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The reports are listed in alphabetical order. The designation of [DOE] or [DFO] after each listing indicates the availability of each report.

For information on DOE designated reports contact

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FEMP Reports
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The five reports published here have not been published previously.

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REFERENCES

- Baker, G.D. 1990. The Economic Performance of the Southern Indian Lake Summer Commercial Fishery, 1988. M.N.R.M. practicum, University of Manitoba, Winnipeg, Man. 85 p. [DFO]
- Baker, R.F., and S. Davies. 1991. Physical, Chemical and Biological Effects of the Churchill River Diversion and Lake Winnipeg Regulation on Aquatic Ecosystems. Can. Tech. Rep. Fish. Aquat. Sci. 1806: v + 53 p. [DFO]
- Barnes, N.E. 1990. Abundance and Origin of Lake Whitefish, Coregonus Clupeaformis (Mitchell) Congregating Downstream of the Missi Falls Control Dam, Southern Indian Lake, Manitoba, M.S. thesis, University of Manitoba, Winnipeg, Man. 180 p. [DFO]
- Duncan, D.A. and D.A. Williamson. 1988. Water Chemistry/Water Discharge Relationships Within the Churchill River Diversion and Lake Winnipeg Regulation Region, Manitoba, Canada. Ecological Report Series No. 88-5. Environment Canada, Winnipeg, Man. 57 p. [DOE]
- Gerard, A. 1989. Hydroelectrical Power Development and River Ice Regime. Ecological Report Series No. 89-6. Environment Canada, Winnipeg, Man. 122 p. [DOE]
- Grapentine, L. et al. 1988. Identification, Collation and Initial Evaluation of Historical Water Quality Data from the Northern Flood Agreement Area, Manitoba. Ecological Report Series No. 89-5. Environment Canada, Winnipeg, Man. 52 p. [DOE]
- E.M.S.I. 1987. Remote Sensing Imagery Catalogue, with Applications to Environmental Monitoring in Northern Manitoba. Ecological Report Series No. 87-3. Environment Canada, Winnipeg, Man. 553 p. [DOE]
- Hanuta, I. 1990. Climate Characterization of the Federal Ecological Monitoring Program Study Area. Environment Canada, Winnipeg, Man. (unpublished) 46 p. [DOE]
- I.D. Systems Ltd. 1991. Effects of the Lake Winnipeg Regulation Project on Waterfowl Use of the Outlet Lakes Area and on the Norway House Band. FEMP Technical Report, Environment Canada, Winnipeg, Man. 72 p. [DOE]
- I.D. Systems Ltd. 1991. Effects of the Churchill River Diversion Project on Waterfowl Use of the Rat-Burntwood Rivers System and on the Nelson House Band. FEMP Technical Report, Environment Canada, Winnipeg, Man. 55 p. [DOE]
- Kellerhals Engineering Services Ltd. 1987. Morphological Effects of the Churchill River Diversion, Volume 1. Ecological Report Series No. 87-5. Environment Canada, Winnipeg, Man. 96 p. [DOE]

- Kellerhals Engineering Services Ltd. 1988. Morphological Effects of the Churchill River Diversion, Volume 2. Ecological Report Series No. 88-4. Environment Canada, Winnipeg, Man. 139 p. [DOE]
- Lapointe, F. et al. 1989. An Interagency Comparison of the Field and Laboratory Procedures for Determining Suspended Sediment. Ecological Report Series No. 89-4. Environment Canada, Winnipeg, Man. 56 p. [DOE]
- Mavros, W.U. and R.A. Bodaly. (in prep) The Effect of the Lake Winnipeg Regulation Hydroelectric Project on the Stock Genetics of Lake Whitefish (*Coregonus Clupeaformis*) in Playgreen Lake and Northern Lake Winnipeg: Preliminary Results. Department of Fisheries and Oceans, Winnipeg, Man. [DFO]
- McCullough, G.K. 1992. Effects of Hydroelectric Development on River Flows and Lake Levels in Northern Manitoba. (in prep.) [DFO]
- McCullough, G.K. 1992. Effects of Hydroelectric Development on the Areas of Lakes in Northern Manitoba. (in prep.) [DFO]
- McKerness, L. 1989 a. A Retrospective Analysis of a Native Information Program. Ecological Report Series No. 89-8. Environment Canada, Winnipeg, Man. 43 p. [DOE]
- McKerness, L. 1989 b. Environment Canada's Federal Ecological Monitoring Program. Ecological Report Series No. 89-1. Environment Canada, Winnipeg, Man. 65 p. [DOE]
- McKerness, L. 1989 c. Sediment Sampling Stations in the Lake Winnipeg, Churchill, Nelson Rivers Area: An Annotated Atlas. Ecological Report Series No. 89-3. Environment Canada, Winnipeg, Man. 111 p. [DOE]
- McLaren Plansearch. 1987. Annotated Bibliography for Churchill-Nelson River Diversion Region. Ecological Report Series No. 87-1. Environment Canada, Winnipeg, Man. 123 p. [DOE]
- Northwest Hydraulic Consultants Ltd. 1987. Assessment of Sediment Effects Churchill River Diversion, Manitoba, Phase I. Ecological Report Series No. 87-4. Environment Canada, Winnipeg, Man. 76 p. [DOE]
- Northwest Hydraulic Consultants Ltd. 1988. Assessment of Sediment Effects, Churchill River Diversion, Manitoba, Phase II. Ecological Report Series No. 88-3. Environment Canada, Winnipeg, Man. 52 p. [DOE]
- Peristy, D. 1989. A Biological Assessment of the Post-Impoundment Commercial Fishery at Southern Indian Lake, Manitoba, with Historical Comparisons. M.N.R.N. practicum, University of Manitoba, Winnipeg, Man. 141 p. [DFO]
- Ralley, W.E. and D.A. Williamson. 1990. Multivariate Analysis of Water Chemistry Changes Following Hydroelectric Development in Northern Manitoba, Canada. Ecological Report Series No. 90-1. Environment Canada, Winnipeg, Man. 90 p. [DOE]

- Ramsey, D.J. 1991. Federal Ecological Monitoring Program Final Water Quality Report. FEMP Technical Report, Environment Canada, Winnipeg, Man. 320 p. [DOE]
- Ramsey, D.J. 1991. Federal Ecological Monitoring Program Final Mercury Report. FEMP Technical Report, Environment Canada, Winnipeg, Man. 140 p. [DOE]
- Strange, N.E., R.A. Bodaly, and R.J.P. Fudge. 1991. Mercury Concentrations of Fish in Southern Indian Lake and Issett Lake, Manitoba, 1975-88: The Effect of Lake Impoundment and Churchill River Diversion. Can. Tech. Rep. Fish. Aquat. Sci. 1824:v + 30 p. [DFO]
- Strange, N.E., R.J.P. Fudge, and R.A. Bodaly. 1991. Post-Impoundment Response of A Boreal Northern Pike (Esox Lucius) Population in Wupaw Bay, Southern Indian Lake, 1976-88. Can. Tech. Rep. Fish. Aquat. Sci. 1823: V + 49 p. [DFO]
- Wiens, A.P. and D.M. Rosenberg. 1991. Surveys of Benthic Macroinvertebrates of Playgreen and Kiskittogisu Lakes, Northern Manitoba. Can. Tech. Rep. Fish. Aquat. Sci. 1814: IV + 21 p. [DFO]
- Wiens, A.P. and D.M. Rosenberg. (in prep). The Churchill River Impoundment and Diversion: Effects on Benthic Invertebrates of Downstream Lakes. Can. Tech. Rep. Fish. Aquat. Sci. 1814: IV + 21 p. [DFO]
- Usher, P.J. and M.S. Weinstein. 1991. Towards Assessing the Effects of Lake Winnipeg Regulation and Churchill River Diversion on Resource Harvesting in Native Communities in Northern Manitoba. Can. Tech. Rep. Fish. Aquat. Sci. 1794: vi + 69 p. [DFO]

SPECIAL NOTE

Conclusions, recommendations or opinions expressed in reports prepared for the Federal Ecological Monitoring Program are those of the authors and do not necessarily represent those of the sponsoring departments.

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Federal Ecological Monitoring Program
Final Water Quality Report

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Executive Summary

Environment Canada and the Department of Fisheries and Oceans undertook in 1986 a five year program of ecological monitoring in northern Manitoba which has come to be known as the Federal Ecological Monitoring Program (FEMP). The overall purpose of this program is to address Canada's obligations under the Northern Flood Agreement (NFA) with respect to requirements for ecological monitoring and the identification and disclosure of adverse effects of Churchill River diversion (CRD) and Lake Winnipeg regulation (LWR). This report presents the results of the water quality monitoring program implemented by Environment Canada as part of the FEMP. Current (1987-1989) water quality is described, statistically significant changes in physical and chemical water quality since the LWCNR project began operation are documented and, where possible, the cause of these changes is determined. Any adverse effects directly resulting from changes in water quality caused by the Project are identified. The definition of an adverse effect employed in this report is a statistically significant change in water quality, caused by the hydroelectric developments, which has a detrimental effect on one or more of the traditional water uses of the communities. A detrimental effect is a change in one or more water quality variables such that the variable is not within the acceptable limits of one or more of the Canadian Water Quality Guidelines for drinking water, recreational use, or protection of aquatic life.

Current water quality was documented in a three year study (January 1987 to October 1989) of 61 physical and chemical water quality parameters at 11 sites on the Rat, Burntwood, Footprint, upper and lower Nelson, Jack, and Aiken rivers. The hydrologic regimes at 4 of the sites were modified by CRD (Rat R. above Threepoint L.; the Burntwood R. at the outlet of Threepoint L., at Thompson, and at the inlet to Split Lake), while 3 sites were affected by LWR (Nelson R. below Sea River Falls and below the confluence with the Grass R.; the Jack R. (west channel) at Norway House). A site on Nelson R. at the outlet to Split Lake was affected by both projects. The remaining three sites were on unregulated rivers (Burntwood R. above Threepoint L., Footprint R. above Footprint L., Aiken R.) to provide a measure of current water quality in natural rivers in the study area and, where sufficient historical data existed, to determine the degree of natural variation in water quality between the pre- and post-development study periods. Characteristics of water quality examined in the FEMP study included the annual mean values of the physical and chemical variables, their range of variation over the 3 years of study, differences between the ice-free and ice-cover seasons, and the relationship, if any, with river discharge. These data were compared with all available data for the pre-development, construction, and immediate post-development periods to determine if significant changes had occurred and if these changes can be attributed to LWR, CRD, or to both projects.

A study of the bacteriological quality of recreational waters in and around the communities of Cross Lake, Norway House, Split Lake, York Landing, and Nelson House also was undertaken to evaluate the current quality of waters for recreational (principally swimming) uses. The study did not examine the effect of the hydroelectric projects on bacterial water quality.

Current Water Quality and Compliance with Canadian Guidelines

Physical and chemical water quality at all 11 sites generally meets the Canadian Water Quality Guidelines for drinking water, recreational uses, and protection of aquatic life although there are a number of exceptions. Mean turbidity exceeds the Canadian drinking water guideline and mean extractable Al and total Cu exceed the aquatic life guidelines at all sites. At the natural river sites and on the upper Nelson River (i.e., above Split Lake), mean turbidity is in the range of 6 to 19 NTU. Turbidity on the CRD between Threepoint and Split lakes is higher, ranging from 20 to 35 NTU, due to shoreline and bank erosion caused by the diversion. Levels of total Pb, Cd, and Zn occasionally exceed the respective aquatic life guidelines in the natural rivers and at sites on the CRD, but never in the Jack River or in the Nelson River above Split Lake.

The natural rivers and sites on the CRD are distinguished by mean extractable Fe exceeding the drinking water and aquatic life guidelines. Mean extractable Fe is below the drinking water and aquatic life guidelines in the Nelson River at sites above Split

Lake and in the Jack River. Both guidelines are exceeded in the Nelson River at the outlet of Split Lake. The high Fe levels in Split Lake are a direct result of CRD.

The consistent occurrence of extractable Al and total Cu concentrations in excess of the aquatic life guidelines, and of extractable Fe levels above both the aquatic life and drinking water guidelines in the natural rivers, suggests a review of the applicability of these guidelines in the study area is in order.

Bacterial Studies

The studies of fecal coliform levels in recreational waters indicate contamination is not a significant concern. No clear evidence of contamination was found at any site, including the vicinity of sewage treatment plant outfalls where contamination might be expected. However, the limitations of the sampling program with respect to the number and frequency of samples prevent conclusion that fecal coliform levels comply with national water quality guidelines.

Changes in Water Quality Related to the Hydroelectric Developments Norway House Area

Impacts of LWR on water quality in the Nelson River below Sea River Falls and in the Jack River could not be assessed due to limited (Sea River Falls) or non-existent (Jack River) pre-development data records.

Nelson House and Threepoint Lake Area

The only adverse effect of CRD on water quality in the Rat River above Threepoint Lake was an approximately 3-fold increase in turbidity between the 1972-1973 pre-development study period and the 1987-1989 FEMP study. This increase was caused primarily by bank and shoreline erosion in the southern part of Notigi Reservoir.

Many other significant changes also occurred, although none can be classified as adverse. Annual mean conductivity, alkalinity (total), hardness (total), TOC, ortho PO_4 , Ca, and reactive Si decreased by 15 to 60%, while Na, K, Cl, and F, increased by 10-80%. These changes were a reflection of the differences in water quality between the Churchill River and the pre-existing Rat River. In most cases, the changes were no greater than the difference between the Rat and Churchill rivers. The only exception was TOC, in which the decrease in concentration was greater than the difference between the rivers. This suggests some loss of TOC is occurring as the Churchill River passes through Southern Indian Lake and Notigi Reservoir. The total P concentration in the Rat River did not change following diversion despite a lower concentration in the Churchill River. Release of P from flooded soils and vegetation probably accounts for this lack of a decrease. Dissolved Fe, Mn, Cu, Pb, and Zn concentrations were not affected by diversion. The impact of diversion on other metal species is uncertain due to the lack of pre-development data.

Impacts of diversion on water quality in the Burntwood River

at the outlet of Threepoint Lake could not be quantified due to the lack of pre-development data for this site.

Burntwood River at Thompson

The only adverse effect of CRD on water quality in the Burntwood River at Thompson was a doubling in turbidity due to bank and shoreline erosion along the Burntwood River below Threepoint Lake. A number of other changes occurred, primarily due to the introduction of softer Churchill River water. These included decreases in conductivity, alkalinity, hardness, reactive Si, dissolved Ca and Mg, dissolved Fe, and extractable Al and Mn. Most changes did not occur until after the autumn of 1977, when the diversion was fully operational, although the initial increase in turbidity and a temporary increase in hardness occurred during CRD construction (June, 1974, to August, 1976), when Rat River flows were cut off at the outlet of Notigi Lake.

Total P concentrations approximately doubled between the construction and immediate post-impoundment periods, probably due to release of P from flooded organic soils and vegetation. Pre-development levels were re-established before 1987 but this does not mean the P release has ended. The annual mean total P concentration in the Burntwood River is still more than twice that in the Churchill River near Leaf Rapids, with most of the P entering the water column as particulates.

Concentrations of dissolved and extractable Al have not been elevated by CRD. A previous report of increased extractable Al

following diversion was the product of the mistaken classification of pre-development dissolved Al data as extractable Al.

Split Lake Area

Burntwood River at Inlet to Split Lake

No quantitative assessment of water quality changes in the Burntwood River at the inlet to Split Lake was possible due to the lack of pre-development data.

Nelson River at Inlet to Split Lake

Several significant changes in water quality were noted in the Nelson River above Split Lake between the 1971-1974 pre-development study period and the FEMP study, but none of these changes could be directly attributed to LWR. The FEMP study was conducted during a period of very low Nelson River flows caused by drought conditions in western Canada. The changes could be due, all or in part, to the lower flows.

Nelson River at Outlet of Split Lake

An increase in extractable Fe, such that the annual mean concentration at the outlet now exceeds by 30% the Canadian Water Quality Guidelines for drinking water and protection of aquatic life, has been the only adverse effect of CRD on water quality in Split Lake. This increase, and significant decreases in TDS, DIC, and dissolved As were inferred on the basis of current differences in concentration between the Burntwood and Nelson rivers and the

relative contribution of these rivers to total inflow. Statistical analysis of pre- and post-development data for 24 variables revealed significant declines had occurred in 10 (pH, conductivity, hardness (total), alkalinity (total), Ca, Mg, K, Na, Cl, and SO₄). All but the decline in pH were a direct result of diversion.

Churchill River diversion has not caused a significant increase in annual mean water turbidity or suspended sediment concentration (NFR) or a decrease in water transparency despite the much higher suspended sediment concentrations in the Burntwood River than in the Nelson River. Much of the suspended sediment load of the Burntwood River appears to settle out near the mouth, preventing a general increase in suspended sediment levels in the lake or downstream. Sediment deposition also appears to have prevented increased concentrations of extractable Al and total Ni and V in Split Lake.

Water Quality - Water Discharge Relationships

The natural rivers, sites on the Nelson River upstream of Split Lake, and the Burntwood River at Thompson prior to diversion were characterized by having many discharge-correlated variables, with negative correlations dominating. Negatively correlated variables generally included the major cations (Na, K, Ca, Mg) and anions (Cl, F, SO₄) and associated variables (conductivity, alkalinity, hardness, TDS), some metals, DIC, reactive Si, and dissolved N species (TDN, NO₂-NO₃, total NH₃). Positive correlates generally included turbidity, NFR, colour, and the organic carbon

species (DOC and TOC).

The discharge relationships were profoundly different at sites affected by CRD. In the Rat River above Threepoint Lake and in the Burntwood River below Threepoint Lake, at Thompson, and at the inlet of Split Lake, all correlations with discharge were positive following diversion. This change is directly attributable to shoreline erosion caused by CRD.

A shift from negative correlations prior to development to positive correlations during the FEMP study also was evident in the Nelson River at the outlet to Split Lake. The role of CRD in this change is unclear due to the unusually low flows on the Nelson River during the period of study.

The water quality - water discharge relationships derived from the FEMP study data offer limited potential for impact prediction or management. The strongest correlations were found at sites where there is the least need for their application; the natural rivers and the Nelson River at Sea River Falls. Most relationships at sites on the CRD were too weak (i.e., $r^2 < 0.25$) to have much predictive ability although there were several important exceptions, including the relationships for NFR and turbidity in the Burntwood River at Thompson and for turbidity and alkalinity in the Burntwood River at the inlet to Split Lake. The weaker discharge relationships at sites on the CRD than at sites off the diversion appear to be due to the relatively less dynamic discharge regime on the CRD.

Mitigation of Adverse Effects

There are few options for mitigation of the adverse impacts of CRD on water quality and considerably more information on the ecological consequences of these impacts is required before any mitigation measure can be recommended. Additional bank stabilization along the Burntwood River might be considered for control of the turbidity problem. Reduction of diversion flows should reduce extractable Fe concentrations in Split Lake, with a 50% reduction in flow required to reach compliance with the current Canadian Water Quality Guidelines.

Recommendations for Further Study

The FEMP study is a major step in documentation of the ecological impacts of the Churchill-Nelson hydroelectric development. This process is by no means complete, however, with a number of outstanding questions remaining. The following activities are recommended to address these questions.

- Investigate the potential for inference of water quality changes in the Jack River and Nelson River below Sea River Falls using pre-development data from upstream sites.
- Examine ecological impacts of elevated turbidity in the Rat-Burntwood system.
- Extend the water quality database for the Burntwood River at Thompson to examine the long-term evolution of water quality on the Churchill River diversion.

- Evaluate the predictive ability of the water quality - water discharge relationships at sites where these relationships appear to have some application.
- Review the applicability of the Canadian Water Quality Guidelines for Fe, Al and Cu to the NFA area and develop site-specific guidelines if necessary.
- Investigate the potential for localized increases in turbidity in Split Lake related to CRD.
- Evaluate the ecological impact of sediment deposition in Split Lake near the mouth of the Burntwood River including the potential for mobilization of associated heavy metals.
- Monitor water quality in Split Lake during periods of higher Nelson River flows for differentiation of water quality changes due to low flows from any changes due to the hydroelectric projects.

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1.0 Introduction

Environment Canada and the Department of Fisheries and Oceans undertook in 1986 a five year program of ecological monitoring in northern Manitoba which has come to be known as the Federal Ecological Monitoring Program (FEMP). The overall purpose of this program is to address Canada's obligations under the Northern Flood Agreement (NFA) with respect to requirements for ecological monitoring and the identification and disclosure of adverse effects of Churchill River diversion (CRD) and Lake Winnipeg regulation (LWR).

The NFA was signed between Canada, Manitoba, Manitoba Hydro, and the Northern Flood Committee (NFC) which represents the 5 Indian Bands affected by the Lake Winnipeg Regulation and the Churchill-Nelson Rivers Diversion Project (LWCNR). The objective of the NFA is to ensure that all persons represented by the NFC who are adversely affected by the project receive fair and equitable treatment.

This report presents the results of the water quality monitoring program implemented by Environment Canada as part of the FEMP. Existing water quality is described, changes in physical and chemical water quality since the LWCNR project began operation are documented and, where possible, the cause of these changes is determined. Any adverse effects directly resulting from changes in water quality caused by the Project are described.

The specific objectives of the study were to:

- 1.) document and assess changes in water quality which have or are continuing to occur since the hydroelectric developments in the study area were completed;
- 2.) document and assess changes in water quality resulting from the developments which have had an adverse effect on the water uses of the five NFA communities;
- 3.) identify water quality conditions which exceed generally accepted national guidelines for identified water uses, but for which there is insufficient information to identify cause, and;
- 4.) develop and recommend a water quality management program which will provide information necessary to document future changes in water quality and for use by water quality managers.

The purpose of these objectives is to fulfil the water quality aspect of Canada's obligations under Clauses 17.5 and 17.1 of the NFA. Clause 17.5 refers to monitoring adverse effects of the project and states "In particular but without limitation, monitoring of adverse effects of the Project pursuant to the Lake Winnipeg, Churchill and Nelson Rivers Study Board recommendations shall be planned and implemented so as to provide such information

as may be necessary to give effect to this agreement". Clause 17.1 states, "Hydro and Canada and Manitoba, severally and jointly, undertake to implement such recommendations of the Lake Winnipeg, Churchill, and Nelson Rivers Study Board Report which affect the Communities and which fall within their respective or joint jurisdictions". One such recommendation is recommendation Number 10 which states "That appropriate government departments and agencies develop and implement a long-term coordinated ecological monitoring and research program to allow impact evaluation and to assist in the ongoing management of the affected area".

Thus, the two primary tasks of the water quality monitoring program were to document existing water quality, as required by Clause 17.1 (recommendation 10), and to identify adverse effects, as required by Clause 17.5.

1.1 Definition of an Adverse Effect

The definition of an adverse effect employed in this study is a change in water quality which has a detrimental effect on one or more of the traditional water uses of the five NFA communities (Seidl 1987). Traditional water uses of the communities which are considered in this report include drinking water, recreation, and aquatic life. Aesthetic impacts also were considered. As suggested by Seidl (1987), the definition of a detrimental effect employed in this report is a significant change in water quality since LWCNR Project construction, which is not attributable to natural causes or to causes other than the hydroelectric project, and which causes

one or more parameters to exceed the Canadian Water Quality Guidelines (CCREM 1987) for drinking water, recreational water use, and protection of aquatic life.

The CCREM guidelines have been established by the Canadian Council of Resource and Environment Ministers to provide basic information about the effects of water quality parameters on uses in order to assess water quality issues and concerns and to establish water quality objectives for specific sites (CCREM 1987). The guidelines were developed following review of existing guidelines from many sources. Those guidelines considered to be appropriate for Canadian environmental conditions were adopted. Those found inappropriate, were either adopted after modification on the basis of scientific information specific to Canadian conditions or, if no data existed on which to base a modification, no guideline was established.

Use of the CCREM guidelines to define detrimental or harmful changes in water quality and, hence, adverse effects represents a balance between the need to prevent harm to biota or water users, while recognizing that most systems have some capacity to absorb or assimilate some change without causing any harm. The CCREM guideline level for a variable is set below the level at which physiological effects have been shown to first appear in the most sensitive species and life stage which have been tested. The guidelines therefore protect all forms of aquatic life and all life stages during indefinite exposure (Williamson 1988).

Use of the CCREM guidelines to define an adverse effect is somewhat limiting in that guidelines have not been established for all variables, the guidelines are not based on site-specific cause-effect studies in northern Manitoba, and they do not take into account the effect of cumulative stresses caused by the simultaneous exceedence of guideline values by two or more water quality variables. Modification of the guidelines to accommodate cumulative effects was beyond the scope of this study. Site-specific information on biological responses to the water quality changes which have occurred at the study sites is largely non-existent, necessitating reliance on the more general guidelines. The documentation of adverse effects of hydroelectric development in northern Manitoba should be considered as an ongoing process; information on the nature of changes in one environmental component (e.g., water quality) often is necessary to focus subsequent studies in other areas (e.g., benthos or fish feeding efficiency). Future studies may reveal adverse effects which are not presently apparent.

1.2 Limitations to the Assessment of Impacts on Water Quality

The scope and findings of any retrospective analysis of changes in water quality, such as that presented here, are dependent upon the nature and extent of pre-development data. While it may be possible in some cases to estimate pre-development concentrations for some variables, most statements of significant change are dependent upon the existence of sufficient pre-

development data, of comparable quality, for the application of a statistical test. Unfortunately, the pre-development water quality database is not complete for any of the sites considered in this report. In particular, the pre-development nutrient and metals databases were incomplete for most sites. Specific data limitations are identified throughout the text.

1.3 Environment Canada FEMP Study Plan

A water quality monitoring program was implemented in late 1986 to document existing water quality. This program was developed in co-operation with the Department of Fisheries and Oceans, Manitoba Natural Resources, and Manitoba Environment to provide the most complete coverage of the NFA area that was possible within the overall program budget. Results of studies by Manitoba Environment are presented in Playle (1986), Playle and Williamson (1986), Duncan and Williamson (1988), and Ralley and Williamson (1990). Data gathered by Manitoba Natural Resources are documented in Green (1990) and are summarized and analyzed by Ramsey et al. (1989) and Berger and Ramsey (1991).

The Environment Canada program included a three year study of physical and chemical water quality at 11 sites on the Rat, Burntwood, Footprint, upper and lower Nelson, Jack, and Aiken rivers to document current conditions (Figure 1). All data collected under this program are documented in Strange (1991). In this report, the FEMP study data are compared with all available pre-development data for the sites to identify statistically

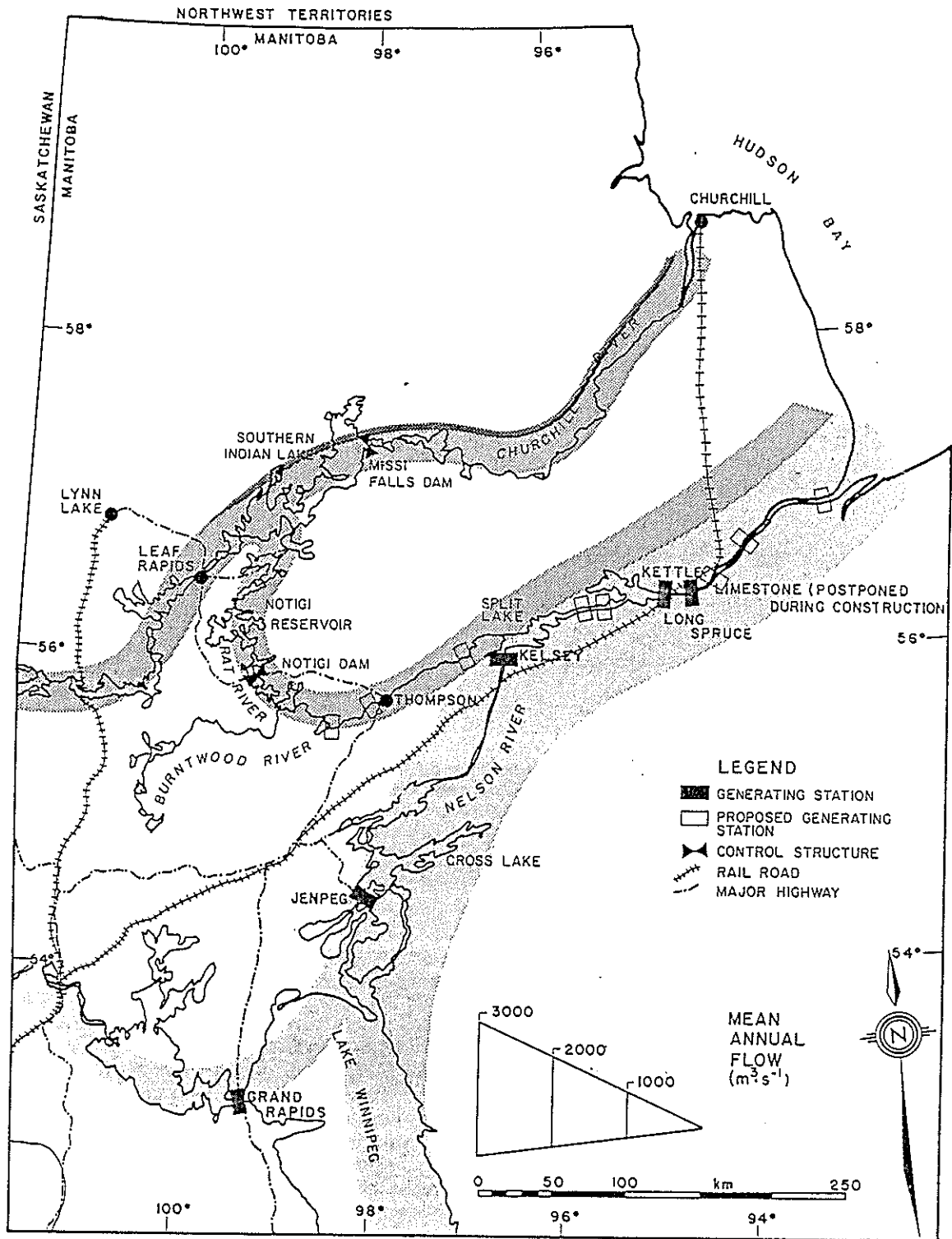


Figure 1. Churchill and Nelson rivers hydroelectric development, indicating the altered flow regime of the rivers. Dark tone indicates relative magnitude of lower Churchill discharge remaining after diversion; mid-tone indicates portion of Churchill River discharge diverted at Southern Indian Lake; light tone indicates Nelson River discharge. From Newbury et al. (1984).

significant changes.

In addition to the study of current physical and chemical water quality, Environment Canada undertook a study of the bacteriological quality of recreational waters in the 5 NFA communities. The bacterial studies were intended specifically for evaluation of the present quality of waters for recreational (principally swimming) uses. The studies were not designed nor intended for evaluation of the effects of the LWCNR Project on bacterial water quality.

1.4 Overview of the Churchill-Nelson Hydroelectric Project

The Churchill and Nelson rivers drain 1.4 million km² of the interior plains of western North America, flowing northeasterly across the Canadian Shield to Hudson Bay. In the mid-1960's, joint Federal-Provincial studies of the feasibility of developing hydroelectric generating capacity on the Churchill and Nelson rivers for markets in southern Canada and the northern United States identified 9 dam sites on the Nelson River (Figure 1). No development of generating stations on the Churchill River was recommended. Instead, diversion of a significant portion of Churchill River flows was proposed to increase flows through the Nelson River dams (Newbury et al. 1984).

A number of diversion schemes were considered but the plan which was implemented in 1976 involved a 3 m rise in the mean level of Southern Indian Lake, diverting 75% (about 835 m³ sec⁻¹) of Churchill River flow to the Nelson River via the Rat and Burntwood

Rivers (Figure 1). Churchill River diversion required the construction of a control dam at Missi Falls, the natural outlet of SIL, the excavation of a short diversion canal between South Bay on SIL and Issett Lake, and the construction of a control structure at the outlet of Notigi Lake which was used to create and now regulates the Notigi Reservoir.

Construction and operation of the diversion have caused dramatic changes in the hydrologic regime of the Rat River. River flow was completely cut off at the Notigi Dam site from June 1974 to August 1976 to facilitate construction of the dam and filling of the Notigi Reservoir (G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB, pers. comm.). The reservoir incorporates several major lakes, flooding in excess of 500 km² of forested land in the Rat River valley. With the beginning of diversion operation in September, 1976, annual mean flows below Notigi Dam increased from approximately 30 m³ sec⁻¹ (Water Resources Branch 1974) to about 865 m³ sec⁻¹ (Manitoba Hydro, unpubl. data).

The Rat River merges with the Burntwood River at Threepoint Lake (Figure 1). Prior to diversion construction, the Rat River contributed approximately 30% of total Burntwood River flows below Threepoint Lake (Water Resources Branch 1974). This contribution accounts for the approximately 30% reduction in Burntwood River flow at Thompson during Notigi Dam construction (Water Survey of Canada 1990).

Diversion produced a substantial increase in mean Burntwood River flows below Threepoint Lake, from an average of about 120 m³

sec⁻¹ between 1958 and 1973, to about 900 m³ sec⁻¹ between 1978 and 1989 (Figure 2). The increase in flow due to diversion developed gradually over a period of about 1 year, reaching full flows in September, 1977.

The increased flows on the lower Rat and Burntwood rivers following diversion produced elevated water levels on the rivers and lakes causing flooding along shorelines, backflooding along stream channels, and incorporation of, and marginal flooding along, several adjacent lakes. Mean levels on Threepoint and Footprint lakes increased approximately 4.5 m. Approximately 99 km² of land was flooded between the Notigi Dam and the inlet to Split Lake (G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB. pers. comm.).

Churchill River diversion caused an approximately 30% increase in mean Nelson River flows below Split Lake, although the relative contribution of the CRD to Nelson River flows varies greatly from year to year. Prior to diversion, mean annual Burntwood River flow at the inlet to Split Lake was approximately 5% of mean Nelson River inflow to the lake (Figure 2). Since diversion in 1976, mean Burntwood River flow has ranged from 29% to 100% of Nelson River flow, with this broad range in relative contribution due primarily to discharge fluctuations on the Nelson River. During the 1987-1989 period of the FEMP study, Burntwood River flows relative to Nelson River flows were the highest since the diversion began operating in 1976. The impact of CRD on water chemistry in Split Lake and sites downstream is dependent on the relative contribution of CRD to

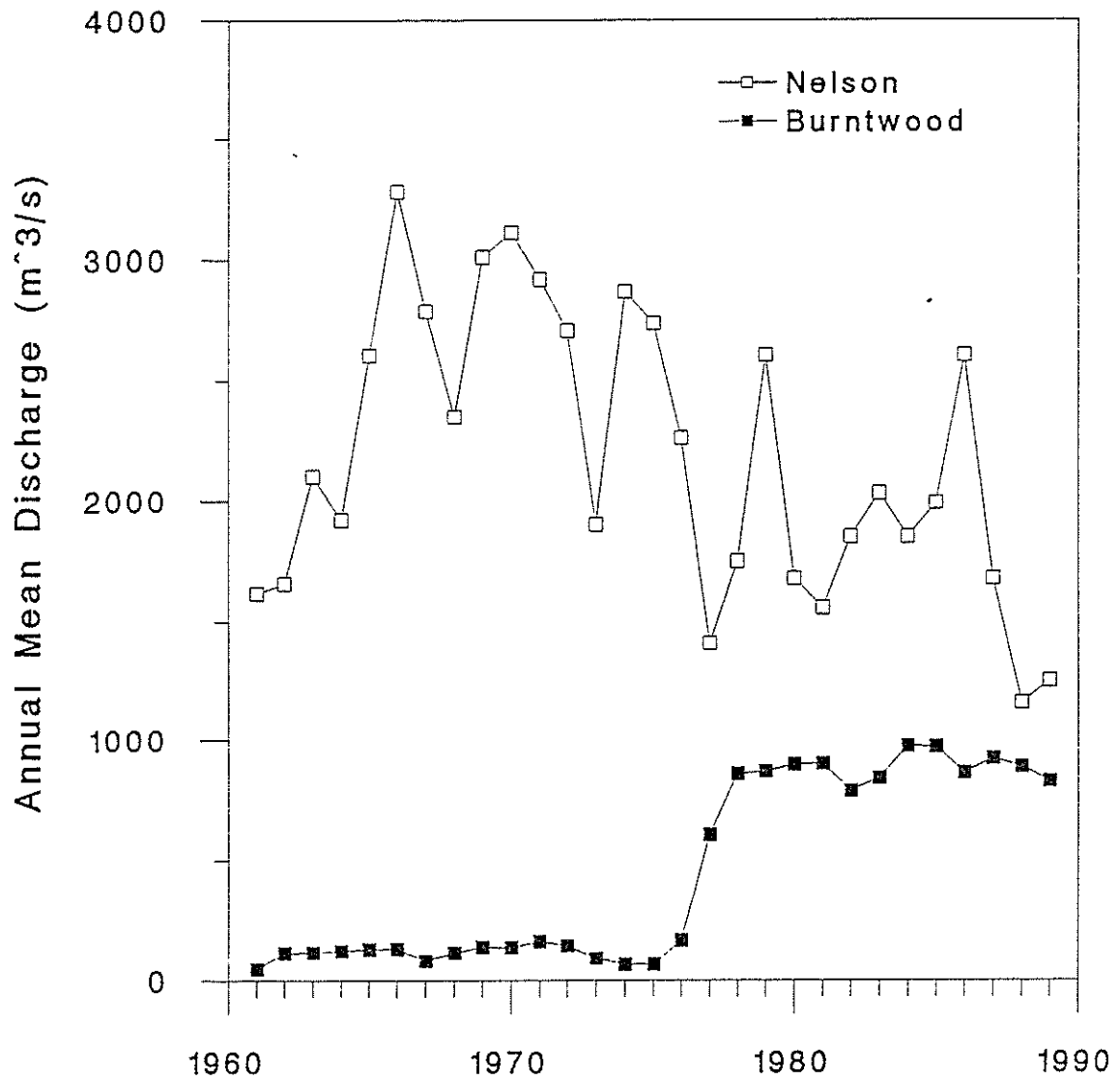


Figure 2. Historical annual mean discharge in the Nelson River at Kelsey Generating Station and in the Burntwood River at Thompson, 1961-1989. Data from Water Survey of Canada (1990).

Nelson River flows (Section 3.5.4). Thus, the changes in Split Lake water quality which are documented in this report probably are more severe than any which occurred in the first 10 years following diversion.

Another significant component of the Churchill-Nelson project is the regulation of levels on 24,400 km² Lake Winnipeg by the Jenpeg Dam, located on the west channel of the Nelson River above Cross Lake, for dependable mid-winter flows (Figure 1; Newbury et al. 1984). The Lake Winnipeg regulation component of the project also involved alteration of the water flow pattern in southern Playgreen Lake, due to the construction of Two Mile Channel, and in northern Playgreen Lake by Eight Mile Channel, which now diverts water through Kiskittogisu Lake. The dam maintains water levels on Lake Winnipeg and Playgreen and Kiskittogisu lakes near the high end of the natural range by increasing the mean level and decreasing the range of fluctuation. Backflooding is restricted to lower Kiskittogisu Lake and the downstream reach above the dam. Little Playgreen Lake and the East Channel of the Nelson River are probably subject to slightly higher than natural flows and levels due to increased average Playgreen Lake levels at the outlets into Little Playgreen Lake (G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB. pers. comm.).

2.0 Methods

2.1 Physical and Chemical Water Quality

2.1.1 FEMP Sampling Locations

The physical and chemical water quality sampling program concentrated on three major areas; the Nelson House area on the CRD between Threepoint Lake and Thompson, the major inlets and outlets of Split Lake, and the Norway House area (Figure 3).

In the Threepoint Lake area (Figure 4), stations were situated on the Burntwood River, above the confluence with the Wimapedi River, on the Rat River immediately above Threepoint Lake, on the Footprint River above Footprint Lake, at the outlet of Threepoint Lake, and on the Burntwood River at Thompson, approximately mid-way between Threepoint and Split lakes. In the Split Lake area (Figure 5), stations were situated on the Nelson River above Split Lake, about 1 km below the confluence with the Grass River; on the Burntwood River above Split Lake, about 1 km below the confluence with the Odei River; at the mouth of the Aiken River; and, at the outlet of Split Lake. In the Norway House area (Figure 6), samples were collected from the Jack River at Norway House and from the Nelson River below Sea River Falls.

The sites on the Burntwood River above Threepoint Lake, the Footprint River, and the Aiken River have not been affected by the hydroelectric developments and were included to characterize and provide a measure of the natural variation in water quality on these smaller regional rivers.

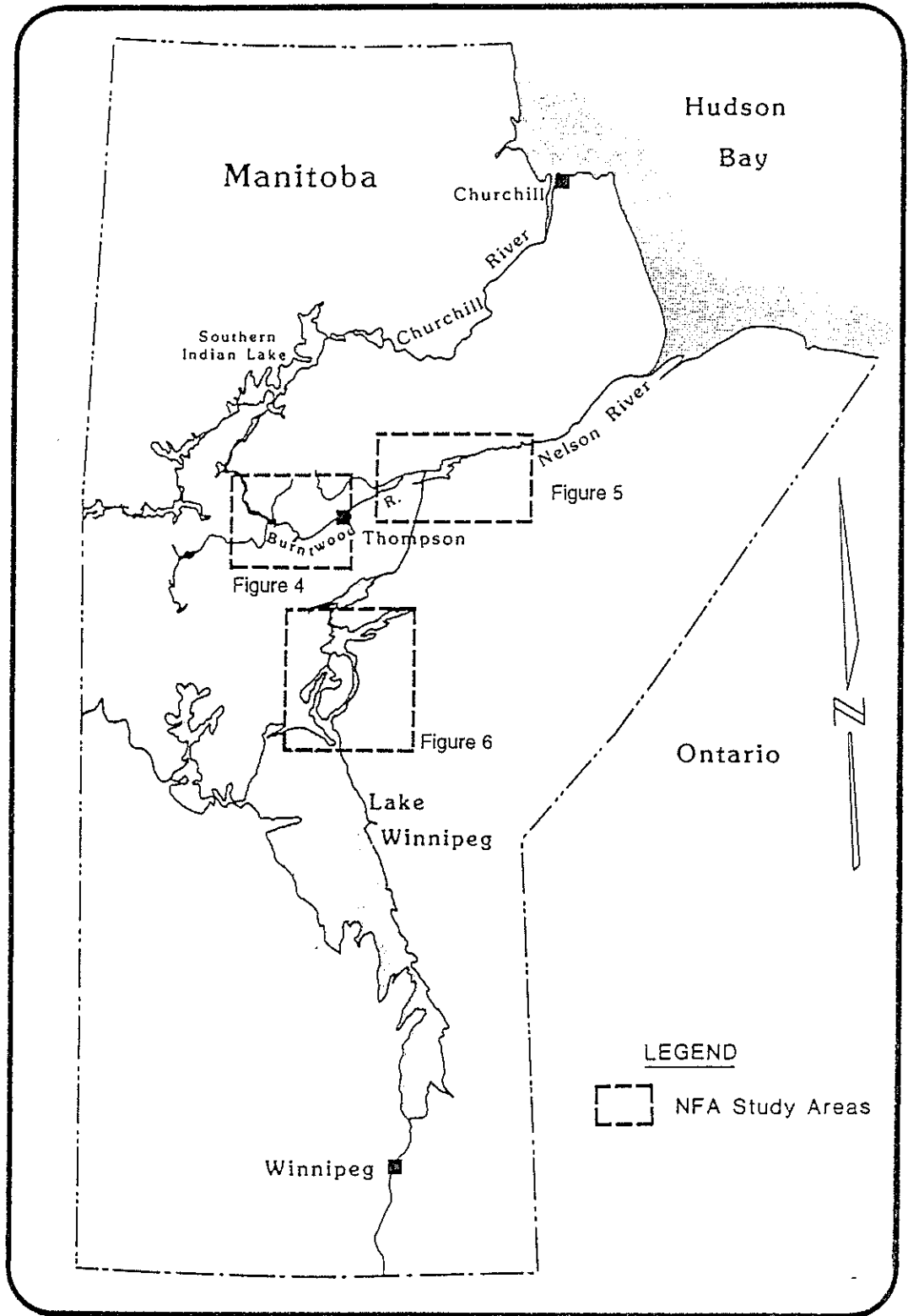


Figure 3.

Location of FEMP water quality study areas. See Figures 4, 5, and 6 for location of FEMP and pre-development sampling locations.

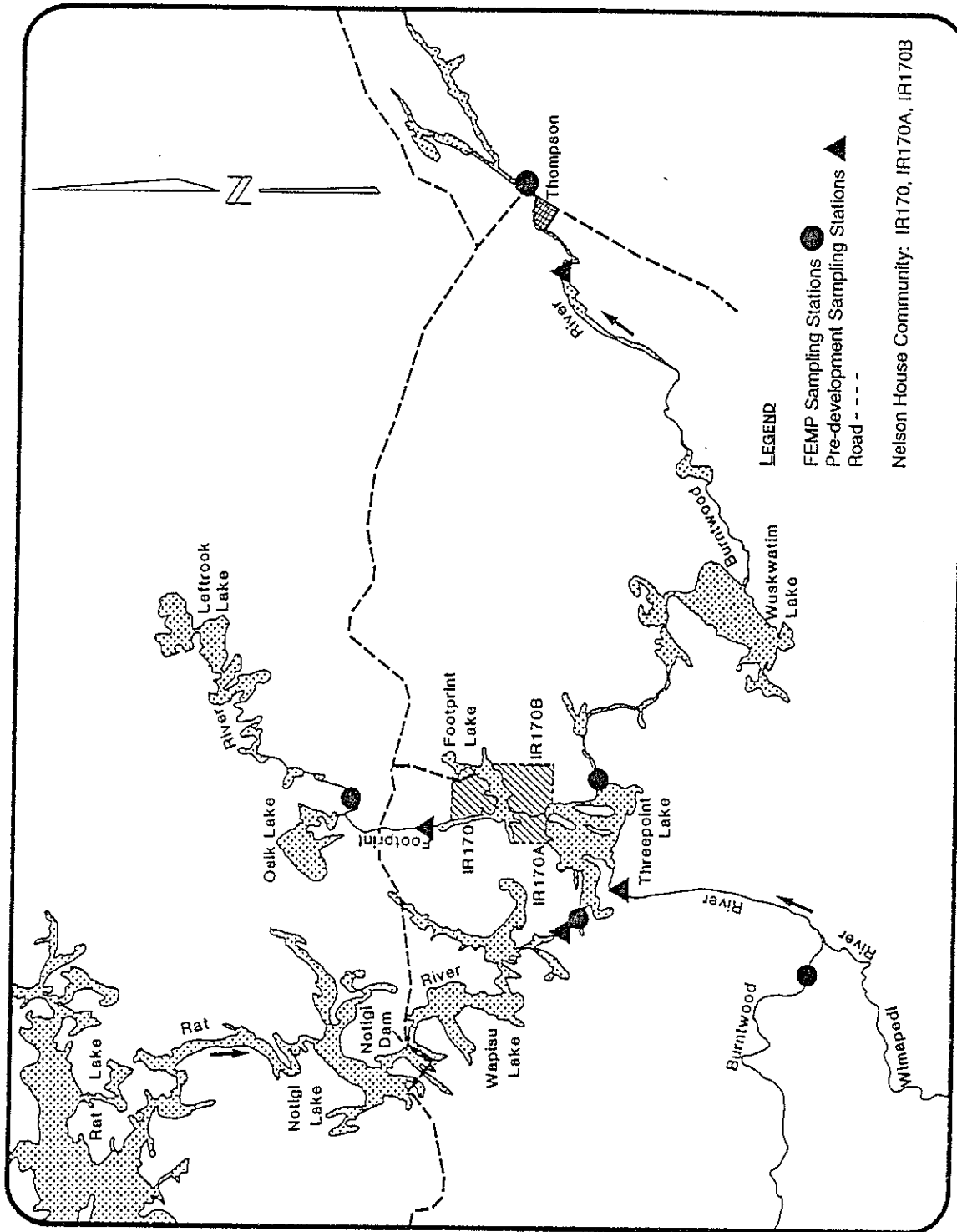


Figure 4. Location of FEMP and pre-development water quality sampling stations in the Nelson House study area.

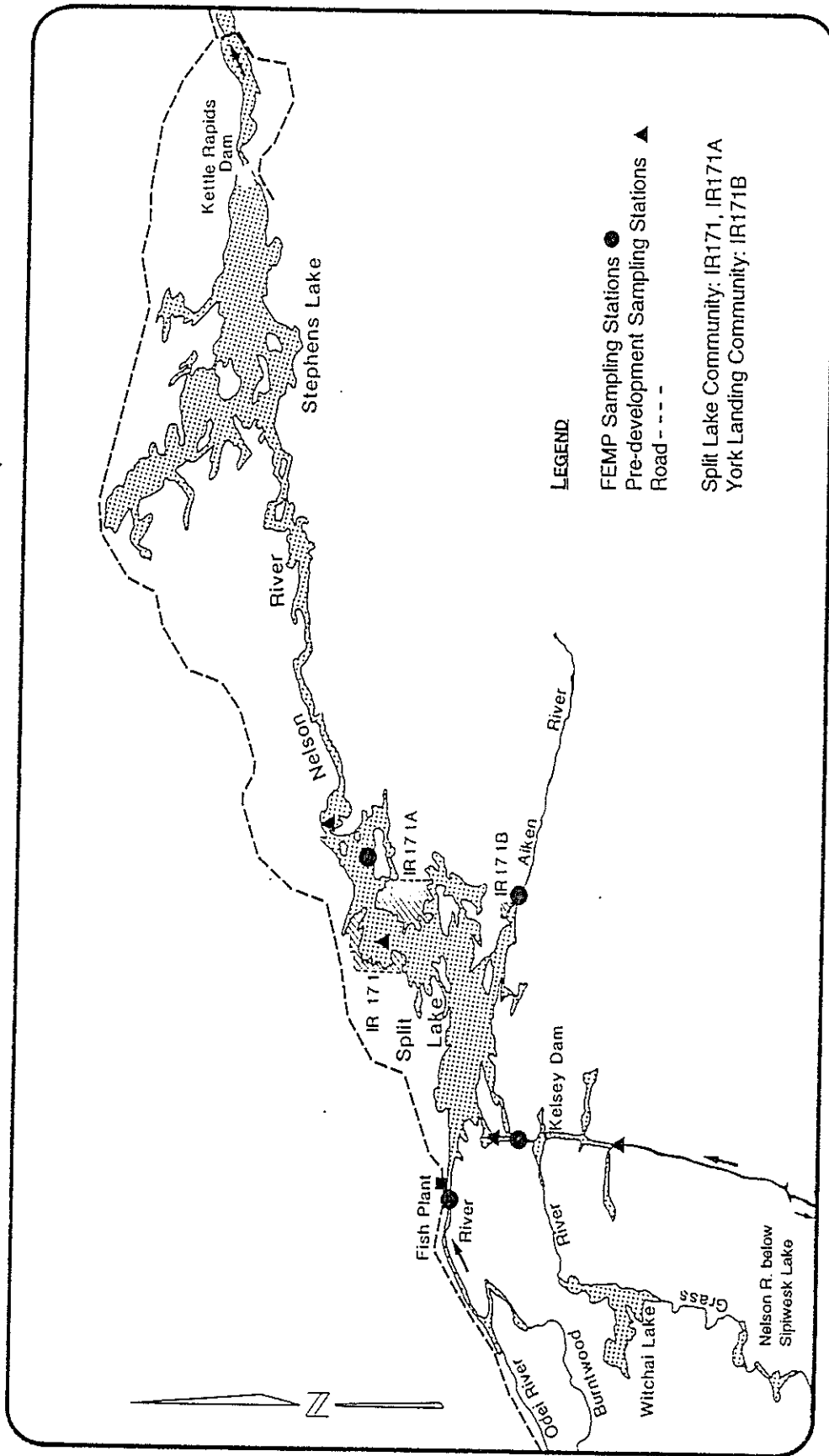


Figure 5. Location of FEMP and pre-development water quality sampling stations in the Split Lake study area.

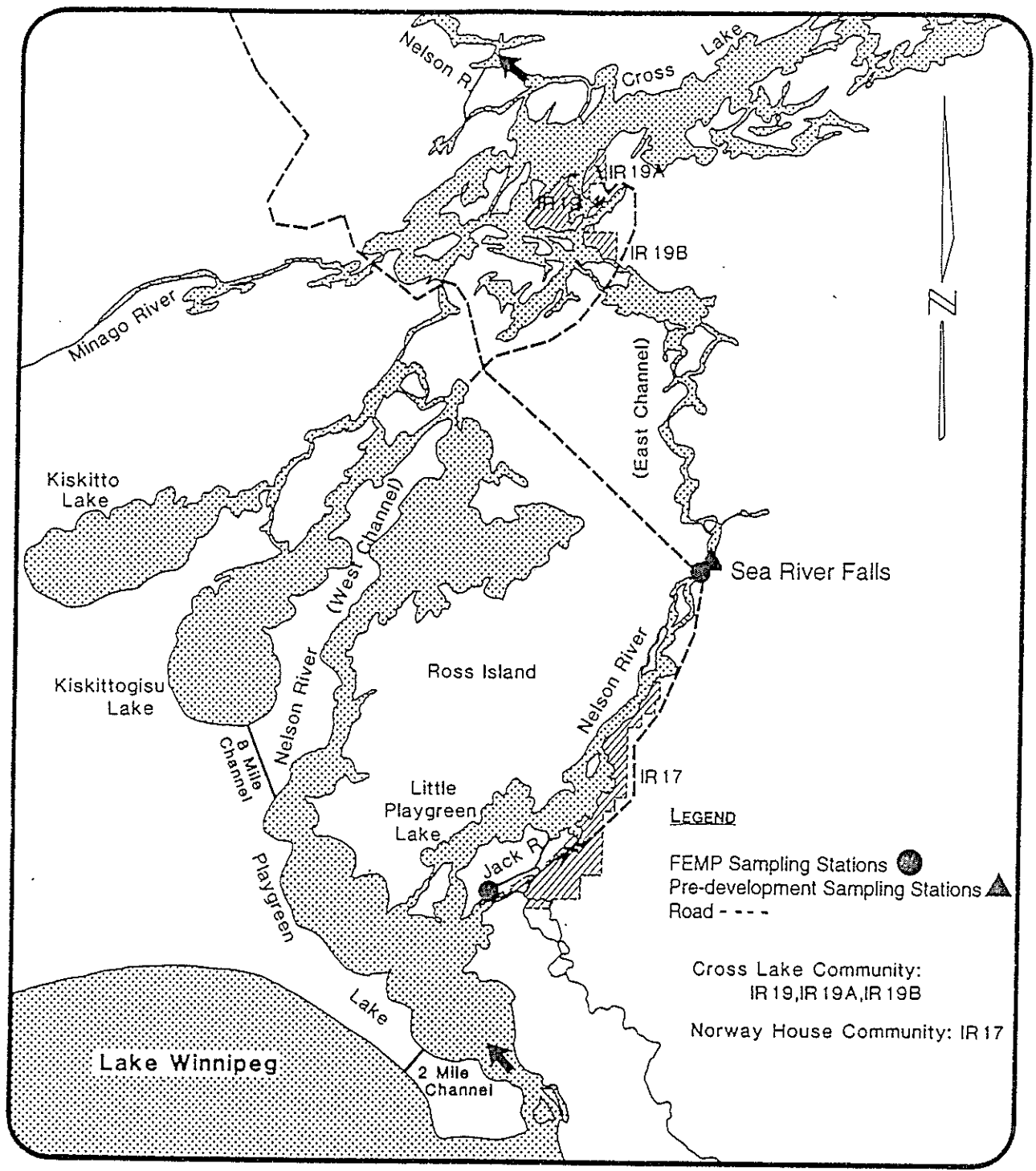


Figure 6. Location of the FEMP and pre-development water quality sampling stations in the Norway House study area.

All sites were sampled monthly except during freeze-up in November and break-up in April. Nine of the 11 sites were sampled twice monthly from June to September in 1987 and 1988. Although samples were collected from 5 of the sites in September, 1986, routine sampling did not begin at most sites until January, 1987. None of the 1986 data was included in the analysis. Sampling at most sites continued until September-October, 1989. Exceptions to this sampling plan were the sites on the Burntwood River above Threepoint Lake and Footprint River above Footprint Lake where sampling began in May 1987 and continued to March 1989. Details of the sampling procedures may be found in Strange (1991). The variables measured in this program, and the corresponding NAQUADAT analysis codes are listed in Table 1.

2.1.2 Historical Data Sources and Data Screening

Pre-development data for the sites were obtained from a number of sources (Table 2). All data, including those gathered in the FEMP study, were screened for inclusion in the analysis of differences between the pre- and post-development periods to ensure that any significant differences were not the product of differences in analytical methods or errors in analyses or data reporting, and that sufficient data were available for application of the statistical test. Data which were included in the analysis had the following characteristics:

Table 1. Description of water quality variables and abbreviations listed in data tables, including the NAQUADAT analysis codes for the FEMP study data.

Abbreviation	Variables (units)	NAQUADAT Code
pH (F)	pH (Field)	10301F
pH (L)	pH (Lab)	10301L
Spec Cond (F)	Specific Conductance at 25°C (Field) ($\mu\text{S cm}^{-1}$)	02041F
Spec Cond (L)	Specific Conductance at 25°C (Lab) ($\mu\text{S cm}^{-1}$)	02041L
Turbidity (F)	Turbidity (Field) (NTU)	02073F
Turbidity (L)	Turbidity (Lab) (NTU)	02081L
Colour	True Colour (Relative units)	02021L
TDS	Total Dissolved Solids (mg L^{-1})	00201L
NFR	Non-filterable Residue (mg L^{-1})	10401L
NFR Fixed	Non-filterable, Fixed Residue (mg L^{-1})	10501L
Tot Alk	Total Alkalinity, as CaCO_3 (mg L^{-1})	10111L
Bicarb	Bicarbonate Alkalinity (mg L^{-1})	06201L
Tot Hard	Total Hardness, as CaCO_3 (mg L^{-1})	10602L
Non-carb Hard	Non-carbonate Hardness (mg L^{-1})	10650L
Free CO_2	Free Carbon Dioxide (mg L^{-1})	06401L
Part P	Particulate Phosphorus (mg L^{-1})	15901L
Ortho P	Ortho Phosphorus (mg L^{-1})	15256D
TDP	Total Dissolved Phosphorus (mg L^{-1})	15103D
Tot P	Total Phosphorus (mg L^{-1})	15406L
Part N	Particulate Nitrogen (mg L^{-1})	07901L
TDN	Total Dissolved Nitrogen (mg L^{-1})	07651L
Diss NO_2NO_3	Dissolved Nitrite and Nitrate (mg L^{-1})	07110D
Tot NH_3	Total Ammonia, as N (mg L^{-1})	07540P
Tot N	Total Nitrogen (mg L^{-1})	07603L
DOC	Dissolved Organic Carbon (mg L^{-1})	06104L
POC	Particulate Organic Carbon (mg L^{-1})	06901L
TOC	Total Organic Carbon (mg L^{-1})	06002L
DIC	Dissolved Inorganic Carbon (mg L^{-1})	06181L
Chl a	Chlorophyll a (mg L^{-1})	06717G
Diss Ca	Dissolved Calcium (mg L^{-1})	20103L
Diss Mg	Dissolved Magnesium (mg L^{-1})	12102L

Table 1. continued.

Abbreviation	Parameters	NAQUADAT Codes
Diss K	Dissolved Potassium (mg L ⁻¹)	19103L
Diss Na	Dissolved Sodium (mg L ⁻¹)	11103L
React Si	Reactive Silica, as SiO ₂ (mg L ⁻¹)	14102L
Diss Cl	Dissolved Chloride (mg L ⁻¹)	17206L
Diss F	Dissolved Fluoride (mg L ⁻¹)	09106L
Diss SO ₄	Dissolved Sulphate (mg L ⁻¹)	16306L
Extr Al	Extractable Aluminum (mg L ⁻¹)	13302P
Diss Al	Dissolved Aluminum (mg L ⁻¹)	13102G
Diss As	Dissolved Arsenic (mg L ⁻¹)	33108D
Tot Ba	Total Barium (mg L ⁻¹)	56001P
Tot Cd	Total Cadmium (mg L ⁻¹)	48002P
Diss Cd	Dissolved Cadmium (mg L ⁻¹)	48102G
Tot Co	Total Cobalt (mg L ⁻¹)	27002P
Tot Cu	Total Copper (mg L ⁻¹)	29005P
Diss Cu	Dissolved Copper (mg L ⁻¹)	29105G
Extr Fe	Extractable Iron (mg L ⁻¹)	26304P
Diss Fe	Dissolved Iron (mg L ⁻¹)	26104G
Tot Pb	Total Lead (mg L ⁻¹)	82002P
Diss Pb	Dissolved Lead (mg L ⁻¹)	82103G
Extr Mn	Extractable Manganese (mg L ⁻¹)	25304P
Diss Mn	Dissolved Manganese (mg L ⁻¹)	25104G
Tot Hg	Total Mercury (mg L ⁻¹)	80011P
Tot Ni	Total Nickel (mg L ⁻¹)	28002P
Diss Ni	Dissolved Nickel (mg L ⁻¹)	28102G
Diss Se	Dissolved Selenium (mg L ⁻¹)	34108D
Tot V	Total Vanadium (mg L ⁻¹)	23002P
Diss V	Dissolved Vanadium (mg L ⁻¹)	23102G
Tot Zn	Total Zinc (mg L ⁻¹)	30005P
Diss Zn	Dissolved Zinc (mg L ⁻¹)	30105G

Table 2. Sources of pre-development data used in the analysis of water quality changes.

Site	Source Station Name (No.)	Period of Record Used	Variables Used
Churchill R. near Leaf Rapids	NAQUADAT MA06EB0002	July 1972 to August 1973	pH (L), Spec Cond (L), Turbidity (L), Tot Alk, Tot Hard, TOC, Diss Na, Diss K, React Si
Rat R. above Threepoint L.	NAQUADAT MA05TF0001	March 1972 to October 1973	pH (L), Spec Cond (L), Turbidity (L), Tot Alk, Tot Hard, TOC, NO ₂ NO ₃ , Ortho P, Tot P, Diss Na, Diss K, Diss Ca, Diss Cl, Diss F, Diss SO ₄ , React Si, Diss Fe, Diss Mn, Diss Cu, Diss Pb, Diss Zn
Footprint R. above Footprint L.	NAQUADAT MA05TF0003	March 1972 to December 1973	pH (L), Spec Cond (L), Turbidity (L), Tot Alk, Tot Hard, TOC, NO ₂ NO ₃ , Ortho P, Tot P, Diss Na, Diss K, Diss Ca, Diss Cl, Diss F, Diss SO ₄ , React Si, Diss Fe, Diss Mn, Diss Cu, Diss Pb, Diss Zn
Burntwood R. above Threepoint L.	NAQUADAT MA05TF0002	March 1972 to December 1973	pH (L), Spec Cond (L), Turbidity (L), Tot Alk, Tot Hard, TOC, NO ₂ NO ₃ , Ortho P, Tot P, Diss Na, Diss K, Diss Ca, Diss Cl, Diss F, Diss SO ₄ , React Si, Diss Fe, Diss Mn, Diss Cu, Diss Pb, Diss Zn
Burntwood R. at Thompson	NAQUADAT MA05TG0001	February 1967 to December 1977	pH (L), Spec Cond (L), Turbidity (L), NFR, Tot Alk, Tot Hard, Tot P, Diss NO ₂ NO ₃ , TOC, Diss Ca, Diss Mg, Diss K, Diss Na, React Si, Diss Cl, Diss F, Diss SO ₄ , Extr Al, Extr Fe, Diss Fe, Extr Mn
	Morelli (1974) Burntwood R. at Thompson		Tot P, Diss Ca, Diss Cl, Diss SO ₄
Nelson R. near outlet to Split L.	NAQUADAT* Split L. at Split L.	July 1972 to October 1973	pH (L), Spec Cond (L), Turbidity (L), Colour, NFR, NFR Fixed, Tot Alk, Tot Hard, Diss Ca, Diss K, Diss Na, React Si, Diss Cl, Diss F, Ortho P, Tot P, Diss NO ₂ NO ₃ , TOC, Diss Cu, Diss Fe, Diss Pb, Diss Mn, Diss Zn
	Cleugh (1974) and Hecky & Harper (1974) Split L. at Outlet	June 1972 to September 1973	pH (L), Spec Cond (F), TDS, NFR, TDP, Susp P, TDN, Susp N, DOC, Susp C, Chla a, Diss Ca, Diss Mg, Diss K, Diss Na, React Si, Diss Cl, Diss SO ₄ , Diss Fe, Diss Mn
Nelson R. near inlet to Split L.	Cleugh (1974) Nelson R. below Sipiweski L.	February to December 1973	pH (L), Spec Cond (F), TDS, NFR, TDP, Susp P, TDN, Susp N, DOC, Susp C, Chla a, Diss Ca, Diss Mg, Diss K, Diss Na, React Si, Diss Cl, Diss SO ₄ , Diss Fe, Diss Mn
	Cleugh (1974) Nelson R. above Kelsey Dam	May to October 1972	pH (L), Spec Cond (F), TDS, NFR, TDP, Susp P, TDN, Susp N, DOC, Susp C, Chla a, Diss Ca, Diss Mg, Diss K, Diss Na, React Si, Diss Cl, Diss SO ₄ , Diss Fe, Diss Mn
	Cleugh (1974) Nelson R. below Kelsey Dam	May 1972 to September 1973	pH (L), Spec Cond (F), TDS, NFR, TDP, Susp P, TDN, Susp N, DOC, Susp C, Chla a, Diss Ca, Diss Mg, Diss K, Diss Na, React Si, Diss Cl, Diss SO ₄ , Diss Fe, Diss Mn
Nelson R. below Sea River Falls	NAQUADAT MA05UB0006	June 1971 to May 1972	pH (L), Spec Cond (L), Turbidity (L), NFR, NFR Fixed, Tot Alk, Tot Hard, Diss NO ₂ NO ₃ , TOC, Diss K, Diss Na, React Si, Diss Cu, Diss Pb, Diss Zn

* Contains data from Morelli (1974)

- were the product of comparable analytical methods in the pre-and post-development periods;
- included measurements in at least 2 calendar years (either at the same site or at closely located replicate sites) before and 2 calendar years after development with at least 3 measurements in each year and with samples in both the ice-free and ice-cover seasons (ensures enough data to screen outlying values);
- no more than 20% of the measurements in a period were below the analytical limit of detection (avoids use of mean values which are below detection); and,
- all outlying values (i.e., unusually high or low relative to other measurements for the period) which could not be confirmed as valid were removed.

Based on these criteria, the minimum dataset which could be included in the analysis consisted of 3 samples (either 2 ice-free and 1 ice-cover or vice-versa). In practice, the historical datasets included in the analysis generally consisted of 5 to 9 samples with 4 to 6 of the samples during the ice-free season.

In cases where data used were below the analytical limit of detection, they were assigned the value of 50% of the limit of detection following the convention employed by Playle and

Williamson (1988).

The comparability of analytical methods was assessed in a number of ways. For the data from the NAQUADAT database, this comparison was relatively straightforward as the analytical methods were identified as parameter codes in the data records. Measurements for which the parameter codes have not changed were considered to be directly comparable. The consequence of any change in parameter code during the period of record on data comparability was investigated further with officials of Environment Canada. If there was any doubt regarding comparability for any parameter, that parameter was excluded from consideration. The analyses of changes in water quality in the Churchill River above Leaf Rapids and in the Footprint, Rat, and Burntwood rivers above Threepoint Lake relied almost exclusively on NAQUADAT data. This provided some assurance that any significant changes detected in the analysis were not due to a change in analytical method. Where possible, this approach was verified by examining temporal changes in the natural rivers. This analysis (see Section 3.1.5) revealed that dramatic changes in SO_4 and NO_2 - NO_3 concentrations occurred in the natural rivers between the pre-development and FEMP study periods. In the absence of any identifiable cause, this suggested there have been some undocumented changes in sampling or analytical methods or in analytical laboratories.

The analyses of changes in water quality in the Split Lake area relied on data from a number of sources including NAQUADAT and the LWCNR reports (Cleugh 1974; Hecky and Harper 1974). The

analytical methods used in these studies were reviewed and compared with those employed in the current study and found to use the same basic procedures (i.e., temperature correction of pH and conductivity values; major ions and dissolved metals by atomic absorption). In the absence of parallel measurements it is not possible to declare the methods identical, but our investigation indicated the data should be suitable for detection of changes of the magnitude to which the system has been subjected (i.e., ± 25 -50% of baseline values).

Although total Hg concentrations were measured in water samples from all 11 sites, these data were not included in the analysis due to inadequacies in the sampling and analytical methods. The detection limits of the method ranged from 0.01 to 0.02 $\mu\text{g L}^{-1}$ over the course of the study, a level which has been shown to be inadequate for the ambient concentrations. Actual levels on the CRD and Nelson River range between 0.001 and 0.003 $\mu\text{g L}^{-1}$ (Ramsey 1990). The sampling procedures employed in the FEMP study also were not adequate for the prevention of sample contamination, as indicated by the occurrence of measurable concentrations at a detection limit which was a factor of 10 higher than actual levels.

2.2 Statistical Analysis

2.2.1 Water Quality - Water Discharge Relationships

At riverine sites, there often is a correlation between the measured value of physical and chemical water quality variables and

river discharge as has been found at a number of sites in northern Manitoba (Duncan and Williamson 1988). Examination of these relationships can provide general information on the nature of impacts related to modification of river flow regimes.

A negative slope, the result of decreasing concentrations with increasing discharge, indicates the primary effect of higher flows is dilution of the substance in question. In this circumstance, the increased flows either do not increase the supply of a substance to the watercourse at all, or that supply does not increase as quickly as flow. Negative slopes for dissolved constituents are characteristic of streams in which baseflow is maintained by sub-surface drainage through the watershed (Hynes 1970). Sub-surface runoff percolates through the soils, picking up solutes along the way. Thus, water quality in receiving streams is a reflection of soil and bedrock composition in the watershed and integrates local variations in chemical composition. Surface flows predominate during storm events or spring run-off. The run-off water is in contact with the soils for a shorter period, resulting in lower solute concentrations. This produces decreasing chemical concentrations with increasing discharge and the corresponding negative correlation.

A positive slope, the product of increasing concentrations with increasing flow, indicates the primary effect of higher flows is to concentrate the substance in question. Particulate constituents (e.g., NFR, turbidity, POC) are often positively correlated with discharge. This can occur if higher flows cause the

introduction of the substance to the water column through bank and shoreline erosion or through surficial erosion within the watershed.

Relationships with discharge also can be used to reduce the magnitude of variation among samples in a period of interest, permitting a greater sensitivity for the detection of significant differences between periods. Knowledge of the relationship with discharge can also be used as an impact prediction and management tool to determine the impact of proposed changes in the discharge regimes of control and generation structures as required by the addition of generating capacity to the Churchill-Nelson project.

The individual water quality parameters were regressed on daily discharge for each period for each site. A linear model was selected as there was no reason to consider a higher order polynomial. In a similar analysis which included several of the sites examined in the present study, Duncan and Williamson (1988) found a 2nd or 3rd order polynomial did not significantly improve the fit of the models.

The data record for each site was stratified according to the sequence of hydroelectric development. These periods are indicated in Table 3. The analysis was performed for each period of the data record to examine changes in the relationship which might be related to the hydroelectric project. If data were collected from more than one location during a period for a given site, as indicated in Table 2, the regression analysis was conducted separately for each location. A correlation was considered

Table 3. Stratification of historical water quality data records by phase of construction. Dates of construction start and completion from G. McCullough, DFO, Winnipeg.

Site	Period Name	Period of Record
Burntwood R. above Threepoint L.	Pre-development	March 1972 to December 1973
Rat R. above Threepoint L.	Pre-development	March 1972 to December 1973
Footprint R. above Footprint L.	Pre-development	March 1972 to December 1973
Nelson R. below Sea River Falls	Pre-development	June 1971 to May 1972
Nelson R. at the inlet to Split L.	Pre-development	May 1972 to September 1973
Nelson R. at the outlet to Split L.	Pre-development	July 1972 to October 1973
Burntwood R. at Thompson	Pre-development	February 1967 to May 1974
	Construction	June 1974 to August 1976
	Immediate Post-diversion	September 1976 to December 1977

significant at $p \leq 0.05$. Sources of discharge data which were used in the analysis are listed in Table 4.

Variables which were significantly correlated with discharge during a period at a given site were standardized to the mean discharge for that period at that site as follows:

$$^{std}[Y]_i = [\hat{Y}]_{ref} - ([\hat{Y}]_i - [Y]_i)$$

where, for a given site:

$^{std}[Y]_i$ is the concentration of Y in the i^{th} sample standardized to mean discharge;

$[Y]_i$ is the observed concentration of Y in the i^{th} sample;

$[\hat{Y}]_i$ is the predicted concentration of Y at the observed mean daily discharge (Q) on the day of the i^{th} sample; and,

$[\hat{Y}]_{ref}$ is the predicted concentration of Y at the mean Q for the period calculated as:

$$[\hat{Y}]_{ref} = m \cdot Q_{ref} + b$$

where:

Q_{ref} is the mean discharge for the period at the site; and,

m and b are the slope and intercept, respectively, of the regression relationship between Y and Q.

This discharge-standardization procedure is depicted graphically in Figure 7.

Standardization to discharge eliminates the variance in Y accounted for by the relationship with Q, but has no effect on

Table 4. Sources of discharge data used in the water quality/water discharge analysis.

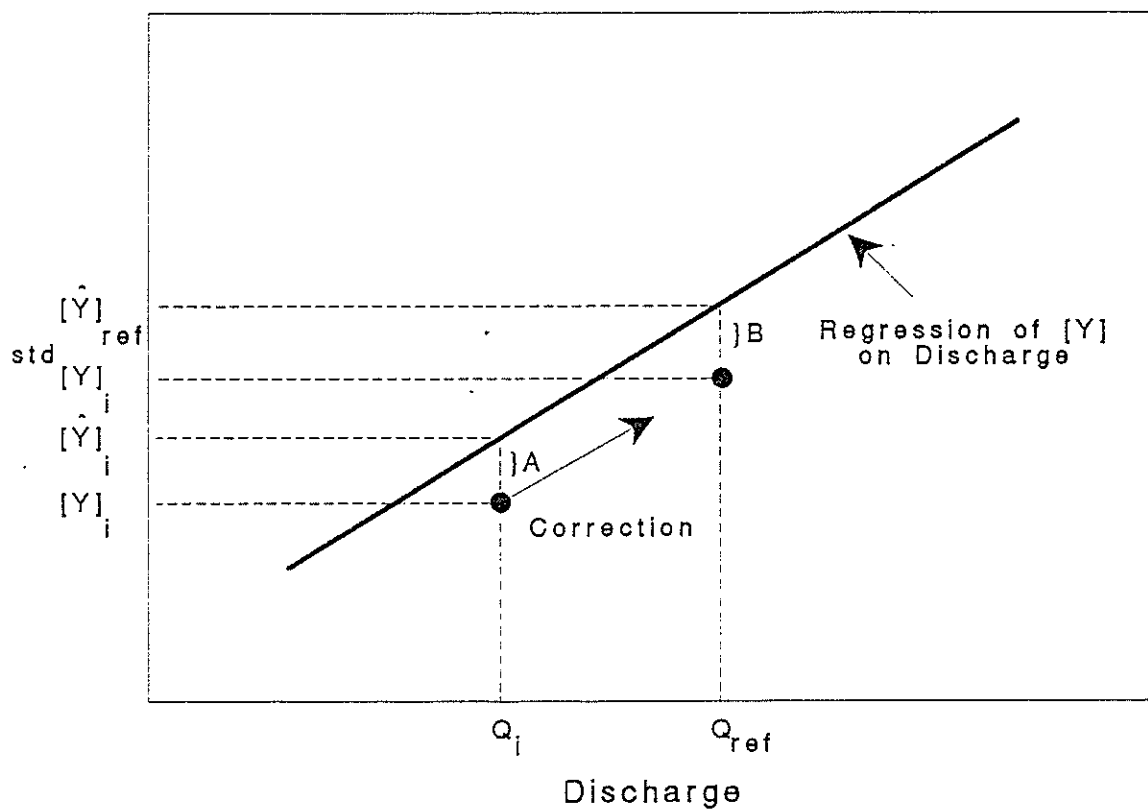
Site	Information Source	Sites Used (W.S.C. codes)	
		Measured Discharge	Calculated Discharge
Jack R. (W. Channel) above Norway House	No Data		
Nelson R. below Sea River Falls	W.S.C.	05UB008	
Burntwood R. near Thompson	W.S.C.	05TG001	
Burntwood R. above Threepoint L.	W.S.C.	05TE002	
Rat R. above Threepoint L.	Man. Hydro	05TF003	
Footprint R. above Footprint L.	W.S.C.	05TF002	
Burntwood R. at outlet of Threepoint L.	Man. Hydro/W.S.C.		05TE002 + 05TF003 + 05TF002
Burntwood R. at inlet to Split L.	W.S.C.		05TG001 + 05TG003 ¹
Nelson R. below confluence with Grass R.	Man. Hydro/W.S.C.		05UE005 ² + 05TD001 ³
Aiken R. at inlet to Split L.	No Data		
Nelson R. at outlet of Split L.	Man. Hydro/W.S.C.	05UF006	

W.S.C. Water Survey of Canada (1990)

¹ Odei R. near Thompson

² Nelson R. at Kelsey Generating Station

³ Grass R. above Standing Stone Falls



Where: $A = [\hat{Y}]_i - [Y]_i$

$B = [\hat{Y}]_{ref} - {}^{std}[Y]_i$

Let: $A = B$

Therefore: ${}^{std}[Y]_i = [\hat{Y}]_{ref} - ([\hat{Y}]_i - [Y]_i)$

Figure 7. Graphical representation of the procedure used to standardize chemistry data to discharge. See text for further definition of terms. Figure modified from Jones et al. (1985).

other sources of variance in Y, thereby reducing the total variance in Y. Reduction of the total variance in Y can improve the sensitivity of an analysis of change in Y over time because the smaller the variance in Y, the smaller the change necessary for it to be statistically significant (Snedecor and Cochran 1980). This approach is commonly used in the analysis of temporal trends in mercury levels in fish tissues in order to remove the influence of covariates such as fish size or age (Ramsey 1991).

Discharge-standardized values, when a significant correlation with discharge indicated they could be calculated, were used in the analysis of changes in water quality between the pre- and post-development periods. The use of standardized values has been indicated in all tables. None of the figures contained in this report presents standardized values.

2.2.2 Analysis of Changes in Water Quality

For the Burntwood, Rat, and Footprint rivers above Threepoint Lake, the Nelson River at the inlet to Split Lake, and the outlet of Split Lake, the water quality data records amount to a short period immediately before the start of construction (1972-1973) and the data collected in the FEMP study (Table 3). The significance of changes in water quality at these sites was examined using Student's t-test (Snedecor and Cochran 1980). Replicates were the annual mean values for each variable. In most cases, successive years of data from the same source (e.g., NAQUADAT; Table 2) comprised the replicates for a period. However, replicates for the

pre-development period both at the Nelson River inlet to Split Lake and in the vicinity of the Split Lake outlet were comprised of data from several sources (Table 2) in order to provide a sufficient number of replicates for analysis. In those instances where data from more than one location were used, the annual mean values for each location were treated as individual replicates.

The data record for the Burntwood River at Thompson is more extensive and was stratified into four periods (Table 3). The significance of changes in water quality at this site was examined using one-way analysis of variance (ANOVA; Snedecor and Cochran 1980) followed by the Tukey/HSD multiple comparison test on all possible pairwise comparisons (Wilkinson 1988).

The use of discharge-standardized values in the analysis of temporal changes is comparable to the use of analysis of covariance (Snedecor and Cochran 1980), with discharge as the covariate. Covariance analysis could have been used in this study, however, standardization to discharge followed by ANOVA or a t-test was found to be more expeditious.

Data were screened for violations of the assumption of homogeneity of variance using Bartlett's test (Snedecor and Cochran 1980). When such violations occurred, the variance usually could be stabilized with a log-transformation although a square-root transformation was occasionally necessary. Notably, inhomogeneity of variance never caused a false significant difference. Stabilization of the variance by transformation generally caused the significance of a difference to increase rather than decrease.

The use of annual means as replicates differs from the approach used in previous analyses of water quality changes in the NFA area by Playle and Williamson (1986) and Guilbault et al. (1979). In these studies, the individual measurements were used as replicates. Annual means were used in the present study to ensure independence of the experimental units (Lettenmaier 1976).

All data manipulations and statistical analyses were performed on an IBM compatible 386 computer using SYSTAT Version 4.0 (Wilkinson 1988).

2.2.3 Seasonal Differences

One of the strong points of the FEMP water quality study was the collection of samples under ice-cover as well as during the ice-free season. This permits analysis of current differences in water quality between the ice-cover and ice-free seasons and evaluation of the significance of changes in seasonal patterns related to the hydroelectric developments at sites where sufficient pre-development data are available.

The comprehensive statistical analysis of significant seasonal differences in a water quality database of the size considered in this study is a substantial task. In order to reduce the analysis to a manageable size, the data were screened for the probable occurrence of seasonal differences, as indicated by winter measurements falling outside the range of summer measurements. All occurrences of seasonal differences revealed by this screening process, both in the pre-development and FEMP datasets, are

identified in the report along with graphs illustrating the timing and magnitude of these differences. Seasonal data are not presented for variables in which seasonal differences were not evident based on the screening criteria.

No statistical analysis of any apparent changes in seasonal water quality patterns between the pre- and post-development periods was undertaken due to the general deficiency of the pre-development datasets. None of the sites where initial screening indicated changes in the seasonal water quality patterns may have occurred had a sufficient number of pre-development measurements (i.e., at least 3 in a season and at least 2 years of data both pre- and post-development) for application of a statistical analysis.

2.3 Bacteriological Water Quality

The bacteriological water quality study sampled recreational waters at all 5 NFA communities. At Cross Lake, Norway House, Split Lake, and Nelson House the studies were comprised of two components, a seasonal sampling program and an intensive sampling program. Only the seasonal sampling program was conducted at York Landing. Although a number of bacterial parameters (fecal and total coliform and fecal streptococcus) were measured in these studies, only the results of the fecal coliform counts are presented in this report. The fecal coliform bacteria count is the only parameter for which a water quality guideline has been established.

The seasonal sampling program involved the collection of single samples from 7 to 12 widely separated stations near each community on five occasions between late May and late September, 1987. Stations included popular and remote beach areas, raw water sources for water treatment plant intakes, and sewage treatment plant outfalls. Station numbers, descriptions, and locations are listed in Appendix 1.

The seasonal sampling program was conducted to provide a general indication of bacteriological water quality in each of the communities. A major deficiency of this program was its non-compliance with the sampling frequency specified in the Health and Welfare Canada Guidelines for Recreational Water Quality of at least 5 samples within a 30 day period (Health and Welfare Canada 1989). The sites were sampled on 5 occasions, but over a period of almost 120 days. Thus, the results of the seasonal study program can not be used to determine if recreational water quality is in strict compliance with the guideline. Rather, they provide a general measure of bacteriological water quality and can be used to identify sites requiring further examination.

The intensive sampling program involved the collection of 10 to 11 samples from each of 1 or 2 sites in the communities of Norway House, Cross Lake, Split Lake, and Nelson House on 2 dates in August, 1987. Sampling sites were located near sewage treatment plant outfalls and water treatment plant intakes. In Cross Lake, sampling was concentrated within 150 m of the water treatment plant intake on both sampling dates. In Norway House, 7 of the 11 samples

on August 11 and 7 of the 10 samples on August 24 were collected within 180 m of the sewage treatment plant outfall with the remaining samples on both dates from the vicinity of the water treatment plant. Station numbers, descriptions and locations, and sampling and analytical methods are listed in Appendix 1.

2.3.1 Data Interpretation

The suitability of water at the study sites for recreational use was assessed by comparison with the Health and Welfare Canada (1989) Guideline for Recreational Water Quality which specifies the geometric mean fecal coliform bacteria count should be lower than 200 bacteria per 100 mL. The geometric mean is the antilog of the arithmetic mean of the $\log_{10}$ -transformed bacterial counts (Elliot 1977). This guideline has been established to minimize the risk of swimmers developing gastroenteritis. Epidemiological studies have shown that the risk of contracting gastroenteritis is about 0.6% at a fecal coliform count of 200 bacteria per 100 mL (McGovern 1983).

As indicated in Section 2.2, the geometric mean fecal coliform counts from the seasonal study only provide a general measure of recreational water quality. This sampling frequency might be sufficient for the identification of serious violations of the water quality guideline but is inadequate for assessment of strict compliance. This inadequacy of the seasonal sampling program was alleviated in large measure by the intensive sampling program, which collected up to 20 samples from a single site within a period of 14-21 days. The geometric mean fecal coliform counts from the

intensive sampling program can therefore be used to assess compliance with the Health and Welfare Canada guideline. These intensive studies typically were conducted in the vicinity of sewage treatment plant outfalls; sites with the highest potential for bacterial contamination and therefore sites of the greatest concern.

3.0 Results and Discussion

3.1 Natural Rivers

3.1.1 Current Water Quality

Water quality in the three natural, unregulated, rivers (Burntwood R. above Threepoint Lake; Footprint R. above Footprint Lake; Aiken R.) was characterized by a number of variables which exceeded one or more of the Canadian guidelines for drinking water, recreational use, and protection of aquatic life. Mean turbidity values exceeded the drinking water guideline at all three sites (Tables 5, 6, 7, and 8). All measurements, both lab and field, from the Burntwood River exceeded the guideline as did all field measurements from the Footprint River.

The guideline for turbidity has been set to meet both health and aesthetic concerns. Turbidity can affect disinfection efficiency, biological nutrient availability, trihalomethane formation, and can carry toxic heavy metals and biocides (CCREM 1987). It should be kept in mind that the prevalence of turbidity values in excess of the Canadian guideline is not an indicator of pollution, only that the water should be treated for turbidity

Table 5. Canadian water quality guidelines (CCREM 1987) for drinking water, recreational use, and protection of aquatic life.

Variables	Drinking	Recreational	Aquatic Life
pH	6.5-8.5	5.0-9.0	6.5-9.0
Turbidity	5 NTU	<50 NTU	
Colour	15 TCU		
TDS	500 mg L ⁻¹		
NFR			increase of 10 mg L ⁻¹ _a
Extractable Al			0.005 mg L ⁻¹ (pH≤6.5) 0.1 mg L ⁻¹ (pH>6.5)
Dissolved As	0.05 mg L ⁻¹		0.05 mg L ⁻¹
Total Ba	1.0 mg L ⁻¹		
Total Cd	0.005 mg L ⁻¹		0.0002-0.0008 mg L ⁻¹ _b
Total Cu	1.0 mg L ⁻¹ _c		0.002 mg L ⁻¹
Extractable Fe	0.3 mg L ⁻¹ _c		0.3 mg L ⁻¹
Total Pb	0.05 mg L ⁻¹		0.001-0.002 mg L ⁻¹ _d
Extractable Mn	0.05 mg L ⁻¹		
Total Ni			0.025-0.065 mg L ⁻¹ _e
Dissolved Se	0.01 mg L ⁻¹		0.001 mg L ⁻¹
Total Zn	5.0 mg L ⁻¹		0.03 mg L ⁻¹
Dissolved NO ₂ NO ₃	10 mg L ⁻¹		
Total NH ₃			2.2 mg L ⁻¹ _f 1.37 mg L ⁻¹ _g
Dissolved Cl	250 mg L ⁻¹		
Dissolved F	1.5 mg L ⁻¹		
Dissolved SO ₄	500 mg L ⁻¹		

a Background NFR ≤100 mg L⁻¹b 0.0002 mg L⁻¹ at hardness 0-60 mg L⁻¹
0.0008 mg L⁻¹ at hardness 60-120 mg L⁻¹

c Aesthetic

d 0.001 mg L⁻¹ at hardness 0-60 mg L⁻¹
0.002 mg L⁻¹ at hardness 60-120 mg L⁻¹e 0.025 mg L⁻¹ at hardness 0-60 mg L⁻¹
0.065 mg L⁻¹ at hardness 60-120 mg L⁻¹

f pH 6.5, 10°C

g pH 8.0, 10°C

Table 6. Summary statistics for physical and chemical water quality variables measured in the Burntwood River above Threepoint Lake; May, 1987, to March, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.5	0.2	7.9	7.2
	Lab		7.40	0.20	7.97	7.00
Conductivity	Field	$\mu\text{S cm}^{-1}$	101	22	142	24
	Lab		112	17	167	82
Turbidity	Field	NTU	23 ⁺	7	47 ⁺	12 ⁺
	Lab		19 ⁺	4	27 ⁺	13 ⁺
Colour	True	Relative	41 ⁺	10	55 ⁺	25 ⁺
TDS		mg L^{-1}	57	10	71	25
NFR	Total	mg L^{-1}	14	7	33	6
	Fixed	mg L^{-1}	11	5	27	4
Alkalinity	Total	mg L^{-1}	50.1	6.3	61.4	36.2
	Bicarbonate	mg L^{-1}	63.4	13.1	115.9	44.1
Hardness	Total	mg L^{-1}	51.2	5.6	61.7	39.6
	Non-carbonate	mg L^{-1}	1.5	1.3	3.5	0
Free CO ₂		mg L^{-1}	3.32	1.92	6.70	0
Phosphorus	Particulate	mg L^{-1}	0.020	0.011	0.062	<0.006
	Ortho PO ₄	mg L^{-1}	0.003	0.004	0.022	<0.003
	Total Dissolved	mg L^{-1}	0.016	0.009	0.050	0.006
	Total	mg L^{-1}	0.036	0.016	0.112	0.021
Nitrogen	Particulate	mg L^{-1}	0.102	0.099	0.529	0.010
	Total Dissolved	mg L^{-1}	0.454	0.104	0.771	0.280
	NO ₂ -NO ₃	mg L^{-1}	0.049	0.059	0.205	<0.010
	Total NH ₃	mg L^{-1}	0.035	0.021	0.096	0.012
	Total	mg L^{-1}	0.56	0.14	1.07	0.37
Carbon	DOC	mg L^{-1}	11.96	0.88	14.00	10.40
	POC	mg L^{-1}	0.724	0.470	2.640	0.310
	TOC	mg L^{-1}	12.68	1.13	15.24	10.87
	DIC	mg L^{-1}	11.7	1.8	14.7	8.5
Chlorophyll-a		mg L^{-1}	0.004	0.005	0.022	<0.001
Ca	Dissolved	mg L^{-1}	13.0	1.4	15.8	10.1
Mg	Dissolved	mg L^{-1}	4.6	0.5	5.4	3.5
K	Dissolved	mg L^{-1}	0.96	0.15	1.29	0.75
Na	Dissolved	mg L^{-1}	2.5	0.4	3.3	1.9

Table 6. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L ⁻¹	3.25	0.92	5.23	1.69
Cl	Dissolved	mg L ⁻¹	1.0	0.2	1.5	0.7
F	Dissolved	mg L ⁻¹	0.07	0.01	0.10	0.06
SO ₄	Dissolved	mg L ⁻¹	2.4	0.7	4.1	1.0
Al	Extractable	mg L ⁻¹	0.443 ⁺⁺⁺	0.148	0.700 ⁺⁺⁺	0.191 ⁺⁺⁺
	Dissolved	mg L ⁻¹	0.100	0.082	0.443	<0.100
As	Dissolved	mg L ⁻¹	0.0003	0.0001	0.0004	0.0001
Ba	Total	mg L ⁻¹	<0.08		<0.08	<0.08
Cd	Total	mg L ⁻¹	0.0001	0.0002	0.0009 ⁺⁺⁺	<0.0001
	Dissolved	mg L ⁻¹	<0.0001		0.0001	<0.0001
Co	Total	mg L ⁻¹	<0.0005	0.0003	0.0013	<0.0005
Cu	Total	mg L ⁻¹	0.0061 ⁺⁺⁺	0.0081	0.0296 ⁺⁺⁺	0.0012
	Dissolved	mg L ⁻¹	0.0017	0.0005	0.0026	0.0011
Fe	Extractable	mg L ⁻¹	0.642 ^{+,+++}	0.165	1.050 ^{+,+++}	0.183
	Dissolved	mg L ⁻¹	0.142	0.077	0.350	0.035
Pb	Total	mg L ⁻¹	0.0009	0.0007	0.0028 ⁺⁺⁺	<0.0007
	Dissolved	mg L ⁻¹	<0.0007		<0.0007	<0.0007
Mn	Extractable	mg L ⁻¹	0.041	0.016	0.090 ⁺	0.005
	Dissolved	mg L ⁻¹	0.014	0.017	0.056	<0.002
Ni	Total	mg L ⁻¹	0.0015	0.0005	0.0026	0.0007
	Dissolved	mg L ⁻¹	0.0008	0.0005	0.0018	<0.0005
Se	Dissolved	mg L ⁻¹	0.0001	0.0001	0.0003	<0.0001
V	Total	mg L ⁻¹	0.0015	0.0013	0.0070	<0.0005
	Dissolved	mg L ⁻¹	0.0005	0.0003	0.0010	<0.0005
Zn	Total	mg L ⁻¹	0.0041	0.0033	0.0127	0.0009
	Dissolved	mg L ⁻¹	0.0016	0.0010	0.0030	0.0005
Discharge	Daily	m ³ sec ⁻¹	17.9	11.8	44.0	3.0

⁺ Exceeds limit for drinking water.

⁺⁺ Exceeds limit for recreational use.

⁺⁺⁺ Exceeds limit for protection of aquatic life.

Table 7. Summary statistics for physical and chemical water quality variables measured in the Footprint River above Footprint Lake; May, 1987, to March, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.7	0.5	8.5	7.0
	Lab		7.48	0.33	7.87	6.15
Conductivity	Field	$\mu\text{S cm}^{-1}$	178	38	249	112
	Lab		191	42	268	121
Turbidity	Field	NTU	12 ⁺	6	36 ⁺	6 ⁺
	Lab		9 ⁺	3	18 ⁺	4
Colour	True	Relative	20 ⁺	8	40 ⁺	10
TDS		mg L^{-1}	99	26	147	41
NFR	Total	mg L^{-1}	8	4	20	4
	Fixed	mg L^{-1}	6	3	16	2
Alkalinity	Total	mg L^{-1}	96.0	21.7	138.0	56.8
	Bicarbonate	mg L^{-1}	117.0	26.5	168.2	69.2
Hardness	Total	mg L^{-1}	96.2	22.0	139.4	61.2
	Non-carbonate	mg L^{-1}	1.2	1.7	6.7	0
Free CO ₂		mg L^{-1}	5.39	5.92	22.29	0
Phosphorus	Particulate	mg L^{-1}	0.018	0.007	0.041	0.007
	Ortho PO ₄	mg L^{-1}	<0.003	0.001	0.003	<0.003
	Total Dissolved	mg L^{-1}	0.012	0.005	0.031	0.005
	Total	mg L^{-1}	0.030	0.007	0.051	0.018
Nitrogen	Particulate	mg L^{-1}	0.093	0.049	0.248	0.040
	Total Dissolved	mg L^{-1}	0.458	0.069	0.610	0.280
	NO ₂ -NO ₃	mg L^{-1}	0.022	0.027	0.110	<0.010
	Total NH ₃	mg L^{-1}	0.043	0.031	0.124	0.012
	Total	mg L^{-1}	0.55	0.10	0.86	0.41
Carbon	DOC	mg L^{-1}	11.27	0.75	13.00	10.00
	POC	mg L^{-1}	0.640	0.260	1.270	0.340
	TOC	mg L^{-1}	11.91	0.78	14.00	10.57
	DIC	mg L^{-1}	22.8	5.4	33.1	13.6
Chlorophyll-a		mg L^{-1}	0.004	0.005	0.024	<0.001
Ca	Dissolved	mg L^{-1}	27.5	6.7	40.5	17.6
Mg	Dissolved	mg L^{-1}	6.7	1.3	9.3	4.2
K	Dissolved	mg L^{-1}	0.90	0.23	1.37	0.59
Na	Dissolved	mg L^{-1}	2.4	0.5	3.5	1.5

Table 7. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L ⁻¹	2.82	1.45	5.07	0.25
Cl	Dissolved	mg L ⁻¹	1.0	0.3	1.8	0.7
F	Dissolved	mg L ⁻¹	0.07	0.01	0.09	0.05
SO ₄	Dissolved	mg L ⁻¹	1.9	0.7	3.3	<0.2
Al	Extractable	mg L ⁻¹	0.190 ⁺⁺⁺	0.097	0.410 ⁺⁺⁺	<0.100
	Dissolved	mg L ⁻¹	<0.100	0.029	0.148	<0.100
As	Dissolved	mg L ⁻¹	0.0003	0.0001	0.0004	0.0001
Ba	Total	mg L ⁻¹	<0.08		<0.08	<0.08
Cd	Total	mg L ⁻¹	<0.0001	0.0001	0.0003	<0.0001
	Dissolved	mg L ⁻¹	<0.0001	0.0001	0.0002	<0.0001
Co	Total	mg L ⁻¹	<0.0005	0.0002	0.0011	<0.0005
Cu	Total	mg L ⁻¹	0.0232 ⁺⁺⁺	0.1060	0.5530 ⁺⁺⁺	0.0005
	Dissolved	mg L ⁻¹	0.0011	0.0007	0.0027	0.0005
Fe	Extractable	mg L ⁻¹	0.420 ^{+,+++}	0.246	1.150 ^{+,+++}	0.091
	Dissolved	mg L ⁻¹	0.123	0.125	0.438	0.010
Pb	Total	mg L ⁻¹	0.0011	0.0030	0.0161 ⁺⁺⁺	<0.0007
	Dissolved	mg L ⁻¹	<0.0007	0.0001	0.0007	<0.0007
Mn	Extractable	mg L ⁻¹	0.057 ⁺	0.078	0.296 ⁺	<0.002
	Dissolved	mg L ⁻¹	0.047	0.084	0.286	<0.002
Ni	Total	mg L ⁻¹	0.0009	0.0003	0.0016	<0.0005
	Dissolved	mg L ⁻¹	0.0007	0.0003	0.0012	<0.0005
Se	Dissolved	mg L ⁻¹	0.0001	0.0001	0.0002	<0.0001
V	Total	mg L ⁻¹	0.0008	0.0004	0.0014	<0.0005
	Dissolved	mg L ⁻¹	<0.0005	0.0002	0.0006	<0.0005
Zn	Total	mg L ⁻¹	0.0040	0.0099	0.0515 ⁺⁺⁺	0.0004
	Dissolved	mg L ⁻¹	0.0009	0.0010	0.0032	<0.0003
Discharge	Daily	m ³ sec ⁻¹	3.0	2.3	9.0	0

⁺ Exceeds limit for drinking water.

⁺⁺ Exceeds limit for recreational use.

⁺⁺⁺ Exceeds limit for protection of aquatic life.

Table 8. Summary statistics for physical and chemical water quality in the Aiken River at the inlet to Split Lake, January, 1987, to October, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.5	0.3	8.1	6.9
	Lab		7.57	0.28	7.96	6.74
Conductivity	Field	$\mu\text{S cm}^{-1}$	185	85	360	66
	Lab		196	92	370	75
Turbidity	Field	NTU	7 ⁺	3	16 ⁺	2
	Lab		6 ⁺	4	18 ⁺	1
Colour	True	Relative	46 ⁺	25	120 ⁺	8
TDS		mg L^{-1}	104	51	212	27
NFR	Total	mg L^{-1}	5	4	18	<1
	Fixed	mg L^{-1}	3	2	8	<1
Alkalinity	Total	mg L^{-1}	84.9	36.4	195.0	32.4
	Bicarbonate	mg L^{-1}	105.1	43.8	237.7	39.5
Hardness	Total	mg L^{-1}	91.3	36.5	187.2	39.4
	Non-carbonate	mg L^{-1}	6.5	4.7	17.3	0
Free CO ₂		mg L^{-1}	5.82	8.85	47.58	0
Phosphorus	Particulate	mg L^{-1}	0.012	0.009	0.038	<0.006
	Ortho PO ₄	mg L^{-1}	<0.003	0.002	0.010	<0.003
	Total Dissolved	mg L^{-1}	0.011	0.006	0.030	0.004
	Total	mg L^{-1}	0.023	0.010	0.060	0.008
Nitrogen	Particulate	mg L^{-1}	0.067	0.064	0.404	0.020
	Total Dissolved	mg L^{-1}	0.398	0.087	0.756	0.230
	NO ₂ -NO ₃	mg L^{-1}	0.011	0.014	0.058	<0.010
	Total NH ₃	mg L^{-1}	0.037	0.066	0.385	0.007
	Total	mg L^{-1}	0.47	0.14	1.16	0.29
Carbon	DOC	mg L^{-1}	12.16	4.09	23.10	3.90
	POC	mg L^{-1}	0.527	0.776	4.900	0.100
	TOC	mg L^{-1}	12.69	4.50	28.00	4.40
	DIC	mg L^{-1}	20.3	8.7	46.8	7.8
Chlorophyll-a		mg L^{-1}	0.003	0.002	0.008	<0.001
Ca	Dissolved	mg L^{-1}	25.3	9.7	54.7	11.5
Mg	Dissolved	mg L^{-1}	6.9	3.2	13.2	2.6
K	Dissolved	mg L^{-1}	0.93	0.68	2.55	0.16
Na	Dissolved	mg L^{-1}	5.1	5.1	17.9	0.7

Table 8. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L ⁻¹	2.27	2.45	13.41	0.45
Cl	Dissolved	mg L ⁻¹	5.7	5.8	20.7	0.8
F	Dissolved	mg L ⁻¹	0.07	0.03	0.12	0.03
SO ₄	Dissolved	mg L ⁻¹	7.7	7.9	27.0	1.1
Al	Extractable	mg L ⁻¹	0.107 ⁺⁺⁺	0.059	0.226 ⁺⁺⁺	<0.100
	Dissolved	mg L ⁻¹	<0.100	0.022	0.153	<0.100
As	Dissolved	mg L ⁻¹	0.0003	0.0001	0.0006	0.0001
Ba	Total	mg L ⁻¹	<0.08		<0.08	<0.08
Cd	Total	mg L ⁻¹	<0.0001	0.0001	0.0003	<0.0001
	Dissolved	mg L ⁻¹	<0.0001		0.0001	<0.0001
Co	Total	mg L ⁻¹	<0.0005	0.0003	0.0022	<0.0005
Cu	Total	mg L ⁻¹	0.0029 ⁺⁺⁺	0.0041	0.0170 ⁺⁺⁺	<0.0005
	Dissolved	mg L ⁻¹	0.0011	0.0008	0.0030	<0.0005
Fe	Extractable	mg L ⁻¹	0.438 ^{+,+++}	1.008	5.980 ^{+,+++}	0.101
	Dissolved	mg L ⁻¹	0.268	0.879	5.210	0.017
Pb	Total	mg L ⁻¹	<0.0007	0.0003	0.0014	<0.0007
	Dissolved	mg L ⁻¹	0.0009	0.0011	0.0036	<0.0007
Mn	Extractable	mg L ⁻¹	0.093 ⁺	0.339	1.920 ⁺	0.005
	Dissolved	mg L ⁻¹	0.073	0.292	1.640	<0.002
Ni	Total	mg L ⁻¹	0.0005	0.0003	0.0013	<0.0005
	Dissolved	mg L ⁻¹	<0.0005	0.0004	0.0015	<0.0005
Se	Dissolved	mg L ⁻¹	0.0001	0.0001	0.0002	<0.0001
V	Total	mg L ⁻¹	<0.0005	0.0003	0.0013	<0.0005
	Dissolved	mg L ⁻¹	<0.0005	0.0001	0.0006	<0.0005
Zn	Total	mg L ⁻¹	0.0030	0.0038	0.0186	<0.0003
	Dissolved	mg L ⁻¹	0.0012	0.0014	0.0041	<0.0003

⁺ Exceeds limit for drinking water.

⁺⁺ Exceeds limit for recreational use.

⁺⁺⁺ Exceeds limit for protection of aquatic life.

reduction before distribution in a municipal water system.

Colour also generally exceeded the Canadian guideline for human consumption (Tables 6, 7, and 8). Mean colour in the Footprint River was about 30% above the guideline, while Aiken and Burntwood river colour values were about 3 times the guideline level.

The drinking water guideline for colour was set primarily for aesthetic reasons, with the value of 15 TCU representing the lowest concentration in a glass of water which is detectable to the eye (CCREM 1987). Control of colour also is indirectly related to protection of human health, as the chemicals which are responsible for production of the colour (e.g., humic and fulvic acids) can also carry toxic materials such as heavy metals.

Mean concentrations of extractable Al and Fe and total Cu exceeded the guideline for protection of aquatic life in all 3 natural rivers (Tables 6, 7, and 8). The maximum total Pb concentrations exceeded the aquatic life guideline in the Burntwood and Footprint rivers as did peak total Zn in the Footprint River. The consistent violation of the aquatic life guidelines for Al, Fe, and Cu, and the occasional violation of the guidelines for Pb and Zn therefore appear to be a natural characteristic of these local rivers. The mean extractable Fe concentration also exceeded the drinking water guideline in all three rivers, as did mean extractable Mn in the Aiken and Footprint rivers and maximum extractable Mn in the Burntwood River.

3.1.2 Variation Among the Rivers

There were no significant differences among the three rivers in the annual mean concentrations of many variables, including: most nutrient species (particulate P, ortho PO_4 , TDP, total P, particulate N, TDN, total N, DOC, POC, and TOC), most major ions (Mg, K, Na, Cl, SO_4 , and F), pH, dissolved As and Se, and several metal species (total Cu and Zn, total Pb, dissolved Fe, and extractable and dissolved Mn; Table 9). The Burntwood River differed from the Footprint and Aiken rivers in having lower conductivity, TDS, hardness, DIC, and Ca, and higher turbidity and extractable Al. Burntwood River NFR also was marginally higher than in the Aiken River.

The substances which are responsible for the colour of a stream also can carry heavy metals (CCREM 1987). Both the Burntwood and Aiken rivers were highly coloured relative to the Footprint River, yet none of the differences in metals among the rivers was accounted for by this difference in colour. Levels of extractable Fe and total Ni and V were highest in the Burntwood River and lowest in the Aiken River (Table 9).

3.1.3 Water Quality - Water Discharge Relationships

Water-quality discharge relationships were examined for just two of the three natural rivers, and then only for the FEMP study period. There are no discharge data for the Aiken River and discharge data for the natural Burntwood and Footprint river sites are available only for the post-development period.

Table 9. Evaluation of the significance of differences in selected water quality parameters among the Aiken River at the inlet to Split Lake, the Burntwood River above Threepoint Lake and the Footprint River above Footprint Lake. Values listed are the mean and standard deviation (s.d.) for the respective site in 1987 and 1988, and the p value from the Tukey/HSD pairwise comparison.

pH (Lab)			Conductivity (Lab)			Turbidity (Lab)		
	Mean	s.d.	Aiken	BWR	Mean	s.d.	Aiken	BWR
Aiken	7.59	0.06			170	18	Aiken	6
BWR	7.41	0.06	0.110		112.0 ¹	4	BWR	19
FPR	7.51	0.05	0.465	0.343	179.0 ²	19	FPR	9
								0.047
								0.855
								0.353
								0.031
Colour (True)			TDS			NFR (Lab)		
	Mean	s.d.	Aiken	BWR	Mean	s.d.	Aiken	BWR
Aiken	45.0	6			90	5	Aiken	4
BWR	42.0 ¹	2	0.896		57.0 ¹	3	BWR	14.0 ¹
FPR	19.0 ¹	6	0.030	0.039	93.0 ¹	6	FPR	9.0 ¹
								0.011
								0.888
								0.270
								0.280
Alkalinity (Total)			Hardness (Total)			Part P		
	Mean	s.d.	Aiken	BWR	Mean	s.d.	Aiken	BWR
Aiken	75.3	0.4			81.5	4.2	Aiken	0.011
BWR	50.0 ²	1.6	0.013		51.1 ¹	0.8	BWR	0.020
FPR	90.2 ²	6.1	0.054	0.004	90.1 ¹	9.1	FPR	0.017
								0.003
								0.170
								0.554

Table 9. continued.

Ortho P	TDP				Total P			
	Mean	s.d.	Aiken	BWR	Mean	s.d.	Aiken	BWR
Aiken	0.003	0.001			Aiken	0.021	0.004	
BWR	0.003	0.001	0.856		BWR	0.036	0.007	0.092
FPR	0.002	<0.001	0.856	0.581	FPR	0.030	0.002	0.282
								0.454
Part N	TDN				Total N			
	Mean	s.d.	Aiken	BWR	Mean	s.d.	Aiken	BWR
Aiken	0.060	0.002			Aiken	0.45	0.02	
BWR	0.105	0.027	0.135		BWR	0.56	0.10	0.322
FPR	0.097	0.009	0.208	0.869	FPR	0.55 ¹	0.03	0.366
								0.989
DOC	POC				TOC			
	Mean	s.d.	Aiken	BWR	Mean	s.d.	Aiken	BWR
Aiken	11.55	0.92			Aiken	11.98	0.94	
BWR	12.06 ¹	0.36	0.695		BWR	12.78 ¹	0.16	0.433
FPR	11.15	0.21	0.788	0.388	FPR	11.77	0.17	0.930
								0.311

Table 9. continued.

DIC	Chlorophyll a					Diss Ca				
	Mean	s.d.	Aiken	BWR		Mean	s.d.	Aiken	BWR	
Aiken	18.0	0.9			Aiken	22.7	0.6			
BWR	11.7 ¹	0.1	0.031		BWR	13.0 ¹	0.2	0.015		
FPR	21.3 ¹	2.0	0.148	0.010	FPR	25.8	2.5	0.238	0.007	
Diss Mg	Diss K					Diss Na				
	Mean	s.d.	Aiken	BWR		Mean	s.d.	Aiken	BWR	
Aiken	6.1	0.6			Aiken	4.0	1.5			
BWR	4.6 ¹	0.1	0.109		BWR	2.6 ²	0.1	0.363		
FPR	6.3 ¹	0.6	0.874	0.076	FPR	2.3 ¹	0.1	0.280	0.955	
React Si	Diss Cl					Diss F				
	Mean	s.d.	Aiken	BWR		Mean	s.d.	Aiken	BWR	
Aiken	1.69	0.01			Aiken	0.07	<0.01			
BWR	3.16 ¹	0.17	0.033		BWR	0.07	<0.01	0.682		
FPR	2.49	0.50	0.147	0.214	FPR	0.07 ¹	<0.01	1.000	0.682	

Table 9. continued.

Diss Mn			Tot Ni			Diss Se		
	Mean	s.d.	Aiken	BWR		Mean	s.d.	Aiken
Aiken	0.006	0.001			Aiken	0.0001	<0.0001	
BWR	0.014 ¹	0.006	0.824		BWR	0.0001	0.0001	1.000
FPR	0.021	0.023	0.586	0.890	FPR	0.0001	<0.0001	1.000
Tot V			Tot Zn					
	Mean	s.d.	Aiken	BWR		Mean	s.d.	Aiken
Aiken	0.0004	0.0001			Aiken	0.0032	0.0019	
BWR	0.0013	0.0011	0.015		BWR	0.0040	0.0021	0.796
FPR	0.0009	0.0004	0.080	0.105	FPR	0.0036	0.0045	0.723
								0.989

¹ Discharge-standardized variables.

General characteristics of the discharge relationships in the natural rivers included relatively large numbers of discharge correlated variables, the dominance of negative slopes, and the frequent occurrence of moderate to strong correlations (i.e., $r^2 > 0.4$). Colour, turbidity, NFR, and particulate nutrient species typically were positively correlated with discharge. Conductivity, major ions, related variables such as alkalinity and hardness, metals and dissolved nutrients generally were negatively correlated. Specific characteristics of the discharge relationships are discussed below.

3.1.3.1 Burntwood River above Threepoint Lake

More variables (25) were correlated with discharge at this site than at any of the other sites in the study (Figure 8). Negative correlates included conductivity (field and lab), TDS, alkalinity (total and bicarbonate), hardness (total), TDN, dissolved $\text{NO}_2\text{-NO}_3$, total NH_3 , DIC, major ions (Ca, Mg, K, Na, Si, Cl), and metals (dissolved Fe, dissolved and extractable Mn). Positive correlates included turbidity (field), colour, NFR (total and fixed), DOC, and TOC. Coefficients of determination (r^2) ranged from <0.2 to >0.8 with 11 of the regressions on discharge accounting for at least 40% of the variance in the dependent variables.

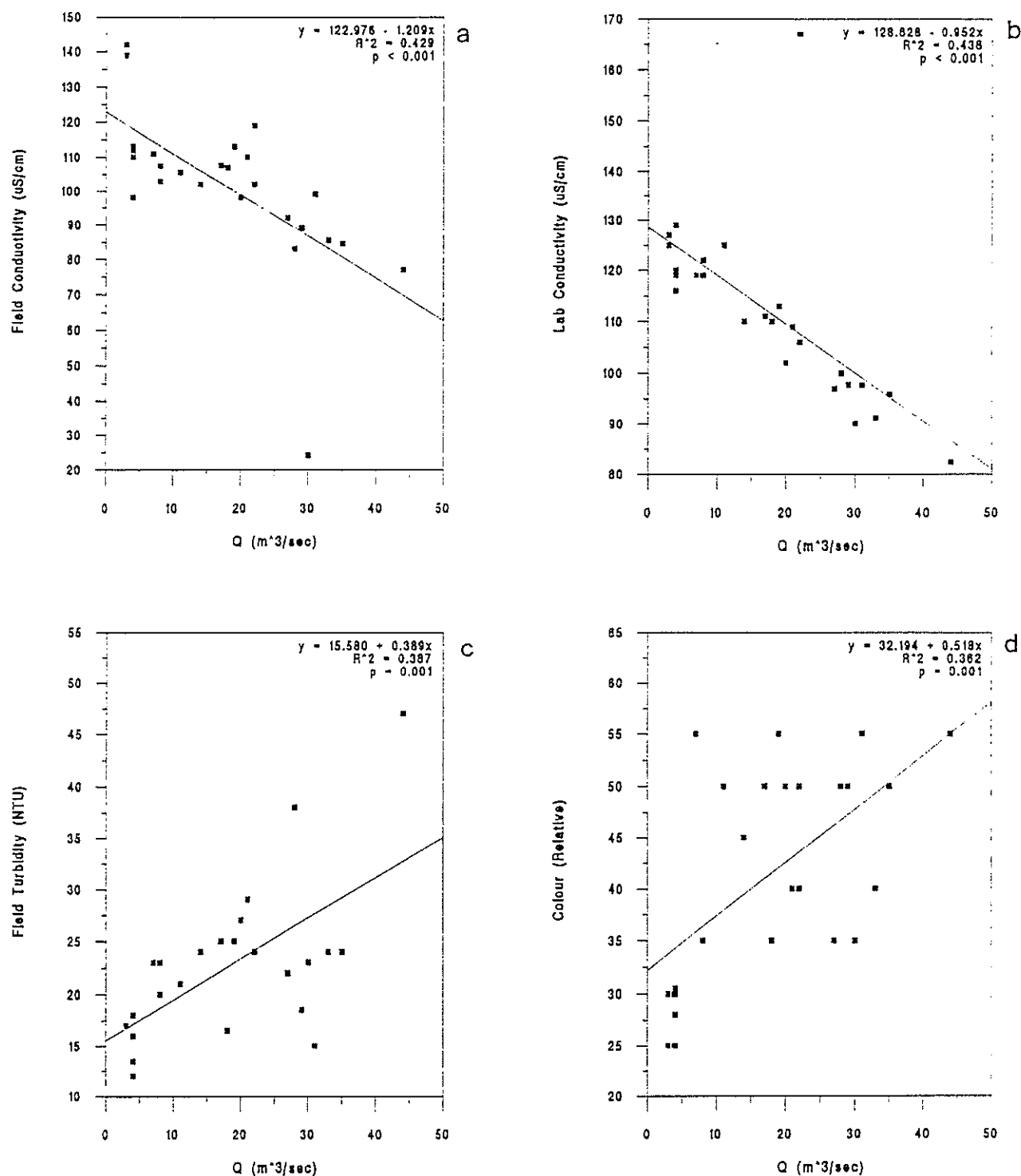


Figure 8. Water chemistry - water discharge relationships in the Burntwood River above Threepoint Lake, 1987-1989. a. field conductivity b. lab conductivity c. field turbidity d. colour e. TDS f. total hardness g. total NFR h. fixed NFR i. total alkalinity j. bicarbonate alkalinity k. TDN l. dissolved $\text{NO}_2\text{-NO}_3$ m. total NH_3 n. DOC o. TOC p. DIC q. dissolved Ca r. dissolved Mg s. dissolved K t. dissolved Na u. reactive Si v. dissolved Cl w. dissolved Fe x. extractable Mn y. dissolved Mn.

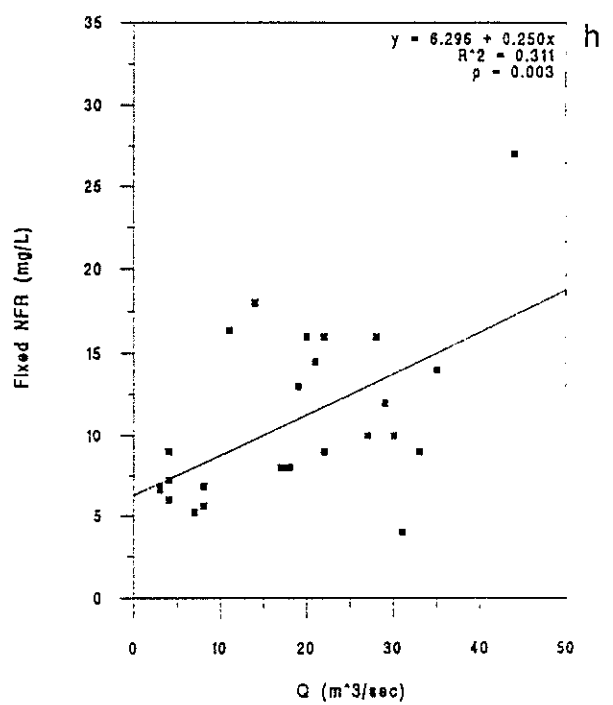
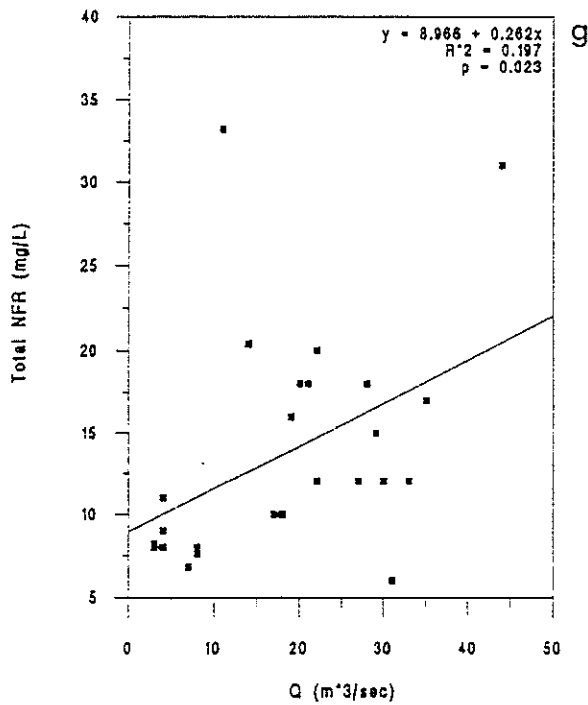
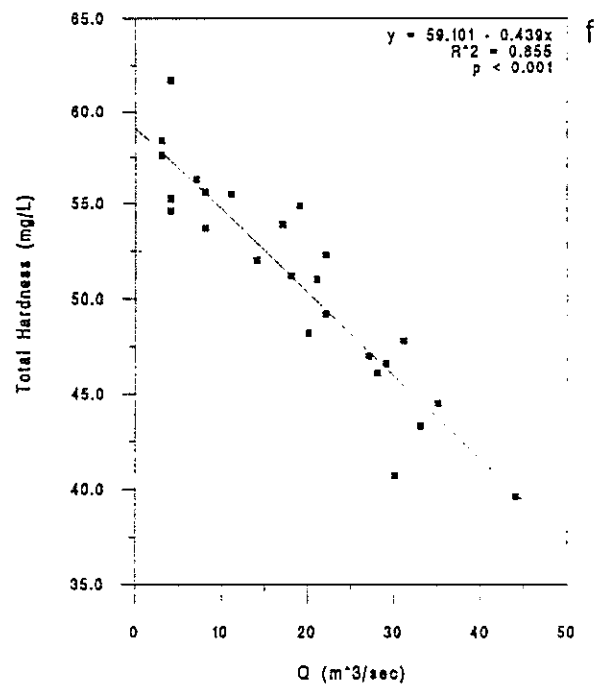
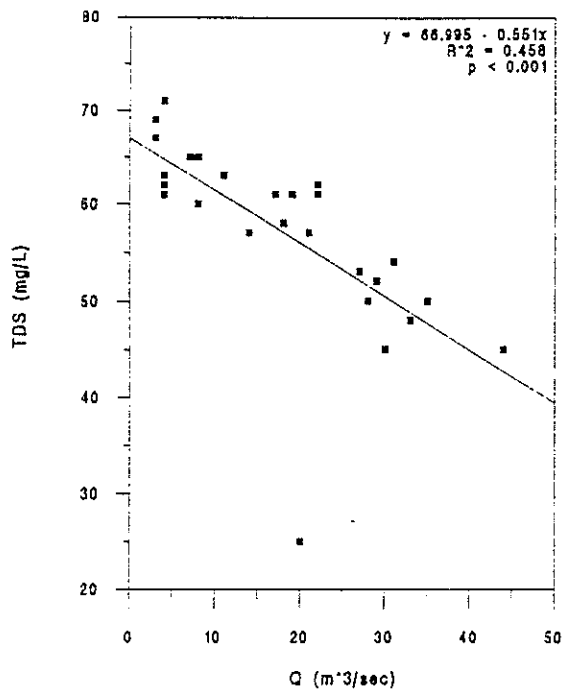


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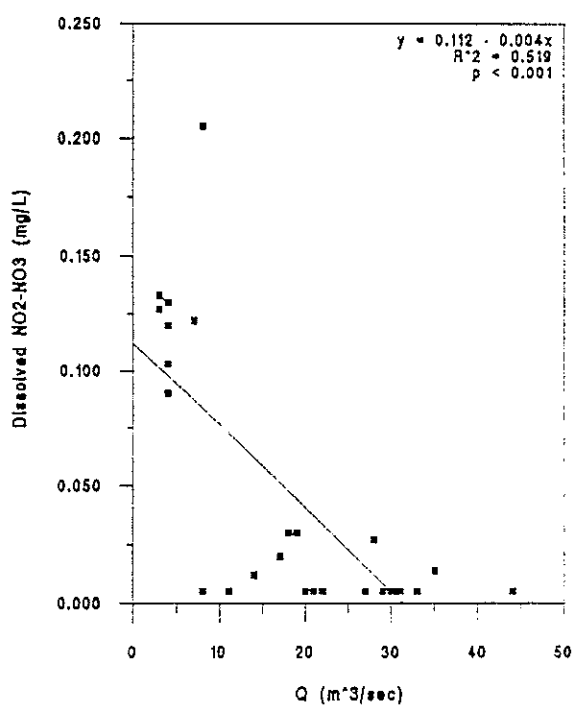
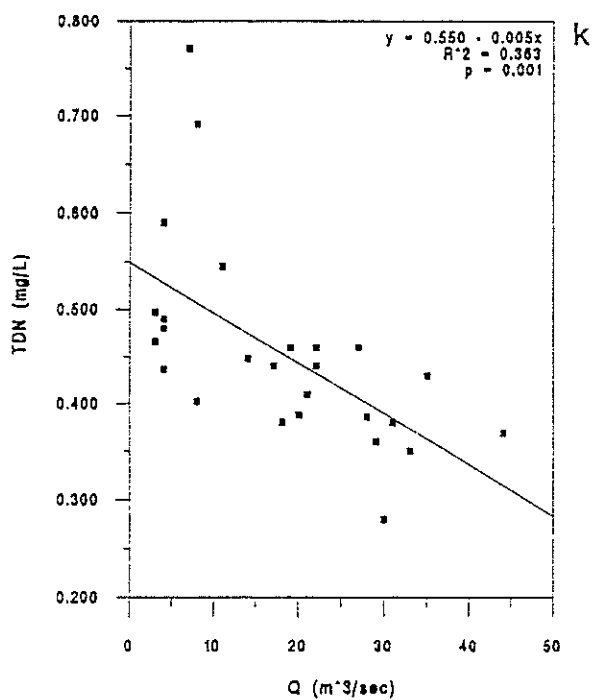
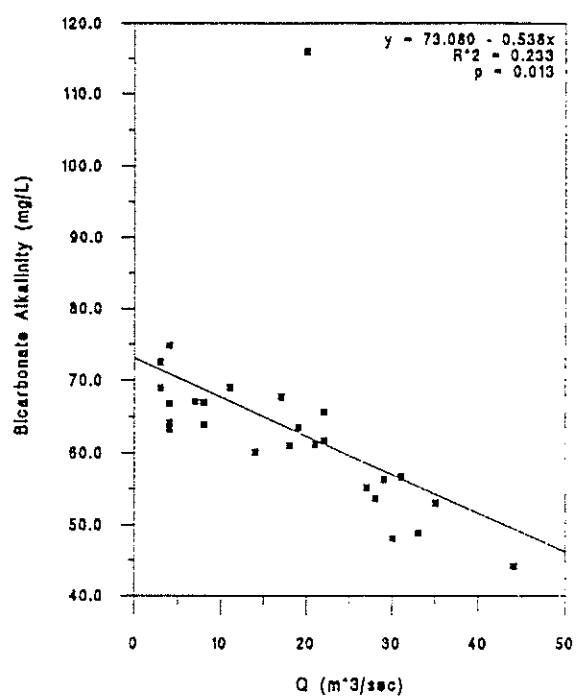
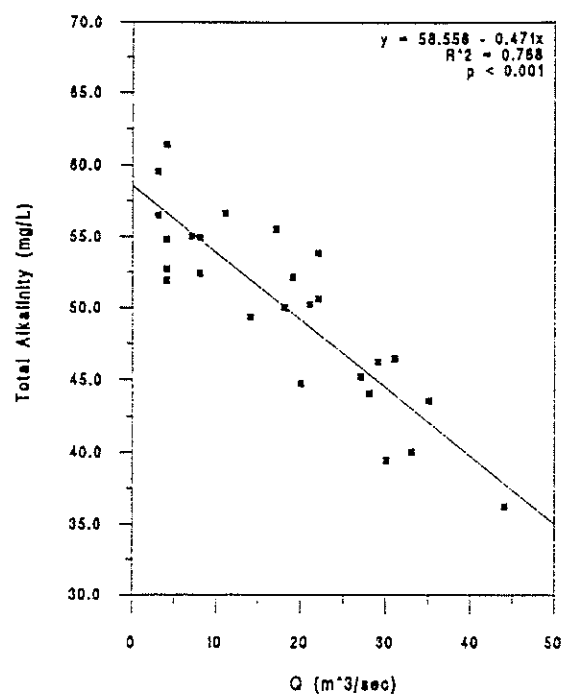


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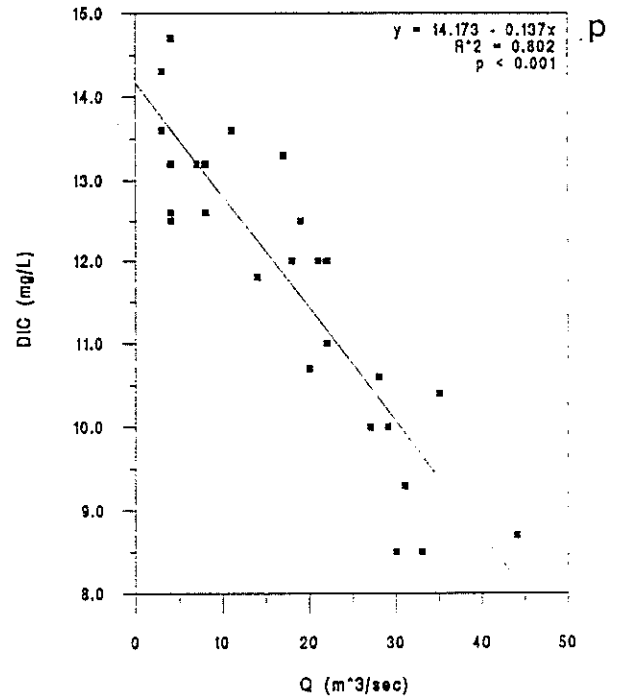
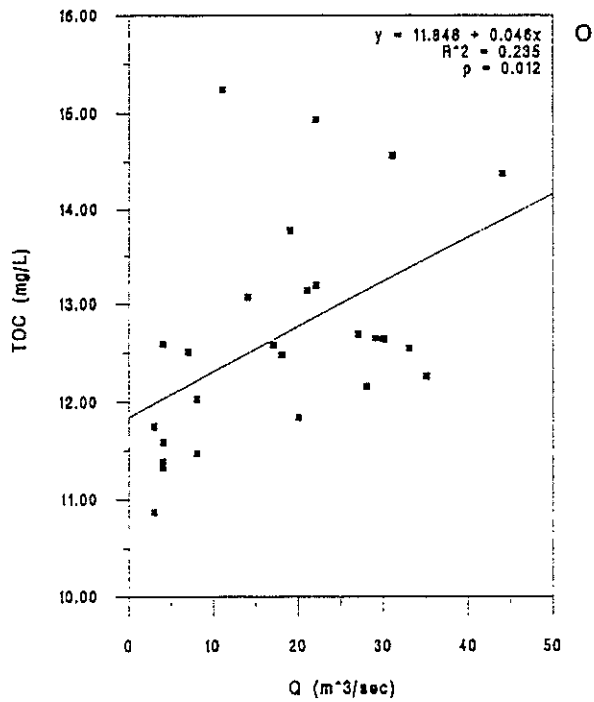
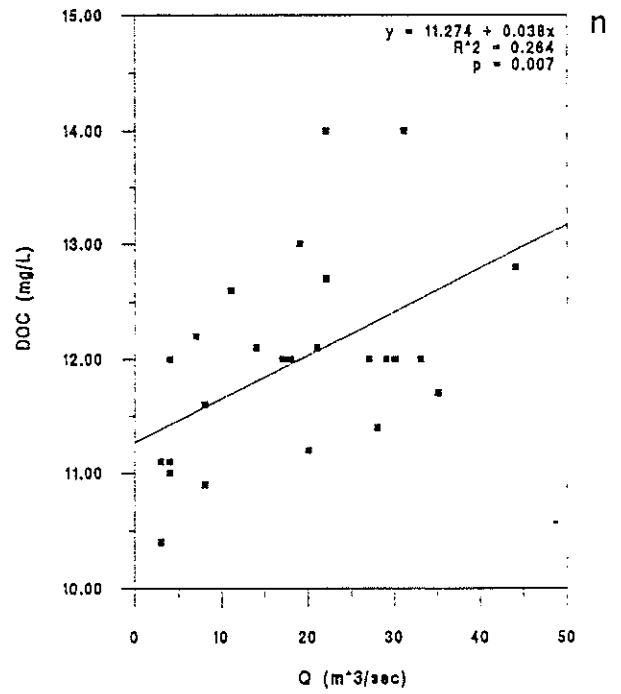
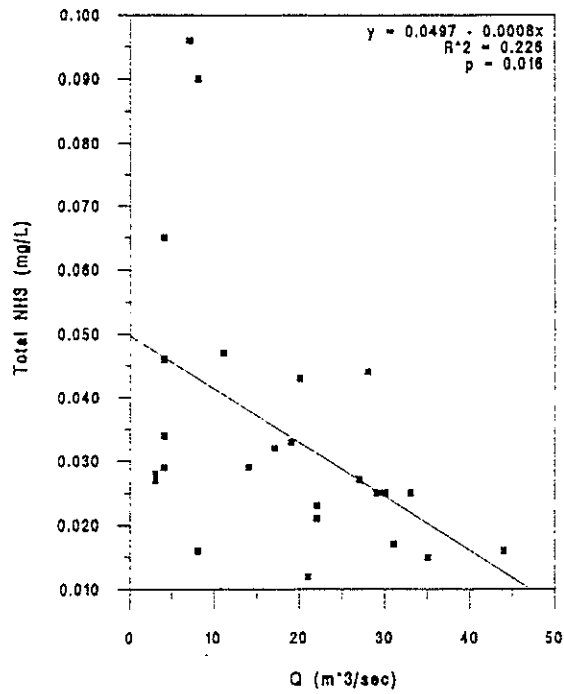


Figure 8. Continued.

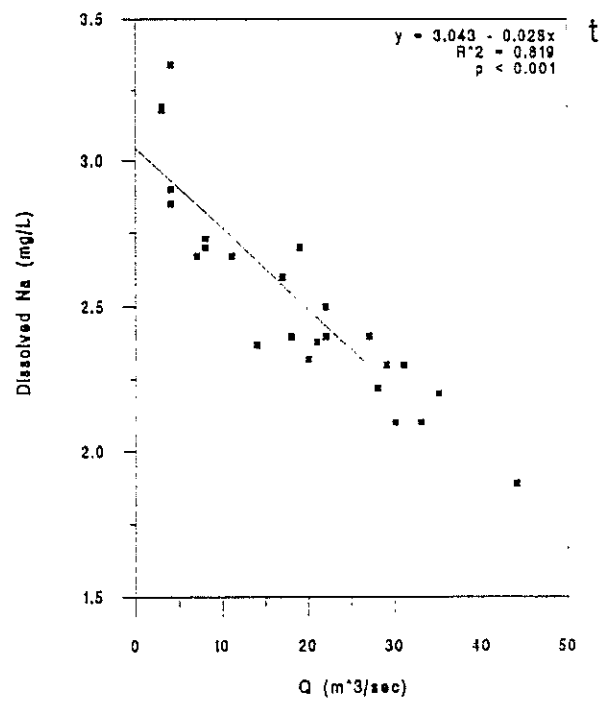
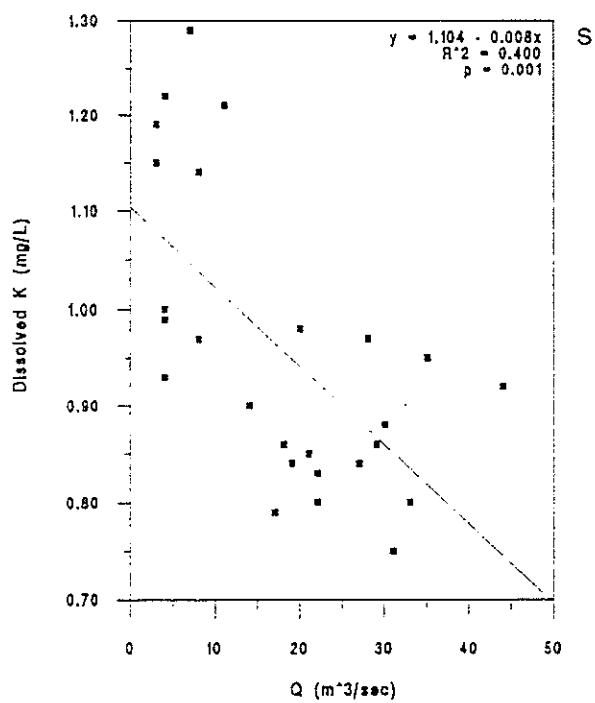
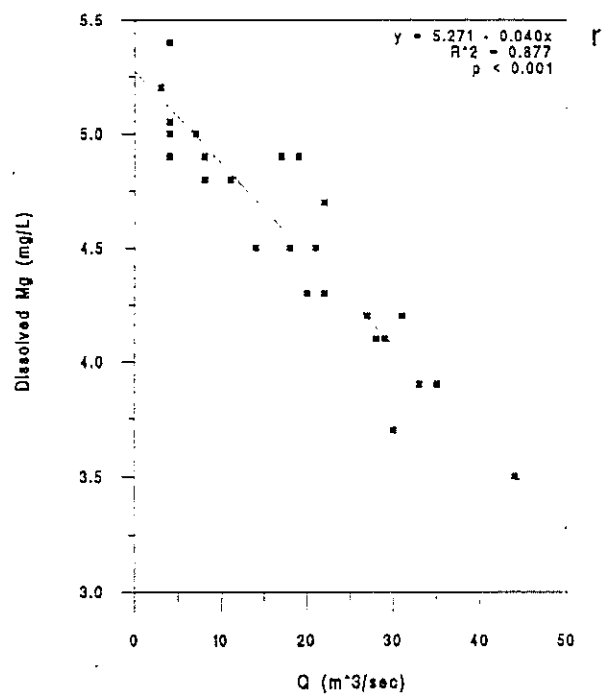
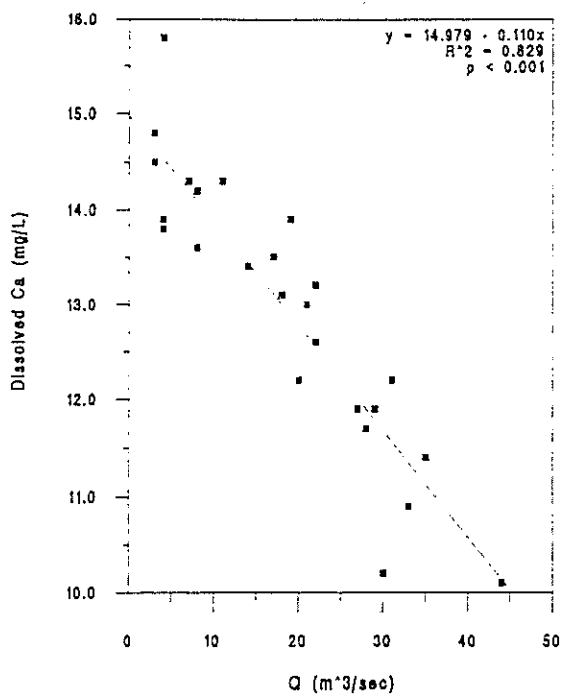


Figure 8. Continued.

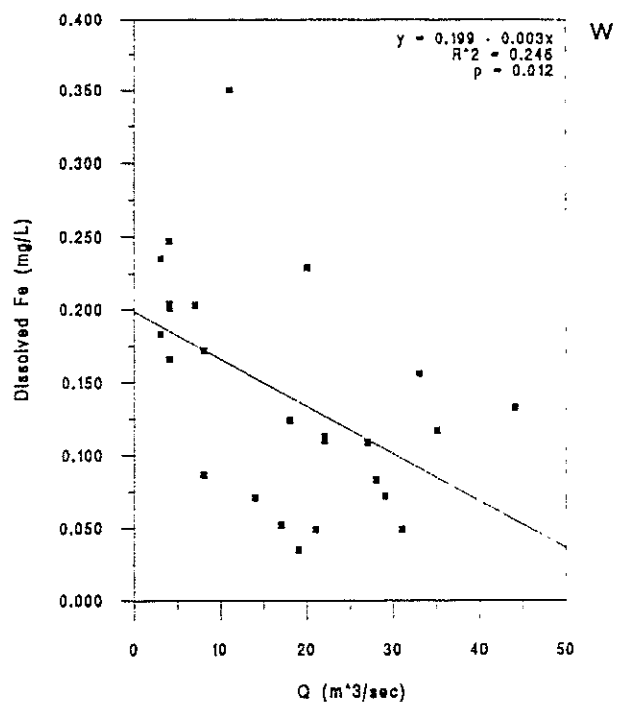
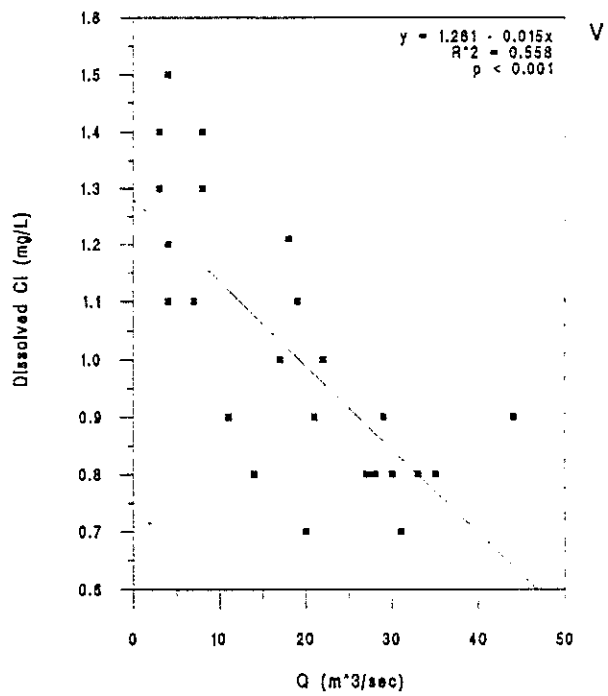
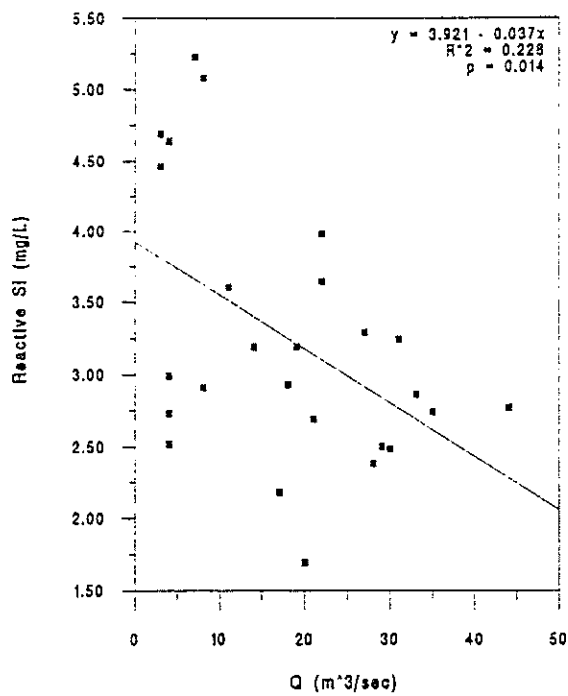


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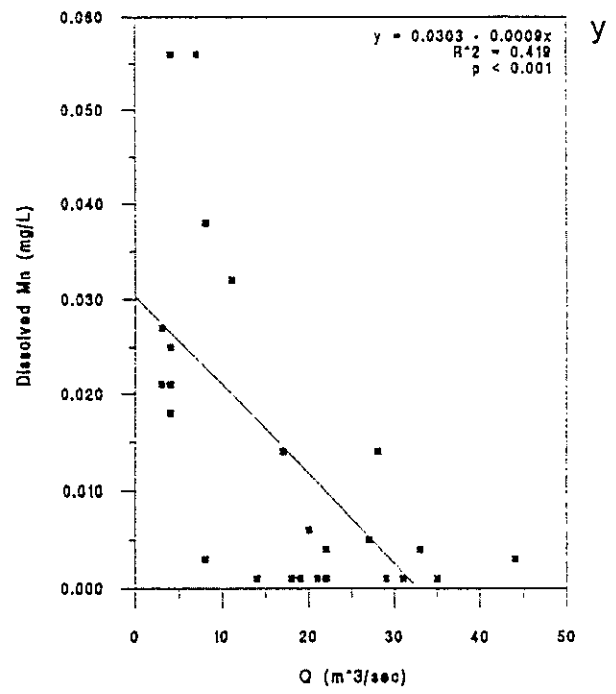
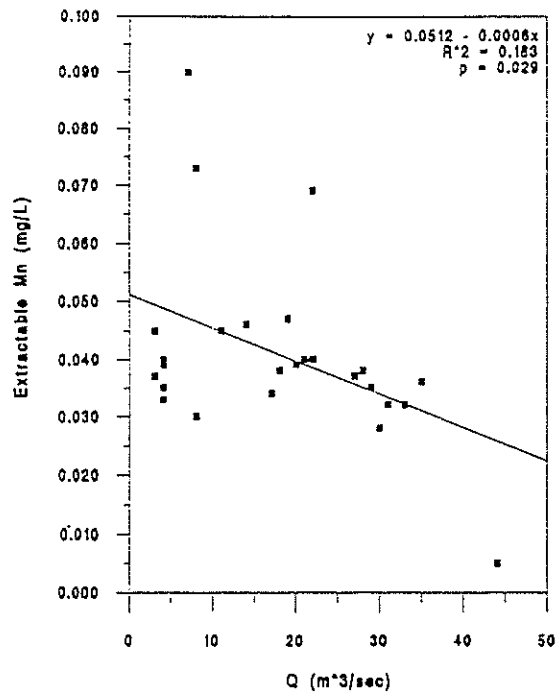


Figure 8. Continued.

3.1.3.2 Footprint River above Footprint Lake

Most of the 17 significant correlations with discharge had a negative slope, as in the Burntwood River above Threepoint Lake (Figure 9). The only positive correlates were colour and NFR (total and fixed). There was a marginally significant positive correlation with turbidity (lab) as well. Field turbidity was not correlated with discharge ($r^2=0.089$, $p=0.085$, $n=24$). Negative correlates included conductivity (field and lab), TDS, alkalinity (total and bicarbonate), hardness (total), TDN, total N, DIC, some major ions (Mg, Na, F), dissolved Ni, and dissolved As. Coefficients of determination ranged from <0.2 to >0.8 , with most between 0.15 and 0.40.

3.1.4 Seasonal Differences

3.1.4.1 Burntwood River above Threepoint Lake

There were few differences in water quality between the ice-free and ice-cover seasons in the Burntwood River above Threepoint Lake. Colour and chlorophyll a were lower under ice-cover while concentrations of dissolved Mg, Na, and Zn were higher under ice-cover than during the open water season (Figure 10). The lower chlorophyll a under ice is probably due to the imposition of light limitation on algal growth by accumulated snowcover.

The pre-development dataset for this site included just one winter of 4 measurements (Dec. 1972, and Jan., Feb., Mar, 1973). All of the measurements made under ice-cover during this pre-development study period were within the range for the open water

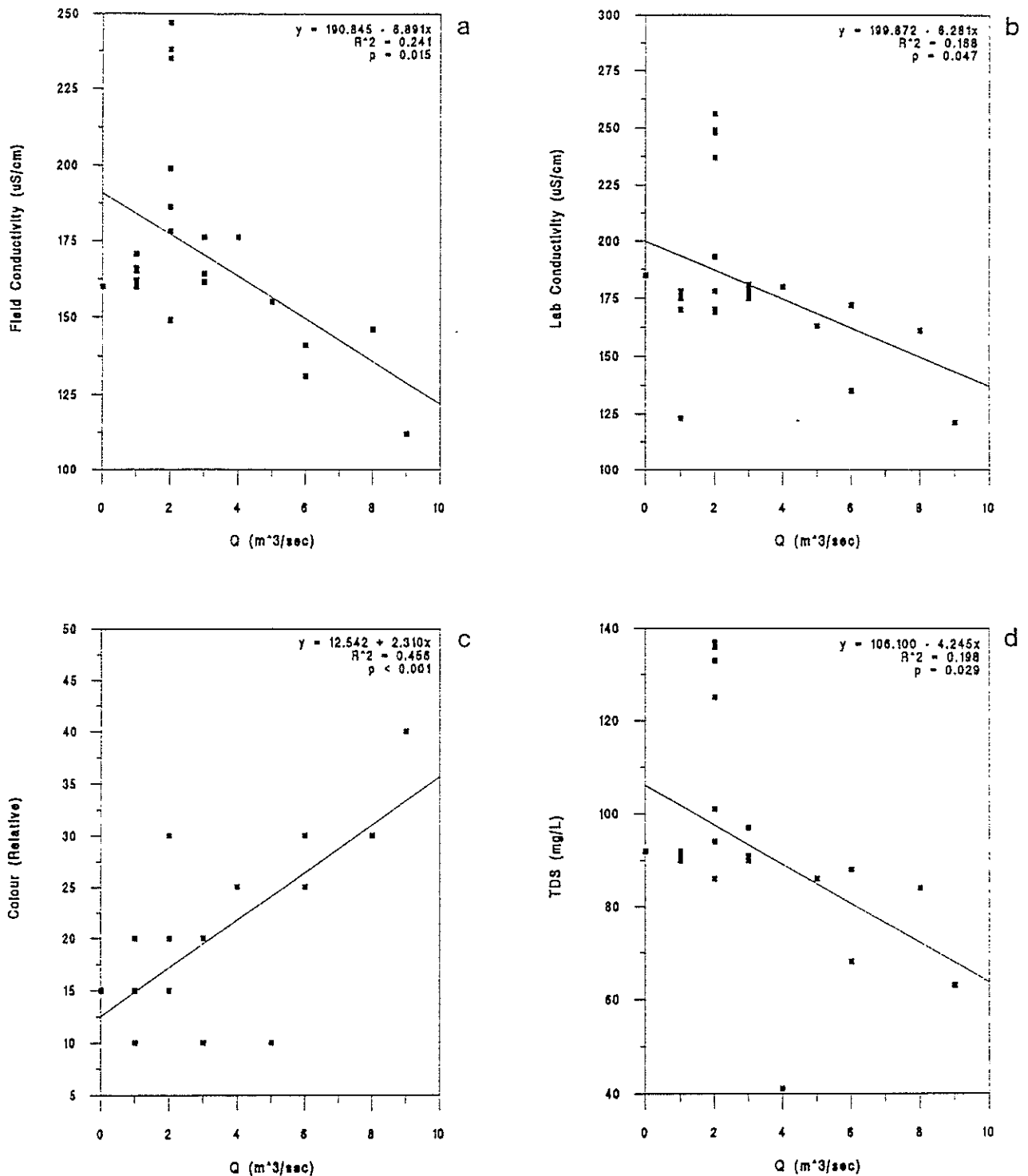


Figure 9. Water chemistry - water discharge relationships in the Footprint River above Footprint Lake, 1987-1989. a. field conductivity b. lab conductivity c. colour d. TDS e. total NFR f. fixed NFR g. total alkalinity h. bicarbonate alkalinity i. total hardness j. TDN k. total N l. DIC m. dissolved Mg n. dissolved Na o. dissolved F p. dissolved As q. dissolved Ni r. lab turbidity.

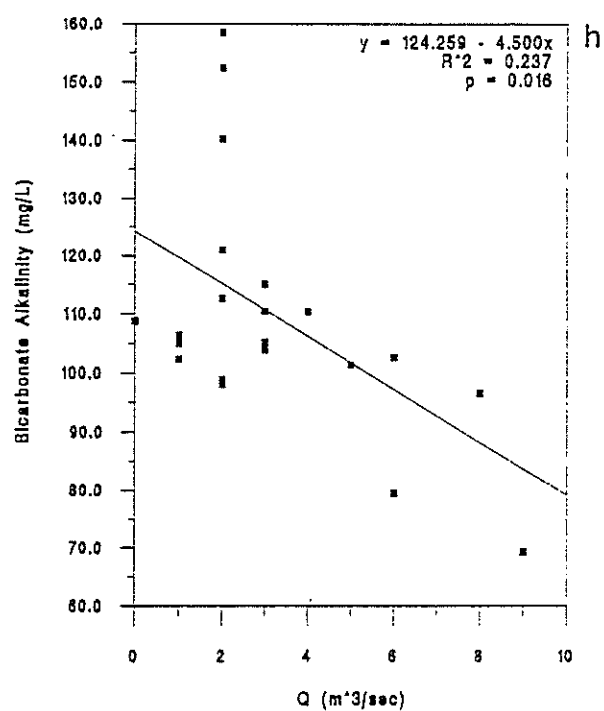
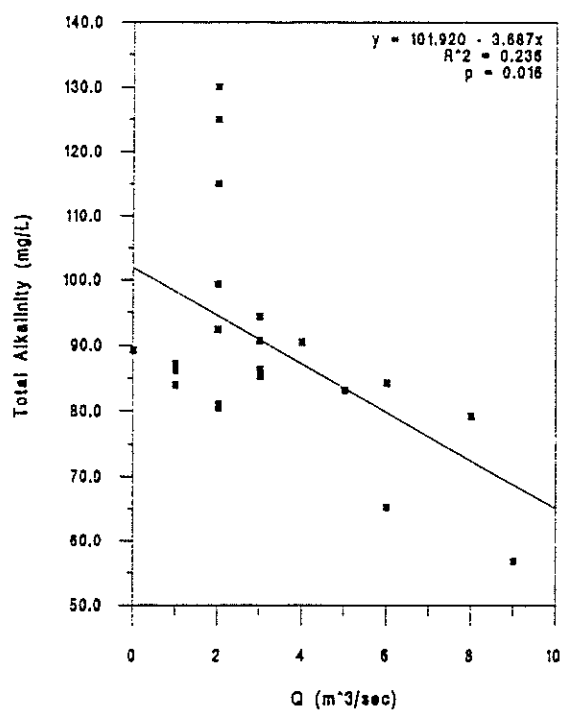
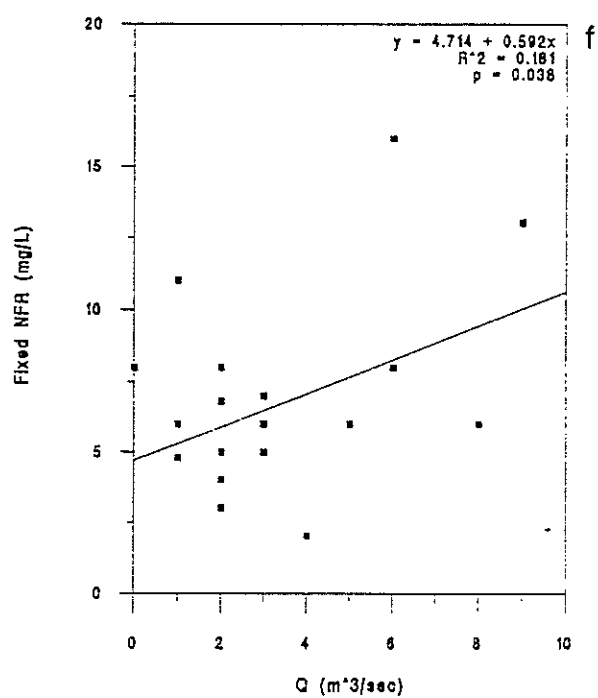
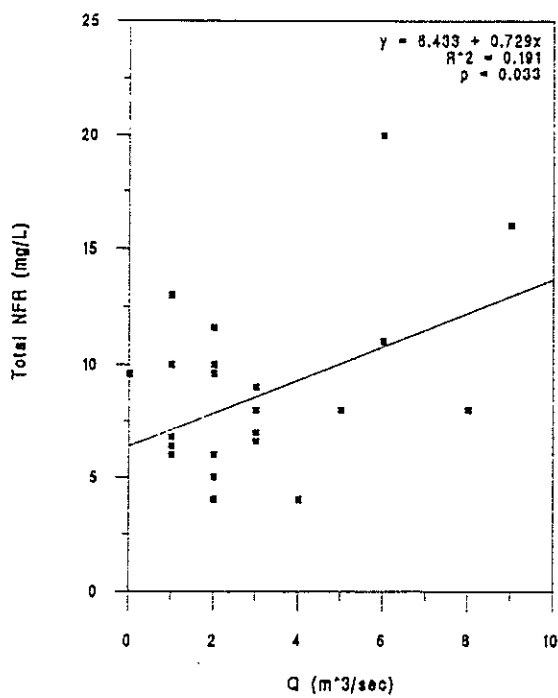


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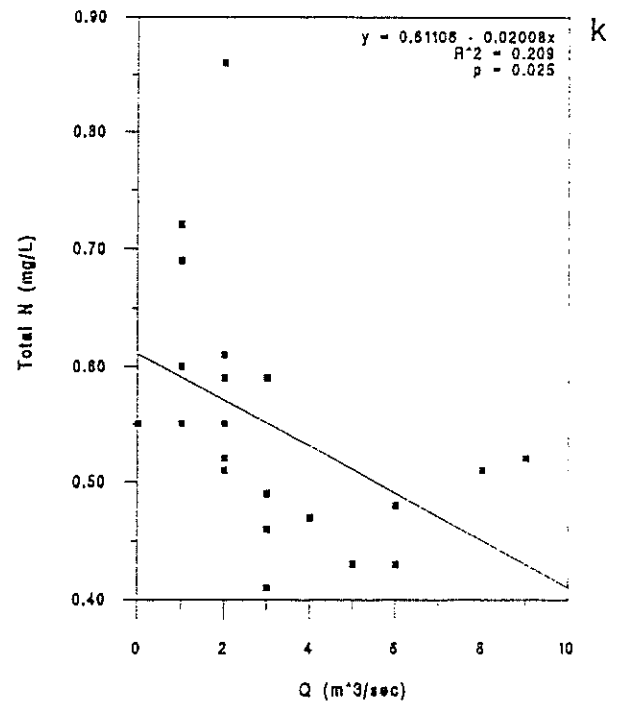
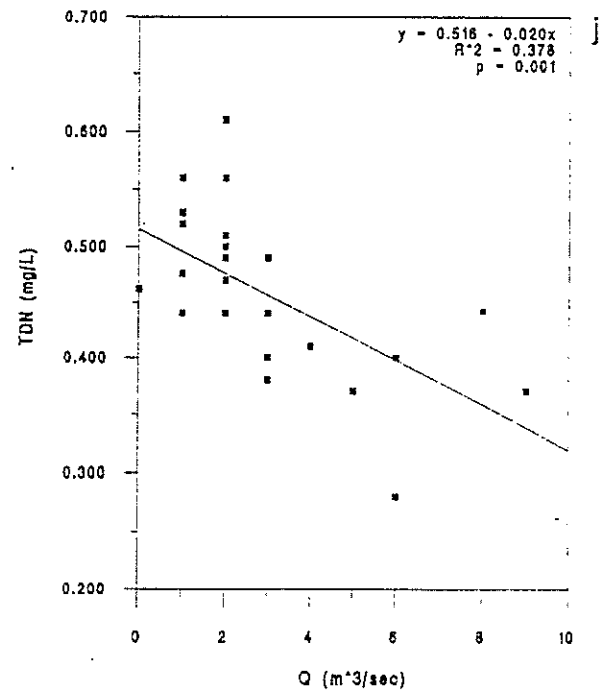
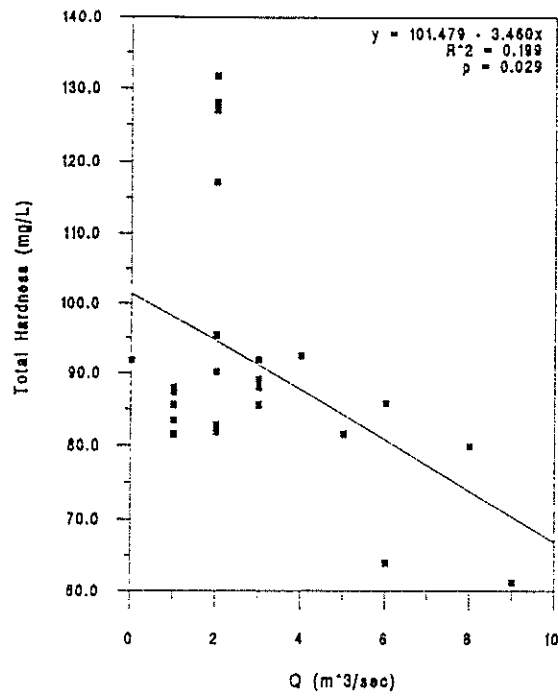


Figure 9. Continued.

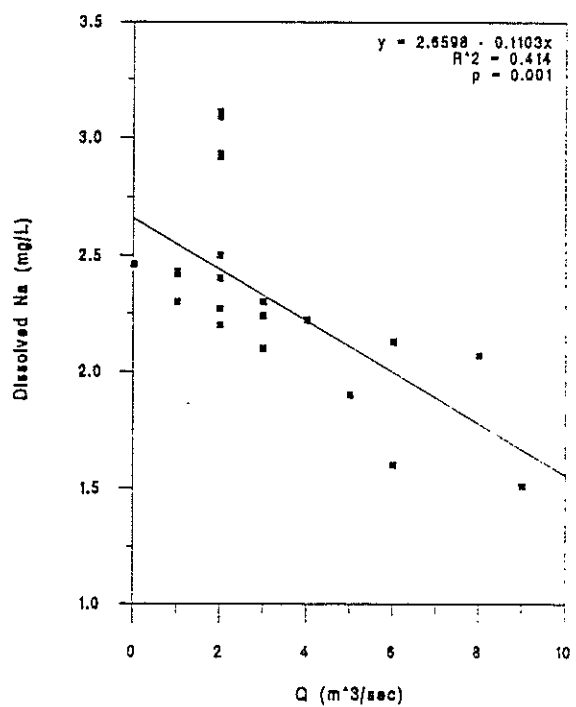
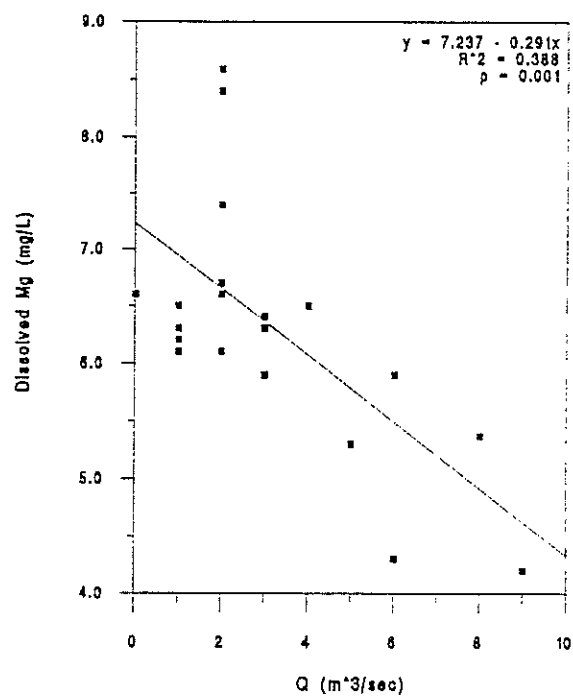
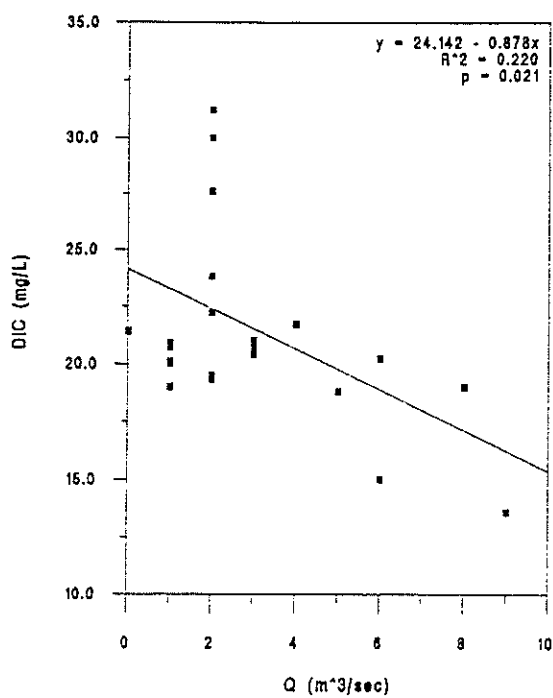


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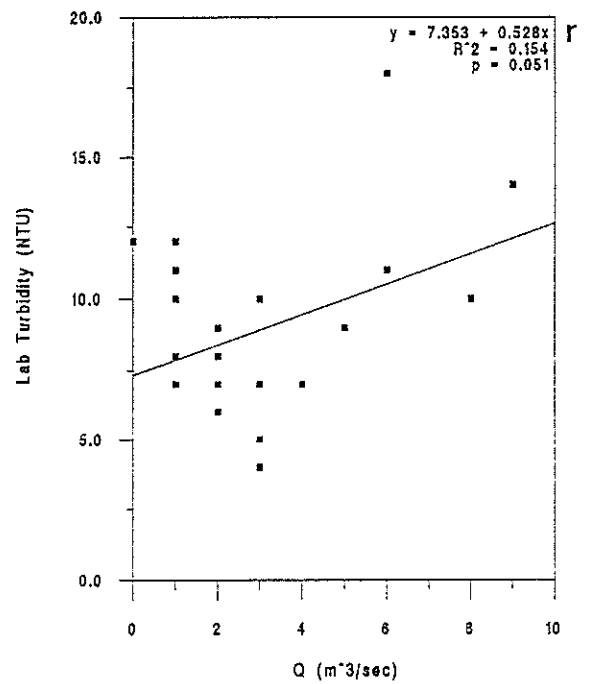
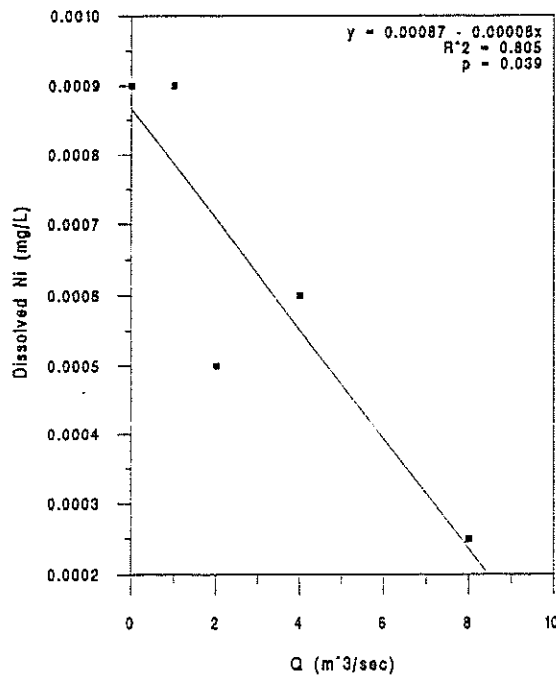
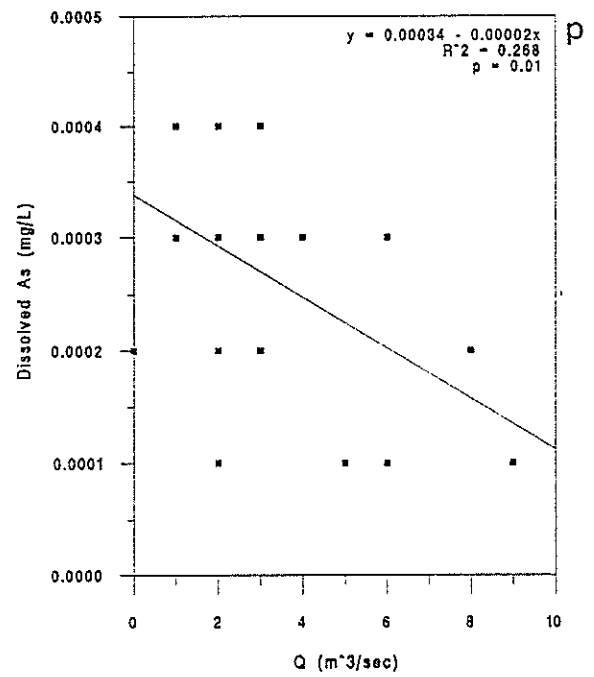
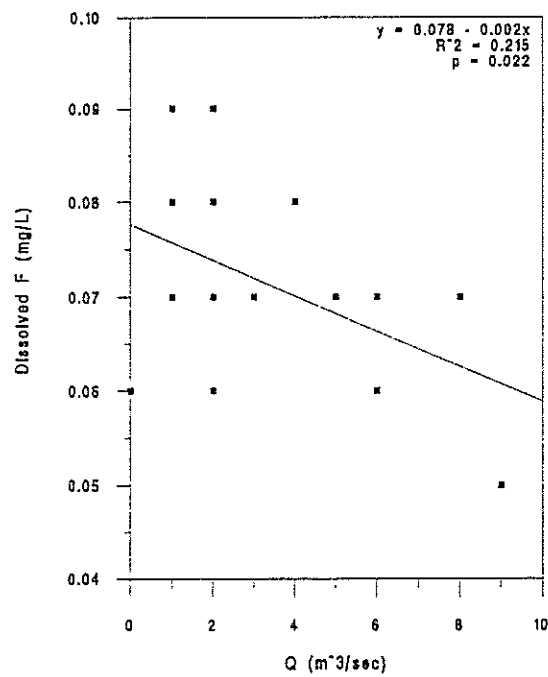


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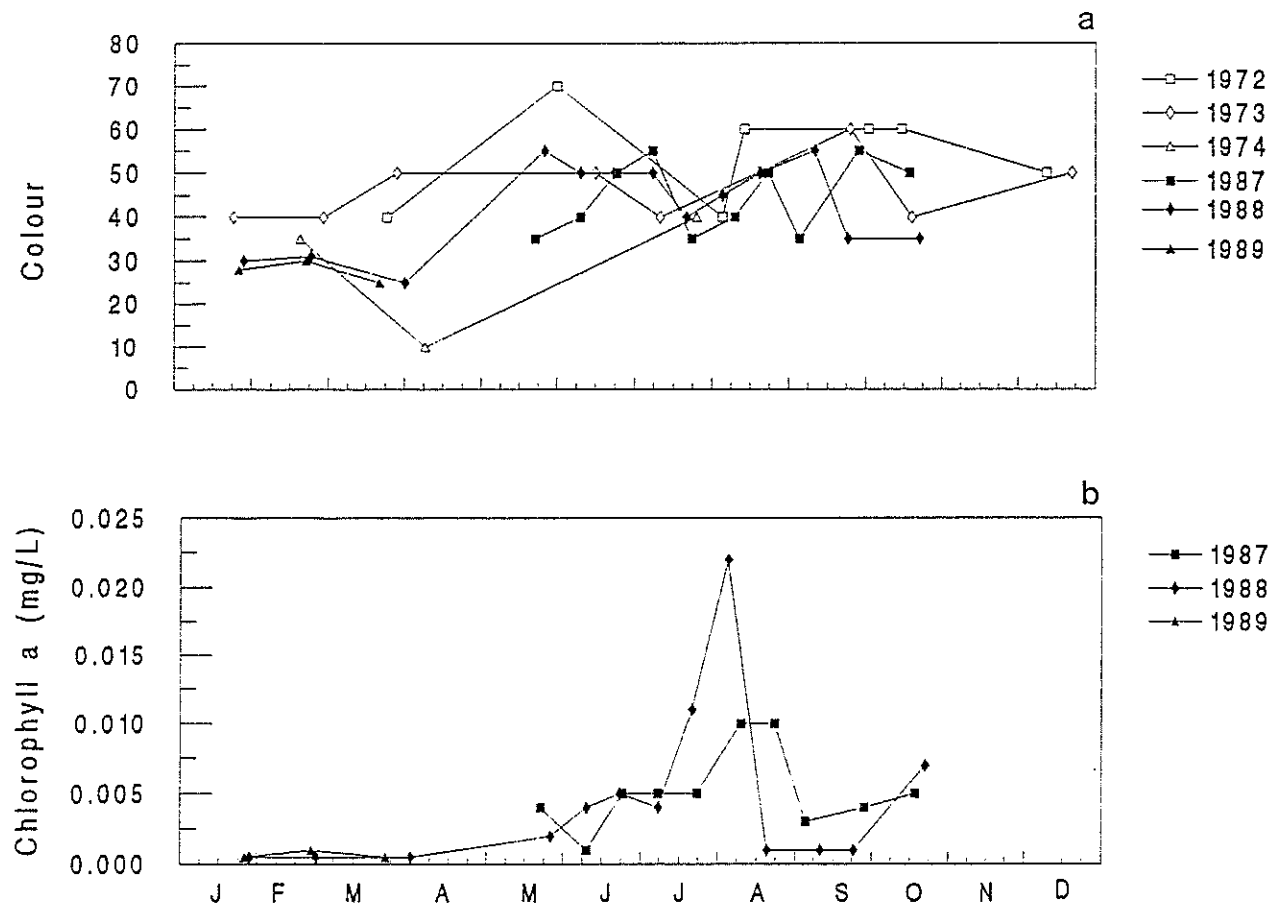


Figure 10. Seasonal water quality trends in the Burntwood River above Threepoint Lake. a. colour b. chlorophyll a c. dissolved Na d. dissolved Mg e. dissolved Zn.

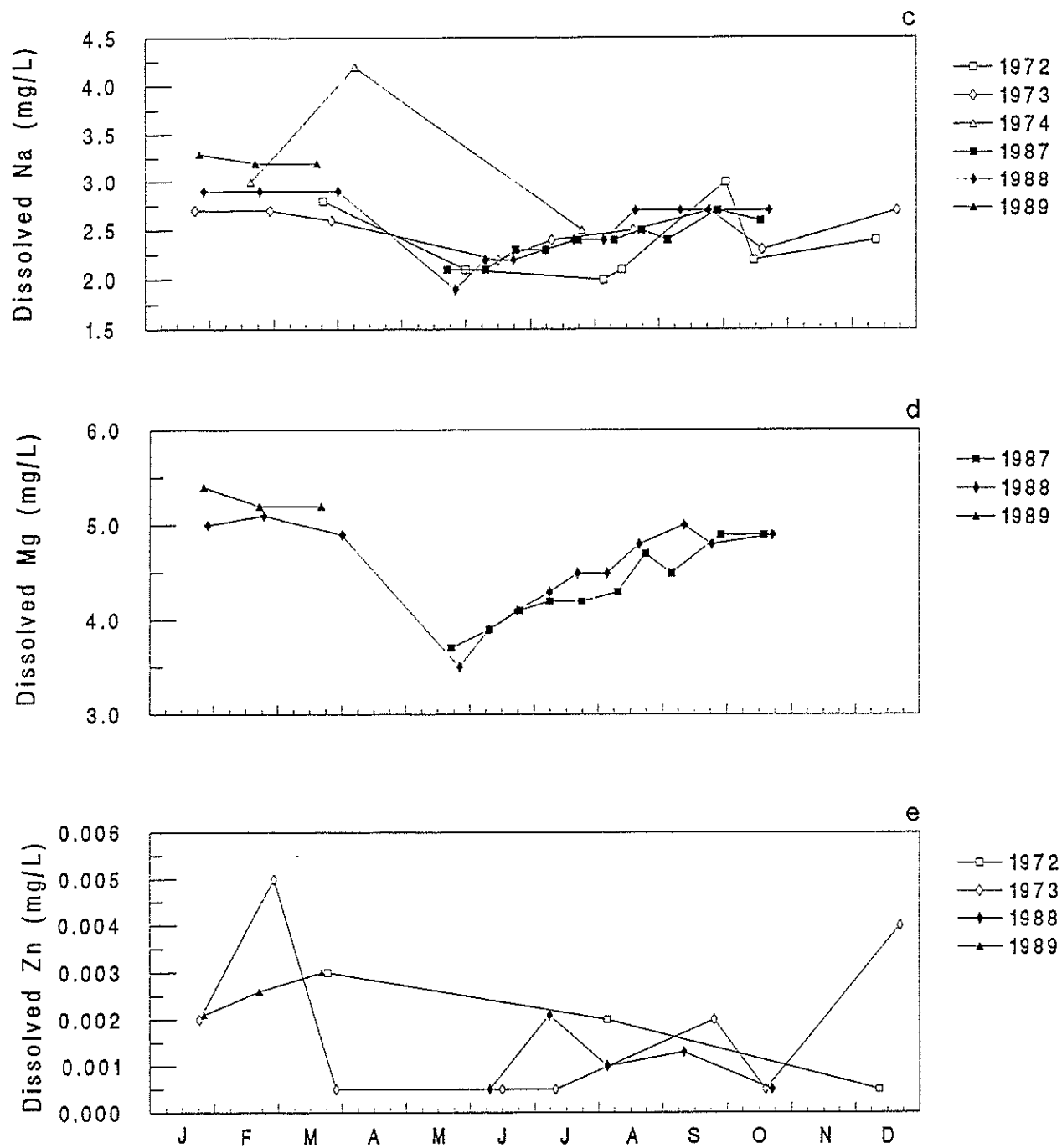


Figure 10. Continued.

season. The pre-development measurements tended to be more variable within a season, making any seasonal difference less distinct than during the FEMP study. This was the case for the pre-development Na data, in which the range during the ice-cover period overlapped that in the ice-free season although the highest values typically occurred in winter, as in the FEMP study period (Figure 10).

3.1.4.2 Footprint River above Threepoint Lake

There were clear differences in the values for many variables between the ice-free and ice-cover seasons during the 1987-1989 FEMP sampling period (Figure 11). Conductivity, alkalinity, (total and bicarbonate), hardness (total), most major ions (Ca, Mg, K, Na, Cl), reactive Si, DIC, dissolved Fe, and extractable and dissolved Mn were higher under ice cover, while dissolved Cu and chlorophyll a were lower. The lower chlorophyll a levels under ice are most probably a result of reduced light penetration through the ice and snow cover. The higher levels of some variables under ice (conductivity, hardness, alkalinity, Mg, Na, and DIC) are possibly explained by their negative correlations with river discharge and the low flows which occurred on the river in winter. Monthly mean discharges from December to April range from 1.27 to 2.49 m³ sec⁻¹ compared with 3.30 to 7.56 m³ sec⁻¹ from May to November (Water Survey of Canada 1990). This is, at best, a partial explanation as none of the other variables which increased in winter was correlated with river discharge. The lower dissolved Cu concentrations in winter also were not the product of a

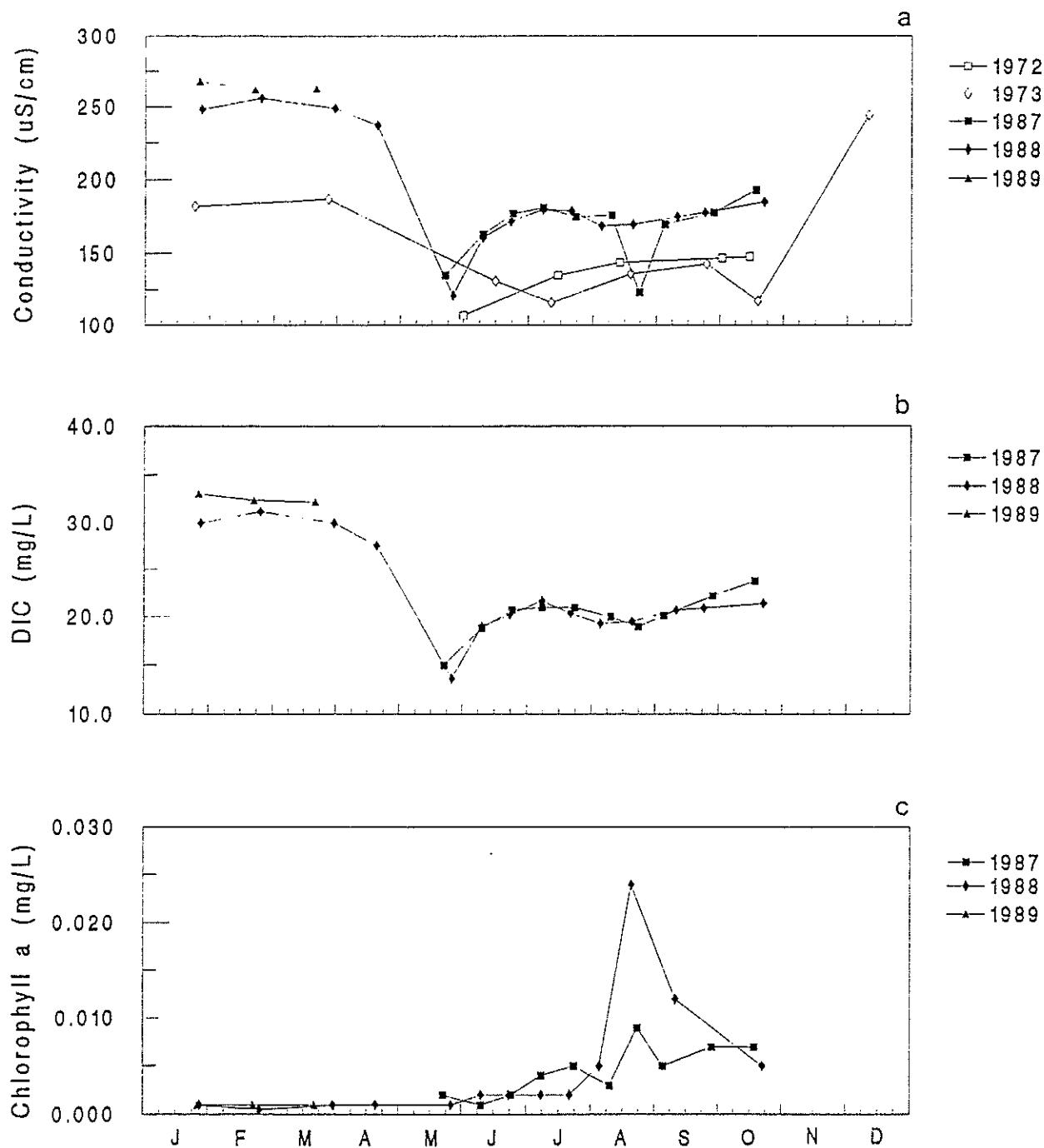


Figure 11.

Seasonal water quality trends in the Footprint River above Footprint Lake. a. lab conductivity b. DIC c. chlorophyll a d. total alkalinity e. bicarbonate alkalinity f. total hardness g. dissolved Na h. reactive Si i. dissolved Cl j. dissolved Ca k. dissolved Mg l. dissolved K m. dissolved Cu n. dissolved Zn o. dissolved Fe p. extractable Mn q. dissolved Mn.

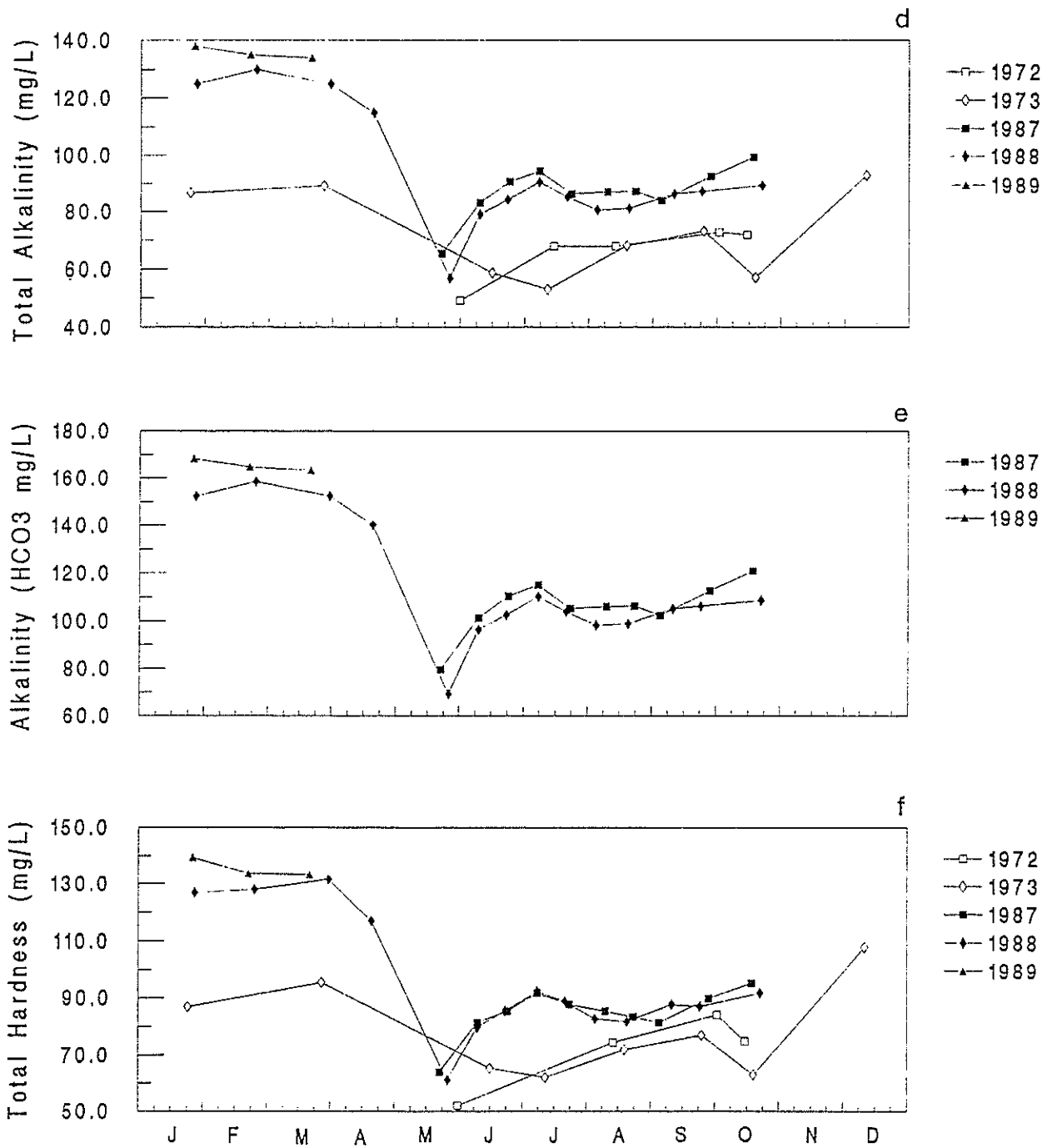


Figure 11. Continued.

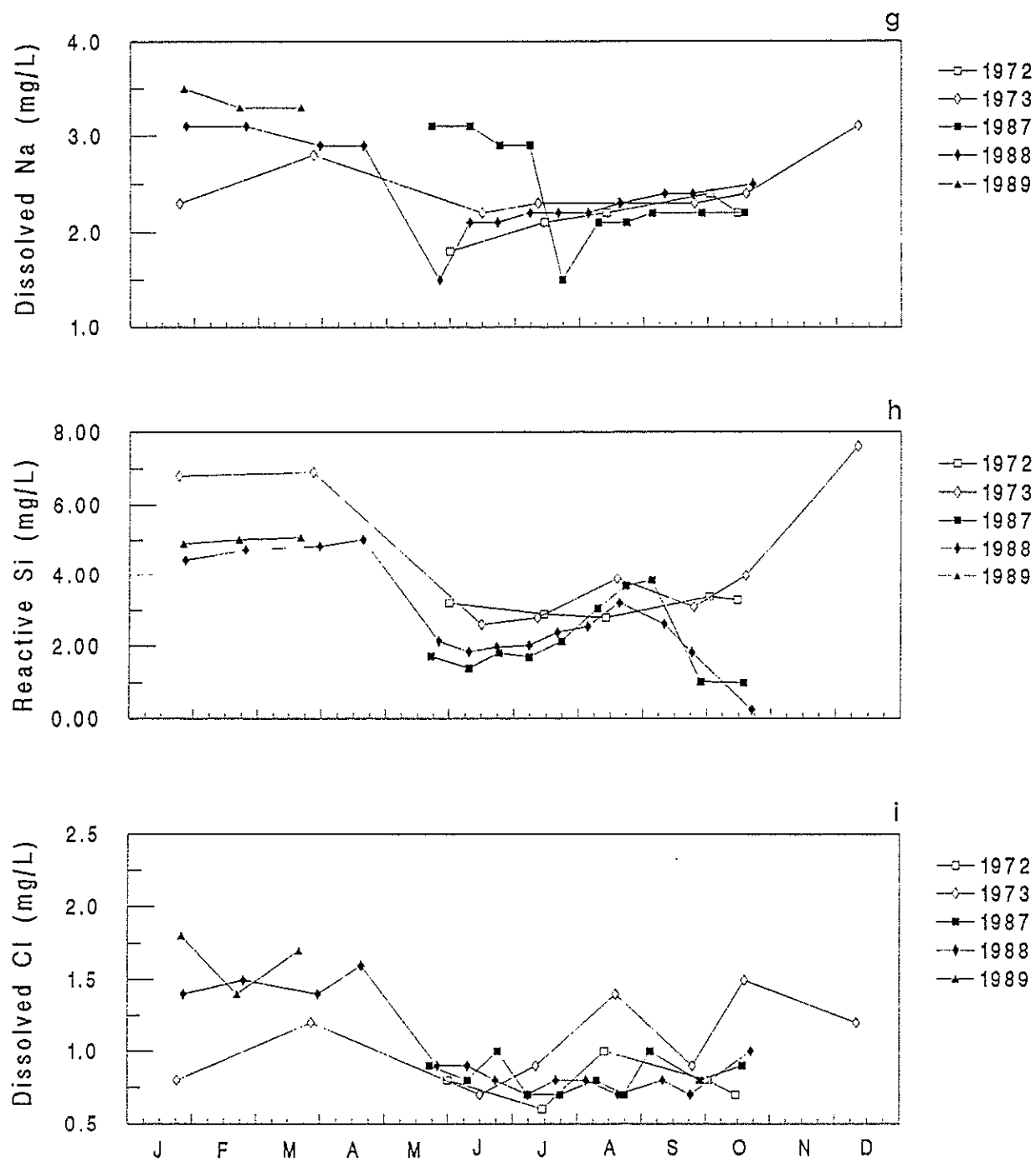


Figure 11. Continued.

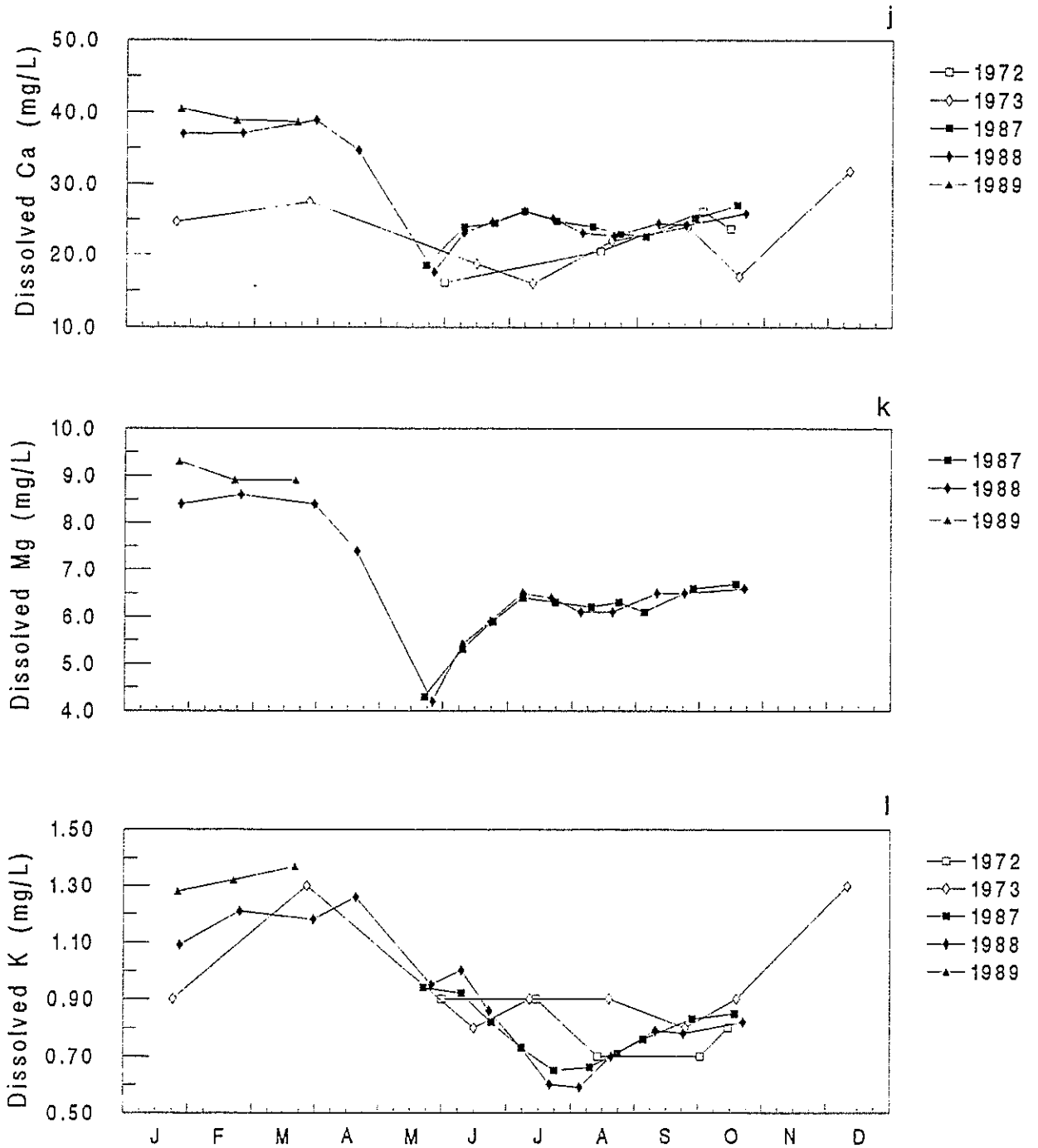


Figure 11. Continued.

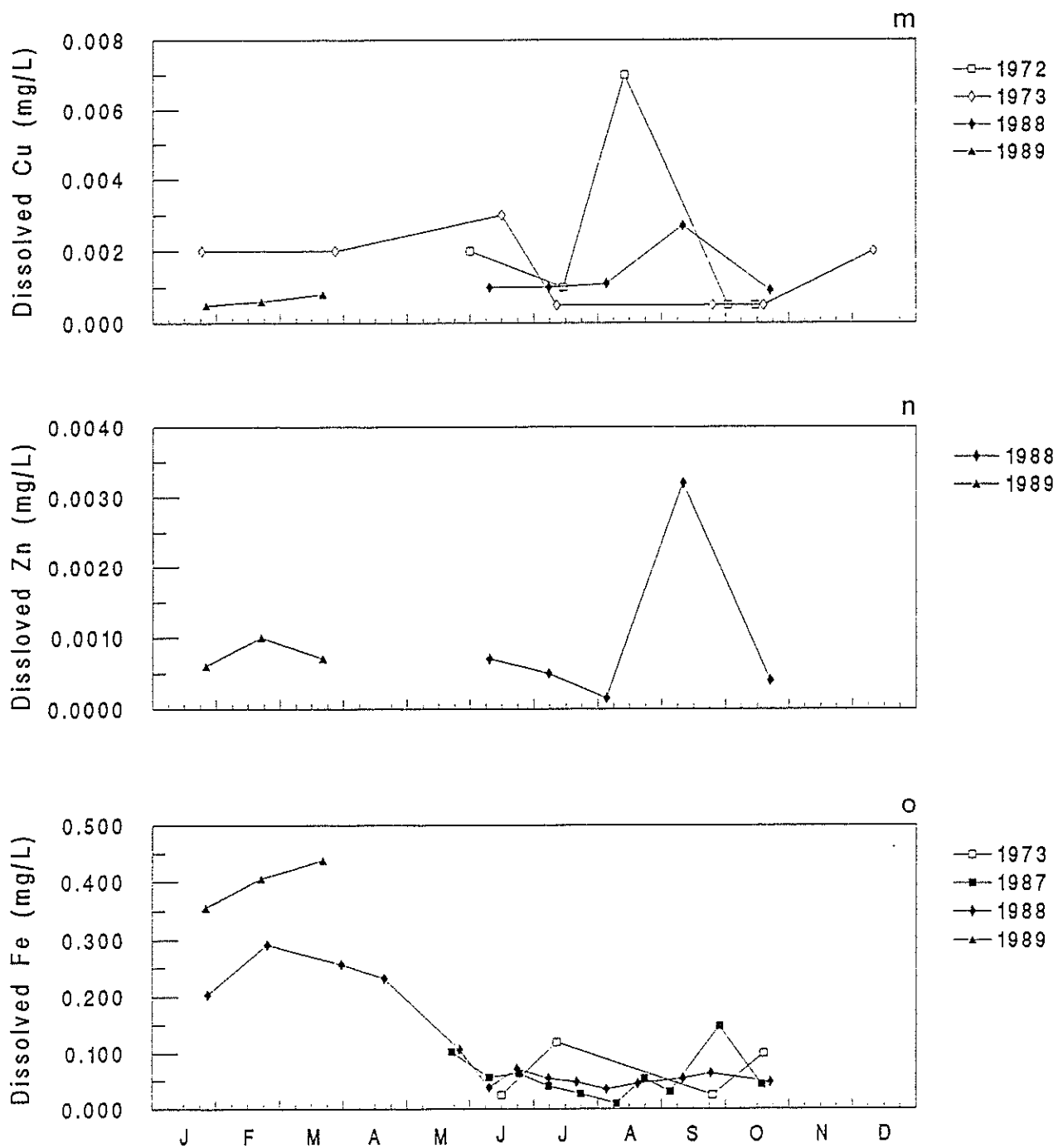


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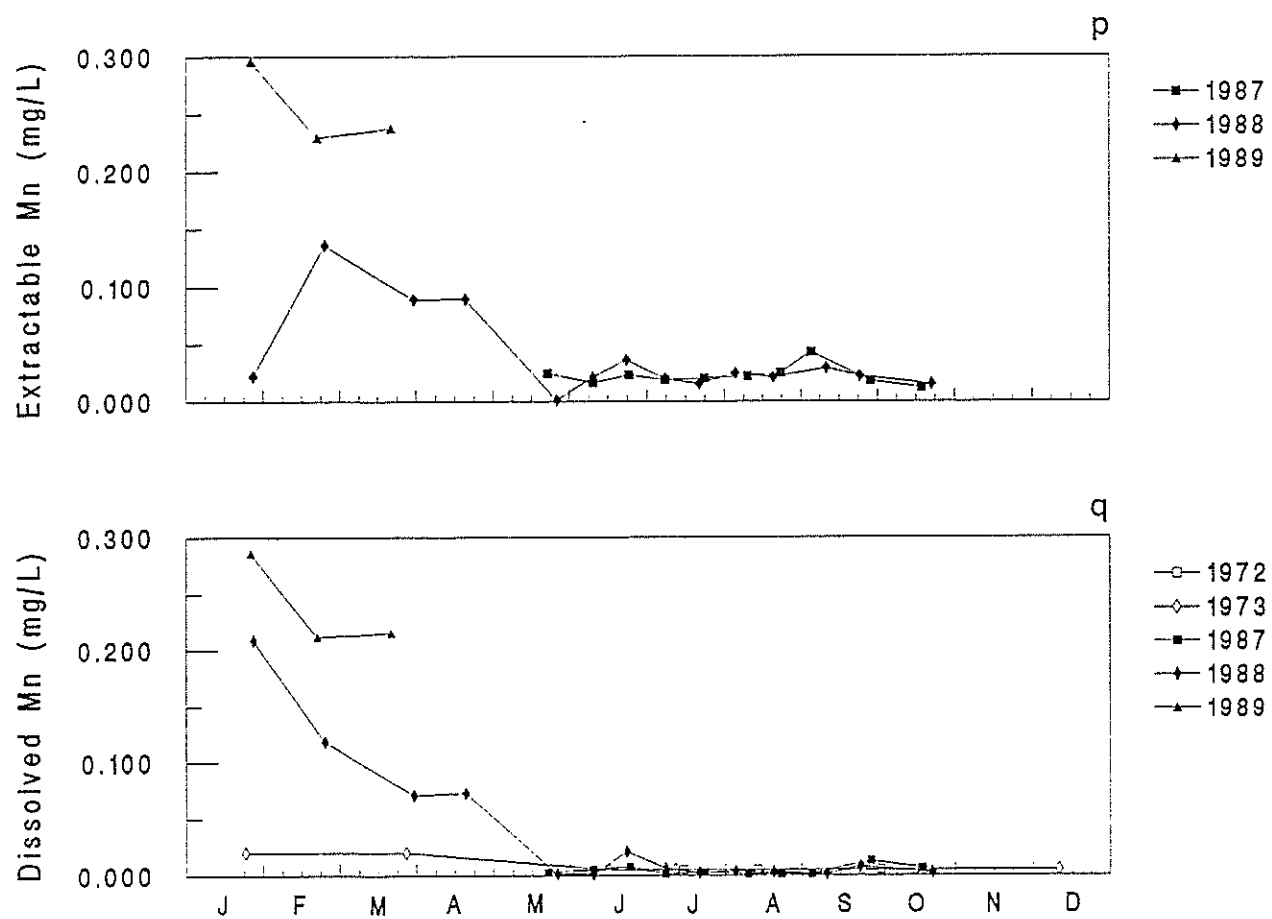


Figure 11. Continued.

relationship with discharge.

The pre-development database for this site consists of just 4 measurements of most variables under ice cover in the winters of 1972-73, and 1973-74. Qualitative examination of those variables common to the pre- and post-development periods revealed the same general pattern was present prior to development (Figure 11). Conductivity, alkalinity (total), hardness (total), ortho PO_4 , dissolved Na, K, Ca, and Mn, and reactive Si all occurred in higher concentrations under ice-cover. No clear seasonal trend was evident in dissolved Cl. None of the other variables which showed distinct seasonality during the FEMP study period was measured with sufficient frequency prior to development to examine seasonal patterns. The seasonal differences during the pre-development period are probably explained by lower discharge in winter, although the nature of any water quality - water discharge relationships during the pre-development period is unknown.

3.1.4.3 Aiken River

Distinct differences between the ice-free and ice-cover seasons were evident in 9 variables (Figure 12). Alkalinity (total and bicarbonate), hardness (total), dissolved Ca, total NH_3 , DIC, and dissolved As and Zn were higher under ice cover, while chlorophyll a was lower. This is generally the same pattern as in the Footprint River, suggesting a common cause. It is not known if any of these variables was correlated with discharge, however, due to the lack of flow measurements for this river.

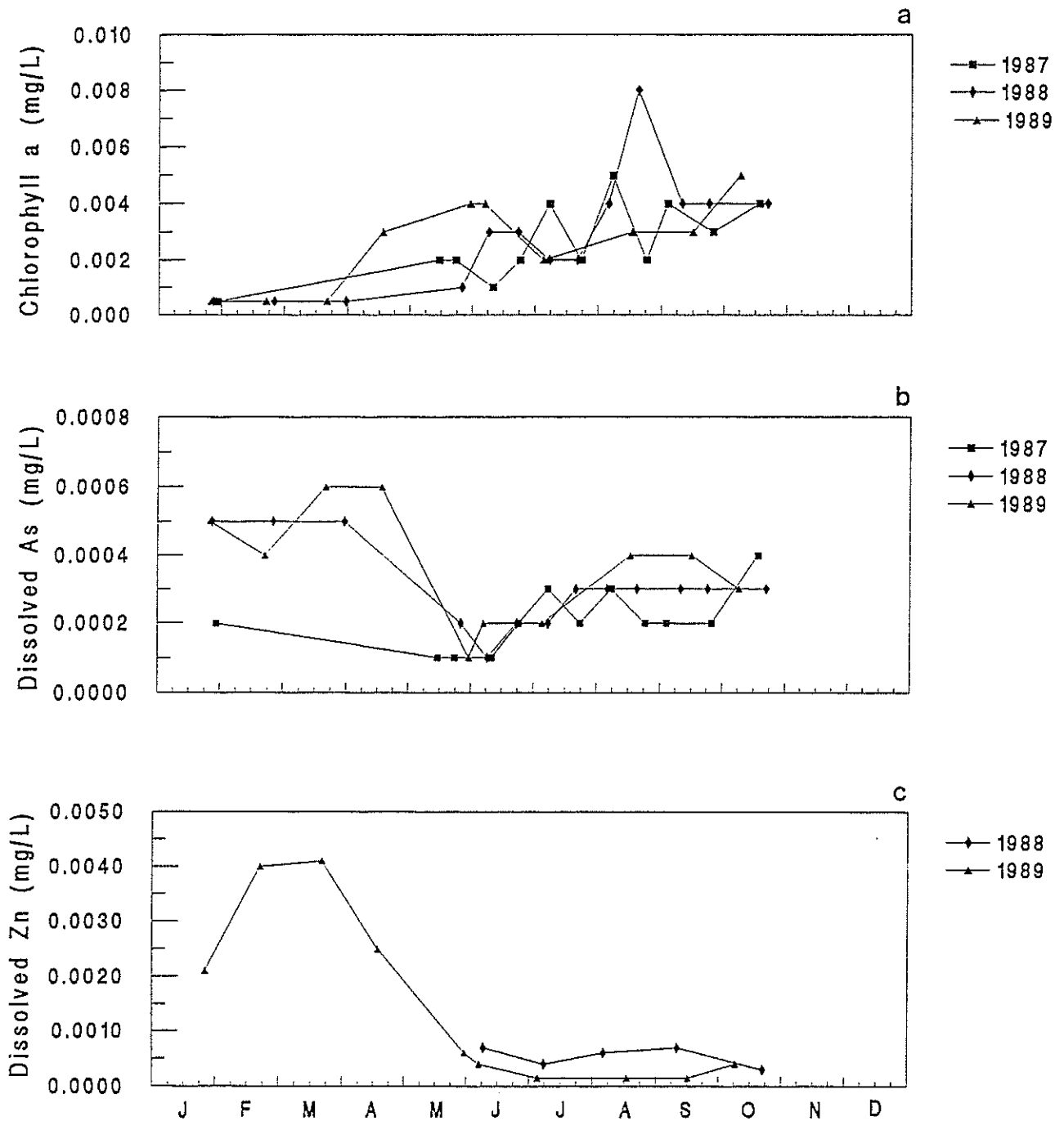


Figure 12. Seasonal water quality trends in the Aiken River. a. chlorophyll a b. dissolved As c. dissolved Zn d. dissolved Ca e. total NH_3 f. DIC g. total alkalinity h. bicarbonate alkalinity i. total hardness.

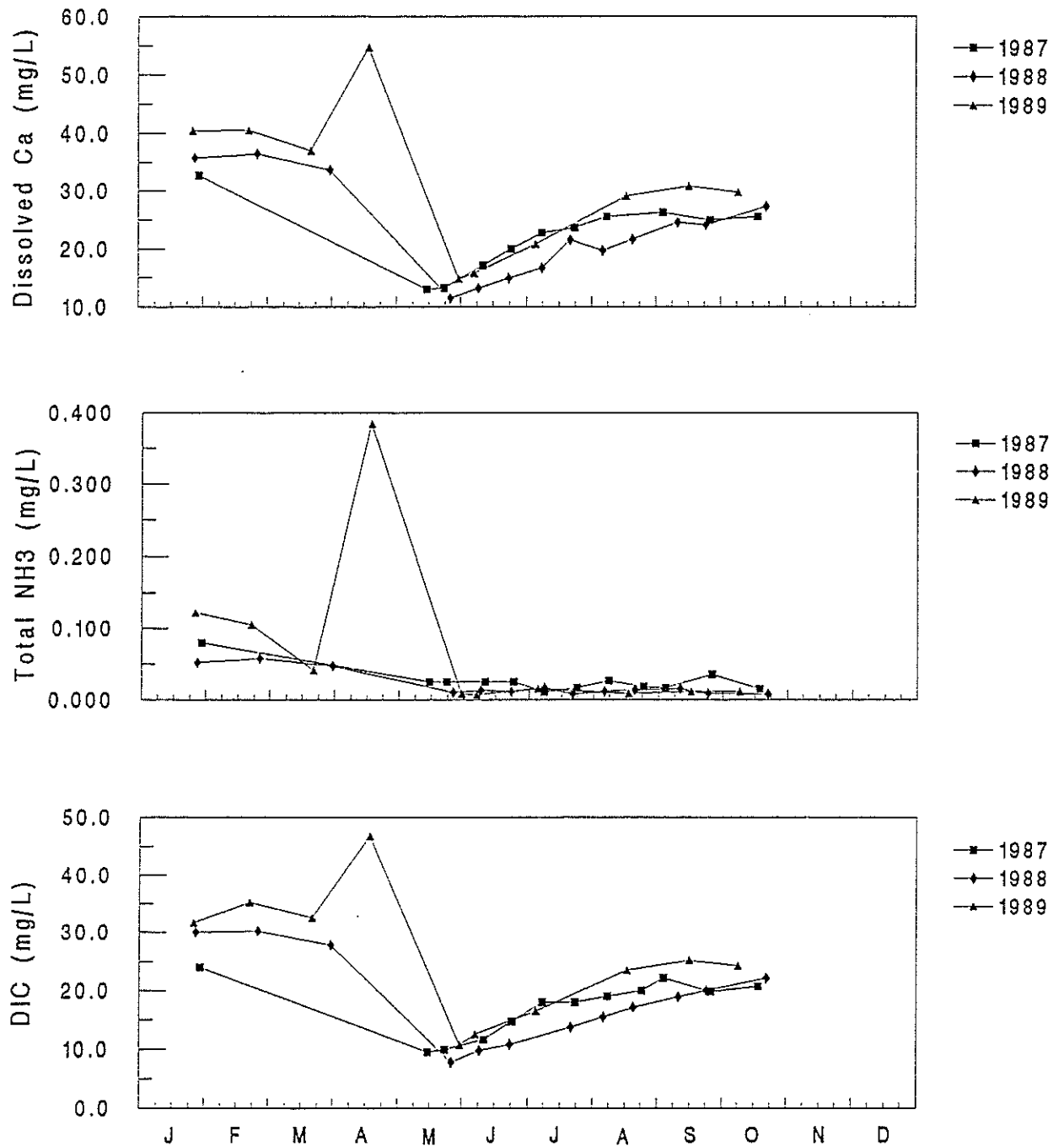


Figure 12. Continued.

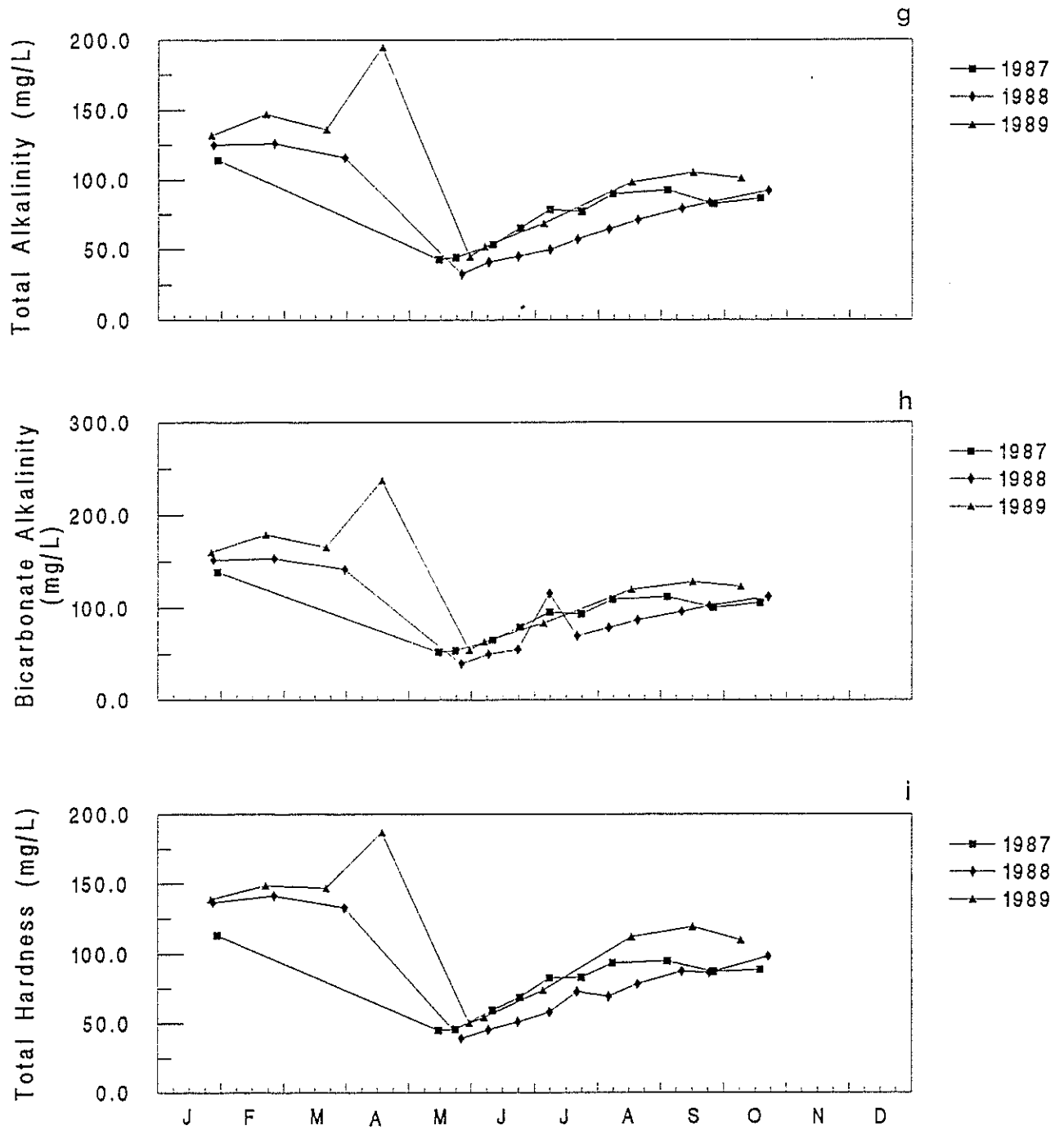


Figure 12. Continued.

3.1.5 Long-term Variation in Water Quality

Analyses of changes in water quality in the natural rivers between the pre- and post-development periods were possible only for the Burntwood and Footprint river sites. There are no pre-development water quality data for the Aiken River site.

Statistical comparisons of changes between the 1972-1973 pre-development study period and the 1987-1988 post-development period were possible for 17 variables at both sites (Tables 10 and 11). The only significant change detected in the Burntwood River was a decrease in pH, although marginally significant ($0.05 < p < 0.08$) decreases in dissolved Ca and SO_4 also occurred. A significant decrease in pH also was found in the Footprint River, along with decreases in NO_2 - NO_3 and SO_4 and a marginally significant increase in alkalinity (total). These findings indicate a good measure of long-term stability in the chemistry of these natural rivers.

The decreases in pH at both sites probably were an artefact of undocumented modifications in analytical equipment or procedures. Equipment (e.g., probe sensitivity) and procedures (e.g., agitation for a faster but incorrect reading) for pH measurement have changed considerably in the past 15 years (M. Stainton, DFO Winnipeg, pers. comm.) although such modifications in what appears to be a simple analysis often are not considered to represent a change in method. The decreases in pH also are inconsistent with the other changes in water chemistry which occurred between the pre- and post-development study periods. An increase in acid deposition should increase the SO_4 concentration and decrease alkalinity (Schindler

Table 10. Evaluation of the significance of changes in selected water quality parameters in the Burntwood River above Threepoint Lake between 1972-1973 and 1987-1988. X_{pre} denotes the mean for the years 1972 and 1973. X_{post} denotes the mean for the years 1987 and 1988. p is the probability of Student's t -statistic for the comparison.

	pH	Conductivity $\mu S\ cm^{-1}$	Turbidity NTU	Tot Alk $mg\ L^{-1}$	Tot Hard $mg\ L^{-1}$	TOC $mg\ L^{-1}$	NO_2NO_3 $mg\ L^{-1}$	Ortho PO_4 $mg\ L^{-1}$	Tot P $mg\ L^{-1}$	Diss Na $mg\ L^{-1}$
Burntwood River										
p	0.028	0.681	0.202	0.953	0.222	0.250	0.226	0.553	0.295	0.394
X_{pre}	7.8	110.	14.	50.2	55.6	14.2	0.290	0.004	0.027	2.3
X_{post}	7.43	112. ¹	19.	50.0 ¹	51.1 ¹	12.78 ¹	0.048 ¹	0.003	0.036	2.5 ¹
	Diss K $mg\ L^{-1}$	Diss Cl $mg\ L^{-1}$	Diss F $mg\ L^{-1}$	Diss SO_4 $mg\ L^{-1}$	React Si $mg\ L^{-1}$	Diss Fe $mg\ L^{-1}$	Diss Mn $mg\ L^{-1}$	Diss Cu $mg\ L^{-1}$	Diss Pb $mg\ L^{-1}$	Diss Zn $mg\ L^{-1}$
p	0.859	1.00	1.00	0.066	0.095	0.240	i.d.	i.d.	i.d.	i.d.
X_{pre}	1.0	1.0	0.07	3.3	3.6	0.10	<0.01	0.091	0.059	0.062
X_{post}	0.94 ¹	1.0 ¹	0.07	2.4	3.16 ¹	0.125 ¹	0.011 ¹	0.0017	<0.0007	0.0011

i.d. Insufficient data.

¹ Discharge-standardized variables.

Table 11. Evaluation of the significance of changes in selected water quality parameters in the Footprint River above Footprint Lake between 1972-1973 and 1987-1988. X_{pre} denotes the mean for the years 1972 and 1973. X_{post} denotes the mean for the years 1987 and 1988. p is the probability of Student's t -statistic for the comparison.

	pH	Conductivity $\mu S\ cm^{-1}$	Turbidity NTU	Tot Alk $mg\ L^{-1}$	Tot Hard $mg\ L^{-1}$	TOC $mg\ L^{-1}$	NO_2NO_3 $mg\ L^{-1}$	Ortho PO_4 $mg\ L^{-1}$	Tot P $mg\ L^{-1}$	Diss Na $mg\ L^{-1}$
Footprint River										
p	0.027	0.193	0.162	0.062	0.186	0.492	0.001	0.293	0.499	0.939
X_{pre}	7.9	146.	7.	68.9	75.0	18.0	0.577	0.004	0.026	2.3
X_{post}	7.50	179. ¹	9.	90.2 ¹	90.1 ¹	11.78	0.020	0.003	0.030	2.3 ¹
	Diss K $mg\ L^{-1}$	Diss Ca $mg\ L^{-1}$	Diss F $mg\ L^{-1}$	Diss SO_4 $mg\ L^{-1}$	React Si $mg\ L^{-1}$	Diss Fe $mg\ L^{-1}$	Diss Mn $mg\ L^{-1}$	Diss Cu $mg\ L^{-1}$	Diss Pb $mg\ L^{-1}$	Diss Zn $mg\ L^{-1}$
p	0.677	0.189	0.106	0.042	0.248	0.758	i.d.	i.d.	i.d.	i.d.
X_{pre}	0.9	22.2	0.06	4.5	3.9	0.08	<0.01	0.002	<0.001	0.003
X_{post}	0.85	25.8	0.08 ¹	1.8	2.49	0.084	0.021	0.0013	<0.0007	0.0010

i.d. Insufficient data.

¹ Discharge-standardized variables.

et al. 1980), yet decreases in SO_4 appear to have occurred at both sites and an increase in alkalinity occurred in the Footprint River.

There is no obvious explanation for the 25-60% decreases in SO_4 at both sites nor for the almost 96% decrease in $\text{NO}_2\text{-NO}_3$ in the Footprint River. The same declining trend in $\text{NO}_2\text{-NO}_3$ was evident in the Burntwood River as well, although the 83% lower value during the FEMP study was not significantly different from the pre-development value (Table 10). The higher mean concentrations of both parameters during the pre-development period were not the product of outliers or occasional high values, concentrations of both variables were consistently higher throughout the pre-development study period.

It is particularly difficult to understand how the much lower $\text{NO}_2\text{-NO}_3$ concentrations during the FEMP study could be a product of natural variation. Measurements of NO_3 concentrations reported in a pre-development study of the Rat-Burntwood system suggest problems with the analytical method used to produce the $\text{NO}_2\text{-NO}_3$ data in the NAQUADAT file may be the cause. Concentrations of NO_3 in the Rat-Burntwood system, including Threepoint Lake, reported by Cleugh (1974) were in the range of <0.001 to 0.007 mg L^{-1} between June and September, with values in the range of 0.050 to 0.125 mg L^{-1} under ice cover. Concentrations of $\text{NO}_2\text{-NO}_3$ measured during the FEMP study were comparable to the NO_3 concentrations reported by Cleugh. Although Cleugh just measured NO_3 , this is the dominant contributor to $\text{NO}_2\text{-NO}_3$ in the Rat-Burntwood system. Concentrations of NO_2 in the

system typically are $<0.001 \text{ mg L}^{-1}$ between June and September (Berger and Ramsey 1991). Long-term monitoring of NO_3 levels in natural waters at the Experimental Lakes Area in northwestern Ontario, using the same method employed by Cleugh, have not shown the dramatic decreases evident in the NAQUADAT data. The trend is for slightly increasing NO_3 concentrations due to increasing atmospheric deposition (R.E. Hecky, DFO, Winnipeg, MB. pers. comm.). Because of the uncertain validity of the pre-development NO_2 - NO_3 data in the NAQUADAT file, any decreases detected in the analysis of temporal changes associated with hydroelectric development which were based on NAQUADAT data were considered to be an artefact unless they were supported by measurements from other sources.

There appears to be a similar problem with the pre-development SO_4 measurements. Decreases of 31 to 60% seem to have occurred in the Burntwood and Footprint rivers and in the Churchill River near Leaf Rapids (pre: 3.6 mg L^{-1} , NAQUADAT; post: 2.5 mg L^{-1} , G. McCullough, DFO, Winnipeg, MB. unpubl. data.), all natural sites, between the pre-development study period and the FEMP study. Again it seems unlikely that such a change might naturally occur in 3 separate drainage basins. A change in SO_x emissions from the INCO smelter in Thompson also can be ruled out as all 3 sites lie upwind of this source. Again, the most likely cause of this shift is a change in analytical methods or laboratories. Because of this uncertainty in the pre-development SO_4 data, it generally was not possible to determine if the hydroelectric developments had any

impact on SO₄ concentrations.

The pre-development database unfortunately is deficient in its coverage of metal species. In particular, measurements of extractable or total metals generally were made on fewer than 4 dates between 1971 and 1974. Several dissolved metal species were routinely analyzed (Tables 10 and 11), although occasionally with higher limits of detection than employed in the FEMP study. The FEMP study also contributed to this poor coverage of metals with its limited examination of dissolved species. None of dissolved Mn, Cu, Pb, or Zn was analyzed in 1987, and sampling was terminated after March 1989, resulting in just one full calendar year of study. This eliminated the possibility of a statistical analysis for these species. Nevertheless, qualitative comparison of the available data suggests dissolved Cu, Pb, and Zn levels perhaps were lower during the FEMP study, while dissolved Mn may have been higher (Tables 10 and 11). There was no significant change in the dissolved Fe concentration.

3.1.6 Uses of the Natural River Data

The natural rivers were originally included in the FEMP study in the hope that they might serve as surrogates for pre-development conditions at sites where historical data were deficient or lacking (Seidl 1987). However, there is no means of establishing the validity for their general use as control sites. This would require a long-term comparison of water quality in the regulated and natural rivers prior to development to demonstrate similarities in

water quality and in their responses to variations in climatic conditions. This sort of comparison was not possible for any of the natural rivers included in this study because of the limited pre-development databases for all of the sites. Data from the site on the Burntwood River above Threepoint Lake clearly are useful for interpretation of the cause and nature of water quality changes in the lower Burntwood River below Threepoint Lake, as this was the dominant source of flow prior to diversion. It is quite unlikely that current water quality in the natural rivers would be quantitatively comparable to pre-development water quality in the Nelson or Churchill rivers due to the large differences in discharge and drainage area. The natural rivers would be dominated by the influence of local geology and soils, while the larger Churchill and Nelson integrate the effects of geology, soils, land use, and climate from as far west as the Rocky Mountains. Therefore, the natural river data were not generally employed as controls in the analysis of water quality changes due to hydroelectric development although the Burntwood and Footprint river data were used to assist in identification of the cause of water quality changes on the Rat and Burntwood rivers. The natural river data also are useful for interpretation of chemical budgets in the Churchill-Nelson system as they can be used to define the limits of natural runoff contributions to chemical fluxes in the regulated system.

3.2 Norway House Area

3.2.1 Nelson River below Sea River Falls

3.2.1.1 Current Water Quality

With a few important exceptions, physical and chemical water quality at this site generally was well within Canadian guidelines for human consumption, recreational use, and the protection of aquatic life. The exceptions included turbidity, which always exceeded the drinking water guideline, and extractable Al and total Cu, for which the mean concentrations exceeded the respective guidelines for protection of aquatic life by factors of 1.8 and 2.4 (Table 12). Other variables (extractable Fe and Mn, colour) occasionally exceeded the drinking water guideline. These sporadic exceedences are characteristic of natural rivers and streams in northern Manitoba, occurring at all of the sites sampled in this study.

3.2.1.2 Water Quality - Water Discharge Relationships

A number of variables were negatively correlated with Nelson River discharge during the 1987-1989 study period (Figure 13). These included pH (lab), alkalinity (total and bicarbonate), hardness (total and non-carbonate), conductivity (field and lab), TDS, most major ions (Ca, Mg, K, Na, Cl, SO₄), dissolved NO₂-NO₃, total NH₃ and DIC. Positive correlates with discharge included turbidity (field) and NFR (total and fixed), particulate N, POC, and extractable Fe. The sparse pre-development water quality data precluded an analysis of changes in discharge relationships since

Table 12. Summary statistics for physical and chemical water quality variables measured in the Nelson River below Sea River Falls; January, 1987, to September, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.8	0.3	8.3	7.3
	Lab		7.86	0.20	8.13	7.49
Conductivity	Field	$\mu\text{S cm}^{-1}$	292	30	331	220
	Lab		309	36	355	212
Turbidity	Field	NTU	10 ⁺	3	16 ⁺	6 ⁺
	Lab		8 ⁺	3	13 ⁺	4
Colour	True	Relative	9	4	20 ⁺	3
TDS		mg L^{-1}	163	19	189	124
NFR	Total	mg L^{-1}	6	3	10	3
	Fixed	mg L^{-1}	5	3	10	2
Alkalinity	Total	mg L^{-1}	103.0	10.6	120.0	82.3
	Bicarbonate	mg L^{-1}	125.6	12.9	146.3	100.3
Hardness	Total	mg L^{-1}	117.7	12.9	138.7	89.4
	Non-carbonate	mg L^{-1}	14.7	3.6	20.9	4.5
Free CO ₂		mg L^{-1}	3.53	2.53	8.62	0
Phosphorus	Particulate	mg L^{-1}	0.013	0.007	0.026	<0.006
	Ortho PO ₄	mg L^{-1}	<0.003	0.002	0.010	<0.003
	Total Dissolved	mg L^{-1}	0.010	0.005	0.023	0.004
	Total	mg L^{-1}	0.023	0.006	0.032	0.010
Nitrogen	Particulate	mg L^{-1}	0.051	0.022	0.090	<0.010
	Total Dissolved	mg L^{-1}	0.300	0.034	0.370	0.230
	NO ₂ -NO ₃	mg L^{-1}	0.013	0.014	0.047	<0.010
	Total NH ₃	mg L^{-1}	0.023	0.012	0.050	0.006
	Total	mg L^{-1}	0.35	0.04	0.43	0.27
Carbon	DOC	mg L^{-1}	7.63	0.67	8.98	6.55
	POC	mg L^{-1}	0.385	0.145	0.760	0.160
	TOC	mg L^{-1}	8.02	0.63	9.56	7.02
	DIC	mg L^{-1}	25.0	2.2	28.8	19.0
Chlorophyll-a		mg L^{-1}	0.002	0.002	0.006	<0.001
Ca	Dissolved	mg L^{-1}	29.3	3.1	34.6	22.3
Mg	Dissolved	mg L^{-1}	10.8	1.2	12.7	8.2
K	Dissolved	mg L^{-1}	2.27	0.25	2.67	1.84
Na	Dissolved	mg L^{-1}	15.9	2.5	19.6	10.9

Table 12. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L ⁻¹	0.91	0.54	2.40	0.25
Cl	Dissolved	mg L ⁻¹	18.2	3.4	23.8	12.0
F	Dissolved	mg L ⁻¹	0.10	0.01	0.14	0.08
SO ₄	Dissolved	mg L ⁻¹	24.1	3.7	28.4	16.3
Al	Extractable	mg L ⁻¹	0.181 ⁺⁺⁺	0.094	0.385 ⁺⁺⁺	<0.100
	Dissolved	mg L ⁻¹	<0.100	0.015	0.100	<0.100
As	Dissolved	mg L ⁻¹	0.0006	0.0001	0.0008	0.0005
Ba	Total	mg L ⁻¹	<0.08	0.01	0.10	<0.08
Cd	Total	mg L ⁻¹	<0.0001	0.0001	0.0003	<0.0001
	Dissolved	mg L ⁻¹	<0.0001	0.0001	0.0002	<0.0001
Co	Total	mg L ⁻¹	<0.0005	0.0001	0.0007	<0.0005
Cu	Total	mg L ⁻¹	0.0047 ⁺⁺⁺	0.0054	0.0208 ⁺⁺⁺	0.0011
	Dissolved	mg L ⁻¹	0.0019	0.0006	0.0027	0.0011
Fe	Extractable	mg L ⁻¹	0.197	0.072	0.365 ^{+,+++}	0.087
	Dissolved	mg L ⁻¹	0.033	0.016	0.063	0.010
Pb	Total	mg L ⁻¹	0.0008	0.0005	0.0020	<0.0007
	Dissolved	mg L ⁻¹	0.0011	0.0020	0.0073	<0.0007
Mn	Extractable	mg L ⁻¹	0.016	0.012	0.067 ⁺	0.005
	Dissolved	mg L ⁻¹	0.004	0.003	0.011	<0.002
Ni	Total	mg L ⁻¹	0.0012	0.0004	0.0022	<0.0005
	Dissolved	mg L ⁻¹	0.0009	0.0003	0.0016	0.0007
Se	Dissolved	mg L ⁻¹	0.0002	0.0001	0.0003	<0.0001
V	Total	mg L ⁻¹	0.0007	0.0005	0.0015	<0.0005
	Dissolved	mg L ⁻¹	<0.0005	0.0002	0.0007	<0.0005
Zn	Total	mg L ⁻¹	0.0032	0.0022	0.0083	0.0009
	Dissolved	mg L ⁻¹	0.0009	0.0007	0.0024	<0.0003
Discharge	Daily	m ³ sec ⁻¹	291	163	521	0

⁺ Exceeds limit for drinking water.

⁺⁺ Exceeds limit for recreational use.

⁺⁺⁺ Exceeds limit for protection of aquatic life.

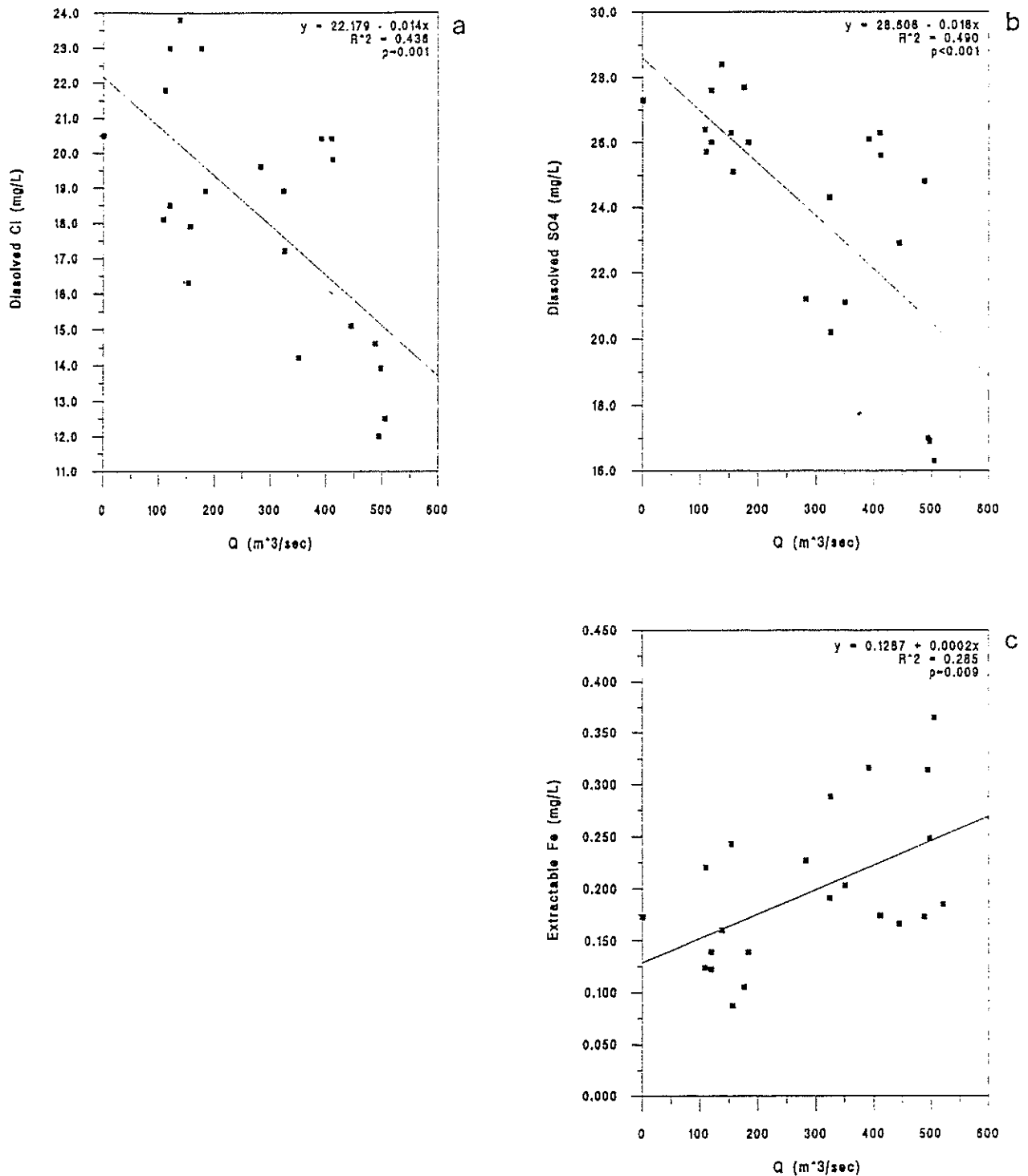


Figure 13.

Water chemistry - water discharge relationships in the Nelson River below Sea River Falls, 1987-1989. a. dissolved Cl b. dissolved SO₄ c. extractable Fe d. dissolved Ca e. dissolved Mg f. dissolved K g. dissolved Na h. dissolved NO₂-NO₃ i. total NH₃ j. POC k. DIC l. bicarbonate alkalinity m. particulate N n. total hardness o. non-carbonate hardness p. TDS q. total alkalinity r. total NFR s. fixed NFR t. lab pH u. field turbidity v. field conductivity w. lab conductivity.

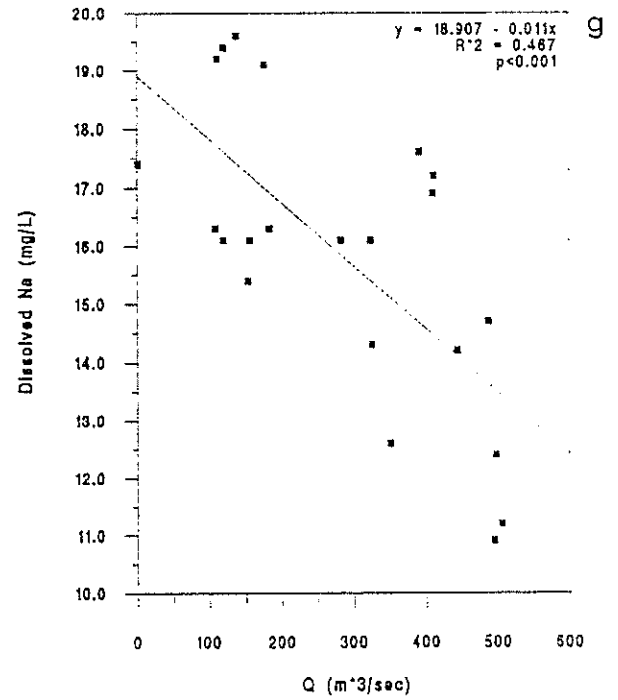
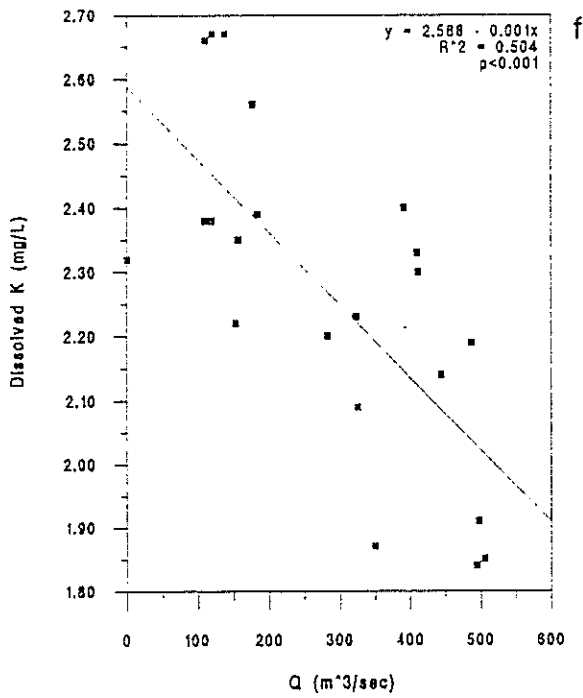
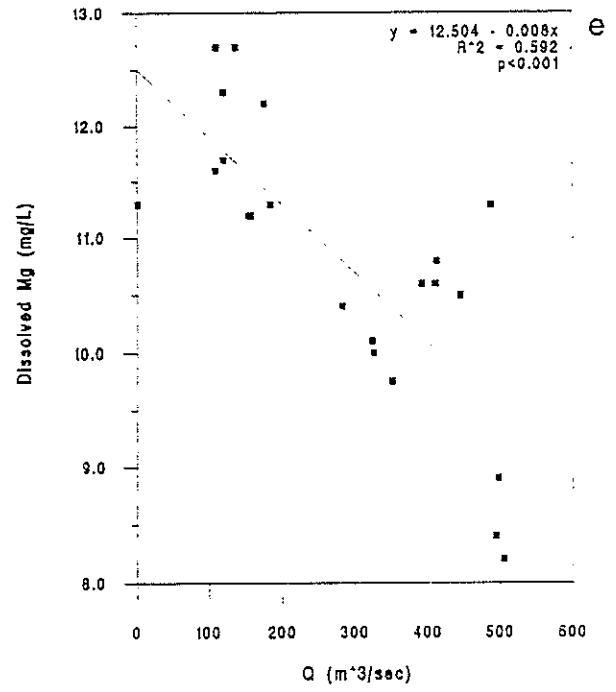
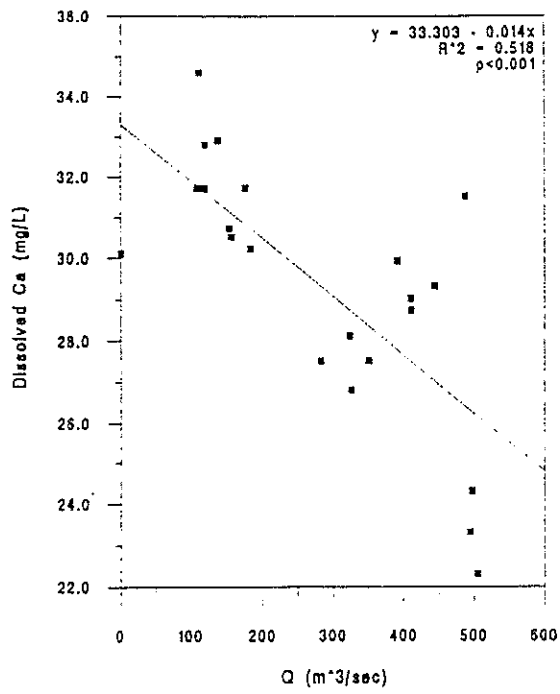


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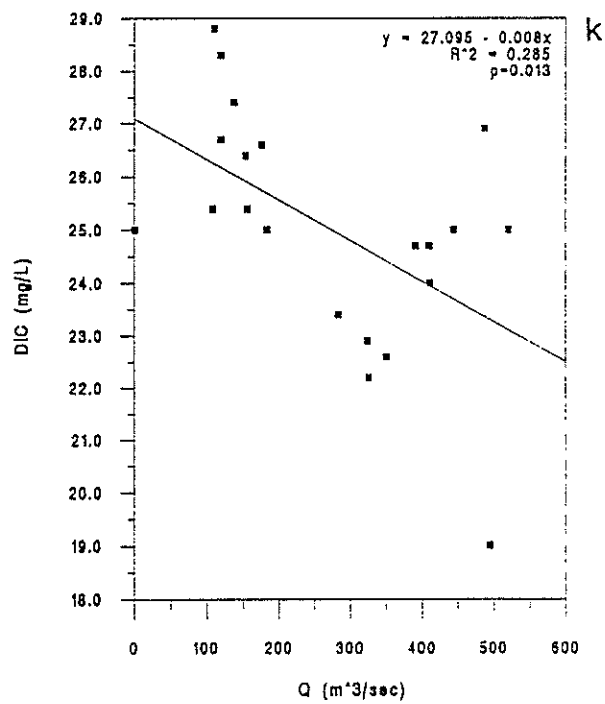
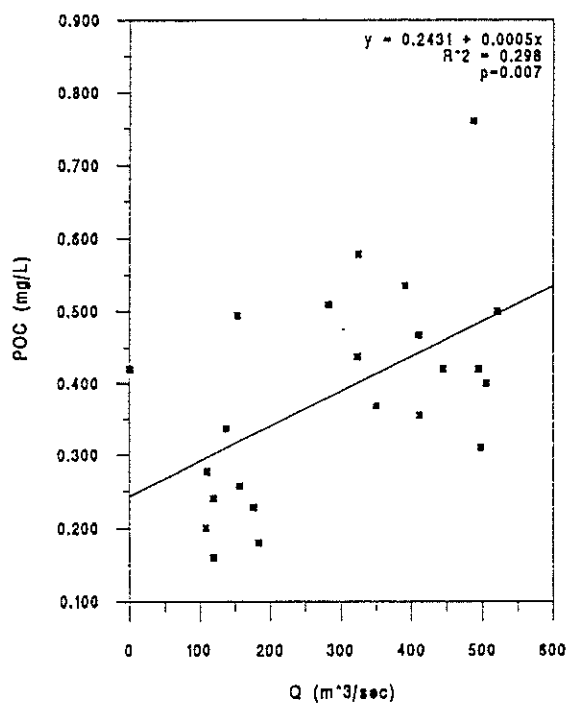
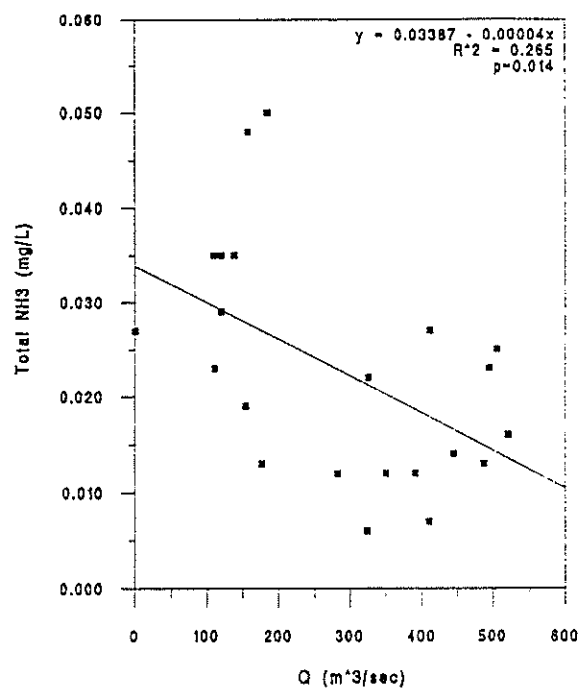
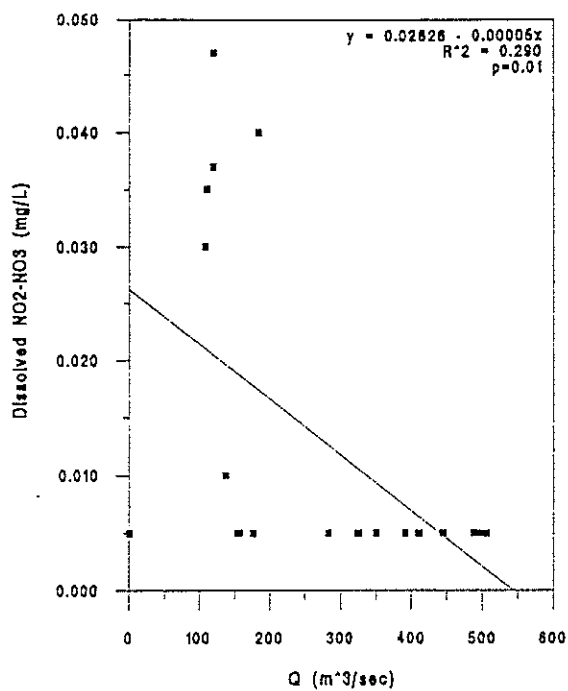


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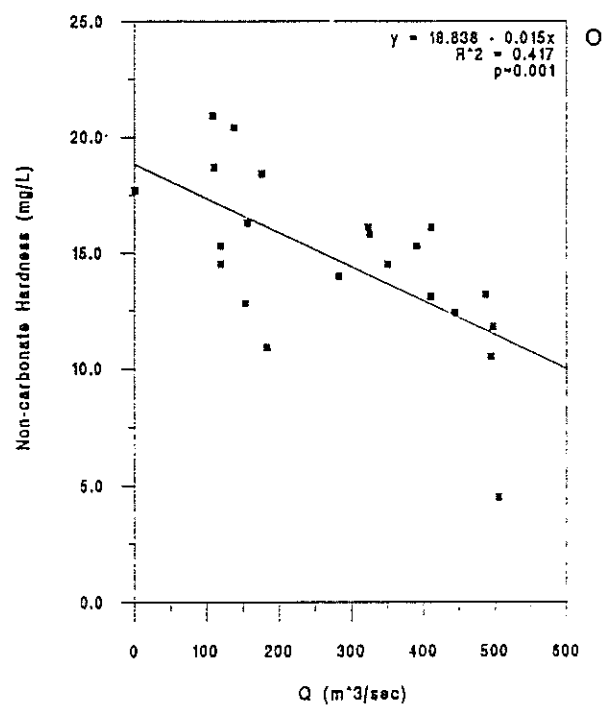
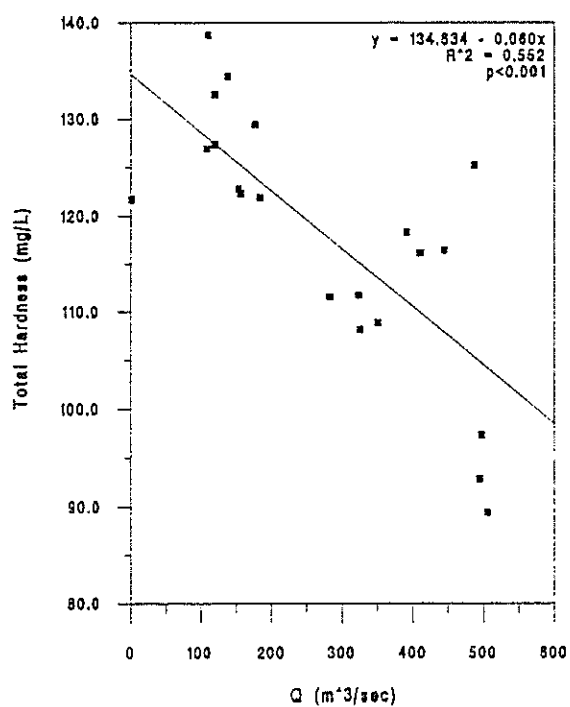
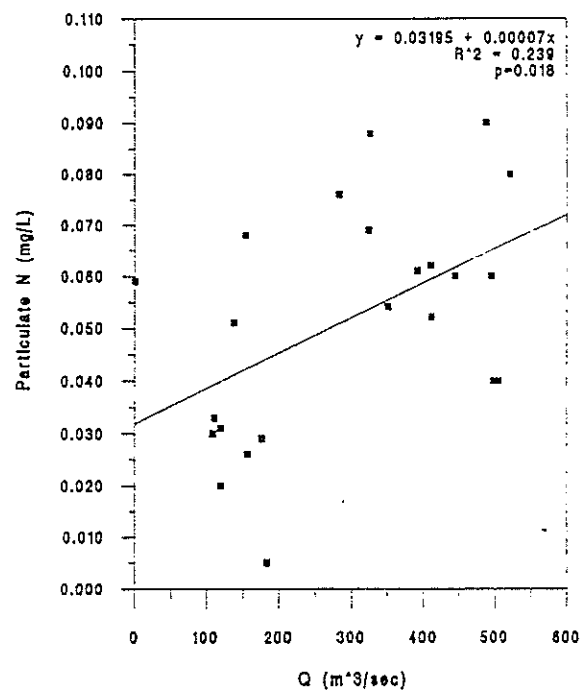
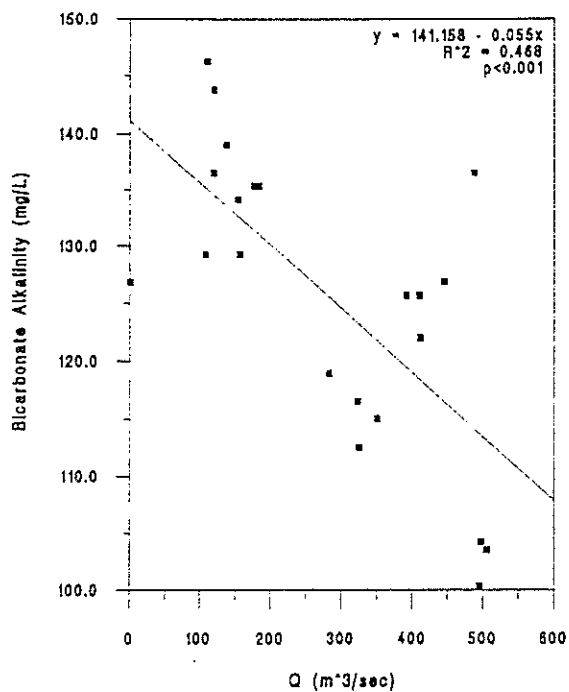


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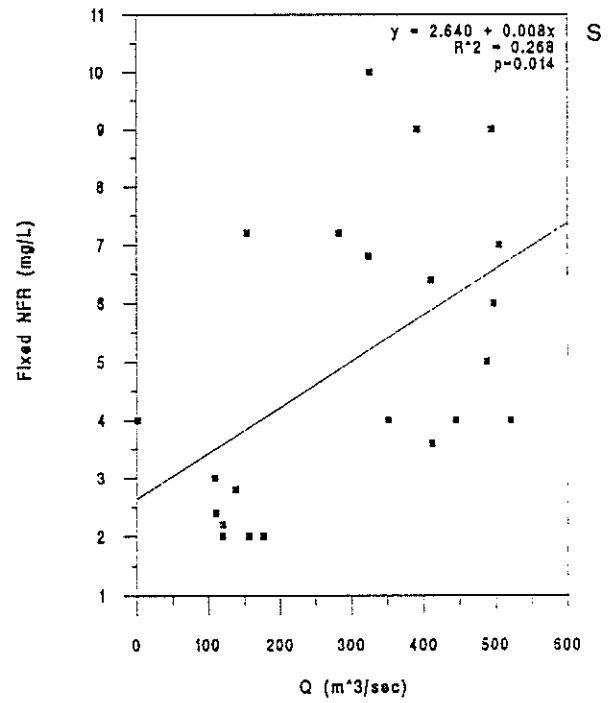
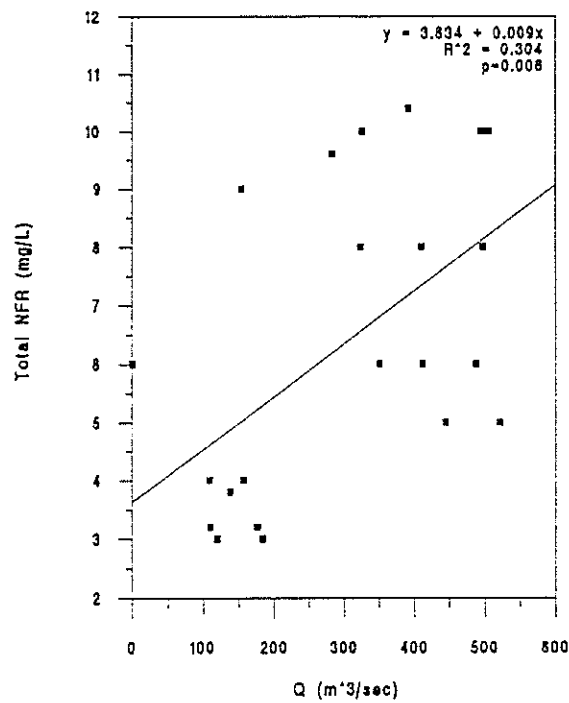
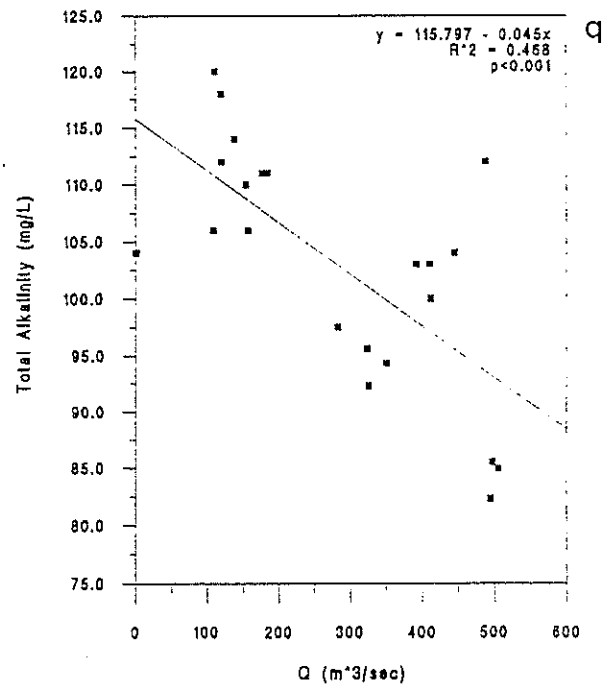
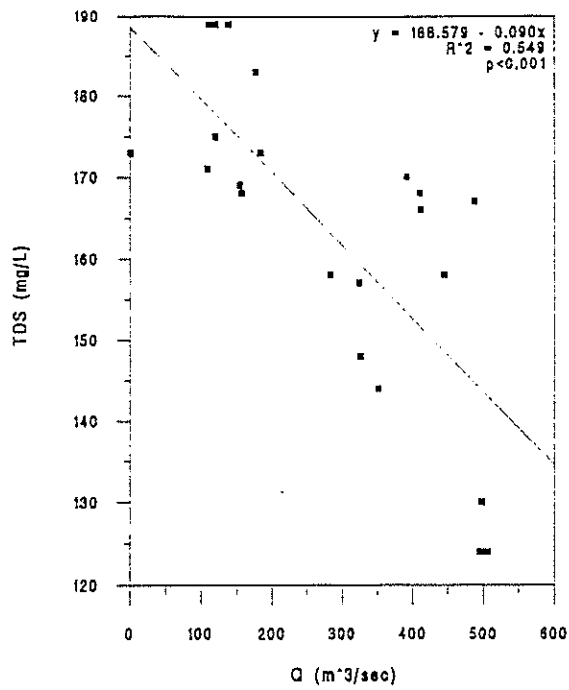


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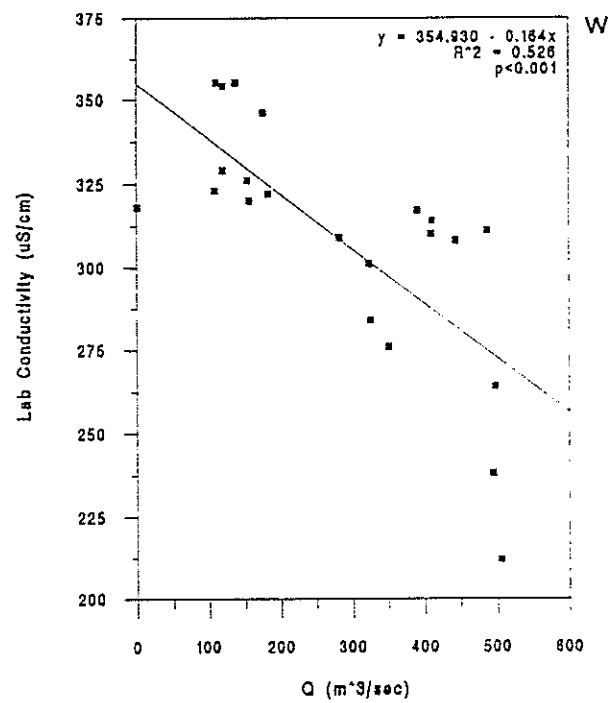
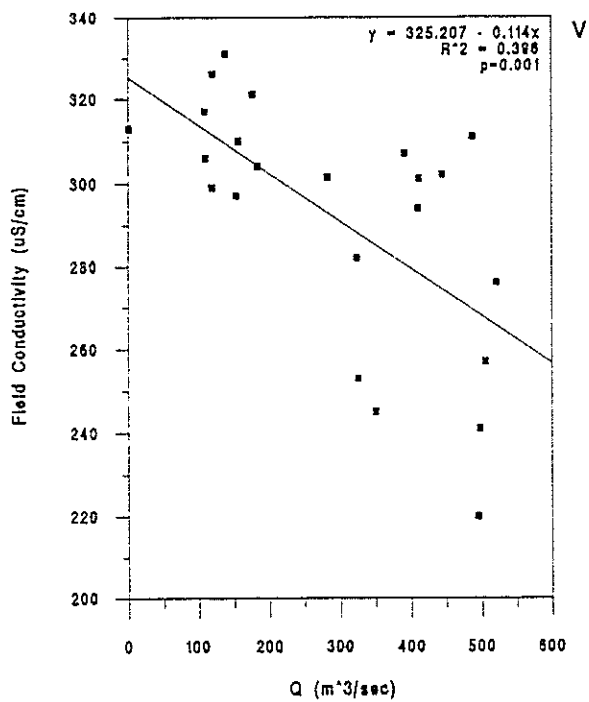
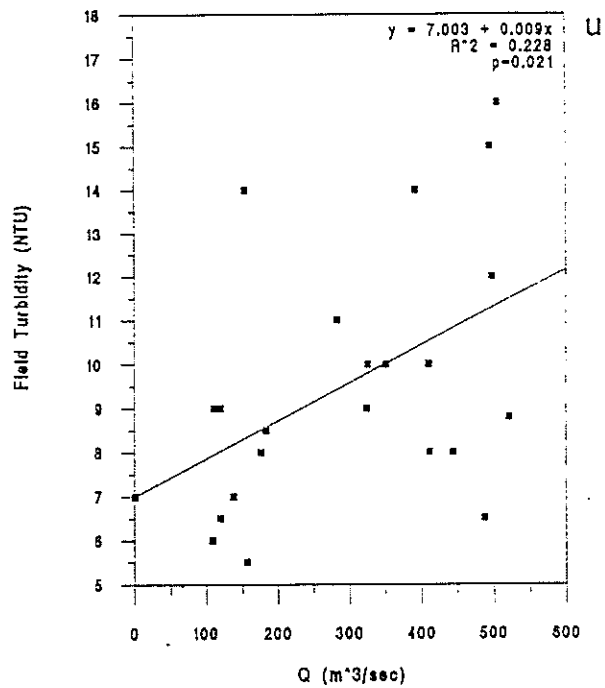
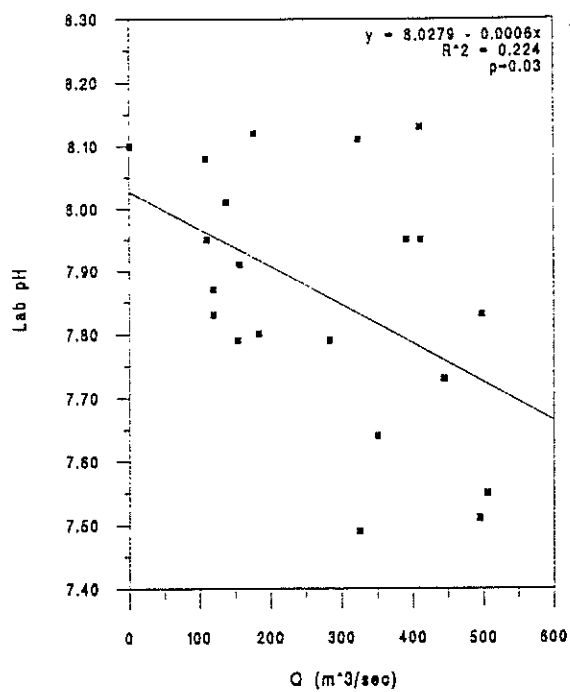


Figure 13. Continued.

LWR.

Most of the significant water quality - water discharge relationships at this site should be of some use for impact prediction and management purposes. All of the 23 significant correlations accounted for at least 21% of the variance in the dependent variable and all but two of the correlations (dissolved $\text{NO}_2\text{-NO}_3$ and total NH_3) were a reflection of a general linear trend in the data. The most useful discharge correlations are those involving conductivity (lab), TDS, alkalinity (total and bicarbonate), hardness (total and non-carbonate), dissolved Cl and SO_4 , and dissolved Ca, Mg, K, and Na in which more than 40% of the variance was accounted for by the regression.

Duncan and Williamson (1988) examined the linear relationships between discharge and water chemistry in the Nelson River at Norway House, about 20 km upstream of Sea River Falls. As in the present study, they found conductivity, alkalinity, Ca, and Mg were negatively correlated and turbidity was positively correlated with discharge. Contrary to the present study, pH was found to be positively correlated with discharge. There is no clear reason for this discrepancy between the studies.

3.2.1.3 Seasonal Differences

The range of values in the ice-cover season overlapped the range in the ice-free season for all but 4 variables (total and fixed NFR, dissolved Mg, and reactive Si) during the 1987-1989 FEMP study period. Total and fixed NFR and reactive Si were lower under

ice cover while Mg was higher (Figure 14). Three of the variables were correlated with discharge, NFR (total and fixed) positively and Mg negatively (Figure 13). With discharge at Sea River Falls typically lower under ice cover (220 to 232 m³ sec⁻¹, Dec. to Mar.) than in the open water season (297 to 527 m³ sec⁻¹, May to Nov., Water Survey of Canada 1990), these seasonal differences can be attributed to the fluctuations in discharge. Although the relationship with discharge generally accounts for the observed seasonal differences, it fails to account for the very few variables in which seasonal differences occurred given the large number of variables that were correlated with discharge. Discharge fluctuations also do not explain the summer maximum in reactive Si at this site. Release or resuspension from upstream lake sediments (Wetzel 1983), or contributions in eroded bank and shoreline materials are possible explanations, supported by the summer peak in NFR, although no specific site can be identified with the available information.

The pre-development dataset was comprised of 5 to 6 measurements between June 1971 and May 1972, with 2 to 3 of these under ice-cover. This dataset was judged to be inadequate for assessment of the occurrence of seasonal patterns.

3.2.1.4 Changes in Water Quality Since Lake Winnipeg Regulation

The pre-development data record for this site did not meet the minimum requirements for a statistical comparison, as noted above. Qualitative comparison indicates no marked change in physical and

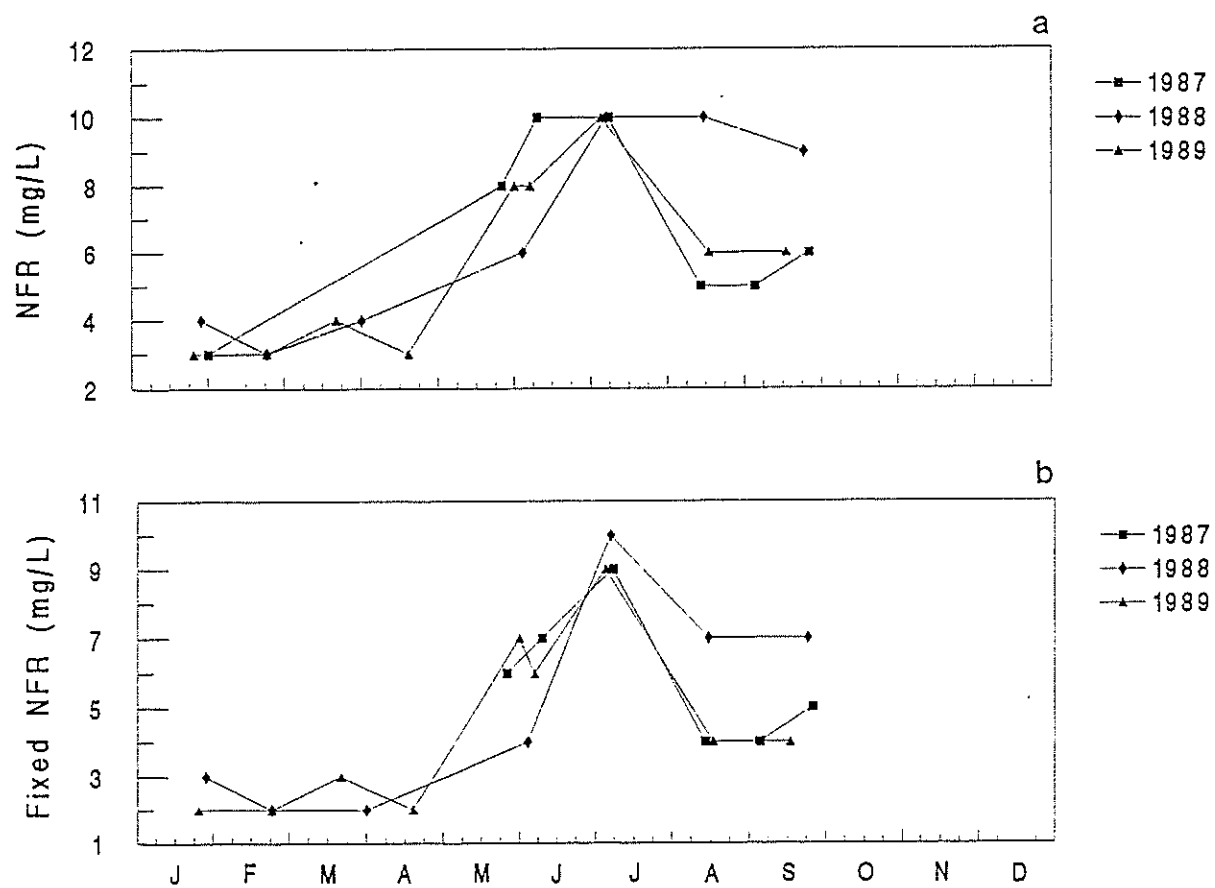


Figure 14. Seasonal water quality trends in the Nelson River below Sea River Falls. a. total NFR b. fixed NFR c. dissolved Mg d. reactive Si.

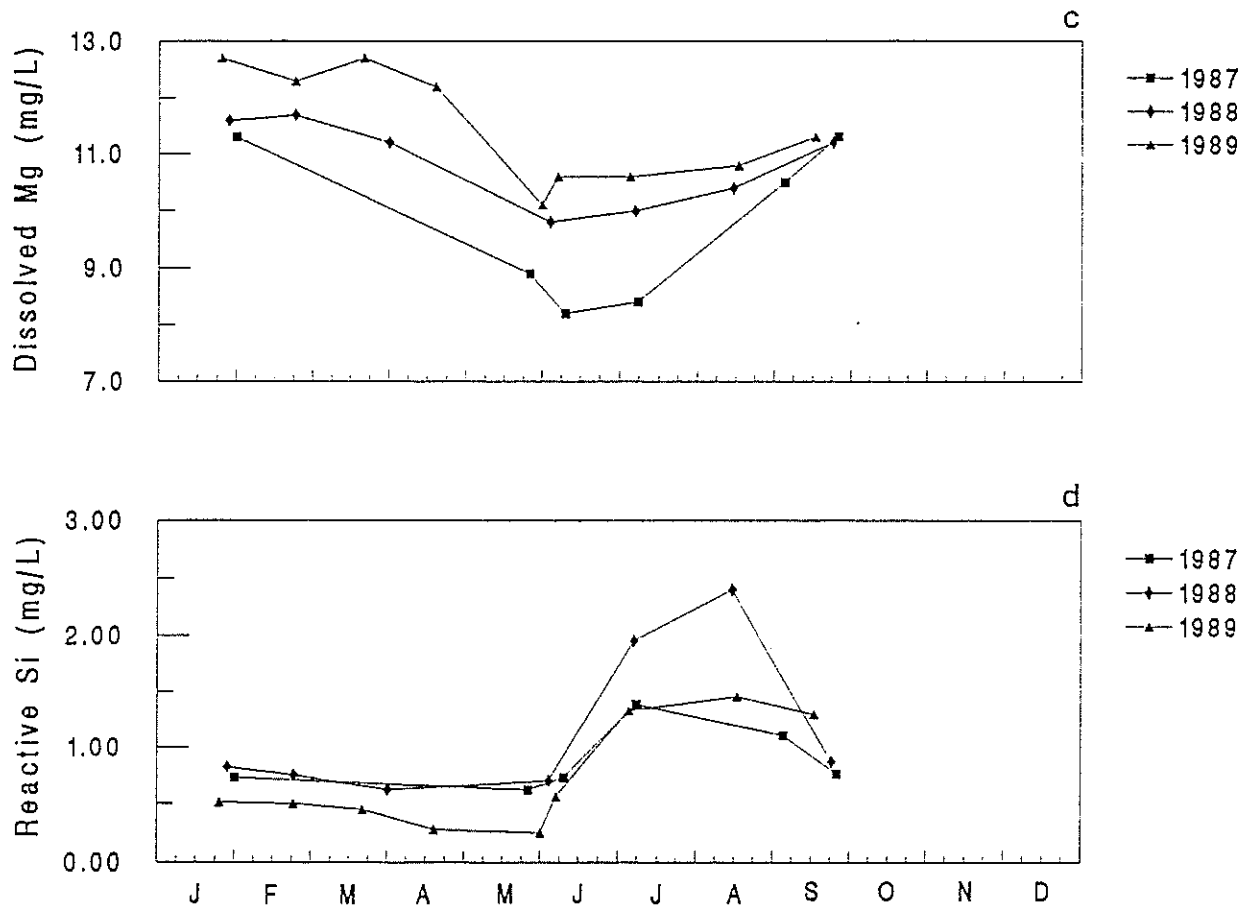


Figure 14. Continued.

chemical water quality. For the majority of variables (turbidity, TOC, $\text{NO}_2\text{-NO}_3$, pH, NFR (total and fixed), hardness, Na, K, and dissolved Pb), the pre-development mean is within 1 standard deviation (s.d.) of the post-development mean (Tables 12 and 13). The difference is less than 2 s.d. of the post-development mean for most other variables (conductivity, alkalinity, and reactive Si). Thus the pre-development data generally fall within the approximate 95% confidence limits for the post-development means, indicating no significant change. The only exceptions to this trend are dissolved Cu and Zn. Concentrations of these species prior to development are 5 to 10 times higher than the post-development means and are outside the range measured in the 1987-1989 sampling period. However, the pre-development data represent single analyses so it would be unwise to place much emphasis on these apparent changes.

Playle and Williamson (1986) found a number of small changes in their more extensive database for the Nelson River near Norway House. Total organic and inorganic carbon, total P, TKN, SO_4 , and turbidity were 16-36% higher between mid-1876 and 1984 than from 1971 to mid-1976. These changes were generally attributed to shoreline degradation caused by level regulation and channel construction.

The only way of assessing the impacts of LWR regulation on water quality at Sea River Falls, in the absence of sufficient pre-development data for the site, is to use pre-development data for some other adequately sampled site. The Nelson River at Norway House is the only possible choice, as it was routinely sampled from

Table 13. Summary statistics for physical and chemical water quality variables measured in the Nelson River below Sea River Falls; June, 1971, to May, 1972. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Lab		7.7	0.2	7.9	7.4
Conductivity	Lab	$\mu\text{S cm}^{-1}$	252	46.2	307	182
Turbidity	Lab	NTU	9.2 ⁺	3.6	13.0 ⁺	2.8
NFR	Total	mg L^{-1}	7			
NFR	Fixed	mg L^{-1}	2			
Alkalinity	Total	mg L^{-1}	85.5	13.5	103	66.1
Hardness	Total	mg L^{-1}	103.5	17.0	129.0	81.4
Nitrogen	$\text{NO}_2\text{-NO}_3$	mg L^{-1}	0.09	0.08	0.14	0.03
Carbon	TOC	mg L^{-1}	8.5	3.5	12	5
K	Dissolved	mg L^{-1}	2.2	0.5	2.9	1.7
Na	Dissolved	mg L^{-1}	13.3	2.2	16.0	10.5
Si	Reactive	mg L^{-1}	1.9	0.4	2.4	1.5
Cu	Dissolved	mg L^{-1}	0.01			
Pb	Dissolved	mg L^{-1}	<0.001			
Zn	Dissolved	mg L^{-1}	0.011			
Discharge	Daily	$\text{m}^3 \text{sec}^{-1}$	476	174	688	323

⁺ Exceeds limit for drinking water.

1972 to 1974.

In order to use data from a surrogate site, it is necessary to demonstrate that water quality at the two sites should have been the same prior to development. This probably was the case for the Norway House and Sea River Falls sites provided there were no natural sites of erosion, significant local tributaries, or man-made structures or facilities between the sites which might contribute to significant differences. This validation would require some field reconnaissance and consultation with individuals knowledgeable in the recent history of development (i.e., past 25 years) along this reach of the Nelson River. Such a validation process was outside the scope of the present study, but should be considered as part of any subsequent investigation.

3.2.2 Jack River

3.2.2.1 Current Water Quality

Physical and chemical water quality in the Jack River was generally good, with levels for most parameters well below the Canadian guidelines for human consumption, recreational use, and the protection of aquatic life (Table 14). As at most other sites in the FEMP study, there were a number of exceptions to this trend. The mean extractable Al and total Cu concentrations exceeded the guideline levels for protection of aquatic life, as did the peak concentrations of total Pb. Mean turbidity exceeded the drinking water guideline as did the maximum extractable iron concentrations.

Table 14. Summary statistics for physical and chemical water quality variables measured in the Jack River (West Channel); January, 1987, to September, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.8	0.3	8.4	7.4
	Lab		7.90	0.19	8.19	7.44
Conductivity	Field	$\mu\text{S cm}^{-1}$	293	25	330	237
	Lab		314	27	349	245
Turbidity	Field	NTU	10 ⁺	6	34 ⁺	5
	Lab		9 ⁺	5	26 ⁺	3
Colour	True	Relative	8	4	15	3
TDS		mg L^{-1}	165	16	185	122
NFR	Total	mg L^{-1}	7	4	20	2
	Fixed	mg L^{-1}	5	4	17	1
Alkalinity	Total	mg L^{-1}	103.7	9.9	117.0	77.4
	Bicarbonate	mg L^{-1}	124.3	15.7	142.6	76.4
Hardness	Total	mg L^{-1}	118.5	10.1	133.5	93.1
	Non-carbonate	mg L^{-1}	14.7	3.5	20.4	9.1
Free CO ₂		mg L^{-1}	3.38	2.27	8.01	0
Phosphorus	Particulate	mg L^{-1}	0.012	0.007	0.032	<0.006
	Ortho PO ₄	mg L^{-1}	<0.003	0.002	0.009	<0.003
	Total Dissolved	mg L^{-1}	0.010	0.005	0.025	0.004
	Total	mg L^{-1}	0.022	0.006	0.037	0.015
Nitrogen	Particulate	mg L^{-1}	0.050	0.026	0.101	<0.010
	Total Dissolved	mg L^{-1}	0.300	0.045	0.470	0.230
	NO ₂ -NO ₃	mg L^{-1}	0.011	0.011	0.048	<0.010
	Total NH ₃	mg L^{-1}	0.025	0.017	0.070	0.006
	Total	mg L^{-1}	0.35	0.05	0.50	0.28
Carbon	DOC	mg L^{-1}	7.30	0.46	8.10	6.59
	POC	mg L^{-1}	0.387	0.162	0.753	0.150
	TOC	mg L^{-1}	7.68	0.43	8.59	7.12
	DIC	mg L^{-1}	24.8	2.5	28.1	18.3
Chlorophyll-a		mg L^{-1}	0.002	0.002	0.005	<0.001
Ca	Dissolved	mg L^{-1}	29.5	2.4	33.0	23.6
Mg	Dissolved	mg L^{-1}	10.9	1.1	12.6	8.3
K	Dissolved	mg L^{-1}	2.30	0.21	2.71	1.87
Na	Dissolved	mg L^{-1}	16.2	2.2	19.4	11.3

Table 14. continued

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L ⁻¹	0.70	0.46	1.91	0.23
Cl	Dissolved	mg L ⁻¹	18.6	2.9	23.3	12.8
F	Dissolved	mg L ⁻¹	0.11	0.02	0.16	0.08
SO ₄	Dissolved	mg L ⁻¹	24.4	3.3	29.0	17.4
Al	Extractable	mg L ⁻¹	0.176 ⁺⁺⁺	0.126	0.442 ⁺⁺⁺	<0.100
	Dissolved	mg L ⁻¹	<0.100		<0.100	<0.100
As	Dissolved	mg L ⁻¹	0.0006	0.0001	0.0008	0.0004
Ba	Total	mg L ⁻¹	<0.08	0.01	0.08	<0.08
Cd	Total	mg L ⁻¹	0.0001	0.0001	0.0004	<0.0001
	Dissolved	mg L ⁻¹	<0.0001	0.0001	0.0002	<0.0001
Co	Total	mg L ⁻¹	<0.0005	0.0001	0.0007	<0.0005
Cu	Total	mg L ⁻¹	0.0034 ⁺⁺⁺	0.0038	0.0159 ⁺⁺⁺	0.0011
	Dissolved	mg L ⁻¹	0.0021	0.0013	0.0061	0.0013
Fe	Extractable	mg L ⁻¹	0.187	0.106	0.457 ^{+,+++}	0.070
	Dissolved	mg L ⁻¹	0.025	0.012	0.052	0.004
Pb	Total	mg L ⁻¹	0.0007	0.0006	0.0027 ⁺⁺⁺	<0.0007
	Dissolved	mg L ⁻¹	0.0017	0.0025	0.0077	<0.0007
Mn	Extractable	mg L ⁻¹	0.012	0.007	0.034	<0.002
	Dissolved	mg L ⁻¹	0.003	0.002	0.006	<0.002
Ni	Total	mg L ⁻¹	0.0012	0.0004	0.0020	0.0005
	Dissolved	mg L ⁻¹	0.0010	0.0008	0.0030	<0.0005
Se	Dissolved	mg L ⁻¹	0.0002	0.0001	0.0004	<0.0001
V	Total	mg L ⁻¹	0.0007	0.0004	0.0017	<0.0005
	Dissolved	mg L ⁻¹	<0.0005	0.0003	0.0014	<0.0005
Zn	Total	mg L ⁻¹	0.0032	0.0031	0.0131	0.0005
	Dissolved	mg L ⁻¹	0.0013	0.0015	0.0045	<0.0003

+ Exceeds limit for drinking water.

++ Exceeds limit for recreational use.

+++ Exceeds limit for protection of aquatic life.

3.2.2.2 Seasonal Differences

There were few clear differences in water quality between the ice-free and ice-cover seasons during the 1987-1989 FEMP study. NFR was lower under ice-cover, as was chlorophyll a, while dissolved Cd and Ni concentrations were higher under ice cover (Figure 15). The lower chlorophyll a under ice is consistent with the trend at other sites. The lower NFR and higher Cd and Ni under ice could be a product of lower winter flows, but this cannot be confirmed due to the lack of discharge data for the site.

3.2.2.3 Water Quality Changes Since Lake Winnipeg Regulation

There are no pre-development water quality data for this site and the immediate post-development data record amounts to a pair of samples collected in November, 1976, and June, 1977. Consequently there is no means of directly determining to what extent, if any, water quality has changed since Lake Winnipeg regulation. It may be possible to use pre-development data for another upstream site as a surrogate, as suggested for the site at Sea River Falls. The only upstream site which was adequately sampled prior to development is the Nelson River at Warren Landing, which has been sampled since 1969. Verification of this site as a suitable surrogate, as described above for the Sea River Falls surrogate, would also be required. Again, this verification process was outside the scope of the present study.

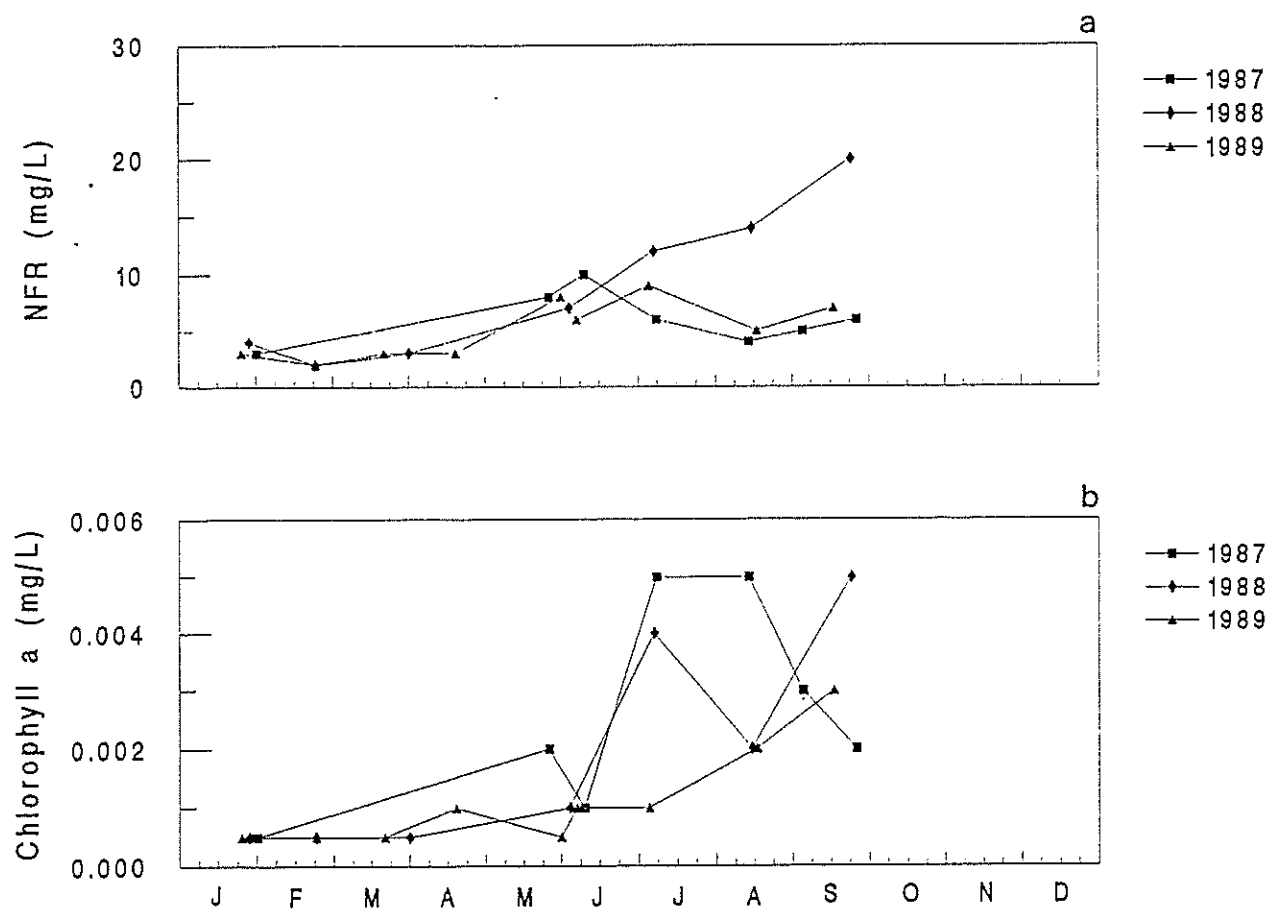


Figure 15. Seasonal water quality trends in the Jack River (West Channel) near Norway House. a. total NFR b. chlorophyll a c. dissolved Cd d. dissolved Ni.

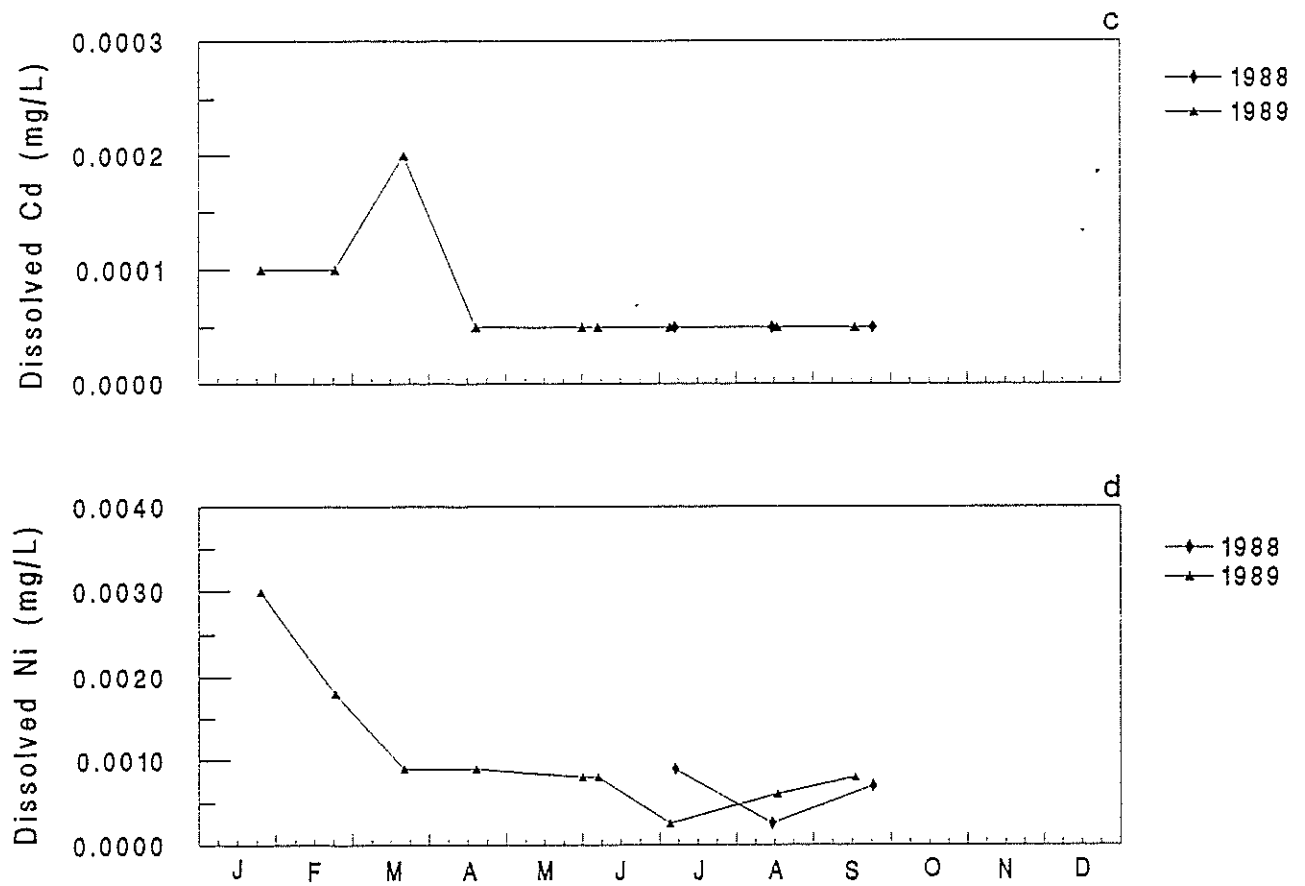


Figure 15. Continued.

3.2.3 Bacterial Studies

3.2.3.1 Norway House

Geometric mean fecal coliform bacterial counts for the seasonal study ranged from 3 per 100 mL at Historic Dock by Bay Store Bridge to 39 per 100 mL at Government Dock (Figure 16). Counts in samples from the Sewage Outlet Bay at Church Point, Green Dock, School Dock, Asiniwapiska, Bridge into Fort Island, and Historic Dock by Bay Store Bridge were consistently below 200 per 100 mL. Counts in 3 of the 4 samples from Government Dock also were well below 200 per 100 mL, but the September 22 count was 2230.

Intensive studies of bacterial water quality were conducted in the vicinity of the sewage and water treatment plants on August 11 and 24, 1987. On the first survey, fecal coliform counts ranged between 0 and 11 bacteria per 100 mL with a geometric mean in the 10 samples of 5 per 100 mL (Figure 17a). Counts ranged between 0 and 567 bacteria per 100 mL on the second survey with a geometric mean of 15 per 100 mL (Figure 17b). The overall geometric mean count for both intensive surveys was 9 fecal coliform bacteria per 100 mL. These analyses indicate bacteriological water quality in the vicinity of the water and sewage treatment plants is excellent for recreational uses, with a geometric mean fecal coliform count just 5% of the Canadian guideline of 200 bacteria per 100 mL. The regular samples also indicate bacterial water quality is generally within the Canadian guideline. It is not known if water quality at all sites is in strict compliance with the guideline, as the sampling frequency did not meet the 5 samples per 30 days specified

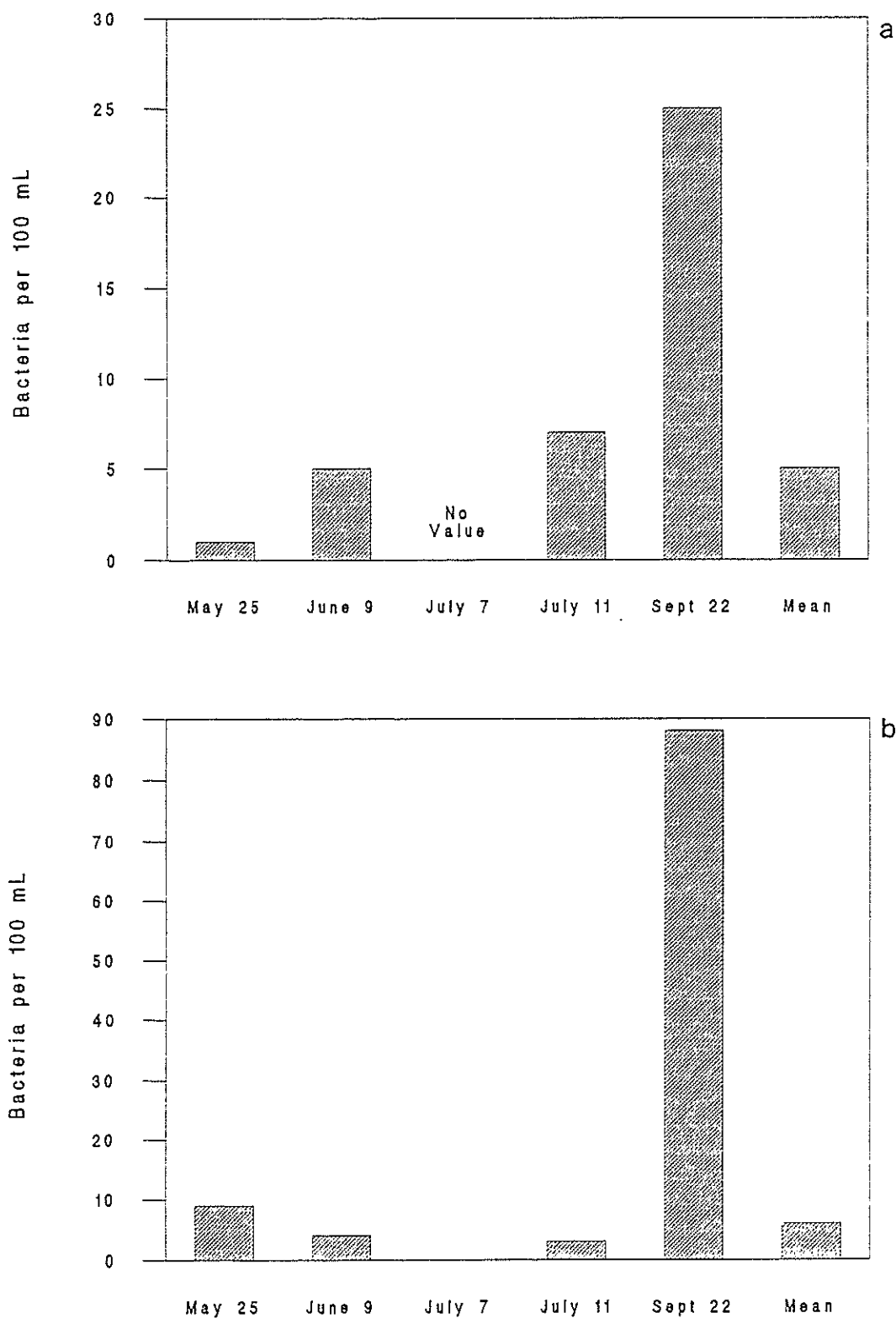


Figure 16. Fecal coliform bacteria counts from the seasonal study in the vicinity of Norway House. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Asiniwapiska b. Bridge into Fort Island c. Government Dock d. Green Dock e. Historic Dock at Bay Store Bridge f. School Dock g. Sewage Outlet Bay at Church Point. See Appendix 1 for site locations and descriptions.

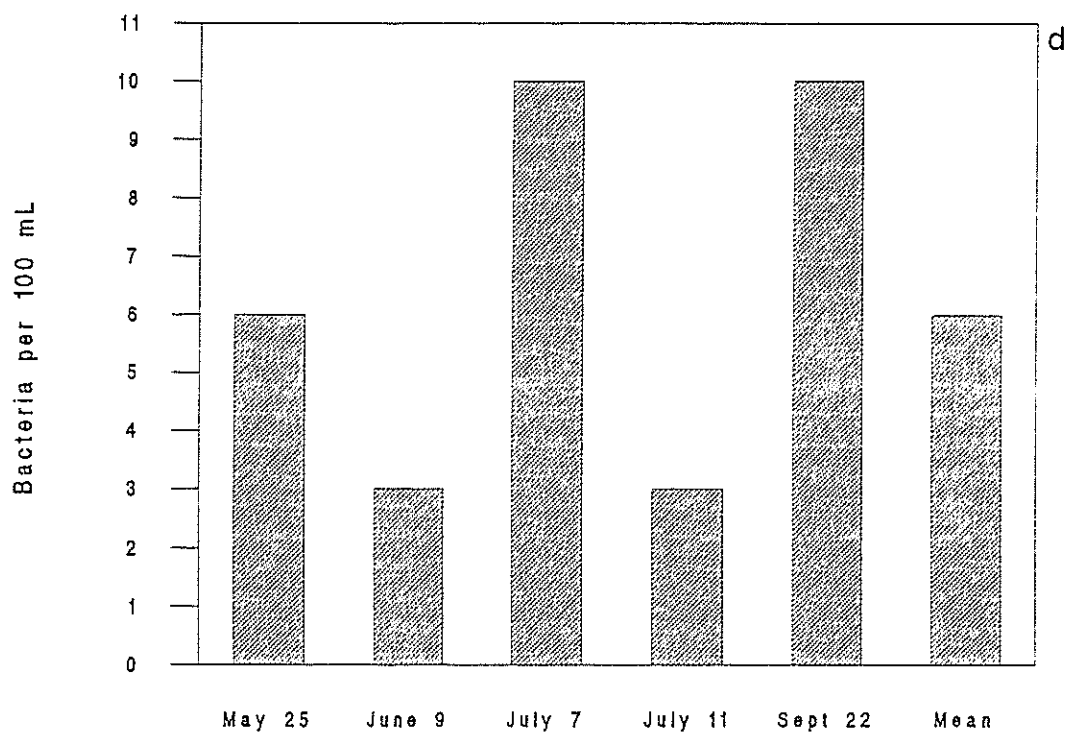
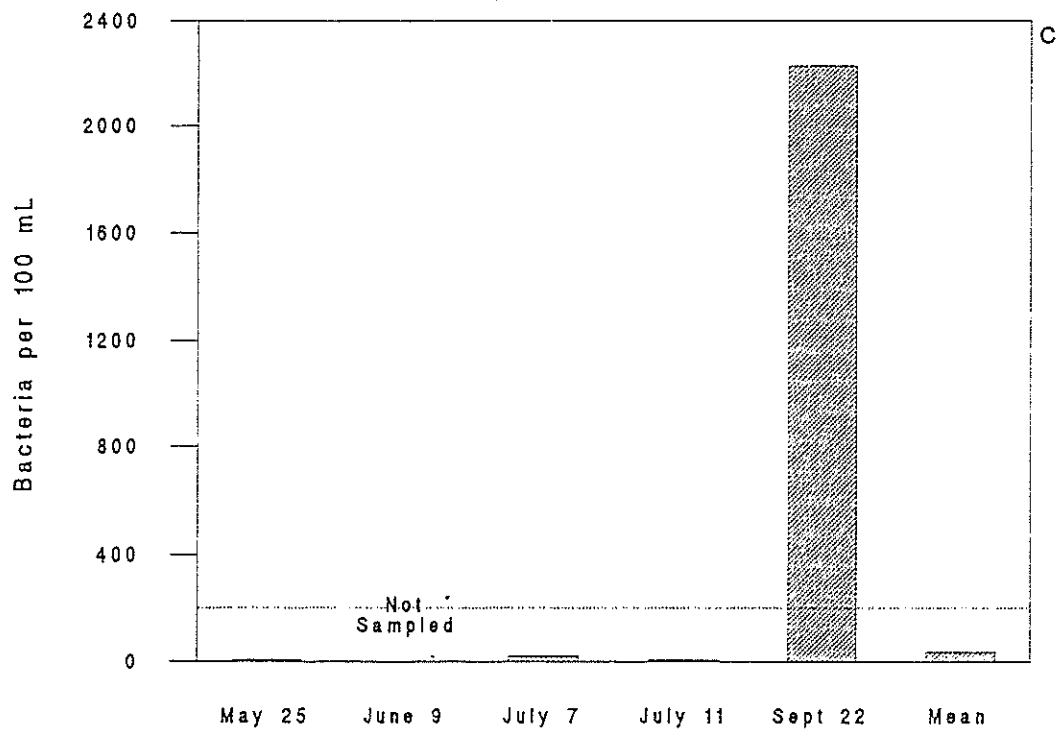


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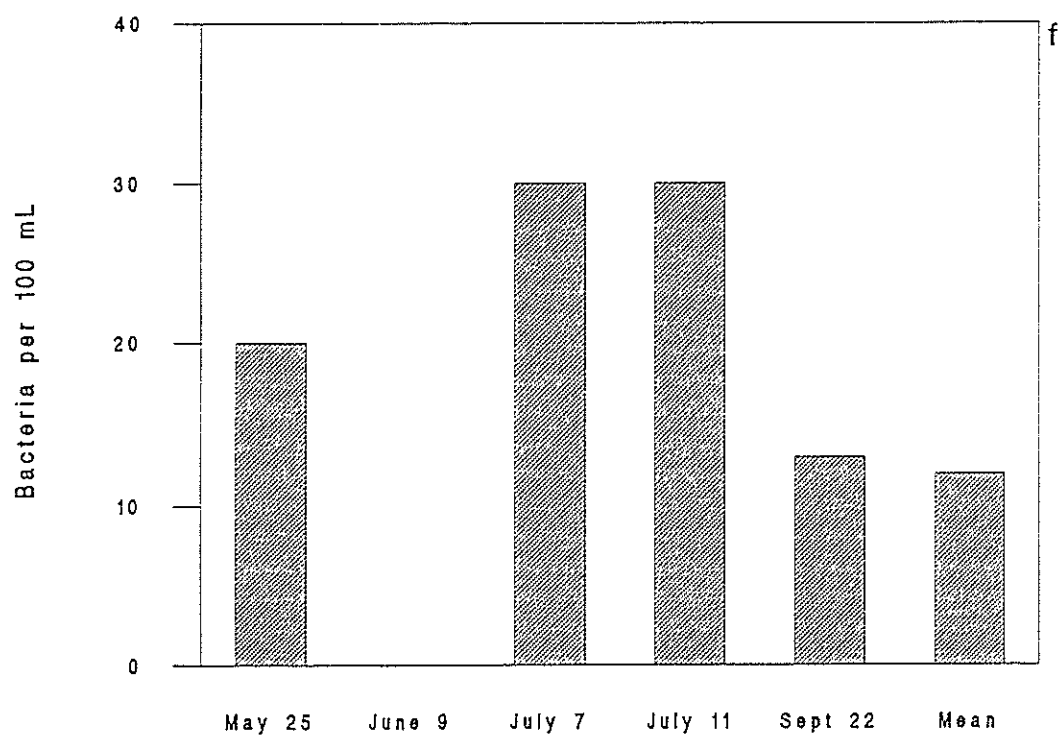
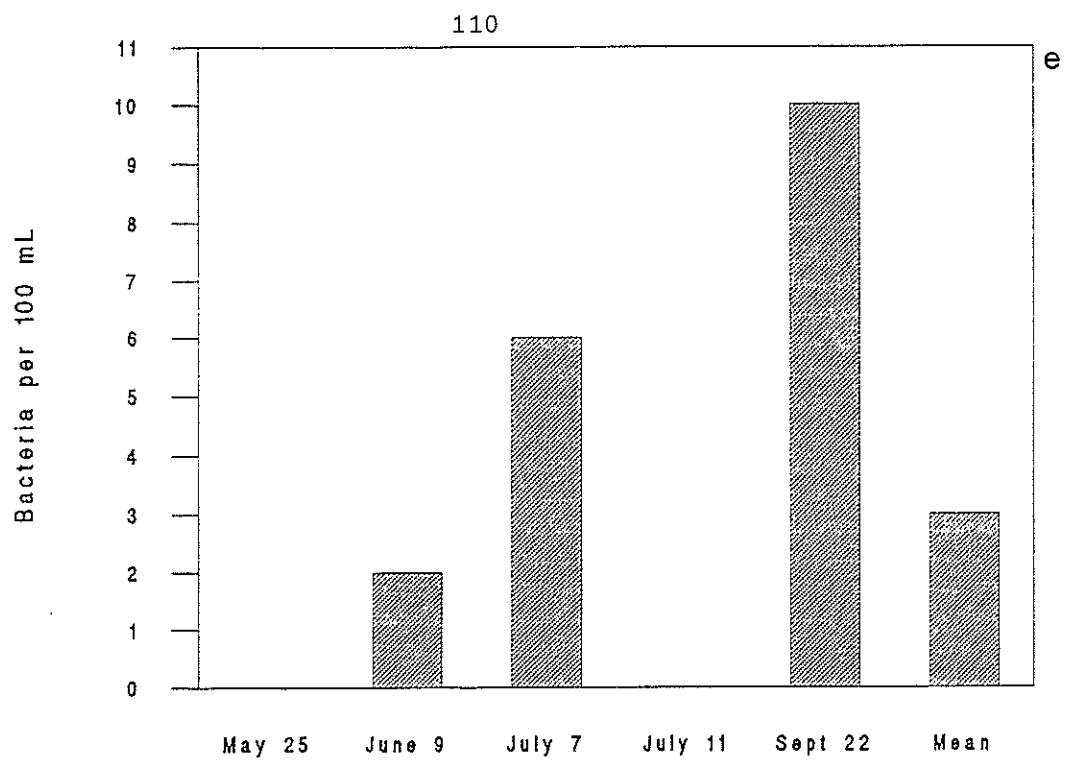


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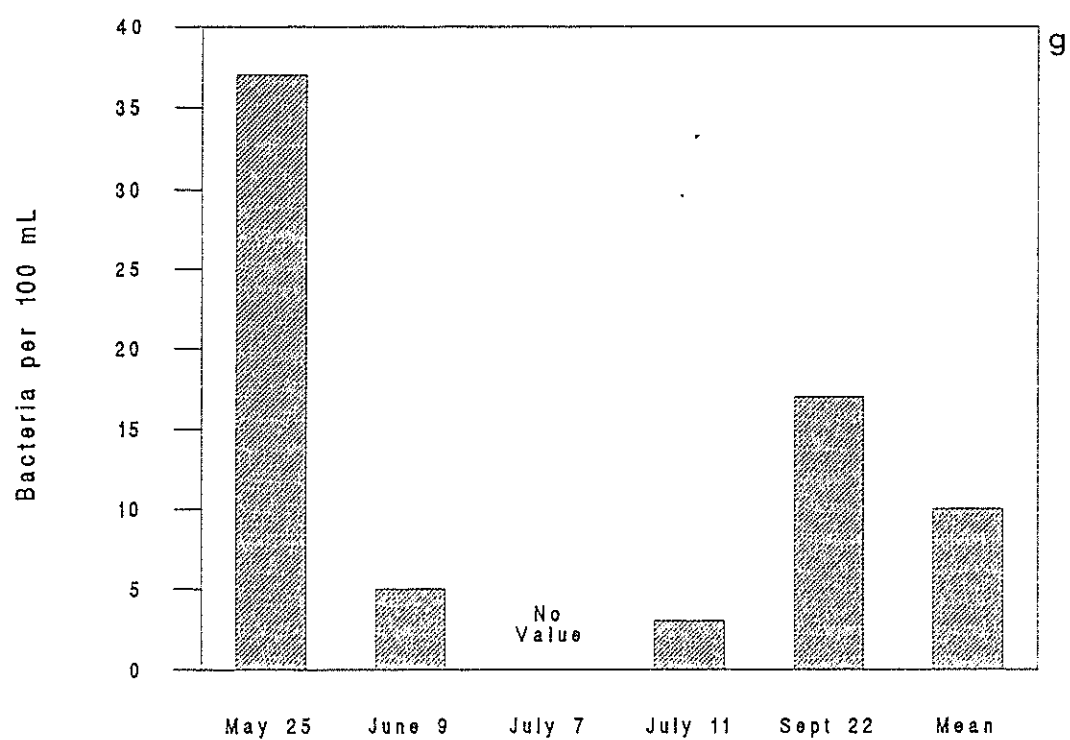


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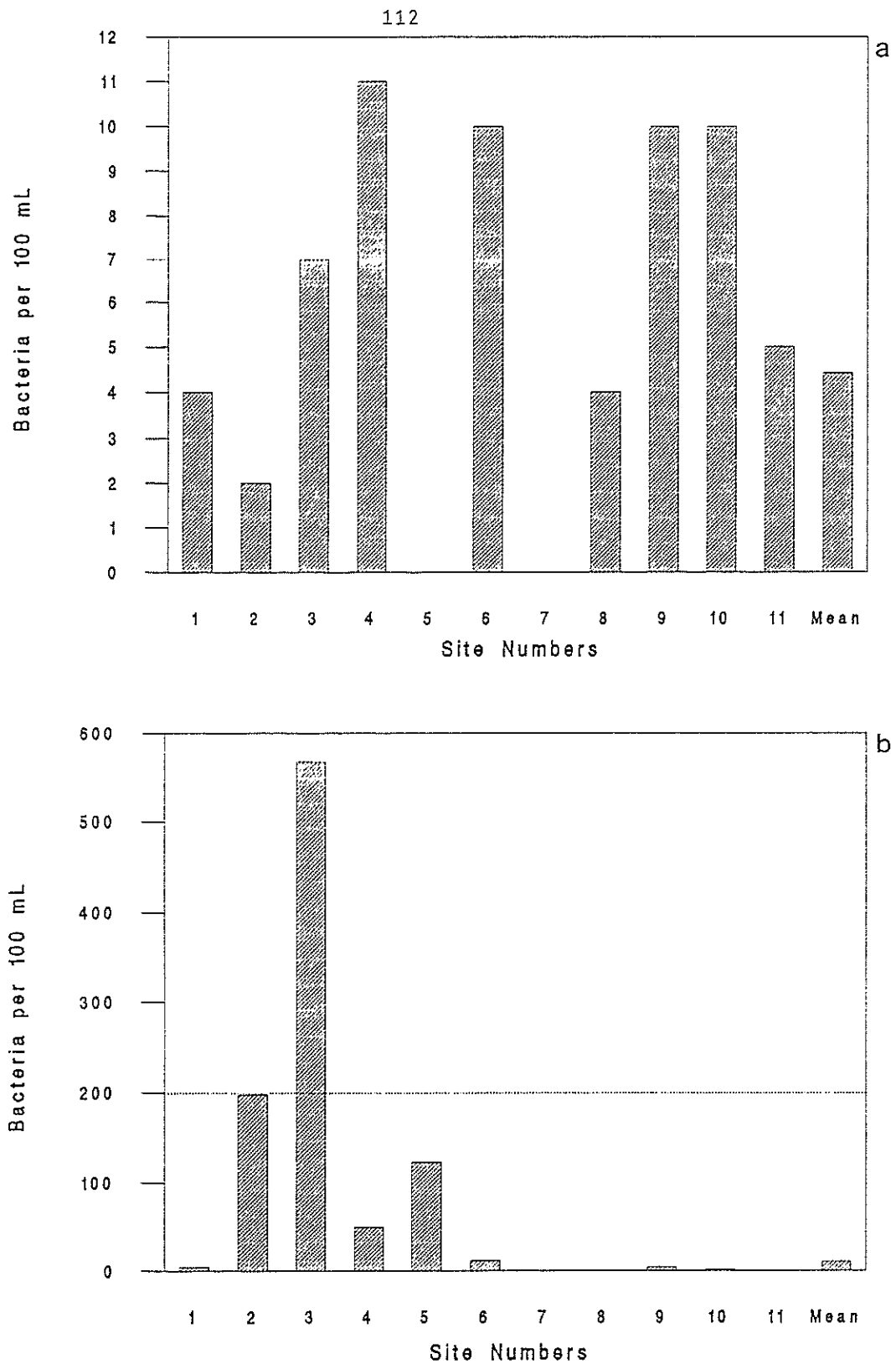


Figure 17. Fecal coliform bacteria counts from the intensive studies in the vicinity of the Norway House Sewage Treatment Plant Outfall. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Sampled 11 August 1987 b. Sampled 24 August 1987. See Appendix 1 for site descriptions and locations.

under the guideline. Future studies should ensure the designated sampling frequency is met at all sites.

3.2.3.2 Cross Lake

Geometric mean fecal coliform bacteria counts for the seasonal study ranged from 6 per 100 mL at Beardy's Point to 44 at Wapak Swimming Area (Figure 18). Fecal coliform counts in water samples from Saggitawak-Swimming Area and Beardy's Point were consistently below 200 per 100 mL. One of the five samples from each of the Saggitawak Bridge, Wapak Swimming Area, and Water Treatment Plant Intake exceeded 200 per 100 mL.

Intensive studies of bacterial water quality in the vicinity of the water treatment plant intake were conducted on August 11 and 25, 1987. Fecal coliform counts were below 10 per 100 mL in 14 of the 18 samples, and counts exceeded 200 per 100 mL in just 2 samples (Figure 19). The geometric mean of the counts on the August 11 survey was 21 per 100 mL compared with 4 per 100 mL on the August 25 survey. The overall geometric mean for both surveys was 9 counts per 100 mL. The two high counts, 1030 and 2280, occurred at sites 6 and 8, respectively on August 11. By August 25, counts at these sites had dropped to 1 and 7 bacteria per 100 mL, indicating the high bacterial numbers were relatively transient.

On the basis of the intensive studies, bacteriological water quality in the area of the water treatment plant intake can be classified as excellent for recreational purposes, with a geometric mean fecal coliform count for the study less than 10% of the

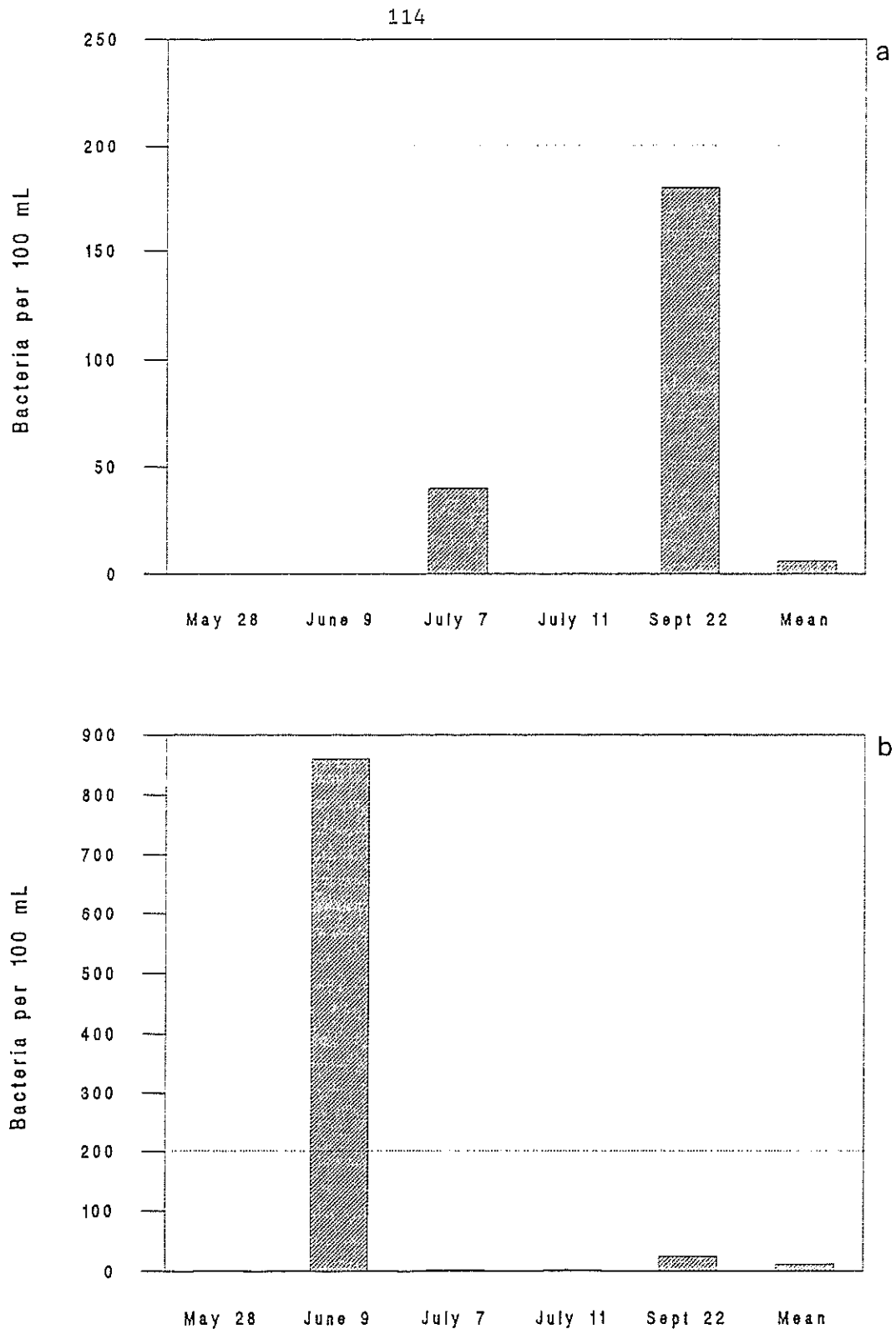


Figure 18. Fecal coliform bacteria counts from the seasonal study in the vicinity of Cross Lake. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline.
 a. Beardy's Point b. Saggitawak Bridge c. Saggitawak Swimming Area d. Wapak Swimming Area e. Left of the Water Treatment Plant Intake. See Appendix 1 for site descriptions and locations.

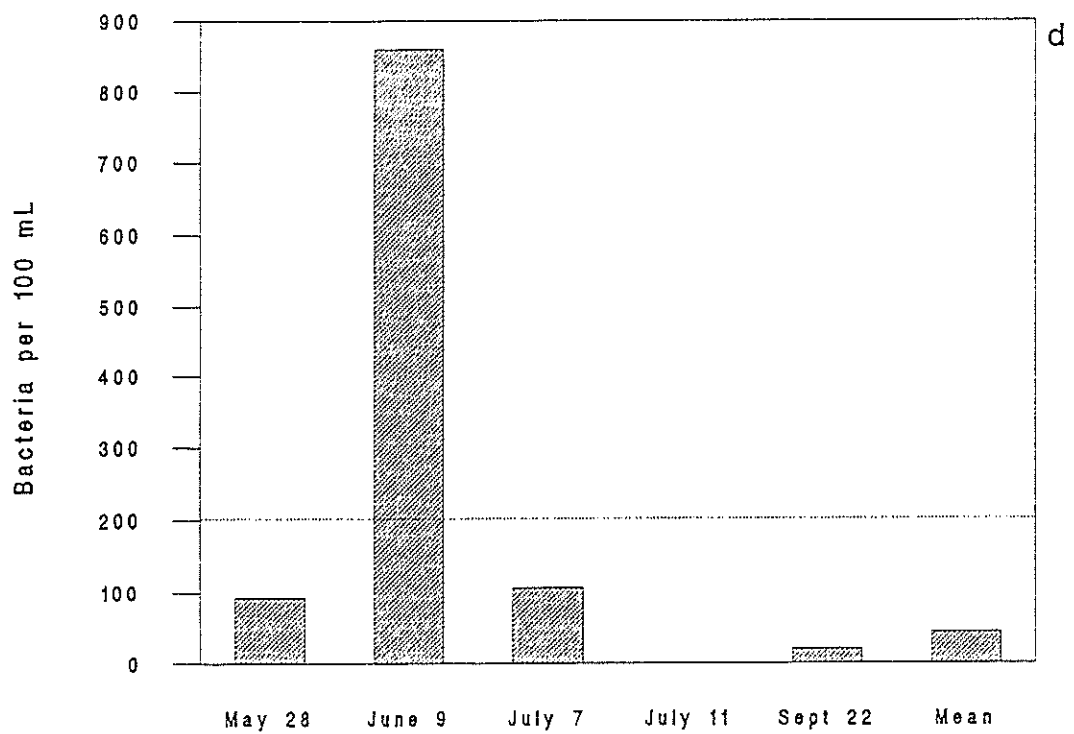
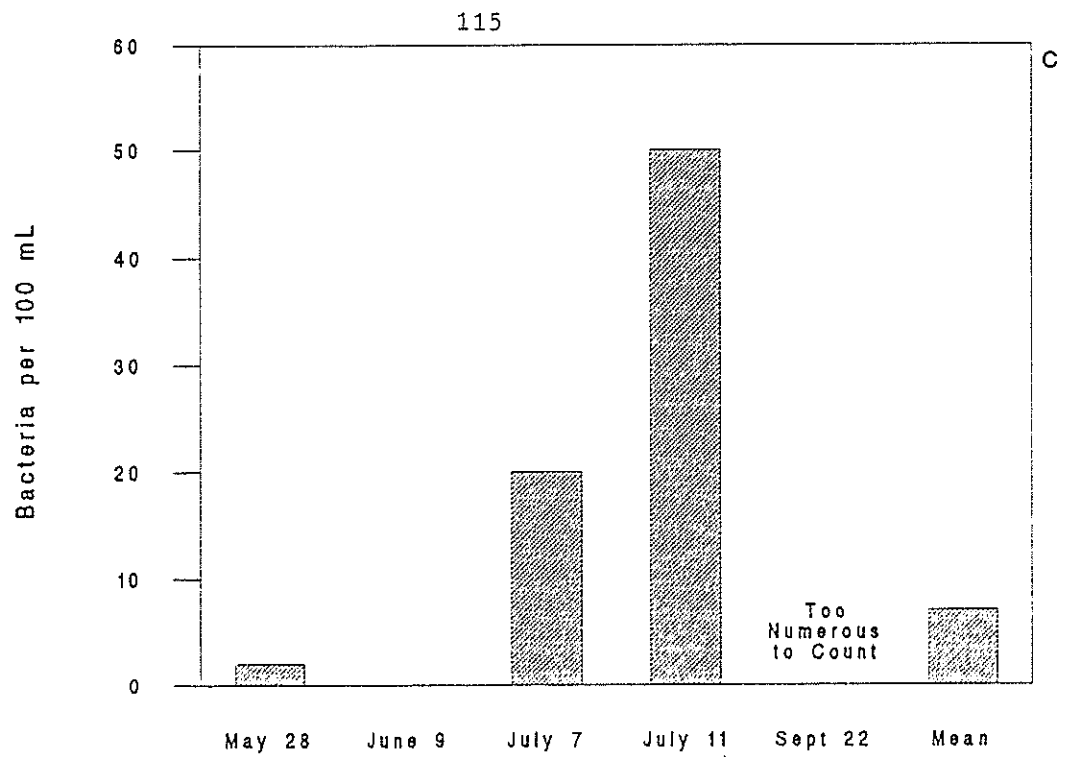


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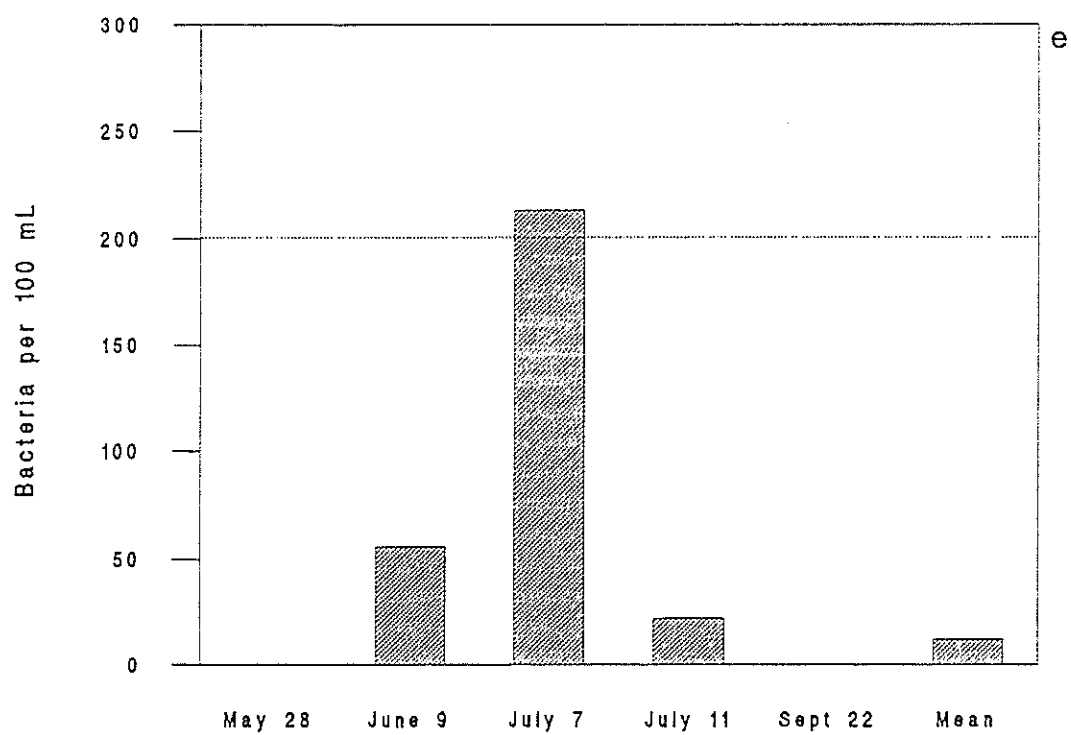


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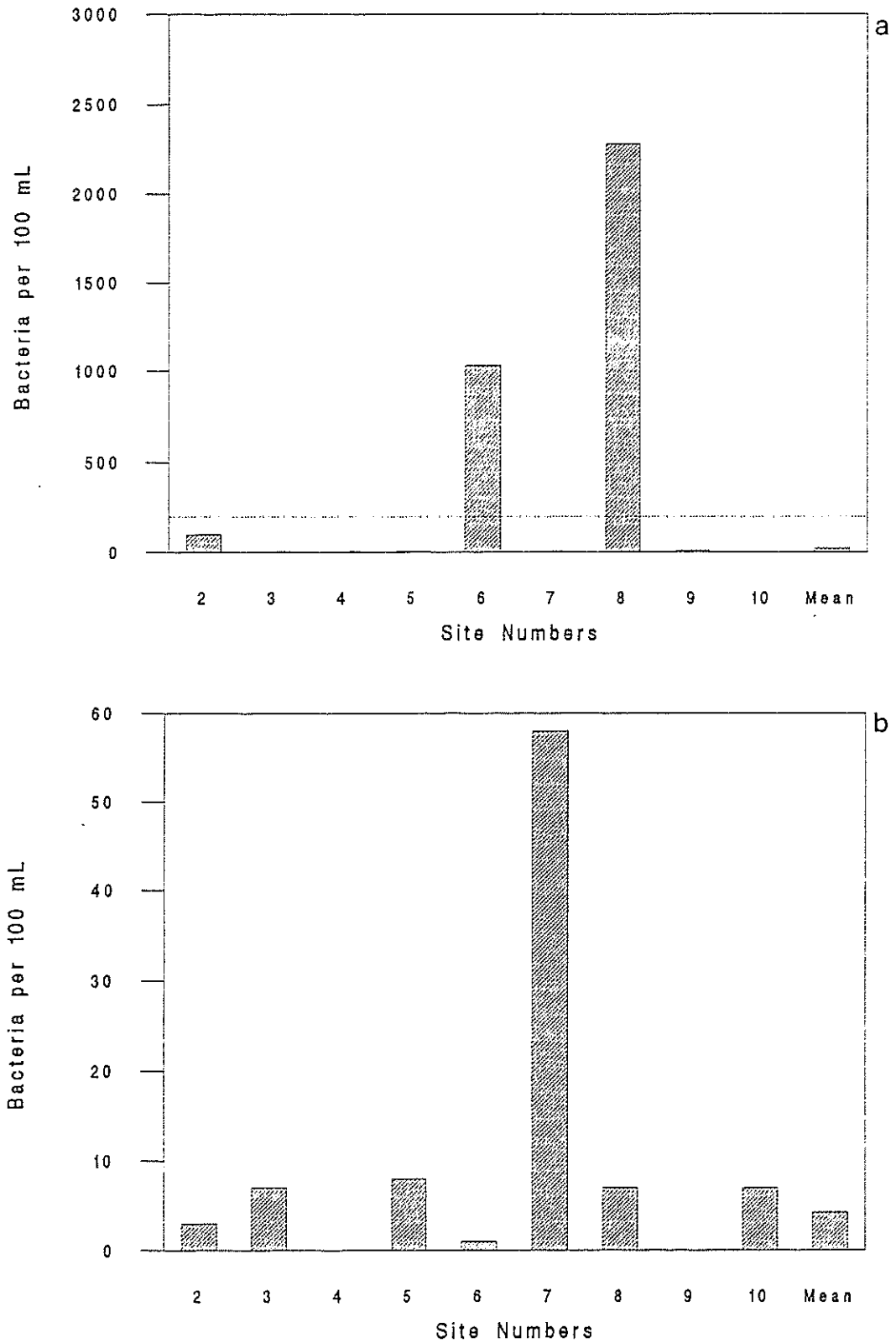


Figure 19. Fecal coliform bacteria counts from the intensive studies in the vicinity of the Cross Lake Water Treatment Plant Intake. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Sampled 11 August 1987 b. Sampled 25 August 1987. See Appendix 1 for site descriptions and locations.

Canadian guideline of 200 bacteria per 100 mL. The results of the seasonal study indicate recreational waters in the Cross Lake community appear to be of good quality for recreational uses. The occasional high bacterial count should not be considered as evidence of contamination as the confidence interval for an estimate based on a single sample is very broad. Williamson (1988a) indicates a count of 1000 bacteria per 100 mL in a single sample is within the 90% confidence interval for a true bacterial density of 200 per 100 mL. Sampling was not of sufficient frequency in the seasonal study to determine if quality was in strict compliance with the Canadian guideline. Future studies should ensure that at least five samples are collected within a 30 day period at each site of interest.

3.3 Threepoint Lake and Nelson House Area

3.3.1 Rat River above Threepoint Lake

3.3.1.1 Current Water Quality

Water quality in the Rat River was generally similar to that in the natural rivers. Among the variables for which water quality guidelines have been established, most were consistently well below the guideline values (Table 15). The exceptions generally were the same as in the natural rivers. Turbidity always exceeded the drinking water guideline and peak levels (48 NTU) approached the recreational use guideline. Mean colour was at the high end of the acceptable range for untreated drinking water, with peak values exceeding the guideline. Extractable Al concentrations always

Table 15. Summary statistics for physical and chemical water quality variables measured in the Rat River above Threepoint Lake; September, 1986, to October, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.5	0.3	7.9	7.0
	Lab		7.67	0.17	8.15	7.29
Conductivity	Field	$\mu\text{S cm}^{-1}$	99	10	115	79
	Lab		102	7	120	86
Turbidity	Field	NTU	19 ⁺	8	48 ⁺	9 ⁺
	Lab		17 ⁺	8	46 ⁺	11 ⁺
Colour	True	Relative	14	4	20 ⁺	5
TDS		mg L^{-1}	53	4	62	46
NFR	Total	mg L^{-1}	10	7	42	5
	Fixed	mg L^{-1}	9	6	38	4
Alkalinity	Total	mg L^{-1}	46.4	3.1	53.2	40.8
	Bicarbonate	mg L^{-1}	56.6	3.8	64.9	49.7
Hardness	Total	mg L^{-1}	45.9	3.1	54.0	39.9
	Non-carbonate	mg L^{-1}	0.4	0.7	2.6	0
Free CO ₂		mg L^{-1}	3.36	2.42	9.19	0
Phosphorus	Particulate	mg L^{-1}	0.016	0.009	0.039	<0.006
	Ortho PO ₄	mg L^{-1}	0.003	0.003	0.012	<0.003
	Total Dissolved	mg L^{-1}	0.010	0.006	0.024	0.003
	Total	mg L^{-1}	0.026	0.008	0.049	0.013
Nitrogen	Particulate	mg L^{-1}	0.049	0.021	0.120	<0.010
	Total Dissolved	mg L^{-1}	0.235	0.043	0.330	0.170
	NO ₂ -NO ₃	mg L^{-1}	0.019	0.025	0.088	<0.010
	Total NH ₃	mg L^{-1}	0.016	0.011	0.042	<0.005
	Total	mg L^{-1}	0.28	0.05	0.44	0.22
Carbon	DOC	mg L^{-1}	6.04	0.73	8.80	3.79
	POC	mg L^{-1}	0.363	0.153	0.900	0.160
	TOC	mg L^{-1}	6.40	0.71	8.96	4.53
	DIC	mg L^{-1}	10.9	0.7	12.8	9.6
Chlorophyll-a		mg L^{-1}	0.002	0.001	0.004	<0.001
Ca	Dissolved	mg L^{-1}	12.3	1.0	14.2	10.4
Mg	Dissolved	mg L^{-1}	3.7	0.3	4.6	3.3
K	Dissolved	mg L^{-1}	1.02	0.07	1.24	0.93
Na	Dissolved	mg L^{-1}	2.7	0.3	3.5	2.1

Table 15. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L ⁻¹	1.86	0.73	3.38	0.69
Cl	Dissolved	mg L ⁻¹	1.1	0.2	1.6	0.7
F	Dissolved	mg L ⁻¹	0.09	0.01	0.13	0.07
SO ₄	Dissolved	mg L ⁻¹	2.5	0.6	3.6	1.1
Al	Extractable	mg L ⁻¹	0.357 ⁺⁺⁺	0.116	0.712 ⁺⁺⁺	0.161 ⁺⁺⁺
	Dissolved	mg L ⁻¹	0.078	0.050	0.239	<0.100
As	Dissolved	mg L ⁻¹	0.0002	0.0001	0.0003	<0.0001
Ba	Total	mg L ⁻¹	<0.08	0.02	0.13	<0.08
Cd	Total	mg L ⁻¹	<0.0001	0.0001	0.0004 ⁺⁺⁺	<0.0001
	Dissolved	mg L ⁻¹	<0.0001	0.0001	0.0002	<0.0001
Co	Total	mg L ⁻¹	<0.0005	0.0002	0.0012	<0.0005
Cu	Total	mg L ⁻¹	0.0053 ⁺⁺⁺	0.0091	0.0400 ⁺⁺⁺	0.0005
	Dissolved	mg L ⁻¹	0.0012	0.0010	0.0040	<0.0005
Fe	Extractable	mg L ⁻¹	0.435 ^{+,+++}	0.131	0.926 ^{+,+++}	0.143
	Dissolved	mg L ⁻¹	0.100	0.085	0.479	0.033
Pb	Total	mg L ⁻¹	0.0008	0.0006	0.0021 ⁺⁺⁺	<0.0007
	Dissolved	mg L ⁻¹	0.0008	0.0009	0.0035	<0.0007
Mn	Extractable	mg L ⁻¹	0.019	0.013	0.081 ⁺	0.004
	Dissolved	mg L ⁻¹	0.003	0.004	0.016	<0.002
Ni	Total	mg L ⁻¹	0.0011	0.0006	0.0032	<0.0005
	Dissolved	mg L ⁻¹	<0.0005	0.0003	0.0010	<0.0005
Se	Dissolved	mg L ⁻¹	0.0001	0.0001	0.0004	<0.0001
V	Total	mg L ⁻¹	0.0009	0.0004	0.0018	<0.0005
	Dissolved	mg L ⁻¹	<0.0005	0.0002	0.0008	<0.0005
Zn	Total	mg L ⁻¹	0.0049	0.0057	0.0184	0.0006
	Dissolved	mg L ⁻¹	0.0016	0.0024	0.0081	<0.0003
Discharge	Daily	m ³ sec ⁻¹	814	104	990	665

⁺ Exceeds limit for drinking water.

⁺⁺ Exceeds limit for recreational use.

⁺⁺⁺ Exceeds limit for protection of aquatic life.

exceeded the aquatic life guideline, by a factor of 3.5 on average and by as much as a factor of 7. Extractable Fe exceeded the drinking water and aquatic life guidelines in all samples; by about 45% on average. The total Cu concentration exceeded the aquatic life guideline by a factor of 2.7 on average, and by as much as a factor of 20 at peak concentrations. Peak concentrations of total Cd and total Pb also exceeded the respective aquatic life guidelines. Peak extractable Mn exceeded the drinking water objective.

3.3.1.2 Water Quality - Water Discharge Relationships

Positive correlations dominated the significant relationships between water quality and discharge at this station. These positive correlates included TDS, alkalinity (total and bicarbonate), total hardness, ortho PO_4 , dissolved Ca and Se (Figure 20). Negative correlates were particulate N and POC.

The dominance of positive correlations at this site is the reverse of the situation in the natural rivers and in the Burntwood River at Thompson prior to diversion, where negative correlations dominated (Sections 3.1.3 and 3.4.2.1). This suggests negative correlations probably dominated at the Rat River site prior to diversion as well. The current nature of the discharge relationships most likely is a product of shoreline erosion along the lower Rat River.

None of these correlations is of much use for impact prediction or management. The regressions accounted for just 15 to

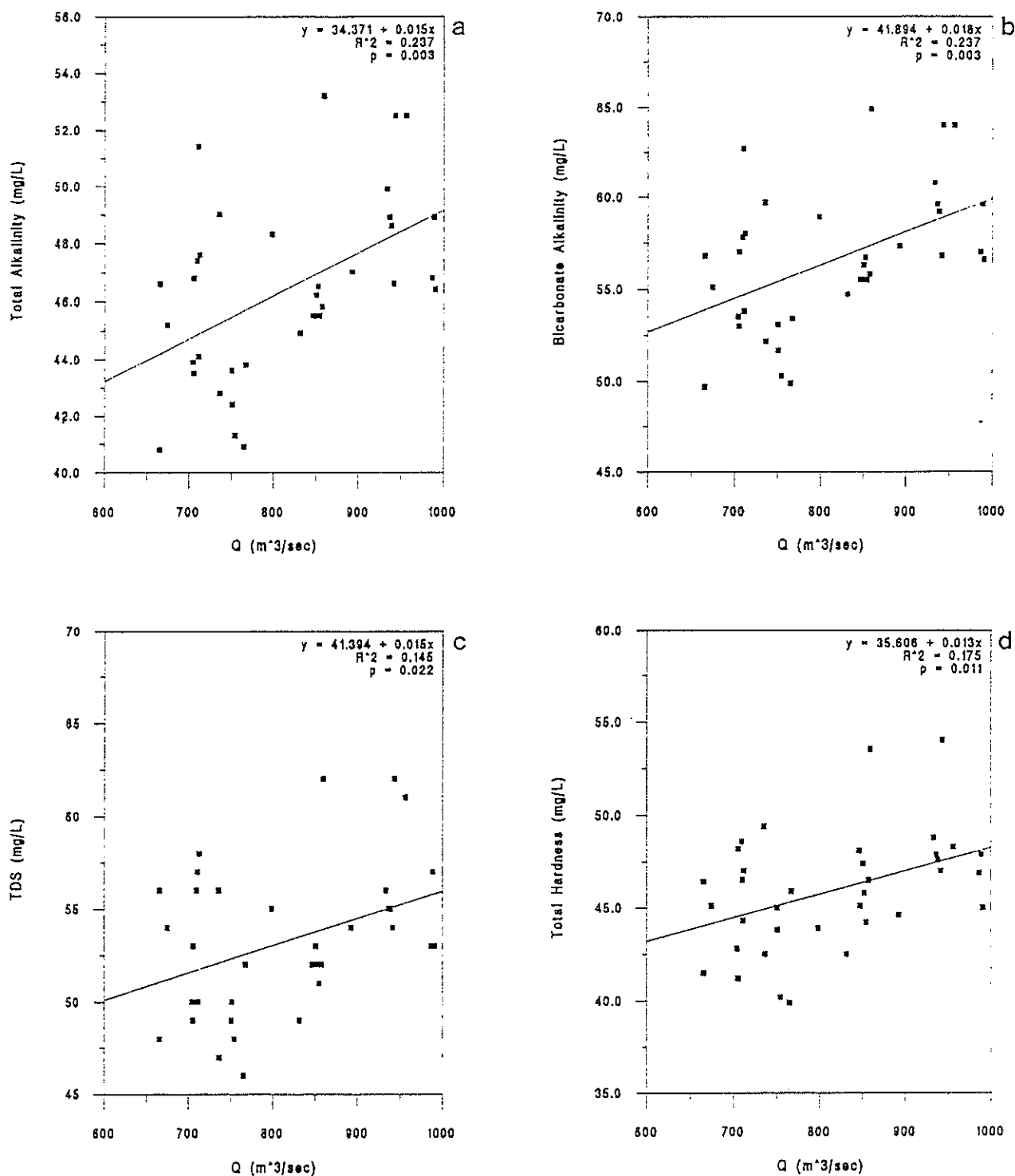


Figure 20. Water chemistry - water discharge relationships in the Rat River above Threepoint Lake, 1987-1989. a. total alkalinity b. bicarbonate alkalinity c. TDS d. total hardness e. ortho PO₄ f. particulate N g. POC h. dissolved Ca i. dissolved Se.

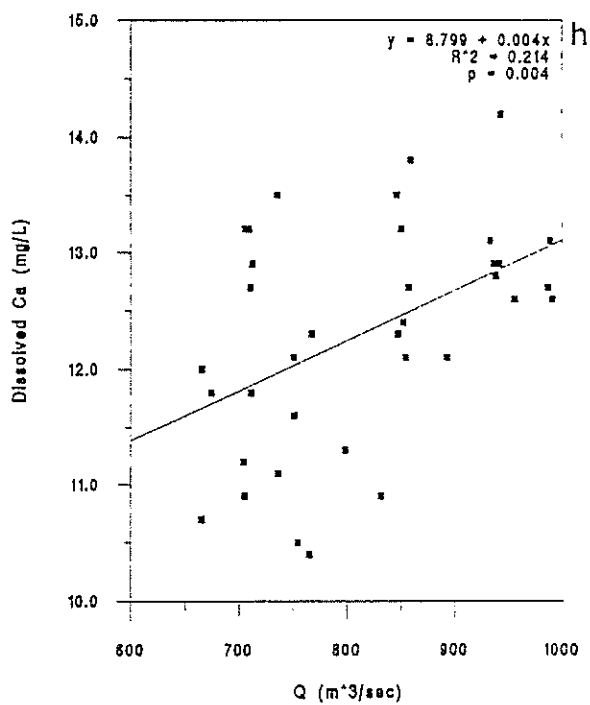
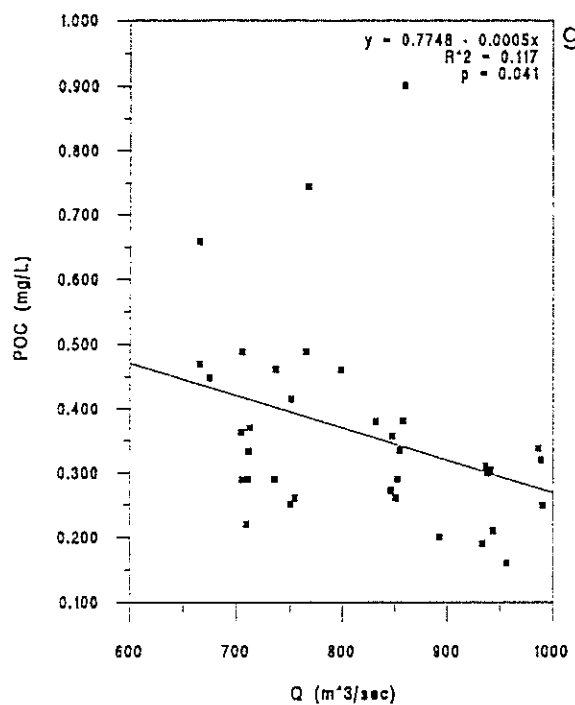
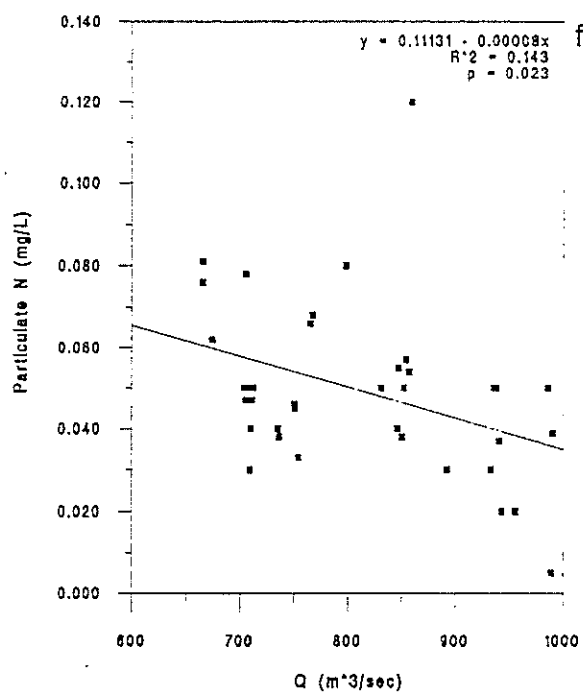
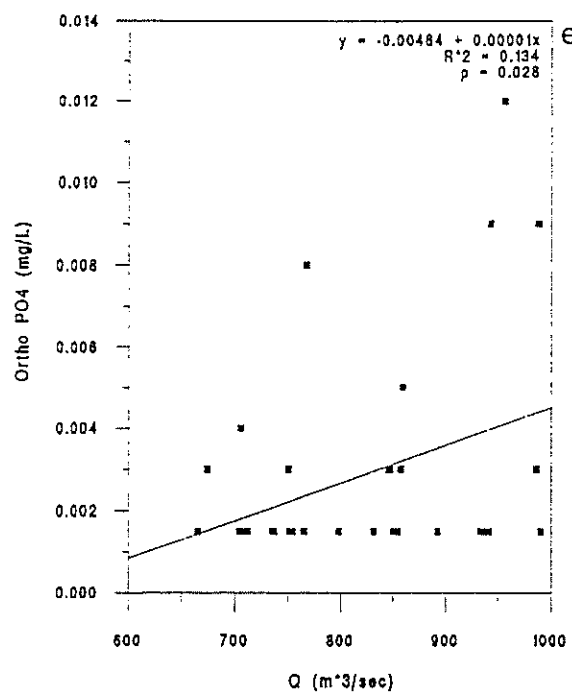


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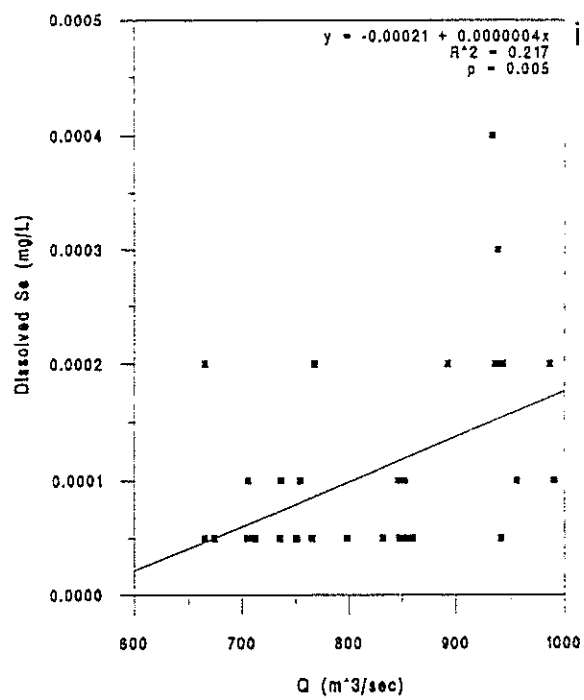


Figure 20. Continued.

25% of total variance in the dependent variables and none of the discharge-correlated variables was associated with adverse impacts either at this site or downstream.

3.3.1.3 Seasonal Differences

Few variables displayed distinct differences between the ice-cover and ice-free seasons during the FEMP study, although TDN, dissolved $\text{NO}_2\text{-NO}_3$, total N, and total Ni occurred in higher concentrations under ice while chlorophyll a concentrations were higher in the ice-free season (Figure 21). None of these variables was correlated with discharge, eliminating discharge fluctuations as a possible cause of the seasonality. The higher concentrations of total N and TDN in winter can largely be attributed to the higher dissolved $\text{NO}_2\text{-NO}_3$ levels. Reduced algal growth rates and standing crops under ice, and therefore lower dissolved N demand, can lead to a buildup of dissolved N constituents, and particularly of NO_3 (Wetzel 1983). Chlorophyll a concentrations were lower under ice-cover, supporting this hypothesis. The lower algal biomass under ice-cover was probably a result of reduced light penetration leading to light limitation. There is no obvious explanation for the higher Ni levels in winter.

The pre-development dataset included only one winter (Dec. 1972, Jan. and Feb. 1973) of measurements. This precluded a statistical analysis of changes in seasonal patterns since development, but was sufficient for a qualitative analysis of seasonal differences. This analysis indicated there was no evidence

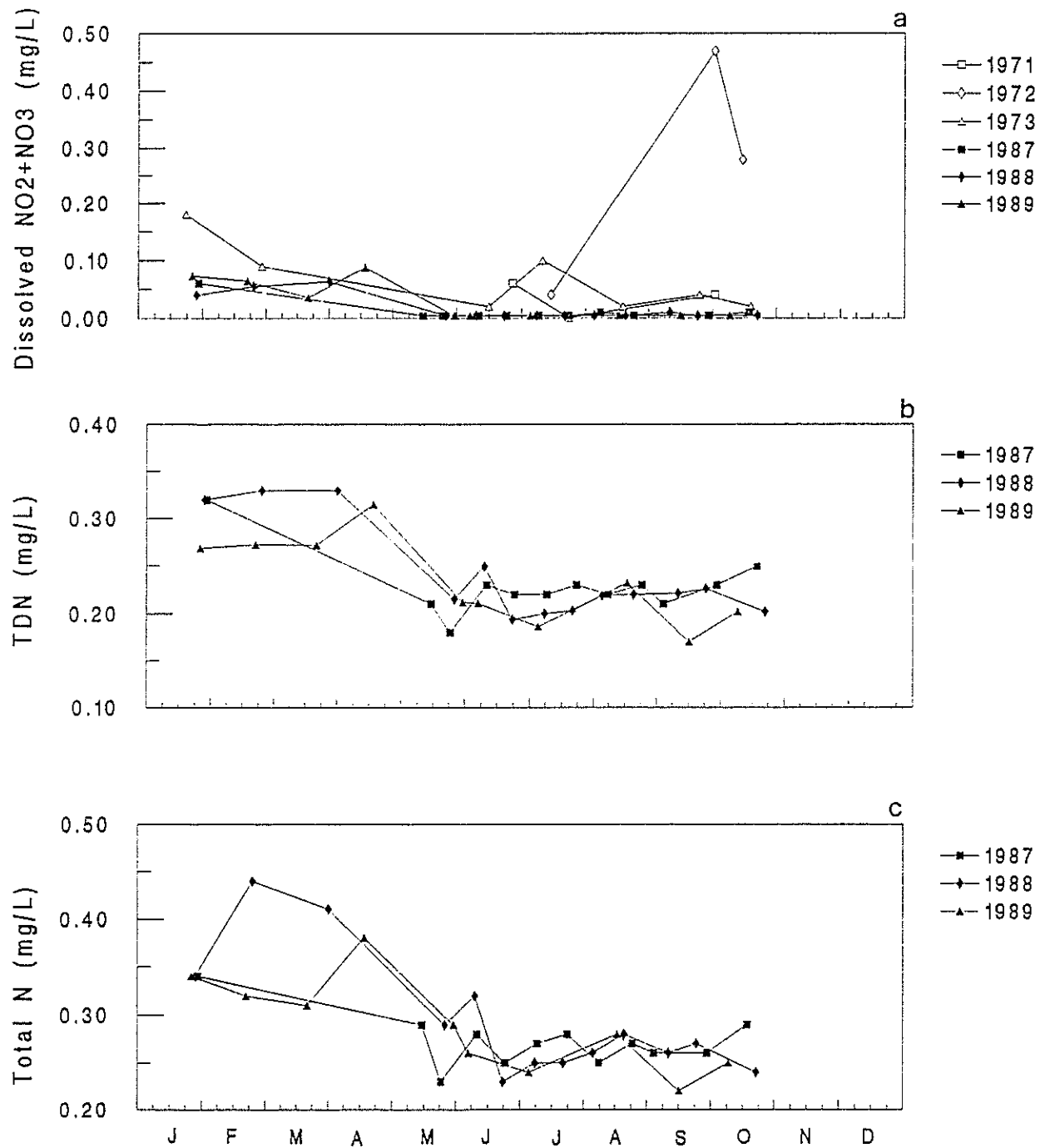


Figure 21. Seasonal water quality trends in the Rat River above Threepoint Lake. a. dissolved $\text{NO}_2\text{-NO}_3$ b. TDN c. total N d. chlorophyll a e. dissolved Zn f. total Ni.

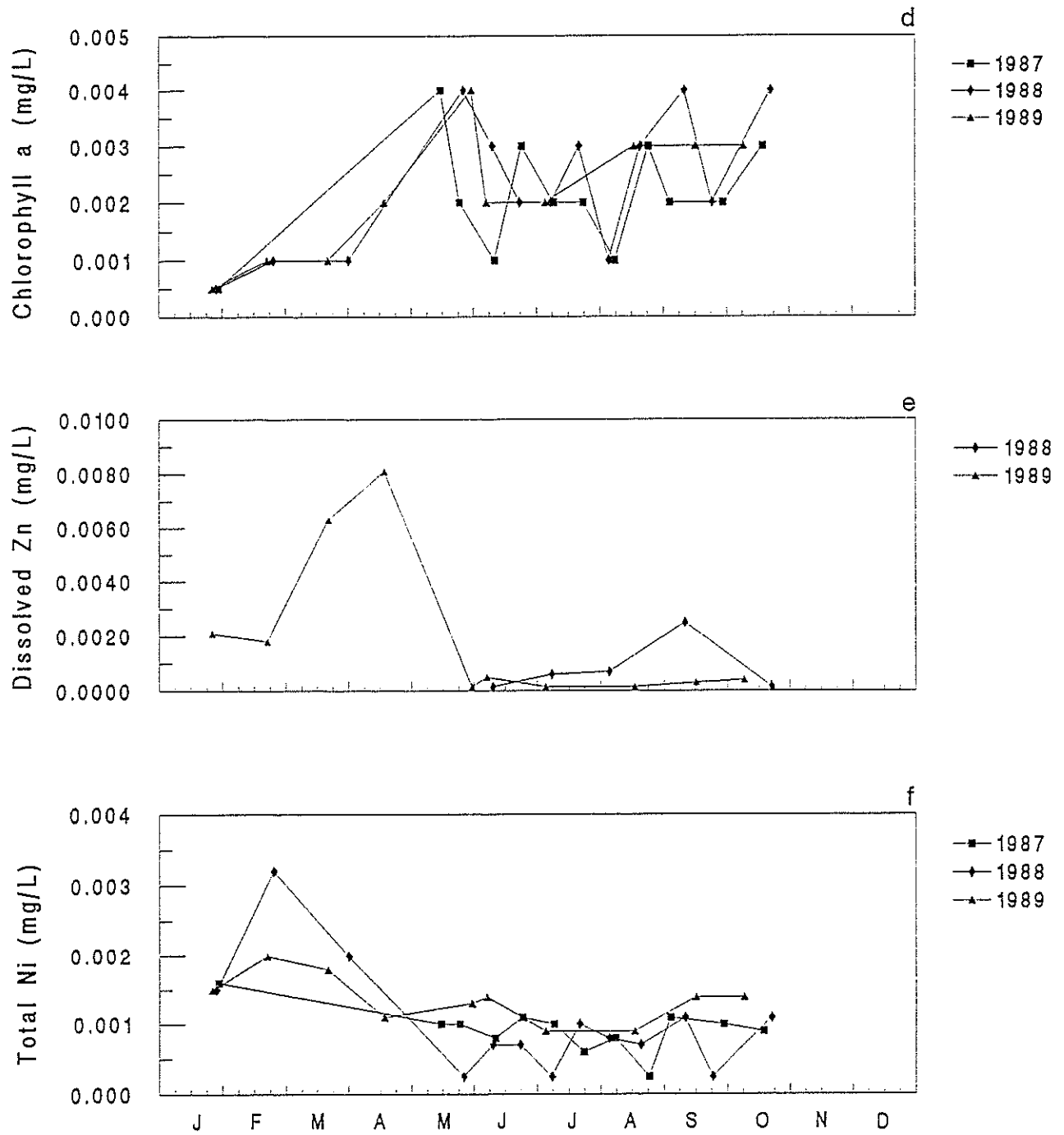


Figure 21. Continued.

of differences between the ice-cover and ice-free seasons in any of the measured variables. However, most of the variables which displayed seasonal trends following diversion were not measured in the pre-diversion studies. Only dissolved $\text{NO}_2\text{-NO}_3$ was consistently measured before diversion, and the range of concentrations under ice-cover overlapped the range during the ice-free season. However, the lowest values tended to occur in summer and the highest in winter. The seasonal difference during the FEMP study was just more pronounced than it was prior to diversion. This could be a product of improved analytical or sampling methods rather than representing any effect of CRD.

3.3.1.4 Changes in Water Quality Since Churchill River Diversion

Annual mean conductivity, alkalinity (total), hardness (total), TOC, ortho PO_4 , Ca, and reactive Si all decreased by 15 to 60% between the pre-diversion (1972-1973) and post-diversion (1987-1989) sampling periods (Table 16). Significant increases in Na, K, Cl, and F, by 10-80%, also occurred. All of these changes are consistent with the differences in water quality between the Churchill and Rat rivers which existed prior to diversion (Table 17).

The changes in Ca, Cl, F, and ortho PO_4 appear to have been a direct result of the replacement of local Rat River flows with water from the Churchill River. However, the changes in conductivity, alkalinity, hardness, Na, and K were smaller than would be expected if the only impact of CRD was substitution of the

Table 16. Evaluation of the significance of changes in selected water quality parameters in the Rat River above Threepoint Lake following Churchill River diversion. X_{pre} denotes the mean for the pre-development years 1972 and 1973. X_{post} denotes the mean for post-development years 1987 through 1989. p is the probability of Student's t -statistic for the comparison.

	pH	Conductivity $\mu S\ cm^{-1}$	Turbidity NTU	Tot Alk $mg\ L^{-1}$	Tot Hard $mg\ L^{-1}$	TOC $mg\ L^{-1}$	NO_2NO_3 $mg\ L^{-1}$	Ortho PO_4 $mg\ L^{-1}$	Tot P $mg\ L^{-1}$	Diss Na $mg\ L^{-1}$
Rat River										
p	0.064	0.002	0.006	0.013	<0.001	0.023	0.049	0.032	0.536	0.010
X_{pre}	7.9	120.	5.	56.4	61.2	10.7	0.237	0.005	0.023	2.2
X_{post}	7.66	102.	17.	46.4 ¹	45.9 ¹	6.38	0.019	0.002 ¹	0.026	2.7
	Diss K $mg\ L^{-1}$	Diss Ca $mg\ L^{-1}$	Diss F $mg\ L^{-1}$	Diss SO_4 $mg\ L^{-1}$	React Si $mg\ L^{-1}$	Diss Fe $mg\ L^{-1}$	Diss Mn $mg\ L^{-1}$	Diss Cu $mg\ L^{-1}$	Diss Pb $mg\ L^{-1}$	Diss Zn $mg\ L^{-1}$
p	0.004	<0.001	0.049	0.630	0.001	0.485	i.d.	1.000	i.d.	0.728
X_{pre}	0.9	18.2	1.0	2.7	3.6	0.07	<0.01	0.002	<0.001	0.007
X_{post}	1.02	12.3 ¹	1.1	2.5	1.83	0.096	0.003	0.0010	0.0007	0.0014

i.d. Insufficient data.

¹ Discharge-standardized variables.

Table 17. Comparison of water quality in the Rat River before (1972-1973) and after (1987-1989) Churchill River diversion with water quality in the Churchill River near Leaf Rapids (1972-1973). All values are annual means. Units are mg L^{-1} except where noted.

	Rat River		Churchill River
	Pre-Diversion	Post-diversion	
pH (units)	7.9	7.66	7.6
Conductivity ($\mu\text{S cm}^{-1}$)	120	102	91
Turbidity (NTU)	5	17	5
Alkalinity (Total)	56.4	46.4	40.4
Hardness (Total)	61.2	45.9	41.9
TOC	10.7	6.38	8.7
NO_2NO_3	0.237	0.019	0.157
Ortho PO_4	0.005	0.002	0.003
Total P	0.023	0.026	0.017
Dissolved Na	2.2	2.7	3.0
Dissolved K	0.9	1.02	1.2
Dissolved Ca	18.2	12.3	12.5
Dissolved Cl	1.0	1.1	1.2
Dissolved F	0.005	0.09	0.08
Dissolved SO_4	2.7	2.5	3.6
Reactive Si	3.6	1.83	2.6
Dissolved Fe	0.07	0.096	0.05
Dissolved Mn	<0.01	0.003	<0.01
Dissolved Cu	0.002	0.0010	0.002
Dissolved Pb	<0.001	0.0007	<0.001
Dissolved Zn	0.007	0.0014	0.002

water source. Shoreline erosion along the diversion route may be contributing Na, K, Mg, and carbonates, thereby slightly mitigating the changes in these variables which otherwise would have occurred. Conversely, the decreases in TOC and SO_4 were greater than might be expected. The excess decrease in SO_4 is probably explained by changes in analytical methodology, but the greater decrease in TOC is rather surprising. The flooded vegetation and organic soils in SIL and Notigi Reservoir might rather be expected to contribute organic C to the water column, yet the TOC concentration in the Churchill River appears to decrease after passing through these flooded areas. The cause of this decrease cannot be determined with the available information as it is not known if the loss is occurring in one or both organic carbon fractions (i.e., dissolved or particulate). Possible causes include sedimentation of particulates in Notigi Reservoir or the binding of DOC to other suspended material and subsequent sedimentation. This issue will be examined more extensively in the analysis of water quality data for SIL and Notigi Reservoir being conducted by DFO.

A 92% decline in $\text{NO}_2\text{-NO}_3$ also was detected in the analysis (Table 16). This drop is similar to those which occurred in the Burntwood River above Threepoint Lake and in the Footprint River and, therefore, this decrease cannot be attributed to diversion.

Total P concentrations in the Rat River were not lower during the FEMP study than prior to diversion despite lower concentrations in the Churchill River (Table 17). The release of P from flooded soils and vegetation in new reservoirs is well-documented (e.g.,

Baxter and Glaude 1980). This finding suggests phosphorus continues to be released from flooded terrain as late as 13 years after impoundment and diversion. The lack of data for this site between the start of diversion and the FEMP study prevents examination of the temporal trends in P release which may have been higher in the past.

There also has been an increase in turbidity, by a factor of 3.2, since CRD (Table 16). It is not known if a corresponding increase in suspended sediment (NFR) concentrations has occurred at this site due to the lack of pre-diversion measurements. The increase in turbidity was probably caused by eroding glacial clays in the southern part of Notigi Reservoir or along the Rat River above Threepoint Lake. Erosion is a problem on Southern Indian Lake and in the northern part of Notigi reservoir as well, but this material is thought to be largely retained within Notigi Reservoir (Northwest Hydraulic Consultants 1988). The lack of a correlation between turbidity and discharge at this site ($r^2 < 0.001$, $p = 0.97$, $n = 36$) indicates most of the increase in turbidity is due to shoreline erosion in Notigi Reservoir. A positive correlation with discharge would be expected if erosion along the Rat River was the primary cause.

Concentrations of dissolved Fe, Mn, Cu, Pb, and Zn in the Rat River have not been affected by CRD (Table 16). The impact, if any, of diversion on other metal species is unknown due to the lack of pre-development data.

Water quality changes in the Rat River system and in Threepoint Lake have also been examined in studies by Manitoba Natural Resources, Fisheries Branch (Berger and Ramsey 1991, Ramsey et al. 1989). These studies included sample collection in the summers (June-September) of 1986 through 1989 from Rat and Threepoint lakes to document current water quality, and the comparison of these data with published pre-diversion measurements. The water quality changes in the Rat River above Threepoint Lake detected in the present analysis are generally consistent with the changes in Rat and Threepoint lakes following CRD reported in the Fisheries Branch studies. Berger and Ramsey (1991) found lower conductivity, Ca, and reactive Si and higher concentrations of Na and K in both lakes following diversion. Lower Mg concentrations, a variable not found in the pre-development Rat River database, also were found in both lakes following diversion. There was some discrepancy in the impact of CRD on major anions in the two lakes. Concentrations of Cl and SO₄ decreased in Rat Lake but there was no change in either variable in Threepoint Lake. The decrease in Rat Lake was explained by higher concentrations of Cl and SO₄ in the lake than in the Churchill River prior to diversion. The lack of a change in either variable in Threepoint Lake was explained by similar concentrations of both variables in the lake and Churchill River. This finding clarifies the status of the impact of diversion on SO₄ concentrations in the Rat River; a significant decrease must have occurred in the Rat River for there to be no significant change in Threepoint Lake.

The pre-development data sources included in the Berger and Ramsey (1991) study provided better coverage of the nutrient variables in the Rat River system than was possible in the present study; more nutrient variables were measured in Rat Lake both before and after diversion and the NO_3 measurements were not troubled by the analytical problems associated with NO_2 - NO_3 in the Rat River. Berger and Ramsey (1991) did not consider Threepoint Lake nutrient changes in detail because the pre-development database for the lake amounted to analyses on a single sample.

Concentrations of the N species in Rat Lake were most affected by CRD. In particular, NO_3 and suspended N concentrations in Rat Lake between 1986 and 1989 were just 23-25% of the 1973 values. Concentrations of DIC, DOC, and suspended C decreased by 26 to 50%, with the lower DOC consistent with the lower post-diversion TOC levels found in the present study. In contrast to the present study, Berger and Ramsey (1991) found suspended P concentrations in Rat Lake dropped 36% on average following diversion while TDP was not affected. Rat Lake P is equally divided between the dissolved and suspended fractions. Therefore, the lower suspended P levels must have caused an approximately 18% decrease in total P, yet this decrease was not evident in the Rat River above Threepoint Lake. This lack of change in the Rat River may be a product of P release from flooded areas between Rat and Threepoint lakes although this possibility remains to be investigated.

3.3.1.5 Adverse Effects

The only adverse effect of CRD on water quality in the Rat River above Threepoint Lake was the 3-fold increase in turbidity. Prior to diversion, mean turbidity met the Canadian drinking water guideline of 5 NTU without treatment. The water should now be treated for removal of turbidity before consumption.

Increased turbidity also has a number of potential ecological implications, although the significance of these impacts remains to be quantified. The higher turbidity might interfere with the ability of sight-feeding fishes to capture food and increased sediment deposition on fish spawning beds can impair whitefish egg survival (Fudge and Bodaly 1984). The elevated turbidity would also be expected to reduce primary productivity in lakes on the Rat River due to reduced light penetration (Hecky and Harper 1974).

3.3.2 Burntwood River at Outlet of Threepoint Lake

3.3.2.1 Current Water Quality

This site was characterized by violations of water quality guidelines similar to those observed in the three rivers which flow into Threepoint Lake. All turbidity measurements exceeded the drinking water guideline, with peak values exceeding the recreational use objective by about a factor of 3 (Table 18). Mean colour and maximum extractable Mn exceeded the drinking water objectives while the mean extractable Fe concentration was about 60% higher than the guidelines for both drinking water and protection of aquatic life. Mean extractable Al exceeded the

Table 18. Summary statistics for physical and chemical water quality variables measured in the Burntwood River at the outlet to Threepoint Lake; September, 1986, to October, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.5	0.3	8.1	7.0
	Lab		7.59	0.17	7.86	7.28
Conductivity	Field	$\mu\text{S cm}^{-1}$	102	10	119	80
	Lab		105	5	118	97
Turbidity	Field	NTU	23 ⁺	25	160 ^{+, ++}	10 ⁺
	Lab	NTU	20 ⁺	21	140 ^{+, ++}	8 ⁺
Colour	True	Relative	16 ⁺	5	35 ⁺	10
TDS		mg L^{-1}	54	4	62	47
NFR	Total	mg L^{-1}	12	21	134	3
	Fixed	mg L^{-1}	11	21	126	2
Alkalinity	Total	mg L^{-1}	47.9	3.2	54.6	42.6
	Bicarbonate	mg L^{-1}	58.4	4.0	66.6	51.9
Hardness	Total	mg L^{-1}	47.3	2.9	53.9	42.0
	Non-carbonate	mg L^{-1}	0.6	1.4	7.0	0
Free CO ₂		mg L^{-1}	3.09	2.25	9.44	0
Phosphorus	Particulate	mg L^{-1}	0.019	0.018	0.119	<0.006
	Ortho PO ₄	mg L^{-1}	0.003	0.003	0.012	<0.003
	Total Dissolved	mg L^{-1}	0.010	0.005	0.023	0.003
	Total	mg L^{-1}	0.029	0.018	0.130	0.018
Nitrogen	Particulate	mg L^{-1}	0.054	0.043	0.280	<0.010
	Total Dissolved	mg L^{-1}	0.234	0.039	0.330	0.174
	NO ₂ -NO ₃	mg L^{-1}	0.016	0.020	0.063	<0.010
	Total NH ₃	mg L^{-1}	0.012	0.010	0.051	<0.005
	Total	mg L^{-1}	0.29	0.06	0.59	0.23
Carbon	DOC	mg L^{-1}	6.32	0.62	8.50	5.48
	POC	mg L^{-1}	0.380	0.317	2.100	0.150
	TOC	mg L^{-1}	6.70	0.74	8.80	5.77
	DIC	mg L^{-1}	11.3	0.7	13.1	9.8
Chlorophyll-a		mg L^{-1}	0.003	0.001	0.007	<0.001
Ca	Dissolved	mg L^{-1}	12.7	0.9	14.7	10.9
Mg	Dissolved	mg L^{-1}	3.8	0.3	4.6	3.4
K	Dissolved	mg L^{-1}	1.03	0.07	1.24	0.94
Na	Dissolved	mg L^{-1}	2.7	0.3	3.5	2.2

Table 18. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L ⁻¹	1.91	0.71	3.36	0.76
Cl	Dissolved	mg L ⁻¹	1.1	0.2	1.7	0.8
F	Dissolved	mg L ⁻¹	0.09	0.01	0.13	0.07
SO ₄	Dissolved	mg L ⁻¹	2.4	0.6	4.3	0.8
Al	Extractable	mg L ⁻¹	0.377 ⁺⁺⁺	0.209	1.360 ⁺⁺⁺	<0.100
	Dissolved	mg L ⁻¹	<0.100	0.038	0.179	<0.100
As	Dissolved	mg L ⁻¹	0.0002	0.0001	0.0003	<0.0001
Ba	Total	mg L ⁻¹	<0.08	0.02	0.13	<0.08
Cd	Total	mg L ⁻¹	0.0001	0.0001	0.0006 ⁺⁺⁺	<0.0001
	Dissolved	mg L ⁻¹	<0.0001		0.0002	<0.0001
Co	Total	mg L ⁻¹	<0.0005	0.0003	0.0020	<0.0005
Cu	Total	mg L ⁻¹	0.0044 ⁺⁺⁺	0.0059	0.0264 ⁺⁺⁺	0.0005
	Dissolved	mg L ⁻¹	0.0010	0.0004	0.0017	0.0005
Fe	Extractable	mg L ⁻¹	0.477 ^{+,+++}	0.303	2.010 ^{+,+++}	0.135
	Dissolved	mg L ⁻¹	0.103	0.109	0.627	0.020
Pb	Total	mg L ⁻¹	0.0008	0.0009	0.0046 ⁺⁺⁺	<0.0007
	Dissolved	mg L ⁻¹	0.0009	0.0015	0.0050	<0.0007
Mn	Extractable	mg L ⁻¹	0.020	0.028	0.179 ⁺	0.006
	Dissolved	mg L ⁻¹	0.004	0.004	0.017	<0.002
Ni	Total	mg L ⁻¹	0.0013	0.0009	0.0059	<0.0005
	Dissolved	mg L ⁻¹	0.0005	0.0004	0.0015	<0.0005
Se	Dissolved	mg L ⁻¹	0.0001	0.0001	0.0005	<0.0001
V	Total	mg L ⁻¹	0.0010	0.0006	0.0039	<0.0005
	Dissolved	mg L ⁻¹	<0.0005	0.0001	0.0006	<0.0005
Zn	Total	mg L ⁻¹	0.0040	0.0040	0.0182	0.0007
	Dissolved	mg L ⁻¹	0.0011	0.0015	0.0056	<0.0003
Discharge	Daily	m ³ sec ⁻¹	832	105	1020	670

⁺ Exceeds limit for drinking water.

⁺⁺ Exceeds limit for recreational use.

⁺⁺⁺ Exceeds limit for protection of aquatic life.

aquatic life guideline by a factor of 3.7 on average and by as much as a factor of 13.6 at peak concentrations. Total Cu exceeded the aquatic life guideline by a factor of 2 on average and by as much as a factor of 13. Peak total Cd and total Pb also exceeded the respective aquatic life guidelines.

3.3.2.2 Water Quality - Water Discharge Relationships

All significant correlations between water quality and discharge were positive at this site. These correlates included conductivity (lab), TDS, alkalinity (total and bicarbonate), total hardness, and extractable Al (Figure 22). The slopes and intercepts of discharge relationships for alkalinity, TDS and hardness at this site were the same as in the Rat River above Threepoint Lake as was the generally weak character of the correlations. Only 12-21% of the variance in the dependent variables was accounted for by the regression on discharge. There was no resemblance between the water chemistry - discharge relationships in the unregulated inflows (Burntwood and Footprint rivers) to those at the outflow (Figures 8, 9, and 22). Again, the discharge correlations for this site are of limited use for water quality management or prediction due to the weakness of the relationships and the lack of a correlation for any variable responsible for an adverse effect at this site or downstream.

No examination of the pre-development relationships between water quality and discharge is possible for this site due to the lack of pre-development discharge and water quality data. These

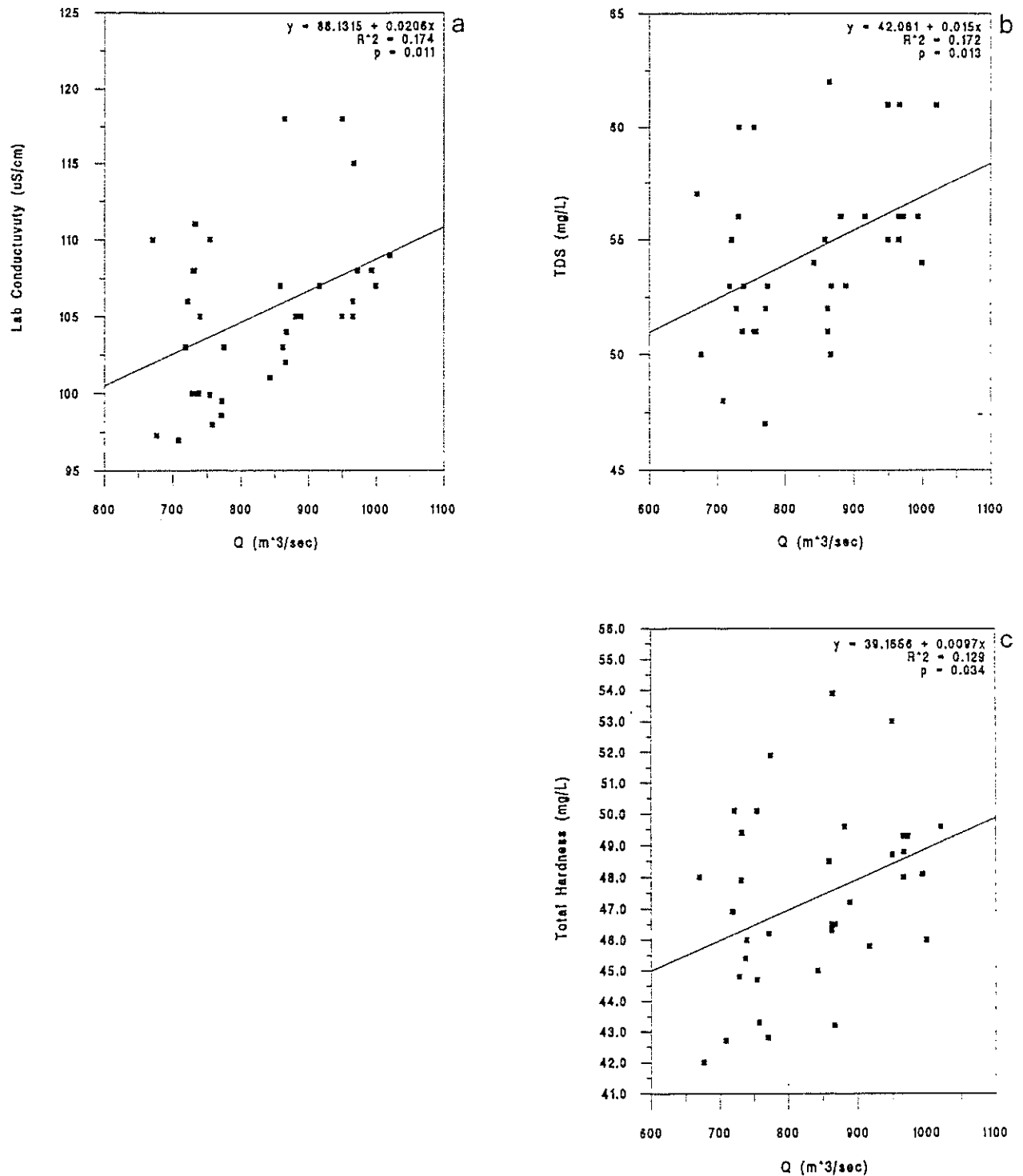


Figure 22. Water chemistry - water discharge relationships in the Burntwood River at the outlet of Threepoint Lake, 1987-1989. a. lab conductivity b. TDS c. total hardness d. total alkalinity e. bicarbonate alkalinity f. extractable Al.

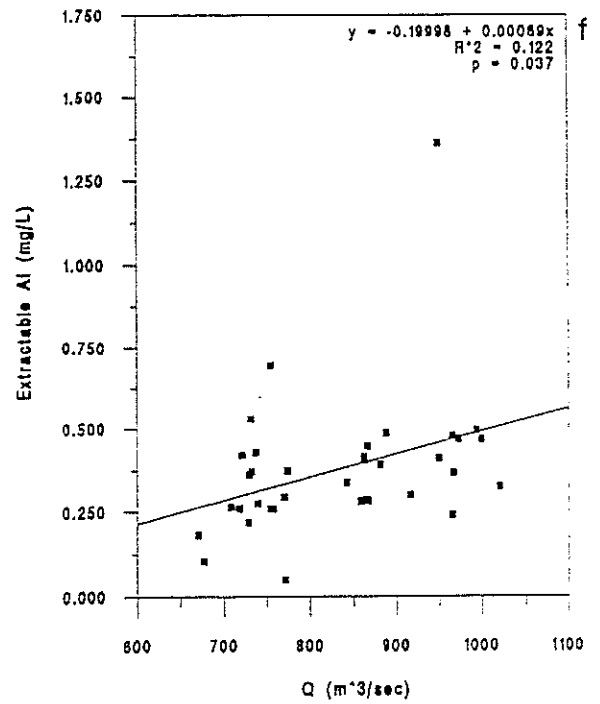
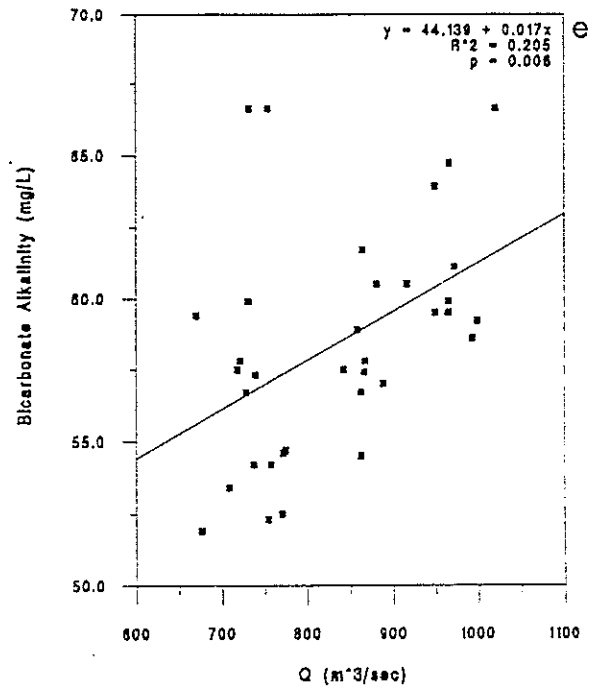
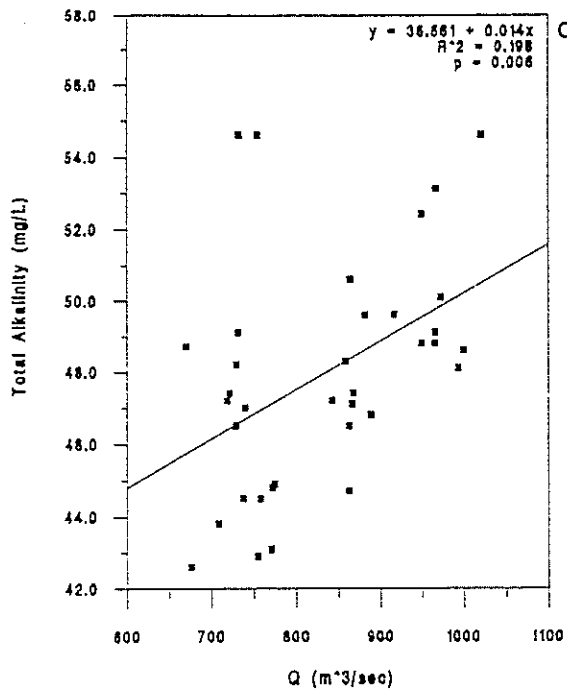


Figure 22. Continued.

relationships probably were most similar to those in the Burntwood River above Threepoint lake as this was the major inflow to the lake prior to diversion.

3.3.2.3 Seasonal Differences

Few differences in water quality were evident between the ice-free and ice-cover seasons during the 1987-1989 FEMP study period. As in the Rat River at the inlet to Threepoint Lake, TDN and dissolved $\text{NO}_2\text{-NO}_3$ concentrations were higher under ice-cover, probably due to the same cause as in the Rat River (Figure 23). Dissolved Zn concentrations also were much higher under ice-cover. The highest dissolved Zn concentrations occurred under ice cover in all three major inflows to Threepoint Lake as well, although the seasonality was not as distinct as at the outlet. The cause of this winter increase is unknown. The dynamics of Zn in freshwaters has been little studied (Wetzel 1983).

3.3.3 Nelson House Bacterial Testing

Geometric mean fecal coliform bacteria counts at the eight survey sites generally ranged between 1 and 6 bacteria per 100 mL for the seasonal study (Figure 24). The only exception was School Bay, where the geometric mean count was 56 per 100 mL. Fecal coliform counts in water samples from Industrial Bay, Church Bay (East Ramp), Government Dock, Inspiration Point, New Development Area (Across from Cemetery), and Metis Beach were consistently below 200 per 100 mL. Counts in 4 of the 5 samples from Nursing

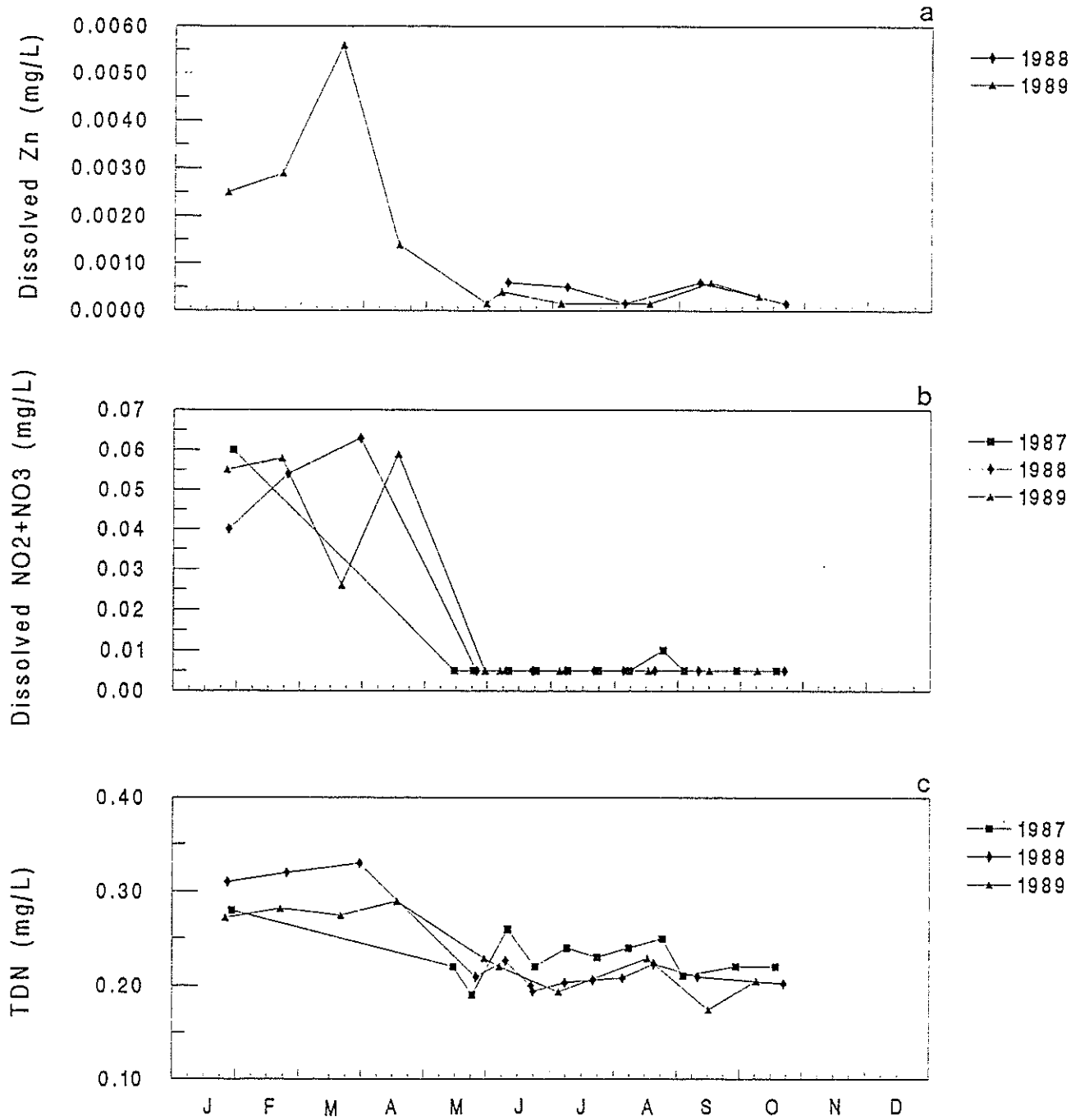


Figure 23. Seasonal water quality trends in the Burntwood River at the outlet of Threepoint Lake. a. dissolved Zn b. dissolved NO₂-NO₃ c. TDN.

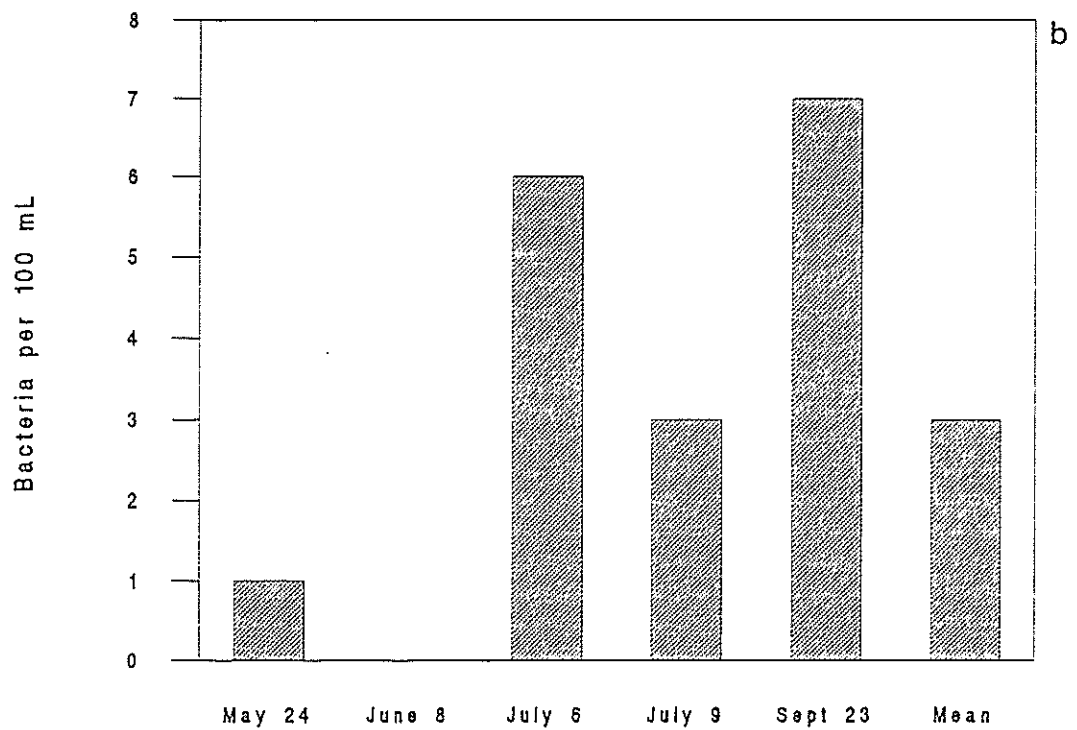
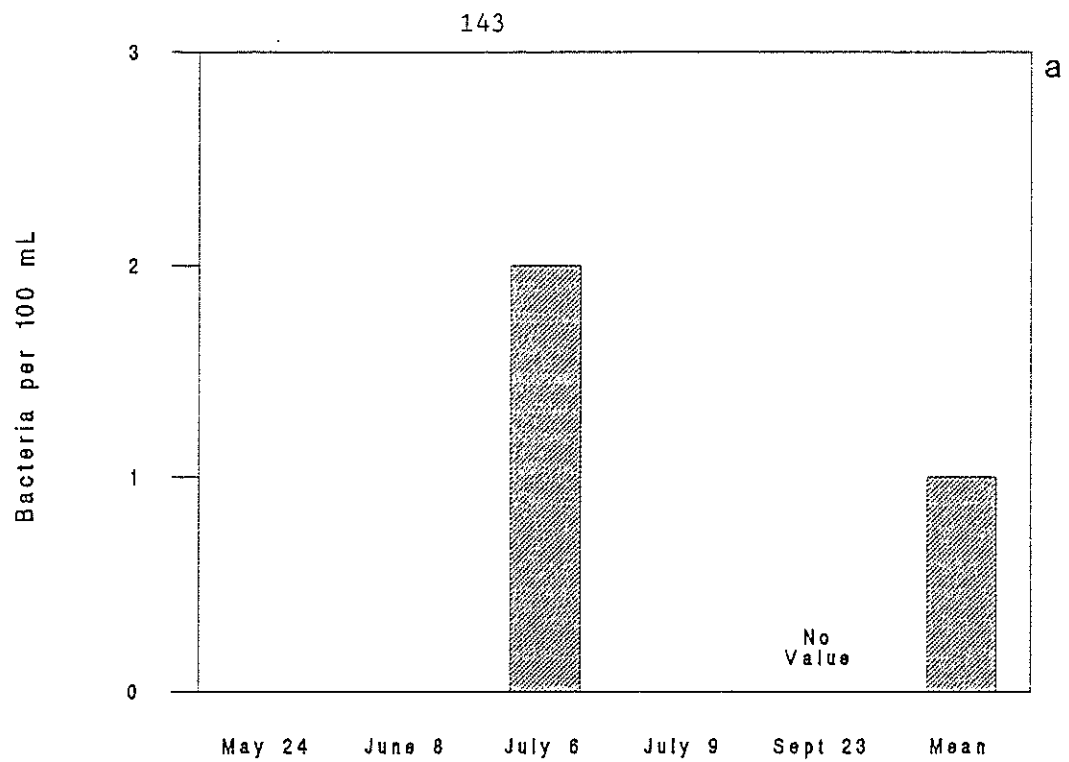


Figure 24. Fecal coliform bacteria counts from the seasonal study in the vicinity of Nelson House. Mean is the geometric mean. Dotted line indicates Canadian Water Quality Guideline.
 a. Church Bay (East Ramp) b. Government Dock c. Industrial Bay d. Inspiration Point
 e. Metis Beach f. New Development Area (Across from Cemetery) g. Nursing Station Bay
 h. School Bay.

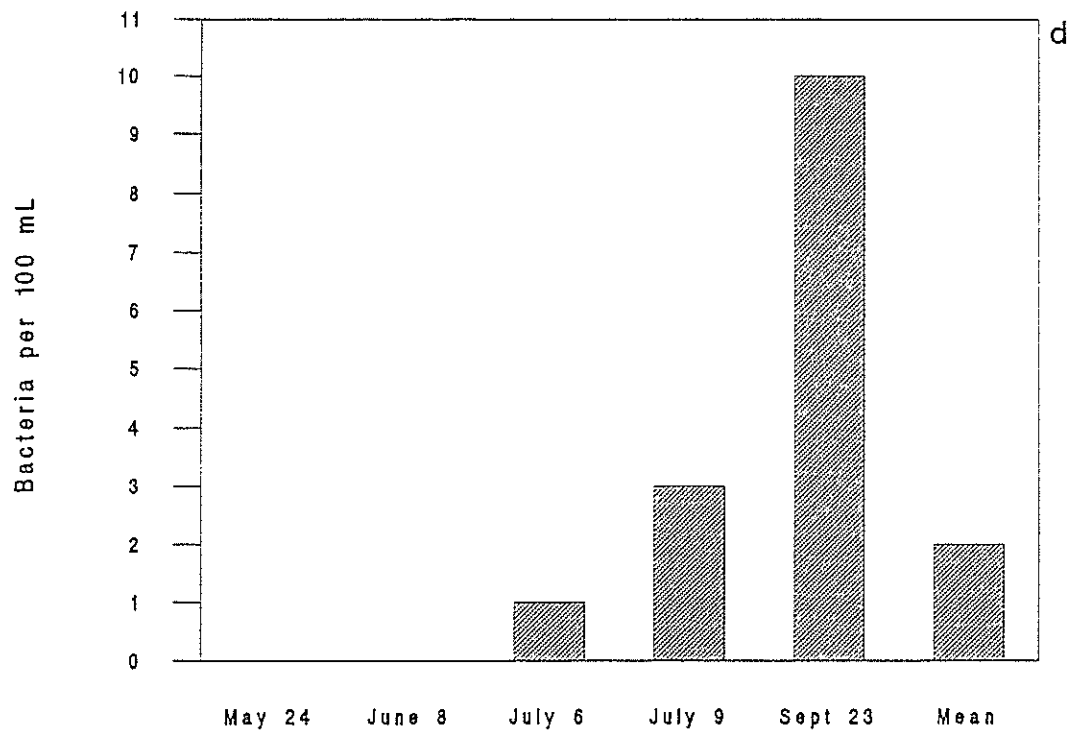
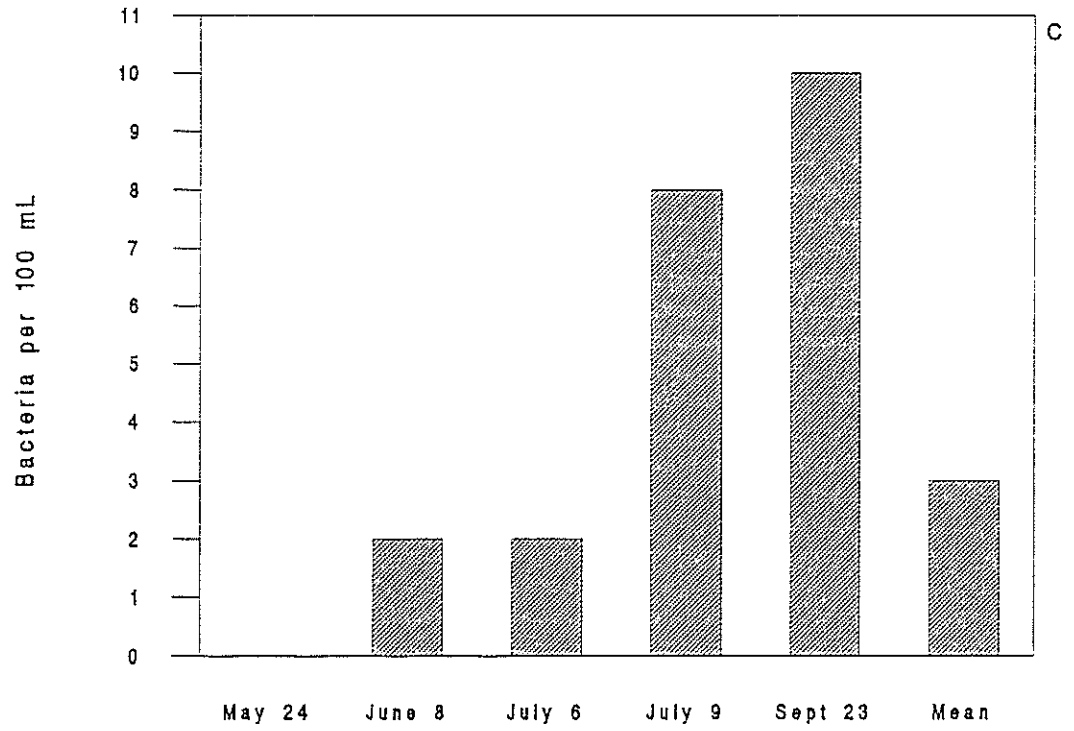


Figure 24. Continued.

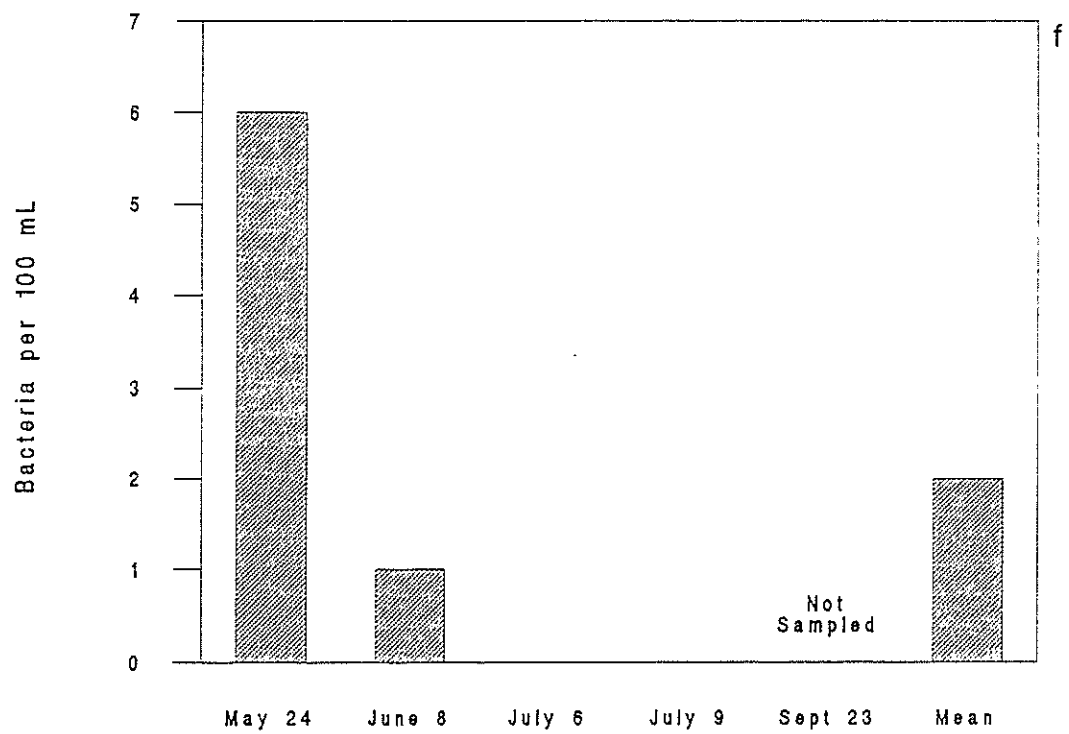
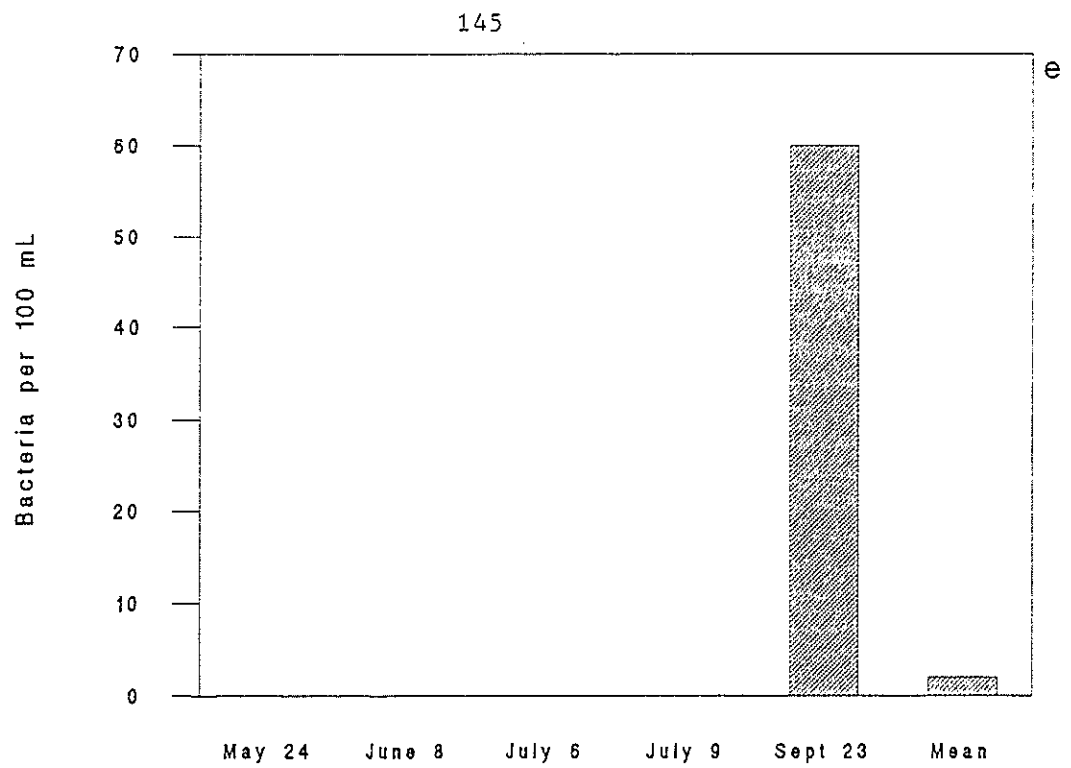


Figure 24. Continued.

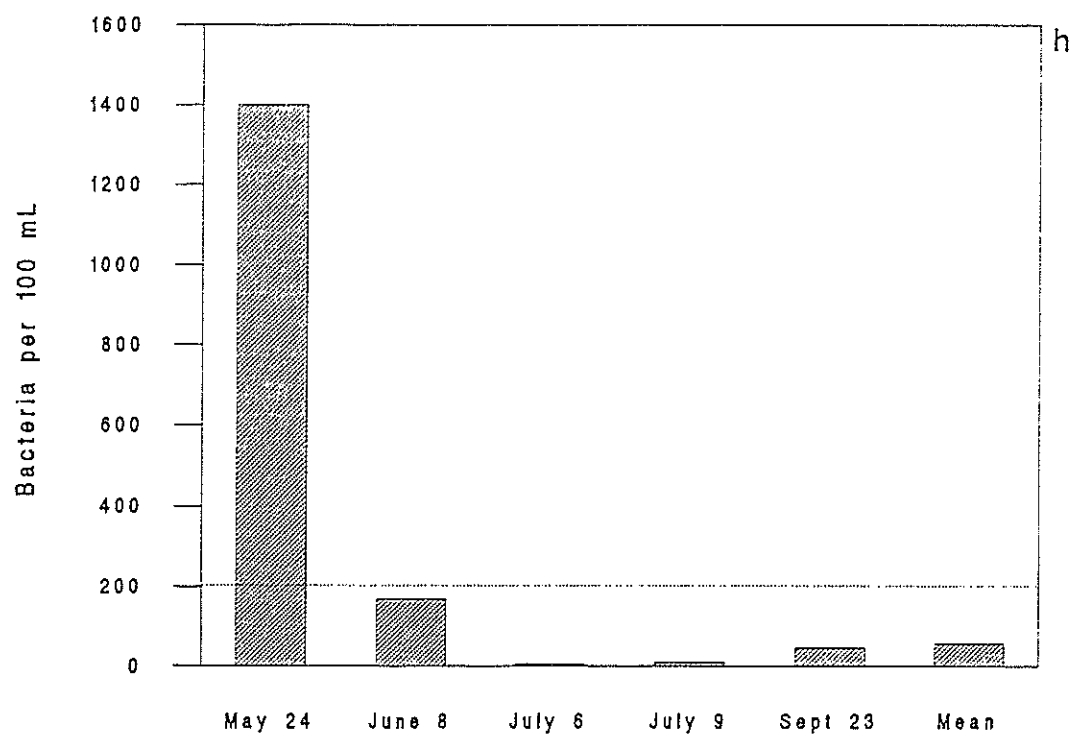
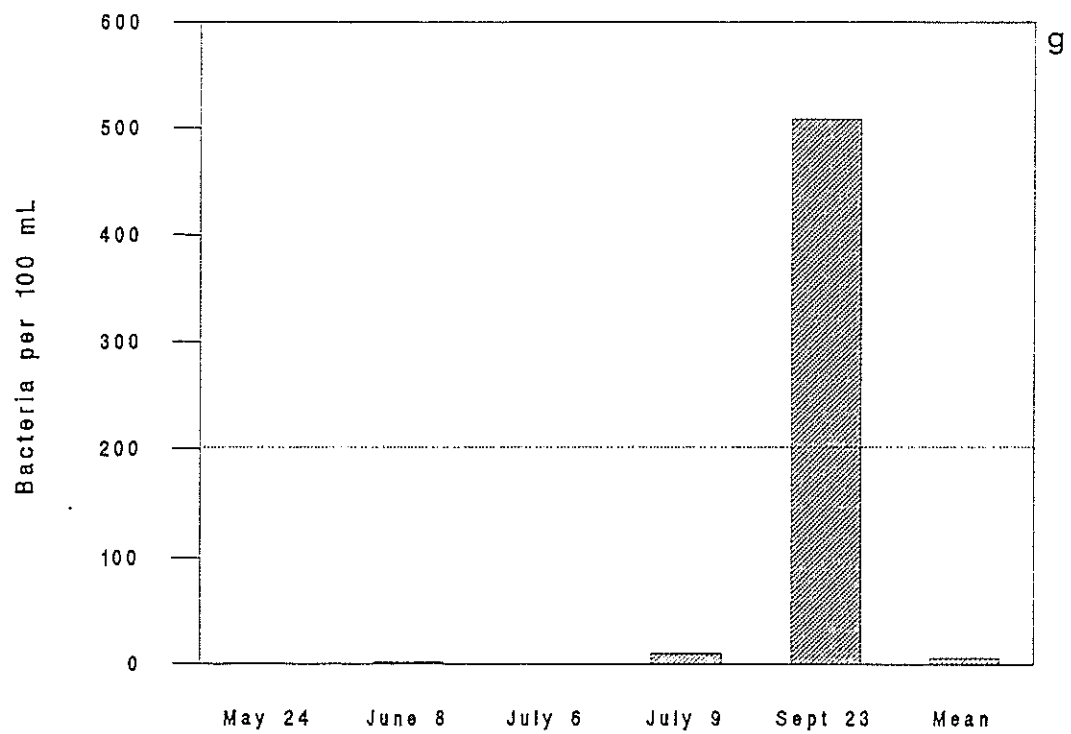


Figure 24. Continued.

Station Bay were well below 200 bacteria per 100 mL while a count of 508 per 100 mL was found on September 23, 1987.

Intensive studies of bacterial water quality in the vicinity of the sewage treatment plant outfall were undertaken on August 12 and 25, 1987. Fecal coliform counts in 16 of the 19 samples were below 50 per 100 mL (Figure 25). Counts in the other 3 samples ranged between 530 and 4100 per 100 mL. The geometric mean of the sample counts was 38 per 100 mL with 16% of the samples exceeding 400 counts per 100 mL. The few high coliform counts were transient. At sites 3, 7, and 8, respectively, counts were 4100, 43, and 530 per 100 mL on August 12, compared with 7, 1910, and 0 on August 25.

Bacteriological water quality at the Nelson House study sites was generally acceptable for recreational water uses, as indicated by the Health and Welfare Canada Recreational Water Quality Guideline of 200 bacteria per 100 mL. The intensive study demonstrated this criterion was met even in the area of the sewage treatment plant outfall.

3.4 Burntwood River at Thompson

The Burntwood River at Thompson is the only site in the FEMP study which was sampled before, during, and immediately after construction of an hydroelectric project in addition to the current study. This site also has the best baseline data record of any in the study, sampled continuously from 1967 to 1974. Only 10 variables were common to the four periods, however, thus seriously limiting the nature of the water quality changes which can be

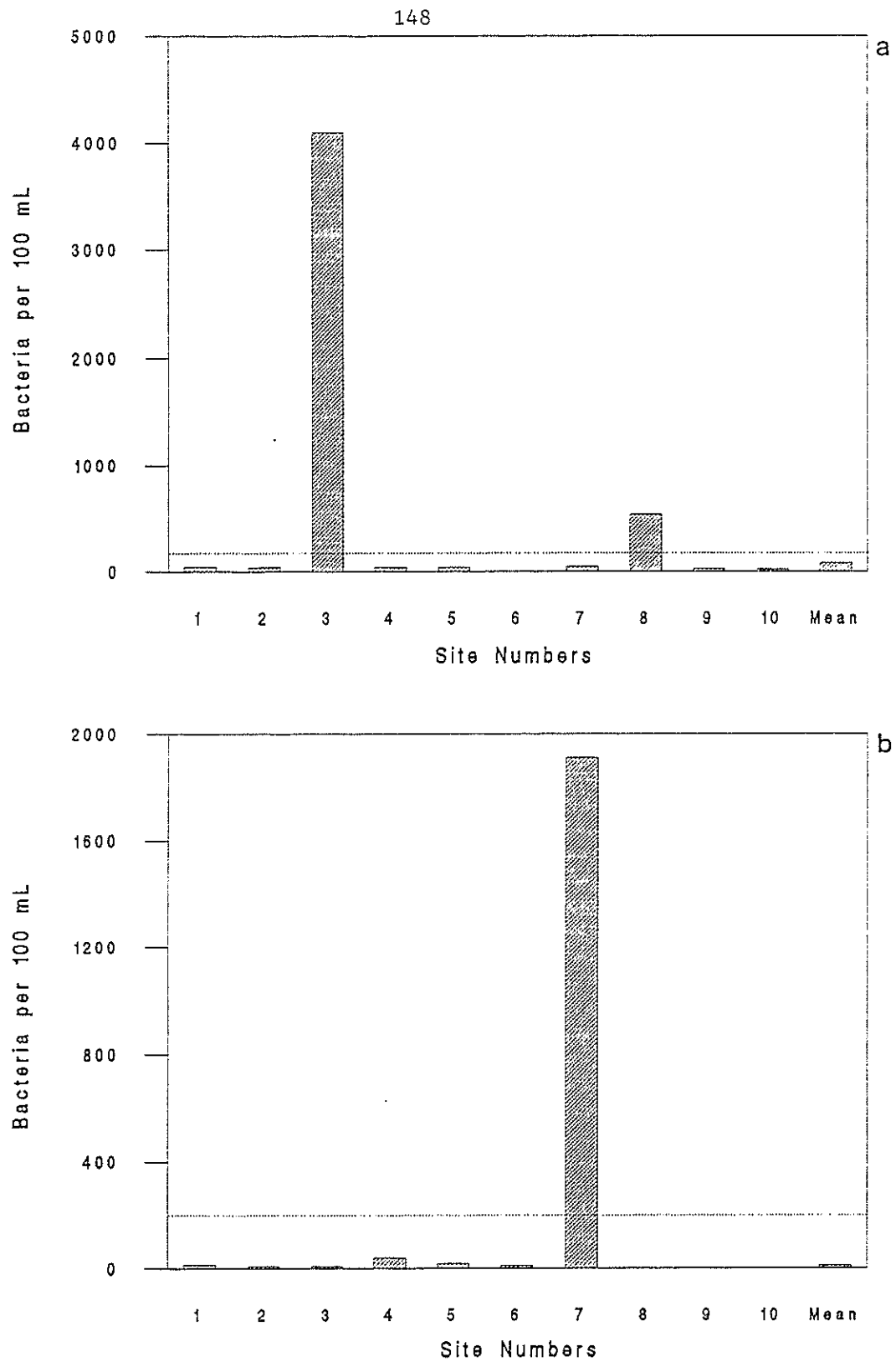


Figure 25. Fecal coliform bacteria counts from the intensive study in the vicinity of the Nelson House Sewage Treatment Plant Outfall. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Sampled 12 August 1987 b. Sampled 25 August 1987. See Appendix 1 for site locations and descriptions.

conclusively stated. A number of variables were added to the analytical program during the construction period, and changes have occurred in several of these since diversion. The lack of measurements in the pre-diversion period raises some question about the directness of any relationship between these changes and CRD. In some cases there may be indirect evidence that the change is applicable to the pre-development period, but in other instances, there is no way to be certain.

3.4.1 Current Water Quality

Water quality in the Burntwood River near Thompson had similar characteristics to the other sites in the local (Burntwood, Footprint, and Aiken rivers) and Churchill River drainages. Levels of most parameters were well within Canadian guidelines, and those which exceeded the guidelines were the same as in the local drainages. All turbidity measurements exceeded the drinking water guideline, by about a factor of 6 on average (Table 19). The peak level measured in the field also exceeded the guideline for recreational uses by almost 30%. Colour exceeded the drinking water objective by about 40% on average and by as much as 100%. Extractable Fe exceeded the aquatic life and drinking water objectives by a factor of 2 on average and by as much as a factor of 3. All measurements of extractable Al exceeded the aquatic life guideline; by a factor of 4.9 on average and by as much as a factor of 8.3. The mean total Cu concentration was a factor 2 higher than the aquatic life guideline on average and as much as a factor 12

Table 19. Summary statistics for physical and chemical water quality in the Burntwood River at Thompson, January, 1987, to October, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.6	0.3	8.2	7.1
	Lab		7.68	0.24	7.96	6.80
Conductivity	Field	$\mu\text{S cm}^{-1}$	111	10	142	90
	Lab		116	6	133	104
Turbidity	Field	NTU	31 ⁺	9	65 ^{+,++}	18 ⁺
	Lab		27 ⁺	6	37 ⁺	15 ⁺
Colour	True	Relative	21 ⁺	5	30 ⁺	10
TDS		mg L^{-1}	60	4	68	54
NFR	Total	mg L^{-1}	17	5	32	7
	Fixed	mg L^{-1}	15	5	29	6
Alkalinity	Total	mg L^{-1}	53.1	3.1	59.9	46.4
	Bicarbonate	mg L^{-1}	64.7	3.8	73.0	56.6
Hardness	Total	mg L^{-1}	52.8	3.2	62.7	47.6
	Non-carbonate	mg L^{-1}	0.6	1.2	6.5	0
Free CO ₂		mg L^{-1}	2.86	2.01	7.97	0
Phosphorus	Particulate	mg L^{-1}	0.023	0.008	0.042	<0.006
	Ortho PO ₄	mg L^{-1}	0.003	0.005	0.027	<0.003
	Total Dissolved	mg L^{-1}	0.012	0.005	0.026	0.003
	Total	mg L^{-1}	0.036	0.007	0.056	0.025
Nitrogen	Particulate	mg L^{-1}	0.048	0.020	0.090	<0.010
	Total Dissolved	mg L^{-1}	0.249	0.047	0.436	0.190
	NO ₂ -NO ₃	mg L^{-1}	0.022	0.038	0.210	<0.010
	Total NH ₃	mg L^{-1}	0.015	0.011	0.050	<0.005
	Total	mg L^{-1}	0.30	0.04	0.48	0.23
Carbon	DOC	mg L^{-1}	6.49	0.55	7.60	5.58
	POC	mg L^{-1}	0.457	0.163	0.940	0.224
	TOC	mg L^{-1}	6.95	0.64	8.54	6.02
	DIC	mg L^{-1}	12.4	1.2	14.4	6.7
Chlorophyll-a		mg L^{-1}	0.002	0.001	0.005	<0.001
Ca	Dissolved	mg L^{-1}	14.6	1.0	17.7	12.9
Mg	Dissolved	mg L^{-1}	4.0	0.2	4.6	3.6
K	Dissolved	mg L^{-1}	1.05	0.07	1.28	0.95
Na	Dissolved	mg L^{-1}	2.7	0.3	3.4	2.2

Table 19. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L ⁻¹	2.01	0.71	3.32	0.78
Cl	Dissolved	mg L ⁻¹	1.2	0.2	1.8	0.8
F	Dissolved	mg L ⁻¹	0.09	0.01	0.14	0.07
SO ₄	Dissolved	mg L ⁻¹	2.6	0.7	5.1	1.7
Al	Extractable	mg L ⁻¹	0.491 ⁺⁺⁺	0.154	0.826 ⁺⁺⁺	0.230 ⁺⁺⁺
	Dissolved	mg L ⁻¹	<0.100	0.043	0.179	<0.100
As	Dissolved	mg L ⁻¹	0.0002	0.0001	0.0003	<0.0001
Ba	Total	mg L ⁻¹	<0.08		<0.08	<0.08
Cd	Total	mg L ⁻¹	<0.0001	0.0001	0.0002	<0.0001
	Dissolved	mg L ⁻¹	0.0001	0.0001	0.0003	<0.0001
Co	Total	mg L ⁻¹	<0.0005	0.0002	0.0011	<0.0005
Cu	Total	mg L ⁻¹	0.0040 ⁺⁺⁺	0.0051	0.0240 ⁺⁺⁺	0.0009
	Dissolved	mg L ⁻¹	0.0014	0.0006	0.0034	0.0008
Fe	Extractable	mg L ⁻¹	0.624 ^{+,+++}	0.189	0.931 ^{+,+++}	0.256
	Dissolved	mg L ⁻¹	0.076	0.041	0.156	0.014
Pb	Total	mg L ⁻¹	0.0009	0.0006	0.0027 ⁺⁺⁺	<0.0007
	Dissolved	mg L ⁻¹	0.0012	0.0016	0.0057	<0.0007
Mn	Extractable	mg L ⁻¹	0.021	0.005	0.030	0.005
	Dissolved	mg L ⁻¹	0.004	0.003	0.012	<0.002
Ni	Total	mg L ⁻¹	0.0018	0.0014	0.0089	0.0007
	Dissolved	mg L ⁻¹	0.0007	0.0004	0.0015	<0.0005
Se	Dissolved	mg L ⁻¹	0.0001	0.0001	0.0003	<0.0001
V	Total	mg L ⁻¹	0.0013	0.0005	0.0025	<0.0005
	Dissolved	mg L ⁻¹	<0.0005	0.0002	0.0007	<0.0005
Zn	Total	mg L ⁻¹	0.0035	0.0030	0.0172	0.0012
	Dissolved	mg L ⁻¹	0.0012	0.0013	0.0051	<0.0003
Discharge	Daily	m ³ sec ⁻¹	874	102	1080	718

* Exceeds limit for drinking water.

** Exceeds limit for recreational use.

+++ Exceeds limit for protection of aquatic life.

higher. The maximum total Pb levels measured at this site also exceeded the aquatic life guideline while the mean concentration was within 10% of the guideline level.

3.4.2 Water Quality - Water Discharge Relationships

3.4.2.1 Pre-development, 1967-1974

Most correlations between water quality and discharge were negative during the pre-development period. The negative correlates included conductivity (lab), total alkalinity, total hardness, dissolved Na, and reactive Si (Figure 26). The only positive correlate was turbidity (lab). Most of the regressions on discharge accounted for 40 to 50% of the variance in the dependent variables, although the 22% of the variance in reactive Si was attributable to discharge.

3.4.2.2 Diversion Construction, 1974-1976

All significant correlations were negative during the construction period (Figure 27), during which Rat River flows at the outlet of Notigi Lake were completely cut off. All of the discharge-correlated variables in this period (conductivity (lab), total alkalinity, total hardness, and dissolved Na) were common to the pre-development period as well. The y-intercepts of these relationships varied little between the two periods but the slopes were generally steeper during the construction period. The goodness of fit of the correlations for the construction period were as high or higher than prior to development, with the regressions

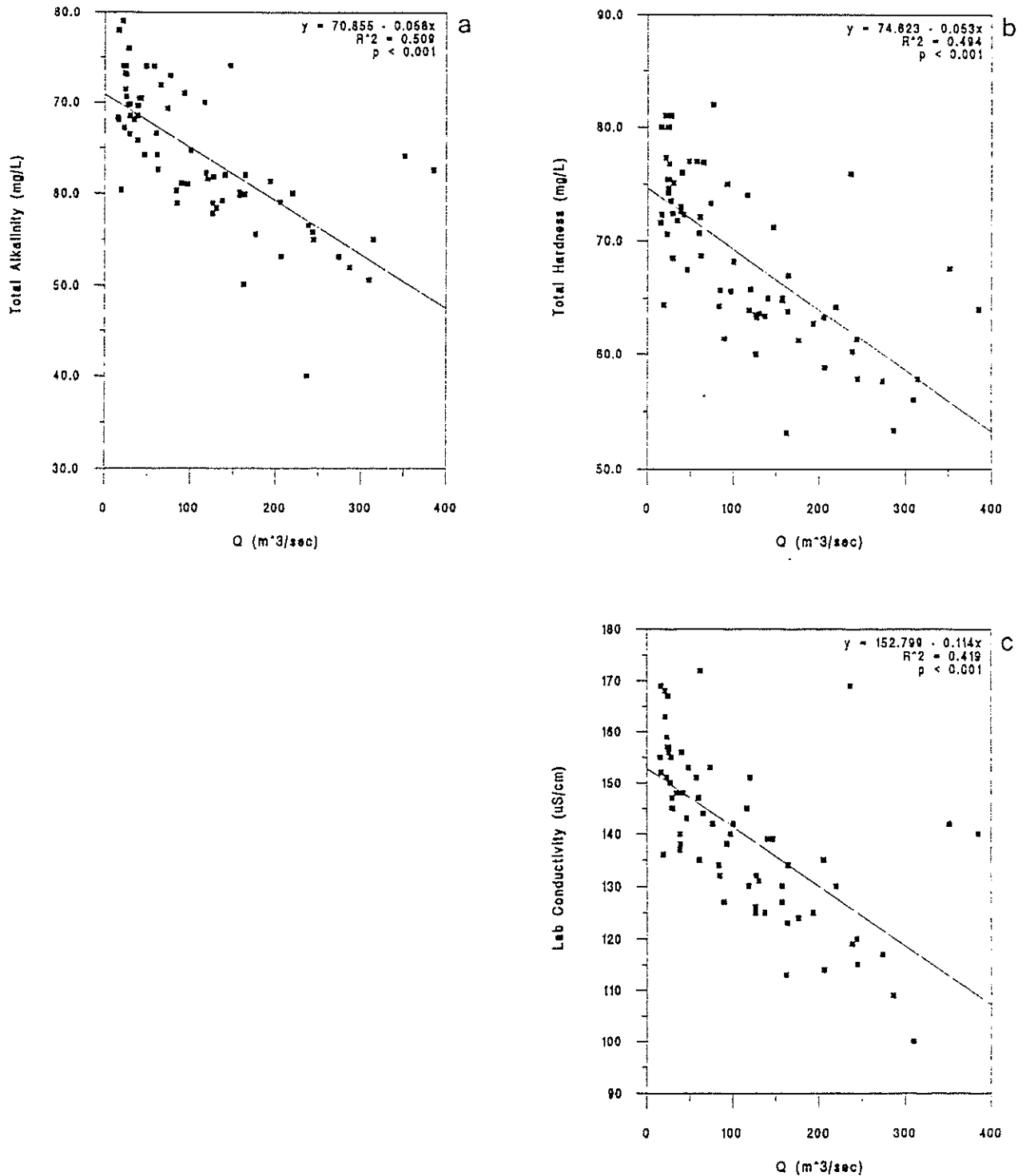


Figure 26. Water chemistry - water discharge relationships in the Burntwood River at Thompson for the pre-development period; from February 1967 to May 1974. a. total alkalinity b. total hardness c. lab conductivity d. lab turbidity e. dissolved Na f. reactive Si.

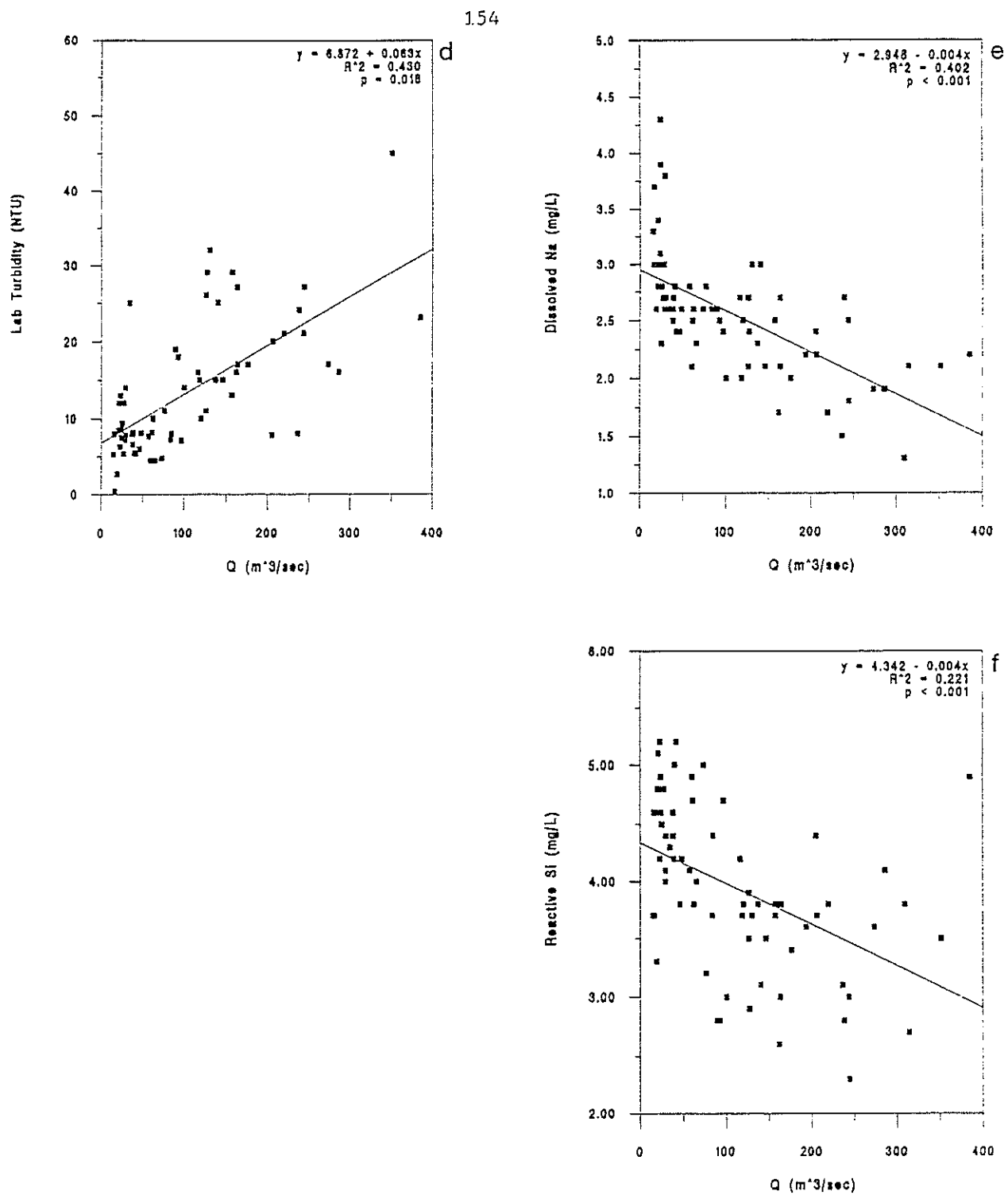


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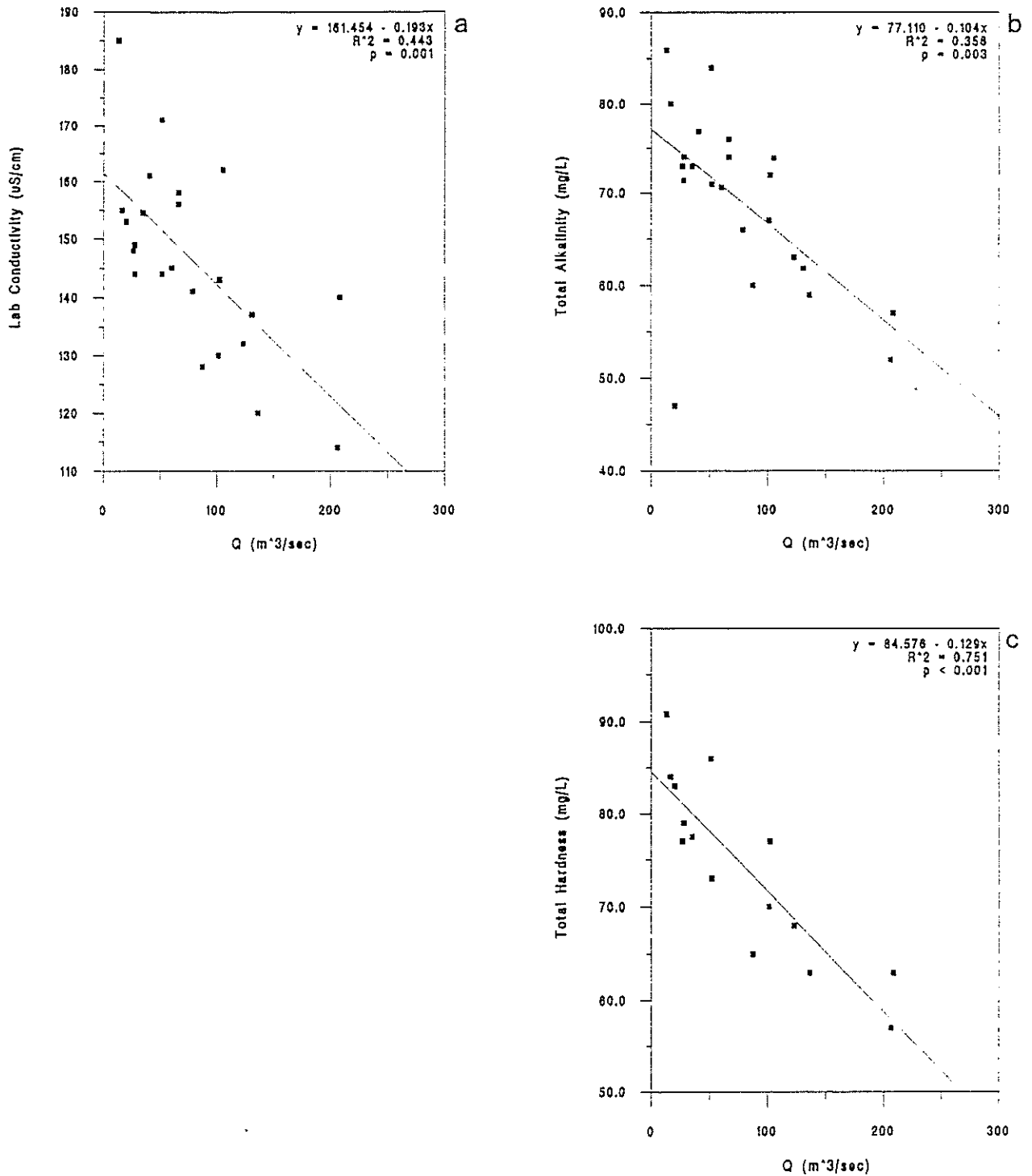


Figure 27. Water chemistry - water discharge relationships in the Burntwood River at Thompson for the construction period; from June 1974 to August 1976. a. lab conductivity b. total alkalinity c. total hardness d. dissolved Mg e. dissolved Na.

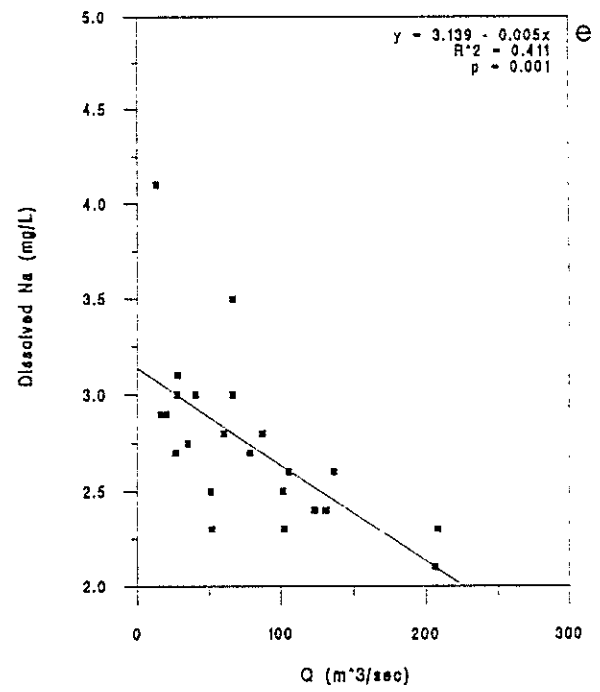
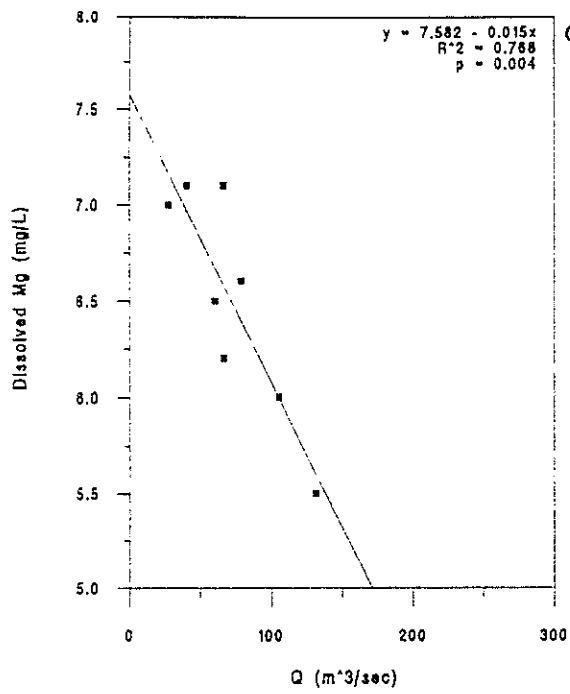


Figure 27. Continued.

accounting for 36 to 77% of the variance. Turbidity and reactive Si, which were correlated with discharge in the pre-development period, were not correlated with discharge during the construction period.

Dissolved Mg was not measured in the pre-development period. Its correlation with discharge is consistent with that of hardness, of which Mg is a major component (APHA 1985).

3.4.2.3 Immediate Post-diversion, 1976-1978

Again, all significant correlations were negative during this period (Figure 28). The correlates included total alkalinity and total hardness, as in both the pre-development and construction periods, reactive Si, as in the pre-development period, and pH. Turbidity, conductivity, and dissolved Na and Mg were not correlated with discharge in this period. The slopes of the alkalinity, hardness, and reactive Si relationships were substantially lower than in the preceding periods, although the y-intercepts remained similar. All four regressions accounted for 30 to 40% of the variance in the dependent variables, similar to those in the pre-development period.

3.4.2.4 Current, 1987-1989

All significant correlations with discharge were positive at this site (Figure 29), as found upstream at the outlet of Threepoint Lake. Correlates included turbidity (lab), NFR (total and fixed), alkalinity (total and bicarbonate), particulate and

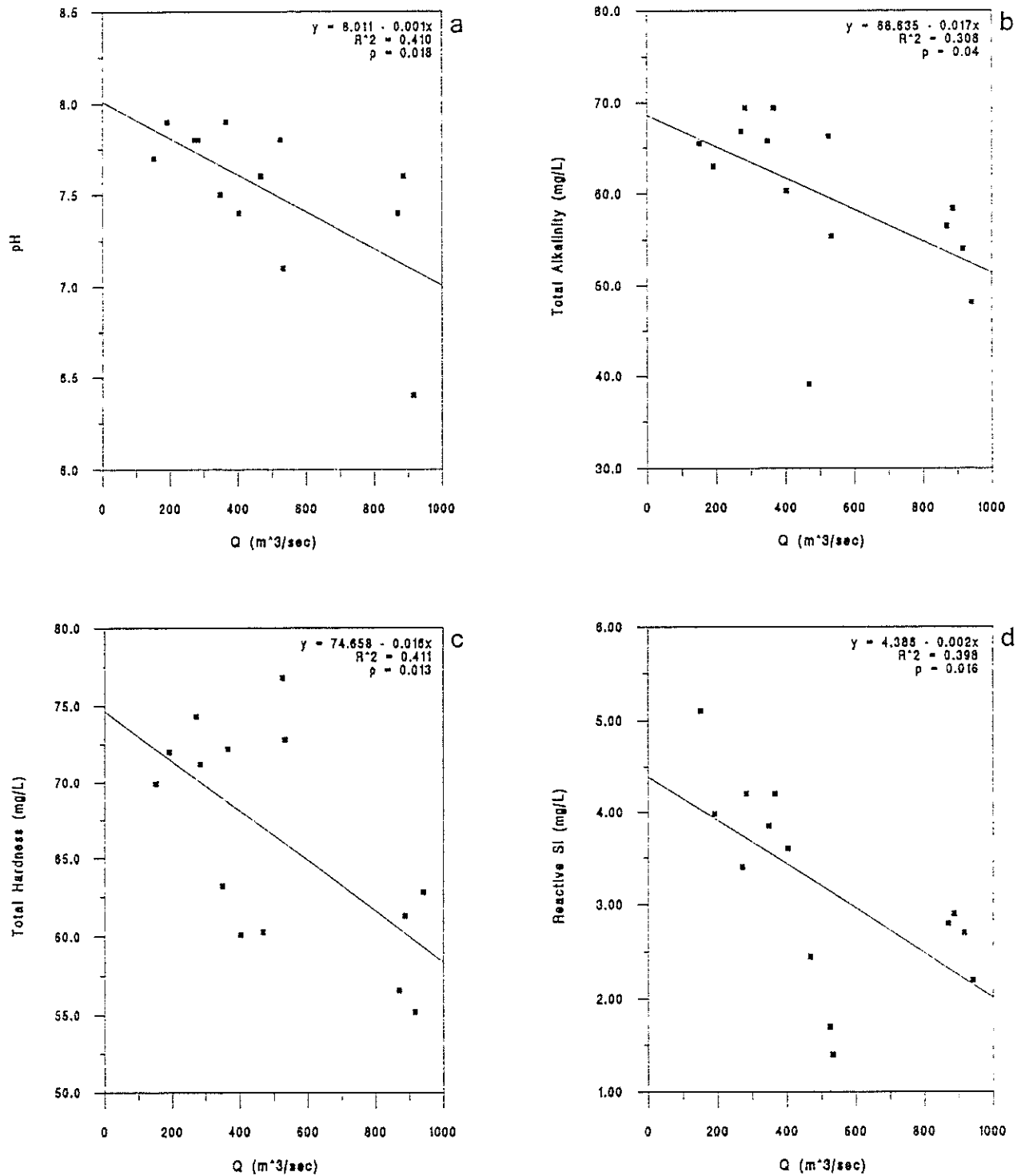


Figure 28. Water chemistry - water discharge relationships in the Burntwood River at Thompson for the immediate post-diversion period; from September 1976 to December 1977. a. pH b. total alkalinity c. total hardness d. reactive Si.

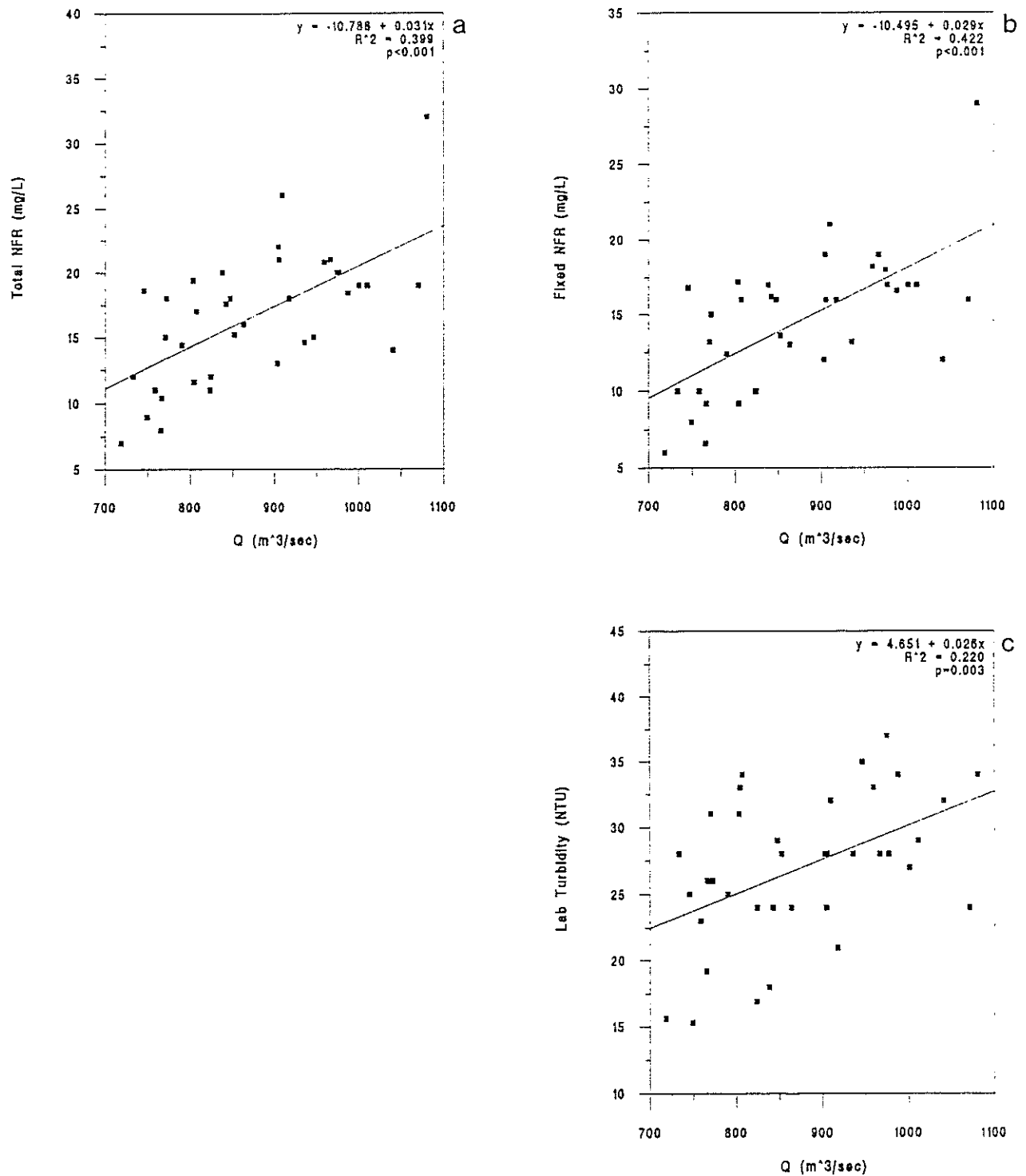


Figure 29.

Water chemistry - water discharge relationships in the Burntwood River at Thompson, 1987-1989. a. total NFR b. fixed NFR c. lab turbidity d. total alkalinity e. bicarbonate alkalinity f. particulate P g. total P h. dissolved K i. dissolved Na j. total Cu k. extractable Fe l. extractable Mn m. dissolved Se n. total Zn.

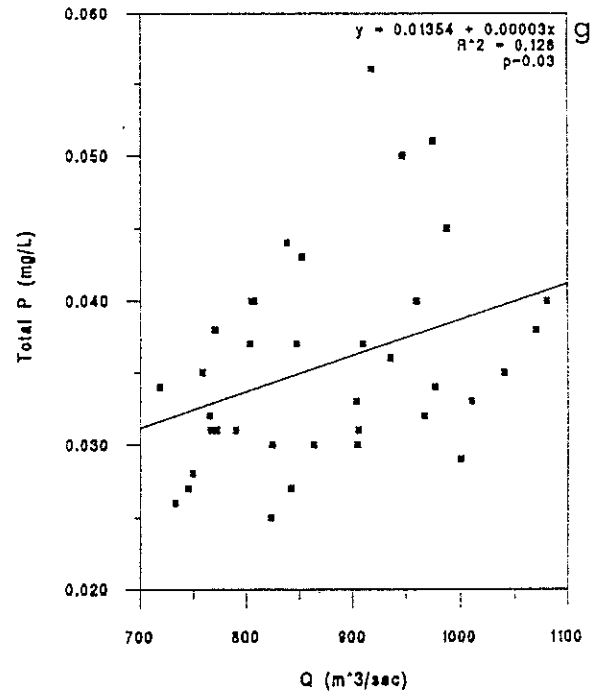
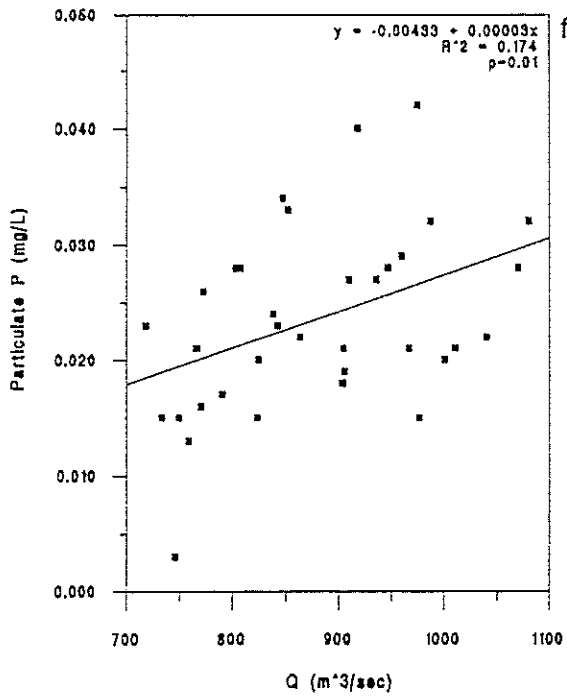
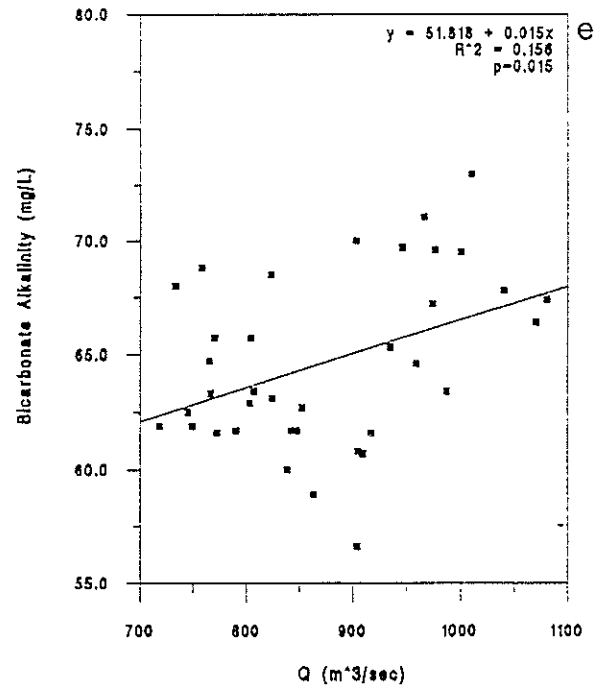
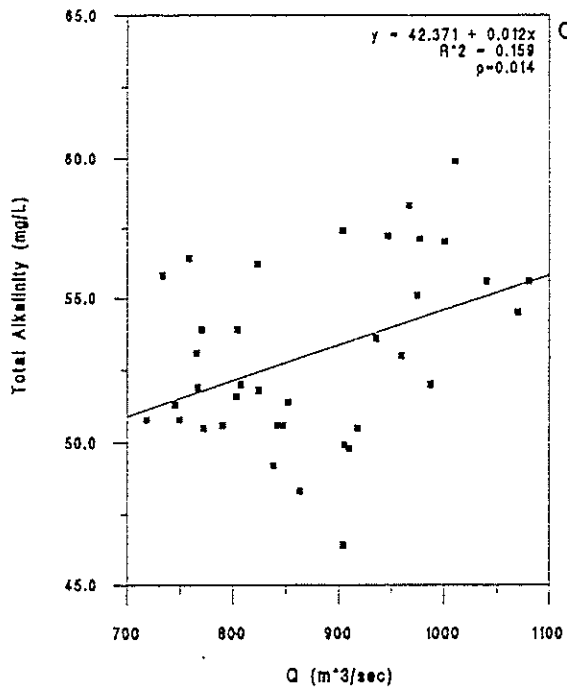


Figure 29. Continued.

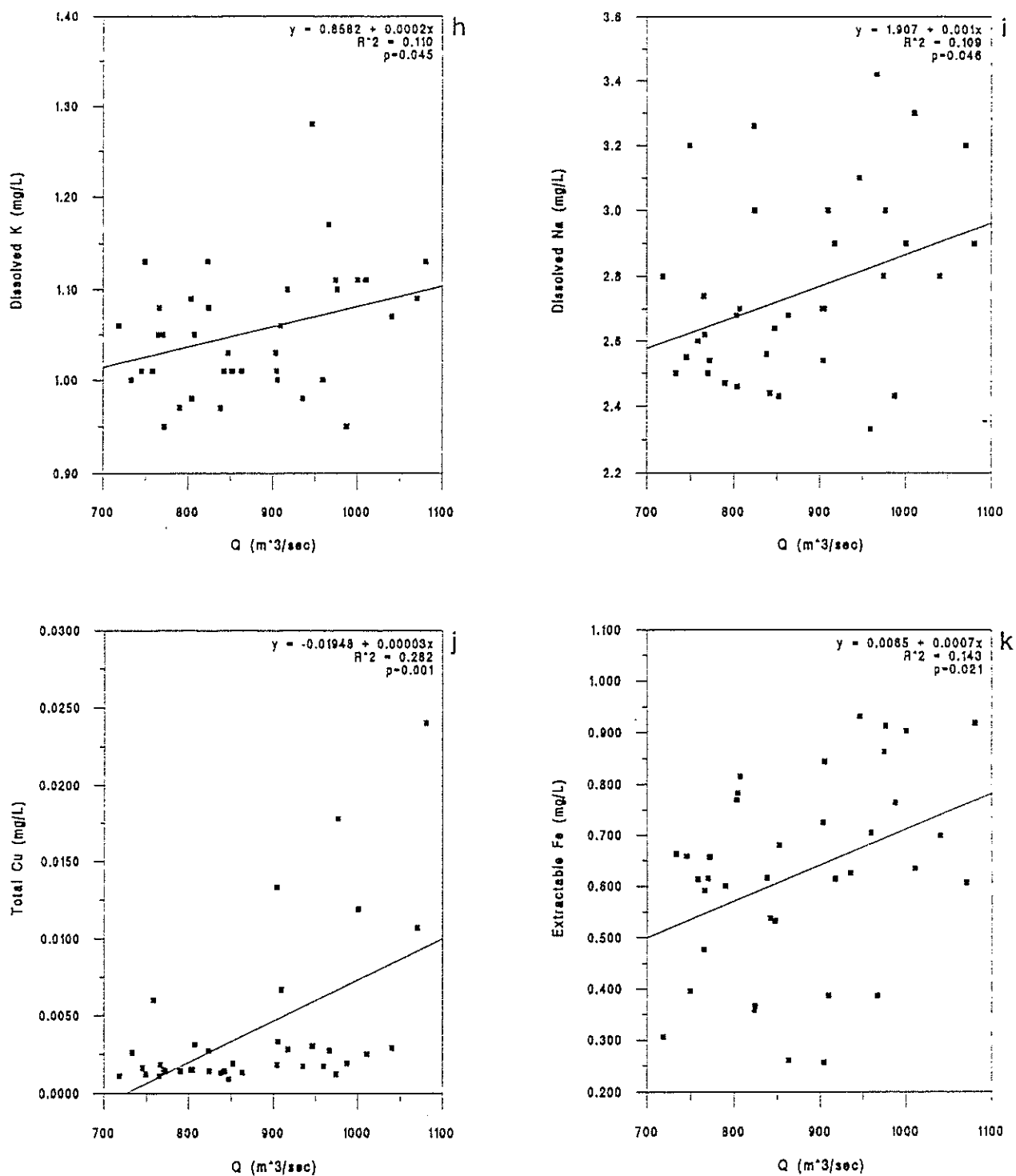


Figure 29. Continued.

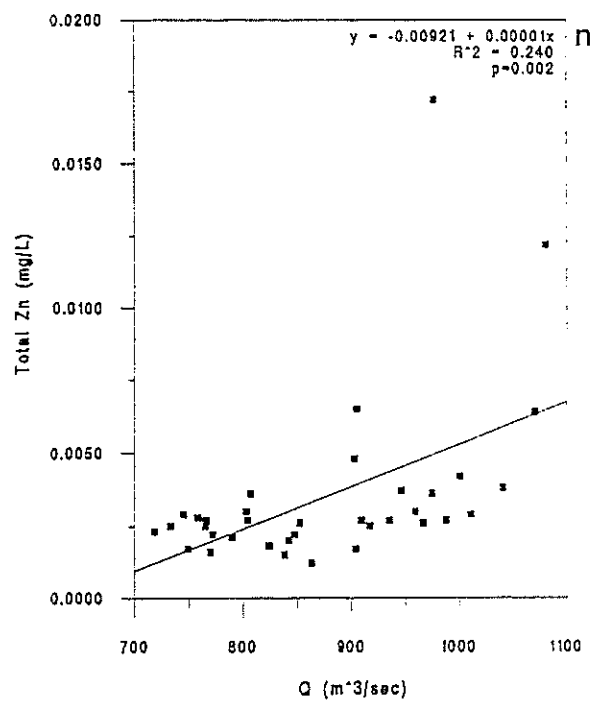
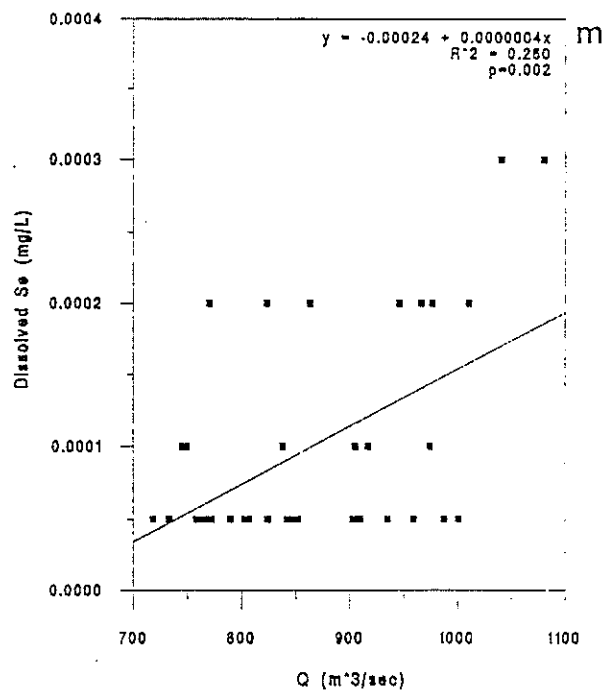
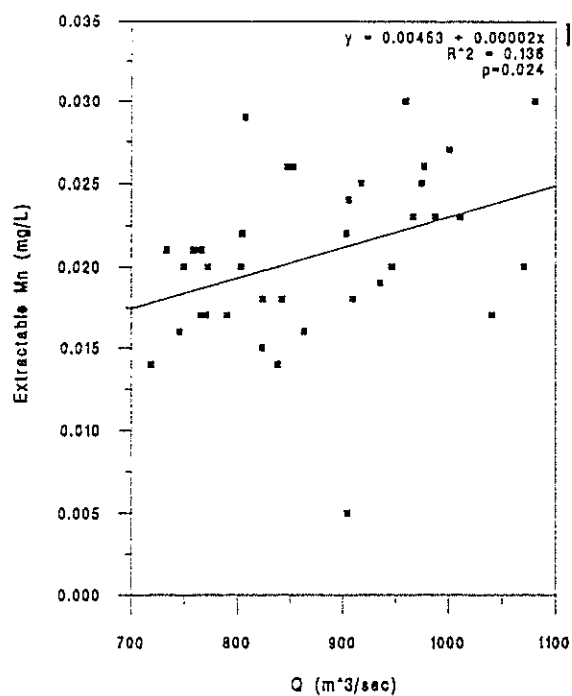


Figure 29. Continued.

total P, dissolved K and Na, total Cu and Zn, extractable Fe and Mn, and dissolved Se.

The shift in the slopes of the alkalinity and Na relationships from negative to positive indicates an important change in the factors regulating concentrations of these parameters. Increasing flows generally had a diluting effect prior to diversion. The current positive correlations indicate an accelerating rate of supply with increasing discharge. Increasing rates of bank and shoreline erosion with increasing discharge are the most probable source of these materials.

The occurrence of positive correlations with discharge for NFR and turbidity in the Burntwood River at Thompson but not upstream at the outlet of Threepoint Lake is indicative of shoreline erosion occurring between these points. Erosion hotspots have been identified on the southern shore of Wuskwatim Lake and along the Burntwood River between Gods and Early Morning rapids (Kellerhals 1988).

Few of these significant correlations shows much promise for the prediction of impacts which might result from a change in the operation regime of the diversion or for management of impacts related to the current operation regime. Although more variables were correlated with discharge during the FEMP study, the relationships generally were much weaker than in the preceding periods. Regression on discharge accounted for 20% or less of the variance in alkalinity (total and bicarbonate), P (particulate and total), dissolved K and Na, and extractable Fe and Mn. The general

occurrence of weaker correlations following diversion is probably a result of the smaller range of fluctuation in discharge, relative to the mean discharge, than during the pre-development period. The coefficient of variation (c.v.) of daily discharge was 122% of the mean for the period between 1967 and 1978, compared with a c.v. of just 12% of the mean during the 1987-1989 FEMP study period.

It is unlikely that any of the discharge relationships for the FEMP study period will be of use for predictive purposes due to the substantial portion of total variance (80-90%) that remains unaccounted for. Although 21-30% of the variance in total Cu and Zn and dissolved Se was accounted for by regression on discharge, the utility of these relationships is questionable because the correlations were driven by a few points rather than representing a general linear trend in the data. Only the discharge relationships for NFR (total and fixed) and turbidity appear to have predictive value as they account for 22-42% of the variance in the dependent variables and the linear model represents the general trend in the data.

3.4.3 Seasonal Differences

As was the case at many other sites, chlorophyll a concentrations were generally lower under ice-cover than during the open water season (Figure 30). This probably reflects the reduced light penetration through the snow and ice. There were no other differences in water quality between the open water and ice-cover seasons during the 1987-1989 FEMP study. No other variables

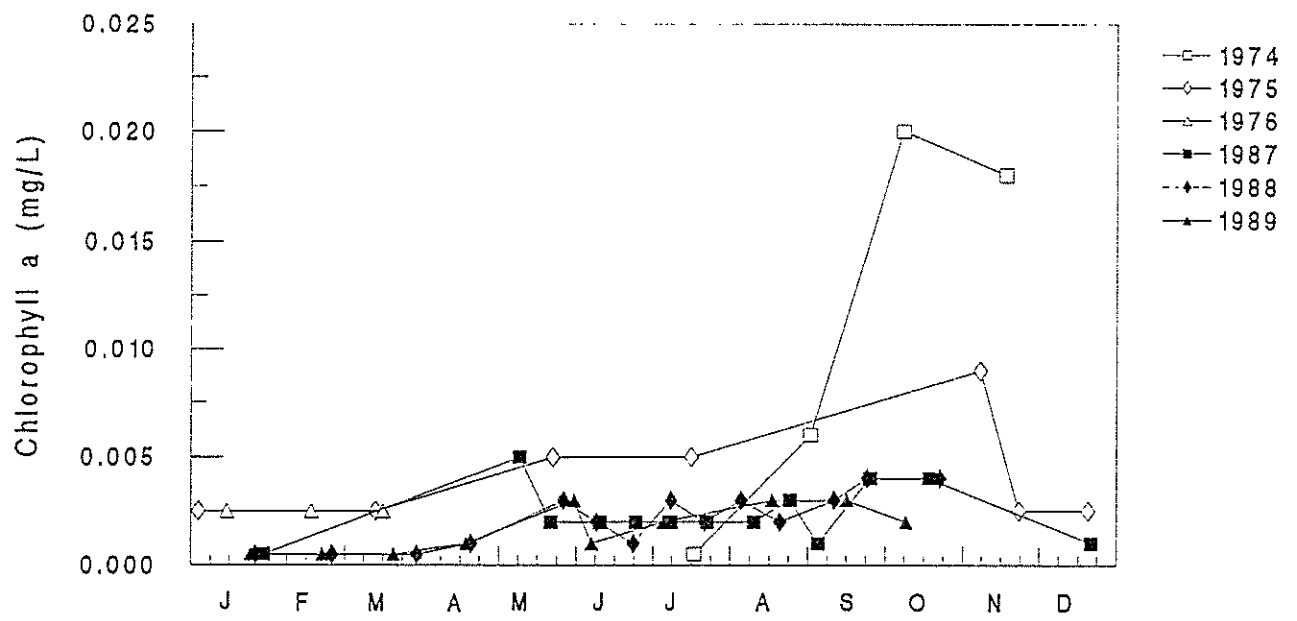


Figure 30. Seasonal trends in chlorophyll a concentrations in the Burntwood River at Thompson.

followed any definable seasonal pattern. This trend of similar water quality in summer and winter was prevalent during the pre-development and construction periods as well.

3.4.4 Changes in Water Quality Since Churchill River Diversion

Significant changes associated with the construction period included a 64% increase in turbidity and a 12% increase in hardness (Table 20). Turbidity remained at or near this elevated level after the diversion began operating. Conversely, hardness declined to the pre-development level in the immediate post-diversion period, with a further decline occurring after 1978. Hardness was 50% lower in the 1987-1989 period than before diversion.

The only significant change between the pre-development and immediate post-diversion periods was a 230% increase in total P (Table 20). This change coincided with the start of CRD operation, as there was no difference in total P concentration between the pre-development and construction periods. Marginally significant changes between these periods included a 19% increase in TOC and a 29% increase in turbidity.

Considerably more changes in water quality become evident when the FEMP study results are compared with the pre-development data. Turbidity was 94% higher during the FEMP study, while conductivity, alkalinity, and hardness were about 20% lower; dissolved Ca and SO₄ were 32 to 34% lower; TOC, reactive Si, and dissolved Fe were 45 to 49% lower; and dissolved NO₂NO₃ was 84% lower (Table 20). The decrease in conductivity was first evident between the construction

Table 20. Evaluation of the significance of changes in selected water quality parameters in the Burntwood River at Thompson following Churchill River diversion. Values listed are the mean and standard deviation (s.d.) for the respective period and the p value from the Tukey/HSD pairwise comparison. Pre denotes the pre-development period from 1967 to May 1974. Const denotes the construction period between June 1974 and August 1976. Post is the immediate post-diversion period from September 1976 to December 1977. Current represents the FEMP study period between January 1987 and October 1989.

pH (Lab)				Conductivity (Lab)				Turbidity (Lab)			
	Mean	s.d.	Pre	Const	Post		Mean	s.d.	Pre	Const	Post
Pre	7.66	0.15				Pre	141.0 ¹	7	Pre	14.0 ¹	2
Const	7.67	0.06	1.000			Const	148.0 ¹	6	Const	23.0	1
Post	7.45 ¹	0.07	0.310	0.109		Post	125.0	25	Post	18.0	<1
Current	7.70	0.10	0.981	0.979	0.120	Current	115.0	2	Current	27.0 ²	2
							0.031	0.013		<0.001	0.249
							0.459			0.001	0.001
Alkalinity (Total)				Hardness (Total)				NTR (Lab)			
	Mean	s.d.	Pre	Const	Post		Mean	s.d.	Pre	Const	Post
Pre	65.0 ¹	3				Pre	69.0 ¹	3	Pre	11.0	10
Const	70.0 ¹	4	0.205			Const	77.0 ¹	4	Const	8.0	6
Post	60.0 ¹	2	0.132	0.004		Post	66.0 ¹	<1	Post	13.0	4
Current	53.0 ¹	1	0.003	<0.001	0.034	Current	53.0	<1	Current	16.0 ¹	1
							<0.001			0.890	0.946
							<0.001			0.438	
TOC				Diss NO ₃ NO ₂				Total P			
	Mean	s.d.	Pre	Const	Post		Mean	s.d.	Pre	Const	Post
Pre	12.63	1.11				Pre	0.140	0.159	Pre	0.028	0.009
Const	14.00	1.00	0.335			Const	i.d.		Const	0.039	0.010
Post	15.00	1.41	0.074	0.950		Post	i.d.		Post	0.065	0.007
Current	6.94	0.33	0.001	<0.001	<0.001	Current	0.022	0.006	Current	0.036 ²	0.003
							0.029			0.565	0.970
										0.016	0.016

Table 20. continued.

Reactive Si						Diss Na						Diss K					
	Mean	s.d.	Pre	Const	Post		Mean	s.d.	Pre	Const	Post		Mean	s.d.	Pre	Const	Post
Pre	3.94 ¹	0.25				Pre	2.5 ¹	0.2				Pre	1.00	0.12			
Const	4.10	0.76	0.973			Const	2.8 ¹	0.2	0.095			Const	1.06	0.10	0.928		
Post	3.25 ¹	0.50	0.117	0.062		Post	2.5	<0.1	1.000	0.081		Post	1.25	0.35	0.111	0.408	
Current	2.01	0.12	0.001	0.001	0.005	Current	2.7 ¹	0.1	0.349	0.948	0.313	Current	1.06 ¹	0.02	0.959	1.000	0.484
Diss Mg						Diss Ca						Diss SO ₄					
	Mean	s.d.	Pre	Const	Post		Mean	s.d.	Pre	Const	Post		Mean	s.d.	Pre	Const	Post
Pre	i.d.					Pre	21.3	2.7				Pre	4.1	0.2			
Const	6.5 ¹	0.1				Const	21.5	0.8	0.996			Const	4.0	0.3	0.995		
Post	i.d.					Post	i.d.					Post	4.0	0.6	0.996	1.000	
Current	4.0	0.1	<0.001			Current	14.6	0.1	0.009	0.011		Current	2.7	0.3	0.007	0.009	0.017
Diss Cl						Diss F						Extr Fe					
	Mean	s.d.	Pre	Const	Post		Mean	s.d.	Pre	Const	Post		Mean	s.d.	Pre	Const	Post
Pre	1.2	0.2				Pre	i.d.					Pre	i.d.				
Const	2.7	2.2	0.279			Const	0.06	i.d.				Const	0.813	0.050			
Post	i.d.					Post	0.12	0.07				Post	0.786	0.008		0.826	
Current	1.2	0.1	0.983	0.177		Current	0.09	0.01			0.291	Current	0.628 ¹	0.078	0.096	0.096	0.046

Table 20. continued.

Diss Fe	Extr Mn					Extr Al				
	Mean	s.d.	Pre	Const	Post	Mean	s.d.	Pre	Const	Post
Pre	0.138	0.027				Pre	i.d.			
Const	0.150	i.d.				Const	0.870	0.207		
Post	i.d.					Post	0.545	0.021	0.067	
Current	0.077	0.010	0.022			Current	0.488	0.095	0.029	0.744

1. Discharge-standardized variables.

and immediate post-development periods, declining further between December, 1977, and January, 1987.

Several variables were not routinely measured until the construction period. Among these, significant reductions in dissolved Mg, by 38%, and in extractable Mn and Al, by 44%, occurred between the 1974-1976 construction period and the 1987-1989 FEMP study (Table 20).

The increase in turbidity during the construction period was not simply a result of the reduced flows. Based on the turbidity-discharge relationship for the pre-development period (Figure 26d), the 30% decrease in flow should have produced a 16% decline in turbidity rather than the observed 64% increase. Although not accounted for by the reduced flow volume, the increase in turbidity was indirectly related to the flow reduction which was caused by the damming of the Rat River at the outlet to Notigi Lake. During the pre-development period, Rat River turbidity was about 50% of that in the Burntwood River above Threepoint Lake (Tables 10 and 16) so that removal of this dilution source would be expected to increase turbidity in the Burntwood River below the confluence with the Rat River. However, loss of Rat River flows during the construction period accounts for an increase in turbidity of only 15% (i.e., loss of 30% of total flow with turbidity 50% of that in Burntwood R.; $0.3 \times 0.5 = 0.15$) leaving most of the increase unaccounted for. Possible causes include some form of construction activity, such as the stabilization of river banks in the vicinity of Thompson, dike construction upstream near Manasan Falls, or the

instream work in construction of the Notigi Dam, as well as a natural increase in turbidity in the Burntwood River above Threepoint Lake. None of these causes can be verified due to the lack of data for the Rat River between Notigi and Thompson and the Burntwood River between Threepoint Lake and Thompson during the construction period.

The persistence of elevated turbidity at Thompson beyond the construction period was partly due to the introduction of more turbid water from the Rat River but the primary cause was bank and shoreline erosion below Threepoint Lake. Turbidity in the Rat River above Threepoint Lake was 250% higher during the FEMP study than before diversion, but this was just sufficient to raise turbidity to the natural level in the Burntwood River above Threepoint Lake (Table 10). This increase in Rat River turbidity was probably due to shoreline erosion in the southern region of Notigi Reservoir and to bank and shoreline erosion which may be occurring on the river between the Notigi Dam and Threepoint Lake. Despite the increased turbidity in the Rat River, turbidity at both sites was still 60% lower than at Thompson, indicating elevated turbidity in the Rat River is not the primary cause of the elevated turbidity at Thompson. None of this additional turbidity was generated within Threepoint Lake, as indicated by the similar values in the Rat and Burntwood rivers above Threepoint Lake and in the Burntwood River below Threepoint Lake (Tables 6, 15, and 18). Erosion along the Burntwood River between the lake outlet and Thompson must be responsible. Potential sources include documented erosion sites on

the south shore of Wuskwatim Lake and on the Burntwood River between Early Morning Rapids and Gods Rapids (Kellerhals 1988).

The elevation in hardness during construction is directly attributable to the reduction in river flows during this period. The discharge-hardness relationship for the construction period (Figure 27c) predicts a mean hardness of 70 mg L^{-1} at the mean discharge for the pre-development period (1967-1973; $126 \text{ m}^3 \text{ sec}^{-1}$; Water Survey of Canada 1990). This compares favourably to the actual pre-development mean hardness of 69 mg L^{-1} (Table 20).

The declines in conductivity, alkalinity, hardness, dissolved Ca, TOC, and reactive Si are a direct reflection of the lower values of these variables in the Churchill River at Leaf Rapids (Table 17), indicating diversion was the cause. The apparent decline in Mg between the construction and current study periods also is explained by a lower concentration in the Churchill River. The annual mean Mg concentration at Opachuanau Rapids, on the Churchill River at the inlet to Southern Indian Lake, was 2.72 mg L^{-1} in 1972 (Cleugh 1974), compared with 6.5 mg L^{-1} in the Burntwood River during the construction period (Table 20). The decline in SO_4 was of the same magnitude as the declines in the natural rivers and, therefore, was attributed to a change in methodology. It clearly is not due to diversion. The decrease in TOC was the product of lower TOC concentrations in the Churchill River and of additional losses which occur as the water flows through SIL and Notigi Reservoir (Section 3.3.1.4).

In addition to the present study, the effect of CRD on water quality in the Burntwood River at Thompson has been examined in some detail in two previous studies. Playle and Williamson (1986) evaluated the significance of changes in 16 water quality parameters between the pre-diversion period, represented by data from January 1972 through June 1976, and the post-diversion period which included data from July 1976 to March, 1984. Guilbault et al. (1979) examined changes in four water quality variables (total P, dissolved Ca and K, extractable Al) related to CRD. They considered data from the beginning of the NAQUADAT data record in 1967 (or later, depending on the variable) through 1977. The division point between their pre- and post-diversion periods was not defined in their report.

Playle and Williamson (1986) identified significant changes in 7 variables following Churchill River diversion. As in the present study, they found significant decreases in conductivity, by 21%, and in alkalinity, Ca, and Mg by 15-17%. The magnitudes of the changes in conductivity and alkalinity are about the same as found in this study, while their reported decreases in Ca and Mg were about half those reported here. They also found a significant increase in total P, by 66%, and a small but significant 9% decrease in TIC. In contrast to the present study, Playle and Williamson (1986) reported a significant increase in NFR, by 73%, and no change in turbidity or hardness.

The discrepancies between the studies can be explained by differences in the periods over which changes were examined and the

way in which the data records were divided. Playle and Williamson (1986) included the construction period (June 1974 through August 1976) as baseline data and did not consider data prior to 1972. Thus the construction period, in which a significant increase in turbidity occurred, represented about half their pre-development data record, obscuring any change (Figure 31b). This illustrates the importance of stratifying the data as much as possible in the initial analysis to consider the effects of all phases of a development. Data from different phases can always be pooled after it has been determined there are no significant differences.

The finding by Playle and Williamson (1986) of a significant increase in NFR following diversion appears to be a result of the restriction of their database to measurements from 1972 and later. When the entire pre-development data record for 1968 through May 1974 is included in the analysis, the difference disappears. NFR concentrations generally tended to be lower and were much less variable from 1972 to 1976 than between 1968 and 1972 (Figure 31a). There does not appear to be any justification for the exclusion of the pre-1972 data on the basis of sampling or analytical incompatibility. It must be concluded there has been no detectable change in annual mean NFR related to CRD.

Notwithstanding the lack of a significant change in mean NFR, it appears from Figure 31a that the minimum NFR has increased since diversion. This is most apparent in the measurements between 1987 and 1989. During this period, NFR never was lower than 7 mg L⁻¹ while 19 of the 45 (42%) measurements made before September, 1976,

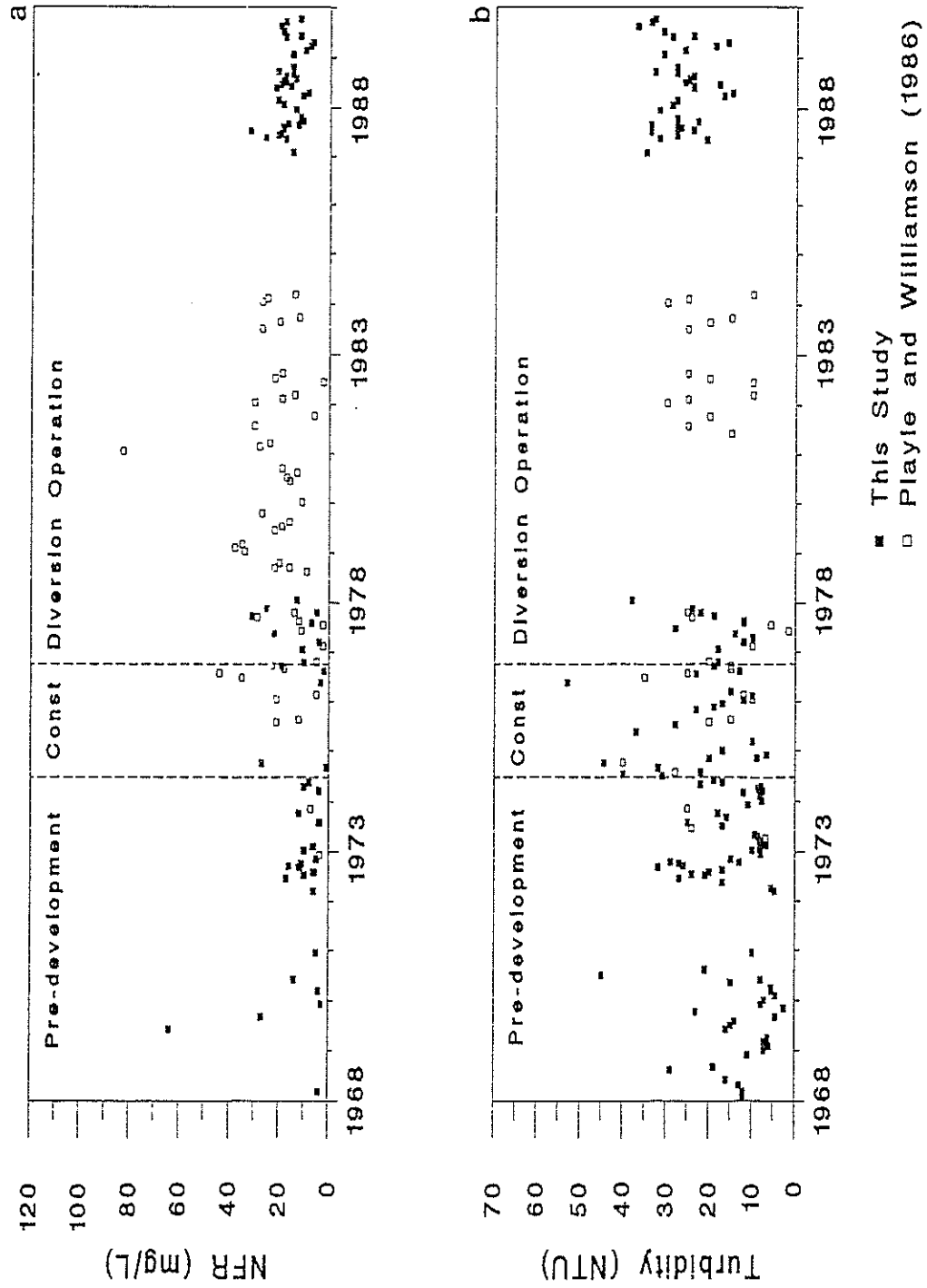


Figure 31. Temporal trends in (a) NFR, (b) turbidity, (c) alkalinity, (d) hardness, and (e) total P in the Burntwood River at Thompson based on data analyzed by Playle and Williamson (1986) and data analyzed in this study.

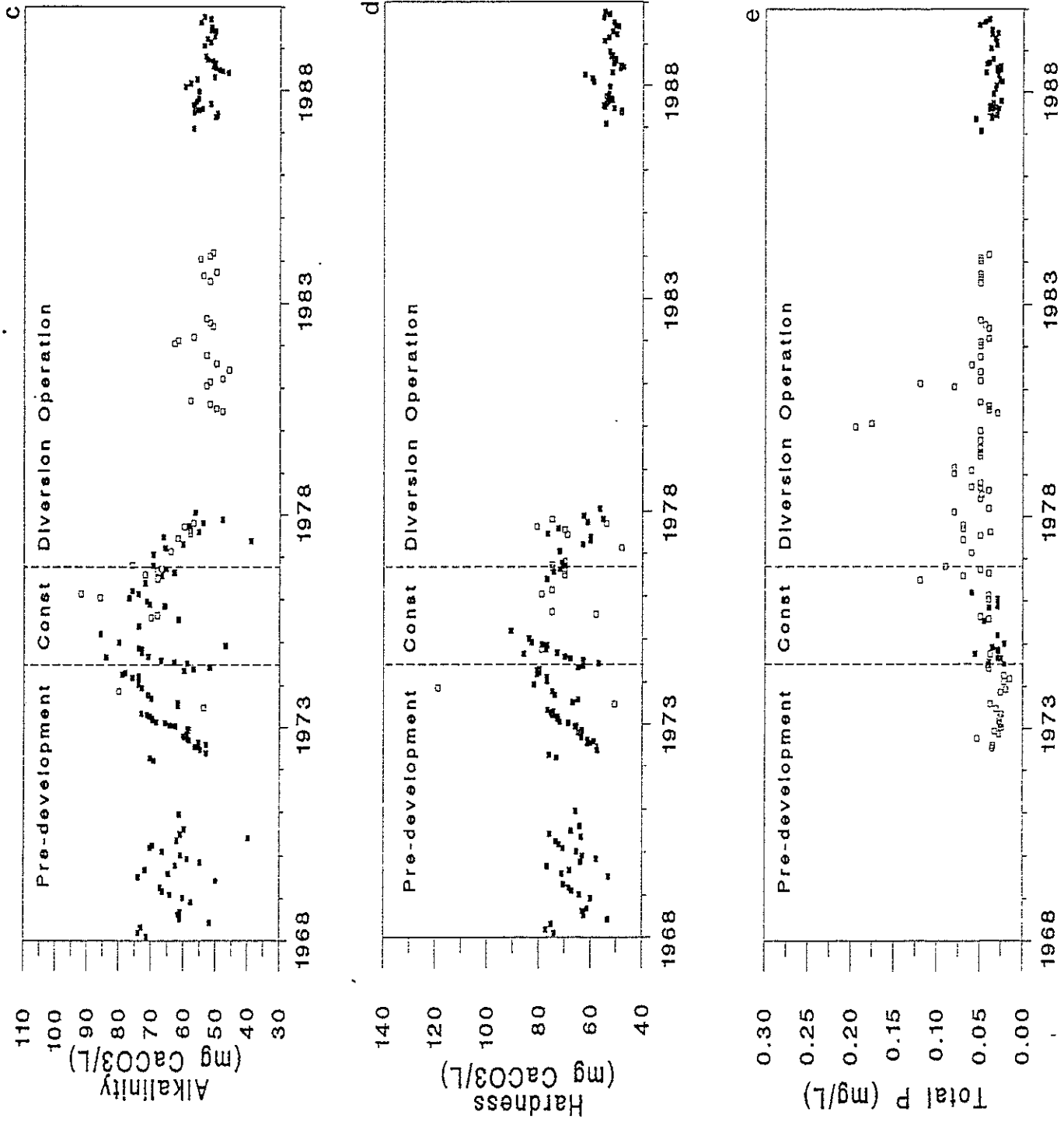


Figure 31. Continued.

■ This Study
 □ Playle and Williamson (1986)

were between 1 and 7 mg L⁻¹.

Playle and Williamson (1986) identified the paradox of elevated NFR in the absence of a change in turbidity. Although it now appears that it is turbidity which increased significantly in the absence of a change in NFR, the paradox remains unresolved. One explanation for this apparent contradiction is the extreme variability in NFR compared to turbidity, particularly in the pre-development period (Figure 31 a and b). The detection of a significant difference therefore requires a smaller change in turbidity than in NFR. Both parameters may show the same general trend over time, as appears to be the case, but only one will show a significant increase. Turbidity may provide a better indication of the change in suspended sediment levels at this site due to CRD because of the more extensive pre-development data record for this variable. Over 100 measurements of turbidity were taken prior to the start of construction compared with 45 NFR measurements, therefore mean turbidity is less strongly influenced by occasional high values.

The lack of a significant change in hardness in the analysis by Playle and Williamson (1986) clearly was a result of the short data record available to them at the time of their study (Figure 31d). When the FEMP study data are added to this record, the decrease becomes apparent.

The increase in total P reported by Playle and Williamson (1986) appears to be a valid conclusion based on the analysis presented here (Table 20) and the data in Figure 31e. Similar

increases were observed in the back-flooded area of Stephens Lake (Hecky and Harper 1974) and at Southern Indian Lake and other sites on the CRD immediately after impoundment (R.E. Hecky, Department of Fisheries and Oceans, Winnipeg, MB, pers. comm.). The elevated total P levels probably were a result of the release of P from flooded soils and vegetation through leaching and decomposition. The flux of P from the flooded materials upstream may appear to have been relatively short-lived, peaking in about 1980 and returning to pre-development values some time between 1981 and 1987. The results of this study indicate total P concentrations in the 1987-1989 study period were not significantly different from the levels prior to diversion. However, the P flux may still be occurring, as indicated by higher total P concentrations in the Rat and Burntwood rivers than in the Churchill River. The end point of the process is not the pre-development concentration for the site, but is the level in the source of flow. A long-term impact of CRD will be a net decrease in the total P concentration in the Rat and Burntwood rivers. Based on the rate of decline in total P from the peak of 0.083 mg L^{-1} in 1980 to 0.036 mg L^{-1} during the 1987-1989 FEMP study, the annual mean concentration in the Burntwood River should reach that in the Churchill River (0.017 mg L^{-1}) within the next decade. This can only be confirmed by repeated monitoring at intervals of about 5 years.

Guilbault et al. (1979) reported a decrease in dissolved Ca, as found in this study and by Playle and Williamson (1986), as well as increases in dissolved K and extractable Al which were not

detected in the other studies. In the present study, a significant decrease in extractable Al was found between the construction and current periods using the same pre-diversion data set analyzed by Guilbault et al. (1979). This discrepancy appears to be an artefact of the previous study having compared dissolved Al in the pre-diversion period with post-diversion measurements of extractable Al. Even when measured in the same sample, the extractable Al concentration will be much higher than the dissolved Al concentration because the extractable value includes Al bound to particulates while the dissolved value only includes Al in solution. In the current study, the extractable Al concentration in the Burntwood River at Thompson was, on average, about 5 times higher than the dissolved Al concentration (Table 19). This confusion of parameters is indicated by the number of pre-development extractable Al measurements, 5, and the mean, 0.04 mg L^{-1} , stated by Guilbault et al. (1979). Our review of the database found 5 measurements of dissolved Al during the construction period, with a mean of 0.04 mg L^{-1} , compared with 12 extractable Al values with a mean of 0.870 mg L^{-1} . No extractable or dissolved Al measurements were made in the pre-development period. Not only is there no evidence to support an increase in extractable Al, there is no mechanism to explain such an increase. Between 1987 and 1989, extractable Al concentrations in the Rat and Burntwood rivers above Threepoint Lake were similar (Tables 6 and 15). A significantly higher concentration in the Rat River would be required to support the occurrence of a significant increase due to diversion.

Although a significant decline in extractable Al occurred between the construction and current study periods, it cannot be concluded that concentrations have declined from the natural level, in the absence of pre-development data for Al or any correlate. The upstream data collected in the FEMP study suggest a more reasonable conclusion might be that extractable Al levels increased during the construction period. Concentrations in the Burntwood and Rat rivers above Threepoint Lake between 1987 and 1989 were not only similar to each other, but were about the same as measured at Thompson as well (Tables 6, 15, and 19). The only way a lower concentration could have developed at Thompson is if the level in the Burntwood River above Threepoint Lake was higher during the pre-development period. Given the general stability of water chemistry in the upper Burntwood River (Section 3.1.4.1), this seems unlikely. Thus, it is the higher concentration during construction, rather than the lower current value, which appears to require explanation.

The finding of a significant increase in dissolved K following diversion by Guilbault et al. (1979) was most probably an artefact of their approach to the statistical analysis. The individual measurements over time were treated as replicates which may have violated the assumption of independence of observations required for valid application of Student's t-test (Snedecor and Cochran 1980). Time-series measurements are typically inter-dependent due to seasonal variations in environmental factors (Lettenmaier 1976). The use of dependent observations as replicates has the same effect as using the same measurement more than once, artificially

increasing the sample size and decreasing the variance. This dependence of observations can be avoided by calculating means for relatively long periods. Annual means of time-series measurements, as used in this study, generally are not inter-correlated and, therefore, can be used as replicates in parametric statistical analyses (Lettenmaier 1976). When the analysis is repeated on the same data set using annual means as replicates, no significant change is found (Table 20).

3.4.5 Effect of CRD on Heavy Metals

The erosion of inorganic sediments and associated heavy metals introduces the potential for the occurrence of elevated concentrations of metals in the water. This concern does not apply to all metals however, but only to those for which the eroding shoreline materials represent an important source. In the Burntwood River at Thompson, the eroding sediments appear to be a source of Fe, Mn, and Zn, as indicated by their positive correlations with river discharge. Despite these contributions through erosion, concentrations of Fe and Mn did not become elevated following CRD. Instead, concentrations of these metals were lower during the FEMP study than during the construction period (Table 20). This paradox is explained by 32-54% lower Fe and Mn concentrations in the Rat River (Table 15), which now dominates flows on the lower Burntwood River, than in the upper Burntwood River which dominated flows prior to diversion (Table 10). Concentrations of most metals are lower in the Rat River than in the Burntwood River (Tables 10 and

15), thus the CRD was largely self-mitigating with respect to the impact of erosion on the concentrations of heavy metals.

Although concentrations of metals have not increased due to CRD, it should be kept in mind that the mass flux of metals down the Burntwood River has increased substantially due to the large increase in river flow. It is not known if the increased loading will have an impact on metal availability at sites downstream, and particularly at sites of sediment deposition.

3.4.6 Adverse Effects

The only adverse effect of Churchill River diversion on physical and chemical water quality in the Burntwood River near Thompson has been the 100% increase in turbidity caused primarily by the erosion of glacial clays along the Burntwood River between Threepoint Lake and Thompson. The increase in turbidity exceeded the increase permitted under the Canadian Water Quality Guidelines for recreational water quality and aesthetics by more than a factor of 2. These guidelines allow for a 5.0 NTU increase in turbidity in waters such as the Burntwood River where turbidity is <50 NTU. Thus, the primary impact of CRD has been visual and physical, causing the water to appear "muddier" than it did prior to diversion and probably increasing the cost of turbidity reduction for those communities which use the Burntwood River for drinking water.

Although the safety of the water for human consumption has not been compromised by CRD, such an obvious visual change may have

caused water users to become concerned about the safety of their drinking water source. Over 10 years passed between the first increase in turbidity during construction of the diversion and the first detailed analysis of changes by Playle and Williamson (1986). This has left much time for local residents to wonder about, and develop their own opinions regarding, the nature and significance of changes which have occurred in their water source. Indeed, turbid water has been equated with poor quality water by some northern residents, while clear water is thought to be safe for drinking (Seidl et al. 1990). This study has shown the increased turbidity caused by CRD has not affected the quality of water in the Burntwood River for human consumption other than the increase in turbidity itself.

It could be argued that all of the other significant changes in water quality since CRD have been beneficial, actually improving the quality of water for domestic uses. The water is now slightly softer and has a lower major ion content although these improvements are small compensation for the degraded appearance of the river and the increased cost of drinking water treatment.

Algal standing crops and rates of photosynthesis in the Burntwood River below Thompson were severely light-limited prior to diversion (Hecky and Harper 1974). The increased turbidity due to diversion therefore may not have had a significant effect on the primary producers in this reach of the Burntwood River.

3.5 Split Lake Area

3.5.1 Burntwood River at Inlet to Split Lake

3.5.1.1 Current Water Quality

Water quality at this site was typical for the Burntwood River below the confluence with the CRD. Concentrations of all but two parameters were the same as at the Thompson site (Table 21). These exceptions were the annual mean total and fixed NFR concentrations which were 45-50% higher at the inlet to Split Lake than at Thompson. This may be a product of the documented bank erosion site near First Rapids (Kellerhals 1988).

The same variables exceeded the guideline values for drinking water, recreation, and aquatic life at this site as found in the natural rivers (Table 22). All measurements of turbidity always exceeded the drinking water guideline, and the maximum values also exceeded the guideline for recreational use. Mean colour exceeded the drinking water guideline by about 47%. Maximum extractable Mn exceeded the drinking water guideline by a factor of 3.5 and the mean was at the guideline level. Extractable Fe exceeded the guidelines for both drinking water and aquatic life by a factor of 2.6 on average and by as much as a factor of 9.5. Extractable Al exceeded the aquatic life guideline by a factor of 5.9 on average and by as much as a factor of 21. The total Cu concentration was 2.4 times the aquatic life guideline on average with maximum concentrations as much as a factor of 14 above the guideline value. Peak total Cd and total Pb also exceeded the aquatic life guidelines.

Table 21. Comparison of current water quality in the Burntwood River at Thompson and at the inlet to Split Lake. $X_{t,lm}$ denotes the 1987-1989 mean for the Burntwood River at Thompson. $X_{s,pl}$ denotes the 1987-1989 mean for the Burntwood River at the inlet to Split Lake. p is the probability of Student's t -statistic for the comparison.

Location	pH	Conductivity $\mu S\ cm^{-1}$	Turbidity NTU	Colour Relative	TDS $mg\ L^{-1}$	NFR $mg\ L^{-1}$	NFR Fixed $mg\ L^{-1}$	Tot Alk $mg\ L^{-1}$	Tot Hard $mg\ L^{-1}$	Diss Ca $mg\ L^{-1}$
Burntwood R.										
p	0.954	0.761	0.201	0.988	0.704	0.015	0.026	0.727	0.525	0.156
$X_{t,lm}$	7.69	115.	27.0 ¹	21.	60.0	16.3 ¹	14.7 ¹	53.0 ¹	53.0	14.6
$X_{s,pl}$	7.76	123.	35.0 ¹	22.	64.7	24.3	21.3	56.7 ¹	56.7	15.9
	Diss Mg $mg\ L^{-1}$	Diss K $mg\ L^{-1}$	Diss Na $mg\ L^{-1}$	React Si $mg\ L^{-1}$	Diss Cl $mg\ L^{-1}$	Diss F $mg\ L^{-1}$	Diss SO ₄ $mg\ L^{-1}$	Ortho P $mg\ L^{-1}$	TDP $mg\ L^{-1}$	Tot P $mg\ L^{-1}$
p	0.991	1.000	0.997	0.999	0.998	0.982	0.943	1.000	1.000	0.810
$X_{t,lm}$	4.0	1.06 ¹	2.7 ¹	2.00	1.2	0.09 ¹	2.7	0.003	0.012	0.036 ¹
$X_{s,pl}$	4.1	1.05	2.8	1.96	1.2	0.10	2.9	0.003	0.012	0.039
				TDN $mg\ L^{-1}$	Diss NO ₂ -NO ₃ $mg\ L^{-1}$	Susp N $mg\ L^{-1}$	DOC $mg\ L^{-1}$	TOC $mg\ L^{-1}$	DIC $mg\ L^{-1}$	Chlorophyll a $mg\ L^{-1}$
p				1.000	0.992	0.459	0.716	0.381	0.260	0.990
$X_{t,lm}$				0.248	0.022	0.049	6.48	6.94	12.4	0.002
$X_{s,pl}$				0.248	0.019	0.063	6.78 ¹	7.41 ¹	13.4	0.002

Table 21. continued.

Location	Extr Al mg L ⁻¹	Diss Al mg L ⁻¹	Diss As mg L ⁻¹	Tot Cd mg L ⁻¹	Diss Cd mg L ⁻¹	Tot Co mg L ⁻¹	Tot Cu mg L ⁻¹	Diss Cu mg L ⁻¹	Extr Fe mg L ⁻¹	Diss Fe mg L ⁻¹	
Burntwood R.											
p	0.274	0.998	1.000	1.000	1.000	0.835	1.000	0.997	0.393	0.573	
X _{thin}	0.488	0.080	0.0002	0.0001	0.0001	0.0004	0.0042 ¹	0.0014	0.628 ¹	0.077	
X _{sp1}	0.596	0.082	0.0002	0.0001	0.0001	0.0005	0.0047	0.0015	0.787	0.091	
	Tot Pb mg L ⁻¹	Diss Pb mg L ⁻¹	Extr Mn mg L ⁻¹	Diss Mn mg L ⁻¹	Tot Ni mg L ⁻¹	Diss Ni mg L ⁻¹	Diss Se mg L ⁻¹	Tot V mg L ⁻¹	Diss V mg L ⁻¹	Tot Zn mg L ⁻¹	Diss Zn mg L ⁻¹
p	0.994	0.991	0.107	1.000	0.280	0.183	1.000	0.502	1.000	0.890	0.959
X _{thin}	0.0010	0.0010	0.021 ¹	0.004	0.0018	0.0006	0.0001 ¹	0.0013	0.0004	0.0035 ¹	0.0010
X _{sp1}	0.0011	0.0007	0.031	0.004 ¹	0.0023	0.0011	0.0001 ¹	0.0016	0.0004	0.0046	0.0007 ²

¹ Discharge-standardized variables.

Table 22. Summary statistics for physical and chemical water quality in the Burntwood River at the inlet to Split Lake, September, 1986, to October, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.6	0.2	8.2	7.2
	Lab		7.76	0.19	8.17	7.33
Conductivity	Field	$\mu\text{S cm}^{-1}$	120	12	138	86
	Lab		123	6	138	107
Turbidity	Field	NTU	34 ⁺	11	70 ^{+,++}	17 ⁺
	Lab		35 ⁺	10	71 ^{+,++}	20 ⁺
Colour	True	Relative	22 ⁺	6	40 ⁺	10
TDS		mg L^{-1}	65	3	71	58
NFR	Total	mg L^{-1}	24	16	107	10
	Fixed	mg L^{-1}	21	15	100	8
Alkalinity	Total	mg L^{-1}	56.9	3.2	64.7	49.1
	Bicarbonate	mg L^{-1}	69.4	3.9	78.9	59.9
Hardness	Total	mg L^{-1}	56.6	3.1	62.9	48.1
	Non-carbonate	mg L^{-1}	0.7	1.0	3.7	0
Free CO ₂		mg L^{-1}	2.45	1.79	8.87	0
Phosphorus	Particulate	mg L^{-1}	0.027	0.014	0.088	0.011
	Ortho PO ₄	mg L^{-1}	0.003	0.003	0.017	<0.003
	Total Dissolved	mg L^{-1}	0.012	0.006	0.029	<0.003
	Total	mg L^{-1}	0.039	0.013	0.102	0.021
Nitrogen	Particulate	mg L^{-1}	0.062	0.026	0.123	<0.010
	Total Dissolved	mg L^{-1}	0.248	0.039	0.370	0.200
	NO ₂ -NO ₃	mg L^{-1}	0.019	0.025	0.077	<0.010
	Total NH ₃	mg L^{-1}	0.014	0.007	0.030	<0.005
Carbon	Total	mg L^{-1}	0.31	0.04	0.40	0.24
	DOC	mg L^{-1}	6.78	0.76	9.10	5.70
	POC	mg L^{-1}	0.635	0.316	2.100	0.270
	TOC	mg L^{-1}	7.40	0.89	9.72	6.11
Chlorophyll-a	DIC	mg L^{-1}	13.4	0.7	14.9	11.8
		mg L^{-1}	0.002	0.001	0.005	<0.001
Ca	Dissolved	mg L^{-1}	15.9	0.9	17.6	13.0
Mg	Dissolved	mg L^{-1}	4.1	0.2	4.6	3.6
K	Dissolved	mg L^{-1}	1.05	0.06	1.21	0.96
Na	Dissolved	mg L^{-1}	2.8	0.3	3.5	2.4

Table 22. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L ⁻¹	1.96	0.68	3.39	0.89
Cl	Dissolved	mg L ⁻¹	1.2	0.2	1.5	0.9
F	Dissolved	mg L ⁻¹	0.09	0.02	0.15	0.08
SO ₄	Dissolved	mg L ⁻¹	2.9	0.9	7.1	1.5
Al	Extractable	mg L ⁻¹	0.592 ⁺⁺⁺	0.315	2.140 ⁺⁺⁺	0.235 ⁺⁺⁺
	Dissolved	mg L ⁻¹	<0.100	0.058	0.255	<0.100
As	Dissolved	mg L ⁻¹	0.0002	0.0001	0.0003	<0.0001
Ba	Total	mg L ⁻¹	<0.08	0.01	0.11	<0.08
Cd	Total	mg L ⁻¹	<0.0001	0.0001	0.0003 ⁺⁺⁺	<0.0001
	Dissolved	mg L ⁻¹	<0.0001	0.0001	0.0003	<0.0001
Co	Total	mg L ⁻¹	0.0005	0.0004	0.0021	<0.0005
Cu	Total	mg L ⁻¹	0.0048 ⁺⁺⁺	0.0062	0.0278 ⁺⁺⁺	0.0010
	Dissolved	mg L ⁻¹	0.0014	0.0006	0.0032	0.0010
Fe	Extractable	mg L ⁻¹	0.774 ^{+,+++}	0.416	2.860 ^{+,+++}	0.299
	Dissolved	mg L ⁻¹	0.090	0.053	0.208	<0.007
Pb	Total	mg L ⁻¹	0.0010	0.0007	0.0034 ⁺⁺⁺	<0.0007
	Dissolved	mg L ⁻¹	0.0007	0.0008	0.0035	<0.0007
Mn	Extractable	mg L ⁻¹	0.030	0.027	0.177 ⁺	0.004
	Dissolved	mg L ⁻¹	0.004	0.002	0.011	<0.002
Ni	Total	mg L ⁻¹	0.0023	0.0011	0.0077	0.0010
	Dissolved	mg L ⁻¹	0.0011	0.0005	0.0021	<0.0005
Se	Dissolved	mg L ⁻¹	0.0001	0.0001	0.0004	<0.0001
V	Total	mg L ⁻¹	0.0016	0.0008	0.0047	<0.0005
	Dissolved	mg L ⁻¹	<0.0005	0.0002	0.0009	<0.0005
Zn	Total	mg L ⁻¹	0.0045	0.0033	0.0181	0.0011
	Dissolved	mg L ⁻¹	0.0009	0.0007	0.0021	<0.0003
Discharge	Daily	m ³ sec ⁻¹	908	105	1131	719

⁺ Exceeds limit for drinking water.

⁺⁺ Exceeds limit for recreational use.

⁺⁺⁺ Exceeds limit for protection of aquatic life.

No analysis of changes in water quality since CRD is possible for this site due to the lack of pre-development data.

3.5.1.2 Water Quality - Water Discharge Relationships

Positive correlations with discharge dominated at this site during the 1987-1989 study period, consistent with our findings at sites upstream on the Burntwood River. The positive correlates included turbidity (lab and field), alkalinity (total and bicarbonate), free CO₂, DOC, TOC, and dissolved Na and Se (Figure 32). Negative correlates with discharge were dissolved Mn and Zn. Again, none of the relationships showed much impact prediction or management potential. Nine of the 11 significant correlations accounted for just 10-20% of the variance in the dependent variable, although 2 correlations accounted for 25-35% of the variance.

The most useful of these water quality - water discharge relationships for prediction and management purposes are those involving turbidity and alkalinity (Figure 32). Although none of these correlations accounted for more than 20% of the variance, all reflected a general linear trend in the data. The strength of these relationships might be improved with additional data collection which may reduce the influence of the few outlying values. The discharge relationship for dissolved zinc also has potential for predictive purposes but, with the relatively few points on the curve (Figure 32k; n=14), additional data collection is advised to determine if the relationship holds.

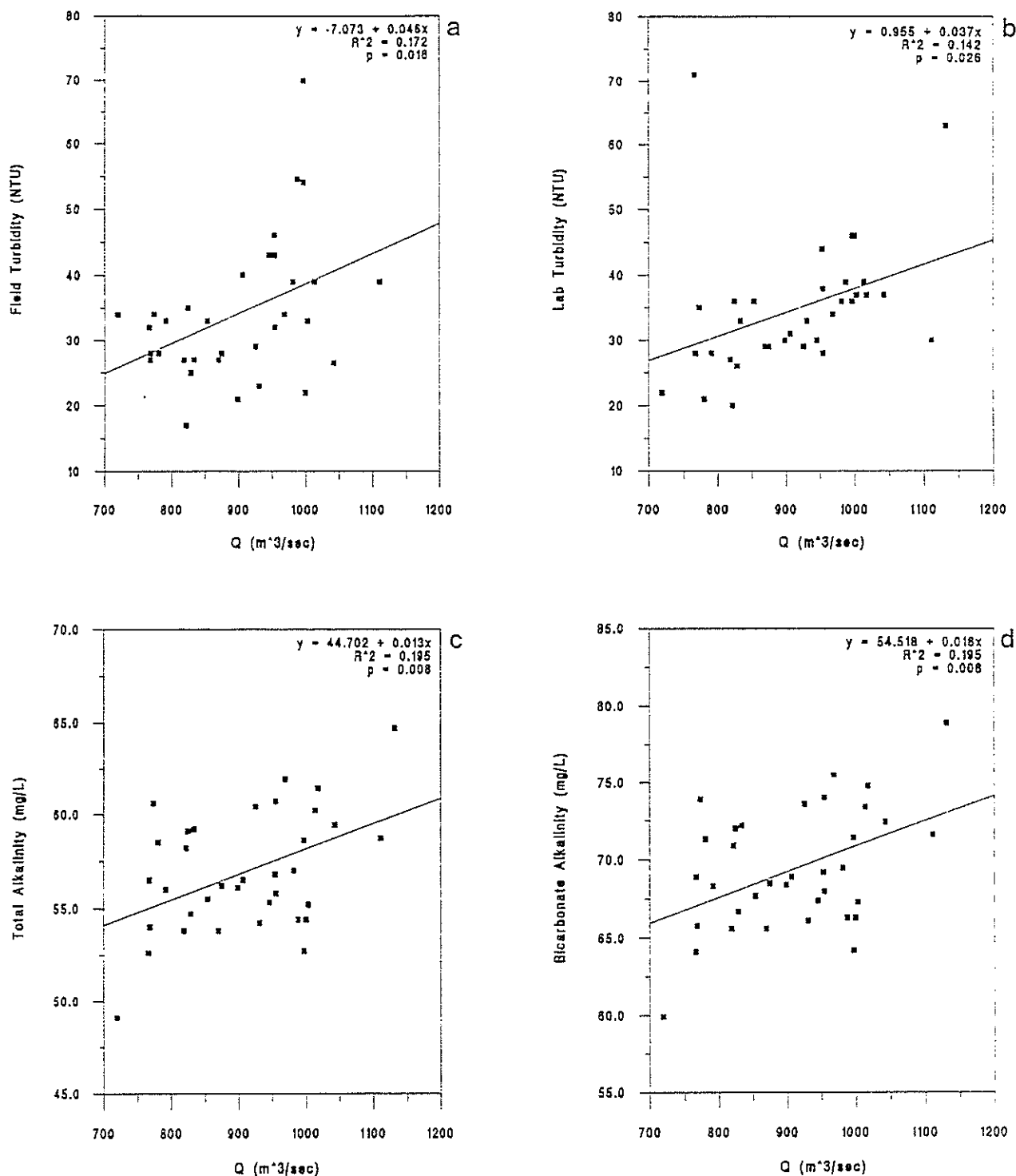


Figure 32. Water chemistry - water discharge relationships in the Burntwood River at the inlet to Split Lake, 1987-1989. a. field turbidity b. lab turbidity c. total alkalinity d. bicarbonate alkalinity e. free CO₂ f. DOC g. TOC h. dissolved Na i. dissolved Mn j. dissolved Se k. dissolved Zn.

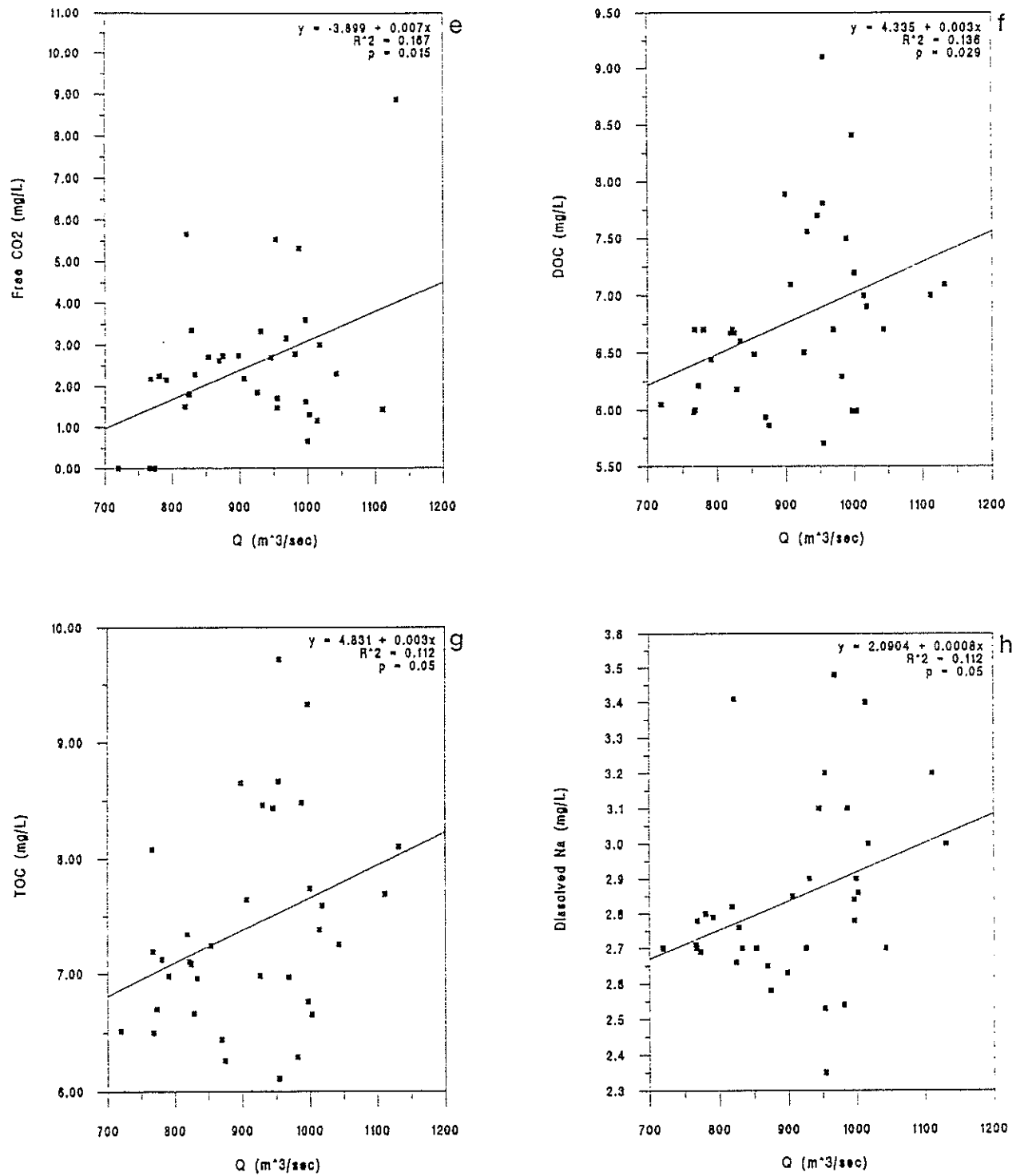


Figure 32. Continued.

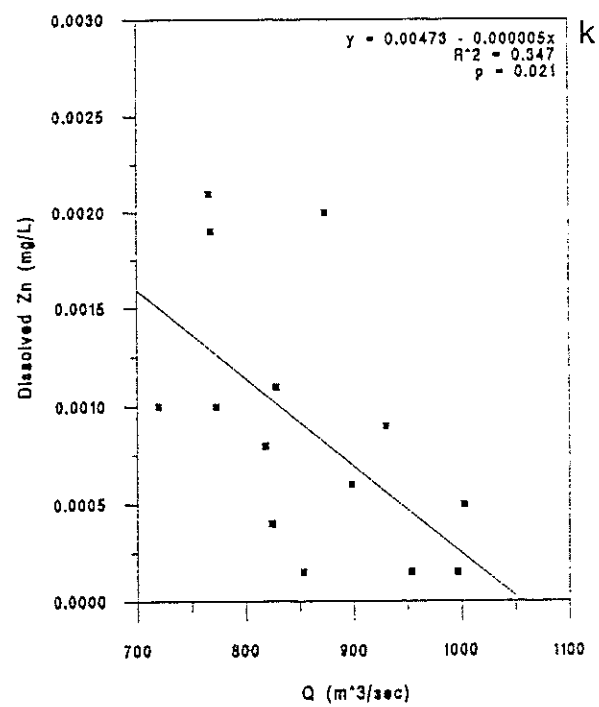
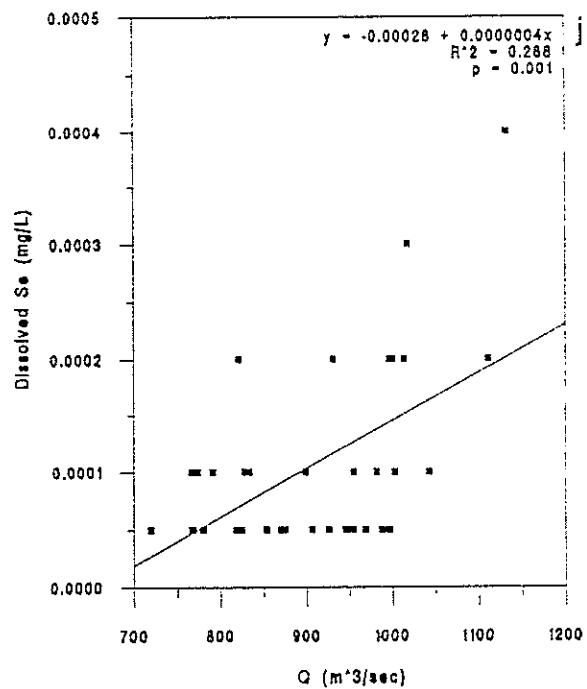
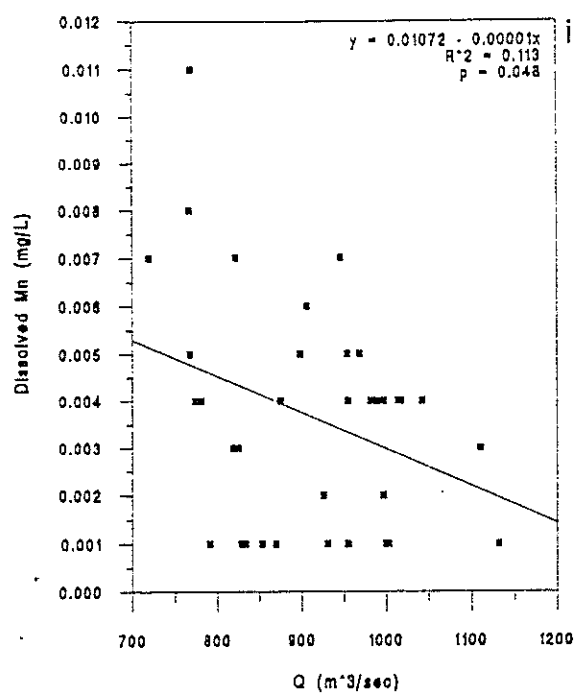


Figure 32. Continued.

No discharge analysis was possible for the pre-development period due to the lack of water quality data, although changes similar to those which occurred upstream at Thompson probably have occurred here as well.

3.5.1.3 Seasonal Differences

As found at a number of sites upstream on the Burntwood River, concentrations of dissolved NO_2 - NO_3 were higher under ice-cover while chlorophyll a levels were lower under ice-cover than in the open-water season (Figure 33). Causes of these seasonal patterns are probably the same as at the upstream sites.

3.5.2 Nelson River below Grass River at Inlet to Split Lake

3.5.2.1 Current Water Quality

Physical and chemical water quality in the Nelson River at the inlet to Split Lake was not significantly different in any aspect from that at Sea River Falls, upstream (Table 23). As at all other sites in the study, mean turbidity exceeded the drinking water guideline as did peak colour (Table 24). Mean extractable Al and total Cu exceeded the respective aquatic life guidelines by a factor of 1.9 on average while the peak values were 4 and 8 times the guideline values. The maximum extractable Fe concentration was about 40% above the aquatic life and drinking water guidelines although the mean value was 36% below these limits.

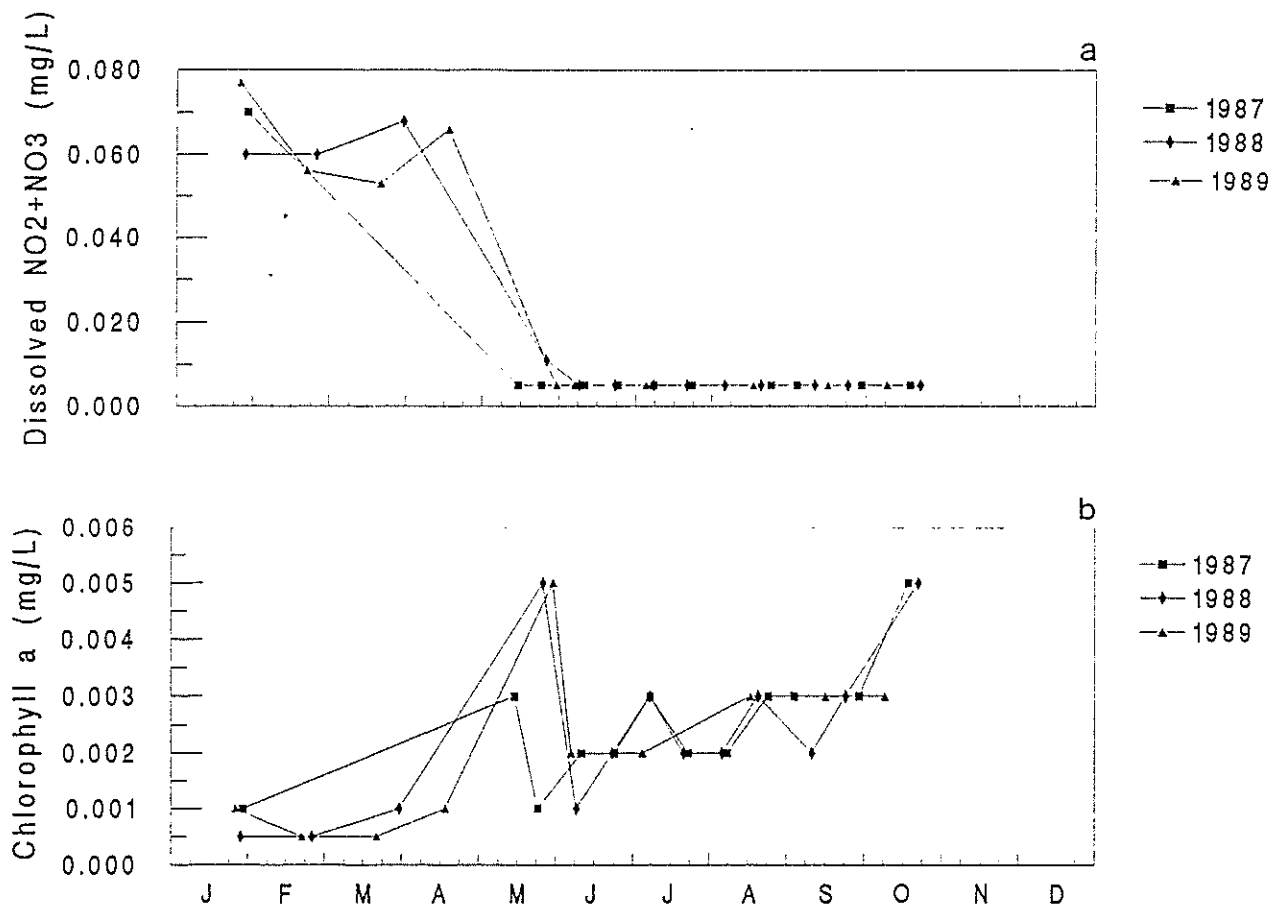


Figure 33. Seasonal water quality trends in the Burntwood River at the inlet to Split Lake. a. dissolved NO₂-NO₃ b. chlorophyll a.

Table 23. Comparison of current water quality in the Nelson River below Sea River Falls and at the inlet to Split Lake. $X_{\text{se},rf}$ denotes the 1987-1989 mean for the Nelson River below Sea River Falls. X_{sep1} denotes the 1987-1989 mean for the Nelson River at the inlet to Split Lake. p is the probability of Student's t -statistic for the comparison.

Location	pH	Conductivity $\mu\text{S cm}^{-1}$	Turbidity NTU	Colour Relative	TDS mg L^{-1}	NFR mg L^{-1}	NFR Fixed mg L^{-1}	Tot Alk mg L^{-1}	Tot Hard mg L^{-1}	Diss Ca mg L^{-1}
Nelson R.										
p	0.999	0.904	0.877	1.000	0.921	0.985	0.878	0.999	0.989	0.999
$X_{\text{se},rf}$	7.83 ¹	305.0 ¹	8.	9.	161.7 ¹	6.0 ¹	5.0 ¹	102.0 ¹	117.0 ¹	29.1 ¹
X_{sep1}	7.85	293.0	9.	9.	154.7	5.0	3.3	103.0	115.0	29.3
	Diss Mg mg L^{-1}	Diss K mg L^{-1}	Diss Na mg L^{-1}	React Si mg L^{-1}	Diss Cl mg L^{-1}	Diss F mg L^{-1}	Diss SO_4 mg L^{-1}	Ortho P mg L^{-1}	TDP mg L^{-1}	Tot P mg L^{-1}
p	0.855	0.610	0.753	0.475	0.774	0.982	0.832	0.097	0.946	0.940
$X_{\text{se},rf}$	10.7 ¹	2.25 ¹	15.6 ¹	0.93	17.7 ¹	0.10	23.7 ¹	0.002	0.010	0.023
X_{sep1}	10.2	2.06	13.8	1.24	15.3	0.10 ¹	21.1 ¹	0.004 ¹	0.011	0.022
				TDN mg L^{-1}	Diss $\text{NO}_2\text{-NO}_3$ mg L^{-1}	Susp N mg L^{-1}	DOC mg L^{-1}	TOC mg L^{-1}	DIC mg L^{-1}	Chlorophyll a mg L^{-1}
p				1.000	0.999	0.995	0.961	0.874	0.979	1.000
$X_{\text{se},rf}$				0.301	0.014 ¹	0.051 ¹	7.67	8.05	24.8 ¹	0.002
X_{sep1}				0.305	0.016	0.048	7.51	7.79	24.4	0.002

Table 23. continued.

[illegible]¹ Discharge-standardized variables.

Table 24. Summary statistics for physical and chemical water quality in Nelson River below Grass River, September, 1986, to October, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.7	0.3	8.3	7.1
	Lab		7.85	0.21	8.16	7.30
Conductivity	Field	$\mu\text{S cm}^{-1}$	273	43	345	185
	Lab		291	38	360	232
Turbidity	Field	NTU	12 ⁺	7	45 ⁺	4
	Lab		9 ⁺	3	18 ⁺	5
Colour	True	Relative	9	5	20 ⁺	<5
TDS		mg L^{-1}	154	21	195	119
NFR	Total	mg L^{-1}	5	2	11	2
	Fixed	mg L^{-1}	4	2	9	<1
Alkalinity	Total	mg L^{-1}	102.7	10.4	123.0	85.3
	Bicarbonate	mg L^{-1}	125.4	12.4	149.9	104.0
Hardness	Total	mg L^{-1}	114.4	13.4	141.9	91.4
	Non-carbonate	mg L^{-1}	11.9	4.6	21.1	3.9
Free CO ₂		mg L^{-1}	4.25	3.52	15.28	0
Phosphorus	Particulate	mg L^{-1}	0.011	0.005	0.020	<0.006
	Ortho PO ₄	mg L^{-1}	0.003	0.002	0.011	<0.003
	Total Dissolved	mg L^{-1}	0.011	0.003	0.020	0.005
	Total	mg L^{-1}	0.022	0.005	0.032	0.012
Nitrogen	Particulate	mg L^{-1}	0.048	0.059	0.380	<0.010
	Total Dissolved	mg L^{-1}	0.306	0.039	0.380	0.210
	NO ₂ -NO ₃	mg L^{-1}	0.016	0.016	0.054	<0.010
	Total NH ₃	mg L^{-1}	0.023	0.011	0.060	0.005
	Total	mg L^{-1}	0.35	0.04	0.43	0.24
Carbon	DOC	mg L^{-1}	7.55	0.50	8.70	6.34
	POC	mg L^{-1}	0.285	0.082	0.520	0.140
	TOC	mg L^{-1}	7.83	0.49	8.86	6.64
	DIC	mg L^{-1}	24.3	2.7	29.5	19.8
Chlorophyll-a		mg L^{-1}	0.002	0.002	0.010	<0.001
Ca	Dissolved	mg L^{-1}	29.2	3.1	35.4	23.5
Mg	Dissolved	mg L^{-1}	10.1	1.4	13.0	7.8
K	Dissolved	mg L^{-1}	2.03	0.29	2.64	1.61
Na	Dissolved	mg L^{-1}	13.6	2.9	19.1	9.2

Table 24. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L ⁻¹	1.29	0.66	2.94	0.34
Cl	Dissolved	mg L ⁻¹	14.9	3.8	22.5	9.2
F	Dissolved	mg L ⁻¹	0.10	0.01	0.12	0.08
SO ₄	Dissolved	mg L ⁻¹	20.7	4.6	30.0	13.1
Al	Extractable	mg L ⁻¹	0.188 ⁺⁺⁺	0.101	0.418 ⁺⁺⁺	<0.100
	Dissolved	mg L ⁻¹	<0.100	0.016	0.119	<0.100
As	Dissolved	mg L ⁻¹	0.0007	0.0002	0.0010	0.0004
Ba	Total	mg L ⁻¹	<0.08		<0.08	<0.08
Cd	Total	mg L ⁻¹	<0.0001	0.0001	0.0003	<0.0001
	Dissolved	mg L ⁻¹	<0.0001		0.0002	<0.0001
Co	Total	mg L ⁻¹	<0.0005		0.0005	<0.0005
Cu	Total	mg L ⁻¹	0.0037 ⁺⁺⁺	0.0043	0.0159 ⁺⁺⁺	0.0010
	Dissolved	mg L ⁻¹	0.0020	0.0007	0.0031	0.0010
Fe	Extractable	mg L ⁻¹	0.196	0.074	0.409 ⁺⁺⁺	0.045
	Dissolved	mg L ⁻¹	0.043	0.030	0.178	0.013
Pb	Total	mg L ⁻¹	<0.0007	0.0004	0.0017	<0.0007
	Dissolved	mg L ⁻¹	0.0011	0.0020	0.0072	<0.0007
Mn	Extractable	mg L ⁻¹	0.010	0.003	0.020	0.004
	Dissolved	mg L ⁻¹	0.003	0.003	0.018	<0.002
Ni	Total	mg L ⁻¹	0.0011	0.0004	0.0023	0.0005
	Dissolved	mg L ⁻¹	0.0009	0.0004	0.0017	<0.0005
Se	Dissolved	mg L ⁻¹	0.0002	0.0001	0.0005	<0.0001
V	Total	mg L ⁻¹	0.0008	0.0004	0.0014	<0.0005
	Dissolved	mg L ⁻¹	<0.0005	0.0002	0.0010	<0.0005
Zn	Total	mg L ⁻¹	0.0024	0.0020	0.0077	0.0006
	Dissolved	mg L ⁻¹	0.0010	0.0009	0.0027	<0.0003
Discharge	Daily	m ³ sec ⁻¹	1271	352	2253	734

+ Exceeds limit for drinking water.

++ Exceeds limit for recreational use.

+++ Exceeds limit for protection of aquatic life.

3.5.2.2 Water Chemistry - Water Discharge Relationships

3.5.2.2.1 Pre-development

Pre-development data from 3 sites were used in the analysis, with the discharge analysis conducted separately on each data set. At the Nelson River below Sipiwesk Lake, the only significant correlation was a positive relationship with NFR, which accounted for 67% of the variance (Figure 34). At the Nelson River above Kelsey Generating Station, 9 variables were correlated with discharge. Negative correlates included conductivity (field), dissolved Na, K, Mg, Ca, SO_4 , TDS, and dissolved Fe (Figure 35). The sole positive correlate was POC. All of the regressions accounted for at least 45% of the variance in the dependent variables. The correlations of discharge with Ca, K, SO_4 , and conductivity were particularly strong, accounting for 78 to 93% of the variance.

At the Nelson River below Kelsey Generating Station, 4 variables were significantly correlated with discharge (Figure 36). Positive correlates were NFR and TDN while the negative correlates were conductivity (field) and TDS. The regressions generally were not as strong as found above the Kelsey Dam, accounting for 30 to 66% of the variance in the dependent variables.

There are some general similarities in the discharge relationships at the three stations. Conductivity and TDS were both negatively correlated with discharge both above and below Kelsey, while NFR was positively correlated with discharge below Sipiwesk Lake and below Kelsey. Little significance should be attached to

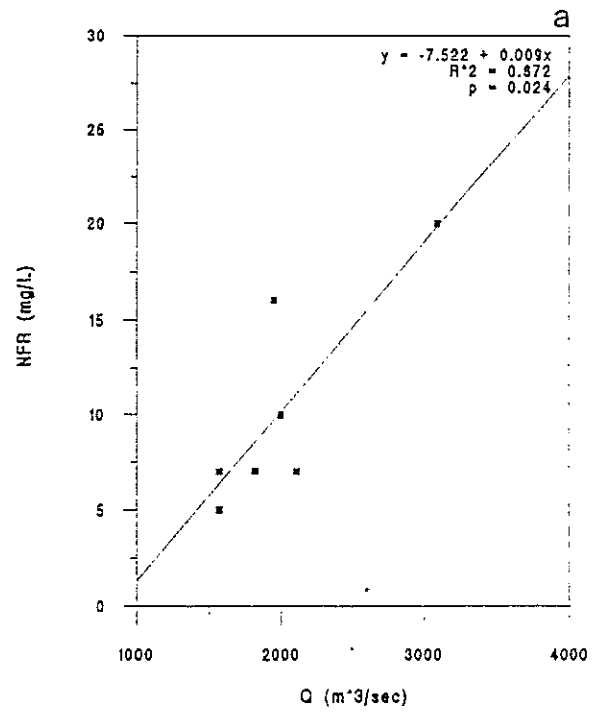


Figure 34. Water chemistry - water discharge relationships in the Nelson River below Sipiwesk Lake, 1972-1973. a. NFR.

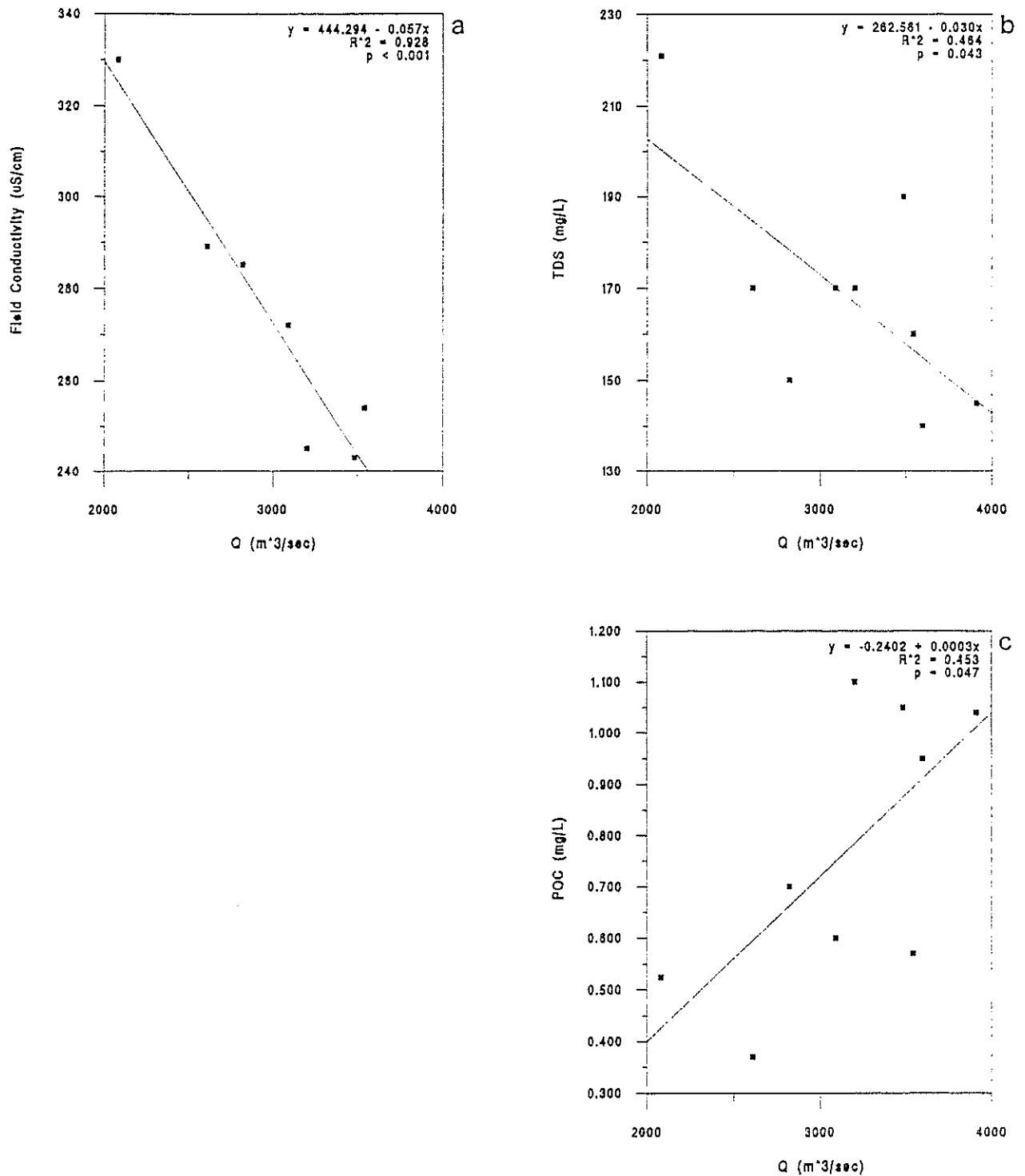


Figure 35. Water chemistry - water discharge relationships in the Nelson River above Kelsey Generating Station, 1972-1974. a. field conductivity b. TDS c. POC d. dissolved Na e. dissolved SO_4 f. dissolved Fe g. dissolved Ca h. dissolved Mg i. dissolved K.

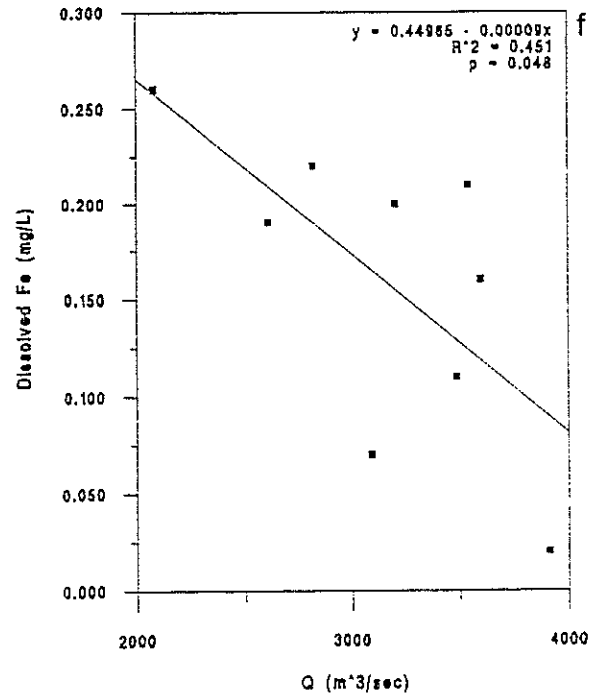
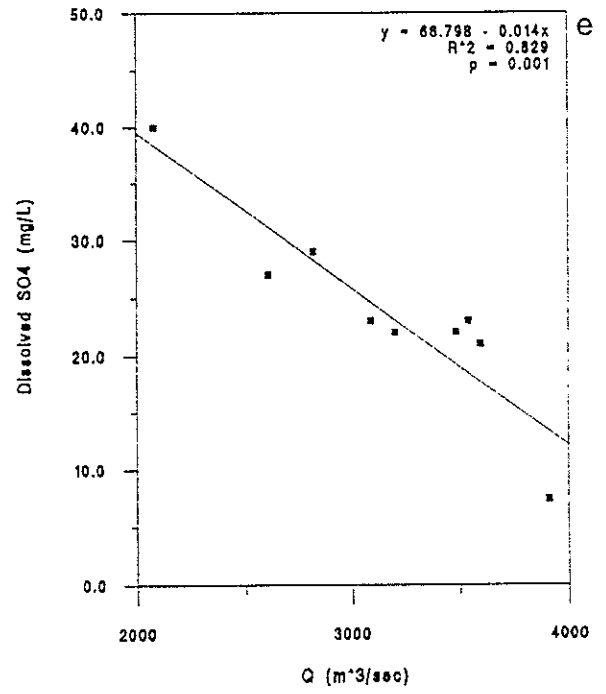
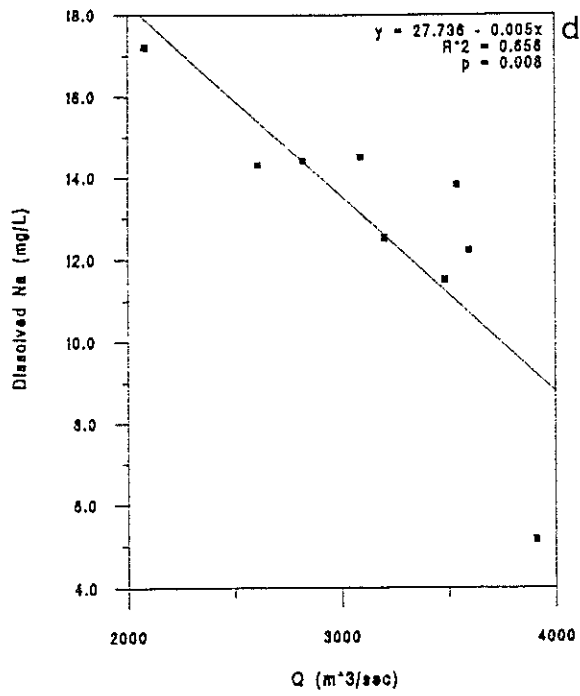


Figure 35. Continued.

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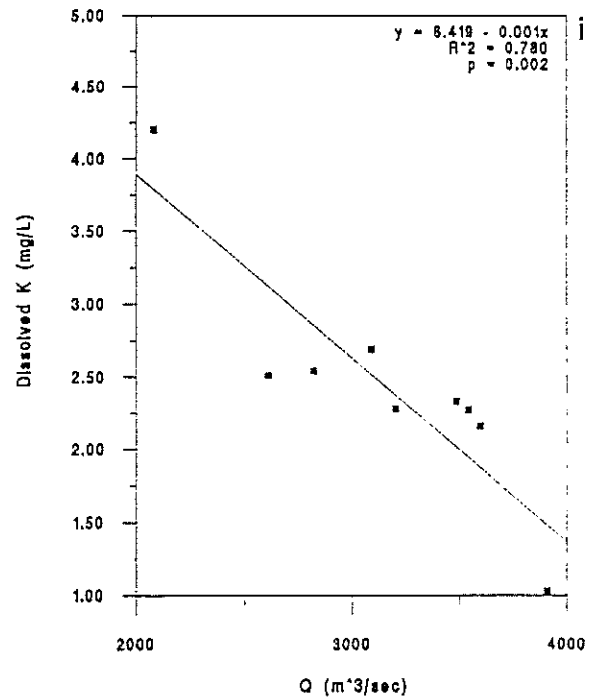
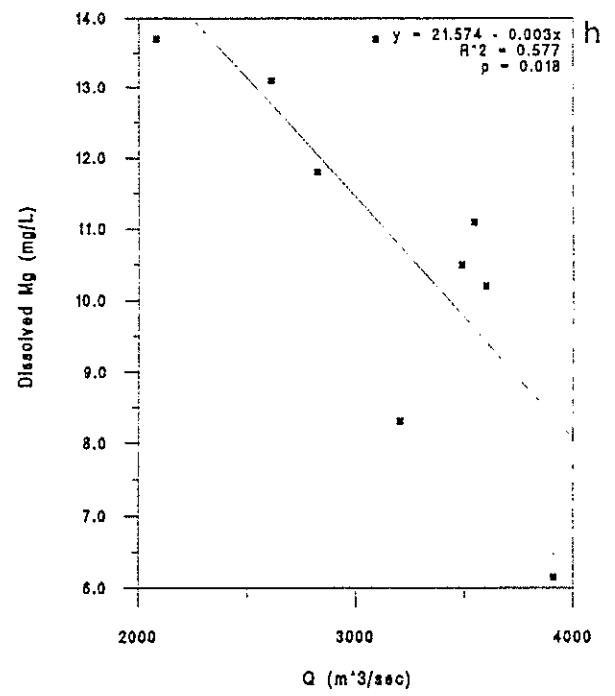
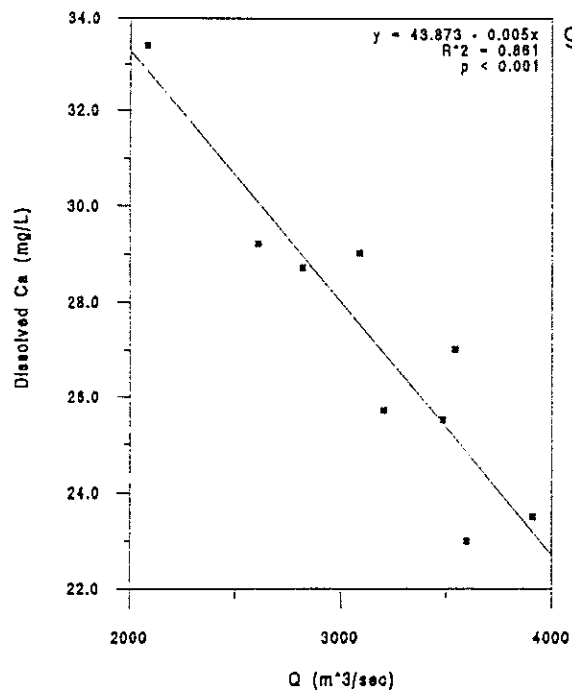


Figure 35. Continued.

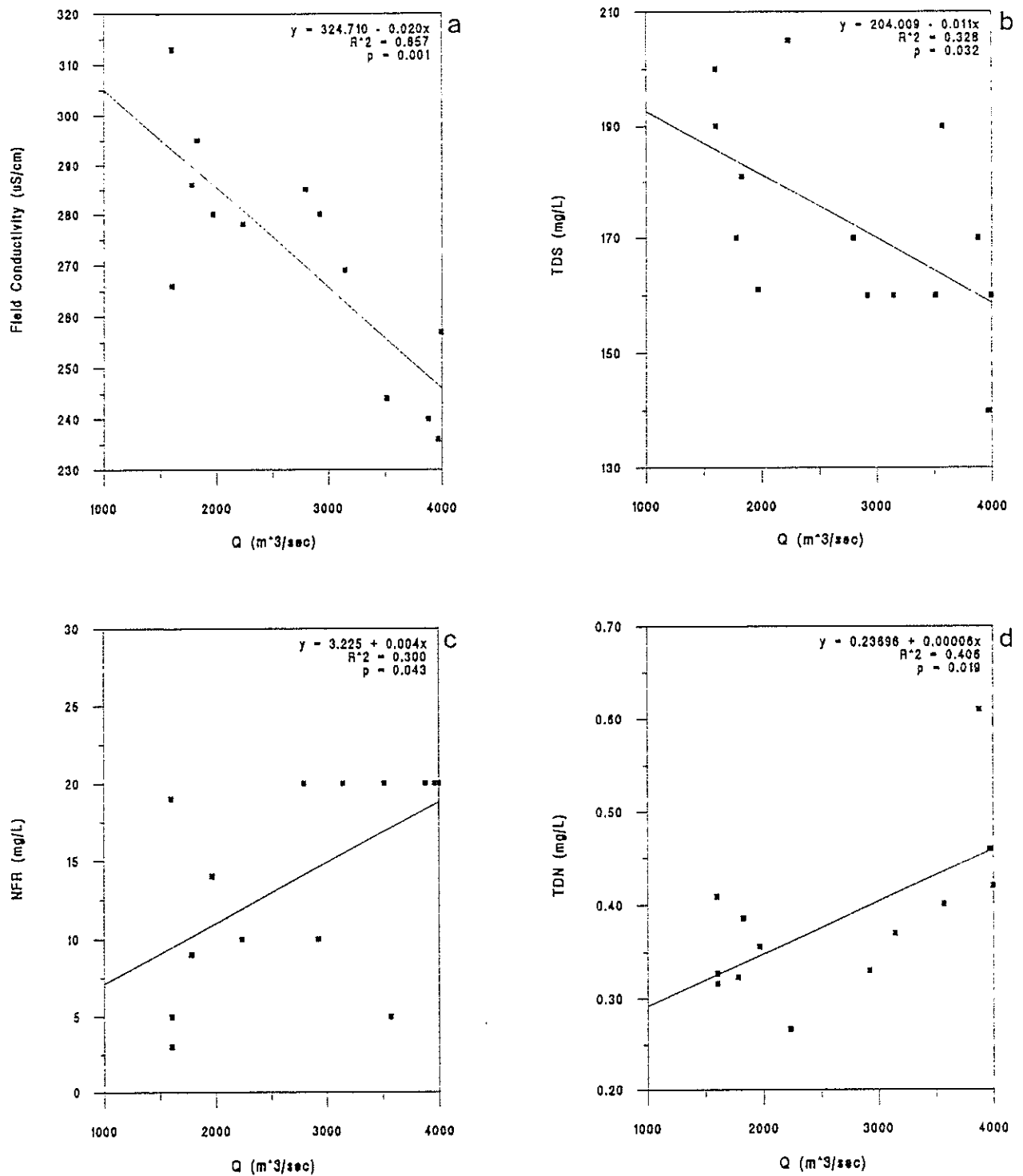


Figure 36. Water chemistry - water discharge relationships in the Nelson River below Kelsey Generating Station, 1972-1974. a. field conductivity b. TDS c. NFR d. TDN.

any specific differences in these common relationships, however, as all are based on few samples, the samples were not collected on the same date at all stations and, therefore, the range of discharge on which the relationships are based was not the same at each station.

The pre-development discharge relationships at this site also follow the trends evident in the natural rivers and in the Burntwood River at Thompson prior to development. Negative correlates predominated with positive correlations restricted to variables associated with particulate matter or suspended sediments.

3.5.2.2.2 Current

Fewer variables were significantly correlated with discharge at this site than at sites in the Burntwood River drainage or upstream on the Nelson River. Most of the significant correlations had positive slopes. These included ortho PO_4 , dissolved fluoride, SO_4 , and Cd, and total Cd (Figure 37). The only negative correlate was total N. None of these correlations was particularly strong, with all but one accounting for just 10-20% of the variance in the dependent variable.

None of the significant correlations with discharge in the current data resembles any which existed during the pre-development (1972-1973) period, but the cause of these changes is less clear than was the case at sites on the Burntwood River. The changes are superficially similar to those caused by CRD (i.e., positive slopes for variables which might otherwise be expected to be negative

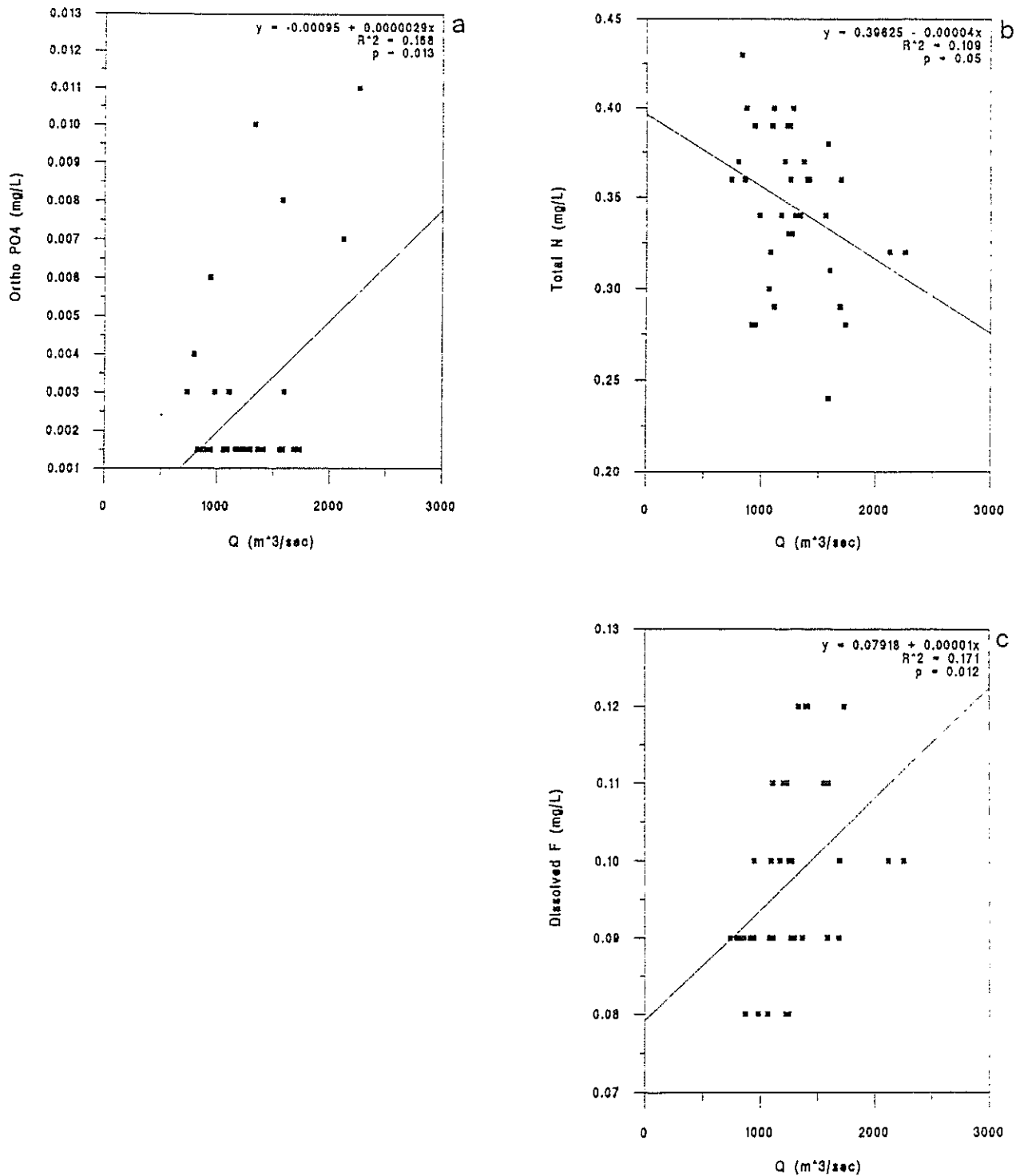


Figure 37. Water chemistry - water discharge relationships in the Nelson River at the inlet to Split Lake, 1987-1989. a. ortho PO₄ b. total N c. dissolved F d. dissolved SO₄ e. dissolved Cd f. total Cd.

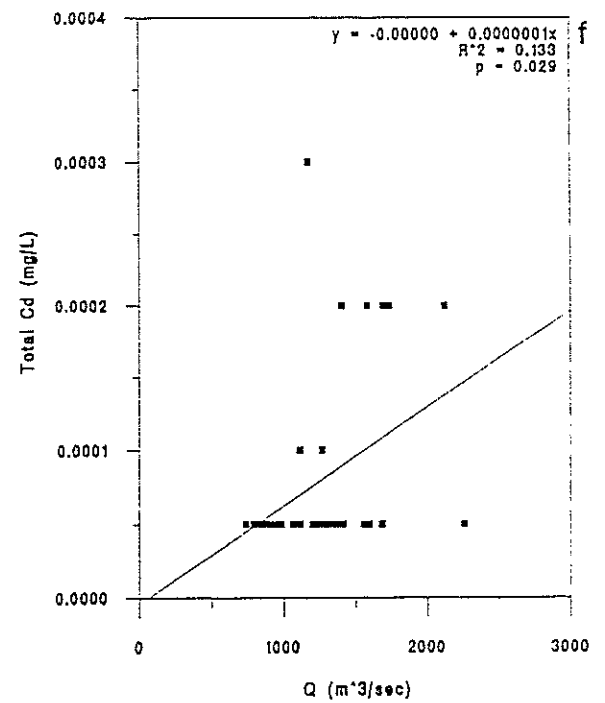
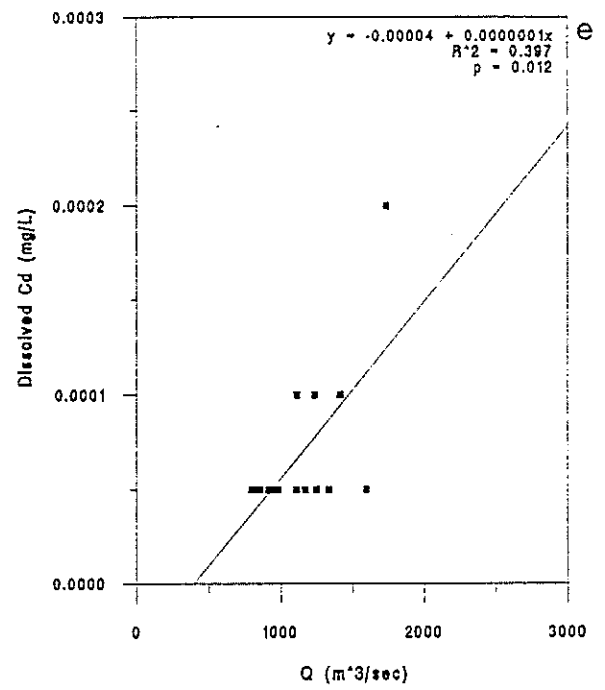
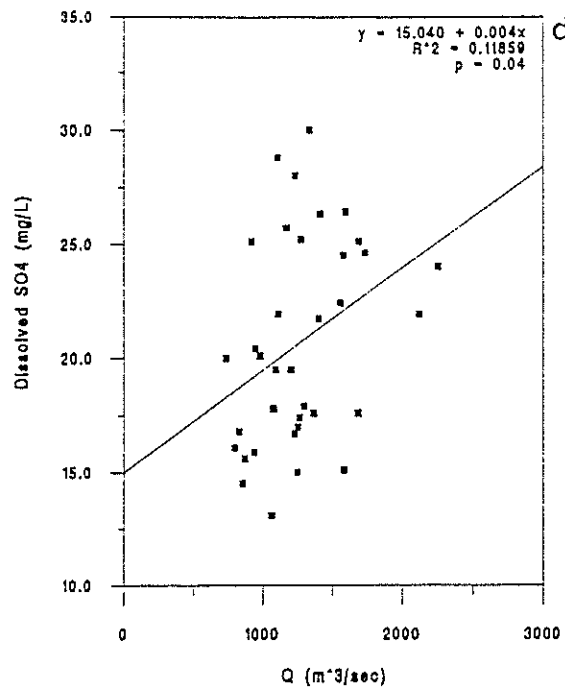


Figure 37. Continued.

correlates of discharge). Alternatively, these changes may be a result of the low discharges and rather narrow range of discharge fluctuation during the FEMP study compared with the pre-development period, resulting in little overlap in the discharges which were common to the two periods. It is questionable if any of the correlations observed in the FEMP data would prevail over a broader range of discharges. On this basis, it cannot be determined if LWR has had any effect on the water quality - water discharge relationships at this site. Additional sampling during periods of higher discharge is required to determine if the relationships observed during the FEMP study are representative of current conditions or if some other relationships represent the usual post-development condition. The discharge relationships derived from the FEMP data are not recommended for use in prediction or management without additional data collection over a wider range of discharges.

3.5.2.3 Seasonal Differences

Few variables showed marked differences between the open-water and ice-cover seasons during the 1987-1989 FEMP sampling period although conductivity, hardness (total), dissolved Mg, and dissolved $\text{NO}_2\text{-NO}_3$ tended to be higher under ice cover (Figure 38). The cause of the higher $\text{NO}_2\text{-NO}_3$ under ice is probably the same as discussed in Section 3.3.1.3. The elevated hardness is a result of the higher Mg, but the cause of the elevated Mg is unclear. Mg was not correlated with discharge at this site and, in general, is not

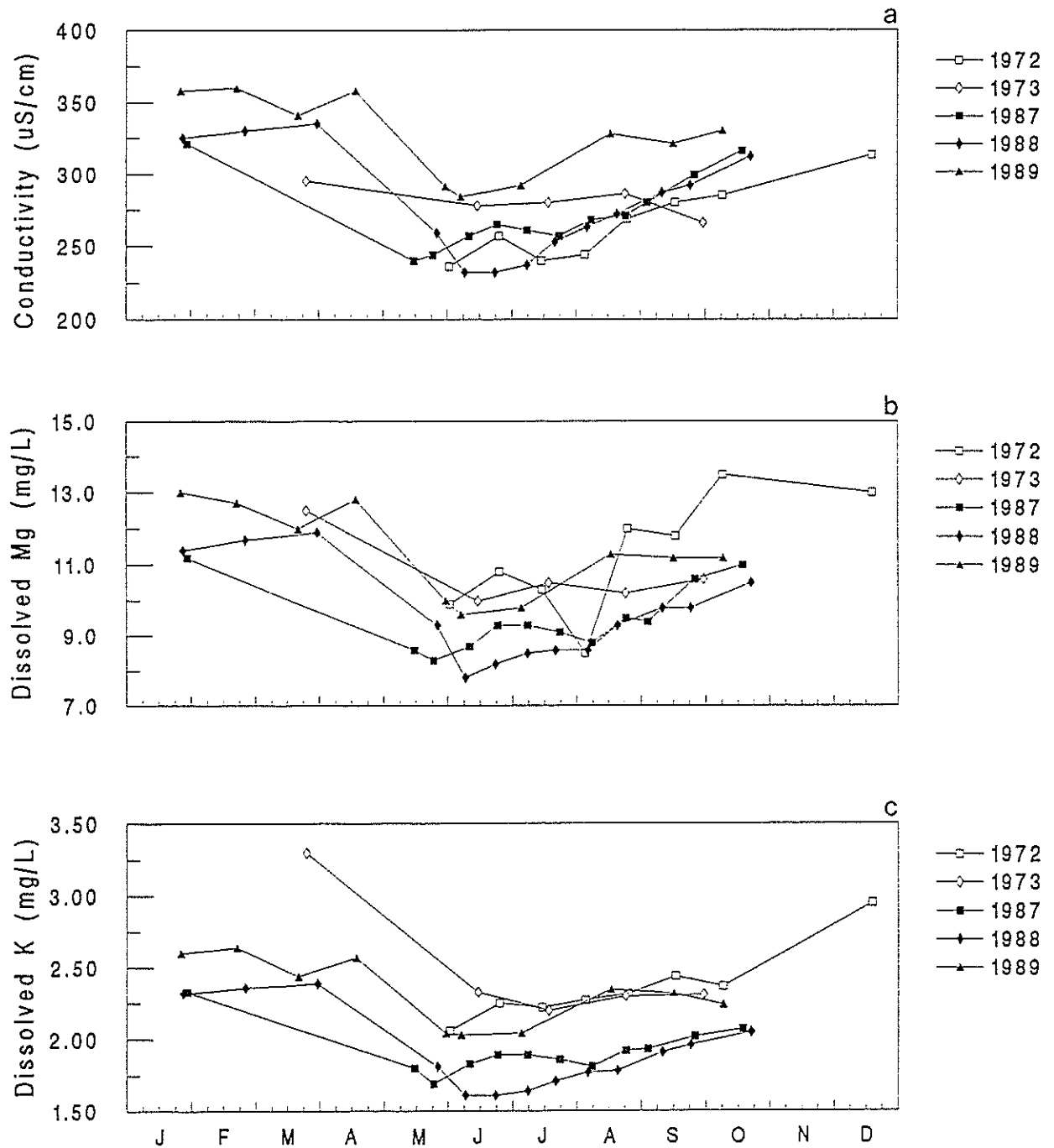


Figure 38.

Seasonal water quality trends in the Nelson River at the inlet to Split Lake. a. lab conductivity b. dissolved Mg c. dissolved K d. total hardness e. dissolved $\text{NO}_2\text{-NO}_3$ f. DIC.

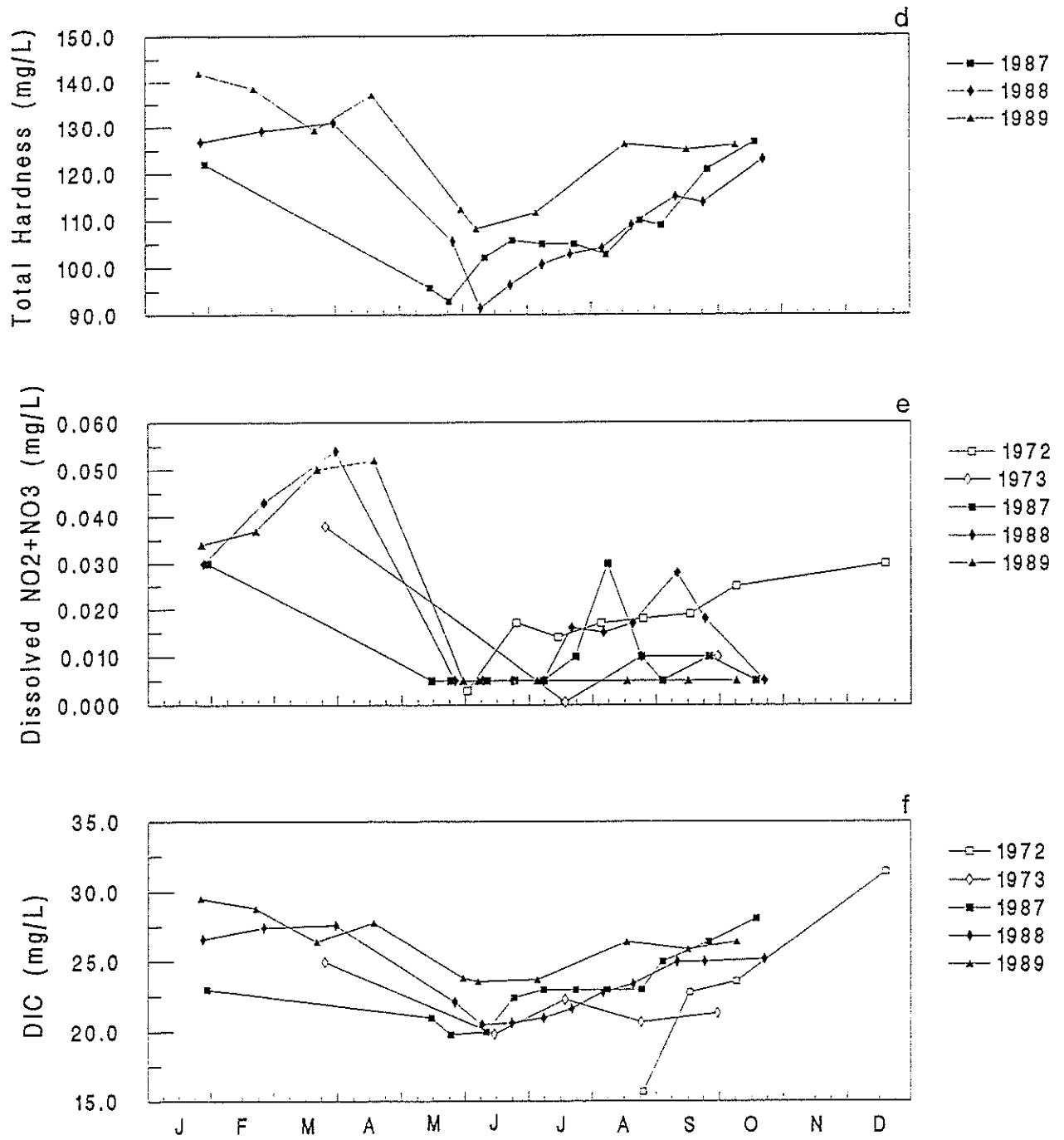


Figure 38. Continued.

known for marked seasonal variations due to its considerable solubility and minor biotic demand (Wetzel 1983). Conductivity was not correlated with discharge so its seasonality also cannot be explained by discharge variations.

The pre-development database for this site is poor for the examination of differences between the ice-free and ice-cover seasons. Only one winter measurement was taken from each of the Nelson River above Kelsey Dam (Mar. 1973) and the Nelson River below Sipiwesk Lake (Feb. 1973), while 2 were taken from the Nelson River below Kelsey Dam (Dec. 1972 and Mar. 1973). These limited data preclude a statistical analysis of changes in seasonal water quality patterns. Qualitative analysis of the data from the Nelson River below Kelsey Dam reveals higher conductivity, dissolved K and Mg, DIC, and NO_3 under ice-cover than during the open water season (Figure 38). Conductivity and dissolved K and Mg all were negatively correlated with discharge prior to development (Figure 36), with the lower winter discharges a possible explanation for the increases under ice cover. Again, the increased $\text{NO}_3\text{-N}$ under ice-cover probably was a product of reduced photosynthetic demand. The same process may explain the increase in DIC under ice. DIC is the inorganic carbon source for photosynthetic organisms (Wetzel 1983), but this explanation seems improbable given the relatively high buffering capacity of the Nelson River waters.

Although it appears the seasonal patterns of several variables were somewhat different during the FEMP study than prior to LWR, it would be inappropriate to attribute these changes to the

hydroelectric development. The low Nelson River flows during the FEMP study may have been responsible. The pre-development seasonal trends were based on a single winter and a minimum number of observations so it is not known if these apparent changes exceed the range of natural variation. The influence of LWR on seasonal trends in water quality should be clarified through additional monitoring during periods of higher flow.

3.5.2.4 Changes in Water Quality Since Lake Winnipeg Regulation

Significant changes occurred in 6 water quality variables between the 1972-1973 pre-development study period and the 1987-1989 FEMP study (Table 25). pH declined by 5%. NFR, dissolved K, TDP, and TDN also declined by 67, 15, 45, and 22%, respectively. The reactive Si concentration increased by 72%. A marginally significant decline ($p=0.075$) in chlorophyll a, by 51%, also occurred.

There is no clear relationship between these changes in water quality and LWR. The decline in pH was comparable to that observed in the natural rivers and is probably due to changes in analytical equipment or procedure. All of the other water quality changes appear to have been restricted to the reach between Sipiwesk and Split lakes. No significant changes in any of the nitrogen or phosphorus species, reactive Si, major ions (including dissolved K), turbidity, NFR, TDS, conductivity, suspended carbon, or colour on the Nelson River mainstem in Sipiwesk Lake were found in a comparison of baseline data from 1972-1973 with data for the period

Table 25. Evaluation of the significance of changes in selected water quality parameters in the Nelson River near the inlet of Split Lake following Lake Winnipeg regulation. X_{pre} denotes the mean for pre-development years, 1972 and 1973. X_{post} denotes the mean for the post-development years, 1987 through 1989. p is the probability of Student's t -statistic for the comparison.

[illegible] X_{pre} Station 37a

2 Station 38

Station 39

¹ X_{post} Discharge-standardized variables.

from 1987 to 1989 (Berger and Ramsey 1991). There were no significant changes in NFR, reactive Si, or major ion concentrations in Cross Lake during the same period (Berger and Ramsey 1991). Significant changes in several parameters were reported by Playle and Williamson (1986) for the Nelson River at Norway House, Cross Lake, and Sipiwesk Lake, although none can account for the observed changes in the Nelson River at the inlet to Split Lake.

Some of these changes might be explained by a natural evolution of limnological conditions in the Kelsey Generating Station forebay. Severe shoreline erosion became evident in the forebay after flooding (Water Resources Branch 1974), and the observed reduction in NFR might be an indication these shorelines are beginning to stabilize. Alternatively, erosion rates may have decreased temporarily as a result of the lower Nelson River flows. Between 1987 and 1989, Nelson River flows were at or near the lowest observed in the past 30 years (Figure 2). Based on the correlation between NFR and discharge in the Nelson River below Sipiwesk Lake for the 1972-1973 sampling period (Figure 34), mean NFR at the mean 1987-1989 discharge ($1271 \text{ m}^3 \text{ sec}^{-1}$) is estimated to be 5 mg L^{-1} , compared with 15 mg L^{-1} at the mean 1972-1973 discharge ($2305 \text{ m}^3 \text{ sec}^{-1}$). The observed reduction in NFR is identical (Table 25) and therefore appears to be completely accounted for by the reduction in discharge. Neither of these explanations can be ruled out with the data at hand. Additional sampling after the current drought conditions have subsided is required to properly determine

if LWR has had any long-term impact on water quality at this site.

3.5.3 Differences in Nelson and Burntwood River Water Quality

Water quality in the Nelson and Burntwood rivers differed significantly in many aspects in the 1987-1989 study period. Concentrations of most major ions and values of related variables (i.e., conductivity, alkalinity, DIC, hardness, and TDS) were significantly lower in the Burntwood River (Table 26). Most values were 45-60% lower, although Na, Cl, and SO₄ concentrations were 80-92% lower.

The Burntwood River was significantly more turbid, by about a factor of 4, and carried an almost 5-fold higher suspended sediment (NFR) concentration than the Nelson River (Table 26). Several metal species also occurred in significantly higher concentrations in the Burntwood River. Levels of extractable Al, Fe, and Mn were 3 to 4 times higher while total Ni and V levels were a factor of 2 higher. A similar trend was evident in the dissolved metals, although the difference was not as large and fewer metals were affected. The mean dissolved Al concentration was 50% higher in the Burntwood River and dissolved Fe was 2.3 times higher. In contrast to the metals, concentrations of dissolved As (a semi-metal) and dissolved Se (a metalloid) in the Burntwood River were 71 and 50% of levels in the Nelson River.

No clear trend was evident among the nutrient species (Table 26). Concentrations of reactive Si and total P were significantly higher, by 58 and 77%, respectively, in the Burntwood River while

Table 26. Comparison of current water quality in the Nelson and Burnwood rivers at the inlet to Split Lake. X_{nei} denotes the 1987-1989 mean for the Nelson River. X_{bwr} denotes the 1987-1989 mean for the Burnwood River. p is the probability of Student's t -statistic for the comparison.

Location	pH	Conductivity $\mu S\ cm^{-1}$	Turbidity NTU	Colour Relative	TDS $mg\ L^{-1}$	NPR $mg\ L^{-1}$	NPR Fixed $mg\ L^{-1}$	Tot Alk $mg\ L^{-1}$	Tot Hard $mg\ L^{-1}$	Diss Ca $mg\ L^{-1}$
Split Lake (inlet)	p	0.896	<0.001	0.006	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	X_{nei}	7.85	9.0	9.	154.7	5.0	3.3	103.0	115.0	29.3
	X_{bwr}	7.76	35.0 ¹	22.	64.7	24.3	21.3	56.7 ¹	56.7	15.9
	Diss Mg $mg\ L^{-1}$	Diss K $mg\ L^{-1}$	Diss Na $mg\ L^{-1}$	React Si $mg\ L^{-1}$	Diss Cl $mg\ L^{-1}$	Diss F $mg\ L^{-1}$	Diss SO ₄ $mg\ L^{-1}$	Ortho P $mg\ L^{-1}$	TDP $mg\ L^{-1}$	Tot P $mg\ L^{-1}$
	p	<0.001	<0.001	0.020	<0.001	0.982	<0.001	0.624	0.946	<0.001
	X_{nei}	10.2	13.8	1.24	15.3	0.10 ¹	21.1 ¹	0.004 ¹	0.011	0.022
	X_{bwr}	4.1	2.8	1.96	1.2	0.10	2.9	0.003	0.012	0.039
				TDN $mg\ L^{-1}$	Diss NO ₂ -NO ₃ $mg\ L^{-1}$	Susp N $mg\ L^{-1}$	DOC $mg\ L^{-1}$	TOC $mg\ L^{-1}$	DIC $mg\ L^{-1}$	Chlorophyll a $mg\ L^{-1}$
	p			0.053	0.998	0.438	0.062	0.587	<0.001	1.000
	X_{nei}			0.305	0.016	0.048	7.51	7.79	24.4	0.002
	X_{bwr}			0.248	0.019	0.063	6.78	7.41 ¹	13.4	0.002

Table 26. continued.

Location	Extr Al mg L ⁻¹	Diss Al mg L ⁻¹	Diss As mg L ⁻¹	Tot Cd mg L ⁻¹	Diss Cd mg L ⁻¹	Tot Co mg L ⁻¹	Tot Cu mg L ⁻¹	Diss Cu mg L ⁻¹	Extr Fe mg L ⁻¹	Diss Fe mg L ⁻¹
Split Lake										
(inlet)										
p	<0.001	0.006	<0.001	1.000	1.000	0.189	0.998	0.451	<0.001	0.001
X _{me1}	0.186	0.055	0.0007	0.0001 ¹	0.0001	0.0003	0.0037	0.0020	0.193	0.039
X _{pwF}	0.596	0.082	0.0002	0.0001	0.0001	0.0005	0.0047	0.0015	0.787	0.091
	Tot Pb mg L ⁻¹	Diss Pb mg L ⁻¹	Extr Mn mg L ⁻¹	Diss Mn mg L ⁻¹	Tot Ni mg L ⁻¹	Diss Ni mg L ⁻¹	Tot V mg L ⁻¹	Diss V mg L ⁻¹	Tot Zn mg L ⁻¹	Diss Zn mg L ⁻¹
216										
p	0.484	0.996	<0.001	0.946	0.003	0.675	0.002	0.991	0.45	0.978
X _{me1}	0.0006	0.0010	0.010	0.003	0.0011	0.0009	0.0008	0.0005	0.0025	0.0010
X _{pwF}	0.0011	0.0007	0.031	0.004	0.0023	0.0011	0.0016	0.0004	0.0046	0.0007 ¹

¹ Discharge-standardized variables.

TDN and DOC were marginally higher, by 23 and 11%, in the Nelson River. The higher total P concentration in the Burntwood River is entirely attributable to the suspended fraction as there were no differences in TDP or ortho PO₄ between the rivers.

3.5.4 Nelson River at Outlet of Split Lake

3.5.4.1 Current Water Quality

All physical and chemical water quality data gathered at this site during the FEMP study are summarized in Table 27. Water quality at the outlet of Split Lake shared a number of general similarities with sites upstream on the Nelson River. Mean turbidity and maximum colour exceeded the drinking water guidelines and the mean extractable Al and total Cu exceeded the guidelines for protection of aquatic life. However, water quality here differed from the other sites on the Nelson River in the magnitude by which these guidelines were exceeded. Turbidity exceeded the guideline by a factor of 2 on average at the other Nelson River sites, compared with a factor of 4 at the outlet of Split Lake. Mean extractable Al also was about a factor of 2 above the guideline value at the upstream sites compared with a factor of 3.2 at the outlet.

Other differences between the Split Lake outlet and sites on the upper Nelson River include an extractable Fe concentration exceeding the drinking water and aquatic life guidelines by about 30% on average and by as much as a factor of 2. Upstream, peak extractable Fe concentrations never were more than 50% over the

Table 27. Summary statistics for physical and chemical water quality in the Nelson River at the outlet of Split Lake, September 1986, to October, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.7	0.3	8.3	7.1
	Lab		7.80	0.21	8.15	7.17
Conductivity	Field	$\mu\text{S cm}^{-1}$	212	33	278	130
	Lab		227	30	296	180
Turbidity	Field	NTU	22 ⁺	12	88 ^{+,++}	13 ⁺
	Lab		18 ⁺	5	35 ⁺	11 ⁺
Colour	True	Relative	14	6	30 ⁺	5
TDS		mg L^{-1}	118	16	154	93
NFR	Total	mg L^{-1}	10	4	21	4
	Fixed	mg L^{-1}	8	3	19	3
Alkalinity	Total	mg L^{-1}	84.1	8.3	101.0	70.0
	Bicarbonate	mg L^{-1}	102.5	10.2	123.1	85.3
Hardness	Total	mg L^{-1}	91.4	10.3	114.6	75.4
	Non-carbonate	mg L^{-1}	7.3	3.5	18.4	0
Free CO ₂		mg L^{-1}	3.63	3.15	12.73	0
Phosphorus	Particulate	mg L^{-1}	0.017	0.006	0.030	<0.006
	Ortho PO ₄	mg L^{-1}	0.003	0.003	0.015	<0.003
	Total Dissolved	mg L^{-1}	0.011	0.004	0.020	0.003
	Total	mg L^{-1}	0.028	0.005	0.040	0.020
Nitrogen	Particulate	mg L^{-1}	0.052	0.023	0.110	<0.010
	Total Dissolved	mg L^{-1}	0.296	0.078	0.660	0.210
	NO ₂ -NO ₃	mg L^{-1}	0.030	0.074	0.404	<0.010
	Total NH ₃	mg L^{-1}	0.016	0.009	0.047	0.005
	Total	mg L^{-1}	0.35	0.08	0.71	0.24
Carbon	DOC	mg L^{-1}	7.27	0.45	8.70	6.50
	POC	mg L^{-1}	0.428	0.102	0.720	0.250
	TOC	mg L^{-1}	7.72	0.47	8.95	6.93
	DIC	mg L^{-1}	19.8	2.1	24.2	15.7
Chlorophyll-a		mg L^{-1}	0.003	0.002	0.007	<0.001
Ca	Dissolved	mg L^{-1}	24.0	2.4	29.0	20.3
Mg	Dissolved	mg L^{-1}	7.7	1.1	10.3	6.0
K	Dissolved	mg L^{-1}	1.63	0.22	2.20	1.22
Na	Dissolved	mg L^{-1}	9.2	2.1	14.5	5.8

Table 27. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L ⁻¹	1.54	0.53	2.60	0.69
Cl	Dissolved	mg L ⁻¹	9.2	2.8	16.7	4.8
F	Dissolved	mg L ⁻¹	0.10	0.02	0.15	0.08
SO ₄	Dissolved	mg L ⁻¹	13.5	3.4	21.3	8.6
Al	Extractable	mg L ⁻¹	0.315 ⁺⁺⁺	0.126	0.651 ⁺⁺⁺	<0.100
	Dissolved	mg L ⁻¹	<0.100	0.025	0.138	<0.100
As	Dissolved	mg L ⁻¹	0.0005	0.0002	0.0009	0.0002
Ba	Total	mg L ⁻¹	<0.08		<0.08	<0.08
Cd	Total	mg L ⁻¹	<0.0001	0.0001	0.0003	<0.0001
	Dissolved	mg L ⁻¹	<0.0001	0.0001	0.0003	<0.0001
Co	Total	mg L ⁻¹	<0.0005	0.0001	0.0007	<0.0005
Cu	Total	mg L ⁻¹	0.0050 ⁺⁺⁺	0.0075	0.0308 ⁺⁺⁺	0.0010
	Dissolved	mg L ⁻¹	0.0020	0.0010	0.0042	0.0011
Fe	Extractable	mg L ⁻¹	0.389 ^{+,+++}	0.110	0.672 ^{+,+++}	0.129
	Dissolved	mg L ⁻¹	0.070	0.055	0.355	0.023
Pb	Total	mg L ⁻¹	0.0009	0.0011	0.0066 ⁺⁺⁺	<0.0007
	Dissolved	mg L ⁻¹	0.0012	0.0015	0.0049	<0.0007
Mn	Extractable	mg L ⁻¹	0.015	0.004	0.025	0.003
	Dissolved	mg L ⁻¹	0.004	0.004	0.021	<0.002
Ni	Total	mg L ⁻¹	0.0015	0.0004	0.0025	0.0006
	Dissolved	mg L ⁻¹	0.0011	0.0004	0.0021	0.0006
Se	Dissolved	mg L ⁻¹	0.0001	0.0001	0.0005	0
V	Total	mg L ⁻¹	0.0011	0.0005	0.0020	<0.0005
	Dissolved	mg L ⁻¹	<0.0005	0.0003	0.0009	<0.0005
Zn	Total	mg L ⁻¹	0.0040	0.0049	0.0215	0.0006
	Dissolved	mg L ⁻¹	0.0013	0.0018	0.0055	<0.0003
Discharge	Daily	m ³ sec ⁻¹	2641	577	3817	1530

+ Exceeds limit for drinking water.

++ Exceeds limit for recreational use.

+++ Exceeds limit for protection of aquatic life.

water quality objectives. Similarly, total Pb levels never exceeded the water quality guidelines at upstream sites yet peak concentrations at the Split Lake outlet were as much as a factor of 3 above the aquatic life guideline. Although concentrations of some metals were higher at the Split Lake outlet than at upstream sites on the Nelson River, the extractable Mn concentration generally was lower, never approaching the drinking water guideline. These higher Al and Fe levels appear to be more characteristic of the Churchill and Burntwood River drainages than of the Nelson River (Section 3.5.3).

On the basis of the mean concentrations of total phosphorus, total nitrogen, and chlorophyll a, Split Lake can be classified as mesotrophic according to the model of Vollenweider (1979).

3.5.4.2 Water Chemistry - Water Discharge Relationships

3.5.4.2.1 Pre-development

Pre-development data for the outlet of Split Lake comes from 2 sites, one near the Split Lake community and the other near the outlet. Water quality - discharge relationships at these sites were examined separately.

Only four variables were correlated with discharge at the site near the Split Lake community, all negatively, in the 1972-1974 period. These included conductivity (lab), total alkalinity, total hardness, and dissolved K (Figure 39). The correlations accounted for 37 to 65% of the variance in the dependent variables.

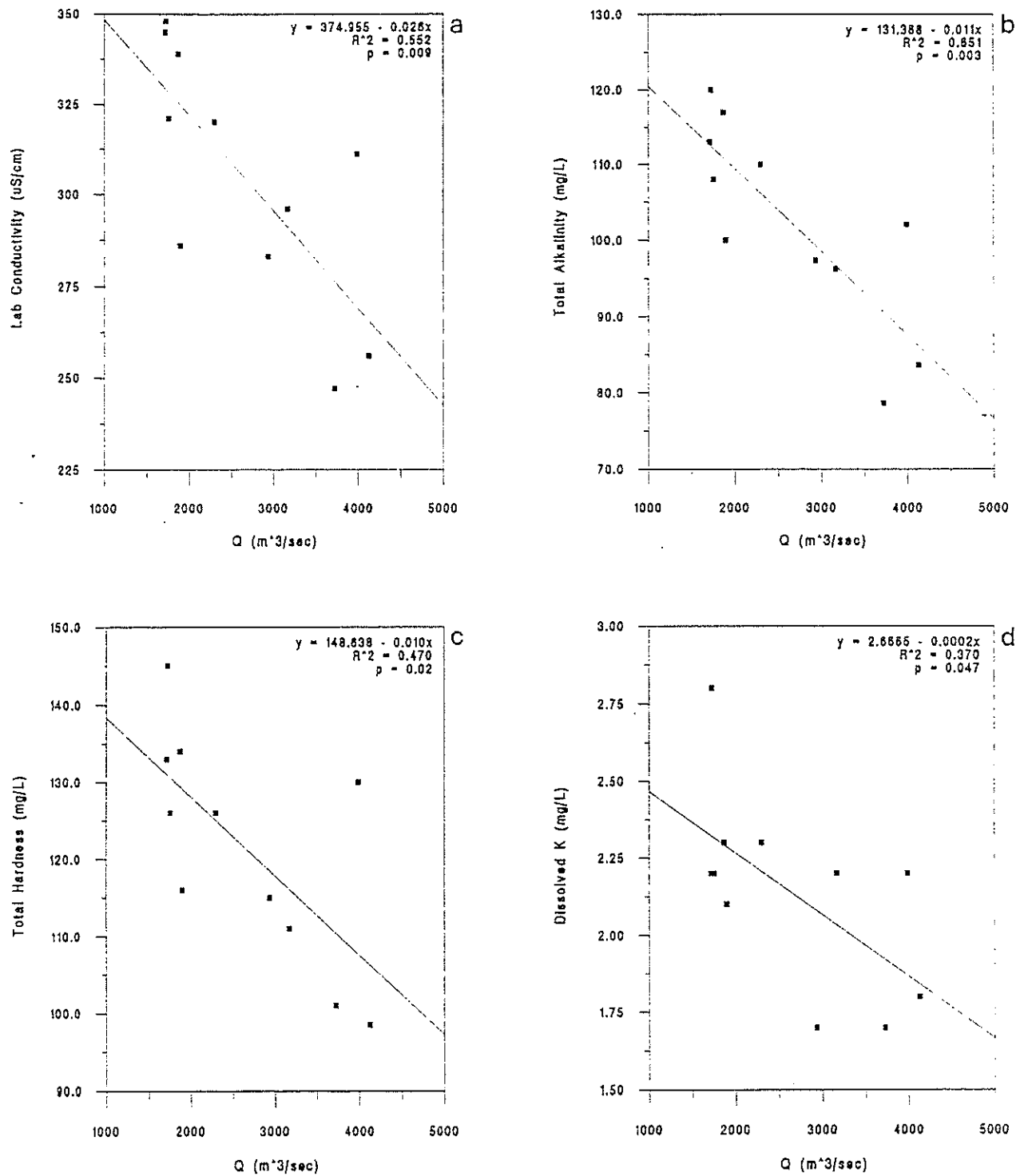


Figure 39. Water chemistry - water discharge relationships in Split Lake at Split Lake, 1972-1974.
a. lab conductivity b. total alkalinity c. total hardness d. dissolved K.

Six variables were negatively correlated with discharge at the Split Lake outlet in the 1972-1973 period (Figure 40). Two of these variables, conductivity (field) and dissolved K, were measured both here and at the site near the community and were correlated with discharge at both sites. Consistent with the correlation between hardness and discharge at the site near the community, Mg, a major component of hardness (APHA 1985), was correlated with discharge at the outlet. Other correlates with discharge at the outlet site included TDS and dissolved Na and SO_4 . The regressions on discharge accounted for 66 to 92% of the variance in the dependent variables.

3.5.4.2.2 Current

Only 3 variables were significantly correlated with discharge in the 1987-1989 study period. All correlations were positive and included DOC, TOC, and dissolved Fe (Figure 41). The correlation between discharge and TOC was purely a result of the correlation with DOC. On average, 94% of TOC occurred as DOC (Table 27) resulting in significant collinearity of these variables. None of these relationships was particularly strong, as appears to be characteristic of sites influenced by the CRD, with the regression on discharge accounting 13 to 26% of the variance in the dependent variable.

The discharge relationship for dissolved Fe should not be used for predictive purposes, as it was clearly driven by the single high value (Figure 41c), unless the relationship can be verified by further sampling. The discharge relationships for TOC and DOC

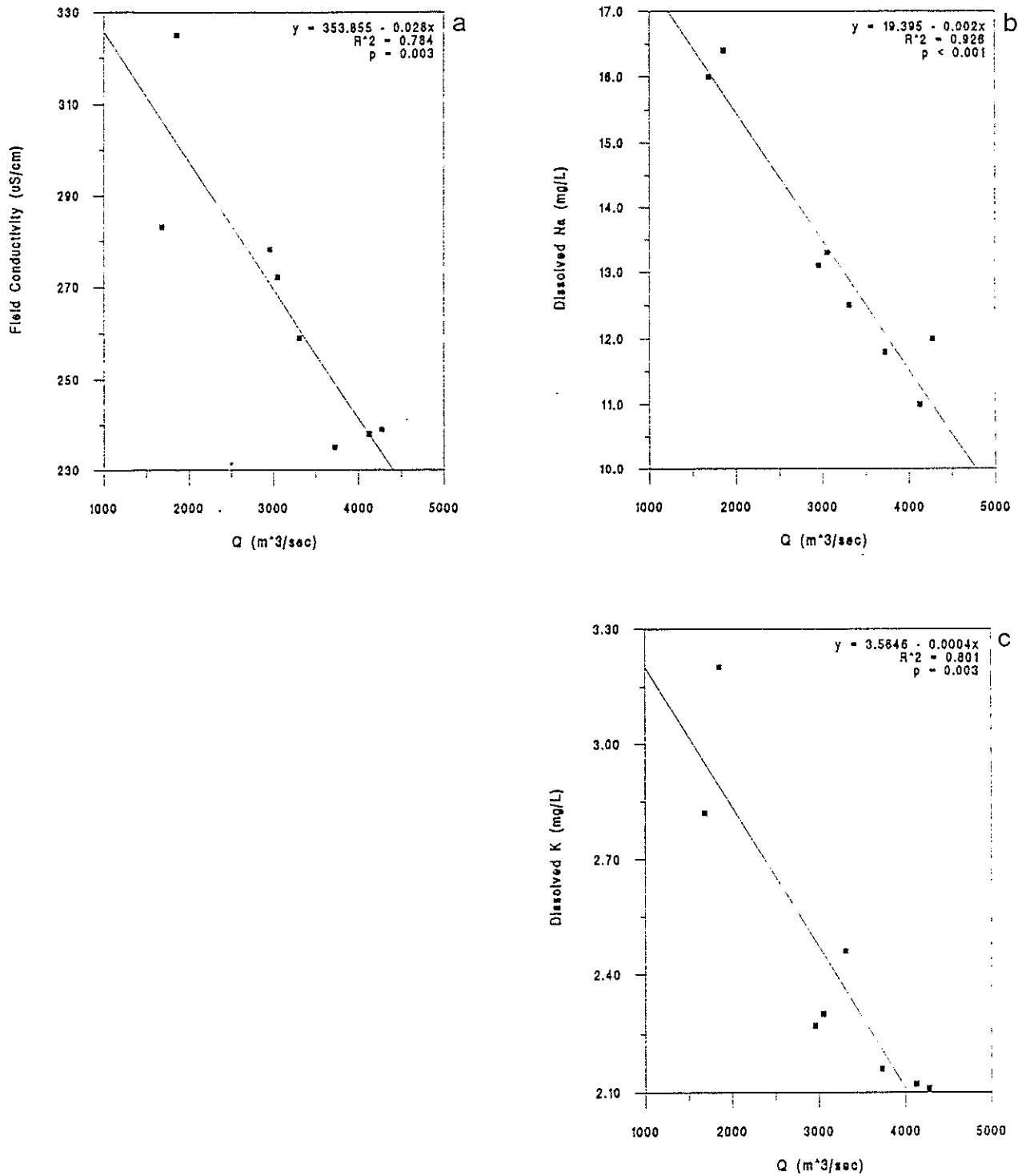


Figure 40. Water chemistry - water discharge relationships near the outlet of Split Lake, 1972-1973.
 a. field conductivity b. dissolved Na c. dissolved K d. dissolved Mg e. dissolved SO_4
 f. TDS.

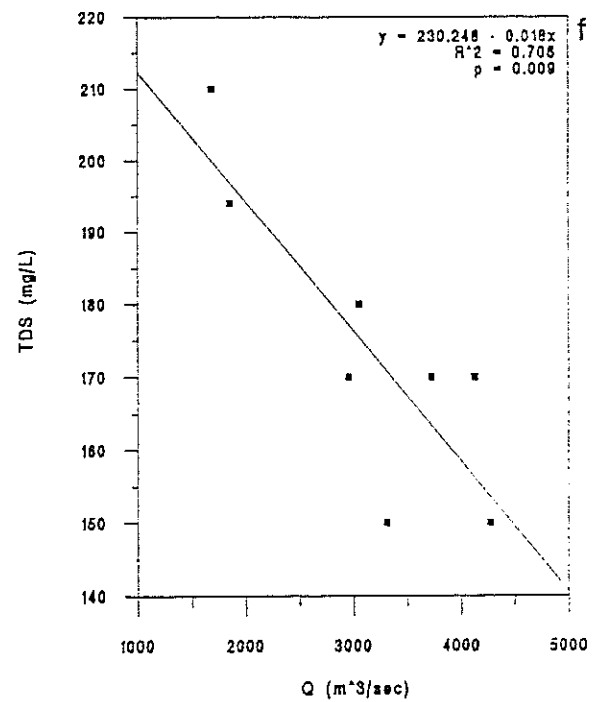
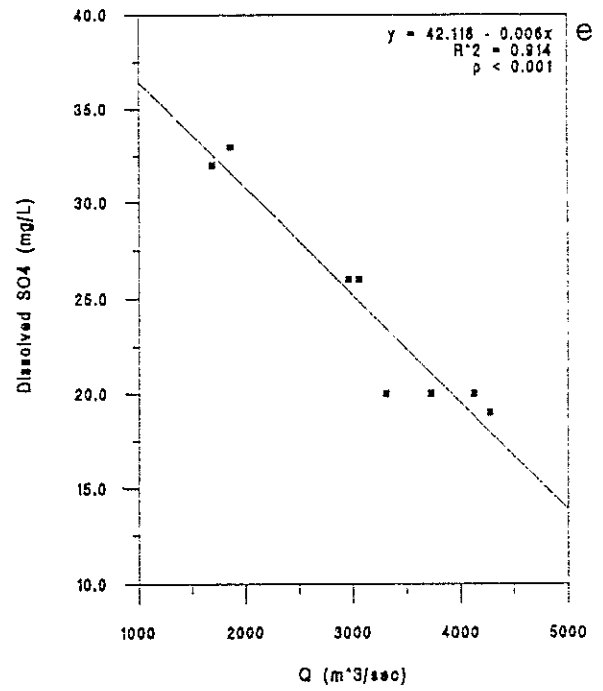
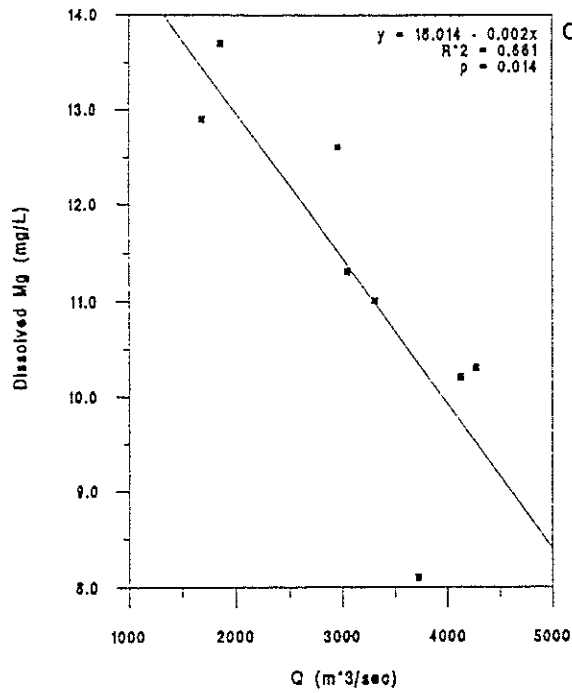


Figure 40. Continued.

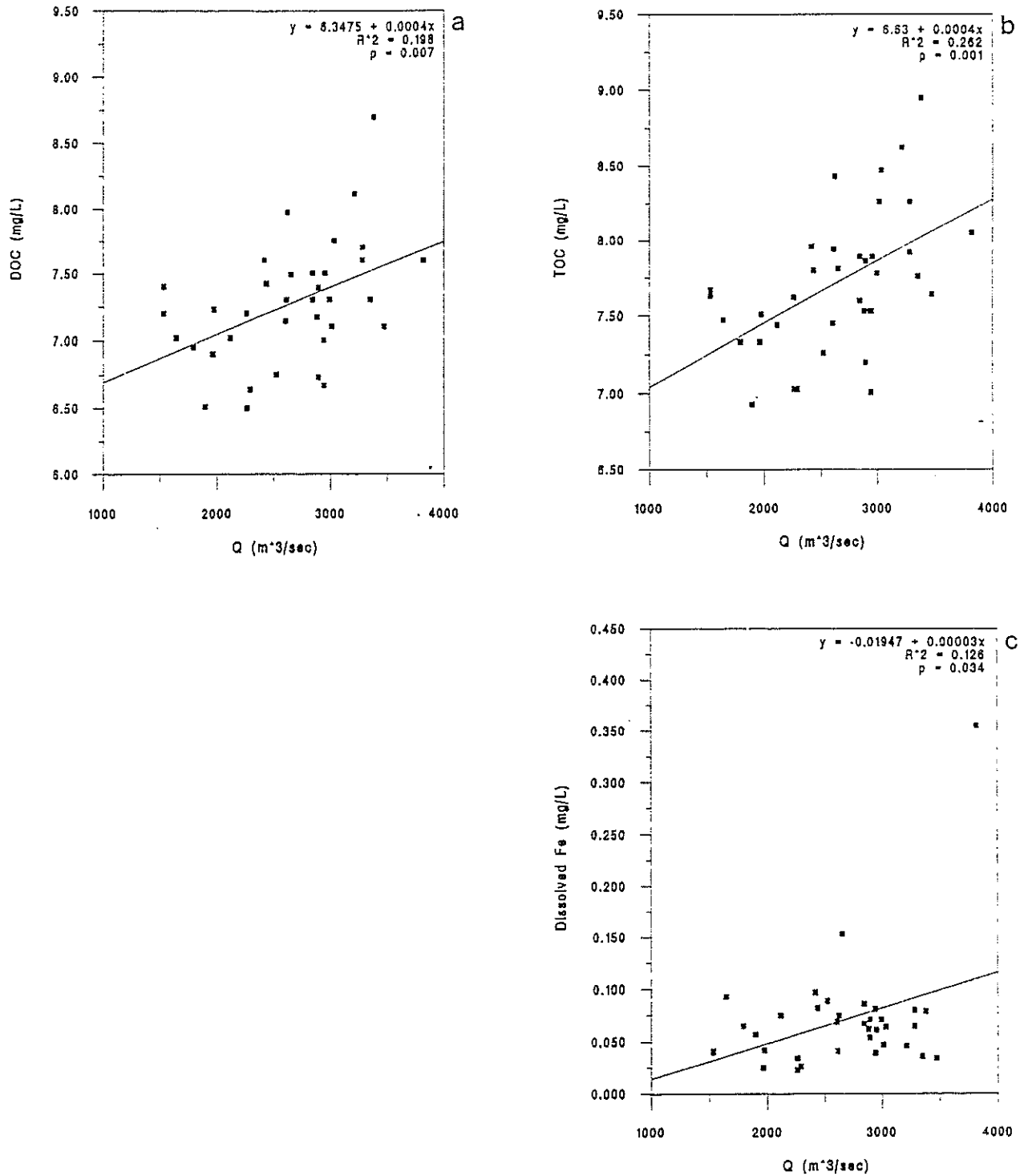


Figure 41. Water chemistry - water discharge relationships in the Nelson River at the outlet of Split Lake, 1987-1989. a. DOC b. TOC c. dissolved Fe.

appear to have some predictive ability as they are consistent with the general linear trends in the data.

Water chemistry - discharge relationships in the vicinity of the Split Lake outlet appear to have changed considerably between the 1972-1973 pre-development period and the 1987-1989 FEMP study. None of the variables which were correlated with discharge prior to development was correlated with discharge in the current study. All post-development relationships had a positive slope, compared to the negative slopes which prevailed before development. This shift in the nature of the discharge relationship, from negative to positive, occurred at other sites which have been affected by CRD. This change cannot conclusively be attributed to CRD, however, because other potential causes cannot be ruled out with the available data. Other potential causes include the low Nelson River flows during the FEMP study (Section 1.4), and the reversal of the hydrologic regime in the lake as a result of LWR. Discharge records for the pre-development period (1961-1975) indicate monthly mean discharge at the Kelsey Dam was $1852\text{--}2105 \text{ m}^3 \text{ sec}^{-1}$ under ice cover (Dec.- Mar.) compared with $2749\text{--}3202 \text{ m}^3 \text{ sec}^{-1}$ during the ice-free season (May-Oct.; Water Survey of Canada 1990). Since 1978, monthly mean flows through Split Lake (sum of Nelson R. and Burntwood R. flows) have been higher during the ice-cover season ($2872\text{--}3089 \text{ m}^3 \text{ sec}^{-1}$, Dec. to Mar.) than during the ice-free season ($2402\text{--}2755 \text{ m}^3 \text{ sec}^{-1}$, May to Oct., Water Survey of Canada 1990). It should be possible to separate the effects of low flows due to drought from those due to CRD and LWR when the drought conditions subside. In

the meantime, the water quality - water discharge relationships based on the FEMP study should not be used for prediction or management purposes until the cause of the changes can be clarified.

3.5.4.3 Seasonal Differences

There appears to have been a change in the patterns of seasonal variation of several variables in Split Lake since CRD and LWR, although there is insufficient pre-development data on which to base a statistical analysis. Prior to development, conductivity, alkalinity (total), hardness (total), and dissolved Na, K, and SO_4 all were slightly higher under ice cover than during the open water season (Figure 42). These increases under ice cover appear to be the combined result of lower river flows in winter and the negative correlations of these variables with discharge.

In contrast to the pre-development period, few variables occurred in different concentrations under ice cover than in the open water season during the FEMP study, and none was common to the pre- and post-development periods. Those which showed some evidence of seasonal variation after development included TDN, dissolved $\text{NO}_2\text{-NO}_3$, and total N (which was comprised predominantly of dissolved species) and dissolved Zn, all of which occurred in higher concentrations under ice-cover (Figure 42). The occurrence of increased $\text{NO}_2\text{-NO}_3$ concentrations in winter, and consequently of TDN and total N, is generally consistent with the higher levels in both major inflows during the ice-cover period (Figures 33 and 38).

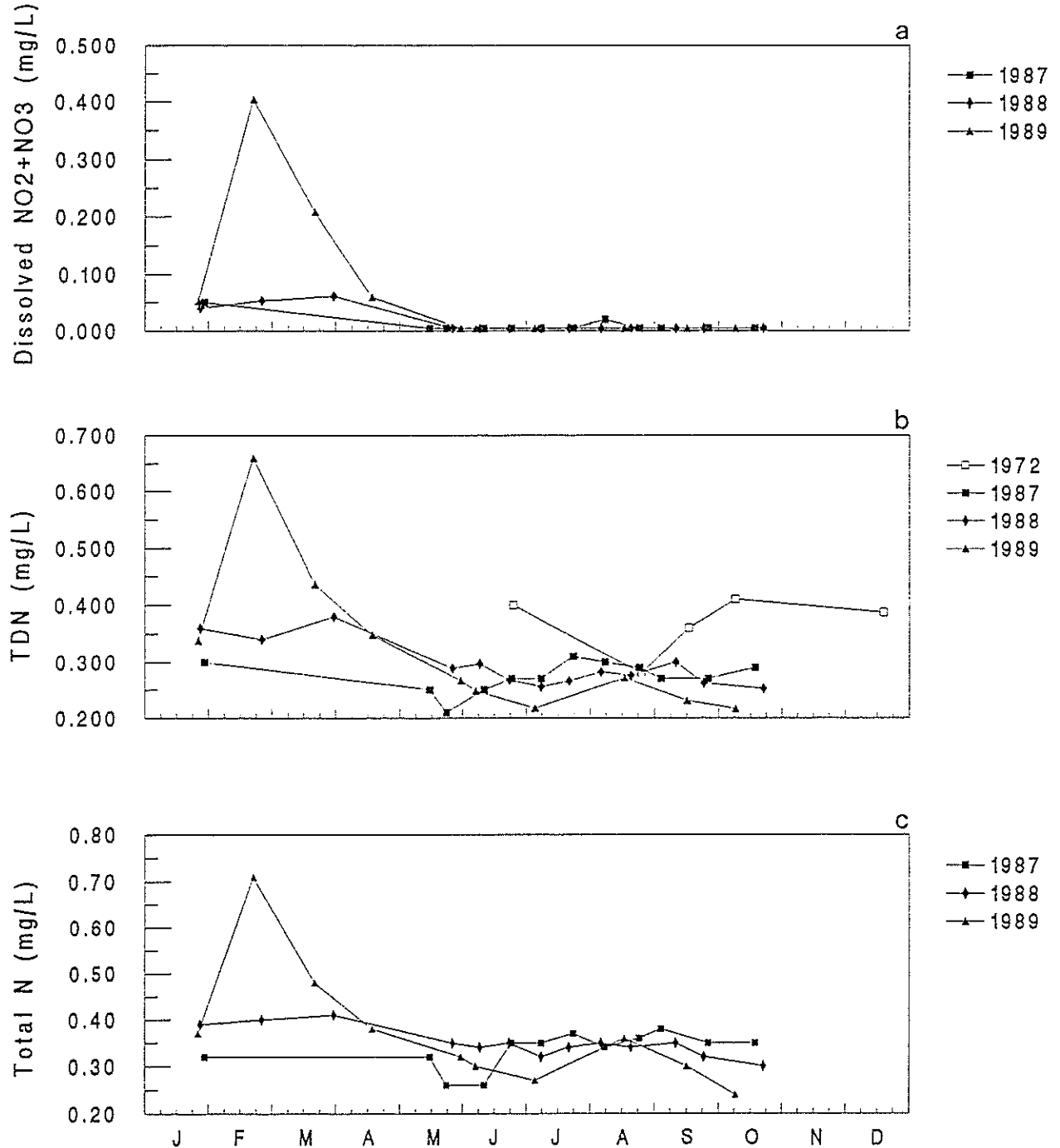


Figure 42. Seasonal water quality trends in the Nelson River at the outlet of Split Lake. a. dissolved $\text{NO}_2\text{-NO}_3$ b. TDN c. total N d. colour e. dissolved SO_4 f. dissolved Zn g. total hardness h. lab conductivity i. total alkalinity j. dissolved Ca k. dissolved K l. dissolved Na.

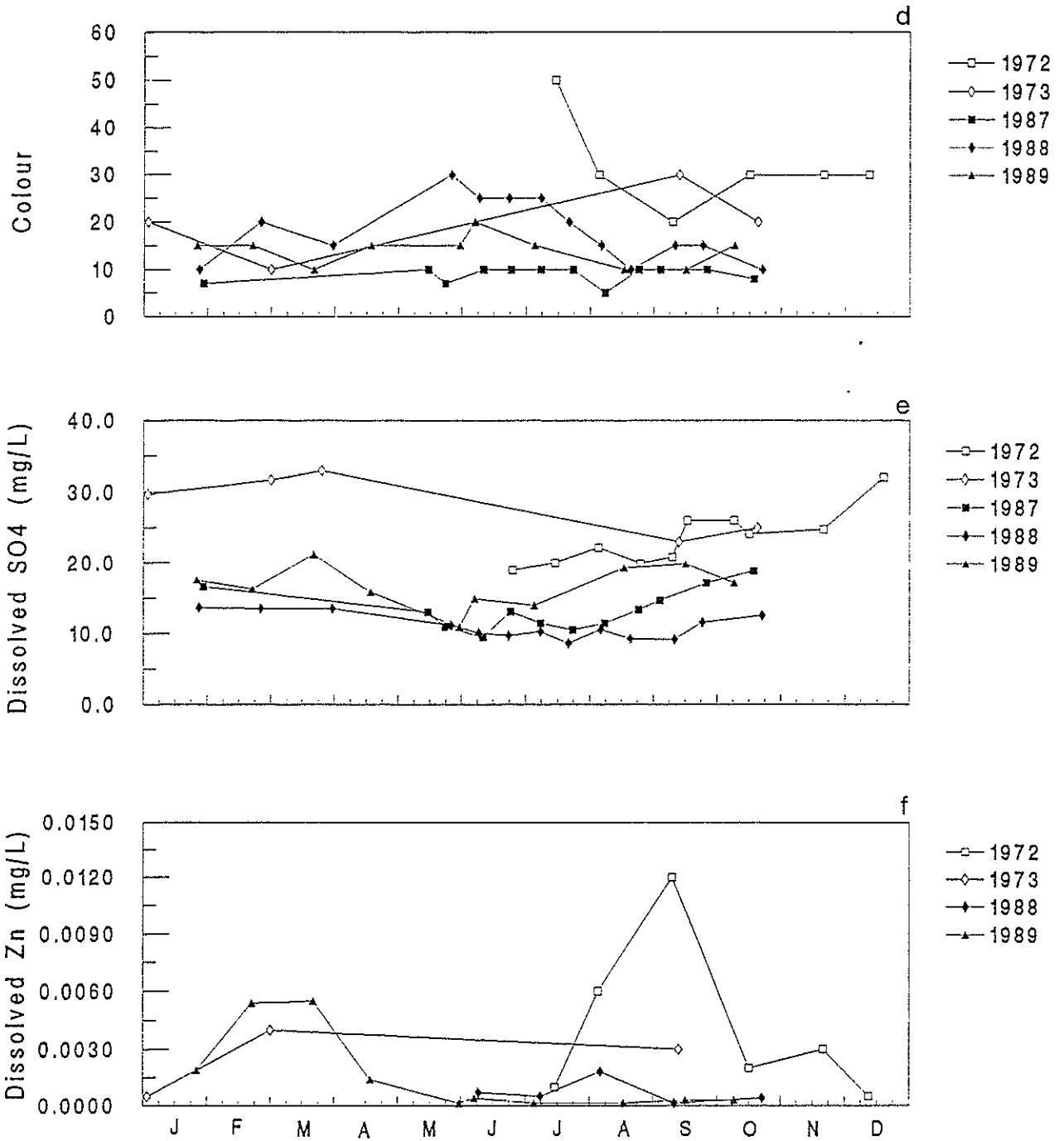


Figure 42. Continued.

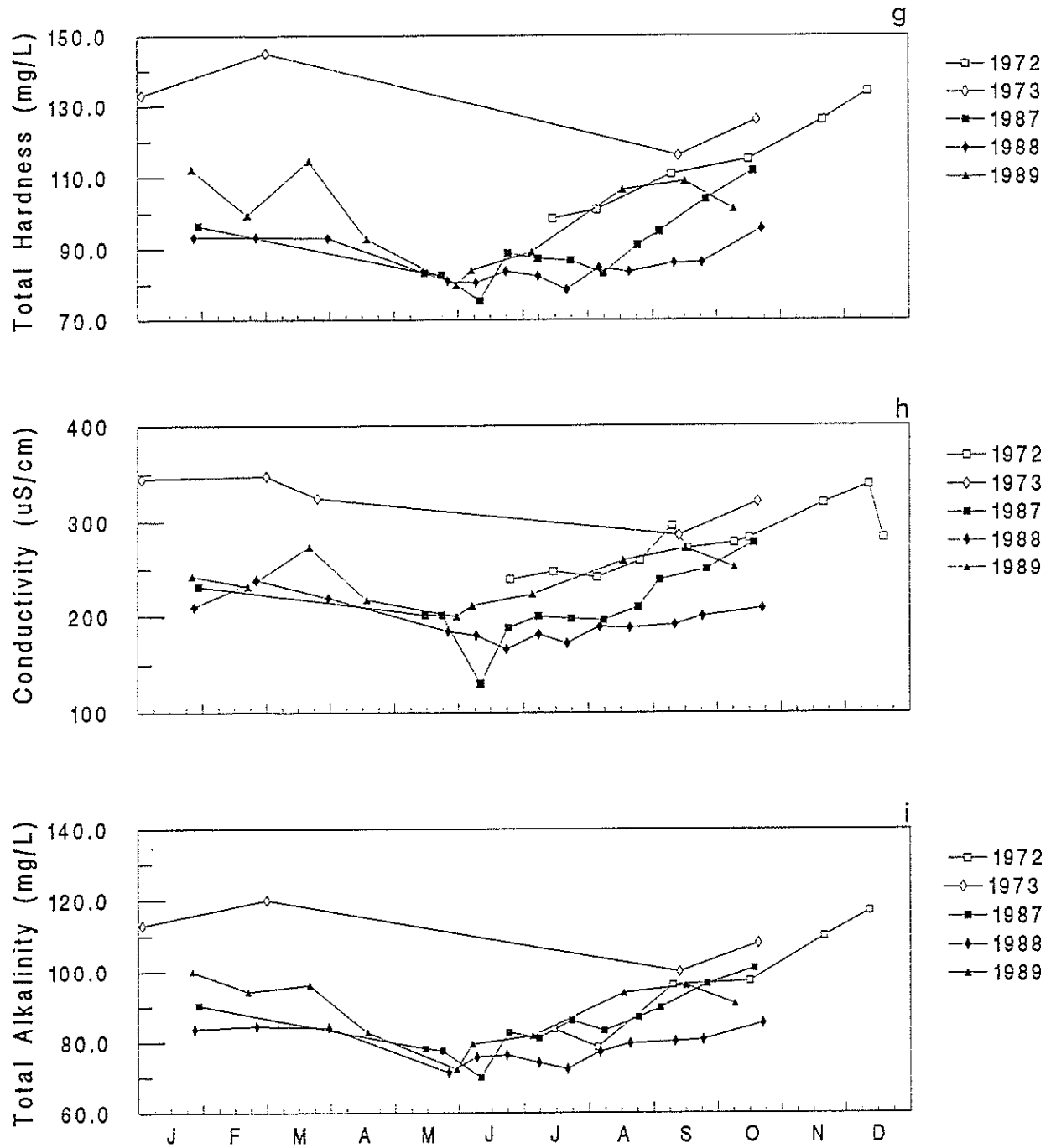


Figure 42. Continued.

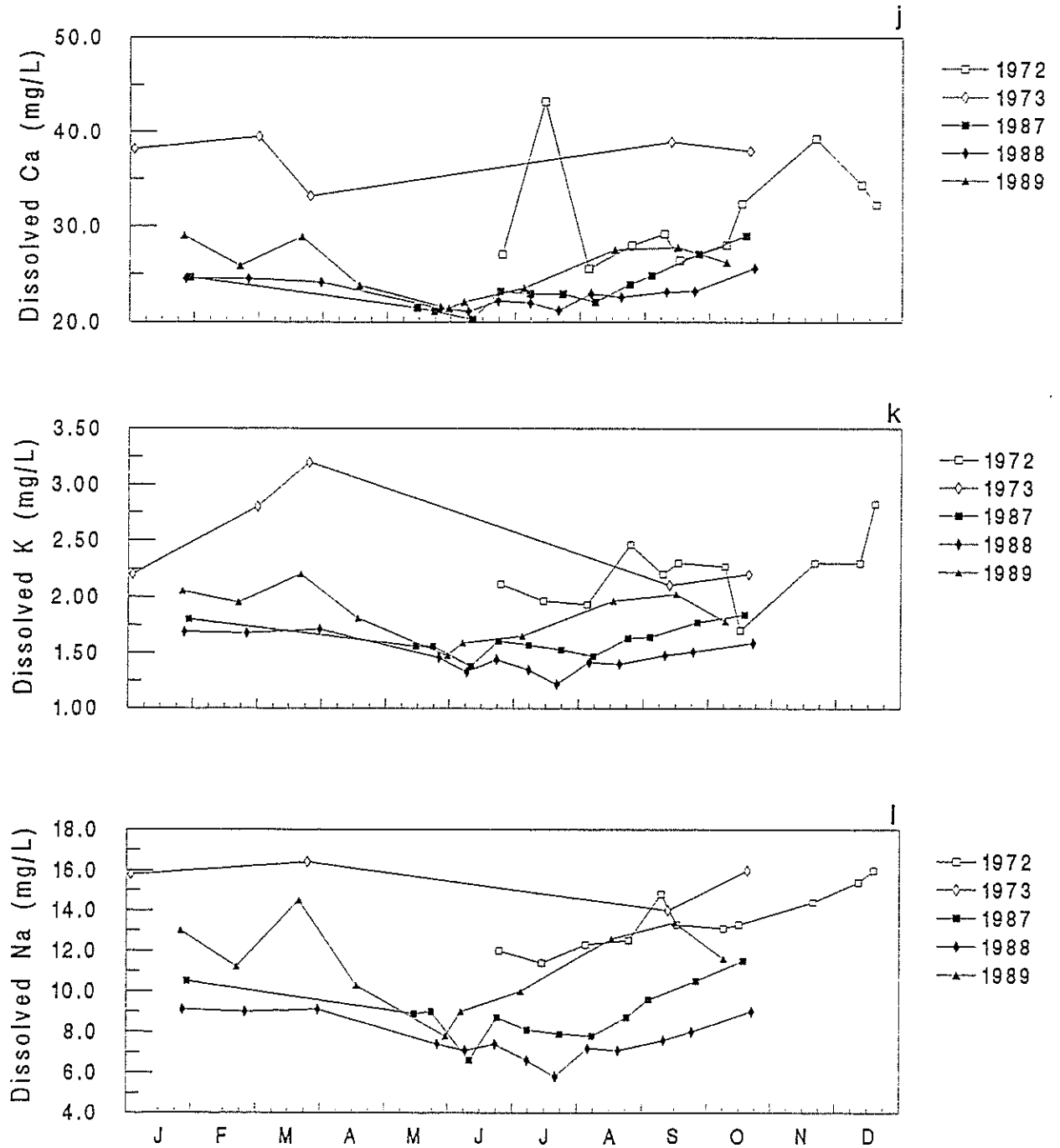


Figure 42. Continued.

Generation of $\text{NO}_2\text{-NO}_3$ within Split Lake also seems to play a role, as indicated by the much higher concentrations in February and March, 1989, than in the inflows. The winter maximum in dissolved Zn is not explained by higher loading, as no seasonal differences were evident in either major inflow at this time.

The absence of seasonal differences in several variables during the FEMP study could be due to one or more of the several changes in the water regime or the changes in water quality which have occurred since the pre-development study period. As noted in Section 1.4, Nelson River flows during the FEMP study period were the lowest in the 30 year record and this may have disrupted the normal discharge relationships and the accompanying seasonal trends. Another possible cause is the reversal of the natural seasonal flow regime due to operation of the LWR project. Peak flows now occur during the winter rather than during spring run-off and the annual range in monthly mean flow also has decreased since CRD and LWR came into operation. Prior to development, the difference between the highest and lowest monthly mean flows was $1350 \text{ m}^3 \text{ sec}^{-1}$, but this difference dropped to $687 \text{ m}^3 \text{ sec}^{-1}$ after development (Water Survey of Canada 1990). These factors may have been sufficient to eliminate the rather small seasonal differences which prevailed prior to development. Finally, most of the changes in water quality caused by CRD involved the same variables in which seasonal trends were evident prior to development but not during the FEMP study, suggesting a common cause. Clearly, there is no way to identify the most probable cause of these changes in seasonality

with the available data. This assessment will require additional sampling during periods of higher Nelson River flows to determine the role of the naturally low flows in these changes.

3.5.4.4 Changes in Water Quality Since Lake Winnipeg Regulation and Churchill River Diversion

3.5.4.4.1 Pre/Post Comparison

Significant changes in 10 water quality variables occurred in the outlet basin of Split Lake between the 1972-1973 pre-development studies and the 1987-1989 FEMP study. pH declined by 3% while conductivity, hardness (total), alkalinity (total), and dissolved Ca, Mg, K, Na, Cl, and SO₄ declined by 26, 24, 19, 29, 29, 27, 35, 38, and 47%, respectively (Table 28).

Berger and Ramsey (1991) also examined changes in water quality related to Churchill River diversion at 2 sites on Split Lake. Data for the June to September period collected by Manitoba Natural Resources, Fisheries Branch, in 1987, 1988, and 1989 were compared with pre-development data from 1972 and 1973 contained in the LWCNR study reports (Cleugh 1974; Hecky and Harper 1974; Morelli 1974), the same pre-development database employed in the present analysis. The sites examined were the outlet of Split Lake, near the sampling site for the FEMP study, and a site in the plume of the Burntwood River about 5 km from the inlet to Split Lake (Figure 5). Substantial reductions in major ion concentrations following CRD were found at the site near the Burntwood River inlet, with levels of Ca, Mg, Na, K, Cl, and SO₄ lower by 32, 48,

Table 28. Evaluation of the significance of changes in selected water quality parameters in the Nelson River near the outlet of Split Lake following Churchill River diversion and Lake Winnipeg regulation. X_{pre} denotes the mean for pre-development years, 1972 and 1973. X_{post} denotes the mean for the post-development years, 1987 through 1989. p is the probability of Student's t -statistic for the comparison.

[illegible]

66, 46, 83, and 71%, respectively, due to increased Burntwood River flows. No other significant changes were detected at this site although a marginally significant reduction in $\text{NO}_3\text{-N}$ and a marginal increase in TDP were indicated. Similar changes were found at the site near the Split Lake outlet, although the decreases were not as large as near the Burntwood River inlet due to additional mixing with the Nelson River. Levels of Na, Cl, SO_4 , and colour were significantly lower, by 24, 36, 32, and 55%, respectively. It is not known if a similar decrease in colour occurred near the Burntwood River inlet due to the lack of pre-development data for this location. Mg was not included in the analysis of changes at the outlet station, again due to a shortage of pre-development data.

The lack of a significant change in Ca and K and the finding of a significant reduction in colour near the Split Lake outlet in the Berger and Ramsey (1991) study contrast with the significant reductions in Ca and K and lack of a significant change in colour in the present study. These discrepancies appear to be related to the restriction of the Berger and Ramsey (1991) study to the summer (June-September) season. Prior to CRD, Ca and K concentrations in Split Lake near the outlet tended to be higher in autumn and winter than in summer, with summer concentrations similar to those currently observed year-round (Figure 42). Thus, the reductions in the annual mean Ca and K levels detected in the present analysis were specifically due to lower levels during autumn and winter. In contrast, the reported reduction in colour appears to have been due

to lower values in summer, but not necessarily in other seasons (Figure 42).

Berger and Ramsey (1991) also found water transparency at both sites, as indicated by Secchi disk visibility, was the same between 1987 and 1989 as in 1972-1973. Transparency was lower near the mouth of the Burntwood River than at other locations in the lake throughout the 1987-1989 study period, but this was the case prior to CRD as well. This absence of a change in transparency is consistent with the finding of no significant change in turbidity in the present study.

Similar changes to those found in this study and reported by Berger and Ramsey (1991) also were reported for several variables by Playle and Williamson (1986), with conductivity, hardness, alkalinity, and dissolved Ca, SO_4 , and Mg declining by 19, 12, 27, 11, 36, and 18%, respectively. No significant change in colour was detected. In contrast to the present study, they did not find any significant change in the concentrations of dissolved K, Na, or Cl, and reported a significant 88% increase in turbidity.

The disparate conclusions of the Playle and Williamson (1986) study can be explained by a number of factors. Their different conclusion regarding a change in turbidity appears to have been due to the exclusion of some pre-development data. In particular, they did not include data for 1972 and 1973 reported in Morelli (1974). These data were not included in the analysis by Playle and Williamson (1986) because of a concern for a lack of data comparability indicated by different units of measurement. The data

in Morelli (1974) were reported in Jackson Turbidity Units (JTU), compared with Nephthelometric Turbidity Units (NTU) in all later studies. Investigation revealed this concern was unfounded. Ed Sorba (Ward Lab, Manitoba Environment, pers. comm.) indicated the Morelli (1974) data were analyzed following the same procedure as later data. There have been changes in the type of nephthelometer used since 1974, but the method has otherwise remained unchanged. Although the Morelli (1974) turbidity values were reported in JTU, they were measured with a nephthelometer. On the basis of this information, the Morelli (1974) data were included in our analysis. When these data are added to the data considered by Playle and Williamson and reanalysed, the t-test also indicates no significant change ($p=0.773$; Figure 43).

All three studies concluded there has been no change in NFR at this site as a result of LWR and CRD. This appears consistent with the absence of significant change in turbidity or water transparency. Although NFR in the Burntwood River is significantly higher than in the Nelson River, the majority of this higher suspended sediment load appears to be lost through sedimentation shortly after entering Split Lake. Although all studies indicate that any impact of CRD on suspended sediment levels in Split Lake is not widespread, the zone of impact has not yet been adequately defined. Future studies should focus on the immediate area of delta formation.

Playle and Williamson (1986) did not detect a significant change in dissolved K because their data were the product of an

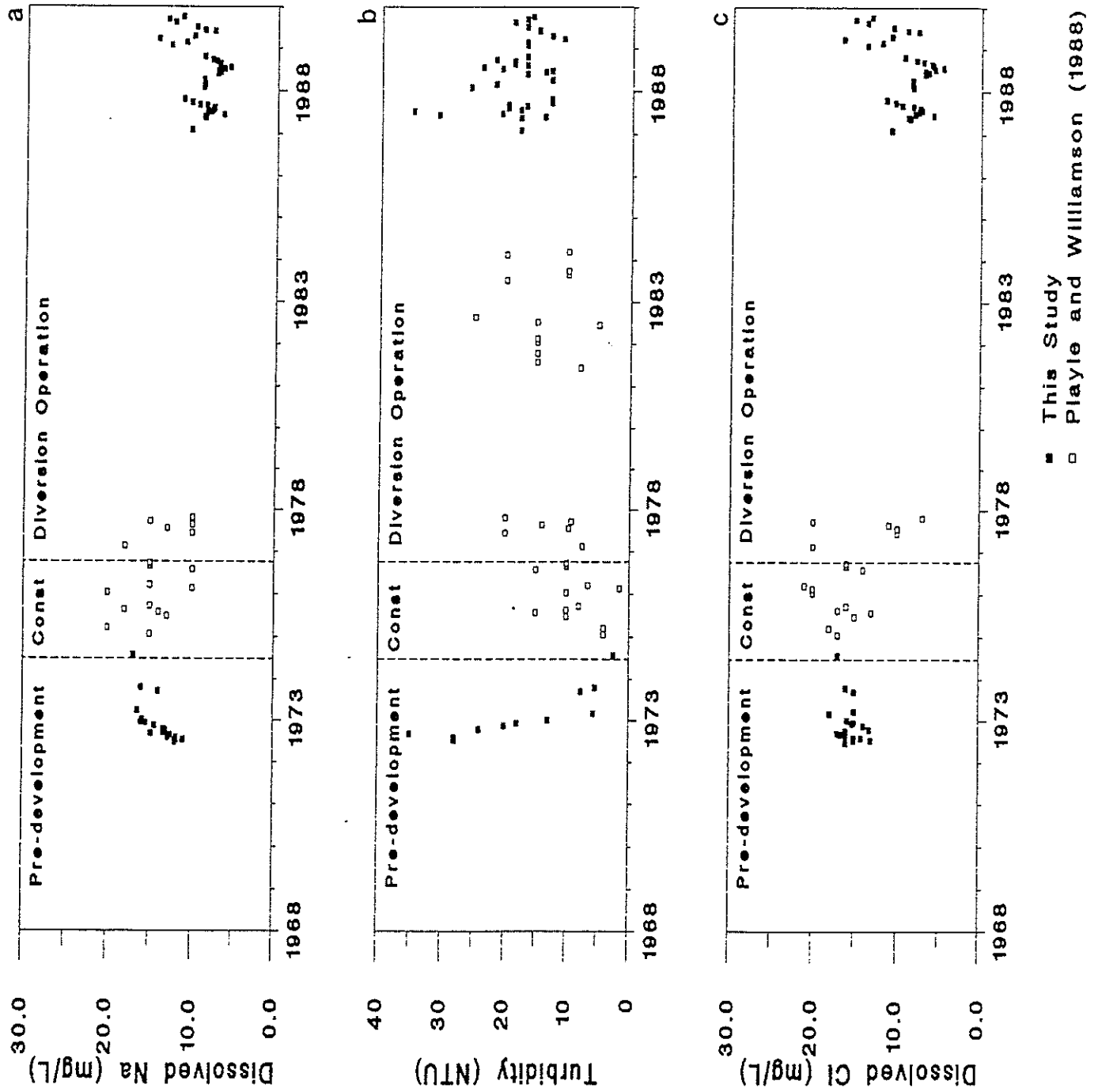


Figure 43. Temporal trends in (a) dissolved Na, (b) turbidity, and (c) dissolved Cl in the Nelson River at the outlet to Split Lake based on data analyzed by Blouin and Williamson (1988) and the data collected by this study.

analytical method with inadequate sensitivity for the ambient K levels. The method had a detection limit of 2 mg L^{-1} (Playle 1986) which would not permit discrimination of the observed decrease from 2.26 to 1.65 mg L^{-1} (Table 28).

The lack of significant changes in Na and Cl in the Playle and Williamson (1986) study appears to have been a result of the short data records available to them. In particular, the post-development data records for these variables only extended from July, 1976, to October, 1977. Considering that full diversion flows were not achieved until September, 1977, their period of record was clearly too short to examine the effect of operational flows. In contrast, the present analysis is perhaps more indicative of the worst case impact of CRD on Na and Cl levels in Split Lake due to the unusually high relative contribution of the CRD to total Split Lake inflows during the study period.

Most of the changes in Split Lake water quality are solely attributable to the increased flows of softer water from the Churchill River drainage basin into the lake as a result of Churchill River diversion. Conductivity, hardness, and the concentrations of all major ions in the Burntwood River are significantly lower than in the Nelson River (Table 26), and no changes in conductivity or in dissolved Ca, Na, Cl, or SO_4 occurred in the Nelson River above Split Lake over the same period (Section 3.5.2.3). About 60% of the reduction in dissolved K near the outlet of Split Lake can be attributed to the lower concentration in the inflowing Nelson River, with the balance of the reduction due to

the increased Burntwood River inflows. Only the small decrease in pH at the Split Lake outlet can be entirely attributed to changes in the Nelson River.

The decrease in colour during the summer is not directly attributable to CRD. Colour in the Burntwood River at the inlet to Split Lake was more than twice as high as in the Nelson River (Table 26). Insufficient pre-development colour data are available for the Nelson River to determine if there has been a reduction in this source.

In summary, CRD has caused demonstrably significant reductions in hardness, alkalinity, and major ion concentrations in Split Lake through the introduction of softer Churchill River drainage water flowing to the lake. Although the quantity of sediment delivered to Split Lake via the Burntwood River has increased since CRD (Northwest Hydraulic Consultants 1988), this has not had a significant effect on turbidity, suspended sediment concentration (NFR), or water transparency in the majority of the lake due to sediment deposition in Split Lake near the mouth of the Burntwood River.

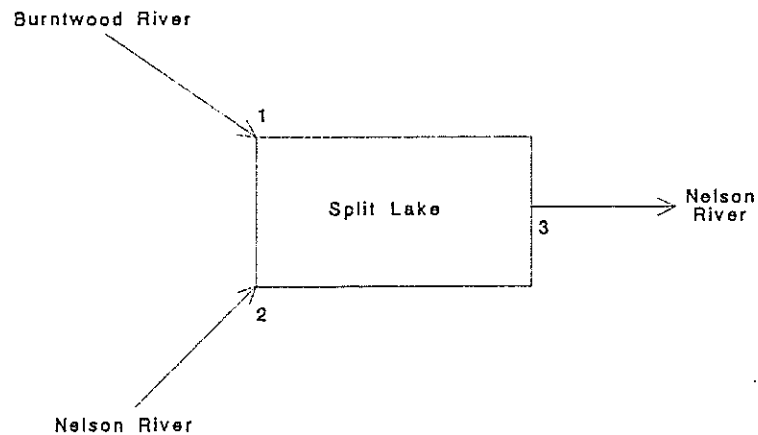
3.5.4.4.2 Modelled Changes

The dominance of Split Lake hydrology by the Nelson River prior to diversion, and now jointly by the Nelson and Burntwood rivers, makes it possible to predict changes in water quality which may have occurred near the outlet of Split Lake in the absence of pre-development data for this site. This modelling approach is

based on the assumption that, prior to diversion, the concentration of a parameter at the outlet of Split Lake would have been approximately the same as in the Nelson River at the inlet to Split Lake. Therefore, the finding of a significant difference in the concentration of a parameter between the Nelson River inflow and the Split Lake outflow after diversion, in the absence of a pre/post comparison to the contrary, is evidence of a change due to CRD. This approach was judged to be reasonable on the basis of the dominance of total inflows to Split Lake by the Nelson River prior to diversion, the short residence time for the lake (approximately 6 days based on the lake volume, $1208.44 \times 10^6 \text{ m}^3$ from Cherepak, 1990, and mean annual outflow volume from 1987 to 1989 of $71 \text{ km}^3 \text{ year}^{-1}$, G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB, pers. comm.) and the lack of a significant increase in shoreline erosion following diversion, as indicated by no change in turbidity, transparency, or NFR at the outlet.

For this approach to be valid, it must be shown that the present concentration of a variable in the Split Lake outflow is adequately predicted from the concentrations and discharges in the two major lake inflows, and that the pre-development concentration in the outflow was adequately predicted by the concentration in the Nelson River at the inflow to Split Lake. The present (post-diversion) concentration of a variable in the Split lake outflow was found to be readily predicted using the numerical model described in Figure 44. In this model, the only inflows considered are the Nelson and Burntwood rivers. Losses to sediments and

A. Simplified Model of Split Lake Hydrology



B. Model Derivation

1. Pre-development

- a. Basic model, $Q_3C_3 = Q_2C_2 + Q_1C_1$
- b. But, $Q_1 = 0$, relative to Q_2 , which makes $Q_3C_3 = Q_2C_2$
- c. Finally, $Q_3 = Q_2$, Therefore $C_3 = C_2$

2. Post-development

- a. Basic model, $Q_3C_3 = Q_2C_2 + Q_1C_1$
- b. Let $Q_1 + Q_2 = Q_3$, $Q_1 = 0.34 * Q_3$, and $Q_2 = 0.66 * Q_3$ - based on discharge data
- c. Then $Q_3C_3 = (0.66 Q_3)C_2 + (0.34Q_3)C_1$
- d. Which reduces to $C_3 = 0.66C_2 + 0.34C_1$

Where Q_n = discharge at site 1, 2, or 3

C_n = concentration of substance (mg L^{-1}) at site 1, 2, or 3

Q_nC_n = flux of substance at site 1, 2, or 3

Figure 44. (A) Simplified model of Split Lake hydrology, and (B) derivation of numerical models used to predict the concentration of a substance in the Nelson River near the outflow of Split Lake based on the concentrations in the inflowing Burntwood and Nelson rivers and the relative contribution of these rivers to discharge at the outflow. Model predictions are compared with observed values in Table 29.

contributions from local inflows and direct precipitation were considered to be unimportant. The model was tested by comparing the predicted and observed annual mean concentrations in the Split Lake outflow for the 1987-1989 study period for all variables which had changed significantly between the pre- and post-diversion periods. The model predicted the concentration in the outflow with a mean (± 1 s.d.) error of 5 (± 3)% (Table 29), clearly indicating the importance of inflow volume and concentration to outflow chemistry.

The annual mean values for these variables at the Split Lake outlet prior to development were predicted equally well using a model which considered only the mean values in the Nelson River at the inlet to Split lake (Figure 44). The model predicted the mean value in the outflow with a mean (± 1 s.d.) error of 7 (± 5)% (Table 29). Therefore, the finding of a significant difference in concentration between the Nelson River inflow and the outlet of Split Lake during the 1987-1989 study period should be indicative of a change caused by diversion.

Among the many variables for which there are no pre-development data, only 3, dissolved As and extractable Fe and Mn, occurred in significantly different concentrations at the Split Lake outlet than at the Nelson River inlet during the 1987-1989 study period (Table 30). The excellent agreement between the observed and predicted outflow concentrations of dissolved As and extractable Fe (Table 29) indicates a direct effect of the Burntwood River on the concentrations of these variables in Split Lake. The increase in extractable Fe attributable to CRD was

Table 29. Comparison of observed and predicted values for selected water quality parameters in the Nelson River at the outlet of Split Lake following Churchill River diversion. Pre- and post-development predicted values have been calculated as in Figure 44. BWR denotes the Burntwood River. NR denotes the Nelson River. All values in mg L^{-1} , except where noted.

	Post-Development					Pre-Development				
	Inflow Value		Outflow Value			Outflow Value			Outflow Value	
	BWR	NR	Predicted	Observed	% Error	Predicted	Observed	% Error	Predicted	Observed
Conductivity ($\mu\text{S cm}^{-1}$)	123	293	235	228	3	273	307	10		
TDS	65	155	124	119	4	177	171	4		
Alkalinity (Total)	57	103	87	84	4		115			
Hardness (Total)	57	115	95	91	4		120			
DIC	13.4	24.4	20.7	19.9	4	22.2	22.7	2		
Dissolved Ca	15.9	29.0	24.5	24.1	2	28.5	33.9	16		
Dissolved Mg	4.1	10.2	8.1	7.7	5	11.1	10.9	2		
Dissolved K	1.05	2.06	1.72	1.65	4	2.43	2.26	7		
Dissolved Na	2.8	13.8	10.1	9.3	9	14.08	14.30	2		
Dissolved Cl	1.2	15.3	10.5	9.5	11	17.23	15.3	13		
Dissolved SO_4	2.94	21.1	14.9	13.7	9	24.5	26.0	6		
Dissolved As	0.0002	0.0007	0.0005	0.0005	0					
Extractable Fe	0.787	0.193	0.395	0.387	2					
Extractable Mn	0.031	0.010	0.018	0.015	20					

Table 30. Comparison of current water quality in the Nelson River at the inlet and outlet to Split Lake. X_{in} denotes the 1987-1989 mean for the Nelson River at the inlet to Split Lake. X_{out} denotes the 1987-1989 mean for the Nelson River at the outlet to Split Lake. p is the probability of Student's t-statistic for the comparison.

Location	pH	Conductivity $\mu S\ cm^{-1}$	Turbidity NTU	Colour Relative	TDS $mg\ L^{-1}$	NFR $mg\ L^{-1}$	NFR Fixed $mg\ L^{-1}$	Tot Alk $mg\ L^{-1}$	Tot Hard $mg\ L^{-1}$	Diss Ca $mg\ L^{-1}$
Nelson R.										
p	0.989	0.005	0.001	0.487	0.007	0.200	0.142	0.001	0.002	0.001
X_{in}	7.85	293	9	9	154.7	5.0	3.3	103.0	115.0	29.3
X_{out}	7.81	228	18	14	118.7	9.7	8.0	84.3	91.7	24.1
	Diss Mg $mg\ L^{-1}$	Diss K $mg\ L^{-1}$	Diss Na $mg\ L^{-1}$	React Si $mg\ L^{-1}$	Diss Cl $mg\ L^{-1}$	Diss F $mg\ L^{-1}$	Diss SO_4 $mg\ L^{-1}$	Ortho P $mg\ L^{-1}$	TDP $mg\ L^{-1}$	Tot P $mg\ L^{-1}$
p	0.004	0.034	0.025	0.625	0.040	1.000	0.034	0.197	0.999	0.092
X_{in}	10.2	2.06	13.8	1.24	15.3	0.10 ¹	21.1 ¹	0.004 ¹	0.011	0.022
X_{out}	7.7	1.65	9.3	1.51	9.5	0.10	13.7	0.003	0.011	0.027
				TDN $mg\ L^{-1}$	Diss NO_2-NO_3 $mg\ L^{-1}$	Susp N $mg\ L^{-1}$	DOC $mg\ L^{-1}$	TOC $mg\ L^{-1}$	DIC $mg\ L^{-1}$	Chlorophyll a $mg\ L^{-1}$
p				0.989	0.915	0.969	0.818	0.997	0.002	0.886
X_{in}				0.305	0.016	0.048	7.51	7.79	24.4	0.002
X_{out}				0.297	0.034	0.053	7.26 ¹	7.71 ¹	19.9	0.003

Table 30. continued.

Location	Extr Al mg L ⁻¹	Diss Al mg L ⁻¹	Diss As mg L ⁻¹	Tot Cd mg L ⁻¹	Diss Cd mg L ⁻¹	Tot Co mg L ⁻¹	Tot Cu mg L ⁻¹	Diss Cu mg L ⁻¹	Extr Fe mg L ⁻¹	Diss Fe mg L ⁻¹
Nelson R.										
P	0.188	0.941	0.001	1.000	1.000	0.997	0.994	1.000	0.001	0.110
X _{in}	0.186	0.055	0.0007	0.0001 ¹	0.0001 ¹	0.0003	0.0037	0.0020	0.193	0.039
X _{out}	0.308	0.059	0.0005	0.0001	0.0001	0.0003	0.0051	0.0020	0.387	0.064 ¹
	Tot Pb mg L ⁻¹	Diss Pb mg L ⁻¹	Extr Mn mg L ⁻¹	Diss Mn mg L ⁻¹	Tot Ni mg L ⁻¹	Diss Ni mg L ⁻¹	Tot V mg L ⁻¹	Diss V mg L ⁻¹	Tot Zn mg L ⁻¹	Diss Zn mg L ⁻¹
P	0.554	1.000	0.051	0.946	0.550	0.675	0.283	1.000	0.668	0.990
X _{in}	0.0006	0.0010	0.010	0.003	0.0011	0.0009	0.0008	0.0005	0.0025	0.0010
X _{out}	0.0010	0.0010	0.015	0.004	0.0015	0.0011	0.0011	0.0005	0.0041	0.0012

¹ Variables standardized with discharge.

sufficient to cause the mean concentration at the outlet to exceed the guidelines for human consumption and protection of aquatic life by about 30%. Without CRD, the mean concentration at the outlet would have been about 35% below these guidelines given the present concentration in the inflowing Nelson River (Table 29). Based on the post-development model described in Figure 44, A 50% reduction in Burntwood River flows would be required for the mean extractable Fe concentration in Split Lake to comply with the guidelines.

Extractable Mn appears to behave less conservatively than Fe, with a 20% higher predicted concentration in the outlet than was observed (Table 29). Perhaps this 20% error was due to the sedimentation of Mn attached to particulates in the delta which is forming near the Burntwood River inlet. Although the impact is smaller than predicted, the higher concentration in the outflow than in the Nelson River is still attributable to the Burntwood River.

Sedimentation of particulate-bound metals would also explain the few significant differences in metals concentrations in the Nelson River between the inlet and the outlet to Split Lake (Table 30) despite the higher concentrations of several metals in the Burntwood River (Table 26). Site-specific investigations in the area of sediment deposition would be required to confirm this hypothesis.

3.5.5 Adverse Effects

The only adverse effect of Churchill-Nelson hydroelectric development on physical and chemical water quality in Split Lake has been the approximately two-fold increase in the extractable Fe concentration caused by CRD. This impact was most severe during the 1987-1989 sampling period, due to the high relative contribution of the CRD to Split Lake inflows. The elevation in extractable Fe is a product of the higher concentration in the Burntwood River and the increased contribution of this river to total Split Lake inflow due to diversion. As a result, the mean extractable Fe concentration in Split Lake exceeded the Canadian guidelines for drinking water and protection of aquatic life in the 1987-1989 study period. Although high extractable Fe is common in the northern drainages (i.e., Churchill, Burntwood, Footprint, and Aiken rivers; Sections 3.1 and 3.3), it is not characteristic of sites on the Nelson River.

It is unlikely that the elevated Fe concentration will have any toxic effects on aquatic life. The Canadian guideline (0.3 mg L^{-1}) is much more stringent than those of the Province of Manitoba (1.0 mg L^{-1} ; Williamson 1988b) or the United States (1.0 mg L^{-1} ; U.S. EPA 1976). The mean and maximum concentrations are well below the EC_{50} for fathead minnow eggs (1.5 mg L^{-1}), the most sensitive fish species tested which has been tested (CCREM 1987). Some invertebrates may be affected by concentrations as low as 0.32 mg L^{-1} , which is near the average for the 1987-1989 study period (CCREM 1987). However, the toxicity tests were conducted using dissolved

Fe in the absence of binding agents like suspended sediments. Most of the Fe in Split lake is in particulate form. The mean dissolved Fe concentration at the Split Lake outlet was just 0.07 mg L^{-1} , well below the level shown to have toxic effects on any species.

The effect of the increase in extractable Fe on the quality of drinking water for the Split Lake community is probably insignificant. The naturally high turbidity in the lake requires that the water be treated for turbidity removal prior to distribution. Surface water treatment plants operated for turbidity removal also remove Fe from raw waters (CCREM 1987). Most (>80%) of the Fe in Split Lake water is in particulate form, so turbidity removal should be effective in reducing Fe below the drinking water guideline.

The impact of the increased Fe concentration on those who use water directly from the lake is probably insignificant as well. Toxicity is not a significant concern, as the guideline level was set for aesthetic rather than health considerations (Health and Welfare Canada 1989). Potential impacts of excessive Fe include staining of laundry and plumbing, unpleasant taste and colour, and the promotion of undesirable biological growths within a waterworks (Health and Welfare Canada 1980). However, these impacts have not occurred upstream on the Burntwood River, where extractable Fe concentrations are naturally higher than in Split Lake. It should be possible to avoid any impact of the elevated Fe in Split Lake water by allowing particulates to settle out before use resulting in the removal of much of the Fe.

The perception among Split Lake community residents is that the lake has become more turbid since hydroelectric development on the Nelson and Churchill rivers began (Seidl et al. 1990). The appearance of a delta at the mouth of the Burntwood River since the diversion began operation is an obvious indication of the large quantities of suspended sediment entering the lake. However, this study and Berger and Ramsey (1991) have shown there has been no significant change in turbidity, water transparency, or NFR beyond the area of sediment deposition. The immediate area of the growing delta may be a site of severe adverse impact, but the nature and extent of any impacts in the delta area remain to be documented.

An aspect of the environmental effects of hydroelectric development in northern Manitoba which remains to be examined is the effect of construction and operation of the Kelsey Generating Station on water quality and biota in Split Lake and other downstream waters. This study has shown CRD did not cause an elevation in levels of turbidity or NFR or a reduction in water transparency, yet the prevailing belief among Split Lake residents is that the water is more turbid now than in the past. It is possible that the perceived higher turbidity may have been caused by the Kelsey Dam rather than the diversion. The difficulty in examining this possibility is the virtual absence of data prior to construction of the dam, and the paucity of data for the immediate post-development period. Unless some other data can be found and validated, it may not be possible to completely document the effects of hydroelectric development on Split Lake.

None of the other significant changes in Split Lake water quality can be considered to represent an adverse effect. Indeed, the generally lower ionic content and hardness of Split Lake water since CRD could be interpreted as an improvement in water quality, albeit a relatively small one.

3.5.6 Split Lake Bacterial Testing

Geometric mean fecal coliform bacteria counts for the seasonal study ranged between 6 and 37 per 100 mL at the five sites (Figure 45). Counts at the Bay Store Beach Area, Beach Point (between nursing station and Bay store), and Second Beach (near ferry), were consistently below 200 per 100 mL. Counts in 4 of the 5 samples from Nursing Station Bay and First Beach also were well below 200 per 100 mL. Single high counts occurred at First Beach on May 27 (3,030), and at Nursing Station Bay on September 21 (217). Although water quality at the seasonal study sites appears to be excellent for recreational uses, as indicated by geometric mean counts below 200 bacteria per 100 mL, it is not possible to conclude that water quality is in strict compliance with the recreational water quality guideline because sampling frequency did not meet the guideline requirement of at least 5 samples within a 30 day period.

Intensive studies of bacterial water quality were conducted in the area of the sewage treatment plant outfall on August 12 and in waters adjacent to the proposed sewage lagoon on August 26, 1987. All fecal coliform counts were below 100 per 100 mL on August 12 with a geometric mean of 21 per 100 mL for the 10 samples (Figure

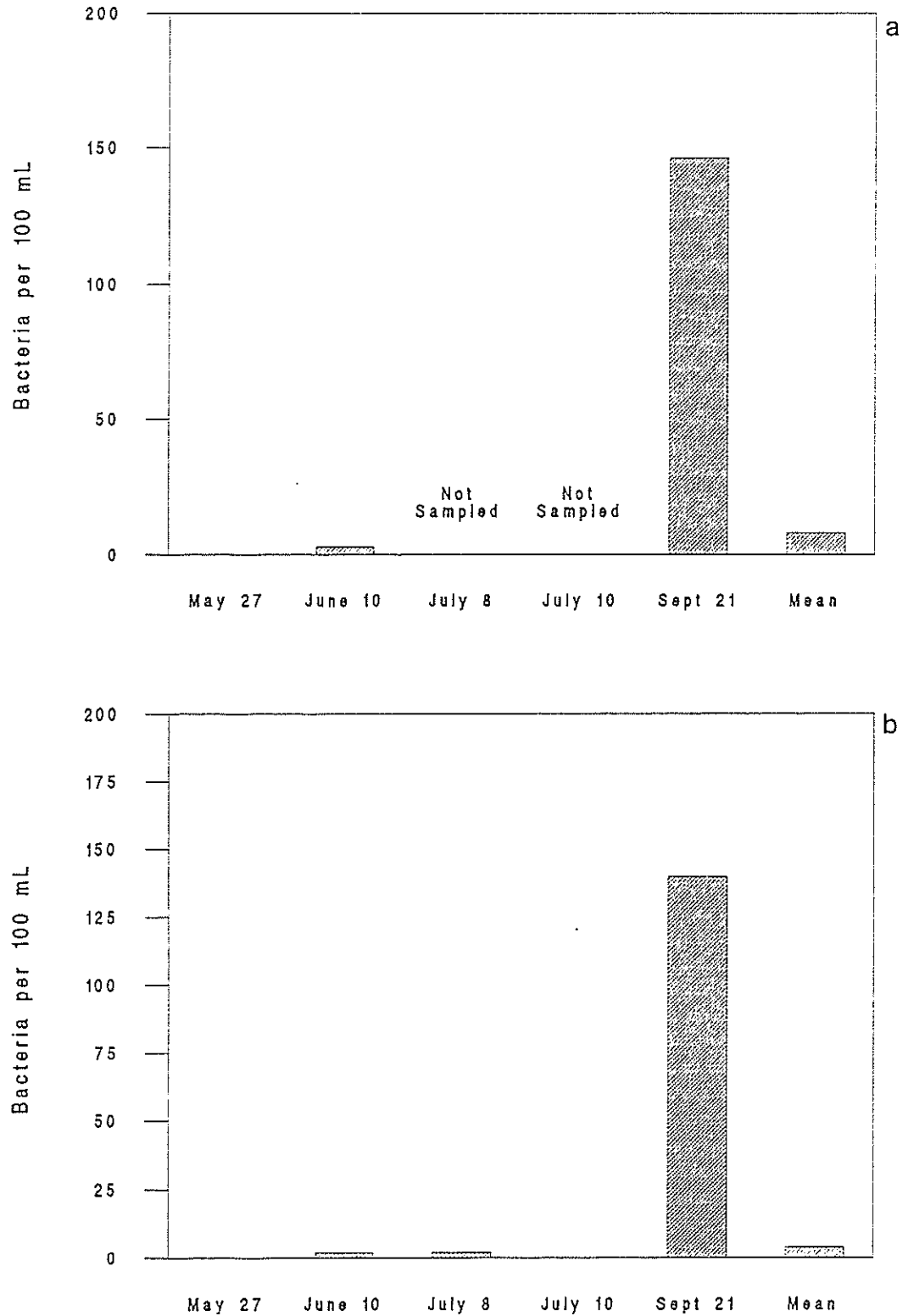


Figure 45. Fecal coliform bacteria counts from the seasonal study in the vicinity of Split Lake. Mean is the geometric mean. Dotted line indicates Canadian Water Quality Guideline. a. Bay Store Beach Area b. Beach Point between Nursing Station and Bay Store c. First Beach d. Nursing Station Bay e. Second Beach near Ferry. See Appendix 1 for site locations and descriptions.

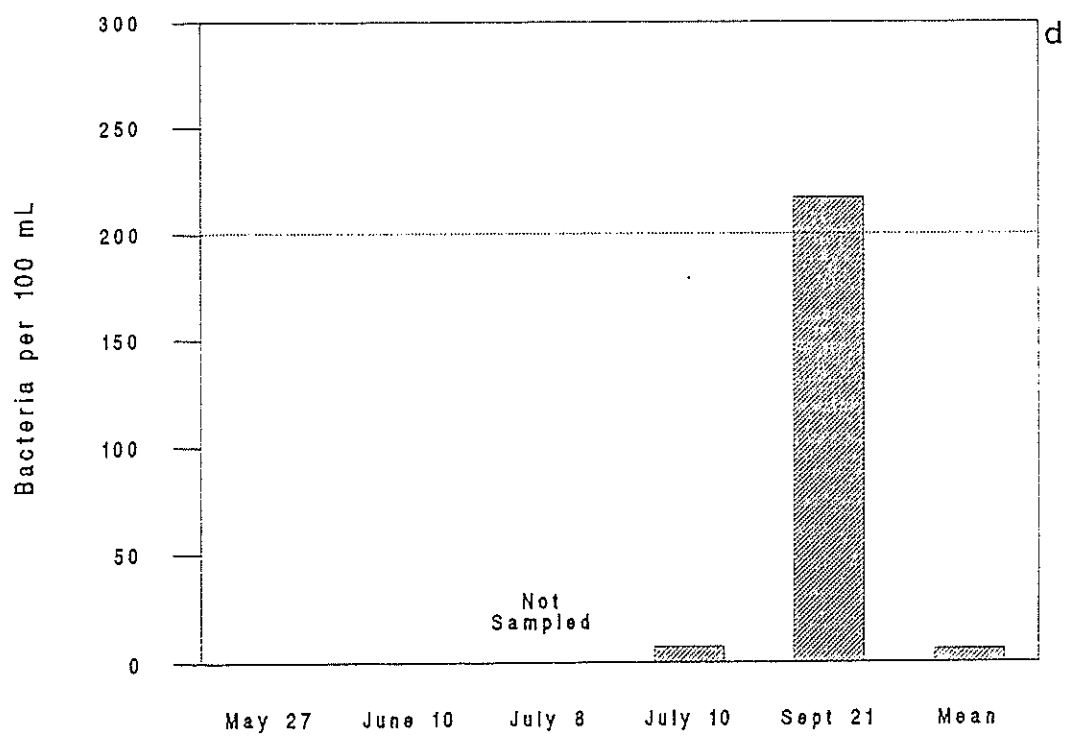
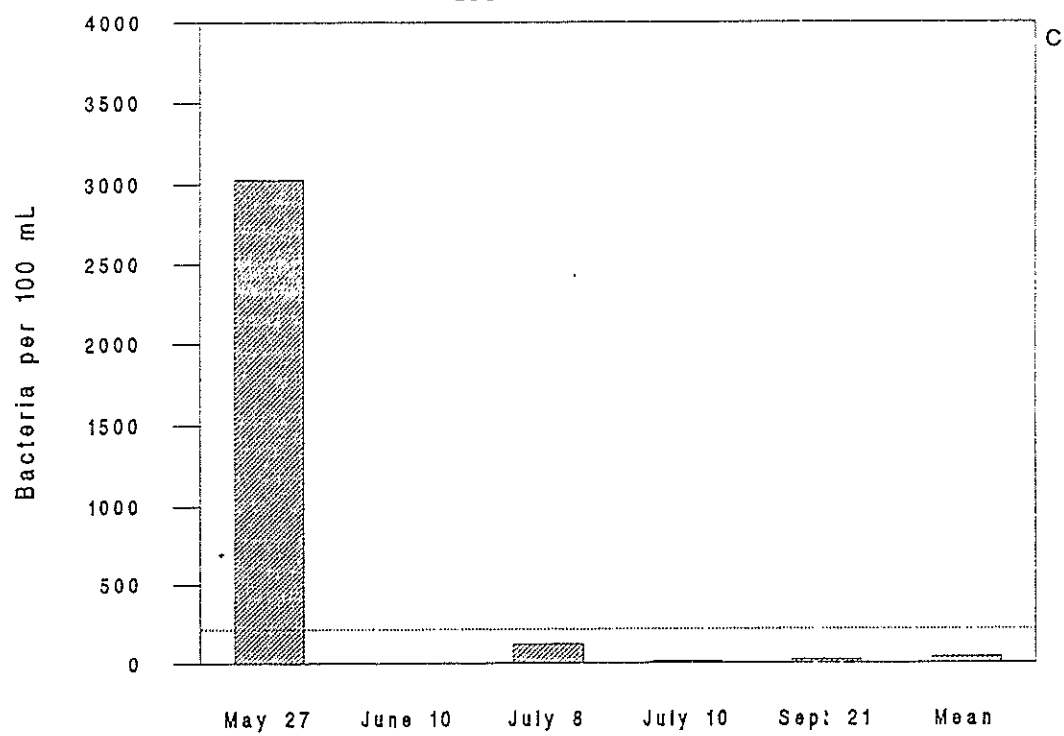


Figure 45. Continued.

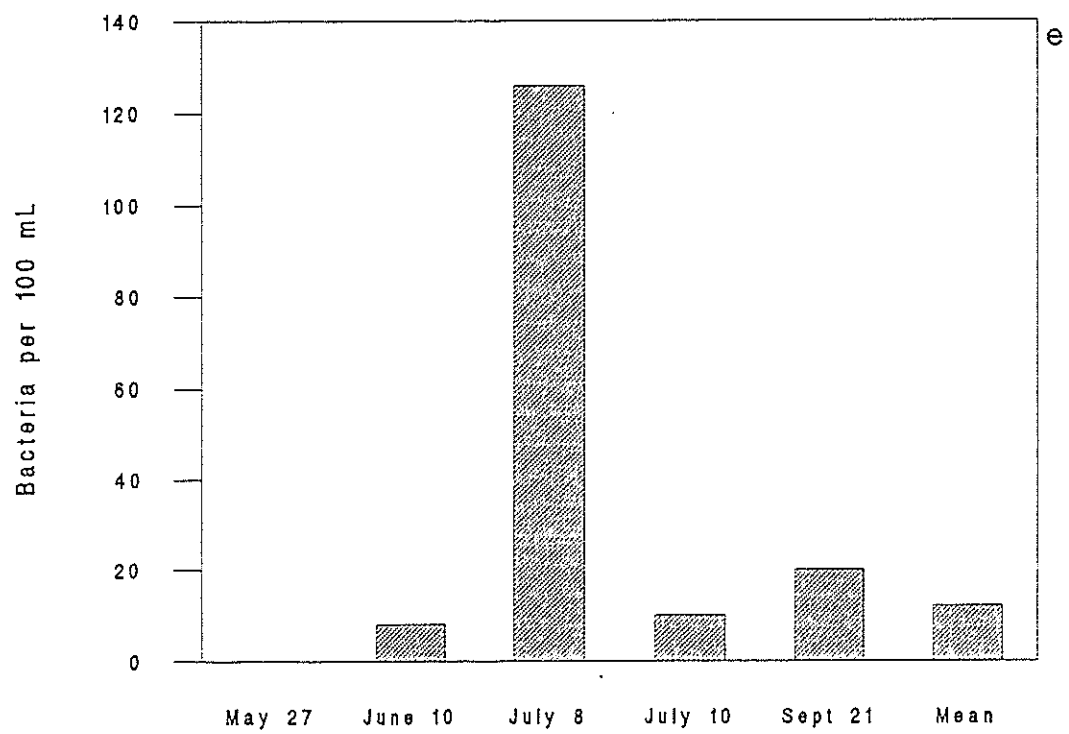


Figure 45. Continued.

46). Fecal coliform counts were below 200 bacteria per 100 mL in 9 of the 10 samples on August 26, with a very high count, 8490 bacteria per 100 mL, in one sample (Figure 46). The geometric mean count for the second survey was 8 bacteria per 100 mL. Bacteriological water quality at these sites appears to be excellent for recreational uses with geometric mean fecal coliform counts at 5-10% of the Canadian guideline.

3.5.7 York Landing Bacterial Testing

Geometric mean fecal coliform bacteria counts for the study season ranged between 1 and 10 per 100 mL at the survey sites at York Landing (Figure 47). Counts were consistently below 20 per 100 mL in all samples from the Gravel Pit, Water Treatment Plant, and Island Beach, and in 4 of the 5 samples from the Ferry Dock. The single high value at Ferry Dock, 460 per 100 mL, occurred on September 21. No intensive surveys were conducted at York Landing. The very low geometric mean fecal coliform counts in comparison to the Canadian recreational water quality guideline of 200 bacteria per 100 mL suggests bacteriological water quality is excellent for recreational uses. However, as was the case at the other 4 communities, sampling was less frequent than the 5 samples per 30 day period specified by the guideline so it is not known if water quality was in strict compliance. Any future studies should ensure that sampling meets the guideline requirements.

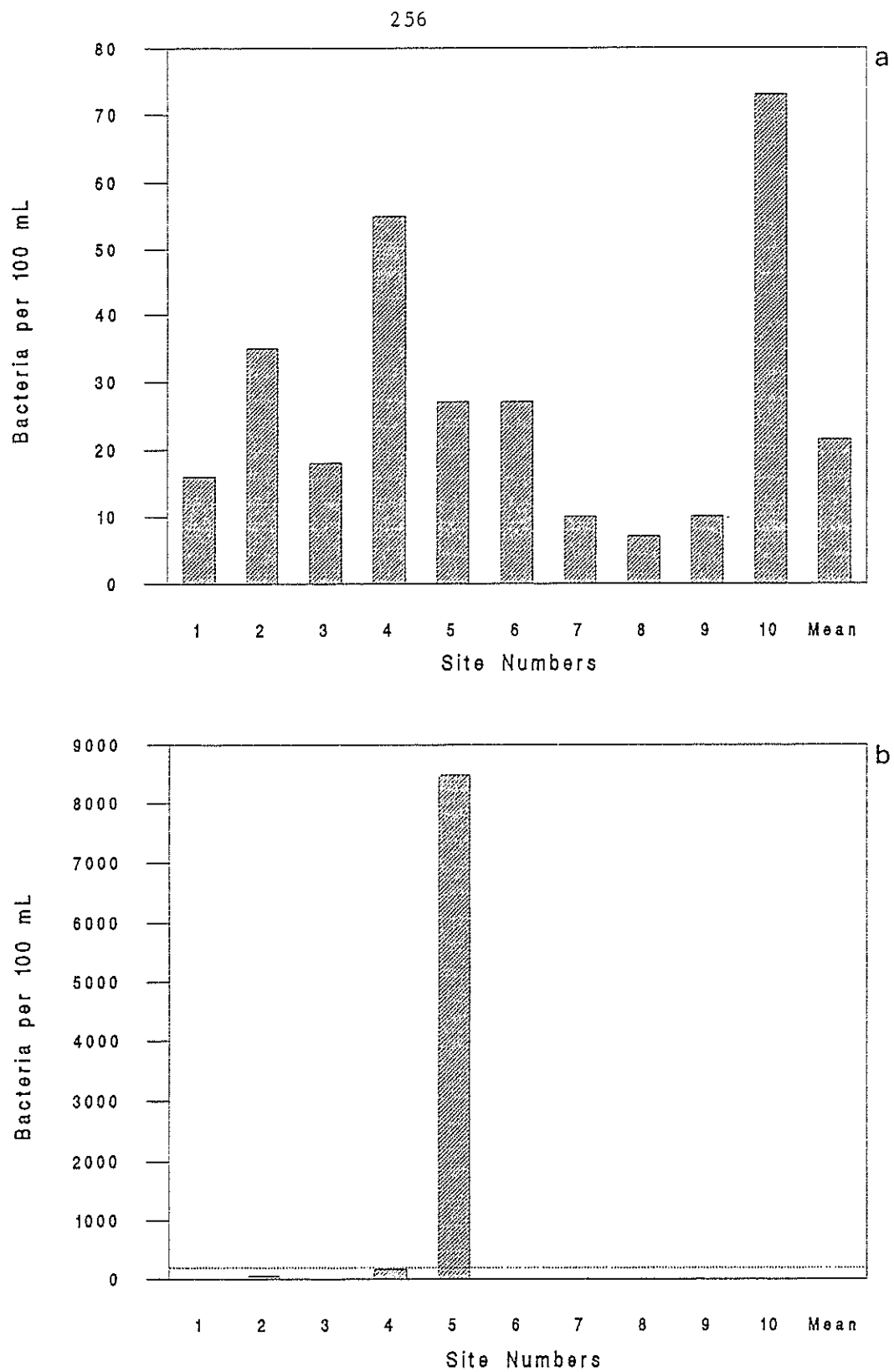


Figure 46. Fecal coliform bacteria counts from the intensive studies in the vicinity of the Split Lake Sewage Treatment Plant Outfall. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Sampled 12 August 1987 b. Sampled 26 August 1987. See Appendix 1 for site locations and descriptions.

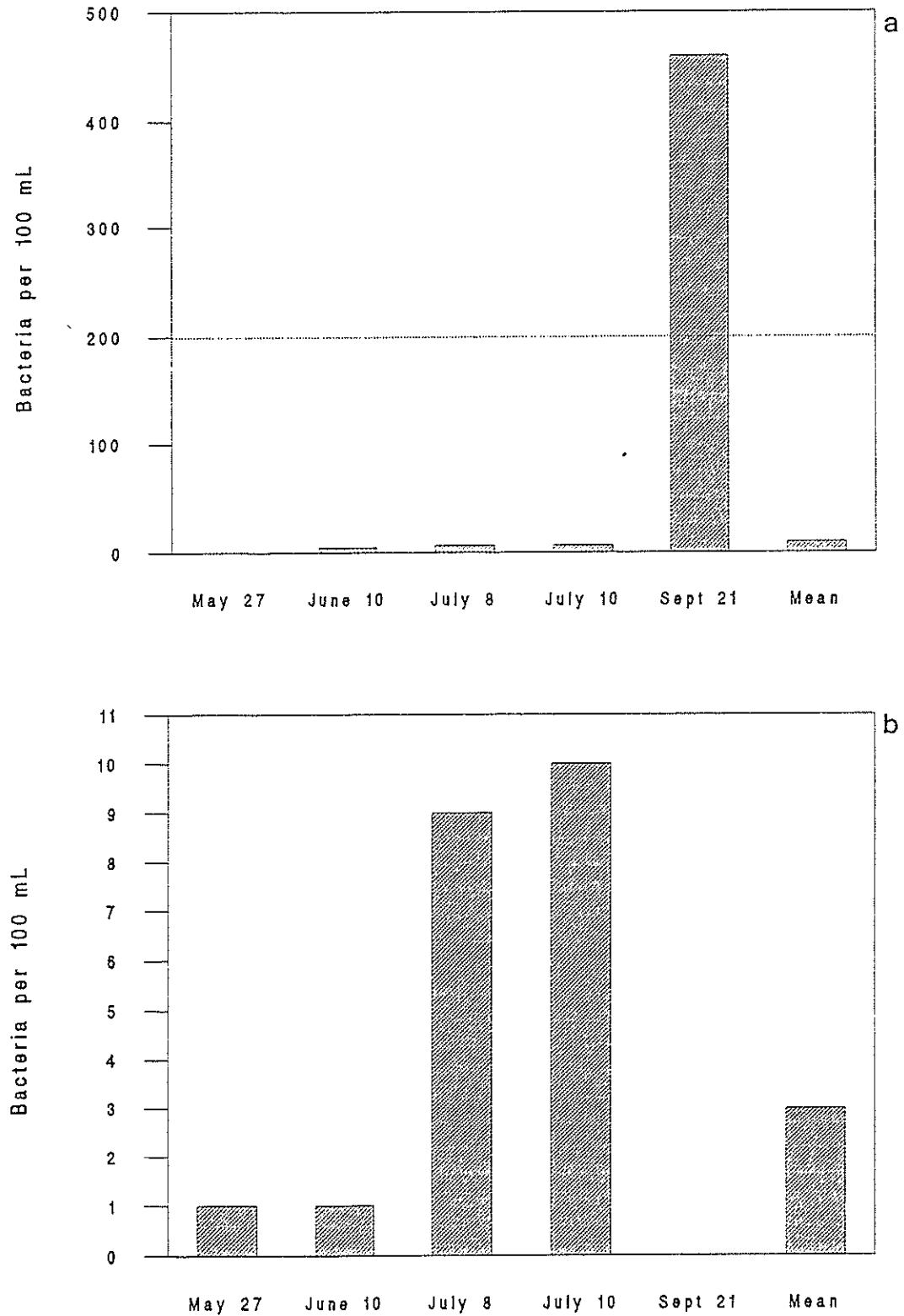


Figure 47. Fecal coliform bacteria counts from the seasonal study in the vicinity of York Landing. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Ferry Dock b. Gravel Pit c. Island Beach d. Water Treatment Plant. See Appendix 1 for site locations and descriptions.

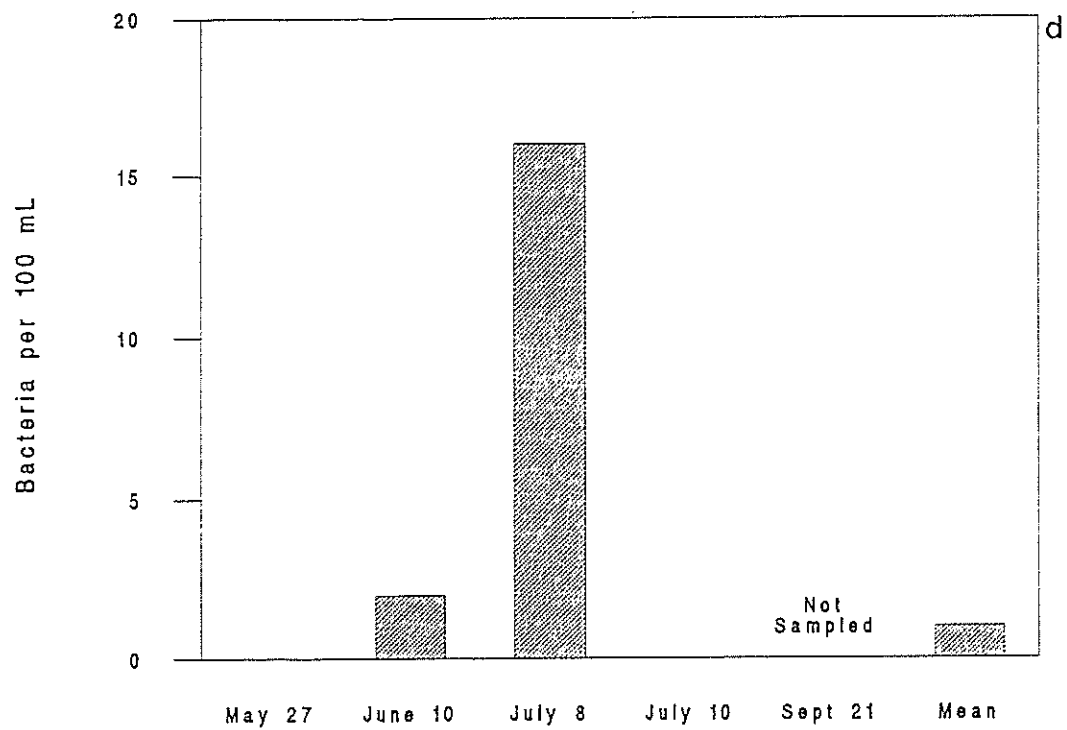
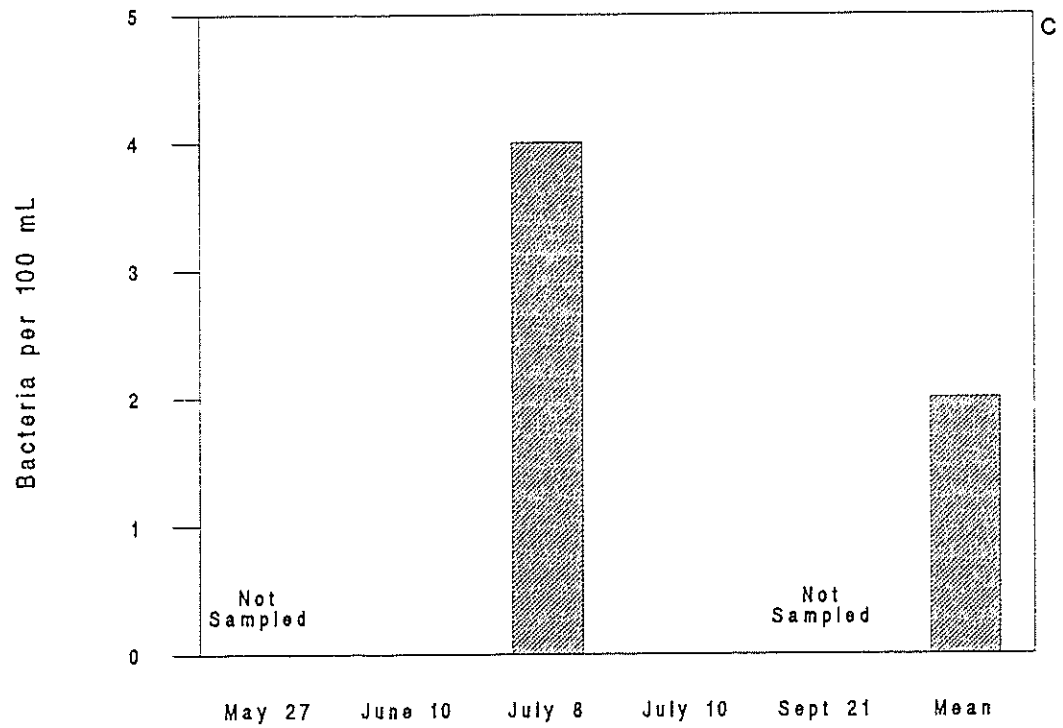


Figure 47. Continued.

4.0 Conclusions

4.1 Current Water Quality

Physical and chemical water quality at all 11 sites generally meets the Canadian Water Quality Guidelines for drinking water, recreational uses, and protection of aquatic life although there are a number of exceptions. Mean turbidity exceeded the Canadian drinking water guideline and mean extractable Al and total Cu exceeded the aquatic life guidelines at all sites (Tables 31, 32, 33, and 34). At the natural river sites and on the upper Nelson River (i.e., above Split Lake), mean turbidity was in the range of 6 to 19 NTU. Turbidity on the CRD between Threepoint and Split lakes was higher, ranging from 20 to 35 NTU, due to shoreline and bank erosion caused by the diversion. Extractable Al and total Cu concentrations at sites where the hydrologic regime was altered by hydroelectric development generally were within the ranges found in the natural rivers.

The natural rivers and sites on the CRD were distinguished by mean extractable Fe exceeding the drinking water and aquatic life guidelines. Mean extractable Fe was below the drinking water and aquatic life guidelines in the Nelson River at sites above Split Lake and in the Jack River. Both guidelines were exceeded in the Nelson River at the outlet of Split Lake. The high levels in Split Lake appear to be a direct result of CRD. Similarly, sporadic exceedences of the aquatic life guidelines for total Pb, Cd, and Zn occurred in the natural rivers and at sites on the CRD, but never in the Jack River or in the Nelson River above Split Lake. Total Pb

Table 31. Water quality variables in the natural rivers which exceeded one or more of the Canadian Water Quality Guidelines. Listed values are the mean, standard deviation (s.d.), maximum and minimum values during the 1987-1989 FEMP study period.

Variables	Units	Mean	s.d.	Maximum	Minimum
Aiken River at the inlet to Split Lake					
Turbidity (Lab)	NTU	6.0 ⁺	4.0	18.0 ⁺	1.0 ⁺
Extractable Al	mg L ⁻¹	0.107 ⁺⁺⁺	0.059	0.226 ⁺⁺⁺	<0.100
Total Cu	mg L ⁻¹	0.0029 ⁺⁺⁺	0.0041	0.0170 ⁺⁺⁺	<0.0005
Extractable Fe	mg L ⁻¹	0.438 ^{+,+++}	1.008	5.980 ^{+,+++}	0.101
Extractable Mn	mg L ⁻¹	0.093 ⁺	0.339	1.920 ⁺	0.005
Burntwood River above Threepoint Lake					
Turbidity (Lab)	NTU	19.0 ⁺	4.0	27.0 ⁺	13.0 ⁺
Extractable Al	mg L ⁻¹	0.443 ⁺⁺⁺	0.148	0.700 ⁺⁺⁺	0.191 ⁺⁺⁺
Total Cd	mg L ⁻¹	0.0001	0.0002	0.0009 ⁺⁺⁺	<0.0001
Total Cu	mg L ⁻¹	0.0061 ⁺⁺⁺	0.0081	0.0296 ⁺⁺⁺	0.0012
Extractable Fe	mg L ⁻¹	0.642 ^{+,+++}	0.165	1.050 ^{+,+++}	0.183
Total Pb	mg L ⁻¹	0.0009	0.0007	0.0028 ⁺⁺⁺	<0.0007
Extractable Mn	mg L ⁻¹	0.041	0.016	0.090 ⁺	0.005
Footprint River above Footprint Lake					
Turbidity (Lab)	NTU	9.0 ⁺	3.0	18.0 ⁺	4.0
Extractable Al	mg L ⁻¹	0.190 ⁺⁺⁺	0.097	0.410 ⁺⁺⁺	<0.100
Total Cu	mg L ⁻¹	0.0232 ⁺⁺⁺	0.1060	0.5530 ⁺⁺⁺	0.0005
Extractable Fe	mg L ⁻¹	0.420 ^{+,+++}	0.246	1.150 ^{+,+++}	0.091
Total Pb	mg L ⁻¹	0.0011	0.0030	0.0161 ⁺⁺⁺	<0.0007
Extractable Mn	mg L ⁻¹	0.057 ⁺	0.078	0.296 ⁺	<0.002
Total Zn	mg L ⁻¹	0.0040	0.0099	0.0515 ⁺⁺⁺	0.0004

⁺ Exceeds limit for drinking water.
⁺⁺ Exceeds limit for recreational use.
⁺⁺⁺ Exceeds limit for protection of aquatic life.

Table 32. Water quality variables at sites in the Norway House area which exceeded one or more of the Canadian Water Quality Guidelines. Listed values are the mean, standard deviation (s.d.), maximum and minimum values during the 1987-1989 FEMP study period.

Variable	Units	Mean	s.d.	Maximum	Minimum
Nelson River below Sea River Falls					
Turbidity (Lab)	NTU	10.0 ⁺	3.0	16.0 ⁺	6.0 ⁺
Extractable Al	mg L ⁻¹	0.181 ⁺⁺⁺	0.094	0.385 ⁺⁺⁺	<0.100
Total Cu	mg L ⁻¹	0.0047 ⁺⁺⁺	0.0054	0.0208 ⁺⁺⁺	0.0011
Extractable Fe	mg L ⁻¹	0.197	0.072	0.365 ^{+,+++}	0.087
Extractable Mn	mg L ⁻¹	0.016	0.012	0.067 ⁺	0.005
Jack River (West Channel)					
Turbidity (Lab)	NTU	9.0 ⁺	5.0	26.0 ⁺	3.0
Extractable Al	mg L ⁻¹	0.176 ⁺⁺⁺	0.126	0.442 ⁺⁺⁺	<0.100
Total Cu	mg L ⁻¹	0.0034 ⁺⁺⁺	0.0032	0.0159 ⁺⁺⁺	0.0011
Extractable Fe	mg L ⁻¹	0.187	0.106	0.457 ^{+,+++}	0.070
Total Pb	mg L ⁻¹	0.0007	0.0006	0.0027 ⁺⁺⁺	<0.0007

⁺ Exceeds limit for drinking water.
⁺⁺ Exceeds limit for recreational use.
⁺⁺⁺ Exceeds limit for protection of aquatic life.

Table 33. Water quality variables at sites on the Churchill River Diversion which exceeded one or more of the Canadian Water Quality Guidelines. Listed values are the mean, standard deviation (s.d.), maximum and minimum values during the 1987-1989 FEMP study period.

Variable	Units	Mean	s.d.	Maximum	Minimum
Rat River above Threepoint Lake					
Turbidity (Lab)	NTU	17.0 ⁺	8.0	46.0 ⁺	11.0 ⁺
Extractable Al	mg L ⁻¹	0.357 ⁺⁺⁺	0.116	0.712 ⁺⁺⁺	0.161 ⁺⁺⁺
Total Cd	mg L ⁻¹	<0.0001	0.0001	0.0004 ⁺⁺⁺	<0.0001
Total Cu	mg L ⁻¹	0.0053 ⁺⁺⁺	0.0091	0.0400 ⁺⁺⁺	0.0005
Extractable Fe	mg L ⁻¹	0.435 ^{+,+++}	0.131	0.926 ^{+,+++}	0.143
Total Pb	mg L ⁻¹	0.0008	0.0006	0.0021 ⁺⁺⁺	<0.0007
Extractable Mn	mg L ⁻¹	0.019	0.013	0.081 ⁺	0.004
Burntwood River at the outlet to Threepoint Lake					
Turbidity (Lab)	NTU	23.0 ⁺	25.0	160.0 ^{+,++}	10.0 ⁺
Extractable Al	mg L ⁻¹	0.377 ⁺⁺⁺	0.209	1.360 ⁺⁺⁺	<0.100
Total Cd	mg L ⁻¹	0.0001	0.0001	0.0006 ⁺⁺⁺	<0.0001
Total Cu	mg L ⁻¹	0.0044 ⁺⁺⁺	0.0059	0.0264 ⁺⁺⁺	0.0005
Extractable Fe	mg L ⁻¹	0.477 ^{+,+++}	0.303	2.010 ^{+,+++}	0.135
Total Pb	mg L ⁻¹	0.0008	0.0009	0.0046 ⁺⁺⁺	<0.0007
Extractable Mn	mg L ⁻¹	0.020	0.028	0.179 ⁺	0.006
Burntwood River at Thompson					
Turbidity (Lab)	NTU	27.0 ⁺	6.0	37.0 ⁺	15.0 ⁺
Extractable Al	mg L ⁻¹	0.491 ⁺⁺⁺	0.154	0.826 ⁺⁺⁺	0.230 ⁺⁺⁺
Total Cu	mg L ⁻¹	0.0040 ⁺⁺⁺	0.0051	0.0240 ⁺⁺⁺	0.0009
Extractable Fe	mg L ⁻¹	0.624 ^{+,+++}	0.189	0.931 ^{+,+++}	0.256
Total Pb	mg L ⁻¹	0.0009	0.0006	0.0027 ⁺⁺⁺	<0.0007

⁺ Exceeds limit for drinking water.
⁺⁺ Exceeds limit for recreational use.
⁺⁺⁺ Exceeds limit for protection of aquatic life.

Table 34. Water quality variables at sites in the Split Lake area which exceeded one or more of the Canadian Water Quality Guidelines. Listed values are the mean, standard deviation (s.d.), maximum and minimum values during the 1987-1989 FEMP study period.

Variable	Units	Mean	s.d.	Maximum	Minimum
Burntwood River at the inlet to Split Lake					
Turbidity (Lab)	NTU	35.0 ⁺	10.0	71.0 ^{+,++}	20.0 ⁺
Extractable Al	mg L ⁻¹	0.592 ⁺⁺⁺	0.315	2.140 ⁺⁺⁺	0.235 ⁺⁺⁺
Total Cd	mg L ⁻¹	<0.0001	0.0001	0.0003 ⁺⁺⁺	<0.0001
Total Cu	mg L ⁻¹	0.0048 ⁺⁺⁺	0.0062	0.0278 ⁺⁺⁺	0.0010
Extractable Fe	mg L ⁻¹	0.774 ^{+,+++}	0.416	2.860 ^{+,+++}	0.299
Total Pb	mg L ⁻¹	0.0010	0.0007	0.0034 ⁺⁺⁺	<0.0007
Extractable Mn	mg L ⁻¹	0.030	0.027	0.177 ⁺	0.004
Nelson River below Grass River					
Turbidity (Lab)	NTU	9.0 ⁺	3.0	10.0 ⁺	5.0
Extractable Al	mg L ⁻¹	0.188 ⁺⁺⁺	0.101	0.418 ⁺⁺⁺	<0.100
Total Cu	mg L ⁻¹	0.0037 ⁺⁺⁺	0.0043	0.0159 ⁺⁺⁺	0.0010
Extractable Fe	mg L ⁻¹	0.196	0.074	0.409 ⁺⁺⁺	0.045
Nelson River at the outlet of Split Lake					
Turbidity (Lab)	NTU	18.0 ⁺	5.0	35.0 ⁺	11.0 ⁺
Extractable Al	mg L ⁻¹	0.315 ⁺⁺⁺	0.126	0.651 ⁺⁺⁺	<0.100
Total Cu	mg L ⁻¹	0.0050 ⁺⁺⁺	0.0075	0.0308 ⁺⁺⁺	0.0010
Extractable Fe	mg L ⁻¹	0.389 ^{+,+++}	0.110	0.672 ^{+,+++}	0.129
Total Pb	mg L ⁻¹	0.0009	0.0011	0.0066 ⁺⁺⁺	<0.0007

⁺ Exceeds limit for drinking water.
⁺⁺ Exceeds limit for recreational use.
⁺⁺⁺ Exceeds limit for protection of aquatic life.

occasionally exceeded the aquatic life guideline at the outlet of Split Lake although these occasional high levels are not directly attributable to the CRD.

The consistent exceedence of the aquatic life guideline for extractable Al and total Cu, and of both the aquatic life and drinking water guidelines for extractable Fe in the northern drainages suggests a review of the applicability of these guidelines in the study area is in order. It remains to be determined if the naturally high levels of these variables are responsible for any impairment of drinking water quality or have any other ecological significance. If not, then it may be appropriate to develop specific water quality objectives for the area to accommodate these naturally higher levels.

4.2 Temporal Variation in Natural River Water Quality

The natural rivers (Burntwood River above Threepoint Lake, Footprint River above Footprint Lake) were distinguished by considerable stability in water quality between the 1971-1974 pre-development sampling period and the 1987-1989 FEMP study. Significant changes were detected in just 3 of the 17 variables for which comparable pre- and post-development data existed. Decreases in pH were detected in both the Footprint and Burntwood rivers. Decreases in NO_2 - NO_3 and SO_4 occurred in the Footprint River and a marginally significant decrease in SO_4 occurred in the Burntwood River. All of these changes appear to have been due to alterations in sampling or analytical methods or to changes in the laboratories

which conducted the analyses. Any comparable decreases in pH, SO_4 , or NO_2 - NO_3 at the sites affected by hydroelectric development were, therefore, attributed to methodological sources.

Long-term temporal trends in metals could not be examined in the natural rivers. Data on extractable or total metal species are essentially non-existent for the pre-development period. Several dissolved metals were routinely analyzed in the pre-development period, but the design of the FEMP study provided just one year of post-development measurements, eliminating the possibility of a statistical comparison.

No pre-development water quality data exist for the Aiken River, precluding an analysis of temporal trends at this site.

4.3 Changes in Water Quality Related to the Hydro Developments

4.3.1 Norway House Area

Impacts of LWR on water quality in the Nelson River below Sea River Falls and in the Jack River could not be assessed due to limited (Sea River Falls) or non-existent (Jack River) pre-development data records.

4.3.2 Nelson House and Threepoint Lake Area

The only adverse effect of CRD on water quality in the Rat River above Threepoint Lake was an approximately 3-fold increase in turbidity (Table 35) caused by bank and shoreline erosion on the Rat River below Notigi Dam and in the southern part of Notigi Reservoir. The primary source of this turbidity appears to be

Table 35. Water quality variables which changed significantly following Churchill River diversion. X_{pre} denotes the mean for the pre-development years 1971 through 1974. X_{post} denotes the mean for the post-development years 1987 through 1989.

Variable	Units	X_{pre}	X_{post}
Rat River above Threepoint Lake			
Conductivity (Lab)	$\mu\text{S cm}^{-1}$	120	102
Turbidity (Lab)	NTU	5	17
Alkalinity (Total)	mg L^{-1}	56.4	46.4
Hardness (Total)	mg L^{-1}	61.2	45.9
TOC	mg L^{-1}	10.7	6.38
$\text{NO}_2\text{-NO}_3$	mg L^{-1}	0.237	0.019
Ortho PO_4	mg L^{-1}	0.005	0.002
Dissolved Na	mg L^{-1}	2.2	2.7
Dissolved K	mg L^{-1}	0.9	1.02
Dissolved Ca	mg L^{-1}	18.2	12.3
Dissolved Cl	mg L^{-1}	1.0	1.1
Dissolved F	mg L^{-1}	0.05	0.09
Reactive Si	mg L^{-1}	3.6	1.83

Notigi Reservoir.

Many other significant changes also occurred, although none can be classified as adverse. Annual mean conductivity, alkalinity (total), hardness (total), TOC, ortho PO_4 , Ca, and reactive Si in the Rat River decreased by 15 to 60% between the 1972-1973 pre-development sampling period and the FEMP study (Table 35). Significant increases in Na, K, Cl, and F, by 10-80%, also occurred. All of these changes are directly attributable to diversion and are a reflection of differences in water quality between the Churchill River and the pre-existing Rat River. In most cases, the changes were no greater than the difference between the Rat and Churchill rivers, although the decrease in TOC was greater. This suggests some loss of TOC from the water column in passing through SIL and Notigi Reservoir. Decreases in SO_4 and NO_2 - NO_3 were comparable to those in the natural rivers and were attributed to changes in analytical methodology.

The total P concentration in the Rat River did not change following diversion despite a lower concentration in the Churchill River. Release of P from flooded soils and vegetation probably accounts for this lack of a decrease.

Dissolved Fe, Mn, Cu, Pb, and Zn concentrations were not affected by diversion. The impact of diversion on other metal species is uncertain due to the lack of pre-development data.

Impacts of diversion on water quality in the Burntwood River at the outlet of Threepoint Lake could not be quantified due to the lack of pre-development data for this site. Significant changes

would be expected and these should be qualitatively similar to those in the Rat River. However, the impact of CRD on turbidity in the outflow was probably much smaller than in the Rat River. Turbidity in the Rat River after diversion was the same as the natural level in the Burntwood River which accounted for about 60% of the outflow discharge prior to diversion.

4.3.3 Burntwood River at Thompson

The only adverse effect of CRD on water quality in the Burntwood River at Thompson is elevated turbidity due to bank and shoreline erosion. Many other changes in water quality occurred in addition to the increase in turbidity (Table 36), but none can be classified as an adverse effect based on the definition employed in this study. All of the other significant changes fall in the category of an improvement in water quality due to the introduction of softer Churchill River water.

Most of the changes in water quality did not occur until after the autumn of 1977, when the diversion was fully operational, although several were evident during CRD construction (from June, 1974, to August, 1976), when Rat River flows were cut off at the outlet of Notigi Lake, or during the immediate post-diversion period (September, 1976, to December, 1977), as flows were gradually increased. Turbidity and hardness increased during the period of CRD construction (Table 36). The increase in hardness was due to reduced flows on the Burntwood River during Notigi Dam construction and was restricted to the construction period. About

Table 36. Water quality variables in the Burntwood River at Thompson which changed significantly following Churchill River diversion. X_{pre} denotes the pre-development period from 1967 to May 1974. X_{const} denotes the construction period between June 1974 and August 1976. X_{post} is the immediate post-diversion period from September 1976 to December 1977. $X_{current}$ represents the FEMP study period between January 1987 and October 1989.

Variable	Units		
		X_{pre}	X_{const}
Turbidity (Lab)	NTU	14	23
Hardness (Total)	mg L ⁻¹	69	77
		X_{pre}	X_{post}
Total P	mg L ⁻¹	0.028	0.065
		X_{pre}	$X_{current}$
Conductivity (Lab)	$\mu\text{S cm}^{-1}$	141	115
Turbidity (Lab)	NTU	14	27
Alkalinity (Total)	mg L ⁻¹	65	53
Hardness (Total)	mg L ⁻¹	69	53
TOC	mg L ⁻¹	12.63	6.94
Dissolved NO ₂ -NO ₃	mg L ⁻¹	0.140	0.022
Reactive Si	mg L ⁻¹	3.94	2.01
Dissolved Ca	mg L ⁻¹	21.3	14.6
Dissolved SO ₄	mg L ⁻¹	4.1	2.7
Dissolved Fe	mg L ⁻¹	0.138	0.077
		X_{const}	X_{post}
Conductivity (Lab)	$\mu\text{S cm}^{-1}$	148	125
Alkalinity (Total)	mg L ⁻¹	70	60
Hardness (Total)	mg L ⁻¹	77	66
Total P	mg L ⁻¹	0.039	0.065

Table 36. continued.

Variable	Units		
		X_{const}	X_{current}
Conductivity (Lab)	$\mu\text{S cm}^{-1}$	148	115
Alkalinity (Total)	mg L^{-1}	70	53
Hardness (Total)	mg L^{-1}	77	53
TOC	mg L^{-1}	14.00	6.94
Reactive Si	mg L^{-1}	4.10	2.01
Dissolved Mg	mg L^{-1}	6.5	4.0
Dissolved Ca	mg L^{-1}	21.5	14.6
Dissolved SO_4	mg L^{-1}	4.0	2.7
Extractable Mn	mg L^{-1}	0.037	0.021
Extractable Al	mg L^{-1}	0.870	0.488
		X_{post}	X_{current}
Turbidity (Lab)	NTU	18	27
Alkalinity (Total)	mg L^{-1}	60	53
Hardness (Total)	mg L^{-1}	66	53
TOC	mg L^{-1}	15.00	6.94
Total P	mg L^{-1}	0.065	0.036
Reactive Si	mg L^{-1}	3.25	2.01
Dissolved SO_4	mg L^{-1}	4.0	2.7
Extractable Fe	mg L^{-1}	0.786	0.628
Extractable Mn	mg L^{-1}	0.030	0.021

25% of the increase in turbidity during construction was due to elimination of inflow from the Rat River, with the balance of the increase possibly due to in-stream work and bank stabilization activities.

Total P concentrations approximately doubled between the construction and immediate post-impoundment periods (Table 36), probably due to release of P from flooded organic soils and vegetation. Pre-development levels were re-established before 1987 but this does not mean the P release has ended. The annual mean total P concentration in the Burntwood River is still more than twice that in the Churchill River near Leaf Rapids. Most of the P enters the water column as particulates; TDP at Thompson is 50% higher than in the Churchill River while particulate P is 300% higher.

Significant water quality changes between the pre-development and current (1987-1989) study periods included decreases in conductivity, alkalinity, hardness, $\text{NO}_2\text{-NO}_3$, reactive Si, and dissolved Ca, SO_4 , and Fe, and an approximate doubling of turbidity (Table 36). Decreases in dissolved Mg and in extractable Al and Mn occurred between the construction and current study periods. The primary cause of the increase in turbidity was shoreline and bank erosion on the Burntwood River below Threepoint Lake. All of the decreases, with the exception of SO_4 and $\text{NO}_2\text{-NO}_3$, were due to the introduction of softer Churchill River water. The decreases in both SO_4 and $\text{NO}_2\text{-NO}_3$ were attributed to changes in analytical methods, laboratories, or both.

Concentrations of dissolved and extractable Al have not been elevated by CRD. A previous report of increased extractable Al following diversion was the product of the mistaken classification of pre-development dissolved Al data as extractable Al.

4.3.4 Split Lake Area

4.3.4.1 Burntwood River at Inlet to Split Lake

The changes water quality at this site caused by CRD undoubtedly were similar to those documented upstream at Thompson. No quantitative assessment of water quality changes at this site since CRD was possible due to the lack of pre-development data.

4.3.4.2 Nelson River at Inlet to Split Lake

Several significant differences in water quality were noted for the Nelson River above Split Lake between the 1971-1974 pre-development study period and the FEMP study (Table 37) but none of these changes could be directly attributed to LWR. The FEMP study was conducted during a period of very low Nelson River flows caused by drought conditions in western Canada. These changes could therefore be due in part, or in total, to the lower flows. There was no way of isolating the impact of LWR from any drought effects because the FEMP study comprised the entire post-development data record for the site.

Table 37. Water quality changes at sites in the Split Lake area following Lake Winnipeg regulation and Churchill River diversion. X_{pre} denotes the mean for the pre-development years 1972 and 1973. X_{post} denotes the mean for the post-development years 1987 through 1989. For modelled changes, X_{pre} is the current concentration in the Nelson River at the inlet to Split Lake.

Variable	Units	X_{pre}	X_{post}
Nelson River at the inlet to Split Lake			
pH (Lab)	units	8.24	7.85
NFR	mg L ⁻¹	15	5
Dissolved K	mg L ⁻¹	2.43	2.06
TDP	mg L ⁻¹	0.020	0.011
TDN	mg L ⁻¹	0.388	0.304
Reactive Si	mg L ⁻¹	0.72	1.24
Nelson River at the outlet to Split Lake			
Based on a pre/post comparison			
pH (Lab)	units	8.06	7.81
Conductivity (Lab)	$\mu\text{S cm}^{-1}$	307	228
Hardness (Total)	mg L ⁻¹	120.0	91.7
Dissolved Ca	mg L ⁻¹	33.9	24.1
Dissolved K	mg L ⁻¹	2.26	1.65
Dissolved Na	mg L ⁻¹	14.3	9.3
Dissolved Cl	mg L ⁻¹	15.3	9.5
Dissolved SO ₄	mg L ⁻¹	26.0	13.7
Based on modelled changes			
Dissolved As	mg L ⁻¹	0.0007	0.0005
Dissolved Fe	mg L ⁻¹	0.193	0.387
Extractable Mn	mg L ⁻¹	0.010	0.015

4.3.4.3 Nelson River at Outlet of Split Lake

An increase in extractable Fe, such that the annual mean concentration at the outlet now exceeds the Canadian Water Quality Guidelines for drinking water and protection of aquatic life by 30% (Table 37), has been the only adverse effect of CRD on water quality in Split Lake. This increase, and significant decreases in TDS, DIC, and dissolved As were inferred on the basis of current differences in concentration between the Burntwood and Nelson rivers and the relative contribution of these rivers to total inflow.

Churchill River diversion has not caused a significant increase in annual mean water turbidity or suspended sediment concentration (NFR) or a decrease in water transparency despite the much higher suspended sediment concentrations in the Burntwood River than in the Nelson River. Much of the suspended sediment load of the Burntwood River appears to settle out near the mouth, preventing a general increase in suspended sediment levels in the lake or downstream. Sediment deposition also appears to have prevented increased concentrations of extractable Al and total Ni and V in Split Lake.

Statistical analysis of pre- and post-development data for 24 variables revealed significant declines had occurred in 8 (Table 37). All but the decline in pH were a direct result of diversion. The decrease in pH at the Split Lake outlet is consistent with the decrease which occurred in the Nelson River near the inlet and in the natural rivers during the same period, indicating the change is

not related to the hydroelectric developments. None of these significant changes based on pre/post comparisons represents an adverse effect.

4.4 Water Quality - Water Discharge Relationships

The natural rivers, sites on the Nelson River upstream of Split Lake, and the Burntwood River at Thompson prior to diversion were characterized by having many discharge-correlated variables, with negative correlations dominating. Negatively correlated variables generally included the major cations (Na, K, Ca, Mg) and anions (Cl, F, SO_4) and associated variables (conductivity, alkalinity, hardness, TDS), some metals, DIC, reactive Si, and dissolved N species (TDN, $\text{NO}_2\text{-NO}_3$, total NH_3). Positive correlates at these sites generally included turbidity, NFR, colour, and the organic carbon species (DOC and TOC).

The discharge relationships were profoundly different at sites on the CRD. In the Rat River above Threepoint Lake and in the Burntwood River at the outlet to Threepoint Lake, at Thompson, and at the inlet to Split Lake all correlations with discharge were positive following diversion. This is evidence of the increased importance of eroded shoreline materials in regulating water quality. Whereas water quality typically was highest during periods of peak flow prior to diversion, water quality now decreases with increasing river flows.

The discharge relationships also were weaker, in general, at the sites on the CRD than in the natural rivers. This appears to be

a product of the less dynamic discharge regime following diversion. Although mean discharge on the Rat and Burntwood rivers was substantially elevated due to the diversion, the variation in discharge relative to the mean was an order of magnitude lower.

It is unclear if LWR has had any effect on water quality - water discharge relationships in the Nelson River at the inlet to Split Lake or if either LWR or CRD has affected the relationships at the outlet of Split Lake. The relationships during the FEMP study differed from those which prevailed prior to development, and the changes at the outlet to Split Lake were similar to those which occurred at sites on the CRD, but the effect of the hydroelectric developments could not be separated from effects of the low Nelson River flows during the FEMP study.

The water quality - water discharge relationships derived from the FEMP study data offer limited potential for impact prediction or management. The strongest correlations were found at sites where there is the least need for their application; the natural rivers and the Nelson River at Sea River Falls. Most relationships at sites on the CRD were too weak (i.e., $r^2 < 0.25$) to have much predictive ability. Several important exceptions included the relationships for NFR and turbidity in the Burntwood River at Thompson, and the relationships for turbidity and alkalinity in the Burntwood River at the inlet to Split Lake. None of the water quality - water discharge relationships for the Nelson River at the inlet or outlet to Split Lake should be used for impact prediction or management purposes until the influence of the low Nelson River

flows on these relationships can be determined.

4.5 Seasonal Variations in Water Quality

A number of distinct seasonal trends were evident in the comparison of water quality under ice cover and in the open water season during the 1987-1989 study period. Lower chlorophyll a concentrations occurred under ice cover at several sites (Jack R., Rat R. above Threepoint L., Burntwood R. at Thompson, and Burntwood R. at the inlet to Split Lake). This may be a result of the imposition of light limitation on algal growth by accumulated snow and ice cover. Elevated $\text{NO}_2\text{-NO}_3$ concentrations under ice-cover also occurred at most sites (Rat R. above Threepoint Lake, Burntwood R. at the inlet and at the outlet of Threepoint Lake and at the inlet to Split Lake, and the Nelson R. at the inlet and outlet to Split Lake). This could be a result of lower photosynthetic demand for nitrogen at the lower light levels and cooler temperatures under ice cover. At some sites (Rat R. above Threepoint Lake, Burntwood R. at the outlet of Threepoint Lake, Nelson R. at the outlet of Split Lake), the higher $\text{NO}_2\text{-NO}_3$ levels under ice contributed to higher TDN and, in some cases (Rat R.), higher total N concentrations.

A variety of other variables displayed marked seasonality at specific sites, with a larger number of seasonally variant parameters in the smaller natural rivers (i.e., Footprint and Aiken rivers) than in the larger Burntwood, Rat, and Nelson rivers. These variables included major ions and associated variables like

conductivity, alkalinity, hardness and in some cases metals and NFR. Most of these variables increased under ice cover, although NFR typically was lower in winter. For variables which were correlated with discharge, these seasonal differences were accounted for by seasonal variations in river discharge.

At sites where water quality measurements were made prior to development, samples under ice cover generally were much less frequent than in the FEMP study. This restricted the examination of changes in seasonal patterns related to the hydroelectric developments to a qualitative analysis. The analysis was further restricted by the few variables which were common to the pre- and post-development periods at many sites. Although several changes in seasonal patterns were noted, none could conclusively be attributed to the hydroelectric developments. The most complete analysis was possible for the Burntwood River at Thompson, where measurements were routinely made under ice-cover in the pre-development and construction periods. Only chlorophyll a showed marked seasonality in any of the pre-development, construction, or current study periods.

5.0 Impact Mitigation and Management Options

5.1 Burntwood River - Threepoint Lake to Split Lake

The major impact of CRD on this reach of the Burntwood River has been the increase in turbidity due to shoreline and bank erosion along the Burntwood River and upstream on the Rat River and in southern Notigi Reservoir. Elevated turbidity also is the only

adverse water quality impact currently evident in this section of the CRD. Natural bank stabilization processes should eventually contribute to a reduction in turbidity, but the length of time this will require and the level of recovery which will be achieved are difficult to predict.

5.2 Split Lake

The development of measures to mitigate adverse effects of CRD on Split Lake water quality does not appear to be justifiable at this time. Notwithstanding the several significant changes in water quality related to CRD, the only adverse effect of the diversion on water quality which has been documented thus far is the increase in mean extractable Fe above the drinking water and aquatic life guidelines.

Although the guidelines are now exceeded in Split Lake, there is no reason to believe that drinking water quality or aquatic life have actually been adversely affected. Naturally high Fe levels are prevalent in the study area. However, should information become available in the future which suggests significant environmental degradation has occurred as a result of this increase in Fe, requiring mitigation at source rather than treatment before consumption, this should be possible through management of the relative contribution of Burntwood River flows to Nelson River flows. A 50% reduction in Burntwood River flows should reduce the concentration in Split Lake to the guideline value.

6.0 Recommendations for Further Study

6.1 Norway House Area

The FEMP sampling program adequately characterized current water quality at these sites and represents a satisfactory baseline for assessment of the impact of any future hydroelectric projects. The impact of LWR on water quality at these sites could not be determined due to the shortage of pre-development data. This deficiency cannot be alleviated by further sampling.

It may be possible to infer the nature of any water quality changes which have occurred at the Jack River or Sea River Falls sites by comparing the FEMP study data with existing pre-development data for upstream sites. Water quality in the Nelson River at Warren Landing, sampled since 1969, might be a suitable surrogate for the Jack River. Similarly, data for the Nelson River at Norway House might be a surrogate for Sea River Falls. In order to use pre-development data from these surrogate sites, it must be determined that there are, or were, no natural differences in water quality between the FEMP sites and their respective surrogates or at least that any natural differences between the sites were not similar in character to those which might be caused by the hydroelectric development. Furthermore, this validation of the surrogate sites must be done indirectly, using information on natural erosion sites and locations of man-made facilities, due to the paucity or complete lack of pre-development data for the FEMP sites.

6.2 Burntwood River - Threepoint Lake to Split Lake

The only adverse impact of Churchill River diversion on water quality in the Burntwood River system has been the increase in turbidity due to bank and shoreline erosion. The higher turbidity in the Rat River appears to be from Notigi Reservoir while the increase in the Burntwood River is primarily due to erosion occurring Threepoint Lake. Although this change can be classified as an adverse impact on the basis of the Canadian Water Quality Guidelines, and the general nature of any associated ecological changes can be stated on the basis of studies conducted elsewhere, any specific impacts of increased turbidity on the ecology of the Burntwood River remain largely unknown, as most studies have focused on lakes in the Rat River valley. Investigation of the ecological effects of increased turbidity in the system will be necessary for the establishment of water quality objectives to be met by hydroelectric generating stations which are constructed on the Churchill-Nelson system in the future.

Unfortunately, any assessment of biological impacts will be more limited by the availability of pre-development data than was the analysis of water quality impacts presented in this report. Nevertheless, some review of biological impacts related to increased turbidity on the Burntwood River is warranted, if only to provide adequate documentation of the limitations of any such analysis.

The overall impact of elevated turbidity is determined by both the magnitude and duration of the increase. This report documents

magnitude, but none of the data gathered to date provide an indication of duration. The eroding banks and shorelines should gradually stabilize over time, but the lack of a significant decline in turbidity since diversion indicates this may be a very slow process. An estimate of duration is necessary both for complete documentation of the impact of CRD on the sediment regime in the Burntwood River, as well as for assessment of the impact of future modifications to the operation regime.

There is no need for continued broad-spectrum water quality monitoring on the Burntwood River for assessment of the major impacts of CRD. Additional data collection will never make up for deficiencies in the pre-development database and, unless there is a change in the way the diversion is managed, there is little reason to expect further substantial changes in water quality.

Although the major changes in water quality appear to have occurred, the length of time required for the system to reach a new equilibrium is unknown. Thus there is strong justification for the continued monitoring of water quality at a minimum of one site. Located below the major reservoirs on the CRD, the Thompson site is well-placed to examine long-term changes in water quality on the CRD. Other sites might be considered, but the Thompson site has the most complete data record of any site in the Churchill-Nelson system and is readily accessible. Annual monitoring is probably unnecessary, with a 5 year interval between monitoring periods providing adequate coverage. Each monitoring period should be a 2-year study which repeats the FEMP study sampling and analytical

methods exactly. A number of variables could be dropped from the program to reduce cost without losing important information. Deleted variables could include all those which were consistently at or below the limit of detection, including total Ba, dissolved and total Cd, total Co, and total Hg. Other variables which might be dropped include the field measurements of pH, conductivity, and turbidity because all of the pre-development data are laboratory values.

Any future field activities should include an examination of the predictive ability of the water quality - water discharge relationships identified in the present study. Such studies would involve the measurement of discharge-correlated variables over a range of discharges, and the comparison of predicted and observed values.

6.3 Split Lake

Contrary to the perception of some members of the Split Lake community, this and other studies have shown CRD has not caused a general increase in turbidity or other suspended sediment-related variables in Split Lake. This does not necessarily indicate the perception of increased turbidity is incorrect, only that any increase is not widespread. The diversion has undoubtedly had a substantial impact on sediment deposition rates in Split Lake near the mouth of the Burntwood River, and possibly has affected sediment chemistry in this region as well as causing a local elevation in turbidity. Documentation of the extent and magnitude

of sediment-related impacts in this part of the lake is required. Variables monitored should include the rate and location of sediment deposition and the chemical composition of the sediments with emphasis on metals and factors regulating metal mobilization.

The FEMP and other sampling programs which have been conducted in Split Lake examined changes in turbidity and other variables in the central basin and outlet. This approach was necessary to detect any major, wide-spread, changes but could easily have missed localized impacts or changes around the perimeter of the lake. The higher lake level since diversion does not appear to have caused widespread shoreline erosion, as expected since the increase was within the natural range of lake levels. It is not known if any isolated or localized increases in erosion have occurred, and this possibility should be investigated. This might be done using remote sensing techniques accompanied by appropriate ground-truthing surveys.

The increase in extractable Fe in Split Lake is the only change in water quality which can be categorized as adverse, yet it is unlikely that this increase has had an effect on drinking water quality or on aquatic life. The prevalence of sites in the NFA study area where extractable Fe naturally exceeds the Canadian water quality guidelines for consumption and protection of aquatic life suggests the guidelines may be inappropriate for this area. A more complete review of information on the effect of Fe on drinking water quality and aquatic life than was possible in this report should be conducted to assess the applicability of the guideline in

the area. Additional monitoring of extractable Fe levels in Split Lake may be required if the current elevated levels are shown to have impaired water quality.

The low Nelson River flows during the FEMP study period made it difficult to determine if the apparent changes in water quality at the inlet to Split Lake, and in the water quality - water discharge relationships at the inlet and outlet to Split Lake, were due to the hydroelectric developments or to low flow conditions. There is a need for continued monitoring over a wider range of flows to discriminate natural from project-induced changes, if any have occurred.

The monitoring program should be consistent with the design of the FEMP study, with a similar sampling frequency, although a number of the variables measured in the FEMP study could be dropped from any subsequent program in order to minimize cost without any significant reduction in information, as identified above. A two year program would be the minimum to permit a statistical analysis.

6.4 Bacteriological Studies

The bacteriological studies in recreational waters conducted under the FEMP were limited in the scope of their conclusions due to non-compliance with the sampling requirements specified in the water quality guidelines at most of the study sites. The need for complete documentation of recreational water quality compliance with the fecal coliform guideline should be reviewed with the appropriate responsible agencies and the communities to determine

if additional sampling, according to the guideline protocol, is required.

6.5 General Recommendations

The results of the FEMP water quality study indicate the major changes in water quality caused by the Churchill-Nelson hydroelectric development have been increases in turbidity due to shoreline erosion, the release of phosphorus from flooded materials, the loss of organic carbon from the water column, and changes in water quality related to the mixing of water masses from different sources. These features should be considered in the design of monitoring programs for assessment of the effects of additional generation projects in the system. The requirements of a monitoring program for a specific generating station will be very dependent on its location, the proposed extent of flooding, and the operation regime.

The FEMP water quality study placed considerable emphasis on the measurement of metals. This emphasis could be reduced considerably in any future studies. Few changes in metals concentrations have occurred in the Churchill-Nelson system due to hydroelectric development, and then only as a result of the mixing of water masses with naturally different metals concentrations. There is no evidence of a general mobilization of metals from eroded soils and vegetation.

Unfortunately, the total mercury data could not be used because the limit of detection of the analytical method was well in

excess of the demonstrated actual concentrations and some apparent sample contamination. Should it be considered necessary to measure total mercury concentrations in water in the future, the sampling program should be designed to measure levels in the 0.5 to 3.0 ng L⁻¹ concentration range. This will require a dedicated mercury sampling program which employs clean sampling technique and an appropriate analytical method. The utility of any program to measure total mercury levels in water is questionable, given that other studies have shown the elevated mercury levels in fish from flooded lakes along the CRD are in no way related to total mercury concentrations in water or sediments, that total mercury levels in water are not elevated by impoundment, and that natural levels are well below current drinking water standards.

The number and location of stations sampled in the FEMP study appear to have been sufficient for documentation of the impacts at the sites which have been developed to date, covering the major components of the system. Additional stations would not improve the assessment due to the absence of pre-development data. The data collected to date also provide a substantial baseline for assessment of the impacts of future projects although additional stations may be required to examine specific or localized impacts. It would be premature to identify additional station locations at this time without knowledge of the specific nature of future projects. Three years of study prior to construction, with a sampling frequency comparable to that of the FEMP study, would be required to establish pre-development baseline conditions at any

new sampling station.

The results of the FEMP study indicate there is really no substitute for solid pre-development data. The natural rivers were somewhat useful for examining natural levels of change in the few variables which were common to the pre- and post-development periods, but were of no use for the filling in of data gaps at the regulated sites. The pre-development databases for the natural rivers were just as limited as for the regulated sites. The Aiken River data were of no use in this regard as there were no data from the pre-development period.

7.0 References

- American Public Health Association (APHA). 1985. Standard Methods for the Examination of Water and Waste Water. American Public Health Association, Washington, D.C.
- Baxter, R.M., and P. Glaude. 1980. Environmental effects of dams and impoundments in Canada: experience and prospects. Can. Bull. Fish. Aquat. Sci. 205: 34p.
- Berger, G.M., and D.J. Ramsey. 1991 (in review). Limnological changes in lakes of the Rat-Burntwood and Nelson river systems following Churchill River diversion and Lake Winnipeg regulation. Manitoba Department of Natural Resources, Fisheries Branch MS Report No. 91-XX.
- Canadian Council of Resource and Environment Ministers (CCREM). 1987. Canadian Water Quality Guidelines. Water Quality Objectives Division, Water Quality Branch, Inland Waters Directorate, Environment Canada.
- Cleugh, T.R. 1974. Hydrographic survey of lakes on the Lower Churchill and Rat-Burntwood rivers and reservoirs and lakes on the Nelson River. Lake Winnipeg, Churchill and Nelson Rivers Study Board, 1971-1975. Tech. Rep. Append. 5, Vol. 2E. Queen's Printer, Winnipeg, Manitoba.
- Duncan, D.A., and D.A. Williamson. 1988. Water chemistry / water discharge relationships within the Churchill River diversion and Lake Winnipeg Regulation Region, Manitoba, Canada. Manitoba Department of Environment and Workplace Safety and Health, Water Standards and Studies, Report 87-8. vi + 57p.
- Elliot, J.M. 1977. Some Methods for the Statistical Analysis of Samples of Benthic Invertebrates. Second Edition. Freshwater Biological Association Scientific Publication No. 25.
- Fudge, R.J.P., and R.A. Bodaly. 1984. Postimpoundment winter sedimentation and survival of lake whitefish (*Coregonus clupeaformis*) eggs in Southern Indian Lake, Manitoba. Can. J. Fish. Aquat. Sci. 41:701-705.
- Green, D.J. 1990. Summary of water chemistry data for the Nelson River and Rat-Burntwood lakes, 1986-1989. Manitoba Department of Natural Resources, Fisheries Branch MS Report No. 90-XX.
- Guilbault, R.A., W.D. Gummer, and V.T. Chacko. 1979. The Churchill Diversion: Water Quality Changes in the Lower Churchill and Burntwood Rivers. Fisheries and Environment Canada, Water Quality Interpretive Report No. 2. xi + 9p.

- Health and Welfare Canada. 1989. Guidelines for Canadian Drinking Water Quality. Fourth Edition. Minister of Supply and Services Canada.
- Health and Welfare Canada. 1983. Guidelines for Canadian Recreational Water Quality. Minister of Supply and Services Canada. 75p.
- Health and Welfare Canada. 1980. Guidelines for Canadian Drinking Water Quality 1978. Supporting Documentation. Supply and Services Canada, Hull.
- Hecky, R.E., and R.J. Harper. 1974. Phytoplankton and primary production of the lower Churchill lakes, Rat-Burntwood lakes, and Nelson River lakes and reservoirs. Lake Winnipeg, Churchill and Nelson Rivers Study Board, 1971-1975. Tech. Rep. Append. 5, Vol. 2F. Queen's Printer, Winnipeg, Manitoba.
- Hynes, H.B.N. 1970. The Ecology of Running Waters. University of Toronto Press, Toronto. 555 pp.
- Kellerhals Engineering Services Ltd. 1988. Morphological effects of the Churchill River diversion, volume 2, feasibility studies and monitoring recommendations. Northern Flood Agreement Manitoba, Ecological Report Series, Report Number 88-4.
- Lettenmaier, D.P. 1976. Detection of trends in water quality data from records with dependent observations. Water Resources Research. 12(5): 1037-1046.
- Morelli, M. 1974. Results of the water sampling program in northern Manitoba, 1972-74. Lake Winnipeg, Churchill and Nelson Rivers Study Board, 1971-1975. Tech. Rep. Append. 2, Vol. 2A. Queen's Printer, Winnipeg, Manitoba.
- Newbury, R.W., G.K. McCullough, and R.E. Hecky. 1984. The Southern Indian Lake Impoundment and Diversion. Can. J. Fish. Aquat. Sci. 41:548-557.
- Northwest Hydraulic Consultants Ltd. 1988. Assessment of sediment effects, Churchill River diversion, Manitoba, phase II report. Northern Flood Agreement Manitoba, Ecological Report Series, Report Number 88-3.
- Playle, R.C. 1986. Water chemistry changes associated with hydroelectric development in northern Manitoba: The Churchill, Rat, Burntwood, and Nelson rivers. Manitoba Department of Environment and Workplace Safety and Health, Water Standards and Studies, Report 86-8, Water Quality Data Supplement.

- Playle, R.C., and D.A. Williamson. 1986. Water chemistry changes associated with hydroelectric development in northern Manitoba: The Churchill, Rat, Burntwood, and Nelson rivers. Manitoba Department of Environment and Workplace Safety and Health, Water Standards and Studies, Report 86-8. viii + 50p.
- Ralley, W.E., and D.A. Williamson. 1990. Multivariate analysis of water chemistry changes following hydroelectric development in northern Manitoba, Canada. Manitoba Department of Environment, Water Standards and Studies Section, Report No. 90-.
- Ramsey, D.J. 1991. Analysis of fish mercury data from lakes on the Rat-Burntwood and Nelson River systems, 1983-1989. Manitoba Department of Natural Resources, Fisheries Branch MS Report No. 91-14, 84 + x pp.
- Ramsey, D.J. 1990. Measurements of methylation balance in Southern Indian Lake, Granville Lake, and Stephens Lake, Manitoba, 1989. Northern Flood Agreement Manitoba, Ecological Report Series, Report Number 90-3.
- Ramsey, D.J., L. Livingston, I. Hagenson, and D.J. Green. 1989. Evolution of limnological conditions in lakes of the Nelson and Rat-Burntwood river systems after Churchill River diversion and Lake Winnipeg regulation: I. An Overview. Manitoba Department of Natural Resources, Fisheries Branch, MS Report No. 89-15. vii + 89p.
- Schindler, D.W., R. Wagemann, R.B. Cook, T. Ruzsyczynski, and J. Prokopowich. 1980. Experimental acidification of Lake 223, Experimental Lakes Area: background data and the first three years of acidification. Can. J. Fish. Aquat. Sci. 37(3):342-354.
- Seidl, P. 1987. Water quality proposal for Northern Flood Agreement Study 1987/88. Environment Canada, Western and Northern Region, Water Quality Branch.
- Seidl, P., B. Dutka, V. Spence, and R. Webster. 1990. Self-administered bacteriological assessment program at Split Lake, Manitoba. Northern Flood Agreement Manitoba, Ecological Report Series, Report No. 90-2. v + 45p.
- Seidl, P., and B. Dutka. 1987 in prep. Microbiological survey of Northern Flood Agreement Community Recreational Water.
- Snedecor, G.W., and W.G. Cochran. 1980. Statistical Methods. Seventh Edition. The Iowa State University Press, Ames, Iowa.
- Strange, N.S. 1991. Water quality data for sites affected by Churchill River diversion and Lake Winnipeg regulation, northern Manitoba, 1986-1989.

- U.S. EPA. 1976. Quality Criteria for Water. U.S. Environmental Protection Agency, Washington, D.C. EPA-440/0-76-023.
- Vollenweider, R.A. 1979. Das Nährstoffbelastungskonzept als Grundlage für den externen Eingriff in den Eutrophierungsprozess stehender Gewässer und Talsperren. Z. Wasser-u. Abwasser-Forschung 12:46-56.
- Water Resources Branch. 1974. Physical Impact Study. Lake Winnipeg, Churchill and Nelson Rivers Study Board, 1971-1975. Tech. Rep. Append. 2, Vol. 1E. Queen's Printer, Winnipeg, Manitoba.
- Water Survey of Canada. 1990. Historical streamflow summary, Manitoba to 1989. Supply and Services Canada.
- Wetzel, R.G. 1983. Limnology, second edition. Saunders College Publishing, Toronto. 767 pp.
- Wilkinson, L. 1988. SYSTAT: The System for Statistics. SYSTAT Inc. Evanston, IL.
- Williamson, D.A. 1988a. A four year study of bacteriological characteristics at recreational beaches, Manitoba, Canada. Manitoba Department of Environment and Workplace Safety and Health, Water Standards and Studies, Report No.88-7.
- Williamson, D.A. 1988b. Manitoba Surface Water Quality Objectives. Manitoba Department of Environment and Workplace Safety and Health, Water Standards and Studies.

8.0 Glossary of Technical Terms, Chemical Symbols, and Units of Measure

Al: Chemical symbol for the element aluminum.

As: Chemical symbol for the element arsenic.

Ba: Chemical symbol for the element barium.

Biocide: a substance which is destructive to many different organisms (e.g., DDT).

Ca: Chemical symbol for the element calcium.

Cd: Chemical symbol for the element cadmium.

Chlorophyll a: A photosynthetic pigment in plants. The concentration is used as a measure of algal biomass.

Cl: Chemical symbol for the element chlorine.

Co: Chemical symbol for the element cobalt.

Coefficient of variation (c.v.): Calculated as the ratio of the standard deviation to the mean, expressed as a percentage. A measure of the degree of variation in an observed quantity (e.g., river discharge).

CRD: Churchill River diversion.

Cu: Chemical symbol for the element copper.

Detection limit: The smallest concentration or amount of a substance which can be reported as present with a specified degree of certainty by a definite, complete, analytical method.

DIC: Dissolved inorganic carbon.

Dissolved metal: A metal constituent in an unacidified sample that passes through a 0.45 μm membrane filter.

EC₅₀: Median effective concentration. The concentration of material in water to which test organisms are exposed that is estimated to be effective in producing some sublethal response in 50% of the test organisms. The EC₅₀ is usually expressed as a time-dependent value (e.g., 24-h or 96-h EC₅₀).

Extractable metal: The concentration of a metal in solution after treatment of an unfiltered sample with hot dilute mineral acid. Applies to a method which will remove most but not all of a metal from a sample.

F: Chemical symbol for the element fluorine.

FEMP: Federal Ecological Monitoring Program.

Fulvic acid: One of three classes of humic substances defined as those substances which are soluble in both acid and base.

Guideline (water quality): A numerical concentration limit or narrative statement which has been established to support and maintain a designated water use.

Hg: Chemical symbol for the element mercury.

Humic acid: One of three classes of humic substances defined as those substances which precipitate with acidification.

Humic substance: Form the majority of organic matter in soil and water. Consist of dark-coloured and acidic compounds with molecular weights ranging from hundreds to thousands of daltons. Resistant to microbial degradation, these substances tend to persist in aquatic systems.

K: Chemical symbol for the element potassium.

LWCNR: Lake Winnipeg and Churchill and Nelson Rivers Project.

LWR: Lake Winnipeg regulation.

Mg: Chemical symbol for the element magnesium.

Mn: Chemical symbol for the element manganese.

N: Chemical symbol for the element nitrogen.

Na: Chemical symbol for the element sodium.

NFA: Northern Flood Agreement.

NFC: Northern Flood Committee.

NFR: Non-filterable residue. A measure of the concentration of suspended sediments in water. Defined as that mass of material in a volume of water which does not pass through a 0.45 μm membrane filter.

NH_3 : Chemical symbol for the compound ammonia.

Ni: Chemical symbol for the element nickel.

NO_2 - NO_3 : Chemical symbol used to identify the compounds nitrite (NO_2) and nitrate (NO_3).

ortho PO_4 : Phosphorus occurring as the compound PO_4^{3-} . Considered to represent the concentration of inorganic soluble phosphorus, the form directly available to aquatic plants.

P: Chemical symbol for the element phosphorus.

Pb: Chemical symbol for the element lead.

pH: Value which represents the acidity or alkalinity of an aqueous solution. It is defined as the negative logarithm of the concentration of hydrogen ions in a solution.

POC: Particulate organic carbon.

Primary production: The production of organic matter from inorganic material by organisms.

Q: Symbol used to represent river discharge.

Range: The difference between the lowest and highest value in a set of data.

React Si: Soluble reactive silica.

Replicate: A repeated measurement of a variable under specific conditions.

Se: Chemical symbol for the element selenium.

Si: Chemical symbol for the element silicon.

SIL: Southern Indian Lake.

SO_4 : Chemical notation for the compound sulphate which consists of 1 sulphur and 4 oxygen atoms.

Sublethal: Involving a stimulus below the level which causes death.

Suspended metal: Those metals in an unacidified sample which are retained on a $0.45 \mu\text{m}$ membrane filter.

TDN: Total dissolved nitrogen.

TDP: Total dissolved phosphorus.

TDS: Total dissolved solids.

TIC: Total inorganic carbon.

TKN: Total Kjeldahl nitrogen. Total concentration of organic nitrogen and ammonia nitrogen.

TOC: Total organic carbon.

Total metal: The concentration of a metal in an unfiltered sample that is digested in strong nitric acid. Refers to a method which removes all traces of a metal from a sample for analysis.

True colour: The colour of water resulting from substances which are totally in solution: not to be mistaken for apparent colour resulting from colloidal or suspended matter.

V: Chemical symbol for the element vanadium.

Zn: Chemical symbol for the element zinc.

Appendix 1: Bacteriological Study Methods

Sampling Site Descriptions and Locations

Seasonal bacterial studies were conducted in the vicinity of the communities of Norway House, Cross Lake, Nelson House, Split Lake, and York Landing. Seasonal study site numbers and descriptions are listed in Tables A1, A2, A3, A4, and A5 and site locations are identified in Figures A1, A2, A3, A4, and A5. Intensive bacterial studies were conducted near all communities except York Landing. Intensive study site numbers and descriptions are listed in Tables A6, A7, A8, and A9 and site locations are identified in Figures A6, A7, A8, A9, A10, A11, A12, and A13.

Sampling and Analytical Methods

Water samples for fecal coliform determination were collected from shoreline beach areas near the NFA communities by wading into the water, or by boat in deeper embayment areas. Subsurface (0.5 m depth) water samples were collected in 1 L wide-mouthed autoclaved glass jars. The mouth of the sample was pointed in the upstream direction of the waterbody to eliminate any potential contamination from the sample collector, then immediately sealed. Samples were placed in ice-cooled coolers or trays and transported to the Water Quality Branch facilities in Thompson, Manitoba, for analysis.

Fecal coliform bacterial densities in the water samples were determined using the membrane filter (MF) procedure, as described in APHA (1985), with the following modifications. Known volumes of the raw water samples were vacuum filtered through 47 mm diameter,

0.45 μm pore-size, HC type, membrane filters. Sample volume was determined from site observations. Three volumes, ranging from 1 to 100 mL, were normally filtered for each station. Samples were incubated at 44.5 °C for 48 hours. Incubations commenced within 6 hours of sample collection. The results of the 3 bacterial counts for each sample were standardized to a 100 mL volume. Reported values are the arithmetic mean of the 3 counts.

Table A1. Bacteriological water quality sampling sites, Norway House, Manitoba, 1987. See Figure A1 for site locations.

	Site Number	Site Description
Seasonal Study	1	Green Dock adjacent to Band Office
	2	School Dock
	3	Government Dock, east of STP outfall
	4	Rocky Place near graveyard
	5	Asiniwapiska, "where the rocks are"
	6	Sewage Outlet Bay at Church Point
	7	Bridge into Fort Island
	8	Historic Dock by Bay Store Bridge
	9	Water intake

STP Sewage Treatment Plant

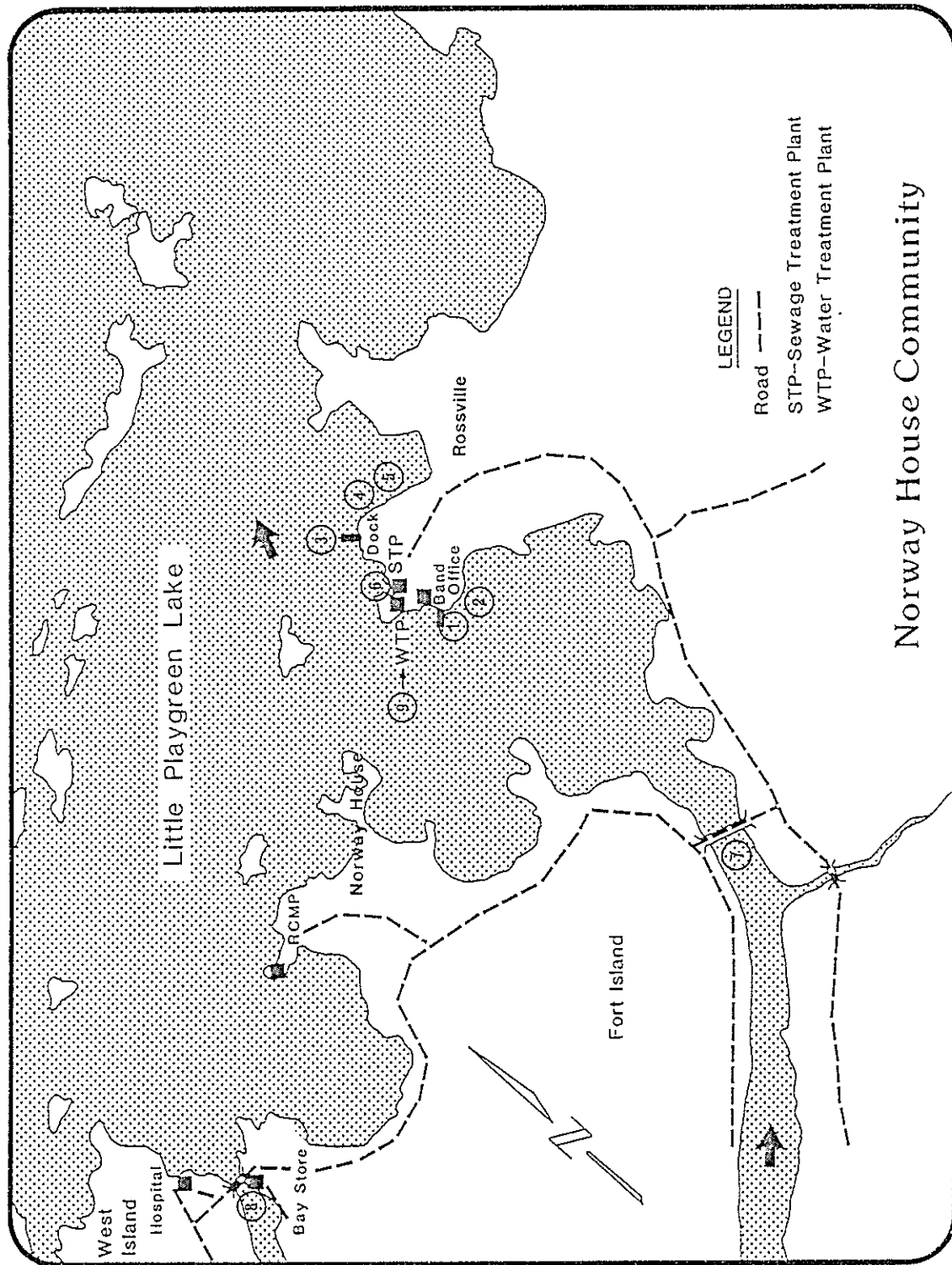


Figure A1. Location of 1987 seasonal bacterial study sampling sites near the community of Norway House, Manitoba. See Table A1 for site descriptions.

Table A2. Bacteriological water quality sampling sites, Cross Lake, Manitoba, 1987. See Figure A2 for site locations.

	Site Number	Site Description
Seasonal Study	1	Saggitawak, swimming area near Roman Catholic church
	2	Beardy's Point, swimming area near Saggitawak Bridge
	3	Saggitawak Bridge
	4	Wapak swimming area, near Junior High School, power line, and sewage area
	5	Natimeek Water Treatment Bay, to the right of water intake
	6	Natimeek - Robinson's Point, Water Treatment Bay to the left of the water intake
	7	Sand Bay
	8	Catherine's Point

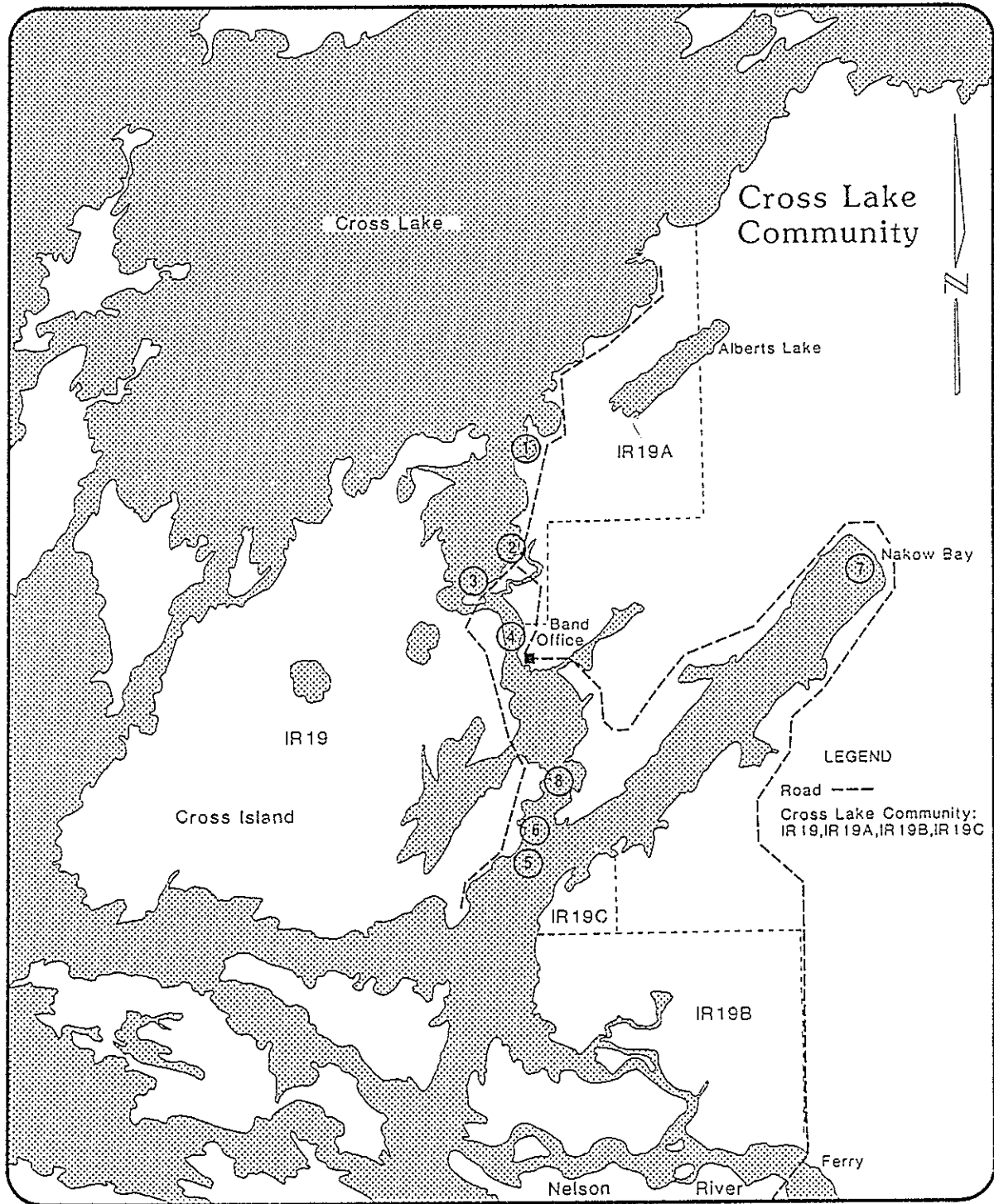


Figure A2. Location of 1987 seasonal bacterial study sampling sites near the community of Cross Lake, Manitoba. See Table A2 for site descriptions.

Table A3. Bacteriological water quality sampling sites, Nelson House, Manitoba, 1987. See Figure A3 for site locations.

	Site Number	Site Description
Seasonal Study	1	Metis Beach
	2	New Development Area, across from cemetery
	3	Church Bay (East Ramp), nearest the church
	4	Church Bay (West Ramp), furthest from the church
	5	Industrial Bay, by Causeway #2 near old industrial area
	6	School Bay, at outlet of sewage effluent
	7	Nursing Station Bay, near dock by pumphouse
	8	Inspiration Point, near dock, west of Nursing Station Bay
	9	Government Dock, most popular dock, just west of Bay Store, below the embankment
	10	Michael Bay
	11	WTP, raw water inside pumphouse
	12	Dog Point, bay east of Industrial Bay at Causeway #1

WTP Water Treatment Plant

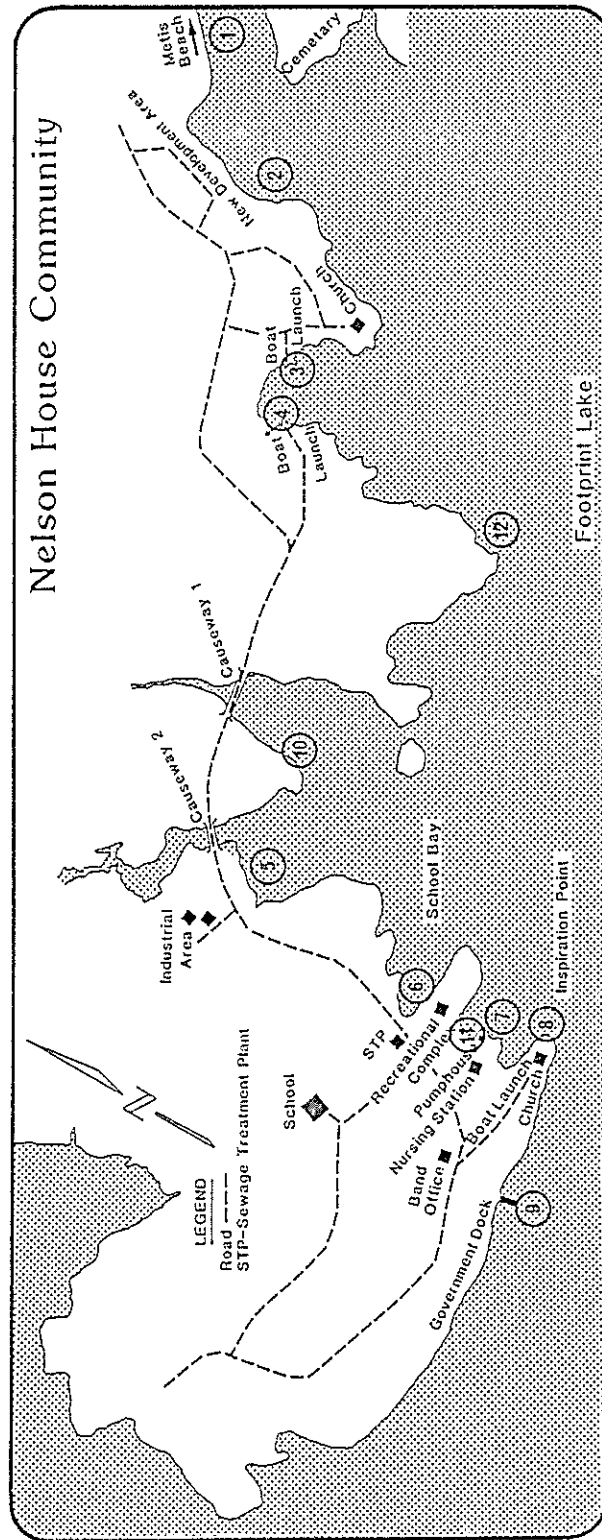


Figure A3. Location of 1987 seasonal bacterial study sampling sites near the community of Nelson House, Manitoba. See Table A3 for site descriptions.

Table A4. Bacteriological water quality sampling sites, Split Lake, Manitoba, 1987. See Figure A4 for site locations.

	Site Number	Site Description
Seasonal Study	1	Nursing Station Bay, opposite Nursing Station
	2	Bay Store Beach Area, at end of Bay Store back road
	3	First Beach, adjacent to Sewage outfall
	4	Beach Point, between Nursing Station and Bay Store
	5	Second Beach, near Ferry Dock
	6	School A (1)
	7	School B (3)
	8	WTP, treated water
	9	WTP, untreated water
	10	Water Treatment Plant Beach, east of old plant
	11	Water Treatment Bay, west of old plant
	12	Sewage Treatment Bay, near First Beach

WTP Water Treatment Plant

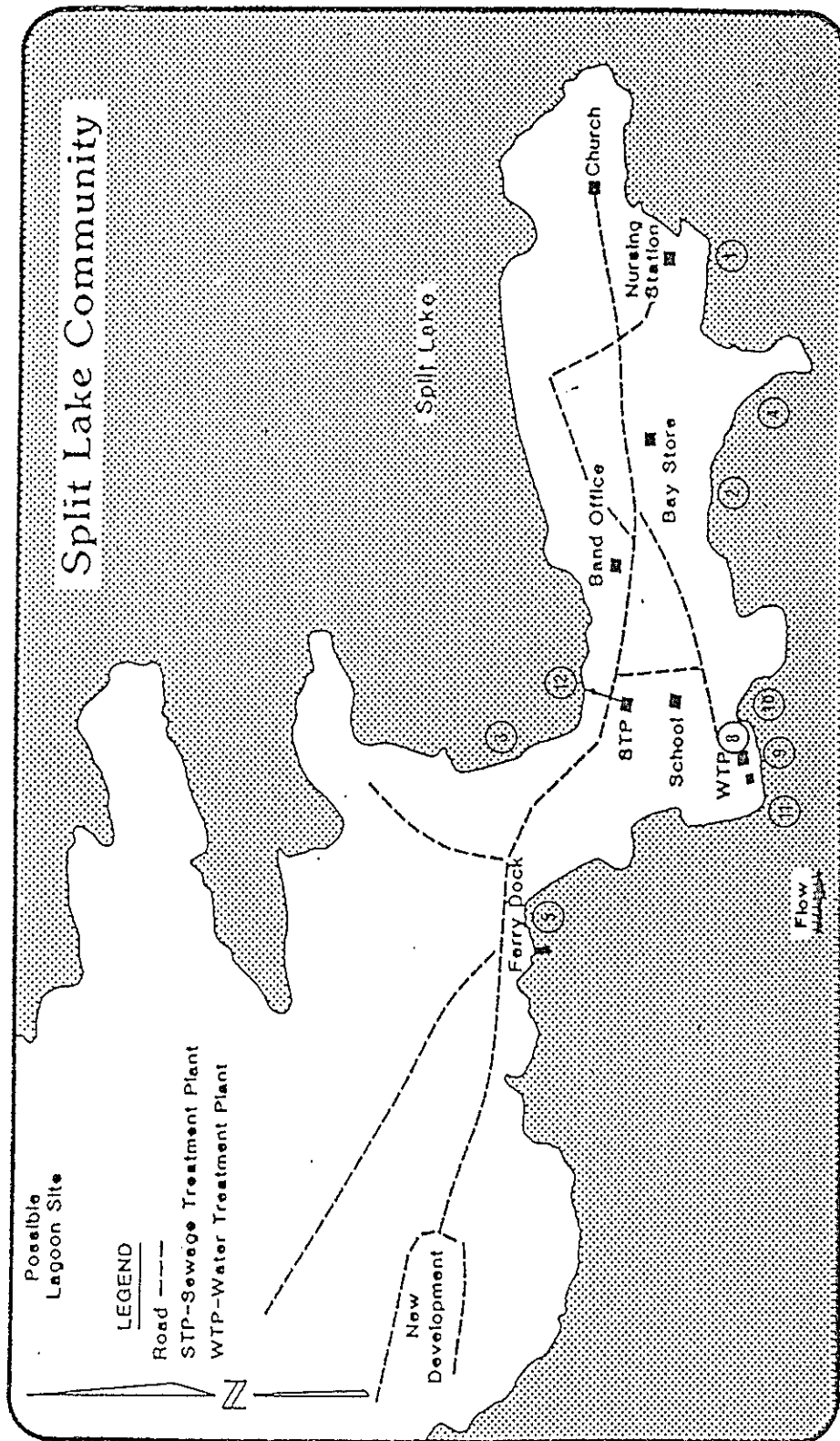


Figure A4. Location of 1987 seasonal bacterial study sampling sites near the community of Split Lake, Manitoba. See Table A4 for site descriptions.

Table A5. Bacteriological water quality sampling sites, York Landing, Manitoba, 1987. See Figure A5 for site locations.

	Site Number	Site Description
Seasonal Study	1	Gravel Pit
	2	Water Treatment Plant holding tank
	3	Ferry Dock, east side
	4	Drainage Ditch outflow into river on west side of Ferry Dock
	5	Island Beach
	6	Sandy Beach
	7	Bearass Beach

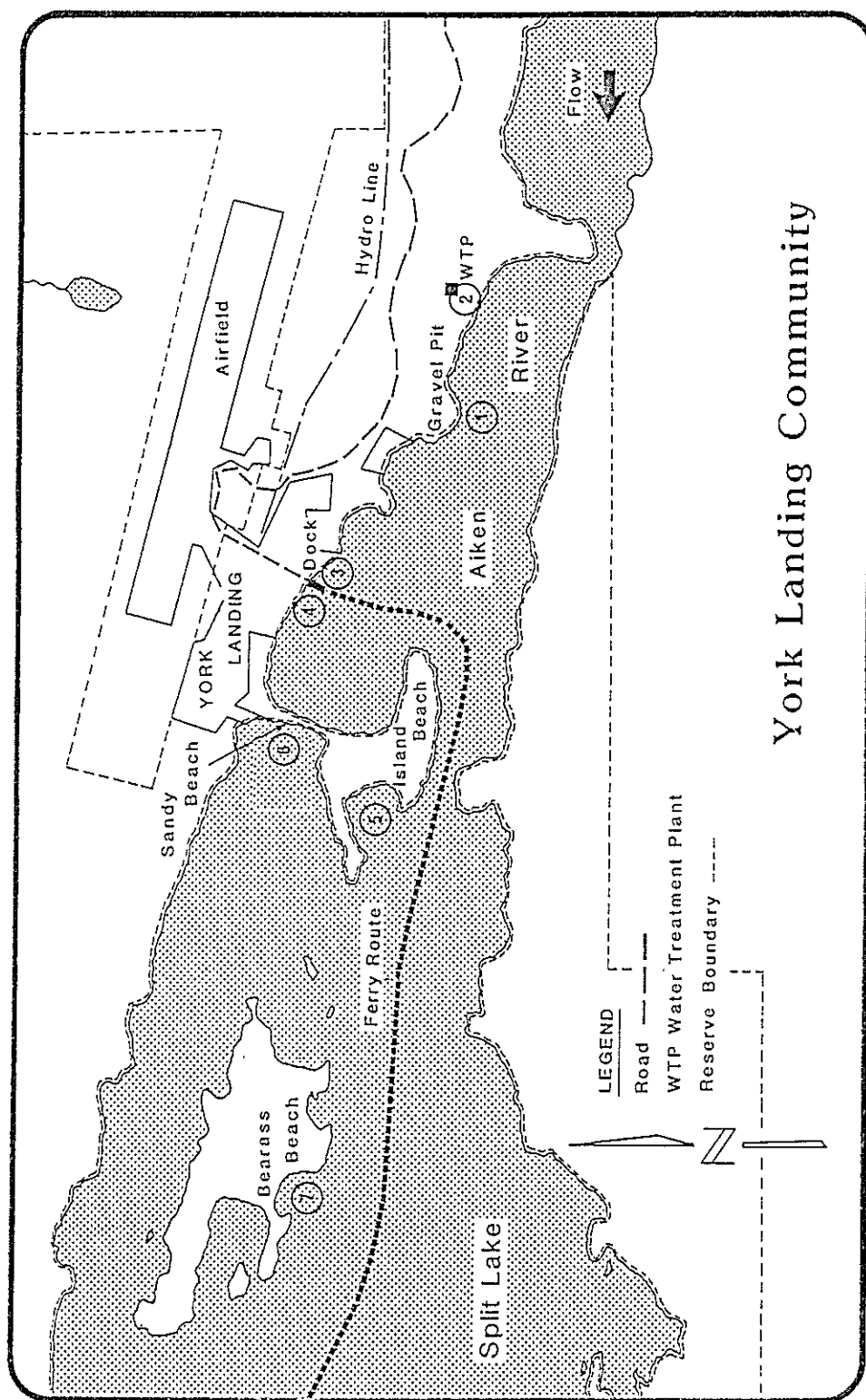


Figure A5. Location of 1987 seasonal bacterial study sampling sites near the community of York Landing, Manitoba. See Table A5 for site descriptions.

Table A6. Bacteriological water quality sampling sites, Norway House, Manitoba, 1987. See Figures A6 and A7 for site locations.

	Site Number	Site Description
Intensive Study	1	10 m from shoreline of STP in small bay
August 11	2	30 m from STP outfall, near south shore
	3	30 m from STP outfall, near north shore opposite warning sign of underwater cable
	4	60 m from STP outfall and 40 m from warning sign of underwater cable
	5	5 m from south shore and 60 m from STP outfall
	6	200 m from STP and 180 m from dock on south shore
	7	Adjacent to dock on south shore, 180 m from STP outfall
	8	20 m from Monument on Church Point
	9	10 m from WTP intake
	10	20 m from WTP intake
	11	10 m from island opposite WTP, 60 m from WTP
Intensive Study	1	100 m from dock on shoreline, locale of STP outfall
August 24	2	150 m from STP outfall, 50 m from south shore
	3	10 m from south shore and 150 m from STP small dock
	4	20 m from south shore at small wharf before Government Dock
	5	100 m from south shore opposite small wharf and site 4
	6	At the Government Dock, 10 m from south shore (seasonal study site 3)
	7	150 m from STP outfall opposite Church Point, north west shoreline
	8	50 m from north west shoreline next to Church Point (transect of Church Island)
	9	WTP intake, church in line with rock 250 m from shore (seasonal study site 9)
	10	200 m from WTP intake, across from WTP

STP Sewage Treatment Plant discharge in Rossville

WTP Water Treatment Plant

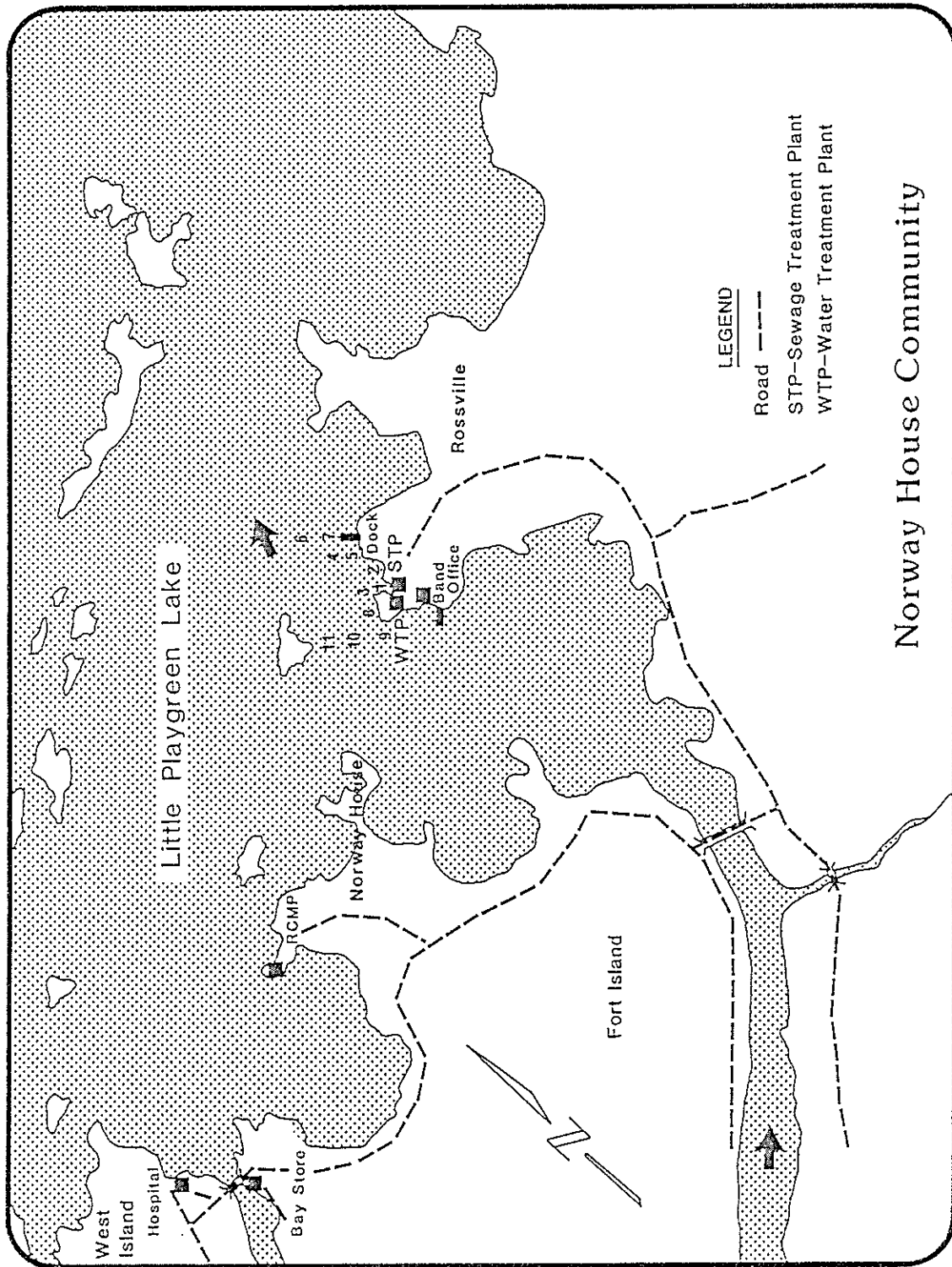


Figure A6. Location of bacterial sampling sites for the August 11, 1987, intensive study near the community of Norway House, Manitoba. See Table A6 for site descriptions.

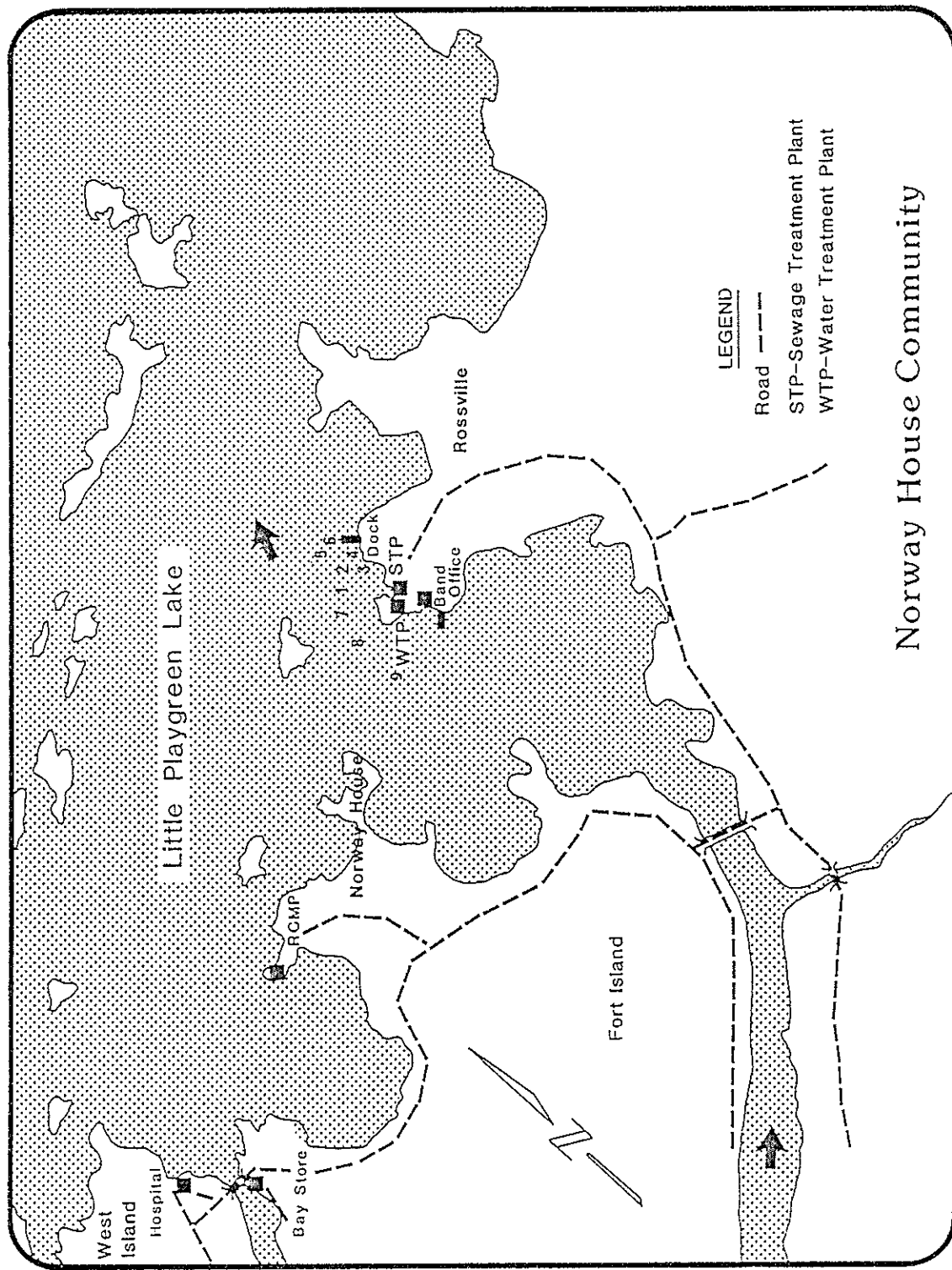


Figure A7. Location of bacterial sampling sites for the August 24, 1987, intensive study near the community of Norway House, Manitoba. See Table A6 for site descriptions.

Table A7. Bacteriological water quality sampling sites, Cross Lake, Manitoba, 1987. See Figures A8 and A9 for site locations.

	Site Number	Site Description
Intensive Study	1	WTP, raw water
August 11	2	150 m from wet well
	3	10 m from wet well
	4	75 m from wet well
	5	150 m from wet well, in line with intake trench
	6	10 m from wet well, in line with intake trench
	7	75 m from wet well, in line with intake trench
	8	150 m from wet well, in line with wet well and house
	9	100 m from wet well, in line with wet well and house
	10	10 m from wet well, in line with wet well and house
	11	Water Treatment Bay, at water intake, intensive study site 7
Intensive Study	1	WTP, raw water
August 25	2	10 m from wet well
	3	75 m from wet well
	4	150 m from wet well
	5	10 m from wet well, in line with intake trench
	6	75 m from wet well, in line with intake trench
	7	150 m from wet well, in line with intake trench
	8	10 m from wet well, in line with wet well and house
	9	100 m from wet well, in line with wet well and house
	10	150 m from wet well, in line with wet well and house

WTP Water Treatment Plant

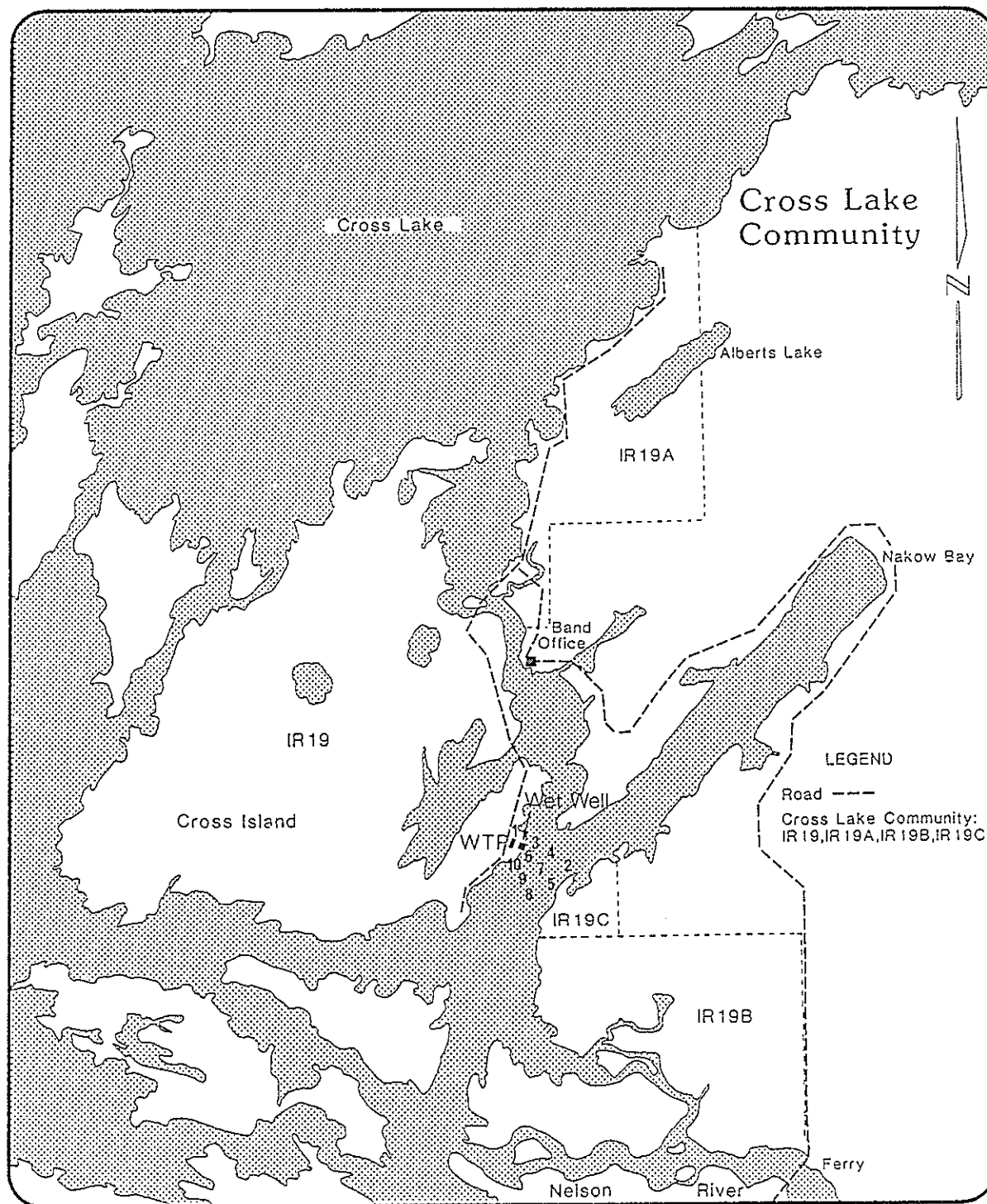


Figure A8. Location of bacterial sampling sites for the August 11, 1987, intensive study near the community of Cross Lake, Manitoba. See Table A7 for site descriptions.

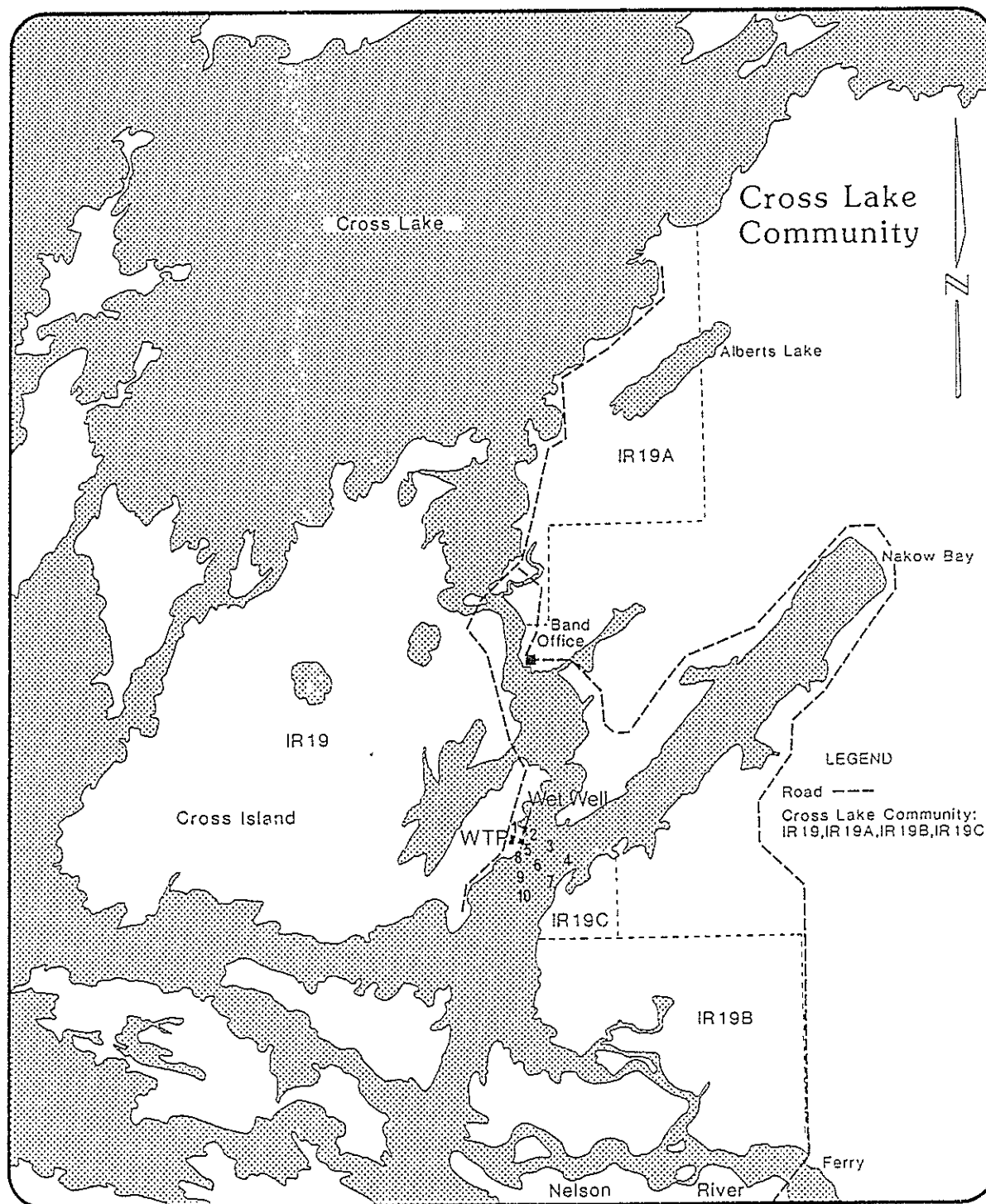


Figure A9. Location of bacterial sampling sites for the August 25, 1987, intensive study near the community of Cross Lake, Manitoba. See Table A7 for site descriptions.

Table A8. Bacteriological water quality sampling sites, Nelson House, Manitoba, 1987. See Figures A10 and A11 for site locations.

	Site Number	Site Description
Intensive Study	1	60 ft. from sewage outflow, intensive study site 6
August 12	2	25 ft. from shore, 300 ft. from sewage outflow, intensive study site 6 (in line with lone white tree and Duncan Hall)
	3	150 ft. from shore, 300 ft. from sewage outflow, intensive study site 6, (in line with lone white tree and Duncan Hall)
	4	mid-bay, 400 ft. from sewage outflow
	5	100 ft. from shore, 400 ft. from sewage outflow
	6	40 ft. from shore, 800 ft. from sewage outflow
	7	275 ft. from shore, in line with white house on land, 800 ft. from sewage outflow
	8	850 ft. from sewage outflow, 175 ft. from shore
	9	100 ft. from shore, halfway around bend from School Bay to Water Treatment Bay (across from cemetery)
	10	120 ft. from shore, at edge of bend between School Bay and Water Treatment Bay
	11	Water Treatment Bay, at water intake, seasonal study site 7
Intensive Study	1	20 m from sewer outfall in mid-bay
August 25	2	10 m from west shore next to Duncan Hall
	3	50 m from west shore next to Duncan Hall
	4	130 m from sewer outfall in mid-bay
	5	130 m from sewer outfall 30 m from west shore (Duncan Hall)
	6	260 m from sewer outfall and 13 m from west shore next to a floating dock on point
	7	260 m from sewer outfall and 100 m from the west shore
	8	30 m from south shore, west of site 6
	9	30 m from south shore, west of site 8, next to the edge of point of land
	10	at the water intake pipe opposite the pumphouse

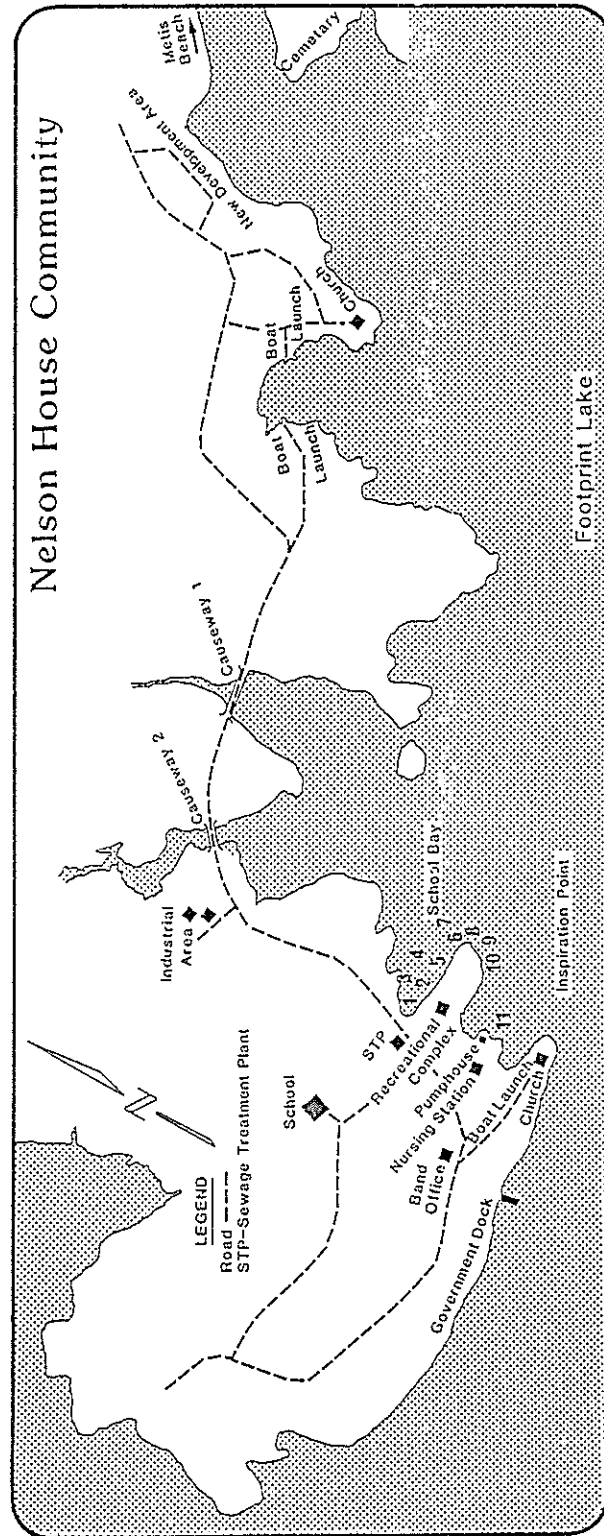


Figure A10. Location of bacterial sampling sites for the August 12, 1987, intensive study near the community of Nelson House, Manitoba. See Table A8 for site descriptions.

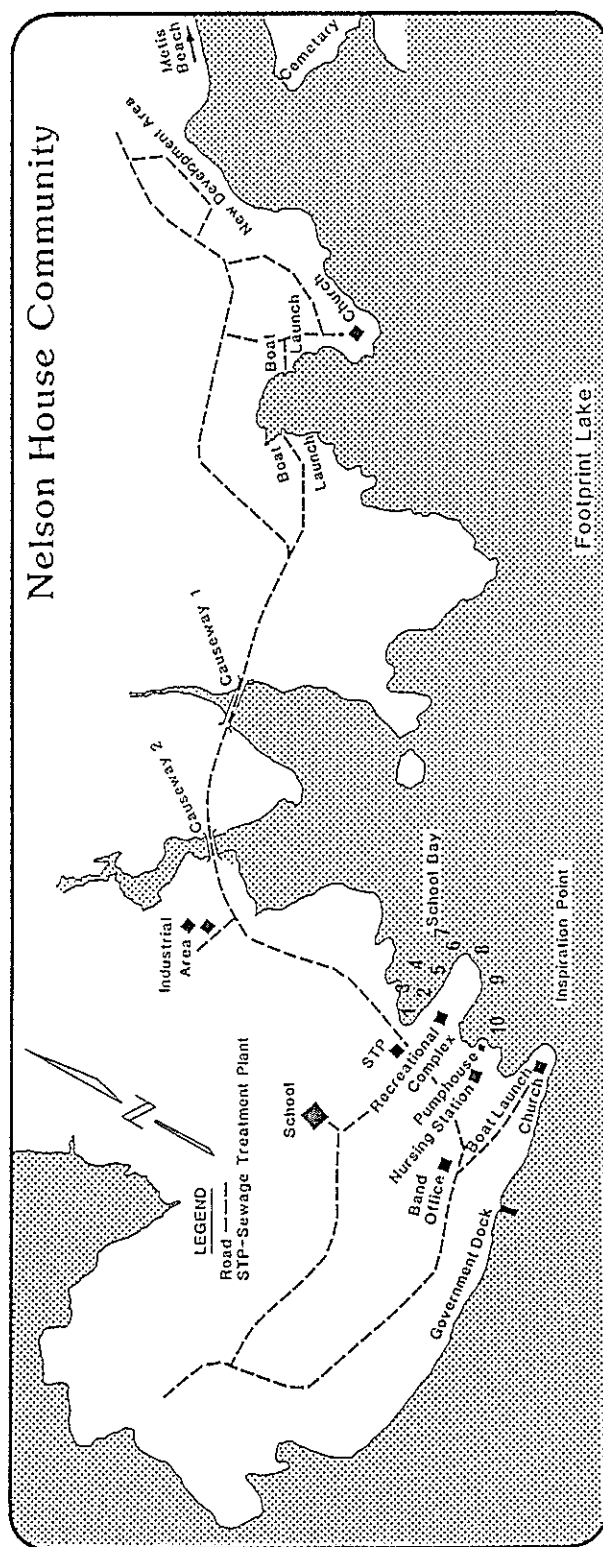


Figure A11. Location of bacterial sampling sites for the August 25, 1987, intensive study near the community of Nelson House, Manitoba.
See Table A8 for site descriptions.

Table A9. Bacteriological water quality sampling sites, Split Lake, Manitoba, 1987. See Figures A12 and A13 for site locations.

	Site Number	Site Description
Intensive Study	1	NW corner of 1st Beach Bay past bedrock outcropping
August 12	2	NW at junction of bedrock outcrop and beach, 10 m from beach
	3	SW side of 1st beach at edge of beach and rock deposits, 10m from beach
	4	SW of STP outfall opposite large boulder on shoreline, 10 m from shore
	5	10 m from and opposite to the STP outfall
	6	30 m from and opposite to the STP outfall
	7	N side of 1st Beach Bay opposite STP outfall by 100 m and east of station 1
	8	East of station 7, 30 m from N shore of Bay
	9	S of station 8, mid-bay and 50 m from south shore
	10	10 m from S shore opposite transformer on old shack
Intensive Study	1	150 m from centroid west shore bay
August 26	2	150 m from centroid on shore of bay, between sites 1 and 3
	3	150 m from centroid east shore of bay
	4	75 m from centroid east shore of bay
	5	75 m from centroid on shore of bay, between sites 4 and 6
	6	75 m from west shore of bay
	7	20 m from centroid on mid-shore of bay
	8	20 m from east shore of bay
	9	Seasonal Study Site #12, STP in 1st Beach Bay
	10	Seasonal Study Site #11, WTP site, west of plant

STP Sewage Treatment Plant

WTP Water Treatment Plant

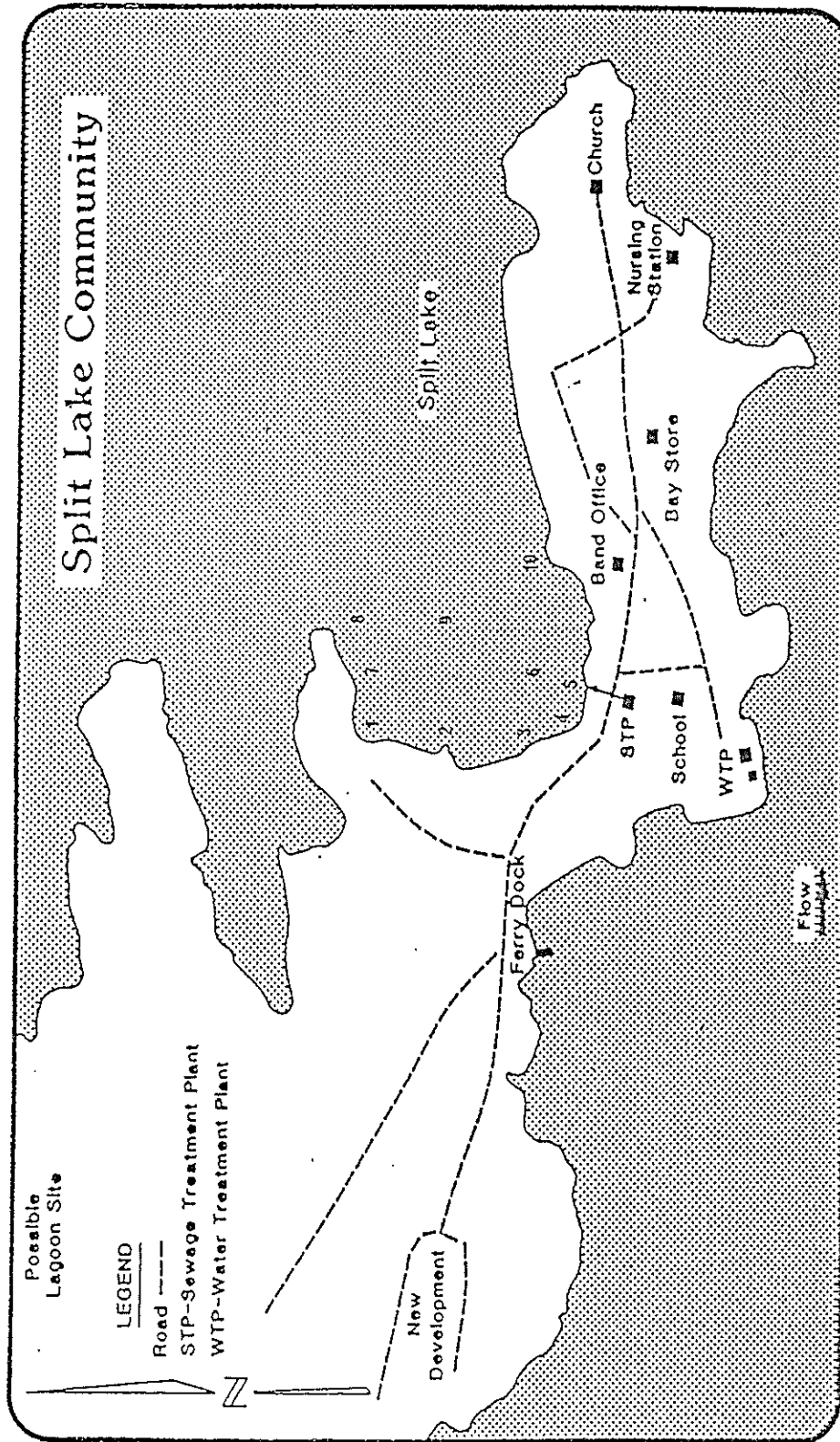


Figure A12. Location of bacterial sampling sites for the August 12, 1987, intensive study near the community of Split Lake, Manitoba. See Table A9 for site descriptions.

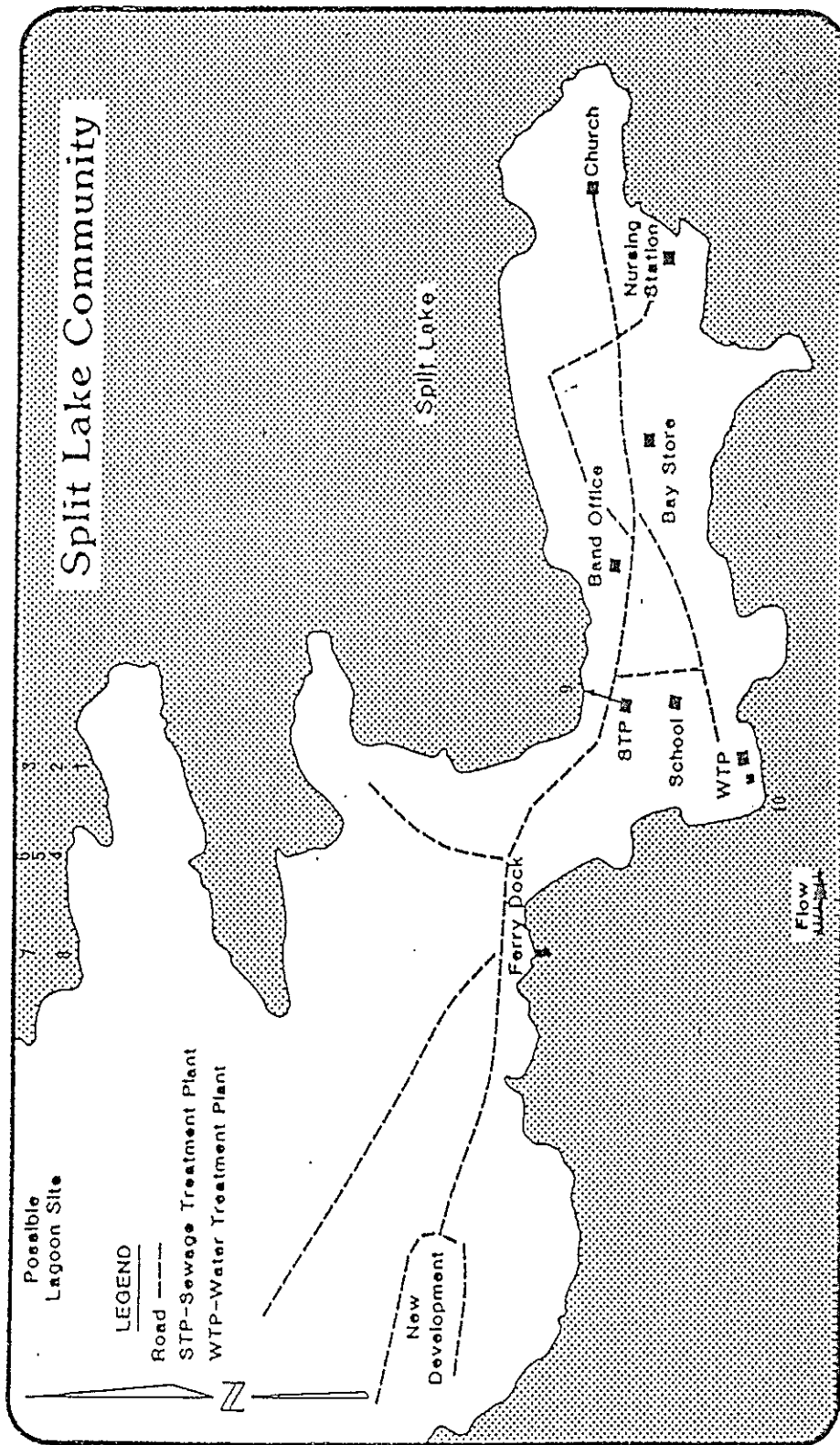


Figure A13. Location of bacterial sampling sites for the August 26, 1987, intensive study near the community of Split Lake, Manitoba. See Table A9 for site descriptions.

**THE EFFECT OF THE LAKE WINNIPEG REGULATION HYDROELECTRIC PROJECT
ON THE STOCK GENETICS OF LAKE WHITEFISH (Coregonus clupeaformis)
IN PLAYGREEN LAKE AND NORTHERN LAKE WINNIPEG: PRELIMINARY RESULTS**

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Crescent, Winnipeg, MB R3T 2N6.

ABSTRACT

The main objective of this study was to test the null hypothesis that the construction of the Lake Winnipeg Regulation Project (LWR) has not caused significant changes in the genetic relationships among lake whitefish (Coregonus clupeaformis) stocks in the LWR development area. A secondary objective was the description of the post-LWR genetic stock composition of lake whitefish in Playgreen Lake and the north basin of Lake Winnipeg. To accomplish this, the genetic composition of six spawning aggregations of lake whitefish collected in 1989 was examined and compared to the genetic composition of four spawning aggregations collected in 1975 (Kristofferson and Clayton 1990). Starch gel electrophoresis of tissue isozymes was utilized. Sixteen loci were deemed polymorphic in the 1989 sample but due to certain constraints posed by the 1975 samples, only five polymorphic loci were used for temporal analysis between 1989 and 1975. Allelic frequencies differed significantly among stocks ($P < 0.05$) at two of the five polymorphic loci (MDH-B1,2) compared, indicating that lake whitefish in northern Lake Winnipeg and Playgreen Lake can be differentiated into two distinct stocks (Playgreen Lake and Lake Winnipeg). These two differentiated stocks were present before LWR and they have remained genetically distinct after the construction of LWR. These preliminary results indicate that allelic frequencies have been stable over time and support the null hypothesis that the genetic structure of the stocks has not changed since the construction of LWR. Morphological analyses should be conducted on the lake whitefish samples to determine if additional structuring exists amongst the six spawning aggregations that were sampled, and to determine if the stability seen in isozyme allele frequencies is reflected in the morphology of the spawning aggregations.

INTRODUCTION

Larkin (1972) has defined a stock of fish as a population which shares a common environment and gene pool. This definition implies a degree of reproductive isolation between various stocks of a particular fish species in a particular area. For freshwater fishes, reproductive isolation of stocks often results from the homing behavior of individual fish to return to the place of hatching. Horrall (1981) has discussed the mechanisms for the reproductive isolation of stocks of freshwater fishes. Reproductive isolation will generally be expected to result in genetic differences between stocks, which will tend to accumulate due to random genetic drift and/or selection. Larkin's definition of fish stocks also implies that different fish stocks of the same species utilize different environments for spawning, egg incubation and early life stages, when environmental influences can alter morphological traits. Therefore, differing environmental conditions will lead to morphological differences between stocks which are not based solely on genetic differences, i.e. they are at least in part environmentally induced. This is especially the case for coregonid fishes in which extreme morphological plasticity is present (Lindsey 1981).

The genetic differences expected among various stocks of fish and the environmental differences present in their habitats have allowed the development of methods to identify and differentiate stocks (Ihssen et al. 1981). Stock characteristics are usually measured in samples of fish taken from spawning areas at the time of spawning, when stocks would be expected to have segregated themselves into reproductively isolated units. Morphological differences between stocks result from genetic and/or environmental differences and these can be measured directly. Differences in behavioral or physiological characters can also

be tested. The geographic limits to stock movements are often tested by the marking of fish (e.g., Pollard 1973). Direct assessment of genetic differences is possible by a number of means, the most common of which is the electrophoresis of alternate forms of functional enzymes (i.e. isozymes). Genetic characteristics, as measured by electrophoresis, are not environmentally modifiable and therefore a primary tool in identifying stocks of fish. Gel electrophoresis has been used to study intraspecific genetic variation of coregonids (Clayton and Franzin 1970; Clayton et al. 1973; Franzin and Clayton 1977; Imhof et al. 1980; Vuorinen 1984). Starch gel electrophoresis is the separation of a mixture of electrically charged molecules in an electric field through a starch gel. It is one of the most useful techniques yet devised for studying genetic variability within and among populations of organisms (Aebersold et al. 1987).

The study of fish stocks is of great importance in fisheries management and in the assessment of environmental impacts. Stocks are the natural and logical unit of management for fisheries and each stock should in theory be harvested separately because each stock can have distinct rates of growth, reproduction and mortality so fishing efforts must be distributed in accordance to the productivity of each stock. Mixed harvesting of stocks can lead to inappropriate harvest rates for individual stocks and even the extirpation of some stocks. Stock genetics is also an increasingly utilized tool in the measurement and study of environmental impacts on fish populations and fisheries. For example, Bodaly et al. (1984) were able to show that a significant change in the stock genetics of lake whitefish (Coregonus clupeaformis) had taken place in Southern Indian Lake, Manitoba and adjacent water bodies. These changes took place concurrently with disruptions in fish populations caused by the impoundment of the lake and the blockage of fish migrations via the natural outlet of the lake.

Populations of coregonid fishes in large lake systems have been shown in many instances to be divided into discrete stocks (e.g., Imhof et al. 1980; Casselman et al. 1981). In a study of lake whitefish populations in Lake Winnipeg and connecting water bodies which was based on samples collected in 1975, Kristofferson (1978 and Kristofferson and Clayton 1990), utilized morphological comparisons to identify three different stocks of lake whitefish in the north end of Lake Winnipeg and Playgreen Lake. Distinct stocks were noted for Grand Rapids on the northwest shore of Lake Winnipeg, for Big Black, Poplar and Berens Rivers on the northeast shore of Lake Winnipeg, and for Little Playgreen Lake, on the Nelson River outlet of Lake Winnipeg. Four genetic loci were found to be polymorphic but there were no significant differences in allelic frequencies for these populations (Kristofferson 1978; Kristofferson and Clayton 1990). These published results form part of the pre-development baseline for the present study.

There is evidence that there are movements of lake whitefish between Playgreen Lake and the north end of Lake Winnipeg. Some stocks migrate downstream from their feeding grounds in Lake Winnipeg to utilize Playgreen Lake for spawning during late fall and then reverse their migration upstream into Lake Winnipeg to overwinter and feed. Pollard (1973) tagged 2 934 lake whitefish at Warren Landing, at the outlet of Lake Winnipeg, between September 25th and October 7th 1970 and noted that 16 fish were recaptured downstream in the river channels (mouth of Gunisao River) by Norway House and in Little Playgreen Lake. Changes in the geographic and/or temporal distribution of flows between Lake Winnipeg and Playgreen Lake might be expected to have disrupted migration patterns between the two lakes, and possibly also the genetic structure of stocks in the area. Subsequent to the construction and operation of the Lake Winnipeg Regulation (LWR) hydroelectric project, fishermen on Playgreen Lake claimed that

the spatial distributions and seasonal movements of lake whitefish in the Lake Winnipeg and Playgreen Lake areas have changed. Fishermen have also reported decreased catches of lake whitefish and have attributed this decline in catch to deleterious effects caused by the construction of LWR.

The main emphasis of this investigation was to determine whether the genetic structuring of the populations of lake whitefish in northern Lake Winnipeg and in Playgreen Lake has changed as a result of the construction and subsequent operation of LWR. This study deals with the identification of stocks of lake whitefish in northern Lake Winnipeg and in Playgreen Lake (pre-LWR and post-LWR) by utilizing biochemical genetic techniques for the analysis of genetically determined characteristics. The null hypothesis that there has been no significant change will be tested by comparing the genetic stock structure of lake whitefish populations in 1989 to the genetic stock structure present in 1975. Additional genetic analyses of 1975 samples (Kristofferson 1978) will be conducted to provide a more detailed genetic baseline. The approach deals with the testing of samples from spawning aggregations of fish for both within-group and among-group homogeneity (Shaklee 1984; Winans 1980; Richardson 1983).

The identification of post-LWR stocks of lake whitefish utilizing genetic analysis (allelic frequencies), in the Playgreen Lake and Lake Winnipeg north basin area, is a secondary objective of this study. These results should be generally useful in the management of commercial fisheries in the area.

STUDY AREA AND THE LAKE WINNIPEG REGULATION PROJECT

Lake Winnipeg is a very large (surface area greater than 24,000 km²) lake in central Manitoba which collects water from the Saskatchewan, Red, and Winnipeg rivers and other local drainage basins. This catchment area totals more than one million square kilometers. Prior to LWR, all water exited Lake Winnipeg through Warren Landing, flowing into Playgreen Lake (Fig. 1). The majority of this flow went into the north basin of Playgreen Lake and eventually to the west channel of the Nelson River. A part of the flow entered Little Playgreen Lake and the complex of channels around the community of Norway House, becoming the East Channel downstream. The East and West channels of the Nelson joined at Cross Lake to form the Nelson River proper. The Nelson River has a mean annual flow of about 2000 m³ s⁻¹.

The Lake Winnipeg Regulation Project was constructed by Manitoba Hydro over the period 1971-1976. The primary aim of the project was to regulate the level of Lake Winnipeg to provide larger assured winter flows into the Nelson River for hydroelectric power production on the lower Nelson. The Jenpeg control dam regulates flows through the west channel of the Nelson River, and indirectly, the level of Lake Winnipeg. The secondary aim of the LWR project was to increase the outflow capacity from Lake Winnipeg. Warren Landing is shallow and hydraulically inefficient during periods of low water levels. A variety of artificial channels and dike configurations serve to facilitate flows out of Lake Winnipeg. In the area of southern Playgreen Lake, the only change in the physical configuration of the system was the construction of Two Mile Channel (2MC) to carry a portion of the outflow of Lake Winnipeg into Playgreen Lake. The south basin of Playgreen Lake has a surface area of 275 km², a mean depth of about 3 m and a

maximum depth of about 13 m (MacLaren Plansearch 1985). The north basin of Lake Winnipeg has a surface area of 17,520 km², a mean depth of 13.3 m and a maximum depth of 19 m (Brunskill et al. 1980).

Since the completion of 2MC in 1976 and the operation of LWR, the geographic and seasonal distribution of flows through the Playgreen Lake area have been altered. A higher proportion of flow now proceeds by the west channel of the Nelson River than was the case prior to LWR. Also, a significant proportion of the flow utilizing Warren Landing now leaves Lake Winnipeg via 2MC. During periods of low water levels on Lake Winnipeg, 2MC carries the majority of the flow into Playgreen Lake (e.g., almost 70% in 1977; MacLaren Plansearch 1985). During normal and high water levels in Lake Winnipeg, Warren Landing carries most of the outflow of Lake Winnipeg (about 60-70%; MacLaren Plansearch 1985). Water levels in Playgreen Lake are now lower at the end of winter (March) and higher at the commencement of winter (October) than they were before LWR, although the lake is regulated within its historic range of water levels.

METHODS

Post-LWR sampling of lake whitefish was conducted in late October 1989 at a time close to that when lake whitefish spawning was expected to occur. Samples of approximately fifty mature lake whitefish were collected by gill netting from six different sites in northern Lake Winnipeg and Playgreen Lake (Fig. 1). Thus these samples likely are representative of fish segregated into stocks. Fish were frozen and shipped to the Freshwater Institute in Winnipeg, where they were processed. For genetic (biochemical) analyses, a sample of red and white muscle and a sample of liver was removed from each fish. These tissue samples were then frozen (-30°C) for later use in electrophoretic analyses.

Frozen samples (-30°C) from four identical locations (except Gunisao R. and Two Mile Channel (Fig. 1)) sampled by Kristofferson in 1975 were obtained to determine pre-development stock structure. Biochemical analyses for these four samples were conducted using white muscle tissue extracts.

Horizontal starch gel electrophoresis was performed following the methodology outlined by Vuorinen (1984) and under the electrophoretic conditions described in Bodaly et al. (1991). Thirty-five loci were screened for samples collected in 1989 whereas 10 loci were screened for samples collected in 1975. Table 1 gives the genetic loci examined, their abbreviations, and the tissue of primary expression. Table 2 lists alleles observed at polymorphic loci with their relative mobilities on electrophoretic gels. Allele frequencies of polymorphic loci were calculated for samples from each location. Phenotypic proportions were tested for Castle-Hardy-Weinberg (C-H-W) equilibrium by Chi-square contingency tables of observed and expected phenotype frequencies. Inter-stock homogeneity was evaluated by using the G-statistic test based on maximum likelihood ratios

(Sokal and Rohlf 1981). Previous studies have shown that the following enzyme loci are likely to be polymorphic in these populations of lake whitefish and therefore useful in comparing the various spawning aggregations genetically: MDH-B1,2 (treated as two loci with equal allelic frequencies), IDDH-1,2 (SDH), G3PDH-1, G3PDH-3, LDH-B2 (liver), IDHP-3, IDHP-4, and MEP-3,4 (Bodaly et al. 1991; Casselman et al. 1981; and Kristofferson and Clayton 1990). In performing statistical analyses, BIOSYS-1 was used (Swofford and Selander 1981).

RESULTS

Genetic variation was found at sixteen of the loci screened. Eleven of these were not used for subsequent analysis: six loci had unreliable models (G3PDH-3, IDHP-3, MEP-1,2, and MEP-3,4); and only rare alleles (average frequencies over all populations <0.01) were expressed at five loci (AAT, EST, GPI-1, PGM-2, SOD). Variation found at the remaining five loci fit existing genetic models (Vuorinen 1984). Observed MDH-B1,2, G3PDH-1, IDHP-4 and LDH-B2 phenotypic frequency distributions were in compliance with expected Castle-Hardy-Weinberg distributions for both pre-LWR and post-LWR samples, with the exception that the pre-LWR Warren Landing sample G3PDH-1 allele frequencies violated equilibrium frequencies. MDH-B1,2 allele frequencies observed in 1989 differed significantly (G statistic, 3 d.f. = 9.48, $P < .05$) among the subpopulations being compared (Table 3,4). Similar samples were pooled a posteriori as suggested by Kristofferson and Clayton (1990) in order to determine which stocks differed.

The samples taken in 1989 indicated that lake whitefish from Warren Landing and Little Playgreen Lake had similar MDH-B1,2 allelic frequencies (G statistic, 1 d.f. = 0.32, $P > 0.05$) and comprised one grouping (Fig. 2). Two additional samples from 1989 (Gunisao R. & Two Mile Channel) were grouped with similar samples from Big Black R. and Grand Rapids. Inter-group differences were significant at $P < 0.05$ (G statistic, 1 d.f. = 4.61).

These "pooled" groupings were similar to results obtained for the pre-development (1975) samples. Among-group allele frequency differences at the MDH-B1,2 loci for the 1975 samples were significant at $P < 0.05$ (G statistic, 1 d.f. = 3.80). Little Playgreen L. and Warren Landing samples were grouped together as were the Grand Rapids and Big Black R. samples (intra-group

homogeneity nonsignificant at the 5% level) (Fig. 3). Allele frequencies at the MDH-B1,2 loci for the grouped samples were not significantly different between 1975 and 1989 (Warren Landing and Little Playgreen samples, 1975 vs. 1989: G statistic, 3 d.f. = 0.82, $P > 0.05$; Grand Rapids and Big Black River samples, 1975 vs. 1989: G statistic, 3 d.f. = 2.68, $P > 0.05$).

IDHP-4, G3PDH-1 and LDH-4 frequencies were not significantly different among samples at the 5% level for both pre-LWR and post-LWR sampling periods.

DISCUSSION

The analysis of MDH-B1,2 allelic frequency distributions indicated that, prior to LWR, at least two different stocks of lake whitefish existed in the area of northern Lake Winnipeg and Playgreen Lake. These two stocks were, first, fish utilizing sites in the north basin of Lake Winnipeg for spawning (Grand Rapids and Big Black River), and second, fish utilizing Little Playgreen Lake for spawning. Fish caught in the area of Warren Landing had allele frequencies similar to those caught in Little Playgreen Lake. These genetic similarities and differences were found to be present in 1975, before LWR, and in 1989, after LWR. Thus, the preliminary conclusion of this study is that there is insufficient evidence to reject the null hypothesis that there has been no significant change in the genetic stock structure of lake whitefish in the area affected by LWR. The preliminary results of this study, therefore, suggest that changes in the fishery claimed by commercial fishermen on Playgreen Lake following LWR construction and operation are not attributable to changes in stock structure of lake whitefish. This conclusion is tentative and preliminary, however, for a number of reasons. First, the analysis in this report is based on only two genetic loci. Others will probably be included in subsequent analyses as genetic models are refined and a more extensive data set is interpretable. Second, an analysis of morphological data for both 1975 and 1989 samples will eventually be included in this study. This further work may either confirm or contradict the conclusions based on the preliminary analysis presented here.

Because genetic differences indicate distinct stocks whereas similarities do not prove identity, more than two stocks may be present in the LWR area. It is in fact known already that there are more than two morphological stocks of lake

whitefish in the LWR area. Kristofferson and Clayton (1990) found that the Grand Rapids, Big Black River, and Little Playgreen spawning stocks were separable on the basis of morphological characteristics. Other stocks are also probably present. Although biochemical results presented here indicated that fish captured in Playgreen Lake proper (at 2MC) and in the Gunisao River have allele frequencies at the MDH-B1,2 genetic loci similar to Lake Winnipeg stocks, this finding may be the result of coincidence. It seems more likely, based on the geographic distribution of spawning areas, that these samples represent one or two additional genetic stocks. That is, it seems unlikely that fish spawning in the Gunisao River are in fact part of the same stock as fish spawning on the west or east side of the north basin of Lake Winnipeg. One other possibility is that the 2MC stock is passing through Playgreen Lake to spawn in the Gunisao River. Further examination of biochemical genetic and morphological data will shed light on these questions.

A basic limitation of this study is the relatively small number of samples taken before LWR in the north basin of Lake Winnipeg and especially in the area of Playgreen lake. It should be emphasized, however, that the pre-LWR study was not conducted with the objective of providing a baseline data set against which to measure changes caused by LWR.

Differences in allele frequencies detected between stocks in northern Lake Winnipeg and Playgreen Lake probably reflect some degree of reproductive isolation among stocks and confer biochemical individuality upon each stock (Kristofferson and Clayton 1990). This isolation could be induced by innate homing of lake whitefish to natal spawning grounds. This would restrict gene flow between stocks and tend to perpetuate the discrete characteristics of each group. The degree of geographic isolation of stocks by distance also affects

reproductive isolation by reducing effective rates of gene flow resulting from straying between adjacent spawning areas. Richardson et al. (1986) have contrasted two models which could underly the formation of genetic stocks of fishes. The isolation by distance model predicts that gradual geographically related changes occur in allele frequencies, that the effects at each locus are independent, and that there will be no geographical discontinuities in allele frequencies. The discrete stock model is supported when intra-stock allele frequencies are homogeneous and in equilibrium, when there are inter-stock differences in allelic frequencies, and when there are discontinuities in allele frequencies at the same geographical locations for several loci (Richardson et al. 1986). In the case of lake whitefish stocks, probably both distance and homing are acting to cause genetic differences between adjacent stocks.

FUTURE INVESTIGATIONS

Additional biochemical investigations combined with morphological methodologies should be carried out for the pre- and post-LWR samples. Additional stock subdivisions may be discerned. The morphological results may allow for the identification of additional morphological stocks not ascertained by genetic analysis. As mentioned above, Kristofferson and Clayton (1990) were able to discriminate three different stocks in northern Lake Winnipeg and Playgreen Lake and the possibility exists to discriminate between three more stocks by examination of post-LWR samples. Phenotypic characteristics will be of practical importance in delineating stocks and in confirming the results of biochemical analyses. In addition, Savvaitova et al. (1989) have hypothesized that environmental impacts (such as LWR) may affect the morphological heterogeneity of fish stocks. This will be tested by the application of morphological analyses in the pre- and post-LWR lake whitefish samples.

REFERENCES

- Abersold, P.B., G.A. Winans, D.J. Teel, G.B. Milner, and F.M. Utter. 1987. Manual for starch gel electrophoresis: a method for the detection of genetic variation. U.S. Department of Commerce, NOAA Tech. Rep. NMFS No. 61.
- Bodaly, R.A., T.W.D. Johnson, R.J.P. Fudge, and J.W. Clayton. 1984. Collapse of the lake whitefish (Coregonus clupeaformis) fishery in Southern Indian Lake, Manitoba, following lake impoundment and river diversion. Can. J. Fish. Aquat. Sci. 41: 692-700.
- Bodaly, R.A., J. Vuorinen, R.D. Ward, M. Luczynski, and J.D. Reist. 1991. Genetic comparisons of new and old world coregonid fishes. J. Fish. Biol. 38: 37-51.
- Brunskill, G.J., S.E.M. Elliott, and P. Campbell. 1980. Morphometry, hydrology, and watershed data pertinent to the limnology of Lake Winnipeg. Can. MS Rep. Fish. Aquat. Sci. 1556: v + 23 p.
- Casselman, J.M., J.J. Collins, E.J. Crossman, P.E. Ihssen, and G.R. Spangler. 1981. Lake whitefish (Coregonus clupeaformis) stocks of the Ontario waters of Lake Huron. Can. J. Fish. Aquat. Sci. 38: 1772-1789.
- Clayton, J.W., and W.G. Franzin. 1970. Genetics of multiple lactate dehydrogenase isozymes in muscle tissue of lake whitefish (Coregonus clupeaformis). J. Fish. Res. Bd. Can. 27: 1115-1121.
- Clayton, J.W., W.G. Franzin, and D.N. Tretiak. 1973. Genetics of glycerol-3-phosphate dehydrogenase isozymes in white muscle of lake whitefish (Coregonus clupeaformis). J. Fish. Res. Bd. Can. 30: 187-193.
- Franzin, W.G., and J.W. Clayton. 1977. A biochemical genetic study of zoogeography of lake whitefish (Coregonus clupeaformis) in western Canada. J. Fish. Res. Bd. Can. 34: 617-625.
- Horrall, R.M. 1981. Behavioral stock-isolating mechanisms in Great Lakes fishes with special reference to homing and site imprinting. Can. J. Fish. Aquat. Sci. 38: 1481-1496.
- Ihssen, P.E., H.E. Booke, J.M. Casselman, J.M. McGlade, N.R. Payne, and F.M. Utter. 1981. Stock identification: materials and methods. Can. J. Fish. Aquat. Sci. 38: 1838-1855.
- Imhof, M., R. Leary, and H.E. Booke. 1980. Population or stock structure of lake whitefish, Coregonus clupeaformis, in northern Lake Michigan as assessed by isozyme electrophoresis. Can. J. Fish. Aquat. Sci. 37: 783-793.
- Kristofferson, A.H. 1978. Evidence for the existence of subpopulations of lake whitefish, Coregonus clupeaformis (Mitchill) in Lake Winnipeg. M.Sc. thesis, Department of Zoology, University of Manitoba, 94 p.

Kristofferson, A.H., and J.W. Clayton. 1990. Subpopulation status of lake whitefish (Coregonus clupeaformis) in Lake Winnipeg. Can. J. Fish. Aquat. Sci. 47: 1484-1494.

Larkin, P.A. 1972. The stock concept and management of Pacific salmon. H.R. MacMillan Lectures in Fisheries, Univ. British Columbia, Vancouver, B.C. 231 p.

Lindsey, C.C. 1981. Stocks are chameleons: plasticity in gill rakers of coregonid fishes. Can. J. Fish. Aquatic. Sci. 38: 1497-1506.

MacLaren Plansearch Inc. 1985. Ecological study of Playgreen Lake, Manitoba. Report to Manitoba Hydro, March 1985.

Pollard, W.R. 1973. A preliminary evaluation of whitefish movements in the north basin of Lake Winnipeg and between Lake Winnipeg and Playgreen Lake. Manit. Dep. Mines Resour. Environ. Manage. 34 p.

Richardson, B.J. 1983. Distribution of protein variation in skipjack tuna (Katsuwonus pelamis) from the central and south-western Pacific. Aust. J. mar. freshw. Res. 34: 231-251.

Richardson, B.J., P.R. Baverstock, and M. Adams. 1986. Allozyme electrophoresis. A handbook for animal systematics and population studies. Academic Press, Sydney. 410 p.

Savvaitova, K.A., V.G. Markaryan, and E.A. Dorofeyeva. 1989. Changes in morphometric characters of races of the Sevan Trout, Salmo ischchan, under anthropogenic influence (monitoring data). Voprosy Ikhtiologii, No. 2. pp. 198-208.

Shaklee, J.B. 1984. The utilization of isozymes as gene markers in fisheries management and conservation. In Isozymes: Current topics in biological and medical research. Vol. 11: Medical and other applications. p. 213-247.

Sokal, R.R., and F.J. Rohlf. 1981. Biometry, 2nd edition. W.H. Freeman. San Francisco. 573 p.

Swofford, D.L. and R.B. Selander. 1981. BIOSYS-1: a FORTRAN program for the comprehensive analysis of electrophoretic data in population genetics and systematics. J. Hered. 72: 281-283.

Vuorinen, J. 1984. Electrophoretic expression of genetic variation and duplicate gene activity in vendace, Coregonus albula (Salmonidae). Hereditas 101: 85-96.

Winans, G.A. 1980. Geographic variation in the milkfish Chanos chanos. I. Biochemical evidence. Evolution 34: 558-574.

Table 1. Enzymes screened with number of loci and tissues (M=muscle; L=liver). All loci were examined for the 1989 samples; G3PDH-1, IDHP-3, IDHP-4, LDH-B2, MDH-B1,2, MEP-1,2, MEP-3, MEP-4 only were examined for the 1975 samples.

Enzyme name	Enzyme number	Abbreviation	No. loci screened	Tissue
Aspartate aminotransferase	2.6.1.1	mAAT	1	M
		sAAT	2	M
Alcohol dehydrogenase	1.1.1.1	ADH	1	L
Creatine kinase	2.7.3.2	CK-A	2	M
Esterase	3.1.1.1	EST	1	L
Glycerol-3-phosphate dehydrogenase	1.1.1.8	G3PDH	2	M
Glucose-6-phosphate isomerase	5.3.1.9	GPI-A	2	L
		GPI-B	2	M
L-Iditol dehydrogenase [Sorbitol dehydrogenase]	1.1.1.14	IDDH	2	L
Isocitrate dehydrogenase	1.1.1.42	mIDHP	2	M
		sIDHP	2	L
Lactate dehydrogenase	1.1.1.27	LDH-A	2	M
		LDH-B	2	L
Malate dehydrogenase	1.1.1.37	sMDH-A	2	L
		sMDH-B	2	M
NADP ⁺ -dependent malic enzyme	1.1.1.40	mMEP	2	M
		sMEP	2	L
Phosphogluconate dehydrogenase	1.1.1.44	PGDH	1	M, (L)
Phosphoglucomutase	5.4.2.2	PGM	2	M, (L)
Superoxide dismutase	1.15.1.1	sSOD	1	L

 Table 2. Relative mobilities of alleles at polymorphic loci under electrophoretic conditions explained in the text. Alleles and locus nomenclature follow Bodaly et al. (1991) which should be consulted for allele designations in previous studies.

Locus	Allele	Mobility
G3PDH-1	a	0
	b	-85
	d	210
IDHP-3	a	100
	d	190
IDHP-4	a	100
	c	118
LDH-B2	a	100
	b	90
MDH-B1,2	a	120
	b	100
MEP-1,2	a	100
	b	40
MEP-3,4	a	100
	b	135

Table 3. Sample sizes and allele frequencies at five polymorphic loci examined in four lake whitefish samples from 1975.

Locus and alleles	Population			
	Little Playgreen	Warren Landing	Grand Rapids	Big Black River
G3PDH-1				
N	50	51	53	55
a	.250	.157	.226	.309
b	.270	.216	.245	.245
d	.480	.627	.528	.445
IDHP-4				
N	24	24	36	22
a	.896	.854	.944	.864
c	.104	.146	.056	.136
LDH-B2				
N	51	51	54	55
a	.020	.020	.028	.036
b	.980	.980	.972	.964
MDH-B1,2				
N	50	51	53	54
a	.450	.461	.274	.380
b	.550	.539	.726	.620

Table 4. Sample sizes and allele frequencies at five polymorphic loci examined in six lake whitefish samples from 1989.

Locus and alleles	Population					
	Little Playgreen	Warren Landing	Grand Rapids	Big Black River	2MC	Gunisao River
G3PDH-1						
N	50	49	50	50	50	50
a	.220	.214	.200	.250	.210	.230
b	.320	.184	.220	.290	.300	.280
d	.460	.602	.580	.460	.490	.490
IDHP-4						
N	50	48	50	50	47	50
a	.850	.865	.790	.880	.851	.740
c	.150	.135	.210	.120	.149	.260
LDH-B2						
N	50	48	50	50	50	50
a	.040	.042	.090	.090	.100	.050
b	.960	.958	.910	.910	.900	.950
MDH-B1,2						
N	50	48	50	50	50	50
a	.470	.510	.290	.350	.350	.370
b	.530	.490	.710	.650	.650	.630

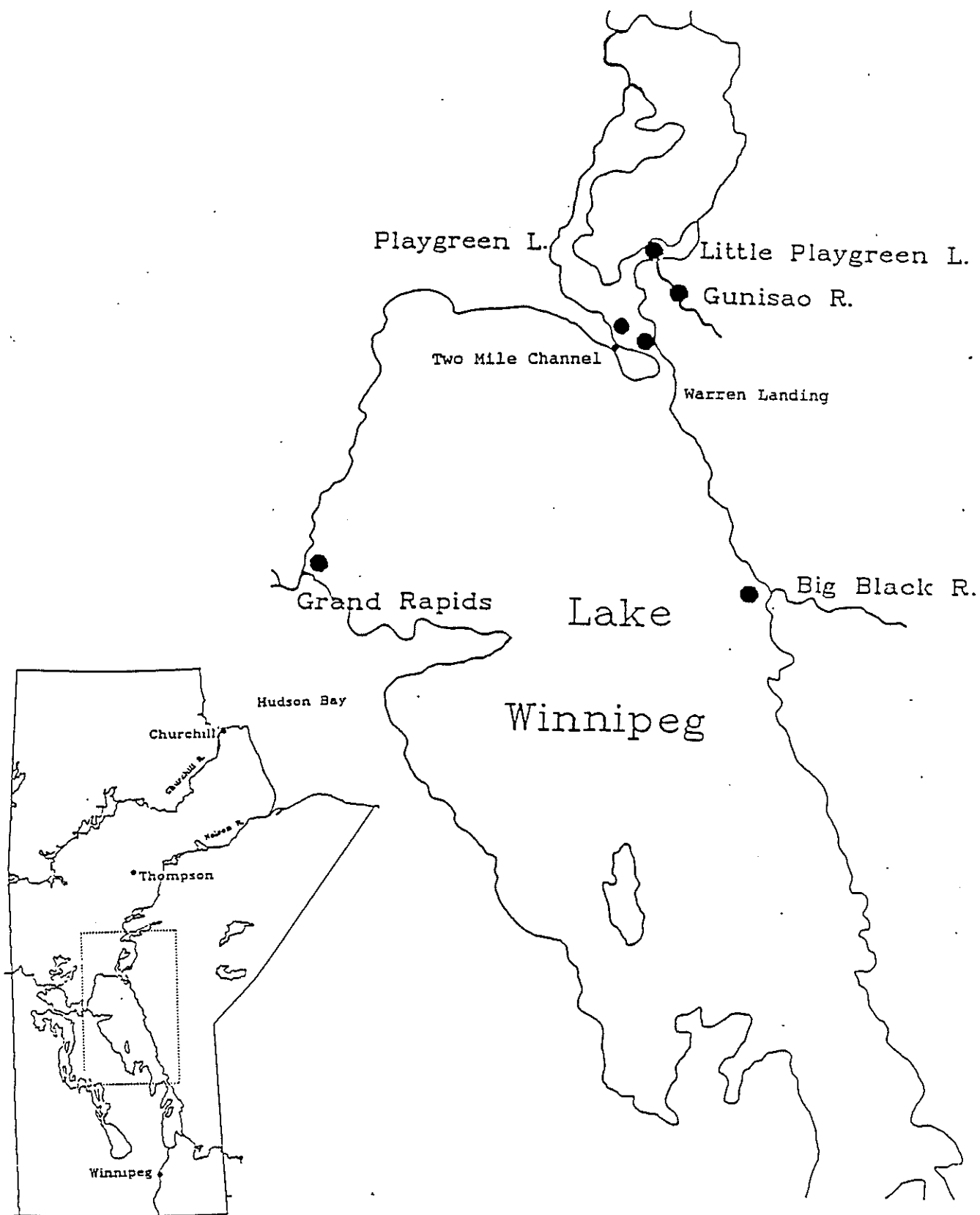


Fig. 1 Location of sampling sites in northern Lake Winnipeg and Playgreen Lake.

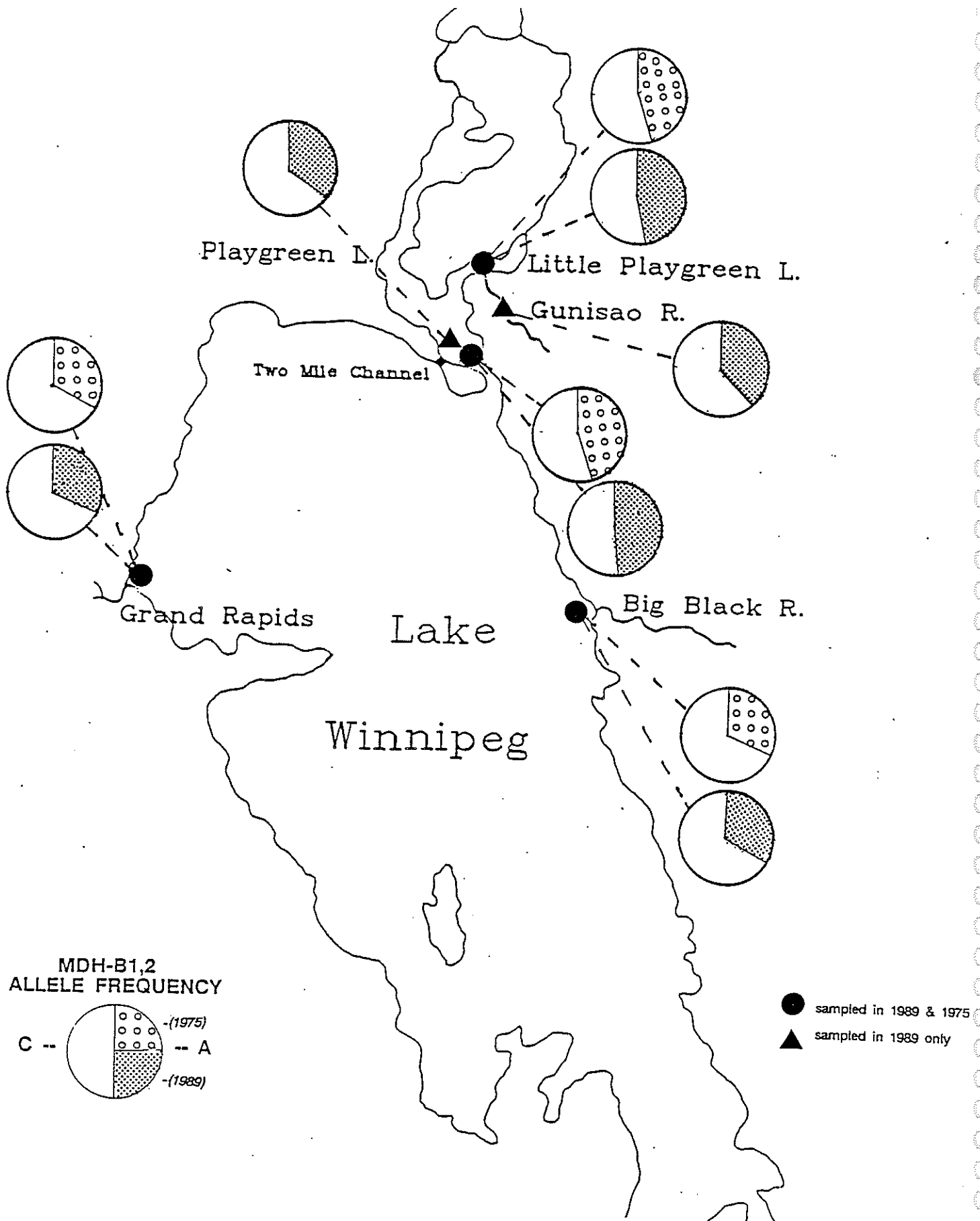


Fig. 2. Relative allele frequencies of MDH-B1,2 alleles for post-LWR (1989) and pre-LWR (1975) samples.

