

6.0 REGIONAL SYNTHESIS

This part of the RAMP 2007 Technical Report presents regional assessments of the status of aquatic environmental resources considered by RAMP and the possible influence of focal projects and other developments on those resources at the regional level. This regional assessment consists of two parts for hydrology, water quality, benthic invertebrate communities and sediment quality, and fish populations:

- An assessment for the Athabasca River, representing the ultimate receiving environment for potential aquatic effects of focal projects and other developments in the Athabasca oil sands region; and
- A regional assessment for the rest of the RAMP FSA, represented by the watersheds and lakes considered in Section 5.

This section concludes with a presentation of the 2007 results for the Acid-Sensitive Lakes component, which by its design is regional in scope.

6.1 CLIMATE AND HYDROLOGY

6.1.1 Summary of Hydrologic Conditions in the Athabasca River

The assessed hydrologic effects of focal projects and other oil sands development activities in the RAMP FSA up to and including 2007 are summarized in Table 6.1-1. All hydrologic measurement endpoints are calculated to be lower in the operational hydrograph than in the baseline hydrograph, indicating these measurement endpoints are less than what they would have been in the absence of focal projects and other oil sands development activities. This is largely because of water withdrawals and assumed decreased natural runoff from oil sands development areas. The percent change varies from -0.2% to -1.8% depending on the specific measurement endpoint. The impact on low flows is greater in percentage terms than on high flows, because the more or less constant withdrawals from the Athabasca River are proportionately larger during low-flow than during high-flow periods. These estimated changes in hydrologic measurement endpoints for 2007 are assessed as Negligible.

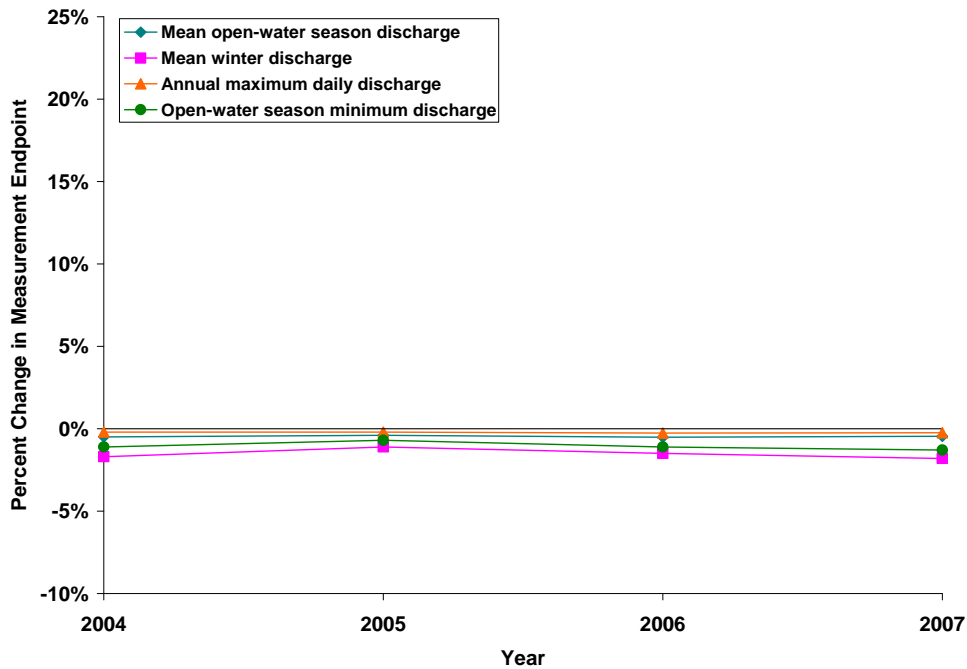
Table 6.1-1 Summary of hydrologic conditions of the Athabasca River in 2007 with respect to oil sands developments (cumulative effects case).

Measurement Endpoint ¹	Baseline Value (m ³ /s)	Operational Value (m ³ /s)	Percent Change	Assessment
Mean open-water (1 May to 31 October) season discharge	1,010	1,000	-0.4%	Negligible
Mean winter (1 November to 31 March) discharge	191	187	-1.8%	Low
Annual maximum daily discharge	3,410	3,400	-0.2%	Negligible
Open-water season minimum daily discharge	290	286	-1.2%	Low

¹ All as measured at RAMP Station S24, Athabasca River below Eymundson Creek.

Trends in the values of these measurement endpoints are provided in Figure 6.1-1. For the four years 2004 - 2007, the hydrologic effects of focal projects and other oil sands developments on the Athabasca River are assessed as being negligible in magnitude and negative in direction. The past three years (2005 to 2007) indicate a trend toward increasing negative effects on both open-water season and mean winter discharges.

Figure 6.1-1 Trends in assessed hydrologic effects on the Athabasca River from focal projects and other oil sands developments.



Note: Negligible: $\pm 2\%$; Low: $\pm 2\%$ to $\pm 5\%$; Moderate: $\pm 5\%$ to $\pm 15\%$; High: $> 15\%$ or $< -15\%$

6.1.2 Regional Assessment of Hydrologic Conditions at the RAMP FSA Level

The magnitude of change in each watershed in 2007 for each measurement endpoint is presented in Table 6.1-2.

Most of the hydrologic assessments are rated as negligible with the exception of the Muskeg, Tar, Poplar, and Fort Creek watersheds in which hydrologic effects are assessed as ranging from low to high, depending on the measurement endpoint. Specific water withdrawals and releases, and water diversions, were the focal project activities with the greatest influence in 2007 on hydrologic conditions in these watersheds, including:

- Effects of water withdrawals from the Athabasca River on winter flows;
- Discharges via the Aurora Clean Water Diversion into Stanley Creek and on into the Muskeg River;
- Increased flows into Poplar Creek via the Beaver River diversion and Poplar Creek spillway; and
- Additional minor withdrawals and discharges from watercourses in other watersheds including the Tar River.

Table 6.1-2 Summary of 2007 hydrologic assessment for RAMP FSA watersheds.

Watershed	Hydrologic Measurement Endpoint			
	Mean Open-Water Season Discharge	Mean Winter Discharge	Annual Maximum Daily Discharge	Minimum Open-Water Season Discharge
Athabasca River	Negligible	negligible (-)	Negligible	negligible (-)
Muskeg	low (-)	moderate (+)	low (-)	high (+)
Steepbank	Negligible	<i>not measured</i>	Negligible	negligible
Tar	moderate (+)	<i>not measured</i>	low (+)	high (+)
MacKay	negligible	negligible (+)	Negligible	negligible
Calumet	negligible	<i>not measured</i>	Negligible	negligible
Ells	negligible	Negligible	Negligible	negligible
Firebag	negligible	Negligible	Negligible	negligible
Christina	No hydrometric monitoring station at the mouth of the Christina River			
Hangingstone	negligible	<i>not measured</i>	<i>not measured</i>	negligible
Poplar	high (+)	<i>not measured</i>	high (+)	high (+)
Fort Creek	moderate (+)	<i>not measured</i>	moderate (+)	moderate (+)

Assessments based on comparisons of estimated incremental change in hydrologic measurement endpoints with criteria used in oil sands EIAs (RAMP 2005b). Negligible: $\pm 2\%$; Low: $\pm 2\%$ to $\pm 5\%$; Moderate: $\pm 5\%$ to $\pm 15\%$; High: $> 15\%$ or $< -15\%$. "not measured" means hydrologic information was not obtained for times of year for which the measurement endpoint is applicable.

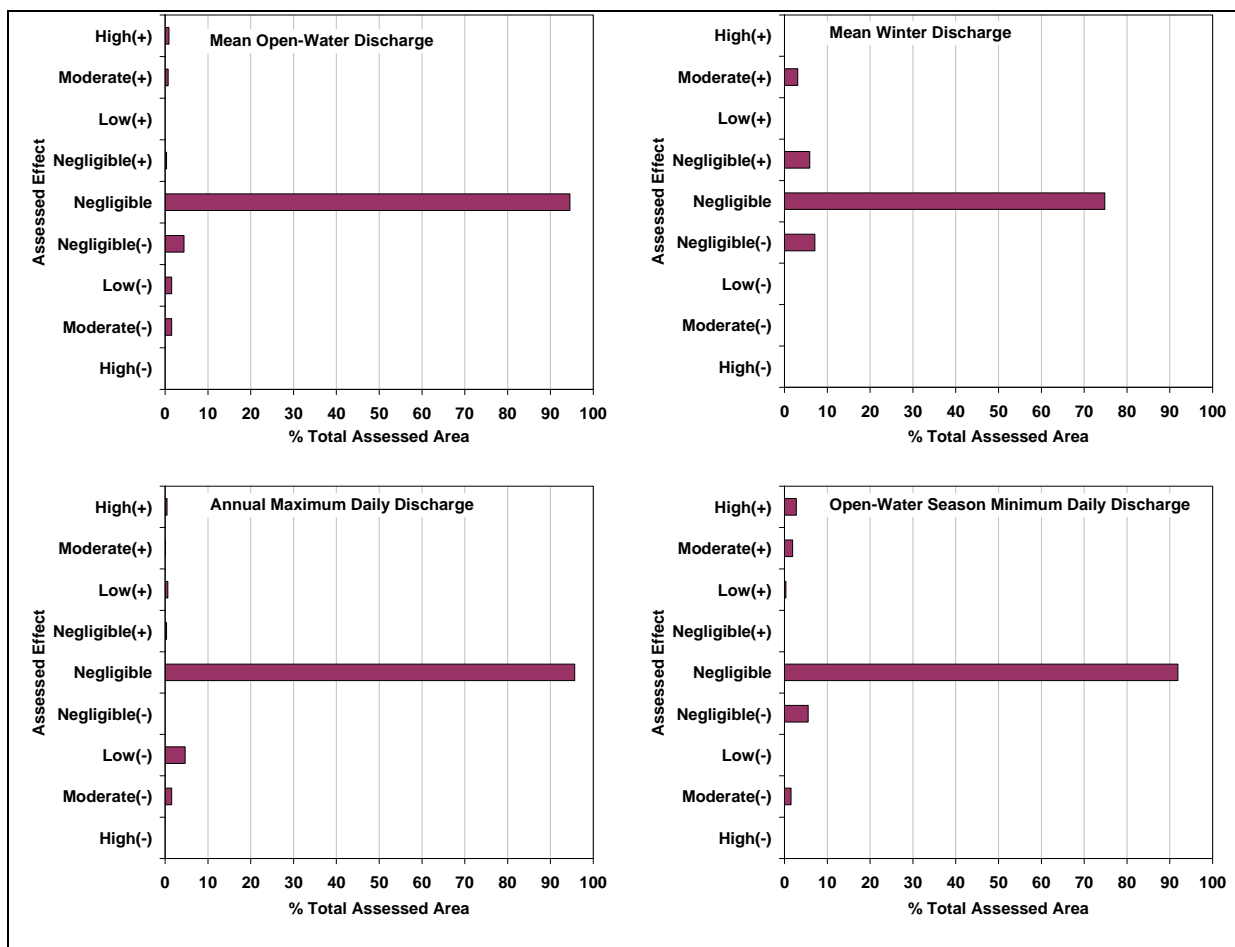
Direction indicators (+ or -) indicate an increase or decrease in discharge. Direction indicators are shown only for estimated impacts of a minimum of $\pm 0.5\%$.

Activities that caused land change resulting in closed-circuited areas were the focal project activities that had the second greatest influence on hydrologic conditions in 2007 in RAMP FSA watersheds; these land change activities were as important an effect on hydrologic conditions in the Muskeg River in 2007 as the Aurora Clean Water Diversion.

Activities that caused land change resulting in areas that were not closed-circuited were focal project activities that generally had minor effects on hydrologic conditions in RAMP FSA watersheds in 2007; the exceptions to this were the Tar and Fort Creek watersheds. The cumulative hydrologic effects of focal project activities plus all other active oil sands projects in the RAMP FSA are estimated to be only marginally greater than the hydrologic effects of the focal projects alone.

The average estimated percent change from 2004 to 2007 in each of the four hydrologic measurement endpoints are presented in Figure 6.1-2, which shows the percent of the area assessed each year falling under each effect classification. In all cases, most of the assessed area has experienced negligible hydrologic impacts. In cases where the estimated effects are not negligible, they often consist of decreases in maximum discharge or increases in minimum discharge. For example, trends in hydrologic impacts on the Muskeg River are shown in Figure 6.1-3. Compared to the Athabasca River (Figure 6.1-1), assessed hydrologic effects of focal projects and other oil sands developments are stronger on the Muskeg River; the measurement endpoints with changes that are estimated as Moderate or High in 2006 and 2007 are the mean winter discharge and the minimum open-water season discharge, both of which have increased with development.

Figure 6.1-2 Average percent change in hydrologic measurement endpoints in RAMP FSA, 2004 to 2007.

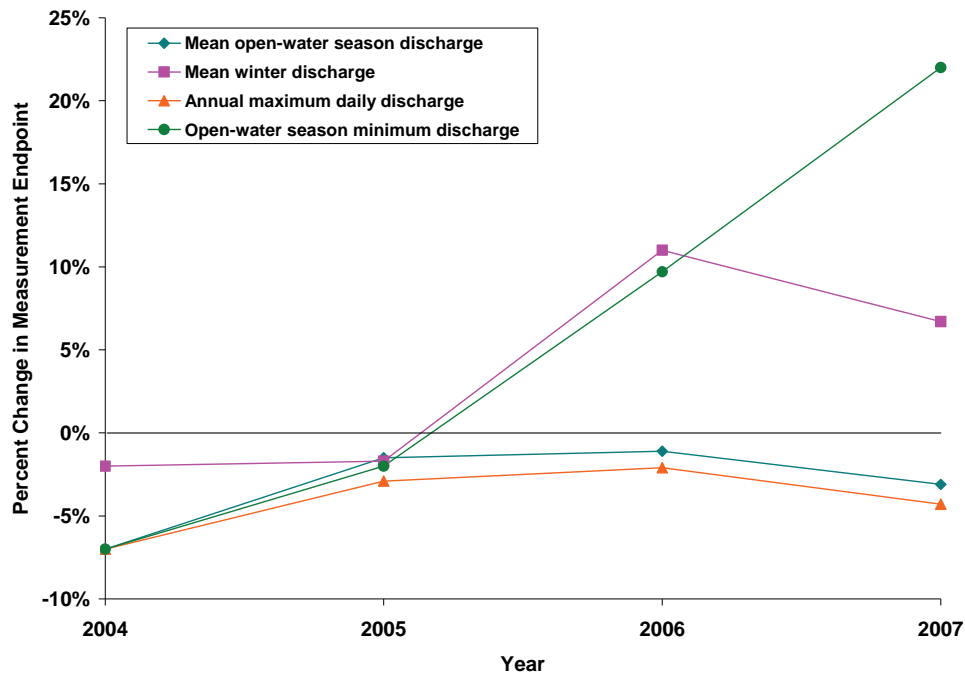


Assessed effects based on comparisons of estimated incremental change in hydrologic measurement endpoints with criteria used in oil sands EIAs (RAMP 2005b). Negligible: $\pm 2\%$ change; Low: $\pm 2\%$ to $\pm 5\%$; Moderate: $\pm 5\%$ to $\pm 15\%$; High: $> 15\%$ or $< -15\%$

Direction indicators (+ or -) indicate an increase or decrease in discharge. Direction indicators are shown only for estimated impacts of a minimum of $\pm 0.5\%$.

Therefore, while there have been changes in key hydrologic measurement endpoints in particular watersheds that have been moderate to high, hydrologic effects of focal projects and other oil sands developments at a regional level have been largely negligible to 2007.

Figure 6.1-3 Trends in assessed hydrologic effects on the Muskeg River from focal projects and other oil sands developments.



Note: Negligible: $\pm 2\%$; Low: $\pm 2\%$ to $\pm 5\%$; Moderate: $\pm 5\%$ to $\pm 15\%$; High: $> 15\%$ or $< -15\%$

6.2 WATER QUALITY

RAMP water quality data from fall were compared with regional reference values, to assess the likelihood that observations in 2007 fell outside the range of regional natural variability. These reference ranges were developed from all historical observations at all reference stations with similar water quality characteristics. Stations with similar water-quality characteristics were determined through an objective classification analysis, as described in Section 3 (supporting data and computations are presented in Appendix D).

The following groups (clusters) of water quality stations were assigned:

- **Cluster 1:** Eastern and southern tributaries and lakes, including stations in the Muskeg, Steepbank, Firebag, Clearwater, and Christina watersheds, plus Kearl and McClelland lakes;
- **Cluster 2:** Western tributaries, Athabasca floodplain lakes, and small tributaries to the Athabasca, including stations in the Calumet, Tar, Mackay, Ells, and Hangingstone watersheds, plus Fort, McLean, Poplar, and Beaver creeks, and Shipyard and Isadore's lakes; and
- **Cluster 3:** Athabasca River mainstem and delta.

RAMP 2007 water-quality data were presented and assessed at a station- and watershed-specific level in Section 5. This section examines various water-quality endpoints at a regional scale, through presentation of water-quality data by group (cluster), by year (i.e.,

all historical reference data versus 2007 data), and by station status in 2007 (i.e., reference versus potentially exposed). Variables selected for regional analysis included a subset of key measurement endpoints presented in Section 5, and variables with values that frequently exceeded water-quality guidelines in 2007. Figure 6.2-2 to Figure 6.2-15 present box-and-whisker plots of the distribution of water-quality values observed for various endpoints, with boxes describing the 25th to 75th percentiles of observations (the median, or 50th percentile appears as a central line within each box), error bars describing the 5th and 95th percentiles, and symbols (x) representing individual data points (outliers) that fall outside the 5th-to-95th percentile range.

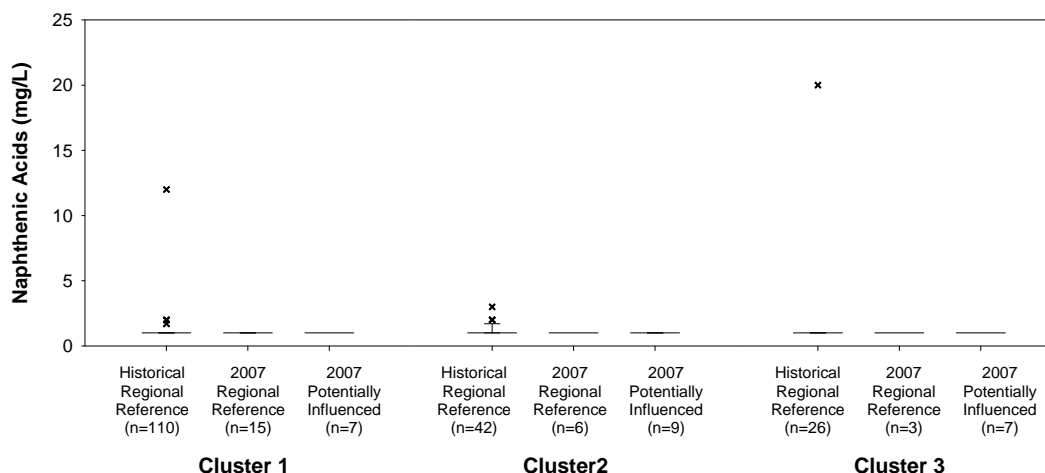
6.2.1 Water Quality Variables Associated with Oil Sands

6.2.1.1 Naphthenic Acids

Naphthenic acids include a wide variety of predominantly alkylated, cycloaliphatic carboxylic acids, which are natural constituents of petroleum hydrocarbons, including bitumen occurring in the oil sands region (Scott *et al.* 2005). Naphthenic acids are released during processing of bitumen, and may occur at high concentrations in oil-sands tailing waters. Although these tailing waters are not released to the aquatic environment through effluent discharges, naphthenic acids are a key measurement endpoint for the RAMP water-quality component, given they are specific indicators of bitumen-related hydrocarbons on water quality.

Naphthenic acids have almost always been non-detectable in water at all stations monitored by RAMP since 1997 (i.e., <1 mg/L). This also was the case in fall 2007, when naphthenic acids were not detected at any station (Figure 6.2-1). Historically, these acids have been detected at concentrations at or near the detection limit at some tributary stations (i.e., Clusters 1 and 2).

Figure 6.2-1 Naphthenic acids in waters of the RAMP study area, 2007 and historical data.



Two observations of relatively high concentrations of naphthenic acids exist in the RAMP database, both at reference stations sampled on September 17, 1998: at the upper Muskeg River (MUR-6); and at the Athabasca River mainstem upstream of Donald Creek, west bank (ATR-DC-W). These two extreme values may be erroneous, given: the large discrepancy between these two values and all other values observed by RAMP from 1997

to 2007; that both samples were collected on the same day (which may suggest lab error); and that the water sample collected from the opposite (east) bank at ATR-DC that day showed non-detectable naphthenic acids.

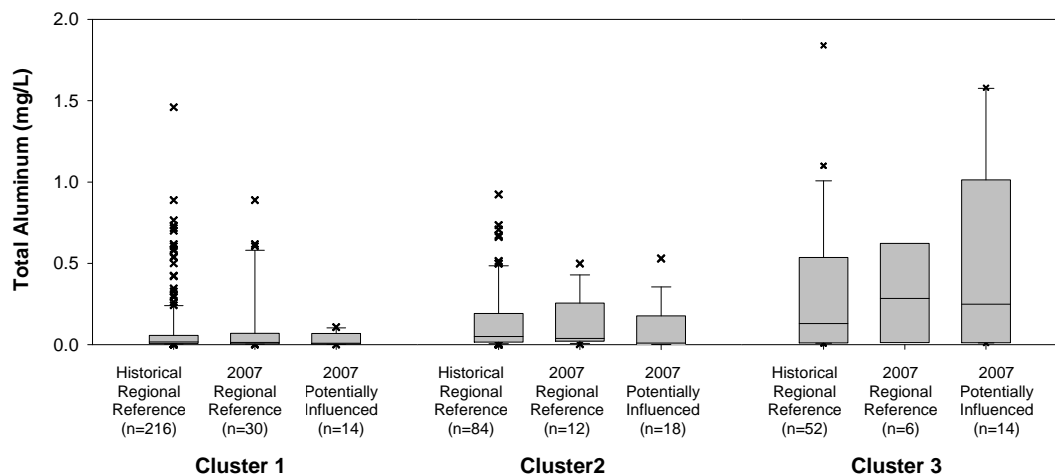
The absence of detectable concentrations of naphthenic acids from any station in 2007 indicates an absence of measurable effects of oil-sands development or other human activities on this monitoring endpoint.

6.2.2 Other Water Quality Variables

6.2.2.1 Aluminum

Total aluminum concentrations measured in the study area by RAMP since 1997 are summarized and compared in Figure 6.2-2; the CCME water-quality guideline for protection of aquatic life for aluminum (0.1 mg/L) also is presented in this graph.

Figure 6.2-2 Total aluminum in waters of the RAMP study area, 2007 and historical data.



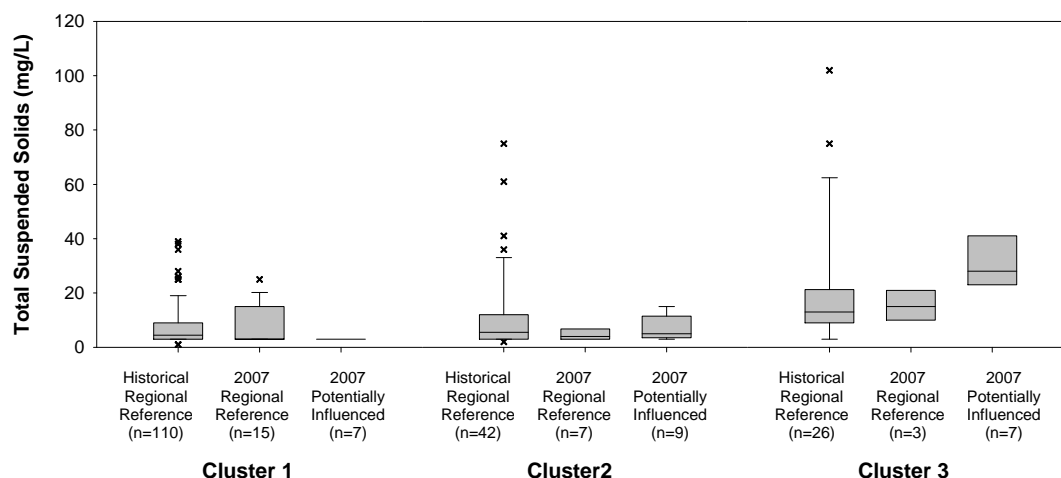
Although total aluminum concentrations have been highly variable in waters of the study area, clear differences between regional clusters of stations are apparent: values for Cluster 1 (eastern and southern tributaries) generally are low relative to other clusters (although exceedence of the CCME guideline has still been frequent); and those in Cluster 3 (Athabasca River mainstem) have typically been much higher, well above the CCME guideline. Values for Cluster 2 stations (western and smaller tributaries) generally were intermediate between values observed for Cluster 1 or 3 stations.

In 2007, median total aluminum concentrations generally were higher than those of historical reference data, for all station clusters. However, all 2007 data fell within the range of historical reference data. Median and interquartile (i.e., 25th-to-75th-percentile) concentrations at potentially influenced stations in 2007 were similar to, or less than, those in 2007 at reference stations belonging to Clusters 1 or 2. Median total aluminum concentrations at potentially exposed stations in the Athabasca River mainstem (Cluster 3) in 2007 were lower than those at reference (upstream) stations in 2007, but higher than the majority of historical reference observations at the three RAMP stations located upstream of oil-sands influences (i.e., stations at Donald Creek and upstream of

Fort McMurray). However, 75th and 95th percentiles for potentially exposed stations in 2007 were higher than regional reference data for 2007 and historically.

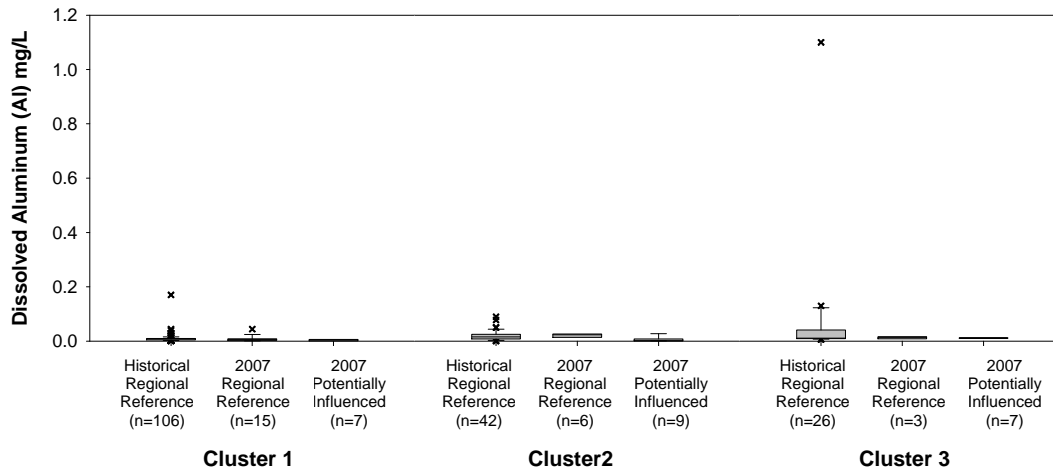
Aluminum is the most abundant metal on Earth, and commonly occurs in aquatic environments in particulate form, which is not readily bioavailable; its aquatic toxicity is strongly associated with its dissolved form, whose toxicity is highly dependant on pH, hardness, and dissolved organic carbon, increases in any of which generally reduce aluminum toxicity (Butcher 1988). In the complete RAMP water-quality dataset, total aluminum is more highly correlated with total suspended solids than any other variable ($r_s=0.726$, $n=335$, $r_{crit}=|0.09|$). Concentrations of total suspended solids in the study area, within and among clusters, and between 2007 and historical observations, show a very similar distribution to those of total aluminum, with highest TSS concentrations in the Athabasca River mainstem (Figure 6.2-3).

Figure 6.2-3 Total suspended solids in waters of the RAMP study area, 2007 and historical data.



Because much of the aluminum present in the aquatic environment is in particulate form and; therefore, not readily bioavailable, the British Columbia government uses an aluminum guideline based specifically on dissolved aluminum, with a chronic (30-day) guideline of 0.05 mg/L (Government of British Columbia 2006). In fall 2007, concentrations of dissolved aluminum in waters of the RAMP study area were below this BC chronic guideline at all stations (Figure 6.2-4), although concentrations exceeding this guideline (i.e., up to 0.09 mg/L) were observed in spring 2007 at Athabasca River mainstem stations ATR-DC-E and ATR-DC-W, and western tributary stations TAR-2, CAR-2 and ELR-2 (all reference stations).

Figure 6.2-4 Dissolved aluminum in waters of the RAMP study area, 2007 and historical data.



As is apparent from Figure 6.2-4, dissolved aluminum values have been consistently low throughout the study area since 1997, with the exception of a single high value of 1.1 mg/L found at station ATR-DC-W in September 2001. This single observation may be erroneous, given the concentration of dissolved aluminum along the other (east) bank of the river at the time of sampling was 0.05 mg/L.

6.2.2.2 Iron

Concentrations of iron also frequently exceed its screening guideline in rivers of the RAMP study area, which in this case is the 1987 CCME guideline for protection of aquatic life of 0.3 mg/L (CCME 2007). CCME does not provide supporting information regarding the foundation of this guideline; the CCME guideline for iron in drinking water (also 0.3 mg/L) is aesthetics-based. No water-quality guidelines for the protection of aquatic life for iron exist for Alberta or British Columbia. There is no national standard for iron in the United States; most state-based standards are 1.0 mg/L or higher (Iowa 2005). Naturally occurring, dissolved iron may contribute much of the stained colour of muskeg-related waters, such as those in the RAMP study area.

In the RAMP study area, both total and dissolved iron have frequently exceeded the CCME guideline in water, although maximum concentrations of total iron have been much higher than those observed for dissolved iron (Figure 6.2-5, Figure 6.2-6). However, total and dissolved iron concentrations in 2007 were similar or lower at potentially exposed stations than at reference stations, and values at all stations sampled in 2007 were within the range of regional reference data. In the complete RAMP water-quality dataset, dissolved iron was mostly strongly correlated with total colour and dissolved organic carbon ($r_s=0.660$ and 0.545 , respectively, $n=335$, $r_{crit}=|0.09|$). Total iron was associated with these other variables, but also with total suspended solids, which would be expected, given the large amount of suspended, particulate iron found at most stations (inferred through subtracting dissolved from total fractions).

Figure 6.2-5 Total iron in waters of the RAMP study area, 2007 and historical data.

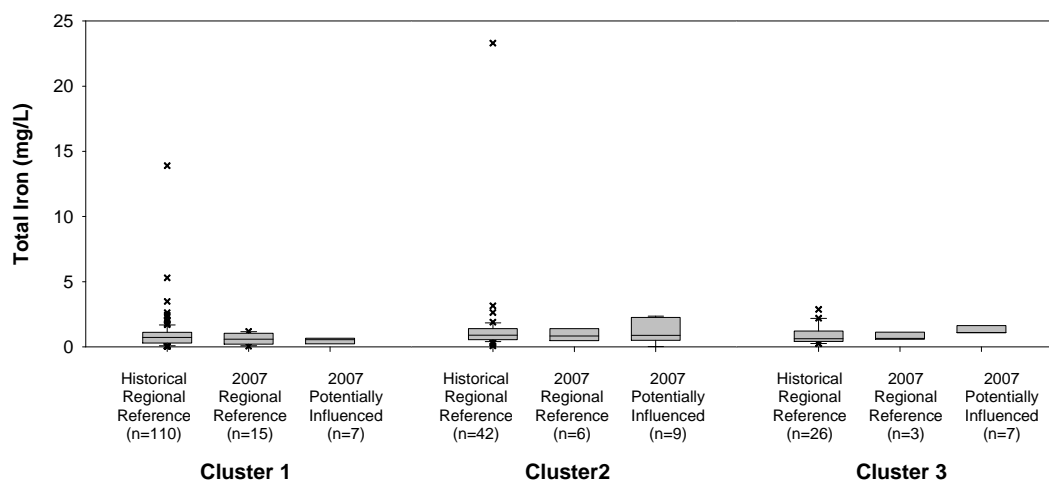
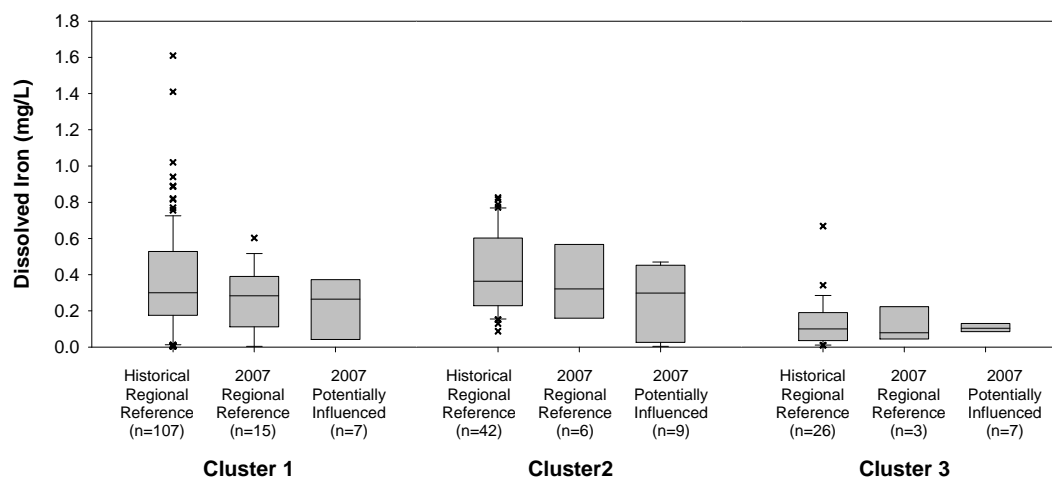


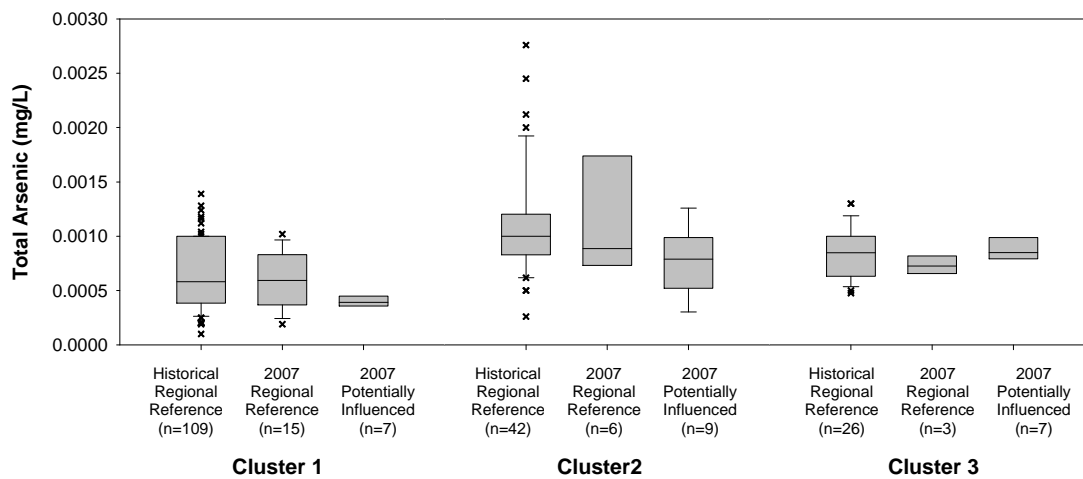
Figure 6.2-6 Dissolved iron in waters of the RAMP study area, 2007 and historical data.



6.2.2.3 Arsenic

Total arsenic in waters of the RAMP study area in 2007 generally was present at similar concentrations in potentially influenced stations and reference stations; all concentrations measured in 2007 were within the range of historical regional reference data (Figure 6.2-7). All arsenic concentrations observed by RAMP since its inception in 1997 (n=225) have been below the CCME guideline of 0.005 mg/L for protection of aquatic life, and the Health Canada guideline of 0.010 mg/L for drinking water (CCME 2007, Health Canada 2007).

Figure 6.2-7 Total arsenic in waters of the RAMP study area, 2007 and historical data.



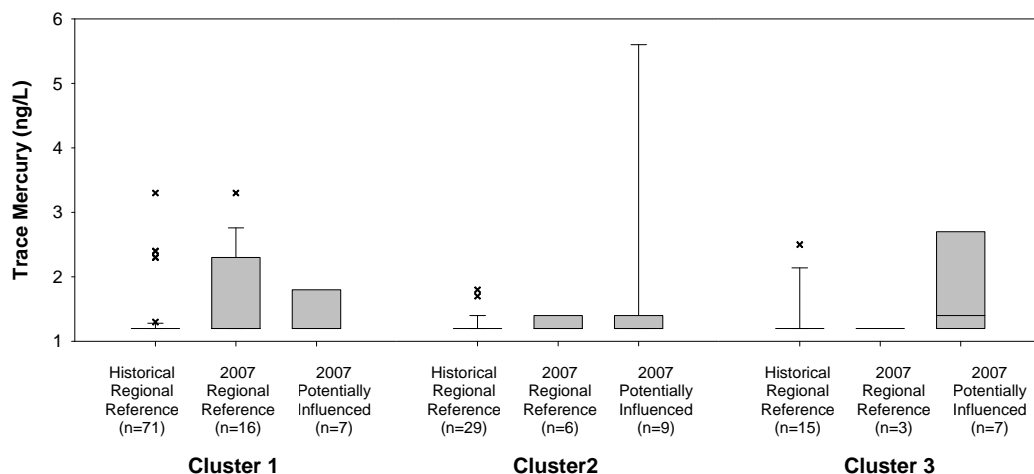
6.2.2.4 Mercury

Concentrations of total mercury (ultra-trace) in waters of the RAMP study area in 2007 generally were higher than those observed in previous years, at both potentially influenced stations and reference stations. The high 95%-percentile value shown in Figure 6.2-8 for 2007 at potentially influenced stations in Cluster 2 corresponds to an observation of 5.6 ng/L total mercury in the lower Tar River (TAR-1). This value was outside the ranges of historical observations and natural variability for any water quality group in fall sampling, and exceeded the CCME water-quality guideline for the protection of aquatic life of 5 ng/L. It is unknown whether this historically high mercury concentration in the lower Tar River was related to any activities associated with the development of the CNRL Horizon project; future monitoring data from this location in 2008 and onwards will help to clarify if this individual observation was characteristic of a longer-term change in water quality.

This guideline also was exceeded in observations in spring 2007 at reference station ELR-2 (upper Ells River, 12 ng/L), and in the Athabasca River mainstem downstream of development (ATR-DD, near Susan Lake, 16 and 19 ng/L along east and west banks, respectively). Along the Athabasca River mainstem in fall, most (6/10) total mercury measurements yielded non-detectable results (i.e., <1.2 ng/L), while concentrations ranging from 1.4 to 3.2 ng/L were observed at stations along the west bank of the river upstream of the Steepbank and Muskeg rivers and near Susan Lake, and in a cross-channel composite sample taken upstream of the Firebag River.

Mercury was not included in correlation analyses with other water-quality variables because it has been undetectable in over 50% of samples collected by RAMP since 1997.

Figure 6.2-8 Total mercury in waters of the RAMP study area, 2007 and historical data.

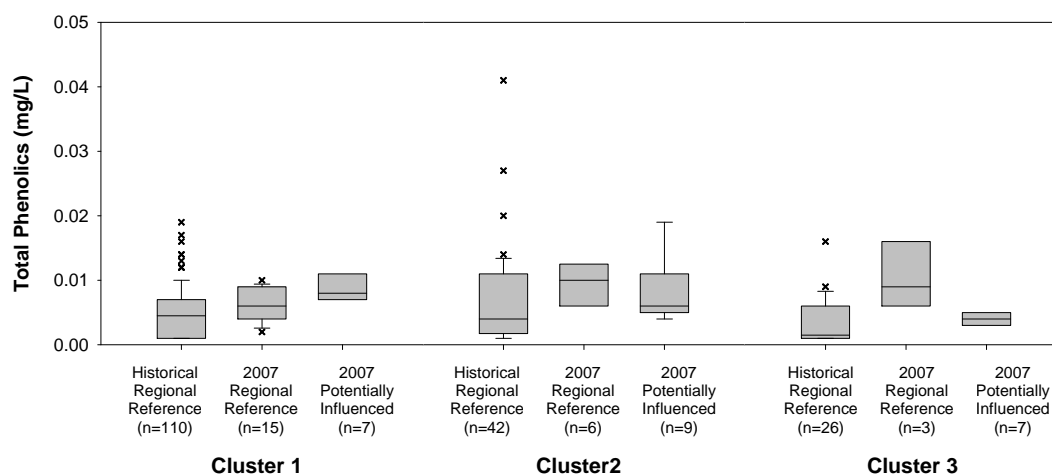


6.2.2.5 Total Phenols

Phenols are a large, complex group of acidic compounds that are hydroxyl derivatives of aromatic hydrocarbons. They are produced through the natural decomposition of plant materials, but also may occur in coal tar (CCME 1999, Government of British Columbia 2002).

Although there is a CCME guideline for the protection of aquatic life for phenols (0.004 mg/L, or 4 µg/L), this guideline is specific for mono- and dihydric phenols (i.e., those with one or two hydroxyl groups), and is therefore not applicable to the RAMP analyte, which encompasses a wide variety of phenolic compounds, including polyhydric species. Alberta Environment (1999) provides a chronic guideline for “phenolics” of 0.005 mg/L (5 µg/L), which was derived from interim guidelines prepared by the Alberta government in 1977; this guideline was used as a screening value in Section 5 of this report. British Columbia (2006) presents water-quality guidelines for specific phenol compounds (i.e., 3- and 4-hydroxyphenol) and for all other non-halogenated phenols of 0.05 mg/L (50 µg/L). Water at several stations in the RAMP area exceeded the AENV chronic guideline for phenolics in fall 2007, at reference and potentially exposed locations (Figure 6.2-9), although all values were within the range of historical RAMP observations for each station cluster. All phenol concentrations in 2007 and historically were below the BC guideline of 0.05 mg/L. Total phenols were most highly correlated with dissolved organic carbon and total colour ($r_s=0.395$ and 0.341 , respectively), suggesting that phenols in regional waters are associated with dissolved humic substances in water.

Figure 6.2-9 Total phenols in waters of the RAMP study area, 2007 and historical data.



6.2.2.6 Nutrients

Dissolved nitrogen and phosphorus are key variables affecting the primary productivity of aquatic ecosystems. In the majority of observations in the RAMP water-quality dataset, most nitrogen present is comprised of organic nitrogen, as indicated by the very strong correlation ($r_s=0.987$, $n=335$) between total nitrogen (TN) and total Kjeldahl nitrogen (TKN, which includes organic nitrogen and free ammonium), and the typical absence of detectable concentrations of inorganic nitrogen (i.e., nitrate-nitrite or ammonium). TN shows a strong correlation with dissolved organic carbon (DOC) in the RAMP dataset (further suggesting that most nitrogen in study-area waters is organically bound), and indeed is generally higher in stations in Clusters 1 and 2, which also exhibit higher DOC (Figure 6.2-10, Figure 6.2-11). Total nitrogen concentrations generally have been lowest in the Athabasca River mainstem (Cluster 3).

Figure 6.2-10 Total nitrogen in waters of the RAMP study area, 2007 and historical data.

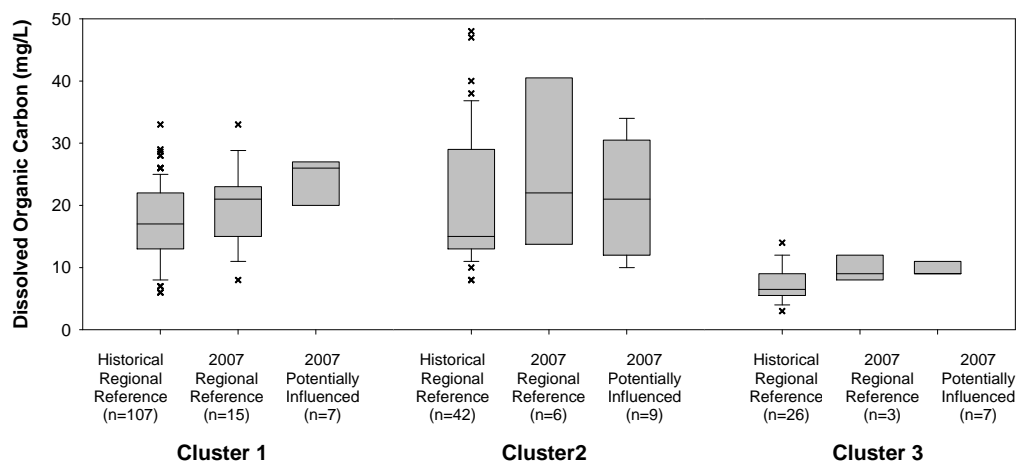


Medians and ranges of TN concentrations were similar between reference and potentially influenced stations in 2007, and were within the range of historical regional reference values, except in Cluster 2, where one station (lower Tar River, TAR-2) exhibited high TN

relative to other stations (indicated by the high 95th-percentile bar for potentially influenced stations).

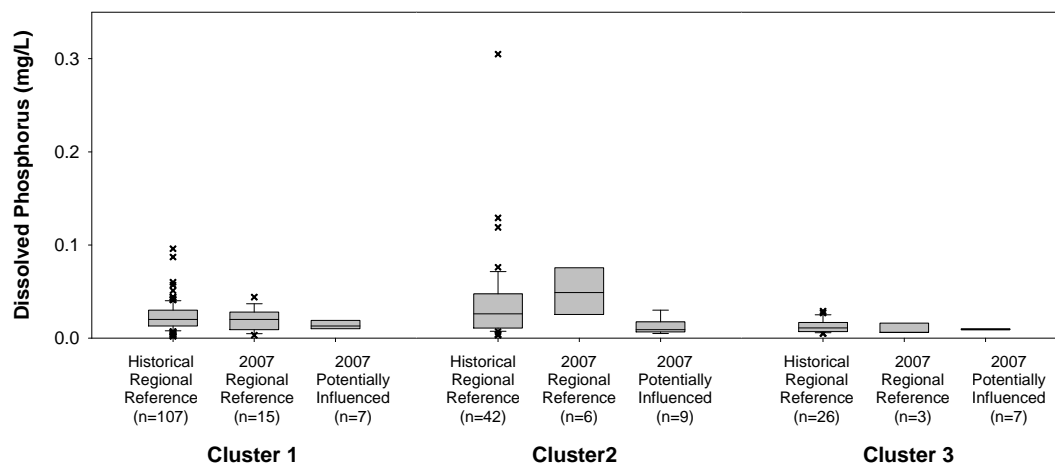
As discussed in Section 5, not only did TN fall outside the range of historical and regional reference values at this station, but the composition of nitrogen was predominantly inorganic nitrogen—including free ammonium and nitrate—rather than organic nitrogen. This composition strongly suggests an effect of the sewage-treatment-plant discharge from CNRL's Horizon project, which discharges to the Tar River upstream of the RAMP sampling location.

Figure 6.2-11 Dissolved organic carbon in waters of the RAMP study area, 2007 and historical data.



Total dissolved phosphorus (TDP) also has exhibited relatively high variability among stations and years since RAMP began sampling in 1997, particularly in stations in tributary stations in Clusters 1 and 2 (Figure 6.2-12). Although TDP concentrations in 2007 for reference stations in Cluster 2 were generally higher than those observed typically observed historically, TDP concentrations at all stations sampled by RAMP in 2007 were within the range of historical regional reference values, and were lower at potentially influenced stations than reference stations in all clusters.

Figure 6.2-12 Total dissolved phosphorus in waters of the RAMP study area, 2007 and historical data.



6.2.2.7 Major Ions

Concentrations of sulphate and chloride measured by RAMP in fall 2007 and historically, organized by station cluster and classification, appear in Figure 6.2-13 and Figure 6.2-14; concentrations of total dissolved solids (TDS) appear in Figure 6.2-15.

Although median concentrations of sulphate and chloride were generally similar in 2007 to those observed in previous years in each station cluster, the range of values observed was generally greater for both reference and potentially influenced stations in 2007 than for historical observations, particularly for stations in clusters 1 and 2. The upper quartile of both sulphate and chloride values for potentially influenced stations in Cluster 2 in 2007 was higher than those for reference stations in 2007, and outside the range of historical regional reference data. Stations with sulphate or chloride concentrations in the upper range of this group of potentially influenced stations in fall 2007 included the lower Tar River (TAR-1), lower Beaver River (BER-1), and Shipyard Lake (SHL-1). As discussed in Section 5, the lower Tar River has been affected by the development of the CNRL Horizon project since 2005, and lower Beaver River has received flow predominantly from minor tributaries and seepage from Syncrude's Mildred Lake site since the upper Beaver River was routed into the Poplar Creek system in 1974. Although Shipyard Lake is not directly affected by oil-sands development, significant upward trends in several ions, including sulphate, have been observed over the duration of monitoring (see Section 5.12). The change in concentrations of these anions from regional reference ranges supports the hypothesis that project-specific effects on water quality may be occurring in these waterbodies.

Figure 6.2-13 Total sulphate in waters of the RAMP study area, 2007 and historical data.

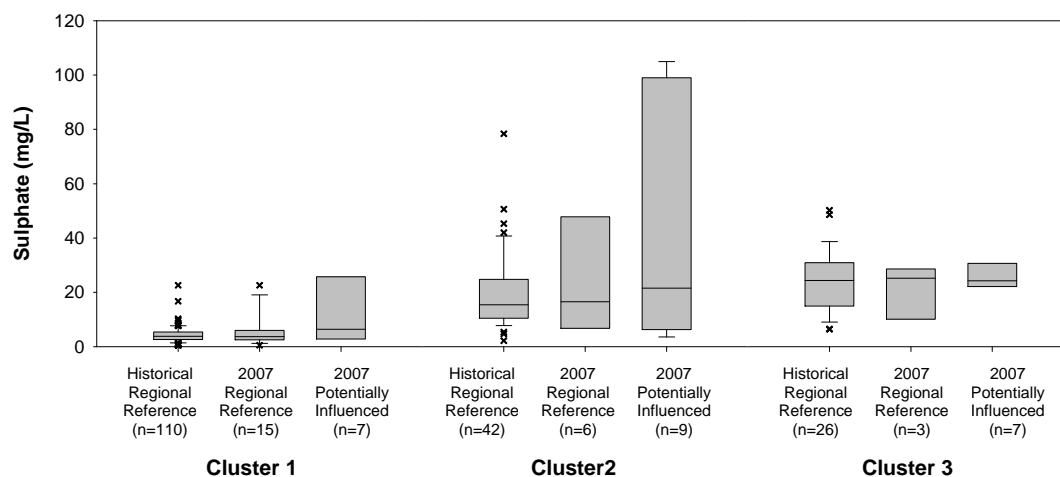


Figure 6.2-14 Total chloride in waters of the RAMP study area, 2007 and historical data.

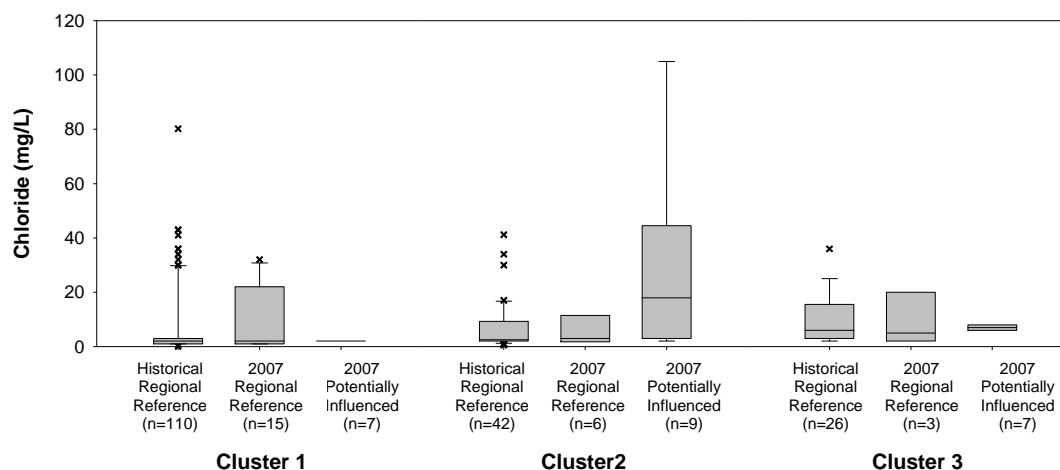
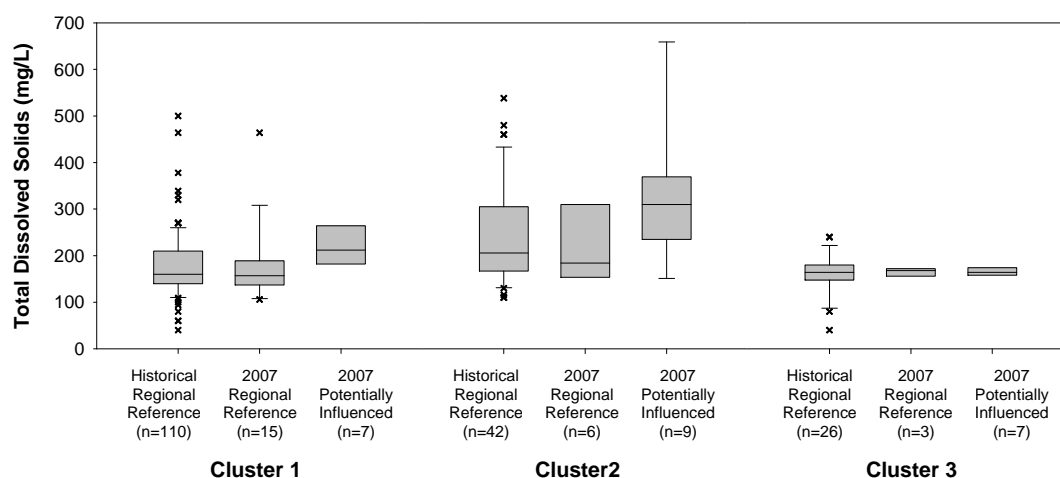


Figure 6.2-15 Total dissolved solids in waters of the RAMP study area, 2007 and historical data.



6.2.3 Summary

With some exceptions, water quality data collected by RAMP in fall 2007 was similar for all key measurement endpoints between stations defined as potentially exposed to oil-sands development and those defined as reference stations. Most data from reference and potentially influenced stations in 2007 fell within the range of historical observations from previous years, although 2007 data in aggregate did exhibit relative increases or decreases in some variables relative to historical data, such as ultra-trace mercury, which was detected more frequently in 2007 than in previous years, at both reference and potentially influenced stations.

Excursions from historical and regionally-typical water quality at specific stations included the following:

- **Lower Tar River:** Increases in total nitrogen, and speciation of nitrogen, that suggest effects of the treated-sewage discharge from CNRL's Horizon project on water quality;
- **Lower Beaver River:** Regionally high concentrations of major anions (sulphate and chloride) that may be related to seepage from Syncrude's Mildred Lake site into this drainage, which had most of its flow diverted away to Poplar Creek in 1974; and
- **Shipyard Lake:** Increasing concentrations of several major ions, including sulphate, which in 2007 exhibited a concentration outside the regional range of natural variability.

6.3 BENTHIC INVERTEBRATE COMMUNITIES AND SEDIMENT QUALITY

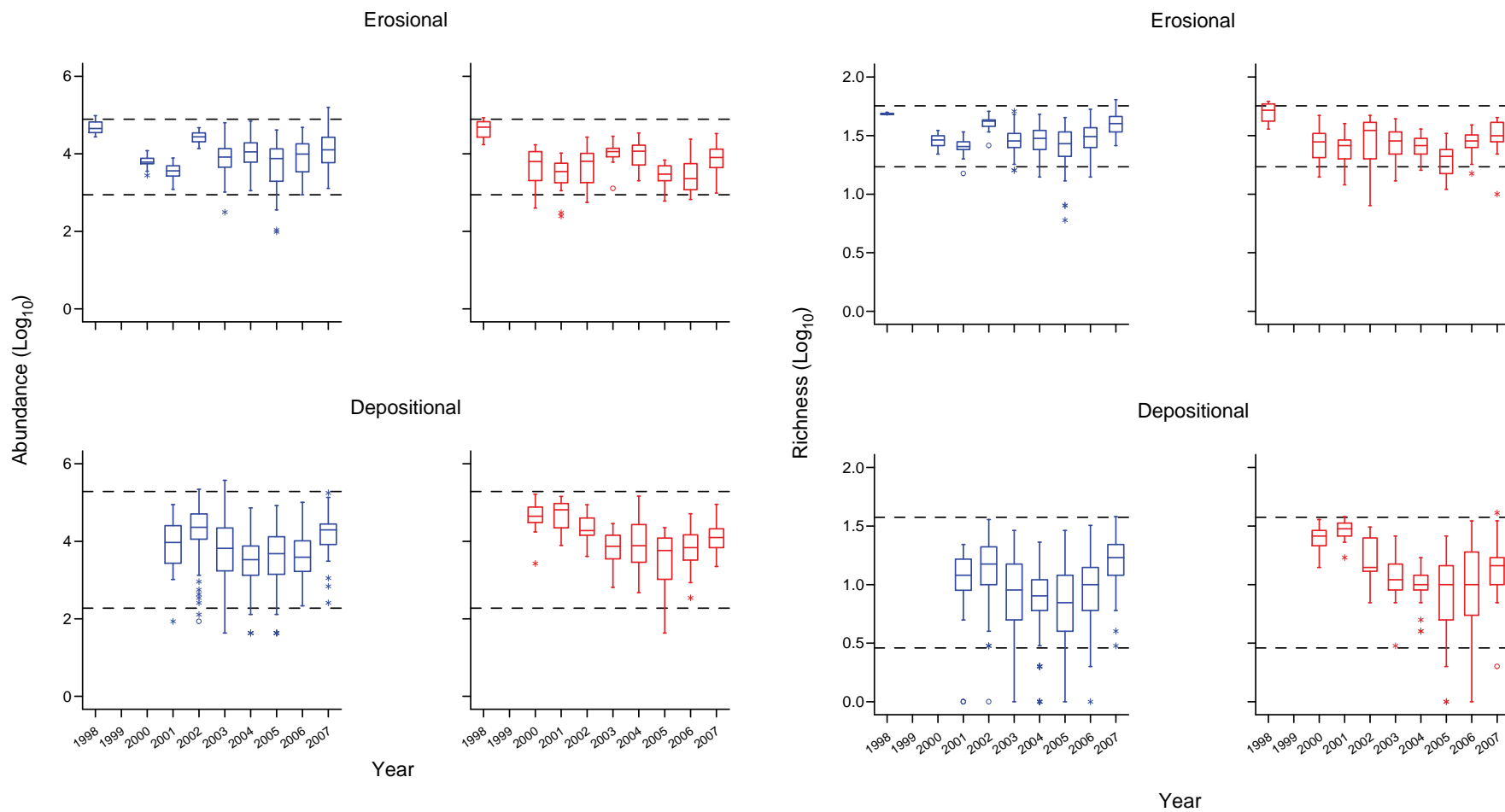
6.3.1 Regional Assessment of Benthic Invertebrate Community Conditions at the RAMP FSA Level

Variations in indices of composition of *reference* and *potentially influenced* reaches and lakes are illustrated as box plots for each year (Figure 6.3-1 to Figure 6.3-4).

In the case of all three major habitat classes (erosional reaches, depositional reaches, and lakes), variations within and among reaches (and lakes) designated as *potentially influenced* have generally been within the normal (background) range of variability as observed in the reference reaches (and lakes). By using the estimated 95% region for the reference data to quantify the normal range of variability, about 5% of observations of both reference and exposure data would be expected to fall outside the normal range. There was, thus, a percentage (not estimated) of records falling outside the normal range in almost each year for almost each measurement endpoint. The distributions of index values, however, were similar for reaches (and lakes) designated as reference and potentially influenced, suggesting a lack of obvious impairment among the reaches (and lakes) within the RAMP FSA as a result of focal project and other oil sands development activities.

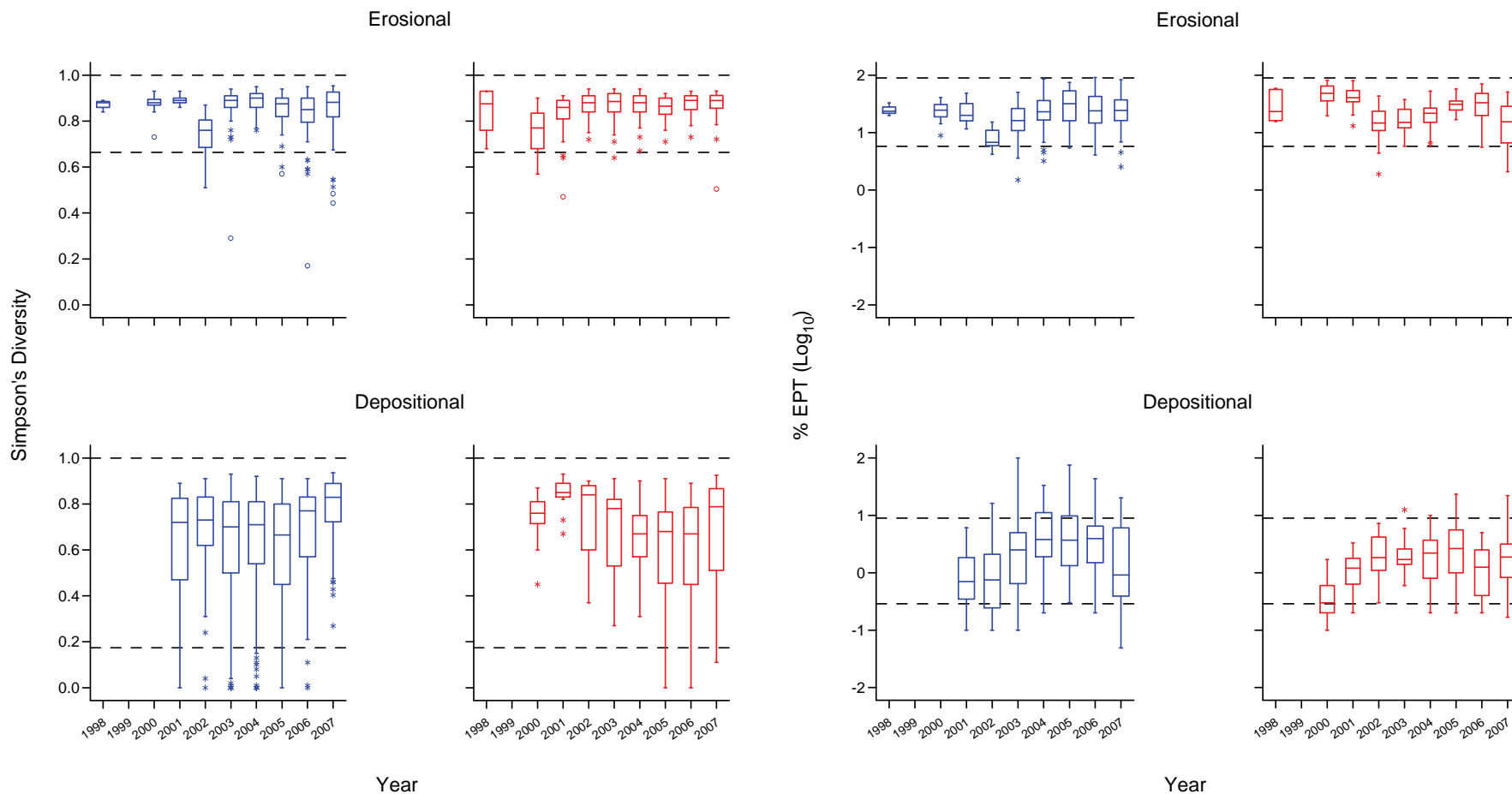
This overview of these data, and conclusion of no major effects, is supported by the "within-river (and within lake)" analyses conducted in Chapter 5. Detailed statistical analyses within reaches (and lakes) did not provide convincing evidence of effects of focal projects and other oil sands developments of a magnitude that are of concern.

Figure 6.3-1 Annual variation in total benthic community abundance and taxa richness for river reaches in the RAMP FSA.



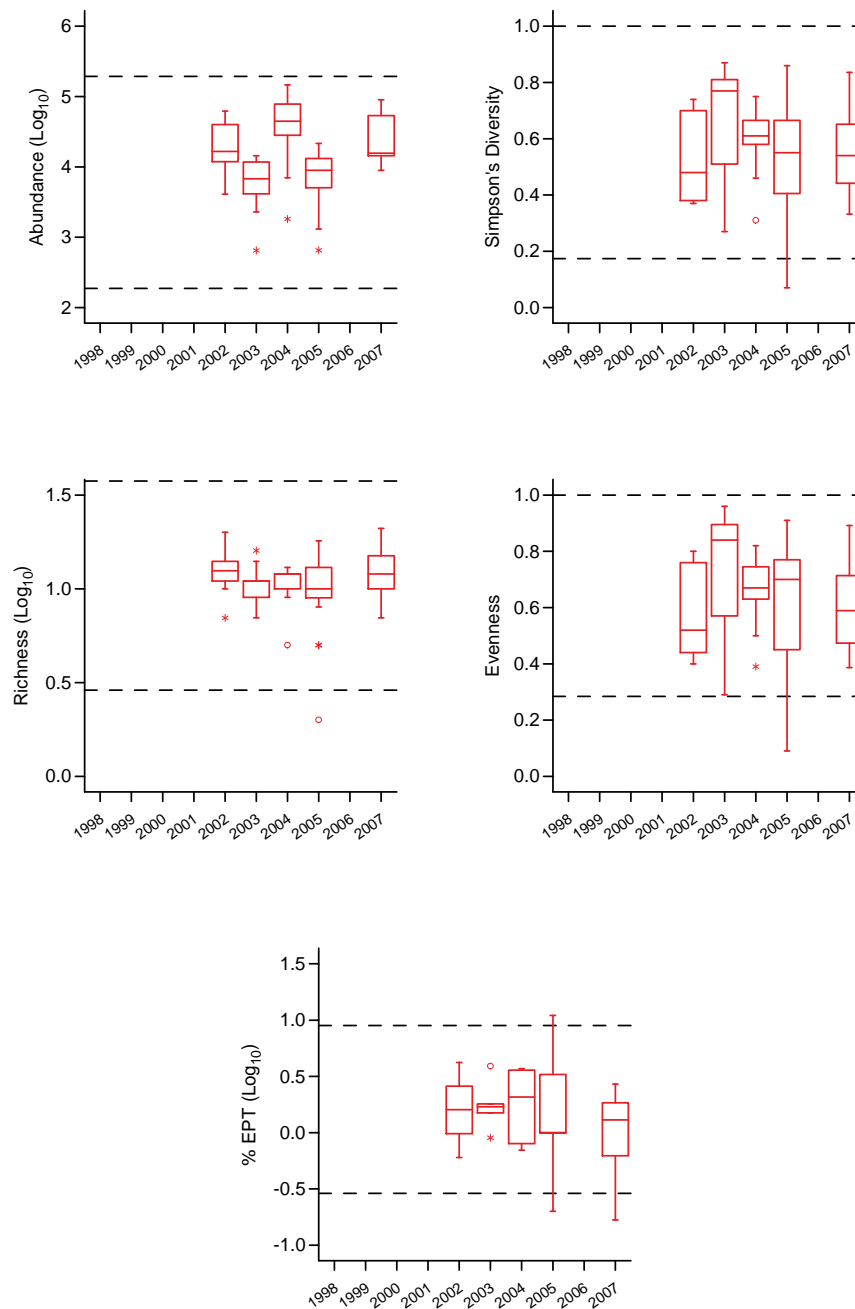
Note: Blue: *reference*; Red: *potentially influenced*. The normal range of variation for reference reaches is depicted by dashed lines.

Figure 6.3-2 Annual variation in benthic community diversity and percent EPT for river reaches in the RAMP FSA.



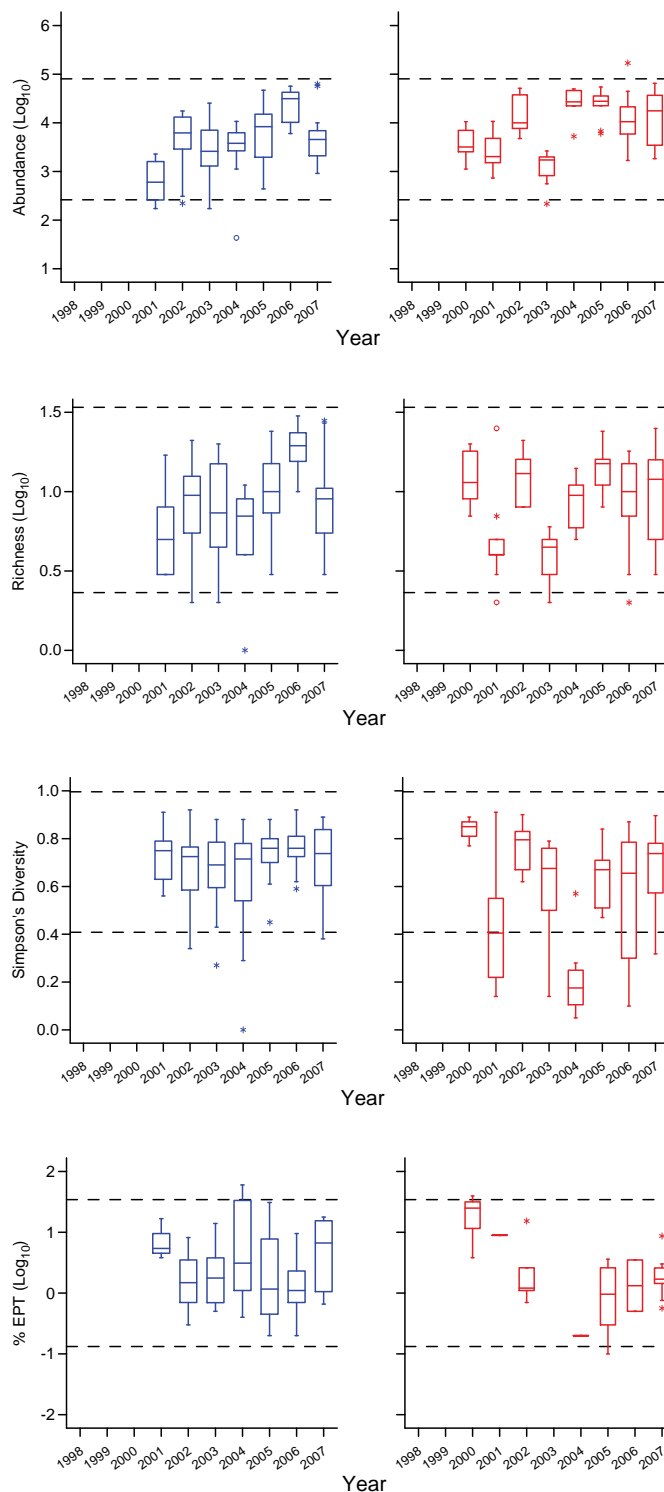
Note: Blue: *reference*; Red: *potentially influenced*. The normal range of variation for reference reaches is depicted by dashed lines.

Figure 6.3-3 Annual variations in benthic invertebrate community measurement endpoints for the Athabasca River Delta.



The normal range of variation for reference reaches is depicted by dashed lines.

Figure 6.3-4 Annual variation in benthic invertebrate community measurement endpoints for lakes in the RAMP FSA.



Note: Blue: *reference*; Red: *potentially influenced*. The normal range of variation for reference lakes is depicted by dashed lines.

6.3.2 Sediment Quality

6.3.2.1 Spatial and Temporal Trends in Sediment Quality

Total Hydrocarbons

Total hydrocarbons in sediments have been measured by RAMP since 1997. Until 2005, this was assessed using the Alberta Environment variable Total Recoverable Hydrocarbons (TRH); from 2005 onwards, total hydrocarbons has been assessed using the recently established CCME summary variable Total Petroleum Hydrocarbons (TPH, which is a sum of four molecular-weight-specific fractions). This change to the CCME four-fraction variable was made because it provided greater resolution of different hydrocarbon fractions, and also because associated environmental-quality guidelines were concurrently established for these fraction-specific variables, which did not exist for the previous TRH variable. It should be noted that both TRH and TPH variables were developed for application to assessments of terrestrial soils, rather than aquatic sediments. Further information and discussion of the CCME petroleum hydrocarbon variables may be found in CCME (2003).

Comparison of TRH and TPH data from duplicate samples collected by RAMP in 2005 found a best-fit relationship of $TPH = 2.183 \times TRH$ (RAMP 2006, Appendix E). Data collected by RAMP using the CCME four-fraction test since 2005 has shown that most hydrocarbons present in regional sediments are comprised of high-molecular-weight species (i.e., those falling in fractions 3 and 4, with more than 16 carbon atoms). Heavy oils, asphalts, and many PAHs (of petrogenic or biogenic origin) fall within these fractions; for RAMP 2006 and 2007 sediments, total PAHs were generally strongly correlated with F3 and F4 fractions, and with total hydrocarbons ($r_s = 0.770$, 0.714 and 0.759 , respectively; Appendix F).

Observed concentrations of total hydrocarbons in sediments of tributaries to the Athabasca River since 1997 appear in Figure 6.3-5. Graphs in this figure include TRH (1997 to 2005) and TPH (2005 and 2007, shown at a 1:2 vertical scale relative to TRH). Stations considered *reference* in the year of sampling show green background shading, while those considered *potentially influenced* in the year of sampling show a blue background. A similar presentation of total hydrocarbons in sediments of the Athabasca River mainstem and delta appears in Section 5.1.

Total concentrations of hydrocarbons have been highly variable within and among stations since sampling by RAMP began, and between stations defined as *reference* and those defined as *potentially influenced*. Historically, highest concentrations of total hydrocarbons have been observed in the Calumet River (2005 and 2006, upper and lower, *reference*), Stanley Creek (2003, *potentially influenced*), Shipyard Lake (2004, *potentially influenced*), and McLean Creek (1999 and 2000, *potentially influenced*). Highest concentrations of total hydrocarbons observed in 2007 were at Kearl Lake and McClelland Lake, both *reference* stations.

However, the organic carbon content of sediments may be an important determinant of the concentrations of hydrocarbons (given their hydrophobic nature and tendency to sorb to organic particles), and may confound comparisons among stations and years (e.g., see Lamberson *et al.* 2000); in the RAMP 2006/2007 sediment dataset, total hydrocarbons was significantly but weakly correlated with TOC, [$r_s = 0.475$]; Appendix F).

Therefore, concentrations of total hydrocarbons in sediments normalized to 1% organic carbon also were calculated and are presented in Figure 6.3-6.

Highest carbon-normalized concentrations of total hydrocarbons in sediments observed by RAMP since 1997 have occurred in the lower Ells River (2006 and 2007, *reference*), the lower Steepbank River (1997 and 2005, *potentially influenced*), McLean Creek (1999 and 2005, *potentially influenced*) and the lower Calumet River (2006, *reference*). No spatial trends were apparent at any station except possibly an upward trend in the lower Ells River, which has yet to experience significant oil-sands development.

Based on these observations, and results for the Athabasca River delta reported in Section 5.1, a regional-level effect of oils-sands development on concentrations of total hydrocarbons in sediments is not suggested.

Polycyclic Aromatic Hydrocarbons (PAHs)

In 2007, highest total concentrations of PAHs were observed in sediments from the middle Muskeg River (MUR-D2, *potentially influenced*), lower Firebag River (FIR-D1, *reference*), and lower Ells River (ELR-D1, *reference*) (Figure 6.3-7). However, when normalized to 1% organic carbon, the lower Ells River exhibited total PAH concentrations several times higher than those from any other station. Concentrations of PAHs in sediments of the Athabasca River delta were generally low or intermediate between those from tributaries with relatively high PAHs and those with relatively low PAHs, consistent with historical observations by Evans *et al.* (2002).

Concentrations were dominated by alkylated forms, with parent PAHs comprising a very small fraction of total PAH concentrations. This is consistent with a petrogenic origin of these PAHs, and consistent with observations by others that PAHs in lower Athabasca regional sediments are petrogenic in origin and predominantly alkylated, in areas affected or unaffected by oil-sands development (e.g., Wayland *et al.* 2008).

Spatial and temporal trends in total PAHs in sediments sampled by RAMP, in absolute and carbon-normalized concentrations, appear in Figure 6.3-8 and Figure 6.3-9. Stations with highest total PAH concentrations over time, before normalization to 1% TOC, included the middle Muskeg River (*potentially influenced*), the lower Ells River (*reference*), Stanley Creek (*potentially influenced*), McLean Creek (*potentially influenced*) and the lower Steepbank River (*potentially influenced*). Following normalization to 1% TOC, highest total PAH concentrations in sediment since 1997 have been observed in the lower Ells River (*reference*), the upper Steepbank River (*reference*) and McLean Creek (*potentially influenced*). Given the strong correlation between total PAHs and total hydrocarbons in the dataset, it is unsurprising that most of these stations also exhibited some of the highest observed concentrations of total hydrocarbons.

Figure 6.3-5 Concentrations of total hydrocarbons in sediments sampled by RAMP in tributaries to the Athabasca River, 1997 to 2007.

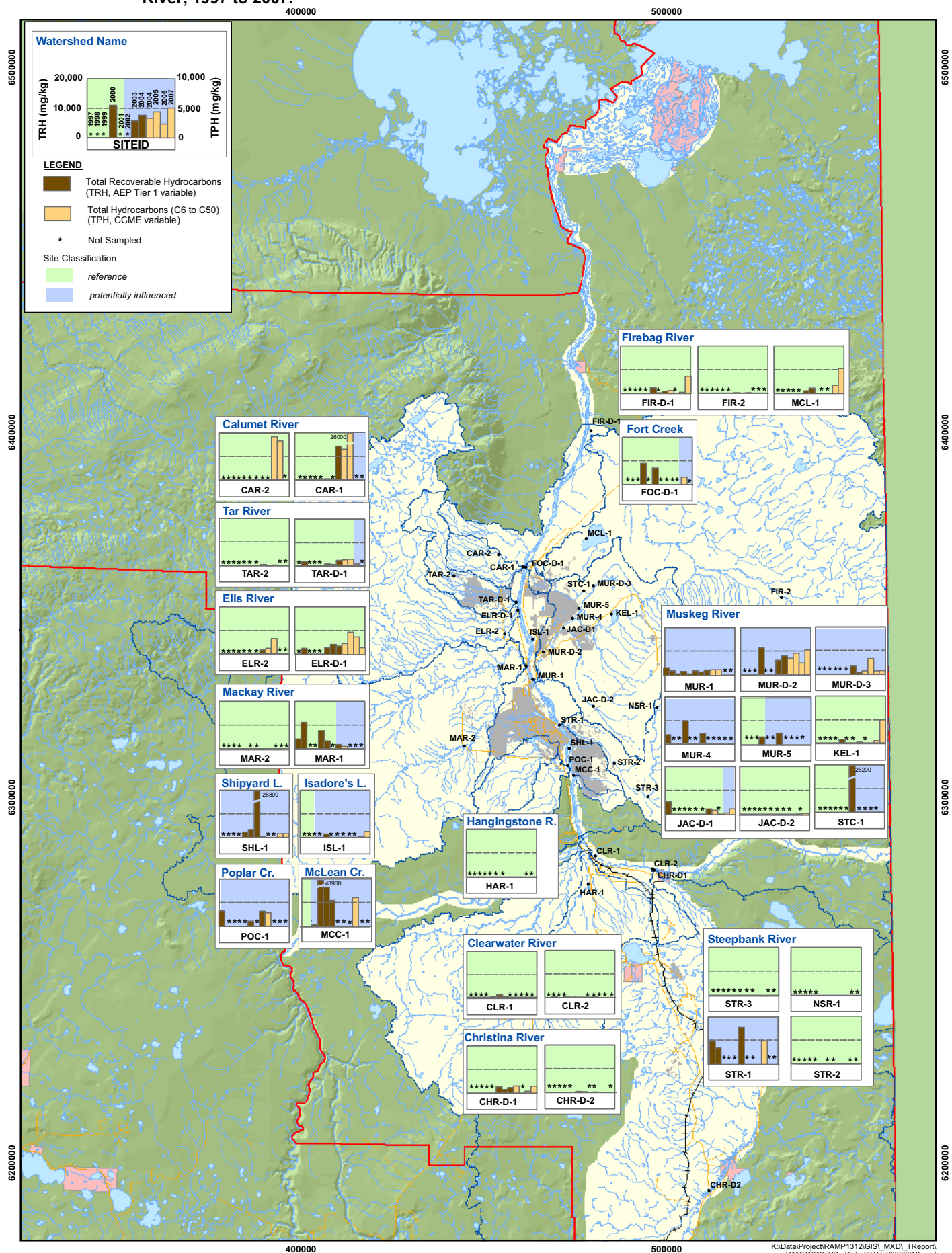


Figure 6.3-6 Carbon-normalized concentrations of total hydrocarbons in sediments sampled by RAMP in tributaries to the Athabasca River, 1997 to 2007.

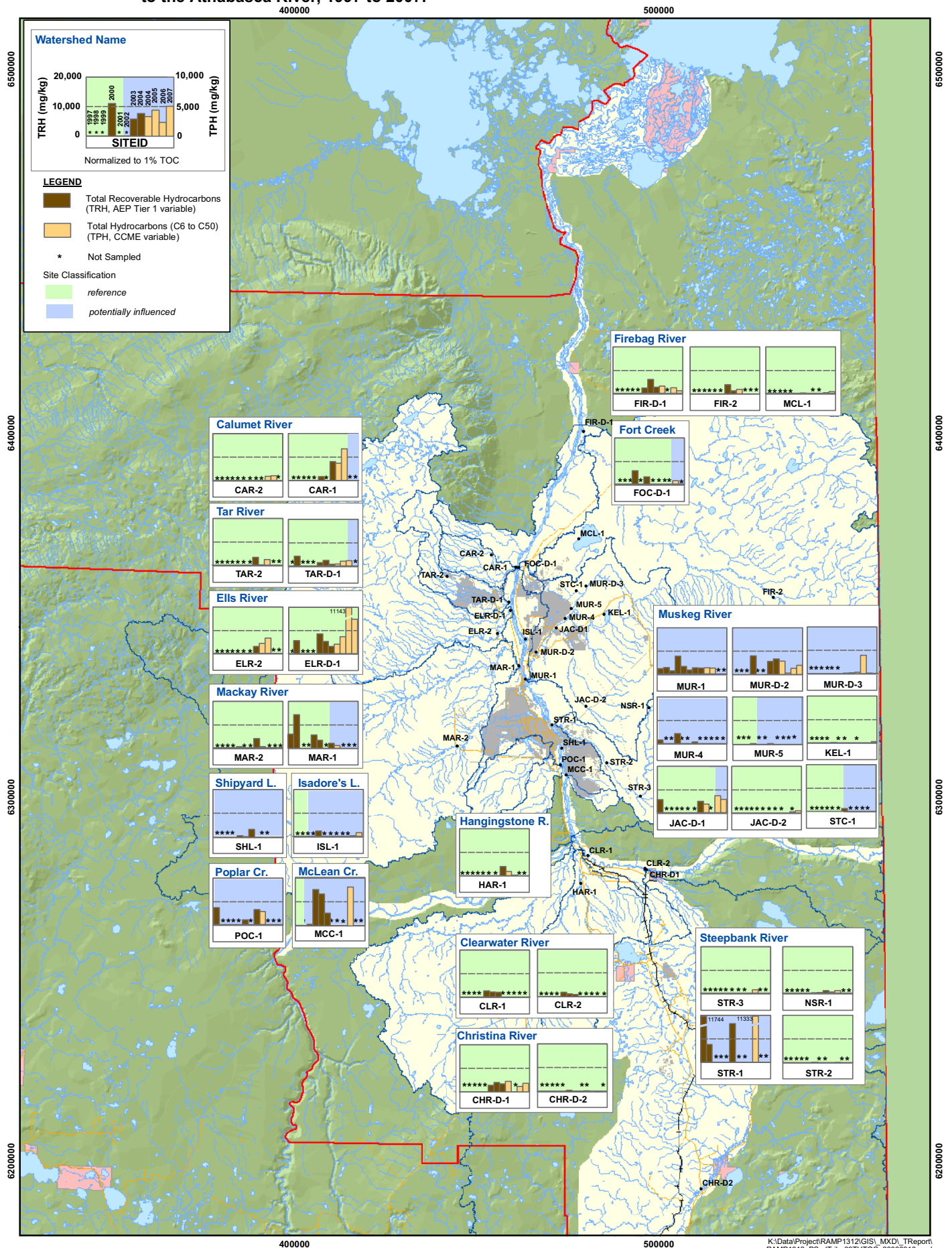


Figure 6.3-7 Total concentrations of parent and alkylated PAH in sediments collected by RAMP in 2007, including concentrations normalized to 1% organic carbon.

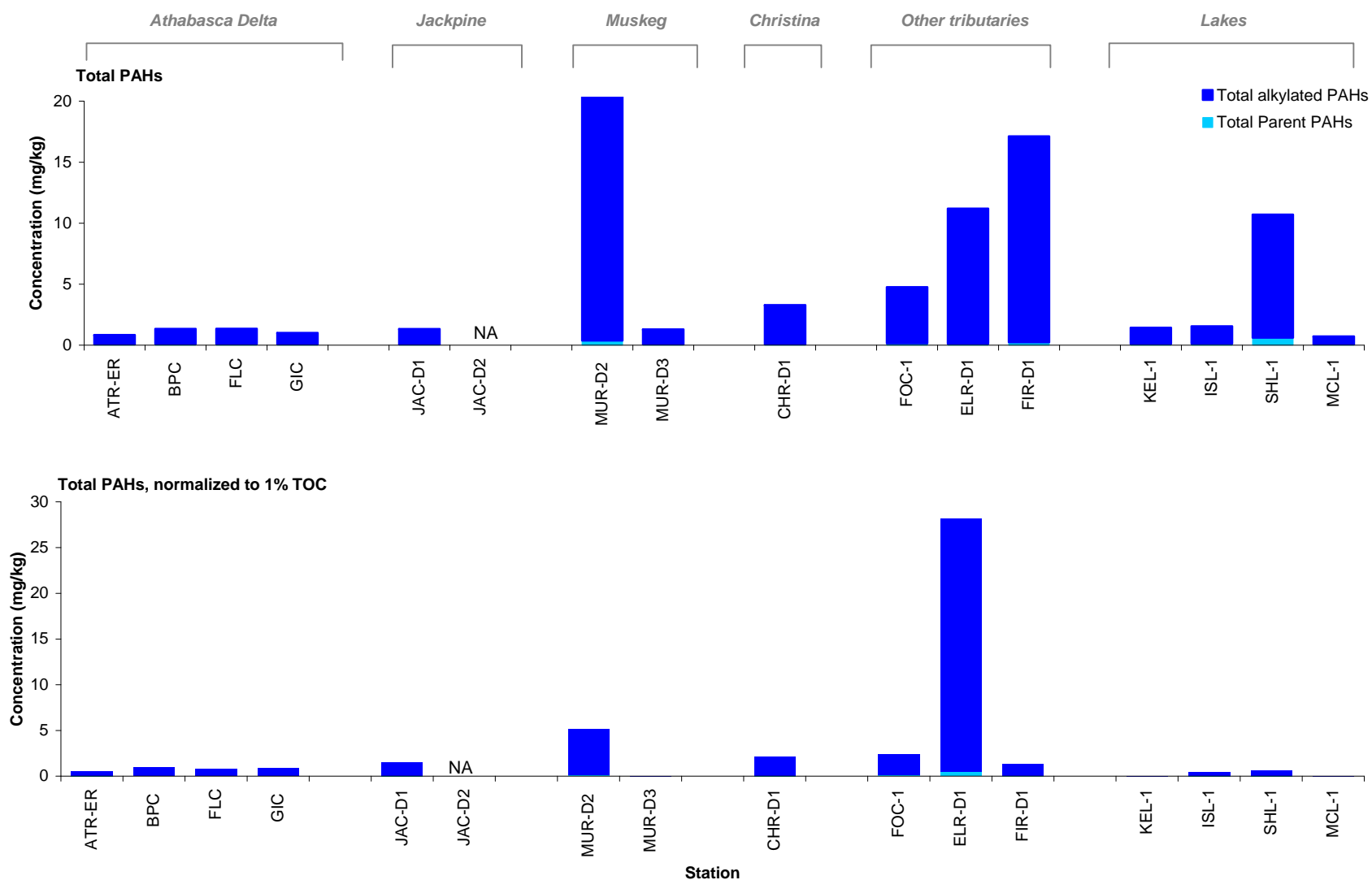
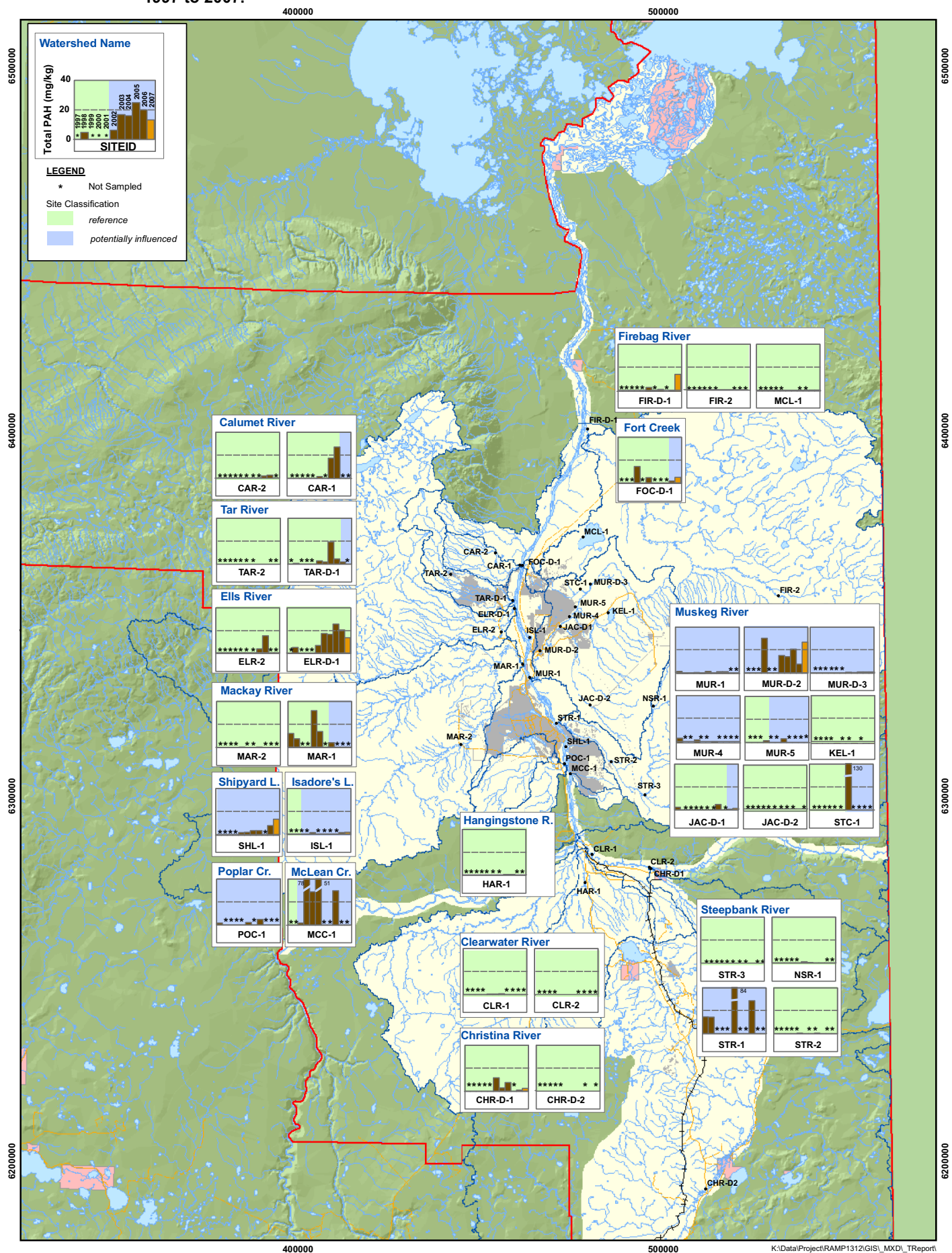
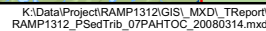


Figure 6.3-8 Concentrations of total PAHs in sediments sampled by RAMP in tributaries to the Athabasca River, 1997 to 2007.



500000



Metals

Most metals measured in RAMP sediments are highly inter-correlated. Principal component analysis (PCA) of metals data in sediments (2006 and 2007 data, $n=30$; Appendix F) found that the first derived principal component (total metals PC1) explained over 65% of the total variance in this dataset, and that most metals measured (i.e., 15 of 21 included in PCA) were strongly correlated (i.e., $r_s > 0.75$) with this single PC. These relationships indicate a generally consistent composition of metals in sediments throughout the study area. Several metals were excluded from PCA because they were non-detectable in most samples collected in 2006 and 2007, including antimony (non-detectable in 100% of samples), beryllium (73% non-detects), bismuth (100%), cadmium (73%), mercury (100%), molybdenum (60%), silver (100%), thallium (70%), tin (97%), and uranium (50%).

Concentrations of total metals in sediments sampled by RAMP are presented in Figure 6.3-10, both in absolute concentrations and in concentrations normalized to percent fine sediments (i.e., silt and clay). For 2006 and 2007 RAMP sediments, total metals PC1 was strongly positively correlated with fine fractions of sediment (i.e., $r_s = 0.850$ versus %-silt, and 0.714 versus %-clay), but strongly negatively correlated with %-sand ($r_s = -0.869$), indicating that metals concentrations were nearly always higher in fine rather than coarse sediments. Total metals concentrations in sediments in 2007 were relatively variable among stations (i.e., from below 30 to nearly 700 mg/kg), with highest concentrations in the upper Muskeg River (MUR-D3, *reference*) and Isadore's Lake (*reference*). Following normalization to the percent fines, sediment metals exhibited much more consistent concentrations among stations, with highest concentrations in the lower Ells River (*reference*), Kearl Lake (*reference*), and the upper Muskeg (*reference*). Concentrations of metals in sediments were very similar at all Athabasca delta stations, and were intermediate between highest and lowest concentrations observed in tributaries (Figure 6.3-10).

Arsenic has been frequently been measured by RAMP at concentrations near or above the CCME ISQG of 5.9 mg/kg in sediments since 1997 (Figure 6.3-11). The highest concentrations of arsenic observed in sediments sampled by RAMP have been in the upper Tar and upper Calumet rivers in 2005 (both *reference*) and in Stanley Creek in 2003 (potentially influenced). The observed concentration at Stanley Creek was the only one to have exceeded the CCME PEL (i.e., 18.5 mg/kg vs. 17 mg/kg PEL). It should be noted that these high-arsenic sediment samples were taken from slow-flowing or wetland areas, and contained large amounts of plant material (for example, Stanley Creek sediments in 2003 were over 40% organic carbon); arsenic has been shown to accumulate in plants, although it does not biomagnify between trophic levels (ATSDR 2007). Generally, no consistent differences in sediment-borne arsenic were apparent between *reference* and *potentially influenced* stations, or over time, with the potential exception of a short-term increase, followed by a similar decrease, in arsenic levels in the lower Tar and Ells rivers from 2002 to 2007. Generally, concentrations of arsenic in sediments collected by RAMP since 1997 are consistent with, or lower than, those observed by others throughout lower the Athabasca-Slave-Mackenzie river basin (DeBoer *et al.* 2007).

Figure 6.3-10 Total metals concentrations in sediments collected by RAMP in 2007, including concentrations normalized to fine-sediment fraction (i.e., %silt+clay).

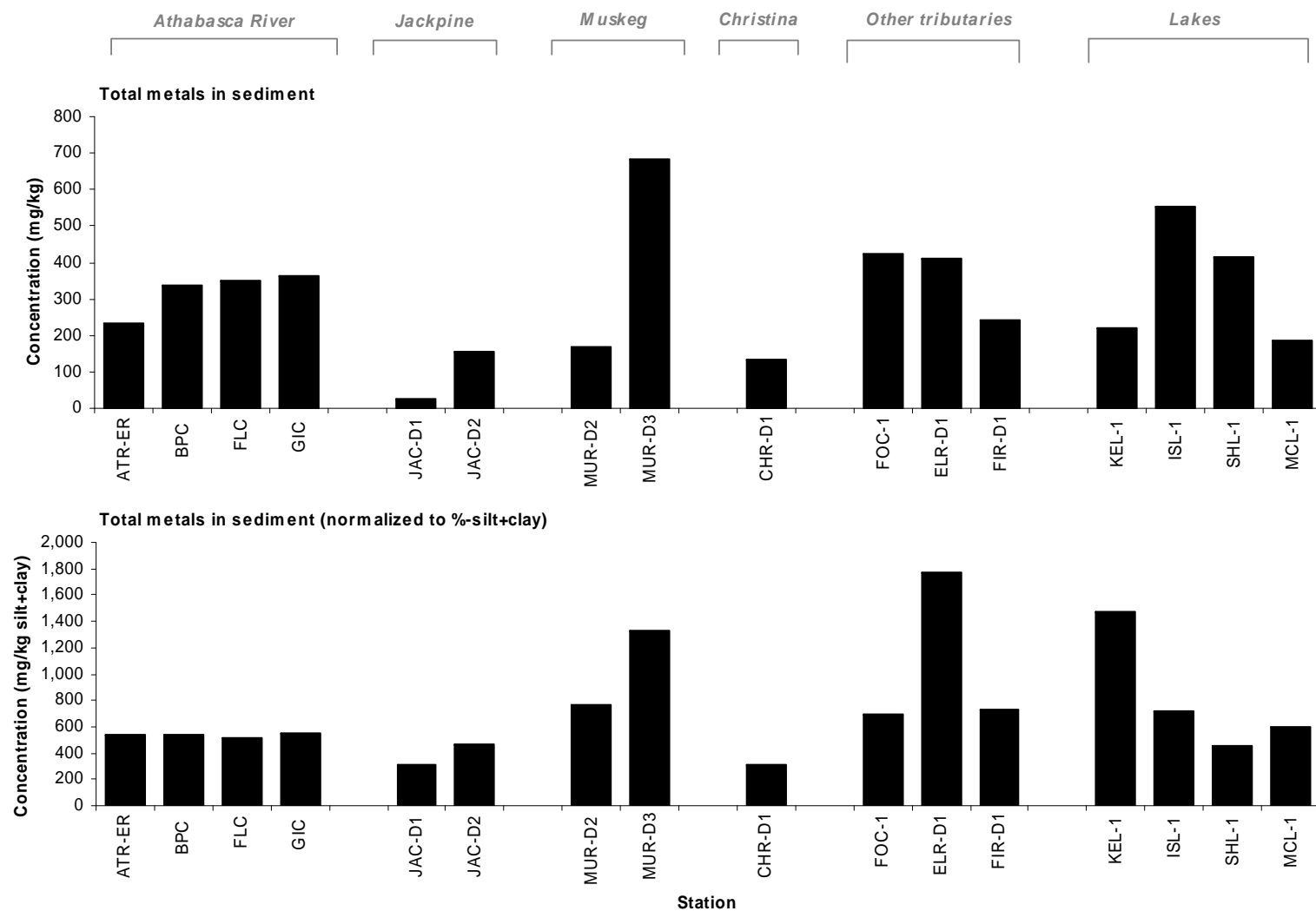
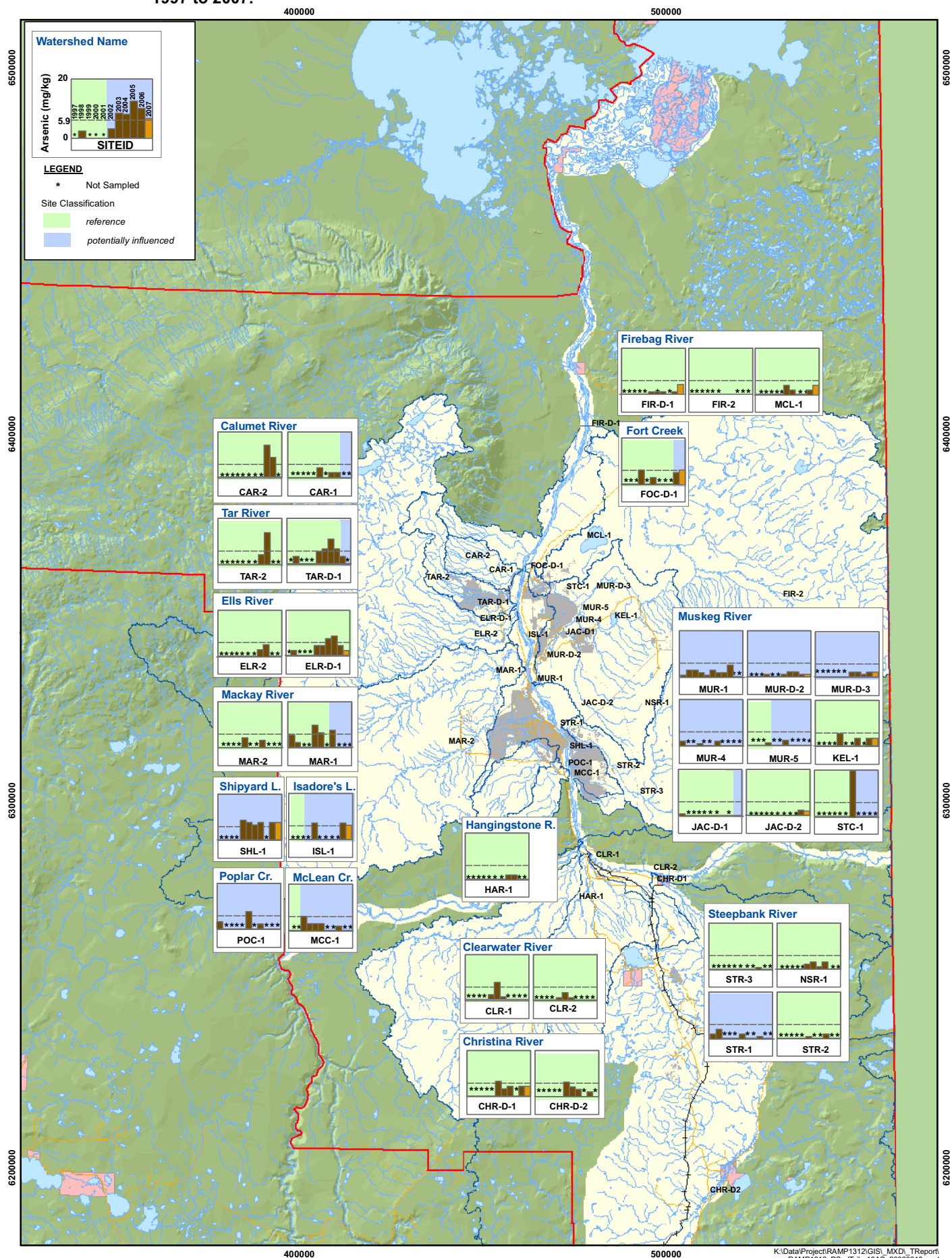


Figure 6.3-11 Concentrations of total arsenic in sediments sampled by RAMP in tributaries to the Athabasca River, 1997 to 2007.



6.3.2.2 Relationships Between Sediment Quality and Benthic Invertebrate Community Endpoints

The RAMP sediment quality component was integrated with the benthic invertebrate component in 2006. This was done for various reasons, a key one of which was the ability to directly examine relationships between sediment quality and the structure of benthic invertebrate communities. Results of correlation analysis of these endpoints appear in Table 6.3-1 below.

Table 6.3-1 Correlations (Spearman's coefficients) between benthic invertebrate community and sediment quality endpoints, 2006 and 2007.

Sediment Endpoint	Benthic Invertebrate Endpoint				
	Abundance	Taxa Richness	Simpson's Diversity	Evenness	%-EPT
Physical Variables					
% Clay	0.289	-0.035	-0.014	0.007	0.080
% Sand	<i>-0.471</i>	0.01	0.119	0.113	0.085
% Silt	0.509	-0.035	-0.187	-0.201	-0.226
Total organic carbon	0.106	0.152	<i>0.425</i>	<i>0.437</i>	0.288
Inorganic carbon	0.503	0.145	-0.07	-0.148	-0.141
Total carbon	0.204	0.138	<i>0.343</i>	<i>0.384</i>	0.265
Hydrocarbons & PAHs					
CCME F2	0.084	0.085	-0.028	0.026	-0.255
CCME F3	0.179	0.124	0.24	<i>0.333</i>	0.007
CCME F4	0.002	-0.011	0.13	0.199	-0.196
CCME total HC	0.118	0.104	0.256	<i>0.342</i>	-0.032
Naphthalene	<i>0.421</i>	0.137	0.068	-0.004	0.155
Retene	<i>0.327</i>	0.216	<i>0.321</i>	<i>0.343</i>	0.249
Total dibenzothipenes	<i>0.337</i>	0.167	0.118	0.167	-0.008
Total PAHs	0.253	0.07	0.109	0.143	-0.209
High-mol.-weight PAHs	0.208	-0.055	-0.083	-0.134	-0.390
Low-mol.-weight PAHs	0.295	0.162	0.128	0.195	-0.07
Predicted PAH Toxicity	0.189	0.042	0.019	0.005	-0.146
Metals					
Total metals PC1	0.214	-0.167	-0.106	-0.084	-0.134

n=30; Critical value of $r_s=|0.309|$; values in italics indicate significant correlation; values in bold indicate moderate correlation (i.e., $|0.50| > r_s > |0.75|$).

Although several correlations were statistically significant (i.e., $r_s > |0.309|$), there were no strong correlations between sediment and benthic community endpoints. Only two moderate correlations were observed: between invertebrate abundance and %-silt ($r_s=0.509$); and between abundance and inorganic carbon ($r_s=0.503$). However, when considered with observed correlations among sediment variables and in the context of invertebrate habitat requirements, several of the weak but significant correlations observed are consistent. Generally, invertebrate communities in these depositional

environments exhibited higher abundance in sediments with finer particle sizes (except EPTs – mayflies, stoneflies and caddisflies – which were less abundant in more depositional environments, as would be expected given their preferred habitats are erosional).

Although benthic abundance also was significantly, positively correlated with concentrations of hydrocarbons and PAHs in sediments, these chemicals were themselves strongly correlated with fine, carbon-rich sediments (which were associated with abundance). There were no significant correlations between predicted PAH toxicity (an estimate of the likely toxicity of PAH mixtures present in sediments at each sampled location, based on octanol-water partitioning coefficients [K_{ow}] and known aquatic toxicity of each PAH species) and any invertebrate endpoint. Taken together, these results suggest that the depositional nature of these habitats exerted a stronger influence on benthic invertebrate communities than concentrations of hydrocarbons or PAHs. Negative correlations between hydrocarbons/PAHs and EPT taxa likely reflect similar relationships with physical habitat suitability.

Total metals in sediments were not significantly correlated with any benthic invertebrate endpoint.

6.3.2.3 Summary

Sediments in the RAMP study area naturally contain hydrocarbons and PAHs at concentrations that may exceed environmental-quality guidelines. Spatial and temporal comparisons of sediment quality since monitoring by RAMP began in 1997 do not indicate any consistent trends over time in concentrations of hydrocarbons or metals, any consistent differences in sediment quality between *reference* and *potentially influenced* stations, or any relationships between sediment chemistry and composition of benthic communities.

6.4 FISH POPULATIONS

The 2007 RAMP fish component included surveys in the following watersheds:

- Fish inventories on the Athabasca and Clearwater rivers;
- Chemical analyses of fish tissue collected from the Clearwater River;
- Sentinel species monitoring on the Athabasca and Elks rivers; and
- Chemical analyses of fish tissue collected from the following regional lakes: Gregoire and Namur lakes.

The intention of this section is to provide a regional context for measurement endpoints in fish populations monitored during programs completed in 2007 in relation to programs conducted in waterbodies during historical RAMP fish programs (1997-2006), surveys completed prior to 1997 (i.e., prior to RAMP) and regional studies completed in and surrounding the RAMP study area. Endpoints, which include mercury concentrations in fish tissue, condition factor, gonadosomatic index and liver somatic index, species diversity and catch per unit effort, were evaluated temporally and spatially, particularly as they relate to oil sands development.

6.4.1 Mercury in Fish Tissue

The RAMP fish tissue program collected samples from Key Indicator Resource (KIR) species in Gregoire Lake (walleye, lake whitefish and northern pike), Namur Lake (lake trout) and the Clearwater River (northern pike) in 2007. As a consistent concern for communities living in Northern Alberta, the program was designed to assess mercury concentrations in fish tissue frequently consumed by humans. Health Canada provides

human consumption mercury guideline concentrations in fish tissue for subsistence fishers (0.2 mg/kg) and general consumers (0.5 mg/kg) (CFIA 2003, Health Canada 1978, as cited in Lockhart *et al.* [1995]). Results for northern pike collected from the Clearwater River indicated that 19% of fish exceeded the Health Canada subsistence fisher guideline; none exceeded the general consumer guideline. Similarly, 38% of walleye collected from Gregoire Lake exceeded the subsistence fisher guideline (none exceeded the guideline for general consumers) and 46% of northern pike exceeded the subsistence fisher guideline (one fish exceeded the general consumer guideline). Lake whitefish from Gregoire Lake posed the lowest consumption risk as none were found to exceed subsistence or general consumer guidelines. Results for lake trout collected from Namur Lake represented the highest consumption risk with 75% of fish exceeding the subsistence fisher guideline; seven of those fish exceeding the general consumer guideline.

To provide a regional context for these mercury levels, Figure 6.4-1 displays the 2007 fish tissue results relative to mercury concentrations in fish tissue from waterbodies not currently impacted by focal projects and from previous RAMP sampling (Grey *et al.* 1995, Golder 2004, RAMP 2003, RAMP 2004). Mercury concentrations in each waterbody were averaged over all individuals sampled for each species (male and female individuals were combined given the small variation in mercury concentrations observed between the two sexes).

Regionally, eleven of the twenty-two lakes (50%) sampled for walleye showed an exceedance of the mean mercury concentration of the subsistence fisher guideline but none exceeded the general consumer guideline; ten of the twenty-one lakes (48%) sampled for northern pike showed an exceedance of the subsistence fisher guideline, but no exceedances of the general consumer guideline; there were no guideline exceedances of mercury concentrations in lake whitefish. Mean mercury concentrations in waterbodies downstream of oil sands development (i.e., Athabasca River, Lake Athabasca, Lake Claire and the Muskeg River) fell within range of regional mercury concentrations from fish in waterbodies beyond the influence of oil sands development. Only one lake, Namur Lake, was sampled for lake trout in 2000 (Evans *et al.* 2005, Doetzel 2007) and 2007; in both years the average mercury concentration in lake trout exceeded the subsistence guideline but not the general consumer guideline.

For available temporal data, the mean mercury concentration standardized to fish weight in northern pike in the Clearwater River was higher in 2004 than in 2007, whereas the mean concentration in females in Gregoire Lake (considered a *reference* waterbody) was higher in 2007 than 2002 but lower for male northern pike. Lake whitefish mercury concentrations in 2007 in Gregoire Lake were slightly higher than 2002 for both males and females. The average mercury concentration in walleye in Gregoire Lake was higher in 2007 than 2002. Lake trout mean mercury concentration in Namur Lake (considered a *reference* waterbody) was higher in 2007 than 2000. Given there were no other lakes sampled for lake trout, it was difficult to make regional comparisons, however, mercury is generally high in lake trout given their high position on the food chain (MRBB 2003). The limited temporal data available suggest that the variability in mercury concentrations over time can be high, with no consistent pattern among species or direction.

Historical regional assessments of mercury levels have shown some evidence that concentrations are generally high in freshwater lakes and rivers in Northern Canada (INAC 2003, MRBB 2003, Lockhart *et al.* 2005). Mercury naturally occurs in soils, bedrock and peatland areas and is introduced into the aquatic environment via runoff through surrounding soils or during periods of dewatering (Grigal 2003); anthropogenic inputs of mercury come from fossil fuel combustion released first into the atmosphere through emissions and then as depositional fallout to aquatic environments, possibly through

long-range transport to areas not directly impacted by development (Rada *et al.* 1989). Concern from a fishers perspective arises because mercury bioaccumulates in fish in the toxic form of methyl-mercury when it is transformed microbially in sediment (Ullrich *et al.* 2001). Given that a number of species of interest to the communities in this area (i.e., northern pike and walleye) are piscivorous species, mercury levels are much higher than in species lower on the food chain. Factors affecting the amount of methyl-mercury in a fish include size, age, trophic status, and sediment and water chemistry (INAC 2003).

Studies have shown that mercury is naturally present in uncontaminated freshwater fish at concentrations of 0.2 mg/kg, but can be as high as 1 mg/kg in waters near natural geological sources of mercury (Craig 1986, Ullrich *et al.* 2001). Sampled waterbodies shown in Figure 6.4-1 fall within this natural range of mercury concentration. In addition to natural sources, the degree to which atmospheric deposition contributes to mercury levels in freshwater lakes and rivers is still unknown as it is difficult to differentiate mercury sources in fish tissue. There is no known evidence to indicate that oil sands development contribute to atmospheric mercury. However, results from a study with the Northern Rivers Ecosystem Initiative (2002) completed from 1997 to 1999 showed that average atmospheric concentrations of mercury in the Fort Chipewyan area, downstream of Fort McMurray, were lower (1.37 ng/m³) than average concentrations in Canada (1.60 ng/m³) recorded by the Canadian Atmospheric Mercury Network (ARQP 2002).

Given the factors mentioned above and the variability of mercury in fish both spatially and temporally, the influence of natural versus anthropogenic sources on levels of mercury observed in fish in this region merits further research.

6.4.2 Fish Inventory Program

In 2007, the inventory program was conducted on the Athabasca and Clearwater rivers in spring and fall to assess relative population abundance of large-bodied species (i.e., walleye, white sucker, northern pike, longnose sucker and goldeye). Given the fish inventory is a community driven program with a focus on large-bodied species commonly caught in local subsistence, sport and commercial fisheries, significant measurement endpoints of interest are relative abundance (as estimated by catch per unit effort) and species diversity. Catch per unit effort was calculated per one hundred seconds of electrofishing for all large-bodied species combined in each sampling area. Species diversity as a function of species richness and relative abundance was also calculated for each sampling area using Simpson's index of diversity.

Given that the inventory program was not conducted on a regional scale, the 2007 results were compared with fish inventory programs conducted from 1987 to 2006 on the Athabasca River and from 2003 to 2006 on the Clearwater River during spring and fall to provide an overall assessment of measurement endpoints (Figure 6.4-2).

Catch per unit effort (CPUE) in all areas has fluctuated over time with no clear decreasing or increasing trends. On a temporal scale, comparisons of CPUE in years prior to RAMP (1987-1996) showed that the CPUE in 2007 either fell within or exceeded historical ranges (Poplar area - 1.08 to 2.19, Steepbank area - 0.74 to 6.68, Muskeg area - 0.75 to 2.03 and Tar-Ells area - 0.56 to 1.28). Given all areas on the Athabasca River are classified as *potentially influenced*, comparisons with CPUE from the Clearwater River (reference) were evaluated. Mean CPUE in all areas in the Athabasca River fell within the range of CPUE in the *reference* areas (Clearwater River - 0.62 to 4.8) across sampling years indicating a consistent natural range of variability in relative abundance.

Generally, species diversity at all sampling reaches showed large fluctuations over time, but was generally greater than 0.5 in all sampling years. With the exception of the Poplar

area, species diversity in 2007 fell within the historical range (i.e., prior to 1997) (Poplar area – 0.44 to 0.78, Steepbank area – 0.31 to 0.68, Muskeg area – 0.39 to 0.71 and Tar-Ells area – 0.46 to 0.65). On the Athabasca River, across all years in each *potentially influenced* area, species diversity was within the *reference* area range (Clearwater River 0.35 – 0.76) indicating changes in *potentially influenced* areas consistent with areas not currently influenced by oil sands development.

Temporal changes and spatial differences between *reference* and *potentially influenced* fish populations will continue to be monitored as more data is collected throughout the Athabasca and Clearwater rivers.

6.4.3 Sentinel Species Program

A number of sentinel species programs have been conducted under RAMP (Table 3.4-7). The measurement endpoints used to determine changes between *reference* and *potentially influenced* sites in the sentinel species programs have been condition factor (K) for lethal and non-lethal sampling programs and gonadosomatic index (GSI) and liver somatic index (LSI) for lethal sampling programs. These measurements were evaluated against the Environment Canada (2005) impact criterion of a $\pm 10\%$ or $\pm 25\%$ difference between the *potentially influenced* and *reference* sites for condition and LSI/GSI, respectively. To provide a regional context for condition, GSI and LSI in fish in the RAMP study area measured in the RAMP sentinel species programs are provided in Figure 6.4-3; this figure shows the measurement endpoints over time at each site, including the years and sites where mean measurement endpoints exceeded the impact criterion.

In 1999, 2004 and 2006, mean condition factor in slimy sculpin at the Lower Muskeg River site (*potentially influenced*) exceeded the -10% criterion relative to the reference sites at the Upper Steepbank River, the Horse River and the Dunkirk River; in 2005 and 2007, mean condition in longnose dace at the Lower Ells River site was greater than the +10% criterion (2005) and less than the -10% criterion (2007), respectively, relative to the Upper Ells River site (both currently classified *reference*); in 2007 fish at Site 3 on the Athabasca River (upstream of the Muskeg River confluence) had a mean condition factor greater than the +10% threshold. The gonadosomatic index of fish in the Muskeg and Steepbank rivers in 2001 exceeded the -25% threshold. The liver somatic index of fish in the Athabasca River at Site 5 (downstream of the Firebag River) in 2002 exceeded the +25% criteria relative to the *reference* site upstream of oil sands development.

Generally, trout-perch in the Athabasca River at *potentially influenced* sites showed higher condition, GSI and LSI than trout-perch at *reference* sites. Although there was some variability among sites, this response does not suggest an industry-related effects but rather a greater availability of food (Gray *et al.* 2002). Contrastingly, in the tributaries, slimy sculpin primarily had lower condition, GSI and LSI at *potentially influenced* sites compared to *reference* sites, a response suggesting reduced availability of resources potentially through reduced food abundance, or increased competition for food or habitat (Gibbons and Munkittrick 2004). However, without further follow-up studies, it is difficult to know whether this is related to development activities in the Muskeg watershed, particularly given the lack of effects found in benthic invertebrates, water quality and sediment quality. As well, the inconsistency in the response of longnose dace in the Ells River, a reference watershed, highlights the sensitivity of small-bodied fish species to conditions of the aquatic environment in which they reside. Accordingly, it is important to develop a strong baseline to facilitate the identification of an impact versus natural variability. Regardless, to date, the observed responses in sentinel species monitored in *potentially influenced* sites have not exhibited large-scale differences relative to reference populations.

Figure 6.4-1 Mercury concentrations in tissue of fish captured during the fish tissue sampling program in the RAMP focal study area and in other waterbodies in Northern Alberta, 1989-2007.

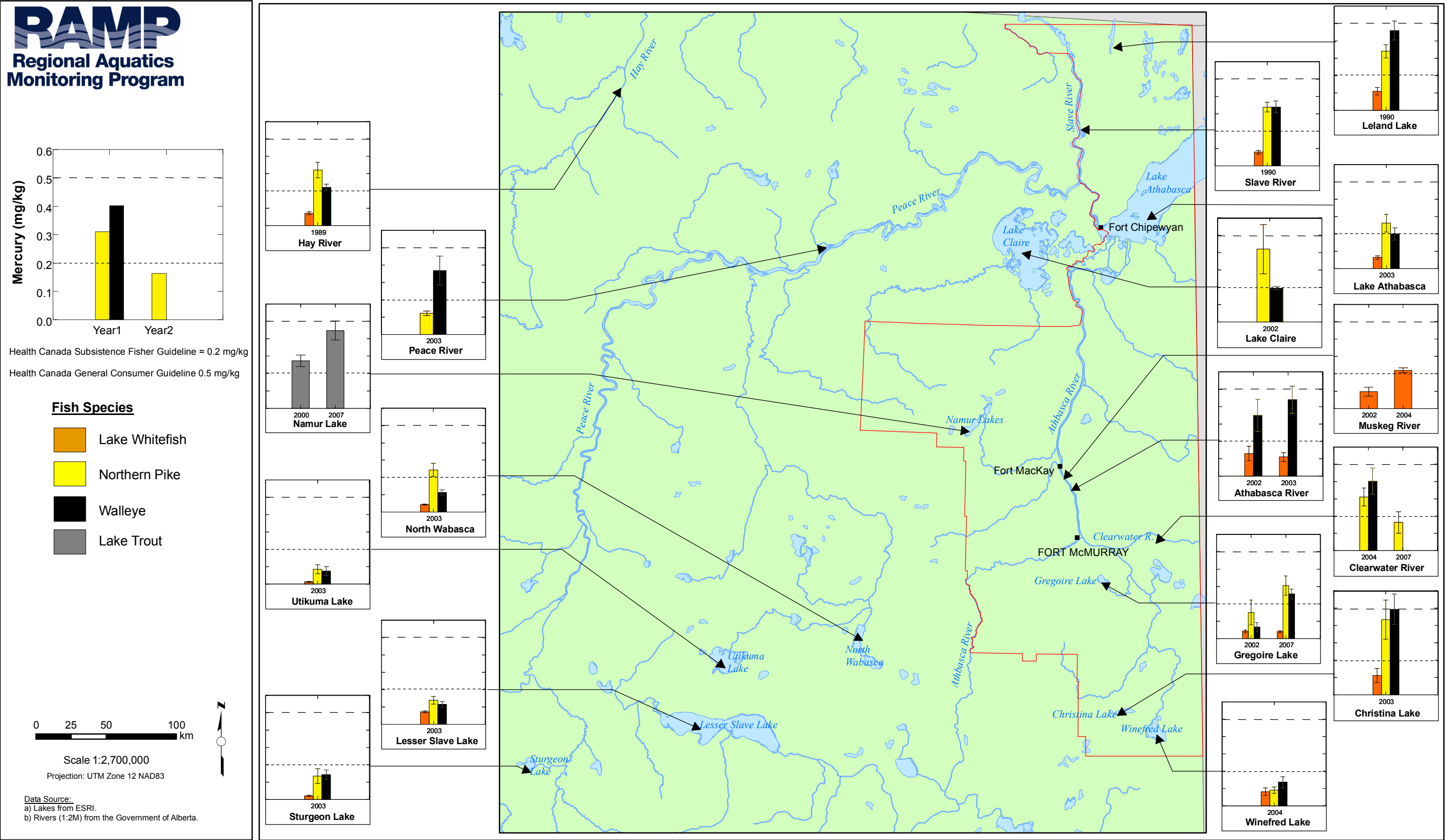


Figure 6.4-2 Species diversity and catch per unit effort (CPUE) of large-bodied fish captured during the spring and fall fish inventory programs in the Athabasca and Clearwater rivers, 1987-2007.

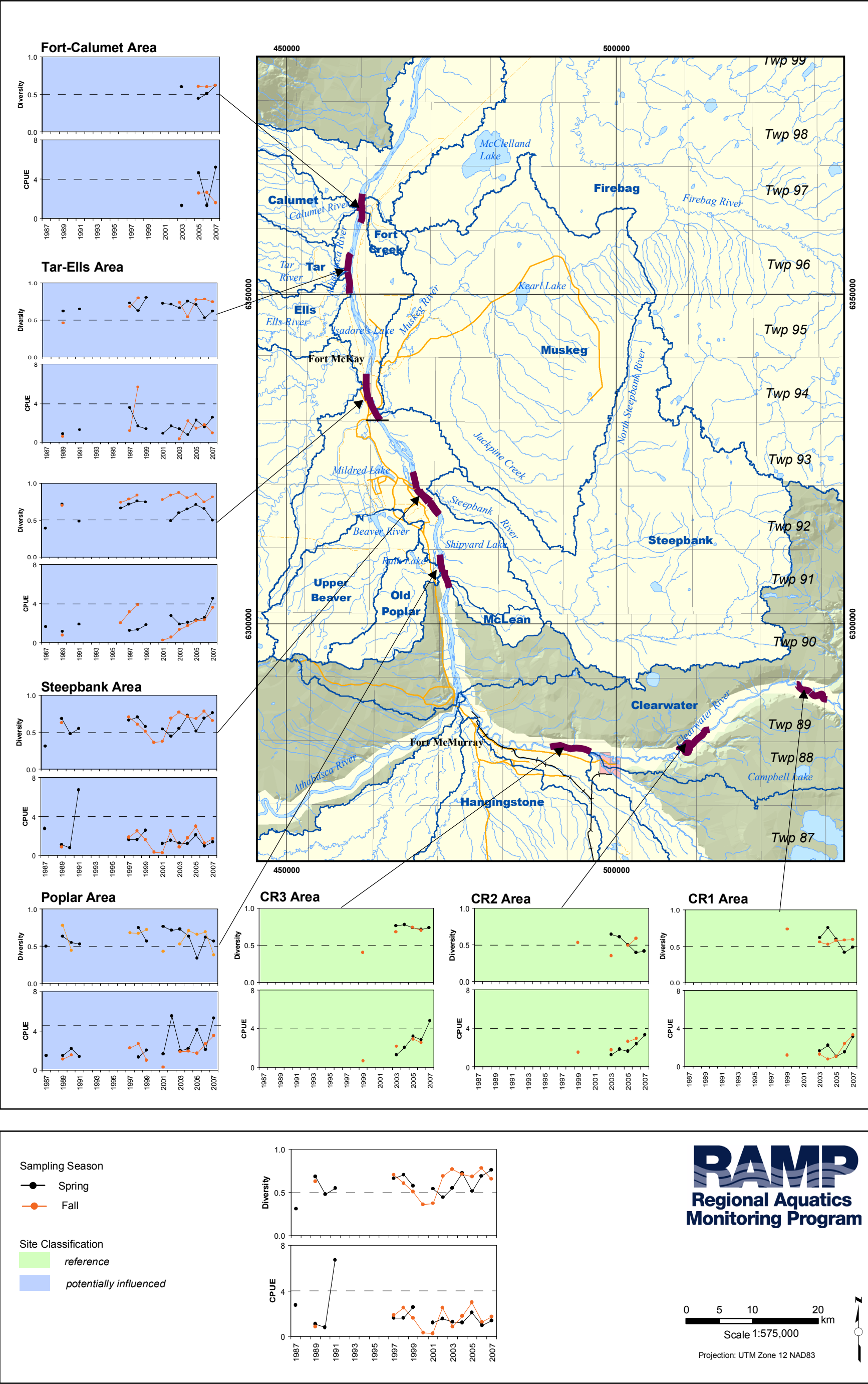
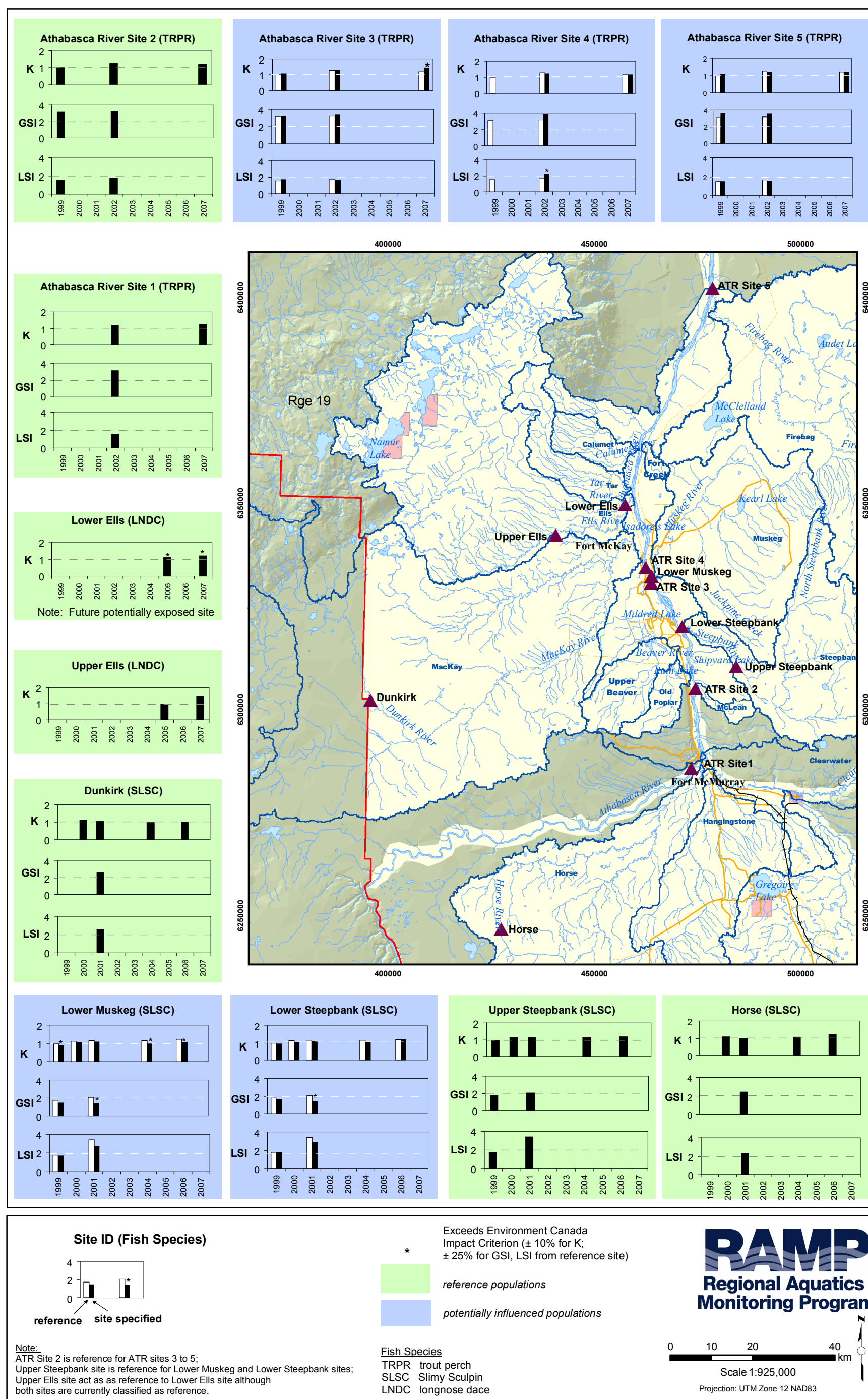


Figure 6.4-3 Condition (K), gonadosomatic index (GSI), and liver somatic index (LSI) of fish in regional waterbodies captured during the sentinel species programs in the RAMP focal study area, 1997-2007.



6.5 ACID-SENSITIVE LAKES

This section presents the results of the Acid-Sensitive Lakes (ASL) component of RAMP for 2006. A general description of the 50 RAMP lakes is provided as well as three primary analyses of the RAMP ASL lake dataset:

- **Between-Year Comparison of ASL Measurement Endpoints** An Analysis of Variance (ANOVA) to determine whether there have been any significant changes in the ASL measurement endpoints over the five years of data available for the 50 lakes;
- **Calculation of Critical Loads of Acidity and Critical Load Exceedances** A Calculation of the Critical Load of acidity (CL) for each RAMP ASL lake and a comparison of the CL values to recent estimates of Potential Acid Input (PAI) for each ASL lake; and
- **Trends in ASL Measurement Endpoints** An analysis of potential trends in ASL measurement endpoints in individual lakes.

These primary analyses are supported by the additional data analysis, the results of which are presented in Appendix H:

- The chemical characteristics of the RAMP ASL lakes were reviewed with the addition of the 2006 data. Summary statistics were calculated on the updated dataset that now includes five years of data on all 50 lakes;
- Trace metal concentrations in the RAMP ASL lakes were summarized and relationships between metal concentrations, lake location and chemistry were noted; and
- Estimates of the seasonal variability in water quality variables in ten of the ASL lakes were updated with the 2006 data and summary statistics were calculated.

6.5.1 General Characteristics of the 50 RAMP Lakes

The chemical parameters measured in the 50 RAMP lakes from 1999 to 2007 are summarized in Table 6.5-1. Chemically the RAMP lakes cover a large range of lake types from softwater to hardwater. Historically, the pH of the lakes has ranged from 3.97 to 9.46 with a median value of 6.76. Gran alkalinity has ranged from negative values to 1802 µeq/L with a median of 198 µeq/L. Concentrations of sulphate are relatively low and range from non-detectable to 16.7 mg/L with a median concentration of 1.13 mg/L. By conventional standards, most of the RAMP lakes are considered humic with a median dissolved organic carbon (DOC) concentration of 21.5 mg/L (Korteleinen et al. 1989; Forsius 1992; Driscoll et al. 1991). Over 50% of the RAMP ASL lakes are considered to be highly sensitive or moderately sensitive to acidification (Section 3.5.1).

Lakes having “unusual” chemistry were identified in the 2007 monitoring data as those falling below or above the 5th and 95th percentile for pH, Gran alkalinity, and DOC (Table 6.5-22). These lakes were essentially the same lakes identified in 2006 (RAMP 2007). Three lakes (168/A21, 169/A24 and 448 Clayton L.) had very low (zero or negative) levels of Gran alkalinity. Two of these lakes were found in the Stony Mountains upland region while the other was found in the Birch Mountains. The highest values of Gran alkalinity and buffering capacities in the RAMP ASL lakes were found in Lakes 270, 271 and Kearl Lake, located northeast of Fort McMurray. Lakes having the lowest pH were Lakes 169/A24 and 172/A59 in the Stony Mountains and Clayton Lake (448) in the Birch Mountains. Lakes 270, 271 and Kearl Lake had the highest pH.

Table 6.5-1 Summary of the chemical characteristics of the RAMP ASL Lakes.

Parameter	Minimum		Maximum		Mean		Median		5th Percentile 2007	95th Percentile 2007
	1999- 2007	2007	1999- 2007	2007	1999- 2007	2007	1999- 2007	2007		
pH	3.97	4.28	9.46	8.16	6.55	6.51	6.76	6.66	4.88	7.67
Total Alkalinity (µeq/L)	ND	ND	1784	1784	310	345	214	224	29.24	1219.2
Gran Alkalinity (µeq/L)	-57.2	-19.6	1802	1802	298	337	194	198	2.03	1216.9
Specific Cond. (µS/cm)	10.5	10.5	180.4	180	43.6	47.0	32.4	33.5	11.6	120
Turbidity (NTU)	0.38	0.40	53.0	14.0	3.91	3.71	1.80	2.20	0.52	11.7
Colour (TCU)	8.00	11.5	948	948	149	151	119	113	19.9	316
Sodium (mg/L)	0.18	0.69	10.4	10.4	1.93	2.28	1.26	1.33	0.73	7.16
Potassium (mg/L)	ND	0.12	2.40	2.11	0.51	0.59	0.44	0.44	0.19	1.26
Calcium (mg/L)	0.20	0.75	32.2	16.57	5.65	5.20	4.62	4.57	1.02	13.8
Magnesium (mg/L)	0.11	0.27	13.6	6.08	1.79	1.71	1.40	1.28	0.36	4.34
Bicarbonate (mg/L)	ND	ND	109	109	18.8	21.1	13.1	13.8	1.80	74.0
Chloride (mg/L)	ND	0.08	2.64	2.52	0.37	0.31	0.19	0.19	0.10	1.02
Sulphate (mg/L)	ND	ND	16.7	14.1	2.21	2.59	1.13	1.51	0.39	10.7
Total Diss. Nitrogen (µg/L)	105	273.00	2891	1830	853	736	695	640	312	1376
Ammonia (µg/L)	ND	ND	1509	233	42.0	20.9	15.7	14.5	ND	54.3
Nitrate + Nitrite (µg/L)	ND	ND	733	302	21.4	16.1	3.00	2	ND	51.5
Total Phosphate (µg/L)	3.60	7.00	341	163	56.5	62.9	40.9	52.0	16.7	133
Diss. Phosphate (µg/L)	1.20	3.00	156	102	20.3	22.0	11.7	13.0	3.00	69.5
Diss. Inorg. Carbon (mg/L)	0.03	0.20	20.3	19.4	3.17	3.69	2.00	2.20	0.40	14.0
Diss. Organic Carbon (mg/L)	6.82	7.20	81.2	45.9	22.7	21.3	21.5	20.0	10.7	39.3
Chlorophyll <i>a</i> (µg/L)	0.60	1.94	371	162	20.3	20.0	9.64	10.8	2.85	57.4
Iron (mg/L)	ND	0.02	3.88	1.58	0.34	0.28	0.16	0.11	0.02	0.86
Silica (mg/L)	ND	ND	5.04	5.04	1.10	1.06	0.62	0.57	ND	3.34
Total Nitrogen (µg/L)	0.00	274	6558	3250	1256	1056	998	887	380	2586
Sum Base Cations (µeq/L)	0.00	38.2	2291	1643	563	515	430	444	108	1466
Aluminum (µg/L)	0.10	0.10	681	430	71.0	55.9	22.3	16.00	0.60	224.10

Shaded Parameters are measurement endpoints for the ASL program;

ND = non-detectable

Table 6.5-2 RAMP ASL lakes with chemical characteristics either below the 5th or above the 95th percentile of the 2007 values.

Lake	Region	pH	Gran Alkalinity (µeq/L)	DOC (mg/L)
5th Percentile, 2007		4.88	2.03	10.7
95th Percentile, 2007		7.67	1217	39.3
168 (A21)	Stony Mountains	5.01	-19.6	17.4
169 (A24)	Stony Mountains	4.73	-13.2	15.8
172 (A59)	Stony Mountains	4.81	28	32.4
175 (P13)	Birch Mountains	7.35	835	44.1
436 (L18) Namur	Birch Mountains	7.05	431	7.7
444 (L25) Legend	Birch Mountains	6.67	188	7.2
448 (L29) Clayton	Birch Mountains	4.28	0	13.1
270	NE of Fort McMurray	7.79	1327	22.8
271	NE of Fort McMurray	8.16	1244	19.9
418/Kearl	Northeast of Fort McMurray	7.88	1802	24.5
118 (L107)	Canadian Shield	7.06	446	10.5
165 (A42)	West of Fort McMurray	6.88	368	41.6
223 (P94)	West of Fort McMurray	7.22	70.	45.9

Red values represent those values below the 5th percentile for that variable in the 2007 data.

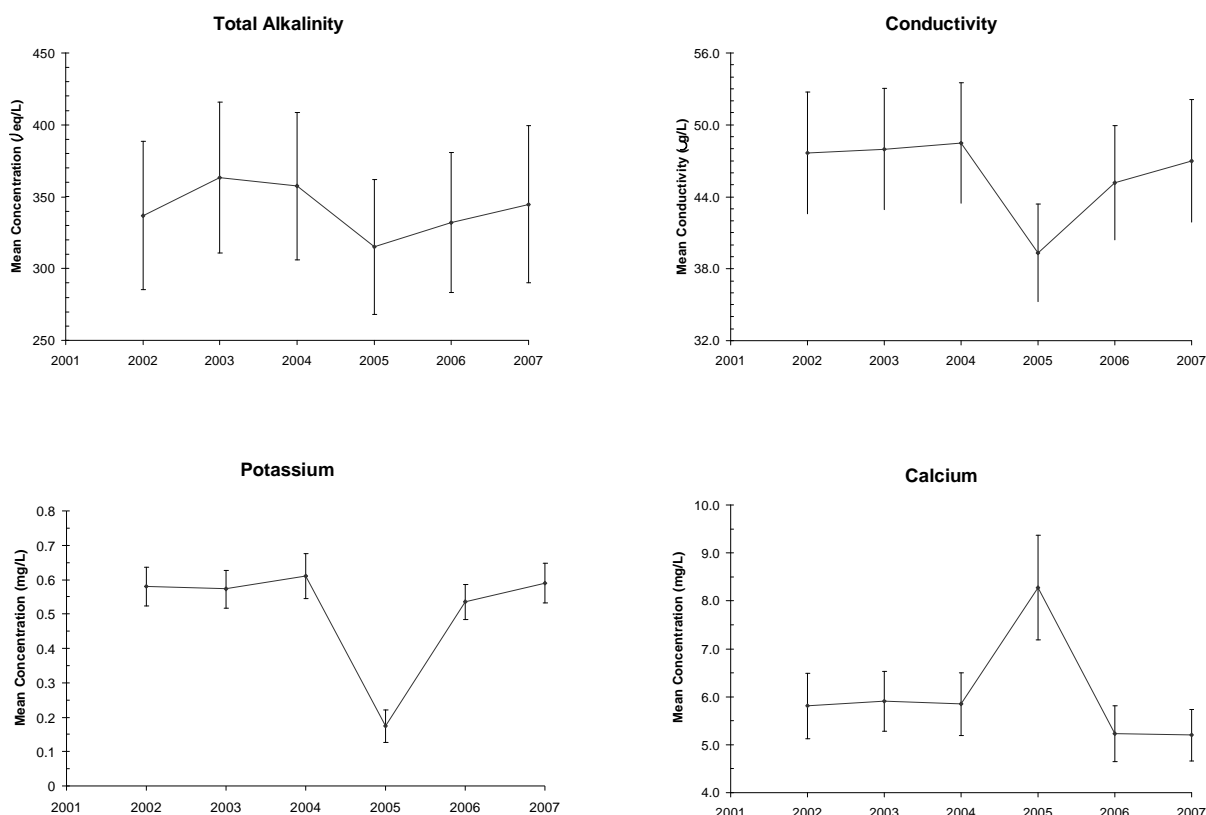
Blue values represent those values above the 95th percentile.

The lowest levels of DOC were found in Namur Lake (436) and Legend Lake (444) in the Birch Mountains and in Lake 118/L107 in the Canadian Shield. The highest concentrations of DOC were found in Lake 165/A42 and Lake 223 West of Fort McMurray sub-region and in Lake 175, a pond in the Birch Mountains. As indicated in previous RAMP reports, lakes with low levels of Gran alkalinity were generally the same lakes having low pH, high DOC and low conductivity. These were often fairly small, shallow lakes found in the upland regions. Unique to the RAMP lakes are those lakes like Lake 221 that are simultaneously high in pH and high in DOC. Most coloured (high DOC) lakes are typically low in pH (Korteinen et al. 1989).

6.5.2 Between-Year Comparison of ASL Measurement Endpoints

As in previous years, an Analysis of Variance (ANOVA) was performed in order to determine whether there have been any significant changes in ASL measurement endpoints over the six years of data available for the 50 ASL lakes (2002-2007). No significant changes in ASL measurement endpoints were observed over the 6 years. However, as noted in the 2006 technical report, significant changes were observed in 2005 for two variables (potassium and calcium) that affect the sum of base cations. Distinct changes in conductivity and Gran alkalinity were also observed in 2005, although these were not statistically significant. These changes were related to high rates of precipitation and surface runoff in 2005. Conditions appeared to have returned to normal in 2006 and normal conditions continued into 2007 (Figure 6.5-1).

Figure 6.5-1 Mean concentrations of selected variables over all the RAMP lakes and years of the ASL component.



Note: Error bars represent one standard error of the mean

6.5.3 Critical Loads of Acidity and Critical Load Exceedances

The critical loads of acidity (CL) were calculated for each RAMP lake for the years 1999 to 2006 using the Henriksen steady state water chemistry model modified to include the contribution of organic anions as both strong acids and weak organic buffers (WRS 2006; RAMP 2005a). The critical load is an inherent property of each lake that defines the greatest load of acidifying substances that will not cause ecological damage to the lake. The CL therefore represents a measure of the acid-sensitivity of a lake. The lower the critical load the more sensitive the lake to acidification.

As in 2006, the runoff to each lake, a term in the Henriksen model, was calculated both from traditional hydrometric methods and from analysis of heavy isotopes of oxygen (^{18}O) and (^2H) in each lake. Table 6.5-3 presents the two estimates of runoff and critical loads of acidity between 2002 and 2007. The isotopically-derived runoff values were greater than the hydrometrically derived values in 20 lakes, lower in 27 lakes and identical in one lake. The greatest discrepancies were observed for lakes having the highest rates of runoff.

Using the hydrometrically derived runoff, the critical loads in 2007 ranged from $-0.070 \text{ keq H}^+/\text{ha}/\text{y}$ to $1.333 \text{ keq H}^+/\text{ha}/\text{y}$ with a median of $0.198 \text{ keq H}^+/\text{ha}/\text{y}$ (Table 6.5-5). With the isotopically derived runoff, critical loads ranged from $-0.048 \text{ keq H}^+/\text{ha}/\text{y}$ to $1.781 \text{ keq H}^+/\text{ha}/\text{y}$ with a median CL of $0.174 \text{ keq H}^+/\text{ha}/\text{y}$. The individual

CL values for each lake often differed significantly although the means and median critical loads for the entire lake population were quite similar for the two methods.

Mean critical loads in 2007 for the two methods (hydrometric/isotopic) in the six sub-regions were calculated as follows:

- Stony Mountains: 0.029/0.036 keq H⁺/ha/y
- West of Fort McMurray: 0.407/0.161 keq H⁺/ha/y
- North-East of Fort McMurray: 0.492/0.405 keq H⁺/ha/y
- Birch Mountains: 0.254/0.219 keq H⁺/ha/y
- Canadian Shield: 0.212/0.524 keq H⁺/ha/y
- Caribou Mountains: 0.155/0.479 keq H⁺/ha/y.

Low critical loads observed in the upland regions (the Birch Mountains, the Caribou Mountains and the Stony Mountains) and in the Canadian Shield are consistent with the findings of previous RAMP reports (RAMP 2004, 2005a, 2006). Negative critical loads were observed in many of the lakes, especially in the Stony Mountains sub-region. These lakes may be the most acid-sensitive of the 50 RAMP ASL lakes.

6.5.3.1 Comparison of Critical Loads of Acidity to Modelled Potential Acid Input

The critical load of acidity was compared to modeled rates of acid deposition for each lake published in the Deer Creek Joslyn North Mine Project EIA for the Approved Case (Deer Creek Energy 2006). The approved case includes the depositional effects of all existing and approved projects in the region. Acid input was expressed in units of Potential Acid Input (PAI) which represents the total annual deposition of nitrogen and sulphur in both wet and dry forms minus the neutralizing effects of base cations. The PAI for lakes in the Caribou Mountains and the Canadian Shield regions was estimated from an air modeling study conducted by Alberta Environment using the RELAD model and was equivalent to background PAI values (no industrial input).

Lakes having the modelled PAI greater than the critical load are identified in Table 6.5-3. The percentage of such lakes ranged from a low of 32.7% (16 of 49 lakes) in 2005 to a high of 48.0 (24 of 50 lakes) in 2006 (Table 6.5-4). The use of the isotopically derived runoff resulted in four additional lakes in this category. The percentage of ASL lakes in which the modeled PAI is greater than the critical load is higher than the 8% reported for 399 regional lakes in WRS (2006). The higher proportion in the ASL lakes largely reflects the bias in lake selection in which the most poorly-buffered lakes in the region were chosen preferentially (see Appendix H). The estimates of PAI are also biased high. By incorporating both approved and existing industries in the calculation of the PAI, the estimates of PAI reported in Table 6.5-3 represent future risk (not current risk) to the ASL lakes. For comparison to other regions, Henriksen et al. (2002) reported that 11% to 26 % of lakes in four sensitive regions of Ontario had levels of PAI exceeding the critical load. Their study did not include modifications to the model for organic anions or use of isotopic estimates of runoff. A modeled PAI greater than the critical load of a lake does not mean that acidification is imminent but that there is a potential risk of acidification. Other factors, such as the influence of highly buffered groundwater seepage to each lake should also be considered. Table 6.5-5 summarizes the key chemical characteristics of the lakes having the modelled PAI greater than the critical load. As expected, these are small lakes of low pH, low conductivity, low ANC, and high in DOC. A large proportion of these lakes are found in the Stony and Birch Mountain regions.

Table 6.5-3 Critical loads of acidity in the RAMP ASL Lakes, 2002 to 2007.

ID No.	Original RAMP Desig.	Runoff (Hydro) m³/s	Runoff (Isotopic) m³/s	Mean pH	Mean Gran Alk. (µeq/L)	Mean DOC (mg/L)	Mean SBC (mg/L)	Critical Load of Acidity (keq H⁺/ha/y)								
								2002	2003	2004	2005	2006 (Hydro)	2006 (Isotopic)	2007 (Hydro)	2007 (Isotopic)	PAI¹
Stony Mountains Sub-Region																
168	A21	0.0404	0.0474	4.95	36.0	20.19	145	-0.089	-0.079	-0.087	-0.118	-0.081	-0.096	-0.070	-0.082	0.186
169	A24	0.0264	0.0323	4.65	-5.98	20.63	116	-0.124	-0.071	-0.205	-0.132	-0.104	-0.127	-0.033	-0.040	0.177
170	A26	0.0238	0.0140	5.43	-4.2	14.96	156.6	-0.030	-0.028	-0.036	-0.047	-0.045	-0.027	-0.012	-0.007	0.186
167	A29	0.0131	0.0150	5.75	17.5	15.15	165.8	-0.028	-0.019	-0.002	0.004	0.033	0.038	-0.002	-0.002	0.145
166	A86	0.0147	0.0093	6.55	123	17.51	278.9	0.094	0.101	0.109	0.110	0.100	0.063	0.104	0.066	0.117
287	25	0.0223	0.0335	5.05	-8.73	16.04	116.9	-0.056	-0.055	-0.075	-0.077	-0.068	-0.103	-0.032	-0.048	0.179
289	27	0.0216	0.0275	6.43	64.8	12.27	179.0	0.019	0.029	0.035	0.035	0.030	0.038	0.044	0.055	0.175
290	28	0.0124	0.0130	5.71	34.8	20.94	203.0	0.004	0.033	-0.008	-0.007	0.012	0.012	-0.014	-0.015	0.181
342	82	0.0291	0.0085	6.68	166	26.68	355.8	0.208	0.181	0.165	0.125	0.182	0.053	0.122	0.036	0.120
354	94	0.0162	0.0240	7.14	359	24.50	528.9	0.322	0.225	0.213	0.226	0.179	0.265	0.186	0.275	0.141
West of Fort McMurray Sub-Region																
165	A42	0.0639	0.0245	6.93	326.	45.92	654.8	0.388	0.373	0.553	0.706	0.455	0.175	0.359	0.138	0.121
171	A47	0.0115	0.0044	6.37	131	20.32	333.4	0.217	0.167	0.152	0.253	0.207	0.079	0.168	0.064	0.120
172	A59	0.1781	0.0339	5.24	41.2	33.36	267.6	0.038	0.001	0.002	-0.023	-0.075	-0.014	-0.061	-0.012	0.076
223	P94	0.0019	0.0003	7.32	779	48.27	1435.2	1.120	1.031	1.054	1.399	1.004	0.153	0.829	0.126	0.258
225	P96	0.0034	0.0027	7.32	624	32.41	931.1	0.745	0.595	0.666	0.825	0.669	0.539	0.506	0.408	0.238
226	P97	0.0057	0.0056	6.85	329	30.82	663.4	0.328	0.346	0.266	1.377	0.238	0.235	0.277	0.273	0.353
227	P98	0.0070	0.0025	7.26	637	32.25	907.3	0.969	0.956	0.917	0.462	1.042	0.378	0.857	0.311	0.307
267	1	0.1182	0.0138	7.71	748	23.37	943.5	1.055	1.024	0.994	1.091	0.732	0.086	0.630	0.074	0.214
Northeast of Fort McMurray Sub-Region																
452	L4	0.0920	0.0675	5.78	74.0	24.69	284.6	0.070	0.070	0.078	0.143	0.073	0.053	0.095	0.070	0.222
470	L7	0.1010	0.0376	6.43	160	28.76	418.8	0.170	0.190	0.141	0.307	0.707	0.263	0.357	0.133	0.646
471	L8	0.0450	0.0257	6.81	332	20.88	575.8	0.528	0.622	0.527	0.659	0.340	0.194	0.527	0.301	0.607
400	L39	0.0501	0.0855	6.76	171	15.26	386.0	0.157	0.157	0.144	0.073	0.316	0.539	0.251	0.428	0.085
268	E15	0.0472	0.0472	7.01	365	39.95	636.7	0.520	0.465	0.400	0.505	0.092	0.054	0.421	0.245	0.206
182	P23	0.0296	0.0254	7.68	748	17.61	975.5	0.294	1.084	2.017	2.008	0.443	0.379	1.333	1.143	0.250
185	P27	0.0172	0.0175	5.34	65.3	30.86	283.6	0.035	0.017	-0.095	0.233	-0.030	-0.030	0.035	0.035	0.220
209	P7	0.0072	0.0095	6.11	129	25.12	313.8	0.141	0.163	0.112	0.089	0.109	0.145	0.143	0.189	0.195
270	4	0.0411	0.0371	8.23	1,400	32.31	1743.1	1.382	1.318	1.408	1.705	1.037	0.936	0.904	0.816	0.181
271	6	0.0485	0.0388	8.53	1,350	27.76	1619.0	1.293	1.449	1.931	1.369	1.009	0.807	0.856	0.685	0.133
418	Kearl	0.1690	0.2329	7.96	1,564	22.93	1543.8	NA	1.280	1.290	1.664	1.192	1.643	1.293	1.781	0.367

Shaded values represent critical loads less than the Potential Acid Input obtained from the 2006 Deer Creek Joslyn North Mine EIA, Deer Creek Energy (2006).

¹ Estimate of PAI was based on SO₂ deposition alone except for lakes receiving Nitrogen deposition above a threshold value of 9 kg/ha/y.

Hydro – runoff estimated using traditional hydrometric methods; Isotopic – runoff estimated using analysis of heavy isotopes of oxygen and hydrogen.

Table 6.5-3 (Cont'd.)

ID No.	Original RAMP Desig.	Runoff (Hydro) m³/s	Runoff (Isotopic) m³/s	Mean pH	Mean Gran Alk. (µeq/L)	Mean DOC (mg/L)	Mean SBC (mg/L)	Critical Load of Acidity (keq H ⁺ /ha/y)								PAI ¹
								2002	2003	2004	2005	2006 (Hydro)	2006 (Isotopic)	2007 (Hydro)	2007 (Isotopic)	
Birch Mountains Sub-Region																
436	L18	0.3250	0.1485	7.18	394	8.48	639	0.235	0.239	0.226	0.313	0.225	0.103	0.231	0.105	0.122
442	L23	0.0430	0.1848	6.75	145	13.81	271.9	0.087	0.074	0.065	0.074	0.059	0.252	0.074	0.317	0.094
444	L25	0.1765	0.6413	6.81	165	8.85	305.0	0.088	0.097	0.099	0.134	0.109	0.396	0.111	0.403	0.096
447	L28	0.0448	0.1130	5.21	25.3	27.80	224.2	-0.016	-0.025	0.002	-0.025	-0.039	-0.099	0.001	0.004	0.056
448	L29	0.0330	0.0461	4.21	-5.43	15.84	80.3	-0.127	-0.090	-0.073	-0.111	-0.117	-0.163	-0.025	-0.035	0.086
454	L46	0.1690	0.1026	6.78	226	23.62	635.9	0.394	0.375	0.365	0.374	0.303	0.184	0.482	0.292	0.097
455	L47	0.1016	0.1422	6.79	230	22.56	740.3	0.282	0.241	0.958	0.324	0.272	0.381	0.286	0.400	0.074
457	L49	0.0666	0.1164	6.51	141	22.47	578.2	0.301	0.260	0.283	0.234	0.210	0.367	0.205	0.358	0.085
464	L60	0.1630	0.0730	7.06	286	19.88	607.5	0.408	0.420	0.501	0.422	0.319	0.143	0.356	0.159	0.078
175	P13	0.0120	0.0028	7.92	911	46.64	1398.2	1.198	1.235	2.149	1.449	1.099	0.254	0.818	0.189	0.145
199	P49	0.0044	0.0013	6.61	157	18.69	310.3	0.245	0.215	0.237	0.247	0.305	0.092	0.191	0.058	0.172
Canadian Shield Sub-Region ²																
473	A301	0.1756	0.0581	7.26	402	15.22	593.1	0.210	0.194	0.189	0.264	0.197	0.065	NA	NA	0.014
118	L107	0.0092	0.0806	7.26	436	10.68	627.8	0.118	0.116	0.114	0.168	0.109	0.956	0.101	0.882	0.007
84	L109	0.3537	0.0974	7.04	355	18.87	565.8	0.409	0.394	0.341	0.496	0.386	0.106	0.294	0.081	0.014
88	O-10	0.0118	0.0094	6.85	212	22.88	435.2	0.178	0.189	0.138	NA	0.166	0.133	NA	NA	0.014
90	R1	0.0788	0.0974	7.04	298	17.70	465.0	0.318	0.311	0.279	0.408	0.311	0.384	0.418	0.517	0.014
Caribou Mountains Sub-Region ²																
146	E52	0.0439	0.1510	7.05	374	23.39	629.7	0.377	0.365	0.350	0.531	0.349	1.201	0.347	1.193	0.027
152	E59	0.0124	0.3079	6.79	174	13.06	328.5	0.023	0.025	0.026	0.031	0.021	0.531	0.025	0.622	0.027
89	E68	0.1576	0.1072	6.84	235	22.74	484.6	0.258	0.274	0.223	0.395	0.262	0.179	0.216	0.147	0.027
91	O-1/E55	0.0044	0.0122	6.31	98.9	21.59	403.1	0.020	0.029	0.038	0.536	0.064	0.178	0.082	0.228	0.027
97	O-2 E67	0.1109	0.2180	6.60	166	22.85	365.1	0.201	0.187	0.149	0.081	0.134	0.238	0.104	0.205	0.027

Shaded values represent critical loads exceeded by the Potential Acid Input obtained from the 2006 Deer Creek Joslyn North Mine EIA, Deer Creek Energy (2006).

¹ Estimate of PAI was based on SO₂ deposition alone except for lakes receiving Nitrogen deposition above a threshold value of 9 kg/ha/y.

² PAI obtained from OPTI 2002 EIA representing background values (no industry).

Hydro – runoff estimated using traditional hydrometric methods; Isotopic – runoff estimated using analysis of heavy isotopes of oxygen and hydrogen.

Table 6.5-4 Summary of critical loads in ASL lakes.

	2002	2003	2004	2005	2006 (Hydro)	2006 (Isotope)	2007 (Hydro)	2007 (Isotope)
No. of Lakes	49	50	50	49	50	50	48	48
Minimum CL	-0.127	-0.090	-0.205	-0.132	-0.117	-0.163	-0.070	-0.048
Maximum CL	1.382	1.449	2.149	2.008	1.192	1.643	1.333	1.781
Average CL	0.306	0.335	0.387	0.435	0.291	0.252	0.300	0.285
Median CL	0.210	0.192	0.177	0.253	0.202	0.164	0.198	0.174
No. of lakes in which the PAI is greater than the CL	21	19	19	16	19	24	19	23
Proportion of lakes in which the PAI is greater than the CL (%)	42.9	38.0	38.0	32.7	38.0	48.0	39.6	47.9

Table 6.5-5 Chemical characteristics of lakes having the modelled PAI greater than the critical load in 2007.

Lake	Original Name	pH	Gran Alkalinity (µeq/L)	Conductivity (µS/cm)	DOC (mg/L)	Lake Area (km ²)
168	A21	5.01	-19.6	12.14	17.4	1.38
169	A24	4.73	-13.2	11.83	15.8	1.45
170	A26	5.56	10	11.22	13.5	2.78
167	A29	5.89	31.8	11.48	13.7	1.05
171	A47	6.57	71.6	31.90	22.7	0.8
172	A59	4.81	28	22.00	32.4	108
166	A86	6.55	166.6	29.80	18.6	2.17
267	1	7.33	639.8	68.80	21.7	2
287	25	5.16	5.8	10.49	11.0	2.18
289	27	6.35	78.4	16.25	11.4	1.83
290	28	5.86	49	14.96	19.4	0.544
342	82	6.60	164.6	27.40	25.9	2
452	L4	5.62	71.2	19.90	21.6	0.610
470	L7	6.48	164.6	27.10	24.7	0.330
471	L8	6.94	339.8	41.10	16.9	0.6
436	L18	7.05	431	66.10	7.7	44
442	L23	6.60	180.6	24.80	13.2	3.44
444	L25	6.67	187.6	28.90	7.2	17
447	L28	5.43	46.8	20.30	25.7	1.30
448	L29/Clay	4.28	0	16.44	13.1	0.650
400	L39	6.57	164.6	26.60	12.3	1.2
268	E15	6.72	266.2	35.00	32.0	1.9
152	E59	6.63	198.6	30.10	12.0	9.53
91	O-1	6.27	114.6	25.00	22.0	0.800
97	O-2	6.72	196.6	29.20	18.8	3.10
209	P7	6.34	139.4	25.00	23.6	0.1
185	P27	5.06	69	27.00	34.9	3.94
199	P49	6.46	140.4	25.00	16.2	0.1
223	P94	7.22	705.2	104.00	45.9	

These are lakes with PAI greater than critical load, regardless of the method of calculation (hydrometric or isotopic).

6.5.4 Trends in ASL Measurement Endpoints in Individual Lakes

Potential trends in the ASL measurement endpoints in 31 individual lakes were examined using the Mann-Kendall non-parametric test (Gilbert 1987). In addition, key measurement endpoints (pH, Gran alkalinity, sulphate, sum of base cations and nitrates) were charted in Shewhart control plots for 10 lakes considered most at risk to acidification. These control plots can assist in detecting trends before significant change has occurred.

6.5.4.1 Mann-Kendall Trend Analysis

As the Mann-Kendall test deals only with differences between successive points (rather than their magnitude) small, insignificant differences can result in conclusions of a trend being present, when none really exists (false positives). To reduce the number of false positive effects, estimates of analytical error obtained from the laboratory (Appendix H) were incorporated in the trend analysis by eliminating, as statistically insignificant, all difference-pairs in the Mann-Kendall calculations that were less the percent error at the relevant concentration.

The results of the Mann Kendall trend analysis were very similar to those results observed in 2006 (Table 6.5-6). Almost the same significant trends were evident in both years. As in 2006, these trends were often inconsistent with any conceivable acidification scenario:

- The only significant change in pH was positive (1 lake) rather than negative (if acidification were evident);
- All significant changes in sulphate, the primary acidifying agent, were decreases;
- Gran alkalinity decreased significantly in three lakes and increased in five; none of the significant decreases in Gran alkalinity was associated with significant increases in sulphate;
- Total alkalinity decreased significantly in two lakes and increased in seven lakes; Base cations decreased significantly in five lakes and increased in three lakes. Acidification should initially result in an increase in base cations as these ions are stripped from soils in catchments receiving acid deposition. None of the increases in base cations were associated with significant increases in sulphate and all three were associated with significant increases (rather than decreases) in Gran or total alkalinity; and
- There were no significant changes in dissolved organic carbon. A decrease in DOC is expected in acidifying lakes (Schindler et al. 1992).

The conclusions of the Mann Kendall trend analysis are essentially those drawn in 2006. There is no evidence from the water chemistry of the RAMP lakes from 1999 to the present to suggest that acidification is occurring to individual lakes.

Table 6.5-6 Results of Mann-Kendall trend analyses on ASL measurement endpoints.

Lake ID	Original RAMP Designation	pH (units)	Total Alkalinity µeq/L	Gran Alkalinity µeq/L	Calcium (mg/L)	Sulphate (mg/L)	Total Dissolved Nitrogen (µg/L)	Nitrates and Nitrites (µg/L)	Dissolved Organic Carbon	Sum Base Cations (µeq/L)	Potential Acid Input (keq/ha/y)
168	A21	6	0	-10	-25	-22	-11	-12	-15	-22	0.186
169	A24	7	3	-12	6	-9	-18	-9	5	2	0.177
170	A26	-1	-6	2	-9	-16	-12	8	-4	-7	0.186
167	A29	5	5	13	4	-2	-1	1	2	17	0.145
166	A86	3	17	9	19	7	13	11	16	22	0.117
165	A42	14	20	11	18	-4	-1	6	5	7	0.120
171	A47	8	20	2	18	6	-4	-4	12	18	0.076
172	A59	-15	-15	-16	-3	-12	-11	-7	-6	-16	0.222
452	L4	16	20	17	8	2	1	-9	2	4	0.646
470	L7	2	-7	8	-11	0	2	4	-3	9	0.607
471	L8	-1	-20	-10	-9	-1	-10	-1	-13	-16	0.085
400	L39	-3	-13	-7	-28	0	-1	3	7	-8	0.206
268	E15 (L15b)	-10	-17	-21	-23	14	-13	-2	-1	-17	0.122
436	L18	16	20	17	8	2	1	-9	2	4	0.094
442	L23	0	7	5	-13	-21	0	5	7	-18	0.096
444	L25	6	18	21	1	-11	1	-6	1	4	0.056
447	L28	12	16	10	-6	-3	-5	-18	-3	-6	0.086
448	L29	4	0	2	-8	-4	-2	-1	-8	-4	0.097
454	L46	-4	-18	-6	-14	-12	-7	-7	5	-15	0.074
455	L47	6	-11	3	-5	-4	-13	7	9	-9	0.085
457	L49	-4	-5	-12	-14	-16	6	6	10	-25	0.078
464	L60	6	16	10	1	-25	-1	-1	14	-8	0.007
118	L107	5	2	6	-15	3	-8	-5	13	-12	0.014
84	L109	3	-13	-17	-16	-2	-1	-11	2	-17	0.014
88	O-10	7	6	-2	1	-2	-5	-1	-1	-19	0.014
90	R1	8	15	3	4	9	-8	-4	-5	9	0.027
146	E52	0	8	15	4	3	-6	2	7	6	0.027
152	E59	1	16	16	4	-16	10	7	8	9	0.027
89	E68	-13	-8	0	-10	-18	0	-11	2	-10	0.027
91	O-1/E55	-6	-8	-16	-19	5	7	-12	-8	-23	0.027
97	O-2 E67	30	31	22	23	-15	-6	-20	14	24	0.027

Numbers represent the S statistic used in the analysis. Negative values represent overall decreases in a variable and positive values represent increases. Shaded values are statistically significant.

6.5.4.2 Control Charting of Endpoint Parameters

The choice of the ten lakes selected for control charting was based on an acidification risk factor calculated from the ratio of PAI to the value of the critical load (Table 6.5-7). The greater this ratio in a lake, the greater is the risk for acidification. The 10 lakes with the highest ratios are indicated in Table 6.5-7. All but one of these lakes is found in the Stony Mountains, Birch Mountains and Muskeg River uplands. If acidification is occurring, it should be evident first in these lakes.

The control plots follow standard analytical control chart theory where control limits representing two and three standard deviations are plotted on the graphs with the points and the mean value (Gilbert 1987). The lines at two standard deviations represent warning limits while the lines at three standard deviations identify distinct outliers. A trend in an endpoint parameter is often assumed if three consecutive points fall on the same side outside of the two standard deviation warning limits or one point outside of the three standard deviation control limits.

The plots for pH, Gran alkalinity, sulphates, nitrates and DOC for the ten lakes (Figure 6.5-2 to Figure 6.5-7) indicate that only isolated exceedances of the two standard deviation warning limits were observed (e.g., pH and base cations in Lake 170, nitrates in Lakes 447, 170 and 172 and sulphate in Lakes 447 and 167). The parameters in each case appear to return to more normal values afterwards. Gran alkalinity shows the same anomaly identified in 2005 in Section 6.5.2, a sudden decrease in most of the lakes attributed to the high rates of runoff and precipitation that year. The year 1999 is also identified as unusual with high levels of base cations, pH and DOC in various lakes especially Lake 170/A26 in the Stony Mountains. Nitrates were highly variable between both years and lakes. In general, no distinct trends are evident to suggest that change is occurring. Control plots for Gran alkalinity for each of the fifty lakes are shown in Figure 6.5-8. This figure permits tracking of potential changes in this key endpoint parameter by sub-region.

Control plots for the ASL endpoint parameters will be updated yearly over the RAMP program and their ability to detect change will improve as more data are collected and better estimates of natural variability emerge.

6.5.5 Summary of Conditions

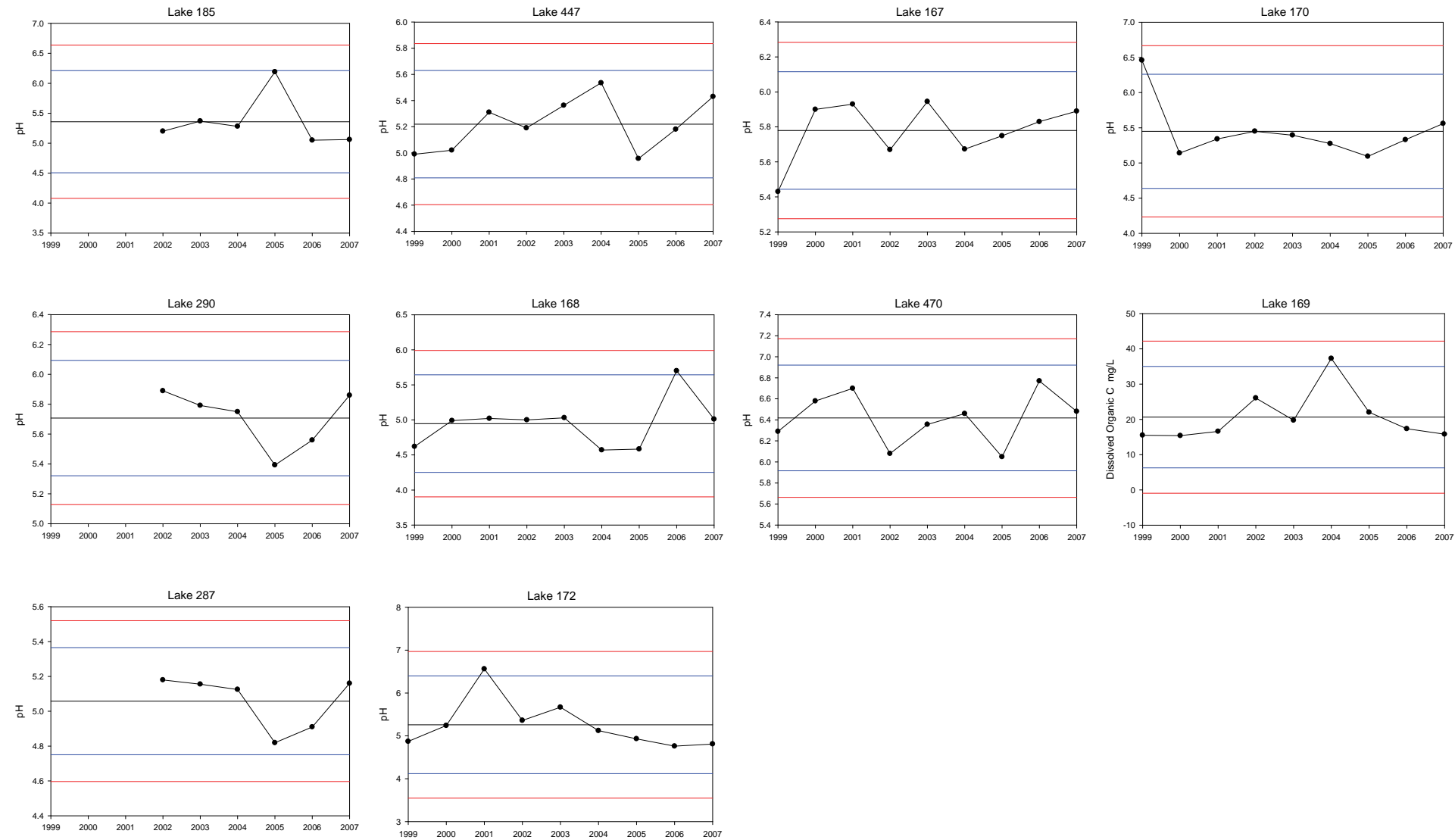
These results of the analysis of 2007 RAMP ASL lake data in conjunction with historical RAMP ASL lake dataset suggest that there has been no significant change in the overall chemistry of the 50 RAMP ASL lakes in 2007 compared to previous years. Based on the results of the trend analysis and the control plotting, there is no evidence to conclude that there have been any significant changes in lake chemistry in the RAMP ASL lakes over the monitoring period.

Table 6.5-7 Calculation of the Acidification Risk Factor for Individual RAMP ASL Lakes.

Lake No.	Original Designation	Critical Load (keq/ha/y)	PAI (keq H ⁺ /ha/y)	Acidification Risk Factor PAI/CL
118	L107/Weekes	0.882	0.007	0.008
146	E52/Fleming	1.193	0.027	0.023
90	R1	0.517	0.014	0.027
152	E59/Rocky I.	0.622	0.027	0.043
91	O-1/E55	0.228	0.027	0.118
97	O-2/E67	0.205	0.027	0.132
84	L109/Fletcher	0.081	0.014	0.173
89	E68/Whitesand	0.147	0.027	0.184
455	L47	0.400	0.074	0.185
271	6	0.685	0.133	0.194
400	L39	0.428	0.085	0.199
418	Kearl	1.781	0.367	0.206
182	P23	1.143	0.25	0.219
270	4	0.816	0.181	0.222
457	L49	0.358	0.085	0.238
444	L25/Legend	0.403	0.096	0.238
442	L23/Otasan	0.317	0.094	0.297
454	L46/Bayard	0.292	0.097	0.332
464	L60	0.159	0.078	0.490
354	94	0.275	0.141	0.512
225	P96	0.408	0.238	0.584
175	P13	0.189	0.145	0.765
268	E15	0.245	0.206	0.839
165	A42	0.138	0.121	0.879
227	P98	0.311	0.307	0.987
209	P7	0.189	0.195	1.032
436	L18/Namur	0.105	0.122	1.158
226	P97	0.273	0.353	1.293
166	A86	0.066	0.117	1.778
171	A47	0.064	0.12	1.862
471	L8	0.301	0.607	2.015
223	P94	0.126	0.258	2.041
448	L29/Clayton	-0.035	0.086	2.477
267	1	0.074	0.214	2.904
199	P49	0.058	0.172	2.969
289	27	0.055	0.175	3.162
452	L4 (A-170)	0.070	0.222	3.175
342	82	0.036	0.12	3.363
287	25	-0.048	0.179	3.733
169	A24	-0.040	0.177	4.392
470	L7	0.133	0.646	4.871
168	A21	0.036	0.186	5.212
185	P27	0.035	0.22	6.200
172	A59	-0.012	0.076	6.509
290	28	-0.015	0.181	11.936
447	L28	0.004	0.056	15.188
170	A26	-0.007	0.186	26.659
167	A29	-0.002	0.145	63.774

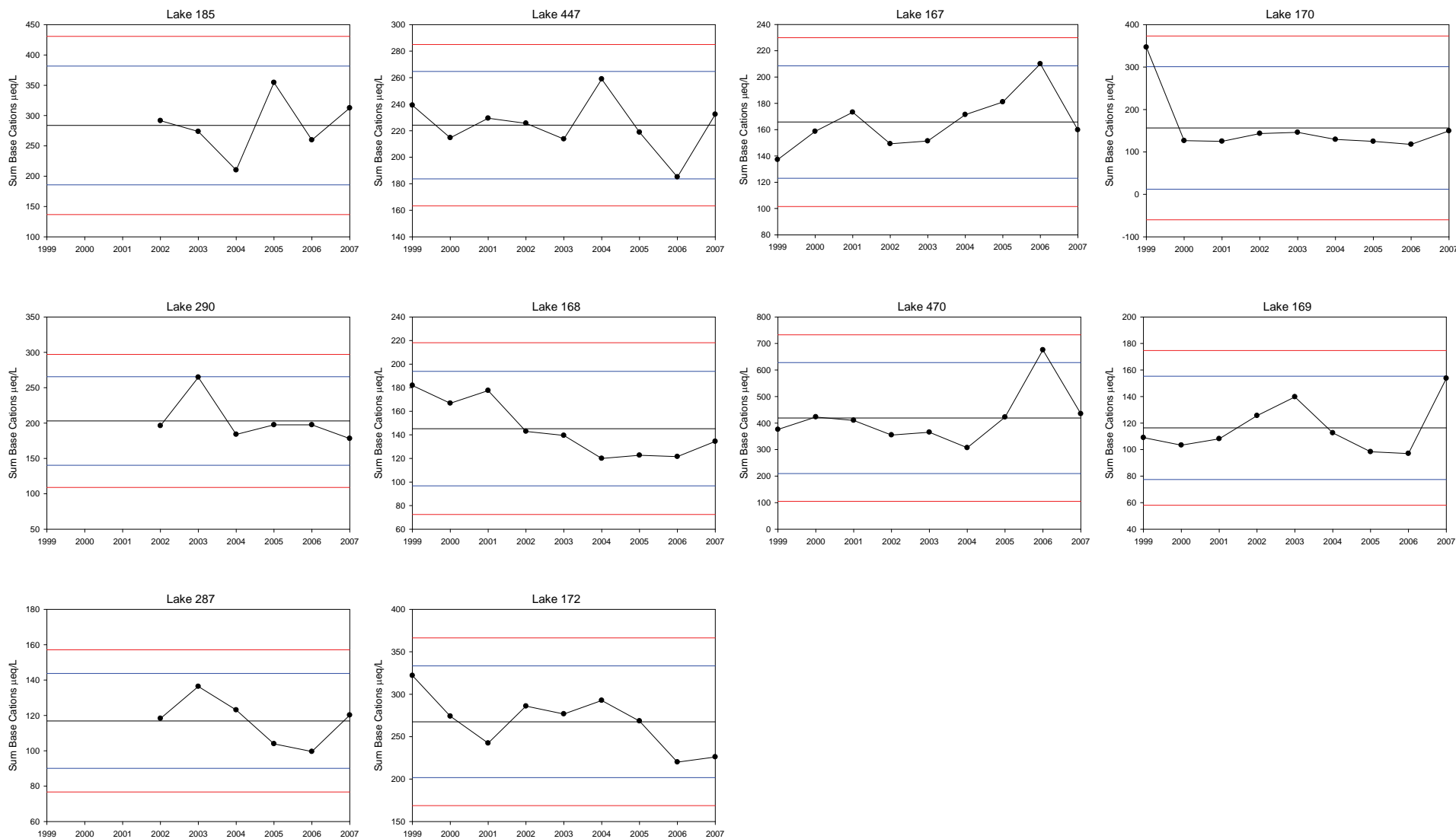
Shaded lakes represent those lakes most at risk to acidification

Figure 6.5-2 Shewhart control charts of pH in the ten RAMP ASL lakes most at risk to acidification.



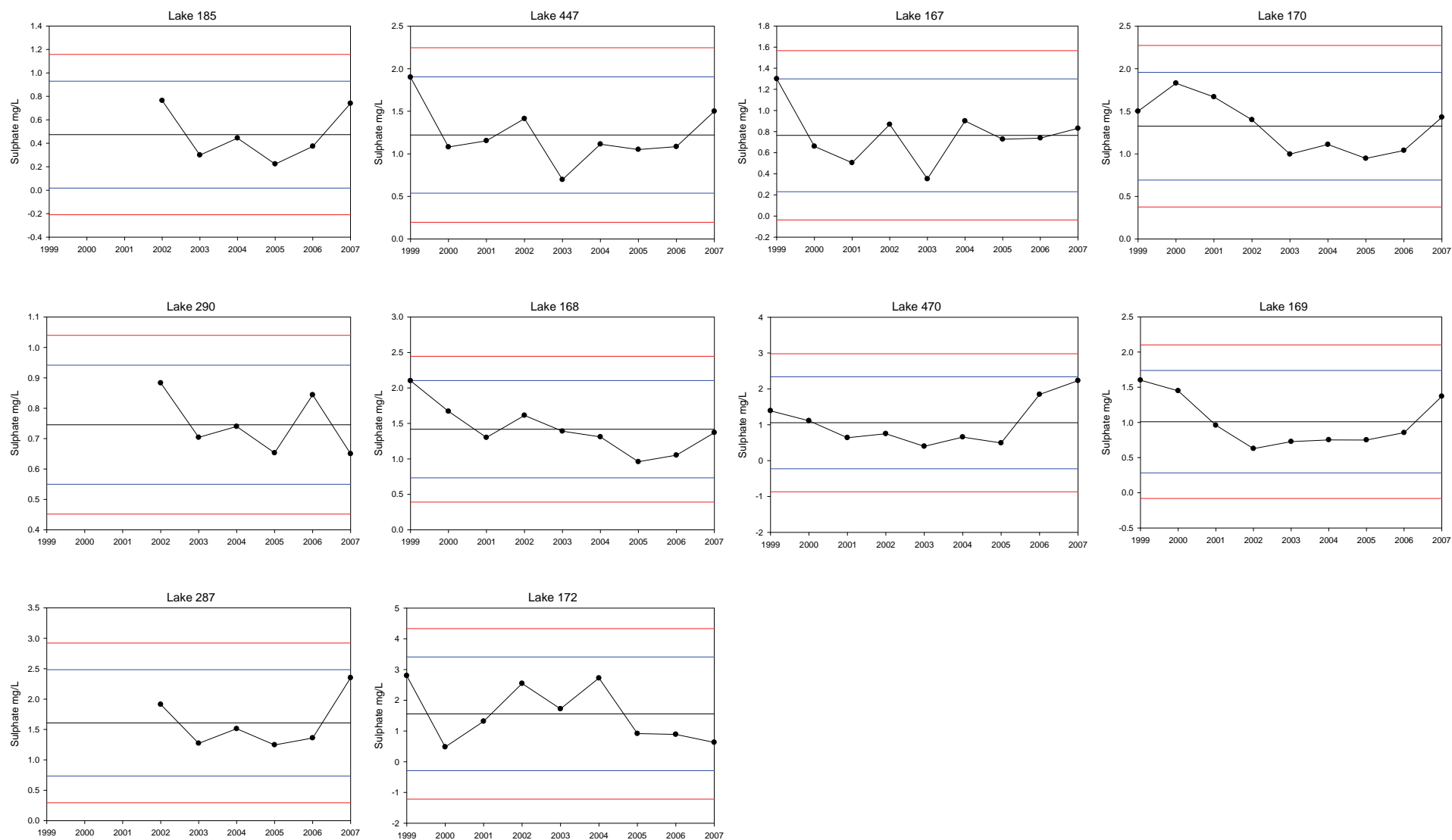
Blue lines: ± 2 standard deviations; Red lines: ± 3 standard deviations; black line - mean

Figure 6.5-3 Shewhart control charts of the sum of base cations in the ten RAMP ASL lakes most at risk to acidification.



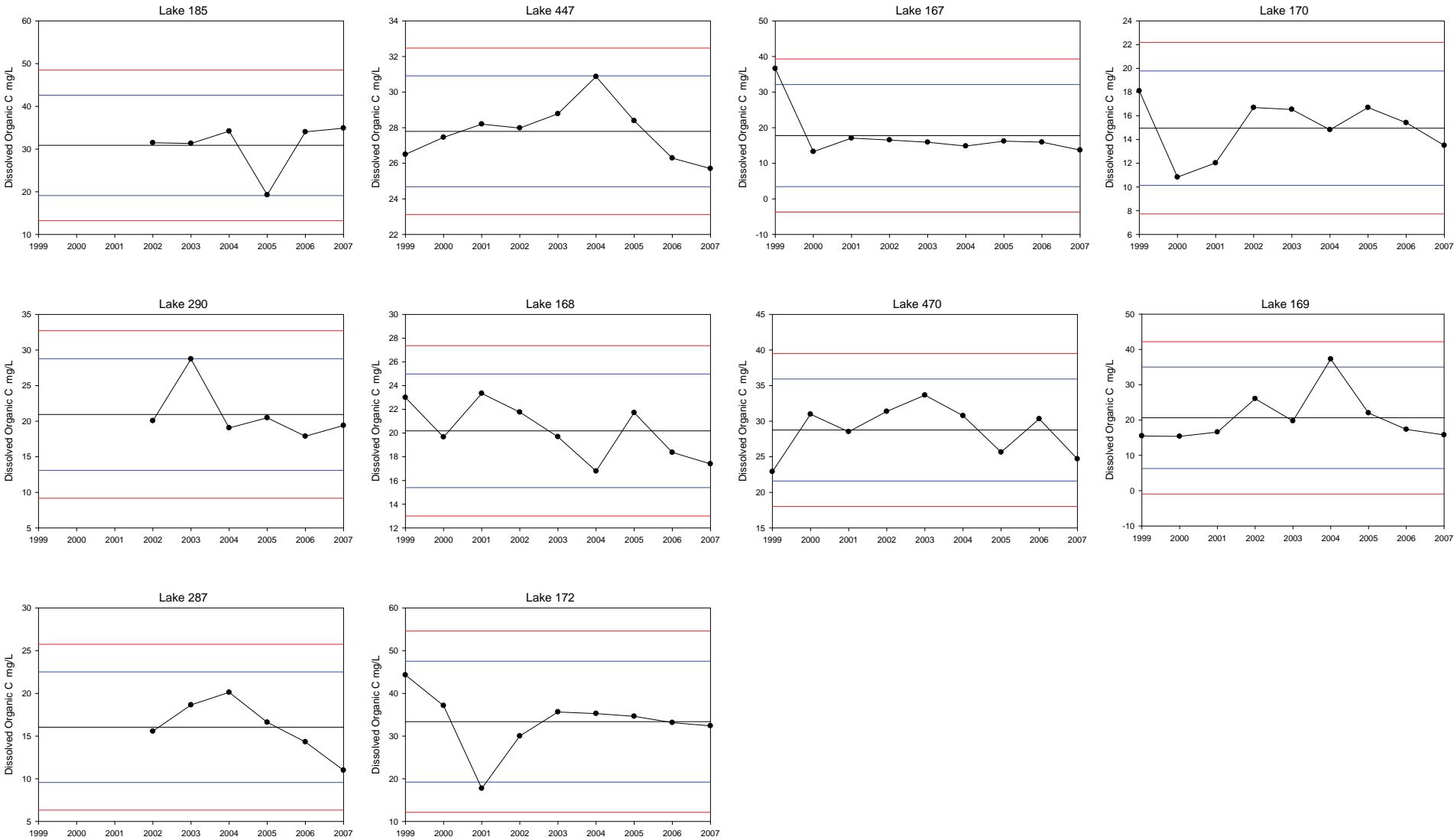
Blue lines: ± 2 standard deviations; Red lines: ± 3 standard deviations; black line - mean

Figure 6.5-4 Shewhart control charts of sulphate in the ten RAMP ASL lakes most at risk to acidification.



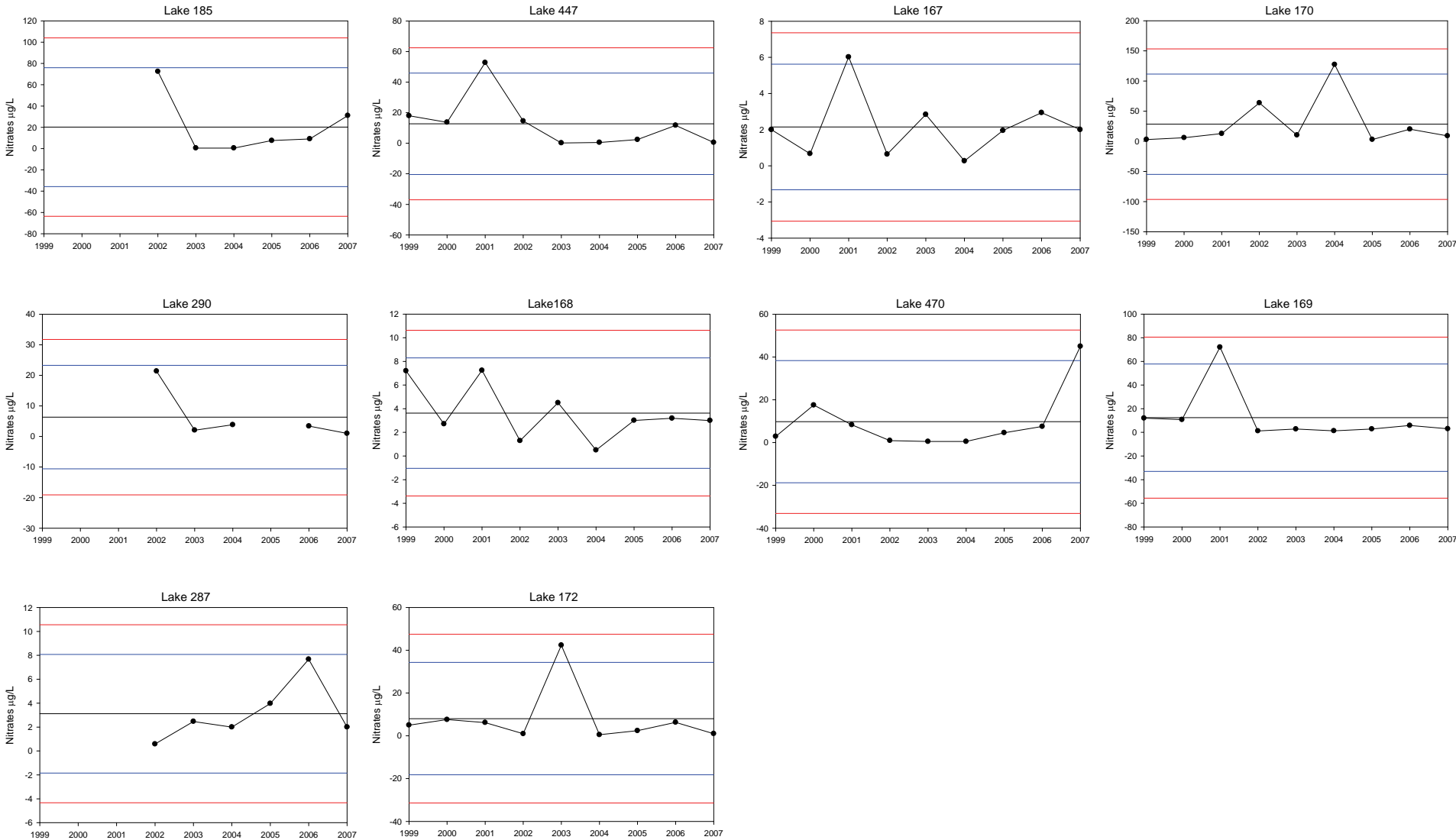
Blue lines: ± 2 standard deviations; Red lines: ± 3 standard deviations; black line - mean

Figure 6.5-5 Shewhart control charts of dissolved organic carbon in the ten RAMP ASL lakes most at risk to acidification.



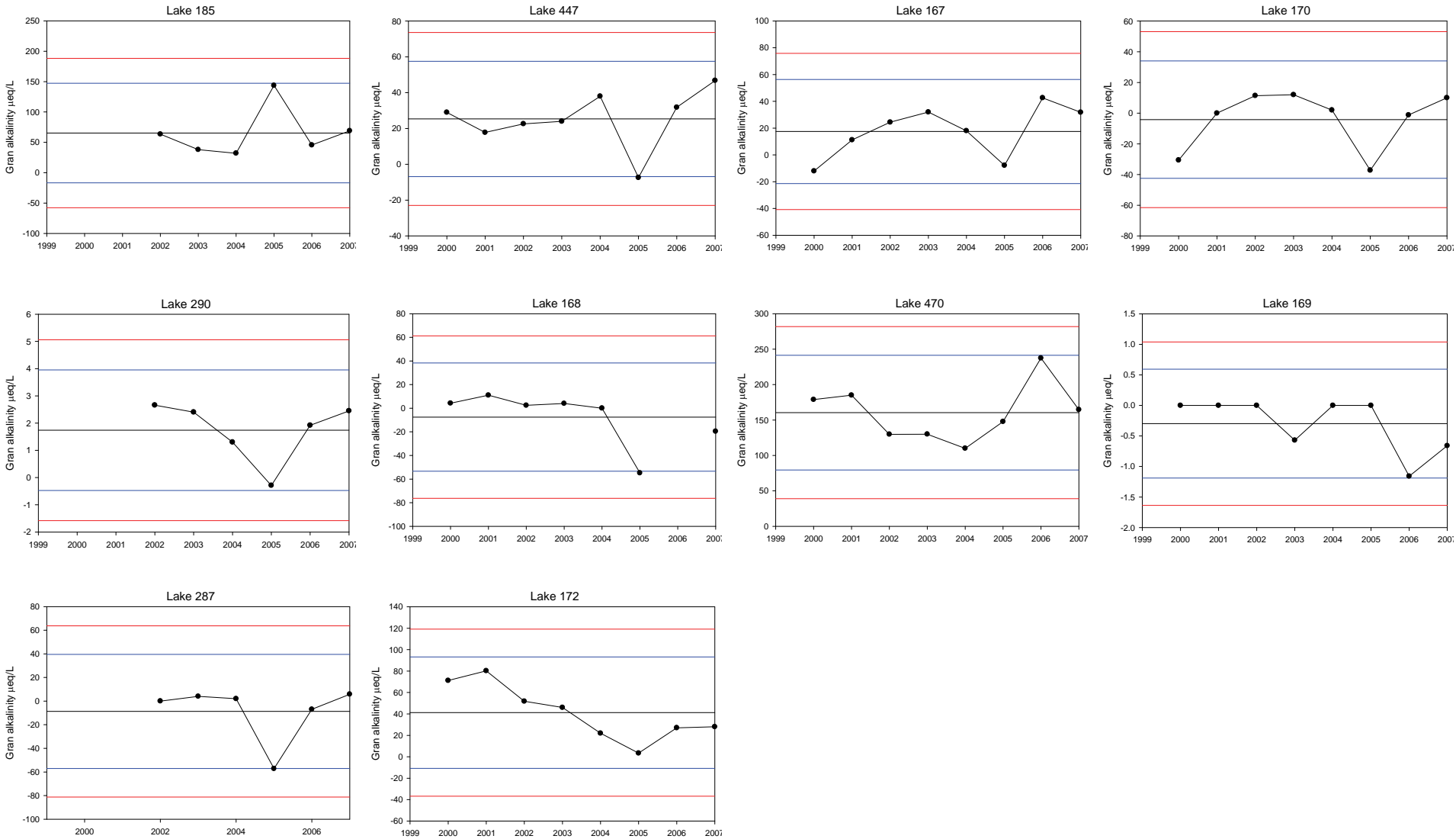
Blue lines: ± 2 standard deviations; Red lines: ± 3 standard deviations; black line - mean

Figure 6.5-6 Shewhart control charts of nitrates in the ten RAMP ASL lakes most at risk to acidification.



Blue lines: ± 2 standard deviations; Red lines: ± 3 standard deviations; black line - mean

Figure 6.5-7 Shewhart control charts of Gran alkalinity in the ten RAMP ASL lakes most at risk to acidification.



Blue lines: ± 2 standard deviations; Red lines: ± 3 standard deviations; black line - mean

Figure 6.5-8 Control Plots of Gran alkalinity for each of the 50 ASL lakes.

