6.0 REGIONAL SYNTHESIS

This part of the RAMP 2008 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 2008 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 2008 are summarized in Table 6.1-1. Mean open-water season discharge, mean winter discharge, annual maximum daily discharge, and open-water season minimum daily discharge are all 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 sand development activities. This is largely because of water withdrawals and estimated decreased natural runoff from oil sands development areas. The percent change varies from 0.5% to 2.2% 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. The estimated changes in hydrologic measurement endpoints for 2008 are assessed as Negligible-Low.

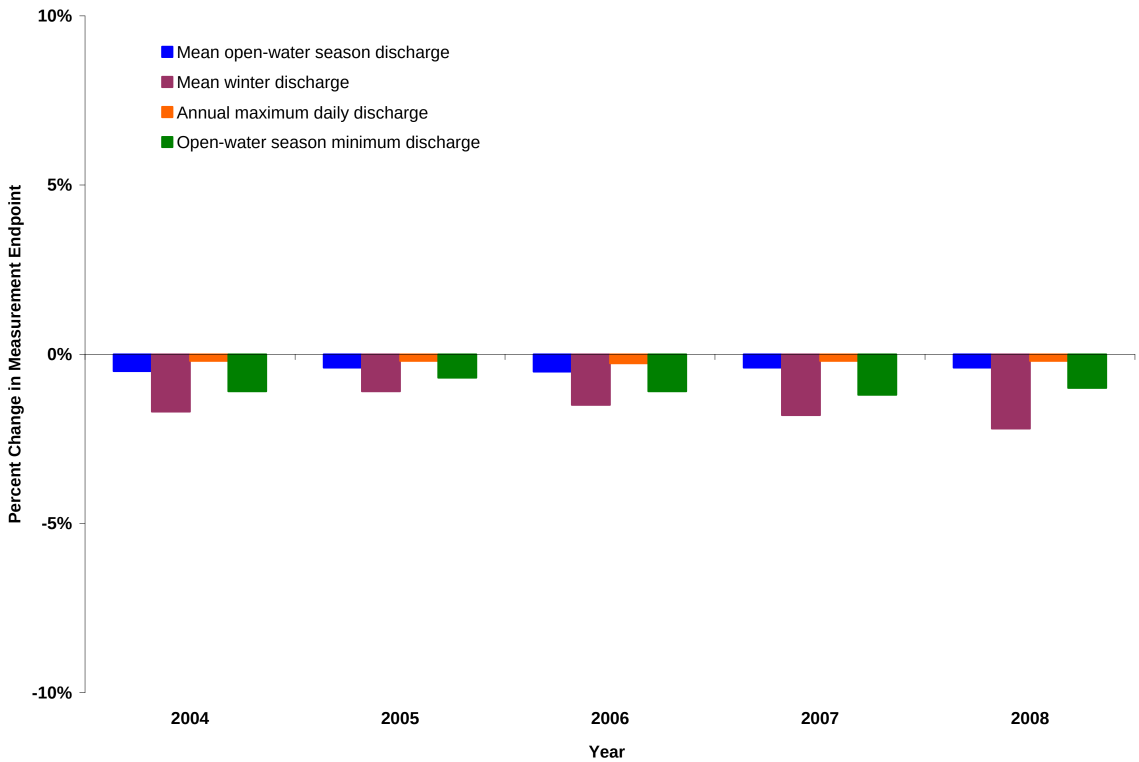
Table 6.1-1 Summary of hydrologic conditions of the Athabasca River in 2008 with respect to oil sands developments.

Measurement Endpoint	*Baseline* Value (m^3/s)	Operational, *Test* Value (m^3/s)	Percent Change	Assessment
Mean open-water (1 May to 31 October) season discharge	906	899	-0.8%	Negligible-Low
Mean winter (1 November to 31 March) discharge	190	186	-2.2%	Negligible-Low
Annual maximum daily discharge	1,830	1,820	-0.5%	Negligible-Low
Open-water season minimum daily discharge	315	311	-1.1%	Negligible-Low

Note: Focal projects plus all other oil sands projects that were active as of 2008 are included in this analysis.

Trends in the values of these measurement endpoints are provided in Figure 6.1-1. For the five years, 2004 to 2008, the hydrologic changes as a result of focal projects and other oil sands developments on the Athabasca River are assessed as Negligible-Low. The past four years (2005 to 2008) indicate a slight trend of decreasing river discharges in winter but no definite trend in other measurement endpoints.

Figure 6.1-1 Trends in assessed hydrologic changes in Athabasca River as a result of focal projects and other oil sands developments.



6.1.2 Regional Assessment of Hydrologic Conditions at the RAMP FSA Level

The assessed change in each watershed in 2008 for each measurement endpoint is summarized in Table 6.1-2.

Most of the hydrological assessments are rated as Negligible-Low with the exception of the Muskeg, Tar, Poplar, Mills Creek and Fort Creek watersheds in which hydrologic changes are assessed as ranging from Moderate 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 2008 on hydrologic conditions in these watersheds, including:

- 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

- Reduced flows in Tar River due to the filling of Canadian Natural compensation lake in spring and redirection of flow into a tailings pond for the remainder of the year.

Table 6.1-2 Summary of 2008 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-Low	Negligible-Low	Negligible-Low	Negligible-Low
Muskeg	Negligible-Low	Negligible-Low	Moderate (-)	Negligible-Low
Steepbank	Negligible-Low	*not measured*	Negligible-Low	Negligible-Low
Tar	High (-)	*not measured*	High (-)	High (-)
MacKay	Negligible-Low	Negligible-Low	Negligible-Low	Negligible-Low
Calumet	Negligible-Low	*not measured*	Negligible-Low	Negligible-Low
Ells	Negligible-Low	Negligible-Low	Negligible-Low	Negligible-Low
Firebag	Negligible-Low	Negligible-Low	Negligible-Low	Negligible-Low
Christina	No hydrometric monitoring station at the mouth of the Christina River			
Hangingstone	Negligible-Low	*not measured*	Negligible-Low	Negligible-Low
Poplar Creek	High (+)	*not measured*	Negligible-Low	High (+)
Mills Creek	Moderate (-)	Moderate (-)	Moderate (-)	Moderate (-)
Fort Creek	Moderate (+)	*not measured*	Moderate (+)	Moderate (+)

Assessments based on comparisons of estimated incremental change in hydrologic measurement endpoints with criteria used in Section 5.0: Negligible-Low: ± 5%; Moderate: ±15%; High: > ± 15%.

"not measured" means hydrologic information was not obtained for times of year for which the measurement endpoint is applicable.

Direction indicators (+ or -) indicate a detected increase or decrease in discharge in observed, *test*, conditions as compared to estimated discharge in estimated, *baseline* conditions. Direction indicators are shown only for estimated impacts of a minimum of ± 5% (i.e., Moderate or High).

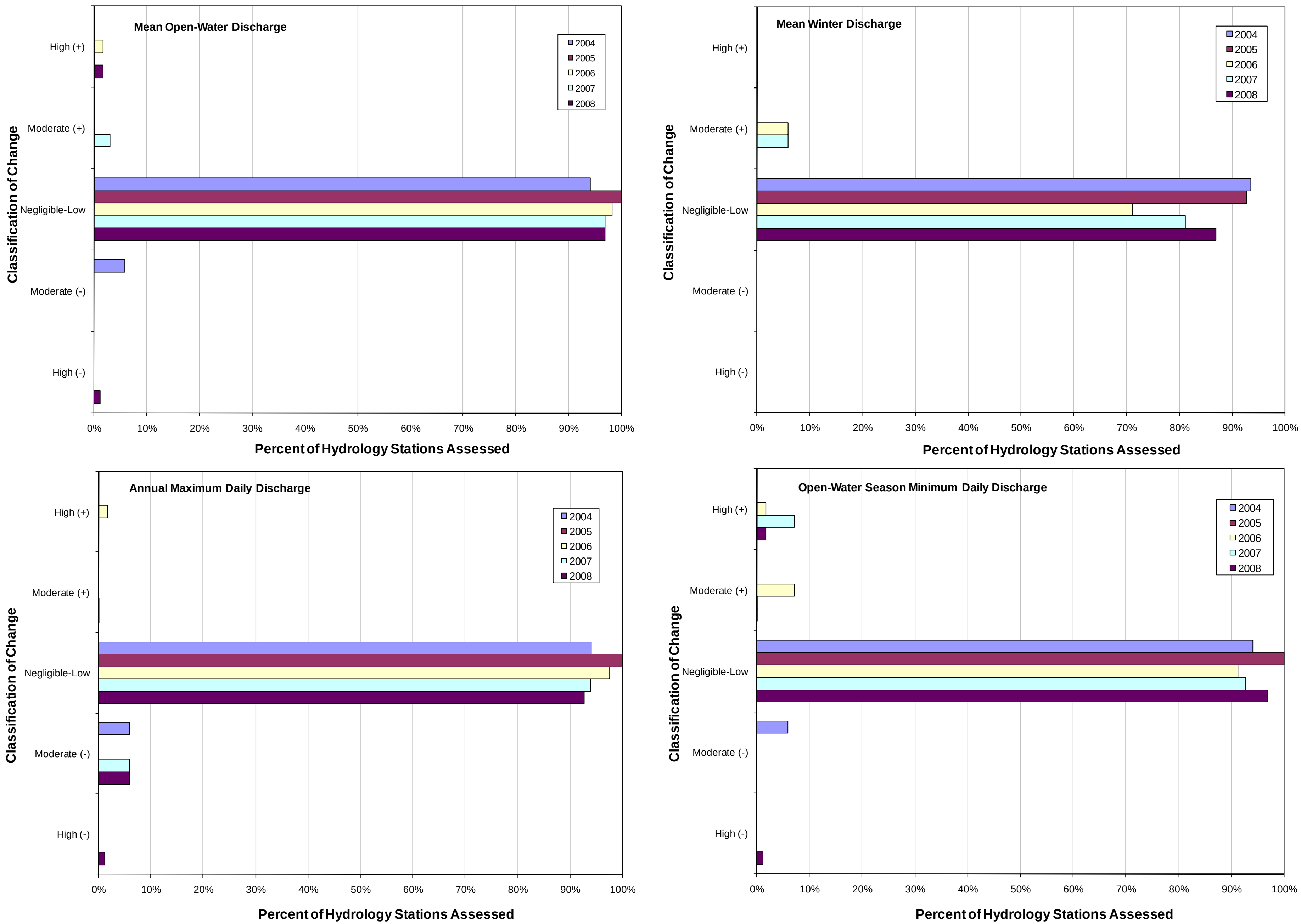
Activities resulting in closed-circuited areas were the focal project activities that had the second greatest influence on hydrological conditions in 2008 in RAMP FSA watersheds. The largest areas of closed-circuited land are in the Athabasca local catchment, followed by the Muskeg, Tar, and Steepbank watersheds.

Activities resulting in land change areas that were not closed-circuited generally had minor influences on hydrologic conditions in RAMP FSA watersheds in 2008; the largest change occurred in the Athabasca local catchments and the Muskeg and Steepbank watersheds.

The hydrologic changes from focal project activities plus all other active oil sands projects in the RAMP FSA are estimated to be only marginally greater than the hydrologic changes from the focal projects alone.

The average estimated percent change from 2004 to 2008 in each of the four measurement endpoints are presented in Figure 6.1-2, which shows the percent of the area assessed each year falling under each change classification. In all cases, most of the assessed area has experienced Negligible-Low hydrologic changes.

Figure 6.1-2 Change in hydrologic measurement endpoints among hydrology stations monitored by RAMP, 2004 to 2008.



6.2 WATER QUALITY

RAMP water quality data from fall were compared with regional *baseline* values, to assess the likelihood that observations in 2008 fell outside the range of regional natural variability. These *baseline* ranges were developed from all historical observations at all *baseline* 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, Poplar, Beaver, and Hangingstone watersheds, plus Fort, McLean, , and Shipyard and Isadore's lakes; and
- **Cluster 3:** Athabasca River mainstem and delta.

RAMP 2008 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 *baseline* data versus 2008 data), and by station status in 2008 (i.e., *baseline* versus *test*). Variables selected for regional analysis included a subset of key measurement endpoints presented in Section 5, and other variables with values that frequently exceeded water-quality guidelines in 2008. Figure 6.2-1 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 25^{th} to 75^{th} percentiles of observations (the median, or 50^{th} percentile appears as a central line within each box), error bars describing the 5^{th} and 95^{th} percentiles, and symbols (×) representing individual data points (outliers) that fall outside the 5^{th}-to-95^{th} percentile range.

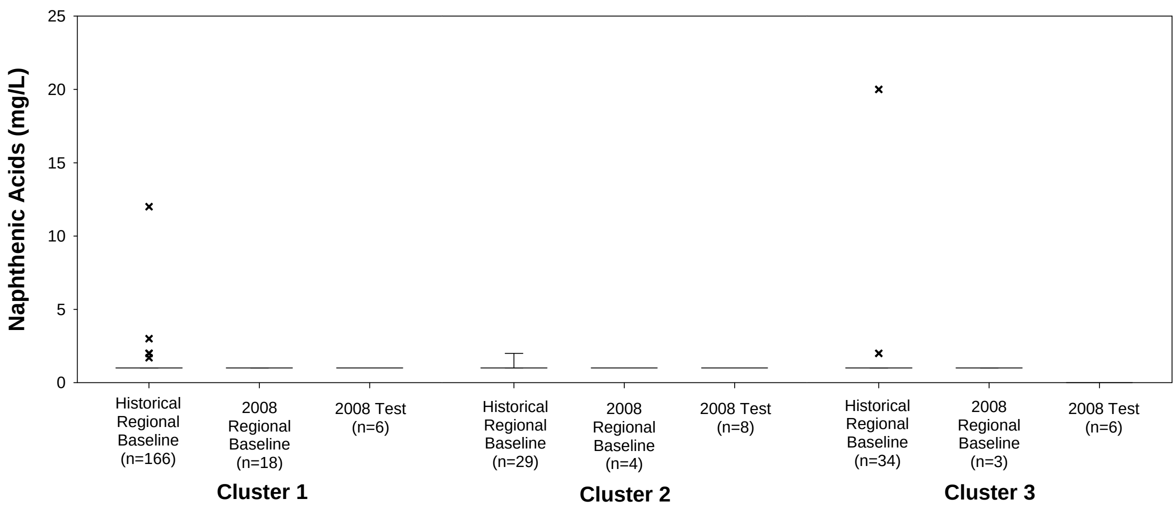
6.2.1 Water Quality Variables Associated with Oil Sands

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 2008 (Figure 6.2-1), when naphthenic acids were not detected at any station except Stanley Creek (*test*, in the Muskeg River watershed), where a value of 1 mg/L, equal to the detection limit, was measured. An identical value was measured in Stanley Creek water by RAMP in 2001, when the station was classified as *baseline*. However, there is considerable uncertainty associated with measurement of any water quality variable at its detection limit. Naphthenic acids have previously been detected at concentrations at or near the detection limit at some tributary stations belonging to Clusters 1 and 2.

Figure 6.2-1 Naphthenic acids in waters of the RAMP FSA, 2008 and historical data.



Two observations of relatively high concentrations of naphthenic acids exist in the RAMP database, both at *baseline* stations sampled on September 17, 1998: at the upper Muskeg River (MUR-6, *baseline*); and at the Athabasca River mainstem upstream of Donald Creek, west bank (ATR-DC-W, *baseline*). 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 2008 is consistent with an absence of measurable effects of oil-sands development or other human activities on this measurement endpoint.

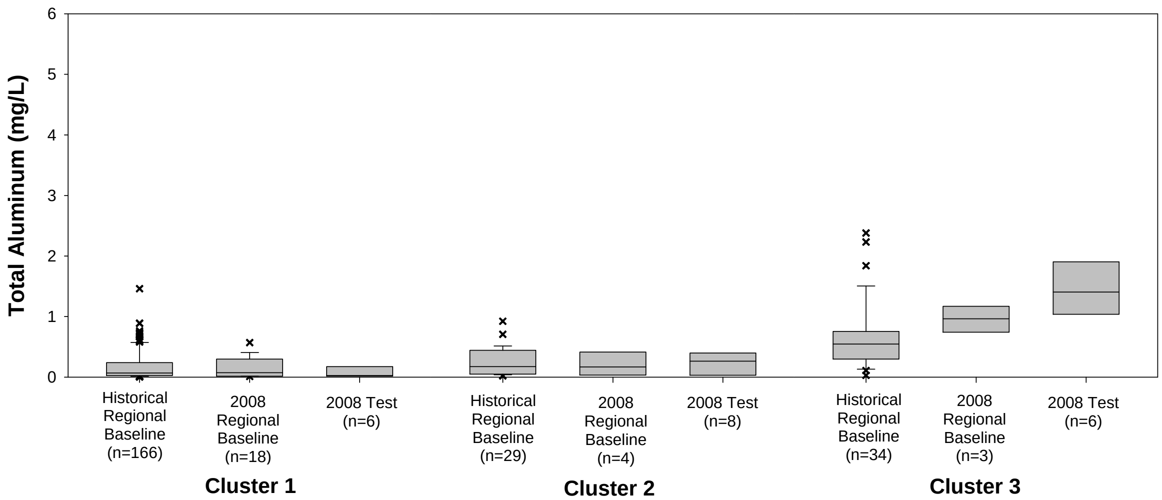
The RAMP 2009 program will include improved detection limits for naphthenic acids (i.e., 0.1 mg/L, instead of 1.0 mg/L), which should result in an increased frequency of detectable values, and yield a better understanding of *baseline* naphthenic acid concentrations in the region. As of April 2008, discussions with analytical laboratories were ongoing regarding refinement of this analytical test and its associated detection limit.

6.2.2 Other Water Quality Variables

6.2.2.1 Aluminum

Total aluminum concentrations measured in the RAMP FSA since 1998 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 FSA, 2008 and historical data.



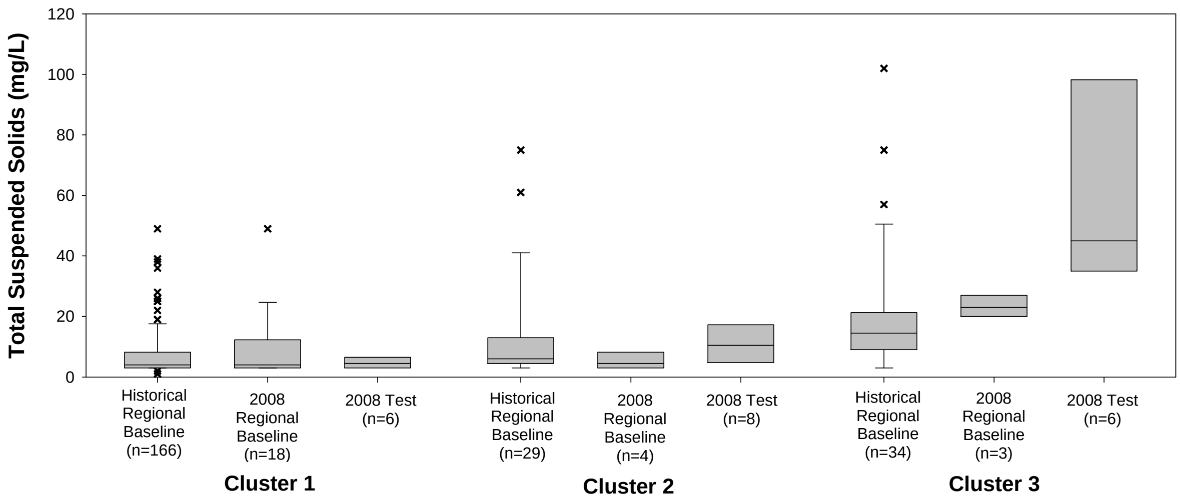
Although total aluminum concentrations have been highly variable in waters of the RAMP FSA, clear differences between regional clusters of stations are apparent: values for Cluster 1 and 2 generally are low relative to Clusters 3 (although exceedance of the CCME guideline has still been frequent); those in Cluster 3 (Athabasca River mainstem) have typically been much higher, well above the CCME guideline.

In 2008, median total aluminum concentrations for stations in Clusters 1 and 2 (tributaries and lakes) generally were similar to those of historical *baseline* data, and all 2008 data fell within the range of historical *baseline* data for these clusters. Median and interquartile (i.e., 25^{th}-to-75^{th}-percentile) concentrations at *test* stations in 2008 were similar to, or less than, those in 2008 at *baseline* stations belonging to Clusters 1 or 2.

Median total aluminum concentrations in 2008 at stations in the Athabasca River mainstem (Cluster 3) defined as *test* (i.e., all downstream of ATR-DC) were generally higher than those at upstream *baseline* stations (ATR-DC and ATR-UFM) in 2008, and higher than the majority of historical *baseline* observations at the three RAMP stations located upstream of oil-sands influences (i.e., stations at Donald Creek and upstream of Fort McMurray). However, 75^{th} and 95^{th} percentiles for downstream *test* stations in 2008 were within the range of historical RAMP observations at these *baseline* stations.

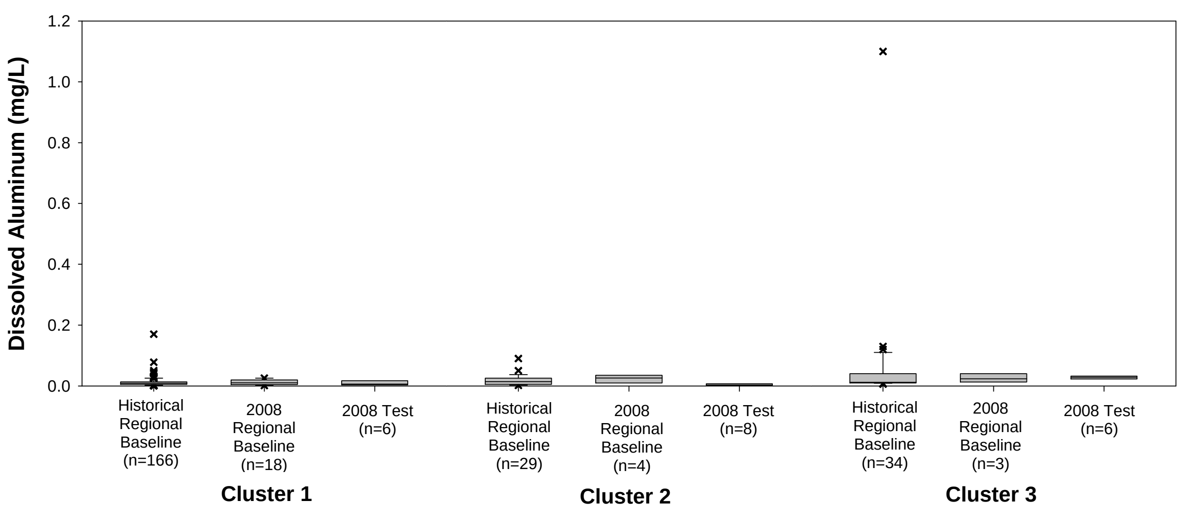
Aluminum is the most abundant metal on Earth, and commonly occurs in aquatic environments in particulate form, which is not readily bioavailable; its aquatic toxicity is strongly associated with its dissolved form, whose toxicity is highly dependant on pH, hardness, and dissolved organic carbon, increases 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 (as of 2008, r_s=0.760, n=396, r_{crit}=|0.099|). Concentrations of total suspended solids in the RAMP FSA, within and among clusters, and between 2008 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 FSA, 2008 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 (B.C. 2006). In fall 2008, concentrations of dissolved aluminum in waters of the RAMP FSA were below this BC chronic guideline at all stations (Figure 6.2-4).

Figure 6.2-4 Dissolved aluminum in waters of the RAMP FSA, 2008 and historical data.



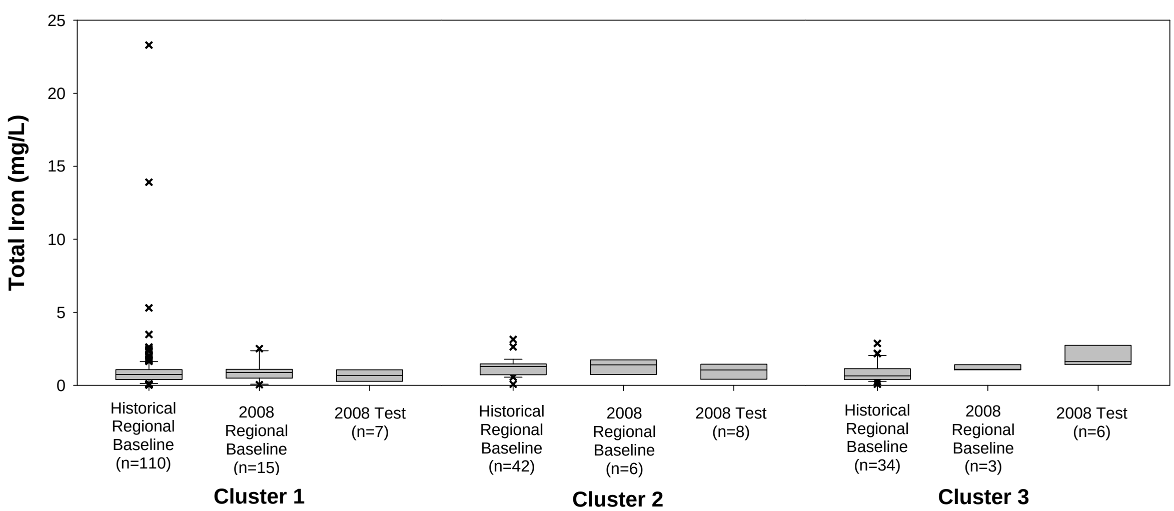
As is apparent from Figure 6.2-4, dissolved aluminum values have been consistently low throughout the RAMP FSA since 1997, with the exception of a single high value of 1.1 mg/L found at station ATR-DC-W in September 2001. This single observation may be erroneous, given the concentration of dissolved 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 FSA, 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 surface-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 (e.g., Government of Iowa 2005). Naturally occurring, dissolved iron may contribute much of the stained colour of muskeg-related waters, such as those in the RAMP FSA.

In the RAMP FSA, 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 (see Section 5 and Figure 6.2-5, Figure 6.2-6). However, total and dissolved iron concentrations in 2008 were similar or lower at *test* stations than at *baseline* stations for tributary or lake station (Clusters 1 or 2), and values at all stations sampled in 2008 were within the range of regional *baseline* data. In the complete RAMP water-quality dataset, dissolved iron was most strongly correlated with total colour and dissolved organic carbon (as of 2008, r_s=0.710 and 0.550, respectively, n=396, r_{crit}=|0.099|). 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 FSA, 2008 and historical data.



6.2.2.3 Arsenic

Total arsenic in waters of the RAMP FSA in 2008 generally was present at similar concentrations at *test* stations and *baseline* stations; all concentrations measured in 2008 were within the range of historical regional *baseline* data (Figure 6.2-7). All total arsenic concentrations observed by RAMP since its inception in 1997 (n=486) 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 6.2-6 Dissolved iron in waters of the RAMP FSA, 2008 and historical data.

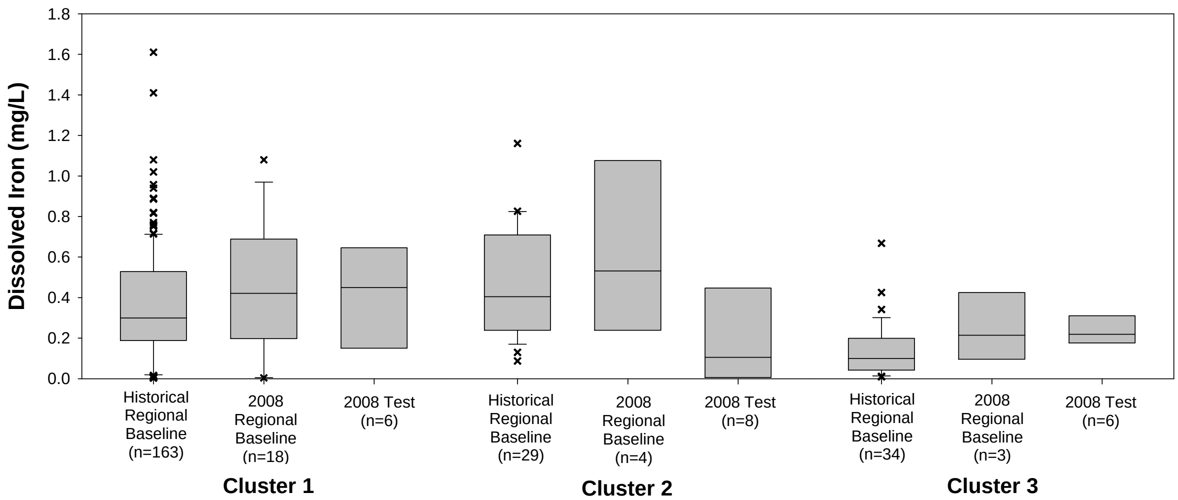
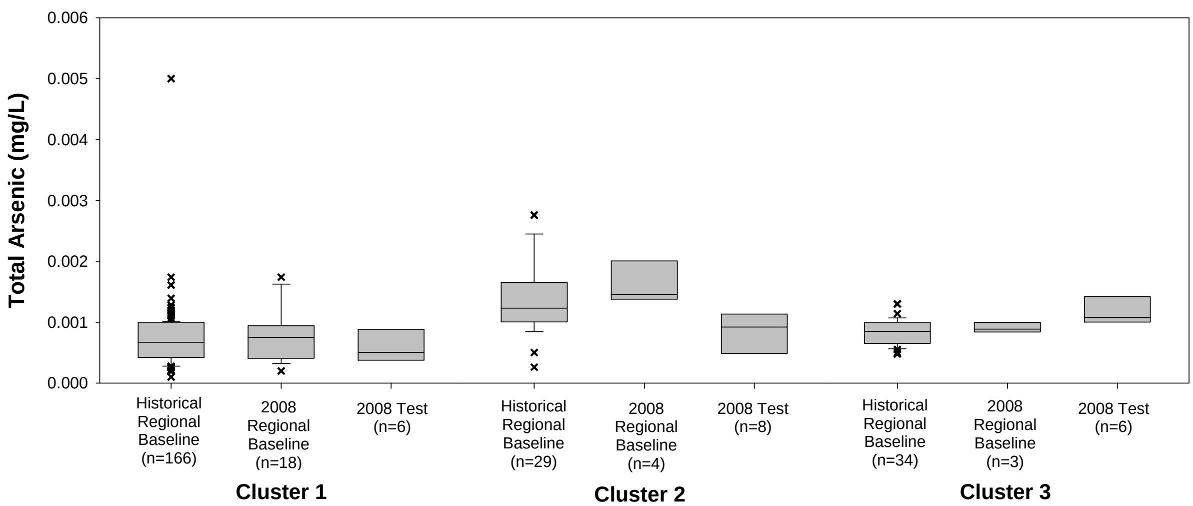


Figure 6.2-7 Total arsenic in waters of the RAMP FSA, 2008 and historical data.

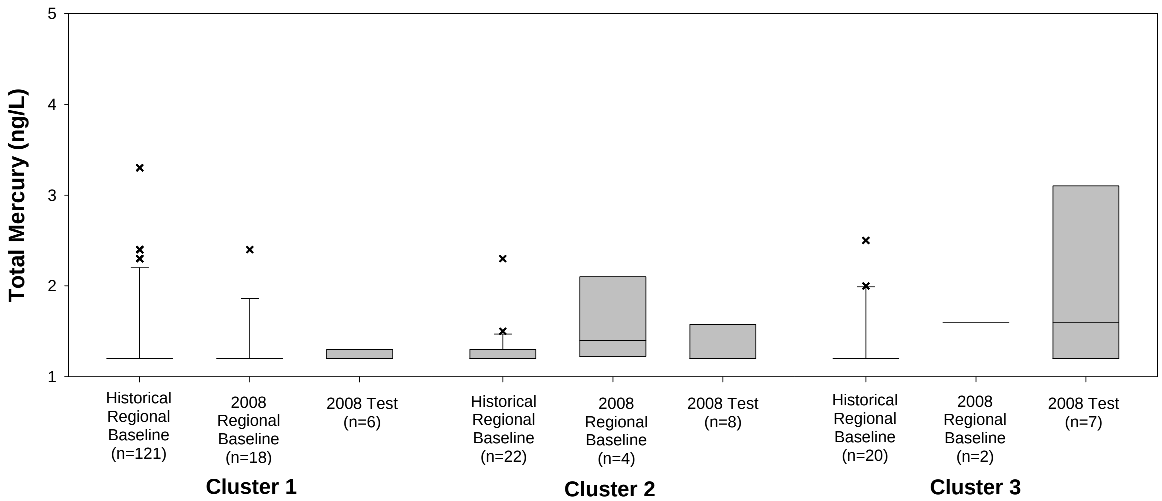


6.2.2.4 Mercury

Concentrations of total mercury (ultra-trace) in waters of the RAMP FSA in 2007 and 2008 have generally been higher than those observed in previous years, at both *test* stations and *baseline* stations. However, in 2008, median values of mercury were similar or lower in *test* stations than in *baseline* stations for all three clusters; for clusters 1 and 2, this median was at the detection limit of 1.2 ng/L, given most observations were non-detectable (Figure 6.2-8). One observation in fall 2008, of 8.1 ng/L at lower Beaver River station BER-1, exceeded the CCME water-quality guideline for the protection of aquatic life of 5 ng/L.

This guideline also was exceeded in observations in spring 2008 at *baseline* station BER-2 (upper Beaver River, 5.6 ng/L), and in the lower Mackay River (MAR-1, 6.4 ng/L). Mercury was not detectable in 34 of 50 water samples collected by RAMP in 2008, and was below the 5 ng/L guideline in all but three samples.

Figure 6.2-8 Total mercury in waters of the RAMP FSA, 2008 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 1999a), 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 water quality variable, which encompasses a wide variety of phenolic compounds, including polyhydric species. AENV (1999b) 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 2008, at *baseline* and *test* locations (Figure 6.2-9), although all values were within the range of historical RAMP observations for each station cluster. Median values for *test* stations were lower than those for *baseline* stations for all three clusters in 2008. All phenol concentrations in 2008 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 (as of 2008, r_s=0.450 and 0.400, respectively, n=396), suggesting that phenols in regional waters are associated with dissolved humic substances in water.

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 (as of 2008, r_s=0.980, n=396) 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 6.2-9 Total phenols in waters of the RAMP FSA, 2008 and historical data.

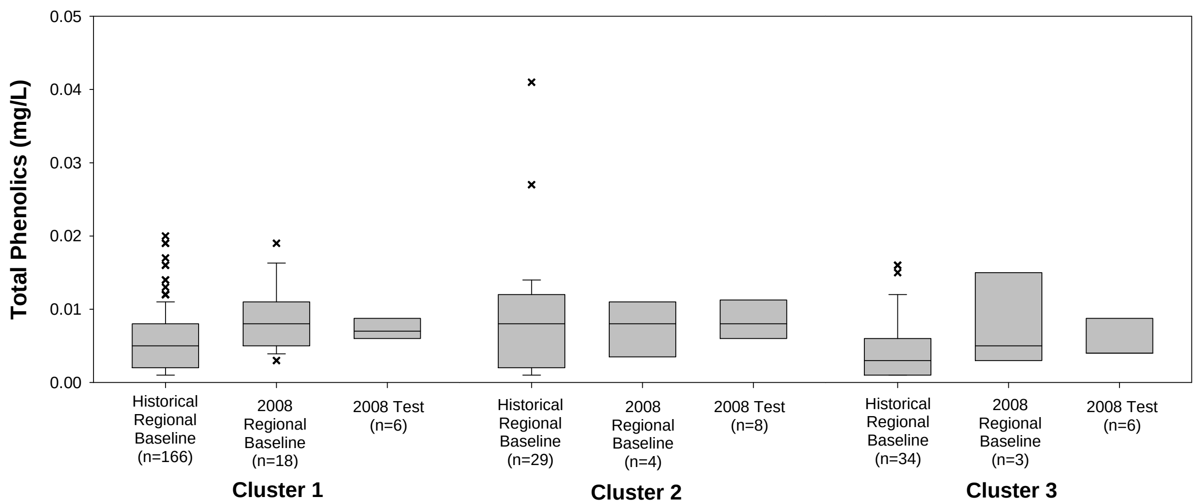
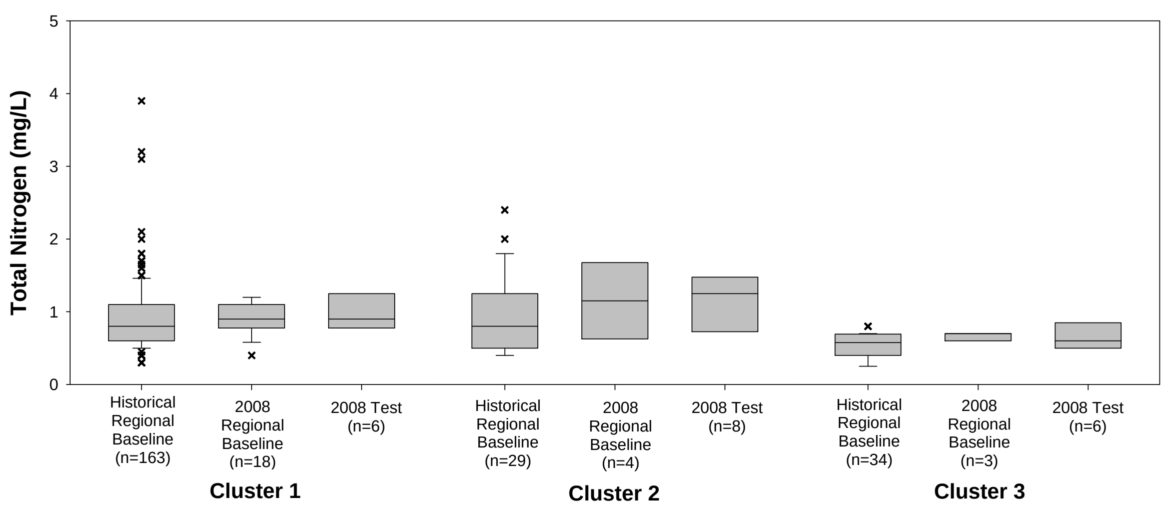
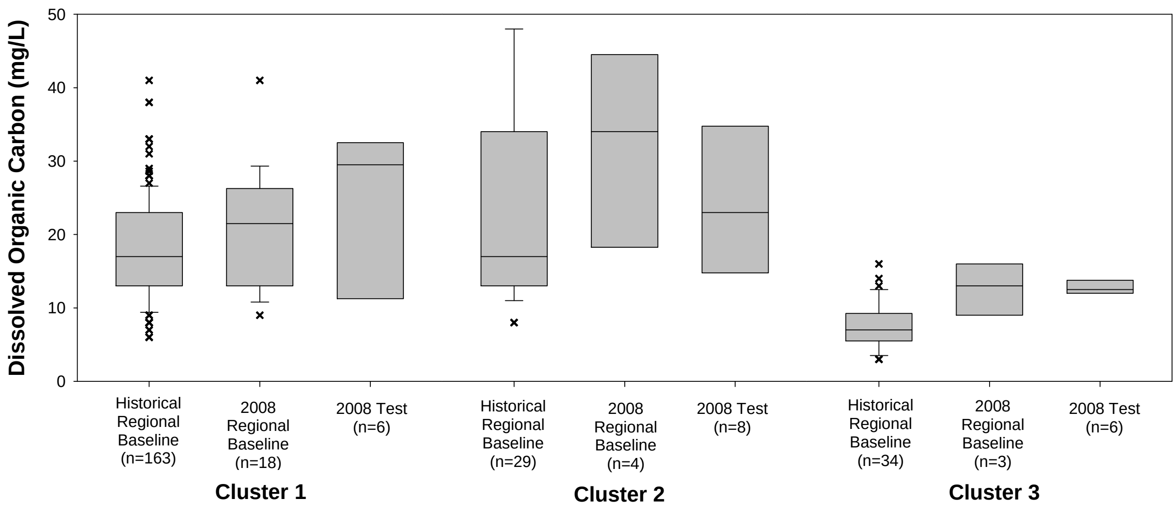


Figure 6.2-10 Total nitrogen in waters of the RAMP FSA, 2008 and historical data.



Medians and ranges of TN concentrations were similar between *baseline* and *test* stations in 2008, and were within the range of historical regional *baseline* values, except in Cluster 2, where one station (lower Tar River, TAR-1) exhibited very high TN relative to other stations (i.e., 4.3 mg/L). The majority of this nitrogen at TAR-1 was present as nitrate (3.4 mg/L) and organic forms (0.8 mg/L); ammonia was not detectable in the sample from TAR-1 in fall 2008 (<0.05 mg/L). This composition of nitrogen species at TAR-1 differed from 2007, when nitrogen was comprised predominantly inorganic nitrogen (free ammonium and nitrate), rather than organic nitrogen.

Figure 6.2-11 Dissolved organic carbon in waters of the RAMP FSA, 2008 and historical data.



Total dissolved phosphorus (TDP) also has exhibited relatively high variability among stations and years since RAMP began sampling in 1998, particularly in stations in tributary stations in Clusters 1 and 2 (Figure 6.2-12). In fall 2008, median TDP concentrations at *test* stations of all clusters were below median values at *baseline* stations; these medians also fell within the range of historical regional *baseline* values.

6.2.2.7 Major Ions

Concentrations of sulphate and chloride measured by RAMP in fall 2008 and historically, organized by station cluster and classification, appear in Figure 6.2-13 andFigure 6.2-14; concentrations of total dissolved solids (TDS) appear in Figure 6.2-15. Although median concentrations of sulphate were generally similar in 2008 to those observed in previous years in each station cluster, the range of values observed was generally greater for both *baseline* and *test* stations in 2007 and 2008 than for historical observations, particularly for stations in Cluster 2. The upper quartile of chloride values for *test* stations in Cluster 2 in 2008 was higher than that for *baseline* stations in 2008, and outside the range of historical regional *baseline* data; this also was observed in 2007 (RAMP 2008). Stations with sulphate or chloride concentrations in the upper range of this group of *test* stations in fall 2008 included the lower Tar River (TAR-1), lower Beaver River (BER-1), Shipyard Lake (SHL-1), and Isadore's Lake.

6.2.3 Summary

With some exceptions, water quality data collected by RAMP in fall 2008 was similar for all key measurement endpoints between *test* and *baseline* stations. Most data from *baseline* and *test* stations in 2008 fell within the range of historical observations from previous years.

Figure 6.2-12 Total dissolved phosphorus in waters of the RAMP FSA, 2008 and historical data.

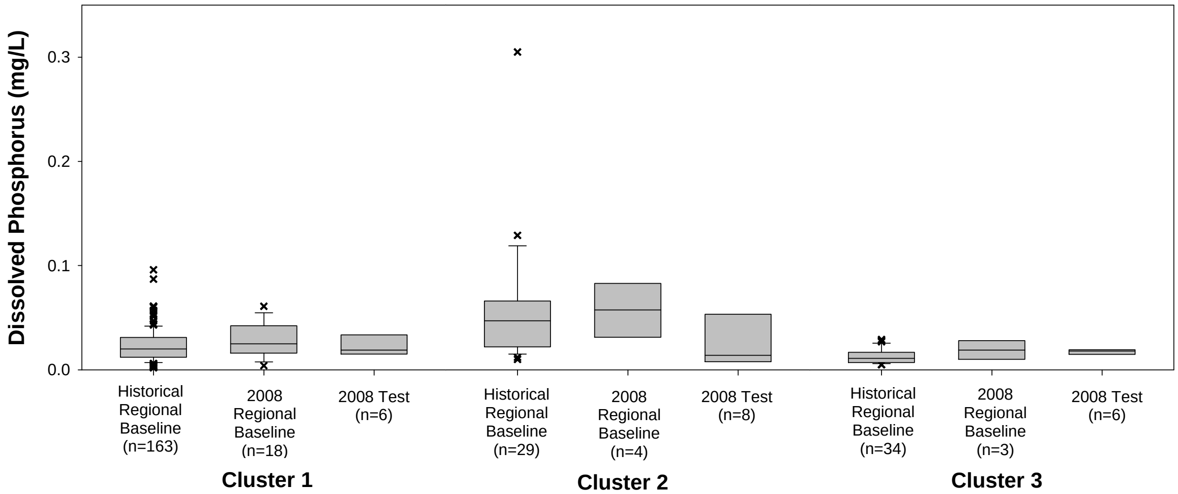


Figure 6.2-13 Total sulphate in waters of the RAMP FSA, 2008 and historical data.

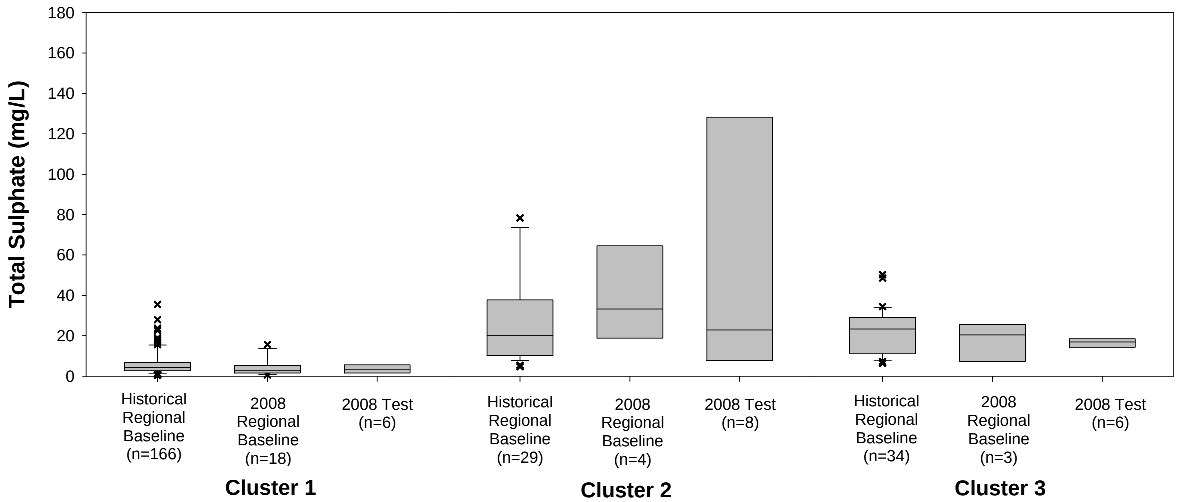


Figure 6.2-14 Total chloride in waters of the RAMP FSA, 2008 and historical data.

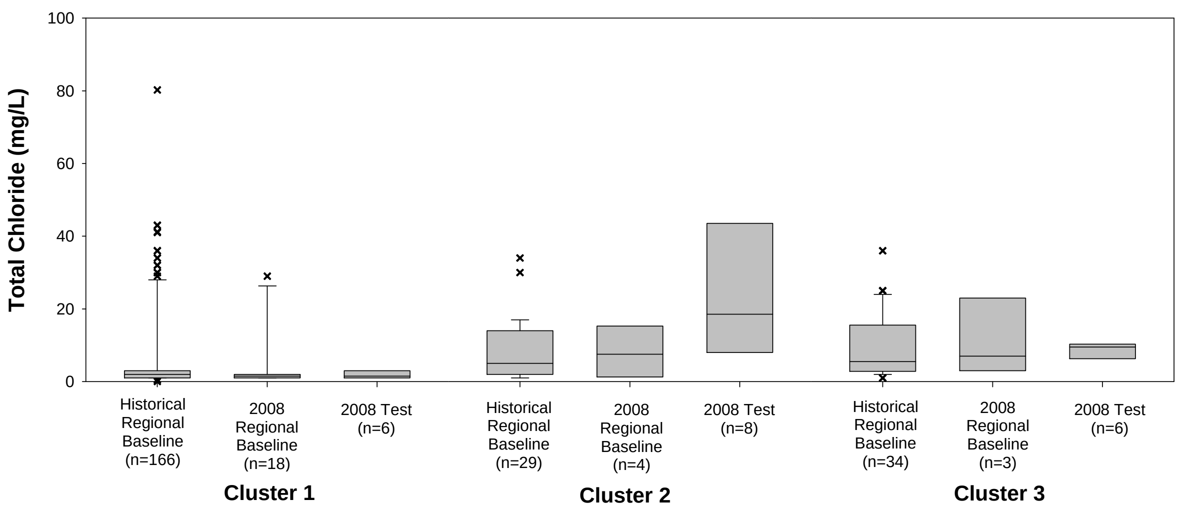
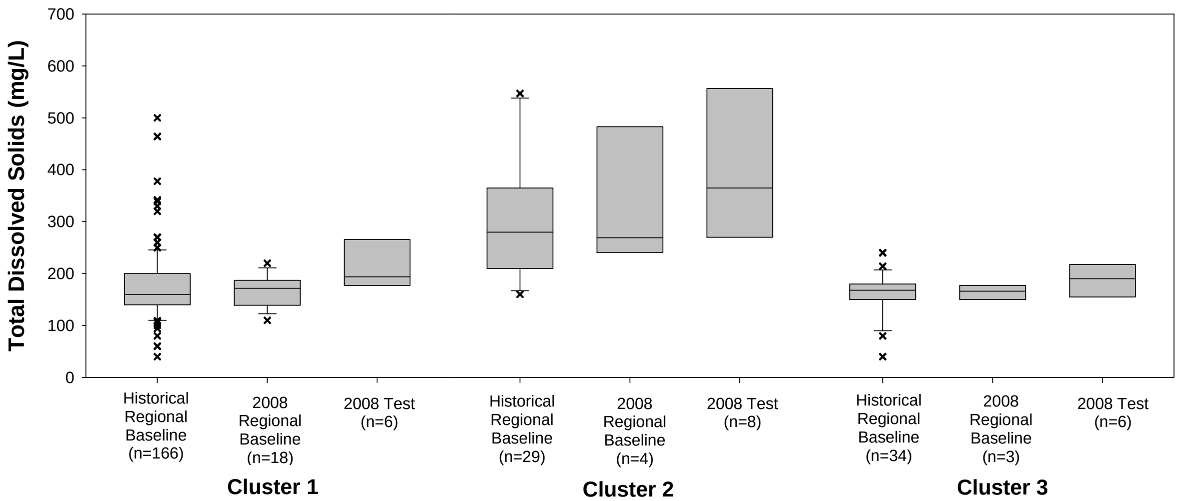


Figure 6.2-15 Total dissolved solids in waters of the RAMP FSA, 2008 and historical data.



6.3 BENTHIC INVERTEBRATE COMMUNITIES AND SEDIMENT QUALITY

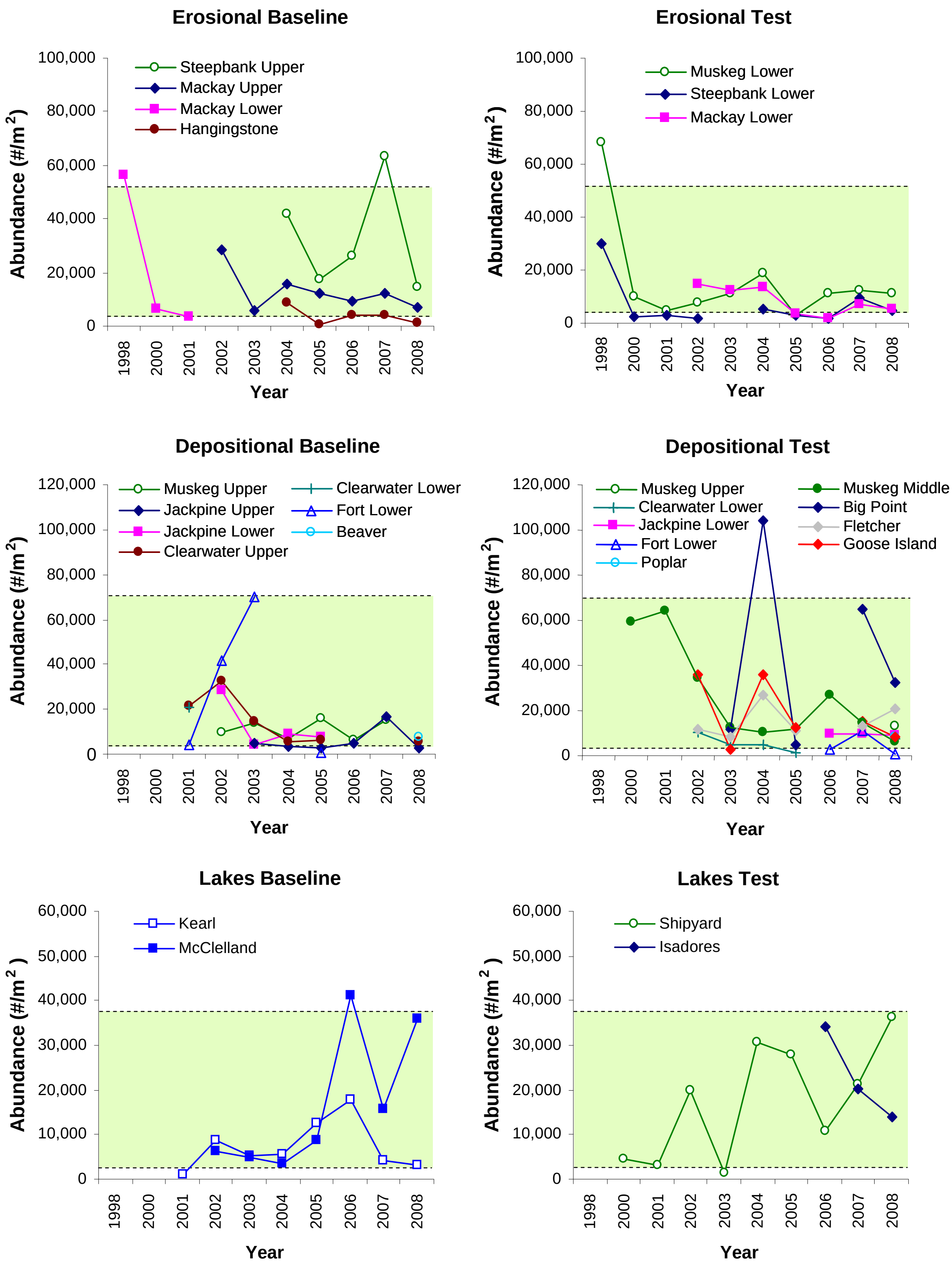
6.3.1 Regional Assessment of Benthic Invertebrate Community Conditions at the RAMP FSA Level

The background conditions for each index and major habitat class are illustrated in the figures below (Figure 6.3-1 through to Figure 6.3-5). Community composition differs fundamentally between erosional and depositional river reaches, and between lakes and rivers. The normal ranges of variation in benthic invertebrate community measurement endpoints in *baseline* depositional reaches, *baseline* erosional reaches, and *baseline* lakes has been computed to provided guide-posts for interpreting regional variations (Figure 6.3-1 through Figure 6.3-5).

The bounds on each measurement endpoint that define the normal range are statistical estimates based on the 5th and 95th percentile of observations from reaches or lakes that are classified as being in a *baseline* condition. One would predict, then that 5% of observations will naturally fall outside the normal range of variation, and that appears to be the case for systems in a *baseline* condition. That feature of the data is observed by inspection of the time plots of *baseline* data for all five of the conventional benthic invertebrate community measurement endpoints. By inspection of the time trends for reaches and lakes classified as *test* we observe that watercourses classified as *test* have had similar frequency of exceedances of the 5th and 95th percentile range. This therefore indicates that, at the regional level, variations within and among reaches (and lakes) designated as *test* have generally been within the normal (background) range of variability as observed in *baseline* reaches (and lakes).

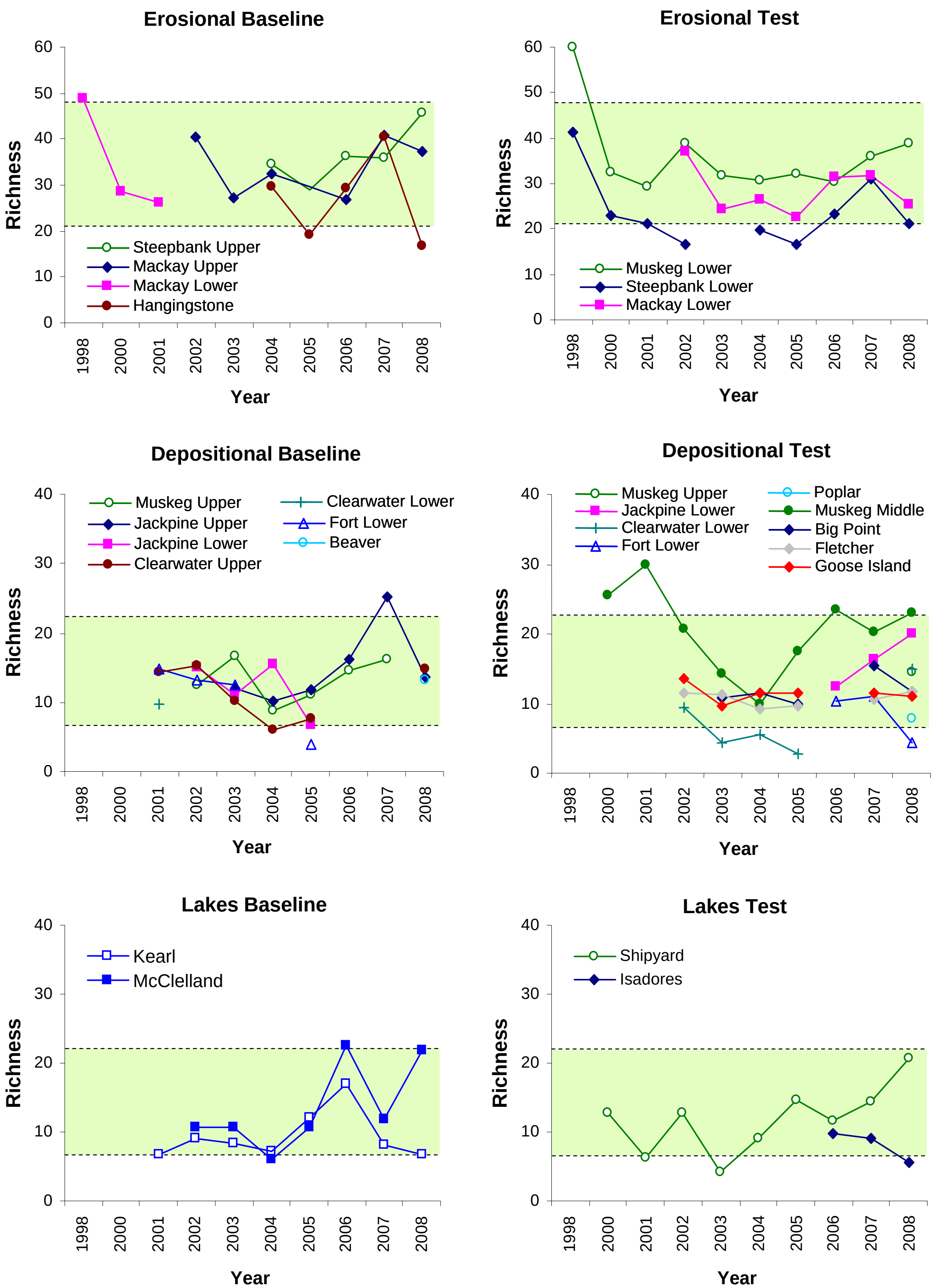
In addition, most differences in benthic invertebrate community measurement endpoints between *test* reaches and *baseline* reaches in watersheds were not significant as of 2008. The exceptions to this in 2008 were the lower Steepbank and lower Poplar rivers, lower Fort Creek, and Isadore's Lake, all of which had a number of significant differences between *test* and *baseline* reaches (and lakes), and values of measurement endpoints that were below the 5th percentile of *baseline* ranges for the particular habitat type in the RAMP FSA.

Figure 6.3-1 Variations in total benthic community abundance across years for river reaches, Athabasca Delta stations and lakes in the RAMP FSA.



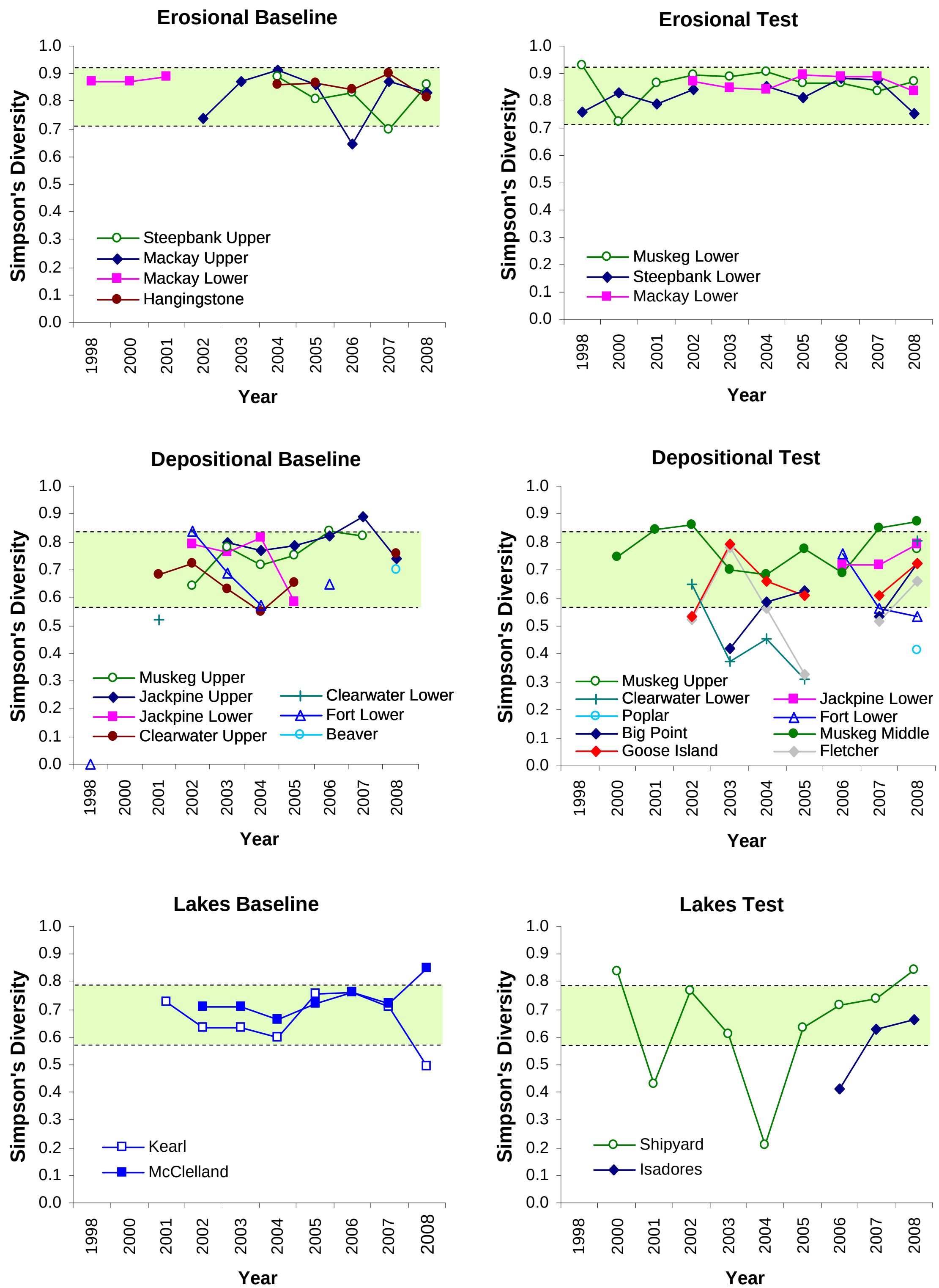
Note: the normal range of variation (5th and 95th percentiles) is depicted by dashed lines.

Figure 6.3-2 Variations in benthic community taxa richness across years for river reaches, Delta Stations and Lakes in the RAMP FSA.



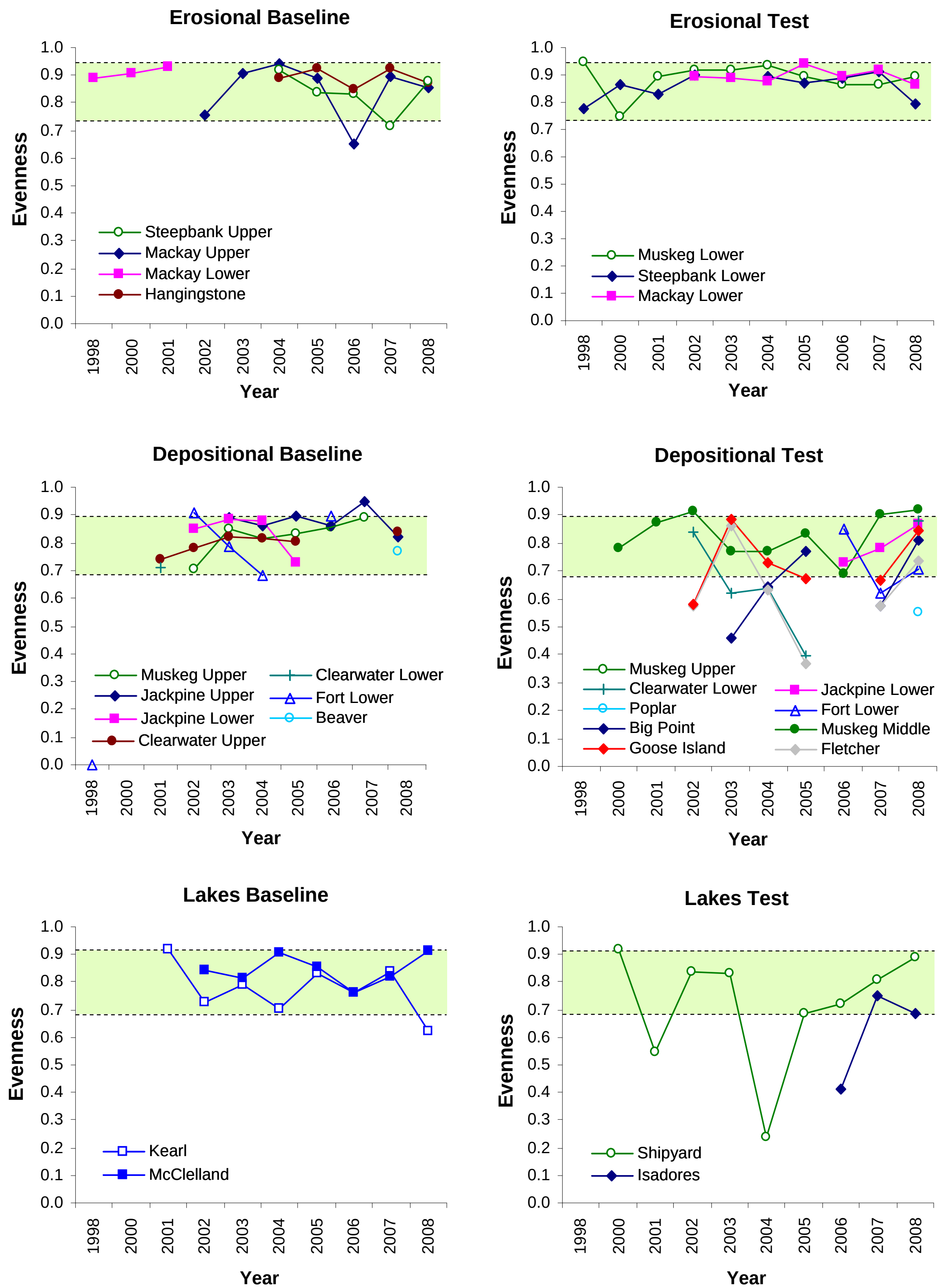
Note: the normal range of variation (5th and 95th percentiles) is depicted by dashed lines.

Figure 6.3-3 Variations in benthic community Simpson's Diversity across years for river reaches, Delta Stations and Lakes in the RAMP FSA.



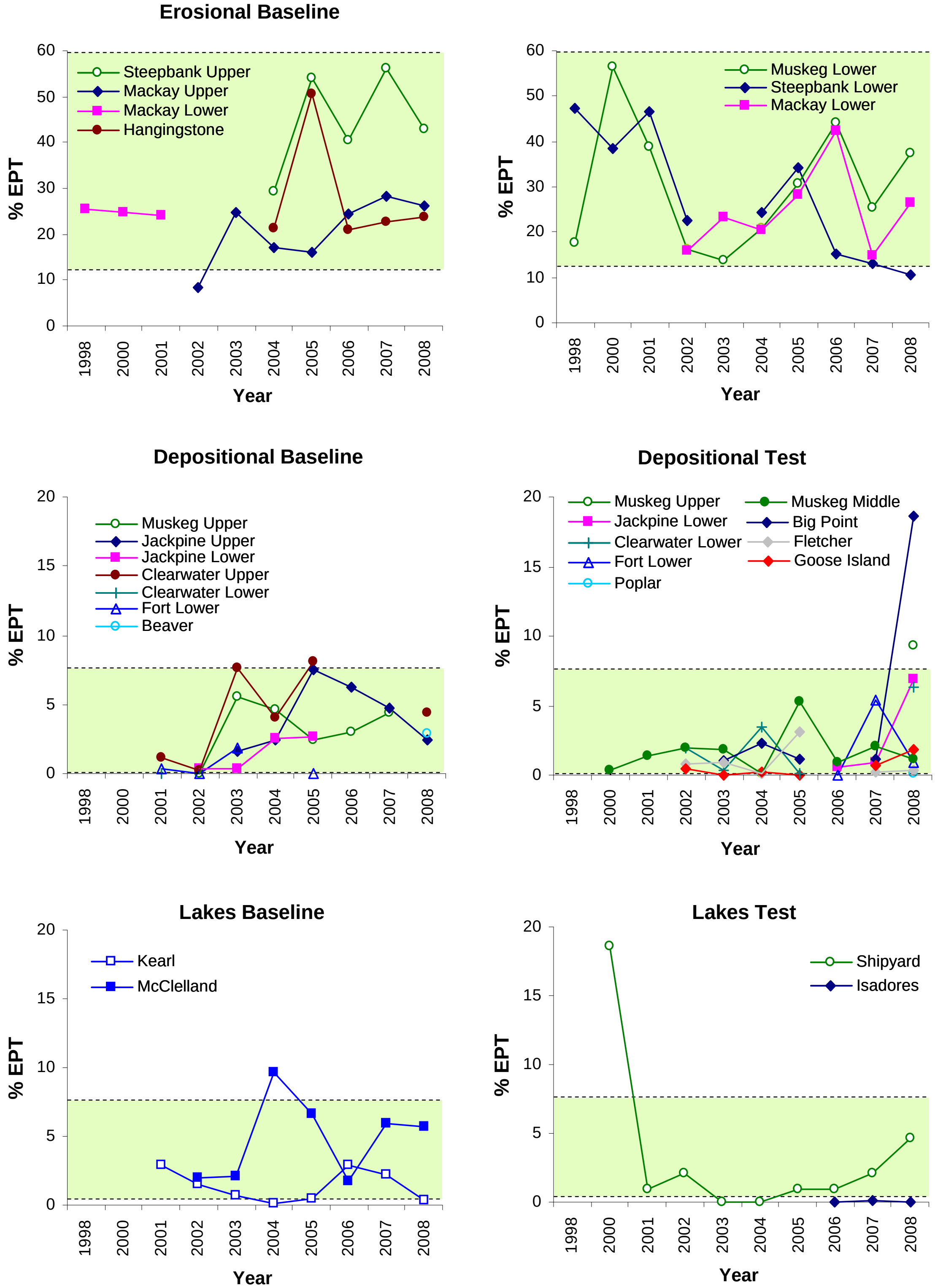
Note: the normal range of variation (5th and 95th percentiles) is depicted by dashed lines.

Figure 6.3-4 Variations in benthic community Evenness across years for river reaches, Delta Stations and Lakes in the RAMP FSA.



Note: the normal range of variation (5th and 95th percentiles) is depicted by dashed lines.

Figure 6.3-5 Variations in benthic community %EPT for river reaches in the RAMP FSA.



Note: the normal range of variation (5th and 95th percentiles) is depicted by dashed lines.

An interesting observation from inspection of the data was that there appears to have been a region-wide reduction in average number of taxa in 2003/2004 in both *baseline* and *test* depositional and erosional river reaches, and region-wide reduction in percent EPT in erosional reaches in that same time period. That observation was more acutely apparent in depositional rivers (*baseline* and *test*). Two *baseline* depositional reaches had average taxa richness per sample just below the 5^{th} percentile of *baseline* observations, as did the lower Clearwater River. Since that time, the average taxa richness per sample has increased for both *baseline* and *test* reaches, including the Clearwater River. These phenomena of the data are unlikely to have been sampling artifacts since the field crews and taxonomic laboratory have been the same since 2003. Rather, they appear to reflect natural region-wide variations.

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 (2001).

Comparison of TRH and TPH data from duplicate samples collected by RAMP in 2005 found a best-fit relationship of TPH=2.183(TRH) (RAMP 2006, Appendix E). Data collected by RAMP using the CCME four-fraction test since 2005 has shown that most hydrocarbons in regional sediments are comprised of high-molecular-weight species (i.e., those 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; in sediments sampled by RAMP from 2006 to 2008, total PAHs were correlated with F3 and F4 fractions, and with total hydrocarbons (r_s=0.53, 0.47 and 0.52, respectively; Appendix F).

Observed concentrations of total hydrocarbons in sediments of Athabasca river tributaries 1998 appear in Figure 6.3-6. Graphs in include TRH (1997 to 2005) and TPH (2005 to 2008, shown at a 1:2 vertical scale relative to TRH). Stations considered *baseline* in the year of sampling show green background shading, while those considered *test* 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 *baseline* and those defined as *test*. Historically, highest concentrations of total hydrocarbons have been observed in the Calumet River (2005 and 2006, upper and lower, *baseline*), Stanley Creek (2003, *test*), Shipyard Lake (2004, *test*), and McLean Creek (1999 and 2000, *test*). Highest concentrations of total hydrocarbons observed in 2008 were at Kearl Lake and McClelland Lake, both *baseline* 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-to-2008 sediment dataset, total hydrocarbons was significantly and moderately correlated with TOC (r_s=0.63; Appendix F). Therefore, concentrations of total hydrocarbons in sediments normalized to 1% organic carbon also were calculated and are presented in Figure 6.3-7. Adjustment of hydrocarbon concentrations for organic content affects particularly concentrations in sediments of lakes, where organic carbon content is typically high.

The highest carbon-normalized concentrations of total hydrocarbons in sediments observed by RAMP since 1997 have occurred in the lower Ells River (2006 and 2007, *baseline*), the lower Steepbank River (1997 and 2005, *test*), McLean Creek (1999 and 2005, *test*), the lower Calumet River (2006, *baseline*), and, in 2008, Fort Creek (*test*). 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)

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 (*test*), the lower Ells River (*baseline*), Stanley Creek (*test*), McLean Creek (*test*) and the lower Steepbank River (*test*). Following normalization to 1% TOC, highest total PAH concentrations in sediment since 1997 have been observed in the lower Ells River (*baseline*), the upper Steepbank River (*baseline*) and McLean Creek (*test*). 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.

In 2008, highest total concentrations of PAHs were observed in sediments from lower Fort Creek (FOC-D1, *test*), followed by Shipyard Lake (SHL-1, *test*) and lower Jackpine Creek (JAC-D1, *baseline*) (Figure 6.3-10). 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).

Figure 6.3-6 Concentrations of total hydrocarbons in sediments sampled by RAMP in tributaries to the Athabasca River, 1997 to 2008.

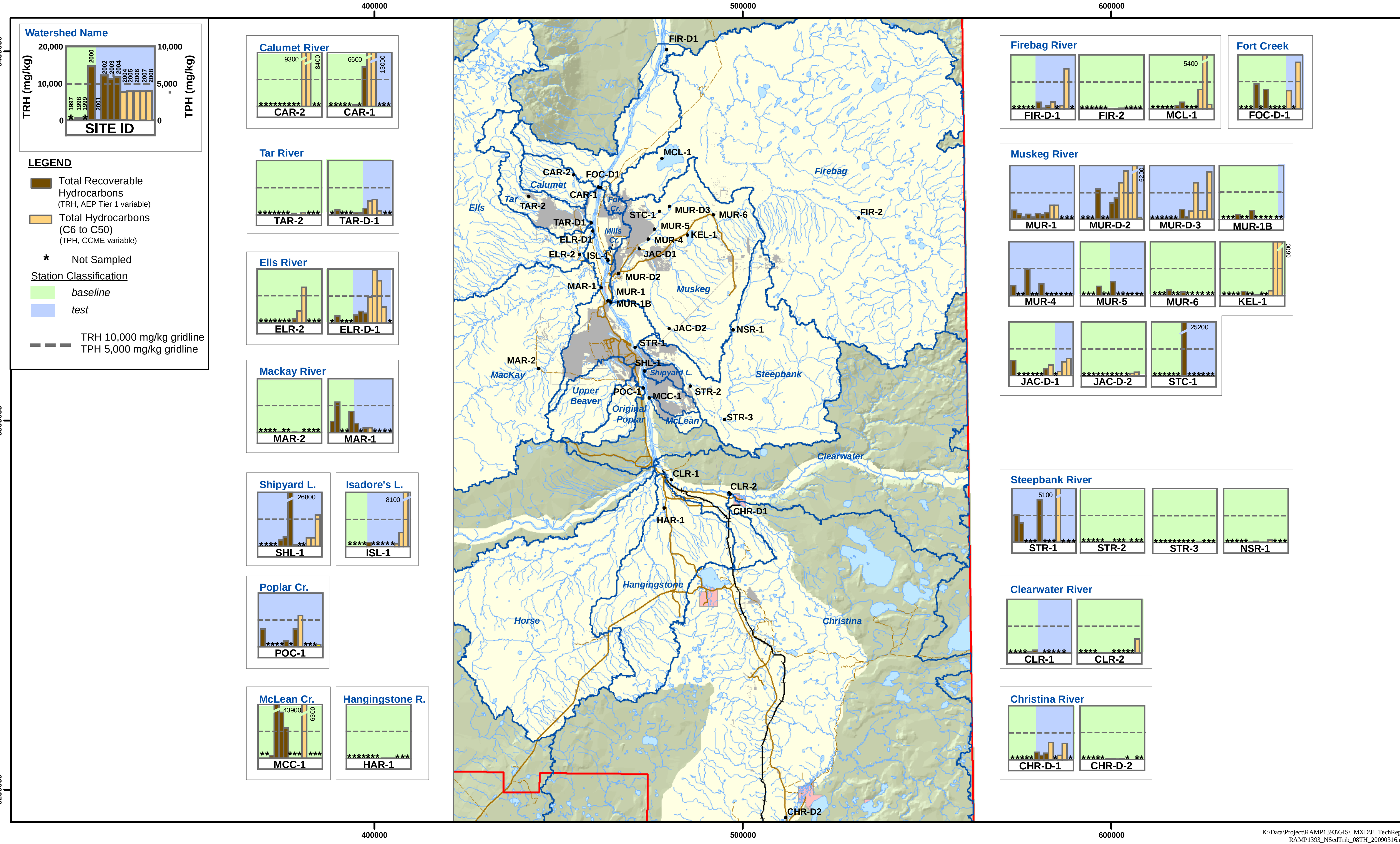


Figure 6.3-7 Carbon-normalized concentrations of total hydrocarbons in sediments sampled by RAMP in tributaries to the Athabasca River, 1997 to 2008.

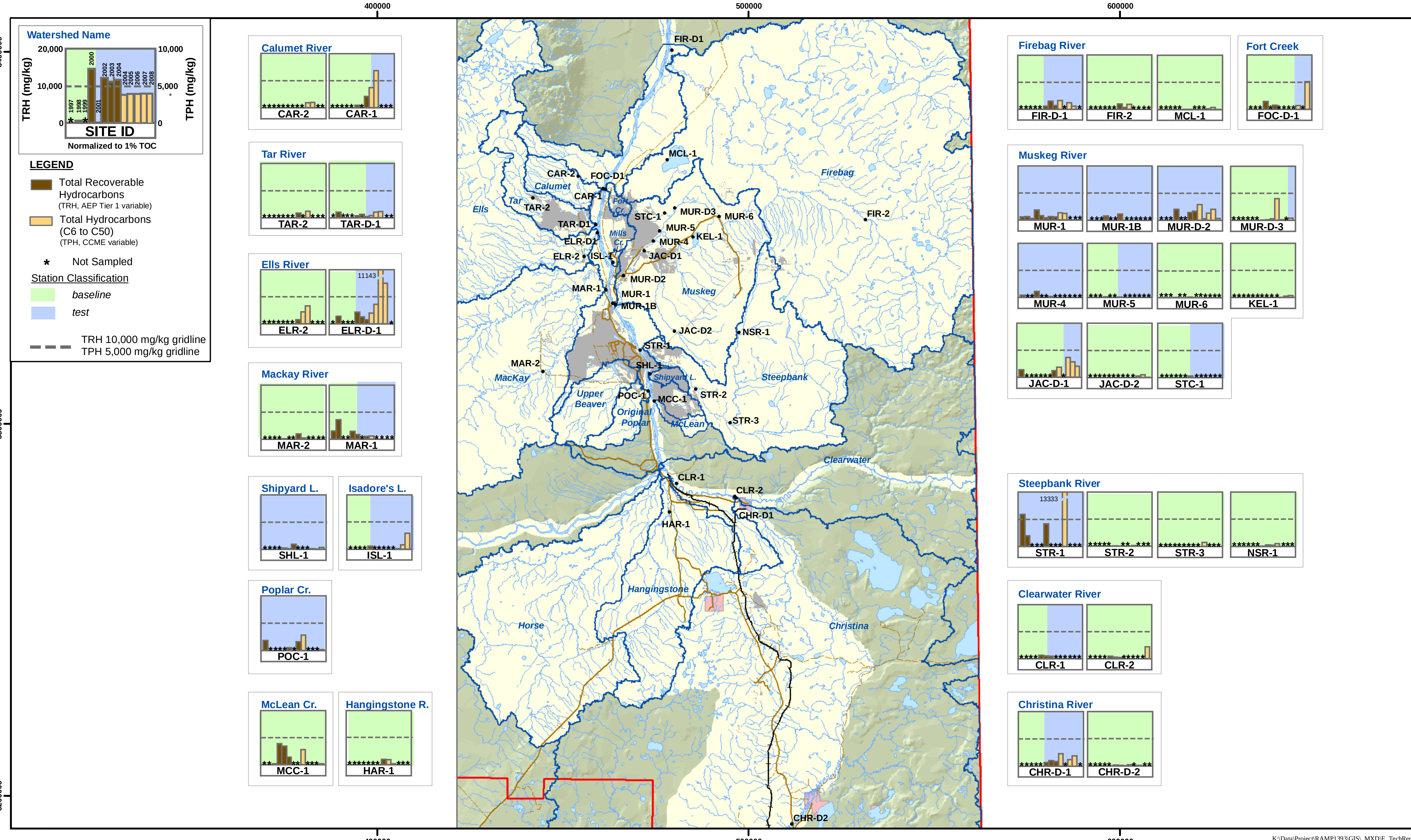


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Figure 6.3-8 Concentrations of total PAHs in sediments sampled by RAMP in tributaries to the Athabasca River, 1997 to 2008.

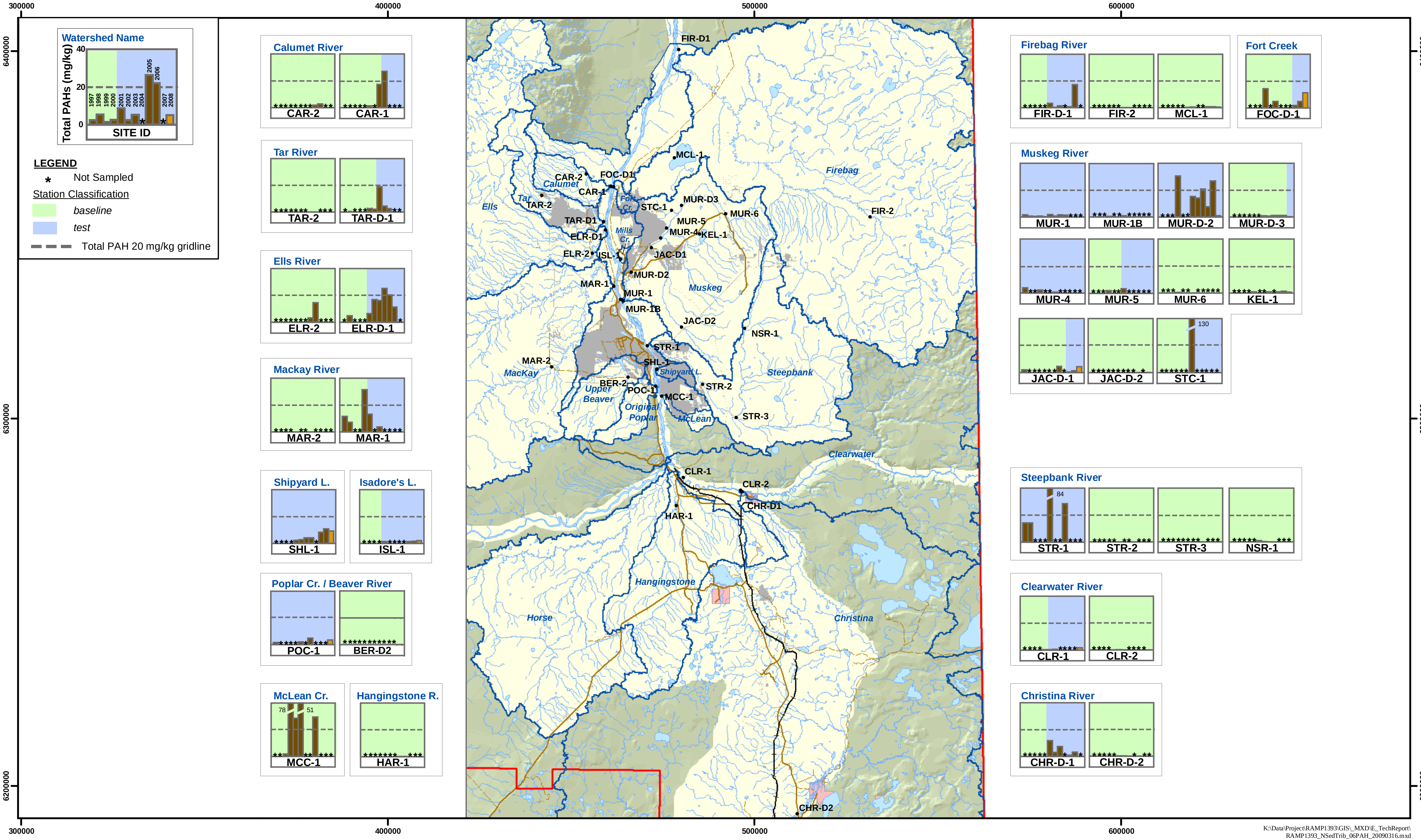


Figure 6.3-9 Carbon-normalized concentrations of total PAHs in sediments sampled by RAMP in tributaries to the Athabasca River, 1997 to 2008.

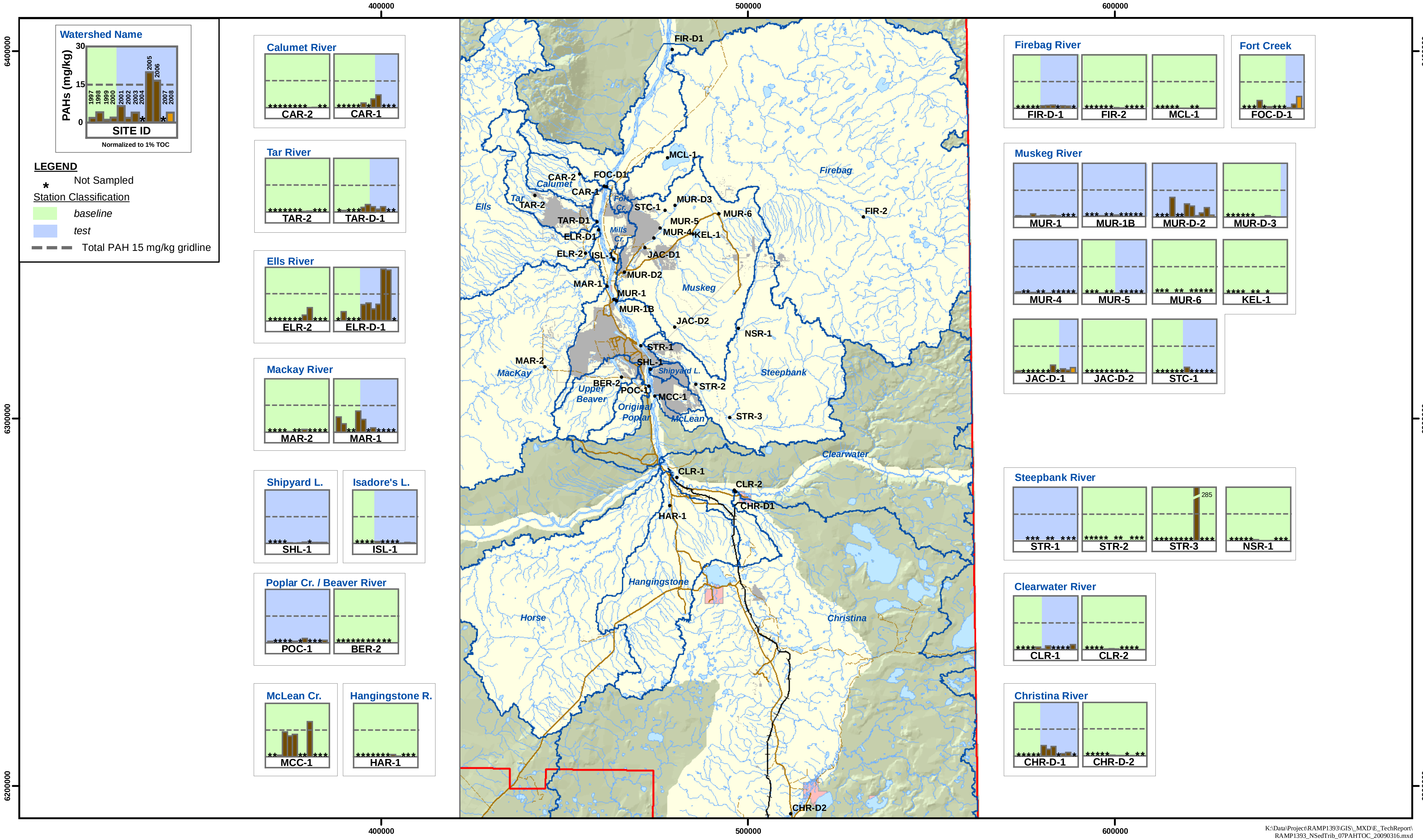
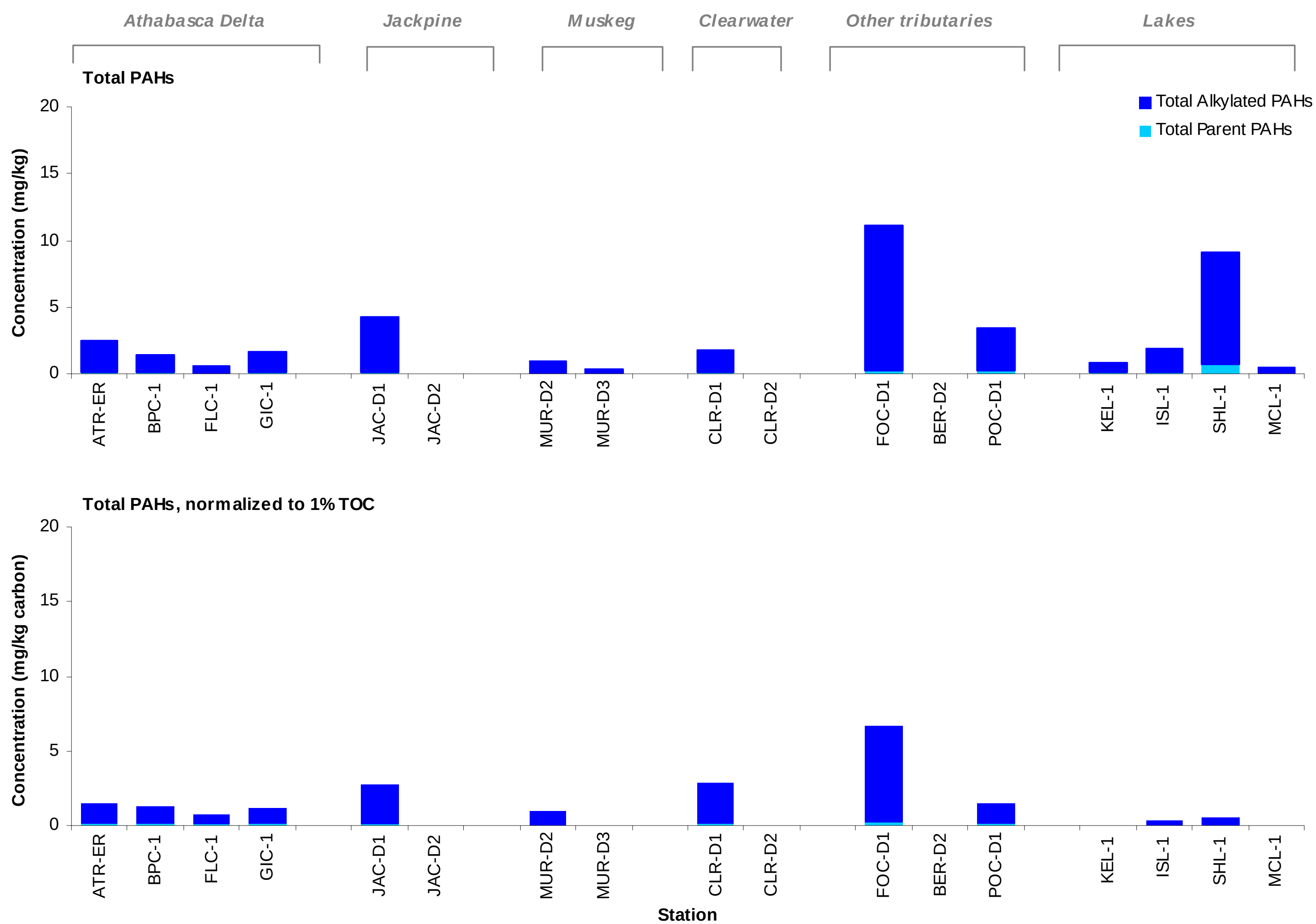


Figure 6.3-10 Total concentrations of parent and alkylated PAH in sediments collected by RAMP in 2008, including concentrations normalized to 1% organic carbon.



Metals-Arsenic

Arsenic has been frequently been measured by RAMP at concentrations near or above the CCME Interim Sediment Quality Guideline (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 *baseline*) and in Stanley Creek in 2003 (*test*). The observed concentration at Stanley Creek was the only one to have exceeded the CCME Probable-Effects Level (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 bio-magnify between trophic levels (ATSDR 2007). In 2008, highest arsenic concentrations were found in Isadore's Lake (7.3 mg/kg) and lower Poplar Creek (6.1 mg/kg). Generally, no consistent differences in sediment-borne arsenic were apparent between *baseline* and *test* 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 throughout the Athabasca-Slave-Mackenzie basin (DeBoer *et al.* 2007).

Metals-General

Most metals measured in RAMP sediments are highly inter-correlated. Principal component analysis (PCA) of metals data in sediments (2006 to 2008 data, n=48; Appendix F) found that the first derived principal component (total metals PC1) explained approximately 65% of the total variance in this dataset, and that most metals measured (i.e., 20 of 25 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 RAMP FSA.

Concentrations of total metals in sediments sampled by RAMP are presented in Figure 6.3-12, both in absolute concentrations and in concentrations normalized to percent fine sediments (i.e., silt and clay). For RAMP sediments collected from 2006 to 2008, total metals PC1 was strongly positively correlated with fine fractions of sediment (r_s=0.87 versus %-silt, and 0.77 versus %-clay), but strongly negatively correlated with %-sand (r_s=-0.84), indicating that metals concentrations were nearly always higher in fine rather than coarse sediments. Total metals concentrations in sediments in 2008 were relatively variable among stations (i.e., from below 30 to nearly 500 mg/kg), with highest concentrations in lower Poplar Creek (POC-D1, *test*) and Shipyard Lake (SHL-1, *test*). Following normalization to the percent fines, sediment metals exhibited much more consistent concentrations among stations, with highest concentrations in the lower Fort Creek (*test*) and lower Poplar Creek (POC-D1, *test*). Concentrations of metals in sediments were similar at all Athabasca delta stations, and were intermediate between highest and lowest concentrations observed in tributaries (Figure 6.3-12).

Figure 6.3-11 Concentrations of total arsenic in sediments sampled by RAMP in tributaries to the Athabasca River, 1997 to 2008.

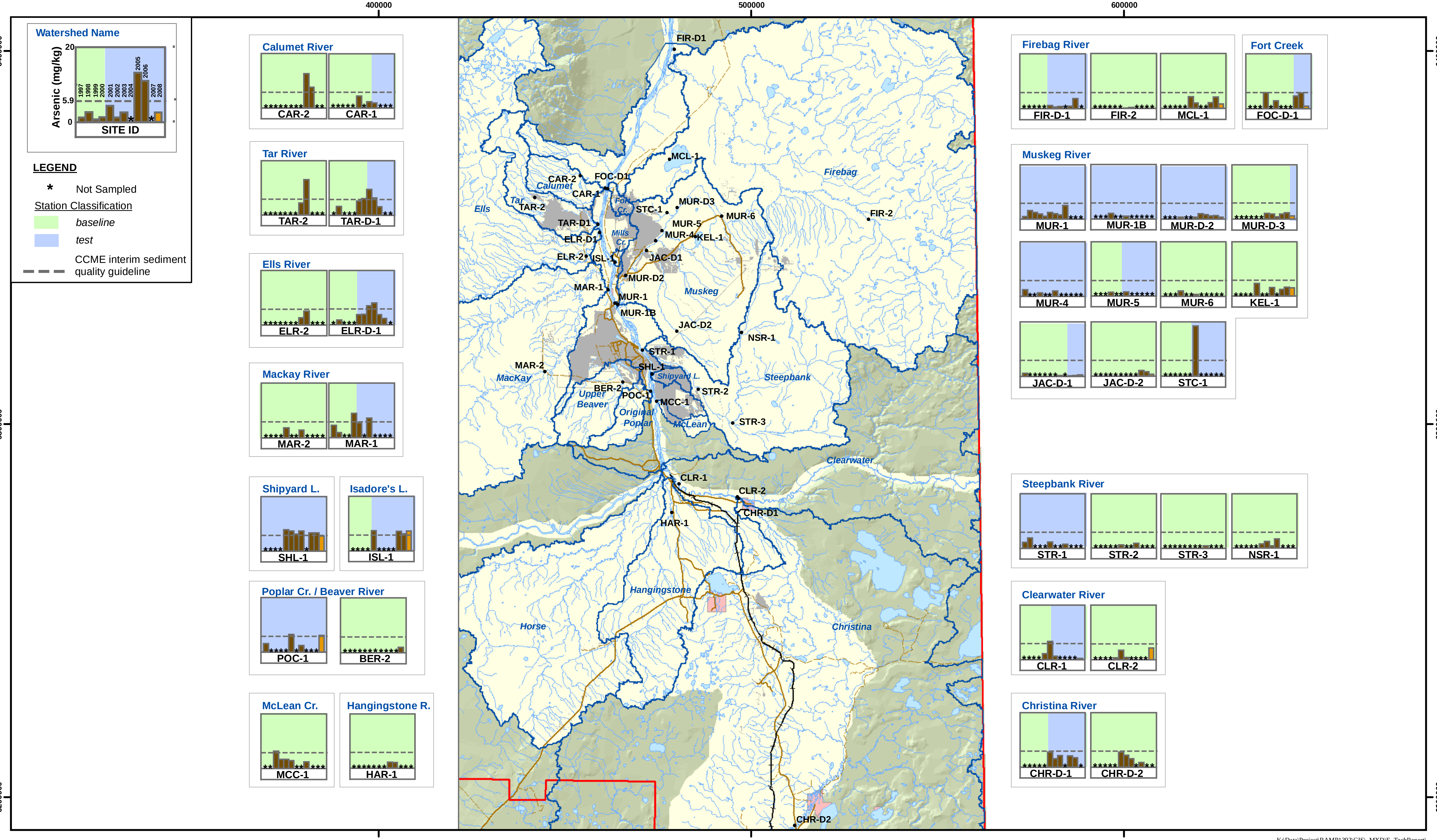


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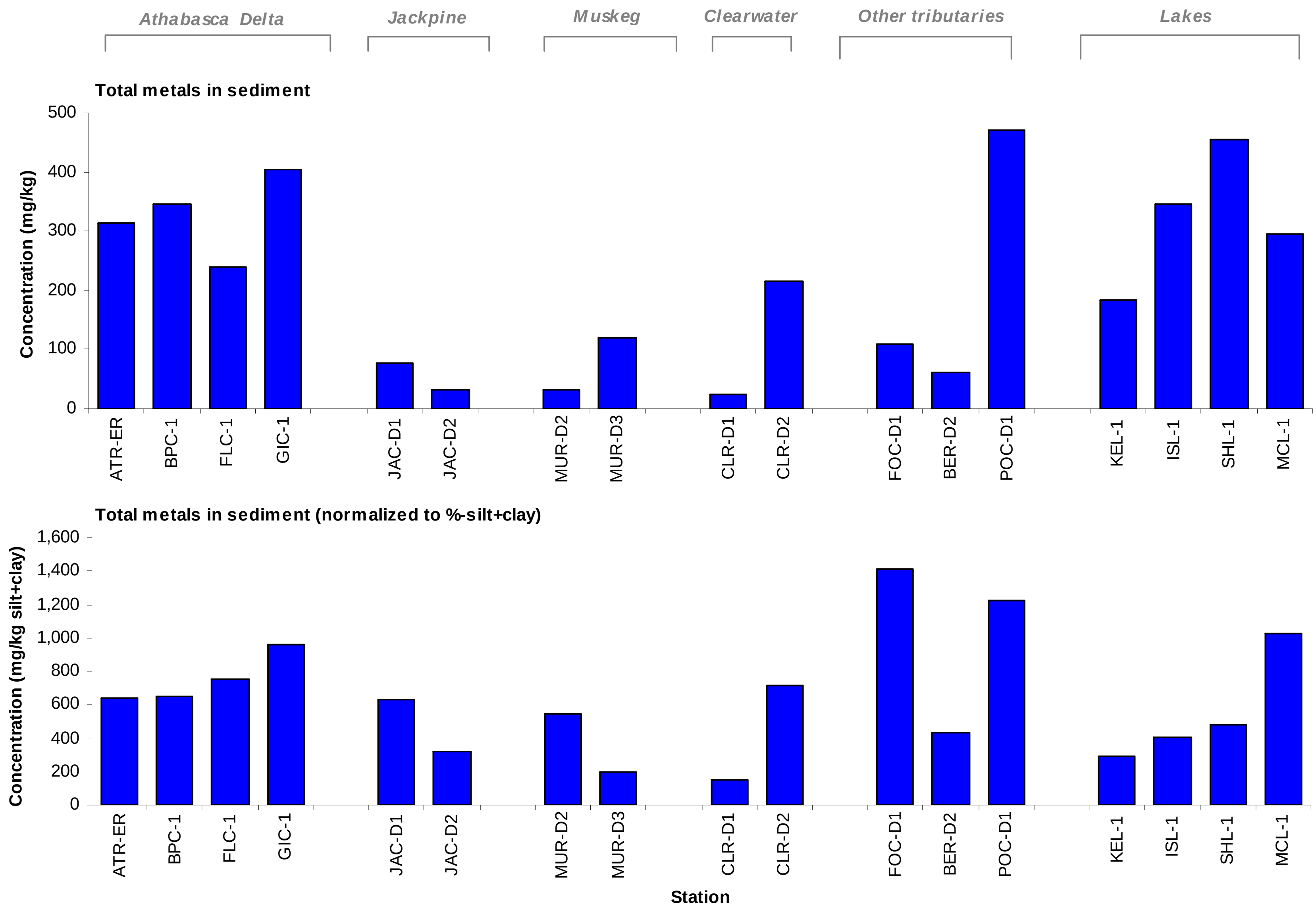
Figure 6.3-12 Total metals concentrations in sediments collected by RAMP in 2008, including concentrations normalized to fine-sediment fraction (i.e., %silt + clay).



Sediment Toxicity

Survival and growth of larvae of Chironomus tentans and amphipod Hyalella azteca in sediments collected from the RAMP FSA, relative to laboratory controls are shown in Figure 6.2-13 and Figure 6.3-14. For all sediments sampled, Chrionomus and Hyalella survival were within the range of laboratory baseline samples. At least three of five replicate sediment samples from several reaches showed growth rates of Chironomus or Hyalella that exceeded the range of laboratory controls: lower Jackpine Creek (JAC-D1, *test*, higher Chironomus growth), Isadore's Lake (ISL-1, *test*, higher Hyalella growth), and upper Beaver River (BER-D2, *baseline*, higher Hyalella growth). No sediments from either *test* or *baseline* sampling reaches exhibited lower growth than laboratory controls.

Figure 6.3-13 Survival and growth of *Chironomus* in sediments collected from the RAMP FSA in 2008, relative to laboratory baseline samples.

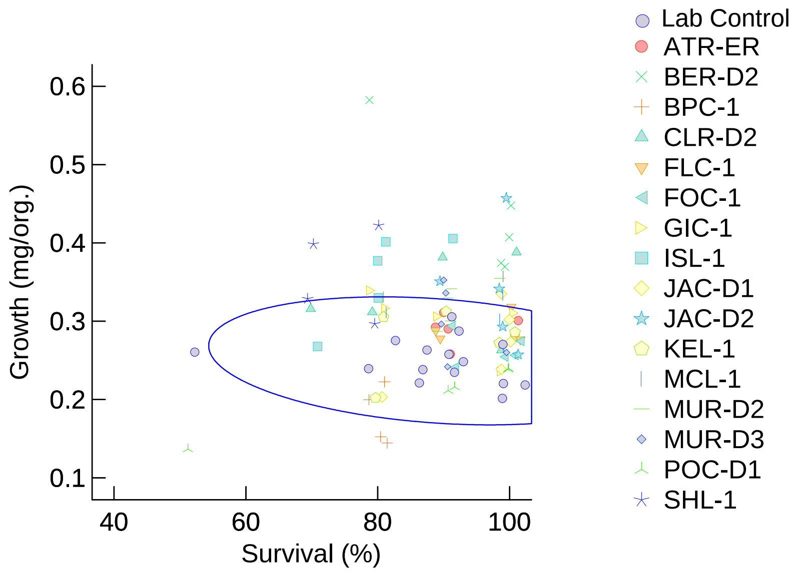
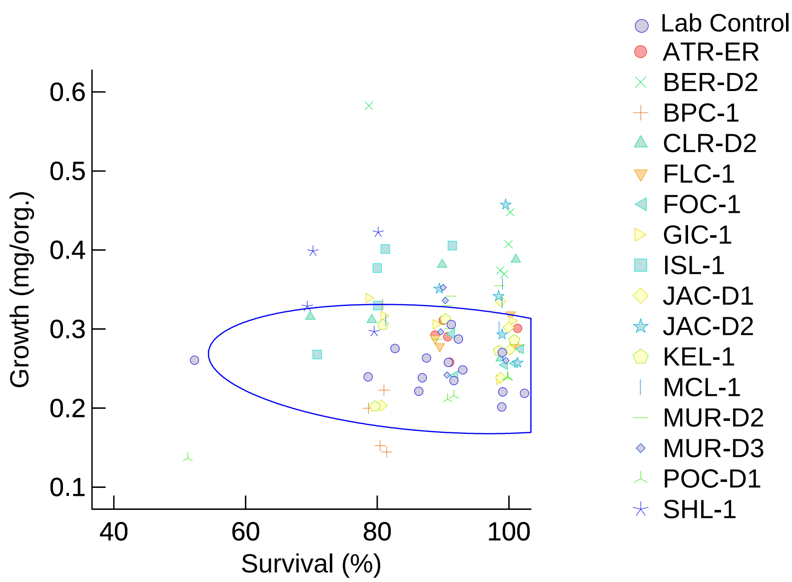


Figure 6.3-14 Survival and growth of *Hyalella azteca* in sediments collected from the RAMP FSA in 2008, relative to laboratory baseline samples.



6.3.2.2 Relationships Between Sediment Quality and Benthic Invertebrate Community Measurement Endpoints

Although several correlations between sediment quality and benthic invertebrate community measurement endpoints were statistically significant (i.e., $r_s>|0.284|$, Table 6.3-1), there were no moderate or strong correlations between sediment and benthic community endpoints, except a positive correlation between invertebrate abundance and total metals. When considered with observed correlations among sediment variables and in the context of invertebrate habitat requirements, several weak but statistically 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 metals, hydrocarbons and PAHs in sediments, these chemicals were themselves strongly correlated with fine, carbon-rich sediments (which were associated with abundance).

Attempts to examine more complex relationships between benthic abundance or richness and various sediment quality variables through stepwise, multiple regression (Appendix F) were confounded by the high level of autocorrelation among dependent (sediment quality) variables. When sediment-quality variables were summarized into six orthogonal principal components prior to multiple regression (to eliminate autocorrelation among dependant variables), no sediment-quality summary variables (PCs) showed strong correlation with either benthos abundance or richness (i.e., $R^2 \leq 0.16$).

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.

Table 6.3-1 Correlations (Spearman's coefficients) among benthic invertebrate community and sediment quality measurement endpoints, 2006 to 2008.

Sediment Endpoint	Benthic Invertebrate Endpoint				
	Abundance	Taxa Richness	Simpson's Diversity	Evenness	%-EPT
Physical Variables					
% Clay	*0.35*	*-0.31*	-0.15	-0.21	-0.29
% Sand	*-0.43*	0.22	0.14	0.19	*0.32*
% Silt	*0.47*	-0.17	-0.15	-0.20	*-0.37*
Total organic carbon	*0.38*	0.04	0.05	0.08	0.00
Inorganic carbon	*0.48*	-0.06	-0.12	-0.09	-0.23
Total carbon	*0.43*	0.02	0.06	0.08	-0.02
Hydrocarbons & PAHs					
CCME F2 (C10-C16)	-0.08	0.02	-0.13	-0.09	-0.04
CCME F3 (C16-C34)	*0.29*	0.04	0.01	0.07	-0.01
CCME F4 (C34-C50)	0.14	0.00	-0.01	0.04	-0.02
CCME TPH (C6-C50)	0.20	0.06	0.09	0.15	0.08
Total PAHs	0.28	0.04	0.07	0.11	-0.10
Naphthalene	*0.48*	-0.09	-0.07	-0.07	-0.08
Retene	*0.35*	0.13	0.24	0.31	0.17
Total dibenzothipenes	0.23	-0.04	-0.07	-0.03	-0.16
Metals					
Total metals (PC1)	***0.53***	-0.28	-0.22	-0.26	*-0.41*

n=48; Critical value of r_s=|0.283|; values in italics indicate significant correlation; values in bold indicate moderate correlation (i.e., $|0.50|>r_s>|0.75|$).

6.3.2.3 Summary

Sediments in the RAMP FSA 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 *baseline* and *test* stations, or any relationships between sediment chemistry and composition of benthic invertebrate communities.

6.4 FISH POPULATIONS

The 2008 RAMP Fish Population component included fish inventories on the Athabasca and Clearwater rivers, chemical analyses of fish tissue collected from the Athabasca River, and chemical analyses of fish tissue collected from the following regional lakes: Gardiner and Big Island lakes.

The intention of this section is to provide a regional context for measurement endpoints in fish populations monitored during programs completed in 2008 in relation to programs conducted in waterbodies during historical RAMP fish programs (1997-2007), surveys completed prior to 1997 (i.e., prior to RAMP) and regional studies completed in and surrounding the RAMP FSA. Endpoints, which include mercury concentrations in fish tissue, condition factor, species evenness and catch-per-unit-effort (CPUE), were evaluated temporally and spatially, particularly as they relate to oil sands development.

To provide a regional context for the Fish Population component, Figure 6.4-1 displays the 2008 fish tissue results relative to mercury concentrations in fish tissue from waterbodies not currently downstream of focal projects and from previous RAMP sampling (DFO 1984, Grey *et al.* 1995, Golder 2004, RAMP 2003, RAMP 2004, RAMP 2008). 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). In addition, given that the inventory program was not conducted on a regional scale, the 2008 results were compared with fish inventory programs conducted from 1987 to 2007 on the Athabasca River and from 2003 to 2007 on the Clearwater River during spring, summer and fall to provide an overall assessment of measurement endpoints (Figure 6.4-2).

6.4.1 Mercury in Fish Tissue

The RAMP fish tissue program collected samples from Key Indicator Resource (KIR) species in Gardiner and Big Island lakes (walleye, lake whitefish and northern pike), and the Athabasca River (walleye and lake whitefish) in 2008. 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) (Health and Welfare Canada 1979, as cited in Lockhart *et al.* [2005]). 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 2004, 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).

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, 1989-2008.

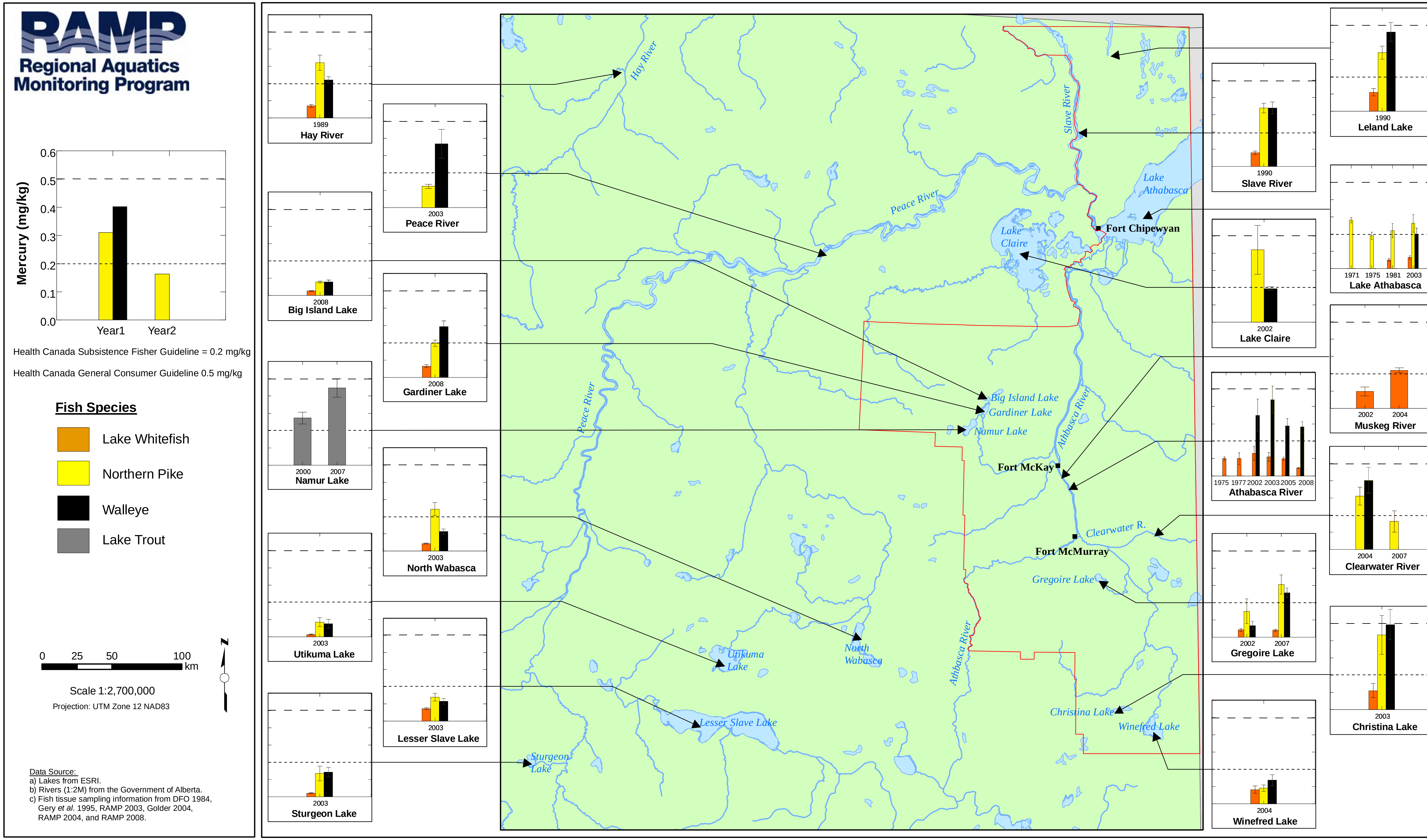


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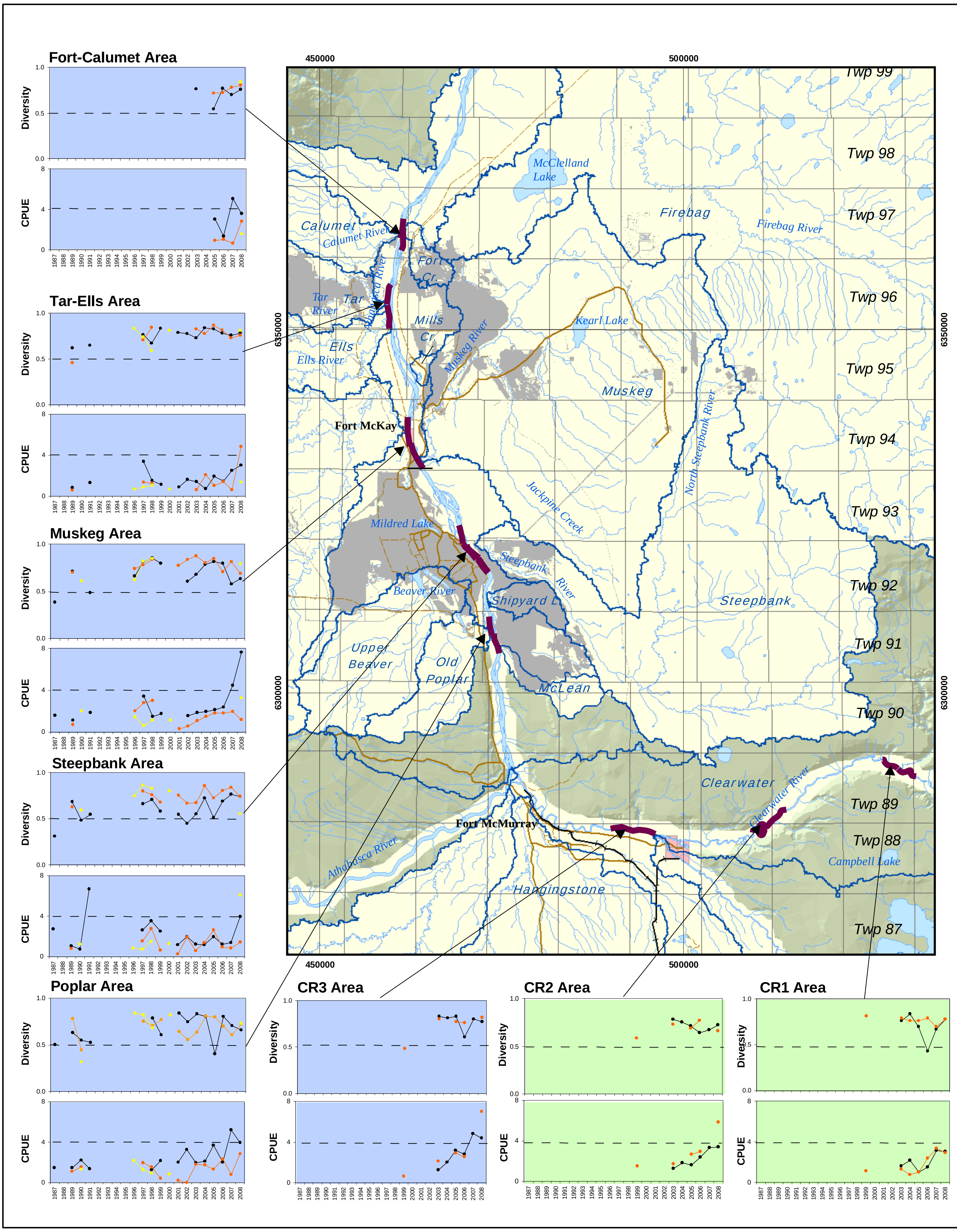
Figure 6.4-2 Species diversity and catch per unit effort (CPUE) of large-bodied fish captured during the spring, summer, and fall fish inventory programs in the Athabasca and Clearwater rivers, 1987-2008.



LEGEND

- Lake/Pond
- River/Stream
- Major Road
- Secondary Road
- Railway
- First Nations Reserve
- RAMP Regional Study Area
- RAMP Focus Study Area
- Land Change Area as of 2008
- Watershed Boundary
- Fish Sampling Reaches

Sampling Season
- Spring
- Summer
- Fall

Site Classification
- *baseline*
- *test*

RAMP
Regional Aquatics Monitoring Program

Data Sources:
a) Lake/Pond, River/Stream, Major Road, Secondary Road, Railway, First Nation Reserve, and Hillshade from 1:250,000 National Topographic Data Base (NTDB).
b) Watershed Boundaries from the Cumulative Management Association (CEMA).
c) Land Change Areas Delineated from 10m SPOT-5 (May, July and August 2008) Multispectral Imagery.

Township and Range designations are relative to W4M.

0 5 10 20 km
Scale 1:575,000
Projection: UTM Zone 12 NAD83

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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 several species of interest to the communities in this area (i.e., northern pike and walleye) are piscivorous species, mercury levels are higher than in species lower on the food chain (i.e., lake whitefish). Factors affecting the amount of methyl-mercury in a fish include size, trophic status, and sediment and water chemistry (INAC 2003).

Results for lake whitefish collected from the Athabasca River indicated that 0% of fish exceeded the Health Canada subsistence fisher or general consumer guideline. Similarly, there were no lake whitefish in Big Island Lake or Gardiner Lake that exceeded the Health Canada guidelines. Results for walleye collected from the Athabasca River indicated that 62% (16 of 16) of fish exceeded the Health Canada subsistence fish guideline and of that 62%, 3% (3 of 16) of fish exceeded the general consumer guideline (or the guideline for the commercial sale of fish). Similarly, 52% (16 of 31) of fish exceeded the subsistence fisher guideline in Gardiner Lake and of that 52%, 50% (8 of 16) exceeded the general consumer guideline. In Big Island Lake, there was only one walleye that exceeded the subsistence fisher guideline; all other fish had mercury levels below any guideline values. Results for northern pike collected from Gardiner Lake indicated that 42% (5 of 12) of fish exceeded the Health Canada subsistence fisher guideline but no fish had mercury concentrations exceeding the general consumer guideline; in Big Island Lake, there were no guideline exceedances in any northern pike. Given lake whitefish is lower in trophic status than walleye and northern pike, it is expected that mercury concentrations in this species are lower.

Regionally (Figure 6.4-1), eleven of the seventeen waterbodies (64%) sampled for walleye over time showed an exceedance of the subsistence fisher mercury concentration guideline (0.2 mg/kg) but none exceeded the general consumer guideline (0.5 mg/kg); ten of the seventeen waterbodies (59%) 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 northern pike and walleye 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; the maximum mean mercury concentration in lake whitefish was measured in the Athabasca River in 2002, but since then, concentrations have decreased. Mercury concentrations in Lake Athabasca lake whitefish and northern pike measured in the early 1970s, prior to major oil sands development were consistent or higher than mercury concentrations in fish in 2008 (DFO 1984) (Figure 6.4-1).

For available temporal data, the mean mercury concentration standardized to fish weight in lake whitefish in the Athabasca River has decreased over time for both males and females, whereas the mean concentration in walleye from the Athabasca River has decreased over time in males and remained fairly consistent for females. On a spatial scale, in 2008, mercury results from the Athabasca River, downstream of oil sands development were similar to results from the two regional lakes, outside of oil sands development, for lake whitefish and walleye. 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 (Ullrich *et al*. 2001). Sampled waterbodies shown in Figure 6.4-1 fall within this natural range of mercury concentration.

As mentioned in the focal projects updates in the RAMP FSA in Section 2.2, there is muskeg dewatering and land clearing during oil sands development, which could lead to increased levels of mercury in watercourses within the developed areas (Grigal 2003), resulting in increases in mercury in muscle tissue of fish in these watercourses. However, the temporal trends of mercury data in walleye and lake whitefish in the Athabasca River do not indicate an increase coinciding with increased development. Given 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 2008, the inventory program was conducted on the Athabasca and Clearwater rivers in spring, summer (Athabasca) 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 CPUE), condition of fish and species evenness. CPUE was calculated per one hundred seconds of electrofishing for all large-bodied species combined in each sampling area.

CPUE in all areas of the Athabasca River has fluctuated over time with no clear decreasing or increasing trends. There are no *baseline* reaches on the Athabasca, the fish inventory is conducted in areas of the river downstream of development (*test* reaches). Therefore, comparisons of change in fish populations relative to oil sands development can only be made temporally, comparing CPUE in years prior to RAMP (1987-1996) versus CPUE from 1997-2008. CPUE in 2008 either fell within or exceeded historical ranges in each area where a fish inventory is conducted in spring, summer and fall (Figure 6.4-2).

CPUE in all areas of the Clearwater River has generally shown increasing trends over time (Figure 6.4-2). The two *baseline* reaches (CR1 and CR2) were compared to CPUE in the *test* reach, CR3, to assess changes. For both spring and fall, the CPUE in the *test* reach is higher than the CPUE in both *baseline* reaches, indicating that the relative abundance of fish populations in an area downstream of oil sands development does not show negative trends.

Species diversity in all sampling areas of the Athabasca River showed large fluctuations without any increasing or decreasing trends over time. With the exception of the Steepbank area in summer, species diversity in 2008 fell within the historical range (i.e., prior to 1997). Species diversity in all sampling areas of the Clearwater River showed minor fluctuations over time and was generally consistent across years and areas. Diversity in the lower *test* reach was similar to diversity in the upper *baseline* reaches.

6.5 ACID-SENSITIVE LAKES

This section presents the results of the Acid-Sensitive Lakes (ASL) component of RAMP for 2008. As the lakes are located across all the various watersheds, the ASL component is presented only as part of the Regional Synthesis. A general description of the 50 RAMP lakes is provided, as well as three primary analyses of the RAMP ASL lake dataset to examine changes or trends in measurement endpoints indicative of potential acidification of these lakes:

- **Between-Year Comparison of ASL Measurement Endpoints** An Analysis of Variance (ANOVA) to determine whether there have been any significant changes in the mean values of the ASL measurement endpoints over the 10 years of monitoring data available for the 50 lakes[1];

- **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 using the Mann-Kendall test and Shewhart control charts.

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 2008 data. Summary statistics were calculated on the updated dataset that now includes nine or ten years of data on the 50 lakes. Using multivariate principal components analysis and Piper plots, the ASL lakes were categorized and grouped according to lake chemistry;
- The database on trace metal concentrations in the RAMP ASL lakes was updated and summarized statistically. 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 2008 data and summary statistics were calculated. Due to the high seasonal variability in many endpoint parameters in the ASL lakes, the importance of sampling at the same point in the hydrological and biological cycles of the lakes was stressed.

6.5.1 General Characteristics of the 50 RAMP Lakes-2008

The chemical variables measured in the 50 RAMP lakes from 1999 to 2008 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 1,802 µeq/L with a median of 193 µeq/L. Concentrations of sulphate are relatively low and range from non-detectable to 16.7 mg/L with a median concentration of 1.15 mg/L.

By conventional standards, most of the RAMP lakes are considered humic with a median dissolved organic carbon (DOC) concentration of 21.4 mg/L (Korteleinen *et al.* 1989, Forsius 1992, Driscoll *et al.* 1991). Over 60% of the RAMP ASL lakes are considered to be highly sensitive or moderately sensitive to acidification by classifications based on pH, Gran alkalinity and Critical Load (Section 3.5.1). In general, nitrates are quite low (median 3 µg/L), although some individual lakes may have nitrate concentrations two orders of magnitude greater than the median. Total phosphorus covers a broad range from 3.6 µg/L to 341 µg/L with a median of 39.9 µg/L.

[1] Not all 50 lakes were sampled in every year from 1999 to 2008; see Table 3.5-4 for sampling years in each lake.

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

Variable	Mean		Median		Minimum		Maximum		5th Percentile	95th Percentile
	1999-2008	2008	1999-2008	2008	1999-2008	2008	1999-2008	2008	2008	2008
Lab pH	6.55	6.55	6.76	6.66	3.97	4.12	9.46	8.34	4.75	7.91
Total Alkalinity (µeq/L)	311	316	214	208	0.00	0.00	1784	1727	21.2	1085
Gran Alkalinity (µeq/L)	298	295	193	187	-57.2	-44.2	1802	1720	-18.1	1073
Specific Cond. (µS/cm)	45.6	44.2	30.3	30.0	10.5	13.0	180	175	14.5	106.8
Total Dissolved Solids (mg/L)	67.4	71.1	61.3	59.0	0.02	0.02	219	214	28.2	142
Turbidity (NTU)	3.92	3.99	1.88	2.17	0.321	0.321	53.0	27.7	0.701	11.8
Colour (TCU)	151	163	123	133	8.00	9.30	948	476	21.1	384
Sodium (mg/L)	1.98	2.36	1.33	1.63	0.184	0.640	10.4	9.70	0.718	6.41
Potassium (mg/L)	0.518	0.578	0.44	0.43	0.000	0.120	2.40	2.11	0.17	1.232
Calcium (mg/L)	5.61	5.34	4.63	4.66	0.20	0.38	32.2	20.1	1.10	13.3
Magnesium (mg/L)	1.79	1.71	1.40	1.28	0.114	0.270	13.6	6.08	0.357	4.34
Bicarbonate (mg/L)	18.9	19.5	13.1	12.7	0.92	0.00	109	105	1.30	66.2
Chloride (mg/L)	0.363	0.334	0.196	0.220	0.02	0.07	2.64	2.39	0.074	1.27
Sulphate (mg/L)	2.28	2.71	1.15	1.30	0.175	0.430	16.7	16.4	0.554	11.2
Total Dissolved Nitrogen (µg/L)	848	808	698	740	105	332	2891	2270	394	1572
Ammonia (µg/L)	39.3	15.0	15	10	0.35	1.00	1509	69	1	39.6
Nitrate + Nitrite (µg/L)	20.5	13.5	3	3	0.02	0.5	733	271	0.7	30.8
Total Phosphate (µg/L)	55.3	46.4	39.9	33.9	3.60	6.00	341	166	12.1	121.4
Dissolved Phosphate (µg/L)	20.3	19.7	11.5	11.0	1.20	4.00	156	120	5	65
Dissolved Inorganic Carbon (mg/L)	3.17	3.18	1.98	1.90	0.0269	0.100	20.3	19.4	0.24	11.0
Dissolved Organic Carbon (mg/L)	22.8	23.1	21.4	20.1	6.80	7.30	81.2	52.1	9.88	39.5
Chlorophyll *a* (µg/L)	20.5	21.6	9.61	9.55	0.60	0.86	371	125	2.30	65.8
Iron (mg/L)	0.37	0.60	0.18	0.38	0.001	0.01	3.88	3.65	0.03	2.06
Total Nitrogen (µg/L)	1237	983	984	800	274	324	6558	4300	415	2244
Total Kjeldahl Nitrogen (µg/L)	1216	969	958	759	273	323	6552	4296	412	2243
Sum base cations (µeq/L)	556	535	439	428	38.19	78.3	2291	2005	143	1277
Dissolved Aluminum (µg/L)	70.84	70.2	24.90	38.8	0.10	0.47	681	422	0.95	284.8

Shaded variables are measurement endpoints for the ASL program.

Lakes having "unusual" chemistry were identified in the 2008 monitoring data as those with values below or above the 5th and 95th percentile for the three measurement endpoints of pH, Gran alkalinity, and DOC (Table 6.5-2). These lakes were in many cases the same lakes identified in previous years (e.g., RAMP 2008). Three lakes (168/A21, 287/25 and Clayton Lake) had very low or negative levels of Gran alkalinity. All three lakes are found in upland regions, two in the Stony Mountains and one in the Birch Mountains. These lakes were also associated with the lowest values of pH. The highest values of Gran alkalinity and buffering capacities in the RAMP ASL lakes were found in Lakes 270/4, 271/6 and Kearl Lake, located northeast of Fort McMurray. These lakes also had the highest values of pH in all ASL lakes. The lowest levels of DOC were found in two Birch Mountains Lakes (Namur and Legend lakes) and one shield lake (Weekes Lake). The highest concentrations of DOC were found in Lake 165/A42, and Lake 223 both in the West of Fort McMurray sub-region.

Table 6.5-2 RAMP ASL lakes with chemical characteristics either below the 5th or above the 95th percentile in the 2008 data.

Lake	Region	pH	Gran Alkalinity (μeq/L)	DOC (mg/L)
5th percentile, 2008		**4.75**	**-18.1**	**9.88**
95th percentile, 2008		**7.91**	**1073**	**39.5**
168 (A21)	Stony Mountains	**4.6**	**-33.6**	21.9
287 (25)	Stony Mountains	**4.67**	**-44.2**	13.9
Clayton Lake 448 (L29)	Birch Mountains	**4.12**	**-35.6**	15.9
Namur Lake 436 (L18)	Birch Mountains	6.86	429	**7.30**
Legend Lake 444 (L25)	Birch Mountains	6.51	151	**7.70**
Weekes Lake 118 (L107)	Canadian Shield	7.61	476	**9.00**
270 (4)	Northeast of Fort McMurray	**8.11**	**1341**	32.6
271 (6)	Northeast of Fort McMurray	**8.34**	**1208**	14.9
Kearl L. 418	Northeast of Fort McMurray	**8.32**	**1720**	35.5
165 (A42)	West of Fort McMurray	6.69	313	**44.7**
223 (P94)	West of Fort McMurray	7.37	666	**52.1**

Blue values represent those values below the 5th percentile for that variable in the 2008 data.

Red values represent those values above the 95th percentile for that variable in the 2008 data.

As indicated in previous RAMP reports (RAMP 2005, 2006, 2007, 2008), 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 set of RAMP ASL lakes are those lakes that are simultaneously high in pH and high in DOC (e.g., Kearl Lake). Most coloured (high DOC) lakes are typically low in pH (Korteleinen *et al*. 1989). The other variables characterizing the chemistry of the ASL lakes are discussed in Appendix H.

6.5.2 Between-Year Comparisons of ASL Measurement Endpoints

An Analysis of Variance (ANOVA) was performed in order to determine whether there have been any significant changes in the ASL measurement endpoints over the seven years when all 50 lakes were sampled consistently (2002-2008).

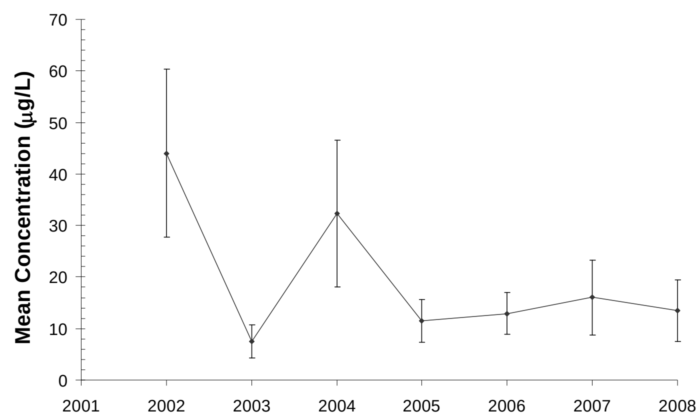
Nitrate was the only ASL measurement endpoint to show a significant change over the seven years (Kruskal-Wallis non-parametric test). The mean concentration of nitrate fell from a peak of 44.0 µg/L in 2002 to 7.5 µg/L in 2003, increased to 32.3 µg/L in 2004 and fell to 11-16 µg/L between 2005 and 2008 (Table 6.5-3;

Figure 6.5-1). Nitrates are extremely variable in the ASL lakes with a coefficient of variation between 200%-300% over the seven years of monitoring (Table 6.5-3). The extreme variability in this measurement endpoint makes it very difficult to detect a change in nitrates that would indicate lake acidification. Overall, there is no evidence of an increase in nitrates between 2002 and 2008 as expected under an acidification scenario triggered by nitrogen emissions from oil sands developments. There is, therefore, no indication that acidification is occurring from nitrogen deposition.

Table 6.5-3 Summary of nitrate concentrations in the RAMP ASL lakes, 2002-2008.

Value	2002	2003	2004	2005	2006	2007	2008
N	49	50	50	49	48	48	49
Mean (mg/L)	44.0	7.50	32.3	11.5	12.9	16.1	13.5
Median (mg/L)	5.26	0.5	0.995	2.96	5.44	2	3
SD	114	22.3	101	28.7	28.1	50.6	41.8
CV (%)	260	298	313	250	217	315	309

Figure 6.5-1 Mean concentration of nitrate 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 2008 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 2005). 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 and 2007, 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 (^{2}H) in each lake. Table 6.5-4 presents the two estimates of runoff and critical loads of acidity between 2002 and 2008. The isotopically-derived runoff values were greater than the hydrometrically derived values in 16 lakes, lower in 31 lakes and identical in two lakes. The greatest discrepancies were observed for lakes having the highest rates of runoff.

Using the hydrometrically derived runoff, the critical loads in 2008 ranged from -0.088 keq H^+/ha/y to 1.551 keq H^+/ha/y with a median of 0.308 keq H^+/ha/y (Table 6.5-5). Using the isotopically derived runoff, critical loads ranged from -0.122 keq H^+/ha/y to 2.137 keq H^+/ha/y with a median CL of 0.302 keq H^+/ha/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 2008 for the two methods (hydrometric/isotopic) in the six sub-regions were calculated as follows:

- Stony Mountains: 0.026/0.020 keq H^+/ha/y;
- West of Fort McMurray: 0.544/0.238 keq H^+/ha/y;
- North-East of Fort McMurray: 0.510/0.501 keq H^+/ha/y;
- Birch Mountains: 0.280/0.226 keq H^+/ha/y;
- Canadian Shield: 0.295/0.417 keq H^+/ha/y; and
- Caribou Mountains: 0.174/0.568 keq H^+/ha/y.

Low critical loads observed in the upland regions (the Stony Mountains, the Birch Mountains and the Caribou Mountains) and in the Canadian Shield are consistent with findings in previous RAMP reports (RAMP 2005, 2006, 2007, 2008). Negative critical loads were observed in many of the lakes, especially in the Stony Mountains sub-region. By the critical load criterion, these lakes are 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 Total E&P Joslyn North Mine Project EIA for the Planned Development Case (Deer Creek Energy 2006). 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) (Foster *et al.* 2001).

Lakes with modelled PAI values greater than the critical load are identified in Table 6.5-4. The percentage of such lakes ranged from a low of 33.3% (16 of 49 lakes) in 2005 to a high of 48.0% (24 of 50 lakes) in 2006 (Table 6.5-5). In 2008, the use of the isotopically derived runoff in the calculations resulted in two additional lakes in which the PAI exceeds the critical load compared to results using the hydrometrically derived runoff values. Generally, the number of lakes with PAI values exceeding the critical load was higher when the isotopically derived runoff was used in the calculations.

Table 6.5-4 Critical loads of acidity in the RAMP ASL Lakes, 2002 to 2008.

NOX-SOX GIS No.	Original RAMP Designation	Runoff (Hydro) (m^3/s)	Runoff Isotopic m^3/s	Mean pH	Mean DOC (mg/L)	Mean Gran Alk. (µeq/L)	2002	2003	2004	2005	2006 Hydro	2006 Isotopic	2007 Hydro	2007 Isotopic	2008 Hydro	2008 Isotopic	PAI 2006[1]
Stony Mountains Subregion																	
168	A21	0.0404	0.0474	4.91	20.4	28.3	-0.089	-0.079	-0.087	-0.118	-0.081	-0.096	-0.070	-0.082	-0.069	-0.081	0.186
169	A24	0.0264	0.0323	4.67	20.5	-4.7	-0.124	-0.071	-0.205	-0.132	-0.104	-0.127	-0.033	-0.040	-0.083	-0.102	0.177
170	A26	0.0238	0.0140	5.42	15.0	-2.1	-0.030	-0.028	-0.036	-0.047	-0.045	-0.027	-0.012	-0.007	0.003	0.002	0.186
167	A29	0.0131	0.0150	5.76	16.0	19.8	-0.028	-0.019	-0.002	0.004	0.033	0.038	-0.002	-0.002	-0.033	-0.038	0.145
166	A86	0.0147	0.0093	6.53	17.2	125.6	0.094	0.101	0.109	0.110	0.100	0.063	0.104	0.066	0.141	0.089	0.117
287	25	0.0223	0.0035	5.00	15.7	-13.8	-0.056	-0.055	-0.075	-0.077	-0.068	-0.103	-0.032	-0.048	-0.040	-0.061	0.179
289	27	0.0216	0.0275	6.47	13.0	66.9	0.019	0.029	0.035	0.035	0.030	0.038	0.044	0.055	0.030	0.038	0.175
290	28	0.0124	0.0130	5.74	20.5	38.3	0.004	0.033	-0.008	-0.007	0.012	0.012	-0.014	-0.015	0.003	0.003	0.181
342	82	0.0291	0.0085	6.59	26.5	159	0.208	0.181	0.165	0.125	0.182	0.053	0.122	0.036	0.090	0.026	0.120
354	94	0.0162	0.0240	7.09	24.2	355	0.322	0.225	0.213	0.226	0.179	0.265	0.186	0.275	0.220	0.326	0.141
West of Fort McMurray Subregion																	
165	A42	0.0639	0.0245	6.91	45.8	324.6	0.388	0.373	0.553	0.706	0.455	0.175	0.359	0.138	0.419	0.161	0.121
171	A47	0.0115	0.0044	6.40	21.1	145.1	0.217	0.167	0.152	0.253	0.207	0.079	0.168	0.064	0.332	0.127	0.120
172	A59	0.1781	0.0339	5.20	34.0	39.4	0.038	0.001	0.002	-0.023	-0.075	-0.014	-0.061	-0.012	0.046	0.009	0.076
223	P94	0.0019	0.0003	7.33	41.7	763	1.120	1.031	1.054	1.399	1.004	0.153	0.829	0.126	0.996	0.152	0.258
225	P96	0.0034	0.0027	7.33	32.7	616	0.745	0.595	0.666	0.825	0.669	0.539	0.506	0.408	0.574	0.462	0.238
226	P97	0.0057	0.0056	6.88	32.0	325	0.328	0.346	0.266	1.377	0.238	0.235	0.277	0.273	0.373	0.368	0.353
227	P98	0.0070	0.0025	7.31	32.4	604	0.969	0.956	0.917	0.462	1.042	0.378	0.857	0.311	1.071	0.389	0.307
267	1	0.1182	0.0138	7.71	23.4	748	1.055	1.024	0.994	1.091	0.732	0.086	0.630	0.074	NA	NA	0.214
Northeast of Fort McMurray Subregion																	
452	L4	0.0920	0.0675	5.79	25.5	75.8	0.070	0.070	0.078	0.143	0.073	0.053	0.095	0.070	0.100	0.073	0.222
470	L7	0.1010	0.0376	6.42	29.2	158	0.170	0.190	0.141	0.307	0.707	0.263	0.357	0.133	0.238	0.089	0.646
471	L8	0.0450	0.0257	6.85	21.2	334	0.528	0.622	0.527	0.659	0.340	0.194	0.527	0.301	0.567	0.324	0.607
400	L39	0.0501	0.0855	6.79	13.8	174	0.157	0.157	0.144	0.073	0.316	0.539	0.251	0.428	0.204	0.348	0.085
268	E15	0.0809	0.0472	7.07	39.0	364	0.520	0.465	0.400		0.092	0.054	0.421	0.245	0.509	0.297	0.206
182	P23	0.0296	0.0254	7.55	17.8	656	0.294	1.084	2.017	2.008	0.443	0.379	1.333	1.143	0.199	0.171	0.250
185	P27	0.0172	0.0175	5.30	31.4	64.6	0.035	0.017	-0.095	0.233	-0.030	-0.030	0.035	0.035	0.041	0.041	0.220
209	P7	0.0072	0.0095	6.14	23.9	137	0.141	0.163	0.112	0.089	0.109	0.145	0.143	0.189	0.311	0.411	0.195
270	4	0.0411	0.0371	8.22	32.4	1392	1.382	1.318	1.408	1.705	1.037	0.936	0.904	0.816	1.021	0.922	0.181
271	6	0.0485	0.0388	8.51	25.9	1329	1.293	1.449	1.931	1.369	1.009	0.807	0.856	0.685	0.873	0.698	0.133
418	Kearl L.	0.1690	0.2329	8.02	25.0	1590	NA	1.280	1.290	1.664	1.192	1.643	1.293	1.781	1.551	2.137	0.367

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 SO_2 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 (Cont'd.)

NOX-SOX GIS No.	Original RAMP Designation	Runoff (Hydro) (m^3/s)	Runoff Isotopic m^3/s	Mean pH	Mean DOC (mg/L)	Mean Gran Alk. (µeq/L)	2002	2003	2004	2005	2006 Hydro	2006 Isotopic	2007 Hydro	2007 Isotopic	2008 Hydro	2008 Isotopic	PAI 2006[1]
Birch Mountains Subregion																	
436	Namur	0.3250	0.1485	7.14	8.4	398	0.235	0.239	0.226	0.313	0.225	0.103	0.231	0.105	0.269	0.123	0.122
442	L23	0.0430	0.1848	6.73	13.7	146	0.087	0.074	0.065	0.074	0.059	0.252	0.074	0.317	0.093	0.398	0.094
444	Legend	0.1765	0.6413	6.78	8.7	164	0.088	0.097	0.099	0.134	0.109	0.396	0.111	0.403	0.119	0.433	0.096
447	L28	0.0448	0.1130	5.21	27.8	24.2	-0.016	-0.03	0.002	-0.025	-0.039	-0.099	0.001	0.004	0.008	0.019	0.056
448	Clayton	0.0330	0.0461	4.20	15.9	-9.2	-0.127	-0.09	-0.073	-0.111	-0.117	-0.163	-0.025	-0.035	-0.088	-0.122	0.086
454	L46	0.1690	0.1026	6.77	23.9	231	0.394	0.375	0.365	0.374	0.303	0.184	0.482	0.292	0.480	0.291	0.097
455	L47	0.1016	0.1422	6.77	22.5	224	0.282	0.241	0.958	0.324	0.272	0.381	0.286	0.400	0.301	0.422	0.074
457	L49	0.0666	0.1164	6.48	22.6	137	0.301	0.260	0.283	0.234	0.210	0.367	0.205	0.358	0.247	0.433	0.085
464	L60	0.1630	0.0730	7.04	20.1	285	0.408	0.420	0.501	0.422	0.319	0.143	0.356	0.159	0.395	0.177	0.078
175	P13	0.0120	0.0028	7.84	45.5	905	1.198	1.235	2.149	1.449	1.099	0.254	0.818	0.189	0.959	0.222	0.145
199	P49	0.0044	0.0013	6.67	18.3	158	0.245	0.215	0.237	0.247	0.305	0.092	0.191	0.058	0.293	0.089	0.172
Canadian Shield Subregion																	
473	A301	0.1756	0.0581	7.30	14.9	405	0.210	0.194	0.189	0.264	0.197	0.065			0.230	0.076	0.014[2]
118	Weekes	0.0092	0.0806	7.30	10.5	440	0.118	0.116	0.114	0.168	0.109	0.956	0.101	0.882	0.133	1.167	0.007[2]
84	L109	0.3537	0.0974	7.05	18.8	355	0.409	0.394	0.341	0.496	0.386	0.106	0.294	0.081	0.441	0.121	0.014[2]
88	O-10	0.0118	0.0094	6.87	22.5	210	0.178	0.189	0.138	NA	0.166	0.133	NA	NA	0.251	0.201	0.014[2]
90	R1	0.0788	0.0974	7.07	17.5	302	0.318	0.311	0.279	0.408	0.311	0.384	0.418	0.517	0.422	0.521	0.014[2]
Caribou Mountain Subregion																	
146	E52	0.0439	0.1510	7.03	23.0	383	0.377	0.365	0.350	0.531	0.349	1.201	0.347	1.193	0.455	1.565	0.027[2]
152	E59	0.0124	0.3079	6.77	12.9	176	0.023	0.025	0.026	0.031	0.021	0.531	0.025	0.622	0.028	0.695	0.027[2]
89	E68	0.1576	0.1072	6.81	22.0	227	0.258	0.274	0.223	0.395	0.262	0.179	0.216	0.147	0.195	0.132	0.027[2]
91	O-1/E55	0.0044	0.0122	6.33	21.6	90.8	0.020	0.029	0.038	0.536	0.064	0.178	0.082	0.228	0.085	0.237	0.027[2]
97	O-2 E67	0.1109	0.2180	6.57	22.3	176	0.201	0.187	0.149	0.081	0.134	0.238	0.104	0.205	0.107	0.211	0.027[2]

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 SO_2 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 6.5-5 Summary of critical loads in ASL lakes (2002-2008).

	2002	2003	2004	2005	2006 Hydro	2006 Isotopic	2007 Hydro	2007 Isotopic	2008 Hydro	2008 Isotopic
No. Lakes	49	50	50	48	50	50	48	48	49	49
Minimum CL	-0.127	-0.090	-0.205	-0.132	-0.117	-0.163	-0.070	-0.082	-0.088	-0.122
Maximum CL	1.382	1.449	2.149	2.008	1.192	1.643	1.333	1.781	1.551	2.137
Average CL	0.306	0.335	0.387	0.434	0.291	0.252	0.300	0.283	0.230	0.177
Median CL	0.210	0.192	0.177	0.250	0.202	0.164	0.198	0.174	0.308	0.302
No. of lakes in which the PAI is greater than the CL	21	18	19	16	19	24	19	23	18	20
Proportion of lakes in which the PAI is greater than the CL (%)	42.9	36.0	38.0	33.3	38.0	48.0	39.6	47.9	36.7	40.8

The percentage of ASL lakes in which the modeled PAI is greater than the critical load (40.8%) is considerably higher than the 8% of 399 regional lakes reported in a study conducted for the NOxSOx Management Working Group within CEMA (WRS 2006). The higher proportion of ASL lakes largely reflects a bias in the selection of lakes for the RAMP program 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 development input in the calculation of the PAI, the estimates of PAI reported in Table 6.5-4 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; this study did not include modifications to the model for organic anions or use of isotopic estimates of runoff.

A modeled PAI value 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 must also be considered in assessing the risks of acidification. Table 6.5-6 summarizes the key chemical characteristics of the lakes with modelled PAI values greater than the critical load. As expected, these are small lakes of low pH, low conductivity, low acid neutralizing capacity (ANC), and high DOC, primarily found in the Stony and Birch Mountain regions.

6.5.4 Trends in ASL Measurement Endpoints in Individual Lakes

Potential trends in the ASL measurement endpoints in all 50 individual lakes were examined using the Mann-Kendall non-parametric test (Gilbert 1987).

Table 6.5-6 Chemical characteristics of lakes having the modelled PAI greater than the critical load in 2008.

Lake	Original Name	pH	Gran Alkalinity (µeq/L)	Conductivity (µS/cm)	DOC (mg/L)	Lake Area (km^2)
168	A21	5.01	-33.6	12.14	21.9	1.38
169	A24	4.73	5.2	11.83	19.3	1.45
170	A26	5.56	14.4	11.22	15.5	2.78
167	A29	5.89	38.2	11.48	23.7	1.05
172	A59	4.81	25.2	22	39.5	108
166	A86	6.38	145.4	27.6	14.9	1.05
287	25	5.16	-44.2	10.49	13.9	2.18
289	27	6.35	79.6	16.25	17.3	1.83
290	28	5.86	59.6	14.96	18.1	0.544
342	82	6.6	114.8	27.4	25.1	2
452	L4	5.62	90.4	19.9	32.6	0.61
470	L7	6.48	138.6	27.1	32.8	0.33
471	L8	6.94	347.8	41.1	24.3	0.6
442	L23	6.6	152	24.8	12.9	3.44
447	L28	5.43	15.2	20.3	28	1.3
448	L29/Clay	4.28	-35.6	16.44	15.9	0.65
182	P23	6.77	104.2	25.1	18.7	0.281
185	P27	5.06	60.8	27	34.4	3.94
199	P49	6.46	164	25	0.7	0.1
223	P94	7.22	665.8	104	2.4	

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

A Mann-Kendall trend analysis was conducted for each measurement endpoint for each of the 50 ASL lakes using the Finnish program MAKSENS (Salmi *et al.* 2002). The program calculates the Mann-Kendall statistic S on lakes with fewer than 10 years of data. For lakes with at least 10 years of data, a normal approximation test is applied to calculate the test statistic Z. Table 6.5-7 presents the results of the analysis. The value of the S or Z statistic is presented for each endpoint/lake. Statistical significance is indicated by shading. It must be noted that the Mann-Kendall test is a non-parametric test which simply subtracts successive values and ranks the differences as negative or positive.

As in previous years, the results of the trend analysis are difficult to explain as a simple process of acidification. While there are two lakes (342 and 354; both in the Stony Mountains) showing significant decreases in pH, there is no significant increase in sulphate or nitrates in these lakes that would account for this decrease. Nitrate and sulphate are the primary acidifying agents. Over all 50 lakes, there are no significant changes in nitrate concentrations, one significant decrease in sulphate concentration (Lake 89, Caribou Mountains) and one significant increase in sulphate (Lake 268; N-E Fort McMurray).

Table 6.5-7 Results of Mann-Kendall trend analyses on ASL measurement endpoints.

Lake ID	Original RAMP Designation	pH		Total Alkalinity		Gran Alkalinity		Calcium		Sulphate		Total Dissolved Nitrogen		Nitrates and Nitrites		Dissolved Organic Carbon		Sum Base Cations		Potential Acid Input keq H^+/ha/y
		S	Z	S	Z	S	Z	S	Z	S	Z	S	Z	S	Z	S	Z	S	Z	
168	A21	11	0		-0.99	-16			**-2.5**		-1.79		-1.79		-0.36		-1.07		-1.79	0.186
169	A24		0.72		0.99	-4			0.72		-0.18		**-2.15**		-0.45		0.72		0.36	0.177
170	A26		-0.18		-0.54	10			-0.72		-0.89		-1.07		1.07		-0.09		0	0.186
167	A29		0.63		0.81	18			0.27		0.54		-0.72		0.09		0.89		**1.97**	0.145
166	A86	-10		**18**		12		**23**		10		12		4		10		**28**		0.117
287	25	-11		**-15**		-3		**-15**		3		-11		8		-11		-1		0.179
289	27	-3		7		5		2		11		-5		14		1		13		0.175
290	28	3		-3		3		-9		-3		-7		-11		-9		-5		0.181
342	82	**-17**		**-15**		-12		-13		1		-9		3		-1		-13		0.120
354	94	**-17**		-7		-3		-9		7		-9		3		-5		-7		0.141
165	A42		0.72		1.25	8			1.79		-0.72	2		6			0		0.72	0.121
171	A47		1.07		**2.68**	10			**2.5**		0.54		0		-1.07		1.61		**2.33**	0.120
172	A59		-1.53		-1.43	-20			0	-1.07			-1.07		-0.54		-0.54		-0.54	0.076
223	P94	-9		-9		-12		-2		-5		-19		-10		3		-9		0.258
225	P96	-3		-9		-9		-7		7		**-15**		5		1		-9		0.238
226	P97	1		-3		-3		-3		3		-7		-3		5		-1		0.353
227	P98	7		-7		-3		1		-3		-4		-1		-5		-1		0.307
267	1	-3		**-11**		-9		-9		1		-9		1		-1		-9		0.214
452	L4		-0.45		**-2.33**	-4			-0.72		-0.36		1.61		0.72		0.54		0	0.222
470	L7		0		-1.43	0			-0.54		0.18		-0.54		0.63		0.18		0.54	0.646
471	L8		0.36		**-2.5**	**-22**			-1.25		0.54		-0.54		0.36		-0.36		-1.43	0.607
400	L39		0.63		-0.89	-4			**-2.5**		0.18		-0.36		0.36		0.18		-0.89	0.085
268	E15 (L15b)	0		-16		**-22**		**-26**		**22**		-14		-5		-4		-16		0.206
182	P23	-3		1		1		1		5		9		11		5		-1		0.250

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

Table 6.5-7 (Cont'd.)

Lake ID	Original RAMP Designation	pH		Total Alkalinity		Gran Alkalinity		Calcium		Sulphate		Total Dissolved Nitrogen		Nitrates and Nitrites		Dissolved Organic Carbon		Sum Base Cations		Potential Acid Input keq H^+/ha/y
		S	Z	S	Z	S	Z	S	Z	S	Z	S	Z	S	Z	S	Z	S	Z	
185	P27	-8		1		3		-1		1		9		6		9		5		0.220
209	P7	-2		7		7		3		2		17		12		-11		1		0.195
270	4	-5		-11		-9		-11		13		-9		5		-3		-13		0.181
271	6	-1		-13		-13		-9		9		-15		-2		-15		-11		0.133
418	Kearl L.	1		3		3		-1		3		3		1		7		3		0.367
436	L18		0.63		3.5	28			1.25		0.89		0.36	-13			-0.54		1.07	0.122
442	L23		0		1.43	8			-1.61		-1.79		0		0.45		-1.43		-1.25	0.094
444	L25		0.27		1.35	6			-0.54		-1.43		0	-6			0		1.07	0.096
447	L28		1.07		1.61	4		-2.25			-0.18		-0.89		-1.53		0		0	0.056
448	L29	-4		-7		-3		-12		-4		0		-2		-8		-4		0.086
454	L46		-0.72		-1.07	-4			-0.89		-0.72		-0.36		-0.72		0.72		-0.89	0.097
455	L47		-0.18		-1.61	-2			-0.72		0		-0.89		0.89		1.07		-0.36	0.074
457	L49		-0.99		-0.89	-18			-1.79		-1.35		0.54		0.72		1.07		-2.68	0.085
464	L60		0		1.07	12			-0.09		-1.97		0.72		-0.58		1.61		-1.07	0.078
175	P13	-7		-11		-7		-7		-17		-15		-1		-9		-9		0.145
199	P49	-1		-5		3		-5		-1		7		14		-1		-3		0.172
473	A301	9		-6		-5		-9		9		1		5		-5		1		0.014
118	L107	16		5		13		-10		16		-8		-2		8		-8		0.007
84	L109		0.54		-1.43	-22			-1.97		0.18		-0.36		-0.72	0			-1.97	0.014
88	O-10	16		2		-3		-4		5		-10		-2		-4		-18		0.014
90	R1		0.89		0.72	8		0.54		1.25		-1.07		0.09		-0.36		1.07		0.014
146	E52		-0.18	26		24			0.89		0.00	-2			0.00		0.18		1.25	0.027
152	E59		0.00		2.50	18			0.81		-1.61		1.25		0.54		0.18		1.43	0.027
89	E68	-18		-12		-12		-18		-20		-10		-12		0		-18		0.027
91	O-1/E55		-0.45		0.18	-7			-2.68		-0.54		-1.07		-1.25		0.89		2.50	0.027
97	O-2 E67		1.79		2.42	18			1.79		0.89		0.54		-1.43		-2.15		-2.86	0.027

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

Gran alkalinity decreases in three lakes (471, 268 both N-E of Fort McMurray and 84 in the Canadian Shield) and increases in two lakes (436 in the Birch Mountain and 146 in the Caribou Mountains). As the Canadian Shield Lakes do not receive any emissions from oil sands development, the downward trend in this variable in Lake 84 must be attributed to factors other than acidification. Only in Lake 268 is there a significant decrease in Gran alkalinity associated with a significant increase in sulphate.

Total alkalinity decreased significantly in five lakes and increased in five lakes. None of the decreases in alkalinity is associated with significant increasing trends in sulphate or nitrates. Base cations decreased significantly in four lakes and increased in four 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 two of the four were associated with significant increases (rather than deceases) in Gran or total alkalinity.

Dissolved organic carbon showed significant decreases in two lakes (Lake 27, N-E of Fort McMurray and 97 in the Caribou Mountains). A decrease in DOC is expected in acidifying lakes (Schindler *et al.* 1992). As the Caribou Mountains are remote from acid emissions and considered a *baseline* area, the decrease in DOC in Lake 97 must be attributed to factors other than acidification.

As in previous years it is difficult to draw a definite conclusion from the results of the Mann-Kendall trend analysis that acidification of the ASL lakes is or is not occurring. Most of the evidence suggests that acidification is not occurring. However, 2008 is the first year in which a significant decrease in Gran alkalinity is associated with a significant increase in sulphate in Lake 268. This lake, located in the North-East of Fort McMurray sub-region receives a significant PAI of 0.206 keq H^+/ha/y (Table 6.5-4). It is also the first year that significant decreases in pH were observed in lakes 342 and 354, even though these trends were not explained by increases in acidifying agents (sulphates and nitrates).

6.5.5 Control Charting of Measurement Endpoints

Ten lakes were selected for control charting based on an acidification risk factor calculated from the ratio of PAI to the critical load value from Table 6.5-4; the greater the ratio in a lake, the greater the risk for acidification. The 10 lakes with the highest ratios are indicated in Table 6.5-8. 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. This group of lakes is slightly different from the group of lakes examined in 2007. Three lakes (169, 168, and 447) in the 2007 group were excluded in 2008 on the basis of slight decreases in their acidification risk factors this year. For continuity with 2007, these lakes were also charted in control plots along with the 10 lakes in Table 6.5-8.

The control plots follow standard analytical control chart theory where control limits representing two and three standard deviations are plotted with the actual data 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 a measurement endpoint is often assumed if three consecutive years fall on the same side outside of the two standard deviation warning limits or one year outside of the three standard deviation control limit.

The control plots for Gran alkalinity for each of the fifty lakes are shown in Figure 6.5-2. This figure permits tracking of potential changes in this key measurement endpoint by sub-region.

As in 2008, the control plots for pH, Gran alkalinity, sulphates, nitrates and DOC for the 13 lakes (Figure 6.5-3 to Figure 6.5-8) indicate that only isolated exceedances of the two standard deviation warning limits occur (e.g., pH and base cations in Lake 170, nitrates in Lakes 447, 170 and 172, Gran alkalinity in Lakes 167 and 170 and sulphate in Lake 447). The variables in each case appear to return to more normal values after each exceedance. Gran alkalinity shows the same anomaly identified in 2005 (Section 6.5.2), a sudden decrease in most of the 13 lakes attributed to 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 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, although cyclical trends are evident in some variables such as sulphate in Lakes 169 and 287. The decrease in pH noted in Lake 342 in the trend analysis (Section 3.5.4.1) is evident in Figure 6.5-3.

Control plots for the ASL measurement endpoints 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.

6.5.6 Summary of Conditions

The results of the analysis of the 2008 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 2008 compared to previous years. Based on the results of the trend analysis and the control plotting, there is no overwhelming evidence to conclude that there have been any significant changes in lake chemistry in the ASL lakes attributable to acidification, although at least one lake shows trends consistent with an acidification scenario. These lakes will be monitored and tracked in future years of the RAMP program.

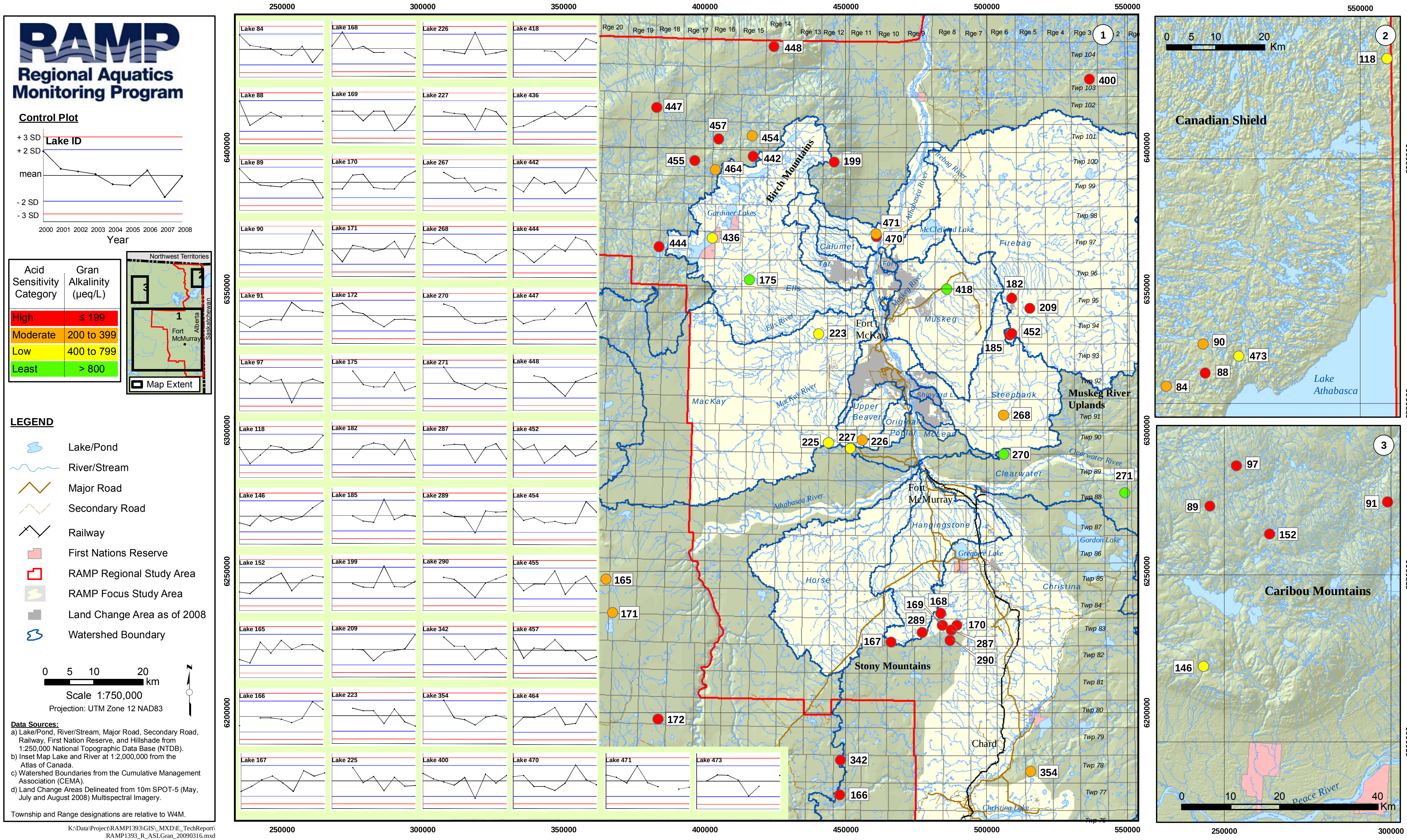
Table 6.5-8 Calculation of the Acidification Risk Factor for Individual RAMP ASL Lakes.

Lake No.	Original Designation	Sub-Region	Critical Load (keq/ha/y)	PAI (keq H+/ha/y)	Acidification Risk Factor PAI/CL
118	L107	Canadian Shield	1.1673	0.007	0.006
146	E52	Caribou Mountains	1.5652	0.027	0.017
90	R1	Canadian Shield	0.5214	0.014	0.027
152	E59	Caribou Mountains	0.6946	0.027	0.039
88	O-10	Canadian Shield	0.2006	0.014	0.070
91	O-1/E55	Caribou Mountains	0.2371	0.027	0.114
84	L109	Canadian Shield	0.1213	0.014	0.115
97	O-2/E67	Caribou Mountains	0.2107	0.027	0.128
418	Kearl L.	N-E Fort McMurray	2.1366	0.367	0.172
455	L47	Birch Mountains	0.4219	0.074	0.175
473	A301	Canadian Shield	0.0760	0.014	0.184
271	6	N-E Fort McMurray	0.6984	0.133	0.190
270	4	N-E Fort McMurray	0.9218	0.181	0.196
457	L49	Birch Mountains	0.4326	0.085	0.196
89	E68	Caribou Mountains	0.1324	0.027	0.204
444	L25	Birch Mountains	0.4330	0.096	0.222
442	L23	Birch Mountains	0.3977	0.094	0.236
400	L39	N-E Fort McMurray	0.3483	0.085	0.244
454	L46	Birch Mountains	0.2915	0.097	0.333
354	94	Stony Mountains	0.3263	0.141	0.432
464	L60	Birch Mountains	0.1769	0.078	0.441
209	P7	N-E Fort McMurray	0.4110	0.195	0.474
225	P96	W. Fort McMurray	0.4624	0.238	0.515
175	P13	Birch Mountains	0.2219	0.145	0.653
268	E15	N-E Fort McMurray	0.2972	0.206	0.693
448	L29	Birch Mountains	-0.1224	0.086	0.703
165	A42	W. Fort McMurray	0.1608	0.121	0.753
227	P98	W. Fort McMurray	0.3887	0.307	0.790
171	A47	W. Fort McMurray	0.1274	0.120	0.942
226	P97	W. Fort McMurray	0.3680	0.353	0.959
436	L18	Birch Mountains	0.1231	0.122	0.991
166	A86	Stony Mountains	0.0888	0.117	1.318
182	P23	N-E Fort McMurray	0.1705	0.250	1.466
223	P94	W. Fort McMurray	0.1519	0.258	1.699
169[1]	A24	Stony Mts.	-0.1017	0.177	1.740
471	L8	N-E Fort McMurray	0.3240	0.607	1.873
199	P49	Birch Mountains	0.0888	0.172	1.938
168[1]	A21	Stony Mountains	-0.0811	0.186	2.295
447[1]	L28	Birch Mountains	0.0191	0.056	2.931
287	25	Stony Mountains	-0.0606	0.179	2.952
452	L4	N-E Fort McMurray	0.0732	0.222	3.034
167	A29	Stony Mountains	-0.0376	0.145	3.856
289	27	Stony Mountains	0.0385	0.175	4.548
342	82	Stony Mountains	0.0263	0.120	4.562
185	P27	N-E Fort McMurray	0.0414	0.220	5.309
470	L7	N-E Fort McMurray	0.0887	0.646	7.287
172	A59	W. Fort McMurray	0.0087	0.076	8.772
290	28	Stony Mountains	0.0028	0.181	63.818
170	A26	Stony Mountains	0.0019	0.186	98.391

Shaded lakes represent those lakes most at risk to acidification.

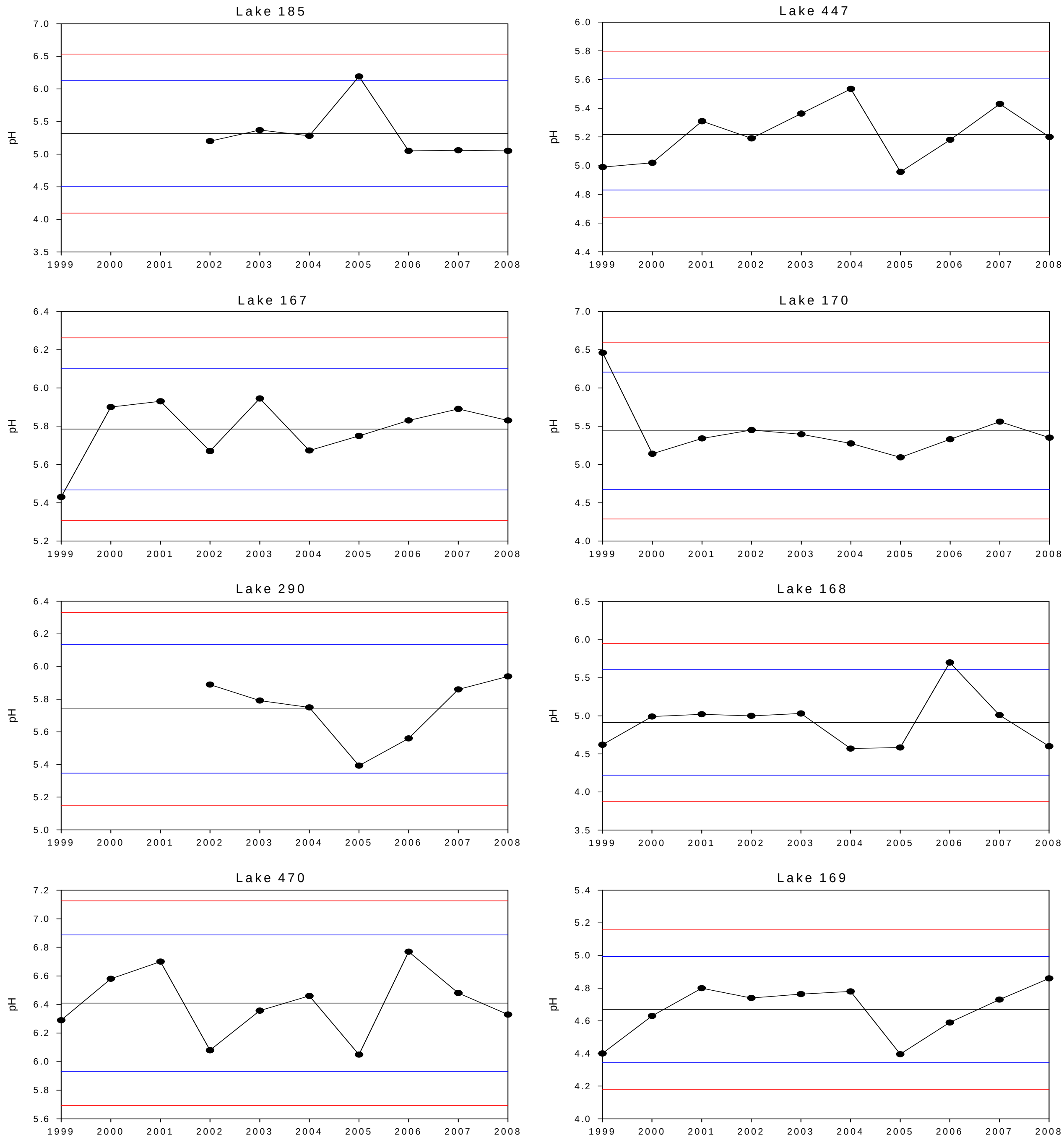
[1] Lakes from 2007 included in the control charting for continuity.

Figure 6.5-2 Control Charts of Gran alkalinity for each of the 50 ASL Lakes, 2000 to 2008 and the Acid Sensitivity of each ASL Lake in 2008.



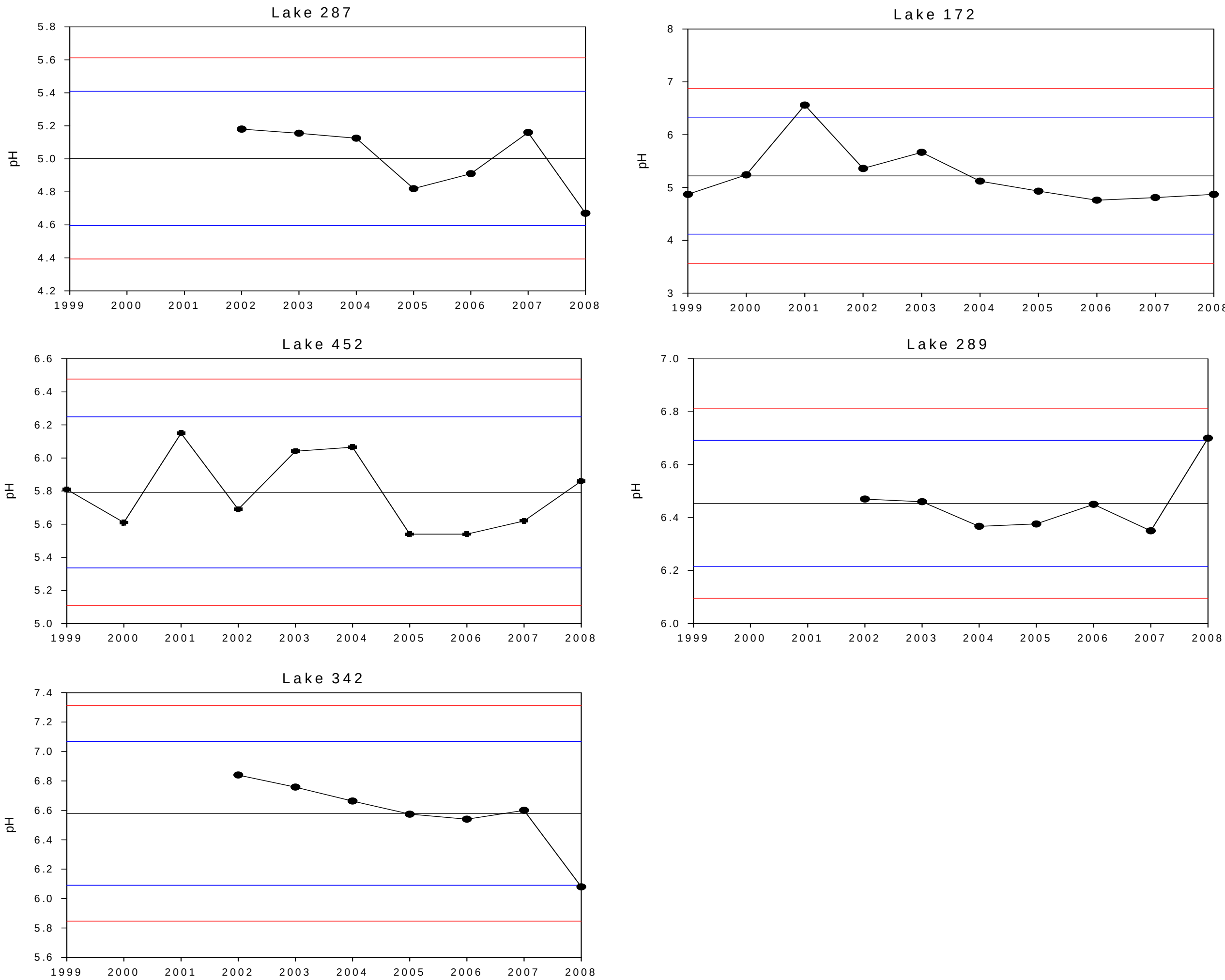
K:\Data\Project\RAMP1393\GIS_MXD\E_TechReport\RAMP1393_R_ASLGran_20090316.mxd

Figure 6.5-3 Shewhart control charts of pH in the ten RAMP ASL lakes most at risk to acidification and three lakes from 2007.



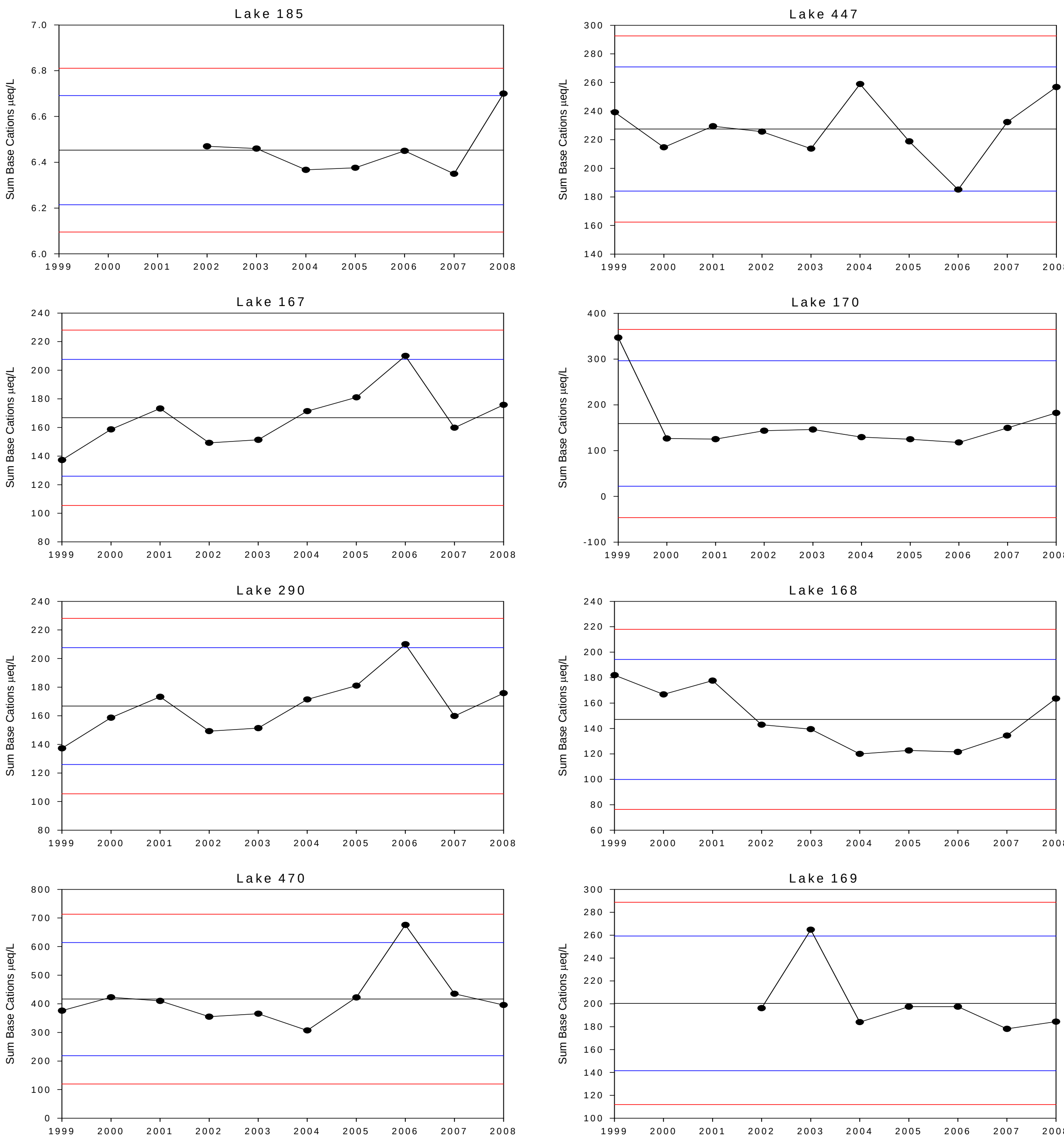
Blue lines: ±2 standard deviations; Red lines: ± 3 standard deviations; black line – mean

Figure 6.5-3 (Cont'd.)



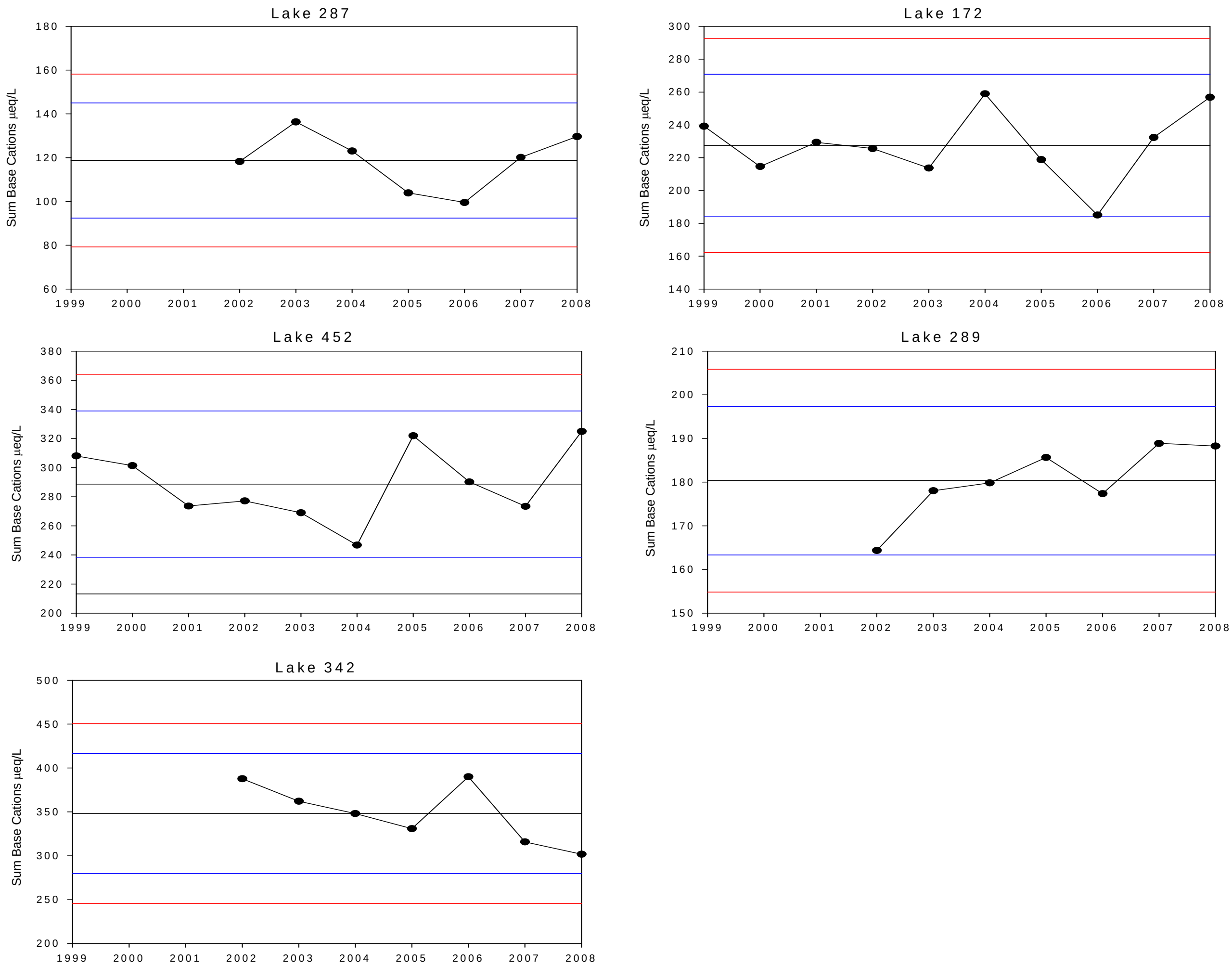
Blue lines: ±2 standard deviations; Red lines: ± 3 standard deviations; black line – mean

Figure 6.5-4 Shewhart control charts of the sum of base cations in the ten RAMP ASL lakes most at risk to acidification and three lakes from 2007.



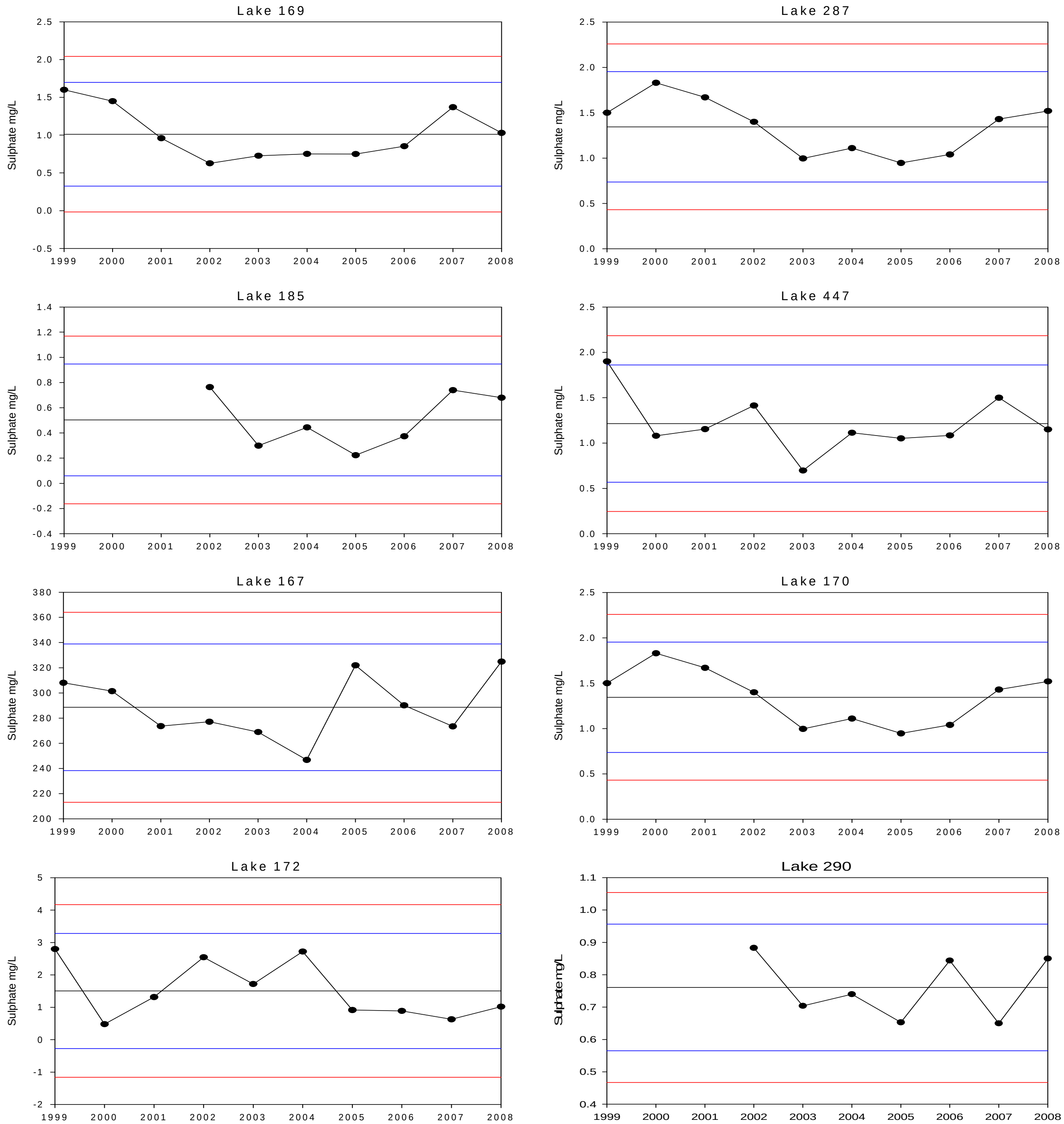
Blue lines: ±2 standard deviations; Red lines: ± 3 standard deviations; black line - mean

Figure 6.5-4 (Cont'd.)



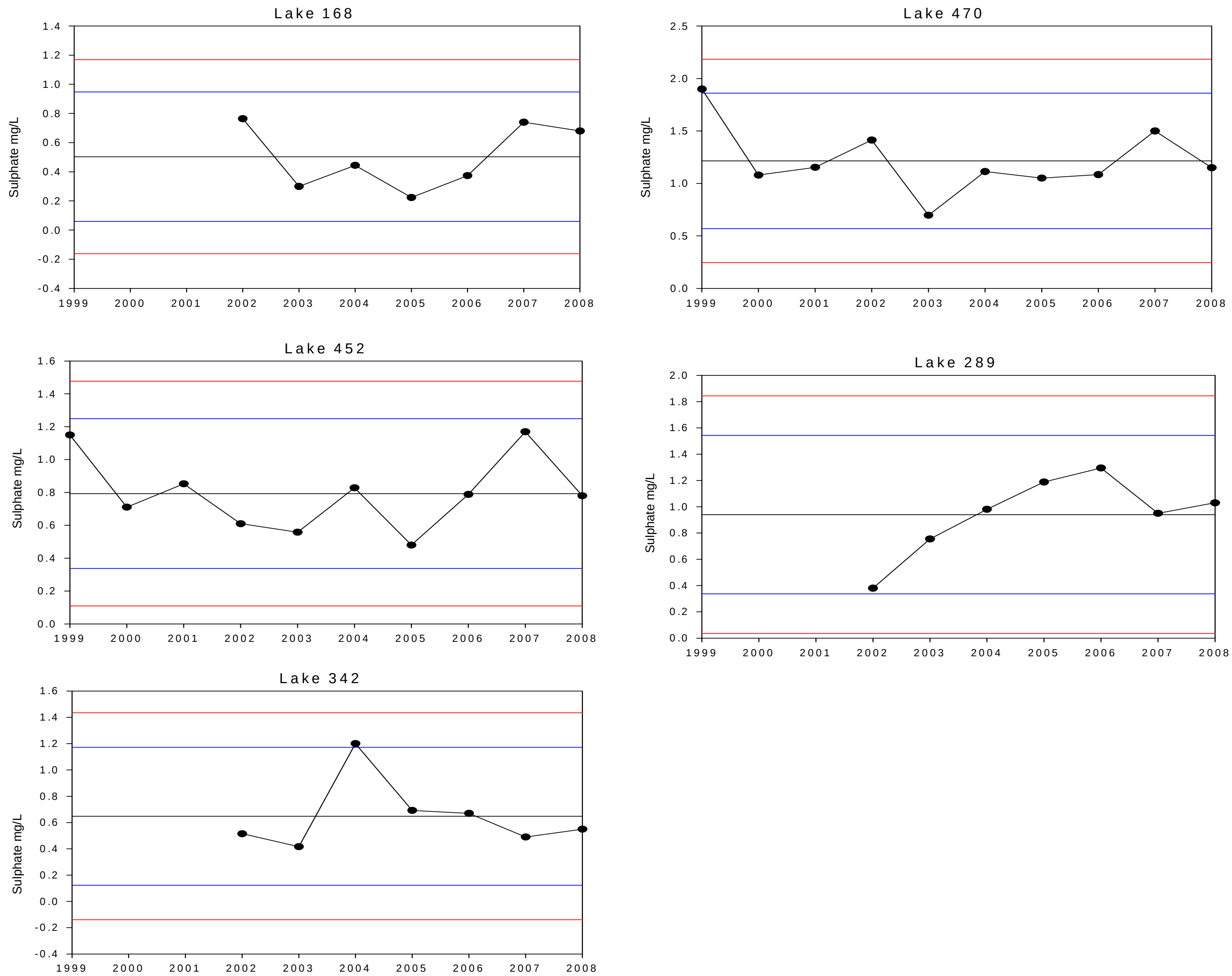
Blue lines: ±2 standard deviations; Red lines: ± 3 standard deviations; black line - mean

Figure 6.5-5 Shewhart control charts of sulphate in the ten RAMP ASL lakes most at risk to acidification and three lakes from 2007.



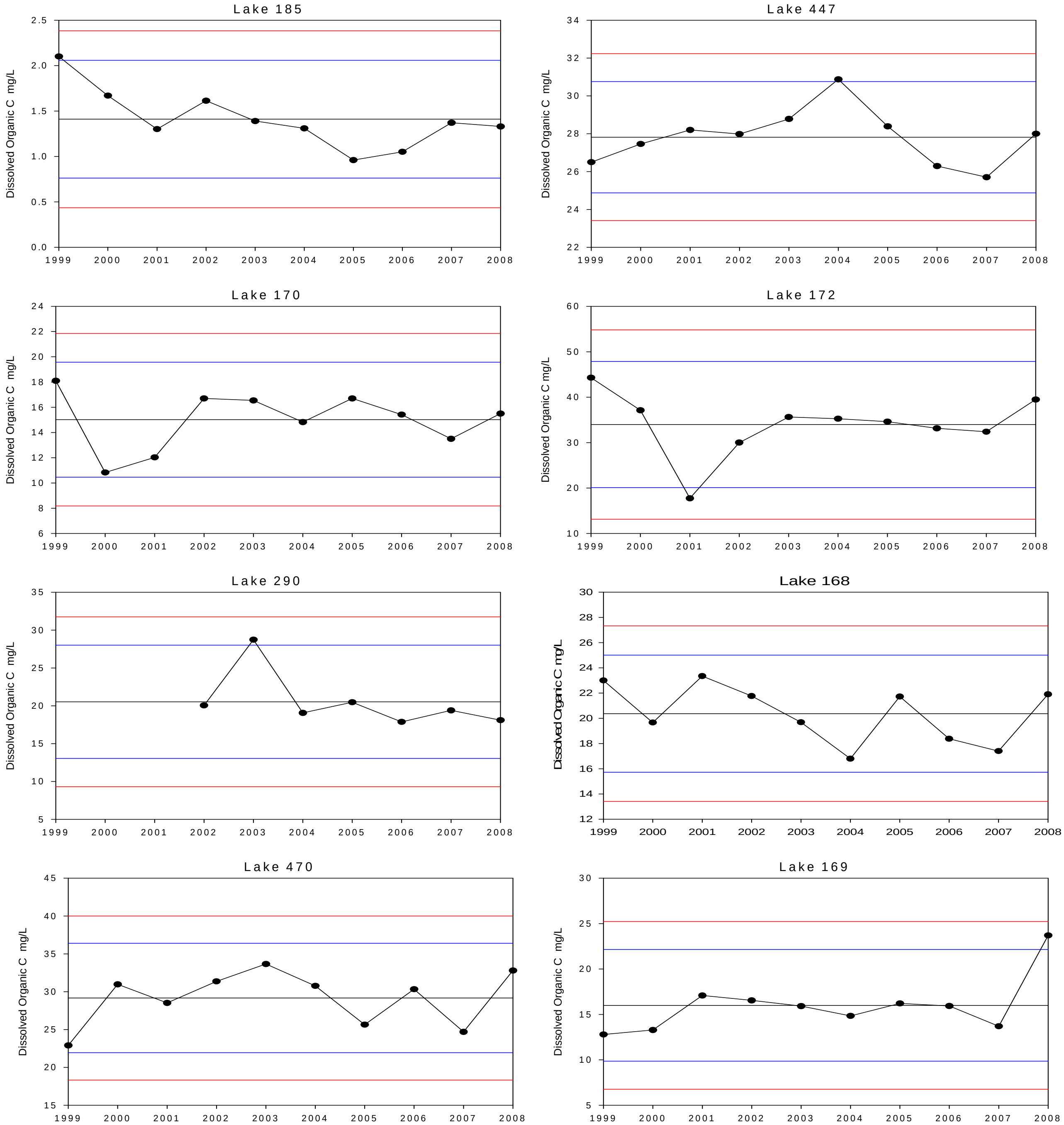
Blue lines: ±2 standard deviations; Red lines: ± 3 standard deviations; black line - mean

Figure 6.5-5 (Cont'd.)



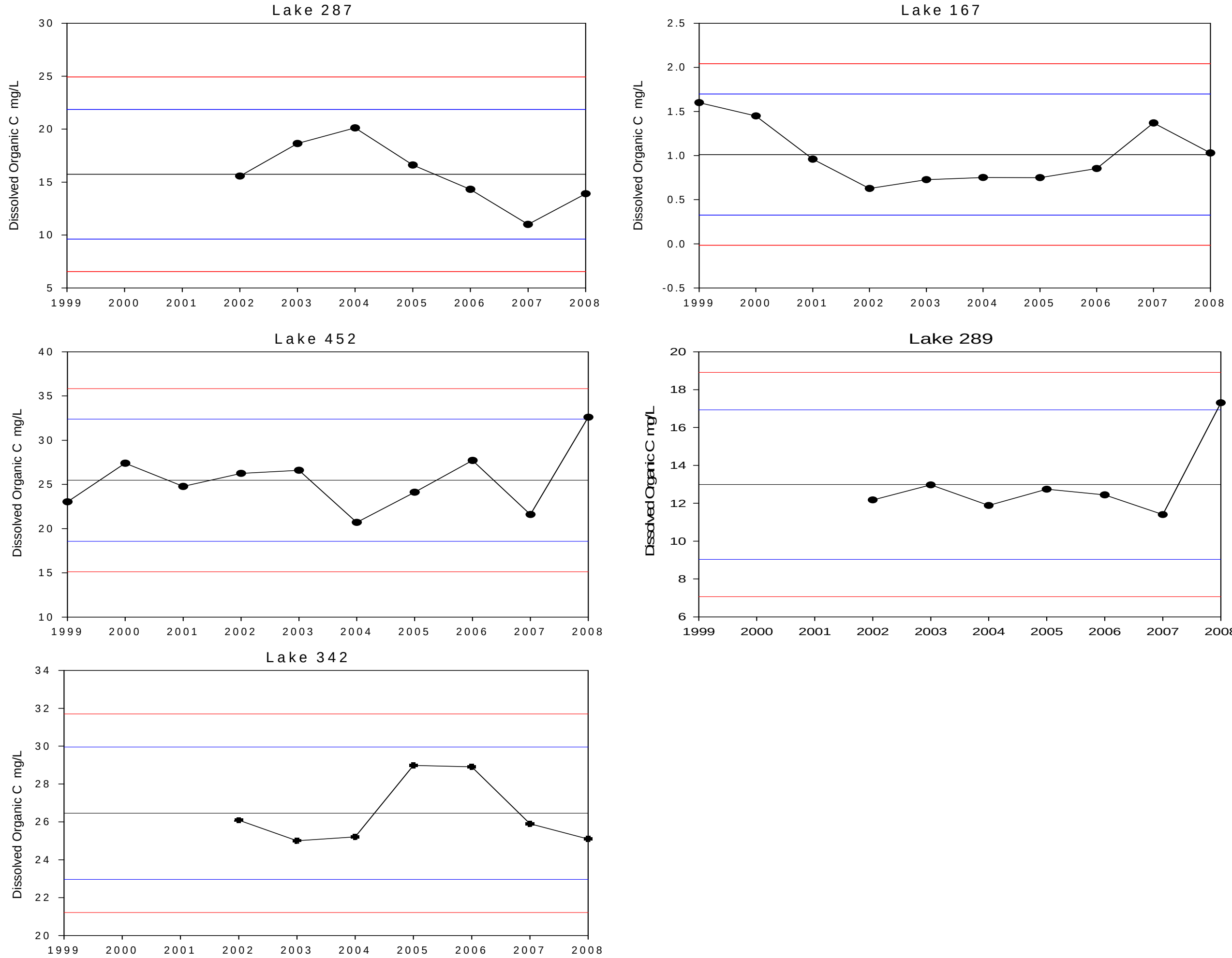
Blue lines: ±2 standard deviations; Red lines: ± 3 standard deviations; black line - mean

Figure 6.5-6 Shewhart control charts of dissolved organic carbon in the ten RAMP ASL lakes most at risk to acidification and three lakes from 2007.



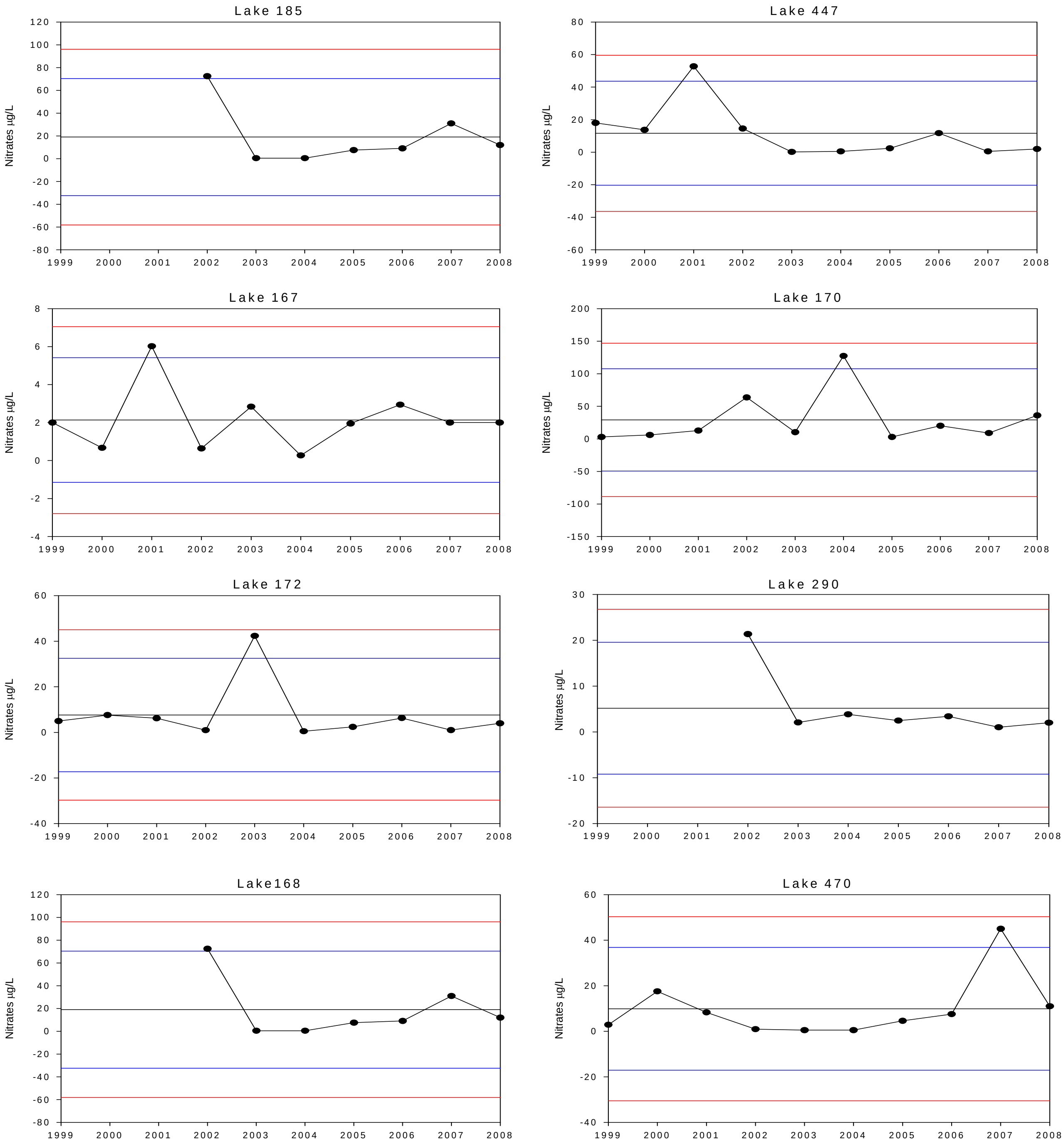
Blue lines: ±2 standard deviations; Red lines: ± 3 standard deviations; black line - mean

Figure 6.5-6 (Cont'd.)



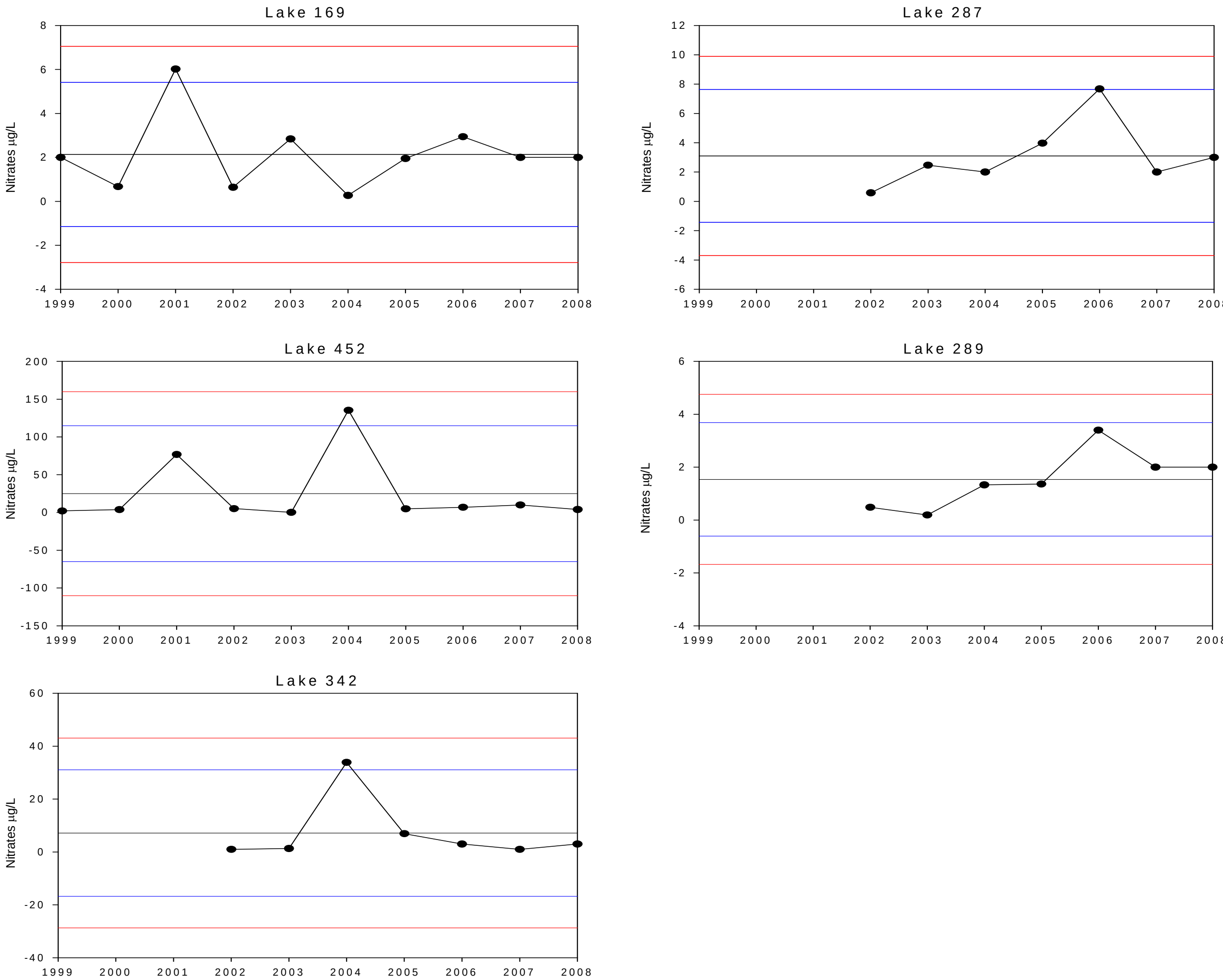
Blue lines: ±2 standard deviations; Red lines: ± 3 standard deviations; black line - mean

Figure 6.5-7 Shewhart control charts of nitrates in the ten RAMP ASL lakes most at risk to acidification and three lakes from 2007.



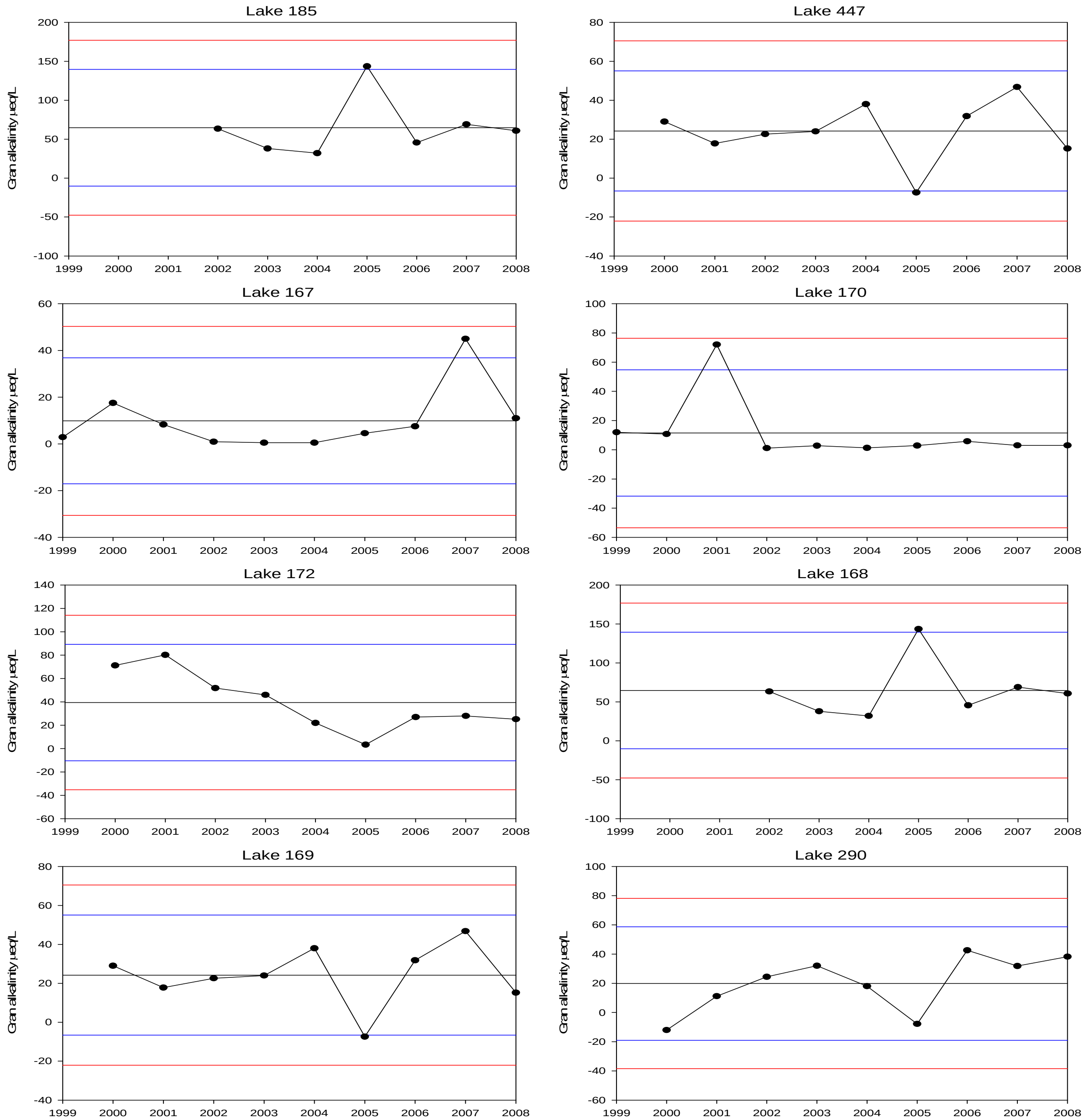
Blue lines: ±2 standard deviations; Red lines: ± 3 standard deviations; black line - mean

Figure 6.5-7 (Cont'd.)



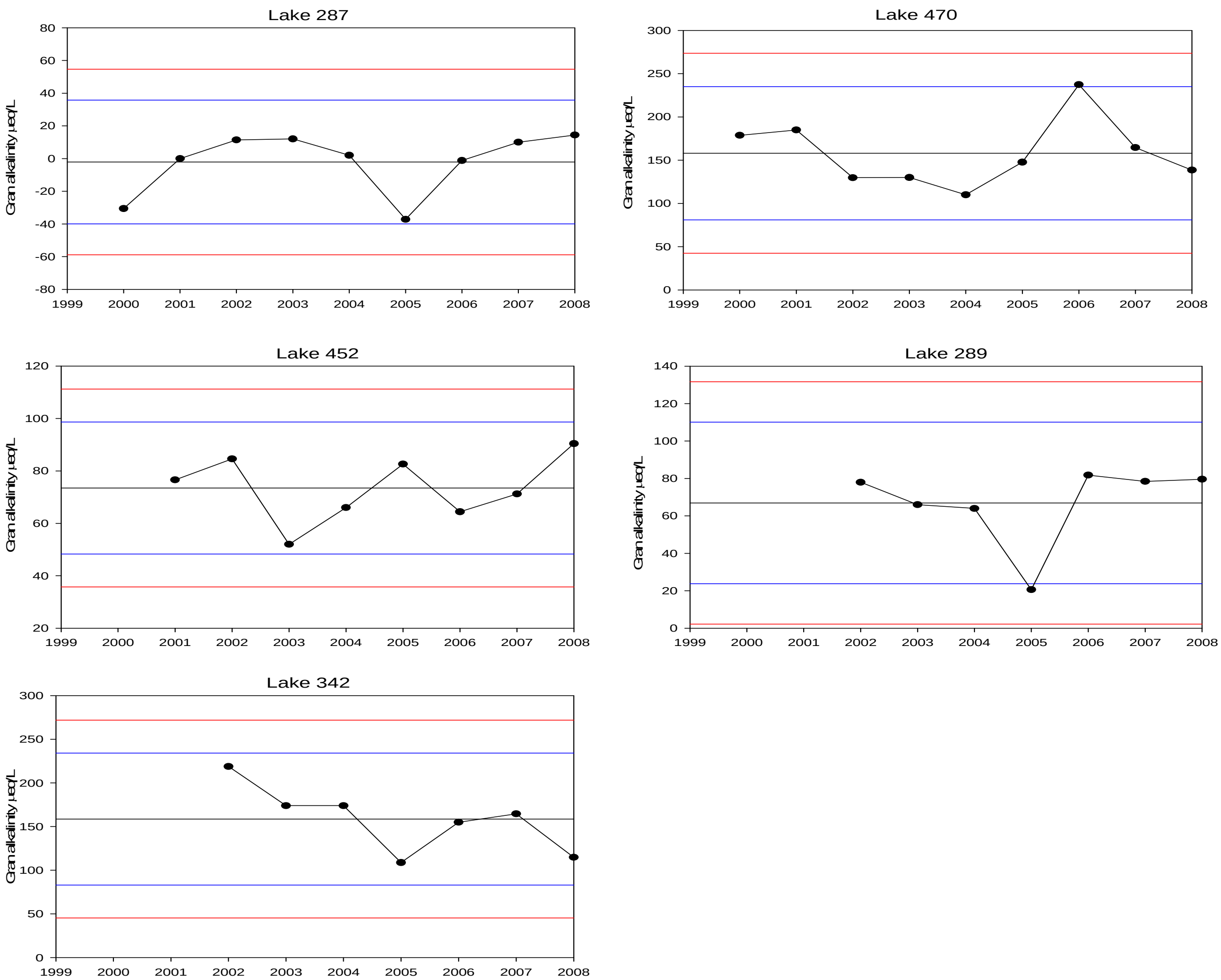
Blue lines: ±2 standard deviations; Red lines: ± 3 standard deviations; black line – mean

Figure 6.5-8 Shewhart control charts of Gran alkalinity in the ten RAMP ASL lakes most at risk to acidification and three lakes from 2007.



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

Figure 6.5-8 (Cont'd.)



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