Serpent River Watershed
State of the Environment Report

Prepared for:

Rio Algom Limited
And
Denison Mines Inc.
Elliot Lake, ON

Prepared by:

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July 2011

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EXECUTIVE SUMMARY

Uranium mining was undertaken in the Elliot Lake area of north eastern Ontario for approximately forty years. The mines generally operated from the late 1950's to the mid 1960's and again from the early 1970's until the early 1990's when most of the mines ceased operations. In total, there are eleven decommissioned mining operations and associated tailings management areas (TMAs) located in the Serpent River Watershed. The TMAs are in the long-term care and maintenance phase following closure that includes effluent treatment, source and watershed monitoring and TMA care and maintenance. All of the TMAs discharge to the Serpent River Watershed, except Pronto which discharges to the north shore of Lake Huron. The long-term care and maintenance of these sites is the responsibility of Rio Algom Limited and Denison Mines Inc.

As part of the closure and decommissioning process, Rio Algom and Denison developed a focused and integrated performance monitoring network. The comprehensive monitoring and management strategy clearly defined and delineated the purpose for all monitoring activities through three integrated program; the TMA operational monitoring program (TOMP), the source area monitoring program (SAMP) and the Serpent River watershed monitoring program (SRWMP).

The objective of this Serpent River Watershed State of the Environment Report was to integrate recent (2005 to 2009) monitoring data from the TOMP, SAMP, and SRWMP to provide an assessment of current TMA performance and the conditions in the downstream Serpent River Watershed relative to TMA sources.

In-Basin Quality

Since decommissioning, conditions in the TMA basins have improved and basin water quality is generally at or near Environmental Impact Statement (EIS)-predicted levels. Water quality has continued to improve in recent years (2003 to 2007) based on decreasing concentrations of radium-226, sulphate, and uranium, as well as increasing pH levels, at most TMAs. Exceptions were observed at Denison TMA-1 and Stanleigh TMA where radium-226 has been increasing in surface water at both TMAs, and pH has been decreasing at Denison TMA-1. While radium-226 concentrations were found to be decreasing over the past five years at most TMAs and remain within the range specified in the EIS sensitivity analysis, sulphate concentrations have also been decreasing and studies on radium release mechanisms suggest that decreases in sulphate over time may result radium release from the tailing to the overlying water column of the basins. In order to develop an understanding of the mechanisms controlling radium-226 releases to basin surface water, EcoMetrix was

retained to investigate radium-226 activities in solids (submerged tailings and treatment solids), porewater, and basin water at both the Quirke and Panel TMAs. These studies concluded that as aqueous sulphate concentrations decline, there is an increased dissolution of barium sulphate to which radium is associated, whereby radium is released from the tailings. Based on this assessment, the concentration of radium-226 in the porewater of flooded basins is not expected to exceed 5.5 Bq/L and the overlying water column is expected to remain below 1.8 Bq/L. It is expected that radium concentrations in porewater will stabilize over time once the dissolution of barium sulphate re-equilibrates with aqueous sulphate concentrations. Assuming there are no new sources of radium to the TMAs, radium concentrations in porewater should decline as the amount of soluble material in the tailings diffusion zone decreases. It is likely that the increases in radium-226 observed at Denison and Stanleigh TMA are associated with declining sulphate concentrations.

TMA Discharges

Primary mine discharges, which contribute the majority of chemical loadings to the receiving environment, have also been improving over time. Where trends were detected, radium-226, sulphate, and uranium concentrations decreased in TMA effluents. The only exception to this was at Stanleigh, where radium-226 concentrations have been increasing slightly in response to decreasing sulphate concentrations in the basin.

At some TMAs (Denison, Stanrock and Pronto), effluent pH showed a decreasing trend but this appeared to be associated with either changes in treatment or possibly the effect of higher flows in 2008 and 2009. In all cases, effluent pH remains circum neutral.

Trend analysis for 2003-2009 data indicated barium concentrations have been increasing at the primary discharge locations (CL-06, D2, D-3, P-14 and Q-28) of the flooded basins, but this was largely due to greater barium chloride use in 2008 and/or 2009 in response to increased flows. In all cases barium concentrations in discharges were well below toxicity thresholds.

Over the past five years, effluent quality has consistently achieved discharge criteria at all TMAs. With few exceptions, effluent has also been consistently non-lethal to *Daphnia magna* and rainbow trout with no mortality reported in semi-annual acute toxicity tests. Similarly, survival and reproduction of *Ceriodaphnia dubia* were not affected by exposure to 100% effluent in most tests conducted over the past five years at all TMAs.

Direct seepage releases from the TMAs to the receiving environment only occur in the Quirke Lake sub-watershed. While metal concentrations tend to be highest and pH lowest in these sources, their loads to the receiving environment are low compared to primary discharges and background (upstream) loads. As noted in the previous SOE report (Minnow 2009a), the radium load within the Serpent River downstream of the Denison TMA discharge (D-5) was substantially greater than the loading from the Denison TMA or the upstream watershed (D-4) suggesting a radium source within the river. In 2009, EcoMetrix conducted a study to investigate the difference in loadings within the River and found elevated radium-226 sediment concentrations (14 Bq/g) between stations D4 and D5. The barium and sulphate depth profiles in sediment and water (porewater and overlying water) mirrored the radium profiles, indicating that these profiles are likely caused by the settling/accumulation of historical treatment solids. The loadings from this area are consistent with the recovery of historically accumulated sediments releasing radium to the water column. Diffusion modelling indicated that radium-226 release from the sediment should decrease with time.

Watershed Conditions

The improvements within the TMAs were reflected in the downstream watershed. With few exceptions, mean surface water concentrations of mine related substances were less than the SRWMP benchmarks and, where concentrations exceeded the benchmark, they did not exceed toxicological thresholds. Furthermore, metal concentrations (cobalt, manganese, radium-226, sulphate and uranium) in surface water have been decreasing over time, and pH has been increasing.

In locations where sediment concentrations were above benchmarks, concentrations of barium, cobalt, iron, manganese and nickel appeared to decrease or remain stable over the past ten years (1999 to 2009). Statistical comparisons of 1999 versus 2009 sediment concentrations indicated few statistically significant differences (1999 vs. 2009), except: a) a significant increases in sediment iron and manganese concentrations in Quirke Lake; b) an increase in sediment radium-226 in McCabe Lake, and c) decreases in sediment cobalt, manganese, nickel and radium-226 concentrations in Hough Lake. Overall, the data indicate a very slow rate of change in sediment quality.

Sediment toxicity tests using *Hyalella azetca* showed reduced survival and growth in samples from Pecors, McCarthy and Nordic compared to reference lakes and laboratory control samples. These results did not correspond with sediment chemistry since McCarthy and Pecors lakes had some of the lowest sediment concentrations of mine-related substances. The observed response may be related to total organic carbon (TOC) which was much lower in McCarthy and Pecors lakes than in the lab control or the reference lake. Growth and survival of *Chironomus dilutus* did not differ between exposure and reference lakes.

The benthic invertebrate communities of all mine-exposed lakes were statistically different from reference lakes with respect to at least one of the benthic community metrics. The exposure areas showed a pattern of lower benthic invertebrate density and CA1 scores, along with higher CA2 and CA3 scores than the pooled reference areas, indicative of a mine-related signature. The communities in Quirke, McCabe, and May lakes showed more significant differences from the mean reference community than the other lakes (i.e., more metrics differed), but the magnitudes of difference were larger at Quirke and McCabe than May when differences were expressed as a percentage of the reference mean or the number of reference area standard deviations. The benthic communities in Elliot and McCarthy Lakes were most similar to the mean reference community, differing only with respect to CA-3 score.

It is clear that year-to-year variation is a significant component of community change in lake benthic communities, against which reference-exposure differences must be assessed in future years. Despite the variability among years, it appears that the significant pattern of deviations from reference mean values for the exposure lakes generally decreased through the three cycles of study, from 4 out of 5 metrics in 1999, to 3 out of 5 in 2004, and only 2 out of 5 metrics in 2009. These changing patterns of deviation are evidence in support of a hypothesis of gradual recovery from initial (1999) impact evaluation in exposure lakes. In most cases, the metrics for mine-exposed lakes fell within the reference lake range, especially when Rochester Lake was considered. Therefore, the patterns of effect suggested by the data in 2009 are based on relative small shifts away from the mean reference condition and may have little or no ecological consequence when considered in terms of the range of values exhibited by reference lakes in the area.

Risks to Wildlife and Humans

A special investigation was undertaken to better estimate dose and risk by making measurements to confirm or adjust assumptions used in previous dose and risk estimates. The data collected as part of the special investigation proved adequate to resolve the outstanding questions with respect to dose and risk estimates within the Serpent River Watershed. Dose estimates received by aquatic biota and riparian wildlife in the six watershed lakes were less than the respective UNSCEAR (1996) benchmarks of 10 mGy/d and 1 mGy/d. The incremental radiation doses received by generic human receptors (residing at the lake and consuming local fish and game) at the six watershed lakes, ranged from 0.023 to 0.288 mSv/a, all less than the public dose limit of 1 mSv/a. The calculated dose to a Serpent River First Nation harvester was 0.062 mSv/a (total) or 0.049 mSv/a (incremental) based on realistic use of the six watershed lakes, and 0.060 mSv/a (total) or

0.047 mSv/a (incremental) based on a projected future use scenario. All these doses are less than the public dose limit of 1 mSv/a (incremental).

Summary

In Summary, the TMAs are performing well in terms of meeting EIS predictions and reflecting improving conditions. The Serpent River Watershed is responding to these improvements, with water quality responding (improving) more rapidly than sediment and benthic invertebrates. Nevertheless, the benthic community has shown a pattern of improvement over the past ten years. Updated dose and risk estimates based on measured values indicate that dose is below established benchmarks for aquatic and riparian biota and humans.

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1.0 INTRODUCTION

1.1 Site and Program History

Uranium mining was undertaken in the Elliot Lake area of northeastern Ontario for approximately forty years. The mines generally operated from the late 1950's to the mid 1960's and again from the early 1970's until the early 1990's when most of the mines ceased operations (Table 1.1). In total, there are eleven decommissioned mining operations located in the Serpent River Watershed (Quirke I and Quirke II, Panel, Denison, Spanish-American, Can-met, Stanrock, Stanleigh, Milliken, Lacnor, Nordic, Buckles), and one other (Pronto) is located near the north shore of Lake Huron (Figure 1.1). Associated with the mine sites are eleven decommissioned tailings management areas (TMAs) of which seven are flooded (Denison TMA-1, Denison TMA-2, Panel, Quirke, Spanish-American, Milliken and Stanleigh) and four are vegetated (Lacnor, Nordic, Pronto and Stanrock). Tailings were also historically deposited in Buckles Creek adjacent to the Nordic TMA and Sheriff Creek adjacent to the Milliken mine. These areas are included within the licensed areas.

Final decommissioning and closure of the Quirke, Panel, Denison, Stanrock and Spanish-American properties was undertaken between 1992 and 1996. The Stanleigh Mine and the historic properties (*i.e.,* mine sites that operated in the 1950's and 1960's only; Table 1.1) were decommissioned from 1997 to 2000 and, in the case of Stanleigh, was not complete until 2002 (*i.e.,* when flooding was completed). The TMAs are currently in long-term care and maintenance following closure that includes effluent treatment, source and watershed monitoring and TMA care and maintenance. All of the TMAs discharge to the Serpent River Watershed, except Pronto which discharges to the north shore of Lake Huron. The long-term care and maintenance of these sites is the responsibility of Rio Algom Limited and Denison Mines Inc.

At the time of closure, each mine had its own environmental monitoring program conducted under an operating license from the Atomic Energy Control Board (AECB), the predecessor of the Canadian Nuclear Safety Commission (CNSC), and/or a Certificate of Approval (CofA) from the Ontario Ministry of the Environment (MOE). As part of the environmental approvals for the closure and decommissioning plans, Rio Algom and Denison evaluated their existing monitoring requirements in terms of their relevance to current and closure conditions. In 1997, the two companies began reviewing the existing environmental data, together with predicted changes associated with decommissioning, the latter of which was outlined in Environmental Impact Statements (EIS). The first outcome was the development of the Serpent River Watershed Monitoring Program (SRWMP) to replace the various mine-specific

Table 1.1: Elliot Lake mines - operating history, size and cover type.

Site [d]	Operating Period	TMA Tailings (million tonnes)	Area (ha)	Cover Type
Panel	Feb 1958 - June 1961; 1979 - Aug 1990	16.0	123	flooded
Denison (depostited in TMA-1 and TMA-2)	May 1957 - Apr 1992	59.7; 3	240	flooded
Lacnor	Sep 1957 - Jul 1960	2.7	27	vegetated
Milliken	Apr 1958 - June 1964	0.08 [a]	23.1	flooded
Nordic/Buckles [b]	Jan 1957 - Jul 1968	12.0	117.3	vegetated
Pronto	Aug 1958 - 1970	4.4 [c]	47	vegetated
Quirke	Sep 1956 - Feb 1961; Aug 1968 - 1992	46.0	192	flooded
Spanish-American	May 1958 - Feb. 1959	0.45	12	flooded
Stanleigh	Mar 1958 - June 1960; 1983 - June 1996	20.5	411	flooded
Stanrock and Canmet	1958 - late 1964 and Oct 1957 - Mar 1960	5.7	52	vegetated

Notes

[a] Majority of Milliken tailings (5.7 Mt) deposited at Stanleigh TMA, volume given for tailings deposited in Milliken TMA.

[b] Includes 0.04 Mt of contaminated sediment consisting of fine tailings and $Ba(Ra)SO_4$ in 10.3 ha Buckles Creek

[c] Includes 2.1Mt of uranium tailings and 2.3Mt of copper tailings

[d] Denison Mines Inc. owns the Denison and Stanrock properties and Rio Algom Limited owns the Quirke, Panel, Spanish-American, Lacnor, Nordic, Milliken, Stanleigh and Pronto properties.

Adopted from Table 5.2.2 CNSC, 2002.

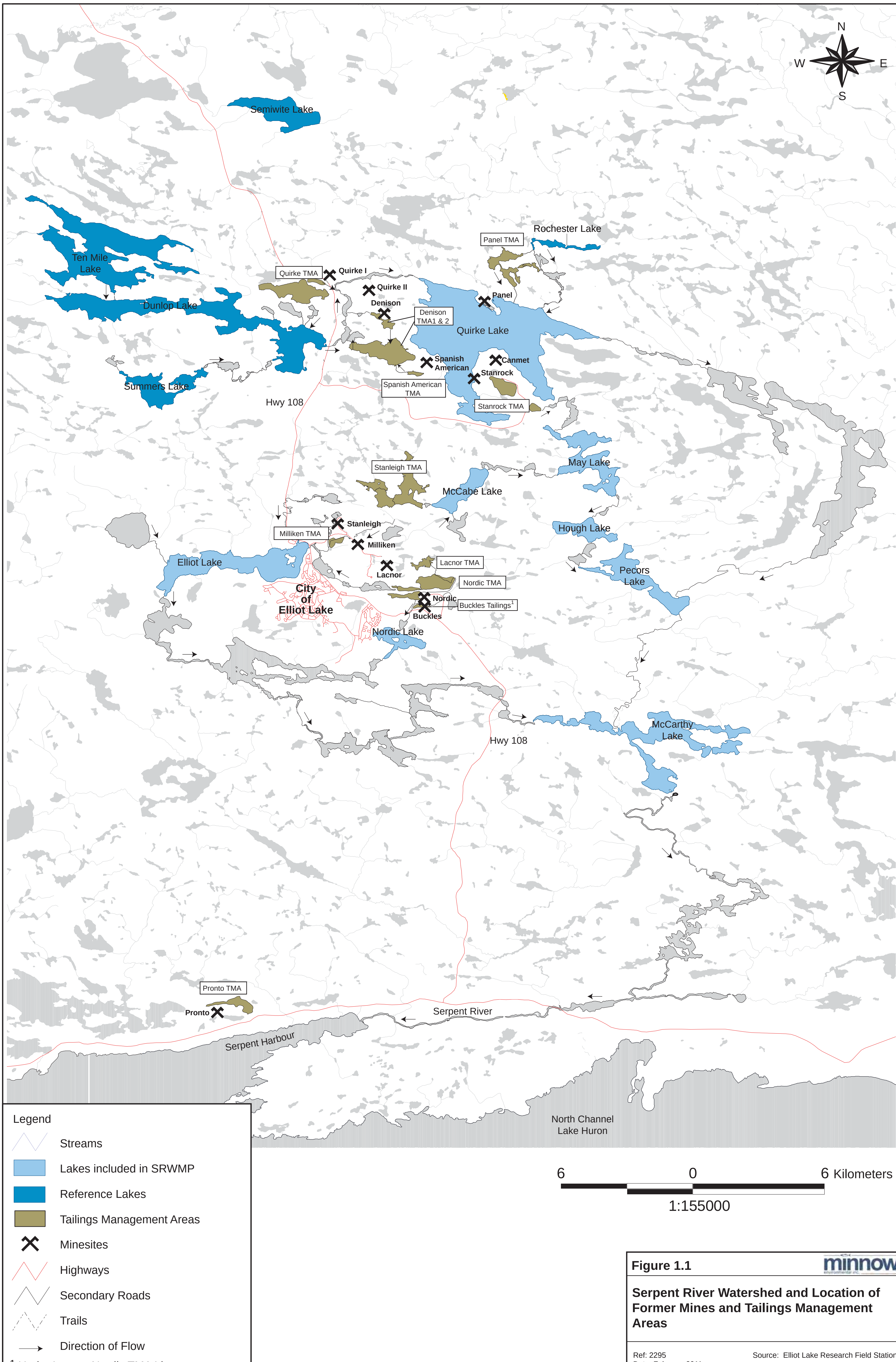

Figure 1.1

Serpent River Watershed and Location of Former Mines and Tailings Management Areas

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Date: February 2011

Source: Elliot Lake Research Field Station receiving environment monitoring programs with one comprehensive, harmonized watershed monitoring program. A companion program, the In-Basin Monitoring Program (IBMP), was also developed to assess the health risks to biota potentially feeding at each of the aquatic and vegetated TMAs. These programs were approved and implemented in 1999 (Beak, 1999a,b).

The Source Area Monitoring Program (SAMP) was the third program to evolve from the rationalization of the monitoring requirements associated with the licenses and certificates of approvals for the closed mines near Elliot Lake (Minnow 2002a). The purpose of the SAMP is to monitor the nature and quantity of constituents being discharged from the TMAs to the Serpent River Watershed (SRW). Therefore, the program focuses on monitoring stations that represent the final points of release from each TMA to the watershed. The SAMP was designed to complement the SRWMP and IBMP in terms of monitoring locations, variables and sampling frequency, and thus ensure that the overall monitoring framework is comprehensive and interpretable. The SAMP was approved in 2002 and implemented January 1, 2003.

The fourth and final program involved updating the monitoring requirements associated with internal TMA management, referred to as the TMA Operational Monitoring Program (TOMP; Minnow 2002b). The TOMP was designed to track TMA performance and support decisions regarding the management of the TMAs. The TOMP program was implemented concurrently with the SAMP in January 2003.

The end result of the rationalized monitoring programs for the Elliot Lake mine sites was the development of a comprehensive monitoring and management strategy that clearly defined and delineated the purpose for all monitoring activities. This ensured that all monitoring was objective-driven and would allow for modifications to be made over time in response to demonstrated conditions.

Each of the monitoring programs has been developed in consultation with and approved by the Elliot Lake Joint Review Group (JRG). The JRG is a multi-stakeholder committee comprised of representatives from the Canadian Nuclear Safety Commission (CNSC), Department of Fisheries and Oceans (DFO), Environment Canada (EC), Ontario Ministry of Environment (MOE), Ontario Ministry of Natural Resources (MNR), Ontario Ministry of Labour (MOL) and the Ontario Ministry of Northern Development, Mines and Forestry (MNDMF). The JRG continues to participate in the programs through the review of monitoring and design reports for the SAMP, the TOMP, and the SRWMP.

To date two SRWMP reports have been completed; the Cycle 1 report which captured the first year of water quality monitoring (1999 to 2000) as well as the first sediment and biological monitoring study implemented in 1999 (Minnow and Beak 2001) and the Cycle 2 report which presented the 2005 sediment and biological monitoring results as well as water quality data collected throughout the watershed during the first five years of the program (Minnow 2005). In 2008, Rio Algom and Denison mines prepared a "State of the Environment" (SOE) report (Minnow 2009a) which assessed conditions at each of the TMAs based on the SAMP, TOMP and IBMP and integrated the findings for the various TMAs with conditions observed in the watershed (SRWMP). This report captured data collected from the inception of these programs to the end of 2006. Based on the findings of the SOE report and previous SRWMP reports (Minnow 2005, Minnow and Beak 2001), the Cycle 3 SRWMP design was prepared along with revised SAMP and TOMP study designs (Minnow 2009b,c,d). The revised study designs were reviewed by the CNSC and JRG and approved in July 2009. Concurrent with the revised designs, the In-Basin Monitoring Program was discontinued as it had provided sufficient information to achieve its original objective. Therefore, the SRWMP, SAMP and TOMP are the monitoring programs that are currently in place at the closed Denison Mines Inc (DMI) and Rio Algom Limited (RAL) mines in Elliot Lake.

1.2 Project Background

To date the findings of the SRWMP have been reported separately and then summarized and referenced in a State of the Environment Report which provided details on the TMA performance and discharges. As the scope of the SRWMP retracts in response to improved conditions within the watershed, and the focus of the program shifts towards the source areas, the integration between the SRWMP, SAMP, and TOMP becomes more important. To better address the relationships between TMA performance, source area releases and watershed conditions it was agreed that one interpretive report be prepared which integrates the findings from all the three programs (SRWMP, SAMP and TOMP). This document; called the Serpent River Watershed State of the Environment Report, has been prepared to present and integrate the results of the three monitoring programs. The scope of the document includes:

- TMA performance (TOMP) for each TMA with a description of water management, water quality (surface, porewater and groundwater), reagent consumption, effluent compliance, and effluent toxicity;

- Source discharge concentrations and loads to the watershed from TMA effluent and seepage locations (SAMP) in terms of both spatial and temporal patterns;
- Conditions within the Serpent River Watershed based on water (2005-2009), sediment (2009), and benthic invertebrate (2009) monitoring results, including comparisons to previous study results and predictions, as well as recommendations for monitoring in subsequent cycles; and
- The findings of a special investigation conducted to better define dose and risk to human receptors.

1.3 Project Objectives and Approach

The objective of this Serpent River Watershed State of the Environment Report is to integrate recent monitoring data from the TOMP, SAMP and SRWMP to provide an assessment of current TMA performance and the conditions in the downstream Serpent River Watershed relative to TMA sources. In order to achieve this objective a number of goals were identified:

- Assess TMA performance relative to discharge criteria as well as performance objectives and predictions made in the Environmental Impact Statements (EIS);
- Evaluate mine sources (TMA releases) in terms of concentrations and loads to the Serpent River Watershed (SRW) and utilize trend analysis to anticipate future conditions in source contributions to the watershed; and
- Assess watershed conditions relative to TMA sources through water and sediment quality and benthic invertebrate community composition.

To meet the project objective and goals a weight of evidence approach was used that incorporated existing performance, trend analysis, loadings assessment and downstream conditions relative to established criteria and expected conditions (EIS predictions).

1.4 Report Organization

This report is organized in the following fashion. Section 2.0 presents the methodology used in the collection of samples and assessment of data. Section 3.0 presents the TMA performance for each TMA (TOMP) and Section 4.0 provides an assessment of TMA sources (SAMP) within sub-watersheds of the Serpent River so that multiple TMA sources to the same receiver may be considered together. The findings of the SRWMP are presented in Section 5.0. The updated risk assessment (special investigation) is summarized Section

6.0. Conclusions and recommendations based on the report are presented in Section 7.0. References cited throughout the report are provided in Section 8.0. Supporting information for the methods is provided in Appendix A. A complete data quality assessment for the TOMP, SAMP and SRWMP (2005 to 2009) is presented in Appendix B. Raw data and supporting information for the TOMP, SAMP and SRWMP are presented in Appendices C to E respectively. The results of the special investigation are presented in Appendix F.

2.0 METHODS

This report is a compilation of data associated with three monitoring programs implemented at the Elliot Lake closed mine sites – the Serpent River Watershed Monitoring Program (SRWMP), Source Area Monitoring Program (SAMP) and Tailings Operational Monitoring Program (TOMP). The data collected through these programs over the past five years (2005 to 2009) are assessed in detail herein, as well as older data, as appropriate, for the purpose of assessing temporal trends.

Methods employed for sample/data collection and analyses for all components of these programs are described in the following sections.

2.1 Sample/Data Collection

Surface water samples are collected under all three program (SRWMP, SAMP and TOMP), while groundwater and porewater samples are collected through TOMP only (Table 2.1). In addition, effluent samples are collected for toxicity testing as part of the SAMP. Other samples, such as sediment and benthic invertebrates, are collected as part of the SRWMP. Sampling methods are described below.

2.1.1 Water Chemistry and Toxicity

Water samples are collected under the SRWMP, SAMP and TOMP, with 16, 22, and 121 stations monitored, respectively (Table 2.2). Under these programs four types of water samples are collected:

- Influent and effluent samples at TMA treatment plants;
- Surface water samples within basins, at discharge points including seepages, and in the Serpent River watershed (Figure 2.1);
- Porewater within TMA basins ; and
- Groundwater outside of TMAs.

Specific monitoring variables for each station depend on the program objectives and station type. Station locations, monitoring frequency and variables for each program are listed in Tables 2.3, 2.4 and 2.5.

Collection of water samples is the responsibility of Denison Environmental Services (DES), which administers the operation and monitoring of the closed mines under contract to Rio Algom Limited and Denison Mines Inc. DES follows standard operating procedures (SOPs)

Table 2.1: Types of data collected through each sampling program.

Data Collected	Sampling Program		
	TOMP	SAMP	SRWMP
Water Quality			
Surface Water	x	x	x
Groundwater	x		
Porewater	x		
Water Flow	x	x	
Water Elevation	x		
Water Toxicity			
Acute Toxicity		x	
Sublethal Toxicity		x	
Sediment Characteristics			x
Sediment Chemistry			x
Benthic Invertebrates			x

Table 2.2: Current monitoring stations included in the SRWMP, SAMP and TOMP.[c]

	Serpent River Watershed Monitoring Program (SRWMP)[a]	Source Area Monitoring Program (SAMP)	TMA Operational Monitoring Program (TOMP)[b]			
			Effluent Control Point	Operational Data		
				Surface	Groundwater	Porewater
Panel	SR-01	P-02, P-03, P-05, P-11, P-14, and P-36	P-14	ECA-349, P-13, P-15, P-21, P-36	P-31, P-16 A, P-20,	
Quirke	Q-09, Q-20	ECA-398, Q-22, Q-23, Q-27, Q-28	Q-28	Q-03, Q-04P, Q-05, Q-24 (renamed Cell 16S), Q-29, Q-30 (renamed Cell 14), Q-47 (renamed Cell 15), and Q-48 (renamed Cell 17)	QPW1-1,4,8, 95QW-3A,C,D, 95QW-4, 95QW-5A,D	90DK-14-5 C; DK15-2 (A-D); DK15-4 (A-D); DK16-2 (A-D); DK17-2 (A-D)
Lacnor/Nordic	SC-01, SR-08	N-12	N-19	L-03, ECA-131, ECA-132, N-17, N-18, N-20, N-22, NWPH	M-12-1,3,6,9; M-13-1,3,6,9; M-14-1,3,6,9; 95N-4A,B; 95N-7A,B; 95N-11; 95N-12A,B; 95N-13A,C,E; 95N-14A,B,C; 95N-16A,C,E; 95N-17A,B,C	UW7(2,4,6), UW9(1-3)
Milliken	M-01	MPE	N/A			
Stanleigh	SR-06	CL-06	CL-06	CL-04, CL-05	SGW-3, SGW-4	
Spanish-American	N/A	N/A	N/A	ECA-128		
Pronto	N/A	LL-01, PR-01	PR-04	PR-02, PR-03		
Denison	D-5, D-6	D-2, D-3, D-9, D-16	D-2, D-3	D-1, D-22, D-25	BH91-D9A; BH91-DG4B; BH91-D1A,B; BH91-D3A,B	
Stanrock	DS-18	DS-4, DS-16	DS-4	DS-1, DS-2, DS-3, DS-5, DS-6	BH91-SG1A;BH91-SG3A,B; BH98-16A; BH98-15A	BH91-SG2A,D; PN-ST3-P3,5,6,8;
Reference	D4, P-22, SR-05, SR-14, SR-18, SR-19	SR-16, SR-17				
TOTAL STATIONS [d]	**16[a]**	**24**	**8[b]**	**34**	**56**	**25**

[a] SRWMP stations are not intended to be associated with a single source (TMA). Many stations integrate conditions from several TMAs.
[b] Includes some stations identifed as SAMP stations (i.e. stations that serve multiple purposes).
[c] Number of groundwater and porewater stations represents the number of wells monitored (i.e. A-C)

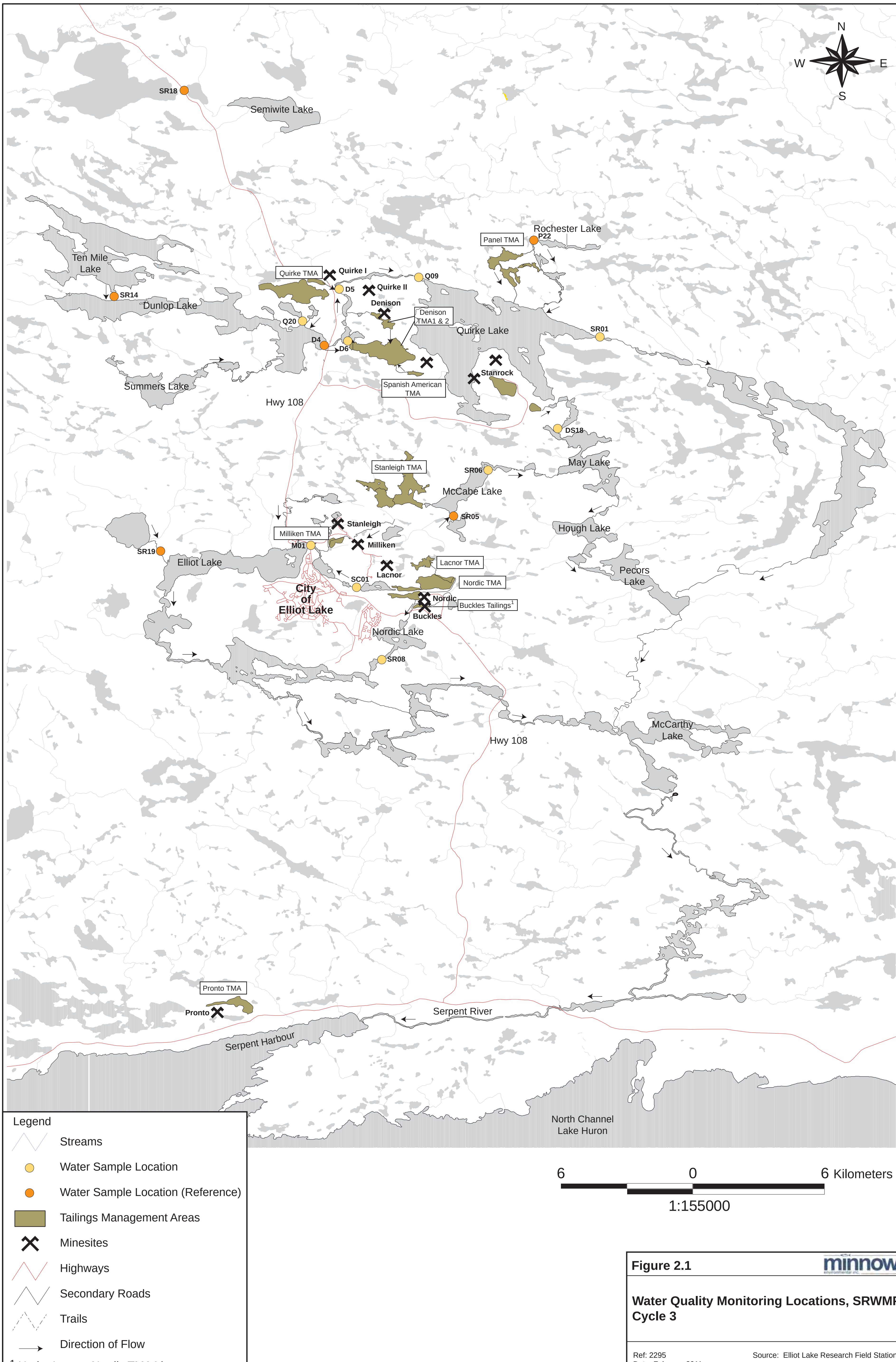

Figure 2.1

Water Quality Monitoring Locations, SRWMP Cycle 3

Table 2.3: Current Cycle 3 SRWMP water quality sample locations and frequencies.

Station	Location / Description	Type	UTM (North)	UTM (East)	Current Frequency
D4	Dunlop Lake Outlet (Q-14)	reference	5148783	373383	S
P-22	Rochester Creek @ Rochester Lake Outlet	reference	5153231	382747	S
SR-05	Canyon Lake Outlet	reference	5141190	379159	Q
SR-14	Ten Mile Creek at Inlet to Dunlop Lake	reference	5151063	363621	A
SR-18	Outlet of Jim Christ Lake	reference	5160540	366863	S
SR-19	Inlet to Elliot Lake	reference	5139744	365666	Q
D-5	Serpent River between Denison and Quirke TMAs	exposed	5151274	374006	Q
D-6	Cinder Lake Outlet	exposed	5148477	374404	Q
DS-18	Halfmoon Lake Outlet	exposed	5145050	383761	Q
M-01	Sherriff Creek @ Highway 108	exposed	5139798	372727	Q
Q-09	Serpent River Below Quirke TMA Effluent	exposed	5152097	377264	Q
Q-20	Evans Lake Outlet to Dunlop Lake	exposed	5150036	372333	A
SC-01	Westner Lake Outlet	exposed	5137964	374604	A
SR-01	Quirke Lake Outlet	exposed	5149300	385824	A
SR-06	McCabe Lake Outlet	exposed	5143518	380551	S
SR-08	Nordic Lake Outlet	exposed	5133920	375365	Q
Total Samples/Analytes		27			44

M= Monthly, S=Semi-Annual, A=Annual, 0 = no sampling

Change in frequency occurred as of January 1, 2010

Table 2.4: Current Cycle 3 SAMP stations, substances and frequencies.

TMA	Location	Type	Description	Parameter[f]					
				flow	pH	Sulphate	Radium-226	SAMP metals[a]	toxicity
Denison	D-2[c]	Primary	Stollery Lake Outlet	D	W	M	M	M	2
	D-3[c]	Primary	TMA-2 Effluent at Denison Mine access road	D	W	M	M	M	
	D-9	Seepage	Seepage at Dam 17	Q	Q	Q	Q	Q	
	D-16	Seepage	Seepage at Dam 9	Q	Q	Q	Q	Q	
Quirke	ECA-398	Seepage	Quirke II north of access road	Q	Q	Q	Q	Q	
	Q-22	Drainage	Quirke II Drainage south of access road	Q	Q	Q	Q	Q	
	Q-23	Drainage	Swamp Outlet west of Dam K1	Q	Q	Q	Q	Q	
	Q-27	Seepage	Dam J Toe Seepage		Q	Q	Q	Q	
	Q-28[c,d]	Primary	Final Treated Effluent	W	W	M	M	M	2
Panel	P-02	Seepage	Downstream of Dam B	Q	Q	Q	Q	Q	
	P-03	Drainage	Beaver Pond C Outlet	Q	Q	Q	Q	Q	
	P-05	Drainage	Swamp Outlet north of Dam E		Q	Q	Q	Q	
	P-11	Drainage	Panel Creek Outlet at Quirke Lake	Q	Q	Q	Q	Q	
	P-14[b,c,d,e]	Primary	Final Treated Effluent	W	W	M	M	M	2
Stanrock	DS-4	Primary	Orient Lake Outlet (Final Point of Control)	W	W	M	M	M	2
	DS-16	Drainage	Quirke Lake Delta	Q	Q	Q	Q	Q	
Stanleigh	CL-06[c,d]	Primary	Final Treated Effluent	W	W	M	M	M	2
Milliken	MPE	Primary	Milliken Park Effluent		M	M	M	M	2
Nordic	N-12	Primary	Buckles Creek at Hwy. 108	M	M	M	M	M	2
Pronto	LL-01	Drainage	Pronto Creek at Inlet to Lake Lauzon	Q	Q	Q	Q	Q	
	PR-01	Primary	Pronto Discharge Channel at Highway 17	M	M	M	M	M	2

[a] SAMP metals - barium, cobalt, iron, manganese, uranium
[b] P-14 will revert to P-36 upon ETP shut down.
[c] This station is also TOMP effluent station and requirements will be harmonized to serve both programs
[d] Sampled when treatment plant is operating
[e] Flow is based on influent flow to the ETP at P-13.
[f] DOC and hardness have been added effective January 1, 2010
D =daily, W = weekly, M = monthly, 2 = twice per year, Q = quarterly

Table 2.5: Substances and frequency of TOMP data collected.

			Parameters and Frequencies											
TMA	TOMP Stations	Station Type/Purpose	Elevation	Flow	pH	Conductivity	Sulphate	Total radium-226	Lime or NaOH Consumption	Barium Chloride Consumption	TSS	Acidity	Iron	SAMP Metals[c]
Denison	D-1	Basin performance (primary), ETP operations	W	D	D		Q	M	M	M		Q		Q
	D-22	ETP operations			W		Q	M		M		Q		Q
	D-3	Effluent		D[d]	W		M	W			W			M[d]
	D-2	Effluent		D[d]	W		M	W			W			M[d]
	D-25	Basin performance (secondary)			S		S	S				S	S	
	BH91-D1A,B, BH91-D3A,B, BH91-DG4B, BH91-D9A	Groundwater			A		A					A	A	
S.A.[h]	ECA-128	Basin performance (primary)	M[i]	Q	Q		Q	Q				Q		Q
Quirke	Q-05[j]	Basin performance (primary), ETP operations	W	D	M		Q	M	M	M		Q		Q
	Q-03[j]	ETP operations			W									
	Q-04P[j]	ETP operations			D									
	Q-28[j]	Effluent		W[d]	W		M	W			W			M[d]
	Q-29	Perimeter monitoring	W	W[i]										
	Cell 14, 15, 16S, 17	Basin performance (secondary)	M[i]		S		S	S				S	S	
	90DK-14-5C; DK15-2(A-D); DK15-4(A-D); DK16-2(A-D); DK17-2(A-D)	Porewater			A		A					A	A	
	QPW1-1,4,8; 95QW-3A,C,D; 95QW-4, 95QW-5A,D	Groundwater			A		A					A	A	
Panel	P-13[j]	Basin performance (primary), ETP operations	W	D	M		Q	M	M	M		Q		Q
	ECA-349[j]	ETP operations			D									
	P-14[j], P-36[j]	Effluent		-[f]	W		M	W			W			M[d]
	P-15	Perimeter				M								
	P-21	Basin performance (secondary)	M[j]		S		S	S				S	S	
	P-16A, P-20, P-31	Groundwater			A		A					A	A	
Stanrock	DS-2	Basin performance (primary), ETP operations		D	D		Q	M	M	M		Q		Q
	DS-3	ETP operations			D									
	DS-4	Effluent		W[d]	W		M	W			W			M[d]
	DS-1	Additional pH control, radium monitoring		W	W			Q						
	DS-6	Additional pH control		W	W									
	DS-5	Seepages and surface water internal to TMA		Q	Q	Q								
	PN-ST3-P3,5,6,8; BH91-SG2A,D	Porewater			A		A					A	A	
	BH91-SG1A, BH98-16A, BH98-15A, BH91-SG3A,B	Groundwater			A		A					A	A	
Stanleigh	CL-04[j]	Basin performance (primary), ETP operations	W	D	M		Q	M	M	M		Q		Q
	CL-05[j]	ETP Operations			D									
	CL-06[j]	Effluent		W[d]	W		M	W			W			M[d]
	SGW-3, SGW-4[e]	Groundwater			A		A					A	A	
Lacnor/Nordic	L-03	Basin performance (primary)	M[i]	Q	Q		Q	Q				Q		Q
	N-17	Basin performance (primary), ETP operations		D	M		Q	M	M			Q		Q
	N-18	ETP operations			D									
	N-19	Effluent		W	W		M	W			W			M
	N-22	Basin performance (secondary)		M[i]	S		S	S				S	S	S
	ECA-132	Basin performance (secondary)	M[i]	M[i]	M[i]		S	S				S	S	S
	NWPH	Basin performance (secondary)		M[i]	S		S	S				S	S	S
	ECA-131, N-20	Basin performance (secondary)			Q		Q	Q				Q		Q
	UW7-2,4,6; UW9-1,2,3	Porewater			A		A					A	A	
	M-12-1,3,6,9; M-13-1,3,6,9; M-14-1,3,6,9; 95N-4A,B; 95N-7A,B; 95N-11; 95N-12A,B; 95N-13A,C,E; 95N-14A,B,C; 95N-16A,C,E; 95N-17A,B,C	Groundwater			A[g]		A[g]					A[g]	A[g]	
Pronto	PR-02[j]	Basin performance (primary), ETP operations	W	D	M		Q	M	M	M		Q		Q
	PR-03[j]	ETP operations			D									
	PR-04[j]	Effluent		W	W		M	W			W			M

[a] D - Work days, W - Weekly, M - Monthly, S - Semi-annually, A - Annually, Q-Quarterly
[b] Also elevation
[c] SAMP metals are barium, cobalt, iron, manganese, and uranium
[d] Monitoring requirement of SAMP
[e] Relocated to Settling Pond Dam
[f] No flow monitoring at P-14 because <1% additional flow between P-13 and P-14
[g] A one-time modelling exercise was recommended by Ecometrix to confirm flow conditions and potentially modify future GW monitoring under TOMP. In the meantime, GW monitoring at Nordic will continue will cotinue at previously identified TOMP stations.
[h] Spanish-American
[i] During the snow-free period (April - November)
[j] Sampled when treatment plant is operating

that address all aspects of sample collection and management for the TOMP, SAMP and SRWMP from sample collection to laboratory submissions, data entry, validation and response. The SOPs ensure that that the data produced are consistent with the objectives of these programs, regulatory requirements, and industry standards (Table 2.6). The detailed SOPs are provided in their entirety in Appendix A. DES maintains contracts for various chemical analyses with SGS Laboratory, Becquerel and Aquatox Testing and Consulting Inc.

Water samples collected for chemical analyses were shipped to SGS Lakefield Research Limited in Lakefield, ON, for chemical analysis based on established methods. Water samples collected for toxicity testing were submitted to Aquatox Testing and Consulting Inc. (Aquatox) in Guelph, ON, for acute (*Daphnia magna* and rainbow trout) and sub lethal (*Ceriodaphnia dubia*) testing following Environment Canada (2000 a, b and 2007) methods.

2.1.2 Sediment Chemistry and Toxicity

Sediment samples were collected between September 14 and September 23, 2009 as part of the Cycle 3 SRWMP, consistent with the timing of previous field programs. The samples were collected from 13 lakes, five of which were reference (Figure 2.1). Five stations were sampled in each lake where benthic macroinvertebrate samples were also collected (Table 2.7 and Appendix Figures A.1 to A.13). This represents an increase from three stations per lake in past studies. Where possible, samples were collected from the same locations sampled in Cycles 1 and 2. The approximate location of each lake sample was identified on a bathymetric map prior to the field program. The station map, Cycle 1 and 2 station locations based on Global Positioning System (GPS) data, and a depth sounder were used to find the stations in the field. In order to achieve comparable substrate amongst sampling locations, two existing stations (DUL-09-01 and QL-09-5) were relocated as indicated on Appendix Figures A.1 and A.9 respectively. An average depth of 15 m was targeted for all lake sample locations, although some stations were positioned at depths slightly shallower or deeper to ensure that comparable substrates were sampled across lakes (Appendix Table A.1).

Two types of sediment samples were collected at each station: one for metal and radium-226 analysis and the other for analysis of total organic carbon (TOC) and particle size distribution. Sediment samples for analysis of metals and radium-226 were collected using a Tech-Op corer equipped with a 4-inch diameter lexan core tube. The use of the 4-inch corer necessitated taking a total of three to four cores (five cores were taken at one station in Pecors Lake) to meet minimum sample volume requirements for chemical analyses. The corer was deployed from a boat with care taken to control the rate of descent and to maintain

Table 2.6: List of Operating Procedures associated with the implementation of the SAMP and the TOMP.

Procedure Name	Operating Procedure Number
Control Limit Maintenance	PR8.7.2.02
Data Entry	PR8.7.3.01
Data Validation	PR8.7.3.02
Field Conductivity Determination	PR8.6.3.03
Field pH Determination	PR8.6.3.01
Field Sampling Quality Control	PR8.5.3.01
Flow Determination	PR8.6.4.02
Groundwater Sampling	PR8.6.2.01
Surface Water Grab Sampling	PR8.6.1.01
Toxicity Sampling	PR8.6.1.03
Water Quality Data Quality Assessment	PR8.5.4.01
Water Quality Assessment and Response Plan	PR8.0.0.01

Table 2.7: Cycle 3 sediment and benthic monitoring locations, number of stations and sediment parameters.

Station	Location/Description	Type	Number of Samples[1]	Parameters									Toxicity		Benthic Community
				Barium	Cobalt	iron	manganese	nickel	radium-226	uranium	TOC	Grain Size	*Hyallela azteca*	*Chironomus riparius*	
DUL	Dunlop Lake	Reference	5	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔
TML	Ten Mile Lake	Reference	5	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔		✔
RL	Rochester Lake	Reference	5	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔		✔
SL	Semiwite Lake	Reference	5	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔
SUL	Summers Lake	Reference	5	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔		✔
QL[2]	Quirke Lake	Exposure	5	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔		✔
ML	McCabe Lake	Exposure	5	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔
MAL	May Lake	Exposure	5	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔		✔
HOL	Hough Lake	Exposure	5	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔		✔
PL	Pecors Lake	Exposure	5	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔		✔
EL	Elliot Lake	Exposure	5	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔
NL	Nordic Lake	Exposure	5	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔		✔
MCL	McCarthy Lake	Exposure	5	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔		✔

[1] Three of the five stations were located at the same stations used in previous cycles with two additional stations placed at similar depth.

[2] No additional stations were added at Quirke Lake as it has had five stations in both previous cycles.

the corer in a vertical position during ascent. After it penetrated the sediment, the corer was carefully retrieved to the surface and an extruder was inserted into the bottom of the core tube to prevent any slippage. Core samples were rejected if there was any evidence of slippage, if there was any evidence that the core did not adequately penetrate the substrate, or if there was any evidence of disturbance of the sediment-water interface. The number of rejected cores, penetration depths and visible sediment characteristics (*i.e.*, the presence of epibenthic organisms or stratification) were recorded on field sheets.

Water in the core tube was decanted with a siphon hose prior to extruding sediments. Siphoning was stopped when there was approximately 2 to 3 cm of water remaining above the sediment surface. The core extruder was used to push sediments upwards towards the top of the core tube in a controlled fashion with care taken to minimize suspension of fines. In the event of suspension, momentum was stopped allowing the solids to re-settle. Once the sediment was near the top of the tube, an extrusion collar marked in 1-cm intervals was carefully aligned on the top of the tube and the sediment was extruded upwards to a depth of 1 cm. A core slicer (box design) was then carefully inserted between the tube and the collar, the collar removed and the sample transferred from the slicer to labelled Ziploc bags (double-bagged).

After sampling for metals and radium-226 was complete, additional sediment samples were collected for analysis of particle size and TOC using a petite ponar grab sampler. Surficial sediment (top 3 cm) was carefully removed from each of two intact grabs using a stainless steel spoon and composited into a Ziploc bag (double-bagged).

Sediment samples collected for the analysis of metals and radium-226 were submitted to Maxxam Analytics in Mississauga, ON, (Maxxam) where they were subsequently homogenized and dried, and a sub-sample (dry powder) was sent to Becquerel Laboratories, Mississauga, Ontario for radium-226 analysis. Sediments collected for metal content were digested using aqua regia (3:1 hydrochloric to nitric acid) and analyzed by inductively coupled plasma (ICP) (Table 2.8). Sediment samples for radium-226 analysis were digested using nitric, hydrochloric and hydrofluoric acids (which frees the radium-226 from the matrix for separation and analysis (MD-4871)) then analyzed for radium-226 activity using alpha spectroscopy (BQ-RAD-ALPHA).

Sediments collected for the analysis of particle size and TOC were also submitted to Maxxam. Particle size was analyzed using sieve and hydrometer methods while total organic carbon was analyzed using a Leco Carbon Analyzer (Table 2.8). Sediment was collected from one station at each of the 13 lakes for sediment toxicity testing using

Table 2.8: SRWMP sediment quality analytical methods.

Parameter (mg/kg)		Sediment Analytical Method	MDL
Particle Size	Particle Size (%)	Sieve and Hydrometer	0.1
TOC	Total Organic Carbon (%)	Leco Carbon Analyzer	0.1
Ba	Barium	ICP-AES	0.1
Co	Cobalt	ICP-AES	0.09
Fe	Iron	ICP-AES	0.2
Mn	Manganese	ICP-AES	0.03
Ni	Nickel	ICP-AES	0.1
Ur	Uranium	Flurometric AA	0.5
^{226}Ra	Radium-226 (Bq/kg)	Alpha Spectroscopy	5.0

AA - Atomic Absorption

ICP-AES - Inductively Coupled Plasma Atomic Emission Spectrometer

MDL - Method Detection Limit

Environment Canada (1997) methods for assessing 14-day survival and growth of *Hyalella azteca*. The selected station represented the location with the highest previously reported radium-226 concentration (Minnow 2005). Additional sediment samples collected from the same stations in McCabe, Elliot, Dunlop and Semiwite Lakes were tested for toxicity to *Chironomus dilutus* using a 10-day survival and growth test (Environment Canada 1997b). The chironomid tests were conducted to investigate observations in previous benthic surveys of fewer chironomid species in some lakes (McCabe and Elliot lakes), along with two reference lakes (Dunlop and Semiwite). Approximately 5-L of sediment was collected into a bucket by taking multiple grabs with a petite ponar. The samples were refrigerated at 4ºC and shipped to Aquatox in Guelph, Ontario for toxicity testing. Survival and growth were computed for samples from each lake and statistical comparisons were made among lakes and relative to a laboratory control (Appendix E).

2.1.3 Benthic Community Monitoring

Benthic macroinvertebrate samples were collected from 13 lakes (8 mine exposed and 5 reference) to assess potential impacts associated with the decommissioned mines (Figure 2.1; Appendix Figures A.1 to A.13). The samples were collected from the same locations as sediment samples (Section 2.1.2) so that the benthic communities could be considered relative to sediment composition and chemical quality. Each station was geographically referenced using a GPS (Appendix Table A.1).

To the extent possible, sampling methods employed in the 2009 Cycle 3 program were consistent with both the 2004 Cycle 2 and 1999 Cycle 1 program to allow for comparison of results between cycles. Five grab samples were composited at each station, as was done in Cycle 2, to provide a more representative sample (three grab samples were composited at each station in Cycle 1). Comparison of 2009 data to 2004 and 1999 was still possible as benthic invertebrate abundance data for all studies were expressed on a per m^2 basis.

The samples were collected using a petite ponar grab (0.023 m^2). Given the low productivity typical of profundal areas in lakes of the Canadian Shield, a small sieve size (250 μm) was used to optimize the number of individuals and taxa captured. The samples were transferred to a 250-micron sieve bag and rinsed with site water to remove sediment particles. Reduced samples were transferred to 1-L wide-mouth plastic jars and preserved with 100% buffered formaldehyde to a minimum level of 10% formalin within 8 hours of collection. An internal label was placed into each sample bottle to ensure correct sample identification.

All benthic samples (60) were submitted to Zaranko Environmental Assessment Services (ZEAS) in Nobleton, Ontario. The QA/QC procedures and methods for the benthic component are outlined in Section 2.3.

Upon arrival at the ZEAS laboratory, benthic samples were checked to ensure that they were adequately preserved in the field and clearly and correctly labelled. Prior to detailed sorting, the samples were washed free of formalin in a sieve of the appropriate size. At this time, a stain was added to the samples to aid in sorting recovery. No problems with preservation or sample labelling were reported. The material retained by the sieve was sorted with the aid of a stereomicroscope at a magnification of ten times. Benthic invertebrates were sorted from the debris into major taxonomic groups (*i.e.*, order or family levels) and placed in vials containing 70% ethanol. The benthic invertebrates were then identified to the lowest practical level, which in most cases was genus or species, and enumerated by a senior taxonomist.

2.1.4 Supporting Measurements Associated with Benthic Community Sampling

At each benthic community sample station, a number of supporting measurements were taken, both at the surface (30 cm below surface) and bottom (50 cm above bottom) of the water column, including temperature, dissolved oxygen, pH and conductivity (Appendix Table B.1). These measurements were made using a YSI 556 or a YSI 85 Multimeter. At Station 2 in each lake, a temperature and dissolved oxygen profile was taken at multiple depths from the surface to bottom of the water column to determine stratification conditions and the depth of the thermocline. Accuracy of the meters was assured by daily calibration and frequent verification to achieve performance specifications (Table 2.9). In any case where verification or calibration failed to meet known values, the meter/probe was either re-calibrated or replaced, if possible. All manufacturers' instructions for maintenance and calibration of multimeters were followed at all times. If meter failure occurred, backup procedures included the measurement of temperature using a thermometer, dissolved oxygen using a Hach Kit (which gives an estimation of dissolved oxygen concentration to ± 0.1 mg/L), and pH using pH strips. Any incidence of meter failure and the use of these backup measures were recorded on the field sheets (Appendix E). During the field program the conductivity measurement on the YSI 85 meter would not calibrate properly and therefore conductivity could not be measured at some lakes (Dunlop, McCabe, Quirke, Semiwite, and Ten Mile).

Other field observations included weather conditions, water depth, any deviations from standard sampling gear and conditions, details of unusual events and habitat conditions. A

Table 2.9: Data quality objectives and specifications for field equipment.

Equipment	Operator	Field pH			Conductivity (uS/cm)				Dissolved Oxygen (mg/L)				Temperature (°C)		
		MDL (DQO 0.1)	MDD (DQO 0.01)	Accuracy (DQO 10%)	MDL (DQO 0 uS/cm)	Measurement Range	MDD (DQO 1 uS/cm)	Accuracy (DQO 10%)	MDL (DQO 0 mg/L)	Measurement Range	MDD (DQO 0.01)	Accuracy (DQO 20%)	MDL (no DQO stipulated)	MDD (DQO 0.1)	Accuracy (DQO 20%)
Orion pH Meter	DES	0.01	0.01	± 0.01									1.0	-	± 0.01
Omega PHH-320	DES	0.01	0.01	± 0.02									1.0	-	± 0.05
YSI 85	Minnow/ DES	0.01	0.01	<20%[a]	0	0 to 499.9	0.1	0.5%	0	0 to 20	0.01	± 0.03	-5.0	0.1	± 0.4°C
					0	0 to 4,999	1.0	0.5%							
					0	0 to 49,999	10	0.5%							
					0	0 to 200,000	100	0.5%							
YSI 556	Minnow	0.01	0.01	<20%[a]	0	0 to 2,000	2	3%	0	0 to 20	0.01	<10%[b]	-5.0	0.1	± 0.1°C
					0	2,000 to 20,000	10	3%							
					0	20,000 to 100,000	50	4%							
Hach Kit	Minnow								0	0 to 10	0.1	± 0.1			

[a] Instrument accuracy reported as ±0.01 to 0.04 pH units, depending on model. Reported accuracy measurements greater than 3 mg/L. Lowest value measured between September 1999 and September 2004 was 3.2.

[b] Instrument accuracy reported as ±0.3 mg/L. Reported percentage assumes dissolved oxygen measurements greater than 3 mg/L. Lowest value measured between September 1999 and September 2004 was 6.5 mg/L.

DES - Denison Environmental Services

MDL - Method Detection Limit

MDD - Minimum Detectable Difference

DQO - Data Quality Objective

GPS was used to record the Universal Transverse Mercator (UTM; NAD 83) position of all stations (Appendix Table A.1).

2.2 Data Entry and Extraction

Water data generated through the various monitoring programs were entered into an electronic database (emLine). Data entered or imported with any values outside the established data quality assessment limits were highlighted. Prior to being accepted (*i.e.,* posted) in the database, any highlighted data were reviewed and validated through a QA process (see procedures PR8.7.3-01, PR8.7.3-02 and PR8.7.2-02 Appendix A).

Monthly and annual data reports were generated from the database to meet reporting requirements for various regulatory programs. The data retrieval is managed by Denison Environmental Services (DES), the care and maintenance contractor for both of the licensees. Retrieval methods and rationales employed by DES to satisfy data requests are described in Appendix A. The nature of the data retrieval request can affect the type and configuration of the data reported from the emLine system. For this reason, summary statistics presented in this report (e.g., sample sizes, annual means) may vary slightly from annual means presented in the Annual Operating, Care and Maintenance (OCM) Reports. For example, reported annual OCM averages are based on data collected solely for “regulated” monitoring and reporting; whereas the data extracted for this report included all available data (e.g., also “Internal” & “Special Project” data).

Data extracted from field sheets (SRWMP) were entered into Excel spreadsheets, and checked by a second person to assure no errors were made in the data entry process. Laboratory results for sediment samples were reviewed relative to submission Chain of Custodies (COCs), method detection limits (MDLs) and Data Quality Objectives (DQOs). Laboratory data was copied and/or entered into Excel spreadsheets, which again was checked and verified for accuracy by a second reviewer. Benthic invertebrate data was provided in Excel spreadsheets, so re-entering of data was not required. For the special investigation, all raw data was provided directly to EcoMetrix, where it was then entered into tables and used in the risk assessment (Appendix F).

2.3 Data Quality Control and Assessment

A variety of factors can influence the chemical measurements made in environmental monitoring and thus affect the accuracy and precision of the data. Inconsistencies in sampling or laboratory methods, use of instruments that are inadequately calibrated or which cannot measure to the desired level of accuracy and contamination of samples in the field or laboratory are just some of the potential factors that can lead to the reporting of data that do not accurately reflect actual environmental conditions. Depending on the magnitude of the problem, this has potential to affect the reliability of any conclusions made from the data. Therefore, it is important to ensure that monitoring programs incorporate appropriate steps to control the non-natural sources of data variability (*i.e.*, minimize the variability that does not reflect natural spatial and temporal variability in the environment) and thus assure the quality of the data.

There are data quality objectives (DQOs) and procedures (e.g. PR8.5.4-01 in Appendix A) for each of the monitoring programs (the SAMP, the TOMP and the SRWMP) to ensure data generated from these programs are representative of conditions at specific monitoring locations and times. DQOs are statements of desired sensitivity, precision and accuracy and are used to assess data acceptability. In other words, DQOs determine the level of confidence with which the data can be used to derive conclusions. DQOs previously established for the SAMP, TOMP and SRWMP (Tables 2.10 and 2.11) consider the intended use of the data and the technical feasibility of collecting data of such quality.

DQOs for water samples included negligible contaminant levels in all blanks and rinses, acceptable variability between field duplicates and laboratory replicate samples, efficient recovery from spikes and minimal bias in analytical estimates for certified reference materials. DQOs respecting field and laboratory duplicates, as well as matrix spike recoveries were also established for sediment samples.

Quality assurance/quality control (QA/QC) practices for benthic invertebrate sampling followed Environment Canada (2002) guidance for sub-sampling precision and sorting recovery. Duplicate sub-samples were analyzed for at least 10% of samples to verify that sub-sampling precision was within 20% (Table 2.10). Ten percent of the samples were also re-sorted to verify that less than 5% of total organisms were missed (sorting recovery).

Toxicity test QA/QC involved adherence to requirements defined in (Aquatox's) internal standard laboratory protocols and in toxicity methods (EPS 1/RM/32, Environment Canada 1997b; EPS 1/RM/33, Environment Canada 1997a). These pertained to aspects such as organism health/culturing, data entry, reference toxicant testing, control of test conditions, and report completeness. In addition, there were specific validity criteria specified by the test methods, such as minimal control organism mortality and achieving minimum organism growth requirements.

Table 2.10: Data quality objectives for the SRWMP.

Measurements	Units	Detection Limit	Field & Lab Blank Criterion	Analytical Precision (Duplicates)	Analytical Accuracy		Field Precision (Duplicates)
					Spike	CRM[b]	
Field Measurements							
pH	pH units	0.1	-	0.01 or 0.02[a]	-	-	10%
conductivity	µmho/cm	0.01	-	0.05[a]	-	-	10%
dissolved oxygen	mg/L	0.1	-	0.03[a]	-	-	20%
temperature	°C	0.1	-	0.01 or 0.05[a]	-	-	20%
flow	L/s	varies w method	-	0.1[a]	-	-	30%
Laboratory Water Chemistry							
barium	mg/L	0.005	0.01	10%	20%	20%	20%
cobalt	mg/L	0.0005	0.001	10%	20%	20%	20%
iron	mg/L	0.02	0.04	10%	20%	20%	20%
manganese	mg/L	0.002	0.004	10%	20%	20%	20%
radium-226	Bq/L	0.005	0.01	20%	20%	-	20%
sulphate	mg/L	0.1	0.2	10%	20%	20%	20%
uranium	mg/L	0.0005	0.001	10%	20%	20%	20%
Laboratory Sediment Chemistry							
barium	mg/kg	0.5	-	20%	30%	30%	40%
cobalt	mg/kg	0.2	-	20%	30%	30%	40%
iron	mg/kg	20	-	20%	30%	30%	40%
manganese	mg/kg	0.5	-	20%	30%	30%	40%
nickel	mg/kg	0.5	-	20%	30%	30%	40%
radium-226	Bq/kg	5	-	20%	30%	30%	40%
uranium	mg/kg	0.1	-	20%	30%	30%	40%
grain size	%	0.1	-	20%	30%	30%	40%
TOC	%	0.05	-	20%	30%	30%	40%
Benthos							
Organism Recovery		-	-	90%	-	-	-
Subsampling Precision		-	-	20%	-	-	-
Subsampling Accuracy				20%			
Sediment Toxicity							
Chironomus dilutus		-	70% control surv.	20% control CV	-	± 3 SD in ref tox	-
Hyalella azteca		-	70% control surv.	20% control CV	-	± 3 SD in ref tox	-

[a] Minimum Detectable Difference as identified in instrument manual rather than measurement of analytical precision using replicate samples.

[b] CRM (Certified Reference Material).

Table 2.11: Field and laboratory data quality objectives for SAMP/TOMP stations.

Parameter	Units	Receiving Environment Criteria		Targeted Detection Limit	Minimum Detectable Difference	Field Blank Criteria	Laboratory Blank Criteria	Field Precision	Laboratory Precision	Laboratory Spikes	Laboratory Accuracy (CRM)
Field Parameters											
Conductivity	µmho/cm	-	c	0.1	0.05	-	-	20%	-	-	-
Flow	L/s	-	c	method	method	-	-	-	-	-	-
pH	pH units	6.5 - 8.5	a	0.1	0.01 or 0.02	-	-	20%	-	-	-
Laboratory Parameters											
Acidity	mg/L	-	c	1.0	-	2	2	20%	10%	-	20%
Barium	mg/L	0.0531	b	0.005	-	0.01	0.01	20%	10%	20%	20%
Cobalt	mg/L	0.0009	a	0.0005	-	0.001	0.001	20%	10%	20%	20%
Iron	mg/L	0.87	b	0.02	-	0.04	0.04	20%	10%	20%	20%
Manganese	mg/L	0.8	d	0.002	-	0.004	0.004	20%	10%	20%	20%
Radium	Bq/L	1.0	a	0.005	-	0.01	0.01	20%	20%	20%	-
Sulphate	mg/L	100	d	0.1	-	0.2	0.2	20%	10%	20%	20%
TSS	mg/L	-	c	1	-	2	-	20%	10%	-	20%
Uranium	mg/L	0.005	a	0.0005	-	0.001	0.001	20%	10%	20%	20%

a - Provincial Water Quality Objectives

b - Cycle 2 SRWMP Benchmarks

c - no criteria set

d - British Columbia Water Quality Guidelines (BCMOE 2006)

e - Canadian Water Quality Guidelines (CCME 2003)

Data Quality Assessment (DQA) is the process of evaluating how well laboratory test results compare with pre-established DQOs and thus determines the confidence that can be placed in conclusions derived from the data. A comprehensive data quality assessment was undertaken for the SRWMP, SAMP and TOMP data and is presented in Appendix B.

2.4 Data Evaluation

Numerous types of data were compiled, synthesized and assessed for this project, including:

- Water quality data from TOMP and SAMP, including TMA surface water, seepage, porewater, groundwater, and effluent stations, as well as surface water quality data from SRWMP;
- Other data related to TMA management, including water levels and regent use;
- Effluent toxicity data;
- Flow data from TMA discharges, seepages and within the downstream receiving environment, which were used to compute loadings; and
- Sediment and biological data from the SRWMP.

The approaches followed for analysis of these different types of data are described below.

2.4.1 Water Samples

TMA porewater samples were collected annually, with some samples taken from multiple depths/horizons (typically labelled as A, B, C, D, etc.) per station. Each porewater sample was analyzed for pH, acidity, iron, and sulphate. Conductivity replaced sulphate measurement in 2003 until 2006, but conductivity was discontinued and sulphate analysis was resumed in 2007. All data were tabulated and presented in the appendix corresponding to each TMA. Trend analysis was completed, as described in Section 2.4.3. Significant trends were summarized in tables and all significant trends were plotted and presented in appendices.

Groundwater quality has been monitored on a yearly basis, typically at locations down-gradient of tailings dams. Samples were analyzed for pH, acidity, sulphate and iron. Consistent with porewater, sulphate replaced conductivity in 2007. Trend analysis was completed, as described in Section 2.4.3. Significant trends were summarized in tables and all significant trends were plotted and presented in appendices.

Surface water within the TMA and the SRW was monitored for substances and at frequencies that were specific to the objectives of each monitoring program (*i.e.*, TOMP, SAMP and SRWMP). Concentrations of all variables monitored within TMAs (*i.e.*, in basins), and in effluent, seepages, and downstream surface water stations were compared to SRWMP benchmarks for receiving water quality (described below). It is recognized that mine sources (effluent and seepage) are not expected to achieve criteria for receiving environment quality, but such comparisons were made to identify potential variables or sources of concern relative to the downstream receiving environment. Based on expected minimum 10-fold dilution downstream of the mine discharges, concentrations of 10x the appropriate receiving environment criteria were sometimes presented as the relevant basis for comparison of discharge water quality.

SRWMP benchmarks were based on water quality criteria for protection of aquatic life or the upper range of background (reference area) concentrations (except for pH for which the lower background range was relevant). Water quality criteria that were considered included Ontario's Provincial Water Quality Objectives (PWQO; OMOEE 1994) and Canadian Water Quality Guidelines (CWQG; CCME 2003). For manganese and sulphate, which have no PWQO or CWQG, British Columbia Water Quality Guidelines (BCWQG) were used (BCMOE 2006). The upper range of background concentrations was calculated as (mean + 1.699 * standard deviation; Appendix Table E.1). With the exception of pH, the highest value of the applicable water quality criteria and background concentration was selected as the benchmark for evaluation of water quality at mine-exposed stations (Table 2.12). To detect potential mine-related reductions in water pH, the lower PWQO limit of pH 6.5 was applied in data evaluation instead of the lower background value of 6.0, based on previous input from the CNSC.

2.4.2 Water Elevations and Effluent Treatment Efficacy

TMA elevations were assessed relative to operating levels specified in site-specific Operating Care and Maintenance Plans (Rio Algom sites) and Tailings Management Area Operating Manuals (Denison sites).

The TMA effluent treatment facilities in Elliot Lake neutralize acidity and remove metals through the addition of lime (in most cases) or caustic soda (sodium hydroxide). Barium chloride is also added at most treatment plants for removal of radium-226. Reagent use was evaluated relative to treated effluent volume to assess changes in reagent consumption over time.

Table 2.12: Serpent River receiving environment benchmarks, 2005-2009.

Station		Upper limit of Background[a]	Provincial Water Quality Objective
Barium	mg/L	0.047	-
Cobalt	mg/L	0.0007	0.0009
DOC	mg/L	5.6	
Iron	mg/L	0.47	0.30
Manganese	mg/L	0.098	-
pH	pH units	6.3	6.5
Radium	Bq/L	0.006	1.0
Sulphate[b]	mg/L	6.3	100
Uranium	mg/L	0.0006	0.005

Shaded value indicates selected benchmark.

[a] Upper limit of background based on data collected from reference stations 2005 - 2000 (Appendix Table E.1)

[b] BCMOE sulphate guideline used as there is no PWQO for sulphate (BCMOE 2006).

Routine toxicity testing is conducted as an additional measure of the quality of treated water released from the TMAs. Semi-annual acute lethality tests are performed using rainbow trout (Environment Canada 2000b) and *Daphnia magna* (Environment Canada 2000a), while 1-week survival and reproduction tests are performed using *Ceriodaphnia dubia* (Environment Canada 2007).

2.4.3 Trend Analysis

Analyses of temporal changes in water quality were performed on data from all surface water, seepage, porewater and groundwater stations. Specifically, trends were assessed for porewater and groundwater stations for the period 1990 to 2009 based on pH, sulphate and iron levels. While acidity is also measured in porewater and groundwater, changes in analytical methods in 2006 precluded the use of prior data and such that there were too few data to conduct trend analysis. Surface water and seepage quality trends during the period 2003-2009 were also assessed for all SAMP and TOMP locations based on radium-226, sulphate, uranium, pH, barium, cobalt, iron, manganese, and acidity (TOMP only). Trends were assessed for all SRWMP stations for the period 2000 to 2009 based on concentrations of pH, radium-226, sulphate, uranium, barium, cobalt, iron and manganese.

Prior to trend analysis, concentrations reported as less than the method detection limit (MDL) were replaced with concentrations representing one-half the MDL for that variable. In some cases, method detection limits varied over time (e.g., cobalt), which had the potential to alter or mask actual trends, so detectable concentrations that were less than the maximum MDL were also taken at half the maximum MDL. Abnormally high MDLs were not used as the maximum MDLs, but rather were removed prior to the trend analysis.

Station sampling frequency varied from annual to weekly, depending on the monitoring program and specific location being sampled (Tables 2.3, 2.4 and 2.5). For variables measured more frequently than annually, seasonal variability in concentrations needed to be considered in assessing trends over time. This necessitated that data for each variable and station be organized into common time periods across years, ranging from monthly to annual (depending on the monitoring frequency for each variable at each station), which are hereafter referred to as “seasons”. For stations sampled weekly, monthly averages were computed and months represented “seasons”. In some cases, data for two or more months were grouped into a “season” (if different months were sampled within a “season” in different years) and/or data were averaged (if multiple values existed within a defined “season” within a given year). Therefore, there were as few as one or as many as 12 “seasons” of data for a given variable and monitoring station. Trend analysis was performed if there were >7 years (SRWMP), or >5 years (SAMP and TOMP) of concentrations reported within a season.

Trends were separately analyzed for each season using Spearman rank correlation (r_s) between variable concentrations and years (SPSS 2006; McLeod et al., 1991). This identified any statistically-significant temporal trends within seasons. Rank correlations do not require normally distributed data, and a significant correlation does not necessarily imply a linear increasing or decreasing trend. However, results do indicate where a significant increase or decrease in concentration has occurred over time.

For locations and variables for which multiple seasons were assessed for significant correlations (trends), van Belle tests were applied to test for differences among seasonal trends, and test the common (combined) trend over all seasons. Van Belle and Hughes (1984) and Gilbert (1987) describe application of the tests to the Mann-Kendall statistic (S); Paine (1998) describes application of the tests to Spearman rank correlations (r_s). First, trend correlations for each season were divided by their standard errors (SE) to convert them to standard normal deviates (Z_i). For Spearman r_s, SE=$\frac{1}{\sqrt{n-1}}$, where n=the number of years included in the trend analysis, and:

$$Z_i = r_s\left(\sqrt{n-1}\right)$$

Trend Z values were then compared among the m seasons using van Belle tests for homogeneity of trends:

$$\chi^2_{\mathrm{H}} = \sum\left(Z_i - \overline{Z}\right)^2$$

with df=m–1 for χ^2_{H}. The common trend over all seasons was then tested using:

$$\chi^2_{\mathrm{T}} = \overline{Z}^2 m$$

with df=1 for χ^2_{T}. Mean trend correlations ($\bar{r}_s$) were then calculated by weighting r_s by1/SE= $\sqrt{n-1}$. Van Belle and Hughes (1984) suggest that common trends should not be tested when differences among seasons (*i.e.*, χ^2_{H}) are significant at p<0.01. In this study, common trends were tested and $\bar{r}_s$ calculated for all stations and variables, but cases where χ^2_{H} was significant at p<0.05 were noted. For (seasonal and common) trend analysis where the number of years was less than 10, the *p*-value was obtained from the table of critical values (Zar 1984). Common trends for each station and for each variable were tabulated with significant trends highlighted.

2.4.4 Loadings Estimates

Annual loadings (2005 to 2009) of various monitored variables were developed for:

- TMA direct (controlled) discharge locations;
- TMA seepage locations; and
- Downstream locations within the Serpent River Watershed.

Loadings were computed to compare contributions from background sources and TMAs, and to assess the relative contribution of each TMA and the cumulative loads at downstream locations throughout the watershed.

Loadings from TMA discharge locations were based on monitoring results (flow and concentration) for each year (2005 to 2009). Weekly flow and concentration data measured during discharge periods at the main TMA discharge locations (2005-2009) were used to calculate weekly loads (kg/wk or Bq/wk). Weekly loads were summed to estimate annual loads for each variable. In some instances, loads were computed by averaging concentrations for dates immediately before and after a date when flow but no concentration data were available.

Flows for seepage locations were based on either design flows reported in the EIS documents or mean flows from site monitoring data, whichever was higher[1] (Table 2.13). These flow rates were multiplied by mean annual concentrations (2005 to 2009) for the same station to roughly estimate annual loads for each variable.

Loadings were also estimated for 14 monitoring stations within the SRW which were located either upstream or downstream of various TMA sources. Loadings were estimated by pro-rating data from a Water Survey of Canada (WSC) flow gauging station (02CD006 Serpent River upstream of Quirke Lake) based on watershed areas. Watershed areas were taken from previously published reports or from historical WSC data for each of the downstream locations (Table 2.14). Mean annual flow was determined for each year (2005 to 2009) at

[1] The design flow was used at P-03 as it was believed to be more representative of annual average conditions.

Table 2.13: Non-point source discharge design and measured flow values.

TMA	SAMP Station	Purpose	Receiver	Design Flow	Measured Flow Data							Design Flow Reference
					Mean	Minimum	Maximum	SD	Count	Length of Record		
										Starting Date	Final Date	
				(L/sec)	(L/sec)	(L/sec)	(L/sec)					
Panel	P-02	Seepage from Dam B	Rochester Creek	2	0.8	0.1	5.0	1.1	42	9/12/1991	10/12/1994	Table 6.2.4 -Quirke & Panel EIS[b]
	P-03	Pond C discharge -SW	Rochester Creek	10.7	24.3	5.9	54.4	26.3	3	4/27/2009	10/26/2009	Table 6.2.4 -Quirke & Panel EIS[b]
	P-05	Drainage downstream of Dam E	Rochester Creek	8.03	no flow data							Table 6.2.4 -Quirke & Panel EIS[b]
	P-11	Site drainage	Panel Creek P-26	NA	21.8	0.0	155.8	34.04	20	1/24/2005	10/26/2009	
Quirke	ECA-398	Site drainage	Serpent River Upstream of Q-09	[d]	1.6	0.0	10.0	2.43	39	1/10/2005	10/13/2009	
	Q-22	Site drainage	Serpent River Upstream of Q-09	[d]	8.9	0.5	50.0	12.52	20	1/10/2005	10/13/2009	
	Q-23	Swamp Downstream of Dam K	Dunlop Lake	[d]	46.7	2.7	129.7	71.95	3	5/5/2009	10/21/2009	
	Q-27	Seepage from Dam J	Evans Lake	0.1	0.0	0.0	0.0	0.00	32	2/2/1991	2/3/2000	Table 6.2.2 -Quirke & Panel EIS[b]
Lacnor/Nordic[a]	All sources captured through monitoring at N-12 thus no non-point source discharge											
Milliken	All sources captured through monitoring at MPE thus no non-point source discharge											
Stanleigh[a]	All sources captured through monitoring at CL06 thus no non-point source discharge											
Spanish American	All sources captured through Denison TMA thus no non-point source discharge											
Pronto	LL-01	Upstream Source to Lake Lauzon	Lake Lauzon		10.1	1.1	30.0	8.85	23	1/5/2005	10/14/2009	
Denison	D-3	Lower Williams Lake Discharge	Serpent River Upstream of D-5	0.3	8.1	0.0	161.0	14.04	640	1/4/2005	12/29/2009	Table 6.2.2 -Denison & Stanrock EIS[c]
	D-9	Seepage at Dam 17	Quirke Lake	3.4	3.5	1.3	10.8	2.74	20	1/4/2005	10/6/2009	Table 6.2.2 -Denison & Stanrock EIS[c]
	D-16	Seepage at Dam 9	Quirke Lake	0.3	1.3	0.2	5.7	1.68	20	1/4/2005	10/6/2009	Table 6.2.2 -Denison & Stanrock EIS[c]
Stanrock	DS-16	Drainage from Dam G and J	Quirke Lake	0.7	4.0	0.0	57.8	6.46	308	1/4/2005	10/5/2009	Table 6.2.2 -(Dams B, C, D)Denison & Stanrock EIS[c]

shade denotes the flow values used for loading calculations presented within the SOE for seepage locations

[a] some Lacnor mine site, Stanleigh mine site and Stanleigh Dam A seepage reports to the MPE watershed but these are accounted for in MPE loadings from Milliken

[b]. Tables 6.2.2 and 6.2.4 (Rio Algom Limited 1995)

[c] - Table 6.2.2 - Estimated Long Term Values (Denison Mines Limited 1995)

[d] - specific predictions for seepage or runoff flow from these areas were not included in EIS but loadings considered representative of these areas were included in general TMA predictions.

NA - not available

Table 2.14: Watershed areas and prorated flow estimates[a] for stations within the Serpent River watershed, 2005 to 2009.

Station	Description	Watershed Area (Km^2)	Mean Flow (L/s)[a]						Drainage Area Source
			2005	2006	2007	2008	2009	Mean Annual Flow	
SR-01	Quirke Lake Outlet	319	3,280	4,238	3,182	5,661	5,376	4,348	WSC (02CD003)
M-01	Elliot Lake Inlet	18.56	191	247	185	329	313	253	Senes 2007[b]
Q-20	Evans Lake Outlet	1.08	11	14	11	19	18	15	S. Kam e-mail June 14th 2007
DS-18	Halfmoon Lake Outlet	11.6	119	154	116	206	196	158	Table 6.3.3 Denison & Stanrock EIS
SR-05	Canyon Lake Outlet	7.57	78	101	75	134	128	103	Topo map 41 J10
SR-06	McCabe Lake Outlet	32.8	337	436	327	582	553	447	Senes 2007[b]
SR-08	Nordic Lake Outlet	32.3	332	429	322	573	544	440	Senes 2007[b]
D-6	Outlet of Cinder Lake	4.13	42	55	41	73	70	56	Topo map 41 J10
D-4	Outlet of Dunlop Lake	109	1,121	1,448	1,087	1,934	1,837	1,486	WSC (02CD002)
MPE	Outlet of Sherriff Creek Park	13.5	138	179	134	239	227	183	Golder 2004
Q-09	Quirke Lake Inlet	157	1,614	2,086	1,566	2,786	2,646	2,140	WSC (02CD006)
	Serpent River @ Hwy 17	1350	13,263	16,346	12,092	23,558	22,753	17,602	WSC (02CD001)
D-5	Serpent River downstream of Denison	118	1,213	1,568	1,177	2,094	1,989	1,608	Table 6.3.3 Denison & Stanrock EIS
SC-01	Westner Lake Outlet	2.37	24	31	24	42	40	32	Golder Westner Lake Outlet Berm Report

WSC - Water Survey of Canada (Station Identification)

[a] Flows calculated based on mean annual flow data from Quirke Lake Inlet, Water Survey of Canada data.

[b] Data provided by Senes 2007 taken from EIS loading predictions.

each location and pro-rated flow estimates were multiplied by mean annual concentrations to roughly estimate annual loads at SRW monitoring stations.

2.4.5 Sediment Quality

Similar to the approach taken for water quality data, sediment quality data were analyzed to identify variables that were elevated relative to quality benchmarks and to identify locations with elevated concentrations. Spatial patterns were assessed relative to TMA discharges, and where possible, temporal changes were evaluated by comparing 2009 to 2004 and 1999 sediment data. Sediment data were also used in the assessment of the benthic macroinvertebrate communities to identify potential relationships between benthic community composition and sediment quality.

Sediment concentrations were compared to Ontario's Provincial Sediment Quality Guidelines (PSQG; OMOE 1993; iron, manganese), guidelines proposed by Thompson et al. (2005; nickel, uranium, radium-226) and upper background (reference area) concentrations. The upper range of background concentrations was defined as the mean (2009) + (2.145 * standard deviation).

2.4.6 Benthic Invertebrates

Benthic community data evaluation included

- Statistical comparisons of communities downstream of mine discharges relative to reference communities based on key benthic community metrics (density, number of taxa, and first three Correspondence Analysis (CA) axes);
- Correlation analysis of benthic metric and physical-chemical variables to identify potential relationships that might explain reference-exposure benthic community differences, and
- Comparison of Cycle 3 (2009) data to results from Cycles 1 (1999) and 2 (2004).

Benthic invertebrate community data were subjected to a data quality assessment to verify overall data quality prior to their use in data analysis (Appendix B).

Invertebrate density (individuals/m^2) was calculated based on the known area sampled (*i.e.*, 0.232 m^2). The benthic diversity metric "number of taxa" (also known as taxon richness) was calculated based on lowest-practical-level taxonomy, excluding any life stages that could not be conclusively identified as separate taxa.

Benthic invertebrate community structure was also assessed using a multivariate technique known as correspondence analysis (CA). CA extracts “axes”, representing weighted vectors of species abundances, which can be thought of as new variables summarizing community composition. The greatest variation among either taxa or stations is explained by the first axis, with other axes accounting for progressively less variation. The method is influenced by rare species, so those taxa occurring at ≤10% of stations are eliminated from the data sets before analysis, and interpretation of results must consider the potentially biasing effects of those taxa remaining which still are not present at most stations. After screening and data reduction, abundances were log10 (x+1) transformed. Scores for both taxa and stations were calculated using the ADE-4 statistical software package (Thioulouse et al. 1997) to evaluate the associations of organisms and stations.

All benthic invertebrate community metrics were summarized by separately reporting mean, minimum, maximum, standard deviation, standard error and sample size for each study area (*i.e*., lake).

Exploratory evaluation of relative densities, number of taxa and Correspondence Analysis results were used to identify the reference lakes which would best serve to identify any mine-related differences between mine-effluent-exposed and reference lakes. For the resulting reference/exposure comparisons, a pooled reference mean was calculated from the mean values of the reference lakes (n=4 lakes, omitting Rochester Lake for reasons described in Section 5.3.1), and these data were compared to mean values for each exposure lake (n=5 replicate stations) using *a priori*, user-defined contrasts in ANOVA. User-defined contrasts are tests of hypotheses constructed prior to Analysis of Variance (ANOVA). As such, these independent tests, which are conceptually similar to t-tests between two groups, are not adjusted for multiple comparisons, and are more powerful at detecting differences than are *post-hoc*, pair-wise comparisons, especially when many groups are compared *post-hoc*. Since the user-defined contrasts are *a priori* tests, their results trump those of ANOVA and *post-hoc*, pair-wise comparisons. Accordingly, and by convention, ANOVA results are not reported for these comparisons. For all comparisons between areas, heterogeneity of variances was tested with Levene’s test and, when necessary, tests that allow for unequal variances were used when comparing areas.

Benthic invertebrate community surveys in Canada are generally expected to have sufficient power to detect a difference (effect size) of ± two standard deviations (SDs) of the reference mean (Environment Canada 2002). Therefore, for each significant difference between reference and mine-exposed areas, the magnitude of the difference between area means was expressed as the number of reference mean SDs as follows:

magnitude of difference = (exposure mean – reference mean) / SD of the reference mean

Exposure means were also expressed as a percentage of the reference area mean to convey magnitudes of difference between areas.

Correlation analysis was carried out between the five primary benthic metrics (density, number of taxa, CA axes 1-3) and 13 habitat-related variables, including sediment concentrations of mine-indicator substances. With 65 simultaneous comparisons, correlations significant at the p-level of 0.05 should be interpreted cautiously, since several (5%) of the correlations could be expected to occur by chance alone. For this reason, correlations significant at a more stringent level of $p<0.001$ were also noted. All correlations significant at the unadjusted p-level of 0.05 were examined in scatter plots to verify the magnitude and significance of the relationships, which can be falsely inflated by the leveraging effect of outliers.

Ecological and habitat requirements of benthic taxa were considered in data interpretation as supported by standard references (e.g., Merritt and Cummins 1996; Weiderholm 1983; Wiggins 1996).

2.5 Special Investigation

A special investigation was undertaken to allow for better estimates of dose and risk by taking measurements to confirm or adjust assumptions made in previous dose and risk estimates. Risk assessments have been conducted in the watershed as part of the Environmental Assessments conducted in support of mine decommissioning (Rio Algom 1995, Denison Mines 1995, AECB 1997. CNSC 2002), the 1999 SRWMP (Minnow and Beak 2001) and the State of the Environment Report (Minnow 2009a). Within the receiving environment, estimates of dose and risk have been based on a number of assumptions with respect to:

- Secular equilibrium of lead-210 and polonium-210 with radium-226 in sediment.
- Negligible contribution of the thorium-232 decay chain to dose estimates;
- Bioaccumulation factors in fish;
- Resource use and consumption by local First Nations persons; and

- Occupancy of downstream lakes by waterfowl.

A detailed study was conducted in 2009 to confirm these assumptions and generate comprehensive dose estimates based on measured data. The study focused on six lakes for which human health risks were estimated as part of the SOE (Quirke, McCabe, Nordic, Elliot, May, and McCarthy). In each lake, water, sediment, forage fish, and macrophytes were sampled and analyzed for U-nat, Th-230, Ra-226, Po-210, Pb-210, Th-232, Th-228, and Ra-228 (*i.e.* both the uranium-238 and thorium-232 decay chains). A complete description of the methods employed in the collection of these samples is provided in Appendix F (EcoMetrix 2011a).

Dietary intake and usage by First Nations people was documented through a consumption survey conducted by SRFN fishers and hunters and their families (SRFN, 2010). Interviews were conducted with 21 fisher/hunter respondents selected to be representative of the community. Each respondent reported: number of household residents, annual household consumption of fish, waterfowl and other game (by species), and harvest distribution by species and location. Household consumption was divided by the number of household residents to estimate the annual consumption per person in each household. A detailed description of the survey is provided in Appendix F (EcoMetrix 2011a).

In addition, an assessment of waterfowl usage in the key lakes noted above (Quirke, McCabe, Nordic, Elliot, May, and McCarthy) was conducted in the fall of 2009 (i.e. when waterfowl are staging). Observations by field crew on the species and number of waterfowl present took place over one or two days per lake; the time required for completion of water, sediment, macrophyte, and fish collection. At Quirke Lake however, three days were spent collecting samples and therefore waterfowl observation was extended to three days. Field crews generally surveyed most of the area within each lake with the exception of McCarthy where access to a western portion of the lake was difficult due to the presence of a beaver dam.

The information from this study was used to update the human health risk assessment prepared for the SOE report, provide an estimate of dose and risk to aquatic biota and riparian wildlife within these lakes and address the specific assumptions used in previous dose estimates. A detailed description of the method used to estimate dose and risk to aquatic biota, riparian wildlife and human receptors is provided in Appendix F (EcoMetrix 2011a).

3.0 TMA PERFORMANCE

Within the Serpent River Watershed there are eleven TMA's, although one of these, Spanish American, discharges to the Denison TMA complex. TMAs have either a vegetative cover or a water cover, both of which are intended to inhibit oxidation and acidification of tails. In water-covered TMAs (flooded) excess water flows from the TMA to an effluent treatment plant prior to discharge. In vegetated TMAs, seepage from the TMA is collected in pond structures or ditches and treated prior to discharge.

The performance of the TMAs is monitored and assessed through the TMA Operational Monitoring Program (TOMP) which includes the assessment of:

- Water cover on flooded basins;
- Surface water quality within the basins;
- Porewater quality within the basins (where monitored);
- Groundwater quality down-gradient of the TMAs; and
- Treatment performance (reagent use and effluent compliance).

Releases to the environment are monitored under the Source Area Monitoring Program (SAMP) which captures site drainage, seepages, and final effluent. Releases are discussed in the context of common sub-watersheds within the SRW in Section 4.0.

Performance of each TMA is presented in the following sections.

3.1 Denison TMA

3.1.1 Basin History and Modifications

The Denison mine and mill operated from 1957 to 1992. Over this time, a total of 63 million tonnes of uranium ore were milled. Tailings were deposited into two bedrock-lined basins, TMA-1 (formerly Bear Cub Lake and Long Lake) and TMA-2 (formerly Upper Williams Lake). Tailings in TMA-2 are contained by an engineered dam to the northwest (Dam 1) and bedrock between TMA-2 and TMA-1 (Figure 3.1). TMA-2 was used from start-up until it was filled in the early 1960s. After TMA-2 was filled, tailings were discharged into the Bear Cub Lake basin, which eventually merged with the Long Lake basin to form TMA-1. Sixty million tonnes of tailings are contained in TMA-1 by five engineered perimeter dams (Dam 9, Dam 10, Dam 16, Dam 17 and Dam 18) representing a total area of approximately 240 ha (Figure 3.1). In general, the Denison TMAs were decommissioned as flooded tailings following mine closure in 1992, with decommissioning largely completed in late 1996. Specifically, from 1992 to 1995 beached tailings on the east side of TMA-1 were hydraulically dredged and

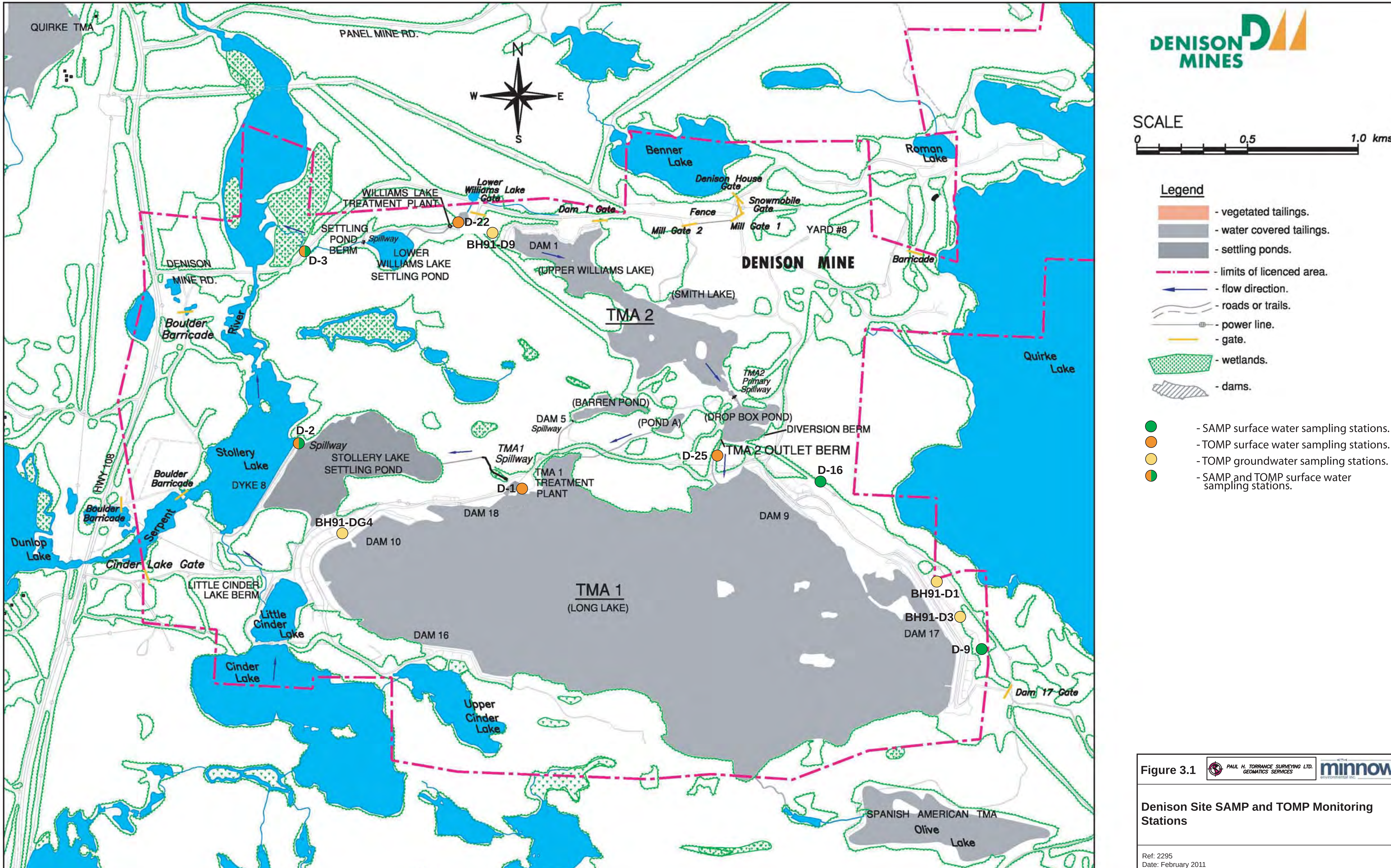

Figure 3.1

Denison Site SAMP and TOMP Monitoring Stations

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placed into deeper areas on the west side of TMA-1. From 1993 to 1996, tailings from TMA-2 were hydraulically relocated to TMA-1 and to the underground workings, leaving a total of 3.3 million tonnes of tailings in TMA-2 within an area of 40 hectares. In addition, all tailings on the rock shoreline were washed into the TMA-2 basin in 1997. The Dam 10 stability and seepage reduction berms were completed by 1996. The stabilization of the remaining dams in TMA-1 for closure was also completed by 1996.

Effluent/decant from TMA-2 flows into TMA-1 via the TMA-2 spillway. Seepage from TMA-2 is treated at the Lower Williams Lake Treatment Plant and discharged to the Serpent River at D-3. The Denison Effluent Treatment Plant (ETP) is located on the north shore of TMA-1 where effluent is treated prior to discharge to Stollery Lake, which then discharges into the Serpent River (Figure 3.1).

Within the Denison TMA, surface water and ground water are monitored under the TOMP and the locations, substances and frequency monitored are specific to the station type (Table 3.1; Figure 3.1). Data from the TOMP stations are summarized in the following sections and presented in Appendix C (Appendix Tables C.1.2- C.1.8).

3.1.2 Water Management

Water cover at the Denison TMA is used to inhibit oxidation and acidification of tailings. Since 2005 (start of reporting period), water levels were consistently above the minimum operating level of 9144.5 ft (Figure 3.2). Water levels were highest in 2008 and 2009 due to higher precipitation in these years.

3.1.3 Basin Surface Water Quality

Surface water quality is monitored at three stations: the ETP influent from TMA 1 (D-1) and TMA 2 (D-22) and the overflow between TM2 and TMA 1 (D-25; Figure 3.1).

Since decommissioning (1992 to 1997), concentrations of radium-226, sulphate and uranium have decreased and pH has remained neutral (Figure 3.3). Concentrations of radium-226 and sulphate are near the 50-year post-decommissioning predictions (*i.e.* 2040) (Figure 3.3).

More recently (2003-2009), radium-226 has increased and pH decreased in Denison TMA-1 (Table 3.2). These trends appear to be attributed to a step change in 2008, which may be associated with the decrease in sulphate over time (i.e. since 2000) and/or the higher water levels in 2008 and 2009 (Appendix Figures C.1.1 and C.1.3) compared to a relatively dry

Table 3.1: TOMP monitoring stations, substances, and frequencies[a] at Denison TMA.

			Parameters and Frequencies[a]										
TMA	**TOMP Stations**	**Station Type/Purpose**	**Elevation**	**Flow**	**pH**	**Sulphate**	**Total radium-226**	**Lime or NaOH Consumption**	**Barium Chloride Consumption**	**TSS**	**Acidity**	**Iron**	**SAMP Metals[b]**
Denison	D-1	Basin performance (primary), ETP operations	W	D	D	Q	M	M	M		Q		Q
	D-22	ETP operations			W	Q	M		M		Q		Q
	D-3	Effluent		D[c]	W	M	W			W			M[d]
	D-2	Effluent		D[c]	W	M	W			W			M[d]
	D-25	Basin performance (secondary)			S	S	S				S	S	
	BH91-D1A,B, BH91-D3A,B, BH91-DG4B, BH91-D9A	Groundwater			A	A					A	A	

[a] D - Work days, W - Weekly, M - Monthly, S - Semi-annually, A - Annually, Q-Quarterly

[b] SAMP metals are barium, cobalt, iron, manganese, and uranium

[c] Monitoring requirement of SAMP

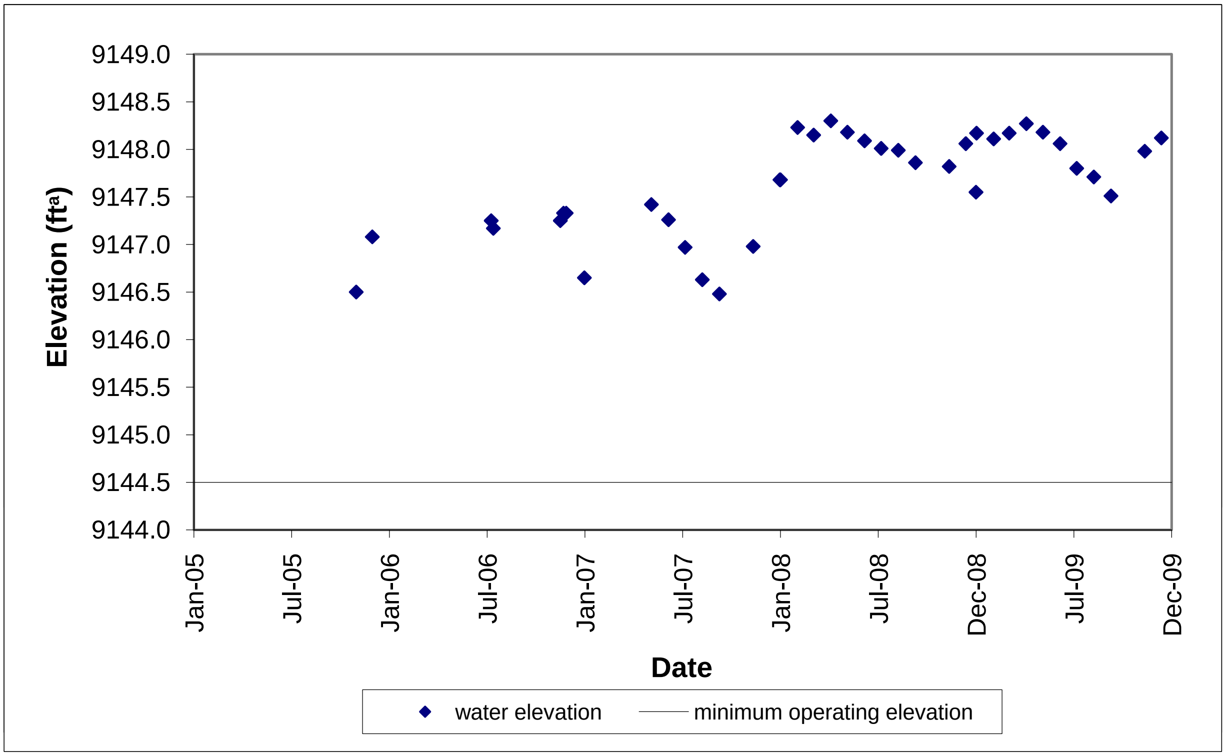


Figure 3.2: Water level at Denison TMA-1 relative to minimum operating elevation.

[a] Elevation is based on Denison Mine datum. Historically Quirke Lake was given an arbitrary elevation of 9,000 feet and all elevations are relative to that value.

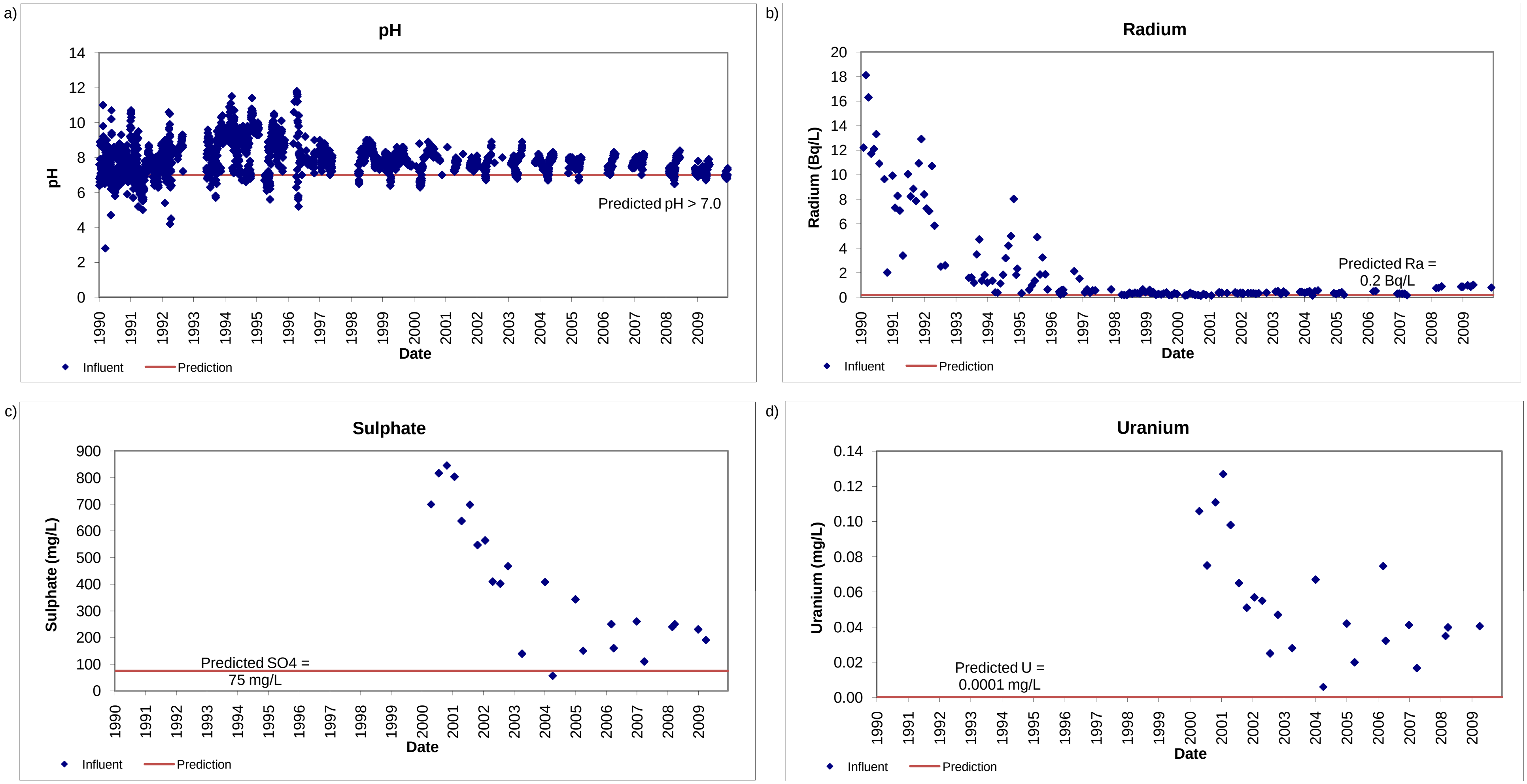


Figure 3.3: Water quality at the Denison TMA-1 ETP influent (D-1) relative to predictions for 50 years (2040) post-decomissioning.

Table 3.2: Summary of water quality trends[a] at TOMP monitoring stations, Denison TMA, 2003 to 2009.

Station ID	Type/Location	Number of Seasons Used in Common Trend[c]	Acidity	Barium	Cobalt	Iron	Manganese	pH	Radium-226	Sulphate	Uranium
D-1	TMA-1 Influent	1 to 4	-[b]	-	-	-0.559	-	-0.586	0.500	0.679	0.700
D-25	Spillway between TMA-1 and TMA-2	2	-	-	-	-	-	0.048	0.378	-	-
D-22	Influent to ETP at TMA-2	4 to 12	-	-	-	-0.016	-	-0.014	-0.267	-0.567	-[d]

decreasing trend, significant at $p<0.05$

increasing trend, significant at $p<0.05$

[a] Based on rank correlation coefficients (rho) for common (combined) season trends, shown in table.

[b] "-" denotes that this parameter was not included in the trend analysis for that particular station due to insufficient data (e.g. there were <5 years worth of data for that parameter)

[c] Seasons used varied for substances based on suitability of data for trend analysis

[d] high MDLs for Uranium from 2003 to 2005 precluded ability to statistically assess trends at this station

2007. However, data during this period is limited due to minimal flows through the TMA[2] and additional data will be required to verify the trend and, if necessary, determine the cause. To address this issues, DMI has implemented additional monitoring in TMA-1 such that the basin water quality will be monitored (pH sulphate and radium-226) during periods of zero discharge. Radium-226 and pH levels at D-1 achieve PWQO before treatment and are much lower than values observed immediately following closure (Figure 3.3). Within TMA-2, radium-226 and sulphate concentrations have been decreasing over time (Table 3.2).

3.1.4 Groundwater Quality

Four locations (wells) are sampled annually for iron, pH, sulphate and acidity; two are located down-gradient of Dam 17 (BH91-D1 and BH91-D3), one is down-gradient of Dam 1 (BH91 D-9), and one is down-gradient of Dam 10 (BH91-DG4; Figure 3.1).

Down-gradient of Dam 17 at the east end of TMA-1 groundwater quality has significantly improved since decommissioning (1991-2009), with iron concentrations decreasing and pH levels increasing to neutral levels. However, down-gradient of Dam 10 at the west end of TMA-1, pH in groundwater has been decreasing (Table 3.3) consistent with pH in surface water within the basin (Station D-1, Table 3.2).

Down-gradient of Dam 1 in TMA 2 (BH91-D9A) groundwater quality has not improved over time, based on concentrations of iron that have significantly increased while pH levels have decreased (Table 3.3; Appendix Figure C.1.5).

3.1.5 Treatment Performance

The primary ETP for the Denison TMA is located at the outlet of TMA-1 with a second ETP at TMA-2 to treat seepage from this basin as well as from a historical tailings spill (Figure 3.1). The TMA-1 ETP uses both caustic soda and barium chloride to reduce acidity and radium-226, respectively. Generally, barium chloride and caustic soda consumption (kg/yr) was higher in 2008 and 2009, which is likely associated with lower pH and increased radim-226 in TMA-1 influent during this period. In addition, higher precipitation in these years caused the ETP to operate for more days which also contributed to the increase in reagent consumption (Figure 3.4). Caustic soda was not used in 2007 as no treatment for pH was required (pH was 7.8).

[2] Influent water chemistry is not monitored when the TMA is not discharging.

Table 3.3: Summary of water quality trends[ab] in TOMP groundwater in Denison TMA, 1991[d] to 2009.

Location	Station	Depth (m)	Dates	Iron	pH	Sulphate
Downgradient of Dam 1 (TMA-2)	BH91 D9A	22	1991-2009	0.913	-0.756	-[c]
Downgradient of Dam 10 (TMA-1)	BH91 DG4B	10.9	1996-2009	0.481	-0.736	-
Downgradient of Dam 17 (TMA-1)	BH91 D1B	45	1991-2009	-0.067	0.510	0.664
	BH91 D1A	66	1991-2009	-0.729	0.867	-0.582
Downgradient of Dam 11 (TMA-1)	BH91 D3B	21	1991-2009	-0.565	0.892	-0.515
	BH91 D3A	48	1991-2009	-0.511	0.804	-0.582

decreasing trend, significant at $p<0.05$

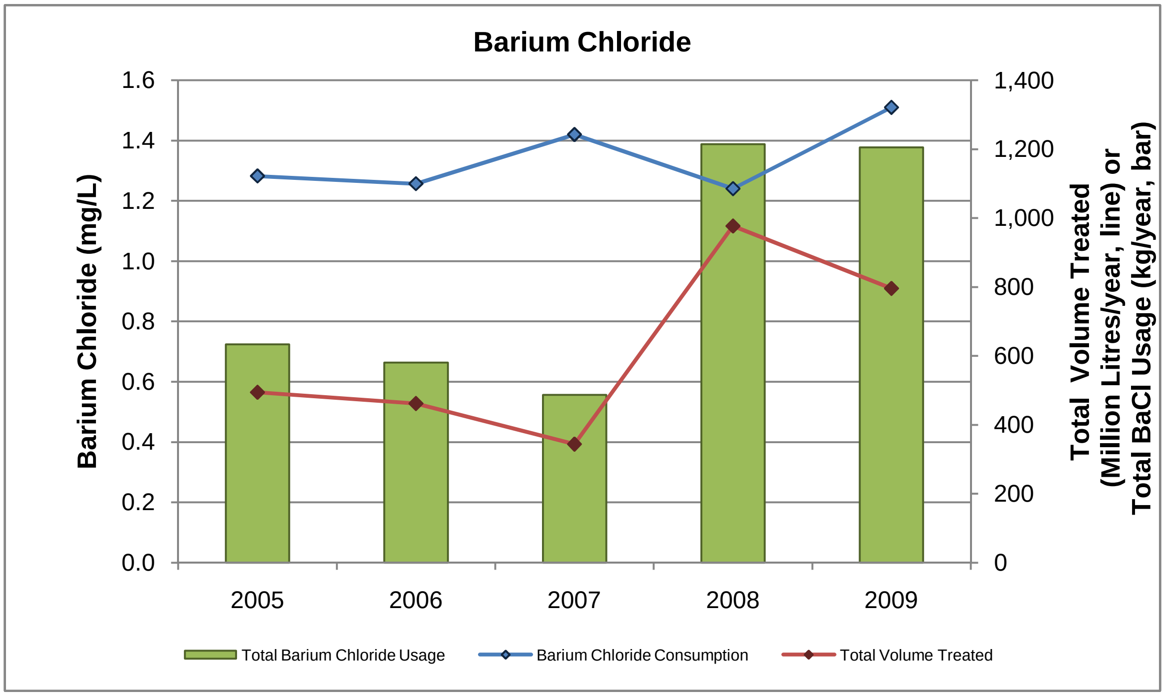
increasing trend, significant at $p<0.05$

[a] Trends were not assessed for acidity because a change in analytical technique in 2006 meant that the data were not comparable before and after that time.

[b] Based on rank correlation coefficients (rho) for common (combined) season trends, shown in table.

[c] "-" denotes that this parameter was not included in the trend analysis for that particular station due to insufficient data (e.g. there were <5 years worth of data for that parameter)

[d] This is the earliest year included in the trend analysis, but not all stations have data going back to 1991.



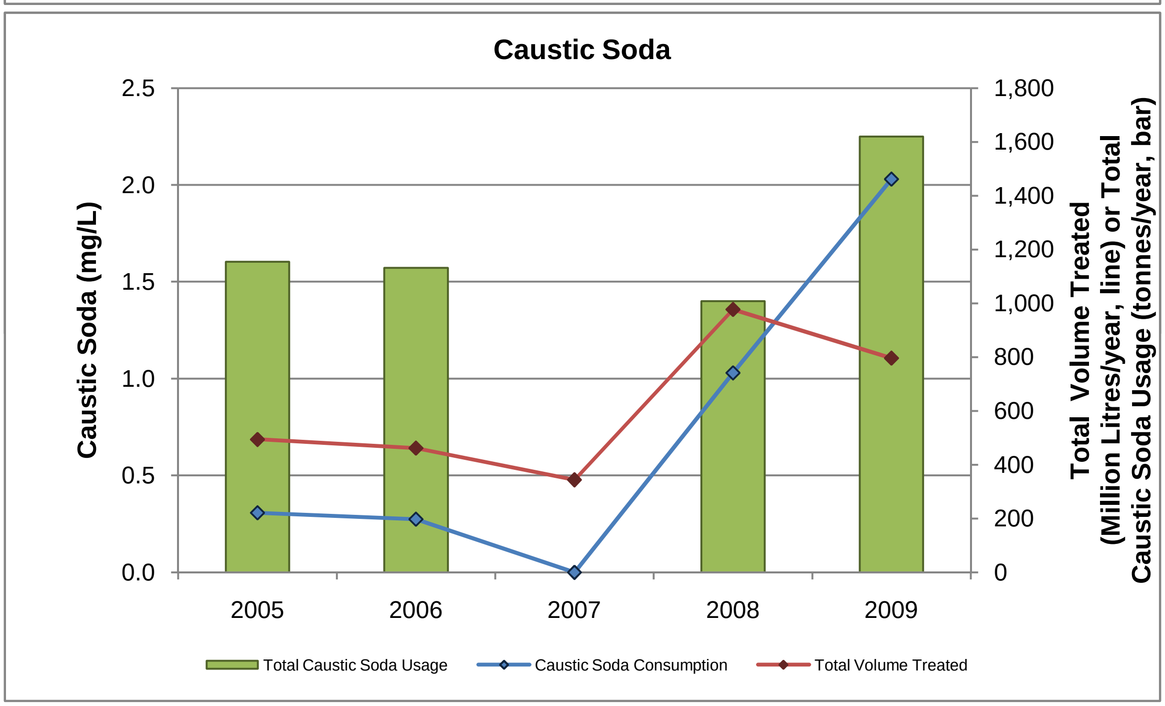


Figure 3.4: Comparison of total reagent consumed versus total volume treated at Denison TMA-1 from 2005-2009.

The historical spill and seepage from TMA-2 is treated with barium chloride to reduce radium-226 concentrations (currently no treatment for pH). Reagent use has been relatively stable over the past five years, likely associated with a stable vegetative cover, reductions in radium-226 concentrations in TMA-2 influent and that seepage flow rates are less influenced by precipitation (Figure 3.5).

Treated, effluent quality is monitored at the outlet of each ETP (TMA-1 is monitored at D-2 and TMA-2 is monitored at D-3) and over the past five years effluent quality has consistently achieved discharge criteria (Figures 3.6 and 3.7). While one radium-226 measurement at each location was greater than the monthly mean discharge criterion (Figures 3.6 and 3.7), the values were well below the individual grab sample criterion of 1.11 Bq/L (Appendix Table D.1.1).

Effluent has also been consistently non-lethal to *Daphnia magna* and rainbow trout, with no mortality reported in semi-annual acute toxicity tests (Table 3.4). Similarly, survival and reproduction of *Ceriodaphnia dubia* was not affected by exposure to 100% effluent in any tests conducted over the past five years (Table 3.4).

3.1.6 Summary

Water cover over tailings was consistently maintained at the Denison TMAs over the past five years. Since decommissioning, concentrations of radium-226, sulphate and uranium have decreased and are near the 50-year post decommissioning predictions (*i.e.* 2040). More recently sulphate and radium-226 concentrations have continued to decrease in TMA-2, but radium-226 has been increasing and pH decreasing in surface water at TMA-1. The trends at TMA-1 appear to be attributed to a step change in 2008, possibly related to decreases in sulphate over time and/or higher water levels in 2008 and 2009. However, additional data is required to verify the trend and, if required, determine the cause. Radium-226 and pH levels at D-1 achieve PWQO before treatment and are much lower than values observed immediately following closure. Groundwater down-gradient of the east end of TMA reflects improving conditions since decommissioning, based on decreasing iron concentrations and increasing pH. However, at the west end of TMA 1 and down-gradient of TMA-2, groundwater pH has been decreasing and iron increasing. Reagent use has increased in recent years reflecting increased radium-226 and decreased in pH in ETP influent, as well as the impact of higher flows in 2008 and 2009 necessitating a longer treatment period. Regardless, effluent quality has consistently achieved discharge criteria over the past five years and all tests to *Daphnia magna*, rainbow trout and *Ceriodaphnia dubia* were non-toxic.

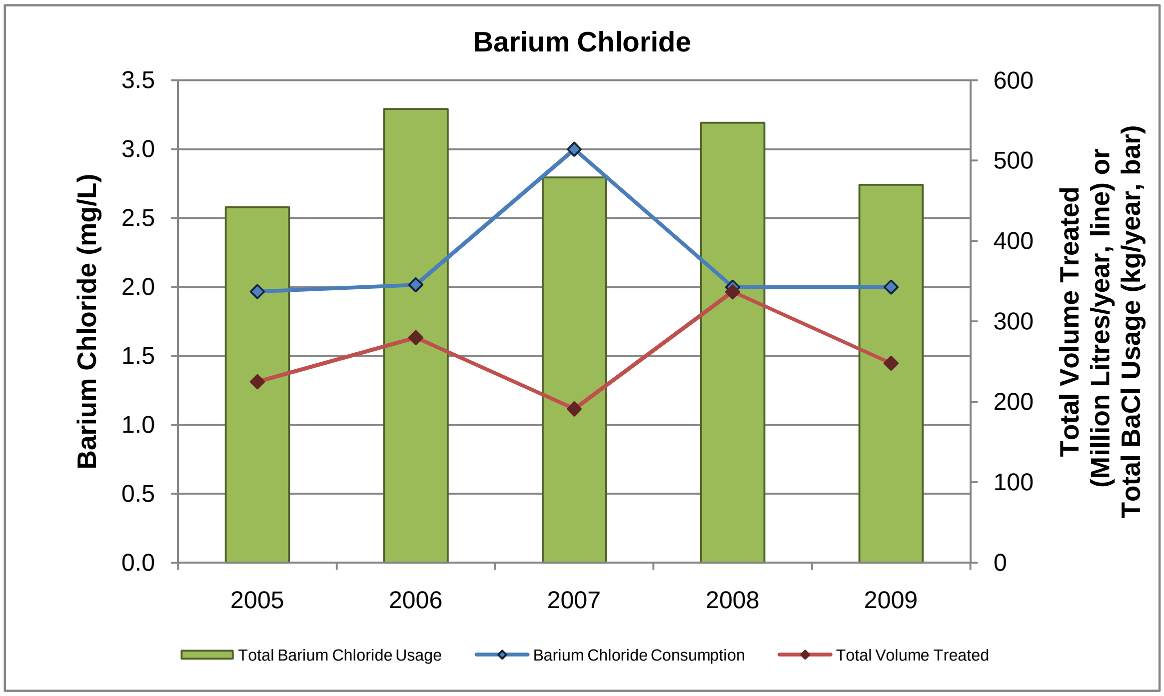


Figure 3.5: Comparison of total reagent consumed versus total volume treated at Denison TMA-2 from 2005-2009.

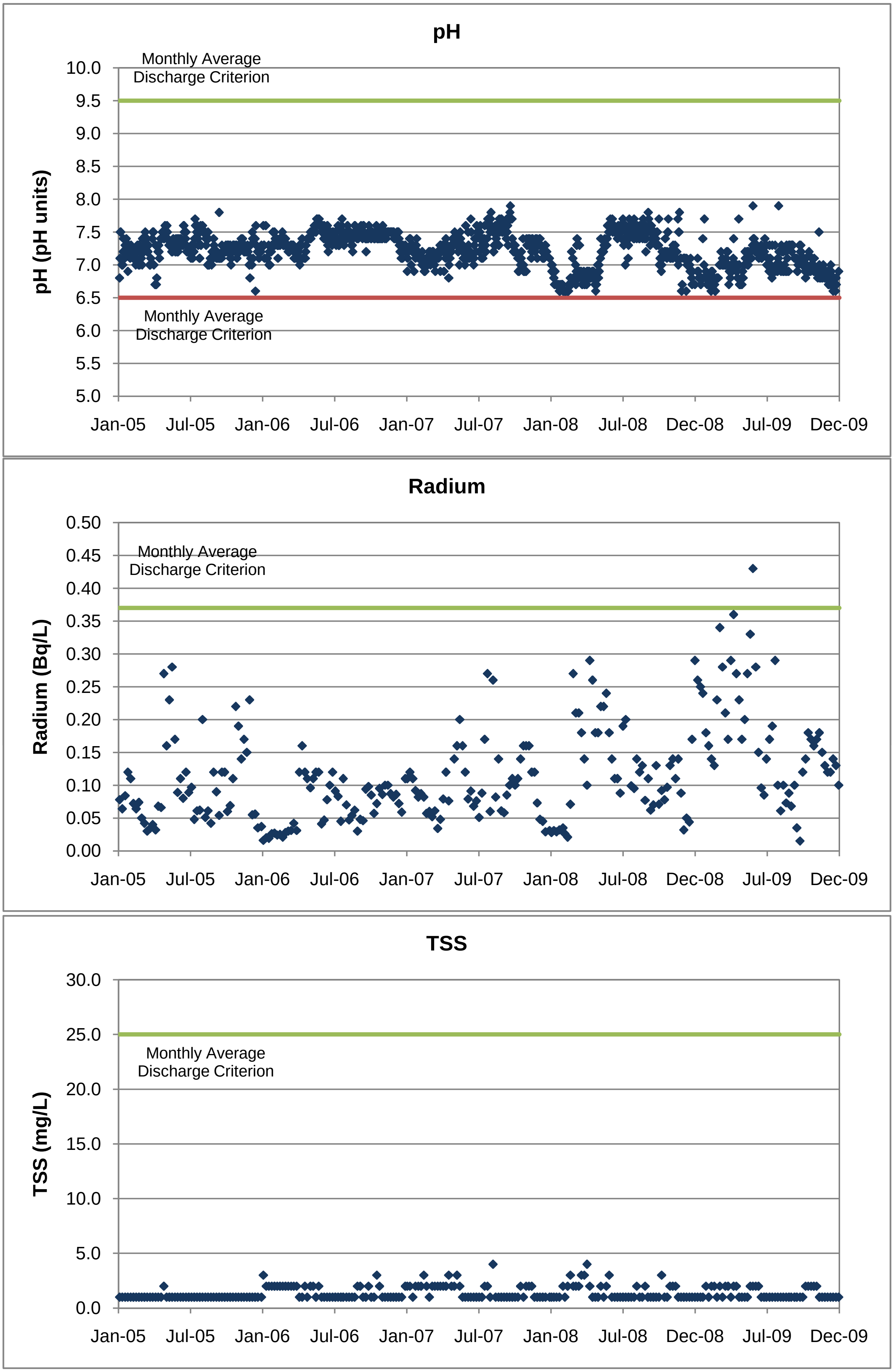


Figure 3.6: Effluent concentrations versus monthly average discharge criteria at Denison TMA station D-2.

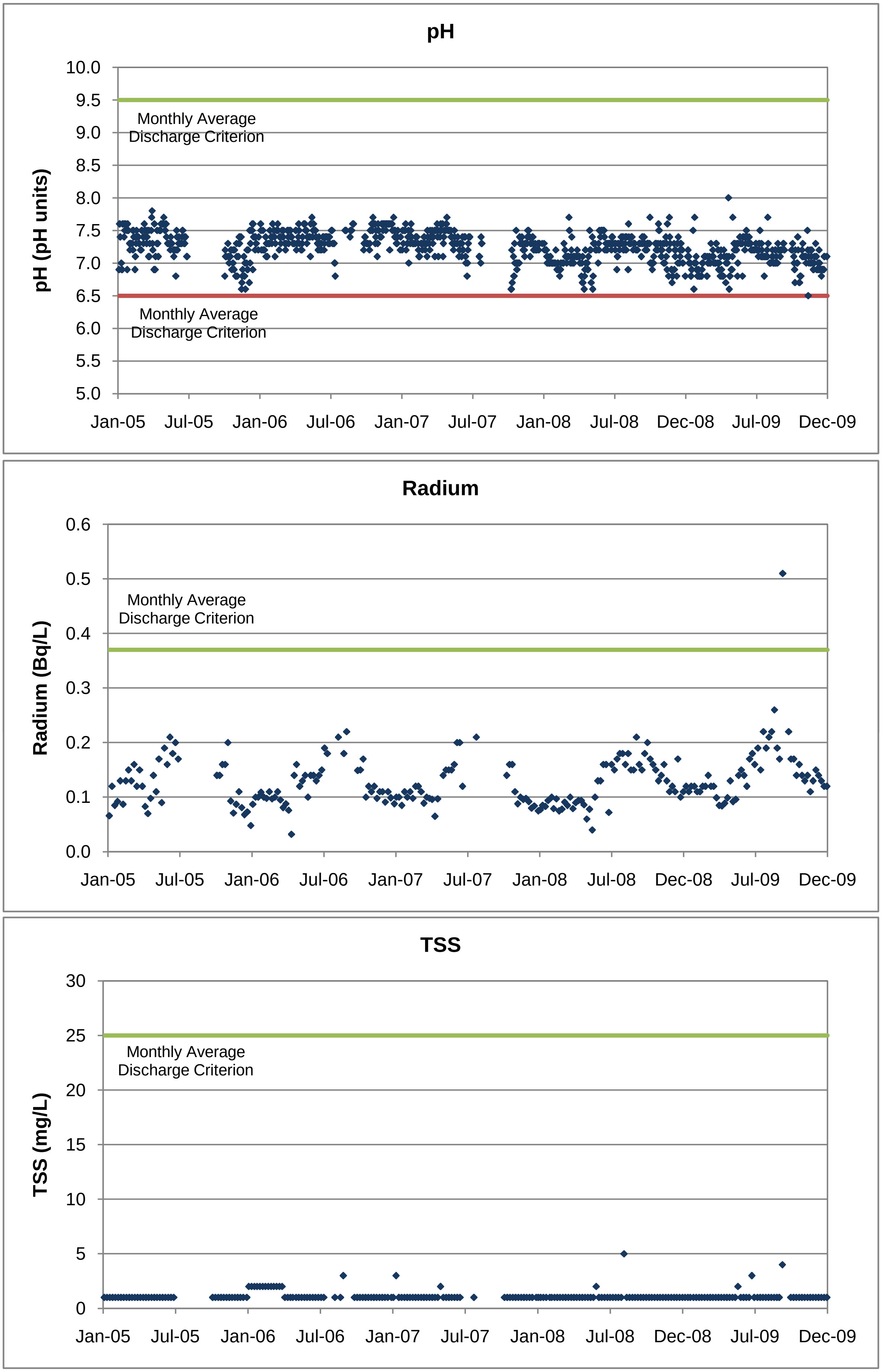


Figure 3.7: Effluent concentrations versus monthly average discharge criteria at Denison TMA station D-3.

Table 3.4: Toxicity test results for samples collected at Denison TMA station D-2, 2005 - 2009.

Sample Date (month-year)	Acute Toxicity (% mortality)		Survival and Reproduction (IC25[d] as % effluent)
	Daphnia magna [a]	rainbow trout [b]	*Ceriodaphnia dubia* [c]
May-05	0	0	100
November-05	0	0	100
May-06	0	0	100
December-06	0	0	100
June-07	0	0	100
October-07	0	0	100
June-08	0	0	100
October-08	0	0	100
May-09	0	0	100
October-09	0	0	100

[a] *Daphnia magna* 48-hr LC50 test (Environment Canada 2000a).

[b] Rainbow trout 96-hr LC50 test (Environment Canada 2000b).

[c] *Ceriodaphnia dubia* survival and reproduction test (Environment Canada 2007).

[d] Effluent concentration causing 25% inhibition relative to control organisms.

3.2 Spanish-American TMA

3.2.1 Basin History and Modifications

The Spanish-American mine and mill operated from 1958 to 1959. During that time the mine deposited approximately 0.45 million tonnes of tailings into the Spanish-American TMA.

In 1994, approximately 90,000 m^3 of exposed tailings beaches at the eastern end of Spanish-American TMA were relocated to the western end of the basin providing a nominal water cover depth of 0.9 m at the eastern perimeter and 1.5 m in the centre of the basin. Two engineered berms (North and South berms) were installed at the western outlet to flood the basin and confine the 10.92 ha Spanish-American TMA. Lime slurry was added to the basin during and after flooding (summers of 1994 to 1996) to achieve the target surface water pH of 7.0.

There is no ETP at the Spanish-American TMA. Drainage from the 37-hectare Spanish-American TMA watershed (owned by Rio Algom Limited), is monitored at station ECA-398 as it passes through the South Berm spillway to Denison TMA-1 (owned by Denison Mines Inc.; Figure 3.8). Station ECA-128 is monitored under the TOMP and the substances and frequency monitored are specific to the station type (Table 3.5; Figure 3.8). Data from ECA-128 are summarized in the following section and presented in Appendix C (Appendix Table C.2.1).

3.2.2 Basin Surface Water Quality

Surface water quality is monitored at the outlet of the Spanish American prior to its discharge to Denison TMA-1 (ECA-128). Effluent from the TMA is treated at Denison TMA-1 prior to discharge to the Serpent River Watershed. Routine monthly inspections of the Spanish American TMA indicate that the water cover in the TMA was consistently maintained and exposed tailing were not observed.

Over the past seven years (2003-2009), radium-226 has increased and pH and sulphate have decreased in the basin (Table 3.6). Increases in radium-226 concentration are likely associated with the decrease in sulphate concentrations within the basin and association of radium with residual iron hydroxides which are re-suspended and released during spring turn over as evidenced by elevated iron and radium in spring 2008 and 2009 samples (Appendix Table C.2.1). Work completed by EcoMetrix (Appendix G) indicates that as aqueous sulphate concentrations decline, there is an increased dissolution of barium sulphate to which radium is associated, whereby radium is released from the tailings. It is expected that radium concentrations in porewater will stabilize over time once the dissolution of barium

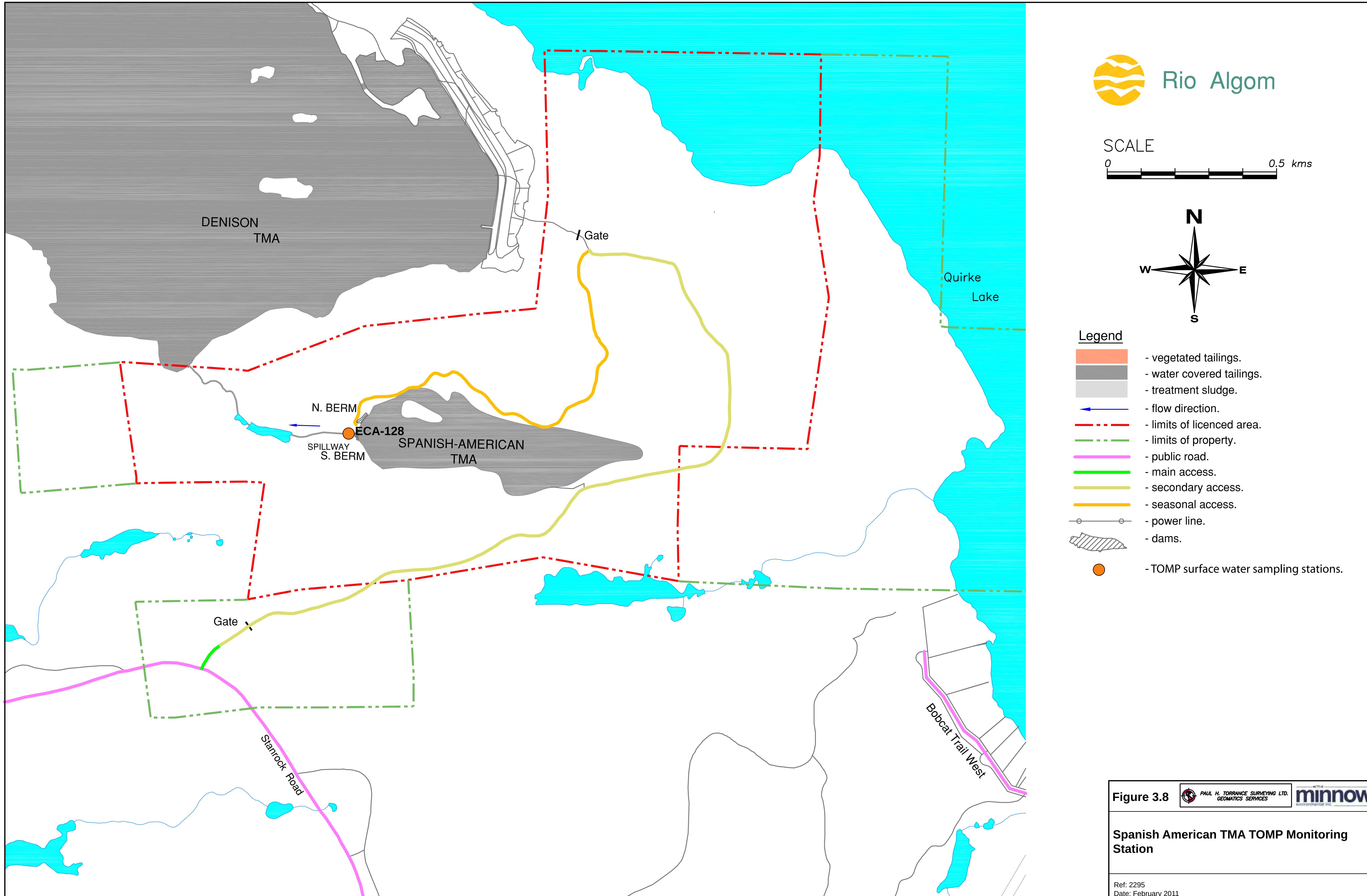


Figure 3.8

Spanish American TMA TOMP Monitoring Station

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Table 3.5: TOMP monitoring stations, substances, and frequencies[a] at Spanish American TMA.

			Parameters and Frequencies[a]										
TMA	TOMP Stations	Station Type/Purpose	Elevation	Flow	pH	Sulphate	Total radium-226	Lime or NaOH Consumption	Barium Chloride Consumption	TSS	Acidity	Iron	SAMP Metals[b]
Spanish American	ECA-128	Basin Performance	M[c]	Q	Q	Q	Q				Q		Q

[a] D - Work days, W - Weekly, M - Monthly, S - Semi-annually, A - Annually, Q-Quarterly

[b] SAMP metals are barium, cobalt, iron, manganese, and uranium

[c] During the snow-free period (April - November)

Table 3.6: Summary of water quality trends[a] at TOMP monitoring stations, Spanish American TMA, 2003 to 2009.

Station ID	Type/Location	Number of Seasons Used in Common Trend[c]	Acidity	Barium	Cobalt	Iron	Manganese	pH	Radium-226	Sulphate	Uranium
ECA-128	Sp. Am. TMA Effluent	3	-[b]	-	-	0.240	-	-0.478	0.578	-0.557	-0.204

decreasing trend, significant at $p<0.05$ (blue)

increasing trend, significant at $p<0.05$ (green)

[a] Based on rank correlation coefficients (rho) for common (combined) season trends, shown in table.

[b] "-" denotes that this parameter was not included in the trend analysis for that particular station due to insufficient data (e.g. there were <5 years worth of data for that parameter)

[c] Seasons used varied for substances based on suitability of data for trend analysis

sulphate re-equilibrates with aqueous sulphate concentrations. As there is no new source of radium to the TMA, radium concentrations in porewater and releases to surface water should decline as the amount of soluble material in the tailings diffusion zone decreases.

3.3 Quirke TMA

3.3.1 Basin History and Modifications

The Quirke TMA is located approximately 13 km north of the City of Elliot Lake and immediately north of Dunlop Lake. The Quirke mine and mill operated from 1956 to 1961, and again from 1968 to closure in 1990. Over this period, the Quirke mill produced approximately 42 million tonnes of tailings which along with four million tonnes of waste rock were deposited into the Quirke TMA.

The Quirke TMA was decommissioned as flooded tailings following mine closure in 1990 and covers a surface area of 192 ha. This TMA is composed of five terraced cells (Cells 14 to 18) within a bedrock-rimmed basin, separated by engineered, low-permeability dykes (Figure 3.9). Cell 14 at the west end of the basin was formed by raising Dyke 14 in 1991-1992 and provides a minimum 0.6 metre depth of water cover over the tailings. The downstream cells and dykes were constructed sequentially between 1994 and 1995. The last cell (Cell 18) is approximately 14 metres lower than Cell 14 creating a west to east cell-to-cell seepage gradient across the basin. Water is taken from Gravel Pit Lake to Cell 14 to replenish and maintain the water cover in Cell 14. In 1997 till blankets were applied to selected sections of the upstream sides of Dyke 14 and Dyke 15 to reduce the seepage flow. In the winter of 2003, a till blanket was extended across the entire length of Dyke 14 and a diffusion barrier was applied to 68% of Cell 14. The combined seepage from Cells 14 and 15 is approximately 45 L/sec with seepage from Cell 14 estimated at 35 L/sec (Golder 2011; Appendix H).

An *in-situ* lime addition program was initiated in 1995 whereby lime slurry is added to the cells on a seasonal basis to accelerate neutralization of historic acidity. Overflow from the Quirke TMA and its drainage basin is treated with lime (neutralization and metals removal) and barium chloride (radium removal) at the Quirke ETP prior to discharge into the Serpent River.

Within the TMA, surface water, porewater and ground water are monitored under the TOMP and the locations, substances and frequency monitored are specific to the station type (Table 3.7; Figure 3.9) Data from the TOMP stations are summarized in the following sections and presented in Appendix C (Appendix Tables C.3.2 – C.3.17).

Table 3.7: TOMP monitoring stations, substances, and frequencies[a] at Quirke TMA.

		Parameters and Frequencies[a]										
TOMP Stations	Station Type/Purpose	Elevation	Flow	pH	Sulphate	Total radium-226	Lime or NaOH Consumption	Barium Chloride Consumption	TSS	Acidity	Iron	SAMP Metals[b]
Q-05[d]	Basin performance (primary), ETP operations	W	D	M	Q	M	M	M		Q		Q
Q-03[d]	ETP operations			W								
Q-04P[d]	ETP operations			D								
Q-28[d]	Effluent		W[c]	W[c]	M[c]	W			W			M[c]
Q-29	Perimeter monitoring	W	W									
Cell 14, 15, 16S, 17	Basin performance (secondary)	M[e]		S	S	S				S	S	
90DK-14-5C; DK15-2(A-D); DK15-4(A-D); DK16-2(A-D); DK17-2(A-D)	Porewater			A	A					A	A	
QPW1-1,4,8; 95QW-3A,C,D; 95QW-4, 95QW-5A,D	Groundwater			A	A					A	A	

[a] D - Work days, W - Weekly, M - Monthly, S - Semi-annually, A - Annually, Q-Quarterly

[b] SAMP metals are barium, cobalt, iron, manganese, and uranium

[c] Monitoring requirement of SAMP

[d] Sampled when treatment plant is operating

[e] During the snow-free period (April - November)

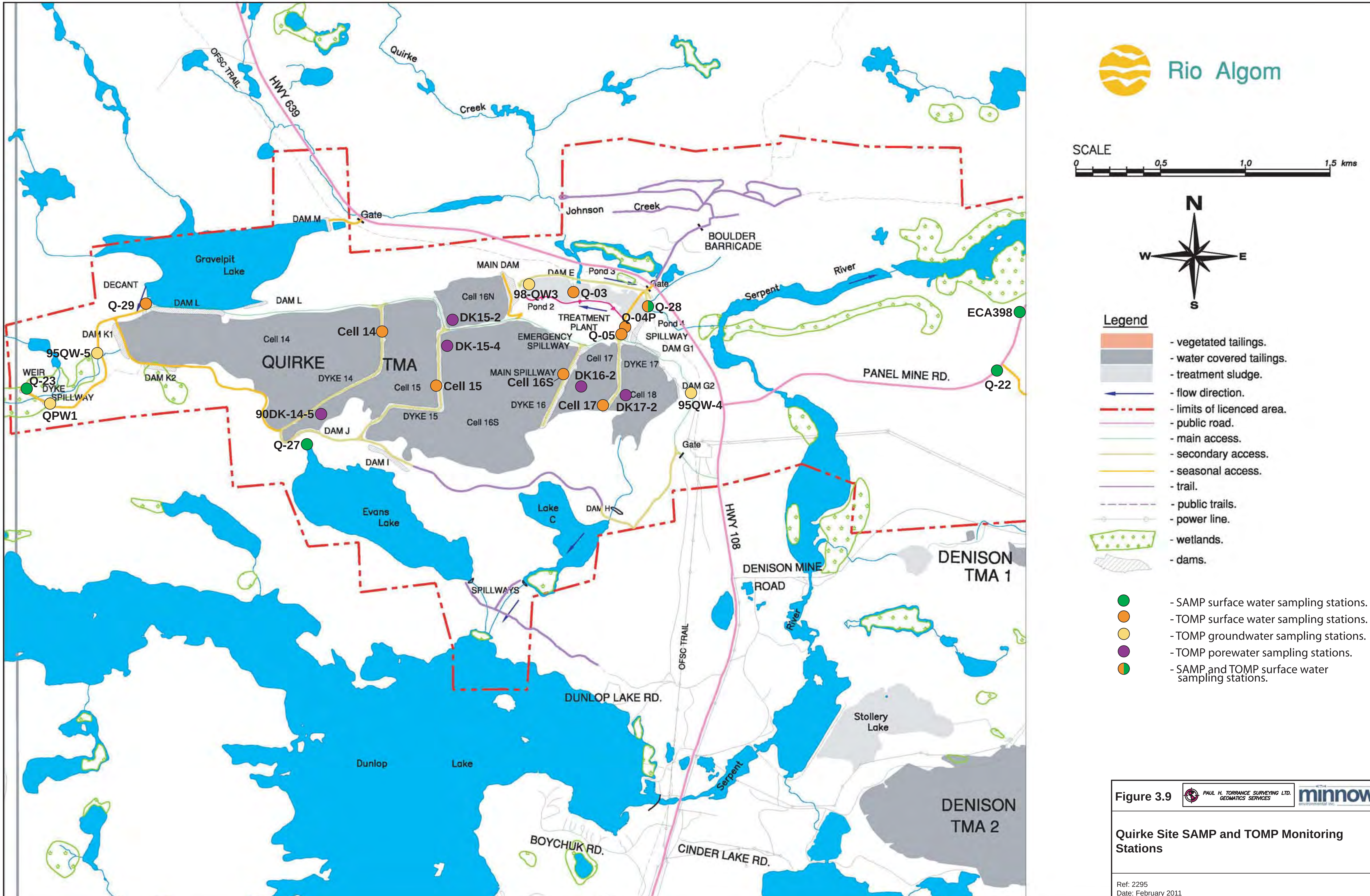

Figure 3.9

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minnow environmental inc.

Quirke Site SAMP and TOMP Monitoring Stations

Ref: 2295
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3.3.2 Water Management

Since the five cells of Quirke TMA are terraced, water elevations are lower in each progressive cell (Figure 3.10). Water from the first cell (Cell14) flows into the next cell until it reaches Cell 18 where it is treated prior to discharge to the Serpent River (Figure 3.10).

Application of the till blanket and diffusion barrier to Cell 14 in 2003 reduced seepage losses from Cell 14 from 50 L/s to 35 L/sec (Golder 2011; Appendix H) and following re-flooding of the cell in 2004, average water elevations within Cell 14 (2005 – 2009) have been maintained at 14 cm below the spillway overflow pipe (invert elevation of 377.77 masl) with a maximum depth below spillway elevation of 35 cm occurring in August of 2009. Water elevations in Cell 15 during the same time period have been maintained, on average, at 15 cm below the spillway overflow pipe (invert elevation of 373.74 masl) with a maximum depth below spillway elevation of 68 cm occurring in October of 2007. All other cells have remained at or above spillway invert elevation for the reporting period (Figure 3.10).

Since application of the till blanket and diffusion barrier to the cell margins, water elevation changes in Cell 14 do not result in exposure of tailings. However, prolonged periods of low precipitation can result in seasonal exposure of tailing in Cell 15. The lowest recorded water elevation (372.6 masl) occurred in April 2001 following a 1 in 50 year low precipitation event. EcoMetrix (2011b; Appendix H) was retained to assess potential acidity releases at elevations 0.5 m above and below this 1 in 50 year return event and determined:

- Annual acidity loadings from Cell 14 and Cell 15 are 1.14 and 0.88 tonnes of CaO per year and represent only 1% of the total annual Quirke lime consumption;
- Conservatively estimated acidity loads represent potential lime demands from 1 to 5 tonnes per year at Cell 15 water elevations of 373.0 masl and 372.0 masl respectively. These very conservative potential acidity loads are higher than acidity loads estimated from on-going monitoring data yet still represent only 1 to 3% of total annual lime consumption at the Quirke facility.

Water elevations in Cell 18 were consistently within the upper and lower operating limit for the TMA (Figure 3.10).

3.3.3 Basin Surface Water Quality

Basin surface water quality is monitored at five stations: the spillway of each cell (Cell 14, 15, 16S and 17) and at the ETP influent from Cell 18 (Q-05; Table 3.7; Figure 3.9).

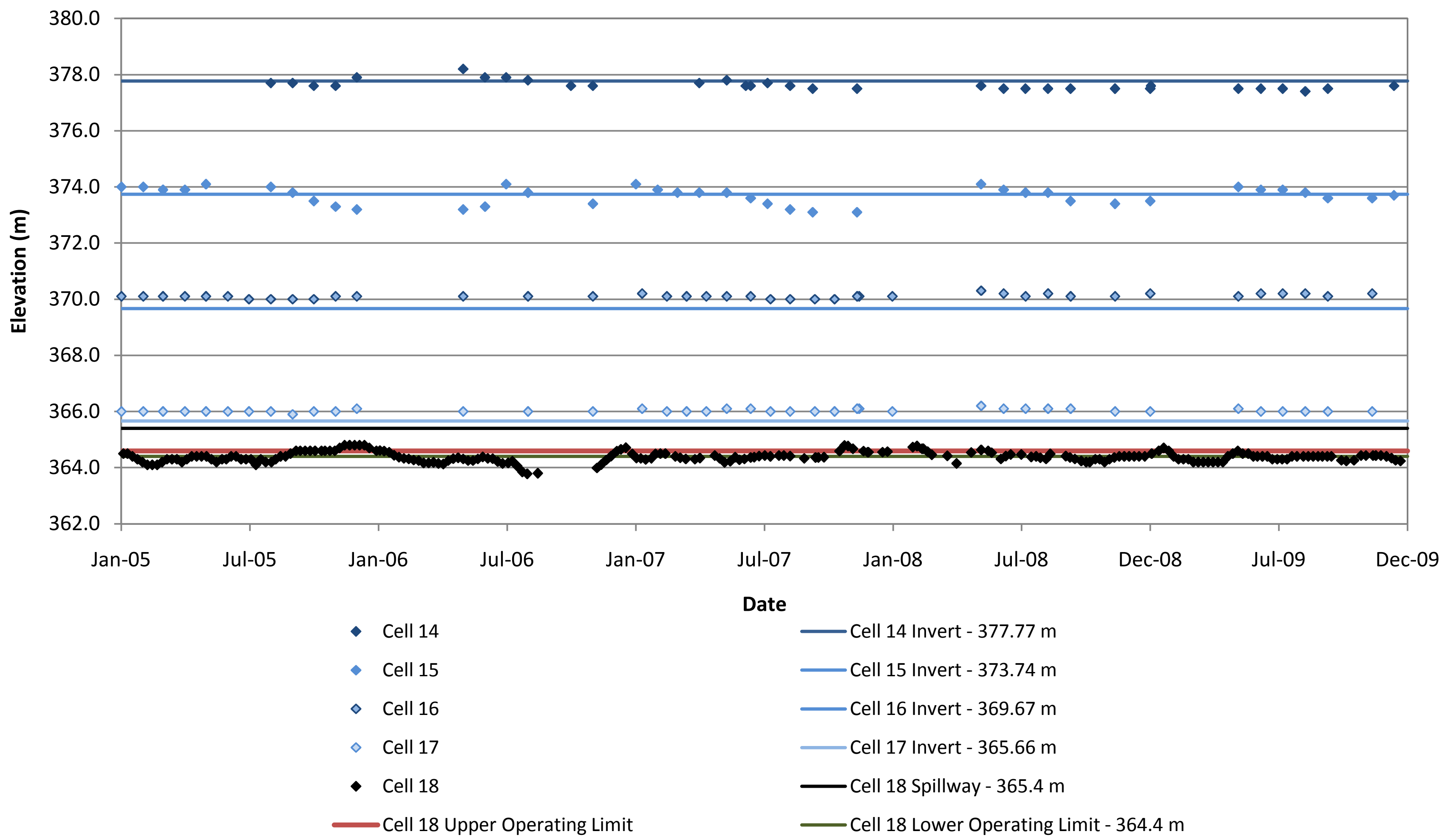


Figure 3.10: Water levels in cells of Quirke TMA.

Since decommissioning (1990 to 1996), treatment plant influent concentrations of sulphate and uranium have decreased, and pH has increased to near neutral levels (Figure 3.11). Concentrations of radium-226 increased slightly between 1992 and 2002 but have been relatively stable since then (Figure 3.11). Concentrations of radium-226, sulphate and uranium are approaching the 50 year post decommissioning predictions (*i.e.* 2040) (Figure 3.11).

More recently (2003-2009), surface water has continued to improve with significant reductions in acidity, sulphate and uranium and increased pH at Q-05 due to ongoing lime additions in Cell 16 and 17 (Table 3.8). Also acidity has decreased in Cell 14 in response to the diffusion barrier installed in 2003. During the installation, the water in the cell was drawn down and tailings were temporally exposed causing oxidation and acid production. After construction and cell re-flooding, the acid from oxidized tailings was flushed out and has slowly been reducing over time.

Although radium concentrations throughout the basin remain stable and within the EIS sensitivity analysis ranges (0.7 to 2.9 Bq/L), studies on radium release mechanisms suggest that the observed decreases in sulphate over time may result in increased radium concentrations within the basin. In order to develop an understanding of the mechanisms controlling radium-226 releases to basin surface water and to provide an upper bound radium-226 activity that may be observed in basin water, RAL retained EcoMetrix to investigate radium-226 activities in solids, porewater, and basin water in Cell 14. A complete description of the study findings is provided in Appendix G (EcoMetrix 2011c) and are summarized below:

- Barium concentrations and radium activities in porewater were well correlated, indicating that radium activities in porewater are controlled by similar mechanisms to the control of barium concentrations in porewater.
- The observed curvi-linear relationship between barium and sulphate porewater concentrations is consistent with the theoretical solubility of barium sulphate, whereby a decrease in sulphate porewater concentration will result in an increase in barium concentration in the porewater. Although the relationship was weaker for radium, it was evident that sulphate concentrations in porewater could also control for radium solubility in porewater.
- The concentration gradients between porewater and the overlying water column indicate an upward diffusion and mass transport of radium-226 from porewater to the

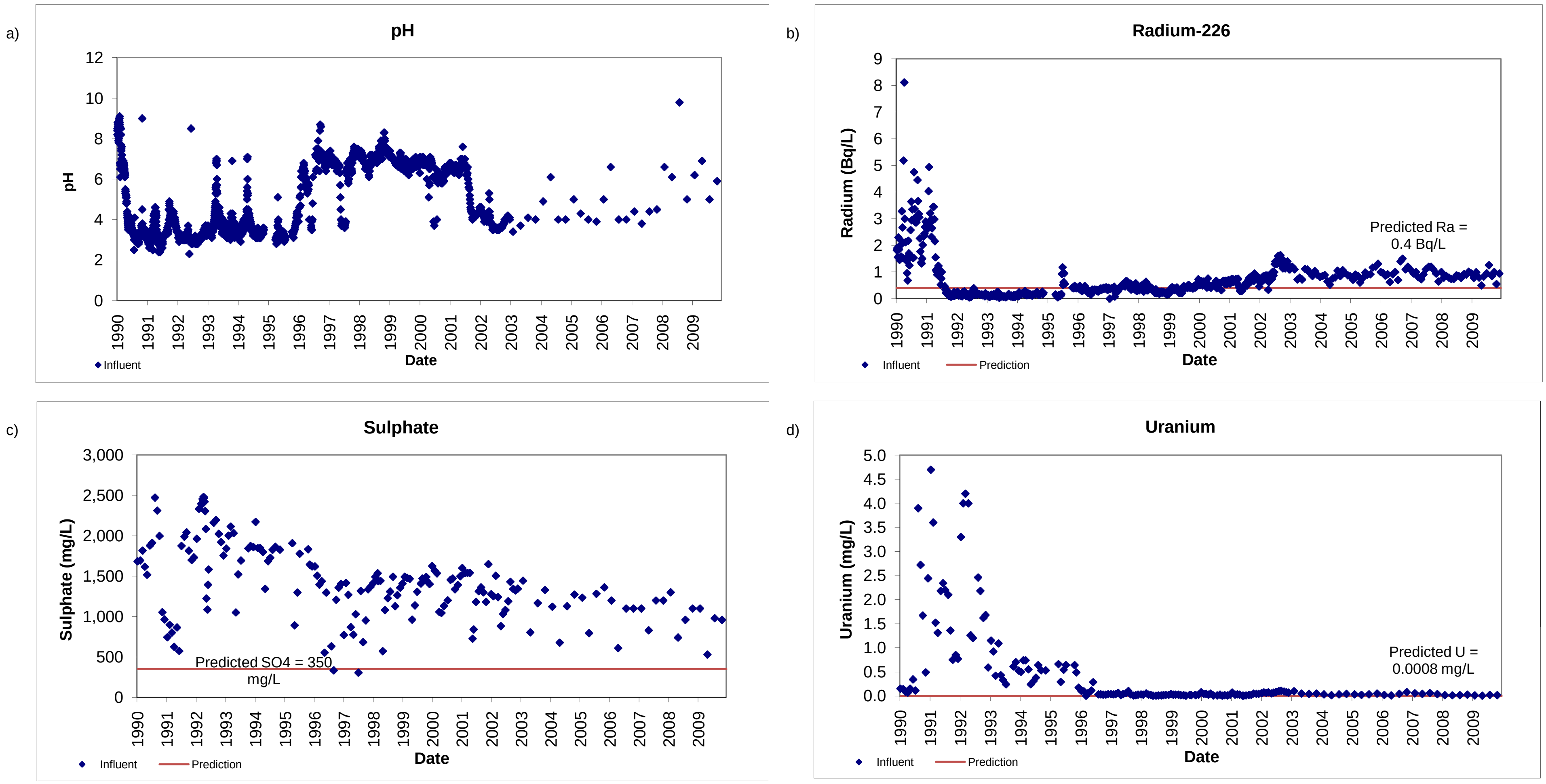


Figure 3.11: Water quality at the Quirke TMA ETP influent (Q-05) relative to predictions for 50 years (2040) post-decomissioning.

Table 3.8: Summary of water quality trends[a] for TOMP monitoring stations, Quirke TMA, 2003 to 2009.

Station ID	Type/Location	Number of Seasons Used in Common Trend[c]	Acidity	Barium	Cobalt	Iron	Manganese	pH	Radium-226	Sulphate	Uranium
Cell 14	Cell 14 at Spillway	1 to 2	-0.640	-[b]	-	-	-	0.200	0.036	-0.359	-
Cell 15	Cell 15 at Spillway	1 to 2	-	-	-	-	-	0.114	0.432	-0.500	-
Cell 16S	Cell 16S at Spillway	1 to 2	-	-	-	-	-	0.556	0.268	-0.872	-
Cell 17	Cell 17 at Spillway	1 to 2	-	-	-	-	-	0.438	0.089	-0.872	-
Q-05	Treatment Plant Influent	4 to 12	-0.710	-0.365	-0.366	-0.304	-0.219	0.718	-0.104	-0.574	-0.643

decreasing trend, significant at $p<0.05$
increasing trend, significant at $p<0.05$

[a] Based on rank correlation coefficients (rho) for common (combined) season trends, shown in table.

[b] "-" denotes that this parameter was not included in the trend analysis for that particular station due to insufficient data (e.g. there were <5 years worth of data for that parameter)

[c] Seasons used varied for substances based on suitability of data for trend analysis.

overlying water. This is further supported by the close agreement between observed radium-226 activities in the basin water compared to the calculated activities using the diffusive flux measures determined from Cell 14 core samples.

- Mass transport theory indicates that the concentrations in the basin cannot exceed those in the porewater, assuming no flow through Cell 14. Therefore, sediment data indicate that under the study conditions, an upper boundary for radium activities in the basin water is about 1.5 Bq/L.

3.3.4 Porewater

Porewater is monitored annually for acidity, pH, iron and sulphate in each of the five dykes within the Quirke TMA (Table 3.7 and Figure 3.9).

Porewater at the Quirke TMA represents surface water flushing through the dykes, and so it is not surprising that porewater demonstrated similar trends to basin surface water. Sulphate and iron concentrations decreased over time (1990 to 2009) while pH increased at almost all locations and depths (Table 3.9; Appendix Figures C.3.1-C.3.10).

In shallow (3-5 m) and mid depth (6-10m) porewater samples, pH achieves levels predicted in the EIS for 2040 (*i.e.*, 50 year post-closure). In deeper (11-15m) porewater, pH is approaching the predicted level (Figure 3.12).

3.3.5 Groundwater Quality

Four locations (wells) are sampled annually for acidity, pH, iron and sulphate. One well is located at the east end of the TMA(QW4), one is down-gradient of the main dam (95QW3 A,C,D) at the north end of the TMA, and the other two are located down-gradient of Dam K1 at the west end of the TMA (95QW5 (A,D) and QPW1(1,4, 8); Figure 3.9).

At the north end of the TMA, down-gradient of the Main Dam (95QW3) a significant increase in pH and decrease in sulphate indicated improved ground water quality over time (Table 3.9). Down-gradient of Dam G-2 at the east end of the TMA (95QW-4) pH levels have significantly decreased and sulphate has increased over time, although pH remains near neutral and sulphate has not increased since 2005 (Table 3.9; Appendix Figure C.3.9). Similarly, down-gradient of Dam K1 (QPW1) iron and sulphate have been increasing in deeper wells with concentrations possibly stabilizing since 2005 (Table 3.9). These trends likely reflect the slow flushing of contaminants in the west end of the basin since flooding in 1990.

Table 3.9: Summary of water quality trends[ab] in TOMP porewater and groundwater in Quirke TMA, 1990[c] to 2009.

Type	Location	Station	Depth (m)	Dates	Iron	pH	Sulphate
porewater	cell 15 below dyke 15	DK14-5C	5.91	1991-2009	-0.482	0.600	-0.090
	cell 16 below dyke 15	DK15-2D	4.13	1995-2009	-0.975	0.798	-0.593
		DK15-2C	5.5	1995-2009	-0.988	0.763	-0.705
		DK15-2B	7.25	1995-2009	-0.981	0.720	-0.744
		DK15-2A	10.24	1995-2009	-0.952	0.768	-0.778
	cell 16S below dyke 15	DK15-4D	4.01	1995-2009	-0.969	0.900	-0.912
		DK15-4C	5.61	1995-2009	-0.974	0.846	-0.872
		DK15-4B	7.08	1995-2009	-0.987	0.677	-0.960
		DK15-4A	10.3	1995-2009	-0.987	0.639	-0.948
	cell 17 below dyke 16	DK16-2D	4.01	1995-2009	-0.930	0.752	-0.608
		DK16-2C	5.6	1995-2009	-0.887	0.682	-0.535
		DK16-2B	7.1	1995-2009	-0.987	0.785	-0.462
		DK16-2A	10.21	1995-2009	-0.130	-0.084	0.049
	cell 17 below dyke 17	DK17-2D	3.91	1995-2009	-0.705	0.746	0.117
		DK17-2C	5.57	1995-2009	-0.225	0.267	-0.486
		DK17-2B	7	1995-2009	0.090	0.459	-0.097
		DK17-2A	12.17	1995-2009	0.512	0.841	0.527
groundwater	downgradient of main dam	95QW3D	4.6	1995-2009	0.248	0.838	-0.455
		95QW3C	9	1995-2009	-0.301	0.871	-0.815
		95QW3A	20.7	1995-2009	-0.512	-0.121	-0.679
	downgradient of dam G2 at east end of TMA	95QW4	10	1995-2009	-0.258	-0.629	0.605
	downgradient of dam K1	95QW5D	4.3	1995-2009	-0.216	-0.311	-0.039
		95QW5A	9.75	1995-2009	0.279	-0.061	0.267
	downgradient of dam K1, upgradient of dyke 23	QPW1-1	2.1	1991-2008	0.221	-0.608	0.046
		QPW1-4	11.4	1990-2009	0.632	-0.323	0.141
		QPW1-8	23.9	1990-2009	0.603	-0.361	0.917

decreasing trend, significant at $p<0.05$

increasing trend, significant at $p<0.05$

[a] Trends were not assessed for acidity because a change in analytical technique in 2006 meant that the data were not comparable before and after that time.

[b] Based on rank correlation coefficients (rho) for common (combined) season trends, shown in table.

[c] This is the earliest year included in the trend analysis, but not all stations have data going back to 1990.

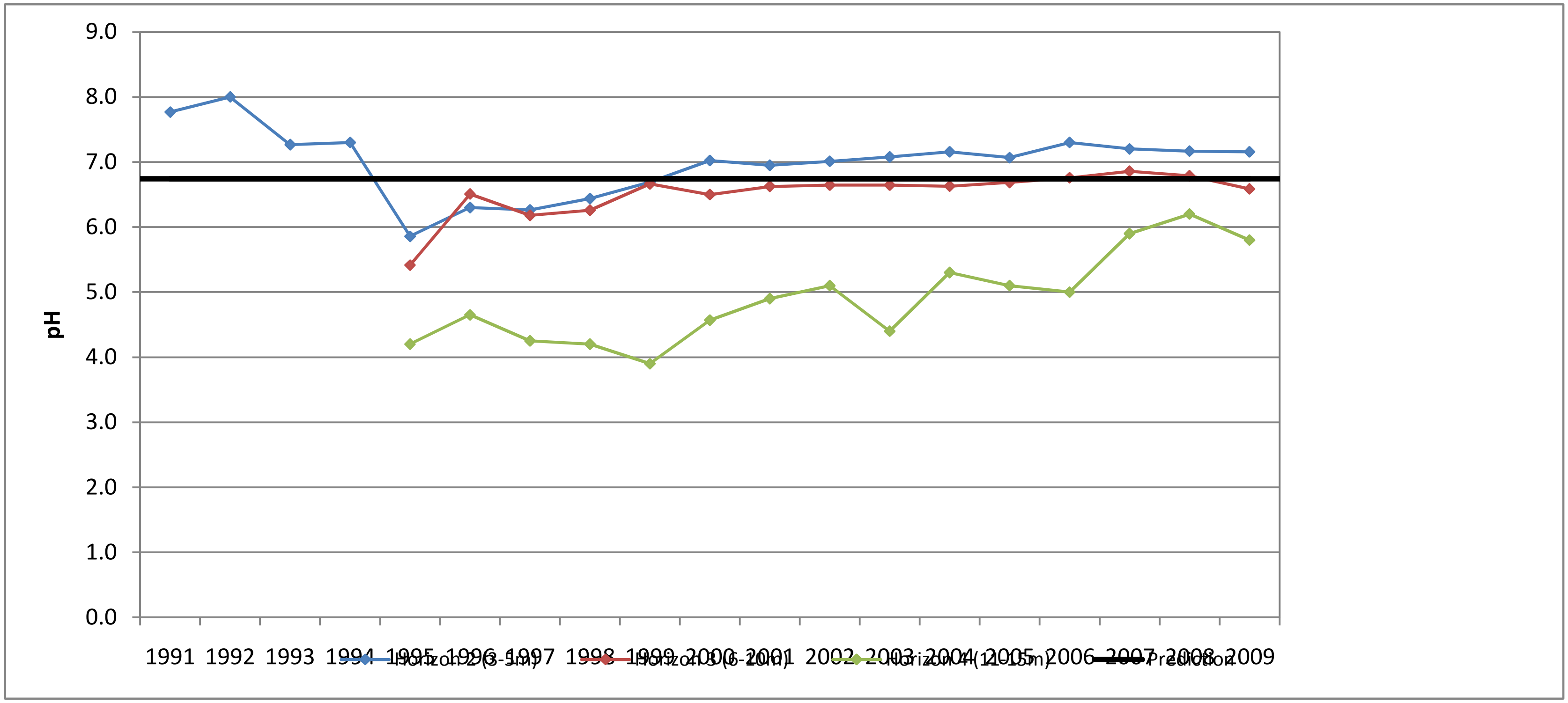


Figure 3.12: Comparison of mean porewater pH at various depths to EIS (2040) prediction, Quirke TMA, 1993-2009.

Horizon 2 - DK14-5C, DK15-2C, DK15-2D, DK15-4C, DK15-4D, DK16-2C, DK16-2D, DK17-2C, DK17-2D
Horizon 3 - DK15-2A, DK15-2B, DK15-4A, DK15-4B, DK16-2A, DK16-2B, DK17-2A, DK17-2B
Horizon 4 - DK17-2A

3.3.6 Treatment Performance

The Quirke TMA ETP is located at the spillway from Cell 18 (Figure 3.9). Treatment includes both lime and barium chloride to reduce acidity and radium-226 respectively. Combined annual lime consumption for both *in-situ* lime addition and treatment plant operations has remained relatively stable during the reporting period while the barium chloride consumption rate has declined from 1.2 to 0.6 mg/L (Figure 3.13).

Treated effluent quality is monitored at the outlet of the ETP settling pond (Q-28) and over the past five years has consistently achieved discharge criteria (Figure 3.14; Appendix Table C.3.1).

Effluent has also been consistently non-lethal to *Daphnia magna* and rainbow trout with no mortality reported in semi-annual acute toxicity tests (Table 3.10). Similarly, survival and reproduction of *Ceriodaphnia dubia* was not affected by exposure to 100% effluent in any of the tests conducted over the past five years (Table 3.10).

3.3.7 Summary

Tailings water cover in the Quirke TMA has been maintained, with water levels within operational range limits. In-basin surface water and porewater quality has been improving over time and generally achieves EIS predictions (*i.e.* the TMA is performing as anticipated). Groundwater down-gradient of the main dam has been improving over time, while the groundwater down-gradient of Dam K1 has shown decreasing pH and increasing concentrations of iron and sulphate. It is expected that these trends are representative of the initial flushing of historical porewaters from the TMA following flooding. In the past five years effluent quality consistently achieved discharge criteria and all tests to *Daphnia magna*, rainbow trout and *Ceriodaphnia dubia* were non-toxic. Overall, the Quirke TMA is performing well and conditions are improving over time.

3.4 Panel TMA

3.4.1 Basin History and Modifications

The Panel TMA is located 19 km northeast of the City of Elliot Lake, immediately north of Quirke Lake. The TMA is comprised of two bedrock-rimmed basins, the Main Basin and the South Basin, and contains a total of approximately 16 million tonnes of tailings and waste rock produced during two operating periods 1958 to 1961 and, following rehabilitation and upgrading, from 1979 to closure in 1991(Rio Algom 1995).

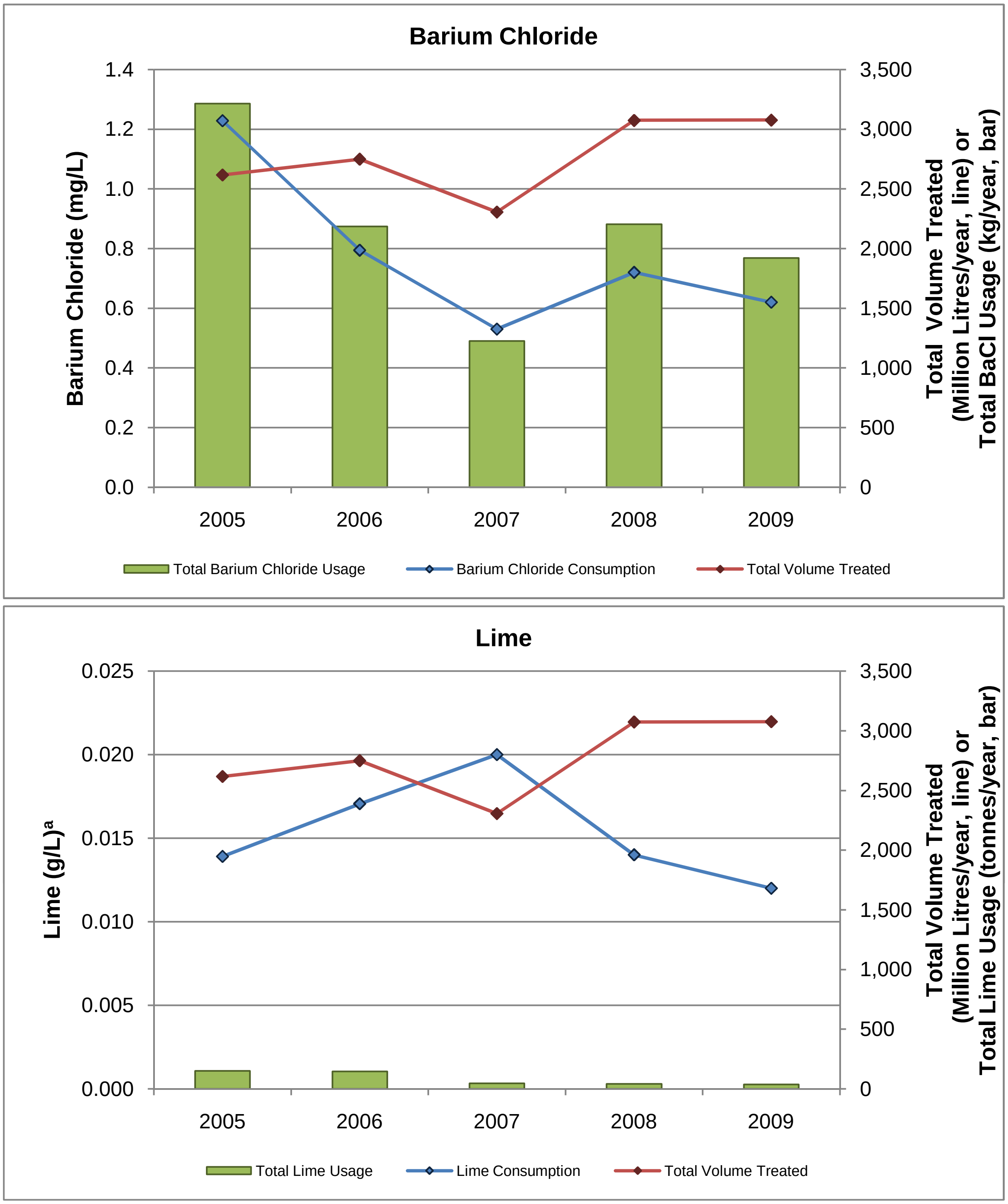


Figure 3.13: Comparison of total reagent consumed versus total volume treated at Quirke TMA from 2005-2009. [a] including in situ cell lime additions.

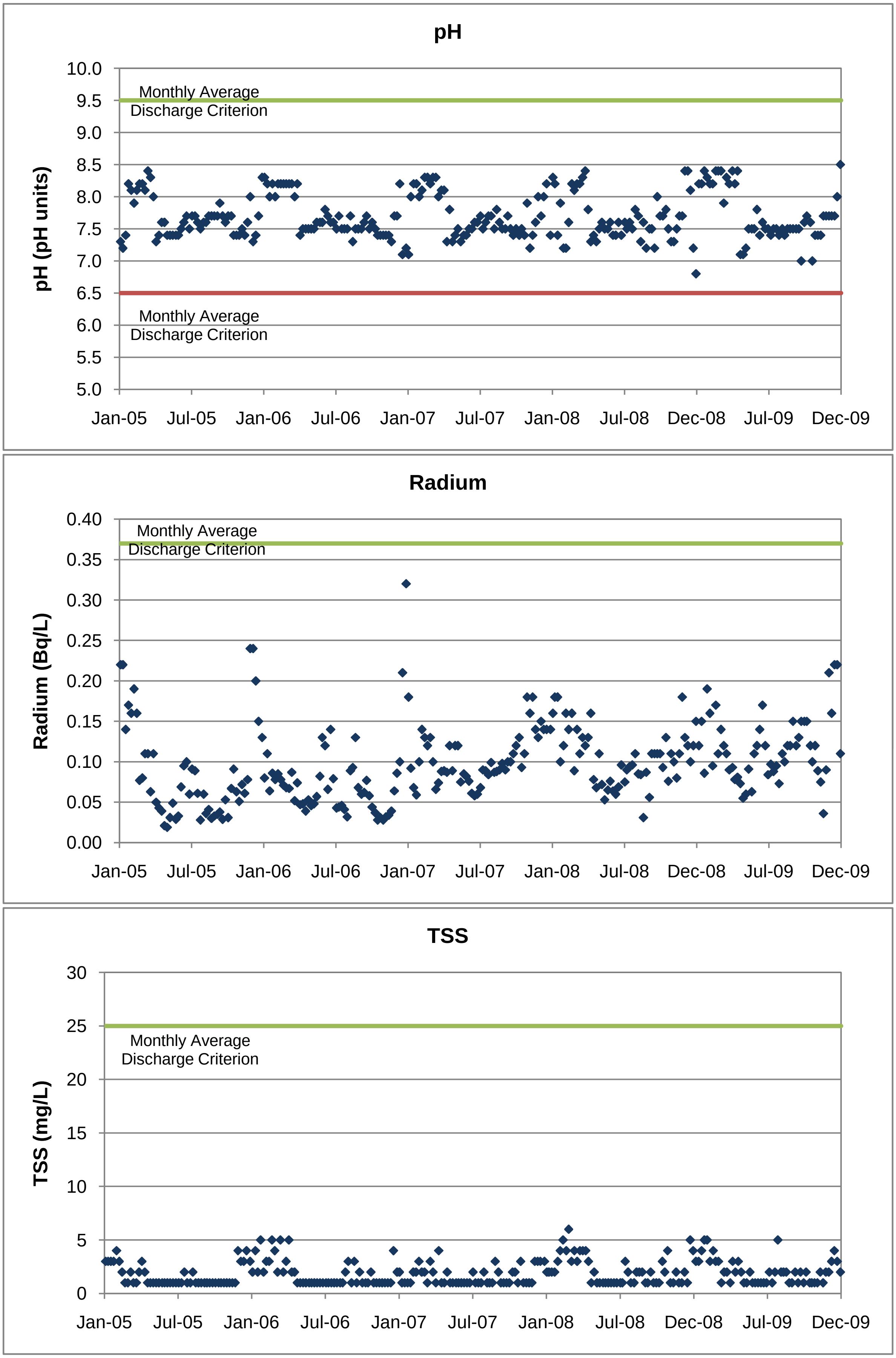


Figure 3.14: Effluent concentrations versus monthly average discharge criteria at Quirke TMA station Q-28.

Table 3.10: Toxicity test results for samples collected at Quirke TMA station Q-28, 2005 - 2009.

Sample Date (month-year)	Acute Toxicity (% mortality)		Survival and Reproduction (IC25[d] as % effluent)
	Daphnia magna [a]	rainbow trout [b]	*Ceriodaphnia dubia* [c]
May-05	0	20	100
November-05	0	0	100
May-06	0	0	100
November-06	0	0	100
May-07	3	0	100
November-07	0	0	100
May-08	0	0	100
November-08	0	0	100
May-09	0	0	100
November-09	0	0	100

[a] *Daphnia magna* 48-hr LC50 test (Environment Canada 2000a).

[b] Rainbow trout 96-hr LC50 test (Environment Canada 2000b).

[c] *Ceriodaphnia dubia* survival and reproduction test (Environment Canada 2007).

[d] Effluent concentration causing 25% inhibition relative to control organisms.

The Main Basin is contained by four engineered low-permeability dams (Dams B, D, E, and H) and has a total area of approximately 84 hectares (Figure 3.15). The South Basin, which contains a small quantity of tailings deposited in the late 1950s, is retained by two engineered low-permeability dams (Dams A and F) that have maintained the 39-ha basin in a flooded state since 1978 (Rio Algom 2000; Figure 3.15). Dam K and Berms W1, W2 and W3 were constructed in 1978 to divert run-off from the sub-watershed north of the Main Basin, east to Rochester Creek through Channel Y (Figure 3.15). Additional surface run-off is diverted away from the west side of the Main Basin to Panel Creek and Quirke Lake via Channel Z, which was also constructed in 1978 (Figure 3.15). This resulted in a drainage area of 177 hectares for the Main Basin. The South Basin, which receives inflow from the Main Basin, also receives surface water drainage from its own 119-ha watershed area.

Neutralization of tailing in the mill was practiced during all operational phases of the mine. Starting in 1974 and until construction of the new plant in 1981, lime and barium chloride were mixed in a small treatment plant adjacent to the mill and pumped to the basins via a two-inch line during the frost-free season. Treatment solids settled in what is now the South Basin and treated effluent was discharged to Rochester Creek via Dam A. As part of the 1978 facility upgrading, the current treatment plant and settling ponds were constructed in the vicinity of Dam F and treated effluent was directed towards Quirke Lake.

The Panel TMA was decommissioned through flooding, with the Main Basin draining into the South Basin via a spillway. The overflow from the South Basin enters the ETP where it is treated with a mixture of lime slurry and barium chloride to neutralize acidity and remove radium. The water level in the Main Basin reached its target elevation in 1994, after which lime slurry was added *in situ* on a seasonal basis (until 1999) to increase the pH in both basins on a seasonal basis. Rehabilitation of the Panel TMA was completed in 1999 with the construction of an overflow spillway at the west abutment of Dam F in the South Basin and the construction of an engineered earthfill dam at the outlet of Pond C to Rochester Creek (Pond C berm; Figure 3.15). Pond C contains a small volume of fine tailings and treatment solids and receives seepage from Dam A and run-off from its 65-ha drainage area.

Within the TMA, surface water and groundwater are monitored under the TOMP and the locations, substances and frequency monitored are specific to the station type (Table 3.11 and Figure 3.15). Data from the TOMP stations are summarized in the following sections and presented in Appendix C (Appendix Tables C.4.2-C.4.8).

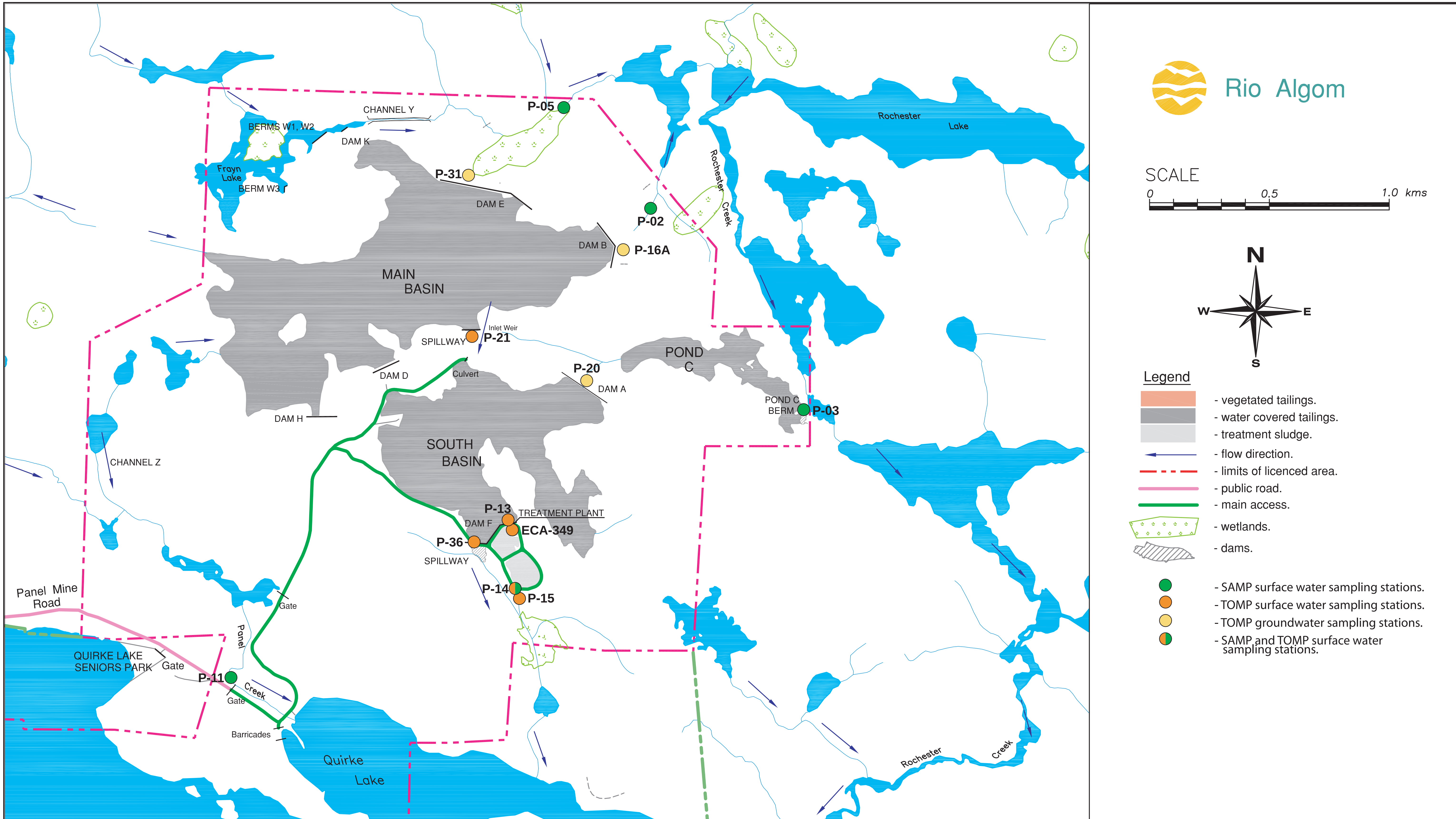


Figure 3.15

Panel Site SAMP and TOMP Monitoring Stations

Ref: 2295
Date: February 2011

Table 3.11: TOMP monitoring stations, substances, and frequencies[a] at Panel TMA.

		Parameters and Frequencies[a]											
TOMP Stations	Station Type/Purpose	Elevation	Flow	pH	Conductivity	Sulphate	Total radium-226	Lime or NaOH Consumption	Barium Chloride Consumption	TSS	Acidity	Iron	SAMP Metals[b]
P-13[f]	Basin performance (primary), ETP operations	W	D	M		Q	M	M	M		Q		Q
ECA-349[f]	ETP operations			D									
P-14[f], P-36[f]	Effluent		-[d]	W[c]		M[c]	W			W			M[c]
P-15	Perimeter				M								
P-21	Basin performance (secondary)	M[e]		S		S	S				S	S	
P-16A, P-20, P-31	Groundwater			A		A					A	A	

[a] D - Work days, W - Weekly, M - Monthly, S - Semi-annually, A - Annually, Q-Quarterly

[b] SAMP metals are barium, cobalt, iron, manganese, and uranium

[c] Monitoring requirement of SAMP

[d] No flow monitoring at P-14 because <1% additional flow between P-13 and P-14

[e] During the snow-free period (April - November)

[f] Sampled when treatment plant is operating

3.4.2 Water Management

Water levels are monitored in both the Main and South basins of the Panel TMA. The Main Basin water elevation is generally maintained above the spillway invert (393.2 m), although a bedrock outcrop down-gradient of the spillway tends to retain water in the spillway to an elevation above 393.4m (Figure 3.16). In the South Basin, an operating practice is used to maintain a consistent water elevation while minimizing treatment plant start and stop cycles. Generally water is drawn down in the fall to maximize winter storage capacity and avoid winter operation of the ETP (e.g., period when ETP is least efficient). At the time the last State of the Environment Report (Minnow 2009a) was prepared, Rio Algom established winter and summer operating elevations for the South Basin to minimize fluctuations in water elevations. In the fall/winter, a draw down elevation of 379.6 m is used with a restart target of 380.15 m (0.55 m fluctuation in water level) whereas in the summer the draw down elevation is 380.0 m with a restart target of 380.34 (0.34 m fluctuation). Since 2008, water levels in the South Basin have been more stable (Figure 3.16). Over the past five years, water levels in the Main Basin were maintained high enough to ensure consistent water cover of tailings (Figure 3.16).

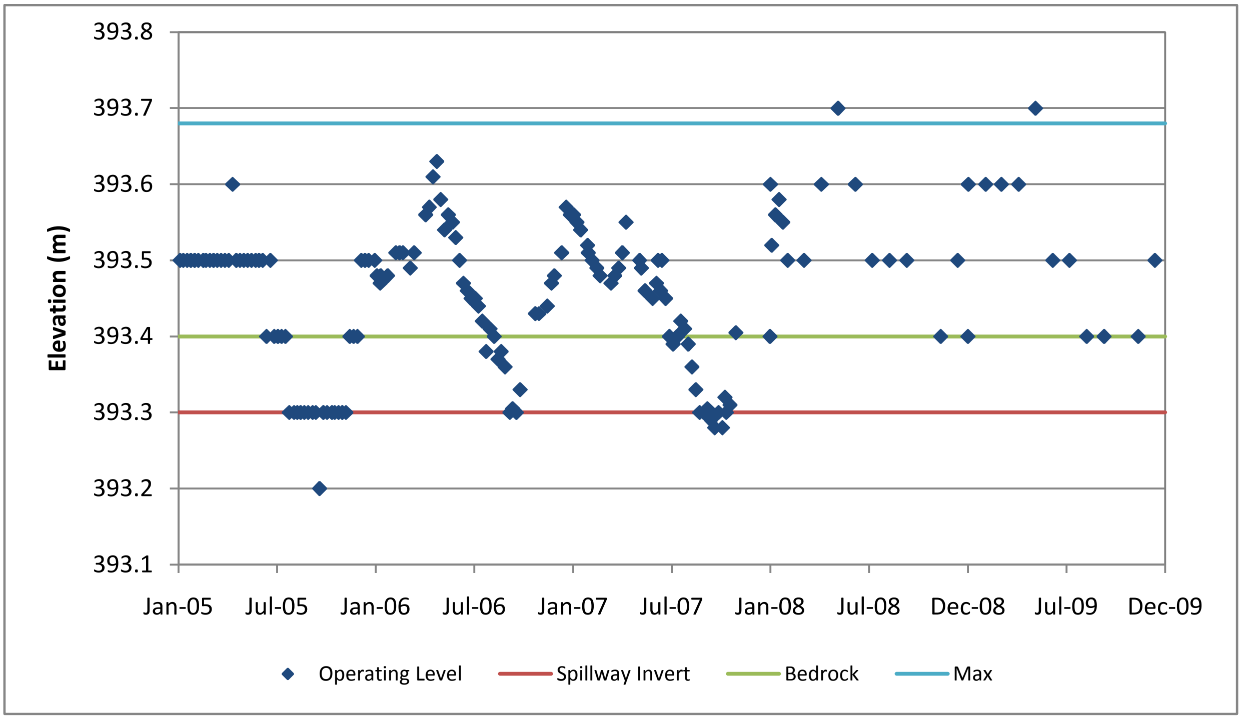
3.4.3 Basin Surface Water Quality

Surface water quality is monitored at five stations: the spillway of the Main Basin (P-21), the ETP influent (P-13) and effluent (P-14), the ETP pH probe (ECA-349) and the ETP settling pond underflow drainage (P-15; Table 3.11; Figure 3.15).

Since decommissioning (1990 to 1999) radium-226, sulphate and uranium concentrations have decreased and pH has increased to near neutral (Figure 3.17) such that concentrations are approaching the 50 year post decommissioning predictions (*i.e.* 2040) (Figure 3.17).

More recently (2003-2009) surface water has continued to improve with significant reductions in the concentrations of acidity, radium-226, sulphate and uranium and increased pH at the ETP influent (P-13; Table 3.12; Appendix Figure C.4.1) At the ETP influent, pH meets the discharge criterion and radium-226 concentrations are approaching the criterion (Appendix Figure C.4.1). At the outlet of the Main Basin both pH and radium-226 achieve discharge criteria prior to treatment (Appendix Table C.4.7).

While radium-226 concentrations were found to be decreasing over the past five years and remain within the range specified in EIS sensitivity analysis (0.4 to 1.4 Bq/L), sulphate has also been decreasing and studies on radium release mechanisms suggest that decreases in sulphate over time may result in radium release from the tailing to the overlying water column a)



b)

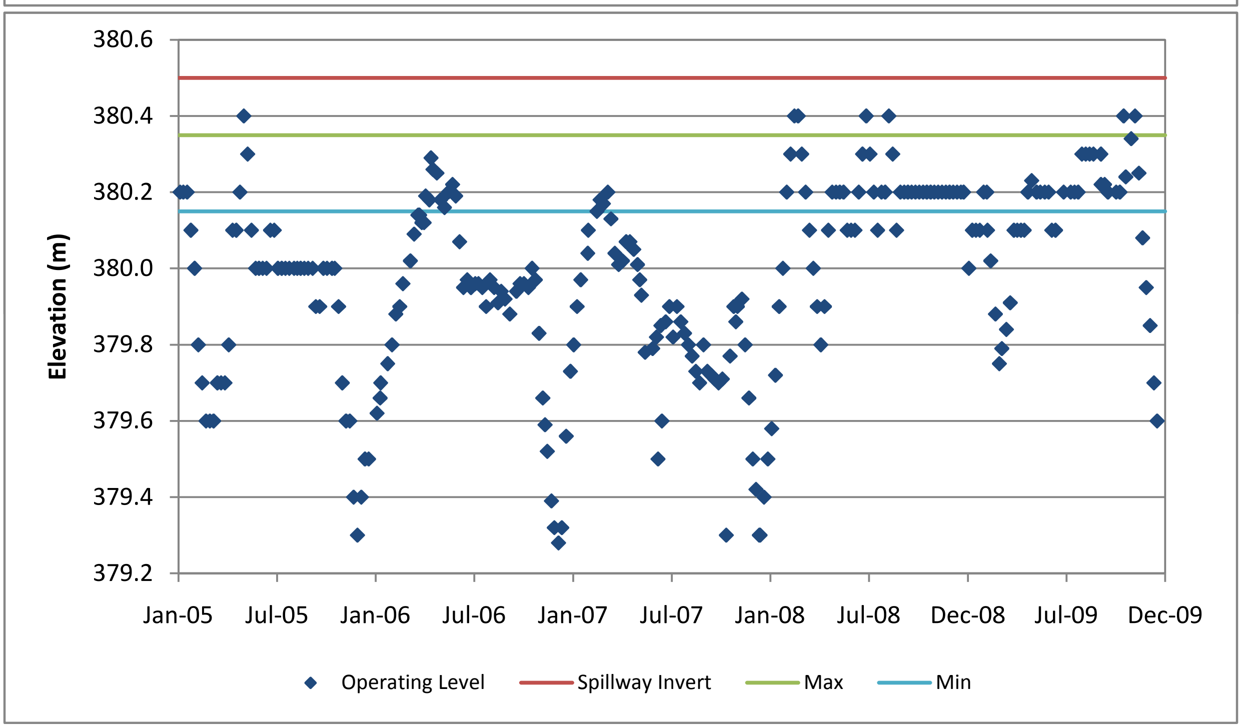


Figure 3.16: Water level at Panel main basin (a) and south basin (b) relative to minimum operating elevation.

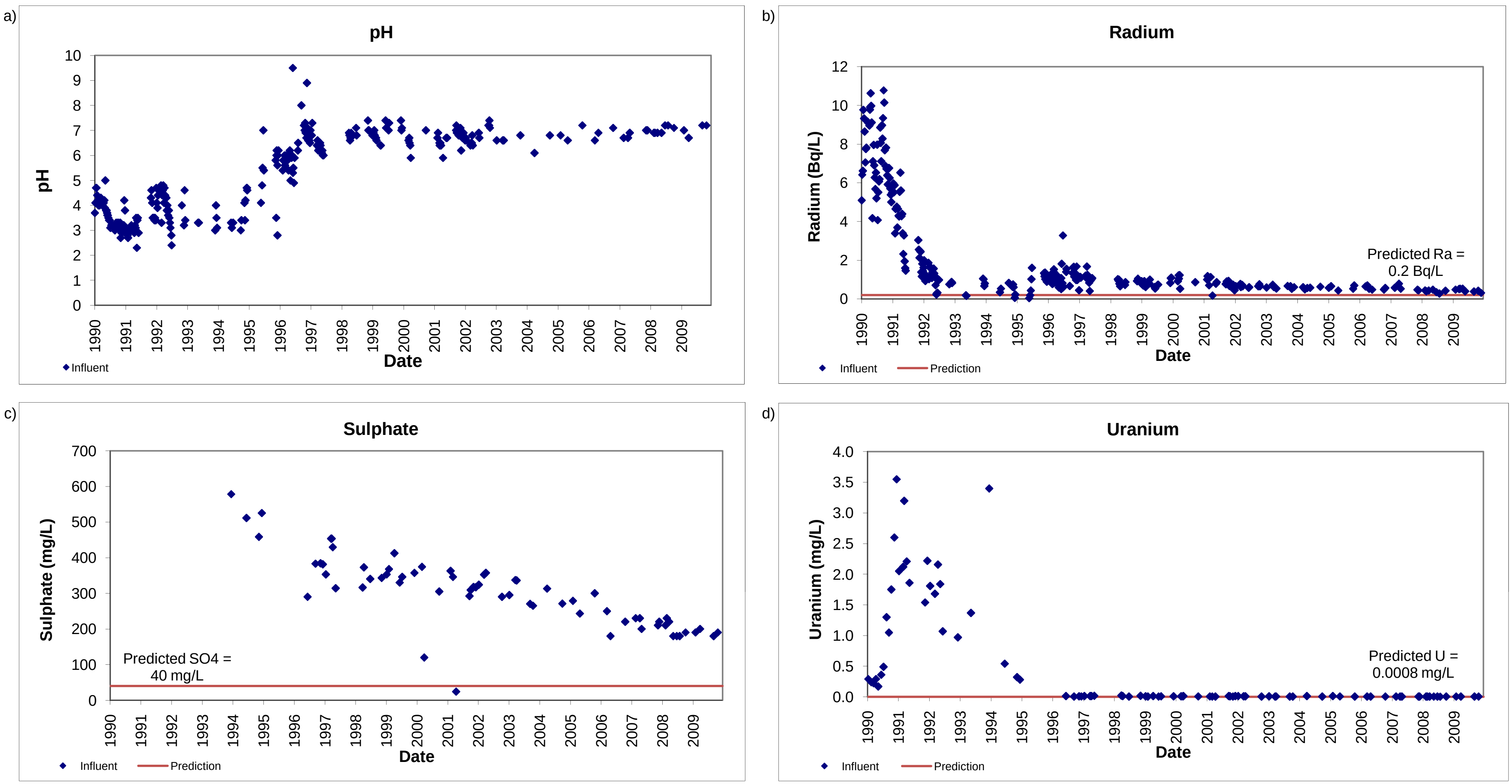


Figure 3.17: Water quality at the Panel TMA ETP influent (P-13) relative to predictions for 50 year (2040) post-decomissioning.

Table 3.12: Summary of water quality trends[a] for TOMP monitoring stations, Panel TMA, 2003 to 2009.

Station ID	Type/Location	Number of Seasons Used in Common Trend[c]	Acidity	Barium	Cobalt	Iron	Manganese	pH	Radium-226	Sulphate	Uranium
P-21	Main Basin Outflow	2	ND[b]	-[d]	-	-	-	-0.144	-0.171	-	-
P-13	ETP Influent	2 to 4	-0.870	0.510	-0.282	-0.331	0.276	0.741	-0.689	-0.904	-0.850

decreasing trend, significant at $p<0.05$

increasing trend, significant at $p<0.05$

[a] Based on rank correlation coefficients (rho) for common (combined) season trends, shown in table.

[b] ND denotes that this parameter was not included in the trend analysis for that particular station due to >50% non-detectable concentrations in the samples available for the ana

[c] Seasons used varied for substances based on suitability of data for trend analysis.

[d] "-" denotes that this parameter was not included in the trend analysis for that particular station due to insufficient data (e.g. there were <5 years worth of data for that paramete

of the basin. In order to develop an understanding of the mechanisms controlling radium-226 releases to basin surface water, RAL retained EcoMetrix to investigate radium-226 activities in solids (submerged tailings and treatment solids), porewater, and basin water in the Panel TMA. A complete description of the study findings is provided in Appendix G (EcoMetrix 2011d) and summarized below:

- Barium concentrations and radium activities in porewater were correlated, suggesting that radium-226 may behave similarly to barium, although another secondary mechanism may also influence radium release from solids. A strong correlation between calcium and sulphate in the sediment indicates that gypsum is present. Therefore, the solubility of sulphate (present as gypsum) likely controls the release of radium (associated with barite) into porewater.

- A correlation between barium and sulphate suggests that barium (and therefore radium) release to porewater is controlled by the solubility of barite and the sulphate concentrations in the porewater. Correlations between radium and sulphate in porewater were relatively weak, however, when all data is combined (both Panel and Quirke cores from the EcoMetrix studies) a stronger relationship is evident, where radium begins to release from solids when sulphate in porewater decreases to below 250 mg/L.

- Sulphate concentrations in the TMA pore water were high (ranging 190 to 1,800 mg/L), and therefore radium release into pore water is expected to be low. Therefore, a conservative upper bound for pore water radium was suggested at 5.5 Bq/L, the maximum concentration observed in the 2006 Pond C sediment samples where sulphate concentrations in porewater were significantly lower (minimum observed concentration 75.3 mg/L) and are consistent with maximum concentrations observed at the Quirke TMA.

- Mass transport theory indicates that the concentrations in the basin cannot exceed those in the porewater. Therefore, diffusive flux indicates that based on a pore water upper bound of 5.5 Bq/L, an upper boundary for radium activities in the basin water is in the range of 0.65 to 1.79 Bq/L.

3.4.4 Groundwater Quality

Three locations (wells) are sampled annually for acidity, pH, iron and sulphate. Two wells are located in the Main Basin down-gradient of Dams E (P-31) and B (P-16A) and one is located down-gradient of Dam A (P-20) in the South Basin (Figure 3.15).

Since decommissioning, groundwater in the Main Basin down-gradient of Dam B (P-16A) showed a significant increase in sulphate and decrease in pH over time (1990-2009), although conditions have been stable or possibly improving since 2005 (Table 3.13; Appendix Figure C.4.2). These trends are representative of acidic waters from early decommissioning being flushed through the groundwater. No significant trends were found at the other groundwater station down-gradient of the Main Basin (P-31) although the data tend to reflect the same pattern. In the South Basin down-gradient of Dam A (P-20 – towards Pond C) sulphate in groundwater has decreased over time (Appendix Figure C.4.3) consistent with the trend observed in South Basin surface water (Table 3.12).

3.4.5 Treatment Performance

Surface water from the Panel Main Basin discharges to the South Basin. Overflow from the South Basin is treated at the ETP and associated settling ponds prior to discharge to the receiving environment (P-14; Figure 3.15). The TMA ETP uses both lime (used caustic soda 2003 to 2007) and barium chloride to reduce acidity and radium-226 levels, respectively. Reintroduction of lime as the neutralizing agent in 2007 has enabled reduction in the barium chloride addition rate by 0.5 mg/L although total consumption increased in 2008 and 2009 due to higher treatment volumes (Figure 3.18).

Treated effluent is monitored at the outlet of the ETP settling pond (P-14) and over, the past five years, effluent quality has consistently achieved discharge criteria (Figure 3.19; Appendix Table C.4.1). Effluent has also been consistently non-lethal to *Daphnia magna* and rainbow trout with no mortality reported in semi-annual acute toxicity tests (Table 3.14). Similarly, survival and reproduction of *Ceriodaphnia dubia* was not affected by exposure to 100% effluent in any tests conducted over the past five years (Table 3.14).

3.4.6 Summary

Tailings water cover at the Panel TMA has been maintained and since 2008, water levels within the South Basin have been more stable than in previous years. In-basin surface water quality has been improving over time and is near or achieving the 50-year EIS predictions (*i.e.* the TMA is performing as anticipated). Since decommissioning, groundwater down-gradient of the Main Basin showed a significant increase in sulphate and decrease in pH over time (1990-2009), although conditions have been stable or possibly improving since 2005. In the South Basin down-gradient of Dam A, groundwater sulphate has decreased over time consistent with the trend observed in surface water. In the past five years effluent quality consistently achieved discharge criteria and all tests to *Daphnia magna*, rainbow trout

Table 3.13: Summary of water quality trends[ab] in TOMP groundwater in Panel TMA, 1990[c] to 2009.

Location	Station	Depth (m)	Dates	Iron	pH	Sulphate
downgradient of dam A (south basin)	P-20	13.9	1990-2009	-0.374	-0.428	-0.902
downgradient of dam B (main basin)	P-16A	24.8	1990-2009	-0.086	-0.751	0.699
below dam E (main basin)	P-31	9.97	1996-2009	0.012	-0.332	0.169

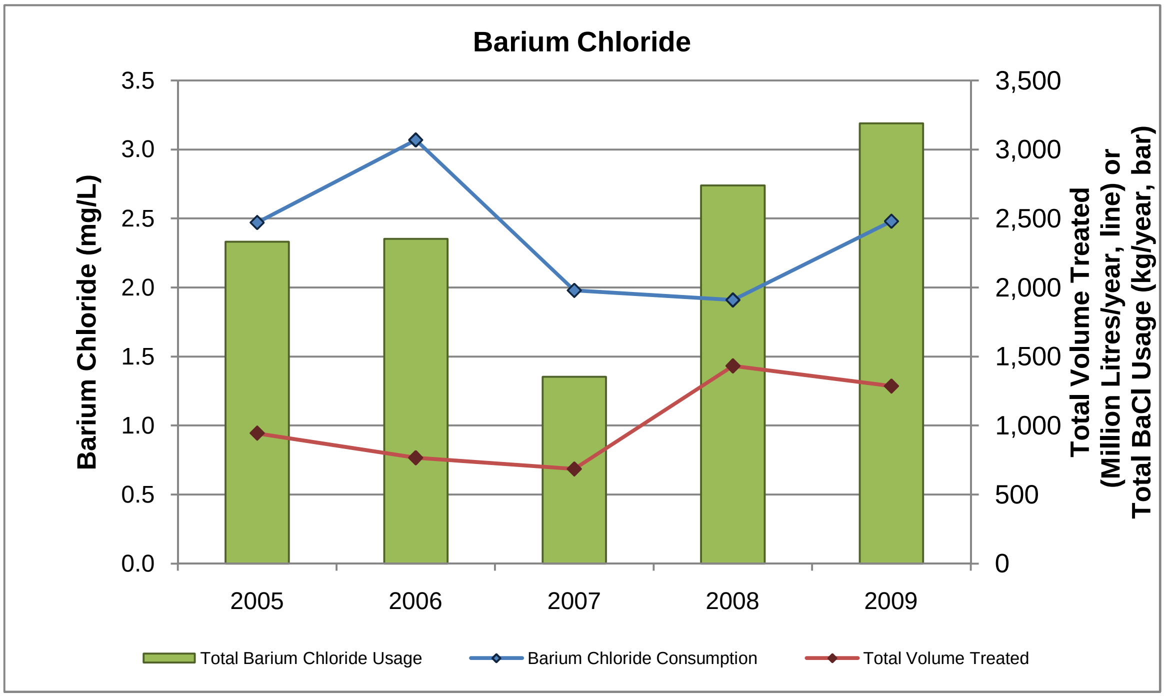
decreasing trend, significant at $p<0.05$ (blue)

increasing trend, significant at $p<0.05$ (green)

[a] Trends were not assessed for acidity because a change in analytical technique in 2006 meant that the data were not comparable before and after that time.

[b] Based on rank correlation coefficients (rho) for common (combined) season trends, shown in table.

[c] This is the earliest year included in the trend analysis, but not all stations have data going back to 1990.



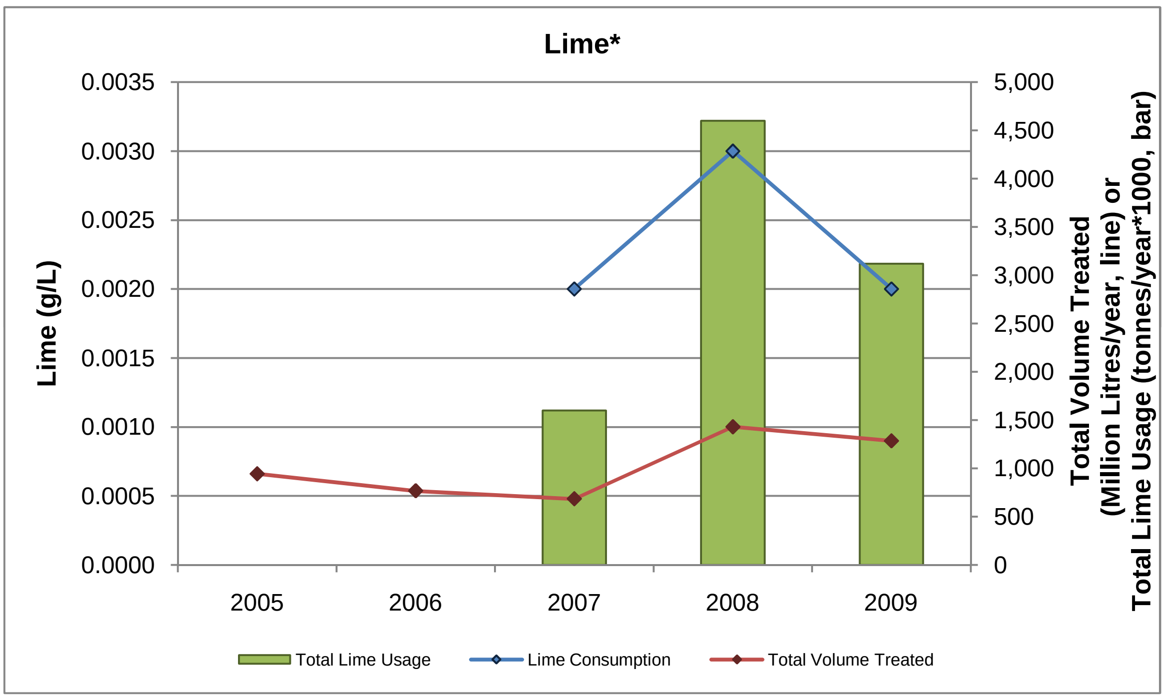


Figure 3.18: Comparison of total reagent consumed versus total volume treated at Panel TMA from 2005-2009 (* Caustic Soda in 2005 & 2006).

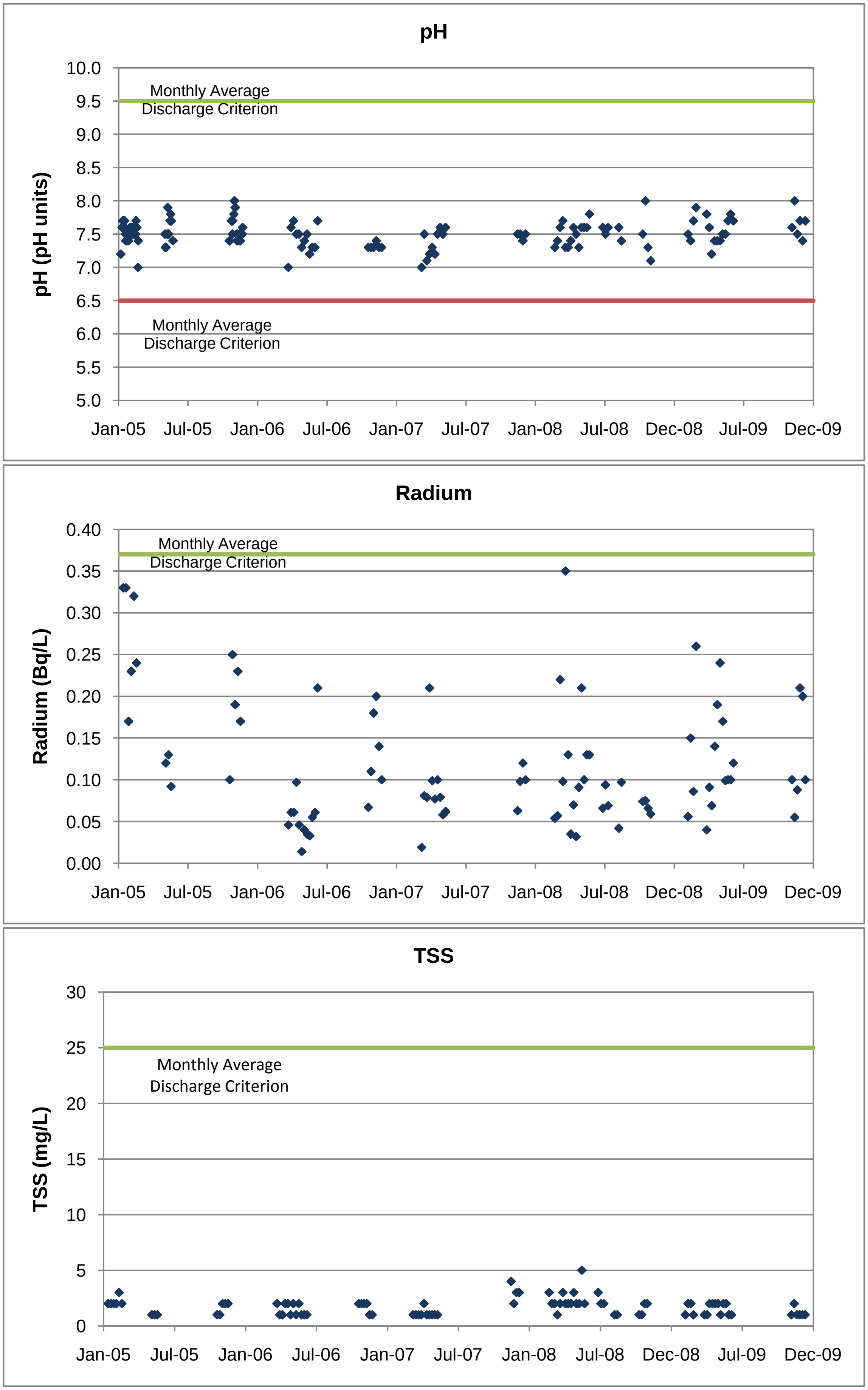


Figure 3.19: Effluent concentrations versus monthly average discharge criteria at Panel TMA station P-14.

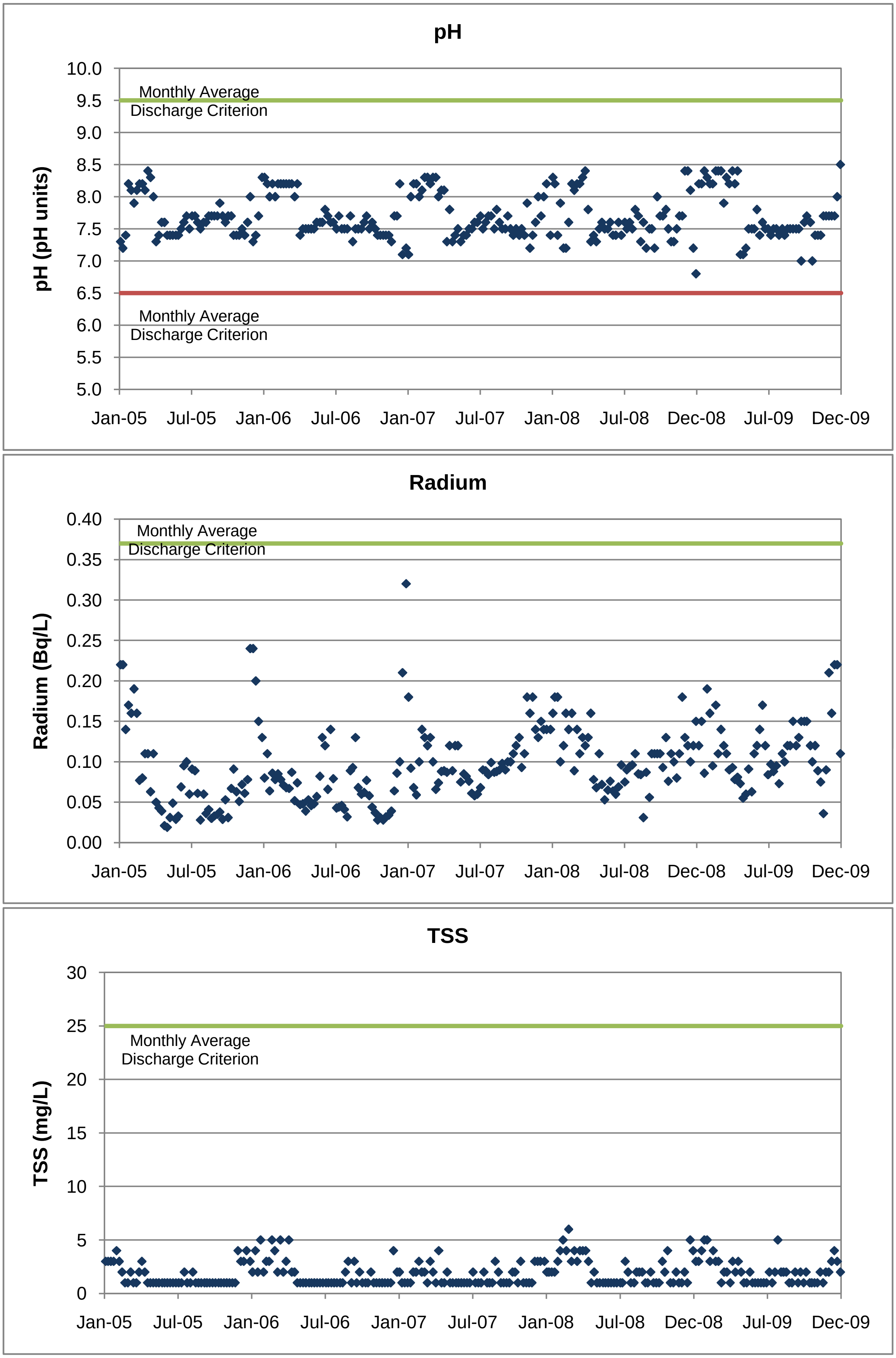


Figure 3.14: Effluent concentrations versus monthly average discharge criteria at Quirke TMA station Q-28.

and *Ceriodaphnia dubia* were non-toxic. Overall, the Panel TMA is performing well and conditions are improving over time.

3.5 Stanrock TMA

3.5.1 Basin History and Modifications

Stanrock Uranium Mines Limited and Can-met Exploration Limited began mining operations in early 1958. Both companies discharged their tailings to the natural basin of a small lake located immediately south of the mines that became the Stanrock TMA (Figure 3.20). On March 24, 1960, Can-met Exploration Limited amalgamated with Consolidated Denison Mines Limited and shortly thereafter operations at the Can-met mine were suspended. In 1964, underground operations at the Stanrock mine were also suspended, at which time the discharge of tailings ceased. Approximately 5.7 million tonnes of tailings were produced and stored within the 52-hectare Stanrock TMA over the course of mine operations.

Between 1964 and 1970, leaching solution, supplemented by water from Quirke Lake, was employed to leach uranium from the underground mine. The uranium-bearing liquor was processed in the mill ion exchange circuit to recover uranium, and then was returned underground. Excess solution was neutralized and discharged to the Stanrock TMA. In 1973, Denison Mines amalgamated with Stanrock Mines and, from 1978 to 1983, the Stanrock mine was re-established and underground development was carried out as part of an Ontario Hydro expansion. During this time, underground mine water was processed and neutralized mine water was discharged to the Stanrock TMA. A small amount of ore was processed in the Denison mill.

An "*In Situ* Management Plan" using a vegetation cover was chosen as the preferred option for decommissioning the Stanrock TMA. In accordance with the decommissioning plan, the following major activities were completed to decommission the Stanrock TMA between 1997 and 1999:

- Construction of a new rock cut spillway near Dam A;
- Construction of new low permeability engineered Dams A, B, C, and D;
- Reconstruction of Dam K and spillway to provide additional sludge storage capacity;
- Relocation of sludge within Moose Lake;
- Upgrading of Dam F to ensure long-term stability;
- Upgrading of Orient Lake outlet berm;

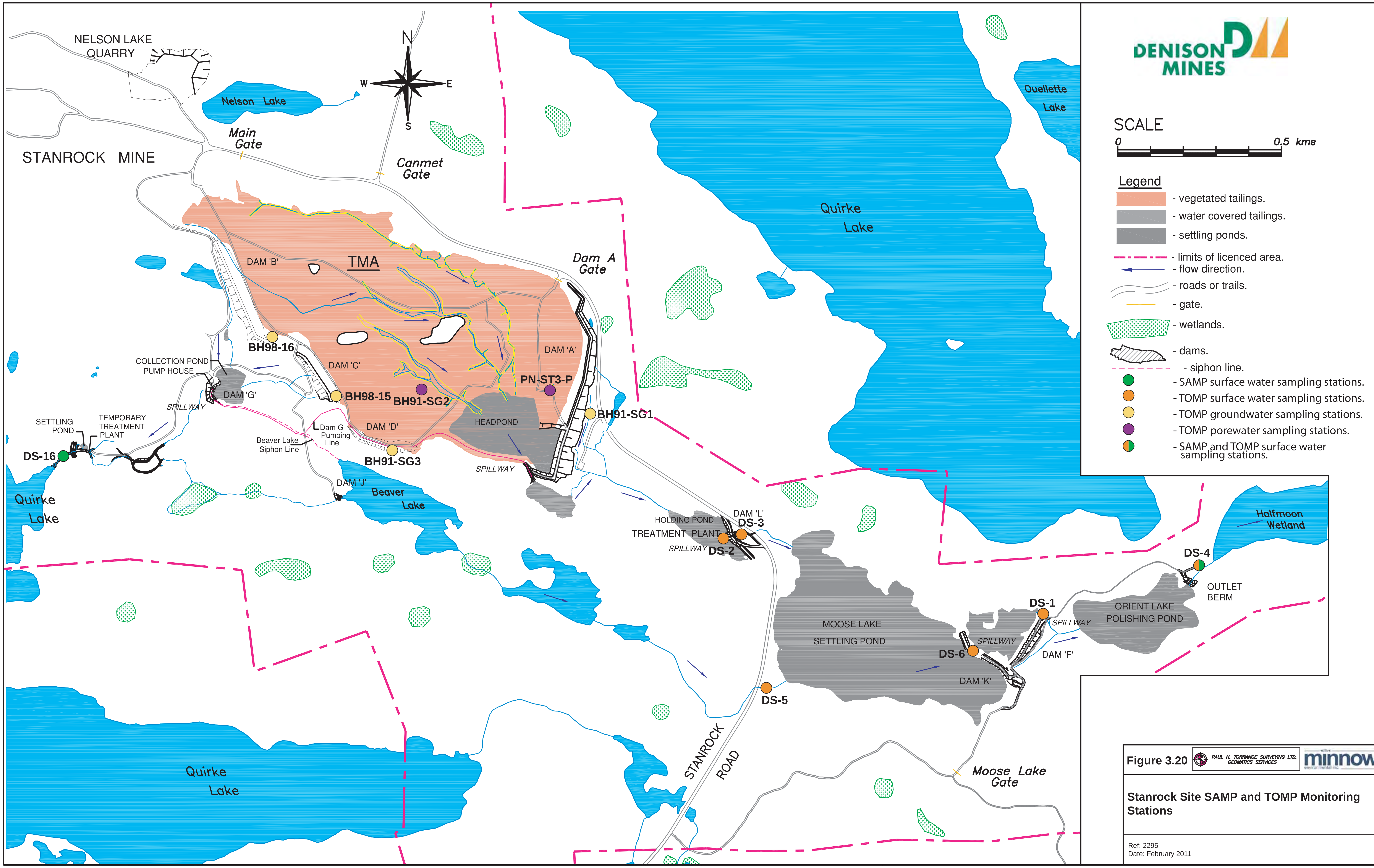


Figure 3.20

Stanrock Site SAMP and TOMP Monitoring Stations

- Remediation of spilled tailings;
- Establishment of rock lined channels on the Stanrock TMA for surface drainage;
- Vegetation of the tailings;
- Installation of new monitoring piezometers in the new dams and tailings to measure water levels; and
- Construction of a new water treatment plant that allowed for storage of untreated water and improved reagent mixing with untreated effluent.

In 1997 and 1998 new containment dams were constructed downstream of the existing structures. The dams incorporated a water retaining core of compacted till that is founded on bedrock. The bedrock foundation beneath the dam core was grouted to minimize seepage. Filters and drains were provided to prevent internal erosion and a build-up of porewater against the dam. Construction of the low permeability dams began in 1997 and was completed in 1998.

Approximately 40 ha of the Stanrock TMA were vegetated in 1998 with the remainder, in the area of the main headpond, being completed in 1999. Although there is a small headpond, water is generally not impounded in the TMA, but drains from the surface and passes through a spillway near Dam A to the Stanrock treatment plant. Seepage from Dams B and C is collected in the Dam G Collection Pond and pumped to the Dam A spillway where it flows downstream to the ETP holding pond for treatment at the ETP located to the southeast of the TMA (Figure 3.20). Treated effluent is discharged into the Moose Lake settling pond which flows into Orient Lake for further polishing and eventually to Halfmoon Lake, which is the first downstream receiver after the final point of control (DS-4, Orient Lake Outlet). Currently, DMI is in the process of replacing beaver dams at the outlet of Halfmoon Wetland with engineered berms to better contain treatment solids and tailings associated with an historical spill that occurred in 1964. The project is expected to be completed by the spring of 2011.

Since early 2005, Beaver Lake water, which receives seepage from Dam D, has been siphoned to the Dam G Collection Pond (and thereafter pumped to the ETP) to reduce untreated seepage overflow to Moose Lake.

In the summer of 2005, an issue arose regarding historic low pH water entering Quirke Lake from an area downstream of the Dam G Collection Pond. This area was a result of a historical tailings spill that occurred in 1964. In 2000, tailings were removed from Quirke

Lake and placed within the Stanrock TMA. In addition, the two tailings deposits (upper and lower) between Dam G and Quirke Lake were covered with a layer of sand and gravel to attenuate gamma and to raise the water table and saturate the tailings. The drainage pathway was directed around the two deposits in order to reduce the flushing of contaminants from the covered tailings.

Additional measures were taken in October of 2005 in order to address the low pH entering Quirke Lake at DS-16, which included the installation of a temporary sodium hydroxide treatment system located downstream of the outlet of the lower tailings deposit. A sludge collection basin was excavated in the lower tailings deposit immediately downstream of the sodium hydroxide addition point. Three concrete measuring weirs were also installed on the flow path between Dams G and J and Quirke Lake and a more rigorous sampling program was implemented. These measures were undertaken in order to better understand the mechanisms that were taking place in the area below Dam G, such that a final solution to deal with the low pH water could be determined. Based on the supplemental data obtained for this area, an improvement plan was designed and approved by the CNSC in consultation with other members of the Elliot Lake JRG. The improvement work involved the removal of tailings in the upper and lower wetland areas and construction of; fresh water diversions, a seepage collection pond, dam and spillway, and pumping station at the receiving end of the lower wetland to collect surface runoff and seepage water. This remedial work was completed in November of 2010. The water collected from these works is pumped to the Dam G Collection Pond and eventually through to the Dam A headpond. The water then drains through the spillway to the ETP for treatment, and discharge to Halfmoon Lake via the Moose Lake settling pond and Orient Lake polishing pond.

Based on the supplemental data obtained for this area, an improvement plan has been designed and is currently being reviewed by the Elliot Lake Joint Review Group. The proposed maintenance work will include removal of tailings in the upper and lower wetland areas and construction of a seepage collection pond, dam, and pumping station at the receiving end of the lower wetland to collect surface runoff and seepage water. These waters will be pumped to the Dam G Collection Pond and eventually through the Dam A spillway to the ETP for treatment, and discharge to Halfmoon Lake via the Moose Lake settling pond and Orient Lake polishing pond.

Within the TMA, surface water, porewater and ground water are monitored under the TOMP and the locations, substances and frequency monitored are specific to the station type (Table

3.15; Figure 3.20) Data from the TOMP stations are summarized in the following sections and presented in Appendix C (Appendix Tables C.5.2-C.5.12).

3.5.2 Basin Surface Water Quality

Stanrock is a vegetated TMA and as such there is no surface water within the TMA. Surface water runoff and seepage are collected in a holding pond and represent the influent to the ETP treatment plant (DS-2). In addition, water within downstream settling ponds (DS-6) and polishing ponds (DS-1), as well as the final effluent (DS-4), are monitored (Figure 3.20).

Since 2003, TMA water quality at the ETP influent has improved with significant reductions in radium-226 and sulphate (Table 3.16; Appendix Figure C.5.1). Influent radium-226 is now below the discharge criterion (0.37 Bq/L) but sulphate remains elevated and acidity continues to require treatment.

3.5.3 Porewater

Porewater is monitored annually at two locations in the Stanrock TMA: up-gradient of Dam A (PN-STP3) and up-gradient of Dam D (BH91SG2) (Table 3.15; Figure 3.20) for acidity, pH, iron and sulphate.

Up-gradient of Dam D, tailings porewater showed a significant increase in pH over time (1991 to 2009; Table 3.17; Appendix Figure C.5.5). Up-gradient of Dam A (PN-STP3) pH increased significantly in the shallow porewater (5.94 m), but decreased significantly over the same time at the deepest sampling depth (20.91 m; Table 3.17, Appendix Figure C.5.8). Iron increased significantly at both the shallow and deep sampling depths (Table 3.17; Appendix Figure C.5.7 and C.5.8). The increase in pH in shallower wells and the decrease in deeper wells likely reflect the on-going flushing of historic acidity from the tailing porewater over time.

Porewater pH at all depths except the deepest (>26 m) achieved the EIS predicted level for 2010, indicating that the TMA is performing as expected (Figure 3.21).

3.5.4 Groundwater Quality

Four groundwater locations are sampled annually for acidity, pH, iron and sulphate: one well is located down-gradient of each of the TMA Dams; A (BH91-SG1), B (BH98-16), C (BH98-15) and D (BH98-SG3 Figure 3.20).

Down-gradient of Dam A groundwater is assessed at 5.49m. Here both iron and pH levels have significantly increased over time (1991-2009; Table 3.17; Appendix Figure C.5.2)

Table 3.15: TOMP monitoring stations, substances, and frequencies[a] at Stanrock TMA.

		Parameters and Frequencies[a]										
TOMP Stations	**Station Type/Purpose**	**Flow**	**pH**	**Conductivity**	**Sulphate**	**Total radium-226**	**Lime or NaOH Consumption**	**Barium Chloride Consumption**	**TSS**	**Acidity**	**Iron**	**SAMP Metals[b]**
DS-2	Basin performance (primary), ETP operations	D	D		Q	M	M	M		Q		Q
DS-3	ETP operations		D									
DS-4	Effluent	W[c]	W		M	W			W			M[d]
DS-1	Additional pH control, radium monitoring	W	W			Q						
DS-6	Additional pH control	W	W									
DS-5	Seepages and surface water internal to TMA	Q	Q	Q								
PN-ST3-P3,5,6,8; BH91-SG2A,D	Porewater		A		A					A	A	
BH91-SG1A, BH98-16A, BH98-15A, BH91-SG3A,B	Groundwater		A		A					A	A	

[a] D - Work days, W - Weekly, M - Monthly, S - Semi-annually, A - Annually, Q-Quarterly

[b] SAMP metals are barium, cobalt, iron, manganese, and uranium

[c] Monitoring requirement of SAMP.

Table 3.16: Summary of water quality trends[a] for TOMP monitoring stations, Stanrock TMA, 2003 to 2009.

Station ID	Type/Location	Number of Seasons Used in Common Trend[b]	Acidity	Barium	Cobalt	Iron	Manganese	pH	Radium-226	Sulphate	Uranium
DS-2	Treatment Plant Influent	3 to 12	-0.130	0.364	0.221	0.142	0.311	-0.141	-0.458	-0.561	-0.253

decreasing trend, significant at p<0.05

increasing trend, significant at p<0.05

[a] Based on rank correlation coefficients (rho) for common (combined) season trends, shown in table.

[b] Seasons used varied for substances based on suitability of data for trend analysis.

Table 3.17: Summary of water quality trends[ab] in TOMP porewater and groundwater in Stanrock TMA, 1991[d] to 2009.

Type	Location	Station	Depth (m)	Dates	Iron	pH	Sulphate
porewater	upgradient of dam D	BH91 SG2A	33.31	1991-2009	0.274	0.643	-[c]
	upgradient of dam A	PN-ST3-P5	2.64	1999-2009	0.800	0.420	-
		PN-ST3-P3	5.94	1991-2009	-0.103	0.508	-
		PN-ST3-P6	11.58	1991-2009	0.409	0.387	-
		PN-ST3-P8	20.91	1991-2009	0.932	-0.552	-
groundwater	downgradient of dam A	BH91 SG1A	5.49	1991-2009	0.631	0.764	-
	downgradient of dam B	BH98-16A	5.49	1999-2009	-0.764	0.019	-
	downgradient of dam C	BH98-15A	7.86	1999-2009	-0.300	0.583	-
	downgradient of dam D	BH91 SG3B	5.85	1999-2009	-0.067	-0.280	-
		BH91 SG3A	8.78	1999-2009	-0.939	-0.165	-

decreasing trend, significant at $p<0.05$
increasing trend, significant at $p<0.05$

[a] Trends were not assessed for acidity because a change in analytical technique in 2006 meant that the data were not comparable before and after that time.

[b] Based on rank correlation coefficients (rho) for common (combined) season trends, shown in table.

[c] "-" denotes that this parameter was not included in the trend analysis for that particular station due to insufficient data (e.g. there were <5 years worth of data for that paramet

[d] This is the earliest year included in the trend analysis, but not all stations have data going back to 1991.

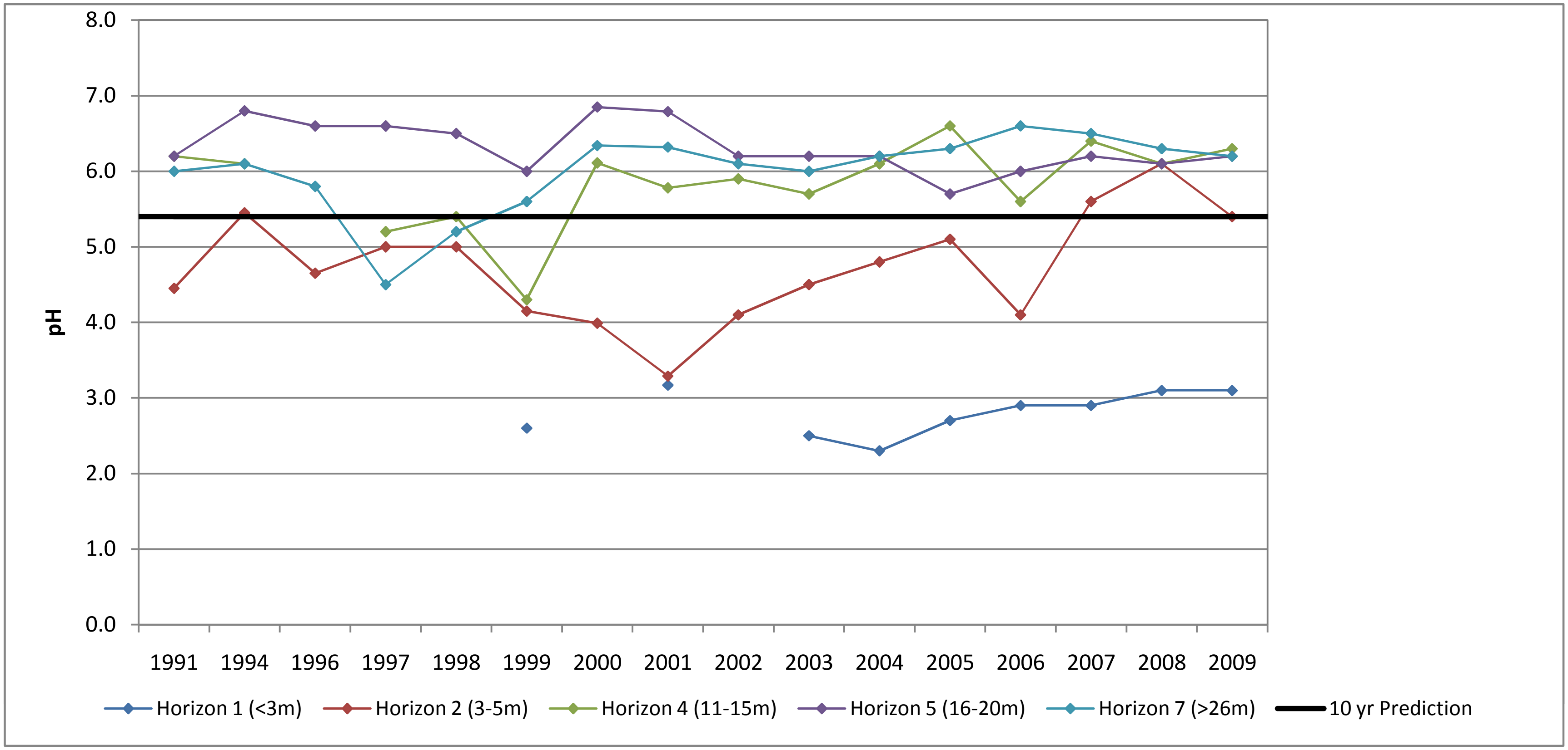


Figure 3.21: Comparison of mean porewater pH at various depths to EIS (2010) prediction, Stanrock TMA, 1991-2009.

Horizon 1 - PN-STP3-P5

Horizon 2 - PN-STP3-P3, BH91-SG2D

Horizon 4 - PN-STP3-P6

Horizon 5 - PN-ST3-P8

Horizon 7 - BH91-SG2A

consistent with porewater trends. Down-gradient of Dams B and D, iron concentrations significantly decreased (Table 3.17; Appendix Figures C.5.3 and C.5.4).

3.5.5 Treatment Performance

Water collected at Stanrock TMA is treated at the Stanrock ETP, then flows through a settling and polishing pond prior to discharge into Halfmoon Lake (Figure 3.20). Treatment includes both lime and barium chloride to reduce acidity and radium-226, respectively. Consistent with a reduction in radium-226 concentrations in the ETP influent, barium chloride consumption rates have decreased over the past five years, although the total usage has remained similar to other years likely due to higher treatment volumes in 2008 and 2009 (high precipitation years) (Figure 3.22). Lime usage was similar to previous years even though the volume treated in 2008 and 2009 was higher (Figure 3.22).

Following treatment, effluent quality is monitored at the outlet the polishing pond (DS-4). Over the past five years effluent quality has consistently achieved discharge criteria (Figure 3.23; Appendix Table C.5.1). Effluent has also been consistently non-lethal to *Daphnia magna* and rainbow trout with no mortality reported in semi-annual acute toxicity tests (Table 3.18). Similarly, survival and reproduction of *Ceriodaphnia dubia* was not affected by exposure to 100% effluent in any tests conducted over the past five years except for one sample collected in October 2007 (Table 3.18), in which reproduction was affected at an effluent concentration of 86%. However, it is expected that effluent concentrations would be diluted to less than 86% in the receiving environment.

3.5.6 Summary

Since 2003, TMA water quality at the ETP influent has improved with significant reductions in radium-226 and sulphate. Influent radium-226 is now below the discharge criterion (0.37 Bq/L) but sulphate remains elevated and pH continues to require treatment. Porewater pH has been increasing except at the deepest well and as a result, pH levels are for the most part, achieving levels predicted in the EIS for 2010. However, iron in porewater down-gradient of Dam A has been increasing over time, as has iron in groundwater down-gradient of Dam A. Groundwater down-gradient of Dams B and D showed a significant decrease in iron since decommissioning. Barium chloride consumption rate in the ETP has decreased over the past five years as a result on decreasing radium-226 concentrations in the ETP influent. Lime usage has remained stable. Effluent quality has consistently achieved discharge criteria over the past five years and has consistently been non-lethal to *Daphnia magna* and rainbow trout with no mortality reported in semi-annual acute toxicity tests.

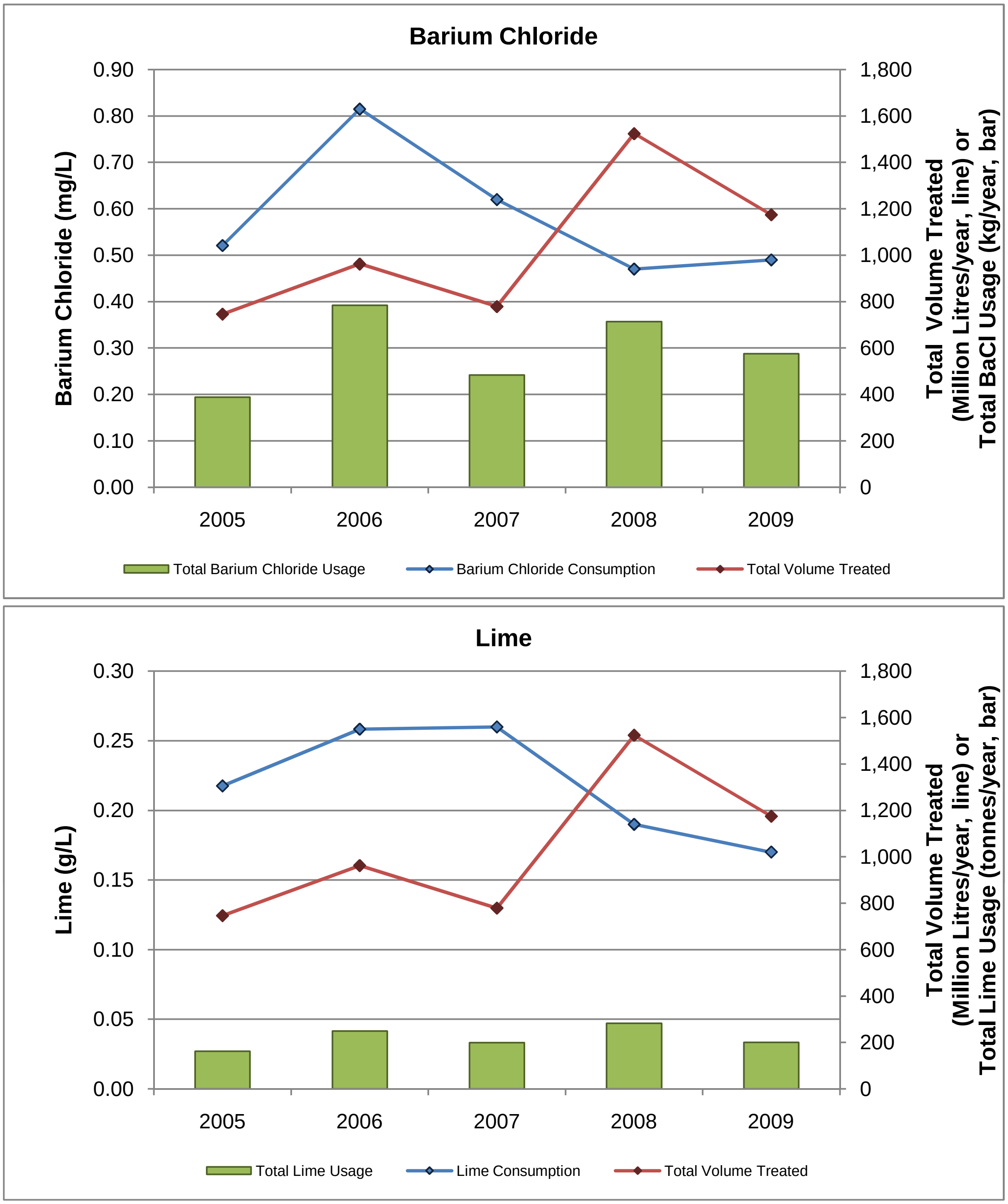
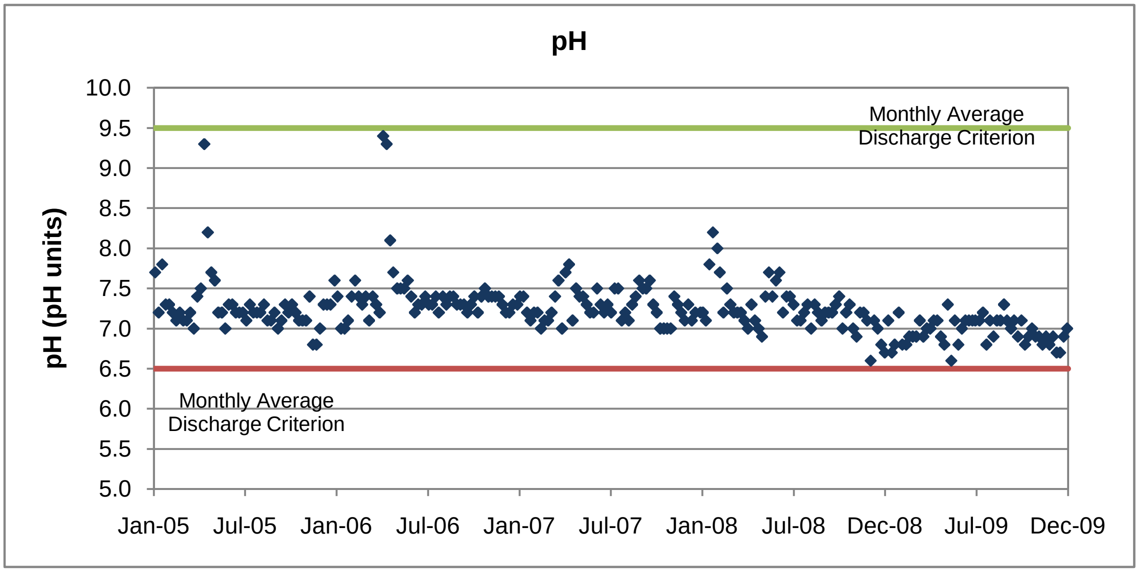
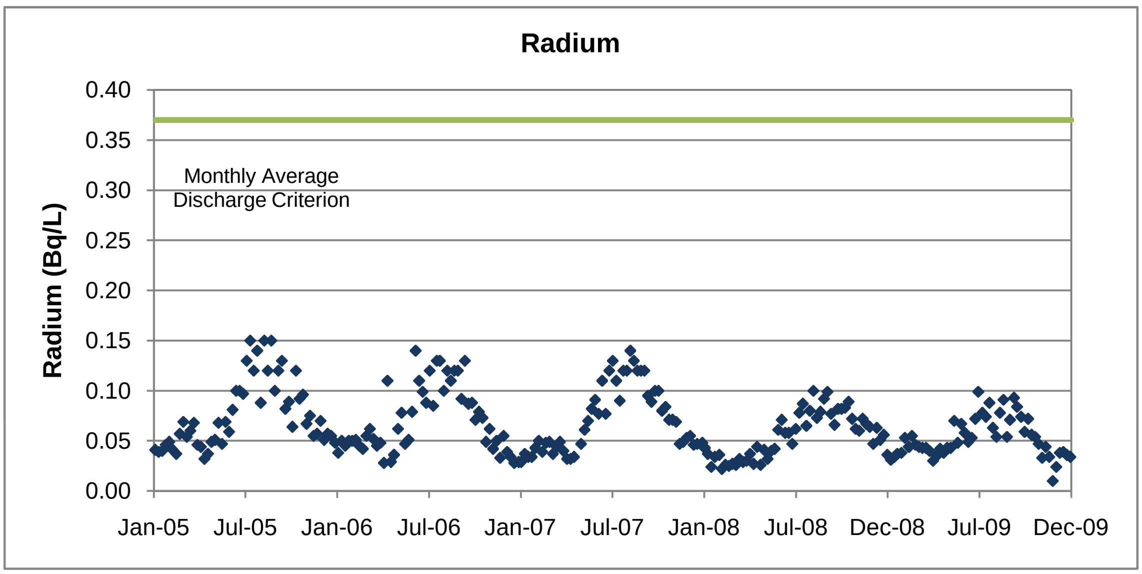


Figure 3.22: Comparison of total reagent consumed versus total volume treated at Stanrock TMA from 2005-2009.





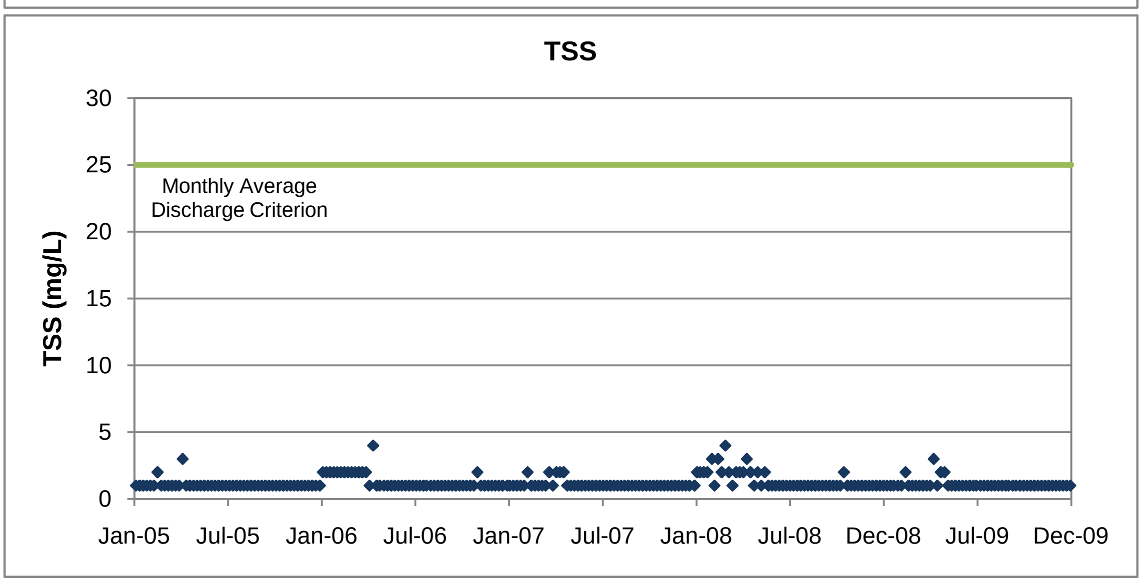


Figure 3.23: Effluent concentrations versus monthly average discharge criteria at Stanrock TMA station DS-4.

Table 3.18: Toxicity test results for samples collected at Stanrock TMA station DS-4, 2005 - 2009.

Sample Date (month-year)	Acute Toxicity (% mortality)		Survival and Reproduction (IC25[d] as % effluent)
	Daphnia magna [a]	**rainbow trout** [b]	***Ceriodaphnia dubia*** [c]
May-05	0	0	100
November-05	0	0	100
May-06	0	0	100
November-06	0	0	100
June-07	0	0	100
October-07	0	0	86
June-08	0	0	100
October-08	0	0	100
May-09	0	0	100
October-09	0	0	100

[a] *Daphnia magna* 48-hr LC50 test (Environment Canada 2000a).

[b] Rainbow trout 96-hr LC50 test (Environment Canada 2000b).

[c] *Ceriodaphnia dubia* survival and reproduction test (Environment Canada 2007).

[d] Effluent concentration causing 25% inhibition relative to control organisms.

Similarly, survival and reproduction of *Ceriodaphnia dubia* was not affected by exposure to 100% effluent in any tests except for slight reproductive impairment (IC25 86%) in one sample collected in October 2007.

3.6 Stanleigh TMA

3.6.1 Basin History and Modifications

The Stanleigh TMA is located 5 km north east of the City of Elliot Lake and contains 20 million tonnes of tailings from both the Milliken and Stanleigh mines and mills (Figure 3.24). During the initial operating period, 5.7 million tonnes were deposited in the west arm of the basin from the Milliken mill (1958 to 1964) and 1.7 million tonnes from the Stanleigh mill (1957 to 1960). In the mid 1960s, a lime and barium chloride treatment plant was constructed at the outlet of the West Arm with treatment solids settling in what is now the South Arm and treated effluent discharged to McCabe Lake through a concrete structure upstream of the current Dam B.

As part of the Stanleigh mill reactivation in the early 1980s, Dams 9, 10, R3 and R5 were constructed north and west of the basin to reduce the TMA watershed from 22 km^2 to 13.32 km^2. Five low-permeability engineered structures were constructed at bedrock lows around the basin to form the 350-ha TMA. During the second operating period an additional 12.8 million tonnes of tailings and waste rock were deposited in the basin, predominantly in the West Arm but also in the North Arm during later operating years.

An ETP was built at the TMA outlet in 1981, to treat effluent during operations. The ETP consisted of a reagent addition building and a filtration plant for treatment solids removal. Effluent from the Stanleigh TMA was treated and then discharged into McCabe Lake until 1998/1999, when, as part of the decommissioning of the Stanleigh Mine, the five perimeter dams were raised to allow flooding of the basin between 1998 and 2002. During this time, no treated effluent was discharged but the basin was neutralized by lime slurry addition to minimize acidity and metal concentrations.

Once treated effluent discharge resumed in 2003, water from the flooded TMA basin was siphoned over Dam B, and treated in the ETP prior to being released to McCabe Lake. The ETP operated for four to seven months per year depending upon the amount of snow and rainfall received. In 2007 the complex sand filtration treatment plant was replaced with a relatively simple conventional system similar to those used at all the other Rio Algom TMAs (e.g., Quirke, Panel, Nordic and Pronto). The new treatment system incorporates a Settling

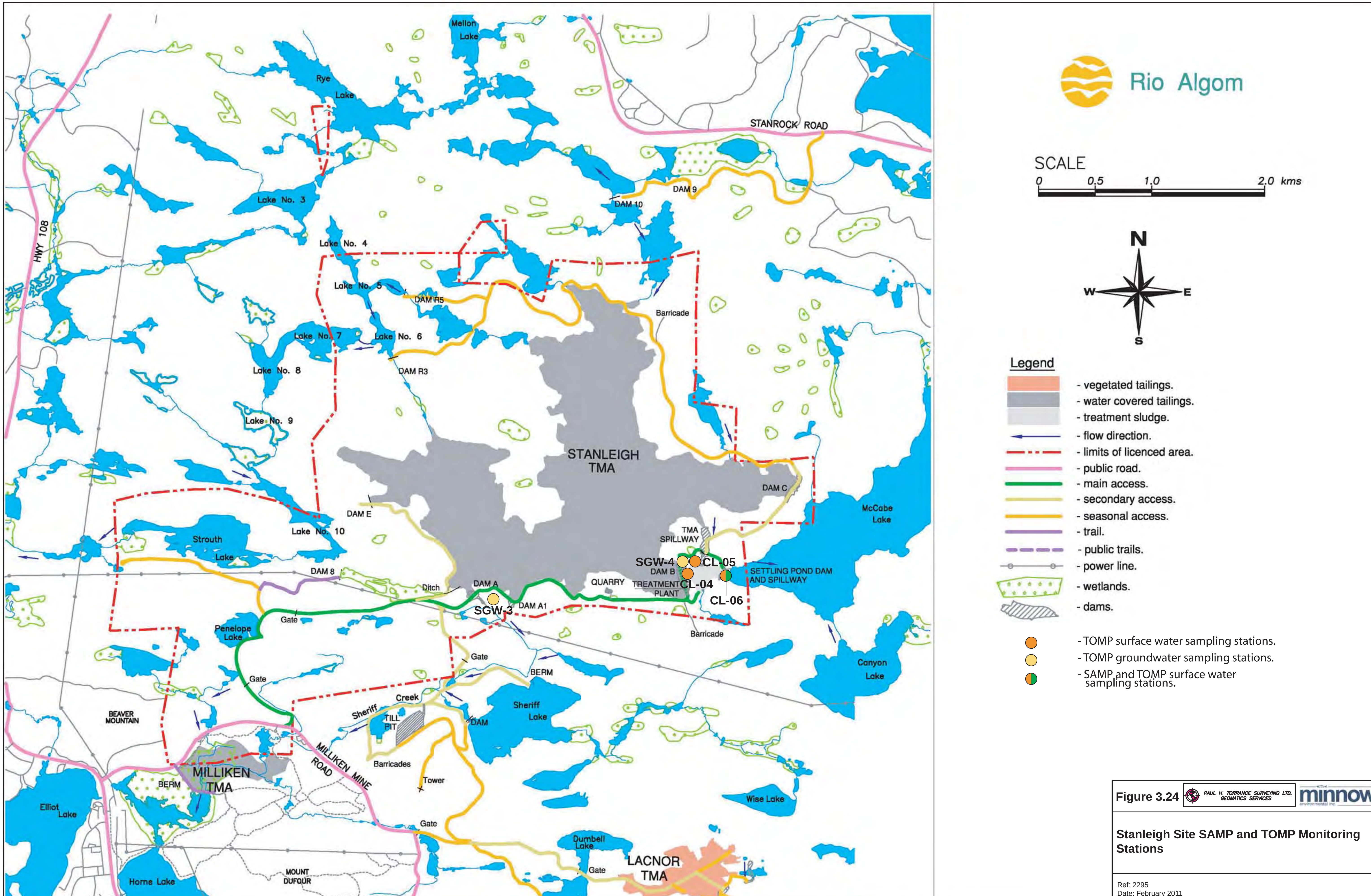


Figure 3.24

Stanleigh Site SAMP and TOMP Monitoring Stations

Ref: 2295
Date: February 2011

Pond for removal of solids created through the construction of the Settling Pond Dam downstream of the ETP

Within the TMA, surface water and groundwater are monitored under the TOMP and the locations, substances and frequency monitored are specific to the station type (Table 3.19 and Figure 3.24). Data from the TOMP stations are summarized in the following sections and presented in Appendix C (Appendix Tables C.6.2- C.6.5).

3.6.2 Water Management

Water levels within the flooded basin were consistently above the minimum operating level from 2005 to 2009 (Figure 3.25). In 2007, water in the TMA basin was drawn down to allow for the replacement of the ETP during the summer and fall of 2007. Increases in treatment volume and duration were required in the spring of 2008 to treat the water held in storage during the 2007 construction. By mid 2008 water levels within the TMA basin were within the established operating range (Figure 3.25).

3.6.3 Basin Surface Water Quality

Surface water quality is monitored at three stations within the TMA: the ETP Influent (CL-04) a pH probe in the ETP (CL-05) and final effluent (CL-06; Figure 3.24).

Concentrations of radium, sulphate and uranium have decreased and pH has increased to near neutral since basin flooding (Figure 3.26). Concentrations of sulphate and uranium are achieving 2012 predictions and radium-226 concentrations are near predicted values (Rio Algom 1997; Figure 3.26).

Surface water trends (2003-2009) indicate improvement based on significant reductions in acidity, iron, manganese, sulphate, and uranium in ETP influent (CL-04; Table 3.20; Appendix Figure C.6.1). Increases in radium-226 concentration since 2004 are likely associated with the decrease in sulphate concentrations within the basin. Work completed by EcoMetrix (Appendix G) indicates that as aqueous sulphate concentrations decline, there is an increased dissolution of barium sulphate to which radium is associated, whereby radium is released from the tailings. It is expected that radium concentrations in porewater will stabilize over time once the dissolution of barium sulphate re-equilibrates with aqueous sulphate concentrations. Assuming there is no new source of radium to the TMA, radium concentrations in porewater and releases to surface water should decline as the amount of soluble material in the tailings diffusion zone decreases.

Table 3.19: TOMP monitoring stations, substances, and frequencies[a] at Stanleigh TMA.

		Parameters and Frequencies[a]										
TOMP Stations	Station Type/Purpose	Elevation	Flow	pH	Sulphate	Total radium-226	Lime or NaOH Consumption	Barium Chloride Consumption	TSS	Acidity	Iron	SAMP Metals[b]
CL-04[e]	Basin performance (primary), ETP operations	W	D	M	Q	M	M	M		Q		Q
CL-05[e]	ETP Operations			D								
CL-06[d,e]	Effluent		W[c]	W[c]	M[c]	W			W			M[c]
SGW-3, SGW-4[d]	Groundwater			A	A					A	A	

[a] D - Work days, W - Weekly, M - Monthly, S - Semi-annually, A - Annually, Q-Quarterly

[b] Proposed SAMP metals are barium, cobalt, iron, manganese, and uranium

[c] Monitoring requirement of SAMP.

[d] Relocated to Settling Pond Dam.

[e] Sampled when treatment plant is operating.

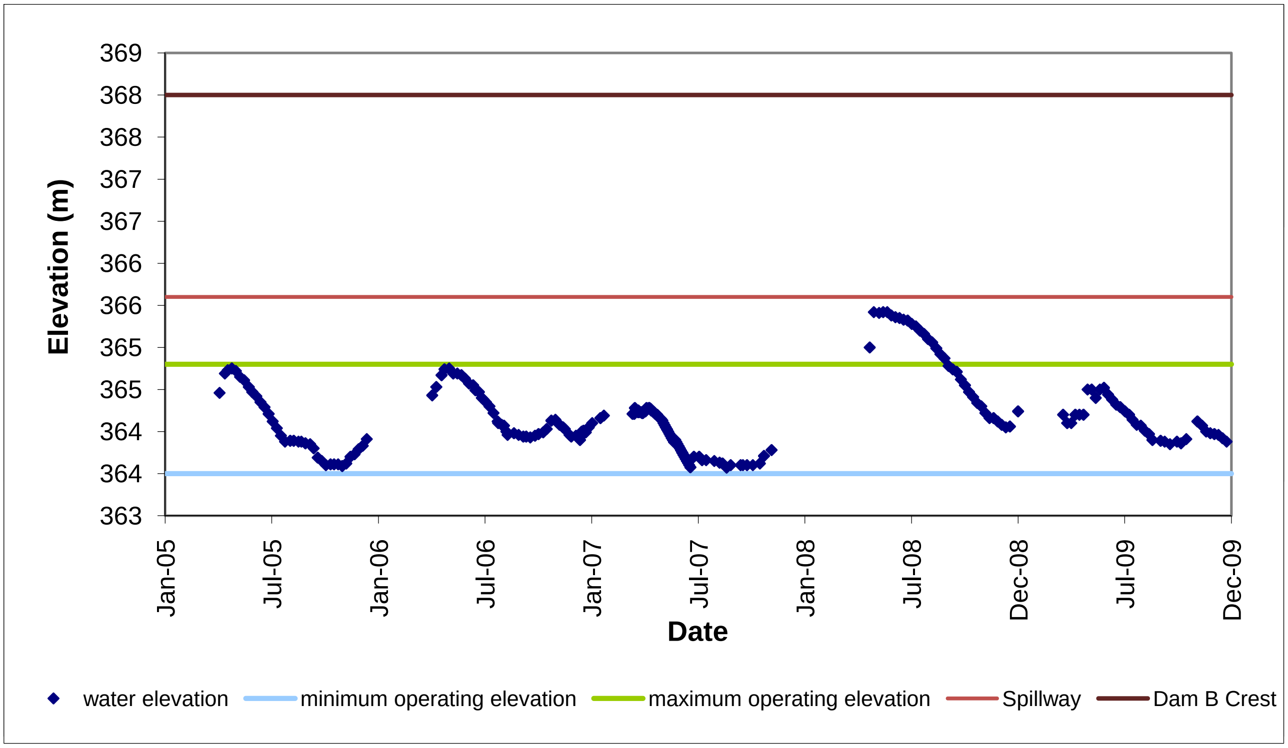


Figure 3.25: Water level at the Stanleigh TMA relative to minimum operating elevations.

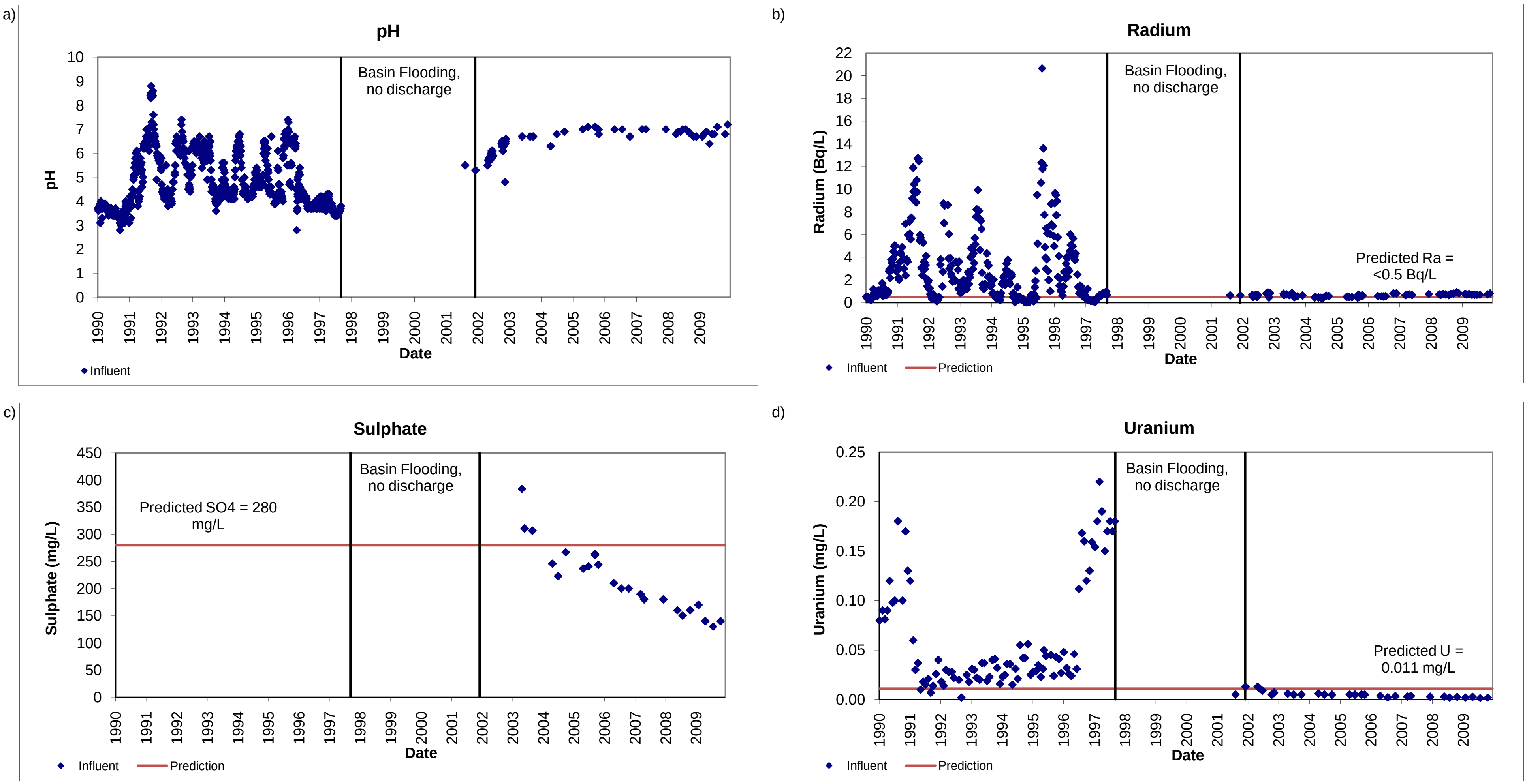


Figure 3.26: Water quality at the Stanleigh TMA ETP influent (CL-04) relative to predictions for 10 years (2012) post-decomissioning.

Table 3.20: Summary of water quality trends[a] for TOMP monitoring stations, Stanleigh TMA, 2003 to 2009.

Station ID	Type/Location	Number of Seasons Used in Common Trend[b]	Acidity	Barium	Cobalt	Iron	Manganese	pH	Radium-226	Sulphate	Uranium
CL-04	Treatment Plant Influent	3 to 7	-0.870	*-0.229* [c]	-0.455	0.764	-0.949	-0.150	0.454	-0.968	-0.702

decreasing trend, significant at p<0.05

increasing trend, significant at p<0.05

[a] Based on rank correlation coefficients (rho) for common (combined) season trends, shown in table.

[b] Seasons used varied for substances based on suitability of data for trend analysis.

[c] Italic text mean monthly correlations significantly different, but common trend value provided.

Influent pH achieves discharge criteria however, basin water still requires treatment to achieve the discharge criterion for radium-226 (Appendix Table C.6.2).

3.6.4 Groundwater Quality

Two locations (wells) are sampled annually for acidity, pH, iron and sulphate: down-gradient of Dam A (SGW-3) and down-gradient of Dam B (SGW-4; Figure 3.24).

Over the past 10 years (1999-2009) ground water quality down gradient of Dam A (towards Sheriff Creek) has improved, with significant decreases in iron and sulphate concentrations and increases in pH reflecting similar trends observed within the basin for iron and sulphate (Table 3.21; Appendix Figure C.6.2). Groundwater quality downstream of Dam 3B has remained stable with neutral pH and low iron (>0.3 mg/L; Table 3.21).

3.6.5 Treatment Performance

Surface water from the Stanleigh Basin is treated at the ETP and associated settling ponds prior to discharge to the receiving environment (CL-06; Figure 3.24). Treatment includes both lime and barium chloride additions to reduce acidity and radium-226 respectively. Treatment volume and reagent use were higher in 2008 and 2009 relative to previous years because excess water accumulated in the basin during the ETP replacement and due to higher precipitation in those years (Figure 3.27). Lime and barium chloride consumption rates have increased following replacement of the ETP, but remain within the design range based on the Panel ETP which has similar influent.

Following treatment, effluent quality is monitored at the settling pond outlet (CL-06) and over the past five years effluent quality has consistently achieved discharge criteria (Figure 3.28; Appendix Table C.6.1). While individual radium-226 concentrations exceeded the grab sample action limit during spring turnover in 2008 and 2009, these values were below the grab sample criterion of 1.11 Bq/L (Appendix Table D.6.1). Since the commissioning of the new ETP, effluent has been consistently non-lethal to *Daphnia magna* and rainbow trout with no mortality reported in semi-annual acute toxicity tests (Table 3.22). Prior replacement of the ETP, three samples were found to be acutely toxic to *Daphnia magna* (November 2005 and 2006 and June 2007; Table 3.22). Reproduction of *Ceriodaphnia dubia* was not affected by exposure to 100% effluent in any of the tests conducted over the past five years (Table 3.22).

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[3] This well was lost during the ETP construction in 2007 and therefore trends could only be assessed up to 2006.

Table 3.21: Summary of water quality trends[ab] in TOMP groundwater in Stanleigh TMA, 1999 to 2009.

Location	Station	Depth (m)	Dates	Iron	pH	Sulphate
downgradient dam A	SGW3	6.04	1999-2009	-0.955	0.954	-0.817
downgradient dam B[c]	SGW4	4.24	1999-2006	0.095	-0.452	-0.714

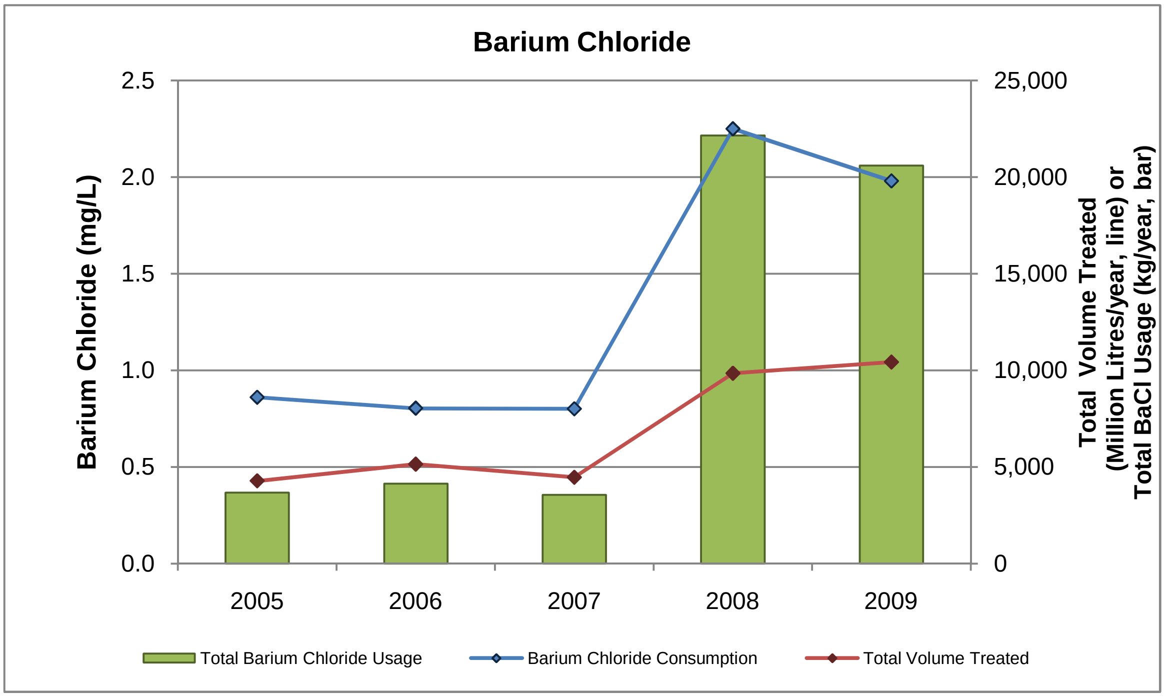
(blue) decreasing trend, significant at p<0.05

(green) increasing trend, significant at p<0.05

[a] Trends were not assessed for acidity because a change in analytical technique in 2006 meant that the data were not comparable before and after that time

[b] Based on rank correlation coefficients (rho) for common (combined) season trends, shown in table.

[c] SGW4 was lost during construction of the new Stanleigh TMA ETP and therefore the record of data ends in 2006.



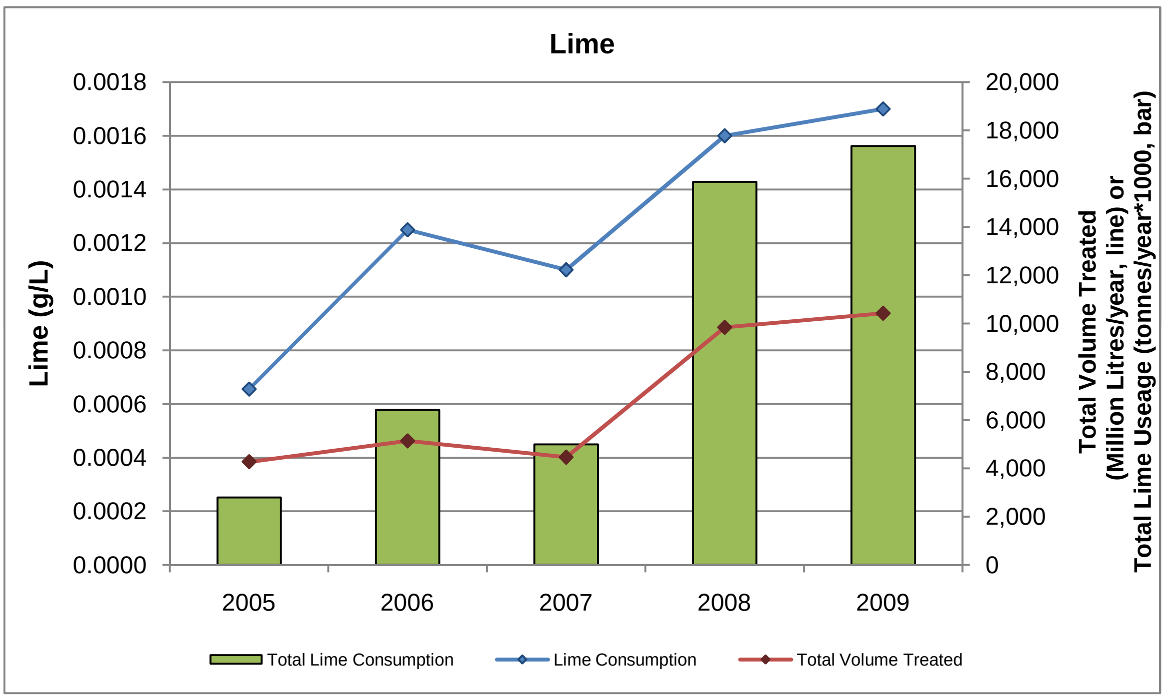


Figure 3.27: Comparison of total reagent consumed versus total volume treated at Stanleigh TMA from 2005-2009.

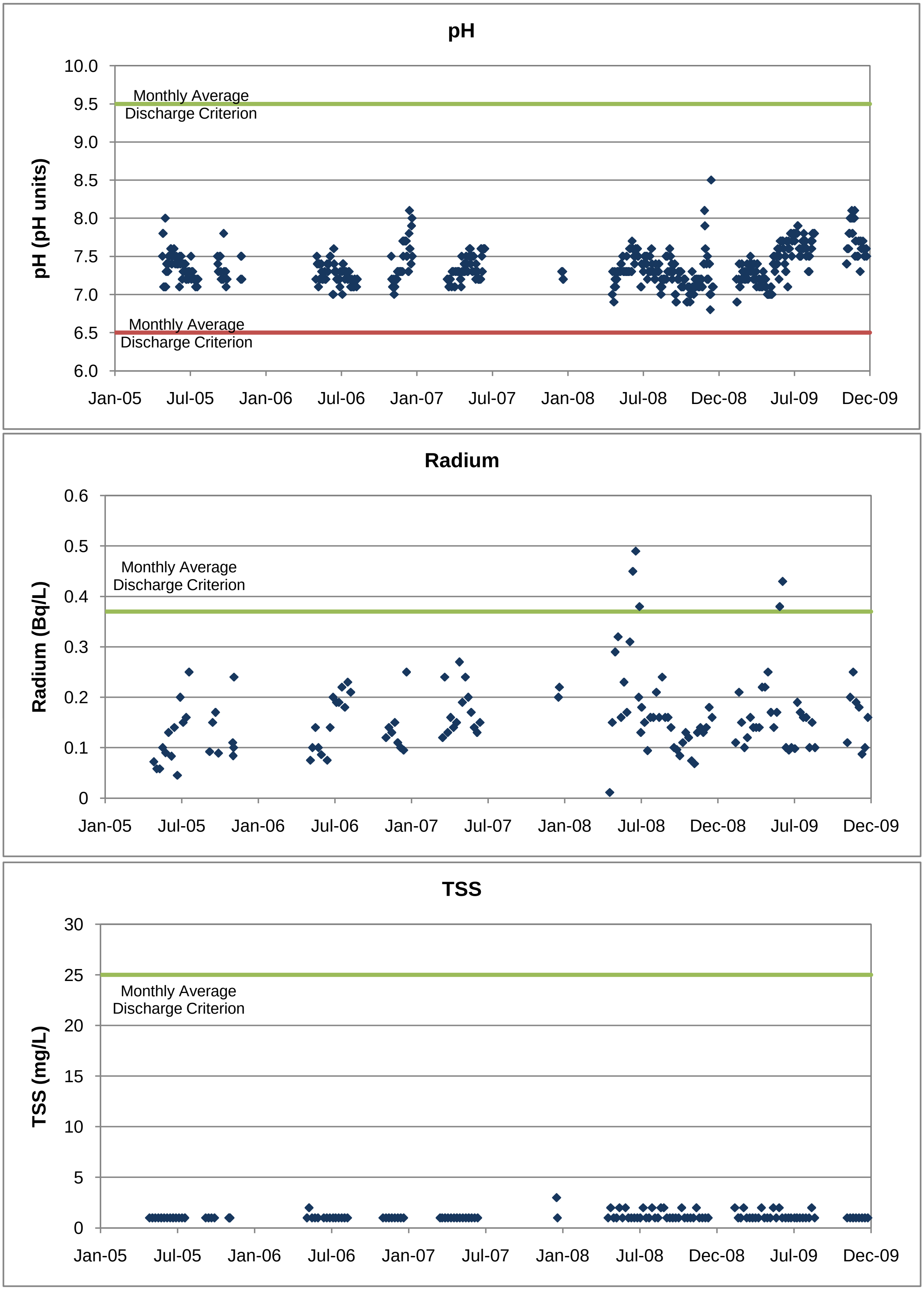


Figure 3.28: Effluent concentrations versus monthly average discharge criteria at Stanleigh TMA effluent station CL-06.

Table 3.22: Toxicity test results from samples collected at Stanleigh TMA station CL-06, 2005 - 2009.

Sample Date (month-year)	Acute Toxicity (% mortality)		Survival and Reproduction (IC25[d] as % effluent)
	Daphnia magna [a]	rainbow trout [b]	*Ceriodaphnia dubia* [c]
May-05	0	0	100
November-05	10	0	100
June-06	0	0	100
November-06	16.7	0	100
June-07	13	0	100
June-08	0	0	100
November-08	0	0	100
May-09	0	0	100
November-09	0	0	100

[a] *Daphnia magna* 48-hr LC50 test (Environment Canada 2000a).

[b] Rainbow trout 96-hr LC50 test (Environment Canada 2000b).

[c] *Ceriodaphnia dubia* survival and reproduction test (Environment Canada 2007).

[d] Effluent concentration causing 25% inhibition relative to control organisms.

3.6.6 Summary

Water levels within the flooded basin (2005 to 2009) were consistently above the minimum operating level from 2005 to 2009. In-basin surface water quality has been improving over time and generally achieves EIS predictions (*i.e.* the TMA is performing as anticipated). Over the past seven years (2003-2009) surface water has continued to improve with significant reductions in acidity, iron, manganese, sulphate and uranium in ETP influent. Radium-226 concentrations within the basin have been increasing over this same period in response to decreasing sulphate concentrations. It is expected that radium concentrations in porewater will stabilize over time once the dissolution of barium sulphate re-equilibrates with aqueous sulphate concentrations. Assuming there is no new source of radium in the TMA, radium concentrations in porewater should decline as the amount of soluble material in the tailings diffusion zone decreases. Groundwater conditions have either been stable (down-gradient of Dam B) or improving (down-gradient of Dam A) since TMA decommissioning. Since the commissioning of the new ETP effluent quality consistently achieved discharge criteria and all tests to *Daphnia magna*, rainbow trout and *Ceriodaphnia dubia* were non-toxic. Overall, the Stanleigh TMA is performing well.

3.7 Milliken TMA

3.7.1 Basin History and Modifications

The Milliken TMA is located 2 km northeast of the City of Elliot Lake and south of the Milliken Mine Road in an area locally referred to as the Sheriff Creek Sanctuary. The Milliken mine and mill operated from 1958 to 1964 and directed 5.7 million tonnes of tailings to the Stanleigh TMA. During this operating period an estimated 76,500 tonnes of tailings were released to Sheriff Creek in an area (17 ha) later rehabilitated to form the Milliken TMA. Remediation took place in the late 1970s by placing three feet of sandy gravel fill over a portion of the tailings to form playing fields and flooding the remaining tailings to form a wetland. In 1997, a berm was constructed at the outlet of the wetland to maintain water cover over the tailings. The resulting Sheriff Creek Sanctuary is now an important wildlife habitat area enjoyed by local naturalist groups.

Upstream of Sheriff Lake, Sheriff Creek receives drainage from a remediated tailings spill area down-gradient of Stanleigh TMA Dam A (see Stanleigh Section 3.5.1). Until its closure in 1996, the Stanleigh mine influenced the quality of water discharging from Penelope Lake, which drains into the north perimeter of the Milliken TMA (Figure 3.29). Similarly, the re-habilitated Lacnor Mine site, (closed in 1960 and rehabilitated in 1999), influences the quality of Lacnor Creek, which flows into the southeast corner of the TMA (Figure 3.29).

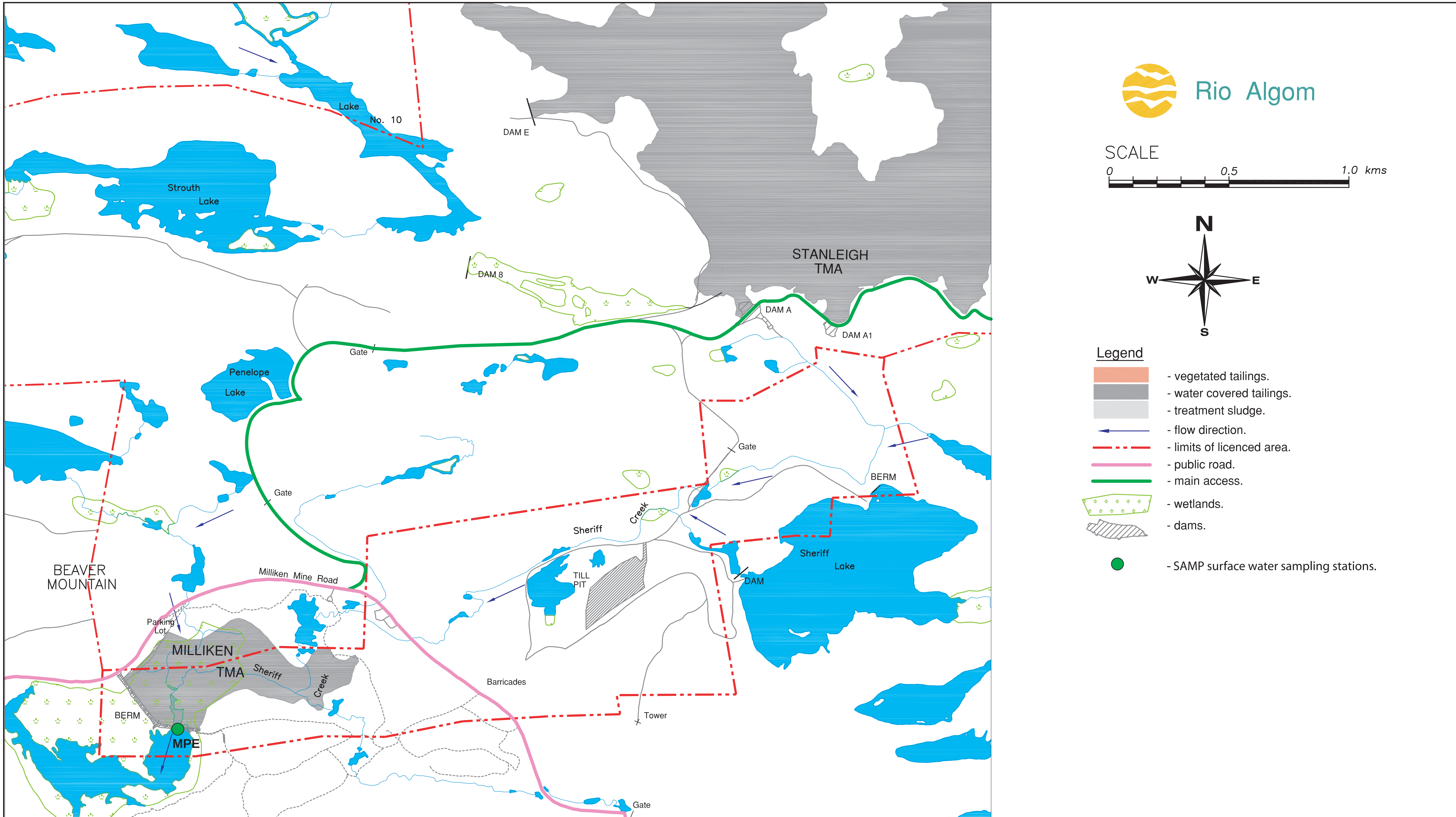


Figure 3.29

Milliken Site SAMP and TOMP Monitoring Stations

Ref: 2295
Date: February 2011

One monitoring station (MPE) was retained at the Milliken TMA outlet under the SAMP to track the combined inputs from all upstream sources and releases to the Serpent River Watershed (Appendix Table D.6.1).

3.7.2 Surface Water Quality and Discharge

Surface water quality is monitored at the outlet of the Milliken TMA (MPE) and reflects conditions within the TMA.

Effluent from the Milliken TMA discharges to a downstream wetland and joins the outflow from Horne Lake before entering Elliot Lake (Figure 3.29). Water quality at MPE generally meets receiving water criteria (see Section 4.3 for a discussion of discharge quality).

Since 2005, water samples collected at MPE have been non-toxic to both *Daphnia magna* and rainbow trout, with no mortality reported in semi-annual acute toxicity tests (Table 3.23). Similarly, survival and reproduction of *Ceriodaphnia dubia* were not affected by exposure to 100% effluent (Table 3.23).

3.8 Lacnor and Nordic TMAs

3.8.1 Basin History and Modifications

Lacnor TMA

The Lacnor TMA is located approximately 7 km east of the City of Elliot Lake and immediately north of the Nordic TMA. The Lacnor mine operated from 1957 to 1960 and milled approximately 2.7 million tonnes of ore. The resulting tailings were deposited in a natural valley 2 km east of the mill/mine and are contained by two pervious waste rock dams (Figure 3.30). The Lacnor TMA covers an area of 27 ha and has a watershed of 100 ha.

Following mine closure in 1960, decommissioning of the Lacnor TMA commenced, with re-vegetation efforts during the 1970s being a major component of the decommissioning plan. However, much of the seeding and planting on bare tailings failed over time due to acidic conditions (Rio Algom 2000). In 1998 and 1999, an engineered cover was placed over the tailings, which consisted of a layer of blast rock to form a capillary break and a layer of till at surface to serve as a growth medium. Limestone (200 kg/ha) was applied below the capillary break and fertilizer (500 kg/ha of 15-15-15) was applied prior to seeding. The cover areas were re-vegetated in 1999 through seeding of grasses and legumes and isolated tree plantings. Permanent rock channels were also installed to prevent erosion.

Seepage and runoff from the Lacnor TMA are collected in a holding pond at the east end of the TMA prior to discharge through a spillway to the Nordic Main TMA (Figure 3.30).

Table 3.23: Toxicity test results from samples collected at Milliken TMA station MPE, 2005 - 2009.

Sample Date (month-year)	Acute Toxicity (% mortality)		Survival and Reproduction (IC25[d] as % effluent)
	Daphnia magna [a]	rainbow trout [b]	*Ceriodaphnia dubia* [c]
May-05	0	0	100
November-05	0	0	100
May-06	0	0	100
November-06	0	0	100
May-07	0	0	100
November-07	0	0	100
May-08	0	0	100
November-08	0	0	100
May-09	0	0	100
November-09	0	0	100

[a] *Daphnia magna* 48-hr LC50 test (Environment Canada 2000a).

[b] Rainbow trout 96-hr LC50 test (Environment Canada 2000b).

[c] *Ceriodaphnia dubia* survival and reproduction test (Environment Canada 2007).

[d] Effluent concentration causing 25% inhibition relative to control organisms.

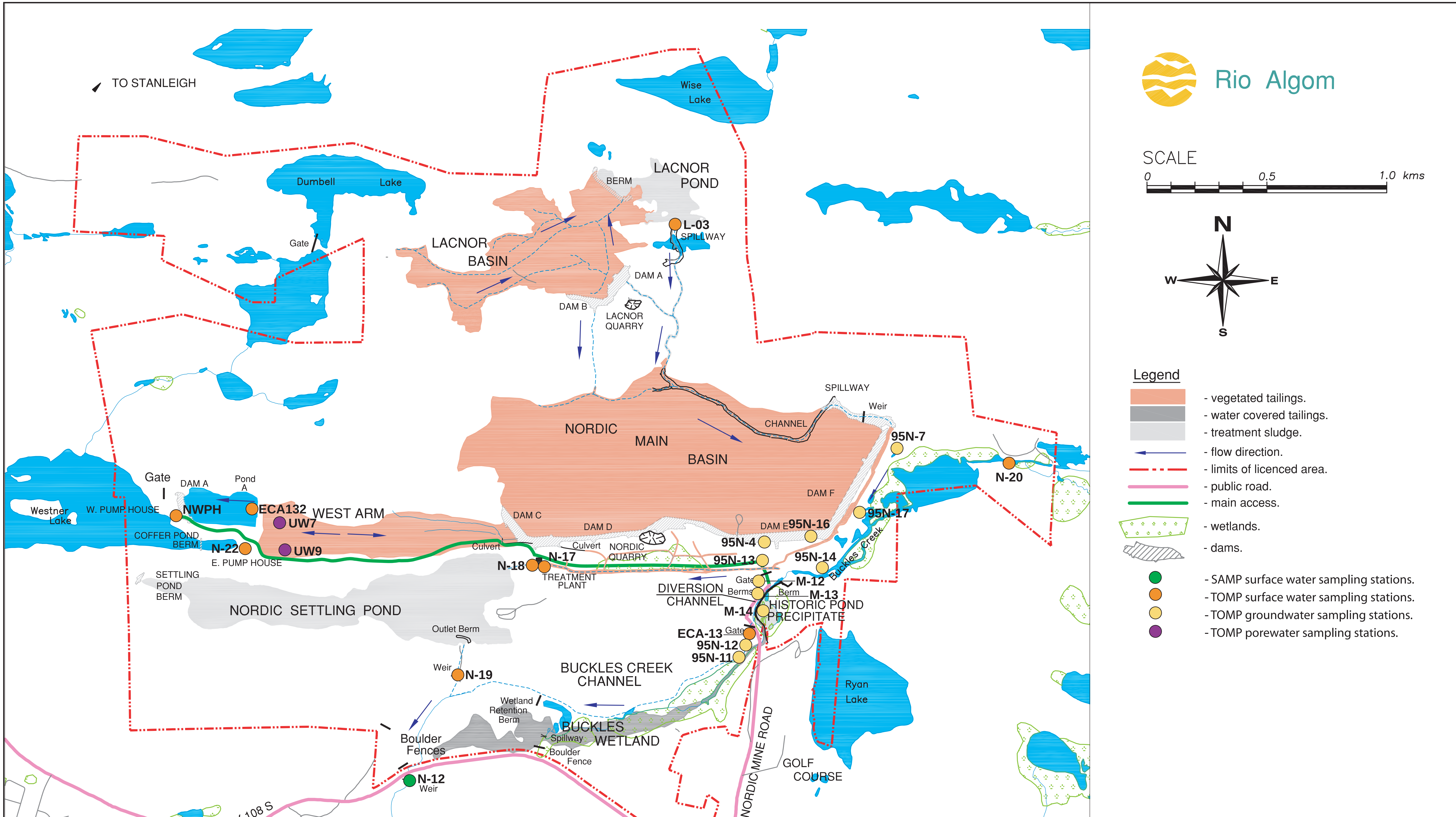


Figure 3.30

Lacnor, Nordic, Buckles Site SAMP and TOMP Monitoring Stations

Ref: 2295
Date: February 2011

Nordic TMA

The Nordic TMA is also located approximately 7 km east of the City of Elliot Lake immediately south of the Lacnor TMA. The Nordic mine operated from 1957 to 1968 and the Nordic mill produced approximately 12 million tonnes of tailings. Tailings were deposited to the Nordic TMA, which is composed of two areas (Nordic Main and Nordic West Arm) with a total area of approximately 107 hectares (Figure 3.30). Nordic Main is approximately 1,500 m long by 600 m wide and was constructed using mine waste embankments. Nordic West Arm is approximately 1,000 m long by 100 m wide.

The Nordic TMA was re-vegetated in the late 1970s (Rio Algom 2000). In 1998 and 1999, layers of rock (serving as a capillary break) and till were placed in areas of the West Arm which exhibited poor drainage and were prone to erosion, and thus tended to have relatively poor vegetative cover. These areas have been successfully re-vegetated

Seepage and runoff from Nordic Main are collected in a perimeter Effluent Collection Ditch (ECD) constructed in 1971. The ECD collects drainage from the Lacnor TMA at the north perimeter of Nordic Main which flows around the Nordic TMA to the Nordic ETP (located at the southwest corner of Nordic Main), for treatment prior to discharge into the Nordic Settling Pond (Figure 3.30). The ECD was lowered in 1994 and the Settling Pond was lowered by 0.6 m in 1997 to improve interception of tailings porewater and reduce groundwater contamination of Buckles Creek located south of Nordic Main. The treatment plant, where lime is added to neutralize acidity and remove metals (predominantly iron), was replaced in 1999. Treated effluent discharges to Buckles Creek and subsequently Nordic Lake (Figure 3.30).

The majority of seepage and runoff from the Nordic West Arm drains in an easterly direction and is directed by a series of ditches to the Nordic ETP for treatment. Runoff from the western portion of the Nordic West Arm is collected in Pond A, then pumped into the Nordic Settling Pond. The East and West Collection Ponds were constructed in 1989 to intercept seepage from Pond A and the West Arm, respectively, and pump it to the Settling Pond. In 2004, a coffer berm was constructed downstream of the East Collection Pond to facilitate removal of a small tailings spill discovered following the beaver dam break at the outlet of Westner Lake in 2003. In 2009, a pump well was installed in the Coffer Pond, and the pumping systems of Pond A, East Collection Pond and West Collection Pond were upgraded to manage a 1 in 100 year return, 15-day rain-on-snow design hydrological event.

During mine operations Buckles Creek was diverted to provide water for mining and milling and run-off from the Nordic Main was piped to the original Buckles Creek bed. From 1965 to 1975, barium chloride was used to treat the radium in Buckles Creek, with radium precipitates settling in a beaver pond (located by the mine road) and the creek bed. In the late 1970's, the precipitates were covered with fill and the Buckles Creek Channel was relocated to isolate the flow from historic deposits. Maintenance of the Buckles Creek Channel in 2005, included lining the section of channel above the point of confluence with the Nordic Settling Pond with rip rap and restoring the berm isolating the historic precipitate pond. Performance monitoring of diversion channel indicated that construction activities had lowered the creek elevation relative to the ECD resulting in increased groundwater seepage to Buckles Creek. Modifications to the diversion stream bed in 2006 reversed this flow restoring groundwater interception by the ECD as designed.

Monitoring station L-03 monitors releases from the Lacnor TMA to the Nordic TMA (Table 3.24). Within the Nordic TMA, surface water, porewater and groundwater are monitored under the TOMP and the locations, substances and frequency monitored are specific to the station type (Table 3.24 and Figure 3.30). Data from the TOMP stations are summarized in the following sections and presented in Appendix C (Appendix Tables C.7.2- C.7.23).

3.8.2 Basin Surface Water Quality

Surface water quality at the Lacnor/Nordic TMA is monitored at a number of stations to assess conditions associated with the various tailings deposits (Figure 3.30):

- Seepage and surface runoff from the Lacnor TMA are captured in the Lacnor Pond and is monitored at L-03 (Appendix Table C.7.4);
- Surface runoff from the Nordic TMA West Arm is collected in Pond A and monitored at ECA-132 with seepage from Pond A monitored at NWPH and seepage from the Nordic West Arm monitored in the East Seepage Collection Pond at N-22 (Appendix Tables C.7.3 and C.7.9 respectively);
- Seepage and runoff from the Nordic Main TMA and eastern sections of the West Arm of the Nordic TMA are monitored at the ETP influent (N-17; Appendix Table C.7.5); and
- Contributions from the Nordic Main TMA historic groundwater plume to Buckles Creek are monitored at ECA 131 (Appendix Table C.7.2).

Table 3.24: TOMP monitoring stations, substances, and frequencies[a] at Nordic TMA.

		Parameters and Frequencies[a]									
TOMP Stations	Station Type/Purpose	Elevation	Flow	pH	Sulphate	Total radium-226	Lime or NaOH Consumption	TSS	Acidity	Iron	SAMP Metals[b]
L-03	Basin performance (primary)	M[d]	Q	Q	Q	Q			Q		Q
N-17	Basin performance (primary), ETP operations		D	M	Q	M	M		Q		Q
N-18	ETP operations			D							
N-19	Effluent		W	W	M	W		W			M
N-22	Basin performance (secondary)		M[d]	S	S	S			S	S	S
ECA-132	Basin performance (secondary)	M[d]	M	M[d]	S	S			S	S	S
NWPH	Basin performance (secondary)		M	S	S	S			S	S	S
ECA-131, N-20	Basin performance (secondary)			Q	Q	Q			Q		Q
UW7-2,4,6; UW9-1,2,3	Porewater			A	A				A	A	
M-12-1,3,6,9; M-13-1,3,6,9; M-14-1,3,6,9; 95N-4A,B; 95N-7A,B; 95N-11; 95N-12A,B; 95N-13A,C,E; 95N-14A,B,C; 95N-16A,C,E; 95N-17A,B,C	Groundwater			A[c]	A[c]				A[c]	A[c]	

[a] D - Work days, W - Weekly, M - Monthly, S - Semi-annually, A - Annually, Q-Quarterly

[b] SAMP metals are barium, cobalt, iron, manganese, and uranium

[c] A one-time modelling exercise was recommended by Ecometrix to confirm flow conditions and potentially modify future GW monitoring under TOMP. In the GW monitoring at Nordic will continue will cotinue at previously identified TOMP stations.

[d] During the snow-free period (April - November)

Since 2003, sulphate concentrations at L-03 have decreased significantly (Table 3.25; Appendix Figure C.7.3). Currently surface water quality in the Lacnor Pond is acidic (pH < 3.5) with elevated iron (> 20 mg/L; Appendix Table C.7.4).

Surface water associated with the Nordic West Arm has also improved with significant reductions in radium-226 concentrations at the East Seepage Collection Pond (N-22) and decreasing acidity within Nordic Pond A (ECA-132; Table 3.25). Pond A was limed following the upgrading of Dam A in 2000 and the step change in acidity in 2006 is likely associated with the change in acidity analytical method in the same year.

Decreasing concentrations of acidity and radium-226 upstream of the Buckles Creek wetland (ECA-131; Table 3.25) are associated with: 1) remediation work conducted in 2005 to isolate the Wetland and Historic Precipitate Pond from the Diversion Channel, and 2) streambed modifications completed in 2006 which restored groundwater gradients towards the ECD and away from Buckles Creek.

Since 2003, water quality in the TMA influent (N-17) has significantly improved with decreasing concentrations of acidity, radium-226, sulphate, and uranium (Table 3.25; Appendix Figure C.7.4). Similarly, ETP effluent has also improved over the past seven years with significant decreases in cobalt, manganese and radium-226 and increased pH consistent with the upward adjustment of the treatment plant pH set point in 2004 (Table 3.25).

3.8.3 Porewater

Porewater is monitored annually for acidity, pH, iron and sulphate at two locations (north and south) in the west arm of the Nordic TMA (UW-7 and UW-9; Table 3.24; Figure 3.30).

Since 1993, iron has been significantly decreasing at both porewater locations. Iron concentrations at UW7 (shallowest depth) have decreased from about 2,000 mg/L in 1992 to about 500 mg/L in 2009 (Appendix Figure C.7.8). Sulphate was found to be significantly increasing at UW7-2 (8 m) (Table 3.26; Appendix Figure C.7.7).

Porewater pH at the north end of the West Arm (UW-7) has significantly increased in the deepest well and reflects a step change improvement following the upgrading of Dam A in 2000 (Table 3.26; Appendix Figure C.7.9). Similarly, porewater pH at the south end of the West Arm (UW9) has also been increasing over time and may also represent a response to the improvements in Dam A (Table 3.26 and Appendix Figure C.7.11). The pH in deep

Table 3.25: Summary of water quality trends[a] for TOMP monitoring stations, Lacnor/Nordic TMA, 2003 to 2009.

Station ID	Type/Location	Number of Seasons Used in Common Trend[c]	Acidity	Barium	Cobalt	Iron	Manganese	pH	Radium-226	Sulphate	Uranium
L-03	Lacnor Tailings Discharge	3	-[b]	-	-	-0.166	-	-0.426	-0.048	-0.713	0.338
ECA-132	Nordic Pond A upstream of Westner seepage	2	-0.870	-	-	-	-	*0.1*[d]	*0.245*	-	-
N-22	West Arm Pump Discharge (East Seepage Collection Pond)	2	-0.500	-	-	-	-	-0.198	-0.669	-	-
N-20	Buckles Creek Upstream of Nordic Plume	4	ND[e]	-	-	-	-	-0.083	-0.397	-	-
ECA-131	Buckles Creek at Mine Road	4	-0.540	-	-	-	-	0.070	-0.555	-	-
N-17	Treatment Plant Influent	4 to 12	-0.536	0.149	0.237	0.062	-0.018	-0.220	-0.325	-0.692	-0.696
N-19	Final Treated Effluent	12	-	0.083	-0.371	-0.004	-0.601	0.363	-0.530	-	-0.018

decreasing trend, significant at $p<0.05$

increasing trend, significant at $p<0.05$

[a] Based on rank correlation coefficients (rho) for common (combined) season trends, shown in table.

[b] "-" denotes that this parameter was not included in the trend analysis for that particular station due to insufficient data (e.g. there were <5 years worth of data for that parameter)

[c] Seasons used varied for substances based on suitability of data for trend analysis.

[d] Italic text mean monthly correlations significantly different, but common trend value provided.

[e] ND denotes that this parameter was not included in the trend analysis for that particular station due to >50% non-detectable concentrations in the samples available for the analysis.

Table 3.26: Summary of water quality trends[ab] in TOMP porewater and groundwater in Lacnor/Nordic TMA, 1993[c] to 2009.

Type	Location	Station	Depth (m)	Dates	Iron	pH	Sulphate
porewater	Nordic west arm, porewater north	UW7-4	5.14	1993-2009	-0.861	0.057	-0.409
		UW7-2	8.23	1993-2009	0.168	0.007	0.645
		UW7-6	16	1996-2009	-0.235	0.627	0.578
	Nordic west arm, porewater south	UW9-3	4.27	1993-2009	0.436	-0.401	0.518
		UW9-2	6.4	1993-2009	-0.175	0.615	-0.055
		UW9-1	8.53	1993-2009	-0.596	0.224	-0.155
groundwater	downgradient of ECD at northeast corner Nordic main	95N7B	3.69	1995-2009	-0.531	-0.600	0.429
		95N7A	7.72	1995-2009	-0.552	-0.613	-0.190
	downgradient of ECD at east perimeter Nordic main	95N17C	3.49	1995-2009	0.495	-0.656	-0.006
		95N17B	8.09	1995-2009	0.522	-0.500	-0.333
		95N17A	12.68	1995-2009	0.698	0.109	-0.491
	downgradient of ECD at southeast corner Nordic main	95N14C	3.49	1995-2009	0.402	-0.320	-0.610
		95N14B	7.6	1995-2009	0.354	-0.348	-0.176
		95N14A	11.39	1995-2009	-0.011	0.155	-0.486
	upgradient of ECD at head Nordic plume	95N13E	2.82	1995-2009	-0.886	0.641	-0.591
		95N13C	9.61	1995-2009	-0.886	0.624	-0.664
		95N13A	15.36	1995-2009	-0.845	0.197	-0.309
	upgradient of ECD at southeast corner Nordic main	95N16E	3.86	1995-2009	-0.825	0.705	-0.648
		95N16C	11.03	1995-2009	-0.986	0.810	-0.628
		95N16A	18.21	1995-2009	-0.848	0.498	-0.269
	upgradient of ECD at south perimeter Nordic main	95N4B	5.31	1995-2009	-0.839	-0.104	-0.452
		95N4A	9.91	1995-2009	-0.601	0.687	0.433
	downgradient of ECD, south of M-14; adjacent to ECA-131	95N12B	3.67	1995-2009	-0.560	-0.537	-0.707
		95N12A	6.87	1995-2009	-0.108	-0.274	-0.084
	downgradient of ECD, south of 95N-12	95N11	4.34	1995-2009	-0.699	-0.454	0.714
	downgradient of ECD south of 95N-13	M12-9	2.5	1994-2009	-0.797	0.781	-0.270
		M12-6	5.49	1993-2009	-0.868	0.804	-0.857
		M12-3	6.54	1993-2009	-0.775	0.587	-0.524
		M12-1	13.41	1993-2009	0.061	0.831	0.857
	downgradient of ECD south of M-12	M13-9	2.04	1993-2009	-0.742	-0.150	-0.719
		M13-6	5.46	1993-2009	-0.938	0.736	-0.967
		M13-3	6.43	1993-2009	-0.248	0.760	-0.733
		M13-1	11.46	1994-2009	0.110	0.219	-0.669
	downgradient of ECD south of M-13; west of historic precipitate pond	M14-9	1.8	1998-2009	-0.575	0.282	-0.975
		M14-6	3.84	1998-2009	-0.400	-0.604	-0.600
		M14-1	8.75	1998-2009	0.310	0.096	-0.359
		M14-3	12.83	1998-2009	-0.690	-0.222	-0.900

decreasing trend, significant at $p<0.05$

increasing trend, significant at $p<0.05$

[a] Trends were not assessed for acidity because a change in analytical technique in 2006 meant that the data were not comparable before and after that time.

[b] Based on rank correlation coefficients (rho) for common (combined) season trends, shown in table.

[c] This is the earliest year included in the trend analysis, but not all stations have data going back to 1993.

porewater has been near neutral since the step change but pH at other depth horizons has remained slightly acidic (Figure 3.31).

3.8.4 Groundwater Quality

Groundwater quality is monitored annually at several locations down-gradient of the Nordic TMA (Figure 3.30; Table 3.24) to assess the effectiveness of measures to remediate the plume migrating south from the Main Tailings Basin:

- The eastern perimeter and north east corner of the Nordic Main TMA are monitored at stations 95N-14, 95N-17 and 95N-7;
- The southern perimeter of the TMA, up gradient of the Effluent Collection Ditch (EDC), is monitored at stations 95N-4, 95N-13 and 95N-16; and
- Groundwater down-gradient of the Nordic Main TMA and ECD is monitored at wells M-12, M-13, M-14, 95N-12 and 95N-11.

Along the eastern perimeter of the Nordic TMA, groundwater pH has been decreasing over time with significant trends in the shallower groundwater at 95N7 (north east corner) and 95N17 (eastern perimeter; Table 3.26; Appendix Figures C.7.21 and C.7.22), although pH is near neutral along the eastern perimeter stations (95N17 and 95N14; Appendix Figure C.7.32; Appendix Tables C.7.21 and C.7.23) Iron increased in the deeper groundwater (12 m) along the eastern perimeter at 95N17A (Table 3.26; Appendix Figure C.7.31) although iron concentrations remain low (<5.0 mg/L).

Along the southern perimeter of the Nordic Main (95N-4, 95n-13 and 95N-16), groundwater quality has been improving over time indicating that the oxidation processes may have peaked and loadings are decreasing. At all three wells and at all depths, iron concentrations have decreased over time (1995 to 2009; Table 3.26). The most dramatic reduction in iron concentrations has occurred in the shallow (<5m) and mid depth (10m) wells (Appendix Figures C.7.19 to C.7.20 and C.7.25 to C.7.30). Consistent with the decrease in iron concentrations, pH levels have significantly increased in these same wells and are now near neutral along the southern perimeter (Table 3.26; Appendix Figures C.7.19, C.7.26, C.7.27, C.7.29 and C.7.30). Some improvements in sulphate concentrations have been noted as well, which are likely associated with lower oxidation of tailings (Table 3.26; Appendix Figures C.7.26, and C.7.30).

Remedial measures have been undertaken down-gradient of the Nordic Main TMA and ECD in order to reduce Nordic groundwater seepage to Buckles Creek. In 1994 the ECD was

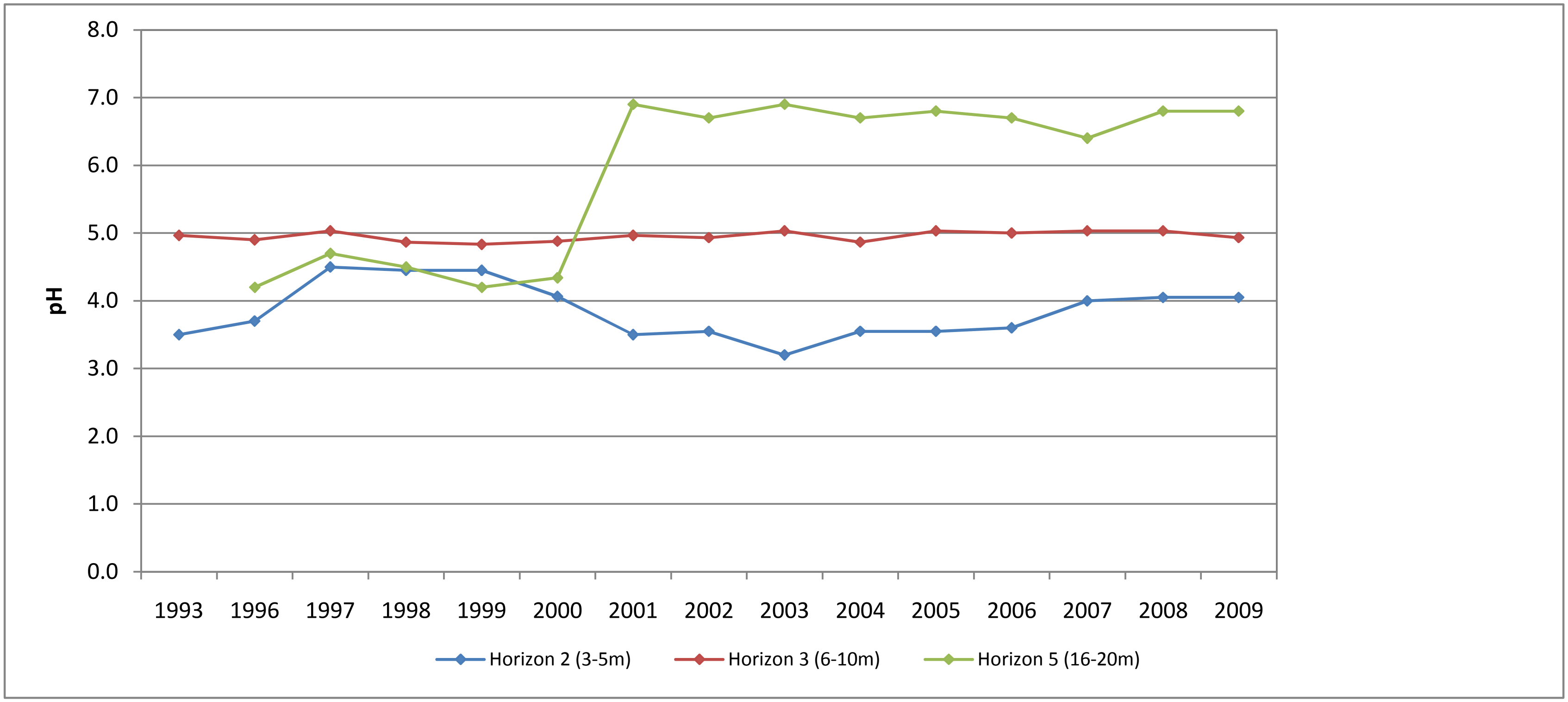


Figure 3.31: Comparison of porewater mean pH at various depths, Lacnor/Nordic TMA, 1993-2009.

Horizon 2 - UW7-4, UW9-3
Horizon 3 - UW7-2, UW9-2, UW9-1
Horizon 5 - UW7-6

lowered and in 1997 the Settling Pond was also lowered (0.6 m) to improve interception of porewater from the tailings and reduce seepage to Buckles Creek located immediately east and south of the Nordic TMA. These measures proved effective in improving groundwater quality down-gradient of the ECD, with significant reductions in iron and commensurate increases in pH at most locations (Table 3.26; Appendix Figures C.7.12-C.7.18 and C.7.23-C.7.24). Review of routine monitoring data including groundwater elevations and chemistry and the chemistry in Buckles Creek indicated that the ECD has effectively been capturing seepage from the TMA and shallow groundwater (EcoMetrix 2011e; Appendix I). In addition, sulphate concentrations have decreased in several wells over time (1993-2009; Table 3.26; Appendix Figures C.7.14, C.7.16 and C.7.17). Sulphate concentrations increased at one well ((M-12-1); however, this trend appears to be leveraged by one 1995 value (Appendix Figure C.7.12).

3.8.5 Treatment Performance

Effluent from the Nordic and Lacnor TMAs is treated at N-17 (ETP influent) and released at N-19, the compliance point for effluent treatment. Surface water affected by the Nordic and Lacnor TMAs, as well as Buckles Creek wetland (e.g. historical tailing deposit) is monitored downstream of N-19 at N-12, which flows into Nordic Lake (Figure 3.30). The ETP at Nordic uses lime to neutralize acidity and reduce metals (predominantly iron). Barium chloride is not required at the Nordic ETP because radium is co-precipitated with the iron hydroxides formed by lime addition. Total annual lime consumption has remained relatively stable over past five years with lower consumption rates (mg/L) observed during peak flow years (e.g. 2008; Figure 3.32).

Following treatment, effluent quality is monitored at the outlet of the Nordic Settling Pond (N-19). Over the past five years effluent quality has consistently achieved discharge criteria (Figure 3.33; Appendix Table C.7.1). Effluent has also been consistently non-lethal to *Daphnia magna* and rainbow trout with no mortality reported in semi-annual acute toxicity tests (Table 3.27). Similarly, survival and reproduction of *Ceriodaphnia dubia* was not affected by exposure to 100% effluent in any tests conducted over the past five years (Table 3.27).

3.8.6 Summary

Surface water quality has improved in all areas of the Lacnor/Nordic TMA with decreasing concentrations in acidity, radium-226 and sulphate. The improvements are the result of remedial measures implemented at the TMA and presumed lower oxidation rates within the

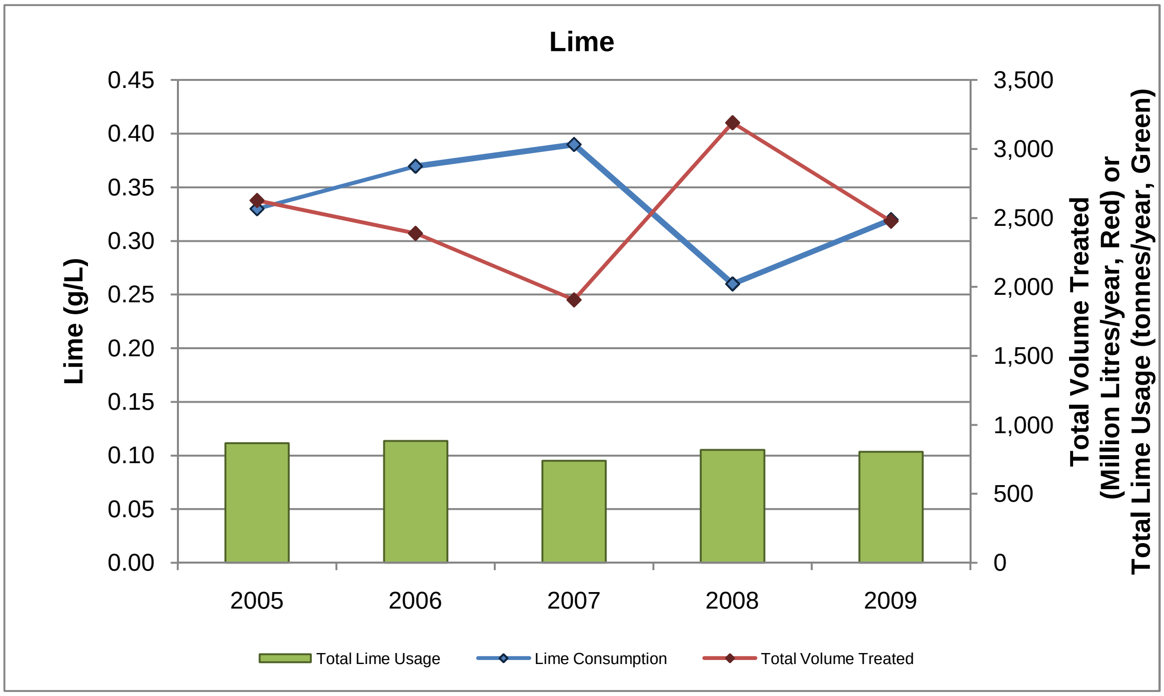


Figure 3.32: Comparison of total reagent consumed versus total volume treated at Nordic TMA from 2005-2009.

Figure 3.33: Effluent concentrations versus monthly average discharge criteria at Nordic TMA station N-19.

Table 3.27: Toxicity test results from samples collected at Lacnor/Nordic TMA station N-12, 2005 - 2009.

Sample Date (month-year)	Acute Toxicity (% mortality)		Survival and Reproduction (IC25[d] as % effluent)
	Daphnia magna [a]	rainbow trout [b]	*Ceriodaphnia dubia* [c]
May-05	0	0	100
November-05	0	0	100
May-06	0	0	100
November-06	0	0	100
May-07	0	0	100
November-07	0	0	100
May-08	0	0	100
November-08	0	0	100
May-09	0	0	100
November-09	0	0	100

[a] *Daphnia magna* 48-hr LC50 test (Environment Canada 2000a).

[b] Rainbow trout 96-hr LC50 test (Environment Canada 2000b).

[c] *Ceriodaphnia dubia* survival and reproduction test (Environment Canada 2007).

[d] Effluent concentration causing 25% inhibition relative to control organisms.

tailings. Porewater associated with the Nordic West Arm has either been stable or improved as indicated by decreasing iron concentrations and increasing pH levels. Groundwater down-gradient of the Nordic Main Basin has also significantly improved, reflecting remediation efforts in the ECD and settling pond and lower oxidation rates within the tails. In the past five years treated effluent consistently achieved discharge criteria and all tests to *Daphnia magna*, rainbow trout and *Ceriodaphnia dubia* were non-toxic. Overall, the Lacnor/Nordic TMA is performing well and conditions are improving over time.

3.9 Pronto TMA

3.9.1 Basin History and Modifications

The Pronto TMA is located on the north side of Highway 17, approximately 10 km east of Blind River. The Pronto mine operated from 1955 to 1960 and, over that period, the Pronto mill processed approximately 2.1 million tonnes of uranium ore. In 1960, the mill was converted to process copper ore from an adjacent mine and, from 1960 to 1970, produced approximately 2 million tonnes of copper tailings. In 2009, approximately 33,000 tonnes of rock fill from adjacent residential properties were relocated to the Pronto TMA. Tailings are located in a 47-hectare natural rock basin contained by Dam A, constructed of a glacial till core with a waste rock shell (Figure 3.34).

A high water table (close to the surface) at the Pronto TMA, serves to reduce acid generation (Rio Algom 2000). However, in the eastern portion of the TMA the saturation extended to surface which precluded traditional direct liming and seeding and as such a successful vegetative cover could not be maintained. To resolve this problem, rock-lined drainage ways were installed in the eastern portion of the TMA during the winter of 1999-2000. Then six tonnes/ha of limestone and 500 kg/ha fertilizer were applied to bare areas in the spring of 1999 and a 30-cm depth of biosolids (in the form of paper mill sludge) were spread over a 20.9-ha area from 1999 to 2001. These measures have been effective in maintaining a 100% vegetative cover following program completion in 2001.

The East and West Collection Ditches, upgraded in 1999, direct seepage and runoff from the TMA into the Holding Pond. Water from the Holding Pond is directed through the Pronto treatment plant at a rate of 100 to 200 L/s which operates for two to four months per year. Lime and barium chloride are added in the treatment plant to promote metals and radium precipitation in the Settling Pond prior to release of treated water to the Downstream Pond. The treatment plant, originally constructed in 1971, was upgraded in 1979 and 1993 prior to being replaced with the current structure in 1997. Dam F was constructed at the west end of

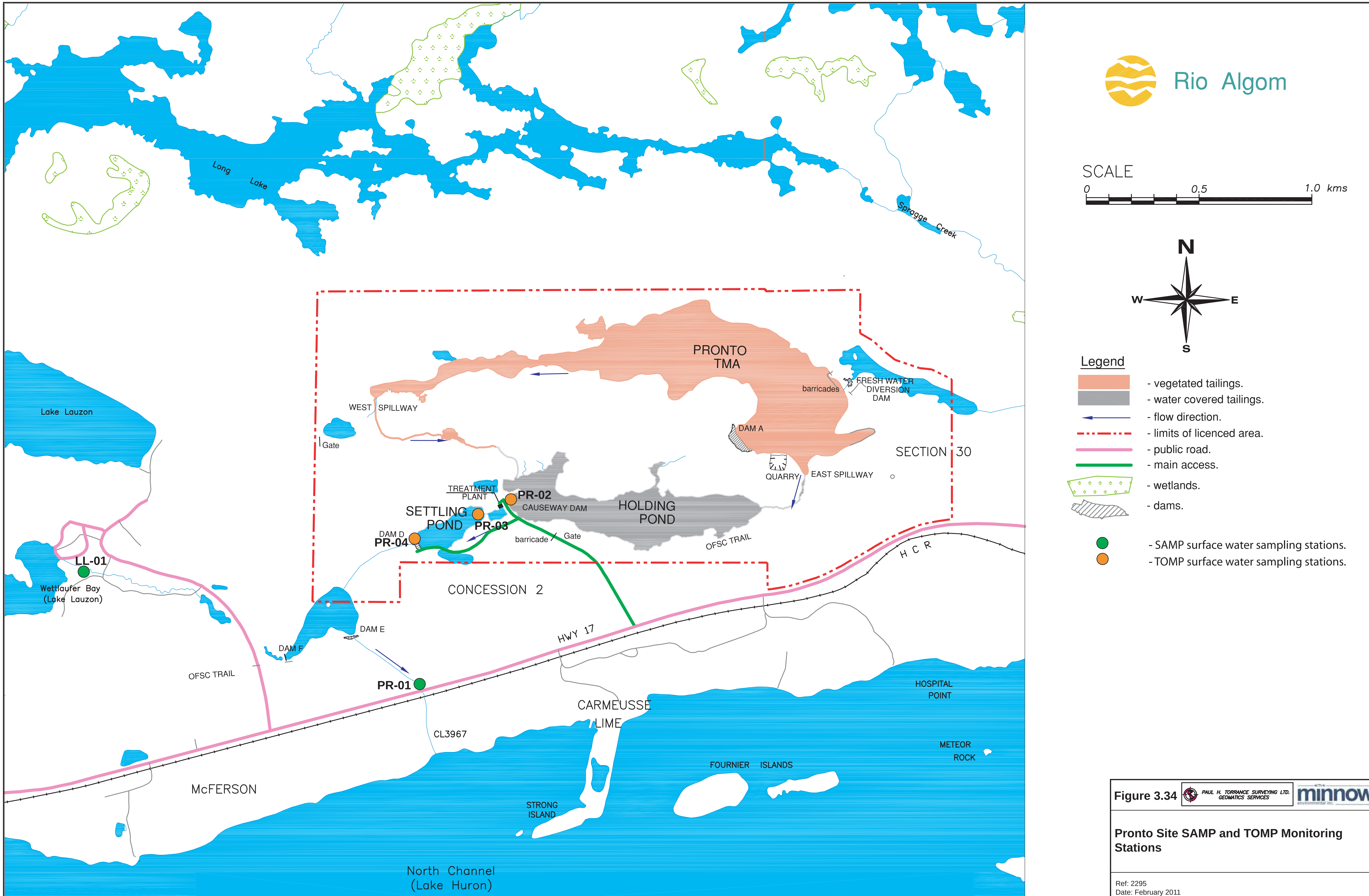


Figure 3.34

Pronto Site SAMP and TOMP Monitoring Stations

Ref: 2295
Date: February 2011

the Downstream Pond in the late 1970s and upgraded in 1998 and 2007 to direct flow away from Lake Lauzon towards Lake Huron via the Pronto Discharge Channel (Figure 3.34).

Within the TMA, surface water is monitored under the TOMP and the locations, substances and frequency monitored are specific to the station type (Table 3.28; Figure 3.34) Data from the TOMP stations are summarized in the following sections and presented in Appendix C (Appendix Tables C.8.2 – C.8.4).

3.9.2 Water Elevations

Operating elevations in the Holding Pond were established to ensure adequate storage capacity to contain and treat "the Timmins Storm" (193 mm in 12 hrs; elevation 196.5 m), and also provide adequate water cover to prevent freeze-up of the influent pipe (elevation 197.7 m). The water levels within the Holding Pond at the Pronto TMA are monitored regularly at PR-02 and have been maintained within the operating limits during routine operations (Figure 3.35). The Holding Pond has been drawn down below normal operating elevations on several occasions to facilitate construction activities including the treatment plant replacement in 1997, Causeway Dam upgrading in 1998, and the ATV trail re-routing in 2006.

3.9.3 Basin Surface Water Quality

Surface water quality at the Pronto TMA is monitored at three stations to assess conditions downstream of the tailings deposition area (Figure 3.34):

- Seepage and surface runoff for the Pronto TMA are captured in the Holding Pond which is monitored at PR-02 (Appendix Table C.8.2);
- pH within the ETP is monitored at PR-03 (Appendix Table C.8.3); and
- Final effluent is monitored at the outlet of the Pronto Settling Pond (PR-04; Appendix Table C.8.4).

Over the past twenty years, concentrations of radium-226 and uranium as well as pH levels have remained relatively stable, while some reduction in sulphate was observed in the past ten years (Figure 3.36). Similarly, over the past seven years, there were no significant trends detected in surface water with the exception of barium concentrations in final effluent associated with reductions in barium chloride use in the ETP (Table 3.29).

Table 3.28: TOMP monitoring stations, substances, and frequencies[a] at Pronto TMA.

		Parameters and Frequencies[a]										
TOMP Stations	**Station Type/Purpose**	**Elevation**	**Flow**	**pH**	**Sulphate**	**Total radium-226**	**Lime or NaOH Consumption**	**Barium Chloride Consumption**	**TSS**	**Acidity**	**Iron**	**SAMP Metals[b]**
PR-02[c]	Basin performance (primary), ETP operations	W	D	M	Q	M	M	M		Q		Q
PR-03[c]	ETP operations			D								
PR-04[c]	Effluent		W	W	M	W			W			M

[a] D - Work days, W - Weekly, M - Monthly, S - Semi-annually, A - Annually, Q-Quarterly

[b] SAMP metals are barium, cobalt, iron, manganese, and uranium

[c] Sampled when treatment plant is operating.

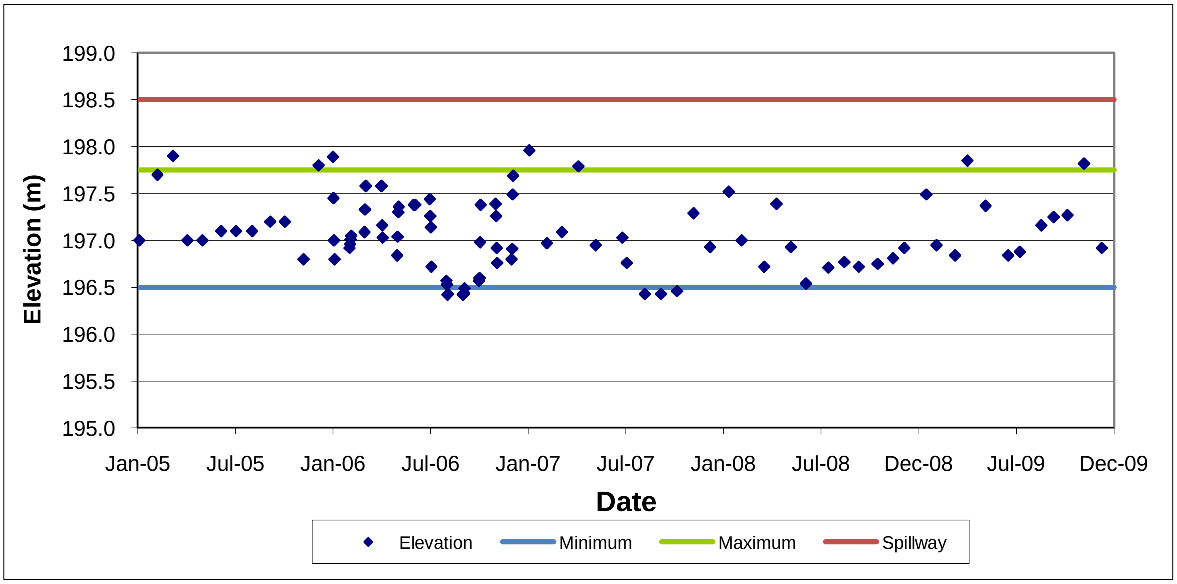


Figure 3.35: Water level at PR-02 relative to minimum operating elevation.

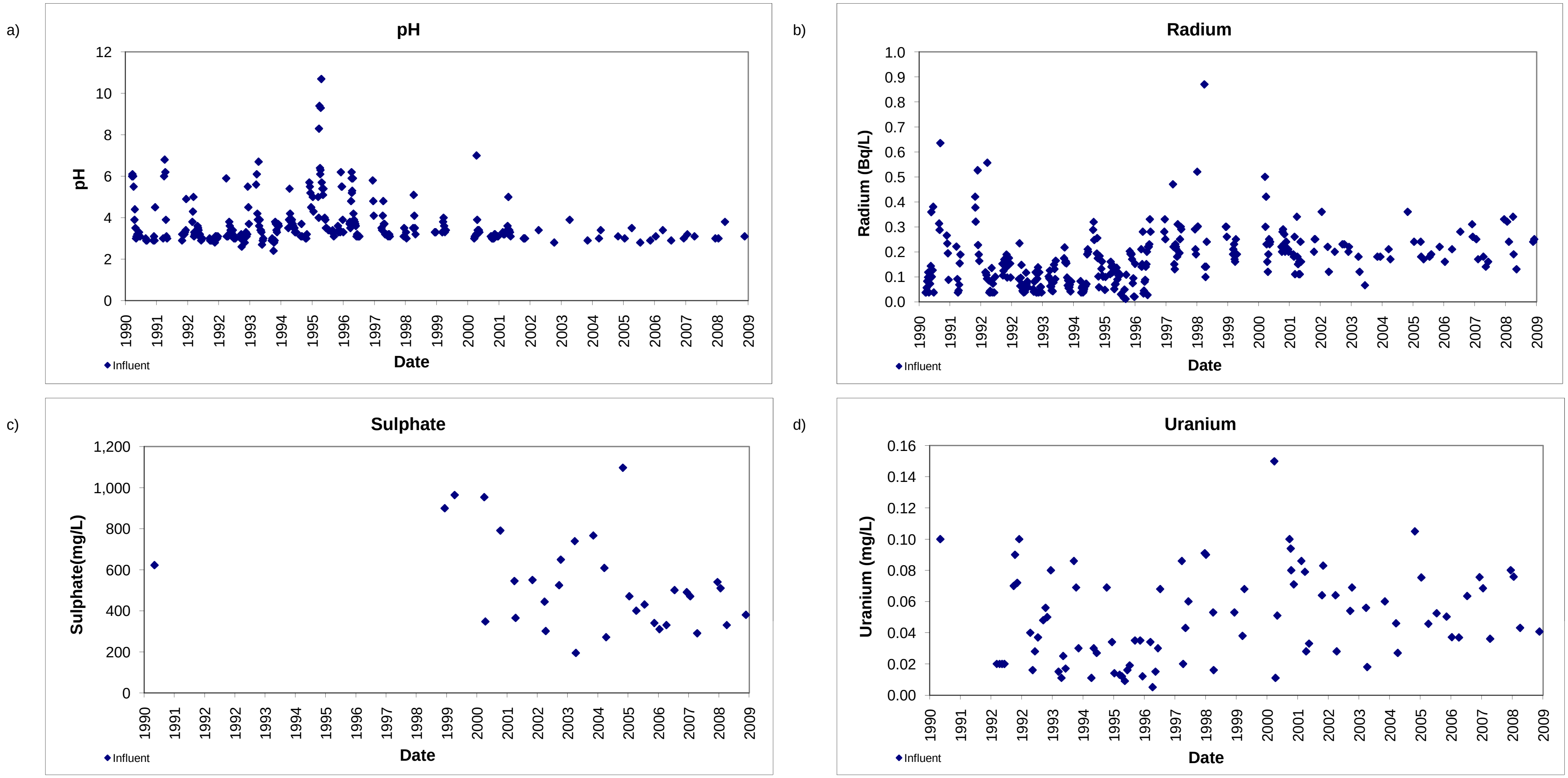


Figure 3.36: Water quality at the influent (PR-02) of the Pronto TMA treatment plant.

Table 3.29: Summary of water quality trends[a] for TOMP monitoring stations, Pronto TMA, 2003 to 2009.

Station ID	Type/Location	Number of Seasons Used in Common Trend[c]	Acidity	Barium	Cobalt	Iron	Manganese	pH	Radium-226	Sulphate	Uranium
PR-02	Treatment Plant Influent	3 to 6	-0.440	0.447	0.381	0.405	0.214	0.299	0.286	-0.094	0.298
PR-04	Final Treated Effluent	4 to 6	-[b]	-0.821	0.170	-0.267	-0.058	0.354	0.041	-	0.047

decreasing trend, significant at $p<0.05$

increasing trend, significant at $p<0.05$

[a] Based on rank correlation coefficients (rho) for common (combined) season trends, shown in table.

[b] "-" denotes that this parameter was not included in the trend analysis for that particular station due to insufficient data (e.g. there were <5 years worth of data for that parameter)

[c] Seasons used varied for substances based on suitability of data for trend analysis.

3.9.4 Treatment Performance

Effluent from the Pronto TMA is treated in an ETP downstream of the Holding Pond (PR-02) (ETP influent) and released at PR-04, the compliance point for effluent treatment. Treatment has included both lime and barium chloride to reduce acidity and radium-226 respectively. However, since 2005, the ETP has been reducing its barium chloride use and in 2009 it was not used in the treatment process because co-precipitation with lime was sufficient to reduce radium-226 levels to less than the discharge criterion (Figure 3.37; Table 3.29). The lime consumption rate has remained stable during the reporting period (Figure 3.37).

Following treatment, effluent quality is monitored at the outlet the Settling Pond (PR-04) and over the past five years effluent quality has consistently achieved discharge criteria (Figure 3.38; Appendix Table C.8.1). One iron concentration in a single grab sample exceeded the action limit of 1.0 mg/L and triggered implementation of a response plan that resulted in compliance with discharge criteria. Effluent has been consistently non-lethal to *Daphnia magna* and rainbow trout with no mortality reported in semi-annual acute toxicity tests (Table 3.30). Similarly, survival and reproduction of *Ceriodaphnia dubia* was not affected by exposure to 100% effluent in any tests conducted over the past five years (Table 3.30).

3.9.5 Summary

Water levels within the Holding Pond have been maintained above the minimum operating levels. Surface water quality has been consistent over time with the exception of decreasing barium concentrations in TMA effluent associated with reductions in barium chloride use in the ETP. The TMA has been reducing barium chloride use over the past five years with no barium chloride used in 2009, because lime precipitation proved adequate to reduce radium-226 concentrations below the discharge criterion. In the past five years treated effluent consistently achieved discharge criteria and all tests to *Daphnia magna*, rainbow trout and *Ceriodaphnia dubia* were non-toxic.

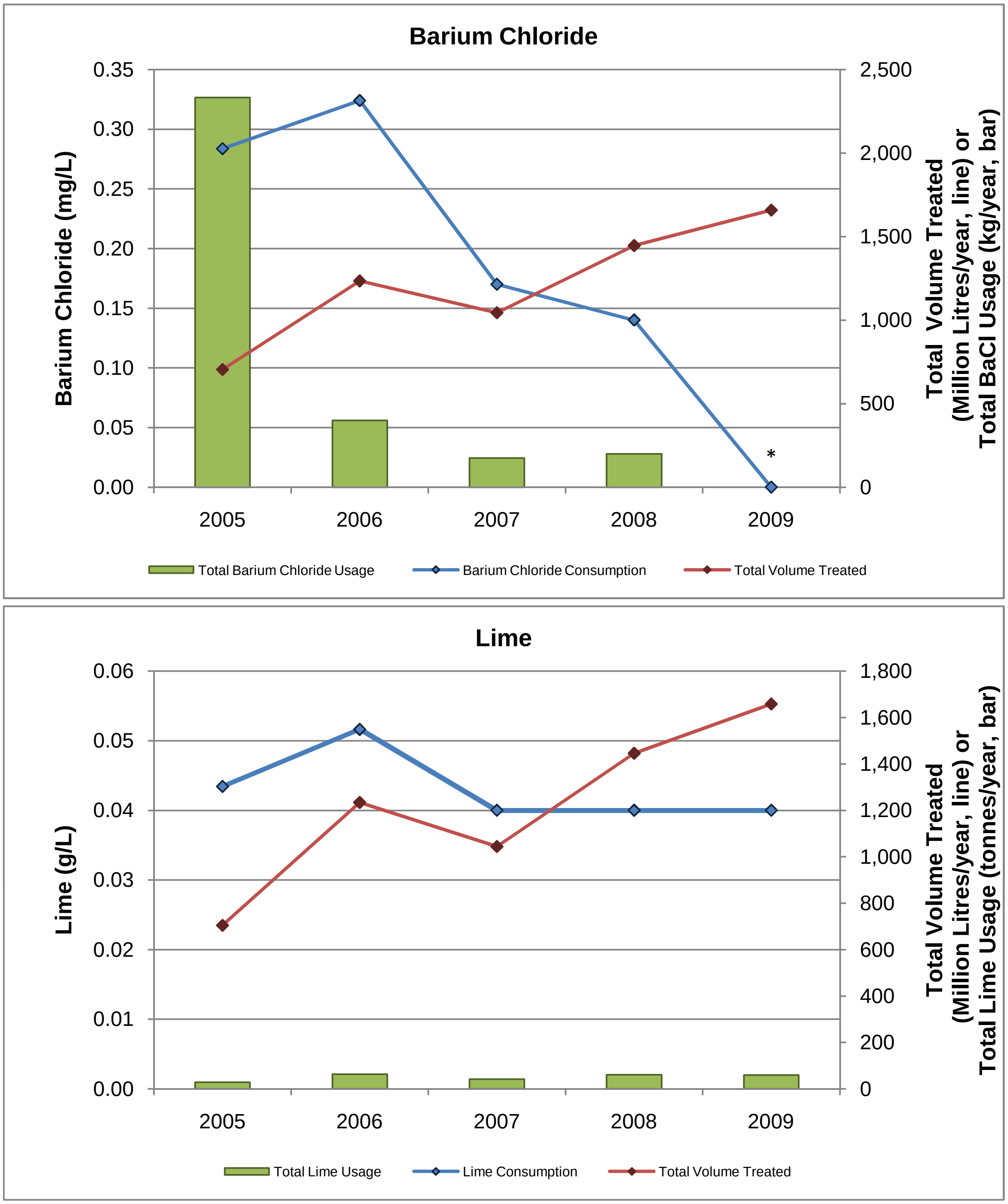


Figure 3.37: Comparison of total reagent consumed versus total volume treated at Pronto TMA from 2005-2009 (* no BaCl used in 2009).

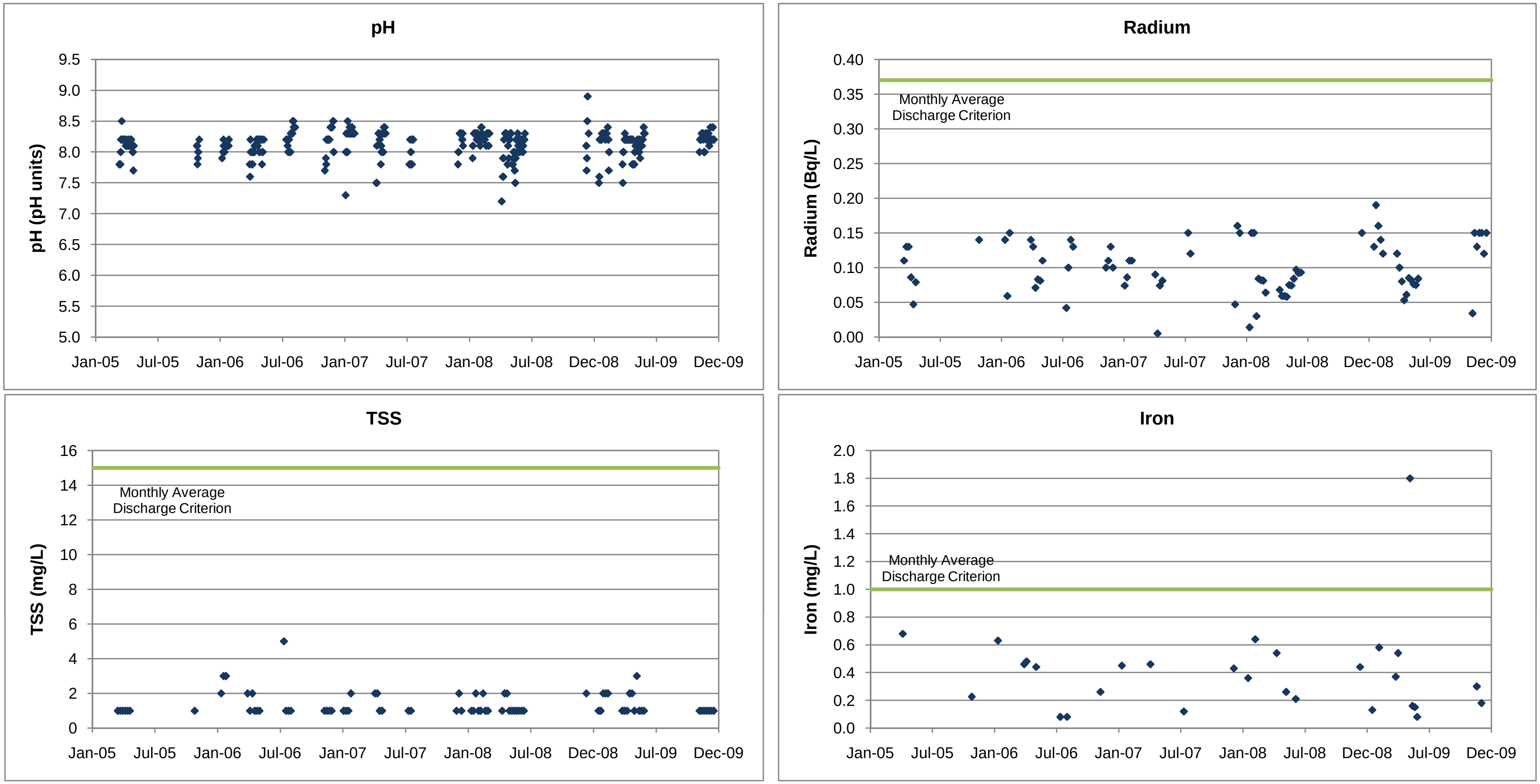


Figure 3.38: Effluent concentrations versus monthly average discharge criteria at Pronto TMA station PR-04.

Table 3.30: Toxicity test results from samples collected at Pronto TMA station PR-01, 2005 - 2009.

Sample Date (month-year)	Acute Toxicity (% mortality)		Survival and Reproduction (IC25[d] as % effluent)
	Daphnia magna [a]	rainbow trout [b]	*Ceriodaphnia dubia* [c]
April-05	0	0	100
October-05	0	0	100
April-06	0	0	100
November-06	0	0	100
April-07	0	0	100
December-07	0	0	100
May-08	0	0	100
December-08	0	0	100
April-09	0	0	100
November-09	3	0	100

[a] *Daphnia magna* 48-hr LC50 test (Environment Canada 2000a).

[b] Rainbow trout 96-hr LC50 test (Environment Canada 2000b).

[c] *Ceriodaphnia dubia* survival and reproduction test (Environment Canada 2007).

[d] Effluent concentration causing 25% inhibition relative to control organisms.

4.0 SOURCES TO THE WATERSHED

Mine releases to watershed, including effluent, seepage and site runoff are captured through the Source Area Monitoring Program (SAMP; Table 4.1). Data for each discharge are presented in Appendix D. Results are discussed below on a sub-watershed basis so that mine sources to the watershed may be considered cumulatively. Concentrations within mine releases have been compared to receiving water benchmarks[4] for the Serpent River Watershed (SRW). While mines sources are generally not expected to achieve standards for receiving environment quality, comparisons were made because in many instances mine effluents are at or approaching these standards. Based on watershed area ratios, a minimum 10-fold dilution is expected downstream of the mine discharges. Thus, a concentration of 10x the appropriate receiving environment criterion is a more relevant comparison for discharges and such values are also discussed as appropriate. Trend analysis was conducted on SAMP data since the inception of the program (2003 to 2009) to determine substances and locations reflecting statistically significant changes in concentrations.

4.1 Quirke Lake Sub-watershed Sources

Within the Quirke Lake sub-watershed there are primary (effluent) and secondary (seepage/runoff) discharges from three TMAs (Denison, Quirke and Panel; Figure 4.1) In addition, seepage form the Stanrock TMA also discharges to Quirke Lake, resulting in four TMA sources to the Quirke Lake sub-watershed. As part of the SRWMP, water quality is monitored both upstream and downstream of these sources (Figure 4.1).

4.1.1 Discharge Quality and Loads

With few exceptions, mean mine discharge concentrations (2005-2009) of cobalt, iron, manganese, radium-226, sulphate and uranium achieved PWQO or were less than 10 times PWQO in mine sources (Figure 4.2). Concentrations of barium and sulphate tended to be highest in the primary discharges while concentrations of metals (Co, Fe, Mn and U) were highest and pH lowest in secondary discharges (seepages) (Figure 4.2). The seepages with the highest concentrations were ECA 398 (cobalt, uranium and pH), DS-16 (cobalt, manganese), D-9 (cobalt, iron, and manganese), D-16 (manganese) and Q-23 (pH). While

[4] The Serpent River Watershed benchmarks are based on the upper limit of background or PWQO whichever is higher. For sulphate and manganese the BCMOE guideline was used as there is no PWQO for this substance.

Table 4.1: SAMP stations, parameters and frequencies[a].

TMA	Location	Type	Description	Parameter[b] flow	pH	Sulphate	Radium-226	SAMP metals[c]	toxicity
Denison	D-2[d]	Primary	Stollery Lake Outlet	D	W	M	M	M	2
	D-3[d]	Primary	TMA-2 Effluent at Denison Mine access road	D	W	M	M	M	
	D-9	Seepage	Seepage at Dam 17	Q	Q	Q	Q	Q	
	D-16	Seepage	Seepage at Dam 9	Q	Q	Q	Q	Q	
Quirke	ECA-398	Seepage	Quirke II north of access road	Q	Q	Q	Q	Q	
	Q-22	Drainage	Quirke II Drainage south of access road	Q	Q	Q	Q	Q	
	Q-23	Drainage	Swamp Outlet west of Dam K1	Q	Q	Q	Q	Q	
	Q-27	Seepage	Dam J Toe Seepage		Q	Q	Q	Q	
	Q-28[d,e]	Primary	Final Treated Effluent	W	W	M	M	M	2
Panel	P-02	Seepage	Downstream of Dam B	Q	Q	Q	Q	Q	
	P-03	Drainage	Beaver Pond C Outlet	Q	Q	Q	Q	Q	
	P-05	Drainage	Swamp Outlet north of Dam E		Q	Q	Q	Q	
	P-11	Drainage	Panel Creek Outlet at Quirke Lake	Q	Q	Q	Q	Q	
	P-14[d,e,f,g]	Primary	Final Treated Effluent	W	W	M	M	M	2
Stanrock	DS-4	Primary	Orient Lake Outlet (Final Point of Control)	W	W	M	M	M	2
	DS-16	Drainage	Quirke Lake Delta	Q	Q	Q	Q	Q	
Stanleigh	CL-06[d,e]	Primary	Final Treated Effluent	W	W	M	M	M	2
Milliken	MPE	Primary	Milliken Park Effluent		M	M	M	M	2
Nordic	N-12	Primary	Buckles Creek at Hwy. 108	M	M	M	M	M	2
Pronto	LL-01	Drainage	Pronto Creek at Inlet to Lake Lauzon	Q	Q	Q	Q	Q	
	PR-01	Primary	Pronto Discharge Channel at Highway 17	M	M	M	M	M	2

[a] Frequencies: D =daily, W = weekly, M = monthly, 2 = twice per year, Q = quarterly

[b] DOC and hardness will be added to the SAMP program effective January 2010.

[c] SAMP metals - barium, cobalt, iron, manganese, uranium

[d] This station is also TOMP effluent station and requirements will be harmonized to serve both programs.

[e] Sampled when treatment plant is operating.

[f] P-14 will revert to P-36 upon ETP shut down.

[g] Flow will be based on influent flow to the ETP at P-13.

Rochester Lake
P-22
P-05
P-02
Panel TMA
P-03
P-14
P-11
Panel
Quirke I
Quirke TMA
Q-28
Q-23
Q-27
Evans Lake
Q-20
Q-09
ECA398
Q-22
Quirke II
D-5
D-3
Denison
Denison TMA1 & 2
Quirke Lake
SR01
Dunlop Lake
D-2
D-6
D-4
D-16
D-9
Cinder Lake
Canmet
Spanish American
Stanrock
Stanrock TMA
D-16
Spanish American TMA
Legend
Streams
Mine discharge (SAMP)
Receiving environment (SRWMP)
Tailings Management Areas
Minesites
Roads
Direction of Flow
0
1,000 m
N
Figure 4.1
minnow
Quirke Lake Sub-watershed Mine Source and Receiving Environment Stations
Ref: 2295
Date: February 2011
Source: Elliot Lake Research Field Station

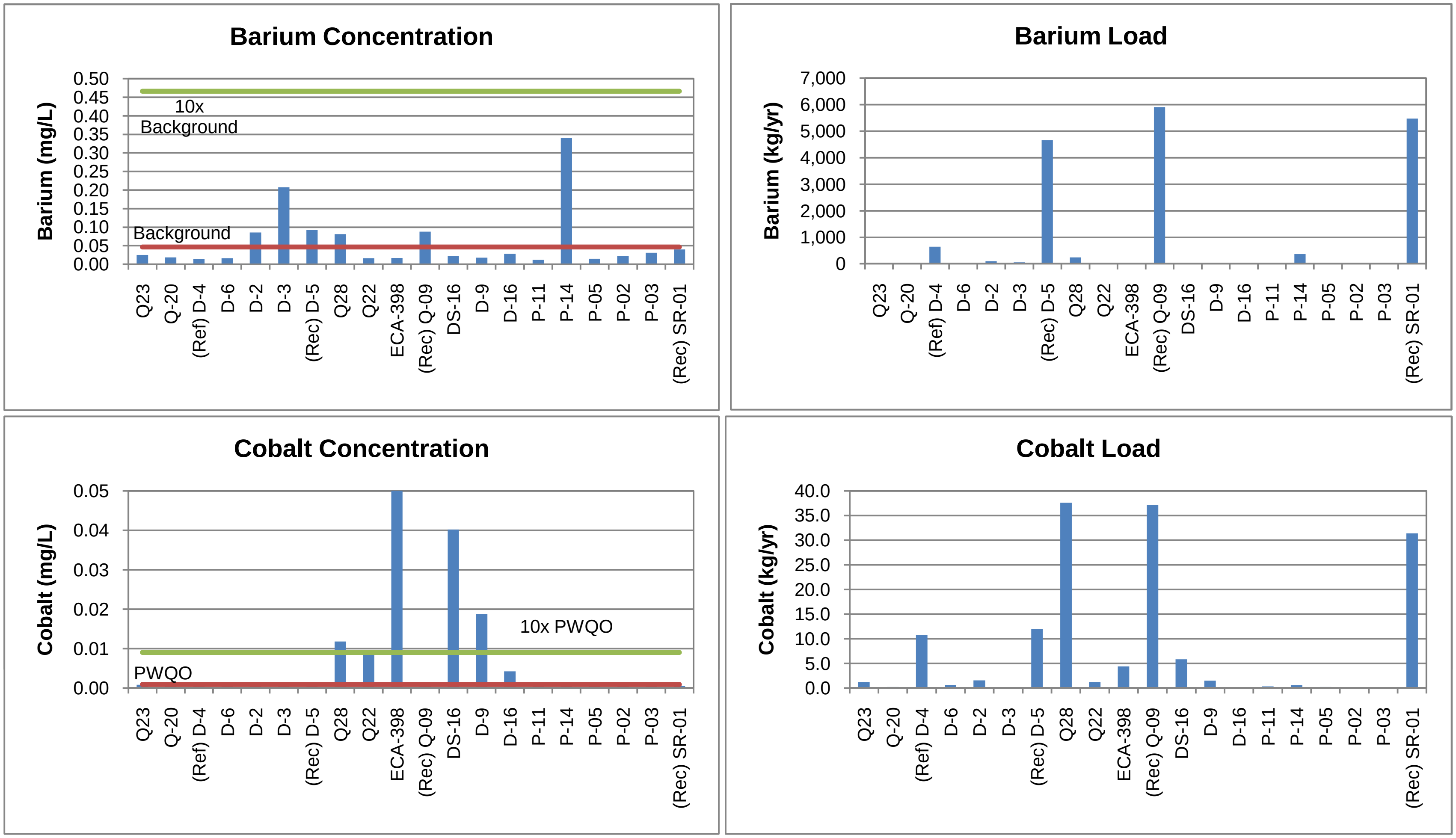


Figure 4.2: Mean concentrations and loads at monitoring stations upstream of Quirke Lake outlet, 2005-2009, (Rec) denotes receiving environment station, (Ref) denotes reference station.

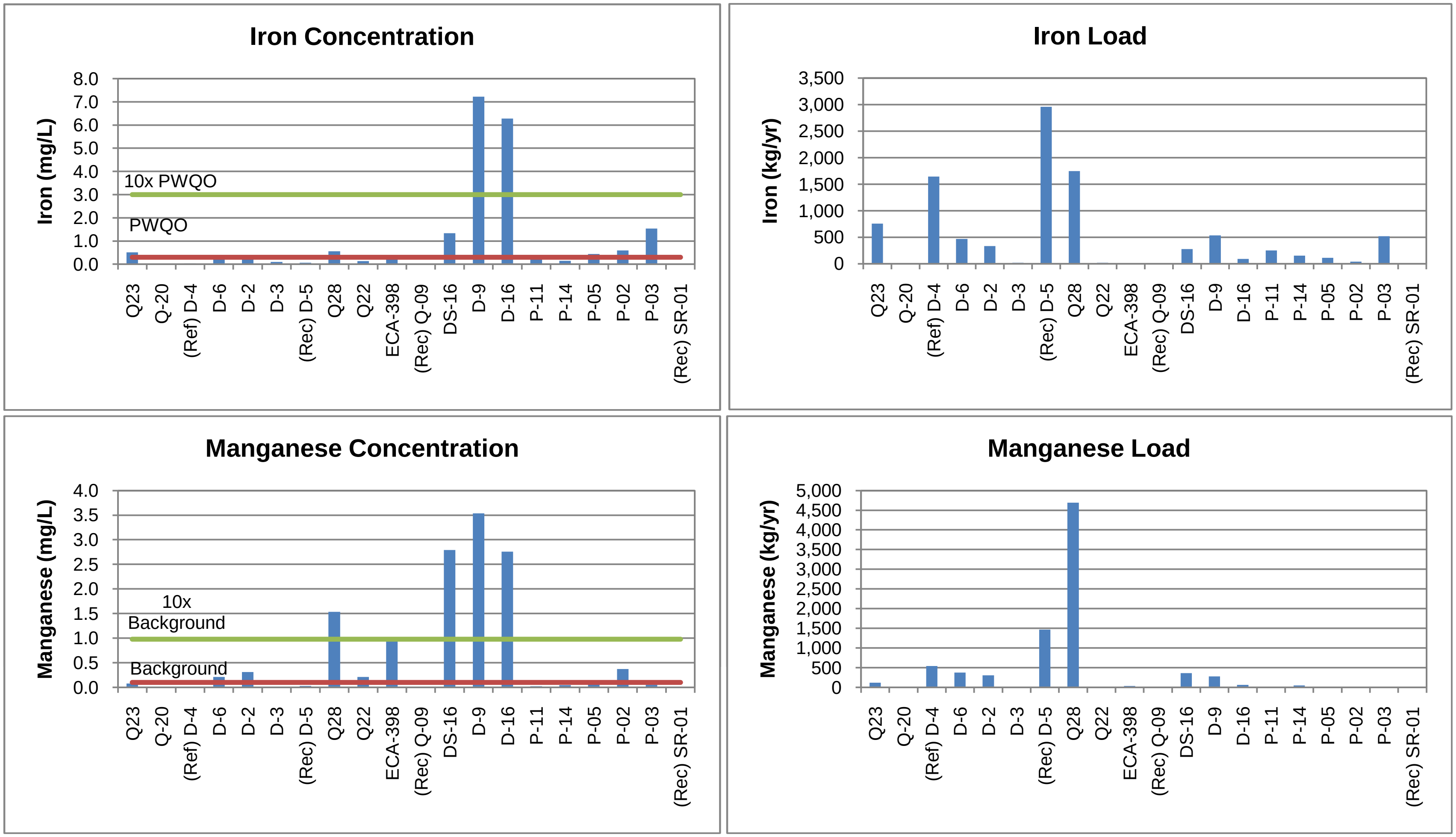


Figure 4.2: Mean concentrations and loads at monitoring stations upstream of Quirke Lake outlet, 2005-2009, (Rec) denotes receiving environment station, (Ref) denotes reference station.

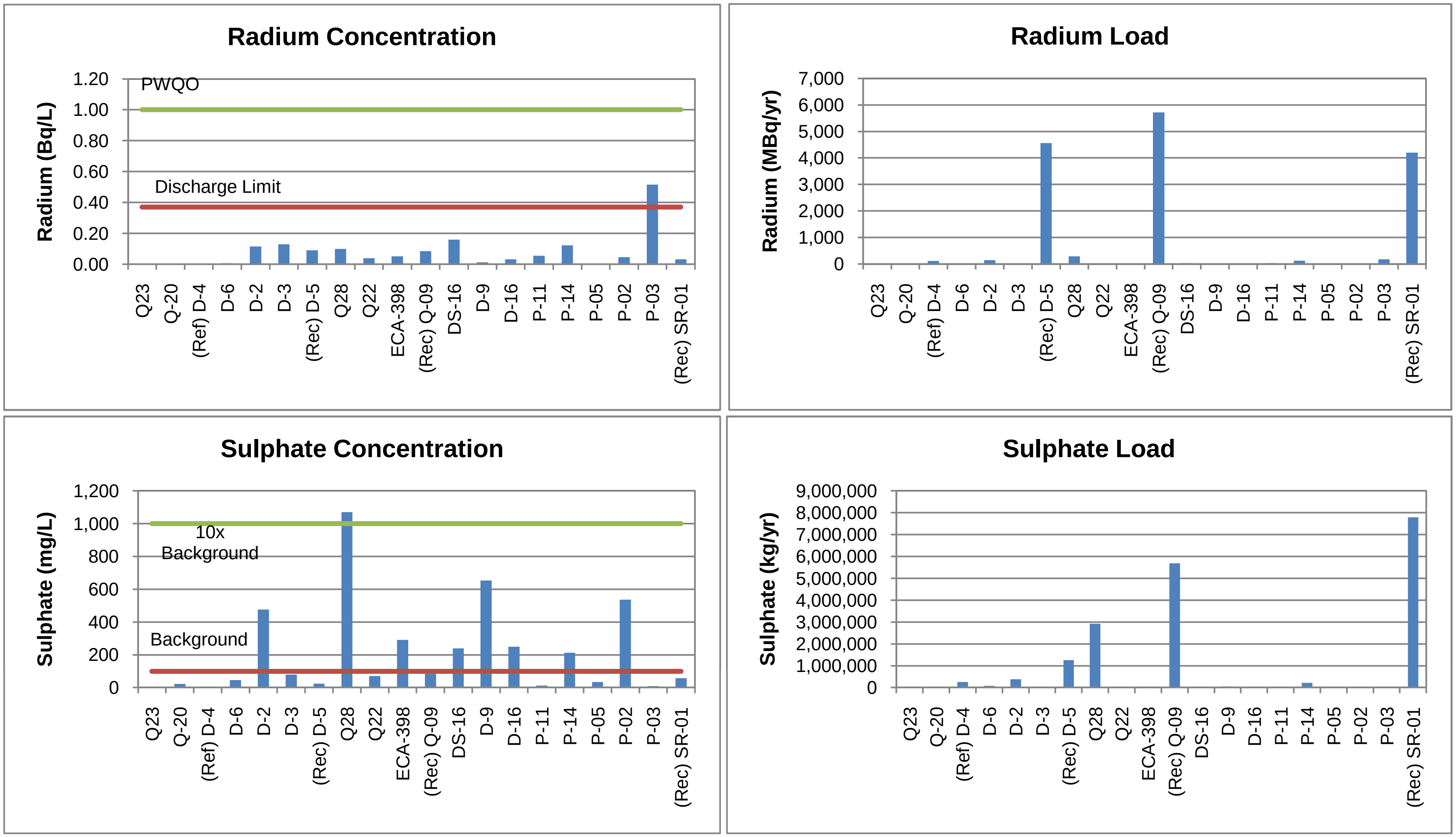
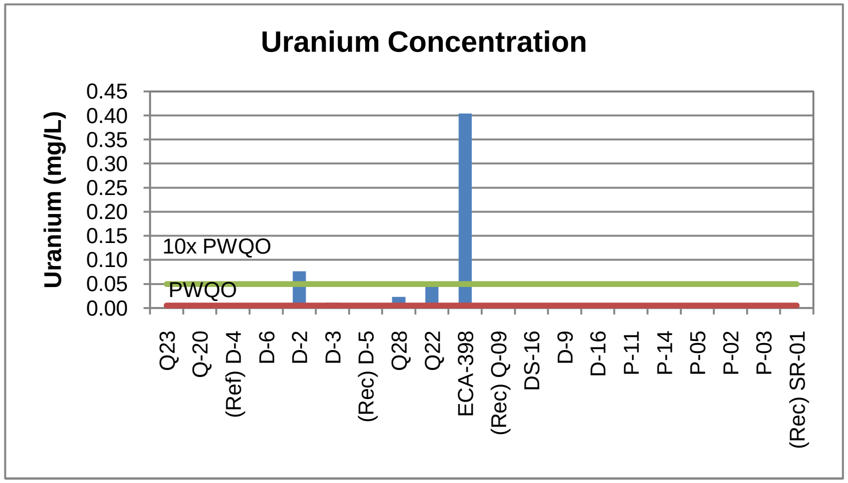
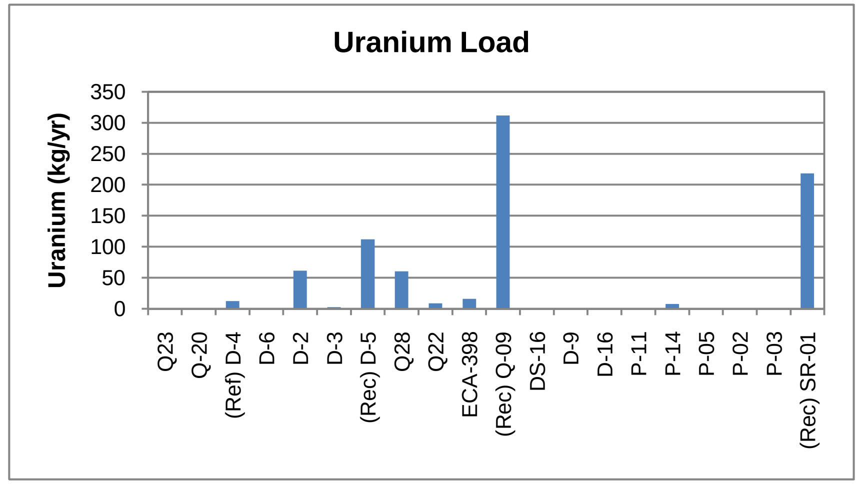


Figure 4.2: Mean concentrations and loads at monitoring stations upstream of Quirke Lake outlet, 2005-2009, (Rec) denotes receiving environment station, (Ref) denotes reference station.





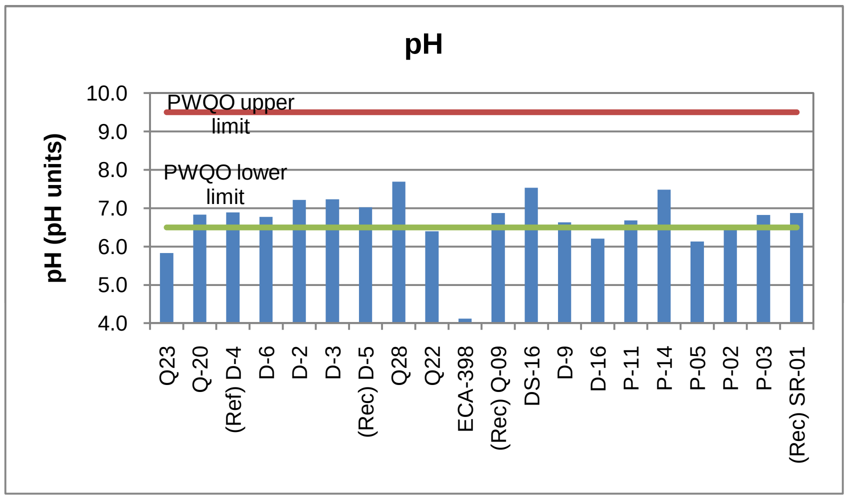


Figure 4.2: Mean concentrations and loads at monitoring stations upstream of Quirke Lake outlet, 2005-2009, (Rec) denotes receiving environment station, (Ref) denotes reference station.

these concentrations were high, the associated loadings contributed to the watershed were low compared to primary discharges and background (upstream) loads (Figure 4.2).

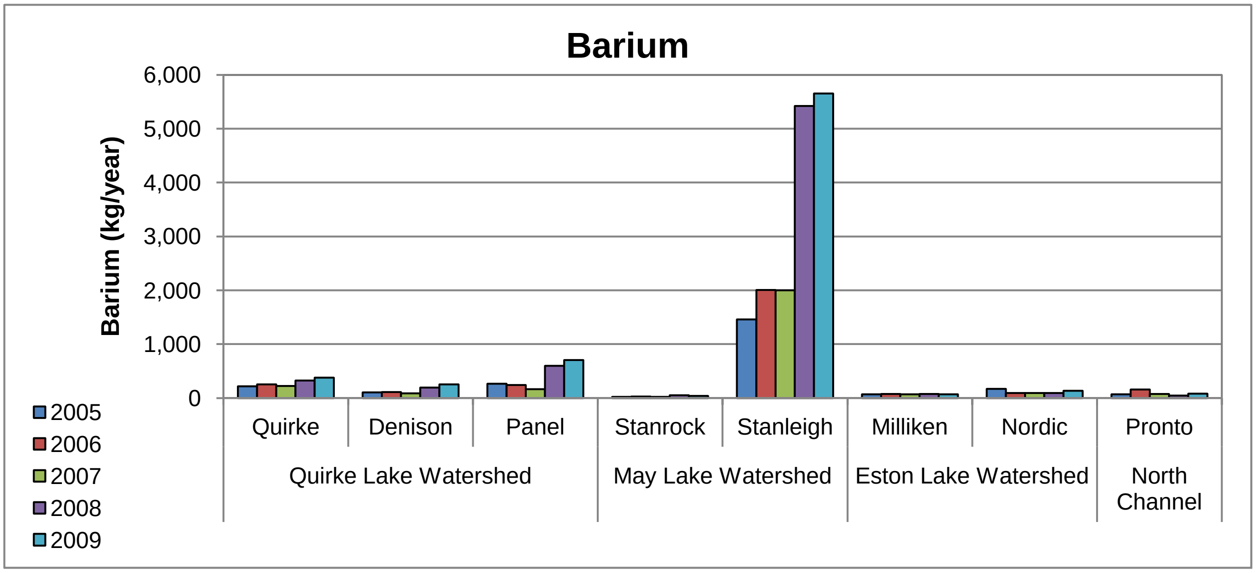
In terms of the relative loadings among TMAs within the Quirke Lake sub-watershed, the Quirke TMA tended to have highest loading of most metals (cobalt, manganese, sulphate and uranium), except for barium and radium-226, for which Panel TMA contributed slightly higher loads (Figure 4.3). Within Quirke TMA, 60 to 80% of annual loads were associated with the primary discharge (Appendix Figure D.2.1). At Panel, over 90% of barium load was from the primary discharge, whereas only 50% of the radium-226 load is from primary discharge with about 30% from Pond C (P-03) (Appendix Figure D.3.1).

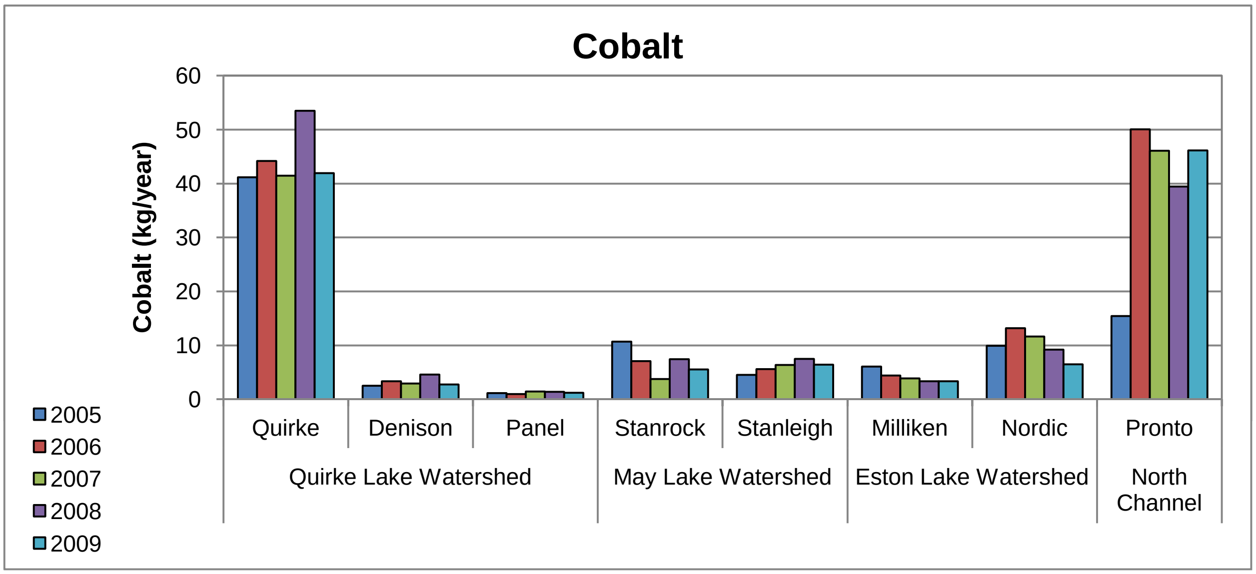
As noted in the previous SOE report (Minnow 2009a), the radium load within the Serpent River downstream of the Denison TMA discharge (D-5) was substantially greater than the loading from the Denison TMA (Figure 4.2) or upstream watershed (D-4) suggesting a radium source within the river. In 2009, EcoMetrix conducted a study to investigate the difference in loadings within the River (Appendix G). Sediment sampling conducted in 2009 found elevated radium-226 concentrations (14 Bq/g) between stations D4 and D5, which indicated a source of radium-226 in the Serpent River. The barium and sulphate depth profiles in sediment and water (porewater and overlying water) mirrored the radium profiles, indicating that these profiles are likely caused by the settling/accumulation of historical treatment solids. Modelling of radium releases (load) based on the diffusion and mass transport of radium from the sediment agreed well with those observed in this report (e.g., the modelled cumulative load was 3,420 MBq/a compared with 3,884 MBq/a calculated in this report), and agreed with loading averages from 2003 to 2006 (Minnow 2009a). These loadings are therefore consistent with the recovery of historically accumulated sediments releasing radium to the water column. Diffusion modelling indicated that radium-226 release from the sediment should decrease with time.

Loadings from all upstream mine sources do not result in concentrations in the receiving environment that are above SRW benchmarks (Figure 4.2).

4.1.2 Source Trends

Cobalt, manganese, sulphate, radium-226 and uranium concentrations have decreased or been stable over the past seven years in all discharges to Quirke Lake (Table 4.2). Barium concentrations increased over time at the primary discharge locations (D2, D-3, P-14 and Q-28) (Table 4.2) largely due to greater barium use in 2008 and/or 2009 in response to increased flows (Figure 4.4).





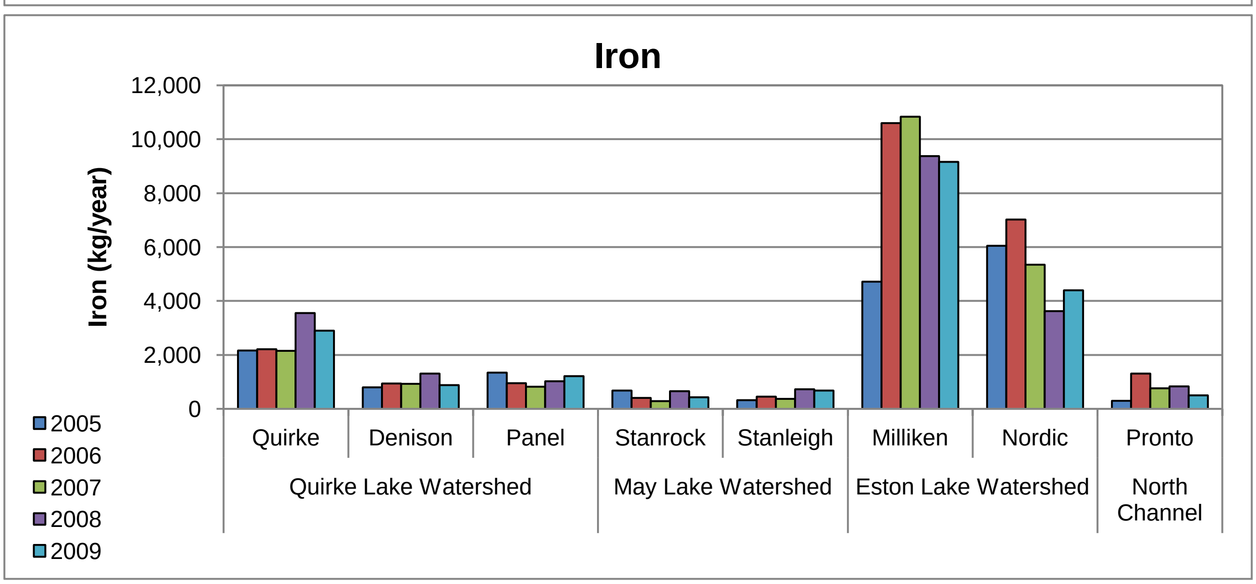
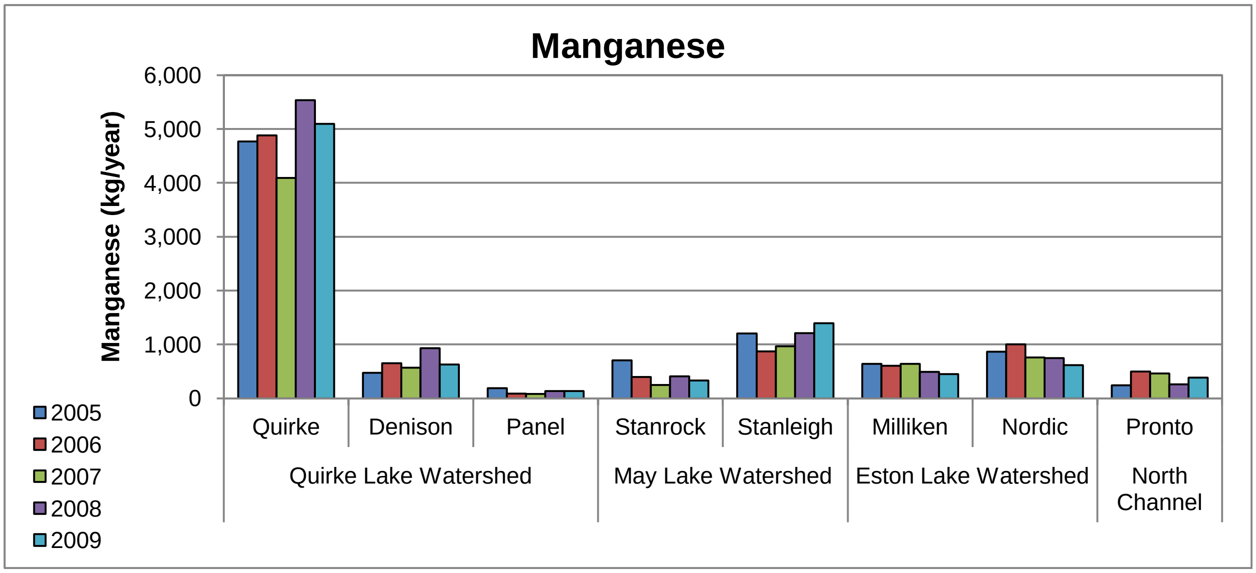
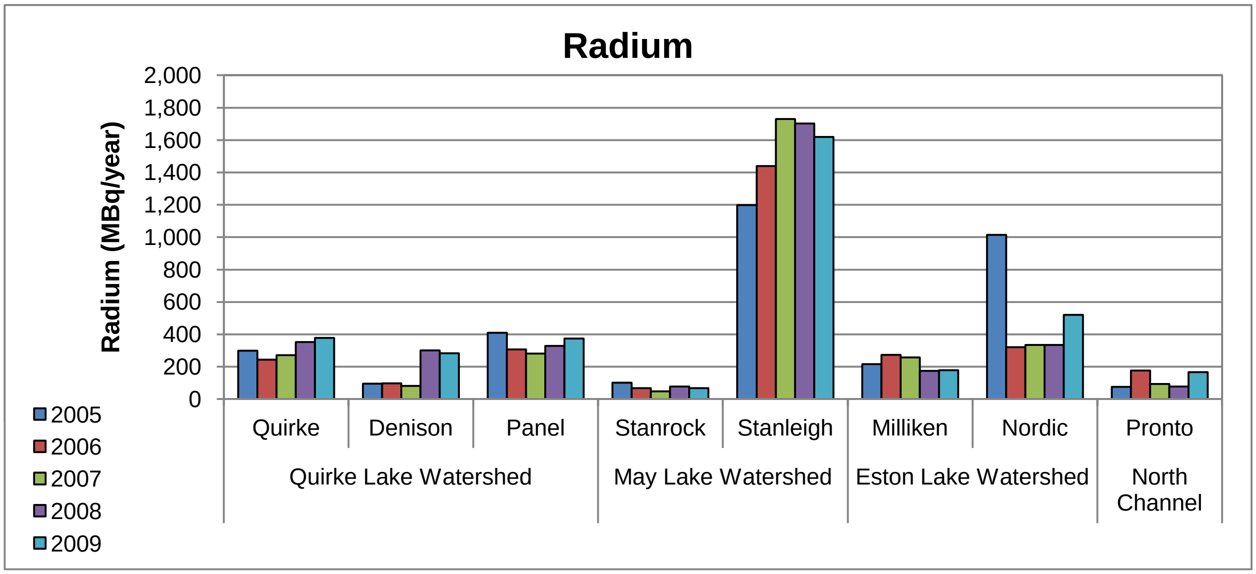


Figure 4.3: Annual TMA loadings by watershed (2005-2009).





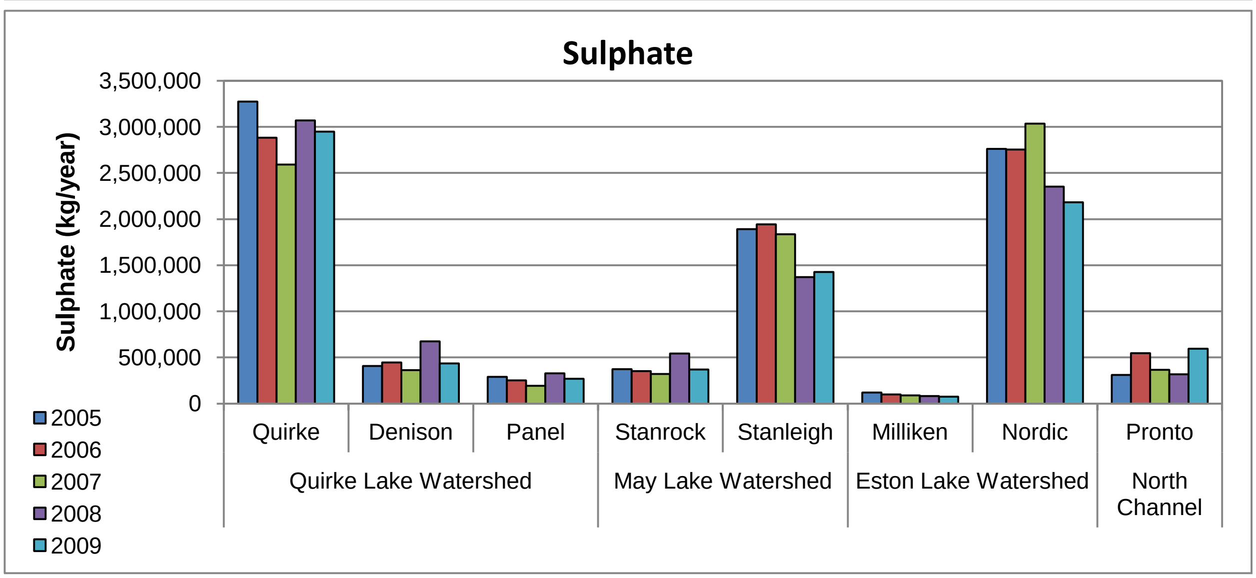


Figure 4.3: Annual TMA loadings by watershed (2005-2009).

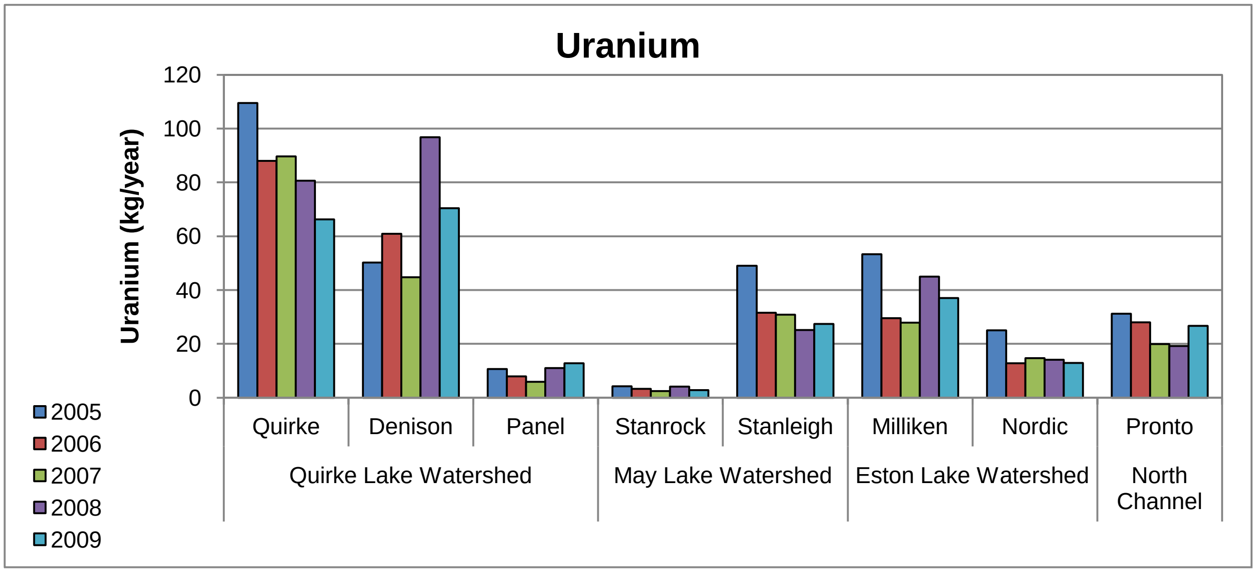


Figure 4.3: Annual TMA loadings by watershed (2005-2009).

Table 4.2: Summary of water quality trends[a] for SAMP water quality monitoring stations in Denison, Quirke, Panel, and Stanrock, 2003 to 2009.

TMA	Station ID	Type	Number of Seasons Used in Common Trend[d]	Barium	Cobalt	Iron	Manganese	pH	Radium-226	Sulphate	Uranium
Denison	D-2	Primary	12	0.313	-0.221	0.333	-0.352	-0.585	-0.080	-0.688	*-0.181*
	D-3	Primary	10 to 12	0.598	ND[c]	0.116	0.157	-0.597	0.170	-0.519	*-0.135*
	D-9	Seepage	4	*0.399*[b]	-0.715	-0.581	-0.036	0.332	*-0.303*	-0.170	-0.410
	D-16	Seepage	4	-0.270	-0.571	0.286	-0.071	0.628	-0.565	-0.393	ND
Quirke	ECA-398	Seepage	7	0.484	-0.624	0.017	-0.389	0.494	-0.416	-0.850	-0.909
	Q-22	Drainage	4	-0.131	-0.523	-0.307	-0.330	0.548	-0.402	-0.340	-0.652
	Q-23	Drainage	1 to 4	-0.232	0.102	0.081	-0.362	0.033	-0.927	-0.558	ND
	Q-27	Seepage	2 to 3	0.582	0.175	-0.260	0.069	-0.039	-0.275	0.315	-0.900
	Q-28	Primary	12	0.522	-0.401	0.391	-0.421	0.107	0.098	-0.704	-0.585
Panel	P-02	Seepage	2 to 4	-0.115	-0.366	-0.018	-0.304	0.426	-0.384	-0.875	-0.342
	P-03	Drainage	4	*0.139*	ND	*-0.265*	*-0.139*	-0.509	0.200	-0.143	ND
	P-05	Drainage	2 to 4	-0.248	0.004	0.045	-0.125	-0.356	-0.533	-0.192	ND
	P-11	Drainage	2 to 4	-0.158	-0.550	-0.246	-0.479	0.056	-0.568	-0.596	-0.205
	P-14	Primary	4 to 5	0.676	-0.386	-0.302	-0.226	-0.301	-0.701	-0.886	-0.612
Stanrock	DS-16	Drainage	1 to 4	-0.040	-0.804	-0.402	-0.580	0.777	-0.569	-0.453	-0.714

decreasing trend, significant at $p<0.05$

increasing trend, significant at $p<0.05$

[a] Based on rank correlation coefficients (rho) for common (combined) seasonal trends, shown in table.

[b] Italic text mean monthly correlations significantly different, but common trend value provided.

[c] ND denotes that this parameter was not included in the trend analysis for that particular station due to >50% non-detectable concentrations in the samples available for the analy

[d] Seasons used varied for sbustances based on suitability of data for trend analysis.

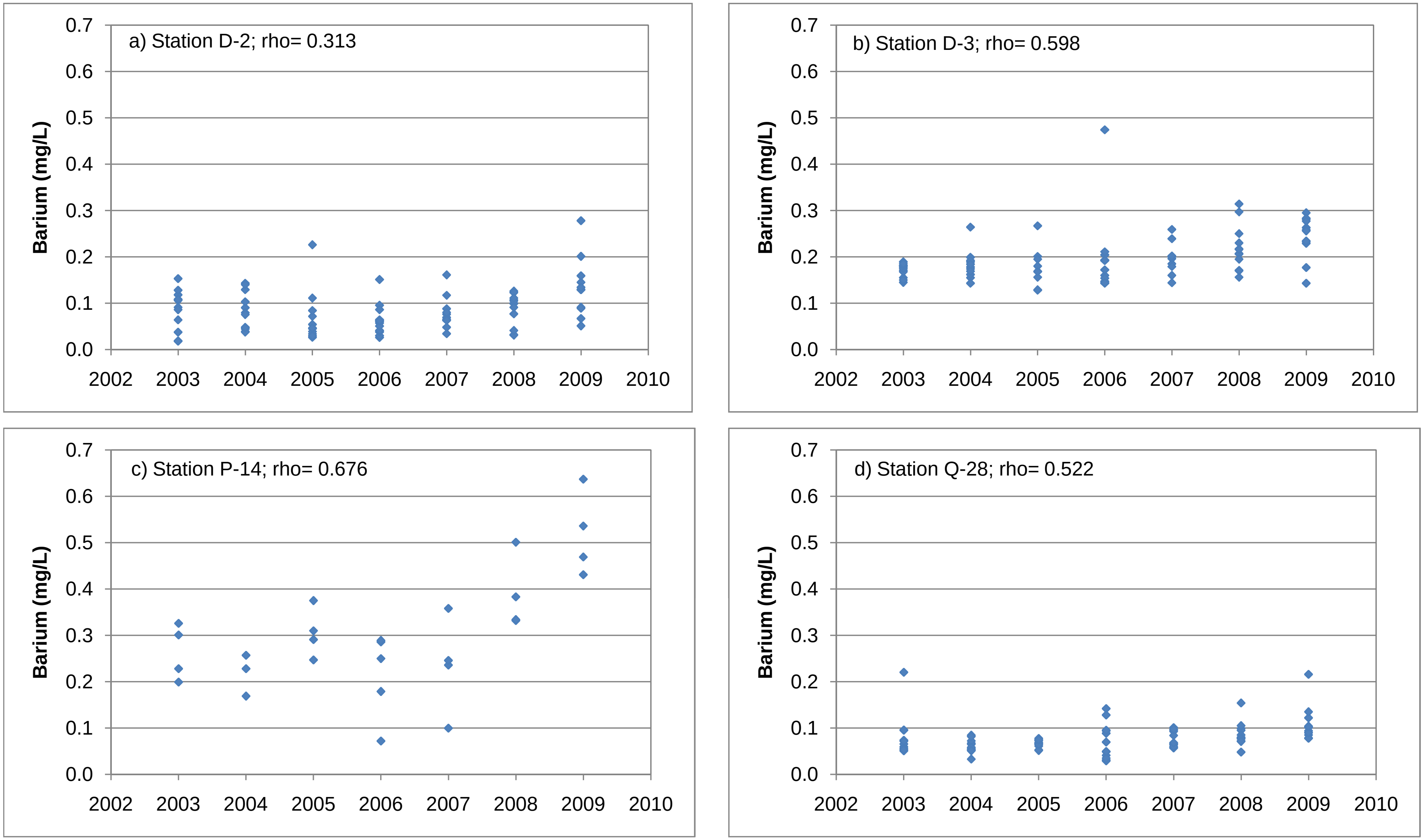


Figure 4.4: Significant common trends for barium from 2003 to 2009 at a) Station D-2, b) Station D-3, c) Station P-14, and d) Station Q-28, SAMP

Iron concentrations increased in the primary discharge at both Denison (D-2) and Quirke (Q-28) TMA from 2003 to 2009. Both trends were influenced by data from 2008 which may have reflected shorter retention times (*i.e.*, less settling of solids) in the settling ponds under the combined condition of ice cover and higher winter and early spring flows (Appendix Figures D.1.2 and D.2.6), as iron did not increase within either basin (D-1 and Q-05; Sections 3.1and 3.2). Despite the increasing trends, iron concentrations in effluent remained low (< 1.5 mg/L).

Discharge pH increased at all discharge locations except at Panel Pond C (P-03) and Denison primary discharge locations (D-2 and D-3; Table 4.2). The decrease in pH at the Denison TMA is largely due to a step change in ETP influent pH in 2008 and 2009, possibly associated with decreasing sulphate concentrations since 2000 and/or higher water levels in 2008 and 2009. At both of these locations, pH remains neutral and above the discharge criteria and PWQO.

Trends indicating improving water quality (decreasing metal concentrations and increasing pH) at DS-16 are associated with the installation of a treatment system in 2005.

4.2 May Lake Sub-watershed Sources

Within the May Lake sub-watershed there are two TMA's: Stanrock, with primary discharges to Halfmoon Lake, and Stanleigh, with primary discharges to McCabe Lake. There are no seepages from these TMAs that drain directly to the May Lake sub-watershed. Both Halfmoon and McCabe Lake drain to May Lake. As part of the SRWMP, water quality is monitored both upstream and downstream of the TMA sources (Figure 4.5).

4.2.1 Discharge Quality and Loads

Concentrations from source discharges are generally less than the SRW benchmarks with exception of barium at the Stanleigh discharge (CL-06) and sulphate at both Stanleigh and Stanrock TMA (DS-4) discharge (Figure 4.6). Barium concentrations in the Stanleigh TMA effluent (mean of 0.39 mg/L) are well below levels considered to be toxic to aquatic biota (>8 mg/L; WHO 2001; USEPA 2007). Similarly, sulphate concentrations in the Stanrock (<400 mg/L) and Stanleigh (<250 mg/L) discharges are less than concentrations that would be expected to be toxic to aquatic biota (about 500 mg/L; Mount and Gulley 1992; Singleton 2000; Davies 2007). Generally, concentrations in the immediate downstream receiving environment are less than the SRW benchmarks. Further downstream, water quality in May Lake consistently met the SRW benchmarks and so it was judged to meet acceptability

Figure 4.5

May Lake Sub-watershed Mine Source and Receiving Environment Stations

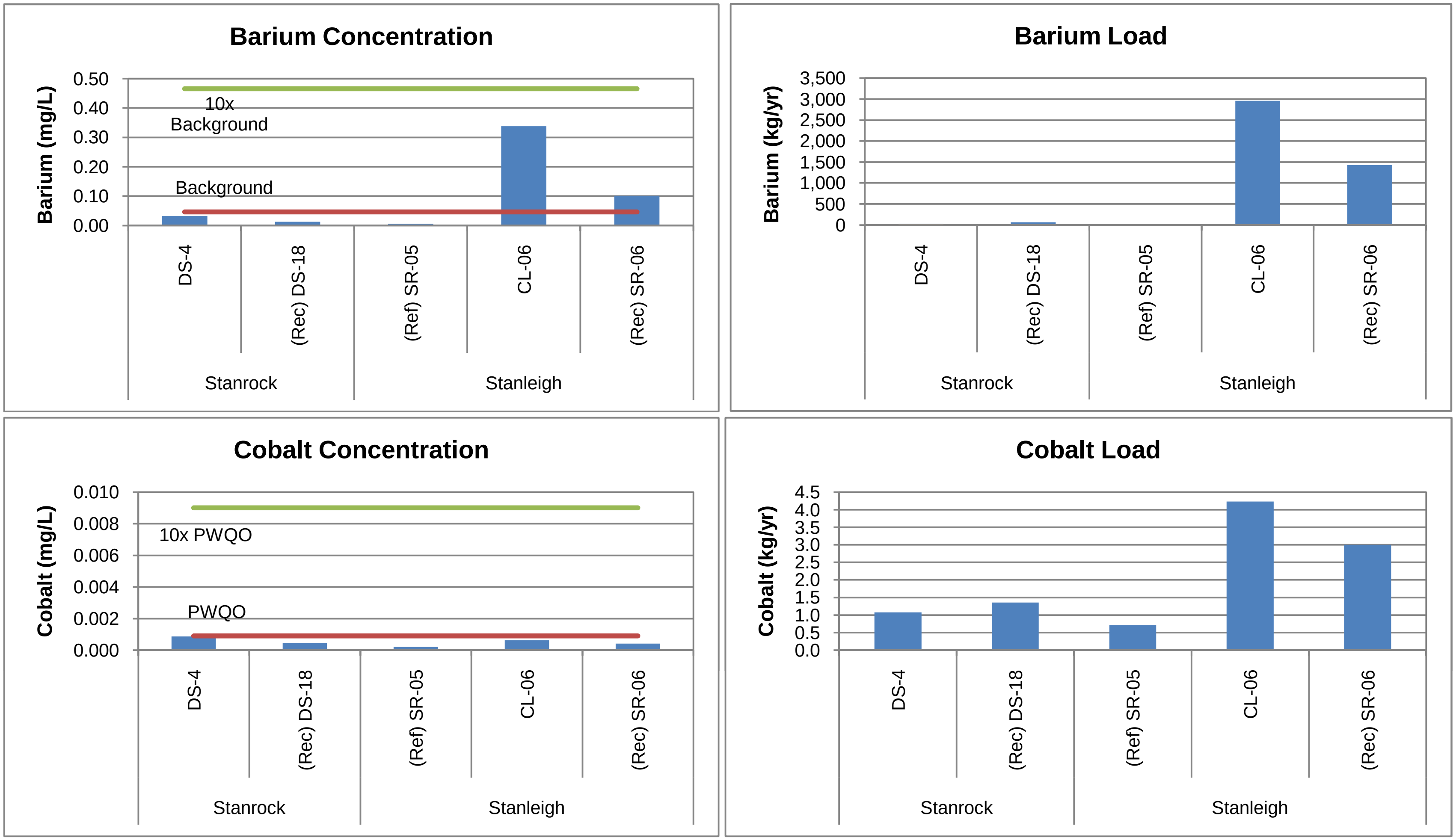


Figure 4.6: Mean concentrations and loads at monitoring stations downstream of Stanrock and Stanleigh TMAs, 2005-2009 (Rec) denotes receiving environment station, (Ref) denotes reference station.

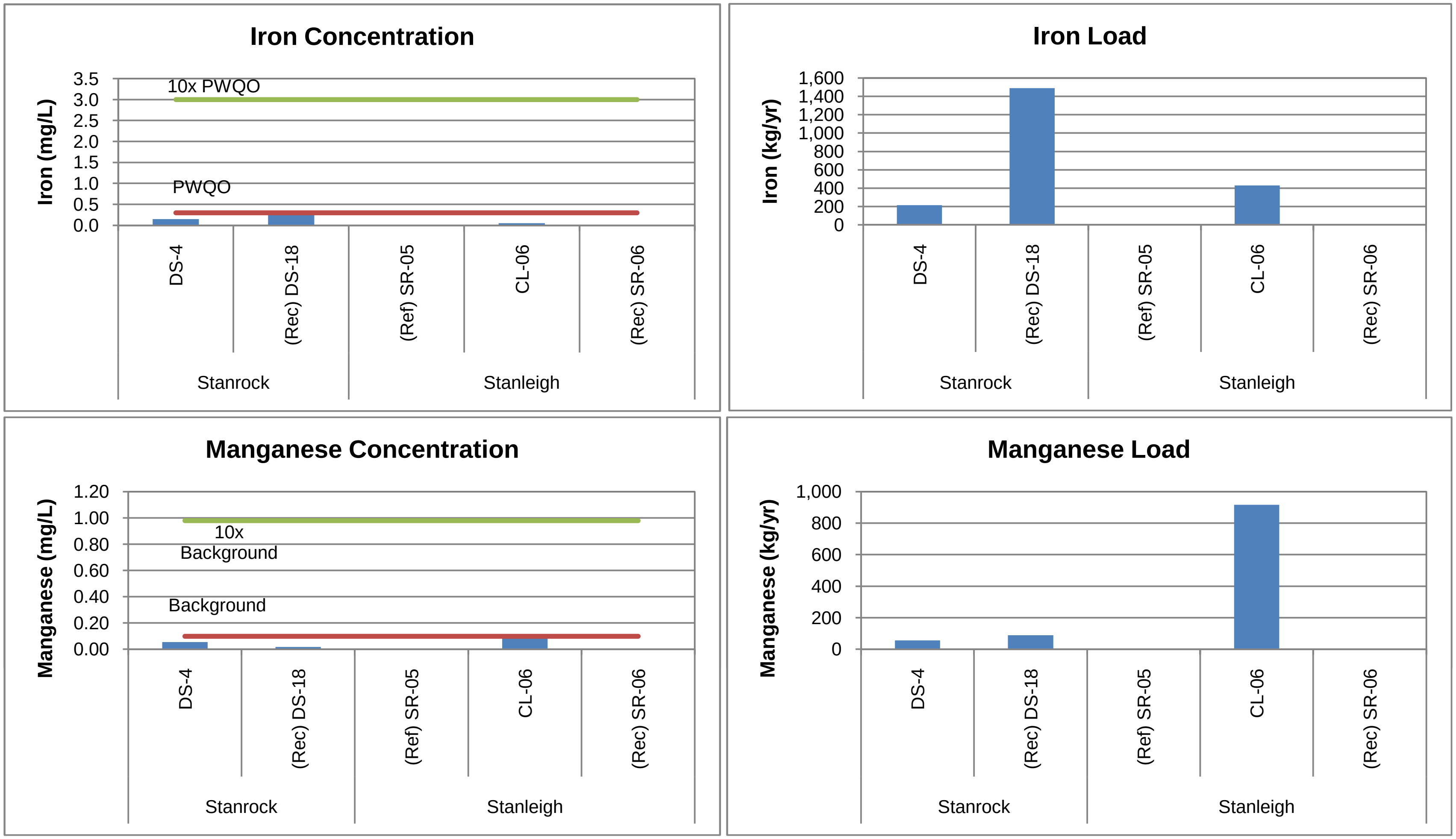


Figure 4.6: Mean concentrations and loads at monitoring stations downstream of Stanrock and Stanleigh TMAs, 2005-2009, (Rec) denotes receiving environment station, (Ref) denotes reference station. Iron and manganese not measured at SR-05 and SR-06.

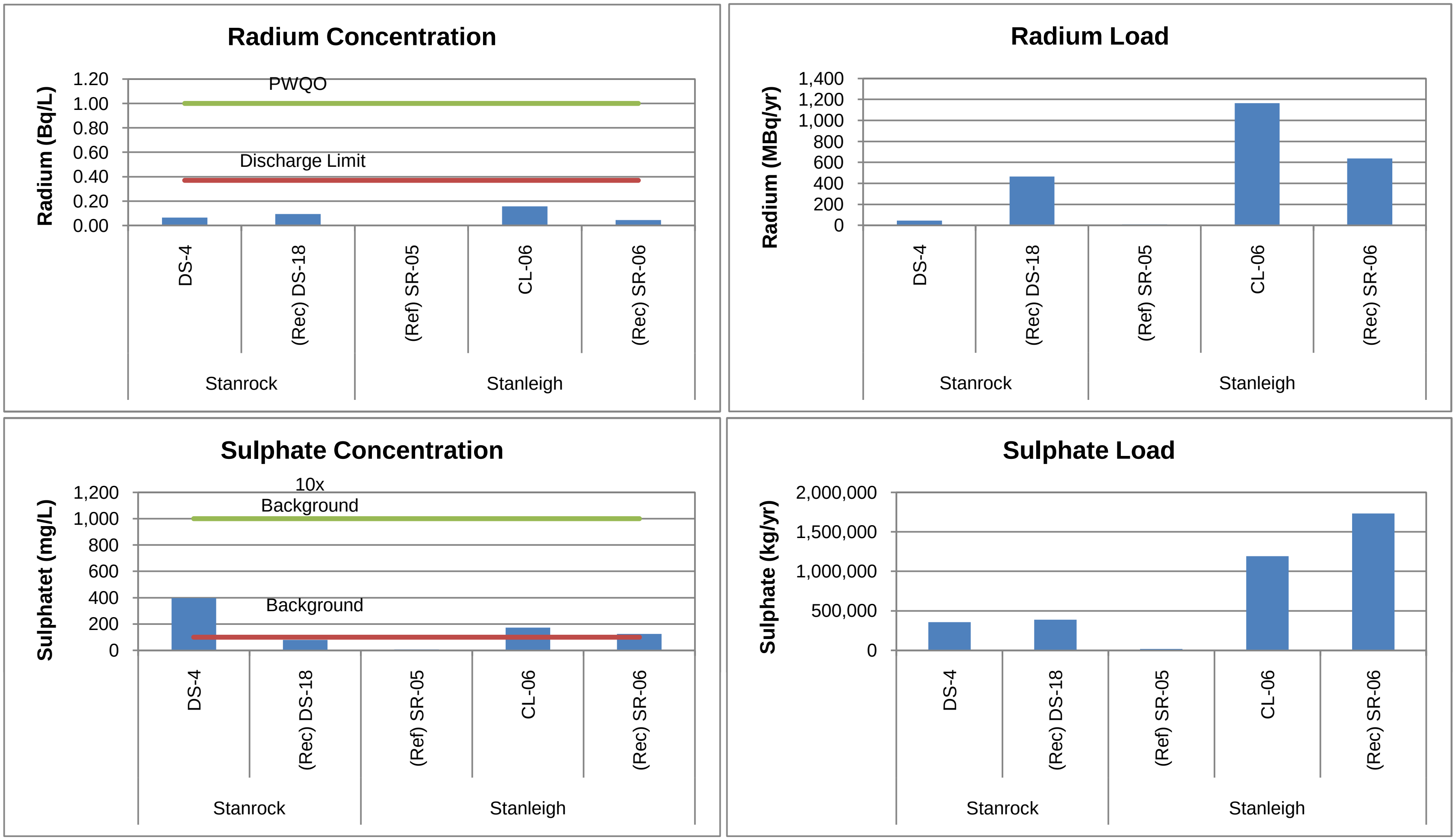


Figure 4.6: Mean concentrations and loads at monitoring stations downstream of Stanrock and Stanleigh TMAs, 2005-2009, (Rec) denotes receiving environment station, (Ref) denotes reference station.

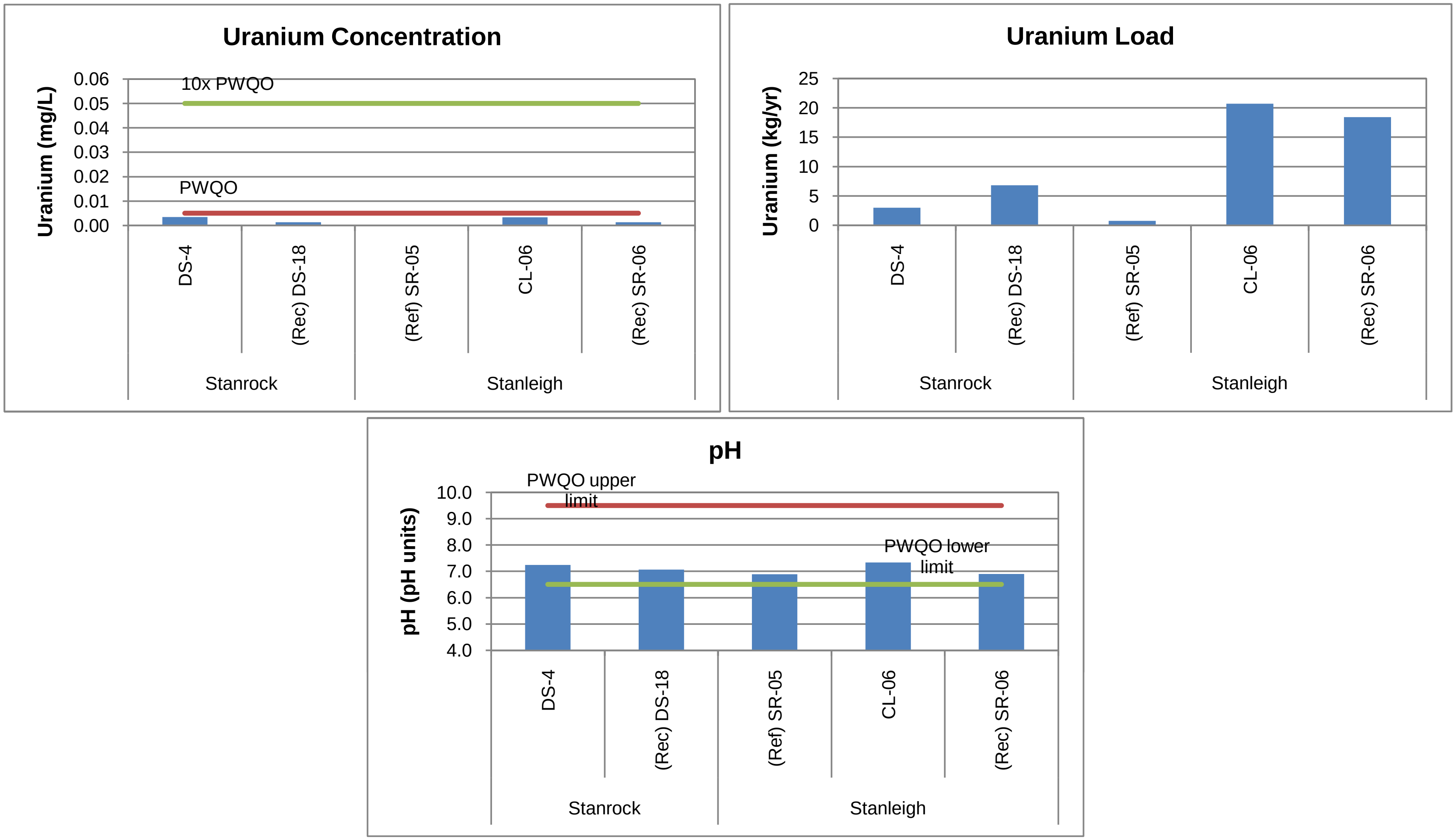


Figure 4.6: Mean concentrations and loads at monitoring stations downstream of Stanrock and Stanleigh TMAs, 2005-2009, (Rec) denotes receiving environment station, (Ref) denotes reference station.

criteria and removed as a SRWMP water quality station for the Cycle 3 Study Design (Minnow 2009b).

Loadings of most substances monitored are higher from the Stanleigh TMA than from the Stanrock TMA (Figure 4.6). However, water quality downstream in McCabe Lake achieves SRW benchmarks for all substances except barium and these concentrations remain well below toxicity thresholds.

4.2.2 Trends

Effluent concentrations of sulphate and manganese at the Stanleigh TMA have been decreasing over time (2003 to 2004) and uranium concentrations were so low in final effluent (more than 50% of values were less the MDL of 0.0005 mg/L) that trend analysis could not be conducted (Table 4.3). Consistent with ETP influent, effluent radium-226 and consequently barium concentrations have been increasing over the same period. The increase in radium concurrent with a decrease in sulphate concentrations within the basin is consistent with the work completed by EcoMetrix (Appendix G) which indicates that as aqueous sulphate concentrations decline, there is an increased dissolution of barium sulphate to which radium is associated, whereby radium is released from the tailings. It is expected that radium concentrations in porewater will stabilize over time once the dissolution of barium sulphate re-equilibrates with aqueous sulphate concentrations. Assuming there is no new source of radium to the TMA, radium concentrations in porewater should decline as the amount of soluble material in the tailings diffusion zone decreases. Radium-226 concentrations remain below the discharge criterion (0.37 Bq/L) and well below the PWQO (1.0 Bq/L).

Concentrations of radium-226, sulphate and uranium in effluent from the Stanrock TMA (DS-4) have been decreasing over time (Table 4.3). Barium concentrations in the Stanrock effluent exhibited a significant increasing trend but the increase is not associated with radium-226 concentrations as these have been decreasing over the same period. The trend in barium appears to be leveraged by 2008 when higher flows resulted in longer discharge periods (i.e. greater barium loading). Effluent pH has also been decreasing over the same period but remains neutral (Table 4.3; Appendix Figure D.4.2).

4.3 Esten Lake Sub-Watershed Sources

Within the Esten Lake sub-watershed, there are two TMA's: Milliken TMA, with primary discharges into Elliot Lake via Sherriff Creek, and Nordic TMA, with primary discharges into Nordic Lake via Buckles Creek. There are no seepages that drain directly to receiving

Table 4.3: Summary of water quality trends[a] for SAMP monitoring stations in Stanleigh and Stanrock, 2003 to 2009.

TMA	Station ID	Type	Number of Seasons Used in Common Trend[c]	Barium	Cobalt	Iron	Manganese	pH	Radium-226	Sulphate	Uranium
Stanleigh	CL-06	Primary	4 to 6	0.902	-0.391	0.415	-0.569	-0.024	0.618	-0.916	ND[b]
Stanrock	DS-4	Primary	7 to 12	0.451	0.151	0.097	-0.169	-0.238	-0.505	-0.241	-0.757

(blue) decreasing trend, significant at $p<0.05$
(green) increasing trend, significant at $p<0.05$

[a] Based on rank correlation coefficients (rho) for common (combined) seasonal trends, shown in table.
[b] ND denotes that this parameter was not included in the trend analysis for that particular station due to >50% non-detectable concentrations in the samples available for the a
[c] Seasons used varied for sbustances based on suitability of data for trend analysis.

environments. Both Elliot and Nordic Lakes drain to Esten Lake. Surface water is monitored downstream of both TMAs at the inlet to Elliot Lake (M-01) and the outlet of Nordic Lake (SR-08) respectively (Figure 4.7). Surface water was previously monitored further downstream of both discharges at the outlet of Depot Lake, but surface water quality was of sufficient quality that monitoring was discontinued at this location.

4.3.1 Discharge Quality and Loads

Concentrations of most substances in the Milliken and Nordic final discharges achieve receiving environment criteria (*i.e.*, below the SRW benchmarks; Figure 4.8). Only iron concentrations were greater than the SRW benchmark in both TMA effluents, but concentrations in the receiving environment were near or below the benchmark. Sulphate was elevated in the Nordic TMA effluent but substantially reduced in the downstream receiving environment to concentrations (250 mg/L), which would not be expected to affect freshwater biota; freshwater biota are usually unaffected by sulphate concentrations less than 500 mg/L at water hardness of at least 50 mg/L (Mount and Gulley 1992; Singleton 2000; Davies 2007).

With the exception of iron and uranium, Nordic TMA loads for all other measured substances were higher than from the Milliken TMA (Figure 4.8). Loadings from the Milliken TMA are likely over-estimated because flow at this location is prorated by drainage area (*i.e.*, measured concentrations are not synoptic with actual flows) but the highest concentrations occur under no flow conditions (due to re-mobilization of metals under anoxic conditions).

Other than sulphate concentration downstream of the Nordic TMA, loadings from these facilities are not resulting in concentrations in the receiving environments above SRW benchmarks.

4.3.2 Trends

Identified trends were indicative of improving water quality in mine discharges (Table 4.4). Where a trend was detected, concentrations of barium, cobalt, manganese, radium-226, sulphate and uranium were decreasing and pH was increasing. The trends at Nordic reflect improvements associated with the Buckles Creek diversion work conducted in 2005 and trends observed in the ETP influent (Appendix Figure C.7.4; Table 3.23).

4.4 Pronto

The Pronto TMA is outside the Serpent River Watershed and effluent from the TMA discharges to a drainage ditch that flows south and discharges into Lake Huron (Figure 4.9). Final effluent, monitored in the Discharge Channel at PR-01, reports directly to the North

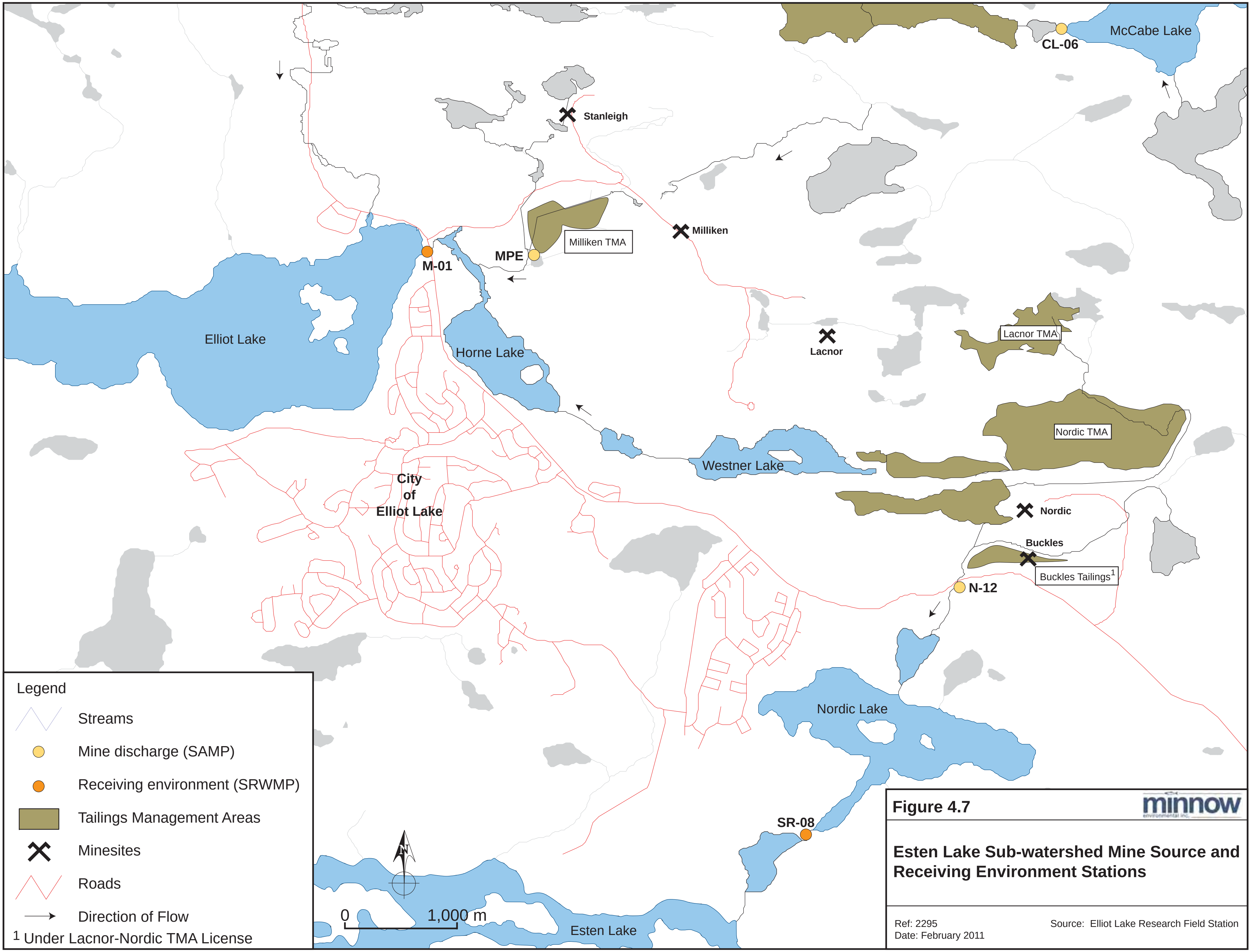


Figure 4.7

Esten Lake Sub-watershed Mine Source and Receiving Environment Stations

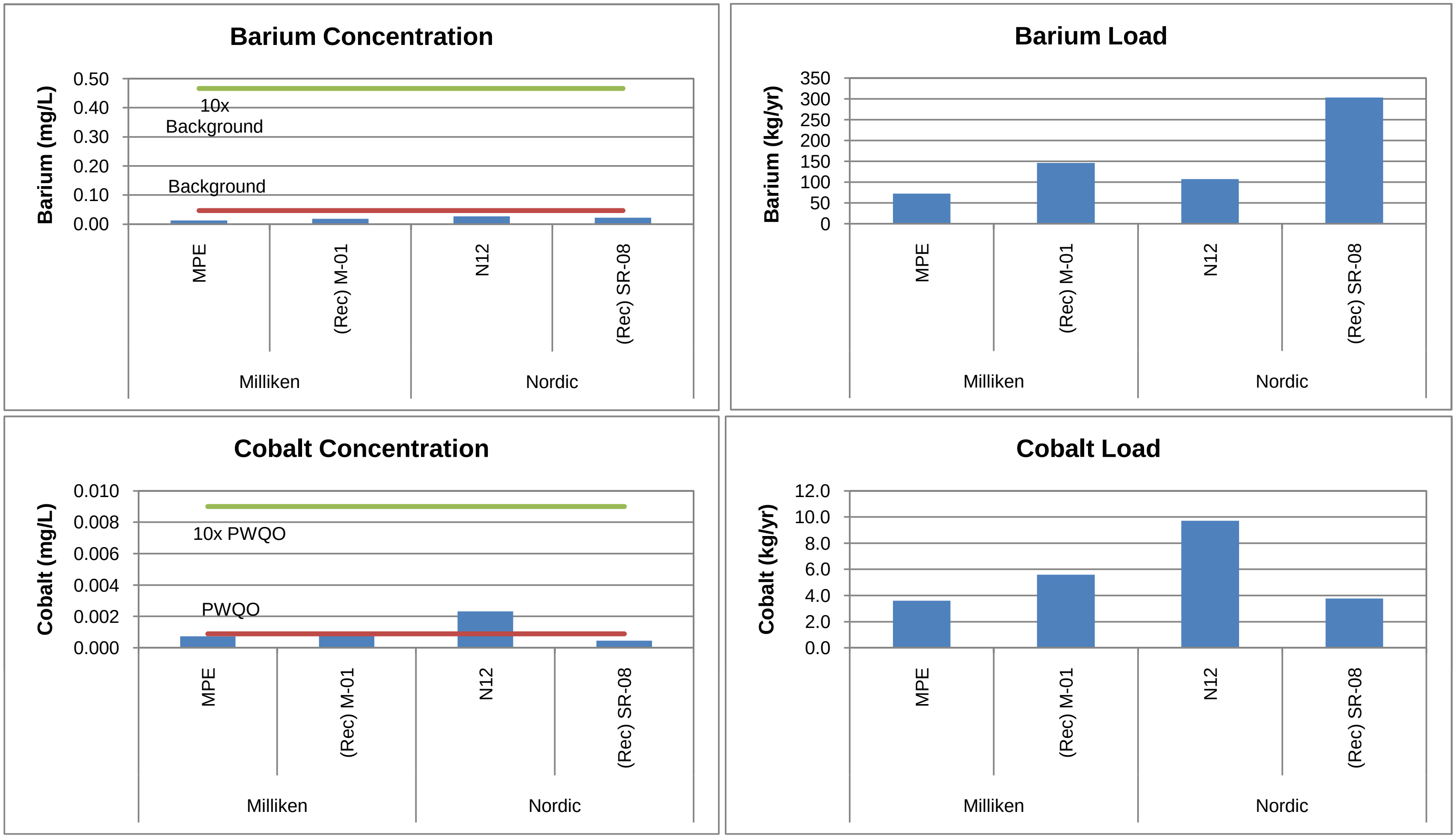


Figure 4.8: Mean concentrations and loads at monitoring stations downstream of Milliken and Nordic TMAs, 2005-2009, (Rec) denotes receiving environment station.

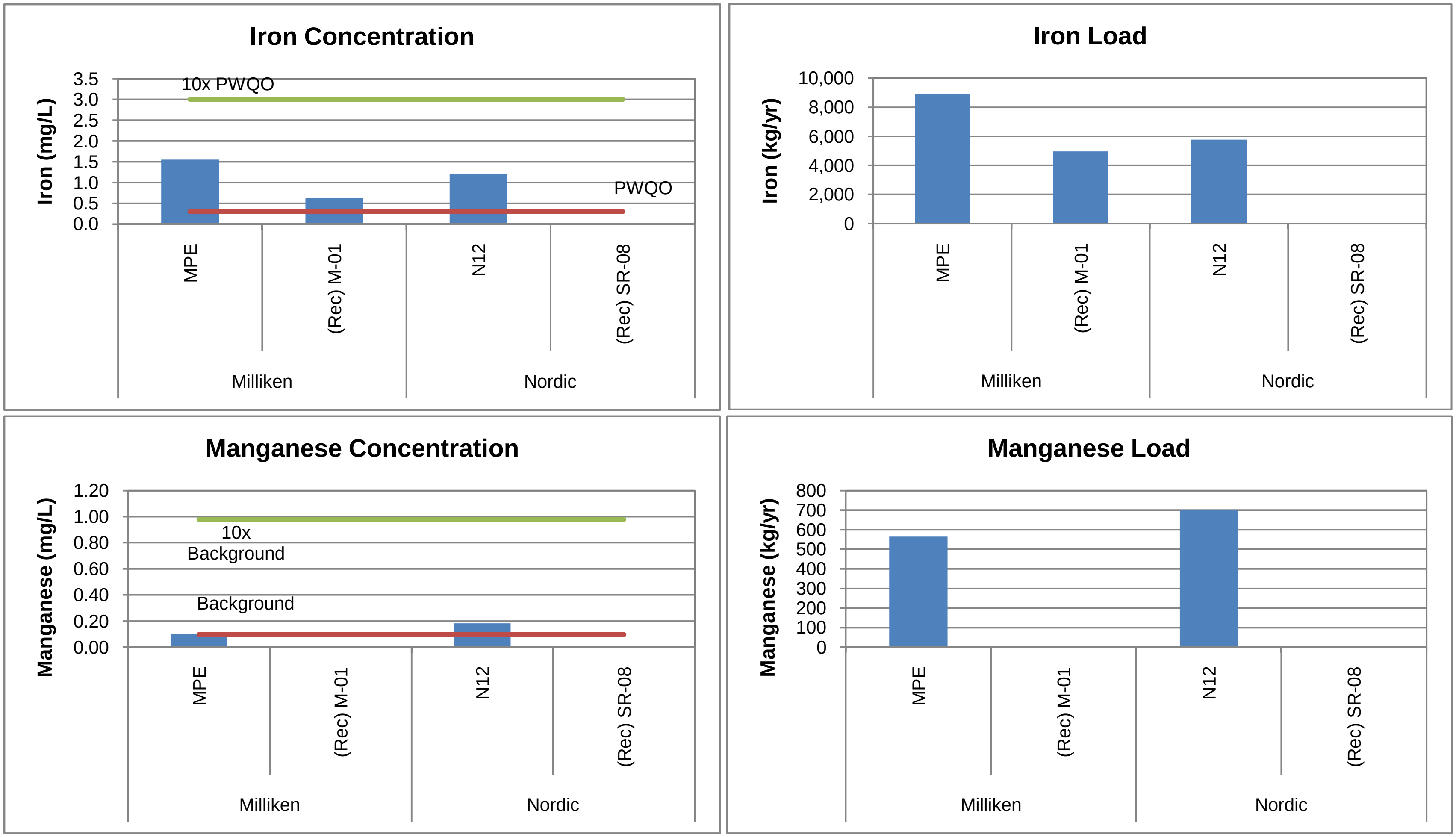


Figure 4.8: Mean concentrations and loads at monitoring stations downstream of Milliken and Nordic TMAs, 2005-2009, (Rec) denotes receiving environment station. Iron not measured at SR-08 and manganese not measured at M-01 and SR-08.

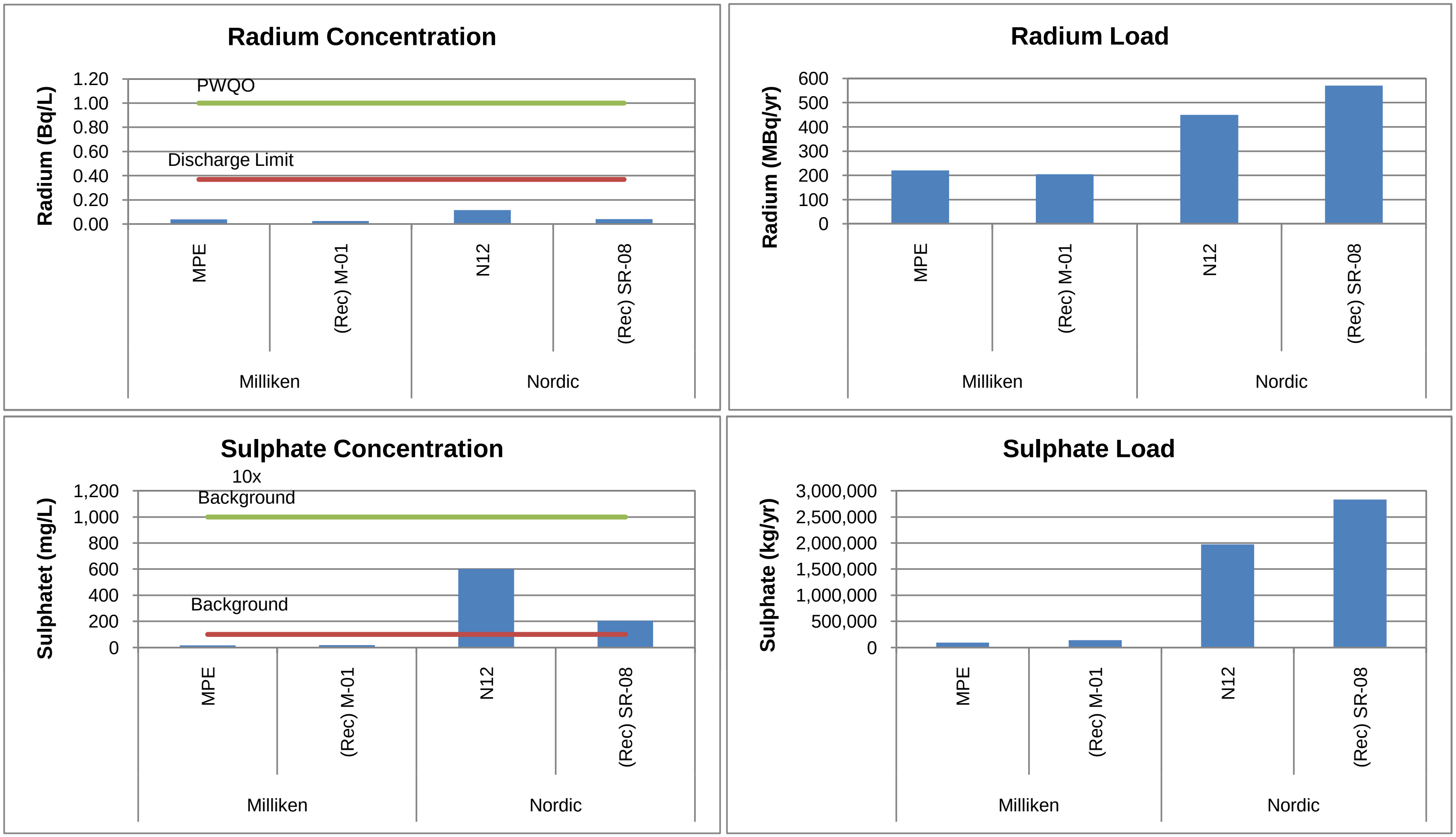


Figure 4.8: Mean concentrations and loads at monitoring stations downstream of Milliken and Nordic TMAs, 2005-2009, (Rec) denotes receiving environment station.

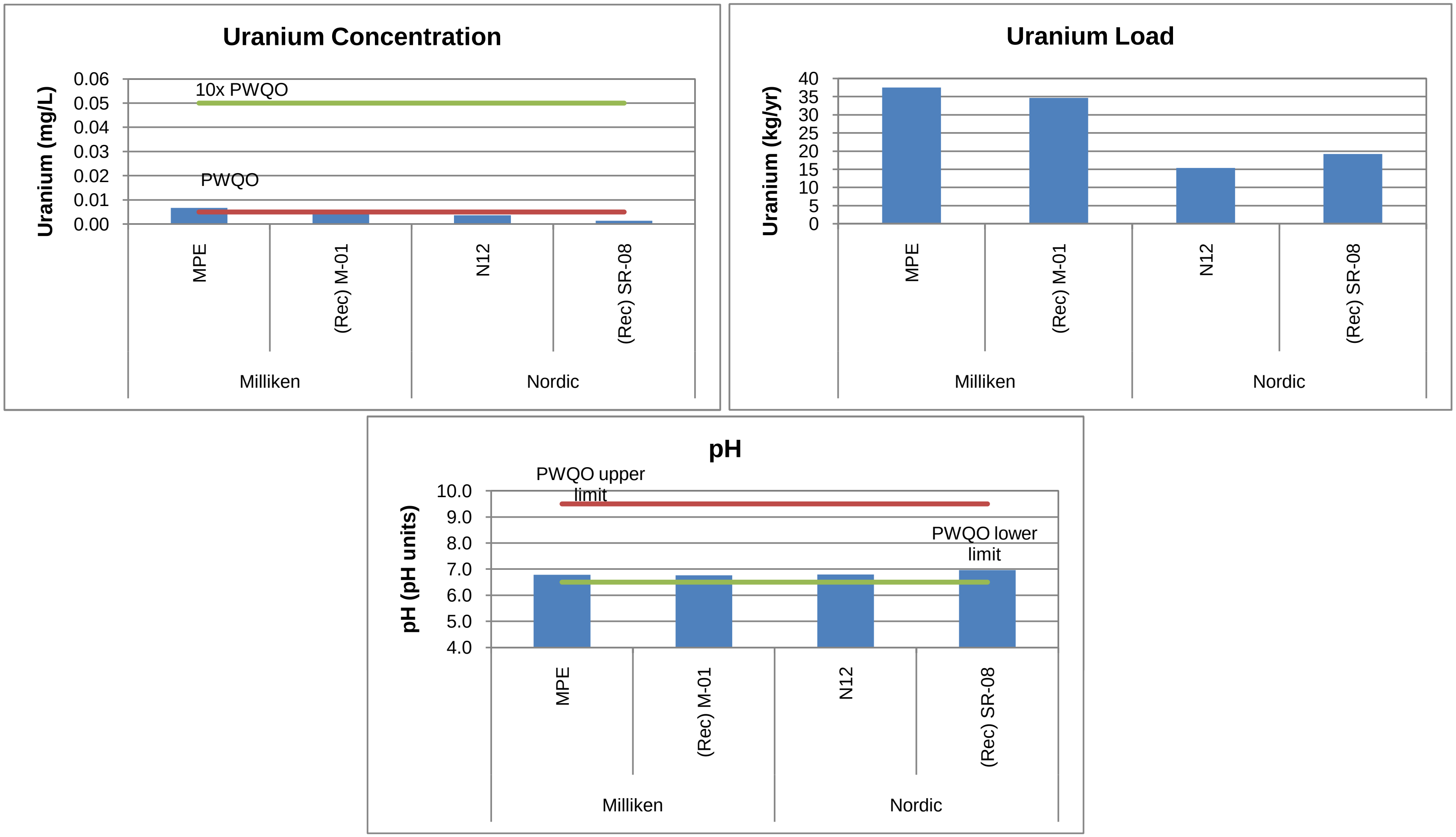


Figure 4.8: Mean concentrations and loads at monitoring stations downstream of Milliken and Nordic TMAs, 2005-2009, (Rec) denotes receiving environment station.

Table 4.4: Summary of water quality trends[a] for SAMP monitoring stations in Nordic and Milliken, 2003 to 2009.

TMA	Station ID	Type	Number of Seasons Used in Common Trend[c]	Barium	Cobalt	Iron	Manganese	pH	Radium-226	Sulphate	Uranium
Nordic	N-12	Primary	11 or 12	-0.593	-0.059	*-0.095*[b]	-0.108	0.581	-0.583	0.013	-0.528
Milliken	MPE	Primary	6 to 12	0.076	-0.496	-0.037	-0.271	0.258	-0.511	-0.630	0.009

decreasing trend, significant at $p<0.05$ (blue)
increasing trend, significant at $p<0.05$ (green)

[a] Based on rank correlation coefficients (rho) for common (combined) seasonal trends, shown in table.

[b] Italic text mean monthly correlations significantly different, but common trend value provided.

[c] Seasons used varied for sbustances based on suitability of data for trend analysis.

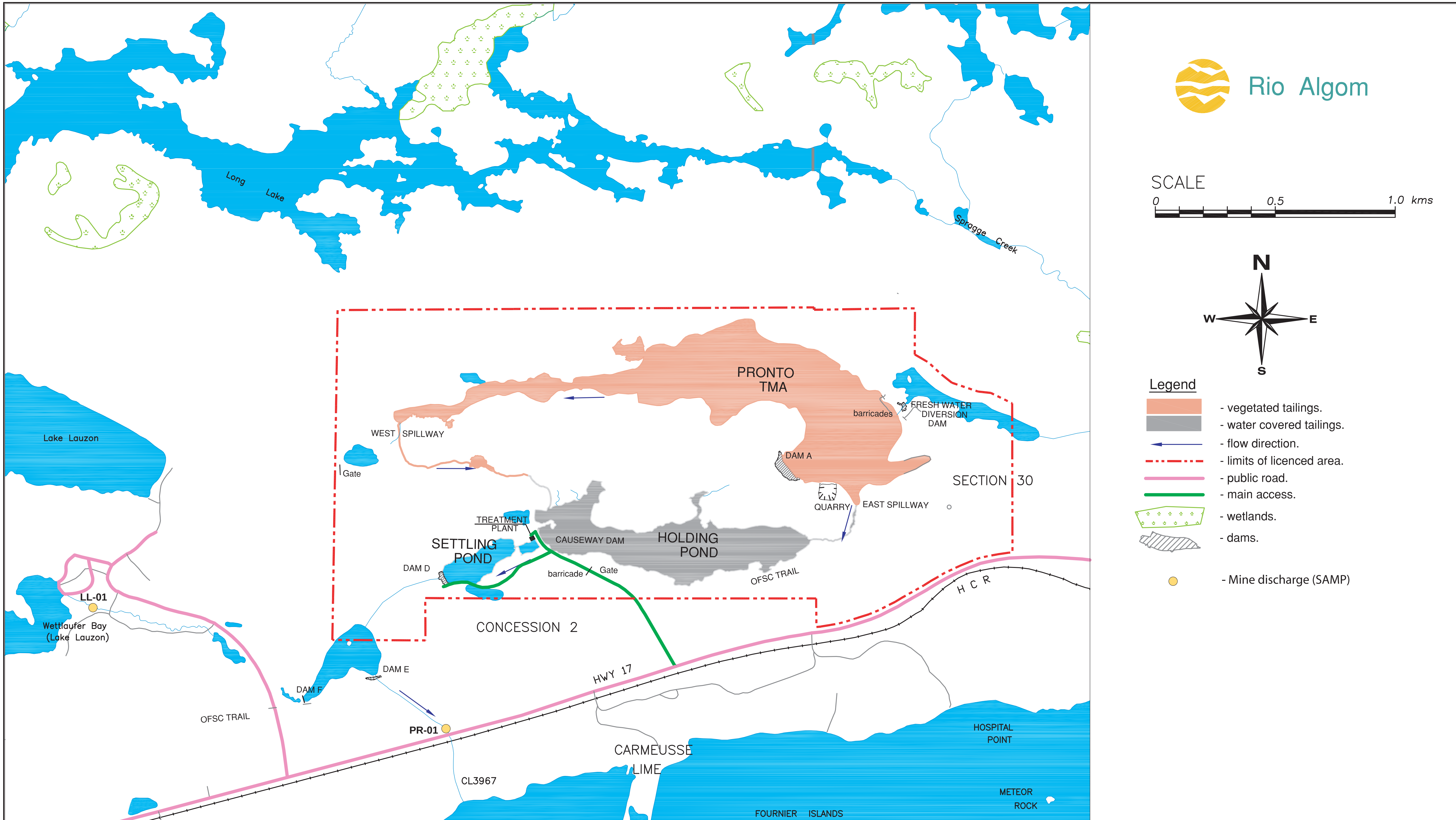


Figure 4.9

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minnow environmental inc.

Pronto Mine Source Monitoring Stations

Ref: 2295
Date: February 2011

Channel of Lake Huron, whereas site drainage to Pronto Creek (LL-01) reports to Lake Lauzon. Water quality monitoring downstream of PR-01 (in Pronto discharge channel and Lake Huron) is not included in the receiving environment monitoring program (SRWMP) due to confounding influences immediately downstream of the TMA discharge, including a rail line, Highway 17, and drainage from a lime calcining plant which enters Lake Huron adjacent to the Pronto discharge channel. Therefore the discussion that follows is limited to discharge quality.

4.4.1 Water Quality and Trends

With the exception of cobalt and uranium, concentrations of other substances monitored at the primary discharge (PR-01) are approaching or below the SRW benchmarks (Figure 4.10). Mean cobalt concentrations at PR-01 are about twenty times the SRW benchmark (PWQO) but mean uranium concentrations are only about 3 times the benchmark. Drainage to Lake Lauzon achieves receiving environment criteria for all substances (Figure 4.10).

Loads from the primary discharge (PR-01) are substantially greater (about 8 to 10 times) greater than those to Lake Lauzon (Figure 4.10).

Concentrations of barium, manganese, radium-226, sulphate, and uranium in site drainage (LL-01) have been decreasing since 2007 and are associated with repairs to Dam F that same year (Table 4.5; Appendix Figure D.8.1).

Decreasing concentrations of barium in the primary discharge were associated with a reduction in the use of barium chloride for treatment; in 2009, the TMA stopped using barium chloride as influent concentrations of radium-226 were sufficiently low that both pH and radium-226 could be treated with lime. Effluent pH has also been decreasing over time but remains near neutral. (Appendix Figure D.8.2).

4.5 Summary

Generally, concentrations of mine related substances were at or near receiving environment benchmarks established for the SRW in mine discharges during the period 2005 to 2009. Few discharges had concentrations more than ten times the benchmark and those discharges that did, tended to be seepages with relatively low flow. Therefore, seepage loads were small relative to primary discharge and background loads. With few exceptions, loads from mine sources were not sufficient to cause mean receiving environment concentrations to be above SRW benchmarks. Trends in discharge quality tended to indicate improvements over time and were consistent with trends observed within the TMAs.

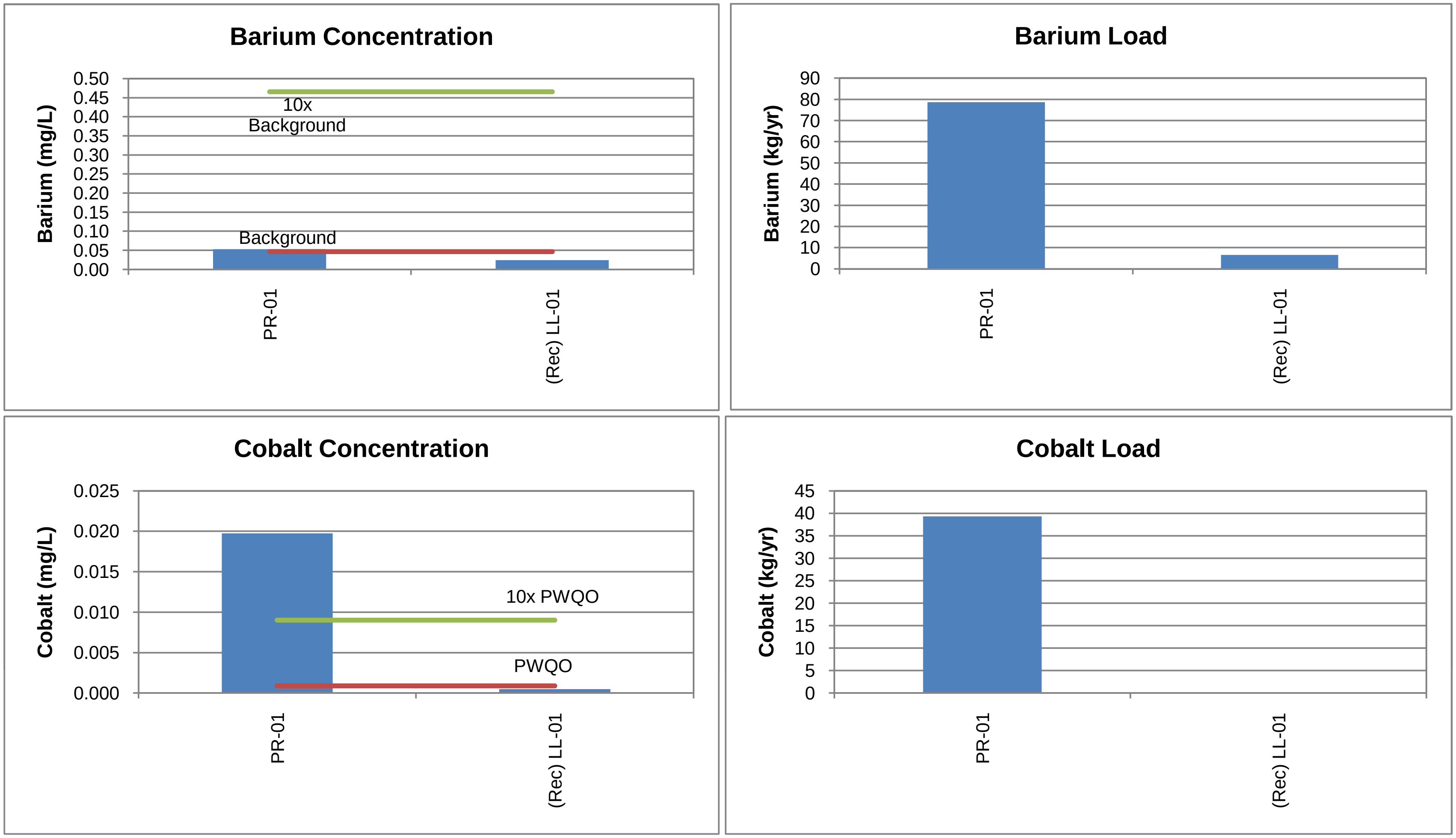


Figure 4.10: Mean concentrations and loads at monitoring stations downstream of Pronto TMA, 2005-2009, (Rec) denotes receiving environment station.

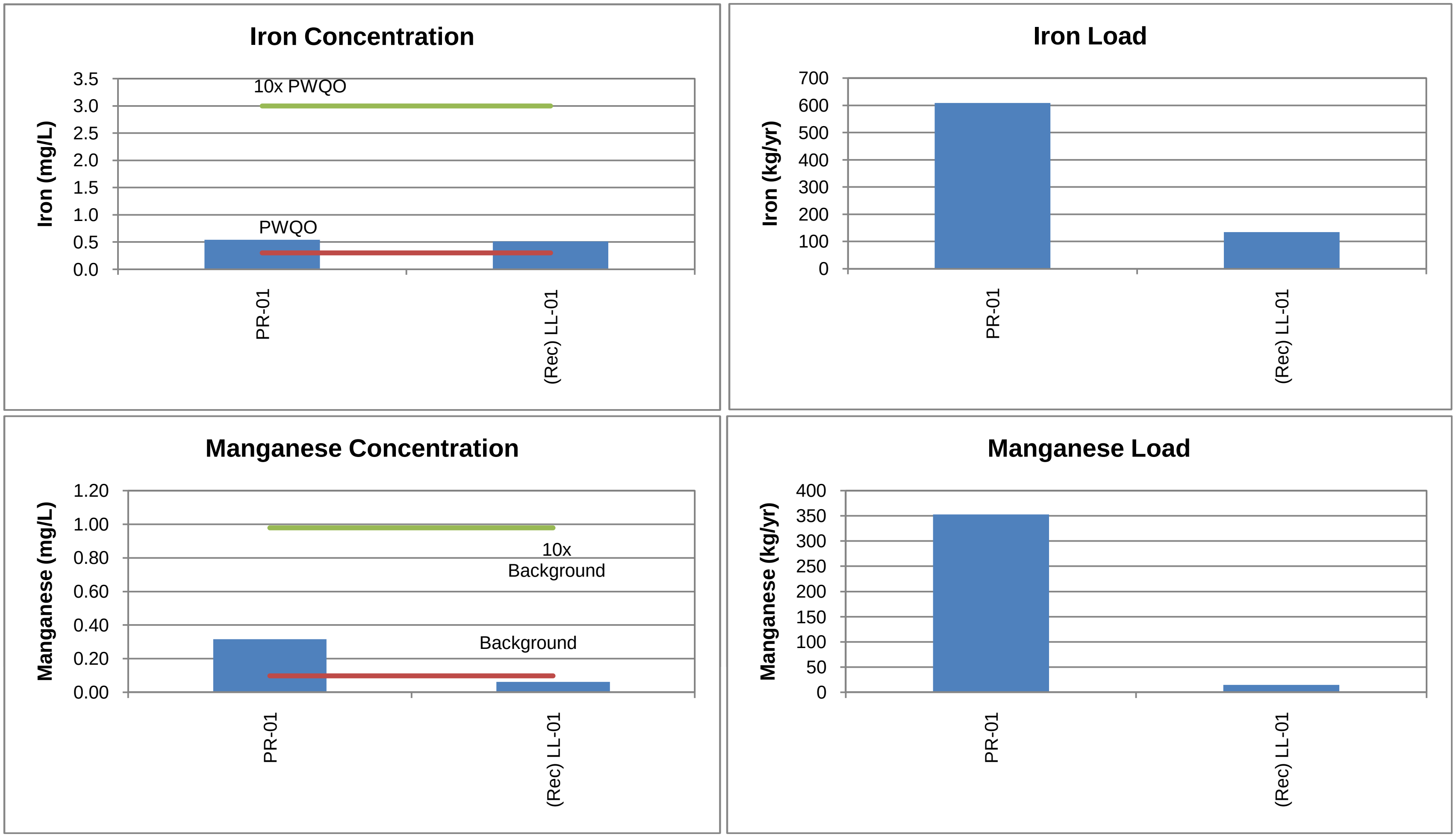


Figure 4.10: Mean concentrations and loads at monitoring stations downstream of Pronto TMA, 2005-2009, (Rec) denotes receiving environment station.

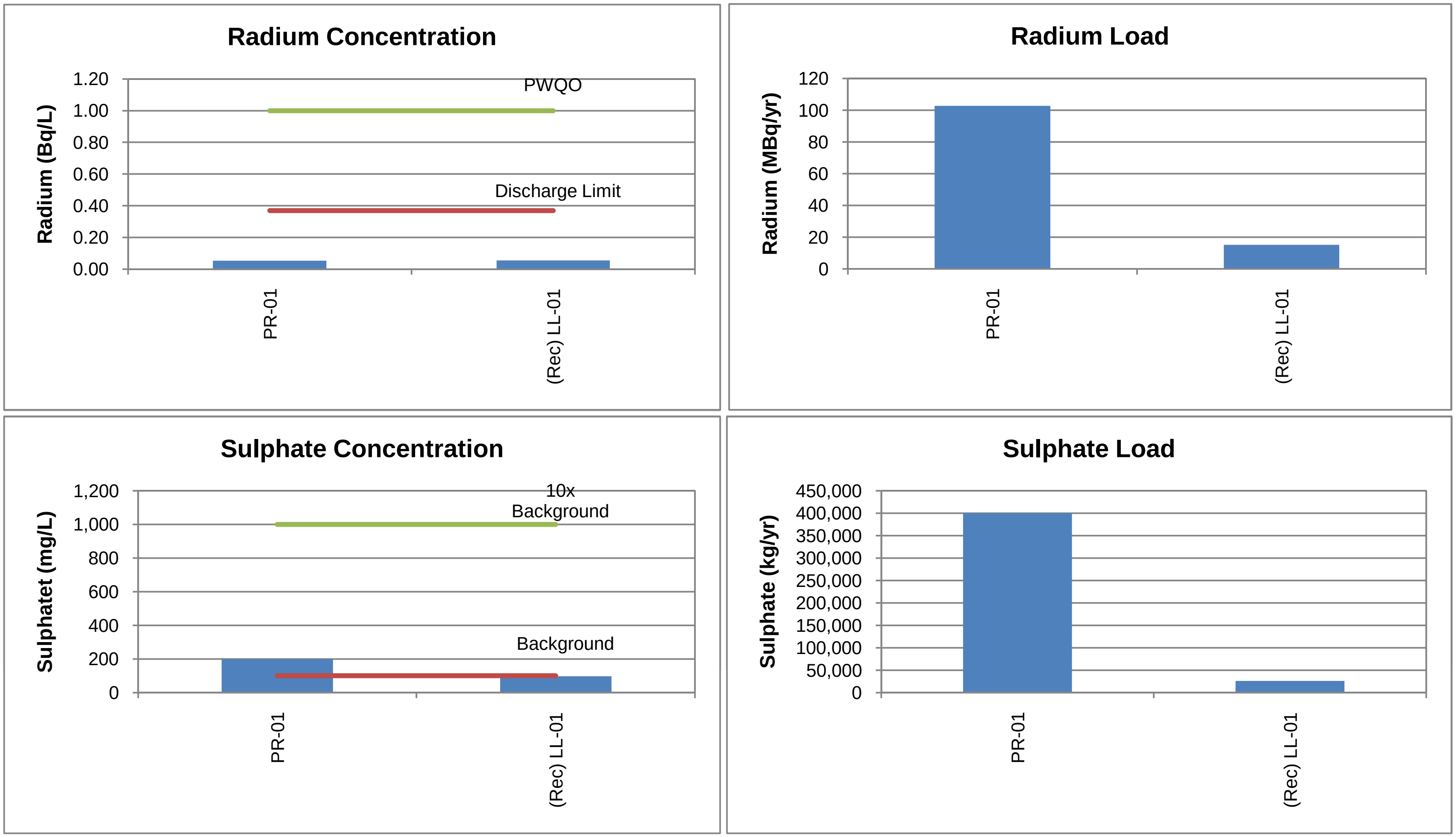
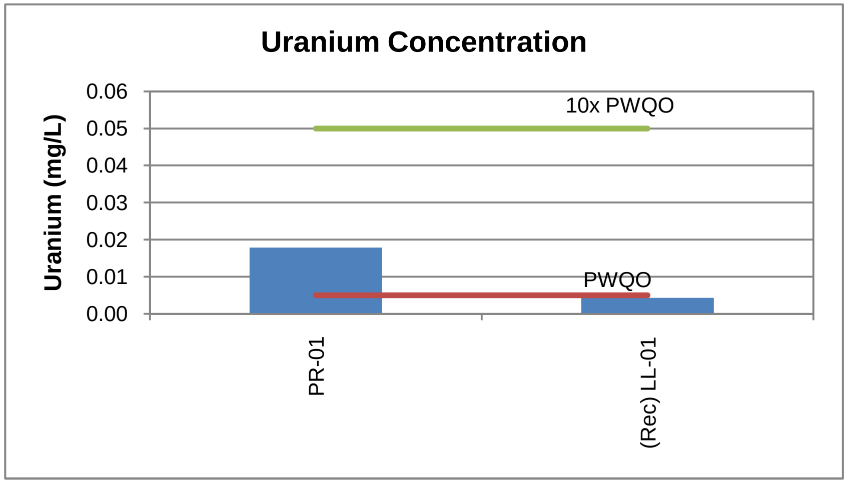
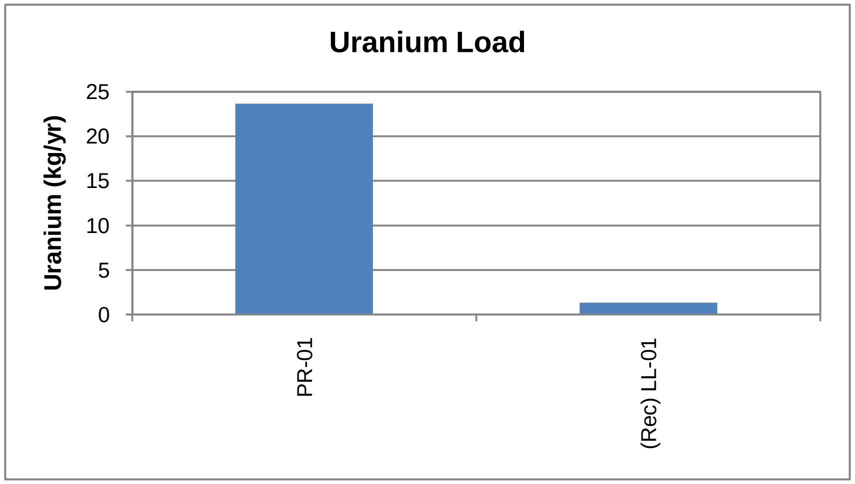


Figure 4.10: Mean concentrations and loads at monitoring stations downstream of Pronto TMA, 2005-2009, (Rec) denotes receiving environment station.





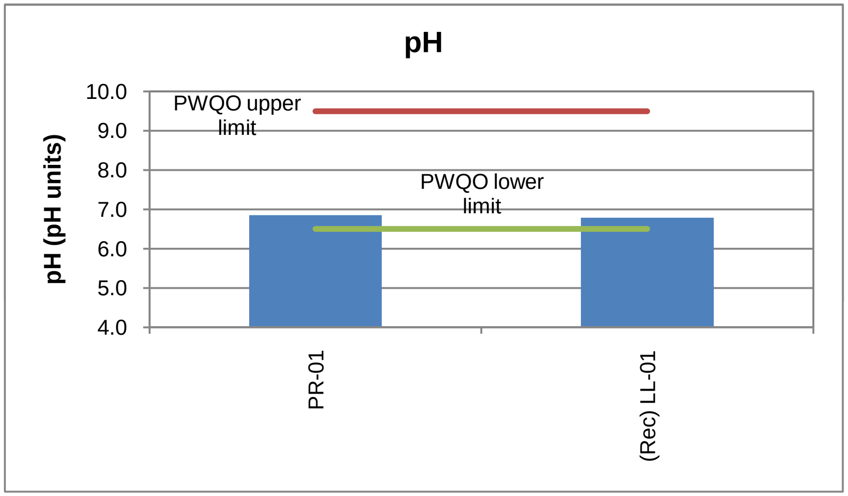


Figure 4.10: Mean concentrations and loads at monitoring stations downstream of Pronto TMA, 2005-2009, (Rec) denotes receiving environment station.

Table 4.5: Summary of water quality trends[a] for SAMP water quality monitoring stations in Pronto, 2003 to 2009.

TMA	Station ID	Type	Number of Seasons Used in Common Trend[c]	Barium	Cobalt	Iron	Manganese	pH	Radium-226	Sulphate	Uranium
Pronto	LL-01	Drainage	2 to 4	-0.604	ND[b]	-0.196	-0.500	-0.130	-0.652	-0.580	-0.862
	PR-01	Primary	12	-0.330	0.181	0.007	0.006	-0.255	0.163	-0.075	0.087

decreasing trend, significant at $p<0.05$

increasing trend, significant at $p<0.05$

[a] Based on rank correlation coefficients (rho) for common (combined) seasonal trends, shown in table.

[b] ND denotes that this parameter was not included in the trend analysis for that particular station due to >50% non-detectable concentrations in the samples availab

[c] Seasons used varied for sbustances based on suitability of data for trend analysis.

5.0 SERPENT RIVER WATERSHED

5.1 Water Quality

With few exceptions, mean and median water concentrations (2005 to 2009) were less than SRWMP benchmarks for most substances (Table 5.1; Figure 5.1). Mean (<0.20 mg/L) and maximum (<0.35 mg/L) barium concentrations were above the benchmark (background) downstream of TMA treatment plant discharges (D-5, Q-09 and SR-06), but remained well below levels considered to be toxic to aquatic life (8.0 mg/L; WHO 2001 and USEPA 2007). Mean cobalt concentrations were only above the benchmark (PWQO) at SC-01 and then only in 2005. Since 2006, mean cobalt concentrations have been below PWQO at all stations with maximum concentrations being close to the PWQO (Figure 5.1). Mean iron and manganese concentrations continued to be higher than respective benchmarks (background) at M-01 and D-6 but have not been increasing over time (Table 5.2). Mean and median concentrations of sulphate were greater than the benchmark (B.C. Guideline for the protection of aquatic life) at SR-06 and SR-08. However, sulphate concentrations were generally less than 250 mg/L, and thus below levels expected to be toxic to freshwater biota (500mg/L - Mount and Gulley 1992; Singleton 2000; Davies 2007).

Temporally, metal concentrations have generally been decreasing downstream of the TMAs over time (2000-2009), while pH has been increasing. Specifically, where a trend was detected for cobalt, manganese, radium-226, sulphate and uranium, concentrations were significantly decreasing (Table 5.2). Concentrations of iron have been increasing at stations D-6 and DS-18, but a similar trend was not observed in the upstream TMA source (D-1 and DS-4; Sections 3.1.3 and 3.4.2 respectively) and 85% of the iron samples at these stations remained at or below the benchmark (Table 5.1).

At SR-06, barium concentrations have been increasing and pH decreasing. The increase in barium is associated with an increase in barium chloride use at the Stanleigh ETP. As iron and sulphate concentrations in the influent decrease, there are fewer solids to react with the barium chloride and form the precipitate that removes radium. Following construction of the new treatment plant, lime and barium chloride addition rates were increased based on operating experience under similar operating conditions at Panel. The decrease in pH reflected changes in operating conditions. Between 1998 and 2002 there was no discharge from the TMA and the lake reflected more alkaline conditions associated with ETP operations. In 2007, pH deceased when the new treatment plant was being constructed and there was no discharge from the TMA from June 15 to December 15, 2007 (Appendix Figure E.13).

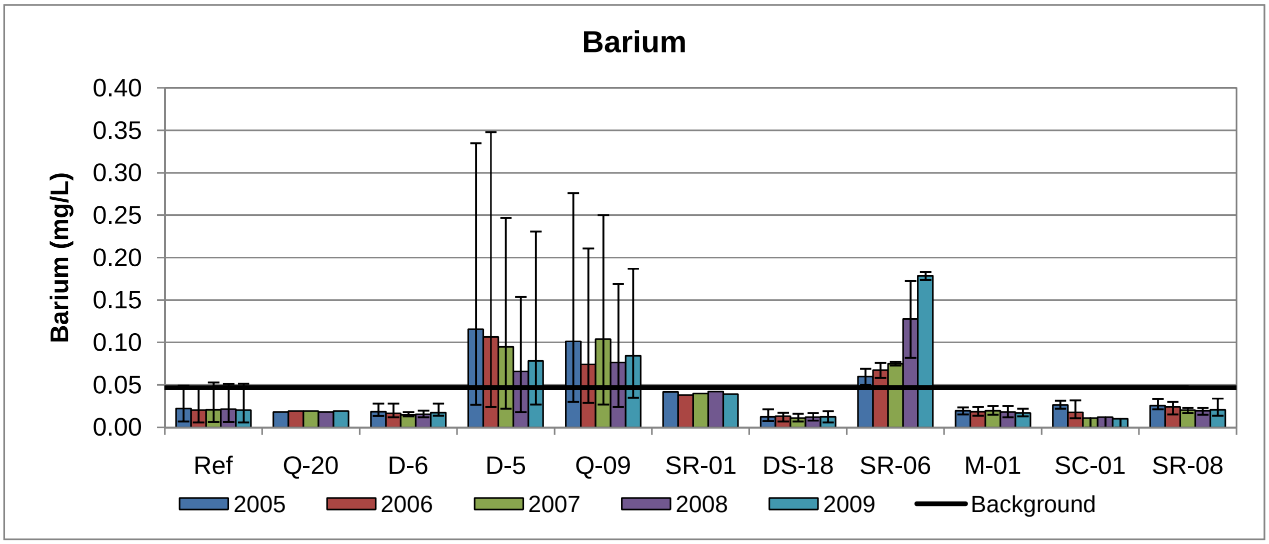
Table 5.1: Percent of samples exceeding selected benchmarks (shaded values) at SRWMP stations, 2005-2009.

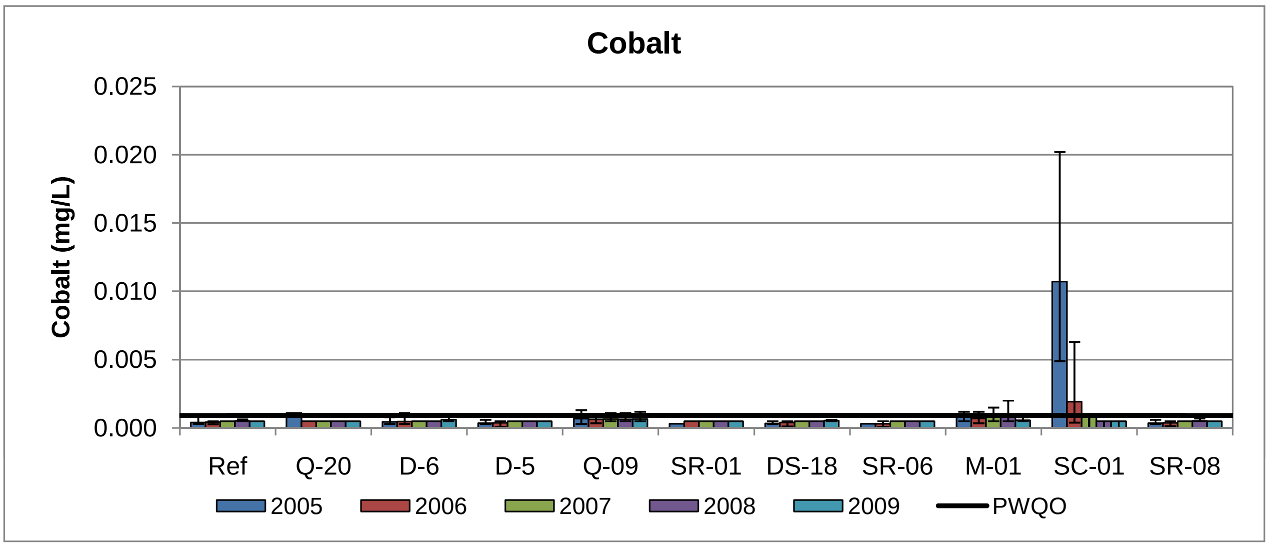
Station	# of Samples	Barium	Cobalt	Iron	Manganese	pH	Radium	Sulphate[b]	Uranium
		mg/L	mg/L	mg/L	mg/L	pH units	Bq/L	mg/L	mg/L
Upper limit of Background		**0.047**	**0.0007**	**0.47**	**0.098**	**6.3**	**0.006**	**6.3**	**0.0006**
PWQO[a]		**-**	**0.0009**	**0.30**	**-**	**6.5**	**1.0**	**100**	**0.005**
D-5	60	48%	0%	0%	0%	0%	0%	0%	5%
D-6	57	0%	5%	14%	65%	2%	0%	12%	0%
DS-18	60	0%	0%	15%	0%	0%	0%	20%	0%
M-01	50	0%	22%	56%	na	4%	0%	0%	24%
Q-09	60	52%	15%	na	na	0%	0%	17%	25%
Q-20	5	0%	20%	na	na	0%	0%	0%	0%
SC-01	16	0%	69%	0%	na	18%	0%	0%	0%
SR-01	5	0%	0%	na	na	0%	0%	0%	0%
SR-06	10	100%	0%	na	na	0%	0%	60%	0%
SR-08	60	0%	0%	na	na	2%	0%	97%	0%

[a] Provincial Water Quality Objectives (OMOEE 1994)

[b] Sulphate criterion based on BCMOE

na - Parameter not sampled at respective station.





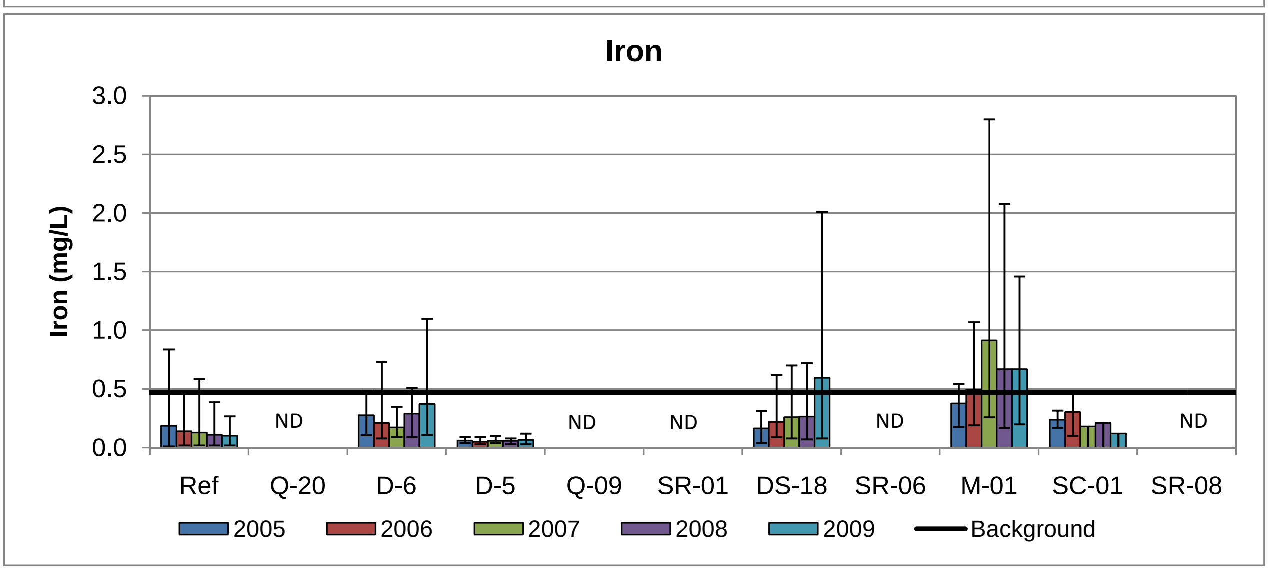
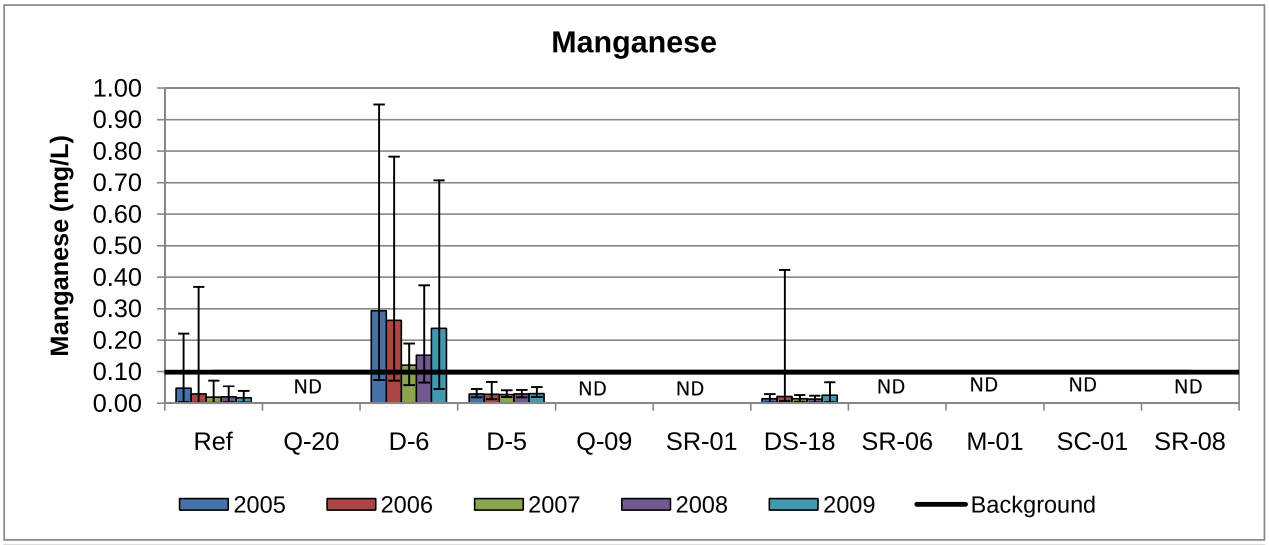
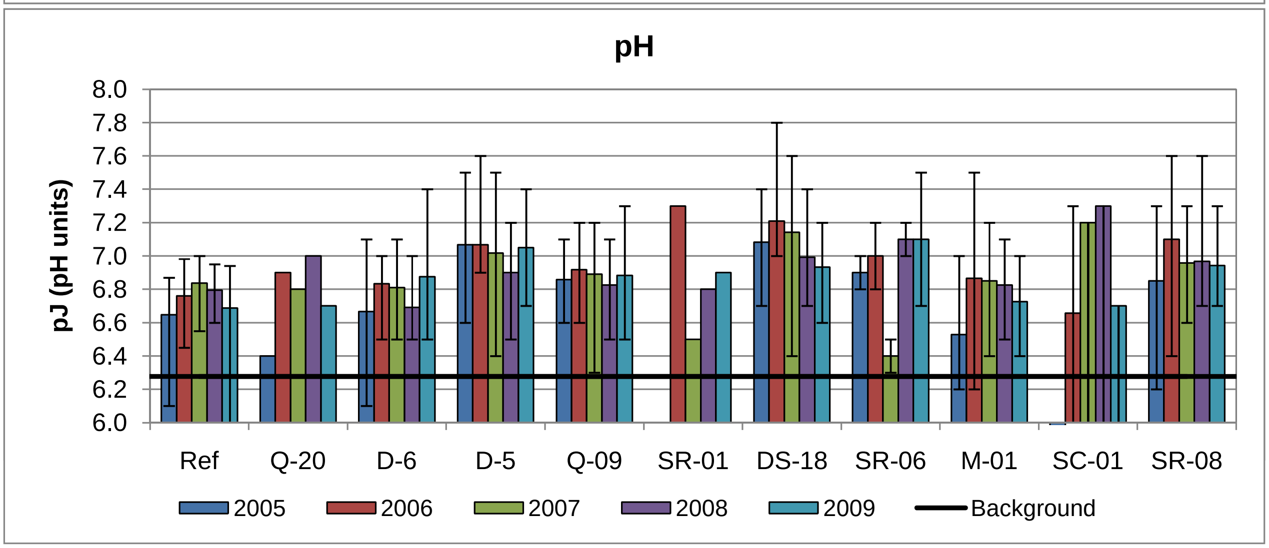


Figure 5.1: Mean, minimum and maximum water concentrations over time at mine exposed stations relative to pooled reference stations and water quality benchmarks. ND denotes no data available for that station.





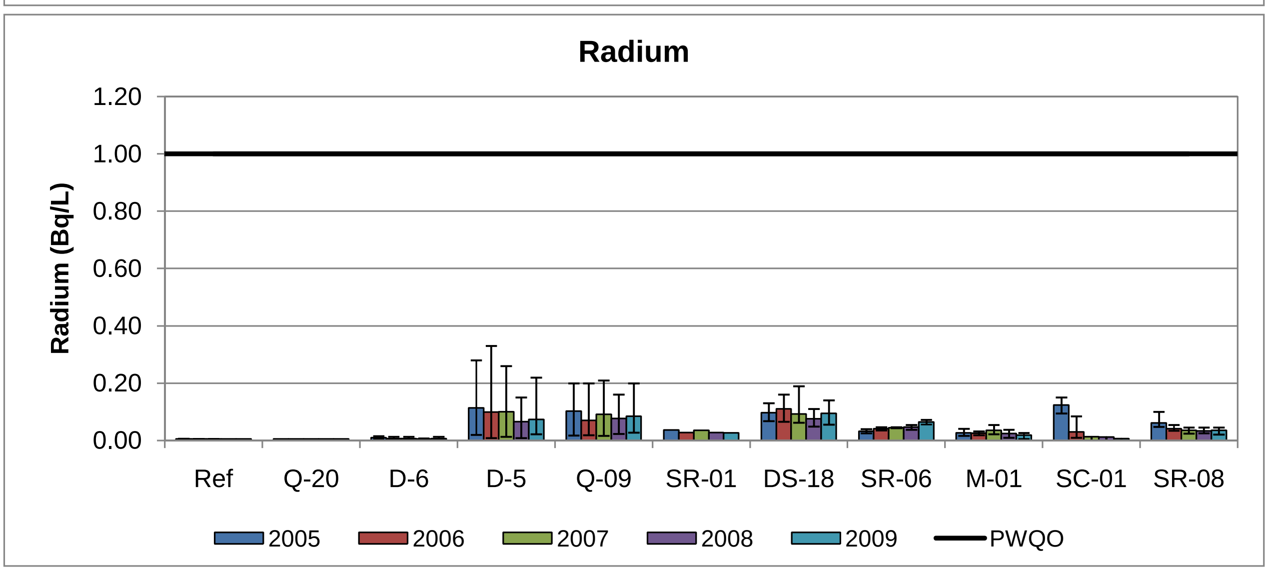
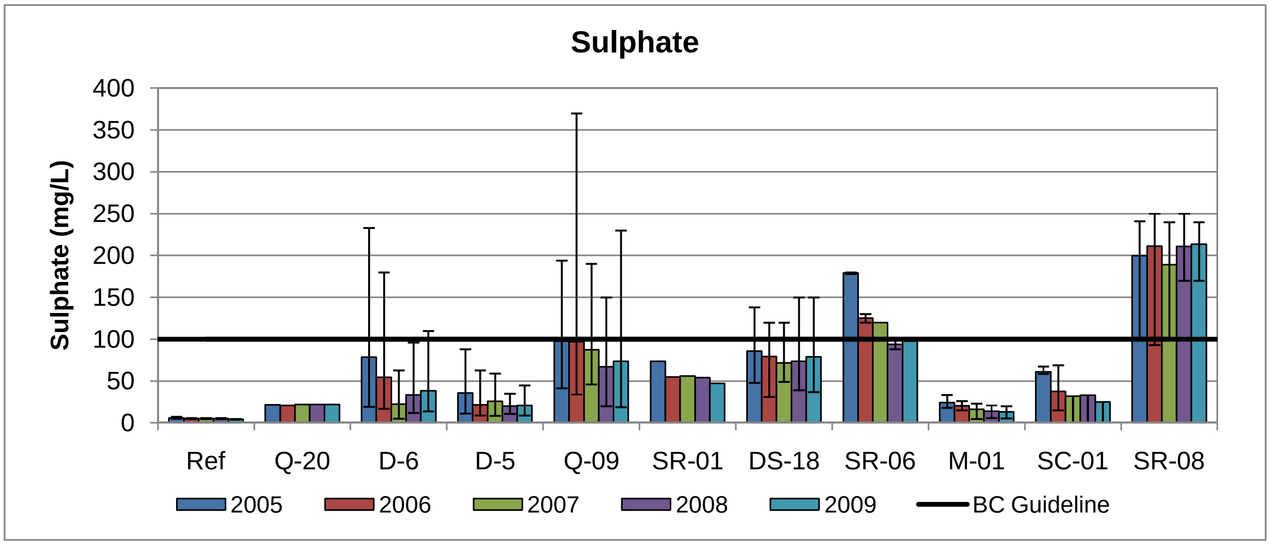


Figure 5.1: Mean, minimum and maximum water concentrations over time at mine exposed stations relative to pooled reference stations and water quality benchmarks. ND denotes no data available for that station.



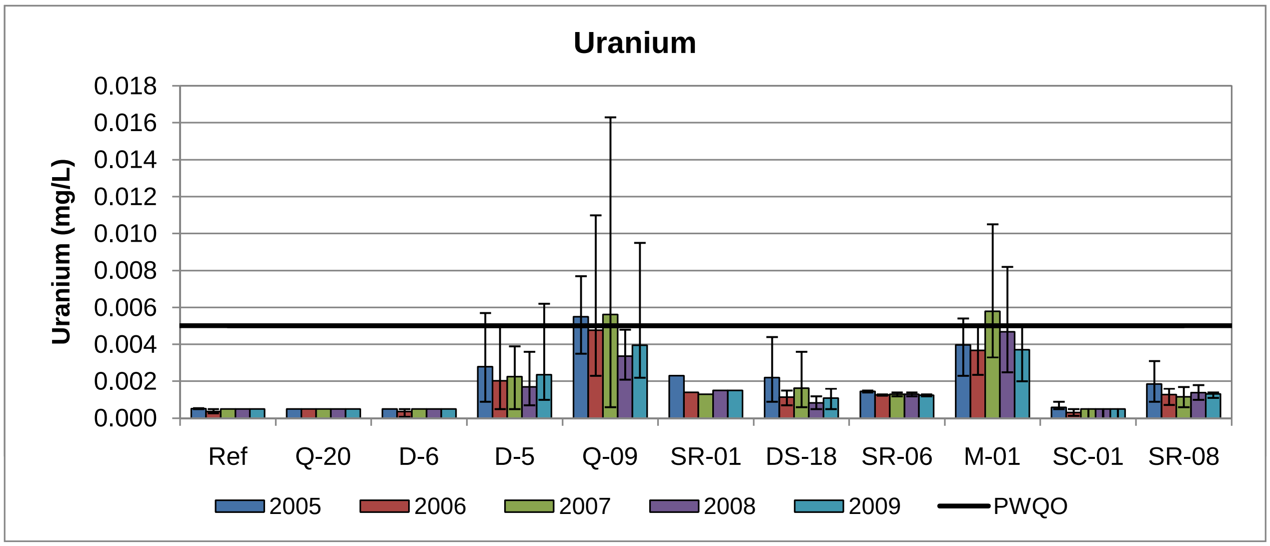


Figure 5.1: Mean, minimum and maximum water concentrations over time at mine exposed stations relative to pooled reference stations and water quality benchmarks. ND denotes no data available for that station.

Table 5.2: Summary of water quality trends[a] for Serpent River monitoring stations, 2000 to 2009.

Station ID	Number of Seasons Used in Common Trend[b]	Barium	Cobalt[c]	Iron[d,e]	Manganese[d]	pH[e]	Radium-226[c]	Sulphate	Uranium[c,e]
Reference Stations									
D-4	2	0.165	ND	0.645	0.621	-0.069	ND	-0.593	ND
P-22	2	0.435	ND	-	-	-0.038	ND	-0.515	ND
SR-05	10	-0.012	ND	-	-	0.070	ND	-0.786	ND
SR-14	1	-0.215	ND	-	-	0.0243	ND	-0.608	ND
SR-18	2	-0.099	ND	-	-	0.289	ND	-0.721	ND
SR-19	12	-0.191	ND	-	-	0.087	ND	-0.579	ND
Exposed Stations									
D-5	12	-0.124	ND	*-0.134*	-0.367	-0.011	-0.405	-0.412	*-0.276*
D-6	12	-0.093	ND	0.244	-0.046	*0.010*	-0.290	-0.258	ND
DS-18	12	-0.121	ND	0.368	-0.321	-0.084	-0.668	-0.442	-0.254
M01	10	-0.229	-0.219	-0.004	-	0.414	-0.660	-0.619	0.162
Q09	12	0.038	-0.292	-	-	-0.095	-0.374	-0.244	-0.379
Q20	1	0.622	ND	-	-	0.582	-0.834	-0.264	ND
SC-01	1 or 2	-0.360	ND	0.446	-	0.655	-0.739	-0.053	ND
SR-01	1	0.422	ND	-	-	0.387	-0.887	-0.967	-0.845
SR-06	2	0.984	ND	-	-	-0.572	0.394	-0.935	-0.977
SR08	12	0.172	ND	-	-	-0.076	-0.416	-0.539	-0.740

decreasing trend, significant at p<0.05

increasing trend, significant at p<0.05

[a] Based on rank correlation coefficients (rho) shown in table for common (combined) seasonal trends.

[b] Seasons used varied for substances based on suitability of data for trend analysis.

[c] ND denotes that this parameter was not included in the trend analysis for that particular station due to >50% non-detectable concentrations in the samples available for the analysis.

[d] "-" denotes that this parameter was not included in the trend analysis for that particular station due to the absence of data (e.g. there were <5 years worth of data for that parameter)

[e] Italic text mean monthly correlations were significantly different, but common trend value provided was not necessarily significant.

Water quality downstream of the TMAs is meeting EIS predictions. Recent concentrations of sulphate and radium-226 downstream of the TMAs were better than the 1999 cumulative predications or in the case of Stanleigh, the 2012 predicted values (Table 5.3). Observed trends reflected decreasing concentrations of both radium-226 and sulphate over time and therefore concentrations appear to be on target for achieving predicted values for 2099.

Generally, water quality downstream of the TMAs achieved receiving water criteria and is improving over time.

5.2 Sediment Quality

Substrate particle size characteristics were very consistent among lakes assessed within the SRWMP consisting of 10 to 15% clay, 45 to 50% silt and 30 to 35% sand (Figure 5.2). Sediment TOC ranged from about 4 % in McCarthy and Pecors lakes to a mean of 9% in the reference lakes (reference range 6.5 to 13.9).

Mean sediment metal concentrations downstream of the TMAs were typically less than the severe effect level (SEL) of the Provincial Sediment Quality Guideline's[5] but greater than the lower effect level (LEL) or background[6] concentrations (Table 5.4). The highest concentrations of most substances were found in McCabe Lake, where concentrations were above background and the LEL and, in the case of iron and manganese, higher than the SEL. Nordic Lake also had elevated concentrations of most substances relative to most other lakes assessed (Table 5.4). Typically Hough Lake had the lowest concentrations of the mine-exposed lakes, with concentrations of barium, cobalt, iron, manganese and uranium being below the background benchmark (Table 5.4).

In locations where sediment concentrations were above benchmarks, concentrations of barium, cobalt, iron, manganese and nickel appeared to decrease or remain stable over the past ten years (1999 to 2009) (Figure 5.3). However, statistical comparisons of 1999 versus 2009 sediment concentrations indicated few statistically significant differences (Appendix Table E.29). For example an apparent increase in nickel in McCabe Lake since 1999 was not significant (Appendix Table E.29). Similarly, uranium concentrations notably increased between 2004 and 2009 at all exposure locations, but there was no statistical difference in

[5] The PSQG were used, where available, for all substances monitored except uranium and radium-226 which were compared to SEL and LEL values cited in Thompson et al. (2005)

[6] The SRWMP background values were typically greater than the LEL, and were greater than the SEL for iron and manganese.

Table 5.3: Concentration predictions at SRWMP stations compared to 2009 values.

TMA	Predicted vs Measured	Year	Sulphate (mg/L)	Radium-226 (Bq/L)	Uranium[a] (mg/L)
SR-01	Cumulative Prediction [b]	1999	173	0.067	-
	Current	2009	47	0.026	0.0015
	Cumulative Prediction [b]	2099	23	0.042	-
DS-18	Cumulative Prediction [b]	1999	215	0.170	-
	Current	2009	79	0.094	0.0011
	Cumulative Prediction [b]	2099	53	0.051	-
SR-06	Current	2009	99	0.064	0.0013
	2012[c] Prediction	2012	32	0.1	0.0029
	Cumulative Prediction [b]	2099	11	0.026	-

[a] Predicted uranium values converted from Bq/L to mg/L.

[b] Prediction values for 1999 and 2099 based on cummulative effects assessment (CNSC 2002).

[c] The 2012 prediceted value represents the 2005 year prediction presented in the CSR (1997) because delays in construction and flooding of the TMA caused a shift in the representative time line for the graphs of predicted concentrations.

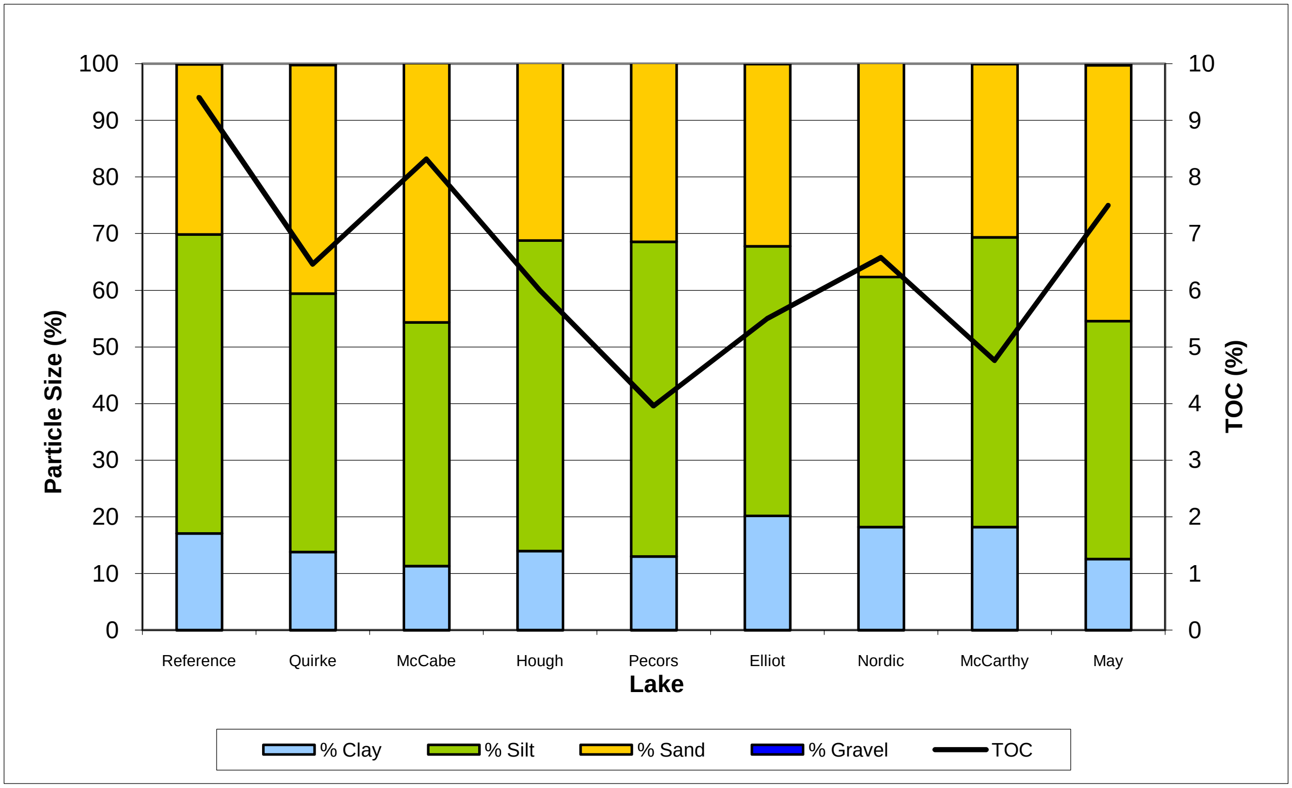


Figure 5.2: Sediment particle size distribution and total organic carbon content, SRWMP 2009.

Table 5.4: Summary of lake sediment quality relative to background concentrations and sediment quality guidelines, SRWMP 2009. Concentrations that were above background or LEL[a], whichever was higher, are highlighted in green. Values above the SEL[b] were highlighted in blue.

	Barium (mg/kg)				
Background	481				
LEL	-				
SEL	-				
	Lake	Mean	SD	Min	Max
Ranked Area Means	McCabe	2,090	1,879	380	4,200
	Quirke	706	430	240	1,400
	Semiwhite	432	112	310	560
	Nordic	294	98	130	390
	Elliot	218	65.0	130	300
	McCarthy	160	32.4	120	190
	May	143.2	67.6	96	260
	Rochester	138	21.7	110	160
	Dunlop	136	48.9	52	170
	Summers	98.6	13.5	86	120
	Pecors	98.0	17.6	75	120
	Ten Mile	81.6	24.7	51	110
	Hough	80.4	8.88	70	90

	Cobalt (mg/kg)				
Background	28.3				
LEL	-				
SEL	-				
	Lake	Mean	SD	Min	Max
Ranked Area Means	McCabe	175	82.0	76	290
	Nordic	109	49.0	25	150
	McCarthy	101	26.0	71	120
	Elliot	74	14.1	59	89
	Pecors	40	4.24	33	43
	Quirke	38.4	26.7	20	84
	May	31.8	10.7	21	49
	Hough	26.8	3.27	22	30
	Summers	19.6	8.20	10	30
	Rochester	19.0	4.06	13	24
	Dunlop	14.4	5.00	6.8	19
	Semiwhite	12.6	1.98	9.9	14
	Ten Mile	6.86	1.34	5.0	8.2

	Iron (mg/kg) [c]				
Background	54,783				
LEL	20,000				
SEL	40,000				
	Lake	Mean	SD	Min	Max
Ranked Area Means	McCabe	75,400	17,358	51,000	100,000
	May	73,600	15,805	59,000	100,000
	Nordic	69,000	27,902	33,000	110,000
	Quirke	57,800	11,584	42,000	68,000
	Elliot	52,000	9,460	10,000	63,000
	Hough	51,400	8,678	39,000	60,000
	McCarthy	49,800	12,276	33,000	65,000
	Semiwhite	34,400	13,088	20,000	49,000
	Pecors	33,400	10,359	18,000	43,000
	Rochester	30,800	10,849	24,000	50,000
	Dunlop	28,400	12,876	11,000	42,000
	Summers	28,400	11,675	17,000	46,000
	Ten Mile	9,700	1,034	8,200	11,000

	Manganese (mg/kg) [c]				
Background	6,918				
LEL	460				
SEL	1,100				
	Lake	Mean	SD	Min	Max
Ranked Area Means	Nordic	19,460	10,904	300	26,000
	McCabe	16,800	11,862	2,000	35,000
	McCarthy	12,360	3,694	8,000	16,000
	Elliot	10,760	6,163	3,000	18,000
	May	5,340	2,735	2,400	9,100
	Semiwhite	5,140	2,988	1,900	8,400
	Quirke	4,140	2,387	2,000	8,000
	Pecors	3,060	1,387	1,500	4,300
	Hough	2,880	606	2,000	3,400
	Dunlop	2,534	1,712	670	5,300
	Summers	1,384	981	290	2,600
	Rochester	710	112	590	850
	Ten Mile	518	141	340	1,060

	Nickel (mg/kg)				
Background	29.7				
LEL	23.4 [d]				
SEL	484 [d]				
	Lake	Mean	SD	Min	Max
Ranked Area Means	McCabe	100.8	36.4	68	160
	Elliot	53.6	5.37	47	59
	Nordic	44.0	6.82	37	52
	McCarthy	43.2	7.22	33	53
	Hough	40.2	3.03	37	45
	May	38.8	14.0	21	59
	Pecors	35.4	6.47	26	42
	Quirke	25.4	8.82	16	38
	Semiwhite	23.6	2.07	21	26
	Rochester	22.8	3.19	18	27
	Dunlop	21.4	5.46	12	26
	Summers	18.6	1.82	17	21
	Ten Mile	17.6	4.72	12	22

	Uranium (mg/kg)				
Background	6.50				
LEL	104.4 [d]				
SEL	5,874 [d]				
	Lake	Mean	SD	Min	Max
Ranked Area Means	Quirke	352	144	180	530
	McCabe	326	149	230	590
	Elliot	170	40.0	120	220
	Nordic	154	41.6	110	220
	McCarthy	138	27.7	110	180
	Pecors	114	11.4	100	130
	May	92.4	13.0	75	110
	Hough	87.4	5.32	78	91
	Rochester	5.04	1.79	3.1	7
	Semiwhite	4.16	0.439	3.6	4.8
	Dunlop	3.82	1.23	1.8	4.8
	Ten Mile	3.32	0.779	2.4	4.3
	Summers	2.70	0.235	2.4	2.9

	Ra-226 (Bq/g)				
Background	0.27				
LEL	0.6 [d]				
SEL	14.4 [d]				
	Lake	Mean	SD	Min	Max
Ranked Area Means	McCabe	13.8	1.30	12	15
	Nordic	4.78	1.68	2.3	6.8
	Quirke	3.64	2.26	1.1	7
	May	2.40	0.806	1.2	3.3
	Hough	1.90	0.367	1.6	2.5
	Elliot	1.592	0.364	0.96	1.9
	McCarthy	1.552	0.653	0.86	2.3
	Pecors	0.672	0.211	0.38	0.92
	Summers	0.158	0.0746	0.09	0.28
	Rochester	0.154	0.0546	0.10	0.24
	Semiwhite	0.154	0.0796	0.07	0.27
	Dunlop	0.088	0.0363	0.05	0.14
	Ten Mile	0.064	0.0230	0.04	0.1

Selected background value or LEL (whichever was higher) or observed concentrations that exceeded selected background value or LEL.

SEL or concentrations that exceeded the SEL.

Bold text indicates reference lakes

[a] Lowest effect level, Ontario Provincial Sediment Quality Guidelines (OMOE 1993).

[b] Severe effect level, Ontario Provincial Sediment Quality Guidelines (OMOE 1993).

[c] Values not compared to SEL since upper range of background values exceeds SEL.

[d] Guidelines proposed by Thompson et al. (2005)

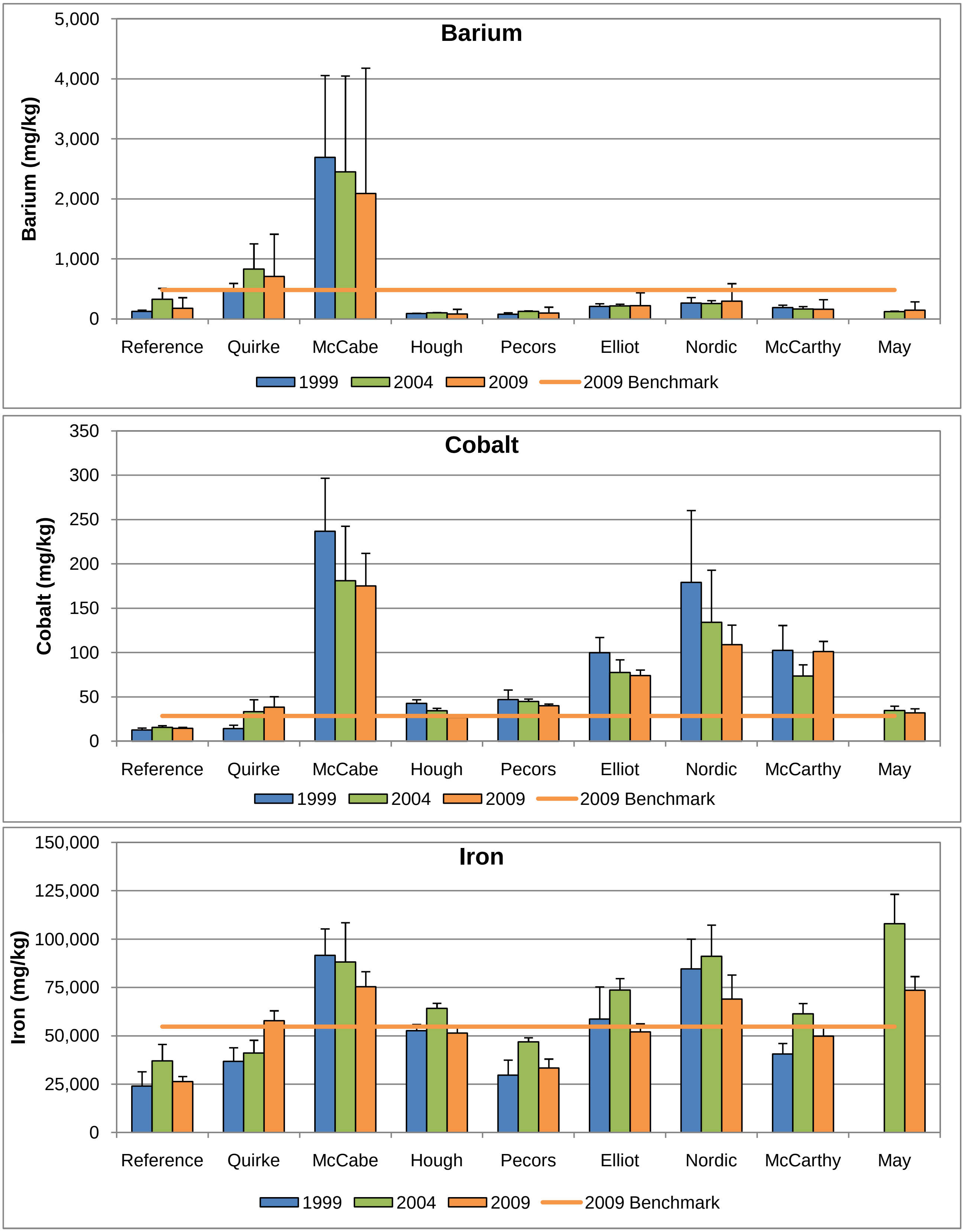


Figure 5.3: Mean lake sediment concentrations (± SE) for 1999 (cycle 1, n=3), 2004 (cycle 2, n=3), and 2009 (cycle 3, n=5).

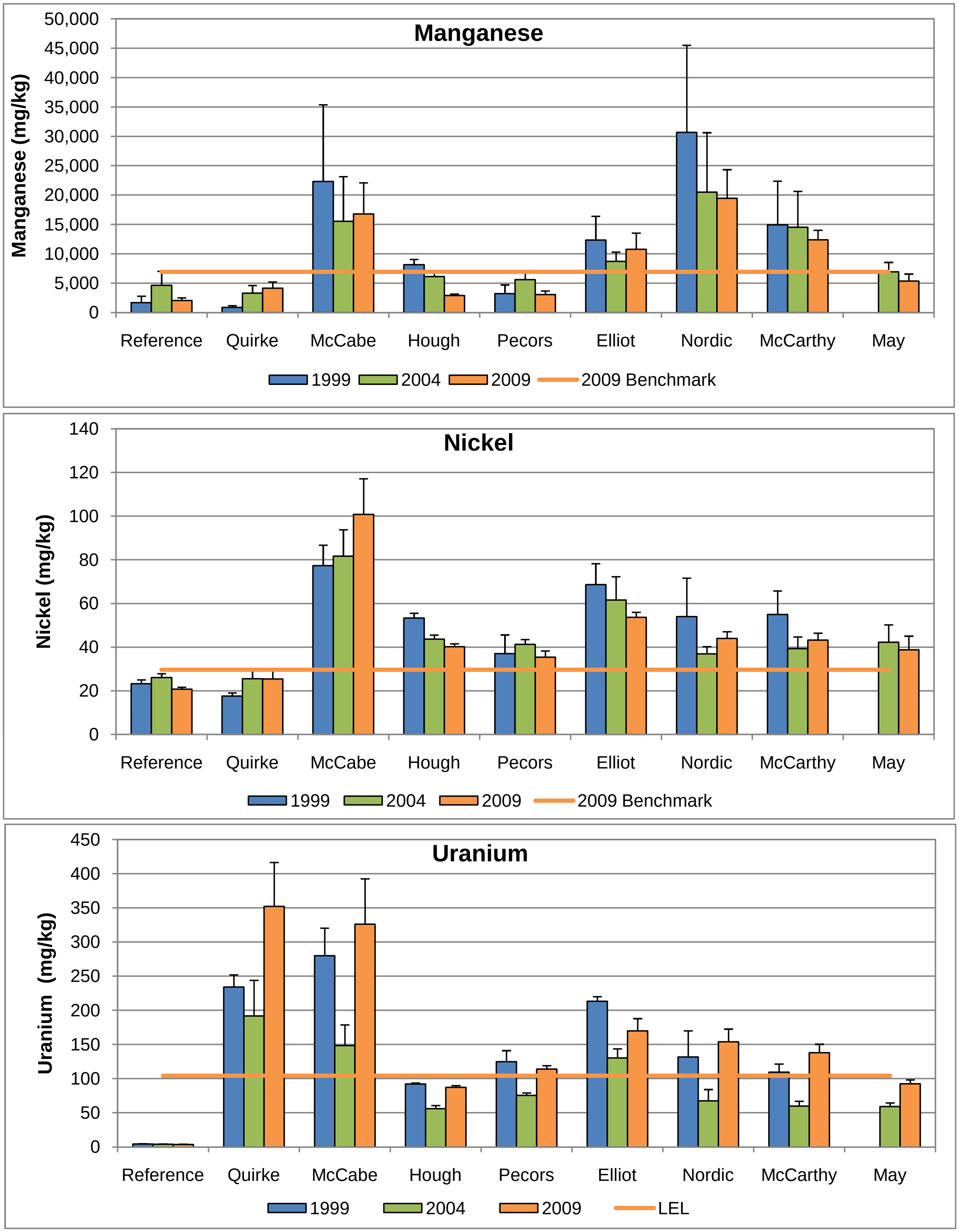


Figure 5.3: Mean lake sediment concentrations (± SE) for 1999 (cycle 1, n=3), 2004 (cycle 2, n=3), and 2009 (cycle 3, n=5).

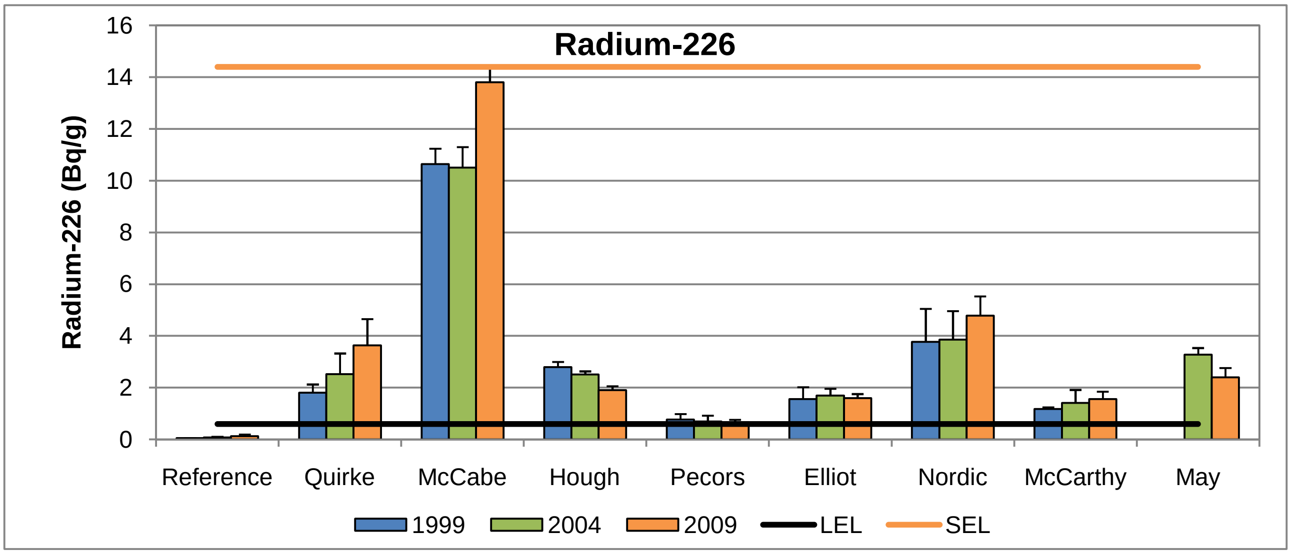


Figure 5.3: Mean lake sediment concentrations (± SE) for 1999 (cycle 1, n=3), 2004 (cycle 2, n=3), and 2009 (cycle 3, n=5).

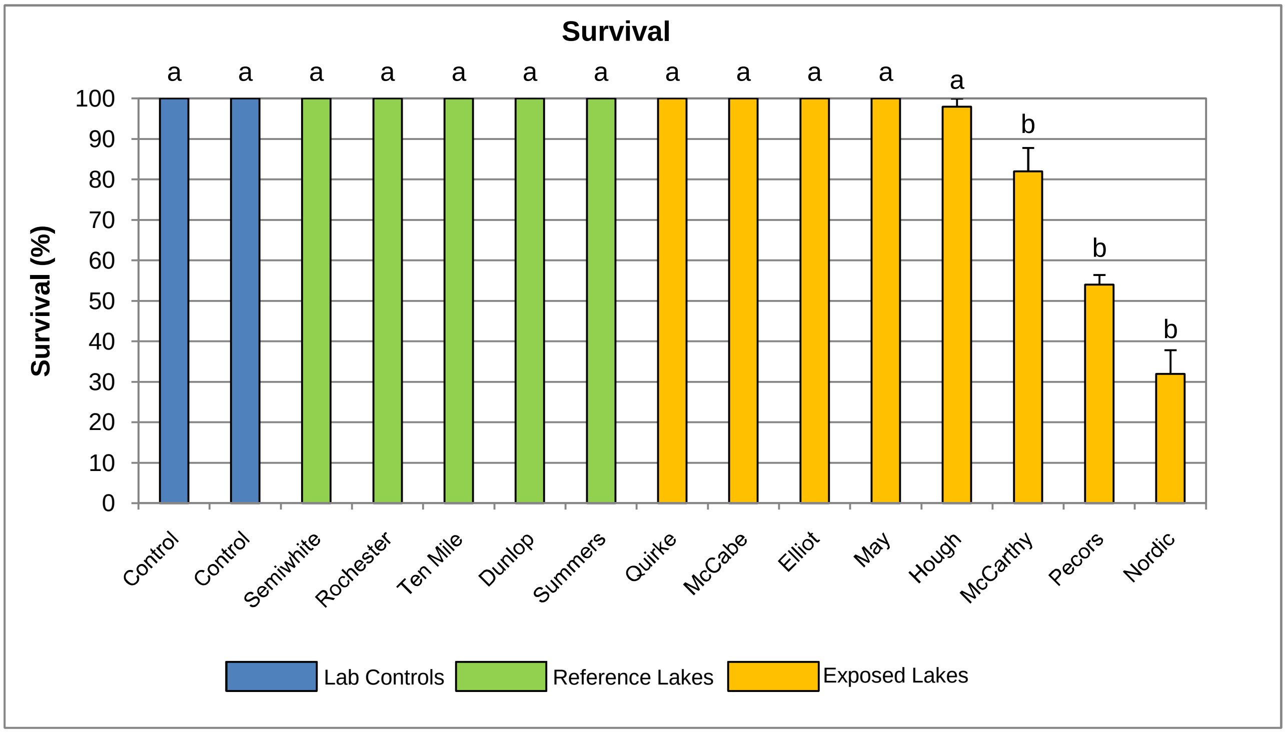
uranium concentrations between 1999 and 2009 in any of the lakes sampled (Appendix Table E.29). The only significant differences in mine-exposed lakes was an increase in iron and manganese in Quirke Lake, an increase in radium-226 in McCabe Lake and a decrease in cobalt, manganese, nickel and radium-226 concentrations in Hough Lake (1999 vs. 2009),. Overall, the data indicate a very slow rate of change in sediment quality.

Sediment quality was further assessed through toxicity testing using *Hyalella azteca* at all lakes monitored within the SRWMP and using *Chironomus dilutus* at selected lakes (Figures 5.4 and 5.5 respectively). Survival of *Hyalella azteca was* significantly reduced by exposure to sediments from McCarthy, Pecors and Nordic Lakes relative to the laboratory control and SRWMP reference lake sediments. In addition, sediment from these lakes and Quirke Lake produced statistically reduced growth in *Hyalella azteca* (Figure 5.4). These results did not correspond with sediment chemistry since McCarthy and Pecors lakes had some of the lowest sediment concentrations of mine-related substances. The observed response may be related to direct or indirect effects of TOC, which was lower in McCarthy and Pecors Lakes (4.6 and 5.5%) than in the lab control (8.9%) or the reference lakes (6.5 to 13%). Depending on the substance, TOC may influence the bioavailability of metals in sediment. Growth and survival of *Chironomus dilutus* did not differ between exposure and reference lakes (Figure 5.5).

5.3 Benthic Invertebrate Communities

5.3.1 Data Exploration

Raw benthic community data from 1999, 2004 and 2009 were combined for preliminary exploration of the data. Where taxonomy changed between years, taxa were, if necessary, collapsed to a coarser level of identification. Twenty-three taxa were retained and used in correspondence analysis (CA). The first three axes of the CA contained 36.2% of the total inertia (or variance) in the original benthic abundance data set (Appendix Table E.37b). The first axis explained 14.4% of the variance and summarized variation principally in the taxa *Rhyacodrilus montana* (an oligochaete worm), *Bezzia* (a “No-See-Um” biting midge), and immature tubificids both with, and without, diagnostic hair chaetae (Appendix Table E.37a). CA Axis-2 summarized 12.7% of the variation, and positive scores on this axis were characterized by high relative abundance of immature tubificids with hair chaetae as well as *Dicrotendipes* and *Paracladopelma* chironomids, while strongly negative scores were associated with immature tubificids without hair chaetae, *Chaoborus punctipennis*, and Harpacticoida (copepods). The third CA axis analysed explained 9.1% of the variation and principally described a continuum of higher abundance of *Pisidium* fingernail clams (low CA



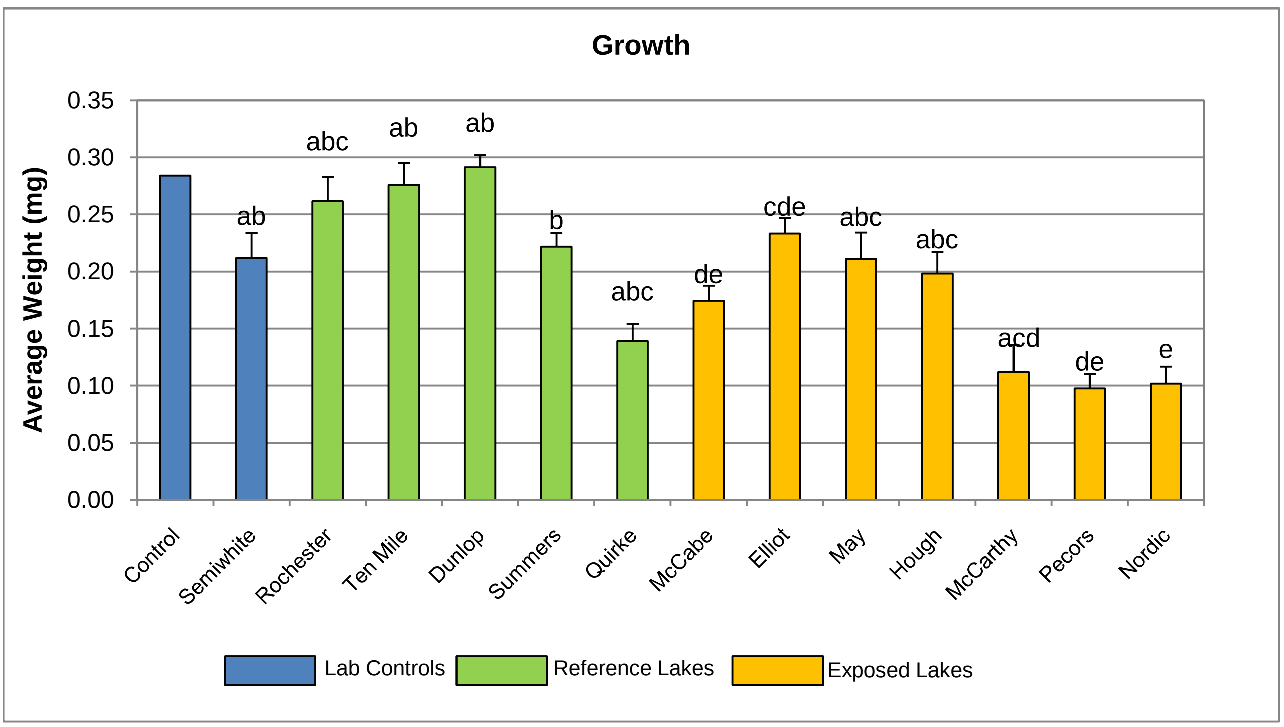


Figure 5.4: Survival and growth (+ SE) of *Hyalella azteca* exposed to sediment samples, SRWMP 2009. Lakes with similar letters above bars were not significantly different (p<0.05).

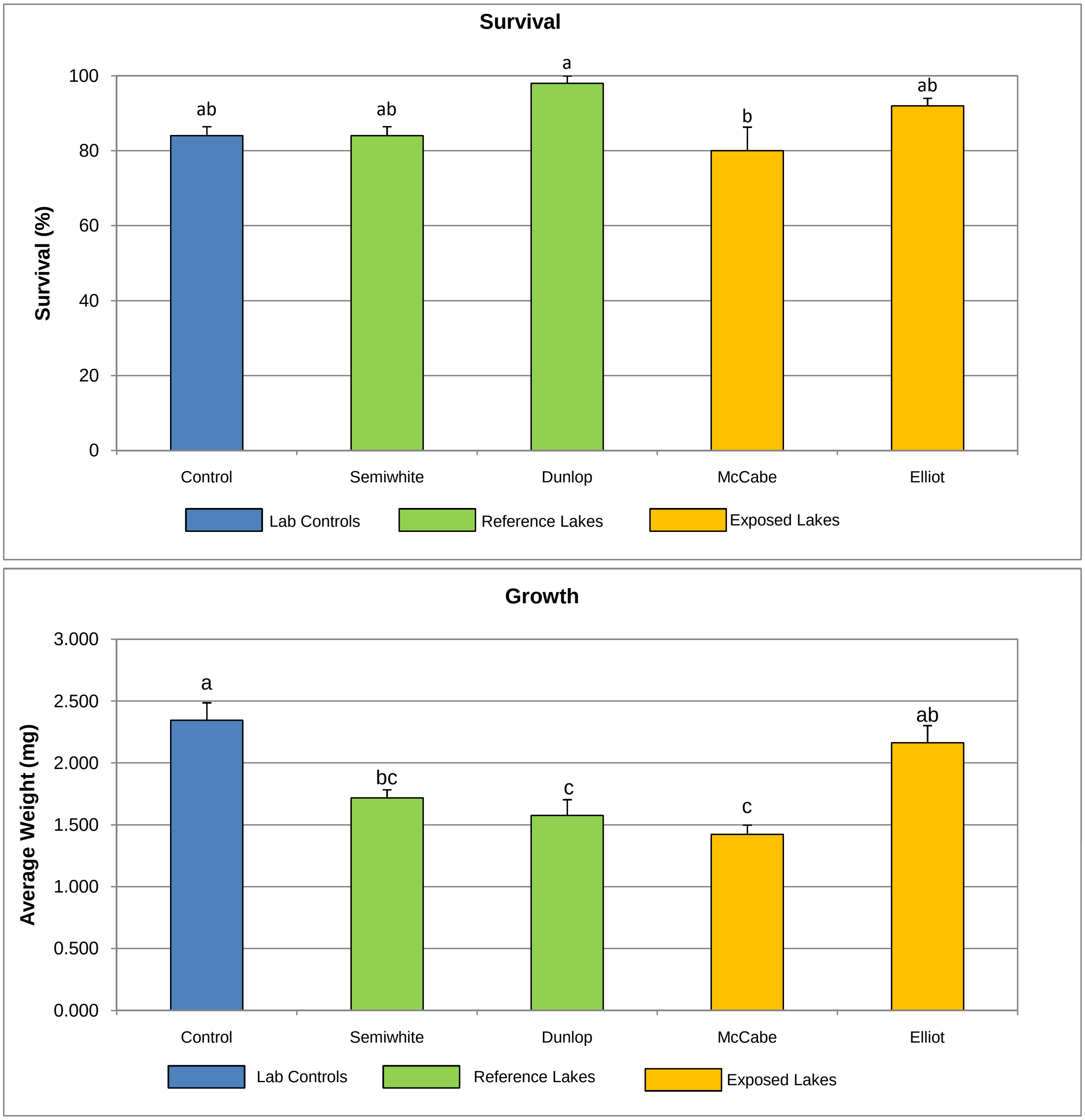


Figure 5.5: Survival and growth (+ SE) of *Chironomus dilutus* exposed to sediment samples, SRWMP 2009. Lakes with similar letters above bars were not significantly different (p<0.05).

Axis-2 scores) to higher abundance of immature tubificids lacking hair chaetae (positive CA Axis-2 scores; Appendix Table E.37a).

CA scores, plotted as means ± 95% confidence intervals for each lake and year, showed that reference area Rochester Lake (RL) had benthic invertebrate communities which, in all three sample years, were considerably different from other reference areas as summarized by CA axes 2 and 3 (Figure 5.6, Appendix Figure E.15). RL stations had very low CA Axis-2 scores, indicating high relative abundance of immature tubificids lacking hair chaetae, and also high relative abundance of the planktonic phantom midge *Chaoborus punctipennis* and harpacticoid copepods. Other reference lakes had low relative abundance of these taxa, and higher relative abundance of immature tubificids with hair chaetae, and of the chironomid larvae *Dicrotendipes* and *Paracladopelma*. Accordingly, these other reference lakes had higher CA Axis-2 scores than RL (Figure 5.6). On CA Axis-3, RL had higher values than other reference lakes, indicating again the dominance of immature tubificids lacking hair chaetae, whereas other reference lakes tended to have few of these tubificids, but higher relative abundance of *Pisidium* fingernail clams.

It may be relevant that RL had more organic sediments (14.2% TOC) and lower water column DO near the sediment-water interface (30%) than all other lakes (4.0-9.9% TOC and 46-132% DO) (Appendix Tables E.12 and E.31).

The clear difference between the benthic community found at RL and the benthic community in other reference lakes would inflate the variance around the means of the community metrics for reference lakes, resulting in a less rigorous test of the hypothesized differences between reference and exposure lakes in the study. Accordingly, a decision was made to remove RL stations from the reference lake data set for all comparisons with mine effluent exposure lakes. Therefore, for reference/exposure comparisons, a pooled reference mean was calculated from the mean values of the reference lakes (n=4 lakes, omitting RL as described above) in each study year, and these data were compared to mean values for each exposure lake (n=5 replicate stations) using *a priori*, user-defined contrasts in ANOVA.

5.3.2 Reference/Exposure Comparisons for 2009 Data

In 2009, the only exposure lake showing a statistically significant difference in benthic community density from the pooled reference community was Quirke Lake (Figure 5.7a), where density was reduced by less than two standard deviations (2SD) from the reference mean (Appendix Table E.43), which may not be biologically meaningful. It is noteworthy, however, that, while not statistically significant in individual comparisons (p>0.1), all exposure lakes except Pecors Lake (PL) had lower mean density than the reference mean (Appendix

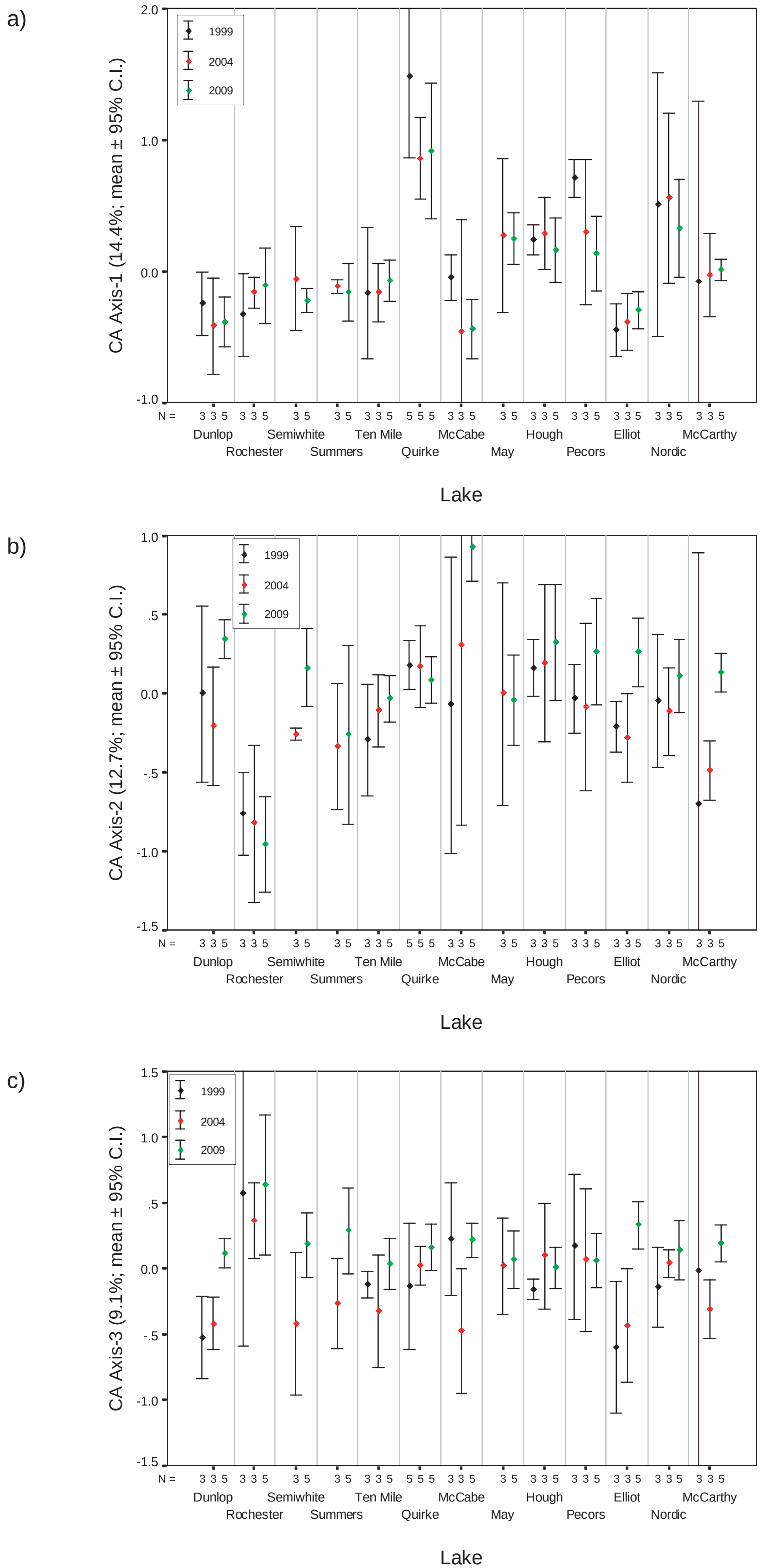


Figure 5.6: Exploratory correspondence analysis of benthic community data at Serpent River watershed areas: 1999, 2004, 2009.

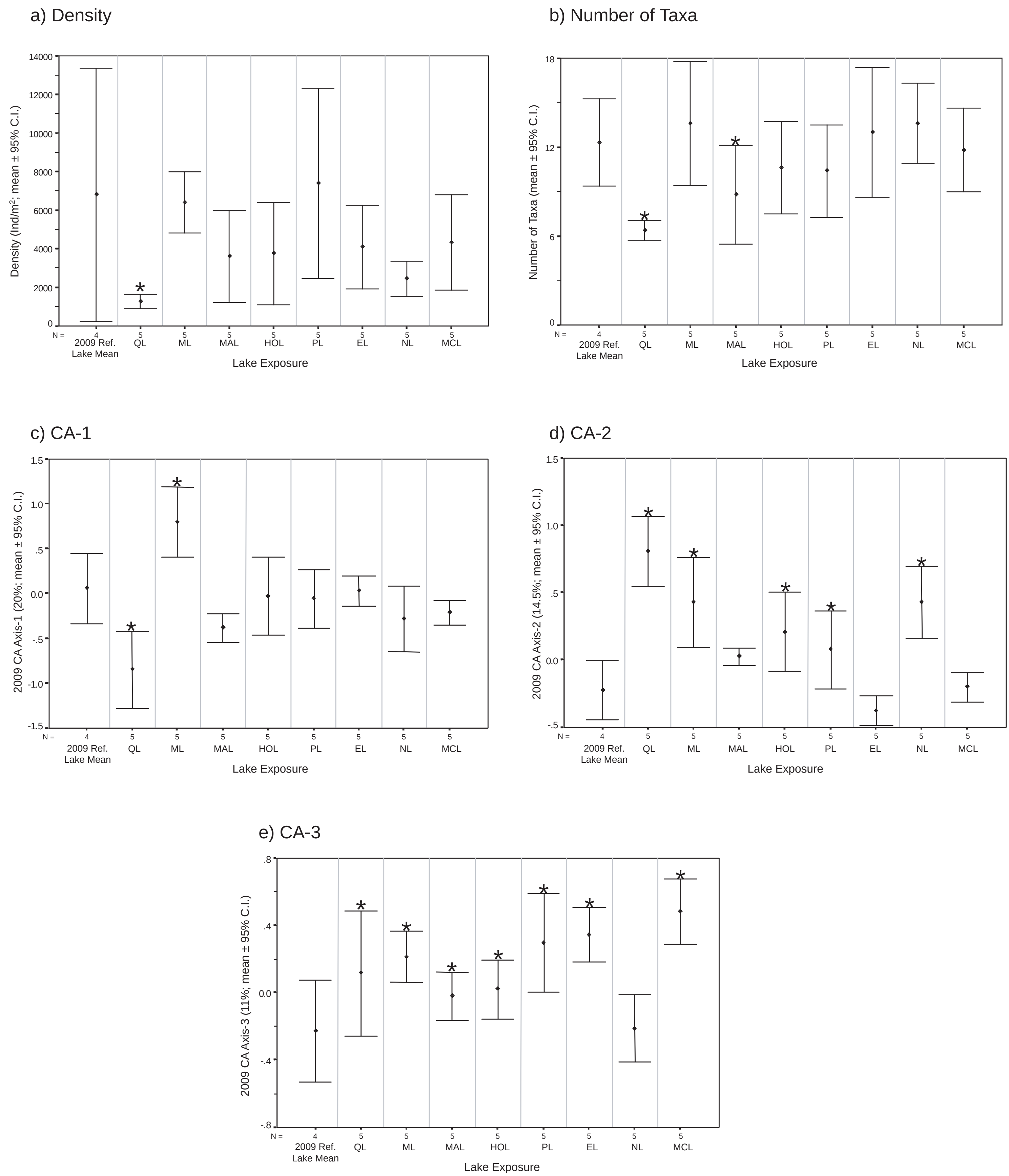


Figure 5.7: Benthic community characteristics in mine-exposed lakes of the Serpent River watershed relative to the pooled reference lakes. Asterisks (*) indicate exposure lakes that were statistically different from reference for each metric (p<0.1).

Table E.43). This pattern of differences, significant by Wilcoxon Signed ranks (WSR; $n = 8$; $p < 0.05$) test, suggests mild impairment of density may be characteristic of effluent exposed lakes.

Both Quirke Lake (QL) and May Lake (MAL) had significantly fewer benthic invertebrate taxa when compared to the reference lake mean, though only the difference at QL was more than 2SD from the reference mean (Appendix Table E.43). The pattern of differences (positive or negative change from reference) was not significant (WSR test, $n = 8$; $p > 0.05$).

CA of 2009 benthic abundance data explained 46.5% of the variance in abundance among stations (Appendix Table E.39a). In the exposure lakes, QL, MAL, and NL had significantly lower CA Axis-1 scores than the reference mean, with a departure of 3.7 SDs from the reference mean for QL, but less than 2 SDs for MAL and NL (Figure 5.7c; Appendix Table E.43). Low CA Axis-1 values indicated higher relative abundance of such taxa as the oligochaete worm *Rhyacodrilus montana* and midge larvae of the genus *Chironomus* (Appendix Table E.39a). In contrast, McCabe Lake (ML) had significantly higher CA Axis-1 scores than reference (3SD), indicating higher relative abundance than in reference of immature tubificids with hair chaetae, and of *Dicrotendipes* chironomids (Figure 5.7c; Appendix Table E.43). The other exposure lakes had mean CA Axis-1 values similar to reference.

All exposure areas, except Elliot Lake (EL) and McCarthy Lake (MCL) had CA Axis-2 values significantly higher than the pooled mean of the reference lakes, with all differences being more than 2SD from the reference mean except at May Lake (MAL) (Figure 5.7d; Appendix Table E.43). High values of CA Axis-2 indicated benthic communities with greater abundance of *Dicrotendipes* chironomids, and some oligochaete taxa (immature tubificids with hair chaetae, and *Rhyacodrilus montana*) (Appendix Table E.39a). The pattern of greater CA Axis-2 values at exposure lakes than the reference lakes was statistically significant (WSR test: $n = 8$; $p < 0.05$). The low CA-2 scores of reference, EL and MCL corresponded to higher abundance of the facultative planktonic predatory phantom midge larva *Chaoborus punticpennis* (Appendix Table E.39a).

All exposure areas with the exception of NL had significantly greater CA Axis-3 scores than the pooled reference lakes (Figure 5.7; Appendix Table E.43). Strongly positive scores on this axis were indicative of greater abundance of *Chaoborus punctipennis*, whereas strongly negative scores indicated greater abundance of harpacticoid copepods (Appendix Table E.39a). Only 4 exposure lakes (ML, PL, EL, MCL) differed from the reference area mean by more than 2 reference SDs (Appendix Table E.43), but overall, the pattern of differences from reference were uniformly positive across all exposure lakes, and therefore constituted a significant overall reference-exposure difference (Appendix Table E.43; WSR test, n = 8; $p < 0.05$).

In summary, the benthic invertebrate communities of all mine-exposed lakes were statistically different from the mean of reference lakes with respect to at least one of the benthic community metrics (Tables 5.5 and 5.6). The communities in Quirke, McCabe, and May lakes showed more significant differences from the mean reference community than the other lakes, but the magnitudes of difference were larger at Quirke and McCabe than May when differences were expressed as a percentage of the reference mean or the number of reference area standard deviations. The benthic communities in Elliot and McCarthy Lakes were most similar to the mean reference community, differing only with respect to CA-3. Overall, the exposure areas showed a pattern of lower benthic invertebrate density and CA1 scores, along with higher CA2 and CA3 scores than the pooled reference areas (Table 5.4), indicative of a mine-related signature.

5.3.3 Correlations between Benthic Metrics and Supporting Measures

A total of 26 correlations between habitat variables and benthic community characteristics were significant at $p<0.05$, but only nine were significant at a more stringent level of $p<0.001$ (Table 5.7). Most correlations were associated with one or more of four patterns (Appendix Figure E.18):

1. Influence of exposed Quirke Lake (QL) stations, which had relatively greater sample depth, secchi depth, and sediment uranium concentrations, and where mean benthic invertebrate density, number of taxa, and CA-1 scores were low and CA2 scores were high relative to the other reference and exposed lake benthic invertebrate communities.

2. Influence of reference Ten Mile Lake (TML) stations, which had relatively large station depth, Secchi depth, and sediment TOC, as well as low metal levels.

3. Influence of McCabe Lake stations (ML), which had highest or second highest mean sediment concentrations of all mine-related substances (radium-226, barium, cobalt, iron, manganese, nickel, uranium), and high scores on all three CA axes.

4. Generally higher sediment concentrations of mine-related substances, and higher water pH, along with higher CA2 and CA3 scores in mine-exposed compared to reference lakes.

Table 5.5: Summary of benthic community comparisons for 2009, showing the magnitude of difference from reference (as percent of reference mean) and differences that were statistically significant (shaded).

Lake	Code	Density	No. of Taxa	Correspondence Analysis (2009 data only)		
				CA1	CA2	CA3
Quirke	QL	-81	-48	-1521	456	150
McCabe	ML	-6	11	1231	289	193
May	MAL	-47	-28	-739	110	90
Hough	HOL	-45	-14	-150	191	109
Pecors	PL	8	-15	-200	133	229
Elliot	EL	-40	6	-53	-67	251
Nordic	NL	-64	11	-579	288	7
McCarthy	MCL	-37	-4	-463	11	311

Table 5.6: Summary of benthic community comparisons for 2009, showing lakes that differed differed signficantly from reference (√) and cases where such differences were more than two reference area standard deviations (# SDs in parentheses).

Lake	Code	Density	No. of Taxa	Correspondence Analysis (2009 data only)		
				CA1	CA2	CA3
Quirke	QL	✓	✓ (-3.2)	✓ (-3.7)	✓ (7.6)	✓
McCabe	ML			✓ (3.0)	✓ (4.8)	✓ (2.3)
May	MAL		✓	✓	✓	✓
Hough	HOL				✓ (3.2)	✓
Pecors	PL				✓ (2.2)	✓ (2.7)
Elliot	EL					✓ (3.0)
Nordic	NL			✓	✓ (4.8)	
McCarthy	MCL					✓ (3.7)

Table 5.7: Correlations between benthic metrics and sediment measures, SRWMP 2009.

		Barium (mg/kg)	Cobalt (mg/kg)	Iron (mg/kg)	Manganese (mg/kg)	Nickel (mg/kg)	Ra-226 (Bq/g)	Uranium (mg/kg)	TOC (%)	Depth (m)	Secchi Depth (m)	DO (% sat)	pH	Fines (%; silt + clay)
Density (Ind./m2)	Pearson Correlation	-0.006	-0.130	-0.375	-0.190	-0.067	-0.086	-0.285	-0.038	-0.186	0.016	0.204	-0.134	0.190
	Sig. (2-tailed)	0.9655	0.3235	0.0032	0.1456	0.6131	0.5117	0.0271	0.7709	0.1542	0.9029	0.1173	0.3312	0.1450
	N	60	60	60	60	60	60	60	60	60	60	60	55	60
Number of Taxa	Pearson Correlation	0.105	0.109	-0.141	0.118	0.104	0.112	-0.225	-0.084	-0.450	-0.336	-0.182	0.001	0.173
	Sig. (2-tailed)	0.4249	0.4069	0.2830	0.3710	0.4310	0.3929	0.0842	0.5224	0.0003	0.0087	0.1635	0.9932	0.1850
	N	60	60	60	60	60	60	60	60	60	60	60	55	60
2009 CA Axis-1 (20.0%)	Pearson Correlation	0.346	0.234	-0.092	0.141	0.381	0.347	-0.150	0.168	-0.496	-0.258	-0.087	-0.003	0.209
	Sig. (2-tailed)	0.0067	0.0722	0.4827	0.2829	0.0026	0.0066	0.2532	0.1997	0.0001	0.0469	0.5100	0.9815	0.1097
	N	60	60	60	60	60	60	60	60	60	60	60	55	60
2009 CA Axis-2 (14.5%)	Pearson Correlation	0.377	0.254	0.386	0.217	0.204	0.500	0.578	-0.163	0.435	0.328	0.213	0.492	-0.342
	Sig. (2-tailed)	0.0030	0.0500	0.0023	0.0961	0.1173	0.0000	0.0000	0.2147	0.0005	0.0106	0.1030	0.0001	0.0075
	N	60	60	60	60	60	60	60	60	60	60	60	55	60
2009 CA Axis-3 (11.%)	Pearson Correlation	0.127	0.384	0.331	0.265	0.371	0.213	0.411	-0.381	-0.049	-0.438	-0.413	0.212	-0.192
	Sig. (2-tailed)	0.3335	0.0025	0.0097	0.0405	0.0035	0.1026	0.0011	0.0027	0.7119	0.0005	0.0010	0.1204	0.1409
	N	60	60	60	60	60	60	60	60	60	60	60	55	60

Correlation is significant at the 0.0014 level (2-tailed, p = 0.05 adjusted for 35 simultaneous tests).
Correlation is significant at the 0.05 level (2-tailed).
Correlation is significant at the 0.01 level (2-tailed).

Overall, the correlation analysis indicated that reference-exposure differences may be attributable to a combination of mine-related and/or non-mine-related factors and the specific causal factors likely differ among lakes.

5.3.4 Comparison of 2009 Benthic Metrics to Previous Years (1999, 2004)

Combining all reference and exposure station data within study years, excluding Rochester Lake, the five benthic community metrics (density, number of taxa, CA1, CA2, CA3) were tested by ANOVA across the three years for which data are available: 1999, 2004, 2009. Significant year-to-year variation was found for all five metrics (Appendix Table E.44). Pair-wise, post-hoc comparisons of year means indicated that benthic density increased monotonically over the 3 years, though only the comparison of 1999 to 2009 resulted in a significant post-hoc comparison (Figure 5.8a; Appendix Table E.45). Number of taxa was significantly lower in 1999 than in either 2004 or 2009, with the latter two years having similar numbers of taxa (Figure 5.8b).

In Correspondence Analysis (CA), CA Axis-1 score decreased monotonically over the three cycles of study (Figure 5.8c), but only the comparison of 1999 to 2009 was statistically significant ($p = 0.08$; Appendix Table E.45). This is supportive of a trend over these years from a community with high relative abundance of *Rhyacodrilus montana* and *Bezzia*, to a community more dominated by immature tubificids with hair chaetae, and by *Cyclocalyx* fingernail clams. CA Axis2 scores remained constant in 1999 and 2004, but decreased significantly in 2009, indicating a community shift from *Pisidium* and *Chaoborus* dominance in the former 2 years, to dominance by *Dicrotendipes* and *Paracladopelma* chironomids, along with immature tubificids with hair chaetae, and *Cyclocalyx* (Figure 5.8d). Year-to-year variation on CA Axis-3 scores also showed consistency between 1999 and 2004, again with a significant change in 2009. This trend indicated again a move towards a *Cyclocalyx* dominated community from one with higher relative abundance of immature tubificids with hair chaetae, *Dicrotendipes*, and *Pisidium*. The apparent contradictions in CA Axis-2 and CA Axis-3 represent, in the case of CA Axis-3, variation in the abundance of immature tubificids (+ hair chaetae) and *Dicrotendipes* that was not correlated with the abundance of *Cyclocalyx*.

For the five metrics considered above, the pattern of deviation from the reference mean in each year was then examined for the exposure lakes' means using Wilcoxon Signed Ranks (WSR) test. Differences between exposure lake means and the mean of four reference lakes were computed as positive or negative differences, and the ranked magnitudes were tested for non-randomness of pattern. The differences also were calculated as percent deviation from reference mean.

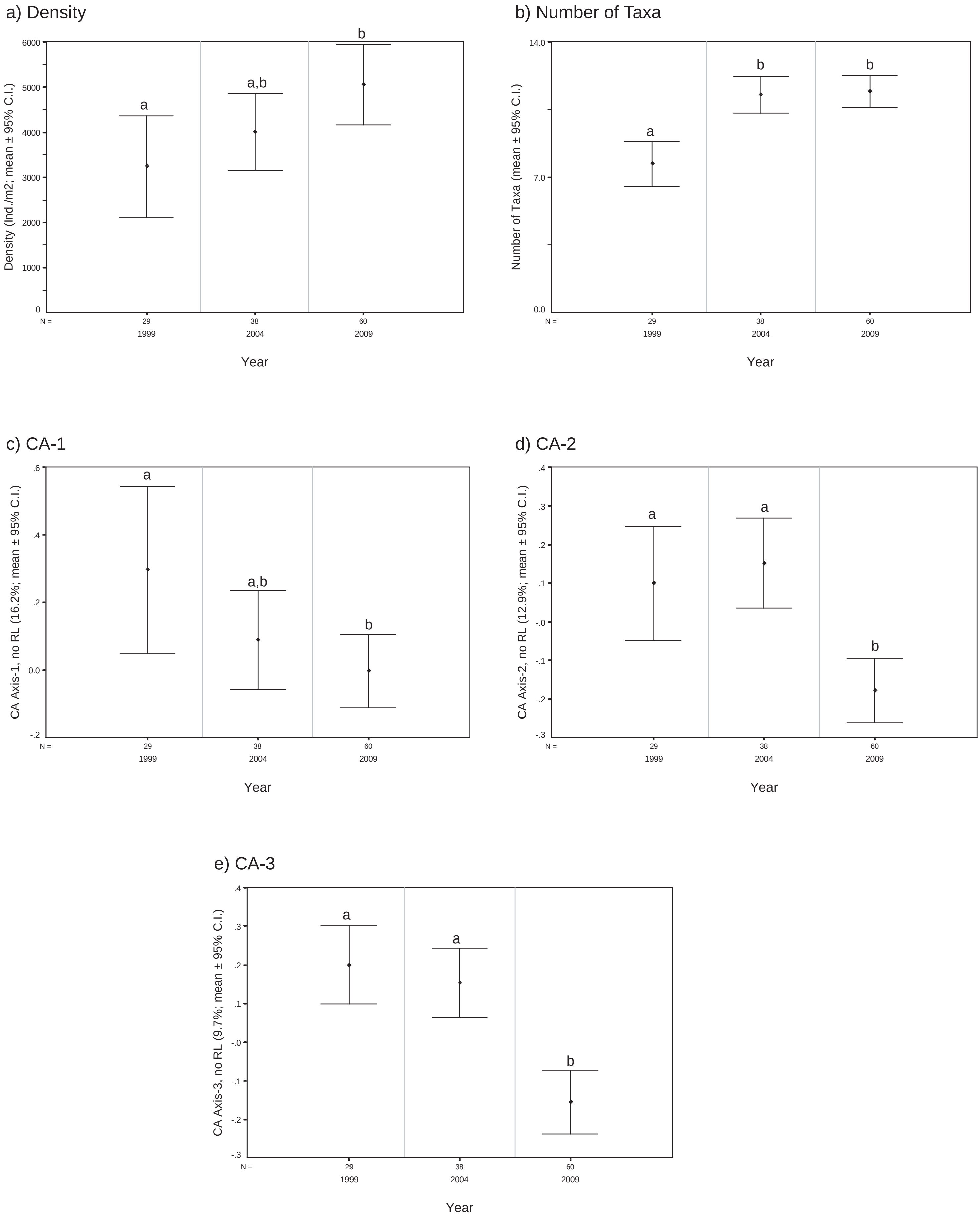


Figure 5.8: Benthic invertebrate community metrics for combined reference and exposure stations among years (1999, 2004, 2009). Years with similar letters were not significantly different (p > 0.1).

In 1999, density and number of taxa in seven exposure lakes was less than the reference mean values more often than would be expected by chance (WSR p = 0.078 and p = 0.031, respectively, Table 5.8). McCabe Lake (ML) was the only lake with a slightly greater mean density than the mean of the four reference lakes, and Hough, Pecors, and Quirke lakes all showed substantial decreases from reference mean density (Appendix Table E.46). Number of taxa was most decreased from reference in Pecors, Quirke, and Hough lakes, and only Elliot Lake had a very slight positive deviation from the reference mean. Scores on CA Axis-1 in 1999 showed significant positive deviation from reference mean in the suite of exposure lakes (WSR p = 0.047; Table 5.8), indicating an exposure community with high relative abundance of *Rhyacodrilus montana* and *Bezzia* compared to that found at reference lakes. Only Elliot Lake had CA Axis-1 scores disparate from other exposure lakes, in a negative deviation from reference mean. Quirke and Pecors lakes showed the greatest increase in CA Axis-1 scores when compared to reference (Appendix Table E.46). No reference-exposure differences were found by WSR for CA Axis-2 scores, but CA Axis-3 scores in the exposure lakes were less than the reference mean more often than expected (WSR p = 0.047). In total, 4 of 5 metrics examined by WSR tests showed significant patterns of deviation from the reference mean in 1999 (Table 5.8).

The second study year: 2004, all eight exposure lakes had lower mean density than the reference mean, again representing a significant pattern (WSR p = 0.008; Table 5.8; Appendix Table E.46). The largest deviations were noted at May, McCabe, and Pecors lakes (Appendix Table E.46). No significant pattern of deviation was found for number of taxa in 2004, and each exposure lake appeared to show improvement in deviation from reference mean over the 1999 data (Appendix Table E.46). Scores on CA Axis-1 continued to show significantly more positive deviations from reference mean than expected by chance (WSR p = 0.055; Table 5.8) though the magnitude of positive deviation from reference was reduced in most lakes (Appendix Table E.46). Exposure lakes also showed a pattern of significant negative deviation from reference mean for CA Axis-2 scores in 2004 (WSR p = 0.039), with only Elliot and McCarthy lakes showing small positive deviations from the reference mean (Appendix Table E.46). No significant pattern of deviations was detected for CA Axis-3 scores in 2004. In total, three of the five metrics examined by WSR tests showed significant patterns of deviation from the reference mean in 2004 (Table 5.8).

In 2009, density continued to show a pattern of significantly more and larger negative deviations from reference than would be expected by chance alone (WSR p = 0.023; Table 5.6). Only Pecors Lake showed a small positive deviation of 8% from reference mean density, whereas Quirke Lake (-81%) and Nordic Lake (-64%) showed substantial negative

Table 5.8. Benthic community metrics for which there was a significant pattern of increase (↑) or decrease (↓) among mine-exposed lakes relative to the reference mean ($p<0.1$).

Metric	1999	2004	2009
Density	↓	↓	↓
Number of Taxa	↓		
CA Axis 1[a]	↑	↑	
CA Axis 2		↓	↓
CA Axis 3	↓		
Total Metrics for Which Exposure Lakes Differed from Reference	**4**	**3**	**2**

[a] CA for all years and locations combined, except RL.

deviation from reference (Appendix Table E.46). Number of taxa showed no significant pattern of deviation from the reference mean in 2009, though Quirke Lake showed the greatest negative deviation from the reference mean (-48%; Appendix Table E.46). Likewise, there was no pattern of reference-exposure differences in CA Axis 1 scores in 2009, though here too Quirke Lake showed the greatest deviation from reference, at +522% (Appendix Table E.46). Exposure lakes continued to show a significant pattern of very strong negative deviations from reference for CA-2 in 2009 (WSR p = 0.023; Table 5.8), with a mean deviation of –635% across the set of exposure lakes (Appendix Table E.46). No differences between reference and exposure were noted by WSR test for CA Axis-3 scores in 2009. Overall, only two of five metrics tested showed reference-exposure patterns of deviation in this latest year of study.

It is clear that year-to-year variation is a significant component of community change in these lake benthic communities, against which reference-exposure differences must be assessed in future years. Despite the variability between years, it appears that the pattern of deviations from reference mean values for the exposure lakes generally decreased through the three cycles of study, from 4 out of 5 metrics in 1999, to 3 out of 5 in 2004, and to only 2 out of 5 metrics in 2009. These changing patterns of deviation are evidence in support of a hypothesis of gradual recovery from initial (1999) impact in exposure lakes, but indicate that small deviations from reference means persist in both the density and community structure of exposure lakes as of 2009.

Previous study reports were unable to conclusively identify mine-related impacts on benthic invertebrate communities for several reasons:

- The inclusion in previous studies of sampling areas representing a much broader range of habitat types (deep and shallow lakes plus erosional and depositional rivers) resulted in considerable data “noise” that obscured effects that were detectable in 2009 based on a greatly reduced data set focused on deep lakes only.
- Previous benthic assessments included the use of numerous metric which contributed to the data noise across the three types of habitat that were previously assessed.
- Removal of Rochester Lake shifted the reference mean value away from the exposure lake values and also reduced data noise (less variability associated with the reference lake mean).

- With the addition of data from 2009, temporal patterns could be investigated for the first time.
- Mine-related effects have been and continue to be very subtle.

Furthermore, comparison of the 2009 mean benthic community metrics for each mine-exposed lake relative to the mean and range of values represented by reference lakes, (including Rochester Lake) show that while statistical differences were detected between individual mine-exposed lakes and the combined reference lake mean (Table 5.6), in most cases, the metrics for mine-exposed lakes fell within the reference lake range, especially when Rochester Lake was considered (Table 5.9). Therefore, the patterns of effect suggested by the data in 2009 are based on relative small shifts away from the mean reference condition and may have little or no ecological consequence when considered in terms of the range of values exhibited by reference lakes in the area.

5.4 Summary

Water quality continues to improve in the Serpent River Watershed with metal concentrations in surface water decreasing over time and pH increasing. Where a trend was detected for cobalt, manganese, radium-226, sulphate and uranium, they were decreasing. With few exceptions, mean surface water concentrations of mine related substances are less than the SRWMP benchmark and where concentrations exceed the benchmark they do not exceed toxicological thresholds. While surface water quality has dramatically improved since decommissioning and the inception of the SRWMP, sediment is changing slowly with few statistical differences found between 1999 and 2009. Sediment toxicity results were not consistent with sediment chemistry showing reduced survival in lakes with some of the lowest sediment concentrations. Pecors, McCarthy and Nordic Lake had reduced survival and growth in test with Hyalella azetca. However, results of Chironomus dilutus test showed no difference between exposure and reference lakes measures for growth or survival

The benthic invertebrate communities of all mine-exposed lakes were statistically different from the mean of reference lake values with respect to at least one of the benthic community metrics. The exposure areas showed a pattern of lower benthic invertebrate density and CA1 scores, along with higher CA2 and CA3 scores than the pooled reference areas, indicative of a mine exposure signature. The communities in Quirke, McCabe, and May lakes showed more significant differences from the mean reference community than the other lakes (i.e., more metrics differed), but the magnitudes of difference were larger at Quirke and McCabe than May when differences were expressed as a percentage of the

Table 5.9: Comparison of 2009 benthic invertebrate communities to reference lake values including Rochester Lake.

Benthic Community Metric	Mean Values in Mine-Exposed Lakes		Reference Lakes Included in Statistical Evaluations[a]		Rochester Lake
			Mean, excl. Rochester	Range of Means	
Density (Ind./m2)	Quirke	1285	6826	2987-7406	866
	McCabe	6409			
	May	3600			
	Hough	3750			
	Pecors	7400			
	Elliot	4086			
	Nordic	2440			
	McCarthy	4334			
Number of Taxa	Quirke	6.40	12.3	11-15	6
	McCabe	13.60			
	May	8.80			
	Hough	10.60			
	Pecors	10.40			
	Elliot	13.00			
	Nordic	13.60			
	McCarthy	11.80			
CA Axis-1 all lakes 2009 (14.4%)[b]	Quirke	0.917	-0.21	-0.069 to -0.387	-0.111
	McCabe	-0.443			
	May	0.247			
	Hough	0.165			
	Pecors	0.134			
	Elliot	-0.299			
	Nordic	0.328			
	McCarthy	0.013			
CA Axis-2, all lakes 2009 (12.7%)[b]	Quirke	0.083	0.053	-263 to 0.346	-0.957
	McCabe	0.932			
	May	-0.043			
	Hough	0.323			
	Pecors	0.263			
	Elliot	0.262			
	Nordic	0.112			
	McCarthy	0.132			
CA Axis-3, all lakes 2009 (9.7%)[b]	Quirke	0.159	0.153	0.034-0.284	0.635
	McCabe	0.211			
	May	0.066			
	Hough	0.005			
	Pecors	0.059			
	Elliot	0.330			
	Nordic	0.137			
	McCarthy	0.187			

[a] Dunlop, Summers, Semiwhite, Ten Mile

[b] The CA results shown here were based on analysis that included Rochester Lake and thus differ from the CA results presented in Figure 5.7.

reference mean or the number of reference area standard deviations. The benthic communities in Elliot and McCarthy Lakes were most similar to the mean reference community, differing only with respect to CA-3 score. The pattern of deviations from reference mean values for the exposure lakes generally decreased through the three cycles of study, from 4 out of 5 metrics in 1999, to 3 out of 5 in 2004, and to only 2 out of 5 metrics in 2009. This supports a hypothesis of gradual recovery from initial (1999) impact in exposure lakes, but indicate that deviations from the reference means persist in both the density and community structure of exposure lakes as of 2009. Such differences were not detected in previous studies due to the “noise” associated with previous inclusion of reference Rochester Lake in the deep lake community evaluations, assessment of a larger suite of benthic community metrics, and from parallel assessments of shallow lakes as well as depositional and erosional stream habitats. Most important, is that metrics for mine exposed areas were generally within or near the range of reference lake values indicating that the detected reference exposure differences were minor and possibly of no ecological consequence. Therefore, the 2009 study design provides a sensitive measure by which to track on-going improvements within the watershed.

6.0 SPECIAL INVESTIGATION

The special investigation study was undertaken in six lakes of the Serpent River Watershed in 2009, in order to clarify several issues pertinent to estimation of radiological dose and risk to natural biota and humans utilizing the watershed lakes and to provide an updated estimate of dose and risk to biota and humans based on the data collected. The six lakes studied were McCabe, May, Elliot, Nordic, Quirke and McCarthy Lake. A complete description of the study findings is provided in Appendix F (EcoMetrix 2010).

Based on measures collected as part of this study several questions with respect to assumptions used in dose estimates were resolved, as follows:

- Lead-210 (Pb-210) and polonium-210 (Po-210) are at secular equilibrium in the lake sediments, as would be expected from their half-lives. The average Po/Pb ratio in sediments was 1.01, with a range from 0.87 to 1.18, and no upstream-downstream pattern.
- Radionuclides of the thorium-232 (Th-232) decay chain are clearly elevated above background in May and Quirke Lake sediments, although the Th-232 concentration is only about $1/10^{th}$ of the Th-230 concentration. The contribution of the Th-232 decay chain to total dose was usually 10% or less, except for May Lake where 4 of 8 receptors had Th-232 decay chain contributions greater than 10%, and for aquatic plants where contributions exceeded 10% in most lakes and reached 25% in May Lake.
- Bioaccumulation factors (BAFs) derived from the flooded basins were generally similar to those derived from the watershed lakes for aquatic plants, although the U value was slightly lower in the basins, and the Pb value was slightly higher. Fish BAFs derived from the basins were consistently lower than those derived from the watershed lakes. Po-210 BAFs were not determined in either case due to non-detection of Po-210 in water; however, Po-210 in fish tissue was consistently higher than Pb-210, by a factor of 22 on average.
- The high observed Po/Pb ratio in fish indicates that fish to duck transfer factors for Po-210, previously determined in the flooded basins using a Pb BAF to estimate Po-210 in fish were most likely overestimated by at least a factor of 10. Correction for this error produces a transfer factor of 5.45 d/kg for fish-eating ducks, which is more in line with the Health Canada (2007) generic value of 2.5 for birds.

- A survey of fish and wildlife consumption by SRFN fishers and hunters and their families (SRFN, 2010) produced more realistic values for fish and wildlife intake rates than those used previously, and also indicated the fraction of harvest likely to come from the six watershed lakes and from Lake Huron. These data were utilized, along with measured radionuclide concentrations in the six lakes and Lake Huron, to estimate the dose received by SRFN members.

6.1 Ecological Dose and Risk

The radionuclide concentrations from the special investigation studies were utilized to calculate radiation doses received by aquatic biota and riparian wildlife in the six watershed lakes. The calculated doses to fish, aquatic plants and benthos were well below the UNSCEAR (1996) benchmark dose of 10 mGy/d. The largest doses to aquatic biota occurred at Quirke Lake, where the doses to fish, aquatic plants and benthos were 0.92, 2.61 and 0.256 mGy/d, respectively. For all aquatic biota, the largest component of dose was internal. The largest contributor to dose was generally Po-210 for fish and benthic invertebrates, while the dose was more evenly distributed for aquatic macrophytes, with Ra-226 and short-lived radon daughters usually making the largest contribution.

The radiation doses to riparian wildlife were less than the UNSCEAR (1996) benchmark dose of 1 mGy/d. The largest doses to riparian wildlife occurred at Quirke Lake, where the doses to mallard, scaup, merganser, muskrat and mink, were 0.263, 0.094, 0.793, 0.407 and 0.124 mGy/d, respectively. For all riparian biota, the largest component of dose was usually internal. The largest contributor to dose was Po-210 for waterfowl, and Ra-226 with short-lived radon daughters for muskrat. For mink, one or the other of these contributors was predominant.

6.2 Human Dose and Risk

The radionuclide concentrations from the special investigation studies were utilized to calculate radiation doses received by generic human receptors at the six watershed lakes (receptor assumed to reside there and take all fish and game from there). The calculated doses ranged from 0.036 to 0.301 mSv/a, all less than the public dose limit of 1 mSv/a, before background correction. Background dose from the same pathways was estimated at 0.013 mSv/a. Therefore, incremental doses ranged from 0.023 to 0.288 mSv/a. The smallest doses were at McCarthy, Elliot and Nordic lakes, whereas the largest dose was at Quirke Lake. The dose at Quirke Lake was dominated by consumption of mallard ducks, and was driven by the high concentration of Po-210 in aquatic macrophytes at Quirke Lake. However, macrophytes were collected in Quirke Lake from a former tailings deposition area near Panel Mine and thus likely over estimate typical macrophyte uptake within the lake. The estimated dose at Quirke Lake without the waterfowl component is 0.072 mSv/a (total) or 0.064 mSv/a (incremental).

The calculated dose to a Serpent River First Nation member was based on realistic use of the six watershed lakes, and of Lake Huron, as determined from the survey of harvesters (SRFN, 2010). Most of the harvest comes from Lake Huron. For an actual use scenario the dose was 0.062 mSv/a (total) or 0.049 mSv/a (incremental). For a future use scenario the dose was 0.060 mSv/a (total) or 0.047 mSv/a (incremental). All these doses are less than the public dose limit of 1 mSv/a (incremental). The use of Serpent Harbour water and sediment data to represent Lake Huron may overestimate the Lake Huron component of dose.

The contributions of water, fish, moose and waterfowl to the SRFN dose are approximately 28%, 37%, 25% and 10%, respectively, with slight variations between actual use and future use scenarios.

6.3 Summary

The data collected as part of the special investigation proved adequate to resolve the outstanding questions with respect to dose and risk estimates within the Serpent River Watershed. Dose estimates received by aquatic biota and riparian wildlife in the six watershed lakes were less than the UNSCEAR (1996) benchmarks of 10 mGy/d and 1 mGy/d respectively. The incremental radiation doses received by generic human receptors at the six watershed lakes (receptor assumed to reside there and take all fish and game from there), ranged from 0.023 to 0.288 mSv/a, all less than the public dose limit of 1 mSv/a. The calculated dose to a Serpent River First Nation harvester was 0.062 mSv/a (total) or 0.049 mSv/a (incremental) based on realistic use of the six watershed lakes, and 0.060 mSv/a (total) or 0.047 mSv/a (incremental).based on a future use scenario. All these doses are less than the public dose limit of 1 mSv/a (incremental).

7.0 CONCLUSIONS AND RECOMMENDATIONS

7.1 Conclusions

The objective of this Serpent River Watershed State of the Environment Report was to integrate recent monitoring data from the TOMP, SAMP, and SRWMP to provide an assessment of current TMA performance and the conditions in the downstream Serpent River Watershed relative to TMA sources. The report presents data from the 2009 SRWMP and TOMP and SAMP data for 2005 to 2009. Key conclusions drawn from the analysis and interpretation of the data are as follows:

In-Basin Quality

Since decommissioning, conditions in the TMA basins have improved and basin water quality is generally at or near EIS-predicted levels. Water quality has continued to improve in recent years (2003 to 2007) based on decreasing concentrations of radium-226, sulphate, and uranium, as well as increasing pH levels, at most TMAs. Exceptions were observed at Denison TMA-1 and Stanleigh TMA where radium-226 has been increasing in surface water at both TMAs, and pH has been decreasing at Denison TMA-1. The trends at Denison TMA-1 appear to be attributable to a step change that occurred in 2008, possibly associated with decreases in sulphate over time (i.e. since 2000) and/or the higher water levels in 2008 and 2009. At the Stanleigh TMA, increasing radium-226 concentrations since 2004 were associated with a decrease in sulphate concentrations within the basin; as aqueous sulphate concentrations decline, there is an increased dissolution of barium sulphate to which radium is associated, whereby radium is released from the tailings. It is expected that radium concentrations in porewater will stabilize over time once the dissolution of barium sulphate re-equilibrates with aqueous sulphate concentrations. Assuming there are no new sources of radium to the TMA, radium concentrations in porewater should decline as the amount of soluble material in the tailings diffusion zone decreases.

Generally, trends in porewater concentrations reflected those observed in surface water within the basins, but trends in groundwater were more variable. For example, at the Nordic TMA, groundwater has improved in response to remedial measures implemented over the past five years. By comparison, deep groundwater at Quirke and Panel TMAs continued to show increasing concentrations of sulphate and decreasing pH, likely associated with the historical plume of acidity and the slow rate of groundwater flow.

TMA Discharges

Primary mine discharges, which contribute the majority of chemical loadings to the receiving environment, have also been improving over time. Where trends were detected, radium-226, sulphate, and uranium concentrations decreased in TMA effluents. The only exception to this was at Stanleigh, where radium-226 concentrations have been increasing slightly in response to decreasing sulphate concentrations in the basin.

At some TMAs (Denison, Stanrock and Pronto), effluent pH showed a decreasing trend but this appeared to be associated with either changes in treatment or possibly the effect of higher flows in 2008 and 2009. In all cases, effluent pH remains circum neutral.

Trend analysis for 2003-2009 data indicated barium concentrations have been increasing at the primary discharge locations (CL-06, D2, D-3, P-14 and Q-28)of the flooded basins, but this was largely due to greater barium chloride use in 2008 and/or 2009 in response to increased flows. In all cases barium concentrations in discharges were well below toxicity thresholds.

Over, the past five years, effluent quality has consistently achieved discharge criteria at all TMAs. With few exceptions, effluent has also been consistently non-lethal to *Daphnia magna* and rainbow trout with no mortality reported in semi-annual acute toxicity tests. Similarly, survival and reproduction of *Ceriodaphnia dubia* were not affected by exposure to 100% effluent in most tests conducted over the past five years at all TMAs.

Direct seepage releases from the TMAs to the receiving environment, only occur in the Quirke Lake sub-watershed. While metal concentrations tend to be highest and pH lowest in these sources, their loads to the receiving environment are low compared to primary discharges and background (upstream) loads. As noted in the previous SOE report (Minnow 2009a), the radium load within the Serpent River downstream of the Denison TMA discharge (D-5) was substantially greater than the loading from the Denison TMA or the upstream watershed (D-4) suggesting a radium source within the river. In 2009, EcoMetrix conducted a study to investigate the difference in loadings within the River and found elevated radium-226 sediment concentrations (14 Bq/g) between stations D4 and D5. The barium and sulphate depth profiles in sediment and water (porewater and overlying water) mirrored the radium profiles, indicating that these profiles are likely caused by the settling/accumulation of historical treatment solids. The loadings from this area are consistent with the recovery of historically accumulated sediments releasing radium to the water column. Diffusion modelling indicated that radium-226 release from the sediment should decrease with time.

Watershed Conditions

The improvements within the TMAs were reflected in the downstream watershed. With few exceptions, mean surface water concentrations of mine related substances were less than the SRWMP benchmarks and, where concentrations exceeded the benchmark, they did not exceed toxicological thresholds. Furthermore, metal concentrations (cobalt, manganese, radium-226, sulphate and uranium) in surface water have been decreasing over time, and pH has been increasing.

In locations where sediment concentrations were above benchmarks, concentrations of barium, cobalt, iron, manganese and nickel appeared to decrease or remain stable over the past ten years (1999 to 2009). Statistical comparisons of 1999 versus 2009 sediment concentrations indicated few statistically significant differences (1999 vs. 2009), except: a) a significant increases in sediment iron and manganese concentrations in Quirke Lake; b) an increase in sediment radium-226 in McCabe Lake, and c) decreases in sediment cobalt, manganese, nickel and radium-226 concentrations in Hough Lake. Overall, the data indicate a very slow rate of change in sediment quality.

Sediment toxicity tests using *Hyalella azetca* showed reduced survival and growth in samples from Pecors, McCarthy and Nordic compared to reference lakes and laboratory control samples. These results did not correspond with sediment chemistry since McCarthy and Pecors lakes had some of the lowest sediment concentrations of mine-related substances. The observed response may be related to TOC which was much lower in McCarthy and Pecors lakes than in the lab control or the reference lake. Depending on the substance, TOC may influence the bioavailability of metals in sediment. Growth and survival of *Chironomus dilutus* did not differ between exposure and reference lakes (Figure 5.5).

The benthic invertebrate communities of all mine-exposed lakes were statistically different from reference lakes with respect to at least one of the benthic community metrics. The exposure areas showed a pattern of lower benthic invertebrate density and CA1 scores, along with higher CA2 and CA3 scores than the pooled reference areas, indicative of a mine-related signature. The communities in Quirke, McCabe, and May lakes showed more significant differences from the mean reference community than the other lakes (i.e., more metrics differed), but the magnitudes of difference were larger at Quirke and McCabe than May when differences were expressed as a percentage of the reference mean or the number of reference area standard deviations. The benthic communities in Elliot and McCarthy Lakes were most similar to the mean reference community, differing only with respect to CA-3 score.

It is clear that year-to-year variation is a significant component of community change in lake benthic communities, against which reference-exposure differences must be assessed in future years. Despite the variability among years, it appears that the significant pattern of deviations from reference mean values for the exposure lakes generally decreased through the three cycles of study, from 4 out of 5 metrics in 1999, to 3 out of 5 in 2004, and only 2 out of 5 metrics in 2009. These changing patterns of deviation are evidence in support of a hypothesis of gradual recovery from initial (1999) impact evaluation in exposure lakes, but indicate that deviations from reference means persist in both the density and community structure of exposure lakes as of 2009. However, in most cases, the metrics for mine-exposed lakes fell within the reference lake range, especially when Rochester Lake was considered. Therefore, the patterns of effect suggested by the data in 2009 are based on relative small shifts away from the mean reference condition and may have little or no ecological consequence when considered in terms of the range of values exhibited by reference lakes in the area.

Risks to Wildlife and Humans

A special investigation was undertaken to allow for better estimates of dose and risk by making measurements to confirm or adjust assumptions used in previous dose and risk estimates. The data collected as part of the special investigation proved adequate to resolve the outstanding questions with respect to dose and risk estimates within the Serpent River Watershed. Dose estimates received by aquatic biota and riparian wildlife in the six watershed lakes were less than the respective UNSCEAR (1996) benchmarks of 10 mGy/d and 1 mGy/d. The incremental radiation doses received by generic human receptors (residing at the lake and consuming local fish and game) at the six watershed lakes, ranged from 0.023 to 0.288 mSv/a, all less than the public dose limit of 1 mSv/a. The calculated dose to a Serpent River First Nation harvester was 0.062 mSv/a (total) or 0.049 mSv/a (incremental) based on realistic use of the six watershed lakes, and 0.060 mSv/a (total) or 0.047 mSv/a (incremental) based on a projected future use scenario. All these doses are less than the public dose limit of 1 mSv/a (incremental).

Summary

In Summary, the TMAs are performing well in terms of meeting EIS predictions and reflecting improving conditions. The Serpent River Watershed is responding to these improvements, with water quality responding (improving) more rapidly than sediment and benthic invertebrates. Nevertheless, the benthic community has shown a pattern of improvement over the past ten years. Updated dose and risk estimates based on measured values indicate that dose is below established benchmarks for aquatic and riparian biota and humans.

7.2 Recommendations

Based on the findings of this report the following recommendations are provided:

- The groundwater monitoring locations at the Nordic TMA should be rationalized to reflect improvements in groundwater interception as recommended in the EcoMetrix Nordic Groundwater Study (Appendix I).
- Conditions are expected to continue to improve, but the rate of change in sediment and benthic invertebrates is slow, so consideration should be given to reducing the frequency of monitoring to once every 10 years.
- When the next SRWMP is implemented the list of exposure lakes to be included should be reduced to remove those lakes showing limited or no effects on benthic invertebrates (Elliot, Hough and McCarthy).

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