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Watershed-lake interactions on the Boreal Plain and the impact of disturbance: fire in the wetlands, logging in the uplands

Impacts of natural disturbance and forest harvesting on water quality, primary producers and invertebrate communities in lakes of the Boreal Plain

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ABSTRACT

Water quality in lakes on the Boreal Plain was examined pre- and/or post disturbance by fire or logging. Indirect gradient analyses of chemical data from headwater lakes in undisturbed watersheds on the Boreal Plain, indicated a clear separation between those in wetland-dominated watersheds (57 to 100% wetland with variable proportions of bog, fen, swamp and marsh cover) and upland-dominated watersheds (0 to 44% wetland cover). In the former, percentage wetland cover in the watershed was positively correlated with total phosphorus (TP, r^2 = 0.78, primarily bog), total nitrogen (TN, r^2 = 0.46) and dissolved organic carbon (DOC, r^2 = 0.74) concentrations. In undisturbed settings, rich fens appeared to sequester both total phosphorus and total nitrogen. In upland-dominated lakes, the ratio of catchment area to lake volume (CA/LV) was the strongest watershed correlate of total phosphorus concentration (r^2 = 0.57) while most limnetic nitrogen and dissolved organic carbon were generated *in situ*. Colour concentration, being highest in wetland lakes, was correlated with isotopically defined effective drainage basin area (eDBA)/LV (r^2 = 0.66). Higher Chlorophyceae and Cryptophyta biomasses in wetland- than in upland-dominated systems may coincide with greater NH_4 availability.

Eleven headwater lakes in Alberta's Boreal Plain were monitored for nutrients and phytoplankton, two years before and two years after variable watershed harvesting (harvesting mean 16%, range 0-38%), as part of the Terrestrial and Riparian Organisms, Lakes and Streams (TROLS) program. After harvesting, variations in annual precipitation resulted in lake water residence times that differed by an order of magnitude from one year to the next. During the first post-treatment year, total phosphorus concentrations increased (overall 40%) in most lakes; however response was most consistent in lakes that were shallow, and the water column mixed or weakly thermally stratified. Chlorophyll *a*, cyanobacteria (*Aphanizomenon-Anabaena*), and cyanotoxins (MCLR) increased after harvesting, primarily in shallow lakes. Post-treatment change in TP concentration was strongly related to weather (greatest response in wet year) and relative drainage basin size (drainage basin area to lake volume, r^2 = 0.82, $P \ll 0.01$). There was no evidence that buffer strip width (20 m, 100 m, 200 m) influenced lake response. IN lakes with relatively large watersheds (catchment area to lake volume ratio ≥ 4) logging in or near rich fens linked with the lake appears to enhance lake TP concentration. These results suggest that activities within the entire watershed should be the focus of catchment-lake interactions.

In addition, biogeochemistry of ten headwater lakes in burnt peatland-conifer catchments and 14 in unburnt catchments, was evaluated throughout a summer two years following forest fire in a boreal sub-arctic region of northern Alberta. Cation exchange within burnt catchments resulted in proton flux and a 9% reduction in mean pH. Lakes in burnt catchments contained > 2-

fold higher (*P* << 0.01) mean concentrations of total, total dissolved and, soluble reactive phosphorus, 1.5-fold higher 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 compared to reference lakes. Total phosphorus (TP) concentration explained 86% of the variance in reference lake chlorophyll concentration (CHL) but was not related to CHL in burnt lakes. Analysis of CHL-TP residuals suggested algae in burn-impacted lakes were light-limited. With the addition of five lakes burnt between 1961 and 1985, time-since-disturbance and percent disturbance, combined, explained 74% of variance in TP among burnt lakes. Fire caused increased flux of materials to the study lakes with slow recovery over decades. Overall, burnt lakes had increased TP levels over reference lakes in the Caribou Mountains. Caribou Mountain reference lakes had overall lower TP concentrations than wetland-dominated watersheds in central Alberta. In upland dominated watersheds, TP concentrations changed similarly to the Caribou Mountains. In upland vs wetland dominated sites, other ions and forms of phosphorus responded differntly.

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INTRODUCTION

As elsewhere, surface water quality and aquatic ecosystems within the Boreal Plain ecozone of western Canada are influenced by inputs (suspended and dissolved) of terrestrial origin entering via surface and subsurface runoff. Drainage basin vegetation is an important contributor to runoff chemistry (Engstrom 1987). Wetland cover in particular may have a large impact on Boreal Plain water quality, as a source of dissolved organic carbon (DOC) and, under certain circumstances, of much higher concentrations of nitrogen and other ions relative to upland ecosystems (Richardson 1989; Halsey et al. 1997). Disruption of drainage basin vegetation cover and soils by natural and human-derived processes can potentially influence groundwater and surface runoff water quality and ultimately the limnology of receiving water bodies (Dillon and Kirchner 1975; Likens 1984; Byron and Goldman 1989; Bayley et al. 1992).

During the past 50-100 years, accelerated eutrophication (excessive phytoplankton and macrophyte growth) coincided in many Boreal Plain lakes with agricultural and urban development in their drainage basins (Cooke and Prepas 1998; Manning et al. 1999), with negative consequences for water quality (Prepas et al. 1997) and aquatic biodiversity (Schindler 1987). Timber harvesting, which has also significantly increased external nutrient loading patterns to lakes in other regions (Likens et al. 1970; Keenan and Kimmins 1993; Rask et al. 1993), has undergone substantial expansion in northern Alberta during the past decade (CCFM 1997). In light of this recent development, there is an understandable demand for information on the impact of timber harvesting and other land uses on surface water quality in the Boreal Plain. However, at the onset of this study, no information existed on drainage basin-surface water interactions for this region, with the exception of recent studies focusing on streams (Munn and Prepas 1986; Cooke and Prepas 1998) and wetlands (Halsey et al. 1997).

Knowledge of drainage basin-water quality interactions in the absence of disturbance is a prerequisite for aquatic resource management in the Boreal Plain ecozone. Starting in 1995, SFMN researchers undertook a widely-based field program to determine how drainage basin disturbance (fire and timber harvesting) might influence water quality and biotic responses in headwater lakes and streams in the boreal mixed-wood forests of western Canada. The objectives of this project were to:

i) assess the land-aquatic interactions for lakes in undisturbed drainage basins on the Boreal Plain, specifically the relationship between wetlands (defined as bogs, fens, swamps and marshes), relative catchment (drainage basin + lake) area, drainage basin slope, deciduous and conifer cover, and water quality variables such as nutrients (e.g., total phosphorus (TP), DOC and NH_4^+ concentrations) and dominant ions;

ii) use the above relationships to develop models describing the influences of forest harvesting and natural disturbance on water quality and aquatic biodiversity and community structure; and to

iii) compare and contrast effects documented for eutrophic Boreal Plain lakes in Alberta with a similar project in Québec focusing on less productive Boreal Shield lakes.

This report focuses on lake physical-chemical relationships; hydrological, wetlands and phytoplankton data are also provided where appropriate.

The selection of lakes in western Canada turned out to raise major challenges. There were no background data for our study on drainage basin-lake interactions for the Boreal Plain ecozone, although ample data have been assembled over the past two decades for the Boreal Shield. Timber harvesting is a very recent phenomenon for the Boreal Plain, relative to other regions of North America; thus, no data on long-term disturbance impacts are available. It may take many years of chronic disturbance before water quality impacts are assembled for the Boreal Plain lakes; eutrophication in Alberta lakes with agricultural and residential perturbation was often not evident until several decades after settlement (Mitchell and Prepas 1990). Although large areas of Boreal Shield and Boreal Plain forests were burnt in 1995, the Boreal Shield fires included merchantable timber, while the Boreal Plain fires were generally in wetland areas with less commercial importance. We were challenged with identifying patterns in lakes where drainage basins ranged from >50% wetland to white spruce/trembling aspen- (upland) dominated, and where the two forms of disturbance of interest were almost completely segregated between wetlands (fire) and uplands (harvesting). We chose to study small (mean surface area ~100-200 ha) headwater lakes within recent or projected fire- and harvest-impact sites in order to:

i) maximise the potential for large proportions of fire or harvest disturbance within small relative to large drainage basins;

ii) minimise variation in water quality associated with the relatively more complex morphometry of large lakes and the moderating influence of upstream lakes on drainage basin runoff; and

iii) allow comparisons between water quality trends documented for Boreal Plain systems with similar disturbance studies on the Boreal Shield that are also surveying small headwater lakes.

We were limited in the number of these lakes, particularly with burnt and harvested drainage basins, which could be landed safely by floatplane. Originally, we could only land on lakes with a minimum longitudinal axis of 1 km in length. During 1998, we began a working partnership with a small Alberta-based charter company with appropriate aircraft and personnel that enabled us to select lakes with longitudinal axes as short as 500 m. Still, the size of the area of the Boreal Plain that we could economically access by air (109 600 km^2) excluded many potential study lakes.

To help address these challenges, the Boreal Plain water quality project within SFMN was linked to the TROLS (Terrestrial, Riparian, Organisms, Lakes & Streams) project, as well as its partner, CMRP (Caribou Mountains Research Partnership) project. These projects focus in part on the effects of forest harvesting (TROLS), and fire (CMRP) on water quality and biota in boreal lakes. We have compiled data for 67 remote lakes in northern Alberta (28 this study, 11 TROLS and 28 CMRP), making it the most extensive study of its kind in western Canada. As outlined in the original SFM Network proposal, we include water quality data generated by TROLS, CMRP and, as the partnership evolved, with Little Red River and Tallcree First Nations within this study because:

i) the Boreal Plain ecozone includes drainage basins with large variations in topography, soil types and proportions of coniferous, deciduous and wetlands vegetative cover. To concentrate on disturbance effects, a substantial database is necessary to identify and partition out the background factors related to drainage basin characteristics that influence water quality;
ii) our data set facilitates comparisons of disturbance impacts on lakes from aspen- (TROLS), coniferous- and permafrost- (CMRP), and mixedwood-dominated drainage basins (this study);
iii) these data enable us to examine how the proportion of drainage basin disturbance influences the magnitude of response within lake ecosystems. Currently, we have water quality data for systems burnt during 1995, ranging from an average of 19% (core lakes, this project) to >80% (CMRP) of mean drainage basin area; and
iv) these data provide the opportunity to examine the generalities from intensive and process-orientated studies (e.g., hydrology, cyanobacteria, nutrient enrichment) initiated by the TROLS project.

We are also working with TROLS researchers in their experimental studies to identify the role of riparian buffer strips in minimising harvesting impacts on water quality and aquatic biota. While all provincial forest practice codes in Canada use riparian buffer strips to manage the land-water interface, current riparian buffer strip regulations in Alberta were developed outside the Boreal Plain ecozone, and thus are without ecological basis for this region. TROLS, with co-operation from our study, will provide a scientific basis for the design and management of riparian buffer strips for Boreal Plain forest practice codes. Our database is maintained separately from that of TROLS, yet data from each can be combined where appropriate to address specific questions. Our project also co-operated with TROLS personnel on fieldwork logistics.

This report provides an overview of results to date from the Boreal Plain water quality project, covering January 1996 to March 2000:

- results from studies of undisturbed wetland- and upland-dominated catchments on the Boreal Plain;
- results of forest harvesting impacts on Boreal Plain Lakes (TROLS);
- forest fire impacts on Boreal lakes in a permafrost dominated region of northern Alberta.

Appendix I lists the students involved with this project and SFM Network publications.

LANDSCAPE VARIABLES INFLUENCING NUTRIENTS AND PHYTOPLANTKON COMMUNITIES: A COMPARISON OF WETLAND- AND UPLAND-DOMINATED CATCHMENTS

The Boreal Forest covers over one-fifth of Canada's landmass. Geological and climatic variation across this land base result in two major sub regions: the larger, wetter and nutrient-poor Boreal Shield of central and eastern Canada and the drier and nutrient-rich Boreal Plain of western Canada. On the Boreal Shield, differences in lake morphometry, drainage basin topography and land use explain variability in water quality (Schindler 1971; Dillon et al. 1991; D'Arcy and Carignan 1997), while similar relationships have yet to be developed for the Boreal Plain. Lakes on the Boreal Plain are situated on low-relief, often poorly-drained sedimentary tills in contrast to the typically higher relief of Shield drainages (Mitchell and Prepas 1990). Relative to lakes on the Boreal Shield, lakes on the Boreal Plain tend to be phosphorus- (P) and plankton-rich (D'Arcy and Carignan 1997; Mitchell and Prepas 1990). On an annual basis, euphotic total phosphorus (TP) in lake water on the western Boreal is derived mainly through internal sediment phosphorus recycling (Shaw and Prepas 1990), atmospheric phosphorus inputs being relatively unimportant compared to Boreal Shield lakes (Shaw et al. 1989; D'Arcy and Carignan 1997). In summer, high cyanobacterial biomass is typical of phytoplankton communities on the Boreal Plain (Zhang and Prepas 1996), while chrysophytes and other flagellates dominate Boreal Shield communities (Kling and Holgrem 1972). Given differences in drainage basin geomorphology, nutrient dynamics, and phytoplankton communities, *in situ* models developed for drainage basin water quality interactions on the Boreal Shield (e.g., Rasmussen et al. 1989 for colour, dissolved organic carbon) will likely require modifications for the Boreal Plain.

Another feature which distinguishes the two Boreal subregions is the amount and type of wetlands; the Boreal Plain (Alberta) having more than twice the wetland (bogs, fens, marshes, swamps) coverage of the Boreal Shield (Quebec) (NWWG 1998). The extent of peatland, (i.e., bogs and fens), coincides with increased dissolved nitrogen, phosphorus and organic carbon concentrations in lakes on the Boreal Plain (Halsey et al. 1997) and Boreal Shield (Dillon et al. 1991). Peatlands can vary markedly in water discharge and thus in their influence on surface water chemistry (Halsey et al. 1997). The amount of water produced by surface and subsurface flow forms an effective watershed (eDBA, defined with stable isotopes by Gibson and Prowse 1998) within the total drainage basin. The size of an eDBA depends on the resistance water encounters flowing from precipitation to lake and depends on surface storage, soil storage and hydraulic conductivity, groundwater levels and the seasonal water balance (Soranno et al. 1996). Effective DBA may be more closely linked with water quality in regions such as the Boreal Plain where low drainage basin slope and variable runoff are associated with extensive peatland cover.

Data collected from headwater lakes having watersheds with minimal disturbance and that contained a range from near zero to 100% wetland cover in their watersheds were used to test hypotheses generated from studies based primarily on the Precambrian Shield and for peatland-dominated lake basins. These hypotheses were: 1) physical features, such as water residence time, lake depth, drainage basin area to lake area ratio (DBA/LA), slope and vegetation cover are effective predictors of lakewater total phosphorus concentrations; 2) nitrogen concentrations will be higher in wetland-dominated systems; 3) phytoplankton communities within wetland-dominated systems will have higher biomasses of N-dependent taxa than upland-dominated systems; 4) in-lake dissolved organic carbon and colour are related to the extent of wetland coverage within drainage basins; and 5) eDBA is a better predictor of constituent inputs, particularly in wetland-dominated watersheds than DBA or catchment area (CA: the sum of DBA and LA).

Study Area

Euphotic water quality and phytoplankton samples, and morphometric and watershed characteristics were collected for 26 headwater lakes located within a 109,600-km^2 area (Fig. 1) within the Mixedwood ecoregion of the Boreal Plain ecozone (Strong 1992). Soil types within the study region range from eutric brunisols/gray luvisols in upland to organics/gleysols in

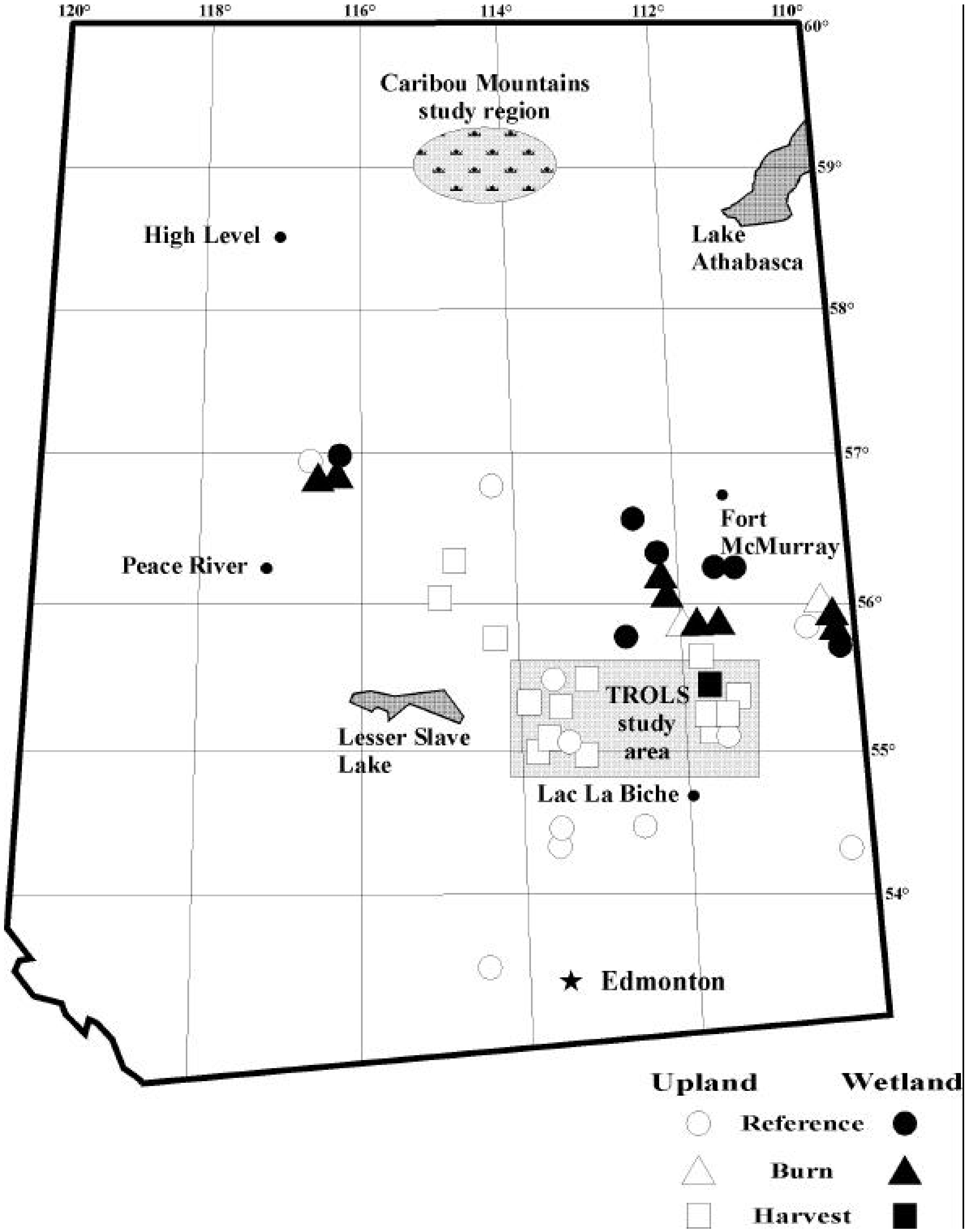

Fig. 1. Locations of Boreal Plain lakes studied by this study (SFMN) and the TROLS project, 1996-1998. The Caribou Mountains study region is also indicated.

wetland areas. Boreal Plain wetlands are categorized as peatland (bogs, poor fens (PFEN), rich fens (RFEN)) and non-peatland (swamps and marshes (SWAMP); Halsey et al. 1997). Bogs are dominated by bryophytes, notably *Sphagnum* while tree cover, when present, is limited to *Picea mariana*. Fens vary in cover from open (dominated by *Carex*, *Scirpus* and *Eriophorum*), shrubby (*Betula* and *Salix*) to wooded (*Picea mariana*, *Larix laricina*, *Betula* and *Salix*). Swamps and marshes are predominately forest/shrub (swamps) and *Carex*/*Scripus*/*Typha* (marshes). Boreal Mixedwood uplands are dominated by *Populus tremuloides*, *Populus balsamifera* (DECID) and *Picea glauca* (CONIF) stands, with a diverse understory of herbs and shrubs (Strong 1992). Thirty-year average annual precipitation for the region ranged from 388 to 432 mm (33% as snow), while mean annual air temperature ranged from 0.7 to 1.4°C. During the main sampling periods (June to September 1996 and 1997), the region received twice the average precipitation compared with the 30-year normal, depending on location (Environment Canada, unpubl. data). The two main sampling years were relatively wet. The lakes were chosen as those which were slated for, or near those slated for, harvesting in or near 1997, or which were near areas burnt in the 1995 wetland-dominated fires. None of the study systems have been disturbed (>5% of vegetation removed) within the past 50 years.

Sampling and Analyses

Lake selection, sample collection, and data analyses are outlined in Prepas et al. (2000, in prep.). The database for this section includes 38 independent summer observations of water quality on 26 lakes. Each of these 38 independent estimates represents summer mean euphotic water quality based on one (n = 4), three (n = 22), four to five (n = 10) or nine to ten (n = 2) sampling dates.

Results and Discussion

The lakes surveyed during 1996 and 1997 (Prepas et al., 2000, in prep., Table 1) are smaller and shallower (mean LA, and Z_{mean}: 0.9 km^2 and 2.2 m respectively, n = 22) than Boreal Plain lakes from the Atlas of Alberta Lakes (Mitchell and Prepas 1990) (mean LA and Z_{mean}: 31 km^2, n = 42, and 6.7 m, n = 41 respectively; Prepas et al., 2000, in prep., Table 2). Consequently, four small headwater lakes in undisturbed watersheds, Long, Moore, Narrow and Sauer (mean LA and Z_{mean}: 4.1 km^2 and 7.8 m respectively, n = 4) were added to our survey set to better reflect the range of physical characteristics reported for the Boreal Plain. Mean DBA/LV of these four systems was one-quarter that for the other upland-dominated systems in our data set (1.3 versus 4.9 respectively), but other physical parameters did not differ overall despite noticeable extremes in individual systems (Prepas et al., 2000, in prep., Table 1). The inclusion of these lakes allowed us to assemble the best data set available to address our objectives.

Most previously studied Boreal Plain lakes are located in road-accessible, inhabited areas of Alberta, where wetlands were sometimes drained for agricultural or other water management purposes (Mitchell and Prepas 1990). In these lakes, alkalinity, conductivity, pH and base cation concentrations were on average higher (Prepas et al., 2000, in prep., Table 2) than in our study lakes with relatively unperturbed drainages (Table 1). However, our study lakes reflect the range of trophic conditions in the Boreal Plain since ranges in TP, chl *a*, NO_3 and DOC concentrations are comparable to data for the region.

Lake productivity indices were up to an order of magnitude higher compared to D'Arcy and Carignan's (1997) Boreal Shield lakes (mean TP: 54 and 8.7µg· L^{-1}; chl *a*: 19 and 2.7 µg· L^{-1}, respectively). Although mean DBA in this study is larger than that of the Shield study (10 and 2.3 km^2 respectively), drainage ratios were similar (mean DBA/LA: 8.3 and 7.5 respectively). Wetland coverage per drainage basin in this study was on average 25 times that reported for the Shield study (32 and 1.3% respectively). Upland conifer cover in Boreal Plain systems averaged less than half that of Shield systems (18 and 44% of DBA respectively).

Indirect gradient analysis of chemical data indicated a clear separation between lakes with > 50% wetland (range 57% to 100%) and those with <50% wetland (range 0 to 44%) in their watershed (Prepas et al., 2000, in prep., Fig. 2). The gradient analysis confirmed that vegetative features of the catchment including % wetland and % upland and physical features related to relative drainage basin area were substantive contributors to water chemistry. This initial data compilation suggested that in addition to relationships with the entire data set (ALL) we should explore those with wetland- and upland-dominated catchments, independently.

When the lakes were divided into wetland- and upland-dominated systems the physical and chemical differences suggested by gradient analyses were apparent (Tables 1 and 2). Mean and maximum depths were over 2.5-times greater as were drainage basin slopes, while eDBA were 2.5 times smaller in lakes with upland-dominated drainage basins, but other morphometric and physical drainage basin parameters were not detectably different. Bogs and deciduous forest were the dominant vegetation in wetland- and upland-dominated drainages respectively. Although mean in-lake TP, chl *a* and TN concentrations were similar ($P > 0.5$), wetland-dominated systems were on average more coloured, with lower pH and conductivity than upland-dominated systems (186 vs. 38 mg· L^{-1} Pt, 5.7 vs. 7.6, and 265 vs. 80 µS· cm^{-1} respectively; $P <$ 0.001 for all comparisons).

Within the combined data set, DBA/LV was strongly linked with chl *a*, TP and colour (r = 0.58, 0.62, 0.64, $P << 0.01$ respectively), less strongly with TDP (r = 0.49, $P < 0.05$) and not detectably with nitrogen. Associations between CA/LV and DBA/LV were indistinguishable. Water residence was negatively related with TP and DOC (-0.45, -0.46, $P < 0.05$) and negatively related to colour (-0.84 $P << 0.01$). Presence of wetlands in the drainage, particularly bogs, were positively linked with lakewater DOC concentrations and colour (r = 0.58, 0.78 respectively), and negatively associated with lakewater pH, alkalinity, base cation concentrations and conductivity (r = -0.42 (Na) $P < 0.05$; -0.55 to -0.71 $P < 0.001$ for the remainder). Drainage basin slope was not related to nutrient concentration in this study in direct contrast to the Precambrian Shield (D'Arcy and Carignan 1997), probably because slopes were on average lower and covered a smaller range compared with the Shield study (ranges: 0.3 to 11% and 6 to 31% respectively). We removed Long, Moore, Narrow and Sauer lakes to determine if these deep lakes had influenced any of the relationships and re-examined our correlation matrix. Relationships between chl *a*, TP, TDP, DOC and colour, and DBA/LV and water residence time did not change appreciably with the removal of the four deeper lakes.

In the combined data set, TP concentrations were most strongly related to CA/LV (r^2 = 0.42, $P < 0.001$), and almost 60% of TP variability was explained by a combination of CA/LV and rich fen (RFEN) cover (Eq. 1). DBA/LV explained essentially the same variability in TP as CA/LV (39 and 42% respectively). In contrast to our initial hypotheses, TP concentrations in Boreal Plain lakes, all with relatively low watershed slope (11%) were unrelated to watershed slope.

$$\text{ALL: } \log(\text{TP}) = 1.57 + 0.38 \log(\text{CA/LV})^{**} - 0.48 \arcsin(\text{RFEN})^{**};\ r^2 = 0.59 \qquad [1]$$

In upland-dominated systems, CA/LV alone explained 57% of TP variability, while percent fen cover explained an additional 20% of TP variation (Eq. 2). Schindler (1971) suggests a similar relation between P and CA/LV for Boreal Shield lakes considering phytoplankton (chl *a* concentration and midsummer phytoplankton production and biomass) are positively related to CA/LV. Relative watershed size is a strong driving factor for TP concentration in our upland lakes, producing 10 fold more phosphorus per unit DBA than lakes on the Precambrian Shield (Carignan et al. 2000).

$$\text{UPLAND: } \log(\text{TP}) = 1.54 + 0.56 \log(\text{CA/LV})^{**} - 0.68 \arcsin(\text{RFEN})^{**};\ r^2 = 0.77 \qquad [2]$$

In wetland-dominated systems, percent wetland (WETL) and conifer cover explained most of the TP variance (78% and 87% respectively). Addition of sampling year (YEAR) for lakes in the wetland-dominated drainages added 7% to the TP variability accounted for by % wetland cover (Eq. 3).

$$\text{WETLAND: } \log(\text{TP}) = -222.17 + 0.01\,(\text{WETL})^{**} + 0.11\ \text{YEAR}^{*};\ r^2 = 0.85 \qquad [3]$$

Consistent with our preliminary data review, the relationship between percent wetland coverage and lakewater TP concentration (Prepas et al., 2000, in prep, Fig. 3B) suggests that our systems are from two populations, one with a positive link overall between wetland cover and TP, the other with a TP sequestering aspect associated with rich fens. By separating wetland cover into its three constituents, we see that bogs apparently release TP (r = 0.84, P << 0.01) poor fens play a neutral role (P > 0.3), and like our upland-dominated systems, rich fens sequester TP (r = -0.88, P << 0.01). Similarly to bogs in the watersheds of our study lakes, Dillon et al. (1991), D'Arcy and Carignan (1997) and Halsey et al. (1997) also reported positive relationships between wetland coverage and water-column P concentration, despite the relatively low wetland coverage of the first two studies (up to 25 and 6% respectively). Apparently, wetlands with organic soil do not retain P as effectively as forested systems (Halsey et al. 1997).

Consistent with our hypotheses, nitrogen concentrations were related to type and extent of wetland cover. Mean DIN for our upland-dominated lakes was comparable to values reported by D'Arcy and Carignan (1997) for Boreal Shield lakes (40 and 31 $\mu g \cdot L^{-1}$ respectively). Yet, most (~80%) DIN was NH_4 in the Boreal Plain lakes, compared to NO_3 in D’Arcy and Carignan’s (1997) Boreal Shield study. The predominance of NH_4 in Boreal Plain lakes was due to greater wetland cover and water logged soils which should favour the export of reduced nitrogen species such as NH_4 (Dillon et al. 1991). As a result, percent wetland cover explained 40% of NH_4 variability in the wetland-dominated systems (r^2 = 0.39, P < 0.01).

The TN pool in our study lakes was composed of 75% TDN on average (Table 1), of which > 90% was dissolved organic nitrogen. In the combined data set, mean and maximum depth and percent rich fen and upland conifer coverage were inversely related with TN concentrations but explained $\leq$ 35% of the variation. However when watersheds were separated based on > and < 50% wetland coverage, TN was strongly related with percent upland conifer (r = -0.87), wetland (r = 0.68), bog (r = 0.60) and fen cover (r = -0.57) and eDBA/LV (r = 0.66) in the former; (P < 0.001 for all), while TN was negatively linked with Z_{max} in the latter (r = -0.75, P < 0.001). Mean lake depth and wetland parameters together accounted for > 70% of TN concentration variation (Eq. 4 and 5):

$$\text{WETLAND: } \log(\text{TN}) = 1.99 + 0.89\ \arcsin(\text{WETL})^{**} + 0.81\ \log(Z_{mean})^{**};\ r^2 = 0.91 \qquad [4]$$

$$\text{UPLAND: } \log(\text{TN}) = 3.39 - 0.56 \log(Z_{mean})^{**} - 0.53 \arcsin(\text{RFEN})^{**};\ r^2 = 0.73 \qquad [5]$$

Wetlands immediately adjacent to our upland-dominated lakes are fens, which appear to retain nitrogen to some degree (Halsey et al. 1997). Whereas bogs that form the majority of total wetland area export nitrogen similar to the bogs elsewhere in Alberta (Halsey et al. 1997) and on the Shield (Dillon et al. 1991). The connection, although weaker than for TP, between watershed variables and lake TN concentrations is strongest in wetland-dominated systems. Denitrification and other *in situ* processes related to mean depth were the dominant forces controlling nitrogen concentration in lakes in upland-dominated watersheds.

Mean summer chl *a* concentrations covered 1.5 orders of magnitude (Table 1), and as expected, TP and TN concentrations were the best overall predictors of chl *a* (r^2 = 0.70, 0.43; P << 0.01). The chl *a*/TP relationship did not vary detectably among the combined, wetland- and upland-dominated data sets and was not enhanced by the addition of other independent variables. Total phytoplankton biomass (BIOM) was most closely related to TP concentration in both wetland- and upland-dominated systems but these relationships were weaker than those for chl *a* (r^2 = 0.42, 0.52). In a survey of 208 temperate upland-dominated lakes, chl *a* and mean depth were inversely related (r = -0.60; Duarte and Kalff 1989). We found a similar relationship within our upland- (r = -0.62, P < 0.001), but not wetland-dominated systems (P > 0.6) which may be related to higher colour and consequently reduced light penetration in the latter systems (Table 1).

Cyanobacteria were the dominant phytoplankton in all study lakes, followed by Peridineae and Cryptophyceae within wetland- and upland-dominated systems respectively (Prepas et al., 2000, in prep., Fig. 4). Within wetland-dominated systems, Chlorophyta and Chrysophyceae biomasses were equally represented, but Chrysophyceae exceeded Chlorophyta in upland-dominated systems. Diatoms were scarce during the July to September study period, relative to lakes on the Boreal Shield. We detected no differences in total phytoplankton biomasses between wetland- and upland-dominated systems, but Chlorophyta and Peridineae biomasses were higher in wetland- than in upland-dominated systems (Prepas et al., 2000, in prep., Fig. 4) and compared to data collected on other lakes in the region (Mitchell and Prepas 1990). Higher Chlorophyta biomasses in wetland- than in upland-dominated systems, despite equivalent TP concentrations, may coincide with greater DIN availability in the former. In contrast, Cryptophyceae biomass was higher in lakes with upland-dominated catchments. Thus consistent with our original hypothesis and patterns in DIN, N-dependent Chlorophyta were higher in lakes in wetland- rather than upland-dominated drainages.

Table 1. Summary water quality data for the 26 Boreal Plain study lakes. Data consist of independent annual estimates collected during the study period (1996-1998), or are from Mitchell and Prepas (1990). Water quality parameters in wetland- and upland-dominated systems were compared with two-tailed *t*-tests; probabilities (*P*) are included. *SE* = ± 1 standard error, Min. = minimum, Max. = maximum. Extinction coefficient data were unavailable for N37 (1997 only), Narrow, Sauer, Long and Moore lakes.

		Wetland-dominated (n = 14)				Upland-dominated (n = 24)				P	Total (n = 38)	
Variable	Abbreviation	Mean	*SE*	Min.	Max.	Mean	*SE*	Min.	Max.		Mean	*SE*
TP ($\mu g \cdot L^{-1}$)	TP	57	9.4	26	162	52	6.8	12	160	0.67	54	5.5
TDP ($\mu g \cdot L^{-1}$)	TDP	19	2.5	5.3	42	17	2.0	5.2	44	0.54	17	1.6
Chl *a* ($\mu g \cdot L^{-1}$)	CHLA	21	4.6	2.0	57	18	3.4	2.0	76	0.67	19	2.7
NH_4^+ ($\mu g \cdot L^{-1}$)	NH_4	136	90	6.4	1271	35	8.3	2.5	149	0.27	72	34
NO_3^- ($\mu g \cdot L^{-1}$)	NO_3	11	6.7	1.1	99	5.5	0.6	1.1	13	0.38	7.7	2.5
DIN ($\mu g \cdot L^{-1}$)	DIN	148	92	7.5	1281	40	8.6	5.2	154	0.26	80	34
TDN ($\mu g \cdot L^{-1}$)	TDN	858	123	373	2177	899	90	212	2129	0.79	884	72
TN ($\mu g \cdot L^{-1}$)	TN	1227	194	604	3297	1208	126	279	2825	0.94	1215	106
TN:TP	TN:TP	23	2.1	13	38	26	2.0	8.1	39	0.30	25	1.5
SO_4^{2-} ($mg \cdot L^{-1}$)	SO_4	4.2	2.2	0.2	25	7.2	3.1	0.1	74	0.44	6.0	2.1
Cl^- ($mg \cdot L^{-1}$)	Cl	0.3	0.03	0.1	0.6	1.5	0.8	0.1	20	0.18	1.0	0.5
Na^+ ($mg \cdot L^{-1}$)	Na	2.4	0.7	0.3	8.8	10	4.6	1.0	113	0.09	6.3	2.3
DOC ($mg \cdot L^{-1}$)	DOC	20	2.1	10	33	15	1.0	10	29	0.05	17	1.1
Extinction coefficient (m^{-1}), base e	ε	3.8	0.5	1.3	8.7	2.3	0.3	0.9	5.9	0.02	2.9	0.3
Colour ($mg \cdot L^{-1}$ Pt)	COL	186	29	29	358	38	5.2	8.0	125	<0.001	92	16
Alkalinity ($mg \cdot L^{-1}$)	ALK	28	8.4	2.9	95	110	14	40	389	<0.001	80	11
HCO_3^- ($mg \cdot L^{-1}$)	HCO_3	34	10	3.6	116	125	15	49	379	<0.001	91	12
PH	pH	5.7	-----	4.9	7.4	7.6	-----	6.9	8.7	<0.001	7.3	-----
Si ($\mu g \cdot L^{-1}$)	Si	703	194	40	2750	1563	235	56	4383	<0.001	1246	177
Ca^{2+} ($mg \cdot L^{-1}$)	Ca	10	3.2	0.9	37	24	1.9	10	52	<0.001	19	2.0
Mg^{2+} ($mg \cdot L^{-1}$)	Mg	2.3	0.6	0.3	7.1	11	1.7	3.7	48	<0.001	7.4	1.2
K^+ ($mg \cdot L^{-1}$)	K	0.6	0.2	0.2	2.4	3.6	0.5	0.7	11	<0.001	2.3	0.3
Conductivity ($\mu S \cdot cm^{-1}$)	COND	80	23	11	271	265	33	100	910	<0.001	197	27

Although total cyanobacterial biomasses were similar, absolute and relative biomasses of Oscillatoriales were greater in upland- than in wetland-dominated systems (Prepas et al., 2000, in prep., Fig. 5). Oscillatoriales are abundant in other upland-dominated systems in northern Alberta (E. Prepas. D. Planas, unpubl. data). *Microcystis* spp. was positively linked with NO_3 concentrations in wetland-dominated systems (r^2 = 0.61, $P << 0.01$) and to a lesser extent, so were non-nitrogen fixing cyanobacteria as a group (r^2 = 0.29, $P < 0.05$). Of all the groupings we reviewed, biomass of *Microcystis* spp. was most strongly associated with TP concentrations particularly in wetland-dominated systems (r^2 = 0.71, $P << 0.1$).

Colour concentrations are dependent on dissolved humic matter and negatively influence underwater light penetration, limiting colonization depths for primary producers (Chambers and Prepas 1988). Colour was strongly associated with wetland and bog and less so with fen cover in the combined data set (r = 0.81, 0.78 ($P << 0.01$) and 0.44 ($P < 0.05$) respectively). In contrast to relationships with TP, TN and DOC, eDBA/LV was more strongly linked with colour (COL) than CA/LV and DBA/LV (r = 0.81, 0.69 and 0.64 respectively); eDBA/LV combined with drainage basin slope (SLOPE) explained 74% of colour variability (Eq. 6). Colour also coincided negatively with water residence time (r = -0.84), which is consistent with in-lake degradation of humic matter (Rasmussen et al. 1989).

$$\text{ALL: } \log(\text{COL}) = 1.82 + 0.51 \log(\text{eDBA/LV})^{**} - 0.06\ \text{SLOPE}^{**};\ r^2 = 0.76 \qquad [6]$$

Drainage basin slope in Eq. 6 could represent the transition from organic peatlands towards mineral soils, which export less humic matter or distinct drainage patterns between upland- and wetland- dominated watersheds. Although catchment slopes are low in our study ($\leq$ 11%), they explained one-quarter of colour variability as a single predictor. Based on a study focused on the Precambrian Shield in Labrador Rasmussen et al. (1989) interpreted the absence of a relationship between colour and wetland cover as due to the overriding influence of slope. In the Labrador study and peatland-dominated subarctic plateau (McEachern et al. 2000), DOC and colour were highly correlated (r^2 = 0.86 and 0.89 respectively) while in our study the relationship was much weaker (r^2 = 0.48). Much of the DOC in Boreal Plain lakes may be produced *in situ* (Curtis and Prepas 1993). As expected, within the combined data set, colour and chl *a* concentration accounted for most of the variability in light extinction (r^2 = 0.80, $P << 0.01$). As predicted, wetland (primarily bog) coverage in the drainage, along with the variable eDBA/LV, were strongly associated with colour concentrations in all our lake water.

Studies based in Labrador, Finland and northern Alberta with relatively high wetland coverage (up to 25, 58 and 85% respectively) noted strong relationships between wetland coverage and in-lake organic carbon concentrations (Engstrom 1987; Kortelainen 1993; Halsey et al. 1997). While in the peatland dominated systems of the Caribou Mountains, DBA/LA explained up to 70% of DOC variability (McEachern et al. 2000). DOC concentrations in our combined data set were positively associated with percent wetland cover and CA/LV, but given the strong autochthonous input relationships were relatively weak (r^2 = 0.32 and 0.28 respectively). The latter was strongest within the upland-dominated systems, where CA/LV and mean depth explained 24 and 38% of DOC variability respectively.

In contrast, in the wetland-dominated subset, percent wetland cover and eDBA explained considerably more DOC variability (r^2 = 0.74 and 0.64 respectively). Engstrom (1987) predicted that DOC and water residence time are strongly related in systems where DBA/LA is > 4. In our study, DOC was inversely related to water residence time (r = -0.46) and not detectably related to DBA/LA (mean = 8.0). DOC concentration and watershed slope were weakly related (r^2 = 0.16, P < 0.02) while this relationship was much stronger (r^2 = 0.40) in lakes on the Boreal Shield (D'Arcy and Carignan 1997). In sharp contrast to colour, the connection between vegetative cover and DOC concentrations in lakewater on the Boreal Plain was stronger in wetland-dominated drainages than upland-dominated drainages. *In situ* generation of DOC was likely the prevailing force in upland-dominated drainages.

Mean water residence time (τ) was more than 20-fold longer for upland- than for wetland-dominated lakes (11 and 0.5 yr respectively, Table 2), attributable to the deeper lake basins and smaller eDBAs within the upland-dominated systems. Positive associations (r 0.65) between lakewater residence time and alkalinity, conductivity, HCO_3^-, Mg^{2+} and K^+ suggest the influence of hydrologic setting, specifically lake flushing rates for some but not all dominant ions. Differences between estimated isotopically defined eDBA and topographically defined DBA were less pronounced within wetland- compared to upland-dominated catchments. On average, wetland- and upland-dominated eDBAs were about ¾and ¼ of the DBA, respectively (Table 2); the former reflecting a relatively higher wetted-fraction or contributing area within the drainage basin and the latter reflecting lower overall runoff contributions from upland areas consistent with loss to groundwater. Within the 26-lake data set, runoff was positively related to bog area (r^2 = 0.87; P < 0.001), but was not detectably linked with DBA, suggesting that bogs play a key role in influencing surface runoff in our study region. Effective DBA is positively related with colour and percentage bog cover (r^2 = 0.40 and 0.37; P < 0.001) and negatively related with drainage basin slope and percent upland cover (r^2 = 0.30 and 0.47; P < 0.001). In wetland-dominated lakes, eDBA is a stronger correlate with DOC and TN than DBA. With the whole data set, eDBA was more strongly associated with lakewater colour than DBA, suggesting a connection with colour producing wetlands.

Table 2. Drainage basin and lake morphometry summary data for the 26 study lakes; data are from this study and Mitchell and Prepas (1990). Physical parameters in wetland- and upland-dominated systems were compared with two-tailed *t*-tests; probabilities (*P*) are included. *SE* = ± 1 standard error, Min. = minimum, Max. = maximum. Data for eDBA, eDBA/LV and τ consist of independent annual estimates collected during 1996 and 1997, or were taken from Mitchell and Prepas (1990). Other parameters were assumed constant over the study period (1996-1998). eDBA and eDBA/LV were unavailable for Narrow, Sauer, Long and Moore lakes.

		Wetland-dominated (*n* = 7)				Upland-dominated (*n* = 19)				*P*	Total (*n* = 26)	
Parameter	Abbreviation	Mean	*SE*	Min.	Max.	Mean	*SE*	Min.	Max.		Mean	*SE*
Catchment area (km^2)	CA	9.41	2.82	0.75	20.42	12.23	4.78	0.58	88.24	0.62	11.47	3.55
Drainage basin area (km^2)	DBA	8.27	2.64	0.56	18.80	10.81	4.37	0.49	82.40	0.62	10.13	3.25
Effective DBA (km^2)	eDBA	6.30	1.15	2.63	15.88	2.46	0.52	0.10	9.45	0.007	4.04	0.65
Slope (%)	SLOPE	1.1	0.5	0.3	3.9	4.1	0.5	1.1	11	<0.001	3.3	0.5
Upland conifer (% of DBA)	CONIF	21	6.7	0	43	17	2.8	1.2	42	0.53	18	2.7
Upland deciduous (% of DBA)	DECID	3.1	2.8	0	20	64	4.0	30	85	<0.001	47	6.1
Wetlands (% of DBA)	WETL	76	7.1	57	100	17	2.3	0	44	<0.001	33	5.8
Bogs (% of DBA)	BOG	44	11	11	85	2.4	1.1	0	20	<0.001	14	4.7
Total fens (% of DBA)	FEN	30	5.6	7.8	45	8.8	1.9	0	30	<0.001	15	2.7
Rich fens (% of DBA)	RFEN	12	6.8	0	39	8.8	1.9	0	30	0.56	9.8	2.2
Poor fens (% of DBA)	PFEN	17	7.3	0	45	0.01	0.01	0	0.2	0.004	4.7	2.4
Swamps & marshes (% of DBA)	SWAMP	1.9	1.0	0	7.0	5.4	0.9	0	12	0.003	4.4	0.7
Lake elevation (m)	ELEV	613	36	505	730	633	11	549	740	0.44	625	12.5
Lake area (km^2)	LA	1.15	0.24	0.19	2.00	1.42	0.52	0.09	9.28	0.65	1.34	0.39
Mean depth (m)	Z_{mean}	1.3	0.3	0.7	2.5	3.6	0.7	0.9	14	0.004	3.0	0.6
Maximum depth (m)	Z_{max}	3.1	0.7	1.8	5.8	8.5	2.0	1.8	38	0.02	7.1	1.6
Lake volume (m^3 x 10^6)	LV	1.80	0.70	0.19	5.72	8.00	4.19	0.27	77.40	0.17	6.31	3.09
Water residence time (years)	τ	0.5	0.2	0.1	2.3	10.8	5.7	0.2	100	0.08	7.0	3.65
Drainage basin area/lake volume	DBA/LV	6.49	2.49	1.28	19.66	3.71	0.89	0.50	16.7	0.32	4.47	0.94
Effective DBA/lake volume	eDBA/LV	6.68	1.41	0.58	18.34	2.28	0.51	0.13	7.65	0.01	4.09	0.75
Drainage basin area/lake area	DBA/LA	6.69	1.66	1.86	13.21	8.93	1.17	1.52	21.29	0.29	8.33	0.97
Catchment area/lake volume	CA/LV	7.37	2.59	1.96	21.15	4.13	0.94	0.60	17.80	0.29	5.05	0.99

FOREST HARVEST IMPACTS ON WATER QUALITY AND AQUATIC BIOTA: INTRODUCTION TO THE TROLS LAKE PROGRAM

Water quality in lakes is a function of both internal and external processes. Disturbances in a watershed (e.g., agriculture, deforestation) can alter external nutrient inputs and runoff (Cooke and Prepas 1998; Lamontagne et al. 2000), thereby changing water quality. Internal recycling of added nutrients through biogeochemical processes (Riley and Prepas 1984) can be a concern, particularly in lakes within sedimentary basins (Shaw and Prepas 1989). Relatively small disturbance in the watershed may be sufficient to disrupt the P-binding potential of the bottom sediments (Manning et al. 1999), thereby increasing the chance of internal P release, and subsequent eutrophication, in lakes on the Boreal Plain.

Post-harvest changes in nutrient concentrations have been documented in streams (Holopainen and Huttunen 1998) and occasionally in lakes (Rask et al. 1998). Impacts at many trophic levels (primary producers to vertebrates) have been attributed to these changes (Rask et al. 1998). Buffer strips, unharvested areas left around lakes and streams, may be effective in moderating water quality changes in disturbed watersheds through retention of runoff and water-soluble nutrients (Ahtiainen 1992). Some information exists on the effectiveness of buffer strips on the Canadian Boreal Shield and west coast of Canada (Newbold et al. 1980), however there is no information on the Boreal Plain. Given differences in forest composition, till depth, precipitation patterns and forestry practices, there is little possibility to extrapolate from studies in other regions to the Boreal Plain of western Canada.

Lakes on the Boreal Plain have algal communities characterized by a relatively high proportion of cyanobacteria (Trimbee and Prepas 1987). Relative or absolute changes in available nutrient concentrations, particularly as it impacts on P and/or N, may create conditions conducive to further proliferation of cyanobacteria (Kotak et al. 1995). Cyanobacteria and their associated toxins may impact aquatic food-webs (Ghadouani et al. 1998). In oligo-mesotrophic lakes of the Boreal Forest in Eastern Canada, recent studies also indicate that harvesting may impact planktonic communities through changes in nutrients and organic matter in lake water (Planas et al. 2000). Primary productivity and biological oxygen demand increase together, often resulting in habitat restriction and/or loss of oxygen sensitive fauna (Dinsmore et al. 1999). The TROLS program was created to evaluate the effects of forest harvesting and buffer strips on Terrestrial and Riparian Organisms, Lakes and Streams.

Here, we present a summary of the TROLS lake program results and the deviation of lakes in the Western Aquatic Group which were sampled following logging (11 to 30% cut in drainage basin area). We focus on annual trends and examine the hypotheses that: (1) changes in TP and IN concentrations in lakes on the Boreal Plain can be predicted from a combination of external and internal features, (2) there is no evident increase in nitrogen with cutting, (3) increased TP concentration following disturbance will be mirrored in chlorophyll *a* concentrations and a proportional increase in cyanobacteria and associated cyanotoxins.

Materials and Methods

The 12 TROLS lakes are clustered in three regional blocks of four lakes each: South Pelican Hills (SPH), South Calling Lake (SCL) and Lac La Biche (LLB) (Fig. 1). Each block had a lake designed as: reference (no cutting within 800 m, the maximum width feasible given timber requirements), 20 m, 100 m, or 200 m buffer strip. Following lake selection, accidental harvesting occurred in one of the reference drainage basins (LLB800; February 1994; 11% within the drainage basin, Prepas et al., 2000, in press, Table 1). Most forest harvesting was conducted between October 1996 and February 1997. A few trees were removed around SCL20 to connect cutblocks in February 1998. Although the original plan was to result in 30% harvesting within 800 m of the treatment lakes, the overall percent cut in the targeted watersheds was 23% within 800 m and 13% within the drainage basin. In 1997 the economic climate precluded correcting the discrepancies. The locations, watershed characteristics and selection criteria for the lakes are described by Prepas et al. (2000, in press).

Water samples were prepared, stored and analyzed as described by Prepas et al. (2000, in press) for total phosphorus (TP), total dissolved phosphorus (TDP), nitrate + nitrite (NO_3 + NO_2), ammonium (NH_4), total dissolved nitrogen (TDN), particulate nitrogen (PN), major cations and anion, dissolved organic carbon (DOC), chlorophyll *a*, and colour. Samples for chl *a* analyses for the TROLS lakes were filtered through a 20 ìm Nitex™ mesh and thus we could separate edible (EP: < 2-20 ìm) and non-edible (NEP: > 20 ìm) fractions calculated from total chl *a* – EP. Samples for microcystin (MCLR) analyses were collected from a depth of 2 m to the surface at the main site (Kotak et al. 1995), and analyzed following Zurawell et al. (1999). Assignment of dramatic increases ( ) in euphotic zone TP to internal and external sources is outlined in Prepas et al. (2000, in press).

Lakes were divided into two categories, stratified (z_{max} 6 m) and weakly or non-stratified or shallow (for headwater z_{max} < 6 m), based on thermal profiles from June through August (i.e. summer) over this four-yr study. The water column of the former was thermally stratified (i.e., temperature changed 1 ºC at some depth changes of 1 m) for most (75%) observations, while for the latter the water column was mixed most sampling days. Within the two categories and following patterns in lake nutrient concentrations in undisturbed lakes on the Boreal Plain (Prepas et al., 2000, in prep.), data presentations were organized from largest to smallest ratio of drainage basin area to lake volume or relative drainage basin area (DBA/V). Four lakes were added from the harvested lakes sampled by Western Aquatic Group (reference report done summer 1999). The average amount harvested in those watersheds was 20%, from 1994 to 1996.

Results and Discussion

The TROLS lakes are situated in an endorheic region in which annual long-term (1970 to 1994) lake evaporation exceeds precipitation by 11% (529 mm· yr^{-1}, 477 mm· yr^{-1} respectively), and runoff is low (25 mm· yr^{-1} to 208 mm· yr^{-1}). Evaporation is relatively constant year-to-year, and small changes in precipitation have a large influence on runoff (Prepas et al., 2000, in press, Fig. 1), which during this study varied over 10-fold (27 mm· yr^{-1} to 274 mm· yr^{-1}). The study consisted of two dispersed relatively dry (1995, 1998 up to 33% less than 25-yr mean precipitation) and two consecutive wet (1996, 1997 up to 28% more than 25-yr mean) years, and two consecutive 2-yr periods with summer air temperatures less than (1995, 1996) and greater than (1997, 1998) the long-term average (Prepas et al., 2000, in press, Fig. 1). The study area has low relief (average watershed slope 1.1% to 8%) and the lakes are relatively shallow (mean depth < 5 m; Prepas et al., 2000, in press, Table 1). The drainage to lake basin area ratios (DBA/A_o) of the headwater lakes varied from 5 to 22, typical of headwater lakes in the boreal forest (Lamontagne et al. 2000; Prepas et al., 2000, in prep.). Average 4-yr WRT for these lakes varied from 1.8 yr to > 17 yr (Prepas et al., 2000, in press, Table 1), while estimates for individual years in individual lakes ranged from 0.36 yr to > 50 yr (Prepas et al., 2000, in press, Fig. 2). Lake years when WRT 1 yr, provide the best opportunity to evaluate external impacts in a short-term study such as this one.

Percent cut in the DBA of the study lakes varied from 0% to 32%, and within 800 m from 0% (SCL800, SPH800) to 37% (SPH20) and was not consistent with the buffer width treatment (Prepas et al., 2000, in press, Table 1). In these low relief areas, drainage basin boundaries may vary year-to-year (previous section). For example, during wet years some drainage basins can become joined, whereas in dryer years they remain distinct. Because of this variability, all the study lakes, particularly SCL800 (the only lake with 0% cut in the drainage basin and lowest (1.1%) watershed slope likely incurred inputs from nearby harvesting activity during wet years.

Although relatively shallow, the water column of all lakes was at least occasionally thermally stratified during summer. Thermal stratification increased in the post-treatment years, relative to the cooler pre-treatments years (illustrated with SPH100, Fig 2b). Dissolved oxygen (DO) profiles indicated strong chemical stratification in all lakes, with persistent anoxic conditions (DO < 1 mg· L^{-1}) over the bottom sediments in the deeper lakes (illustrated with SPH20, Fig 2a). Thermal and chemical stratification was strongest in 1998, a relatively dry and warm year. Following thermal stratification then mixing, these lakes have a high probability of enhanced surface water P concentrations from internal sources (e.g., Riley and Prepas 1984).

Euphotic zone mean specific conductivity ranged from 110 µS· cm^{-1} to 383 µS· cm^{-1} in these $CaHCO_2$ dominated lakes (Prepas et al., 2000, in press, Table 2). pH was consistently high; 4-yr means ranged from 7.8 to 9.1. The pH of the riverine SCL200 was an average full unit higher, averaging 9.7, likely from an abundant macrophyte or attached algal population. Overall, the lake water was weakly coloured, with all but one of the headwater lakes having mean colour ≤ 48 mg· L^{-1} Pt. LLB200 had coloured water (84 mg· L^{-1} Pt), as well as the highest SO_4, NH_4 and DOC concentrations (average 60 mg· L^{-1}, 583 µg· L^{-1} and 28 mg· L^{-1}, respectively) (Prepas et al., 2000, in press, Table 2). As the highest values were recorded in the dry pre-treatment year, they were likely from internal sources.

In the two pre-treatment years, TP concentrations in the study lakes were consistent (49.6 ± 5.7 µg· L^{-1} and 52.8 ± 7.8 µg· L^{-1} yr 1 and 2, respectively; mean of 51.2 ± 4.7 µg· L^{-1}). In the first post-harvest year, TP concentrations increased overall (χ^2 = 38, P < 0.05) to 73 (± 17) µg· L^{-1}, whereas in the drier yr 4, mean TP dropped to 48 (± 9) µg· L^{-1} (Prepas et al., 2000, in press, Table 3). Particularly in the post-treatment years, within lake category TP concentration increased with increased DBA/V. Buffer width was not related to changes in pre- and post-TP concentration (Fig. 3a). Preliminary data from cut lakes in other areas (discussed in section 1), show that logging near or adjacent to rich fens increases TP release.

Overall TP concentrations in the stratified lakes were about 2/3 of the weakly and non-stratified lakes (Prepas et al., 2000, in press, Table 3). In the pre-treatment years, mean summer TP concentration remained fairly consistent overall in the stratified lakes (43.9 ± 7.7 µg· L^{-1}, 42.4 ± 6.8 µg· L^{-1}), while in the weakly or non-stratified lakes a small increase in yearly means (56.4 ± 8.2 µg· L^{-1} to 65.3 ± 14.0 µg· L^{-1}) was not significant (P ≥ 0.05). Mean TP increased in both lake categories after harvesting likely due to increased saturation of the organic layer in near shore areas (Evans et al. 2000), although the increase was only detectable in the shallow lakes ($\chi 2$ = 20, P < 0.05). As similar runoff conditions in yr 4 (Prepas et al., 2000, in press, Fig. 1) produced a much reduced response compared to this first post-treatment year (overall change 3 µg· L^{-1} vs. 23 µg· L^{-1}), our results are consistent with a harvest enhanced wet-year response. The post-treatment TP increase in the shallow lakes lasted only one year. In contrast, in the

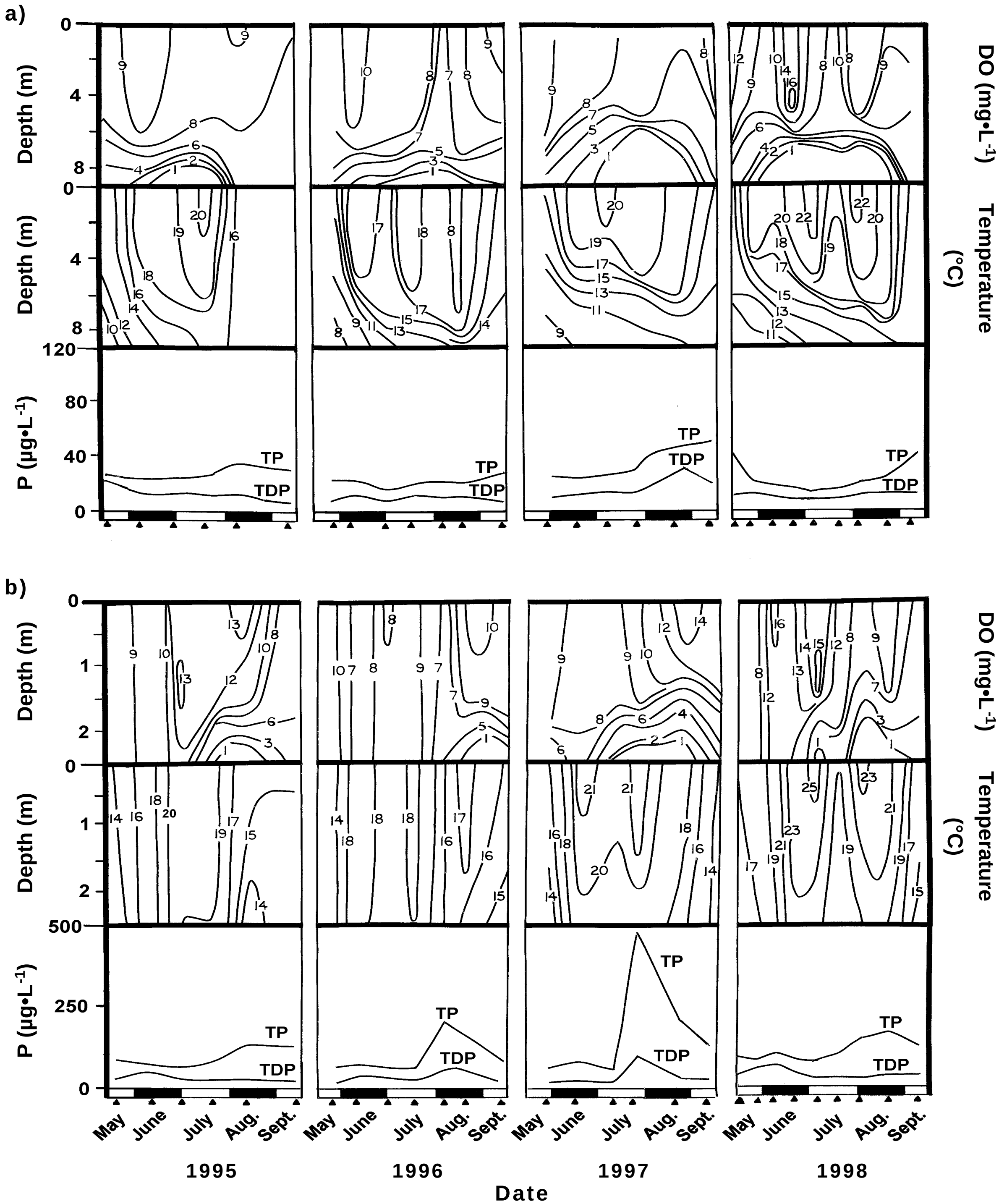

Figure 2 a & b: Thermal (°C) and chemical (dissolved oxygen) data organized over time (4-yr study) and depth in a deep (SPH20:a) upper part) and shallow (SPH100: b) lower part) study lake. Euphotic zone total phosphorus (TP) concentrations presented for the same time

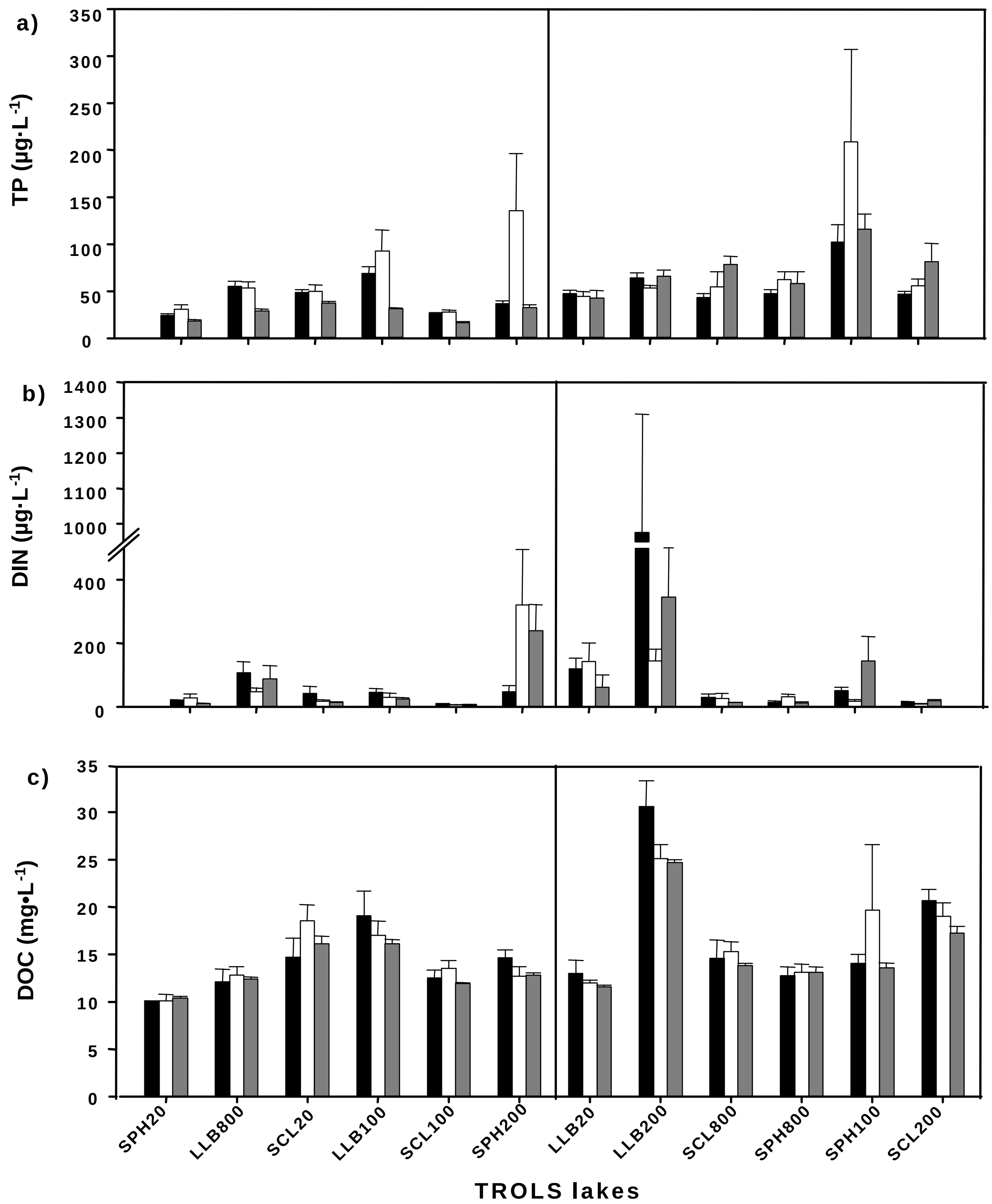

Figure 3. a) TP, b) DIN, and c) DOC concentrations during pre-harvest (1995-96 combined: in black) and post harvest years (1997 and 1998; white and hatched, respectively). Lakes are organized according to thermal stability (first six are well stratified lakes while last six are weakly or non-stratified lakes), and in each sub-set arranged in order of decreasing water residence time. Bars represent means (SE) for euphotic zone samples collected between June and August of each year.

stratified lakes, TP concentrations dropped ($\chi2 = 45$, $P < 0.05$) in the second post-treatment year (Prepas et al., 2000, in press, Table 3). When TP concentration data from the two pre-treatment years were pooled and compared to similar post-treatment data and related to physical features, relative drainage basin area (DBA/V) was the strongest correlate ($r^2 = 0.82$, $P << 0.01$).

The first section of this report illustrated that in headwater lakes such as the TROLS, the DBA/V or relative catchment area is the physical feature which explained the most variation in lake water TP concentration. In the TROLS lakes, DBA/V and WRT are highly correlated ($r^2 = 0.89$, $P << 0.01$). In the wetter first post-treatment year, log-transformed changes in TP concentration were best related to DBA/V ($r^2 = 0.58$, $P < 0.01$) and WRT ($r^2 = 0.47$, $P < 0.02$). When percent harvested in the drainage basin was added as a second independent variable to the regression, it contributed a non-detectable 9% to the change in TP concentration ($P > 0.05$). Thus, in the wetter first post-treatment year, overall increase in TP concentration, although variable, was best related to relative drainage basin area ($\chi^2 = 44$, $P < 0.01$). In the drier second post-treatment year when the shortest lake WRT was 4.1 yr (Prepas et al., 2000, in press, Fig. 2), a small overall decrease in TP was registered relative to pre-treatment conditions (Prepas et al., 2000, in press, Table 3). A large (63 $\mu g \cdot L^{-1}$ to 28 $\mu g \cdot L^{-1}$) drop in the deeper (i.e. thermally stratified) lakes was offset by a more modest change (85 $\mu g \cdot L^{-1}$ to 72 $\mu g \cdot L^{-1}$) in the shallower lakes.

Although relative watershed size partially explained post-treatment changes in TP, internal factors also played a role. In the pre-treatment years, episodic increases in TP concentrations divided themselves roughly into 1/3 internal and 2/3 external inputs, and overall, these inputs represented 23% of the average TP concentration (51 $\mu g \cdot L^{-1}$) in the lake water. In contrast, although baseline TP remained relatively constant in the post-treatment years (44 $\mu g \cdot L^{-1}$ to 35 $\mu g \cdot L^{-1}$), external loading represented a larger portion (just over 40%) of the episodic inputs in yr 3, compared to 12% in yr 4. Internal P loading was most clearly defined in the warmer and drier fourth year of the study. For example, TP concentration began increasing in the euphotic zone in the latter half of July 1998 in both the deeper SPH20 and shallower SPH100 lakes (Fig. 2), following a mixing event where accumulated deepwater P was transferred to the euphotic zone (similar to Riley and Prepas 1984). In contrast, one year earlier (July 1997), a large increase in euphotic zone TP concentration (> 400 $\mu g \cdot L^{-1}$, Fig. 2b) was registered in the relatively rapidly flushed SPH100 (WRT 0.5 yr in 1997), associated with the largest rain event that month (45 mm in 24 h). During the same storm event, the more slowly flushed SPH20 (WRT 3.4 yr in 1997) registered a modest increase (17 $\mu g \cdot L^{-1}$) in euphotic zone TP concentration (Fig. 2a). Increased TP concentration in yr 3 was strongest in those lakes with relatively short WRT and large DBA/V ratio (SPH100, SPH200), while in yr 4 increases were strongest in the shallower and more easily mixed lakes (SCL800, SPH800).

When the harvested lakes from the Western Aquatic project and TROLS were placed on the relative watershed area to TP graph generated in the first section of this report another trend emerged. For those lakes with a relatively large watershed area (CA/LV $\geq$ 4 km^2/m$^3 \cdot 10^6$), logging near the rich fens connected to the lakes appears to enhance post treatment TP concentration in lake water (Fig. 4).

Dissolved IN concentrations were highly variable (e.g., 20-fold in LLB200, Fig. 3b) while the range of variation in mean DOC concentrations was narrow over all lakes and years (9 mg$\cdot$ L^{-1} to 33 mg$\cdot$ L^{-1}, Fig. 3c). The highest DIN and DOC concentrations were recorded in LLB200, which also registered high SO_4 concentrations and colour (Fig. 3b and 3c). DIN increased in the post-treatment years in SPH200, likely related to removal of a beaver dam and a drop in water level of over 1 m in yr 3, a condition which persisted through yr 4. Episodic internal loading of N from decaying organic matter may explain the predominance of NH_4 as the prevalent form of DIN (Prepas et al., 2000, in press, Table 2). DOC peaked only in the stratified lakes SCL20 and LLB100 (Fig. 3c), and in the non-stratified lake LLB200 having the highest colour (Prepas et al., 2000, in press, Table 2). DIN and DOC dynamics appear unrelated to drainage basin treatments.

Overall, during the first two post-treatment years in this study, evidence was strongest for a treatment TP effect in those lakes and years with a relatively short (1 yr) WRT. The nutrient response of lakes to changing weather conditions was influenced by thermal regime. Other studies of the influence of harvesting on water quality in northern forested areas have focused on relatively deep lakes which are characterized by soft waters (Carignan et al. 2000; Lamontagne et al. 2000; Rask et al. 1998). In these studies of dimictic lakes, TP, as well as other leachable nutrient (DOC, DIN) concentrations increased following generally greater watershed disturbance (average 40% and 24% respectively compared with 17% in this study). Ours is the first study we are aware of with relatively shallow lakes where the P response following watershed harvest was generally magnified where the water column was weakly or non-stratified, and DIN and DOC concentrations did not respond detectably.

Prior to harvesting, mean total chlorophyll *a* for all lakes (Tchl *a*) was 16 µg$\cdot$ L^{-1}, comprised equally of edible (EP) and non-edible (NEP) phytoplankton (8 µg$\cdot$ L^{-1}) (Prepas et al., 2000, in press, Table 3). Stratified and non-stratified lakes supported similar mean Tchl *a*, 16.8 µg$\cdot$ L^{-1} and 15.7 µg$\cdot$ L^{-1}, respectively. However, EP chl *a* represented 60% of Tchl *a* in the non-stratified lakes and only 42% in the stratified lakes. After harvest, overall Tchl*a* was 20 µg$\cdot$ L^{-1}

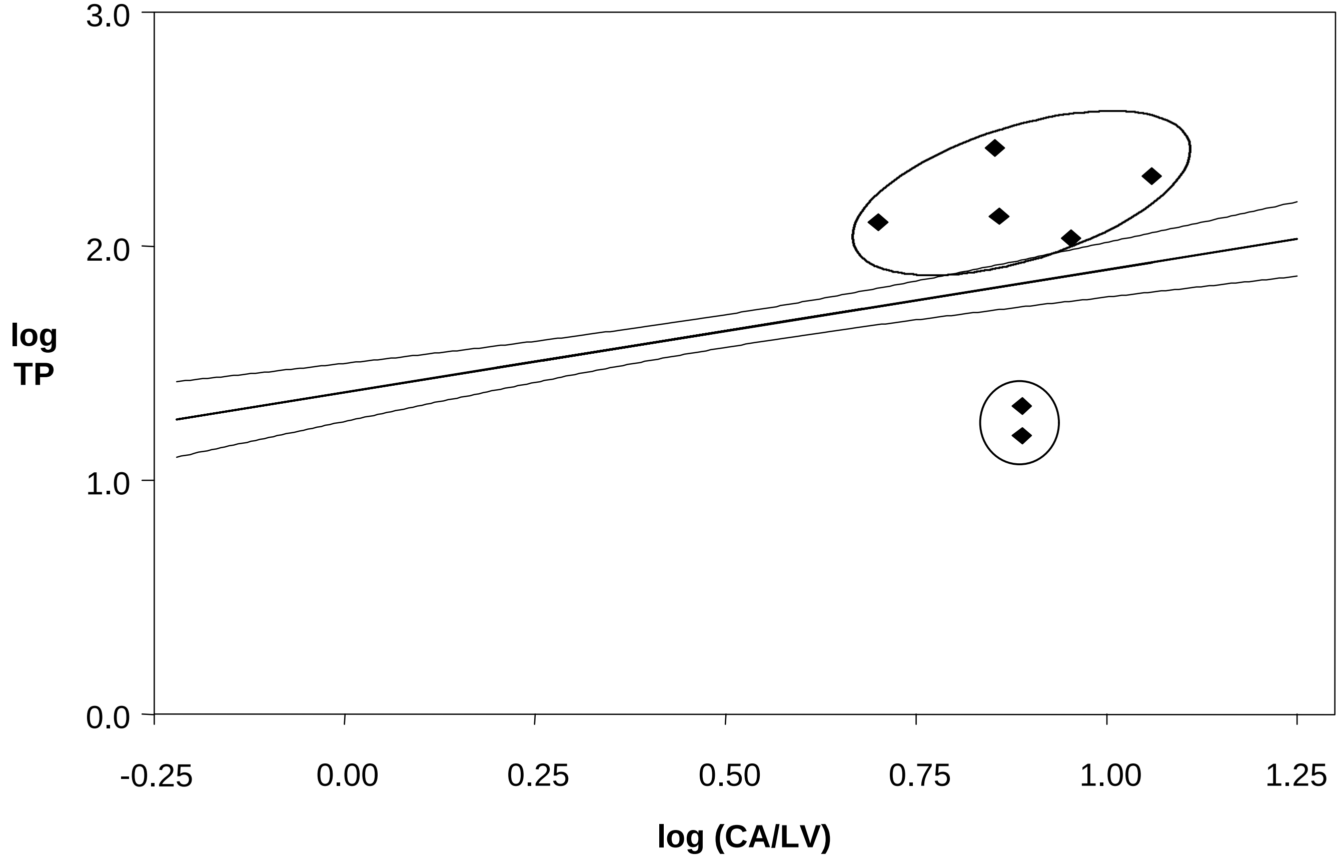

Figure 4. 95% confidence interval and regression of TP concentration (μg/L) and catchment area divided by lake volume (CA/LV $km^2/m^3 \bullet 10^6$) in upland dominated lakes on the Boreal Plain (from first section, this report). Circles represent lakes with a modest amount of harvesting in the watershed (20%) with CA/LV ≥ 4. The lakes in the circle above the regression line have a relatively large amount of harvesting in the rich fens (>60%) connected to the lake, while lakes in the smaller circle below the regression line have no (<10%) harvesting in fens connected to the lake. Lakes with smaller CA/LV (4) and harvesting (<20%) fall within the 95% confidence intervals of the regression line.

and 22 μg· L^{-1} in yr 3 and yr 4, respectively, for all lakes. Post-harvest Tchl*a*, EP and NEP chl *a* were lower in the stratified than in the non-stratified lakes (Prepas et al., 2000, in press, Table 3 and Fig 5a), and EP chl *a* represented only 40% of Tchl*a* in all lakes overall.

Cyanobacteria (CYA) contributed 43% and 67% of phytoplankton biomass overall pre and post-harvest, respectively (Prepas et al., 2000, in press, Fig. 5b). In lakes where CYA were scarce or absent (LLB100 overall, LLB200 and SPH800 in pre-harvest years), the more abundant taxa were the flagellates (mainly Cryptophyta and Dinoflagellates). In lakes where CYA were present, heterocystous nitrogen-fixers (N_2-fixers) (Nostocales) were the most abundant, with *Aphanizomenon-Anabaena* (Ap-An) being the most common species association (SCL20, LLB20, SHP200, SPH100), and *Gloeothrichia echinulata* being the dominant species in LLB800 (Prepas et al., 2000, in press, Fig. 5c). Within the CYA-dominated lakes, Oscillatoriales followed the N_2-fixers in importance; however, in SCL100 Oscillatoriales were the dominant algae.

Although NEP chl *a* increased overall after harvesting, phytoplankton response in the post-treatment years varied depending on lake thermal structure. In stratified lakes, EP and NEP chl *a* decreased after harvesting ($P < 0.05$), whereas in non-stratified lakes NEP chl *a* increased ($P < 0.01$). The increase in NEP chl *a* in weakly or non-stratified lakes corresponded with an increase in biomass of total cyanobacteria, mainly Nostocales and *Microcystis* (only during yr 4). Furthermore, MCLR, the most common cyanotoxin in prairie lakes, increased 10-fold in yr 3 relative to pre-treatment ($t = 4.9$, $P < 0.001$), and remained high in yr 4 (2.9 ± 1.4 μg· g^{-1}, 23 ± 5.3 μg· g^{-1}, 39 ± 28 μg· g^{-1} MCLR, respectively). The MCLR increase was primarily observed in the non-stratified lakes ($t = 4.9$, $P < 0.001$). Relationships between MCLR and phytoplankton were particularly strong with *Aphanizomenon gracile* ($r^2 = 0.58$ and 0.51, $P < 0.05$, < 0.02) and *Limnothrotix redeckii* ($r^2 = 0.45$ and 0.43, $P < 0.03$ in the stratified and non-stratified lakes respectively). Cyanobacteria and associated toxins have been identified previously south of our study lakes in regions with watershed disturbance (Kotak et al. 1995). In our study, pre-treatment MCLR concentrations were low, while both cyanobacterial biomass and the predominant cyanotoxin MCLR increased in the post-harvest period.

Increased cyanobacterial dominance is expected with enhanced P content of lake water, and has been reported elsewhere both in sedimentary and hardrock basins (Rask et al. 1998). However, no other study has monitored a cyanotoxin before and after watershed disturbance, nor within the context of contrasting mixing regimes. Given the variability within cyanobacterial and cyanotoxin communities (Kotak et al. 1995), and the dynamic biogeochemistry of these lakes, an in depth monitoring and expanded treatment program will be required to comprehend phytoplankton responses to watershed disturbance.

FOREST FIRE INDUCED IMPACTS ON WATER QUALITY WITH EMPHASIS ON BOREAL SUBARCTIC LAKES

Interest in fire as a natural disturbance impacting catchment biogeochemistry and surface water quality is rekindling as a relative 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). The 1995 fires in the province of Alberta were all based in wetland-dominated areas. The only fires left to burn somewhat freely were in the boreal subarctic, thus presenting an unusual opportunity to evaluate the impact of fire on surface waters.

Water chemistry has been studied in burnt catchments with small proportions of peatlands. A comparison of burnt and non-burnt 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 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 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.

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 1:20,000 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 (McEachern et al., 2000, Table 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 (Fig. 1) burnt between 1961 and 1985, were sampled monthly after ice-out from late June to late August/early September. 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 the same larger drainage. Lakes within each of 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 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 bank full 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. Water samples were collected and analyzed as outlined by McEachern et al. (2000).

Eight lakes were sampled in central Alberta following the 1995 wetland-dominated Marianna lakes fire. These wetland-dominated lakes had on average 19% of their watershed burnt (SFMN reference report prepared July 1999).

Results

Several physical characteristics of lakes and their catchments differed between burnt and reference systems (McEachern et al., 2000, Table 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•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 (Wilks λ = 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· L^{-1}) were all below the world average for fresh water (2.33 meq· L^{-1}). There were no detectable differences in mean total base cation concentrations (McEachern et al., Fig. 2) between burnt (0.61 meq· L^{-1}) and reference (0.72 meq· 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· L^{-1} $CaCO_3$, P = 0.07; McEachern et al., 2000, Fig. 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· L^{-1}, respectively. Fire did not affect chloride concentrations, which averaged 0.15 and 0.13 mg· 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 (McEachern et al., 2000, Fig. 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 (McEachern et al., 2000, Fig. 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.

TP concentration in the Caribou Mountain reference lakes was on average 26 µg· L^{-1} lower than in the lakes with wetland dominated watersheds in central Alberta. Following fire TP concentrations in upland dominated watersheds changed similarly to the Caribou Mountains (Prepas et al 2000, SFMN report), however other ions and forms of phosphorus were different in upland versus wetland dominated sites. Response of lakes in watersheds of the weakly burned watersheds on t he Boreal Plain is in progress of review.

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 (See section 1). 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 (McEachern et al., 2000, Table 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 (McEachern et al., 2000, Table 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 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 (r^2 = 0.83, P << 0.01, n = 13; McEachern et al., 2000, Fig. 5). 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.

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 (McEachern et al., 2000, Table 3). Colour and DOC tended to increase at a faster rate with drainage ratio in burnt lakes compared to reference lakes (McEachern et al., 2000, Table 3), however, the slopes of these relationships were not distinguishable (ANCOVA, P =0.09 and P = 0.20 respectively).

Water in the previously (1961-1985) burnt lakes had phosphorus concentrations intermediate between recently (1995) and non-burnt lakes. 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): DI = %Disturbance $\cdot$ e^{-Kt}, 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 $K = 0.693 \cdot T^{-1}$ 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 (McEachern et al., 2000, Table 3). We included reference conditions in the analysis by forcing the intercept through mean TP and DP concentrations for reference lakes 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. The increase in TP concentrations in the burnt lakes with respect to those in the reference lakes was relatively large, considering that TP as well as TN concentrations in the Caribou Mountain reference lakes were, on average, lower than those in the other reference study lakes (see section 1). Elevated phosphorus and nitrogen concentrations following fire have also been 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 burnt 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).

CONCLUSION

In conclusion, this study found that bogs release more phosphorus than other vegetation categories, while rich fens appear to sequester more phosphorus and nitrogen than other vegetation categories in undisturbed lakes on the Boreal Plain. In upland-dominated lakes, the relative size of the watershed (DBA/LV) was the strongest predictive factor for TP concentration. In contrast, lakewater colour concentration was strongly linked with the wetland connectivity parameter eDBA/LV. *In situ* nitrogen concentrations were positively linked to % wetland cover. Phytoplankton community composition in these lakes was consistent with greater availability of inorganic nitrogen in the lakes with wetland-dominated catchments.

Although the effects of harvesting were not related to buffer strip width, we found that even a modest disturbance in the watershed can enhance euphotic zone TP under specific climatic and watershed or lake conditions. In this two post-treatment year study, lakes with the relatively largest drainage basin, and thus shorter residence time, had the strongest response which was both magnified and extended in the weakly or non-thermally stratified lakes. Changes in the nutrients status of the lakes were reflected immediately in the phytoplankton and associated toxin communities. We predict that in these slowly flushed systems, where P cycling from bottom sediments is the norm, the response may continue for a decade or more and be positively related to runoff.

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.

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Appendix I

SFM Network Students Supervised by E.E. Prepas 1996-2000

Student	Degree	Date	Supervisor	Project Title
E. Allen	M.Sc.	1999-2002	Prepas	Methyl mercury quantification in disturbed lakes
N. Armstrong*	M.Sc.	Completed April 2000	Prepas, Planas	Contribution of littoral and pelagic algal photosynthesis in Boreal Plain lakes
T. Charette	M.Sc.	1998-2001	Prepas	Long-term response of phytoplankton to fire in the Boreal Plain
P. McEachern	Ph.D.	1996-2001	Prepas	Contributing area disturbance and surface water response in a permafrost area
S. Nicopoulos	M.Sc.	1998-2001	Planas, Prepas	Effects of forest harvesting on epiphytic communities
A. Patoine	Ph.D.	1995-2001	Pinel-Alloul, Prepas	Impact of watershed disturbance on zooplankton communities
I. Whitson	Ph.D.	1996-2001	Prepas	The soil-water interface in a disturbed landscape

* N. Armstrong is currently employed as a Water Quality Specialist with Manitoba Environment.

SFM Network Publications by E.E. Prepas as of Dec. 2000

Summary of SFM Network Publications

Category	Number
Refereed Publications	10
Publications submitted to refereed journals	5
Conference presentations	30
Workshop presentation and invited talks	21
Technical reports	8

Published and In Press Manuscripts:

Chen, Weiping, D. Schopflocher, B. Fowler, J. White, E.E. Prepas, D. Prince, S. Gabos. 1999. Polycyclic aromatic hydrocarbons in sediment following forest fires. Organohalogen Compounds. 43:417-420.

Covich, A.P., Fritz, S.C., Lamb, P.J., Marzolf, R.D., Matthews, W.J., Poiani, K.A., Prepas, E.E., Richamn M.B., Winter, T.C. 1997. Potential effects of climate change on aquatic ecosystems of the Great Plains of North America. Hydrol. Proc. 11:993-1021.

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Gabos, S., Schopflocher, D., Fowler, B., White, J., Prepas, E., Prince, D., Weiping, Chen. 1999. Polycyclic aromatic hydrocarbons in water, fish, and deer liver samples following forest fires. Organohalogen Compounds. 43:329-333.

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Patoine, A., B. Pinel-Alloul, E.E. Prepas and R. Carignan. 2000. Do logging and forest fires influence zooplankton biomass in Canadian Boreal Shield Lakes? Can. J. Fish. Aquat. Sci. 57(Suppl. 2):155-164.

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Reedyk, S., Cooke, S., Burchett, G., Prepas, E.E. 1997. Design for an inexpensive continuous digital output water level recorder. Water Resour. Res. 33:1523-1526.