University of Alberta

Surface water quality in northern Alberta: the characteristics, hydrologic controls and potential impacts from forest fire and logging

by

Preston Mershon McEachern ©

A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the requirements for the degree of Doctor of Philosophy

in

Environmental Biology and Ecology

Department of Biological Sciences

Edmonton, Alberta

Fall 2003

Your file Votre référence
ISBN: 0-612-88021-4
Our file Notre référence
ISBN: 0-612-88021-4

University of Alberta

Library Release Form

Name of Author: Preston Mershon McEachern

Title of Thesis: Surface water quality in northern Alberta: the characteristics, hydrologic controls and potential impacts from forest fire and logging.

Degree: Doctorate of Philosophy

Year this Degree Granted: 2003

15 Aug 2003

University of Alberta

Faculty of Graduate Studies and Research

The undersigned certify that they have read, and recommend to the Faculty of Graduate Studies and Research for the acceptance, a thesis entitled *Surface water quality in northern Alberta: the characteristics, hydrologic controls and potential impacts from forest fire and logging* submitted by Preston Mershon McEachern in partial fulfillment of the requirements for the degree of Doctorate of Philosophy, Environmental Biology and Ecology.

Elon S. Verry

Abstract

I present the results from a six-year investigation of hydrologic processes and surface water chemistry. The study occurred in peatland-dominated boreal forest basins spanning poorly- to well-drained soils on glacial till with discontinuous permafrost. The first in the series of studies investigated the impacts of fire on lake water chemistry, phytoplankton communities and hydrology, the second details hydrologic processes in six study catchments, and the third investigates hydrologic and chemistry impacts from experimental emulation silviculture. The primary results:

- Elevated nutrient concentrations were apparent in lake waters from burnt drainages. These changes were unprecedented in the fire literature for phosphorus and dissolved organic carbon.
- Recovery to pre disturbance conditions spanned decades.
- Lakes with drainage basin to lake volume ratios less then ten were particularly sensitive to fire impacts.
- Enhanced phosphorus and carbon loading from burnt organic soils cauased nitrogen and light limitation of phytoplankton growth.
- Phytoplankton species richness was reduced in lakes impacted by forest fire.
- Emulation silviculture impacted water yield and chemical flux from well drained and poorly drained catchments.

The studies occurred on traditional lands of the Little Red River Cree and proffered a unique opportunity to include Indigenous knowledge (IK). IK identified important management issues, ecosystem relationships and several impacts that could be quantified with further investigation. The overriding hypothesis of IK was that forestry and fire negatively impact hydrology and water quality because debris caused poor drainage. Subsequent shrub growth amplified impacts on animals, such as moose and buffalo, and on LRRTC relationships with the landscape. Variable and conflicting

responses arose from differences in experience that could be linked to site-specific ecology. As in empirical science, IK generalizes from facts but fails to explain them with precision. The context of IK data should be used to investigate the basis for different responses to similar themes in the same manner we search for ecological factors to explain model residuals. Shared information and the diversity of responses by the LRRTC enabled a comprehensive tableau of ecosystem concepts to emerge forming the basis for a deeper but complementary understanding of ecosystem linkages.

Acknowledgments

Funding for research on the Caribou Mountain lakes was provided by The Sustainable Forest Management Network, Little Red River/ Tallcree First Nations and Daishowa-Marubeni International Ltd. Funding for the watershed studies presented in Chapters five and six was provided by the Forest Resources Improvement Association of Alberta in cooperation with Tolko-High Level Lumber Division. Additional funding was received from the Canadian Circumpolar Institute (NSTP) for gathering indigenous knowledge presented in Chapter seven. I was supported by an NSERC- Postgraduate Industrial Scholarship in partnership with Daishowa-Marubeni Ltd. Additional support was provided by an NSERC Discovery Grant to Dr. E. E. Prepas.

The number of people who assisted me in this project was both large and constantly evolving. I make a modest effort to acknowledge each of them for their contribution to my development as a scientist and to my thesis. I thank my supervisor, Dr. Ellie Prepas, for granting me limitless freedom and support to develop the project and still hold reigns on precision and accuracy. Dr. D. Chanasyk played an unofficial co-supervisory role during the timber harvesting component of the research and Dr. J. Volpe was a co-supervisor during my exam, and Dr. Bill Samuel worked sedulously to ensure I developed as a student and remained focused on my thesis. Several collegues were instrumental in contributing ideas on which this thesis drew. Most notable were contributions by Dr. John Gibson through development of isotopic methods for hydrologic inference (Chapters 2 and 3), Dr. Patricia McCormack whos insistance that I find “a common landscape” to approach Indigenous knowledge resulted in rewarding work with the Little Red River Tallcree First Nation (Chapter 7), Dr. Dolores Planas while I was contemplating phytoplankton dynamics in these systems and response to enhanced nutrients (Chapter 4), and Dr. P. Dinsmore for assistance in the lake study (Chapter 2). Several chapters benefited from reviews by J. Burke, Dr. D. Canfield, Dr. R. Carignan, and several anonymous victims. Invaluable were the many hours S. Paquet invested training me in phytoplankton identification and providing a large share of the sample identification presented in Chapter 4. In addition to the papers I published from my thesis with these collegues, another paper, published in the Journal of Hydrology with Dr. Gibson as primary author, resulted from our collaboration. Dr. Dale Vitt contributed to the early development of my study.

I thank Chief J. Sewepagaham, C. Nanooch, and the Little Red River community for supporting this research on their traditional lands. I cannot overemphasize the value of entrenched support from both Prepas and McCormack for the IK component when disdain echoed from some lips in science. S. Stanley, your input was appreciated and missed. I thank Bob Wynes. His vision and support at DMI allowed me to initiate the harvesting study. Unwavering support from the staff at Tolko-HLLD made much of this project possible. I would particularly like to acknowledge T. Hoffman, B. Avery, M. Lecoeur, B. Asmussen and R. Chemago. I relied heavily on the LFS who provided space in their bunkhouse and in particular on J. Penner who welcomed us at his fire tower, voluntarily helped process samples late into the night, and amiably retrieved us when our vehicle could not pass the tower road. T. Prows donated equipment that made work in 1997 possible and vegetation classification by L. Halsey was appreciated. Finally, my exam benefited from all participants. A special note of thanks for Dr. Sandy Verry for input as an external examiner and Jim Webb for representing Chief Johnson.

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DBA	topographically defined catchment area minus lake surface area, ha
CA	contributing area, ha
EA	effective area, ha
δ_I	isotope composition of lake inflow, ‰ Standard Mean Ocean Water or SMOW
δ_Q	isotope composition of lake outflow, ‰ SMOW
$\delta_{E,}$	isotope composition of evaporated water, ‰ SMOW
δ_L	mean ice-free isotope composition of lake water, ‰ SMOW
δ_P	isotope composition of precipitation, ‰ SMOW
m	free-surface evaporation factor, dimensionless
h	ambient atmospheric humidity, dimensionless
δ^*	Limiting isotopic enrichment factor
δ_h	isotope composition of atmospheric humidity, ‰ SMOW
ε	isotopic fractionation factor, dimensionless
τ	Lake water retention time, y^{-1}
Z_{mean}	mean lake depth, m
E_L	long-term mean annual lake evaporation rate, $mm\Sigma y^{-1}$
R	mean annual drainage basin runoff, mm
P	mean annual precipitation, mm
Q_o	annual lake discharge, $m^3 \bullet y^{-1}$
A_i	total area of a distinctive soil in the CA
F_i	flux rate, $mg \bullet m^{-2} \bullet y^{-1}$
T_i	transmissivity, dimensionless

Chapter 1: Introduction and overview

Forest fire is a disturbance that modifies catchment biogeochemistry and surface water quality. Changes in vegetation, soil and water following a wild fire are sufficiently extensive that many researchers conclude fire is the fundamental dynamic shaping boreal forest ecology (Wright 1983; Kimmins 1984). Evolution of boreal ecology to the repeated disturbance of fires over the last 8000 years has lead to a management model based on disturbance termed emulation silviculture (McRae et al. 2001). Emulation silviculture presupposes that the functional benefits of fire can be attained by logging in patterns similar to burnscapes or, at the least, the impacts of logging can be naturally ameliorated within the benchmarks set by fire (Hunter 1993). The supposition that harvesting entire drainage basins produces acceptable changes in surface water chemistry derives from studies that have been unable to detect impacts in lake water chemistry following forest fire rather than evidence demonstrating that emulation silviculture is appropriate for water quality management. Most Canadian provinces are now considering emulation silviculture as a tool that provides greater flexibility in planning cutblock sizes and locations. Clear evidence and tools are required that assess impacts of emulation silviculture across the diverse boreal landscape. Otherwise we risk unleashing a chimera, an impossible beast cobbled together from disparate components, a lion's head, a goat's body, a serpent's tail - an affront to biology.

Fire impact studies suggest the effects on water quality are fire- and region-specific and depend on the fire's severity, the locale's organic soil characteristics, and the watershed's hydrologic regime. Conclusions of minimal water quality impacts from forest fire derive from a knowledge base developed primarily in southern forests with thin (< 10 cm) organic soils and long growing seasons. Recent data suggest instead that fire can have large impacts on lake water chemistry and biota in Canadian boreal lakes where organic soils play a large role in hydrology and nutrient processing (Carignan et al. 2000). The impacts of fire on lakes may include increases in eutrophying nutrients and deleterious metals. These changes are not considered desirable in the management of aquatic systems and likely should not to be emulated across a landscape.

Chapters two, three and four of my thesis present the results from a study investigating the impacts of fire on lake-water chemistry, algal communities and hydrology. Chapter five provides a detailed investigation of hydrologic processes in six study catchments in Canada's western boreal forest. These peatland-dominated catchments represent a range from poorly drained to well drained soils, in a region where deep organic soils have developed on glacial till and discontinuous permafrost. Chapter five sets the stage for an investigation into hydrologic and chemistry impacts from experimental emulation silviculture on stream waters presented in Chapter six. The studies occurred on traditional lands of the Little Red River Cree and Tallcree First Nations, which proffered a unique opportunity to include a larger historic and ecosystem scope through input by Aboriginal people. In this northern boreal setting the deep organic soils, discontinuous permafrost, short growing season and magnitude of disturbance were expected to exaggerate the impacts of both forest fire and logging on surface water quality. The physical setting for experimental work and long oral tradition offered an extreme test of emulation silviculture not presented elsewhere.

An understanding of fire impacts is essential to build a landscape management tool or even to determine if and where emulation silviculture is appropriate for aquatic ecosystems. A general model representing current understanding of fire impacts would predict increased base cation and nutrient concentrations in surface waters from mineralization and changes in base cation-exchange capacity of burnt organic soil. However, the magnitude of ion flux from catchments is a function of fire severity, with low potential for changes in flux rates when organic soil-layer damage is moderate (e.g., Richter 1982; MacLean 1983; Vitt 1984) and high potential for changes in flux rates when fires are severe (McColl 1975; Bayley et al. 1992; Minshall 1997). When organic soil damage is high, enhanced nutrient-flux should be transmitted to lake biota. Increased nutrient concentrations should augment phytoplankton biomass as well as suspended organic and inorganic lacustrine seston. In Chapter two, I investigate changes in lake water chemistry, seston and phytoplankton biomass following an opportune forest fire where peatland-dominated drainage basins in a discontinuous permafrost region were burnt. Damage to organic soils was variable; however, the hypothesis that flux rates of nutrients increase as a result of fire was confirmed at magnitudes unprecedented in other lake studies. Nutrient concentrations in burnt-drainage lakes were elevated several-fold while lake water nutrient concentrations remained elevated in burn-impacted lakes over a 40-y history. A model was constructed that combined spatial and temporal impacts of fire on lake water chemistry and is the first study in lakes to demonstrate impacts that extend for decades post fire.

Past limnological surveys have compared water chemistry, light penetration and phytoplankton-community characteristics in lakes from disturbed and undisturbed catchments (e.g., Paterson et al. 1998; Rask 1998; Carignan et al. 2000; Planas et al. 2000). The Gordian knot in such surveys is controlling for hydrologic variation among catchments. This variation can obscure changes in water chemistry due to catchment disturbance. While catchment features such as slope and vegetation have been widely used to examine landscape relationships to water chemistry (e.g., Prepas et al. 2001), analysis of direct hydrologic patterns has often been hindered by the paucity of runoff data in remote areas of Canada's boreal forests. Incorporation of hydrologic indices into lake surveys may reduce unaccounted variation in water chemistry and resolve mechanisms transmitting disturbance to water quality impacts (e.g., Gibson et al. 2002). Chapter three reviews the application of stable-isotope mass-balance methods to determine hydrologic indices and presents an analysis of the hydrology/water-chemistry linkages for the 24 lakes introduced in Chapter two. Chapter three confirms the hypothesis that contributing areas have a preponderant influence on runoff chemistry (e.g., Beven and

Kirkby 1979; Hill 1993). Chapter three demonstrates the poor connectivity of contributing areas in peatland dominated catchments like the Caribou Mountains and presents a novel method for calculating both contributing-area and effective contributing-area to surface waters.

If nutrient concentrations increase in lake water following drainage basin disturbances such as forest fire, these impacts should be transmitted to phytoplankton, where success of groups or individual species are partly determined by water chemistry (e.g., Kilham and Kilham 1984; Tilman 1986). However, few studies have linked changes in phytoplankton diversity or productivity commensurate with intense anthropogenic nutrient loading (e.g., Edmondson 1991) to the consequences of fire or logging. On Canada's western Boreal Plain, soils are nitrogen deficient and result in naturally low nitrogen-to-phosphorus (N:P) ratios in lake water. Low N:P ratios result in cyanobacterial blooms in lakes, particularly of nitrogen-fixing Nostocales (e.g., Riley and Prepas 1984). In Alberta, Nostocales blooms are associated with high dissolved oxygen-depletion rates during winter and degraded water quality (Riley and Prepas 1984). Lakes with low N:P ratios may be particularly sensitive to eutrophication following forest fire or logging activities in regions where internal and external phosphorus-loading is high relative to nitrogen loading. Strong nitrogen retention in organic soil, reduced lake water N:P ratio and increased lake water colour were consequences of fire in the Caribou Mountains. Chapter two indicated that lakes in burnt catchments had lower N:P ratios and higher colour than lakes in unburnt catchments. Chapter four characterizes the phytoplankton communities in the same group of lakes to determine if phytoplankton biomass was nitrogen limited when N:P ratios were generally below 12 in lake waters as a consequence of forest fire. The concomitant impact of nitrogen limitation on lake phytoplankton communities is also presented. Hypotheses were subsequently tested: that nitrogen- and light-limitation of phytoplankton would prevail, that phytoplankton community richness would be diminished, and that species adapted to low-nitrogen and light conditions would dominate in burn-impacted lakes. Nitrogen limitation and reduced community richness was confirmed for burn-impacted lakes. Enhanced dominance by Nostocales in burn-impacted lakes was not detected because Nostocales dominated the phytoplankton community in all Caribou Mountain lakes. However, phytoplankton communities in burn-impacted lakes were dominated by *Aphanizomenon* sp. while unimpacted lakes were dominated by *Anabaena* spp. In Chapter four, I provide further evidence that increased nitrate concentrations in burn-impacted lakes may be an important factor limiting increased success of Nostocales.

To understand how fire influences processes that control surface-water chemistry and ecology requires identification of the source-area dynamics and timing of runoff-generation in impacted drainage basins. The prediction of flow paths and flow rates for runoff routed through peatlands is complicated by the anisotropic nature of peat soils and the variable storage capacity of peatlands (Schwartz and Milne-Home 1982; Holecek 1988). Frost and permafrost within northern peatlands amplifies the complexity of modeling water flow through these systems (Pietroniro et al. 1996). In the study area, the forest developed over deep glacial tills where elucidation of recharge, flow, and discharge from local and regional groundwater aquifers is encumbered by variable hydrologic properties that change seasonally (Devito et al. 2000; Evans et al. 2000). Several detailed hydrologic investigations of peatland systems can be assembled to form a picture of seasonal patterns occurring at the catchment scale in my study region. Runoff should be pronounced in the spring when snowmelt encounters low soil-permeability and flows over ice to fill depressions and wetlands (Woo and Winter 1993). As spring progresses, there is a diminishing probability for overland-flow from infiltration-excess because infiltration rates of the aerobic peat layer (acrotelm) increases (Ingram 1983). The glacial tills regnant throughout the study have variable hydraulic conductivities (Evans et al. 2000) ranging from clays (10^{-10} cm/s) to sand and gravel lenses (10^{-1} cm/s). Water, infiltrating from overlaying peat, moves as pipe-flow along the peat-mineral interface if clays are encountered (Hill 1993; Evans et al. 1999) but infiltrates down to aquifers through lenses of high hydraulic conductivity (Evans et al. 2000). Groundwater discharges in peatland catchments as return-flows through fens or as pipe-flows along peat-mineral interfaces (Evans et al. 1999). Return-flow through peat, even within riparian areas, can be restricted to the peat-mineral interface by hydraulic conductivities below 10^{-7} in overlaying compact, anaerobic peat of the catotelm (Evans et al. 1999) and the concrete layer of frost subsumed in the peat well into midsummer (Woo and Winter 1993). Lateral flow through peat often discharges at depressional storage areas such as fens as the final path to surface runoff (e.g., Branfireun and Roulet 1998; Metcalfe and Buttle 2001).

Storm discharge from fens and other saturated peatlands often occurs as piston-flow where precipitation forces previously infiltrated water from soils. Older water produces discharge with the distinct chemistry of peatlands (Branfireun and Roulet 1998). Thus elevated mineral-ion concentrations can be produced during all but the heaviest storms as old water is preferentially discharged (Cirmo and McDonnell 1997). Intersection of water at the peat-mineral interface may produce ion rich water even from ombotrophic peatlands where surface pore-waters are chemically dilute (Vitt and Chee 1990). If piston-flow is the predominant process generating water discharge from peatlands, stream-water chemistry should be dominated by groundwater of mineral or peat origin. The importance of mineral-versus-peat sources should depend on the availability of highly conductive lenses in mineral soils, slope characteristics for producing return flow, and watertable fluctuations within peatlands. The degree to which overland runoff is chemically similar to precipitation should depend on the distribution of wetlands and should occur only when the watertable is exceptionally high. Chapter five investigates hydrology and water chemistry data in six boreal forest catchments and combines data from groundwater wells with stream samples to determine the relative contributions of overland flow and groundwater from organic or mineral soils to catchment water discharge. Chapter five demonstrates that snowmelt proffers the highest discharge rates and the greatest proportion to annual total discharge in the study catchments. This chapter also specifies the degree to which snowmelt-runoff flows overland or through shallow soil pathways, reveals the importance of piston-flow from groundwater, and quantifies how these hydrologic processes differ between sloped catchments and flat catchments. Finally, Chapter five examines the strategic role of peatlands in determining runoff response to storm events and stream chemistry. The focus of Chapter five is to define the role of hydrologic source areas for streams draining northern boreal forests where peatlands and permafrost influence surface water chemistry.

Emulation silviculture provides a model wherein trees are harvested over large contiguous areas that include riparian stands. Emulation silviculture does not protect riparian features such as peatlands from equipment traffic. Chapter five identified the importance of peatlands in controlling solute flux to surface waters (*see also* Vitt et al. 1994; Prepas et al. 2001) suggesting that even exiguous changes in hydrologic processes could confer significant changes in the flux of nutrients to surface waters (e.g., Evans et al. 2000; Prepas et al. 2001). Quantifying changes in surface water chemistry following catchment disturbance can be difficult and may require long time series to detect change due to high variability in surface and soil water chemistry. However, analytical tools exist that are sensitive to ecosystem change over relatively short time periods that have as yet not been applied to timber harvesting. Chapter six combines End Member Mixing Analysis (EMMA, Neal 1990; Muller 1993) with Randomized Intervention Analysis (RIA, Carpenter et al. 1989) into a novel method for detecting forest disturbance impacts on hydrologic processes and biogeochemistry. EMMA provides inference for stream source-waters from diverse catchment soils, commensurately addressing uncertainties in hydrology that often confound interpretation of catchment-scale studies. RIA offers a robust technique for testing that changes observed following an intervention are non-random and therefore the result of a disturbance.

The hypotheses addressing transport of waters and solutes examined in Chapter five were reexamined in Chapter six after timber harvesting to quantify the potential impacts of emulation silviculture on surface waters. Based on the dominant hydrologic processes identified in sloped and lowland catchments in Chapter five, several predictions were proposed for the post harvesting condition. Anticipated changes in catchments with moderate slope were: the rapid discharge of meltwater would intensify, peak discharge would be elevated during snowmelt and summer storms, and total water yield would increase. However, total elemental flux from the sloped basins would remain relatively unchanged because enhanced water yield would derive from rapid discharge of dilute surface runoff. Conversely, in lowland basins where a large proportion of surface soils were saturated: discharge of meltwater would remain unchanged in duration and peak, watertables would rise, total water yield would also increase but over the entire ice-free season and runoff would be increasingly dominated by water derived from organic soils and peatlands. Nitrogen and mineral ion export from mineral soils would decline while concentrations of elements from organic soils, such as organic carbon, would increase following harvesting in lowland areas. I speculated that the total flux of all elements from lowlands would be amplified because more discharge would be routed through soil pathways. The study catchments did not all respond as predicted and Chapter six identifies how physical characteristics of slope, percent peatland cover, aspect, differences in timber harvesting and other features in each catchment interacted to nullify or amplify the hypothesized hydrological and biogeochemical diatheses.

The previous chapters provided a scientific investigation of the hydrologic, biogeochemical, and surface-water quality purlieu with a focus on forest management from a natural disturbance ambit. Chapter seven provides an analyses of human interaction in the study region as described by Indigenous people who have sustained themselves through a precise understanding of ecology. Chapter seven adds breadth to the interplay between fire, human disturbance, and the forest ecosystem with the unique historical and observation-based perspective of Indigenous knowledge. However, inclusion of Indigenous knowledge into a physical science thesis is rare because scientific culture does not readily accept other forms of knowledge (Franklin 1995; Stevenson 1996). Hence, Chapter seven presents a framework for scientists to grasp the significance of Indigenous knowledge and subsequently presents the knowledge shared by local people. The first intention in Chapter seven is to create a forum for discourse between Indigenous knowledge and science and, thus broaden the scope of the thesis with an historic analysis provided by local knowledge.

In Chapter seven, I provide a working definition of Indigenous knowledge and propose that this knowledge be approached at two levels: first, as general theory, and second, as context assessment. Like empirical models in ecology, theories from Indigenous knowledge can be interpreted as representative of mean conditions that have been simplified from complex and detailed information. If properly formulated, these theories can speak to scientists as general relationships between ecosystem parameters. However, as demonstrated in Chapter six, ecosystems do not respond uniformly to environmental changes and will not rigorously adhere to a given set of empirically derived hypotheses. Addressing why certain systems do not follow general expectations (based on our understanding) can be achieved by examining details, which I define as the physical context specific to the system. In addition to describing general ecosystem relationships, Indigenous knowledge is location specific and has evolved through generations of observing ecosystem responses to disturbances such as forest fire. Assessment of the physical context from which individuals draw their knowledge can help describe the full range of potential responses to environmental fluctuations or disturbances. Contextual and individual perceptions are analogous to ecological variation; close examination should reveal intricate substrata. Context assessment can supplement the knowledge base and help modify general theories so they more accurately reflect the dynamics of a specific ecosystem. Chapter seven illustrates how ecologists can view Indigenous knowledge through the lens of statistical power, details the pitfalls of including local knowledge, and attempts to bridge the chasm between ecological science and non-scientific understanding by proposing a model that identifies similarities in language and culture between the two.

The ultimate goal of Chapter seven is to include the perspectives and knowledge of the Little Red River and Tallcree First Nations, two Indigenous Cree nations of northern Alberta. Their Indigenous knowledge of ecosystem dynamics is presented, particularly in regard to the relationship between anthropogenic activities and water quantity and quality in a local watersheds. Interviews were conducted with individuals to gather empirical "data" from their personal experience and broader knowledge that identified relationships among water and the local climate, vegetation, and animal behavior. Little Red River and Tallcree perceptions of water quality perturbations and water quantity fluxes from natural and anthropogenic sources are explored to signal prominent features and values important to local use of their land-water resources.

The thesis is summarized in Chapter eight. The individual studies are assembled into a construct that segues from the broad impacts of forest fire on lake water chemistry into the general then specific impacts on hydrology and aquatic ecology. Changes in hydrology and water chemistry following emulation silviculture are then compared to those from forest fire. The results suggest that discretionary acceptance of emulation silviculture is required. The summary reemphasizes the sensitivity of the study-catchments to disturbance but acknowledges the variability in response due to physical features. Several factors promote sensitivity to disturbance and these are summarized: small catchment-area-to-lake-volume ratios, connectivity patterns among extensive peatlands, and sloped compared to lowland or flat catchments. Lowland catchments may be more sensitive to nutrient perturbations following emulation silviculture, whereas sloped catchments may be more sensitive to the timing and magnitude of runoff generation. Although there are challenges to including information of people intimately connected to the local environment, there are also benefits from the larger perspective that could only be provided by local input. Knowledge derived from longterm observation of the land provides a longer thread to weave the temporally limited scientific studies of my thesis into a combined and more complete tapestry. With this historical perspective, the summary suggests timber harvesting practices may produce similar changes in water quality, vegetation, and animal behavior as does fire but with negative impacts over longer periods of time.

References

Bayley, S. E., D. W. Schindler, K. G. Beaty, B. R. Parker and M. Phosphorus. Stainton (1992). Effects of multiple fires on nutrient yields from streams draining boreal forest and fen watersheds nitrogen and phosphorus. *Can. J. Fish. Aquat. Sci.* **49**(3): 584-596.

Beven, K. J. and M. J. Kirkby (1979). A physically based, variable contributing area model of basin hydrology. *Hydrol. Sci. Bull.* **24**(43-69).

Branfireun, B. A. and N. T. Roulet (1998). The baseflow and storm flow hydrology of a Precambrian Shield headwater peatland. *Hydrol. Process.* **12**(1): 57-72.

Carignan, R., P. D'Arcy and S. Lamontagne (2000). Comparative impacts of fire and forest harvesting on water quality in Boreal Shield lakes. *Can. J. Fish. Aquat. Sci.* **57**(Supp/2): 105-117.

Carpenter, S. R., T. M. Frost, D. Heisey and T. K. Kratz (1989). Randomized intervention analysis and the interpretation of whole-ecosystem experiments. *Ecology* **70**(4): 1142-1152.

Cirmo, C. M. and McDonnell, J. J. (1997). Linking the hydrologic and biochemical controls of nitrogen transport in near-stream zones of temperate-forested catchments: A review. *J. Hydrol.* **199**: 88-120.

Devito, K. J., I.F. Creed, R. L. Rothwell, and E.E. Prepas (2000). Landscape controls on phosphorus loading to boreal lakes: Implications for the potential impacts of forest harvesting. Can. J. Fish. Aquat. Sci., 57: 1977-1984.

Edmondson, W. T. (1991). The uses of ecology : Lake Washington and beyond. Seattle, University of Washington Press.

Evans, J. E., E. E. Prepas, K. J. Devito and B. G. Kotak (2000). Phosphorus dynamics in shallow subsurface waters in an uncut and cut subcatchment of a lake on the Boreal Plain. *Can. J. Fish. Aquat. Sci.* **57**(Supp. 2): 60-72.

Evans, M. G., T. P. Burt, J. Holden and J. K. Adamson (1999). Runoff generation and water table fluctuations in blanket peat: evidence from UK data spanning the dry summer of 1995. *J. Hydrol.* **221**(3-4): 141-160.

Franklin, S. (1995). Science as culture, cultures of science. *Annual Review Anthropol.* **24**: 163-184.

Gibson, J. J., E. E. Prepas and P. McEachern (2002). Quantitative comparison of lake throughflow, residency and catchment runoff using stable isotopes: Modeling and results from a regional survey of Boreal lakes. *J. Hydrol.* **262**: 128-144.

Hill, R. A. (1993). Base cation chemistry of storm runoff in a Forested Headwater Wetland. *Water Resour. Res.* **29**: 2663-2673.

Holecek, G. (1988). Storage-Effective Drainage Runoff Model. *J. Hydrol.* **98**: 295-314.
Hunter, M. L. J. (1993). Natural fire regimes as spatial models for managing boreal forests. *Biological Conserv.* **65**: 115-120.
Ingram, H. A. P. (1983). Hydrology. Ecosystems of the world 4A. Mires: Swamp, bog, fen and moor. A. J. Gore (Ed.). Oxford, Elsevier: 67-158.
Kilham, S. S. and P. Kilham (1984). The importance of resource supply rates in determining phytoplankton community structure. In: Trophic interactions within aquatic ecosystems. D.G. Meyers, J.R. Strickler (eds.) AAAS Selected Symposium 1985. Pg 7-27.
Kimmins, J. P. (1984). Forest Ecology: a foundation for sustainable management. Prentice-Hall, New York. 287-303.
Larsen, C. P. S. and G. M. MacDonald (1998). An 840-year record of fire and vegetation in a boreal white spruce forest. *Ecology* **79**: 106-118.
MacLean, D. A. (1983). Fire and nutrient cycling. In: The role of fire in northern circumpolar ecosystems. R. W. Wien & D. A. MacLean (Eds.). *J. Wiley & Sons Ltd.*: 111-132.
McColl, J. G. (1975). Forest fire: Effects on phosphorus movement to lakes. *Science* **188**: 1109-1111.
McRae, D. J., L. C. Duchesne, B. Freedman, T. J. Lynham and S. Woodley (2001). Comparisons between wildfire and forest harvesting and their implications in forest management. *Environ. Rev.* **9**: 223-260.
Metcalfe, R. A. and J. M. Buttle (2001). Soil partitioning and surface store controls on spring runoff from a boreal forest peatland basin in north-central Manitoba, Canada. *Hydrol. Process.* **15**: 2305-2324.
Minshall, G. W. (1997). Postfire responses of lotic ecosystems in Yellowstone National Park, U.S.A. *Can. J. Fish. Aquat. Sci.* **54**: 2509-2525.
Muller, D. W., Wohlfeil, I. C., Christophersen, N., Hauhs, M, and Seip, H. M. (1993). Chemical Reactiveness of Soil Water Pathways Investigated by Point Source Injections of Chloride in a Peat Bog at Birkenes. *J. Hydrol.* **144**: 101-125.
Neal, C. C., N (1990). Linking Hydrological, Geochemical, and Soil Chemical Processes on the Catchment Scale: An Interplay Between Modeling and Field Work. *Water Resour. Res.* **26**: 3077-3086.
Paterson, A. M., B. F. Cumming, J. P. Smol, J. M. Blais and R. L. France (1998). Assessment of the effects of logging, forest fires and drought on lakes in northwestern Ontario: a 30-year paleolimnological perspective. *Can. J. For. Res.* **28**:1546-1556.
Pietroniro, A., T. Prowse, L. Hamlin, N. Kouwen and E. Soulis (1996). Application of a grouped response unit hydrological model to a northern wetland region. *Hydrol. Process.* **14**: 1245-1261.
Planas, D., M. Desrosiers, S.-R. Groulx, S. Paquet and R. Carignan (2000). Pelagic and benthic algal responses in eastern Canadian Boreal Shield lakes following harvesting and wildfires. *Can. J. Fish. Aquat. Sci.* **57**(Supp 2): 136-145.
Prepas, E. E., D. Planas, J. J. Gibson, D. H. Vitt, T. D. Prowse, W. P. Dinsmore, L. A. Halsey, P. M. McEachern, S. Paquet, G. J. Scrimgeour, W. M. Tonn, C. A. Paszkowski and K. Wolfstein (2001). Landscape variables influencing nutrients

and phytoplankton communities in Boreal Plain lakes of northern Alberta: A comparison of wetland- and upland- dominated catchments. *Can. J. Fish. Aquat. Sci.* **58**: 1286-1299.

Rask, M. K. N., Sirkka-Lisa Markkanen (1998). Forestry in catchments: Effects on water quality, plankton, zoobenthos and fish in small lakes. *Boreal Environmental Research* **3:** 75-87.

Richter, D. D. (1982). Prescribed fire: Effects on water quality and forest nutrient cycling. *Science* **215**(661): 663.

Riley, E. T. and E. E. Prepas (1984). Role of internal phosphorus loading in two shallow productive lakes in Alberta, Canada. *Can. J. Fish. Aquat. Sci.* **41**: 845-855.

Schwartz, F. W. and W. A. Milne-Home (1982). Watersheds in Muskeg Terrain. 1. The Chemistry of Water Systems. *J. Hydrol.* **57**(3-4): 267-290.

Stevenson, M. G. (1996). Indigenous knowledge in environmental assessment. *Arctic* **49**(3): 278-291.

Tilman, D. (1986). Green, bluegreen and diatom algae: taxonomic differences in competitive ability for phosphorous, silicon and nitrogen. *Arch. Hydrobiol.* **106**(4): 473-485.

Vitt, D. H. (1984). The vegetation and water chemistry of four oligotrophic basin mires in northwestern Ontario. *Can. J. Bot.* **62**: 1485-1500.

Vitt, D. H. and W.-L. Chee (1990). The relationship of vegetation to surface water chemistry and peat chemistry in fens of Alberta, Canada. *Vertatio* **89**: 87-106.

Vitt, D. H., L. A. Halsey and S. C. Zoltai (1994). The bog landforms of continental western Canada in relation to climate and permafrost patterns. *Arctic Alpine Res.* **26**(1): 1-13.

Woo, M.-K. and T. C. Winter (1993). The role of permafrost and seasonal frost in the hydrology of northern wetlands in North America. *J. Hydrol.* **141**: 5-31.

Wright, D. H. (1983). Species energy theory: an extension of species area theory. *OIKOS* **41**(3): 496-506.

Chapter 2:
Forest fire induced impacts on phosphorus, nitrogen, and chlorophyll *a* concentrations in boreal subarctic lakes of northern Alberta.

Introduction

Interest in fire as a natural disturbance impacting catchment biogeochemistry and surface water quality is rekindling as a benchmark for anthropogenic impacts. However, studies on the effects of fire on water quality suggest impacts are regionally or fire specific due to organic soil characteristics, severity of fire and hydrologic regime. The magnitude of ion flux from catchments is thought to depend on fire severity because of modifications to cation exchange and biochemical reactions in the organic soil layer (Grier 1975; Stark 1977; Schindler et al. 1980). In experimental fires where a third or less of the organic soil layer was burnt, no detectable changes in soil water and stream water carbon, nitrogen, phosphorus, their inorganic fractions or major cations were reported (e.g. Richter et al. 1982). Following more severe fires, increases in nutrient flux were large but usually short lived (< 5 yr) and changes in surface water concentrations did not exceed inter-annual variation (McColl and Grigal 1975; Bayley et al. 1992; Minshall et al. 1997). These studies have occurred primarily in regions with thin organic soils, whereas studies in burnt permafrost-peatland dominated systems such as those in northern Alberta are rare.

Water chemistry has been studied in burnt catchments with small proportions of peatlands. A comparison of burnt and unburnt portions of a *Sphagnum fallax / Picea mariana* mire demonstrated no detectable long-term (8 yr) impact of fire on water chemistry (Vitt and Bayley 1984). In a review of circumpolar studies MacLean et al. (1983) reported fire rarely burnt the entire organic layer and changes in soil water chemistry did not result in nutrient flux from peat-derived soils. However, wetland catchments had elevated phosphorus flux after severe fires (Bayley et al. 1992). The previous studies give

Accepted: McEachern, P., Prepas, E. E., Gibson, J. J., and Dinsmore, P. (2000). *Can. J. Fish. Aquat. Sci.*, 57(Supp 2): 73-81.

little indication of potential impacts from forest fire in catchments with deep peat soils (0.5 m or more), underlain by glacial till and containing permafrost or seasonal frost lasting a majority of the summer. A lack of data makes it difficult to address the potential impacts of large-scale forest disturbance proposed for catchments of northern Alberta, particularly those containing permafrost where forestry and petroleum activities are increasing.

A unique opportunity to examine extensive damage to peatlands was provided when 129 000 ha of the Caribou Mountains, a Sub-arctic plateau, were razed by fire in 1995. In this single event, one-third of the Plateau was burnt, equaling 50% of the mean annual area burnt in the province of Alberta between 1994 and 1998 (Alberta Environment unpubl. data). Impacts on lake water chemistry were expected because of high fire severity, large proportion of catchments burnt (between 60 and 100%) and hydrology dominated by flow through peatlands. We hypothesized that the 1995 fire in the Caribou Mountains would reduce base cation exchange capacity of peat and increase mineralization of nutrients to produce: a) increased base cation and nutrient concentrations in surface waters, b) corresponding increases in phytoplankton biomass, and c) increased suspended organic and inorganic seston. We also examined long-term (decades) impacts from fire and relationships between catchment characteristics and lake water chemistry.

Materials and methods

Site description

The Caribou Mountains (59° N 115° W) are an erosional remnant forming a large, relatively flat plateau 500 m above the Peace River valley. Peatlands cover 56% of the Caribou Mountains and are predominately underlain by poorly-drained cryosolic and brunisolic soils (Strong and Leggat 1992). Subsurface geology is shale, feldspathic sandstone, and siltstone of deltaic and marine origin. Open forest of black spruce (*Picea marianna*) and an understory of mixed *Sphagnum* spp., feathermosses, brown mosses and lichens dominate vegetation. Prior to the 1995 fire, most spruce stands in burnt and reference catchments originated between 1860 and 1910 (Alberta Vegetation Inventory 1:20000 map series 1983) indicating there had not been a stand-replacing fire in 90 years. Aspen (*Populus tremuloides*) is concentrated in upland areas. Mean percent upland per catchment is less than 30%, but is 83% in one burnt catchment (Table 2-1). Permafrost is estimated between 0.5 and 0.75 m beneath the organic soil surface within continental bogs (Strong and Leggat 1992) which cover an average of 62% of the study catchments. Collapse scars signify degraded permafrost (Vitt et al. 1994) and were present in bogs averaging an additional 19% of the study catchments prior to the fire. In the fire, understory herbaceous, sedge, and lichen cover were incinerated but damage to moss and peat varied. Impact from burning was mostly limited to 0 to 20 cm of peat. In some locations burning continued within deeper peat until the following summer. Late August testing in five catchments revealed no permafrost in the peat layer (down to mineral soil) of burnt areas, whereas it was present in five unburnt continental bogs. Mean May through August temperatures were 10.2 °C (automated climate station), typically with less than 800 growing degree days and between 400 and 450 mm of annual precipitation (Strong and Leggat 1992).

Ten headwater lakes were selected with catchments, where between 50 and 100% of the tree cover was killed by fire (mean 83%, median 90%, SD 15%). These, plus 14 headwater lakes in unburnt reference catchments and five headwater lakes in catchments burnt between 1961 and 1985, were sampled monthly after ice-out from late June to late August/early September (Fig. 2-1). The lakes selected from previously burnt catchments were used in time-since-disturbance and % disturbance analyses only. Among all combined lakes, differences in surface elevation were less than 90 m across the 120 km distance of this study on the Plateau. The study lakes had zero to three neighbours within 20 Km that contributed to the same larger drainage. Lakes within each of these drainage groups were within 15 m surface elevation of their neigbours. Lake surface areas ranged from 2.6 to 1173 ha and mean depths from 0.3 to 11.6 m. All lakes were either polymictic or if stratification was observed (late July for deeper lakes) mixed conditions were restored by the next sampling event (Aug./Sept.) In all lakes, dissolved oxygen concentrations exceeding 5 mg•L^{-1}, 0.5 m above the bottom through the summer. Three stream, four fen and six bog sites were also sampled during the summer of 1997. Two streams, MS1 and MS2, drained adjacent unburnt watersheds of 40 and 6 km^2, respectively. MS3, drained a 1 km^2 watershed that was entirely burnt. Thus the watershed area ratios were 40:6:1 for the three streams. MS2, was gauged for discharge calculations. Samples were collected from MS1, MS2 and MS3 on the same four sample dates which included bankfull and baseflow conditions. Our limited data show the ratio of discharge from MS3 relative to the reference streams was 20:1.5: 1. Based on this limited data set the burnt catchment, MS3, released between 2- and 4-fold more catchment-weighted discharge than the two unburnt catchments.

Ground cover in each catchment was classified as upland-aspen, peatland or open water. Peatlands were subdivided into veneer bog, peat plateau, poor fen and rich fen based on vegetation and slope characteristics identified from 1994 - 1:20000 aerial photographs (Halsey et al. 1997). Percentage of fire disturbance per drainage basin was estimated from 1996 aerial photographs. Bathymetric maps were constructed from depth measurements along 5 to 15 transects on each lake. Depth measurements were recorded by echo-sounding at equal time intervals along transects while traveling at a constant velocity and used to interpolate depth contours with ArcInfo. Catchment slope was calculated by computing elevation gain divided by linear distance to lake shore (CS1: D'Arcy and Carignan 1997) with 10 to 30 transects from topographic high points and intermediate saddles around the watershed.

At the deepest site in each lake, water temperature and dissolved oxygen (YSI 50B), and light penetration (Li-Cor LI-185, LI-192SB flat sensors) profiles were recorded at 0.5-m intervals. Transparency was estimated with a 20-cm Secchi disc. A 12-L euphotic-zone, composite water sample was collected with multiple hauls of a polyvinyl chloride tube to the depth of 1% light penetration or 1 m above the lake bottom. Three opaque 2-L acid-washed polyethylene bottles were rinsed and filled with the composite sample. pH of the composite sample was measured with a Hanna HI9025 meter calibrated before each use.

Triplicate 25-mL aliquots were pipetted into 30-mL glass tubes used directly in digestion processes for total and dissolved nitrogen (TN, DN) and phosphorus (TP, DP) analysis. Unfiltered water was used for TN, TP and ammonium (NH_4^+-N) analyses. Filtrate (Millipore HA, 45 μm) was used for DN, DP and nitrate-nitrite (NO_3^--N) analyses. TN and DN samples were preserved with 10 μL of 40% H2SO4 for storage (1-2 wk), and neutralized with equivalent NaOH prior to digestion in the laboratory. TP and DP were analyzed spectroscopically (5-cm cell) from persulfate-oxidized samples by molybdate blue absorption (Prepas and Rigler 1982). TN and DN concentrations were determined by second derivative spectroscopic analysis of persulfate oxidized samples (Crumpton et al. 1992). Soluble reactive phosphorus (SRP) concentrations were measured by molybdate blue absorption from membrane filtrate within 24 h of collection. Ammonium and nitrate samples were frozen in 125-mL polyethylene bottles for analysis by indophenol blue (NH_4^+-N) and cadmium reduction (NO_3^--N) with automated colorimetry (Technicon methods 100-700 W/B and 155-71W). Total Suspended Solids (TSS), Nonvolatile suspended solids (NVSS), and volatile suspended solids (VSS) were collected on pre-ashed GF/F filters (mean particle retention 0.7 μm) and analyzed in duplicate after APHA (1993). Total and bicarbonate (HCO_3^-) alkalinities were estimated by the Hach phenolpthaline/ bromocresol method (APHA 1993). Filtrate from GF/F filters for cations (Ca^{2+}, Mg^{2+}, Na^+, K^+), dissolved organic carbon (DOC), and anions (Cl^-, SO_4^{2-}) analysis were stored in 60-mL polyethylene bottles. Cation samples were preserved with 60 μL of 40% H_2SO_4 , while DOC and anion samples were refrigerated. Cations were analyzed by atomic absorption flame spectroscopy (Perkin-Elmer AS90, AA3300), DOC by high temperature catalysis (Shimadzu TOC-5000), and SO_4^{2-} and Cl^- concentrations by chromatography (Dionex 2000i/SP). Total suspended chlorophyll was collected on Gelman A/H filters (mean particle retention 1.2 μm) in duplicate from each of the three composite sample bottles. Filters were subsequently desiccated with silica-gel and frozen. Total chlorophyll *a* (CHL) was extracted with 90% ethanol (Sartory and Grobbelaar 1986) and concentrations were determined with fluorometric methods Knowlton (1984).

Summer mean (June-early Sept.) values were used in all analyses. Data are presented as Box and Whisker plots where the box outlines 25% to 75% of the data and whiskers denote 10 to 90% of the data. The line within each box represents the median value. pH values were converted to $[H^+]$ prior to averaging and analysis and converted back to pH for reporting. Data were tested for deviation from a normal distribution using Kolmogorov-Smirnov at $P \leq 0.05$. Data were either normally distributed or were $\log_{10}$-transformed to meet assumptions for normality and were analyzed by *t*-test, Pearson's correlation and univariate least-squares regression. Probability values and correlation coefficients are reported for these tests. We specify when non-normal data were analyzed by non-parametric tests (Mann-Whitney *U*). Analysis of covariance (ANCOVA) was used to test differences in linear regressions between burnt and reference systems. We used discriminant analysis (DA) to determine if reference and burn-impacted lakes were different in measured catchment and lake physical characteristics. Percent open water could not be included in the DA because of zero values. Differences were deemed significant at *P*•d 0.05. All analyses used Statistica 4.1 and StatView 4.5 for Macintosh.

Results

Fire impacts on water chemistry

Several physical characteristics of lakes and their catchments differed between burnt and reference systems (Table 2-1). Reference lakes tended towards larger surface areas (Mann-Whitney $P = 0.02$), lake volumes (Mann-Whitney $P = 0.05$) and percent open water cover per catchment (Mann-Whitney $P = 0.01$). Percent open water averaged 5 and 1 % of reference and burnt catchments, respectively. The small proportion of open water and its concentration in collapse scars, areas of degraded permafrost, makes it unlikely that it contributed to chemical differences in the lakes. Drainage ratios ($Wo \cdot Ao^{-1}$) and surface area to volume ratios, considered more important in determining land-water linkage than Wo or Ao alone, were not different between burn-impacted and reference lakes ($P = 0.6, 0.3$ respectively). Slope was also not different between burn-impacted and reference catchments ($P = 0.3$). Discriminant function analysis, containing drainage ratio, lake area : volume ratio, catchment slope and percent upland contained one root and was not significant (Wilk's $\lambda = 0.75$, Canonical $r = 0.5$, $P = 0.3$). Likewise, substituting Ao, Wo and V for the Wo/V and Wo/Ao ratios did not produce a significant discriminant function ($P = 0.2$). Therefore, it is unlikely that chemical differences observed between reference and burn-impacted lakes resulted from physical characteristics alone.

Total base cation concentrations (Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}), of Caribou Mountain lake water (median 0.52 $meq \cdot L^{-1}$) were all below the world average for fresh water (2.33 $meq \cdot L^{-1}$). There were no detectable differences in mean total base cation concentrations (Fig. 2-2) between burnt (0.61 $meq \cdot L^{-1}$) and reference (0.72 $meq \cdot L^{-1}$) lakes ($P = 0.53$). As a percentage of total cations, Ca^{2+} was lower and K^{+} was higher in burnt compared to reference lakes ($P < 0.05$).

Fire affected lake acidity and ion balances possibly through increases in organic anions in burnt lakes. Lake water in burnt catchments was moderately acidic relative to reference lake water (mean pH = 6.9 and 7.6 respectively, P =□0.04). Burnt lakes demonstrated low total alkalinity compared to reference lakes, however, differences in means were not detectable (16.7 and 30.3 $mg \cdot L^{-1}$ $CaCO_3$, $P = 0.07$, Fig. 2-2). Sulfate concentrations were elevated in burnt lakes (Mann-Whitney $P = 0.01$) compared to reference lakes with median concentrations of 2.4 and 1.5 $mg \cdot L^{-1}$, respectively. Fire did not affect chloride concentrations, which averaged 0.15 and 0.13 $mg \cdot L^{-1}$, for burnt and reference lake water, respectively ($P = 0.3$). Mean total anions (SO_4^{2-} , Cl^- , HCO_3^-, NO_3^-) balanced mean total cations in reference lake water (mean = 0.723 meq•L^{-1}, $P = 1.0$). All lakes in burnt catchments had a deficit in total anions (0.46 meq•L^{-1}, $P << 0.01$). The average deficit of 25% below mean total base cations, suggested a large pool of organic acids in burnt lakes. Most *Sphagnum*-derived acids contain –COOH groups which act as anions in solution while contributing protons (Clymo 1984). Anion deficits were linearly correlated with DOC in burnt ($r^2 = 0.84$, $P << 0.01$, n = 10) and less so in reference lakes ($r^2 = 0.26$, $P = 0.06$, $n = 14$). Fire apparently increased organic anion concentrations along a relationship with DOC that also existed in reference lakes.

Lake water in burnt catchments had elevated phosphorus and nitrogen concentrations. In reference lakes, mean TP, DP and SRP concentrations were 33, 14 and 4 μg•L^{-1}, respectively (Fig. 2-3). Lake water in burnt catchments had 2.6, 3.2 and 6.8-fold higher TP, DP and SRP concentrations, respectively than reference lakes ($P << 0.01$). In reference lakes, mean TN, DN, NO_3^--N, and NH_4^+-N were 655, 488, 3 and 15 μg•L^{-1}, respectively (Fig. 2-4). Burn-impacted lake water contained 1.2-fold higher DN ($P = 0.02$), 3-fold higher NO_3^--N ($P = 0.04$) and 1.4-fold higher NH_4^+-N (Mann-Whitney $P = 0.03$) concentrations. Mean TP and DP in the two reference streams (MS1 and MS2) were 48 and 28 μg•L^{-1} while the burn-impacted stream (MS3) contained 5.4-fold higher TP and half the DP. Mean TN and DN concentrations for water in MS1 and MS2 were 710 and 650 μg•L^{-1}, respectively, while MS3 contained 2.2-fold higher TN and 1.2-fold higher DN concentrations. Increased export of phosphorus and nitrogen from burnt relative to reference watersheds was likely given the 2- to 4-fold higher catchment weighted discharge from MS3.

Caribou Mountain water had high concentrations of DOC, particularly in burnt systems. Lake mean DOC concentration in burn-impacted lake water (25 mg•L^{-1}) was 1.6-fold higher ($P << 0.01$) than in reference lakes (16 mg•L^{-1}). Further, in reference streams, MS1 and MS2, DOC concentrations averaged 33 and 37 mg•L^{-1}, respectively. In

MS3, which drains a burnt area, DOC averaged 28□mg•L^{-1}. The three sampled fens contained almost identical DOC (45, 45, 48□mg•L^{-1}) concentrations despite receiving drainage from burnt and unburnt areas. MS3 exported 3 and 2 times more DOC per unit catchment area than MS2 and MS1, respectively, given 2- to 4-fold higher weighted discharge from MS3.

Caribou Mountain water was highly coloured, especially within burnt catchments. Reference lake water mean colour was 151 and ranged from 26 to 388 mg•L^{-1} Pt, comparable to lake water in the boreal mixedwood ecoregion to the south, where colour ranged from 8 to 358 mg•L^{-1} Pt (Prepas m.s. sub.). Mean colour in burn-impacted lake waters (342 mg•L^{-1} Pt) was 2.3-fold higher ($P << 0.01$) than in reference lakes. Colour was related to DOC in all lake waters (Table 2-3). Variance in light penetration was closely associated with colour of lake water; The natural logarithm of Secchi transparencies were negatively related to colour while light extinction coefficients were positively related to colour (Table 2-3). Mean Secchi transparencies in burn-impacted lakes (0.73 m) were 54% of those in reference lakes ($P << 0.01$), while the mean light extinction coefficient (0.787 m^{-1}) was 1.7-fold higher in burn-impacted lakes ($P << 0.01$) compared to reference lakes. Secchi depth and light extinction were correlated ($r^2 = 0.36$, $P < 0.01$, $n = 24$); when lakes C24 and C45 were removed the relationship was stronger ($r^2 = 0.85$, $P << 0.01$, $n = 22$). Lake C24 had a high extinction coefficient (1.28 m^{-1}) for its Secchi depth (1.0 m) due to continuous surface bloom of Aphanizomenon sp. as flakes. Lake C45 had a high extinction (0.54 m^{-1}) for its Secchi depth (2.8 m) due to wind conditions during sampling. Elevated DOC and colour in burn-impacted lakes reduced transparency and likely enhanced the potential for light limitation of phytoplankton growth after fire.

Fire affected inorganic suspended solids concentrations. Mean nonvolatile suspended solids (NVSS) in reference lakes was 0.65 mg•L^{-1}, whereas burnt lakes contained 2-fold higher NVSS (Mann-Whitney, $P = 0.04$). Volatile suspended solids (VSS) were not different between reference and burnt lakes, averaging 2.6 and 2.7 mg•L^{-1} respectively ($P = 0.88$). Increased transport of inorganic particles from burnt catchments is inferred.

Fire impacts on algal biomass as chlorophyll

Fire did not appear to affect lake water CHL concentrations. Among reference lakes, mean CHL was 12 µg•L^{-1} and $\log_{10}$-transformed values were strongly related to TP (Fig. 2-5, $r^2 = 0.83$, $P << 0.01$, $n = 13$). Despite the much higher phosphorus concentrations observed in burn-impacted lake waters, CHL was not elevated, averaging 13 µg•L^{-1}. There was no detectable relationship between CHL and TP concentrations for burn-impacted lakes ($r^2 = 0.06$, $P = 0.5$). We predicted CHL concentrations for burn-impacted lakes with the CHL-TP model for reference lakes. The residuals between predicted and observed values were positively related to Secchi depth ($r^2 = 0.44$, $P < 0.01$, $n = 10$) and to TN:TP ratio ($r^2 = 0.69$, $P << 0.01$, $n = 10$). The lack of response in CHL to elevated phosphorus in burn-impacted lake waters and the relationship with Secchi depth is consistent with light limitation.

Fire impacts as they relate to catchment features

General patterns relating lake chemistry to catchment features were poor. There were no detectable relationships between nitrogen or phosphorus concentrations of lake waters with drainage ratio ($r^2 < 0.25$, $P > 0.3$). However, colour and DOC were linearly related to log-drainage ratio (Table 2-3). Colour and DOC tended to increase at a faster rate with drainage ratio in burnt lakes compared to reference lakes (Table 2-3), however, the slopes of these relationships were not distinguishable (ANCOVA, P =0.09 and P = 0.20 respectively).

Percent disturbance and time since disturbance

Lakes in the previously (1961-1985) burnt watersheds had phosphorus concentrations intermediate between recently (1995) and non-burnt watersheds. TP and DP concentrations for burnt lakes increased in a positive-linear relationship with percent disturbance (intercept forced through mean TP and DP for reference lakes). TP and DP appeared to decline with the natural logarithm of time-since-disturbance. Data were not normally distributed in time or percent disturbance and contained zero values for reference lakes so we combined both hypothesized disturbance patterns into a single disturbance index (DI):

$$\text{DI} = \%\text{Disturbance} \bullet e^{-Kt} \tag{8}$$

where t = time since disturbance (90 yr for reference lakes) and K = decay constant. The value for K was predicted from an exponential decay function $K = 0.693 \bullet T^{-1}$ where T = half-life which was estimated from the relationship between $\log_{10}$ DP and TP with the natural logarithm of time-since-disturbance. The half-life was estimated at 11 and 20 yr for DP and TP, respectively. The disturbance index explained 74 and 76% ($P << 0.01$) of the variance in TP and DP, respectively, among the 15 impacted lakes (Table 2-3). We included reference conditions in the analysis by forcing the intercept through mean TP and DP concentrations for reference lakes (intercept value from Table 2-3) assuming they represented a zero DI.

Discussion

Catchments in the Caribou Mountains are fundamentally different from those where fire effects have historically been studied. The large proportion of inundated peatlands in the Caribou Mountains created unique nutrient responses following fire. For example, forest fire usually augments nitrate and to a lesser degree, phosphorus export from granitic regions (Lewis 1974, McColl and Grigal 1975, Wright 1976). The relatively severe Caribou Mountains fire caused elevated phosphorus, and to a lesser degree, nitrogen concentrations in lakes which suggested higher phosphorus compared to nitrogen export following fire. Similar findings were reported from other wetland systems. Burnt fens exported phosphorus and retained nitrogen, while burnt upland catchments exported nitrogen and held phosphorus in the granitic Experimental Lakes Area (Bayley et al. 1992). Nitrogen retention in Alberta peatlands typically exceeds 98% (Li and Vitt 1997), a value comparable to the >85% retention reported by Bayley et al. (1992) for burnt fens. Surface water in northern Alberta, where peatlands dominate, could be more sensitive to eutrophication from elevated TP following fire than indicated by upland fire studies because of reduced phosphorus retention in peatlands after fire.

Increase in the flux of divalent cations and potassium from organic soils usually occurs after fire (Tiedmann et al. 1978). Increased flux rates of divalent ions from burned peat likely occurred in the Caribou Mountains. However, the cation exchange capacity of both living and dead peat likely remained intact, liberating protons as cations were exchanged (Clymo 1984). The result was a 5-fold median increase in $[H^+]$, decreased pH and reduced alkalinity in burnt lakes while base cation concentrations did not change. Organic acids associated with elevated DOC in burn-impacted lakes possibly added to overall acidification while contributing organic anions (presumably $R–COO^-$).

The relationship between CHL and TP in reference lakes on the plateau indicates a strong association between phytoplankton biomass and total phosphorus. The slope for $\log_{10}$-transformed values (2.31) was more positive and the intercept (-2.46) more negative than values reported for North America (Nurnberg 1996).

A general model for the movement of water to lakes and streams in the discontinuous permafrost region of boreal Alberta may include groundwater systems or flow through organic peat. Surface runoff is minimal, even during intense storms, because the infiltration capacity of surface moss and peat (*acrotelm*) is about 1 $cm \cdot s^{-1}$ declining to 10^{-4}

to 10^{-6} cm•s^{-1} in anaerobic peat (*catotelm*) layers (Ingram 1983). Groundwater could discharge through stream beds and lakes themselves as they appear to be the only places without permafrost, other than the hydrologically disconnected collapse scars. Inputs from the peat layer should be restricted to a small effective contributing area when the water table is limited to the catotelm. Rain events that raise the water table to the acrotelm should cause a substantial increase in effective contributing area and loading of peat-derived water due to higher hydraulic conductivity in the acrotelm. Macropores, typically occurring at the peat-mineral interface (e.g., Hill 1993), may also be important in delivering water high in DOC and protons. However, water discharge from fens likely had a greater influence on surface water chemistry than bogs by virtue of their higher discharge rates (Halsey et al. 1997). Groundwater in the Caribou Mountains had base cation concentrations 20-fold higher than found in lakes (McEachern unpub. data). The similarity between fen and lake water chemistry and the high cation concentrations of groundwater suggests groundwater discharge to lakes was minimal and restricted to discharge through fens.

Time-since-disturbance and percent disturbance were important factors in nutrient enrichment among burnt lakes. When combined into a disturbance index (DI), they explained 74 and 76% of the variance in lake TP and DP concentrations. An exponential decay model for time-since-disturbance impacts was used because it matched the trend for our limited temporal data. The hypothesis is justifiable because recovery rates should initially be rapid as new growth and microbial communities are re-established followed by a decrease with increasing time as microbial and plant communities stabilize and nutrients are flushed from the lake and lost to sediments.

Our results suggest forest fire had a profound impact on surface water quality in lakes of the Caribou Mountains. These lakes responded to fire with elevated nutrient concentrations because phosphorus and to a lesser degree, nitrogen, liberated during fire and subsequent decomposition was not retained by peat. Though base cations were likely liberated from burnt material, cation exchange with peat resulted in an increase in the flux of protons. Elevated phosphorus concentrations slowly returned to reference conditions depending on both the magnitude of disturbance (% disturbance) and time-since-disturbance. A single disturbance index combining %disturbance and time proved effective, however, a larger range in time is needed to test the hypothesized exponential and linear components of this model. The effects of fire on water chemistry may be larger in peatland-dominated catchments than elsewhere, due to elevated export of phosphorus and long recovery periods.

Table 2-1: Physical parameters for Caribou Mountain study lakes.

ID	W_o (ha)	A_o (ha)	Volume (103 m3)	Z_m (m)	Slope (%)	Upland (%)	Peatland (%)	Disturbance (%)	τ (days)
Reference lakes									
C1	663.2	52.3	510	1.07	1.5	1	94	0	75
C2	614.8	93.4	1910	2.17	2.3	2	95	0	217
C5	1068	76.5	568	0.75	2.4	6	79	0	80
C6	260.4	91.1	261	0.30	3.1	16	79	0	55
C7	569.0	90.0	578	0.66	2.9	6	87	0	61
C8	8208	60.8	612	0.99	1.4	5	85	0	49
C9	146.3	86.1	1563	1.83	2.6	13	86	0	551
C11	204.8	134.4	4071	3.06	6.1	65	35	0	887
C30	3218	536.1	55564	11.60	2.0	28	67	0	770
C32	3018	953.4	27318	2.90	3.0	12	81	0	314
C34	5571	166.5	5840	3.50	4.0	22	77	0	194
C35W	825.0	175.0	18240	5.50	6.1	32	66	0	419
C45	4627	1173	93840	8.00	4.0	14	83	0	660
C47	2732	59.0	697	1.18	6.0	72	25	0	56
Lakes in catchments burnt in 1995									
C17	700.9	164.8	2300	1.54	1.9	83	16	80	105
C23	1053	65.8	2436	4.10	1.9	19	80	90	32
C24	1761	159.8	10332	6.36	3.6	35	62	95	42
C25	820.1	151.9	4278	2.87	4.2	25	72	90	171
C26	1520	36.6	523	1.68	3.3	25	73	75	36
C27	581.1	28.1	220	0.85	6.0	30	69	60	43
C41	32.6	2.6	22	0.86	2.0	43	57	100	68
C42	34.3	7.4	53	0.72	3.0	51	49	98	96
C43	691.9	8.3	50	0.60	4.0	51	45	95	26
C46	359.4	4.4	25	0.58	6.0	44	51	60	44
Lakes in catchments burnt between 1961 and 1985									
C12	642.9	40.5	440	1.22	2.9	6	88	20	101
C13	483.8	175.5	1745	1.04	1.3	12	81	80	133
C14	268.9	62.4	1779	2.93	4.3	27	73	80	298
C15	2257	128.3	2013	1.67	2.8	6	91	70	77
C16	995.1	53.4	1056	2.04	4.2	42	55	80	129

Note: watershed area (Wo), lake surface area (Ao), lake mean depth (Z_m), residence time (τ). Slope, upland and peatland are percentages for each catchment and disturbance is the percent impacted by forest fire.

Table 2-2: Summary of chemical characteristics for Caribou Mountain lakes. Numbers are summer means. Abbreviations are: CHL = chlorophyll *a*, TP, DP and SRP are total, total, dissolved and soluble reactive phosphorus, TN and DN are total and total dissolved nitrogen, NO_3 is NO_3-N, NH_4 is NH_4^+-N, all in $\mu g \bullet L^{-1}$. DOC and DIC are dissolved

ID	CHL	TP	DP	SRP	TN	DN	NO3	NH4	DOC	DIC	VSS
	$\mu g \bullet L^{-1}$	$\mu g \bullet L^{-1}$	$\mu g \bullet L^{-1}$	$\mu g \bullet L^{-1}$	$\mu g \bullet L^{-1}$	$\mu g \bullet L^{-1}$	$\mu g \bullet L^{-1}$	$\mu g \bullet L^{-1}$	$mg \bullet L^{-1}$	$mg \bullet L^{-1}$	$mg \bullet L^{-1}$
C1	9.87	31	16	6	609	514	2.6	17.8	22	6.3	1.35
C2	16.58	38	20	9	744	527	11.6	31.2	18	5.8	3.49
C5	12.17	34	11	6	688	452	1.1	10.1	17	5.4	3.21
C6	14.51	36	10	5	823	474	3.2	13.3	12	4.7	4.63
C7	24.94	46	17	2	756	470	1.4	10.6	17	4.3	4.64
C8	4.68	40	25	14	661	612	5.0	13.6	27	7.6	1.57
C9	9.42	32	13	3	729	528	4.6	9.3	12	7.6	4.28
C11	6.79	31	11	2	750	585	1.2	15.4	12	14.8	1.73
C30	4.74	22	9	3	412	349	7.0	14.4	13	8.3	0.80
C32	7.73	32	12	2	445	345	1.0	10.9	11	3.5	2.20
C34	6.66	26	13	1	550	466	1.7	11.5	18	11.5	1.40
C35W	22.72	38	12	4	710	429	0.6	25.2	16	11.2	3.60
C45	9.22	26	11	1	454	361	4.9	12.2	10	7.7	1.80
C47	12.59	35	17	5	840	723	1.1	13.5	27	10.6	1.76
C17	39.23	125	66	36	1086	561	31.8	48.6	18	6.7	5.92
C23	5.94	79	55	40	552	468	6.1	12.9	22	6.3	1.70
C24	10.67	78	62	40	622	505	16.3	18.7	19	6.4	1.71
C25	24.31	55	21	9	751	421	2.6	12.7	14	6.2	4.16
C26	5.15	109	47	32	718	636	5.7	14.4	30	7.6	1.46
C27	3.89	98	43	24	649	583	12.6	16.1	27	8.4	1.12
C41	11.77	48	28	15	871	741	3.0	16.1	32	9.6	1.78
C42	5.97	40	21	4	757	668	5.6	22.2	21	14.6	1.59
C43	20.05	166	79	52	811	664	1.9	30.0	31	7.1	5.51
C46	7.26	54	33	17	779	734	4.2	15.5	35	8.9	2.02
C12	11.59	36	19	12	809	630	20.3	18.3	23	6.8	2.76
C13	19.74	46	14	2	799	534	3.8	70.9	14	6.5	4.27
C14	9.03	36	15	5	588	456	11.4	91.7	12	12.0	1.42
C15	4.89	80	64	51	607	556	9.0	19.9	27	6.3	1.09
C16	14.29	67	40	21	907	728	42.4	76.2	27	5.4	2.85

continued on next page.

organic and inorganic carbon, VSS and NVSS are volatile and non-volatile suspended solids, Ca^{2+}, Mg^{2+}, Na^{+}, and K^{+} are the cations, SO_4^{2-}, SO_4^{2-} is $^{-}$, Alkalin. is total alkalinity all in $mg \bullet L^{-1}$. TC-TA is total cations – total anions in millequivalents per liter.

NVSS	Ca	Mg	Na	K	SO4	Alkalinity	Colour	TC-TA	pH
$mg \bullet L^{-1}$	$mg \bullet L^{-1}$	$mg \bullet L^{-1}$	$mg \bullet L^{-1}$	$mg \bullet L^{-1}$	$mg \bullet L^{-1}$	$mg \bullet L^{-1} CaCO_3$	$mg Pt \bullet L^{-1}$	$mg \bullet L^{-1}$	
0.34	4.40	1.20	0.83	0.08	1.35	9.75	317	-0.1307	6.70
0.37	5.24	1.33	0.28	0.25	1.13	15.00	209	-0.0623	7.70
0.42	4.37	0.95	0.32	0.17	0.70	12.25	172	-0.0505	7.29
0.43	5.65	1.18	0.42	0.31	0.95	19.75	76	0.0128	7.86
1.13	4.12	0.94	0.41	0.22	0.82	11.25	205	-0.0601	7.08
0.91	7.30	1.59	0.37	0.09	1.55	15.25	389	-0.1755	6.69
0.83	8.06	1.91	0.54	0.43	1.05	32.50	34	0.0823	7.88
0.36	18.89	4.17	2.00	0.52	7.69	71.75	26	0.2147	8.49
1.00	10.40	2.00	0.60	0.40	2.30	36.75	105	0.0687	7.47
0.40	6.00	1.40	0.40	0.30	2.10	10.88	77	-0.1759	7.00
1.00	14.60	2.90	0.90	0.40	2.50	54.38	140	0.1287	7.82
0.90	11.00	2.40	0.90	0.50	4.20	40.50	112	0.1053	8.27
0.60	8.50	2.30	0.50	0.30	1.40	35.25	55	0.0965	7.77
0.41	27.01	7.67	6.01	0.49	48.73	59.00	196	-0.0542	7.68
1.84	6.46	1.62	0.85	0.62	2.71	16.50	240	-0.1192	8.38
2.28	7.18	1.35	0.47	0.57	1.63	16.25	331	-0.1395	7.06
0.84	6.04	1.36	0.57	0.82	2.66	15.50	250	-0.0886	6.94
1.09	7.84	2.27	0.53	0.79	2.70	27.75	112	-0.0052	7.96
1.02	12.07	3.06	0.82	0.50	7.04	25.25	480	-0.2479	7.10
2.87	10.24	2.38	0.91	0.34	1.96	26.50	396	-0.1819	7.17
0.51	5.37	1.78	0.69	1.11	9.51	1.13	424	-0.2474	4.75
0.30	5.98	1.85	0.74	0.70	5.43	16.00	212	-0.0638	6.61
1.51	6.54	1.59	1.20	0.61	7.96	6.75	456	-0.2174	6.06
0.82	10.20	2.68	2.76	0.24	16.62	15.00	523	-0.2079	6.48
0.65	7.35	1.58	0.63	0.05	1.23	16.25	312	-0.1728	7.20
0.62	6.71	1.94	1.30	0.25	1.54	26.50	112	0.0070	7.55
0.64	14.28	3.76	1.26	0.63	5.12	54.75	87	0.1132	7.93
0.81	5.67	1.47	0.81	0.33	3.49	10.00	454	-0.1741	6.60
0.89	7.91	1.98	0.51	0.29	5.34	12.75	344	-0.2161	7.26

Table 2-3: Univariate regressions for independent (X) and dependent (Y) variables.

X	Y	n	Intercept	Slope	r^2	SEr of slope	MS error	*F*	*P*
DOC	Colour	24	-156	19.3	0.89	1.44	2595	175	< 0.0001
Colour	LnSecchi	24	0.62	-0.003	0.72	0.0004	0.066	53.6	< 0.0001
Colour	ε	24	0.32	0.001	0.58	0.0002	0.026	30.9	< 0.0001
Log10Wo/Ao	DOC	24	8.7	10.7	0.62	1.78	21.4	36.3	< 0.0001
Burnt		10	10.1	12.2	0.69	2.87	17.2	17.9	0.003
Ref.		14	9.04	8.0	0.68	1.60	10.8	25.1	0.0003
Log10Wo/Ao	Colour	24	9.17	210	0.57	38.9	10205	29.2	< 0.0001
Burnt		10	41.0	248	0.78	45.9	4377	29.3	0.0006
Ref.		14	18.4	141	0.59	34.8	5131	16.4	0.0016
DI	Log10TP	15	1.52	0.005	0.74	0.001	0.038	40.58	< 0.0001
DI	Log10DP	15	1.13	0.007	0.76	0.001	0.059	43.74	<0.0001

Fig. 2-1: Location of Caribou Mountain study lakes in Alberta, Canada. Reference sites (gray circles) contain lakes with no fire in their catchments since approximately 1910, previously burnt sites (triangles) contain lakes with catchments burnt between 1961 and 1985, burnt sites (black circles) contain lakes in catchments burnt in 1995. An automated climate station (CS) operated for two years.

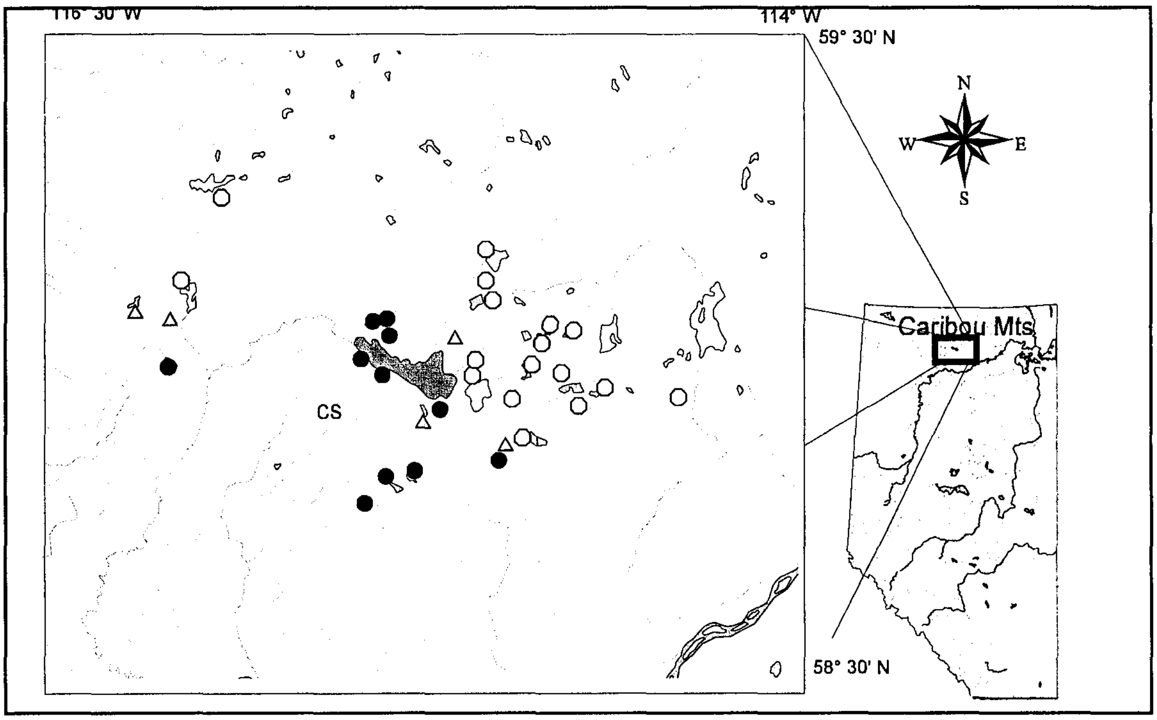


Fig. 2-2: Summer mean total base cations, total anions and pH for lake waters in reference (open boxes, $n = 15$) and burnt (shaded boxes, $n = 10$) catchments. Letters (a & b) indicate differences between reference and burn-impacted lakes ($P < 0.05$).

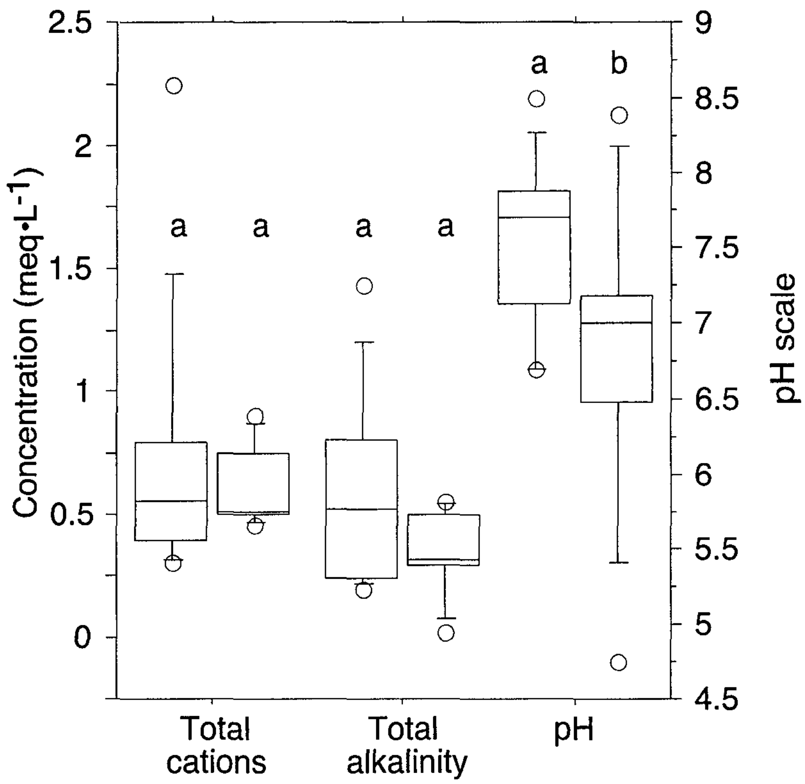


Fig. 2-3: Summer mean concentrations of TP, DP and SRP for reference (open boxes, n = 14), burnt (shaded boxes n = 10) and previously burn-impacted lakes (hatched boxes, n = 5). Letters indicate difference between reference and burnt means (P < 0.05). Analytical statistics were not applied to previously burnt lakes.

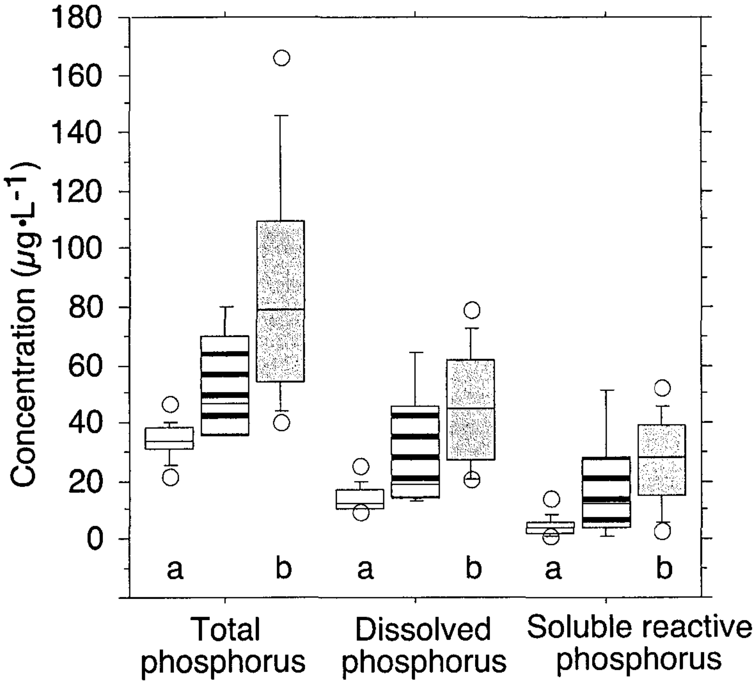


Fig. 2-4: Summer mean TN and DN for reference (open boxes,n = 14), burnt (shaded boxes, *n* = 10) and previously burn-impacted lakes (hatched boxes, n = 5). NO_3^--N and NH_4^+-N are for reference and burnt lakes only. Letters indicate differences in mean concentrations between reference and burnt lakes ($P \leq 0.05$).

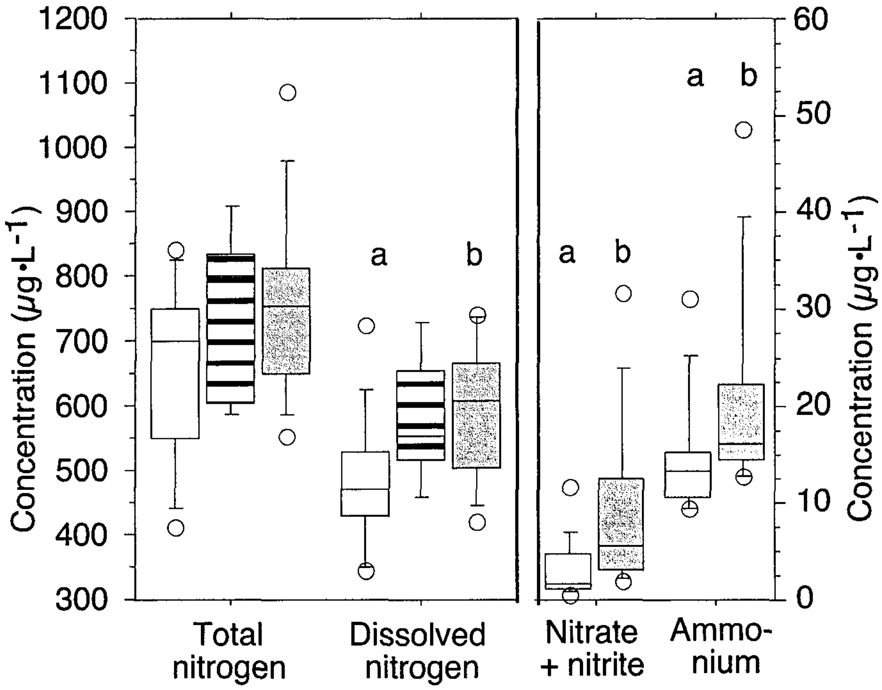
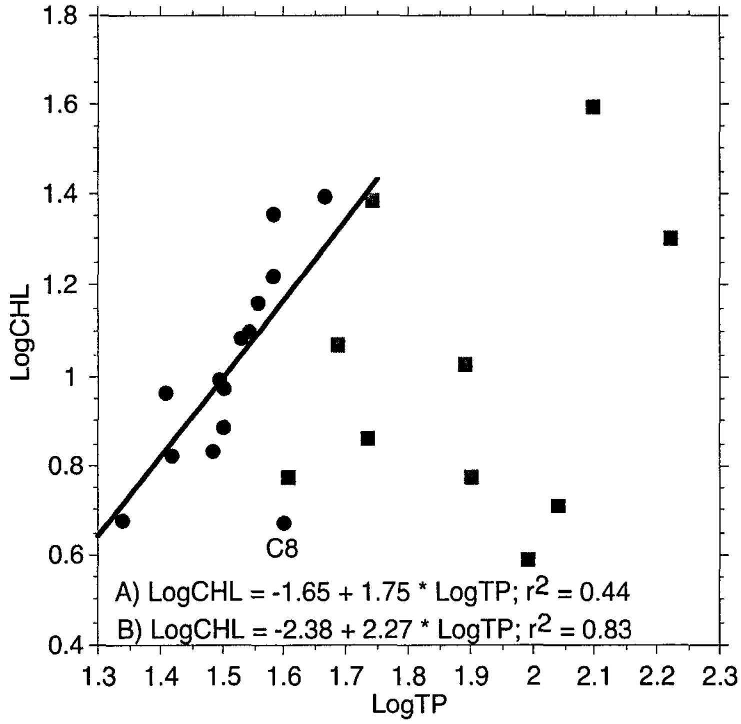


Fig. 2-5: The relationship between CHL and TP (both in μg•L^{-1}) for lakes in reference (circles) and burnt (squares) catchments. Values are $\log_{10}$ transformed, A) equation and line for all reference lakes only. B) Equation without C8 which deviates from other reference lakes because of its high colour (388 mg•L^{-1} Pt).



References

American Public Health Association. APHA 1993. Standard Methods for the Examination of Water and Wastewater. © APHA 1993.

Bayley, S.E., Schindler D.W., Beaty K.G., Parker B.R. and Stainton M.P. 1992. Effects of multiple fires on nutrient yields from streams draining boreal forest and fen catchments nitrogen and phosphorus. Can. J. Fish. Aquat. Sci. **49:** 584-596.

Clymo, R.S. 1984. *Sphagnum*-dominated peat bog: a naturally acid ecosystem. Phil. Trans, R. Soc. Lond. **B305:** 487-499.

Craig, H. and Gordon, L.I. 1965. Deuterium and oxygen-18 variations in the ocean and marine atmosphere. *In:* Satble isotopes in oceanographic studies and paleotemperatures. *Edited by* E. Tongiorgi, Lab. Geologia Nucleare, Pisa. 9-130.

Crumpton, W, Isehart, G.M., and Mithell, P.L. 1992. Nitrate and organic N analyses with second-derivative spectroscopy. Limnol. Oceanogr. **37:** 907-913.

D'Arcy, P., and Carignan, R. 1997. Influence of catchment topography on water chemistry in southeastern Quebec Shield lakes. Can. J. Fish. Aquat. Sci. **54:** 2215-2227.

Grier, C.C. 1975. Wildfire effects on nutrient distribution and leaching in a coniferous ecosystem. Can. J. For. Res. **5:** 599-607.

Halsey, L., Vitt, D.V. and Trew, D.O. 1997. Influence of peatlands on the acidity of lakes in northeastern Alberta, Canada. Water, Air, Soil Pollut. **96:** 17-38

Hill, A.R. 1993. Base cation chemistry of storm runoff in a forested headwater wetland. Wat. Resour. Res. **29:** 266-273.

Ingram, H.A.P. 1983. Hydrology. *In:* Ecosystems of the world 4A. Mires: Swamp, bog, fen and moor. Edited by A.J. Gore Elsevier, Oxford. Pp. 67-158.

Knowlton, M. F. 1984. Flow-through microcuvette for fluorometric determination of chlorophyll. Water res. Bull. **20:** 795-799.

Lewis Jr., W.M.. 1974. Effects of fire on nutrient movement in a South Carolina pine forest. Ecology **55:** 1120-1127

Li, Y. and Vitt, D.H. 1997. Patterns of retention and utilization of aerially deposited nitrogen in Boreal Peatlands. Ecoscience **4:** 106-116

MacLean, D.A., Woodley, S.J., Weber, M.G., and Wein, R.W. 1983. Fire and nutrient cycling. *In:* The role of fire in northern circumpolar ecosystems. *Edited by* R.W. Wien and D.A. MacLean. 111-132.

Prepas, E.E., Rigler, F.H (1982). Improvements in Quantifying the Phosphorus Concentration in Lake Water. Can. J. Fish. Aquat. Sci. **9(6):** 822-829.

McColl, J.G. Grigal, D.F. 1975. Forest fire: Effects on phosphorus movement to lakes. *Science* **188:** 1109-1111.

Minshall, G.W., Robinson, C.T., and Lawrence, D.E. 1997. Postfire responses of lotic ecosystems in Yellowstone National Park, U.S.A. Can. J. Fish. Aquat. Sci. **54:** 2509-2525.

Nurnberg, G. 1996. Trophic state of clear and colored, soft- and hardwater lakes with special consideration of nutrients, anoxia, phytoplankton and fish. Lake Reserv. Man. **12(4):**432-447.

Richter, D.D., Ralston, C.W. and Harms, W.R. 1982. Prescribed fire: Effects on water quality and forest nutrient cycling. Science **215:** 661-663.

Sartory, D. P. and Grobbelaar, J. U. 1986. Extraction of chlorophyll a from freshwater phytoplankton for spectrophotemetric analysis. Hydrobiology. **114:** 117-187.

Schindler, D.W., Newbury, R.W., Beaty,K.G., Prokopowich, J., Ruszcynski, T. and Dalton, J.A. 1980. Effects of a windstorm and forest fire on chemical losses from forested catchments and on the quality of receiving streams. Can. J. Fish. Aquat. Sci. **37:** 328-334.

Stark, N.M. 1977. Fire and nutrient cycling in a Douglas-fir/larch forest. Ecology **58:** 16-30.

Strong, W.L. and Leggat, K.R. 1992. Ecoregions of Alberta. Alberta Energy and Natural Resources.

Tiedemann, A.R., Helvey, J.D, and Anderson, T.D. 1978. Stream chemistry and catchment nutrient economy following wildfire and fertilization in eastern Washington. J. Environ. Qual. **7:** 580-588.

Vitt, D.H., Halsey, L.A, and Zoltai, S.C. 1994. The bog landforms of continental western Canada in relation to climate and permafrost patterns. Arctic Alpine Res. **26:** 1-13.

Vitt, D. H. and Bayley, S.E. 1984. The vegetation and water chemistry of four oligotrophic basin mires in northwestern Ontario. Can. J. Bot. **62:** 1485-1500.

Wright, R.F. 1976. The impact of forest fire on the nutrient influxes to small lakes in northeastern Minnesota. Ecology **57:** 649-663.

Chapter 3:
Assessing hydrologic controls of water chemistry with deuterium and oxygen-18 mass balance in a multi-lake survey comparing burnt and undisturbed catchments.

Introduction

Water chemistry surveys that include a number of lakes are essential in developing theory and management in aquatic science (e.g., Rigler 1982). Patterns in water chemistry are usually compared or contrasted on hypothesized gradients of geology, climate, vegetation or anthropogenic impact (Deevey 1940; Rochelle 1989). The Gordian knot in such surveys is controlling differences in hydrology among catchments, an insurmountable crux in remote areas where little or no hydrologic data are available (e.g. Gibson et al. 2002). While catchment features such as slope and vegetative cover have been widely used to examine potential landscape-water chemistry relationships (e.g., Prepas et al. 2001; McEachern et al. 2000), analysis of direct hydrologic patterns has often been hindered by lack of runoff data in many remote areas of Canada's boreal forest.

Recent studies have applied models that incorporate enrichment rates of the naturally occurring stable isotopes of water, deuterium (^2H) and oxygen-18 (^{18}O), to indirectly estimate catchment runoff to lakes in surveys of headwater (Gibson et al. 2002) and non-headwater systems (Gibson and Edwards 2002). The isotope mass balance approach has been validated in comparison with standard water and energy budget methods in a variety of ecoclimatic regions in northern Canada (Gibson 2002, Gibson et al. 1993, 1996, 1998). The method is a practical field-based alternative for water balance characterization in remote areas because it can be readily incorporated into water quality surveys. Prepas et al. (2001) and Gibson et al. (2002) demonstrated the potential of the technique for distinguishing systematic chemical-hydrologic patterns in wetland versus upland and bog versus fen dominated catchments. In this paper we review the application of stable isotope mass-balance methods for determining hydrologic indices in regional surveys and discuss the hydrology-water chemistry linkages for a case study of 24 lakes where water chemistry was previously compared between burnt and unburnt catchments (McEachern et al. 2000). The hydrology of the lake catchments was complicated by low topographic relief, peatland-dominated vegetation, deep organic soils and discontinuous permafrost. In addition, 10 of the survey lakes resided in catchments that were burnt by a forest fire. We demonstrate how isotope-based hydrologic indices can be applied to investigate the causal relationship between hydrology and lake chemistry, including the influence of forest fires. We also discuss the significance of uncertainty in some of the parameters used in the isotope balance model.

As a corollary to this analysis we develop a contributing and effective area model for linking patterns in water chemistry to variable hydrologic conditions and contributing areas and compare results with nutrient concentrations in the survey lakes. According to Soranno et al. (1996), contributing areas and effective areas are defined respectively as zones that produce runoff and zones that become saturated and deliver solutes. Contributing areas express differences in runoff generation and groundwater recharge or discharge among catchments and can include transbasin transfers of groundwater that can vary widely among lake basins (Shaw and Prepas 1990). Spatial variation in contributing area must be considered both within and among catchments when modeling runoff generation and solute transport to effectively describe surface water chemistry (Beven and Kirkby 1979). The location and temporal fluctuation of variable contributing areas may be particularly important when runoff is routed through soils such as wetlands that form a minor proportion of the total catchment but have an overbearing influence on runoff chemistry (Beven and Kirkby 1979, Hill 1993). Finally, we present and discuss contributing and effective drainage areas as predictors of nitrogen, phosphorous, and carbon concentrations in the survey lakes with the intention of improving our ability to explain patterns in lake chemistry and changes following forest fire.

Site Description

The Caribou Mountains (59° N 115° W) are an erosional remnant forming a large, relatively flat plateau 500 m above the Peace River valley in Alberta, Canada (Fig. 3-1). Peatlands cover 56% of the Caribou Mountains and are underlain by poorly-drained cryosolic and brunisolic soils (Strong and Leggat 1992). Subsurface geology is shale, feldspathic sandstone, and siltstone of deltaic and marine origin. Open forest of black spruce (*Picea marianna*) and an understory of mixed *Sphagnum* spp., feathermosses, brown mosses and lichens dominate vegetation. Permafrost commonly occurs at depths ranging from 0.5 to 0.75 m beneath the organic soil surface within continental bogs (Strong and Leggat 1992), which cover an average of 62% of catchments examined in this study. Collapse scars signify degraded permafrost (Vitt et al. 1994) and were present in bogs covering an additional 19% of the study catchments. Mean May through August temperature (1996-98) was 10.2 °C (automated climate station), typically with less than 800 growing degree days and between 400 and 450 mm of annual precipitation (Strong and Leggat 1992). During 1995, a 129 000 ha forest fire incinerated understory herbaceous, sedge, and lichen cover but damage to moss and peat varied. Moss and peat damage from burning was mostly limited to the upper 0 to 20 cm. Permafrost, detected by manual probing and digging, was present in all five unburnt continental bogs surveyed during late August but was not present in the peat layer (down to mineral soil) at five burnt sites.

Twenty-four headwater lakes were selected from an investigation of fire induced changes in water chemistry (McEachern et al. 2000). Ten lakes were within catchments where 60 to 100% of the tree cover was killed by fire (mean 83%, *s.e.*5%) and 14 lakes were in unburnt reference catchments (Fig. 3-1). Lake surface areas ranged from 2.6 to

1173 ha and mean depths from 0.3 to 11.6 m (Table 3-1). All lakes were either polymictic or if thermally stratified (during July for deeper lakes), mixed conditions were restored by the next sampling event (August). Throughout the summer all lakes had dissolved oxygen concentrations greater than 5 $mg \cdot L^{-1}$, as measured 0.5 m above the lake bottom, indicating generally mixed vertical profiles in the lakes.

Methods

Watersheds were delineated from 1994 series 1:20000 aerial photographs. Ground cover in each catchment was classified as upland, peatland, or open water. Peatlands were subdivided into veneer bog, peat plateau, poor fen and rich fen based on vegetation and slope characteristics identified from the aerial photographs (Halsey et al. 1997). The catchment slope index was calculated by computing elevation gain divided by linear distance to lake shore (CS1: D'Arcy and Carignan 1997). Bathymetric maps were constructed from depth measurements along 5 to 15 transects on each lake and became the basis for estimating lake volume (V). Lake water levels were assumed to remain constant through the sampling period.

The lakes were sampled monthly after ice-off in late June to late August/early September 1997. A 12-L euphotic-zone, composite water sample was collected with multiple hauls of a polyvinyl chloride tube to the depth of 1% light penetration or 1 m above the lake bottom. Light penetration was determined during each lake visit with a Li-Cor LI-185 equipped with surface and submerged LI-192SB sensors. Three opaque 2-L polyethylene bottles were rinsed and filled with the composite sample. The polyethylene bottles were previously acid washed (10% HCl) and rinsed 5 times with deionized-distilled water. A more detailed description of sample collection was presented elsewhere (McEachern et al. 2000).

Three streams, four fen and six bog sites were also sampled during the summer of 1997. Two streams, MS1 and MS2, drained adjacent unburnt watersheds of 40 and 6 km^2, respectively. Stream MS3 drained a 1 km^2 watershed that was entirely burnt. Stream flows were measured with a Girley meter and weighting rod at 3 to 6 points across the channel depending on width. A depth-discharge curve was created for MS2. A pressure transducer was installed in MS2 for discharge calculations every 15 minutes through the summer. Runoff during snowmelt was not captured in its entirety, however, discharge data for the rising and initial falling limbs of the hydrograph were approximated from records obtained for the nearest Water Survey of Canada gauging station, and corrected for basin area. Samples were collected in 2-L polyethylene containers from the center of each stream channel at approximately 50% depth in MS1, MS2 and MS3 on four sample dates that included bank-full and baseflow conditions. An automated ISCO water sampler was installed at MS2 and collected samples at 3-h intervals during the only significant storm event of the 1997 summer. Depression storage and surface water was also collected from standing pools on bogs and from outflows in fens, respectively.

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Twice daily during May to September 1988 and 1989, precipitation amount was recorded and water samples were collected from a standard metric rain gauge with funnel and stand at the Foggy Mountain fire tower located at the southern edge of the Caribou Mountains. Precipitation monitoring was conducted under the auspicies of the Alberta Forest Service climate network, and data are routinely checked for quality by comparison with neighboring towers before being released to the public. Water collected from the gauge was combined in a high-density polyethylene bottle, sealed and immediately frozen. To minimize isotopic fractionation effects, ^{2}H and ^{18}O were analyzed from a sub-sample taken after complete melting and mixing of water in the sealed container. The fire tower was also equipped with a Sacramento gauge from which seasonal precipitation depths were verified. Precipitation samples were collected in 1997 during the lake study but were lost due to vandalism. Snow depths were measured monthly at one site from 1996 to 1999. Four hundred snow samples were also collected with a 10-cm diameter corer along 20 transects during late March of 1999 and 2000. Snow samples were individually bagged and weighed later that day with a digital scale accurate to 0.01 g. Mean snow density was calculated and applied to the 1997 snow depth data to determine snow water equivalent. Randomly selected snow samples from single transects were melted, combined and processed for chemistry analysis following procedures for other water samples.

Groundwater was collected in 1998 and 1999 from wells installed between 1 and 3 m into mineral soil at hill slope sites below the Caribou Mountain lakes. The wells were constructed of 5/8 inch OD PVC with 0.01 mm wide horizontal slits every 6 mm along the bottom 18 cm of the well (Evans et al. 2000). The wells were wrapped in polyester fabric (NILEX MD7407) and installed in augured holes. The wells were capped to minimize evaporation and were completely purged prior to sample collection with a vacuum pump attached to a side-arm flask. The flask was rinsed twice with initial water and samples were collected between 10 min to 1 h after purging depending on recharge rates. Groundwater samples were analyzed for ^2H and ^{18}O only.

Water samples for ^2H and ^{18}O analyses were decanted to new and dry 30□mL polyethylene (HDPE) bottles and returned to the University of Waterloo for analysis by IRMS, observing appropriate quality control procedures (Gibson et al. 2002). Manually-collected water samples for chemical analyses were processed and preserved the same day for later analysis at the University of Alberta. Water samples collected by the ISCO automated sampler at MS2 were similarly processed within three days of collection. Triplicate 25-mL aliquots were pipetted into 30-mL glass tubes used directly in digestion processes for total and dissolved nitrogen (TN, DN) and phosphorus (TP, DP) analysis. Unfiltered water was used for TN and TP analyses. Filtrate (Millipore HA, 45 µm and Gelman magnetic filter tower) was used for DN and DP analyses. TN and DN samples were preserved with 10 µL of 40% H_2SO_4 for storage (1-2 wk), and neutralized with equivalent NaOH prior to digestion in the laboratory. TP and DP were analyzed spectroscopically (5-cm cell) from persulfate-oxidized samples by molybdate blue absorption (Prepas and Rigler 1982). TN and DN concentrations were determined by second derivative spectroscopic analysis of persulfate oxidized samples (Crumpton et al. 1992). Sodium (Na^+), chloride (Cl^-) and dissolved organic carbon (DOC) were analyzed from GF/F filtrate, stored in 60-mL polyethylene bottles. Sodium samples were preserved with 60 µL of 40% H2SO4 , whereas DOC and Cl^- samples were refrigerated. Sodium was analyzed by atomic absorption flame spectroscopy (Perkin-Elmer AS90, AA3300), DOC by high temperature catalysis (Shimadzu TOC-5000), and Cl^- concentrations by chromatography (Dionex 2000i/SP). Eight field blanks of laboratory deinoized distilled water were treated like surface water samples. Of the 96 nitrogen and phosphorus tubes containing field blanks three demonstrated contamination and were easily identified from the triplicates. No contamination was detected for ions or DOC.

Isotope mass balance

Lakewater residence times (t; yr) were calculated for the 24 lakes with a steady-state isotope mass-balance model. The technique is based on the principle that water molecules containing ^2H and ^{18}O are heavier and evaporate at a slower rate from open water bodies than their ^1H and ^{16}O counterparts (Gat and Gonfiantini 1981). Enrichment in heavy isotopes relative to background levels in local precipitation is therefore directly related to the fraction of water lost by evaporation versus liquid outflows, and is closely related to lake water retention time in the present setting. In this paper, isotope mass balance is applied using observed signatures of lake water oxygen ($^{18}O/^{16}O$) and hydrogen ($^2H/^1H$) stable isotope ratios along with supplementary climate data using a steady-state approximation (see Gibson et al. 2002). The isotope mass-balance model, including selected results for the present survey, have been presented in detail elsewhere (Gibson et al. 2002) and a brief summary of the approach is presented below.

The isotope balance for a constant volume lake is given by:

$$I\delta_I = Q\delta_Q + E\delta_{E,} \qquad (1)$$

as derived from the water balance equation, I = Q + E, where the major components of the water balance, total inflow (I), total outflow (Q) and lake evaporation (E), in $m^3 \bullet yr^{-1}$ are multiplied by their respective isotope compositions ($\delta^{18}O$ and δ^2H, ‰ Standard Mean Ocean Water or SMOW). The isotope composition of outflow is assumed equivalent to lake water (δ_L) for well-mixed lakes, and the isotope composition of evaporated moisture (δ_E) is derived with a simplified, one-dimensional Craig-Gordon model for free-surface evaporation (Gat 1995; Gibson et al. 1993). Rearranging (1) and substituting the simplified Craig-Gordon equation for δ_E yeilds:

$$\frac{E}{I} = \frac{(\delta_L - \delta_I)}{m(\delta^* - \delta_L)}, \quad (2)$$

which is dependant on the the isotope composition of lake water and total inflow (δ_L and δ_I, respectively), enrichment slope m (= $h(1-h)^{-1}$), and limiting isotopic enrichment δ^* (= $\delta_h + \varepsilon h^{-1}$), h and δ_h being the ambient atmospheric humidity and its isotope composition, respectively, and ε being the total (equilibrium + kinetic) liquid-vapour isotopic separation which is temperature T and humidity h dependant (see Gat 1995). For the present analysis, the parameters h and ε were spatially interpolated accounting for elevation based on RH and temperature records from Environment Canada climate stations in the region. A flux-weighted algorithm was used to determine h, T and δ_h (see Gibson et al. 2002).

Mean annual isotopic composition in each lake (δ_L) was determined from lakewater samples collected monthly from the beginning to the end of the ice-free period (June through early September). Potential errors in δ_L derive from failure to adequately represent the open water season mean. For example, in stratified lakes, surface waters will become enriched compared to the hypolimnion (Murthy et al. 2002). Surface water samples from stratified lakes will subsequently overestimate retention time and underestimate water yield. Such errors are minimized in the present study as stratification in the study lakes lasted less than one month. Furthermore, June to August differences in†d_L were less than 0.5 0 in ^{18}O for each lake.

The isotope composition of input (δ_I), which is often close to the isotopic composition of precipitation (δ_P), is estimated using several approaches. Long term values of δ_P are estimated graphically as the intersection of the Local Evaporation Line (LEL), determined by linear regression in ^2H-^{18}O space using the mean d_L values determined for all study lakes, and the meteoric water line MWL of Craig (1961). In this case, δ_P is approximately weighted over the average residence time of the lakes (~2 years for this study). A comparative estimate of δ_P is obtained by direct measurement of the isotope content in summer precipitation and snowpack samples, volumetrically weighted for relative contributions to total precipitation. d_P is also modeled by interpolating values from the Global Network for Isotopes in Precipitation database (1998, International Atomic Energy Agency/ World Meteorological Organization, Vienna, Austria). As discussed later on in the paper, the various estimates of d_P are similar, yet produce important divergence in water yield estimates that are useful for gauging sensitivity of the method. In general, graphically-determined estimates of d_P are expected to be most appropriate for lakes with long retention times or where inflows are dominated by local groundwater. Modeled d_P can be determined for time periods as short as one-month (and can be adjusted iteratively to match calculated residence times) but may be prone to uncertainty in interpolation of model parameters. Direct measurement of d_P is the most labour-intensive approach but provides an important baseline for validation of the other methods. Substituting E/I estimates from Eq. (2), the lake water residence time (τ) is determined using:

$$\tau = \frac{\frac{E}{I} * V}{E_L * A_o} \qquad (3)$$

where V is lake volume (m^3), E_L is long-term mean annual lake evaporation rate ($m \cdot yr^{-1}$) for the region (Environment Canada Climate Records) and A_0 is lake surface area. Both lake volume and surface area were assumed to remain constant over the study period. Other presentations of Eq. 3 substitute the equivalent term *mean lake depth* for V/A_o (Gibson et al. 1996, Prepas et al. 2001). The equation adequately represents steady state isotopic enrichment in throughflow and terminal lakes but is not suited to lakes which have substantial volume fluctuations greater than about 5% (Gibson et al. 2002). All of the sampled lakes had continuous outflows and were assumed to be throughflow lakes with minor volumetric fluctuations.

Catchment runoff, contributing and effective areas

Catchment runoff (R) was calculated from lake volume divided by the isotope-based estimate of lake retention time, and measured drainage basin area (DBA, ha). Contributing area (CA, ha) was calculated as:

$$CA = DBA * \frac{R}{P} \qquad (4)$$

where R and P are mean water yield and precipitation (both $mm \cdot yr^{-1}$), respectively. Contributing area within a catchment can change in size in response to precipitation, which is consistent with its broader definition (Beven and Kirkby 1979) but a departure from the application by Soranno et al. (1996) where the contributing area was fixed by catchment topography at a potential maximum for surface runoff. In Eq. 4, contributing area is analogous to the *effective drainage basin area* of Gibson et al. (2002). Contributing area may be useful for addressing among lake patterns in water chemistry because it should be a better representation of the potential linkage between a lake and its catchment than the topographic drainage basin area.

In water quality surveys, water chemistry variations among lakes are often evaluated against differences in size of the drainage basin area. Contributing area refines this concept by linking the lake directly to a field-based (isotopic) estimate of source area. The general hypothesis that lake water chemistry is associated with contributing area assumes flux rates and transport of solutes will be similar among the various catchments. This may not be the case in surveys where soils or other catchment characteristics are inhomogeneous. In particular, complications may arise if the catchments contain significant and differing proportions of surface detention storage, i.e. evaporating surface pools that may retain solutes, alter organic content, and enrich the isotope content of runoff, the latter of which will lead to underestimation of contributing area. In the present survey, conducted within a region with a deep but variable organic layer, as well as variable micro-topography and patchy occurrence of clayey mineral soils, the flux rates of mineral and organic solutes are expected to be highly variable among contributing areas.

A further refinement to address this expected variability among catchments is the *effective area*, which attempts to directly relate this variability in soil flux and transport among catchments by including a chemical tracer. The equation for solute loading proposed by Soranno et al. (1996) can be rearranged to reflect contributions from an unknown effective area (EA, m^2) with known total load. Total load was calculated as the product of the isotope estimate of calculated lake inflows (Q_I, $m^3 \bullet yr^{-1}$) and observed lake concentration ($mg \bullet m^{-3}$) which was divided by flux rate (F, $mg \bullet m^{-2} \bullet yr^{-1}$) and transmissivity (T, dimensionless) to obtain a spatial source for the solute:

$$EA = \frac{Q_I[Na^+]}{\sum_{i=1}^{n} \frac{A_i * F_i * T_i}{CA}} \tag{5}$$

where A_i is the total area of a distinctive soil in the contributing area with flux rate F_i and transmissivity T_i. If effective area is calculated from a conservative element such as Na^+ then T approaches 1, the•A_i for *n* distinct soils equals the contributing area size and variation in effective relative to contributing areas must be related to differences in F_i. Thus, among-catchment variation in effective area derived from Na^+ represents differences in soils through which water flows. The equation assumes Na^+ is a conservative element at equilibrium in the lake and its concentration is directly dependent on loading from the catchment.

In our study, the parameters A_i, and F_i, were not known, and T_i was assumed to be 1. An approximation for the combined AFT term of 4.34 $mg \bullet m^{-2} \bullet yr^{-1}$ (0.189 $meq \bullet m^{-2} \bullet yr^{-1}$) $[Na^+]$ was chosen based on a mean of reported flux rates for similar soils in an arctic basin (Buttle and Fraser 1992) and from north-central Alberta (Shaw and Prepas 1990).

The arbitrary selection of flux makes quantitative values of effective area rough estimates only, however, the issue for this study was to describe variation in an unknown effective area supplying solutes to each lake. Variation in effective area size was properly captured if the assumption that Na^+ loading reflected variation in A_i, and F_i, among basins. Na^+ flux rates were not adjusted for potential impacts of fire. Changes in Na^+ flux rates measured from podsolic soils after forest fire in Montana and burning of slash in British Columbia were reported as insignificant (Stark 1977, Feller and Kimmins 1984), however, these studies recognized accelerated Na^+ loss from some sites may have been obscured by high variability. The purpose of introducing effective area in this paper is to propose a potential refinement to the CA concept used in some lake surveys (Prepas et al. 2001). Due to the many assumptions required at this stage, the effective area calculations should only be viewed as a potential tool whose efficacy should be determined by its ability to explain among lake variability in water chemistry.

The adopted effective area model assumes flux and transmission of target solutes behave in fashion similar to sodium. The model may be applicable to non-conservative substances such as phosphorus, nitrogen and carbon if differences in T_i and generation or removal in the lake can be described. Transmission of non-conservative substances may be predominately determined by differences in weather and can approach one even for particulate bound phosphorus (Soranno et al. 1996). However, if T_i varies widely, the model will not add explanatory power to inferences drawn from contributing area. To address the additional issues of sedimentation and resuspension in lakes, analogues such as lake surface area to mean depth (Osgood 1988) were included. These results are not reported because they did not improve explained variance in nutrient concentrations.

Data handling

Data were tested for deviation from a normal distribution with Kolmogorov-Smirnov at $P \leq 0.05$. Data were either normally distributed or were transformed by $\log_{10}$

or to the negative arcsine square root to meet assumptions for normality prior to analysis by *t*-test, Pearson's correlation and univariate least-squares regression. In some cases, normal data were log transformed to produce linear relationships from variables related by power functions. Nonparametric Mann-Whitney tests were employed when mean values were calculated from fewer than 10 data points. Probability values and correlation coefficients are reported for these tests. Standard error values are reported as ± values in parentheses after mean values. Intercept (α) and slope (β) values are reported from regression analyses. All analyses used Statistica 4.1 (© StatSoft, 1994) and StatView 4.5 (© Abacus 1995) for Macintosh.

Results

Isotope composition of precipitation and inflow

Evaporation from the lake surface causes progressive enrichment in heavy isotopes as retention time increases, with overall buildup beyond δ_I controlled by the *E/I* balance of individual lakes. In a case where significant open-water evaporation does not occur from the catchment, δ_I will be close to δ_P (Gibson et al. 2002). An accurate estimate of δ_P is therefore an important baseline for calculations of water retention time. Long-term δ_P , determined graphically as the intercept of the LEL and MWL of Craig (1961, $\delta^2H=8\ \delta^{18}O+10$), was found to be -165.0 in δ^2H and -21.9 $\delta^{18}O$ for the entire survey of Caribou Mountain lakes (Fig.□2)This long-term δ_P estimate is an average incorporating differences in precipitation at individual lake catchments related to elevation, and variations in groundwater or runoff sources. The confidence of the intercept provides a useful measure of potential error in δ_P due to variability in these sources among catchments. The 95% confidence intervals for the LEL - MWL intercept (Fig. 3-2) are ±6.6 for δ^2H and ± 0.88 for $\delta^{18}O$ corresponding to a potential error of 8% in the graphical δ_P prediction.

We also calculated δ_p from a combination of precipitation measurements. Mean summer precipitation was found to be close to -138 in δ^2H and -19.0 ‰ in $\delta^{18}O$. Snow samples were measured at -193‰ in d^2H and -25.1‰ in $\delta^{18}O$. Snow depth measurements combined with snow water equivalent data suggest snow typically comprised 20% of annual precipitation. Measured δ_p at the sampling station (Fig. 3-2) was therefore close to -149‰ in δ^2H and -20.2‰ in $\delta^{18}O$. These values are enriched by about 9% (1.7‰ and 16‰, respectively) compared to the graphically determined values. This difference may largely be the result of slight interannual variability in the isotopic composition of precipitation. It may also reflect significant measurement error due to evaporation from the rain gauge prior to sampling, or problems with the snow-rainfall weighting approach.

Monthly isotope data for precipitation were also available from climate stations at Edmonton and Ft. Smith as part of the Global Network for Isotopes in Precipitation (Birks et al. 2002). Amount weighted values were spatially interpolated for each study lake to account for latitudinal gradients across the region. The resulting modeled value for δ_p was subsequently corrected for elevation effects. Variability in modeled δ_p for the Caribou Mountain catchments was minor, from about -154 to -153 ‰ for δ^2H and -20.5 to -20.4 ‰ for $\delta^{18}O$ with means of -153 and -20.4 ‰, respectively. The average modeled values were enriched 7% (1.5 and 12 ‰, respectively) compared to the theoretical long-term values, and only slightly depleted compared to the measured values (0.2 and 4 ‰, respectively).

Groundwater and runoff inflows to lakes can generally be assumed to have an isotope composition similar to local δ_p except in the case of non-headwater systems where significant evaporation can occur in upstream lakes, or in lower-lying catchments where regional sources of groundwater may be significant (see Gibson et al. 2002). The assumption of similarity is required to allow substitution of δ_p for combined inputs (δ_I) in Eq. 1. Sampling of springs and wells below the Plateau indicated a median groundwater isotope composition (δ_g) of -148 ‰ for δ^2H and -19.0 ‰ for $\delta^{18}O$ and the mean was not significantly different from modeled or observed δ_P (t-test$_{(0.05, 21)}$, $P > 0.7$) but was enriched 12% over graphically determined δ_P (t-test$_{(0.05, 15)}$, $P << 0.01$). The mean volumetrically-weighted isotope composition of stream water in MS2 was -150 ‰ for δ^2H and -19.5 ‰ for $\delta^{18}O$. Observed and modeled δ_P both closely approximated inputs from groundwater and summer surface runoff. Similar agreement was found for a survey of lakes in Alberta south of the Caribou Mountains (Gibson et al. 2002). However, four groundwater samples were less enriched than modeled δ_P (Fig. 3-2) and resembled long-term graphically-determined values suggesting that some groundwater sources may be systematically depleted in heavy isotopes due to age or preferential recharge during snowmelt. Equivocal selection of a δ_I value for calculating water residence time (τ) was therefore not possible from the Caribou Mountains data, but the various estimates are useful for contraining uncertainty.

Lake and catchment indices

Lake water residence times (τ) calculated from modeled δ_P underestimated those calculated from the graphical method by between 20 and 50% with a mean underestimation of 35% for all lakes. Fractionation of δ^2H is approximately 8-fold more sensitive to evaporation than fractionation of $\delta^{18}O$. Comparison of estimates from the two independent tracers provides another measure of model uncertainty. Hydrologic indices (τ and R) calculated from graphical and modeled δ_P differed widely (Table 3-1). Consistently lower τ from $\delta^{18}O$ compared to δ^2H may indicate that inflow to the lake was slightly isotopically enriched by evaporation. When calculations are based on graphically-determined δ_P values, lake residence time (τ) calculated from $\delta^{18}O$ was consistently lower by an average of 17% and water yield (R) was consistently higher by an average of 30% compared to values calculated from δ^2H. Use of modeled δ_P values resulted in closer agreement between τ estimates for the two tracers (to within about 9%), suggesting that these values are a more accurate representation of inflow to the lakes. Similar, but slightly poorer agreement was obtained using the measured values of δ_P.

Estimates of τ for the 14 reference lakes averaged 2.4 ($\pm$ 0.6) y and 1.7 ($\pm$ 0.4) y when calculated by the graphical and modeled approaches, respectively (Table 3-1). Estimates of *R* for reference catchments averaged 110 ($\pm$ 23) mm and 205 ($\pm$ 39) mm, respectively using the two approaches. Gauged water yield from the MS2 catchment was 114 mm•y^{-1}. Water yields for the region have been estimated at between 75 and 150 mm•y^{-1} (Fisheries and Environment Canada 1978). Interestingly, measured and regional water yields are in closer agreement with isotope-based estimates of *R* using the graphical technique, rather than the other methods. For comparison, estimates of τ and *R* are included in Table 3-1 for both the graphical and modeled approaches. As discussed later, this serves to bracket potential errors associated with the choice of δ_I and will be used to argue that such uncertainty does not limit the ability of the method to characterize hydrologic variability.

Mean τ for the 10 burnt lakes is estimated at 1.0 ($\pm$ 0.2) y and 0.6 ($\pm$ 0.1) y using graphical and modeled δ_P values, respectively. Both approaches suggest shorter residence times in burnt versus reference catchments ($P < 0.05$, Table 3-1). Note that shorter mean values of τ in burnt lakes is a reflection of the smaller volume of this sub-set of lakes rather than any evidence of fire impact on the lake hydrology. Water yield (*R*) from burnt catchments averaged 120 ($\pm$ 27) mm and 226 ($\pm$ 52) mm, respectively using graphical and modeled δ_P. In contrast, *R* was not detectably different in burnt versus reference basins for both estimates ($P > 0.7$). Hence, the hypothesis that runoff is elevated in burnt catchments was not supported by the mean isotope-based estimates of water yield. This may reflect insensitivity to changes in hydrology in the basins which were characterized by poorly connected drainage from relatively small contributing areas.

Contributing areas (CA) ranged from 6 to 4700 ha in size while effective areas (EA) ranged from 0.4 to 260 ha in size (Table 3-1). Contributing areas calculated from the graphical approach were an average of 24 and 25% of drainage basin areas for reference and burnt catchments, respectively. Contributing areas calculated from the modeled approach averaged 40 and 48% of reference and burnt drainage basin areas, respectively. Average Effective areas varied from about 2 to 3% of the drainage basin area (DBA) depending on the choice of approach and no significant mean differences were noted for reference and burnt catchments. Analysis of the overall range in contributing area and drainage basin area estimates for the study catchments may suggest that runoff is produced mainly from small hydrologically active areas. This is apparent as CA/DBA generally declines as DBA increases ($r = -0.68$). Further correlations between watershed parameters and hydrologic indices are provided in Table 3-2 and are discussed below.

Implications of hydrologic connectivity on lake water chemistry are expressed in many models with loading ratios which include a measure of catchment area, representing allochthonous load and a measure of lake size such as volume for dilution. The drainage ratio (DBA/V) assumes the entire catchment influences lake water chemistry. Catchment water yield R was found to decline with increasing drainage ratio in all lakes (Table 3-2). For example, lakes with drainage ratio greater than 8 (median drainage ratio for all lakes) had a mean water yield of 74 mm, compared to 162 mm for lakes with a drainage ratio less than 8. Comparison of water yield and drainage ratio in reference and burnt catchments (Fig. 3-3), reveals a steeper negative slope for burnt catchments (ANCOVA, $P = 0.04$, Table 3-2). Importantly, fire appears to have differently impacted water yield depending on the drainage ratio of the system. In burnt catchments we note as much as a 3-fold increase in water yield from catchments with a drainage ratio of 1, but this effect diminishes as drainage ratio approaches 10 and intersected with reference catchments as drainage ratio approaches 100. In burnt areas, an overall increase in water yields were detected for lakes that had small catchments relative to lake size, presumably because hydrologic connection to the lake was high. Fire also had an apparent impact on solute loading to these lakes as discussed by McEachern et al. (2000). Additional evidence of fire impacts on water chemistry from comparing isotope-based hydrologic indices is presented below.

Variation in lake water chemistry due to hydrology

Burnt catchments contained several fold higher concentrations of nitrogen, phosphorus and dissolved organic carbon compared to lakes in unburnt catchments (McEachern et al. 2000). In this and other regional surveys of lakes (e.g., Prepas et al. 2001) the variation in lake water chemistry due to hydrological differences among catchments was explored to assist in detecting natural versus disturbance impacts due to fire or harvesting. Here we focus on assessing the potential of the isotope-based hydrologic indices for interpretation of the lake chemistry data. Overall, we expected to find poor relationships between nutrient concentrations and drainage ratios, as contributing area and effective area formed small proportions (less than 50% and 8%, respectively) of the drainage areas. Modified drainage ratios replacing drainage basin area with the isotope-based indices contributing area and effective area were calculated for each catchment (Table 3-3) because these should better represent the source areas of water and solute delivery to the lakes. Slope and peatland coverage are also given for comparison (Table 3-3). As contributing area and effective area drainage ratios were found to be highly correlated with the topographic drainage ratio (Fig. 3-4), significant improvements in explaining variance in lake nutrient concentrations were not expected.

Variation in total phosphorus concentration among lakes was in fact best described by the CA/V ratio in unburnt catchments (Table 3-3). Dissolved phosphorus was best described by the DBA/V for lakes in unburnt catchments. No significant relationships were found between TP or DP and any of the hydrologic or physical indices for lakes in burnt catchments. The relatively poor performance of both CA/V and EA/V, as well as DBA/V to explain the observed variance may suggest that in-lake biological and sedimentation processes rather than catchment hydrology were the dominant controls on

TP and DP concentrations. Although they provide no conclusive proof, isotope-based indices are consistent with the physical evidence pointing to predominantly non-hydrologic control mechanisms for phosphorous in Alberta's productive lakes (e.g. Riley and Prepas 1984).

Similarly, variation in TN and DN among lakes in unburnt catchments was found to be unrelated to any of the hydrologic or physical indices (Table 3-3). However, in burnt catchments, among lake variation in TN was found to be inversely correlated (r = -0.86) with the percentage of peatland cover in the basin. DN among burnt-catchment lakes was also positively correlated with the isotope-based CA/V index (r = 0.74; Table 3-3). Catchment slope explained additional DN variation (r = -0.53). The negative relationship between peatland cover and TN was consistent with near complete nitrogen retention in bogs (Li and Vitt 1997), which formed a majority of peatland cover. The relationship between DN and CA/V may reflect contributions from mineral sources through fens in low slope discharge zones. Overall, the strong relationships between CA/V, slope and DN for burnt peatland-rich catchments suggests that fire had a strong impact on catchment hydrology and nitrogen loading.

Variation in lake water concentrations of DOC were best explained by DBA/V for lakes in both unburnt and burnt catchments (Table 3-3). All drainage ratio indices were similar in their ability to explain variation in DOC concentration, with notably good correlations observed between EA/V and DOC for burnt catchments (r^2 = 0.85). Slope (r = -0.24, partial correlation at step 0) and percentage of peatland cover (r = 0.03, partial correlation at step 0) explained additional variation in DOC for lakes in burnt catchments. As with DN, the strong relationships between DOC, isotope indices, slope and peatland cover indicate fire caused increased loading of organic carbon from peatlands, particularly from low-slope discharge zones.

Discussion

Expansion of human disturbance by forestry and other resource activities in northern boreal forests has resulted in concern over the sustainability of water quality in lakes and other aquatic ecosystems (Carignan and Steedman 2000). Recent contributions to our understanding of forest hydrology and the potential impacts of forest disturbance on hydrologic processes have been reviewed with the caveat that a basic understanding is still lacking (Buttle et al. 2000). One of the largest obstacles to integrating hydrological processes in surface water chemistry studies is the insurmountable difficulty of measuring these processes at the catchment scale for multiple lakes. Design of hydrologic networks for investigating forest fire impacts are even more problematic because the studies are usually opportunistic, occurring after the disturbance. Thus studies of fire impacts require a large survey of surface water conditions before any statistical power is realized. Hydrologic networks are seldom available to complement large regional studies and when they are, interpretation can be complicated by position and small footprint of disturbance relative to the hydrologically active areas of the catchment (e.g., Hornbeck et al. 1993, Buttle and Metcalfe 2000). Regional studies of lake water chemistry in Canada have instead relied on catchment characteristics such as slope and vegetation as proxies for hydrological processes, which have been applied with some success (D'Arcy and Carignan 1997, Prepas et al. 2001). Unfortunately, many boreal catchments, particularly in western Canada are characterized by low slope and poor drainage through forested peatlands where variation in vegetation may be a poor representation of dominant hydrological processes. Herein we show that the field-based indices of hydrologic linkage between lake and catchment derived from isotope mass balance may be a practical approach for examining hydrologic variability in boreal forest catchments, including those affected by disturbance.

A majority of studies indicate fire should cause increased catchment runoff (e.g., Wright 1976, Schindler et al. 1980, Hornbeck et al. 1993). However, several studies have noted no change or reduced runoff from burnt areas with deep mineral soils due primarily to reduced snow accumulation and soil moisture (e.g. Rouse 1976, Megahan 1983). Our isotope-based analysis likewise did not detect changes in mean catchment water yield following an extensive fire in boreal peatland catchments. However, the isotope-based analysis did detect increased water yield from burnt catchments that were small relative to the lakes within them. The strong hydrologic connection between such lakes and their drainage areas may account for the increased influence of fire on water yield. In the present study of lakes in the Caribou Mountains, increased water yields were detectable by isotopic methods when DBA/V was below 10. In a subsequent survey of 100 northern Alberta headwater lakes, 80% had DBA/V less than 10. Of those lakes, 100% (n = 18) and 62% (n = 27) of subsets with clear cut logging and forest fire in their catchments, respectively, had drainage ratios less 10 (Prepas et al. in press). A majority of lakes subject to disturbance fall within the group that is most sensitive to increases in catchment water yield following disturbance, and these changes were detectable by the isotope balance approach in the Caribou Mountains. These lakes presumably will also have the greatest potential for changes in water chemistry following disturbance and management options will require some knowledge of contributing and effective areas.

Contributing and effective areas potentially provide additional insight into hydrologic connectivity and export rates, respectively. If export was the predominant control on water chemistry then we would expect the hybrid drainage ratios calculated from these two isotope-based hydrologic indices to be stronger correlates for variance in lake chemistry than drainage ratios calculated from topographic drainage area. Each of the three drainage ratios were highly correlated in the Caribou Mountains dataset. Therefore each offered only minor differences in explaining among lake variation in water chemistry. The isotope method may provide additional insight on export processes for parameters such as DN that are more sensitive to hydrologic exports from the catchment after disturbance.

Our step-wise regression analysis revealed some specific hydrologic and biogeochemical consequences of fire in peatland-rich terrain. Total and dissolved nitrogen were apparently unrelated to catchment features and hydrologic indices for lakes in unburnt catchments, and nitrogen concentrations may be largely determined by internal cycling and the fixation-denitrification balance similar to lakes elsewhere in Alberta (Prepas et al. 2001). Fire apparently altered total nitrogen load causing retention of total nitrogen in burned peatlands but resulted in elevated fluxes of dissolved nitrogen from contributing areas with low slope. In the Caribou Mountains database, rich fens were a larger proportion of peatland area in low slope catchments (McEachern et al. 2000). Increased dissolved nitrogen concentrations in the burnt-catchment lakes may have been derived from enhanced water yield from mineral sources which would typically be discharged through rich fens. Fire in wetlands has been known to reduce their ability to retain dissolved nitrogen causing elevated nitrate export (Bayley et al. 1993). Dissolved organic carbon was found to be primarily related to the fraction of peatland cover, which is a strong indication that this is a primary source. A negative correlation between dissolved nitrogen and basin slope further suggested that loading via fens from burnt catchments may be an important control on water quality following catchment disturbance.

A previous study showed that total and dissolved phosphorus concentrations underwent the most substantial increases in concentration following fire when compared to nitrogen or dissolved organic carbon (McEachern et al. 2000). Phosphorus concentrations were therefore expected to show greatest correlation with the hybrid drainage ratios, although this was not observed. Presumably, substantial elevatation and variable flux rates of phosphorus following fire contributed to a poor correlation between contributing or effective drainage ratio and phosphorus. Total phosphorus concentration in burnt lakes was consistently underestimated by the effective drainage ratio (residual mean = 0.3 for Log_{10} values). Comparing observed concentrations with those predicted from the EA-phosphorus regression indicated that total flux of TP and DP may have been 2-fold higher on average from burnt compared to unburnt effective areas of similar size. This has significant implications for cycling of phosphorus in lakes, which is an important component controlling pelagic concentration. Based on other observations, it is apparent that potential for light and nitrogen limitation of phytoplankton following fire may have disrupted lake phytoplankton growth and nutrient use altering internal cycling in some lakes (McEachern et al. 2002). It was also found that as much as 76% of variation in phosphorus concentrations could be explained by an index including the magnitude of catchment disturbance and time since disturbance (McEachern et al. 2000). Poor correlation between lake phosphorus concentrations and the hydrologic indices for burnt catchments may indicate a majority of phosphorus loading occurred over a short period during or immediately after the fire. Increases in stream water phosphorus concentrations have been found to be pronounced only during fire events with returns to pre-fire loading rates within weeks (Spencer and Hauer 1991). Few studies have observed lasting changes in phosphorus concentration following forest fires (e.g., McColl and Grigal 1975, Minshall et al. 1997). Results from the Caribou Mountains support the hypothesis that fire effects on phosphorus loading may be relatively short-lived with lasting impacts on lake chemistry due to slow internal recovery rather than a continued export of large phosphorus loads from burnt peat.

In most basins, saturated areas along permanent and intermittent channels are believed to comprise the majority of source areas for solute transport (e.g., Pierson and Taylor 1994, Soranno et al. 1996). Representing the size and location of these effective areas is difficult. Improvement in the simple proxy for effective area proposed in this work may be achieved in future by accounting for the distribution and connectivity of saturated or wetland areas within each catchment, rather than just the area. In the Caribou Mountains, there was no apparent relationship between contributing or effective area and saturated surface area, as represented by wetland fen and bog cover. However, the effective area was often directly related to the proportion of upland. In peatland / lichen forest, upland is synonymous with improved drainage (Prepas et al. 2001). The contributing and effective area calculations did not depend solely on the extent of saturated areas such as fens but also on the amount of discharge flowing from those areas. The positive relationship between effective area and upland implied that some catchments sustained higher discharge through saturated groundwater flow or return flow (Dunne 1978, Hill 1993).

In a regional survey of remote catchments, the natural isotopes of water ^{2}H and ^{18}O are shown to be a feasible, practical method for estimating hydrologic conditions, particularly hydrologic variability. They are shown to provide hydrologic indicies that compliment use of physical watershed indicators such as slope and drainage basin area and chemical tracers. Calibration data from one catchment gauged in this study suggests that the isotope method provided comparable estimates of water yield and retention times using a graphical LEL-MWL intercept approach to estimate the isotopic composition of inflow. Contributing areas, calculated from the isotope-derived hydrologic parameters covered similar areas to those determined for other basins with low topographic relief (Prepas et al. 2001, Dunne 1978). Effective area, which used additional chemical parameters to characterize variability in source areas and solute fluxes, was found to be the most effective parameter for explaining among lake variance in nutrient concentrations in the Caribou Mountains if data from burnt and unburnt catchments were combined. Further work addressing the assumptions required in an effective area model may lead to better predictive ability in determining land-water linkages in peatland dominated systems.

Conclusions and Implications

Uncertainty in some aspects of this analysis was exacerbated by short lake water retention times, and difficulty with the verification of model parameters due to remoteness of the study region. However, variability in hydrology predicted by the isotope mass balance model was found to be fairly insensitive to errors in climate parameters (see also Gibson et al. 2002). Our analysis shows that the method can be sensitive to definition of the isotopic composition of inflow or precipitation by several standard approaches. Despite sensitivity in absolute estimates of water yield and residence times, the approach provided a precise set of indices that accurately depicted hydrologic variability and was therefore useful for comparative analysis. While additional hydroclimatic data would have been useful for validation of the methods accuracy, uncertainty in the current approach had no impact on the efficacy as a tool for explaining linkages between patterns in water chemistry and hydrology as error in estimating δ_P results in consistent changes in isotope indices for all lakes.

The elevated nutrient concentrations in Caribou Mountain lake water following fire are unprecedented in current literature. The magnitude of the fire and release of nutrients from burnt peatlands were believed to have caused the observed impacts. Results from this study indicate that increased water yield following forest fire may only be significant for lakes when drainage ratios are small. The results have some additional implications:

1) Impacts from the fire on nitrogen and dissolved organic carbon concentrations were derived from catchment flux through disturbed soil. However, changes in phosphorus concentrations may have been a result of aerial deposition during the fire (e.g. Spencer and Hauer 1991) .
2) Hydrologic connection between lakes and their catchments was generally poor in this peatland environment. Low topographic relief and permafrost likely contributed to poor drainage.
3) Despite poor hydrologic connectivity, impacts from fire were partially due to increased water yield in small catchments.
4) Lakes in small catchments were more likely to be impacted by disturbance than were larger lakes with extensive catchments.

We additionally tested the hypothesis that lake water chemistry should be related to an effective area the size of which can be calculated with a conservative tracer. Due to gross assumptions the effective area calculations did not improve explained variance in water chemistry unless data from burnt and unburnt catchments were combined. Methods for calculating effective areas deserve improvement particularly for regions where surface water quality is linked to characteristics within a limited proportion of the topographically defined catchment.

Table 3-1: Physical and modeled parameters for Caribou Mountain study lakes.

ID	DBA (ha)	Ao (ha)	τ_1 (yr)	τ_2 (yr)	$Yield_1$ (mm)	$Yield_2$ (mm)	CA_1 (ha)	CA_2 (ha)	EA_1 (ha)	EA_2 (ha)
				Lakes in Unburnt Catchments (Reference)						
C1	663.2	52.3	0.69	0.46	88	145	127	210	11.2	18.5
C2	614.8	93.4	1.72	1.22	125	200	167	268	4.9	7.8
C5	1068	76.5	0.64	0.45	54	86	125	200	4.2	6.7
C6	260.4	91.1	0.39	0.30	123	204	70	116	3.1	5.1
C7	569.0	90.0	0.51	0.36	137	220	170	274	7.3	11.8
C8	8208	60.8	0.61	0.40	9	15	161	266	6.4	10.6
C9	146.3	86.1	3.65	2.91	67	98	21	31	1.2	1.8
C11	204.8	134.4	5.85	4.64	47	133	38	59	4.4	12.4
C30	3218	536.1	6.83	4.40	58	367	1440	2560	84.0	149.3
C32	3018	953.4	2.35	1.69	248	397	1624	2602	63.0	100.9
C34	5571	166.5	1.81	1.13	44	82	532	995	50.0	93.4
C35W	825.0	175.0	3.56	2.44	228	375	390	642	37.4	61.6
C45	4627	1173	5.02	3.38	282	488	2709	4682	153.4	265.1
C47	2732	59.0	0.60	0.37	32	60	182	339	121.5	226.0
				Lakes in Burnt Catchments						
C17	700.9	164.8	1.09	0.74	233	380	340	554	31.8	51.9
C23	1053	65.8	1.59	0.83	134	282	302	634	15.2	31.9
C24	1761	159.8	2.30	1.15	210	478	790	1798	48.1	109.6
C25	820.1	151.9	1.60	1.00	247	453	434	795	24.9	45.6
C26	1520	36.6	0.72	0.40	46	90	150	296	13.2	26.0
C27	581.1	28.1	0.50	0.32	62	107	78	135	7.6	13.1
C41	32.6	2.6	0.56	0.37	85	147	6	10	0.4	0.8
C42	34.3	7.4	0.64	0.47	139	231	10	16	0.8	1.4
C43	691.9	8.3	0.31	0.19	18	32	26	46	3.4	6.1
C46	359.4	4.4	0.20	0.10	30	63	22	47	6.8	14.3

Symbols are: drainage basin area (DBA), lake surface area (Ao), lake water residence time (τ), catchment water yield (yield), contributing area (CA) and effective area (EA). Subscripts 1 and 2 are values calculated from theoretical δ_P and modeled δ_P, respectively.

Table 3-2: Univariate regressions of catchment variables (X) and hydrologic variables (Y) for the study lakes. Significance is indicated as <<0.01 when values were < 0.0001.

X	Y	n	α	β	r^2	SE (β)	MSE	F	P
Log_{10} Wo/V	$Log_{10}\ \tau$	24	0.51	-0.51	0.78	0.058	0.042	77.6	<<0.01
Log_{10}DBA/V	$Log_{10}R$	24	2.27	-0.38	0.56	0.0572	0.071	27.6	<<0.01
Burnt		10	2.54	-0.55	0.96	0.039	0.007	192.7	<<0.01
Ref.		14	2.12	-0.35	0.48	0.107	0.091	11.0	0.006
Arc%Upland	Arc%EA	24	0.06	0.129	0.51	0.027	0.001	22.8	<<0.01

Regression parameters are, the number of lakes in the regression (n), intercept (α), slope (β), regression coefficient (r^2), standard error (SE), mean square error (MSE), F-value (F), and probability (P). Catchment area (Wo, m^2), lake volume (V, m^3), residence time (τ, yr), drainage basin area (DBA, m^2), water yield (R, mm)

Table 3-3: Stepwise multivariate linear regression of lake water chemistry, hydrologic indices and catchment characteristics. Values in parentheses are partial correlations and the F-to-enter value at step zero. Otherwise values are combined adjusted r^2 values at each step in the model. One and two astriks are significant at $P < 0.05$ and <<0.01, respectively. Percent values were transformed by negative arcsine square root.

	DBA/V (Log10)	CA/V (Log10)	EA/V (Log10)	Slope (% DBA)	Peatland (% DBA)
TP					
Unburnt	(0.48/2.7)	0.29*	(0.42/1.9)	(0.02/0.003)	(0.27/0.72)
Burnt	(0.26/0.56)	(0.14/0.16)	(0.12/0.11)	(0.31/0.84)	(-0.23/0.45)
DP					
Unburnt	0.31*	(0.61/5.4)	(0.5/2.1)	(-0.27/0.72)	(0.28/0.72)
Burnt	(0.15/0.19)	(0.03/0.008)	(0.04/0.01)	(0.07/0.04)	(-0.23/0.46)
TN					
Unburnt	(0.3/0.89)	(0.29/0.81)	(0.40/1.7)	(0.26/0.69)	(-0.17/0.27)
Burnt	(0.12/0.11)	(0.22/0.42)	(0.28/0.71)	(-0.37/1.2)	0.71**
DN					
Unburnt	(0.44/2.1)	(0.2/0.35)	(0.49/2.8)	(0.16/0.25)	(-0.37/1.4)
Burnt	(0.74/9.7)	0.55**	(-0.53/3.2)	0.89**	(-0.36/1.2)
DOC					
Unburnt	0.69**	(0.69/6.9)	(0.75/12)	(-0.41/1.8)	(-0.02/0.004)
Burnt	0.77**	(0.86/22)	(0.8/14)	0.85**	0.91**

Fig. 3-1: Location of Caribou Mountain study lakes in Alberta, Canada. Reference sites (gray circles) contain lakes with no fire in their catchments since approximately 1910, previously burnt sites (triangles) contain lakes with catchments burnt between 1961 and 1985, burnt sites (black circles) contain lakes in catchments burnt in 1995. An automated climate station (CS) operated for two years.

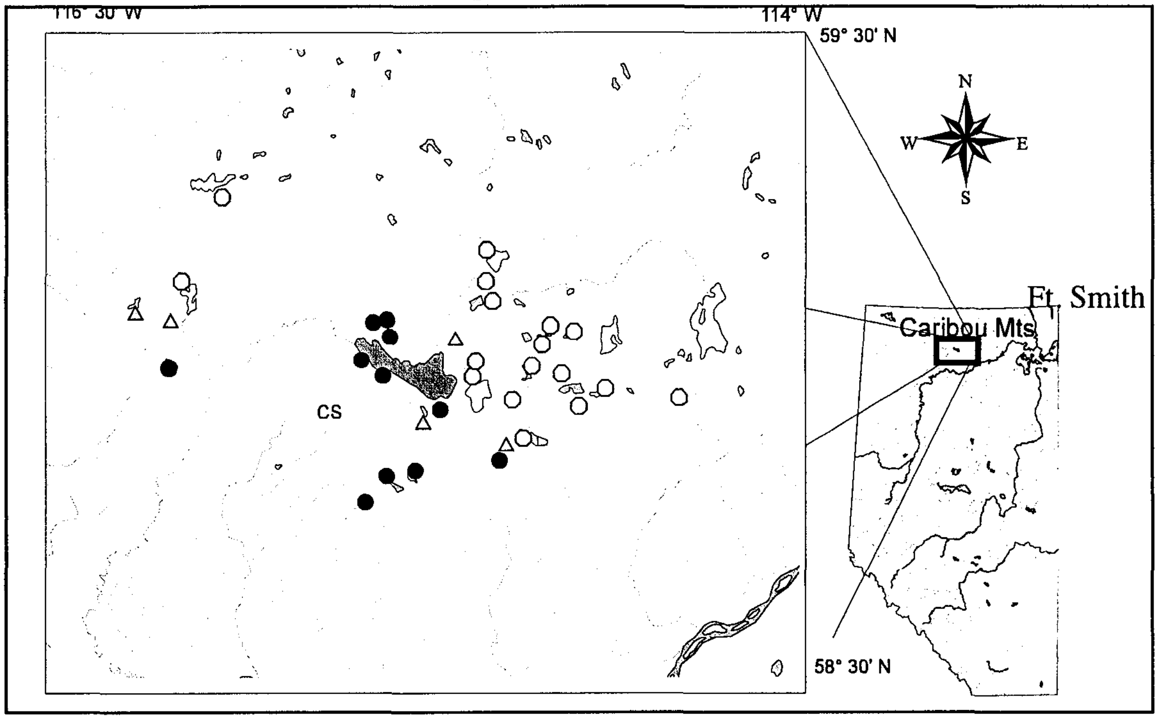


Fig. 3-2: Deuterium and oxygen-18 composition of source waters and lake waters in the Caribou Mountains. Log-term (dp) and volume weighted observed (dp-o) and modeled (dp-m) isotope composition of percipitation are shown. The global mean meteoric evaporation line (MWL, Craig 1961) and the local evaporation line (LEL) are plotted. Also shown are isotope compositions of stream samples and the composition of groundwater samples. The shaded circle represents the mean composition of groundwater.

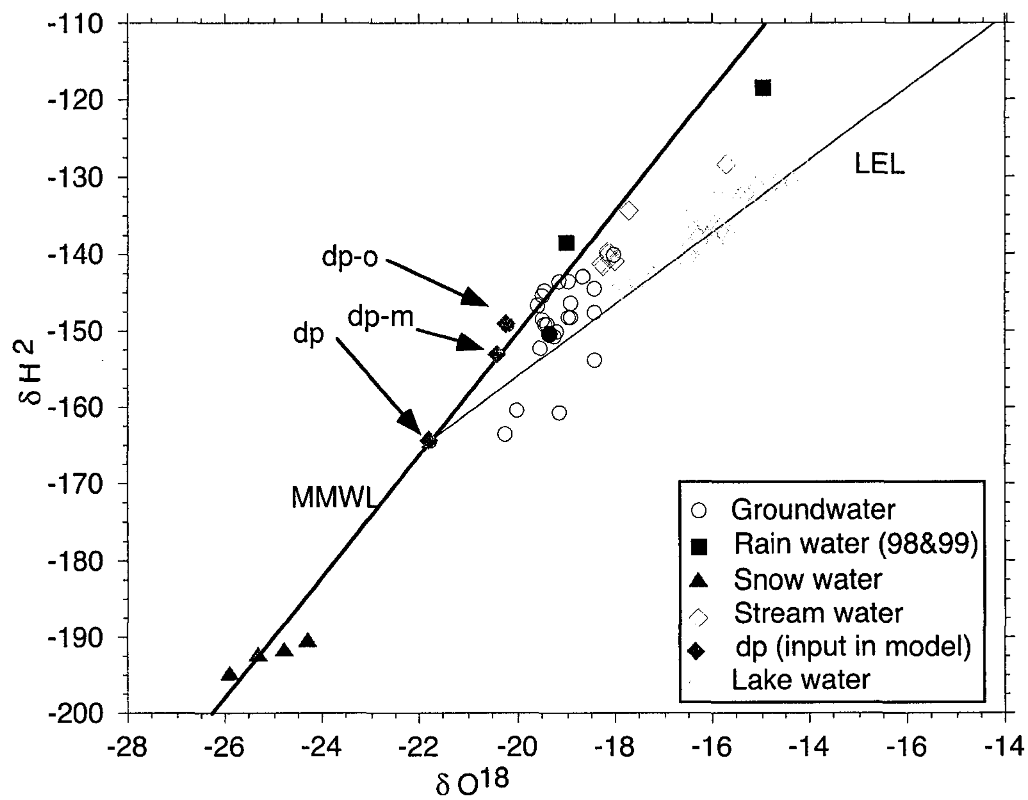
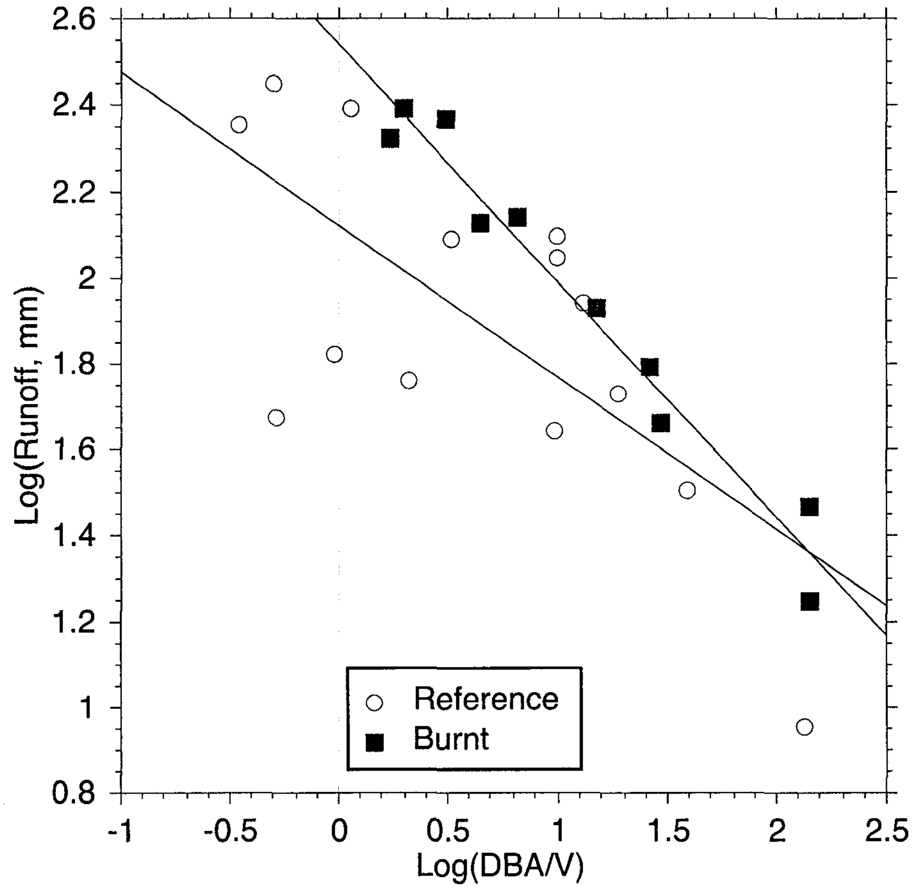
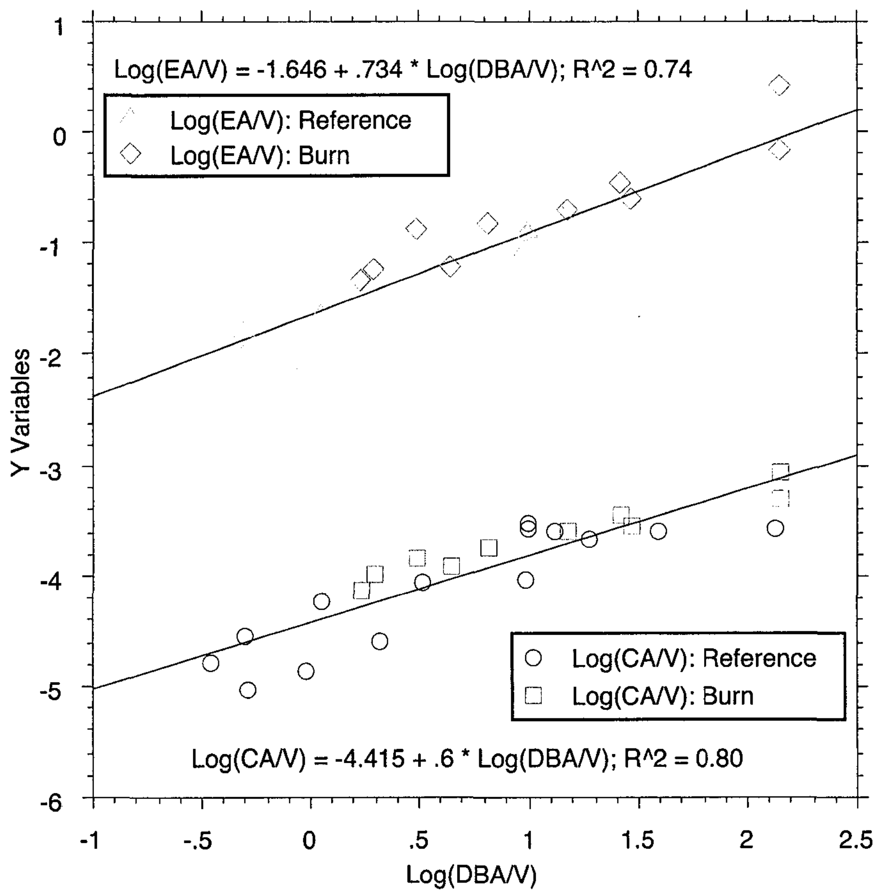


Fig. 3-3: Runoff as a function of drainage basin area : lake volume ratio (DBA/V) for burnt and reference catchments. Regression parameters are in table 2. The slopes of the two regressions are different (P = 0.04)



Log(RO) = 2.535 - 0.546 * Log(DBA/V); R^2 = 0.96 (Burn)
Log(RO) = 2.121 - 0.353 * Log(DBA/V); R^2 = 0.48 (Reference)

Fig. 3-4: Comparison of Effective Area (EA/V) and Contributing Area (CA/V) normalized by lake volume (V) for lakes in burnt and reference catchments. Incorporating effective area in models did not improve explained variance.



References

Bayley, S.E., Schindler D.W., Beaty K.G., Parker B.R. and Stainton M.P. 1992. Effects of multiple fires on nutrient yields from streams draining boreal forest and fen catchments nitrogen and phosphorus. *Can. J. Fish. Aquat. Sci.* 49: 584-596.

Beven, K.J. and Kirkby, M.J. (1979). A physically based, variable contributing area model of basin hydrology. *Hydrol. Sci. Bull.* 24: 43-69.

Birks, S.J., Gibson, J.J., Gourcy, L., Aggarwal, P.K., Edwards, T.W.D., Maps and Animations Offer New Opportunities for Studying the Global Water Cycle. *EOS* 83, (37), 10 September 2002 + electronic supplement (http://www.agu.org/eos_elec/eeshome.html).

Buttle, J. M., Creed, I. F., and Pomeroy, J. W. (2000). Advances in Canadian forest hydrology, 1995-1998. *Hydrol. Process.*, 14: 1551-1578.

Buttle, J. M., and Fraser, K.E. (1992). Hydrochemical fluxes in a high arctic wetland basin during spring snowmelt. *Arct. Antarct. Alp. Res.*, 24(2): 153-164.

Buttle, J. M., and Metcalfe, R. A. (2000). Boreal forest disturbance and streamflow response, northeastern Ontario. *Can. J. Fish. Aquat. Sci.*, 57(Supp 2)(5-18).

Carignan, R., and Steedman, R. J. (2000). Impacts of major watershed perturbations on aquatic ecosystems. *Can. J. Fish. Aquat. Sci.*, 57(Supp 2): 1-4.

Craig, H., (1961). Isotopic variations in meteoric waters, *Science*, 133: 1833-1834.

Crumpton, W, Isehart, G.M., and Mithell, P.L. 1992. Nitrate and organic N analyses with second-derivative spectroscopy. Limnol. Oceanogr. 37: 907-913.

D'Arcy, P., and Carignan, R. (1997). Influence of catchment topography on water chemistry in southeastern Quebec Shield lakes. *Can. J. Fish. Aquat. Sci.*, 54: 2215-2227.

Deevey, E. S. (1940). Limnological Studies in Connecticut: V. A Contirbution to Regional Limnology. *A.C.R.L.*, 238(10): 717-741.

Dunne, T. (1978). "Field studies of hillslope flow processes." In: *Hillslope hydrology*, M. J. Kirkby, (ed.), John Wiley & Sons, Toronto, 227-293.

Evans, J. E., Prepas, E. E., Devito, K. J., and Kotak, B. G. (2000). Phosphorus dynamics in shallow subsurface waters in an uncut and cut subcatchment of a lake on the Boreal Plain. *Can. J. Fish. Aquat. Sci.*, 57(Supp. 2): 60-72.

Feller, M. C., and Kimmins, J. P. (1984). Effects of clearcutting and slash burning on streamwater chemistry ond watershed nutrient budgets in southwestern British Columbia. *Water Resour. Res.*, 20(1): 29-40.

Gat, J.R., (1995). Stable Isotopes of Fresh and Saline Lakes. In: *Physics and Chemistry of Lakes*, A. Lerman, D. Imboden, J.Gat (Eds.) Springer-Verlag, Berlin, pp. 139-165.

Gat, J. R., and Gonfiantini, R. (1981). "Stable isotope hydrology : deuterium and oxygen-18 in the water cycle." *IAEA technical reports series ; no. 210*, Vienna.

Gibson, J.J. (2002). Short-term evaporation and water budget comparisons in shallow arctic lakes using non-steady isotope mass balance *J. Hydrol.* 264: 247-266.

Gibson, J.J., and Edwards, T.W.D. (2002). Regional surface water balance and evaporation-transpiration partitioning from a stable isotope survey of lakes in northern Canada. *Global Biogeochem. Cycles* 16(2): 1010-1029.

Gibson, J.J., Prepas, E.E., and McEachern, P. (2002). Quantitative comparison of lake throughflow, residency, and catchment runoff using stable isotopes: modeling and results from a survey of Boreal lakes. *J. Hydrol.* 262: 128-144.

Gibson, J.J., R. Reid. and C. Spence, (1998). A six-year isotopic record of lake evaporation in the Canadian Subarctic. *Hydrol. Processes*, 12: 1779-1992.

Gibson, J.J., Edwards, T.W.D., Bursey, G.G. and Prowse, T.D., 1993. Estimating evaporation using stable isotopes: quantitative results and sensitivity analysis for two catchments in northern Canada. *Nordic Hydrol.*, 24: 79-94.

Gibson, J. J., Edwards, T. W. D., and Prowse, T. D. (1996). Development and validation of an isotopic method for estimating lake evaporation. *Hydrol. Processes*, 10: 1369-1382.

Halsey, A. L., Vit, D. H., and Trew, D. (1997). Influence of Peatlands on the acidity of lakes in northeastern Alberta, Canada. *Water, Air, and Soil Polution*, 96: 17-38.

Hill, R. A. (1993). Base cation chemistry of storm runoff in a Forested Headwater Wetland. *Water Resour. Res.*, 29(8): 2663-2673.

Hornbeck, J. W., Adams, M. B., Corbett, E. S., Verry, E. S., and Lynch, J. A. (1993). Long-term impacts of forest treatments on water yield: a summary for northeastern USA. *J. Hydrology*, 150: 323-344.

Ivanov, K. E. (1981). *Water Movement in Mirelands*, Academic Press, London England.

Li, Y. and Vitt, D.H. 1997. Patterns of retention and utilization of aerially deposited nitrogen in Boreal Peatlands. *Ecoscience* 4: 106-116.

McColl, J.G. Grigal, D.F. 1975. Forest fire: Effects on phosphorus movement to lakes. *Science* 188: 1109-1111.

McEachern, P., Prepas, E. E., Gibson, J. J., and Dinsmore, P. (2000). Forest fire induced impacts on phosphorus, nitrogen and chlorophyll a concentrations in boreal subarctic lakes of northern Alberta. *Can. J. Fish. Aquat. Sci.*, 57(SUPP/2): 73-81.

Megahan, W. F. (1983). Hydrologic Effects of Clearcutting and Wildfire on Steep Granitic Slopes in Idaho. *Water Resour. Res.*, 19(3): 811-819.

Minshall, G.W., Robinson, C.T., and Lawrence, D.E. 1997. Postfire responses of lotic ecosystems in Yellowstone National Park, U.S.A. *Can. J. Fish. Aquat. Sci.* 54: 2509-2525.

Murthy, J. S. R., Kumar, U. S., Navada, S. V., and Rao, S. M. (2002). Field isotope investigations and numerical simulation of lake hydrodynamics. *Hydrol. Process.*, 16: 1703-1723.

Osgood, R. A. (1988). Lake mixis and internal phosphorus dynamics. *Arch. Hydrobiol.*, 113(4): 629-638.

Pierson, D., and Taylor, C. (1994). The role of surface and sub-surface runoff processes in controlling cation export from a wetland watershed. *Aquat. Sci.*, 56(1): 80-95.

Pietroniro, A., Prowse, T., Hamlin, L., Kouwen, N., and Soulis, E. (1996). Application of a grouped response unit hydrological model to a northern wetland region. *Hydrol. Process.*, 14: 1245-1261.

Prepas, E. E., Planas, D., Gibson, J. J., Vitt, D. H., Prowse, T. D., Dinsmore, W. P., Halsey, L. A., McEachern, P. M., Paquet, S., Scrimgeour, G. J., Tonn, W. M., Paszkowski, C. A., and Wolfstein, K. (2001). Landscape variables influencing nutrients and phytoplankton communities in Boreal Plain lakes of northern Alberta: A comparison of wetland- and upland- dominated catchments. *Can. J. Fish. Aquat. SCi.*, 58: 1286-1299.

Prepas, E. E. & F. H. Rigler (1982). Improvements in quantifying the phosphorus concentration in lake water. *Can. J. Fish. Aquat. Sci.* 39: 822-829.

Rigler, F. H. (1982). Recognition of the possible: an advantage of empiricism in ecology. *Can. J. Fish. Aquat. Sci.*, 39: 1323-1331.

Riley, E.T. and Prepas, E.E. (1984) Role of internal phosphorus loading in two shallow productive lakes in Alberta, Canada. *Can. J. Fish. Aquat. Sci.* 41: 845-855.

Rochelle, B. P. (1989). Regional relationships between geomorphic-hydrologic parameters and surface water chemistry relative to acidic deposition. *J. Hydrol.*, 112(1-2): 103-120.

Rouse, W. R. (1976). Microclimatic changes accompanying burning in Subarctic lichen woodland. *Arct. Antarct. Alp. Res.*, 8(4): 357-376.

Schindler, D. W., Newbury, R. W., Beaty, K. G., Prokopowich, J., Ruszcynski, T., and Dalton, J. A. (1980). Effects of a windstorm and forest fire on chemical losses from forested watersheds and on the quality of receiving streams. *Can. J. Fish. Aquat. Sci.*, 37: 328-334.

Shaw, R. D., and Prepas, E. E. (1990). Groundwater-Lake interactions: II. Nearshore seepage patterns and the contribution of groundwater to lakes in central Alberta. *J. Hydrol.*, 119: 121-136.

Soranno, P.A., S. L. Hubbler, S. R. Carpenter, & R. C. Lathrop (1996). Phosphorus loads to surface waters: A simple model to account for spatial pattern of land use. *Ecolog. Applic.* 6: 865-878.

Spencer, C. N., and Hauer, F. R. (1991). Phosphorus and nitrogen dynamics in streams during a wildfire. *J. North Am. Benthol. Soc.*, 10(1): 24-30.

Stark, N. M. (1977). Fire and nutrient cycling in a Douglas-fir/larch forest. *Ecology*, 58: 16-30.

Strong, W. L., and Leggat, K. R. (1992). Ecoregions of Alberta. Alberta Forestry, Lands and Wildlife, Land Information Services. Edmonton.

Vitt, D.H., Halsey, L.A, and Zoltai, S.C. 1994. The bog landforms of continental western Canada in relation to climate and permafrost patterns. *Arctic Alpine Res.* 26: 1-13.

Winter, T. C. (1981). Uncertainties in estimating the water balance of lakes. *Water Resour. Bull.*, 17(1): 82-115.

Wright, R. F. (1976). The impact of forest fire on the nutrient influxes to small lakes in northeastern Minnesota. *Ecology*, 57: 649-663.

Chapter 4:
Phytoplankton in Boreal SubArctic lakes following enhanced phosphorus loading from forest fire: impacts on species richness, nitrogen and light limitation.

Introduction

Nutrient limitation of phytoplankton biomass and the relative success of phytoplankton groups or individual species are determined in part by lake water chemistry (e.g., Kilham and Kilham 1984, Tilman et al. 1986). In temperate lakes, phytoplankton biomass is typically limited by lake phosphorus (P) concentrations (Schindler 1975, Prepas and Trew 1983). For example, the total nitrogen (N) to total phosphorus ratio (TN:TP) in most temperate lake water is above the cellular mass ratio of 20:1 for freshwater phytoplankton, indicating that N is in greater supply than P relative to cellular requirements (Hecky and Kilham 1988, Hecky et al. 1993). Phosphorus limitation of phytoplankton biomass may become severe at TN:TP ratios above 20 (Hecky et al. 1993) and a TN:TP ratio below 20 suggests phytoplankton could be nitrogen limited. Declines in lake water TN:TP ratios may reduce the ability for non-nitrogen fixing phytoplankton species to compete for resources (Smith 1983).

When P supply is adequate, phytoplankton biomass can be limited by N (Smith 1983, JÛnasson 1996). Nitrogen limitation seems to be mediated by increased loading of P relative to N from either internal or external sources. Mixing of P-rich hypolimnetic water with surface water during the breakdown of thermal stratification is an example of internal P loading (Riley and Prepas 1984, Jones 1990). External P loading can increase after disturbance of forested catchments (Dillon and Kirchner 1975). Forest fire is a common landscape disturbance throughout the boreal forest, causing increased P concentrations in lakes from Canada's eastern Boreal Shield (Carignan et al. 2000) to the

Accepted: McEachern, P., E. E. Prepas & D. Planas (2002). *Lake and Reserv. Manage.* 18(2): 138-148.

SubArctic region of the western boreal forest (McEachern et al. 2000). The soils of the western boreal forest in Alberta are fertile but generally N deficient. As a result, lakes in the western boreal forest tend towards elevated TP concentrations and low TN:TP ratios from both natural and anthropogenic loading. However, lakes in western boreal forests are believed to be predominately phosphorus limited because nitrogen-fixing phytoplankton make up for nitrogen deficiencies (Prepas and Trew 1983). As a result, cyanobacterial blooms are common in lakes of the western boreal forest, particularly of nitrogen fixing Nostocales (Riley and Prepas 1984). These blooms are associated with high winter oxygen depletion rates and poor water quality (Riley and Prepas 1984). The tendency of western boreal lakes towards eutrophic conditions from both internal and external P loading may make them particularly sensitive to eutrophication following catchment disturbances such as forest fire.

Increased lake water concentrations of DOC, colour and increased light extinction coefficient, have been observed following fire in boreal forest catchments (Carignan et al. 2000, McEachern et al. 2000) but little is known about the response of phytoplankton in western lakes. Light limitation of phytoplankton biomass is mediated by the presence of light absorbing material (e.g., dissolved organic carbon (DOC) and suspended solids including phytoplankton cells). A post-fire increase in lake water colour concentration and light extinction could cause increased light limitation of phytoplankton. Combined, light and N limitation following watershed disturbance could result in reduced phytoplankton diversity (Reynolds 1984). A group of lakes in the SubArctic ecoregion of the western boreal forest offered an excellent opportunity to assess N limitation and changes in phytoplankton diversity following watershed disturbance: a 1995 fire impacted 50 to 95% of some headwater lake catchments while leaving other lakes relatively untouched.

Lakes in burnt catchments from this SubArctic ecoregion contained on average 3-fold higher total phosphorus, 5-fold higher soluble reactive phosphorus, 1.2-fold higher total nitrogen and 2.3-fold higher colour concentrations than lakes in unburnt catchments

(Table 4-1). However, pelagic chlorophyll concentrations were not elevated in lakes from burnt compared to unburnt catchments possibly because they had on average half the TN:TP ratio and double the light extinction coefficient of lakes in unburnt catchments. The lower TN:TP ratios and light penetration in lakes from burnt catchments may have resulted in nitrogen or light limitation of phytoplankton biomass (McEachern et al. 2000). An investigation of factors limiting phytoplankton biomass in the study lakes was required to better explain the lack of increased phytoplankton chlorophyll despite exceptional increases in lakewater phosphorus concentration following forest fire.

We described phytoplankton communities in Boreal SubArctic lakes from burnt and unburnt catchments and tested for N, P and light limitation using experimental and field survey data. Our objective was to determine if lake waters with low TN:TP ratios were N limited and if so, what the impact of these low TN:TP ratios would be on phytoplankton communities. We hypothesized that: relatively lower TN:TP ratios and higher colour in water of lakes from burnt catchments would cause N and light limitation of phytoplankton. Further, phytoplankton communities of lakes in burnt catchments would be less rich and would be dominated by species adapted to low N and light conditions. Nitrogen, P and light limitation were tested by modifying nutrient and light regimes within microcosms placed *in situ*. Phytoplankton assemblages in lakes from 10 unburnt and 10 burnt catchments were compared to determine if patterns in phytoplankton dominance were associated with TN:TP ratios, nutrient and colour concentrations that differred between the two groups of lakes.

Methods

The Caribou Mountains in northern Alberta, Canada (59° N 115° W) are erosional remnants forming a large, relatively flat plateau of approximately 12 800 km^2. Peatlands cover 56% of the Caribou Mountains, which are underlain by poorly drained cryosolic and brunisolic soils (Strong and Leggat 1992). Open forest of black spruce (*Picea* *marianna*) and an understory of mixed *Sphagnum* spp., feathermosses, brown mosses and lichens dominate vegetation, including a majority of the peatlands. Hydrology is dominated by direct precipitation and drainage from limited source areas of interconnected peatlands. During 1995, a forest fire incinerated 30% of the Plateau's trees and understory herbaceous, sedge, and lichen cover. Few unburnt areas remained within the fire boundary but damage within the peat soil profile varied. Impact from burning was mostly limited to the top 20 cm of peat.

To test nutrient and light limitation of phytoplankton we selected three lakes spanning a range of nutrient and light conditions observed in lakes from the Caribou Mountains (McEachern et al. 2000). The three lakes, all located within a 20-km radius, were chosen to represent nutrient conditions found in lakewater from burnt and unburnt catchments (Fig. 4-1). Access to lakes in the Caribou Mountains was restricted to float plane and played an additional role in selecting the three lakes. The three lakes were named: Lowratio, Midratio, and Highratio for this study. Lowratio Lake water (C24 from McEachern et al. 2000) had the lowest TN:TP ratio (mean summer ratio = 8) and the highest TP concentration (70 $\mu g \bullet L^{-1}$). It was a headwater lake within a catchment that was 95% burnt in the 1995 forest fire. Midratio Lake water had an intermediate TN:TP ratio (20) and TP concentration (43 $\mu g \bullet L^{-1}$). It was a headwater lake in an undisturbed catchment. Finally, Highratio Lake water had the highest TN:TP ratio (38) and lowest average TP concentration (15 $\mu g \bullet L^{-1}$). Highratio Lake was chosen because its proximity to our 1997 base camp allowed us to monitor the bioassays, and because its water chemistry represented the low end of nutrient concentrations for Caribou Mountain lakes. Highratio Lake was non-headwater and drained a large catchment partially burnt by the 1995 fire (c.a. 20%). However, Highratio Lake retained mesotrophic conditions (Table 4-1) characteristic of lakes in unburnt catchments with the lowest trophic status because its volume (10^9 m^3) was 150 times that of surveyed headwater lakes. Both Highratio and Lowratio Lakes were sampled throughout 1997. Midratio Lake was added in 1998 because it represented elevated nitrogen and phosphorus concentrations found in some lakes from unburnt catchments.

For the lakes originally described in McEachern et al. (2000) and the additional lakes Highratio and Midratio, integrated water samples representative of the euphotic zone were collected with a PVC tube. Triplicate 25-mL aliquots (unfiltered) were pipetted into 30-mL glass tubes used directly in digestion processes for TN and TP analysis. Filtrate (Millipore HA, 0.45 μm) was used for nitrate-nitrite (NO_{2+3}^{-}-N) analysis. Nitrate-nitrite samples were frozen in 125-mL polyethylene bottles for analysis by cadmium reduction with automated colorimetry (Technicon methods 100-700 W/B and 155-71W). Total N samples were preserved with 10 μL of 40% H2SO4 for storage (1-2 wk), and neutralized with equivalent NaOH prior to digestion in the laboratory. Total N concentrations were determined by second derivative spectrophotometric analysis of persulfate oxidized samples (Crumpton et al. 1992). Total P was analyzed by spectrophotometry from persulfate-oxidized samples by molybdate blue absorption (Prepas and Rigler 1982). Dissolved organic carbon was analysed from GF/F filtrate, refrigerated in 60-mL polyethylene bottles until analysis by high temperature catalysis (Shimadzu TOC-5000). Colour was determined spectroscopically from GF/F filtrate (APHA 1995). A Hanna 1230 pH probe was used to measure pH from a subsample of euphotic zone lake water immediately after collection. At the deepest site in each lake, water temperature and light penetration (Li-Cor LI-185, LI-192SA sensors in air and submerged positions) profiles were recorded at 0.5-m intervals. Light extinction coefficient was calculated for for 400-700 nm wavelengths (PAR, $\mu E \bullet m^{-2} \bullet s^{-1}$) and averaged over the euphotic zone without including surface extinction determined when the submerged sensor was at 0.01 m depth. Euphotic zone depth was estimated from the light profile for each sampling period at the depth of 1% of surface PAR. Secchi depth was averaged from two measurements with a 20-cm Secchi disk.

Nutrient stimulation experiments (NSE) were conducted in Lowratio and Highratio lakes concurrently from July 19 to 25, 1997 and in Midratio Lake during the summer of 1998. Each NSE consisted of 10-L polyethylene containers of lake water augmented with N, P, or N+P, plus an untreated control. Triplicate treatment containers were suspended at half the Secchi disk depth for 5 d (Knowlton & Jones 1996). For the NSEs, Midratio and Lowratio lakes received nutrient doses of 1000 $\mu g \bullet L^{-1}$ N and 100 $\mu g \bullet L^{-1}$ P while Highratio Lake received halved doses. In Lowratio Lake, half the Secchi depth was 10% of incident light, subjecting the NSE to 5-fold more light when compared to phytoplankton circulating in the mixed layer which averaged 2% of incident light. Likewise, in Midratio and Highratio lakes, half the Secchi depth was 2-fold and a 5-fold more light compared to their respective epilimnia. Due to accessibility costs, replication through time was not feasible; therefore, results represent a "snapshot" in time.
Lake water collected at NSE initiation and completion, and water from each microcosm, was immediately filtered in triplicate through Gelman AE filters, stored in desiccant and subsequently frozen prior to analysis for chlorophyll *a* (CHL). Total chlorophyll was extracted with 90% ethanol in a water bath, 20 min. at 60 °C, followed by 20 to 24 h of refrigeration (Sartory and Grobbelaar 1986). CHL concentrations were determined with fluorometric methods (Welschmeyer 1994) and a calibration curve from commercial CHL standards checked with spectrophotometric methods (Wintermans and DeMots 1965). Samples were allowed to equilibrate with room temperature prior to analysis. CHL concentrations were uncorrected for pheophytin. The presence of phaeophytin was assessed by comparing fluorescence before and after acidification of each sample with HCL. In a majority of samples, phaeophytin was undetectable and comprised less than 10% of CHL in those where phaeophytin was detected.

NSE phytoplankton response was determined by dividing final CHL by initial CHL in the microcosms and then comparing responses among treatments with replicate means (ANOVA, Bonferoni-Dunn comparison). Limitation by either N or P was inferred from elevated CHL in N or P treated containers compared to control containers. Concurrent limitation by both N and P was inferred from elevated CHL in N+P treated containers relative to controls and those treated by either N or P alone. Light limitation was inferred from increased CHL in control containers relative to lake water CHL over the incubation period.

Phytoplankton species were identified from one sample from each of 10 burnt and 10 unburnt catchments, collected between July 19 and 25, 1997. These 20 lakes for plankton analyses were drawn from the original 24 lakes described by McEachern et al. (2000). Four lakes from unburnt catchments in McEachern et al. were randomly selected and removed from phytoplankton analysis to balance samples from lakes in burnt and unburnt catchments. Lakes in burnt and unburnt catchments were indistinguishable from each other based on physical characteristics including drainage ratio, lake area to volume ratio or landcover features in the catchments (McEachern et al. 2000). Although part of the microcosm experiments, Highratio and Midratio lakes were not included in the search for relationships between phytoplankton taxonomy and lake water chemistry because they did not correspond in physical character or sampling time, respectively, with the 20 surveyed lakes. Samples were stored in amber bottles and preserved with Lugol's solution at the time of collection. Phytoplankton were identified, measured and counted with an inverted inferential microscope (Lund et al. 1958). Species identification followed taxanomic keys of Prescott (1962), Findlay and Kling (1979), Bourelly (1972, 1981a,b, 1985). Species richness was determined as total species identified in a census of a settled sub sample for each lake. Wet weight biomasses were determined by multiplying cell densities by average species dimensions, measured from 10 cells of each species, assuming a density of $1 g/cm^3$.

For statistical analysis, lake water chemistry and phytoplankton data were tested for deviation from a normal distribution based on Kolmogorov-Smirnov procedures. Data were either normally distributed or were $\log_{10}$-transformed to meet assumptions for normality and were analyzed by *t*-test, Pearson's correlation or univariate least-squares regression. Arcsine transformations were applied to proportion data. Probability values and correlation coefficients are reported in the text for these tests. Discriminant analysis followed methods in Statistica 4.1 (© StatSoft, 1994) with orthogonal functions determined from canonical correlation analysis. A power analyses (two-sample *t*-test design) indicated phytoplankton communities from 148 and 244 lakes, respectively, would have been required to determine differences in relative abundance of Nostocales at powers of 80 and 95%. Except for Bonferroni-Dunn comparisons ($P << 0.01$), differences were considered significant at $P \leq 0.05$. All analyses used Statistica 4.1 and StatView 4.5 (© Abacus 1995) for Macintosh.

Results

Impacts of fire on lake water chemistry and phytoplankton biomass

Dramatic differences in phosphorus concentrations and mean TN:TP ratio were apparent in water from lakes in burnt compared to unburnt catchments (Fig.□1)The lower TN:TP ratio in lakes from burnt catchments was presumably a result of enhanced flux of phosphorus relative to nitrogen from burnt peatlands reported by Bayley et al. (1992). CHL and TP were strongly correlated in water from lakes in unburnt catchments (McEachern et al. 2000). However, no similar relationship was apparent for water from lakes in burnt catchments indicating variables, other than phosphorus, were limiting phytoplankton biomass following forest fire. CHL concentration was predicted for the 10 lakes in burnt catchments from the CHL - TP regression for lakes from unburnt catchments and residuals were determined by subtracting observed CHL concentration. The residuals were negatively related to TN:TP ratio (Fig. 4-2, $r^2 = 0.69$, $P < 0.01$, $n = 10$) and to Secchi depth ($r^2 = 0.44$, $P = 0.04$) indicating that limitation of phytoplankton biomass may have switched from phosphorus to nitrogen or light in lakes following forest fire. The occurrence of phosphorus, nitrogen and light limitation was subsequently investigated in microcosm experiments at TN:TP ratios and colour concentrations representative of lakes in burnt and unburnt catchments.

Nutrient limitation in NSE microcosms

The importance of low nitrogen relative to phosphorus concentrations in lakewaters was tested by observing changes in phytoplankton biomass, as CHL concentration, in response to added nitrogen or phosphorus within microcosms. In Highratio Lake (TN:TP = 38), CHL concentration increased 1.5 fold in P amended microcosms (Fig. 4-3) compared to CHL concentration in control microcosms (initial CHL = 6 $\mu g \bullet L^{-1}$). The response indicated P limitation of phytoplankton biomass. CHL concentration in the Highratio Lake N + P microcosms were elevated compared to the P amended microcosms. A larger response in N + P compared to P microcosms may indicate the possibility for reciprical limitation in lake waters (Morris and Lewis 1988). In the Highratio Lake NSEs, reciprical limitation was more likely an artifact of excess P relative to N concentrations in microcosm water treated with P only. In Midratio Lake (TN:TP = 20), CHL concentration in the combined N + P amended microcosms increased an average 7 fold compared to control CHL concentration (initial CHL concentration = 9 $\mu g \bullet L^{-1}$, Fig. 4-3), indicating strong limitation by both N and P. A 2-fold increase in CHL concentration in microcosms amended with N over both control and P amended microcosms (Fig. 4-3, $P << 0.01$) suggests that algal biomass in Midratio Lake was particularly influenced by low N availability during mid summer when the experiment was initiated. In Lowratio Lake (TN:TP = 8), response was limited to microcosms with N amendments. On average, water from N amended microcosms had 1.5-fold more CHL concentration compared to water in control microcosms (initial CHL concentration = 6 $\mu g \bullet L^{-1}$, $P << 0.01$). Both N and N + P amended microcosms indicated N limitation of phytoplankton biomass in water from Lowratio Lake. As mean lakewater TN:TP ratios declined, the three NSEs demonstrated P, concurrent N and P, and N limitation, respectively.

Light Limitation in microcosms

Light limitation of phytoplankton biomass was inferred when CHL concentration in control microcosms increased relative to CHL concentration in the surrounding lake water. In Highratio Lake, CHL concentration in control microcosms increased 1.5 fold when compared to CHL concentration in the lakewater ($P << 0.01$). Likewise, CHL concentraton in Lowratio Lake control microcosms increased 2.5 fold over CHL concentration in the lakewater. In both Highratio and Lowratio lake NSEs, incubation of the microcosms at half the Secchi depth resulted in five times more light exposure than the mixed layer mean. Increased light exposure in the microcosoms relative to the lake likely stimulated additional phytoplankton growth and resulted in increased CHL concentration in control microcosms. Both Highratio and Lowratio lakes contained coloured waters with DOC concentrations close to mean values for lakes from undisturbed catchments in the region and well below mean values for lakes from burnt catchments (Table 4-1). Evidence of light limitation of phytoplankton in Highratio and Lowratio lakes suggests that light limitation of phytoplankton may be common among the majority of lakes in the region that contain similarly coloured waters.

Midratio Lake was less coloured and had the lowest DOC concentration and deepest Secchi depth of the sampled lakes (Table 4-1). During the incubation period, CHL concentration in Midratio Lake water from the euphotic zone increased 2.9 fold from minimum (9 $\mu g \cdot L^{-1}$) to near maximum (26 $\mu g \cdot L^{-1}$) summer values. Although CHL concentration also increased in Midratio Lake control containers (Fig. 4-3, $P << 0.01$), the increase was less than that observed in the lake itself and was likely not a response to improved light conditions in the microcosms. Total phosphorus concentrations increased 10 $\mu g \cdot L^{-1}$ in Midratio Lake water just before the NSEs were initiated and during their incubation; the increased phytoplankton biomass in the Midratio Lake control microcosms could have been a response to enhance phosphorus availability in lakewater when the NSEs were filled. Light did not seem to limit pelagic phytoplankton biomass in Midratio Lake as CHL concentration in the lakewaters increased more than in the control microcosms.

Phytoplankton Assemblages

From the 10 lakes in unburnt catchments, 125 species of phytoplankton were identified. Average species richness was 36, consistent with richness from other lakes in the western boreal forest (Planas and Prepas unpubl. data). Cyanobacteria represented between 15% and 89% of total phytoplankton biomass (Fig. 4-4) and over 90% of cell densities in lakes from unburnt catchments. Dominance patterns within Cyanobacteria were variable with, for example, the N-fixing Nostocales representing from 8% to 88% of total phytoplankton biomass. Lakes in unburnt catchments contained TN:TP ratios and mesotrophic to eutrophic nutrient concentrations ranging between Highratio and Midratio lakes (Fig. 4-1). Phytoplankton assemblages in the 10 lakes from unburnt catchments were therefore expected to show a relationship with TN:TP indicative of phosphorus and nitrogen limitation. Abundance and biomass at order, sub-order or genus levels were not detectably related to N concentration or TN:TP ratio. Clear taxa patterns with N concentration or TN:TP ratio among lakewaters from unburnt catchments were not apparent. Instead, taxa adapted to low N conditions dominated lakes in unburnt catchments consistent with the N+P limitation of phytoplankton demonstrated in Midratio Lake microcosms.

Lakes in burnt catchments contained similar phytoplankton species (Table 4-2) but lower species richness compared to lakes from unburnt catchments. A total of 71 species were identified from lakes in burnt catchments. Average species richness in lakes from burnt catchments was 23, or 36% lower than richness in their unburnt counterparts

($P \ll 0.01$). Cyanobacteria represented from 17% to 99% and Nostocales from 0% to 99% of total phytoplankton biomass in the burnt-catchment lakes (Fig. 4-4). *Aphanizomenon* dominated mean biomass at 40% of total phytoplankton biomass for burnt-catchment lakes compared to *Anabaena,* which dominated in lakes from unburnt catchments (Table 4-2). No genera comprising more than 5% of total biomass were unique to lakes from burnt catchments. Two genera, *Chroomonas* and *Gymnodinium,* identified in three or more lakes in unburnt catchments, were absent from burnt-catchment lakes. Discriminant analyses did not separate phytoplankton communities from lakes in burnt and unburnt catchments at the order, suborder or genus levels ($P > 0.5$). Clear patterns within *Cyanobacteria,* such as dominance by N-fixing Nostocales, were not apparent in lakes from burnt catchments despite a majority having lakewater TN:TP ratios below 10. Differences in phytoplankton communities between lakes in burnt and unburnt catchments were not detected because both groups of lakes were dominated by taxa adapted to N-deficient conditions. However, less common phytoplankton species were absent from lakes in burnt catchments resulting in substantially lower species richness when compared to lakes in unburnt catchments.

Correlation analyses of the combined lake phytoplankton data with lake water chemistry suggested phytoplankton community composition was related to nitrate concentration, pH and DOC or colour concentration. Data from lakes in burnt and unburnt catchments were combined because fire was not a detectable factor in phytoplankton response to chemical variables (ANCOVA, $P > 0.4$). As a proportion of total cells, Cyanobacterial cell abundance was negatively related to nitrate concentration (Table 4-3). Relative Chrysophyte cell abundance decreased with pH and increased with DOC concentration while the inverse was true for Cyanobacteria. Total N concentration increased with relative Cyanobacteria cell abundance while relative abundance in other orders declined. Nostocales showed similar patterns with Cyanobacteria as a whole and were negatively correlated to nitrate and positively correlated to TN concentration. Increases in

Nostocales with decreased nitrate suggested a response of N-fixing plankton to low inorganic N availability.

Results from two of the three NSEs suggested light was a limiting factor; water colour and DOC concentrations, which were closely associated with reduced light penetration (McEachern et al. 2000), seemed to benefit Chrysophyte and Cryptophyte communities (Table 4-3). In addition to advantages under low light conditions due to mobility of both taxa, Cryptophytes and some species of Chrysophytes are mixotrophic (Planas et al. 2000) and may have benefited from heterotrophy or bactivory under elevated DOC concentration. Relationships between taxa cell densities with TN:TP and light extinction coefficients were not detectable ($-0.3 < r < 0.3$, $P > 0.3$) despite lakewater TN:TP ratios from burnt catchments that were less than half and light extinction coefficients almost double that in lakes from unburnt catchments. Relative biomass of phytoplankton groups followed similar patterns as relative cell abundance with lake water chemistry presented in Table 4-3 but correlations were not as strong. Biomass calculations were prone to greater error in our Cyanobacteria dominated samples because errors were multiplied in counting cells and in determining mean volumes, both of which were difficult for mucilage colonies. Although phytoplankton assemblages were related to chemical variables that were impacted by fire, differences in plankton assemblage between lakes in burnt and unburnt catchments were not apparent. Our ability to detect differences between phytoplankton communities from lakes in burnt and unburnt catchments was obscured by the dominance of Cyanobacteria in both lake groups. The apparent dominance of phytoplankton communities by Cyanobacteria may have been accentuated by selecting samples from mid-summer only. However, dominance by Cyanobacteria was expected, given NSEs which indicated nitrogen limitation may be common among lakes in unburnt catchments and likely predominated in lakes from burnt catchments.

Discussion

Phosphorus limitation of phytoplankton biomass is characteristic of lakes in the western boreal forest (Prepas and Trew 1983). Nitrogen limitation has been suggested for eutrophic water typical of western boreal lakes where N-fixing phytoplankton are common (Zhang and Prepas 1996). We demonstrated P limitation in water from one mesotrophic lake (Highratio Lake) and combined N and P limitation in water from a eutrophic lake (Midratio Lake). Many lakes in the western boreal forest contain TN:TP ratios and nutrient concentrations similar to Midratio and Lowratio lakes (Mitchell and Prepas 1990, Prepas et al. 2001). N limitation of phytoplankton may be common in lakes of the western boreal forest.

Nitrogen limitation alone was demonstrated in a lake where the average TN:TP ratio was 8 (Lowratio Lake), less than half and one-quarter that in Midratio and Highratio lakes, respectively. The TN:TP ratio of water in Lowratio Lake was typical for other lakes in burnt catchments. Reduced TN:TP ratio following fire in Boreal SubArctic lakewaters results from increased export of P relative to N from burnt peatlands (McEachern et al. 2000). In addition to the likely widespread occurrence of N limitation of phytoplankton in western boreal forest lakes, forest fire seems to increase the likelihood of N limitation.

Light limitation has been demonstrated for lakes with low light penetration (Knowlton and Jones 1996). We demonstrated light limitation in lakes with light extinction coefficients of ε 2.5 PAR m^{-1} but did not detect light limitation at an extinction coefficient of 1.5 PAR m^{-1}. Light extinction in Boreal SubArctic lakes was directly correlated with high DOC and colour content of lake water, both of which increased following burning of peatland catchments (McEachern et al. 2000). Although lakewater from burnt catchments contained 3 times the mean phosphorus concentration of lakewater from unburnt catchments, mean chlorophyll concentrations were similar between the two groups (McEachern et al. 2000). Of the surveyed lakes in burnt and unburnt catchments, 88% and 67%, respectively, had lower mean epilimnetic light in-comes than Highratio and Lowratio lakes. Light limitation is likely a common feature of lakes in peatland dominated catchments and increased allochthonous DOC following fire enhanced the severity of light limitation.

Phytoplankton species richness was expected to be lower in lakes within burnt catchments because fewer species can survive as N and light limitation become severe (Reynolds 1984, 1994). Declines in phytoplankton species richness have been observed under conditions of increasing external P load and reduced TN:TP ratios (e.g. Tilman 1978), similar to changes following fire in the Caribou Mountains. The decline in phytoplankton species richness by 36% following catchment burning was more severe than what has been reported in the literature, following either forest fires or logging in Boreal Shield catchments (Planas et al. 2000) or logging in Finnish catchments (Rask et al. 1998).

Phytoplankton communities in lakes from burnt catchments should have contained greater relative cell densities and relative biomass of Nostocales if N limitation was more pervasive. Relative cell density and relative biomass of Nostacales were not consistently higher in our samples from burnt lakes despite TN:TP ratios predominantly below 10. However, Nostocales increased relative to other phytoplankton as nitrate concentrations declined, indicating low available N concentration may be a stronger determinant in promoting N-fixing plankton than low TN:TP ratio. Fire may provide no advantage for N-fixing phytoplankton in burnt catchments where nitrate export increases following forest fire (e.g.□Minshall et al. 1997) particularly in peatland catchments (Bayley et al. 1992). We expect that fire may even reduce potential bloom formation of N-fixing phytoplankton in SubArctic and possibly other lakes with peatland dominated catchments because of enhanced light limitation in lake waters following forest fire. In our study, there seemed to be a shift, within the Nostocales, from dominance by *Anabaena* to *Aphanizomenon* in lake waters from unburnt and burnt catchments, respectively

(Table 4-2). In order to determine differences in phytoplankton communities with a reasonable degree of certainty, samples were required from more lakes (n >>100) than were possible in this study. The dominance of phytoplankton by Nostocales in lakes from undisturbed catchments made it unlikely that increases in N-fixing taxa after fire would be detectable, but shifts in dominant genera seemed to occur.

The limitation assays demonstrated a range in nutrient limitation from N through concurrent N+P to P limitation in association with increasing TN:TP ratio in the SubArtic western boreal forest lakes. Phytoplankton community compositions were ambiguous in describing linkages between nutrient limitation and nutrient ratio. However, both NSEs and lake community composition suggested N-deficient conditions could result in N limitation and that N limitation may be common among the undisturbed but naturally productive lakes of Canada's western Boreal forest. Disturbances such as forest fire can cause elevated P loading relative to N in boreal catchments which seem to increase what may be a natural tendency towards N limitation of phytoplankton biomass. Forest fire in peatland environments could result in long term impacts to western boreal lakes, as elevated phosphorus concentrations relative to lakes in unburnt catchments were measured in lakes where their catchments burnt over the past 3 decades (McEachern et al. 2000). Forest fire also caused decreased light penetration and provided nitrate that may inhibit biomass of N-fixing phytoplankton in boreal lakes. Lakes in northern boreal forests provide additional opportunities to review the implications of enhanced allochthonous loading to lake waters where phytoplankton communities thrive in a balance between P, N and light limitation.

Table 4-1: Summer mean values for lakes containing nutrient stimulation experiments. Highratio and Lowratio were sampled in 1997, Midratio was sampled in 1998. Summary data are included from lakes in unburnt and burnt catchments described in 1997 by McEachern et al. (2000). Standard errors are in parentheses.

	Highratio $n = 4$	Midratio $n = 5$	Lowratio $n = 4$	Unburnt $n = 10$	Burnt $n = 10$
TP (μg• L^{-1})	14 (2)	56 (12)	76 (8)	34 (2)	85 (12)
TN (μg• L^{-1})	510 (33)	1003 (58)	592 (43)	717 (27)	759 (47)
TN:TP	38 (2)	20 (2)	8 (1)	22 (2)	11 (2)
NO3 (μg• L^{-1})	51 (16)	4 (2)	16 (6)	6 (3)	9 (3)
DOC (mg• L^{-1})	16.7 (0.5)	16.6 (3.3)	19.2 (0.6)	18.5 (2)	24.9 (2)
CHL (μg• L^{-1})	5 (2)	22 (5)	9 (2)	11 (2)	13 (4)
Colour (mg• L^{-1} [Pt])	210 (60)	83 (16)	248 (2)	180 (36)	342 (42)
Extinction (PAR m^{-1})	2.47(0.06)	1.56 (0.05)	3.68 (0.09)	3.45 (0.40)	5.44 (0.53)
Secchi Depth (m)	2.1 (0.1)	2.2 (0.1)	1.0 (0.1)	1.1 (0.1)	0.7 (0.1)
Euphotic : Mix depth	0.48	1.00	0.10	0.65	0.48

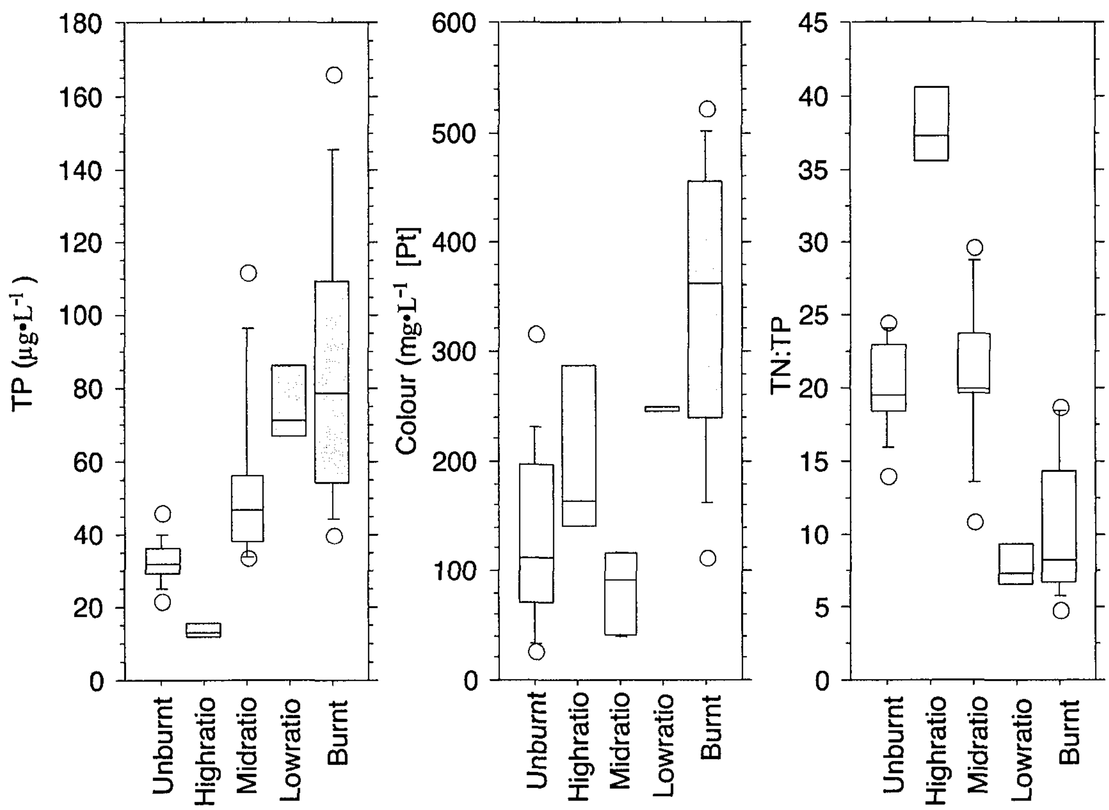
Table 4-2: The 20 most common genera observed in water from lakes in unburnt and burnt catchments. %Biomass is an average of total phytoplankton biomass for the lakes (*n*) in which the genera were identified.

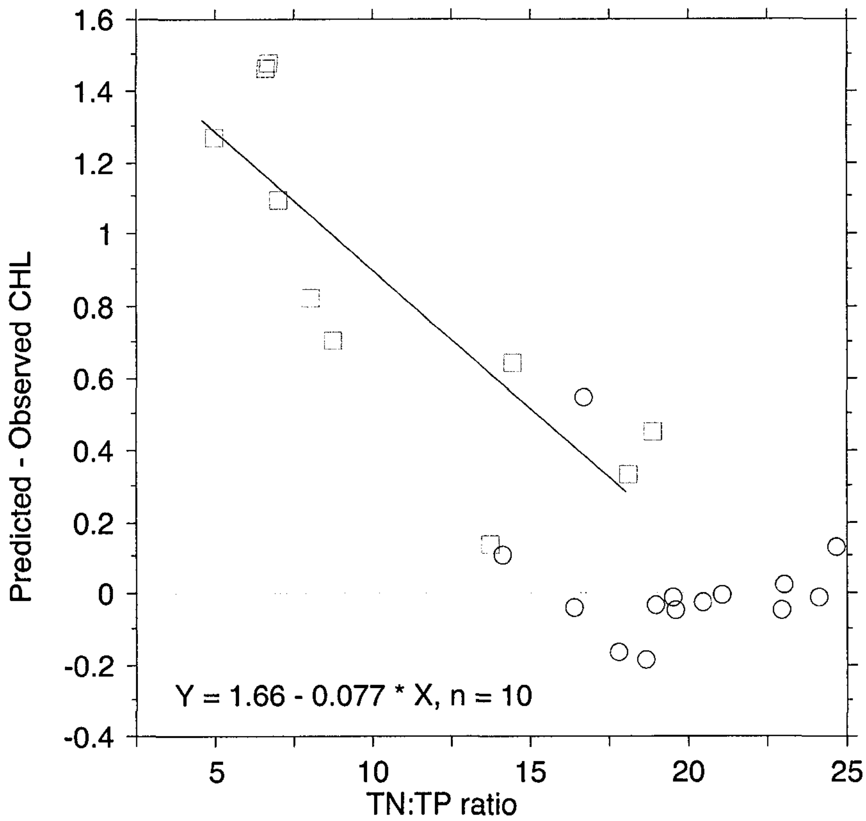
20 most common genera in unburnt catchment lakes	Unburnt		Burnt	
	% Biomass	*n*	% Biomass	*n*
Anabaena	20	10	10	10
Cryptomonas	2	10	4	10
Chromulina	< 1	10	< 1	10
Aphanothece	1	10	2	9
Chlamydomonas	< 1	10	1	6
Coelosphaerium	4	9	3	7
Aphanizomenon	14	8	40	8
Asterionella	2	8	2	6
Erkenia	1	8	2	7
Katablepharis	1	8	1	9
Rhodomonas	2	8	6	8
Ankistrodesmus	1	7	1	8
Ochromonas	1	7	2	7
Uroglena	1	7	1	4
Dinobryon	2	6	< 1	2
Monoraphidium	1	6	0	3
Oocystis	1	5	6	4
Scenedesmus	1	5	0	4
Gymnodinium	1	5	0	0
Melosira	3	5	2	6
Rare, more common in burnt catchment lakes				
Ankyra	0	0	1	4
Cyclotella	< 1	3	1	4
Mallomonas	0	3	1	4

Table 4-3: Pearsons correlation coefficients for relative cell abundance ($cells \cdot L^{-1}$ as a percent of total cell density) of Cyanobacteria (%Cyan.), Cryptophyceae (%Crypto.), Chlorophyceae (%Chloro.), and Chrysophyceae (%Chryso.) with lake water concentration of TN, TP, DOC, NO_{2+3}-N, pH and colour in 10 lakes from unburnt and 10 lakes from burnt catchments. Diatomaceae, Peridineae, and Euglenaceae contribute to total abundance but were not included because correlations with water chemistry were not detected for these groups. Correlations are indicated by one ($P < 0.05$) or two ($P < 0.01$) asterisks.

	%Cyan.	%Crypto.	%Chloro.	%Chryso.
$Log_{10}NO_{2+3}$-N	-0.56 **	0.42	0.42	0.46
Log_{10}TN	0.72 **	-0.51 *	-0.54 *	-0.59 *
Log_{10}TP	0.23	-0.36	0.09	-0.28
Log_{10}Colour	-0.52 *	0.36	0.23	0.52 *
Log_{10}DOC	-0.65 **	0.51 *	0.32	0.56 *
pH	0.64 **	-0.21	-0.48	-0.60 *
%Cyan.		-0.78 **	-0.56 *	-0.87 **
%Crypto.			0.13	0.71 **
%Chloro.				0.16

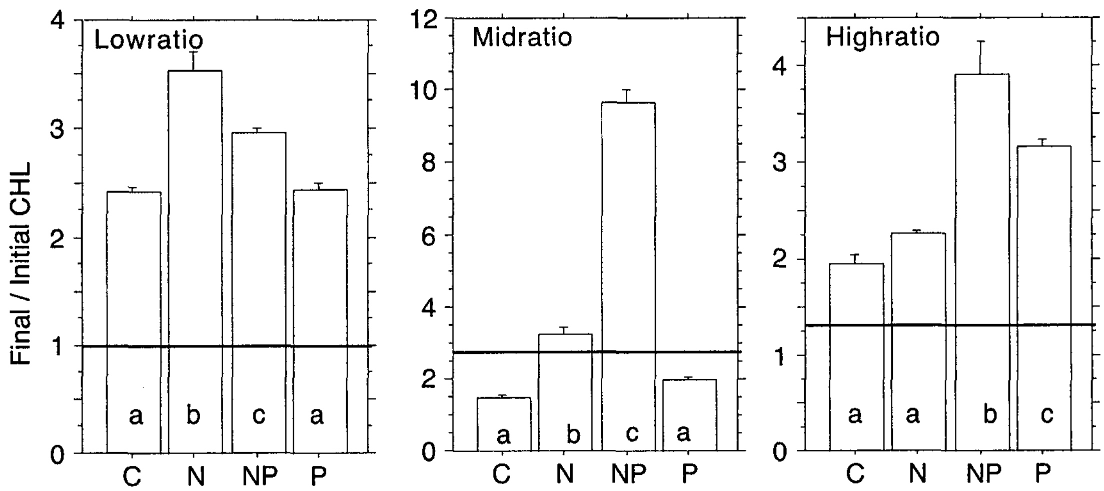
Fig. 4-1: Water chemistry for lakes in the Caribou Mountains summarized from McEachern et al. (2000). Summer mean data from 10 lakes in unburnt and 10 lakes in burnt catchments are compared with sample mean data from Highratio, Midratio, and Lowratio Lakes. Shading indicates data from lakes with burnt catchments.



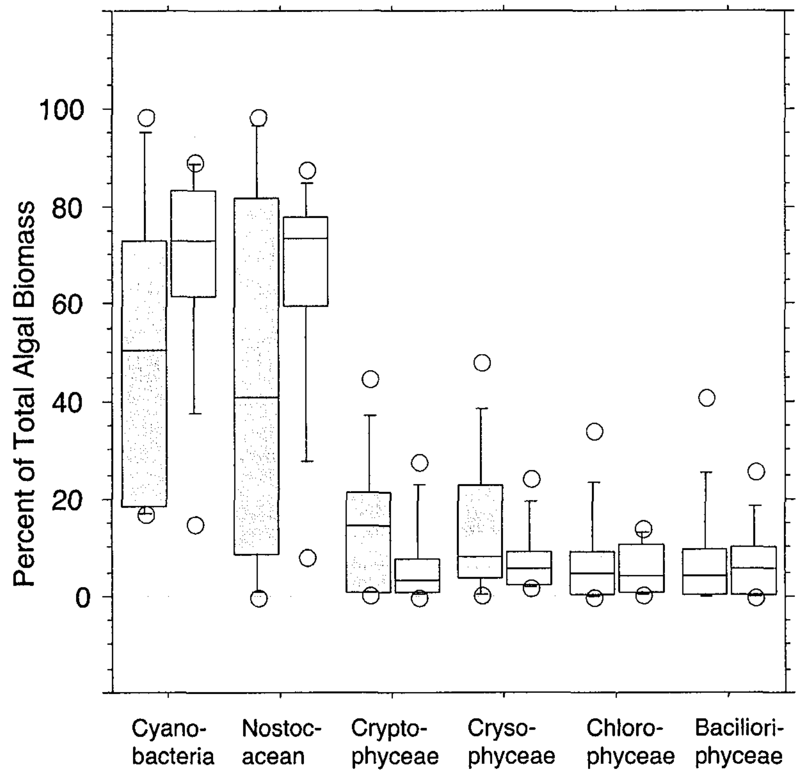


1.6
1.4
1.2
1
0.8
0.6
0.4
0.2
0
-0.2
-0.4
Predicted - Observed CHL
Y = 1.66 - 0.077 * X, n = 10
5
10
15
20
25
TN:TP ratio

Fig. 4-3: Response of CHL in nutrient stimulation experiments (10 L microcosms) in three lakes. Response is final CHL divided by initial CHL and thus represents biomass in response to nutrient ammendments and light. The treatments are C= control, N= +500 to 1000 μg/L [N], P = +50 to 100 μg/L [P] and NP = combined +N +P. Error bars represent one standard error from the mean. Small case letters represent differences in orthogonal comparisons of mean CHL at P < 0.01 (3 X 3 repeated measures, Bonferroni-Dunn). Horizontal line represents final/ initial CHL observed in the lake over the incubation period. All NSE responses are different from final lake CHL at $P < 0.01$.



rcent of total phytoplankton biomass of the main taxa for 10 lakes in unburnt (open boxes) and 10 lakes in burnt (shaded boxes) catchments. The nitrogen fixing suborder of Cyanobacteria, the Nostocaceans, are also presented. Peridineae and Eugleneae are not shown because they comprised < 5% of phytoplankton biomass. The Cyanophycea box includes the Nostocaceans.



References

APHA (1995). Standard Methods for the Examination of Water and Wastewater. American Public Health Association.

Bayley, S. E., Schindler, D. W., Beaty, K. G., Parker, B. R., and Stainton, M. P. (1992). Effects of multiple fires on nutrient yields from streams draining boreal forest and fen watersheds nitrogen and phosphorus. *Can. J. Fish. Aquat. Sci.*, 49: 584-596.

Bourelly, P. 1972. Les algues d'eau douce: Tome I: Les Algues Vertes. N. Boubée editions, Paris, France.

- 1981a. Les algues d'eau douce: Complément Tome I: Algues vertes. N. Boubée editions, Paris, France.

- 1981b. Les algues d'eau douce: Tome II: les algues jaunes et brunes ChrysophycÈes, Phéophycées, Xanthophycées et Diatomes. N. Boubée editions, Paris, France.

- 1985. Les algues d'eau douce: Tome IIII: les algues bleues et rouges les Eugléniens, Peridiniens et Cryptomonadines. N. Boubée editions, Paris, France.

Carignan, R., D'Arcy, P., and Lamontagne, S. (2000). Comparative impacts of fire and forest harvesting on water quality in Boreal Shield lakes. *Can. J. Fish. Aquat Sci.*, 57(Supp 2): 105-117.

Crumpton, W., Isehart G. M., and Mithell P. R. (1992). Nitrate and organic N analyses with second-derivative spectroscopy. *Limnol. Oceanogr.,* 37: 907-913.

Dillon, P. J., and Kirchner, W. B. (1975). The effects of geology and land use on the export of phosphorus from watersheds. *Water Research*, 9: 135-148.

Findlay, D. L. and H. J. Kling, 1979. A species list and pictorial reference to the Phytoplankton of Central and Northern Canada: Part I and Part II. Fisheries and Marine Service, Manuscript Report 1503. Department of Fisheries and the Environment, Winnipeg, Canada.

Hecky, R. E., and Kilham, P. (1988). Nutrient limitation of phytoplankton in freshwater and marine environments: A review of recent evidence on the effects of enrichment. *Limnol. Oceanogr.*, 33: 796-822.

Hecky, R. E., Campbell, P., and Hendzel, L. L. (1993). The stoichiometry of carbon nitrogen and phosphorus in particulate matter of lakes and oceans. *Limnol. Oceanogr.*, 38: 709-724.

Jonasson, P. M. (1996). Limits for aquatic life. *Verh. Internat. Verein. Limnol.,* 26: 1-33.

Jones, C. A. (1990). Internal phosphorus loading related to mixing and dilution in a dendritic shallow prairie lake. *Res. J. Water Pollut. Control*, 62: 847-852.

Kilham, S. S. and P. Kilham (1984). The importance of resource supply rates in determining phytoplankton community structure. In: Trophic interactions within aquatic ecosystems. D.G. Meyers, J.R. Strickler (eds.) AAAS Selected Symposium 1985. Pg 7-27.

Knowlton, M. F., and Jones, J. R. (1996). Experimental evidence of light and nutrient limitation of algal growth in a turbid midwest reservoir. *Arch. Hydrobiol.*, 135(3): 321-335.

Lund, J. W. G., C. Kippling, and E. D. Lecren, 1958. The Inverted Microscope Method of Estimating Algal Numbers and the Statistical Basis of Estimation by counting. *Hydrobiol.* 136:31 46.

McEachern, P., Prepas E. E., Gibson, J. J., and Dinsmore, P. (2000). Forest fire induced impacts on phosphorus, nitrogen and chlorophyll *a* concentrations in boreal subarctic lakes of northern Alberta. *Can. J. Fish. Aquat. Sci.,* 57(Supp/2): 73-81.

Mitchell, P and E. E. Prepas (1990). Atlas of Alberta Lakes. University of Alberta Press, Edmonton, Alberta.

Minshall, G. W., C. T. Robinson and D. E. Lawrence (1997). Postfire responses of lotic ecosystems in Yellowstone National Park, U.S.A. *Can. J. Fish. Aquat. Sci.* 54: 2509-2525.

Morris, D. P. and W. M. Lewis (1988). Phytoplankton nutrient limitation in Colorado mountain lakes. Freshwater Biol. 20: 315-327.

Planas, D., Desrosiers, M., Groulx, S.-R., Paquet, S., and Carignan, R. (2000). Pelagic and benthic algal responses in eastern Canadian Boreal Shield lakes following harvesting and wildfires. *Can. J. Fish. Aquat. Sci.,* 57(Supp 2): 136-145.

Prepas, E. E. & F. H. Rigler (1982). Improvements in quantifying the phosphorus concentration in lake water. *Can. J. Fish. Aquat. Sci.* 39:822-829.

Prepas, E. E., and Trew, D. O. (1983). Evaluation of the phosphorus-chlorophyll relationship for lakes off the Precambrian Shield in western Canada. *Can. J. Fish. Aquat. Sci.,* 40: 27-35.

Prepas, E.E., D. Planas, J.J. Gibson, D.H. Vitt, T.D. Prowse, W.P. Dinsmore, L.A. Halsey, P.M. McEachern, S. Paquet, G.J. Scrimgeour, W.M. Tonn, C.A. Paszkowski, K. Wolfstein (2001). Landscape variables influencing nutrients and phytoplankton communities in Boreal Plain lakes of northern Alberta: A comparison of wetland- and upland- dominated catchments.*Can. J. Fish. Aquat. Sci.* **58:** 1286-1299.

Prescott, G.W. (1962). Alagae of the Western Great Lakes area. Otto Koelts Science Publishers ed. Koenigstein, Germany. Reprint 1982.

Rask, M., K. Nyberg, and S.-L. Markkanen, (1998). Forestry in catchments: Effects on water quality, plankton, zoobenthos and fish in small lakes. *Boreal Environ. Res.* **3:** 75-86.

Reynolds, C. S. (1984). The ecology of freshwater phytoplankton. Cambridge University Press, Cambridge.

Reynolds, C. S. (1994). The long, the short and the stalled: On the attributes of phytoplankton selected by physical mixing in lakes and rivers. *Hydrobiologia*, 289: 9-21.

Riley, E. T. and E. E. Prepas (1984). Role of internal phosphorus loading in two shallow productive lakes in Alberta, Canada. *Can. J. Fish. Aquat. Sci.,* 41: 845-855.

Sartory, D. P. and Grobbelaar, J. U. 1986. Extraction of chlorophyll a from freshwater phytoplankton for spectrophotemetric analysis. Hydrobiology. **114:** 117-187.

Strong, W. L. and K. R. Leggat (1992). Ecoregions of Alberta. Alberta Forestry, Lands and Wildlife, Land Information Services. Edmonton.

Schindler, D. W. (1975). Whole lake eutrophication experiments with phosphorus, nitrogen, and carbon. *Verh. Internat. Verin. Limnol.* 19: 3221-3231.

Smith, V. H. (1983) Low nitrogen to phosphorus ratios favour dominance by blue-green algae in lake phytoplankton. *Science* 221: 669-671.

Tilman, D. (1978). The role of nutrient competition in a predictive theory of phytoplankton population dynamics. *Mitt. Internat. Veriein. Limnol.*, 21: 585-592.

Tilman, D., Kiesling, R., Sterner, R., Kilham, S., and Johnson, F. (1986). Green, bluegreen and diatom algae: Taxonomic differences in competitive ability for phosphorous, silicon and nitrogen. *Arch. Hydrobiol.*, 106(4): 473-485.

Welschmeyer, M. A. (1994). Fluorometric analysis of chlorophyll a in the presence of chlorophyll b and pheopigments. *Limnol. Oceanogr.* 39: 1985-1992.

Wintermans, J. F., and DeMots, A. (1965). Spectrophotemetric characteristics of chlorophylls a and b and phaeophytins in ethanol. *Biochim. Biophys. Acta*, 109: 448-453.

Zhang, Y., and Prepas, E. E. (1996). Regulation of the dominance of planktonic diatoms and cyanobacteria in four eutrophic hardwater lakes by nutrients, water column stability, and temperature. *Can. J. Fish. Aquat. Sci.*, 53: 621-633.

Chapter 5:
Landscape control of water chemistry in northern boreal streams of Alberta.

Introduction

An understanding of source areas and timing for runoff generation from catchments is essential to model and manage aquatic systems. Determining sources for runoff is difficult in northern boreal catchments because they often drain landscapes dominated by peatlands with complex hydrology. The prediction of flow paths and flow rates for runoff in northern boreal catchments is complicated by the anisotropic nature of peat soils and the variable storage capacity of peatlands (Schwartz and Milne-Home 1982a, Holecek 1988). The existence of frost or permafrost further complicates modeling water flow through these systems (Pietroniro et al. 1996). In western Canada, particularly Alberta, the boreal forest developed over glacial tills which add to the difficulty in describing recharge, flow and discharge from local and regional groundwater aquifers (Evans et al. 2000, Devito et al. 2000). Simple tools for describing the interaction of surface soils, groundwater discharge and surface runoff, if it occurs, are required to understand disparities between general models and hydrologic reality in peatland systems and the potential impacts from forestry and other anthropogenic disturbances in these remote watersheds.

Logging in Canada is moving towards a natural disturbance model termed emulation silviculture (McRae et al. 2001), where small watersheds (<10 km^2) could have a majority of their trees removed including those in riparian areas. This paper describes the pre-disturbance hydrology of six streams representing small (<□6 km^2) hillslope and lowland forested peatland catchments in northern Alberta. The hydrologic controls are investigated here to provide context for a study that will evaluate impacts from emulation silviculture. Emulation silviculture is likely to change surface water chemistry by altering the quantity and flow paths for water and the relative importance of overland flow, infiltration and discharge from organic or mineral soils. The hypotheses investigated in this pre-disturbance study were: 1) chemically dilute sources (precipitation, overland flow) are important contributors to stream discharge during snowmelt and summer storm events; 2) a majority of summer discharge is supplied from organic soils; 3) mountain sites will differ from lowland sites by demonstrating greater contributions from groundwater (contributions from organic soil will be less important). The hypotheses derive from a well developed understanding of hydrology in single peatland systems mostly without permafrost (e.g., Branfireun and Roulet 1998, Metcalfe and Buttle 2001). However, complexity and a lack of tools to assess runoff in peatlands has made it difficult to apply general hydrologic knowledge in the discontinuous permafrost zone of boreal forests at the catchment scale. Our current understanding of hydrology in boreal peatlands and the background for chemical mixing analysis refined in this chapter for assessing general catchment hydrology is briefly reviewed below.

Determining the degree to which precipitation, surface soils and groundwater contribute to stream discharge in upland and lowland settings is essential for managing impacts from forestry. Water movement in northern peatlands must account for long-lasting seasonal frost and discontinuous permafrost. Runoff should be pronounced in the spring as snowmelt encounters low soil permeability and flows over ice to fill depressions and wetlands (Woo and Winter 1993). As spring progresses, the probability for runoff to follow surface or shallow soil pathways (throughflow) decreases because infiltration rates in the surface moss and peat layer (*acrotelm*) is high (1 $cm \cdot s^{-1}$: Ingram 1983, Puurveen et al. 1997). The glacial tills that characterize northern Alberta have a wide range in hydraulic conductivities (Evans et al. 2000) from clays (10^{-10} cm/s) to sand and gravel lenses (10^{-3} cm/s). Water infiltrating from overlying peat moves as pipe flow along the peat-mineral interface if clays are encountered (Hill 1993, Evans et al. 1999) and infiltrates to aquifers through lenses of high hydraulic conductivity (Evans et al. 2000). Flow through peat may be restricted to the peat-mineral interface (interflow) by low hydraulic conductivity of clay below and compact, anaerobic peat in the catotelm (Evans et al. 1999) and the existence of concrete frost within peat well into summer (Woo and Winter 1993).

Storm discharge from fens and other saturated peatlands is often described as piston flow where precipitation forces previously infiltrated water from saturated soils. The older water forms discharge with a distinct chemistry of the peatland (Branfireun and Roulet 1998). Peatlands will produce pre-event water under all but the heaviest storms (Cirmo and McDonnell 1997) and the resulting discharge can contain elevated concentrations of mineral ions if piston flow (saturated) or interflow and throughflow (unsaturated) forces water in contact with mineral soils to the stream channel. If piston flow is the predominant process generating water discharge from peatlands, stream water chemistry should be dominated by groundwater of mineral or peat origin and surface runoff with a dilute precipitation signature should be absent. The importance of mineral versus peat sources should depend on the availability of highly conductive lenses in mineral soils, slope characteristics for producing throughflow, and water table fluctuations within peatlands. The degree to which runoff is chemically similar to precipitation should depend on the distribution of wetlands and only occur under exceptionally high water table conditions because piston flow and throughflow would otherwise produce water that had been in contact with organic or mineral soils. These distinctions between sources for stream water are important when considering potential impacts from timber harvesting and can be tested with chemical tracing techniques.

One method for reconstructing the relative importance of peatlands, precipitation and groundwater in runoff generation is end-member mixing analysis (EMMA). The amount of water entering a stream from different source areas in a watershed can be reconstructed from knowledge of source area chemical characteristics (Christophersen and Hooper 1992). Termed end-members, the waters delivered from soil horizons and precipitation form a physical basis for modeling water sources (Ogunkoya & Jenkins 1993). The assumptions required for the existence of end-members highlight their use and potential flaws: 1) the end-members must be consistent across the landscape and chemically distinct from each other; 2) when water is transported and mixed, the chemical parameters are conservative; 3) a reasonable sampling regime can adequately represent the end-members. The importance and potential failures in the assumptions required to implement EMMA have been thoroughly reviewed (Peters et al. 1990, Neal and Christophersen 1990, Muller et al. 1993); however, I discuss violations of each of the assumptions in this study and solutions that allow EMMA to provide insights for assessing dominant soilwater - stream water relationships.

I present hydrology and chemistry data from streams in two regions, one with steep topography and few peatlands, the other with flat topography and catchments dominated by peatlands to test the hypotheses presented in paragraph two. Mixture modeling with chemical constituents through EMMA provided inference as to likely soil horizons contributing water to stream discharge, while similar modeling with the isotopes deuterium and ^{18}O provided inference as to whether discharge was from recent or previous precipitation events.

Methods

Two streams draining the Caribou Mountains and four streams draining the Cameron Lowlands were selected for this study (Fig. 5-1) and were sampled from June 1998 through October 1999. The mountain streams lay in well defined catchments (watersheds MH and MR) dominated by spruce with a moss, lichen and shrub understory typical for Labrador tea-subhygric ecosites (Ecosites of Northern Alberta, Natural Resources Canada). The lowland streams lay in poorly defined catchments (watersheds LH1, LH2, LH3 and LR) with low topographic relief and large networks of interconnected wetlands typical for treed bog and fen ecosites (Table 5-1). The locations were 250 to 300 km south of the zone defined as containing discontinuous permafrost and 100 km (mountain) or 200 km (lowland) north within the zone of sporadic discontinuous permafrost (Prowse 1990). The stream channels were usually narrow (< 3 m) and relatively incised (0.5 to 1.5 m) in both lowland and mountain sites. Long-term hydrologic records from drainages of similar size are not available. However, median discharge in 1999 from the Ponton River, the major Caribou Mountain drainage, was the 40th percentile and total discharge was 12% above average for the last 40 years of record. Flow in 1998 was slightly lower than 1999 making it an average year compared to long-term conditions.

Average May through October and November through April temperatures were 7 and -14 °C, respectively. Mean daily temperatures drop below zero in mid-October and return above zero in mid-April. The warmest month is July with an average temperature of 14 °C and January is the coldest month at -21 °C (Environment Canada Climate Normals). Surface soils in both areas are Orthic Gray Luvisols with roughly equal proportions of illite and montmorillonite clays comprising approximately 40% of the total soil profile (Pawluk 1961). In total, clay and silt average 70-80% of the mineral soil profile. Total exchangeable cations of mineral soils were estimated in both regions at 20 to 40 meq 100 g^{-1} in the A and 24 meq 100 g^{-1} in B horizons, total soluble salts were typically < 10 meq 100 g^{-1} (Alberta Research Council, 1962). Organic soils are 10 to 20□cm deep at well sites and are composed primarily of *Sphagnum* spp., and a variety of feather and brown mosses. In bog and fen sites, organic soils extended down to 1 m. Wells placed in these sites contained ice that thawed completely through the summer or not at all depending on exposure. Geologic formations in the mountain catchments included fine grained feldspathic sandstone with calcareous beds and laminated siltstone of deltaic and marine origin (Dunvegan formation) over marine shale, silt and sand (Shaftesbury formation). The lowland catchments were underlain by the Shaftesbury formation with fossiliferous siltstone and thin beds of concretionary ironstone. Selenitic gypsum crystals were ubiquitous in road fill dugouts in the lowland area. Figure 5-1 and Table 5-1 outline the physical parameters for the study catchments. Streams in the two regions differ in channel slope and percentage of wetlands comprising their drainage basins. Watersheds were estimated from stereo air photo interpretation (1:20 000) and were later ground-truthed on foot with a GPS. Air photo interpretation and ground-truthing differed slightly from boundaries suggested by published topographic maps (Canada NTS, 1:50000) in the upper watersheds for mountain sites and to a larger degree in lowland sites. Watersheds from air photo and ground-truthing are the basis for water yield calculations.

Each stream was divided into 2 (lowland) or 3 (mountain) reaches of approximately equal length. Analyses presented here focus on the most downstream site on each stream. Data from upstream sites are not reported here but were used to check downstream data. At each site, stream discharge was calculated from cross sectional area and stream velocity measurements using a Swoffer current meter at a minimum of five times through the summer. Stream depth was monitored at 15-min intervals with a submerged pressure transducer and discharge was calculated using stage-discharge curves independently derived for each stream. Rating curves were based on data from 1998 through 2001 with 20 or more measurements per stream. Storm discharge was collected at hourly intervals in 1998 with ISCO samplers set to start a cycle with a 1 cm rise in stage. In 1999, ISCO samplers collected 1 L samples every 24 h throughout the ice-free period. The sampling frequency was changed in 1999 as a compromise with poor site access to ensure daily samples were collected. An assessment of 1998 data indicated that sampling at 24 h intervals could adequately capture changes in stream chemistry following storm events.

Except when storms occurred, one-liter samples were collected for analysis every 72 h. During storms events, 24 h interval samples were collected. Data collection began in June 1998; snowmelt data were not collected in 1998. The onset of snowmelt discharge in 1999 was estimated by the first occurrence of change in stage recorded by pressure transducers placed on the ice in the stream channels. Stage measurements could not be converted to discharge volumes during the rising limb of the melt hydrograph because all flow occurred in a channel over the ice surface which changed shape daily. Instead, the rising limb of snowmelt discharge was interpolated by correcting stage measurements to intersect peak discharge assuming the deepest recorded stage corresponded to peak discharge, no influence from channel ice, and accurate estimation of discharge from the stage-discharge curves derived during ice-free periods. The assumption that ice did not influence channel shape at peak discharge was supported by site visits in 1999 and subsequent years corresponding to or within days before and after peak discharge. In lowland sites, snow melt discharge was reconstructed as described above and supplemented with additional information from an Environment Canada gauging station on the Steen River to which the study streams are tributaries.

Precipitation was collected continuously in U.S. Weather Bureau standard rain gauges checked twice daily by Alberta Forest Service staff at two fire towers. The fire towers were located 500 m north of the mountain and 13 km south of the lowland sites. Precipitation depths were also available from shielded Sacramento gauges at both fire towers that were checked in April and October. The rain and Sacramento gauges are part of Alberta's long-term climate network and were installed to Environment Canada standards. The precipitation depth data were validated by the Alberta government prior to public release. Due to the distance between the lowland sites and its closest climate station, precipitation estimates from the lowland fire tower were verified with volumes in on-sight collectors retrieved monthly. Additional 1998 precipitation data were available from an automated recording gauge at the lowland sites, however, the gauge and its replacement were destroyed by bears. Daily precipitation values from the lowland tower were not corrected with the data from the on-site collectors because the monthly sums agreed to within variability of the on-site collectors (±20%). At the mountain fire tower, the total volume of precipitation was combined daily into high density polyethylene bottles and immediately frozen for later chemical analysis. In the lowland catchment, rainfall was collected in three containers with a layer of mineral oil to limit evaporation in 1998. The lowland composite collectors were replaced in 1999 with funnels connected to collection flasks with tygon tubing to increase collected volume and limit organic debris previously trapped in the mineral oil. A loop in the tubing and glass wool maintained a water trap that reduced evaporation. Chemical analyses were performed on a single composite sample created each year for mountain and lowland sites from all precipitation collected at the fire tower and the lowland composite collectors, respectively. Snow was sampled over a three day period between March 27 and April 5 of each year for depth, density and chemistry with a 10-cm diameter core. Between five and ten transects were chosen to represent snow accumulation in different vegetation and aspect conditions for each catchment. Between 20 and 40 cores with associated depth measurements were collected at random distances along each transect. Random cores within each transect were later combined, thawed and processed as water samples for chemistry analyses.

To collect groundwater from mineral and organic horizons for chemistry analysis, 19 and 11 wells were installed in mountain and lowland catchments, respectively. Groundwater wells in mineral soil were installed to 2 to 3 m depths into gravel/sand lenses between layers of clay. The wells were constructed from polyvinylchloride pipe (1 1/4 inch OD) and collected water through horizontal slits (0.01 mm thick, 6 mm apart) over the bottom 30 cm of well length. Wells for organic soils were placed in bog and fen waters to the depth of permafrost or mineral soil and collected water through vertical slits from top to bottom. All wells were placed inside polypropylene fabric (NILEX MD7407, pore size < 100 μm) to limit sediment accumulation in the wells. In two bogs and two fens, wells were placed along assumed flow paths to streams with a well at the uphill margin, the center, and near the base close to the stream. Distance between wells depended on the size of the peatland ranging from 10 to 30 m apart. Groundwater wells were also placed along assumed flow paths. Two uphill wells (mountain) rarely produced water and were abandoned and reinstalled in valley bottoms.

Processing of stream and groundwater samples began within 4 h of collection. ISCO samples ranged between one and 24 d old prior to processing. The first sample (oldest) was always collected at the same time as stream grab samples. Results from the stream grab (processed on the collection day) and the ISCO sample (processed up to 24 d later) were compared to ensure storage did not alter the ion chemistry of the samples. Triplicate 25-mL aliquots were pipetted into 30-mL glass tubes used directly in the digestion process and nutrient analysis. Unfiltered water was used for total nitrogen (TN) and total phosphorus (TP) analyses. Filtrate (Millipore HA, 45 μm) was used for dissolved nitrogen (DN), dissolved phosphorus (DP), soluble reactive phosphorus (SRP), nitrate-nitrite (NO_3^--N) and ammonium analyses (NH_4^+-N). Water for TN and DN samples was preserved with 10 μL of 40% H_2SO_4 for storage (1-2 wk), then neutralized with equivalent NaOH prior to digestion in the laboratory. TP and DP were analyzed spectroscopically (5-cm cell) from persulfate-oxidized samples by molybdate blue absorption (Prepas & Rigler 1982). TN and DN concentrations were determined by second derivative spectroscopic analysis of persulfate oxidized samples (Crumpton *et al.* 1992). Water for ammonium, nitrate and SRP samples were frozen in 125-mL polyethylene bottles for later analysis by indophenol blue (NH_4^+-N), cadmium reduction (NO_3^--N) with automated colorimetry (Technicon methods 100-700 W/B and 155-, respectively) and by molybdate blue absorption (SRP). Total suspended solids (TSS), nonvolatile suspended solids (NVSS), and volatile suspended solids (VSS) were collected on pre-ashed GF/F filters (mean particle retention 0.7 μm) and analyzed in duplicate after APHA (1993). Alkalinity was determined by the Hach phenolpthaline/ bromocresol method (APHA 1993). Filtrate from GF/F filters for cations (Ca^{2+}, Mg^{2+}, Na^+, K^+), dissolved organic (DOC), and anions (Cl^-, SO_4^{2-}) analysis were stored in 60-mL polyethylene bottles. Water for cation and DOC samples were preserved with 40% H_2SO_4, anion samples were refrigerated. Cations were analyzed by atomic absorption flame spectroscopy (Perkin-Elmer AS90, AA3300), DOC by high temperature catalysis (Shimadzu TOC-5000), and SO_4^{2-} and Cl^- concentrations by chromatography (Dionex 2000i/SP).

EMMA followed published methods (Christophersen and Hooper 1992) summarized below. Stream water concentrations for Ca^{2+}, Mg^{2+}, Na^+, K^+, SO_4^{2-}, alkalinity, and DOC were normalized by subtracting the mean and dividing by the standard deviation for each parameter at each site. A correlation table for the normalized values was then input into Principle Components Analysis (Statview 4.1) and resulting eigenvectors were used to project each stream sample into U-space. Samples from groundwater wells, peatland sites, and precipitation were similarly normalized and projected into the unique U-space for each stream. The recommended procedure of selecting median values for potential end-members was appropriate only for precipitation in both mountain and lowland sites. Median values were not appropriate for groundwater and peat water in either mountain or lowland catchments because these demonstrated seasonal hysteresis with individual samples falling along lines connecting maximum values for groundwater, peat water, and precipitation (see Fig. 5-7 for example). Changes in groundwater chemistry coincided with the loss of frost in wells suggesting that groundwater samples prior to July were mixtures that obtained a greater groundwater and lesser peat water signature as frost melted. Thus, the assumption that end-member water chemistry was in equilibrium with the end-member was violated for mean or median values and the suggested method (Christophersen and Hooper 1992) of selecting median end-member values was not valid. July and August ion concentrations were selected instead to represent groundwater end-members as these were maximum values in the seasonal pattern and were assumed to be the closest approximation of soilwater in equilibrium with the soil end-member. The peat water end-member in mountain catchments was similarly determined. Organic soilwater and peat water are synonymous in this study because all organic soil wells were placed within the accumulated peat profile of bogs or fens. The proportion of end-member contribution to each stream sample was determined by solving the set of linear equations (Christophersen and Hooper 1992). One stream sample plotted outside the constraints set by the three end-members. The point was moved to the nearest boundary and the proportions were calculated based on two end-members (Peters et al. 1990). PCA was used to derive U-space. Statistics applied to U-space coordinates and chemistry are based on discriminant analysis (Statistica 6) where indicated and *P* values are reported.

The natural isotopes of water were collected in 30 mL polyethylene bottles for stream grab samples, all groundwater samples and precipitation, including snow collected during March surveys. These, plus additional ISCO stream samples from storm events were analyzed for the relative enrichment of deuterium and ^{18}O compared to Standard Mean Ocean Water (SMOW) by mass spectroscopy at the University of Waterloo. Deuterium and ^{18}O were analyzed from a total of 95 end member samples and 161 stream samples. Isotope samples collected from streams every 72 h could not be analyzed due to budget constraints.

Results

Hydrologic conditions of the study streams

The study streams represent two contrasting hydrologic settings found in northern Alberta; relatively well drained hill slopes and lowlands with poor drainage. Snowmelt was expected to provide the largest water yield to streams in both settings and higher peak discharges were expected in catchments with steeper slopes. This was generally the case for the two hillslope streams where snowmelt provided peak (> 400 L s^{-1}) instantaneous discharge, 50% of total summer discharge and water yields greater than the volume of snow water calculated prior to melt (Fig. 5-2, Table 5-3). Peak flow from summer storms could exceed peak flow from snowmelt in the mountain catchments but did not exceed the total volume or water yield contributed during snowmelt. In lowland catchments, water yield and peak snowmelt discharge was small (10 to 200 L s^{-1}) compared to peak discharge following summer storms (> 1.5-fold higher, Fig. 5-3). Except in LH2, water yield from lowland streams during snowmelt was at least 20-fold lower than from mountain streams (Table 5-3). Snow melt water yield from LH2 was still several fold below yield from mountain catchments. In the catchments with low slope, snowmelt water yield and peak discharge were low because melt waters were stored in soils and wetlands through the spring.

Water yield from mountain streams during snowmelt exceeded estimates of available water from the snow pack (47 mm) and spring precipitation (45 mm) by 1.4-fold while water yield from lowland streams was a fraction (see runoff coefficients, Table 5-3) of the precipitation available during the melt period (82 mm snow and 92□mm rain). Channel ice from previously melted precipitation and groundwater discharge may have contributed to the extra water in mountain streams during the melt period. Channel ice was variable but extensive in the mountain catchments at snowmelt, ranging from 20 to 200□cm thick and 3 to 20 m wide. However, channel ice was estimated to store a maximum of an additional 7 mm in both basins and could not explain the error in snow melt water balance for mountain sites. The elevated runoff compared to available precipitation could not be attributed solely to error in discharge estimates despite difficulties with extensive channel ice. Error in estimating snowmelt discharged was constrained to the ascending limb (see methods) because the onset, peak and recession were accurately represented. The Caribou Mountains are a regional topographic high, a plateau providing groundwater recharge with subsequent discharge as springs into the sloped watersheds below the plateau. These springs were apparent where soil slumping occurred along the steep slopes of the northern watershed boundaries for both mountain streams (Fig. 5-1). The additional 39 mm of water discharged during snowmelt may have derived from this regional groundwater aquifer.

Following snowmelt in mountain streams, mean discharge in June was only 30% of average August through October baseflow indicating a large portion of groundwater supply was depleted or held in soils by frost throughout June. Frost remained in mountain groundwater wells at depths between 1 and 2 m into mid July and may have contributed to reduced discharge from interflow or throughflow. In mountain watersheds, drainage of snowmelt from hillslopes occurred while the ground was largely frozen and resulted in high water yield, poor recharge of soil moisture, and relatively low water yield from subsequent rain events until July precipitation.

An opposite pattern of snowmelt discharge occurred in lowland catchments. Except in catchment LH2, mean discharge over the period was less than 6% of precipitation and snowmelt water. In catchment LH2 discharge approached half of available water from precipitation and snowmelt. Following snowmelt in lowland catchments, 44 mm of rain produced 10 to 100-fold increases in discharge, whereas similar events in mountain catchments failed to produce a rise in stream discharge. In lowland catchments, storage of snowmelt in peatlands, beaver ponds and organic soils resulted in saturated soils, low water yield over the snowmelt period and high water yield during subsequent rain events.

Hillslope and lowland streams displayed contrasting summer discharge patterns as a result of low versus high snowmelt storage, respectively. The mountain streams did not begin to respond to rainfall until mid July despite 110 mm rainfall through June 1999 and 67 mm through June 1998. July stream response in mountain catchments corresponded to the loss of ice in groundwater wells and a rise in water table to the upper 50 cm in some valley bottom wells (Fig. 5-4). In contrast, the lowland streams responded to all precipitation events > 5 mm/d throughout June and July. Discharge in the lowland streams slowly attenuated throughout the summer in correspondence to a gradual 50 cm decline in water table for all mineral soil wells and approximately 20 cm in peatland wells (Fig. 5-5).

Substantial summer precipitation was required in mountain catchments before stream discharge began to increase. Between July 5 and 11, 1999, two storms deposited

21 mm and 101 mm, respectively, over 48 h periods. Discharge responded marginally to the first storm, however, response was rapid and large to the second (Fig. 5-2a) indicating the initial 21 mm was required to fill peatlands and soilwater storage in addition to 81 mm of precipitation between snowmelt and July 5. An almost identical pattern in rainfall and discharge response occurred in 1998 when 30 mm of rain fell on July 8 followed by 144 mm from July 10 through July 12. Again, a rise in stream discharge did not occur until the second storm event, despite the 67 mm that fell in June.

In lowland catchments, stream discharge increased in response to spring rain events. Responses to storms in lowland catchments were markedly different from mountain sites because soils were saturated when rainfall began. An early June event (1999) deposited 44.3 mm over 48 h in the lowlands and seemed essential in releasing stored water from soils and wetlands. All subsequent storms through July that deposited more than 5 mm in 24 h produced noticeable responses in each lowland stream. Except in LH3, each storm response occurred as a peak along a gradual descent in "baseflow" discharge from early June to September (Fig. 5-3). In LH3, streamflow between precipitation events remained close to the detection limit likely because LH3 flowed through a series of wetlands formed in abandoned beaver ponds.

Mixing analysis (EMMA)

Water samples collected from wells, pooled water in fens, the spring snowpack, and precipitation samples were screened as potential end-members for mixing analyses. Figure 5-6 presents a Principle Component Analysis projection of ion concentrations for soil types from which the end members, precipitation, peat water and groundwater were selected. Snow and rain waters were similar between mountain and lowland catchments for the ions incorporated in EMMA (discriminant $P > 0.1$). However, some rain samples from the lowlands contained DOC likely as a contaminant in the composite collectors. Due to the similarity between snow and rain in chemistry parameters for EMMA, the median of combined snow and rain was chosen to represent precipitation end-members in both mountain and lowland catchments.

Groundwater concentrations of all ions and the proportion of ions to total cations and anions from mineral horizons were different between mountain and lowland sites (discriminant, $P << 0.01$). The two regions thus required different estimates of groundwater chemistry despite similarities in geology. In addition, ion concentrations in groundwater from both regions increased through the summer while DOC concentrations declined. Mixing of water from organic and deeper soil horizons likely caused the chemical changes in groundwater samples from May through July. Mixing was particularly apparent for wells located in riparian areas at valley bottoms as these plotted closer to precipitation and peat than wells in upland locations (Fig. 5-7). The closest approximation of a groundwater end-member was assumed to occur at the most extreme distance from dilution by organic or precipitation end-members. Extreme U-space positions for individual groundwater well samples, relative to the origin and other end-members, occurred in upland wells during July or August (Fig. 5-7). Groundwater samples from May, June, and during large storm events occurred along two lines connecting groundwater to extreme positions for precipitation or peat water. Due to the spatial and temporal differences in groundwater, separate mountain and lowland determinations of median U-space position for July and August groundwater samples were used to represent groundwater end-members (Fig. 5-6).

Peat water from both regions were different in their combined ion properties (Fig. 5-6, discriminant, $P << 0.01$). Of the individual ions, Ca^{2+} and total alkalinity were elevated in mountain compared to lowland sites (t-test, $P = 0.03$) while Na^{+} and SO_4^{2-} were in higher concentrations in the lowlands (t-test, $P <<0.01$). Seasonal fluctuation in ion concentrations of peat water, like groundwater, were represented in U-space along a gradient between peat and precipitation end-members in a similar fashion as groundwater from mineral soil. Separate estimates were therefore required for mountain and lowland peat waters (Fig. 5-6).

Mixing analysis supported snowmelt discharge versus storage as the primary differences in hydrologic response between mountain and lowland catchments. Initial snowmelt in the mountain catchments was dominated (> 70%) by groundwater with an organic or mineral signature despite ice in wells and the stream channel. By the second week of melt, groundwater was replaced with precipitation comprising more than 50% and as much as 80% of total discharge (Fig. 5-2). Contributions to discharge from groundwater returned to >60% following snowmelt and remained above 50% through June and early July for catchment MH but in MR were lower by approximately 20% through to September (Fig. 5-2). Contributions to discharge from precipitation in both mountain watersheds remained below 20% through most of June despite substantial rainfall. July storms initially increased discharge with a precipitation signature but stream flow was quickly replaced by peat water and groundwater contributions (Fig. 5-2). The precipitation signature increased (> 40 %) in MH following August storms, however, water from organic and groundwater sources formed the majority of stream discharge in both mountain catchments (Fig. 5-2).

Unlike the mountain sites, source waters in lowland catchments were initially dominated by precipitation which gradually declined through the summer as did instantaneous discharge attributed to baseflow. During snowmelt, precipitation comprised 45 to 70% of stream discharge (Fig. 5-3). As watertables declined in lowland catchments, stream discharges became increasingly dominated by organic sources with the peat water end-member reaching 60 to 80% of total discharge by the onset of Fall ice-cover. Contributions to stream discharge from groundwater were relatively constant or increased marginally through the summer but remained below 30% in all streams. Peat water provided a majority of discharge at peak stream flow during summer storms suggesting that a majority of storm runoff was generated from soilwater stored in peatlands. Movement of water along the peat/mineral interface (throughflow) or directly from saturated peatlands (piston flow) were therefore the predominant processes generating runoff during storms in the lowland streams.

Deuterium and Oxygen-18 mixing analyses

The natural isotopes of water, deuterium and ^{18}O, are useful tracers for differentiating pre-event from event water because water becomes increasingly enriched with heavy isotopes over time as light isotopes evaporate. The enrichment signature is determined by climatic conditions prior to infiltration (Gat and Gonfiantini 1981). Groundwater typically has an isotope composition representative of long-term climatic conditions weighted for relative recharge from winter snow and summer rain (Gibson et al. 2002). Analyses based on ^{2}H and ^{18}O were similar to EMMA but were two component models where relative enrichment represented the mixture between end-members snow water and summer precipitation.

In all streams, isotope signatures during snowmelt did not shift towards less enriched snow. In mountain sites, isotope compositions of stream water at the onset of snowmelt were as much as 10% more enriched than the summer mean isotope composition for groundwater. The implication was that evaporation from the melting snow pack caused substantial enrichment during melt or that groundwater contributed a majority of runoff during the melt period. Water samples collected from fens surprisingly maintained an enrichment similar to groundwater during the melt period having mean ^{18}O and ^{2}H compositions of -18.8 and -145.7‰, respectively (see Fig. 5-8 as an example). Other peatland soils were also enriched compared to snow but were less enriched than stream waters. In lowland catchments, the isotope composition of stream water was enriched by 0.6% during snowmelt recession (when samples were available) compared to the post snowmelt mean for all streams combined. Enrichment during snowmelt indicated older soilwater was an important source of stream discharge during snowmelt recession. All lowland stream waters became successively enriched from July through September consistent with the steady rise in groundwater contributions suggested by EMMA.

In mountain catchments, groundwater in peat and mineral soils were similar in enrichment to summer precipitation (Fig. 5-8), suggesting that as much as 99% of sampled groundwater derived from summer precipitation. In mountain streams, isotope composition indicated that the majority of stream discharge following snowmelt could have derived from either groundwater or summer precipitation (88%), as these two sources were isotopically indistinguishable from each other. In lowland catchments, groundwater from mineral soils was comprised of roughly equal proportions of summer precipitation and snow water. The isotope composition of lowland stream waters indicated that 35% was derived from 1999 snow melt and 65% was derived from summer precipitation, on average. The isotope proportions observed in lowland streams could be achieved with stream discharge composed of approximately 70% groundwater and 30% surface runoff from summer precipitation with no surface runoff from snowmelt. Any mixture between these extremes was possible. During summer storm events, the isotope enrichment of stream waters increased in most cases to levels similar to those of fen waters. For example in three lowland streams the ^{18}O composition changed from -19.5 to -18‰ in three streams during an August storm. However, in LH3 enrichment rose to -13‰ during the same storm. Isotope analyses confirmed that groundwater provided large contributions to stream discharge during snowmelt in all catchments and particularly to stream LH3 which was heavily influenced by fens and wetlands. Summer discharge from mountain streams was comprised of relatively constant proportions of groundwater and rain with little evidence of dilution by less enriched snow water. In lowlands, enrichment slowly increased over the summer consistent with the increase in soilwater and storage of snow water in surface wetlands where evaporative enrichment occurred.

Discussion

Hydrograph shape in the mountain catchments was typical for years when snowmelt is an important component of annual runoff. In such years, peak discharge during snowmelt can be 2 or more times that observed during later rain events (Schwartz and Milne-Home 1982a, Caissie et al. 1996). In similarly-sloped small catchments, the rapid response of streams to snowmelt was attributed to surface runoff and release of pooled meltwaters during periods of high solar insolation (Dunne and Black 1971). In such systems infiltration and groundwater discharge are limited by concrete frost in upper soil horizons until most of the snowpack has melted (Metcalf and Buttle 2001, Woo and Winter 1993, Dunne and Black 1971). Catchments in northern boreal forests, where discontinuous permafrost and long seasonal frost occur, are distinct in that frost is located under a thick organic layer (Woo and Winter 1993) rather than at the soil surface (Dunne and Black 1971). In the mountain catchments, the concrete frost layer in groundwater wells was 20 to 30 cm below the moss surface during winter visits. Melt water infiltrated into the upper unfrozen soils and moved down gradient towards the stream channel and refroze within the top 30 cm of soil. Ice in the mountain stream channels was often stained brown consistent with organic carbon or iron and manganese content and likely derived from groundwater seepage throughout the winter (throughflow and return flow). The ice eventually grew to cover most of the riparian area in the mountain streams.

Stream discharge during snowmelt in the mountain sites was initially dominated by groundwater. In MH this was from mineral soil whereas in MR an organic soil signature was apparent. Interflow and groundwater discharge from the observed springs may have been more important in MH while throughflow exiting as surface runoff (return flow) in riparian wetlands may have predominated in MR. Though unexpected, the two processes may have been occurring through late winter giving the respective signatures to the accumulated channel ice that formed initial stream discharge. Groundwater was quickly replaced by runoff with more dilute chemistry suggestive of rapid discharge through shallow soil pathways which was consistent with the hypothesis and descriptions of other peatland forested sites (e.g., Metcalfe and Buttle 2001). Isotopic analyses confirmed that snowmelt runoff in the mountain streams was initially dominated by highly evaporated water. Combined with the mineral signature, initial melt water was likely of groundwater origin rather than from an evaporated snowpack. Mountain stream waters did not become less enriched as snowmelt progressed indicating that the chemically dilute runoff that followed initial groundwater as identified by EMMA was likely isotopically enriched (evaporated) snowmelt routed through peatlands and in particular through fens where isotopically enriched water was found.

In northern catchments where wetlands and peatlands are common, snowmelt typically results in a saturated landscape and sets the stage for summer drying (Woo and Winter 1993). However, June discharge from MH and MR were 30 and 80%, respectively, of the baseflow discharge following July storms. As hypothesized, melt water did not remain in the mountain catchments to recharge soil moisture. Instead, discharge from mountain catchments exceeded snow water equivalents by 30 to 40% indicating that most snow water was discharged from the catchment even when considering potential errors in calculations of snow-water yield. Discharge from regional groundwater was a potential source of additional water to the mountain streams due to their position below the Caribou Mountains Plateau which may also have contributed to the enriched isotope composition in fens during the snowmelt period.

In lowland catchments, stream discharge during snowmelt did not follow the pattern of high water yield during snowmelt; instead, snow water was retained on the landscape. As a result, peak snowmelt discharges were less than half of summer peak discharges and water yield during snowmelt was as little as half of total summer water yield in LH1 and LR. The watertable remained relatively high within organic soils throughout early June contributing to the rapid response of lowland streams to June precipitation. Following snowmelt, streams in the lowlands followed predictions for saturated catchments with rapid response in stream flow to even small precipitation events. Peak stream discharge following June precipitation corresponded to greater contributions from peat waters, indicating that piston flow rather than overland flow was the dominant runoff process. Stream discharge in similar peatland catchments was also dominated by organic soilwater flowing through riparian fens or along the peat mineral interface of bogs (Schwartz and Milne-Home 1982b, Metcalfe and Buttle 2001). Discharge from wetlands is usually dominated by pre-event water becoming event water only after intense storms (Hill and Waddington 1993, Cirmo and McDonnell 1997). In lowland catchments, isotope analyses indicated that some storms caused a shift in enrichment towards contributions from the event. However, EMMA indicated that summer storms were not large enough to produce a shift to dilute water indicating that even during large storms, residence time and mixing in organic soils was enough to provide water with chemical signatures characteristic of peatland sources.

In the lowland catchments, runoff was routed through bogs and fens. The contribution of peat water to stream discharge increased through the summer despite a decline in watertable. Bogs dominated the headwaters with stream channels formed at the outflows in graminoid and shrub fens. In upstream sites, patterns in source waters were identical to those observed downstream, indicating that the headwater fen complexes played a larger role in stream chemistry than the forest soils and riparian areas downstream where source waters would have indicated greater groundwater contributions.

The adaptation of simple tools to assess the sources for runoff generation was one objective of the pre-disturbance study. I applied mixture modeling using chemical tracers (EMMA) and the natural isotopes of water. EMMA developed from a need to identify soilwater sources for runoff coinciding with the data requirements of hydrochemical models such as MAGIC that predict impacts from human activities by routing water through a limited set of soil layers (e.g. Hooper and Christophersen 1992). EMMA generalizes from a potentially complex set of hydrologic processes to a limited representation of geochemical influence on stream discharge chemistry at the catchment scale. For example, EMMA can provide insight as to the quantities of dilute "precipitation-like" water contributing to stream flow but these could derive from a number of hydrologic processes including incident precipitation, surface runoff or rapid response subsurface flow (Ogunkoya and Jenkins 1993). In this paper, soilwaters themselves were shown to be mixtures differing seasonally and in response to storm events thus violating a requirement that samples represented independent estimates of soilwater in equilibrium with potential end-member soil layers. The working solution was to track seasonal trends in combined chemical variables (U-space) from individual wells and to choose the extremes. The approach differes from the method recommended by Christophersen and Hooper (1992) of choosing a value close to the boundaries of u-space but is justified by the seasonal pattern. This is an improvement over the arbitrary selection of points to encapsulate all the stream data. With this physical basis for seasonality, the approach integrates rates for chemical equilibria with potential mixing between soil layers taking the results further from the physics of water flow and relying instead on the implications for water chemistry. Thus, larger volumes of water that were not in chemical equilibrium with mineral soils could move in June as throughflow than predicted by EMMA. Similar problems were noted for geochemical tracing on the Canadian Shield and were attributed to preferential flow paths that can deliver event-water rapidly through deeper soil pathways (Buttle and Peters 1997). EMMA can therefore underestimate the importance of soil layers in contributing runoff, particularly during rain events. The purpose of applying EMMA in this study was to assign changes in stream water chemistry following emulation silviculture to the combination of soil disturbance, changes in flow pathways and geochemistry which EMMA will accomplish albeit simplified into end-members.

The natural isotopes of water can constrain some of the uncertainty in flow pathways that represent a potential error in EMMA by distinguishing between event and pre-event water (e.g., Hill and Waddington 1993). The isotope composition of all six stream waters were relatively insensitive to snowmelt and precipitation events indicating that most water passed through soil pathways where mixing could occur. EMMA provided insight as to the likely soil sources for stream discharge but these data require the additional information provided by deuterium and ^{18}O, which in these watersheds, suggested soils provided mixing zones, "precipitation-like" water derived from ombotrophic peat and contributions from direct precipitation were limited.

The number of components that can be represented in mixture modeling is another potential limitation for both EMMA and isotopes. EMMA techniques are typically limited to three end-members because solving the set of linear equations becomes increasingly complex and can result in over-determination (Christophersen et al. 1990). Monte-Carlo procedures have been proposed to accurately determine mixing from unlimited end-members (Neal 1991); however, such complexity likely overestimates the certainty with which the end-members themselves can be separated from each other. Three end-members have been applied to deuterium and ^{18}O analyses (e.g. Saito 2000). The ability to distinguish soilwater sources from deuterium and ^{18}O is marginal because the enrichment signature is set at infiltration and is not a result of the soil properties. The general limitations of data and assumptions for hydrologic inference from isotope analyses have been presented in detail for northern Alberta sites (Gibson et al. 2002). When considering hydrologic process relating to streams, isotope analyses are generally based on two component mixing-models (e.g., Hill and Waddington 1993, Buttle and Peters 1997) and, as applied in this paper, are only useful in determining contributions from precipitation sources that differ in isotopic enrichment.

The results from this study indicate two hydrologic patterns for northern boreal catchments on glacial till: a sloped topography where snowmelt is rapidly discharged as near surface runoff and groundwater; and a lowland topography where snowmelt is held in peatlands and other wetlands including beaver ponds. In the former, groundwater from mineral and organic soils contribute equal proportions to runoff throughout summer months. In the latter, discharge from fen/bog complexes dominate runoff. In sloped catchments, isotope enrichment suggested summer precipitation dominated runoff and EMMA indicated that the predominant pathway was through shallow soils. In lowland catchments, piston flow appeared to be a dominant process as discharge derived largely from deeper groundwater. In lowlands, the timing of response to rainfall was determined by the first rain following complete snowmelt. In sloped catchments the timing of response to rainfall was determined by both the deepening of soil frost, increase in soil storage capacity and the occurrence of large rain events to fill available storage. Emulation silviculture may have limited impacts in lowland drainages as long as fen and bog complexes are not disturbed. Avoiding fens and bogs can be operationally difficult during winter harvesting as they may be small, interconnected and require crossing procedures that are currently not part of forestry practices in northern Alberta. In sloped drainages, emulation silviculture is expected to have greater impacts because shallow soil pathways predominate in runoff generation particularly with the removal of riparian trees where return flow is an important source of stream discharge. Water yield also dictates potential impacts from emulation silviculture. Water yield is expected to increase following timber removal (Buttle and Metcalf 2000). This study indicates that lowland sites are sensitive to water yield and, despite being less sensitive to soil disturbance, are likely more sensitive to disturbance during wet years than mountain sites where water yield was comparatively low. Investigation of the interplay between soilwater pathways, dominant sources for stream discharge and the potential impacts from emulation silviculture are on-going in these watersheds.

Table 5-1: Physical parameters for the six study catchments. MH and MR were in the well-drained uplands of the Caribou Mountains. Those with prefix L were in the poorly drained setting of the Cameron Lowlands.

StreamSite	DA (ha)	Channel Slope (%)	Fen (%)	Riparian Bog (%)	Other Bog (%)
MH	208	7	1.1	2.5	5
MR	142	9	2.3	3.6	3
LH1	280	0.6	4.2	2.5	35
LH2	308	0.7	6.4	4.3	58
LH3	540	0.6	6.9	5.5	61
LR	267	0.6	15.7	7.2	66

DA is drainage basin area determined from totpgraphic maps and ground truthing. Vegetation cover was calculated as percent of total drainage basin area. Remaining vegetation area was forested, primarily by white and black spruce with moss organic layers more than 10 cm thick.

Table 5-2: Precipitation data (mm) for study areas. Annual precipitation is from October to October. Snow water equivalent is estimated from spring sampling of snow pack in representative vegetation, slope and aspects conditions for each watershed.

Catchment Type and Year	Total Precipitation (mm)	Total summer precipitation[a] (mm)	Snow water equivalent[b] (mm)
Mountain			
1998	(474)	330 (385)	NA (144)
1999	493 (505)	446 (457)	47 (55)
2000	365 (350)	298 (279)	66 (71) & 30[c]
2001	301	134	67 (66)
30-y mean	381	306	75
Lowland			
1998	(300)	220 (223)	NA (77)
1999	407 (400)	283 (296)	82 (104)
2000	276 (279)	217 (163)	59 (116) & 35[c]
2001	258	168	90 (86)
30-y mean	402	261	141

a. Mid-May to mid-Sept. precipitation from rain gauge. Numbers in parentheses are April to October estimates of precipitation from Sacramento gauges within 100 m of the rain gauges.
b. Snowpack from survey data during early to mid-March. Numbers in parentheses are November to April estimates of precipitation from the Sacramento gauges.
c. Snow water equivalent estimates for harvested basins.

Table 5-3: Discharge characteristics of the study streams. Snowmelt was captured in 1999 only.

Site	Snow melt period	Snowmelt Discharge (m^3 x 10^4)	Snowmelt Runoff Coeficient	Total Stream Discharge (m^3 x 10^4)		Total Runoff Coefficient	
Mountain Sites							
Year	1999	1999	1999	1998	1999	1998	1999
MH	5/14 -6/4	28.98	1.46	11.2	57.7	0.16*	0.56
MR	5/14 -6/2	16.6	1.3	12.2	32.6	0.26*	0.47
Lowland Sites							
LH1	5/1- 6/6	2.4	0.06	0.2	9.2	0.01*	0.1
LH2	5/1- 6/6	19.7	0.4	3.3	47.3	0.05*	0.48
LH3	5/1- 6/6	4.4	0.05	1.2	8.4	0.11*	0.06
LR	5/1- 6/6	1.8	0.04	1.4	7.9	0.03*	0.09

Annual period for calculating total discharges were: 1998 mountain = 6/23/98 to 9/26/98 and 1998 lowland = 7/25/98 to 8/26/98, 1999 mountain = 5/14/99 to 10/7/99 and 1999 lowland = 4/28/99 to 8/23/99.
* Total runoff coefficients for 1998 are based on total stream discharge from June 23 (mountain) or June 25. They do not include snow, rain or discharge prior to project start on these two dates.
Total catchment area (Table 5-1) was used in runoff coefficient calculations.

Fig. 5-1: Location of study catchments, stream sampling sites and nests of ground water wells (1 = peat/fen wells, 2 = groundwater). Dotted blue lines in mountain sites are springs.

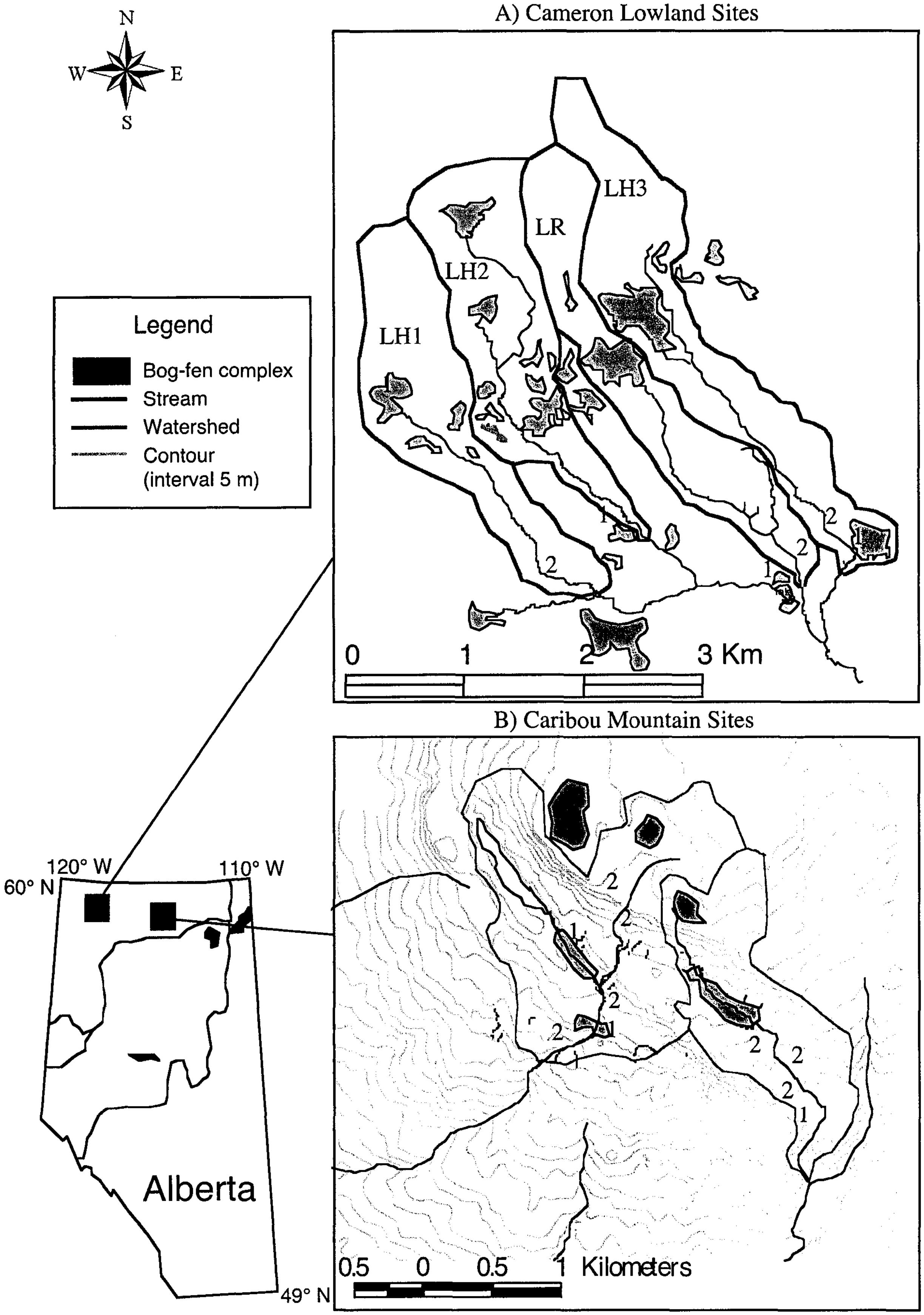


Fig. 5-2: 1999 hyetograph (upper pannel), hydrograph (bold line) and sources of water from EMMA for streams in mountain catchments MH (a) and MR (b). Squares on each hydrograph are discharge measurements used to calibrate the depth-discharge relationship. Light grey is groundwater, dark grey is peat water, and white is precipitation.

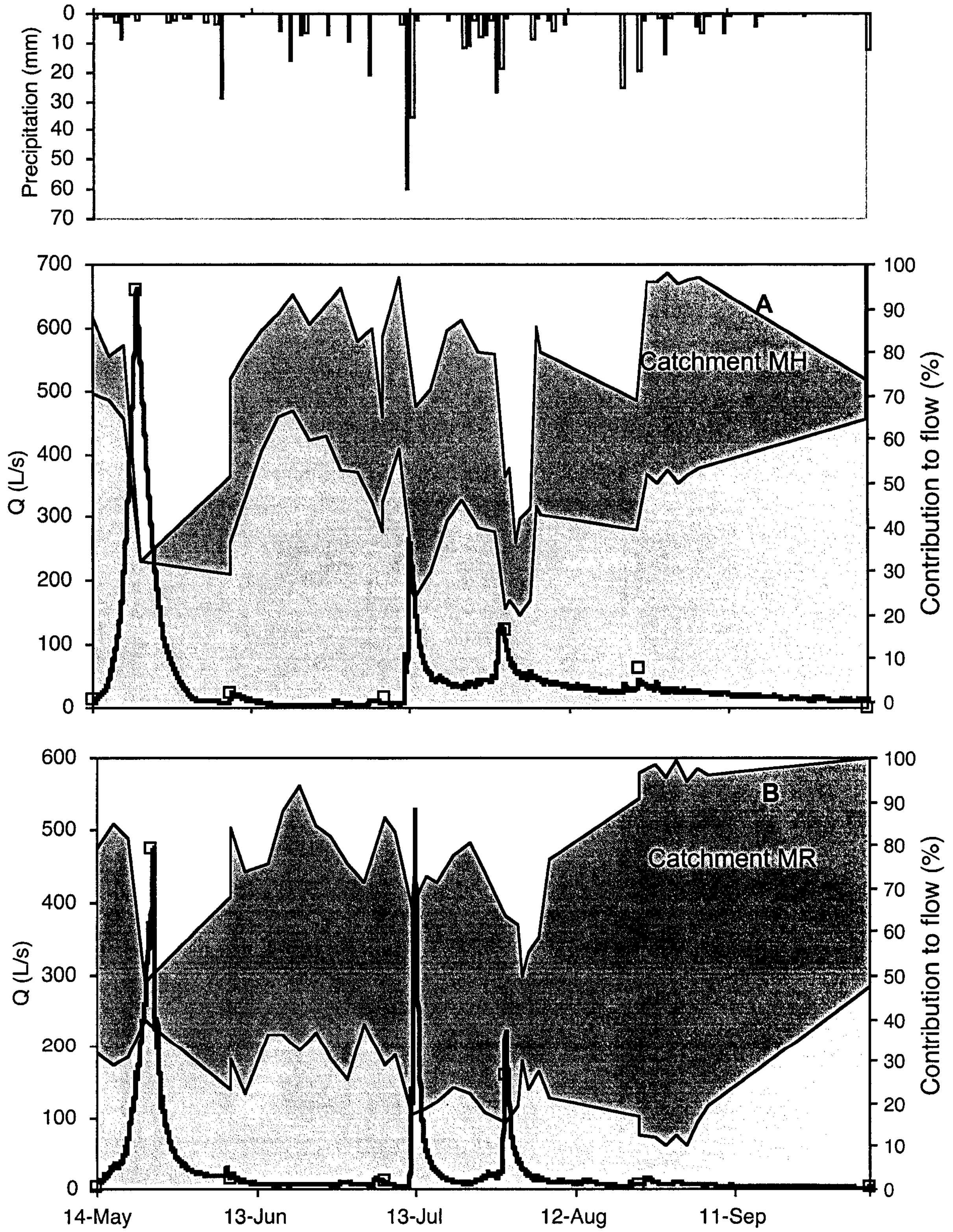


Fig. 5-3: 1999 hyetograph (upper pannel), hydrograph (bold line) and sources of water from EMMA for streams in Lowland catchments LH1 (a) and LH2 (b). Squares on each hydrograph are discharge measurements used to calibrate the depth-discharge relationship. Light grey is groundwater, dark grey is peat water, and white is precipitation. Chemistry during initial snowmelt was unavailable.

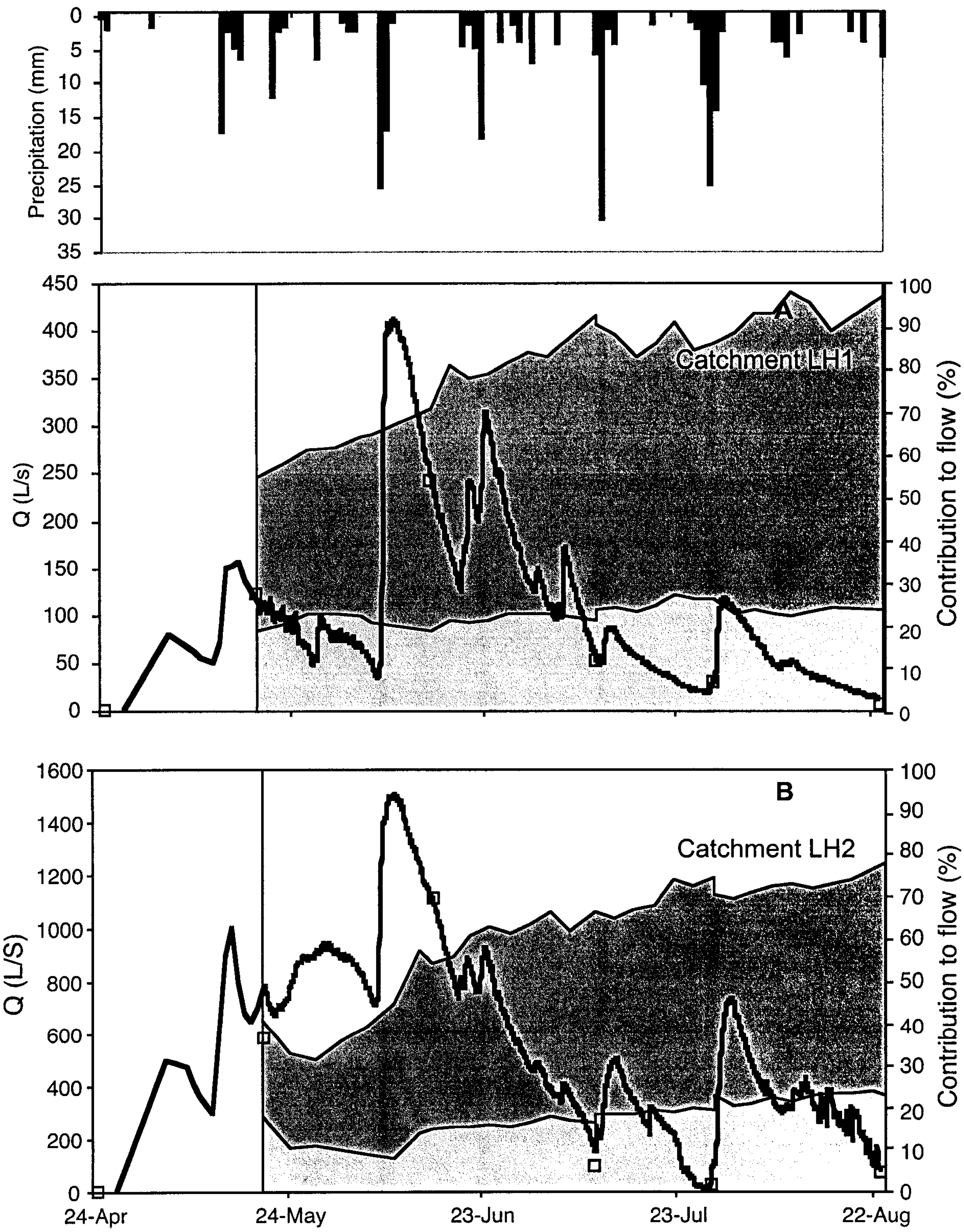


Fig. 5-3 continued: 1999 hyetograph (upper panel), hydrograph (bold line) and sources of water from EMMA for streams in lowland catchments LH3 (a) and LR (b). Squares on each hydrograph are discharge measurements used to calibrate the depth-discharge relationship. Light grey is groundwater, dark grey is peat water, and white is precipitation. Chemistry during initial snowmelt was unavailable.

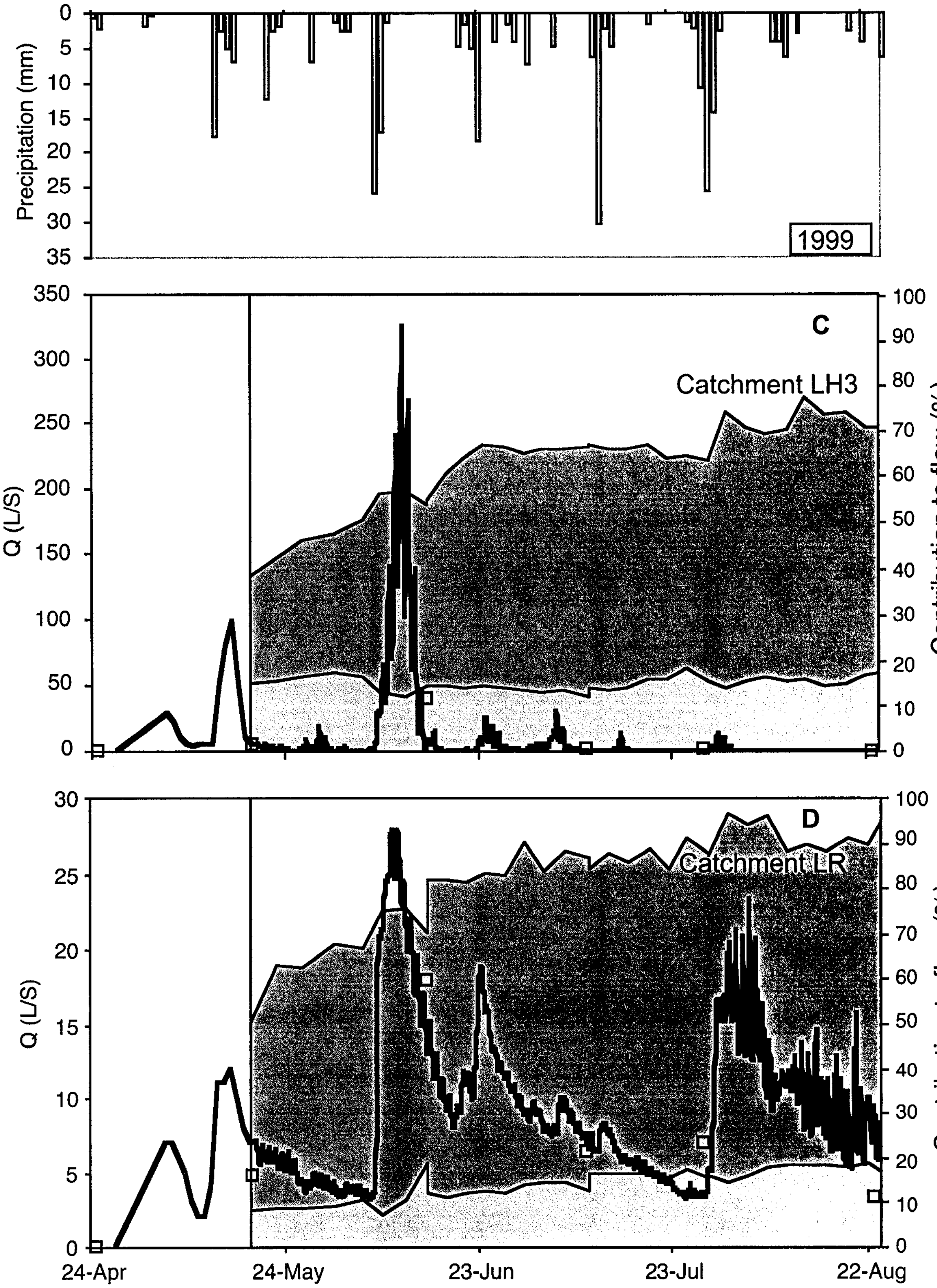


Fig. 5-4: Mountain site mean ice and water depths in a) three wells in MH catchment uplands, b) two wells in MR catchment uplands, c) three wells in MH riparian soils, d) three wells in catchment MR riparian soils, e) three wells in MH riparian bog, f) three wells in MR riparian bog. Standard error bars of the mean are shown for each point. Data from wells that did not intersect the watertable throughout the year were not included.

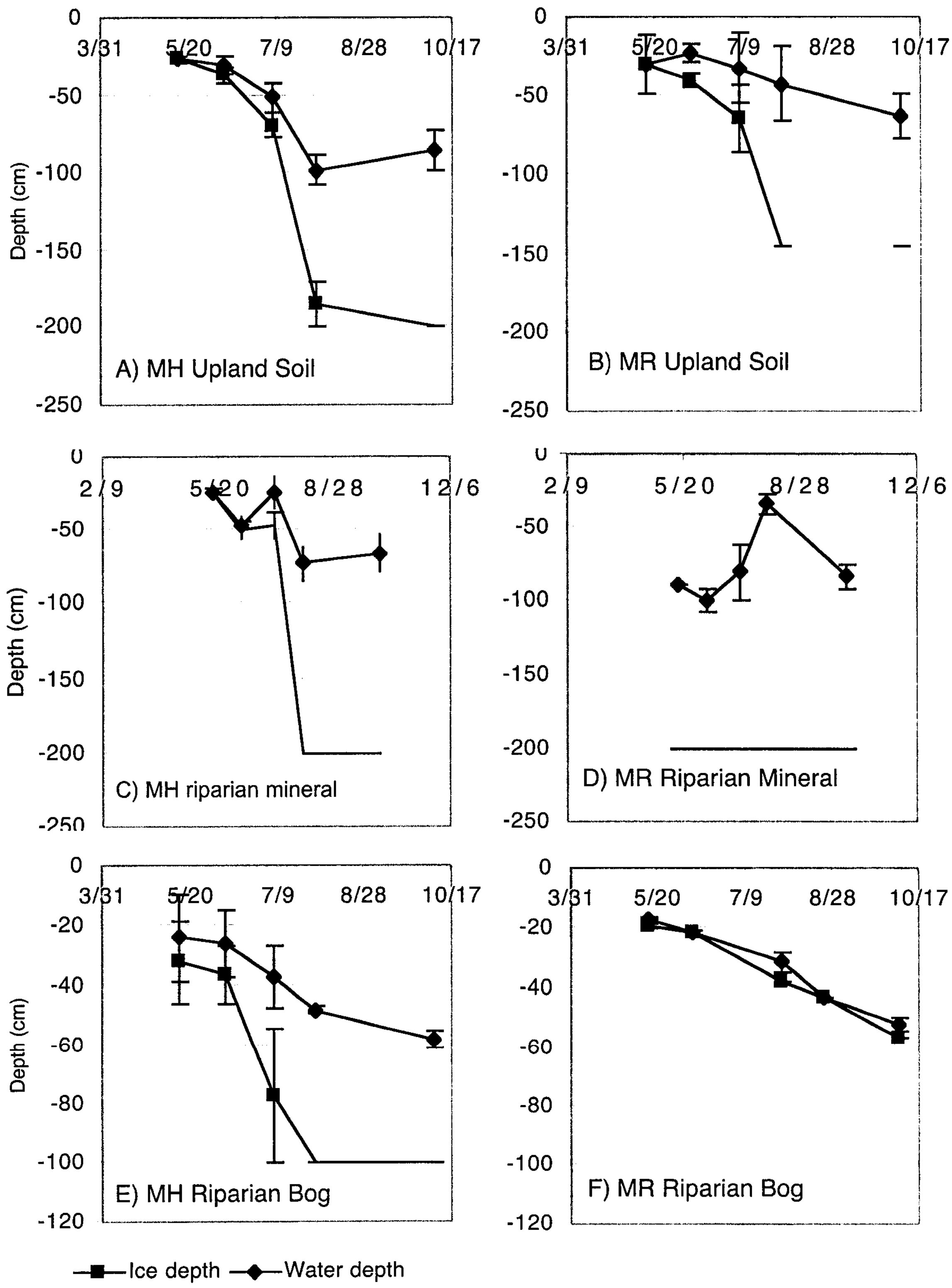


Fig. 5-5: Lowland mean ice and water depths in a) six wells in forest soils, b) three wells in the open bogs/fens at headwaters. There are no standard error bars on the ice depth of bog/fen wells because only one contained ice. The other two bog/fen wells did not contain ice during the sampling period.

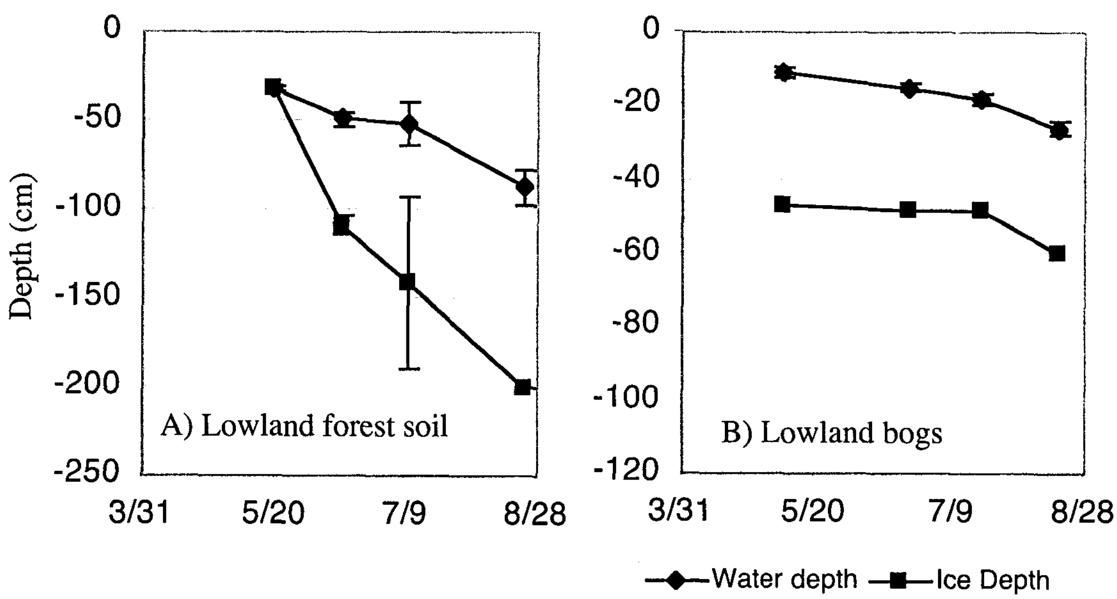


Fig. 5-6: Principle component analysis projection (oblique rotation) of end member chemistry for parameters used in EMMA (Ca, Mg, Na, K, SO4, Alkalinity, DOC). Open symbols are lowland sites, closed symbols are mountain sites. Triangles are groundwater, squares are peatlands, circles are fens, diamonds are rain, inverted triangles are snow. Location of endmember chemistry for EMMA analysis are approximated for lowland peat and groundwater (PL, GL) and mountain peat and groundwater (PM, GM). The precipitation end member for both lowland and mountain sites is centered in the dark cluster and is not identified with text.. All data are from 1999 except precipitation which includes both 1998 and 1999 data.

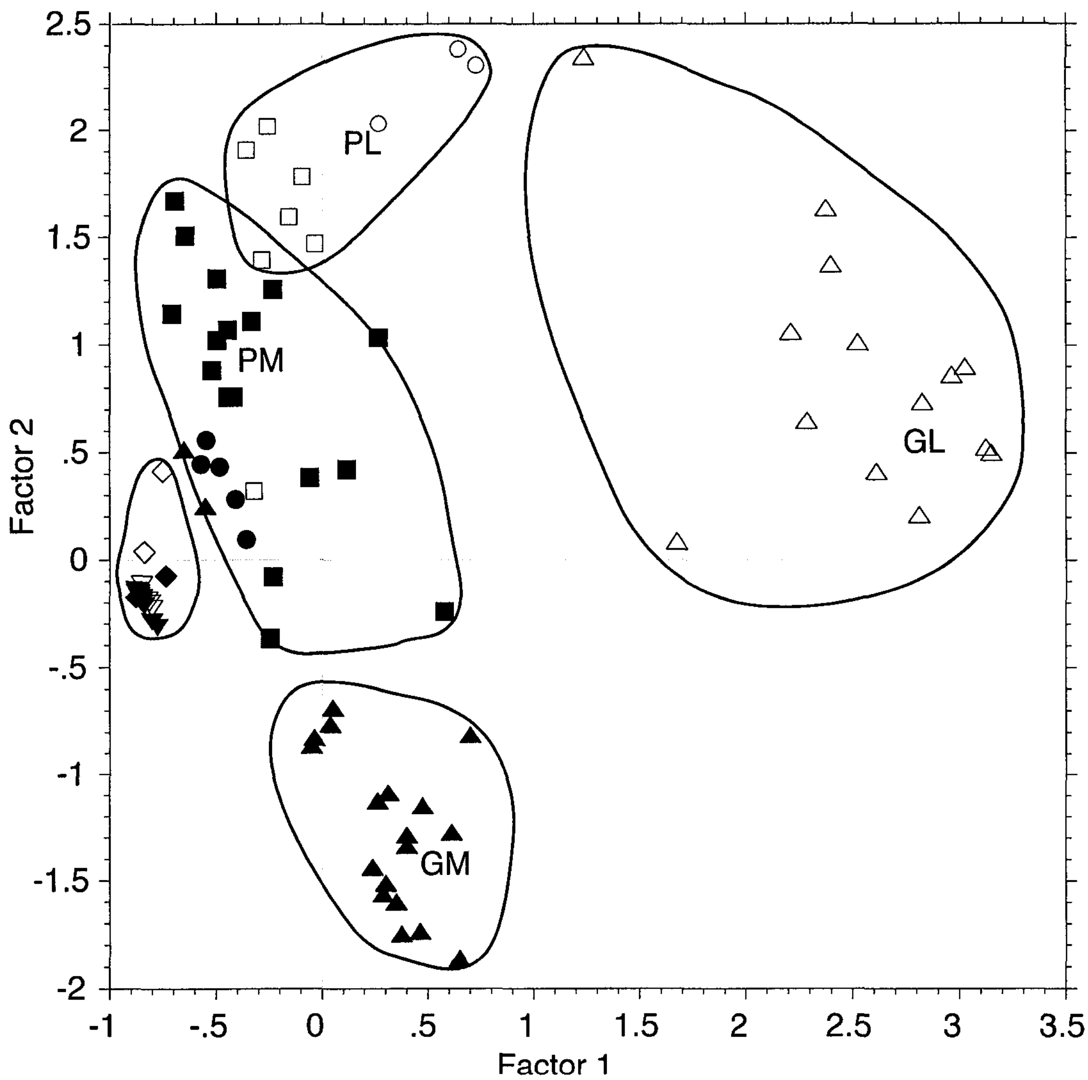


Fig. 5-7: Example end-member mixing analysis diagram for lowlands. Data are identified as stream samples (diamonds) and end-members precipitation (triangles), organic soil water (squares) and groundwater (circles). Data are projected into two factor u-space. End-member data represent combined monthly averages except precipitation which are summer or snow composites. Groundwater is separated into wells at valley bottoms (open circles) and wells in upland positions (closed circles). The arrows linking groundwater points from a valley bottom (riparian) well track changes from May through August. Some upland wells also consistently moved from least negative to most negative values from May through August. The bold triangle represents the separation matrix used to calculate source water proportions for stream samples.

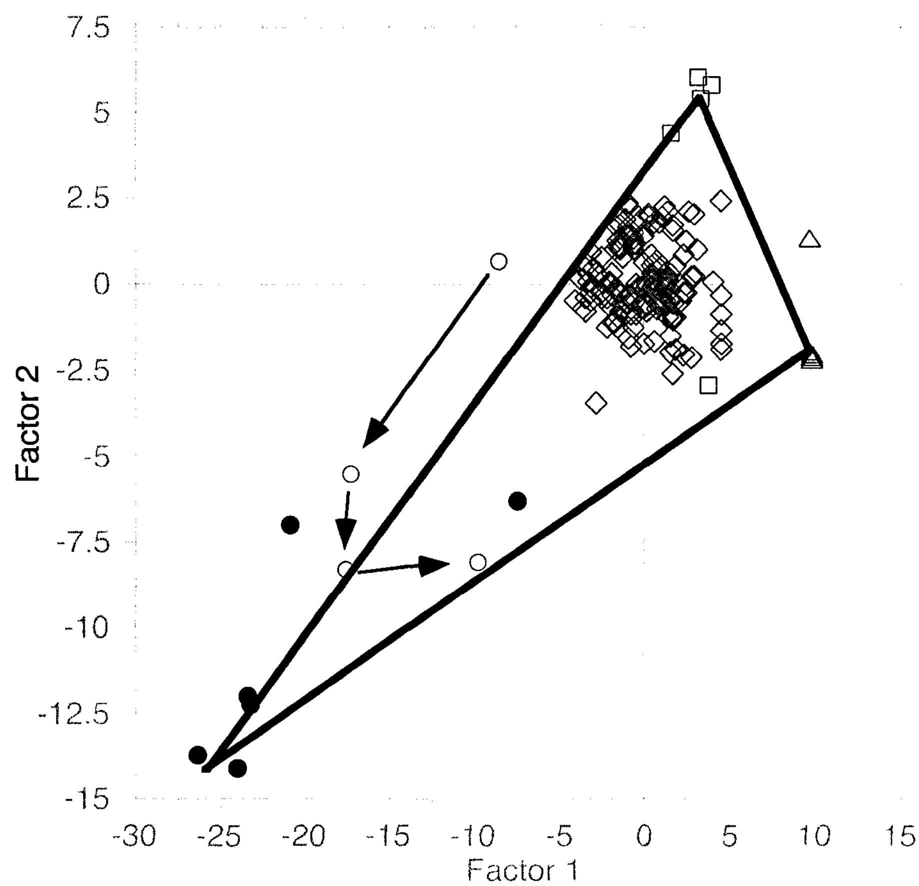
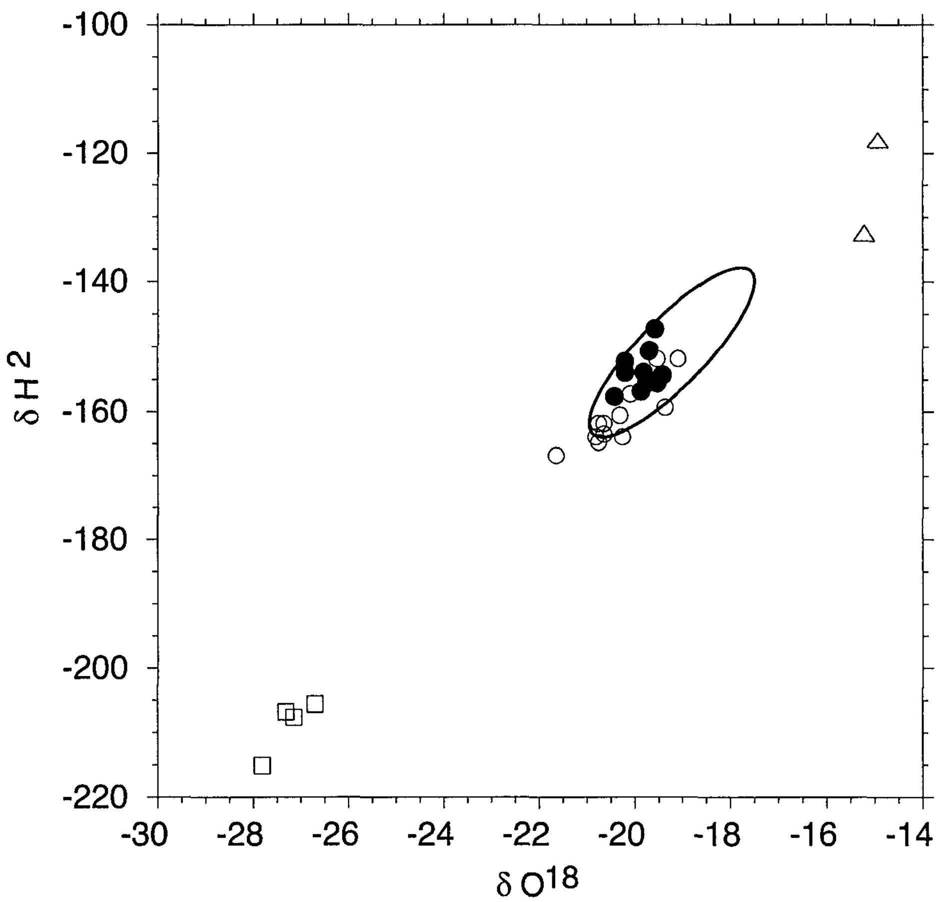


Fig. 5-8: Example deuterium and oxygen-18 signatures for Lowland sites in 1998 and 1999 as they were incorporated in the isotope mixing analysis. Squares are spring snow samples (1999 only), open circles are groundwater, closed circles are peat soil water and triangles are composite precipitation samples for 1998 and 1999. The oval indicates the boundaries for all Lowland stream samples in 1998 and 1999.



References

Branfireun, B. A., and Roulet, N. T. 1998. The baseflow and storm flow hydrology of a Precambrian Shield headwater peatland. *Hydrol. Process.*, 12(1): 57-72.

Buttle, J. M. and D. L. Peters (1997). Inferring hydrological processes in temperate basin using isotopic and geochemical hydrograph separation: a re-evaluation. *Hydrol. Process.* 11: 557-573.

Caissie, D., Pollock, T. L., and Cunjak, R. A. 1996. Variation in stream water chemistry and hydrograph separation in a small drainage basin. *J. Hydrol.*, 178: 137-157.

Christophersen, N., and Hooper, R. P. 1992. Multivariate Analysis of Stream Water Chemical Data: The Use of Principal Components Analysis for the End-Member Mixing Problem. *Water Resour. Res.*, 28(1): 99-107.

Cirmo, C. M., J. J. McDonnell 1997. Linking the hydrologic and biochemical controls of nitrogen transport in near-stream zones of temperate-forested catchments: A review. *J. Hydrol.*, 199 (1-2): 88-120.

Devito, K.J., I.F. Creed, R.L Rothwell, and E.E. Prepas 2000. Landscape controls on phosphorus loading to boreal lakes: Implications for the potential impacts of forest harvesting. *Can. J. Fish. Aquat. Sci.* 57: 1977-1984.

Dunne, T., and Black, R. D. 1971. Runoff processes during snowmelt. *Water Resour. Res.*, 7: 1160-72.

Evans, J.E., E.E. Prepas, K.J. Devito, and B.G. Kotak 2000. Phosphorus dynamics in shallow subsurface waters in an uncut and cut subcatchment of a lake on the Boreal Plain. *Can. J Fish. Aquat. Sci.* 57(Supp/2): 60-72.

Evans, M. G., Burt, T. P., Holden, J., and Adamson, J. K. 1999. Runoff generation and water table fluctuations in blanket peat: evidence from UK data spanning the dry summer of 1995. *J. Hydrol.*, 221(3-4): 141-160.

Gat, J. R., and Gonfiantini, R. 1981. "Stable isotope hydrology : deuterium and oxygen-18 in the water cycle." *IAEA technical reports series ; no. 210*, Vienna.

Gibson, J. J., Prepas, E. E., and McEachern, P. 2002. Quantitative comparison of lake throughflow, residency and catchment runoff using stable isotopes: Modeling and results from a regional survey of Boreal lakes. *J. Hydrol.* 262: 128-144.

Hill, A. R., and Waddington, J. M. 1993. Analysis of storm run-off sources using oxygen 18 in a headwater swamp. *Hydrol. Process.*, 7: 305-316.

Hill, A. R. 1993. Base cation chemistry of storm runoff in a Forested Headwater Wetland. *Water Resour. Res.*, 29: 2663-2673.

Holecek, G. 1988. Storage-Effective Drainage Runoff Model. *J. Hydrol.*, 98(3): 295-314.

Hooper, R. P. and N. Christophersen 1992. Predicting Episodic Stream Acidification in the Southeastern United States: Combining a Long-Term Acidification Model and the End-Member Mixing Concept. *Water Resour. Res.* 28: 1983-1990.

Ingram, H. A. P. 1983. "Hydrology." Ecosystems of the world 4A. Mires: Swamp, bog, fen and moor., A. J. Gore, ed., Elsevier, Oxford, 67-158.

McRae, D. J., Duchesne, L. C., Freedman, B., Lynham, T. J., and Woodley, S. (2001).

Comparisons between wildfire and forest harvesting and their implications in forest management. *Environ. Rev.* 9: 223-260.

Metcalfe, R.A. and J.M. Buttle 2001. Soil partitioning and surface store controls on spring runoff from a boreal forest peatland basin in north-central Manitoba, Canada. *Hydrol. Process.* 15: 2305-2324.

Muller, D.I., I.C. Wohlfeil, N. Christophersen, M. Hauhs, H.M. Seip. (1993). Chemical reactiveness of soil water pathways investigated by point source injections of chloride in a peat bog at Birkenes. *J. Hydrol.* 144: 101-125.

Neal, C. and N. Christophersen 1990. Linking hydrological, geochemical, and soil chemical processes on the catchment scale: An interplay between modeling and field work. *Water Resour. Res.* 26: 3077-3086.

Neal, R. M. 1991. Bayesian mixture modeling by Monte Carlo simulation. University of Toronto Technical Report CRG-TR-91-2.

Ogunkoya, O. O. and A. Jenkins 1993. Analysis of storm hydrograph and flow pathways using a three-component hydrograph separation model. *J. Hydrol.* 142: 71-88.

Pawluk, S. 1961. Mineralogical composition of some grey wooded soils from Glacial till. *Can. J. Soil Sci.* 41: 228-240.

Peters, N. E., N. Christophersen and R. P. Hooper 1990. Modelling Streamwater Chemistry as a Mixture of Soilwater End-Members-An Application to the Panola Mountain Catchment, Georgia, U. S. A. *J. Hydrol.* 116: 321-343.

Pietroniro, A., Prowse, T., Hamlin, L., Kouwen, N., and Soulis, E. 1996. Application of a grouped response unit hydrological model to a northern wetland region. *Hydrol. Process.*, 14: 1245-1261.

Prowse, T. D. (1990). Northern Hydrology: An overview. Northern Hydrology: Canadian Perspectives. T. D. Prowse and C. S. L. Ommanney. Saskatoon, National Hydrology Research Institute. NHRI Sciene Report No. 1: 1-36.

Puurveen, H., R. C. Izaurralde, D. S. Chanasyk, J. R. Williams and R. F. Grant 1997. Evaluation of EPIC's snowmelt and water erosion submodels using data from the Peace River region of Alberta. *Can. J. Soil Sci.* 77(1): 41-50.

Schwartz, F. W., and Milne-Home, W. A. 1982a. Watersheds in Muskeg Terrain. 1. The Chemistry of Water Systems. *J. Hydrol.*, 57(3-4): 267-290.

Schwartz, F. W., and Milne-Home, W. A. 1982b. Watersheds in Muskeg Terrain. 2. Evaluations Based on Water Chemistry. *J. Hydrol.*, 57(3): 291-305.

Saito, T. (2000). Runoff characteristics in a small mountain basin analyzed by the use of hydrogen and oxygen stable isotopes. *Limnology*, 1: 217-224.

Vitt, D. H., and Chee, W.-L. 1990. The relationship of vegetation to surface water chemistry and peat chemistry in fens of Alberta, Canada. *Vertatio*, 89: 87-106.

Woo, M.-K. and T.C. Winter 1993. The role of permafrost and seasonal frost in the hydrology of northern wetlands in North America. *J. Hydrol.*, 141: 5- 31.

Chapter 6:
Impacts of clear cut forest harvesting on hydrology, element export and stream chemistry in catchments of northern Alberta.

Introduction

The boreal forest covers over 15 million km^2 of the northern hemisphere with an incredible diversity of habitats and hydrologic conditions (Gorham 1991). The coniferous and mixed wood trees that thrive in the boreal forest have evolved in balance with the hydrologic and biogeochemical changes that follow periodic large-scale disruptions from wildfire or parasites (Weber and Taylor 1992). In an attempt to approximate spatial patterns of wildfire, emulation silviculture is becoming increasingly popular (McRae et al. 2001). Under this new paradigm, trees are harvested over large contiguous areas that include riparian stands adjacent to waterbodies. Riparian features such as peatlands are not specifically protected from equipment traffic under emulation silviculture. These features can control solute flux to surface waters and can be sensitive to disturbance (Vitt et al. 1994; Prepas et al. 2001).

Despite the importance of boreal forest resources, the hydrologic linkages and potential impacts of disturbance on its aquatic systems are not well understood, particularly in northern Candada (Metcalfe and Buttle 2001). Small changes in hydrologic processes following disturbance can cause large changes in the flux of nutrients to surface waters, especially where nutrient-rich glacial soils occur (Evans et al. 2000; McEachern et al. 2000). The impacts of timber harvesting may be large when riparian buffers are removed, however, complex hydrology can confound the interpretation of catchment-scale studies on disturbance (Devito et al. 2000; Buttle et al. 2001). A method for determining the potential sensitivity of boreal forest catchments to disturbance from hydrologic and vegetation or soil characteristics is required so that water quality can be managed in large-scale resource developments.

In northern Europe, the investigation of disturbance impacts on aquatic systems has successfully linked soil characteristics and water chemistry through End Member Mixing Analysis or EMMA (Neal and Christophersen 1990; Muller et al. 1993). EMMA provides inference into source waters for streams from catchment soils addressing some of the uncertainties in hydrology that confound interpretation of catchment-scale studies. EMMA may be a useful supplement for understanding aquatic chemistry in regions with complex hydrology such as the boreal forest of western Canada. In addition to understanding hydrologic conditions, tools are needed that can detect ecosystem changes after disturbance in the absence of experimental replication. Randomized intervention analysis (RIA) offers a robust technique that tests the likelihood that observed changes following an intervention are a realization of possible random outcomes (Carpenter et al. 1989). The potential power of combining EMMA and RIA as tools for assessing emulation silviculture have as yet not been realized in northern Canada.

This study extends the hydrologic investigation of six boreal forest catchments in northern Alberta (McEachern et al. Submitted) to an assessment of hydrologic and stream chemistry changes following clear cut harvesting. In the two years the basins were studied prior to harvesting, results from EMMA indicated sources for runoff differed markedly between relatively well drained mountain basins and poorly drained lowland basins. Based on the pre-harvest results, we hypothesized that the already rapid discharge of meltwater from mountain catchments would increase after harvesting. In addition, peak discharge would be elevated during snowmelt and summer storms. We expected impacts from harvesting on hydrologic processes would be limited to the surface through the vadose zone with few impacts on deeper hydrologic processes. Total water yield would increase, particularly during melt and storm events due to reduced evapotranspiration and reduced infiltration but impacts on baseflow compared to the pre-harvest condition would be minimal. Total elemental flux from mountain basins was expected to remain unchanged because a switch to rapid discharge and dilute surface runoff would counter enhanced water yield. On the contrary, in lowland basins, watertables were expected to rise with runoff increasingly dominated by water from organic soils and peatlands. Discharge from lowland streams was expected to become more dilute in nitrogen and ions derived from mineral soils, however, the concentration of elements derived from organic soils, such as organic carbon, would increase. The total flux of all elements was expected to increase from lowlands because of increased discharge routed through soil pathways.

Methods

Physical setting

Two streams draining the Caribou Mountains and four streams draining the Cameron lowlands in northern Alberta, Canada were selected for this study (Fig. 6-1). The mountain streams lay in well defined basins (watersheds MH and MR) dominated by spruce with a moss, lichen and shrub understory typical for Labrador tea-subhygric ecosites (Ecosites of Northern Alberta, Natural Resources Canada). The lowland streams lay in poorly defined basins (watersheds LH1, LH2, LH3 and LR) with low topographic relief and large networks of interconnected wetlands typical for treed bog and fen ecosites. Average May through October and November through April temperatures were 7° C and -14° C, respectively. Mean daily temperatures drop below zero in mid-October and return to positive values in mid-April. The warmest month is July at 14° C and January is the coldest month at -21° C. Surface soils in both areas are Orthic Gray Luvisols which have roughly equal proportions of illite and montmorillonite clays comprising 40% of the total soil profile (Pawluk 1961). Clay and silt combined average 70-80%. Total exchangeable cations of surface soils were estimated in both regions at 20 - 40 meq•100 g^{-1} in the A and 24 meq•100 g^{-1} in B horizons; total soluble salts are typically < 10 meq•100 g^{-1} (ARC 1962). Organic soils are typically 10 to 20 cm deep at well sites and composed primarily of *Sphagnum* spp., and a variety of feather and brown mosses. In bog and fen sites, organic soils extended down to 1 m and contained seasonal frost lenses that thawed through the summer depending on exposure. Geologic formations in the mountain catchments include fine grained feldspathic sandstone with calcareous beds and laminated siltstone of deltaic and marine origin (Dunvegan formation) over marine shale, silt and sand (Shaftesbury formation). The lowland basins are underlain by the Shaftesbury formation with fosiliferous siltstone and thin beds of concretionary iron-stone. Selenitic gypsum crystals are ubiquitous in road fill dugouts in the lowland area. The physical parameters for the study basins such as mean slope and percent wetland cover are previously summarized in McEachern and others (submitted). Streams in the two regions differ in channel slope and percentage of wetlands comprising their drainage basins (Table 6-1).

Each stream was divided into 2 (lowland) or 3 (mountain) reaches. At each site, stream discharge was estimated from area and current measurements using a Swoffler current meter at a minimum of five times through the summer. Stream depth was monitored at 15-min. intervals and discharge was calculated using stage-discharge curves independently derived for each stream. Storm discharge was collected at hourly intervals in 1998 with ISCO samplers set to start a 24-h cycle with a 1 cm rise in stage. In 1999, ISCO samplers were set to collect samples every 72 h throughout the ice-free period. Precipitation was monitored twice daily at fire towers proximate to the study watersheds. Precipitation samples were frozen within six hours and later combined for chemical analysis.

To collect groundwater from mineral and organic horizons, a total of 19 and 11 wells were installed in mountain and lowland basins, respectively. Groundwater wells for mineral soil were installed to gravel/sand lenses between layers of clay at 1.5 to 2.5 m depth. The wells collected water through horizontal slits over the bottom 30 cm of well length. Wells for organic soils were placed in bog and fen waters to the depth of perma-frost or mineral soil and collected water through vertical slits from top to bottom. All wells were placed inside polypropylene fabric to limit sediment accumulation in the wells. In two perched bogs and two fens, wells were placed along assumed flow paths to streams with a well at the uphill margin, the center, and near the base proximate to the stream. Additional wells, in LH1 for example, were lost during harvesting activities.

Sample processing and chemical methods

Processing of stream and soil water samples began within four hours of collection. Triplicate 25-mL aliquots were pipetted into 30-mL glass tubes used directly in the digestion process and nutrient analysis. Unfiltered water was used for total nitrogen (TN) and total phosphorus (TP) analyses. Filtrate (Millipore HA, 45 μm) was used for dissolved nitrogen (DN), dissolved phosphorus (DP), soluble reactive phosphorus (SRP), nitrate-nitrite (NO_3^--N) and ammonium analyses (NH_4^+-N). TN and DN samples were preserved with 10 μL of 40% H2SO4 for storage (1-2 wk), then neutralized with equivalent NaOH prior to digestion in the laboratory. TP and DP were analyzed spectroscopically (5-cm cell) from persulfate-oxidized samples by molybdate blue absorption (Prepas & Rigler 1982). TN and DN concentrations were determined by second derivative spectroscopic analysis of persulfate oxidized samples (Crumpton 1992). Ammonium, nitrate and SRP samples were frozen in 125-mL polyethylene bottles for later analysis by indophenol blue (NH_4^+-N), cadmium reduction (NO_3^--N) with automated colorimetry (Technicon methods 100-700 W/B and 155-, respectively) and by molybdate blue absorption (SRP). Total Suspended Solids (TSS), Nonvolatile suspended solids (NVSS), and volatile suspended solids (VSS) were collected on pre-ashed GF/F filters (mean particle retention 0.7 μm) and analyzed in duplicate after APHA (1993). Alkalinity was determined by the Hach phenolpthaline/ bromocresol method (APHA 1993). Filtrate from GF/F filters for cations (Ca^{2+}, Mg^{2+}, Na^+, K^+), dissolved organic and inorganic carbon (DOC/DIC), and anions (Cl^-, SO_4^{2-}) analysis were stored in 60-mL polyethylene bottles. Cation and DOC samples were preserved with 40% H_2SO_4 ; anion samples were refrigerated.

Cations were analyzed by atomic absorption flame spectroscopy (Perkin-Elmer AS90, AA3300), DOC by high temperature catalysis (Shimadzu TOC-5000), and SO_4^{2-} and Cl^- concentrations by chromatography (Dionex 2000i/SP).

End-Member Mixing Analysis (EMMA) methods

EMMA followed published methods (Christophersen and Hooper 1992) summarized below. Stream water concentrations for Ca^{2+}, Mg^{2+}, Na^+, K^+, SO_4^{2-}, alkalinity (HCO_3^-), and DOC were normalized by subtracting the mean and dividing by the standard deviation for each parameter. A correlation table for the normalized values was then input into Principle Components Analysis (Statview 4.1) and resulting eigenvectors were used to project each stream sample into U-space. Samples from groundwater wells, peatland sites, and precipitation were normalized and similarly projected into U-space. The recommended procedure of selecting median values for potential end-members was appropriate for precipitation in both mountain and lowland and for peat waters in lowland basins. Peat water in mountain and groundwater in both mountain and lowland basins demonstrated seasonal hysteresis with individual samples falling along lines connecting maximum values for groundwater, peat water, and precipitation. Changes in groundwater chemistry corresponded to loss of frost in wells, suggesting that groundwater samples prior to July were themselves mixtures that obtained a greater groundwater and lesser peat water signature as frost melted. Hence mean July and August ion concentrations were used to represent groundwater end-members rather than the median for all groundwater samples. The peat water end-member in mountain basins was similarly determined, however, peat water values extended along a line connecting maximum (August) values with precipitation. The proportion of end-member contribution to each stream samples was determined by solving the set of linear equations (Christophersen and Hooper 1992).

Harvesting Treatments

The selection of catchments to be harvested and the magnitude of harvest were determined by the distribution of merchantable trees and the logistics of harvesting them. Although all merchantable trees within treatment basins were to be harvested, some patches were not accessible due to equipment limitations (e.g. slopes over 50°) and a limit of one stream crossing (snow and ice bridge) within each basin. Lowland basins were harvested with fellers and whole trees were skidded to roadside for loading. In the mountain basin, fellers cut-to-length at the stump and forwarders transported cut sections to loading sites. The former harvesting method is the most common in Alberta. The latter is more expensive but is perceived to have a lower impact because equipment traffic is reduced and because unmerchantable trees are undisturbed. Forwarder and skidder traffic occurred to within 15 m of each stream. Within 15 m of the stream, fellers "walked" perpendicularly to within 5 m of the stream channel and reached for merchantable trees along the stream bank. The harvesting methods were similar to existing riparian regulations in Alberta for intermittent streams where a "15 m soft buffer" is applied.

Daily mean data were used to compare stream water discharge and water yield. The total number of discharge values (n_d) differed between years but corresponded to when flows began and to when ice reduced flows to levels below detection limits. The mass flux of sampled chemical parameters was determined from the average of two values collected at 72-hour increments for water chemistry and daily mean water discharge. The total number of water chemistry concentration and flux rate data for each stream in each year (n_c) is presented in Table 6-5. Harvest and reference values for water discharge, flow weighted mean concentration and 72-h flux were paired in time to produce experimental minus reference values during the pre-harvest (Dpre) and post-harvest (Dpost) years. The test statistic Dpre - Dpost was compared to 1000 to 5000 synthesized

Dpre - Dpost values created by Monte Carlo simulation where experimental - reference values were randomly selected from the population of n_d or n_c with replacement irrespective of year (Carpenter et al. 1989).

Results

Pre and post treatment climate

Interannual differences in snow accumulation and precipitation occurred that could influence comparisons of stream discharge between pre and post-harvest years. Total annual precipitation during the pre-harvest year of 1999 was similar to the long-term annual mean for lowland sites but was a wet year in the Mountain sites (Table 6-2). The post-harvest year was comparatively dry in both mountain and lowlands basins compared to the pre-harvest year. Total annual precipitation in 2000 was reduced to approximately 74 and 68% of 1999 values for the mountain and lowland basins, respectively (Table 6-2). Total rainfall (May through September) was 67 and 77% in 2000 compared to 1999 for mountain and lowland basins, respectively. Post-harvest snowpacks were also different; the snowpack was elevated in mountain sites and reduced in lowland sites compared to the pre-harvest year. The snow water equivalent in 2000 was elevated in the reference mountain catchment by 40% but reduced in the reference lowland catchment to 72% when compared to 1999 values. In harvested areas, the mean snow water equivalents in both mountain and lowland basins were approximately half those in undisturbed areas (Table 6-2). Reduced snow accumulation due to harvesting activities was likely related to the timing of snowfall. In 2000, a majority of snow fell prior to and during harvesting activities. Though snow accumulation was low the spring following harvesting, there was no difference one year post-harvest. In the continuing study for 2001, snow fell later in the winter and mean snow water equivalent for accumulated snow was similar in reference and harvested sites despite scarification activities (Table 6-2). I additionally measured snowpacks at sites in two and three year old cutblocks. These sites accumulated 15 to 20% more snow than the aerial mean for forested sites. Snow accumulation in these older cut blocks was similar to accumulation in valley bottoms and open areas such as wetlands. Spring runoff may be reduced following the winter harvest year because the disturbance reduces snow accumulation, however, snow accumulation and spring runoff are not impacted or may be enhanced several seasons after harvesting.

Impacts of clear cut harvesting on groundwater levels

Prior to harvesting, groundwater levels in the mountain basins decreased through summer months as did the depth to concrete frost (Fig. 6-2). Following late July, ice was no longer present in the wells and the water table stabilized or rose closer to the surface through the rest of the open water season. In the untreated mountain basin, groundwater levels were not different between 1999 and 2000 ($P = 0.8$) averaging –71 and –72 cm, respectively at the downstream transect. In the harvested mountain basin, mean water depth rose 24 cm following harvesting ($P = 0.03$) when compared to the mean depth (-62 cm) prior to harvesting (Fig. 6-2). In peatlands surrounded by harvesting activities, mean water depth also increased from –41 cm to –33 cm for the treated mountain basin ($P = 0.004$). Water lost through transpiration prior to harvesting was expected to form runoff after harvesting, however, increased groundwater depth indicated a large proportion may have remained in groundwater storage.

Prior to harvesting in lowland basins, groundwater levels dropped slowly and continuously through the summer (Fig. 6-3). Though continuous, the average drop in groundwater levels during 1999 was 14 cm for lowland wells which was less than the average May through October decline for mountain basins. In 2000, lowland groundwater levels dropped rapidly after snowmelt, possibly due to lower snow accumulation that year. However, groundwater levels rose in response to rainfall in early July which brought water levels up to values comparable with 1999 (Fig 6-3). Mean 1999 and 2000 groundwater levels were -59 and -67 cm in basin LH2 and -58 and -63 cm in basin LH3, and -48 and -54 cm in LR, respectively. Differences in mean groundwater levels were not detectable before and after harvesting at these sites ($P \leq 0.5$). Water levels in LR wells declined an average 6 cm compared to 1999 but were expected to decline further given that precipitation was reduced by 13 cm in 2000. Artificially high levels in LR wells may have been maintained by a downstream beaver dam and may not have reflected dry conditions. Harvesting did not have a clear impact on groundwater levels in lowland catchments, however, the mean values suggest water levels increased from LH2 to LH3 where harvesting was more extensive.

Impacts of harvesting treatments on stream discharge characteristics

In both pre and post-harvest years, the mountain and lowland catchments demonstrated two juxtaposed patterns in runoff with the former dominated by snowmelt and the latter dominated by summer events. In mountain catchments, snowmelt was an important component of annual runoff with peak discharge being two or more times that observed during later rain events (Fig. 6-4a). The rapid response of the mountain streams to snowmelt was likely due to the occurrence of concrete frost in upper soil horizons limiting infiltration and promoting lateral flow and runoff generation (Woo and Winter 1993; Metcalfe and Buttle 2001). Runoff during summer was composed largely of groundwater with infiltration and lateral flow dominating water movement even during storm events. In lowland basins, the snowmelt period did not provide the substantial runoff observed in the mountain basins. Instead, peak snowmelt discharges were typically half of summer peak discharges in the lowland catchments. Like the mountain basins, concrete frost remained in shallow soils of lowland catchments through the snowmelt period (Fig. 6-3). However, snowmelt water was stored in wetlands and in surface pools rather than generating runoff in the lowland catchments. Subsequent summer events in lowland basins produced peak discharges from the saturated landscape that were several fold higher than those during snowmelt.

In the pre-harvest year, the difference between peak snowmelt discharges for the MH and MR basins were consistent with their difference in basin size. As a result both basins had similar water yield during snowmelt in 1999 (Table 6-3). Following harvesting, snowmelt continued to provide the largest annual peak discharge in the mountain basins. However, the timing of peak flow was delayed by two weeks in MH relative to the MR basin (Fig. 6-4b). Two factors seemed to contribute to the delayed peak discharge in the MH basin: a smaller snowpack to provide discharge (Table 6-1) and increased groundwater frozen in soil and the riparian zone. Though only sampled at the gauging station, ice in the riparian zone was 1.5 m thick, double what it had been during pre-harvest years.

In the post-harvest year, the MH basin produced less water relative to its size than did the MR basin (Table 6-3) and the difference between peak discharges narrowed. However, the inter-ecosystem differences for water discharge rates (Dpre and Dpost) were not different during snowmelt periods (Fig. 6-4b). Summer patterns of generally declining flow and response to precipitation remained similar in MH and MR basins. Although the pattern was similar, both summer discharge and water yield from the MH basin declined relative to the MR basin after harvesting ($P \ll 0.01$). Total June through October water yields were 51 mm from the harvested mountain basin compared to 94 mm from the reference mountain basin.

Total water discharge and yield declined for all lowland streams in 2000, likely due to reduced snow accumulation and precipitation. In the lowland reference basin (LR), water yield declined by 46% in 2000 when compared to 1999. Snowmelt discharge from the extensively cut lowland basin (LH1) also declined during snowmelt but remained relatively unchanged when compared to the reference stream (Fig. 6-5a). However, harvesting in LH1 seemed to increase water discharge rates during summer storms (Fig. 6-5b) resulting in a moderate but significant increase in summer discharge rates ($P = 0.02$) and water yield ($P \ll 0.01$) when compared to the reference stream. The moderate harvested lowland stream (LH3) demonstrated enhanced discharge throughout snowmelt and the majority of the summer ($P << 0.01$) and had elevated basin water yield through the open-water season ($P << 0.01$) when compared to the reference stream (Fig. 6-6). Increased discharge from LH1 during storms was likely a result of rapid flushing from surface pools that developed in the LH1 catchment after harvesting. Increased discharge from LH3 may have been related to the large proportion of wetlands (Table 6-1) along harvested sections of the basin. In LH2, with the least proportion harvested, discharge rates and water yield did not increase and in fact appeared to decline relative to the reference basin (Fig. 6-7) in keeping with the observed 3 cm decline in watertable. The interecosytem differences between LH2 and LR for water discharge rates and water yield were not detectably different ($P = 0.12$ and 0.4, respectively). Harvesting caused enhanced water yield from the moderate and extensively cut lowland catchments but did not seem to impact the low harvest basin LH2.

Impacts of harvesting treatments on sources of water

Prior to harvesting, groundwater followed by water from peatland soils formed the majority of runoff in the MH catchment (Table 6-4). In the MR catchment, peat soils supplied the majority of runoff followed by groundwater (Table 6-4). Both basins discharged roughly equal proportions of water with a precipitation signature. Following harvesting in MH, a majority of discharge switched from groundwater to water with a precipitation signature. Discharge derived from precipitation increased 17% relative to the reference basin ($P < 0.01$). The proportion of runoff derived from groundwater sources declined by 22% ($P < 0.01$) relative to the reference basin. Contributions from organic sources in MH increased 5% to replace lost groundwater sources (Table 6-4 and Fig. 6-8). Although harvesting did not appear to impact water yield from MH during snowmelt, it did cause more water to be exported from surface or shallow soil sources.

The increase in relative importance of precipitation following harvesting occurred after snowmelt and continued through the summer to mid-August. During this period basin MH produced 24% more water with a precipitation signature on average than MR whereas the two basins had been matched to within 1% during the pre-harvest year. Summer discharge in the MH basin may reflect contributions from the elevated water table that occurred following harvesting. Although summer discharge from MH was low relative to the MR basin, the water was supplied by rain events, surface storage and perhaps shallow soil water or vadose zone water which was dilute compared to deeper groundwater.

Prior to harvesting, source waters in lowland basins were dominated by precipitation during snowmelt (Table 6-4). However, there was no distinct snowmelt response as the importance of precipitation gradually declined through the summer without the sharp rise and fall expected from events. As watertables declined in the lowland basins, stream discharges became increasingly dominated by organic sources reaching 60 to 80% of total discharge by the onset of fall ice-cover. Contributions from groundwater increased marginally through the summer but remained below 20% in all streams. Stream discharge following summer storms was also provided by organic soil waters suggesting that lateral flow through organic soils or discharge from wetlands were the dominant water pathways in lowland basins.

Each of the lowland streams likely behaved differently following harvesting due to differences in the amount of disturbance and the influence of peatlands in each basin. In the catchment with the largest percentage cut (LH1), stream discharge with a precipitation signature increased 17% relative to the reference basin ($P << 0.01$, Table 6-4, Fig. 6-9). In the basin with the lowest percentage cut (LH2), discharge with a precipitation signature increased 6% relative to the reference basin ($P << 0.01$). Both streams saw a relative decline in contributions from organic sources ($P << 0.01$) and no detectable change in relative contributions from groundwater. The intermediate harvest basin (LH3) showed a decline in contributions from precipitation and organic sources and an increase in groundwater sources by 100% ($P << 0.01$, Fig. 6-10). The pattern was reversed in LH3 possibly because road clearing removed debris dams and drained a downstream wetland. In two of three harvested lowland basins, the proportion of discharge with a precipitation signature increased with more harvesting causing a shift to more dilute runoff. In the third catchment, water yield increased markedly after harvesting and was derived from mineral groundwater sources.

Impacts of harvesting treatments on water chemistry

Following harvesting, changes in the relative contributions of groundwater, organic soil water and dilute precipitation waters changed stream chemistry. In the MH basin, the relative increase in precipitation and decline in groundwater resulted in reduced flux rates of base cations and sulfate relative to the MR catchment (Table 6-5). Unlike the major mineral ions, concentrations of nitrogen, phosphorus and carbon increased relative to the reference catchment ($P << 0.01$). Despite higher concentrations, flux rates for nitrogen, phosphorus and carbon from the MH catchment remained similar to the MR catchment because water yield was reduced. The lack of dilution suggests that increased runoff derived from precipitation sources reached the stream channel as lateral flow through shallow soils rich in nitrogen, phosphorus and carbon.

In the lowland basins, flux rates of most parameters declined in the post-harvest year. The decline was partly due to reduced water yield. Year 2000 flux rates in the lowland reference basin were 14 to 50% of flux rates in 1999 for the same parameters (Table 6-5). In basin LH1, flux rates of dissolved phosphorus and K^+ declined and other parameters (DN, Mg^{2+}, Na^+, SO_4^{2-} and DOC) remained unchanged relative to the flux rates of the reference basin. Flux rates of Ca^{2+} did not decline as much in LH1 as they did in LR giving an apparent increase in Ca^{2+} flux post-harvest ($P << 0.01$). Despite the relative decrease in discharge in LH2 following harvesting, flux rates for Ca^{2+}, Mg^{2+}, Na^+,

SO_4^{2-} and DOC declined by only 40 to 70% which, relative to LR, resulted in an apparent increase in flux rate for these parameters ($P << 0.01$). Flux rates in catchment LH2 declined for K^+ ($P << 0.01$) but remained unchanged for DN and DP. In LH3, flux rates increased for all analyzed parameters after harvesting ($P << 0.01$). The large increase in flux from basin LH3 occurred because of the doubling in relative contributions from groundwater in addition to the elevated discharge that occurred in this basin (Table 6-4).

Impacts of harvesting treatments on suspended sediment loads

Potential increases in suspended sediment loads following harvesting are a primary concern for timber and pulp companies. Harvesting can increase suspended sediment loads by reducing bank stability and increasing storm discharge (Kreutzweiser and Capell 2001). In mountain basins, suspended sediments were 90% inorganic soil particles (NVSS). Median concentrations in reference stream MR were 19 and 17 mg•L^{-1} in 1999 and 2000, respectively (Table 6-6). None of the changes in sediment concentrations were significant by non-parametric tests. Samples were too few to perform parametric statistics. In MH, mean and median concentrations of inorganic material declined by 10-fold after harvesting (Table 6-5). The decline in suspended solids following harvesting was likely due to reduced summer water discharge while harvesting operations appeared to have minimal impact on bank stability. In the lowland reference stream, mean and median concentrations of total suspended solids (TSS) remained unchanged between 1999 and 2000 while volatile suspended solids (VSS) appeared to increase and NVSS decreased in 2000. In LR, median VSS was more than 3-fold higher in 2000 compared to 1999. VSS increased in all three treatment streams after harvesting, however, the increases were less than the 3-fold observed in LR. As a result, there was an apparent reduction in suspended organic matter as represented by VSS in all three harvested lowland streams. While organic inputs declined, enhanced soil erosion following harvesting was not evident in non volatile suspended sediment concentrations in the study streams.

Discussion

Potential hydrologic impacts from timber harvesting include reduced infiltration, increased surface runoff and increased export of major nutrients such as nitrogen, phosphorus and base cations (Ahtiainen 1992; Elliott et al. 1998; Prepas et al. 2001). Mechanical compaction, increased soil saturation, reduced evapotranspiration and changes in biotic activity in soils are some of the causes for increased water and solute flux following harvesting (Fredriksen 1971; Megahan 1983; Lamontagne et al. 2000). In Canada's northern boreal forest, harvesting is often focused during periods when the ground is frozen, ostensibly to limit some of the mechanical impacts on soil. However, in a large proportion of the boreal forest, even small impacts on hydrology can cause large changes in water levels within saturated elements of the landscape and changes in surface water chemistry (Buttle and Metcalfe 2000). The boreal forest of northern Alberta, with its deep organic and mineral soils, exemplifies this sensitivity to changes in hydrology (Prepas et al. 2001).

Snowmelt usually occurs sooner and more moisture is lost to ablation following clear-cut harvesting (Verry et al. 1983). The result can be small peaks in stream discharge over an extended melt period (Buttle and Metcalfe 2000). Harvesting in the mountain catchment was expected to cause increased lateral flow of soil water and the rapid generation of runoff from steep slopes and frozen ground (e.g., Megahan 1983). We further hypothesized that rapid discharge of melt water while soils remained frozen would reduce the amount of water available for storage and we expected groundwater levels to remain unchanged after harvesting despite reduced water loss to evapotranspiration. However, groundwater levels increased in our mountain catchment following harvesting, peak discharge during snowmelt was elevated but short lived, and water yield during snowmelt remained similar to that in the reference catchment. Water yield calculations accounted for the reduced snowpack at harvested sites and much of the enhanced ablation that occurred. The implication was that runoff generation during snowmelt was not elevated by harvesting, largely because harvesting resulted in reduced snow accumulation but also because infiltration remained relatively unchanged in the mountain catchment.

Unlike snowmelt, summer water yield declined following harvesting in the mountain catchment. The decline in water yield was contrary to expectations and studies in other locations where reduced evapotranspiration contributed to runoff (Megahan 1983; Hornbeck et al. 1993). Instead, summer water yield was only 36% of pre-harvest yield or a 44% decline relative to the reference basin. Evapotranspiration estimates for MH and MR of 216 and 263 mm in 1999 and 231 mm from MR in 2000 were obtained by water balance, assuming no change in groundwater storage. Roughly 100 mm of water formerly lost to evapotranspiration was expected to become runoff or groundwater storage after harvesting. The higher than expected rise in groundwater levels (240 mm) observed in the MH wells was likely magnified by focusing sampling in valley bottom wells that intersected the watertable throughout the summer. In peatlands surrounded by harvesting, the rise in groundwater level was 80 mm, consistent with expected increases if no additional runoff was produced. Summer water yield from MH declined by 88 mm indicating a potential maximum for increased groundwater storage of 188 mm. A rise in groundwater and reduced water yield suggests harvesting restricted drainage in MH possibly from soil compaction. Recent studies in other northern boreal forest catchments found little support for changes in groundwater residence time following harvesting (Buttle et al. 2001). However, Buttle et al. (2001) indicated that their data were not conclusive and suggest that increased residence time, and the corollary suggested by my data, *viz* reduced drainage from groundwater, still requires investigation as potential post-harvest impacts.

Despite substantial changes in hydrology, studies on the impacts of logging and forest fire on stream chemistry may not distinguish changes in concentration or flux for many compounds from naturally large inter-annual variation (Wright 1976; Minshall 1997). In the mountain basin, water routing was expected to continue to derive from shallow soils following harvesting and changes in water yield were expected to have little impact on the composition of stream water chemistry. Similar findings have been reported from the Turkey Lakes, Ontario study where relatively unchanged soil water pathways resulted in similar stream water compositions of silicate and other ions following harvesting (Hazlett et al. 2001). Changes in hydrology in the harvested mountain basin resulted in stream water chemistry changes for nutrient and mineral ions. The increase in the relative importance of precipitation and shallow organic soil as sources for runoff, but reduced discharge, resulted in phosphorus and organic carbon flux rates that remained unchanged and concentrations that were moderately enhanced after harvesting. Both the concentration and flux rates of mineral ions, such as Ca^{2+}, Mg^{2+}, and Na^{+}, declined following harvesting in the mountain catchment due to increased contributions from shallow organic soils relative to deeper mineral soils and because of reduced summer water yield.

In lowland basins, water discharge and yield increased following harvesting relative to the reference catchment. Like the mountain basin, harvested lowland basins had reduced snowpack compared to the reference catchment. Reduced precipitation in the post-harvest year resulted in lower watertable and water yield overall. In the LR catchment the change was reflected in a decrease in water levels in wells, a 3-fold decrease in water yield and an increase in relative contributions from mineral groundwater compared to pre-harvest years. In the post harvest year, precipitation also increased in overall importance as a source for runoff in LR due to the relative increase in the importance of rapid discharge during summer storms as flow between storms and the relative contribution from organic sources declined.

In two of the three harvested catchments, higher water yield relative to the reference catchment was exemplified by an increase in contributions from precipitation.

Harvesting impacted the third catchment differently with an increase in mineral groundwater. Unlike MH, harvesting in LH3 may have created better drainage when debris dams were removed from upstream and downstream locations and a drainage connection between a large fen complex and the stream was created by equipment traffic. Flux rates of ions and nutrients generally declined or remained unchanged in the two basins where precipitation increased. Base cation flux remained relatively constant from these two catchments because groundwater contributions remained unchanged. Except in LH3, our hypothesis for elevated flux from lowland systems was not supported; increases in discharge did not result in increased flux largely because groundwater contributions declined relative to precipitation. In LH3 flux rates increased markedly because groundwater contributions doubled after harvesting.

EMMA proved a valuable tool for addressing hydrologic processes, their potential changes following harvesting and their role in determining runoff chemistry in the study catchments. Water derived from soil layers was likely a continuum of chemical compositions depending on equilibrium conditions and path length as well as soil heterogeneity. As such, EMMA could not determine explicitly how much water derived from each source. However, EMMA did capture the relative importance of soil layers in determining runoff chemistry, provided the end-members were properly described. Few studies have employed EMMA to determine hydrologic process and subsequently infer reasons for changes following basin disturbance, likely because of the difficulty in determining the end-members. A decision rule that end-members were properly identified could be justified by the behavior of stream chemistry with the following acceptance criteria: did the end-members capture the stream chemistries; did the end-members change and if so did they follow a trajectory that was consistent with mixing from other soil sources. In this study, stream water chemistry was constrained within the bounds of the chosen end-members. End-member chemistry changed through the summer but followed consistent patterns of mixing between other end-members (see Chapter five: McEachern et al.

submitted). This information can be supplemented with determinations of the relative water age using the isotopes deuterium and ^{18}O. We generally found close agreement between changes in soil water chemistry and isotope-determined water age. The results, presented in Chapter five formed a strong basis for selecting maximum concentrations to represent groundwater chemistry rather than median values influenced by mixing of soil waters in wells during spring and storm events. One potential weakness in EMMA is that by including all potential solute tracers in U-space, one limits the ability for independent confirmation of predictions. In our study we used isotope-determined water age to validate EMMA results. Despite potential weaknesses, chemical tracing techniques like EMMA may offer the best catchment scale insight into hydrologic conditions and changes following disturbance in remote areas where describing hydraulic characteristics through intensive instrumentation is not feasible.

Table 6-1: Channel slope, catchment area, relative vegetation cover and harvest area within each catchment.

Stream Site	Ave. Channel Slope (%)	CA (ha)	Harvest Area (% CA)	Fen (% CA)	Riparian Bog (% CA)	Other Bog (% CA)
MH	7	208	55	1.1	2.5	5
MR	9	142	0	2.3	3.6	3
LH1	0.6	280	68	4.2	2.5	35
LH2	0.7	308	24	6.4	4.3	58
LH3	0.6	540	45	6.9	5.5	61
LR	0.6	267	0	15.7	7.2	66

CA= Catchment Area in hectares, The percentage of catchment area in bogs within the riparian area are shown separately from bogs outside the riparian area (other). Site identifiers are M = mountain, L = lowland, R = reference, H = harvested.

Table 6-2: Precipitation data. Annual precipitation is from October 1 to September 30. Snow water equivalent is from spring sampling.

Catchment Type and Year	Total Precipitation (mm)	Total summer precipitation[a] (mm)	Snow water equivalent[c] (mm)
Mountain			
1998	(474)[b]	330 (385)[b]	(144)[b]
1999	493 (505)	446 (457)	47 (55)
2000	365 (350)	298 (279)	66 (71)
cutblock 2000			30[d]
2001	301	134	67 (66)
30-y mean	381	306	75
Lowland			
1998	(300)[b]	220 (223)[b]	(77)[b]
1999	407 (400)	283 (296)	82 (104)
2000	276 (279)	217 (163)	59 (116)
cutblock 2000			35[d]
2001	258	168	90 (86)
30-y mean	402	261	141

a) Mid-May to mid-Sept. precipitation from rain gauge.
b) Number in parentheses are April-Oct estimate of precipitation from Sacramento gauges.
c) Snow pack from 1999 through 2001 estimated from survey measurements mid-March 1999, early March 2000 and 2001. All sites are combined in 1999 and 2001. Reference sites are seperated from harvested sites in 2000. Number in parentheses is a total snowfall estimate from the Sacramento gauge.
d) Second estimate is an average for measurements in harvested sites during early March 2000.

Table 6-3: Annual mean water discharge rates and water yields from the six study streams for the open-water season before and after harvesting.

	Q (L/s)		Yield (mm)	
	Reference	Harvested	Reference	Harvested
Mountain				
99	26.2	45.7	230	277
00	13.7	18.0	134	118.2
Dpre-Dpost	-	12.21	-	0.55
P	-	0.009	-	<0.001
Lowland				
LH1 99	13.7	10.2	33.4	36.9
LH1 00	2.6	5.0	13.1	24.6
Dpre-Dpost	-	-0.15	-	-0.07
P	-	0.016	-	<0.001
LH2 99	13.7	54.6	33.4	179.0
LH2 00	2.6	10.1	13.1	45.1
Dpre-Dpost	-	42	-	-0.01
P	-	0.12	-	-0.01
LH3 99	13.7	12.3	33.4	23.1
LH3 00	2.6	4.2	13.1	10.5
Dpre-Dpost	-	-9	-	-0.07
P	-	<0.001	-	0.008

The total number of measurements (n_d) was 160 for both upland and lowland streams in 2000 and was 147 and 111 for mountain and lowland, respectively, in 1999. Dpre-Dpost is the subtraction of interecosystem differences (experimental-reference) from the preharvest and postharvest years.

Table 6-4: Mean percent contribution to stream waters from the three sampled end members, precipitation, soil water from organic zones and groundwater.

Treatment	Year	Precipitation (%)	Organic (%)	Groundwater (%)
MH	1999	23	30	47
	2000	34	42	24
MR	1999	21	54	25
	2000	16	59	25
LR	1999	17	69	14
	2000	29	50	22
LH1	1999	19	58	23
	2000	49	31	21
LH2	1999	39	44	17
	2000	60	21	19
LH3	1999	37	48	15
	2000	34	35	31

Table 6-5: Summary of volume weighted mean concentration and total flux of chemical constituents from the study streams.

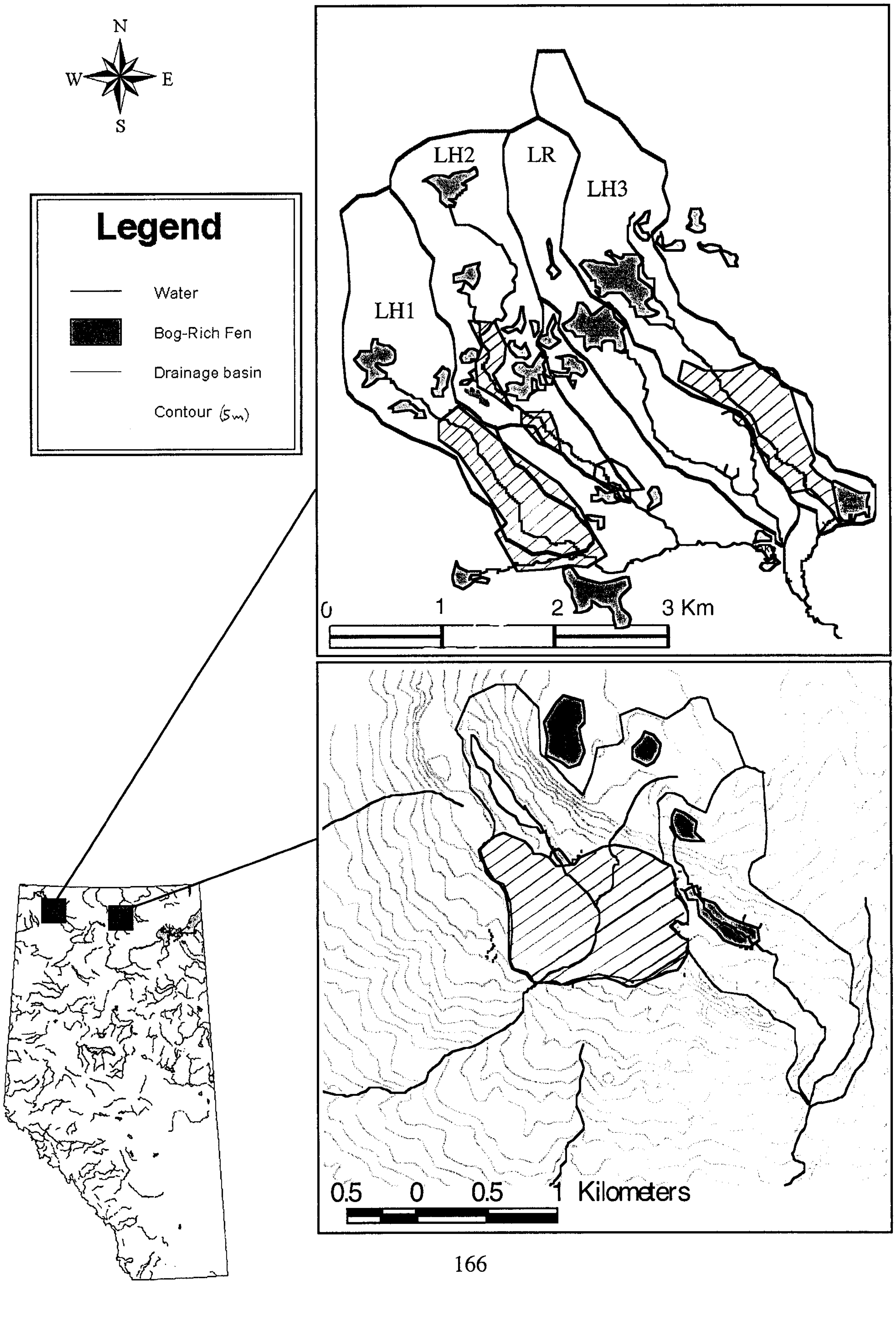
	Mountain Reference		Mountain Harvested		Lowland Reference		Lowland Harvested (LH1)		Lowland Harvested (LH2)			
FWMC (mg/L)	1999	2000	1999	2000	1999	2000	1999	2000	1999	2000	1999	2000
Nc	35	54	39	46	35	45	32	52	33	51	34	53
DN	1.308	0.492	0.896	0.523	0.699	0.973	0.733	0.936	0.945	0.921	0.341	0.996
DP	0.037	0.008	0.025	0.008	0.016	0.037	0.011	0.012	0.015	0.022	0.006	0.020
Ca	281.7	232.5	176.5	152.6	93.4	181.7	168.0	302.8	170.7	200.0	48.4	207.8
Mg	46.5	18.8	33.7	11.0	20.8	27.5	35.5	39.6	34.4	31.2	11.3	25.3
Na	32.2	13.1	15.9	4.3	68.9	57.1	94.1	61.5	58.5	37.0	20.4	43.8
K	3.1	1.1	2.9	1.2	0.5	1.5	0.9	2.2	1.4	2.4	0.8	2.8
SO4	116.5	64.0	122.4	29.8	255.1	319.7	418.8	439.2	360.1	317.9	119.1	264.1
DOC	46.6	11.8	31.2	14.7	29.8	34.6	33.8	35.7	151.4	29.5	14.0	29.2
Flux Rate (g/ha/d)												
DN	11.15	4.11	11.49	3.90	3.53	0.80	1.89	0.38	12.80	3.76	1.05	2
DP	0.316	0.070	0.316	0.062	0.080	0.030	0.028	0.004	0.198	0.062	0.019	0.028
Ca	2402	1943	2265	1139	472	150	434	186	2312	1420	150	638
Mg	396	157	433	82	105	23	92	25	465	158	35	70
Na	275	110	204	32	348	47	243	37	792	269	63	109
K	26.2	9.3	36.9	8.6	2.6	1.2	2.4	0.5	19.0	6.1	2.5	4
SO4	993	535	1570	223	1288	264	1082	317	4875	1518	368	759

Nc is the number of flux rate and flow weighted mean concentrations that could be calculated from sampling in each year.

Table 6-6: Mean total suspended solids (TSS), volatile suspended solids (VSS) and non-volatile suspended solids concentrations in the study streams. Median values are in parentheses

Site	Year	TSS(mg/L)	VSS(mg/L)	NVSS(mg/L)
MH	1999	26.2 (21.1)	4.1 (3.1)	22.1 (17.7)
	2000	27.6 (2.1)	4.0 (0.6)	23.6 (1.5)
MR	1999	149.3 (21.3)	23.7 (2.7)	125.6 (18.5)
	2000	710.6 (20.2)	63.6 (3.4)	647.0 (16.6)
LH1	1999	1.0 (1.0)	0.3 (0.4)	0.7 (0.7)
	2000	1.1 (0.8)	0.7 (0.5)	0.4 (0.4)
LH2	1999	4.1 (2.2)	1.7 (0.7)	2.4 (1.5)
	2000	9.8 (2.2)	6.2 (1.7)	3.6 (0.5)
LH3	1999	0.8 (0.7)	0.5 (0.5)	0.3 (0.3)
	2000	4.2 (1.5)	2.5 (0.9)	1.7 (0.6)
LR	1999	1.70 (1.7)	0.9 (0.6)	0.8 (0.6)
	2000	4.56 (3.2)	3.4 (2.2)	1.1 (0.7)

Fig. 6-1: Location of study catchments, stream sampling sites and harvested areas (hatched polygons).



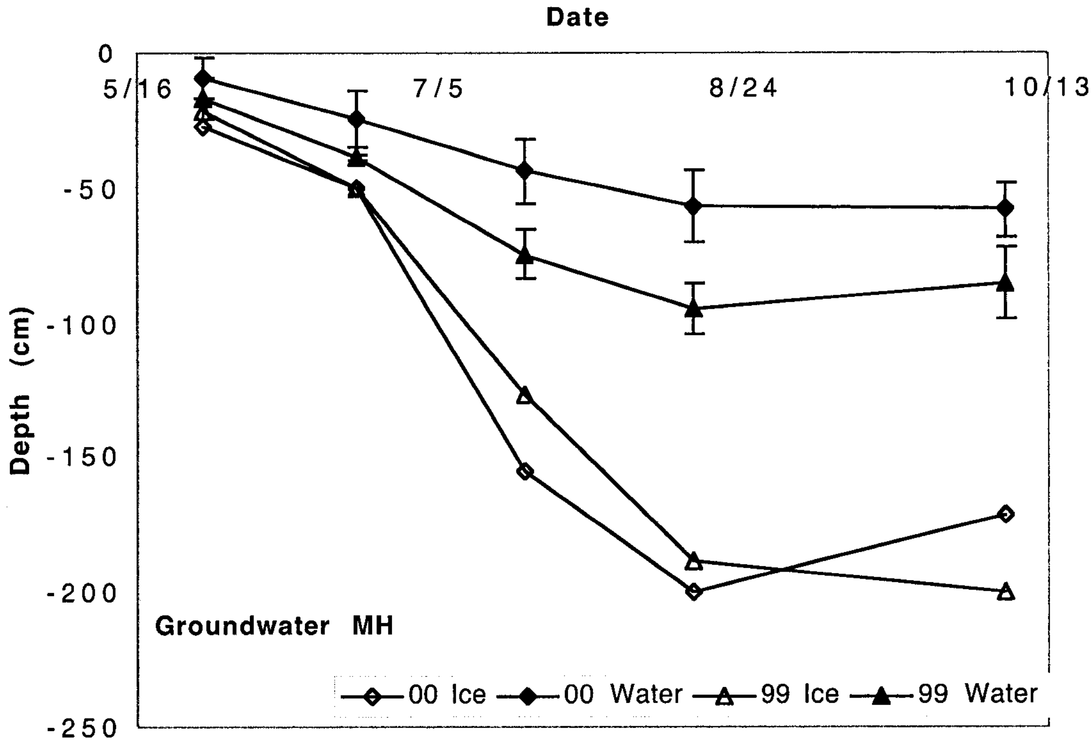


Fig. 6-2a: Mean groundwater well depths for two combined transects in the harvested (MH) mountain basin. Values were linearly interpolated between dates in 1999 to match dates in 2000. Standard errors of the mean are shown for water depth.

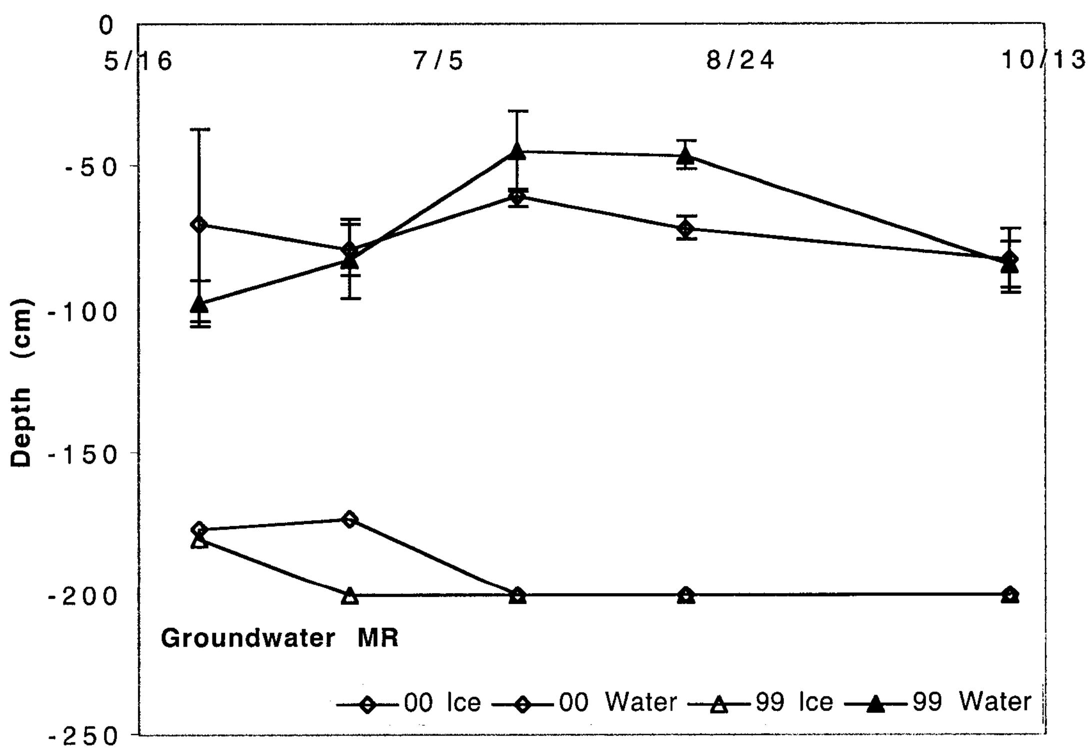


Fig. 6-2b: Mean groundwater well depths for two combined transects in the reference (MR) mountain basin. Values were linearly interpolated between dates in 1999 to match dates in 2000. Standard errors of the mean are shown for water depth.

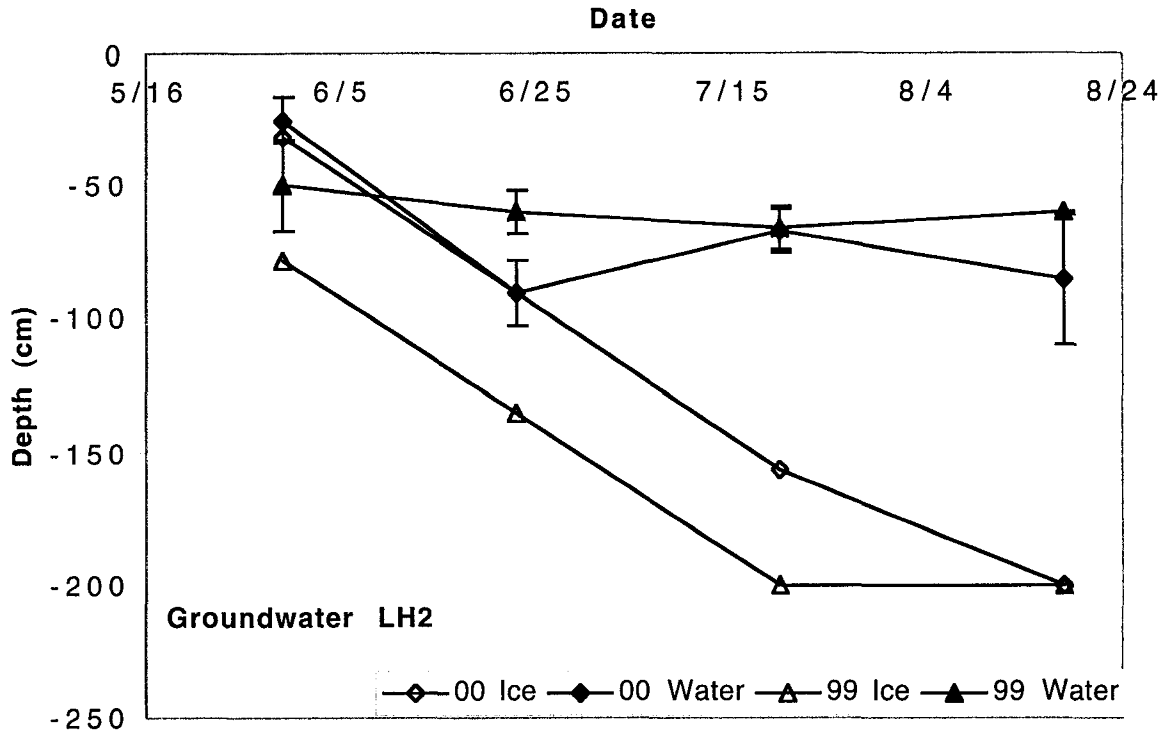


Fig. 6-3a: Groundwater well depths from a transect in lowland basin LH2 (low harvest treatment). Values were linearly interpolated between dates in 1999 to match dates in 2000. Standard errors of the mean are shown for water depth.

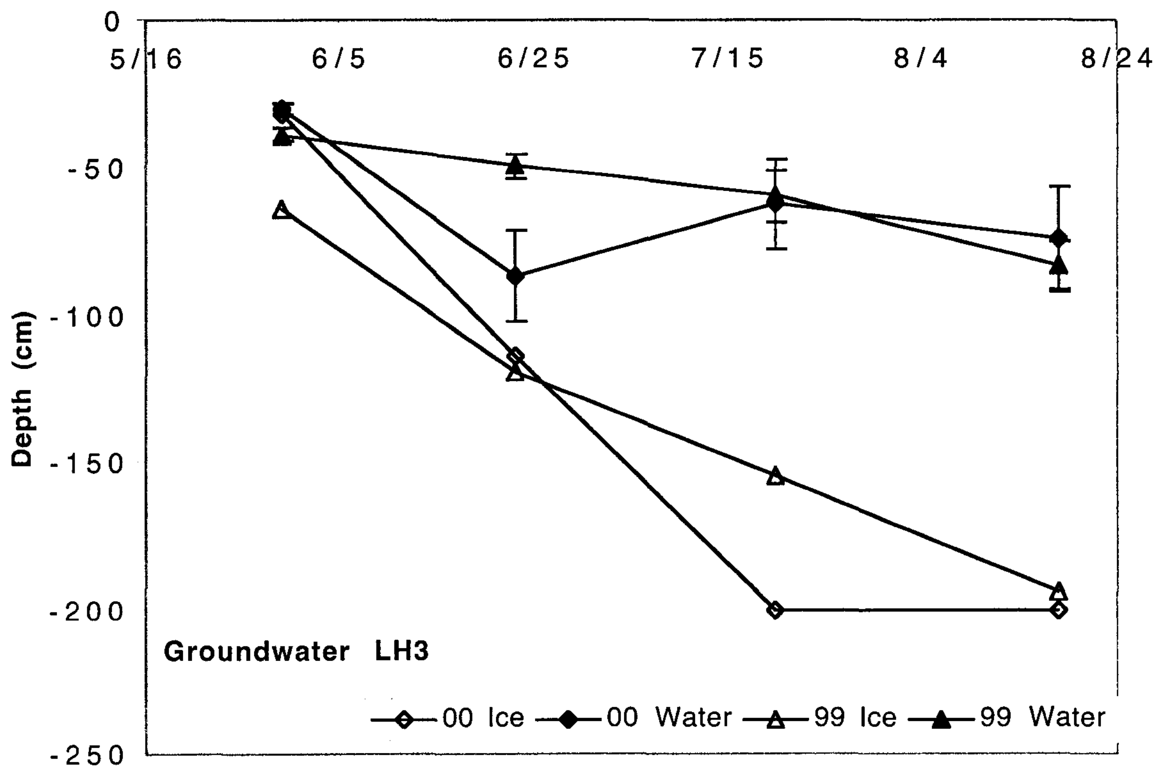


Fig. 6-3b: Groundwater well depths from a transect in lowland basin LH3 (moderate harvest treatment). Values were linearly interpolated between dates in 1999 to match dates in 2000. Standard errors of the mean are shown for water depth.

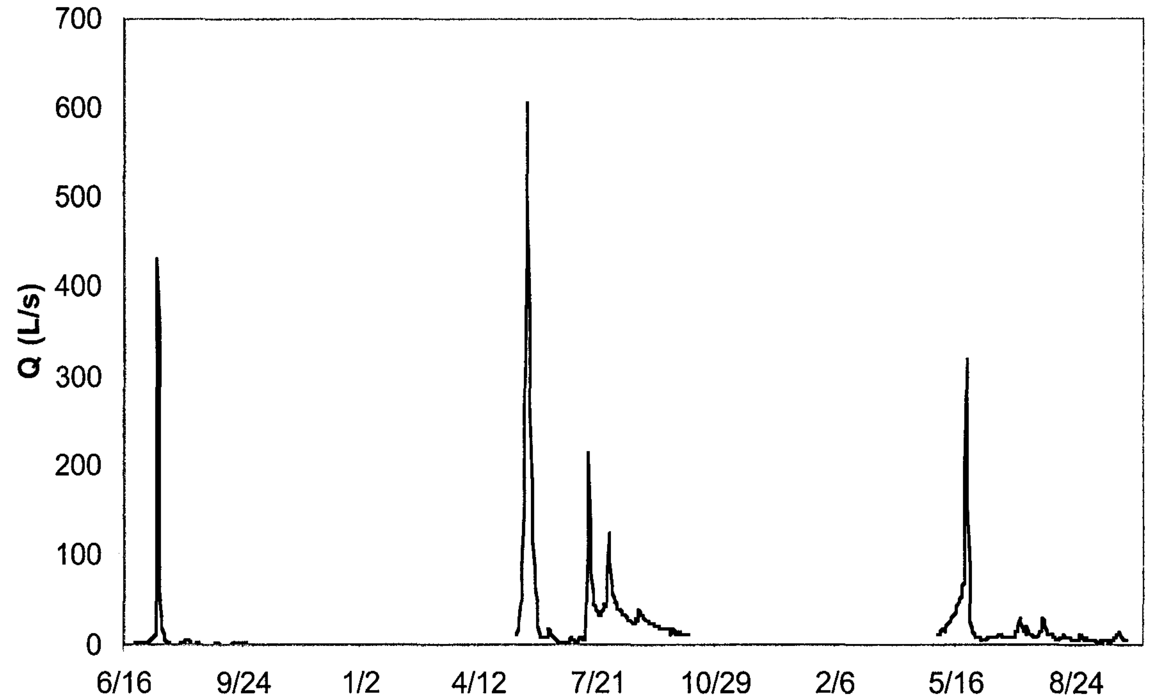


Fig. 6-4a: Hydrograph for the mountain harvested catchment (MH) from June 1998 through October 2000. Harvesting occurred during the winter of 1999-2000.

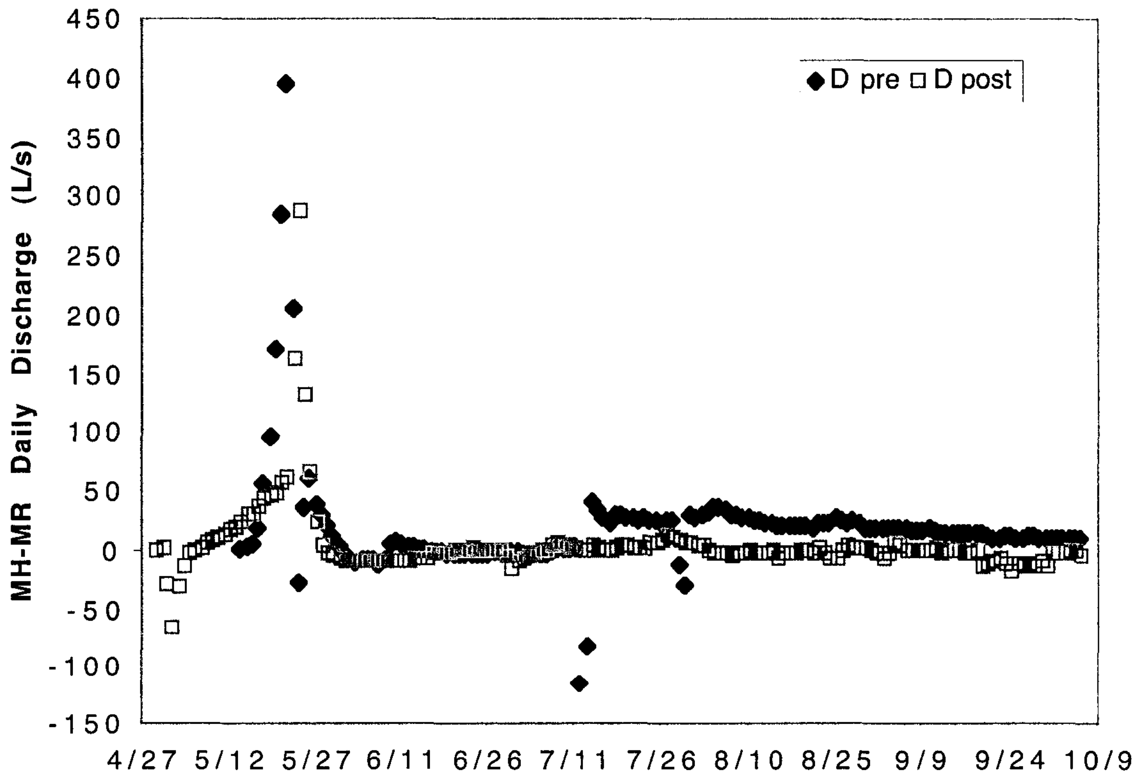


Fig. 6-4b: Differences in daily mean water discharge between the treatment (MH) and reference(MR) mountain catchments. Data show difference in discharge (MH-MR) for pre-harvest (Dpre 1999) and post-harvest (Dpost 2000) years.

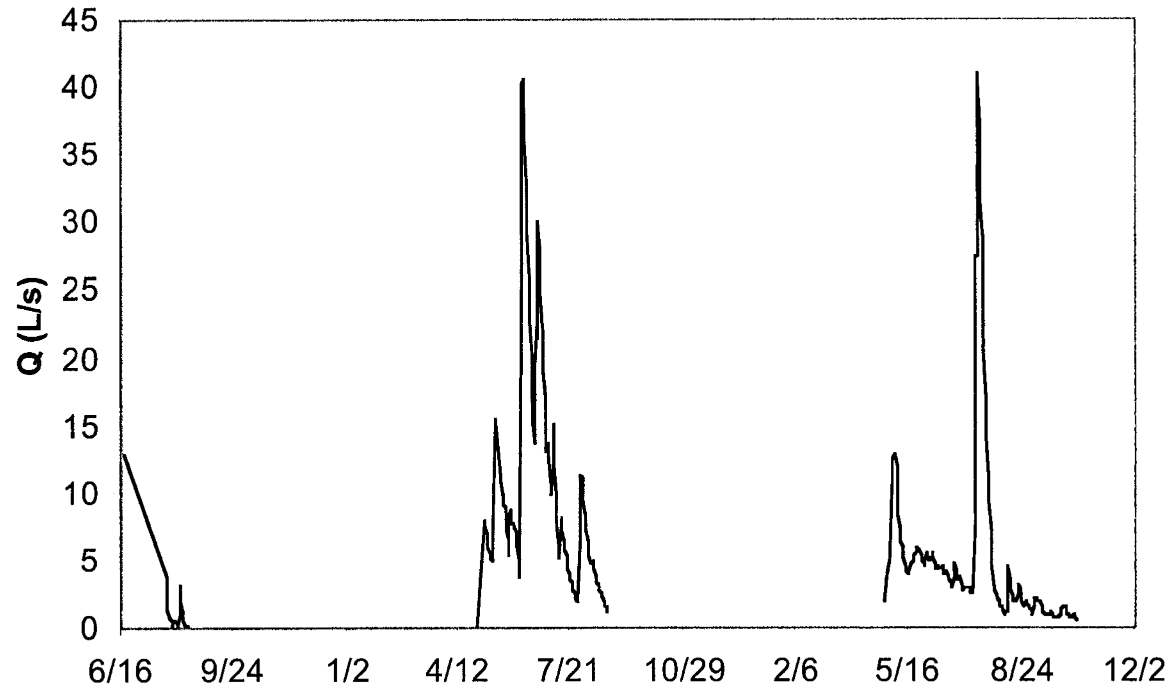


Fig. 6-5a: Hydrograph for the lowland stream with highest harvest treatment (LH1) catchment from June 1998 through October 2000. Harvesting occurred during the winter of 1999-2000. Data from 1998 were not included in statistical analyses.

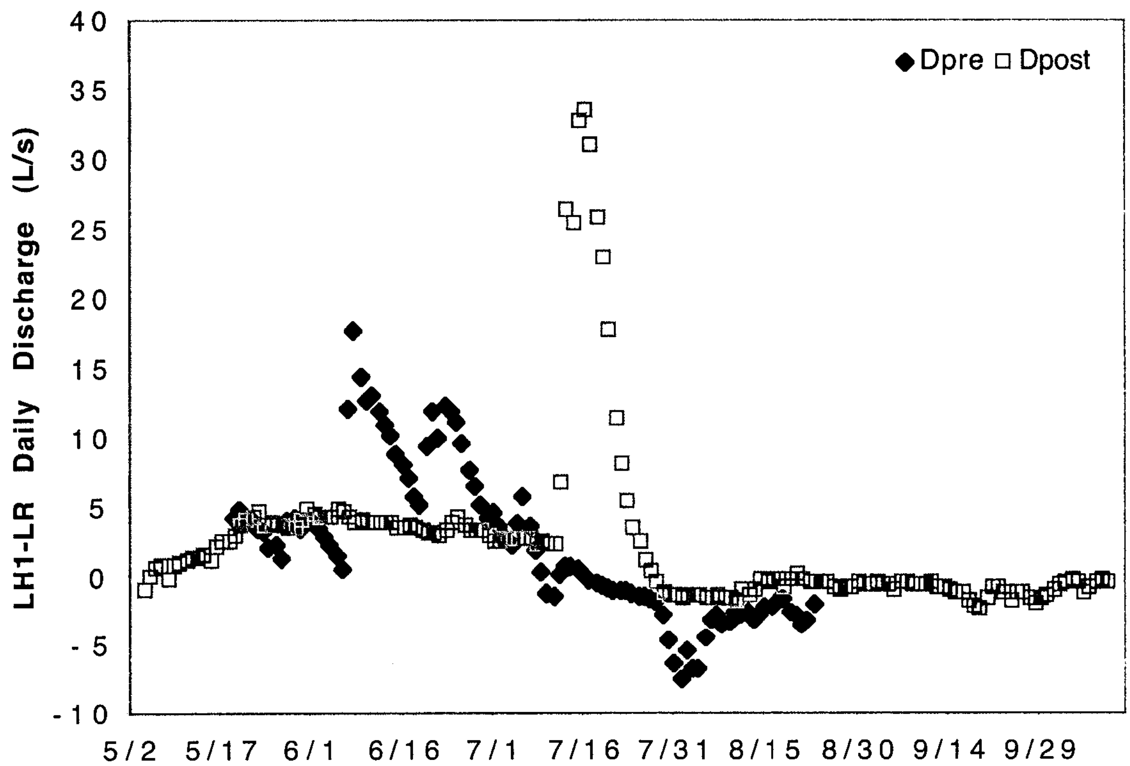


Fig. 6-5b: Differences in daily mean water discharge for extensive lowland treatment (LH1) and reference (LR) catchments. Data show differences in discharge (LH1-LR) for pre-harvest (Dpre) and post-harvest (Dpost) years.

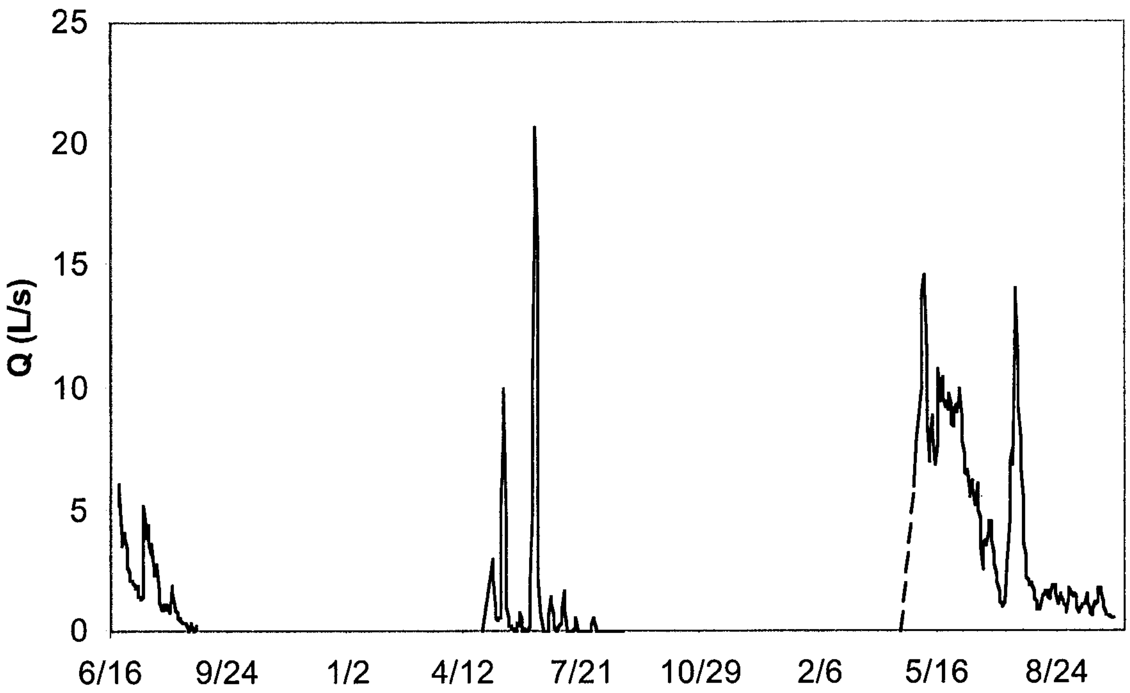


Fig. 6-6a: Hydrograph for the lowland stream with moderate harvest treatment (LH3) catchment from June 1998 through October 2000. Harvesting occurred during the winter of 1999-2000.

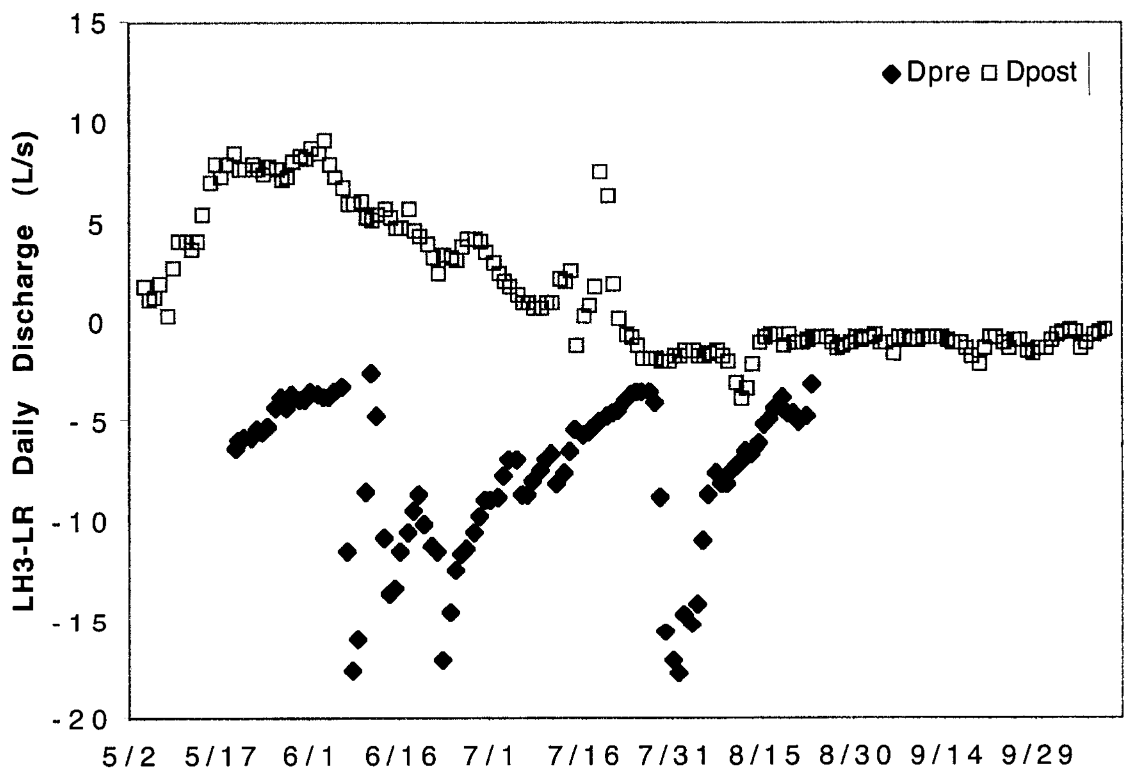


Fig. 6-6b: Differences in daily mean water discharge for moderate lowland treatment (LH3) and reference (LR) catchments. Data show differences in discharge (LH3-LR) for pre-harvest (Dpre) and post-harvest (Dpost) years.

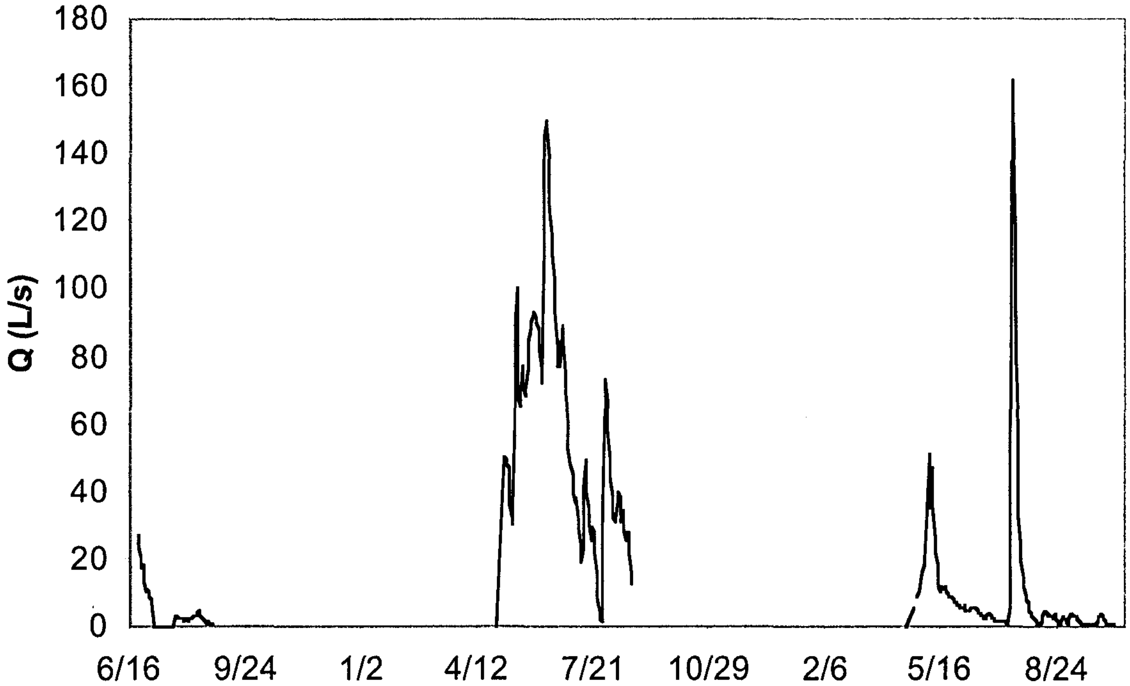


Fig. 6-7a: Hydrograph for the lowland stream with low harvest treatment (LH2) catchment from June 1998 through October 2000. Harvesting occurred during the winter of 1999-2000.

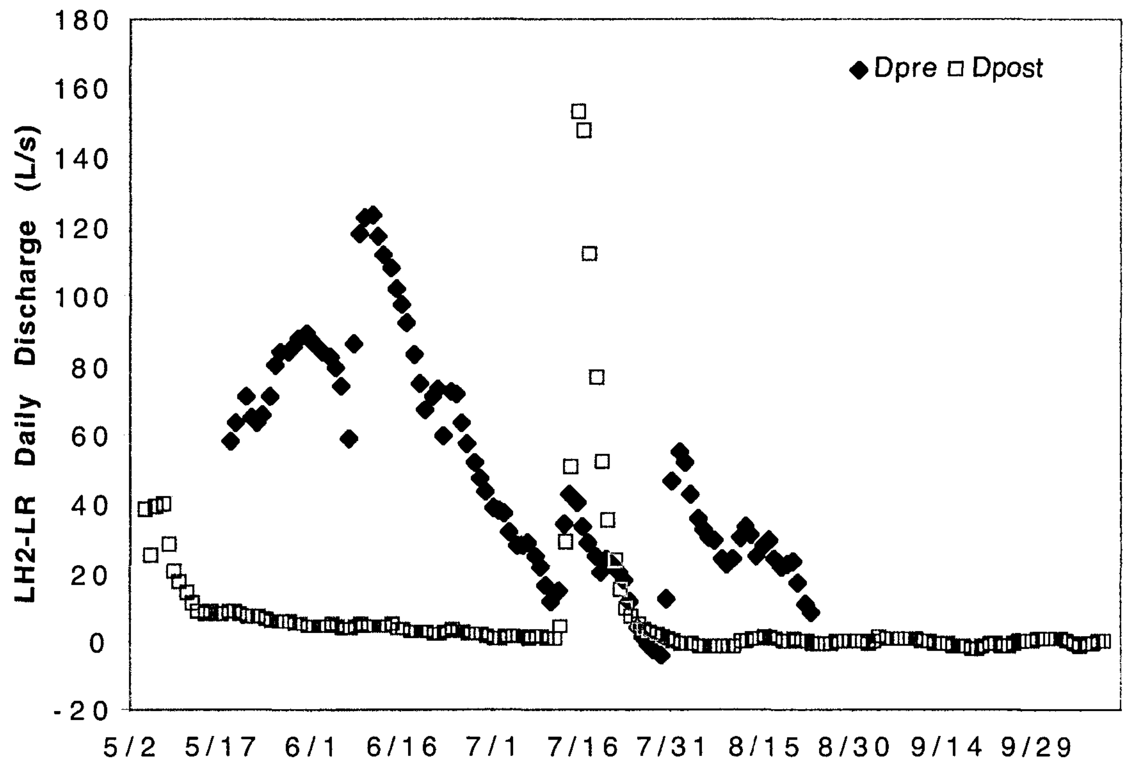


Fig. 6-7b: Differences in daily mean water discharge between lowland treatment with low percent harvested (LH2) and reference (LR) catchments. Data show difference in discharge (LH-LR) for pre-harvest (Dpre) and post-harvest (Dpost) years.

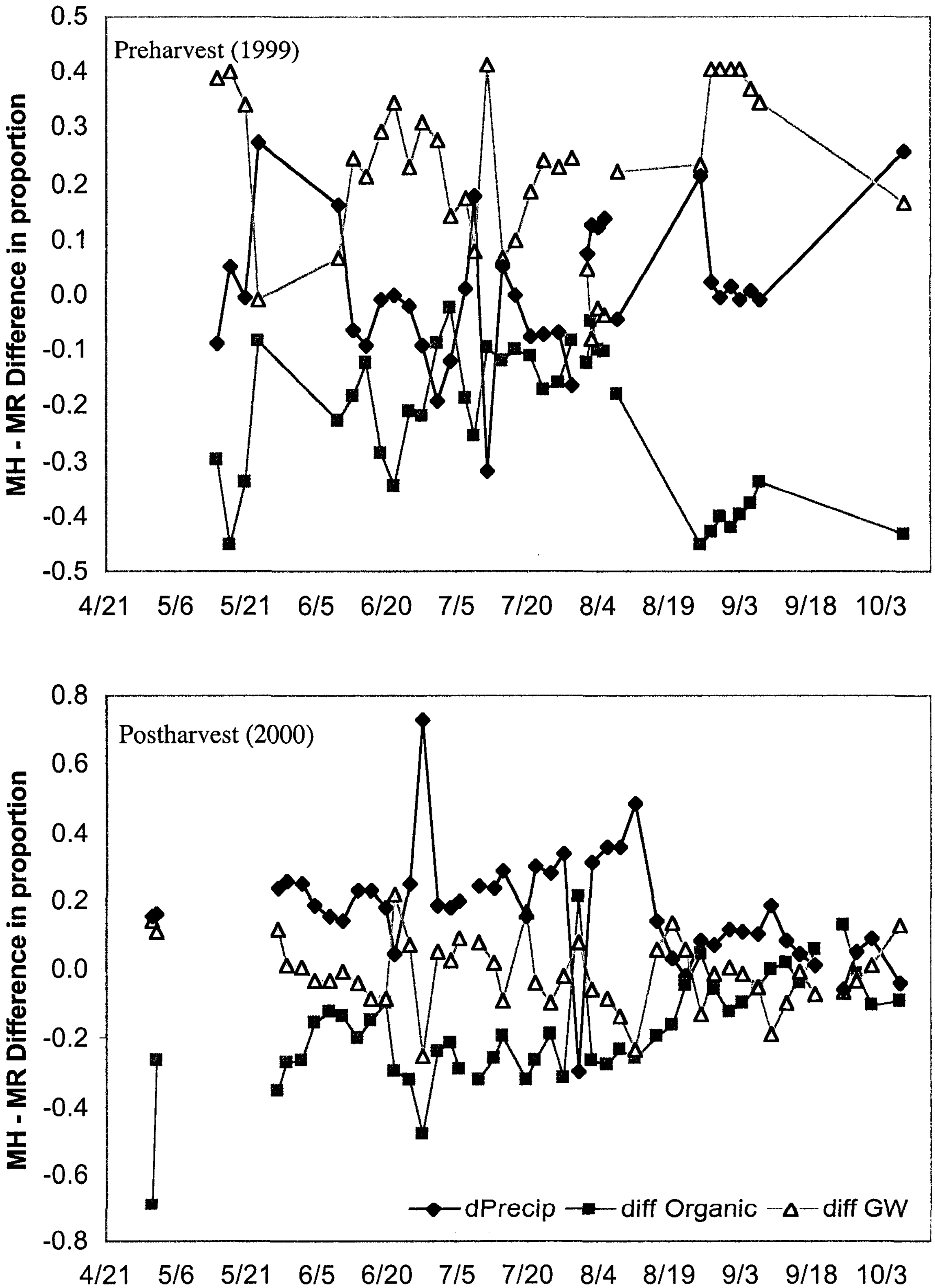


Fig. 6-8: Difference in source water contributions to mountain streams. Data compares proportion of stream discharge derived from three sources, mineral groundwater (GW) organic soil water and surface runoff or precipitation.

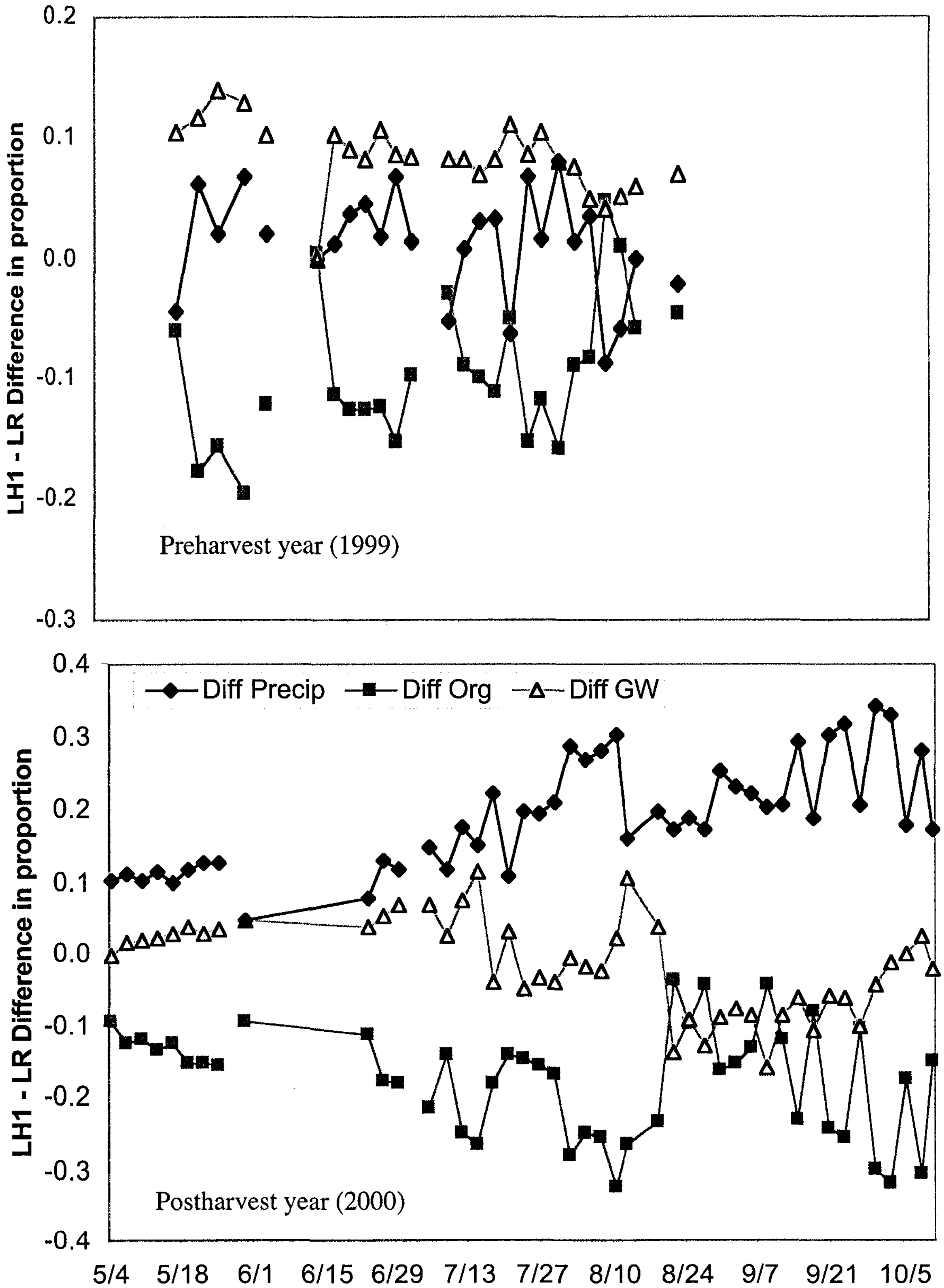


Fig. 6-9: Difference in source water contributions to lowland stream LH1. Data compares proportion of stream discharge derived from three sources, mineral groundwater (GW) organic soil water and surface runoff or precipitation. Note relative increase in precipitation and decline in groundwater contributions compared to the preharvest year. Stream LH2 followed a similar pattern.

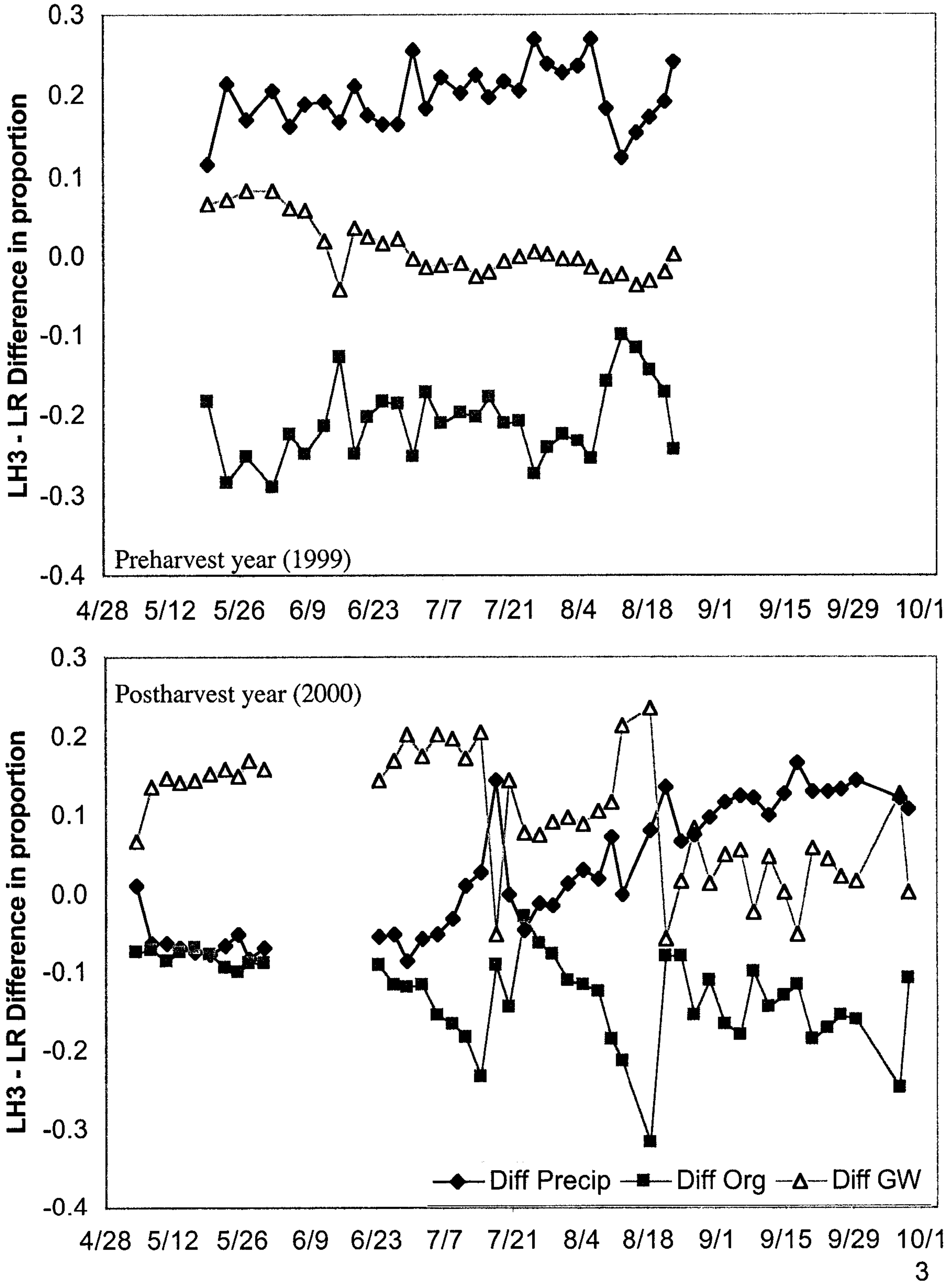


Fig. 6-10: Difference in source water contributions to lowland stream LH3. Data compares proportion of stream discharge derived from three sources, mineral groundwater (GW) organic soil water and surface runoff or precipitation. Note increase in groundwater and organic water with decline in precipitation comaperd to the preharvest year.

References

APHA (1993). Standard Methods for the Examination of Water and Wastewater. © American Public Health Association.

Ahtiainen, M. (1992). The effects of clear-cutting and scarification on the water quality of small brooks. *Hydrobiol.* **243/244**: 465-473.

ARC (1962). Soil survey maps and chemical analyses of Alberta soils. Edmonton, AB, Alberta Research Council.

Buttle, J. M., P. W. Hazlett, C. D. Murray, I. F. Creed, D. S. Jeffries and R. Semkin (2001). Prediction of groundwater characteristics in forested and harvested basins during spring snowmelt using a topographic index. *Hydrol. Process.* **15**: 3389-3407.

Buttle, J. M. and R. A. Metcalfe (2000). Boreal forest disturbance and streamflow response, northeastern Ontario. *Can. J. Fish. Aquat. Sci.* **57**(Supp. 2): 5-18.

Carpenter, S. R., T. M. Frost, D. Heisey and T. K. Kratz (1989). Randomized intervention analysis and the interpretation of whole-ecosystem experiments. *Ecology* **70**(4): 1142-1152.

Christophersen, N. and R. P. Hooper (1992). Multivariate Analysis of Stream Water Chemical Data: The Use of Principal Components Analysis for the End-Member Mixing Problem. *Water Resour. Res.* **28**(1): 99-107.

Crumpton, W. G. (1992). Nitrate and organic N analyses with second-derivative spectroscopy. *Limnol. Oceanogr.* **37**(4): 907-913.

Devito, K. J., I. F. Creed, R. L. Rothwell and E. E. Prepas (2000). Landscape controls on phosphorus loading to boreal lakes: Implications for the potential impacts of forest harvesting. *Can. J. Fish. Aquat. Sci.* **57**: 1977-1984.

Elliott, J. A., B. M. Toth and R. J. Pomeroy (1998). Soil moisture storage in mature and replanted sub-humid boreal forest stands. *Can. J. Soil Sci.* **78**: 17-27.

Evans, J. E., E. E. Prepas, K. J. Devito and B. G. Kotak (2000). Phosphorus dynamics in shallow subsurface waters in an uncut and cut subcatchment of a lake on the Boreal Plain. *Can. J. Fish. Aquat. Sci.* **57**(Supp. 2): 60-72.

Fredriksen, R. L. (1971). Comparative chemical water quality - Natural and disturbed streams following logging and slash burning. Symposium for Forest Land Uses and Stream Environments [October 19-21, 1970], Oregon State Univ.

Gorham, E. (1991). Northern peatlands role in the carbon cycle and probable responses to climatic warming. *Ecological Applications* **1**(2): 182-195.

Hazlett, P. W., R. G. Semkin and F. Beall (2001). Hydrologic pathways during snowmelt in first-order stream basins at the Turkey Lakes Watershed. *Ecosystems* **4**(6): 527-535.

Hornbeck, J. W., M. B. Adams, E. S. Corbett, E. S. Verry and J. A. Lynch (1993). Long-term impacts of forest treatments on water yield: a summary for northeastern USA. *J. Hydrol.* **150**: 323-344.

Kreutzweiser, D. P. and S. S. Capell (2001). Fine sediment deposition in streams after selective forest harvesting without riparian buffers. *Can. J. For. Res.* **31**: 2134-2142.

Lamontagne, S., R. Carignan, P. D'Arcy, Y. T. Prairie and D. Pare (2000). Element export in runoff from eastern Canadian Boreal Shield drainage basins following forest harvesting and wildfires. *Can. J. Fish. Aquat. Sci.* **57**(SUPP/2): 118-128.

McEachern, P., D. S. Chanasyk and E. E. Prepas (Submitted). Landscape control of catchment hydrology, solute flux and water chemistry of streams in northern boreal Alberta. *J. Hydrol.*

McEachern, P., E. E. Prepas, J. J. Gibson and W. P. Dinsmore (2000). Forest fire induced impacts on phosphorus, nitrogen, and chlorophyll a concentrations in boreal subarctic lakes of northern Alberta. *Can. J. Fish. Aquat. Sci.* **57**(Supp. 2): 73-81.

McRae, D. J., L. C. Duchesne, B. Freedman, T. J. Lynham and S. Woodley (2001). Comparisons between wildfire and forest harvesting and their implications in forest management. *Environ. Rev.* **9**: 223-260.

Megahan, W. F. (1983). Hydrologic effects of clearcutting and wildfire on steep granitic slopes in Idaho. *Water Resour. Res.* **19**(3): 811-819.

Metcalfe, R. A. and J. M. Buttle (2001). Soil partitioning and surface store controls on spring runoff from a boreal forest peatland basin in north-central Manitoba, Canada. *Hydrol. Process.* **15**: 2305-2324.

Minshall, G. W. (1997). Postfire responses of lotic ecosystems in Yellowstone National Park, U.S.A. *Can. J. Fish. Aquat. Sci.* **54**: 2509-2525.

Muller, D. I., I. C. Wohlfeil, N. Christophersen, M. Hauhs and H. M. Seip (1993). Chemical reactiveness of soil water pathways investigated by point source injections of chloride in a peat bog at Birkenes. *J. Hydrol.* **144**(1-4): 101-125.

Neal, C. and N. Christophersen (1990). Linking hydrological, geochemical, and soil chemical processes on the catchment scale: An interplay between modeling and field work. *Water Resour. Res.* **26**(12): 3077-3086.

Pawluk, S. (1961). Mineralogical composition of some grey wooded soils developed from glacial till. *Can. J. Soil Sci.* **41**: 228-240.

Prepas, E.E., Rigler, F.H (1982). Improvements in Quantifying the Phosphorus Concentration in Lake Water. Can. J. Fish. Aquat. Sci. **9(6):** 822-829.

Prepas, E. E., B. Pinel-Alloul, D. Planas, G. Methot, S. Paquet and S. Reedyk (2001). Forest harvest impacts on water quality and aquatic biota on the Boreal Plain: Introduction to the TROLS Lake Program. *Can. J. Fish. Aquat. Sci.* **58**: 421-436.

Verry, E. S., J. R. Lewis and K. N. Brooks (1983). Aspen clear-cutting increases snowmelt and storm flow peaks in north central Minnesota. *Water Resour. Bull.* **19**: 59-67.

Vitt, D. H., L. A. Halsey and S. C. Zoltai (1994). The bog landforms of continental western Canada in relation to climate and permafrost patterns. *Arctic Alpine Res.* **26**(1): 1-13.

Weber, M. G. and S. W. Taylor (1992). The use of perscribed fire in the management of Canada's forested lands. *For. Chron.* **68**: 324-334.

Woo, M.-K. and T. C. Winter (1993). The role of permafrost and seasonal frost in the hydrology of northern wetlands in North America. *J. Hydrol.* **141**: 5-31.

Wright, R. F. (1976). The impact of forest fire on the nutrient influxes to small lakes in northeastern Minnesota. *Ecology* **57**: 649-663.

Chapter 7:
Insights for watershed management from Indigenous knowledge.

This thesis has focused on science, ostensibly to inform the management of forest resources. One limitation of experimental ecology is that, in most cases, its temporal perspective is limited. A glimpse of historic conditions can be obtained from the knowledge of locals such as Indigenous people familiar with traditional subsistence activities or non-Indigenous trappers and agriculturalists. A second limitation in ecological science is in its application to management process. The application of science-based knowledge in sustainable development requires a human perspective. My research occurred in part on the traditional lands of the Little Red River Cree and Tallcree First Nations (LRRTC), and understanding LRRTC Indigenous knowledge was instrumental to delineating both a historic perspective of local ecology and human values for sustainable management.

Science follows recognized and regimented methodology with resultant data "types" that scientists are comfortable interpreting. Collecting and analyzing Indigenous knowledge (IK) may not be as straightforward for ecologists who are unfamiliar with social science methods and jargon. The difficulties in conceptualizing and approaching IK and facilitating consultation with First Nations has kept most scientists from utilizing a potentially large knowledge base held by Indigenous people. In this chapter I suggest a model for moving from consultation as data gathering to one of knowledge exchange that can be complementary with science. The model should help elevate consultation from an onerous policy requirement to an effective tool that marries Indigenous knowledge with science. Examples are provided in this chapter demonstrating how a symbiosis between Indigenous and scientific knowledge can indicate ecosystem sensitivity to disturbance and reasons for that sensitivity which are followed with recommendations on silviculture techniques that are appropriate to reduce shrub growth and enhance regeneration. From a First Nation perspective, my study contributed to mutual scientific and Aboriginal exchange and provided a model for coaugmenting Indigenous and scientific knowledge. From a scientific perspective, the study contributed empirical theory deduced from long-term observation (data) and familiarity with the local ecosystem. The First Nations also provided watershed management concepts and helped identify what should be managed.

Twenty-one members from the LRRTC contributed to the study through interviews and facilitation of participatory learning in trapping and hunting activities. In this study, Indigenous knowledge was approached as a deductive process that could be formulated as empirical theory. I construct a framework for a common ideological landscape between IK and scientific knowledge by equating assessment of context with assessment of residuals in empirical theory.

Initial analysis indicated consistent themes among LRRTC interviewees that were analogous to scientific theories. These theories linked water quality impacts of forestry to debris dams in streams, subsequent stagnation and poor drainage. The impacts of forestry on hydrology was a small component of knowledge shared during this study. The non-hydrologic impacts of forestry activities on flora and fauna were due to excessive shrub growth and poor forest regeneration. Impacts of forestry activities on most animals were considered negative. Impacts on other animals such as moose were variable, depending on shrub-growth densities following clear-cutting. In some cases, negative impacts were due to increased difficulty in accessing fauna, which constituted a change in the Aboriginal relationship with fauna and not necessarily a change in the fauna themselves. As with any empirically supported theory, the residuals (data differing from the mean mathematical expression) provided insight about the intricacies of ecosystem function. The specific hydrologic setting with which a particular informant was familiar was the context of that person's knowledge, and collectively, the knowledge of the 21 informants provided a more detailed and comprehensive understanding of the variable interactions among landscape, hydrology, biota and humans in their ecosystem. Once context was addressed, ecosystem linkages of vegetation with hydrology and animal behavior, as understood by LRRTC, were resolved.

Why is Indigenous knowledge avoided by scientists?

Incorporating observations about the natural environment collected outside scientific method is generally discouraged in ecological theses. Polemics that scientific methods provide the only acceptable pathway to informed knowledge have sparked contentious debates and publications (e.g., Labinger and Collins 2001). These debates provide an interesting philosophical analysis of science, knowledge and dogma that should be required reading for scientists. Indigenous knowledge is one example where pundits both from within ecology and anthropology criticize its collection and use in ecological science (e.g. Cruikshank 1998, Howard & Widdowson 1996). Arguments against including IK in a science-based thesis derive from a recognition that ecological components of that knowledge are a small part of Aboriginal people's social framework (Stevenson 1996). The knowledge includes religious or spiritual components and makes no pretense at "objectivity." Defining cultural and spiritual components of IK and winnowing these from objective ecological observation may be an intractable problem (Heuscher 1979) or even destructive to the integrity and meaning of the knowledge (Cruikshank 1998). Other oft-cited reasons ecologists refrain from including IK are difficulties understanding the process and relevance of its transmission among Aboriginal people (Ohmagari and Berkes 1997), the context in which IK has meaning (Johnson 1996), the biases of researchers who document IK (Stevenson 1998), and overreaching or tendentious expectations (Noland and Gallagher 1981). Despite these drawbacks, many Aboriginal people are favorably inclined to share their knowledge with the scientific community, and inclusion of IK in management policy may become common throughout Canada, as it is now throughout the Northwest Territories (GNWT 1991).

The complexity of negotiating a knowledge system that is unfamiliar to ecologists is a common theme for the failure to use Indigenous knowledge in ecological investigations. These obstacles are made more labyrinthine for ecologists by the additionally unfamiliar protocols through which most social scientists approach the transfer of knowledge and the decorum required in cultural study. A step forward for ecologists was provided by Stevenson (1996), where the complex interrelationships between ecological, spiritual and cultural components of knowledge are clearly defined within a framework of Indigenous knowledge. Indigenous knowledge contains both traditional and nontraditional components, the former being handed down from one generation to another (GNWT 1991) and the latter being a cynosure between tradition and western scientific thinking through individual exposure to western science. A focus on IK as a whole rather than components such as "traditional ecological knowledge" can release the ecologist from the need to navigate the complex social constructs that glue various forms of knowledge together under the umbrella of IK. Another benefit of approaching the knowledge system as a whole rather than mining certain components is that this wholistic approach reduces the potential for appropriation of IK by ecologists (Stevenson 1996). Stevenson further argued that all IK could be considered contemporary because its meaning and value are defined by a current frame of reference which is in agreement with definitions of oral narrative (Morrow and Schneider 1995). While the approach suggested by Stevenson (1996) removes complexity by promoting general searches for meaning from the syntheses that is IK, an understanding of how IK provides ecosystem knowledge is required to effectively communicate meaning to people immersed in science. A model is developed in the following text that demonstrates the complementary nature of traditional knowledge and nontraditional or contemporary knowledge, their place within IK as a whole and how IK can be assessed in a framework relevant to ecologists.

Although it may seem a violation of scientific method, failure of IK to satisfy requirements of objectivity should not hinder its inclusion in ecological science. To address why, I begin by outlining the empirical approach of science and develop the thesis that Indigenous knowledge shares many similarities with scientific investigation. Science relies on the telamon of observable facts void of intellectual meaning. A scientist might begin by exploring facts or by forming a hypothesis (Fig. 7-1b). Once facts are combined with a hypothesis they are interpreted and imbued with meaning beyond the actual observations, a combination which is no longer fact (Rigler and Peters 1995). They are no longer facts because the interpretation has qualitative characteristics that may be erroneous. The facts are better viewed now as outcomes within a hypothetical construct or model. As well, observations may be an inaccurate representation of fact due to a fundamental or intractable bias (Hume 1776). The correspondence between observation and fact notwithstanding, the combination of fact and hypothesis becomes theory. A common misconception is that scientific conclusions or theories are facts when it is more appropriate to view them as concepts based on factual information. This distinction between fact and the constructs science creates based on fact is essential for subsequent discussion below where I outline similarities between scientific models and IK.

Science is about hypothesis and theory. Some theoretical sciences such as quantum physics are armed with a paucity of facts. Nevertheless, facts are an essential ingredient in science because they are the measure of accuracy to the degree they can be quantified. In physics, limited fact-scaffolds on which to hang theory are generated at arrant expense (e.g., super-colliders) but which are essential, despite the lamentations of those engaged in the study of scientific knowledge (e.g., Franklin 1995). In Figure 7-1c, a theory that has dominated aquatic science is the relationship between algal biomass and phosphorus concentration. The theory traces its history through many scientists, particularly in the 1960's and 70's (e.g., Edmondson 1969, Schindler 1973) but was formalized by Dillon and Rigler in 1974. A weakness with this theory is that for any given phosphorus concentration there is a wide range in possible chlorophyll concentrations (Fig. 7-1c). Thirty years of research has elucidated why the range exists, but an all-encompassing theory remains elusive. Instead, aquatic scientists incorporate a familiarity with a particular lake or group of lakes to explain deviations from the phosphorus-chlorophyll theory. For example, lakes with high concentrations of phosphorus may become nitrogen-limited and thus have lower algal biomass than predicted by the theory. The incorporation of detailed knowledge about a specific system to modify an accepted theory is common in science. The logical positivists criticize scientists for modifying theory with "unique conditions" as a method ensuring dogmatic theories are not rebuked (Kuhn 1996). This myopic view of science fails to focus on the possibility that invoking unique conditions produces testable hypotheses that can contribute to scientific progress. Whether a strength or weakness of scientific investigation, modifying information is similar to the "context" of IK. Context is defined as the location, culture, and even social position of the Aboriginal informants that lead to individual interpretations of an Aboriginal person's world (Cruikshank 1981). A strength of IK is in context assessment, where individual differences in observations become the unique conditions or hypotheses and where marriage of IK and science can occur.

The ecologist can view IK as a collection of hypotheses supported by facts unknown to the ecologist, similar to the phosphorus-chlorophyll hypothesis in Fig. 7-1b. A parallel construct for IK is presented in Figure 2 with the hypothesis that deliberate burning improves grazing habitat for ungulates. Facts have been compiled (e.g., Larsen and MacDonald 1998) that quantitatively support the hypothesis originally proposed through the study of IK (e.g., Lewis 1989) because natural scientists are more comfortable with quantitative methodology (Schwarz and Wein 1997). However, ecosystems respond differently, sometimes demonstrating no benefit from burning (e.g., Larsen and MacDonald 1998). How does one access the requisite information to elucidate the range of potential response to burning? The answer lies in IK, that knowledge specific to a particular ecosystem that has evolved through generations of observing ecosystem responses to burnt landscapes.

Indigenous knowledge should be approached at two levels: first, as general theory, and second, as context assessment. Like empirical models, IK theories can be generally interpreted by an ecologist as a representation of mean conditions simplified from complex and detailed information. Whether the complex underpinnings of IK are known or understood may be immaterial if the ecologist accepts stochastic uncertainty. The second approach is in assessing why systems respond differently from general theory through assessment of the context in which the IK derived. Contextual and individual understanding can be equated with the inherent ecological variation of a region; close examination should reveal complex underpinnings that can help modify general theory to reflect processes in specific ecosystems more accurately.

The inability to assign a number representing validity or certainty of IK may cause discomfort for ecologists. The model analogy is a useful tool for defining the power of IK. Scientists rely on the statistical power of an experiment to determine sampling and information levels required. In a power analysis the relationship between sample size and probability of statistical significance is explored. Figure 7-3 illustrates how the analogy can be applied to IK and helps define types of knowledge. Shared information may contain general and specific knowledge which may have been transmitted from past generations -traditional knowledge- or acquired through recent experience.

Traditional knowledge is transferred from past generations and is therefore valuable in providing understanding and sense of place within the broad ecological or social purlieu in which it evolved (Heuscher 1979, Ohmagari and Birkes 1997). When assessing responses from two people, considerable overlap in their shared knowledge may be noted, but as one includes more people, the overlap in shared knowledge will be increasingly restricted and the likelihood of finding generally relevant information will increase (Fig. 7-3). Knowledge that has survived generations of transfer is more likely applicable to general ecosystem interactions that are consistent across space and time and is therefore relevant to and shared to a greater degree among members of the community. Following this logic, consistent themes detected in IK represent general ecosystem relationships, or theory, the power of which can be quantified by both the number of informants where it is detected and the variability in caveats or context that modify an individuals knowledge relative to the group. The variability pertains to the modification of knowledge an individual has made based on personal experience or context.

Recent knowledge in IK derives from the traditional in much the same way that science progresses from previously published work. For the individual, recent IK is simply a personal evolution of traditional IK (Morrow and Schneider 1995). Recent knowledge is therefore tied to general empirical theories found in IK but may differ because it is specific to a certain ecological context (location). This information may be less widely shared but is equally important because it is through the differences in response that the intricacies of ecosystems can be examined (Rigler and Peters 1995). The distinction between recent and traditional can thus derive from a recognition that ecosystems change within the time scale of "traditional." In this thesis, the distinction between traditional and recent knowledge is a product of first identifying IK theory, the general ecological patterns shared across informants, from the variability in response which is individual adaptation to a local setting. Not only does this logic provide a model for assessing the power of information, it removes the need to separate traditional knowledge from what could be viewed as the nebulous concept of IK. The ecological context in which the knowledge of an individual was created, whether from recent observation or modification of traditional ideas to the local setting, provided the greatest insights.

How Indigenous knowledge can inform watershed management

Industrial development is often at odds with subsistence use of resources. Northern Alberta is no exception, where co-management of resources between the provincial government, industry and Aboriginal communities is an often desired but unattained goal amongst often fractious or competing parties. The LRRTC are building a foundation of mutual understanding and respect through the creation of partnerships in order to facilitate co-management practices (Sewapagaham 1998). The purpose of this study was to develop a watershed management model for forestry in dialogue with the LRRTC in order to incorporate their perspectives of disturbance impacts on ecology linked to surface water. The primary objective, stated by the LRRTC representatives, was to educate scientists, in this case me, in the values of LRRTC traditional lifestyles and linkages to the landscape. The inclusion of Indigenous knowledge in the watershed management model was an aspiration they recognized might not occur over the course of the project but was perceived as an ancillary benefit. From a science perspective, I expected that IK would provide additional information about hydrology in the study region. Interviews were conducted with individuals from the LRRTC to gather empirical "data" in order to identify relationships between water and the local climate, vegetation, and animal behavior; to identify perceived changes in water quality and quantity due to natural and anthropogenic influences; and to identify important features and values pertinent to LRRTC use of land-water resources.

Methods

Partnerships in the project

The LRRTC are partners in the Sustainable Forest Management Network (SFM), a research organization with academic, industry and governmental participants. Through the SFM, student research projects were funded to investigate the ecological characteristics of LRRTC traditional lands. A majority of projects focussed on aquatic ecosystems, and it was recognized that IK could contribute to these studies. Despite the relevance of IK to the other facets of aquatic ecology that were studied (e.g., fish diversity, habitat, health and the occurrence of fish parasites), this water quality and hydrology project was the only one to attempt reconciliation with IK. Informants from LRRTC seemed open to sharing their knowledge, possibly because this unique attempt at including IK demonstrated respect for local knowledge.

The research proceeded under the broad umbrella of formal agreements between LRRTC and the SFM. A separate written agreement for the specific research in this thesis was considered unnecessary. Broad goals were verbally communicated during a meeting with Chief Johnson Sewapagaham of the Little Red River Cree Nation. The result of this discussion was described in the goals presented above and were captured in a methodology summary submitted to the LRRTC representative Celestan Nanooch prior to initiation of the project. On reflection, reliance on verbal agreements for research specifics was in keeping with the oral tradition in which LRRTC community leaders were comfortable. Adherence to LRRTC goals was facilitated by Celestan Nanooch who, as coordinator of student researchers, had an understanding and experience in ensuring projects met LRRTC needs. Celestan played a crucial bridging role between scientific and Indigenous knowledge. His familiarity with both knowledge systems made it possible to translate understanding as well as language between the general and specific components of both ecological science and IK. Celestan translated meaning between myself and the informants by recognizing our separate interpretations and values that were implied and otherwise not effectively communicated.

Data were collected in two settings: unstructured interviews with key informants and elders, and participant observation while engaged in trapping activities. The study was formally conducted in partnership with the Little Red River Cree First Nation, whose representative, Celestan Nanooch, consulted with the First Nation council, and the elders, as well as arranged consent and interaction with informants. Only one interview was held in Tallcree; the remaining interviews and my field work occurred with members of Little Red River Cree.

Unstructured interviews were held with 21 individuals facilitated by one interpreter (Celestan Nanooch). I was the sole interviewer. The majority of informants were older than 50, and some were community elders. The informants either volunteered or were solicited by Celestan. Elders were esteemed by the community for living by the edicts of traditional culture and for being politically active leaders, educators, and role models. Five informants were younger than 35. One was a woman. The younger informants were selected because of their familiarity with university research and LRRTC co-management objectives. Each informant was acknowledged by the community to be an expert in bush life. Each interview began with an offering of tobacco to the informant, which is a traditional Cree protocol with multifaceted meaning. With Celestan's help, I explained the water quality study, its purposes, objectives, and those of the Indigenous knowledge component. The First Nation provided honoraria to the informants as a token payment of respect from project funds jointly contributed by the Canadian Circumpolar Institute and the Little Red River Cree First Nation. Informants were asked to introduce themselves and describe their relationship with the land. Although informants were guided towards water quality and quantity issues, they gravitated toward their own points of interest, related primarily to fishing, trapping, or hunting in traditional family areas. Most elders described the lives of their parents and themselves from childhood to the present. Some informants described the lifestyle of their grandparents' childhood, referring to that period as "old-times." The majority of information was limited to the 20^{th} century.

Informants spoke about the land in specific terms, sufficient to indicate to which components of the landscape they were referring. Mutually familiar landmarks were used to specify the geographic features and locations contained in their descriptions of the landscape. For example, "below the Caribou Mountains where it is wet" refers specifically to a northern portion of the Peace River lowlands where the break in slope results in poor drainage and a hydrologic discharge zone, but it does not include the entire lowlands. It can be further inferred that the region encompasses the area north and east of John D' Or Prairie where poor drainage has been a historic problem for forest regeneration. Topographic maps of the region were referenced primarily for locations of traplines and in some cases to identify specific landscape features. Table 7-1 indicates the topics covered in question format. However, each informant addressed individual topics according to his/her predilections.

Participant observation with seven of the informants, all of whom were over 50 years old, involved hunting, trapping and fishing over several days. We were periodically accompanied by young adults engaged in hunting and training activities, thus affording me a similitude of the hands-on learning techniques informants customarily use to transmit traditional knowledge. Obvious differences from traditional learning were that our encounters occurred in English and demonstrated as much as possible in a limited time.

Responses to questions were categorized into the general themes in Table 7-1 and assigned one of four possible response codes: 1) positive; 2) negative; 3) neutral; and 4) no response or unsure. Nonparametric *t*-tests were applied to each theme in order to highlight overlap or shared knowledge and thus those with the greatest significance (as in Fig. 7-3). In keeping with the model developed for this research, variable response was a presumed result of individual experience (context), and it was from examining possible reasons for differences in response that the intricacies of hydrologic function and forest regeneration emerged. Discussions arising from each interview were assessed individually within the context of the entire interaction. Details from participant observation were essential because they were linked to an interview's specific area, time, person and experience.

Relative terms such as 'dense brush' had various meanings when defined by informants in an operational context. For example, dense brush was not only thicker than normal but could be so thick as to hinder animal and human movement, as well as growth of the brush itself. Definition of these terms was determined for each individual with additional questions relating their understanding to physical locations with which we were both familiar.

Physical setting

The traditional lands of the LRRTC are shared somewhat between the two First Nations. They include the Caribou Mountains (59°N 115°W), the Peace River lowlands and extend 100 km south of the Peace River. People live in five communities. For Little Red River Cree, they are Garden River in Wood Buffalo National Park (WBNP), Fox Lake, and John D' Or Prairie. The members of Tallcree First Nation are divided between North and South Tallcree. The communities of Fort Vermilion and Beaver Ranch fall within the LRRTC environs (Fig. 7-4). Many of the traplines extending into the Caribou Mountains involve partnerships between members from Little Red River and Tallcree First Nations.

The Caribou Mountains are an erosional remnant forming a large, relatively flat plateau 500 m above the Peace River valley. Peatlands cover 56% of the Caribou Mountains (Strong and Leggat 1992). Open forest of black spruce and an understory of mixed *Sphagnum spp.*, feathermosses, brown mosses and lichens dominate vegetation in the Caribou Mountains. The Peace River lowlands are mixed wood forest of poplar, aspen, pine, white and black spruce. Forest fire is a pervasive disturbance structuring the LRRTC ecosystem. Two major fires are of particular importance to the communities in John D'Or Prairie: one burned the lowlands from Beaver Ranch east to Garden River (c.a. 1950), and the other burnt 129 000 ha on the top of the Caribou Plateau in 1995. The former fire changed the lowland forests from spruce-dominant to the mixed-wood forest of today. The latter fire impacted terrestrial and aquatic resources such that a long recovery period is expected; lakes became more eutrophic, a generally negative condition in the health of an aquatic ecosystem, and impacts prevailed from fires dating back 35 years (McEachern et al. 2000). The region borders the Sub-Arctic, and the Caribou Mountains are underlain by discontinuous permafrost, with the Peace River lowlands being warm enough to support agriculture. Trapping is a winter activity which begins when furs thicken and become plush, when water bodies freeze sufficiently for safe transport, and when government regulations allow trapping of fur-bearing species.

Results and Discussion

I have separated results from the IK study into two parts: 1) a description of LRRTC relationships with the ecosystem; 2) the LRRTC's IK concerning hydrology. Although the division is artificial, it helps organize the information into an LRRTC ecosystem context prior to an assessment of hydrology and watershed management.

Relationship with aquatic systems and animals

Aquatic systems are vital to the LRRTC way of life. Streams provide drinking water and habitat for animals, particularly beaver, on which the LRRTC subsist. Lakes and rivers are used extensively for fishing and provide habitats for the moose and caribou upon which LRTTC also rely. During winter, lakes are vital not only for ice-fishing but also for hunting, because they facilitate access to caribou which preferentially cross lake ice in order to avoid predation by wolves. Wetland areas along lake and pond shores provide habitat for muskrat, another essential food and income source for LRRTC trappers.

The pursuit of fur-bearing species by trapping or hunting occurs in winter when fur quality and access through the frozen landscape are maximal. Prior to the last two or three decades, dog teams remained the primary mode of winter bush travel. Packhorses and walking were the only mode of bush travel during summer months. Frozen streams facilitated winter travel prior to the skidoo, the adoption of which coincided with extensive seismic surveys in northern Alberta. Because skidoos require wider trails than dog teams, seismic lines and rivers are now the preferred routes for winter transport rather than stream channels. Concomitantly, all-terrain vehicles (ATV's) have replaced horses during summer but are limited in their use in wetland areas, particularly in the Caribou Mountains.

Prior to the introduction of a trapline registry by the Province of Alberta and a group system in Wood Buffalo National Park in the 1940s, traplines were associated with families. Association with a family did not imply ownership; anyone from the LRRTC community could use a trapping trail and trapline. However, courtesy dictated that the recognized family associated with a specific trapline be informed by other users to ensure coherent management:

> A trapper controlled the trail that he cut. Others could trap his area but asked permission because the local trapper was considered the authority on how much trapping could occur [on the trappers line] [Alexander Nanooch, LRRTC].

Neither streams nor watersheds defined boundaries of family resource areas because the communal philosophy did not recognize ownership of trails or resources. All 15 informants who discussed boundaries confirmed that, other than expediting winter travel and trapping, streams were no more significant than any other feature of the landscape. The concept of boundaries, both social and hydrologic (watershed), did not appear to be an issue for LRRTC members. Although boundaries with other First Nations were not specifically defined along watersheds, the neighboring Dene Tha used the Hay River and its tributaries (Lea 1996) while the LRRTC remained within an area defined by the lower Peace River and its tributaries. Watersheds may have been important boundaries between the Cree and surrounding First Nations but do not appear to have been important within the two LRRTC First Nations.

Artificial boundaries were implemented with the registration of traplines partly through the responsibility of "ownership" and the formalization of individual trapping areas. Outside WBNP, control rested with the Alberta Government through a registration system. The registering system resulted in greater emphasis on asking permission, or becoming a partner and contributing to registration costs within the LRRTC community. Within WBNP, restricted access was tightly controlled by a federal process. Outside WBNP, either through reduced enforcement or looser rules, trappers seemed to have greater power in managing resources on their lines. The financial burden of registration may have also played a role in an activity that has recently become more culturally significant than financially viable. In essence, registration redounded to constraints on free choice and community cohesiveness embodied in traditional culture.

Watersheds are recognized as ecologically significant by the First Nations despite their irrelevance as social boundaries. Certain streams are recognized as having "better" water than others. Two such examples are the Lawrence River, the drinking water supply for John D'Or, and Wentzel River east of John D' Or:

> People in John D' Or think water from the Lawrence is not good. Water from the Wentzel River is sweet. We drive [30 km] to the Wentzel just to get water which we bring back in jugs. Often people camped at the Wentzel [Celestan Nanooch, LRRTC].

The LRRTC recognize differences between the Wentzel and Lawrence watersheds from the quality of water they produce. From a water chemistry perspective, the Wentzel River, although of similar ion composition, contained two-thirds the ion concentrations of the Lawrence River (McEachern unpubl. data). The softer, less turbid water of the Wentzel River likely contributes to improved taste. Informants offered no explanation for the difference in water quality between the two rivers; however, one informant noted that the

Wentzel basin contained more peatland in its upper reaches, and more sand and aspen forest at lower elevations than did the Lawrence basin. The latter is incised primarily through clay-draining spruce forests at the edge of the Caribou Plateau.

Drinking water preferences among LRRTC members are based on taste, turbidity and flow. Stagnant waters in lakes or streams choked with debris are not considered suitable sources. While turbid waters are disdained, the brown waters draining peatlands are regarded as suitable for drinking water (Lea 1996). Water quality concerns translate into preferred locations for bush camps and travel routes that are influenced by watershed boundaries.

The LRRTC did not alter aquatic systems in the past, nor do they seek to do so today. The most direct aquatic management occurs through their relationship with beavers. Beavers are prevalent in LRRTC lands and actively modify almost every stream system. LRRTC trappers determine the number of beavers that can be removed from particular sites by monitoring the age of individuals caught:

> A [beaver] lodge has three years of kids [kits] and their parents. There may be one or two adult females with one male. The kids are caught easy, the adults are smart. I try to catch some of the adults but if I catch lots of kids I move the traps and leave the adults [Paul Tallcree, LRRTC].

LRRTC trappers do not remove all the beaver from a lodge, ensuring rapid recruitment for future harvest. They also change their beaver trap lines regularly so that beaver populations are not overexploited in each stream.

Managing beaver has direct implications on local hydrology and vegetation. As beaver populations increase they migrate into tributaries and flood more areas, thus changing forests to wetlands. Maintaining the beaver population at a certain level maintains a hydrologic *status quo* in addition to healthy beaver populations.

> Beaver stay in areas where we trap. When we don't trap, beaver numbers increase and the beaver get sick and die. They come back eventually, but when we trap they are always there and they are healthier [Paul Tallcree, LRRTC].

LRRTC interaction with moose also depends on aquatic environments. Moose is a primary food source for the LRRTC community and the focus of most hunting efforts. Moose are hunted primarily in sloughs, swamps and meadows near lakes, along streams and in lowland areas where wetlands form. A majority of informants indicated a preference for hunting moose in aquatic environments rather than in open areas created by cut blocks or forest fire. Population size and access were cited as the primary reasons for this preference:

> There is so much debris after logging that moose stay out of cut blocks. In some cut blocks dense shrub growth occurs and makes it worse for moose and hunters. That's why we don't hunt in cut blocks [Louis Ladouceur, LRRTC].

Avoiding hunting in cut blocks or burnt areas did not imply that these areas were a long-term detriment to moose. A majority of respondents believed moose frequented cut blocks and burnt areas after two to five years. Two respondents noted that moose preferred cut blocks during winter. The relationship of moose to cut or burnt areas was controlled by shrub growth following a disturbance. The magnitude of shrub growth was geographically specific and informants related growth to site hydrology, understory foliage, surface soil, and magnitude of impact to surface soil. Some informants were familiar with areas where shrub growth was not excessive after harvest or fire and thus perceived fewer impacts on moose than did informants familiar with areas where dense shrub growth occurred after a disturbance. Several informants indicated that cut blocks and burnt areas were detrimental to all animals. These responses may have been focused on immediate impacts. All 15 informants agreed that impacts were negative over the first few years following forest fire or logging.

Paul Tallcree is closely connected to an area inside Wood Buffalo National Park and discussed at length the relationship between aquatic systems and the distribution of buffalo. The primary features for WBNP were springs, wetlands, fens and willow swamps, which he called "wet meadows". Wet meadows are extensive south of Garden River and are impacted by the hydrology of the Peace River. Buffalo have adapted to a network of wet meadows south of Garden River that have been maintained by springs and streams. Ancient buffalo trails are deeply carved into the local landscape by the vast numbers of animals that passed over these trails through history, and several critical crossroads inosculate where groundwater springs are observed. Paul Tallcree noted this association and was concerned that changes in hydrology might significantly alter buffalo behavior and distribution. Impacts from the WAC Bennett Dam were of immediate concern because the LRRTC have witnessed these impacts firsthand and are familiar with scientific literature on the Peace River. Climate change was indicated as a future concern. This elder noted a decline in wet meadows that he attributed to less water in the Peace River because of the WAC Bennett dam.

Watershed impacts and management

The LRRTC perceive impacts of fire and forestry through their interaction with animals and their familiarity with the local boreal landscape. An exploratory analysis was initially employed to discern common patterns in the IK data. The impacts of fire and forestry described by informants were combined and summarized in Table 7-2 in an attempt to define the overlapping region of shared IK identified in Figure 7-3. For scientists familiar with empirical approaches to ecological data, the presentation is similar to using correlation analyses to discover patterns and form hypotheses. A general hypothesis can be drawn from the summary in Table 7-2; only forestry adversely impacted water quantity (hydrology), whereas both fire and forestry negatively impacted water quality (chemistry).

The hypothesis can be formalized into a model like Figure 7-2 by proposing a relationship between water quality and disturbance (Fig. 7-5). The summary in Table 7-2 suggests that members of LRRTC believe clear-cutting causes a more precipitous decline in water quality with slower recovery than does fire. Figure 7-5 speaks a language familiar to empirical science; information provided by individuals has been reified and plotted as abstract deviations from the general theory represented by the fire and cut-lines. It is an abstraction because the residuals were not quantifiable and thus represent a qualitative difference perceived by the informants. Like nitrogen or light-limitation impacts on the chlorophyll - phosphorus relationship (Fig. 7-1), the qualitative differences in Figure 7-5 have reifiable ecological significance that can identify and elucidate landscape management practices which may succeed or fail in specific areas.

Water quality and quantity following clear-cuts and fire were large concerns for the LRRTC. All informants indicated that both clear-cuts and fire adversely impacted water quality. Disturbances were always described in relative terms that indicated impacts were generally greater and lasted longer from clear-cut than from fire. Most informants believed water quality returned to pre-disturbance conditions as brush recrudesced. Some indicated time frames for recovery: streams from burnt areas returned to normal within five years; clear-cut areas required a decade. The litmus test for "normal" was determined by whether or not informants would drink from a stream. Several informants believed water quality in burnt areas returned to pre-disturbance conditions within months, a hypothesis that has found support in scientific literature (Spencer and Hauer 1991).

All informants were concerned about a perceived decline in water quantity which they associated with changes in hydrology following WAC Bennett Dam construction. Several scientific reports under the Northern River Basins Study outline the hydrologic impacts of the Bennett Dam on the lower Peace River. These reports could have been the source of some informants' knowledge. However, informants identified locations where reduced ice thickness and increased island formation occurred in the Peace River, as well as reduced water flow in tributary streams which were observations consistent with changing hydrology. Ice thickness is a function of water level and ice dam formation during early winter. During spring melt, ice damming again induces the Peace River to flood its banks and recharge surface water systems in the Peace and Athabasca River deltas.

Several informants were concerned about the loss of sloughs related to the dam through a paucity of spring flooding. Scientific literature in the United States demonstrates that forest and biotic productivity on floodplains depends on seasonal flooding (Toner and Keddy 1997) which was also the conclusion of extensive research in the lower Peace River region (NRBS 1996). The observations of LRRTC members support the need for seasonal flooding in the Peace River, not only to recharge the hydrology of the well-studied Athabasca delta, but also to restore sloughs, swamps and tributary deltas between Fort Vermilion and Garden River.

It was suggested that declines in water quantity and loss of sloughs following forestry activities were the result of dense shrub growth after harvesting, particularly in the Garden River area. Two informants familiar with the lowlands below the Caribou Plateau reported a rise in water table after harvesting which redounded to tree morbidity and death, stream bank instability, and slow regeneration. Both informants witnessed dead trees in the riparian area subsequently collapsing into the stream channel and causing further flooding, due in part to faulty drainage after harvesting from the accumulated debris in stream channels and cut-blocks.

Views on forest regeneration following harvesting were the most diverse because an individual's knowledge was influenced by the disparate physical conditions and subsequent forestry treatments in each trapping area as well as his/her unique forestry experience. When proffering recent knowledge within a specific geographical context, responses were understandably more variable (Fig. 7-3). However, all informants described a general pattern of grasses for the first two years, followed by aspen, alder and other varieties of brush. Informants were generally unhappy with any perceived increase in brush densities, because dense brush inhibits the mobility of larger game animals. Brush growth following clear-cut or fire ranged from "dense with no impacts" to "detrimentally dense". In some areas clear-cuts and fires produced equally dense brush growth, while in other areas they did not. It was noted that brush growth was not prodigious after intense ground fires. Nor was it abundant after harvesting with certain scarification techniques:

> Sometimes fire burns down to the mineral soil. Brush growth is not as dense then and sometimes we see new trees like pine will come back in those areas [Charlie Hamelin, LRRTC].

Reduced brush growth after clear-cutting followed scarification by martini plow and extensive dumping of debris on the site. The martini plow is a scarification technique that simultaneously rips and turns soil by dragging a plow blade behind a bulldozer. Despite consensus that the martini plow reduced problematic shrub growth, half the informants opined that the martini plow caused excessive damage to the landscape.

Forest regeneration through "straight planting" on sites that have not been scarified is becoming increasingly popular but has met with ambiguous success in this region (TOLKO Woodlands Managers, Chemago and Avery pers. com.). Indigenous knowledge can contribute insights to this debate. One informant suggested reduced shrub growth resulted if no scarification treatments were employed:

> Shrub growth is less dense in areas where the moss layer remains intact [Louis Ladouceur, LRRTC].

This observation may have been due to the particular soil conditions at these sites along the Peace River in WBNP, an area where vegetation composition in previously burnt areas was correlated to depth and aerial extent of moss (Timoney et al. 1997).

Informants that supported martini scarification had experience in the forestry industry and were familiar with experiments where different scarification techniques had been applied between John D' Or and Garden River. An equal number of informants abjured martini scarification because the disturbance was viewed as excessive. Some believed limited scarification followed by brush thinning was a better management practice for clear-cuts. Though cited as a potential benefit for reduced shrub growth, respondents did not consider extensive debris from de-limbing a viable management alternative. All respondents believed debris negatively influenced drainage, water quality and wildlife.

Reforestation was considered essential for reducing long-term negative impacts. Beneficial management practices included scarification, no scarification, brush thinning, and a number of harvesting techniques that leave forest reserves in tact. Preferred harvesting techniques included selective harvesting by machine or hand, patch harvesting, and maintenance of larger reserve areas within clear-cuts. Although the consensus was to minimize harvesting impacts, most informants recognized that the local economic future would depend on harvesting activities. All informants were convinced that management would improve when LRRTC gained control over all harvesting activities. Some imponed that changes in management practice would be minimal but would, at the least, allow valued landscape features such as historic sites to be appraised in management plans without unwarranted obtrusion.

Impacts of forest disturbance on most species of animals were considered to be negative, except for impacts on moose, deer and rabbit, which were variable through time and space. Squirrels are a predominant issue because they provide a primary food resource for fur-bearing species, such as marten and are hunted by women and children around their settlements and bush camps. Several informants noted that timber harvesting and trapping compete for the same resource, that of spruce trees, which provide sanctuary for squirrels. Others suggested that a loss of squirrels was not necessarily detrimental to trapping. Several informants indicated that mice and rabbits, which represent a rich food resource for marten, lynx, and foxes, might actually benefit from cut-blocks. Though not desirable, the potential benefit of clear-cuts for rabbits is that trappers relocated to the edge of a clear-cut rather than the forest whenever a clear-cut impacted their trapline. Most informants agreed that management options should factor squirrels into the management equation. One suggestion proffered was to determine if a reduced quantity of coniferous cones were noted and quantified during any given summer which should translate into reduced harvesting the following winter in order to reduce stress on squirrels. The majority of respondents believed larger reserves on higher ground should be left in squirrel habitat because squirrels locate their warrens at elevations well above the riparian zone in order to avoid flooding.

The informants believed that scarification treatments should depend on IK input. Considerations of site hydrology were the biggest component where they felt their knowledge could improve regeneration success. It was generally felt that wet sites, or sites likely to become wet due to harvesting impacts, should not be scarified. Instead these sites should be straight-planted. Informants agreed that some form of scarification reduced shrub growth on upland sites but generally felt less invasive forms such as mounding were better than the deep trenches of martini plows. They felt that through consultation their knowledge could help determine which sites were likely to become wet, which sites should or should not be scarified, what trees should be planted and what post-planting treatments should be implemented.

Aboriginal communities throughout the world have used fire to modify their landscape (Lewis 1982). Northern Alberta is no exception; extensive burning on agricultural land has improved meadows and fireproofed villages and trapping areas (Lewis 1978, Lewis and Ferguson 1988). These prescribed fires could influence water quality if they were extensive or near stream systems. Although all respondents abjured deliberate burning near their traplines or in forest systems because the LRRTC trap for forest-dependant animals adversely impacted by fire, respondents confirmed that LRRTC burned in agricultural areas. In Fox Lake, for example, meadows were burnt in spring. The practice of burning fields subsided as the LRRTC's reliance on horses declined. When asked if prairie was more extensive in the past, most informants alleged there had been no change. The notion that their progenitors may have used fire to increase grasslands for ungulate habitat was viewed with scepticism because, as they pointed out, moose and buffalo in the LRRTC area prefer interconnected wetlands and meadows in woodland habitat, and that after a fire, subsequent willow, alder and poplar brush densities are detrimental to most forest animals. Moreover, there has been recent evidence that fire neither improves grassland in WBNP nor discourages shrub growth. In fact, it promotes the growth of less edible grass species (Larsen and MacDonald 1998, Timoney 1999).

Other reports indicate that trappers used fire in forest areas prior to the 20th Century, and that prairie may have been more extensive in the Peace-Athabasca Region (Lea 1996). Burning appears more prevalent in Tallcree communities and more commonly employed south of the Peace River. With certitude, conflagrations erupted regularly, many of which may have been sparked by Aboriginal people. The question remains whether burning of forest was a deliberate attempt to increase grassland habitat or whether the fires were accidental. Evidence supporting the former exists for Aboriginal societies in the Peace River area (Lewis 1982) but is contradicted by recent LRRTC knowledge, indicating a preeminence of colonizing shrubs not edible grasses follow fire in the LRRTC area. That large fires in forest areas were primarily accidental in recent decades of fire suppression can be supported by a multitude of examples, such as the 1950 conflagration that roared from Beaver Ranch to Garden River. The common practice of leaving campfires smoldering in expectation of returning to the site may be an additional causative factor. An 840-year paleolimnological record from WBNP suggests fire frequency in the region was consistent with natural occurrence between 95 and 185 years (Larsen and MacDonald 1998), and did not increase until European influence after 1860 when fire frequency became every 25 to 49 years (Larsen 1997).

Summary

The resource manager who includes Indigenous knowledge must navigate criticisms that the knowledge: cannot be documented; cannot be understood by a Westerner; cannot be applied beyond the context of its source; was inadvertently misinterpreted by researchers placing it within a familiar context. The manager must also reconcile the cultural tradition of science with the culture of IK. When scientists can define and understand the boundaries within which scientific knowledge is created and how it represents our world, the same process should be applicable to other forms of knowing including Indigenous methods.

Logical arguments that scientific analysis subjugates IK by removing context and meaning are supported by examples of misinterpretation and misuse in resource management (e.g. Johnson 1996). However, the oft-tendentious values of IK can predispose its polemicists to assume an isolationist position similar to the *de haut en bas* apperceptions of science (e.g. Stevenson 1998, 2001). An intermediary position wherein IK is recognized as sharing similarities with other knowledge-based systems, including science methodologies, would be more productive and relevant, despite flaws in generalizing from IK to science. I propose a potential method of assessing IK within a language construct that makes the cynosure of IK palatable to scientific culture.

In this study, IK was assumed to contain extractable data derived empirically through LRRTC experience. Responses from the informants were reduced to positive, negative and neutral symbols to which statistics could be applied. The results were useful because they identified important management issues, ecosystem relationships and the following quantifiable impacts:

- The relationship of LRRTC with most animals is negatively impacted by forest disturbance.
- Water quality is a large and encompassing concern and is negatively impacted by forest disturbance.
- Water quantity and drainage disturbances following logging are primary concerns of the LRRTC.
- Logging externalities, such as changes in hydrology and dense shrub growth, are more deleterious in clear cuts than in firescapes.
- Clear cuts remove potential LRRTC hunting areas; particularly vulnerable are the ungulate habitats.

The basic ecosystem concerns identified through the data approach are summarized by Figures 7-5 and 7-6. Forestry and fire negatively impact hydrology and water quality because debris caused poor drainage. Subsequent shrub growth amplifies impacts on animals, such as moose and buffalo, and on LRRTC relationships with the landscape.

Models like the one proposed in Figure 7-5 generalize from facts (observations) but fail to explain them with precision. The context of IK data should be used to investigate the basis for different responses to similar themes in the same manner we search for ecological factors to explain model residuals. Shared information and the diversity of responses by the LRRTC enabled a comprehensive tableau of ecosystem concepts to emerge which formed the basis for a deeper understanding of ecosystem linkages as summarized in Figure 7-6.

Differences in responses were commonplace. For example, equal numbers of respondents indicated cut blocks had no effect on moose, had positive impact on moose, or were detrimental to moose. These response differences when taken on their own were contradictory. However, when examined within the context, of individual observations, linkages could be drawn. The variation in response seemed particularly linked to how dense shrub growth became, or how soon edible grasses were replaced by shrubs following fire or logging. Negative impacts on beavers from forestry and fire were voiced by those who also indicated de-limbing debris entered streams, tree blow-down occurred, and beaver dams were dynamited during road construction. Those that did not perceive negative impacts on beaver did not share similar experiences of the landscape disturbance from forestry or were familiar with areas where regeneration after the disturbance was more favorable.

Hydrology was recognized as a primary force determining the complex landscape on LRRTC lands. Hydrology dictated the occurrence of sloughs, wet meadows, and peatlands. These vegetation features in turn controlled the distribution of animals from moose, buffalo and caribou to squirrels, small predators and the LRRTC. Forestry was believed to impact hydrology through changes in drainage and vegetation that further influenced water quality and quantity in watershed streams.

On a positive note, the impacts from fire and forestry were not considered irrevocable. Respondents agreed that aquatic systems recovered over time. However, the length of recovery depended on management after disturbance and several recommendations were made to improve recovery. Management recommendations for watersheds following forestry were:

- Improve drainage by limiting debris in stream channels and on cut blocks.
- Improve animal access to cut blocks by limiting debris.
- Reduce shrub growth through scarification and thinning methods.
- Choose appropriate scarification based on consultation with local LRRTC elders or trappers.
- Leave spruce reserves on hill slopes for squirrels.

Table 7-1: Guide for unstructured interviews.

- How are aquatic systems used today? Historically?
- How do/did water systems such as rivers or drainage basins define boundaries between user groups? Are water boundaries important to the individual, family, tribe?
- How do/did people manage their use of aquatic systems to maintain sustainable use of fish or beaver?
- How does a clear cut or fire effect:
    - Animal habitat, behavior, location and use by people?
    - Stream flow and water quality?
- Reforestation concerns:
    - What comes back after fire and after clear-cut, and how long does it take?
    - Does replanting help trees come back?
    - If not, why?
    - How can re-growth be improved?
- When Little Red takes over logging of the forest, what should they do differently?
- What concerns do you have about the forest and forest management?

Table 7-2: Summary of responses to the question of how forest fire and clear-cut harvesting impact animals and water quality. Forest fire and timber harvesting were grouped for this analysis because all respondents indicated that they had similar impacts but differed by degree. Differences between fire and logging impacts are addressed in the results. Zero denotes no perceived impact, -1 denotes a negative impact or no, and +1 denotes a positive impact or yes.

How does a fire or clear-cut impact:	*n*	Impact
Moose	15	0
Caribou	14	-1
Buffalo	16	-1
Deer	4	0
Beaver	15	-1
Marten	14	-1
Squirrel	14	-1
Rabbit	8	0
Water quality - fire	19	-1
Water quality - clear-cut	19	-1
Water quantity - fire	13	0
Water quantity - clear-cut	19	-1
Access	18	-1
Is c.c. worse than fire	17	+1
Do you hunt in clear-cut	12	-1
Do trappers set fires to manage brush	13	-1
Was there more prairie in the past	13	-1

Fig. 7-1: Some common components of a scientific approach in ecology.

Fig. 7-1a: Example data collected from lakes in Alberta. These are the facts of our scientific inquiry.

Lake	TP (μg/L)	Chl a (μg/L)
Cooking	274	158
Eden	16.2	9.2
Ethel	19	6.3
Halfmoon	95.8	46.2
Hasse	21.4	3.3

Fig. 7-1b: A hypothesis that algal biomass depends on the amount of limiting nutrient available. A component hypothesis is that phosphorus is the limiting nutrient. We may further hypothesize about the nature of the relationship by giving values to the function y = f(x) from an understanding of the cellular requirements of algae.

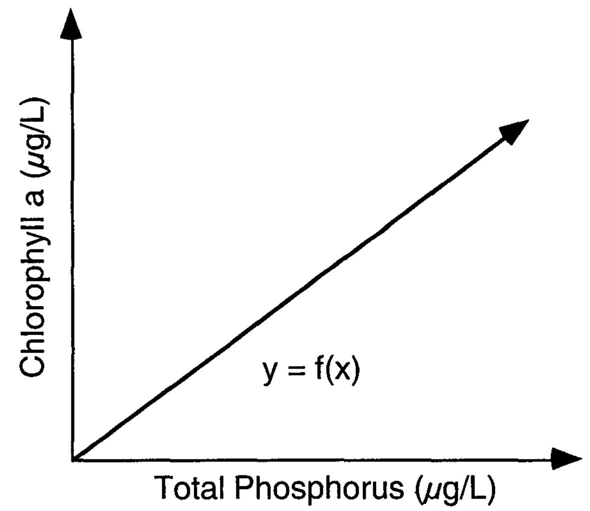
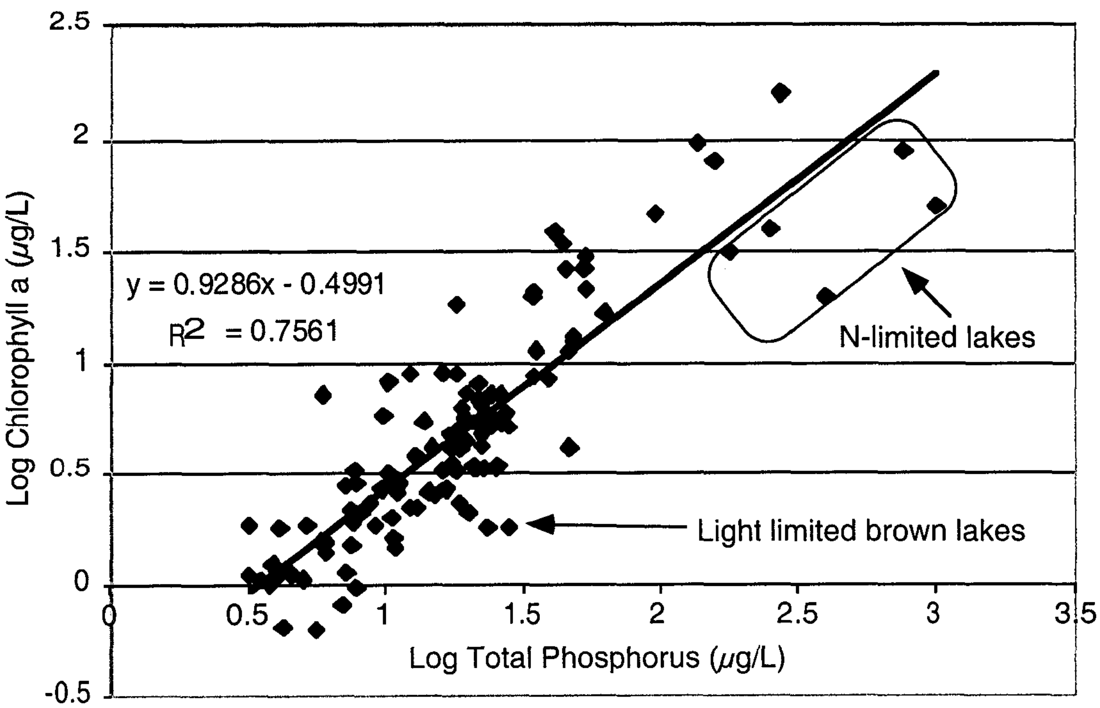


Fig. 7-1c: The theory that chlorophyll concentration depends on phosphorus concentration by a logarithmic relationship. Note the scatter of data. At a log TP of 1.5 the Chl a concentrations range from 0.25 to 1.25, a 10-fold difference! It is not a quantitatively accurate theory. Some of the reasons for inaccuracy can be determined by knowledge of specific lakes, such as those limited by nitrogen or light.



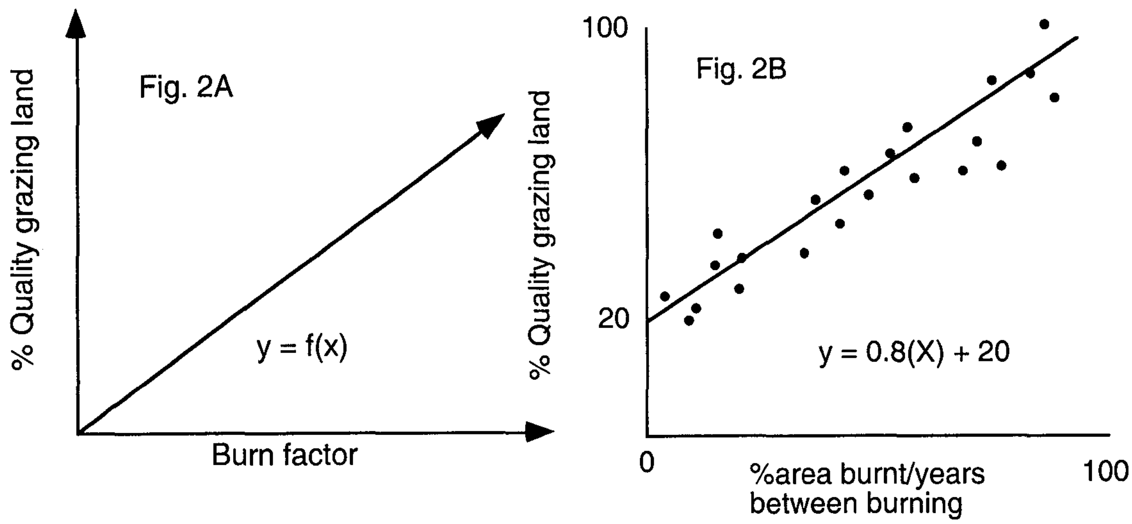


Fig. 7-2: A) An hypothesis that the percentage of quality grazing land depends on some expression of burning. **B)** With some data (facts) an ecologist may quantify the hypothesis. One "fact" might be that in unburnt areas an average of 20% of the area is quality grazing land. The ecologist might further purpose that the y-axis is related to the percentage of area burnt and the frequency with which it is burnt. For example a pasture that is 100% burnt every year will have 100% quality grazing land but a forest that is 20% burnt every 20 years will only have a 1% increase in quality grazing land over unburnt forests.

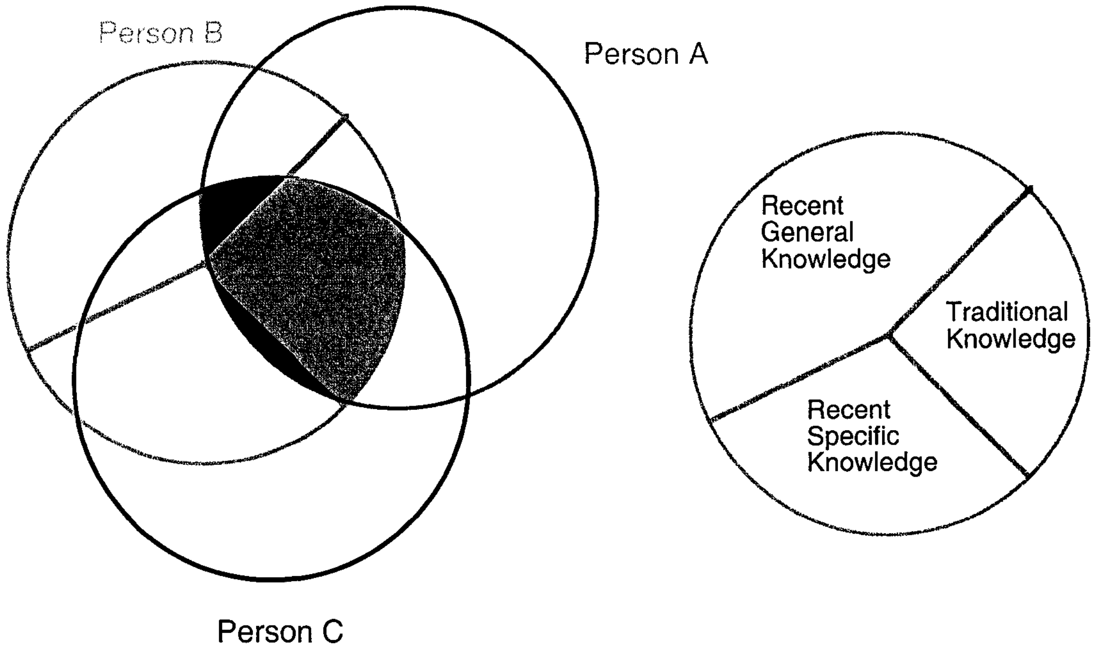
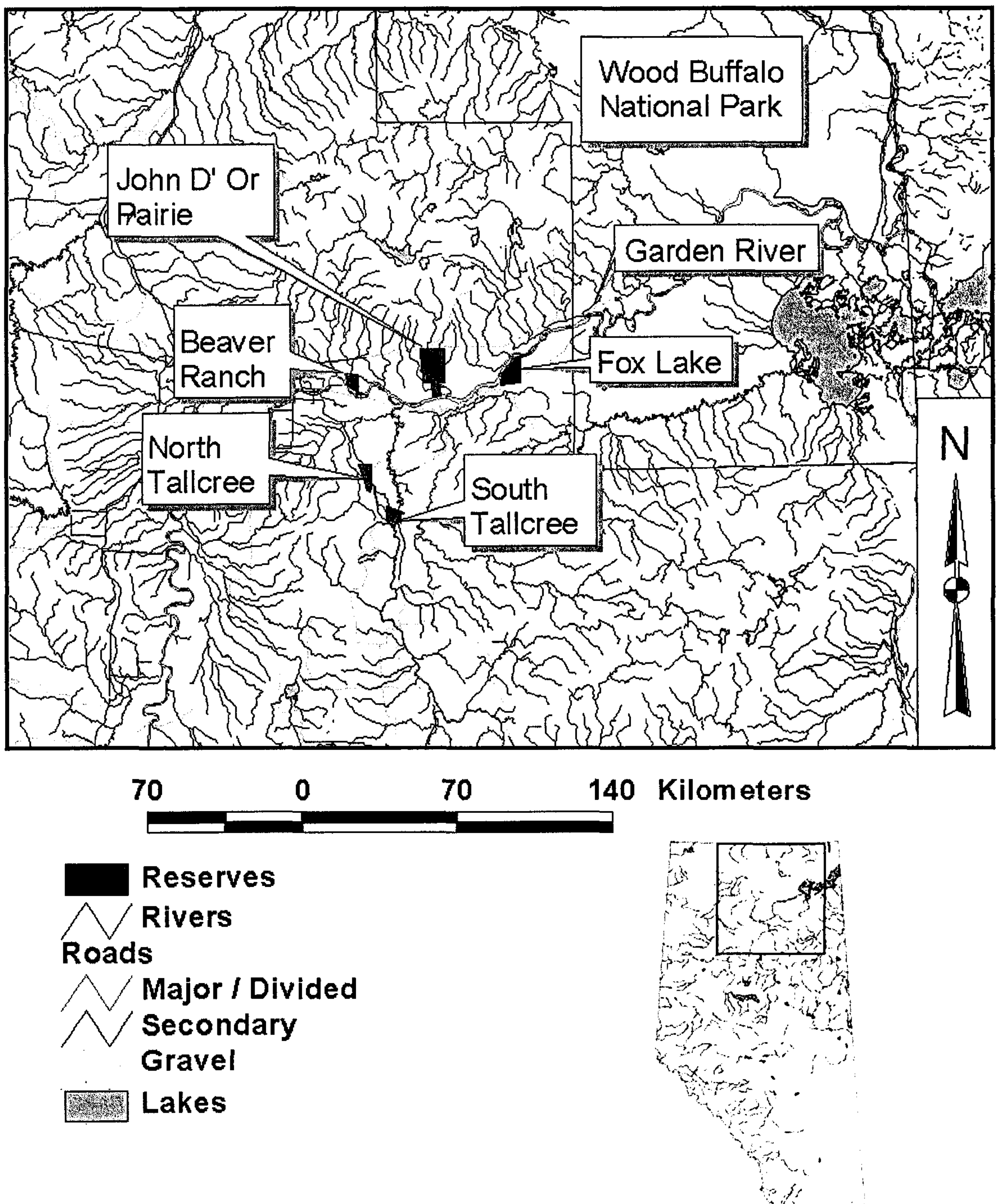


Fig. 7-3: The components of Indigenous knowledge and a hypothesis that the general applicability and power of knowledge is defined by the degree to which it is shared. Indigenous knowledge can contain traditional and recent components. Within recent components, some knowledge is likely to be specific, such as how a certain bear in a certain area behaves. Other recent knowledge may be general, such as how to operate a skidoo. As more people are interviewed, regions of overlapping knowledge become smaller and limited to more generally applicable and significant information. As knowledge from more informants is intersected [in this figure there are three] the reliability of the information to accurately represent ecosystem function should increase. The shaded areas are: green for shared traditional knowledge, blue for shared recent specific knowledge, and red for shared recent general knowledge.

Fig. 7-4: Northern Alberta and the traditional lands of the LRRTC. The lower Peace River, once a major fur trading route flows from southwest to northeast. The six communities that make up LRRTC are shown.



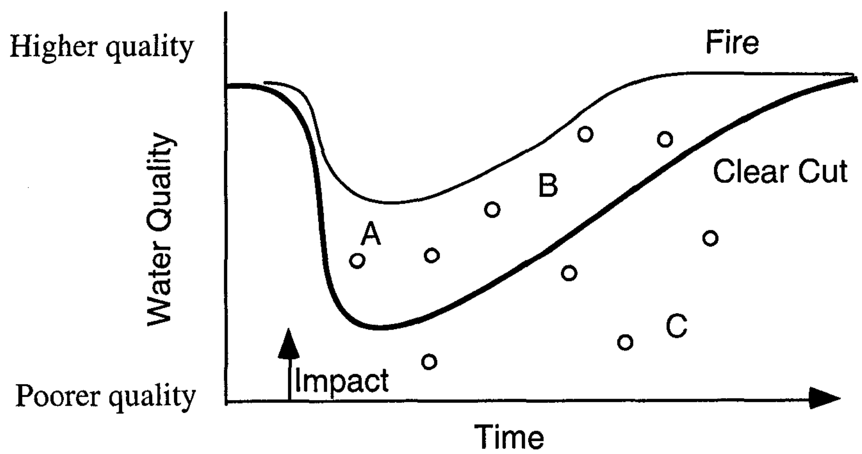


Fig. 7-5: Hypotheses as to the relationship of water quality to time after forest fire and clearcut logging. Hypotheses were derived from the summary in Table 2. Points are qualitative differences from the "general" hypothesis for impacts from harvesting. There is a rapid decline in water quality immediately following fire or logging with a recovery through time. Letters represent potential explanations for residuals based on context of informants response: A) sites that were well drained prior to logging and where debris from logging was minimized, B) similar to A but may also include sites were shrub growth was not excessive, C) sites where logging interfered with already poor drainage through either slash left on site or windthrow following flooding.

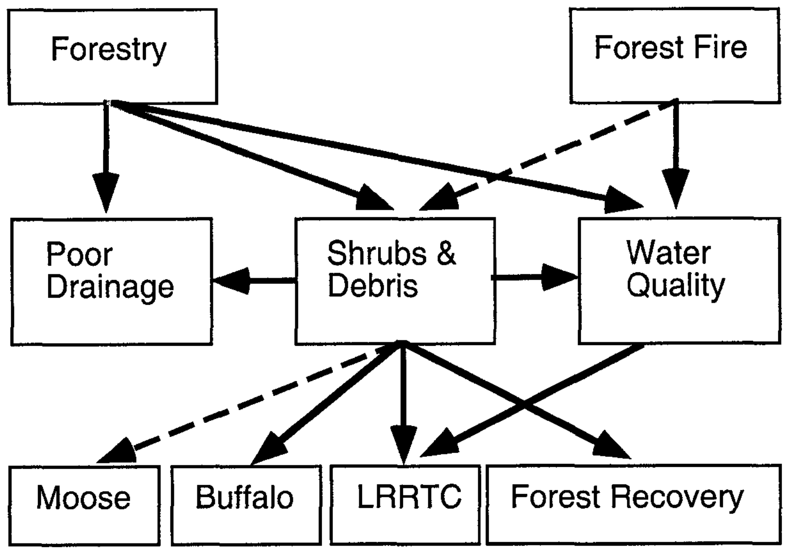


Fig. 7-6: Summary of primary ecosystem concerns following forestry or fire determined through interviews with LRRTC. Figure 5 demonstrated a type of variation in individual responses; Fig. 6 is the detailed understanding of ecosystem components that is both a result of and an explanation of the variation in responses observed in Fig. 5. Solid black lines represent general negative impacts, gray lines indicate a positive impact, and dashed lines indicate a variable impact.

References

Cruikshank, J. (1981). Legend and landscape: Convergence of oral and scientific traditions in the Yukon Territory. Arctic Anthropology 18(2): 67-93.

Cruikshank, J. (1998). The Social Life of Stories: Narrative and Knowledge in the Yukon Territory. Vancouver: UBC Press. Chapter 3: Yukon Arcadia: Oral Tradition, Indigenous Knowledge, and the Fragmentation of Meaning," pp. 45-70.

Dillon P.J. and F.H. Rigler (1974). The phosphorus-chlorophyll relationship in lakes. Limnol. Oceanogr. 19:767-773.

Edmondson, W.T. (1969). Eutrophication in North America. In: Eutrophication: Causes, consequences, Correctives. Washington D.C., National Academy of Sciences p. 124-149.

Franklin, S. (1995). Science as culture, cultures of science. *Annual Review Anthropol.* **24:** 163-184.

G.N.W.T. (1991). Government of the Northwest Territories. Report of the Traditional Knowledge Working Group. Yellowknife.

Heuscher, J.E. (1979). The interpretation of myths and fairy tales. In: Experience forms: Their cultural and individual place and function. (Ed.) G.G.Haydu. 9th International Congress of anthropological and ethnological sciences. The Hague, Mouton.

Howard, A. and F. Widdowson (1996). Traditional knowledge threatens environmental assessment. *Policy Options* 17(9): 34-36.

Hume, D. (1776). Philosophical essays concerning human understanding. In: Reason at work: Introductory readings in philosophy. S.M.Cahn, P.Kitcher, G.Sher and R.J.Fogelin (Eds.). Harcourt Brace Jovanovich. ©1984. San Diego.

Johnson, M. (1996). Lore: capturing traditional environmental knowledge. © 1992 Dene Cultural Institute.

Kuhn, Thomas S. The Structure of Scientific Revolutions. 3d edition. xiv, 212 p. 1996

Labinger, J. A. and H. Collins (2001). The one culture?: a conversation about science. Eds. Labinger, J. A. and H. Collins, The University of Chicago Press, Chicago.

Larsen, C.P.S. and G.M. MacDonald (1998). An 840-year record of fire and vegetation in a boreal white spruce forest. *Ecology* **79:** 106-118.

Larsen, C.P.S. (1997). Spatial and temporal variations in boreal forest fire frequency in northern Alberta. *J. Biogeogr.* **24:** 663-673.

Lea, B. (1996). A report of wisdom synthesized from traditional knowledge component studies. Northern River Basins Study synthesis report. Alberta Environmental Protection. Edmonton, Alberta.

Lewis, H. (1982). Fire Technology and Resource Management in Aboriginal North America and Australia. In: The Regulation of Environmental Resources in Food-Collecting Societies. E.Hunn and N.Williams (Eds.). American Association for the Advancement of Science Selected Symposium Series #67. Westview Press. Boulder, Colorado. pp.45-67.

Lewis, H. (1978). Traditional Uses of Fire by Indians in Northern Alberta. *Current Anthropology* **19**: 401-402.

Lewis, H. T. A. Ferguson (1988). Yards, Corridors and Mosaics: How to Burn a Boreal Forest. *Human Ecology* **16:** 57-77.
Lewis, H. (1989). Non-Agricultural Management of Plants and Animals. In: Wildlife Production Systems, R. J. Hudson, K. R. Drew and L. M. Baskin (Eds.). Cambridge University Press, pp. 54-74.
McCormack, P. (1998). Native homelands as cultural landscapes: De-centering the wilderness paradigm. In: Sacred Lands, Aboriginal worldviews, claims, and conflicts. J.Oakes, K.Kinew and E.Maloney (eds.). Canadian Circumpolar Institute © 1998, University of Alberta. Edmonton.
McEachern, P., E.E. Prepas, John J.Gibson and P.Dinsmore (2000). Forest fire-induced impacts on phosphorus, nitrogen and chlorophyll a concentrations in boreal subarctic lakes of northern Alberta. *Can. J. Fish. Aquat. Sci.* **57(Supp 2)**: 73-81.
Morrow, P. and W. Scheider (1995). When our words return: writing, hearing and remembering oral traditions of Alaska and the Yukon. Utah State University Press. Logan, Utah.
Noland, L.J. and T.Gallagher (1989). Cross-cultural communication for land managers and planners in Alaska. *Agroborealis* **21(1)**: 18-23.
NRBS (1996). Northern River Basins Study Report to the Ministers. Alberta Environment.
Ohmagari, K. And F. Berkes. 1997. Transmission of Indigenous knowledge and bush skills among the Western James Bay Cree women of subarctic Canada. *Human Ecology* **25(2)**: 197-222.
Rigler, F. H. and R. H. Peters (1995). Science and Limnology. Excellence in Ecology Series. International Ecology Institute. Luhe, Germany.
Schindler, D.W. (1973). Eutrophication of lake 227 by addition of phosphate and nitrate: second third and fourth years of enrichment, 1970, 1971, and 1972. *J. Fish. Res. Board. Can.* **30**: 1415-1440.
Schwarz, A.G.and R.W.Wein (1997). Threatened dry grasslands in the continental boreal forests of Wood Buffalo National Park. *Can. J. Bot.* **75**: 1363-1370.
Sewapagaham, Chief Johnson (1998). Traditions for today: Building on cultural traditions. International Indigenous Research Institute, University of Alberta. Edmonton.
Spencer, C.N. and F.R.Hauer (1991). Phosphorus and nitrogen dynamics in streams during a wildfire. *J.N.A.B.S.***10:** 24-30.
Stevenson, M.G. (1996). Indigenous knowledge in environmental assessment. *Arctic* **49(3)**: 278-291.
Stevenson, M.G. (1998). Traditional knowledge in environmental management? From commodity to process. SFM-NCE working paper.
Stevenson, M.G. (2001). No, it's not just like gardening: Traditional knowledge and colonial discourse in the modern world. CINSA Annual Conference Presentation May 29-31 2000, Edmonton, AB.
Strong, W.L.and K.R.Leggat (1992). Ecoregions of Alberta. Alberta Forestry, Lands and Wildlife, Land Information Services. Edmonton, AB.
Timoney, K.P. (1999). Threatened dry grasslands in the continental boreal forests of Wood Buffalo National Park: commentary. *Can.J.Bot.***77**: 913-917.

Toner, M. and P.Keddy, (1997). River hydrology and riparian wetlands: A predictive model for ecological assembly. *Ecol. Appl.* **7**: 236-246.

Summary

Surface-water chemistry is a reflection of the physical setting (Naumann 1929; Hynes 1975) and is described by limnological processes across a scale from macro physiographic gradients to microscopic interactions between organisms and the chemical environment. Regional synoptic surveys categorize lakes into physiographic provinces with similar water chemistry due to geologic, climatic and vegetational characteristics. These surveys help limnologists impute a landscape's water chemistry and hypothesize control mechanisms for surface water chemistry (Rigler 1982). The magnitude of human activities, such as logging or resource harvesting, is a gradient that can be extracted from regional surface water surveys (e.g., Buttle and Metcalfe 2000).

While illustrative in hypothesis form, regional scale knowledge lacks the requisite specificity to be a predictive tool for minimizing human impact (Pace 2001). For example, Prepas et al. (2001) demonstrated that distinctions in hydrologic control for boreal forest lakes of Alberta depend on the proportion of total catchment area comprised of wetlands. Strong correlation coefficients ($r^2 > 0.75$) were obtained between limnetic concentrations of such parameters as phosphorus with catchment features such as size-to-lake volume ratio. However, the regression between catchment size and lake water phosphorus concentration had 95% confidence intervals that spanned a 4-fold range, or half the total phosphorus range observed in the surveyed lakes. While not useful for formulating policy, the Prepas et al. study provided necessary hypotheses linking lake water chemistry to landscape characteristics in a region where boreal forest can overlay deep organic soils and glacial tills. Further development of the hydrologic and surface water chemistry hypotheses presented by Prepas et al. (2001) is required to support sustainable development of natural resources.

My study unfolds in a northern boreal forest where deep organic soils (> 10 cm) contain permafrost and are underlain by glacial till combined with lacustrine and marine sediment. Surface vegetation, biogeochemistry and water chemistry have been shaped not only by natural factors such as geology and climate, but also by centuries of human interaction. Knowledge of some members of the Little Red River and Tallcree First Nations, whose traditional lands includes a portion of my study area, provided another approach to understanding disturbance ecology. My thesis is an exploration of patterns in surface water chemistry, hydrology and biogeochemistry that ties in a historic perspective from Indigenous peoples.

Regional surveys like Prepas et al. (2001) promote inductive reasoning by elucidating trends and controls on surface water chemistry that should be examined at process-oriented scales (Pace 2001). The hydrologic and biogeochemical processes in forest soils and peatland have been studied in detail (e.g., Freeze and Cherry 1979; Ivanov 1981; Stumm and Morgan 1981). However, how these processes interact at catchment scales and with changing climatic factors has not been limned for Canada's northern boreal forest, thus precluding assemblage into predictive models (Pietroniro et al. 1996). My thesis presented results from a regional survey of 24 lakes in the northern boreal forest where permafrost existed in extensive peatland soils. The survey indicated that small hydrologic-contributing areas controlled lake water chemistry and were sensitive to forest fire impacts. Cation exchange within burnt catchments resulted in proton flux and a 9% reduction in mean pH. Compared to lakes in undisturbed catchments, lakes in burnt catchments contained: more than 2-fold higher ($P << 0.01$) mean concentrations of total, total dissolved, and soluble reactive phosphorus; 1.5-fold higher concentrations of dissolved organic carbon ($P << 0.01$); and more than 1.2-fold higher ($P < 0.05$) concentrations of total and total dissolved nitrogen, nitrate/nitrite and ammonium. Elevated nutrient concentrations were apparent in lake waters from drainages burnt over the last four decades. Lake water phosphorus concentrations were particularly sensitive to catchment disturbance in the peatland region of my study, where 74% of variance in total phosphorus could be explained by the temporal and spatial attributes of previous fires. Fire increased materials flux from watersheds to the study lakes. Recovery to pre disturbance conditions spanned decades.

Process-oriented questions arose from the lake survey: Why are contributing areas small, and are they poorly connected? How does water reach lakes in peatland drainages underlain by permafrost? What are the dynamics of lake-catchment area connections? Do peatlands exert disproportionate control over lake water chemistry? What internal processes control a lake's biomass and algal production? These questions required hydrologic and biotic-response data at the scale of individual catchments and lakes. However, with Canada's boreal forest, this vast and remote area provides challenges to the investigation of process-oriented questions. The second chapter outlined a method to overcome these challenges and determine hydrologic parameters for remote catchments from deuterium and oxygen-18 content of water samples. I proposed refinements to two hydrologic indices that correlate water balance with processes controlling solute flux from disturbed and undisturbed drainage basins.

Lake water isotopic composition was used to estimate contributing areas and catchment water yields for each of 24 Caribou Mountain lakes. Contributing hydrologic areas averaged 25% of the drainage basin areas. Catchment water yields determined from the isotope model (110 mm) were comparable to yields determined from stream-discharge measurements (114 mm) and published values for the region (75 to 100 mm). However, water yields for individual catchments ranged from 9 to 282 mm, precluding application of regional hydrologic generalizations to predict individual catchment yields. In catchments both unimpacted and impacted by forest fire, water yield was negatively correlated with lake-basin area to lake-volume ratios, supporting the hypothesis that nearshore areas dominated runoff production. However, lakes in burnt catchments had elevated water yield compared to unburnt catchments when drainage basin to lake volume ratios were less then ten. The results were consistent with the hypothesis that lakes in relatively small drainage basins were particularly sensitive to changes in water chemistry following forest fire because pathways for runoff and groundwater flow were short and more strongly influenced by shallow soil pathways. The result differs from that observed for lakes on the Precambrian Shield where impacts from fire increased with drainage basin size (D'Arcy and Carignan 1997). However, the divergent observations may not be so dissimilar when viewed in the context of hydrology. Thin soils over bedrock make Precambrian Shield lakes more tightly connected to their drainage basin. In northern Alberta, water yields decline as drainage areas increase indicating that a greater proportion of water is lost to recharge and water travels longer flow pathways that do not intersect disturbed soils at the surface. The greater hydrologic connectivity of lakes with small drainage basin area to lake volume ratio in my study is a similar result to that expressed on the Canadian shield for drainage basins with steeper slopes. Steeper slopes (or smaller DBA/V in my study) result in faster groundwater movement, heightening the impacts on receiving lakes from disturbance of surface soils.

Elevated water yields were anticipated from burnt catchments. However, only those with small drainage-basin areas relative to lake volume demonstrated enhanced water yield. I speculated that lakes in larger catchments were not sensitive to changes in water chemistry because contributing areas confluesced through longer flow paths that were less influenced by interaction with impacted surface soil. I attempted to define both contributing areas and effective areas for solute transport that could explain the hydrologic mechanisms for patterns in lake sensitive to changes following forest fire. In the study lakes, effective areas averaged 2 or 3% of drainage areas, suggesting poor drainage and low ion flux from the soils. The small effective areas could be related to the dominance of peatlands and hydrologic processes limited to the surface soils by permafrost and shallow water table depth. The gross assumptions intrinsic to my effective-area model precluded its wide applicability. Process-oriented studies of ion flux from peatland soils could, however, expand the effective-area model's utility. Contributing area was a better descriptor of among lake variation in water chemistry than drainage area for total phosphorus and dissolved nitrogen. The results were consistent with the hypotheses that: water chemistry was determined by limited proportions of the landscape, that lakes with small drainages relative to the lake volume were likely to have a stronger hydrologic connection to their watersheds than lakes sustained by large drainage basins, and that lakes with this stronger hydrologic connectivity were sensitive to water chemistry changes as a result of watershed disturbance.

In situ processes can be as important in controlling lake water chemistry as external factors. Internal loading, phytoplankton response to nutrients, and light penetration through the euphotic zone have implications on trophic status of lakes (e.g., Riley and Prepas 1984; Hecky 1988; Knowlton and Jones 1996). I found that lake waters in burnt catchments contained elevated nitrogen and phosphorus concentrations compared to lake waters in unburnt catchments. Contrary to expectations, elevated concentrations of the phytoplankton biomass indicator chlorophyll *a* were not apparent in burn-impacted lake water despite elevated phosphorus concentrations. I subsequently investigated possible constraints limiting phytoplankton biomass in burn-impacted lake water. A switch from phosphorus to nitrogen limitation and restrictions on phytoplankton uptake of nutrients from light limitation were investigated. Consistent with studies elsewhere, my thesis demonstrated that forest fire in peatland habitats increased the colour of lake water likely due to flux from burnt organic soils (e.g., Bayley et al. 1992; Carignan et al. 2000). My research was a first to demonstrate that elevated water colour, reduced light penetration, and lower nitrogen to phosphorus ratios following fire could cause reduced phytoplankton biomass. I tested nitrogen (N), phosphorus (P) and light limitation of pelagic phytoplankton with *in situ* microcosms placed in three lakes. Nutrient and light-penetration parameters spanned a range representative of the surveyed Caribou Mountain lakes. Phytoplankton species were identified from 10 lakes in unburnt and 10 lakes in burnt catchments to assess post-fire water chemistry influences on phytoplankton assemblages. P and concurrent N + P limitations of phytoplankton biomass were apparent ($P << 0.01$) in water from the two representative lakes. Nitrogen limitation was noted ($P << 0.01$) in the water representative of burn-impacted lakes. Light limitation of phytoplankton biomass was observed in microcosms where water colour was high (> 200 mg/L [Pt]). Phytoplankton species richness was 36% lower ($P << 0.01$) in the ten burnt than in the ten unburnt catchment lakes and phytoplankton communities in all surveyed lakes were dominated by cyanobacteria. My study indicated that phytoplankton communities in boreal forest lakes are particularly sensitive when peatland dominated catchment are burnt because enhance phosphorus and carbon loading from organic soils augment nitrogen and light limitation of phytoplankton growth.

Results in the second through fourth chapters of my thesis link survey and process-oriented data at multiple scales but focused principally on water chemistry and phytoplankton community dynamics following forest fires. Impacts from human activities, particularly logging, constitute a primary concern in boreal Canada. Resource managers (including government and industrial foresters and research scientists) are considering opportunities to limit impacts from human disturbance through emulation silviculture. Emulation silviculture is a model whose proponents postulate that boreal forests evolved to recover from the impacts of forest fire and disease. By logical extension, boreal ecosystems should withstand various human encroachments with impacts that are less than those from forest fire. The disparity between logging and fire impacts on lakes has been thoroughly reviewed elsewhere (e.g., Carignan et al. 2000; McRae et al. 2001). The prevailing hope is that emulation silviculture will maximize profitability while constraining harvesting impacts within sustainable limits (Hunter 1993). As previously demonstrated by Prepas et al. (2001), hydrologic and geochemical control mechanisms of surface water chemistry are different between upland catchments and wetland dominated catchments. This difference has a concomitant influence on the potential impacts from fire or logging in sloped or lowland terrain. I investigated the hydrologic and biogeochemical processes of boreal forest catchments in both well-drained upland and poorly drained lowland settings and then examined the impacts of logging on these processes.

The hydrologic setting of my study region differs from that of forested regions in Canada and the United States that have been the foundation of our current knowledge on fire and logging impacts on surface water chemistry (e.g. McColl 1975; Tiedemann et al. 1979; Paterson et al. 1998). Northern boreal forest hydrology is dominated by flow through peatlands, wherein modeling runoff is complicated by variable water storage, anisotropic hydraulic conductivity, summer frost or permafrost, and a paucity of hydrologic data. I described the hydrology of six peatland-dominated catchments representative of sloped and lowland conditions found in the boreal wetlands of western Canada. In sloped catchments, snowmelt discharged rapidly prior to loss of soil frost. Snowmelt discharge contained between 70 and 90% groundwater during the first few days as seepage that accumulated over the winter in riparian soils and the channel melted first. In subsequent days, groundwater was replaced by surface or near-surface runoff comprising 50 to 80% of stream discharge as shallow flow paths thawed and drew water from peatlands. Peak snowmelt discharge was two-fold higher than peak discharge during subsequent rain events and surface storage of melt water was minimal in sloped catchments. Therefore surface saturation of organic soils was limited and contributed only 20% on average to total summer discharge. Rainfall routed through organic and mineral soils dominated stream discharge, contributing an average of 34 and 46%, respectively, to total stream discharge following snowmelt. However, contributions to stream discharge from surface runoff rose to 60% of total discharge following major storms, indicating that overland flow can be important in sloped basins within peatland dominated boreal forest sites. In lowland catchments, snowmelt was stored. As a result, peak snowmelt discharge was 30 to 66% of summer peak discharge and was an equal mix of older groundwater and snowmelt. Water tables and stream flow in lowland catchments declined through summer, and groundwater routed through organic soils increasingly dominated (> 50 %) mean daily runoff as the summer progressed.

The extensive fires common to northern boreal forests do not discriminate between trees set back from, and proximate to, shorelines and stream banks (Gresswell, 1999). Proponents of emulation silviculture are subsequently attempting to increase the size of cut blocks and include trees previously protected within riparian buffers. Results from my study indicate that limited contributing areas may control surface-water chemistry in peatland dominated sites. Hydrologic contributing areas are not necessarily constrained to areas near surface water that would be managed as riparian areas; however, they typically do occur where soils are frequently saturated (Betson and Marius 1969; Dunne 1978). My survey of lake water chemistry in a peatland and permafrost landscape indicated nutrient and ion fluxes from watersheds are sensitive to soil disturbance in contributing areas, subsequently suggesting that large contributing areas relative to respective catchment areas amplify impacts on surface water chemistry. Adaptable tools to assess hydrologic conditions and potential sensitivity to boreal forest catchment disturbances are a categorical imperative if emulation silviculture intrudes on riparian buffers and impacts on contributing areas.

End-member mixing analysis (EMMA) is a tool that can elucidate hydrologic processes in watersheds and, in particular, the importance of different soils in controlling surface water chemistry. I integrated end member mixing analysis and a statistical tool, randomized intervention analysis, to describe harvesting impacts on hydrology and solute fluxes from sloped and lowland boreal forest catchments. My study identified several unexpected consequences of timber harvesting in an emulation silviculture approach. Water yield during snowmelt did not increase during the first spring following harvesting in sloped catchments but was reduced by 44% throughout the summer. Water levels rose in wells, suggesting that harvesting in the sloped catchments interfered with drainage and runoff patterns. Shallow soil incrementally contributed to runoff generation, with concomitant increases in phosphorus and organic carbon concentrations. Total flux of these nutrients was not altered due to reduced total water yield. Solute flux rates that derived from mineral soils (e.g. Ca^{2+}, Mg^{2+}) declined due to a change in hydrology favoring organic soil pathways. Water yields increased in lowland catchments following harvesting. Flux rates for most parameters remained unchanged in two harvested lowland catchments because increased water yields derived from precipitation and organic soils and were dilute compared to mineral-sourced water. Harvesting in a third lowland basin swelled groundwater contributions and concomitantly increased mineral-ion and nutrient fluxes compared to pre-harvest conditions. Timber harvesting changed the routing of water through soils; EMMA revealed how these changes were manifest as increases or decreases in surface runoff and discharge from organic or mineral soils. Changes in source waters for stream discharge concomitantly explained why flux rates of solutes were or were not impacted by logging.

The dynamics of any "natural" ecosystem includes human interaction with the environment, not simply the physical impacts of a single disturbance. Traditional Aboriginal activities have influenced aquatic environments through a number of direct means, but perhaps most importantly through their relationship with the beaver, an animal that directly modifies hydrology. Indigenous knowledge describes human interactions with a multiplicity of dimensions within a complex environment and ecosystem responses following disturbances. Indigenous knowledge also encapsulates a range of scales from the regional, such as large forest fires, to the local, such as removal of beaver from a single waterbody. Cree knowledge presented in my thesis covered a wide range of impacts on the local landscape from fire, logging and traditional activities. Similarities in attaining understanding of ecosystems between Indigenous knowledge and scientific method were defined and assembled into a model that focused on the physical context in which "data" were acquired. Initial analysis indicated consistent themes among LRRTC interviewees that were similar to scientific theories because they were empirically based and predictive. These theories linked water quality impacts of forestry to debris dams in streams, subsequent stagnation, and poor drainage. Forestry activities created excessive shrub growth and poor forest regeneration with commensurate impacts on faunal habitats. Direct impacts on most animals were small, but on others, such as moose, they were variable depending on shrub-growth densities following clear-cutting. A general hypothesis formed from the consistent theme of LRRTC knowledge was that while fire and logging both degraded water quality and terrestrial landscapes, burnt areas recovered much more quickly than harvested areas.

As with any empirically supported theory, the difference among responses (residuals) begged clarification through context assessment. Identifying and addressing the physical context in which the ecological understanding of individuals was formed helped clarify linkages between vegetation, hydrology and animal behavior. In well-drained areas, recovery from both fire and harvesting was considered relatively rapid, and differences between impacts narrowed for the two types of disturbance. In low lying areas, where soils were prone to saturation, the impacts of logging on hydrology, vegetation and subsequently on animals were considered much more severe than the impacts following forest fire. An examination of context identified differences in hydrologic response between upland and lowland settings as understood by the LRRTC that were consistent with findings from the hydrologic investigation in my thesis. The information from LRRTC added breadth by suggesting how hydrologic changes may be transmitted to vegetation and fauna in the region.

If forest management moves towards a disturbance model, there is a requirement to understand the impacts that are to be emulated. In this thesis I investigated the impacts of a large scale fire that burned entire watersheds because, presumably, harvesting trees from entire watersheds will become a focus of emulation silviculture. The construct for emulation silviculture forces the heterogeneity of forest fires into a single and impossible beast, a chimera. While large scale, stand replacing conflagrations that burn through riparian areas, exist on the boreal landscape, we must ask if the impacts are fully understood. As scientists and managers we should consider the wisdom of unleashing this chimera as a management policy. I investigated the impacts of such a fire and found large changes in lake water chemistry and hydrology that could span decades. The changes in water chemistry were transmitted to the phytoplankton community and possibly to higher trophic levels. Logging under an emulation silviculture approach also caused changes in water chemistry and hydrology. In short, the changes following logging were not necessarily the same as those following fire. Local knowledge, provided by Cree elders was complimentary, indicating that the relative changes following logging could be similar or more extreme than those from fire. My study provides detail on the hydrologic and chemical function of catchments and surface waters in a peatland dominated portion of the boreal forest with discontinuous permafrost. In comparing impacts from wildfire and emulation silviculture practice it further suggests that a natural disturbance model for logging may be a management strategy that we utilize only in physical settings where impacts are likely to be low. Insight into the appropriate application of emulation silviculture will require further understanding of fire and logging impacts in Canada's boreal forest and development of tools to assess these as well as other landscape disturbances.

References

Bayley, S. E., D. W. Schindler, K. G. Beaty, B. R. Parker and M. P. Stainton (1992). Effects of multiple fires on nutrient yields from streams draining boreal forest and fen watersheds nitrogen and phosphorus. *Can. J. Fish. Aquat. Sci.* **49**(3): 584-596.

Betson, R. P. and J. B. Marius (1969). Source areas of storm runoff. *Water Resour. Res.* **5**(3): 574-582.

Buttle, J. M. and R. A. Metcalfe (2000). Boreal forest disturbance and streamflow response, northeastern Ontario. *Can. J. Fish. Aquat. Sci.* **57**(Supp. 2): 5-18.

Carignan, R., P. D'Arcy and S. Lamontagne (2000). Comparative impacts of fire and forest harvesting on water quality in Boreal Shield lakes. *Can. J. Fish. Aquat. Sci.* **57**(Supp/2): 105-117.

D'Arcy, P. and R. Carignan (1997). Influence of catchment topography on water chemistry in southeastern Quebec Shield lakes. *Can. J. Fish. Aquat. Sci.* **54**: 2215-2227.

Dunne, T. (1978). Field studies of hillslope flow processes. Hillslope hydrology. M. J. Kirkby. Toronto, John Wiley & Sons: 227-293.

Evans, J. E., E. E. Prepas, K. J. Devito and B. G. Kotak (2000). Phosphorus dynamics in shallow subsurface waters in an uncut and cut subcatchment of a lake on the Boreal Plain. *Can. J. Fish. Aquat. Sci.* **57**(Supp. 2): 60-72.

Freeze, R. A. and J. A. Cherry (1979). Groundwater. Englewood Cliffs, N.J., Prentice-Hall.

Gibson, J. J., E. E. Prepas and P. McEachern (2002). Quantitative comparison of lake throughflow, residency and catchment runoff using stable isotopes: Modeling and results from a regional survey of Boreal lakes. *J. Hydrol.* **262**: 128-144.

Gresswell, R. E. (1999). Fire and aquatic ecosystems in forested biomes of North *America. Trans. Am. Fish. Soc.* **128:** 193-221.

Hecky, R. E. (1988). Nutrient limitation of phytoplankton in freshwater and marine environments: A review of recent evidence on the effects of enrichment. *Limno. Oceanogr.* **33**: 796-822.

Hunter, M. L. J. (1993). Natural fire regimes as spatial models for managing boreal forests. *Biological Conservation* **65**: 115-120.

Hynes, H. B. N. (1975). The stream and its valley. *Verh. Int. Ver. Theor. Angew. Limnol.* **19**: 1-15.

Ivanov, K. E. (1981). Water Movement in Mirelands. London England, Academic Press.

Knowlton, M. F. and J. R. Jones (1996). Experimental evidence of light and nutrient limitation of algal growth in a turbid midwest reservoir. *Arch. Hydrobiol.* **135**(3): 321-335.

Lamontagne, S., R. Carignan, P. D'Arcy, Y. T. Prairie and D. Pare (2000). Element export in runoff from eastern Canadian Boreal Shield drainage basins following forest harvesting and wildfires. *Can. J. Fish. Aquat. Sci.* **57**(SUPP/2): 118-128.

McColl, J. G. (1975). Forest fire: Effects on phosphorus movement to lakes. *Science* **188**: 1109-1111.

McRae, D. J., L. C. Duchesne, B. Freedman, T. J. Lynham and S. Woodley (2001). Comparisons between wildfire and forest harvesting and their implications in forest management. *Environ. Rev.* **9**: 223-260.

Metcalfe, R. A. and J. M. Buttle (2001). Soil partitioning and surface store controls on spring runoff from a boreal forest peatland basin in north-central Manitoba, Canada. *Hydrol. Process.* **15**: 2305-2324.

Naumann, E. (1929). The scope and chief problems of regional limnology. *Int. Revue Ges. Hydrobiol.* **21**: 432-444.

Pace, M. L. (2001). Prediction in the aquatic sciences. *Can. J. Fish. Aquat. Sci.* **58**(63-72).

Paterson, A. M., B. F. Cumming, J. P. Smol, J. M. Blais and R. L. France (1998). Assessment of the effects of logging, forest fires and drought on lakes in northwestern Ontario: a 30-year paleolimnological perspective. *Can. J. For. Res.* **28**(1546-1556).

Pietroniro, A., T. Prowse, L. Hamlin, N. Kouwen and E. Soulis (1996). Application of a grouped response unit hydrological model to a northern wetland region. *Hydrol. Process.* **14**: 1245-1261.

Planas, D., M. Desrosiers, S.-R. Groulx, S. Paquet and R. Carignan (2000). Pelagic and benthic algal responses in eastern Canadian Boreal Shield lakes following harvesting and wildfires. *Can. J. Fish. Aquat. Sci.* **57**(SUPP 2): 136-145.

Prepas, E. E., D. Planas, J. J. Gibson, D. H. Vitt, T. D. Prowse, W. P. Dinsmore, L. A. Halsey, P. M. McEachern, S. Paquet, G. J. Scrimgeour, W. M. Tonn, C. A. Paszkowski and K. Wolfstein (2001). Landscape variables influencing nutrients and phytoplankton communities in Boreal Plain lakes of northern Alberta: A comparison of wetland- and upland- dominated catchments. *Can. J. Fish. Aquat. Sci.* **58**: 1286-1299.

Rask, M. K. N., Sirkka-Lisa Markkanen (1998). Forestry in catchments: Effects on water quality, plankton, zoobenthos and fish in small lakes. *Boreal Environmental Research.*

Rigler, F. H. (1982). Recognition of the possible: an advantage of empiricism in ecology. *Can. J. Fish. Aquat. Sci.* **39**: 1323-1331.

Riley, E. T. and E. E. Prepas (1984). Role of internal phosphorus loading in two shallow productive lakes in Alberta, Canada. *Can. J. Fish. Aquat. Sci.* **41**: 845-855.

Stumm, W. and J. Morgan (1981). Aquatic Chemistry: An introduction emphasizing chemical equilibria in natural waters. Toronto, Wiley-Interscience.

Tiedemann, A. R., C. E. Conrad, J. H. Dieterich, J. W. Hornbeck, W. F. Megahan, L. A. Viereck and D. D. Wade (1979). Effects of fire on water: A state-of-knowledge review, U.S. Department of Agriculture, Forest Service.