Influence of Watershed Features and Disturbance History on Water Quality in Boreal Shield Streams and Rivers of Eastern Manitoba

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Executive Summary

Water quality and flow were studied in twenty two creeks, streams and rivers in the Manitoba Model Forest area (eastern Manitoba) during the summer of 2004 and the winters of 2004 and 2005. Bank full width ranged from 0.7 to > 29m, and corresponding watershed areas from 1.38 to more than 1,700 km^2. The objectives of the study were four fold: **to** contribute to a long-term data base of water quality in water bodies on the east side of Lake Winnipeg (which is currently lacking), **to** examine how natural watershed features (including watershed size, soil type and forest type) influence regional water quality, **to** examine how natural disturbances (wildfire, insect outbreaks and beaver activity) affect water quality, and **to** compare and contrast this with how forestry activities and agriculture may affect water quality.

The study watersheds spanned a wide gradient in soil type, forest type and disturbance regimes. Two streams were located in agricultural watersheds. All other water bodies were located in forested watersheds. Two of the forested watersheds burned completely in the last 5 years. One of these also had burned completely a decade earlier as well. The average area in the watersheds that burned was 42% (range of 0 to 200% [the latter value represents multiple burns within the same watershed]). Very little forest harvesting occurred in the watersheds within the last 18 years. However, much more harvesting had occurred in the last 60 years (range of 0 to 100%, average of 29% of watershed area). Two watersheds were completely harvested in the last 60 years (i.e., 100% of the land area was logged in each watershed). One of these watersheds also completely burned in this time period, producing a watershed with both a significant fire and logging footprint. Beaver activity in some of the smaller streams and creeks was extensive,

creating an opportunity to investigate the effects of a common, often overlooked and perhaps significant, natural disturbance agent on water quality.

Regionally, the water quality observed in the forested study watersheds were similar to that of other large rivers (Bloodvein, Poplar, Pigeon, Berens) found in the area east of Lake Winnipeg, but were quite distinct from water quality of rivers located in the prairies of southern Manitoba (e.g., Red and Assiniboine rivers). The study streams had much lower concentrations of calcium and phosphorus, as well as lower pH and conductivity than the Red and Assiniboine rivers, reflecting differences in soil type and land use. In general, water quality in the study streams in forested watersheds can be subjectively considered as good to excellent, when compared to water bodies in the prairies.

Both soil and forest type has a significant influence on the water quality in the study streams as determined by relating water quality parameters to soil and forest data available through Geographic Information Systems (GIS). Streams located in watersheds with higher proportions of organic soils typically had higher concentrations of total nitrogen (TN), total phosphorus (TP), dissolved organic carbon (DOC-water color) and sulphate (SO_4). Streams located in watersheds with higher proportions of deep basin (mineral) soils had higher water concentrations of calcium (Ca) and TP and higher pH, alkalinity and conductivity. Streams with predominantly clay bottoms also had higher water turbidity. In contrast, streams located in watersheds with higher proportions of bedrock soils had lower concentrations of TP, TN, and DOC.

Forest type also exerted a significant influence on water quality in the streams, and these relationships were generally consistent with the soil data. The influence of forest type was examined statistically by stratifying the Forest Resource Inventory (FRI) data in 3 different ways: by stratifying the forest into major **Land Types** (e.g., hardwood, softwood, non-productive, etc.), **V-Type Groups** (e.g., groupings of aspen and mixed wood, jack pine, lowland black spruce, etc. that partially account for soil type) and S**tand Types** (e.g., aspen, tamarack, treed muskeg, etc.). The change from Land Type to V-Type Group to Stand Type represents an increase in the degree of forest type detail. The results were consistent regardless of the method of data stratification of the forest.

At a **Land Type** level of resolution, higher stream concentrations of Ca, TP, TDP, TN and higher pH, alkalinity and conductivity were consistently associated with watersheds containing higher proportions of hardwood and hardwood-leading mixedwood (i.e., mixedwood forests with a greater percentage of hardwood species) forests. Higher stream concentrations of SO_4, DOC, TDP and TN were associated with watersheds containing higher proportions of non-productive forest. Upland softwood stands were associated with lower stream TP, TN and Ca concentrations.

At the V**-Type Group** level of resolution, higher stream concentrations of Ca, TP, TDP and TN and higher pH, alkalinity and conductivity were associated with watersheds containing higher proportions of aspen and mixedwood forests, while higher concentrations of DOC and SO_4 were associated with watersheds containing higher proportions of tamarack forests. Lowland black spruce was associated with higher TP and TDP stream concentrations. In contrast, streams in watersheds with higher proportions of jack pine always had lower stream concentrations of Ca, SO_4, TP, TDP, TN and DOC and lower alkalinity and conductivity.

Lastly, at the **Stand Type** scale of resolution, higher stream concentrations of SO_4, TN and DOC and lower pH, alkalinity and conductivity were associated with tamarack forests and treed muskeg, while higher stream concentrations of TP and TDP were associated with black spruce as well as trembling aspen forests. Higher concentrations of Ca, and higher pH, alkalinity and conductivity were associated with forests of trembling aspen.

In general, increasing proportions of hardwood forests, particularly those dominated by aspen, in watersheds resulted in streams with higher concentrations of cations (such as Ca) and phosphorus and nitrogen, and higher pH, alkalinity and conductivity. Increasing proportions of non-productive forest area (including treed muskeg, lowland black spruce and tamarack) in watersheds resulted in streams with higher concentrations of phosphorus and nitrogen, DOC and SO_4 and lower pH and alkalinity. In contrast, higher proportions of jack pine were associated with streams containing lower nitrogen and phosphorus, DOC, Ca, SO_4, pH and alkalinity. Jack pine forests do not appear to be major exporters of dissolved substances. The forest type data therefore agree quite well with the soils data, in terms of their influence on water quality.

We also evaluated the effects of disturbances such as insect (spruce budworm and forest tent caterpillar) outbreaks, low intensity agriculture, forestry and wildfire on water quality. Of the three disturbance regimes, agriculture had the most dramatic impact on water quality. The incidence of insect outbreaks was too low in most watersheds to determine if this form of disturbance has an impact on water quality. Compared to all the streams in forested watersheds (even those experiencing significant levels of logging and fire disturbance), agricultural effects on water quality were more pronounced. For example, TP concentration and TP export were 4x and 7x higher, respectively in agricultural streams, than all other forested streams, Ca concentration and Ca export was 5.9x and 6.7x higher, respectively, and TN concentrations and TN export were 1.9x and 2.8x higher, respectively. In addition, streams with agricultural activity had significantly higher nitrate (NO_3), ammonia (NH_4) and DOC concentrations, as well as higher alkalinity, pH, conductivity and turbidity. Soil type could only partly explain this trend and it therefore appears that permanent removal of forests for even low intensity agriculture can have a major impact on stream water quality. The effects of logging and fire were far less dramatic.

Relative to reference streams (those with no or little fire or harvesting), streams with harvesting had statistically higher concentrations of Ca, TN and DOC and higher concentrations of TP (though not statistically so). In contrast, there were no significant differences in export coefficients for these parameters between harvest and reference streams. There was also no statistical difference between harvest and reference streams with respect turbidity or total suspended solids (TSS). The results suggest that on average, logging may result in increased phosphorus, nitrogen and color in streams. Further, the concentration of SO_4, TDP, TN and DOC in streams was directly proportional to the % of watershed logged. When coupled with other data, it was also apparent that the effects of logging were also influenced by the type of soils in the watersheds. For example, harvesting in watersheds with higher amounts of bedrock soils had less impact on stream water quality than in watersheds with organic or deep basin soils, even when harvesting levels were higher. Conversely, streams with lower amounts of watershed harvesting could result in larger impacts on water quality in watersheds dominated by organic and deep basin soils. On average, there appeared to be no significant effect of harvesting on stream water concentrations of TN, DOC and SO_4 when harvesting was below 30-40% of the watershed area (i.e., the concentrations of these parameters were similar between harvested and reference streams when harvesting was below a threshold of 30-40%). Complete harvesting of watersheds (approximately 100% of the watershed) resulted in significant impacts on TN, DOC and SO_4. These watersheds were also very small, were dominated by organic or deep basin type soils and one watershed was also burned completely. All of these factors contributed to the significant effects on water quality in these streams.

In addition to harvesting, we also examined the aspect of access development (roads, trails) on water quality, as well as hydro transmission (utility) corridors. For watersheds with a long history of forestry activity, a very high percentage of the access development was in the form of winter roads. One would expect that winter roads (which are mainly constructed over non-productive areas and do not involve grading or addition of gravel) would have no detectable effect on stream turbidity or TSS. This was the case. One would also expect that streams in watersheds with a higher density of all-weather roads (and hence, higher erosion potential) could have elevated turbidity and TSS. This hypothesis was supported by our data, but the increase in turbidity and TSS was very small and would not be expected to have any significant effect on water quality or fish habitat. It is likely that any effects (if present) of road construction and stream crossing installations would be localized. We did however find significant positive correlations between winter road density in watersheds and stream water concentrations of SO_4, TDP, TN and DOC. These results should be interpreted with caution, however, as it was not possible to determine the relative contribution of forest type and winter road density to water quality. Higher SO_4, TDP, TN and DOC concentrations observed may have been solely due to forest type. This warrants further investigation. Lastly, we found no indication that the density of all-weather roads was related to SO_4, TDP, TN or DOC, suggesting that on average, all-weather roads have little impact on regional water quality in the Model Forest area. This was also the case for hydro transmission corridors. This should not be surprising, as the footprint of hydro corridors is usually very small relative to the size of watersheds.

Fire also had a noticeable impact on water quality. Streams in burnt watersheds had significantly higher TP, TN, DOC, Ca and conductivity than reference streams. Ca and DOC export were also higher. An interesting comparison can be made between the effects of fire and harvest on water quality. We found that the effects of harvesting on Ca, TP and conductivity, as well as export coefficients for TP, DOC and TN were well within the natural range of variation that streams would experience from fire disturbance. However, streams in harvested watersheds had significantly higher TN and DOC (1.4x and 1.7x higher, respectively) concentrations than streams in burned watersheds. It is not surprising that the effects of fire and harvesting can be different when one considers the nature of the two disturbances.

Finally, we investigated the impacts of beaver activity (specifically, the density of beaver dams and degree of flooding) on water quality through an aerial survey approach. Beaver dams and the incidence of back flooding were minimal on larger streams and rivers. However, smaller streams commonly had many beaver dams, up to 6.2 per km. Our data demonstrate that beaver likely have a significant impact on water quality in smaller streams through reducing water flow, creating stagnant water areas and through back-flooding of riparian areas (meadows and forests). These alterations in hydrology were translated into significant impacts on water quality, particularly during the winter. The occurrence of beaver dams appeared to cause low dissolved oxygen in the water during the summer, likely due to stagnation of water flow, decomposition processes in the organic-rich sediments (which uses up dissolved oxygen) and due to back-flooding of riparian areas (which would also contribute more organic matter for decomposition). The impact was even more severe in winter when ice cover prevents any diffusion of oxygen from the atmosphere into the water. Dissolved oxygen concentrations in the water of streams with significant beaver dam density approached zero in late winter. Not only would this have serious consequences for organisms (e.g., invertebrates, fish) inhabiting these streams, but low dissolved oxygen also resulted in massive release of TP, NH_4, TN and DOC into the water column from the sediments. In some cases, the increase in winter concentrations of TP and NH_4 above summertime concentrations was spectacular, dwarfing any impacts that were observed by fire or logging. For example, winter concentrations of TP and NH_4 increased 500% and 7,500%, respectively over summer concentrations in streams where significant beaver activity was present. This same trend was not observed in larger streams and rivers where beaver activity is minimal. In an effort to quantify the effects of beaver dams on water quality, we developed a simple classification system based on the degree of back flooding created (Class I – no back flooding, Class II – limited back flooding, approximately 5x the normal stream channel width, and Class III – significant back flooding, more than approximately 10x the normal channel width). We hypothesized that the greatest impacts on water quality would be observed in streams with the greatest density of Class III dams. Indeed, we found that there were significant positive correlations between the density (dams/km) of Class III dams and winter TP concentration, winter NH_4 concentration, winter TN concentration and summer TN concentration. There was also a significant negative correlation between density of Class III dams and winter dissolved oxygen concentration. Therefore, beaver dams and their influence on hydrology, decomposition and back flooding can have significant effects on water quality in small streams in eastern Manitoba.

A final objective of this project is to develop simple watershed planning tools for Tembec in order to better incorporate water quality objectives into their forest management planning process. Currently, Tembec limits harvesting to 30% of a watershed over a 7 year period as a method of accommodating water quality concerns. This limit also includes fire disturbance, so that no watershed should have more than a 30% cumulative disturbance over a 7 year period. While our data suggest that this limit could be appropriate, we will be combining our data with that of a companion study that is nearing completion in order to utilize a much larger dataset before developing watershed management tools. This companion study is examining the effects of fire and harvesting on water quality in 100 lakes in and adjacent to the Manitoba Model Forest area and forms the MSc thesis of Kevin Jacobs (Department of Botany, University of Manitoba). The much larger combined dataset will provide a much stronger basis for the development of watershed planning tools.

Lastly, two aspects of the 30% harvesting limit currently employed by Tembec could not be evaluated in our study. Firstly, is 7 years an appropriate timeframe to use before further harvesting can be done in a watershed? Seven years is the length of time after harvest that regeneration surveys are completed to determine if adequate stocking of trees has taken place on harvest sites. This time period likely has no relevance to the time needed for evapotranspiration recovery. Evapotranspiration and flow recovery is currently unknown in the Model Forest area and should be studied using field experiments in experimental watersheds. There are some studies published in the literature that suggest water yield and nutrient concentrations can return to pre-disturbance levels within a short period of time (3-5 years). This needs to be validated in the Manitoba Model Forest area through field experiments. Secondly, the size of watersheds used for planning purposes by Tembec is quite large (up to 500 lm^2). The literature suggests that water quality impacts are likely to be observed only at smaller (<100 km^2) watershed scales. While management planning for very small watersheds (such as those with the largest harvest-related impacts on water quality in our study- 15 km^2), would be unrealistic, we suggest that some of the larger planning watershed areas utilized by Tembec be broken up into smaller management units (of not greater than 100 km^2).

Photographs (in JPG format) of all study stream locations and a complete copy of all raw water chemistry data on CD are included with hardcopy versions of this report inside the back cover of the project report. The report, photos and data are also available on the Manitoba Model Forest website (www.manitobamodelforest.net).

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Many organizations provided direct funding and in-kind support to this study. Initially this study began as a small project on the Black and O'Hanly rivers to respond to concerns by Black River First Nation community members about the status of water quality in the two rivers that run through their community. From there, the study was greatly expanded has become much more comprehensive. This was only possible because of the generous support of our project partners. We wish to thank the Manitoba Model Forest, Manitoba Conservation (Sustainable Development Innovations Fund), Environment Canada (EcoAction funding program), Tembec Inc., Manitoba Hydro, Indian and Northern Affairs Canada (INAC Resource Access Negotiations fund) and Black River First Nation for direct financial support.

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Introduction

The boreal forest, in which the Manitoba Model Forest lies, is the largest terrestrial ecosystem in Canada, stretching from Newfoundland to northern British Columbia and the Yukon Territory. It encompasses nearly 6 million km^2. Canada contains approximately 30% of the world's boreal forest, second only to Russia (NRTEE, 2005). It is made up of a rich tapestry of upland and lowland forests, many types of wetlands, and water bodies ranging from small ponds, creeks and streams to large rivers and lakes. The boreal forest alone in Canada contains an estimated 25% of the world's freshwater resources. The net market value of goods (forest products, oil and gas, mining, etc.) in 2002 was estimated to be $37.8 billion or 4% of the Canadian GDP, and the non-market value of services in 2002 was estimated to be $93 billion, largely consisting of flood control and water purification by peatlands, and to a lesser extent, pest control by birds, nature-related activities and carbon storage (Anielski and Wilson, 2005).

Forested landscapes play a critical role in controlling the hydrologic cycle, including water storage, flow and water quality in watersheds (Hetherington, 1987). This occurs through processes such microclimate regulation (and thus, energy balance), interception of precipitation, transpiration and evaporation, infiltration of water into soils and runoff generation. Removal of the forest canopy therefore, can be expected to have an affect on all of these processes, thereby influencing water flow and possibly water quality in downstream receiving bodies (e.g., lakes, rivers). Fundamental to understanding of the impacts of land use practices (e.g., agriculture, forestry) and natural disturbances (e.g., forest fire) on water quality, is the understanding of the impacts of forest disturbance on hydrology. In the absence of disturbance, watershed characteristics such as the dominance of certain soil or forest types can also have an influence on water quality (Devito and Hill, 1977, Prepas et al., 2001).

The impact of land use and natural disturbance on hydrology has been examined through the use of modeling and field experiments. A simple model used to predict the effects of land use on hydrology is the WRENSS model (Water Resource Evaluation of Non-point Silvicultural Sources), developed by the US Forest Service for evaluating the effects of forest harvesting on water yield change. Changes in water yield are predicted from model inputs of precipitation, watershed aspect (direction-north, south, east and west), forest cover type and density, area of forest removed and cut block size, to name a few. Using this, and other such models, the impact of forest removal on water yield is most influenced by the proportion of the watershed harvested (Keenan and Kimmins, 1993) and less so by the size of the cut blocks. These effects are manifested primarily through a reduction in evapotranspiration caused by removal of forest. For example, WRENSS was used to predict the impacts of forest harvesting on annual water yield on the eastern slopes of the Rocky Mountains (Alberta Energy and Natural Resources, 1984). Harvesting of 30-40% of watershed area resulted in a predicted increase in annual water yield of only 6-10%. Complete removal of forest (i.e., harvesting of 100% of the area) resulted in a predicted increase in annual water yield of 25-29%. Rothwell (1997) found that WRENSS predicted an increase in water yield in the Pasquia-Porcupine Forest Management Area of Saskatchewan of 17-22% based on harvest levels of 22-44% of watershed area. These increases in water yield are not particularly high.

In addition to modeling, studies conducted in experimental watersheds have also demonstrated the impacts of harvesting on annual water yield, peak flows and storm flows. In contrast to modeling, these studies have produced more variable results, likely due to differences in watershed size, climate, soils and forest type. For example, a 31% watershed harvest in Quebec resulted in no detectable increase in water yield (Plamondon and Oullet, 1980), whereas a harvest of 54% (average of 9 watersheds) in Alberta resulted in a 27% increase in water yield, very close to the value predicted by WRENSS (Swanson and Hillman, 1977). Dickson and Daugharty (1982) observed no increase in flows after 92% of a watershed in New Brunswick was harvested. Bosch and Hewlett (1982) conducted a literature review of watershed harvesting trials and indicate that most paired basin studies can not demonstrate an increase in water yield when harvesting is less than 20% of the watershed area. In general, the effects of forest harvesting on water yield in Canada (with the possible exception of coastal British Columbia) is less than elsewhere (Hetherington, 1987).

The effect of logging on water quality in rivers and lakes has been studied extensively in North America. Studies examining the impacts of forest fire on water quality are less common and generally more recent. The magnitude of the impacts and even the direction (positive, negative, neutral) vary considerably from study to study. This is not surprising, as the research has spanned a wide gradient in temperature and precipitation regimes (coastal marine climate versus dry, continental climate), hydrologic regime (snow melt- versus storm runoff-dominated), topography, watershed size, proportion of watershed harvested, forest type (hardwood, mixed woods, upland and lowland softwoods), soil type and productivity and forest harvest techniques (clear cut versus selective cut). Therefore, comparison of the effects of harvesting (and fire) between studies is likely most appropriate within specific regions. In our case, the most appropriate comparisons would be with studies conducted in the boreal shield of Canada.

There have been a few studies conducted in the boreal shield that have examined the impacts of logging and fire on water quality. Steedman (2000) found that harvesting of 45-77% of watershed area around 3 boreal lakes in northwestern Ontario resulted in only marginal changes or no changes at all in a suite of water quality variables including phosphorus, nitrogen, dissolved organic carbon (water color), potassium, chloride, silica and chlorophyll (phytoplankton biomass). In contrast, Carignan et al. (2000) reported significant changes in total phosphorus, total nitrogen and dissolved organic carbon in boreal shield lakes in the Haute-Mauricie region of Quebec after harvesting of between 5-73% of the watershed area. The differences between the two studies may be related to differences in regional climate and differences in the ratio of the lake surface area to watershed area ratios.

Fires also can have a significant impact on water quality and these impacts may or may not be similar to those caused by harvesting. In the study mentioned above by Carignan et al. (2000) in Quebec lakes, wildfire caused increases in total phosphorus and total nitrogen, similar to harvesting. However, fire caused an increase in watershed export of sulphate, opposite to harvested watersheds, and harvesting caused an increase in dissolved organic carbon export, opposite to fire. Bayley et al. (1992) found that wildfires near Kenora, Ontario increased nitrogen and phosphorus concentrations in small streams in forested and wetland-dominated watersheds. It would appear that both harvesting and fire will result in changes to water quality but that the direction of change may be different between the two disturbance types.

Over the last several decades there has been a noticeable shift in forest management from one focused on sustained yield (and thus, maximizing and maintaining harvest levels and the flow of timber to mills) to the concept of sustainable forest management (SFM), which strives to balance ecological and social values with timber production. Water quality is an important ecological and social attribute of aquatic ecosystems that the public place considerable value on in the context of sustainable forest management. Indeed, water quality and sustainable forest management are encapsulated in international-level processes (e.g., Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests; Montreal Process, 1999), national-level policy documents in Canada, such as the National Forest Strategy (National Forest Strategy Coalition, 2003) and the Canadian Council of Forest Minister's Criteria and Indicators of SFM (CCFM, 1995). The "motherhood" statements found in such international- and national-level documents become more concrete and are implemented at the provincial and forest unit scale through application of provincial forest policies (as summarized, for example, in "Applying Manitoba's Forest Policies; Government of Manitoba, 2003), and the development and implementation of long-term forest management plans. These typically incorporate specific objectives for maintaining water quality, along with quantifiable targets, monitoring systems and forest management planning and operating procedures.

Tembec-Pine Falls Forest Resource Management has taken this "Criteria and Indicators" approach, and through public consultation processes, developed several indicators and targets which are designed to maintain water quality on their Forest Management License (FML) in eastern Manitoba. This area overlaps with the Manitoba Model Forest area. Initial targets have been developed, which can be evaluated and tracked through time. One such important target is based on limiting disturbance (including harvesting and fire) to less than 30% of a watershed over seven year periods. The rationale behind this aerial limit to disturbance is based on research in Canada and through modeling (discussed in more detail in the Results and Discussion section). This research and modeling indicates that when disturbance is limited to a low percentage of a watershed (e.g., 30%), harvesting impacts on water flow and water quality will not be observable, or in the worst case, will be minimal and well within the range of impacts expected to occur through natural disturbances such as fire.

Our study examined water quality in 22 rivers, streams and creeks of watersheds found in the Manitoba Model Forest, an area of the province in which there is a lack of water quality information. The objectives of our study were to:

- Contribute to the development of a long-term data base on water quality in the region
- Develop an understanding of how natural watershed features (including watershed size, soil type and forest type) influence water quality
- Develop an understanding of how watershed disturbances (including agriculture, forestry, wildfire, insect outbreaks and beaver activity) influence water quality
- Test the validity and/or applicability of a 30% watershed harvest limitation, and
- Contribute to the development of watershed planning tools for Tembec Industries Inc., who have forest management responsibilities for the area of study

The last objective (development of watershed planning tools) is not part of this project report. To do this, data from the present stream study will be combined with data collected from a companion study on 100 lakes in the region that is just being completed. Together, this larger data set will be used to develop watershed planning tools for the maintenance of water quality. This will form a separate report.

Methods and Materials

Study Sites and Description of the Region

Twenty two rivers, streams and small creeks were selected to be part of the study. One sampling site was established on each water body, with the exception of the Manigotagan and Black rivers, where two sampling sites were established on each. Therefore, there were 24 sampling sites in total. All of the sampling sites are located in the boreal forest of eastern Manitoba (Figure 1), within the boundaries of the Manitoba Model Forest, an area of approximately 1 million hectares. The Model Forest also overlaps with the Tembec Forest Management License (FML) 01. The watersheds of two of the rivers (Manigotagan and Wanipigow) originate in northwestern Ontario. The watersheds for all other rivers, streams and creeks originate in eastern Manitoba. For simplicity, all water bodies studied will be referred to as streams in this report, although there are large differences in bank full width and discharge (discussed later).

Twenty of the study streams are located in the Lac Seul Upland Ecoregion (Ecoregion 90) of Manitoba. This Ecoregion covers most of the eastern side of Lake Winnipeg, as far east as the Albany River in northwestern Ontario. These streams drain directly into Lake Winnipeg or indirectly into Lake Winnipeg via the Winnipeg River. Two of the study streams (Strawberry Creek and Maple Creek) are located in the Lake of the Woods Ecoregion (Ecoregion 91), which is adjacent to the Lac Seul Upland Ecoregion. The Lake of the Woods Ecoregion extends south from the southern part of Lake Winnipeg to the US border, and east to the Rainy River along the

Canada-US border. Strawberry and Maple creeks are found south of, and drain into the Winnipeg River.

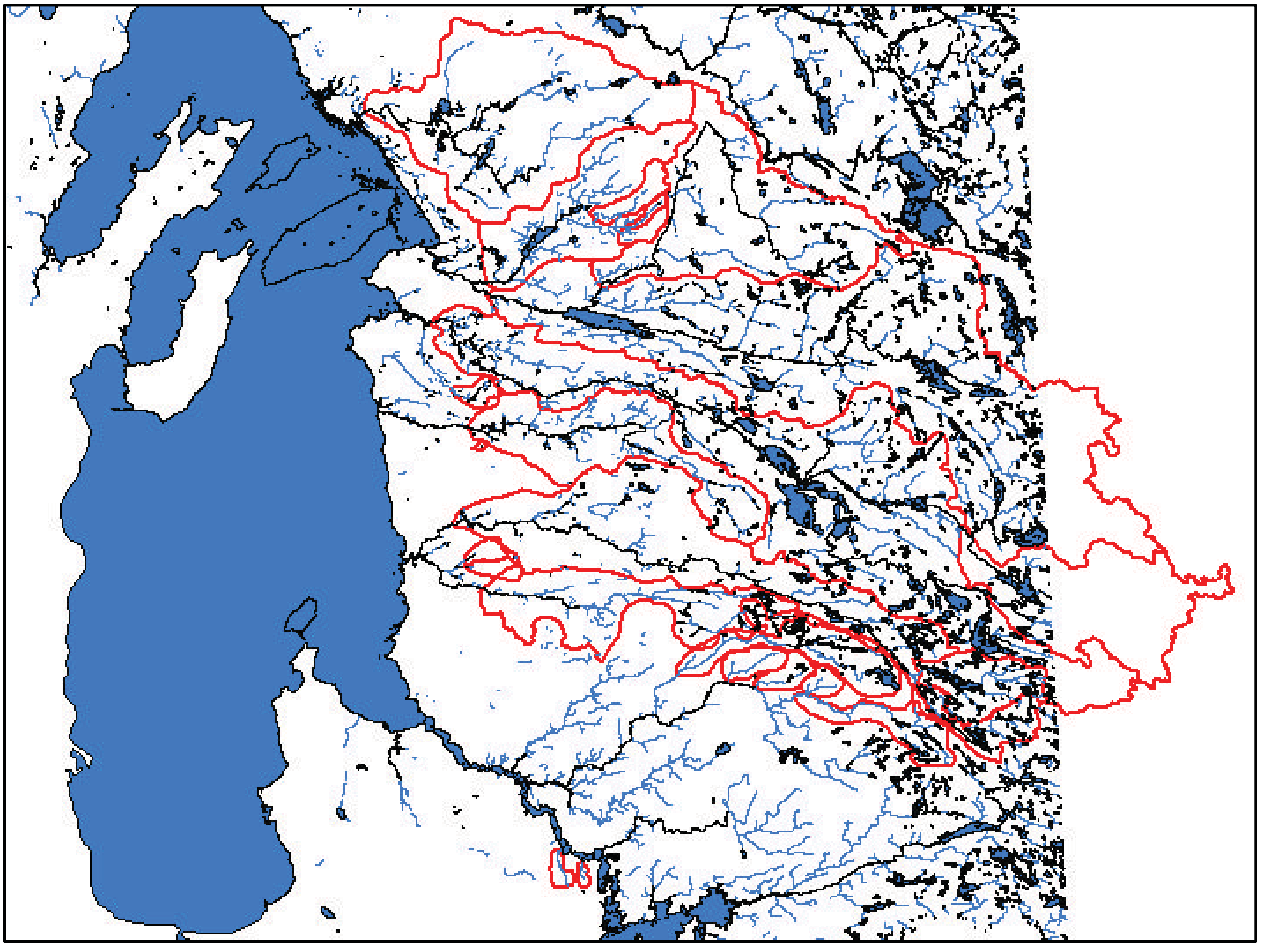


Figure 1. Study streams and their associated watersheds. Note: the watersheds of the Manigotagan and Wanipigow rivers originate in northwestern Ontario. The downstream boundary of each watershed represents the location where the sampling sites are located.

The climate in the Model Forest area is characterized by warm summers and cold winters. Mean summer temperature is 13.5 to 15.5^{o}C, although days > 30^{o}C are not uncommon in July and August. Mean winter temperature is –14.5^{o}C and night time lows at this time of the year can reach –40^{o}C, although this is rare. Precipitation ranges from 500 to 600 mm, with approximately 1/3 of the precipitation falling as snow. Annual precipitation can vary significantly from year to year. The distribution of soil types is quite patchy in the region. In general, soils change from those that are dominated by exposed bedrock with little surficial deposits closer to the Ontario border, to level or gently undulating organic peat deposits overlying lacustrine clay sediments closer to Lake Winnipeg. Upland forests in the region consist of pure softwoods (jack pine, black spruce) and mixed woods (trembling aspen mixed with black spruce, white spruce, jack pine or balsam fir), which are interspersed with lowland forest types (bogs, fens) containing black spruce and tamarack. The area is rich in wetlands, lakes, rivers, streams and small creeks.

Sampling Frequency

For the majority of the streams, sampling during the open water period occurred monthly from May to October, 2004. In addition, most streams were also sampled during the winter, in February and March, 2004 and again in January and March, 2005. However, not all streams were sampled in all time periods. At the start of the program in February, 2004 only 18 stream sites were sampled. In an effort to increase the number of streams in watersheds with historical logging and/or wild fire disturbances, and to include streams in the region with agricultural activity in their watersheds, five streams (O'Hanly mid tributary, Kapukwaywetewonk Creek, Lost Creek, Maple Creek and Strawberry Creek) were added to the program in July, 2004. Therefore, no data were collected in these streams prior to July. In addition, four streams (O'Hanly mid tributary, Kapukwaywetewonk Creek, Lost Creek and Strawberry Creek) were either dry or frozen to the bottom in the winter so that winter water quality samples could not be collected. Some rivers (Wanipigow, Manigotagan at Hwy 304, Rice, Black at Hwy 304 & 314, and Beaver Creek) did not freeze over at the sampling site in winter, while all other rivers, streams and creeks were covered with ice in the winter.

Water Velocity Measurement and Calculation of Discharge

During the open water period (May to October), water velocity was measured in the streams. To measure water velocity, a line transect (tape measure) was stretched across the stream from bank to bank. Depending on the stream width, water depth and velocity was measured every 0.25 m to 1 m across the width of the stream. Water velocity was measured using an AquaSensa RC-2 Water Velocity Meter equipped with an electromagnetic RV4 mini probe. As velocity varies with water depth, velocity was always measured at 0.6 times the water depth, in order to obtain an average velocity reading (Figures 2 and 3). In streams where a defined channel did not exist, or when flow was too great to allow safe measurements, water velocity was measured on the downstream side of culverts (on those streams which had such water crossing structures). In some of the larger rivers (e.g., Manigotagan, Wanipigow), flow was too great at all times to safely allow for water velocity measurements. In some of the other rivers and streams (e.g., Black River, Beaver Creek, O'Hanly River), velocity measurements were only possible during lower flow periods. Therefore, velocity and depth information (and thus, stream discharge – discussed below) is not available for all streams.

Discharge was calculated by creating a cross sectional diagram of each stream using the depth information collected along each transect. Water velocity (m/sec) at each point on the transect was multiplied by the cross sectional area (m^2) calculated between the points along each transect. The product for each measurement was then summed across the entire stream width to produce a discharge value (m^3/sec).

Figure 2. Allison Selinger measuring water velocity in the Black River at Hwy 314 in Nopiming Provincial Park.

Figure 3. Brian Kotak measuring water velocity in Cat Creek.

Water Temperature

Water temperature was measured in two ways. On each sampling visit, surface water temperature was measured with either a handheld CheckTemp Digital Thermometer (Figure 4) or a YSI Model 550A Temperature/Dissolved Oxygen Meter. In addition, hourly water temperature was monitored and recorded using Onset HOBO Water Temp Pro data loggers (Figure 5). For each stream, a data logger was cabled inside of a white PVC pipe, which in turn was cabled to a concrete cinder block. The block was then placed in the water, along the shaded side of the stream. The block was cabled to a nearby tree.

Figure 4. Blaine Johnston measuring water temperature in Kenny Creek using a hand held meter.

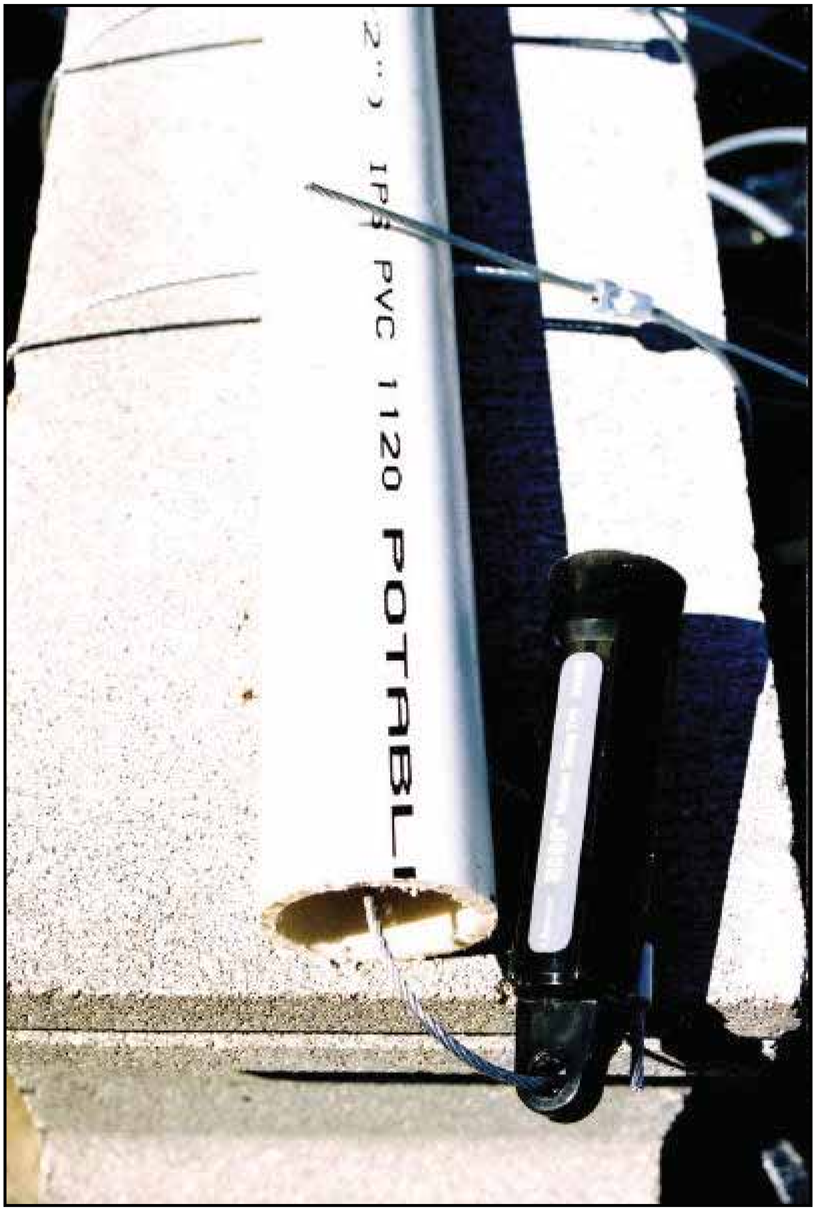


Figure 5. Hobo temp data logger.

<u>Dissolved Oxygen Measurement</u>

Beginning in June 2004, dissolved oxygen concentration in the streams was measured using a YSI Model 550A Temperature/Dissolved Oxygen Meter. Prior to use each day, the meter was calibrated. Membranes and KCl solution were also regularly changed to ensure accuracy of the measurements. Dissolved oxygen was measured *in situ*, within the upper 50 cm of the water column. Care was taken to ensure that the bottom sediments were not disturbed when taking oxygen measurement, as this would affect dissolved oxygen concentrations by mixing anoxic (low oxygen) water of the sediments into the overlying water column. Measurements were always taken on the upstream side when wading in the streams. During the winter, dissolved oxygen measurements were also taken at various depths in the water column. As dissolved oxygen concentrations did not vary with depth (except right at the sediment-water interface), only surface concentrations are reported.

Water Quality Sampling and Water Chemistry Analysis

Surface water (upper 10-50 cm) samples were collected by wading into the streams (when possible) and collecting the water samples directly into the bottles provided by the analytical laboratory. Sample bottles were double rinsed with the stream water prior to collecting the sample. In instances where wading into the streams would create a large disturbance to the bottom sediments and possibly contaminate the sample (e.g., in streams with slow moving water and a mud bottom), samples were collected from the shoreline using a metal scoop on a long handle. Water samples collected for dominant ions (calcium, magnesium, etc.) were acidified in the field with 20% nitric acid. Water samples were placed in a cooler on ice until transport to the analytical laboratory (usually the next day). In the winter, water samples were collected from areas of open water, or when the streams were completely ice-covered, samples were collected from a hole drilled with an ice auger.

Water chemistry analysis was conducted by Envirotest Laboratories, Winnipeg. Water samples were analyzed for pH, alkalinity, conductivity, total phosphorus, total dissolved phosphorus, ammonia, nitrate (includes nitrite), sulphate, total Kjeldahl nitrogen (TKN), total nitrogen (sum of nitrate/nitrite, ammonia and TKN), dissolved organic carbon, cations (calcium, magnesium, potassium, sodium), chloride, total suspended solids and turbidity. Values for total dissolved solids and hardness were calculated. A summary of the analytical methods and references used for the water quality analysis are included in Appendix 1.

GIS Databases and GIS Analysis

Geographic Information Systems (GIS) was used to delineate watershed boundaries of the streams and to compile soil, forest type and disturbance history information in each of the watersheds. For major river systems (e.g., Manigotagan, Wanipigow) watershed boundaries previously developed by PFRA were used. For other rivers, streams and creeks, a combination of 1:50,000 topographic maps, aerial ortho-photography and color infrared photography was used to define watershed boundaries. The lower boundary of each watershed was defined as the sampling point where discharge measurements and/or water quality samples were taken.

In order to quantify various watershed features (soils, forest types, disturbance history), analysis was undertaken using ArcView 3.1 software. GIS data layers were provided by Tembec. GIS layers included the 1997 Forest Resource Inventory (FRI), access development, aquatic features (lakes, rivers, streams and creeks), forest fire distribution, insect outbreak distribution, logging disturbance history and enduring features. All GIS layers are updated annually when appropriate. Each GIS layer contained a myriad of information, much of which was not required for our analysis. For example, while tree species composition (i.e., the percent of each tree species) in each watershed could be calculated, this type of information was thought to be too detailed. Instead, major forest types were assessed. A summary of the GIS layers and types of information used in our study are summarized below:

- Forest Resource Inventory
    - Productive Forested **Land Type**
        - Softwood (S) - >76% of total basal area of a polygon consists of coniferous species
        - Mixedwood (M) – Basal area of all coniferous species is between 51 and 75%
        - Mixedwood (N) – Basal area of all coniferous species is between 26 and 50%
        - Hardwood (H) – Basal area of all coniferous species is less than 25%
    - Non-Productive Forest **Land Type**
        - Treed Muskeg – includes black spruce and tamarack bogs
        - Treed Rock – shallow soils with at least 26% of area treed (non-merchantable trees)
        - Willow/Alder
    - Non-Forested **Land Type**
        - Bare Rock
        - Fields
        - Meadow
        - Marsh
        - Unclassified – includes towns, airstrips, roads, gravel pits, beaver floods
    - Water

- **Stand Type**
- **V-Type Group**

- Access Development – includes all-weather and seasonal road as well as trails and power lines. With respect to roads, only active roads were used in the analysis. Decommissioned forestry roads which have been reforested were not included. Linear feature density (km of linear features per km^2 of watershed) was calculated for each watershed
- Forest Fire Distribution – Two GIS layers were used. A layer for fire history from 1881 to 1975 provided approximate boundaries of major fire events. This information was compiled by Tembec from old fire history maps. Newer fire distribution information has been collected digitally since 1976 and updated annually. This newer data has a sub-hectare spatial resolution.
- Insect Outbreak Distribution – Spatial distribution and severity classes for Forest Tent Caterpillar and Spruce Bud Worm from 1995 and onward.
- Logging Disturbance – Two GIS layers were used. A layer for historic logging history from the 1950s to 1985 provided approximate boundaries for general areas where harvesting took place. This information was compiled by Tembec from old harvest history maps. Newer harvest history distribution information has been collected digitally since 1986, and updated annually.
- Enduring Features (Soils) – This layer provides a general picture of the enduring features (a combination of surficial deposits, soil landscapes and terrain features) which was developed by World Wildlife Fund (1997). Mapping is at a very coarse scale (1:1,000,000). Major enduring feature classes in the Model Forest Region include:

- BR/D – bedrock/dark grey chernozem, BR/F- bedrock/grey brown luvisol, BR/R2 – bedrock/acidic hard rock, BR/Y23 – bedrock/organic mesisol
- DB/D – deep basin/dark grey chernozem, DB/F – deep basin/grey brown luvisol, DB/R2 – deep basin/acidic hard rock, DB/Y23 – deep basin/organic mesisol
- GD/R2 – gaciofluvial deposit/acidic hard rock
- OD/D – organic deposit/dark grey chernozem, OD/F – organic deposit/grey brown luvisol, OD/R2 – organic deposit/acidic hard rock, OD/Y23 – organic deposit/organic mesisol
- T3/R2 – glacial till derived from Precambrian rock/acidic hard rock

Calculation of Export Coefficients

To evaluate the export of dissolved substances (e.g., nutrients) from the watersheds to the streams, export coefficients were calculated. Export coefficients were calculated for calcium, magnesium, potassium, sodium, total phosphorus, total dissolved phosphorus, sulphate, nitrate, ammonia, total nitrogen and dissolved organic carbon. To calculate export coefficients, concentrations (mg/L) of dissolved substances were averaged over the open water period (May to October) and multiplied by the average discharge (m^3/s) for the period. The result was scaled up to an annual value and divided by the net watershed area. Export coefficients are expressed as kg/ha/yr. Export coefficients could not calculated for all streams, as insufficient discharge data were available for some water bodies (particularly the larger rivers). As a result, export coefficients were calculated for only 15 of the 24 sampling sites. In addition, it should be noted that these annual export coefficients are very crude approximations, as they are based on limited data and also do not account for changes in flow or concentrations during the winter, summer storm events or spring runoff periods. However, they do provide a general picture of export from the watersheds.

Statistical Analysis

To evaluate the influence of watershed features such as watershed size, soil type (i.e., enduring features), and forest type on the concentration of dissolved substances in the study streams and export coefficients, simple correlation analysis was used. For these analyses, data from the Wanipigow River, Manigotagan River at Hwy 304 and Manigotagan River at Hwy 314 were excluded, as the GIS information from the Ontario side of the watersheds was not available. In addition, data from Strawberry and Maple creeks were excluded, as agricultural activities appear to have an over-riding influence on water quality.

To evaluate the influence of logging and fire disturbance on dissolved substances in the study streams and export coefficients, the disturbance data were categorized according to disturbance type and time since the disturbance. Harvesting data were categorized in the following time increments: within 5 years since disturbance, 18 years, 30 years and within 60 years since disturbance. This gave a good cross section of recent, midterm and long timeframes since disturbance. The category of 18 years since disturbance was chosen as it coincided with the implementation of accurate, digital harvest area data (in 1986). Fire disturbance history was categorized with the same timeframes. The area of disturbance for each watershed was expressed in hectares as well as a % of each watershed. Total disturbance (logging plus fire) was also calculated. In some cases, this resulted in disturbance rates of greater than 100% in a watershed. When conducting correlation analyses to examine the relationships between disturbance type and time since disturbance on water quality and export coefficients, data from the Wanipigow and Manigotagan rivers were excluded, as GIS information on the Ontario side of their watersheds was not available. In addition, the watersheds for Maple Creek and Strawberry Creek were also excluded as no information on fire and logging were available and are not appropriate. The original forests in these two latter watersheds have been cleared for agriculture since the early 1920s, and thus, the major disturbance in these watersheds is agriculture (for which GIS data was available), not logging or fire. Disturbance such as forest tent caterpillar and spruce budworm outbreaks, as well as the impact of road density on water quality was also examined.

Finally, the data was also categorized in the following manner to examine the broad relationship between disturbance type and water quality and export coefficients: reference watersheds (less than 30% of watershed disturbed by harvesting or fire in the last 60 years), watersheds with harvesting (more than 30% of watershed harvested but less than 30% of watershed burned, in the last 60 years), burned watersheds (more than 30% of watershed burned, but less than 30% of watershed harvested, in the last 60 years) and dual disturbance watersheds (more than 30% of watershed area burned and harvested in the last 60 years). The relationship between disturbance category and water quality and export coefficients was examined by Analysis of Variance (ANOVA).

Assessment of Beaver Activity in Study Streams

To assess the degree of beaver activity in the study streams, a subset of the streams (and their tributaries) were flown in a Cessna 172 on September 15 and 16, 2004. The location of all beaver dams were marked on a map and a visual estimation of the hydraulic impact of the beaver dams on the stream was made. The impact of the beaver dams on flow was categorized as: Class I – no visible impact on flow, Class II – limited back flooding, flooding of approximately 5x the normal stream channel width, Class III – significant back flooding, flooding of more than approximately 10x the normal stream channel width. The density of dams (dams/km) was calculated.

Results and Discussion

General Description of Study Streams and Watersheds

The streams and watersheds were chosen to represent a wide gradient in size, watershed characteristics (soils and forest type) and disturbance history (wild fire, logging, insect outbreaks, and beaver activity).

Stream and Watershed Size

The streams varied considerably in bank full width (i.e., the width across the stream from the top of each bank), from 0.7 m in Kenny Creek to more than 29 m in the Manigotagan River at Hwy 314. Bank full width and GPS coordinates of each study site are found in Table 1.

Table 1. Bank full width of each study stream and GPS location of the study sites where water quality and/or flow measurements were made.

Stream Site	Bank full width (m)	GPS Coordinates*1
Kenny Creek at Translicense Road	0.7	N 50 39.541 W 95 45.255
Lost Creek at Hwy 304	2.1	N 50 56.594 W 96 12.583
O'Hanly mid tributary at Hwy 304	2.3	N 50 47.081 W 96 12.337
Drew Creek at Translicense Road	2.4	N 50 37.515 W 95 29.294
Beaver Creek Crossing 7	3.0	N 51 12.043 W 95 54.464
Kapukwaywetewonk Creek at Hwy 304	3.6	N 50 49.095 W 96 14.337
Beaver Creek Crossing 10	4.0	N 51 11.896 W 95 55.414
Strawberry Creek at Hwy 11	4.2	N 50 27.302 W 96 02.305
Rabbit River at Hwy 314	5.6	N 50 39.090 W 95 24.377
Black River at Hwy 314	5.6	N 50 39.572 W 95 23.549
Maple Creek at Hwy 11	6.5	N 50 28.288 W 96 04.476
Duncan Creek at Hwy 304	7.0	N 51 01.227 W 96 15.034
Beaver Creek Crossing 12	7.3	N 51 12.824 W 95 58.181
Cat Creek at Translicense Road	8.2	N 50 37.515 W 95 29.294
Moose Creek at Translicense Road	9.7	N 50 40.256 W 95 51.050
O'Hanly River at Hwy 304	14.0	N 50 47.081 W 96 12.336
Beaver Creek Main	14.0	N 51 08.012 W 95 57.846
Sandy River at Hwy 304*2	15.0	N 50 58.342 W 96 13.118

Table 1. Continued.

Stream Site	Bank full width (m)	GPS Coordinates*1
English Brook at Rice River Road	16.0	N 51 08.724 W 96 10.697
Black River at Hwy 304	18.0	N 50 51.448 W 96 15.165
Manigotagan River at 304*2	20.5	N 51 06.049 W 96 16.597
Wanipigow River at Rice River Road*2	22.5	N 51 07.971 W 96 10.628
Rice River at Rice River Road	27.0	N 51 21.130 W 96 23.801
Manigotagan River at Hwy 314*2	29.4	N 50 48.020 W 96 21.013

*1 Coordinates are the location where water velocity and/or water quality samples were collected. The sampling sites were located next to highways or logging roads.
*2 Bank full width estimated from GIS.

The study watersheds also varied considerably in size, ranging from 138 hectares (1.38 km^2) to over 170,000 hectares (1,700 km^2). This is shown in Figure 6.

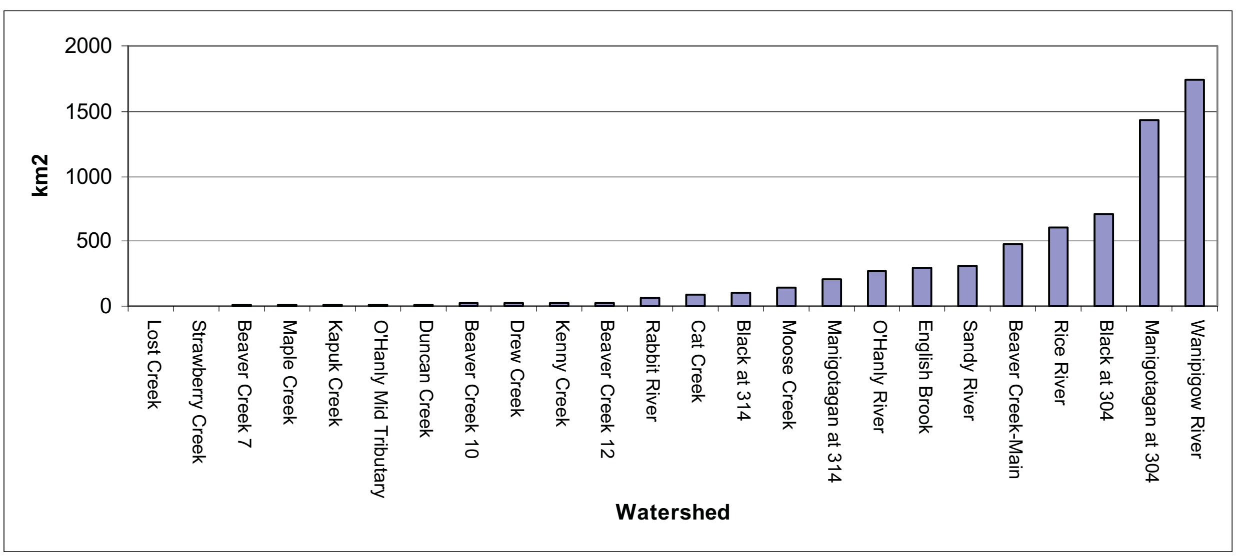


Figure 6. Size of study watersheds.

Two of the study streams (and their watersheds) originate in northwestern Ontario. These are the Wanipigow and Manigotagan rivers.

Soil Characteristics of the Watersheds

Enduring features ("soil types") varied within and between watersheds. Figure 7 shows a spatial representation of the main soil types in two watersheds (Sandy and Rice rivers). The area of 15 of the 24 watersheds were dominated (i.e., > 50% of their area) by acidic bedrock (BR/R2 – Figure 8). BR/R2 was the exclusive soil type (or nearly so) in 5 of the watersheds (Beaver Creek Crossing 7, Beaver Creek Crossing 10, Black River at Hwy 314, Moose Creek and Rabbit River). These watersheds are characterized by rock outcrops with shallow soils. Tree species growing on these areas are generally jack pine and black spruce.

Other watersheds did not fit this pattern and were dominated by other soil types. For example, the watershed of Drew Creek was dominated mainly by organic deposit acidic rock (OD/R2), the watershed of Duncan Creek was dominated by deep basin grey brown luvisols (DB/F), the watershed of Kapukwaywetewonk Creek was dominated by organic deposits and organic mesisol soils (OD/Y23), and the watershed of one of the streams containing significant agriculture (Strawberry Creek) was dominated by bedrock with dark grey chernozem soils (BR/D-Figure 8).

The streams studied therefore represented a good cross section of watershed types with respect to enduring features (soils).

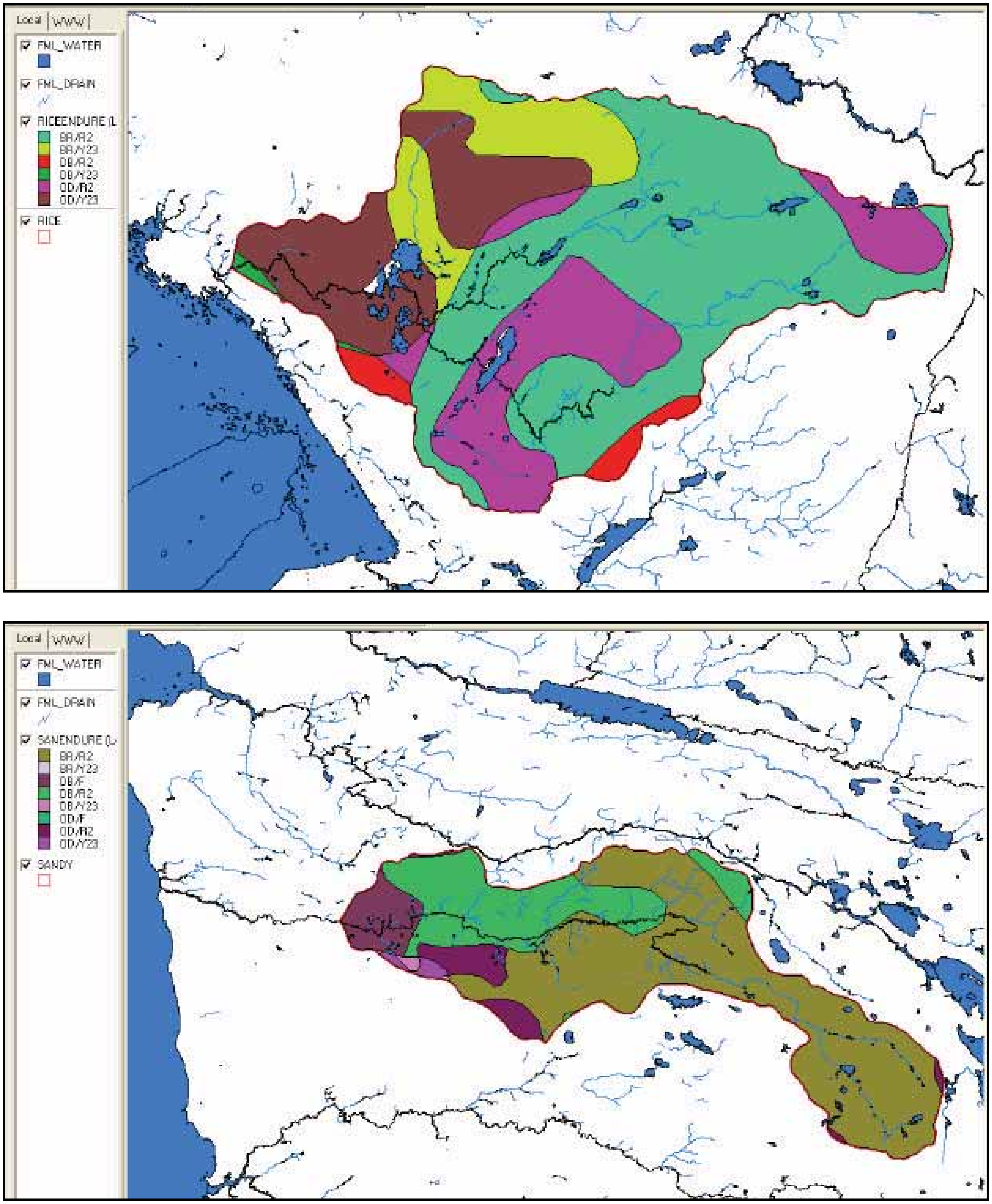


Figure 7. Watersheds of Rice River (top) and Sandy River showing enduring features.

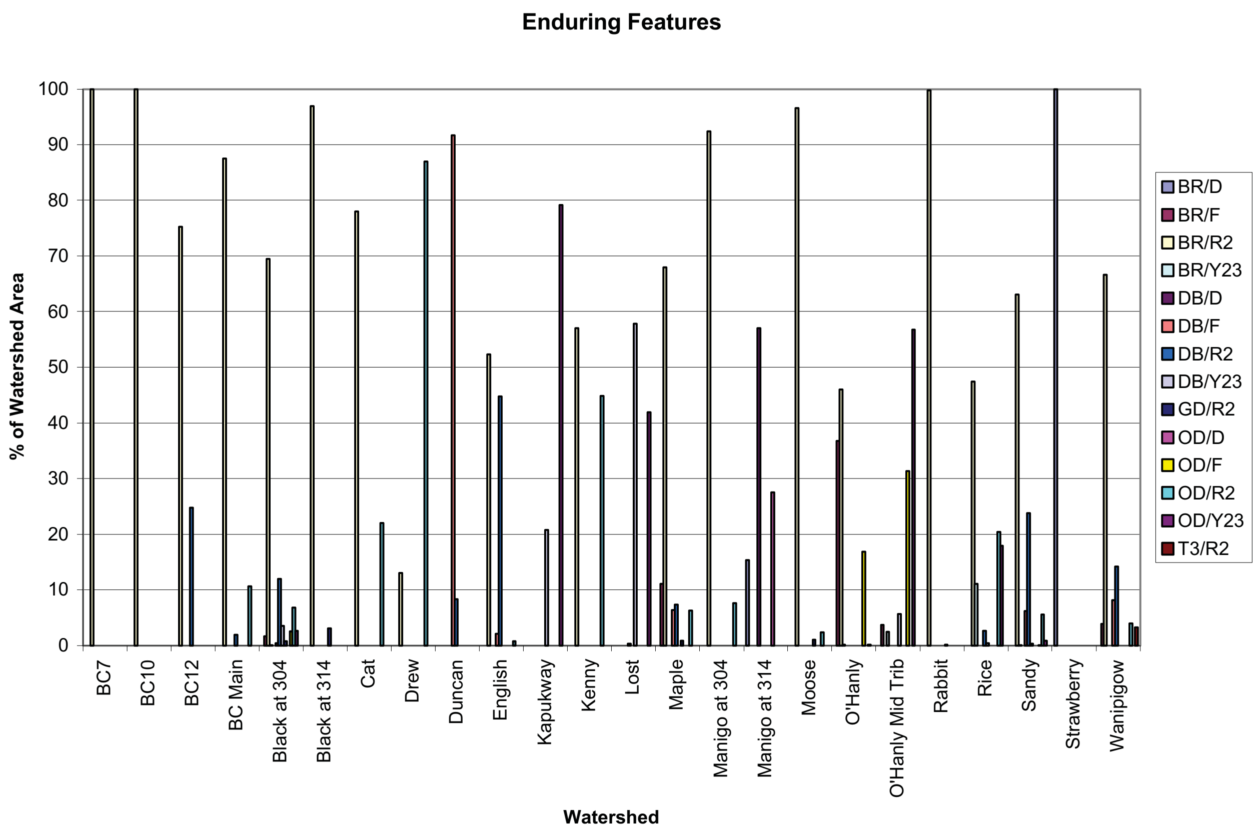


Figure 8. Enduring features in the study watersheds.

Forest Cover Characteristics of the Watersheds

The majority of the watersheds were dominated by productive coniferous forest (jack pine, black spruce). Seventeen of the 24 stream watersheds had their area covered more than 50% by Forest Type S (> 75% basal area of conifer). Very few of the watersheds contained significant amounts of area of hardwood (H) or hardwood-leading mixed wood (N) stands. Duncan Creek was the highest with 30% and 11% of its watershed area in H and N type stands, respectively. All other watersheds had less than approximately 5% of their area in H or N type stands. The area of unproductive forest land in the watersheds varied from 4% to more than 50%. The watersheds of O'Hanly mid tributary and Kapukwaywetewonk Creek contained the highest proportion of non-productive forest at 54 and 42%, respectively, and this was made up of black spruce and tamarack bogs. Only Maple Creek and Strawberry Creek contained significant amounts of non-forested land (81 and 69%, respectively). This represents previously forested land that has been converted to mixed agriculture. All other watersheds contained generally less than 10% non-forested land (which includes bare rock, marshes, converted forest land, human infrastructure, etc). The watersheds encompassing Black River at Hwy 314 and the Rabbit River contained the highest proportion of water (at 15 and 12% respectively), due to their location near the Ontario border (which contains more lakes) and the small size of their watersheds.

Insect Outbreak Disturbance in the Watersheds

The amount of disturbance (human and natural) in the watersheds varied considerably. The area of tree mortality caused by spruce bud worm was generally low in most watersheds. Exceptions were Kapukwaywetewonk Creek (23% of the watershed area affected), Manigotagan River at Hwy 304 (14%) and the Wanipigow River (15%) (Figure 9). Watersheds with the highest amount of forest tent caterpillar damage were the Manigotagan River at Hwy 304 (24% of the watershed area affected), Manigotagan River at Hwy 314 (19%) and the Wanipigow River (13%) (Figure 9).

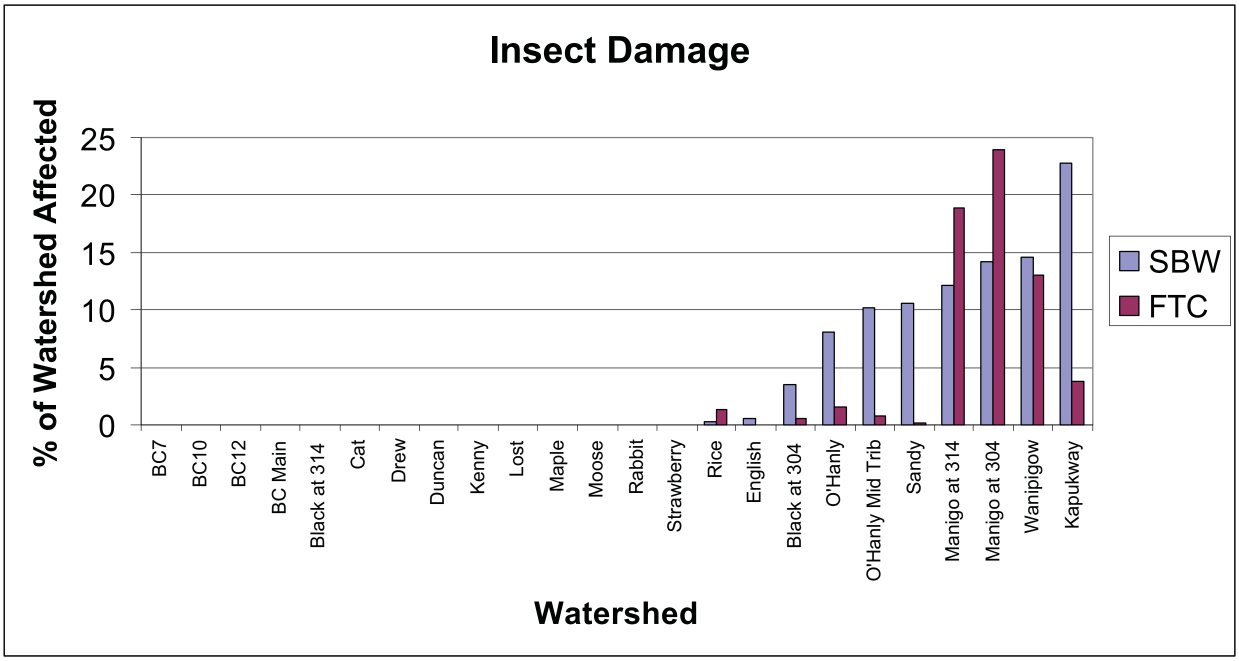


Figure 9. Aerial extent of Spruce Bud Worm (SBW) and Forest Tent Caterpillar (FTC) in study watersheds.

Fire Disturbance in the Watersheds

The occurrence of fire disturbance in the watersheds varied both in area and in timing. Major recent fire events occurred in 1983, 1989 and 1999 in eastern Manitoba. Very large fires occurring in the 1880s, 1929 and 1936 have created the majority of the now older forest stands available for harvest in the region by Tembec.

Few watersheds experienced any fire in the last five years. The two dramatic exceptions to this are Duncan Creek and Lost Creek, in which 100% and 99%, respectively, of their watershed areas burned over a 3 day period in 1999 (Figure 10). Interestingly, the entire watershed of Duncan Creek also burned a decade earlier, in 1989. Approximately 38% of the English Brook watershed burned in the last 18 years, mainly in 1983 and 1989, two "high fire" years in eastern Manitoba (Figure 10). Approximately 15% of the watershed of Manigotagan at Hwy 304 burned in the last 18 years (mainly in 1983, 1987, 1989 and 1999), and 15% of the Sandy River watershed (in 1989 and 1999). Within the last 30 years, many more of the watersheds have experienced significant (more than 30% of the watershed area burned) fire disturbance: 40% watershed disturbance by fire in Beaver Creek Crossing 7 (in 1983), 32% in Beaver Creek Crossing 12 (in 1976 and 1983), 39% in Beaver Creek Main (in 1976 and 1983), 36% in Cat Creek (in 1983), 33% in Drew Creek (in 1983), 73% in Kenny Creek (in 1983) and 42% in Rabbit River (in 1983) (Figure 10).

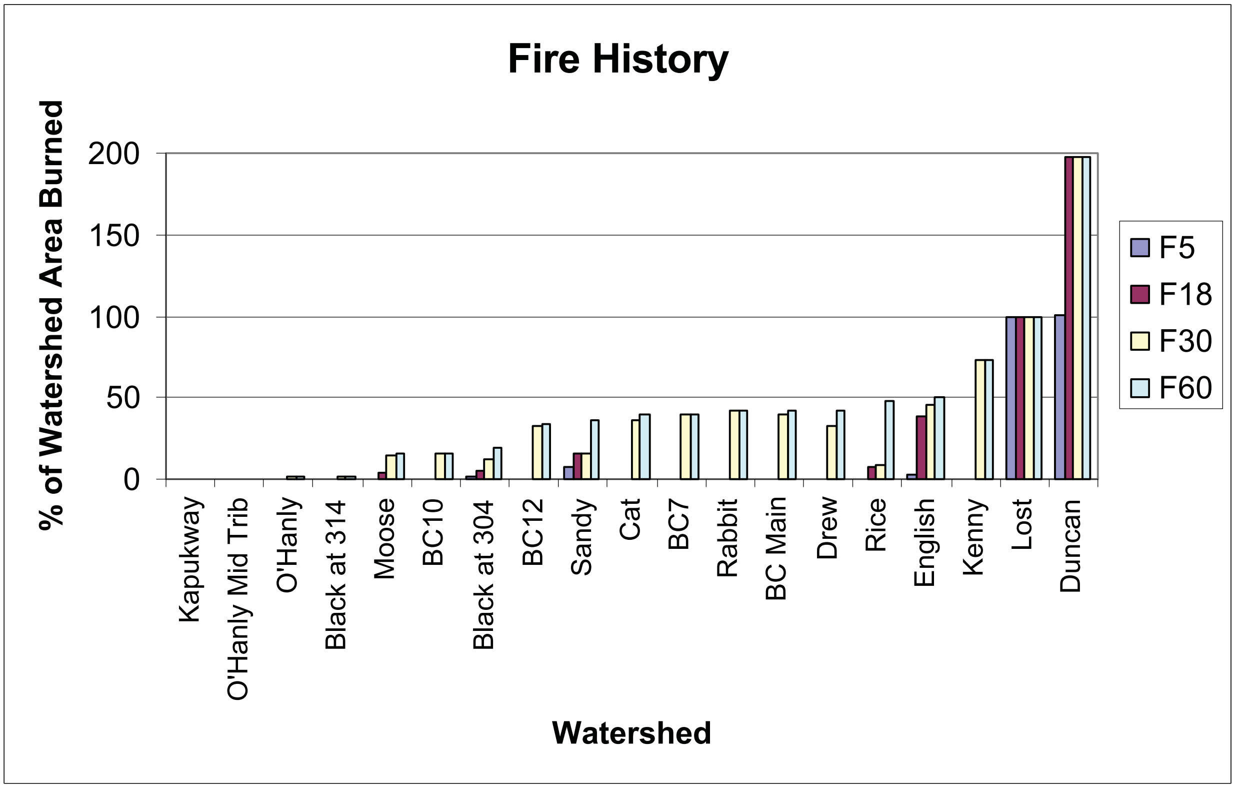


Figure 10. Cumulative area of watershed burned in study watersheds in the past 5, 18, 30 and 60 years. Values > 100% indicates complete spatial overlap in fires.

Harvest Disturbance in the Watersheds

With respect to watershed disturbance caused by logging and associated access (road) development, the history of forestry activity can only be mapped as far back as the 1950s. Although harvesting did occur prior to this, the majority of forestry activity prior to 1950 occurred south of the study area, predominantly along the Winnipeg River. Therefore, the GIS data available regarding forestry activity in the study watersheds can be considered reasonably accurate.

Very little recent (i.e., in the last 5 years) harvesting activity occurred in the study watersheds. In fact, 12 of the watersheds experienced no harvesting in the last 5 years and 7 watersheds experienced harvesting amounting to less than 5% of their watershed area (Figure 11). Approximately 11% of the area in the Lost Creek watershed (which amounts to only 15 hectares) was harvested in 1999. This was a salvage harvest of trees that had already been killed in the 1999 fire that burned the entire watershed. Only 1 watershed experienced any significant harvesting in the last 18 years. In Cat Creek, 24% of the watershed area was harvested in the last 18 years (predominantly occurring from 1986 to 1990), with very little after 1990. Very little harvesting occurred in any of the other watersheds in the last 18 years. In fact, only two additional watersheds have experienced any significant cumulative harvesting in the last 30 years: Kapukwaywetewonk Creek (25% of the watershed harvested, predominantly in 1965 and 1980) and Sandy River (17% of the watershed harvested, predominantly in 1980).

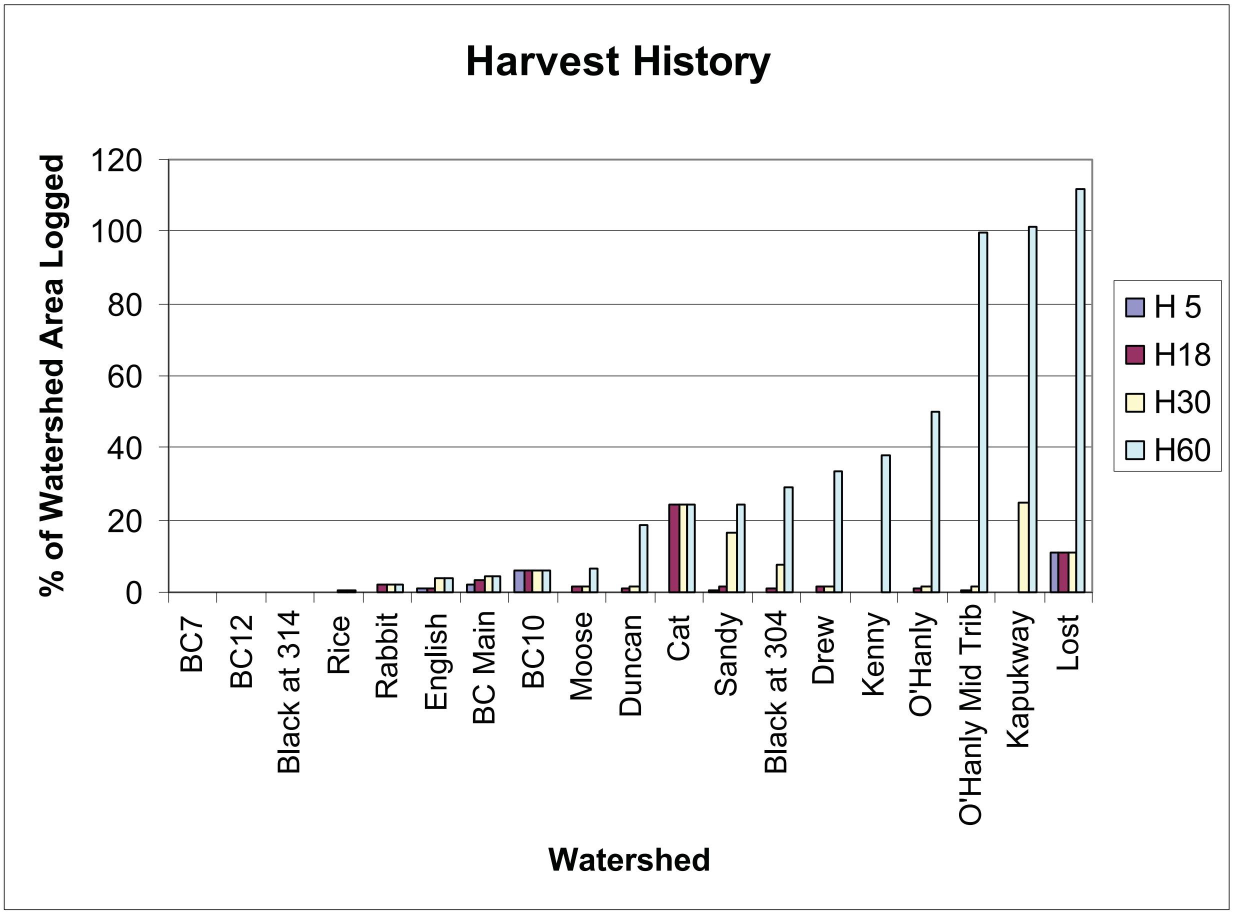


Figure 11. Cumulative area of watershed logged in the study watersheds in the past 5, 18, 30 and 60 years.

The watersheds with the largest cumulative harvest over the last 60 years (from 1945 to 2004) were: 29% in Black River at Hwy 304, 34% in Drew Creek, 100% in Kapukwaywetewonk Creek, 38% in Kenny Creek, 112% in Lost Creek, 50% in O'Hanly River and 100% in O'Hanly mid tributary (Figure 11). The majority of the harvesting that took place in these watersheds occurred between 1945 and 1975, thus, these are not recent harvests.

Cumulative Harvest and Fire Disturbance in the Watersheds

The disturbance data can also be viewed in terms of combined disturbance rates (harvest and fire) in the watersheds. Within the last 5 years and also within the last 18 years, there has been very little combined harvesting and fire disturbance in the study watersheds (i.e., few of the watersheds in the last 18 years experienced a significant amount of both fire and logging) (Figure 12). Only two watersheds have experienced significant harvesting and fire in the last 30 years. In the watershed of Cat Creek, 36% of the watershed area has burned and 24% has been logged since 1974, giving a cumulative total of 60% of the watershed being disturbed. In the watershed of Sandy River, 15% of the watershed area has burned and 17% has been logged since 1974, giving a cumulative total of 32% of the watershed being disturbed. Five of the watersheds have experienced significant harvesting and fire in the last 60 years (Figure 12). In the watershed of Black River at Hwy 304, 19% of the watershed area has burned and 29% has been logged since 1945, giving a cumulative total of 48% of the watershed being disturbed. Other watersheds with significant cumulative disturbance rates in the last 60 years are: Drew Creek (42% burned, 34% logged, 76% cumulative disturbance), Kenny Creek (73% burned, 38% harvested, 110% cumulative disturbance), Lost Creek (99% burned, 100% logged, 199% cumulative disturbance) and Sandy River (35% burned, 24% logged, 59% cumulative disturbance). Some of the cumulative disturbance values exceed 100%, indicating that the there was overlap in the areas burned and harvested in the last 60 years.

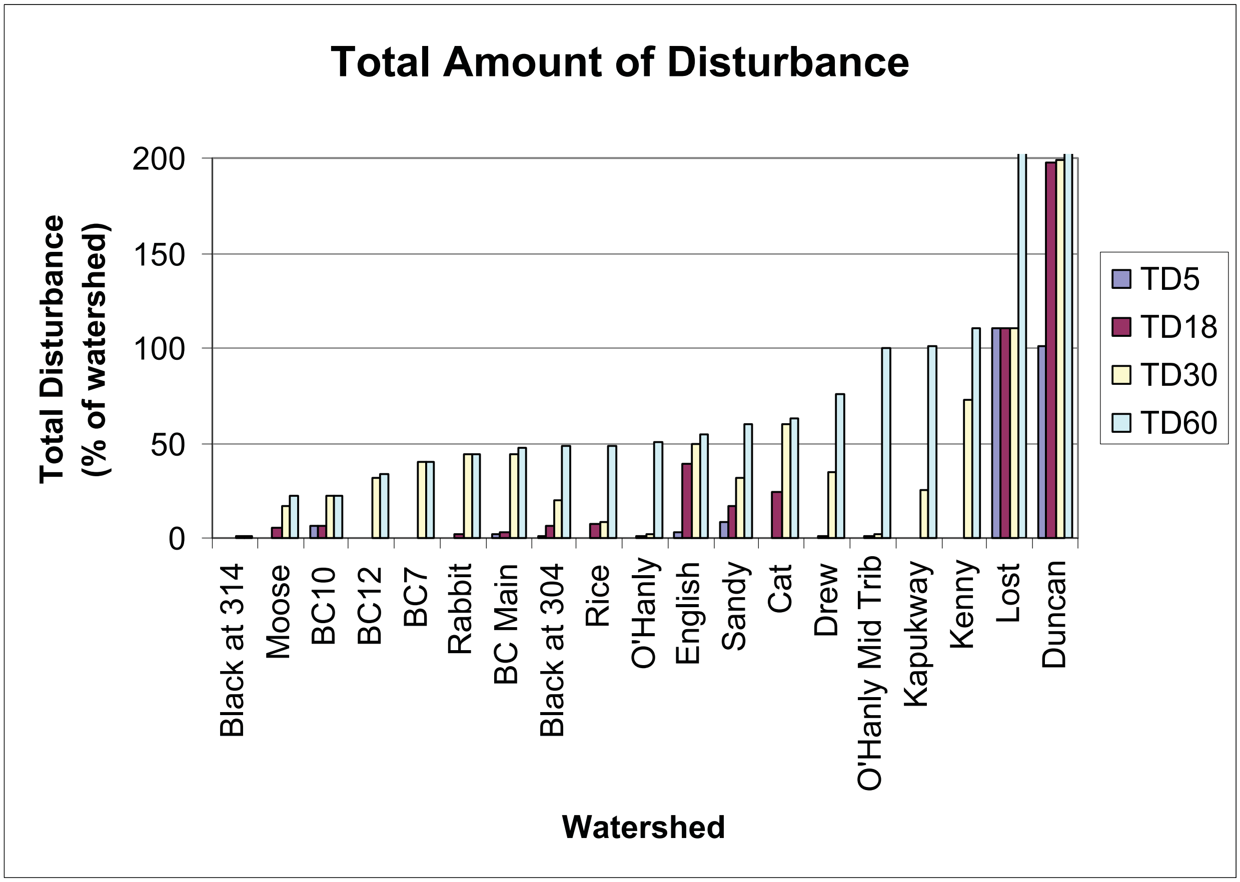


Figure 12. Total amount of disturbance (fire and logging) in the study watersheds in the past 5, 18, 30 and 60 years. Values > 100% indicate spatial overlap in fire and logging.

Linear Features in the Watersheds

Finally, the density of linear features (roads, power lines, trails) in each watershed was calculated using GIS to evaluate the amount of linear disturbance in each watershed. Not surprisingly, the watersheds with the greatest density of linear features are those with the largest road networks; watersheds that have received the highest amount of timber harvesting (O'Hanly mid tributary, Kapukwaywetewonk Creek, Lost Creek – Figure 13). Road development is an integral component of forestry operations.

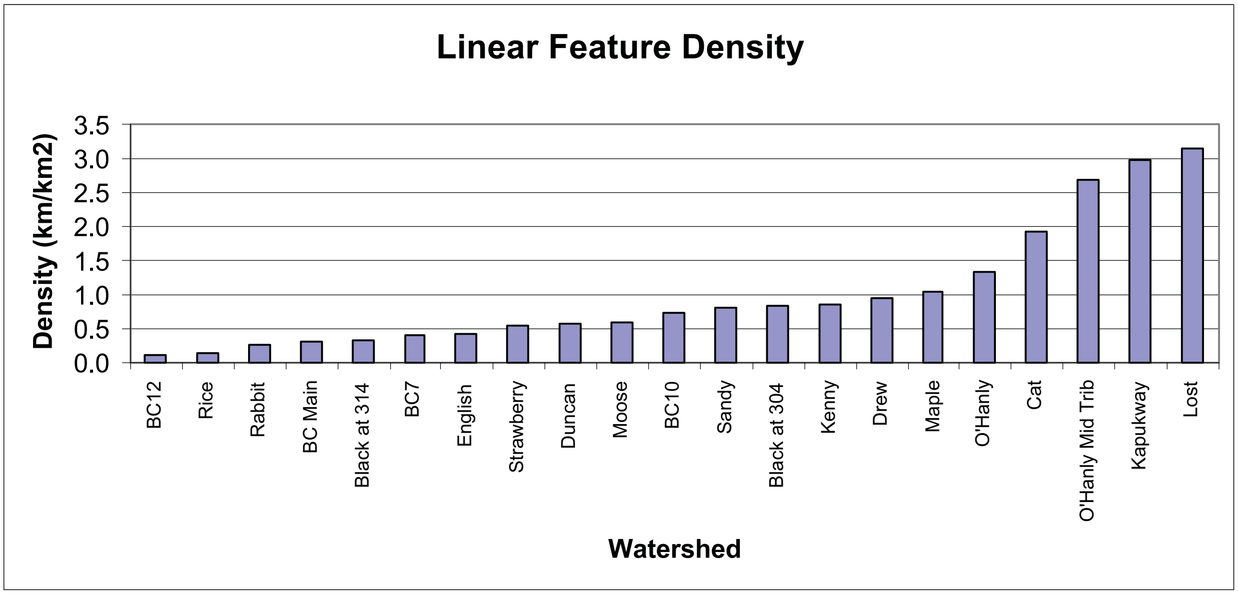


Figure 13. Density of all linear features in the study watersheds. Note: Manigotagan River (at Hwy 304 + 314) and Wanipigow River are missing as the GIS information for the parts of the watersheds in Ontario are lacking.

Regional Context of Water Quality

To put the study streams into context, water quality in the study streams was compared to that of other rivers in the province, including those located on the east side of Lake Winnipeg (and thus found in the same Ecoregion, general soil types, forest types and fire disturbance history) and rivers located in the southern prairie ecosystem. Water quality information on these other rivers was provided by Manitoba Water Stewardship and represent average values for the open water period (May to October). It should be noted that while the data presented for the Red and Assiniboine rivers are relatively recent (i.e., collected in 2004), the data for the Berens, Pigeon and Poplar rivers are much older (i.e., collected from 1969 to 1973). There is no recent water quality data for these rivers. Data from the Bloodvein River were collected from 1970 to 1991.

Conductivity is a general measure of the amount of dissolved substances in the water. In prairie water bodies, conductivity can also be used as an indicator of salinity. Conductivity in the study streams located in forested watersheds was similar to that of the other major rivers (Bloodvein, Berens, Pigeon, Poplar) along the east side of Lake Winnipeg (Figure 14). This is not surprising as the study streams are located in a similar geologic, soil and forest setting as these major rivers in eastern Manitoba. These rivers also experience a similar disturbance regime (mainly fire, and to a limited extent logging). The conductivity of water found in Strawberry Creek and Maple Creek (which were previously forested, but which were cleared for agriculture) was considerably higher than in the forested study stream watersheds to the north. Permanent removal of forest cover and the conversion to agriculture in the watersheds appears to have resulted in a substantial increase in conductivity. It should be noted however, that these two watersheds are located in a different Ecoregion and soil type may also partly explain the difference in conductivity between forested and non-forested study streams (discussed later). Conductivity in the Red and Assiniboine rivers was much higher than all of the other rivers and streams, and likely reflects the prairie characteristics of their watersheds (e.g., fertile soils which are easily eroded) as well as vastly different land use (agriculture, urban development) compared to the rivers and streams in forested watersheds to the north.

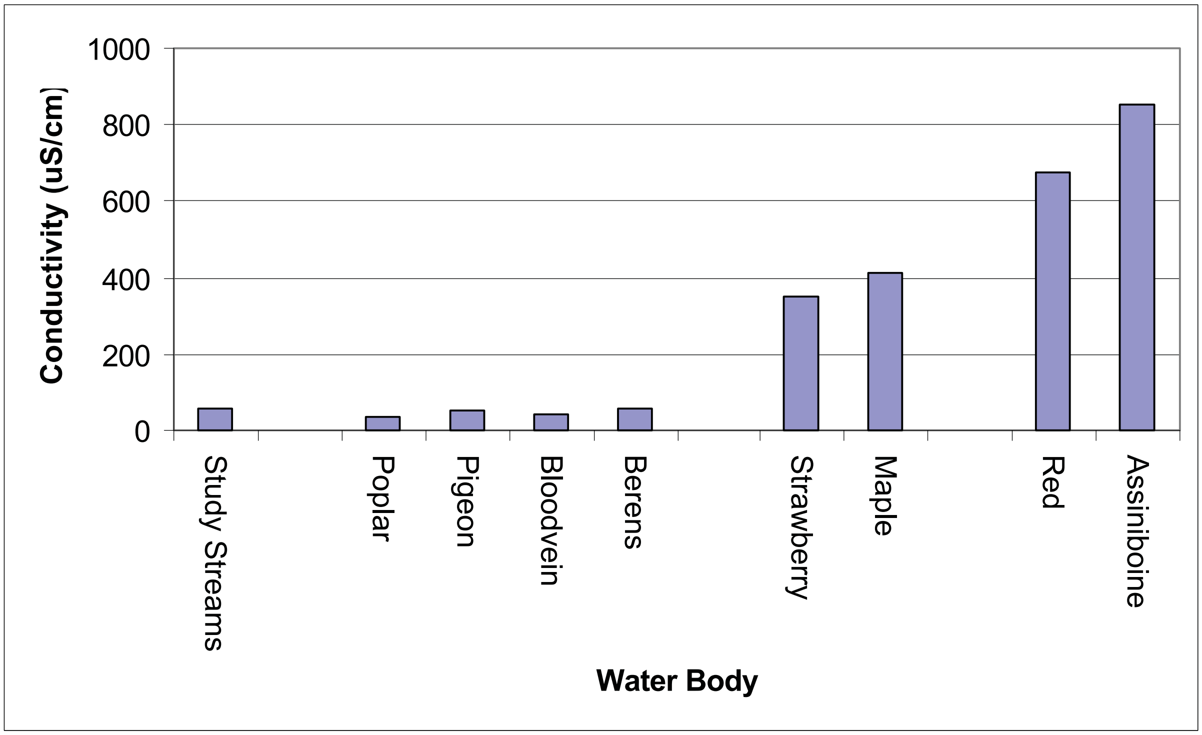


Figure 14. Conductivity of water in the forested study streams, major rivers on the east side of Lake Winnipeg, study streams occurring in agricultural watersheds, and two rivers (Red and Assiniboine) located in the prairie portion of southern Manitoba.

Phosphorus (measured as Total Phosphorus-TP) is one of the main plant nutrients controlling productivity of water bodies. Excessive input of phosphorus has been widely cited as the key cause of eutrophication (nutrient enrichment) of lakes and rivers worldwide. Excessive loading of phosphorus leads to numerous water quality problems including large growths (blooms) of algae, depletion of oxygen from water, fish kills and production of algal toxins, to name a few. In general, TP concentrations in the forested study streams were within the natural range of the major rivers on the east side of Lake Winnipeg (Figure 15), although concentrations of TP in the forested study streams were slightly higher than in the Pigeon, Bloodvein, and Berens rivers. TP concentration in Strawberry Creek and Maple Creek were much higher (3.6 to 4.2 times higher) than in the forested study streams and is likely due to a combination of soil type and agricultural disturbance. TP concentration was the highest in the Red and Assiniboine rivers (9.2 to 11.8 times higher than in the forested study streams), reflecting the influence of productive prairie soils, land use and likely sewage input from major urban centres.

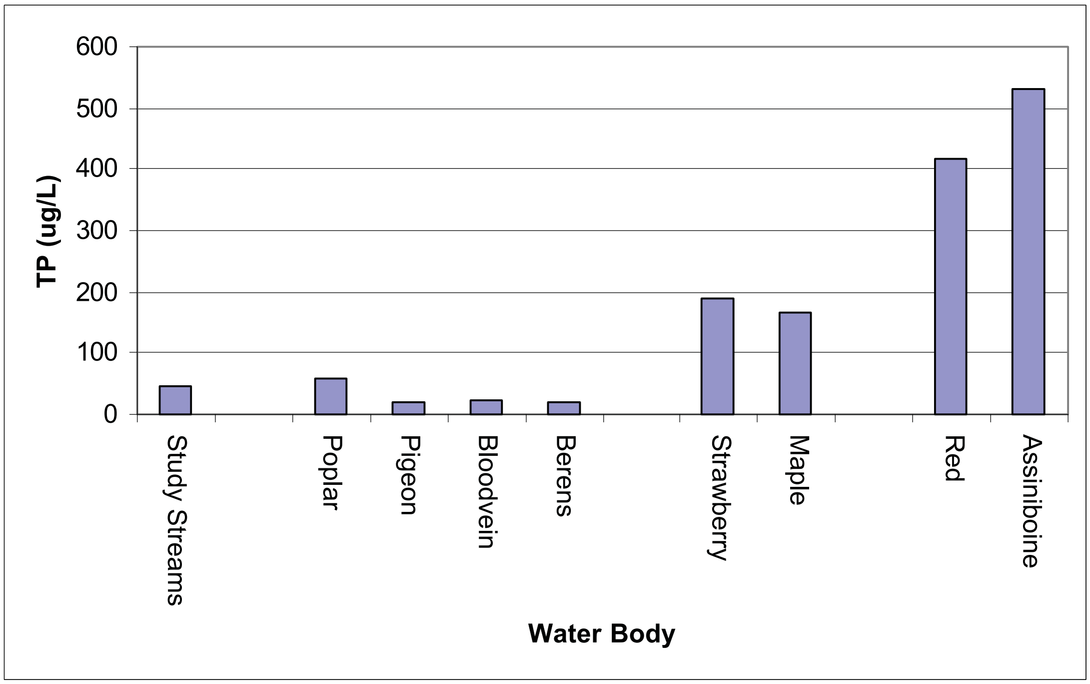


Figure 15. Phosphorus concentration in the forested study streams, major rivers on the east side of Lake Winnipeg, study streams occurring in agricultural watersheds, and two rivers (Red and Assiniboine) located in the prairie portion of southern Manitoba.

Air Temperature and Precipitation in 2004

During the open water period (i.e., May to October) in 2004, the weather was cooler and wetter than the long term average. Table 2 shows the mean daily temperature and precipitation by month for 2004 in comparison to the long-term mean (Environment Canada, 2004) at the Pinawa weather station. Mean daily air temperature was significantly cooler from May to August in 2004 compared to the historical trend from 1971-2000. In addition, precipitation was also well above normal in May, August and September. The effects of the above normal precipitation in particular resulted in higher runoff and stream discharge (discussed in the next section). Prior to 2004, the region was experiencing a much drier period with less snowfall and rainfall than normal.

Table 2. Mean daily temperature and precipitation at Pinawa, Manitoba during May to October in 2004 in comparison to the long-term (1971-2000) average.

Month	**2004 Mean Daily Temperature (^{o}C)**	**Long-term Mean Daily Temperature (^{o}C)**	**2004 Total Precipitation (mm)**	**Long-term Total Precipitation (mm)**
May	6.5	11.4	126.7	59.5
June	12.7	16.2	60.2	94.5
July	17.1	18.9	69.8	78.3
August	13.4	17.7	87.0	71.5
September	13.8	11.8	98.6	64.1
October	5.8	5.1	50.1	45.5

Discharge

As mentioned previously, it was not possible to measure water velocity and calculate discharge in some of the larger rivers and streams due to the excessive flow in 2004, as well as the depth of the rivers. Therefore, discharge data is lacking for the following: Beaver Creek Main, English Brook, Manigotagan River, Moose Creek, Rice River, Sandy River and the Wanipigow River.

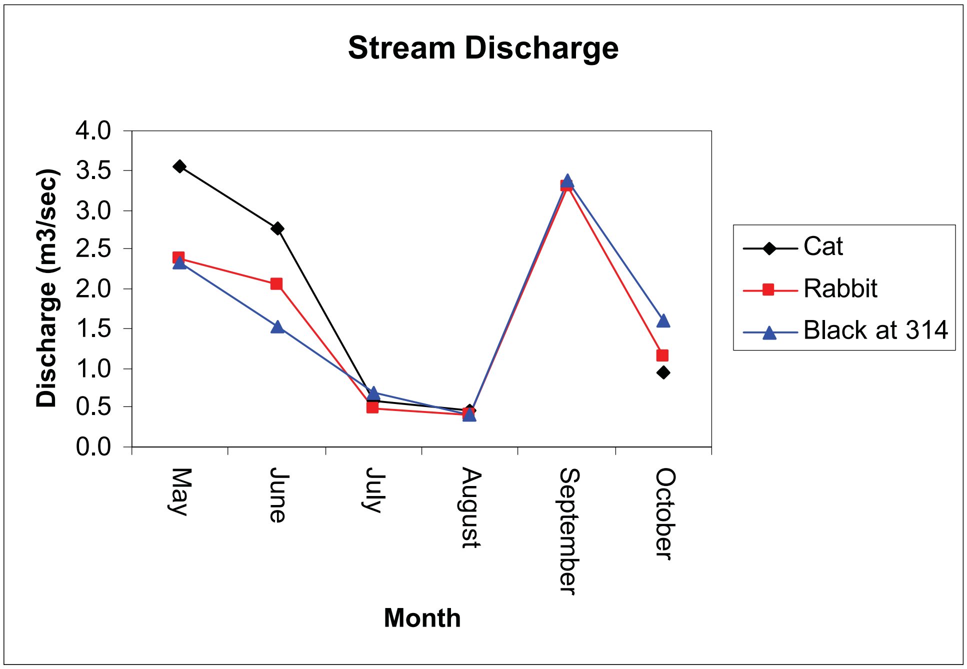
In addition, discharge estimates could not be made on several occasions in some of the small streams and creeks due to the significant impact of beaver dams on water flow. There were instances where beaver dams upstream and downstream of our sampling sites resulted in no detectable flow.

Discharge varied considerably between the study streams and also throughout the season. Maximum recorded discharged ranged from 0.02 m^3/sec in Lost Creek, to 5.14 m^3/sec in Black River at Hwy 304. The maximum discharge for each stream is shown in Table 3.

Table 3. Maximum discharge measured in each of the study streams in 2004.

Water body	**Max. Discharge (m^3/sec)**
Lost	0.02
Strawberry	0.17
O'Hanly mid tributary	0.22
Kapukwaywetewonk Creek	0.25
Maple Creek	0.35
Kenny Creek	0.39
Beaver Creek Crossing 7	0.39
Beaver Creek Crossing 12	0.40
Duncan Creek	0.54
Beaver Creek Crossing 10	0.58
Drew Creek	0.65
Rabbit River	3.29
Black River at 314	3.36
Cat Creek	3.55
O'Hanly River at 304	3.99
Black River at 304	5.14

Discharge varied throughout the season and generally followed the pattern of precipitation. Figure 16 shows the seasonal trend in discharge for 3 larger streams (Cat Creek, Rabbit River and Black River at 314) and 2 smaller streams (Kenny Creek and Beaver Creek Crossing 7). In general, discharge was highest in May, reflecting a delayed spring runoff (due to cold weather) and significant precipitation, and then declined to seasonal lows in July and August. Discharge then peaked again in September. The changes in flow and discharge are also shown graphically in Figure 17.



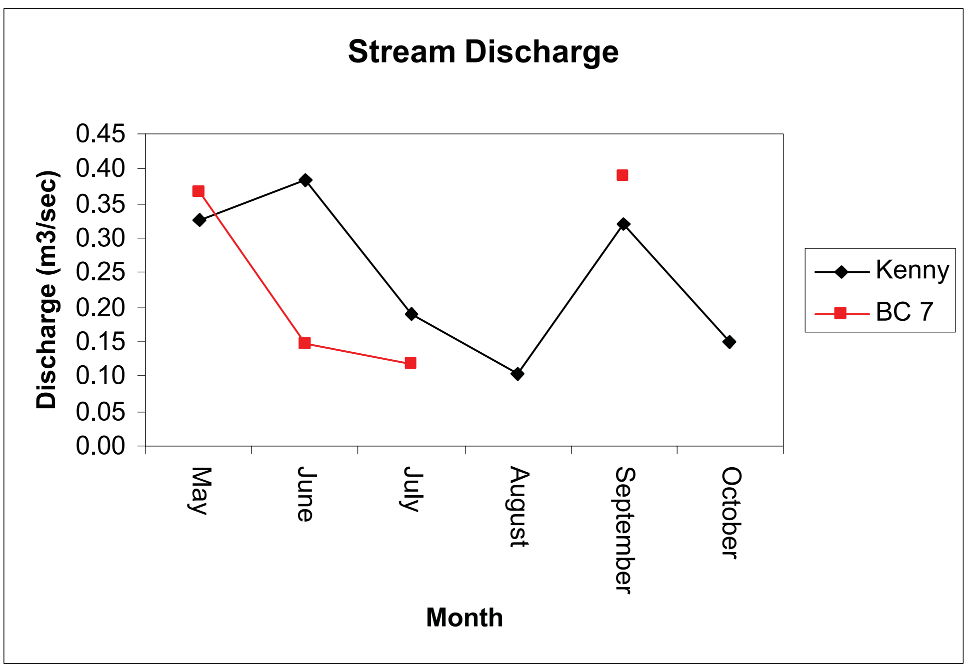


Figure 16. Seasonal variation in discharge (m^3/sec) in larger (top) and smaller (bottom) study streams in 2004.

Figure 17. Cat Creek in June (left) and August (right) 2004, demonstrating significant changes in water level and discharge.

Water Temperature

Water temperature was measured hourly from May to October using Onset HOBO Water Temp Pro data loggers. Data is only shown for a selection of the study streams. In general, the smaller streams and rivers with lower water volume warmed more quickly in the spring and cooled more rapidly in the fall, but the small streams/creeks and large rivers all attained a maximum temperature between 22 and 25^{o}C in the summer (Figures 18, 19 & 20). The one exception to this trend was Kenny Creek, a small creek which was consistently much cooler than the other streams (Figure 18). Peak water temperature in Kenny Creek was approximately 18 to 20^{o}C in mid July. The relatively cool water temperatures may be due to a higher proportion of groundwater feeding the creek, or the occurrence of frozen/cool peat in the watershed. Almost half of the watershed is covered by organic soils (Figure 8). There were occasions when the water level in the streams receded to such an extent that the temperature loggers were located out of the water. This is indicated in Figures 18 to 20 by missing data points.

There was an interesting pattern of diurnal fluctuations in water temperature in all the study streams. The changes in water temperature over a single day were much more pronounced in the smaller streams (Figure 18) than in the larger stream/rivers (Figures 19 & 20). Diurnal variation in water temperature in the small streams (such as Beaver Creek Crossing 10 and Rabbit River-Figure 18) was 7-8^{o}C, whereas diurnal fluctuation in water temperature in the larger study streams (such as the Black, Manigotagan, Rice and Wanipigow rivers-Figures 19 & 20) was generally less than 3^{o}C. The higher water volumes in the latter streams created a more stable thermal environment.

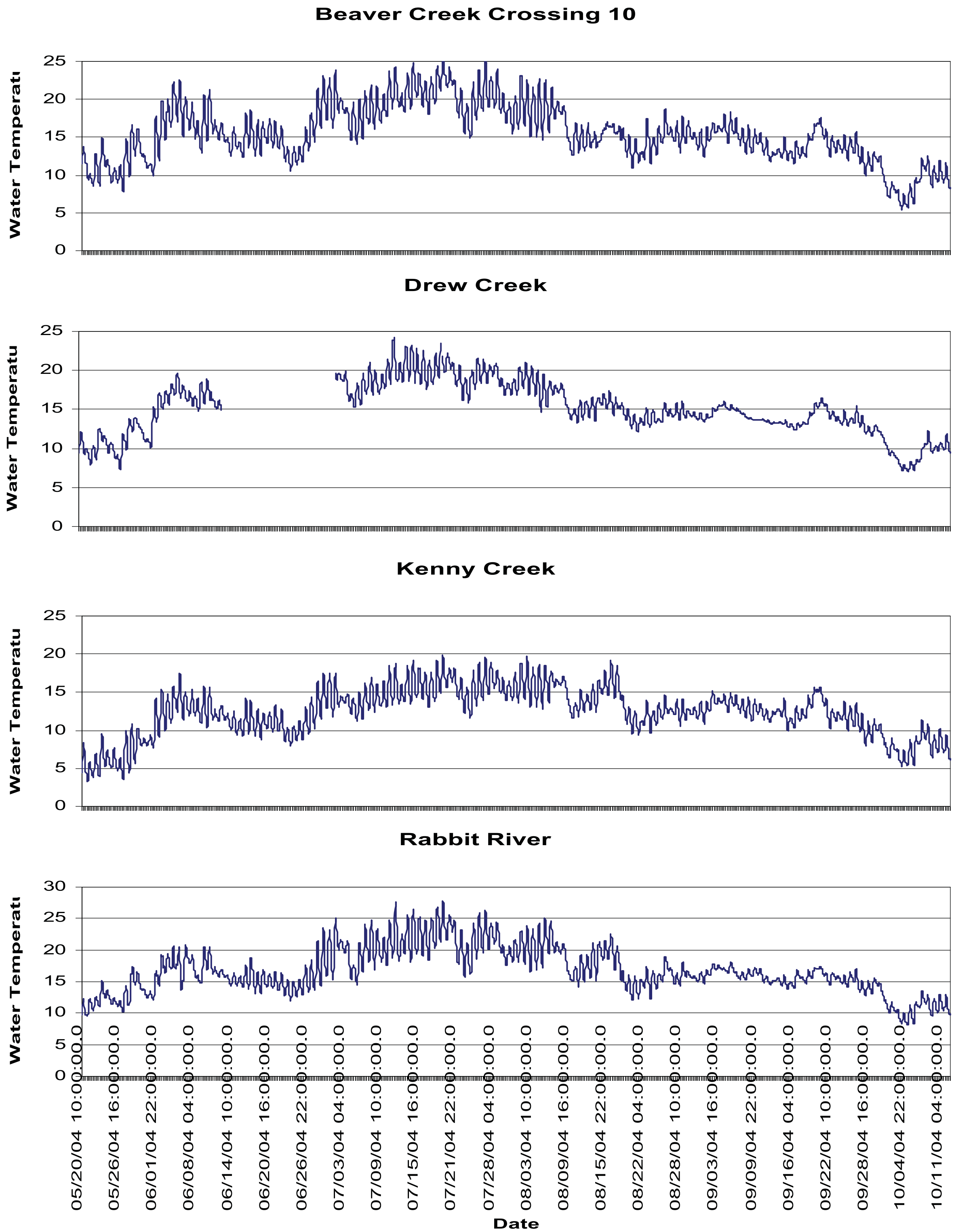


Figure 18. Hourly water temperature in 4 small study streams, recorded using Onset data loggers.

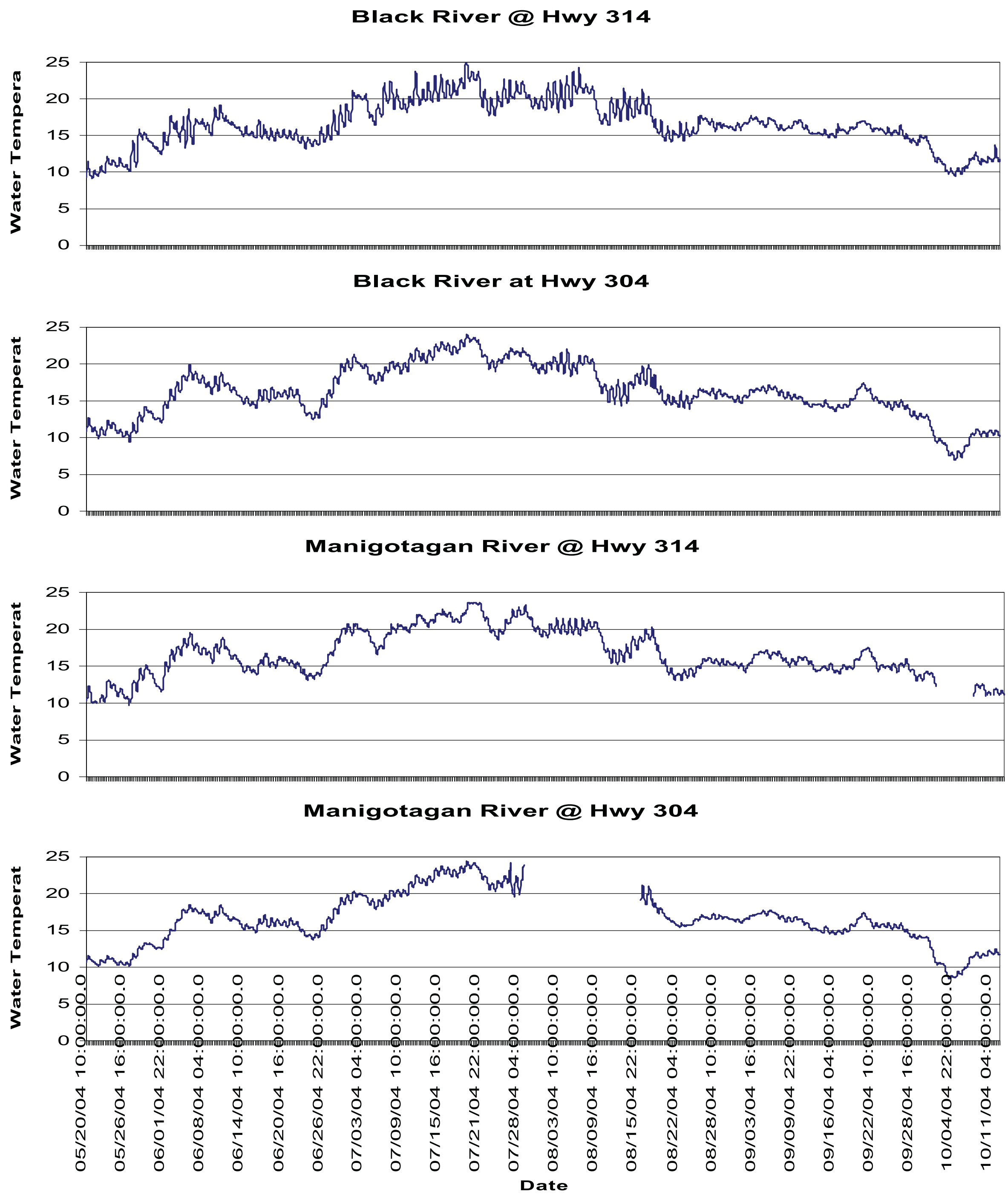


Figure 19. Hourly water temperature in 4 larger study streams, recorded using Onset data loggers.

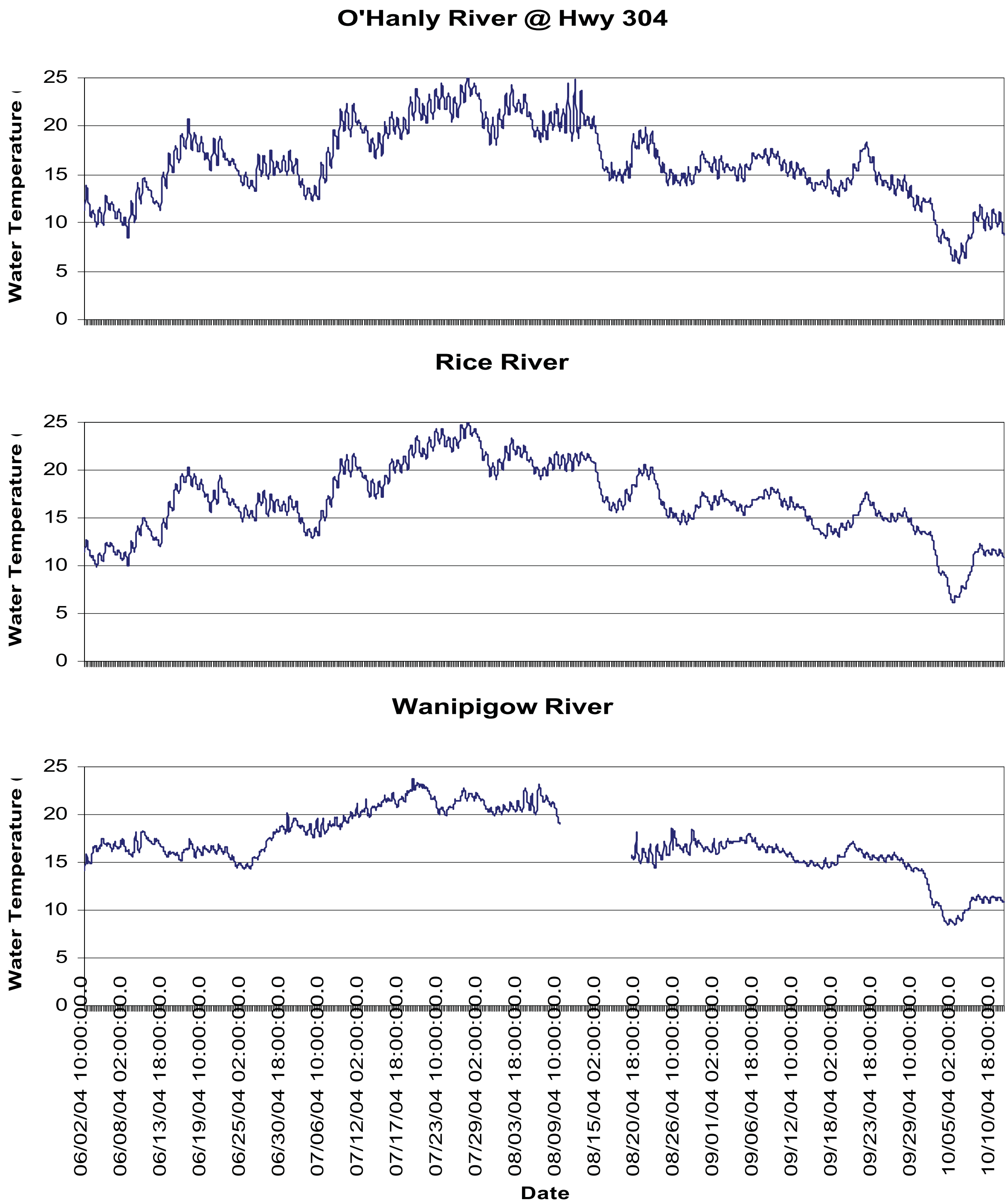


Figure 20. Hourly water temperature in 3 larger study streams, recorded using Onset data loggers.

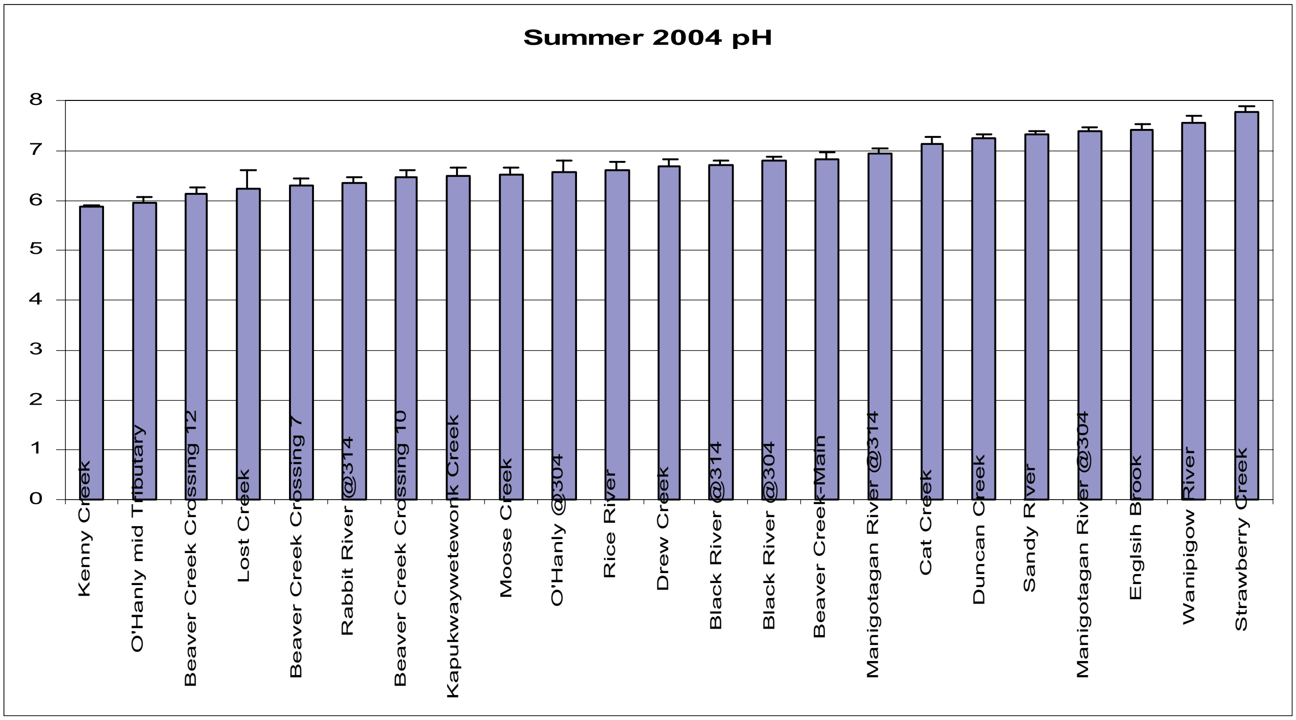
Water Quality in Study Streams

Average Water Quality Conditions and General Relationship to Watershed Characteristics and Disturbance

Water quality in the streams was studied both during the open water period (May to October, 2004) and during the winter (January to March, 2004, 2005). A brief summary of the key water quality characteristics of the streams is provided below, based on mean values from the open water period and winter periods.

pH, Alkalinity and Conductivity

In the summer of 2004, the majority of the study streams had slightly acidic water (Figure 21) while a few had slightly alkaline water. The pH ranged from 5.87 in Kenny Creek to 7.97 in Maple Creek. Most of the study streams in the forested watersheds had low alkalinity, an indicator of buffering capacity to acid. Alkalinity in many of the streams was less than 30 mg/L $CaCO_3$ (Figure 21). Other streams had alkalinity values between 50 and 160 mg/L, with the highest alkalinity values occurring in Maple Creek and Strawberry Creek, two watersheds containing agricultural activity. Alkalinity and pH were correlated (r = 0.74, P<0.0001). Conductivity in the streams ranged from 26 uS/cm in Rabbit River to 412 uS/cm in Maple Creek (Figure 22). In general, most streams in the forested watersheds had low conductivity, indicating a relatively low amount of dissolved ions and low salinity, typical of streams on the Canadian Shield. As with pH and alkalinity, the highest conductivity values occurred in Maple Creek and Strawberry Creek. Higher values for conductivity, pH and alkalinity appeared to be related to soil and forest type in the watersheds, as well as agricultural disturbance (discussed in more detail later), and to a much lesser extent, logging and fire.



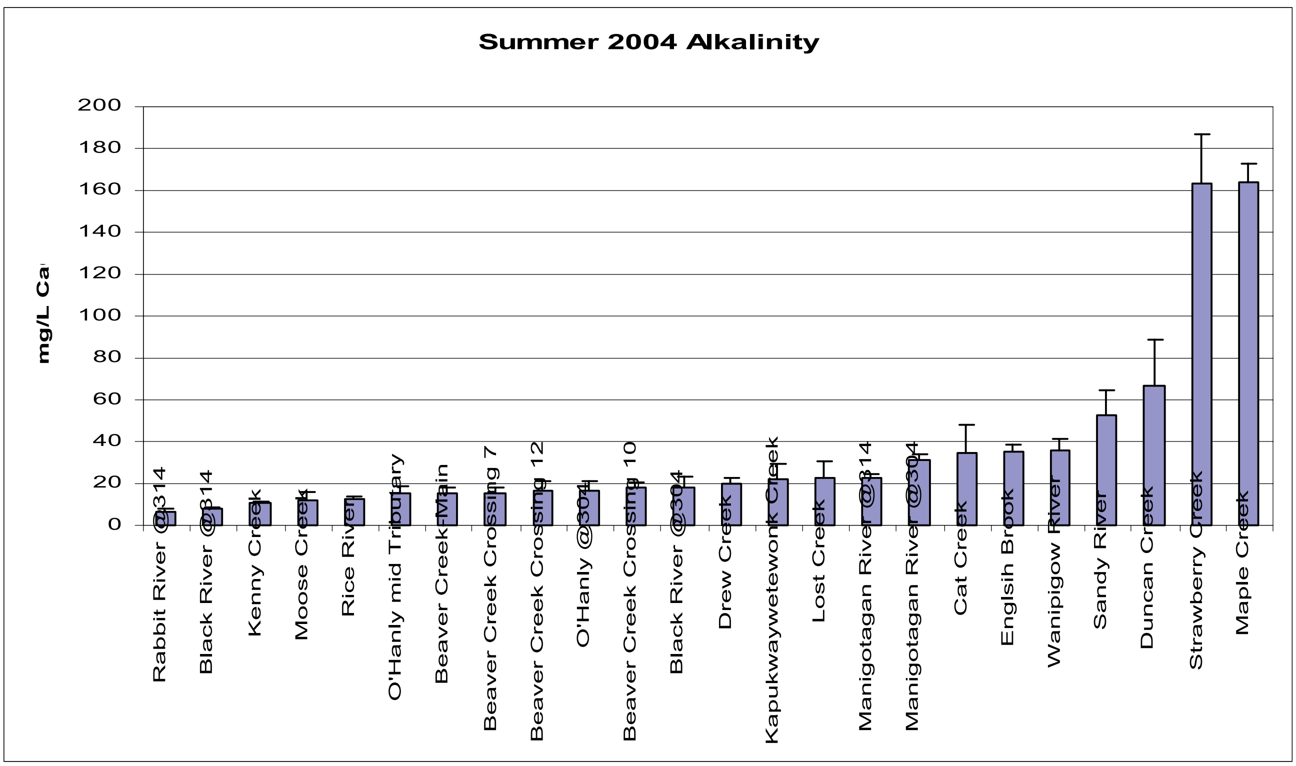


Figure 21. Water pH and alkalinity in the study streams in the summer of 2004. Vertical bars are 1 standard deviation of the mean.

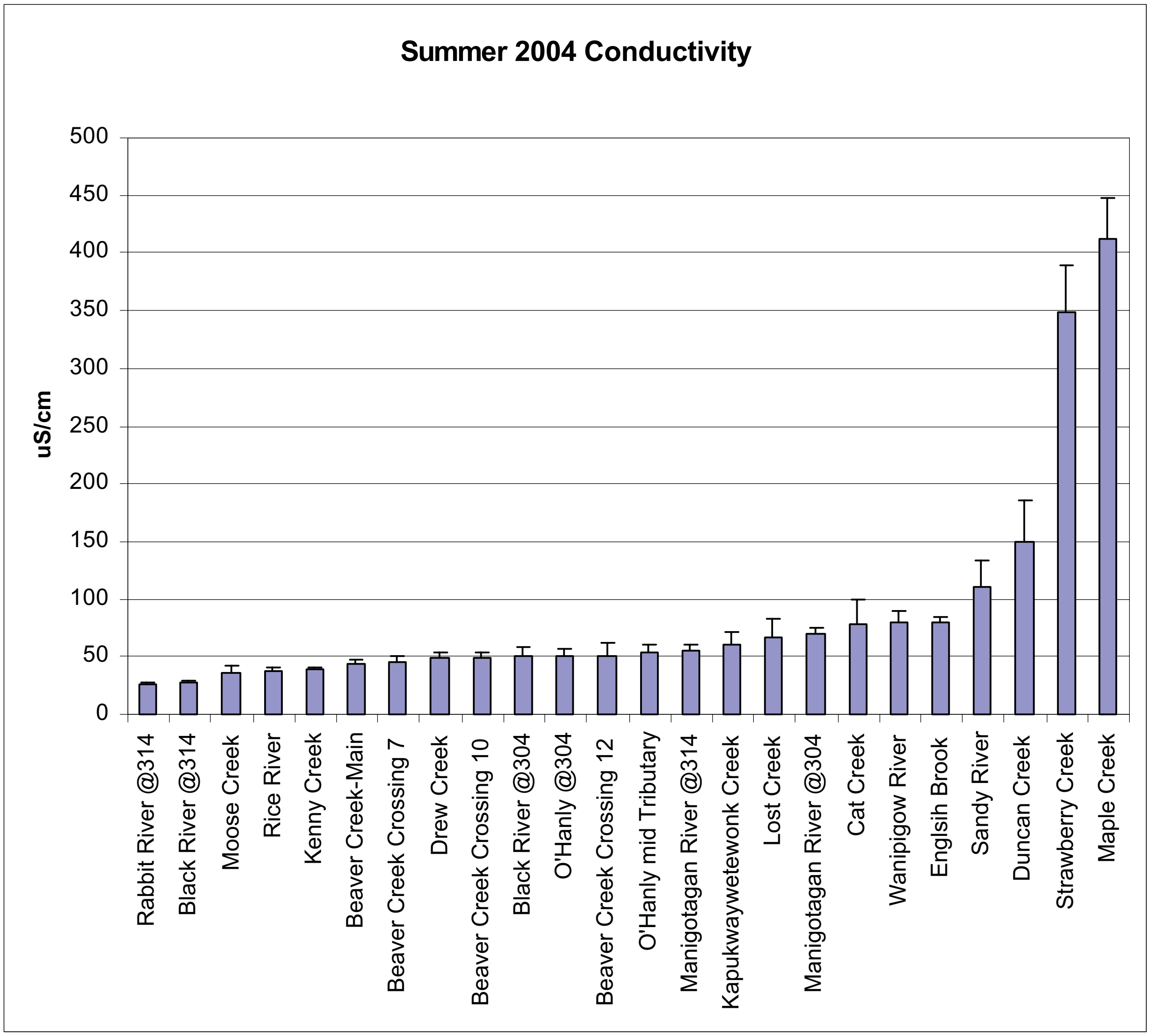


Figure 22. Conductivity in the study streams in the summer of 2004. Vertical bars are 1 standard deviation of the mean.

Turbidity

Many of the study streams can be described as having clear water with little suspended particulates. Turbidity, measured in nephalometric turbidity units (NTU), for the majority of the streams was less than 10 NTU (Figure 23), indicating relatively clear water. Streams with higher turbidity in forested watersheds, such as Cat Creek, Sandy River, and English Brook were associated with stream bottoms containing a high proportion of clay relative to other streams. These streams also tended to have significant local beaver activity, which may have contributed to the turbidity. Strawberry Creek and Maple Creek, in watersheds containing agricultural activity, also had high water turbidity.

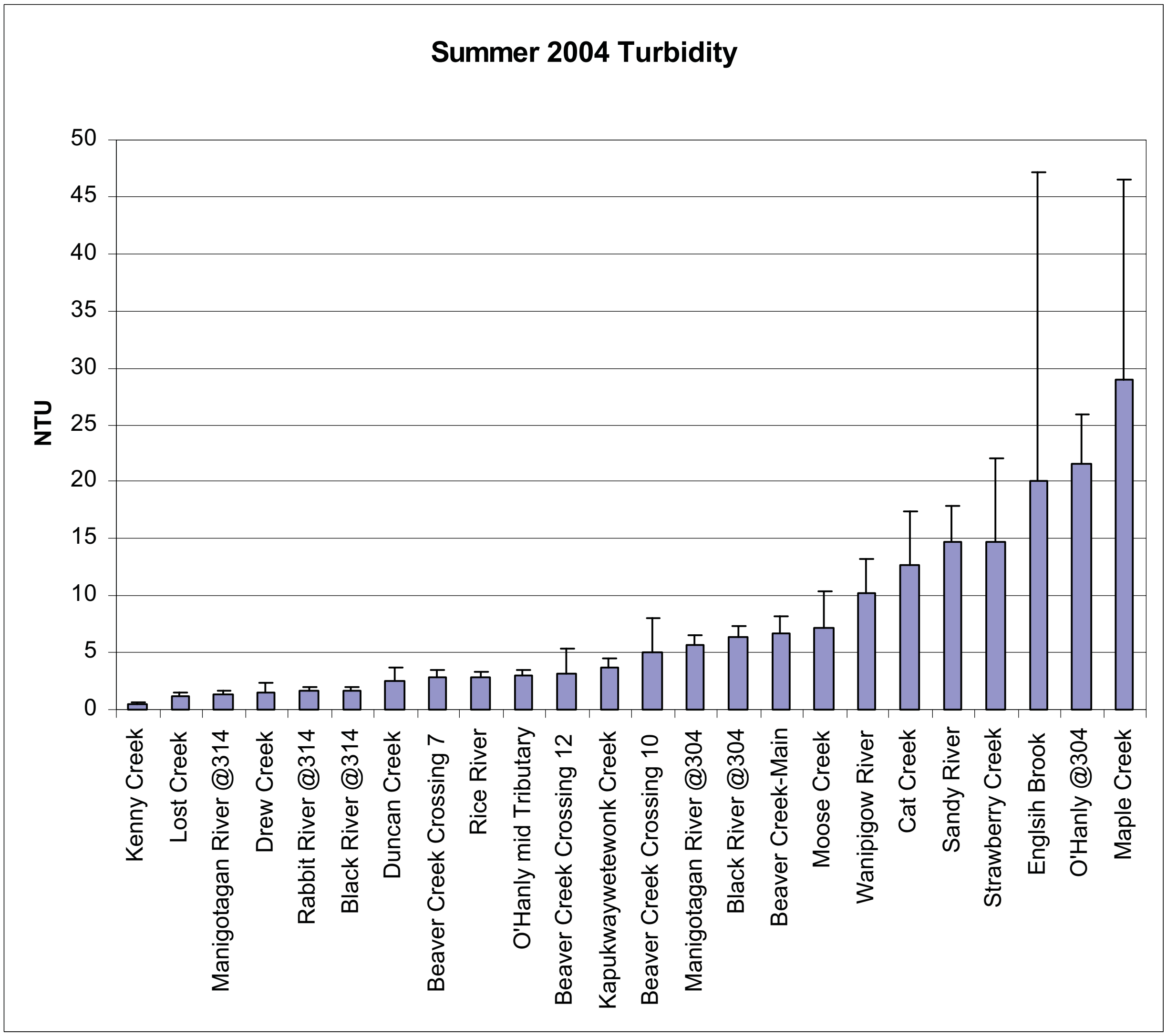


Figure 23. Turbidity in the study streams in the summer of 2004. Vertical bars are 1 standard deviation of the mean.

Dissolved Oxygen

Dissolved oxygen is an important water quality parameter because it can have a significant influence on both the organisms inhabiting a water course, as well as controlling the dynamics of several other water quality variables such as nutrients (Wetzel, 1983). There were several interesting trends in dissolved oxygen in the study streams, both in the summer and in the winter periods.

In the summer of 2004, most of the large rivers and larger streams had dissolved oxygen concentrations that were near saturation conditions (i.e., 80 – 100% saturation-Figure 24). This represents concentrations of dissolved oxygen of 8.0 to 9.5 mg/L. I should not be surprising that even in the large rivers, dissolved oxygen concentrations are not at 100% saturation in the summer. As the water temperature increases throughout the summer, water is able to hold less dissolved oxygen (Wetzel, 1983). The relatively high dissolved oxygen levels in the large rivers and streams throughout the summer also indicates that water turbulence (rapids, water falls) and photosynthesis by algae and rooted plants likely more than offset any decomposition processes (which remove oxygen) occurring in the water. However, this does not seem to be the case for the smaller streams. For example, Beaver Creek Crossings 7 and 12, O'Hanly mid tributary, Duncan Creek and Lost Creek all had water that contain significantly less oxygen than the larger streams and rivers (Figure 24). These smaller streams had water with less than 50% saturation of dissolved oxygen. In terms of dissolved oxygen concentration, this represents concentrations of 2.60 to 5.41 mg/L. There are likely two explanations for this. Firstly, these streams had a significant amount of beaver activity (e.g., dams), which impeded water flow and backed up water on the upstream side of the dams, flooding adjacent riparian areas (sedge meadows and/or forest). Lack of flow will affect the ability of the water to replenish the dissolved oxygen through water cascading over rapids. Secondly, the back flooding, in combination with the lack of flow will enhance decomposition of organic matter in the bottom sediments and in the soils of the back-flooded areas. It therefore appears that beaver activity in the smaller streams may have a detrimental impact on dissolved oxygen in the water, even in the summer. This could be even more significant in the winter when the streams are ice-covered (and thus, cut off from the atmosphere-a source of oxygen). Lack of oxygen can not only have a detrimental effect on organisms such as aquatic invertebrates (insects) and fish, but can have profound impacts on water quality (discussed later).

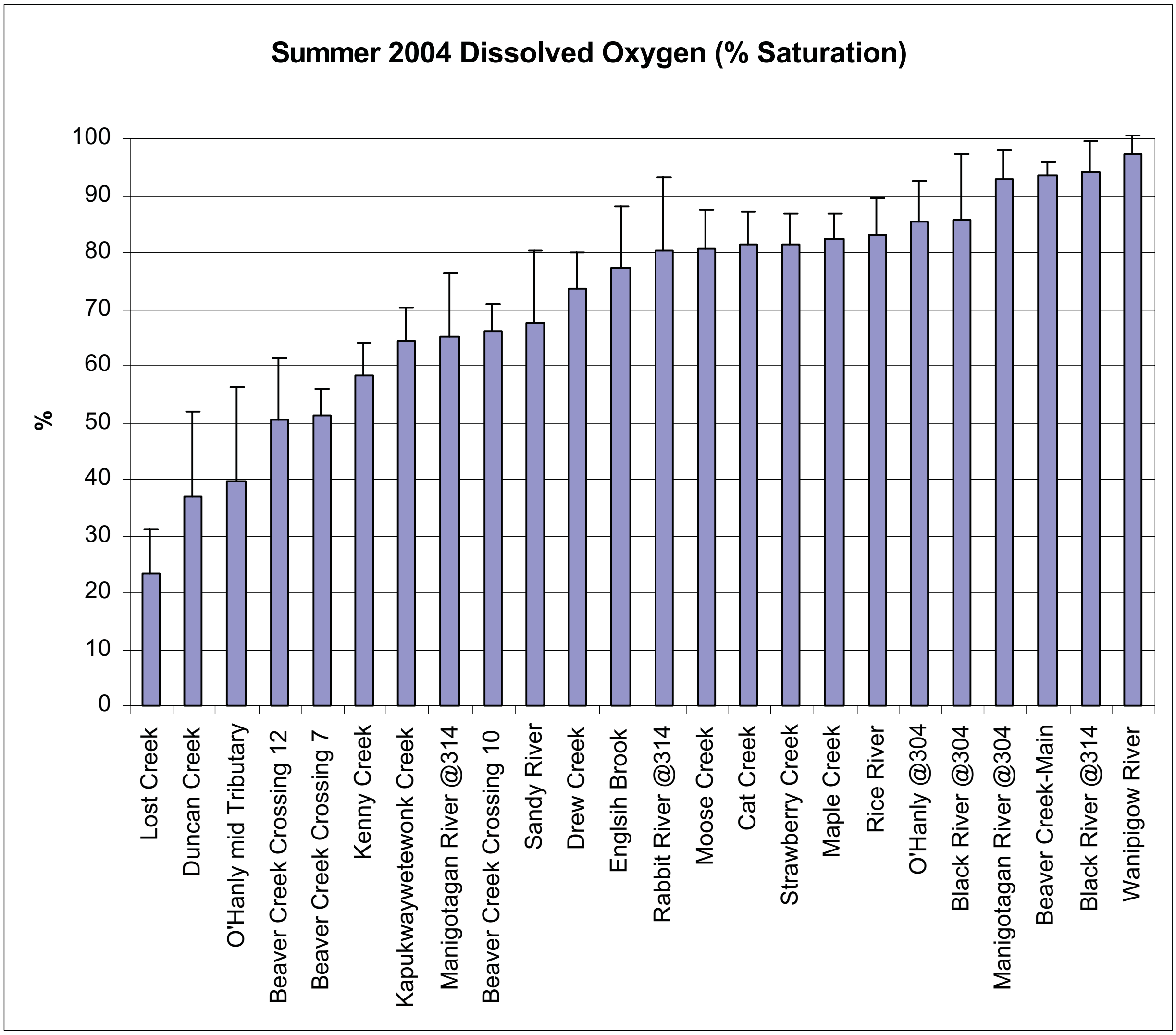


Figure 24. Dissolved oxygen (expressed as % saturation of the water) in the study streams in the summer of 2004. Vertical bars are 1 standard deviation of the mean.

It is interesting to note that Maple Creek and Strawberry Creek, despite their small size, contain more dissolved oxygen than other small streams in the forested watersheds to the north. This may be due to less beaver activity in these drainages or stream bottoms that are dominated by clay, rather than organic muck (as is the case in most of the small streams in the forested watersheds).

A similar pattern in dissolved oxygen was observed in the streams during the winter of 2005 (January to March, 2005). In fact, the trend in low dissolved oxygen in the small streams was even more pronounced in the winter compared to the summer period. Small streams such as Beaver Creek Crossing 7 and 12, Duncan Creek and Kenny Creek contained water having only 6 to 30% dissolved oxygen (Figure 25). This represents dissolved oxygen concentrations of only 0.79 to 4.43 mg/L. The larger rivers containing sections of open water (in areas of rapids or water falls) had much higher dissolved oxygen (80 to 104% saturation). Values in excess of 100% indicate super-saturation, which can occur in cold water with plenty of turbulence to oxygenate the water. It appears that low dissolved oxygen concentrations during the winter in small, ice-covered streams in forested watersheds in eastern Manitoba may be much more common that currently thought. These streams generally go into the winter period with already-low dissolved oxygen levels, and dissolved oxygen is further reduced over the winter period by decomposition processes in the sediments. In many of the smaller streams, no flow could be detected in the winter (due to impediment of water flow by beaver dams), which would further exacerbate the problem. As will be discussed later, low dissolved oxygen does not appear to be related to logging, as two of the streams (Beaver Creek Crossing 7 and 12) have no harvesting in their watersheds), but may be related to fire disturbance.

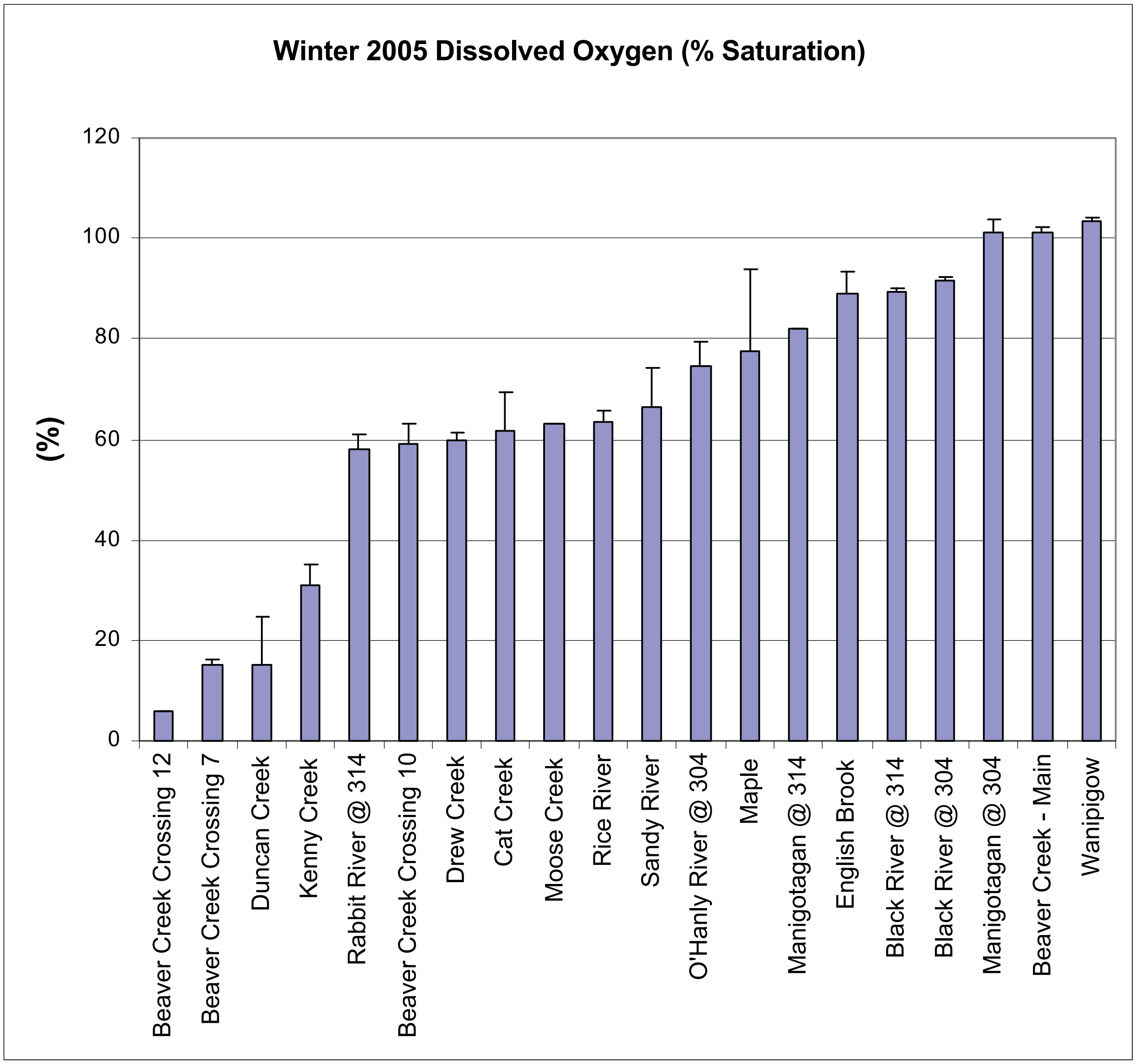


Figure 25. Dissolved oxygen (expressed as % saturation of the water) in the study streams in the winter of 2005. Vertical bars are 1 standard deviation of the mean.

In addition to sampling the surface water (right under the ice) for dissolved oxygen in winter, we also measured dissolved oxygen throughout the water column from right below the ice to the bottom sediments. In all cases, dissolved oxygen levels were uniform from the top to the bottom of the water column, except right at the water-sediment interface, where dissolved oxygen levels dropped off to near 0. This indicates that the sediments are anoxic, likely due to decomposition of organic matter occurring in the sediments.

At low dissolved oxygen concentrations, electronic meters such as the one used in this study, may not provide accurate results. To evaluate this, we took water samples in addition to measuring dissolved oxygen with the meter on one sampling trip in winter. The water samples were sent to Envirotest Laboratories in Winnipeg for dissolved oxygen analysis using a titration method (widely believed to be more accurate than electronic meters). The results indicated that there was a good correspondence between dissolved oxygen measured with the meter, and that measured using the Winkler Titration method (r^2=0.97, P< 0.0001 – Figure 26).

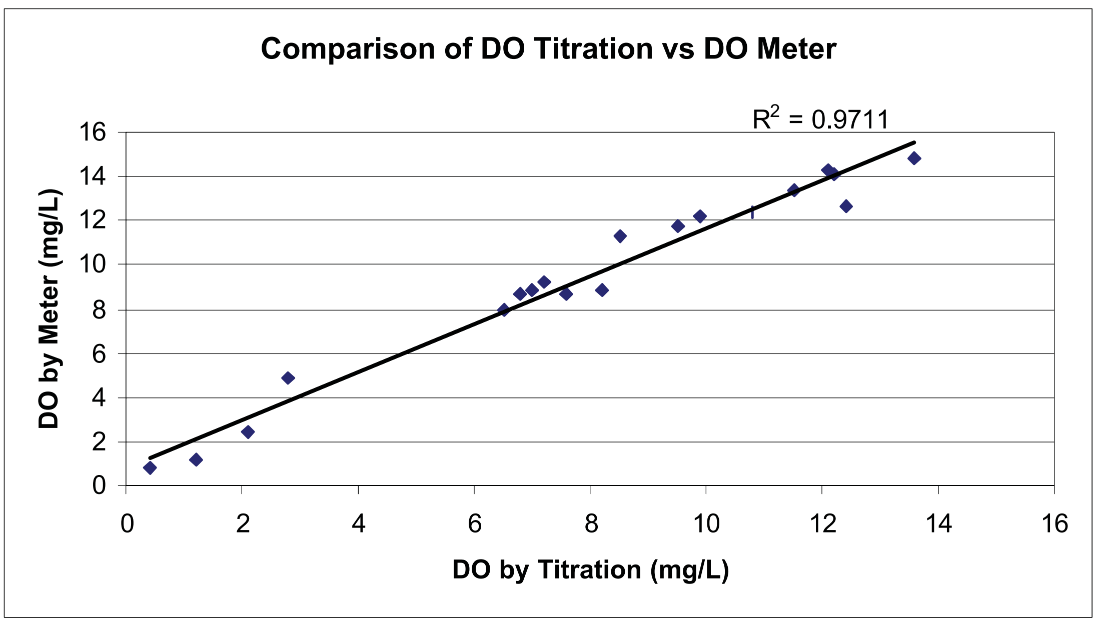


Figure 26. Relationship between dissolved oxygen measure by Winkler titration method and a YSI electronic meter.

Phosphorus

As mentioned previously, phosphorus is a key nutrient that is essential for the growth of algae and macrophytes (higher plants) in aquatic ecosystems. It is also commonly the most limiting of the plant nutrient in Canadian lakes and rivers (Schindler, 1971). Excessive loading of phosphorus to water bodies (through sewage, fertilizers, from soils after land use conversion) has resulted in the eutrophication (nutrient enrichment) of lakes and rivers worldwide, causing a myriad of water quality problems such as massive algal blooms, fish kills, loss of diversity of benthic (bottom-dwelling) organisms and the occurrence of toxic algae. This is currently the case in Lake Winnipeg (Lake Winnipeg Stewardship Board, 2005).

There was a wide range in total phosphorus concentration between the study streams in the summer of 2004. TP values in streams in forested watersheds ranged from 18 and 19 ug/L in two rivers in Nopiming Park (Black River at Hwy 314 and Rabbit River, respectively) to 115 ug/L in O'Hanly mid tributary (Figure 27). It is interesting to note that the streams in forested watersheds with the highest TP concentrations (e.g., O'Hanly River, Sandy River, Duncan Creek, O'Hanly mid tributary) either experienced significant disturbance by fire (Duncan Creek), logging (O'Hanly River, O'Hanly mid tributary) or both fire and logging (Sandy River) and that this effect on TP might be further influenced by soil type. This will be discussed in more detail later. As with many of the other water quality parameters, summer TP concentrations were highest in Maple Creek and Strawberry Creek (166 and 188 ug/L, respectively – Figure 27), almost 4 times higher than the average of the study streams in forested watersheds.

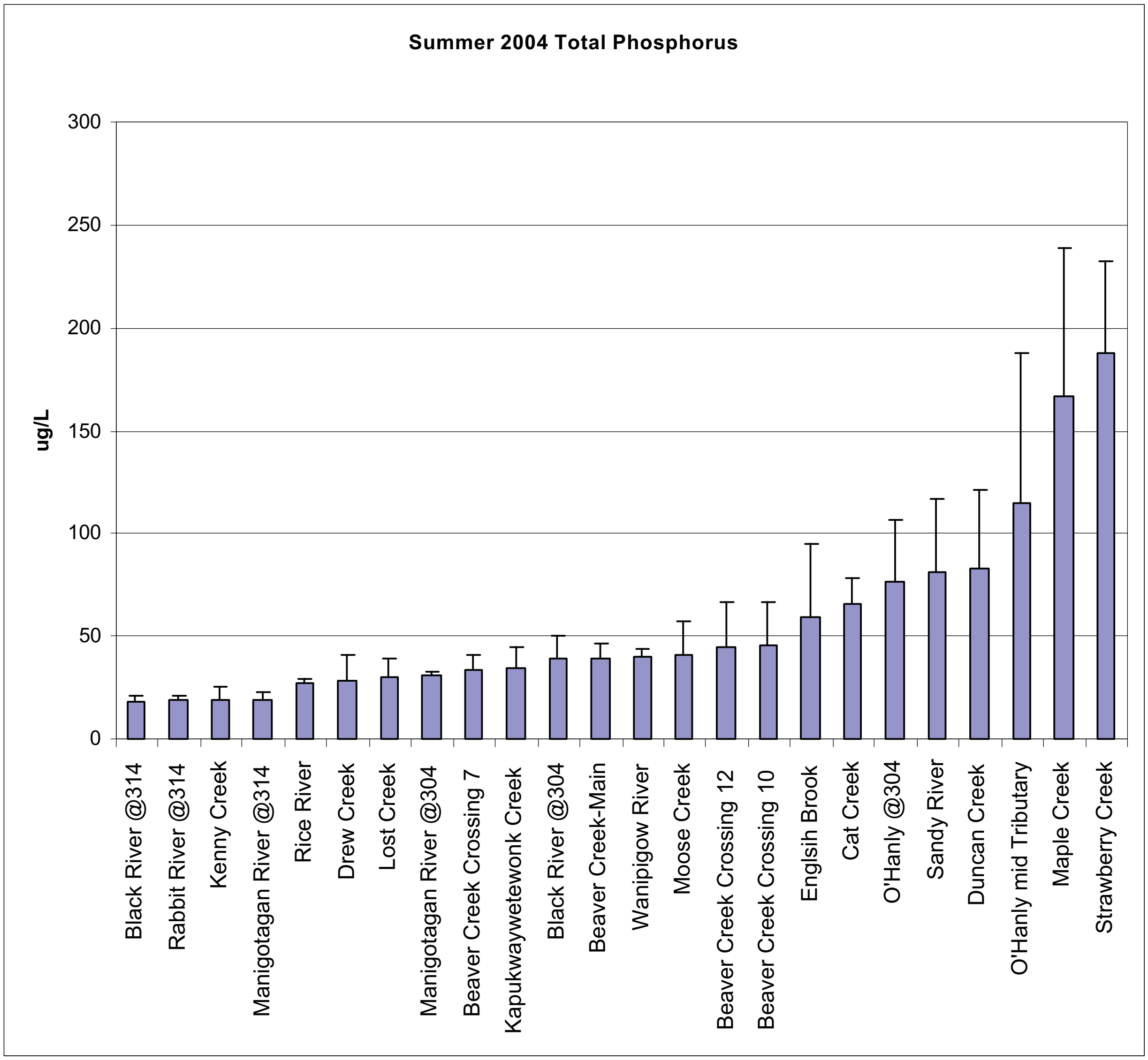


Figure 27. Total Phosphorus concentration in the study streams in the summer of 2005. Vertical bars are 1 standard deviation of the mean.

For many of the streams and even small rivers, there was a dramatic difference in TP between the summer of 2004 and the two winter periods. Figure 28 shows TP concentrations in the study streams during the winters of 2004 and 2005. Several of the smaller streams such as Beaver Creek Crossings 7, 10 & 12 and Drew Creek, which had low TP concentrations in the water

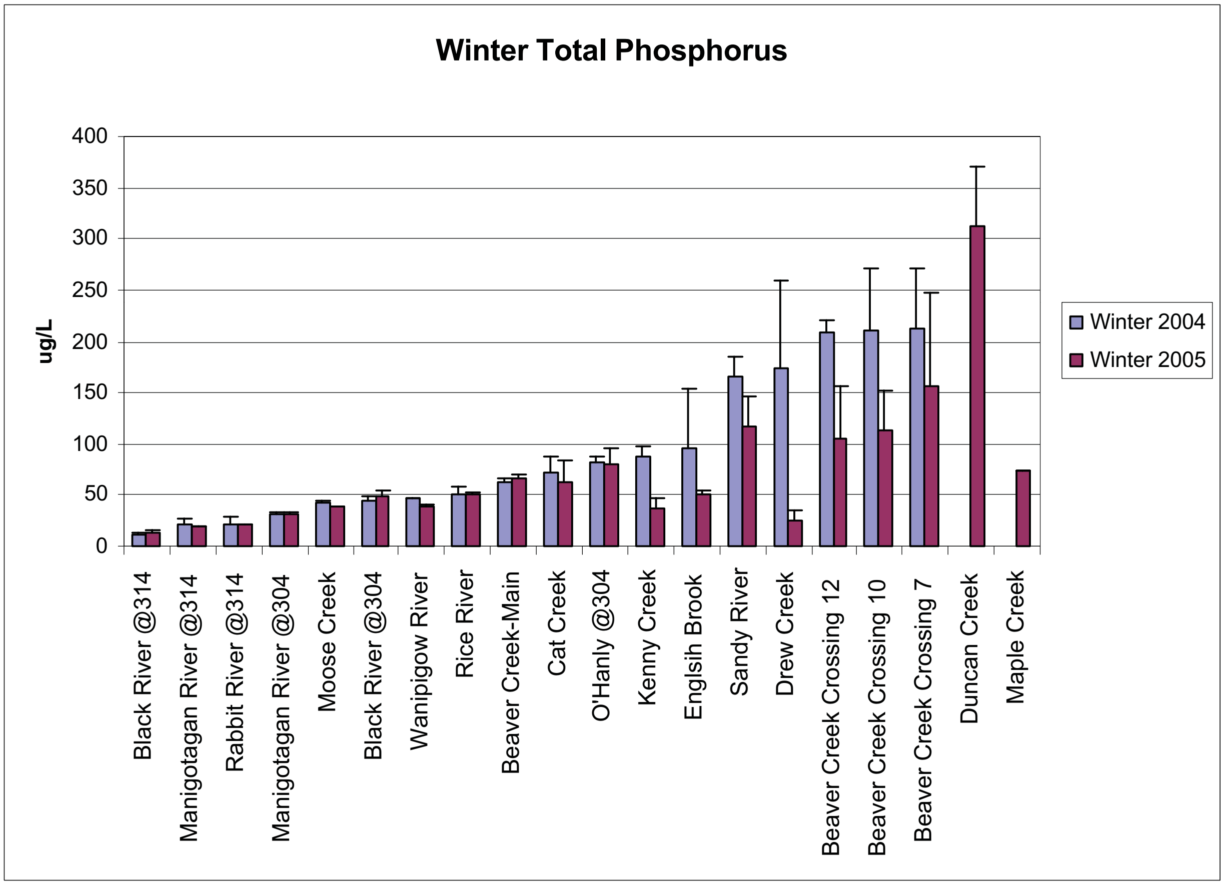


Figure 28. Total Phosphorus concentration in the study streams in the winter of 2004 and 2005. Vertical bars are 1 standard deviation of the mean.

during the summer relative to the other streams, had much higher concentrations in the winter, moving them to the right hand side of the graph in Figure 28. These are also the same streams that contain low dissolved oxygen concentrations during the winter. Low dissolved oxygen levels in the water likely caused the release of phosphorus from the sediments, a common phenomenon observed in lakes worldwide. There was a direct relationship between average winter dissolved oxygen concentrations in the streams and average TP concentrations (r = -0.61, P= 0.005).

For some of the streams and smaller rivers, the reduction in dissolved oxygen over the winter and its impact on phosphorus concentration, was very dramatic. Table 4 shows the percent increase in TP concentration in the winter compared to summer values for the winters of 2004 and 2005. Smaller streams such as Kenny Creek, Beaver Creek Crossing 7, 10 & 12, Drew Creek and Duncan Creek experience massive increases of TP in the winter, relative to the summer (up to a 500% increase). These are also the streams with significant beaver activity, minimal flow (especially in the winter) and low dissolved oxygen in the winter. It is interesting to note that in the winter of 2004, TP in Drew Creek increased by 523% over summer TP values but that in the winter of 2005, TP values were very similar to the summer values (Table 4). This can be explained by a large difference in water flow between the two winters. In the winter of 2004, there was no flow of water. The water was stagnant and due to decomposition, lost most of its dissolved oxygen, and thus released large amounts of phosphorus from the sediments. In contrast, water in Drew Creek flowed continually in 2005, and there was a large patch of open water all winter (providing oxygen to the creek). Thus, we did not observe a large increase in TP in the winter of 2005 compared to that observed in the winter of 2004.

Rivers such as Black River at Hwy 314 actually experienced a moderate decrease in TP in the winter relative to summer concentrations, which is likely related to the much lower flows in winter and therefore less export of TP from the watershed. Other rivers such as the Manigotagan, O'Hanly, Rabbit and Wanipigow show almost no change in TP from summer to winter (Table 4), likely because these rivers all have fairly high dissolved oxygen concentrations throughout the year.

Table 4. Percent increase in TP concentration in the study streams in winter, relative to summer concentrations.

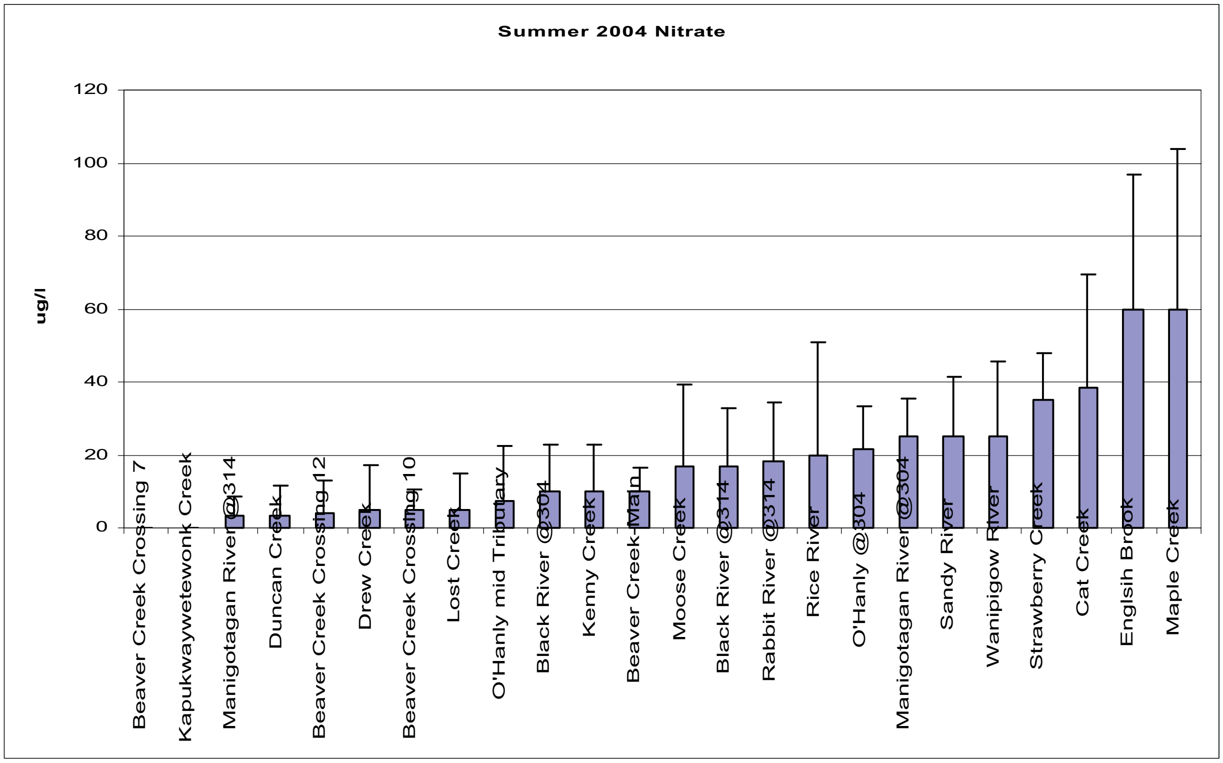
Water Body	% Increase in TP in Winter Compared to Summer	
	Winter 2004	Winter 2005
Black River @ 314	-35	-30
Manigotagan River @ 304	0	-2
Moose Creek	4	-3
Manigotagan River at 314	5	3
O'Hanly River	7	4
Cat Creek	11	-5
Rabbit River	12	10
Black River @ 304	15	23
Wanipigow River	16	-4
Beaver Creek – Main	58	67
English Brook	61	-14
Rice River	91	91
Sandy River	104	44
Kenny Creek	346	87
Beaver Creek Crossing 10	360	147
Beaver Creek Crossing 12	369	139
Beaver Creek Crossing 7	522	358
Drew Creek	523	-6
Duncan Creek	ND	275

ND = no data. Duncan Creek was not sampled in the winter of 2004.

Nitrogen

Inorganic forms of nitrogen, another key nutrient in aquatic ecosystems, were also studied. Figure 29 shows the concentration of nitrate (NO_3) and ammonia (NH_4) during the summer of 2004. Concentrations of NO_3 ranged from non-detectable in Beaver Creek Crossing 7 and Kapukwaywetewonk Creek to 60 ug/L in Maple Creek and English Brook. Concentrations of NH_4 ranged from non-detectable in Manigotagan River at 314 to over 40 ug/L in Maple Creek and the O'Hanly River (Figure 29). Unlike phosphorus, the higher concentrations of inorganic nitrogen in the summer did not appear to be consistently related to soil type or fire or logging history. However, both Maple Creek and Strawberry Creek had relatively high concentrations of inorganic nitrogen, and could be due to agricultural activity in their watersheds.

The same trend for significant increases in TP during the winter, were seen with both NO_3 and NH_4. This trend was more evident in NH_4 and is consistent with the fact that NH_4 is more common in waters with lower dissolved oxygen concentrations than is NO_3. Figure 30 shows the concentration of NH_4 in the streams during the winters of 2004 and 2005. Concentrations in the winter reach over 800 ug/L in Beaver Creek Crossings 7, 10 & 12, Drew Creek and Kenny Creek in 2004. In Kenny Creek, this was an increase of almost 7,500% over the summer concentration. It appears that inorganic nitrogen may be much more sensitive to changes in dissolved oxygen than phosphorus.



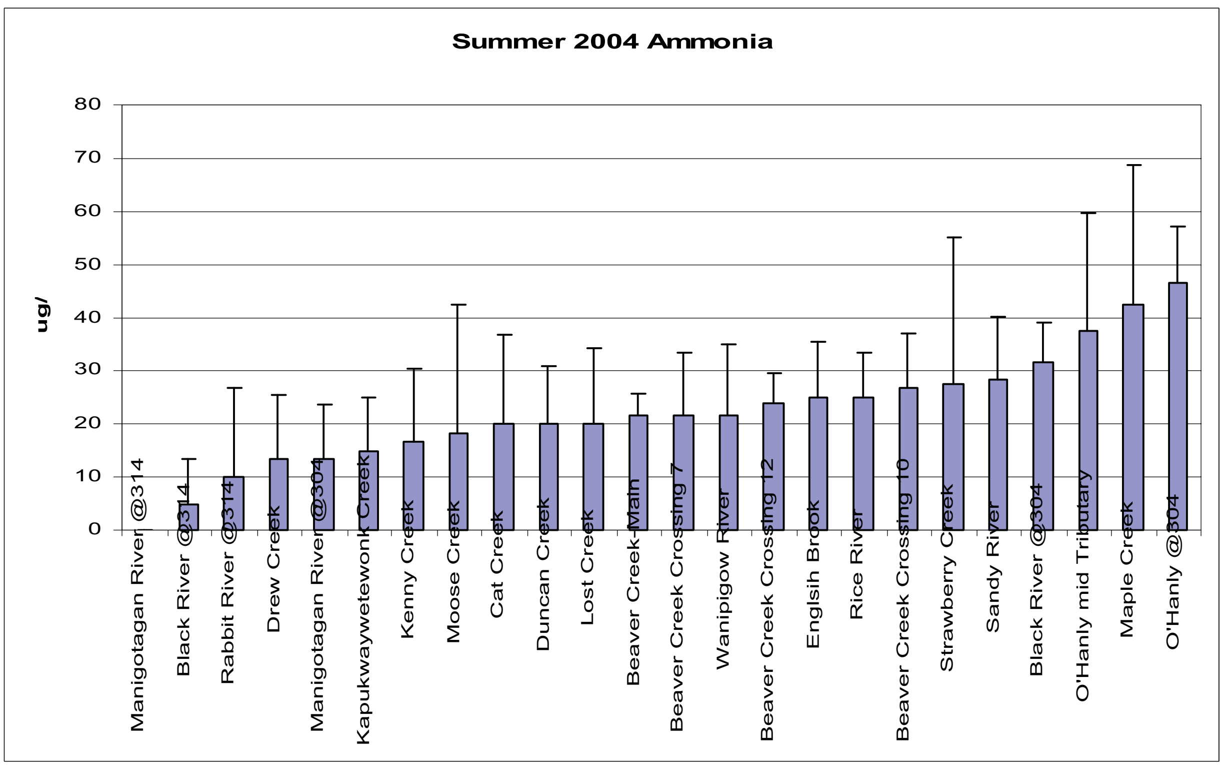


Figure 29. Nitrate (top graph) and ammonia (bottom graph) concentration in the study streams in the summer of 2004. Vertical bars are 1 standard deviation of the mean.

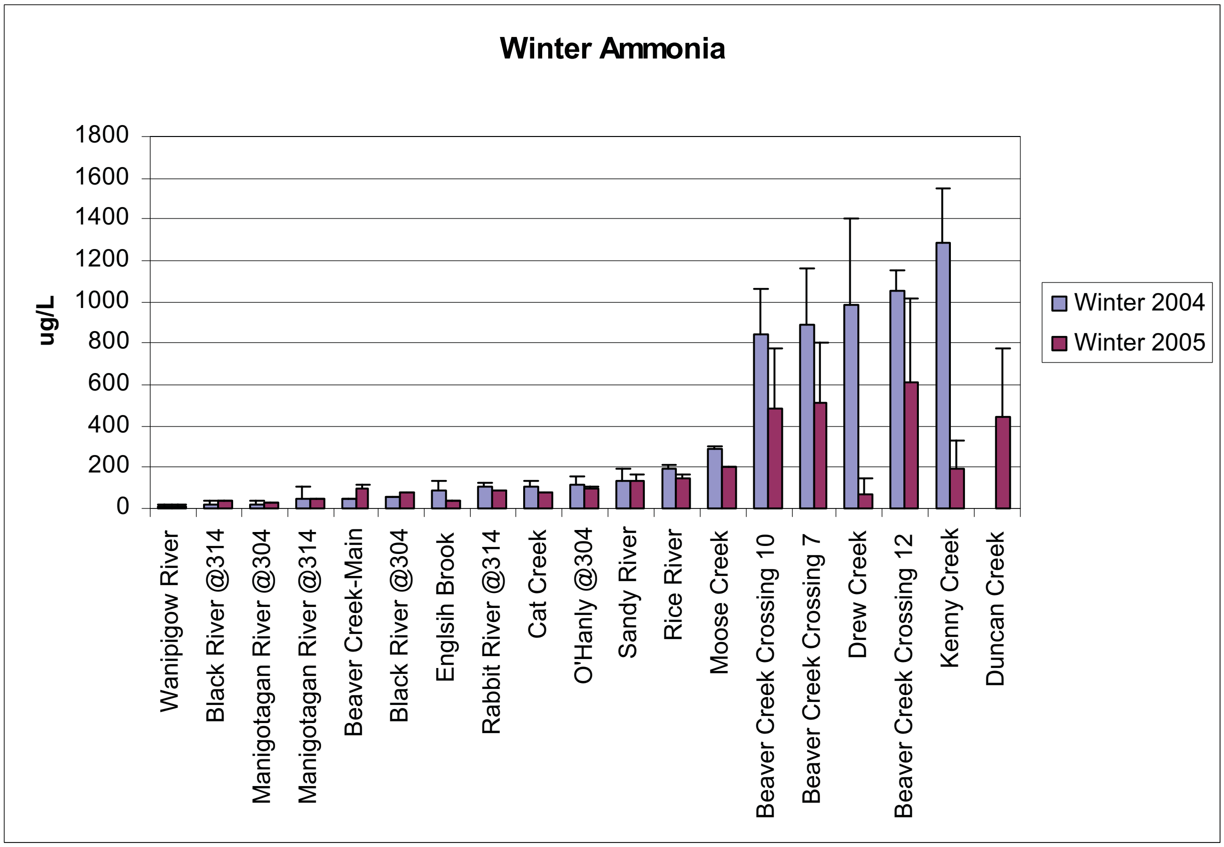


Figure 30. Ammonia concentration in the study streams in the winters of 2004 and 2005. Vertical bars are 1 standard deviation of the mean

In addition to inorganic nitrogen (NO_3 and NH_4), we also monitored Total Nitrogen (TN) in the streams. TN is the sum of inorganic and organic forms of nitrogen. It is calculated by adding the values of NO_3, NH_4 and Total Kjeldahl Nitrogen (TKN – organic nitrogen). TN gives a better indication of the total nitrogen pool in the streams than just NO_3 and NH_4. Summer TN concentrations ranged from approximately 500 ug/L in Black River at 314 and Manigotagan River at 314, to over 2,500 ug/L in Strawberry Creek (Figure 31). Both Strawberry Creek and Maple Creek had water with high TN concentrations, likely a function of the agricultural activity in their watersheds.

In addition, there appears to be a relationship between soil type and disturbance, and TN concentration in the streams. Streams in forested watersheds such as O'Hanly mid tributary, Lost Creek, Duncan Creek , O'Hanly River and Kapukwaywetewonk Creek had high TN concentrations (Figure 31). All of these streams share the common characteristics of watersheds dominated by organic (OD) and deep basin (DB) soils, and either significant logging disturbance (O'Hanly mid tributary-99% of watershed area, O'Hanly River-50%, Kapukwaywetewonk Creek-100%), fire disturbance (Duncan Creek-200% of watershed area) or both logging and fire disturbance (Lost Creek-100% of watershed area by logging and 100% by fire). At the other end of the spectrum are streams with the lowest TN concentrations (Black River at 314 – 521 ug/L TN, Manigotagan River at 304 – 638 ug/L TN) that have watersheds dominated by thin, bedrock soils (BR). Low TN concentration in the water appears to be related to the thin soils in combination with relatively little logging (< 10% of the watershed) and fire (< 33% of the watershed) disturbance. In summary, it appears that organic (OD) and/or deep basin (DB) soils may contribute more TN to the streams that bedrock soils, and that both logging and fire disturbance on OD and DB areas of watersheds may increase TN in receiving streams.

It should be noted however, that there are exceptions to this general trend. For example, Beaver Creek Crossings 10 and 12 both had high (1515 and 1568 ug/L, respectively – Figure 31) TN concentrations, despite occurring in watersheds that are dominated by BR type soils and no to little logging (0 to 6 % of watershed area) and little fire disturbance (16 to 33% of watershed area) in the last 60 years. These two streams did have significant beaver activity however.

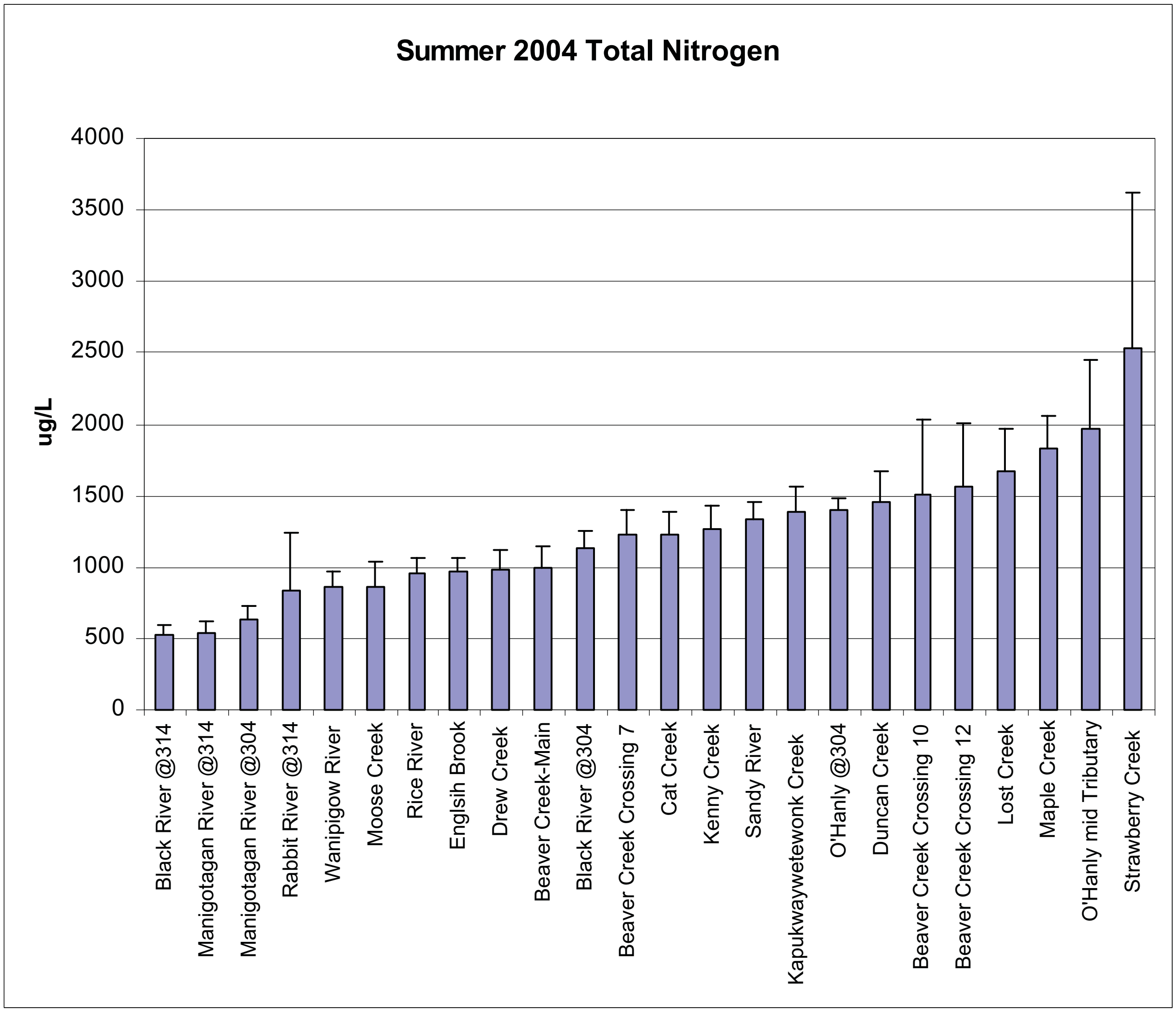


Figure 31. Total Nitrogen concentration in the study streams in the summer of 2004. Vertical bars are 1 standard deviation of the mean

Dissolved Organic Carbon

Water color was monitored in the study streams during the summer of 2004 and in the winters of 2004 and 2005. Water color was measured as Dissolved Organic Carbon (DOC). The higher the DOC value, the more brown the water is in color. Water with a pale straw (yellow/brown) color typically has a DOC value between 10 and 20 mg/L. Dark brown water typically has a DOC value greater than 30 to 40 mg/L. Much of the brown color of water in streams and rivers in the Canadian Shield can be attributed to humic and fulvic acids originating primarily from wetlands, including peatlands (bogs and fens).

All streams studied had at least a pale straw color. Streams with the least amount of color included the Manigotagan River (at Hwy 304 and 314), Black River at 314, Rabbit River and the Wanipigow River. DOC in these rivers ranged from 16 to 26 mg/L (Figure 32). At the other end of the spectrum were streams with extremely colored water. Strawberry Creek, Kapukwaywetewonk Creek, Lost Creek and the O'Hanly mid tributary had the highest DOC concentration, ranging from 57 to 79 mg/L (Figure 32). Of the three streams with the highest DOC concentrations, all are dominated by organic soils (OD in the enduring features classification) and all three have also experienced significant levels of disturbance in the last 60 year. For example, O'Hanly mid tributary (with the highest DOC) and Kapukwaywetewonk Creek had 100% of their watersheds logged in the last 60 years and Lost Creek had all of its watershed logged in the last 60 years as well as 100% of its watershed burn in the last 5 years.

Conversely, streams such as Manigotagan River at 304, Black River at 304 and Rabbit River all had low DOC and watersheds dominated by shallow soils over bedrock (BR/R2 enduring features). The lack of organic soils in these watersheds appears to be responsible for the lack of color in the water, despite the fact that both the Manigotagan at 304 and Rabbit River have had 30 to 40% of their watershed area burned in the last 30 years. It therefore appears that higher inputs of DOC to receiving streams in this region depends on the amount of organic soil in the watershed, as well as the level of disturbance (fire or logging). This is discussed in more detail in later sections.

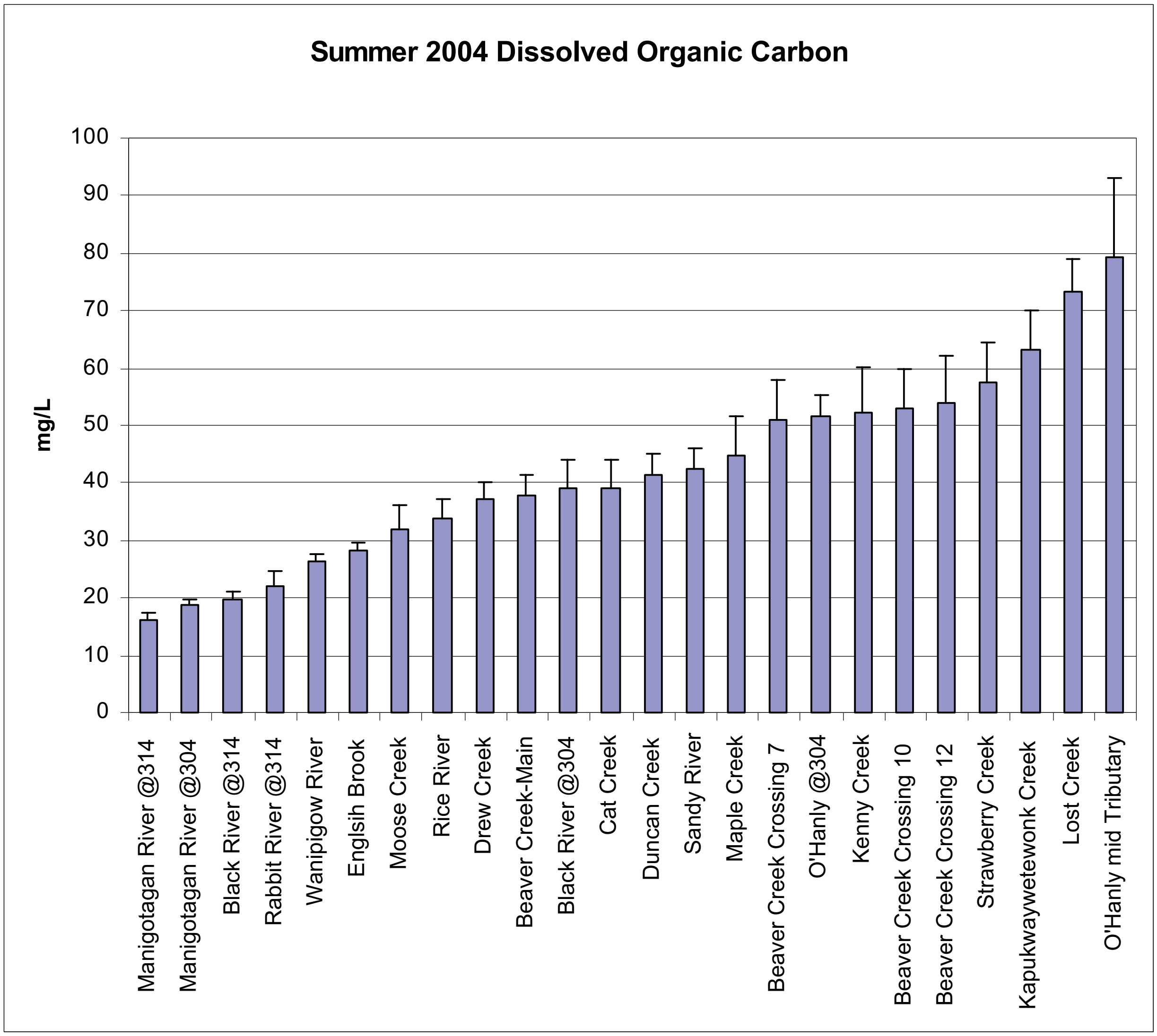


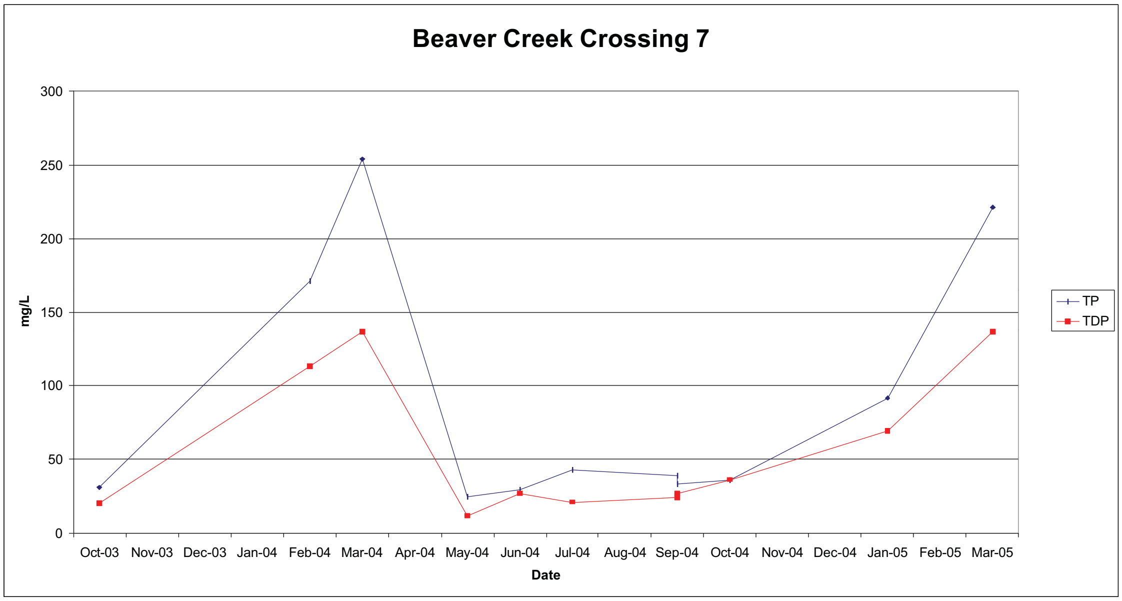
Figure 32. Dissolved Organic Carbon (DOC) concentration in the study streams in the summer of 2004. Vertical bars are 1 standard deviation of the mean

Seasonality of Water Quality in Study Streams

Most of the water quality parameters studied displayed a significant amount of variation throughout the year in each of the streams. There were only a few parameters that did not vary throughout the summer and winter periods. These were cations such as potassium, sodium and magnesium, and pH. Other parameters such as TP, TDP, NO_3, NH_4, TN, DOC, conductivity, turbidity, alkalinity and sulfate (SO_4), were highly variable, and seasonal changes in some appeared related to discharge patterns, while others were not. As discussed previously, concentrations of phosphorus, nitrogen and dissolved oxygen were very different between the summer and winter periods.

Figure 33 demonstrates the seasonal changes in TP and TDP in a small stream (Beaver Creek Crossing 7) and in a large river (Manigotagan River at 304).

TP and TDP display a cyclic pattern in Beaver Creek Crossing 7, with the highest concentrations occurring in the winter, and much lower concentrations in the summer (Figure 33). High concentrations in the winter are likely a result of little or no flow in the stream at this time of the year, coupled with low dissolved oxygen concentrations in the water. Under these conditions, phosphorus diffuses from the sediments into the overlying water. During spring runoff, the creek is diluted with melt water containing much lower phosphorus. Phosphorus levels remain low throughout the summer and fall until the ice forms on the creek again in early winter, after which the concentrations increase again. In contrast to the small streams, TP and TDP concentrations in the Manigotagan River (Figure 33) are generally similar throughout the year, and do not demonstrate the massive increases in TP and TDP in the winter, as observed in the smaller streams.



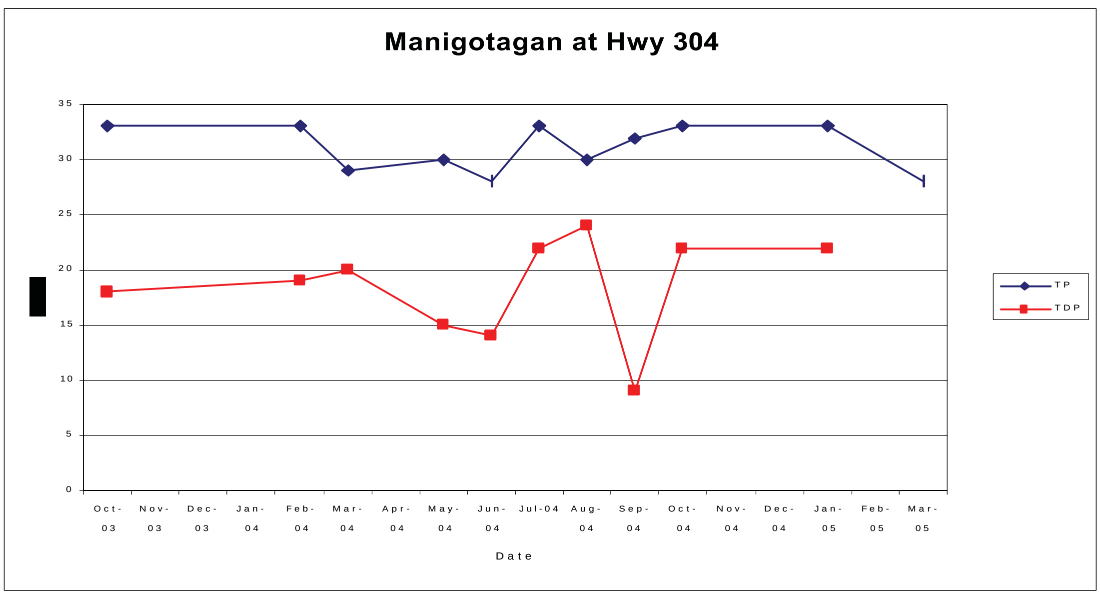


Figure 33. Seasonal changes in TP and TDP in Beaver Creek Crossing 7 (top) and Manigotagan River at Hwy 304 (bottom).

Upstream / Downstream Comparisons of Water Quality

For the majority of the water bodies studied, only one sampling site was chosen per stream. There were two rivers in the study in which two sites was sampled: Black River and Manigotagan River. Upstream sites on both rivers were located in Nopiming Provincial Park, along Highway 314. The downstream sites on both rivers were located near Lake Winnipeg, along Highway 304. For most of the water quality parameters, there were significant differences between the upstream and downstream sites.

Figure 34 shows that in the Black River, concentrations of Ca, TP, DOC and alkalinity and conductivity were higher at the downstream sites, although pH was not. A similar trend was also observed for the Manigotagan River (Figure 34), although the differences between upstream and downstream sites were not as pronounced as those observed in the Black River. The reason(s) for the difference in upstream and downstream water quality can not be easily answered with such a small data set. Differences may be due to differences in soil and forest type and/or disturbance (fire, logging). To better understand the possible cause of these differences, water samples will be collected from multiple sites on the Black, O'Hanly and Manigotagan rivers from the Ontario border to Lake Winnipeg by helicopter in 2005. At the time of this report, the survey has been completed but the data have not been analyzed. This information will be available in a subsequent report.

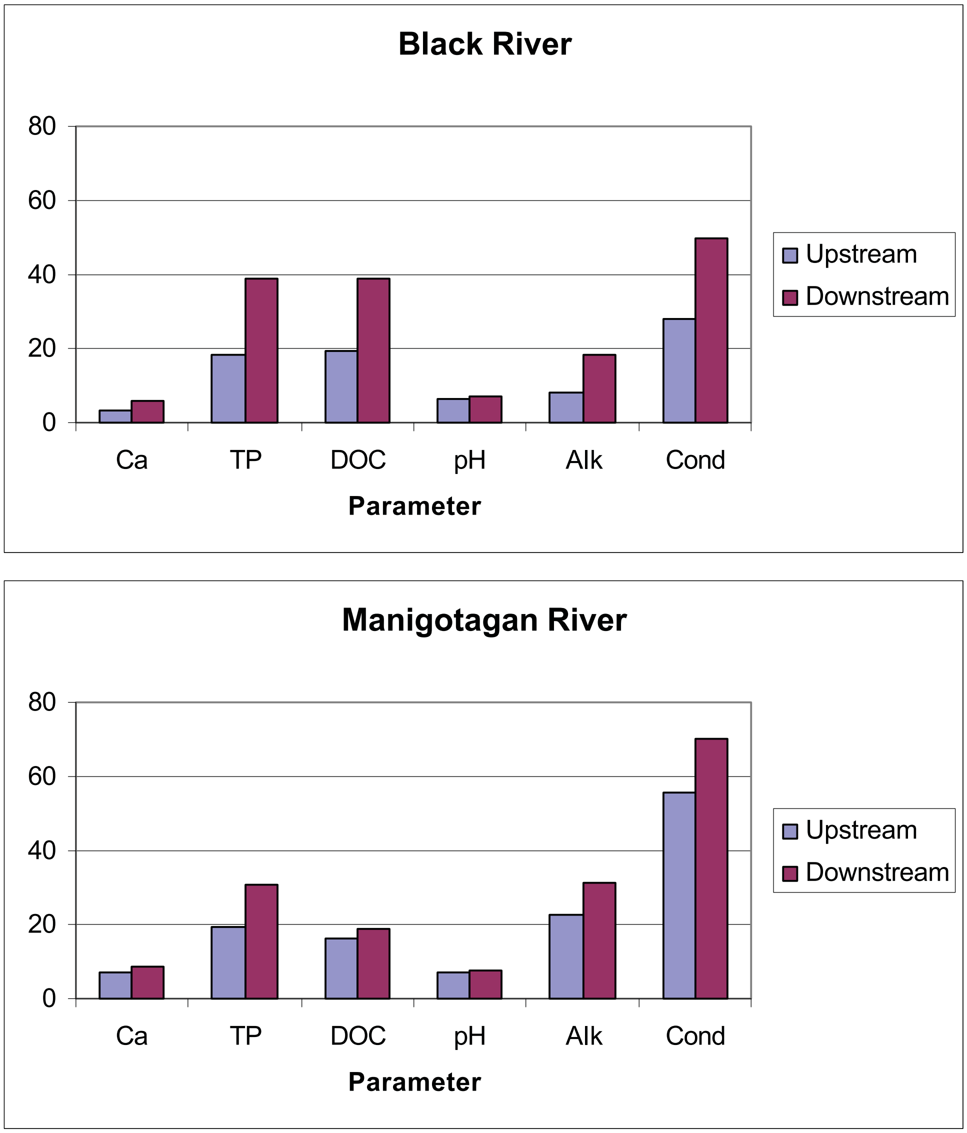


Figure 34. Comparison of upstream and downstream values of various water quality parameters in the Black River (top) and Manigotagan River (bottom). Units for Ca, DOC and alkalinity are mg/L. Units for TP are ug/L. Units for conductivity are uS/cm.

Export Coefficients

Export coefficients are one method of calculating and describing how much dissolved substances (e.g., nutrients) are lost (i.e., exported) from the terrestrial watershed to water bodies such as creeks, streams and rivers. To facilitate comparison of export coefficients between watersheds, the export coefficient values are standardized by watershed size. Export coefficients are reported as kg of dissolved substances exported from watersheds/hectare/year. Most substances are not replenished in soils quickly through natural processes (e.g., such as the weathering of rock) and therefore a high value for an export coefficient could indicate long-term loss of the substance over time from the soils in a watershed. Other substances such as nitrogen can be replenished in soils much more quickly through biological processes such as nitrogen fixation. Agricultural practices such as application of fertilizer are designed to compensate for the long-term loss of nutrients (particularly nitrogen) from soils caused by cropping and erosion. A comparison of the watersheds is provided below for some of the key export coefficients, including Ca, SO_4, TP, TN and DOC. As mentioned previously, these export coefficients need to be interpreted with caution, as most are based on very limited data. Therefore, it is likely more important to view the relative differences in export coefficients between watersheds, rather than the absolute values.

Ca is an important nutrient for terrestrial plants, particularly for tree species. In the boreal shield region of Canada, it is a particularly important terrestrial plant nutrient, as it is in short supply in the thin soils. The loss of Ca to the aquatic environment, through nutrient export processes however, is likely of little consequence to water quality. Ca is not considered an important element in aquatic ecosystems. Figure 35 shows the export coefficient values of Ca for the study watersheds for which export coefficients could be calculated.

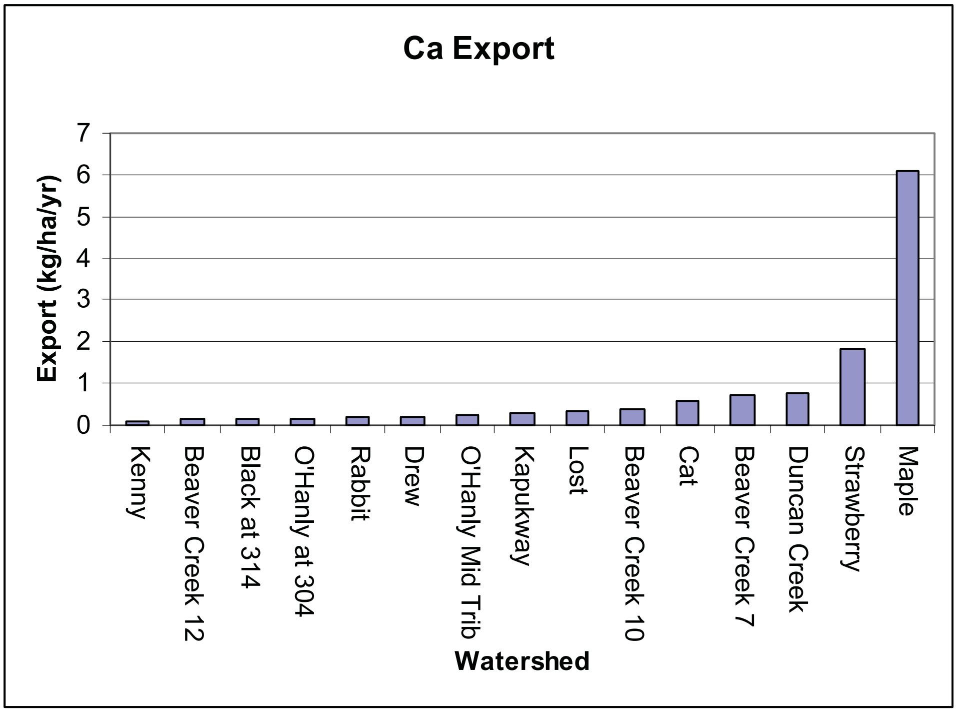


Figure 35. Export coefficients of calcium in the study streams in the summer of 2004.

It is very evident from Figure 35, that Ca export from the two watersheds with agricultural activity was significantly higher than all other streams. Ca export from the Strawberry and Maple creek watersheds was up to 21 times higher than in all of the other streams. This is likely a result of removal of the forest cover and conversion to agricultural land use, as well as due to soil type. Of the non-agricultural watersheds, Duncan Creek had the highest Ca export coefficient. This is likely due to a combination of soil type (100% of the watershed is covered by deep basin soils), forest type (41% of the area is covered by trembling aspen) and disturbance history (the entire watershed was burned twice, in 1989 and again in 1999).

As with Ca, the export coefficient for total phosphorus (TP) was highest in both Maple and Strawberry creeks (Figure 36). TP export was up to 7.4 times higher in the agricultural watersheds that in all other forested study watersheds. The highest TP export coefficient in the forested watersheds was that of Duncan Creek, consistent with the Ca data.

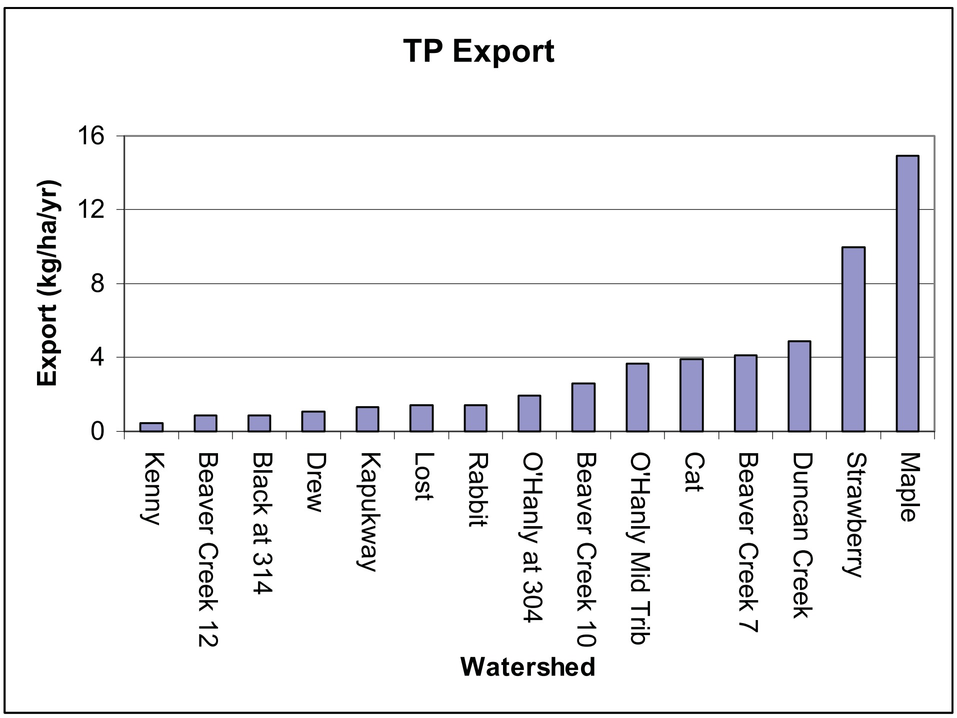


Figure 36. Export coefficients of total phosphorus in the study streams in the summer of 2004.

The export of total nitrogen (TN) from the study watersheds showed a slightly different pattern compared to TP and Ca. For example, while TN export was highest in Maple Creek, and also high in Strawberry Creek, Beaver Creek Crossing 7 also had one of the highest TN export coefficients (Figure 37). This is unexpected, as this watershed is dominated by bedrock (BR)-type soils, has no history of forest harvesting, and has had only minimal fire disturbance (40% of the watershed has burned in the last 30 years) relative to other watersheds. However, beaver activity and the development of beaver floods on this watershed is quite extensive. According to the Forest Resource Inventory (FRI), almost 15% of the watershed area occurs in beaver floods. This is the highest proportion of flooding in any of the study watersheds. As was noted in earlier, back flooding can have a profound influence on nitrogen release from surrounding soils. The relatively high TN export coefficient value for Beaver Creek Crossing 10 may also be due to beaver activity, as this watershed is dominated completely by bedrock soils, has almost no harvesting history (6% of the area in the last 60 years) or fire history (16% of the area in the last 60 years). Finally, the reason for high TN export from Duncan Creek may be due to a combination of productive soils (deep basin soils), significant fire history (the watershed burned twice, completely in the last 18 years) and beaver activity (Duncan Creek has the second highest proportion of beaver flood area of any of the study watersheds).

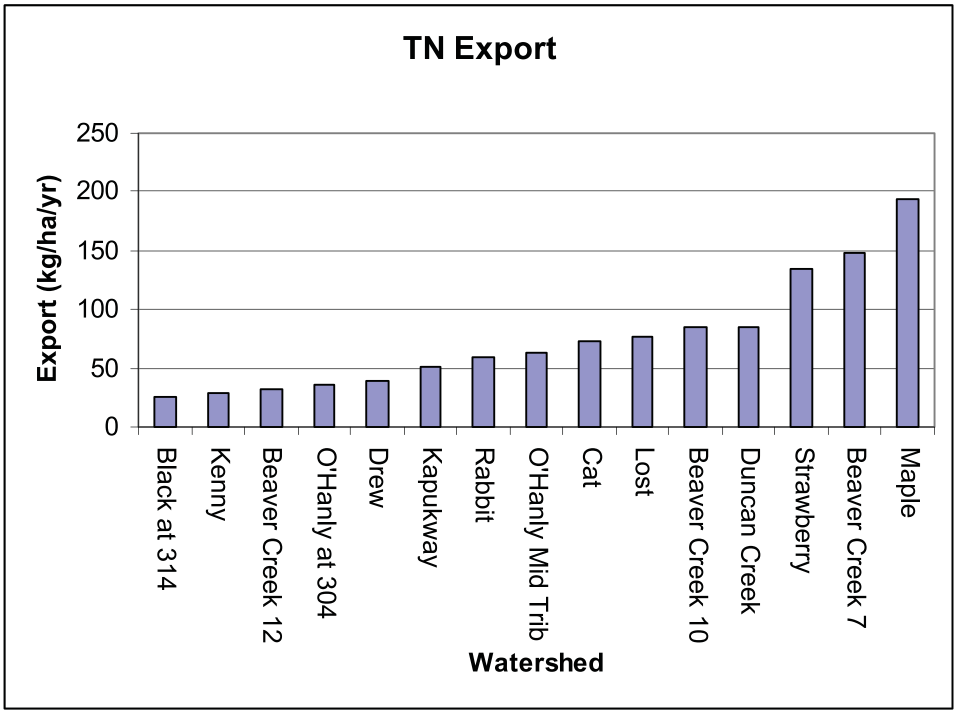


Figure 37. Export coefficients of total nitrogen in the study streams in the summer of 2004.

The regional pattern in export of DOC (color) from the watersheds was different that that of Ca, TP or TN (Figure 38). There also appears to be a complex set of factors that govern DOC movement in watersheds. DOC concentrations, and thus, export is usually high in watersheds with a significant proportion of wetlands (sources of DOC- discussed in more detail later). While this is true for the watersheds of the O'Hanly mid tributary and Kapukwaywetewonk Creek, there are other streams with much higher DOC export coefficients, and yet with much less wetland area (Figure 38). For example, of all the streams, DOC export was highest in Beaver Creek Crossing 7. As mentioned previously with respect to TN, the high DOC export may be due to beaver activity and flooding of terrestrial soils. High DOC export in the Maple and Strawberry Creek watersheds is likely due to loss of forest cover to agricultural land use, while high DOC export in the Lost, Beaver Creek Crossing 10 and Duncan Creek watersheds may be due to soil type, fire and harvesting history, as well as beaver activity.

It appears that nutrient concentrations and export therefore are governed by a complex set of interacting factors, including soils, forest type, disturbance history (fire and logging) and beaver activity. The specific effects of each will be discussed in the next section.

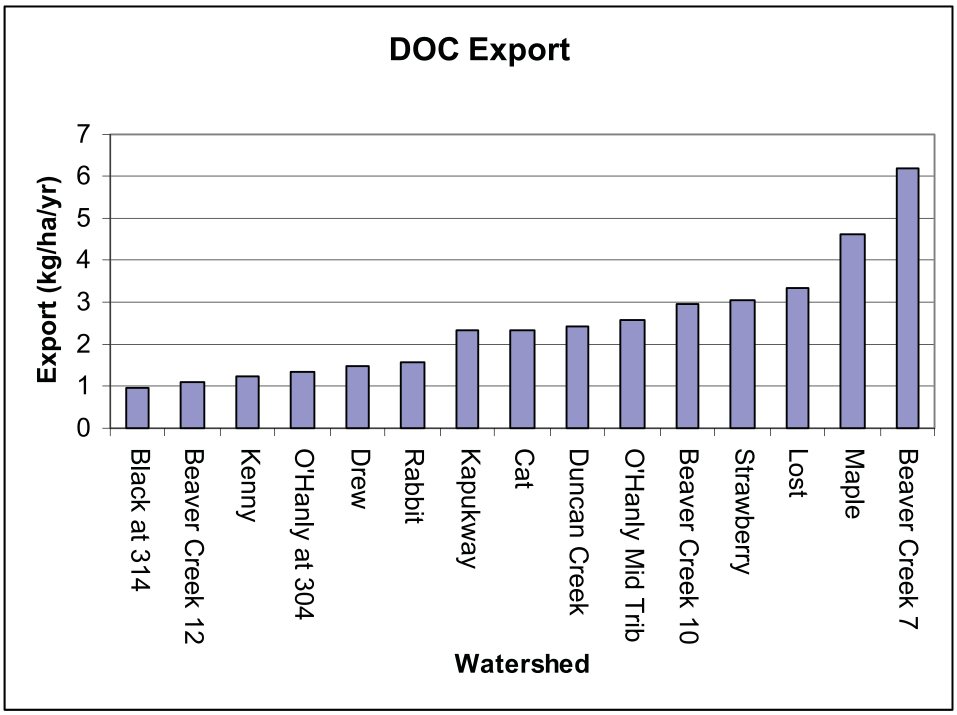


Figure 38. Export coefficients of Dissolved Organic Carbon in the study streams in the summer of 2004.

Influence of Watershed Features on Water Quality

There are three main types of natural watershed features examined in this study that potentially could affect water quality in boreal streams: watershed size, soil type and forest type. At a coarse scale, forest and soil types likely related to one another.

Influence of Watershed Size on Water Quality

Watershed size appeared to have a minor influence on water quality. Rivers and streams in larger watersheds did not necessarily contain higher concentrations of ions (e.g., nutrients, color, etc.) that those in smaller watersheds. Water quality parameters which were correlated to net watershed size (i.e., watershed area excluding the area covered by water) included Ca, NO_3, pH and conductivity, although these correlations were not particularly strong (Table 5).

Table 5. Correlation coefficients and *P* values (in brackets) between watershed size a various water quality parameters.

Parameter	***r (P Value)***
Ca	0.38 (*)
TP	0.19 (NS)
NO_3	0.47 (**)
TN	-0.09 (NS)
DOC	-0.26 (NS)
pH	0.53 (***)
Conductivity	0.35 (*)

P values: NS = not statistically significant (P>0.10), * = P<0.10, ** = P<0.05, *** = P<0.01

Influence of Soil Type of Water Quality

In order to evaluate the influence of soil type on water quality, two separate analyses were conducted. One analysis was more qualitative and looked at the influence of soil type on water quality at a coarse level. The second analysis was more quantitative in nature and involved correlation analysis.

A qualitative examination of the data was performed to look at the influence of broad soil types on stream water quality for 3 key parameters: TP, TN and DOC. To do this, streams with the lowest TP, TN and DOC concentrations, as well as streams with the highest concentrations of TP, TN and DOC were grouped into one of two watershed soil categories: watersheds dominated by bedrock soils, and watersheds dominated by deep basin and organic soils. This is a very coarse way of looking at the influence of soil type on water quality, but is likely more appropriate than a detailed statistical analysis due to the very coarse mapping resolution of soils in the region. Table 6 provides a simple categorization of the watersheds based on two classes of surficial deposit types and their respective impact on TP, TN and DOC. There was a good relationship between soil type at a coarse level of categorization and water quality. Streams located in watersheds that were dominated by bedrock (BR)-type soils consistently had the lowest concentrations of TP, TN and DOC (Table 6). These included the Black River at Hwy 314, Rabbit River and the Manigotagan River at Hwy 314, all located in the eastern part of the Manitoba Model Forest, in an area dominated by bedrock outcrops. At the other end of the spectrum were streams in watersheds that were dominated by richer soils, including deep basin (DB) and deep organic (OD) soils. These streams consistently had the highest concentrations of TP, TN and DOC. These included O'Hanly mid tributary, Duncan Creek, Lost Creek and Kapukwaywetewonk Creek. There were surprisingly few exceptions to these patterns. Only the O'Hanly River at Hwy 304 diverged slightly from this pattern. The O'Hanly River at Hwy 304 had the 4th highest TP and TN concentrations of all of the forested streams (i.e., excluding the two agricultural streams), yet the O'Hanly River watershed is comprised primarily of bedrock soils (82% BR). However, this watershed also has also had 50% of the watershed area logged in the last 60 years. All of the streams with the highest TP, TN and DOC concentrations also have experienced significant levels of watershed harvest, burning or both in the last 60 years. Therefore, water quality appears to be influenced by both soil type and disturbance history.

Table 6. Influence of broad soil types (surficial deposits) in watersheds on water quality in the study streams.

Watershed Class Based on Broad Soil Type (Surficial Deposits)	**Stream Water Quality Characteristics**
Watersheds Dominated by Bedrock (BR) Soils	Low TP (Black at Hwy 314, Rabbit River, Manigotagan at Hwy 314) Low TN (Black at Hwy 314, Rabbit River, Manigotagan at Hwy 314) Low DOC (Manigotagan at Hwy 314 + 304, Black River at Hwy 314, Rabbit River)
Watersheds Dominated by Deep Organic (OD) and/or Deep Basin (BD) Soils	High TP (O'Hanly mid Tributary, Duncan Creek) High TN (O'Hanly mid Tributary, Lost Creek, Duncan Creek, Kapukwaywetewonk Creek) High DOC (O'Hanly mid Tributary, Lost Creek, Kapukwaywetewonk Creek)

The influence of soil type on water quality was also evaluated in a more quantitative manner by use of correlation analyses. The data were initially structured into broad soil categories (called surficial deposits), including bedrock (BR), deep basin (DB), organic deposits (OD), glacial fluvial deposits (GD) and glacial till (T3). A second correlation analysis was then conducted by breaking the broad soil categories (BR, DB, OD, GD, T3) into finer classifications which added the soil landscape component to the surficial deposits category. For example, the BR category was broken down into BR/D, BR/F, BR/R2 and BR/Y23, the OD category into OD/D, OD/F, OD/R2 and OD/Y23, and so forth. However, when this was done, it was evident that several of the finer scale soil categories were rare in the study watersheds. For example, BR/D occurred in only 2 watersheds, DB/D in only 1 watershed, OD/D in only 1 watershed, T3/R2 in only 1 watershed, etc. Therefore, this made statistical analysis of the influence of fine-scale soil categories questionable. As a result, only the broad surficial deposit categories (e.g., BR, OD, DB) were used in the correlation analysis. This is likely appropriate, as the mapping of enduring features was done at a very coarse scale of resolution (1:1,000,000) anyway.

The relationship between broad surficial soil deposits and water quality is presented in Table 7. While Table 6 demonstrates that there is a very strong relationship between soil type and water quality at the two ends of the water quality spectrum in the study streams (i.e., in streams with the highest and lowest TP, TN and DOC concentrations), the correlation analysis using all of the data is less equivocal (Table 7). For many water quality parameters, there is no statistically significant relationship between the proportion of the various soil types in the watersheds and the concentration of key water quality parameters (as identified in Table 7 by the NS – Not Significant designation). The correlation analysis does indicate that there is a relationship between the proportion of OD soils in watersheds and the concentration of SO_4 and DOC in the streams. As the proportion of OD soils increases in the watersheds, streams have higher concentrations of SO_4 and DOC. This is not surprising, as OD soils such as those found in peatlands (black spruce and tamarack bogs and fens) are known to be sources of SO_4 and DOC (Devito and Hill, 1997). There is also a strong negative relationship between the proportion of BR soils in watersheds and the concentration of Ca, TDP, TN, DOC alkalinity and conductivity. Again, this is not surprising as watersheds with a high BR soil proportion tend to have thin soils with little organic or mineral (e.g., clay) content and thus, lower calcium, phosphorus, nitrogen or alkalinity and conductivity. In contrast, there were significant correlations between the proportion of DB soils in watersheds and Ca, pH, alkalinity and conductivity, reflecting the more fertile nature of these soils. Data from a study of 39 lakes in the boreal shield of Quebec also suggest that local variations in bedrock geology and till composition are important in influencing lake water concentrations of Ca, SO_4 and alkalinity (Carignan et al., 2000). While not statistically significant, an increase in the proportion of either OD or DB soils in the watersheds was related to higher nitrogen and phosphorus concentrations in the streams, particularly for TDP and TN. When the OD and DB soil categories were combined, there was even a stronger relationship between this OD/DB category and TDP (r = 0.46) and TN (r = 0.52) than either soil category on their own. From the correlation analysis it therefore appears that streams with higher phosphorus, nitrogen and DOC are found in watersheds with higher proportions of OD and DB soils (and thus, lower proportions of BR soils). In contrast, streams with lower phosphorus, nitrogen and DOC are found in watersheds with lower amounts of OD and DB soils (and thus, more BR soils).

Table 7. Correlation coefficients (*r*) and *P* values (in brackets) between the percentage of each soil type in the watersheds and various water quality parameters.

Water Quality Parameter	**Soil Type**		
	BR	OD	DB
Ca	-0.45 (**)	-0.06 (NS)	0.71 (***)
SO_4	-0.27 (NS)	0.43 (*)	-0.09 (NS)
TP	-0.29 (NS)	0.09 (NS)	0.31 (NS)
TDP	-0.46 (**)	0.29 (NS)	0.31 (NS)
NO_3	0.22 (NS)	-0.29 (NS)	0.01 (NS)
NH_4	-0.03 (NS)	0.03 (NS)	0.01 (NS)
TN	-0.51 (**)	0.35 (NS)	0.33 (NS)
DOC	-0.54 (***)	0.54 (***)	0.15 (NS)
pH	0.00 (NS)	-0.36 (NS)	0.40 (*)
Alkalinity	-0.41 (*)	-0.16 (NS)	0.78 (***)
Conductivity	-0.49 (**)	-0.12 (NS)	0.83 (***)

P values: NS = not statistically significant (P>0.10), * = P<0.10, ** = P<0.05, *** = P<0.01

Influence of Forest Type on Water Quality

The influence of forest type in the watersheds on water quality in the streams can be evaluated at different levels of forest species composition complexity. In this study we have stratified the forest types by three different methods. Firstly, the forest polygons in the Forest Resource Inventory (FRI) were classified at a coarse scale as either hardwood (H), hardwood-leading mixedwood (N), softwood-leading mixedwood (M), softwood (S) or Non-productive (primarily treed muskeg, treed rock, willow/alder). This is known as the FRI **Land Type**. A second classification scheme involved using the FRI **Vegetation Type (V-Type) Groups**. These are groupings of the more detailed V-Types, and include groups such as Lowland Black Spruce (comprised of V30, V31, V32, and V33 V-Types), Jack Pine Conifer (V24, V25, V26) and Black Spruce Conifer (V27, V28, V29), etc. Finally, a third and more detailed classification consisting of the FRI **Stand Type** was used. This classification scheme predominantly classifies the forest polygons by dominant species (e.g., black spruce, trembling aspen, tamarack larch, balsam fir, etc.), although there are also more generic categories as well (e.g., other hardwood, willow/alder, bare rock, meadow, marsh, treed muskeg, etc.) included in this classification scheme. As will be highlighted below, no matter how the forest cover data was categorized, the same general trends emerged with respect to the influence of forest type on water quality in the streams.

FRI Land Type

Table 8 shows the correlation coefficients between the proportion of FRI Land Types in the watersheds and the various water quality parameters in the streams. Higher concentrations of Ca, TP, TDP, and higher pH, alkalinity and conductivity in the streams was related to higher proportions of Hardwood (H) and hardwood-leading mixedwood (N) forests in the watersheds. This trend is opposite for softwood (S) forests. It appears that the productive, upland soils associated with hardwood forests may be sources of more ions (Ca, nutrients such as phosphorus) and provide streams with higher pH, alkalinity and conductivity than softwood (S) forests. In addition, the higher the proportion of S Land Types in the watersheds, the lower the TN concentrations in the stream water. The only Land Type significantly (statistically) associated positively with TN was softwood-leading mixedwoods (Land Type M), although higher TN values were also associated with higher proportions of hardwood (H) and hardwood-leading mixedwood (N) forest stands, but these relationships were not statistically significant. Higher SO_4 and DOC concentrations in the streams were correlated to higher proportions of non-productive forest area in the watersheds. This is consistent with the soil data, where higher SO_4 and DOC was also correlated with higher proportions of OD type soils in the watersheds. There were also weak (and not statistically significant) positive correlations between the area of non-productive forest and phosphorus (as TDP) and nitrogen (as NH_4 and TN) concentrations in the streams. Based on the analysis in Table 8, it appears that hardwood forest stands or hardwood-containing mixedwood stands contribute more ions such as Ca and phosphorus to the streams, while non-productive stands contribute SO_4, DOC and to a certain extent, phosphorus. With the exception of softwood (S) stands, all other stands contribute varyingly to TN inputs to streams.

Table 8. Correlation coefficients and *P* values (in brackets) between the percentage of each forest Land Type (H=hardwood, N=hardwood-leading mixedwood, M=softwood-leading mixedwood, S=softwood, NP=non-productive) in the watersheds and various water quality parameters

Water Quality Parameter	**FRI Land Type**				
	H	N	M	S	NP
Ca	0.79 (***)	0.54 (***)	0.16 (NS)	-0.48 (**)	-0.23 (NS)
SO_4	-0.21 (NS)	0.14 (NS)	0.33 (NS)	-0.40 (*)	0.49 (**)
TP	0.48 (**)	0.25 (NS)	-0.02 (NS)	-0.46 (**)	0.20 (NS)
TDP	0.44 (*)	0.28 (NS)	0.09 (NS)	-0.56 (***)	0.34 (NS)
NO_3	0.02 (NS)	-0.06 (NS)	-0.13 (NS)	0.15 (NS)	-0.10 (NS)
NH_4	0.03 (NS)	0.02 (NS)	0.02 (NS)	-0.23 (NS)	0.24 (NS)
TN	0.19 (NS)	0.27 (NS)	0.42 (*)	-0.49 (**)	0.27 (NS)
DOC	-0.06 (NS)	0.29 (NS)	0.53 (***)	-0.54 (***)	0.45 (*)
pH	0.57 (***)	0.28 (NS)	-0.17 (NS)	-0.09 (NS)	-0.38 (*)
Alkalinity	0.89 (***)	0.46 (**)	0.10 (NS)	-0.37 (*)	-0.38 (*)
Conductivity	0.91 (***)	0.49 (**)	0.17 (NS)	-0.43 (*)	-0.34 (NS)

P values: NS = not statistically significant ($P>0.10$), * = $P<0.10$, ** = $P<0.05$, *** = $P<0.01$

FRI V-Type Group

As mentioned previously, another way to classify forest cover in the FRI is by creating vegetation types (V-Types). However, there are 33 V-Types in the classification system, and this is likely too large and detailed to be of use in this analysis. Therefore, we chose to use V-Type Groups, which group V-Types that are most similar. There are 11 V-Type Groups identified in the FRI, but only 5 that are present in the watersheds to any great degree. Table 9 shows the correlations between the proportion of V-Type Groups in the watersheds and water quality in the streams.

Table 9. Correlation coefficients and *P* values (in brackets) between the percentage of each V-Type Group in the watersheds and various water quality parameters.

Water Quality Parameter	**FRI V-Type Group**				
	Cedar/Tamarack Mixedwood	Lowland Black Spruce	Aspen Hardwood + Mixedwood	White Spruce/Balsam Fir	Jack Pine Conifer
Ca	0.17 (NS)	0.34 (NS)	0.79 (***)	0.09 (NS)	-0.52 (***)
SO*4*	0.42 (*)	0.06 (NS)	-0.08 (NS)	0.32 (NS)	-0.58 (***)
TP	-0.01 (NS)	0.48 (**)	0.45 (**)	-0.07 (NS)	-0.46 (**)
TDP	0.07 (NS)	0.52 (***)	0.43 (*)	0.03 (NS)	-0.54 (***)
NO_3	-0.17 (NS)	-0.01 (NS)	-0.01 (NS)	-0.19 (NS)	0.19 (NS)
NH_4	0.14 (NS)	0.16 (NS)	0.03 (NS)	0.04 (NS)	-0.34 (NS)
TN	0.50 (***)	0.27 (NS)	0.26 (NS)	0.38 (*)	-0.70 (***)
DOC	0.60 (***)	0.12 (NS)	0.10 (NS)	0.51 (***)	-0.75 (***)
pH	-0.23 (NS)	0.16 (NS)	0.52 (***)	-0.18 (NS)	0.01 (NS)
Alkalinity	0.07 (NS)	0.36 (*)	0.83 (***)	0.03 (NS)	-0.38 (*)
Conductivity	0.14 (NS)	0.41 (*)	0.85 (***)	0.09 (NS)	-0.46 (**)

P values: NS = not statistically significant (P>0.10), * = P<0.10, ** = P<0.05, *** = P<0.01

As with the Hardwood (H) FRI Land Type, the aspen hardwood + mixedwood category in the FRI V-Type Group had that strongest correlation with Ca, TP and TDP (Table 9), and was also strongly correlated with pH, alkalinity and conductivity. There were negative correlations between the proportion of Jack Pine V-Type Group in the watersheds and Ca, SO_4, TP, TDP, TN, DOC, alkalinity and conductivity. This is an indication that Jack Pine forests do not contribute significantly to the export of ions from these watersheds. This is also consistent with the soil data in which the bedrock (BR) type soils were negatively correlated with these water quality parameters also. It is interesting to note as well that the soil data is consistent with the relationships existing between the Cedar/Tamarack V-Type Group (which includes tamarack bogs containing Labrador tea) with SO_4 and DOC. Higher concentrations of SO_4 and DOC were correlated with higher proportions of the organic deposit (OD) soil type, which matches well with the Cedar/Tamarack V-Type Group. Finally, there were strong relationships between the proportion of Lowland Black Spruce V-Type Group and both TP and TDP. It appears that this V-Type Group may be a source of phosphorus to the streams.

FRI Stand Type

Lastly, the relationship between forest type and water quality was evaluated by using the FRI Stand Type attributes. The Stand Type classification categorizes stands according to dominant tree species (e.g., trembling aspen, tamarack, white spruce, black spruce, jack pine, etc.) in productive areas, or other characteristics (e.g., beaver floods, bare rock, marsh, meadow, treed muskeg, treed rock, willow/alder) in non-productive areas. Only 5 of the pertinent Stand Types displaying significant correlations are shown below.

Table 10. Correlation coefficients and *P* values (in brackets) between the percentage of each FRI Stand Type in the watersheds and various water quality parameters.

Water Quality Parameter	**FRI Stand Type Group**				
	Tamarack/Larch	Black Spruce	Treed Muskeg	Trembling Aspen	Jack Pine
Ca	0.17 (NS)	0.11 (NS)	-0.23 (NS)	0.79 (***)	-0.43 (**)
SO_4	0.42 (*)	0.37 (*)	0.54 (***)	-0.08 (NS)	-0.57 (***)
TP	-0.02 (NS)	0.59 (***)	0.18 (NS)	0.45 (**)	-0.47 (**)
TDP	0.06 (NS)	0.62 (***)	0.34 (*)	0.43 (**)	-0.58 (***)
NO_3	-0.18 (NS)	-0.03 (NS)	-0.18 (NS)	-0.01 (NS)	0.21 (NS)
NH_4	0.14 (NS)	0.51 (***)	0.19 (NS)	0.03 (NS)	-0.36 (*)
TN	0.51 (***)	0.33 (NS)	0.33 (NS)	0.26 (NS)	-0.65 (***)
DOC	0.60 (***)	0.29 (NS)	0.52 (***)	0.10 (NS)	-0.72 (***)
pH	-0.23 (NS)	-0.02 (NS)	0.44 (**)	0.52 (***)	0.04 (NS)
Alkalinity	0.07 (NS)	0.08 (NS)	0.39 (*)	0.83 (***)	-0.31 (NS)
Conductivity	0.14 (NS)	0.13 (NS)	0.34 (*)	0.85 (***)	-0.39 (*)

P values: NS = not statistically significant (P>0.10), * = P<0.10, ** = P<0.05, *** = P<0.01

There was a good correspondence between Tables 9 and 10 with respect to the correlations between forest types and water quality parameters. Higher SO_4, TN and DOC in the streams were correlated with higher proportions of the Tamarack/Larch and Treed Muskeg Stand Types in the watersheds (Table 10). This was also the case for the Tamarack V-Type Group in Table 9.

Higher TP and TDP were associated with higher proportions of Black Spruce Stand Types in the watersheds (Table 10), similar to that seen with the Lowland Black Spruce V-Type Group in Table 9. Prepas et al. (2001) found that lakes situated in bog/muskeg-dominated watersheds in Alberta had high phosphorus concentrations. Higher Ca, TP, TDP, pH, alkalinity and conductivity were associated with higher proportions of the Trembling Aspen Stand Type (Table 10), consistent with the trends observed for the Aspen Hardwood V-Type Group (Table 9) and the Hardwood (H) component in Table 8.

In summary, forest type in the watersheds was significantly related to various water quality parameters in the streams. In other words, the occurrence and proportion of different forest types in the watersheds had a significant influence of water quality in the streams. Hardwood forests, especially those dominated by trembling aspen, resulted in stream water quality with higher Ca, TP and TDP concentrations, as well as higher pH, alkalinity and conductivity. Watersheds dominated by Jack Pine resulted in streams with less Ca, TP, TDP, TN and DOC concentrations, as well as lower alkalinity and conductivity. Watersheds dominated by lowland tree species such as black spruce and tamarack and muskeg resulted in stream water quality with higher SO_4, TP, TDP, TN and DOC concentrations, and generally lower pH, alkalinity and conductivity. In a general sense as well, the forest cover type data was consistent with the results obtained for broad soil types, but were much more revealing of the relationships between watershed features and water quality.

Influence of Watershed Disturbance on Water Quality

The influence of watershed disturbance on water quality was examined by evaluating disturbance agents such as forest tent caterpillar and spruce budworm outbreaks, road density, logging history in the watersheds, as well as fire history and beaver activity. The results of these analyses are described below.

Insect Outbreaks and Water Quality

Spruce budworm and forest tent caterpillar outbreaks (all severity classes combined) since 1995, the first year where digital data was available, were present in 55% of the forested watersheds. However, the aerial extent of damage was quite low. For spruce budworm, the largest area of damage occurred in Kapukwaywetewonk Creek (23% of the watershed) but the mean and median of the area of damage among all the forested watersheds were only 4.4% and 0% respectively. Similarly, for forest tent caterpillar the largest area of damage was quite low (24% in Manigotagan at Hwy 304), with a mean and median of only 2.9% and 0% among all forested watersheds. Because 45% of the watersheds did not contain any spruce budworm or forest tent caterpillar outbreaks, and because of the low aerial extent of damage where they did occur, it is not possible to examine the effect of these insect outbreaks on water quality in the streams. It is unlikely that such small outbreaks would result in any significant changes to water quality.

Forest Management and Water Quality

Two types of forest management activities were evaluated for their effects on water quality in the streams. Firstly, access development (roads, trails) and power lines (hydro utility corridors-not necessarily related to forestry activities) was evaluated by calculating linear feature density in each watershed and relating this to water quality observed in the streams. Secondly, the amount (area) of forest harvested was calculated at 4 times periods (cumulatively within the last 5 years, last 18 years, 30 years and within the last 60 years) within each watershed and then related to the water quality in the streams.

Linear Features and Water Quality

The total distance of all linear features in the watersheds varied from 3.2 km in Beaver Creek Crossing 7 to 1087 km in the Wanipigow watershed. When watershed size was taken into consideration, linear feature density in the watersheds ranged from 0.11 km/km^2 in Beaver Creek Crossing 12 to 3.14 km/km^2 in Lost Creek (Figure 13). The highest road densities occurred in the smallest watersheds. There were significant correlations between total linear feature density and SO_4 concentration in the streams (r = 0.62), TDP (r = 0.43), TN (r = 0.60) and DOC (r = 0.74), suggesting that higher stream concentrations of these water quality parameters are associated with higher linear feature densities. However, when one examines the linear features in the watersheds in more detail, it is clear that a high proportion of access development in many watersheds is winter (Class 4) roads rather than all weather roads or power lines. This is particularly true for small watersheds. Figure 39 shows the length of winter roads in the watersheds expressed as a percentage of the total length of access development features.

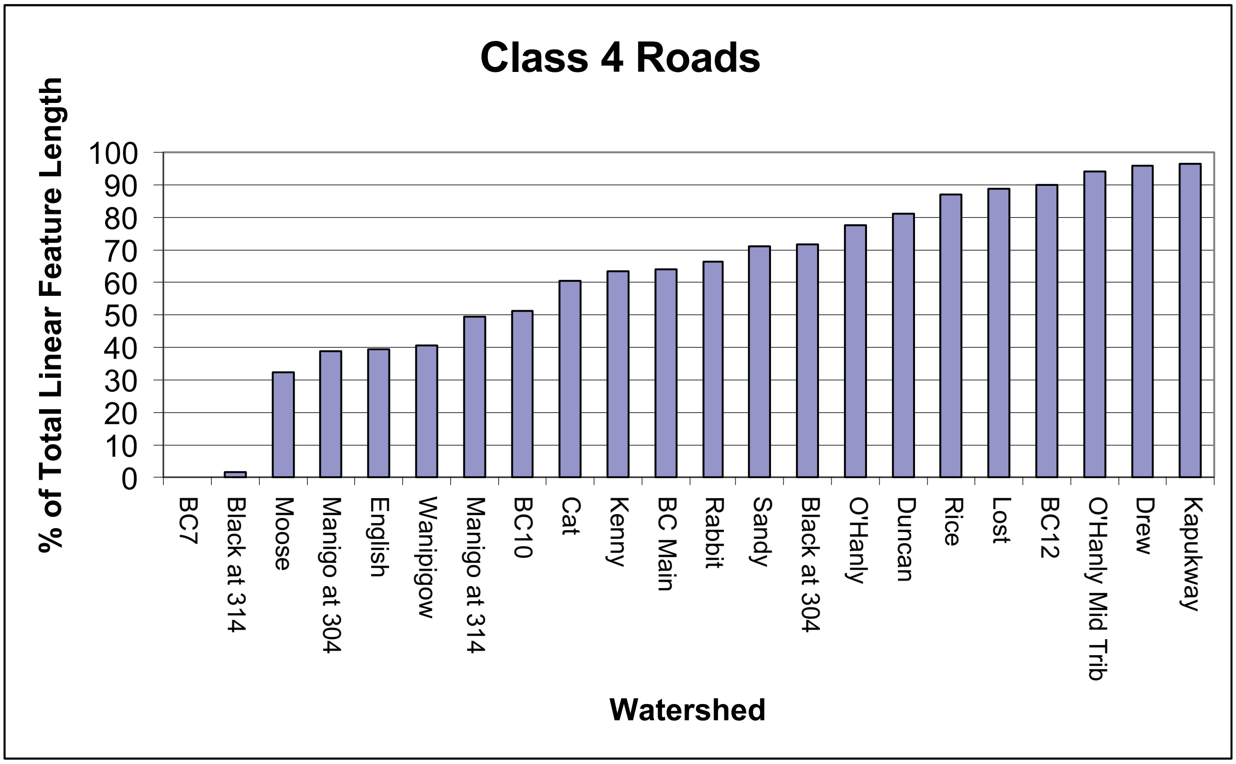


Figure 39. Winter road development in the watersheds, expressed as a percentage of total linear feature length.

When the relationships between water quality and linear feature density are examined separately for winter roads and all-weather roads, a much different pattern emerges. In contrast to the significant relationships between total linear feature density and water quality, there are no significant relationships between all-weather road density and SO_4, TDP, TN or DOC, nor with most other water quality parameters, with two exceptions. There were significant relationships (r = 0.44 and 0.52, respectively) between the density of all-weather roads and stream turbidity and total suspended solids (TSS). This should not be surprising, as most published studies on the effects of forestry activities on water quality have found that road building and the installation of stream crossings can have significant impacts on soil erosion and suspended sediment inputs

(Jablonski, 1986; Nipp, 1991; Rothwell, 1997). However, Martin et al. (2000) indicate that these impacts can be greatly decreased or eliminated by the use of best management practices. In a study of erosion and suspended sediments in harvest areas near Happy Lake, in the Manitoba Model Forest area, Schneider-Vieria (1997) found that a stream crossing constructed with extensive infill and steep slopes caused significant sediment input and sedimentation whereas another crossing with less fill and a much more gentle side slope resulted in no detectable erosion and sedimentation. In addition, harvesting and site scarification did not appear to result in any detectable erosion in the Happy Lake area. It should be pointed out that while our study did find a relationship between all-weather road density and stream turbidity and TSS, the levels of turbidity and TSS were quite low (generally less than 10 NTU or mg/L, respectively), and would not be expected to have a negative impact on water quality nor on fish habitat.

However, in our study there were significant relationships between winter road density and SO_4 ($r = 0.62$, $P = 0.004$), TDP ($r = 0.45$, $P = 0.05$), TN ($r = 0.62$, $P = 0.004$) and DOC ($r = 0.76$, $P = 0.0001$). The data suggest that the relationship between linear feature density and water quality is related mainly to winter roads. It is important to note however, that all of these particular water quality variables are also highly correlated to lowland forest types, including black spruce, tamarack and muskeg. These forest types are also the ones where the majority of winter roads are constructed. From our data, it is not possible to separate the relative contributions of forest type and winter roads to water quality, and thus, properly assess the true impacts of winter roads on water quality. There were no relationships between winter road density and turbidity or TSS, indicating that these types of temporary roads do not result in increased erosion.

Effects of Disturbance Type on Water Quality

There is a growing literature focused on the impacts of forest harvesting on water quality in lakes, rivers and streams in North America. There also appears to be as many potential outcomes (results) as there are studies. For example, research in North America has shown that forest harvesting can cause increases, decreases or no change in major water quality parameters such as phosphorus, nitrogen, suspended sediments, water temperature , algae, benthic invertebrates, and fish. This should not be surprising as these studies span a wide gradient in climate (from maritime climate in the Pacific Northwest and the northeastern United States, to subtropical climate in Georgia, to continental climate in the Canadian boreal forest), topography (from mountainous regions of British Columbia to the eastern slopes of the Canadian Rocky Mountains, to the comparatively flat boreal forest), hydrological regime (snow melt versus storm runoff dominated hydrology), forest types (cedar, hemlock and sitka spruce forests, lodge pole pine, black spruce, jack pine and Ponderosa pine), soil types (from phosphorus rich luvisols in Alberta to relatively nutrient poor soils of the Canadian shield), and forest management techniques such as forest harvesting method (clear cut, selective cut), the proportion of watershed area harvested and scarification methods, to name but a few of the complicating factors. This makes generalizations about the impacts of forest harvesting and the ability to extrapolate results from one study to another more difficult. Despite this, there have been a few studies conducted in the boreal forest which examines the impact of forest harvest on water quality.

A second approach to examining the effects of watershed disturbance on stream water quality was undertaken by classifying the study watersheds into one of five disturbance categories and comparing the average water quality within each of the categories. The categories were:

- Reference watersheds: less than 30% of the watershed area harvested within the last 60 years, less than 30% of the watershed area burned within the last 60 years (3 streams)
- Harvested watersheds: more than 30% of the watershed area harvested in the last 60 years, but less than 30% of the watershed area burned in the last 60 years (3 streams)
- Burned watersheds: more than 30% of the watershed area burned in the last 60 years, but less than 30% of the watershed area harvested in the last 60 years (9 streams)
- Harvest + Fire watersheds: more than 30% of the watershed area harvested in the last 60 years, and more than 30% of the watershed area burned in the last 60 years (4 streams)
- Agricultural watersheds: Maple Creek and Strawberry Creek watersheds (2 streams)

The value of 30% of watershed area was chosen as a break point for defining the lower limits for what is considered a harvesting- or fire-affected watershed and the upper limit for reference watersheds for two reasons. Firstly, Tembec has chosen the 30% limit as a planning tool under their Criteria and Indicators of Sustainable Forest Management approach to managing and reporting on forest management activities in FML 01. Our selection of 30% therefore, allows us to investigate whether differences in water quality can be detected in watersheds with less than and greater than 30% disturbance. Secondly, while there is great variability reported on the impacts of harvesting on water quality, there is some evidence in published studies in the literature that water quality impacts may not be significant if harvesting occurs on less than 30%

of a watershed area. This value should not be taken as an absolute, but merely reflects an approximate lower boundary for observed impacts. This is however, also consistent with predictions that low levels of forest harvesting (below 35%) have no detectable or a minor effect (less than 20-25% increase) on water yield based on modeling using WRENSS (Rothwell, 1997), and ForHyM models (Meng et al., 2002) for boreal forests in Saskatchewan and New Brunswick, and in field studies elsewhere in Canada (Swanson and Hillman, 1977; Plamondon and Ouellet, 1980). In their review of the literature, Bosch and Hewlett (1982) indicate that most paired basin studies can not demonstrate an increase in water yield when harvesting is less than 20% of the watershed area. Rothwell (1997) goes further to suggest that harvest levels could be based on using the ratio of hydrologic recovery age to rotation age as a general guideline. For the Pasquia-Porcupine Forest Management Area in Saskatchewan, this would represent a limit of 25-30% of the watershed areas. Therefore, there is some scientific justification for the 30% harvest value.

Figures 40 through 44 show the relationship between watershed disturbance category and stream concentrations of Ca, TP, TN, DOC and conductivity, as well as export coefficients for these parameters (where appropriate).

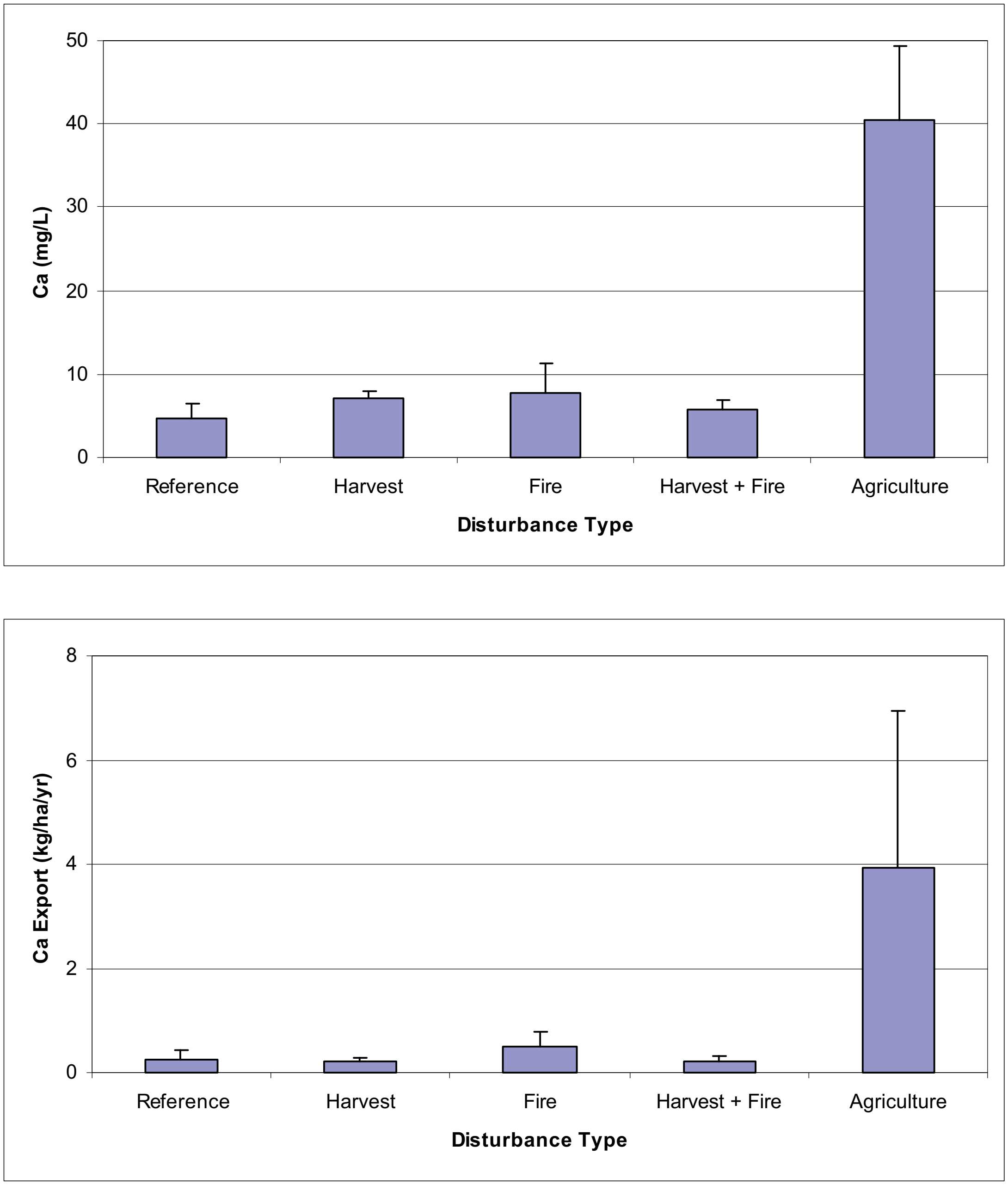


Figure 40. Effects of disturbance type on Ca concentration (top) and Ca export (bottom) in the study watersheds. Vertical bars are one standard deviation of the mean.

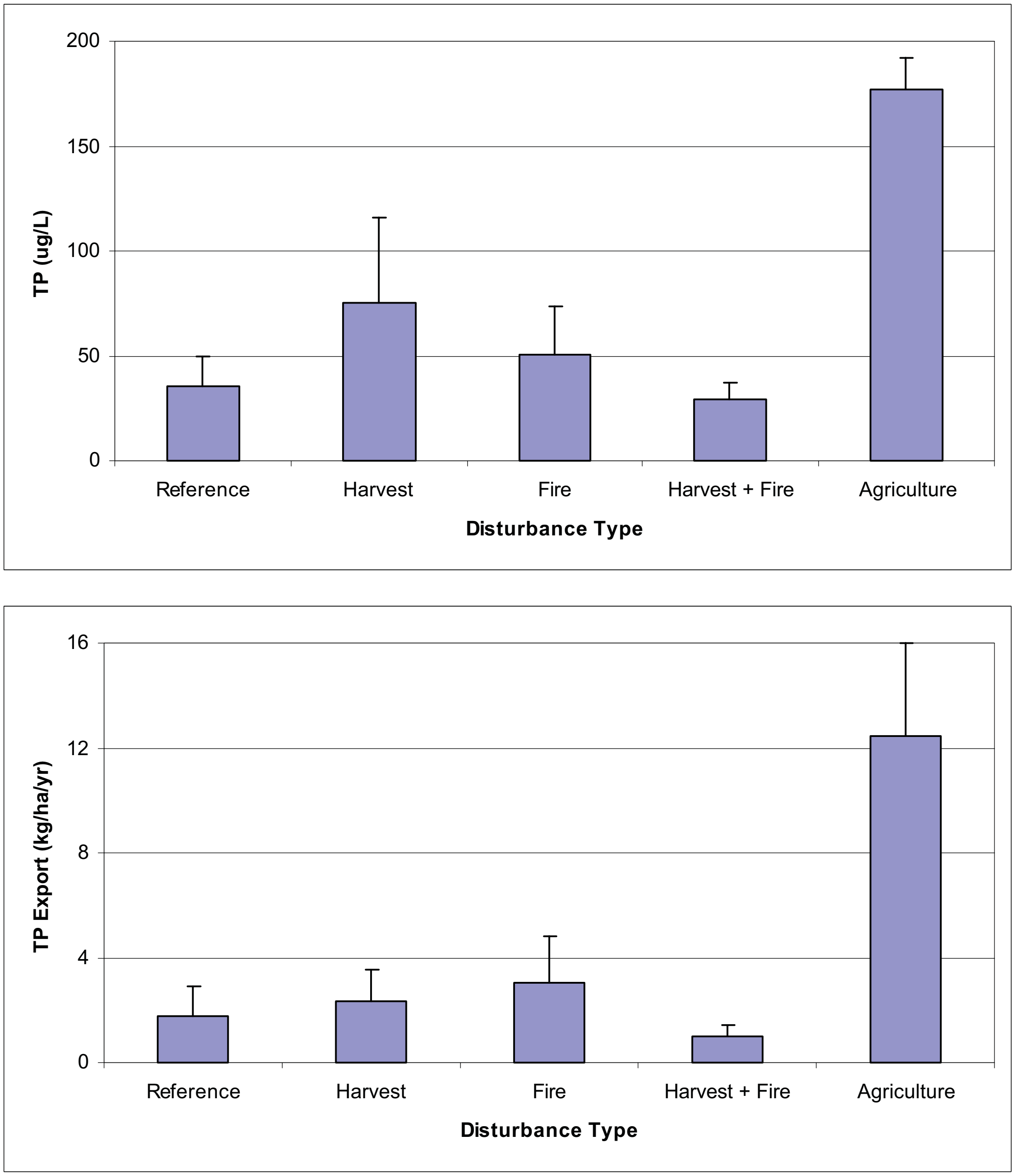


Figure 41. Effects of disturbance type on TP concentration (top) and TP export (bottom) in the study watersheds. Vertical bars are one standard deviation of the mean.

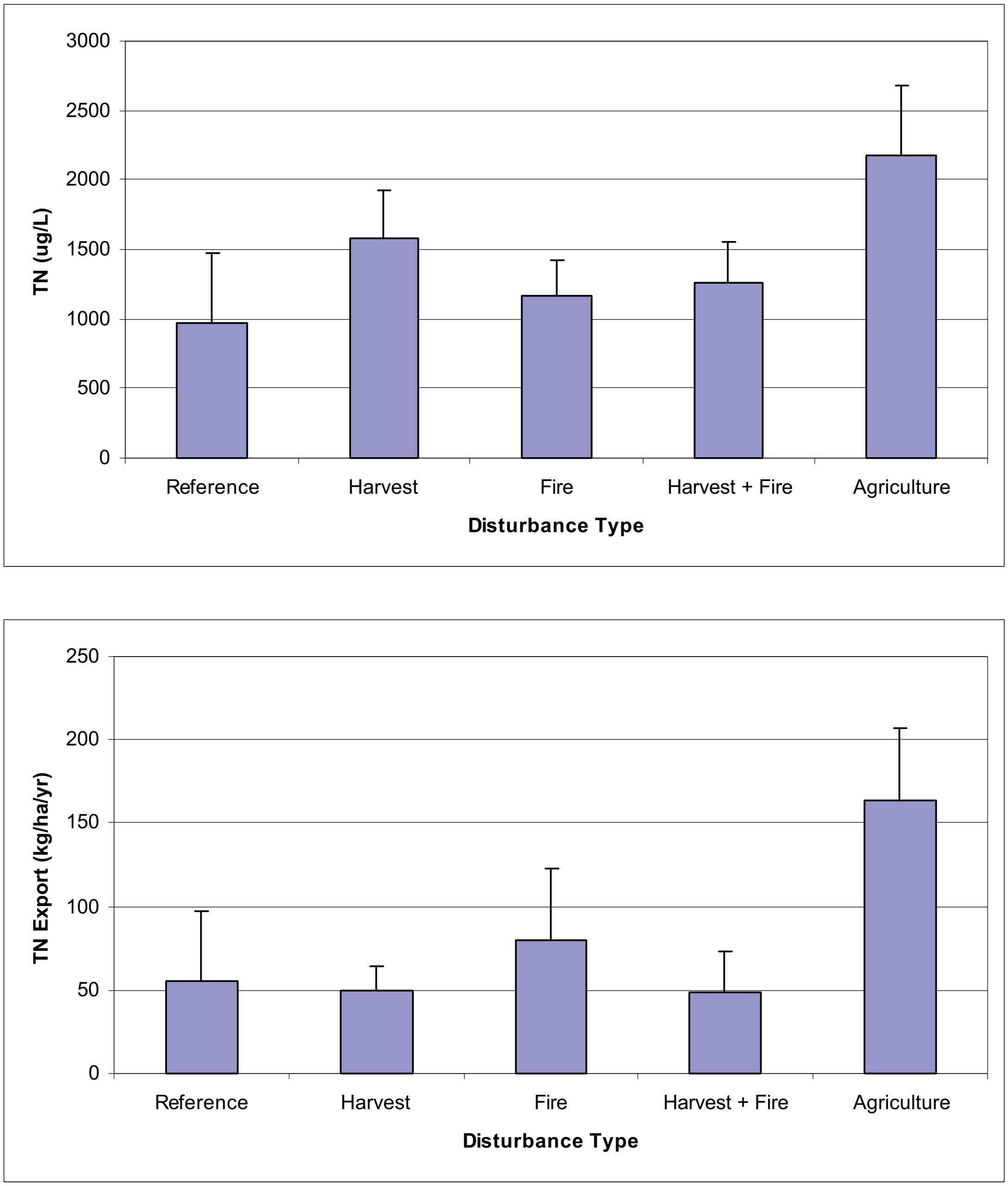


Figure 42. Effects of disturbance type on TN concentration (top) and TN export (bottom) in the study watersheds. Vertical bars are one standard deviation of the mean.

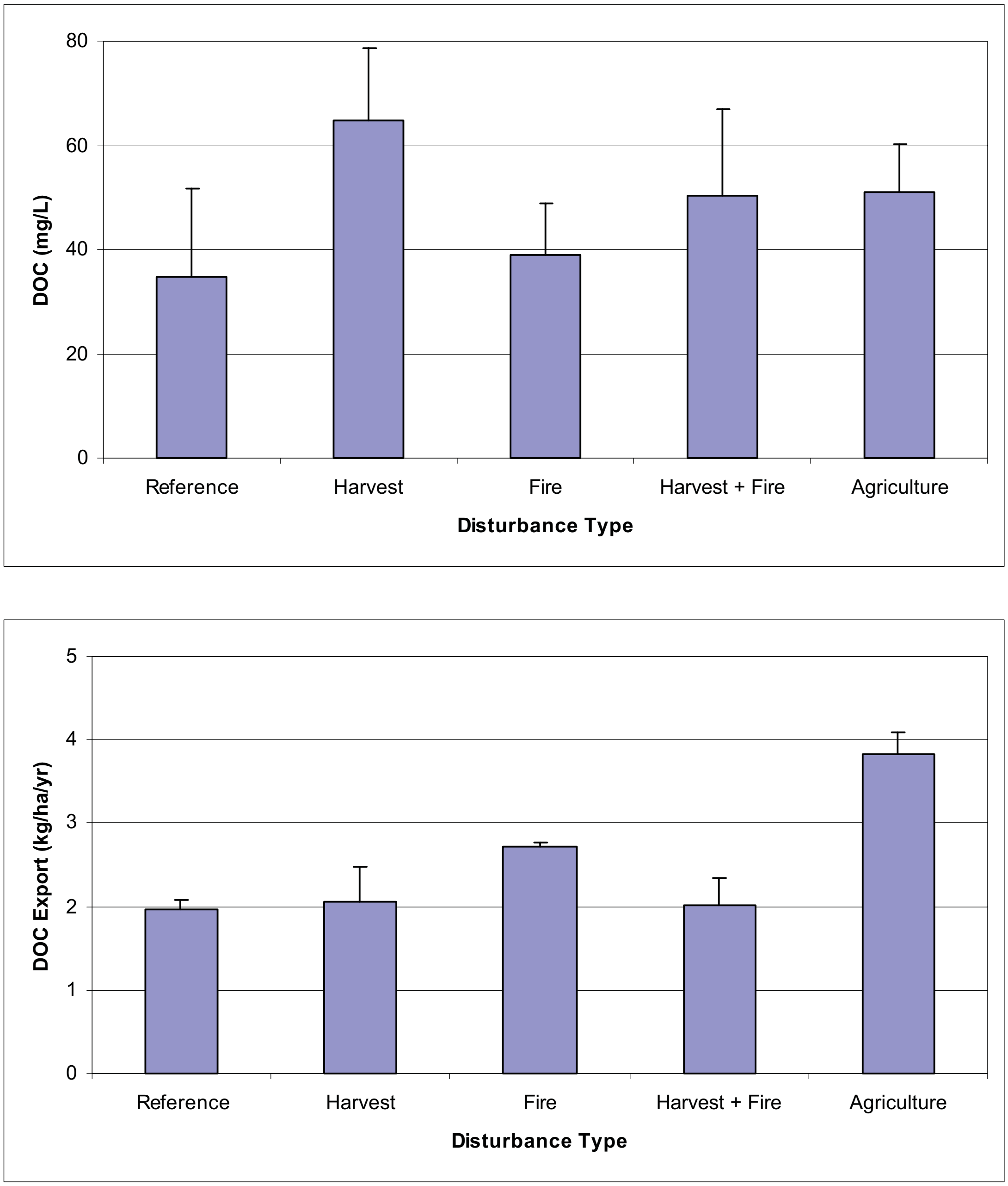


Figure 43. Effects of disturbance type on DOC concentration (top) and DOC export (bottom) in the study watersheds. Vertical bars are one standard deviation of the mean.

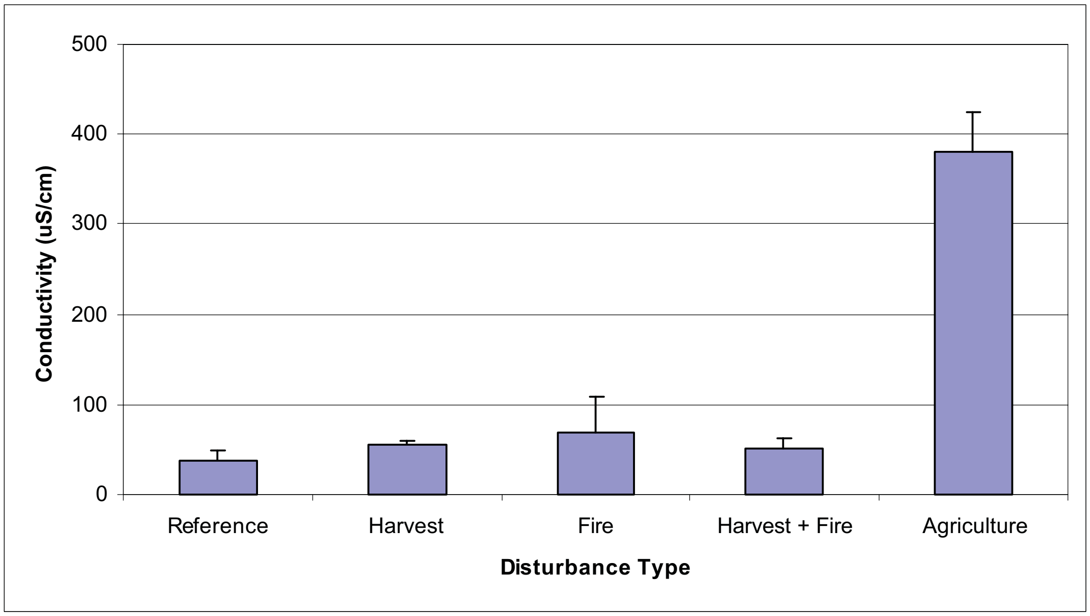


Figure 44. Effects of disturbance type on conductivity in the study watersheds. Vertical bars are one standard deviation of the mean.

Water quality in the agricultural watersheds was quite different from the forested, reference watersheds and watersheds with harvesting, fire or both harvesting and fire disturbances. For example, mean Ca concentration, TP concentration and conductivity were statistically higher in the agricultural disturbance watersheds than all other watershed categories (Figures 40, 41 and 44). Mean TN concentration was also higher in the agricultural watershed than the other watershed categories, although not statistically so (Figure 42). Mean Ca, TP and TN concentrations and conductivity were 5.9, 3.9 and 1.9 times higher, respectively in the agricultural streams than in all of the other streams combined. In contrast, mean DOC concentration in the agricultural watersheds, although higher than in the reference watersheds, was statistically similar to the other disturbance watershed categories (Figure 43). With respect to export coefficients, the agricultural watersheds again had much higher values than any of the other watershed categories. Export coefficients for Ca, TP, TN and DOC were all significantly higher (6.7, 3.1, 2.8 and 1.7 times, respectively) in the agricultural watersheds than in any of the other watersheds combined (Figures 40, 41, 42 and 43). From the data collected, it appears that even low intensity agricultural can have a significant impact on stream water quality.

Harvesting also appeared to have an impact on water quality in the streams, although this impact was generally much less dramatic than that observed for agriculture. Concentrations of Ca, DOC and TN as well as conductivity were statistically higher ($P = 0.03$ to 0.09) in harvested watersheds than in reference watersheds (1.5, 1.9 and 1.6 times higher, respectively). While TP concentrations were 2.1 times higher in harvested watersheds than in reference watersheds, the concentrations were not statistically different ($P>0.01$) due to a large amount of variability in the data. Conductivity was only marginally higher in harvested watersheds compared to reference watersheds. Despite differences in the concentrations of Ca, TP, TN and DOC between harvested and reference watersheds, the export coefficients were not significantly different ($P>0.10$) between the two disturbance categories. In addition, there was no significant differences in turbidity or total suspended solids (TSS) between harvested and reference watersheds (data not shown), although the two agricultural watersheds had much higher turbidity and TSS than any of the forested streams.

In a recent study of the impacts of clear cut harvesting on water quality in lakes of the Coldwater Experimental Watersheds area in northwestern Ontario, Steedman (2000) found only marginal changes or no changes at all in a suite of water quality variables of these boreal shield lakes including TP, TN, DOC, K, Cl, Si and chlorophyll (phytoplankton biomass). In this study, between 45 and 77% of the watershed area around 3 lakes were harvested. In contrast, Carignan et al. (2000) observed significant changes in water quality in boreal shield lakes in the Haute-Mauricie area of Quebec after forest harvesting of between 5 and 73% of the watershed area around 13 lakes, compared to 17 reference lakes. Three years after harvest, DOC concentration and water transparency (measured as light attenuation coefficient) was up to 3 times higher than in reference lakes, TP approximately 2 times higher and TN approximately 1.4 times higher, similar to our results for streams in eastern Manitoba. The differences in response of the lakes between the northwestern Ontario and Quebec boreal shield studies mentioned above may be attributed to differences in the drainage ratios (ratio of lake surface area or volume to watershed area). The lakes in the Carignan et al. (2000) study had higher drainage ratios (and thus, more chance for water movement) than lakes in the Steedman (2000) study. Carignan et al. (2000) developed simple models to predict the impacts of harvesting on water quality, based largely on drainage ratio. This model correctly predicted that lakes with small drainage ratios, such as those found in the Steedman (2000) study, would not likely demonstrate a response to watershed harvesting. Carignan et al. (2000) also found that the longevity of the impacts depended on the water quality parameter in question. For example, mobile ions released by harvesting (e.g., K, Cl, DOC) were rapidly flushed out of the lakes and as a result, concentrations decreased substantially from their peak concentrations within 3 years after harvest. Other constituents (e.g., TP, total organic nitrogen, Ca, Mg) have not decreased, or were still increasing after 3 years.

In our study, the general trends in the effect of fire on water quality was similar to that of harvesting, although statistically significant differences between burned and reference watersheds were not evident. Watersheds with fire had higher concentrations of Ca, TP, TN, DOC and conductivity than reference watersheds (1.7, 1.4, 1.2, 1.1 and 1.8 times higher). In addition, while export coefficients between harvested and reference watersheds were not statistically different, export coefficients for Ca and DOC were statistically higher in burned watersheds than in reference watersheds. Carignan et al. (2000) also observed higher concentrations of TP, TN, Ca and SO_4 in lakes from burned compared to reference watersheds. Carignan et al. (2000) also point out that the chemical form and bioavailability of the exported phosphorus in burned watersheds may be different than that in harvested watersheds. This would have important implications for water quality. Their data suggest that more readily used (i.e., more bioavailable inorganic) forms of phosphorus and nitrogen are exported from burned watersheds compared to harvested watersheds (which export predominantly organic forms of nitrogen and phosphorus). As a result, these released nutrients from burned watersheds would be more available for biological uptake (by algae for example) that the nutrient forms released by harvesting, as observed in the response of phytoplankton and benthic algae in the same Quebec lakes (Planas et al., 2000). Therefore, the higher phosphorus and nitrogen exports from harvested versus burned watersheds does not necessarily indicate that algal growth (as an example) would be enhanced.

Bayley et al. (1992) studied the effects of multiple fires on the nutrient export into streams in forested and fen-dominated watersheds at the Experimental Lakes Area, near Kenora, Ontario. They found that both watershed type (forest versus fen-dominated) and fire intensity was important in explaining the resulting water quality impacts. Wetland basins exported more phosphorus (TDP, TP) and nitrogen (TDN, TN) than the forest basins which were burned, and nutrient export after the second fire (7 years after the first) was not as great as after the first fire. In addition, water quality impacts were greatest in a basin that burned with higher fire intensity than a basin which experienced lower fire intensity.

Prepas et al. (2003) examined the impacts of fire on stream discharge and phosphorus export in streams in the Swan Hills of Alberta. They found that water export from a burned watershed was significantly elevated when considering trends in nearby reference watersheds and despite a regional trend in decreasing precipitation. In addition, Prepas et al. (2003) found that phosphorus export was more significant in the burned watershed, and was highest during storm events compared to base flow conditions. Finally, export of phosphorus from the burned watershed occurred primarily as particulate phosphorus, rather than dissolved phosphorus.

Comparisons of the effects of natural disturbance to harvesting are particularly relevant to the concept of sustainable forest management. Forest management in Canada has been moving towards a system of planning and operations that seek not only to minimize the environmental impact, but to ensure that when impacts do occur, they are well within the range of what would be expected by those caused by natural disturbance agents. Comparisons between the impacts of fire and harvesting on water quality are therefore given below.

In our study, there were no statistically significant differences between the impact of fire and harvesting on water quality for most parameters evaluated. For example, the concentrations of Ca, TP and conductivity were similar between watersheds with fire and watersheds with harvesting. Similarly, there were no statistically significant differences in export coefficients of

TP, DOC and TN between burned and harvested watersheds. However, DOC and TN concentrations were statistically higher in harvested watersheds than in burned watersheds (1.7 and 1.4 times higher, respectively). Carignan et al. (2000) also found that DOC concentrations were higher in lakes of harvested compared to burned watersheds, although the differences were not statistically significant. Lamontange et al. (2000), calculated export coefficients for various water chemistry parameters on the suite of Quebec lakes studied by Carignan et al. (2000), and found that harvested watersheds exported more DOC than burned watersheds.

The higher DOC concentrations in the harvested watersheds and much lower DOC concentrations in burned watersheds can be explained by two factors. Firstly, fire will deplete DOC in soils by burning the organic matter. In harvested watersheds, this organic remains intact. In fact, new woody debris on the forest floor following harvesting would also provide a long-term source (input) to the soil, and thus, to aquatic environments over time. In addition, some of the DOC remaining after fires may become adsorbed to large quantities of ash left on the ground, making this DOC less able to be exported to receiving streams (Wardle et al., 1998). Secondly, difference in DOC concentrations between the two watershed disturbance types is also likely a function of soil type. On average, the percentage watershed area with OD type soils (the soil type with the highest DOC content) in the burned watersheds was only 8.6%, while the harvested watersheds had more than 7 times the area (61%) in OD soils. The harvesting in the watersheds making up the "harvest" category also occurred mainly between 1944 and 1974, a period when the Pine Falls mill was targeting black spruce. Therefore, the harvest in these watersheds likely focused on lowland black spruce areas, containing a high percentage of OD soils. In contrast to our study, McEachern et al. (2000) found that lake water DOC concentrations were much higher in burned watersheds compared to reference watersheds in the boreal forest of northern Alberta. However, the climatic and geologic setting of this area is very different from that of the boreal shield in that the area is underlain by discontinuous permafrost and is dominated by extensive areas of black spruce which is underlain by thick organic soils. Carignan et al. (2000) correctly point out that trying to compare disturbance impacts between vastly different regions of Canada may not be valid.

Different types of watershed-scale disturbance agents (such as agriculture, fire and logging) should be expected to have some differing effects on the magnitude and direction of impacts on water quality. However, all disturbance agents share one process in common that is responsible for triggering impacts on water quality: the disturbance must result in changes to water flow (timing, intensity, duration, etc.) within the watershed and to the receiving streams, in order to result in movement of solutes (e.g., nutrients). Disturbances such as logging and fire result in the dramatic reduction in interception of precipitation by the tree canopy (resulting in a higher proportion of precipitation reaching the soil layer) and a reduction in evapotranspiration (Hetherington, 1987). This can have a net effect of promoting increased saturation of soils, and the movement of more water to channelized flow (e.g., streams and rivers). Thus, forest disturbance leads to increased water yield in watersheds. In general, increased water yield following forest harvest in Canadian streams is much less than that observed in streams in the southern United States, likely due to the cooler temperature and less precipitation (Hetherington, 1987). For Canadian streams in snow-melt driven hydrologic settings, increased water yield and peak flows after harvesting are predicted to occur mainly during spring runoff, as loss of canopy cover (particularly in small cut blocks) tends to cause snow accumulation (Pomeroy and

Granger, 1997). Lack of canopy cover also facilitates more rapid thawing than in the adjacent mature forest, leading to faster and increased peak flows in the spring. This is commonly observed in the field (Swanson and Hillman, 1977). Water yield generally increases in proportion to the fraction of the basin disturbed (Keenan and Kimmins, 1993) and this is also modified by soil depth and precipitation (Hetherington, 1987). However, larger clear cut size or watershed harvest levels may actually reduce flows from harvested areas as snow is redistributed to adjacent forest by increased wind velocity in harvest areas, or through increased ablation processes such as evaporation (Pomeroy and Granger, 1997; Rothwell, 1997). Indeed, Dickson and Daugharty (1982) observed no increase in flows after 92% of a watershed in New Brunswick was harvested. Agriculture would be expected to have a similar, and perhaps more dramatic effect than either fire or logging due to the permanent removal of forest vegetation.

The longevity of the water yield response depends on climate and forest type. For example, evapotranspiration recovery of aspen-dominated forests in Canada after harvesting is thought to occur over a relatively short period of time (approximately 15 years post disturbance), whereas evapotranspiration recovery in coniferous species such as spruce and pine may take 30 years or even longer (Swanson and Hillman, 1977; Rothwell, 1997). Regeneration of the forest, and thus, evapotranspiraton recovery, could take longer in the dryer boreal plain ecozone of Alberta compared to the wetter boreal shield ecozone of central and eastern Canada (Prepas et al., 2003; Smith et al., 2003). Martin et al. (2000) found that water yield, peak flow and most stream water chemistry parameters returned close to preharvest levels in 3 to 6 years after harvest in hardwood-dominated watersheds in Hubbard Brook Experimental Forest in New Hampshire. This suggests that conditions can return to a pre-disturbance state in a much shorter time than predicted by the time needed for complete evapotranspiration recovery alone. These authors also indicate that any transitory impact on water yield and peak flows in smaller, headwater streams will be completely masked in the larger streams and rivers downstream, as water joins from the multiple sub watersheds.

Our study was not capable of examining the effects of either fire or logging on water yield or stream flow due to lack of discharge data on the larger rivers and even some of the smaller streams. Based on the literature, one would expect however, that both forest harvesting and fire would reduce evapotranspiration and increase water yield in our study watersheds. The extent to which water yield increases would also be expected to be proportional to the proportion of watershed disturbed. The longevity of these impacts would likely be shorter in hardwood dominated watersheds, and longer in conifer-dominated watersheds. At the present time, these hypotheses would need to be tested in a more formal experiment. Some of our streams experienced extensive harvesting over the last 60 years (e.g., Kapukwaywetewonk Creek, Lost Creek, O'Hanly mid tributary) and also had some of the highest concentrations of phosphorus, nitrogen, dissolved organic carbon and sulphate. It might be tempting to speculate that the impact of the harvests, which took place in the periods of 1965-1980, 1962-1968 and 1954-1965 in the three streams above, respectively, are still evident in the water quality parameters today. The longevity of the impacts however can not be easily discerned from our study due to the influence of other complicating factors. For example, these streams always likely had higher concentrations of the above water quality parameters due to the dominant soil and forest types in their watersheds.

Tembec currently uses a time period of 7 years as part of their 30% harvest limitation. This period of time is linked to when regeneration surveys are conducted after harvest to verify adequate stocking of trees in harvest areas. This time period however, has no relevance to the time required for recovery of evapotranspiration. Based on the literature, it is likely that recovery of flow and water quality to pre-disturbance values will occur in less time than that predicted for evapotranspiration recovery (less than 30 years for softwoods). Given that harvesting in our study watersheds at a level of less than 40% of watershed area (see next section) resulted in no detectable differences in water quality between harvest and reference watersheds, the 7 year time frame may seem reasonable. However, watersheds with higher levels of harvest may require longer time for recovery. The longevity of impacts should be studied in the Manitoba Model Forest area under more controlled experimental conditions, such as a time series between replicated, paired (harvest, no harvest) basins.

Effects of % Watershed Disturbance by Harvest and Fire on Water Quality

A third and final approach to examining the effects of watershed disturbance on stream water quality was undertaken by relating water quality parameters to the amount of watershed disturbed (% of watershed area disturbed) by logging, fire and both disturbances combined at various time periods since the disturbance. Initially, the analysis was to relate water quality to the cumulative disturbance of logging, fire and both disturbances at 4 times periods: within the last 5 years (i.e., cumulative disturbance from 1999 to 2003), within the last 18 years (1986 to 2003), within the last 30 years (1974 to 2003) and within the last 60 years (1944 to 2003). However, as mentioned in the Methods section, few of the watersheds experienced any significant harvesting within the last 5, 18 or 30 years. In addition, few of the watersheds experienced fire within the last 5 and 18 years. While there have been some watersheds with a significant amount of fire disturbance within the last 30 years, we have decided to use the last 60 years time period for the analysis, to be consistent with the time period used for harvesting. Therefore, we are only examined the effects of % of watershed disturbed at one time period: in the last 60 years.

Table 11 shows the relationships (correlations) between the proportion of the watershed area disturbed (by logging, fire, and both disturbances) within the last 60 years and various water quality parameters.

Table 11. Correlation coefficients and *P* values between the percentage of watershed disturbed by logging, fire and both disturbances, and various water quality parameters.

Water Quality Parameter	**Disturbance Agent**		
	Harvesting	Fire	Both
Ca	0.19 (NS)	0.49 (**)	0.51 (**)
SO_4	0.65 (***)	-0.26 (NS)	0.20 (NS)
TP	0.28 (NS)	0.07 (NS)	0.23 (NS)
TDP	0.45 (**)	0.10 (NS)	0.37 (NS)
NO_3	-0.29 (NS)	-0.09 (NS)	-0.26 (NS)
NH_4	0.23 (NS)	-0.19 (NS)	-0.01 (NS)
TN	0.64 (***)	0.15 (NS)	0.53 (**)
DOC	0.79 (***)	-0.01 (NS)	0.49 (**)
pH	-0.33 (NS)	0.25 (NS)	-0.01 (NS)
Alkalinity	0.02 (NS)	0.63 (***)	0.52 (**)
Conductivity	0.11 (NS)	0.67 (***)	0.61 (***)

P values: NS = not statistically significant (P>0.10), * = P<0.10, ** = P<0.05, *** = P<0.01

There were statistically significant relationships between the area (proportion) of watershed harvested and SO_4, phosphorus (as TDP), TN and DOC concentration in the streams. Higher levels of watershed disturbance were associated with higher concentrations of these parameters in the streams, as shown in Figures 45 to 47.

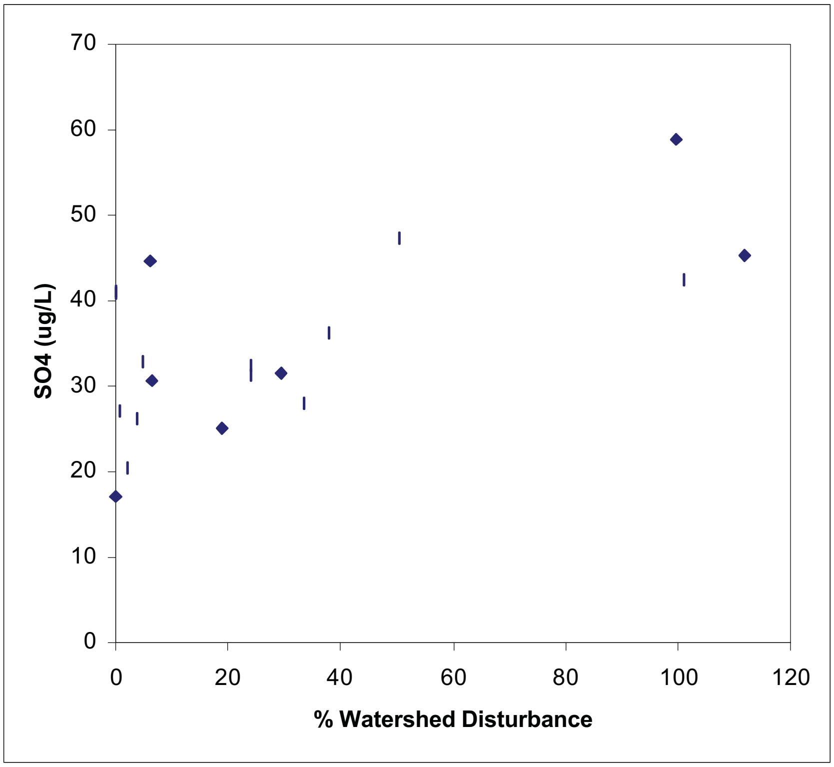
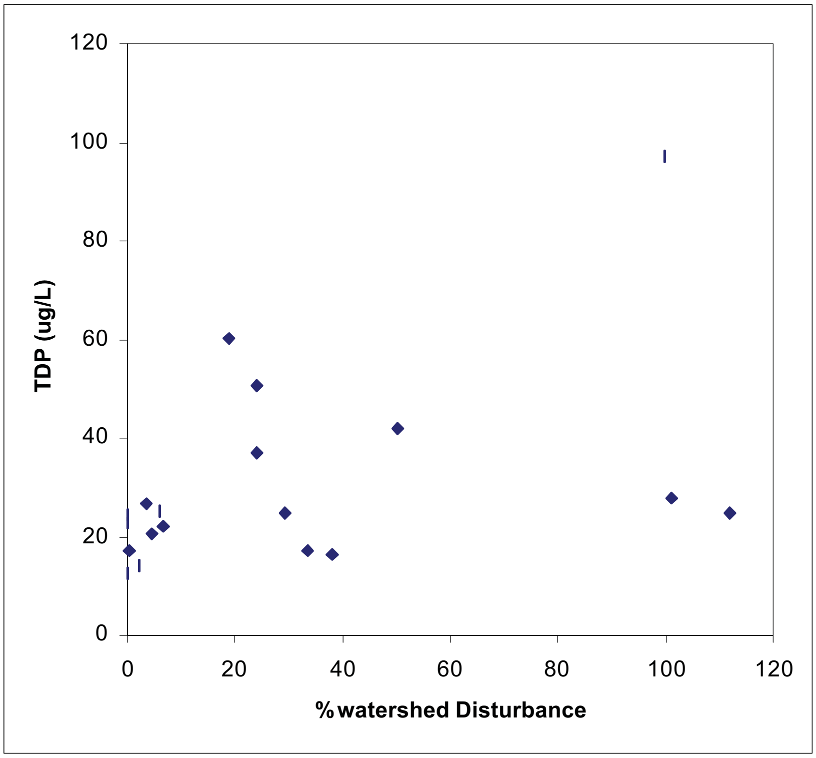


Figure 45. Relationship between % of watershed area harvested and SO_4 concentration in the study streams.



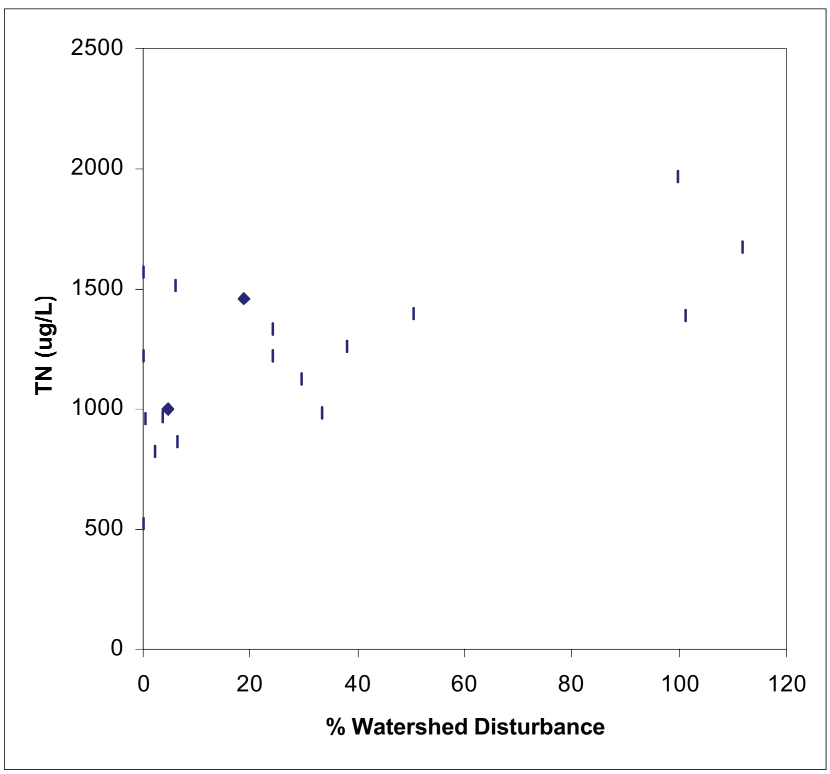


Figure 46. Relationship between % of watershed area harvested and TDP (top) and TN (bottom) concentration in the study streams.

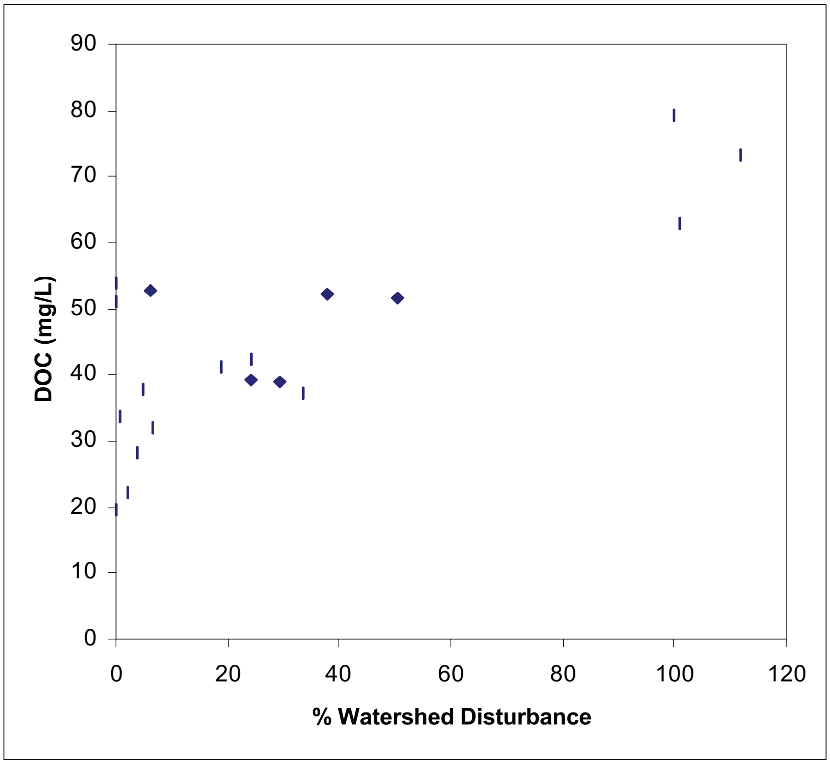


Figure 47. Relationship between % of watershed area harvested and DOC concentration in the study streams.

It is evident from the figures above that one practical method of minimizing the impacts of forest harvesting on water quality (at least for those parameters identified in Figures 45 to 47) would be to limit the area harvested in watersheds to a certain percentage. To our knowledge, this is not a common practice in Canada. However, Louisiana Pacific, in Manitoba has a limit for harvesting under their Environment Act License, of 30%. Tembec (Manitoba operations) has also established a voluntary 30% limit under their Criteria and Indicators of SFM framework.

There are several points that can be made with respect to limiting harvest to a certain percentage (or range) within watersheds to protect water quality. Firstly, at a cursory level, this management approach would seem to be appropriate for some water quality parameters in our study. It is interesting to note that in our study, a harvest level of 40% would maintain SO_4, TN and DOC concentrations in the streams within the range occurring in reference watersheds. This seems to suggest that a limit of 40% would be appropriate for these parameters. This however, does not appear to be the case for phosphorus (Figure 46), and demonstrates the important effect of soil type in the watershed. There were three watersheds (Cat, Duncan, Sandy) with 18-24% of their watersheds harvested in the last 60 years that had much higher stream TDP concentrations than watersheds with no harvesting, or watersheds with a harvest of 30-40%. This can be partially explained by the fact that these watersheds contain more productive soils (i.e., DB + OD soil types) than watersheds with no harvest history (which are dominated by the BR soil type) or those with a 30-40% harvest rate. Due to their soil types, Cat Creek, Duncan Creek and Sandy River have always likely contained higher phosphorus concentrations than the reference streams, even in the absence of disturbance.

It would therefore appear that soil type should possibly be another factor to consider when setting watershed harvest limits. Higher watershed harvesting limits would apply to watersheds with less productive soils (e.g., BR soils) and lower watershed harvesting limits would apply to watersheds with more fertile soils (e.g., DB and OD soils). This should be used as a general guidance, as there are exceptions. For example, the Rice River and English Brook had moderately high amounts of OD and DB soils in their watersheds and almost no harvest activity. Based on this, one would predict that these streams should have higher TDP concentrations when in fact they do not. At the other extreme are Kapukwaywetewonk Creek and Lost Creek, which have had their entire watershed area harvested in the last 60 years (and in the case of Lost Creek, the entire watershed burned as well). One would predict very high stream TDP concentrations, and this is not the case. TDP concentrations ranged from 24-28 ug/L, at the low end of TDP concentrations in any of the streams.

For DOC, the 30% watershed limit is consistent with observations elsewhere in the Canadian boreal forest. In the study of 13 harvested and 17 reference lake watersheds, Carignan et al. (2000) state that in lakes with higher drainage ratios (and thus, higher probability of water quality impacts caused by harvesting), DOC concentrations will be maintained within their natural range of variation if harvesting does not exceed approximately 30% of the watershed. This appears to be consistent with our data.

When restricting harvest to a certain percentage of watershed area as a management tool, one must also take into consideration the size of the watershed. In our study, those watersheds with the highest level of harvesting, and consequently the highest stream water concentrations of SO_4, TN and DOC, are also the smallest watersheds in our study (1.4 to 14 km^2). The impacts of watershed disturbance on water yield and water quality are predicted to be greatest in smaller (<100 km^2) watersheds than larger watersheds (Hetherington, 1987; Rothwell, 1997). Tembec's current approach in eastern Manitoba to limit harvesting to less than 30% of watershed area is based on fairly large watersheds (average of 272 km^2, range of 16-571 km^2). While it is not likely desirable or feasible to manage harvesting on watershed scales of less than 25 km^2, planning and management at watershed scales of 500 km^2 are likely to be meaningless in terms of protection of water quality (Martin et al., 2000). The choice of what watershed scale to manage at is subjective, and not easily answered by our data. We would suggest however, that watershed management should be based on watershed scales of no larger than 100 km^2 (Rothwell, 1997), based on the observation that water impacts are less likely to be observed above this threshold. For Tembec, this would entail splitting up their larger watersheds into smaller management units.

In order to use a watershed harvesting limit as a tool to incorporate water quality objectives into forest management planning, a data set consisting of a much larger suite of watersheds and water bodies is required, spanning a gradient in soil type, forest type, and disturbance history is required. Fortunately, a companion study to ours is currently collecting that information. The Manitoba Model Forest Watershed Planning Tools project has completed a survey of 100 lakes in eastern Manitoba and is currently completing a data analysis identical to the one done in our stream water quality project. In 2006, the lake and stream data sets will be combined in order to develop a suite of simple watershed management tools for use by Tembec in FML 01. As mentioned previously, Tembec is currently using a 30% watershed disturbance limit as a general guideline under their Criteria and Indicators of Sustainable Forest Management approach. This limit not only includes harvesting, but also fire in the watersheds. The stream and lake water quality studies will help determine if this limit is appropriate.

When we compare the relationships between % of watershed area disturbed by logging and fire, the water quality parameters that were most highly correlated to the percentage of watershed harvested, were not the same as those for fire (Table 11). While SO_4, TDP, TN and DOC concentrations were correlated to % watershed harvested, these same parameters were not correlated to the % watershed burned. Only Ca, alkalinity and conductivity were correlated to the proportion of watershed area burned in the last 60 years (Figures 48 and 49).

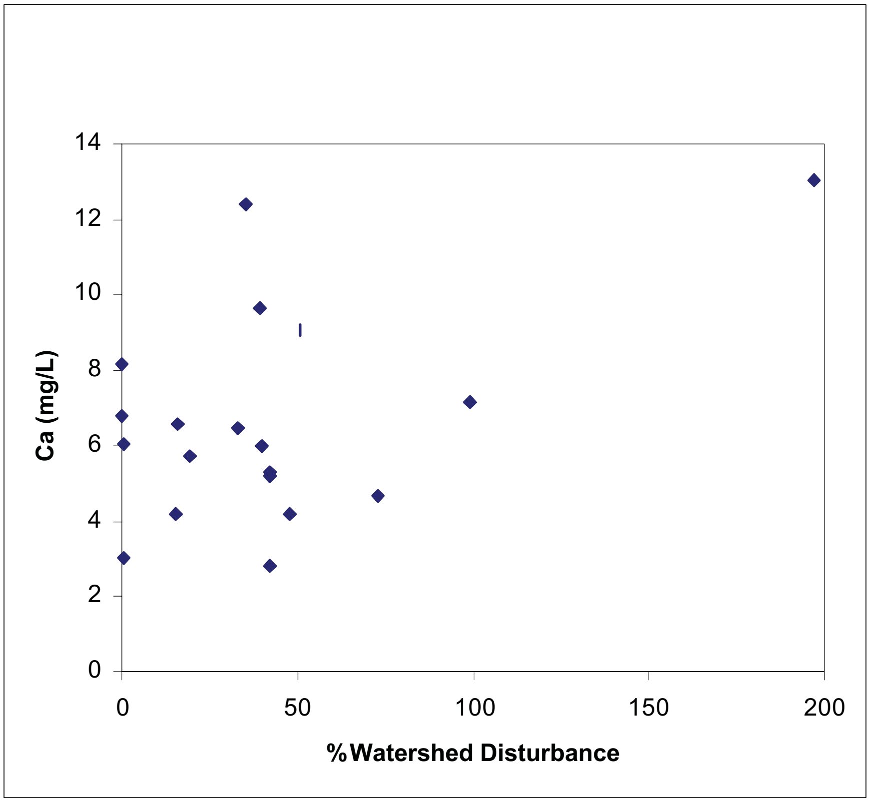


Figure 48. Relationship between % of watershed area burned and Ca concentration in the study streams.

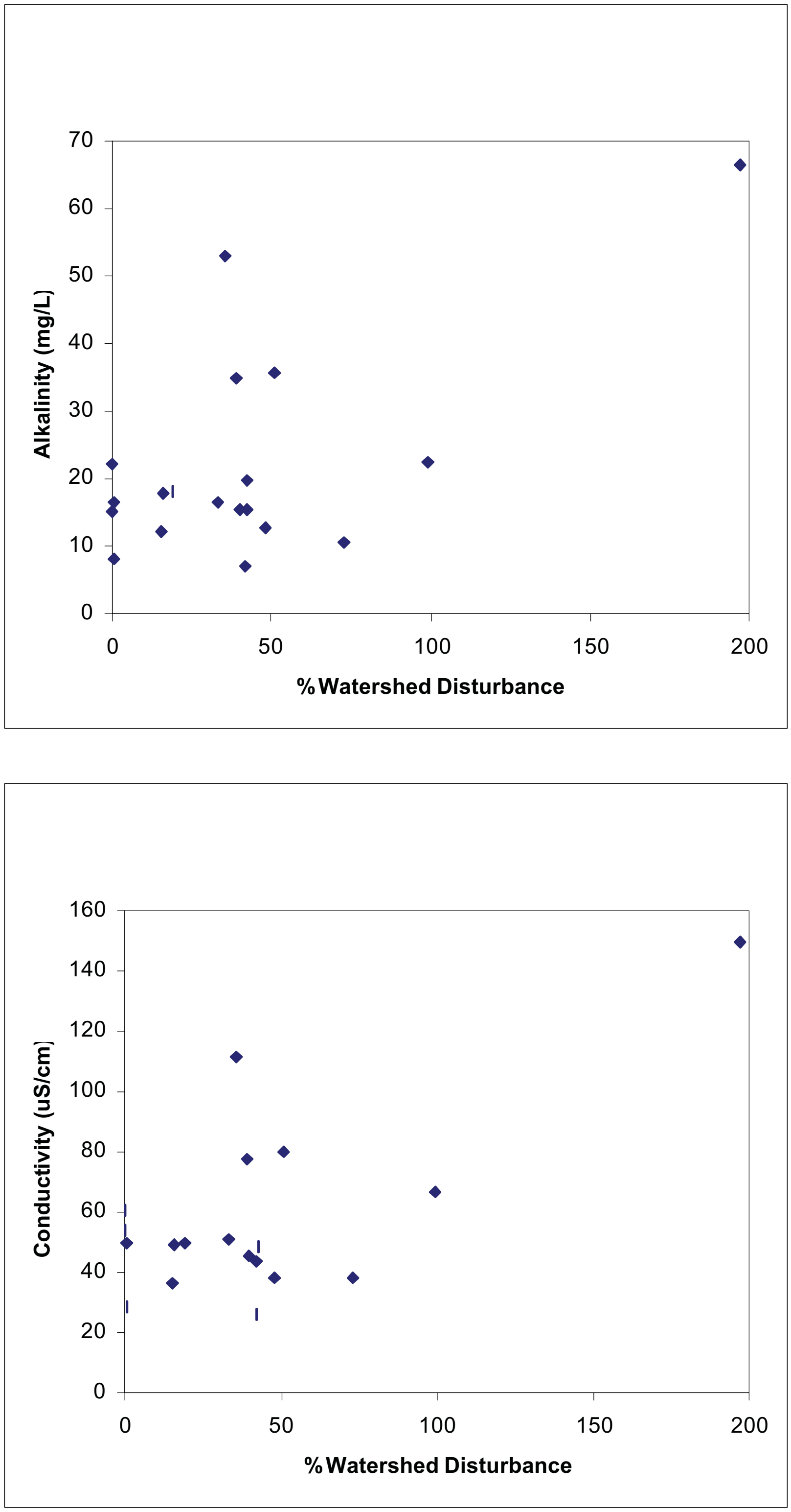


Figure 49. Relationship between % of watershed area burned and alkalinity (top) and conductivity (bottom) in the study streams.

When compared to the relationships observed between % watershed harvest and water quality, there is more variability in the data for the relationships with % watershed burned. In fact the statistical strength of the relationships between % watershed burned and Ca, alkalinity or conductivity is largely due to one watershed (Duncan Creek, the data point at the top right-hand corner of Figures 48 and 49). The Duncan Creek watershed was burned twice (and completely) in the last 18 years, is comprised entirely of DB soils and has a high proportion of aspen hardwood forests relative to other watersheds. In addition, the watershed has a high proportion of area in beaver floods and significant beaver activity. The combination of all these factors contribute to its very high Ca, alkalinity and conductivity compared to other forested watersheds.

<u>Influence of Climate on Water Quality</u>

The influence of watershed characteristics and processes on water quality is governed by a large number of factors. These include not only those already discussed in earlier sections of this report (e.g., soils, forest type, disturbance history) but also the processes that link the flow of water from the terrestrial landscape to the aquatic environment. The movement of substances such as phosphorus, nitrogen, DOC, etc. from soils to water bodies is tightly linked to the hydrologic cycle, and thus, climate. During dry years, one would expect less movement and export of substances from watersheds to receiving waters, and during wet years, more export would be expected to occur. This has been commonly noted as an explanatory variable in many studies. For example, part of the lack of response in harvesting effects during the first year after harvesting in the study by Steedman (2000) for northwestern Ontario lakes was attributed to dry weather. Large increases in soil phosphorus concentration in a study examining the effects of harvesting in aspen-dominated forested catchments in Alberta occurred during very wet years when shoreline areas around one lake were flooded (Evans et al., 2000) and the water table intersected with the organic layer of the soils.

The lack of precipitation during dry years and lack of runoff not only has an impact on the mobility of dissolved substances, but plays a major role in transforming substances from one chemical form to another. For example, during dry years when the water table is drawn downward in soils, considerable mineralization of less mobile organic forms of nitrogen and phosphorus to more mobile, inorganic forms can occur. In addition, high concentrations of SO_4 can be produced though the aeration/oxidation of sulfide in wetlands (Devito and Hill, 1997) during dry years, as well as high concentrations of TP and TN (Bayley et at., 1987; Van dam, 1988; Devito and Dillon, 1993a). In contrast to terrestrial soils, wetlands such as conifer swamps transform inorganic forms of phosphorus and nitrogen into organic forms, particularly during dry years (Devito and Dillon, 1993a). These high concentrations are then easily flushed to receiving streams during wet years. Therefore, one could expect larger exports of dissolved substances after a watershed disturbance, if the disturbance occurs during dry years and then is followed by successive wet years. Devito and Hill (1993a) propose a conceptual model of nutrient generation and transport that states that during periods of dry years or drought, localized areas with little contact with groundwater flow become disconnected from major water flow paths and become "hot spots" for the generation of dissolved substances (nutrients, etc.). Then during wet years, these hot spots are hydrologically reconnected with major water flow paths and become significant sources of nutrient export. This is particularly relevant in the Canadian shield where soils and many wetlands are shallow, and easily disconnected from regional water flow paths. This is likely the case in the Manitoba Model Forest area, especially towards the Manitoba/Ontario border.

Beaver Activity in the Study Watersheds

In September, 2004 an aerial survey of 12 of the study streams was conducted to document the occurrence of beaver dams and the extent of back flooding. A total of 512 km of stream length was flown over two days. This included the main stems of the rivers/stream/creeks as well as some of their major tributaries. Black River and Beaver Creek Main were the two longest water bodies that were flown (132 and 108 km, respectively, Figure 50), and Kenny and Drew creeks were the shortest (5 and 6 km, respectively, Figure 50).

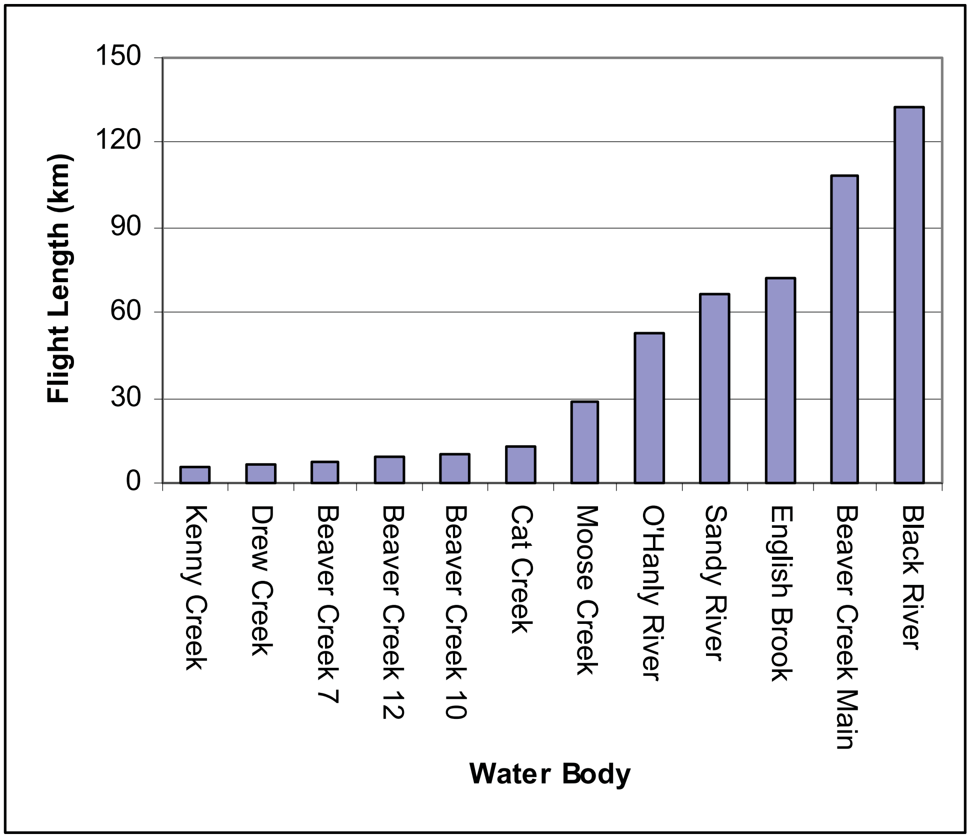


Figure 50. Flight length (km) of aerial surveys for beaver activity on study streams.

As noted in the Methods section, an assessment of the degree of back-flooding was made for each dam with the following classification system: Class I – no significant back flooding, Class II – back flooding of approximately 5 times the normal channel width, and Class III – flooding of more than 10 times the normal channel width. Figure 51 indicates the total number of beaver dams observed on each stream, by flooding class. Beaver Creek Main had the most number of beaver dams recorded in total, partly because this was one of the longest streams flown. English Brook had the second highest number of beaver dams.

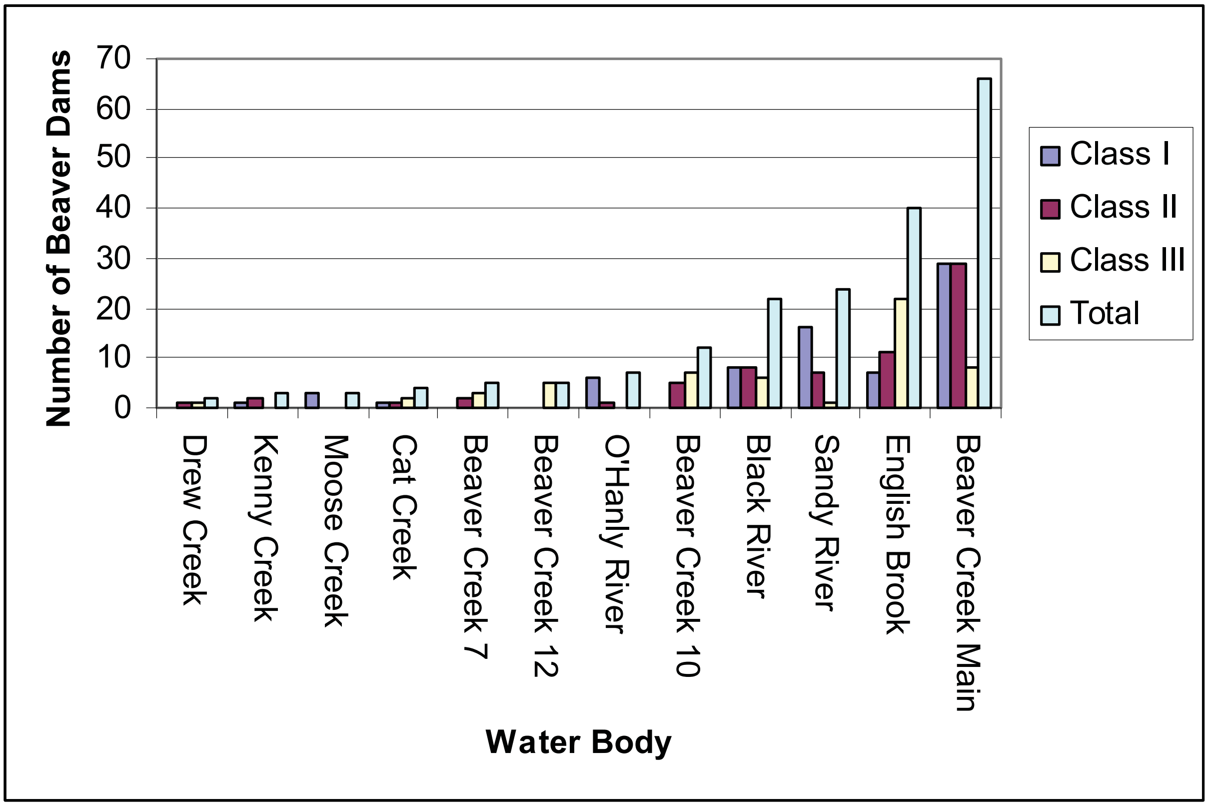


Figure 51. Total number of beaver dams by flooding class during aerial survey.

To account for the differing lengths of each stream, the density of beaver dams (number of dams per km) was calculated. Figure 52 shows the beaver dam density by flooding class in each stream. Generally, the smaller streams and creeks had higher beaver dam densities than did the larger rivers, although small tributaries of the larger rivers also contained significant beaver activity. The small watersheds located in the Tembec Beaver Creek Operating Area (e.g., Beaver Creek Crossing 10, 7 and 12) had the highest overall density of beaver dams (1.1, 0.6 and 0.5 per km, respectively). In addition, a significant proportion of those dams were Class III, causing significant, localized back flooding. The highest density of contiguous beaver dam activity occurred on two un-named creeks near Beaver Creek. On one creek, there were 16 Class

I dams over a 2.6 km distance (giving a density of 6.2 dams per km) and on the other creek there were 14 dams over a 2.9 km distance (giving a density of 4.8 dams per km).

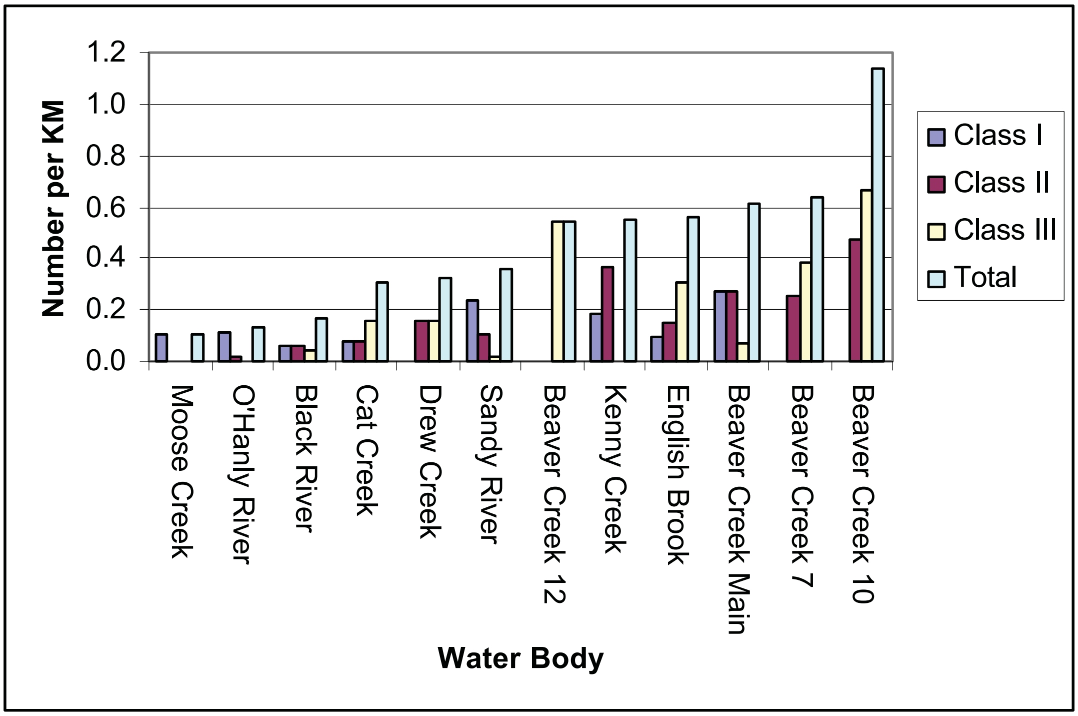


Figure 52. Number of beaver dams per km of stream by flooding class during aerial survey.

The occurrence and density of Class III dams, and its effect on reducing water flow, reducing dissolved oxygen (particularly in the winter) and leaching nutrients from surrounding saturated soils may provide some explanation for the water quality observed in the study streams. For example, winter dissolved oxygen concentration in the streams was negatively correlated to Class III dam density (r = -0.45), winter TP concentration in the streams was positively correlated to Class III dam density (r = 0.55), winter NH_4 and winter TN concentrations were positively correlated to Class III dam density (r = 0.77 and r = 0.75, respectively) and summer TN concentration was positively correlated to Class III dam density (r = 0.51). This suggests that beaver dams and their effects on flow and back flooding may play a significant role in altering water quality in smaller streams and creeks. There are also several published studies that support these conclusions. Devito and Dillon (1993b) found that the initial flooding of uplands and accumulation of organic material in beaver ponds is very important to the phosphorus and nitrogen dynamics of boreal streams in Ontario. They found that dissolved oxygen concentrations approached zero under ice in the winter, leading to significant concentrations of phosphorus and nitrogen (especially ammonia), and that there was a significant export of these nutrients downstream during spring melt. Our results support this as well, particularly with ammonia, which is very sensitive to dissolved oxygen concentration. Similarly, Dillon et al. (1991) studied 32 forested stream catchments in Ontario and found that beaver ponds are sources of high concentrations of TP and total organic nitrogen (TON). Naiman et al. (1994) studied the impacts of beaver on water quality of streams as well as the linkages and movements of nutrients from upland areas to aquatic environments in northern Minnesota They suggest that the net effect of beavers is to translocate elements (e.g., nutrients) from inundated upland forest vegetation and soils to downstream communities and beaver pond sediments. They found that beaver pond sediments can accumulate vast stores of nutrients, which would be released to the water, particularly under low dissolved oxygen conditions (Devito and Dillon, 1993b; our study). Naiman et al. (1994) also state that the export of DOC in streams with beaver activity is several times higher than in streams without beaver. This is also supported by our results.

In conclusion, we found that soil and forest type in watersheds in eastern Manitoba exert a significant influence on stream water quality. Watershed disturbance such as clearing of forest catchments for non-intensive agriculture, had a greater impact on water quality than logging or fire in forested watersheds. While both logging and fire was related to increased stream phosphorus, nitrogen and color, logging appeared to have a larger effect on nitrogen and color than did fire. Stream nitrogen, color and sulphate were related to the proportion of watershed logged, with effects not observable below a watershed harvest threshold level of 40%. This may suggest that the 30% harvest limit currently utilized by Tembec is appropriate, although this warrants further investigation, including the longevity of impacts and the most appropriate watershed size for forest management planning in the Manitoba Model Forest area. Winter road density also appeared to be related to stream concentrations of sulphate, phosphorus, nitrogen and color, although these relationships may be entirely due to the fact that these roads occur on organic soils (which naturally result in higher concentrations of these water quality parameters in streams). Finally, the impact of beaver on water quality in this region can not be ignored. Beaver dams resulted in stream flow reduction (including stagnation of flow) and significant back-flooding of riparian areas along small streams and creeks. As a result, these small water bodies had much lower dissolved oxygen and much higher concentrations of phosphorus, nitrogen (especially NH_4) and color. In some cases, the effects of beaver dwarfed the effects of all other disturbance types.

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Appendix 1. Analytical Methods and References for Water Chemistry Analyses.

Sample Parameter Qualifier key listed:

Qualifier	Description
RAMB	Result Adjusted For Method Blank

Methods Listed (if applicable):

ETL Test Code	Matrix	Test Description	Preparation Method Reference(Based On)	Analytical Method Reference(Based On)
ALK-TOT-WP	Water	Alkalinity		APHA 4500B, 2510B, 2320B, 1998

Alkalinity of water is a measure of its acid neutralizing capacity. Alkalinity is imparted by bicarbonate, carbonate and hydroxide components of water. It is determined by titration with a standard solution of strong mineral acid to the successive HCO3- and H2CO3 endpoints indicated electrometrically.

C-DIS-ORG-WP	Water	Dissolved Organic Carbon		APHA 5310 B

This method is applicable to the analysis of ground water, wastewater, and surface water samples. The form detected depends upon sample pretreatment: Unfiltered sample = TC, 0.45um filtered = TDC. Samples are injected into a combustion tube containing an oxidation catalyst. The carrier gas containing the combustion product from the combustion tube flows through an inorganic carbon reactor vessel and is then sent through a halogen scrubber into a sample cell set in a non-dispersive infrared gas analyzer (NDIR) where carbon dioxide is detected. For total inorganic carbon and dissolved inorganic carbon, the sample is injected into an IC reactor vessel where only the IC component is decomposed to become carbon dioxide.

The peak area generated by the NDIR indicates the TC/TDC or TIC/DIC as applicable. The total organic carbon content of the sample is calculated by subtracting the TIC from the TC.
TOC = TC-TIC, DOC = TDC-DIC, Particulate = Total - Dissolved.

CL-SOL-WP	Water	Chloride Soluble		APHA4500;1998/LACHAT;MAR 1997

Chloride - Colourimetric using Mercuric Thiocyanate

EC-WP	Water	Conductivity		APHA 4500B, 2510B, 2320B, 1998

Conductivity of an aqueous solution refers to its ability to carry an electric current. Conductance of a solution is measured between two spatially fixed and chemically inert electrodes.

ETL-HARDNESS-EXT-WP	Water	Hardness Calculated		Calculated
ETL-ROU-EXT-LOW-WP	Water	Metals for Ion balance		EPA 200.8 Rev 5.4 May 1994
N-TOTKJ-WP	Water	Total Kjeldahl Nitrogen		Quickchem method 10-107-06-2-E Lachat

Samples are digested with a sulphuric acid solution, cooled, diluted with water, and analyzed for ammonia. Total Kjeldahl nitrogen is the sum of free-ammonia and organic nitrogen compounds which are converted to ammonium sulphate through this digestion process. Analysis is performed by Flow Injection
Analysis (FIA). The pH of the digested sample is raised to a known, basic pH by neutralization with a concentrated buffer solution. This neutralization converts the ammonium cation to ammonia. The ammonia produced is heated with salicylate and hypochlorite to produce blue colour which is proportional to the ammonia concentration.

N2N3-SOL-WP	Water	Nitrate + Nitrite Soluble		APHA4500;1998/LACHAT;MAR 1997
NH3-SOL-WP	Water	Ammonia Soluble		APHA4500;1998/LACHAT;MAR 1997

Ammonia - Colourimeric using Salicylate-nitroprusside and hypochlorite, in an alkaline phosphate buffer.

P-TOTAL-WP	Water	Phosphorus, Total		APHA, 1998

Samples are digested using a sulphuric acid-persulphate mixture to convert organic phosphorous to orthophosphate. The samples are analyzed by either the Flow Injection Analysis (FIA) or the Segmented Flow Analysis (SFA) method. The absorbance measured by the instrument is proportional to the concentration of orthophosphate in the sample, and is reported as phosphorous. Samples are analyzed for total or total dissolved phosphorous depending on the sample pretreatment.

P-TOTALDIS-WP	Water	Phosphorus, Total Dissolved		APHA, 1998

Samples are digested using a sulphuric acid-persulphate mixture to convert organic phosphorous to orthophosphate. The samples are analyzed by either the Flow Injection Analysis (FIA) or the Segmented Flow Analysis (SFA) method. The absorbance measured by the instrument is proportional to the concentration of orthophosphate in the sample, and is reported as phosphorous. Samples are analyzed for total or total dissolved phosphorous depending on the sample pretreatment.

PH-WP	Water	pH		APHA 4500B, 2510B, 2320B, 1998

pH of a sample is the determination of the activity of the hydrogen ions by potentiometric measurement using a standard hydrogen electrode and a reference electrode.

SO4-SOL-WP	Water	Sulphate Soluble		APHA4500;1998/LACHAT;MAR 1997

Sulphate - Turbidimetric

SOLIDS-TOTSUS-WP	Water	Total Suspended Solids		APHA 2540