Ekati Diamond Mine

2013 Aquatic Effects Monitoring Program
Part 1 – Evaluation of Effects

DOMINION DIAMOND

EKATI DIAMOND MINE

2013 AQUATIC EFFECTS MONITORING PROGRAM PART 1 - EVALUATION OF EFFECTS

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Glossary and Abbreviations

ERM

Glossary and Abbreviations

Terminology used in this document is defined where it is first used. The following list will assist readers who may choose to review only portions of the document.

AANDC	Aboriginal Affairs and Northern Development Canada
AEMP	Aquatic Effects Monitoring Program. A comprehensive, early-warning monitoring program designed to detect changes in aquatic ecosystems potentially influenced by the Ekati Diamond Mine.
ALS	ALS Environmental Services
Benthic	Pertaining to the bottom region of a water body, on or near bottom sediments or rocks.
Biomass	The amount of living matter as measured on a weight or concentration basis. Biomass is an indication of the amount of food available for higher trophic levels. In the AEMP, phytoplankton biomass is estimated as chlorophyll *a*, and zooplankton biomass is measured as milligrams of dry weight per cubic metre.
CCME	Canadian Council of Ministers of the Environment
CCREM	Canadian Council of Resource and Environment Ministers
CES	Critical Effects Sizes
Chlorophyll	Chlorophyll is a molecule contained in photosynthetic organisms which is required to carry out photosynthesis. It is used as an indicator of phytoplankton biomass in this report.
DDEC	Dominion Diamond Ekati Corporation
DFO	Fisheries and Oceans Canada.
Diatom	Diatoms are a type of single celled algae. They photosynthesize and may live either free-floating in water (as phytoplankton) or attached to substrates (as periphyton). Diatoms contain a silica shell (called a frustule) outside of their cell membrane.
Diptera	Refers to an insect order. Dipterans are the true flies, and are a major component of lake and stream benthos communities. Dipterans are characterized by a single pair of functional wings and include a wide diversity of species. The diptera include the familiar mosquito and black-fly, and are an important food source for fish as larvae. Their abundance and diversity can be used as an indicator of lake or stream water and sediment quality.
Diversity Indices	A measure of how varied a community of organisms is. In general, a healthy ecosystem will support a variety of species and have a high diversity index.
EC	Environment Canada.

Ecology	The study of the interactions between organisms and their environment.
Ecosystem	A community of interacting organisms considered together with the chemical and physical factors that make up their environment.
Effect	Refers to any potential change in the aquatic environment that is a result of project activities associated with the Ekati Diamond Mine.
ENR-GNWT	Northwest Territories Department of Environment and Natural Resources.
EROD	Ethoxyresorufin-O-deethylase
Euphotic Zone	The euphotic zone refers to the upper portion of the water column in which adequate light is present for photosynthesis to occur.
Eutrophication	Refers to the process by which changes occur in a lake due to nutrient input. Changes which can occur include increased primary producer biomass, shifts in the composition of primary producers, increased sediment oxygen demand, and winter dissolved oxygen decline. Eutrophication is a global issue, and is the reason for the use of phosphorus-free detergents and soaps, and sewage treatment plants.
FPK	Fine Processed Kimberlite
Freshet	Freshet refers to a high water flow event within a stream. In snowmelt driven systems such as the Arctic, the term is commonly used to refer to spring hydrology conditions in which the majority of annual water volume passes through streams in a short period of time. At the Ekati Diamond Mine, freshet typically begins in late May or early June, and lasts for about four weeks.
HC	Health Canada
Hydrology	The study of the properties of water and its movement in relation to land.
IEMA	Independent Environmental Monitoring Agency; established in 1997 to serve as a public watchdog for environmental management at the Ekati Diamond Mine as per the Environmental Agreement.
INAC	Indian and Northern Affairs Canada
Invertebrates	Collective term for all animals without a backbone or spinal column.
Kimberlite	An ultrabasic igneous rock that consists mainly of the mineral olivine and is found in volcanic pipes. The name is derived from Kimberley, South Africa, where the rock was first identified. The host rock for diamonds at the Ekati Diamond Mine.
KLSES	Kodiak Lake Sewage Effects Study
KPSF	King Pond Settling Facility. A settlement facility used to store mine water at the Ekati Diamond Mine.
Kurtosis	Measurement of peakedness of a distribution curve, including height and width

Lake Benthos	Lake benthos communities are a group of organisms that live associated with the bottom of lakes. These communities contain a diverse assortment of organisms that have different mechanisms of feeding. The term lake benthos is used interchangeably with lake benthic macroinvertebrates in this report. Lake benthos are an important food source for fish.
Larva	The immature stage, between egg and pupa, of an insect with complete metamorphosis.
Limnology	The study of lakes, including their physical, chemical, and biological processes.
LLCF	Long Lake Containment Facility. An engineered storage site used to confine the fine fraction of the processed kimberlite (i.e., tailings) in Long Lake at the Ekati Diamond Mine.
PDC	Panda Diversion Channel. An engineered channel used to channel water from North Panda Lake to Kodiak Lake.
Photosynthesis	The metabolic process by which carbon dioxide and sunlight are converted to simple sugars and oxygen. Organisms that photosynthesize contain the molecule chlorophyll.
Phytoplankton	Phytoplankton are microscopic primary producers that live free-floating in water. These organisms are single-celled algae that photosynthesize. Some common types of phytoplankton include diatoms and cyanobacteria.
Primary Producers	In this report, primary producers refer to organisms that convert sunlight into food through the process of photosynthesis. Aquatic primary producers can include phytoplankton, periphyton, macrophytes, and submerged vegetation. Only phytoplankton are examined as part of the Ekati Diamond Mine AEMP.
Processed Kimberlite	The residual material left behind when the processing of kimberlite ore has been completed to extract the diamonds.
Pupa	The stage between larva and adult in insects with complete metamorphosis.
Residual Effects	Effects that persist after mitigation measures have been applied.
Runoff Coefficient	A ratio that expresses the precipitation contributing to overland flow in relation to the total precipitation occurring over a given area.
Secchi Depth	Secchi depth is the depth at which a Secchi disc (standardized white and black disc) can no longer be seen when it is lowered into a lake. Secchi depth can be used to calculate the depth of the euphotic zone.
Secondary Producers	Secondary producers derive their food from eating primary producers. Aquatic secondary producers include zooplankton and some lake and stream benthic invertebrates.

Shannon Diversity Index (H)	Is an index defined as: H = -∑pi * ln(pi), where pi is the proportion of the ith species or genera at a sampling station and ∑ indicates that the pi * ln(pi) is summed over all species or genera.
Simpson's Diversity Index (D)	Is considered a dominance index because it weights towards the most abundant species (represents the probability that two individuals selected at random from the population are different species or genera) and is defined as: D = 1 - $\sum(p_i)2$, where p_i is the proportion of the i^{th} species or genera at a sampling station and ∑ indicates that the $(p_i)2$ is summed over all species or genera.
SNP	Surveillance Network Program.
SSWQO	Site-specific Water Quality Objective
Stream Benthos	Stream benthos communities are a group of organisms that live associated with the bottom of streams. These communities contain a diverse assortment of organisms that have different mechanisms of feeding. The term stream benthos is used interchangeably with stream benthic macro-invertebrates in this report. Stream benthos are an important food source for fish.
Tailings	Ground waste material and water (slurry) rejected from a mill or process plant after most of the valuable minerals have been extracted.
TDS	Total Dissolved Solids
TOC	Total Organic Carbon
Trophic Levels	Functional classification of organisms in an ecosystem according to feeding relationships. Primary producers constitute the first trophic level, and convert energy from the sun into food. All other trophic levels depend upon primary producers for their food. Secondary producers (or primary consumers) constitute the second trophic level, and tertiary producers (or secondary consumers) constitute the third trophic level. In a lake, phytoplankton constitute the first trophic level, zooplankton and some benthic organisms the second, and fish the third.
Turbidity	A condition of reduced transparency in water caused by suspended colloidal or particulate material.
US EPA	United States Environmental Protection Agency
Waste Rock	Barren rock or rock too low in grade to be mined or processed economically.
WLWB	Wek'eezhii Land and Water Board
WRSA	Waste Rock Storage Area
Zooplankton	Zooplankton are small animals that live free-floating in the water. They are secondary producers and feed mainly on phytoplankton.

Centimetre	cm	**Metre**	m
Cubic metre	m^3	**Micrometre (micron)**	µ
Degree	°	**Microsiemens**	µS
Degrees Celsius	°C	**Microsiemens per centimetre**	µS/cm
Gram	g	**Milligrams per kilogram**	mg/kg
Greater than	>	**Milligrams per litre**	mg/L
Kilogram	kg	**Millimetre**	mm
Kilometre	km	**Parts per million**	ppm
Less than	<	**Percent**	%
Litre	L	**Plus or minus**	±

1. Introduction

1. Introduction

1.1 BACKGROUND

The Aquatic Effects Monitoring Program (AEMP) at the Ekati Diamond Mine is a requirement specified in Dominion Diamond Ekati Corp.'s (DDEC) Class A Water Licence (W2012L2-0001). Sampling conducted for the 2013 AEMP was permitted through the Aurora Research Institute Scientific Research Licence (15182) issued for the Ekati Diamond Mine for the collection of samples between January 1 and December 31, 2013.

The AEMP is designed to detect changes in the aquatic ecosystem that may be caused by mine activities. The 2013 AEMP was conducted as specified in the document titled *Ekati Diamond Mine: Aquatic Effects Monitoring Program Plan for 2013-2015* (Rescan 2013d). This plan was developed following a detailed review or re-evaluation of 2010 to 2012 AEMP results completed in November of 2012 and presented to stakeholders at a workshop in December 2012 (Rescan 2012d). Stakeholders that participated in the meetings and provided feedback to the program included Environment Canada (EC), Fisheries and Oceans Canada (DFO), Aboriginal Affairs and Northern Development Canada (AANDC), the Yellowknives Dene First Nation, the Independent Environmental Monitoring Agency (IEMA) and the Wek'eezhii Land and Water Board (WLWB; Rescan 2013d).

Following the workshops, the WLWB solicited written comments from stakeholders to consider and provided recommendations to be incorporated into an AEMP design summary for 2013 to 2015. The final AEMP Plan for 2013 to 2015 (Rescan 2013b) incorporated each of the recommendations provided in the 2012 re-evaluation (Rescan 2012a) and two additional requests made by the WLWB. A summary of the changes to the field methods and laboratory methods implemented in 2013 are included in Sections 1.3 and 1.4.

As completed in the past, the 2013 AEMP report includes a Summary Report which provides an overall summary of the evaluation of effects. The main 2013 AEMP report is comprised of three parts:

1. Part 1 - Evaluation of Effects: provides the methods used to assess change in the aquatic environment and summarizes the results of the effects assessments;
2. Part 2 - Data Report: reports on the state of the aquatic environment at the Ekati Diamond Mine in 2013, including the field methodology and results for each of the aquatic environmental components (e.g., physical limnology); and
3. Part 3 - Statistical Report: provides the detailed results of the statistical analyses reported in the effects analysis.

1.2 OBJECTIVES

The objective of the AEMP is to identify changes occurring in the aquatic environment that may be caused by the Ekati Diamond Mine activities. To that end, the following components of the aquatic ecosystem were monitored in 2013:

- hydrology (October 2012 to September 2013);
- under-ice physical limnology (April/May 2013);
- open water season physical limnology (August 2013);

- ice-covered season lake water quality (April/May 2013);
- open water season lake water quality (August 2013);
- open water season stream water quality (June, July, August, and September 2013);
- phytoplankton (August 2013);
- zooplankton (August 2013);
- lake benthos (August 2013); and
- stream benthos (August to September 2013).

Meteorological data are collected year round at the Ekati Diamond Mine between October 2012 and September 2013 and are reported in the AEMP because they are directly related to hydrology at the site (see Section 3.1 of Part 2 - Data Report).

AEMP sediment quality sampling occurs once every three years and was most recently completed in 2011. The next sediment quality monitoring will be conducted in 2014.

AEMP fish community sampling has occurred once every five years and was most recently completed in 2012. As part of a 2011 evaluation of the fish sampling program at the Ekati Diamond Mine, slimy sculpin were proposed as a sentinel species and changes to the 2012 AEMP field sampling program included the addition of slimy sculpin to be assessed with a sampling frequency of once every three years and decrease the sampling frequency of lake trout and round whitefish to once every six years to link it with the sampling frequency of slimy sculpin (and to further minimize total sampling mortality; Rescan 2011d, 2013a). Thus, slimy sculpin monitoring will be conducted in 2015 and monitoring of large-bodied fish (i.e., lake trout and round whitefish) will be conducted in 2018. The use of slimy sculpin as a sentinel species will continue to be evaluated as fish monitoring progresses.

The objective of this report (Part 1 – Evaluation of Effects) is to provide an overall examination of the long term trends in the aquatic environment at the Ekati Diamond Mine. The report consists of five main sections:

1. Effects Evaluation Methods: includes a description of the statistics used in evaluating the data;
2. Effects Evaluation of the Koala Watershed Lakes and Lac de Gras: includes a summary of statistical results and discussion of each evaluated variable including identification of mine related effects and impacts;
3. Effects Evaluation of the King-Cujo Watershed Lakes and Lac du Sauvage: includes a summary of the statistical results and a discussion for each evaluated variable, including identification of mine related effects and impacts;
4. Historical Averages: includes historical averages (by month) of all measured water variables for each of the AEMP lakes in both the Koala Watershed and King-Cujo Watershed for each baseline and monitoring year. Historical values of key hydrological variables are also included; and
5. Estimates of Residence Times: includes residence times for water in lakes downstream of the Long Lake Containment Facility (LLCF). Lake residence times provide an indication of how quickly lake water quality responds to changes in the quality of surface water entering the lake. Lakes with long residence times (large lakes with small or modest inflows) should respond relatively slowly to changes in upstream water quality, while lakes with short residence times should respond relatively quickly.

There are three other components to the aquatic monitoring at the Ekati Diamond Mine, including Surveillance Network Program (SNP), special effects studies and monitoring programs, and environmental baseline studies (Figure 1.2-1). The SNP assesses DDEC's compliance with the Water Licence (W2012L2-0001) and sampling is completed by DDEC staff according to the Water Licence. Data from two SNP sampling stations, located at the two effluent discharge locations, 1616-30 in the Long Lake Containment Facility (LLCF) and 1616-43 in the King Pond Settling Facility (KPSF), are also incorporated into the AEMP for comparative purposes. Special effects studies are carried out on an as-needed basis to answer questions raised by the results of AEMP monitoring that require further investigation or to focus on specific topics by providing additional information not typically collected in the AEMP.

In 2013, the following studies were undertaken as part of the special effects studies and monitoring programs:

- Lac de Gras Water Quality Monitoring Station – a sampling program in the north arm of Lac de Gras beyond the current extent of the AEMP in order to determine if a new water quality monitoring station is required beyond the current site, S3;
- Nero-Nema Stream Water Quality – water hardness concentrations were compared to concentrations of water quality variables with hardness-dependent water quality benchmarks to examine the extent to which there may be differential dilution in hardness and water quality variables with hardness-dependent benchmarks;
- Grizzly Lake Biological Communities – phytoplankton, zooplankton and benthic invertebrates were sampled in August to assess if communities have been altered following observed changes in the under-ice temperature profiles in 2011 and 2012;
- Hydrocarbon Exposure to Fish – a follow-up study to the results of the 2012 EROD (ethoxyresorufin-Odeethylase) activity analyses which indicated greater hydrocarbon exposure in slimy sculpin and round whitefish which may be related to mine activities. Results from this study will be presented separately from the AEMP.

The results of studies 1 to 3 are presented in Part 2 – Data Report. Study 4 is ongoing and results will be presented separately from the AEMP.

Baseline studies are carried out on lakes and streams of the DDEC claim block prior to development in order to define background conditions from which mine effects can be assessed. Baseline studies were carried out by Golder Associates in the Jay Pipe area and will be presented separately.

1.3 OVERVIEW OF THE EKATI DIAMOND MINE ACTIVITIES

1.3.1 Koala Watershed

The following major activities took place in the Koala Watershed during the 2013 AEMP period (October 1, 2012 to September 30, 2013):

- Main camp housed an average of 15,914 people per month;
- Construction:

SUMMARY OF WATER MONITORING PROGRAMS AT EKATI

Surveillance Network Program (SNP)
(Monthly Reports)

- Program specified in Class A Water Licence.
- Designed to measure compliance with water licence.
- Two major compliance points are discharge from Cell E of the Long Lake Containment Facility (LLCF) and discharge from the King Pond Settling Facility (KPSF).
- If water from the two compliance points does not meet water licence criteria, no water is discharged from the facilities.

Aquatic Effects Monitoring Program
(Annual Report)

- Program is a comprehensive monitoring tool.
- Designed to be sensitive and able to determine changes in the aquatic environment regardless of the cause.
- Provides DDEC with information so that mitigation measures can be evaluated and implemented if required before any changes become a significant environmental impact (Adaptive Management).
- Program designed to determine nature and magnitude of changes.
- Program guidelines are provided in Class A Water Licence, but the program is much more comprehensive and flexible compared to the SNP. Program has been updated in response to stakeholder input and information identified during the past 16 years.

Special Effects Monitoring Programs and Studies
(Annual Report or more frequent)
e.g., Panda Diversion Channel Monitoring Program

- Designed to focus on specific issues or provide additional information not collected as part of the AEMP.
- Designed to meet the needs of other regulatory requirements of the mine (e.g., Fisheries Authorizations).

Environmental Baseline Studies

- Baseline monitoring that is conducted in geographical areas not currently being monitored and where future mining activities may occur.
- Sampling design for aquatics is typically based on AEMP design, as baseline data will form part of future AEMP.

- The Beartooth Fine-Processed Kimberlite Slurry Pipeline was completed and put into operation January 2013. This is a 4.5 km pipeline that transports fine processed kimberlite (FPK) from the process plant and deposits the material into the mined-out Beartooth Pit. The pipeline is heat-traced to allow year round operation and constructed out of fused together lengths of high-density polyethylene (HDPE) pipe. While some process plant discharge was deposited into the LLCF in all months of 2013, the majority of discharge was diverted to Beartooth pit between February and June, 2013 (see pumping summary below for details);
- The inlet and outlet sections and related fish habitat features of the Pigeon Stream Diversion channel were completed in the winter of 2013. Consistent with the first phase of construction, the areas were drilled, blasted and excavated to remove the underlying till material, and then back-filled with 6" granite. An engineered liner system was installed, followed by substrate and habitat features. The original Pigeon Stream was kept operating in 2013 and the completed Pigeon Stream Diversion channel was flushed to remove excess sediment materials. Sandbags blocking the PSD were removed at the end of September 2013 immediately prior to freeze-up, effectively opening the PSD. A small amount of construction will be required in the winter of 2013/14 to complete the tie-in sections where flow to the original Pigeon Stream was maintained;
- Modifications were made to several of the LLCF Cell C FPK discharge spigots to direct material further into the discharge area and allow greater utilization of the available space. These modifications involved lengthening the discharge spigots by adding additional lengths of HDPE pipe;
- An access road was constructed at the South End of the airport runway to allow equipment access required to service the approach light towers;
- Construction and improvement activities were completed on the Misery Haul Road. These road improvements were required to allow safe usage of the road by the HaulMax trucks that will transport kimberlite from the Misery Pit back to the Processing Plant. Main construction activities included realigning three areas that contained sharp turns, widening the road in sections that were deemed too narrow for two-way haul traffic, and general improvements to the roadway surface. Additional caribou crossings were installed based on the wildlife data provided by the remote wildlife cameras implemented in 2012; and
- The construction of the Cell B West Road which began in August of 2011 continued as waste rock became available from the underground mine. The road will provide access to the west side of Cell C so that a pipeline can be built to further maximize tailings storage space.

- Fox Pit:
  - Kimberlite ore was transported to the process plant;
  - Waste rock was transported to the Fox Waste Rock Storage Area; and
  - Kimberlite coarse ore rejects were placed in the coarse kimberlite rejects area of the Panda/Koala Waste Rock Storage Area.
- Beartooth Pit:
  - No mining of Beartooth Pit occurred.
- Panda Pit:
  - No mining of Panda Pit occurred.
- Koala North Pit:
  - Kimberlite ore from underground was transported to the process plant; and

- Waste rock and kimberlite coarse ore rejects from the underground were transported to the Panda/Koala Waste Rock Storage Area.

- Koala Pit:
  - Kimberlite ore from underground was transported to the process plant; and
  - Waste rock and kimberlite coarse ore rejects from the underground were transported to the Panda/Koala Waste Rock Storage Area.

- Dewatering and Discharge:
  - FPK, surface sump water and treated effluent from the sewage plant continued to be deposited into the LLCF;
  - FPK discharge and underground minewater were pumped to Beartooth Pit (total volume 1,531,563 m^3 and 316,452 m^3 respectively);
  - Grizzly Lake drawdown for use at main camp continued (total volume 94,094 m^3);
  - Water was pumped from Bearclaw to North Panda Lake from July 8, 2013 to July 20, 2013 (total volume 96,300.33 m^3);
  - Water from Cell E of the LLCF was discharged into Leslie Lake from October 1, 2012 to December 18, 2012 (on going from July, 2011) (total volume = 3,363,244.92 m^3) and from June 18, 2013 to September 30, 2013 (total volume = 3,717,624 m^3) at which point discharge was still ongoing; and
  - All water discharged from Cell E to the receiving environment met effluent quality criteria defined in the Water Licence W2009L2-0001 and W2012L2-0001.

1.3.2 King-Cujo Watershed

The King-Cujo Watershed contains Misery Camp and the KPSF as well as Misery Pit and associated waste rock piles. The following major activities took place in the King-Cujo Watershed during the 2013 AEMP period (October 1, 2012 to September 30, 2013):

- Misery camp was re-opened in April of 2012 and housed an average of 2,245 people per month.

- Misery Pit:
  - Kimberlite ore was transported to the process plant at main camp; and
  - Waste rock was hauled to the Misery Waste Rock Storage Area.

- Dewatering and discharge:
  - Waste Rock Dam water was pumped into the KPSF from July 15 to July 24, 2013 (total volume = 49,149 m^3);
  - No water was pumped from Misery Pit in 2013;
  - Water was pumped from the KPSF to Cujo Lake from July 7 to July 12 (total volume = 66,322.9 m^3);
  - All water pumped from the KPSF to the receiving environment met effluent quality criteria defined in Water Licence W2009L2-0001 and W2012L2-0001; and
  - Desperation Pond water was pumped to Carrie Pond from June 22 to June 27, 2013 (total volume = 24,693.3 m^3). This water also met effluent quality criteria defined in Water Licence W2009L2-0001 and W2012-L2-0001. A fish removal program was carried out in July and August of 2013 in order to remove as many fish as possible from Desperation Pond prior to the infilling half of the pond with waste rock.

The year 2013 was the 16th consecutive year of post-baseline monitoring within the Koala Watershed and Lac de Gras, and the 13th consecutive year of post-baseline monitoring within the King-Cujo Watershed and Lac du Sauvage.

1.4 CHANGES TO EVALUATION OF EFFECTS FOLLOWING THE 2012 RE-EVALUATION

Seven changes were made to the evaluation of effects beginning in 2013, following the 2012 AEMP re-evaluation:

1. The list of evaluated water quality variables was altered to include total barium, total boron, total cadmium, and total vanadium. Meanwhile, total dissolved solids, ortho-phosphate-P, total aluminum, total iron, and total zinc were removed from the list of evaluated variables in both the Koala and King-Cujo watersheds. In the Koala watershed, TOC was added and total copper was removed from the list of evaluated variables.
2. Given that there is now four years of data available, water quality data collected from Leslie-Moose Stream was analyzed in accordance with the analytical approach employed for other water quality stations in the annual AEMP evaluation of effects beginning in 2012. However, the relatively small number of data points available for Leslie-Moose Stream decreases the probability of detecting statistically significant changes in evaluated variables. Thus, graphical analysis was the primary means through which change in evaluated variables and potential mine effects were assessed in Leslie-Moose Stream in 2013.
3. To better distinguish natural variation from potential mine effects in cases where temporal trends in reference lakes do not share a common slope and the trend in the monitored lake differs from a slope of zero, the slope of monitored lakes was compared to the slope of each reference lake in order. Lack of statistical differences between the slope observed in a given monitored lake and at least two reference lakes would indicate natural variability as the underlying cause of temporal trends in the monitored lake. Significant differences between the trend observed in a monitored lake and two or more reference lakes would indicate a potential mine effect. Graphical analysis and best professional judgment were used to assess the likelihood that a given trend resulted from mining operations.
4. To improve model fit of reference lake data, the reference model was selected that best fits the data using AIC to directly compare the ‘fit’ or error associated with each reference model.
5. In the event that both transformed and untransformed data satisfy parametric assumptions, the AIC was used to determine which transformation provides the best fit to the data and used the best fit model in statistical analyses.
6. The coefficient of determination was examined in cases where there is reason to suspect poor model fit for a given variable and waterbody based on graphical analysis. Low R square values would indicate that results of statistical analyses must be interpreted with caution.
7. To provide a more streamlined and explicit discussion on linkages between physical variables and biotic effects as well as trophic effects, the phytoplankton, zooplankton and benthos sections were merged into a single “biology” section (Sections 3.3 and 4.3 of this report).

2. Methods

2. Methods

2.1 SAMPLE COLLECTION

2.1.1 2013 Field Methodology

Field methodologies are kept consistent for all AEMP sampling periods with minor changes made upon stakeholder review. A complete description of field methodologies is provided in the Part 2 - Data Report, in addition to any modifications made to the sampling methods that occurred in the current sampling year.

DDEC personnel conducted all of the ice-covered season sampling and the majority of stream flow measurements. ERM Rescan scientists conducted all of the open water season lake and stream sampling with the assistance of DDEC personnel.

2.1.2 2013 Sampling Locations

The 2013 AEMP lake and stream sampling sites are provided in Table 2.1-1 and shown in Figure 2.1-1. Surface water flow diagrams through the Koala and King-Cujo watersheds are provided in Figure 2.1-2. Bathymetric maps depicting the aquatic sampling locations within each lake are provided in Figures 2.1-3 through 2.1-14 of Part 2 - Data Report.

Table 2.1-1. 2013 AEMP Sampling Locations

Lake Sites	Stream Sites
Reference Watersheds	
Nanuq Lake	Nanuq Outflow[3]
Counts Lake	Counts Outflow
Koala Watershed and Lac de Gras	
Vulture Lake (reference)	Vulture-Polar (reference)
Kodiak Lake	Lower PDC[1]
Grizzly Lake	Kodiak-Little[2]
1616-30 (LLCF)[4]	1616-30 (LLCF)[4]
Leslie Lake	Leslie-Moose[2]
Moose Lake	Nema-Martine
Nema Lake	Slipper-Lac de Gras
Slipper Lake	
Lac de Gras: S2, S3	
King-Cujo Watershed and Lac du Sauvage	
1616-43 (KPSF)[4]	1616-43 (KPSF)[4]
Cujo Lake	Cujo Outflow
Lac du Sauvage: LdS1, LdS2	Christine-Lac du Sauvage
	Mossing Outflow[2]

1: Water quality and hydrology only.
2: Water quality only.
3: Hydrology station was removed in 2003.
4: Water quality and pumping data only.

Most of the AEMP sampling locations within the Koala Watershed are located downstream of mine discharge (Figure 2.1-1). Exceptions include Vulture Lake and Vulture-Polar Stream, which are internal reference sites located upstream of mine discharge in the Koala Watershed. Grizzly Lake, Kodiak Lake, Kodiak-Little Stream, and the Lower Panda Diversion Channel (PDC) are also located upstream of the LLCF, but are in close proximity of the mine which leaves them susceptible to effects from mine activities. Potential effects at these sites stem from fugitive dust deposition (i.e., from roads, the airstrip, and blasting), road runoff, and the potential for spills. In addition, Kodiak Lake and Kodiak-Little Stream are susceptible to effects associated with the weathering of the PDC, an artificial channel constructed to allow fish passage from North Panda to Kodiak Lake. Kodiak Lake and Kodiak-Little Stream are also susceptible to surface runoff from the vicinity of the ammonium nitrate building (situated near the western shore of Kodiak Lake). Downstream of the LLCF, all lakes and streams are susceptible to the quantity and quality of water discharged from the LLCF as far as Lac de Gras, which receives water from the Koala Watershed at its northern end. In addition, Nema Lake and Nema-Martine Stream are located near the active Fox Pit and are susceptible to fugitive dust and seepage from Fox Pit and its associated waste rock storage areas.

All AEMP sampling stations in the King-Cujo Watershed are located downstream of the KPSF or Desperation Pond (Figure 2.1-1). This includes Lac du Sauvage, which receives water from the King-Cujo Watershed along its western shore. The AEMP lakes and streams are therefore susceptible to changes in the quantity and quality of water discharged from the KPSF.

The external reference lakes and streams (Nanuq and Counts lakes and their respective outflows) are located well away from any mine activities, outside of the zone of influence of the mine (Figure 2.1-1). Nanuq Lake is located in the northeast corner of the Ekati Diamond Mine claim block, approximately 26 km from the nearest possible mine influence. Counts Lake is located southeast of the Ekati Diamond Mine Main Camp, approximately halfway between the camp and Misery Pit. The most proximate source of potential mine effects on Counts Lake is Misery Road, which is approximately 5 km from Counts Lake at its closest point.

2.2 EVALUATION METHODS

2.2.1 Evaluation Framework

Evaluation of the AEMP results relies on a hierarchy of steps (Figure 2.2-1). First, data was collected based on the AEMP plan for 2013 to 2015 (Rescan 2013d). The methods and results of the 2013 AEMP sampling program are reported in Part 2 – Data Report of the 2013 AEMP report.

Observed data were evaluated for quality. Any large dataset is likely to contain some outliers or questionable records caused by instrument failure, transcription errors, laboratory errors, etc. Thus, questionable data were identified and excluded prior to the evaluation of effects. However, all of the data collected as part of the sampling program, including data that were excluded from subsequent analyses, is presented in Part 2 – Data Report of the 2013 AEMP report.

The finalized dataset was graphically and statistically analysed to detect possible mine effects. Regression modelling was used to detect any changes that might be occurring in lakes and streams through time and also to determine whether temporal patterns differed between monitored and reference sites. Different regression models were applied to different variables depending on the number of years of data that were available and, in the case of water quality, the proportion of data that were greater than the analytical detection limit (see Section 2.2.4). If statistical analyses were not possible because assumptions or data requirements were not satisfied, variables were subjected to graphical analysis only (see Section 2.2.5). In such cases, aquatic component data were examined for historical trends and spatial gradients.

Figure 2.1-1
AEMP Lake and Stream Sampling Locations, 2013

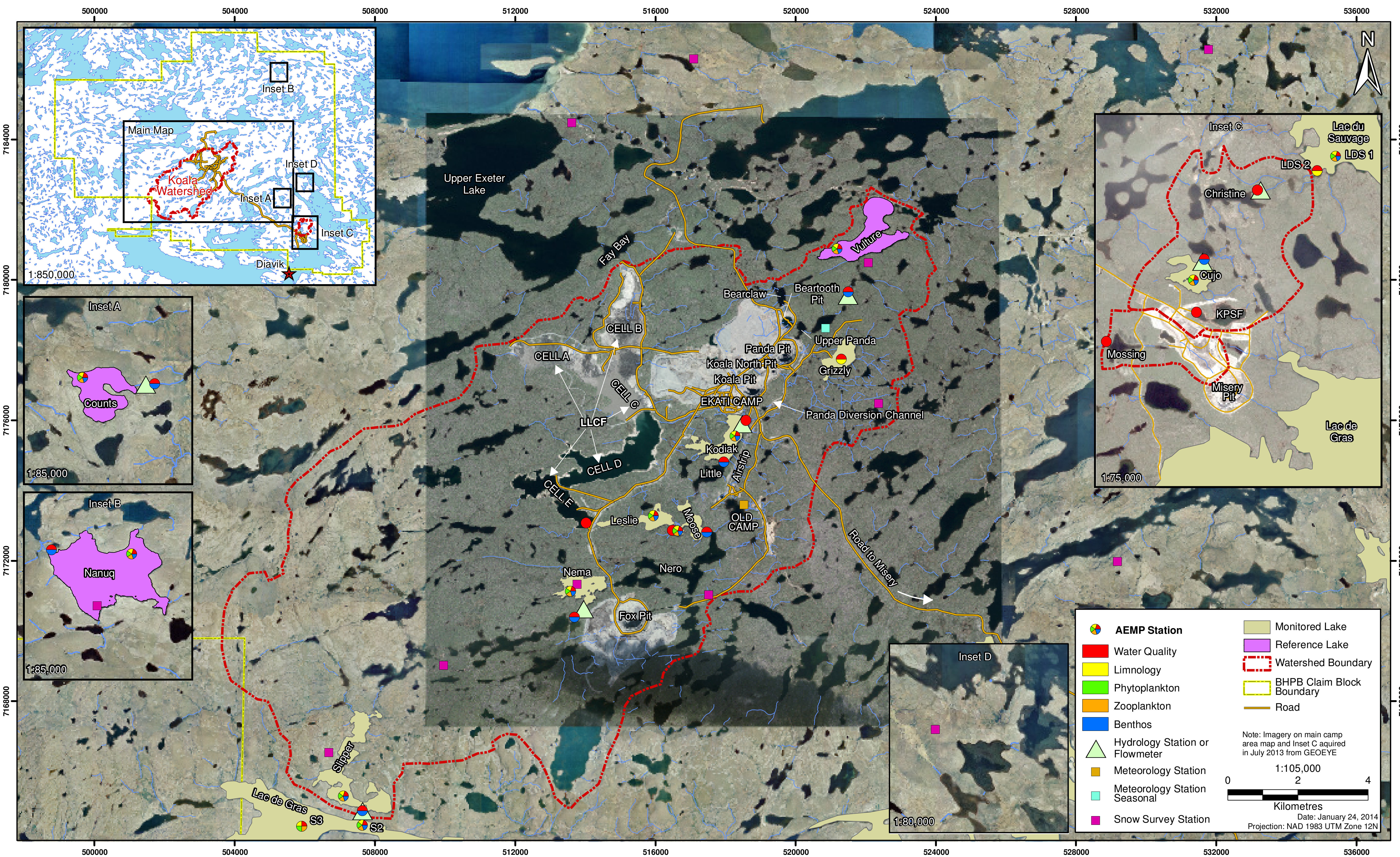
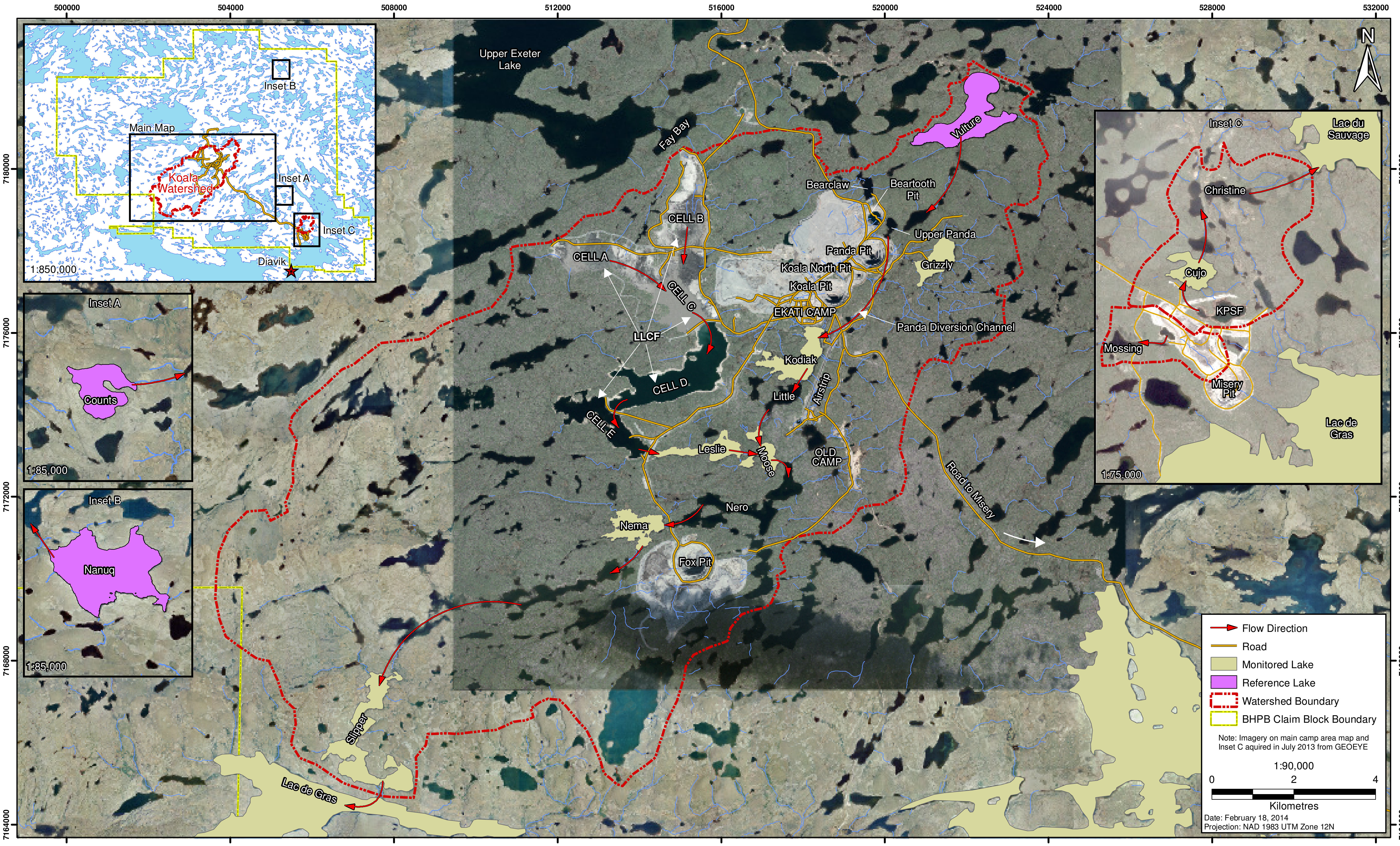
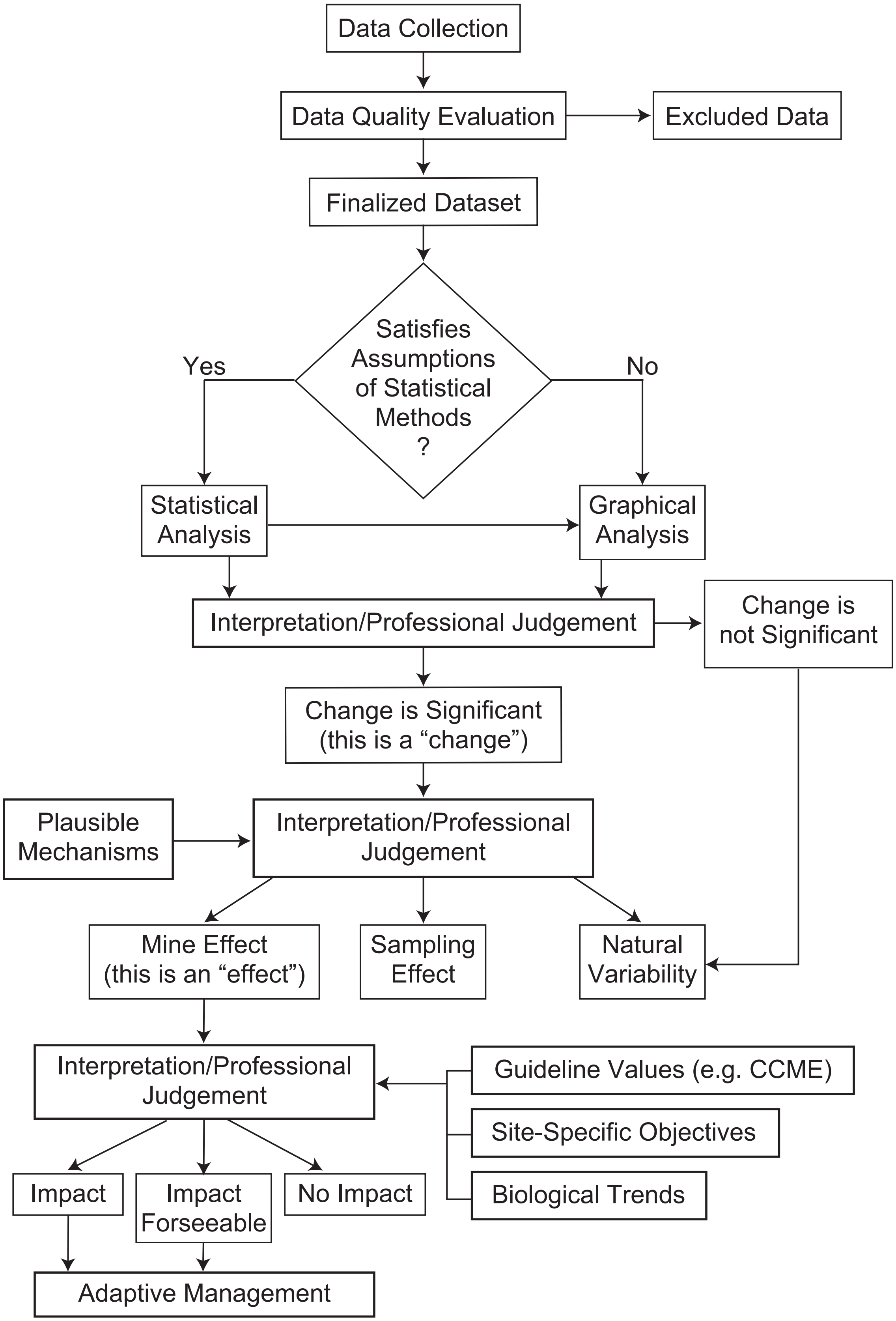


Figure 2.1-2
Surface Water Flow through the AEMP Sampling Area




Data Collection
Data Quality Evaluation
Excluded Data
Finalized Dataset
Satisfies Assumptions of Statistical Methods ?
Yes
No
Statistical Analysis
Graphical Analysis
Interpretation/Professional Judgement
Change is not Significant
Change is Significant (this is a "change")
Plausible Mechanisms
Interpretation/Professional Judgement
Mine Effect (this is an "effect")
Sampling Effect
Natural Variability
Interpretation/Professional Judgement
Guideline Values (e.g. CCME)
Site-Specific Objectives
Biological Trends
Impact
Impact Forseeable
No Impact
Adaptive Management

2.2.2 2013 Sampling Program

Table 2.2-1 summarizes sampling components, frequency, and replication completed during the ice-covered and open water seasons as part of the 2013 AEMP sampling program.

Table 2.2-1. Summary of the 2013 AEMP Sampling Program

Monitoring	Seasonal Frequency	Replication and Depths at each Lake/ Stream per Sampling Event
Lakes		
Water quality	April	n=2 @ mid water column depth
		n=2 @ 2 m from the bottom
	early August	n=3 @ 1 m below surface
		n=3 @ mid water column depth
		n=3 @ 2m from the bottom (Leslie Lake only)
Physical Limnology	April[1]	n=1 profile over deepest part of lake, or at lake station (LdG, LdS)
	early August	n=1 profile over deepest part of lake, or at lake station
Phytoplankton	early August	n=3 @ 1 m
Zooplankton	early August	n=3 vertical hauls from 1 m above bottom to surface, with flowmeter
Benthos	early August	n=3 @ 5-10 m depth (mid)
Streams		
Water quality	June (freshet)	n=2
	early July	
	early/mid-August	
	September (fall high flows)	
Benthos	Early August to early September	n=5
Hydrology manual flow measurements	7 or more times per open water season (Late-May to September)	n=7 or more
Automated station installation	installation prior to freshet, maintenance during manual measurements	n=1
Hydrometric levelling surveys	4 or more times, Late-May to August	n=4 or more

n = number of samples or measurements
1: Dissolved oxygen and temperature profiles were collected 6 times throughout the ice-covered season in Cujo Lake.

2.2.3 Variables Evaluated in 2013

The variables evaluated in the 2013 AEMP included the list of variables of interest identified in the AEMP plan for 2013 to 2015 (Table 2.2-2; Rescan 2013d).

2.2.4 Statistical Analysis

Regression models were used to compare data from each of the monitored lakes to reference lake data over the monitoring period, between 1998 and 2013. If a large number of data (> 60%) were below the analytical detection limit, the lake was excluded from the regression analyses. Either linear mixed effects or tobit regression analyses were fit to the data, depending on the fraction of samples that were below analytical detection limits (see Section 2.2.4.1 and 2.2.4.2), and hypothesis tests were performed to evaluate differences in the level of each variable in the monitored and reference lakes and streams (Section 2.2.4.3). For each variable, observed and fitted values were examined (Section 2.2.4.4), with conclusions drawn based on the statistical results (2.2.4.5). Details of the statistical results for each variable are presented in Part 3 – Statistical Report of the 2013 AEMP report.

Table 2.2-2. Aquatic Variables Evaluated in 2013

Physical Limnology - Lakes	**Water Quality – Lakes and Streams**	**Aquatic Ecology**
Under-ice dissolved oxygen	Physical/Ions	Phytoplankton
Secchi depth	pH	Chlorophyll *a* concentrations
Open water dissolved oxygen[1]	Total Alkalinity	Phytoplankton density
Hydrology[1,2]	Water hardness	Phytoplankton diversity
	Chloride	Relative densities of major phytoplankton taxa
	Potassium	Zooplankton
	Sulphate	Zooplankton biomass
		Zooplankton density
	Nutrients	Zooplankton diversity
	Total ammonia-N	Relative densities of major zooplankton taxa
	Nitrite-N	Lake Benthos
	Nitrate-N	Lake benthos density
	Total phosphate-P	Lake benthos dipteran diversity
	Total organic carbon	Relative densities of major dipteran taxa
	Metals	Stream Benthos
	Total antimony	Stream benthos density
	Total arsenic	Stream benthos dipteran diversity
	Total barium	Relative densities of major dipteran taxa
	Total boron	Stream benthos EPT diversity
	Total cadmium	Relative densities of EPT taxa
	Total copper[3]	
	Total molybdenum	
	Total nickel	
	Total selenium	
	Total strontium	
	Total uranium	
	Total vanadium	

1: Open water season dissolved oxygen and 2013 hydrology results are only reported in Part 2 - Data Report and discussed where relevant in this report.
2: Historical values of key hydrological variables are presented in Section 5.
3: King-Cujo Watershed only.

The statistical methodology outlined above has been used to assess patterns in physical limnology, water quality, sediment quality, and phytoplankton, zooplankton, and benthos communities in monitored lakes and streams in the AEMP since 2007, with minor modifications introduced in 2013, following the 2012 AEMP Re-evaluation (Rescan 2008a, 2012d). The minor modifications introduced in 2013 include the following:

1. To better distinguish natural variation from potential mine effects in cases where temporal trends in reference lakes did not share a common slope and the trend in the monitored lake

differed from a slope of zero, the slope of each monitored lakes was compared to the slope of each reference lake. Significant differences between the trend observed in a monitored lake and two or more reference lakes indicated a potential mine effect. Lack of statistical differences between the slope observed in a given monitored lake and at least two reference lakes indicated that the cause of temporal trend in the monitored lake was related to natural variability. Graphical analysis and best professional judgment was continued to be used to assess the likelihood that a given trend resulted from mining operations.

2. The Aikake Information Criterion (AIC) was used to directly compare the 'fit' or error associated with each reference model. This information was used in combination with reference model testing to ensure the most robust reference model was selected for use in hypothesis testing.

3. In the event that both transformed and untransformed data satisfied parametric assumptions, AIC was used to determine which transformation provided the best fit to the data. This information was used to inform professional judgment with respect to model selection in order to ensure that the best possible model was used in statistical analyses.

8. The coefficient of determination was examined in cases where there was reason to suspect poor model fit for a given variable and waterbody based on graphical analysis. Low R square values indicated that model fit was weak (r^2 < 0.5) or poor (r^2 < 0.2) and that results of statistical analyses must be interpreted with caution.

Other analyses were performed prior to 2007 (e.g., assessment of aquatic variability and repeated measures), with details provided in earlier reports (e.g., Rescan 2010b).

In addition to regression models, current mean values (without including an estimate of error) of selected biological variables (i.e., biomass, density, and diversity indices) were compared against mean baseline values ± 2 standard deviations (SD) following the 2009 AEMP re-evaluation (Rescan 2010c).

2.2.4.1 Linear Mixed Effects (LME) Regression

<u>Model Form</u>

Let y denote a water, sediment, or biological variable of interest (e.g., sulphate concentration or zooplankton density) and $y_i(x)$ be the observation from lake i in year x. The types of model fitted to the data all have the basic regression model form:

$$\textbf{(1)} \qquad y = Lake + Year + Year^2 + Lake*Year + Lake*Year^2,$$

indicating that the mean level of a variable is modeled with separate intercepts, linear and quadratic effects of time in each lake, and random errors.

Separate intercepts allow for differences in the initial values of the variable between lakes and linear effects for changes over time. Quadratic effects are included to allow for non-linearity in the trend. Errors are assumed to be normally distributed with zero mean and the same variance for all lakes. Mathematically, the basic regression model can be written as:

$$\textbf{(2)} \qquad E(y_i(x)) = \beta_{0i} + \beta_{1i}x + \beta_{2i}x^2 \ ,$$

where $E(y_i(x))$ represents the expected (mean) value of the variable in lake i in year x.

Assessing Model Fit

Goodness-of-fit of the regression models was examined through plots of the residuals. Let $y_i(x)$ denote the fitted value for lake ***i*** in year ***x***, defined as:

$$\textbf{(3)} \qquad y_i(x) = \beta_{0\mathrm{i}} + \beta_{1\mathrm{i}}x + \beta_{2\mathrm{i}}x^2 + \varepsilon_x \ ,$$

where ε_x is the predicted value of the random effect that impacts all lakes in year ***i***. The residual for each observation is the difference between the fitted and the observed values:

$$\textbf{(4)} \qquad e_{ix} = y_i(x) - \hat{y}_i(x),$$

which estimates the unexplained variation for lake ***i*** in year ***x***, ε_{ix} . If the key assumption that the true errors are normally distributed with equal variance is satisfied, then these residuals should also be approximately normally distributed and their variance should not depend on either lake or year. Normality of the distribution of residuals for each fitted model was assessed with a normal QQ-plot (see Part 3 - Statistical Report). Plots of the residuals by year and against the fitted values were used to assess homogeneity of the variance over time and against the value of the variable.

A common deviation from this assumption is that variance increases as the value of the variable increases. This often results simply because quantities vary more at larger scales, and is visible as a cone shape in the plot of residuals versus fitted values, with residuals at small fitted values clustering close to zero relative to the residuals at large fitted values. In these cases, the logarithm of the variable was modeled to satisfy approximate normality and stabilise the variance (e.g., total nickel, Part 3 - Statistical Report).

Pseudoreplication

Under the current AEMP, repeated observations from each lake in each month are collected from similar locations at the same time, and the variability between these observations may not reflect the true variation between random replicates from the entire lake in the given month (but see Rescan 2008b). Analyzing these measurements as independent observations may underestimate the true variability, making tests overly sensitive. The simplest method of dealing with pseudoreplication is to average all measurements from each lake in each month to provide a single observation. Because comparisons were made across lakes and across years, averaging the data within one lake has little effect on the tests of interest.

The depth from which water quality samples were collected in the water column was assumed to have no effect on water quality and all observations from the same lake in the same month were combined into a single observation.

Random Variation

The formulae presented above provide a regression model for the mean value of the variable in each lake in each year, but actual measurements are affected by random sources of variation and are distributed about the mean. Potential sources of variation exist on many different levels in the system and may include environmental factors that affect all lakes equally in a given year, factors that affect each lake uniquely, sampling variation that affects the samples taken from a single lake in a single year due to heterogeneity in the water or sediment, and true measurement errors that arise during laboratory analysis. One of the strengths of the regression modeling approach is that some of these sources of variation can be distinguished in order reduce some of the unexplained variation in the measurements, and provide more precise estimates of the true variable means.

As discussed above, measurements from each lake in each year can be averaged to create a single grouped observation without any loss of information. Variation in these values can then be broken into two components: yearly effects that impact the measurements in all lakes/streams and effects that impact each of the monitored and reference lakes individually. These sources of variation are included in the model as random effects, so that the final linear mixed effects model of the average variable value observed in lake *i* in year *x* becomes:

$$\textbf{(5)} \qquad y = Lake + Year + Year^2 + Lake*Year + Lake*Year^2 + Year\text{-}R + Error\text{-}R,$$

or mathematically:

$$\textbf{(6)} \qquad y_i(x) = \beta_{0\mathrm{i}} + \beta_{1\mathrm{i}}x + \beta_{2\mathrm{i}}x^2 + \varepsilon_x + \varepsilon_{ix} \ ,$$

where ε_{xi} and ε_{ix} represent two random variables, the first that affects all lakes in year *x* identically and the second that only affects lake *i*. These random variables are both assumed to follow normal distributions with zero mean and variance σ_x^2 and σ_{ix}^2 respectively. Because these models include both the fixed effects (informative factor levels that influence the mean) and random effects (uninformative factor levels that influence the variance) they are termed mixed-effects models. Thus linear mixed effect (LME) models were used to detect changes in selected variables in monitored lakes.

Baseline Data

Baseline data were collected from l994 to 1997 for the reference lakes and lakes of the Koala Watershed, and from 1999 to 2000 for the King-Cujo Watershed. Ideally, monitoring would include baseline data for each lake in order to account for initial variability before the start of mining at the Ekati Diamond Mine. Unfortunately, the timing of baseline sampling in the Koala Watershed and reference lakes often did not correspond to the time period that was used for the regression analysis (mid-April to early May, and late July to early August). Consequently, baseline data for these lakes were excluded from the statistical analyses. Data from all sampling years were included in the analysis of the lakes of the King-Cujo Watershed because the timing of baseline sampling for these lakes corresponded with post-baseline data collection. Moreover, excluding these data would have left few years of data for fitting the statistical models. Therefore, data collected from 1998 onward were included in the analysis in the King-Cujo Watershed. Interpretations are based on the methods of Wiens and Parker (1995), originally developed for assessing the impact of what are termed ‘accidental events’ (e.g., oil spills), when no baseline data are available. Although no accidental events were observed at the Ekati Diamond Mine, these methods can be applied to monitored lakes and streams of the receiving environment to determine effects of exposure to containment facility discharge. This is discussed further in Section 2.2.4.5.

2.2.4.2 Tobit Regression

Model Form

All of the water and sediment quality variables have detection limits (DLs) below which the laboratory analyses cannot make an accurate measurement. Thus, for some water and sediment quality variables the observed value is below the DL for many of the lakes and years so that only an upper bound is known for these values. Often this upper bound is replaced by half of the DL and statistical analyses are performed as if the value is actually observed. Results from this type of analysis can be misleading, particularly when the DLs are not consistent from year to year. For example, if all observations for a given variable in one lake have been below the DL in every year but the DL for that variable has consistently decreased (perhaps due to improving technology), then the imputed observations will appear to decrease over time. In reality, there is no information to conclude if the value is increasing, decreasing or remaining constant. Further, replacing these values with half of the DL ignores any uncertainty in these observations and the analysis will tend to underestimate the standard deviation of the variables.

A better approach is to perform a 'tobit' regression which properly accounts for the censoring below the DL. In a maximum likelihood analysis of a standard regression model (as above) the likelihood contribution of a single observation *y* given the covariates $x1,...,xp$ and a single error term $\varepsilon \sim N(0,\sigma^2)$ is:

$$\textbf{(7)} \qquad L(y) = (2\pi\sigma^2)^{-1/2} \exp\left(\frac{-1}{2\sigma^2} \left(y - \sum_{i=1}^{p} \beta_i x_i \right)^2 \right),$$

which is simply a normal probability density function of an observation, *y*, with mean $\sum \beta_i x_i$ and variance σ^2 . Now consider the case where *y* is censored and is only known to lie in the interval (*a*,*b*). Tobit regression replaces the likelihood contribution with the integrated density:

$$\textbf{(8)} \qquad L(y) = \int_a^b \exp\left(\frac{-1}{2\sigma^2} \left(y - \sum_{i=1}^{p} \beta_i x_i \right)^2 \right) dy = \Phi\left(\frac{b - \sum_{i=1}^{p} \beta_i x_i}{\sigma} \right) - \Phi\left(\frac{a - \sum_{i=1}^{p} \beta_i x_i}{\sigma} \right),$$

where $\Phi(x)$ is the standard normal cumulative distribution function. The likelihood can then be formed by multiplying the appropriate censored or uncensored contributions for each observation and maximum likelihood inference can be conducted to compute variable estimates and their standard errors and to perform hypothesis tests (Tobin 1958).

Tobit regression can be applied when there is a moderate amount of data missing from each lake. In the analysis of some variables, all, or almost all, of the observations from a given lake are below the DL in all years. In these instances, there is not enough information to estimate the variables of the model associated with that lake, and data for that lake was omitted from the regression analysis. When a monitored lake was omitted from the regression analysis, comparisons involving that lake could not be performed and limited inference was based on plots of the observed data. In a few cases, there were insufficient data to model any of the reference lakes, and so it was not possible to make comparisons between the reference lakes and the monitored lakes. In these cases, simpler comparisons were performed to test whether there was any evidence that the variable values in each monitored lake had changed over time.

Pseudoreplication

The same concern with pseudoreplication in the LME regression models exists in the tobit regression. However, when values were censored it was not possible to average the observations in each lake to obtain a single value for each year and a different solution was necessary. Suppose that observations $y_1,...,y_{n1}$ and $y'_1,....,y'_{n2}$ are available from a given lake in a given year where each y_i is known exactly and each y'_i is censored so that y'_i belongs to the interval (a_i,b_i). Given these observations, the sample average, $\bar{y}$, was bounded such that:

(9) $$a = \frac{\sum_{i=1}^{n_1} y_i + \sum_{i=1}^{n_1} a_i}{n_1 + n_2} < \bar{y} < \frac{\sum_{i=1}^{n_1} y_i + \sum_{i=1}^{n_1} b_i}{n_1 + n_2} = b\ ,$$

and tobit regression was performed with (*a*,*b*) as the censoring interval for the sample mean. If all measurements are known exactly, then $n_2 = 0$ and $a = b = \bar{y}$.

2.2.4.3 Hypothesis Testing

Overview

Once the regression models were fit, hypothesis tests based on the fitted curves were performed to test for differences in the level of each variable in the monitored lakes and in the reference lakes (Figure 2.2-2). Simply put, we aimed to test the hypothesis that the intercept and/or trend of the mean variable value in each monitored lake had the same intercept and/or trend in the reference lakes. If this hypothesis is true, then any differences between the monitored lakes and reference lakes are due to random variation and there is no reason to believe the mine has affected the monitored lakes. If this hypothesis is false, then we conclude that the variable has behaved differently in the monitored and reference lakes, which may suggest a change related to the mine.

However, this comparison has two important caveats. First, the comparison is only sensible if the variable behaves the same over time in all reference lakes. Second, the behavior of a variable may refer to absolute values of the variable (so that any difference between the reference lakes and monitored lakes is important even if this difference is constant over time) or to changes relative to the initial level in each lake (so that differences are not deemed important if the changes in monitored and reference lakes are the same relative to the initial value in each lake). To account for these points, a sequence of tests was performed that attempted to dissect the relationship between the lakes in several steps (Figure 2.2-2). The results of each test in the sequence determined subsequent tests that were performed, and the exact conclusions and the strength of the inference was dependent upon which tests were performed.

Test 1a: Equality among Reference Lakes

The first hypothesis test compared the absolute value of the variable in all three reference lakes to determine if there was any evidence of a difference between the mean variable values in the reference lakes (Figure 2.2-2). The null and alternative hypotheses for the test were:

> H_0: exactly the same pattern of means occurs over time in all three reference lakes
>
> H_a: there is a difference in the pattern of means between at least one pair of the reference lakes.

To state this mathematically, let β_{oc} , β_{1c} and β_{2c} denote the regression coefficients for the model of Counts Lake, β_{oN} , β_{1N} and β_{2N} the coefficients for Nanuq Lake, and β_{oV} , β_{1V} and β_{2V} the coefficients for Vulture Lake. The hypotheses of the test are:

> H_0: $\beta_{oc} = \beta_{oN} = \beta_{oV}$, $\beta_{1c} = \beta_{1N} = \beta_{1V}$, and $\beta_{2c} = \beta_{2N} = \beta_{2V}$
>
> H_a: $\beta_{ij} \neq \beta_{ik}$ for at least one *i* = 0, 1, 2 and *j* ≠ *k*.

Figure 2.2-2

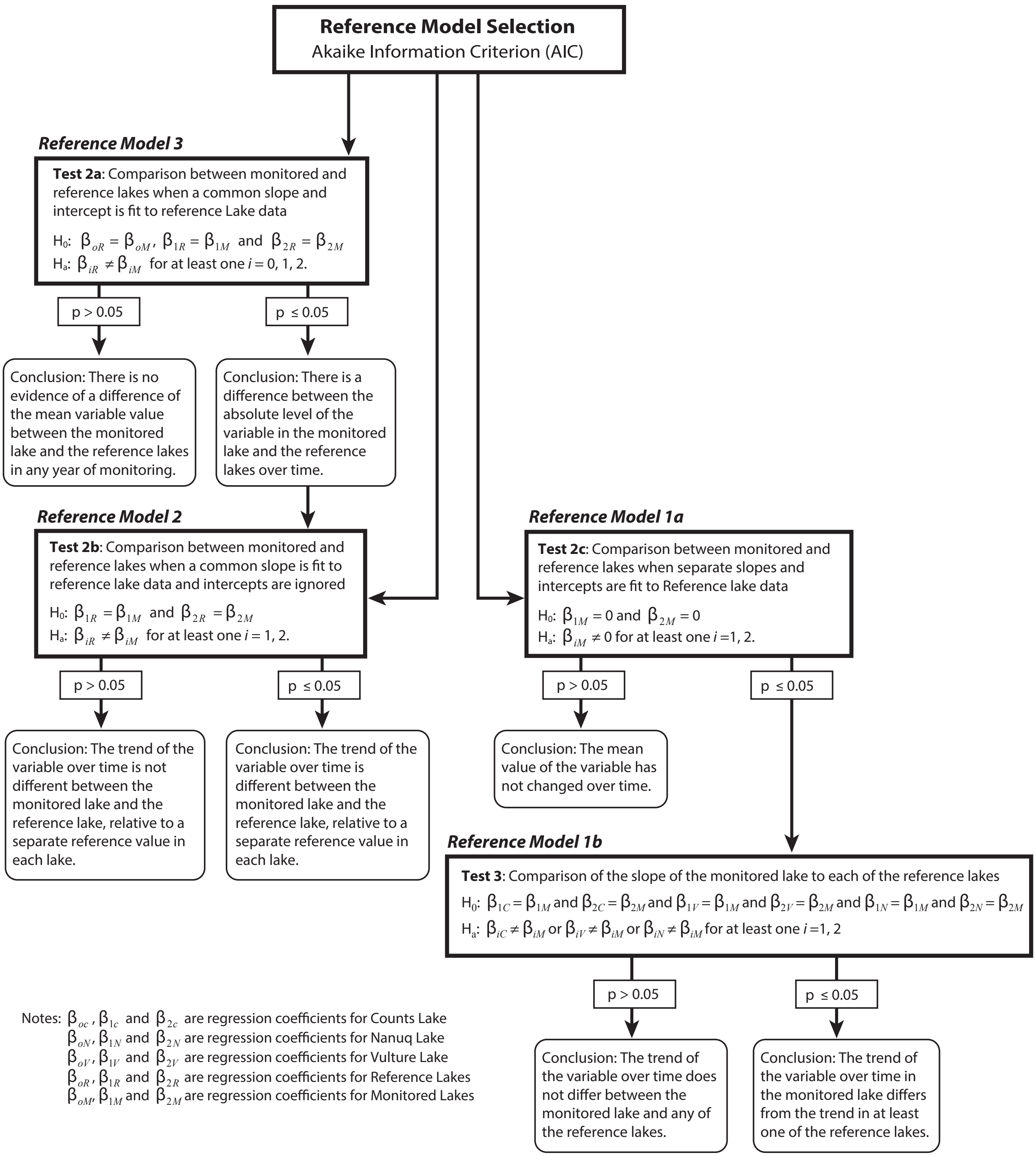
Hypothesis Testing Procedure for Evaluation of Effects

ERM

Model Form: $y_i(x) = \beta_{0i} + \beta_{1i}x + \beta_{2i}x^2 + \varepsilon_x + \varepsilon_{ix}$

or

$$L(y) = \int_a^b \exp\left(\frac{-1}{2\sigma^2}\left(y - \sum_{i=1}^{p}\beta_i x_i\right)^2\right)dy = \Phi\left(\frac{b - \sum_{i=1}^{p}\beta_i x_i}{\sigma}\right) - \Phi\left(\frac{a - \sum_{i=1}^{p}\beta_i x_i}{\sigma}\right)$$



Notes: β_{oc}, β_{1c} and β_{2c} are regression coefficients for Counts Lake
β_{oN}, β_{1N} and β_{2N} are regression coefficients for Nanuq Lake
β_{oV}, β_{1V} and β_{2V} are regression coefficients for Vulture Lake
β_{oR}, β_{1R} and β_{2R} are regression coefficients for Reference Lakes
β_{oM}, β_{1M} and β_{2M} are regression coefficients for Monitored Lakes

If the null hypothesis was not rejected ($p > 0.05$), then we concluded that the same model could account for the observations in all three reference lakes (i.e., there was no evidence to believe that the variable behaves differently between the reference lakes).

Test 2a: Comparisons between Monitored and Reference Lakes when Test 1a is not Rejected

In the case that the null hypothesis of Test 1a was not rejected ($p > 0.05$), a new model can be fit that groups all of the data from the reference lakes (Figure 2.2-2). This allows each monitored lake to be compared to the reference lakes as a single group and decreases the number of coefficients in the model, which increases the power of further tests. Let β_{oR}, β_{1R} and β_{2R} denote the coefficients associated with the reference lakes in the new model. The next set of hypothesis tests compared the model of the mean variable value in each monitored lake with the model for the reference lakes. Let β_{oM}, β_{1M} and β_{2M} denote the coefficients of the new model for one monitored lake. The hypotheses of Test 2a were:

H_0: $\beta_{oR} = \beta_{oM}$, $\beta_{1R} = \beta_{1M}$ and $\beta_{2R} = \beta_{2M}$

H_a: $\beta_{iR} \neq \beta_{iM}$ for at least one $i = 0, 1, 2$.

If H_0 was rejected ($p \leq 0.05$), we would conclude that there was a difference between the absolute value of the variable in the monitored lake and the reference lakes over time. This difference may result from differences in either the intercept or in the trend over time. If H_0 could not be rejected ($p > 0.05$), the analysis provides no evidence of a difference between the mean value in the monitored lake and reference lakes in any year of monitoring.

Test 1b: Further Comparisons among Reference Lakes when Test 1a is Rejected

If the null hypothesis of Test 1a was rejected ($p \leq 0.05$), we would conclude that there was a difference among the reference lakes. This difference may arise either because there is a difference between the reference lakes that was constant over time (so that the means in all reference lakes are parallel through time) or because there are more complicated differences that change over time. This might occur if, for example, there is natural variation between lakes so that the mean value of the variable differed between the lakes but remains constant over time within a particular lake. If the difference is constant over time, a simplified model can be fit to the data from the reference lakes that groups the linear and quadratic effects but allows for different intercepts. Comparisons can then be made to the monitored lakes, ignoring the intercept in each model.

To assess differences in trends, a new test was conducted with the following hypotheses (Figure 2.2-2):

H_0: $\beta_{oN} = \beta_{oV}$, $\beta_{1N} = \beta_{1V}$, and $\beta_{2N} = \beta_{2V}$

H_a: $\beta_{ij} \neq \beta_{ik}$ for at least one $i = 1, 2$ and $j \neq k$.

The conclusions of this test were weaker than the conclusions of Test 1a in that they only pertain to the values of the mean in the reference lakes relative to the intercept in each lake.

Test 2b: Comparisons between Monitored and Reference Lakes if Test 1b is Not Rejected and Following Test 2a

If the null hypothesis of Test 1b is not rejected ($p > 0.05$), then a new set of hypothesis tests can then be performed to compare the relative pattern in each monitored lake to the reference lakes (Figure 2.2-2). Using the notation above, the hypotheses of the new tests are:

H_0: $\beta_{1R} = \beta_{1M}$ and $\beta_{2R} = \beta_{2M}$

H_a: $\beta_{iR} \neq \beta_{iM}$ for at least one i = 1, 2.

Rejecting H_0 leads to the conclusion that the temporal trend of the variable differs between the monitored lake and the reference lake, relative to a separate reference value in each lake. That is, the model of the mean variable values in the monitored lake and the reference lakes are not parallel.

This test was also conducted for each lake following Test 2a. Rejecting the null hypothesis for Test 2a leads to the conclusion that the mean variable values differ between the monitored lake and the reference lakes, but it is not clear what causes this difference. As with the differences among reference lakes, it is possible that the difference is constant over time so that the curves fitted to the mean values are parallel, or that there is a more complicated difference that changes over time. This can be determined with Test 2b. If the null hypothesis of Test 2a was rejected, and the null hypothesis of Test 2b was not rejected, then there is only evidence for a difference in the intercepts of the models. If both null hypotheses were rejected, then there was evidence of a more complicated difference between the monitored and reference lakes.

Test 2c: Comparison for Monitored Lakes when Test 1b is Rejected or when no Reference Lakes are Modelled

If the null hypothesis of Test 1b is rejected ($p \leq 0.05$), then it is not possible to draw conclusions about any similarities between the reference lakes (Figure 2.2-2). When this occurs, it is not sensible to construct tests that compare the observations in the monitored lakes with the reference lakes as a single group. A similar situation arises when none of the reference lakes can be modeled because too many values are less than the analytical detection limit. In either case, the fitted patterns of means in each monitored lake are compared to a constant value to determine if there is evidence that the mean value of the variable has changed over time. The hypotheses of the test are as follows:

H_0: $\beta_{1M} = 0$ and $\beta_{2M} = 0$

H_a: $\beta_{iM} \neq 0$ for at least one i =1, 2.

Rejection of the null hypothesis provides evidence that the mean variable value in the monitored lake has changed over time. Plots of the fitted and observed values are then used to identify the changes.

Test 3: Comparison for Monitored Lakes when Test 2c is Rejected and at Least One Reference Lake is Modelled

If the null hypothesis of Test 2c was rejected ($p \leq 0.05$) and at least one reference lake has been retained in the analyses, the fitted patterns of means in that monitored lake are compared to the slope of each of the individual reference lakes that have been modelled. The hypotheses of these tests are as follows:

H_0: $\beta_{1C} = \beta_{1M}$ and $\beta_{2C} = \beta_{2M}$ and $\beta_{1V} = \beta_{1M}$ and $\beta_{2V} = \beta_{2M}$ and $\beta_{1N} = \beta_{1M}$ and $\beta_{2N} = \beta_{2M}$

H_a: $\beta_{iC} \neq \beta_{iM}$ or $\beta_{iV} \neq \beta_{iM}$ or $\beta_{iN} \neq \beta_{iM}$ for at least one i =*1, 2*

Rejection of the null hypothesis provides evidence that the mean variable value in the monitored lake has changed over time relative to a given reference lake. Lack of statistical differences between the slope observed in a given monitored lake and at least two reference lakes indicate natural variability as the underlying cause of temporal trends in the monitored lake. Significant differences between the trend observed in a monitored lake and two or more reference lakes indicate a potential mine effect.

Structure of the Tests

All of the hypothesis tests outlined above are performed using Wald-type chi-square tests based on normal approximation for maximum likelihood estimation. Each null hypothesis can be written as a matrix equation with the form, $L'\beta = 0$, where *L'* denotes the vector of regression coefficients. The Wald theory then states that the quantity:

$$\textbf{(10)} \qquad X^2 = (L'\hat{\beta})(L'\Sigma L)(\hat{\beta}'L)$$

is approximately distributed as a chi-square with degrees of freedom equal to the row rank of L, where $\hat{\beta}$ is the vector of maximum likelihood estimates and Σ is its estimated variance-covariance matrix. The p-values for the tests are computed from the upper-tail probabilities of this distribution.

2.2.4.4 Plots of Observed and Fitted Values

Plots of the observed and fitted values for each variable were constructed to visually compare the values within and among lakes and to aid in the interpretation of the results of the hypothesis tests. On these plots, the observed mean value of the variable for each lake and year are represented by points identified by a separate symbol and colour for each lake. Lakes and stream located downstream of discharge sites (i.e., the LLCF or KPSF) were assigned colors from a red to blue heat palette that correspond to distance from the discharge site, with red representing close proximity to the LLCF or KPSF and blue representing sites that are furthest downstream of the LLCF or KPSF. When one or more observations in a year were below the detection limit, the plotted value is equal to (a + b)/2, with a and b defined as in equation (9). Fitted values of the mean variable are represented with curves matching the colour for each lake. Error bars about the curves represent the 95% confidence intervals for the annual means.

2.2.4.5 Assumptions and Interpretation of Results

Conclusions about the impact of the Ekati Diamond Mine are drawn from the hypothesis testing (regression analysis) and analysis of the observed and fitted values plots for all evaluated lakes and streams outlined in Sections 2.2.3. These analyses allow for the comparison of trends in monitored lakes and streams and reference lakes and streams over time rather than simple comparisons to baseline data only. The assumptions and interpretations of these comparisons reflect those outlined by Wiens and Parker (1995) originally developed for assessing the effects of accidental environmental impacts (e.g., forest fires and oil spills). In their words,

> *Assessment of the impacts of an unplanned environmental accident is based on correlating injury and exposure: if there truly is an effect, injury will increase with exposure.* (Wiens and Parker 1995; pg 1071).

Although no accidental events were observed at the Ekati Diamond Mine, these methods can be applied to monitored lakes and streams of the receiving environment. Exposure of the monitored lakes to containment facility discharge is determined by a combination of two factors: proximity to the containment facility and time. Lakes closest to a containment facility (hydrologically speaking) should have higher exposure levels and show greater effects. Moreover, as more water is released from a containment facility, exposure increases. Consequently, effects stemming from discharge should increase with time. However, historical effects are also possible. In such cases, effects may have stabilised in monitored lakes but historical increases would have stemmed from earlier discharge. Reference lakes are completely disconnected from containment facilities and therefore have no exposure.

The design used in the analysis of the AEMP data fits what Wiens and Parker (1995) term a level-by-time interaction monitoring design. In this type of design, time-series collected from several sites that differ in their levels of exposure are compared. An interaction between temporal trajectories and exposure (i.e., differences in the time-series for different levels of exposure) are taken as evidence of an impact of the accident. Underlying this interpretive approach is the assumption that monitored sites are in a state of dynamic equilibrium. In other words, it is assumed that the temporal trajectories of the means would be the similar in all lakes in the absence of external impacts. This assumption is tested directly using measurements from the reference lakes (i.e., Tests 1a and 1b). These three lakes all receive the same exposure (i.e., no exposure). As such, the trajectory of the means should be the same in all three lakes if the assumption is correct. Test 1a compares the absolute level of a variable over time in each of the three lakes and allows for stronger conclusions. Failure to reject the null hypothesis of Test 1a provides evidence that not only the relative but absolute means of the variable are the same in the reference lakes over time. If the null hypothesis is rejected, then Test 1b compares the trajectories over time in the three lakes, relative to a separate reference level in each lake. If the null hypothesis is rejected for both tests, then the dynamic equilibrium assumption must be rejected.

If the dynamic equilibrium assumption appears to be satisfied, then the second set of tests is used to compare the monitored lakes to the reference lakes. The monitored lakes represent differing levels of exposure to discharge from the containment facility. Differences between the trajectories in the monitored lake and two or more of the reference lakes are evidence of a mine effect. Evidence is strongest when differences follow the gradient of exposure (such that the greatest change is observed in lakes closest to the containment facility and the least change or no change is observed in the reference lakes) and when the magnitude of any differences increases through time. Such evidence would lead to the conclusion that mine effects are present (see Figure 2.2-1).

As discussed in Section 2.2.2, the interpretation of whether an "effect" is an impact on the environment includes an assessment of whether benchmarks (federal guidelines, provincial guidelines or site-specific water quality objectives (SSWQO)) are exceeded and considers biological trends. The minimum detectable difference is calculated (see below) to aid in the determination of whether the value for a given variable has exceeded a benchmark, within a margin of uncertainty.

<u>Minimum Detectable Difference</u>

Although the minimum detectable difference (MDD) can be calculated for each of the tests performed, the values that arise are not easily interpreted because of the complexity of the hypotheses. Instead, MDD were computed for a simplified test. The MDD aids in the determination of whether a benchmark value has been exceeded, and whether an effect is an impact, with greater certainty.

Suppose that for a specific variable there is some fixed benchmark value that is of particular interest, perhaps the Canadian Council of Ministers of the Environment (CCME) water quality guidelines for the protection of aquatic life or SSWQO, and we wish to know if the concentration of the variable in each lake is above or below this value in the final year of monitoring. The MDD computed in the analysis answers the question “How far below (or above) the value would the mean concentration need to be to reliably detect a difference?” Statistically speaking, this is equivalent to asking for the smallest decrease (or increase) from the guideline value that will provide both sufficiently low Type I and Type II Error probabilities for a hypothesis test comparing the guideline and the fitted mean concentration for the final year obtained from the random effects model.

The minimum detectable difference (MDD), *d*, is the smallest decrease (or increase) in the true concentration relative to the benchmark value that will reliably produce a statistically significant difference between the fitted mean and the benchmark value. In the past, the MDD has been used to aid in the determination of whether a benchmark value has been exceeded, and whether an effect is an impact, with greater certainty. In most cases the MDD is not required to interpret results. This was the case for all results in 2013 as fitted means were clearly greater than, or less than water quality benchmark values.

2.2.4.6 Computing

All steps of the analysis were performed using statistical computing package R 2.15.2 (R Development Core Team 2012). Linear mixed effect regression models were fit using the 'lme' function. Tobit regression analysis was conducted using the 'survreg' function available from the survival package. Results from the statistical tests are provided in Part 3 – Statistical Report of the 2013 AEMP Report.

2.2.5 Graphical Analysis

To ensure robustness in the evaluation of effects, three types of graphical analyses were used to aid in the interpretation of statistical results: visual gradient analysis, historical trend analysis, and graphical analysis of non-replicated data.

2.2.5.1 Visual Gradient

The two main point sources for potential water quality effects in the receiving in environment at the Ekati Diamond Mine are discharge from the LLCF into Leslie Lake and discharge from the KPSF into Cujo Lake. Historical data are therefore presented by location within a given Watershed, in order to identify and assess how these two point sources are affecting downstream lakes and streams. Lakes and stream located downstream of discharge sites (i.e., the LLCF or KPSF) are assigned colors from a red to blue heat palette that correspond to distance from the discharge site, with red representing close proximity to the LLCF or KPSF and blue representing sites that are furthest downstream of the LLCF or KPSF. This enables the identification of concentration gradients from the two point sources and allows the overall downstream distance of effects to be determined. All evaluated variables are visually analyzed for spatial gradient effects.

2.2.5.2 Historical Trend

Historical data are presented for all evaluated variables and are used to evaluate temporal trends and to aid in the interpretation of statistical analyses. For example, a statistically significant difference was found between the trend in total aluminum concentrations in Kodiak Lake and the trend of total aluminum concentrations in reference lakes in April and August of 2011. Visual analysis of the historical trend in total aluminum concentrations indicated that total aluminum concentrations in Kodiak Lake had declined from initially high levels to stabilise at current levels over the previous ten years. Thus, the statistical difference in trend in Kodiak Lake was attributed to this decline, which was not observed in reference lakes.

2.2.5.3 Graphical Analysis of Non-replicated Values

Several variables have characteristics that inhibit statistical analyses:

- Dissolved oxygen (DO) and Secchi depth, which are not replicated; and
- Diversity indices, which are the products of data manipulation. Such data manipulation may result in abnormal data characteristics, including non-normal distributions.

Consequently, these variables are subject to graphical analyses only. For these variables, data from 2013 had to appear different from all of the data collected in baseline years to be considered an effect. For example, if values from 2013 appeared different from 1996 but not from 1994 in Koala Watershed lakes or streams, no mine effects were indicated. However, if values from 2013 appeared different from all of the values prior to 1998, it was concluded that mine activities may have affected the variable in question unless similar trends were observed in both monitored and reference lakes. For Secchi depths, estimates of measurement variability from field trials indicate that observer error could introduce as much as 0.5 m variability, which was taken into consideration during the evaluation of effects.

2.2.6 Best Professional Judgment

The evaluation of effects was conducted by experienced and competent scientists who have first-hand knowledge of the aquatic ecosystems present in the Ekati Diamond Mine claim block. Best professional judgment was used in the evaluation of all variables to determine whether a change was 'significant', if a change was a mine effect, and if the effect was having an impact on the aquatic environment. Statistical results and graphical analyses were examined in concert.

2.3 WATER QUALITY BENCHMARKS

As part of the evaluation framework, benchmark values are important in the determination of mine impacts. Water quality benchmarks include both applicable CCME water quality guidelines (CCME 2013) and calculated site specific water quality objectives (SSWQOs; Table 2.3-1). Other water quality benchmark values also exist for total antimony, total barium and total strontium (Haywood and Drinnan 1983; Fletcher et al. 1996; Golder 2011)

Table 2.3-1. The Ekati Diamond Mine Water Quality Benchmarks Used for the AEMP Evaluation of Effects

Variable	Source	Benchmark Value	Notes
Physical/Ion			
pH	CCREM (1987)	6.5 to 9 pH units	
Chloride	SSWQO (Elphick, Bergh, and Bailey 2011)	116.1 * ln(hardness) – 204.1 (where hardness = 10 – 160)	Hardness as mg/L $CaCO_3$;
Sulphate	SSWQO (Rescan 2012f)	$e^{(0.9116 \times \ln(\text{hardness}) + 1.712)}$ (where hardness < 160)	Hardness as mg/L $CaCO_3$
Potassium	SSWQO (Rescan 2012g)	41	
Nutrients/Organics			
Total Ammonia-N	CCME (2001)	Dependent on pH and temperature (see Table 2.3-3)	
Nitrate-N	SSWQO (Health Canada 1987; Rescan 2012e)	$e^{(0.9518 [\ln(\text{hardness})] - 2.032)}$ (where hardness ≤ 160)	Hardness as mg/L $CaCO_3$
Nitrite-N	CCREM (1987)	0.06	
Total Phosphate-P	CCME (2004)	Trigger value or if phosphorus concentrations increase more than 50% over the average level during baseline years (see Table 2.3-4)	
Total Metals			
Antimony	Fletcher et al. 1996	0.02	
Arsenic	CCME (1999)	0.005	
Barium	Haywood and Drinnan (1983)	1	

(continued)

Table 2.3-1. The Ekati Diamond Mine Water Quality Benchmarks Used for the AEMP Evaluation of Effects (completed)

Variable	Source	Benchmark Value	Notes
Total Metals *(cont'd)*			
Boron	CCME (2009)	1.5	
Cadmium	CCME (2014)	$10^{(0.83 \times \log10(\text{hardness}) - 2.46)}/1000$ (with minimum = 0.0004 where hardness = 0-16 and maximum = 0.00037 where hardness > 280)	Hardness as mg/L $CaCO_3$
Copper	CCREM (1987)	$e^{(0.8545 \times \ln(\text{hardness}) - 1.465)} * 0.2/1000$ (where hardness < 180 and 0.004 where hardness ≥ 180; minimum is 0.002 regardless of water hardness)	Hardness as mg/L $CaCO_3$
Molybdenum	SSWQO (Rescan 2012a)	19.38	
Nickel	CCREM (1987)	$e^{(0.76 \times \ln(\text{hardness}) + 1.06)}/1000$ (where hardness = 60 - 180, 0.025 where hardness < 60, and 0.15 where hardness > 180; minimum = 0.025 regardless of water hardness)	Hardness as mg/L $CaCO_3$
Selenium	CCREM (1987)	0.001	
Strontium	Golder (2011)		
Uranium	CCME (2011)	0.015	
Vanadium	SSWQO (Rescan 2012h)	0.03	

Units are mg/L unless otherwise specified.

The CCME water quality guidelines for the protection of freshwater aquatic life provide useful benchmarks for evaluation of the Ekati Diamond Mine's aquatic environment. For the purpose of the evaluation of effects, the following water quality variables were compared to the applicable CCME guideline value: pH, total ammonia-N, nitrite, total phosphate-P, total arsenic, total boron, total cadmium, total copper, total nickel, total selenium, and total uranium (Table 2.3-1). The guideline value for copper and nickel is hardness-dependent with a minimum value of 0.002 mg/L for copper and 0.025 mg/L for nickel. The CCME water quality guideline for total ammonia-N is a function of pH and temperature and corresponds to total ammonia concentrations as NH_3-N (Table 2.3-2).

Table 2.3-2. Total Ammonia-N Values (as NH_3-N) as a Function of pH and Temperature

	pH							
Temperature (°C)	6.0	6.5	7.0	7.5	8.0	8.5	9.0	10.0
0	190	60	19	6.0	1.9	0.62	0.21	0.035
5	126	40	13	4.0	1.3	0.41	0.14	0.028
10	84	27	8.5	2.7	0.86	0.28	0.10	0.024
15	57	18	5.7	1.8	0.59	0.20	0.073	0.021
20	39	13	4.0	1.3	0.41	0.14	0.055	0.020
25	28	8.7	2.8	0.89	0.29	0.10	0.044	0.018
30	19	6.2	2.0	0.63	0.21	0.077	0.035	0.017

Units are mg/L.
The values presented are equivalent to an unionized ammonia concentration of 0.019 mg/L as NH_3.
Values outside of the shaded area should be used with caution owing to a lack of toxicity data to accurately determine toxic effects at the extreme of these ranges (CCME 2001).

The benchmark for phosphate-P was established using the Canadian Guidance Framework for the Management of Phosphorus in Freshwater Systems (CCME 2004; Environment Canada 2004).

This Framework uses a tiered approach where predefined trigger ranges are based on the trophic status for the lakes being addressed (Table 2.3-3). The trigger ranges are based on the range of total phosphate-P concentrations in water that define the reference trophic status for a site. These ranges are therefore system-specific.

Table 2.3-3. Phosphorus Trigger Ranges for Lakes

Trophic Level	Total Phosphate-P (mg/L)
Ultra-oligotrophic	< 0.004
Oligotrophic	0.004 - 0.01
Mesotrophic	0.01 - 0.02
Meso-eutrophic	0.02 - 0.035
Eutrophic	0.035 - 0.10
Hypereutrophic	> 0.10

The Framework requires further assessment if the upper values of the trigger range is exceeded or total phosphate-P concentrations have increased more than 50% over the average level during baseline years. This 50% increase was deemed by the Ontario Ministry of Environment (1994) as an acceptable increase, beyond which deterioration of water quality from excessive phosphorus levels was observed in Pre-Cambrian Shield lakes. It was also deemed sufficient to protect Arctic lakes (Environment Canada 2004).

Based on the Framework the upper trigger range value or the mean baseline + 50% for the open water season are provided in Table 2.3-4 as benchmarks for the management of phosphorus at the Ekati Diamond Mine in lakes downstream of the LLCF and KPSF.

Table 2.3-4. Total-Phosphate-P Benchmark Concentrations, AEMP Lakes

Lake	Benchmark Value (mg/L)
Nanuq	0.0025
Counts	0.01
Vulture	0.0043
Grizzly	0.01
Kodiak	0.0180
Leslie	0.0096
Moose	0.0077
Nema	0.0091
Slipper	0.01
Lac de Gras (S2 and S3)	0.0054
Cujo	0.01
Lac du Sauvage (LdS1 and LdS2)	0.00069

Site specific water quality objectives used in the 2013 AEMP Evaluation of Effects have been developed at the Ekati Diamond Mine for chloride, sulphate, potassium, nitrate-N, molybdenum, and vanadium (Elphick, Bergh, and Bailey 2011; Rescan 2012a, 2012e, 2012f, 2012g, 2012h). SSWQOs for the Ekati Diamond Mine have been established through a review of water quality guidelines in Canada and the United States, literature in the Ecotox database, and through experimentation using species that are present or closely related to those that are present at the Ekati Diamond Mine (Table 2.3-1). These

SSWQOs provide benchmarks that are ecologically relevant, scientifically defensible, and provide reasonable estimates of concentrations above which the risk of adverse effects may become elevated.

The SSWQO for chloride applies across a range of water hardness values, from 10 to 160 mg/L as $CaCO_3$; a guideline was not established at levels higher than this because the dataset used to establish the SSWQO was limited to this range of water hardness values (Elphick, Bergh, and Bailey 2011). Similarly, a site-specific water quality objective has been developed for sulphate at the Ekati Diamond Mine across a range of hardness values up to 160 mg/L as $CaCO_3$ (Rescan 2012f). The SSWQO for Nitrate-N also was established for the Ekati Diamond Mine receiving waters in 2012 and is dependent on a range of hardness values (Rescan 2012e). The SSWQO for potassium, molybdenum, and vanadium are not hardness-dependent.

3. Evaluation of Effects: Koala Watershed and Lac de Gras

3. Evaluation of Effects: Koala Watershed and Lac de Gras

3.1 PHYSICAL LIMNOLOGY

3.1.1 Variables

Three physical limnology variables were evaluated for potential effects caused by mine activities: temperature, under-ice dissolved oxygen (DO) concentrations, and open water season Secchi depths.

Under-ice DO concentrations were evaluated as opposed to open water season concentrations because they often represent the '*worst-case scenario*'. DO concentrations are generally lowest during the winter because ice cover restricts oxygen diffusion into the water column from the atmosphere, and because of aerobic microbial activity in the sediment. The amount of sunlight penetrating into the water column is also limited by snow and ice cover, thus restricting phytoplankton growth and the production of DO by photosynthesis. Low DO concentrations can inhibit growth and reproduction in zooplankton, benthic invertebrates, and fish, and may lead to mortalities if low DO impedes respiration. The CCME guideline for DO concentrations for cold-water organisms is 9.5 mg/L for early life stages and 6.5 mg/L for other life stages (CCME 2013).

Secchi depths are a measure of water clarity. A reduction in Secchi depth generally indicates increased turbidity due to increases in phytoplankton or other suspended particulates.

3.1.2 Dataset

Under-ice dissolved oxygen and temperature profiles were collected in March, April, or May of each year for the evaluation of effects (Table 3.1-1). Secchi depths were measured during August sampling surveys (Table 3.1-2).

3.1.3 Results and Discussion

3.1.3.1 Under-ice Dissolved Oxygen

Summary: Under-ice temperature profiles have changed through time in Kodiak, Grizzly, Leslie, Nema, and possibly Moose lakes. In Kodiak Lake, the observed changes in the under-ice temperature profiles likely stem from attempts to increase under-ice dissolved oxygen using aerators from 1997 to 2007. The causes of the temporal changes in under-ice temperature profiles downstream of the LLCF and in Grizzly Lake are unclear at this time. There is some evidence of a cooling trend in reference lakes in recent years, though the patterns are not as pronounced in the reference lakes. Thermal stratification resembling that observed in Grizzly Lake was also observed in Vulture Lake in 2013, the one reference lake that is a similar depth to Grizzly Lake, suggesting that changes in thermal profiles may reflect natural climactic variability rather than mine effects. There were no similar changes in the associated DO profiles in Grizzly Lake from 2010-2013. With the exception of decreased under-ice DO concentrations observed in Kodiak Lake over time (reflection of earlier adaptive management to increase under-ice DO concentrations), no mine effects were detected with respect to DO concentrations in the Koala Watershed or Lac de Gras.

Table 3.1-1. Dataset Used for Evaluation of Effects on Under-ice Dissolved Oxygen and Temperature Profiles in Koala Watershed Lakes and Lac de Gras

Year	Nanuq	Counts	Vulture	Grizzly	Kodiak	Leslie	Moose	Nema	Slipper	S2	S3
1994	-	-	-	-	Mar-26	-	-	-	-	-	-
1995	-	-	-	-	-	-	-	-	-	-	-
1996	-	-	-	-	-	-	-	-	-	-	-
1997	-	-	-	-	-	-	-	-	-	-	-
1998	Apr-19	Apr-19	Apr-15	Apr-15	Apr-15	-	Apr-16	-	Apr-19	-	-
1999	Apr-17	Mar-10	Mar-24	Mar-8	Apr-19	-	Apr-17	Apr-17	Apr-17	Mar-25	-
2000	Mar-16	Mar-17	Mar-23	-	Apr-19	-	May-2	May-2	May-2	Mar-22	-
2001	Apr-14	Apr-15	Apr-14	-	Apr-24	-	Apr-15	Apr-15	Apr-15	-	Apr-15
2002	Apr-23	Apr-23	Apr-20	-	Apr-18	-	Apr-20	Apr-18	Apr-23	Apr-23	Apr-23
2003	Apr-12	Apr-13	Apr-14	Apr-16	Apr-17	Apr-15	Apr-15	Apr-14	Apr-15	Apr-15	Apr-15
2004	Apr-18	Apr-17	Apr-18	-	Apr-19	Apr-16	Apr-16	Apr-17	Apr-17	Apr-17	Apr-17
2005	-	-	-	-	Apr 28	Apr-29	Apr-26	Apr-25	Apr-26	Apr-26	Apr-26
2006	Apr-20	Apr-22	Apr-21	Apr-24	Apr-24	Apr-23	Apr-23	Apr-23	Apr-18	Apr-18	Apr-18
2007	-	-	-	-	-	-	-	-	-	-	-
2008	Apr-27	May-3	May-3	May-5	May-6	May-6	May-6	May-5	May-4	May-4	May-4
2009	May-18	May-17	Apr-28	Apr-28	May-2	May-2	Apr-29	Apr-29	May-17	May-18	May-17
2010	Apr-14	Apr-14	Apr-14	Apr-12	Apr-16	Apr-16	Apr-14	Apr-15	Apr-15	Apr-15	Apr-15
2011	Apr-25	Apr-26	Apr-28	Apr-28	Apr-28	Apr-27	Apr-27	Apr-27	Apr-27	Apr-27	Apr-27
2012	Apr-20	Apr-17	Apr-18	Apr-16	Apr-22	Apr-24	Apr-24	Apr-23	Apr-25	Apr-24	Apr-24
2013	Apr-26	Apr-26	Apr-23	Apr-22	Apr-28	Apr-24	Apr-24	Apr-24	Apr-27	Apr-27	Apr-27

Dashes indicate no data were available.

Table 3.1-2. Dataset Used for Evaluation of Effects on Secchi Depths in Koala Watershed Lakes and Lac de Gras

Year	Nanuq	Counts	Vulture	Grizzly	Kodiak	Leslie	Moose	Nema	Slipper	S2	S3
1994*	-	-	Aug-20	Aug-20	-	-	Aug-20	-	Aug-20	Aug-20	-
1995*	-	-	Aug-10	-	Aug-10	-	-	Aug-10	Aug-10	-	-
1996*	-	-	Jul-28	-	Jul-26	-	Jul-27	Jul-26	Jul-26	-	-
1997*	Aug-4	Aug-14	Aug-5	Aug-8	Aug-9	-	Aug-10	Aug-10	Aug-11	Aug-12	-
1998	Aug-4	Aug-14	Aug-7	Aug-8	Aug-10	-	Aug-11	Aug-11	Aug-12	Aug-13	Aug-13
1999	Aug-7	Aug-8	Aug-6	Aug-6	Aug-10	-	Aug-7	Aug-10	Aug-9	Aug-11	Aug-11
2000	Aug-4	Aug-1	Aug-4	Aug-3	Jul-29	-	Jul-30	Jul-30	Jul-31	Aug-3	Aug-3
2001	Aug-1	Jul-30	Aug-2	-	Jul-28	-	Aug-3	Aug-3	Jul-29	Jul-29	Jul-29
2002	Aug-1	Aug-7	Aug-3	-	Aug-2	-	Aug-5	Aug-4	Aug-6	Aug-4	Aug-4
2003	Aug-9	Aug-7	Aug-4	Aug-8	Aug-8	Aug-3	Aug-9	Aug-3	Aug-7	Aug-5	Aug-5
2004	Aug-10	Aug-13	Aug-9	-	Aug-7	Aug-9	Aug-10	Aug-9	Aug-12	Aug-9	Aug-9
2005	Aug-1	Aug-7	Jul-31	Aug-7	Aug-3	Aug-4	Aug-9	Aug-9	Aug-5	Aug-5	Aug-5
2006	Aug-2	Aug-4	Aug-2	Aug-7	Aug-1	Aug-6	Aug-5	Aug-5	Aug-4	Aug-4	Aug-4
2007	Aug-11	Aug-6	Aug-12	Aug-4	Aug-4	Aug-13	Aug-7	Aug-11	Aug-10	Aug-6	Aug-6
2008	Aug-8	Jul-31	Jul-29	Jul-27	Jul-27	-	Jul-29	Jul-29	Jul-29	Aug-7	Aug-7

(continued)

Table 3.1-2. Dataset Used for Evaluation of Effects on Secchi Depths in Koala Watershed Lakes and Lac de Gras (completed)

Year	Nanuq	Counts	Vulture	Grizzly	Kodiak	Leslie	Moose	Nema	Slipper	S2	S3
2009	Jul-30	Aug-1	Jul-30	Aug-2	Aug-8	Aug-5	Jul-30	Jul-30	Aug-3	Jul-31	Jul-31
2010	Aug-5	Aug-7	Aug-5	Aug-4	Aug-5	Aug-3	Aug-3	Aug-5	Aug-6	Aug-5	Aug-5
2011	Aug-2	Aug-5	Aug-5	Aug-1	Aug-5	Aug-2	Aug-3	Aug-5	Aug-3	Aug-4	Aug-4
2012	Aug-1	Aug-8	Aug-12	Aug-2	Aug-6	Aug-8	Aug-9	Aug-7	Aug-8	Aug-3	Aug-2
2013	Aug-3	Aug-1	Aug-1	Jul-31	Aug-6	Aug-1	Aug-5	Aug-6	Aug-5	Aug-2	Aug-2

Dashes indicate no data were available
** = Data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison.*

No statistical analyses could be performed on under-ice dissolved oxygen or temperature profiles because they are not replicated. Thus, graphical analysis and best professional judgment were the primary methods used in the evaluation of effects.

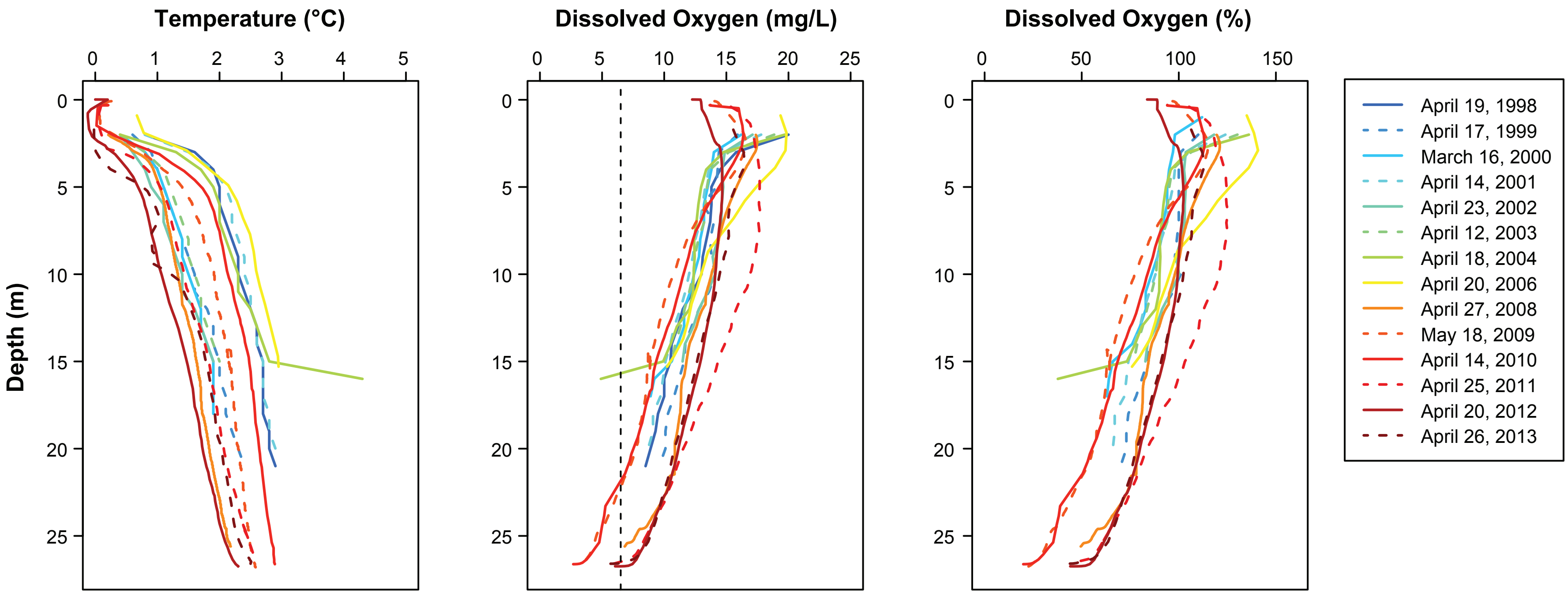
Dissolved oxygen concentrations measured in late April of 2013 were generally within the historical ranges observed in each lake (Figures 3.1-1 and 3.1-2). Dissolved oxygen concentrations were typically greatest just below the surface ice and declined with depth and as temperature increased. Under-ice DO concentrations were greater than the CCME guideline value of 6.5 mg/L throughout the water column in Nanuq, Vulture, Moose, and Slipper lakes and at sites S2 and S3 in Lac de Gras (CCME 2013). DO was less than the 6.5 mg/L CCME guideline in Counts (at depths greater than roughly 9.5 m), Grizzly (at depths greater than roughly 36.5 m), Leslie (at depths greater than 10.5 m), and Nema (at depths greater than 3 m) lakes. However, 2013 under-ice DO profiles, including DO concentrations at deeper depths, were similar to those observed historically in Counts, Leslie, Grizzly, and Nema lakes with no apparent temporal trends. Furthermore, a gradual decline in DO through the ice-covered period is expected: Microbial decomposition and heterotrophic respiration continue to consume oxygen during the ice-covered period, but the production of oxygen is reduced because temperature and light penetration limit the photosynthetic activity of phytoplankton. Ice cover also excludes the dissolution of atmospheric oxygen into the water column. Observed DO profiles from reference lakes indicate that DO concentrations in deeper sections are often less than the CCME threshold during the ice-covered period (Figures 3.1-1a-c). Thus, no mine effects were detected with respect to under-ice DO concentrations in these lakes.

In contrast, DO profiles in Kodiak Lake have changed through time, beginning in 2006 (Figure 3.1-2b). These changes coincide with changes in under-ice temperature over the same period. Specifically, surface layers in Kodiak Lake have become warmer in recent years (Figure 3.1-2b). The trend toward warmer surface waters began in 2006, became more pronounced in 2008, and stabilised through 2013 (Figure 3.1-2b). DO concentrations were less than CCME guidelines throughout most of the water column in 2013, but were greater than historical concentrations at deeper depths (Figure 3.1-2b). Changes in DO and temperature profiles in Kodiak Lake are likely related to attempts to increase under-ice dissolved oxygen using aerators (beginning in 1997). Historically, under-ice DO concentrations in Kodiak Lake have been monitored at regular intervals from February to April, with aeration initiated if results indicated that DO was low. The change in the shape of the under-ice temperature and DO profile in Kodiak Lake corresponds to the first year in which aerators were no longer used (2007). Thus the more recent, stratified temperature and DO profiles likely represent undisturbed conditions in Kodiak Lake, since aerators would cause mixing of the water column which would result in homogeneity of temperature and dissolved oxygen in the water column.

Figure 3.1-1a

Under-ice Dissolved Oxygen and Temperature Profiles for AEMP Reference Lakes, 1998 to 2013

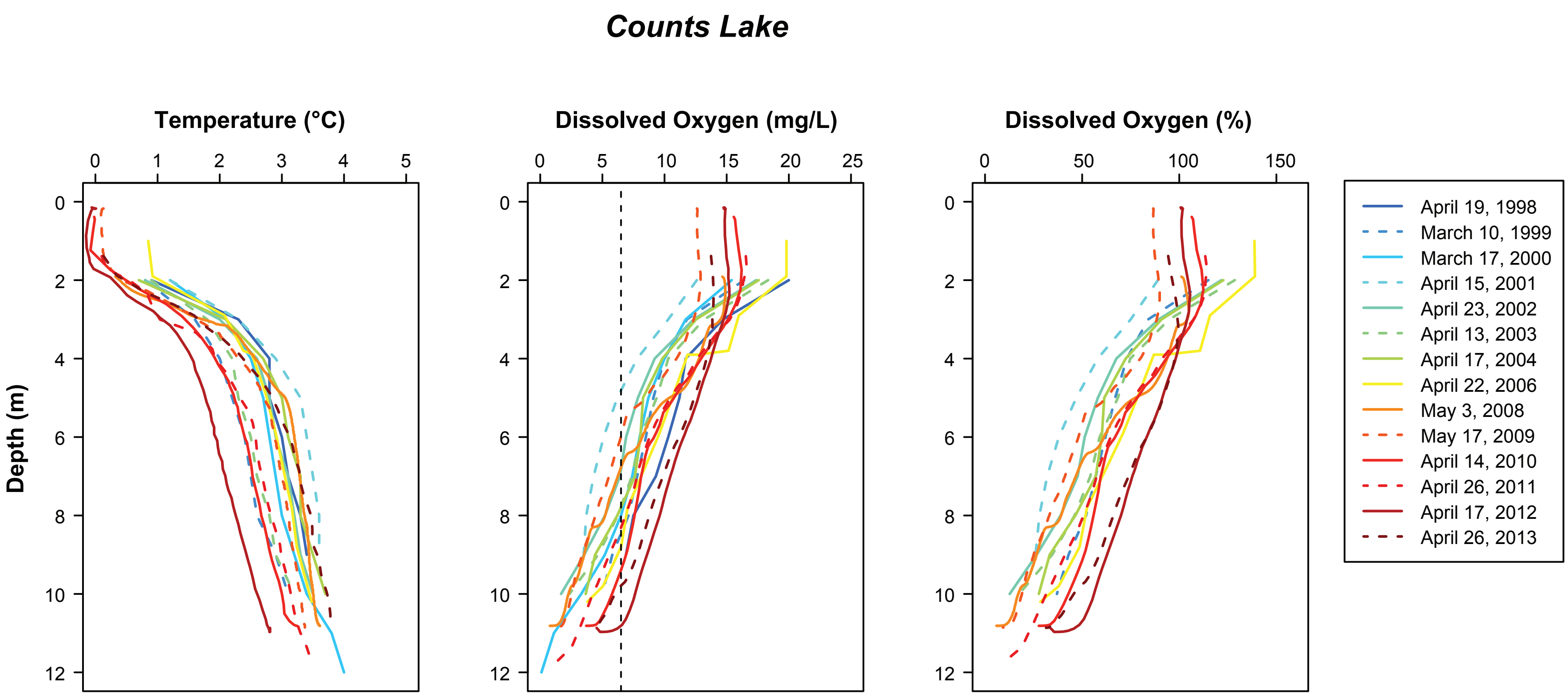
Nanuq Lake



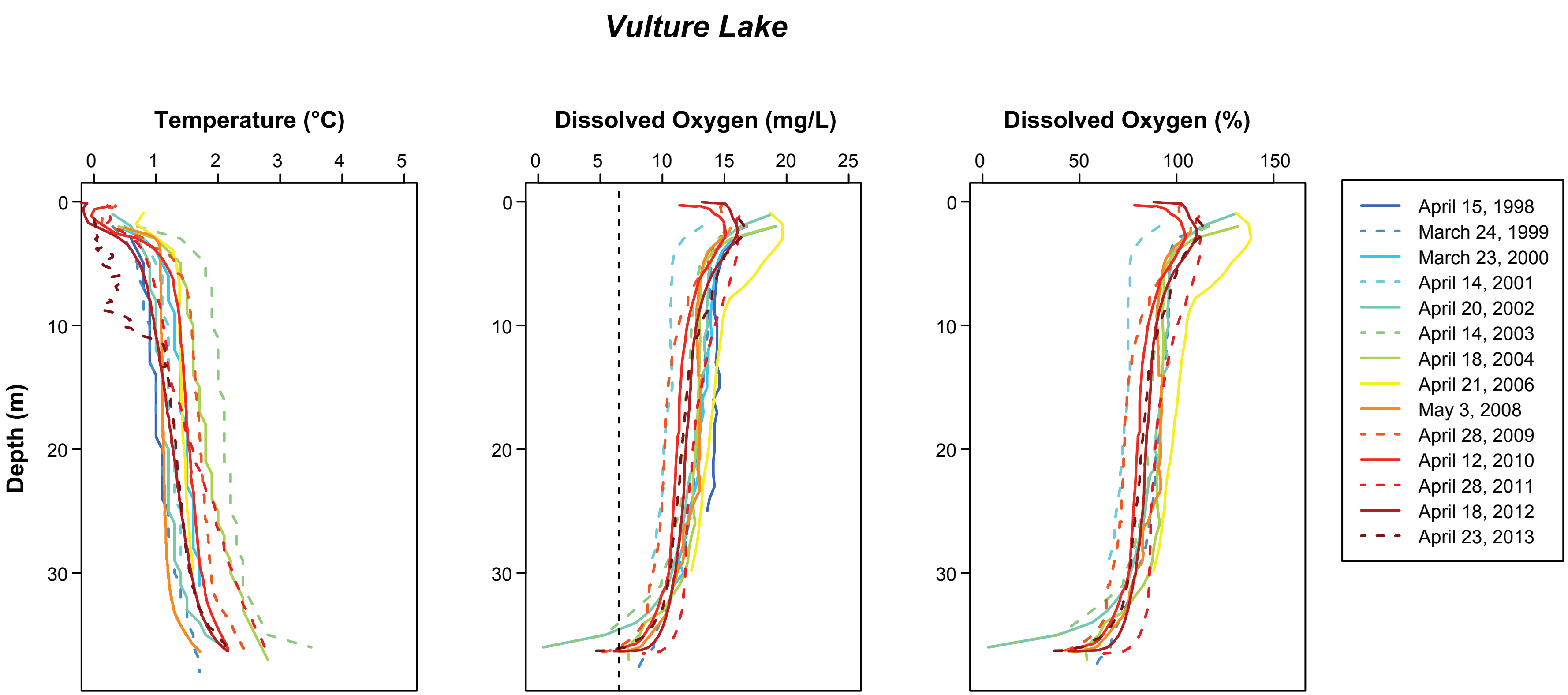
Note: Data collected and supplied by DDEC.
Vertical dashed line represents the CCME guideline for dissolved oxygen (6.5 mg/L).

Figure 3.1-1b

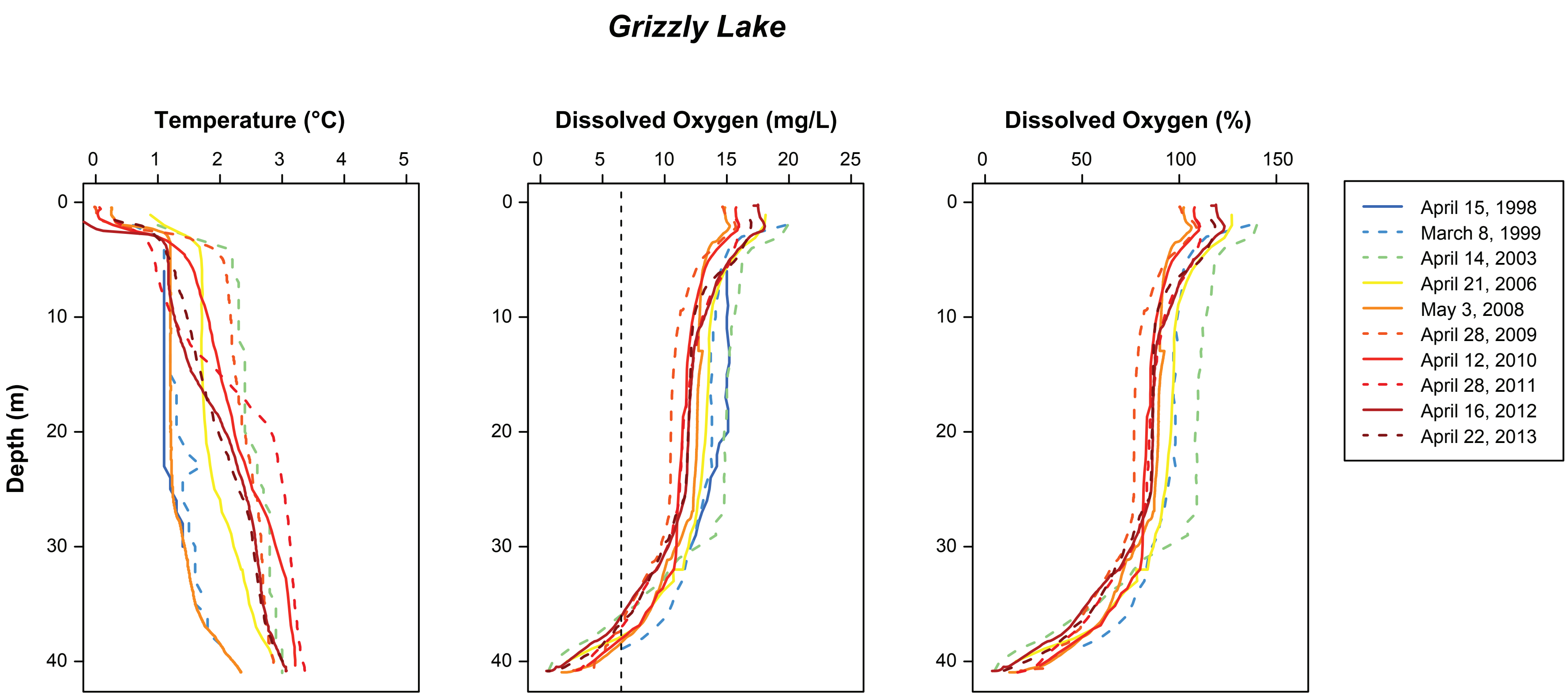
Under-ice Dissolved Oxygen and Temperature Profiles for AEMP Reference Lakes, 1998 to 2013



Note: Data collected and supplied by DDEC.
Vertical dashed line represents the CCME guideline for dissolved oxygen (6.5 mg/L).

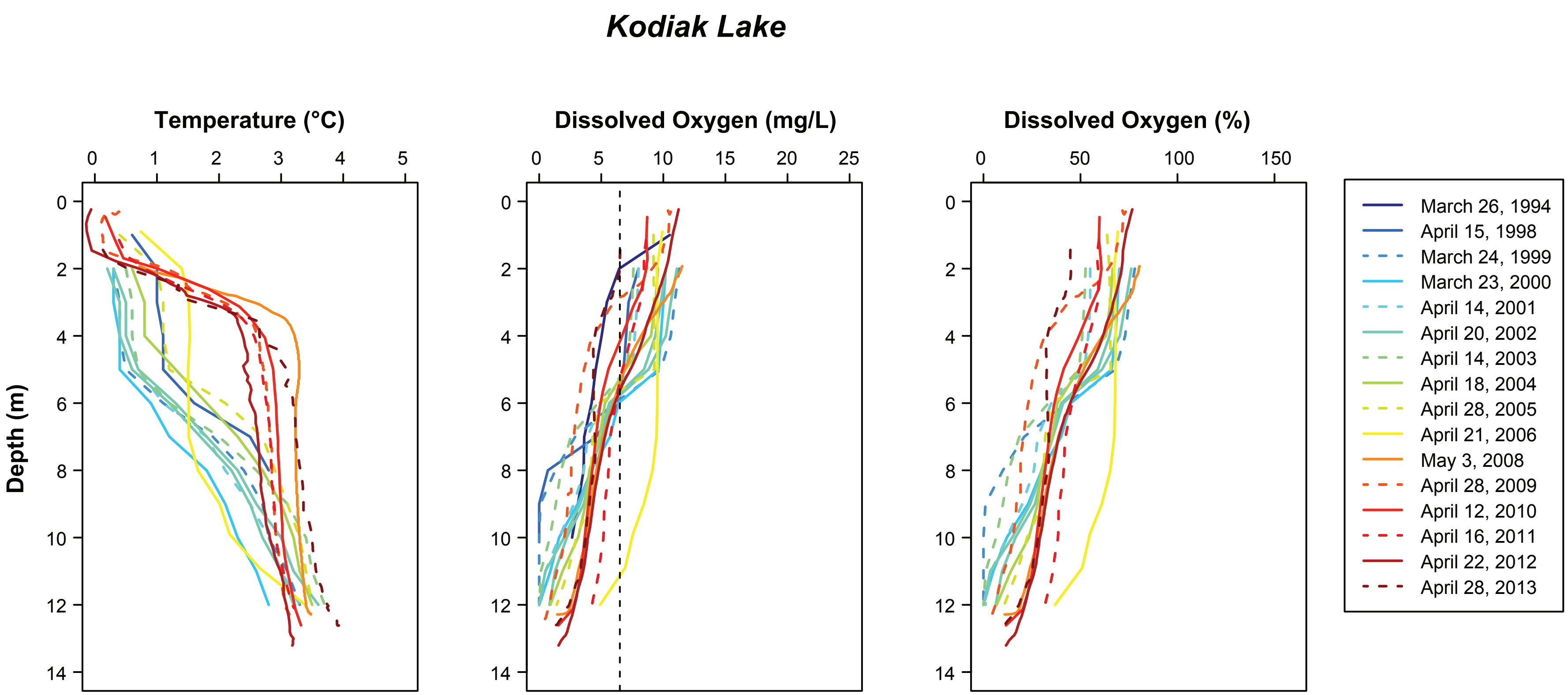


Note: Data collected and supplied by DDEC.
Vertical dashed line represents the CCME guideline for dissolved oxygen (6.5 mg/L).



Note: Data collected and supplied by DDEC.
Vertical dashed line represents the CCME guideline for dissolved oxygen (6.5 mg/L).

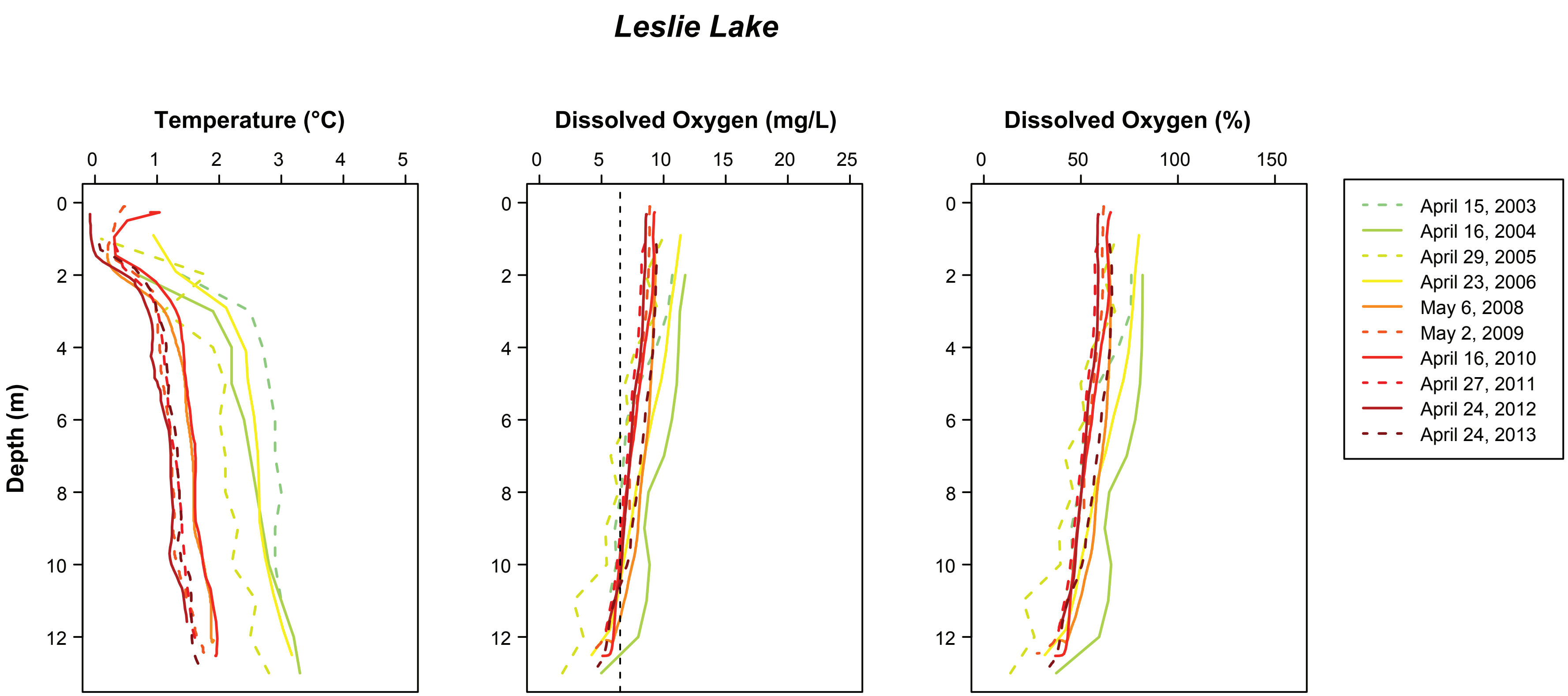
Under-ice Dissolved Oxygen and Temperature Profiles for Koala Watershed Lakes and Lac de Gras, 1994 to 2013



Note: Data collected and supplied by DDEC.
Vertical dashed line represents the CCME guideline for dissolved oxygen (6.5 mg/L).

Figure 3.1-2c

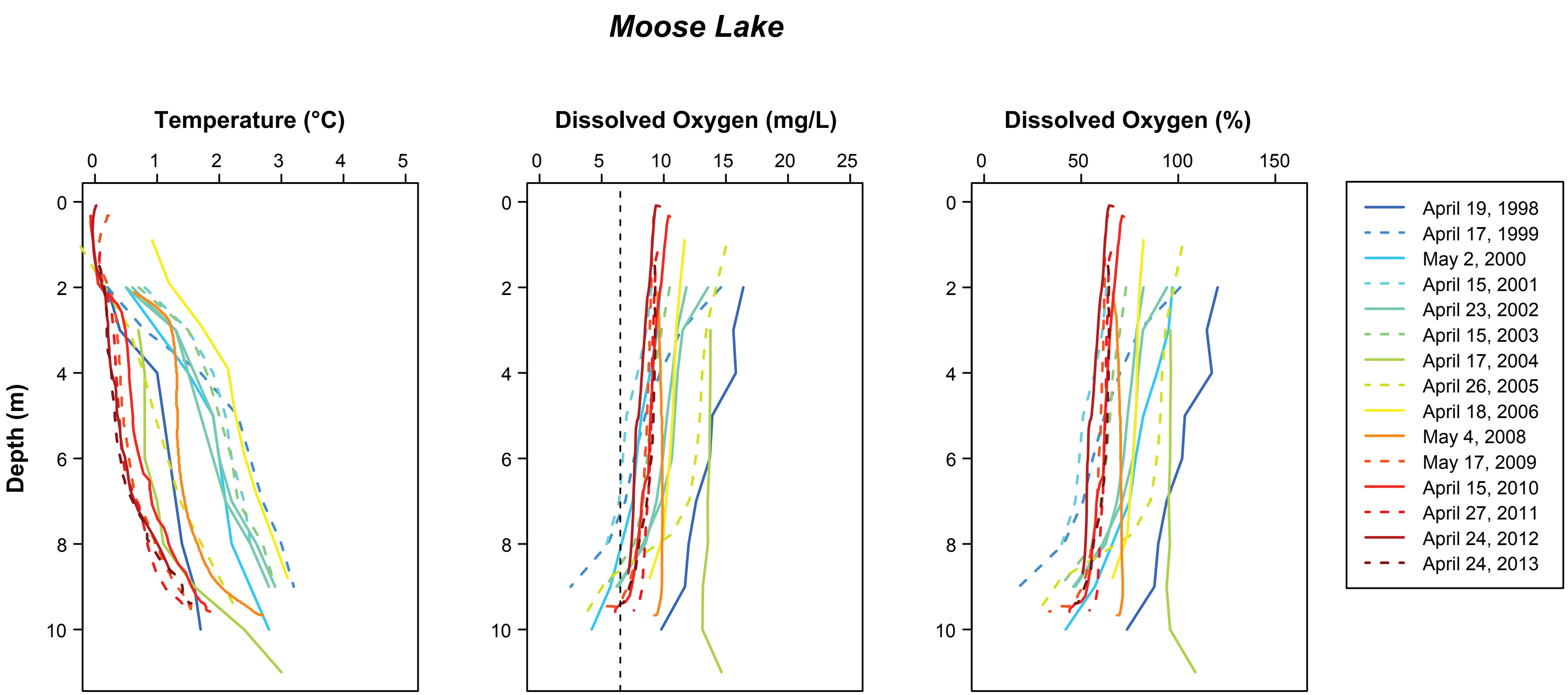
Under-ice Dissolved Oxygen and Temperature Profiles for Koala Watershed Lakes and Lac de Gras, 1994 to 2013



Note: Data collected and supplied by DDEC.
Vertical dashed line represents the CCME guideline for dissolved oxygen (6.5 mg/L).

Under-ice Dissolved Oxygen and Temperature Profiles for Koala Watershed Lakes and Lac de Gras, 1994 to 2013

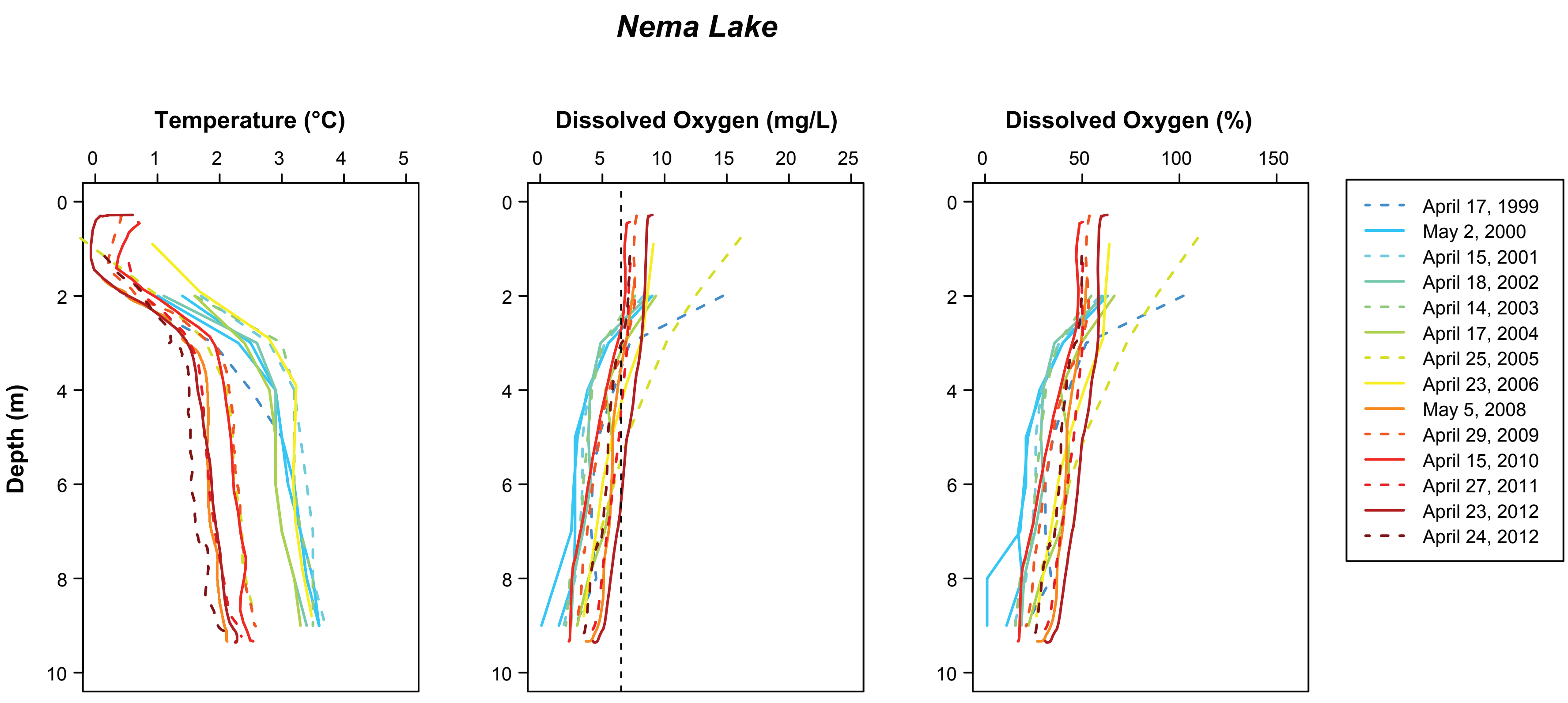
ERM



Note: Data collected and supplied by DDEC.
Vertical dashed line represents the CCME guideline for dissolved oxygen (6.5 mg/L).

Figure 3.1-2e

Under-ice Dissolved Oxygen and Temperature Profiles for Koala Watershed Lakes and Lac de Gras, 1994 to 2013

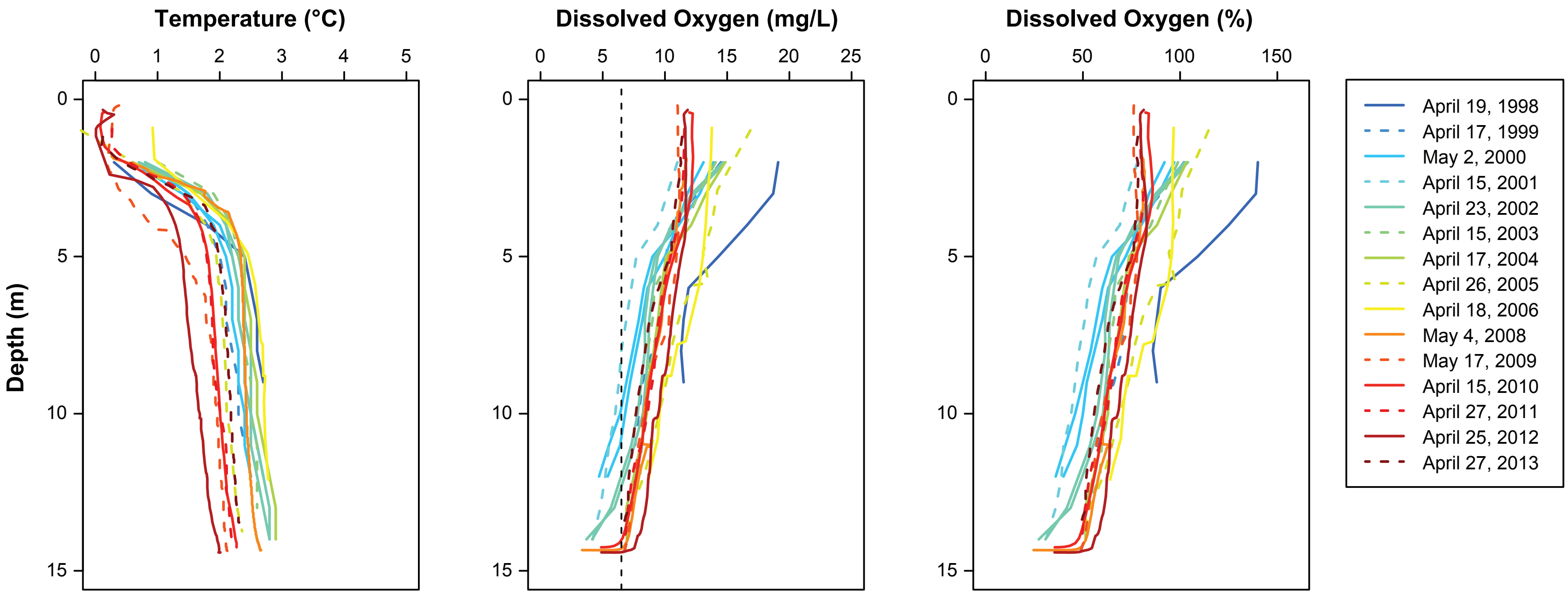


Note: Data collected and supplied by DDEC.
Vertical dashed line represents the CCME guideline for dissolved oxygen (6.5 mg/L).

Under-ice Dissolved Oxygen and Temperature Profiles for Koala Watershed Lakes and Lac de Gras, 1994 to 2013

ERM

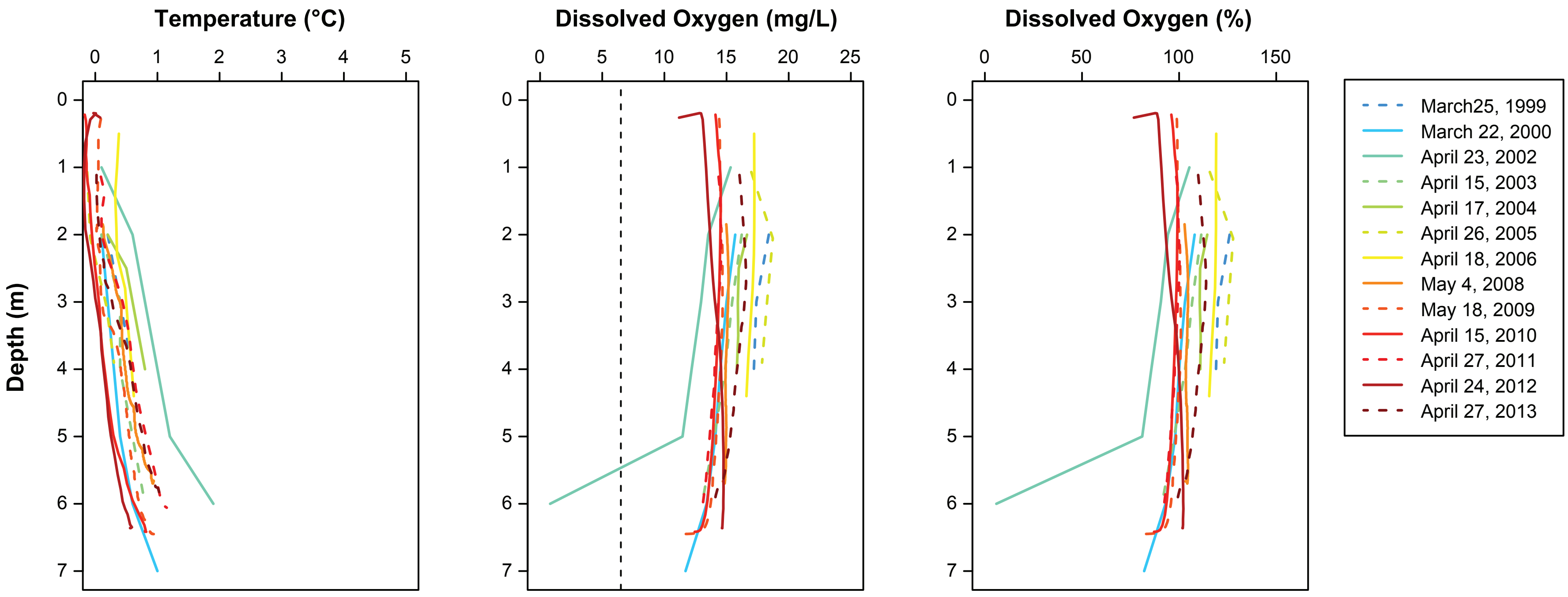
Slipper Lake



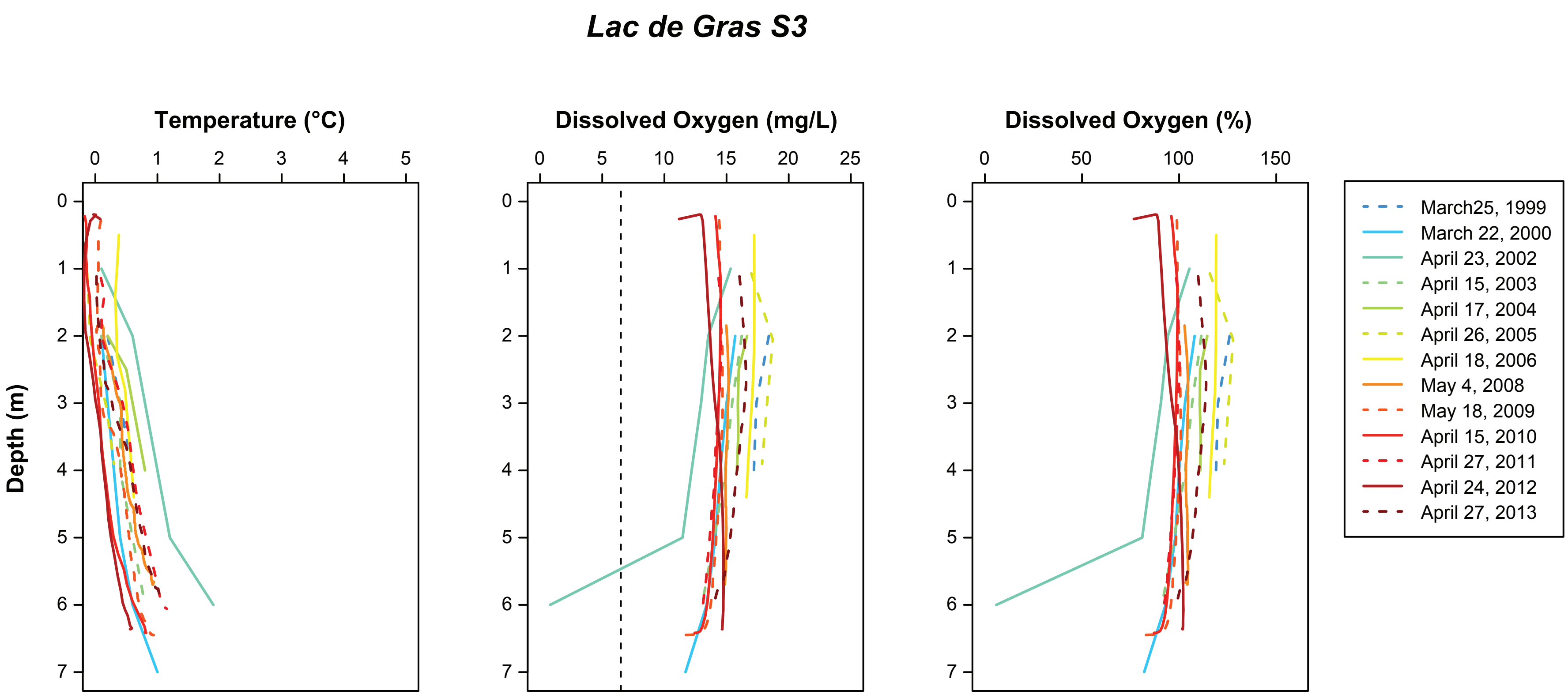
Note: Data collected and supplied by DDEC.
Vertical dashed line represents the CCME guideline for dissolved oxygen (6.5 mg/L).

Under-ice Dissolved Oxygen and Temperature Profiles for Koala Watershed Lakes and Lac de Gras, 1994 to 2013

Lac de Gras S2



Note: Data collected and supplied by DDEC.
Vertical dashed line represents the CCME guideline for dissolved oxygen (6.5 mg/L).



Note: Data collected and supplied by DDEC.
Vertical dashed line represents the CCME guideline for dissolved oxygen (6.5 mg/L).

The over-winter DO monitoring program in Kodiak Lake ceased after 2012, when it was determined that DO concentrations had likely returned to baseline concentrations (BHP Billiton 2012). It is unclear why under-ice DO concentrations were low throughout the water column in late April 2013. Recently, slope stabilizing construction activities took place in the PDC, which flows into Kodiak Lake (see Section 1.2 in Rescan 2012c). However, no construction occurred in the PDC during the winter of 2013 (see Section 1.3.1). Thus, decreased under-ice DO concentrations in Kodiak Lake are likely unrelated to upstream mine activities. Although DO concentrations in Kodiak Lake returned to historical levels and were greater than the CCME guideline in August (see Section 3.3.2 of Part 2 – Data Report), low under-ice DO concentrations suggest that over-winter monitoring of Kodiak Lake should be resumed.

Under-ice temperature profiles in some of the monitored lakes in the Koala watershed and Lac de Gras have been variable through time. Specifically, temperature profiles from Leslie and Nema lakes have shown a trend toward cooling, at all depths, through time (Figures 3.1-2c,and e). Under-ice temperature profiles have also shown some indication of a cooling trend in recent years in Moose Lake (Figure 3.1-2d); however, temperature profiles have been more variable in Moose Lake than in other lakes, likely owing to its comparatively small volume. There is some evidence of a general cooling trend, at all depths, in two of the reference lakes (i.e., Nanuq and Vulture lakes) in recent years (Figures 3.1-1a, and c). In particular, the top 10 m of Vulture Lake was markedly cooler in 2013 (Figure 3.1-1c) and 2013 marked the first year in which Vulture Lake showed signs a thermal stratification. The third reference lake, Counts Lake, has also shown signs of cooling in recent years, but returned to historical temperatures in 2013 (Figure 3.1-1b). Overall, the trends in reference lakes suggest that shifts in temperature profiles in monitored lakes may reflect natural climatic variability rather than mine effects. Temperature profiles in Slipper Lake and Lac de Gras (both sites S2 and S3) were similar to those observed in previous years, with water temperature warming gradually from the surface to the bottom of the lakes (Figures 3.1-2f-g).

In Grizzly Lake, under-ice temperature profiles from 2010-2013 have differed markedly from previous years (Figure 3.1-2a). Surface waters in Grizzly Lake were cooler from 2010-2012 than in previous years. In addition, under-ice temperatures showed a pattern of increase with increasing depth from 2010-2012, rather than remaining homogeneous throughout the water column as in previous years. Although surface temperatures in Grizzly Lake have become warmer in 2013, a similar pattern of increasing temperature with depth is observed (Figure 3.1-2a). There were no similar changes in the associated DO profiles in Grizzly Lake from 2010-2013 (Figure 3.1-2a). Grizzly Lake has been the source of potable water for the main camp since baseline years with no changes to pumping equipment or cycles of water withdrawal since 2004, when water withdrawal was at its peak. It is unclear why the observed thermal profile in Grizzly Lake may have changed in recent years however, thermal stratification was also observed in Vulture Lake in 2013, the one reference lake that is a similar depth to Grizzly Lake, suggesting that changes in thermal profiles may reflect natural climactic variability, rather than mine effects.

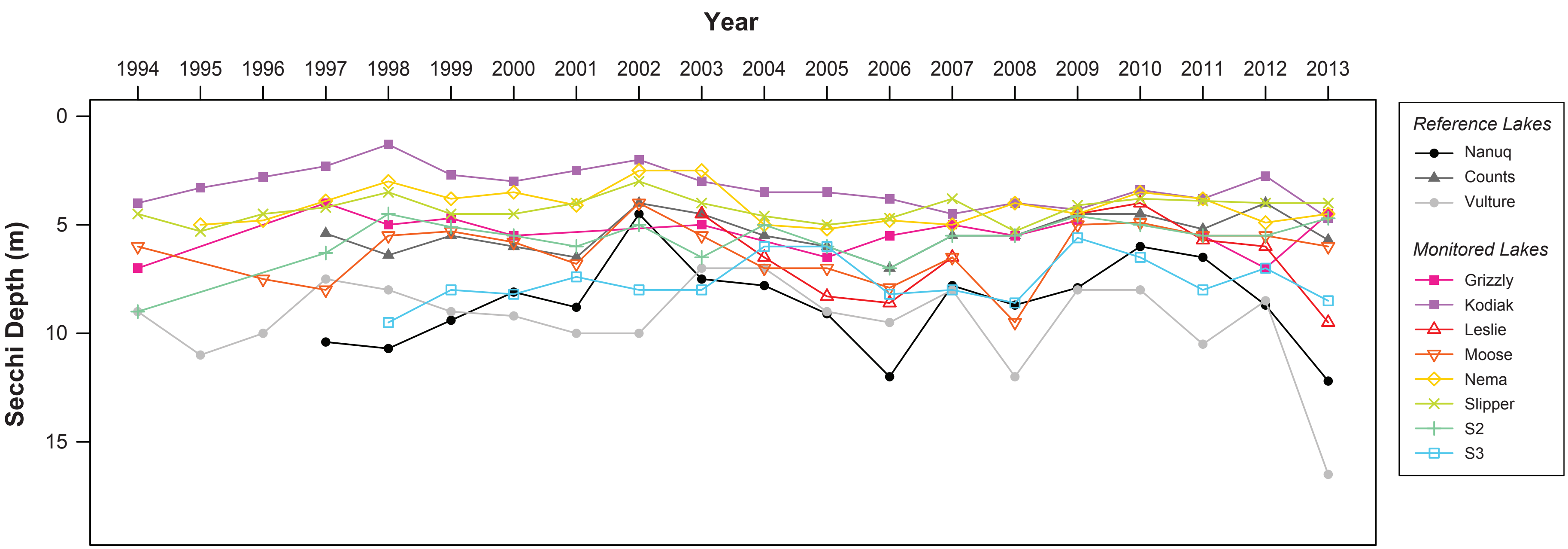
3.1.3.2 Secchi Depth

Secchi depth is an indicator of underwater light conditions in lakes. It can be used as an indicator of changes in water quality or plankton density. Graphical analysis and best professional judgment were used to evaluate if a significant change in Secchi depth occurred in monitored lakes of the Koala Watershed and Lac de Gras (Figure 3.1-3). A value of ± 0.5 m was used as an estimate of error due to sampler bias for interpreting graphical results.

Taking into account estimate error for each year, observed August 2013 Secchi depths were similar to those observed in baseline years in all monitored lakes (Figure 3.1-3). Thus, no mine effects were detected with respect to Secchi depth in monitored lakes of the Koala Watershed or Lac de Gras.

Figure 3.1-3

August Secchi Depths for Koala Watershed Lakes and Lac de Gras, 1994 to 2013



3.2 LAKE AND STREAM WATER QUALITY

3.2.1 Variables

Monitoring water quality in the receiving environment is important for understanding how mining activities may be affecting the watershed. Twenty two water quality variables were evaluated for potential mine effects in lakes and streams in the Koala and King Cujo Watersheds. These included physical variables and anions (pH, total alkalinity, hardness, chloride, sulphate, potassium), nutrients (total ammonia-N, nitrite, nitrate, total phosphate-P, total organic carbon), and total metals (antimony, arsenic, barium, boron, cadmium, molybdenum, nickel, selenium, strontium, uranium, and vanadium). In the King-Cujo Watershed, a 23rd variable, total copper, was also evaluated.

CCME guidelines for the protection of aquatic life exist for 11 of the evaluated water quality variables, including pH, total ammonia-N, nitrite-N, total phosphate-P, total arsenic, total boron, total cadmium, total copper, total nickel, total selenium, and total uranium (see Section 2.3; CCME 2013). In addition, DDEC has established SSWQO for six of these variables, including chloride, sulphate, potassium, nitrate-N, total molybdenum, and total vanadium (see Section 2.3). Other water quality benchmark values also exist for total antimony, total barium and total strontium (see Table 2.3-1 in Section 2.3)

General Physical Variables and Anions

pH

pH plays a major role in the chemical speciation of many metals, their solubility in water, and their overall bioavailability. Thus, pH influences both the availability of nutrients (e.g., phosphates, ammonia, and trace metals) and the toxicity of pollutants. At high pH, many metals form hydroxides or carbonates that are relatively insoluble and usually precipitate out. At low pH, toxic elements and compounds can be released from sediments into the water column (CCME 2013). Changes in pH may also have direct impacts on aquatic organisms. For example pH that is too high or too low can result in physiological stress, which may affect survival, growth, and reproduction.

Total Alkalinity

Alkalinity is a measure of the overall buffering capacity of an aquatic system. It is the sum of all of the components in the water column that act to buffer it against a change in major negative ions including pH, carbonates, bicarbonates, hydroxides, sulphides, silicates, and phosphates. Elevated alkalinity allows for greater stability in pH, which is important for aquatic life. In general, Arctic and sub-Arctic systems tend to have low buffering capacity.

Hardness

Water hardness is a measure of positive major ions including calcium and magnesium. Although there is no CCME guideline for water hardness, some CCME guidelines are hardness-dependent because hardness can affect the toxicity of some elements and compounds (e.g., chloride, sulphate, nitrate, copper, and nickel).

Chloride

Chloride influences osmotic balance and ion exchange and is therefore highly regulated by aquatic organisms. At elevated concentrations, chloride can be toxic and may inhibit survival, growth, and reproduction. Elevated chloride concentrations may also reduce the diversity of organisms present in freshwater systems because organisms that are intolerant of high salinity are likely extirpated.

Sulphate

Sulphur is a non-metallic element found in many mineral compounds that may be released into the aquatic environment as water percolates through rock containing sulphur compounds (Singleton 2000). Sulphate may also enter aquatic systems from atmospheric sources including sulphur dioxide, which is formed by the combustion of fossil fuels and dissolves to form acid rain. In high concentrations, sulphate is toxic to many aquatic organisms, including invertebrates and fish.

Potassium

Potassium plays an important role in nerve function and is therefore required by many aquatic species (Environment Canada 2002). However, potassium can become toxic when concentrations are elevated. Compared to other major ions of earth metals (i.e., magnesium, calcium, and sodium), potassium is substantially more toxic and was therefore selected for evaluation in the AEMP as a 'worst case' indicator ion for earth metals as a whole. Potassium toxicity may decrease as the total ion concentration increases as a consequence of strong interactions with other metals (Trotter 2001).

Nutrients

Nutrients – especially macronutrients such as carbon, nitrogen, and phosphorous - are essential in the synthesis of living material. While these elements are required by plants and animals for survival, growth, and reproduction, changes in both the total and relative concentrations of nutrients (i.e., both the total number of mg/L and the ratio of carbon, nitrogen, and phosphorous in the systems) can have dramatic impacts on aquatic ecosystems. Effects may include changes in the total abundance of individuals present, changes in the relative abundances of the species present, changes in community composition, and reductions in trophic complexity (CCME 2003b). Excessive quantities of macronutrients result in "eutrophic" conditions, which may result in algal blooms, reductions in water clarity, or reduced concentrations of dissolved oxygen as organic matter is degraded through the process of microbial respiration (CCME 2003b). These effects may cascade up the food web, resulting in changes in the abundance, condition, or species composition of fish (CCME 2003b). Other elements and compounds, known as micronutrients, are required in smaller or even trace amounts. These include some metals (e.g., copper, zinc, etc.), which will be discussed in more detail in the metals section. Elevated concentrations of micronutrients are often toxic to aquatic life.

Total Ammonia-N

Total ammonia is a measure of the most reduced inorganic forms of nitrogen in water and includes dissolved ammonia (NH_3) and the ammonium ion (NH_4^+). Ammonia is an important component of the nitrogen cycle: Ammonia readily oxidises to nitrite, and then to nitrate, which is a highly bio-available form of nitrogen. Excessive quantities of ammonia can have deleterious impacts on aquatic systems through eutrophication (CCME 2003b). Ammonia can also be toxic to aquatic organisms, even at low concentrations (Cavanagh et al. 1998). The toxicity of ammonia is strongly dependent on pH and temperature, with toxicity increasing as pH increases and as temperature decreases. pH affects the balance between NH_3 and NH_4^+, with the formation of NH_4^+ favoured at low pH (CCME 2000). Since the non-ionised form, NH_3, is much more toxic than the ammonia ion, toxicity tends to be highest at elevated pHs.

Biological effects of elevated ammonia levels are well documented for fish and include gill lesions, kidney damage, and larval deformities and death (CCME 2000). In comparison, effects of ammonia toxicity to periphyton, phytoplankton, zooplankton, and benthic organisms are not established (CCME 2000). However, concentrations as low as 0.6 mg/L have been shown to result in significant mortality in freshwater algae (Bretthauer 1978). In zooplankton, significant 7-day mortality has been demonstrated at a concentration of 15.2 mg/L in the cladoceran *Ceriodaphinia dubia* (Nimmo et al. 1989).

Nitrite-N

Nitrite is produced through the oxidation of ammonia and is then quickly oxidised to nitrate in the presence of adequate oxygen. Consequently, only trace amounts of nitrite are generally found in surface waters. As with ammonia, nitrite can be toxic to aquatic life at relatively low concentrations. Nitrite toxicity increases with increasing pH (Cavanagh et al. 1998).

Nitrate-N

Nitrate is produced as bacteria oxidise nitrite, which is oxidised from ammonia. Nitrate is a highly bio-available form of nitrogen. It is the primary form of nitrogen used by aquatic primary producers (i.e., macrophytes, periphyton, and phytoplankton) and constitutes between two-thirds and four-fifths of the total available nitrogen in surface waters (Crouzet et al. 1999; CCME 2003b). Excessive quantities of nitrate in relation to other macronutrients can have deleterious effects on aquatic systems through eutrophication, increasing the risk of algal blooms and oxygen depletion, decreasing water clarity, changing species composition, and reducing trophic complexity (CCME 2003a). However, phosphorous often acts as the limiting nutrient in freshwater aquatic systems. Thus outside of toxic effects, increases in nitrogen may have little impact on aquatic systems unless concentrations of available phosphorous also increase (CCME 2003b).

Nitrate is less toxic than ammonia or nitrite, but may reduce the oxygen carrying capacity of blood and interfere with an organism's ability to osmoregulate (Colt and Armstrong 1981). Some forms of nitrate are more toxic than other forms. For example, potassium nitrate (KNO_3) can be as much as five times more toxic to freshwater organisms than sodium nitrate ($NaNO_3$; CCME 2003b). Significant mortality has been observed for benthic invertebrates when exposed to nitrate concentrations as low as 290 mg/L $NaNO_3$ (211.7 mg/L NO_3-N) and 1,657 mg/L (1209.6 mg/L NO_3-N) for cladocerans like *Daphnia magna*, which are an important source of food for fish (CCME 2003b).

Total Phosphate-P

Total phosphate is a combined measure of the inorganic and organic forms of phosphorus. Excess quantities of phosphorous may result in increased primary productivity, which can decrease water clarity and reduce dissolved oxygen as organic material is decomposed by bacteria through the process of respiration (CCME 2004). Such changes can result in "dead zones" where oxygen levels are too low to support aquatic life (Carpenter 2008; Diaz and Rosenberg 2008).

Total Organic Carbon (TOC)

TOC is a measure of the amount of organic material – including both living and decaying tissues – in dissolved and particulate forms in a water column. Increases in the biomass of primary and secondary consumers may result from increases in nutrient levels. Such increases in biomass, some of which are reflected as changes in total organic carbon, may lead to reductions in oxygen as a consequence of increased microbial decomposition rates as these organisms expire. Consequently, there is often an inverse relationship between TOC concentrations and dissolved oxygen concentrations in a system and TOC concentrations may be used as an indicator of change. TOC has been measured as part of the Ekati Diamond Mine AEMP since 2004 in order to better understand patterns in under-ice dissolved oxygen concentrations in Cujo Lake. Consequently, baseline conditions are not defined.

<u>Metals</u>

Metals include both dissolved metals and metals bound to particulate matter in the water column. When the pH of water decreases, metal solubility increases and metal particles become more bioavailable. The effects of metal exposure on physiological processes in aquatic organisms (i.e., algae, macrophytes, invertebrates, and fish) are complex and variable (Connell and Miller 1984). Metal toxicity can be affected by multiple factors including pH, temperature, water hardness, and carbon dioxide concentration (Mullins 1977; Connell and Miller 1984; Westman 1985). Sensitivity to toxicity can also depend on the species, age, sex, and size of an individual. Some metals – known as trace metals, including arsenic, copper, molybdenum, nickel, and vanadium – are required in small amounts by most species for normal physiological function. However, excessive amounts of these metals can be toxic. Metals can also be stored in the tissues of aquatic organisms, and this may result in the accumulation of metals in increasing concentrations in biotic tissues as they are transferred up the food web from primary producers to top predators like fish (a process called "bioaccumulation").

3.2.2 Dataset

Table 3.2-1 summarizes the reference and monitored lakes and streams that were sampled in the Koala Watershed during each sampling period.

Table 3.2-1. Reference and Monitored Lakes and Streams Sampled in the Koala Watershed and Lac de Gras in 2013

Watershed	Month	Lake / Stream	Reference Lakes / Streams Sampled	Monitored Lakes/Streams Sampled
Koala	April	Lake	Nanuq, Counts, Vulture	Grizzly, Kodiak, Leslie, Moose, Nema, Slipper, S2, S3
	August	Lake	Nanuq, Counts, Vulture	Grizzly, Kodiak, 1616-30 (LLCF)[1], Leslie, Moose, Nema, Slipper, S2, S3
	August	Stream	Nanuq Outflow, Counts Outflow, Vulture-Polar	Lower PDC, Kodiak-Little, 1616-30 (LLCF)[1], Leslie-Moose[2], Moose-Nero, Nema-Martine, Slipper-Lac de Gras

1: 1616-30 is the discharge location in the LLCF and is sampled as part of the SNP and AEMP.
2: Leslie-Moose station was added in to the AEMP in 2010.

3.2.2.1 Lakes

For each of the sampling years between 1998 and 2013, lake water quality data was collected for the evaluation of effects between mid-April and mid-May during the ice-covered season (Table 3.2-2) and between late July and mid-August during the open water season (Table 3.2-3). Baseline water quality data, collected between 1994 and 1997, are included in the data summary tables (Tables 3.2-2 and 3.2-3) and illustrated graphically, below, for visual comparison, but were not included in the statistical evaluation of effects. Water from Cell E of the LLCF was discharged into Leslie Lake from June 18, 2013 to September 30, 2013, at which time it was still ongoing. Therefore, August sampling is representative of post-discharge water quality in receiving lakes.

The timing and number of sampling events during the open water season has varied through time as refinements have been made to the sampling protocol. During baseline years, sampling occurred in July and August in 1994, in August in 1995, in July in 1996, and in August in 1997. In 1998, water quality was sampled five times during the open water season. A detailed quantitative analysis was conducted on the 1998 dataset, which resulted in a reduction of the open water season sampling frequency from five to three events per season in 1999. Open water sampling frequency remained at three events per season through 2009 (July, August and September). In 2010, sampling frequency was reduced to once per season, in August, as a result of a detailed review of the historical data carried out as part of the 2009 re-evaluation (Rescan 2010c). Historical lake water quality data – including all sampling events – is presented graphically in Section 5 of this report. Summaries of the 2013 April and August lake water quality data are provided in Part 2 of the AEMP (Part 2 – Data Report).

Table 3.2-2. Dataset Used for Evaluation of Effects on the April (Ice-covered) Water Quality of the Lakes of the Koala Watershed and Lac de Gras

Year	Nanuq	Counts	Vulture	Grizzly	Kodiak	Leslie	Moose	Nema	Slipper	S2	S3
1994[1]	-	-	-	-	-	-	-	-	-		-
1995[1]	-	-	-	-	-	-	-	-	-	-	-
1996[1]	-	-	Apr-18 (1)	-	Apr-18 (1)	-	Apr-17 (1)	Apr-17 (1)	Apr-17 (1)	-	-
1997[1]	-	-	-	-	-	-	-	-	-	-	-
1998	-	-	-	-	-	-	-	-	-	-	-
1999	-	-	-	-	Apr-19 (2)	-	-	-	-	-	-
2000	-	-	-	-	Apr-6 (2), Apr-10 (2)	-	-	-	-	-	-
2001	-	-	-	-	Apr-23 (4)	-	-	-	-	-	-
2002	Apr-19 (4)	Apr-23 (4)	Apr-20 (4)	-	Apr-18 (8)	-	Apr-20 (4)	Apr-18 (4)	Apr-23 (4)	Apr-23 (4)	Apr-23 (4)
2003	Apr-12 (4)	Apr-13 (4)	Apr-14 (4)	Apr-16 (4)	Apr-17 (4)	Apr-15 (4)	Apr-15 (4)	Apr-14 (4)	Apr-15 (4)	Apr-15 (4)	Apr-15 (4)
2004	Apr-18 (4)	Apr-17 (4)	Apr-18 (4)	Apr-19 (4)	Apr-19 (4)	Apr-16 (4)	Apr-16 (4)	Apr-17 (4)	Apr-17 (4)	Apr-17 (2)	Apr-17 (4)
2005	Apr-24 (4)	Apr-24 (4)	Apr-24 (4)	Apr-24 (4)	Apr-28 (4)	Apr-29 (4)	Apr-26 (4)	Apr-25 (4)	Apr-26 (4)	Apr-26 (2)	Apr-26 (4)
2006	Apr-20 (4)	Apr-22 (4)	Apr-21 (4)	Apr-24 (4)	Apr-24 (4)	Apr-23 (4)	Apr-23 (4)	Apr-23 (4)	Apr-18 (4)	Apr-18 (4)	Apr-18 (4)
2007	Apr-21 (4)	Apr-24 (4)	Apr-22 (4)	Apr-26 (4)	Apr-26 (4)	Apr-26 (4)	Apr-27 (4)	Apr-27 (4)	Apr-24 (4)	Apr-23 (4)	Apr-23 (4)
2008	Apr-27 (4)	May-3 (4)	May-3 (4)	May-6 (4)	May-6 (4)	May-6 (4)	May-6 (4)	May-6 (4)	May-4 (4)	May-4 (4)	May-4 (4)
2009	May-11 (4), May-18 (4)	May-17 (4)	Apr-28 (4)	Apr-28 (4)	May-2 (4)	May-2 (4)	Apr-29 (4)	Apr-29 (4)	May-17 (4)	May-18 (4)	May-17 (4)
2010	Apr-14 (4)	Apr-14 (4)	Apr-12 (4)	Apr-12 (4)	Apr-16 (4)	Apr-16 (4)	Apr-14 (4)	Apr-15 (4)	Apr-15 (4)	Apr-15 (4)	Apr-15 (4)
2011	Apr-25 (4)	Apr-26 (4)	Apr-28 (4)	Apr-28 (4)	Apr-28 (4)	Apr-27 (4)	Apr-27 (4)	Apr-27 (4)	Apr-27 (4)	Apr-27 (4)	Apr-27 (4)
2012	Apr-20 (4)	Apr-17 (4)	Apr-18 (4)	Apr-16 (4)	Apr-22 (4)	Apr-24 (4)	Apr-24 (4)	Apr-23 (4)	Apr-25 (4)	Apr-24 (4)	Apr-25 (4)
2013	Apr-26 (4)	Apr-26 (4)	Apr-23 (4)	Apr-22 (4)	Apr-28 (4)	Apr-24 (4)	Apr-24 (4)	Apr-24 (4)	Apr-27 (4)	Apr-27 (2)	Apr-27 (4)

Dashes indicate no data were available
Number of samples is indicated in brackets
[1]Data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison

Between 1998 and 2001, Kodiak Lake water quality samples were collected as part of the Kodiak Lake Sewage Effects Study (KLSES; Rescan 2002). During these years, the timing of sample collection differed from that of the AEMP and different analytical laboratories were used for the KLSES and AEMP. Water quality data that corresponds to the timing of AEMP sample collection was selected from the KLSES and screened for use in the AEMP. Kodiak Lake was included as part of the AEMP sampling program and analysis in 2002, with the timing of sample collection and analytical laboratory consistent with those of the AEMP since this time.

The number of replicates collected and the depth at which replicates are collected during the ice-covered and open water seasons have changed through time. Currently, the AEMP methods include the collection of two replicate water quality samples collected at each of two depths during the ice-covered season: middle of the water column and at 2 m above the sediment surface; and the collection of three replicate water quality samples collected at each of two depths during the open water season: 1 m below the surface and in the middle of the water column.. During the baseline sampling period (1996), only one water quality sample was collected during the ice-covered season, at a depth of 1 m below the ice layer. In 2002, the number of replicates collected during the open water season was reduced from three to two after it was shown that open water season water quality was independent of water column depth in previous years (Rescan 1998). The number of replicates was further reduced, to one replicate with 10% duplication, in 2003 following recommendations from the 2003 AEMP Re-evaluation and refinement report (Rescan 2003). Following the AEMP re-evaluation in 2006 (Rescan 2006), triplicate samples have been collected from each depth in order to provide sufficient data for August-only sampling from 2007 to present.

Table 3.2-3. Dataset Used for Evaluation of Effects on the August (Open Water) Water Quality Koala Watershed Lakes and Lac de Gras

Year	Nanuq	Counts	Vulture	Grizzly	Kodiak	1616-30	Leslie	Moose	Nema	Slipper	S2	S3
1994*	-	-	Aug-13 (5)	Aug-13 (5)	Aug-19 (15)	-	Aug-20 (5)	Aug-22 (5)	-	Aug-15 (5)	Aug-14 (1)	Aug-14 (1)
1995*	-	-	Aug-9 (5)	-	Aug-19 (15)	-	-	-	Aug-11 (5)	Aug-11 (6)	-	-
1996*	-	-	Jul-26 (3)	-	Jul-28 (9)	-	-	Jul-26 (3)	Jul-26 (3)	Jul-26 (3)	-	-
1997*	Aug-4 (9)	Aug-14 (3)	Aug-5 (9)	Aug-7 (3)	Aug-9 (5)	-	-	Aug-10 (3)	Aug-11 (3)	Aug-11 (3)	Aug-12 (1)	Aug-12 (3)
1998	Jul-29 (6), Aug-11 (6)	Jul-29 (3), Aug-14 (3)	Jul-27 (3), Aug-10 (3)	Jul-27 (6), Aug-9 (6)	Jul-28 (12), Aug-11 (12)	Aug-18 (1)	-	Jul-28 (6), Aug-11 (6)	Jul-28 (6), Aug-11 (6)	Jul-31 (6), Aug-12 (6)	Jul-30 (6), Aug-13 (6)	Jul-30 (6), Aug-13 (6)
1999	Aug-7 (6)	Aug-8 (6)	Aug-6 (6)	Aug-6 (6)	Aug-10 (5)	Aug-9 (1)	-	Aug-7 (6)	Aug-10 (6)	Aug-9 (6)	Aug-11 (6)	Aug-11 (6)
2000	Aug-4 (4)	Aug-1 (4)	Aug-4 (4)	Aug-4 (4)	Jul-29 (4)	Jul-31 (1)	-	Jul-30 (4)	Jul-30 (4)	Jul-31 (4)	Aug-3 (4)	Aug-3 (4)
2001	Aug-1 (4)	Jul-30 (4)	Aug-2 (4)	Aug-7 (4)	Jul-28 (5)	Aug-7 (3)	-	Aug-3 (4)	Aug-3 (4)	Jul-29 (4)	Jul-29 (4)	Jul-29 (4)
2002	Aug-1 (4)	Aug-7 (4)	Aug-3 (4)	Aug-2 (4)	Aug-2 (4)	Aug-6 (3)	-	Aug-5 (4)	Aug-4 (4)	Aug-6 (4)	Aug-4 (4)	Aug-4 (4)
2003	Aug-9 (3)	Aug-7 (2)	Aug-4 (2)	Aug-8 (3)	Aug-8 (2)	Aug-2 (2)	Aug-3 (2)	Aug-9 (3)	Aug-3 (2)	Aug-7 (3)	Aug-5 (2)	Aug-5 (2)
2004	Aug-10 (3)	Aug-12 (2)	Aug-9 (2)	Aug-7 (2)	Aug-7 (2)	Jul-26 (2), Aug-2 (2), Aug-11 (4)	Aug-9 (2)	Aug-10 (3)	Aug-9 (2)	Aug-12 (3)	Aug-9 (2)	Aug-9 (2)
2005	Aug-1 (2)	Aug-7 (3)	Jul-31 (2)	Aug-7 (2)	Aug-3 (2)	Aug-2 (2)	Aug-4 (2)	Aug-9 (2)	Aug-9 (2)	Aug-5 (2)	Aug-5 (2)	Aug-5 (3)
2006	Aug-2 (3)	Aug-4 (2)	Aug-2 (2)	Aug-7 (2)	Aug-1 (3)	Jul-26 (2), Jul-27 (1), Jul-29 (1), Jul-31 (1), Aug-4 (1)	Aug-6 (2)	Aug-5 (2)	Aug-5 (2)	Aug-4 (2)	Aug-4 (3)	Aug-4 (2)
2007	Aug-11 (6)	Aug-6 (6)	Aug-12 (6)	Aug-4 (6)	Aug-4 (6)	Jul-28 (2), Aug-12 (2)	Aug-13 (6)	Aug-7 (6)	Aug-11 (6)	Aug-10 (6)	Aug-8 (6)	Aug-6 (6)
2008	Aug-8 (6)	Jul-31 (6)	Jul-29 (6)	Jul-27 (6)	Jul-27 (6)	Aug-4 (2), Aug-24 (2)	Jul-31 (6)	Jul-29 (6)	Jul-29 (6)	Jul-29 (6)	Aug-7 (6)	Aug-7 (6)
2009	Jul-30 (6)	Aug-1 (6)	Jul-30 (6)	Aug-2 (6)	Aug-8 (6)	Aug-3 (2), Aug-6 (2), Aug-10 (2) Aug-17 (2), Aug-24 (2)	Aug-5 (6)	Jul-30 (6)	Jul-30 (6)	Aug-3 (6)	Jul-31 (6)	Jul-31 (6)
2010	Aug-5 (6)	Aug-7 (6)	Aug-5 (6)	Aug-4 (6)	Aug-5 (6)	Aug-3 (2), Aug-9 (2), Aug-16 (2), Aug-23 (2), Aug-30 (2)	Aug-3 (6), Aug-17 (2)**, Aug-31 (2)**	Aug-3 (6), Aug-17 (2)**, Aug-31 (2)**	Aug-5 (6)	Aug-5 (6)	Aug-5 (1), Aug-6 (5)	Aug-5 (3), Aug-6 (3)
2011	Aug-2 (6)	Aug-5 (6)	Aug-5 (6)	Aug-1 (6)	Aug-5 (6)	Jul-31 (2), Aug-2 (1), Aug-8 (2) Aug-14 (1), Aug-24 (1), Aug-29 (1)	Aug-2 (6)	Aug-3 (6)	Aug-5 (6)	Aug-3 (6)	Aug-4 (6)	Aug-4 (6)
2012	Aug-1 (6)	Aug-8 (6)	Aug-7 (6)	Aug-2 (6)	Aug-6 (6)	Jul-30 (1), Aug-4 (2), Aug-6 (1), Aug-14 (1), Aug-21 (1), Aug-27 (1)	Aug-8 (6)	Aug-9 (6)	Aug-7 (6)	Aug-8 (6)	Aug-3 (6)	Aug-2 (6)
2013	Aug-3 (6)	Aug-1 (6)	Aug-1 (6)	Jul-31 (6)	Aug-6 (6)	Jul-29 (1), Aug-5 (3), Aug-12 (1), Aug-19 (1), Aug 26 (2)	Aug-1 (9)[1]	Aug-5 (6)	Aug-6 (6)	Aug-5 (6)	Aug-2 (6)	Aug-2 (6)

Dashes indicate no data were available

Number of replicates is indicated in brackets

1: Three additional bottom depth samples were collected

** = Data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison*

Water quality samples were analyzed by ALS Environmental Services (ALS) in Burnaby, B.C. from 1994 to 1997, by EnviroTest Laboratories in Edmonton, A.B. from 1998 to 2001 (with the exception of Kodiak Lake), by EnviroTest in 2002 and 2003, and by ALS from 2004 to present. Samples collected from Kodiak Lake from 1998 to 2001 were analyzed by the University of British Columbia (dissolved nutrients), EnviroTest (total metals) and ALS (all other variables).

Reductions in detection limits can sometimes mislead data analysis or interpretation, suggesting sample contamination, analytical variability, or temporal patterns in concentrations that do not actually exist. Although lake water quality samples have been collected consistently using General Oceanic FLO (GO-FLO) bottles during the open water season and Niskin bottles during the ice-covered season, analytical detection limits have changed through time. Generally, detection limits have decreased through time as analytical methods have improved. Analytical detection limits for water quality variables are indicated as black dotted lines in figures presented below.

Mean concentrations of water quality variables were calculated for the ice-covered (April) and open water (August) seasons by pooling data from samples collected at all depths under the assumption that water columns are completely mixed. Owing to potential changes in water column structure in Leslie Lake, three replicate water quality samples from the lower strata in Leslie Lake were also collected during August AEMP sampling, to provide a more accurate depiction of open water-season water quality in Leslie Lake beginning in 2013 (Rescan 2013d). The lower strata water quality data will be pooled with upper and middle strata water quality data collected from Leslie Lake as part of the evaluation of effects. Over the years, data were removed from the dataset prior to analysis and interpretation as a result of contamination (Table 3.2-4).

Table 3.2-4. Data Removed from the Historical Lake and Stream Water Quality Dataset for the Koala Watershed and Lac de Gras

Year	Date	Sample ID	Variables	Rationale
1999	April 19	Kodiak (mid and deep)	TDS, Chloride, Sulphate, Potassium, Total Selenium	Unexplained contamination
1999	April 19	Kodiak (mid)	Total Aluminum	Unexplained contamination
1999	August	Lakes and Streams	Total Metals	Contaminated nitric acid provided by lab
2000	April 6 and 10	Kodiak	Ortho-phosphate, Potassium, Total Molybdenum, Total Selenium	Unexplained contamination
2001	August	Lakes and Streams	Ortho-phosphate	Unexplained contamination
2002	August 7	Counts (mid, rep 1)	Total Zinc	Unexplained contamination, >6x replicate concentration
2002	July 31	Nema (1 m, rep 2)	All	Unexplained contamination
2003	August 2	Kodiak-Little (rep 1)	Total Zinc	Unexplained contamination, >10x replicate concentration
2005	April 24	Nanuq (mid, rep 1)	Total Copper	Unexplained contamination
2005	August 9	Moose (mid, rep 2)	Total Phosphate	Unexplained contamination
2007	August 4	Kodiak (1 m, rep 2)	Total Zinc	Unexplained contamination, >40x replicate concentration
2008	May 3	Vulture (mid, rep 1)	Sulphate, Chloride, TDS	Unexplained contamination
2008	August 2	Nanuq Outflow (rep 1)	pH	Much higher than the pH in all reference lakes samples collected in 2008

(continued)

Table 3.2-4. Data Removed from the Historical Lake and Stream Water Quality Dataset for the Koala Watershed and Lac de Gras (completed)

Year	Date	Sample ID	Variables	Rationale
2009	July 31	S3 (mid, reps 1 and 2)	Total Phosphate	Unexplained contamination
2010	August 5	Slipper (rep 1)	All	Unexplained contamination
2011	April 28	Grizzly (deep, reps 1 and 2)	All	Unexplained contamination
2012	August 7	Nema (mid rep 3)	arsenic	Elevated concentrations compared to other replicates, other lakes in the same sampling period, and the same and other lakes historically
2012	September 11	1616-30 (LLCF; rep X)	All metals	Unexplained contamination associated with elevated TSS

3.2.2.2 Streams

Stream water quality data has been collected in June, August, and September of each year between 1998 and 2013. July stream water quality sampling was added to the AEMP program in 2010. In 2013, August stream sampling was representative of post-discharge water quality as water from Cell E of the LLCF was discharged into Leslie Lake from June 18, 2013 to September 30, 2013 and was ongoing at that time. Thus, August 2013 samples were used for the evaluation of effects (Table 3.2-5). Using August samples for the evaluation of stream water quality effects also maintains consistency with the evaluation of lake water quality effects. The Part 2- Data Report provides all stream water quality results for June, July, and September. Baseline water quality data, collected from 1994 to 1997, are included in the data summary tables (Tables 3.2-2 and 3.2-5) and illustrated in Figures 3.2-1 to 3.2-22 for visual comparison, but were not used in the statistical evaluation of effects.

Data from the Lower PDC in 1998 and 1999 were collected as part of the SNP. The timing of SNP sample collection and the laboratory at which analyses were conducted differed from samples collected as part of the AEMP. Lower PDC data collected as part of the SNP were screened and selected to correspond to AEMP sampling dates. Kodiak-Little water quality data collected between 1998 and 2001 were collected as part of the Kodiak Lake Sewage Effects Study (Rescan 2002). Data from Kodiak-Little – which is referred to as station K4 in the KLSES – was screened and selected to correspond to AEMP sampling dates. Kodiak-Little was included as an AEMP sampling location for the first time in 2002.

The number of replicate samples collected at stream sites has varied over the course of the AEMP (Table 3.2-5). From 1994 to 1997, one replicate sample with 10% duplication was collected from each stream. From 1998 to 2002, three replicate samples were collected. In 2003, the number of replicate samples collected changed to two. Two replicate samples have been collected at each stream site since 2003. In 2011, additional data were collected from the Lower PDC in late July and August as part of the PDC Slope Enhancement Project monitoring program and were included in the evaluation of effects.

Leslie-Moose Stream was added to the list of streams that are subject to statistical evaluation in 2012. However, the relatively small number of data points (i.e., four years) available for Leslie-Moose Stream in 2013 decreases the probability of detecting statistically significant changes in evaluated variables. Thus, graphical analysis was the primary means through which change in evaluated variables and potential mine effects were assessed in Leslie-Moose Stream in 2013.

Table 3.2-5. Dataset Used for Evaluation of Effects on the August (Open Water) Water Quality in Koala Watershed Streams and Lac de Gras

Year	Nanuq Outflow	Counts Outflow	Vulture-Polar	Lower PDC	Kodiak-Little	1616-30	Leslie-Moose	Moose-Nero	Nema-Martine	Slipper-Lac de Gras
1994*	-	-	Aug-4 (1)	-	Aug-3 (1)	-	-	-	-	Aug-9 (1)
1995*	-	-	Aug-10 (1)	-	Aug-8 (1)	-	-	-	Aug-10 (1)	Aug-10 (1)
1996*	-	-	Jul-27 (1)	-	Jul-28 (2)	-	-	Jul-27 (1)	Jul-26 (1)	Jul-26 (1)
1998	Aug-18 (3)	Aug-18 (3)	Aug-16 (3)	Aug-17 (1)	Aug-11 (1), Aug-20 (3)	Aug-18 (1)	-	Aug-16 (3)	Aug-21 (3)	Aug-19 (3)
1999	Aug-6 (3)	Aug-7 (3)	Aug-8 (3)	Aug-2 (2)	Aug-11 (1)	Aug-9 (1)	-	Aug-8 (3)	Aug-7 (3)	Aug-7 (3)
2000	Jul-30 (3)	Jul-30 (3)	Jul-30 (3)	Jul-30 (3)	Jul-29 (2)	Jul-31 (1)	-	Jul-29 (3)	Jul-29 (3)	Jul-29 (3)
2001	Aug-7 (3)	Aug-7 (3)	Aug-7 (3)	Aug-7 (3)	Aug-2 (2)	Aug-7 (3)	-	Aug-7 (3)	Aug-7 (3)	Aug-7 (3)
2002	Aug-6 (3)	Aug-6 (3)	Aug-6 (3)	Aug-6 (3)	Aug-6 (3)	Aug-6 (3)	-	Aug-6 (3)	Aug-6 (3)	Aug-6 (3)
2003	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)	-	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)
2004	Aug-11 (2)	Aug-11 (2)	Aug-11 (2)	Aug-11 (2)	Aug-11 (2)	Jul-26 (2), Aug-2 (2), Aug-11 (4)	-	Aug-11 (2)	Aug-11 (2)	Aug-11 (2)
2005	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)	-	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)
2006	Jul-27 (2)	Jul-27 (2)	Jul-27 (2)	Jul-29 (2)	Jul-29 (2)	Jul-26 (2), Jul-27 (1), Jul-29 (1), Jul-31 (1), Aug-4 (1)	-	Jul-27 (2)	Jul-27 (2)	Jul-28 (2)
2007	Aug-3 (2)	Aug-3 (2)	Aug-4 (2)	Aug-5 (2)	Aug-5 (2)	Jul-28 (2), Aug-12 (2)	-	Aug-3 (2)	Aug-4 (2)	Aug-4 (2)
2008	Aug-2 (2)	Aug-1 (2)	Aug-2 (2)	Jul-28 (2)	Aug-1 (2)	Aug-4 (2), Aug-24 (2)	-	Aug-1 (2)	Aug-1 (2)	Aug-1 (2)
2009	Aug-3 (2)	Aug-3 (2)	Aug-4 (2)	Aug-8 (2)	Aug-5 (2)	Aug-3 (2), Aug-6 (2), Aug-10 (2),Aug-17 (2), Aug-24 (2)	-	Aug-5 (2)	Aug-4 (2)	Aug-4 (2)
2010	Aug-1 (2)	Aug-1 (2)	Aug-1 (2)	Aug-2 (2)	Aug-1 (2)	Aug-3 (2), Aug-9 (2), Aug-16 (2), Aug-23 (2), Aug-30 (2)	Aug-1(2)	Aug-1 (2)	Aug-1 (2)	Aug-1 (2)
2011†	Jul-30 (2)	July-30 (2)	July-31 (2)	Jul-31 (2), Aug-6 (1), Aug-13 (1), Aug-21 (1), Aug-28 (2)	Jul-31 (2)	Jul 31 (2), Aug 2 (1), Aug 8 (2) Aug 14 (1), Aug 24 (1), Aug 29 (1)	Jul-30 (2)	Jul-30 (2)	Jul-30 (2)	Jul-30 (2)
2012	Aug-4 (2)	Aug-5 (2)	Aug-5 (2)	Aug-4 (2)	Aug-4 (2)	Jul 30 (1), Aug-4 (2), Aug-6 (1), Aug-14 (1), Aug-21 (1), Aug-27 (1)	Aug-4 (2)	Aug-5 (2)	Aug-4 (2)	Aug-4 (2)
2013	Aug-4 (2)	Aug-4 (2)	Aug-4 (2)	Aug-6 (2)	Aug-7 (2)	Jul 29 (1), Aug-5 (3), Aug-12 (1), Aug-19 (1), Aug-26 (2)	Aug-7 (2)	Aug-7 (2)	Aug-7 (2)	Aug-7 (2)

Dashes indicate no data were available
Number of replicates is indicated in brackets
** = Data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison*
† Additional data were collected from the Lower PDC as part of the PDC Slope Enhancement Project monitoring program.

Stream water quality samples were analyzed in the same manner as lake water quality samples (see Section 3.2.2.1). Over the years, some data were removed from the historical dataset on four occasions as a result of sample contamination (Table 3.2-4).

3.2.3 Statistical Description of Results

Although a complete description of the statistical results for each variable and sampling month is provided in Part 3 – Statistical Results, it was still necessary to provide the statistical summaries in order to support effects conclusions. Thus the results and discussion of each variable includes a table summarizing the best fit model (LME or tobit) for each variable in the reference and monitored lakes and streams that were sampled in the Koala Watershed and Lac de Gras in April (lakes only) and August. The statistical evaluation of effects for each variable follows the model selection process outlined in detail in Section 2.2.4 and Figure 2.2-2. A brief recapitulation of the process is provided here:

- Model fit = 1a was selected whenever more than 60% of the observations in all reference sites were less than detection limits or whenever both the slopes and intercepts of the temporal trends differed among reference sites. Monitored sites were compared to a constant slope of 0.
- Model fit = 1b was selected whenever both the slopes and intercepts of the temporal trends differed among reference sites *and* the trend in monitored sites differed from a constant slope of 0. Monitored sites were compared to the slopes of individual reference sites.
- Model fit = 2 was selected whenever slopes were similar, but intercepts differed, among reference sites. Monitored sites were compared to the common slope of the reference sites; intercepts were ignored.
- Model fit = 3 was selected whenever the slopes and intercepts of the temporal trends were similar among reference sites, unless AIC weights suggested that the reference lakes were better modeled with a separate intercepts and/or slopes. Monitored sites were first compared to the common slope and intercept of the reference sites and then to a reduced model that allowed for differences in intercepts but retained a common slope.

A table describing the model fit selected and the data that was excluded, if any, is included for each variable.

3.2.4 Results and Discussion

3.2.4.1 pH

Summary: Statistical and graphical analyses suggest that pH has increased in all monitored lakes and streams downstream of the LLCF as far as site S3 in Lac de Gras as a result of mine operations. No mine effects were detected at sites that are not downstream of the LLCF. The 95% confidence intervals of the fitted means were within the Canadian Council of Resource and Environment Ministers (CCREM) guideline range of pH 6.5 to 9 in all monitored lakes and streams, except Grizzly Lake. The observed mean for Grizzly Lake was below the lower guideline value during the ice-covered season. However, the observed means were also less than the lower CCREM guideline value in all reference lakes and streams. Observed mean pH was less than the lower CCREM guideline value in all reference lakes in August, in Nanuq Outflow and Vulture-Polar Stream in June, and in all three reference streams in September (see Part 2 – Data Report).

Statistical and graphical analyses indicate that pH has changed through time, relative to reference sites, at all sites downstream of the LLCF as far as Slipper Lake during the ice-covered season and as far as site S3 in Lac de Gras during the open water season, except for Leslie-Moose Stream (Table 3.2-6; Figure 3.2-1). Only four years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends improbable; however, graphical analysis shows that pH levels in Leslie-Moose Stream were similar to levels in the LLCF in all years during which Leslie-Moose Stream was monitored. Graphical analysis also suggests that pH levels were greatest near the LLCF and decreased with downstream distance (Figure 3.2-1).

Table 3.2-6. Statistical Results of pH in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	2	-	Kodiak, Leslie, Moose, Nema, Slipper	-	1-1
Aug	Lake	-	LME	3	Grizzly, Kodiak, 1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2, S3	1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2, S3	-	1-7
Aug	Stream	-	LME	1b	-	-	Lower PDC, 1616-30 (LLCF), Moose-Nero, Nema-Martine, Slipper Lac de Gras	1-13

Dashes indicate not applicable.

At sites that are not downstream of the LLCF, statistical analyses indicate that pH has changed through time, relative to reference sites, in Kodiak Lake during the ice-covered season and in the Lower PDC during the open water season (Table 3.2-6). Graphical analyses also suggest that pH has increased in Kodiak Lake during the ice-covered season, but pH levels in the lower PDC have remained within the range of those observed during baseline years or overlap with values observed in reference streams (Figure 3.2-1). Given that pH in Kodiak Lake, Kodiak-Little, and the Lower PDC has been stable through time during the open water season (Table 3.2-6; Figure 3.2-1), no mine effects were detected at sites that are not downstream of the LLCF.

3.2.4.2 Total Alkalinity

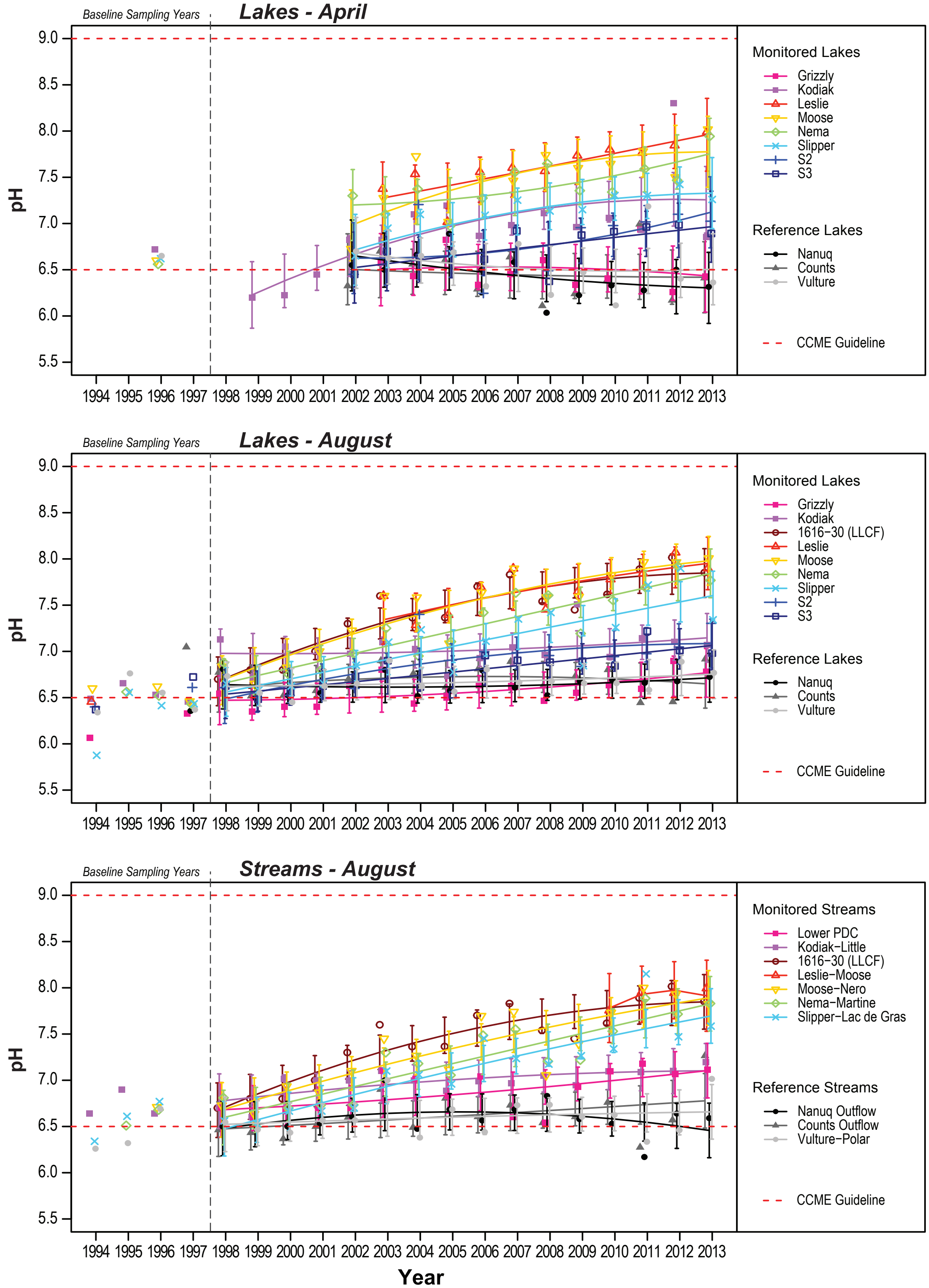
> ***Summary:*** Statistical and graphical analyses suggest that total alkalinity has increased at all sites downstream of the LLCF as far as site S2 in Lac de Gras as a result of mine operations, with total alkalinity decreasing with downstream distance from the LLCF. No mine effects were detected at sites that are not downstream of the LLCF.

Statistical analyses indicate that total alkalinity has changed through time, relative to reference sites, in all monitored lakes and streams downstream of the LLCF as far as site S2 in Lac de Gras in the open water season and as far as Slipper Lake in the ice-covered season (Table 3.2-7). Graphical analysis suggests that total alkalinity has increased through time in all lakes and streams downstream of the LLCF as far as Slipper-Lac de Gras (Figure 3.2-2). Graphical analysis also suggests that total alkalinity decreases with downstream distance from the LLCF (Figure 3.2-2).

Figure 3.2-1

Observed and Fitted Means for pH in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013

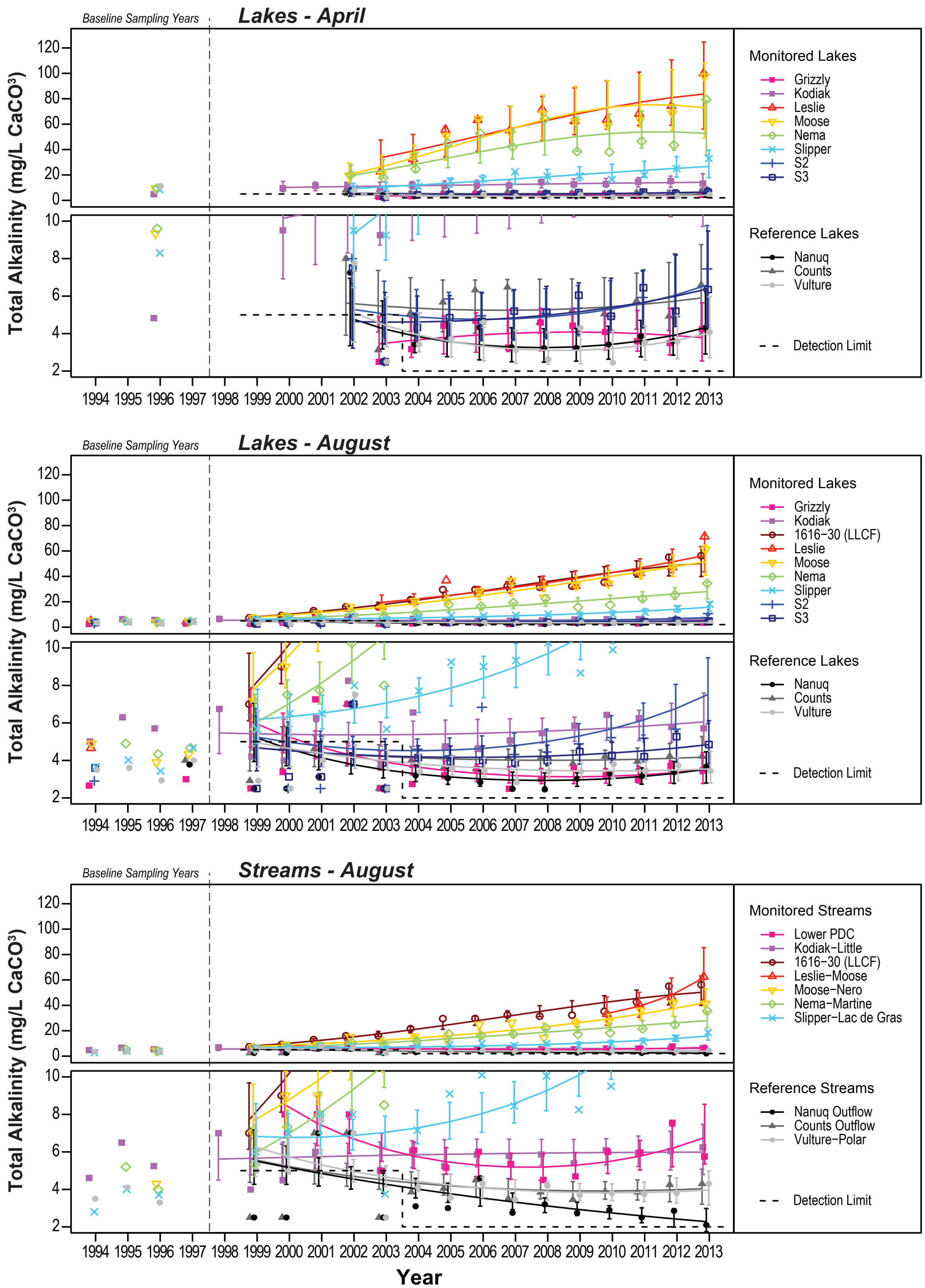
ERM



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
CCME guideline = 6.5 - 9.0.

Figure 3.2-2

Observed and Fitted Means for Total Alkalinity in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.

Table 3.2-7. Statistical Results of Total Alkalinity in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	2	-	Leslie, Moose, Nema, Slipper	-	1-19
Aug	Lake	-	Tobit	2	-	Kodiak, 1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2	-	1-25
Aug	Stream	-	Tobit	2	-	Kodiak-Little, Leslie-Moose, 1616-30 (LLCF), Moose-Nero, Nema-Martine, Slipper-Lac de Gras	-	1-31

Dashes indicate not applicable.

At sites that are not downstream of the LLCF, statistical analyses indicate that trends in total alkalinity differed from those observed in reference lakes and streams in Kodiak Lake and Kodiak-Little during the open water season (Table 3.2-7). However, graphical analysis suggests that total alkalinity has been stable in Kodiak Lake and Kodiak-Little since monitoring began (Figure 3.2-2).

3.2.4.3 Water Hardness

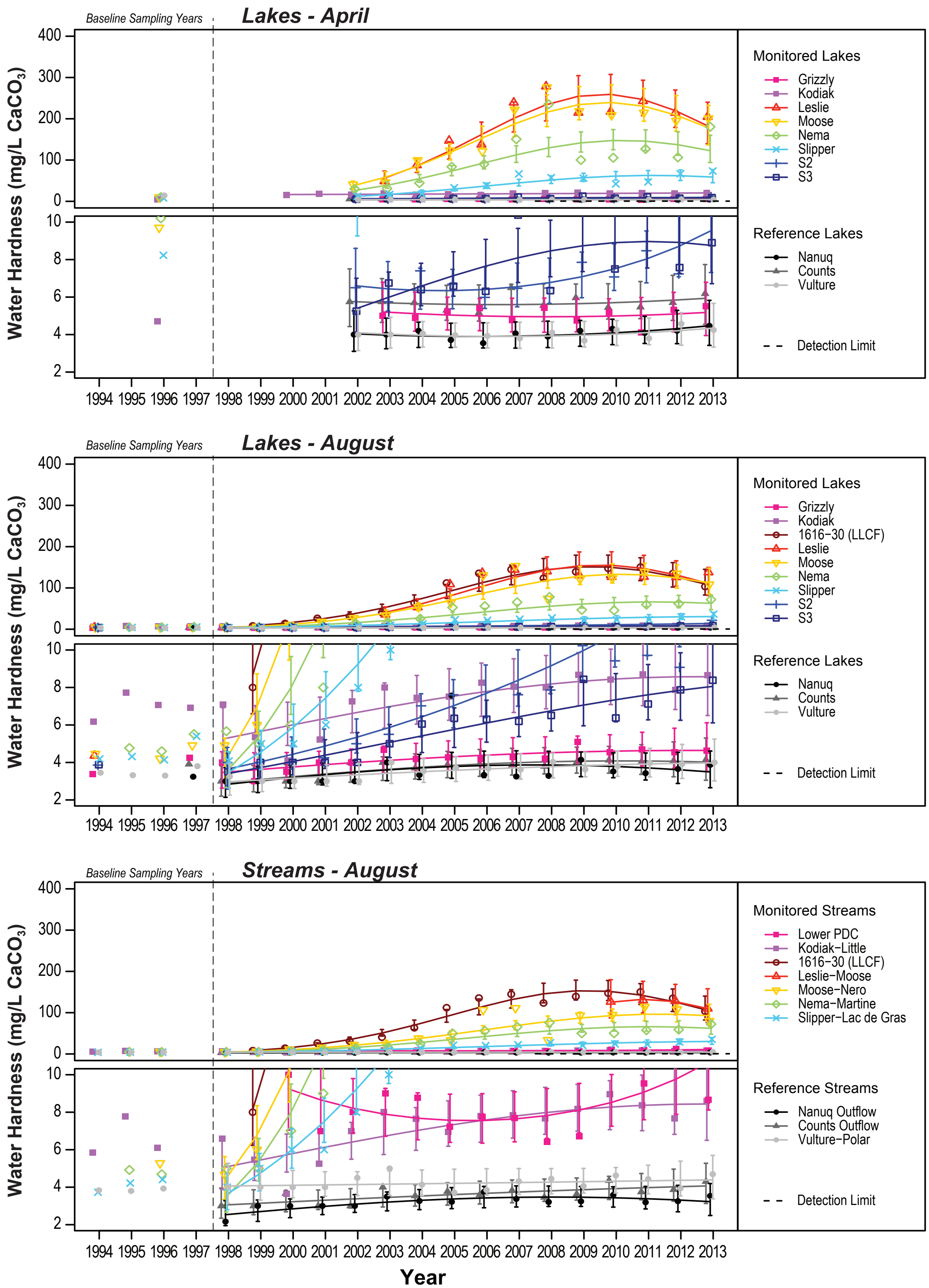
Summary: Statistical and graphical analyses suggest that water hardness has increased in all lakes and streams downstream of the LLCF as far as site S3 in Lac de Gras as a result of mine operations. However, water hardness has stabilised at concentrations greater than observed historical and reference lake concentrations at all sites downstream of the LLCF since 2006. No mine effects were detected at sites that are not downstream of the LLCF.

Statistical analyses indicate that water hardness has changed through time, relative to reference sites, in all lakes and interconnecting streams downstream of the LLCF as far as Slipper Lake during the ice-covered season and as far as site S3 in Lac de Gras during the open water season (Table 3.2-8). Graphical analysis suggests that water hardness has increased in Leslie, Moose, Nema and Slipper lakes and their interconnecting streams since monitoring began, but has stabilised at concentrations greater than historical and reference lake concentrations since about 2006 (Figure 3.2-3). For both lakes and streams, water hardness was greatest near the LLCF and decreased with increasing downstream distance (Figure 3.2-3).

Statistical analyses suggest that water hardness has changed through time in Kodiak-Little (Table 3.2-8). However, statistical analyses indicate that water hardness has been stable through time, relative to reference lakes, during both the ice-covered and open water seasons in Grizzly and Kodiak lakes (Table 3.2-8). Graphical analysis suggests that water hardness has been low and stable through time in Grizzly and Kodiak lakes, the Lower PDC, and Kodiak-Little (Table 3.2-8). No mine effects were detected at sites that are not downstream of the LLCF.

Figure 3.2-3

Observed and Fitted Means for Water Hardness in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.

Table 3.2-8. Statistical Results of Water Hardness in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts: Model Fit = 3	Model Fit = 2	Model Fit = 1	Statistical Report Page No.
Apr	Lake	-	LME	2	-	Leslie, Moose, Nema, Slipper	-	1-37
Aug	Lake	-	LME	3	Grizzly, Kodiak, 1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2, S3	1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2, S3	-	1-43
Aug	Stream	-	LME	1b	-	-	Kodiak-Little, 1616-30 (LLCF), Moose-Nero, Nema-Martine, Slipper-Lac de Gras	1-49

Dashes indicate not applicable.

3.2.4.4 Chloride

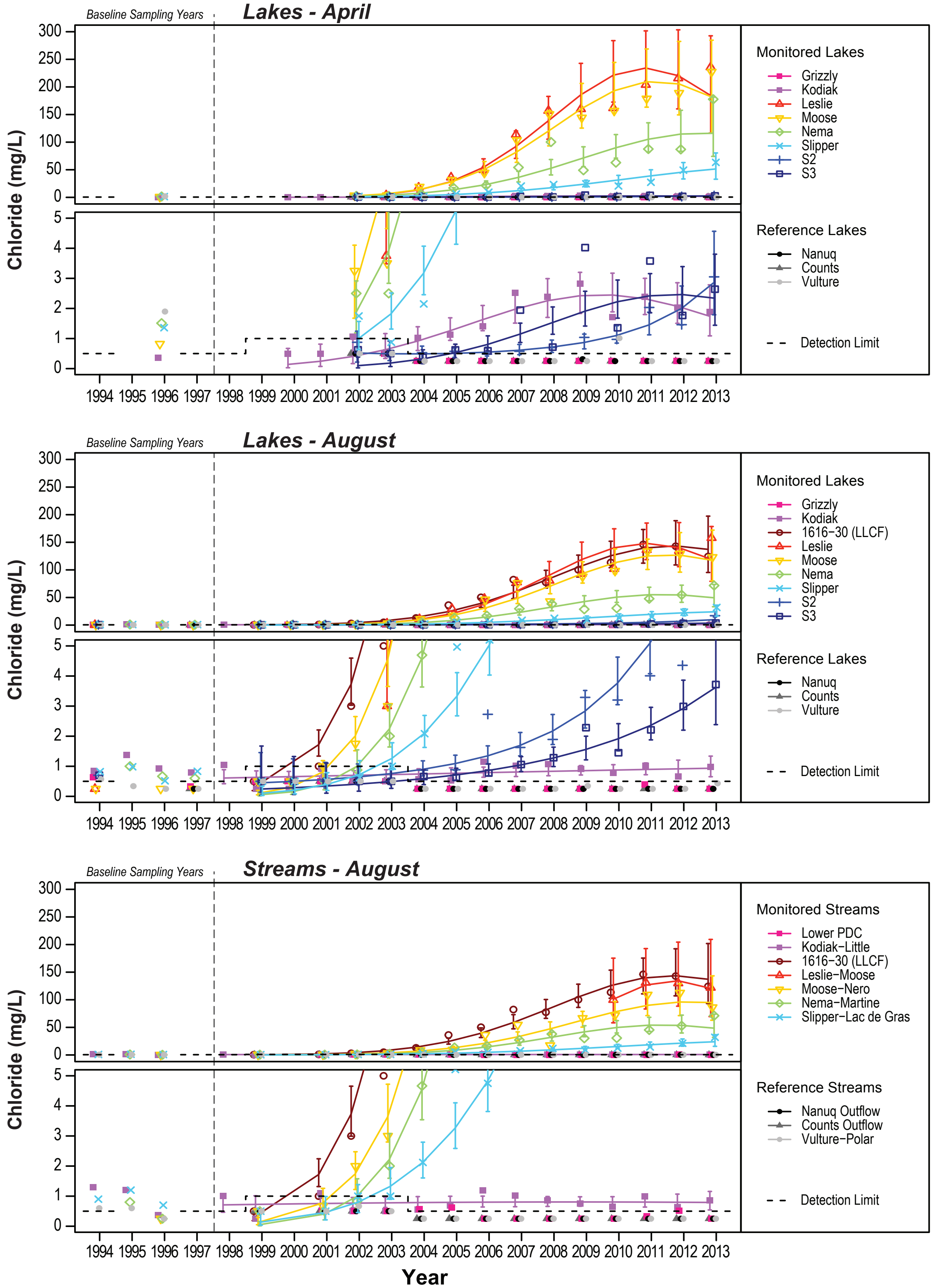
Summary: Statistical and graphical analyses suggest that chloride concentrations have increased in all monitored lakes and streams downstream of the LLCF as far as site S3 in Lac de Gras as a result of mine operations. Although chloride concentrations have increased in Kodiak Lake during the ice-covered season, there has been no change in chloride concentrations during the open water season. Thus, no mine effects were detected at sites that are not downstream of the LLCF. Observed and fitted mean concentrations were less than the hardness-dependent chloride SSWQO at all sites in 2013.

Statistical and graphical analyses indicate that chloride concentrations have increased through time in all monitored lakes and streams downstream of the LLCF as far as site S3 in Lac de Gras during both the ice-covered and open water seasons except for Leslie-Moose Stream (Table 3.2-9; Figure 3.2-4). Only four years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends improbable; however, graphical analysis shows that chloride concentrations in Leslie-Moose Stream were similar to those in the LLCF in all years during which Leslie-Moose Stream was monitored. Graphical analysis suggests chloride concentrations decrease with downstream distance from the LLCF (Figure 3.2-4). At sites that are not downstream of the LLCF, statistical analyses suggest that chloride concentrations have changed through time in Kodiak Lake during the ice-covered season. However, graphical analyses suggest that chloride concentrations in Grizzly and Kodiak lakes, the Lower PDC, and Kodiak-Little have been stable through time (Table 3.2-9; Figure 3.2-4).

The 95% confidence intervals of the fitted mean and the observed mean chloride concentrations were less than the hardness-dependent chloride SSWQO in all monitored lakes and streams in 2013 (Elphick, Bergh, and Bailey 2011). Chloride concentrations were also less than the hardness-dependent chloride SSWQO in all monitored streams in June, July, August and September 2013 (see Part 2 – Data Report).

Figure 3.2-4

Observed and Fitted Means for Chloride Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
SSWQO = 116.6 X ln(Hardness) - 204.1, where hardness = 10 - 160 mg/L.

Table 3.2-9. Statistical Results of Chloride Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	Grizzly, Nanuq, Counts, Vulture	Tobit	1a	-	-	Kodiak, Leslie, Moose, Nema, Slipper, S2, S3	1-55
Aug	Lake	Grizzly, Nanuq, Counts, Vulture	Tobit	1a	-	-	1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2, S3	1-60
Aug	Stream	Lower PDC, Nanuq Outflow, Counts Outflow, Vulture-Polar	Tobit	1a	-	-	1616-30 (LLCF), Moose-Nero, Nema-Martine, Slipper-Lac de Gras	1-65

Dashes indicate not applicable.

3.2.4.5 Sulphate

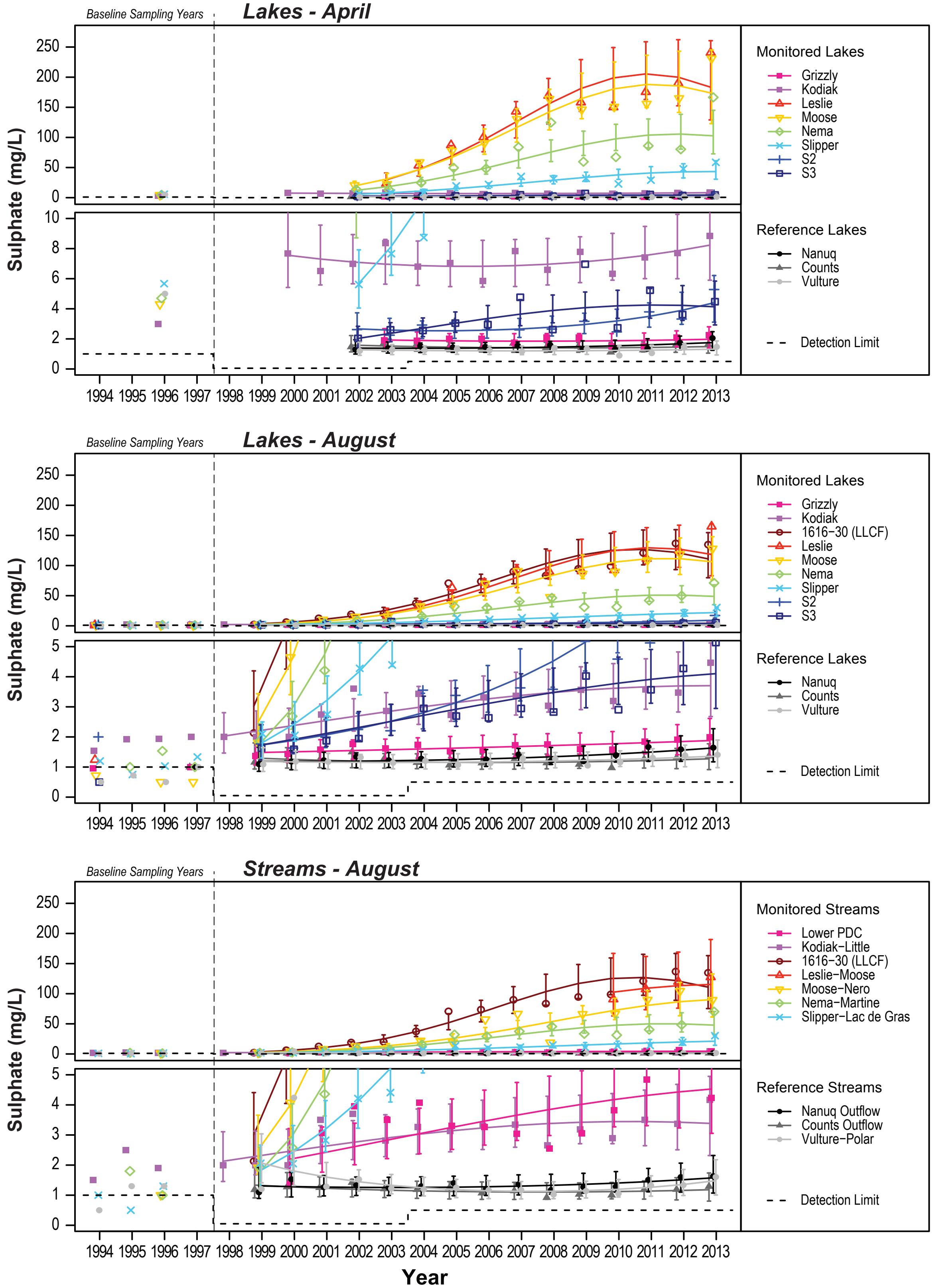
Summary: Statistical and graphical analyses suggest that sulphate concentrations have increased in all monitored lakes and streams downstream of the LLCF as far as site S3 in Lac de Gras as a result of mine operations. In recent years, sulphate in Leslie, Moose, and Nema lakes has stabilised at concentrations greater than observed historical and reference lake concentrations. In Kodiak Lake, a slight increase in sulphate concentrations during the open water season may reflect an effect of mine-related activities at the main camp. Observed and fitted mean concentrations were less than the hardness-dependent sulphate SSWQO at all sites in 2013.

Statistical and graphical analyses indicate that sulphate concentrations have increased through time, relative to reference lakes and streams, in all monitored lakes and streams downstream of the LLCF with the exception of Leslie-Moose Stream (Table 3.2-10; Figure 3.2-5). Only four years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends improbable; however, graphical analysis shows that sulphate concentrations in Leslie-Moose Stream were similar to those in the LLCF in all years during which Leslie-Moose Stream was monitored. In most lakes and streams, sulphate concentrations have stabilised in recent years (Figure 3.2-5) however, sulphate concentrations in 2013 have continued to increase at site S2 during the ice-covered and open water seasons and in Slipper Lake and Slipper-Lac de Gras Stream during the open water season (Figure 3.2-5). Sulphate concentrations decreased with downstream distance from the LLCF (Figure 3.2-5).

At sites that are not downstream of the LLCF, statistical analyses indicate that sulphate concentrations have changed through time, relative to reference sites, in Kodiak Lake and the Lower PDC during the open water season (Table 3.2-10). However, graphical analysis suggests that concentrations have either remained relatively stable over time or have remained within the range of concentrations observed during baseline years at all sites except possibly Kodiak Lake during the open water season (Figure 3.2-5). The increase since about 2002 in sulphate concentrations in Kodiak Lake were observed only during the open water season, but may reflect mine-related activities at the main camp or airport.

Figure 3.2-5

Observed and Fitted Means for Sulphate Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
SSWQO = $e^{(0.9116 \times \ln(Hardness) + 1.712)}$ mg/L, where hardness < 160 mg/L.

Table 3.2-10. Statistical Results of Sulphate Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts: Model Fit = 3	Significant Monitored Contrasts: Model Fit = 2	Significant Monitored Contrasts: Model Fit = 1	Statistical Report Page No.
Apr	Lake	-	LME	2	-	Leslie, Moose, Nema, Slipper, S3	-	1-70
Aug	Lake	-	LME	3	Grizzly, Kodiak, 1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2, S3	Kodiak, 1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2, S3	-	1-76
Aug	Stream	-	LME	1b	-	-	Lower PDC, 1616-30 (LLCF), Moose-Nero, Nema-Martine, Slipper-Lac de Gras	1-82

Dashes indicate not applicable.

The 95% confidence intervals of the fitted mean and the observed mean sulphate concentrations were less than the hardness-dependent sulphate SSWQO in all reference and monitored lakes and streams in 2013 (Rescan 2012f). Sulphate concentrations were also less than the hardness-dependent sulphate SSWQO in all monitored streams in June, July, August and September in 2013 (see Part 2 – Data Report; Rescan 2012f).

3.2.4.6 Potassium

> ***Summary:*** Statistical and graphical analyses suggest that potassium concentrations have increased at all monitored sites that are downstream of the LLCF as far as site S3 in Lac de Gras as a result of mine operations. The observed means exceeded the long-term potassium SSWQO in Leslie and Moose lakes during the ice-covered season. No mine effects were detected at sites that are not downstream of the LLCF.

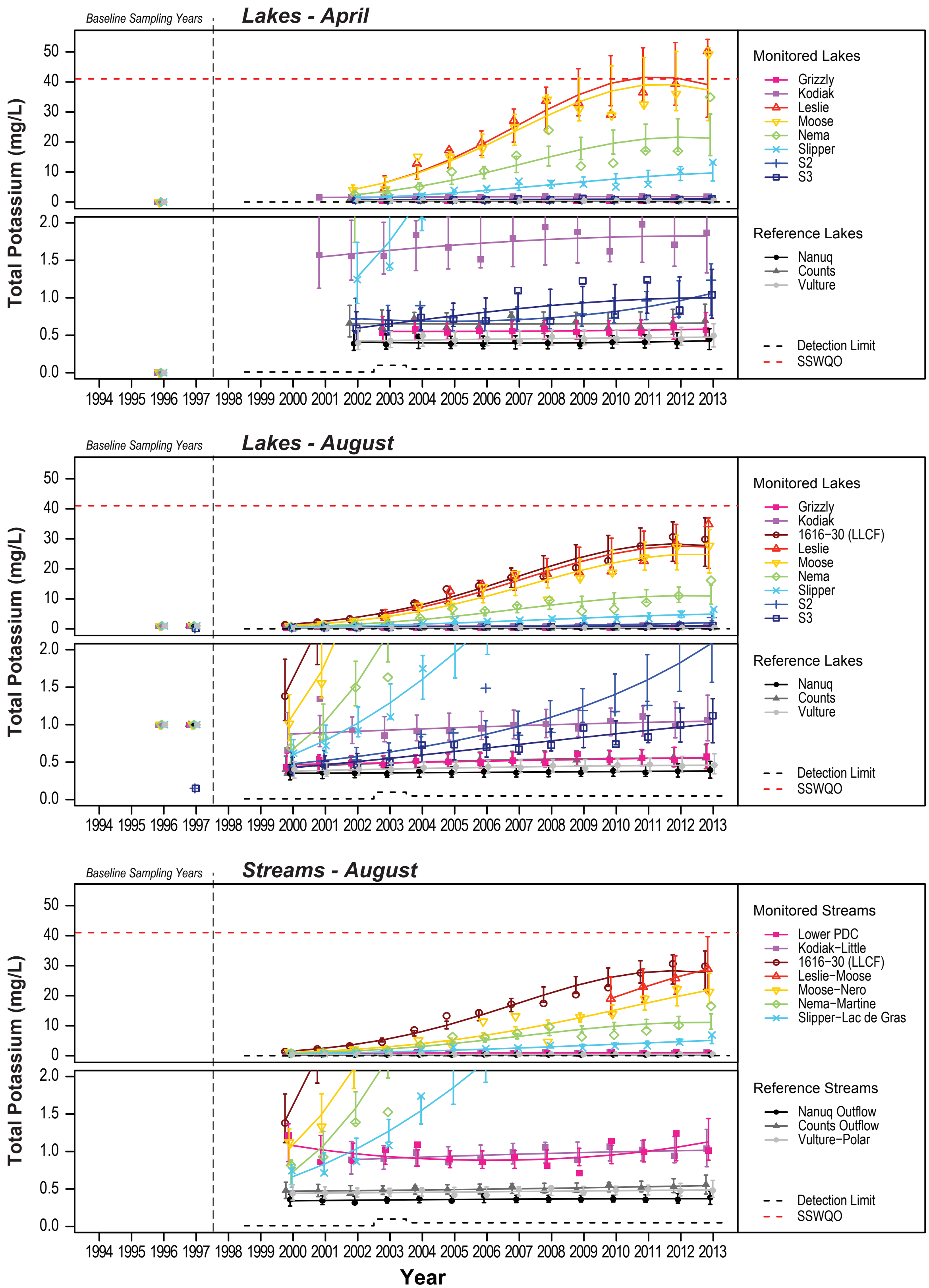
Statistical analyses indicate that temporal trends in potassium concentrations differ from those observed at reference sites at all monitored sites downstream of the LLCF as far as Slipper Lake during the ice-covered season and as far as site S3 in Lac de Gras during the open water season, with the exception of Leslie-Moose Stream (Table 3.2-11). Only four years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends improbable; however, graphical analysis shows that potassium concentrations in Leslie-Moose Stream have been increasing and were similar to those in the LLCF in recent years. Graphical analysis suggests that the concentration of potassium has increased in all monitored lakes and streams downstream of the LLCF as far as site S3 in Lac de Gras, with concentrations decreasing with downstream distance from the LLCF (Figure 3.2-6).

At sites that are not downstream of the LLCF, statistical and graphical analyses indicate that potassium concentrations have been stable through time in all monitored lakes and streams (Table 3.2-11; Figure 3.2-6). No mine effects were detected at sites that are not downstream of the LLCF.

The observed mean potassium concentration exceeded the long-term potassium SSWQO of 41 mg/L in Leslie and Moose lakes during the ice-covered season (Figure 3.2-6; see Part 3 – Statistical Report; Rescan 2012g). Observed potassium concentrations were less than the long-term SSWQO in all monitored streams in June, July, August and September in 2013 (see Part 2 – Data Report; Rescan 2012g).

Figure 3.2-6

Observed and Fitted Means for Potassium Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
SSWQO = 41 mg/L.

Table 3.2-11. Statistical Results of Potassium Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	3	Leslie, Moose, Nema, Slipper	Leslie, Moose, Nema, Slipper	-	1-88
Aug	Lake	-	LME	2	-	1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2, S3	-	1-94
Aug	Stream	-	Tobit	2	-	1616-30 (LLCF), Moose-Nero, Nema-Martine, Slipper-Lac de Gras	-	1-100

Dashes indicate not applicable.

3.2.4.7 Total Ammonia-N

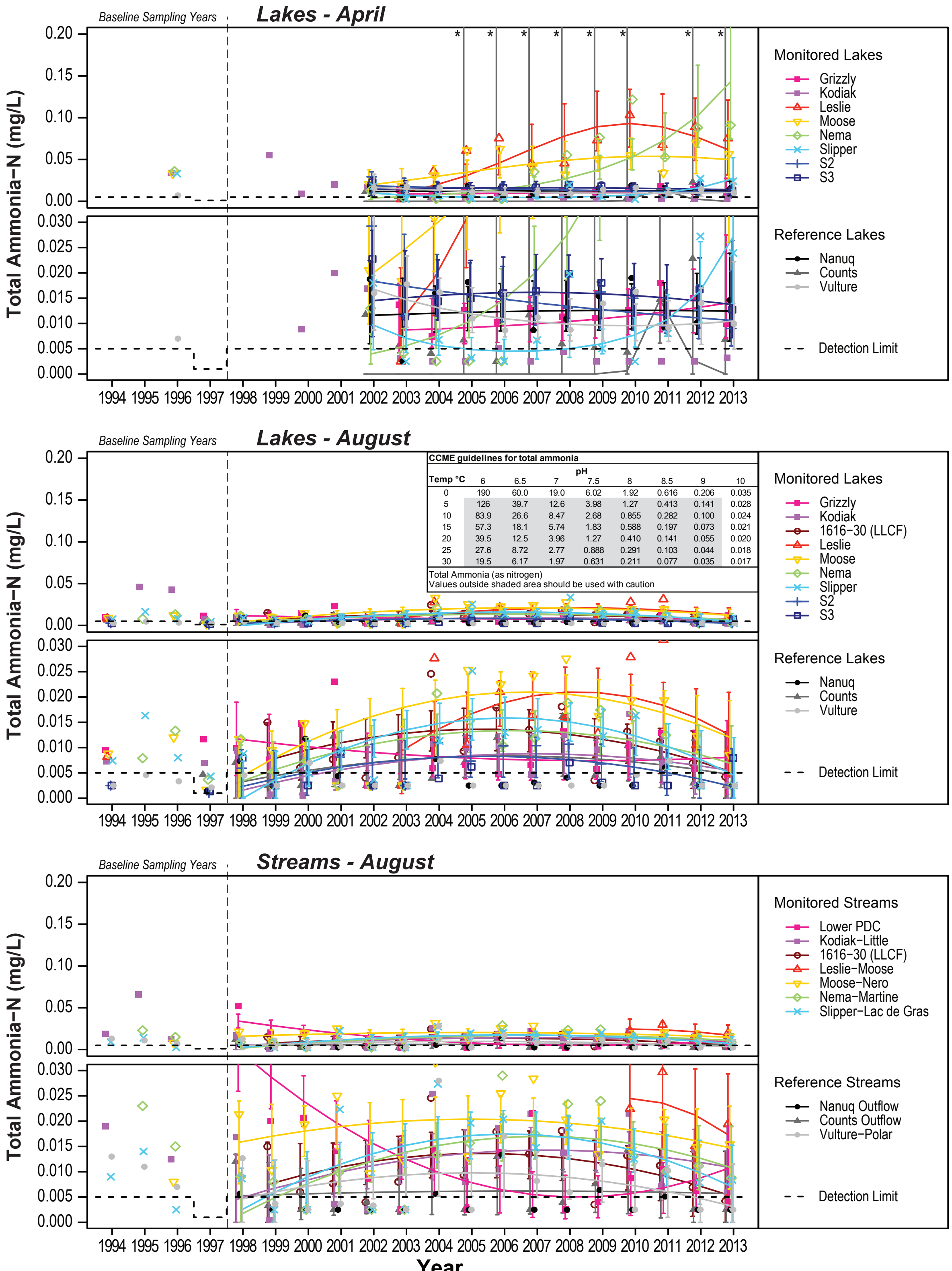
Summary: Statistical and graphical analyses suggest that total ammonia-N concentrations have increased relative to reference lakes at all lakes downstream of the LLCF as far as Slipper Lake, but have remained relatively low and stable in streams. Total ammonia-N concentrations in Leslie, Moose, and Nema lakes has stabilised or decreased in recent years. The 95% confidence interval around the fitted mean total ammonia-N concentration exceeded the pH- and temperature-dependent CCME guideline in Counts Lake during the ice-covered season in 2013. Observed total ammonia-N concentrations were less than pH- and temperature-dependent CCME guidelines at all monitored sites in 2013. No mine effects were detected in lakes or streams that are not downstream of the LLCF.

Statistical analyses indicate that total ammonia-N concentrations have changed through time, relative to reference lakes, at all monitored lakes downstream of the LLCF as far as Slipper Lake during the ice-covered season and in Moose Lake during the open water season (Table 3.2-12). Graphical analysis suggests that total ammonia-N has increased in Leslie, Moose, Nema, and Slipper lakes during both the ice-covered and open water seasons (Figure 3.2-7). Observed concentrations have stabilised or declined in recent years in all lakes except Slipper Lake in which observed concentrations have increased during the ice-covered season in recent years (Figure 3.2-7). Trends are likely more defined during the ice-covered season than in the open water season because oxidisation of ammonia-N to nitrite, then nitrate (a highly bioavailable form of nitrogen) occurs more rapidly during the summer. In streams, total ammonia-N concentrations have been relatively low and stable since monitoring began (Figure 3.2-7). Total ammonia-N concentrations were generally greater in lakes and streams downstream of the LLCF than in reference sites and decreased with downstream distance of the LLCF (Figure 3.2-7). The observed increases in total ammonia-N likely stem from blasting-related ammonia residues in processed kimberlite and can be observed as far downstream as Slipper Lake (Figure 3.2-7).

At sites that are not downstream of the LLCF, statistical analyses indicate that total ammonia-N concentrations have been stable through time at all monitored sites except the Lower PDC during the open water season (Table 3.2-12). However, graphical analysis suggests that total ammonia-N concentrations have declined from initially elevated concentrations in the Lower PDC and have remained within the range of concentrations observed during baseline years in Kodiak Lake (Figure 3.2-7).

Figure 3.2-7

Observed and Fitted Means for Total Ammonia-N Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



CCME guidelines for total ammonia

Temp °C	pH 6	6.5	7	7.5	8	8.5	9	10
0	190	60.0	19.0	6.02	1.92	0.616	0.206	0.035
5	126	39.7	12.6	3.98	1.27	0.413	0.141	0.028
10	83.9	26.6	8.47	2.68	0.855	0.282	0.100	0.024
15	57.3	18.1	5.74	1.83	0.588	0.197	0.073	0.021
20	39.5	12.5	3.96	1.27	0.410	0.141	0.055	0.020
25	27.6	8.72	2.77	0.888	0.291	0.103	0.044	0.018
30	19.5	6.17	1.97	0.631	0.211	0.077	0.035	0.017

Total Ammonia (as nitrogen)
Values outside shaded area should be used with caution

Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
WL = Maximum average concentration premitted in water licence W2009L2-0001. WL Criterion = 2.0 mg/L.
CCME Guideline is pH and temperature dependent (see inset table).
** Upper 95% Confidence Interval on the fitted mean of Counts Lake in April 2005 = $2.10x10^{283}$ mg/L, 2006 = $1.08x10^{250}$ mg/L, 2007 = $1.12x10^{211}$ mg/L, 2008 = $2.35x10^{166}$ mg/L, 2009 = $9.97x10^{115}$ mg/L, 2010 = $8.54x10^{59}$ mg/L, 2012 = $1.49x10^{64}$ mg/L, 2013 = $4.98x10^{131}$ mg/L, and of Nema Lake in 2013 = 0.274 mg/L.*

Table 3.2-12. Statistical Results of Ammonia-N Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	Kodiak	Tobit	1b	-	-	Leslie, Moose, Nema, Slipper	1-106
Aug	Lake	S3, Nanuq, Vulture	Tobit	1b	-	-	Moose	1-112
Aug	Stream	Nanuq Outflow	Tobit	3	Lower PDC, Moose-Nero, Nema-Martine, Slipper-Lac de Gras	Lower PDC	-	1-117

Dashes indicate not applicable.

The 95% confidence interval around the fitted mean exceeded the pH- and temperature-dependent CCME guideline for ammonia-N in Counts Lake during the ice-covered season in 2013 (CCME 2001). Observed total ammonia-N concentrations were less than pH- and temperature-dependent CCME guidelines at all monitored lake and stream sites in 2013 (see Part 2 - Data Report; CCME 2001).

3.2.4.8 Nitrite-N

Summary: Statistical and graphical analyses suggest that nitrite-N concentrations have increased at sites downstream of the LLCF as far as Moose-Nero Stream. Increased nitrite-N concentrations are likely associated with the oxidation of ammonia from blast-residue in processed kimberlite. No mine effects were detected at sites that are not downstream of the LLCF. Observed and fitted mean concentrations were less than the nitrite-N CCREM guideline at all sites in 2013.

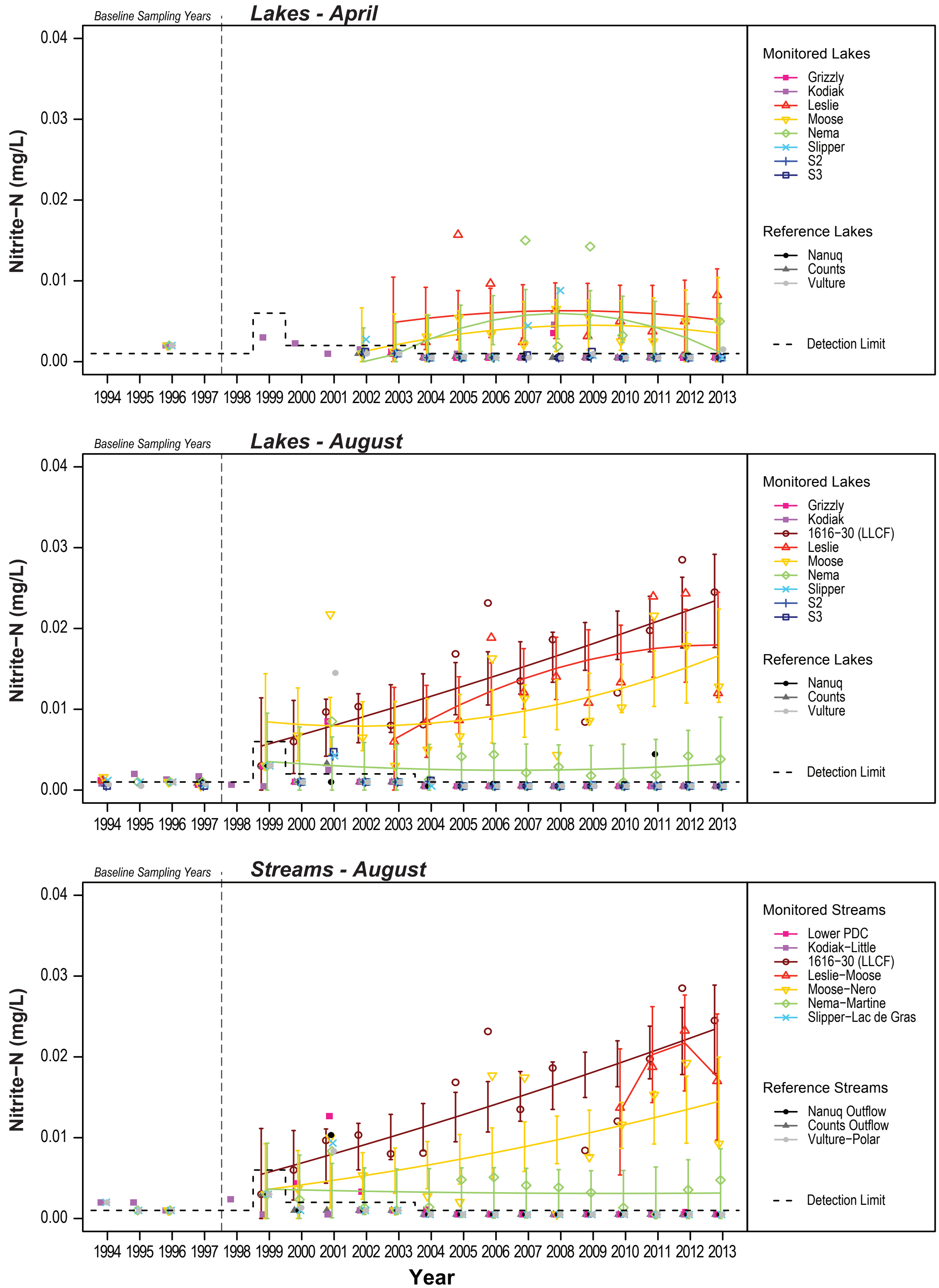
Nitrite-N concentrations were less than detection limits at all reference sites in 2013 (Table 3.2-13; Figure 3.2-8). Statistical analyses suggest that nitrite-N concentrations during the open water season have changed through time in monitored lakes and streams as far as Moose-Nero Stream, except for Leslie-Moose Stream (Table 3.2-13; Figure 3.2-8). Only four years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends improbable; however, graphical analysis shows that nitrite-N concentrations in Leslie-Moose Stream were similar to those in the LLCF in recent years. Graphical analyses suggest that nitrite-N concentrations have increased through time in Leslie and Moose lakes and in Moose-Nero Stream during the open water season (Figure 3.2-8). However, observed concentrations in Leslie and Moose lakes were lower in 2013, which was also reflected in lower observed concentrations in Leslie-Moose and Moose-Nero streams. In general, nitrite-N concentrations decrease with downstream distance of the LLCF as far as Moose-Nero Stream during the open water season. Elevated concentrations of nitrite-N at sites downstream of the LLCF are likely blasting-related, as ammonia residue from processed kimberlite is oxidised to nitrite.

No temporal trends were observed in any of the monitored sites that are not downstream of the LLCF (Table 3.2-13; Figure 3.2-8).

The 95% confidence intervals of the fitted mean and the observed mean nitrite-N concentrations were less than the 0.06 mg/L CCME guideline value for nitrite-N in all reference and monitored lakes in April and August 2013 (CCREM 1987). Nitrite-N concentrations were also less than the CCREM guideline value for nitrite-N in all reference and monitored streams in June, July, August and September in 2013 (see Part 2 - Data Report; CCREM 1987).

Figure 3.2-8

Observed and Fitted Means for Nitrite-N Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
CCME Guideline = 0.06 mg/L.

Table 3.2-13. Statistical Results of Nitrite-N Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	Grizzly, Kodiak, Slipper, S2, S3, Nanuq, Counts, Vulture	Tobit	1a	-	-	None	1-123
Aug	Lake	Grizzly, Kodiak, Slipper, S2, S3, Nanuq, Counts, Vulture	Tobit	1a	-	-	1616-30 (LLCF), Leslie, Moose	1-128
Aug	Stream	Kodiak-Little, Lower PDC, Slipper-Lac de Gras, Nanuq Outflow, Counts Outflow, Vulture-Polar	Tobit	1a	-	-	1616-30 (LLCF), Moose-Nero	1-133

Dashes indicate not applicable.

3.2.4.9 Nitrate-N

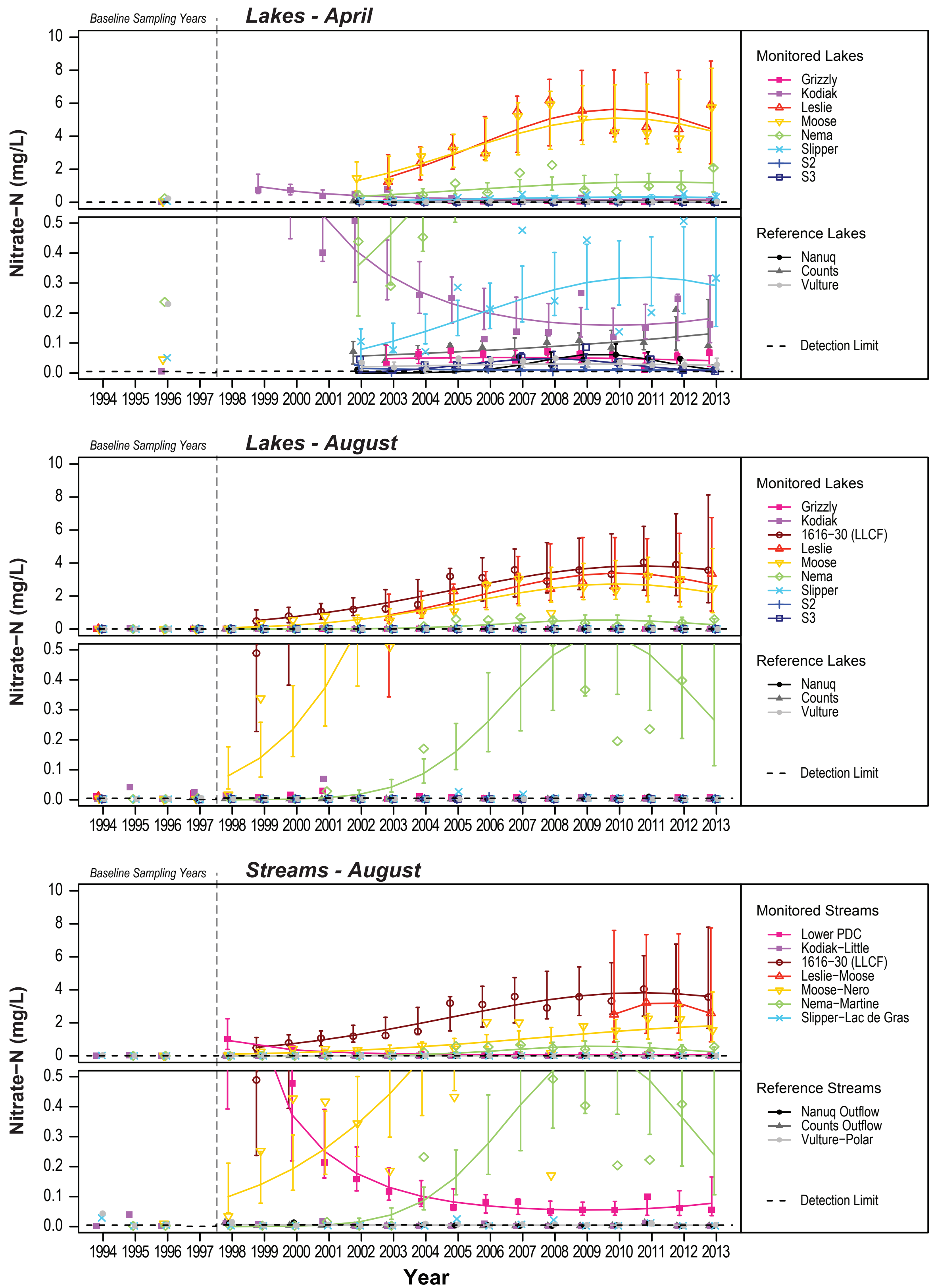
Summary: Statistical and graphical analyses suggest that nitrate-N concentrations have increased in monitored lakes and streams downstream of the LLCF as far as Slipper Lake as a result of mine operations. Increased nitrate-N concentrations downstream from the LLCF are likely associated with the oxidation of ammonia (and then nitrite) associated with the blast-residue in processed kimberlite. In all cases, concentrations have stabilised in recent years No mine effects were detected at sites that were not downstream of the LLCF. Observed and fitted mean concentrations were less than the hardness-dependent nitrate-N SSWQO at all sites in 2013.

Statistical analyses indicate that nitrate-N concentrations have changed through time, relative to reference lakes, in all monitored lakes downstream of the LLCF except site S2 in Lac de Gras during the ice-covered season. Statistical analyses also indicate that nitrate-N concentrations have changed through time in all lakes as far as Moose Lake during the open water season (Table 3.2-14). Statistical analyses also indicate that concentrations have changed through time in Moose-Nero and Nema-Martine streams during the open water season (Table 3.2-14). Graphical analysis suggests that concentrations of nitrate-N have increased through time at monitored lakes and streams downstream of the LLCF as far as Slipper Lake during the ice-covered season and as far as Nema-Martine Stream during the open water season, with concentrations decreasing with downstream distance from the LLCF (Figure 3.2-9). In all cases, concentrations have stabilised in recent years (Figure 3.2-9). The increase in nitrate-N in lakes and streams downstream from the LLCF likely stems from an increase in total ammonia-N associated with blast-residue in processed kimberlite since ammonia oxidises to nitrite, which then oxidises to nitrate,a highly bioavailable form of nitrogen.

Statistical analyses indicate that nitrate-N concentrations have changed through time in Kodiak Lake during the ice-covered season and in the Lower PDC during the open water season (Table 3.2-14). However, graphical analysis suggests that nitrate-N concentrations have decreased through time from initially elevated levels in both the Lower PDC and Kodiak Lake (Figure 3.2-9). Thus, no mine effects were detected at sites that are not downstream of the LLCF.

Figure 3.2-9

Observed and Fitted Means for Nitrate-N Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
SSWQO = $e^{0.9518 \times \ln(Hardness) - 2.032}$ mg/L, where hardness < 160mg/L.

Table 3.2-14. Statistical Results of Nitrate-N Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	Tobit	1b	-	-	Kodiak, Leslie, Moose, Nema, Slipper, S3	1-138
Aug	Lake	Grizzly, Kodiak, Slipper, S2, S3, Nanuq, Counts, Vulture	Tobit	1a	-	-	1616-30 (LLCF), Leslie, Moose	1-144
Aug	Stream	Kodiak-Little, Slipper-Lac de Gras, Nanuq Outflow, Counts Outflow	Tobit	1b	-	-	Lower PDC, 1616-30 (LLCF), Moose-Nero, Nema-Martine	1-149

Dashes indicate not applicable.

The 95% confidence intervals of the fitted mean and the observed mean nitrate-N concentrations were less than the hardness-dependent nitrate-N SSWQO in all reference and monitored lakes in 2013 (Health Canada 1987; Rescan 2012e). Nitrate-N concentrations were also less than the nitrate-N SSWQO in all monitored streams in April, June, July, August and September(see Part 2 – Data Report; Health Canada 1987; Rescan 2012e).

3.2.4.10 Total Phosphate-P

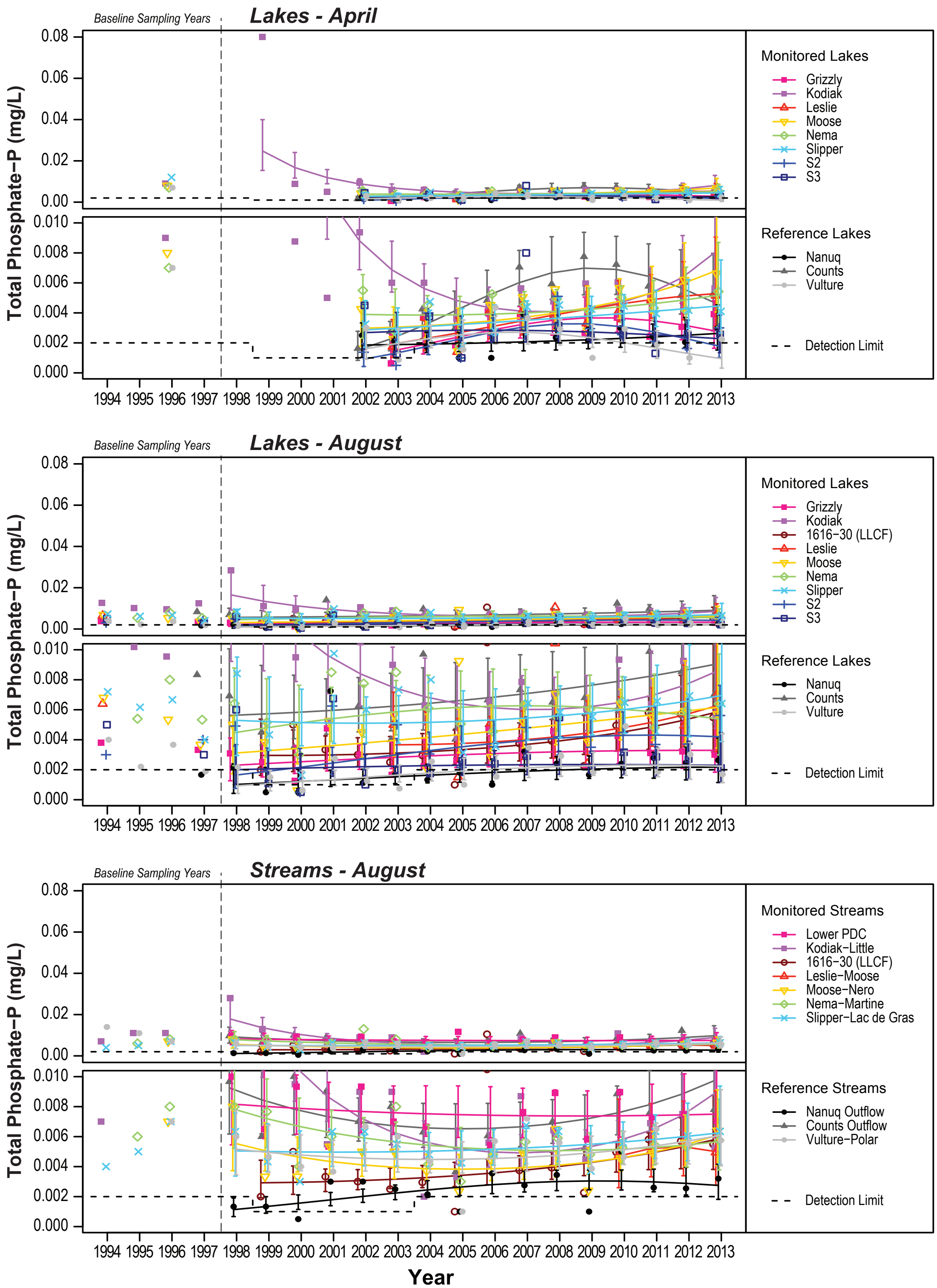
> ***Summary:*** Statistical and graphical analyses suggest that total phosphate-P concentrations have increased in lakes downstream of the LLCF as far as Moose Lake. No mine effects were detected at sites that are not downstream of the LLCF. In several cases, the upper 95% confidence interval on the fitted mean total phosphate-P concentration was greater than the 0.01 mg/L or mean baseline concentrations + 50% triggers during the ice-covered and open water seasons. However, similar patterns were observed in all three reference lakes. The observed mean at site S2 in Lac de Gras also exceeded the benchmark during the open water season, but a similar pattern was also observed in Nanuq Lake.

Statistical and graphical analyses indicate that total phosphate-P concentrations have increased through time, relative to reference lakes, in Leslie and Moose lakes during the ice-covered season (Table 3.2-15). However, total phosphate-P concentrations have shown no signs of change during the open water season in either Leslie or Moose lakes (Table 3.2-15; Figure 3.2-10). Total phosphate-P concentrations have also been stable through time in all monitored streams that are downstream of the LLCF (Table 3.2-15; Figure 3.2-10). The differences between trends observed in the ice-covered and open water season may be related to increased rates of biological uptake during the open water season and are likely related to the addition of phosphorus to the LLCF in 2009, 2010, and 2011.

At sites that are not downstream of the LLCF, statistical analyses indicate that total phosphate-P concentrations have changed relative to reference sites in Kodiak Lake and Kodiak-Little Stream (Table 3.2-15). Graphical analysis suggests that total phosphate-P concentrations have declined from initially high concentrations (Figure 3.2-10). Apparent declines in total phosphate-P concentrations in Kodiak Lake and Kodiak-Little Stream result from anomalously, but uniformly, high concentrations in 1998 and 1999, likely related to input of treated sewage into Kodiak Lake between 1997 and 1999 (see Part 2 – Data Report and Part 3 - Statistical Report; Rescan 2002). In all other years, total phosphate-P concentrations have been relatively low and stable in Kodiak Lake and Kodiak-Little (Figure 3.2-10). Therefore no mine effects were detected.

Figure 3.2-10

Observed and Fitted Means for Total Phosphate-P Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.

Table 3.2-15. Statistical Results of Total Phosphate-P Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	Tobit	1b	-	-	Kodiak, Leslie, Moose	1-154
Aug	Lake	-	Tobit	2	-	Kodiak	-	1-160
Aug	Stream	-	Tobit	1b	-	-	Kodiak-Little, 1616-30 (LLCF)	1-166

Dashes indicate not applicable.

The 95% confidence intervals of fitted mean total phosphate-P concentrations were greater than the 0.01 mg/L trigger set for oligotrophic lakes in the Canadian Guidance Framework for the management of Phosphorus in Freshwater Systems in Slipper and Counts lakes during the open water season in 2013 (Figure 3.2-12; CCME 2004; Environment Canada 2004). The 95% confidence intervals of fitted mean total phosphate-P concentrations were also greater than the recommended benchmark trigger of mean baseline concentration + 50% (Ontario Ministry of Natural Resources 1994; CCME 2004; Environment Canada 2004), in Leslie, Moose, Nema, Nanuq, and Vulture lakes and at site S2 in Lac de Gras during the open water season and in Moose and Nanuq lakes during the ice-covered season in 2013. Although the 95% confidence intervals around the fitted means exceeded trigger or benchmark values in some lakes, the mean observed and fitted concentrations were less than both the 0.01 mg/L trigger concentration and the mean baseline concentration + 50% at all monitored sites. The only exception was site S2, where the observed mean exceeded the benchmark during the open water season. In reference lakes, the observed mean for Nanuq Lake exceeded the benchmark during the open water season while both the observed and fitted means exceeded the benchmark during the ice-covered season. Overall, similar patterns in exceedence of the relevant guideline values were observed in monitored and reference lakes indicating a regional effect and not a mine effect.

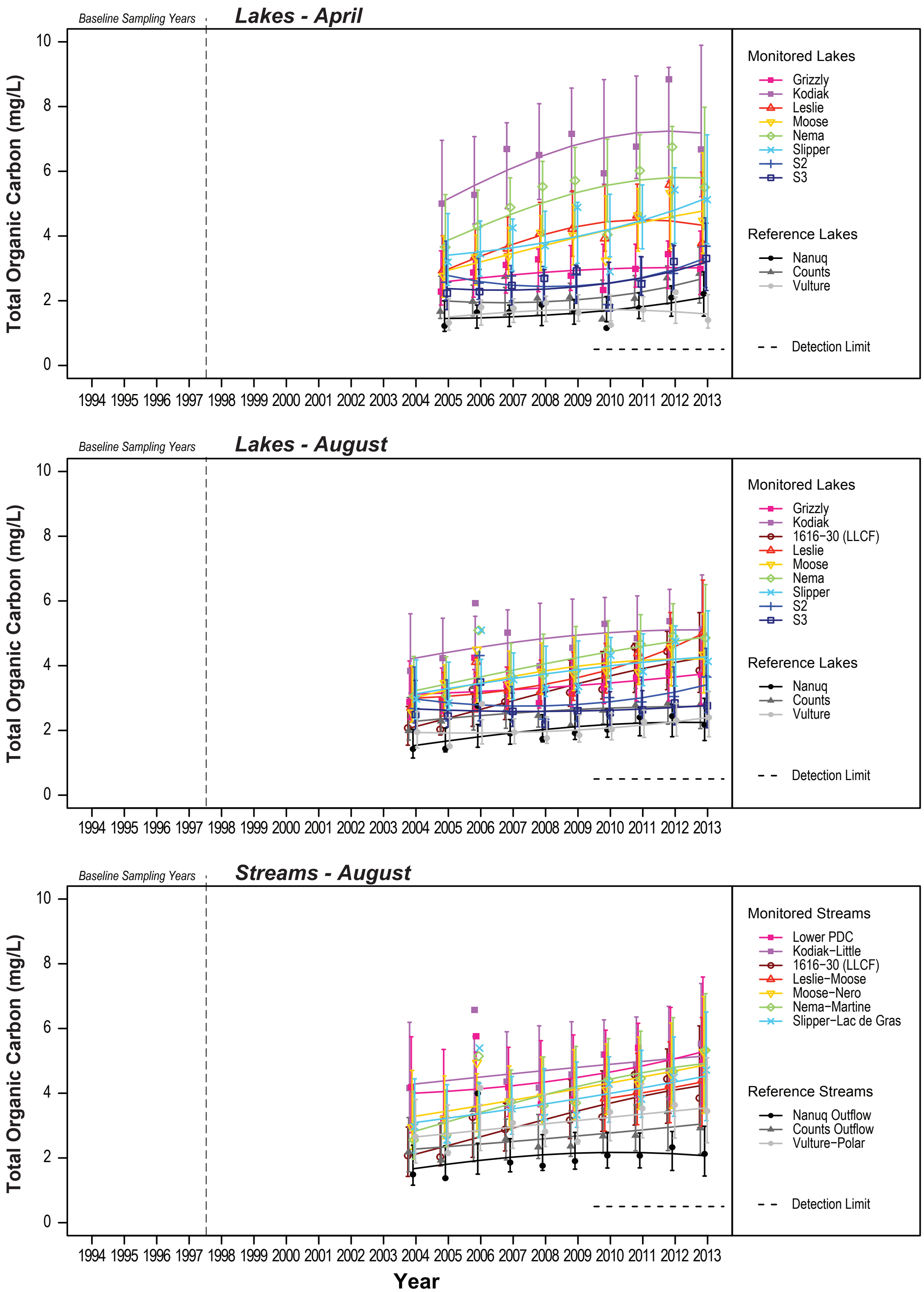
3.2.4.11 TOC

> ***Summary:*** Although graphical and statistical analyses indicate that TOC concentrations have changed through time in all monitored lakes downstream of the LLCF as far as Nema Lake, no clear downstream spatial gradient was present.

Statistical analyses indicate that total organic carbon (TOC) concentrations have changed through time, relative to reference lakes, in all monitored lakes downstream of the LLCF as far as site S2 during both the ice-covered and open water seasons (Table 3.2-16). In contrast, statistical analyses indicate that TOC has been stable through time, relative to reference streams, in all monitored streams downstream of the LLCF. Graphical analysis suggests that TOC may have increased through time in lakes as far downstream as Nema Lake during both the ice-covered season and open water seasons (Figure 3.2-11). However, no clear downstream spatial gradients in TOC concentrations are apparent. For example, TOC concentrations in some downstream lakes (i.e., Nema Lake) have been consistently greater than TOC concentrations in lakes that are closer to the LLCF since monitoring began (Figure 3.2-11). Temporal trends in TOC concentrations are somewhat difficult to discern given the uncertainty in estimating changes in the mean concentrations, as evidenced by relatively large confidence intervals on the fitted means (Figure 3.2-11). The evaluation of changes in TOC concentrations are also complicated by the fact that TOC concentrations were not measured during baseline years. This uncertainty and lack of baseline information makes it difficult to determine whether observed patterns result from mine effects or represent natural regimes. Thus it was concluded that no mine effects were detected at sites that are downstream of the LLCF.

Figure 3.2-11

Observed and Fitted Means for Total Organic Carbon in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.

Table 3.2-16. Statistical Results of Total Organic Carbon in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	1b	-	-	Kodiak, Leslie, Moose, Nema, Slipper, S2	1-172
Aug	Lake	-	LME	1b	-	-	Kodiak, 1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2	1-178
Aug	Stream	-	LME	2	-	none	-	1-184

Dashes indicate not applicable.

At sites that are not downstream of the LLCF, statistical analyses indicate that TOC concentrations have changed through time, relative to reference lakes, in in Kodiak Lake during the ice-covered and open water seasons (Table 3.2-16). However, graphical analysis suggests that TOC concentrations have remained stable through time in Kodiak Lake (Figure 3.2-11). Elevated concentrations in Kodiak Lake compared to all other lakes are likely related to the input of treated sewage between 1997 and 1999 (Figure 3.2-11). No mine effects were detected.

3.2.4.12 Total Antimony

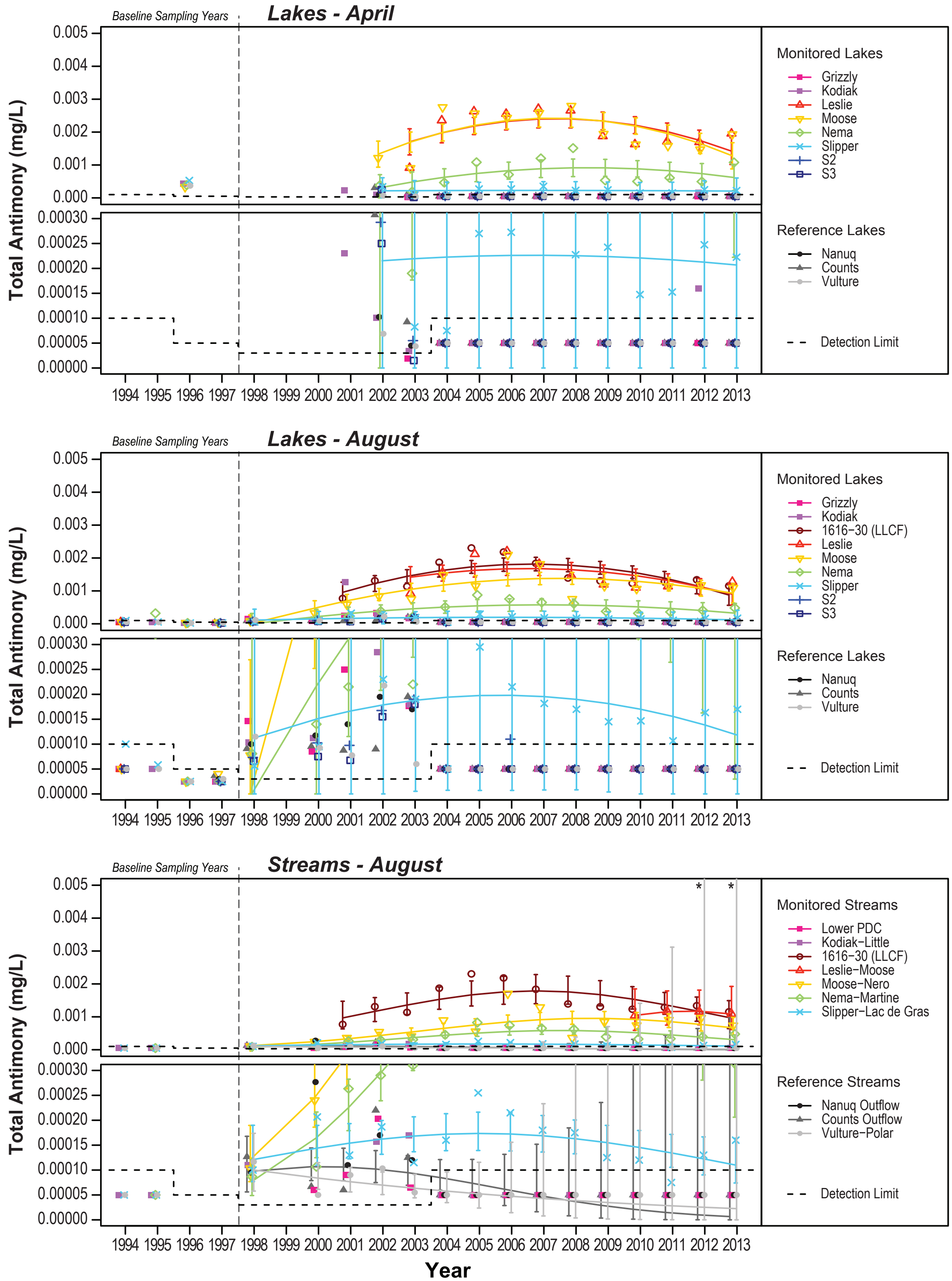
Summary: Together, statistical and graphical analyses suggest that total antimony concentrations have declined in recent years but remain elevated above baseline and reference conditions at monitored sites downstream of the LLCF as far as Moose-Nero Stream during the open water season. No mine effects were detected at sites that are not downstream of the LLCF. Observed and fitted mean concentrations were less than the antimony water quality benchmark (0.02 mg/L) at all sites in 2013.

Statistical analyses indicate that total antimony concentrations have changed through time in Leslie and Moose lakes during the ice-covered and open water seasons and in Moose-Nero and Nema-Martine streams (Table 3.2-17). Graphical analysis suggests that total antimony concentrations increased to peak concentrations in 2007 and have since stabilised or declined during both the ice-covered and open water seasons in Leslie and Moose lakes (Figure 3.2-12). Previous AEMP reports (Rescan 2012b, 2013b) indicated that total antimony concentrations had increased downstream of the LLCF as far as Slipper-Lac de Gras Stream. However, results of the 2013 evaluation of effects suggest that total antimony concentrations have attenuated in recent years and have returned to values within the range of those observed when monitoring began at sites downstream of the LLCF (Figure 3.2-12). However, concentrations of total antimony in lakes and streams downstream of the LLCF as far as Moose-Nero Stream remain above baseline and reference lake concentrations, with concentrations decreasing with downstream distance of the LLCF (Figure 3.2-12). No temporal trends were observed in sites not downstream of the LLCF, where concentrations have generally been less than analytical detection limits since monitoring began (Table 3.2-17; Figure 3.2-12).

The 95% confidence intervals around fitted mean and the observed mean total antimony concentrations in all monitored lakes and streams in 2013 were less than the antimony water quality benchmark (0.02 mg/L; Fletcher et al. 1996). Antimony concentrations were also less than the benchmark in monitored streams in June, July , August and September 2013 (see Part 2 - Data Report; Fletcher et al. 1996).

Figure 3.2-12

Observed and Fitted Means for Total Antimony Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
** Upper 95% Confidence Interval on the fitted mean of Vulture-Polar Stream in August 2012 = 0.0076 mg/L, 2013 = 0.0202 mg/L.*
Water quality benchmark (Fletcher et al. 1996) = 0.02 mg/L.

Table 3.2-17. Statistical Results of Total Antimony Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	Grizzly, Kodiak, S2, S3, Nanuq, Counts, Vulture	LME	1a	-	-	Leslie, Moose	1-190
Aug	Lake	Grizzly, Kodiak,S2, S3, Nanuq, Counts, Vulture	LME	1a	-	-	1616-30 (LLCF), Leslie, Moose	1-195
Aug	Stream	Kodiak-Little, Lower PDC, Nanuq Outflow	Tobit	3	1616-30 (LLCF), Moose-Nero, Nema-Martine, Slipper-Lac de Gras	1616-30 (LLCF), Moose-Nero, Nema-Martine	-	1-200

Dashes indicate not applicable.

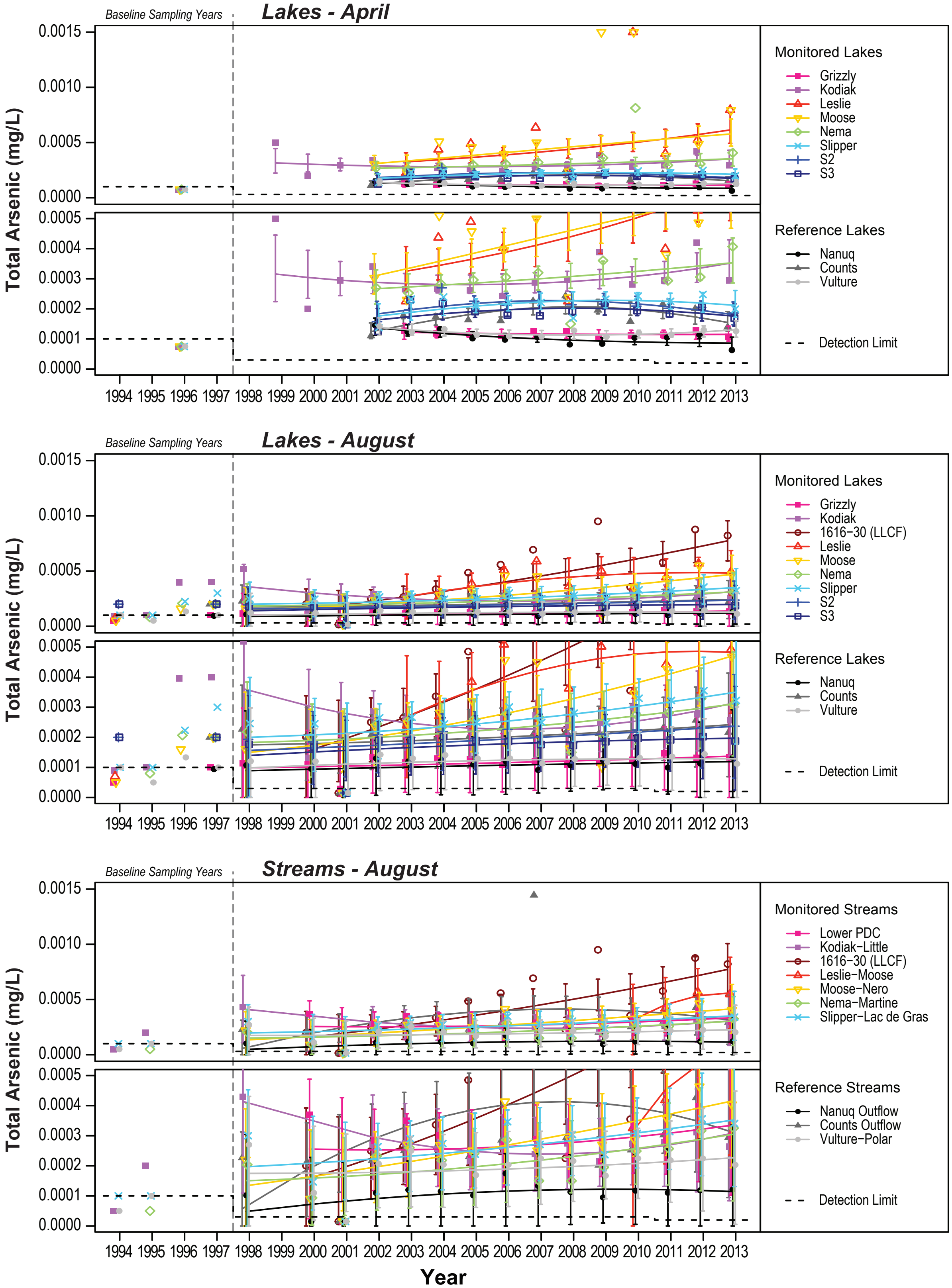
3.2.4.13 Total Arsenic

Summary: Together, statistical and graphical analyses suggest that total arsenic concentrations have increased downstream of the LLCF as far as Moose Lake as a result of mine operations. No mine effects were detected at sites that are not downstream of the LLCF. Observed and fitted mean concentrations were less than the arsenic CCME guideline at all sites in 2013.

Statistical and graphical analyses indicate that total arsenic concentrations have changed through time, relative to reference lakes, in Leslie and Moose lakes during the ice-covered season (Table 3.2-18; Figure 3.2-13). However, no differences in temporal trends were observed between reference and monitored lakes or streams downstream of the LLCF during the open water season (Table 3.2-18; Figure 3.2-13). Together, statistical and graphical analyses suggest that total arsenic concentrations have increased in lakes and streams downstream of the LLCF as far as Moose Lake. However, the observed trend in Leslie and Moose lakes during the ice-covered season may in part be related to variability in detection limits artificially inflating total arsenic concentrations, particularly in 2009 and 2010. Analysis of total arsenic concentrations in water from lakes downstream of the LLCF has become more difficult through time because elevated chloride concentrations can result in matrix interferences during the analysis of total arsenic concentrations in the laboratory. This is because chloride produces a species that has the same mass as the one that is measured for arsenic during Inductively Coupled Plasma Mass Spectrometry (ICPMS). Thus, the mass of arsenic or chloride cannot be distinguished and an accurate concentration of arsenic cannot be determined. To remove the interference, samples must be diluted prior to analysis. However, when samples are diluted, detection limits are increased accordingly. Thus, detection limits are often variable among samples and between years, particularly for Leslie Lake. This has made it somewhat difficult to discern clear patterns in the past. However, although elevated detection limits related to matrix interference were problematic in 2009 and 2010, a new analytical approach was introduced in 2011 (i.e. collision cell ICPMS) and the target detection limit of 0.00002 mg/L was achieved for all ice-covered and open water lake and stream samples except 1616-30 (LLCF) in 2011 and 2012, making it easier to discern trends in the last three years. In 2013, the target detection limit was also achieved in 1616-30 (LLCF).

Figure 3.2-13

Observed and Fitted Means for Total Arsenic Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
WL = Maximum average concentration premitted in water licence W2009L2-0001. WL = 0.50 mg/L. CCME Guideline = 0.005 mg/L.

Table 3.2-18. Statistical Results of Total Arsenic Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	Tobit	1b	-	-	Leslie, Moose	1-206
Aug	Lake	-	Tobit	3	Kodiak, 1616-30 (LLCF), Leslie, Moose, Slipper	1616-30 (LLCF)	-	1-212
Aug	Stream	-	Tobit	2	-	1616-30 (LLCF)	-	1-218

Dashes indicate not applicable.

No temporal changes in total arsenic concentrations were detected at sites that are not downstream of the LLCF (Table 3.2-18). Graphical analysis also suggests that total arsenic concentrations have been stable through in these lakes and streams (Figure 3.2-13). No mine effects were detected at sites that are not downstream of the LLCF.

The 95% confidence intervals around the fitted mean and the observed mean total arsenic concentrations were less than the arsenic CCME water quality guideline (0.005 mg/L) in all lakes and streams during both the ice-covered and open water seasons in 2013 (see Part 2 - Data Report; CCME 1999). Total arsenic concentrations did not exceed CCME guidelines in any of the monitored streams in in June, July, August or September 2013 (see Part 2 - Data Report; CCME 1999).

3.2.4.14 Total Barium

> ***Summary:*** Statistical and graphical analyses suggest that total barium concentrations have increased at all monitored sites that are downstream of the LLCF as far as site S2 in Lac de Gras as a result of mine operations. However, barium concentrations have stabilised at levels greater than observed historical and reference lake concentrations at sites as far downstream as Nema-Martine Stream since 2007. No mine effects were detected at sites that are not downstream of the LLCF. Observed and fitted mean concentrations were less than the barium water quality benchmark (1 mg/L) at all sites in 2013.

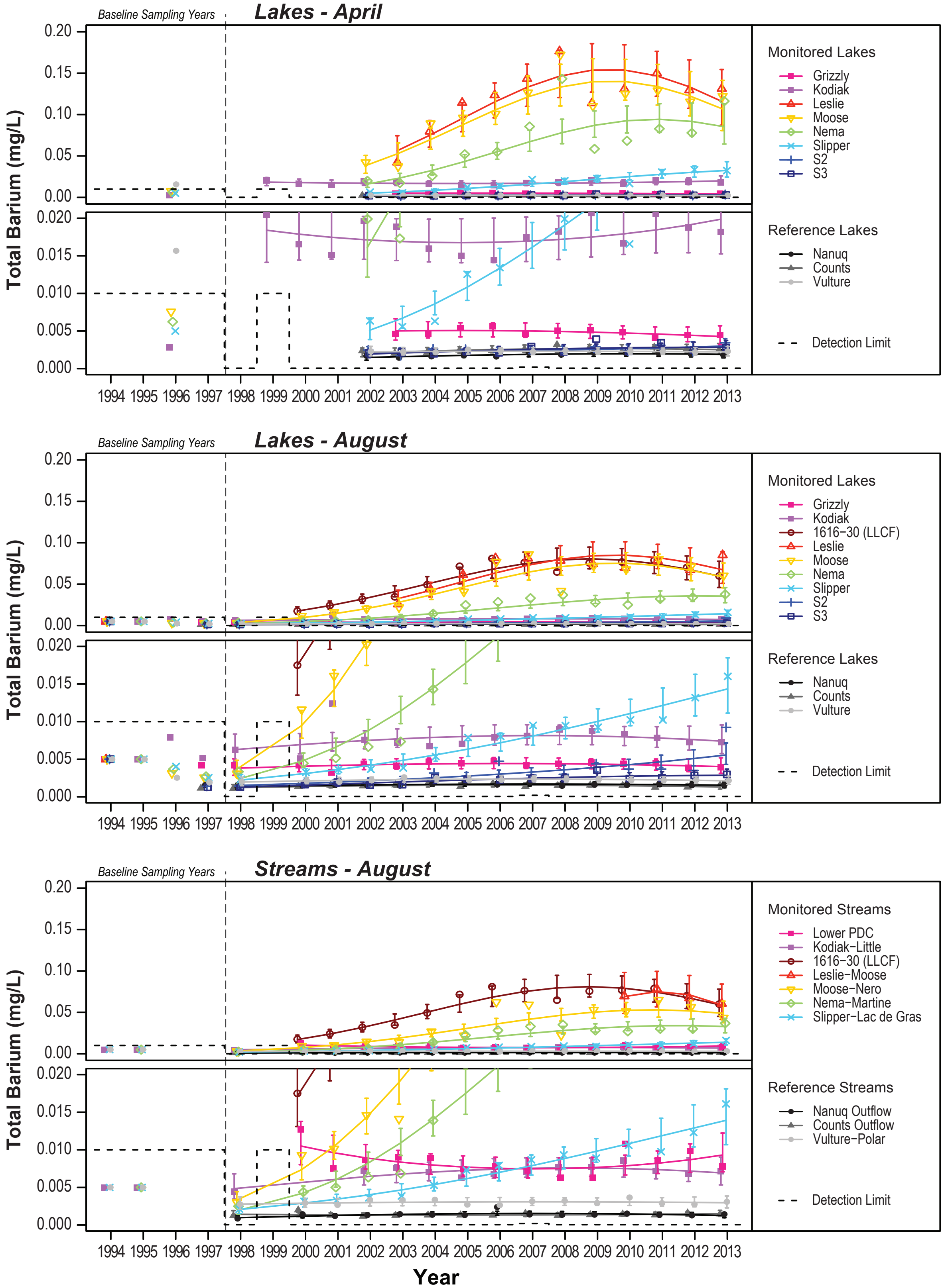
Statistical analyses indicate that temporal trends in total barium concentrations differ from those observed at reference sites at all monitored lakes and streams downstream of the LLCF, with the exception of sites S2 and S3 in Lac de Gras during the ice-covered season and site S3 and Leslie-Moose Stream during the open water season (Table 3.2-19). Graphical analysis suggests that the concentration of total barium has increased in all monitored lakes and streams downstream of the LLCF as far as site S2 in Lac de Gras, with concentrations decreasing with downstream distance from the LLCF (Figure 3.2-14). Graphical analysis also shows that concentrations at sites as far downstream as Nema-Martine Stream have stabilised at levels greater than historical and reference lake concentrations since about 2007 (Figure 3.2-14).

At sites that are not downstream of the LLCF, statistical and graphical analyses indicate that barium concentrations have been stable through time (Table 3.2-19; Figure 3.2-14). Thus, no mine effects were detected at sites that are not downstream of the LLCF.

The 95% confidence intervals of the 2013 fitted mean and the observed mean total barium concentrations were less than the barium water quality benchmark (1 mg/L; Haywood and Drinnan 1983). Total barium concentrations were also less than the benchmark in all monitored streams in June, July, August and September 2013 (see Part 2 - Data Report; Haywood and Drinnan 1983).

Figure 3.2-14

Observed and Fitted Means for Total Barium Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
Water quality benchmark (Haywood and Drinnan 1983) = 1 mg/L.

Table 3.2-19. Statistical Results of Total Barium Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	2	-	-	Leslie, Moose, Nema, Slipper	1-224
Aug	Lake	-	LME	2	-	-	1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2, S3	1-230
Aug	Stream	-	LME	1b	-	-	1616-30 (LLCF), Moose-Nero, Nema-Martine, Slipper-Lac de Gras	1-236

Dashes indicate not applicable.

3.2.4.15 Total Boron

Summary: Statistical and graphical analyses suggest that total boron concentrations have increased at all sites downstream of the LLCF as far as Slipper Lake as a result of mine operations, with total boron concentrations decreasing with increasing downstream distance from the LLCF. No mine effects were detected at sites that are not downstream of the LLCF. Observed and fitted mean concentrations were less than the boron CCME guideline at all sites in 2013.

Statistical analyses indicate that total boron concentrations have changed through time, relative to reference sites, in all monitored lakes and streams downstream of the LLCF as far as Slipper Lake, except for Leslie-Moose Stream (Table 3.2-20). Only four years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends improbable; however, graphical analysis shows that total boron concentrations in Leslie-Moose Stream have been increasing and were similar to those in the LLCF in recent years. Graphical analysis suggests that total boron concentrations have increased through time at all monitored sites as far as Slipper Lake, with concentrations decreasing with downstream distance from the LLCF (Figure 3.2-15). At sites that are not downstream of the LLCF, statistical analyses suggest that total boron concentrations have changed through time in Kodiak Lake during the ice-covered season. In contrast, graphical analyses suggest that total boron concentrations in Grizzly and Kodiak lakes, the Lower PDC, and Kodiak-Little have been stable through time (Table 3.2-20; Figure 3.2-15).

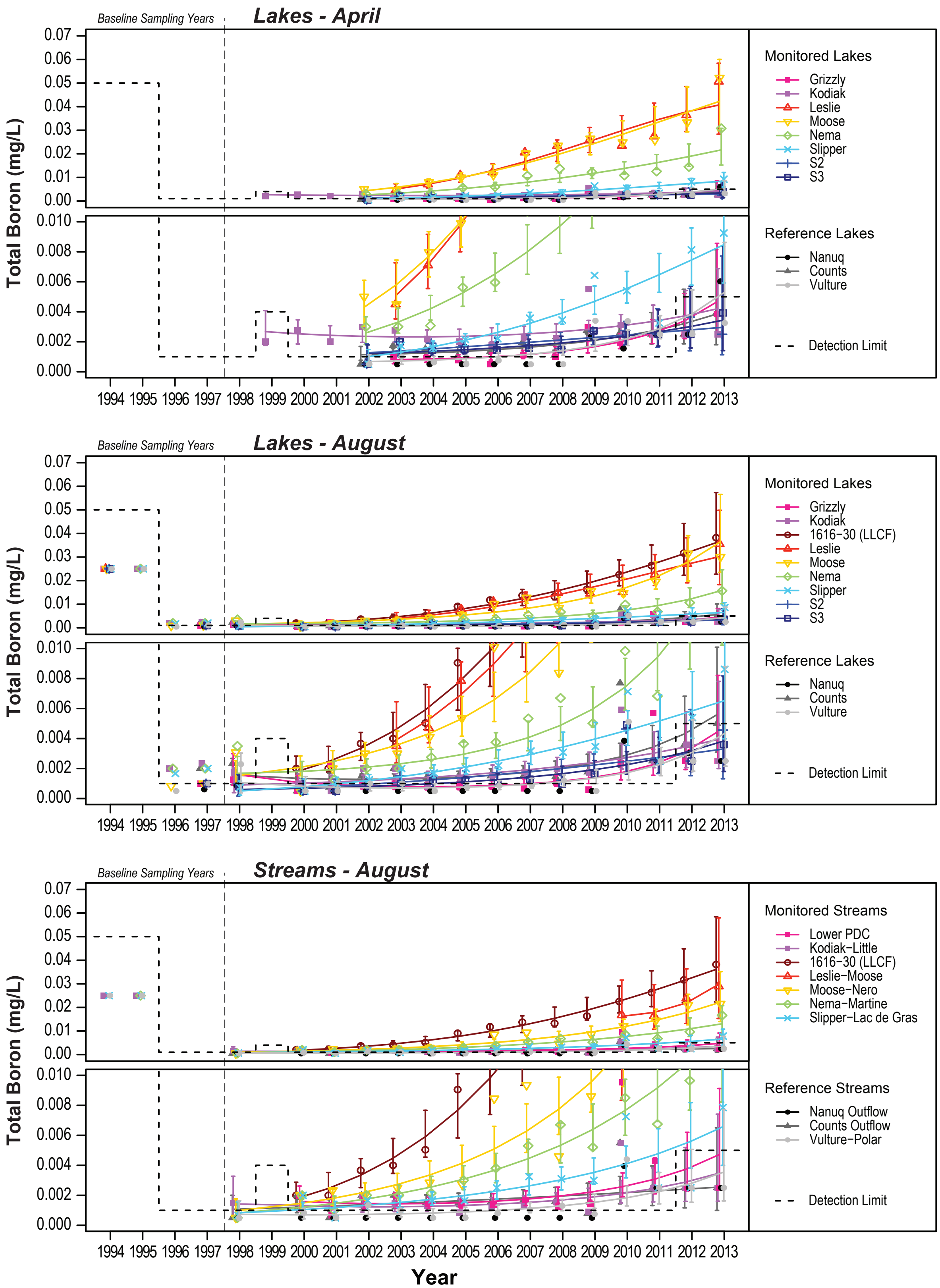
Table 3.2-20. Statistical Results of Total Boron Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	Nanuq	Tobit	2	-	Kodiak, Leslie, Moose, Nema, Slipper	-	1-242
Aug	Lake	Nanuq	Tobit	2	-	1616-30 (LLCF), Leslie, Moose, Slipper	-	1-248
Aug	Stream	Nanuq Outflow	Tobit	2	-	1616-30 (LLCF), Moose-Nero, Nema-Martine	-	1-254

Dashes indicate not applicable.

Figure 3.2-15

Observed and Fitted Means for Total Boron Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
CCME Guideline = 1.5 mg/L.

The 95% confidence intervals of the 2013 fitted mean and the observed mean total boron concentrations were less than the boron CCME guideline (1.5 mg/L; CCME 2009). Total boron concentrations were also less than the CCME guideline in all monitored streams in June, July, August and September 2013 (see Part 2 - Data Report; CCME 2009).

3.2.4.16 Total Cadmium

Summary: Concentrations of total cadmium have generally been below detection limits in all reference and monitored lakes and streams since monitoring began. All observations that were above the detection limit in 2013 were below the hardness-dependent cadmium CCME guideline. No mine effects were detected.

Concentrations of total cadmium were generally less than the detection limit in monitored and reference lakes and streams during both the ice-covered and open water season (Figure 3.2-16). Consequently, all lakes and streams were removed from the statistical analyses and statistical tests were not performed (Table 3.2-21). The detection limit for total cadmium was below the cadmium hardness-dependent CCME guideline for most observations in 2013. Observed cadmium concentrations that were greater than the analytical detection limit were less than the cadmium hardness-dependent CCME guideline (CCME 2014). Thus, it was concluded that no mine effects were detected for total cadmium in lakes and streams downstream of the LLCF and not downstream of the LLCF.

Table 3.2-21. Statistical Results of Total Cadmium Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	ALL	-	-	-	-	-	1-260
Aug	Lake	ALL	-	-	-	-	-	1-263
Aug	Stream	ALL	-	-	-	-	-	1-266

Dashes indicate not applicable.

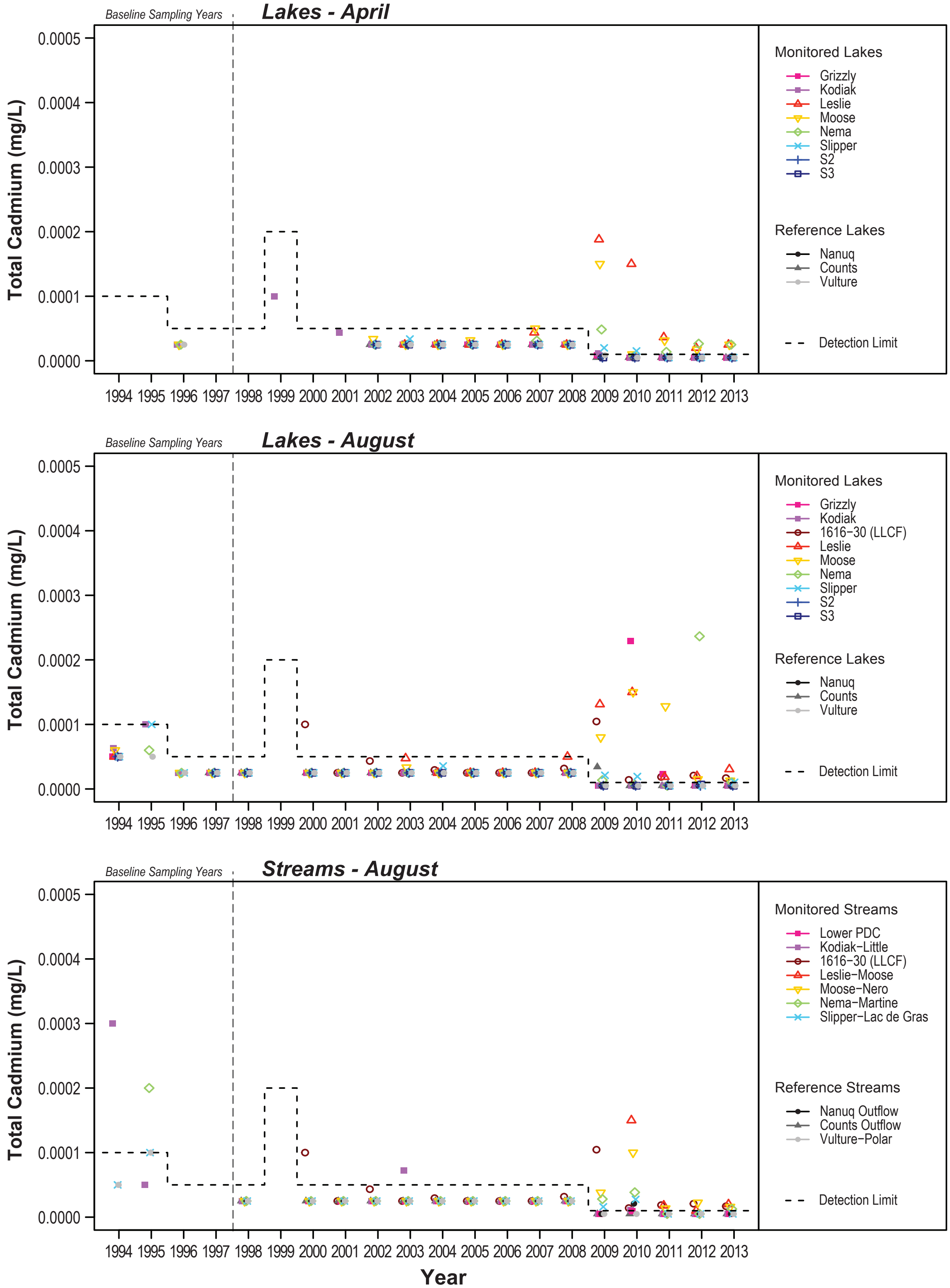
3.2.4.17 Total Molybdenum

Summary: Concentrations of molybdenum have increased in monitored lakes and streams downstream of the LLCF as far as site S3 in Lac de Gras. However, molybdenum concentrations have stabilised at levels greater than observed historical and reference lake concentrations at sites as far downstream as Nema Lake in recent years. In general, concentrations decrease with downstream distance from the LLCF. Observed and fitted mean concentrations were less than the molybdenum SSWQO at all sites in 2013. No mine effects were detected at sites that are not downstream of the LLCF.

Statistical and graphical analyses indicate that total molybdenum concentrations have increased through time in all monitored lakes and streams downstream of the LLCF as far as site S3 in Lac de Gras except Leslie-Moose Stream (Table 3.2-22; Figure 3.2-17). Only four years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends improbable; however, graphical analysis shows that total molybdenum concentrations in Leslie-Moose Stream were similar to those in the LLCF in all years during which Leslie-Moose Stream was monitored. In most cases, total molybdenum concentrations have stabilised in recent years, but there are some indications that concentrations may be increasing in Slipper Lake and at site S2 in Lac de Gras during the ice-covered season and in Slipper-Lac de Gras Stream during the open water season (Figure 3.2-17). In general, total molybdenum concentrations decrease with downstream distance from the LLCF (Figure 3.2-17). Together, graphical and statistical analyses suggest that mine operations have increased total molybdenum concentrations in all monitored lakes and streams downstream of the LLCF.

Figure 3.2-16

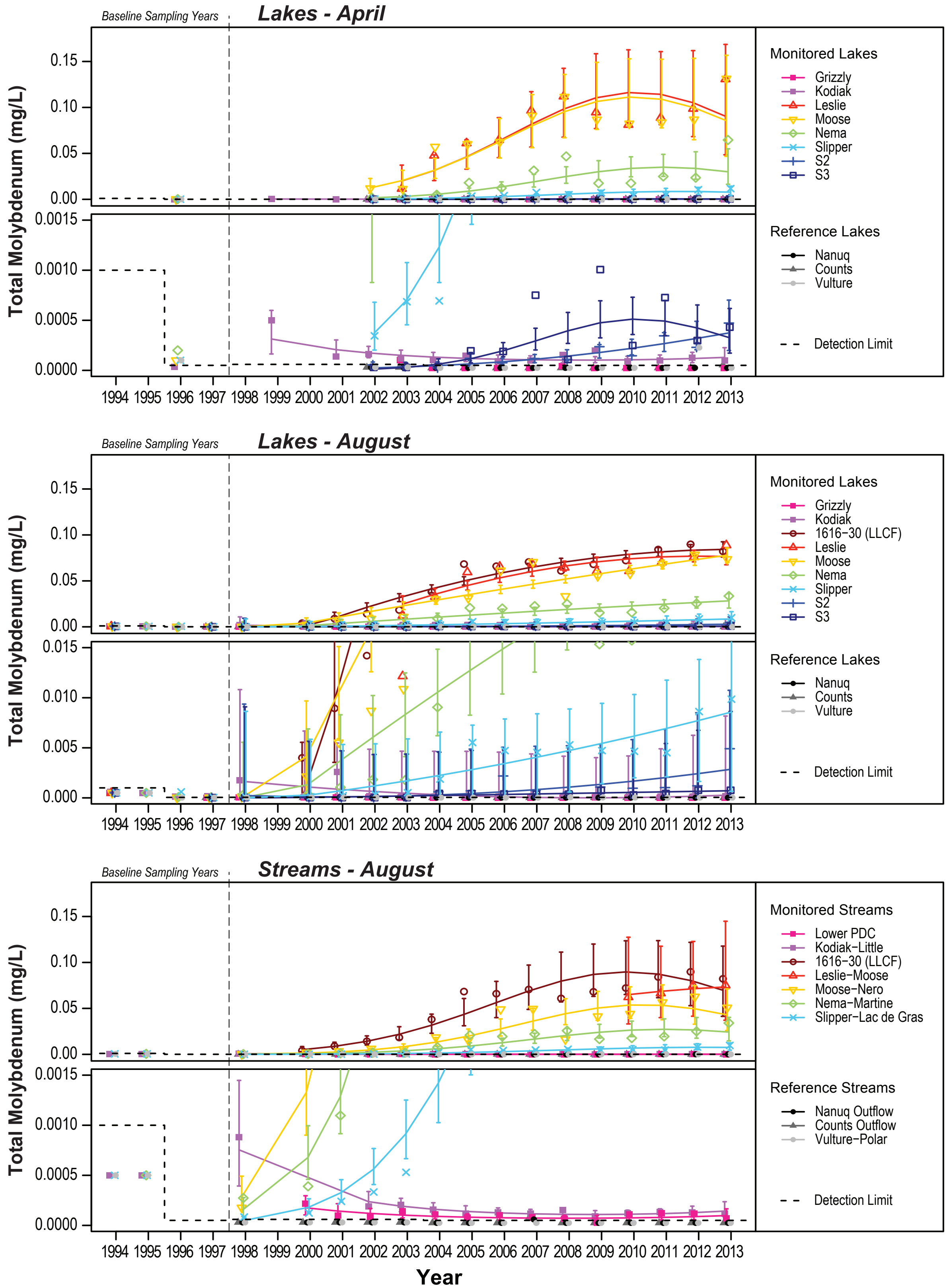
Observed Means for Total Cadmium Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
CCME Guideline = $10^{0.83 X (log10Hardness - 2.46)}$ /1000 mg/L, with minimum = 0.00004 mg/L where hardness = 0-16 mg/L and maximum of 0.00037 mg/L where hardness > 280 mg/L.

Figure 3.2-17

Observed and Fitted Means for Total Molybdenum Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
SSWQO = 19.38 mg/L.

Table 3.2-22. Statistical Results of Total Molybdenum Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	Grizzly, Nanuq, Counts, Vulture	Tobit	1a	-	-	Kodiak, Leslie, Moose, Nema, Slipper, S2, S3	1-269
Aug	Lake	Grizzly, Nanuq, Counts, Vulture	Tobit	1a	-	-	Kodiak, 1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2, S3	1-274
Aug	Stream	Nanuq Outflow, Counts Outflow, Vulture-Polar	LME	1a	-	-	Lower PDC, Kodiak-Little, 1616-30 (LLCF), Moose-Nero, Slipper-Lac de Gras	1-279

Dashes indicate no data were available.

At sites that are not downstream of the LLCF, statistical analyses indicate that total molybdenum concentrations have changed through time in the Lower PDC and Kodiak-Little (Table 3.2-22). However, graphical analysis suggests that concentrations have decreased slightly through time to stabilise at current levels shortly after monitoring began in both of these streams (Figure 3.2-17). Thus, no mine effects were detected at sites that are not downstream of the LLCF.

The 95% confidence intervals around the fitted mean and the observed mean total molybdenum concentrations were less than the molybdenum SSWQO (19.38 mg/L) in all monitored lakes and streams in 2013 during both the ice-covered and open water seasons (see Part 2 - Data Report; Rescan 2012a).

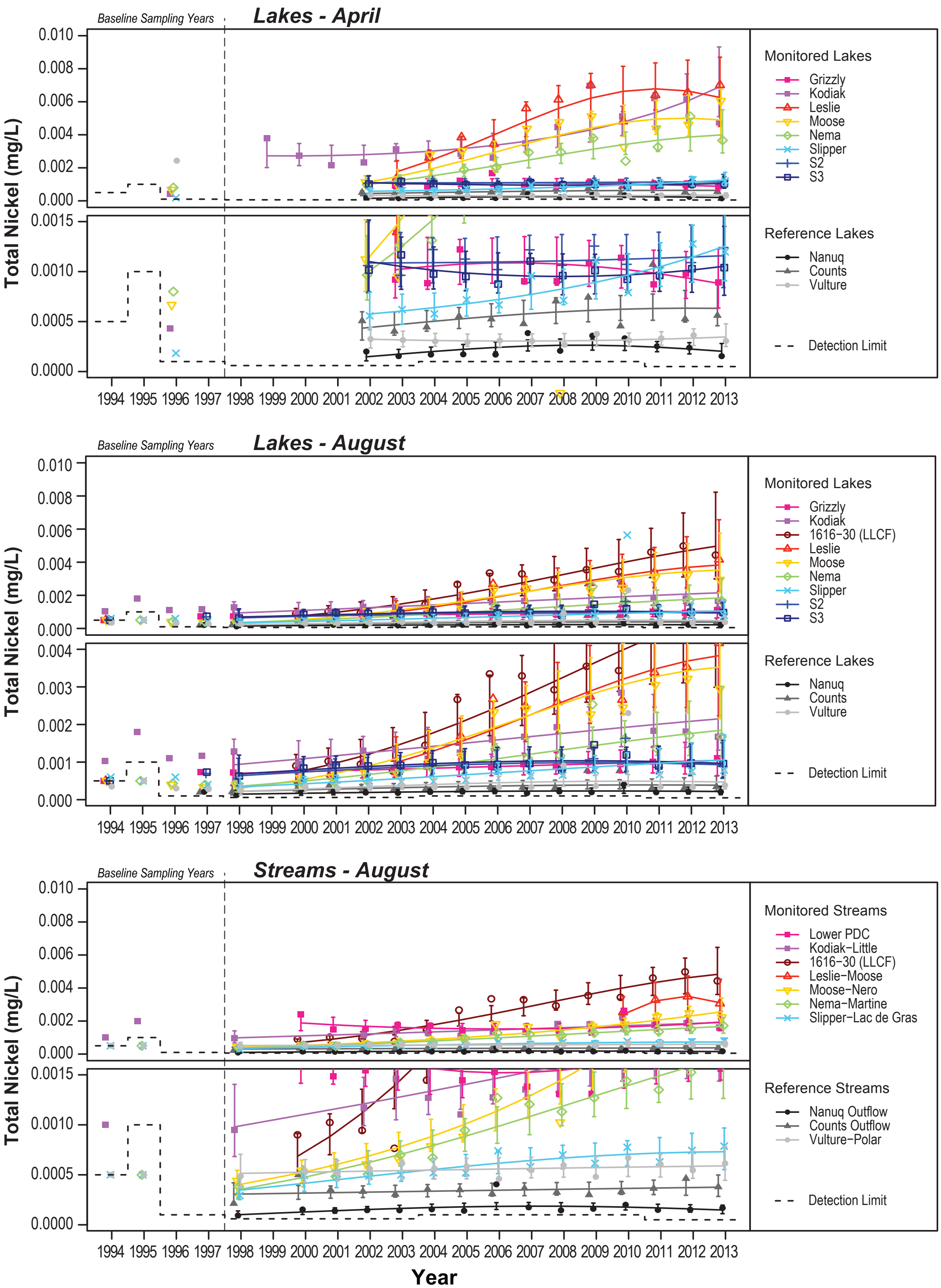
3.2.4.18 Total Nickel

Summary: Statistical and graphical analyses suggest that total nickel concentrations have increased in all lakes and streams downstream of the LLCF as far as Nema-Martine Stream as a result of mine operations. In general, total nickel concentrations decrease with downstream distance from the LLCF. Total nickel concentrations have also increased through time in Kodiak Lake and Kodiak-Little Stream, but the underlying cause of the change is unclear and not confirmed by statistical analysis in the case of Kodiak Lake. Observed and fitted mean concentrations were less than the hardness-dependent nickel CCREM guideline value at all sites in 2013.

Statistical analyses indicate that total nickel concentrations have changed through time in all monitored lakes and streams downstream of the LLCF as far as Slipper-Lac de Gras Stream except Leslie-Moose Stream and Slipper Lake (Table 3.2-23). Graphical analyses suggest that total nickel concentrations have increased through time in all lakes and streams downstream of the LLCF as far as Nema-Martine Stream, but that concentrations have stabilised in Leslie, Moose, and Nema lakes during the ice-covered season in recent years (Figure 3.2-18). Graphical analysis also suggests that total nickel concentrations decrease with downstream distance from the LLCF as far as Nema-Martine Stream (Figure 3.2-18). Together, graphical and statistical analysis suggests that total nickel concentrations have increased in all lakes and streams downstream of the LLCF as far as Nema-Martine Stream as a result of mine operations.

Figure 3.2-18

Observed and Fitted Means for Total Nickel Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
WL = Maximum average concentration premitted in water licence W2009L2-0001. WL = 0.15 mg/L.
CCME Guideline = $e^{0.76 X (lnHardness) + 1.06}$/1000 mg/L, where hardness = 60 - 180 mg/L, 0.025 mg/L where hardness < 60 mg/L, and 0.15 mg/L where hardness > 180 mg/L.

Table 3.2-23. Statistical Results of Total Nickel Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts: Model Fit = 3	Model Fit = 2	Model Fit = 1	Statistical Report Page No.
Apr	Lake	-	LME	2	-	- Leslie, Moose, Nema		1-284
Aug	Lake	-	LME	2	-	-1616-30 (LLCF), Leslie, Moose, Nema		1-290
Aug	Stream	-	LME	1b	-	-	Kodiak-Little, 1616-30 (LLCF), Moose-Nero, Nema-Martine, Slipper-Lac de Gras	1-296

Dashes indicate no data were available.

At sites that are not downstream of the LLCF, statistical analyses indicate that total nickel concentrations have changed through time in Kodiak-Little Stream (Table 3.2-23). Graphical analyses suggest that total nickel concentrations have increased through time in Kodiak-Little Stream and possibly in Kodiak Lake during the ice-covered season (Figure 3.2-18). The lack of statistical differences between the trend in Kodiak Lake and the reference lakes during the ice-covered season may result from a decrease in total nickel concentrations in Kodiak Lake in 2013 (Table 3.2-23; Figure 3.2-18). The source of the observed increase in Kodiak Lake and Kodiak-Little is unclear, but may be related to the construction or weathering of the PDC.

The 95% confidence interval around the fitted mean and the observed mean total nickel concentrations were less than the hardness-dependent nickel CCREM guideline value in all lakes and streams in 2013 (CCREM 1987). Total nickel concentrations were less than the hardness-dependent CCREM guideline for all monitored streams in June, July, August, and September 2013 (see Part 2 - Data Report; CCREM 1987).

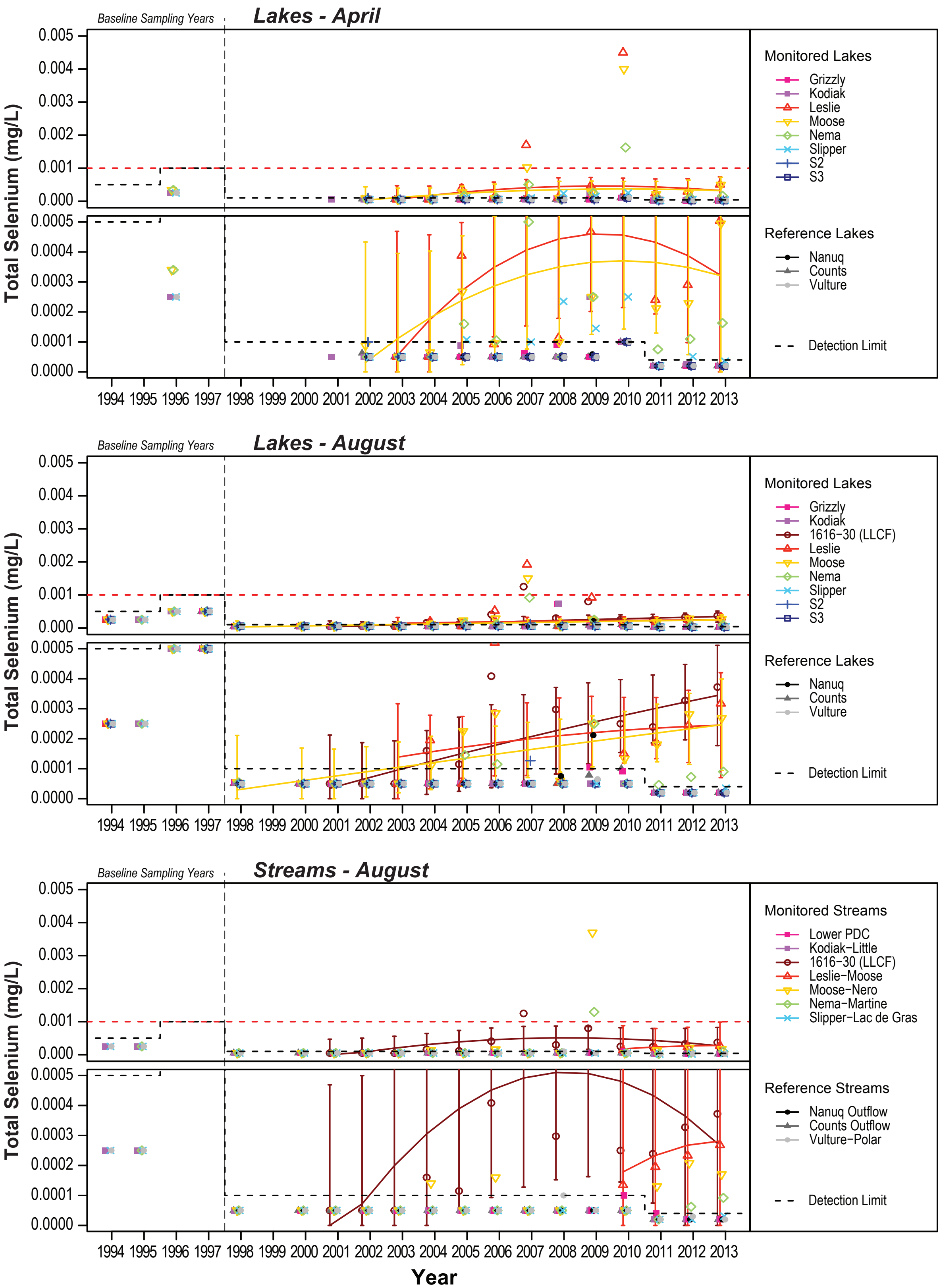
3.2.4.19 Total Selenium

> ***Summary:*** Concentrations of total selenium have generally been below detection limits in all reference and monitored lakes and streams since monitoring began. Observed and fitted mean concentrations were less than the selenium CCREM guideline at all sites in 2013. No mine effects were detected.

With the exception of a few values observed in 2006, 2009, 2010, 2011, 2012, and 2013, concentrations of total selenium were generally less than the detection limit in monitored and reference lakes and streams during both the ice-covered and open water season. Consequently, all lakes and streams were removed from the statistical analyses except Leslie and Moose lakes and Leslie-Moose Stream (Table 3.2-24). Although historically variable detection limits resulting from the matrix interference due to elevated chloride concentrations during ICPMS have made it somewhat difficult to discern clear patterns historically, a change to collision cell ICPMS in 2011 enabled target analytical detection limits of 0.00004 mg/L to be achieved for all samples from all lakes and streams in the Koala Watershed except the LLCF in 2012 (Rescan 2013c). Together, statistical and graphical evidence suggests that total selenium concentrations have been stable through time in all lakes and streams in the Koala Watershed and Lac de Gras (Table 3.2-24; Figure 3.2-19).

Figure 3.2-19

Observed and Fitted Means for Total Selenium Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
CCME Guideline = 0.001 mg/L.

Table 3.2-24. Statistical Results of Total Selenium Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	Grizzly, Kodiak, Nema, Slipper, S2, S3, Nanuq, Counts, Vulture	Tobit	1a	-	-	None	1-302
Aug	Lake	Grizzly, Kodiak, Nema, Slipper, S2, S3, Nanuq, Counts, Vulture	Tobit	1a	-	-	1616-30 (LLCF)	1-307
Aug	Stream	Lower PDC, Kodiak-Little, Moose-Nero, Nema-Martine, Slipper-Lac de Gras, Counts Outflow, Nanuq Outflow, Vulture-Polar	Tobit	1a	-	-	1616-30 (LLCF)	1-312

Dashes indicate not applicable.

Concentrations of total selenium were less than the selenium CCREM guideline (0.001 mg/L) in all lakes and streams during the ice-covered and open water seasons in 2013 (CCREM 1987). Thus, no mine effects were detected at any of the sites.

3.2.4.20 Total Strontium

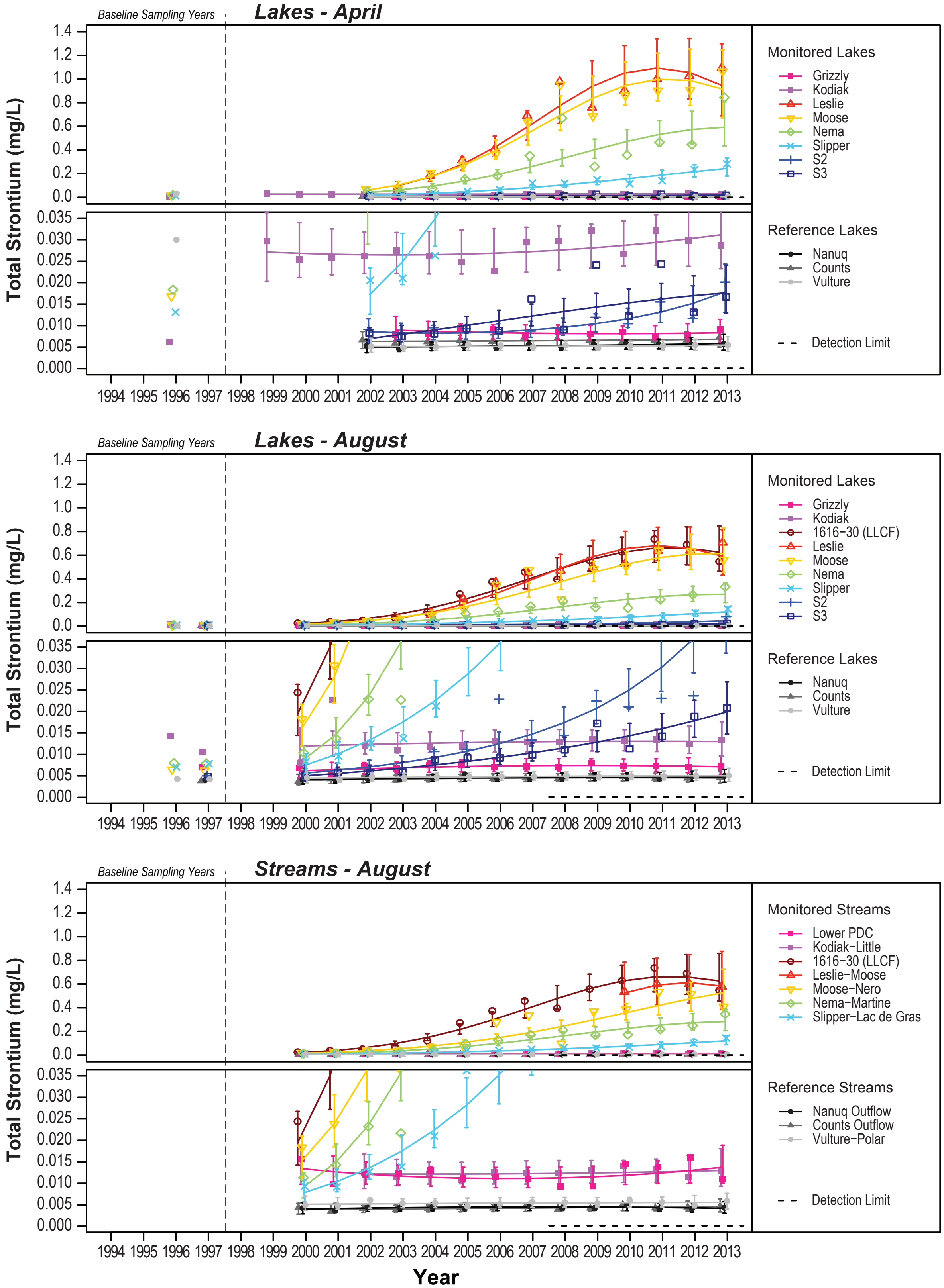
Summary: Statistical and graphical analyses suggest that total strontium concentrations have increased at all sites downstream from the LLCF as far as site S3 in Lac de Gras. No mine effects were detected at sites that are not downstream of the LLCF. Observed and fitted mean concentrations were less than the strontium water quality benchmark (6.242 mg/L) at all sites in 2013.

Statistical and graphical analyses indicate that concentrations of total strontium have increased through time, relative to reference sites, in all monitored lakes and streams downstream of the LLCF (Table 3.2-25; Figure 3.2-20). Graphical analysis also suggests that total strontium concentrations decrease with downstream distance from the LLCF (Figure 3.2-20). In contrast, statistical and graphical analyses suggest that total strontium concentrations have been stable through time at sites not downstream of the LLCF (Table 3.2-25; Figure 3.2-20). Thus, no mine effects were detected at sites that are not downstream of the LLCF.

The 95% confidence interval around the fitted mean and the observed mean total strontium concentrations were below the strontium water quality benchmark (6.242 mg/L) in all lakes and streams during the ice-covered and open water seasons in 2013 (see Part 2 - Data Report; Golder 2011).

Figure 3.2-20

Observed and Fitted Means for Total Strontium Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
Water quality benchmark (Golder 2011) = 6.242 mg/L.

Table 3.2-25. Statistical Results of Total Strontium Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	2	-	Leslie, Moose, Nema, Slipper, S2, S3	-	1-317
Aug	Lake	-	LME	3	Grizzly, Kodiak, 1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2, S3	1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2, S3	-	1-323
Aug	Stream	-	LME	1b	-	-	1616-30 (LLCF), Moose-Nero, Nema-Martine, Slipper-Lac de Gras	1-329

Dashes indicate not applicable.

3.2.4.21 Total Uranium

Summary: Statistical and graphical analyses suggest that total uranium concentrations have increased in all lakes and streams downstream from the LLCF as far as Nema-Martine Stream as a result of mine operations. No mine effects were detected at sites that are not downstream of the LLCF. Observed and fitted mean concentrations were less than the uranium CCME guideline at all sites in 2013.

Statistical and graphical analyses indicate that total uranium concentrations have increased through time, relative to reference sites, in all monitored lakes and streams downstream from the LLCF as far as Nema-Martine Stream, except at Leslie-Moose Stream (Table 3.2-26; Figure 3.2-21). Only four years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends improbable; however, graphical analysis shows that total uranium concentrations in Leslie-Moose Stream have been increasing and were similar to those in the LLCF in recent years. Graphical analysis also indicates that total uranium concentrations decrease with downstream distance from the LLCF, suggesting that effects are a consequence of mine activities (Figure 3.2-21).

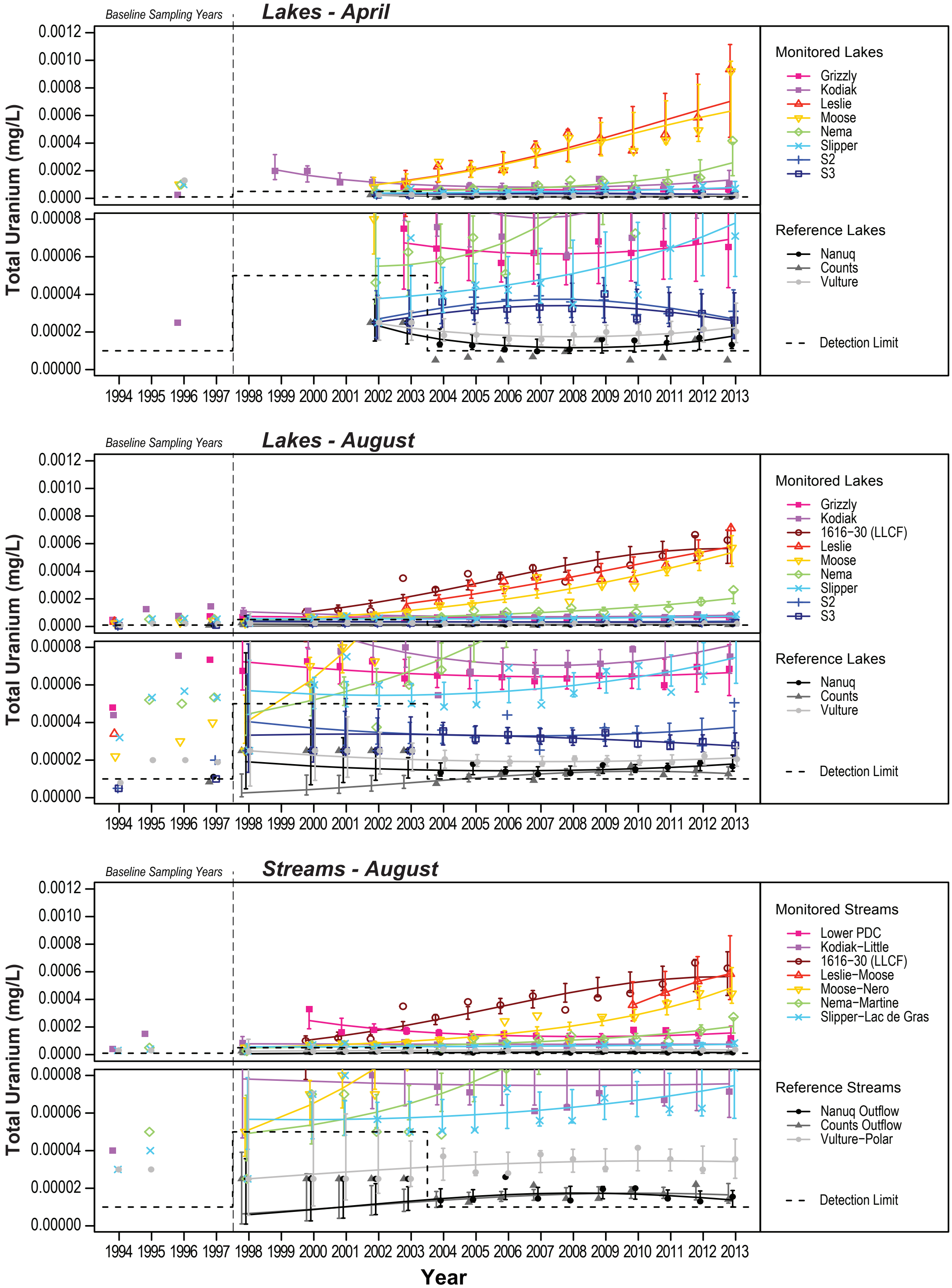
Table 3.2-26. Statistical Results of Total Uranium Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	Counts	Tobit	2	-	Leslie, Moose, Nema	-	1-335
Aug	Lake	-	Tobit	2	-	1616-30 (LLCF), Leslie, Moose, Nema	-	1-341
Aug	Stream	-	Tobit	2	-	Lower PDC, 1616-30 (LLCF), Moose-Nero, Nema-Martine	-	1-347

Dashes indicate not applicable.

Figure 3.2-21

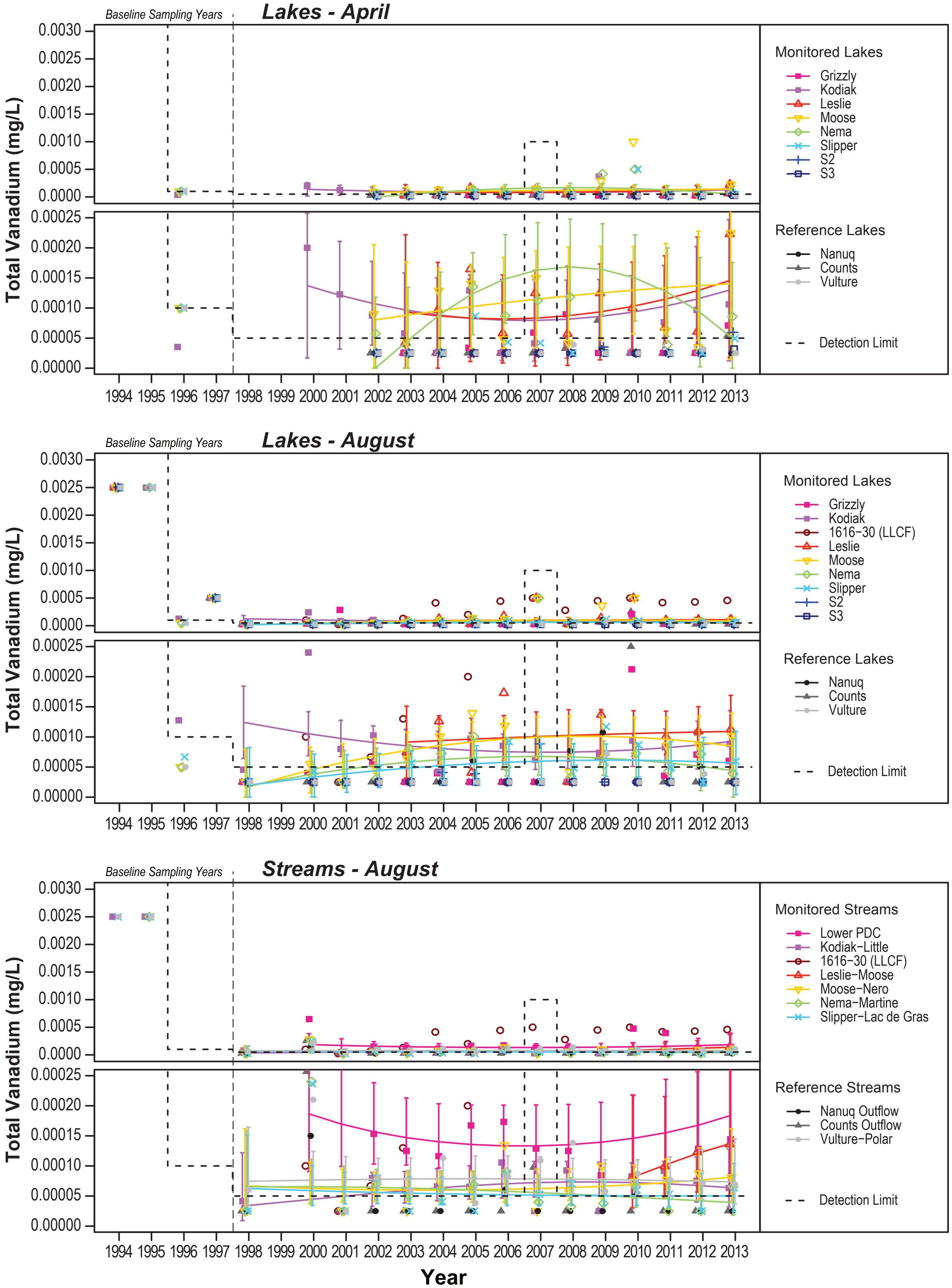
Observed and Fitted Means for Total Uranium Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
CCME Guideline = 0.015 mg/L.

Figure 3.2-22

Observed and Fitted Means for Total Vanadium Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
SSWQO = 0.03 mg/L.

At sites not downstream of the LLCF, statistical analyses indicate that concentrations of total uranium have changed through time, relative to reference streams, in the Lower PDC (Table 3.2-26). However, graphical analysis suggests that total uranium concentrations have declined from initially high concentrations in the Lower PDC since shortly after monitoring began (Figure 3.2-21). Thus, no mine effects were detected at sites that are not downstream of the LLCF.

Observed and fitted mean total uranium concentrations were less than the uranium CCME guideline value (0.015 mg/L) in all reference and monitored lakes and streams during both the ice-covered and open water seasons in 2013 (see Part 2 - Data Report; CCME 2011).

3.2.4.22 Total Vanadium

Summary: Statistical and graphical analyses suggest that total vanadium concentrations have remained low and stable in all monitored sites of the Koala Watershed and Lac de Gras. Observed and fitted mean concentrations were less than the vanadium SSWQO at all sites in 2013. No mine effects were detected.

Statistical and graphical analyses indicate that total vanadium concentrations have been low and stable through time in all lakes and streams in the Koala watershed and Lac de Gras (Table 3.2-27; Figure 3.2-22). Thus, no mine effects were detected in any monitored lakes or streams of the Koala Watershed and Lac de Gras. Observed and fitted mean vanadium concentrations were less than the vanadium SSWQO (0.03 mg/L) in all lakes and streams during the ice-covered and open water seasons in 2013 (see Part 2 - Data Report; Rescan 2012h).

Table 3.2-27. Statistical Results of Total Vanadium Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	Grizzly, Slipper, S2, S3, Counts, Nanuq, Vulture	Tobit	1a	-	-	None	1-353
Aug	Lake	Grizzly, 1616-30 (LLCF), S2, S3, Counts, Nanuq, Vulture	Tobit	1a	-	-	None	1-358
Aug	Stream	1616-30 (LLCF), Counts Outflow, Nanuq Outflow	Tobit	1b	-	-	None	1-363

Dashes indicate not applicable.

3.3 AQUATIC BIOLOGY

The extent to which changes in water quality might result in changes in biological communities is a function of both the relative competitive abilities of different species under different environmental conditions (i.e., their ability to acquire resources, relative to the other species present) and each species' ability to physically tolerate changes in the concentrations of elements and molecules (toxicity). Additional changes in biological communities may result from changes in the taxonomic composition or the nutritional quality of organisms on which higher trophic levels feed.

Results from water quality analyses in the Koala Watershed and Lac de Gras suggest that changes might be expected in biological communities downstream of the LLCF as far as site S3 in Lac de Gras, because concentrations of 18 evaluated water quality variables have increased downstream of the LLCF as a result of mine activities (see Section 3.4). In general, the 95% confidence intervals around the fitted mean and the observed mean concentrations for these 18 water quality variables were below their respective CCME guideline value, SSWQO, or relevant benchmark value (see Section 3.4). Exceptions included pH, total-phosphate-P, and potassium. For pH and total phosphate-P, levels and concentrations in reference lakes or streams also exceeded the applicable CCME guideline value, suggesting that exceedences are not related to mine activities. In contrast, potassium exceedences were unique to the two most upstream monitored lakes (i.e., Leslie and Moose lakes) and are thus likely related to mine activities.

Concentrations of water quality variables that have increased in monitored lakes at the Ekati Diamond Mine for which SSWQO or species sensitivity-based CCME guidelines exist were reviewed as part of the 2012 AEMP Re-evaluation with a specific focus on identifying possible chronic toxic effects on species present in the receiving environment at the Ekati Diamond Mine (Rescan 2012d). As in previous years, concentrations of all the water quality variables in the Koala Watershed and Lac de Gras in 2013 remained below the lowest identified chronic effect level for the most sensitive species; the only exception in 2013 was potassium (Rescan 2012g). In Leslie and Moose lakes, the observed mean potassium concentrations exceeded the potassium SSWQO (41 mg/L; Rescan 2012g) during the ice-covered season. In Leslie Lake, the upper 95% confidence interval of the fitted mean during the ice-covered season also exceeded the lowest identified potassium chronic effect level of 53 mg/L for the most sensitive species (i.e., *Daphnia magna*) (see Section 3.2.4.6; Biesinger and Christensen 1972). Potassium plays an important role in nerve function and is therefore required by many aquatic species (Environment Canada 2002). However, potassium can become toxic at elevated concentrations, and is substantially more toxic than other major ions of earth metals (i.e., magnesium, calcium, and sodium). However, potassium toxicity may decrease as the total ion concentration increases as a consequence of strong interactions with other metals (Trotter 2001).

Concentrations of nutrients are among the water quality variables that have changed through time in the Koala Watershed and changes in nutrients can have an effect on the composition of biological communities that are not related to toxic effects. Accumulating research suggests that the ratio of available elements, especially macronutrients like carbon (C), nitrogen (N), and phosphorous (P), can play an important role in determining community composition and relative abundance by providing a competitive advantage to taxa whose relative elemental requirements best match current conditions (Sterner et al. 1997; Dobberfuhl and Elser 2000; Elser et al. 2000). For example, relatively low nitrogen environments favour phytoplankton species that are capable of fixing nitrogen (i.e., blue-green algae) while those that can take up nitrogen directly from the environment thrive when the relative availability of nitrogen increases (i.e., diatoms; Tillman et al. 1986).

The ratio of available nutrients in the Koala Watershed has shifted through time as nitrogen levels have increased. This coincides with the overall results of the 2012 AEMP Re-evaluation, which suggested that observed changes in biological community composition at the Ekati Diamond Mine likely resulted from inter-specific differences in the competitive ability of different taxonomic groups under changing quantities or ratios of macronutrients like nitrogen or phosphorus, rather than elemental toxicity (Rescan 2012d). As the trends in the evaluated water quality variables in 2013 are consistent with those observed in the 2011 and 2012 AEMP (Rescan 2012b, 2013b) it is expected that the relative availability of macronutrients will continue to be the dominant driver of change in biological communities; however, increasing potassium concentrations may also play a role in explaining changes to species composition observed in 2013. Increasing potassium concentrations may be particularly important for changes in zooplankton composition as the most sensitive species identified in the development of the SSWQO was the cladoceran *Daphnia magna* (Biesinger and Christensen 1972; Rescan 2012g).

3.3.1 Phytoplankton

3.3.1.1 Variables

Phytoplankton are the main source of primary productivity in lake systems. Phytoplankton are also useful indicators of change because they have rapid turn-over times (from hours to days) and are sensitive to physical, chemical, and biological stressors. Previous research has indicated that phytoplankton are some of the most susceptible organisms to changes in water quality variables in lakes (SENES Consultants 2008). Thus, chlorophyll *a* concentrations, phytoplankton density (cells/mL), and phytoplankton diversity (Shannon and Simpson's diversity indices) were evaluated to determine if mine activities have affected phytoplankton communities.

3.3.1.2 Dataset

Phytoplankton data have been collected between late July and early August of each year for the evaluation of effects (Table 3.3-1). Baseline data, which was collected from 1994 to 1997, are included in graphical analysis but not in the statistical evaluation of effects.

Table 3.3-1. Dataset Used for Evaluation of Effects on the Phytoplankton in Koala Watershed Lakes and Lac de Gras

Year	Nanuq	Counts	Vulture	Kodiak	Leslie	Moose	Nema	Slipper	S2	S3
1993*	-	-	-	Aug-15	-	-	-	-	-	-
1994*	-	-	Aug-13	Aug-17	-	-	-	Aug-15	-	-
1995	-	-	-	-	-	-	-	-	-	-
1996*	-	-	Jul-28	Jul-28 (no biomass)	-	Jul-27	Jul-26	Jul-26	-	-
1997*	Aug-4	Aug-14	Aug-5	Aug-9	-	Aug-10	Aug-11	Aug-11	Aug-12	Aug-12
1998	Aug-3	Aug-3	Aug-6	Aug-10	-	Aug-7	Aug-6	Aug-5	Aug-4	Aug-4
1999	Aug-7	Aug-8	Aug-6	Aug-10	-	Aug-7	Aug-10	Aug-9	Aug-11	Aug-11
2000	Aug-4	Aug-1	Aug-4	Jul-29	-	Jul-30	Jul-30	Jul-31	Aug-3	Aug-3
2001	Aug-1	Jul-30	Aug-2	Jul-28	-	Aug-3	Aug-3	Jul-29	Jul-29	Jul-29
2002	Aug-1	Aug-7	Aug-3	Aug-2	-	Aug-5	Aug-4	Aug-6	Aug-4	Aug-4
2003	Aug-9	Aug-7	Aug-4	Aug-8	Aug-3	Aug-9	Aug-3	Aug-7	Aug-5	Aug-5
2004	Aug-10	Aug-13	Aug-9	Aug-7	Aug-9	Aug-10	Aug-9	Aug-12	Aug-9	Aug-9
2005	Aug-1	Aug-7	Jul-31	Aug-3	Aug-4	Aug-9	Aug-9	Aug-5	Aug-5	Aug-5
2006	Aug-2	Aug-4	Aug-2	Aug-1	Aug-6	Aug-5	Aug-5	Aug-4	Aug-4	Aug-4
2007	Aug-11	Aug-6	Aug-12	Aug-4	Aug-13	Aug-7	Aug-11	Aug-10	Aug-8	Aug-6
2008	Aug-8	Jul-31	Jul-29	Jul-27	Jul-31	Jul-29	Jul-31	Jul-29	Aug-7	Aug-7
2009	Jul-30	Aug-1	Jul-30	Aug-8	Aug-5	Jul-30	Jul-30	Aug-3	Jul-31	Jul-31
2010	Aug-6	Aug-7	Aug-5	Aug-5	Aug-3	Aug-3	Aug-5	Aug-5	Aug-6	Aug-6
2011	Aug-2	Aug-5	Aug-5	Aug-5	Aug-2	Aug-3	Aug-5	Aug-3	Aug-4	Aug-4
2012	Aug-1	Aug-8	Aug-7	Aug-6	Aug-8	Aug-9	Aug-7	Aug-8	Aug-3	Aug-2
2013	Aug-3	Aug-1	Aug-1	Aug-6	Aug-1	Aug-5	Aug-6	Aug-5	Aug-2	Aug-2

** = Data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison.*
Dashes indicate no data were available.
Single samples were collected yearly for biomass analysis from 1993 to 1996, triplicate samples were collected from 1997 to 2013.
Triplicate samples were collected annually from 1996 to 2013 for taxonomic analysis.

Prior to 1996, single chlorophyll *a* samples were collected for analysis. Triplicate sampling for chlorophyll *a* and taxonomic composition began in 1996 and has continued to present day. Only taxonomic analyses were conducted for Kodiak Lake phytoplankton data in 1996.

3.3.1.3 Results and Discussion

Chlorophyll *a*

Statistical and graphical analyses indicate chlorophyll *a* concentrations have been stable through time in all monitored lakes (Table 3.3-2; Figure 3.3-1). Compared to mean baseline concentrations ± 2 SD, mean chlorophyll *a* concentrations in 2013 were greater in Moose, Nema, and Slipper lakes and sites S2 and S3 in Lac de Gras (Table 3.3-3). However, a similar pattern was observed in at least one reference lake (i.e., Vulture Lake; Table 3.3-3). Overall, chlorophyll *a* concentrations were greater in 2013 than in 2012 in all monitored and reference lakes, but within the range of historical concentrations. Thus, no mine effects were detected with respect to chlorophyll *a* concentrations.

Table 3.3-2. Statistical Results of Chlorophyll *a* Concentrations in Lakes in the Koala Watershed and Lac de Gras

	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
				Model Fit = 3	Model Fit = 2	Model Fit = 1	
Chlorophyll *a*	-	LME	3	Kodiak, Nema	None	-	1-368

Dashes indicate not applicable.

Table 3.3-3. Mean ± 2 Standard Deviations (SD) Baseline Concentrations of Chlorophyll *a* in each of the Koala Watershed Lakes and Lac de Gras

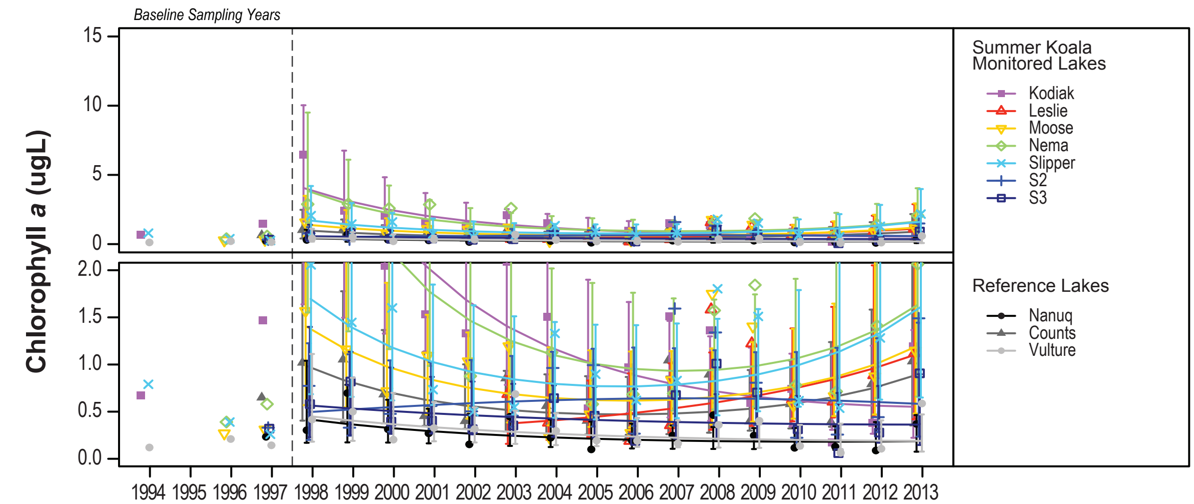
Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD
Nanuq	0.23 (1)	0- 0.51	0.37 ± 0.48
Counts	0.65 (1)	0 - 1.45	1.03 ± 0.82
Vulture	0.15 (2)	0.08- 0.23	0.59 ± 0.09
Kodiak	1.24 (3)	0.46- 2.01	1.19 ± 0.42
Leslie	-	-	1.12 ± 0.46
Moose	0.30 (2)	0 - 0.74	1.15 ± 0.47
Nema	0.53 (2)	0.21 - 0.85	2.06 ± 0.60
Slipper	0.39 (3)	0 - 0.88	2.17 ± 0.79
S2	0.33 (1)	0.13 - 0.53	1.49 ± 1.00
S3	0.32 (1)	0.14 - 0.50	0.91 ± 0.67

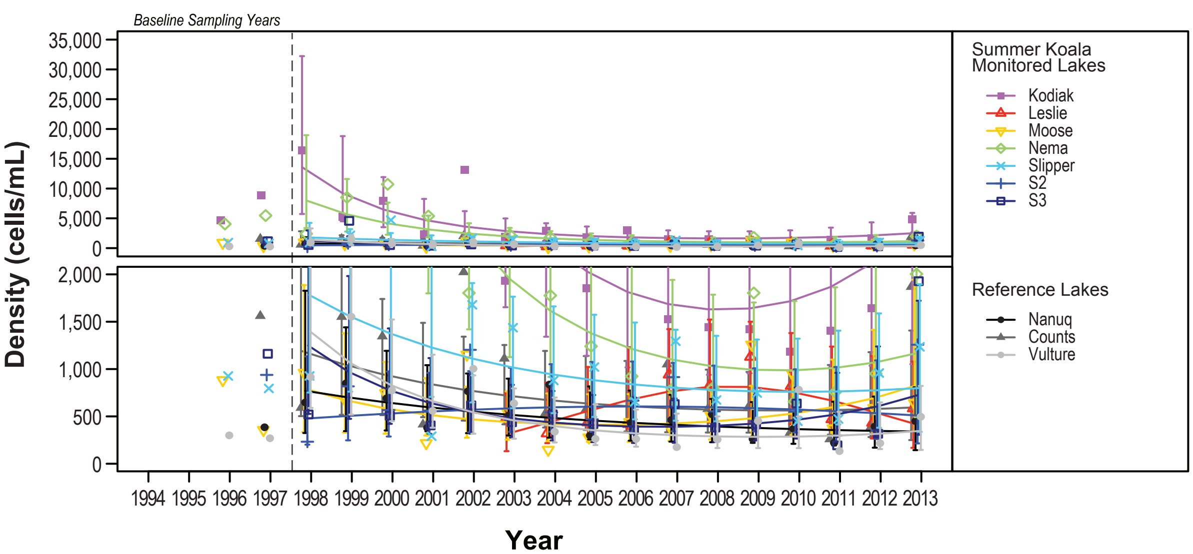
Units are µg/L.
Negative values were replaced with zeros.
N = number of years data were collected.
Dashes indicates no data available.

Density

Statistical analyses indicate that phytoplankton densities have been stable through time, relative to trends observed in reference lakes, in all monitored lakes except in Leslie Lake (Table 3.3-4). However, graphical analysis suggests that phytoplankton densities have been relatively stable through time in Leslie Lake (Figure 3.3-1). In all monitored lakes, phytoplankton densities in 2013 remained within ± 2 SD of the mean observed phytoplankton densities in baseline years (Table 3.3-5). Thus, no mine effects were detected with respect to phytoplankton density.

Observed and Fitted Means for Chlorophyll *a* Concentrations and Phytoplankton Density in Koala Watershed Lakes and Lac de Gras, August 1994 to 2013





Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.

Table 3.3-4. Statistical Results of Phytoplankton Density in Lakes in the Koala Watershed and Lac de Gras

	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
				Model Fit = 3	Model Fit = 2	Model Fit = 1	
Phytoplankton density	-	LME	3	Kodiak, Leslie, Nema, S2	Leslie	-	1-374

Dashes indicate not applicable.

Table 3.3-5. Mean ± 2 Standard Deviations (SD) Baseline Phytoplankton Density in each of the Koala Watershed Lakes and Lac de Gras

Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD
Nanuq	385 (1)	56 - 714	450 ± 118
Counts	1,561 (1)	103 - 3,020	1,868 ± 724
Vulture	284 (2)	76 - 492	496 ± 26
Kodiak	6,778 (2)	1,882 - 11,674	4,845 ± 771
Leslie	-	-	586 ± 226
Moose	620 (2)	0 - 1,551	642 ± 97
Nema	4,757 (2)	329 - 9,185	2,004 ± 805
Slipper	861 (2)	98 - 1,624	1,235 ± 1,120
S2	938 (1)	300 - 1,576	1,257 ± 91
S3	1,161 (1)	0 - 2,385	1,928 ± 1,253

Units are cells/L.
Negative values were replaced with zeros.
N = number of years data were collected.
Dashes indicate no data available.

Diversity

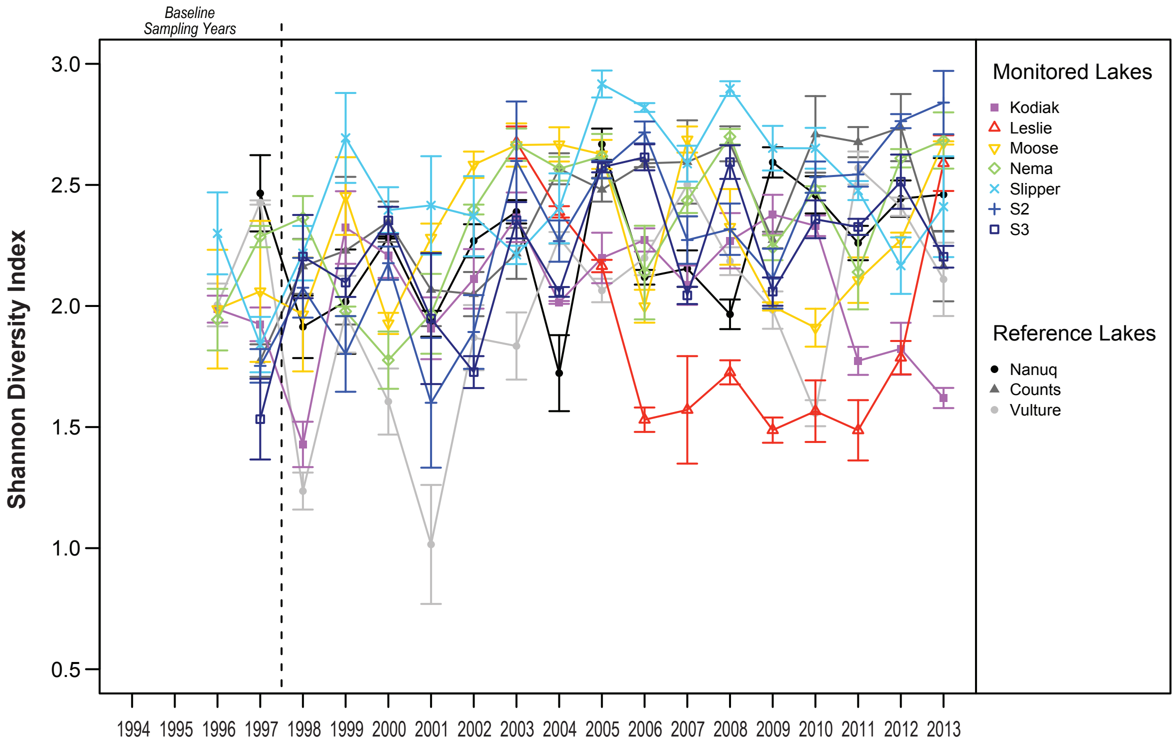
Statistical analyses were not performed on the diversity datasets because the calculation of indices can result in data abnormalities that prevent statistical analysis. Consequently, graphical analyses of temporal trends in diversity indices (Figure 3.3-2) and best professional judgment were the primary methods used in the evaluation of effects. In addition, the average and relative densities of taxa were examined using graphical analyses to identify potential changes in community composition (Figures 3.3-3 to 3.3-8). Note that following recent advances in taxonomic classification, the names of two phytoplankton groups have been updated in 2013 (when comparing to historical AEMP observations): the Cyanophyta are now recognized as the class Myxophyceae and the Pyrrophyta are now recognized as the class Dinophyceae.

Both Shannon and Simpson's diversity indices have varied considerably through time in both monitored and reference lakes (Figure 3.3-2). While the variability makes it somewhat difficult to discern temporal trends, diversity in Leslie Lake decreased between 2006 and 2011, but has returned to historical levels in 2013 (Figure 3.3-2).

Comparisons between mean diversity ± 2 SD in baseline years and mean diversity in 2013 indicate differences between baseline and 2013 values for both Shannon and Simpson's diversity at site S2 in Lac de Gras and in Shannon diversity in Nema Lake and site S3 in Lac de Gras (Table 3.3-6). However, patterns were similar for Shannon diversity in one reference lake (i.e., Counts Lake; Table 3.3-6) and there was no trend in diversity with downstream distance from the mine. In all other lakes, diversity indices remained within two standard deviations of baseline values in 2013 (Table 3.3-6).

Figure 3.3-2

Average Diversity Indices for Phytoplankton in Koala Watershed Lakes and Lac de Gras, August 1996 to 2013



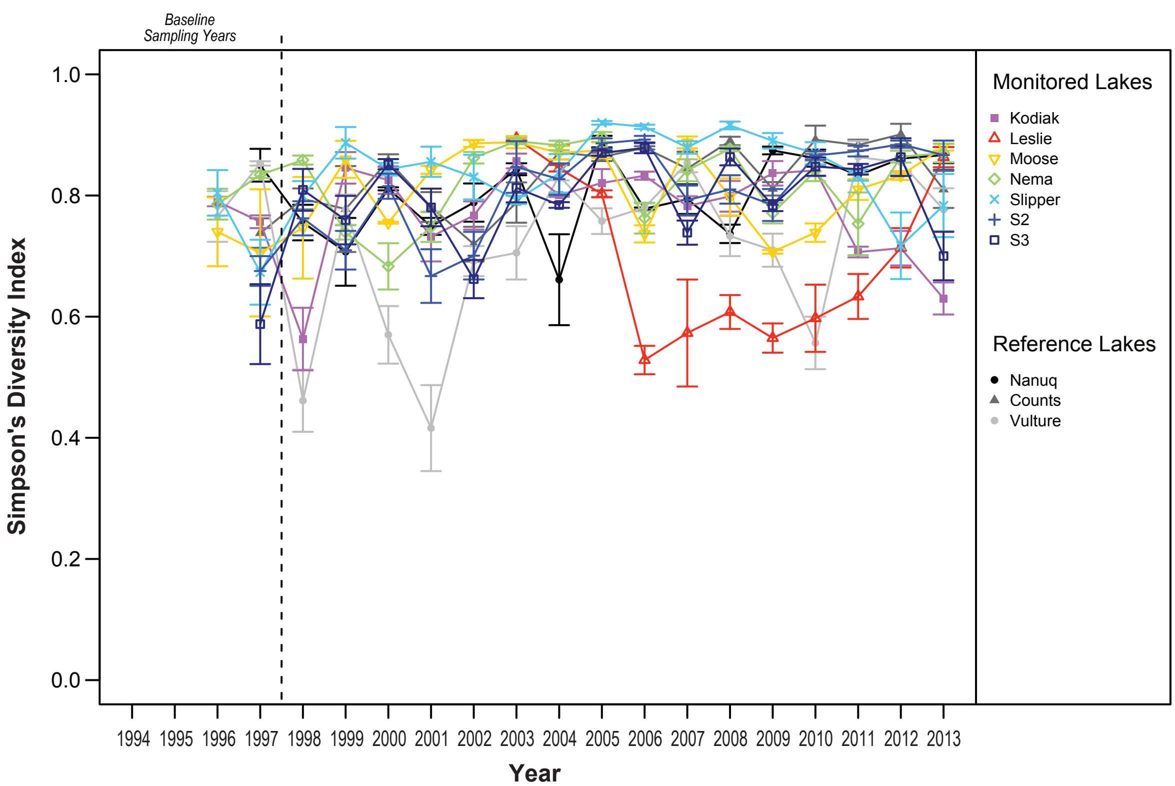


Figure 3.3-3

Average Phytoplankton Density by Taxonomic Group for AEMP Reference Lakes, 1996 to 2013

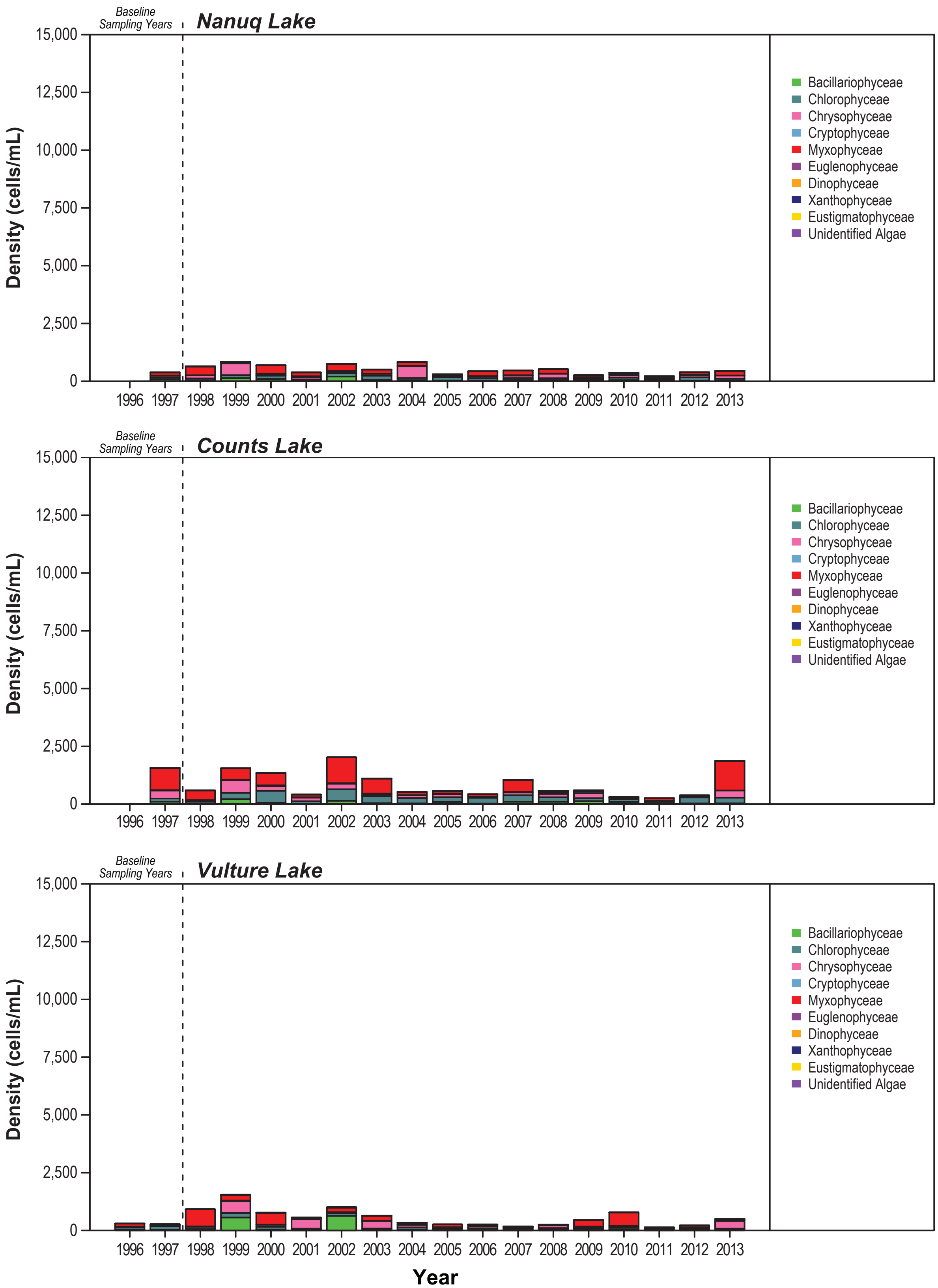
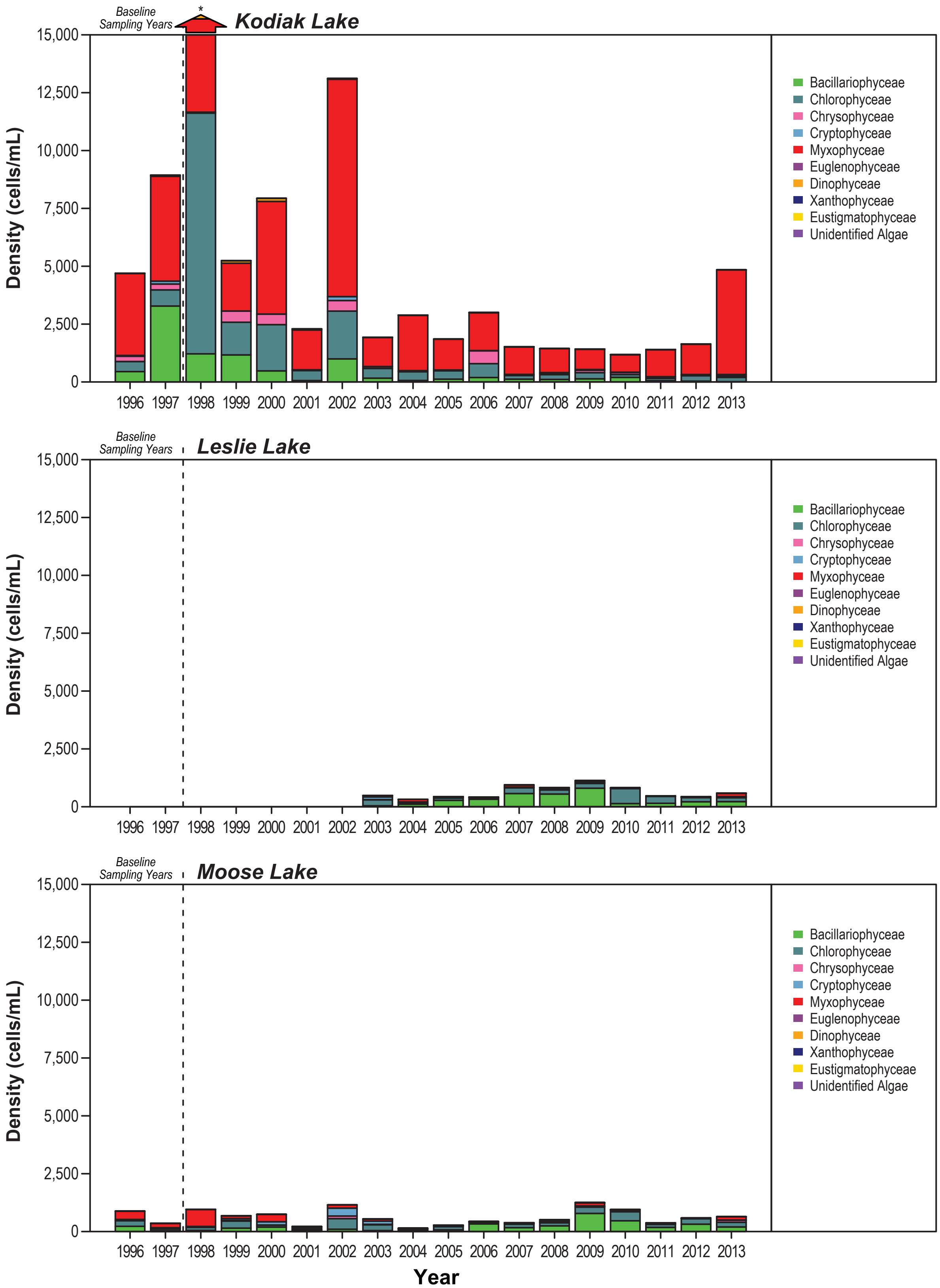


Figure 3.3-4a

Average Phytoplankton Density by Taxonomic Group for Lakes of the Koala Watershed, 1996 to 2013



*Note: * Total density = 16, 434; Myxophyceae = 4, 551; Dinophyceae = 225.*

Figure 3.3-4b

Average Phytoplankton Density by Taxonomic Group for Lakes of the Koala Watershed, 1996 to 2013

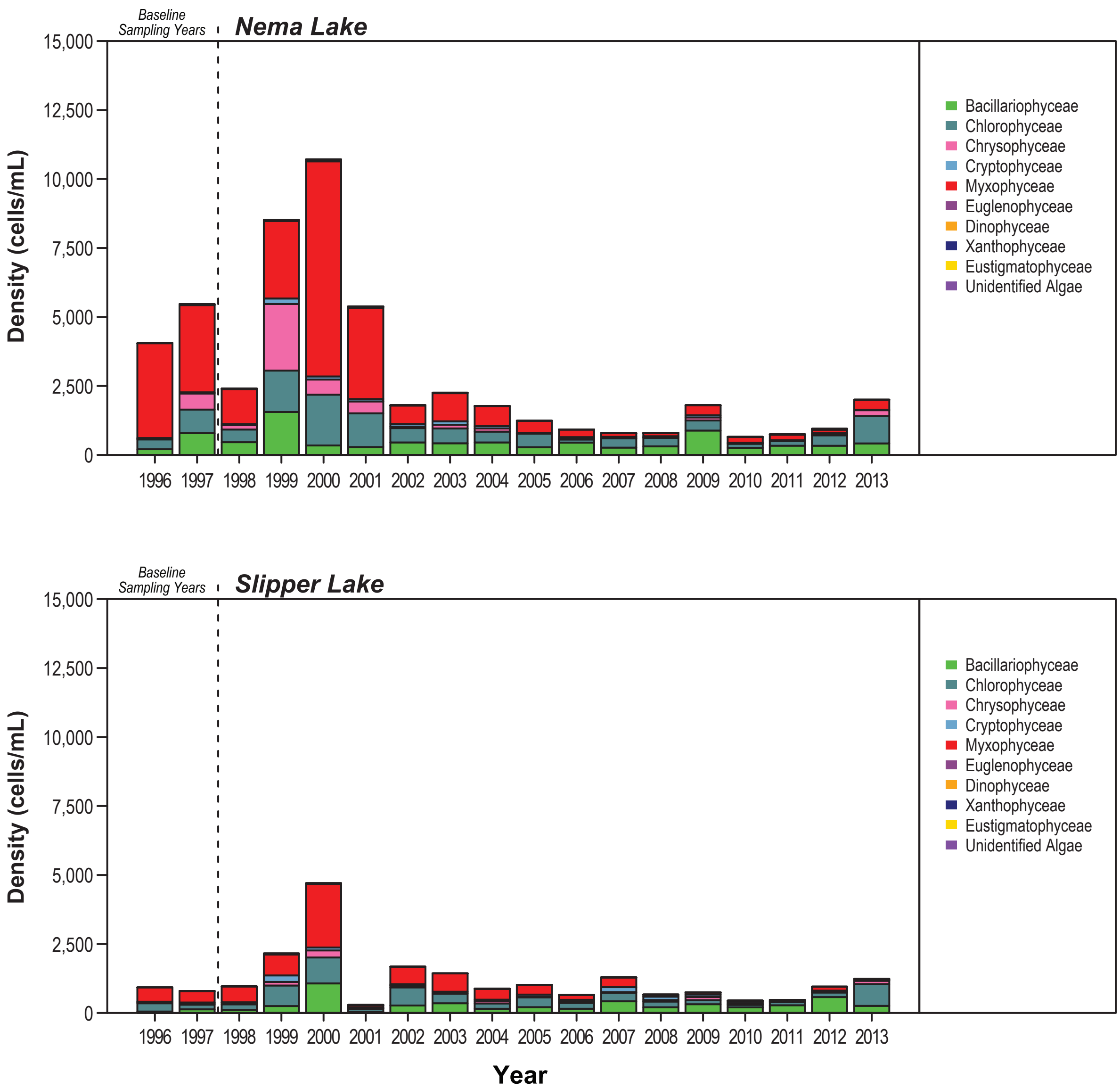
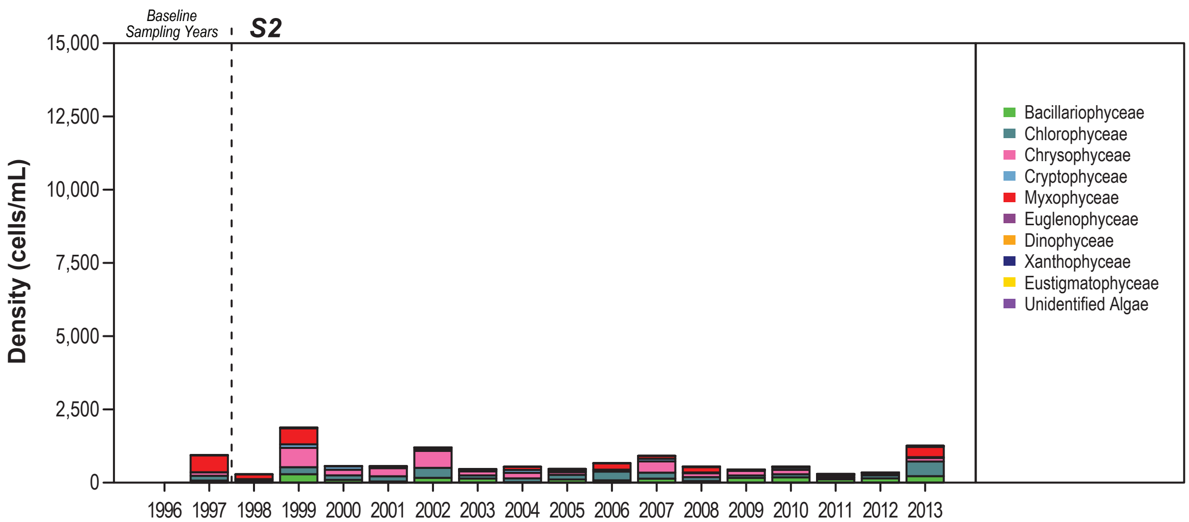


Figure 3.3-5

Average Phytoplankton Density by Taxonomic Group for Lac de Gras, 1996 to 2013



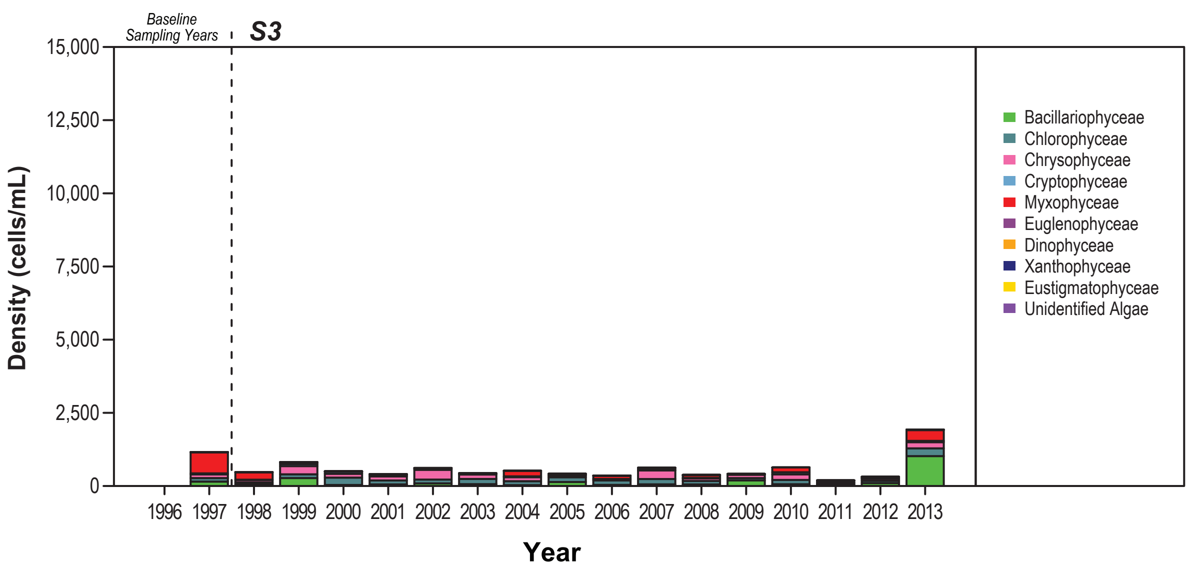


Figure 3.3-6

Relative Densities of Phytoplankton Taxa in AEMP Reference Lakes, 1996 to 2013

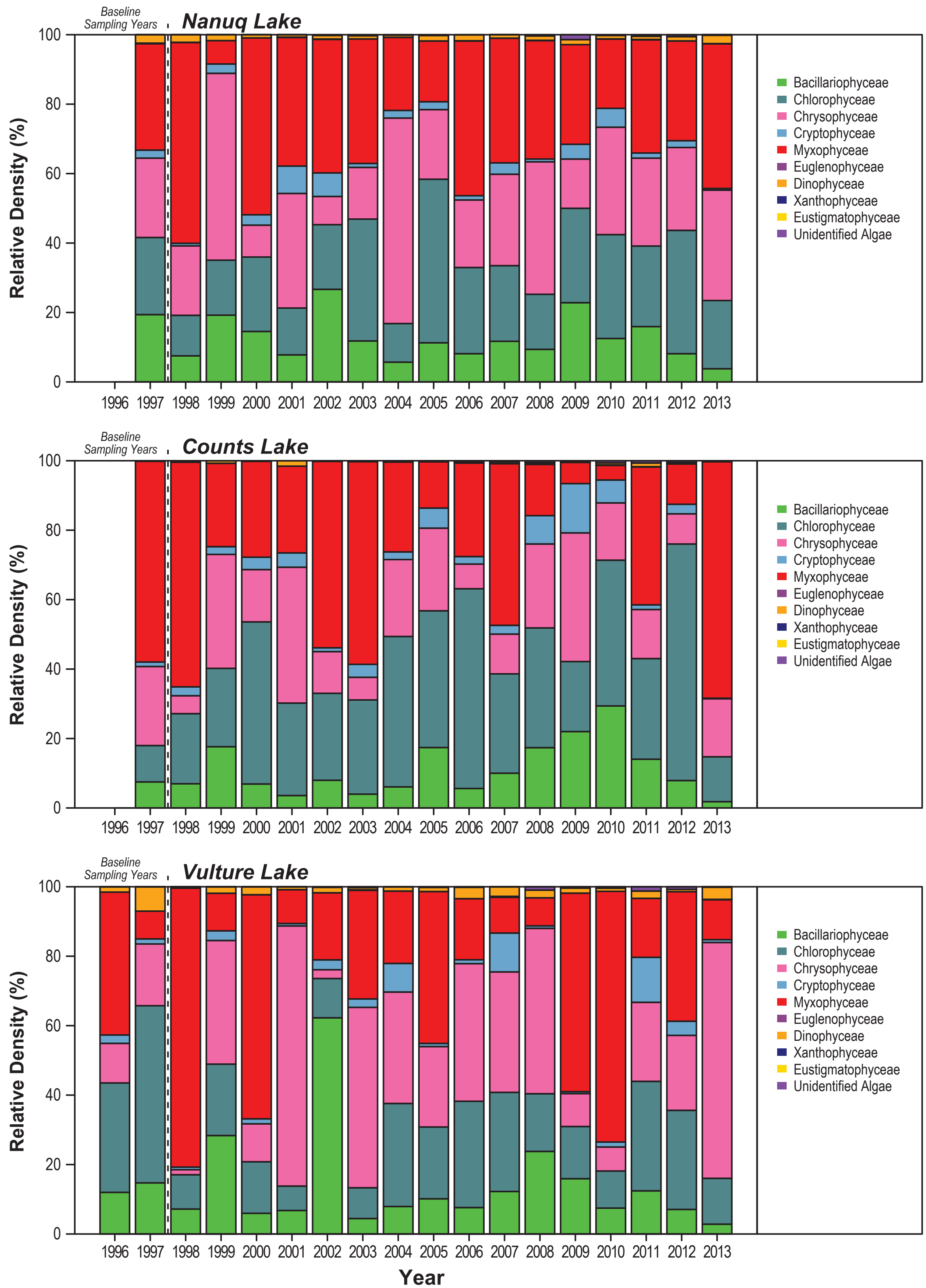


Figure 3.3-7a

Relative Densities of Phytoplankton Taxa in Lakes of the Koala Watershed, 1996 to 2013

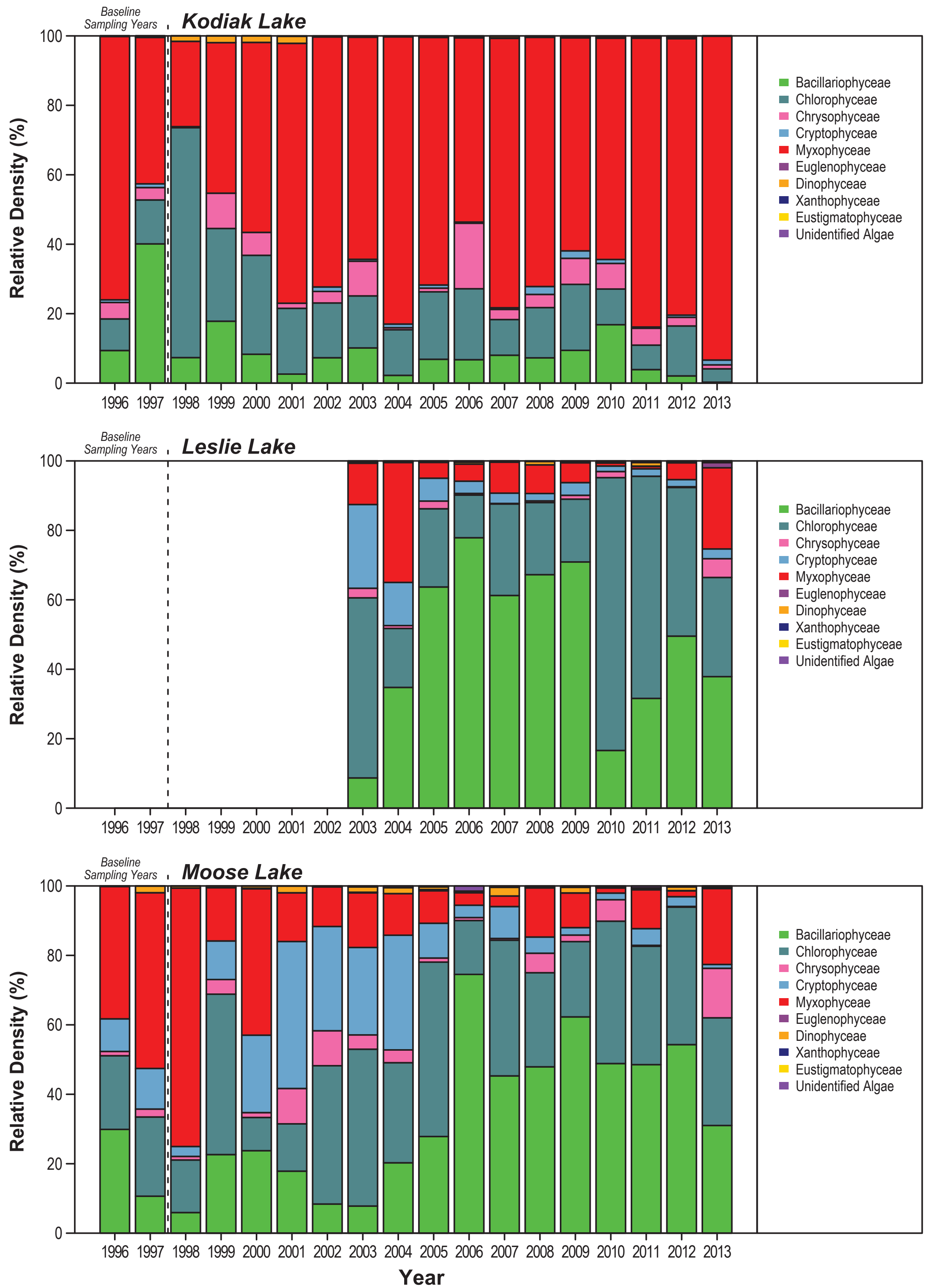


Figure 3.3-7b

Relative Densities of Phytoplankton Taxa in Lakes of the Koala Watershed, 1996 to 2013

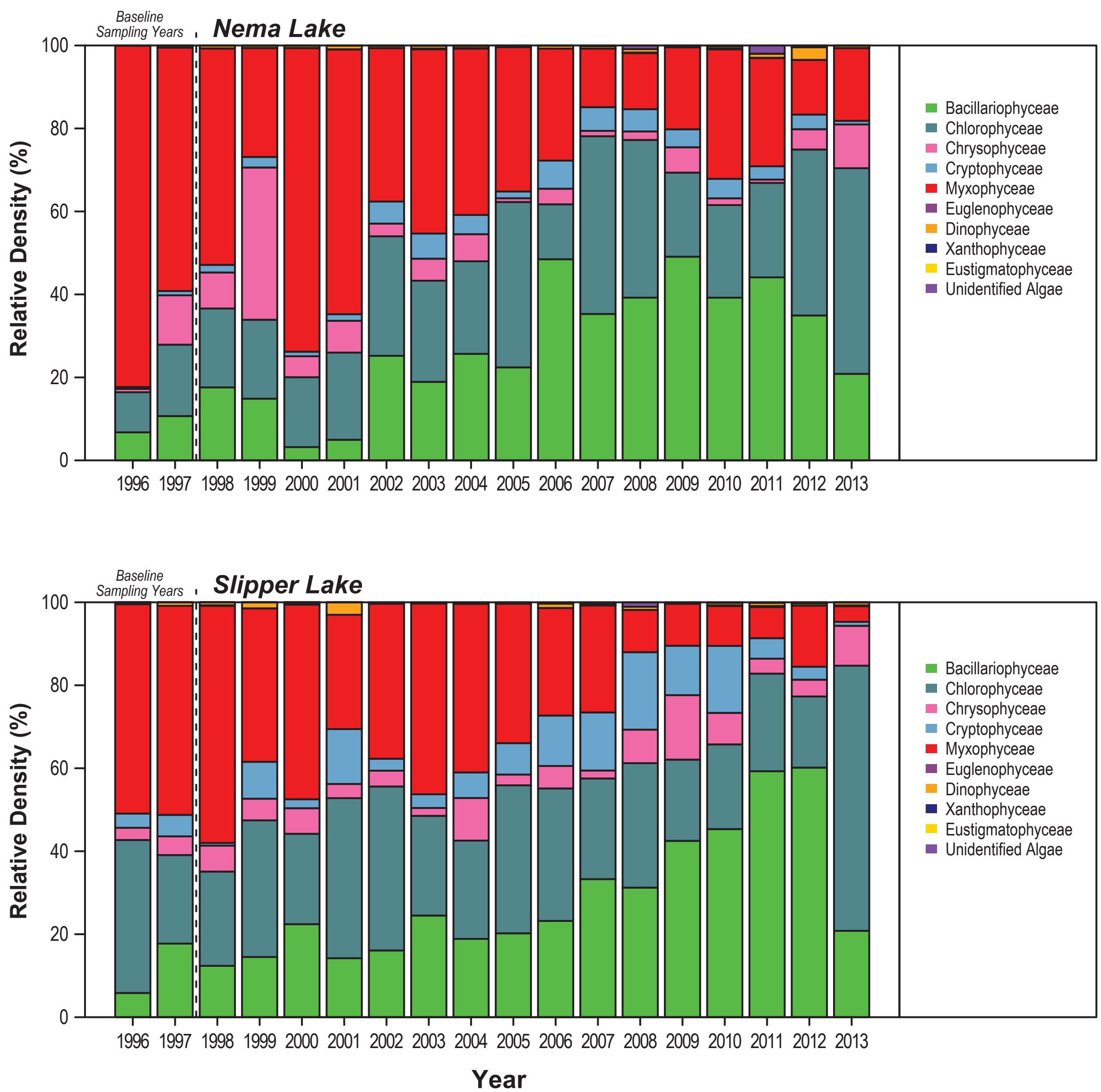


Figure 3.3-8

Relative Densities of Phytoplankton Taxa in Lac de Gras, 1996 to 2013

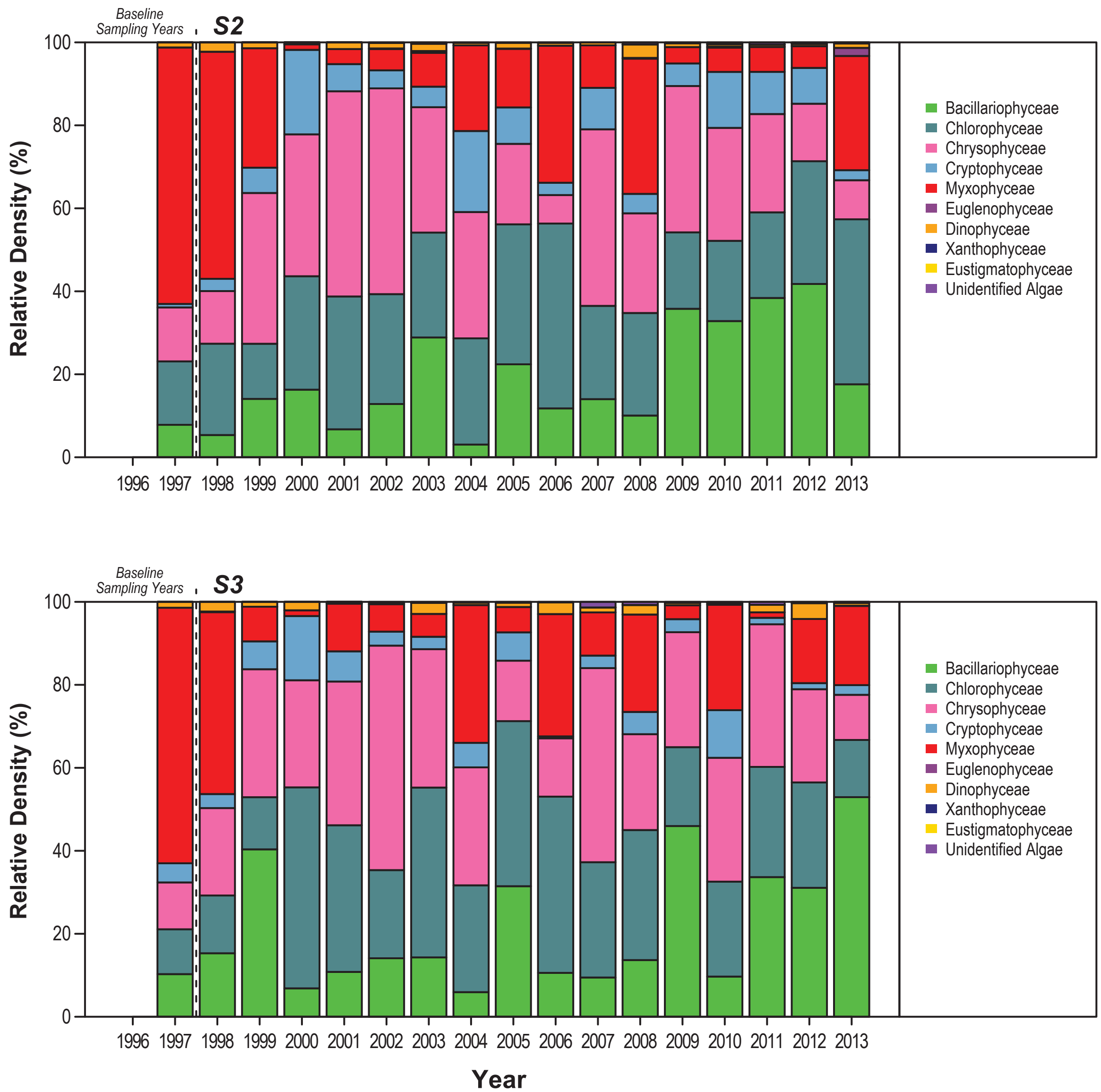


Table 3.3-6. Mean ± 2 Standard Deviations (SD) Baseline Phytoplankton Diversity in each of the Koala Watershed Lakes and Lac de Gras

Lake	Shannon Diversity			Simpson's Diversity		
	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD
Nanuq	2.46 (1)	1.92 – 3.01	2.46 ± 0.26	0.85 (1)	0.76 – 0.94	0.87 ± 0.02
Counts	1.77 (1)	1.54 - 2.01	2.16 ± 0.25	0.74 (1)	0.65 – 0.82	0.81 ± 0.05
Vulture	2.22 (3)	1.71 – 2.72	2.11 ± 0.26	0.81 (3)	0.68 – 0.94	0.78 ± 0.06
Kodiak	1.96 (4)	1.75 – 2.16	1.62 ± 0.07	0.77 (4)	0.73 - 0.82	0.63 ± 0.05
Leslie	-	-	2.59 ± 0.20	-	-	0.86 ± 0.03
Moose	2.02 (2)	1.19 – 2.86	2.67 ± 0.01	0.72 (2)	0.46 – 0.99	0.88 ± 0.01
Nema	2.11 (2)	1.64 – 2.59	2.68 ± 0.20	0.81 (2)	0.73 - 0.89	0.87 ± 0.03
Slipper	2.08 (3)	1.40 – 2.74	2.41 ± 0.36	0.74 (3)	0.54 - 0.94	0.78 ± 0.09
S2	1.75 (1)	1.51 – 1.99	2.84 ± 0.23	0.67 (1)	0.59 – 0.76	0.87 ± 0.04
S3	1.53 (1)	0.95 – 2.11	2.20 ± 0.08	0.59 (1)	0.36 – 0.82	0.70 ± 0.07

N = number of years data were collected.
Dashes indicates data not available.

Together, the evidence suggests that phytoplankton diversity has been stable through time in all monitored lakes of the Koala Watershed and Lac de Gras, except Leslie Lake. Phytoplankton diversity in Leslie Lake has returned to historical levels in 2013.

Graphical analyses of taxonomic composition suggest that the relative density of phytoplankton groups has been shifting through time in lakes downstream of the LLCF as far as site S2 in Lac de Gras (Figures 3.3-6 to 3.3-8). In general, Myxophyceae (blue-green algae) have gradually been replaced by Bacillariophyceae (diatoms), while the relative densities of Chlorophyceae (chlorophytes or green algae), Chrysophyceae (golden algae), and Cryptophyceae (cryptophytes) have remained relatively constant through time (except increases in Chlorophyceae in Leslie Lake from 2010 to 2012; Figure 3.3-7). Graphical analysis of absolute densities suggests that the change in relative abundances likely stems from a decrease in the density of Myxophyceae, while the number of Bacillariophyceae has remained relatively stable (Figures 3.3-4 to 3.3-5). Overall, the extent to which community composition has changed decreases with downstream distance from the LLCF (Figures 3.3-7 to 3.3-8).

The changes in phytoplankton community composition have not adversely affected diversity indices in any lake, other than Leslie Lake. Phytoplankton diversity was low in Leslie Lake from 2006 to 2011, likely reflecting changes in phytoplankton species composition through time. In addition to the decrease in the density of Myxophyceae during that time, Chlorophyceae densities were elevated from 2009 to 2012 (Figure 3.3-4, and 3.3-7). In 2013, diversity in Leslie Lake was comparable to historical levels. The increase in diversity likely reflects a more even distribution across species as the density of Myxophyceae increased and the density of Chlorophyceae decreased.

Increases in the density of Myxophyceae also corresponded to a decrease in the relative density of Bacillariophyceae in Leslie Lake in 2013. Similar patterns were observed in Moose Lake and site S2 in Lac de Gras (Figures 3.3-4 to 3.3-5, and 3.3-7 to 3.3-9). These patterns in community composition in 2013 are more comparable to community composition in baseline years than those observed more recently. Whether these shifts in 2013 indicate the onset of recovery in phytoplankton communities or represent an anomaly in recent trends is unclear at this time. Another potentially important shift was observed in Nema and Slipper lakes in 2013: The absolute density of Chlorophyceae increased in 2013, corresponding to a decrease in the relative density of Bacillariophyceae, resembling patterns observed in Leslie Lake from 2010 to 2012 (Figure 3.3-4 and 3.3-7). However, these changes in community composition did not result in any changes in diversity in Nema or Slipper lakes.

In contrast to the patterns observed in monitored lakes downstream of the LLCF, phytoplankton community composition has been relatively stable through time in all reference lakes and in Kodiak Lake (Figures 3.3-3, 3.3-4a, 3.3-6, and 3.3-7a). Thus, the observed shifts in phytoplankton community composition suggest that mining operations have affected phytoplankton community composition downstream of the LLCF as far as site S2 in Lac de Gras. Hypotheses regarding potential underlying causes of these changes are summarized in the Aquatic Biology Summary below (Section 3.3.5).

Overall, the main changes in phytoplankton community composition observed in lakes downstream of the LLCF has been a shift from blue-green algae to diatoms. Such a shift may cause cascading effects through the foodweb, where changes in phytoplankton composition may be associated with changes in the proportion of edible phytoplankton or the nutritional quality of phytoplankton. Diatoms generally have a higher fatty acid content than blue-green algae, which renders them a better quality food for herbivorous zooplankton (Lamberti 1996 as in Wehr and Sheath 2003). This may lead to changes in the nutrient content, abundance, or taxonomic composition of zooplankton, which may, in turn, cascade upward to affect higher trophic levels from secondary consumers to top predators like fish. While dominant taxa in reference lakes consist mostly of inedible organisms, dominant taxa at sites downstream of the LLCF (as far as site S3 in Lac de Gras) include large fractions of edible species from the genus *Cyclotella* (see Table 3.5-3 in Part 2 – Data Report).

The subsequent shift from diatoms to chlorophytes in Leslie Lake observed between 2009 to 2012, and in Nema and Slipper lakes in 2013, may also affect higher trophic levels. Chlorophytes are usually rare in sub-Arctic freshwater systems in the Northwest Territories (Moore 1978). Of the chlorophytes, *Monoraphidium minitum*, which are solitary cells that have no mucilage envelope and are likely edible by zooplankton, predominated in Nema and Slipper Lakes (see Table 3.5-3 in Part 2 – Data Report).

3.3.2 Zooplankton

3.3.2.1 Variables

Zooplankton are primary and secondary consumers that play an important role in the aquatic food web. Zooplankton feed on phytoplankton or other zooplankton and serve as an important food source for fish. Zooplankton monitoring can be used to help determine the extent to which mine effects have cascaded through the food web. Phytoplankton populations may appear to be suppressed despite increases in overall phytoplankton productivity due to the consumption of phytoplankton by zooplankton. Consequently, changes in the overall productivity may not be reflected in phytoplankton populations, but may be indicated by increases in zooplankton densities or changes in zooplankton community composition. Zooplankton community composition can also be used as an indicator of changes in water quality in the receiving environment as different species occupy different water chemistry niches and have different tolerances to changes in water quality. Therefore, zooplankton biomass (mg dry weight/m^3), density (organisms/m^3), and diversity (Shannon and Simpson's diversity indices) were monitored to detect potential mine effects.

3.3.2.2 Dataset

Zooplankton data have been collected between late July and early August each year from 1994 to 2013 (Table 3.3-7). Zooplankton biomass and taxonomic composition have been monitored using triplicate sampling from 1998 to present. Prior to 1998, zooplankton were monitored for taxonomic composition only. Baseline data, collected between 1994 and 1997, are included in Table 3.3-7 and shown graphically, below, but are not included in the statistical evaluation of effects.

Table 3.3-7. Dataset Used for Evaluation of Effects on Zooplankton in Koala Watershed Lakes and Lac de Gras

Year	Nanuq	Counts	Vulture	Kodiak	Leslie	Moose	Nema	Slipper	S2	S3
1993*	-	-	-	-	-	-	-	-	-	-
1994*	-	-	Aug-8	-	-	-	Aug-10	Aug-10	-	-
1995	-	-	Jul-28	-	-	Jul-27	Jul-26	Jul-26	-	-
1996*	Aug-4	Aug-14	Aug-5	Aug-7	-	Aug-10	Aug-10	Aug-11	Aug-12	Aug-12
1997*	Aug-4	Aug-4	Aug-7	Aug-10	-	Aug-8	Aug-7	Aug-6	Aug-5	Aug-5
1998	Aug-8	Aug-7	Aug-6	Aug-10	-	Aug-7	Aug-10	Aug-9	Aug-11	Aug-11
1999	Aug-4	Aug-1	Aug-4	Jul-29	-	Jul-30	Jul-30	Jul-31	Aug-3	Aug-3
2000	Aug-1	Jul-30	Aug-2	Jul-28	-	Aug-3	Aug-3	Jul-29	Jul-29	Jul-29
2001	Aug-1	Aug-7	Aug-3	Aug-2	-	Aug-5	Aug-4	Aug-6	Aug-4	Aug-4
2002	Aug-9	Aug-3	Aug-4	Aug-8	Aug-3	Aug-9	Aug-3	Aug-7	Aug-5	Aug-5
2003	Aug-10	Aug-13	Aug-9	Aug-7	Aug-9	Aug-10	Aug-9	Aug-12	Aug-9	Aug-9
2004	Aug-1	Aug-7	Jul-31	Aug-3	Aug-4	Aug-9	Aug-9	Aug-5	Aug-5	Aug-5
2005	Aug-2	Aug-4	Aug-2	Aug-1	Aug-6	Aug-5	Aug-5	Aug-4	Aug-4	Aug-4
2006	Aug-11	Aug-6	Aug-12	Aug-4	Aug-13	Aug-7	Aug-11	Aug-10	Aug-8	Aug-6
2007	Aug-8	Jul-31	Jul-29	Jul-27	Jul-31	Jul-29	Jul-31	Jul-29	Aug-7	Aug-7
2008	Aug-8	Jul-31	Jul-29	Jul-27	Jul-31	Jul-29	Jul-31	Jul-29	Aug-7	Aug-7
2009	Jul-30	Aug-1	Jul-30	Aug-8	Aug-5	Jul-30	Jul-30	Aug-3	Jul-31	Jul-31
2010	Aug-6	Aug-7	Aug-5	Aug-5	Aug-3	Aug-3	Aug-5	Aug-5	Aug-6	Aug-6
2011	Aug-2	Aug-5	Aug-5	Aug-5	Aug-2	Aug-3	Aug-5	Aug-3	Aug-4	Aug-4
2012	Aug-1	Aug-8	Aug-7	Aug-6	Aug-8	Aug-9	Aug-7	Aug-8	Aug-3	Aug-2
2013	Aug-3	Aug-1	Aug-1	Aug-6	Aug-1	Aug-5	Aug-6	Aug-5	Aug-2	Aug-2

** = Data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison.*
Dashes indicate no data were available.
Triplicate samples were collected yearly for biomass analysis from 1998 to 2013.
Triplicate samples were collected yearly for taxonomic analysis from 1995 to 2013.

3.3.2.3 Results and Discussion

Biomass

Statistical and graphical analyses suggest that zooplankton biomass has remained relatively stable through time in all monitored and reference lakes (Figure 3.3-9; Table 3.3-8). It was not possible to compare mean zooplankton biomass in 2013 to ± 2 SD of the baseline mean because mean zooplankton biomass was not assessed prior to 1998. No mine effects were detected with respect to zooplankton biomass.

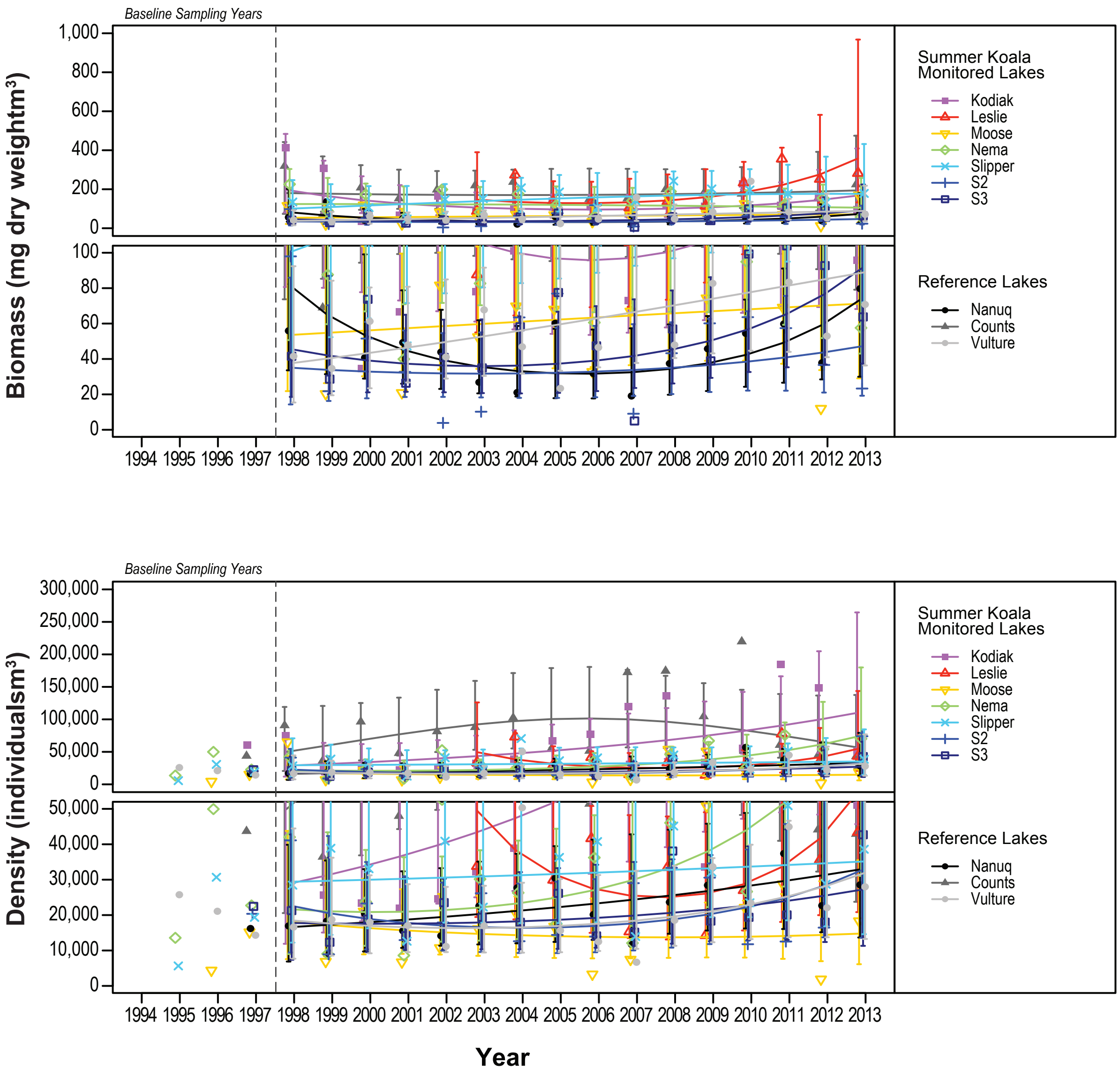
Table 3.3-8. Statistical Results of Zooplankton Biomass in Lakes in the Koala Watershed and Lac de Gras

	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
				Model Fit = 3	Model Fit = 2	Model Fit = 1	
Zooplankton biomass	-	LME	2	-	None	-	1-380

Dashes indicate not applicable.

Figure 3.3-9

Observed and Fitted Means for Zooplankton Biomass and Density in Koala Watershed Lakes and Lac de Gras, August 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.

Density

Statistical analyses indicate that changes in zooplankton densities have been similar through time in all monitored and reference lakes (Table 3.3-9). Graphical analysis suggests that zooplankton densities have been relatively stable in all monitored and reference lakes since monitoring began, though zooplankton densities were elevated, relative to baseline years, in Kodiak Lake in 2011 and 2012 (Figure 3.3-9).

Table 3.3-9. Statistical Results of Zooplankton Density in Lakes in the Koala Watershed and Lac de Gras

	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
				Model Fit = 3	Model Fit = 2	Model Fit = 1	
Zooplankton density	-	LME	3	None	None	-	1-386

Dashes indicate not applicable.

Compared to mean baseline densities ± 2 SD, mean zooplankton densities in 2013 were greater in Nanuq and Nema lakes and sites S2 and S3 in Lac de Gras (Table 3.3-10). No change in zooplankton density was observed in monitored lakes upstream of Nema and the increase observed at Nema, S2 and S3 was similar to that observed in the reference lake Nanuq. Thus, no mine effects were detected with respect to zooplankton density.

Table 3.3-10. Mean ± 2 Standard Deviations (SD) Baseline Zooplankton Density in Each of the Koala Watershed Lakes and Lac de Gras

Lake	Baseline Mean (N)	Mean Baseline Range , ± 2 SD	2013 Mean ± 1 SD
Nanuq	16,209 (1)	13,053 - 19,365	28,547 ± 1,924
Counts	43,710 (1)	33,027 - 54,392	42,894 ± 4,916
Vulture	20,384 (3)	9,704 - 31,064	27,987 ± 1,506
Kodiak	113,472 (3)	0 - 286,939	51,041 ± 5,423
Leslie	-	-	43,452 ± 6,530
Moose	9,782 (2)	0 - 21,900	18,312 ± 2,823
Nema	28,744 (3)	0 - 65,016	70,215 ± 3,407
Slipper	18,562 (3)	0 - 41,464	38,591 ± 4,107
S2	20,360 (1)	15,280 - 25,441	74,007 ± 23,143
S3	22,451 (1)	14,665 - 30,238	42,693 ± 12,091

Units are organisms/m^3.
Negative values were replaced with zeros.
N = number of years data were collected.

Diversity

Statistical analyses were not performed on the diversity datasets because the calculation of indices can result in data abnormalities that prevent statistical analysis. Consequently, graphical analyses of temporal trends in diversity indices (Figure 3.3-10) and best professional judgment were the primary methods used in the evaluation of effects. In addition, the average and relative densities of taxa were examined using graphical analyses to identify potential changes in community composition (Figures 3.3-11 to 3.3-16).

Figure 3.3-10

Average Diversity Indices for Zooplankton in Koala Watershed Lakes and Lac de Gras, August 1995 to 2013

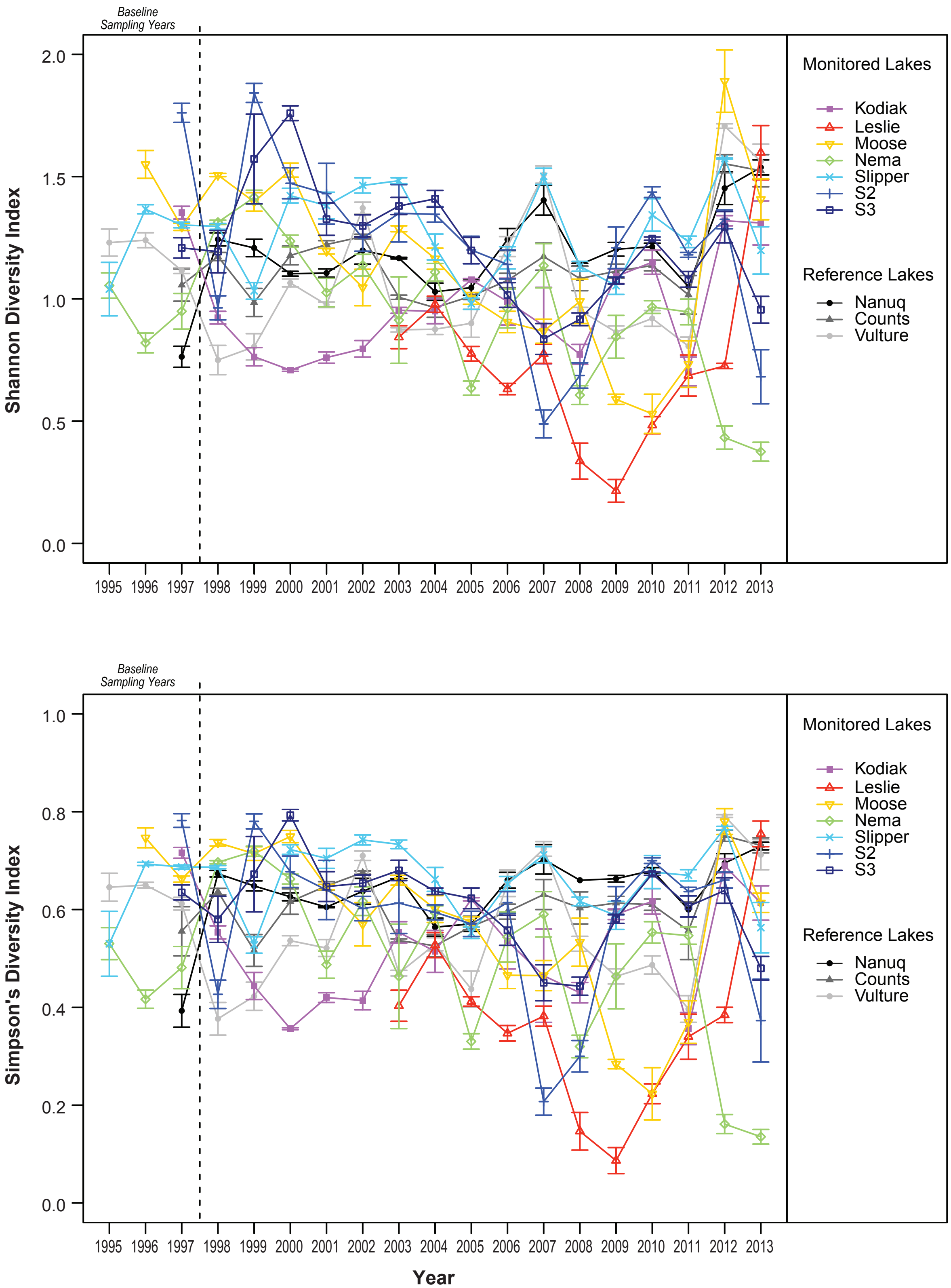


Figure 3.3-11

Average Zooplankton Density by Taxonomic Group for AEMP Reference Lakes, 1995 to 2013

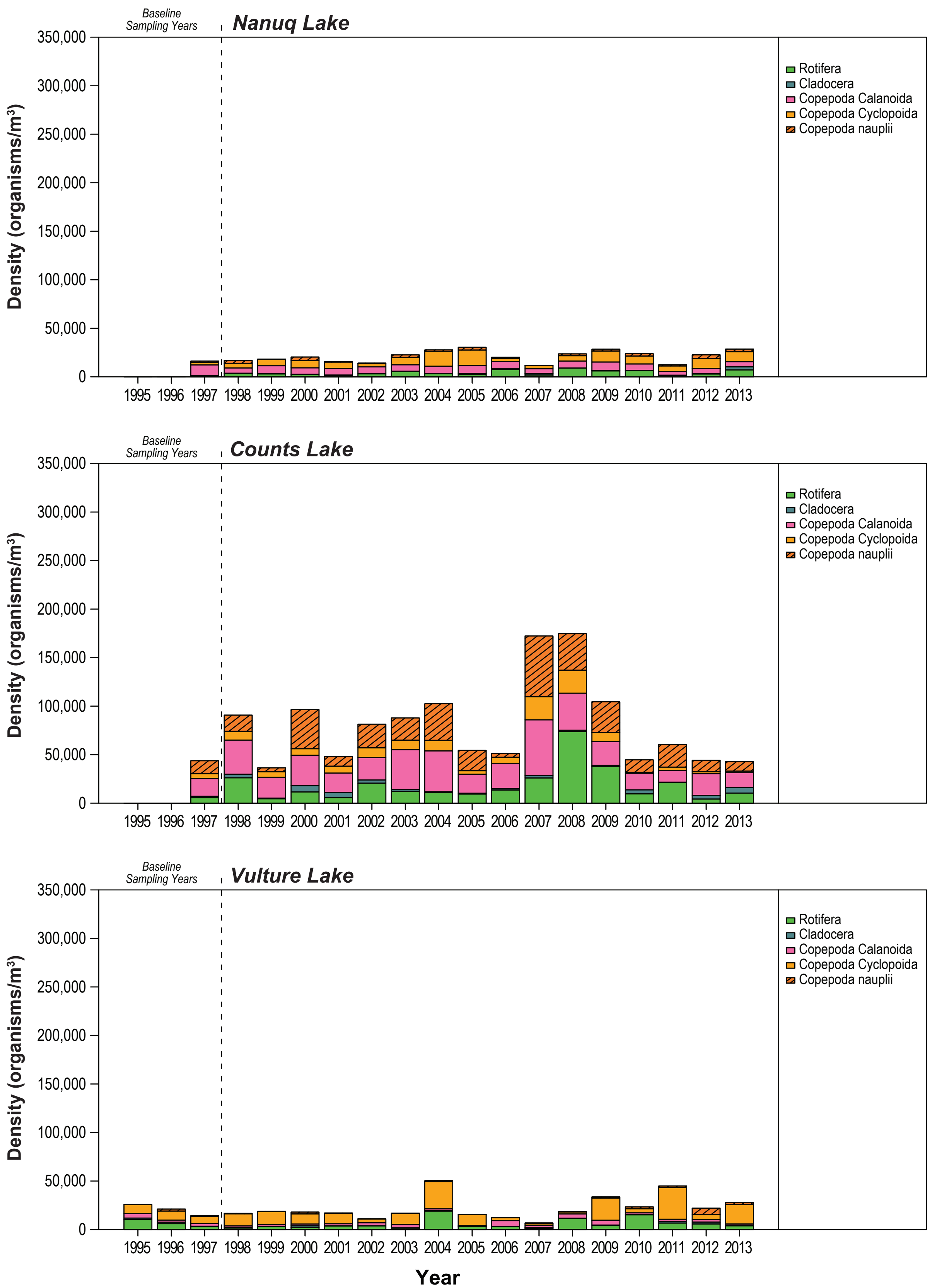


Figure 3.3-12a

Average Zooplankton Density by Taxonomic Group for Lakes of the Koala Watershed, 1995 to 2013

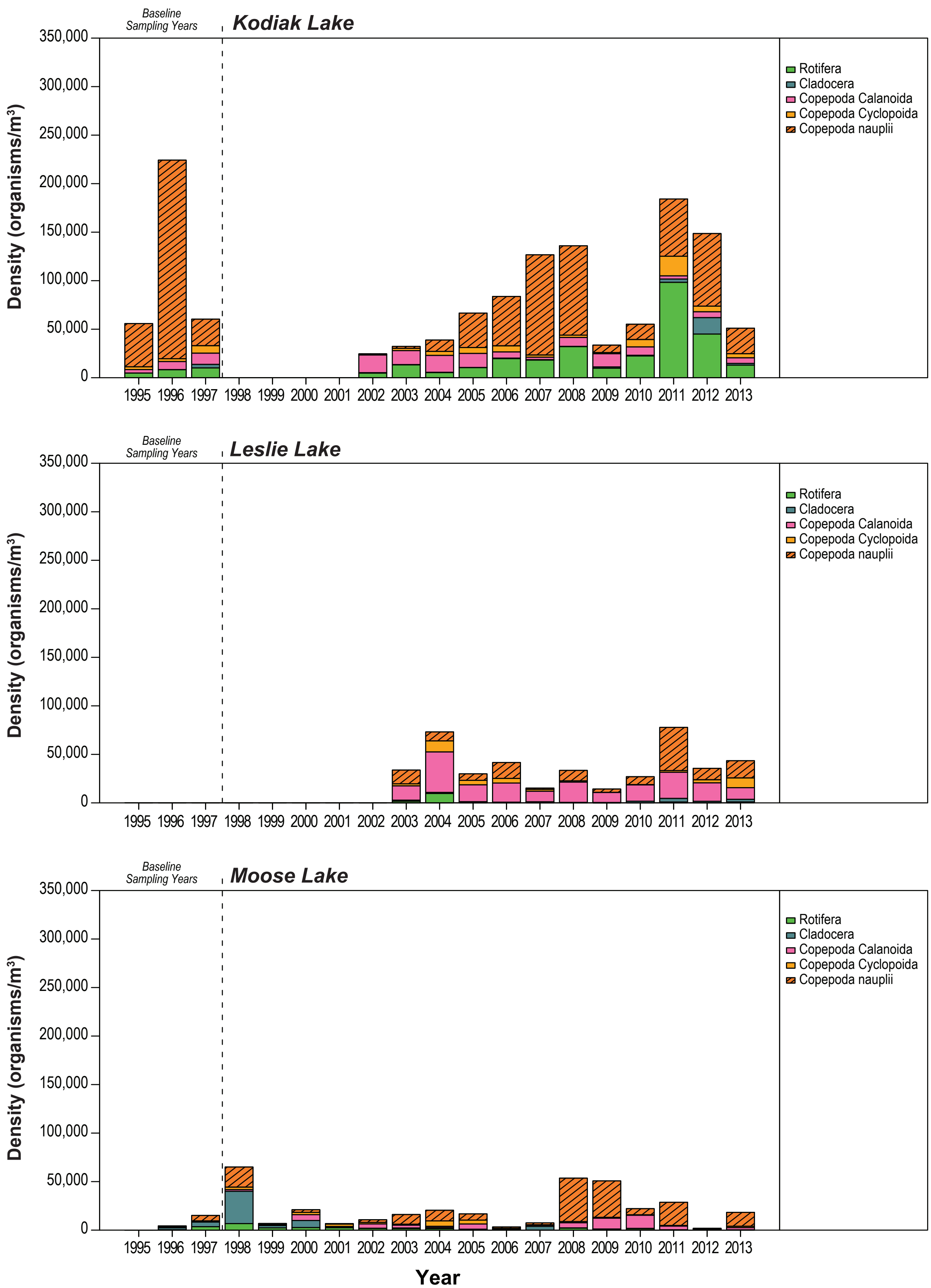
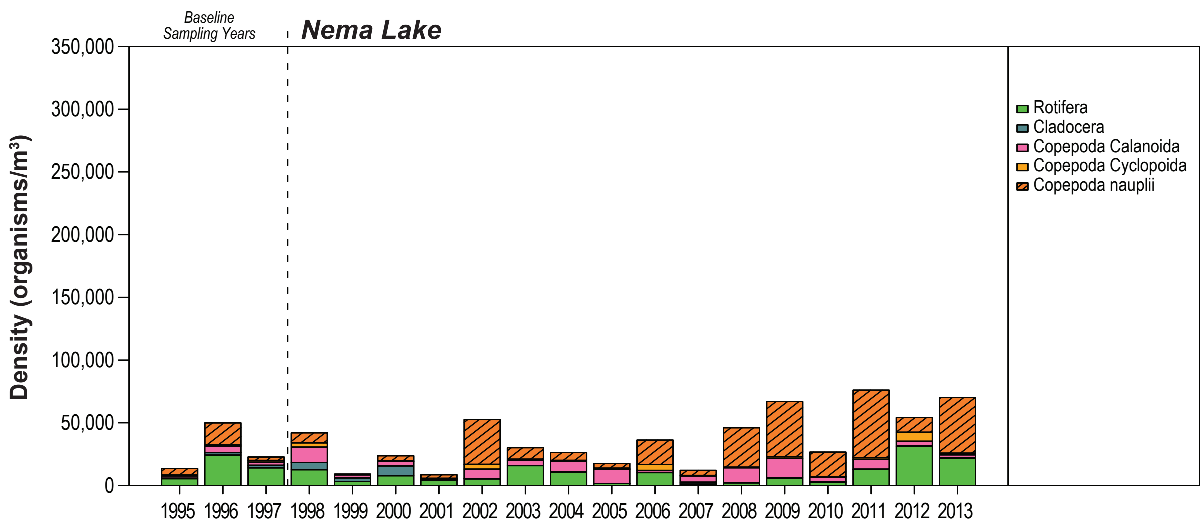


Figure 3.3-12b

Average Zooplankton Density by Taxonomic Group for Lakes of the Koala Watershed, 1995 to 2013



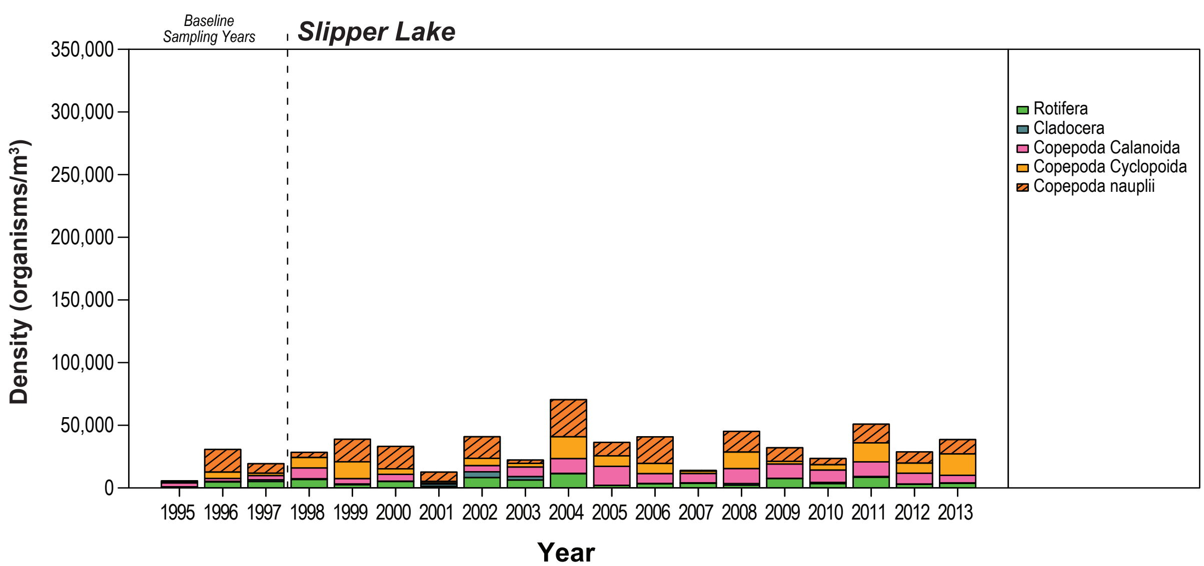
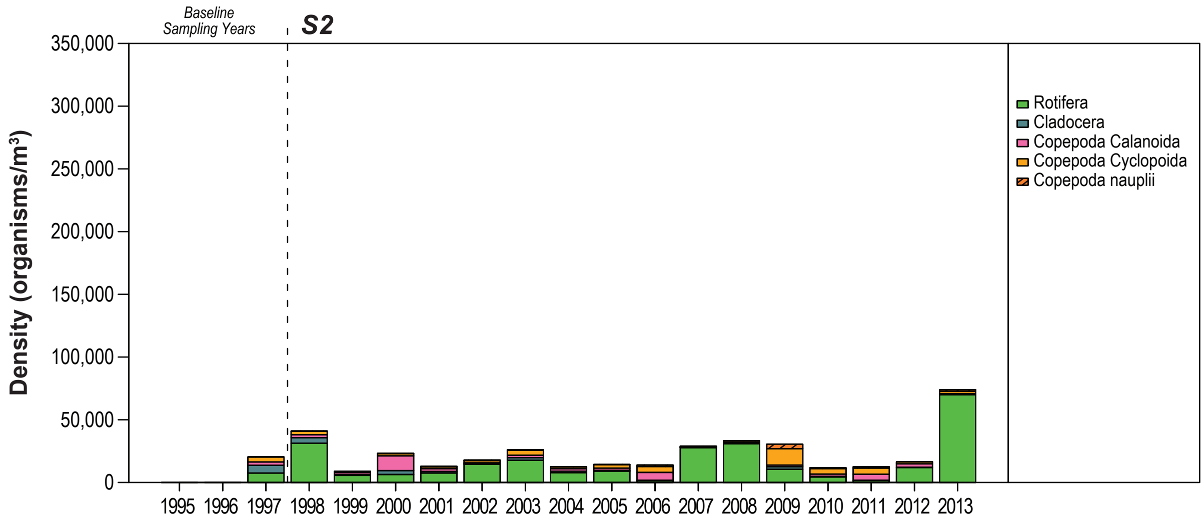


Figure 3.3-13

Average Zooplankton Density by Taxonomic Group for Lac de Gras, 1995 to 2013



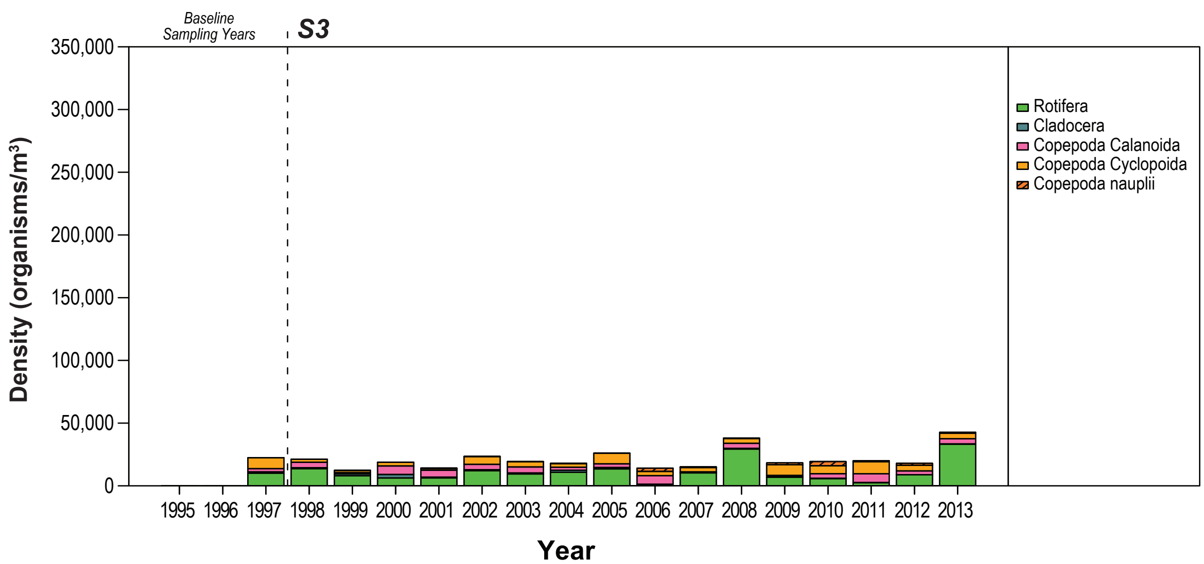


Figure 3.3-14

Relative Densities of Zooplankton Taxa in AEMP Reference Lakes, 1995 to 2013

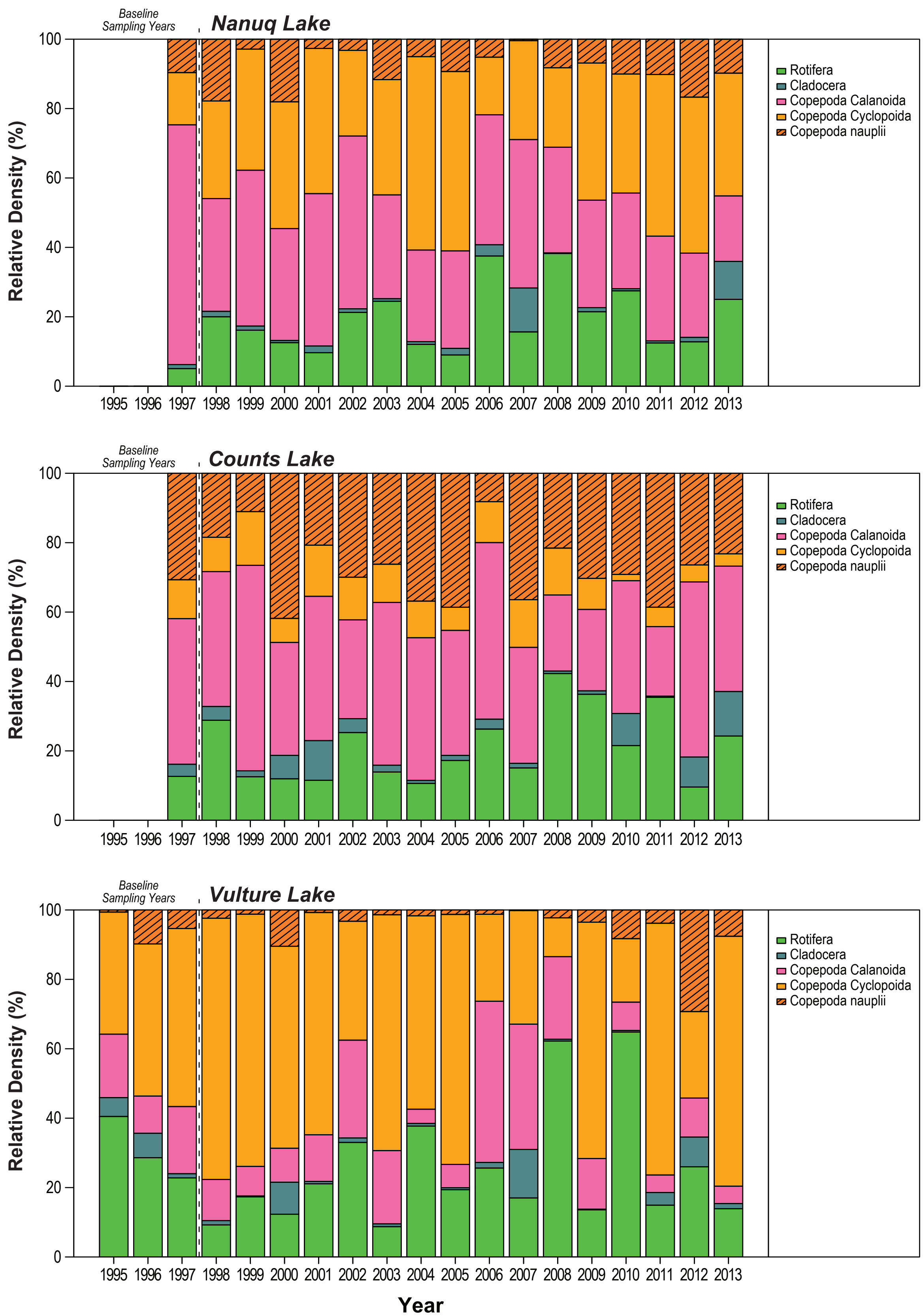


Figure 3.3-15a

Relative Densities of Zooplankton Taxa in Lakes of the Koala Watershed, 1995 to 2013

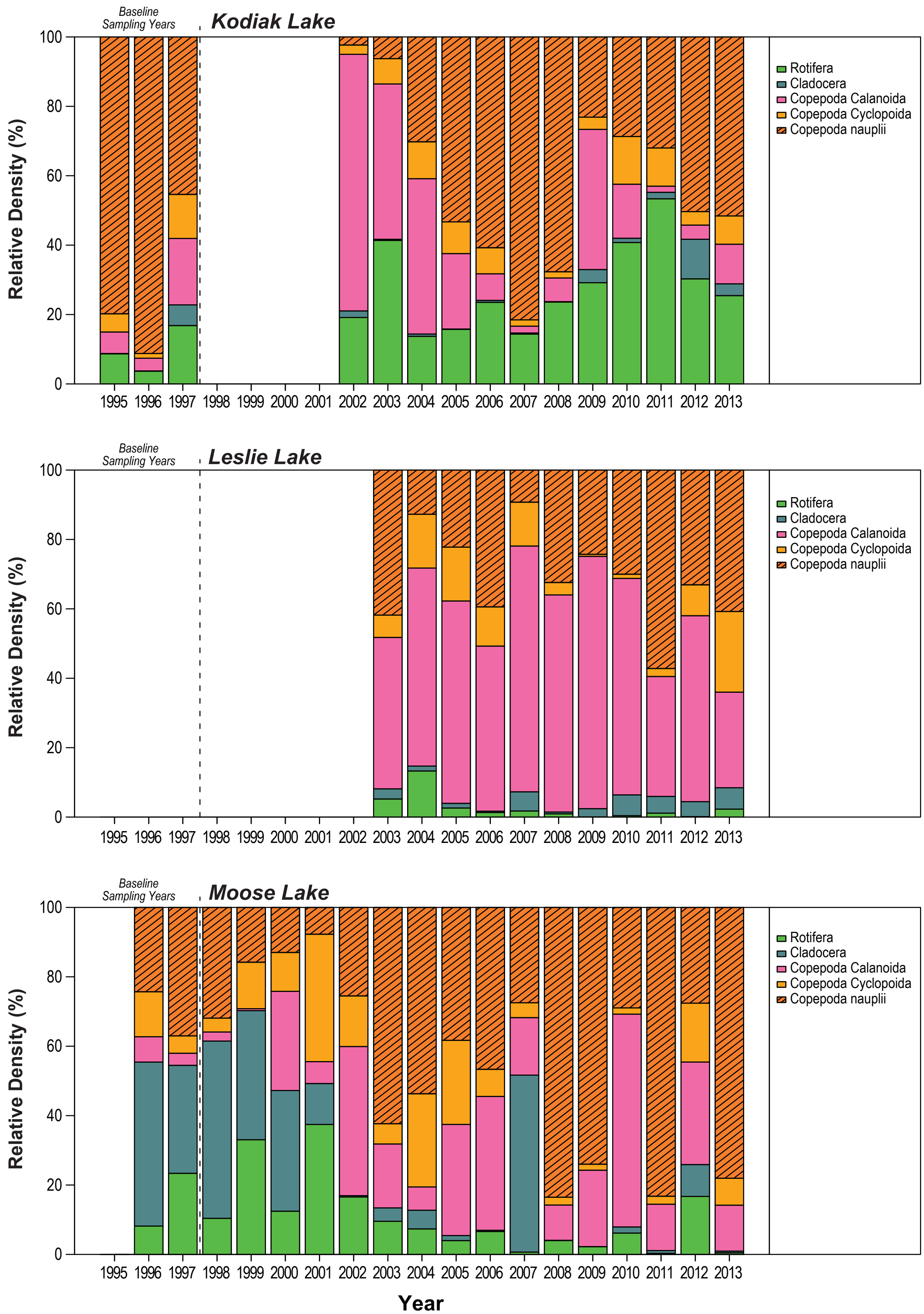
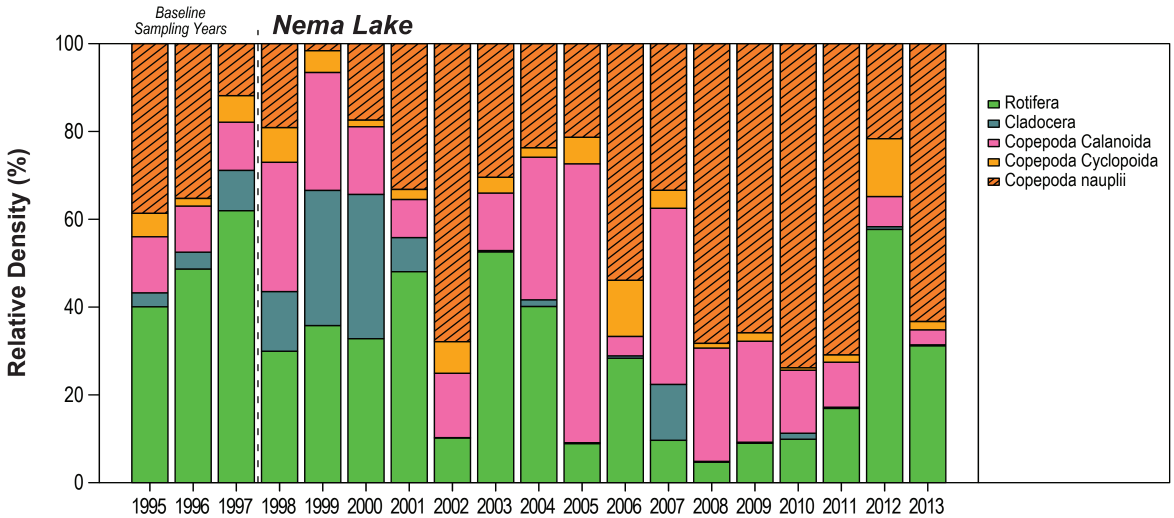


Figure 3.3-15b

Relative Densities of Zooplankton Taxa in Lakes of the Koala Watershed, 1995 to 2013



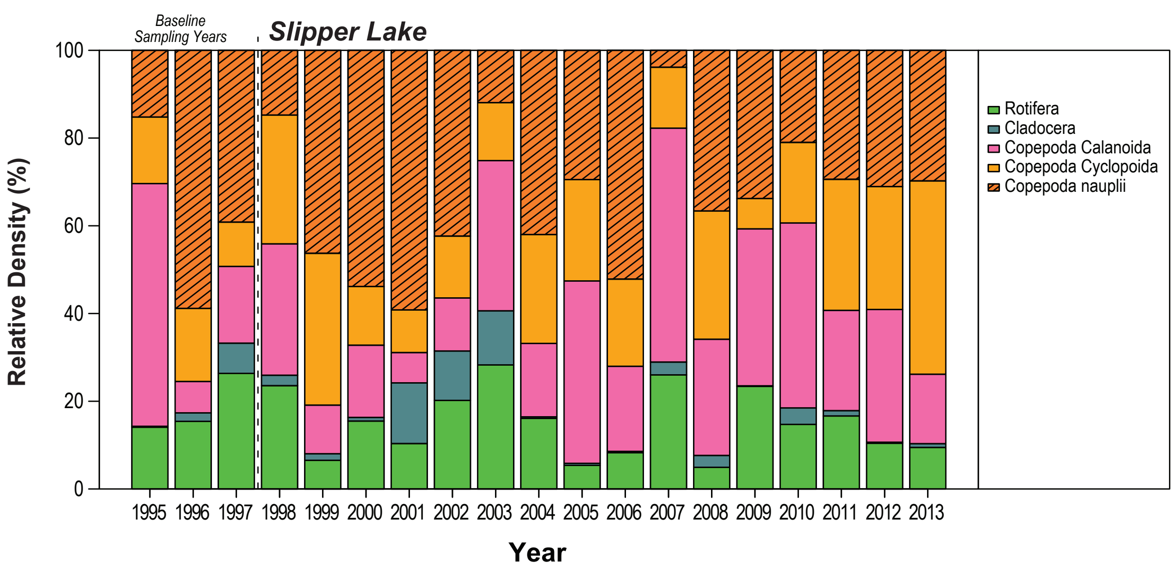
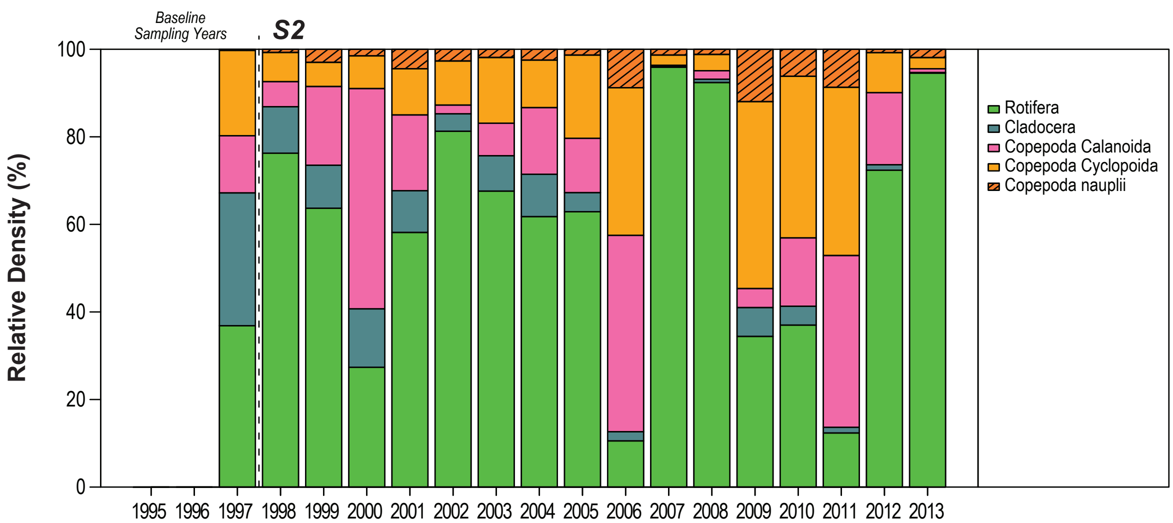
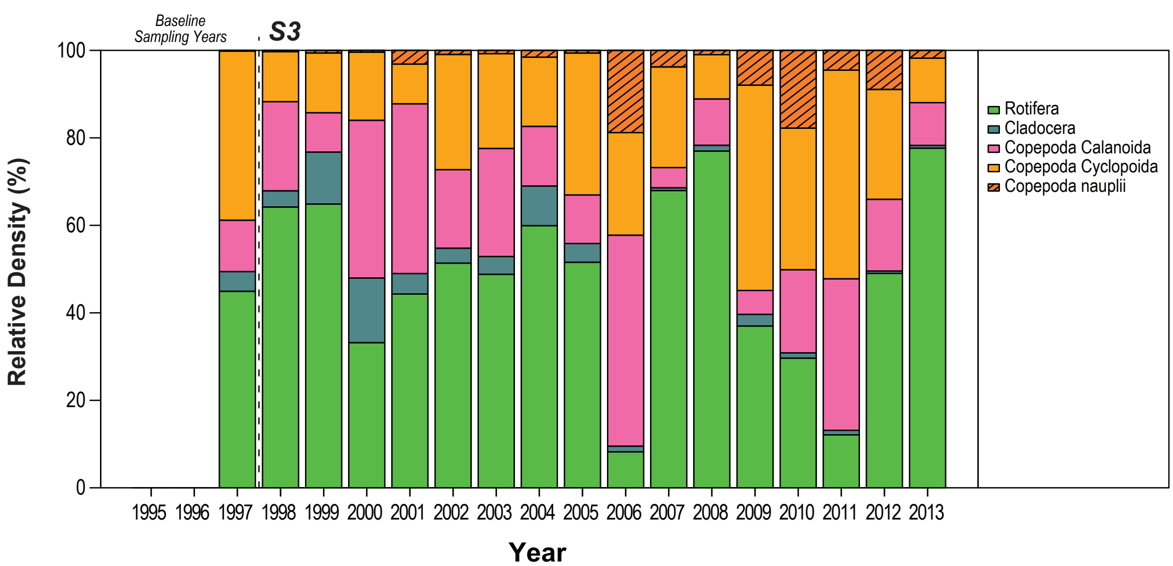


Figure 3.3-16

Relative Densities of Zooplankton Taxa in Lac de Gras, 1995 to 2013





Both Shannon and Simpson's diversity indices have varied considerably through time in both monitored and reference lakes (Figure 3.3-10). While the variability makes it somewhat difficult to discern temporal trends, both Shannon and Simpson's diversity indices have generally declined through time in Leslie and Moose lakes since monitoring began (Figure 3.3-10). However, in both cases, diversity has increased in recent years and was greater than it has ever been in Leslie Lake in 2013. Diversity in Nema Lake has generally been stable through time, but has decreased in recent years (Figure 3.3-10). Compared to mean diversity ± 2 SD in baseline years, mean zooplankton diversity was lower in Nema Lake and sites S2 and S3 in Lac de Gras in 2013 (Table 3.3-11). In contrast, zooplankton diversity in 2013 was greater than in baseline years in all three reference lakes, as well as in two monitored lakes, Moose and Slipper (Table 3.3-11).

Table 3.3-11. Mean ± 2 Standard Deviations (SD) Baseline Zooplankton Diversity in Each of the Koala Watershed Lakes and Lac de Gras

	Shannon Diversity			Simpson's Diversity		
Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD
Nanuq	0.76 (1)	0.61 - 0.91	1.54 ± 0.05	0.39 (1)	0.28 - 0.51	0.73 ± 0.01
Counts	1.06 (1)	0.83 - 1.29	1.52 ± 0.11	0.56 (1)	0.38 - 0.73	0.73 ± 0.03
Vulture	1.19 (3)	1.03 - 1.36	1.56 ± 0.12	0.64 (3)	0.57 - 0.70	0.71 ± 0.05
Kodiak	1.35 (1)	1.26 - 1.44	1.31 ± 0.16	0.72 (1)	0.68 - 0.75	0.61 ± 0.06
Leslie	-	-	1.60 ± 0.19	-	-	0.75 ± 0.05
Moose	1.44 (2)	1.11 - 1.73	1.41 ± 0.14	0.70 (2)	0.60 - 0.81	0.61 ± 0.03
Nema	0.94 (3)	0.68 - 1.21	0.38 ± 0.07	0.48 (3)	0.34 - 0.62	0.14 ± 0.03
Slipper	1.24 (3)	0.88 - 1.59	1.20 ± 0.16	0.64 (3)	0.44 - 0.83	0.56 ± 0.09
S2	1.76 (1)	1.63 - 1.90	0.68 ± 0.19	0.78 (1)	0.73 - 0.83	0.37 ± 0.15
S3	1.21 (1)	1.06 - 1.35	0.96 ± 0.09	0.63 (1)	0.58 - 0.69	0.48 ± 0.04

N = number of years data were collected.
Dashes indicate not available.

The relative densities of different taxonomic groups in reference lakes has remained fairly consistent through time, with rotifers, cladocerans, and calanoid and cyclopoid copepods comprising similar fractions of the total density of zooplankton in each lake through time (Figures 3.3-11 and 3.3-14). In general, copepods (i.e., calanoids, cyclopoids, and nauplii) comprise about two thirds of each community, while rotifers comprise about one third. The remainder of the community is composed of cladocerans, which comprise a consistent, but much smaller, fraction of the total density of organisms present (Figures 3.3-11 and 3.3-14; see Part 2 – Data Report). In contrast to reference lakes, zooplankton community compositions have been more variable in Moose and Nema lakes (Figures 3.3-12 and 3.3-15). With the exception of 2007, recent community compositions in Moose Lake represent a departure from community compositions in baseline years where cladocerans comprised a large fraction (~50%) of total zooplankton density (Figure 3.3-12a). Changes in zooplankton community composition in Nema Lake have resembled those observed in Moose Lake, particularly since 2008, when both rotifers and cladocerans comprised a very small fraction (< 5%) of total zooplankton density (Figure 3.3-12b). Overall, rotifer densities appear to have declined over time in Moose and Nema lakes (Figures 3.3-12 and 3.3-15), a trend that is consistent with results of the 2012 AEMP Re-evaluation (Rescan 2012d). However, rotifer populations in Nema Lake may be recovering because observed densities in 2012 and 2013 were comparable to those observed in baseline years (Figure 3.3-12b). In Leslie Lake, zooplankton compositions have been similar since monitoring began (Figure 3.3-12a and 3.3-15a). However, zooplankton populations have only been monitored in Leslie Lake since 2003. Since that time, communities have more closely resembled the more recent community structure of Moose Lake, with low populations of cladocerans and rotifers and high populations of copepods, when compared to compositions observed in reference lakes (Figure 3.3-12 and 3.3-15). Further downstream from the LLCF, community compositions have been more consistent through time, if somewhat more variable than in reference lakes (Figures 3.3-12 to 3.3-16). In addition, sites S2 and S3 in Lac de Gras differ from other sites in that they tend to be consistently dominated by rotifers through time (Figures 3.3-15 and 3.3-16).

A closer examination of historical population trends at the genera level suggests that the overall decline of cladocerans in Moose and Nema lakes is a function of a reduction in the population densities of *Holopedium gibberum*. With the exception of low abundances recorded in 2009 in Moose Lake, *H. gibberum* has been absent from Moose Lake samples since 2002 and from Nema Lake samples since 2003. *H. gibberum* has been absent from Leslie Lake samples since monitoring began in 2003. Given that *H. gibberum* has been historically common and abundant in both monitored and reference lakes, it is likely that *H. gibberum* has declined in Leslie Lake as well. While the population density of another cladoceran, *Daphnia* sp., has increased in Leslie Lake since 2010, the increase in *Daphnia* populations is relatively small compared to the decline in *H. gibberum* (as evidenced by the overall decline in cladocerans in Moose Lake, Nema Lake, and, presumably, Leslie Lake).

As indicated in Section 3.2.4-5 , the observed mean potassium concentrations in Leslie and Moose lakes exceeded the potassium SSWQO of 41 mg/L (Rescan 2012g). Also, in Leslie Lake, the upper 95% confidence interval of the fitted mean exceeded the lowest identified chronic effect level of 53 mg/L for the most sensitive species (i.e., *Daphnia magna*) during the ice-covered season (Biesinger and Christensen 1972). To date, there is no evidence that elevated potassium concentrations have led to a decline in the density of *Daphnia* sp. in Leslie or Moose lakes. Instead, the observed decline of cladocerans in Leslie and Moose lakes have been linked to a decrease in the density of *Holopedium gibberum*, while *Daphnia* sp. has increased in Leslie Lake since 2010.

The observed densities of two species of rotifers have also declined in Moose and Nema. These include *Conochilus* sp. (a colonial rotifer) and *Kellicottia longispina*. Historically, *Conochilus* has been present in low densities in both monitored and reference lakes, but it has been largely absent from Moose Lake since 2004 and from Nema Lake since 2005. *Conochilus* has also been absent from Leslie Lake samples since monitoring began in 2003. The density of *K. longispina* has also declined through time in lakes downstream of the LLCF as far as Nema Lake and was generally low, compared to reference lakes, in Moose Lake in 2013. However, populations of *K. longispina* continued to show signs of recovery in Nema Lake in 2013. In fact, the low zooplankton diversity observed in Nema Lake in 2013 is likely related to the dominance of *K. longispina*, where populations were more than an order of magnitude greater than other zooplankton genera in Nema Lake in 2013. Thus, the dominance of *Kellicottia longispina* in Nema Lake in 2013 and associated reduction in genera diversity may represent recovery of zooplankton populations toward baseline conditions in Nema Lake and is not cause for concern.

Together, changes in diversity and relative density suggest that mine activities have affected zooplankton community compositions downstream from the LLCF as far as Nema Lake. However, zooplankton diversity and community composition are showing some signs of recovery in Nema Lake. Hypotheses regarding potential underlying causes of changes in zooplankton communities and their potential effects on higher trophic levels are included in Aquatic Biology Summary (see Section 3.3.5).

3.3.3 Lake Benthos

3.3.3.1 Variables

Lake benthos are a group of organisms that live in association with lake sediments. They provide an important source of food for many species of fish. Dipterans (flies) tend to dominate benthic invertebrate communities and are widely used as indicators of ecosystem health, including sediment quality. Thus, lake benthos density (organisms/m^2) and dipteran diversity (Shannon and Simpson's diversity indices) were evaluated for potential mine effects.

3.3.3.2 Dataset

Benthos samples have been collected in triplicate replicates in late July or early August of each year since 1994 (Table 3.3-12). Beginning in 2011, composite samples, consisting of three subsamples per replicate, were collected. Baseline data, collected from 1994 to 1997, were not used in the statistical evaluation of effects but are included below in Table 3.3-12 and shown graphically for visual comparison.

Table 3.3-12. Dataset Used for Evaluation of Effects on the Benthos in Koala Watershed Lakes and Lac de Gras

Year	Nanuq	Counts	Vulture	Kodiak	Leslie	Moose	Nema	Slipper	S2
1994*	-	-	Aug-13	-	-	Aug-22	-	Aug-15	Aug-14
1995*	-	-	Aug-9	Aug-10	-	-	Aug-11	Aug-11	-
1996*	-	-	Jul-27	Jul-27	-	Jul-27	Jul-29	Jul-26	-
1997*	Aug-4	Aug-14	Aug-5	Aug-7	-	Aug-10	Aug-10	Aug-11	Aug-12
1998	Aug-4	Aug-4	Aug-7	Jul-29	-	Aug-8	Aug-7	-	Aug-5
1999	Jul-30	Jul-30	Jul-29	Aug-7	-	Aug-2	Aug-2	Aug-1	Aug-1
2000	Aug-4	Aug-1	Aug-4	Jul-29	-	Jul-30	Jul-30	Jul-31	Aug-3
2001	Aug-1	Jul-30	Aug-2	Jul-28	-	Aug-3	Aug-3	Jul-31	Jul-29
2002	Aug-3	Aug-7	Aug-3	Aug-2	-	Aug-5	Aug-4	Aug-6	Aug-4
2003	Aug-9	Aug-7	Aug-4	Aug-8	Aug-6	Aug-9	Aug-6	Aug-7	Aug-5
2004	Aug-10	Aug-13	Aug-9	Aug-7	Aug-9	Aug-10	Aug-9	Aug-12	Aug-9
2005	Aug-1	Aug-7	Jul-31	Aug-3	Aug-4	Aug-8	Aug-8	Aug-5	Aug-5
2006	Aug-2	Aug-4	Aug-7	Aug-7	Aug-6	Aug-5	Aug-5	Aug-4	Aug-4
2007	Aug-11	Aug-6	Aug-12	Aug-4	Aug-13	Aug-7	Aug-11	Aug-10	Aug-6
2008	Aug-8	Jul-31	Aug-5	Jul-27	Jul-31	Jul-29	Jul-29	Jul-29	Aug-7
2009	Jul-30	Aug-1	Jul-31	Aug-8	Aug-5	Jul-30	Jul-30	Aug-3	Jul-31
2010	Aug-6	Aug-8	Aug-5	Aug-5	Aug-4	Aug-3	Aug-5	Aug-5	Aug-6
2011	Aug-2	Aug-5	Aug-5	Aug-6	Aug-3	Aug-3	Aug-5	Aug-3	Aug-4
2012	Aug-9	Aug-6	Aug-7	Aug-6	Aug-8	Aug-9	Aug-7	Aug-8	Aug-3
2013	Aug-3	Aug-1	Jul-31	Aug-6	Aug-1	Aug-5	Aug-6	Aug-5	Aug-2

** = Data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison.*
Dashes indicate no data were available.

3.3.3.3 Results and Discussion

Benthos Density

Statistical and graphical analyses indicate that the density of benthos has been stable through time in all monitored and reference lakes (Table 3.3-13; Figure 3.3-17). Mean benthos densities in 2013 were also within the range of ± 2 SD of mean densities during baseline years in all reference and monitored lakes except Nema Lake, in which they were greater (a similar pattern for all of the reference lakes; Table 3.3-14). No mine effects were detected.

Table 3.3-13. Statistical Results of Benthos Density in Lakes in the Koala Watershed and Lac de Gras

	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
				Model Fit = 3	Model Fit = 2	Model Fit = 1	
Density	-	LME	2	-	None	-	1-392

Dashes indicate not applicable.

Table 3.3-14. Mean ± 2 Standard Deviations (SD) Baseline Benthos Density in Each of the Koala Watershed Lakes and Lac de Gras

Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD
Nanuq	726 (1)	325 – 1,126	2,573 ± 1,420
Counts	1,289 (1)	0 – 3,212	5,406 ± 2,188
Vulture	852 (4)	0 – 1,960	3,328 ± 2,338
Kodiak	3,471 (3)	0 – 10,430	7,515 ± 2,114
Leslie	-	-	8,514 ± 9,370
Moose	5,683 (3)	0 – 17,195	3,694 ± 2,842
Nema	2,291 (3)	0 – 8,005	14,565 ± 5,964
Slipper	1,641 (4)	0 – 4,218	2,731 ± 275
S2	5,333 (2)	0 – 12,536	1,689 ± 689

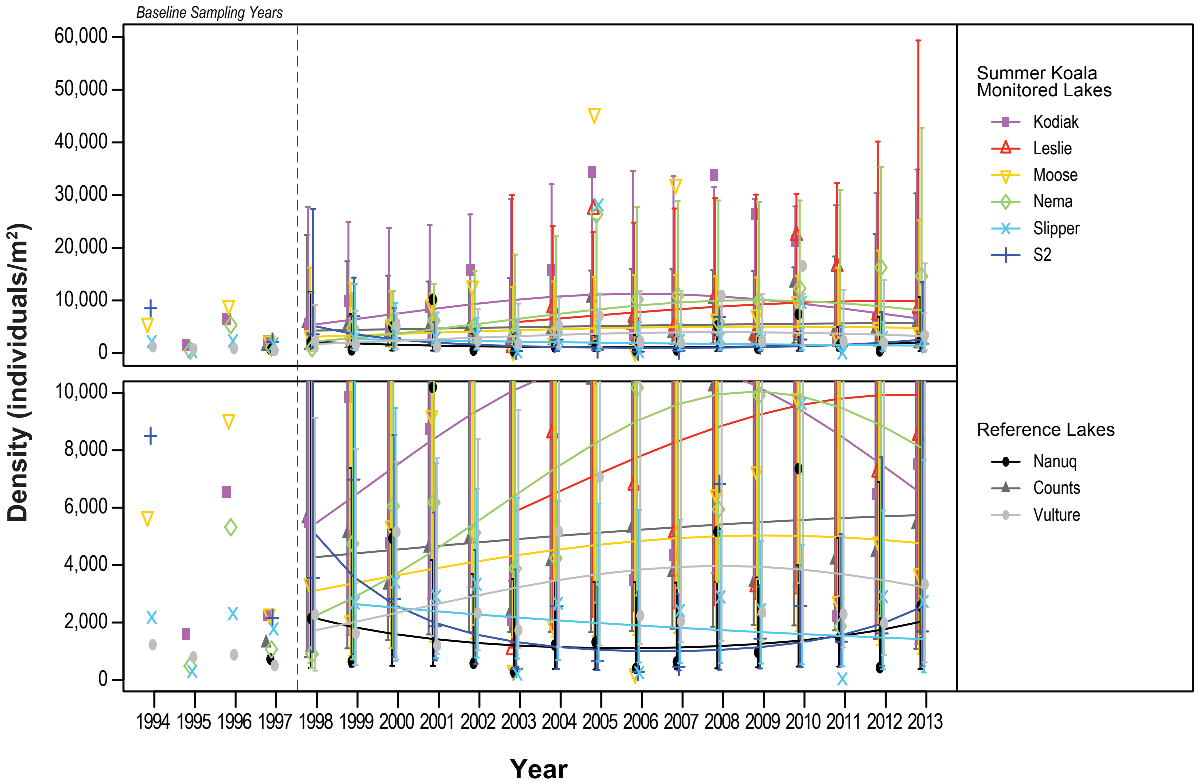
Units are organisms/m^2.
Negative values were replaced with zeros.
N = number of years data were collected.
Dashes indicate data not available.

Dipteran Diversity

Statistical analyses were not performed on the dipteran diversity datasets because the calculation of indices can result in data abnormalities that prevent statistical analysis. Consequently, graphical analyses of temporal trends in diversity indices (Figure 3.3-18) and best professional judgment were the primary methods used in the evaluation of effects. In addition, the average and relative densities of taxa were examined using graphical analyses to identify potential changes in community composition (Figures 3.3-19 to 3.3-22).

Both Shannon and Simpson's diversity indices have varied considerably through time in both monitored and reference lakes since monitoring began (Figure 3.3-18). While the variability makes it difficult to discern temporal trends, it also suggests that diversity tends to fluctuate consistently through time in all lakes (Figure 3.3-18). Simpson's diversity was within ± 2 SD of mean baseline diversities in all monitored lakes, while Shannon diversity was elevated in Kodiak and Moose lakes and at site S2 in Lac de Gras in 2013 (Table 3.3-15). Mean Shannon and Simpson's diversities were also greater than mean baseline densities ± 2 SD in all three reference lakes in 2013, except Counts Lake, in which only Shannon diversity was greater (Table 3.3-15). Therefore, no mine effects were detected with respect to dipteran diversity in lakes of the Koala watershed or Lac de Gras.

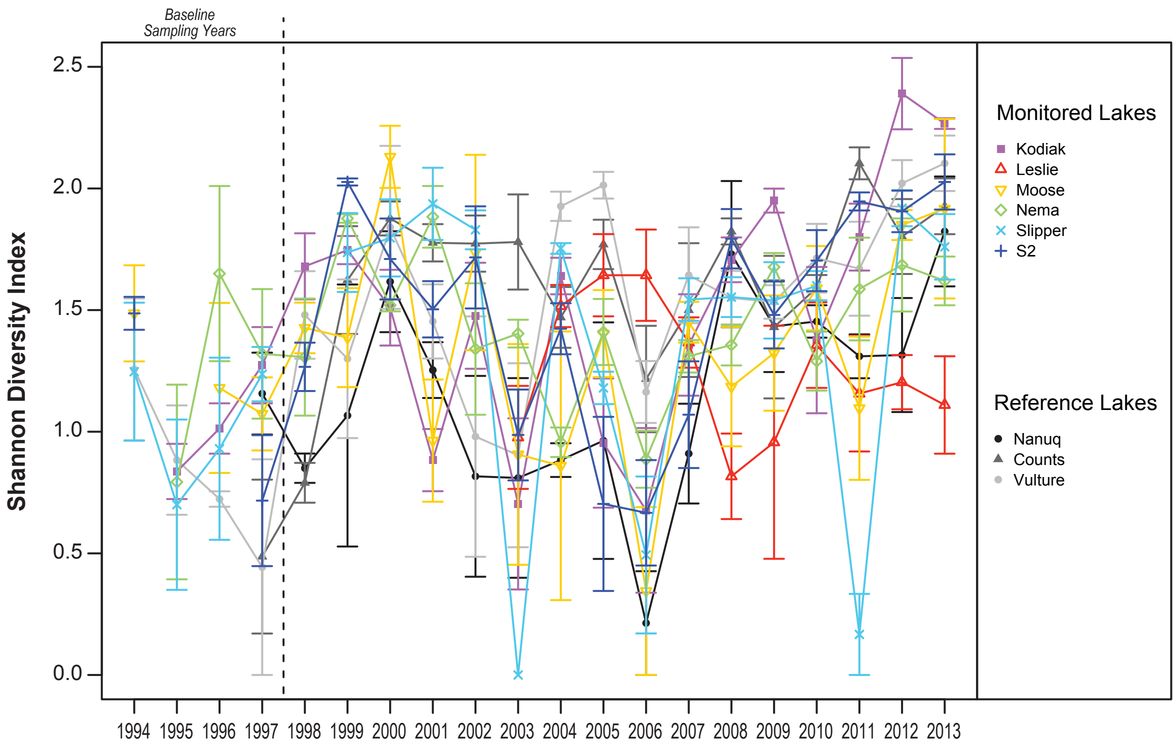
Observed and Fitted Means for Benthos Densities in Koala Watershed Lakes and Lac de Gras, August 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.

Figure 3.3-18

Average Diversity Indices for Benthic Dipterans in Koala Watershed Lakes and Lac de Gras, August 1994 to 2013



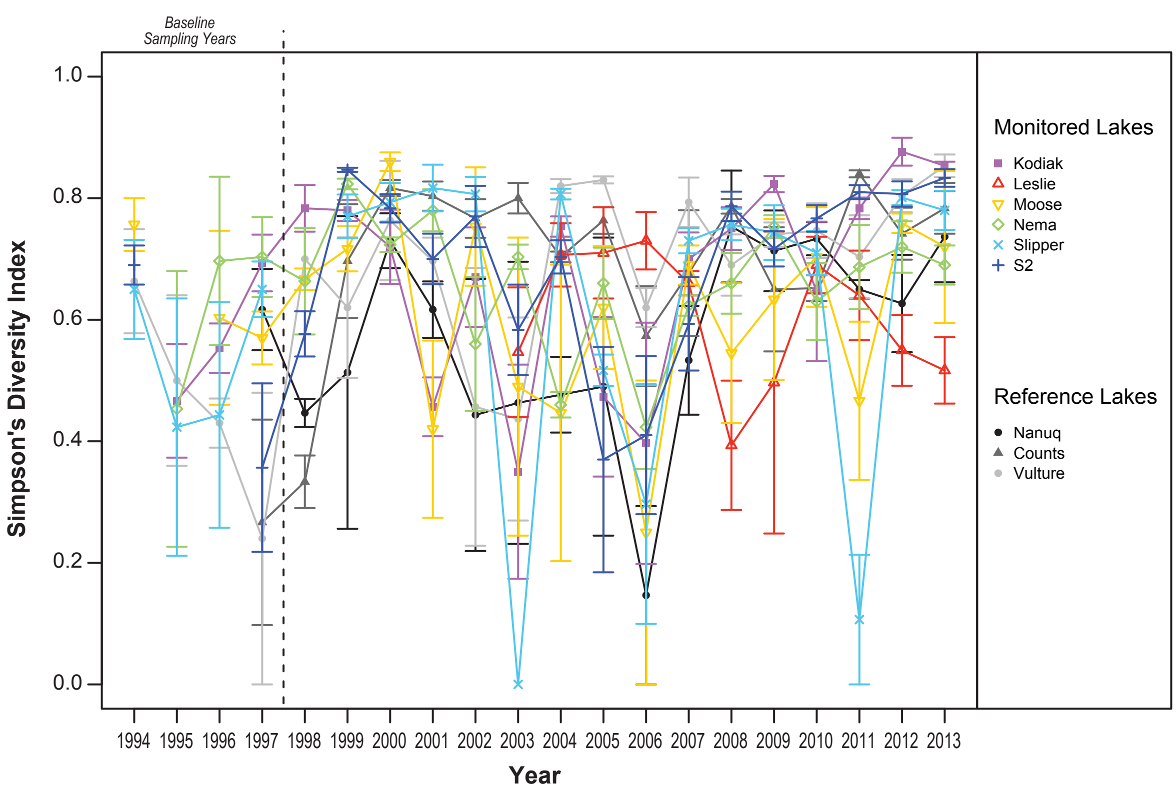


Figure 3.3-19

Average Diptera Density by Taxonomic Group for AEMP Reference Lakes, 1994 to 2013

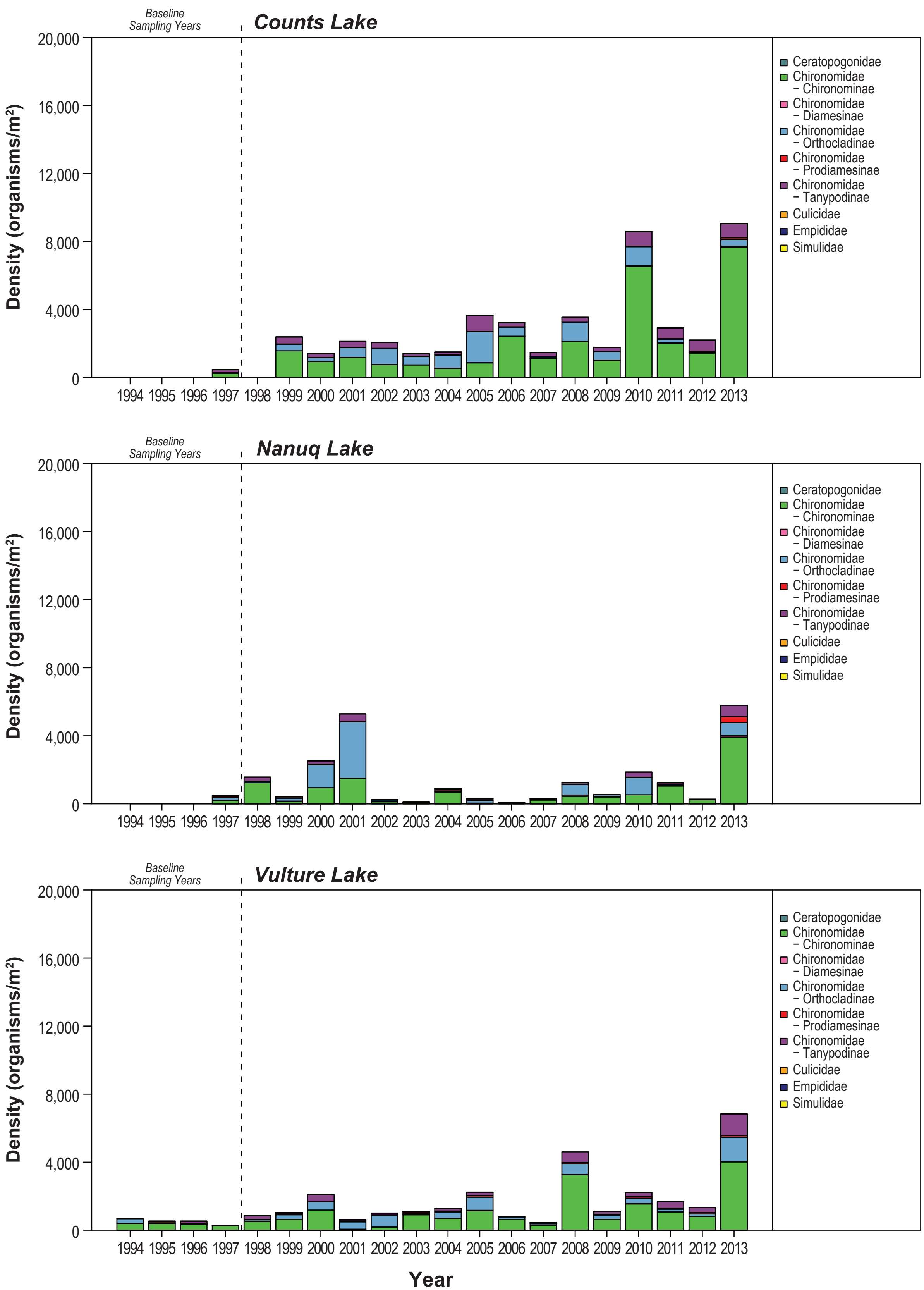


Figure 3.3-20a

Average Diptera Density by Taxonomic Group for Lakes of the Koala Watershed and Lac de Gras, 1994 to 2013

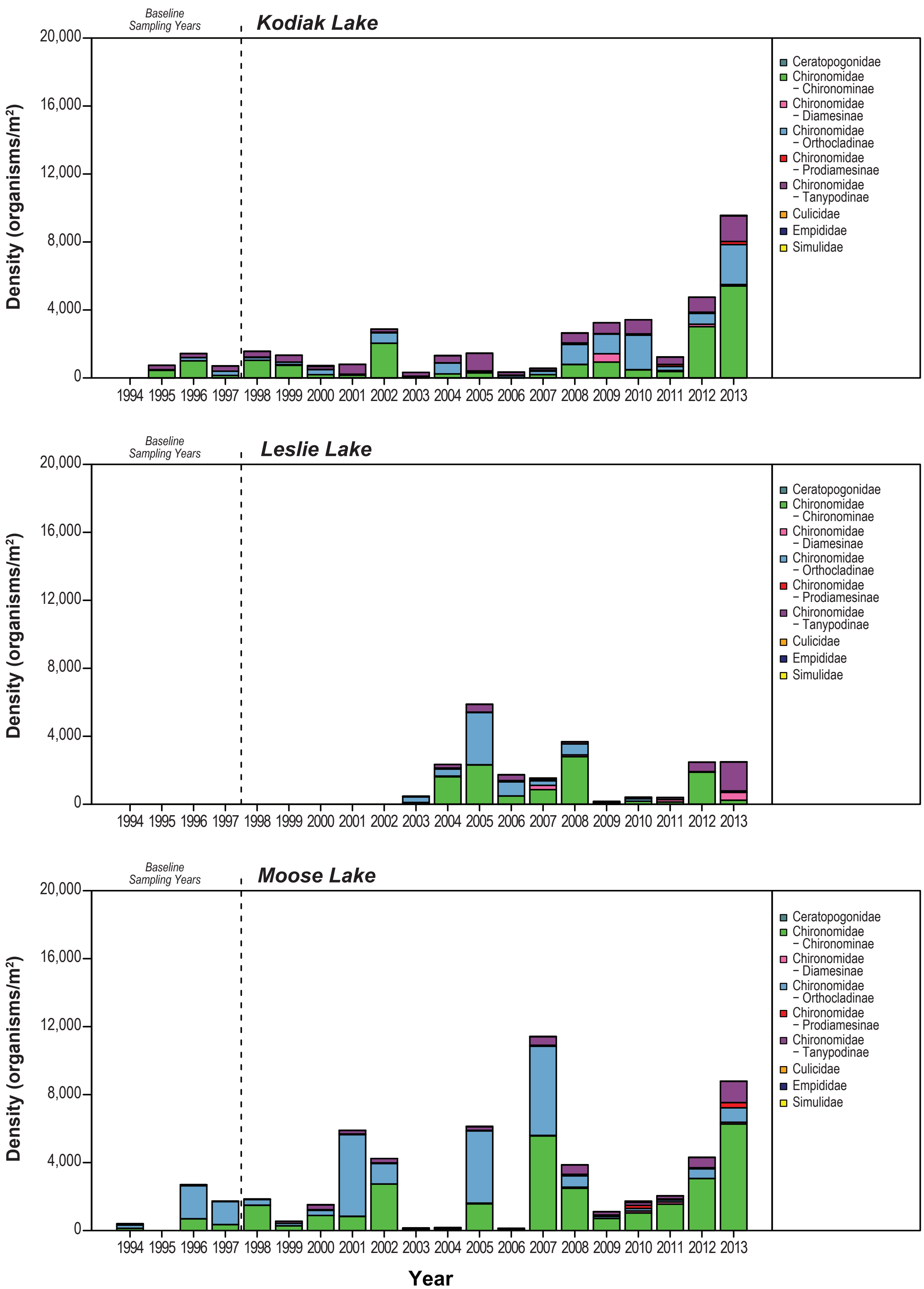


Figure 3.3-20b

Average Diptera Density by Taxonomic Group for Lakes of the Koala Watershed and Lac de Gras, 1994 to 2013

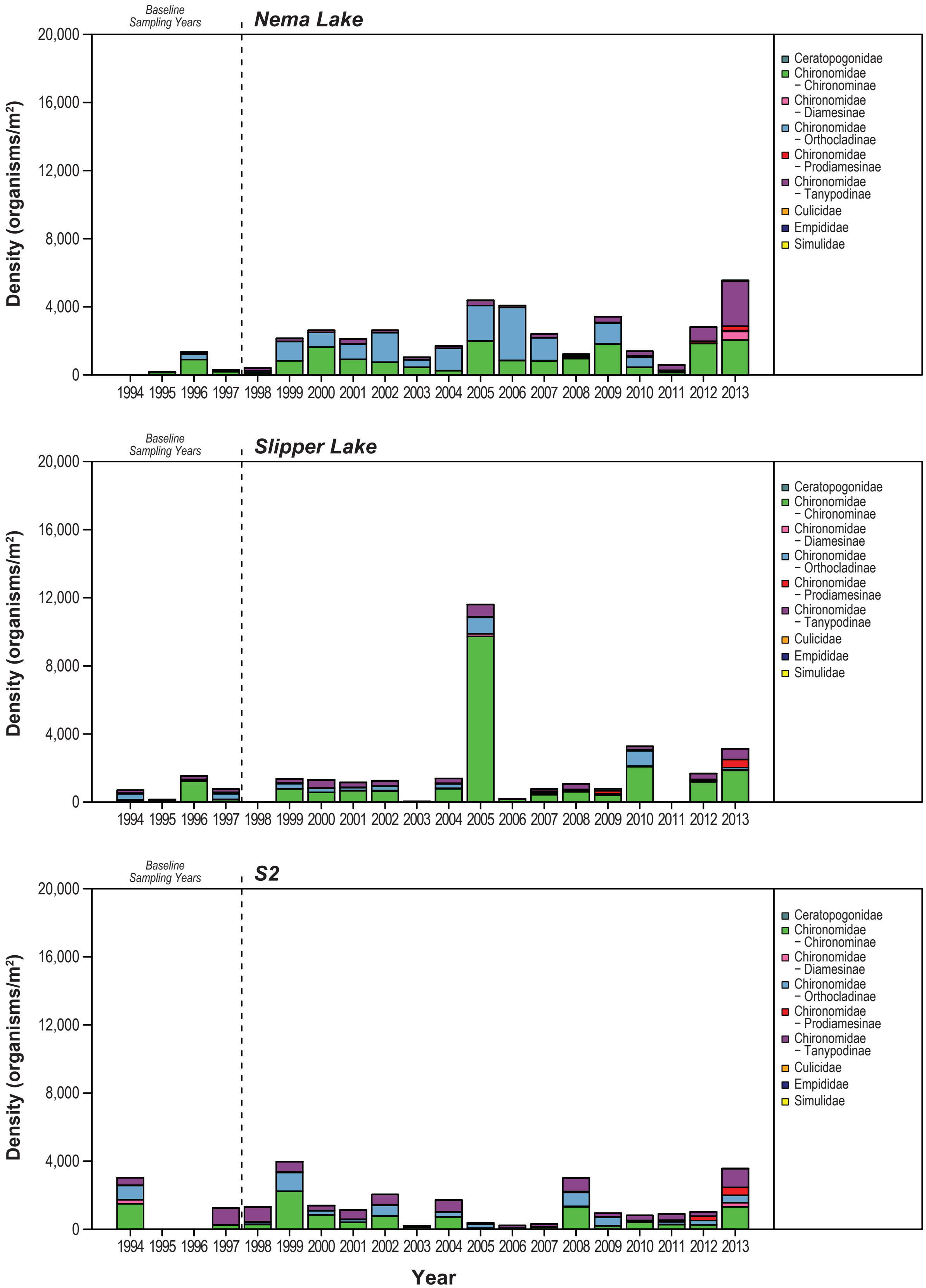


Figure 3.3-21

Relative Densities of Diptera Taxa in AEMP Reference Lakes, 1994 to 2013

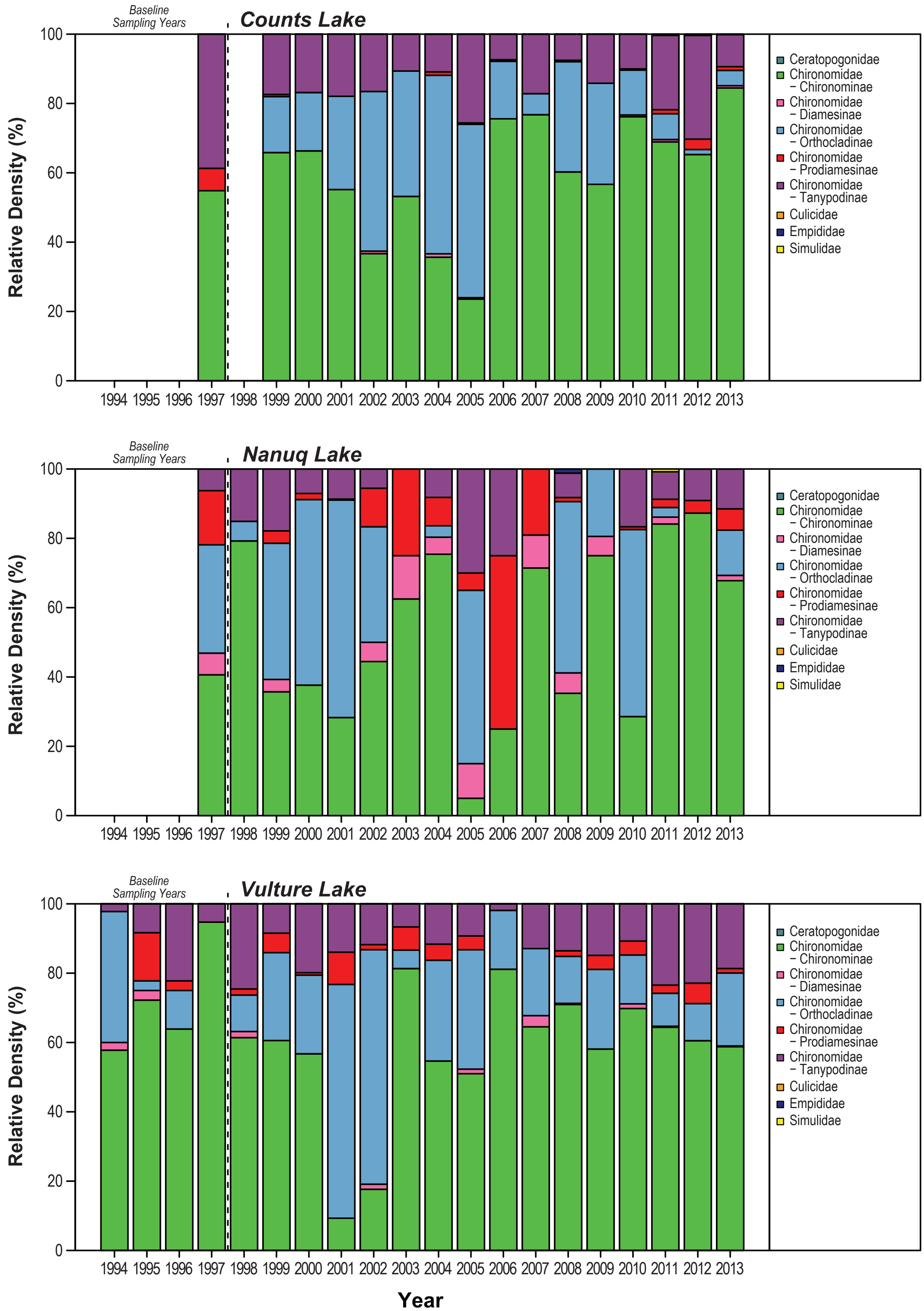
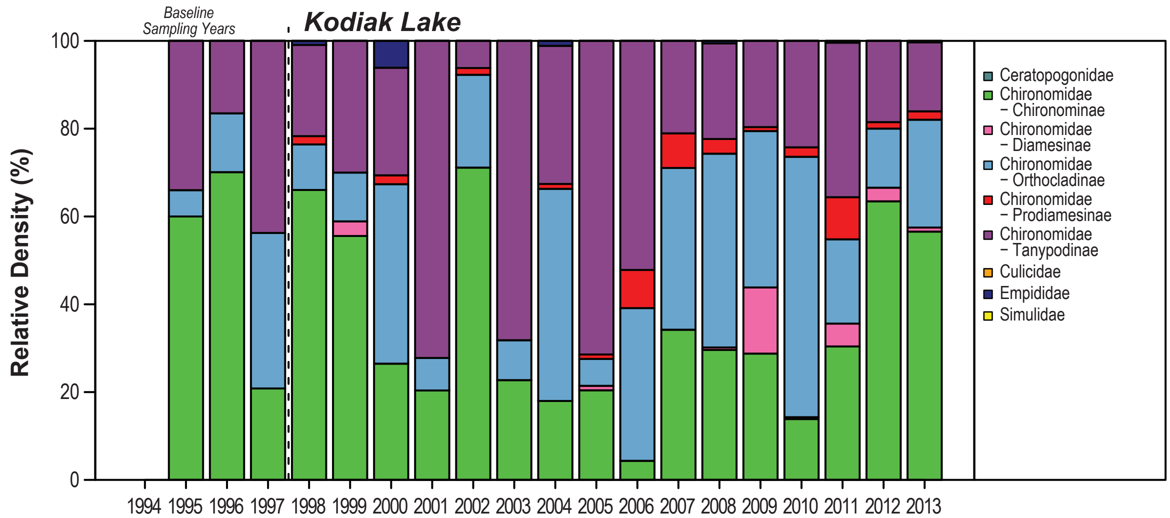
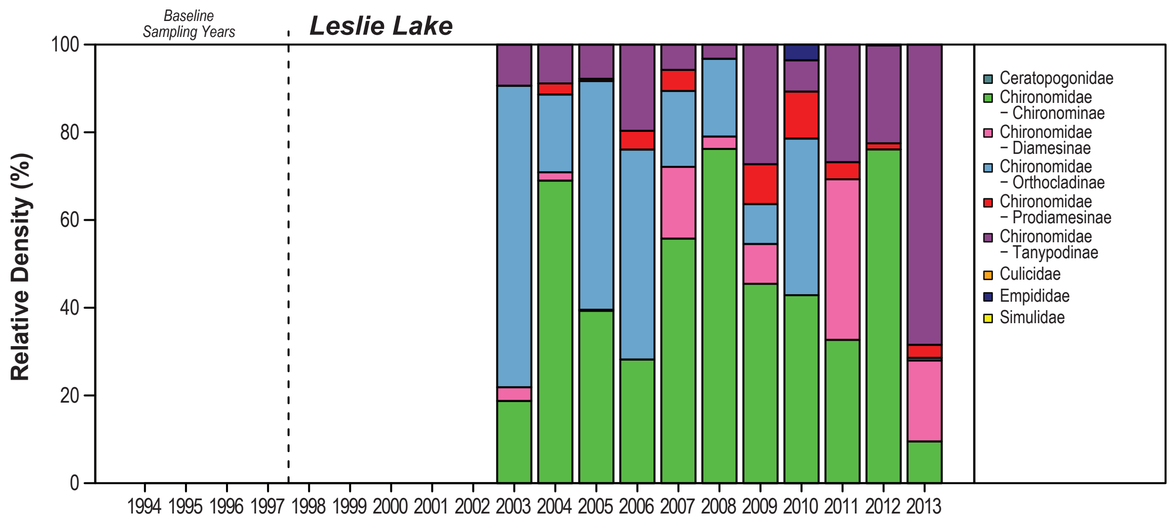


Figure 3.3-22a

Relative Densities of Diptera Taxa in Lakes of the Koala Watershed and Lac de Gras, 1994 to 2013





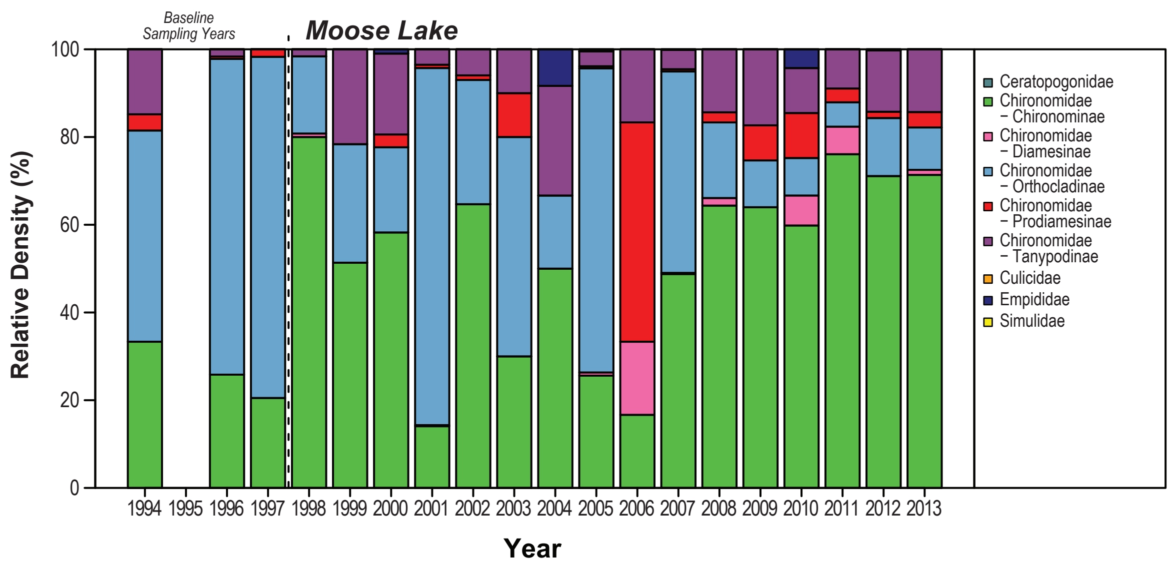


Figure 3.3-22b

Relative Densities of Diptera Taxa in Lakes of the Koala Watershed and Lac de Gras, 1994 to 2013

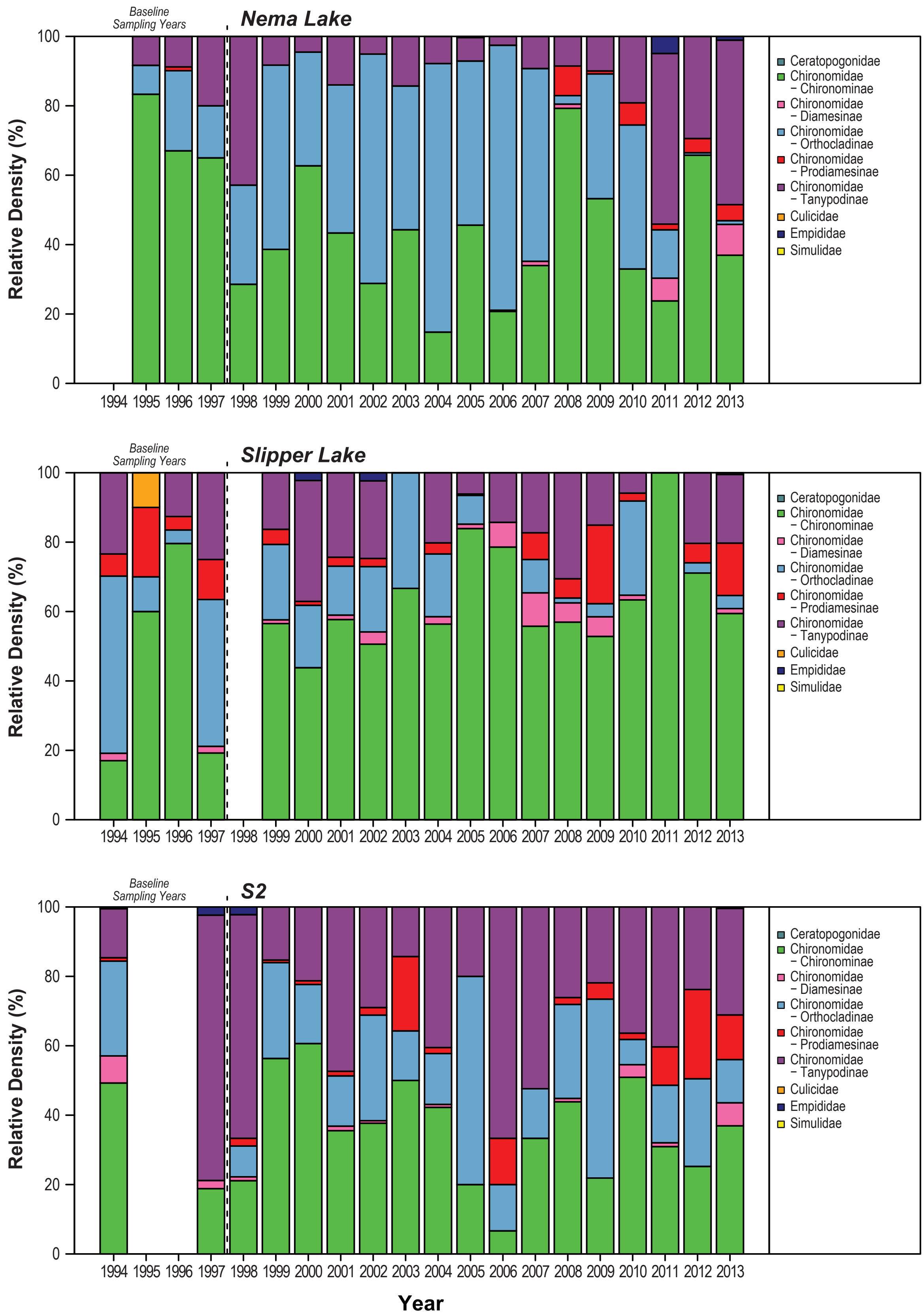


Table 3.3-15. Mean ± 2 Standard Deviations (SD) Baseline Dipteran Diversity in Each of the Koala Watershed Lakes and Lac de Gras

	Shannon Diversity			Simpson's Diversity		
Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD
Nanuq	1.16 (1)	0.57 – 1.75	1.82 ± 0.39	0.39 (1)	0.28 – 0.51	0.74 ± 0.13
Counts	0.49 (1)	0 – 1.59	1.93 ± 0.20	0.27 (1)	0 – 0.86	0.78 ± 0.07
Vulture	0.44 (4)	0 – 1.49	2.10 ± 0.20	0.24 (4)	0.61 – 0.78	0.85 ± 0.03
Kodiak	1.27 (3)	0.73 – 1.81	2.27 ± 0.04	0.69 (3)	0.42 – 0.97	0.85 ± 0.01
Leslie	-	-	1.11 ± 0.35	-	-	0.52 ± 0.09
Moose	1.07 (3)	0.24 – 1.90	1.92 ± 0.64	0.57 (3)	0.25 – 0.89	0.72 ± 0.22
Nema	1.32 (3)	0.04 – 2.60	1.62 ± 0.17	0.70 (3)	0.17 – 1.0	0.69 ± 0.06
Slipper	1.24 (4)	0.23 – 2.24	1.76 ± 0.23	0.65 (4)	0 – 1.0	0.78 ± 0.06
S2	0.72 (2)	0 – 1.76	2.03 ± 0.20	0.36 (2)	0 – 0.83	0.83 ± 0.03

Negative values were replaced with zeros.
For Simpson's diversity, upper confidence intervals >1 were replaced with a value of 1 (i.e., the maximum possible value for Simpson's diversity).
N = number of years data were collected.
- indicates not applicable.

In general, most of the dipteran taxa present belong to the family Chironomidae (Figures 3.3-19 to 3.3-22). Chironomidae are often found in large numbers in freshwater systems as they have a variety of adaptations that allow them to live in a wide variety of environments. The subfamily Chironominae is a particularly diverse and abundant group (Thorp and Covich 2001), while Diamesinae and Orthocladiinae are adapted to cold water environments (Kravtsova 2000).

Graphical analyses suggest that the relative densities of dipteran taxonomic groups have changed through time in Leslie and Moose lakes (Figures 3.3-20a and 3.3-22a). Specifically, the relative densities of Orthocladiinae have decreased, while densities of Diamesinae, Prodiamesinae, and Chironominae have increased through time (Figures 3.3-20a and 3.3-22a). These patterns are consistent with those that were first identified through the multivariate analyses conducted as part of the 2012 AEMP Re-evaluation (Rescan 2012d). In addition, graphical analyses in 2013 suggest that densities of Orthocladiinae in Nema Lake have also decreased, with a coincidental increase in densities of Tanypodinae, and that densities of Prodiamesinae have recently increased at site S2 in Lac de Gras (Figures 3.3-21 and 3.3-24). Although these patterns were not generally observed in reference lakes, graphical analyses in 2013 revealed a more recent trend of decreasing densities of Orthocladiinae with increasing densities of Chironominae in Counts Lake (Figures 3.3-19 and 3.3-20).

Taxonomic data was examined at a finer resolution to determine if abundances of specific genera could explain changes in the relative densities of the Chironomidae subfamilies. In general, it was difficult to detect clear temporal trends at the genera level and some of the trends that were observed in the 2012 AEMP were less apparent this year. Difficulty in discerning trends is due in part to the large amount of variability in genera densities through time and the fact that there are many genera with low abundances and that are often completely absent in any given year. Despite this variability, examination of the data at the genus level may suggest the following patterns:

- The decrease in Orthocladiinae may be related to declines in the density of organisms from the genus *Rheocricotopus* in Moose Lake and *Psectrocladius* in Leslie Lake. However, organisms from these genera have only been previously observed on two occasions in each of these lakes.

In Nema Lake, the decrease in Orthocladiinae may also be related to declines in the density of *Psectrocladius*: organisms from this genus were frequently observed prior to 2002, with only two occurrences in more recent years;

- The increase in Chironominae may be due to recent increases in *Cladotanytarsus* in Leslie and Moose lakes and *Stempellinella* in Moose Lake. In Counts Lake, the increase in Chironominae seems more likely related to recent increases in *Corynocera* and *Stictochironomus*;
- The increase in Prodiamesinae in Leslie and Moose lakes and site S2 in Lac de Gras appears to be related to increases in the density of organisms from the genus *Monodiamesa*;
- The increase in Diamesinae in Leslie and Moose lakes appears to be related to increases in the density of organisms from the genus *Protanypus*; and
- The increase in Tanypodinae in Nema Lake appears related to an overall increase in the density of organisms from the genus *Procladius* over time, as well as a recent increase in organisms of the genus *Ablabesmyia*.

Unfortunately, little information is available on the ecology of these groups and, therefore, the cause of these shifts is unclear. However, results of the 2012 AEMP re-evaluation suggest that changes in the absolute quantities or relative availability of macronutrients like nitrogen and phosphorus are the most likely underlying cause of change in biological communities at the Ekati Diamond Mine rather than the relative sensitivities of different species to changes in water chemistry (see Section 3.3.5; Rescan 2012d).

3.3.4 Stream Benthos

3.3.4.1 Variables

Stream benthos are organisms that live in association with stream sediments. They provide an important source of food for many species of fish. Dipterans (flies) tend to dominate benthic invertebrate communities and are widely used as indicators of ecosystem health, including sediment quality. Organisms from the orders Ephemeroptera, Plecoptera, and Trichoptera (EPT) are also widely used as indicators of stream health because they are often sensitive to disturbance and various sources of pollution. Thus, stream benthos density (organisms/m^2) and dipteran and EPT diversity (Shannon and Simpson's diversity indices) were evaluated for potential mine effects.

3.3.4.2 Dataset

Stream benthos samples have been collected over a one month period from early August to early September of each year since 1995 (Table 3.3-16). Five replicates were collected from each stream in 1995 and between 1999 and 2013, except in Kodiak-Little in 1999 when only three replicates were collected. In 1997 and 1998, triplicate samples were collected from each stream. Although stream benthos samples were collected in 2010, they were not included in the evaluation of effects as a result of laboratory error. Baseline data, which were collected between 1994 and 1997, were not used in the statistical evaluation of effects but are included in Tables 3.3-16 and shown graphically for visual comparison, below.

3.3.4.3 Results and Discussion

<u>Density</u>

Statistical and graphical analyses indicate that benthos density in monitored streams has been stable through time (Table 3.3-17; Figure 3.3-23). Moreover, mean stream benthos density in 2013 did not differ from mean baseline density ± 2 SD in any reference or monitored stream (Table 3.3-18). No mine effects were detected with respect to stream benthos density.

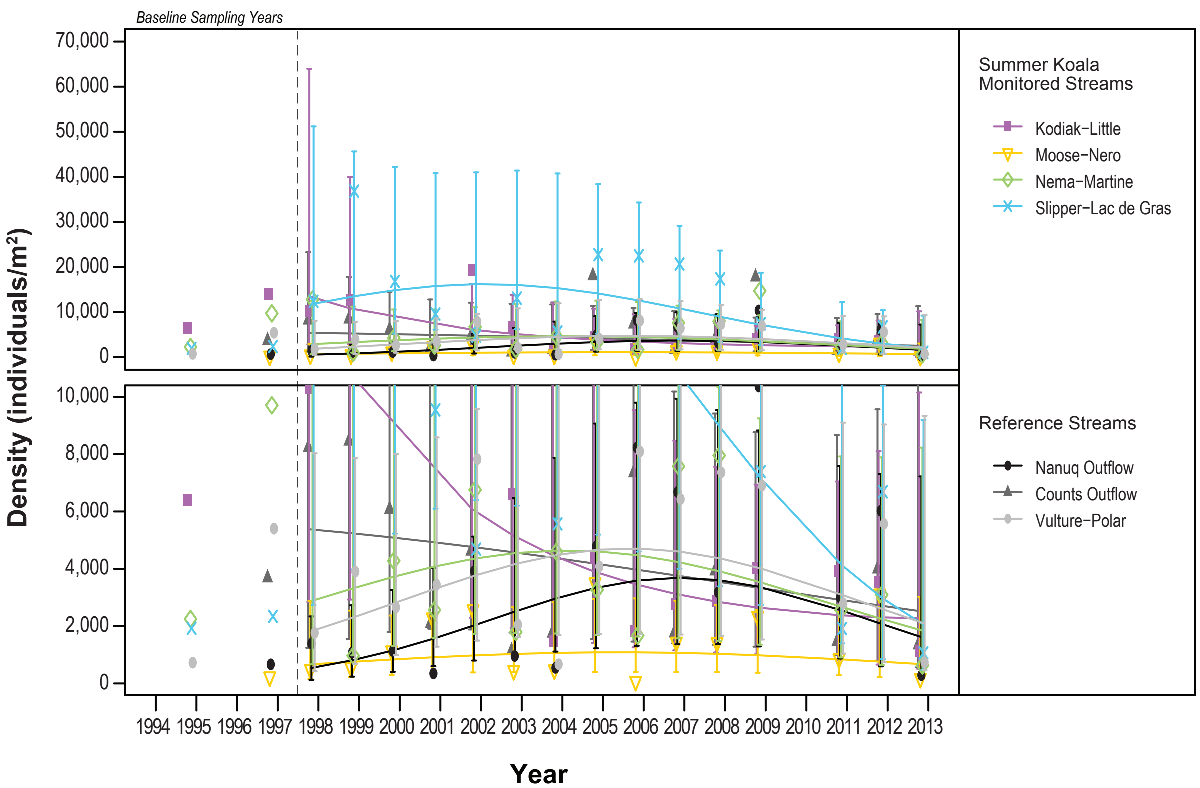
Table 3.3-16. Dataset Used for Evaluation of Effects on Benthos in Koala Watershed Streams

Year	Nanuq Outflow	Counts Outflow	Vulture-Polar	Kodiak-Little	Moose-Nero	Nema-Martine	Slipper-Lac de Gras
1994	-	-	-	-	-	-	-
1995*	-	-	Aug 10 - Sept 14	Aug 8 - Sept 14	-	Aug 10 - Sept 13	Aug 10 - Sept 13
1996*	-	-	-	-	-	-	-
1997*	Aug 10 - Sept 14	Aug 1 - Sept 7	Aug 10 - Sept 14	Jul 31 - Sept 6	Aug 31 - Sept 5	Jul 30 - Sept 3	Jul 30 - Sept 4
1998	Jul 30 - Aug 31	Jul 30 - Aug 31	Jul 30 - Aug 31	Jul 27 - Aug 26	Jul 30 - Sept 1	Jul 30 - Sep 1	Jul 30 -Sept 1
1999	Jul 28 - Aug 28	Jul 28 - Aug 28	Jul 28 - Aug 28	Jul 27 - Aug 27	Jul 28 - Aug 29	Jul 28 - Aug 29	Jul 28 - Aug 2
2000	Jul 28 - Aug 29	Jul 28 - Aug 29	Jul 28 - Aug 29	-	Jul 28 - Aug 30	Jul 28 - Aug 30	Jul 28 - Aug 29
2001	Jul 28 - Aug 29	Jul 28 - Aug 29	Jul 28 - Aug 29	-	Jul 28 - Aug 30	Jul 28 - Aug 30	Jul 28 - Aug 30
2002	Jul 31 - Aug 31	Jul 31 - Aug 31	Jul 31 - Aug 31	Jul 31 - Aug 31	Jul 31 - Aug 31	Jul 31 - Aug 31	Jul 31 - Aug 31
2003	Aug 1 - Sept 6	Aug 1 - Sept 6	Aug 1 - Sept 6	Aug 1 - Sept 6	Aug 1 - Sept 6	Aug 1 - Sep 6	Aug 1 - Sept 6
2004	Aug 11 - Sept 12	Aug 11 - Sept 12	Aug 11 - Sept 10	Aug 11 - Sept 10	Aug 11 - Sept 10	Aug 11 - Sept 10	Aug 11 - Sept 12
2005	Aug 2 - Sept 3	Aug 2 - Sept 3	Aug 2 - Sept 3	Aug 2 - Sept 5	Aug 2 - Sept 5	Aug 2 - Sept5	Aug 2 - Sept 5
2006	Jul 26 - Sept 1	Jul 27 - Sept 1	Jul 27 - Sept 4	Jul 29 - Sept 4	Jul 27 - Sept	Jul 27 - Sept 4	Jul 28 - Sept 3
2007	Aug 3 - Sept 1	Aug 3 - Aug 31	Aug 4 - Sept 3	Aug 5 - Aug 31	Aug 3 - Sept 3	Aug 3 - Sept 1	Aug 3 - Sept 3
2008	Aug 2 - Sept 4	Aug 1 - Sept 4	Aug 2 - Sept 6	Aug 1 - Sept 3	Aug 1 - Sept 3	Aug 3 - Sept 3	Aug 1 - Sept 3
2009	Aug 3 - Sept 4	Aug 3 - Sept 4	Aug 4 - Sept 4	Aug 5 - Sept 6	Aug 3 - Sept 6	Aug 5 - Sept 6	Aug 4 - Sept 6
2010†	-	-	-	-	-	-	-
2011	Jul 30 - Aug 30	Jul 30 - Aug 30	Jul 31 - Aug 31	Jul 31 - Aug 31	Jul 30 - Aug 31	Jul 30 - Aug 31	Jul 30 - Aug 31
2012	Aug 4 - Sept 1	Aug 5 - Aug 31	Aug 4 - Sept 1	Aug 4 - Sept 1	Aug 5 - Sept 1	Aug 4 - Sept 1	Aug 4 - Aug 31
2013	Aug 4 - Sept 3	Aug 4 - Sept 3	Aug 4 - Sept 3	Aug 7 - Sept 4	Aug 7 - Sept 4	Aug 7 - Sept 4	Aug 7 - Sept 4

** = Data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison.*
† Data were collected, but were discarded due to problems with laboratory analyses.
Dashes indicate no data were available.
Five replicates were collected from each stream in 1995 and from 1999 to 2013 except in Kodiak-Little in 1999 when only three replicates were collected.
Triplicate samples were collected in 1997 and 1998.

Figure 3.3-23

Observed and Fitted Means for Benthos Densities in Koala Watershed Streams, August 1995 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.

Table 3.3-17. Statistical Results of Benthos Density in Streams in the Koala Watershed and Lac de Gras

	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
				Model Fit = 3	Model Fit = 2	Model Fit = 1	
Density	-	LME	2	-	None	-	1-398

Dashes indicate not applicable.

Table 3.3-18. Mean ± 2 Standard Deviations (SD) Baseline Benthos Density in Each of the Koala Watershed Streams

Stream	Baseline Mean (N)	Mean Baseline Range ± 2 SD	2013 Mean ± 1 SD
Nanuq Outflow	667 (1)	166 - 1,168	289 ± 104
Counts Outflow	3,685 (1)	0 – 8,242	1,347 ± 1,619
Vulture-Polar	2,479 (2)	0 – 8,551	740 ± 688
Kodiak-Little	9,240 (2)	0 – 23,851	1,124 ± 562

Stream	Baseline Mean (N)	Mean Baseline Range ± 2 SD	2013 Mean ± 1 SD
Moose-Nero	230 (1)	61 – 399	178 ± 132
Nema-Martine	5,044 (2)	0 – 13,208	627 ± 297
Slipper-Lac de Gras	2,079 (2)	0 – 5,623	1067 ± 287

Units are organisms/m^2.
Negative values were replaced with zeros.
N = number of years data were collected.

<u>Dipteran Diversity</u>

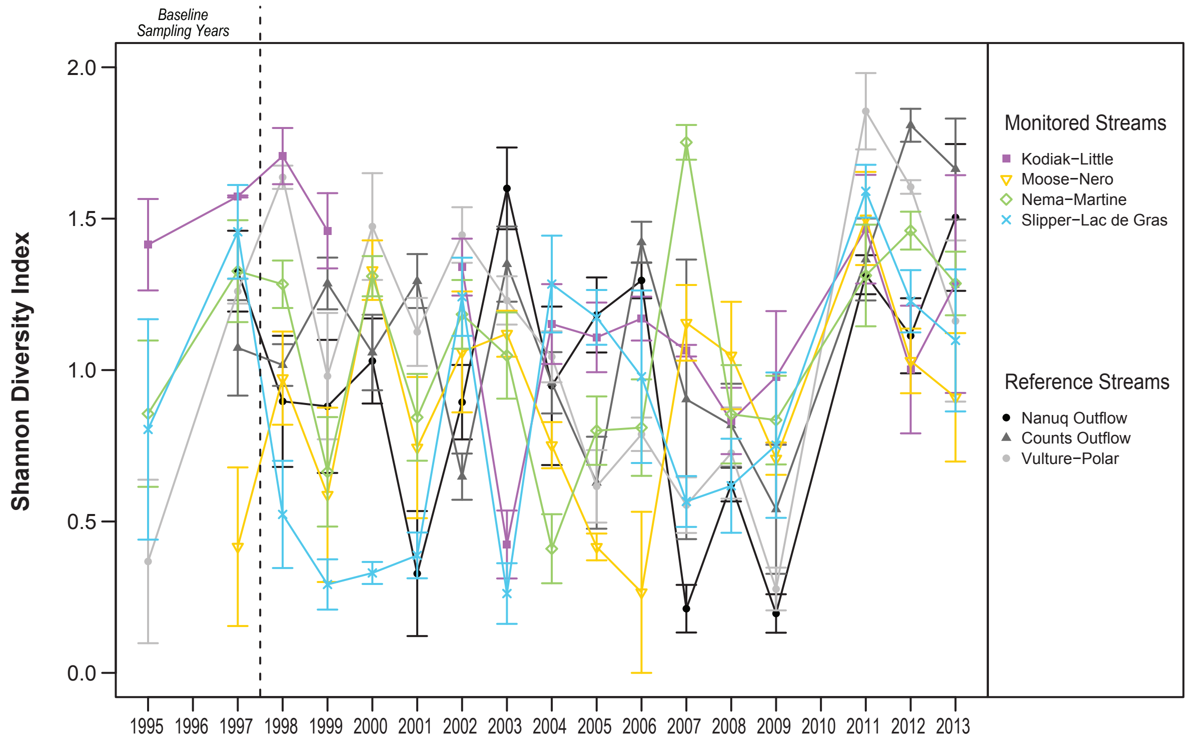
Statistical analyses were not performed on the diversity datasets because the calculation of indices can result in data abnormalities that prevent statistical analysis. Consequently, graphical analyses of temporal trends in diversity indices (Figure 3.3-24) and best professional judgment were the primary methods used in the evaluation of effects. In addition, the average and relative densities of taxa were examined using graphical analyses to identify potential changes in community composition (Figures 3.3-25 to 3.3-28).

Both Shannon and Simpson's dipteran diversity indices have varied considerably through time in both monitored and reference streams since monitoring began (Figure 3.3-24). While the variability makes it difficult to discern temporal trends, Shannon and Simpson's dipteran diversity have not shown signs of directed change through time in any of the monitored or reference streams (Figure 3.3-24). In 2013, mean Shannon and Simpson's dipteran diversities were within ± 2 SD of mean baseline diversities in all cases except Counts Outflow, in which Shannon diversity was greater (Table 3.3-19).

The relative densities of dipteran taxonomic groups have been fairly consistent through time in all monitored and reference streams (Figures 3.3-25 to 3.3-28). However, there was some evidence of a trend toward relatively greater densities of organisms from the sub-family Orthocladiinae and lesser densities of organisms from the sub-family Chironominae through time (Figures 3.3-25 to 3.3-28). This trend was apparent in both reference and monitored streams, which suggests that the trend may result from broader climatic patterns, phenological drift (i.e. changes in the timing of seasonal emergence), or systematic changes in identification or enumeration through time (Figures 3.3-25 to 3.3-28). In general, most of the dipteran taxa present belong to the family Chironomidae. Chironomidae are often found in large numbers in freshwater systems as they have a variety of adaptations that allow them to live in a wide variety of environments. The subfamily Chironominae is a particularly diverse and abundant group (Thorp and Covich 2001), while Orthocladiinae are adapted to cold water environments (Kravtsova 2000). No mine effects were detected with respect to dipteran diversity or taxonomic composition in monitored streams.

Figure 3.3-24

Average Diversity Indices for Benthic Dipterans in Koala Watershed Streams, August 1995 to 2013



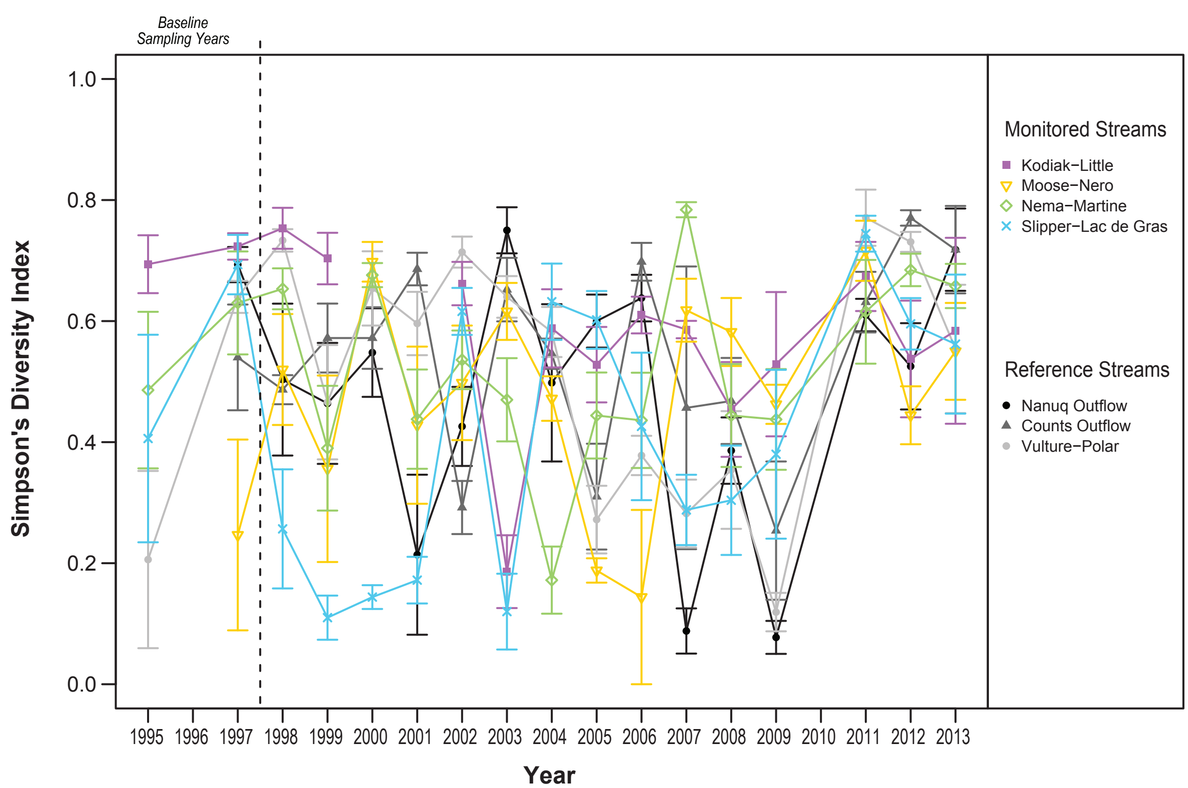


Figure 3.3-25

Average Benthic Dipteran Density by Taxonomic Group for AEMP Reference Streams, 1995 to 2013

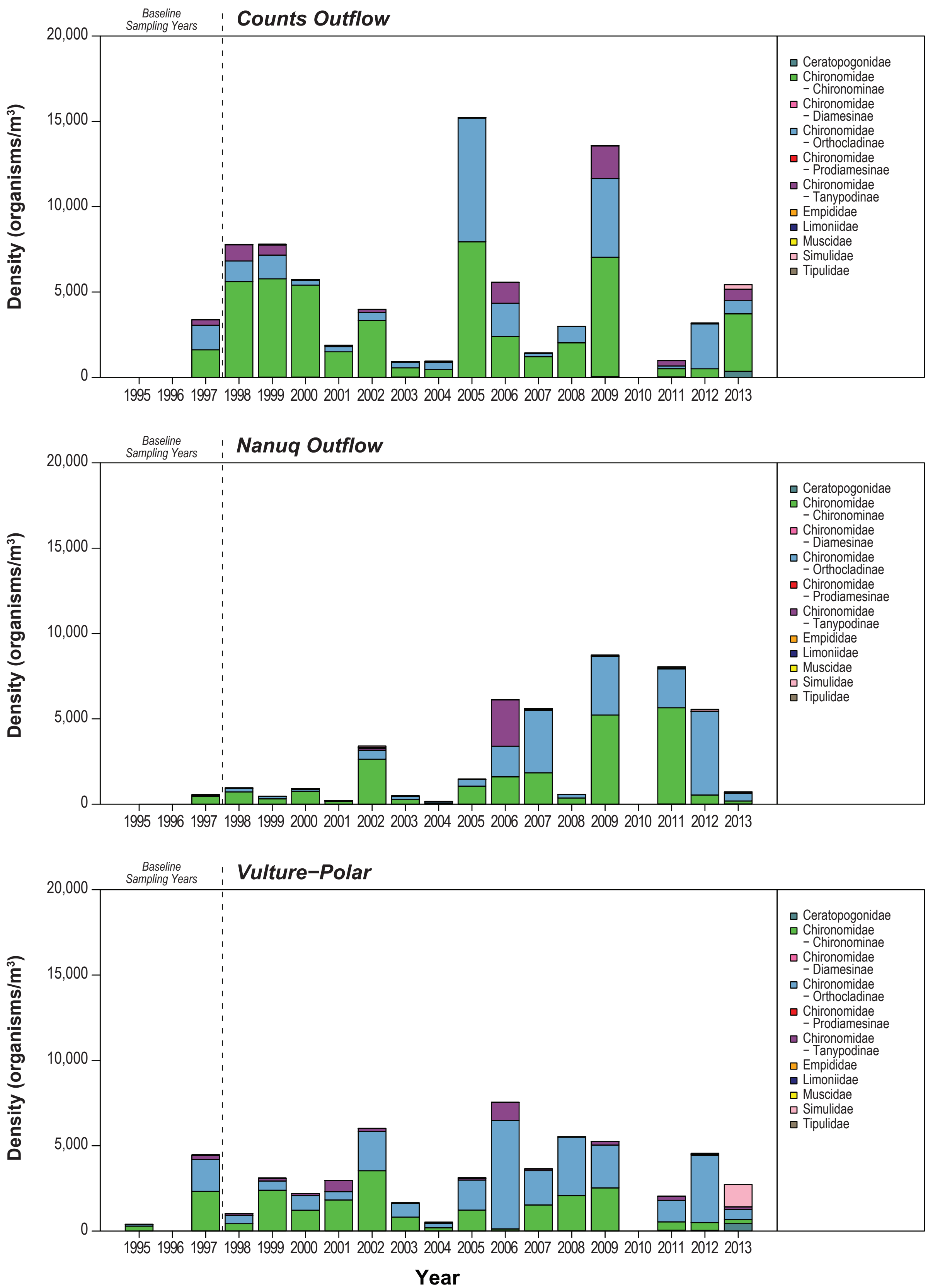


Figure 3.3-26a

Average Benthic Dipteran Density by Taxonomic Group for Streams of the Koala Watershed, 1995 to 2013

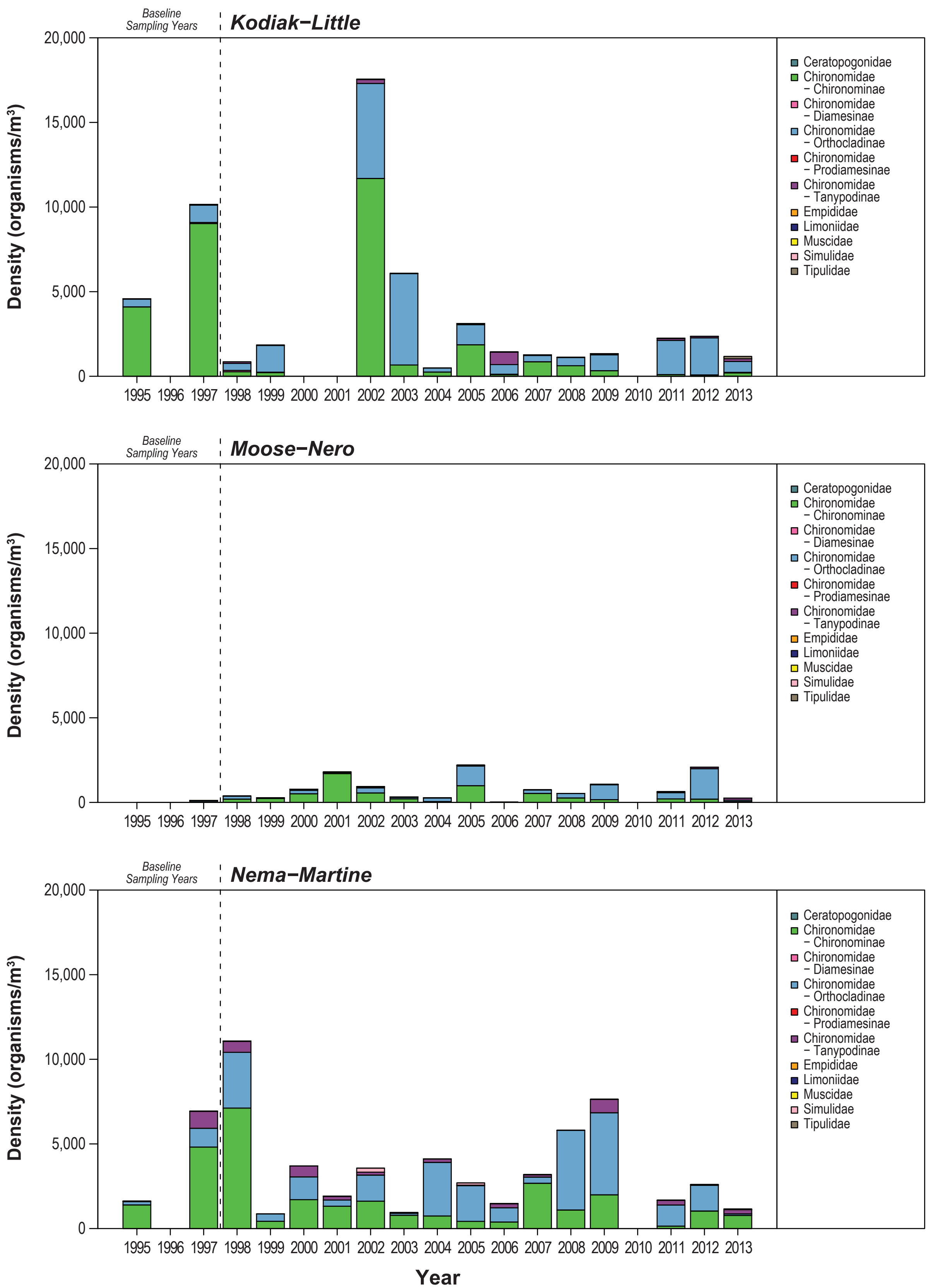
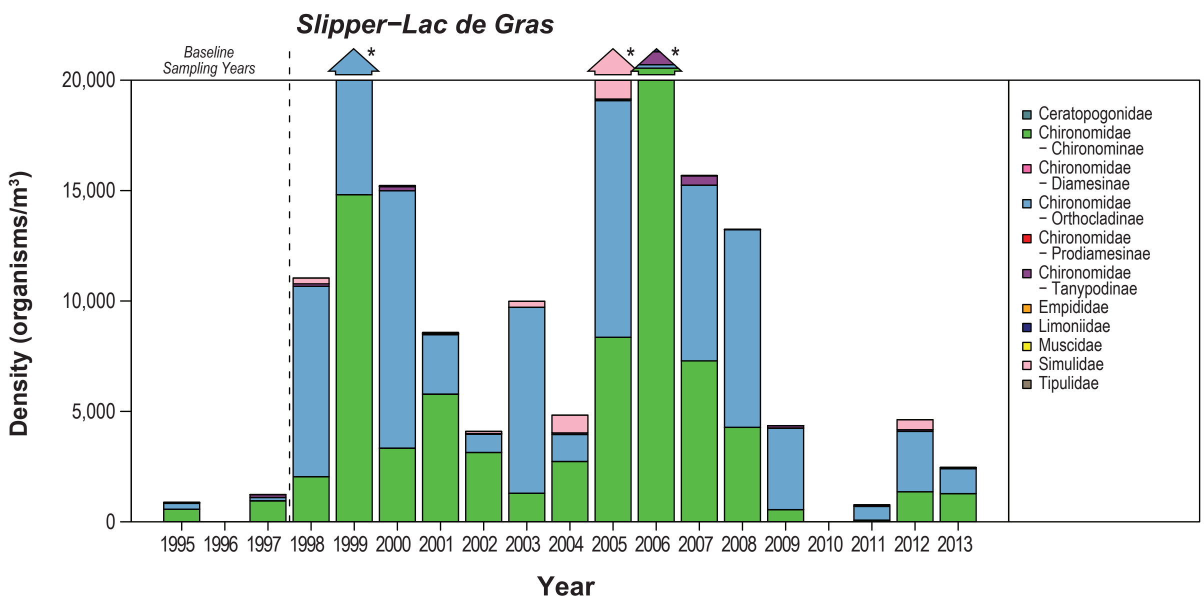


Figure 3.3-26b

Average Benthic Dipteran Density by Taxonomic Group for Streams of the Koala Watershed, 1995 to 2013



*Notes: *1999 total density = 34,948, Orthoclandinae = 20,008;*
2005 total density = 21,524, Simulidae = 2,378;
2006 total density = 21,396, Chironiminae = 20, 438, Orthocladinae = 180, Tanypodinae = 669, Limnoiidae = 67, Tiplulidae = 35

Figure 3.3-27

Relative Densities of Benthic Dipteran Taxa in AEMP Reference Streams, 1995 to 2013

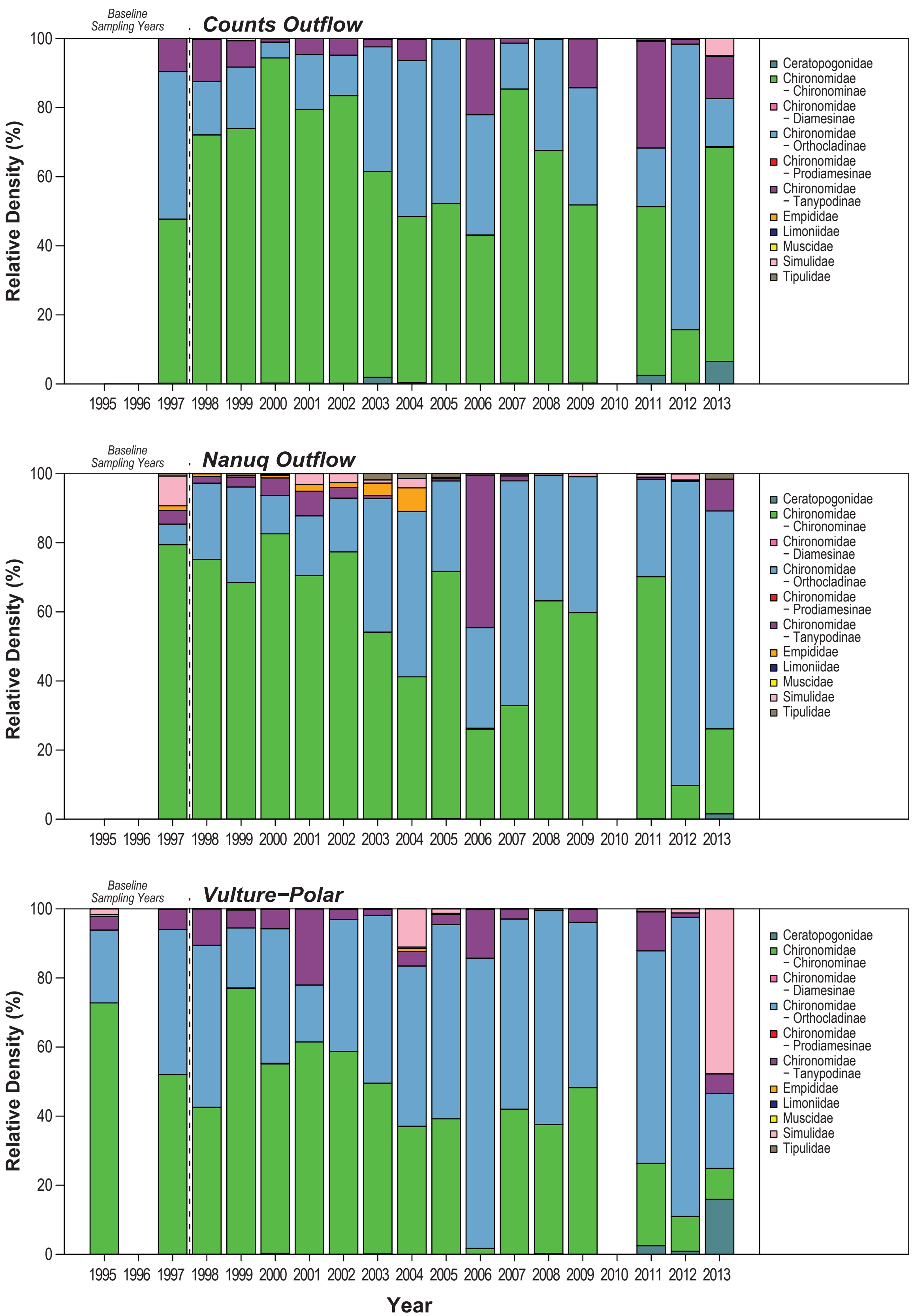


Figure 3.3-28a

Relative Densities of Benthic Dipteran Taxa in Streams of the Koala Watershed, 1995 to 2013

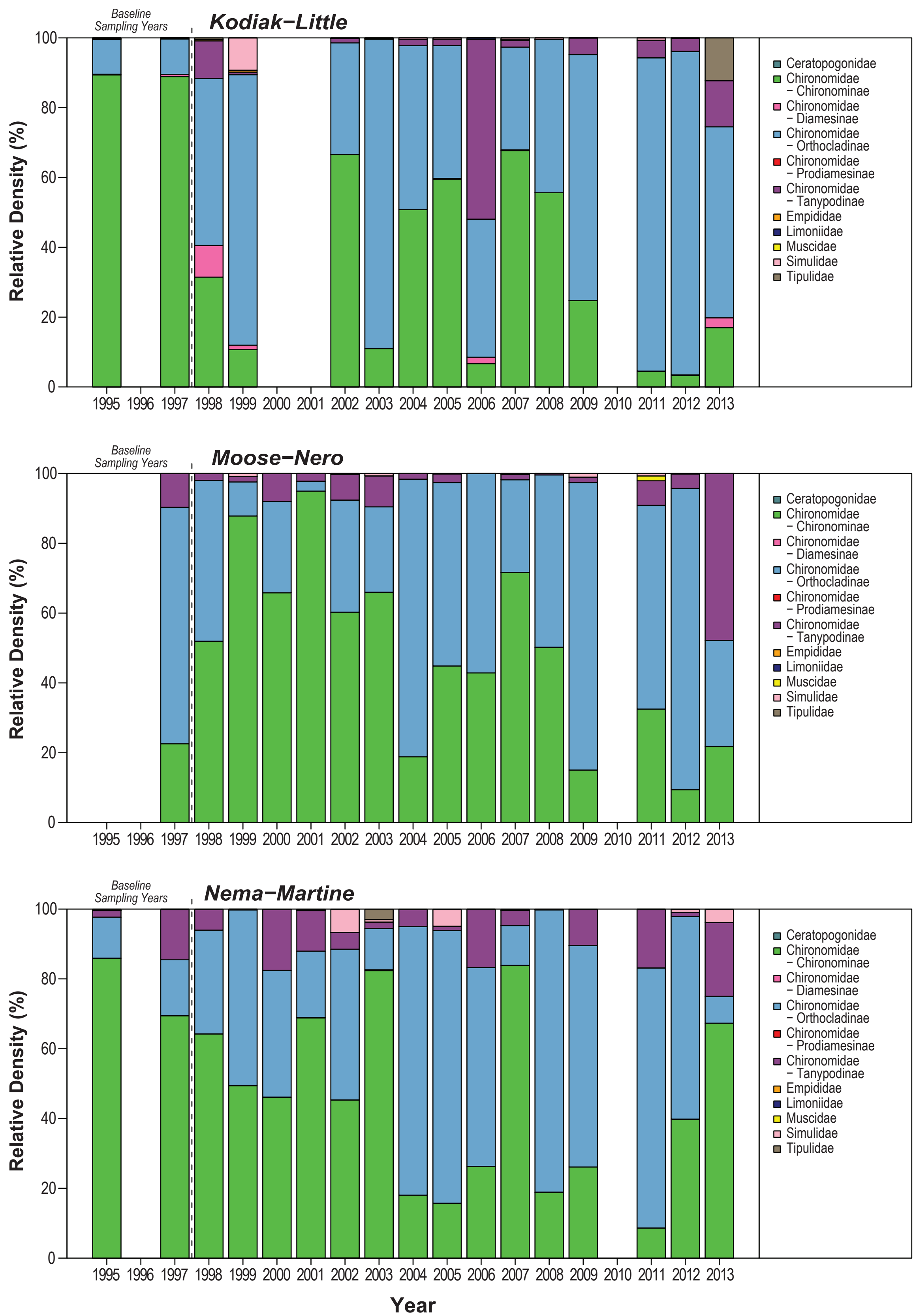


Figure 3.3-28b

Relative Densities of Benthic Dipteran Taxa in Streams of the Koala Watershed, 1995 to 2013

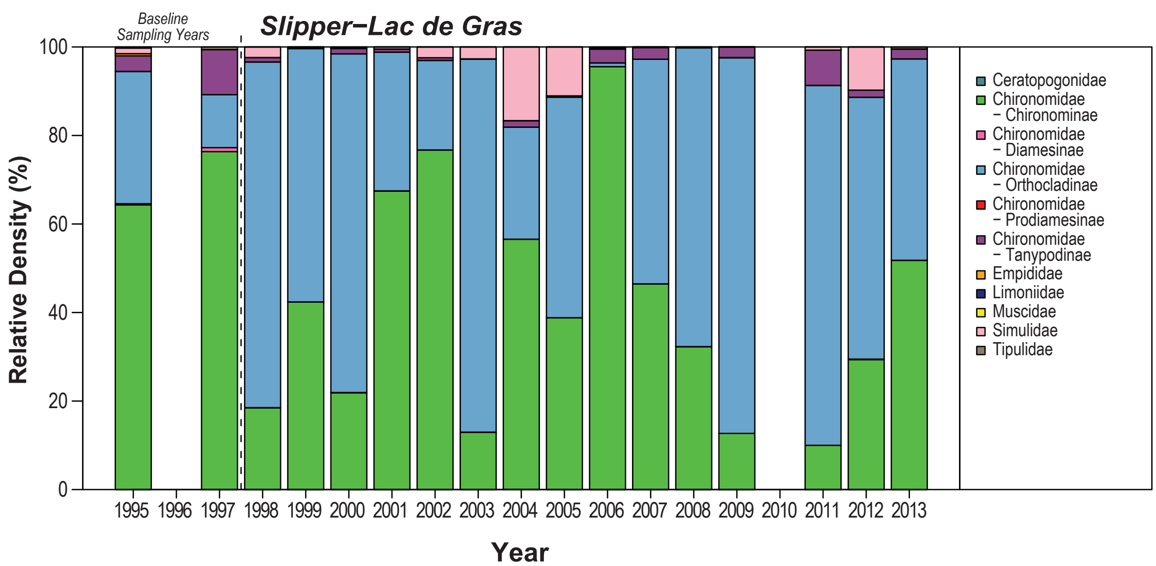


Table 3.3-19. Mean ± 2 Standard Deviations (SD) Baseline Dipteran Diversity in Each of the Koala Watershed Streams

Stream	Shannon Diversity			Simpson's Diversity		
	Baseline Mean (N)	Mean Baseline Range ± 2 SD	2013 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range ± 2 SD	2013 Mean ± 1 SD
Nanuq Outflow	1.33 (1)	0.68 - 1.79	1.50 ± 0.54	0.69 (1)	0.59 – 0.79	0.72 ± 0.15
Counts Outflow	1.07 (1)	0.53 – 1.62	1.66 ± 0.37	0.54 (1)	0.24 – 0.84	0.72 ± 0.16
Vulture-Polar	0.70 (2)	0 – 2.00	1.16± 0.60	0.37 (2)	0 - 1	0.55 ± 0.24
Kodiak-Little	1.47 (2)	0.94 – 2.01	1.28 ± 0.80	0.71 (2)	0.54 - 0.87	0.58 ± 0.34
Moose-Nero	0.42 (1)	0 – 1.32	0.91 ± 0.47	0.25 (1)	0 – 0.79	0.55 ± 0.18
Nema-Martine	1.03 (2)	0.03 – 2.03	1.29 ± 0.23	0.54 (2)	0.05 – 1	0.66 ± 0.08
Slipper-Lac de Gras	1.05 (2)	0 – 2.48	1.10 ± 0.52	0.51 (2)	0 - 1	0.56 ± 0.26

Negative values were replaced with zeros.
For Simpson's diversity, upper confidence intervals >1 were replaced with a value of 1 (i.e., the maximum possible value for Simpson's diversity).
N = number of years data were collected.

3.3.4.4 EPT Diversity

Statistical analyses were not performed on the EPT diversity datasets because the calculation of indices can result in data abnormalities that prevent statistical analysis. Consequently, graphical analyses of temporal trends in EPT diversity indices (Figure 3.3-29) and best professional judgment were the primary methods used in the evaluation of effects. In addition, the average and relative densities of taxa were examined using graphical analyses to identify potential changes in community composition (Figures 3.3-30 to 3.3-33).

Both Shannon and Simpson's EPT diversity indices have varied considerably through time in both monitored and reference streams since monitoring began (Figure 3.3-29). Still, graphical analysis suggests that both Shannon and Simpson's EPT diversity were within the range of values observed in baseline years in all streams in 2013 (Figure 3.3-29). Mean EPT diversities in 2013 were also within ± 2 SD of baseline means in all streams except Counts Outflow, in which Shannon and Simpson's diversity were greater than baseline years (Figure 3.3-29; Table 3.3-20). Relative densities of EPT taxa have been variable through time in all monitored and reference streams. In most cases, the relative densities of EPT taxa have shown no signs of directed change through time (Figures 3.3-30 to 3.3-33). The one possible exception is Kodiak-Little, in which there is some evidence that Nemouridae have been replaced by Brachycentridae through time (Figures 3.3-33a). However, similar patterns were observed in Counts Outflow (Figure 3.3-32). Thus, no mine effects were detected with respect to EPT diversity or taxonomic composition.

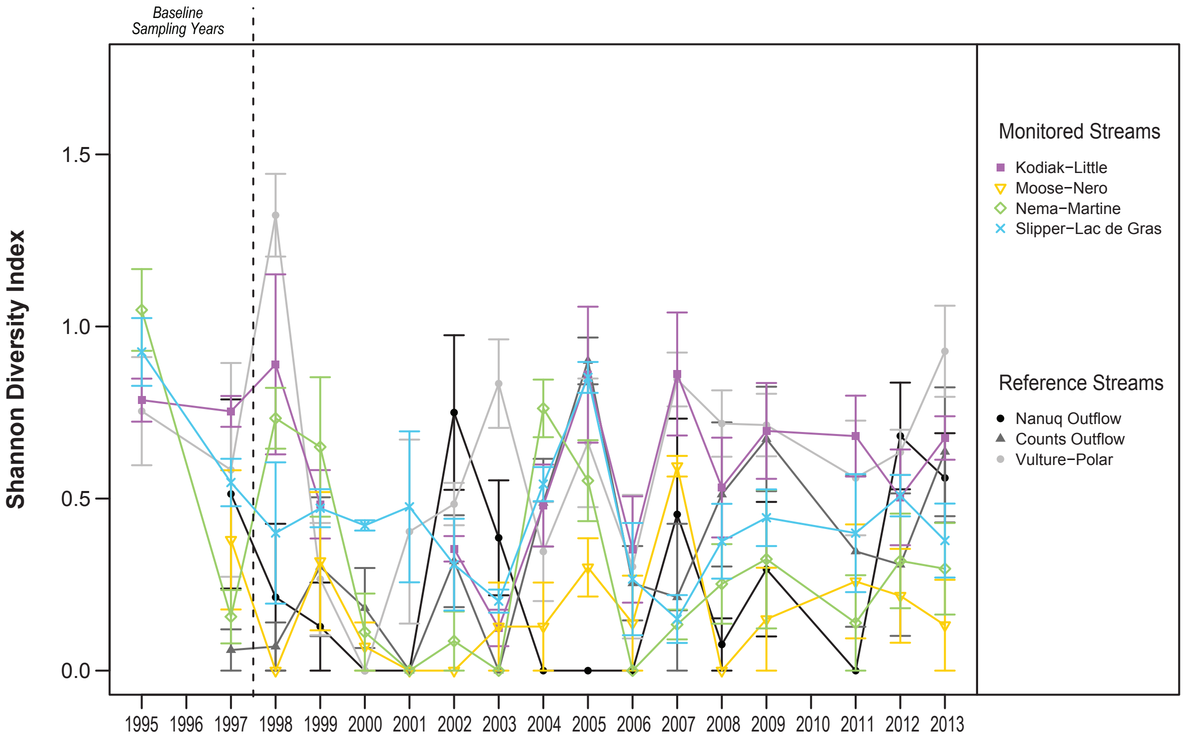
3.3.5 Aquatic Biology Summary

Five changes in biological variables were observed in 2013:

- Altered phytoplankton genera diversity in Leslie Lake;
- Altered taxonomic composition of phytoplankton assemblages in lakes downstream of the LLCF as far as site S2 in Lac de Gras;
- Decreased zooplankton diversity in lakes downstream of the LLC as far as Nema Lake;
- Altered taxonomic composition of zooplankton assemblages in Leslie, Moose, and Nema lakes; and
- Altered taxonomic composition of lake benthos communities in lakes downstream of the LLCF as far as Nema Lake, and at site S2 in Lac de Gras.

Figure 3.3-29

Average Diversity Indices for Benthic EPT Taxa in Koala Watershed Streams, August 1995 to 2013



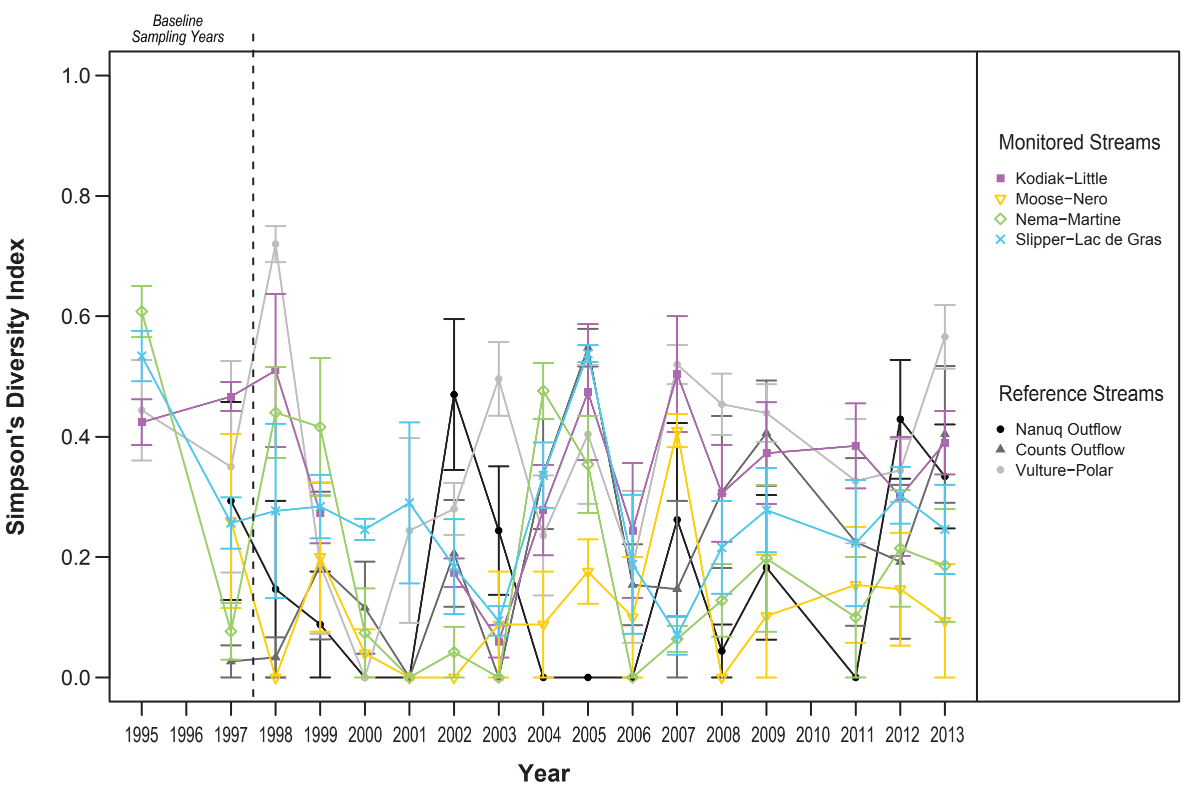


Figure 3.3-30

Average Benthic EPT Density by Taxonomic Group for AEMP Reference Streams, 1995 to 2013

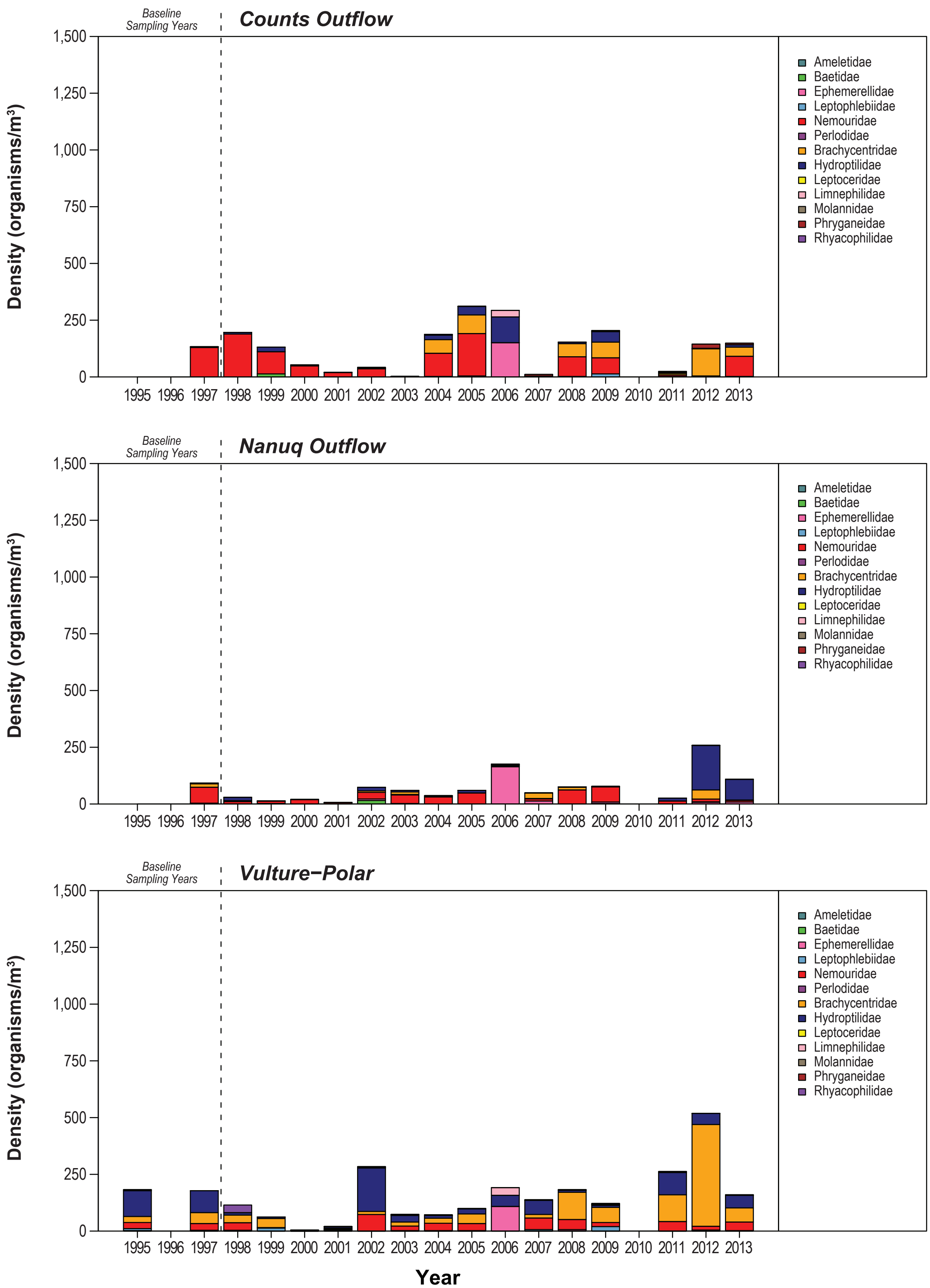
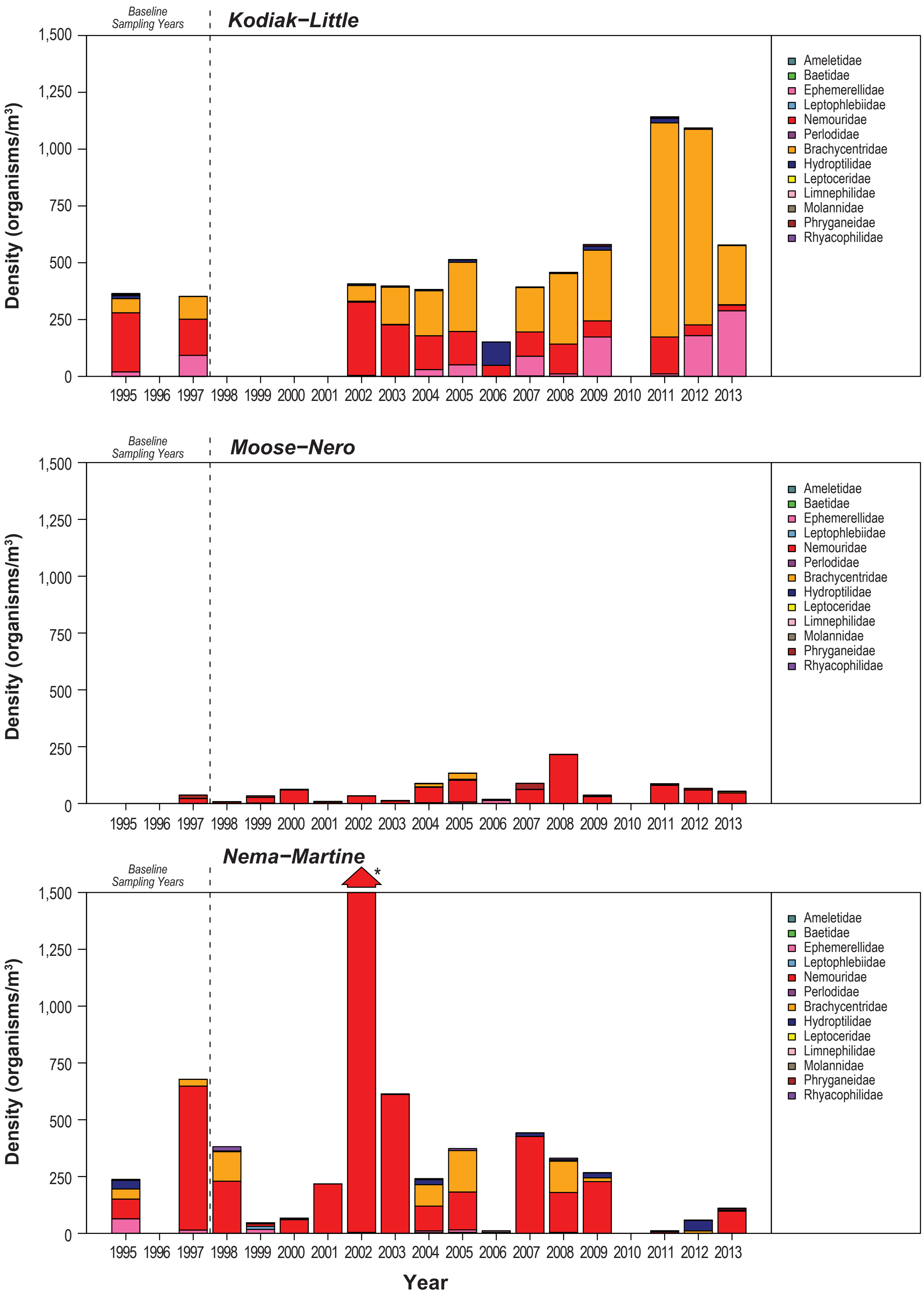


Figure 3.3-31a

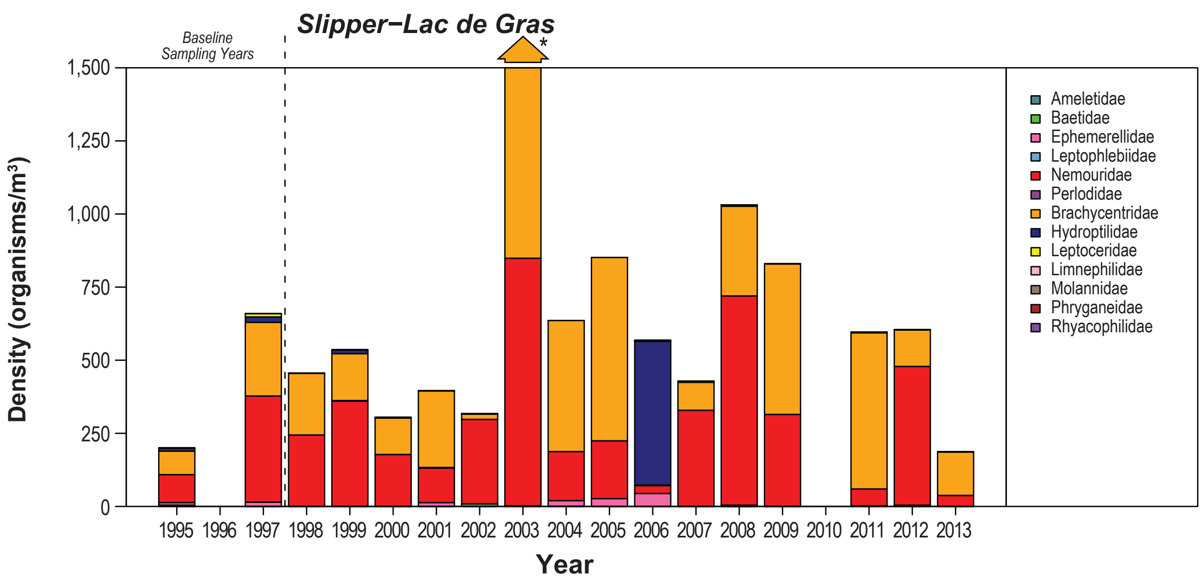
Average Benthic EPT Density by Taxonomic Group for Streams of the Koala Watershed, 1995 to 2013



Notes: 2002 total density = 1,956, Nemouridae = 1,938, Brachycentridae = 7, Hydroptilidae = 7.

Figure 3.3-31b

Average Benthic EPT Density by Taxonomic Group for Streams of the Koala Watershed, 1995 to 2013



Note: 2003 total density = 2,980, Brachycentridae = 2,129, Rhyacophilidae = 2.

Figure 3.3-32

Relative Densities of Benthic EPT Taxa in AEMP Reference Streams, 1995 to 2013

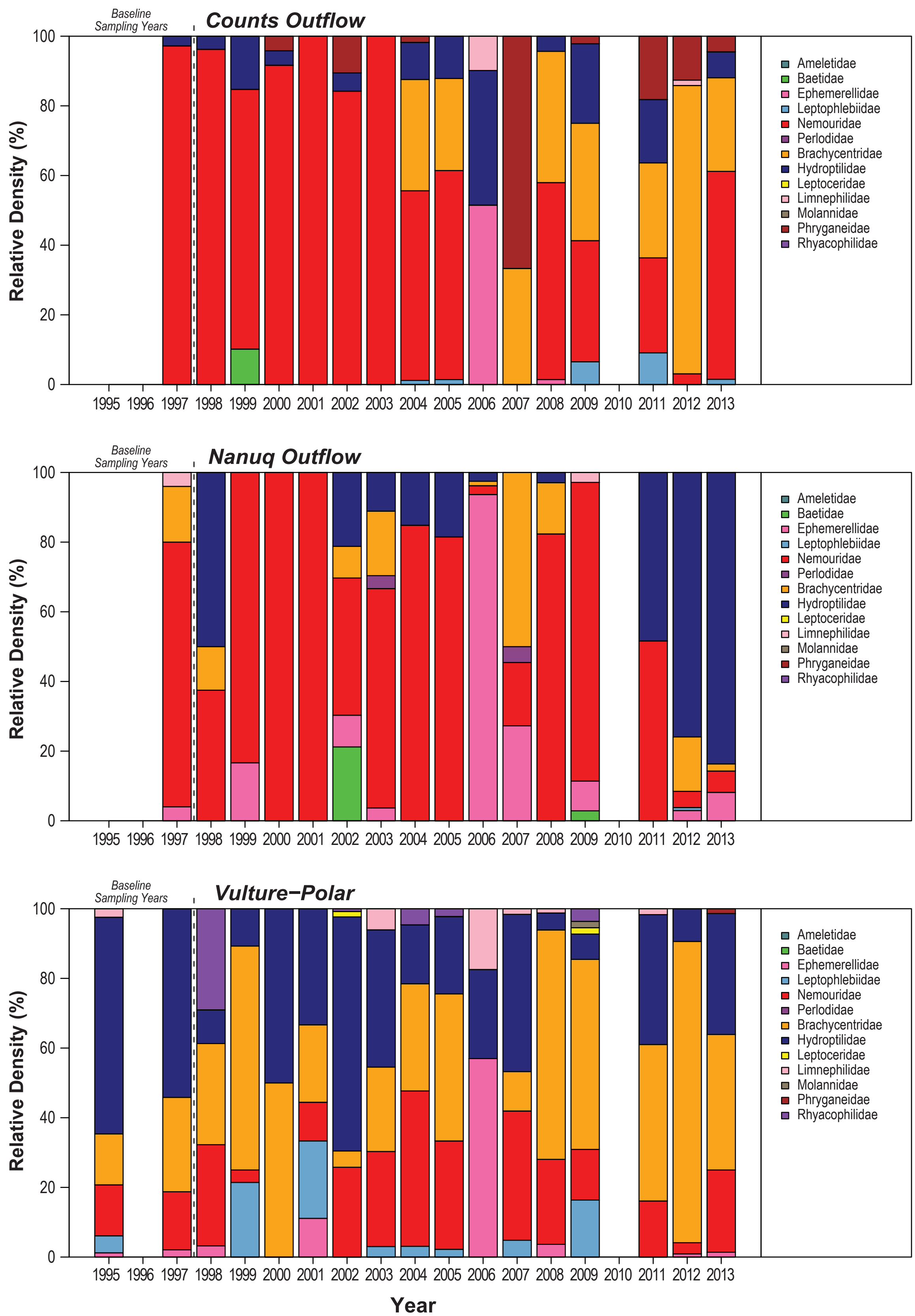


Figure 3.3-33a

Relative Densities of Benthic EPT Taxa in Streams of the Koala Watershed, 1995 to 2013

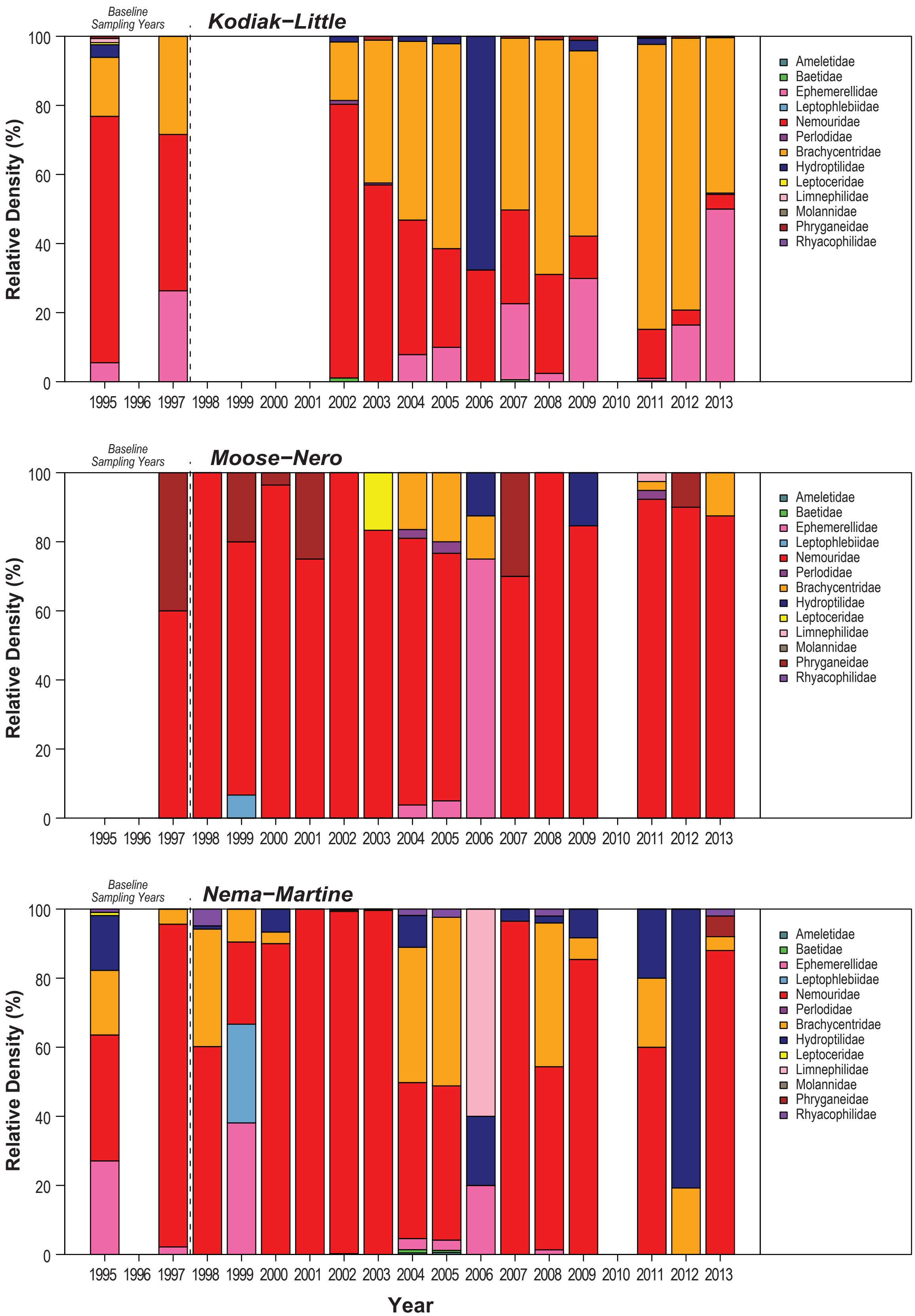


Figure 3.3-33b

Relative Densities of Benthic EPT Taxa in Streams of the Koala Watershed, 1995 to 2013

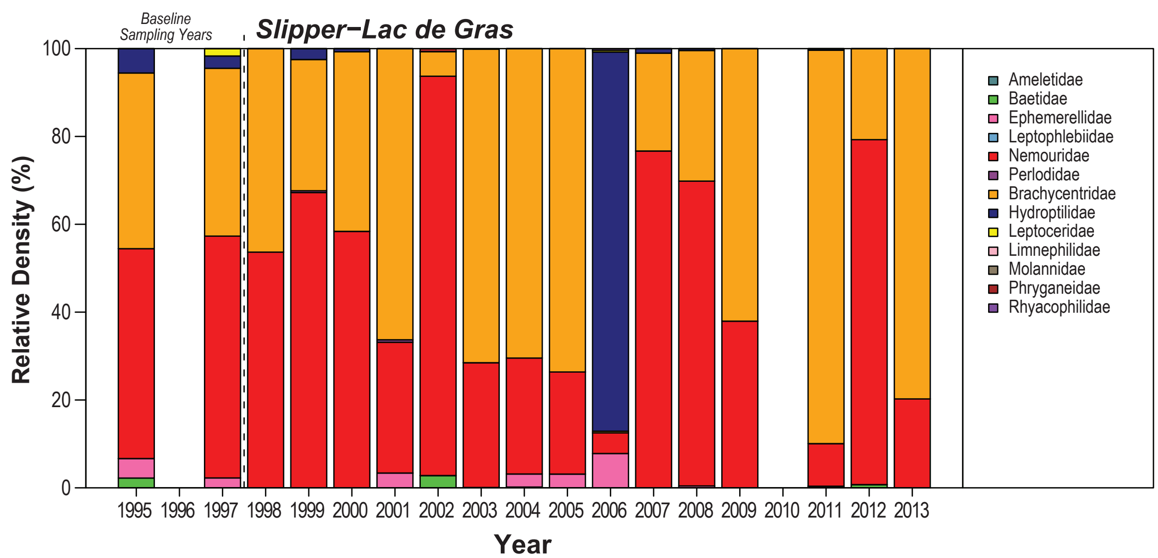


Table 3.3-20. Mean ± 2 Standard Deviations (SD) Baseline Benthic EPT Diversity in Each of the Koala Watershed Streams

	Shannon Diversity			Simpson's Diversity		
Stream	**Baseline Mean (N)**	**Mean Baseline Range, ± 2 SD**	**2013 Mean ± 1 SD**	**Baseline Mean (N)**	**Mean Baseline Range, ± 2 SD**	**2013 Mean ± 1 SD**
Nanuq Outflow	0.51 (1)	0 - 1.47	0.56 ± 0.29	0.29 (1)	0 - 0.86	0.33 ± 0.19
Counts Outflow	0.06 (1)	0 - 0.27	0.64 ± 0.42	0.03 (1)	0 - 0.12	0.40 ± 0.25
Vulture-Polar	0.69 (2)	0 - 1.49	0.93 ± 0.30	0.41 (2)	0 - 0.85	0.57 ± 0.12
Kodiak-Little	0.77 (2)	0.54 – 1.00	0.68 ± 0.14	0.44 (2)	0.30 – 0.58	0.39 ± 0.12
Moose-Nero	0.38 (1)	0 - 1.08	0.13 ± 0.30	0.26 (1)	0 - 0.76	0.10 ± 0.21
Nema-Martine	0.71 (2)	0 - 1.73	0.30 ± 0.30	0.41 (2)	0 - 0.98	0.19 ± 0.21
Slipper-Lac de Gras	0.78 (2)	0.25 – 1.31	0.38 ± 0.24	0.43 (2)	0.10 – 0.76	0.25 ± 0.17

Negative values were replaced with zeros.
N = number of years data were collected.

Phytoplankton diversity has been stable through time in all monitored lakes of the Koala Watershed and Lac de Gras, except Leslie Lake. Phytoplankton diversity in Leslie Lake decreased from 2006 to 2011, but has returned to historical levels in 2013. Phytoplankton community composition has shifted in all lakes downstream of the LLCF as far as site S2 in Lac de Gras, with a decrease in the relative densities of Myxophyceae (blue-green algae) and an increase in the proportion of Bacillariophyceae (diatoms) through time. This shift from blue-green algae to diatoms is likely related to changes in nitrate-N concentrations, which also show a spatial gradient with downstream distance from the LLCF following the onset and subsequent expansion of underground mining operations in 2002 (see Section 3.2.4.9).

Examination of species tolerances with respect to current water quality in the receiving environment suggests that observed changes in biological community composition at the Ekati Diamond Mine likely result from inter-specific differences in the competitive ability of different taxonomic groups under changing quantities or ratios of macronutrients like nitrogen or phosphorus, rather than elemental toxicity (see Section 3.3; Rescan 2012d). Accumulating research suggests that the ratio of available elements, especially macronutrients like carbon (C), nitrogen (N), and phosphorous (P), can play an important role in determining community composition and relative abundance by providing a competitive advantage to taxa whose relative elemental requirements best match current conditions (Sterner et al. 1997; Dobberfuhl and Elser 2000; Elser et al. 2000). At the Ekati Diamond Mine, the ratio of available nutrients in aquatic systems has shifted through time as nitrogen concentrations have increased. Consequently, the composition of primary producers has shifted from those that thrive in high C:N environments because they are capable of fixing nitrogen (i.e., blue-green algae) to those that can take up N directly from the environment and therefore thrive in low C:N environments (i.e., diatoms; Tillman et al. 1986).

The shift in phytoplankton community composition and associated increase in nitrogen in lakes downstream of the LLCF has been recognized for some time and DDEC has undertaken a number of adaptive management actions to reduce the amount of nitrate-N released into the receiving environment. These include the diversion of underground mine water to Beartooth Pit and the addition of phosphorous to Cell D of the LLCF to stimulate nitrogen uptake by phytoplankton (Rescan 2010a, 2011c; Golder 2013). Recent trends in nitrate-N in Cell D and Koala Watershed lakes suggest that such mitigation measures may be working because nitrate-N concentrations have stabilised in recent years (Rescan 2011c; Golder 2013). Although water quality modelling predicts that nitrate concentrations will continue to increase in the LLCF and Koala Watershed lakes downstream of the LLCF (Rescan 2012i), results suggest that nitrogen concentrations have remained stable in 2013 (see Section 3.2.4.9).

A second shift in phytoplankton community composition, toward increased densities of Chlorophyceae, was observed in Leslie Lake from 2010 to 2012 and in Nema and Slipper lakes in 2013 (Figures 3.3-4 and 3.3-7). This second shift in primary producer community composition may be explained by the addition of phosphorous to Cell D of the LLCF from 2009 to 2011 as an adaptive management response to increased nitrate concentrations (Rescan 2011b). The addition of phosphorous to the LLCF ceased in 2011 and in 2013, the phytoplankton assemblage in Leslie Lake returned toward historic community compositions. The increase in Chlorophyceae observed further downstream, in Nema and Slipper lakes, in 2013 may reflect a spatiotemporal lag in the effect of phosphorous additions to Cell D of the LLCF. Chlorophyceae are known to outcompete diatoms at intermediate ratios of N:P (Tillman et al. 1986; Lagus et al. 2004). In addition, concentrations of all the evaluated water quality variables in the Koala Watershed have remained below the lowest identified chronic effect level for the most sensitive species, except potassium (Rescan 2012d, 2012g). However, there was no evidence that elevated potassium concentrations have led to declines in the density of the most sensitive species (see Section 3.3.2). Thus, the correlations between changes in phytoplankton community composition and increases in some water quality variables (e.g., chloride, sulphate, potassium, total arsenic, etc.) may reflect shifts in the relative availability of macronutrients at the Ekati Diamond Mine, rather than species sensitivities to changes in water quality variables.

Although zooplankton biomass and density have been stable through time in all monitored and reference lakes, zooplankton diversity has declined in lakes downstream of the LLCF as far as Nema Lake. Declines in zooplankton diversity have been associated with a shift in community composition that extends as far as Nema Lake. In these lakes, cladocerans (particularly *Holopedium gibberum*) and rotifers (particularly *Conochilus sp.* and *Kellicottia longispina*) have been replaced, to an extent, by copepods. Although diversity increased in Leslie Lake in 2013, the zooplankton community remained dominated by copepods. In contrast, rotifers showed signs of recovery in Nema Lake in 2013, with *K. longispina* dominating community composition to such an extent that zooplankton diversity was relatively low in Nema Lake in 2013. Similar to phytoplankton communities, overall shifts in zooplankton communities showed some evidence of tracking changes in the relative availability of macronutrients, with the relative densities of consumers with high somatic N:P ratios increasing through time and with spatial proximity to the LLCF (e.g., calanoid and cyclopoid copepods; Dobberfuhl and Elser 2000; McCarthy, Donohue, and Irvine 2006). Thus, the observed changes in zooplankton community composition are likely driven, ultimately, by changes in the availability of macronutrients including nitrogen and phosphorus in lakes downstream of the LLCF.

Lake benthos density, dipteran diversity, and dipteran community composition have been variable through time in all monitored and reference lakes. However, the relative densities of dipteran taxonomic communities have changed through time in Leslie and Moose lakes, a pattern that was first identified through the multivariate analyses conducted as part of the 2012 AEMP Re-evaluation (Rescan 2012d). In these lakes, the relative densities of organisms from the Chironomidae subfamily Orthocladiinae (likely from the genera *Rheocricotopus* and *Psectrocladius*) have decreased, while densities of Diamesinae (most likely organisms from the genus *Protanypus*), Prodiamesinae (most likely organisms from the genus *Monodiamesa*), and Chironominae (most likely organisms from the genera *Cladotanytarsus* and *Stempellinella*) have increased through time. Most of these shifts in taxonomic composition began around 2005. In addition, more recent changes in dipteran community composition have been observed in Nema Lake and site S2 in Lac de Gras. Similar to Leslie and Moose lakes, densities of Orthocladiinae (likely from the genera *Psectrocladius*) in Nema Lake have decreased, but with a coincidental increase in densities of Tanypodinae (likely from the genera *Procladius* and *Ablabesmyia*). Meanwhile, overall densities of Prodiamesinae (likely from the genus *Monodiamesa*) have recently increased at site S2 in

Lac de Gras. Little information is available on the ecology of these groups and the cause of these shifts is unclear (Oliver and Dillon 1997). For similar reasons identified for phytoplankton and zooplankton, it is likely that changes in benthos community composition are associated with changes in macronutrient availability, rather than toxic effects.

No mine effects were detected with respect to stream benthos density, dipteran diversity, Ephemeroptera, Plecoptera, Trichoptera (EPT) diversity, or dipteran community composition.

Both zooplankton and lake benthos provide an important source of food for many species of fish. Changes in community composition could have important consequences for fish, especially if preferred prey items are replaced with non-preferred ones. Results of the 2012 AEMP Evaluation of Effects found no evidence of major mine effects on monitored fish populations in the Koala Watershed (Rescan 2012d). Shifts in phytoplankton, zooplankton and benthos communities, do not appear to have influenced fish populations to date. Both round whitefish and lake trout are considered opportunistic feeders where in the absence of strong prey community-wide effects, may not exhibit strong biological changes, including any bioenergetics-related response variables. Furthermore, the mobile nature of these larger-bodied fish populations may also serve to reduce any potential effects. Lakes in the Ekati Diamond Mine study area are not isolated and individual fish are able to move freely between upstream and downstream lakes. This likely serves to buffer any potential effects or may delay the appearance of mine effects.

3.4 SUMMARY

Table 3.4-1 summarizes the evaluation of effects for the Koala Watershed and Lac de Gras. Conclusions regarding the direction of change were drawn from graphical analysis because statistical tests were two-sided and tested only for differences between reference and monitored lakes rather than the direction of change.

Under-ice temperature profiles suggest that there has been a trend towards cooling in all lakes downstream of the LLCF as far as Nema Lake. Although the cause of this shift is unclear, there is also some evidence of a general cooling trend, at all depths, in two of the reference lakes (i.e., Nanuq and Vulture lakes) in recent years, suggesting that shifts in temperature profiles in monitored lakes may reflect natural climatic variability rather than mine effects. In Grizzly Lake, the shape of the temperature profile has changed in recent years. Specifically, from 2011 to 2013, under-ice temperature profiles in Grizzly Lake showed some degree of thermal stratification, with cooler surface temperatures. The cause of the change in Grizzly Lake is unclear; however, thermal stratification was also observed in Vulture Lake in 2013, the one reference lake that is as deep as Grizzly Lake, suggesting that the change in the Grizzly Lake thermal profile may also reflect natural climactic variability, rather than mine effects. In contrast, a warming trend was detected in Kodiak Lake, along with corresponding changes in dissolved oxygen profiles. The observed changes in Kodiak Lake likely stem from DDEC's efforts to improve dissolved oxygen concentrations in Kodiak Lake, which have included the use of aerators beginning in 1997. The changes in the under-ice temperature and DO profiles in Kodiak Lake correspond to the first year in which aerators were no longer used (2007). The current, stratified DO profiles likely represent undisturbed conditions in Kodiak Lake: aerators would cause mixing of the water column which would result in homogeneity of temperature and dissolved oxygen throughout the water column. Despite changes in under-ice profiles, open water season temperature and dissolved oxygen profiles in all monitored lakes were similar to previous years in all lakes. Secchi depths were also similar to those observed in previous years.

Table 3.4-1. Summary of Evaluation of Effects for the Koala Watershed and Lac de Gras

Variable	Change Downstream of the LLCF?	Change Upstream of the LLCF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Physical Limnology							
Under-ice Temperature Profiles	Yes	Yes	Leslie and Nema lakes; Kodiak Lake; Grizzly Lake	Cooling in Leslie, Nema; surface cooling in Grizzly; warming in Kodiak Lake	Unknown downstream of LLCF and Grizzly Lake; aerators in Kodiak Lake	Yes	Current temperature profiles likely represent undisturbed conditions in Kodiak Lake.
Under-ice DO Profiles	No	Yes	Kodiak Lake	Decreased concentrations throughout water column	aerators	Historical	DO concentrations above CCME guidelines throughout most of the water column in all lakes; reference and monitored lakes show similar trends. Current DO profiles likely represent undisturbed conditions in Kodiak Lake.
August Secchi Depths	No	No	-		-	No	-
Lake and Stream Water Quality							
pH	Yes	No	Downstream to site S3	Increase	LLCF	Yes	The observed mean in Grizzly Lake was below CCME guidelines; similar pattern observed in all reference lakes and streams.
Total Alkalinity	Yes	No	Downstream to site S2	Increase	LLCF	Yes	-
Hardness	Yes	No	Downstream to site S3	Increase	LLCF	Yes	-
Chloride	Yes	No	Downstream to site S3	Increase	LLCF	Yes	All 2013 concentrations less than the SSWQO.
Sulphate	Yes	Yes	Downstream to site S3; Kodiak Lake	Increase	LLCF; Possibly mine related activities at the main camp	Yes	All 2013 concentrations less than the SSWQO.
Potassium	Yes	No	Downstream to site S3	Increase	LLCF	Yes	Observed mean concentrations in the ice-covered season in Leslie and Moose lakes exceeded the SSWQO.

(continued)

Table 3.4-1. Summary of Evaluation of Effects for the Koala Watershed and Lac de Gras (continued)

Variable	Change Downstream of the LLCF?	Change Upstream of the LLCF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Lake and Stream Water Quality ***(cont'd)***							
Total Ammonia-N	Yes	No	Downstream to Slipper Lake	Increase	LLCF	Yes	Trend is less clear in streams than in lakes and was not correlated with downstream distance from the LLCF; 95% confidence interval around the 2013 fitted mean exceeded the CCME guideline value in Counts Lake during the ice-covered season.
Nitrite-N	Yes	No	Downstream to Moose-Nero Stream	Increase	LLCF	Yes	All 2013 concentrations less than the CCME guideline.
Nitrate-N	Yes	No	Downstream to Slipper Lake	Increase	LLCF	Yes	All 2013 concentrations less than the SSWQO.
Total Phosphate-P	Yes	No	Downstream to Moose Lake	Increase	LLCF	Yes	The 2013 upper 95% confidence interval around fitted mean in Leslie, Moose, Nema, Slipper, Counts, Nanuq, and Vulture lakes and at site S2 during the open water season and in Moose and Nanuq lakes during the ice-covered season exceeded the CCME trigger range or benchmark values; The observed mean at site S2 and Nanuq Lake during the open water season exceeded the benchmark value; The observed and fitted mean for Nanuq Lake during the ice-covered season exceeded the benchmark value.
Total Organic Carbon	No	No	-	-	Unknown	No	No clear spatial gradient and no baseline data for comparison
Total Antimony	Yes	No	Downstream to Moose-Nero Stream	Increase	LLCF	Yes	All 2013 concentrations less than the water quality benchmark.
Total Arsenic	Yes	No	Downstream to Moose Lake	Increase	LLCF	Yes	All 2013 concentrations less than the CCME guideline.
Total Barium	Yes	No	Downstream to site S2	Increase	LLCF	Yes	All 2013 concentrations less than the water quality benchmark.

(continued)

Table 3.4-1. Summary of Evaluation of Effects for the Koala Watershed and Lac de Gras (continued)

Variable	Change Downstream of the LLCF?	Change Upstream of the LLCF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Lake and Stream Water Quality *(cont'd)*							
Total Boron	Yes	No	Downstream to Slipper Lake	Increase	LLCF	Yes	All 2013 concentrations less than the CCME guideline.
Total Cadmium	No	No	-	-	-	No	All 2013 concentrations above detection limit were less than the CCME guideline.
Total Molybdenum	Yes	No	Downstream to site S3	Increase	LLCF	Yes	All 2013 concentrations less than the SSWQO.
Total Nickel	Yes	Yes	Downstream to Nema-Martine Stream; Kodiak Lake and Kodiak-Little Stream	Increase	LLCF; unknown in Kodiak Lake and Kodiak-Little	Yes	All 2013 concentrations less than the CCME guideline.
Total Selenium	No	No	-	-	-	No	All 2013 concentrations less than the CCME guideline.
Total Strontium	Yes	No	Downstream to site S3	Increase	LLCF	Yes	All 2013 concentrations less than the water quality benchmark.
Total Uranium	Yes	No	Downstream to Nema-Martine Stream	Increase	LLCF	Yes	All 2013 concentrations s less than the CCME guideline.
Total Vanadium	No	No	-	-	-	-	All 2013 concentrations less than the SSWQO.
Phytoplankton							
Chlorophyll *a*	No	No	-	-	-	No	-
Density	No	No	-	-	-	No	-
Diversity	Yes	No	Leslie Lake	Decrease	LLCF	Yes	Diversity returned to historical levels in 2013.
Relative Densities of Major Taxa	Yes	No	Downstream to site S2	(see Notes column)	LLCF	Yes	Decline in relative abundance of blue-green algae and increase in diatoms; replacement of diatoms with green algae in Leslie Lake between 2010 and 2012.

(continued)

Table 3.4-1. Summary of Evaluation of Effects for the Koala Watershed and Lac de Gras (continued)

Variable	Change Downstream of the LLCF?	Change Upstream of the LLCF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Zooplankton							
Biomass	No	No	-	-	-	No	-
Density	No	No	-	-	-	No	-
Diversity	Yes	No	Nema Lake	Decrease	LLCF	Yes	Generally declining over time in Leslie and Moose lakes, but has increased to historical or reference lake levels in 2013. Recent decline in Nema attributed to decline in specific rotifer and cladoceran genera.
Relative Densities of Major Taxa	Yes	No	Downstream to Nema Lake	Decrease in cladocerans	LLCF	Yes	Decrease in proportion of cladocerans possibly as a result of the LLCF discharge.
Lake Benthos							
Density	No	No	-	-	-	No	-
Dipteran Diversity	No	No	-	-	-	No	-
Dipteran Relative Density	Yes	No	Leslie, Moose and Nema lakes and site S2	Decrease in Orthocladinae in Leslie, Moose and Nema lakes; Increase in Diamesinae, Prodiamesinae, Chironominae in Leslie and Moose lakes; Increase in Tanypodinae in Nema Lake; Increase in Prodiamesinae at S2	-	Yes	Changes in community composition may be related to decreases in some genera (*Rheocricotopus*, *Psectrocladius*) and increases in others (*Monodiamesa*, *Protanypus*, *Cladotanytarsus*, *Stempellinella*, *Procladius*, *Ablabesmyia*). A similar pattern of decreased Orthocladiinae with increasing Chironominae (*Corynocera*, *Stictochironomus*) observed in recent years in one reference lake (Counts Lake).

(continued)

Table 3.4-1. Summary of Evaluation of Effects for the Koala Watershed and Lac de Gras (completed)

Variable	Change Downstream of the LLCF?	Change Upstream of the LLCF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Stream Benthos							
Density	No	No	-	-	-	No	-
Dipteran Diversity	No	No	-	-	-	No	-
Dipteran Relative Density	No	No	-	-	-	No	Some changes in taxonomic composition related to broader climatic patterns or systematic changes in in enumeration/identification observed.
EPT Diversity	No	No	-	-	-	No	-
EPT Relative Density	No	No	-	-	-	No	-

Dashes indicate not applicable.
Comparisons to CCME guidelines are for 2013 data only.
DO = dissolved oxygen
CCME = Canadian Council of Ministers of the Environment
SSWQO = Site-specific Water Quality Objective

Grizzly Lake is the source of potable water for the Ekati Diamond Mine's Main Camp and was added to the statistical evaluation of effects for the AEMP in 2009. At present, biological variables and sediment quality are not monitored in Grizzly Lake as part of the AEMP. However, the change in the shape of the temperature profile may have implications for biological communities. Most species have thermal optima (i.e., temperature ranges over which they thrive) (Kravtsova 2000). All ectothermic organisms (i.e., organisms that do not generate their own body heat) are sensitive to changes in temperature, with increases in temperature resulting in higher basal metabolic rates, higher activity levels, shorter lifespans, and smaller body sizes (Angilletta 2010). Thus, changes in temperature can cause shifts in community composition and food web dynamics (Gillooly et al. 2001; Brown et al. 2004; Kingsolver and Huey 2008). Biological variables (i.e., phytoplankton, zooplankton, and benthos) were assessed in Grizzly Lake in 2013 to examine if any changes in biological communities were observed that may be related to the change in temperature profile and results are provided in Section 3.9.3 of Part 2 – Data Report.

Twenty-two water quality variables were evaluated in the 2013 AEMP for the Koala Watershed and Lac de Gras. Of these, concentrations of 18 variables have changed in lakes or streams in the Koala Watershed or Lac de Gras (Table 3.4-1). Although concentrations of eight water quality variables have stabilised at some sites in recent years, concentrations remain elevated above baseline or reference concentrations in all 18 cases. The extent to which concentrations have changed through time generally decreases with downstream distance from the LLCF. Patterns were similar during the ice-covered and open water seasons, though concentrations were sometimes greater during the ice-covered season as a consequence of solute exclusion during freeze up. In reference lakes, concentrations of water quality variables have generally been low and stable through time. Together, the evidence suggests that the observed changes in concentrations in the18 water quality variables identified in Table 3.4-1 in lakes and streams that are downstream of the LLCF are mine effects that stem from the discharge of water from the LLCF into the receiving environment under Water Licence W2012L2-0001. In monitored lakes and streams that are not downstream of the LLCF (i.e., Grizzly Lake, Kodiak Lake and associated streams), only two water quality variables have increased through time: sulphate has increased in Kodiak Lake and total nickel has increased in Kodiak Lake and Kodiak-Little.

CCME guidelines for the protection of aquatic life exist for nine of the evaluated water quality variables, including pH, total ammonia-N, nitrite-N, total arsenic, total boron, total cadmium, total nickel, total selenium, and total uranium (CCME 2013). In addition, DDEC has established SSWQO for six of the evaluated variables, including chloride, sulphate, potassium, nitrate-N, total molybdenum, and total vanadium (see Table 2.3-1). Total phosphate concentrations were compared to lake-specific benchmark trigger values that were established using guidelines set out in the Canadian Guidance Framework for the Management of Phosphorus in Freshwater Systems, the Ontario Ministry of Natural Resources, and Environment Canada (Ontario Ministry of Natural Resources 1994; CCME 2004; Environment Canada 2004). Other water quality benchmarks exist for total antimony, total barium, and total strontium (see Table 2.3-1). In general, the 95% confidence intervals around the fitted mean and the observed mean concentrations were below their respective CCME guideline value, SSWQO, and relevant benchmark value except for pH, total phosphate-P, and potassium (see Table 3.4-1). For pH and total phosphate-P, levels and concentrations in reference lakes or streams also exceeded CCME guidelines, suggesting that exceedences are not related to mine activities. In contrast, potassium exceedences were unique to the two most upstream monitored lakes and are thus likely related to mine activities.

Despite increases in 18 evaluated water quality variables downstream of the LLCF, observed concentrations were generally below water quality benchmark values and thus below concentrations at which toxic effects might be expected. Although potassium concentrations were greater than CCME guidelines in Leslie and Moose lakes, there was no evidence that elevated potassium concentrations have led to declines in the density of the species most sensitive to potassium (see Section 3.3.2; Biesinger and Christensen 1972). Thus, observed changes in biological community composition at the Ekati Diamond

Mine likely result from inter-specific differences in the competitive ability of different taxonomic groups under changing quantities or ratios of macronutrients, rather than elemental toxicity (Rescan 2012d). In phytoplankton communities, community composition has shifted from blue-green algae to diatoms in lakes as far downstream as site S2 in Lac de Gras. A second shift was observed in Leslie, Nema, and Slipper lakes, though community composition in Leslie Lake in 2013 was more similar to reference lake communities than the communities observed from 2010 to 2012. Shifts in zooplankton assemblages have also been observed in Leslie, Moose, and Nema lakes in recent years. Specifically, increases in the densities of copepods have coincided with decreases in the densities of cladocerans and rotifers through time. Some slight changes have also been observed in benthos communities in Leslie, Moose, and Nemalakes, in which relative densities of Orthocladiinae have decreased, while densities of Diamesinae, Prodiamesinae, and Chironominae have increased through time. Although changes in relative abundances of zooplankton and lake benthos could have important cascading effects for higher trophic levels, no evidence to date suggests that monitored fish populations at the Ekati Diamond Mine have been influenced by changes in the relative abundance of prey species (see Section 3.3.5; Rescan 2012d).

4. Evaluation of Effects: King-Cujo Watershed and Lac du Sauvage

ERM

4. Evaluation of Effects: King-Cujo Watershed and Lac du Sauvage

4.1 PHYSICAL LIMNOLOGY

4.1.1 Variables

Two physical limnology variables were evaluated for potential effects caused by mine activities in the King-Cujo Watershed: under-ice dissolved oxygen concentrations and open water season Secchi depths (see Section 3.1.1).

4.1.2 Dataset

Under-ice dissolved oxygen and temperature profiles were collected in March, April, or May of each year for the evaluation of effects (Table 4.1-1). Secchi depths were measured during August sampling surveys (Table 4.1-2).

Table 4.1-1. Dataset Used for Evaluation of Effects on Under-ice Dissolved Oxygen and Temperature Profiles in King-Cujo Watershed Lakes and Lac du Sauvage

Year	Nanuq	Counts	Vulture	Cujo	LdS1
1994	-	-	-	-	-
1995	-	-	-	-	-
1996	-	-	-	-	-
1997	-	-	-	-	-
1998	Apr-19	Apr-19	Apr-15	-	-
1999	Apr-17	Mar-10	Mar-24	-	-
2000	Mar-16	Mar-17	Mar-23	Apr-11	-
2001	Apr-14	Apr-15	Apr-14	Apr-15	Apr-15
2002	Apr-23	Apr-23	Apr-20	Apr-23	Apr-23
2003	Apr-12	Apr-13	Apr-14	Apr-13	Apr-13
2004	Apr-18	Apr-17	Apr-18	Apr-17	Apr-17
2005	-	-	-	Apr-28	Apr-25
2006	Apr-20	Apr-22	Apr-21	Apr-22	Apr-22
2007	-	-	-	-	-
2008	Apr-27	May-3	May-3	May-3	May-4
2009	May-18	May-17	Apr-28	May-3	May-18
2010	Apr-14	Apr-14	Apr-14	Apr-16	Apr-14
2011	Apr-25	Apr-26	Apr-28	Apr-26	Apr-26
2012	Apr-20	Apr-17	Apr-18	Apr-15	Apr-19
2013	Apr-26	Apr-26	Apr-23	May-11	Apr-26

Dashes indicate no data were available.

Table 4.1-2. Dataset Used for Evaluation of Effects on Secchi Depths in King-Cujo Watershed Lakes and Lac du Sauvage

Year	Nanuq	Counts	Vulture	Cujo	LdS1
1994	-	-	Aug-20	-	-
1995	-	-	Aug-10	-	-
1996	-	-	Jul-28	-	-
1997	Aug-4	Aug-14	Aug-5	-	-
1998	Aug-4	Aug-14	Aug-7	-	-
1999	Aug-7	Aug-8	Aug-6	Aug-8	-
2000	Aug-4	Aug-1	Aug-4	Jul-31	Aug-2
2001	Aug-1	Jul-30	Aug-2	Jul-30	Jul-31
2002	Aug-1	Aug-7	Aug-3	Aug-7	Aug-5
2003	Aug-9	Aug-7	Aug-4	Aug-4	Aug-6
2004	Aug-10	Aug-13	Aug-9	Aug-10	Aug-10
2005	Aug-1	Aug-7	Jul-31	Aug-9	Aug-9
2006	Aug-2	Aug-4	Aug-2	Aug-4	Aug-1
2007	Aug-11	Aug-6	Aug-12	Aug-5	Aug-5
2008	Aug-8	Jul-31	Jul-29	Jul-26	Jul-31
2009	Jul-30	Aug-1	Jul-30	Jul-31	Aug-1
2010	Aug-5	Aug-7	Aug-5	Aug-4	Aug-4
2011	Aug-2	Aug-5	Aug-5	Aug-4	Aug-4
2012	Aug-1	Aug-8	Aug-12	Aug-6	Aug-7
2013	Aug-3	Aug-1	Aug-1	Jul-30	Aug-3

Dashes indicate no data were available.

4.1.3 Results and Discussion

4.1.3.1 Under-ice Dissolved Oxygen

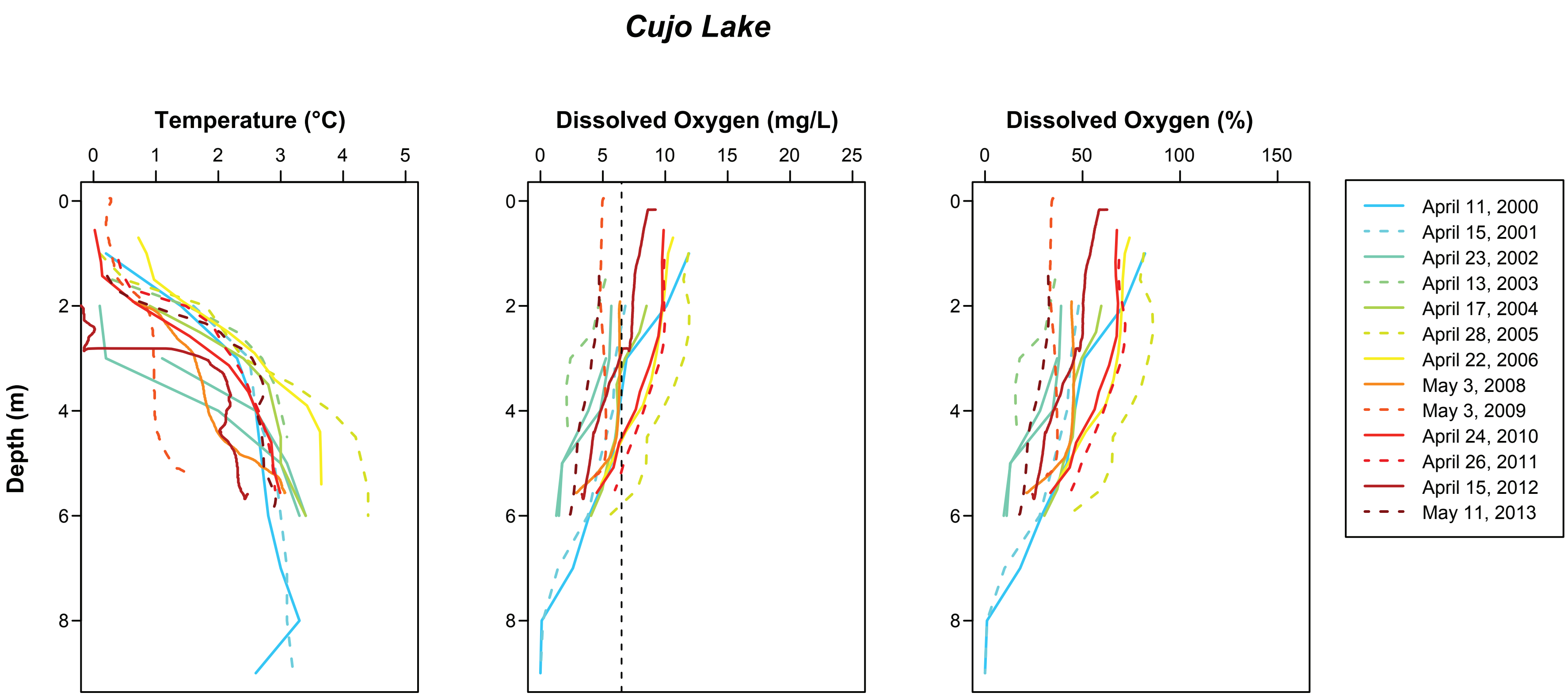
Summary: Under-ice temperature and DO profiles have been consistent through time at all monitored sites in the King-Cujo Watershed and Lac du Sauvage. The concentration of DO in Cujo Lake was less than the 6.5 mg/L CCME guideline throughout the water column. This is consistent with historical DO profiles in Cujo Lake, in which DO concentrations were often less than the CCME guideline value throughout the entire depth profile. No mine effects were detected.

No statistical analyses could be performed on under-ice DO or temperature profiles because they are not replicated. Therefore, graphical analysis and best professional judgment were the primary methods used in the evaluation of potential mine effects on under-ice DO profiles.

Under-ice dissolved oxygen concentrations measured in late April to early May of 2013 were generally consistent with the historical ranges observed in Cujo Lake and site LdS1 in Lac du Sauvage (Figure 4.1-1). DO concentrations in these two lakes were typically greatest close to the ice and declined with depth, as temperature increased (Figure 4.1-1). As in previous years, DO concentrations at site LdS1 in Lac du Sauvage were greater than the CCME guideline of 6.5 mg/L throughout the water column (Figure 4.1-1; CCME 2013). In contrast, DO concentrations in Cujo Lake were less than the CCME guideline at all depths in 2013 (Figure 4.1-1; CCME 2013).

Figure 4.1-1a

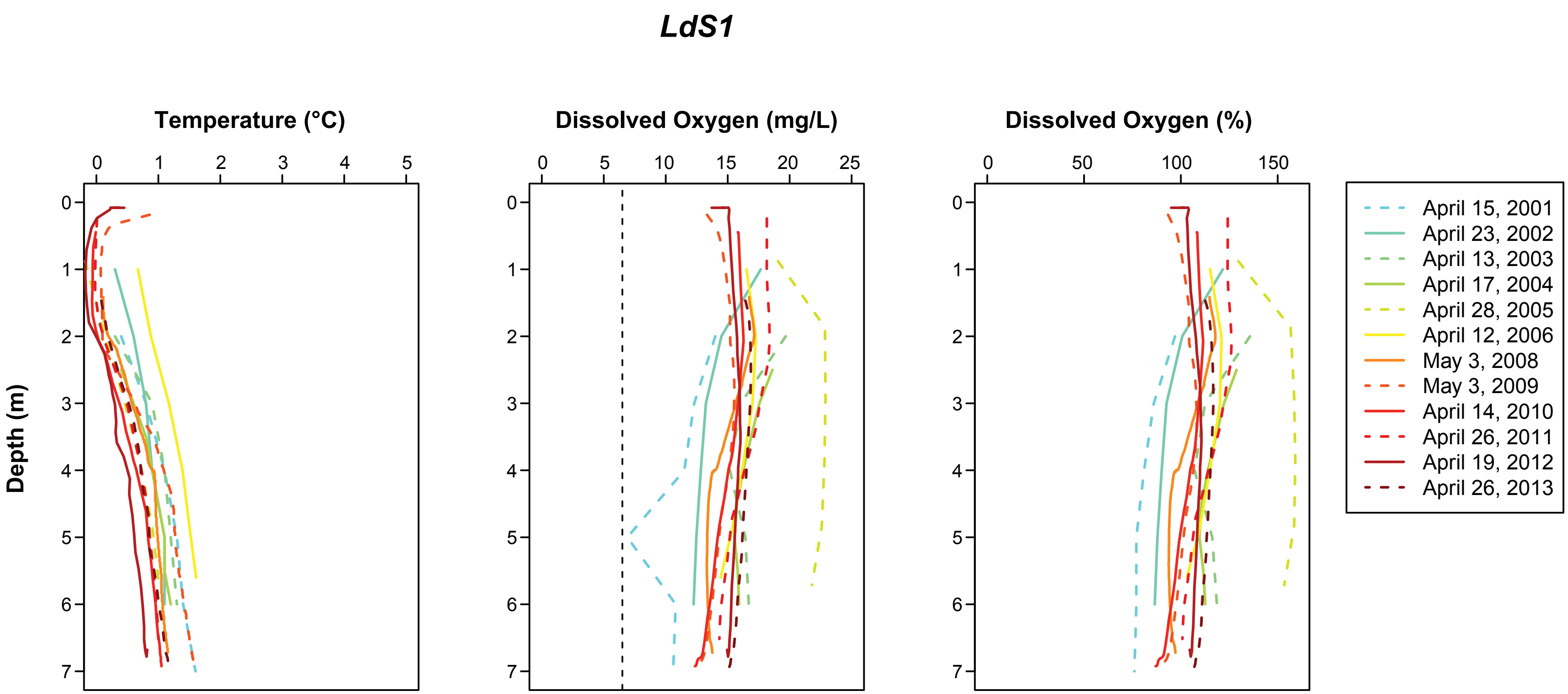
Under-ice Dissolved Oxygen and Temperature Profiles for Cujo Lake, 2000 to 2013



Note: Data collected and supplied by DDEC.
Vertical dashed line represents the CCME guideline for dissolved oxygen (6.5 mg/L).

Figure 4.1-1b

Under-ice Dissolved Oxygen and Temperature Profiles for LdS1, 2001 to 2013



Note: Data collected and supplied by DDEC.
Vertical dashed line represents the CCME guideline for dissolved oxygen (6.5 mg/L).

Additional over-winter monitoring conducted in Cujo Lake in 2013 indicated a decline in under-ice DO from February through to May (Figure 4.1-2), which is consistent with expected patterns in under-ice DO in ice-covered lakes (discussed in Section 3.1.3.1). The concentration of DO in Cujo Lake was less than the 6.5 mg/L CCME guideline throughout most of the water column by early April (Figure 4.1-2; CCME 2013). This reflects historical DO profiles in Cujo Lake, in which DO concentrations have often been less than the CCME guideline value throughout the water column (Figure 4.1-1a). Snow was cleared from Cujo Lake in an attempt to increase the production of DO by photosynthetic organisms from March 30 to April 25, which improved DO profiles on May 6 (Figure 4.1-2). Dissolved oxygen profiles in reference lakes suggest that deeper sections of sub-Arctic lakes are often less than the CCME threshold during the ice-covered period (Figures 3.1-1a and b). Thus, no mine effects were detected with respect to under-ice DO concentrations in monitored lakes in the King Cujo Watershed or Lac du Sauvage in 2013.

2013 temperature profiles in Cujo Lake and at site LdS1 in Lac du Sauvage were similar to those observed in previous years, with water temperature warming from the surface to the bottom of the lakes (Figure 4.1-1). Trends in monitored lakes were also similar to those observed in reference lakes (Figure 3.1-1). No mine effects were detected with respect to under-ice temperatures in the King-Cujo Watershed or Lac du Sauvage.

4.1.3.2 Secchi Depth

Secchi depth is a measure of light penetration in lakes, which is associated with water clarity. Thus, Secchi depth can be used as an indicator of changes in water quality or plankton density. No statistical analyses could be performed on Secchi depths because they are not replicated. Graphical analysis and best professional judgment were used to evaluate whether a significant change in Secchi depth occurred. If a change was detected, variables affecting underwater light conditions (e.g., phytoplankton biomass and density, total suspended solids, turbidity, and TOC) were analyzed to assess the underlying cause of the change in Secchi depth and determine whether mine effects were present. A value of ± 0.5 m was used as an estimate of error due to sampler bias for interpreting graphical results.

Secchi depths in 2013 were within the range of Secchi depths observed during the baseline sampling period in all monitored lakes (Figure 4.1-3). Thus, no mine effects were detected with respect to Secchi depths in the King-Cujo Watershed or Lac du Sauvage.

4.2 LAKE AND STREAM WATER QUALITY

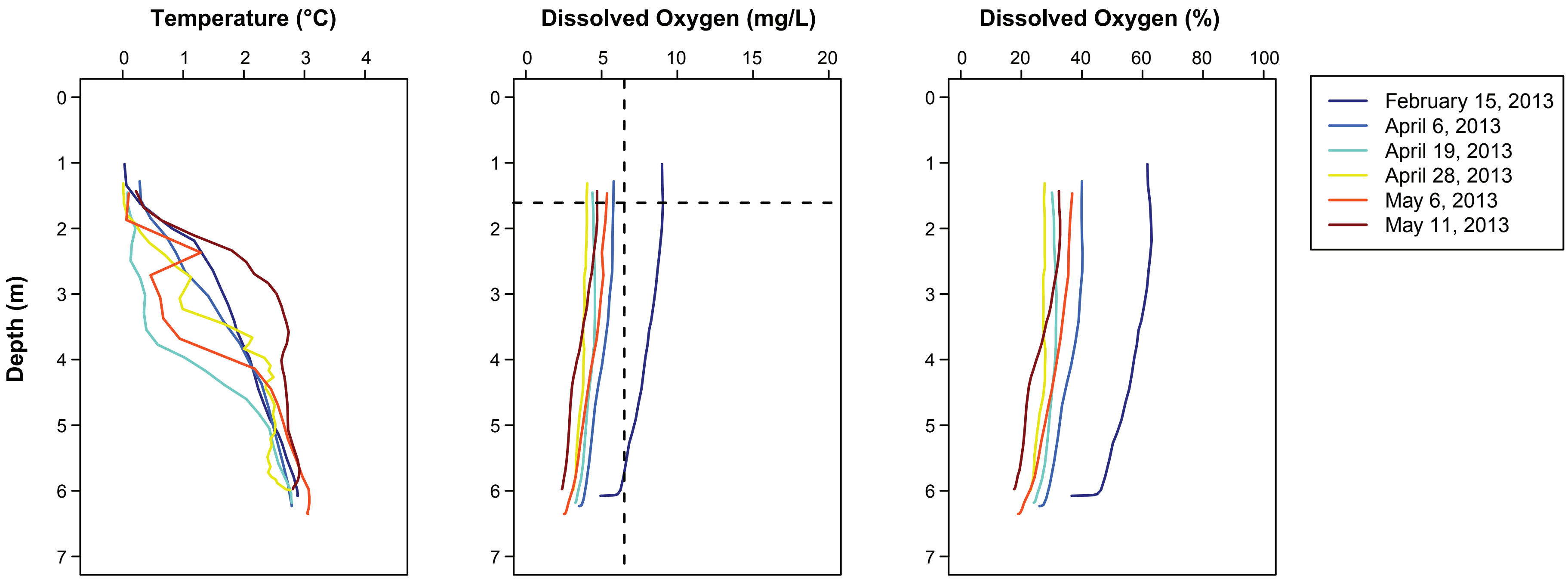
4.2.1 Variables

Twenty-three water quality variables were evaluated for potential mine effects in the King Cujo Watershed and Lac du Sauvage (see Section 3.2.1). CCME guidelines for the protection of aquatic life exist for 11 of the evaluated water quality variables, including pH, total ammonia-N, nitrite-N, total phosphate-P, total arsenic, total boron, total cadmium, total copper, total nickel, total selenium, and total uranium (see Section 2.3; CCME 2013). In addition, DDEC has established SSWQO for six variables, including chloride, sulphate, potassium, nitrate-N, total molybdenum, and total vanadium (see Section 2.3). Other water quality benchmark values also exist for total antimony, total barium and total strontium (see Table 2.3-1 in Section 2.3).

Figure 4.1-2

Dissolved Oxygen and Temperature Profiles for Cujo Lake, Ice-covered Season 2013

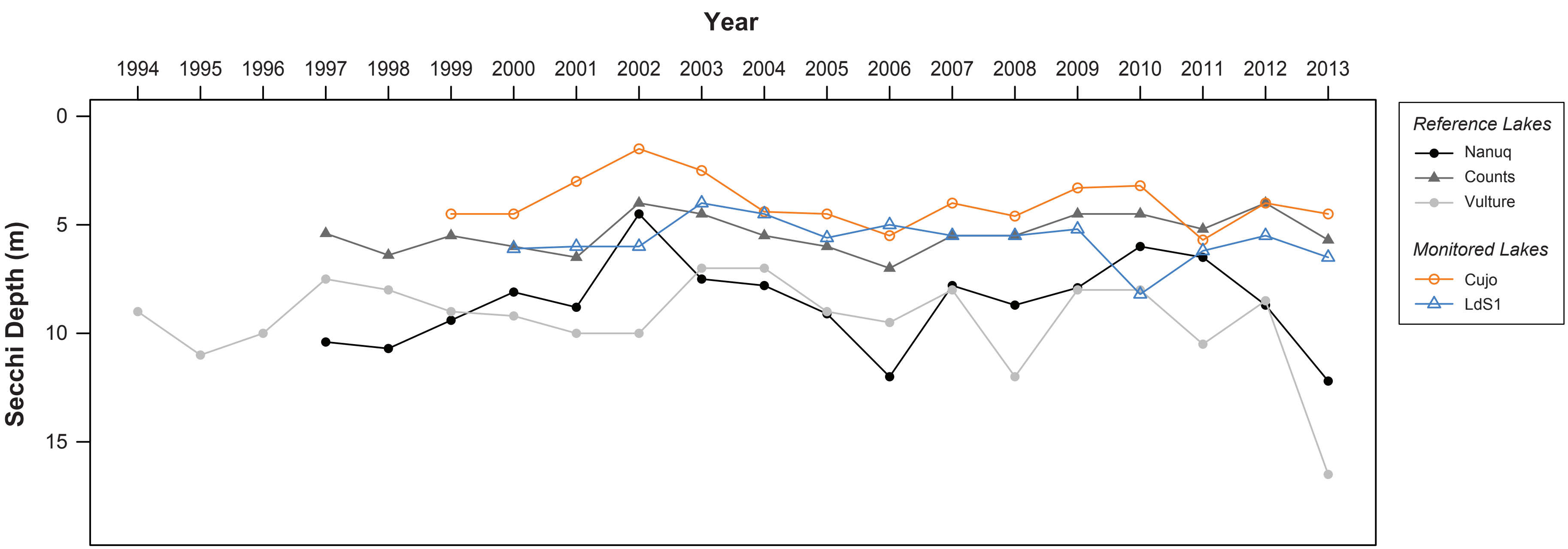
ERM



Note: Horizontal dashed line represents maximum ice thickness; Vertical dashed line represents the CCME guideline for dissolved oxygen (6.5 mg/L).

Figure 4.1-3

August Secchi Depths for King-Cujo Watershed Lakes and Lac du Sauvage, 1994 to 2013



Note: Data collected and supplied by DDEC.

4.2.2 Dataset

4.2.2.1 Lakes

Lake water quality data was collected during the ice-covered season from mid-April to mid-May and/or during the open water season from late July to mid-August of each year from 1998 to 2013 (Tables 4.2-1 and 4.2-2). Water was pumped from the KPSF into Cujo Lake from July 7, 2013 to July 12, 2013. Thus August 2013 samples are reflective of post-discharge water quality. A complete description of the datasets used in the Koala Watershed lakes and Lac de Gras evaluation of effects (i.e., sampling timing, frequency, replication, and laboratory analysis) is provided in Section 3.2.2 of this report. This description also applies to the King-Cujo Watershed lakes and Lac du Sauvage.

Table 4.2-1. Dataset Used for Evaluation of Effects on the April (Ice-covered) Water Quality in King-Cujo Watershed Lakes and Lac du Sauvage

Year	Nanuq	Counts	Vulture	Cujo	LdS1
1994*	-	-	-	-	-
1995*	-	-	-	-	-
1996*	-	-	Apr-18 (1)	-	-
1997*	-	-	-	-	-
1998	-	-	-	-	-
1999	-	-	-	-	-
2000	-	-	-	-	-
2001	-	-	-	-	-
2002	Apr-19 (4)	Apr-23 (4)	Apr-20 (4)	Apr-23 (4)	Apr-23 (4)
2003	Apr-12 (4)	Apr-13 (4)	Apr-14 (4)	Apr-13 (4)	Apr-13 (4)
2004	Apr-18 (4)	Apr-17 (4)	Apr-18 (4)	Apr-17 (4)	Apr-17 (4)
2005	Apr-24 (4)	Apr-24 (4)	Apr-24 (4)	Apr-24 (4)	Apr-25 (4)
2006	Apr-20 (4)	Apr-22 (4)	Apr-21 (4)	Apr-22 (4)	Apr-22 (4)
2007	Apr-21 (4)	Apr-24 (4)	Apr-22 (4)	Apr-27 (4)	May-1 (4)
2008	Apr-27 (4)	May-3 (4)	May-3 (4)	May-3 (4)	May-4 (2)
2009	May-11 (4), May-18 (4)	May-17 (4)	Apr-28 (4)	May-3 (4)	May-18 (4)
2010	Apr-14 (4)	Apr-14 (4)	Apr-12 (4)	Apr-16 (4)	Apr-14 (4)
2011	Apr 25 (4)	Apr 26 (4)	Apr 28 (4)	Apr 26 (4)	Apr 26 (4)
2012	Apr-20 (4)	Apr-17 (4)	Apr-18 (4)	Apr-15 (4)	Apr-19 (4)
2013	Apr-26 (4)	Apr-26 (4)	Apr-23 (4)	Apr-28 (2)	Apr-26 (4)

Dashes indicate no data were available.
Number of replicates is indicated in brackets.
** = Data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison.*

Baseline water quality data collected between 1994 and 1997 were not used in the statistical evaluation of effects, but are included in Tables 4.2-1 and 4.2-2 and illustrated graphically, below, for visual comparison. Station 1616-43 (KPSF) is not sampled during the ice-covered season as part of the AEMP and was not included in the April (ice-covered) regression analysis.

Table 4.2-2. Dataset Used for Evaluation of Effects on the August (Open Water) Water Quality in King-Cujo Watershed Lakes and Lac du Sauvage

Year	Nanuq	Counts	Vulture	1616-43	Cujo	LdS2	LdS1
1994*	-	-	Aug-13 (5)	-	-	-	-
1995*	-	-	Aug-9 (5)	-	-	-	-
1996*	-	-	Jul-26 (3)	-	-	-	-
1997*	Aug-4 (9)	Aug-14 (3)	Aug-5 (9)	-	-	-	-
1998	Jul-29 (6), Aug-11 (6)	Jul-29 (3), Aug-14 (3)	Jul-27 (3), Aug-10 (3)	-	-	-	-
1999	Aug-7 (6)	Aug-8 (6)	Aug-6 (6)	-	Aug-8 (6)	-	-
2000	Aug-4 (4)	Aug-1 (4)	Aug-4 (4)	-	Jul-31 (4)	Aug-2 (2)	Aug-2 (4)
2001	Aug-1 (4)	Jul-30 (4)	Aug-2 (4)	-	Jul-30 (4)	Jul-31 (2)	Jul-31 (4)
2002	Aug-1 (4)	Aug-7 (4)	Aug-3 (4)	Aug-6 (3)	Aug-7 (4)	Aug-5 (4)	Aug-5 (4)
2003	Aug-9 (3)	Aug-7 (2)	Aug-4 (2)	Aug-2 (2)	Aug-4 (2)	Aug-6 (2)	Aug-6 (2)
2004	Aug-10 (3)	Aug-12 (2)	Aug-9 (2)	Aug-11 (2), Aug-19 (2)	Aug-10 (2)	Aug-10 (2)	Aug-10 (2)
2005	Aug-1 (2)	Aug-7 (3)	Jul-31 (2)	Aug-2 (2)	Aug-9 (3)	Aug-9 (2)	Aug-9 (2)
2006	Aug-2 (3)	Aug-4 (2)	Aug-2 (2)	Jul-27 (2), Aug-7 (1)	Aug-4 (2)	Aug-3 (2)	Aug-1 (2)
2007	Aug-11 (6)	Aug-6 (6)	Aug-12 (6)	Aug-3 (2)	Aug-5 (6)	Aug-5 (3)	Aug-5 (6)
2008	Aug-8 (6)	Jul-31 (6)	Jul-29 (6)	Jul-27 (1), Aug-1 (2)	Jul-26 (6)	Jul-31 (3)	Jul-31 (6)
2009	Jul-30 (6)	Aug-1 (6)	Jul-30 (6)	Aug-3 (2), Aug-4 (1)	Jul-31 (6)	Aug-1 (3)	Aug-1 (6)
2010	Aug-5 (6)	Aug-7 (6)	Aug-5 (6)	Aug-1 (2), Aug-02 (2), Aug-16 (2), Aug3-26 (1), Aug-29 (1), Aug-31 (2)	Aug-4 (6)	Aug-4 (3)	Aug-4 (5)
2011	Aug-2 (6)	Aug-5 (6)	Aug-5 (6)	Jul-30 (2), Aug-16 (2)	Aug-4 (6)	Aug-4 (6)	Aug-4 (6)
2012	Aug-1 (6)	Aug-8 (6)	Aug-7 (6)	Aug-5 (2), Aug-31 (2)	Aug-6 (6)	Aug-7 (3)	Aug-7 (6)
2013	Aug-3 (6)	Aug-1 (6)	Aug-1 (6)	Jul-30 (2), Aug 26 (1)	Jul-30 (5)	Aug-3 (3)	Aug-3 (6)

Dashes indicate no data were available.
Number of replicates is indicated in brackets.
** = Data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison.*

For each variable, average values were calculated for April and August of each year by pooling data from all of the depths that were sampled in that month, assuming that all water columns were completely mixed. Laboratory analyses of water quality samples were conducted as described in Section 3.2-2 of this report. Some data were removed from the historical dataset due to sample contamination or laboratory difficulties in sample analysis (see Table 4.2-3).

Table 4.2-3. Data Removed from the Historical Lake and Stream Water Quality Dataset for the King-Cujo Watershed and Lac du Sauvage

Year	Date	Samples	Variables	Rationale
1999	August	Lakes and Streams	Total Metals	Contaminated nitric acid provided by laboratory
2001	August	Lakes and Streams	Ortho-phosphate	Unexplained contamination
2002	August 7	Counts (mid, rep 1)	Total Zinc	Unexplained contamination, > 6x replicate concentration
2005	April 24	Nanuq (mid, rep 1)	Total Copper	Unexplained contamination
2008	May 3	Vulture (mid, rep 1)	Sulphate, Chloride, TDS	Unexplained contamination
2008	August 2	Nanuq Outflow (rep 1)	pH	Much higher than the pH in all reference lakes samples collected in 2008
2009	May 3	Cujo (deep, rep 2)	Ortho-phosphate	Unexplained contamination
2010	-	-	-	-
2011	-	-	-	-
2012	-	-	-	-
2013	July 30	Cujo (shallow, rep 1)	All variables	Contamination detected based on Equipment Blank performed on July 30, 2013
2013	June 9	Christine-Lac du Sauvage (rep 2)	Total Phosphate-P	Identified as an extreme oulier

Dashes indicate no samples were removed.

4.2.2.2 Streams

Stream water quality was collected from late July to mid-August of each year from 1998 to 2013 (Table 4.2-4). Baseline water quality data from reference lakes, collected between 1994 and 1997, were not used in the statistical evaluation of effects, but are included in Table 4.2-4 and shown graphically for visual comparison, below.

Table 4.2-4. Dataset Used for Evaluation of Effects on the August Water Quality in King-Cujo Watershed Streams and Lac du Sauvage

Year	Nanuq Outflow	Counts Outflow	Vulture-Polar	1616-43	Cujo Outflow	Christine-Lac du Sauvage
1994*	-	-	Aug-4 (1)	-	-	-
1995*	-	-	Aug-10 (1)	-	-	-
1996*	-	-	Jul-27 (1)	-	-	-
1997	-	-	-	-	-	-
1998	Aug-18 (3)	Aug-18 (3)	Aug-16 (3)	-	-	-
1999	Aug-6 (3)	Aug-7 (3)	Aug-8 (3)	-	Aug-7 (3)	-
2000	Jul-30 (3)	Jul-30 (3)	Jul-30 (3)	-	Jul-30 (3)	Aug-3 (3)
2001	Aug-7 (3)	Aug-7 (3)	Aug-7 (3)	-	Aug-7 (3)	Aug-7 (3)
2002	Aug-6 (3)	Aug-6 (3)	Aug-6 (3)	Aug-6 (3)	Aug-6 (3)	Aug-6 (3)
2003	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)
2004	Aug-11 (2)	Aug-11 (2)	Aug-11 (2)	Aug-11 (2), Aug-19 (2)	Aug-11 (2)	Aug-11 (2)
2005	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)	Aug-2 (2)
2006	Jul-27 (2)	Jul-27 (2)	Jul-27 (2)	Jul-27 (2), Aug-7 (1)	Jul-27 (2)	Jul-27 (2)

(continued)

Table 4.2-4. Dataset Used for Evaluation of Effects on the August Water Quality in King-Cujo Watershed Streams and Lac du Sauvage (completed)

Year	Nanuq Outflow	Counts Outflow	Vulture-Polar	1616-43	Cujo Outflow	Christine-Lac du Sauvage
2007	Aug-3 (2)	Aug-3 (2)	Aug-4 (2)	Aug-3 (2)	Aug-4 (2)	Aug-4 (2)
2008	Aug-2 (2)	Aug-1 (2)	Aug-2 (2)	Jul-27 (1), Aug-1 (2)	Aug-1 (2)	Aug-1 (2)
2009	Aug-3 (2)	Aug-3 (2)	Aug-4 (2)	Aug-3 (2), Aug-4 (1)	Aug-3 (2)	Aug-3 (2)
2010	Aug-1 (2)	Aug-1 (2)	Aug-1 (2)	Aug-1 (2), Aug-2 (2), Aug-16 (1), Aug-26 (1), Aug-29 (1), Aug-31 (2)	Aug-1 (2)	Aug-1 (2)
2011	Jul-30 (2)	Jul-30 (2)	Jul-31 (2)	Jul-30 (2), Aug-16 (2)	Jul-30 (2)	Jul-30 (2)
2012	Aug-4 (2)	Aug-5 (2)	Aug-5 (2)	Aug-5 (2), Aug-31 (2)	Aug-5 (2)	Aug-4 (2)
2013	Aug-4 (2)	Aug-4 (2)	Aug-4(2)	Jul-30 (2), Aug 26 (1)	Aug-4 (2)	Aug-4 (2)

Dashes indicate no data were available
Number of replicates is indicated in brackets
** = Data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison*

The number of replicate samples collected at stream sites has varied over the course of AEMP monitoring. One replicate sample was collected with 10% duplication in each stream from 1994 to 1997 and three replicate samples were collected from 1998 through 2002. Between 2003 and 2013, two replicate samples were collected at each site.

Stream water quality samples were analyzed as described in Section 3.2.2.1. Data has been removed from the stream water quality historical dataset due to sample contamination or laboratory difficulties in sample analysis (see Table 4.2-3).

4.2.3 Statistical Description of Results

Table 4.2-5 summarizes the reference and monitored lakes and streams that were sampled in the King-Cujo Watershed during each sampling period.

Table 4.2-5. Summary of Reference and Monitored Lakes and Streams Sampled in the King-Cujo Watershed and Lac du Sauvage in 2013

Watershed	Month	Lake / Stream	Reference Lakes / Streams Sampled	Monitored Lakes / Streams Sampled
King-Cujo	April	Lake	Nanuq, Counts, Vulture	Cujo, LdS1
	August	Lake	Nanuq, Counts, Vulture	1616-43 (KPSF)[1], Cujo, LdS2, LdS1
	August	Stream	Nanuq Outflow, Counts Outflow, Vulture Polar	1616-43 (KPSF)[1], Cujo Outflow, Christine-LdS

[1] 1616-43 is monitored as part of the SNP and AEMP

Although a complete description of the statistical results for each variable and sampling month is provided in Part 3 – Statistical Results, it was still necessary to provide the statistical summaries in order to support effects conclusions. Thus the results and discussion of each variable includes a table summarizing the best fit model (LME or tobit) for each variable in the reference and monitored lakes and streams that were sampled in the King-Cujo Watershed and Lac du Sauvage in April (lakes only) and August. The statistical evaluation of effects for each variable follows the model selection process outlined in detail in Section 2.2.4 and Figure 2.2-2. A brief recapitulation of the process is provided here:

- Model fit = 1a was selected whenever more than 60% of the observations in all reference sites were less than detection limits or whenever both the slopes and intercepts of the temporal trends differed among reference sites. Monitored sites were compared to a constant slope of 0.
- Model fit = 1b was selected whenever both the slopes and intercepts of the temporal trends differed among reference sites *and* the trend in monitored sites differed from a constant slope of 0. Monitored sites were compared to the slopes of individual reference sites.
- Model fit = 2 was selected whenever slopes were similar, but intercepts differed, among reference sites. Monitored sites were compared to the common slope of the reference sites; intercepts were ignored.
- Model fit = 3 was selected whenever the slopes and intercepts of the temporal trends were similar among reference sites, unless AIC weights suggested that the reference lakes were better modeled with a separate intercepts and/or slopes. Monitored sites were first compared to the common slope and intercept of the reference sites and then to a reduced model that allowed for differences in intercepts but retained a common slope.

A table describing the model fit selected and the data that was excluded, if any, is included for each variable.

4.2.4 Results and Discussion

4.2.4.1 pH

Summary: Statistical and graphical analyses suggest that pH has increased at all sites downstream of the KPSF as far as Christine-Lac du Sauvage Stream as a result of mine operations, but has stabilized since 2004. Although the lower 95% confidence interval of the 2013 fitted mean was less than the lower CCREM guideline value (pH 6.5) at site LdS1 during both the ice-covered and open water season and at site LdS2 during the open water season, the lower 95% confidence interval of fitted mean pH in all three reference lakes and streams was also less than lower CCREM guideline value in 2013.

Statistical analyses indicate that pH has changed through time, relative to reference lakes and streams, in Cujo Lake, Cujo Outflow, and Christine-Lac du Sauvage Stream (Table 4.2-6). Graphical analysis also suggests that pH has increased through time in all monitored lakes and streams downstream of the KPSF as far as Christine-Lac du Sauvage Stream, but has stabilised since 2004 (Figure 4.2-1). The recent stability in pH may be related to the suspension of open pit mining operations in Misery Pit in April of 2005, though discharge from the KPSF into the receiving environment has continued.

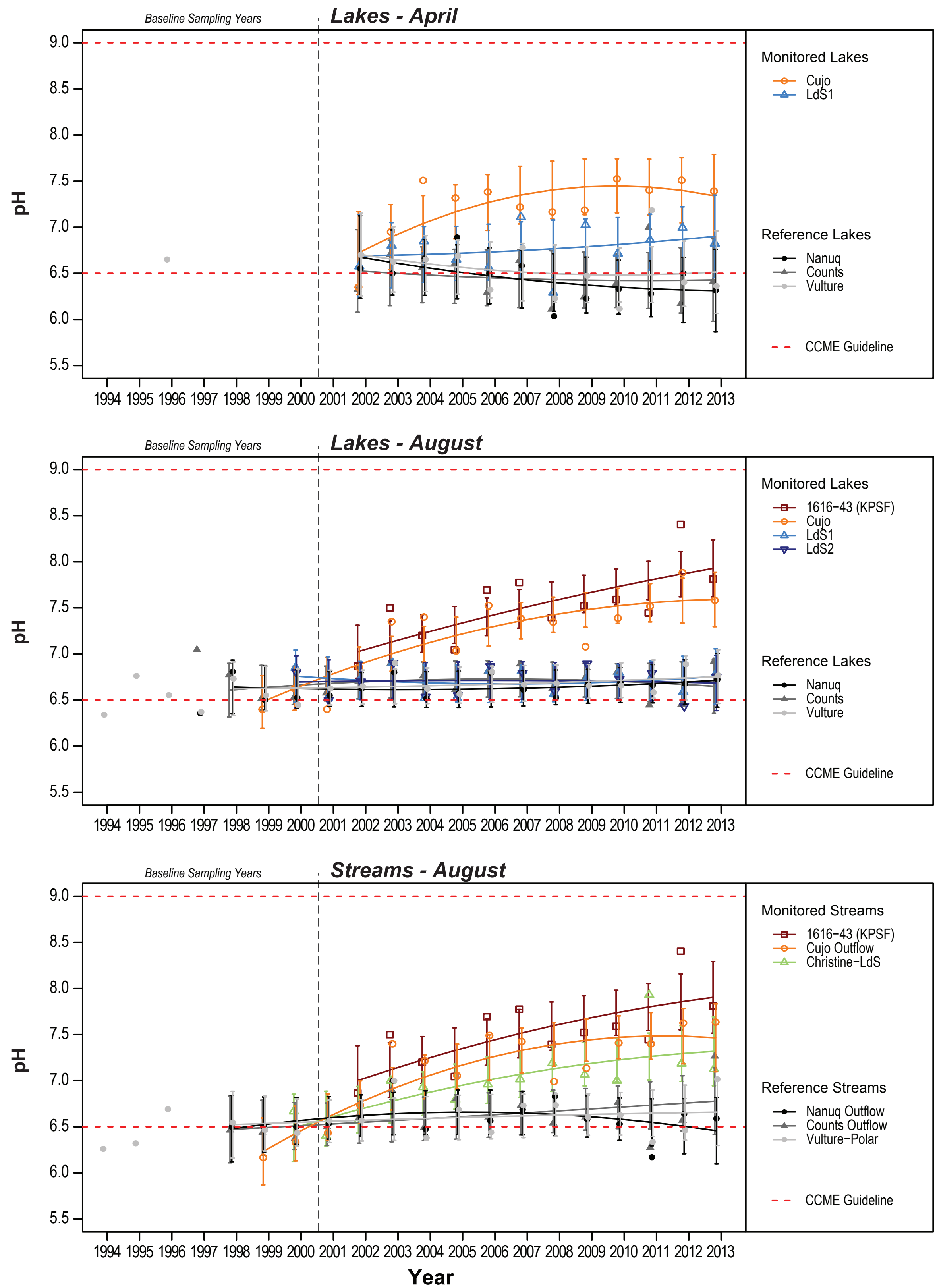
Table 4.2-6. Statistical Results of pH in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	2	-	Cujo		2-1
Aug	Lake	-	LME	3	1616-43 (KPSF), Cujo	1616-43 (KPSF), Cujo	-	2-6
Aug	Stream	-	LME	3	1616-43 (KPSF), Cujo Outflow, Christine- Lac du Sauvage	1616-43 (KPSF), Cujo Outflow, Christine- Lac du Sauvage		2-12

Dashes indicate not applicable.

Figure 4.2-1

Observed and Fitted Means for pH in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
CCME guideline = 6.5 - 9.0

The 2013 lower 95% confidence interval of the fitted mean pH was less than the lower CCREM guideline value (pH 6.5) at site LdS1 in Lac du Sauvage (CCREM 1987). However, the lower 95% confidence interval around the fitted mean pH also was less than the lower CCREM guideline value in all three reference lakes and streams during both the ice-covered and open water seasons. Observed pH in Cujo Lake, Cujo Outflow, and Christine-Lac du Sauvage Stream was within the CCREM guideline values during the ice-covered and open water seasons in 2013 (see Part 2 - Data Report; CCREM 1987).

4.2.4.2 Total Alkalinity

Summary: Statistical and graphical analyses suggest that total alkalinity has increased in lakes and streams downstream of the KPSF as far as Christine-Lac du Sauvage Stream as a result of mine operations. Concentrations have stabilised at most sites in recent years.

Statistical analyses indicate that total alkalinity has changed through time, relative to reference lakes, in all monitored lakes and streams downstream from the KPSF as far as Christine-Lac du Sauvage Stream (Table 4.2-7). Graphical analysis suggests that total alkalinity has increased through time at these sites, but has stabilised since 2005 (Figure 4.2-2). Together, graphical and statistical analyses indicate that total alkalinity has increased downstream of the KPSF as far as Christine-Lac du-Sauvage Stream as a result of mine operations. The recent stability in total alkalinity at concentrations above baseline values in many of the King-Cujo lakes and streams likely results from the suspension of open pit mining in Misery Pit in April 2005, though discharge from the KPSF into the receiving environment has continued.

Table 4.2-7. Statistical Results of Total Alkalinity in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	2	-	Cujo		2-18
Aug	Lake	-	Tobit	2	-	1616-43 (KPSF), Cujo	-	2-23
Aug	Stream	-	Tobit	2	-	1616-43 (KPSF), Cujo Outflow, Christine-Lac du Sauvage	-	2-29

Dashes indicate not applicable.

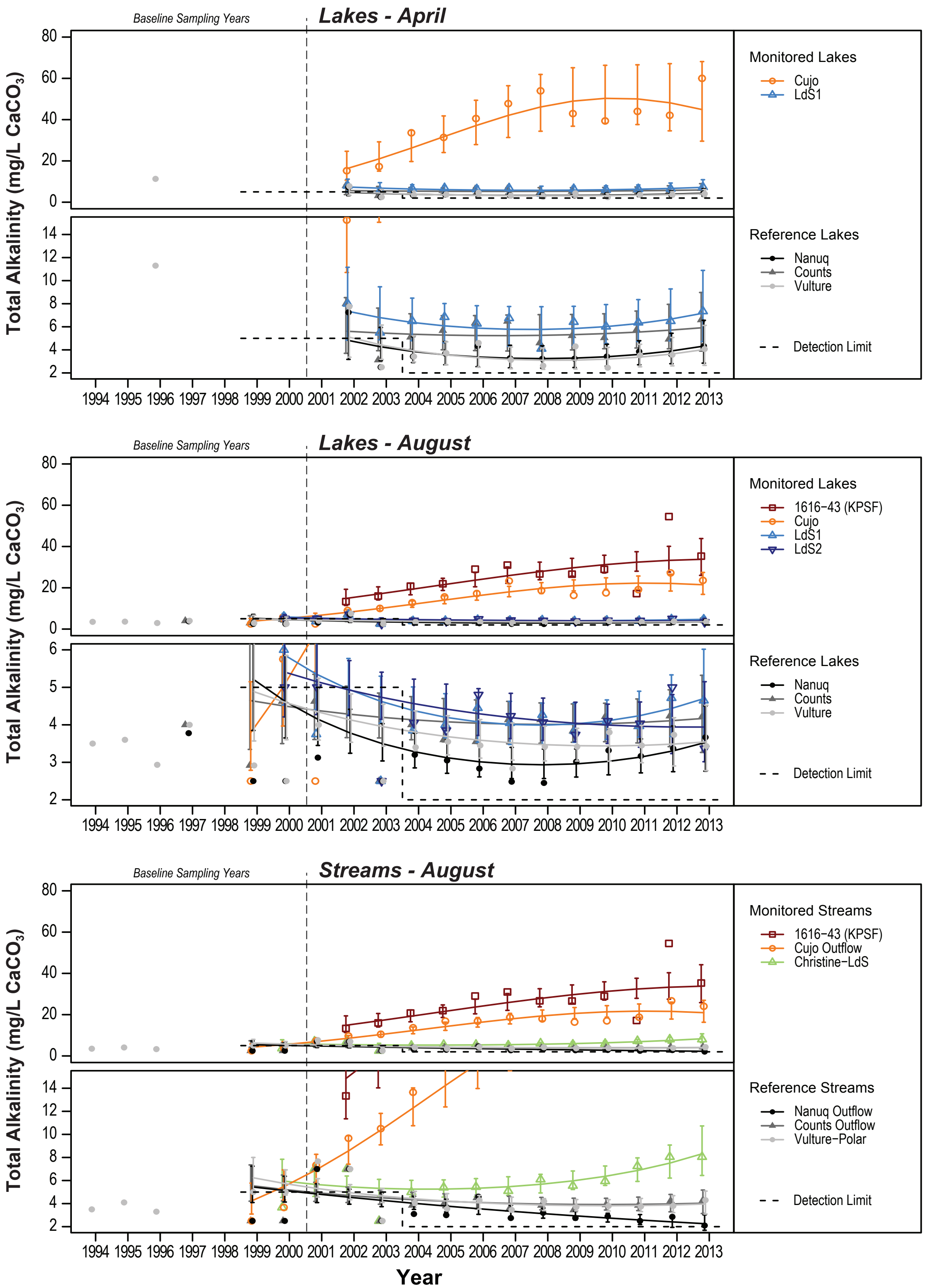
4.2.4.3 Water Hardness

Summary: Statistical and graphical analyses suggest that water hardness has increased at all sites downstream of the KPSF as far as Christine-Lac du Sauvage Stream as a result of mine operations. Concentrations have stabilised at most sites during the open water season in recent years.

Statistical and graphical analyses indicate that water hardness has increased through time, relative to reference lakes and streams, at all sites downstream of the KPSF as far as Christine-Lac du Sauvage Stream (Table 4.2-8). However, graphical analysis also suggests that water hardness has stabilised in Cujo Lake and Cujo Outflow during the open water season in recent years (Figure 4.2-3). The recent stability in water hardness at most sites during the open water season is likely related to the suspension of open pit mining operations in Misery Pit in April of 2005, though discharge from the KPSF into the receiving environment has continued.

Figure 4.2-2

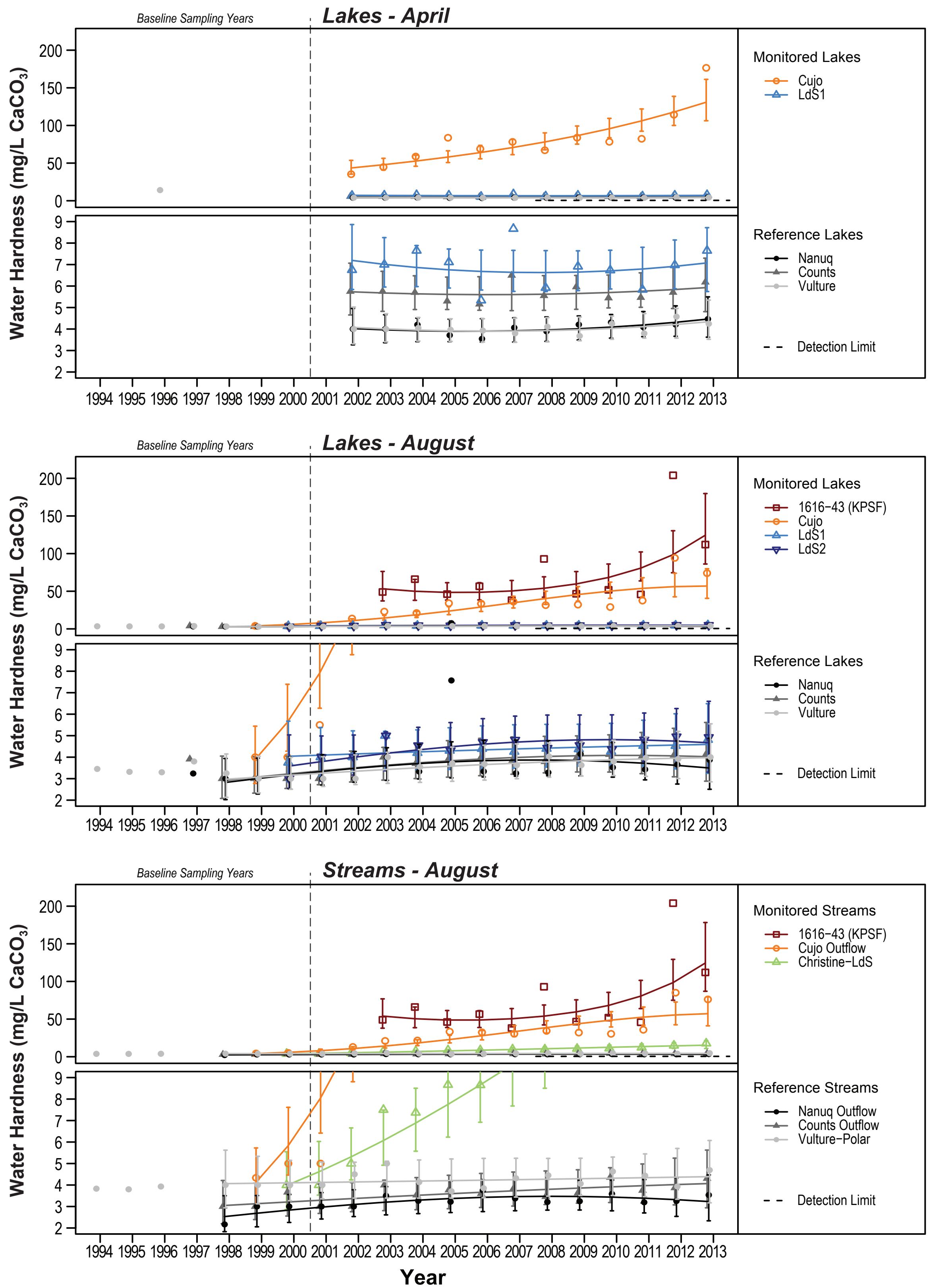
Observed and Fitted Means for Total Alkalinity in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.

Figure 4.2-3

Observed and Fitted Means for Water Hardness in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.

Table 4.2-8. Statistical Results of Water Hardness in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	2	-	Cujo		2-35
Aug	Lake	-	LME	3	1616-43 (KPSF), Cujo, LdS1, LdS2	1616-43 (KPSF), Cujo	-	2-41
Aug	Stream	-	LME	2	-	Cujo Outflow, Christine-Lac du Sauvage		2-47

Dashes indicate not applicable.

4.2.4.4 Chloride

> ***Summary:*** Statistical and graphical analyses suggest that chloride concentrations have increased in all monitored lakes and streams downstream of the KPSF as far as Cujo Outflow as a result of mine operations. The 95% confidence intervals around fitted mean and observed chloride concentrations were less than the hardness-dependent chloride SSWQO in all monitored lakes and streams in 2013.

Statistical analyses indicate that chloride concentrations have changed through time at site LdS1 in Lac du Sauvage during the ice-covered season and in Cujo Lake and Cujo Outflow during the open water season (Table 4.2-9). Graphical analysis suggests that chloride concentrations have decreased through time at site LdS1 in Lac de Gras, likely as a result of a reduction in detection limits rather than actual decreases in chloride concentrations (Figure 4.2-4). Graphical analysis also suggests that chloride concentrations in Cujo Lake and Cujo Outflow have increased through time during the open water season (Figure 4.2-4). Although no temporal trend was statistically identified in Cujo Lake during the ice-covered season, the observed mean chloride concentration in 2013 was greater than in any previous year (Figure 4.2-4). Together, graphical and statistical analyses suggest that chloride concentrations have increased at sites downstream of the KPSF as far as Cujo Outflow as a result of mine activities.

Table 4.2-9. Statistical Results of Chloride Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

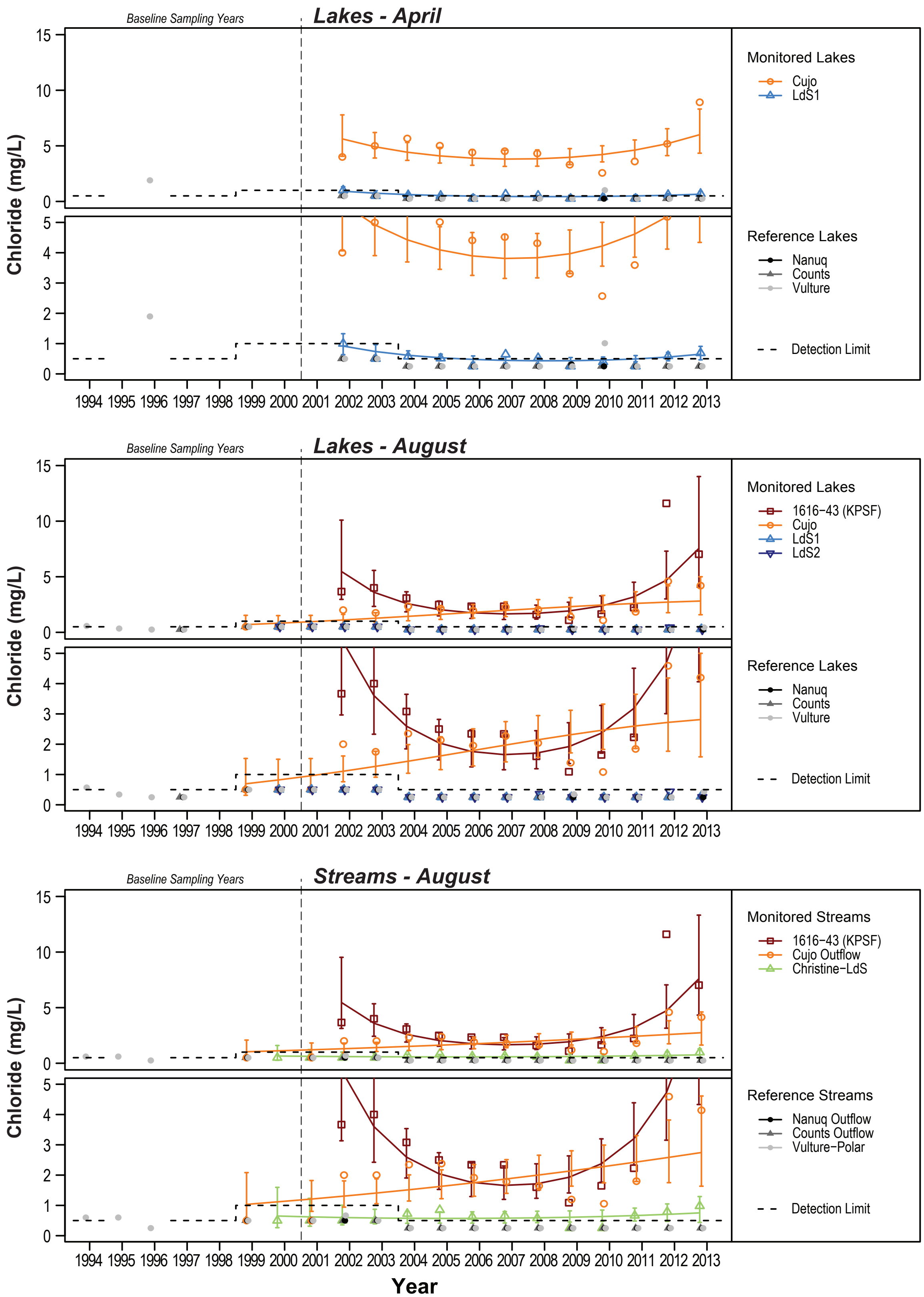
Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	Counts Nanuq Vulture	Tobit	1a	-	-	LdS1	2-52
Aug	Lake	LdS1, LdS2, Counts, Nanuq, Vulture	Tobit	1a	-	-	1616-43 (KPSF), Cujo	2-56
Aug	Stream	Counts Outflow, Nanuq Outflow, Vulture Outflow	Tobit	1a	-	-	1616-43 (KPSF), Cujo Outflow	2-60

Dashes indicate not applicable.

The 95% confidence intervals around the fitted mean and the observed mean chloride concentrations were less than the hardness-dependent chloride SSWQO in all monitored lakes and streams in 2013 (Elphick, Bergh, and Bailey 2011). Chloride concentrations were also less than the SSWQO in all monitored streams in June, July, August and September 2013 (see Part 2 – Data Report).

Figure 4.2-4

Observed and Fitted Means for Chloride Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
SSWQO = 116.6 X ln(Hardness) - 204.1, where hardness = 10 - 160 mg/L.

4.2.4.5 Sulphate

Summary: Statistical and graphical analyses suggest that sulphate concentrations have increased in all monitored lakes and streams downstream of the KPSF as far as Christine-Lac du Sauvage Stream as a result of mine operations. Observed and fitted mean concentrations were less than the hardness-dependent sulphate SSWQO at all sites in 2013.

Statistical analyses indicate that sulphate concentrations have changed through time, relative to reference lakes and streams, in Cujo Lake, Cujo Outflow, and Christine-Lac du Sauvage Stream (Table 4.2-10). Graphical analysis suggests that sulphate concentrations have increased at all monitored sites downstream from the KPSF as far as Christine-Lac du Sauvage Stream (Figure 4.2-5). Increased sulphate concentrations downstream of the KPSF likely reflect increased concentrations within the KPSF in recent years (Figure 4.2-5). Together, graphical and statistical analyses suggest that sulphate concentrations have increased in all lakes and streams downstream of the KPSF as far as Christine-Lac du Sauvage Stream as a result of mine activities.

Table 4.2-10. Statistical Results of Sulphate Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	2	-	Cujo	-	2-64
Aug	Lake	-	LME	3	1616-43 (KPSF), Cujo	1616-43 (KPSF), Cujo	-	2-70
Aug	Stream	-	LME	3	1616-43 (KPSF), Cujo Outflow, Christine-Lac du Sauvage	Cujo Outflow, Christine-Lac du Sauvage	-	2-76

Dashes indicate not applicable.

The 95% confidence intervals around the fitted mean and the observed mean sulphate concentrations were less than the hardness-dependent sulphate SSWQO in all reference and monitored lakes and streams in 2013 (Rescan 2012f). Sulphate concentrations were also less than the SSWQO in all monitored streams in June, July, August and September 2013 (see Part 2 - Data Report; Rescan 2012f).

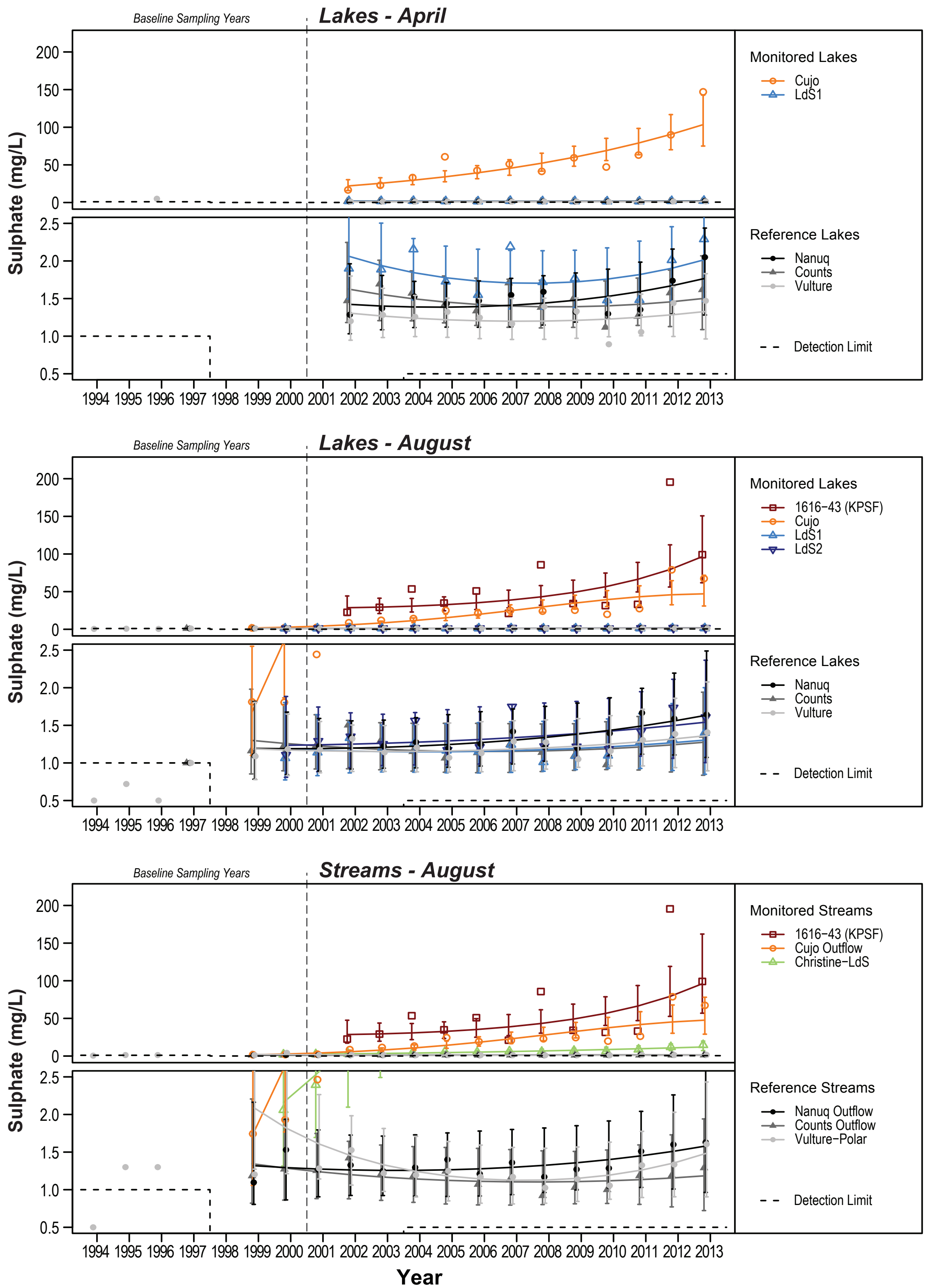
4.2.4.6 Potassium

Summary: Statistical and graphical analyses suggest that potassium concentrations have increased in all monitored lakes and streams downstream of the KPSF as far as Christine-Lac du Sauvage Stream as a result of mine operations. Potassium concentrations were less than the potassium SSWQO at all monitored sites in 2013.

Statistical analyses suggest that potassium concentrations have changed through time, relative to reference sites, at all sites downstream of the KPSF as far as Christine-Lac du Sauvage Stream (Table 4.2-11). Graphical analysis suggests that potassium concentrations have increased at all sites downstream of the KPSF as far as Christine-Lac du Sauvage Stream and decrease with downstream distance from the KPSF (Figure 4.2-6). Although potassium concentrations have stabilised in Cujo Lake and Cujo Outflow during the open water season in recent years, concentrations have continued to rise in Cujo Lake during the ice-covered season and in Christine-Lac du Sauvage Stream during the open water season (Figure 4.2-6). Together, statistical and graphical analyses suggest that potassium concentrations have increased at all sites downstream of the KPSF as far as Christine-Lac du Sauvage Stream as a result of mine operations.

Figure 4.2-5

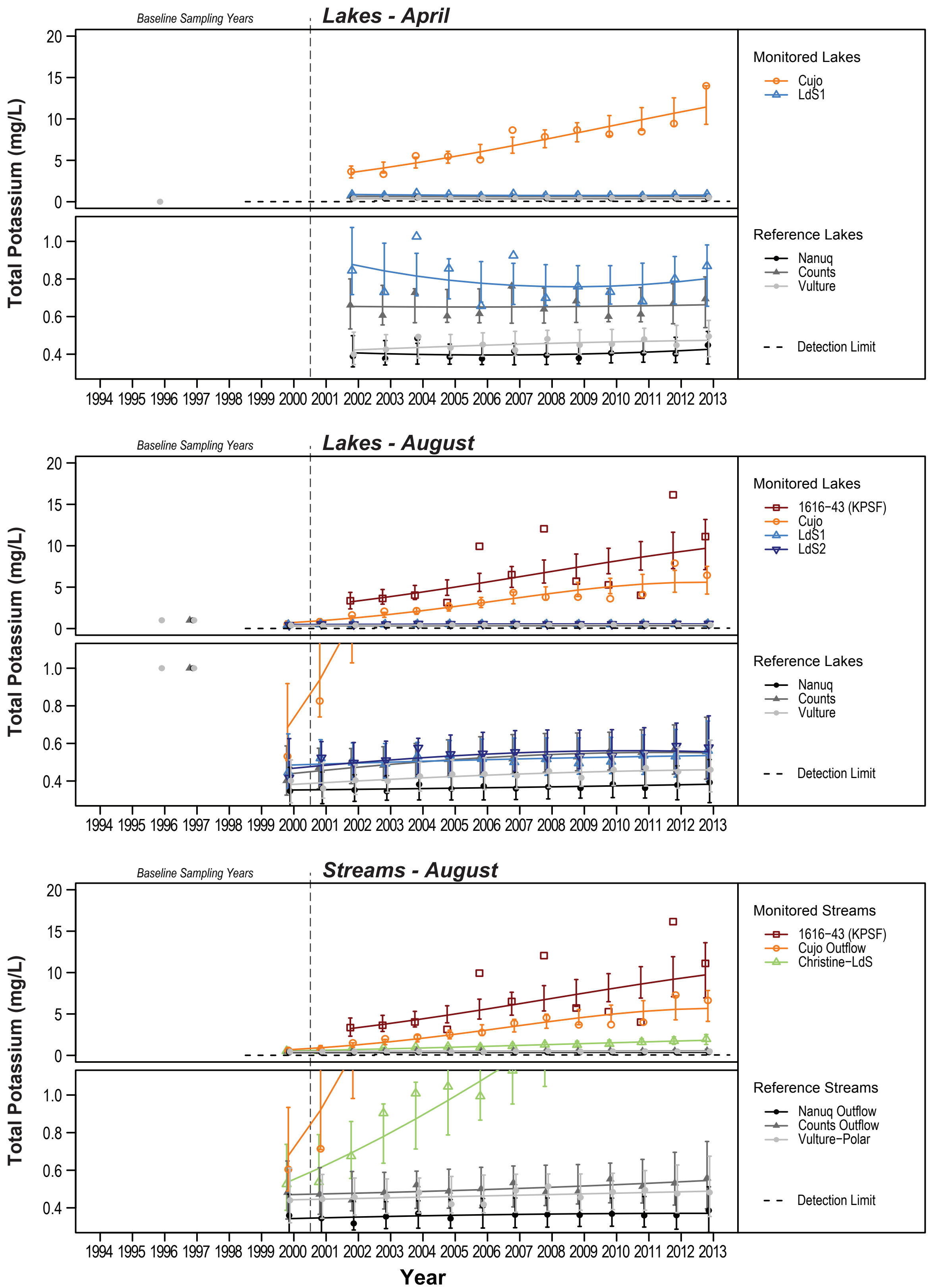
Observed and Fitted Means for Sulphate Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
SSWQO = $e^{(0.9116 \times \ln(Hardness) + 1.712)}$ mg/L, where hardness < 160 mg/L.

Figure 4.2-6

Observed and Fitted Means for Potassium Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
SSWQO = 41 mg/L.

Table 4.2-11. Statistical Results of Potassium Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	3	Cujo	Cujo	-	2-81
Aug	Lake	-	LME	2	-	1616-43 (KPSF), Cujo	-	2-87
Aug	Stream	-	LME	2	-	1616-43 (KPSF), Cujo Outflow, Christine-Lac du Sauvage	-	2-93

Dashes indicate not applicable.

The 95% confidence intervals around the fitted and the observed mean potassium concentrations were less than the long-term potassium SSWQO (41 mg/L) in all monitored lakes and streams (Rescan 2012g). Potassium concentrations in all monitored streams in June, July, August and September 2013 were also less than the long-term potassium SSWQO (see Part 2 - Data Report; Rescan 2012g).

4.2.4.7 Total Ammonia-N

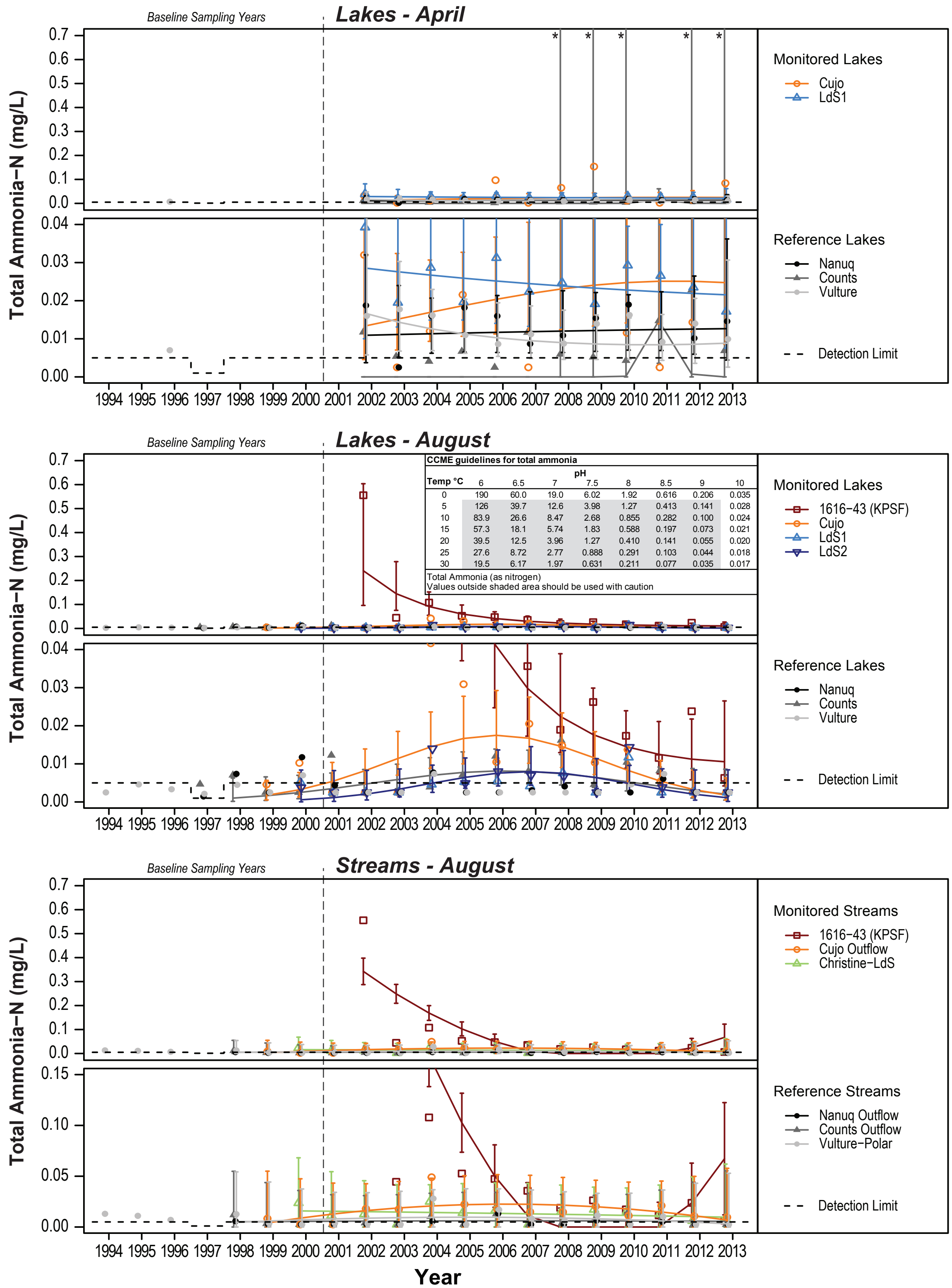
Summary: Statistical and graphical analyses suggest that total ammonia-N concentrations had previously increased in Cujo Lake as a result of mine operations, but have returned to baseline and reference concentrations in recent years. The 95% confidence interval around the fitted mean and observed mean total ammonia-N concentrations were less than the pH- and temperature-dependent ammonia CCME guideline value in all monitored lakes and streams during both the ice-covered and open water seasons in 2013.

Statistical analyses indicate that total ammonia-N concentrations have been stable in all monitored lakes and streams during the ice-covered and open water seasons (Table 4.2-12). Graphical analysis suggests that open water season total ammonia-N concentrations increased to a peak around 2005 in Cujo Lake and have since declined to baseline and reference concentrations (Figure 4.2-7). The recent stability in total ammonia-N concentrations at most sites during the open water season is likely related to the suspension of open pit mining operations in Misery Pit in April of 2005, though discharge from the KPSF into the receiving environment has continued. In addition, total ammonia-N concentrations measured during seepage surveys of the Waste Rock Storage Area (WRSA) have displayed a clear decreasing trend after 2005, when a 5 m deep granite cap was placed over the Misery WRSA to encapsulate acid-producing rocks and some of the stockpiled kimberlite was removed from the area (SRK 2010). Together, statistical and graphical analyses suggest that total ammonia-N concentrations in Cujo Lake have been affected by mine operations historically, but have returned to baseline concentrations in recent years.

The 95% confidence intervals around fitted mean total ammonia-N concentrations were less than the pH- and temperature-dependent ammonia CCME guideline in all monitored lakes and streams in 2013 (CCME 2001). Total concentrations in all monitored streams in June, July, August and September 2013 were also less than pH- and temperature-dependent ammonia CCME guideline (see Part 2 - Data Report; CCME 2001).

Figure 4.2-7

Observed and Fitted Means for Total Ammonia-N Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Temp °C	pH 6	6.5	7	7.5	8	8.5	9	10
0	190	60.0	19.0	6.02	1.92	0.616	0.206	0.035
5	126	39.7	12.6	3.98	1.27	0.413	0.141	0.028
10	83.9	26.6	8.47	2.68	0.855	0.282	0.100	0.024
15	57.3	18.1	5.74	1.83	0.588	0.197	0.073	0.021
20	39.5	12.5	3.96	1.27	0.410	0.141	0.055	0.020
25	27.6	8.72	2.77	0.888	0.291	0.103	0.044	0.018
30	19.5	6.17	1.97	0.631	0.211	0.077	0.035	0.017

Total Ammonia (as nitrogen)
Values outside shaded area should be used with caution

Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
WL = Maximum average concentration premitted in water licence W2009L2-0001.
WL Criterion = 2.0 mg/L. CCME Guideline is pH and temperature dependent (see inset table).
** Upper 95% Confidence Interval on the fitted mean of Counts Lake in April 2008 = 4.64x10^{272} mg/L, 2009 = 1.85x10^{190} mg/L, 2010 = 5.86x10^{98} mg/L, 2012 = 1.82x10^{105} mg/L, and 2013 = 9.34x10^{214} mg/L.*

Table 4.2-12. Statistical Results of Total Ammonia-N Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	Tobit	1b	-	-	None	2-99
Aug	Lake	LdS1, Nanuq, Vulture	Tobit	1b	-	-	1616-43 (KPSF)	2-104
Aug	Stream	Nanuq Outflow	Tobit	3	1616-43 (KPSF)	1616-43 (KPSF)	-	2-109

Dashes indicate not applicable.

4.2.4.8 Nitrite-N

Summary: Nitrite-N concentrations have generally been below detection limits at all sites since monitoring began. Observed concentrations were less than the nitrite-N CCREM guideline at all sites in 2013. No mine effects were detected.

More than 60% of nitrite-N concentration measurements have been below detection limits in all lakes and streams in the King-Cujo Watershed since monitoring began (Table 4.2-13; Figure 4.2-8). No statistical analyses could be performed and no mine effects were detected. The low concentrations of nitrite-N are likely related to low concentrations of total ammonia-N in the King-Cujo Watershed, since nitrite is primarily formed through the oxidisation of ammonia (Figure 4.2-9). Moreover, nitrite is a relatively transient form of nitrogen, which quickly oxidises to produce nitrate.

Table 4.2-13. Statistical Results of Nitrite-N Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	ALL	-	-	-	-	-	2-114
Aug	Lake	ALL	-	-	-	-	-	2-116
Aug	Stream	ALL	-	-	-	-	-	2-118

Dashes indicate not applicable.

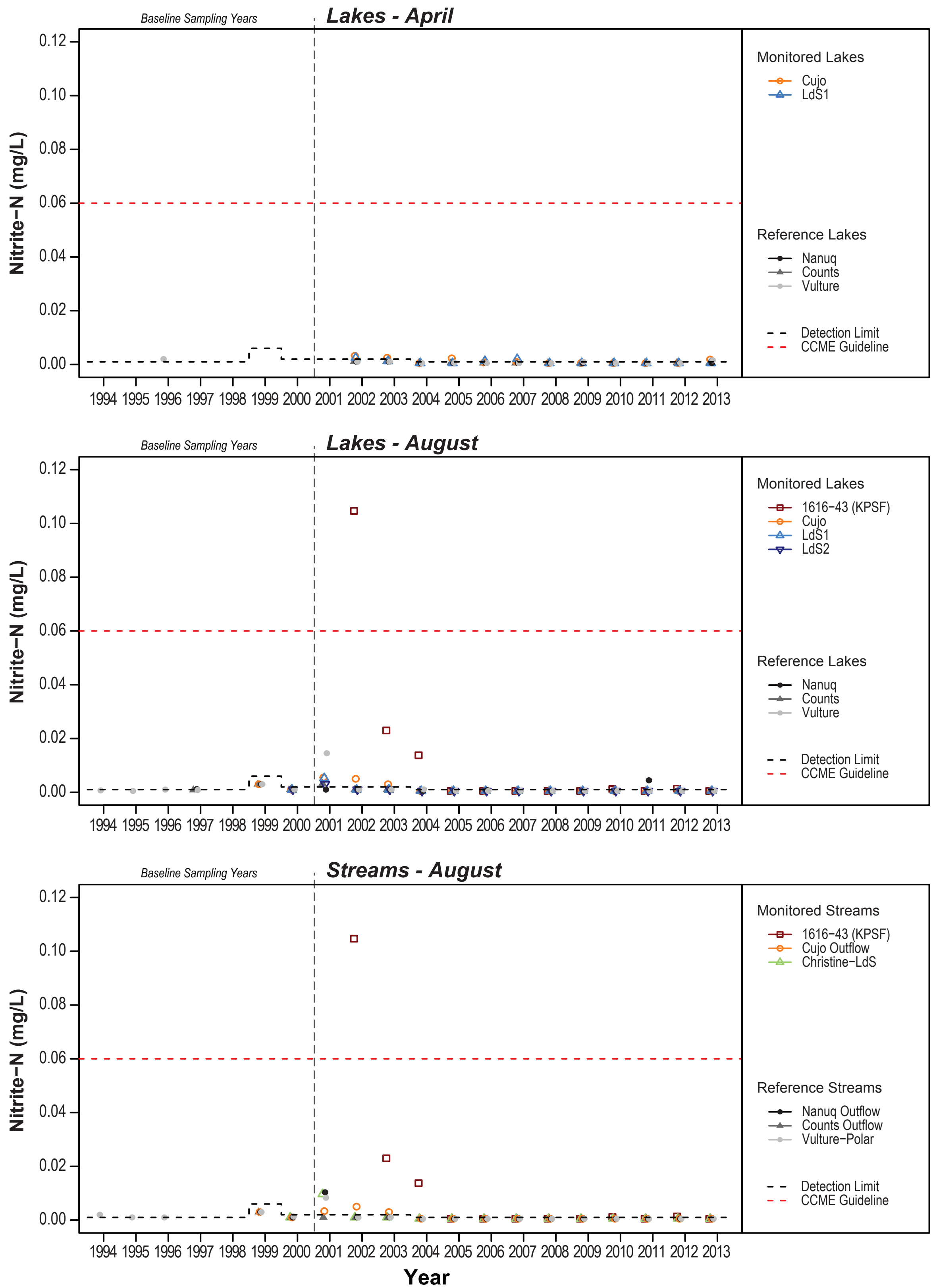
All 2013 observed means were less than the CCREM water quality guideline value for nitrite-N (0.06 mg/L; see Part 2 - Data Report; CCREM 1987). Together, statistical and graphical analyses indicate that mine activities have had no effect on nitrite-N concentrations in the King-Cujo Watershed or Lac du Sauvage. Nitrite-N concentrations in all monitored streams in June, July, August and September 2013 were also less than nitrite-N CCREM guideline (see Part 2 - Data Report; CCREM 1987).

4.2.4.9 Nitrate-N

Summary: Nitrate-N concentrations have declined through time in Cujo Lake during the ice-covered season and have generally been below detection limits during the open water season and at all sites in the King-Cujo Watershed and Lac du Sauvage. Observed and fitted mean concentrations were less than the hardness-dependent nitrate-N SSWQO at all sites in 2013. No mine effects were detected.

Figure 4.2-8

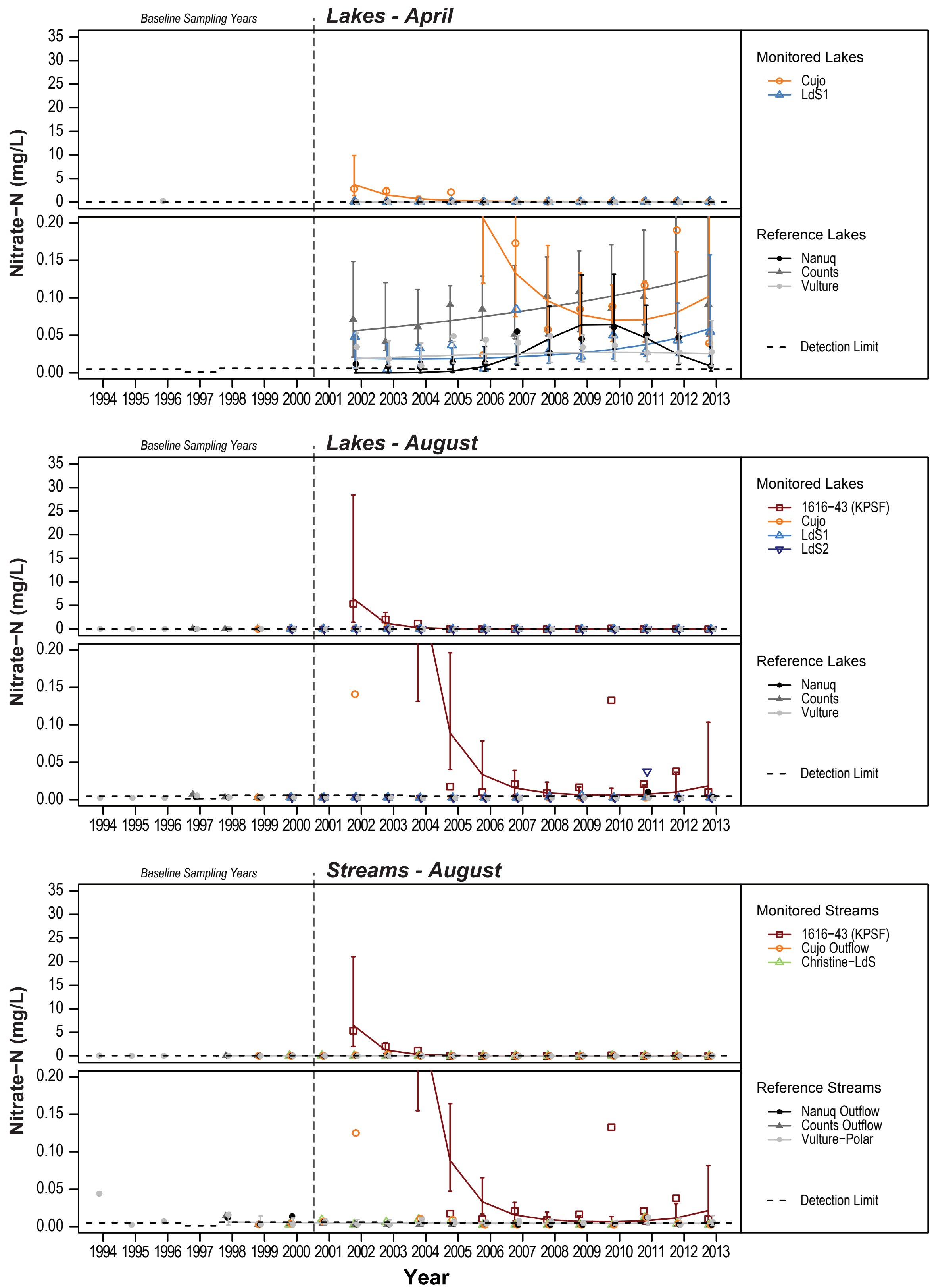
Observed Means for Nitrite-N Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
CCME Guideline = 0.06 mg/L.

Figure 4.2-9

Observed and Fitted Means for Nitrate-N Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
SSWQO = $e^{0.9518 \times \ln(Hardness) - 2.032}$ mg/L, where hardness < 160mg/L.

Statistical analyses indicate that nitrate-N concentrations have changed through time, relative to reference lakes, in Cujo Lake during the ice-covered season (Table 4.2-14). Graphical analysis suggests that nitrate-N concentrations have declined through time in Cujo Lake during the ice-covered season (Figure 4.2-9). The decline in nitrate-N concentrations observed during the ice-covered season may be related to the suspension of open pit mining in Misery Pit in 2005, though discharge from the KPSF into the receiving environment has continued. Declines in total nitrate-N likely reflect decreasing total ammonia-N concentrations, since ammonia oxidises to nitrite, which then oxidises to nitrate. Together, statistical and graphical analyses suggest that mine activities have had no effect on nitrate-N concentrations in the King-Cujo Watershed or Lac du Sauvage.

Table 4.2-14. Statistical Results of Nitrate-N Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	Tobit	2	-	Cujo	-	2-120
Aug	Lake	Cujo, LdS1, LdS2, Counts, Nanuq, Vulture	Tobit	1a	-	-	1616-43 (KPSF)	2-125
Aug	Stream	Cujo Outflow, Christine-Lac du Sauvage, Counts Outflow, Nanuq Outflow	Tobit	1b	-	-	1616-43 (KPSF)	2-129

Dashes indicate not applicable.

The 95% confidence intervals of the fitted mean and the observed mean nitrate-N concentrations were less than the hardness-dependent nitrate-N SSWQO in all reference and monitored lakes and streams (Health Canada 1987; Rescan 2012e). Nitrate-N concentrations in all monitored streams in June, July, August and September 2013 were also less than nitrate-N SSWQO (see Part 2 - Data Report; Health Canada 1987; Rescan 2012e).

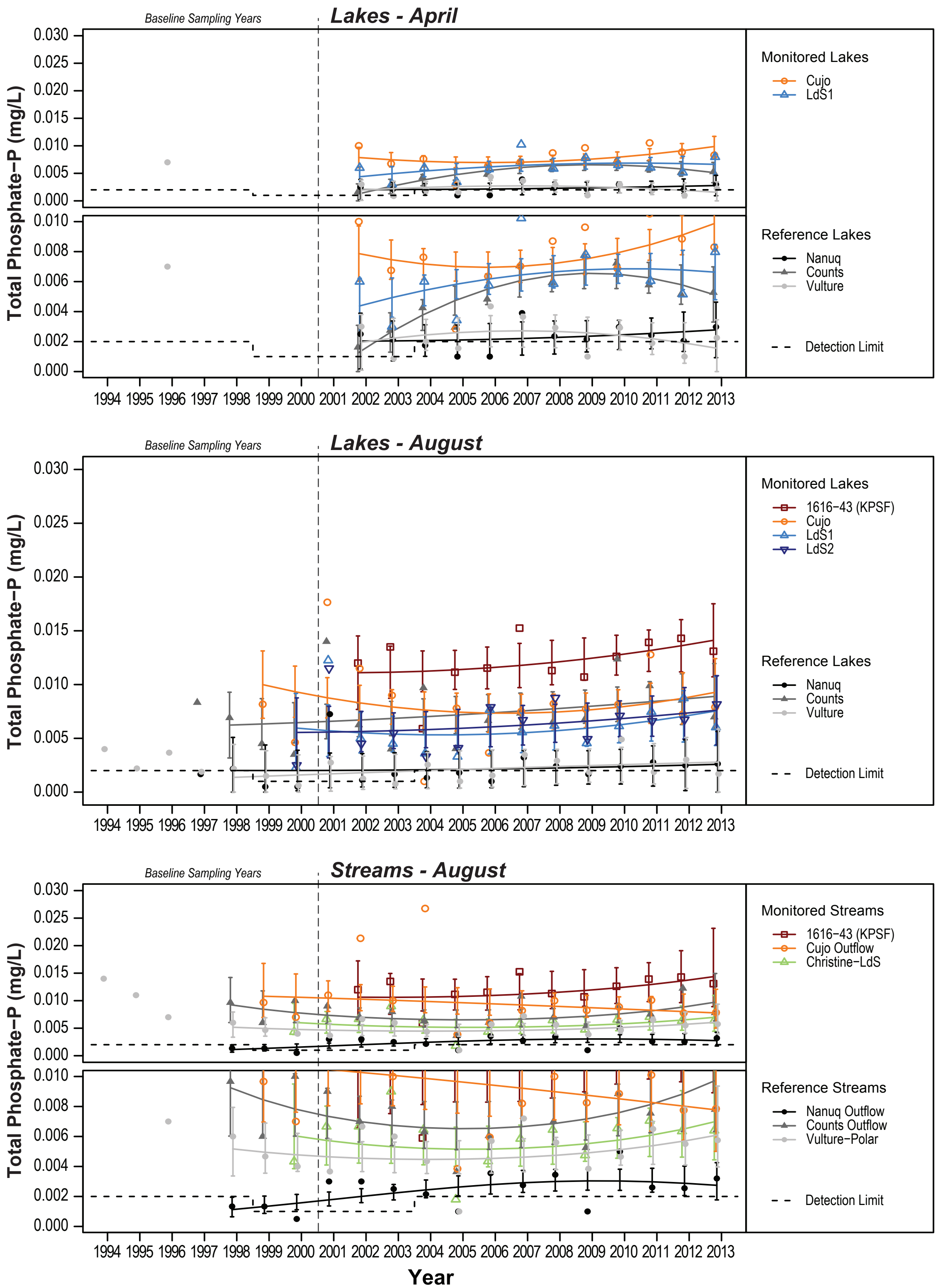
4.2.4.10 Total Phosphate-P

> ***Summary:*** Statistical and graphical analyses suggest that total phosphate-P concentrations have been stable in all monitored lakes and streams in the King-Cujo Watershed and Lac du Sauvage since monitoring began. Although the upper 95% confidence interval around the 2013 fitted mean total phosphate-P concentrations was greater than the 0.01 mg/L or mean baseline concentrations + 50% phosphate-P benchmark in Cujo Lake and at sites LdS1 and LdS2 in Lac du Sauvage, similar patterns were observed in reference lakes. No mine effects were detected.

Statistical analyses indicate that total phosphate-P concentrations have changed through time, relative to reference lakes, in Cujo Lake during the ice-covered season (Table 4.2-15). In contrast, statistical analyses indicate that total phosphate-P concentrations have been stable through time, relative to reference lakes, during the open water season at all sites in the King-Cujo Watershed and Lac du Sauvage (Table 4.2-15). Graphical analysis suggests that total phosphate-P concentrations have been stable through time in all lakes and streams (Figure 4.2-10). Together, statistical and graphical analysis indicates that mining activities have had no effect on total phosphate-P concentrations in the King-Cujo Watershed or Lac du Sauvage.

Figure 4.2-10

Observed and Fitted Means for Total Phosphate-P Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: *Symbols represent observed mean values.*
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.

Table 4.2-15. Statistical Results of Total Phosphate-P Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	Tobit	1b	-	-	Cujo	2-134
Aug	Lake	-	Tobit	2	-	None	-	2-139
Aug	Stream	-	Tobit	2	-	None	-	2-144

Dashes indicate not applicable.

The upper 95% confidence intervals around the fitted mean total-phosphate P concentrations were greater than the 0.01 mg/L trigger set out for oligotrophic lakes in the Canadian Guidance Framework for the management of Phosphorus in Freshwater Systems in Cujo Lake during both the ice-covered and open water season (CCME 2004; Environment Canada 2004). Total phosphate-P concentrations were also greater than the recommended benchmark trigger of mean baseline concentration + 50% (Ontario Ministry of Natural Resources 1994; CCME 2004; Environment Canada 2004) at sites LdS1 and LdS2 in Lac du Sauvage during the ice-covered and open water seasons (see Part 2 – Data Report). The fitted and observed mean concentrations also exceeded the 50% triggers at site LdS1 during the ice-covered and at sites LdS1 and LdS2 during the open water season. However, similar patterns in the 95% confidence intervals were observed in all three reference lakes during the open water season and fitted and observed mean concentrations were greater than the 50% trigger in Nanuq Lake during both the ice-covered and open water seasons.

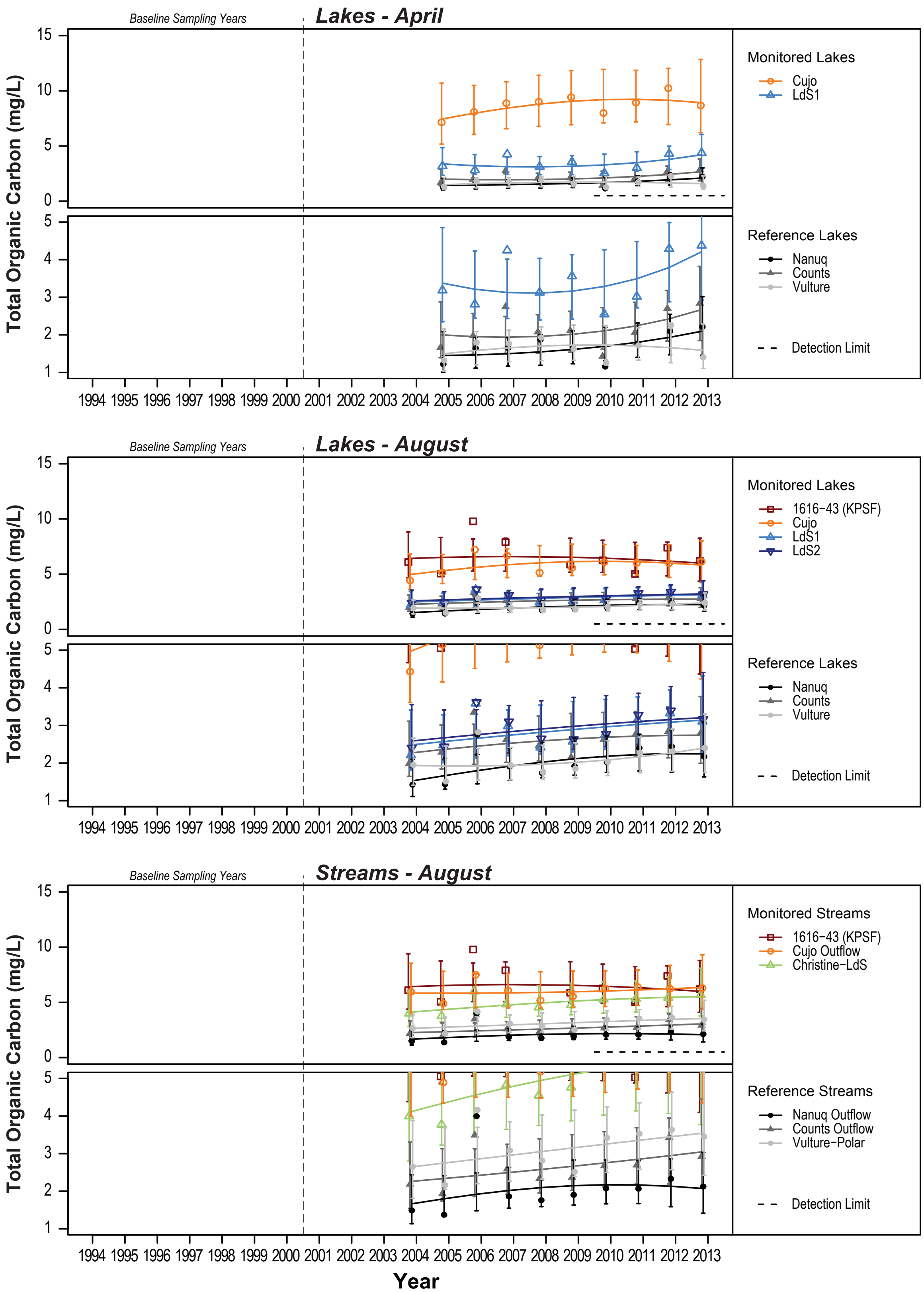
4.2.4.11 TOC

Summary*:* Graphical analysis suggests that TOC concentrations have been elevated, relative to reference sites, in all lakes and streams downstream of the KPSF as far as Christine-Lac du Sauvage Stream, with concentrations decreasing with downstream distance from the KPSF. Moreover, TOC concentrations in Cujo Lake and Cujo Outflow have been similar to those observed in the KPSF since TOC monitoring began. Thus, elevated TOC concentrations in Cujo Lake, Cujo Outflow, and Christine-Lac du Sauvage Stream are likely related to mine operations.

Statistical analyses indicate that TOC concentrations have changed through time at sites LdS1 and LdS2 during the open water seasons (Table 4.2-16). However, model fit for TOC concentrations at sites LdS1 and LdS2 during the open water season was weak and graphical analysis suggests that TOC concentrations have been relatively stable through time at all monitored sites since monitoring began in 2005 (see Part 3 – Statistical Report; Figure 4.2-11). However, graphical analysis suggests that TOC concentrations have been elevated, relative to reference site concentrations, in Cujo Lake, Cujo Outflow, and Christine-Lac du Sauvage Stream, with concentrations decreasing with downstream distance from the KPSF (Figure 4.2-11). TOC concentrations were not measured during baseline years, making it difficult to discern whether the observed patterns result from mine effects or represent natural concentrations in the King-Cujo Watershed. However, TOC concentrations in Cujo Lake and Cujo Outflow were similar to concentrations in the KPSF in all years during which TOC has been measured in the King-Cujo Watershed (Figure 4.2-11). Thus, graphical analyses suggest that elevated TOC concentrations in Cujo Lake and Cujo Outflow likely result from mine operations.

Figure 4.2-11

Observed and Fitted Means for Total Organic Carbon Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.

Table 4.2-16. Statistical Results of TOC in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	2	-	None	-	2-149
Aug	Lake	-	LME	1b	-	-	LdS1, LdS2	2-154
Aug	Stream	-	LME	2	-	None	-	2-159

Dashes indicate not applicable.

4.2.4.12 Total Antimony

Summary: Total antimony concentrations have generally been below detection limits in all monitored lakes and streams since monitoring began. Although the upper 95% confidence interval around the 2013 fitted mean total antimony concentration was greater than the water quality benchmark of 0.02 mg/L in Christine-Lac du Sauvage Stream, a similar result was observed in one of the reference streams. All other concentrations were below the benchmark in 2013. No mine effects were detected.

Statistical and graphical analyses indicate that total antimony concentrations have generally been below detection limits in all monitored lakes and streams in the King-Cujo Watershed and Lac du Sauvage (Table 4.2-17; Figure 4.2-12). No mine effects were detected.

Table 4.2-17. Statistical Results of Total Antimony Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	ALL	-	-	-	-	-	2-164
Aug	Lake	Cujo, LdS1, LdS2, Counts, Nanuq, Vulture	Tobit	1a	-	-	1616-43 (KPSF)	2-166
Aug	Stream	Cujo Outflow, Christine-Lac du Sauvage	Tobit	3	1616-43 (KPSF)	None	-	2-169

Dashes indicate not applicable.

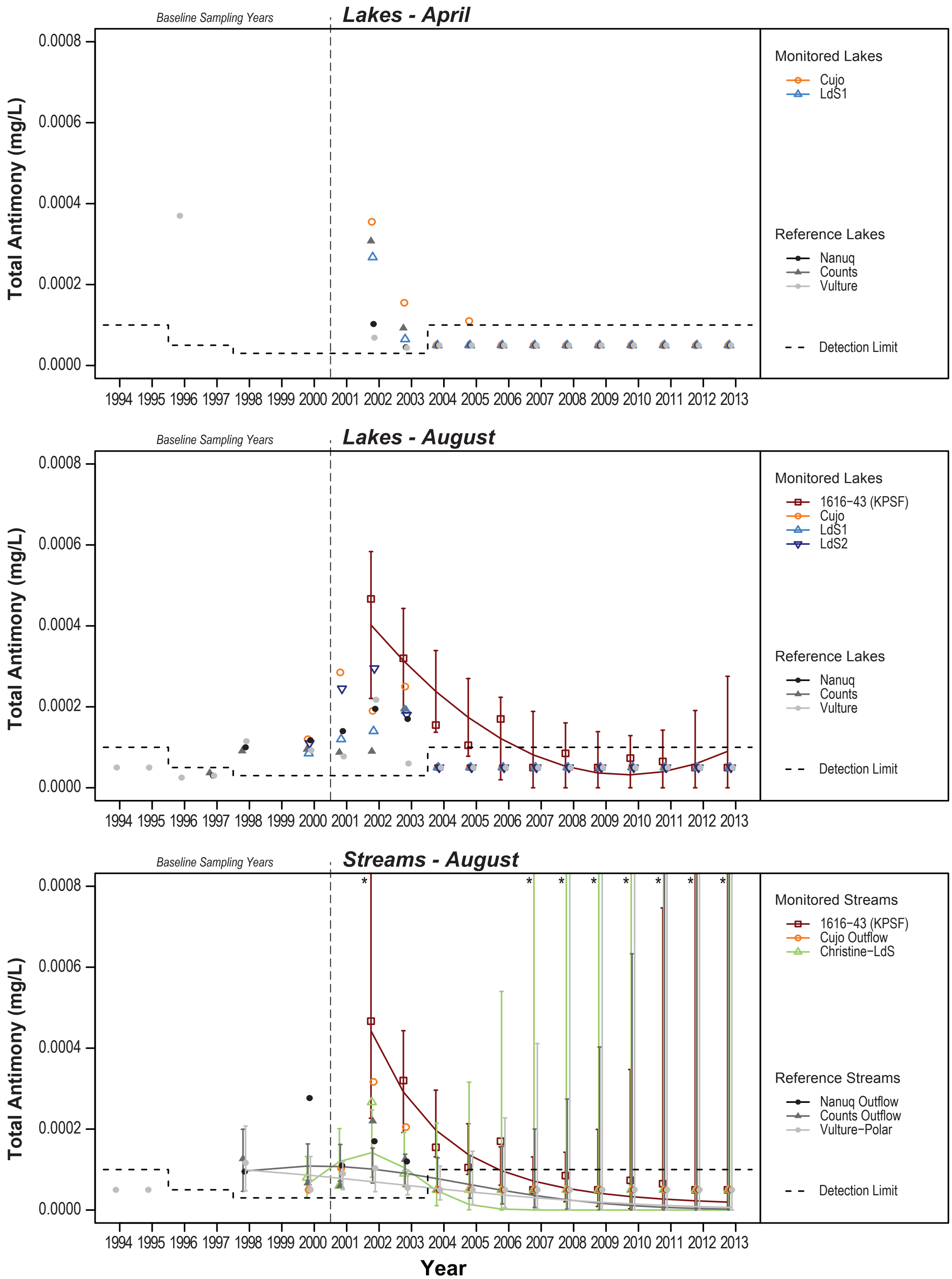
The 95% upper confidence interval around the fitted mean total antimony concentration in Christine-Lac du Sauvage Stream was greater than the water quality benchmark of 0.02 mg/L (Fletcher et al. 1996); however, a similar result was obtained in Vulture-Polar Stream, one of the reference streams. All observed and fitted means were below the water quality benchmark in all reference and monitored lakes and streams in 2013. Total antimony concentrations were also less than the benchmark concentration in monitored streams in June, July, August, and September 2013 (see Part 2 - Data Report; Fletcher et al. 1996).

4.2.4.13 Total Arsenic

Summary: Statistical and graphical analyses suggest that mining operations have had no effect on total arsenic concentrations in lakes and streams in the King-Cujo Watershed and Lac du Sauvage. Observed and fitted mean concentrations were less than the arsenic CCME guideline at all sites in 2013. Thus, no mine effects were detected.

Figure 4.2-12

Observed and Fitted Means for Total Antimony Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
** Upper 95% Confidence Interval on the fitted mean of 1616-43 (KPSF) in 2002 = 0.0009 mg/L; Christine-LdS in 2007 = 0.0011 mg/L, 2008 = 0.0023 mg/L, 2009 = 0.0057 mg/L, 2010 = 0.0159 mg/L, 2011 = 0.0501 mg/L, 2012 = 0.1772 mg/L, and 2013 = 0.7057 mg/L; Counts Outflow in 2011 = 0.0011 mg/L, 2012 = 0.0019 mg/L, and 2013 = 0.0037 mg/L; Vulture-Polar in 2008 = 0.0009 mg/L, 2009 = 0.0020 mg/L, 2010 = 0.0054 mg/L, 2011 = 0.0166 mg/L, 2012 = 0.0580 mg/L, and 2013 = 0.2309 mg/L.*
Water quality benchmark (Fletcher et al. 1996) = 0.02 mg/L.

Statistical analyses indicate that total arsenic concentrations have changed through time in Cujo Lake during the ice-covered season (Table 4.2-18). However graphical analysis suggests that total arsenic concentrations have been elevated, relative to reference sites, but stable in Cujo Lake during the ice-covered season and in Cujo Lake, Cujo Outflow, and at site LdS1 during the open water season (Figure 4.2-13). In contrast, concentrations at sites LdS1 and LdS2 and in Christine-Lac du Sauvage Stream have been stable and similar to those observed in reference sites during the open water season (Figure 4.2-13). Thus it was concluded that no mine effects related to arsenic were detected.

Table 4.2-18. Statistical Results of Total Arsenic Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	1b	-	-	Cujo	2-174
Aug	Lake	-	LME	2	-	None	-	2-179
Aug	Stream	-	Tobit	3	1616-43 (KPSF), Cujo Outflow	None	-	2-185

Dashes indicate not applicable.

The 95% confidence intervals around the fitted mean and observed mean total arsenic concentrations in all monitored and reference lakes and streams during the ice-covered and open water seasons in 2013 were less than the arsenic CCME guideline value (0.005 mg/L) (see Part 2 - Data Report; CCME 1999).

4.2.4.14 Total Barium

Summary: Statistical and graphical analyses suggest that total barium concentrations have increased at all monitored sites downstream of the KPSF as far as Christine-Lac du Sauvage Stream as a result of mine operations. Observed and fitted mean concentrations were less than the barium water quality benchmark (1 mg/L) at all sites in 2013.

Statistical analyses indicate that total barium concentrations have changed through time, relative to reference sites, in Cujo Lake and Cujo Outflow during the open water season (Table 4.2-19). Graphical analysis suggests that total barium concentrations have increased through time in Cujo Lake and Cujo Outflow (Figure 4.2-14). Thus, increases are likely related to mine operations.

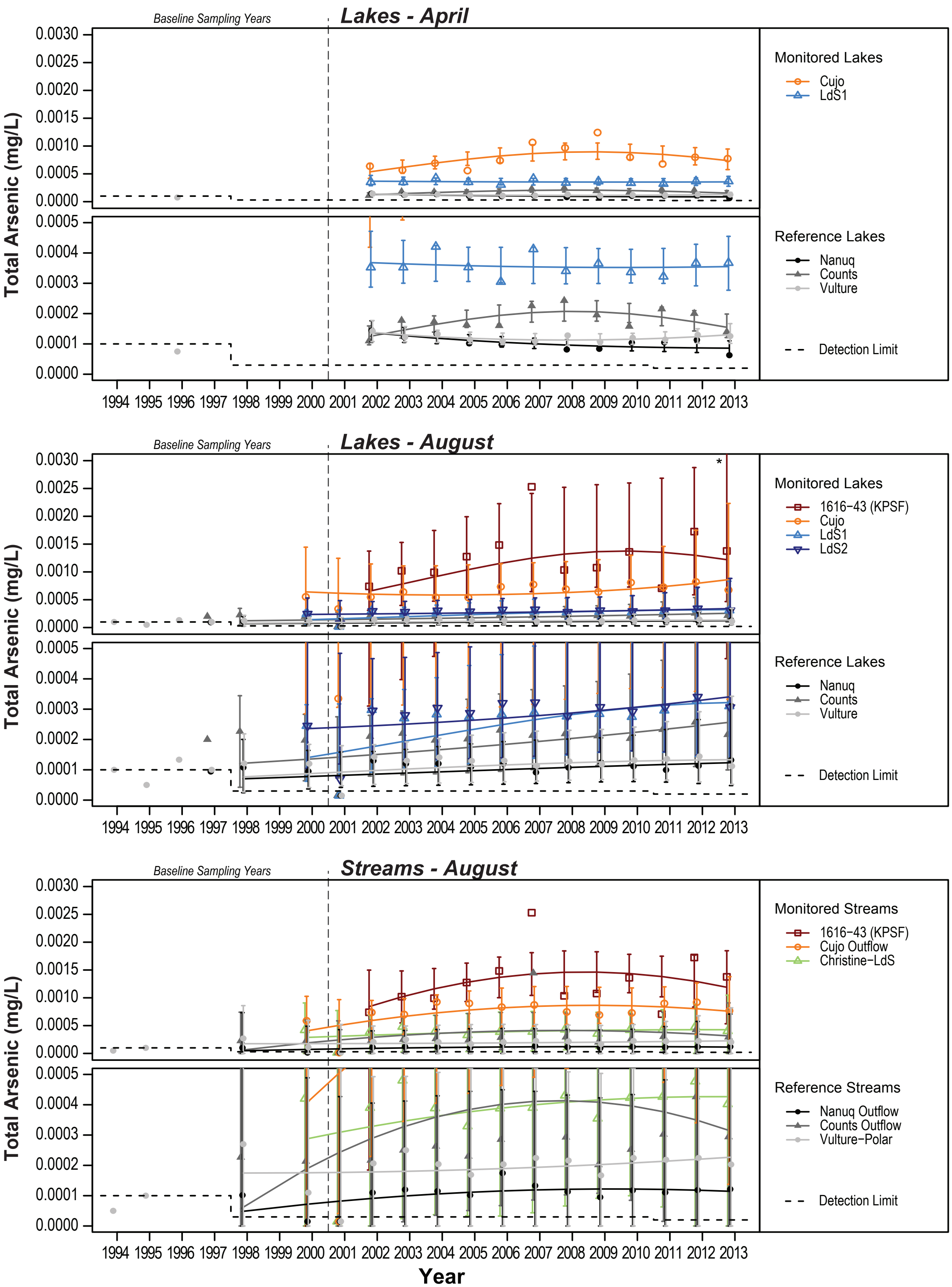
Table 4.2-19. Statistical Results of Total Barium Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	2	-	None	-	2-190
Aug	Lake	-	LME	2	-	Cujo	-	2-196
Aug	Stream	-	LME	2	-	Cujo Outflow	-	2-201

Dashes indicate not applicable.

Figure 4.2-13

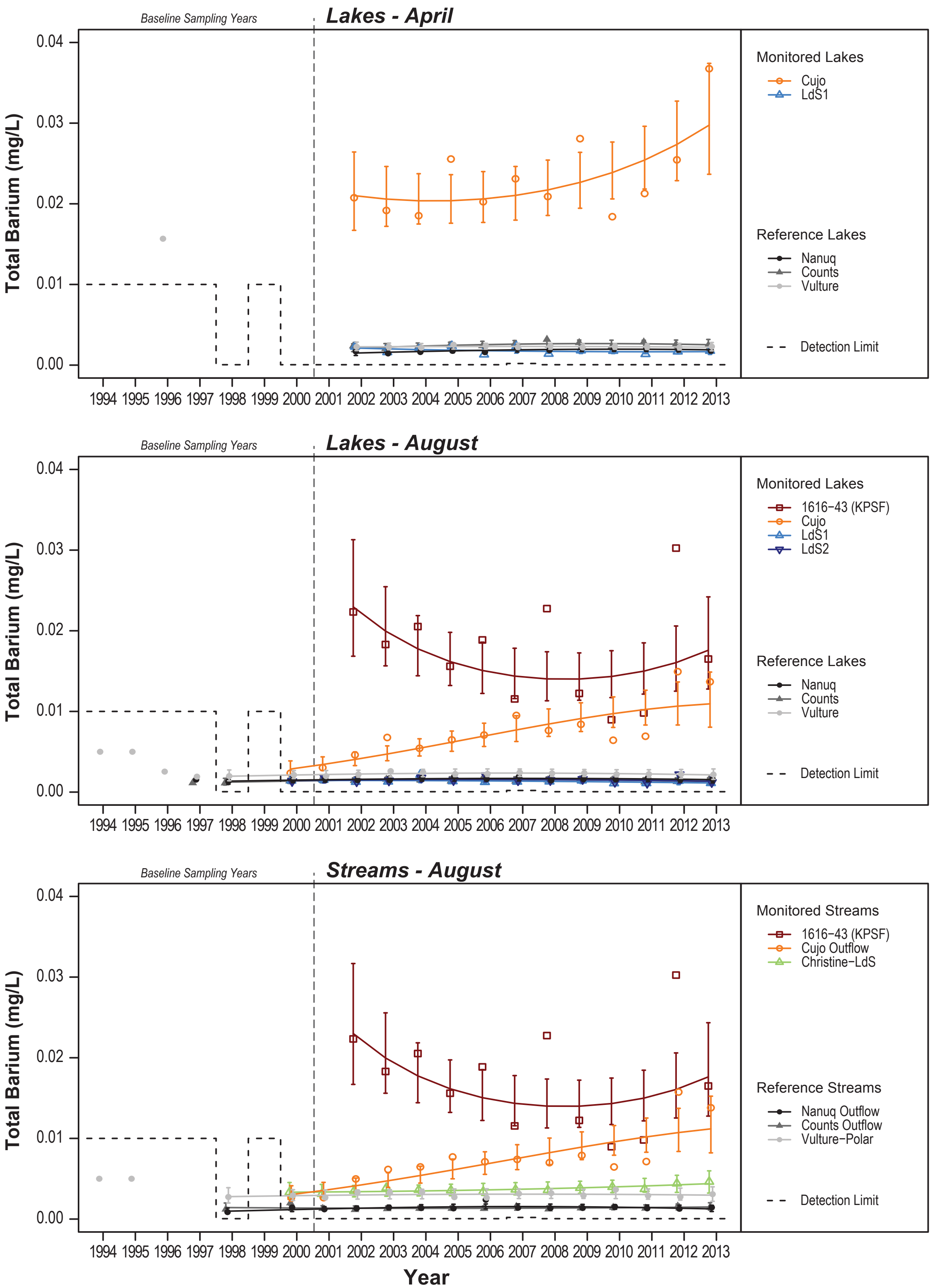
Observed and Fitted Means for Total Arsenic Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
WL = Maximum average concentration premitted in water licence W2009L2-0001. WL = 0.50 mg/L.
CCME Guideline = 0.005 mg/L
** Upper 95% Confidence Interval on the fitted mean of 1616-43 (KPSF) in August 2013 = 0.0032 mg/L.*

Figure 4.2-14

Observed and Fitted Means for Total Barium Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
Water quality benchmark (Haywood and Drinnan 1983) = 1 mg/L.

The 95% confidence intervals of the fitted and observed mean total barium concentrations at all monitored sites in 2013 were below the barium water quality benchmark (1 mg/L; Haywood and Drinnan 1983). Total barium concentrations in all monitored streams in June, July, August and September 2013 were also less than the barium water quality benchmark (see Part 2 - Data Report; Haywood and Drinnan 1983).

4.2.4.15 Total Boron

Summary: Statistical analyses suggest that total boron concentrations have been stable through time. Although concentrations do not differ from those found in reference lakes, graphical analysis suggests that total boron concentrations may have increased through time at all monitored sites downstream from the KPSF as far as Cujo Outflow. All concentrations were less than the CCME guideline in 2013.

Statistical analyses indicate that total boron concentrations have been stable through time, relative to reference sites, in all lakes and streams downstream of the KPSF since monitoring began (Table 4.2-20). Although, graphical analysis suggests that total boron concentrations have increased through time at all monitored sites downstream from the KPSF as far as Cujo Outflow during both the ice-covered and open water seasons, with total boron concentrations decreasing with downstream distance from the KPSF, concentrations do not differ from those found in reference lakes at this time and temporal trends in reference and monitored sites are similar (Figure 4.2-15). Observed mean concentrations in reference lakes are likely artificially elevated in recent years owing to increased detection limits in 2012 and 2013; more than 85% of all observations in reference lakes were below detection limits in 2012 and 2013.

Table 4.2-20. Statistical Results of Total Boron Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	Nanuq, Vulture	Tobit	1a	-	-	None	2-206
Aug	Lake	Nanuq	Tobit	2	-	None	-	2-210
Aug	Stream	Nanuq Outflow	Tobit	2	-	None	-	2-215

Dashes indicate not applicable.

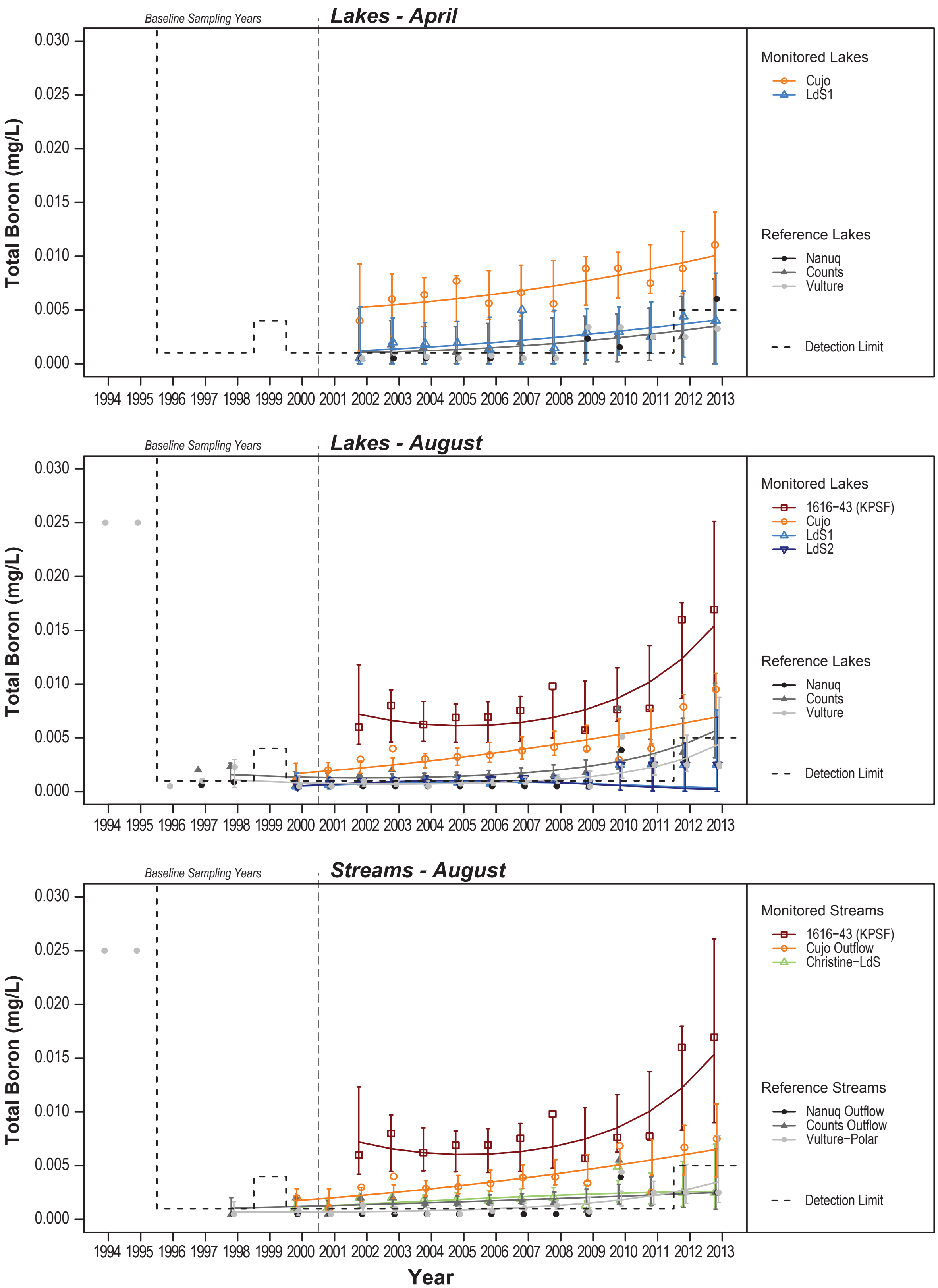
The 95% confidence intervals of the fitted and observed mean total boron concentrations in all monitored lakes and streams in 2013 were less than the boron CCME guideline (1.5 mg/L; CCME 2009).Total boron concentrations in all monitored streams in June, July, August and September 2013 were also less than the boron CCME guideline value (see Part 2 - Data Report; CCME 2009).

4.2.4.16 Total Cadmium

Summary: Concentrations of total cadmium have generally been below detection limits in all reference and monitored lakes and streams since monitoring began. All observations that were greater than the detection limit in 2013 were less than hardness-dependent CCME guidelines. No mine effects were detected.

Figure 4.2-15

Observed and Fitted Means for Total Boron Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
CCME Guideline = 1.5 mg/L.

Concentrations of total cadmium have generally been less than the detection limit in all monitored and reference lakes and streams during both the ice-covered and open water season since monitoring began (Figure 4.2-16). Consequently, all lakes and streams were removed from the statistical analyses (Table 4.2-21). Graphical analysis suggests that total cadmium concentrations have been low and stable through time in all lakes and streams in the King-Cujo Watershed and Lac du Sauvage (Figure 4.2-16).The detection limit for total cadmium was less than the hardness-dependent cadmium CCME guideline in Cujo Lake, Lac du Sauvage and Cujo Outflow during the ice-covered and open water seasons in 2013. The single observed concentration from Cujo Lake that was greater than the detection limit in 2013 was less than the hardness-dependent CCME guideline (CCME 2014). Thus it was concluded that there were no mine effects detected.

Table 4.2-21. Statistical Results of Total Cadmium Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts: Model Fit = 3	Model Fit = 2	Model Fit = 1	Statistical Report Page No.
Apr	Lake	ALL	-	-	-	-	-	2-220
Aug	Lake	ALL	-	-	-	-	-	2-222
Aug	Stream	ALL	-	-	-	-	-	2-224

Dashes indicate not applicable.

4.2.4.17 Total Copper

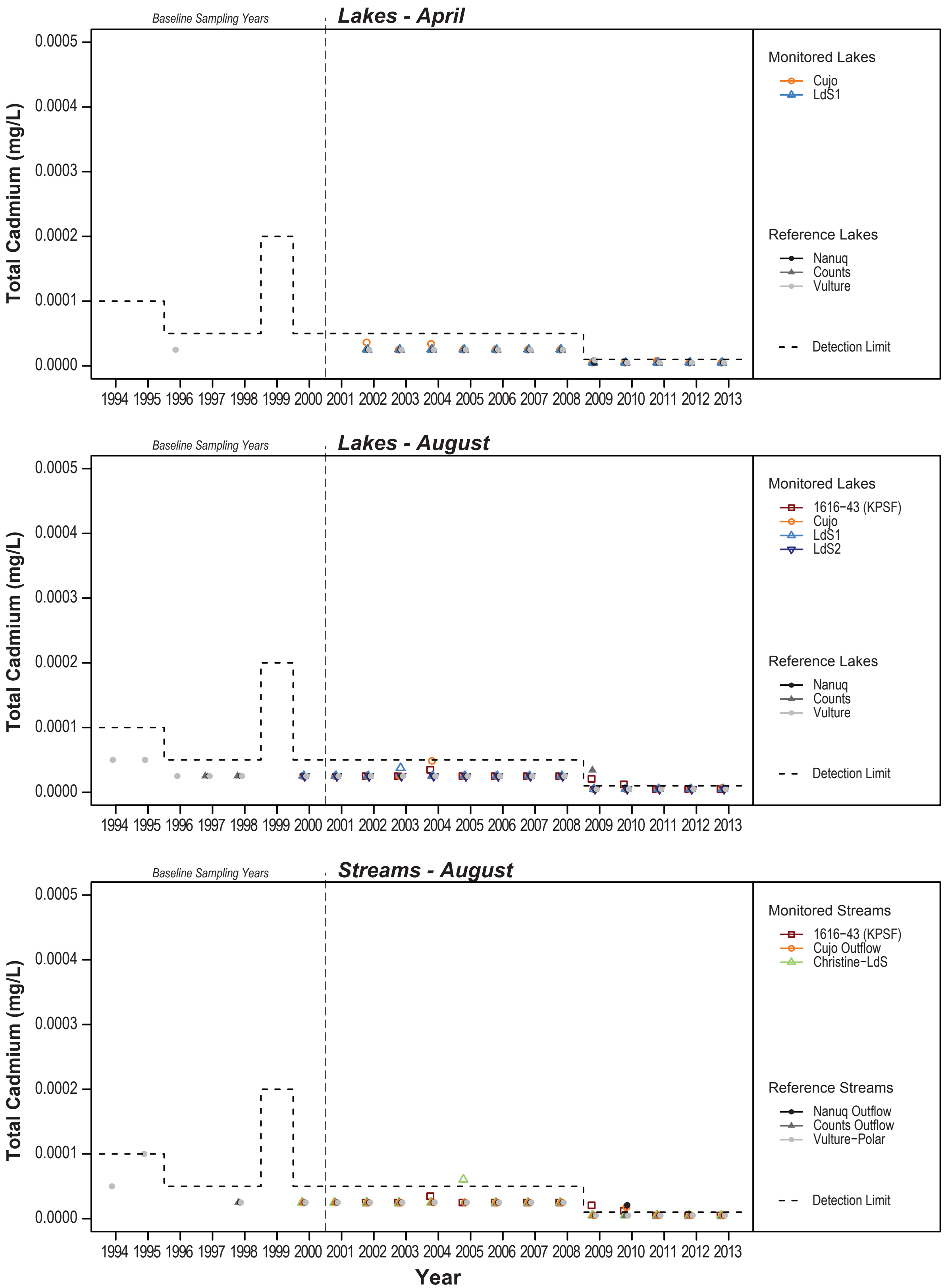
> ***Summary:*** Total copper concentrations have declined in recent years in Cujo Lake and Cujo Outflow and have remained stable at all other monitored sites in the King-Cujo Watershed and Lac du Sauvage. The observed mean total copper concentration in Cujo Lake in 2013 was greater than the hardness-dependent copper guideline value during the open water season. In all other monitored lakes and streams, the 95% confidence intervals around the 2013 fitted mean total copper concentrations during both the ice-covered and open water seasons were less than the hardness-dependent CCREM guideline value. Together, statistical and graphical analyses suggest that total copper concentrations have been affected by mine operations historically, but show no effects of mine operations in 2013.

Statistical and graphical analyses indicate that total copper concentrations have declined through time, relative to reference sites, in Cujo Lake during the ice-covered season (Table 4.2-22; Figure 4.2-17). Previous AEMP reports have suggested that total copper concentrations had increased downstream of the KPSF as far as Cujo Outflow as a result of mine operations (Rescan 2011a). However results of the evaluation of effects in 2011, 2012 and 2013 suggest that total copper concentrations have attenuated in recent years at monitored sites downstream of the KPSF and are now similar to concentrations observed during baseline years (Figure 4.2-17). Thus, no mine effects were detected in 2013.

The 95% confidence intervals around the fitted and observed mean total copper concentrations in all reference and monitored lakes during both the ice-covered and open water seasons in 2013 were less than hardness-dependent copper CCREM guideline value, with the exception of Cujo Lake where the observed mean in August was greater than the copper CCREM guideline (CCREM 1987). Observed total copper concentrations were less than the hardness-dependent copper CCREM guideline value in all monitored and reference streams in 2013 (see Part 2 - Data Report; CCREM 1987).

Figure 4.2-16

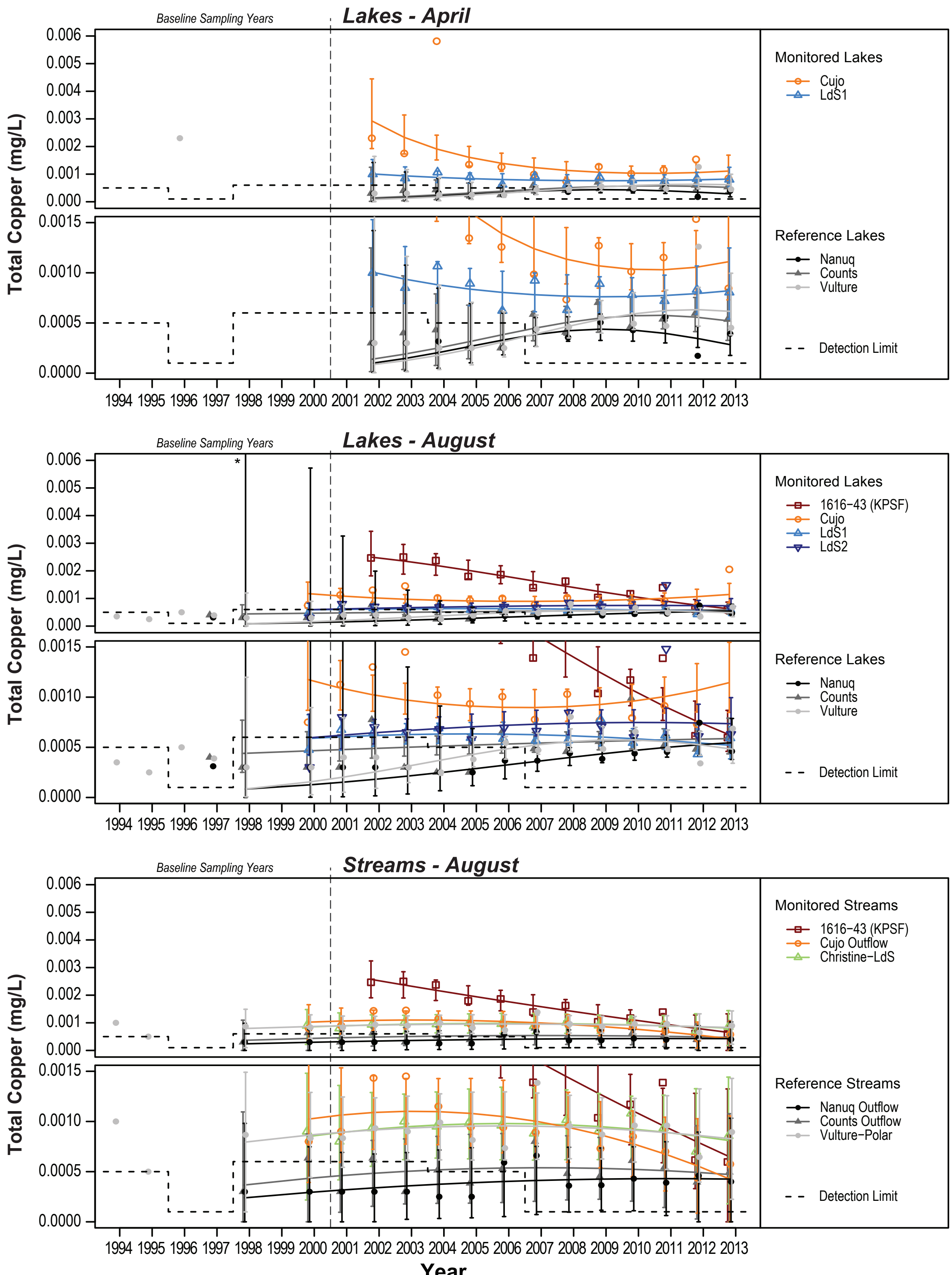
Observed Means for Total Cadmium Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
CCME Guideline = $10^{0.83 X (log10Hardness - 2.46)}$ /1000 mg/L, with minimum = 0.00004 mg/L where hardness = 0-16 mg/L and maximum of 0.00037 mg/L where hardness > 280 mg/L.

Figure 4.2-17

Observed and Fitted Means for Total Copper Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
WL = Maximum average concentration premitted in water licence W2009L2-0001. WL = 0.10 mg/L. CCME Guideline = $e^{0.8545 X (lnHardness) - 1.465 * 0.2}$/1000 mg/L, where hardness < 180 mg/L and 0.004 mg/L where hardness is ≥ to 180 mg/L . Minimum benchmark = 0.002 mg/L.
** Upper 95% Confidence Interval on the fitted mean of Nanuq in August 1998 = 0.022 mg/L.*

Table 4.2-22. Statistical Results of Total Copper Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	Tobit	3	Cujo, LdS1	Cujo	-	2-226
Aug	Lake	-	Tobit	2	-	1616-43 (KPSF)	-	2-231
Aug	Stream	-	Tobit	2	-	1616-43 (KPSF)	-	2-237

Dashes indicate not applicable.

4.2.4.18 Total Molybdenum

Summary: Together, statistical and graphical analyses suggest that total molybdenum concentrations have increased in all lakes and streams downstream of the KPSF as far as Christine-Lac du Sauvage Stream as a result of mine operations. Observed and fitted mean concentrations were less than the molybdenum SSWQO at all sites in 2013.

Statistical analyses indicate that total molybdenum concentrations have changed through time in Cujo Outflow and Christine-Lac du Sauvage Stream (Table 4.2-23). Graphical analysis suggests that total molybdenum concentrations have been elevated, but stable, in Cujo Lake during both the ice-covered and open water seasons and that concentrations in Cujo Outflow have been relatively stable since reaching a peak around 2007 (Figure 4.2-18). Although total molybdenum concentrations in Christine-Lac du Sauvage Stream have generally been less than detection limits since monitoring began, observed concentrations have been greater than detection limits since 2011, suggesting that concentrations may be increasing (Figure 4.2-18). Together, graphical and statistical analyses indicate that total molybdenum concentrations have increased in all lakes and streams downstream of the KPSF as far as Christine-Lac du Sauvage Stream as a result of mine operations (Table 4.2-23; Figure 4.2-18). The 95% confidence intervals around the fitted mean and observed mean total molybdenum concentrations in all lakes and streams during the ice-covered and open water seasons in 2013 were less than the molybdenum SSWQO (19.38 mg/L) (Rescan 2012a).

Table 4.2-23. Statistical Results of Total Molybdenum Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	LdS1, Counts, Nanuq, Vulture	LME	1a	-	-	None	2-242
Aug	Lake	LdS1, LdS2, Counts, Nanuq, Vulture	Tobit	1a	-	-	None	2-246
Aug	Stream	Counts Outflow, Nanuq Outflow, Vulture Outflow	Tobit	1a	-	-	Cujo Outflow, Christine-Lac du Sauvage	2-250

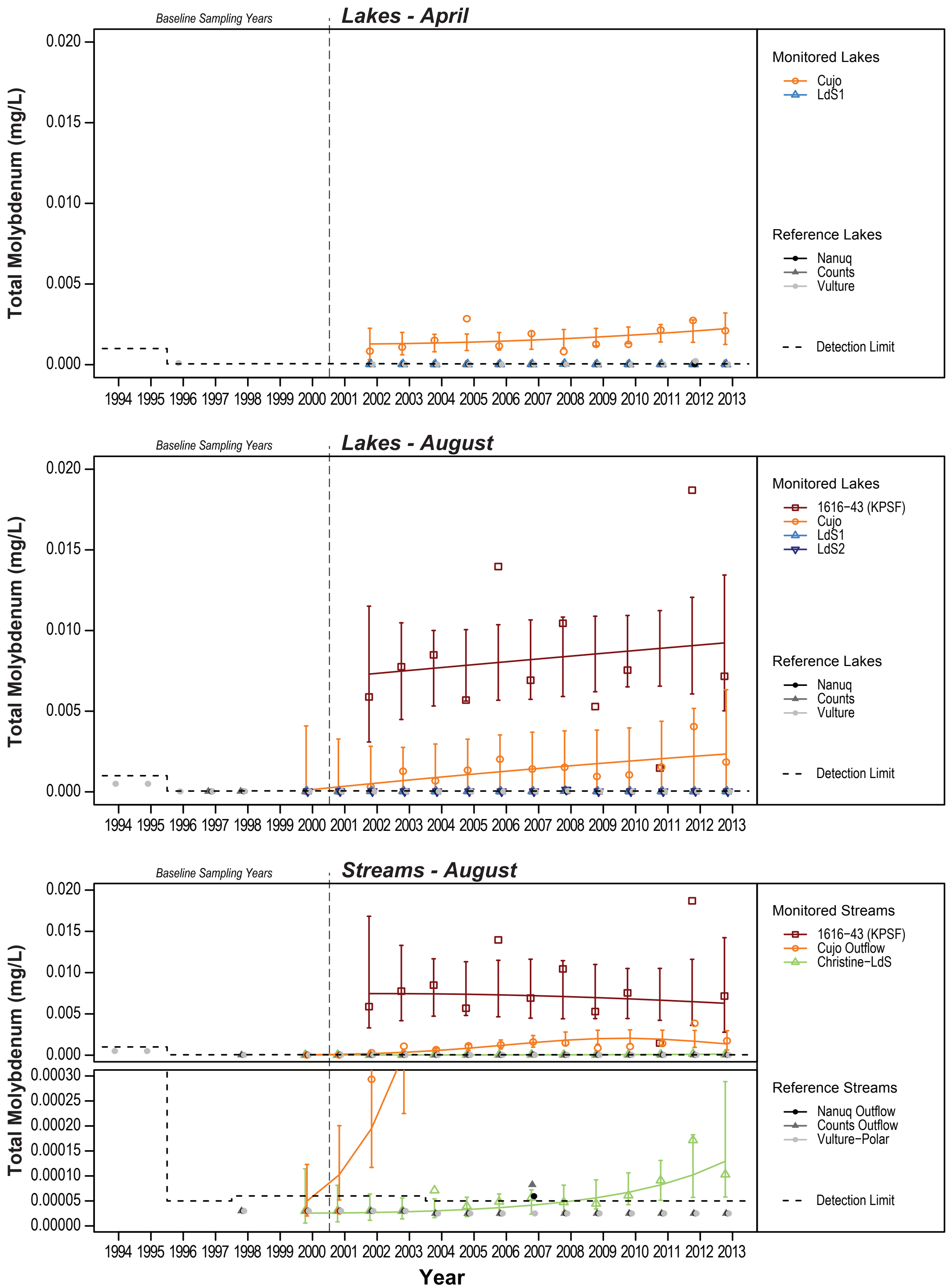
Dashes indicate not applicable.

4.2.4.19 Total Nickel

Summary: Statistical and graphical analyses indicate that total nickel concentrations have been stable through time. Observed and fitted mean concentrations were less than the nickel CCREM guideline value at all sites in 2013. Thus, it was concluded that no mine effects were detected.

Figure 4.2-18

Observed and Fitted Means for Total Molybdenum Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
SSWQO = 19.38 mg/L.

Statistical and graphical analyses indicate that total nickel concentrations have been stable through time in all monitored lakes and streams in the King-Cujo Watershed and Lac du Sauvage (Table 4.2-24; Figure 4.2-19). No mine effects were detected.

Table 4.2-24. Statistical Results of Total Nickel Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	2	-	None	-	2-255
Aug	Lake	-	LME	2	-	None	-	2-260
Aug	Stream	-	LME	1b	-	-	1616-43 (KPSF	2-266

Dashes indicate not applicable.

The 95% confidence intervals around the fitted mean and observed mean total nickel concentrations in 2013 were less than the hardness-dependent nickel CCREM guideline value (CCREM 1987). Total nickel concentrations in all streams in June, July, August, and September 2013 were also below the hardness-dependent nickel CCREM guideline value (see Part 2 - Data Report; CCREM 1987).

4.2.4.20 Total Selenium

> ***Summary:*** Total selenium concentrations have generally been less than the detection limits through time and were less than CCREM guidelines at all monitored and reference sites in 2013. No mine effects were detected.

Statistical and graphical analyses indicate that total selenium concentrations have generally been stable and less than the analytical detection limit through time in all monitored and reference lakes and streams (Table 4.2-25; Figure 4.2-20). Total selenium concentrations were less than the selenium CCREM guideline (0.001 mg/L) in all monitored and reference sites in April, June, July, August, and September 2013 (see Part 2 - Data Report; CCREM 1987). Thus it was concluded that no mine effects were detected.

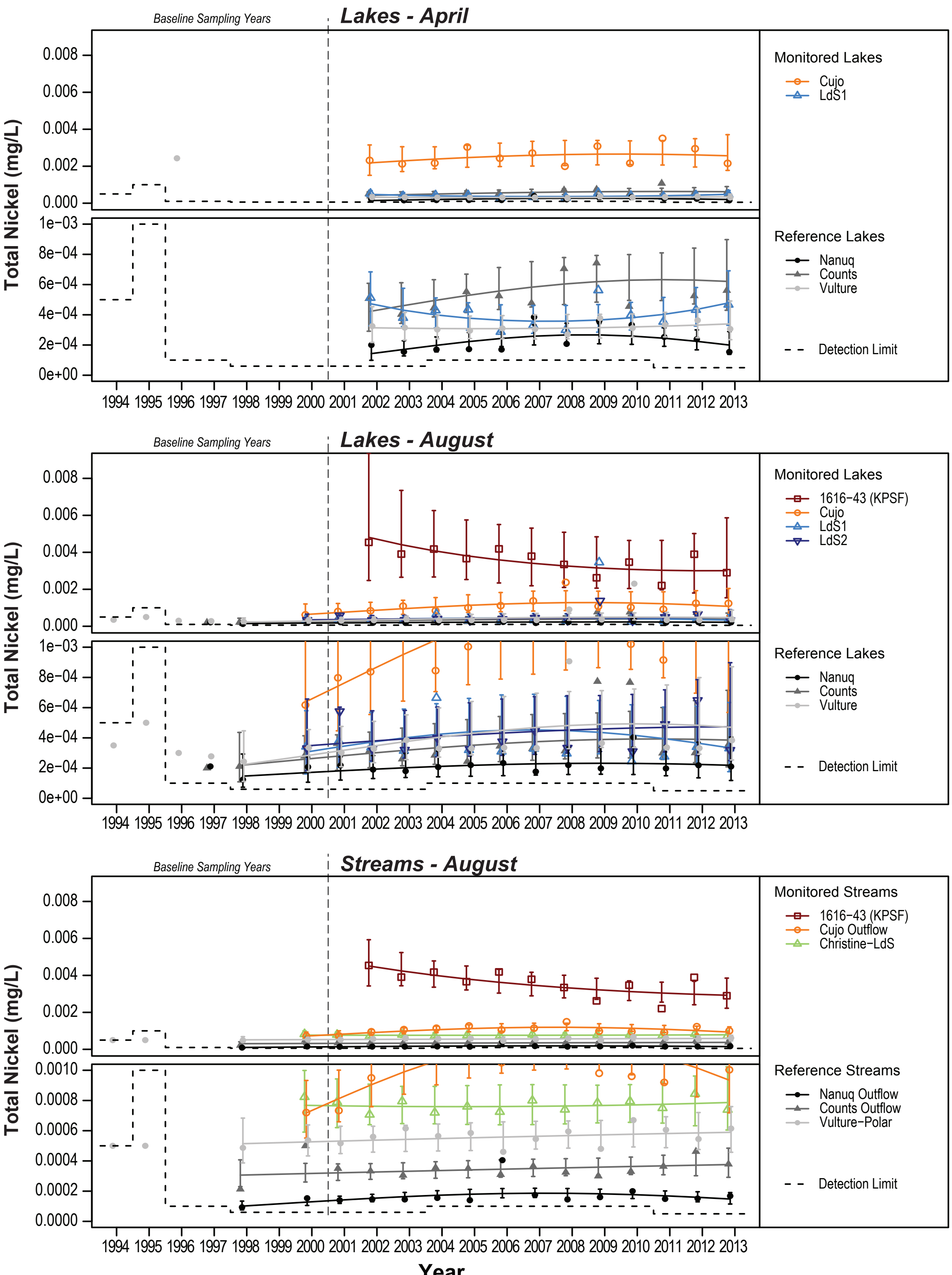
Table 4.2-25. Statistical Results of Total Selenium Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	LdS1, Counts, Nanuq, Vulture	Tobit	1a	-	-	None	2-272
Aug	Lake	LdS1, LdS2, Counts, Nanuq, Vulture	Tobit	1a	-	-	1616-43 (KPSF)	2-276
Aug	Stream	Christine-Lac du Sauvage, Counts Outflow, Nanuq Outflow, Vulture-Polar	Tobit	1b	-	-	1616-43 (KPSF)	2-280

Dashes indicate not applicable.

Figure 4.2-19

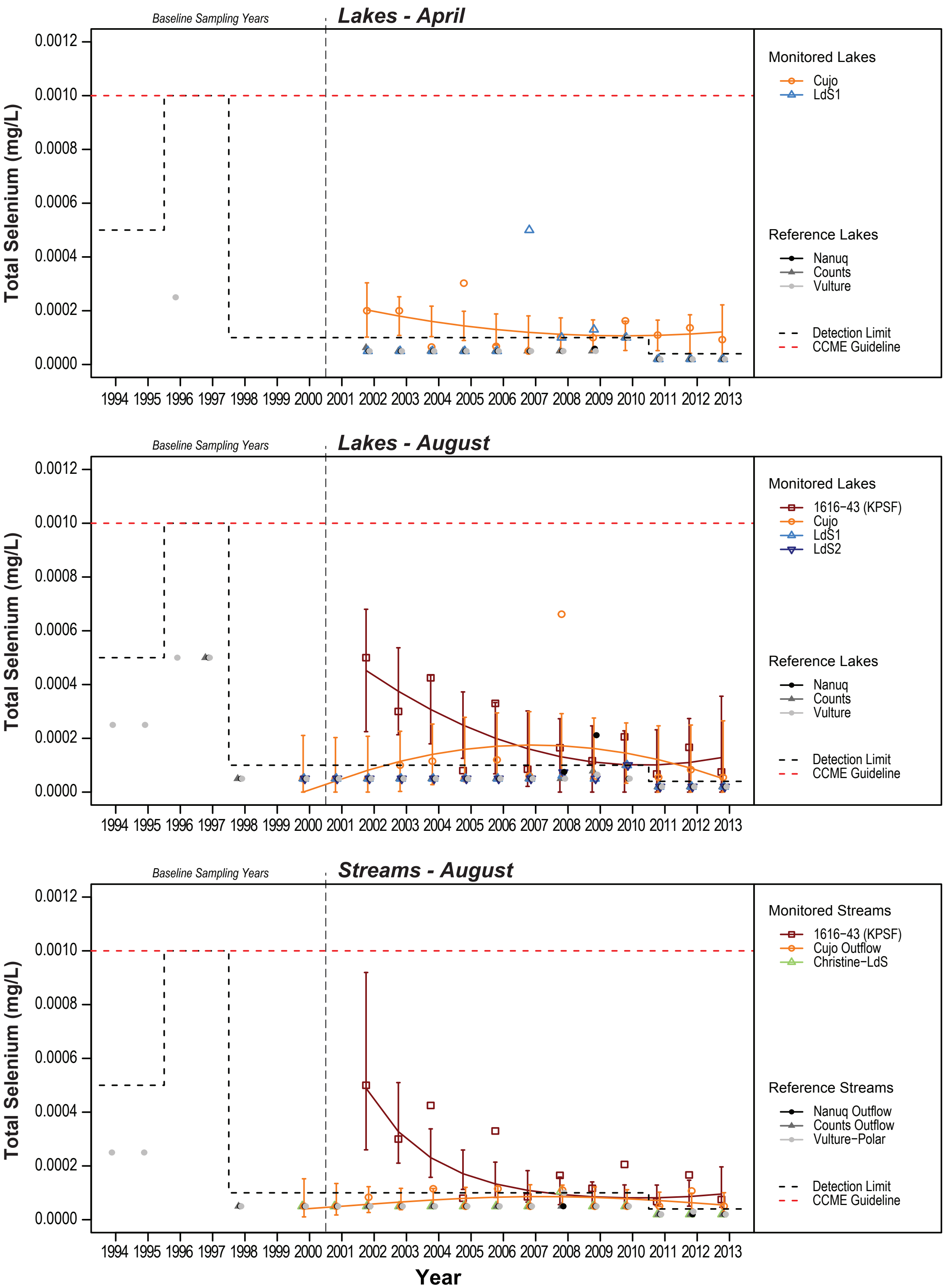
Observed and Fitted Means for Total Nickel Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
WL = Maximum average concentration premitted in water licence W2009L2-0001.
WL = 0.15 mg/L. CCME Guideline = e0.76 X (lnHardness) + 1.06/1000 mg/L, where hardness = 60 - 180 mg/L, 0.025 mg/L where hardness < 60 mg/L, and 0.15 mg/L where hardness > 180 mg/L.

Figure 4.2-20

Observed and Fitted Means for Total Selenium Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
CCME Guideline = 0.001 mg/L.

4.2.4.21 Total Strontium

> ***Summary:*** Statistical and graphical analyses suggest that total strontium concentrations have increased in lakes and streams downstream of the KPSF as far as Cujo Outflow as a result of mine operations. All observed and fitted total strontium concentrations were less than the strontium water quality benchmark value (6.242 mg/L) in 2013.

Statistical and graphical analyses indicate that total strontium concentrations have increased through time, relative to reference sites, in lakes and streams downstream of the KPSF as far as Cujo Outflow (Table 4.2-26; Figure 4.2-21). Graphical analysis also suggests that concentrations decrease with downstream distance from the KPSF, indicating that changes are likely related to mine operations (Figure 4.2-21).

Table 4.2-26. Statistical Results of Total Strontium Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	2	-	Cujo	-	2-284
Aug	Lake	-	LME	3	1616-43 (KPSF), Cujo	1616-43 (KPSF), Cujo	-	2-289
Aug	Stream	-	LME	1b	-	-	1616-43 (KPSF), Cujo Outflow	2-295

Dashes indicate not applicable.

The 95% confidence interval around the fitted mean and the observed mean total strontium concentrations in all lakes and streams during the ice-covered and open water seasons in 2013 were less the strontium water quality benchmark (6.242 mg/L) (Golder 2011).

4.2.4.22 Total Uranium

> ***Summary:*** Statistical and graphical analyses suggest that total uranium concentrations have been stable through time, relative to reference lakes and streams, at all sites downstream of the KPSF. Total uranium concentrations were less than CCME guidelines at all sites. Although mine effects were detected as far as Cujo Outflow in previous years, there was no evidence of mine effects in 2013.

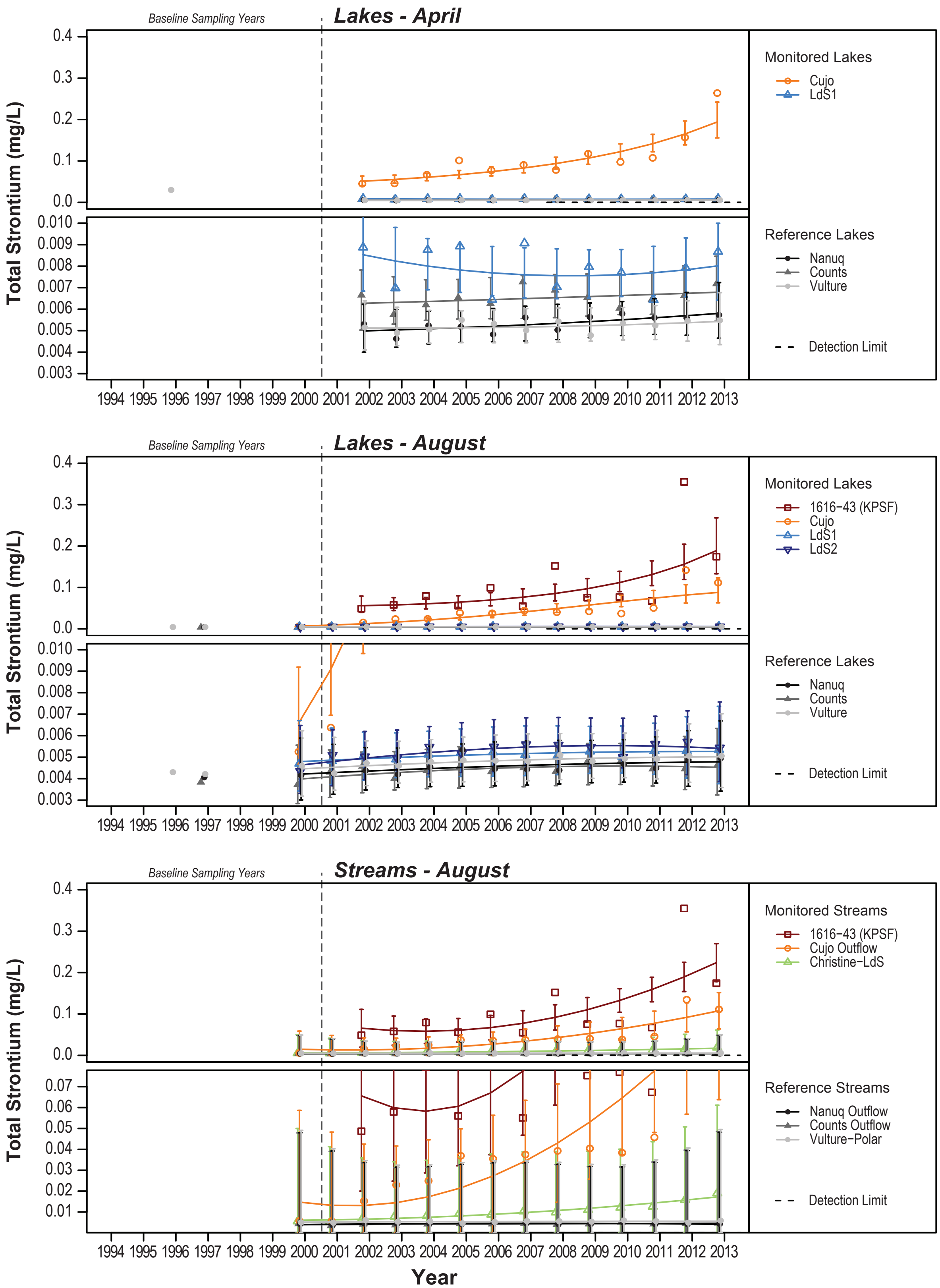
Statistical and graphical analyses suggest that total uranium concentrations have been stable through time, relative to reference lakes, in all lakes and streams in the King-Cujo Watershed and Lac du Sauvage (Table 4.2-27; Figure 4.2-22). Total uranium concentrations observed in all reference and monitored lakes and streams during both the ice-covered and open water seasons in 2013 were less than the CCME uranium guideline (0.015 mg/L; see Part 2 - Data Report; CCME 2011). Although mine effects have been detected as far as Cujo Outflow in previous years, there was no evidence of mine effects – currently or historically - in 2013.

4.2.4.23 Total Vanadium

> ***Summary:*** Statistical and graphical analyses suggest that total vanadium concentrations have been stable through time at all sites downstream of the KPSF. Total vanadium concentrations were less than SSWQO at all sites in 2013. No mine effects were detected.

Figure 4.2-21

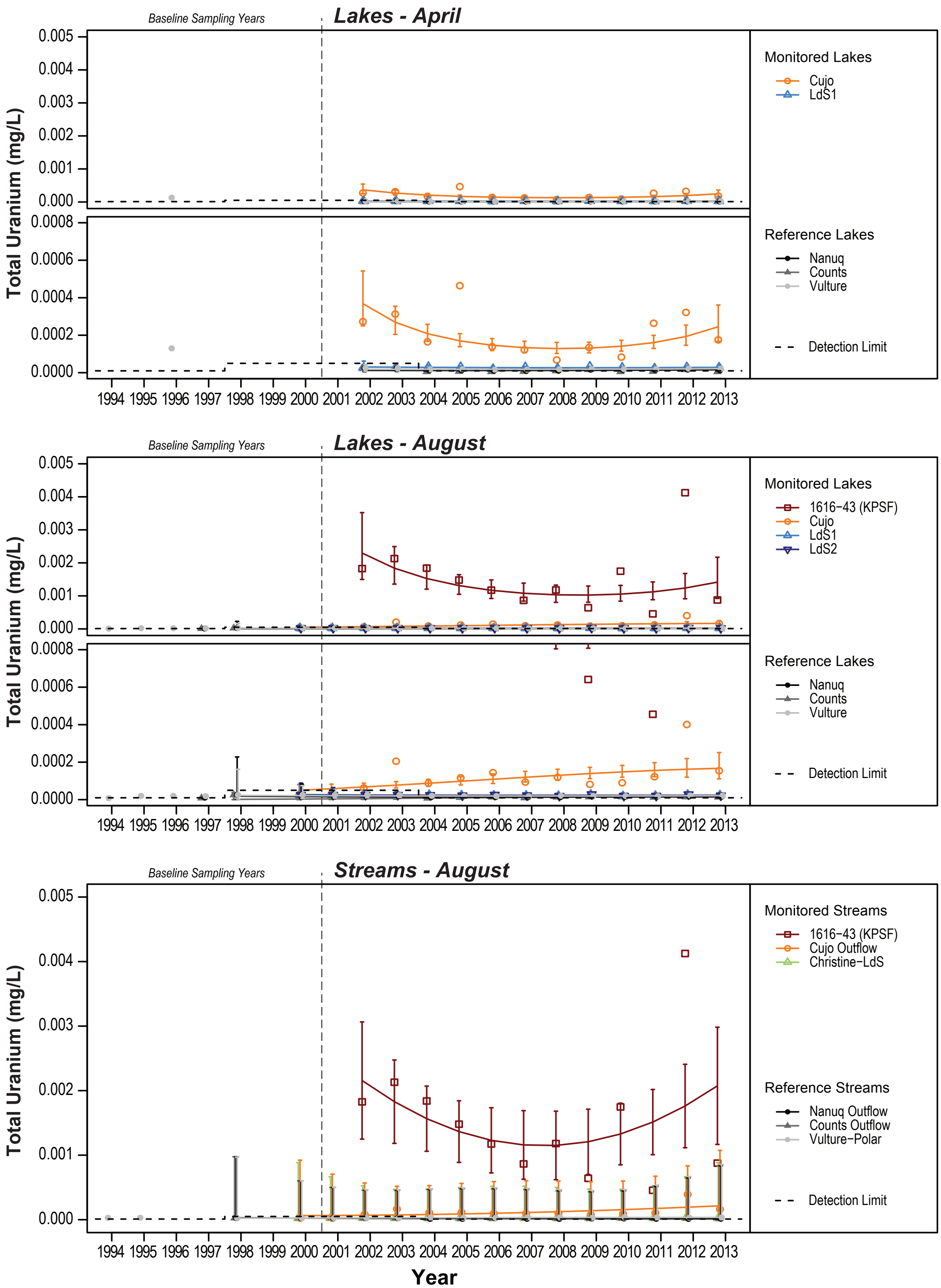
Observed and Fitted Means for Total Strontium Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
Water quality benchmark (Golder 2011) = 6.242 mg/L.

Figure 4.2-22

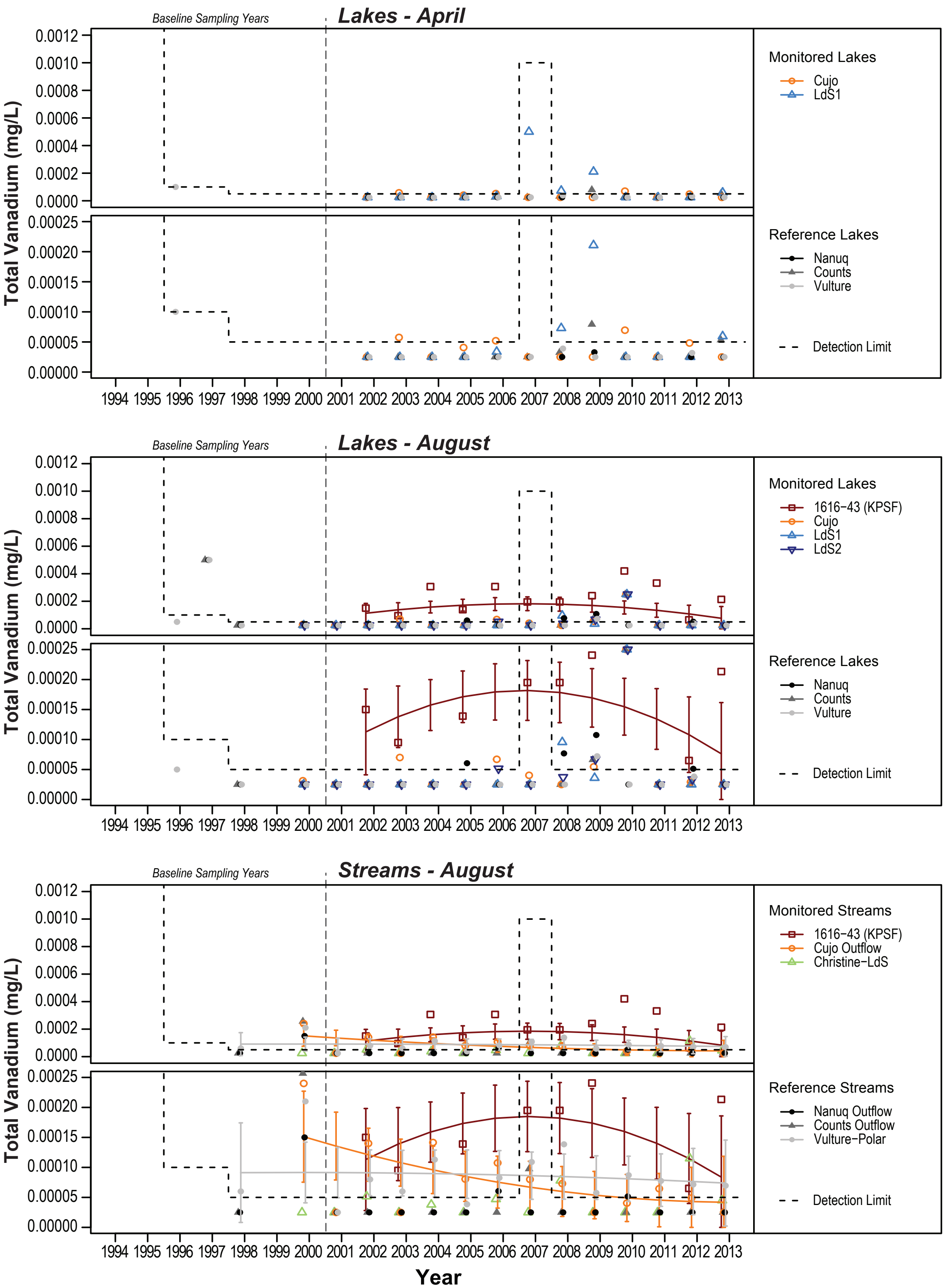
Observed and Fitted Means for Total Uranium Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
CCME Guideline = 0.015 mg/L.

Figure 4.2-23

Observed and Fitted Means for Total Vanadium Concentrations in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
SSWQO = 0.03 mg/L.

Table 4.2-27. Statistical Results of Total Uranium Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams removed from analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	Counts	Tobit	2	-	None	-	2-301
Aug	Lake	-	Tobit	2	-	None	-	2-306
Aug	Stream	-	Tobit	3	1616-43 (KPSF)	None	-	2-312

Dashes indicate not applicable.

Statistical and graphical analyses suggest that total vanadium concentrations have been stable through time in all lakes and streams in the King-Cujo Watershed and Lac du Sauvage (Table 4.2-28; Figure 4.2-23). Total vanadium concentrations observed in all reference and monitored lakes and streams during both the 2013 ice-covered and open water seasons were less than the vanadium SSWQO (0.003 mg/L) (see Part 2 - Data Report; Rescan 2012h). No mine effects were detected.

Table 4.2-28. Statistical Results of Total Vanadium Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake / Stream	Lakes / Streams removed from analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	ALL	Tobit	NA	-	-	-	2-317
Aug	Lake	Counts, Nanuq, Vulture, Cujo, LdS1, LdS2	Tobit	1a	-	-	None	2-319
Aug	Stream	Counts Outflow, Nanuq Outflow, Vulture-Polar, Christine-Lac du Sauvage	Tobit	1a	-	-	None	2-323

Dashes indicate not applicable.

4.3 AQUATIC BIOLOGY

The extent to which changes in water quality variables might result in changes in biological communities is a function of both the relative competitive abilities of different species under different environmental conditions (i.e., their ability to acquire resources, relative to the other species present) and each species' ability to physically tolerate changes in the concentrations of elements and molecules (toxicity). Additional changes in biological communities may result from changes in the taxonomic composition or the nutritional quality of organisms on which higher trophic levels feed.

Results from water quality analyses in the King-Cujo Watershed and Lac du Sauvage suggest that changes might be expected in biological communities downstream of the KPSF as far as Christine-Lac du Sauvage Stream, as concentrations of 13 evaluated water quality variables have increased downstream of the KPSF as a result of mine activities (see Section 4.4). However, with the exception of pH, the 95% confidence intervals around the fitted mean and the observed mean concentrations for these 13 water quality variables were below their respective CCME guidelines, SSWQOs, or other benchmark values (see Section 4.4). The lower 95% confidence interval on the fitted mean pH at site LdS1 was less than the CCME guideline; however, similar patterns were observed in all reference lakes and streams, suggesting that it was not related to mine activities.

Concentrations of water quality variables that have increased in monitored lakes at the Ekati Diamond Mine for which SSWQO or species sensitivity-based CCME guidelines exist were reviewed as part of the 2012 AEMP Re-evaluation with a specific focus on identifying possible chronic toxic effects on species present in the receiving environment at the Ekati Diamond Mine (Rescan 2012d). As in previous years, concentrations of all the water quality variables in the King-Cujo Watershed and Lac du Sauvage in 2013 remained below the lowest identified chronic effect level for the most sensitive species (Rescan 2012d). Thus, populations of even the most sensitive species were not expected to experience deleterious effects as a result of concentrations of the evaluated water quality variables in the Ekati Diamond Mine monitored lakes. In 2013, concentrations of all the water quality variables reviewed remained below the lowest identified chronic effect level for the most sensitive species in the King-Cujo Watershed and Lac du Sauvage.

The overall results of the 2012 AEMP Re-evaluation suggested that observed changes in biological community composition at the Ekati Diamond Mine likely resulted from inter-specific differences in the competitive ability of different taxonomic groups under changing quantities or ratios of macronutrients (i.e., nitrogen and phosphorus), rather than elemental toxicity (Rescan 2012d). Results from the 2012 AEMP Evaluation of Effects found no effects of mine activities on the main evaluated biological variables in the King-Cujo Watershed and Lac du Sauvage with the possible exception of a shift in lake benthos dipteran community composition in Cujo Lake (Rescan 2013b). As the trends in evaluated water quality variables in 2013 were consistent with those observed in 2011 and 2012 (Rescan 2012b, 2013b), there is little reason to expect adverse biological effects in 2013. However, it is expected that the relative availability of macronutrients could continue to be an important driver of change in biological community composition.

4.3.1 Phytoplankton

4.3.1.1 Variables

Phytoplankton are the main source of primary productivity in lake systems. Phytoplankton are also useful indicators of change because they have rapid turn-over times (from hours to days), and are sensitive to physical, chemical, and biological stressors. Previous research has shown that phytoplankton are some of the most susceptible organisms to toxins in lakes (SENES Consultants 2008). Thus, chlorophyll *a* concentrations, phytoplankton density (cells/mL), and phytoplankton diversity (Shannon and Simpson's diversity indices) were evaluated to determine whether mine activities have affected phytoplankton communities.

4.3.1.2 Dataset

Phytoplankton have been collected for analysis between late July and early August of each year for the evaluation of effects (Table 4.3-1). Baseline data, which was collected between 1994 and 1997, are included in graphical analysis but not in the statistical evaluation of effects.

Table 4.3-1. Dataset Used for Evaluation of Effects on the Phytoplankton in King-Cujo Watershed Lakes and Lac du Sauvage

Year	Nanuq	Counts	Vulture	Cujo	LdS1
1993*	-	-	-	-	-
1994*	-	-	Aug-13	-	-
1995	-	-	-	-	-
1996*	-	-	Jul-28	-	-
1997*	Aug-4	Aug-14	Aug-5	-	-

(continued)

Table 4.3-1. Dataset Used for Evaluation of Effects on the Phytoplankton in King-Cujo Watershed Lakes and Lac du Sauvage (completed)

Year	Nanuq	Counts	Vulture	Cujo	LdS1
1998	Aug-3	Aug-3	Aug-6	-	-
1999	Aug-7	Aug-8	Aug-6	Aug-8	-
2000	Aug-4	Aug-1	Aug-4	Jul-31	Aug-2
2001	Aug-1	Jul-30	Aug-2	Jul-30	Jul-31
2002	Aug-1	Aug-7	Aug-3	Aug-7	Aug-5
2003	Aug-9	Aug-7	Aug-4	Aug-4	Aug-6
2004	Aug-10	Aug-13	Aug-9	Aug-10	Aug-10
2005	Aug-1	Aug-7	Jul-31	Aug-9	Aug-9
2006	Aug-2	Aug-4	Aug-2	Aug-4	Aug-1
2007	Aug-11	Aug-6	Aug-12	Aug-5	Aug-5
2008	Aug-8	Jul-31	Jul-29	Jul-26	Jul-31
2009	Jul-30	Aug-1	Jul-30	Jul-31	Aug-1
2010	Aug-6	Aug-7	Aug-5	Aug-4	Aug-4
2011	Aug-2	Aug-5	Aug-5	Aug-4	Aug-4
2012	Aug-1	Aug-8	Aug-7	Aug-6	Aug-7
2013	Aug-3	Aug-1	Aug-1	Jul-30	Aug-3

** = Data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison.*
Dashes indicate no data were available.
Single samples were collected yearly for biomass analysis from 1993 to 1996.
Triplicate samples were collected yearly from 1996 to 2013 for density and diversity analysis.

4.3.1.3 Results and Discussion

<u>Chlorophyll *a*</u>

Statistical and graphical analyses suggest that chlorophyll *a* concentrations have been stable through time, relative to reference lakes, in all monitored lakes (Table 4.3-2; Figure 4.3-1). Mean chlorophyll *a* concentrations were within the range of mean baseline concentrations ± 2 SD in all monitored lakes in 2013 (Table 4.3-3). Thus, no mine effects were detected with respect to chlorophyll *a* concentrations.

Table 4.3-2. Statistical Results of Chlorophyll *a* Concentrations in King-Cujo Watershed Lakes and Lac du Sauvage

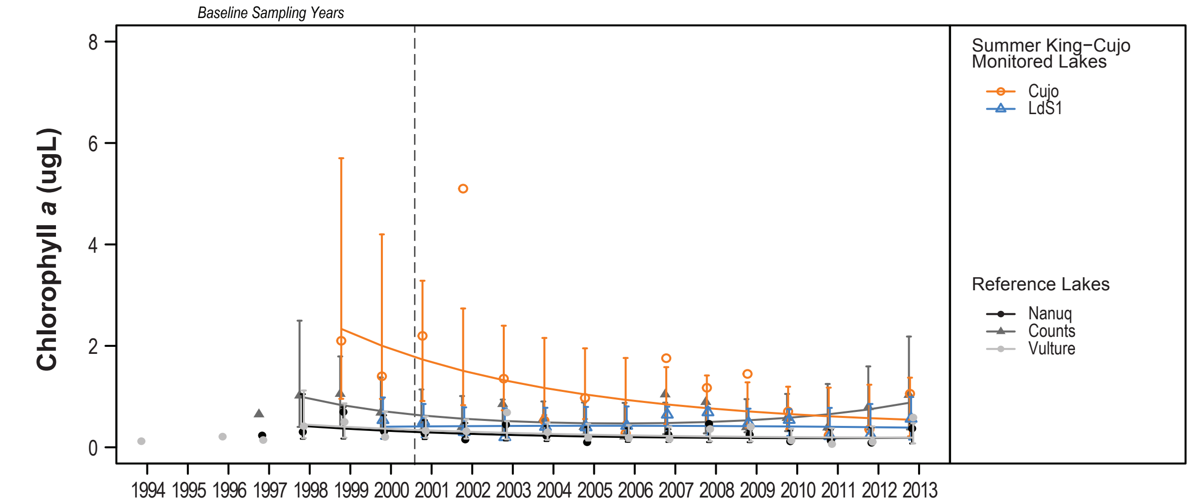
	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
				Model Fit = 3	Model Fit = 2	Model Fit = 1	
Chlorophyll *a*	-	LME	2	-	None	-	2-327

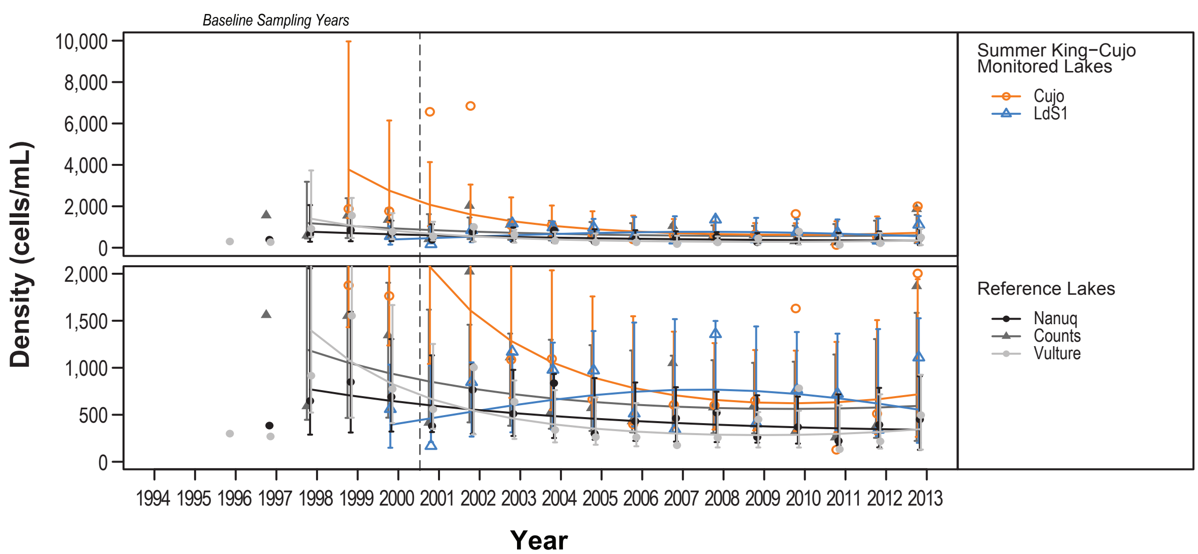
Dashes indicate not applicable.

<u>Density</u>

Statistical and graphical analyses indicate that phytoplankton densities have been stable through time, relative to reference lakes, in all monitored lakes (Table 4.3-4; Figure 4.3-1). Moreover, phytoplankton densities in Cujo Lake and at site LdS1 in 2013 remained within ± 2 SD of the mean observed phytoplankton densities in baseline years (Table 4.3-5). Thus, no mine effects were detected with respect to phytoplankton density.

Observed and Fitted Means for Chlorophyll *a* Concentrations and Phytoplankton Density in King-Cujo Watershed Lakes and Lac du Sauvage, August 1994 to 2013





Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.

Table 4.3-3. Mean ± 2 Standard Deviations (SD) Baseline Concentrations of Chlorophyll *a* in Each of the King-Cujo Watershed Lakes and Lac du Sauvage

Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD
Nanuq	0.23 (1)	0 - 0.51	0.37 ± 0.48
Counts	0.65 (1)	0 - 1.45	1.03 ± 0.82
Vulture	0.15 (2)	0.08 - 0.23	0.59 ± 0.09
Cujo	1.75 (2)	0.93 - 2.57	1.06 ± 0.72
LdS1	0.54 (1)	0.39 - 0.68	0.56 ± 0.16

Units are µg/L.
Negative values were replaced with zeros.
N = number of years data were collected.

Table 4.3-4. Statistical Results of Phytoplankton Density in Lakes in the King-Cujo Watershed and Lac du Sauvage

	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
				Model Fit = 3	Model Fit = 2	Model Fit = 1	
Phytoplankton density	-	LME	3	None	None	-	2-333

Dashes indicate not applicable.

Table 4.3-5. Mean ± 2 Standard Deviations (SD) Baseline Phytoplankton Density in Each of the King-Cujo Watershed Lakes and Lac du Sauvage

Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD
Nanuq	385 (1)	56 – 714	450 ± 118
Counts	1,561 (1)	103 - 3,020	1,868 ± 724
Vulture	284 (2)	76 – 492	496 ± 26
Cujo	1821 (2)	840 – 2,801	2,003 ± 479
LdS1	561 (1)	0 – 1,379	1,109 ± 696

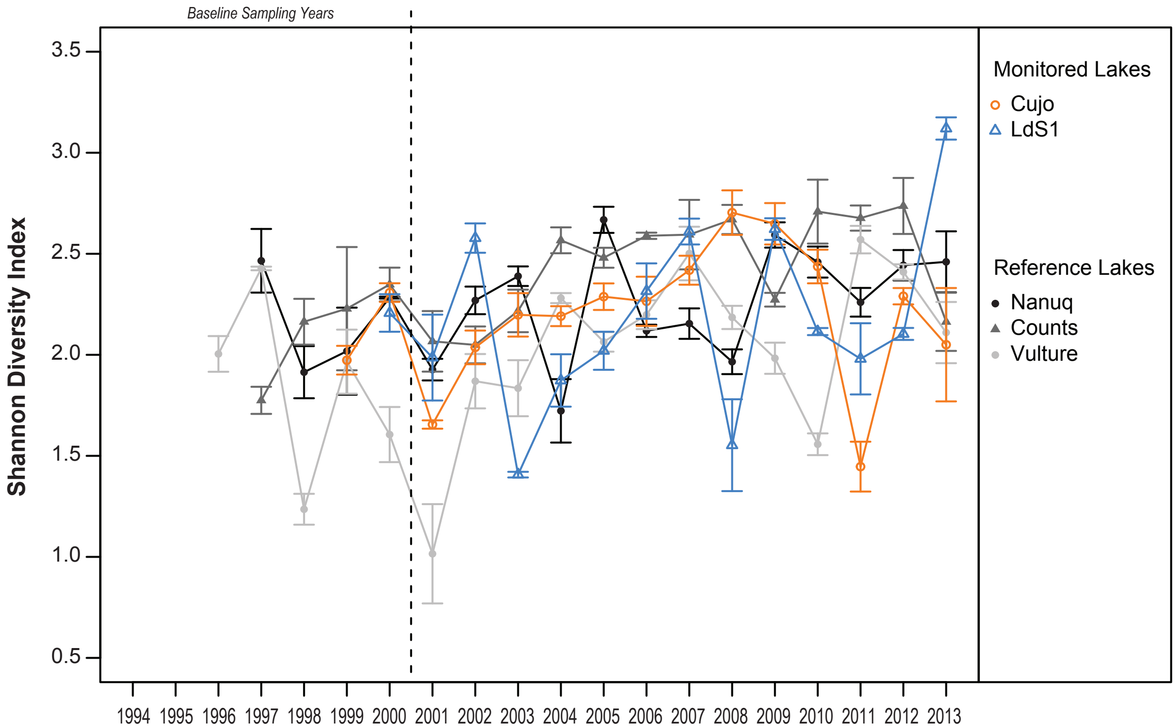
Units are cells/mL.
Negative values were replaced with zeros.
N = number of years data were collected.

Diversity

Statistical analyses were not performed on the diversity datasets because the calculation of indices can result in data abnormalities that prevent statistical analysis. Consequently, graphical analyses of temporal trends in diversity indices (Figure 4.3-2) and best professional judgment were the primary methods used in the evaluation of effects. In addition, the average and relative densities of taxa were examined using graphical analyses to identify potential changes in community composition (Figures 4.3-3 to 4.3-4). Following recent advances in taxonomic classification, the names of two phytoplankton groups have been updated: the Cyanophyta are now recognized as the class Myxophyceae and the Pyrrophyta are now recognized as the class Dinophyceae.

Figure 4.3-2

Average Diversity Indices for Phytoplankton in King-Cujo Watershed Lakes and Lac du Sauvage, August 1996 to 2013



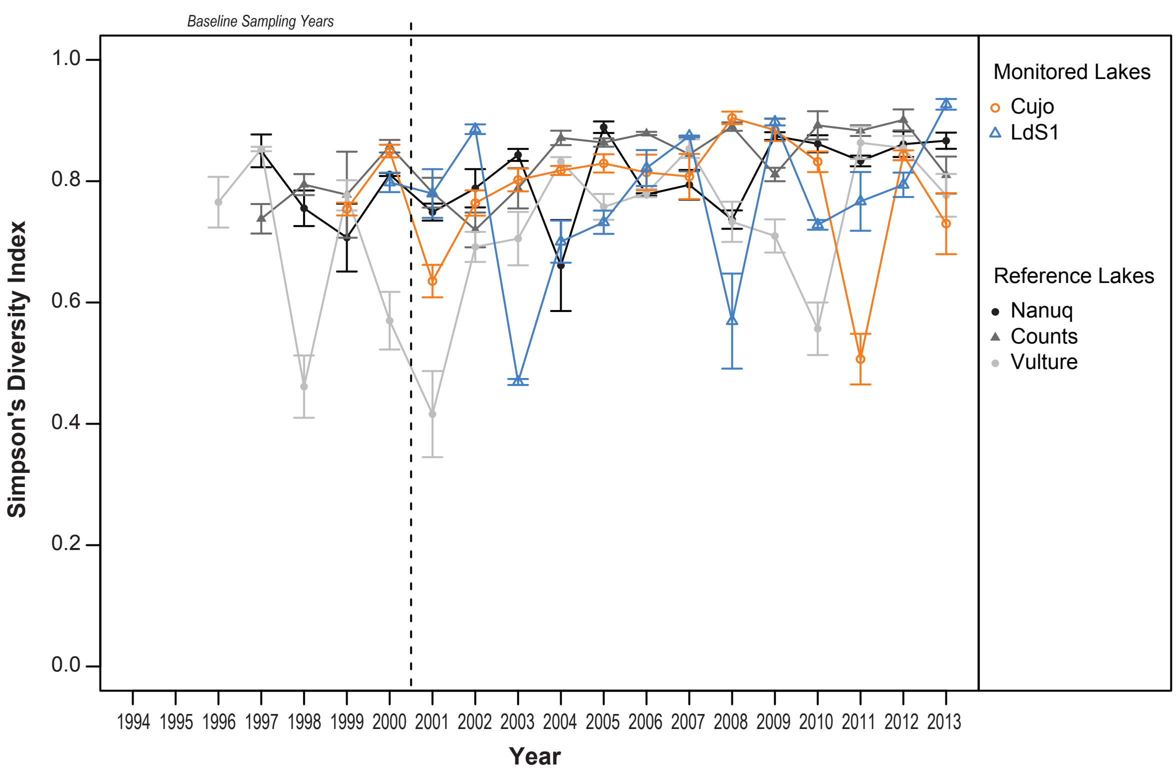
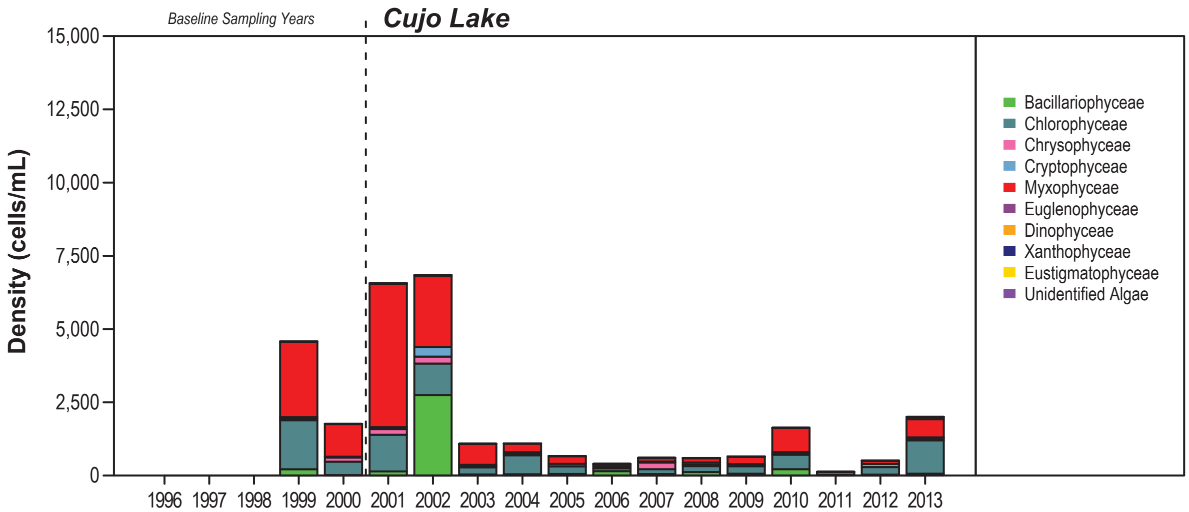


Figure 4.3-3

Average Phytoplankton Density by Taxonomic Group for Lakes of the King-Cujo Watershed and Lac du Sauvage, 1995 to 2013



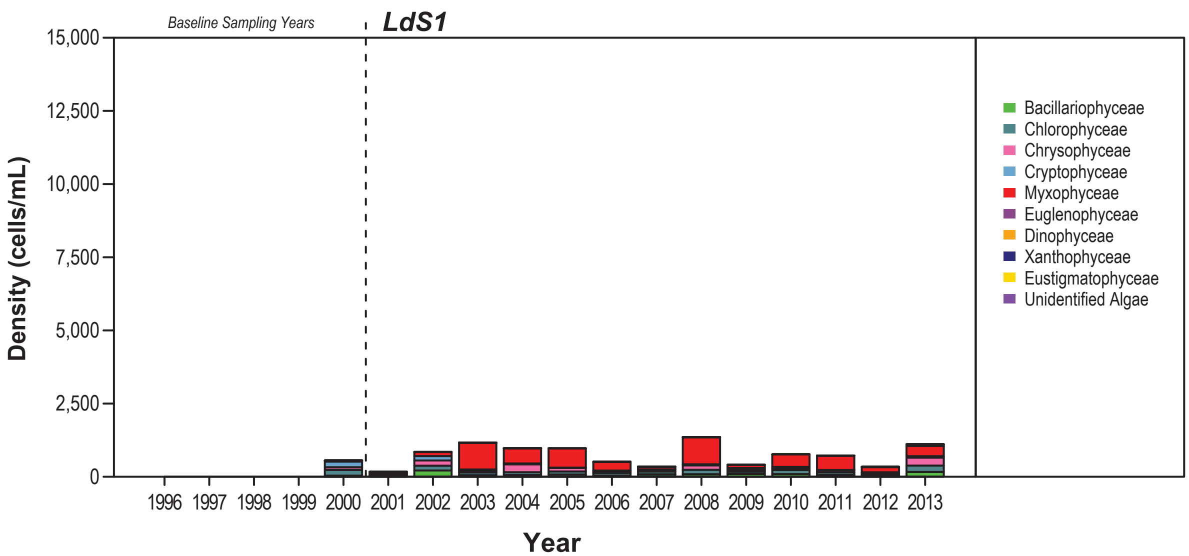
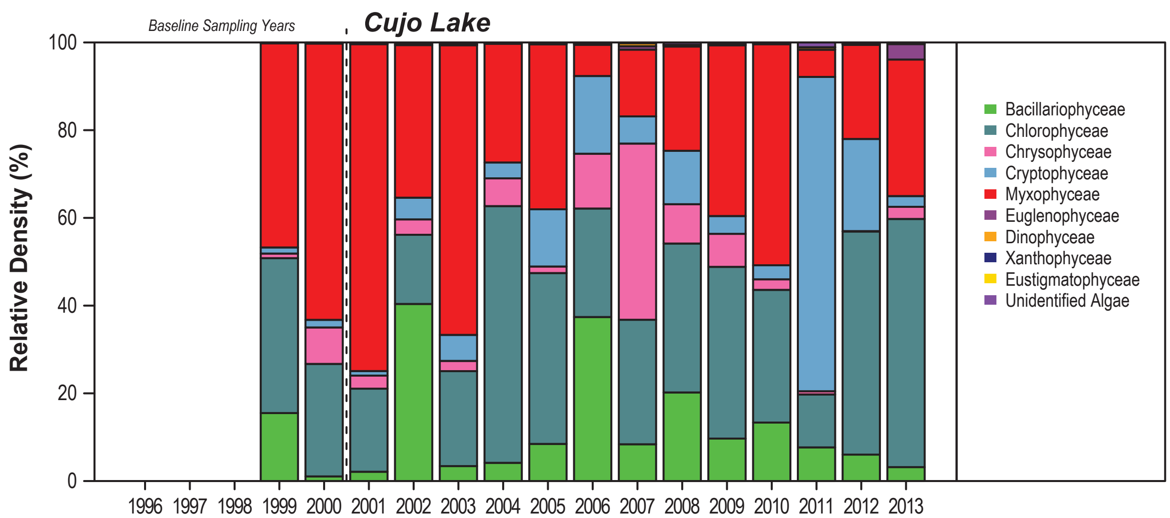
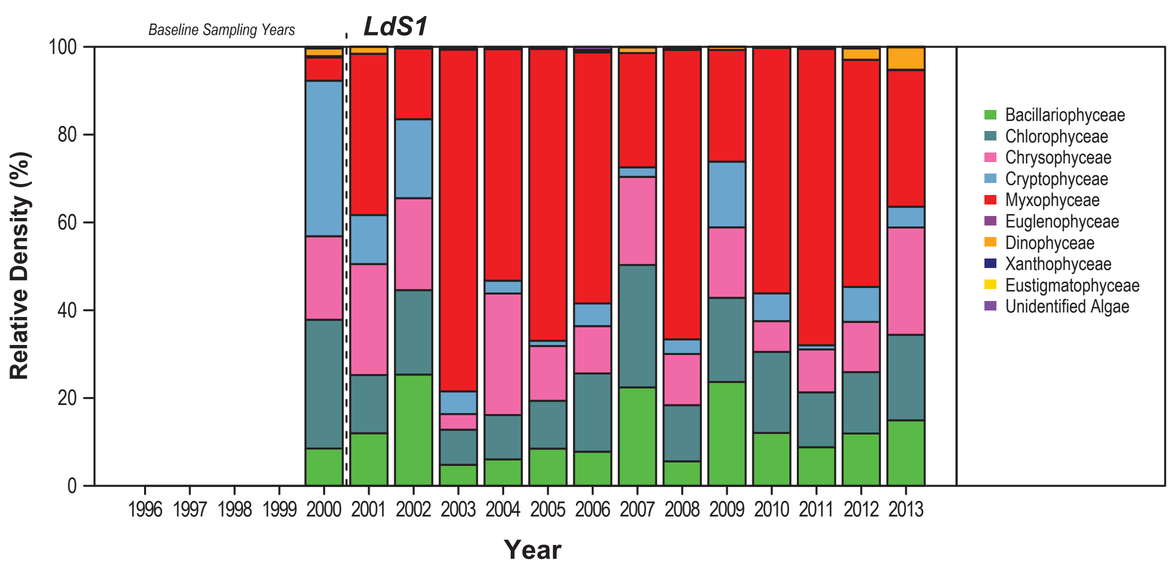


Figure 4.3-4

Relative Densities of Phytoplankton Taxa in Lakes of the King-Cujo Watershed and Lac du Sauvage, 1995 to 2013





Both Shannon and Simpson's diversity indices have varied considerably through time in monitored and reference lakes since monitoring began (Figure 4.3-2). Mean Shannon and Simpson's diversity was greater than ± 2 SD of mean baseline diversity at LdS1 in 2013 (Table 4.3-6). However, a similar pattern for Shannon diversity was observed in one reference lake (i.e., Counts Lake; Table 4.3-6). The increase in diversity observed at LdS1 likely reflects an increase in the absolute densities of Chrysophyceae, Cryptophyceae and Bacillariophyceae, corresponding to a more even distribution in abundance among the phytoplankton groups in 2013 (Figures 4.3-3 and 4.3-4). Graphical analysis of species composition at LdS1 has been variable through time; however, no directional shift in species composition was observed at LdS1 or in any other monitored lake in the King-Cujo Watershed (Figures 4.3-3 and 4.3-4). Thus, no mine effects were detected with respect to phytoplankton diversity or taxonomic composition in lakes of the King-Cujo Watershed or Lac du Sauvage.

Table 4.3-6. Mean ± 2 Standard Deviations (SD) Baseline Phytoplankton Diversity in Each of the King-Cujo Watershed Lakes and Lac du Sauvage

	Shannon Diversity			Simpson's Diversity		
Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD
Nanuq	2.46 (1)	1.92 - 3.01	2.46 ± 0.26	0.85 (1)	0.76 - 0.94	0.87 ± 0.02
Counts	1.77 (1)	1.54 - 2.01	2.16 ± 0.25	0.74 (1)	0.65 - 0.82	0.81 ± 0.05
Vulture	2.22 (3)	1.71 - 2.72	2.11 ± 0.26	0.81 (3)	0.68 - 0.94	0.78 ± 0.06
Cujo	2.14 (2)	1.73 - 2.55	2.05 ± 0.49	0.80 (2)	0.69 - 0.91	0.73 ± 0.09
LdS1	2.21 (1)	1.89 - 2.53	3.12 ± 0.10	0.80 (1)	0.74 - 0.85	0.93± 0.02

N = number of years data were collected.

No mine effects were detected with respect to phytoplankton biomass, density, diversity, or community composition in the King-Cujo Watershed or Lac du Sauvage, thus no cascading effects through the foodweb are expected (i.e., no changes in zooplankton or benthos lake communities).

4.3.2 Zooplankton

4.3.2.1 Variables

Zooplankton are primary and secondary consumers that play an important role in the aquatic food web. Zooplankton feed on phytoplankton or other zooplankton and serve as an important food source for fish. Zooplankton monitoring can be used to help determine the extent to which mine effects have cascaded through the food web. Phytoplankton populations may appear to be suppressed despite increases in overall phytoplankton productivity due to the consumption of phytoplankton by zooplankton. Consequently, changes in the overall productivity may not be reflected in phytoplankton populations, but may be indicated by increases in zooplankton densities or changes in zooplankton community composition. Zooplankton community composition can also be used as an indicator of changes in water quality in the receiving environment as different species occupy different water chemistry niches and have different tolerances to changes in water quality. Therefore, zooplankton biomass (mg dry weight/m^3), density (organisms/m^3), and diversity (Shannon and Simpson's diversity indices) were monitored to detect potential mine effects.

4.3.2.2 Dataset

Zooplankton data have been collected during late July or August each year from 1995 to 2013 (Table 4.3-7). Zooplankton biomass and taxonomic composition has been monitored using triplicate sampling from 1998 to present. Prior to 1998, zooplankton were monitored for taxonomic composition only. Baseline data, collected between 1994 and 1997, are included in Table 4.3-7 and depicted graphically, below, but are not included in the statistical evaluation of effects.

Table 4.3-7. Dataset Used Evaluation of Effects on the Zooplankton in King-Cujo Watershed Lakes and Lac du Sauvage

Year	Nanuq	Counts	Vulture	Cujo	LdS1
1994	-	-	-	-	-
1995*	-	-	Aug-8	-	-
1996*	-	-	Jul-28	-	-
1997*	Aug-4	Aug-14	Aug-5	-	-
1998	Aug-4	Aug-4	Aug-7	-	-
1999	Aug-8	Aug-7	Aug-6	Aug-8	-
2000	Aug-4	Aug-1	Aug-4	Jul-31	Aug-2
2001	Aug-1	Jul-30	Aug-2	Jul-30	Jul-31
2002	Aug-1	Aug-7	Aug-3	Aug-7	Aug-5
2003	Aug-9	Aug-3	Aug-4	Aug-4	Aug-6
2004	Aug-10	Aug-13	Aug-9	Aug-9	Aug-10
2005	Aug-10	Aug-13	Aug-9	Aug-9	Aug-9
2006	Aug-2	Aug-4	Aug-2	Aug-4	Aug-1
2007	Aug-11	Aug-6	Aug-12	Aug-5	Aug-5
2008	Aug-8	Jul-31	Jul-29	Jul-26	Jul-31
2009	Jul-30	Aug-1	Jul-30	Jul-31	Aug-1
2010	Aug-6	Aug-7	Aug-5	Aug-4	Aug-4
2011	Aug-2	Aug-5	Aug-5	Aug-4	Aug-4
2012	Aug-1	Aug-8	Aug-7	Aug-6	Aug-7
2013	Aug-3	Aug-1	Aug-1	Aug-6	Aug-3

** = Data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison.*
Dashes indicate no data were available
Biomass data available for 1998-2013 only.

4.3.2.3 Results and Discussion

<u>Biomass</u>

Statistical and graphical analyses suggest that zooplankton biomass has been relatively stable through time in all monitored and reference lakes (Figure 4.3-5; Table 4.3-8). Zooplankton biomass in 2013 was greater than the range of ± 2 two SD of mean biomass in baseline years in Cujo Lake and lower than the range of ± 2 two SD of mean biomass in baseline years at site LdS1 in Lac du Sauvage (Table 4.3-9). However, observed zooplankton biomass in Cujo Lake at site LdS1 has remained within the range of values observed through time (Figure 4.3-5). Thus, no mine effects were detected with respect to zooplankton biomass.

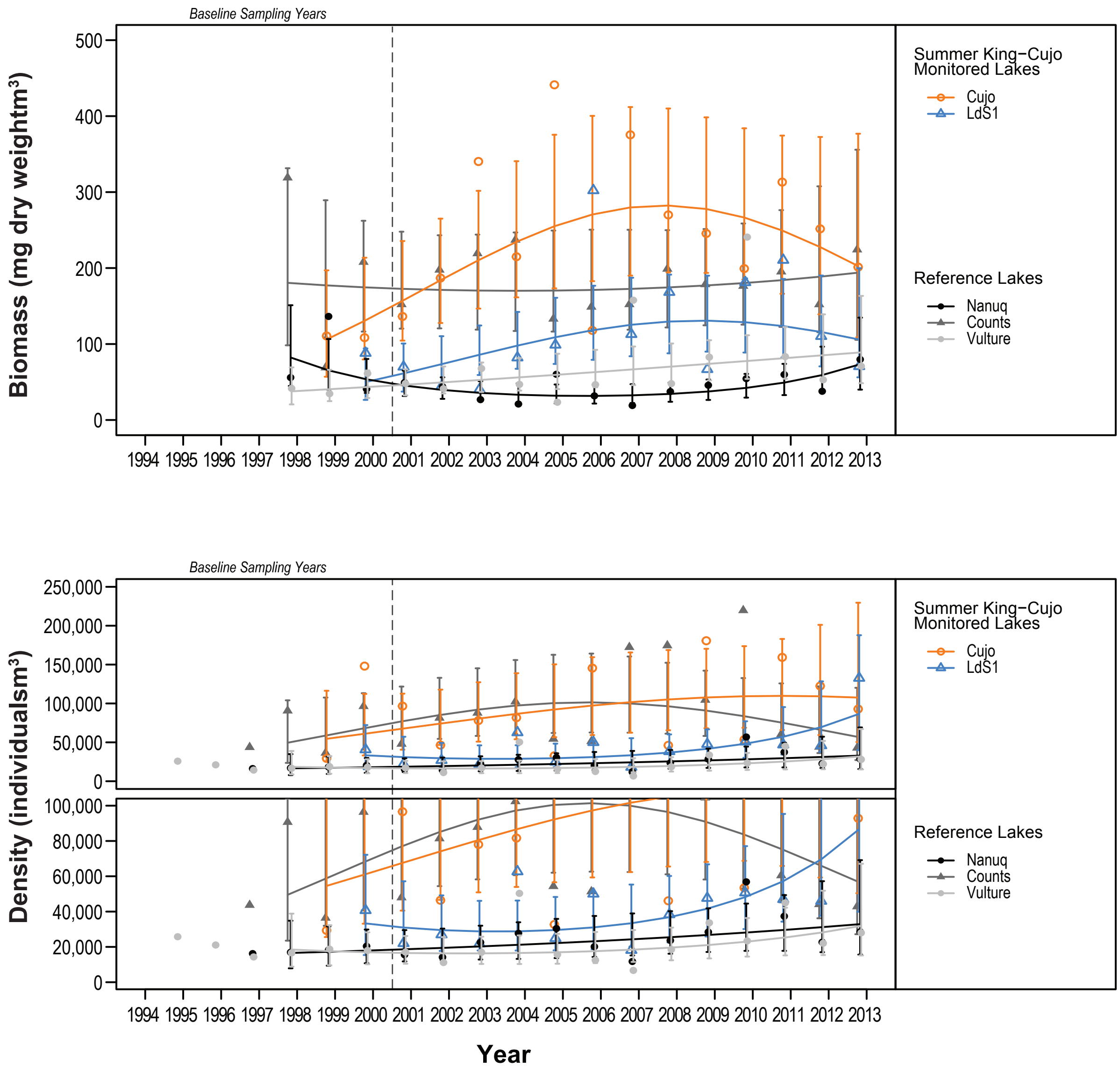
Table 4.3-8. Statistical Results of Zooplankton Biomass in Lakes in the King-Cujo Watershed and Lac du Sauvage

	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
				Model Fit = 3	Model Fit = 2	Model Fit = 1	
Zooplankton biomass	-	LME	1a	-	-	None	2-339

Dashes indicate not applicable.

Figure 4.3-5

Observed and Fitted Means for Zooplankton Biomass and Density in King-Cujo Watershed Lakes and Lac du Sauvage, August 1995 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.

Table 4.3-9. Mean ± 2 Standard Deviations (SD) Baseline Zooplankton Biomass in each of the King-Cujo Watershed Lakes and Lac de Gras

Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD
Nanuq	76 (3)	0 - 168	79.7 ± 6.1
Counts	198 (3)	0 - 418	224.3 ± 100.0
Vulture	46 (3)	21 - 71	70.8 ± 5.4
Cujo	110 (1)	70 - 150	201.3 ± 42.6
LdS1	88 (1)	76 – 100	70.7 ± 25.8

Units are g/m^3.
Negative values were replaced with zeros.
N = number of years data were collected.

Density

Statistical and graphical analyses of observed zooplankton densities suggest that densities have been stable through time in all monitored and reference lakes (Table 4.3-10; Figure 4.3-5). Mean zooplankton densities in 2013 were greater than mean baseline densities ± 2 SD at site LdS1 in Lac du Sauvage (Table 4.3-11). However, 2013 mean zooplankton densities were also greater than baseline densities in one of the reference lakes (i.e., Nanuq Lake) and zooplankton densities in lakes upstream of Lac du Sauvage were similar to baseline densities (Table 4.3-11). Thus, no mine effects were detected with respect to zooplankton density in lakes of the King-Cujo watershed or Lac du Sauvage.

Table 4.3-10. Statistical Results of Zooplankton Density in Lakes in the King-Cujo Watershed and Lac du Sauvage

	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
				Model Fit = 3	Model Fit = 2	Model Fit = 1	
Zooplankton density	-	LME	3	None	None	-	2-344

Dashes indicate not applicable.

Table 4.3-11. Mean ± 2 Standard Deviations (SD) Baseline Zooplankton Density in Each of the King-Cujo Watershed Lakes and Lac du Sauvage

Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD
Nanuq	16,209 (1)	13,053 - 19,365	28,547 ± 1,923
Counts	43,710 (1)	33,027 - 54,392	42,894 ± 4,916
Vulture	20,384 (3)	9,704 - 31,064	27,987 ± 1,506
Cujo	88,687 (2)	0 – 219,780	92,857 ± 53,152
LdS1	40,734 (1)	32,780 – 48,688	132,681 ± 12,275

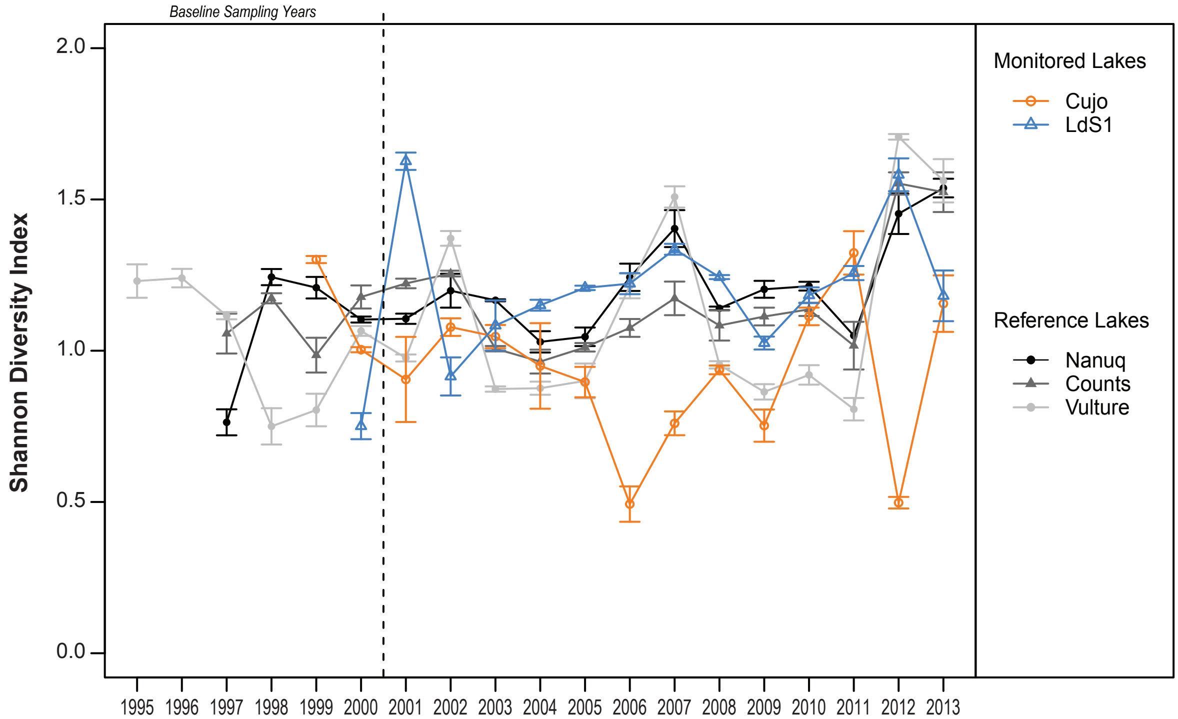
Units are organisms/m^3.
Negative values were replaced with zeros.
N = number of years data were collected.

Diversity

Statistical analyses were not performed on the diversity datasets because the calculation of indices can result in data abnormalities that prevent statistical analysis. Thus, graphical analyses of temporal trends in diversity indices (Figure 4.3-6) and best professional judgment were the primary methods used in the evaluation of effects. In addition, the average and relative densities of taxa were examined using graphical analyses to identify potential changes in community composition (Figures 4.3-7 and 4.3-8).

Figure 4.3-6

Average Diversity Indices for Zooplankton in King-Cujo Watershed Lakes and Lac du Sauvage, August 1995 to 2013



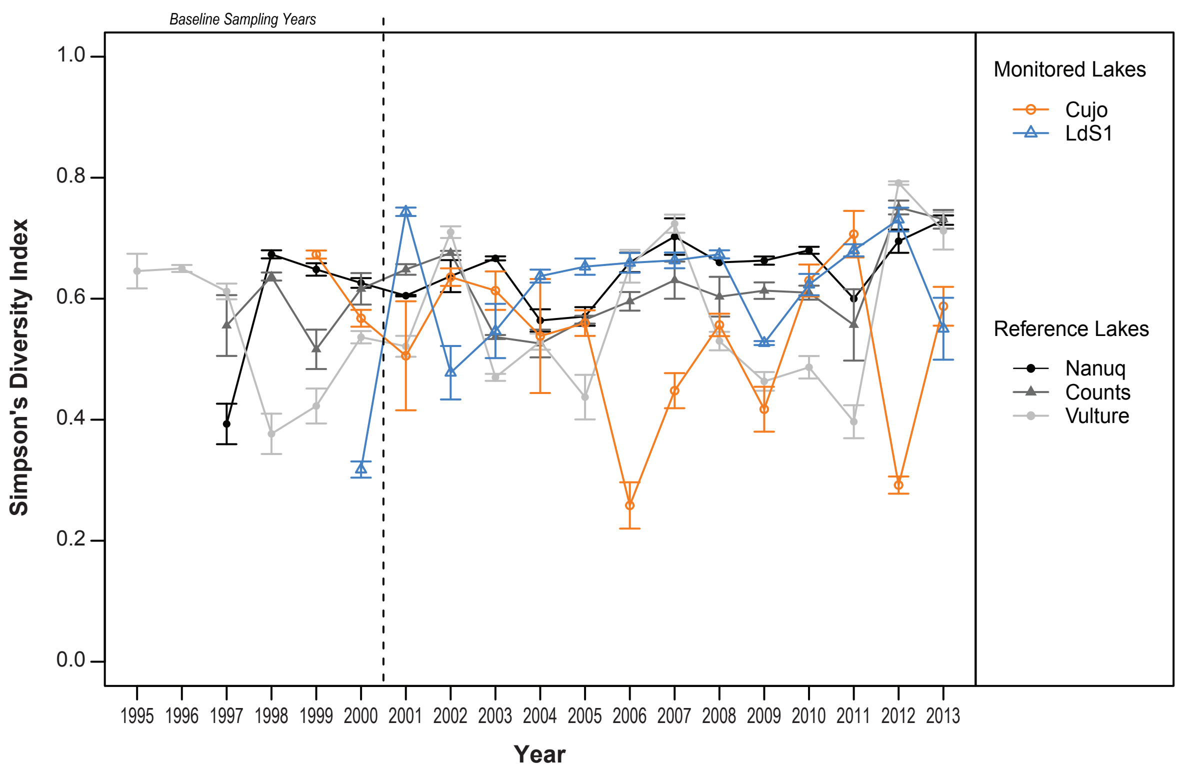
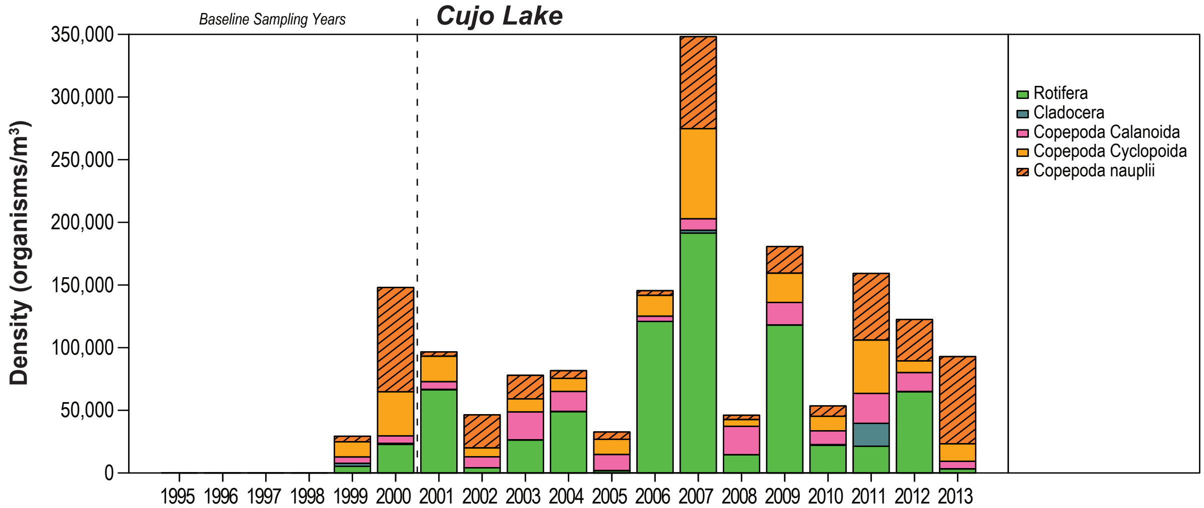


Figure 4.3-7

Average Zooplankton Density by Taxonomic Group for Lakes of the King-Cujo Watershed, 1995 to 2013



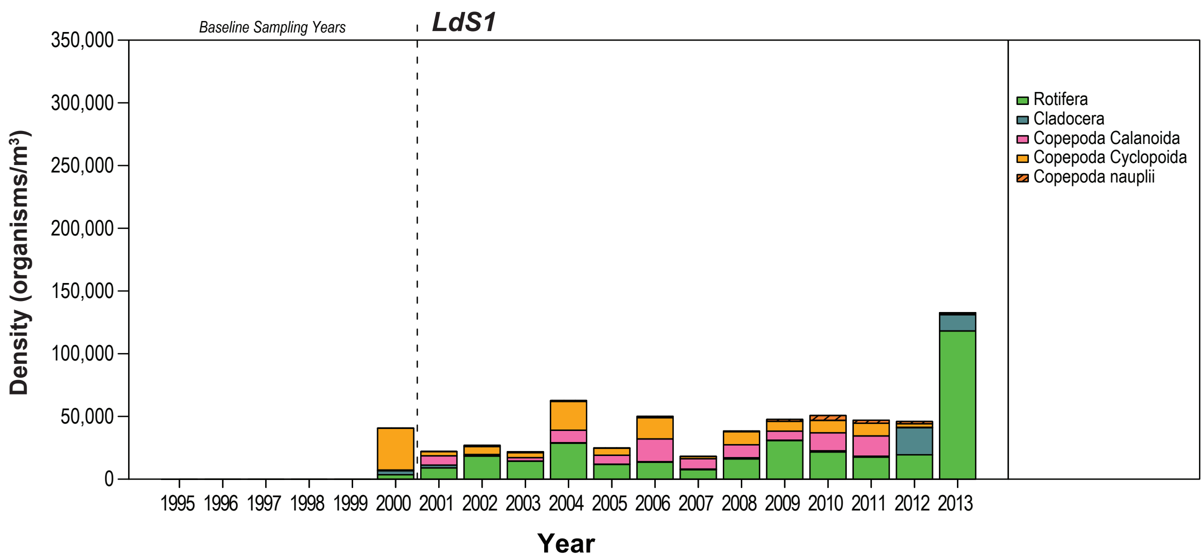
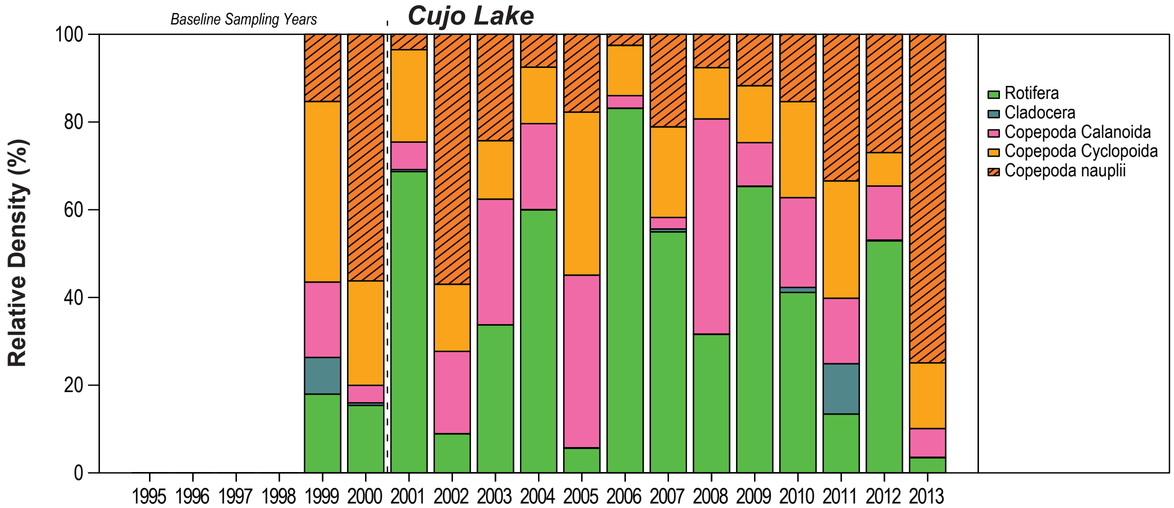
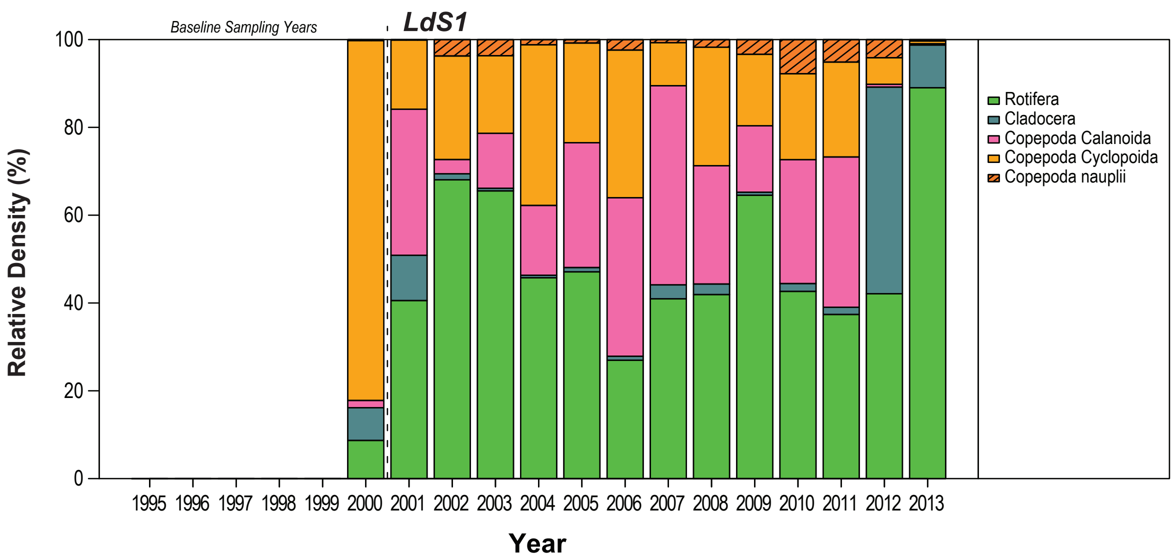


Figure 4.3-8

Relative Densities of Zooplankton Taxa in King-Cujo Watershed Lakes, 1995 to 2013





Both Shannon and Simpson's diversity indices have varied considerably through time in both monitored and reference lakes (Figure 4.3-6). While the variability makes it somewhat difficult to discern temporal trends, both Shannon and Simpson's diversity indices have remained relatively stable through time in all monitored and reference lakes (Figure 4.3-6). Mean diversity in 2013 was within the range of two standard deviations of the baseline mean diversity in Cujo Lake, but was higher at site LdS1 in Lac du Sauvage and in all reference lakes (Table 4.3-12). The relative densities of different zooplankton taxonomic groups has remained largely consistent through time in both Cujo Lake and at site LdS1 in Lac du Sauvage, though community composition is somewhat more variable through time in Cujo Lake than in either the reference lakes or at site LdS1 in Lac du Sauvage (Figure 4.3-9 to 4.3-10). Zooplankton community composition at site LdS1 has differed from those observed in previous years. Specifically, copepods have been replaced by rotifers and cladocerans. However, no such pattern was observed upstream at Cujo Lake. Thus it was concluded that no mine effects were detected.

Table 4.3-12. Mean ± 2 Standard Deviations (SD) Baseline Zooplankton Diversity in Each of the King-Cujo Watershed Lakes and Lac du Sauvage

	Shannon Diversity			Simpson's Diversity		
Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD
Nanuq	0.76 (1)	0.61 – 0.91	1.54 ± 0.05	0.39 (1)	0.28 – 0.51	0.73 ± 0.01
Counts	1.06 (1)	0.83 – 1.29	1.52 ± 0.11	0.56 (1)	0.38 – 0.73	0.73 ± 0.03
Vulture	1.19 (3)	1.03 – 1.36	1.56 ± 0.12	0.64 (3)	0.57 – 0.70	0.71 ± 0.05
Cujo	1.15 (2)	0.82 - 1.48	1.16 ± 0.16	0.62 (2)	0.50 – 0.74	0.59 ± 0.06
LdS1	0.75 (1)	0.60 – 0.90	1.18 ± 0.15	0.32 (1)	0.27 – 0.36	0.55 ± 0.09

N = number of years data were collected.

Although no mine effects were detected with respect to zooplankton diversity, the rotifer *Conochilus sp*. and the cladoceran *Holopedium gibberum*, have been largely absent from Cujo Lake since 2002. This trend in the densities of these two species is similar to that observed downstream of the LLCF (see Section 3.3.2.3). Hypotheses regarding potential underlying causes of changes in zooplankton communities and their potential effects on higher trophic levels are included in Aquatic Biology Summary (see Section 4.3.5).

4.3.3 Lake Benthos

4.3.3.1 Variables

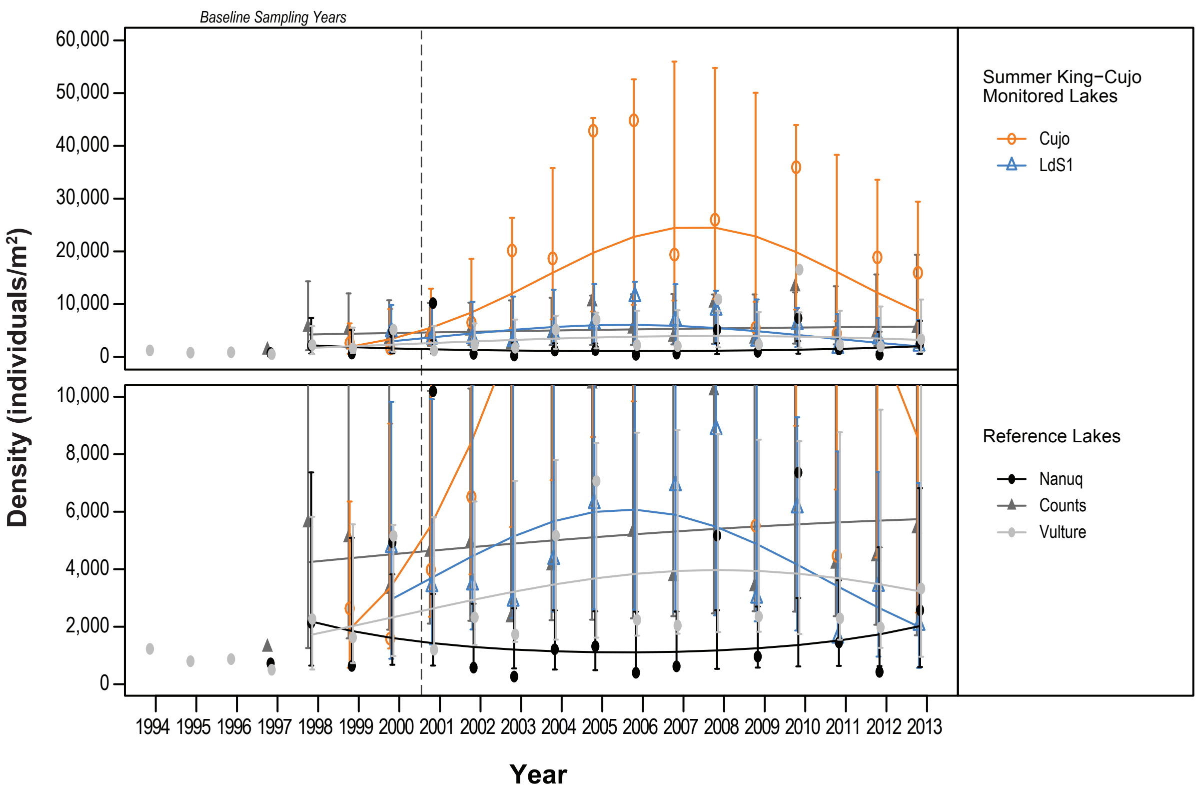
Lake benthos are a group of organisms that live in association with lake sediments. They provide an important source of food for many species of fish. Dipterans (flies) tend to dominate benthic invertebrate communities and are widely used as indicators of ecosystem health, including sediment quality. Thus, lake benthos density (organisms/m^2) and dipteran diversity (Shannon and Simpson's diversity indices) were evaluated for potential mine effects.

4.3.3.2 Dataset

Benthos samples have been collected in triplicate replicates in late July or early August of each year since 1994 (Table 4.3-13). Beginning in 2011, composite samples, consisting of three subsamples per replicate, were collected. Baseline data was collected between 1994 and 1997, and was not used in the statistical evaluation of effects but are included in Table 4.3-13 and shown graphically, below, for visual comparison.

Figure 4.3-9

Observed and Fitted Means for Benthos Densities in King-Cujo Watershed Lakes and Lac du Sauvage, August 1994 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.

Figure 4.3-10

Average Diversity Indices for Benthic Dipterans in King-Cujo Watershed Lakes and Lac du Sauvage, August 1994 to 2013

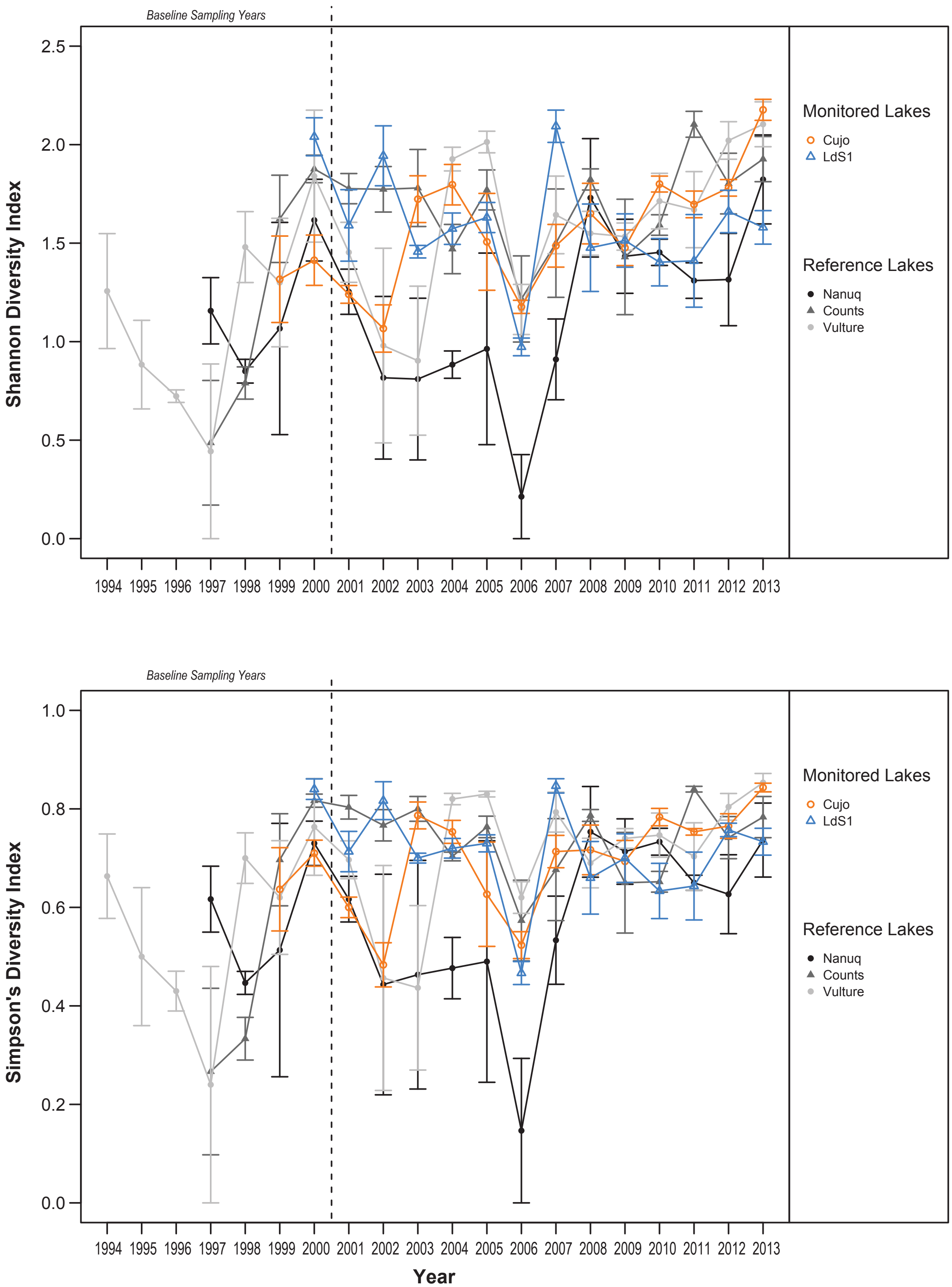


Table 4.3-13. Dataset Used for Evaluation of Effects on Benthos in King-Cujo Watershed Lakes and Lac du Sauvage

Year	Nanuq	Counts	Vulture	Cujo	LdS1
1994*	-	-	Aug-13	-	-
1995*	-	-	Aug-9	-	-
1996*	-	-	Jul-27	-	-
1997*	Aug-4	Aug-14	Aug-5	-	-
1998	Aug-4	Aug-4	Aug-7	-	-
1999	Jul-30	Jul-30	Jul-29	Jul-31	-
2000	Aug-4	Aug-1	Aug-4	Jul-31	Aug-2
2001	Aug-1	Jul-30	Aug-2	Jul-30	Jul-31
2002	Aug-3	Aug-7	Aug-3	Aug-7	Aug-5
2003	Aug-9	Aug-7	Aug-4	Aug-4	Aug-6
2004	Aug-10	Aug-13	Aug-9	Aug-10	Aug-10
2005	Aug-1	Aug-7	Jul-31	Aug-8	Aug-8
2006	Aug-2	Aug-4	Aug-2	Aug-4	Aug-1
2007	Aug-11	Aug-6	Aug-12	Aug-5	Aug-5
2008	Aug-8	Jul-31	Aug-5	Jul-26	Jul-31
2009	Jul-30	Aug-1	Jul-30	Jul-31	Aug-1
2010	Aug-6	Aug-8	Aug-5	Aug-4	Aug-4
2011	Aug-2	Aug-5	Aug-5	Aug-4	Aug-4
2012	Aug-9	Aug-6	Aug-7	Aug-7	Aug-7
2013	Aug-3	Aug-1	Jul-31	Jul-30	Aug-3

** = Data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison.*
Dashes indicate no data were available.

4.3.3.3 Results and Discussion

<u>Density</u>

Statistical analyses indicate that temporal trends in benthos density in Cujo Lake differed significantly from the common slope of the reference lakes (Table 4.3-14). However, graphical analyses suggest that observed benthos density has been fairly consistent through time in all monitored and reference lakes (Figure 4.3-9). Mean benthos density in 2013 was greater than the range of ± 2 SD observed in baseline years in Cujo Lake, but similar patterns were observed in all three reference lakes (Table 4.3-15). Mean benthos density at site LdS1 in 2013 was lower than the range of ± 2 SD observed in baseline years; however, benthos density has remained within the range of values observed through time at site LdS1 (Figure 4.3-9). Thus, it was concluded that no mine effects were detected with respect to benthos density.

Table 4.3-14. Statistical Results of Benthos Density in Lakes in the King-Cujo Watershed and Lac du Sauvage

	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
				Model Fit = 3	Model Fit = 2	Model Fit = 1	
Density	-	LME	2	-	Cujo	-	2-349

Dashes indicate not applicable.

Table 4.3-15. Mean ± 2 Standard Deviations (SD) Baseline Benthos Density in Each of the King-Cujo Watershed Lakes and Lac du Sauvage

Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean, ± 1 SD
Nanuq	726 (1)	325 – 1,126	2,573 ± 1,420
Counts	1,289 (1)	0 – 3,212	5,406 ± 2,188
Vulture	852 (4)	0 – 1,960	3,328 ± 2,338
Cujo	2,111 (2)	0 – 4,379	15,941 ± 7,610
LdS1	4,755 (1)	3,021 – 6,490	2,049 ± 732

Units are organisms/m^2.
Negative values were replaced with zeros.
N = number of years data were collected.

Dipteran Diversity

Statistical analyses were not performed on the diversity datasets because the calculation of indices can result in data abnormalities that prevent statistical analysis. Consequently, graphical analyses of temporal trends in diversity indices (Figure 4.3-10) and best professional judgment were the primary methods used in the evaluation of effects. In addition, the average and relative densities of taxa were examined using graphical analyses to identify potential changes in community composition (Figures 4.3-11 and 4.3-12).

Both Shannon and Simpson's diversity indices have varied considerably through time in both monitored and reference lakes since monitoring began, though less variability has been observed in most reference and monitored lakes since 2007 (Figure 4.3-10). While the variability makes it somewhat difficult to discern temporal trends, both diversity indices have been relatively stable through time in both Cujo Lake and at site LdS1 in Lac du Sauvage (Figure 4.3-10). However, diversity was greater in Cujo Lake in 2013 than in any previous year (Figure 4.3-10).

Shannon diversity in 2013 was greater than the range of ± 2 SD of the mean in baseline years in Cujo Lake (Table 4.3-16). Simpson's diversity in 2013 was within the range of ± 2 SD of of the mean in baseline years in Cujo Lake, but below the range of baseline values at site LdS1 in Lac du Sauvage (Table 4.3-16). Mean Shannon and Simpson's diversity indices were greater than mean baseline densities ± 2 SD in all three reference lakes in 2013, except Counts Lake, in which only Shannon diversity was greater (Table 4.3-16). Although a decrease in Simpson's diversity relative to baseline years was detected at site LdS1 in Lac du Sauvage in 2013, there is no clear temporal trend in either Shannon or Simpson's diversity at site LdS1 and therefore it was concluded that no mine effects were detected.

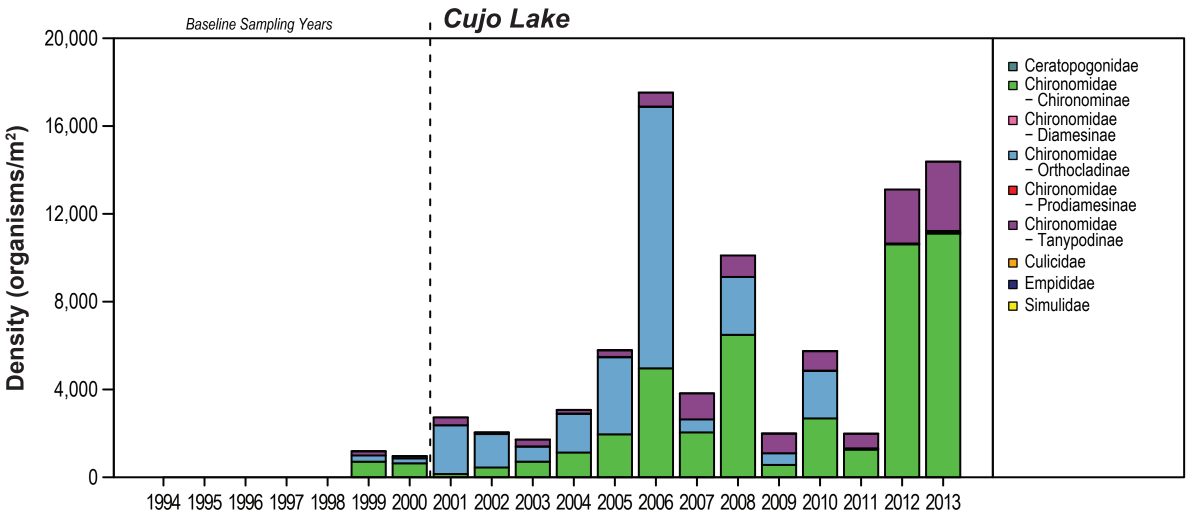
Table 4.3-16. Mean ± 2 Standard Deviations (SD) Baseline Dipteran Diversity in Each of the King-Cujo Watershed Lakes and Lac du Sauvage

	Shannon Diversity			Simpson's Diversity		
Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD
Nanuq	1.16 (1)	0.57 - 1.75	1.82 ± 0.39	0.39 (1)	0.28 - 0.51	0.74 ± 0.13
Counts	0.49 (1)	0 - 1.59	1.93 ± 0.20	0.27 (1)	0 - 0.86	0.78 ± 0.07
Vulture	0.44 (4)	0 - 1.49	2.10 ± 0.20	0.24 (4)	0.61 - 0.78	0.85 ± 0.03
Cujo	1.37 (2)	0.80 – 1.93	2.18 ± 0.09	0.67 (2)	0.46 – 0.88	0.84 ± 0.02
LdS1	2.04 (1)	1.71 – 2.37	1.58 ± 0.15	0.84 (1)	0.77 - 0.91	0.73 ± 0.05

Negative values were replaced with zeros.
N = number of years data were collected.

Figure 4.3-11

Average Density of Diptera Taxa for Lakes of the King-Cujo Watershed, 1994 to 2013



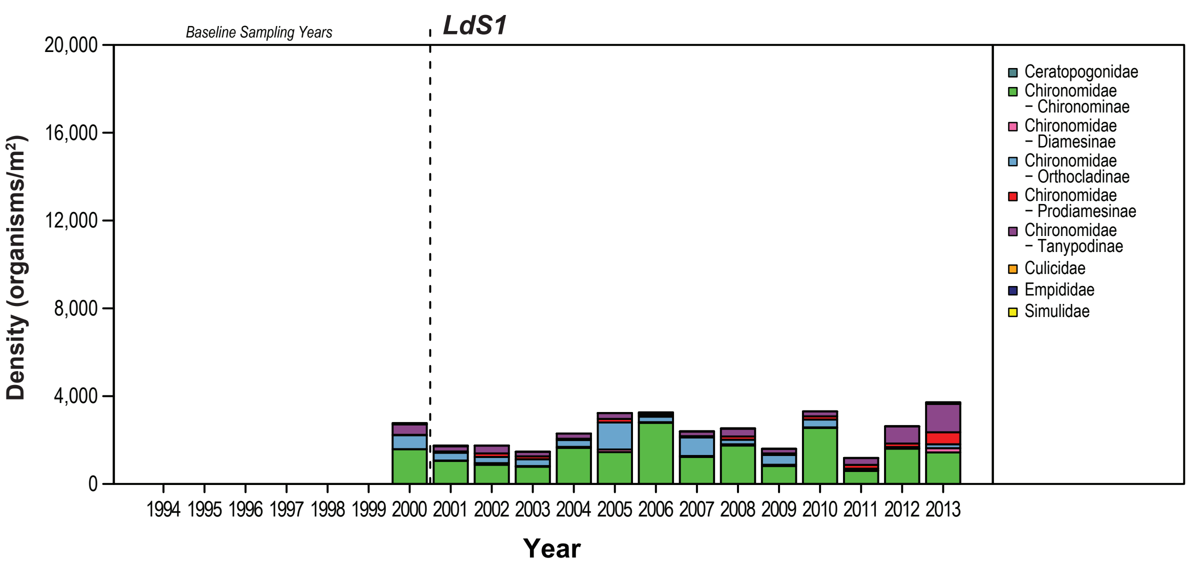
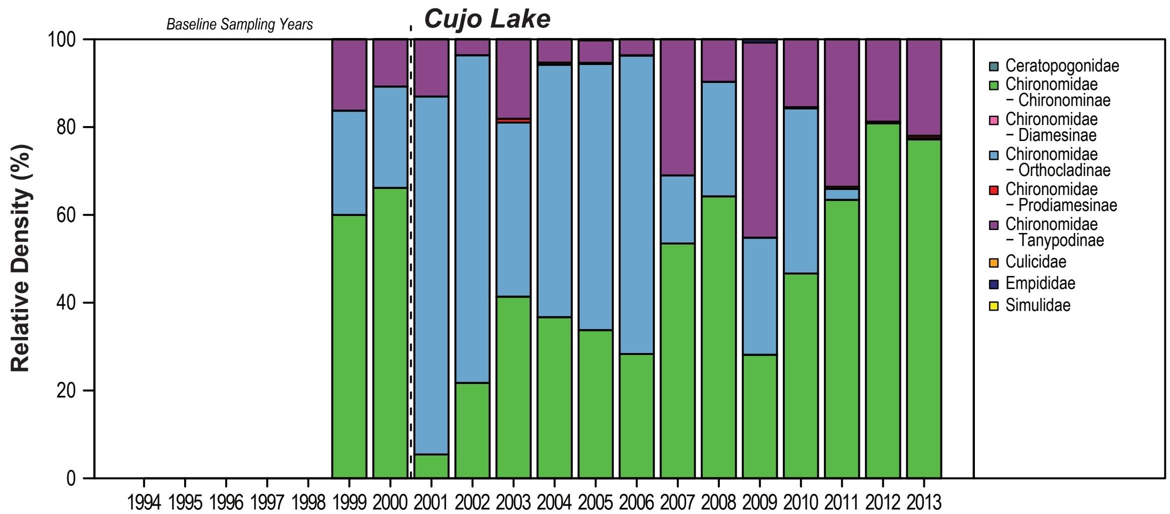
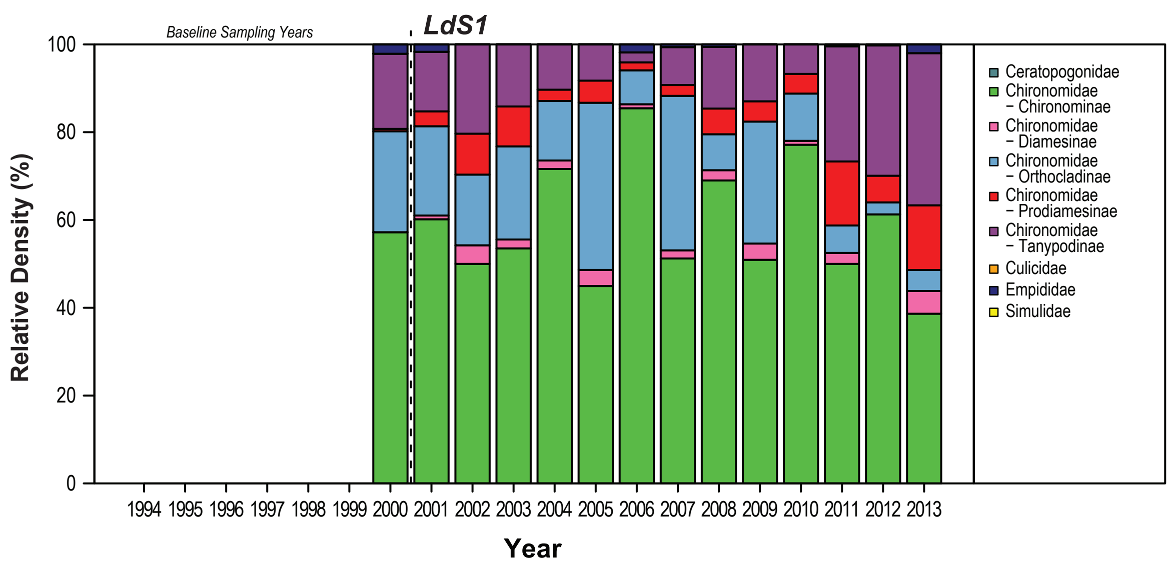


Figure 4.3-12

Relative Density of Diptera Taxa in Lakes of the King-Cujo Watershed, 1994 to 2013





Graphical analyses suggest that the relative densities of dipteran taxonomic groups have changed through time in Cujo Lake (Figures 4.3-11 and 4.3-12). Specifically, the relative densities of Orthocladiinae have decreased, while densities of Chironominae, Tanypodinae and Prodiamesinae have increased (Figures 4.3-11 and 4.3-12). These patterns are consistent with those that were first identified through the multivariate analyses conducted as part of the 2012 AEMP Re-evaluation (Rescan 2012d). In addition, graphical analyses in 2013 suggest that densities of Orthocladiinae at site LdS1 in Lac du Sauvage have decreased through time, with a coincidental increase in densities of Tanypodinae and Prodiamesinae (Figures 4.3-11 and 4.3-12). Although these patterns were generally not observed in reference lakes, graphical analyses in 2013 reveal a more recent trend of decreasing Orthocladiinae densities with coincidental increases in Chironominae densities in Counts Lake (Figures 3.3-19 and 3.3-22).

Taxonomic data was examined at a finer resolution to determine whether densities of specific genera could explain changes in the relative densities of the Chironomidae subfamilies, Orthocladiinae, or Tanypodinae. In general, it was difficult to detect clear temporal trends at the genera level owing to large variability through time and low densities of many genera that frequently result in the absence of particular genera in a given year. Some of the trends that were described in the 2012 AEMP (Rescan 2013) were less apparent in 2013. Despite this variability, examination of the genera data may suggest the following patterns:

- The decrease in Orthocladiinae in Cujo Lake may be related to declines in the density of organisms from the genera *Psectrocladius* and *Zalutschia*, while at site LdS1 in Lac du Sauvage, the decrease in Orthocladiinae appears was most likely related to declines in the density of organisms from the genus *Heterotanytarsus*;
- The increase in Chironominae in Cujo Lake may be due to recent increases in *Cladotanytarsus*, *Corynocera,* and *Stictochironomus*. In Counts Lake, the increase in Chironominae seems more likely to be related to recent increases in *Corynocera* and *Stictochironomus*;
- The increase in Prodiamesinae in Cujo Lake and site LdS1 in Lac du Sauvage appears to be related to increases in the density of organisms from the genus *Monodiamesa*; and
- The increase in Tanypodinae in Cujo Lake and site Lds1 in Lac du Sauvage appears to be related to an overall increase in the density of organisms from the genus *Procladius* over time, as well as a recent increase in organisms of the genus *Ablabesmyia*.

Unfortunately, little information is available on the ecology of benthic invertebrates and, therefore, the cause of these shifts is unclear (Oliver and Dillon 1997). However, results of the 2012 AEMP Re-evaluation suggested that changes in the absolute quantities or relative availability of macronutrients like nitrogen and phosphorus are the most likely underlying cause of change in biological communities at the Ekati Diamond Mine rather than the relative sensitivities of different species to changes in water chemistry (Rescan 2012d).

4.3.4 Stream Benthos

4.3.4.1 Variables

Stream benthos are organisms that live in association with stream sediments. They provide an important source of food for many species of fish. Dipterans (flies) tend to dominate benthic invertebrate communities and are widely used as indicators of ecosystem health, including sediment quality. Organisms from the families Ephemeroptera, Plecoptera, and Trichoptera (EPT) are also widely used as indicators of stream health because they are often sensitive to disturbance and various sources of pollution. Thus, stream benthos density (organisms/m^2) and dipteran and EPT diversity (Shannon and Simpson’s diversity indices) were evaluated for potential mine effects.

4.3.4.2 Dataset

Stream benthos samples have been collected over a one month period from early August to early September of each year since 1995 (Table 4.3-17). Five replicates were collected from each stream in 1995 and between 1999 and 2013. In 1997 and 1998, triplicate samples were collected from each stream. Although stream benthos samples were collected in 2010, they were not analyzed as a result of laboratory error. Baseline data, which were collected between 1994 and 1997, were not used in the statistical evaluation of effects but are included in Table 4.3-17 and are depicted graphically, below, for visual comparison.

Table 4.3-17. Dataset Used for Evaluation of Effects on the Benthos in King-Cujo Watershed Streams

Year	Nanuq Outflow	Counts Outflow	Vulture-Polar	Cujo Outflow
1994	-	-	-	-
1995*	-	-	Aug 10 – Sept 14	-
1996*	-	-	-	-
1997*	Aug 10 – Sept 14	Aug 1 – Sept 7	Aug 10 – Sept 14	-
1998	Jul 30 – Aug 31	Jul 30 – Aug 31	Jul 30 – Aug 31	-
1999	Jul 28 – Aug 28	Jul 28 – Aug 28	Jul 28 – Aug 28	Jul 28 – Aug 28
2000	Jul 28 – Aug 29	Jul 28 – Aug 29	Jul 28 – Aug 29	Jul 28 – Aug 29
2001	Jul 28 – Aug 29	Jul 28 – Aug 29	Jul 28 – Aug 29	Jul 28 – Aug 29
2002	Jul 31 – Aug 31	Jul 31 – Aug 31	Jul 31 – Aug 31	Jul 31 – Aug 31
2003	Aug 1 – Sept 6	Aug 1 – Sept 6	Aug 1 – Sept 6	Aug 1 – Sept 6
2004	Aug 11 – Sept 12	Aug 11 – Sept 12	Aug 11 – Sept 10	Aug 11 – Sept 12
2005	Aug 2 – Sept 3	Aug 2 – Sept 3	Aug 2 – Sept 3	Aug 2 – Sept 5
2006	Jul 26 – Sept 1	Jul 27 – Sept 1	Jul 27 – Sept 4	Jul 27 – Sept 2
2007	Aug 3 – Sept 1	Aug 3 – Aug 31	Aug 4 – Sept 3	Aug 3 – Sept 3
2008	Aug 2 – Sept 4	Aug 1 – Sept 4	Aug 2 – Sept 6	Aug 1 – Sept 4
2009	Aug 3 – Sept 4	Aug 3 – Sept 4	Aug 4 – Sept 4	Aug 3 – Sept 4
2010[†]	-	-	-	-
2011	Jul 30 - Aug 30	Jul 30 - Aug 30	Jul 31 - Aug 31	Jul 31 - Aug 30
2012	Aug 4 – Sept 1	Aug 5 – Aug 31	Aug 4 – Sept 1	Aug 5 – Aug 31
2013	Aug 4 – Sept 3	Aug 4 – Sept 3	Aug 4 – Sept 3	Aug 4 – Sept 3

** = Data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison.*
[†] Data were collected, but were not analyzed as a result of laboratory error.
Dashes indicate no data were available.
Five replicates were collected from each stream in 1995 and from 1999 to 2013.
Triplicate samples were collected in 1997 and 1998.

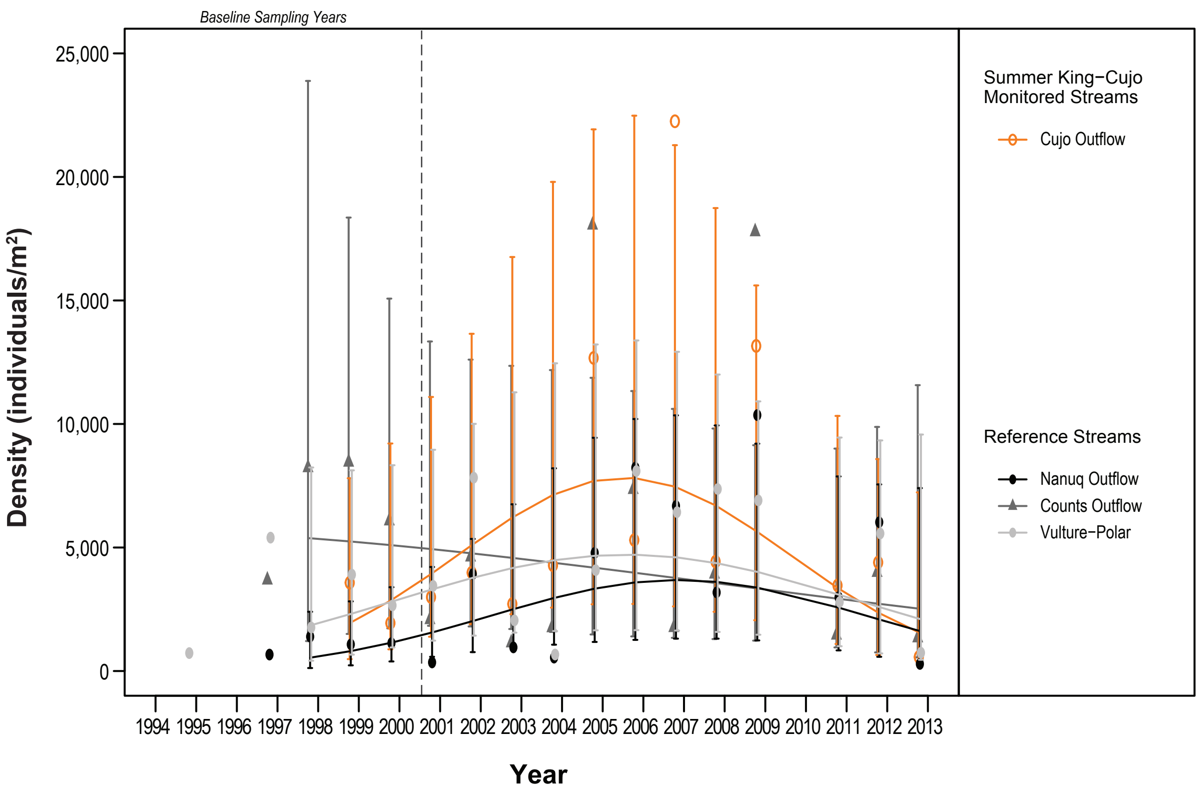
4.3.4.3 Results and Discussion

Density

Statistical and graphical analyses suggest that the density of stream benthos has been stable through time in all monitored and reference streams (Table 4.3-18; Figure 4.3-13). Mean density of stream benthos in Cujo Outflow in 2013 was within the range of ± SD of the baseline mean in Cujo Outflow (Table 4.3-19). Thus it was concluded that no mine effects were detected.

Figure 4.3-13

Observed and Fitted Means for Benthos Densities in King-Cujo Watershed Streams, August 1995 to 2013



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.

Table 4.3-18. Statistical Results of Benthos Density in Streams in the King-Cujo Watershed

	Lakes / Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
				Model Fit = 3	Model Fit = 2	Model Fit = 1	
Density	-	LME	2	-	None	-	2-354

Dashes indicate not applicable.

Table 4.3-19. Mean ± 2 Standard Deviations (SD) Baseline Benthos Density in Each of the King-Cujo Watershed Streams

Stream	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean, ± 1 SD
Nanuq Outflow	667 (1)	166 - 1,168	289 ± 104
Counts Outflow	3,685 (1)	0 – 8,242	1,347 ± 1,619
Vulture-Polar	2,479 (2)	0 - 8,551	740 ± 688
Cujo Outflow	2,758 (1)	360 – 5,155	567 ± 329

Units are organisms/m^2.
Negative values were replaced with zeros.
N = number of years data were collected.

Dipteran Diversity

Statistical analyses were not performed on diversity datasets because the calculation of indices can result in data abnormalities that prevent statistical analysis. Consequently, graphical analyses of temporal trends in diversity indices (Figure 4.3-14) and best professional judgment were the primary methods used in the evaluation of effects. In addition, the average and relative densities of taxa were examined using graphical analyses to identify potential changes in community composition (Figures 4.3-15 to 4.3-16).

Both Shannon and Simpson's stream dipteran diversity indices have varied considerably through time in both monitored and reference streams since monitoring began (Figure 4.3-14). While the variability makes it somewhat difficult to discern temporal trends, both Shannon and Simpson's dipteran diversity in Cujo Outflow have generally been higher since 2011 when compared to previous years (Figure 4.3-14). In particular, diversity in Cujo Outflow declined between 2002 and 2009, but has since returned to baseline values (Figure 4.3-14). Mean stream dipteran diversity in 2013 was within the range ± 2 SD of baseline means in all streams except Counts Outflow, in which Shannon diversity was greater (Table 4.3-20).

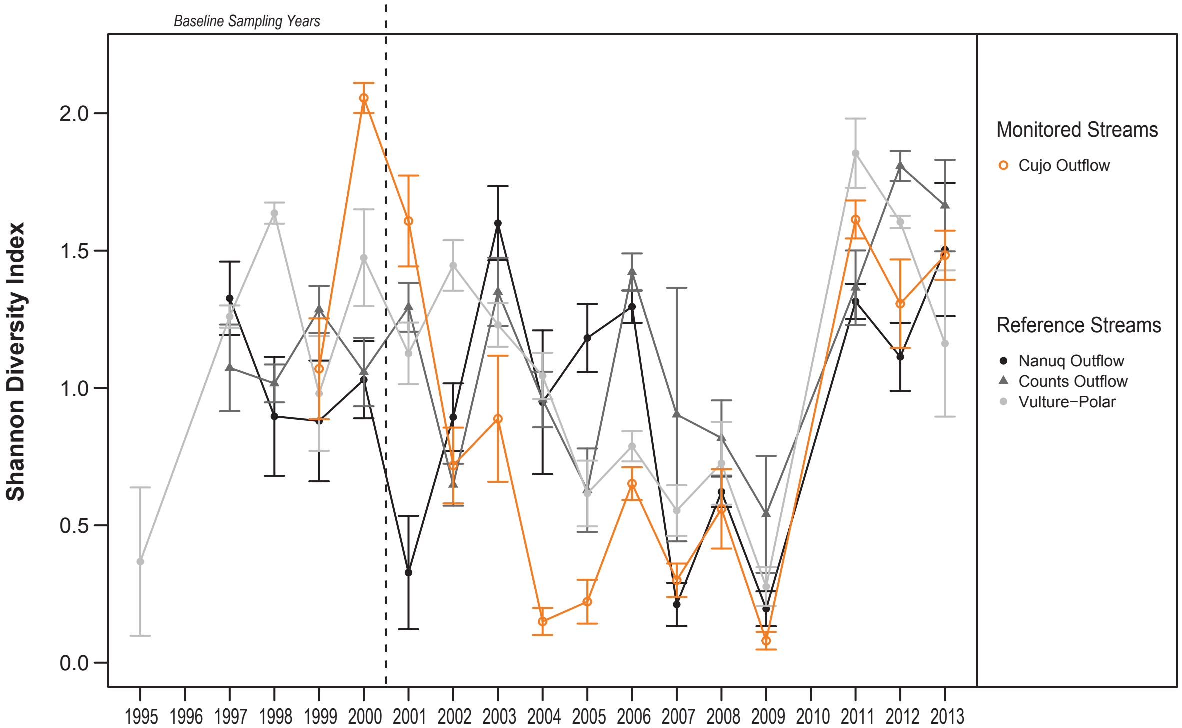
Table 4.3-20. Mean ± 2 Standard Deviations (SD) Baseline Dipteran Diversity in Each of the King-Cujo Watershed Streams

	Shannon Diversity			Simpson's Diversity		
Stream	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD
Nanuq Outflow	1.33 (1)	0.86 - 1.79	1.50 ± 0.54	0.69 (1)	0.59 – 0.79	0.72 ± 0.15
Counts Outflow	1.07 (1)	0.53 - 1.62	1.66 ± 0.37	0.54 (1)	0.24 – 0.84	0.72 ± 0.16
Vulture-Polar	0.70 (2)	0 – 2.00	1.16 ± 0.60	0.37 (2)	0 - 1	0.55 ± 0.24
Cujo Outflow	1.56 (1)	0.38 – 2.75	1.48 ± 0.16	0.69 (1)	0.26 – 1	0.75 ± 0.04

Negative values were replaced with zeros.
For Simpson's diversity, upper confidence intervals >1 were replaced with a value of 1 (i.e., the maximum possible value for Simpson's diversity).
N = number of years data were collected.

Figure 4.3-14

Average Diversity Indices for Benthic Dipterans in King-Cujo Watershed Streams, August 1995 to 2013



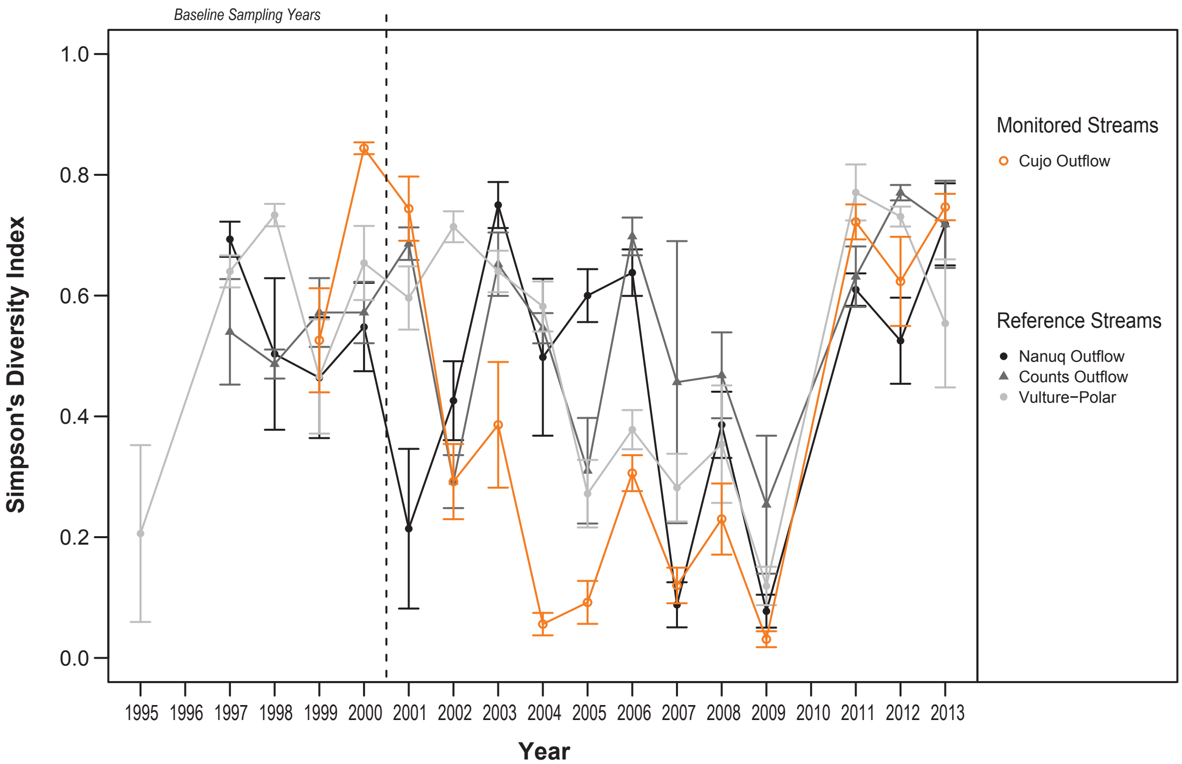


Figure 4.3-15

Average Benthic Dipteran Density by Taxonomic Group in Streams of the King-Cujo Watershed, 1995 to 2013

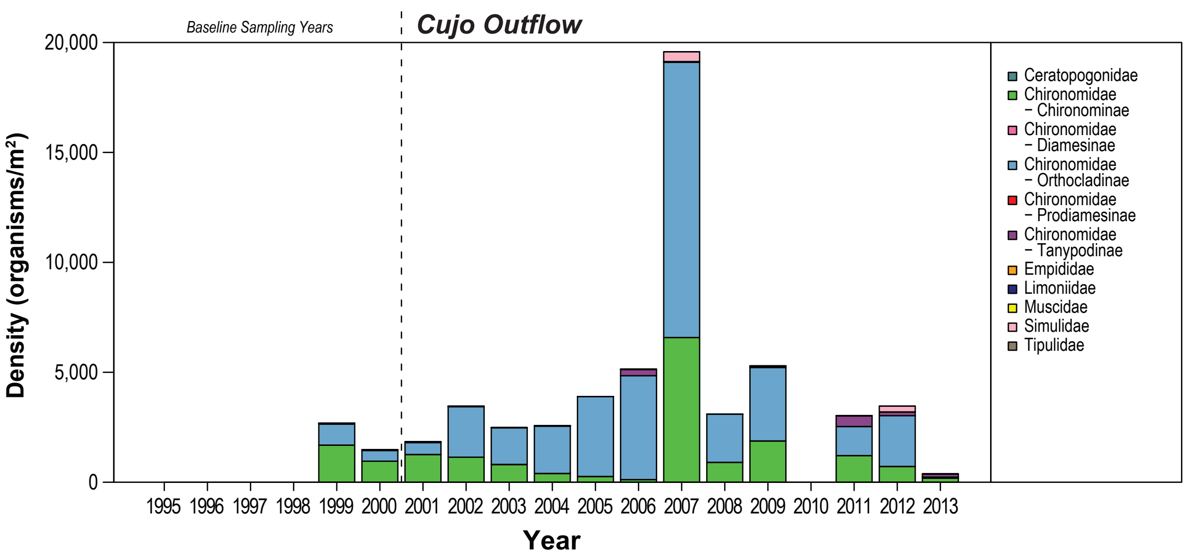
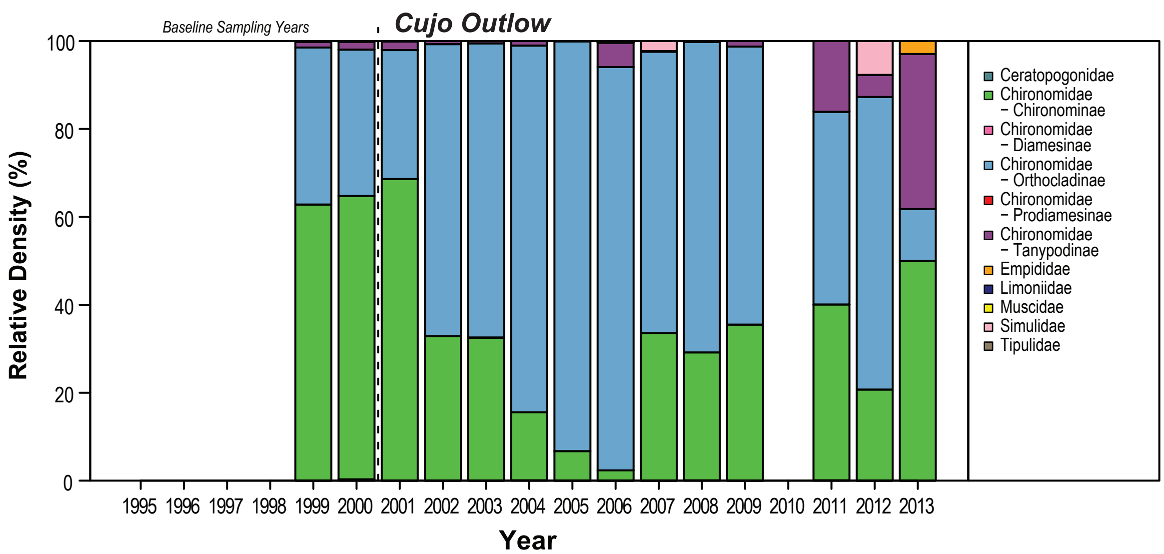


Figure 4.3-16

Relative Densities of Benthic Dipteran Taxa in Streams of the King-Cujo Watershed, 1995 to 2013



The relative densities of dipteran taxonomic groups have been consistent through time in all monitored and reference streams (Figures 4.3-15 to 4.3-16). However, as in the Koala Watershed, there was some evidence of a trend toward relatively greater densities of organisms from the sub-family Orthocladiinae and lower densities of organisms from the sub-family Chironominae through time in all monitored and reference streams (Figures 3.3-27, 3.3-30, 4.3-15, and 4.3-16). Observations in 2013 indicate that this trend may be reversing in both Counts and Cujo outflows. There is also some evidence that densities of organisms from the sub-family Tanypodinae may be increasing in Counts and Cujo outflows over the last three years (Figures 3.3-27, 3.3-30, 4.3-15, and 4.3-16). Overall, observed trends were similar in both reference and monitored streams, which suggests that any changes in stream benthos community composition may result from broader climatic patterns or systematic changes in identification or enumeration through time. Of the two subfamilies of Chironomidae that tend to dominate benthic community composition in the King-Cujo Watershed, the subfamily Chironominae is a particularly diverse and abundant group (Thorp and Covich 2001), while Orthocladiinae are adapted to cold water environments (Kravtsova 2000). Thus no mine effects were detected with respect to stream dipteran diversity or taxonomic composition.

<u>EPT Diversity</u>

Statistical analyses were not performed on the diversity datasets because the calculation of indices can result in data abnormalities that prevent statistical analysis. Consequently, graphical analyses of temporal trends in diversity indices (Figure 4.3-17) and best professional judgment were the primary methods used in the evaluation of effects. In addition, the average and relative densities of taxa were examined using graphical analyses to identify potential changes in community composition (Figures 4.3-18 and 4.3-19).

Both Shannon and Simpson's EPT diversity indices have varied considerably through time in both monitored and reference streams since monitoring began (Figure 4.3-17). While the variability makes it somewhat difficult to discern temporal trends, both Shannon and Simpson's EPT diversity have remained within the range of historical values observed in Cujo Outflow and all reference streams through time (Figure 4.3-17). Mean EPT diversity in 2013 was within the range of two standard deviations of baseline years in all monitored streams (Table 4.3-21). Both Shannon and Simpson's EPT diversity in Counts Outflow were equal to zero in 2013 as only one EPT taxa (*Nemoura* sp.) was present in samples (Table 4.3-21). Values of zero for stream EPT diversity are common in the historical record at the Ekati Diamond Mine and have been observed on several occasions in both monitored and reference streams. Relative densities of EPT taxa have been variable through time in all monitored and reference streams and show no signs of directed change in the monitored stream (Figures 3.3-34, 3.3-37, 4.3-18, and 4.3-19). Thus no mine effects were detected with respect to EPT diversity or taxonomic composition.

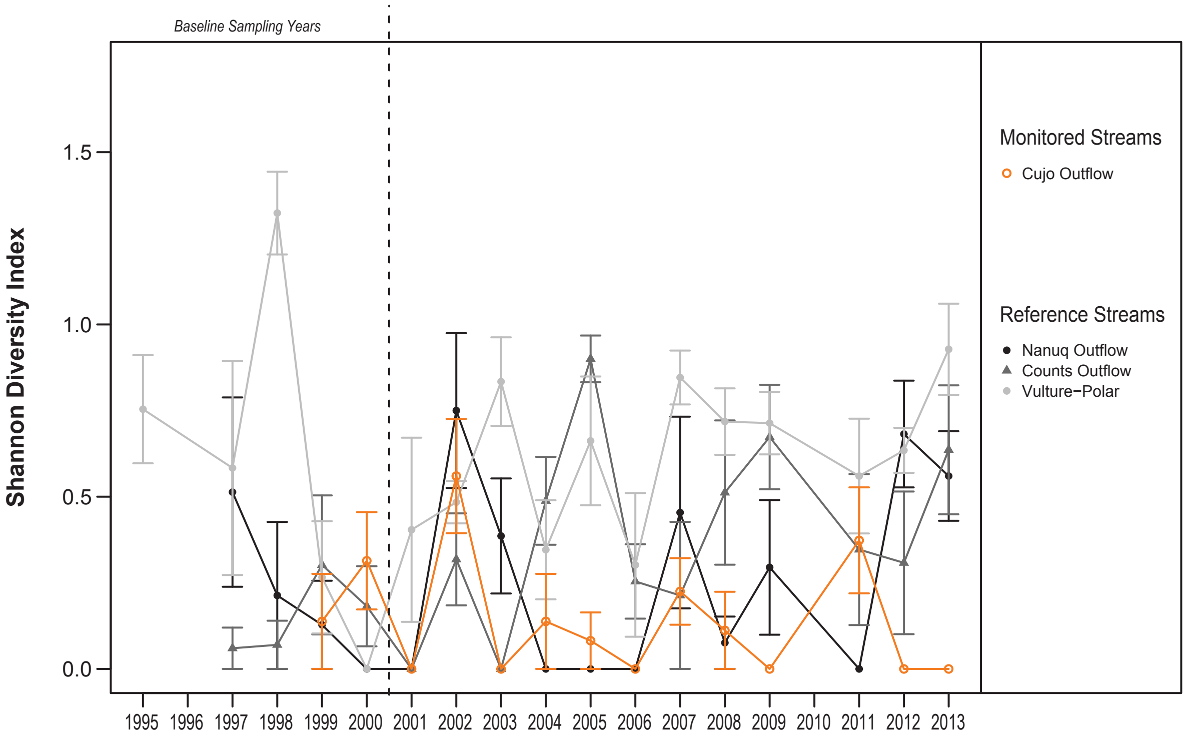
Table 4.3-21. Mean ± 2 Standard Deviations (SD) Baseline EPT Diversity in Each of the King-Cujo Watershed Streams

	Shannon Diversity			Simpson's Diversity		
Stream	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2013 Mean ± 1 SD
Nanuq Outflow	0.51 (1)	0 - 1.47	0.56 ± 0.29	0.29 (1)	0 - 0.86	0.33 ± 0.19
Counts Outflow	0.06 (1)	0 - 0.27	0.64 ± 0.42	0.03 (1)	0 - 0.12	0.40 ± 0.25
Vulture-Polar	0.69 (2)	0 - 1.49	0.93 ± 0.30	0.41 (2)	0 - 0.85	0.57 ± 0.12
Cujo Outflow	0.23 (1)	0 - 0.84	0	0.16 (1)	0 - 0.59	0

Negative values were replaced with zeros.
N = number of years data were collected.

Figure 4.3-17

Average Diversity Indices for Benthic EPT Taxa in King-Cujo Watershed Streams, August 1995 to 2013



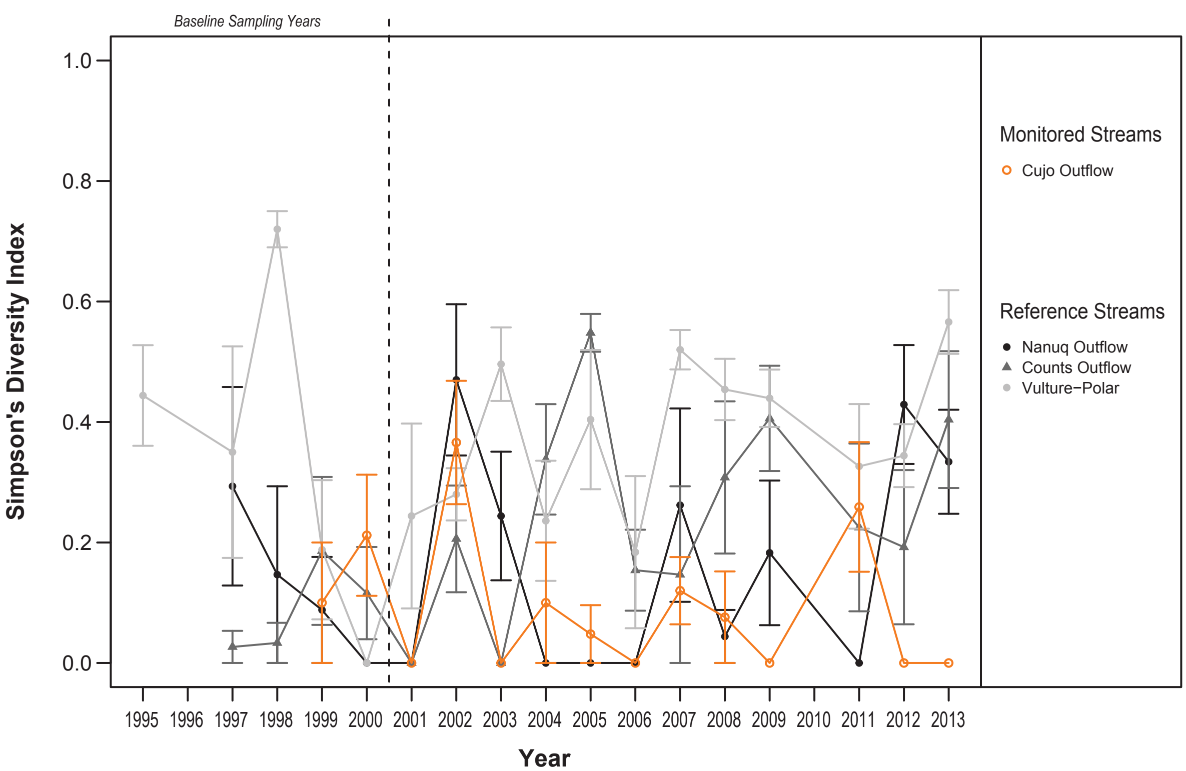


Figure 4.3-18

Average Benthic EPT Density by Taxonomic Group in Streams of the King-Cujo Watershed, 1995 to 2013

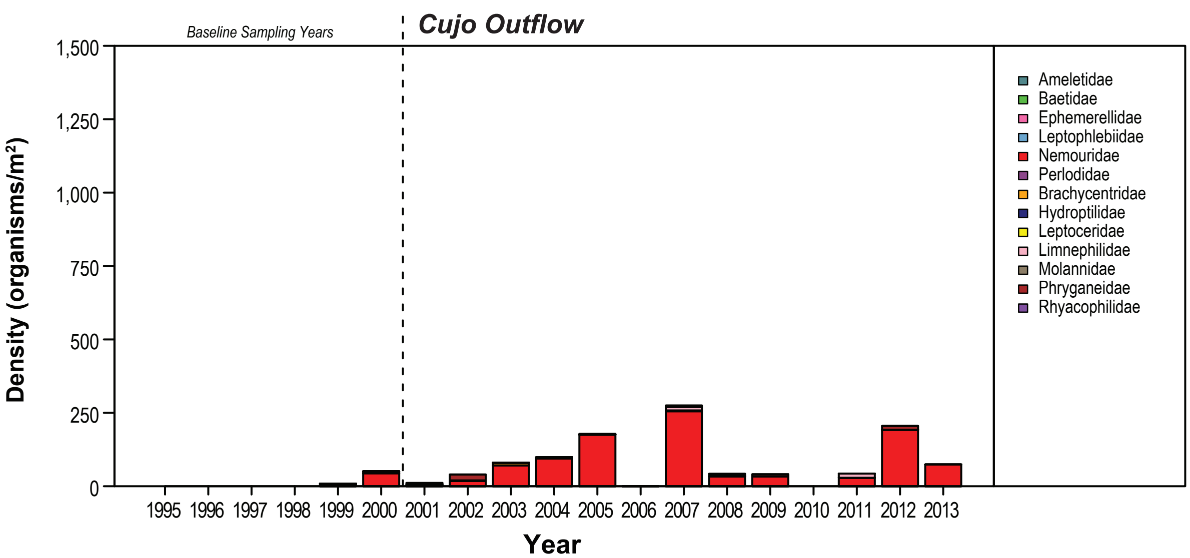
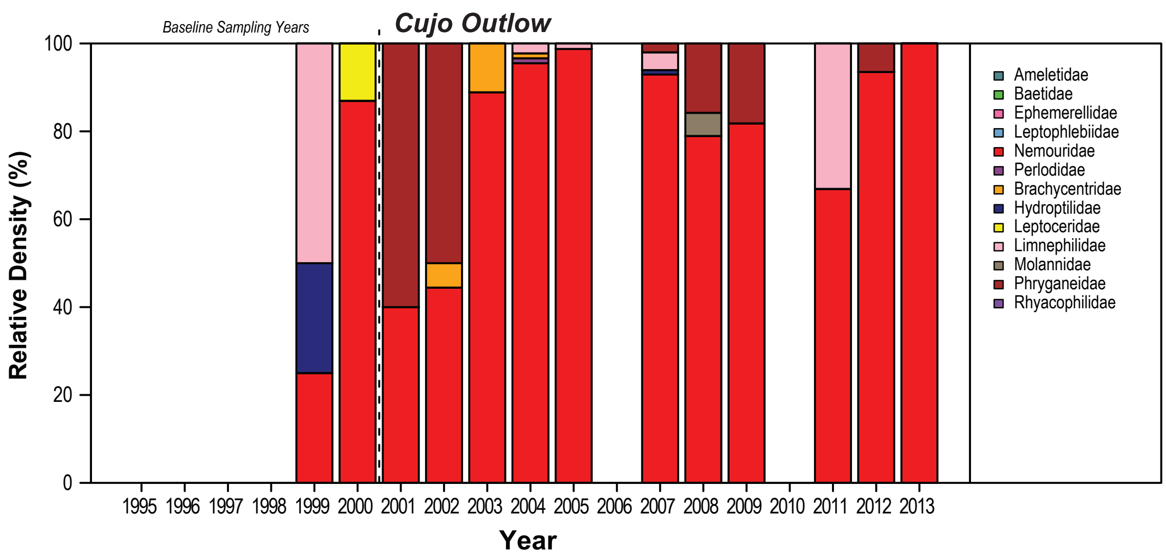


Figure 4.3-19

Relative Densities of Benthic EPT Taxa in Streams of the King-Cujo Watershed, 1995 to 2013



4.3.5 Aquatic Biology Summary

Only one change in biological variables was observed in 2013:

- change in lake benthos dipteran community composition in Cujo Lake and site LdS1.

No mine effects were detected with respect to phytoplankton biomass, density, diversity, or community composition in the King-Cujo Watershed or Lac du Sauvage.

Zooplankton biomass, density, diversity, and overall community composition have remained relatively stable through time in Cujo Lake and site LdS1 in Lac du Sauvage. However, although no mine effects were detected with respect to zooplankton diversity or community composition, a close examination of zooplankton species compositions suggests that the rotifer *Conochilus sp*. and the cladoceran *Holopedium gibberum*, have been largely absent from Cujo Lake since 2002. A similar trend was observed in lakes downstream of the LLCF. *Conochilus sp.* returned to Cujo Lake in 2011, but was once again absent from Cujo Lake in 2012 and 2013. The reason for the change in composition of cladoceran genera remains unclear.

Lake benthos density has been stable through time in all monitored and reference lakes of the King-Cujo Watershed and Lac du Sauvage. Although dipteran diversity has been variable through time, diversity has been relatively stable in monitored and reference lakes since 2007. Shifts in the benthos community composition have been observed in Cujo Lake and at site LdS1 in Lac du Sauvage, in which the relative densities of organisms from the Chironomidae sub-family Orthocladiinae (most likely organisms from the genera *Psectrocladius* and *Zalutschia* in Cujo Lake and from the genus *Heterotanytarsus* at site LdS1) have decreased through time while densities of organisms from the subfamilies Tanypodinae (most likely organisms from the genera *Procladius* and *Ablabesmyia*) and Prodiamesinae (most likely organisms from the genus *Monodiamesa*) have increased through time. Organisms from the subfamily Chironominae (likely organisms from the genera *Cladotanytarsus*, *Corynocera* and *Stictochironomus*) have also increased through time in Cujo Lake. Most of these changes began in Cujo Lake in 2005 and were first identified through the multivariate analyses conducted as part of the 2012 AEMP re-evaluation (Rescan 2012d). The shift in taxonomic composition was more recently observed at site LdS1 in 2013. Unfortunately, little information is available on the ecology of these benthic invertebrates and the cause of these shifts is unclear (Oliver and Dillon 1997). However, these shifts are similar to those that have occurred in Leslie and Moose lakes in the Koala Watershed and concentrations of all the evaluated water quality variables in the King-Cujo Watershed have remained below the lowest identified chronic effect level for the most sensitive species. Thus, the observed changes in lake benthos community composition are likely driven, ultimately, by changes in the availability of macronutrients including nitrogen and phosphorus in lakes downstream of the KPSF.

No mine effects were detected with respect to stream benthos density, dipteran diversity or EPT diversity, or dipteran or EPT community composition in the King-Cujo Watershed.

Lake benthos provide an important source of food for many species of fish. Changes in community composition could have important consequences for fish, especially if preferred prey items are replaced with non-preferred ones. Similar to the Koala Watershed, results of the 2012 AEMP Evaluation of Effects found no evidence of major mine effects on monitored fish populations in the King-Cujo Watershed (Rescan 2012d). Thus, shifts in lake benthos communities do not appear to have influenced fish populations to date. Both round whitefish and lake trout are considered opportunistic feeders where in the absence of strong prey community-wide effects, may not exhibit strong biological changes, including any bioenergetics-related response variables. Furthermore, the mobile nature of these larger-bodied fish populations may also serve to reduce any potential effects. Lakes in the Ekati Diamond Mine study area are not isolated and individual fish are able to move freely between upstream and downstream lakes. This likely serves to buffer any potential effects or may delay the appearance of mine effects.

4.4 SUMMARY

Table 4.4-1 summarizes the evaluation of effects for the King-Cujo Watershed and Lac du Sauvage. Conclusions regarding the direction of change were drawn from graphical analysis because statistical tests were two-sided and tested only for differences between reference and monitored lakes rather than the direction of change.

No mine effects were detected with respect to physical limnology variables (i.e., temperature, dissolved oxygen, and Secchi depths) in monitored lakes during either the ice-covered or open water season in 2013 (Table 4.4-1). Under-ice DO concentrations were greater than the CCME guideline value of 6.5 mg/L throughout the majority of the water column in most monitored sites in the King-Cujo Watershed and Lac du Sauvage(CCME 2013). In Cujo Lake, DO measurements were less than CCME guidelines throughout most of the water column by early April 2013, through to the last sampling date during the ice-covered season, though some improvement occurred following snow clearance in April. This pattern reflects historical DO profiles in Cujo Lake. Data from reference lakes suggests that deeper sections of sub-Arctic lakes are generally less than the CCME threshold during the ice-covered period (Figures 3.1-1a-b). Although, the low under-ice dissolved oxygen concentrations in Cujo Lake may be related to elevated TOC concentrations in Cujo Lake, dissolved oxygen and TOC concentrations were not measured during baseline years, making it difficult to discern whether the correlation results from mine operations or represents undisturbed conditions in the King-Cujo Watershed.

A total of 23 water quality variables were evaluated for lakes and streams in the King-Cujo Watershed and Lac du Sauvage in the 2013 AEMP. Of these, concentrations of 13 variables have changed through time in monitored sites downstream of the KPSF (Table 4.4-1). In two cases (total copper, and total ammonia-N), concentrations have returned to baseline concentrations in recent years, with no mine effects detected since 2012 (Table 4.4-1; Rescan 2013b). Concentrations remain elevated above baseline or reference concentrations in ten cases (Table 4.4-1). In one case, TOC, concentrations have been consistently elevated in comparison to reference lakes and streams downstream as far as Christine-Lac du Sauvage, but have not increased over time.

TOC is a measure of the amount of live and decomposing organic matter in the water column. Increases in TOC may reflect increases in available nutrients, which stimulate the growth and reproduction of aquatic organisms. In oligotrophic (i.e., nutrient poor) systems, like those found in the sub-Arctic, changes in nutrient levels may not be detected because of the speed with which available nutrients are incorporated into biotic material. Thus, increases in TOC may indicate an increase in the overall productivity of a system, which may also be reflected in changes in the biomass of primary producers (e.g., phytoplankton, periphyton), primary consumers (e.g., zooplankton, benthic invertebrates), or secondary consumers (e.g., fish), depending on how far up the food web the changes have progressed. However, neither the biomass nor the density of phytoplankton, zooplankton, or benthos has changed through time in the King-Cujo Watershed or Lac du Sauvage (see Sections 4.3). Increases in TOC may also be associated with reductions in dissolved oxygen because bacteria consume oxygen as they decompose organic matter. Under-ice dissolved oxygen concentrations in Cujo Lake have historically been less than the CCME guidelines throughout the majority of the water column (Figure 4.1-1). Thus, the observed elevated TOC in Cujo Lake, relative to reference lakes and Lac du Sauvage, could be related to the observed low dissolved oxygen concentrations. However, dissolved oxygen and TOC concentrations were not measured during baseline years, making it difficult to discern whether the correlation results from mine operations or represents undisturbed conditions in the King-Cujo Watershed.

Table 4.4-1. Summary of Evaluation of Effects for the King-Cujo Watershed and Lac du Sauvage

Variable	Change Dowstream of the KPSF?	Locations Changes Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Physical Limnology						
Under-ice Temperature Profiles	No	-	-	-	-	-
Under-ice DO Profiles	No	-	-	-	-	-
August Secchi Depths	No	-	-	-	-	-
Lake and Stream Water Quality						
pH	Yes	Downstream to Christine-Lac du Sauvage Stream	Increase	KPSF	Yes	Lower 95% CI on fitted 2013 means at site LdS1 during the ice-covered and open water season and at site LdS2 during the open water season was below the CCME guidelines; similar patterns observed in all reference lakes and streams.
Total Alkalinity	Yes	Downstream to Christine-Lac du Sauvage Stream	Increase	KPSF	Yes	-
Hardness	Yes	Downstream to Christine-Lac du Sauvage Stream	Increase	KPSF	Yes	-
Chloride	Yes	Downstream to Cujo Outflow	Increase	KPSF	Yes	All 2013 concentrations less than the SSWQO.
Sulphate	Yes	Downstream to Christine-Lac du Sauvage Stream	Increase	KPSF	Yes	All 2013 concentrations less than the SSWQO.
Potassium	Yes	Downstream to Christine-Lac du Sauvage Stream	Increase	KPSF	Yes	All 2013 concentrations less than the SSWQO.
Total Ammonia-N	Yes	Downstream to Cujo Lake	Increase	KPSF	Yes	Concentrations in Cujo Lake have returned to baseline and reference concentrations in recent years. All 2013 concentrations less than the CCME guideline.
Nitrite-N	No	-	-	-	-	All 2013 concentrations less than the CCME guideline.
Nitrate-N	No	-	-	-	-	All 2013 concentrations less than the SSWQO.
Total Phosphate-P	No	-	-	-	No	Upper 95% CI around the fitted means exceeded benchmark values in Cujo Lake and sites LdS1 and LdS2; similar patterns observed in reference lakes.
Total Organic Carbon	Yes	Downstream to Christine-Lac du Sauvage Stream	Stable at elevated concentrations since monitoring began	KPSF	Yes	Baseline concentrations not sampled, but downstream spatial gradient present.

(continued)

Table 4.4-1. Summary of Evaluation of Effects for the King-Cujo Watershed and Lac du Sauvage (continued)

Variable	Change Dowstream of the KPSF?	Locations Changes Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Lake and Stream Water Quality ***(cont'd)***						
Total Antimony	No	-	-	-	No	Upper 95% CI around the fitted mean exceeded the benchmark value in Christine-Lac du Sauvage Stream and the reference Vulture-Polar Stream
Total Arsenic	No	-	-	-	No	All 2013 concentrations less than the CCME guideline.
Total Barium	Yes	Downstream to Christine-Lac du Sauvage Stream	Increase	KPSF	Yes	All 2013 concentrations less than the water quality benchmark.
Total Boron	Yes	Downstream to Christine-Lac du Sauvage Stream	Increase	KPSF	Yes	All 2013 concentrations less than the CCME guideline.
Total Cadmium	No	-	-	-	No	All 2013 concentrations less than the CCME guideline.
Total Copper	Yes	Downstream to Cujo Outflow	Increase	KPSF	Historical	All 2013 concentrations less than the CCME guideline.
Total Molybdenum	Yes	Downstream to Christine-Lac du Sauvage Stream	Increase	KPSF	Yes	All 2013 concentrations less than the SSWQO.
Total Nickel	No	-	-	-	No	All 2013 concentrations less than the CCME guideline.
Total Selenium	No	-	-	-	No	All 2013 concentrations less than the CCME guideline.
Total Strontium	Yes	Downstream to Christine-Lac du Sauvage Stream	Increase	KPSF	Yes	All 2013 concentrations less than the strontium water quality benchmark-
Total Uranium	No	-	-	-	-	All 2013 concentrations less than the CCME guideline.
Total Vanadium	No	-	-	-	-	All 2013 concentrations less than the SSWQO.
Phytoplankton						
Chlorophyll *a*	No	-	-		No	-
Density	No	-	-	-	No	-
Diversity	No	-	-	-	No	-
Relative Densities of Major Taxa	No	-	-	-	No	-
Zooplankton						
Biomass	No	-	-	-	No	-
Density	No	-	-	-	No	-

(continued)

Table 4.4-1. Summary of Evaluation of Effects for the King-Cujo Watershed and Lac du Sauvage (completed)

Variable	Change Dowstream of the KPSF?	Locations Changes Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Zooplankton *(cont'd)*						
Diversity	No	-	-	-	No	-
Relative Densities of Major Taxa	No	-	-	-	No	The rotifer *Conochilus sp*. and the *Holopedium gibberum* has been largely absent from Cujo Lake since 2002.
Lake Benthos						
Density	No	-	-	-	No	-
Dipteran Diversity	No	-	-	-	No	-
Dipteran Relative Density	Yes	Cujo Lake and site LdS1 in Lac du Sauvage	Decrease in Orthocladinae; Increase in Chironominae, Prodiamesinae, Tanypodinae in Cujo Lake; Increase in Tanypodinae, Prodiamesinae at LdS1	-	Yes	Changes in community composition may be related to decreases in some genera (*Psectrocladius*, *Zalutschia*, *Heterotanytarsus*) and increases in others (*Monodiamesa*, *Cladotanytarsus*, *Corynocera*, *Stictochironomus*, *Procladius*, *Ablabesmyia*). A similar pattern of decreased Orthocladiinae with increasing Chironominae (*Corynocera*, *Stictochironomus*) observed in recent years in one reference lake (Counts Lake).
Stream Benthos						
Density	No	-	-	-	No	-
Dipteran Diversity	No	-	-	-	No	-
Dipteran Relative Density	No	-	-	-	No	Some changes in taxonomic composition related to broader climatic patterns or systematic changes in in enumeration/identification observed.
EPT Diversity	No	-	-	-	No	-
EPT Relative Density	No	-	-	-	No	-

Dashes indicate not applicable.
Comparisons to CCME guidelines are for 2013 data only.
DO = dissolved oxygen
CCME = Canadian Council of Ministers of the Environment
SSWQO = Site-specific Water Quality Objective

Overall, the extent to which concentrations of water quality variables have changed through time generally decreases with downstream distance from the KPSF. Patterns were similar during the ice-covered and open water seasons, though concentrations were sometimes elevated during the ice-covered season, relative to the open water season, as a consequence of solute exclusion during freeze up. In reference lakes, concentrations have generally been low and stable through time. Together, the evidence suggests that the observed changes in concentrations in the variables listed in Table 4.4-1 are mine effects that stem from the discharge of water from the KPSF into the receiving environment under Water Licence W2012L2-0001.

CCME guidelines for the protection of aquatic life exist for ten of the evaluated water quality variables, including pH, total ammonia-N, nitrite-N, total arsenic, total boron, total cadmium, total copper, total nickel, total selenium, and total uranium (CCME 2013). In addition, DDEC has established SSWQO for six of the evaluated variables, including chloride, sulphate, potassium, nitrate-N, total molybdenum, and total vanadium (see Table 2.3-1). Total phosphate concentrations were compared to lake-specific benchmark trigger values that were established using guidelines set out in the Canadian Guidance Framework for the Management of Phosphorus in Freshwater Systems, the Ontario Ministry of Natural Resources, and Environment Canada (Ontario Ministry of Natural Resources 1994; CCME 2004; Environment Canada 2004). Other water quality benchmarks also exist for total antimony, total barium, and total strontium (see Table 2.3-1). With the exception of pH, the 95% confidence intervals around the fitted mean and the observed mean concentrations were below their respective CCME guideline value, SSWQO, or relevant benchmark value. The lower 95% confidence interval on the fitted mean pH at site LdS1 was less than the CCME guideline; however, similar patterns were observed in all reference lakes and streams, suggesting that it is not related to mine activities.

Despite increases in 13 evaluated water quality variables downstream of the KPSF, observedconcentrationswere generally below water quality benchmark values. This suggests that concentrations of water quality variables remain less than the concentrations at which toxic effects might be expected. Thus, observed changes in biological community composition at the Ekati Diamond Mine likely result from inter-specific differences in the competitive ability of different taxonomic groups under changing quantities or ratios of macronutrients, rather than elemental toxicity (Rescan 2012d). A shift in the community composition of lake benthos species was found in Cujo Lake and at site LdS1 in Lac du Sauvage. The underlying cause of this shift was attributed to changes in the relative availability of macronutrients (see Section 4.3.1.3 and 4.3.5).A shift in lake benthos community composition was observed in Cujo Lake and at site LdS1 in Lac du Sauvage. The underlying cause of this shift was attributed to changes in the relative availability of macronutrients (see Section 4.3.1.3 and 4.3.5). Although changes in relative densities of lake benthos species could have important cascading effects for higher trophic levels, no evidence to date suggests that monitored fish populations at the Ekati Diamond Mine have been influenced by changes in the relative abundance of prey species (see Section 4.3-5; Rescan 2012d).

5. Historical Lake Water Quality and Stream Hydrology

5. Historical Lake Water Quality and Stream Hydrology

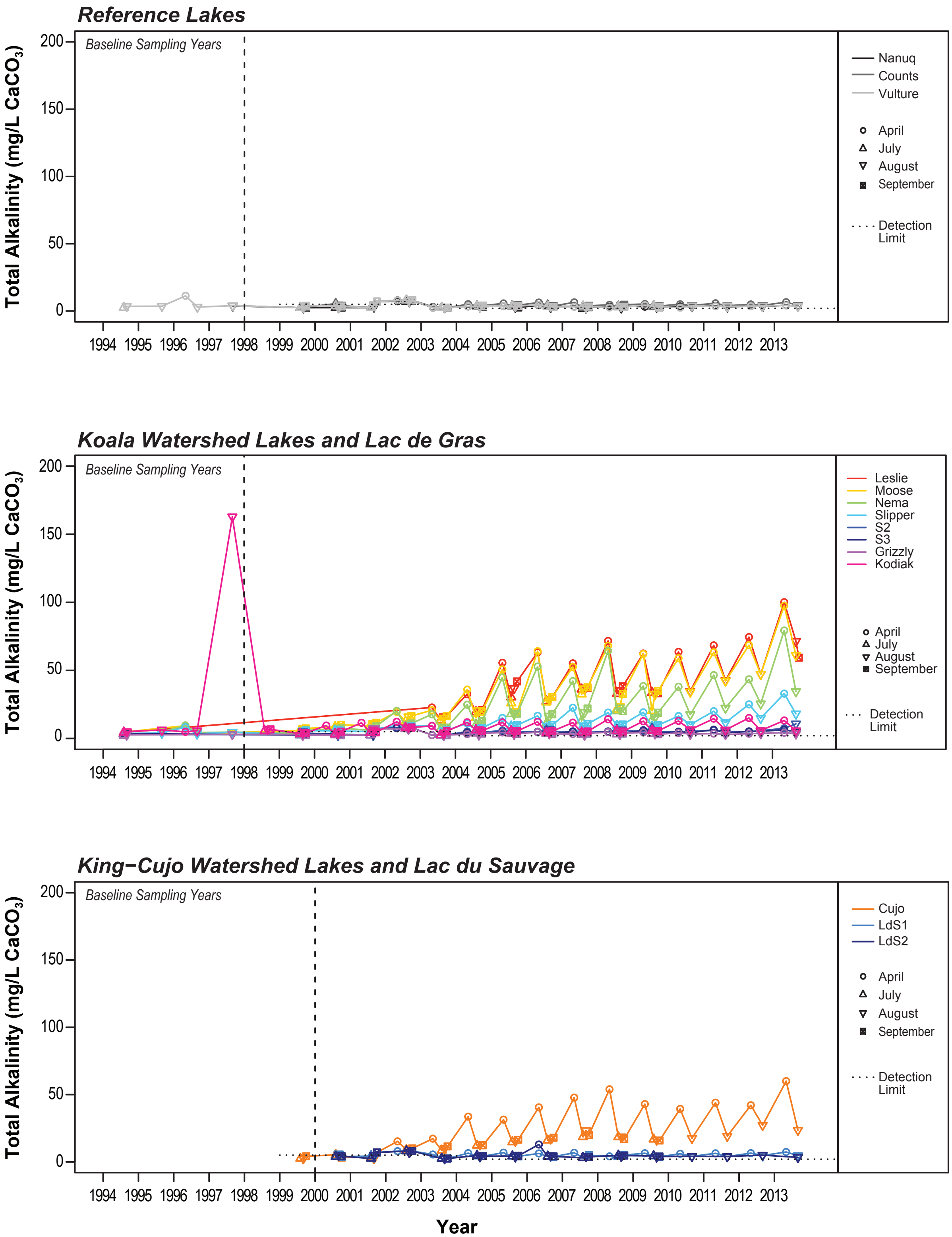
The AEMP evaluation of effects focuses on detecting changes in 22 lake water quality variables in the Koala Watershed and Lac de Gras and 23 lake water quality variables in the King-Cujo Watershed and Lac du Suavage, using samples collected in August of each year (see Sections 2.2, 3.2, 4.2). However, lake water quality samples are collected and screened for 47 water quality variables in the laboratory (Table 5-1). In addition, prior to 2010 lake water quality was sampled in July and September in addition to the April and August sampling. Historical averages for each of the 46 water quality variables for the three reference lakes (Nanuq, Counts, and Vulture) and each of the lakes that is monitored for water quality in the Koala and King-Cujo Watersheds are presented below (Figures 5-1 to 5-46). CCME water quality guidelines for the protection of aquatic life are provided where applicable (CCME 2013). More recently, DDEC has established SSWQOs for six variables, including chloride, potassium, sulphate, nitrate, molybdenum, and vanadium (see Section 2.3). In addition, water quality benchmarks for antimony, barium, manganese and strontium were adopted in 2012, and for vanadium in 2013 (see Section 2.3). Analytical detection limits are also included, with the lowest detection limit presented in cases where detection limits varied between lakes and months within the same year.

Table 5-1. AEMP Water Quality Variables

Variables	Figure Number	Variables	Figure Number
Physical/Ion		**Total Metals**	
Total Alkalinity	5-1	Aluminum	5-22
Bicarbonate	5-2	Antimony	5-23
Carbonate	5-3	Arsenic	5-24
Conductivity	5-4	Barium	5-25
Hydroxide	5-5	Beryllium	5-26
pH	5-6	Boron	5-27
Chloride	5-7	Cadmium	5-28
Potassium	5-8	Calcium	5-29
Total Silicon	5-9	Chromium	5-30
Sulphate	5-10	Cobalt	5-31
Total Suspended Solids	5-11	Copper	5-32
Turbidity	5-12	Iron	5-33
Hardness	5-13	Lead	5-34
Ion Balance	Not Shown	Magnesium	5-35
Total Dissolved Solids	5-14	Manganese	5-36
		Mercury	5-37
Nutrients/Organics		Molybdenum	5-38
Total Ammonia-N	5-15	Nickel	5-39
Nitrate-N	5-16	Selenium	5-40
Nitrite-N	5-17	Silver	5-41
Orthophosphate	5-18	Sodium	5-42
Total Phosphate-P	5-19	Strontium	5-43
Total Organic Carbon	5-20	Uranium	5-44
Total Kjeldahl Nitrogen	5-21	Vanadium	5-45
		Zinc	5-46

Figure 5-1

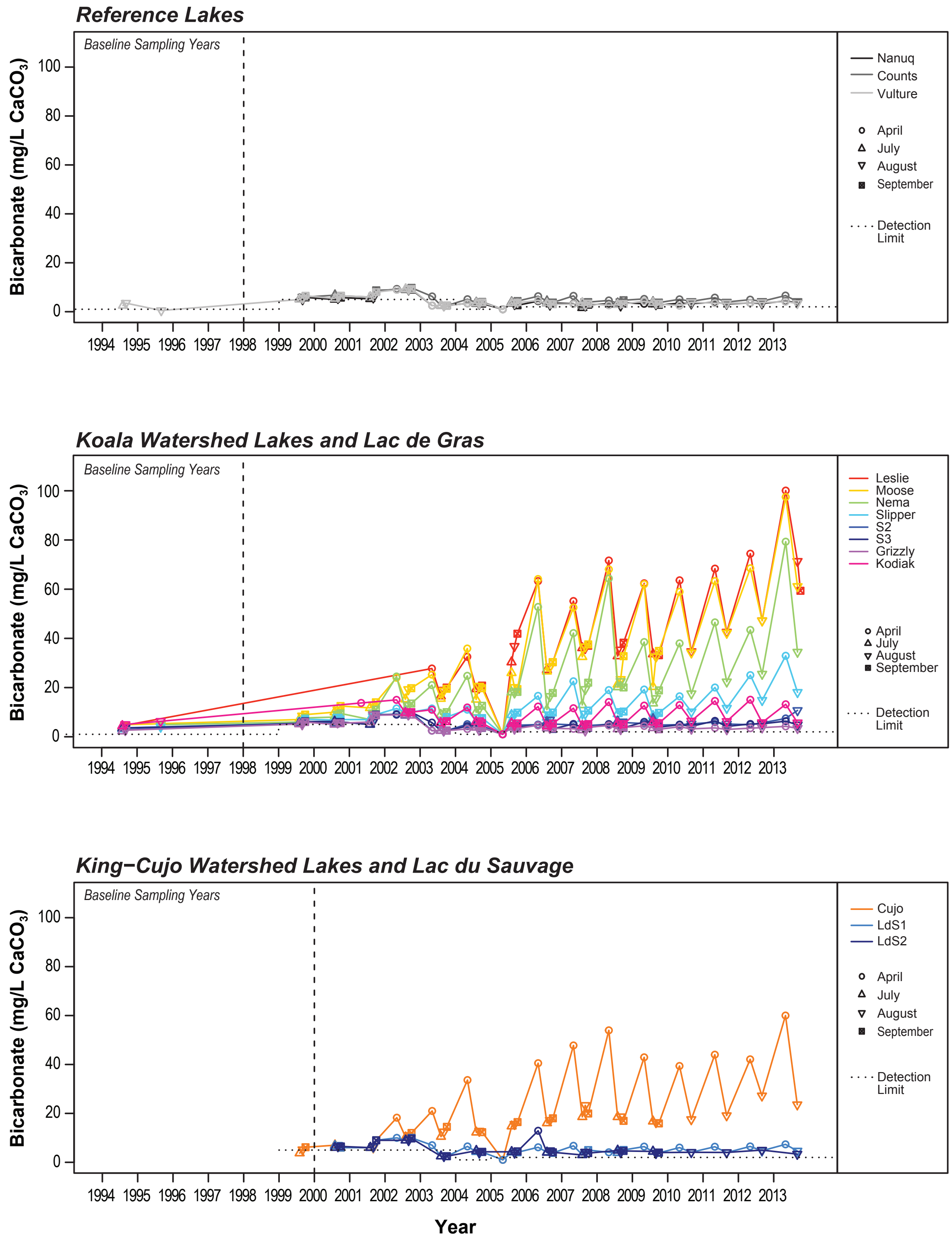
Total Alkalinity
at AEMP Lake Sites, 1994 to 2013



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 5-2

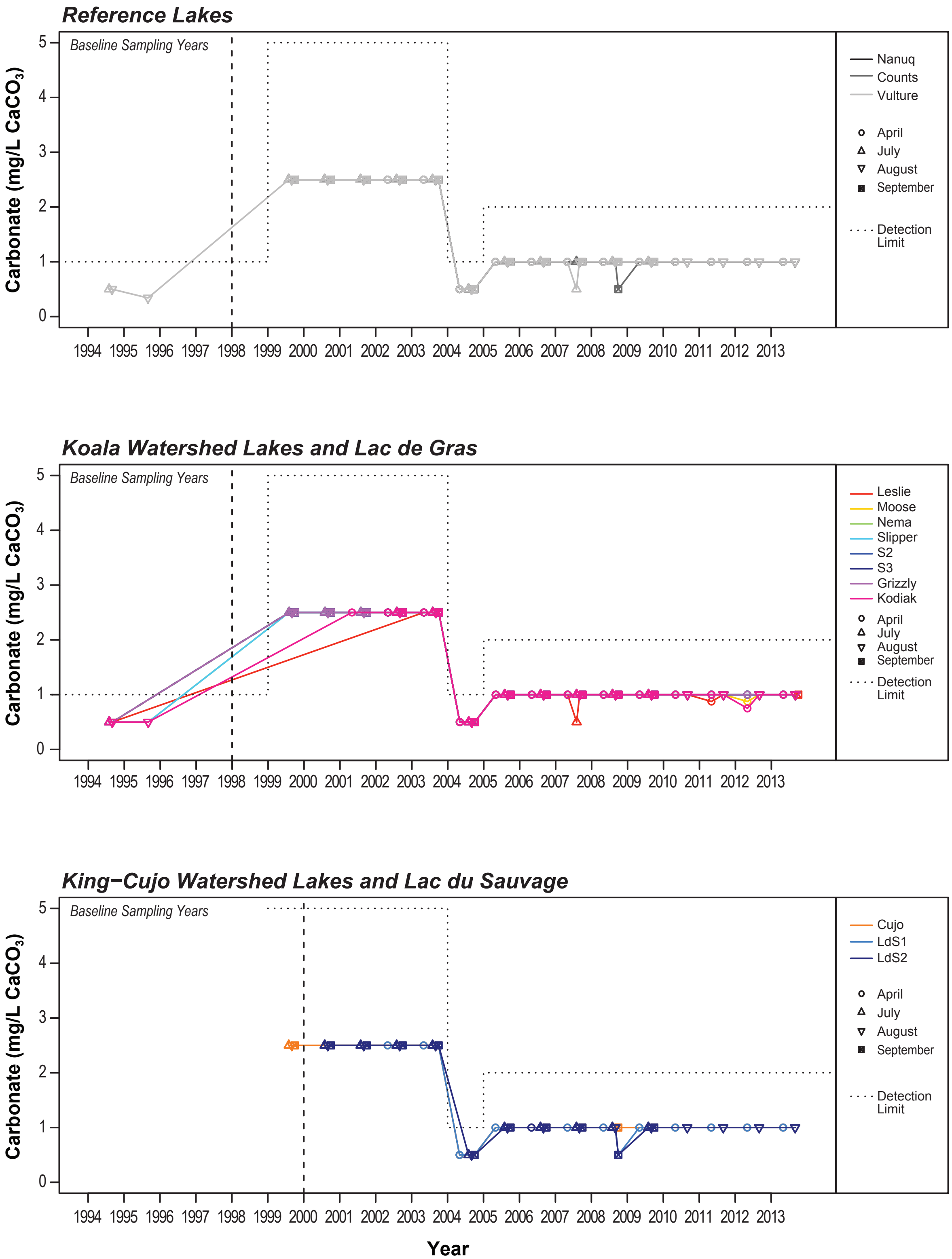
Bicarbonate Concentrations at AEMP Lake Sites, 1994 to 2013



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 5-3
Carbonate Concentrations at AEMP Lake Sites, 1994 to 2013

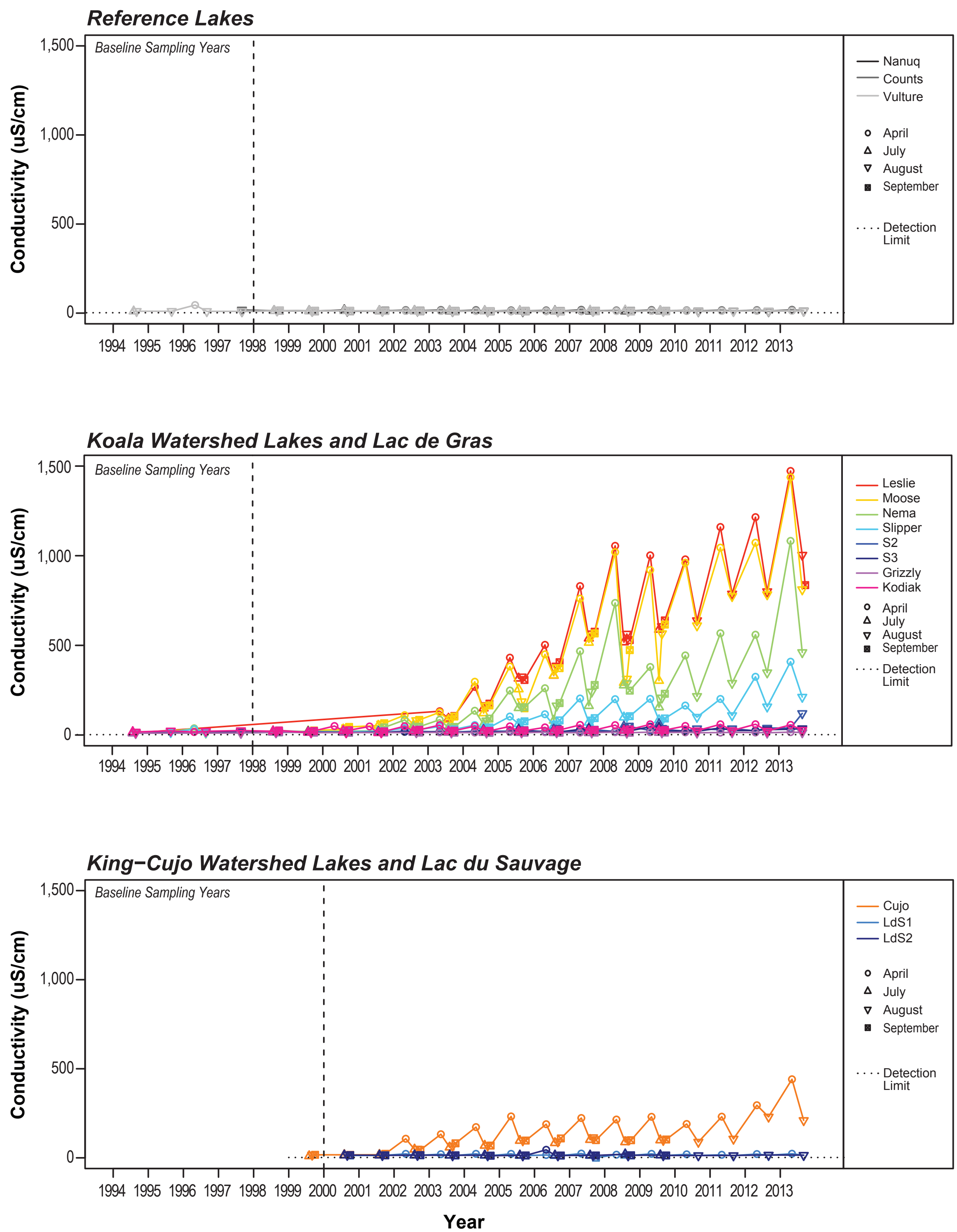
ERM



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 5-4

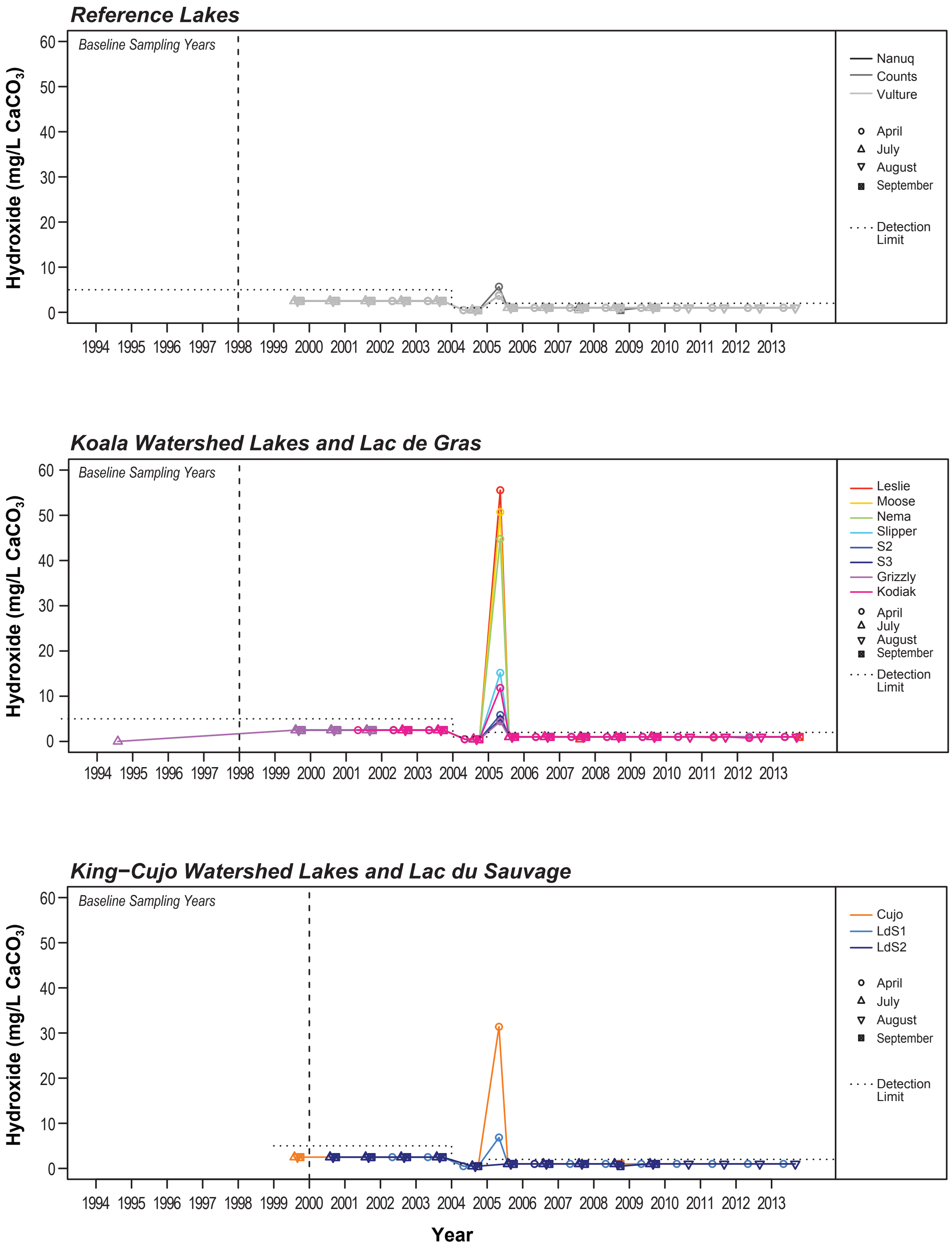
Conductivity at AEMP Lake Sites, 1994 to 2013



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

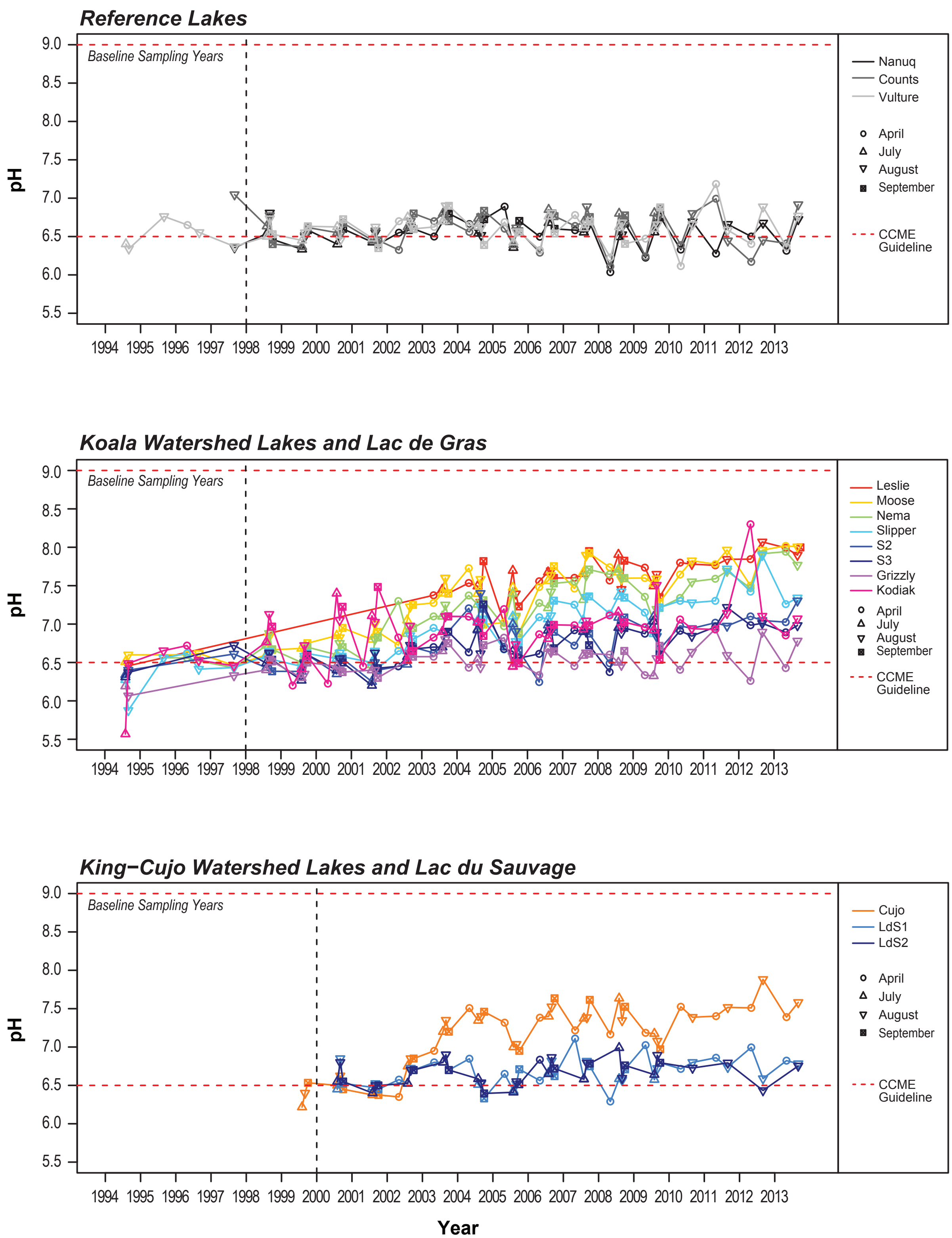
Figure 5-5

Hydroxide Concentrations at AEMP Lake Sites, 1994 to 2013



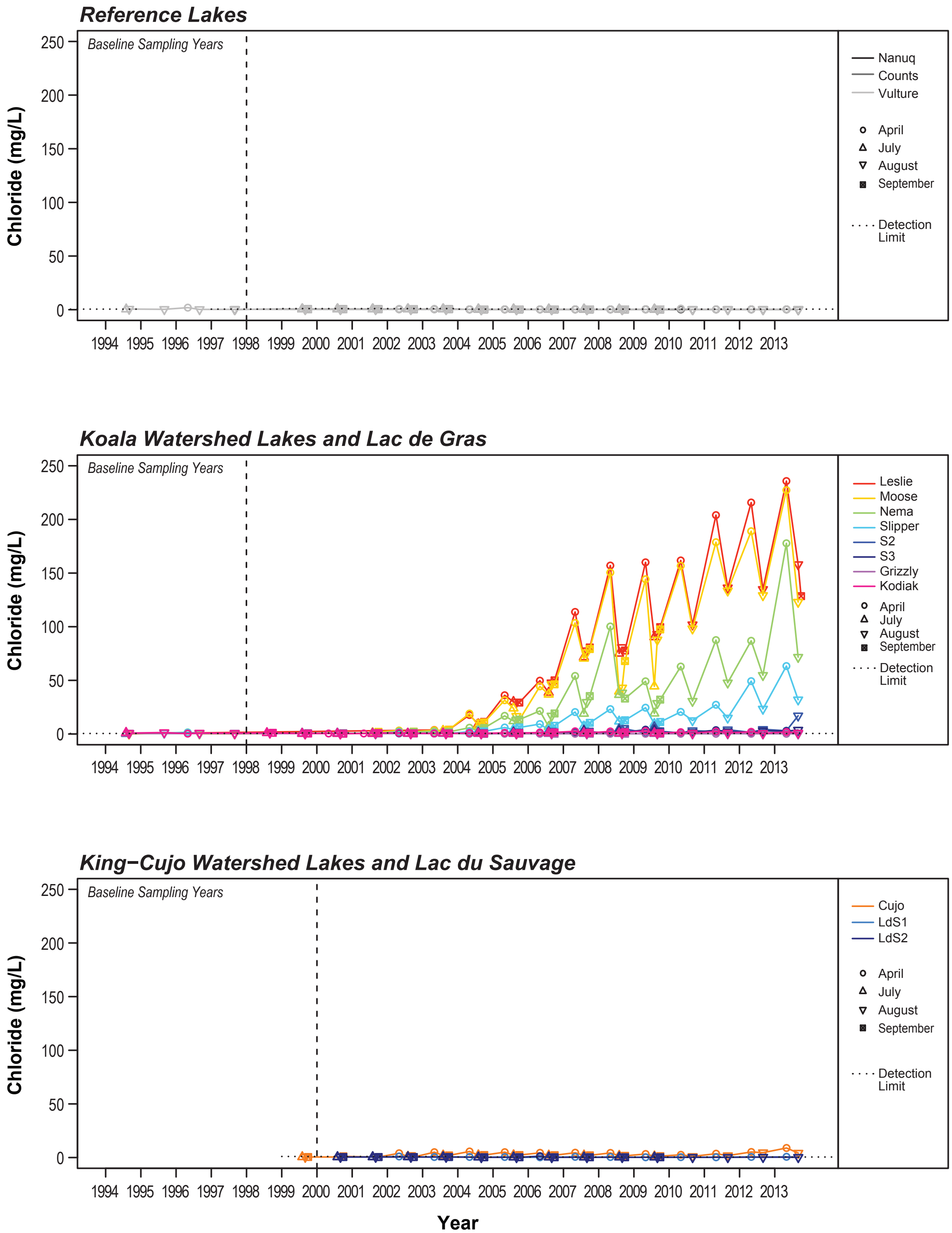
Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 5-6
pH at AEMP Lake Sites, 1994 to 2013



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME Guidelines are 6.5-9.0.

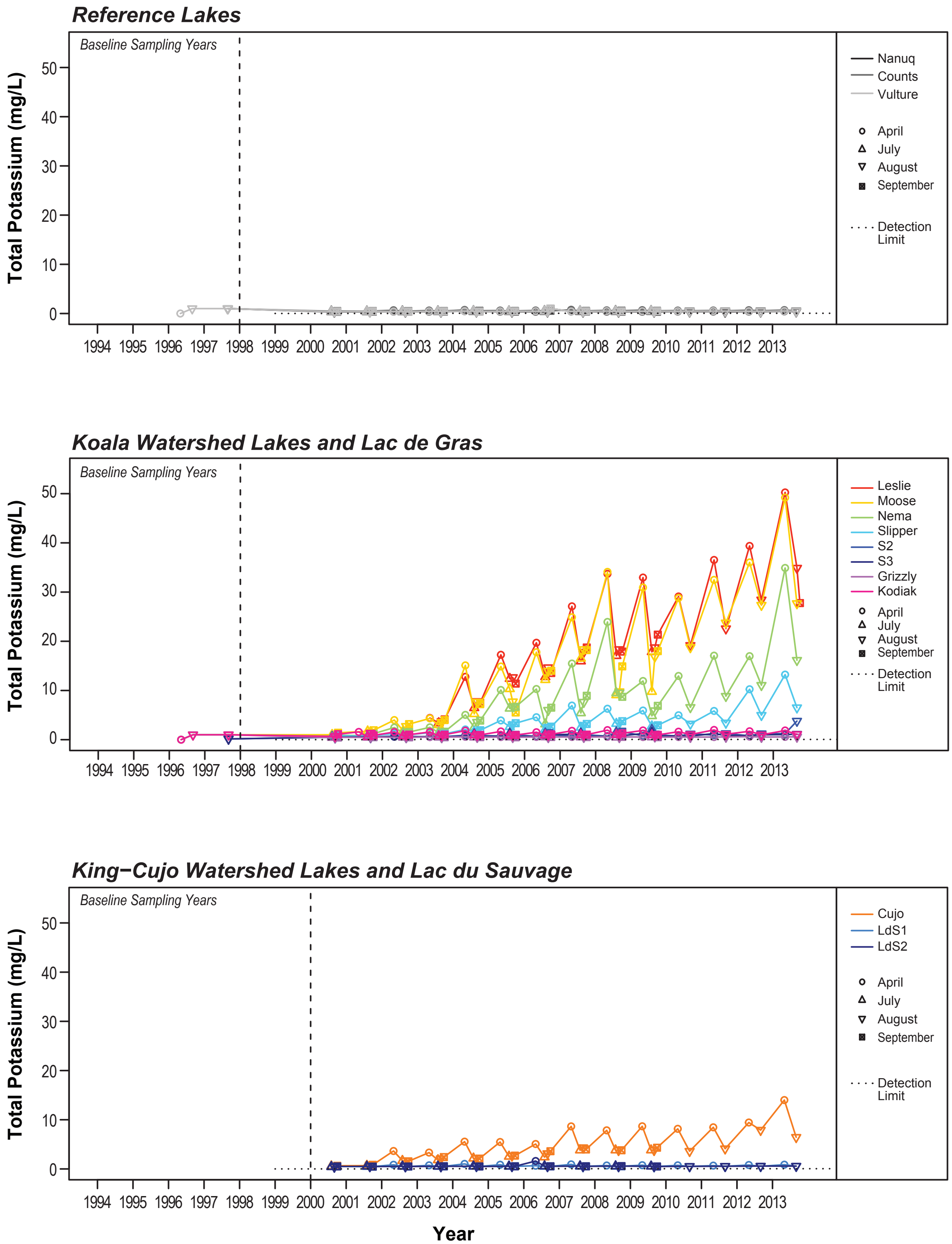
Figure 5-7
Chloride Concentrations at AEMP Lake Sites, 1994 to 2013



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
SSWQO = 116.6 X ln(Hardness) - 204.1 mg/L, where hardness = 10 - 160 mg/L.

Figure 5-8

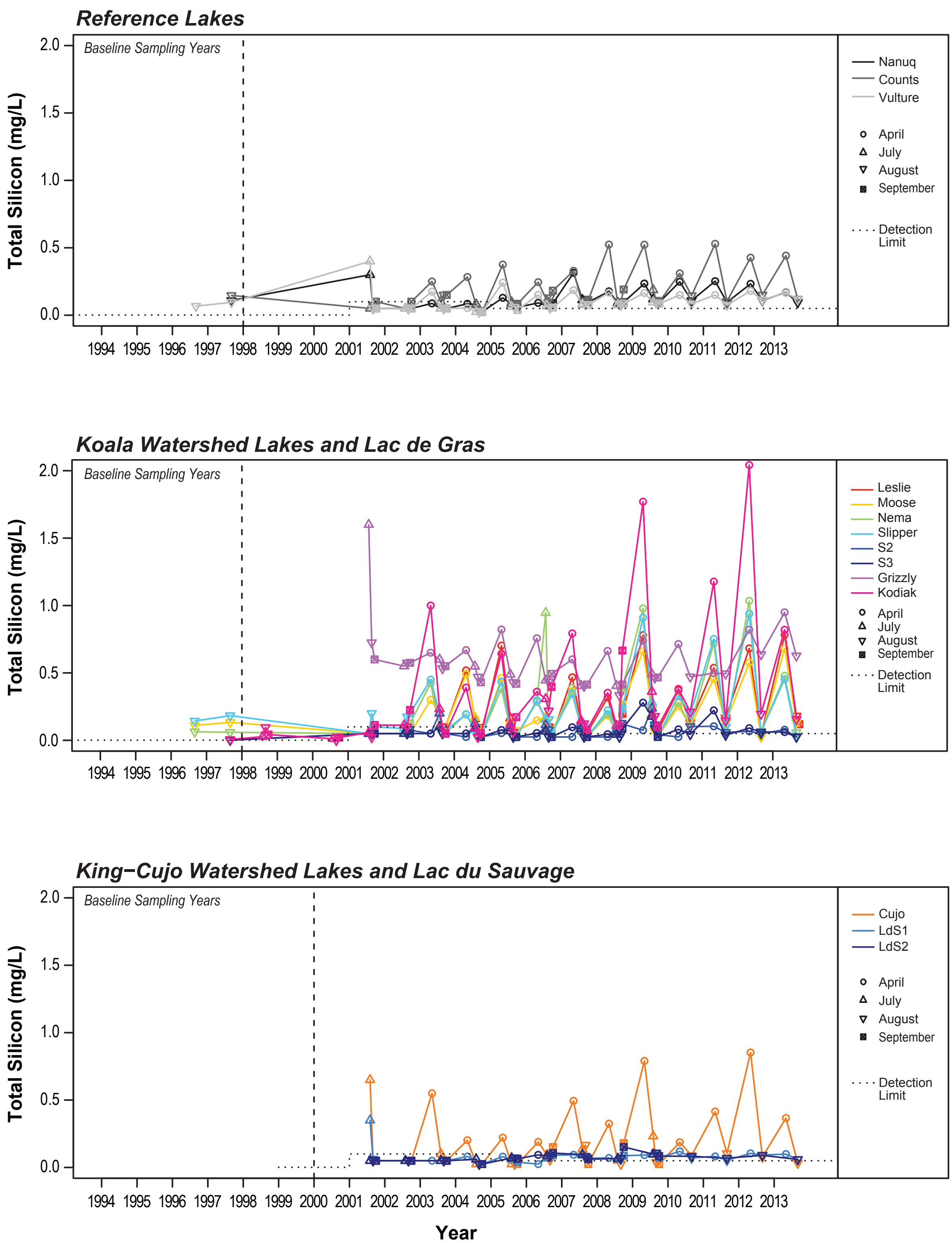
Total Potassium Concentrations at AEMP Lake Sites, 1994 to 2013



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
SSWQO = 41 mg/L.

Figure 5-9

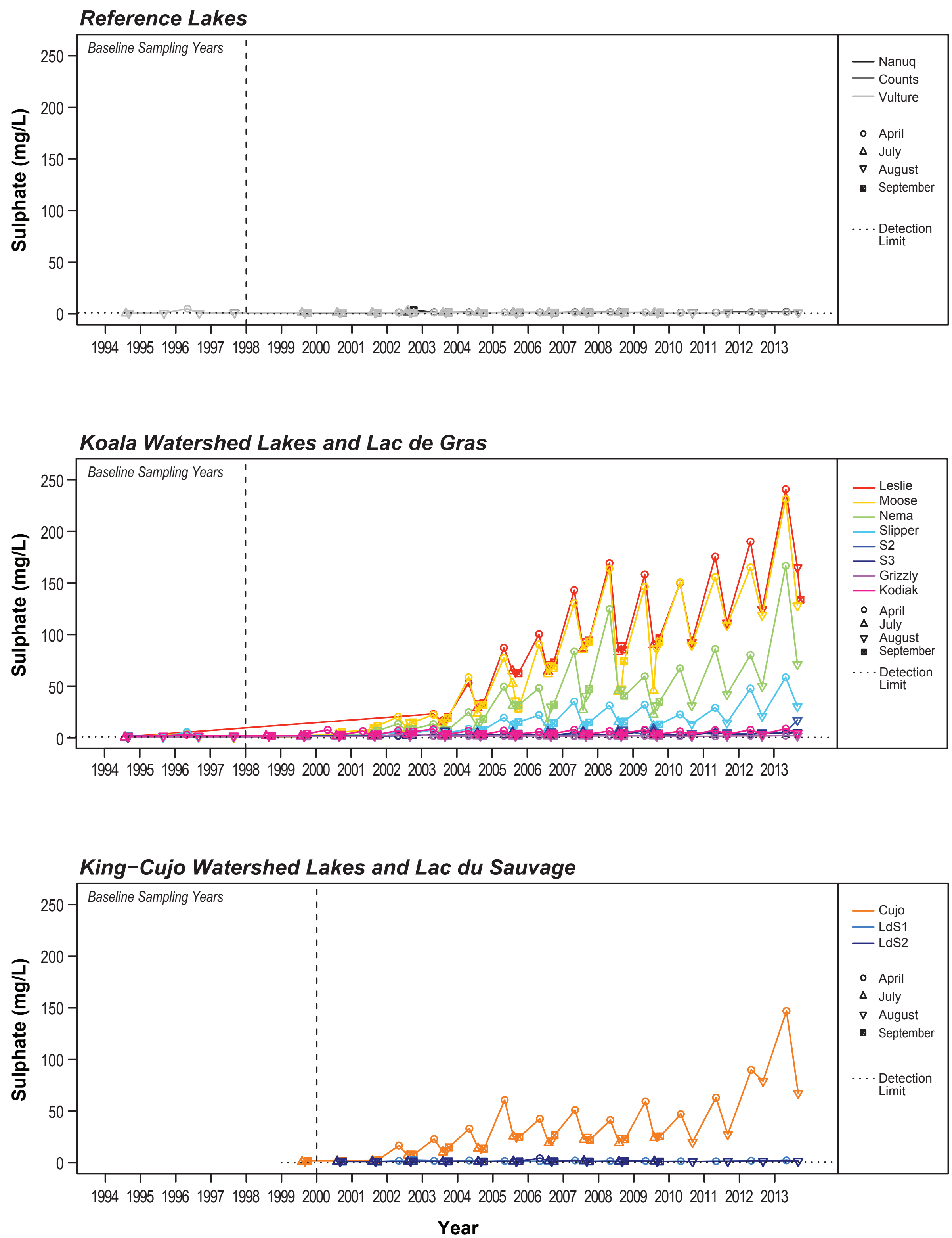
Total Silicon Concentrations at AEMP Lake Sites, 1994 to 2013



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 5-10

Sulphate Concentrations at AEMP Lake Sites, 1994 to 2013

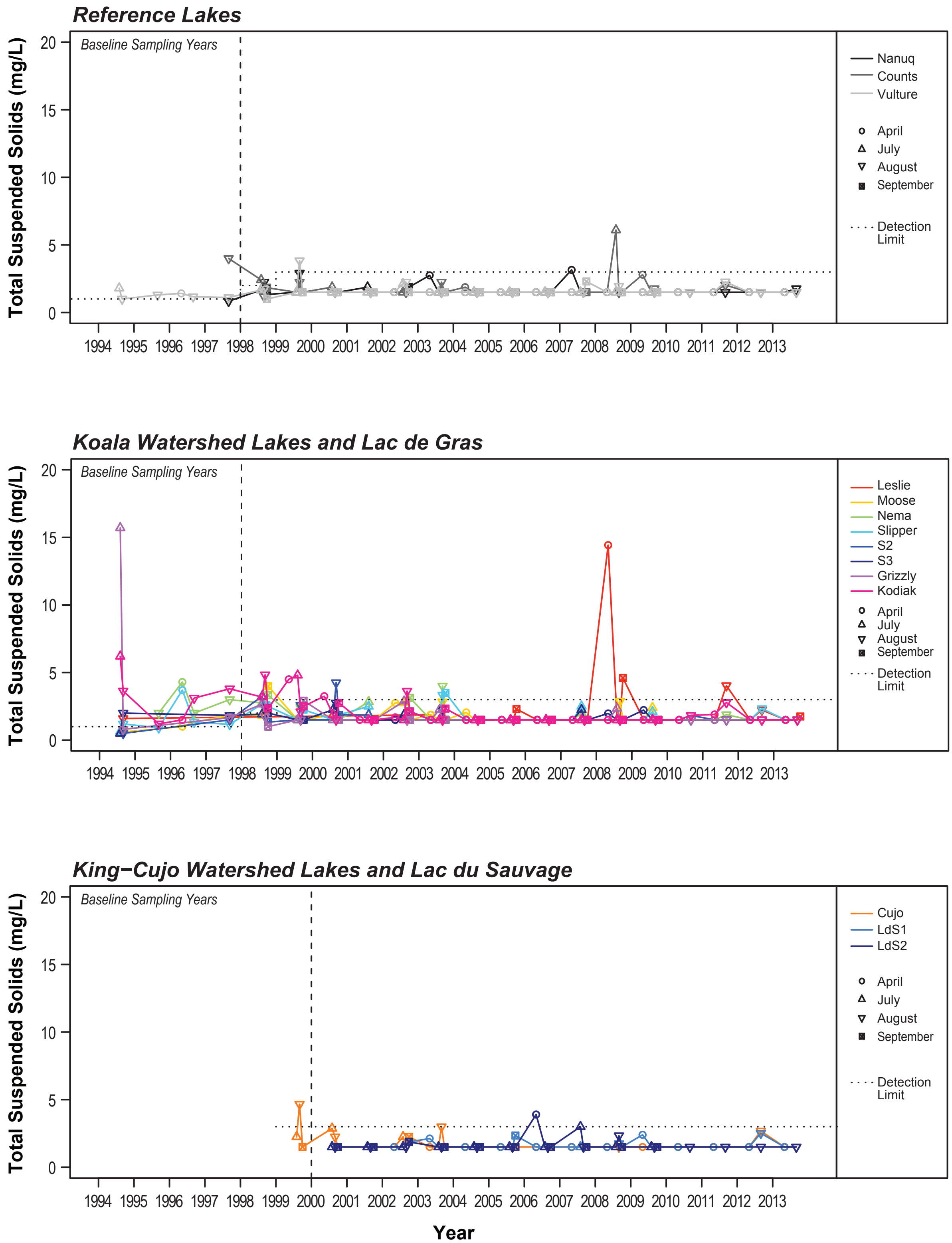


Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
SSWQO = $e^{(0.9116 \times \ln(\text{Hardness}) + 1.712)}$ mg/L, where hardness < 160 mg/L.

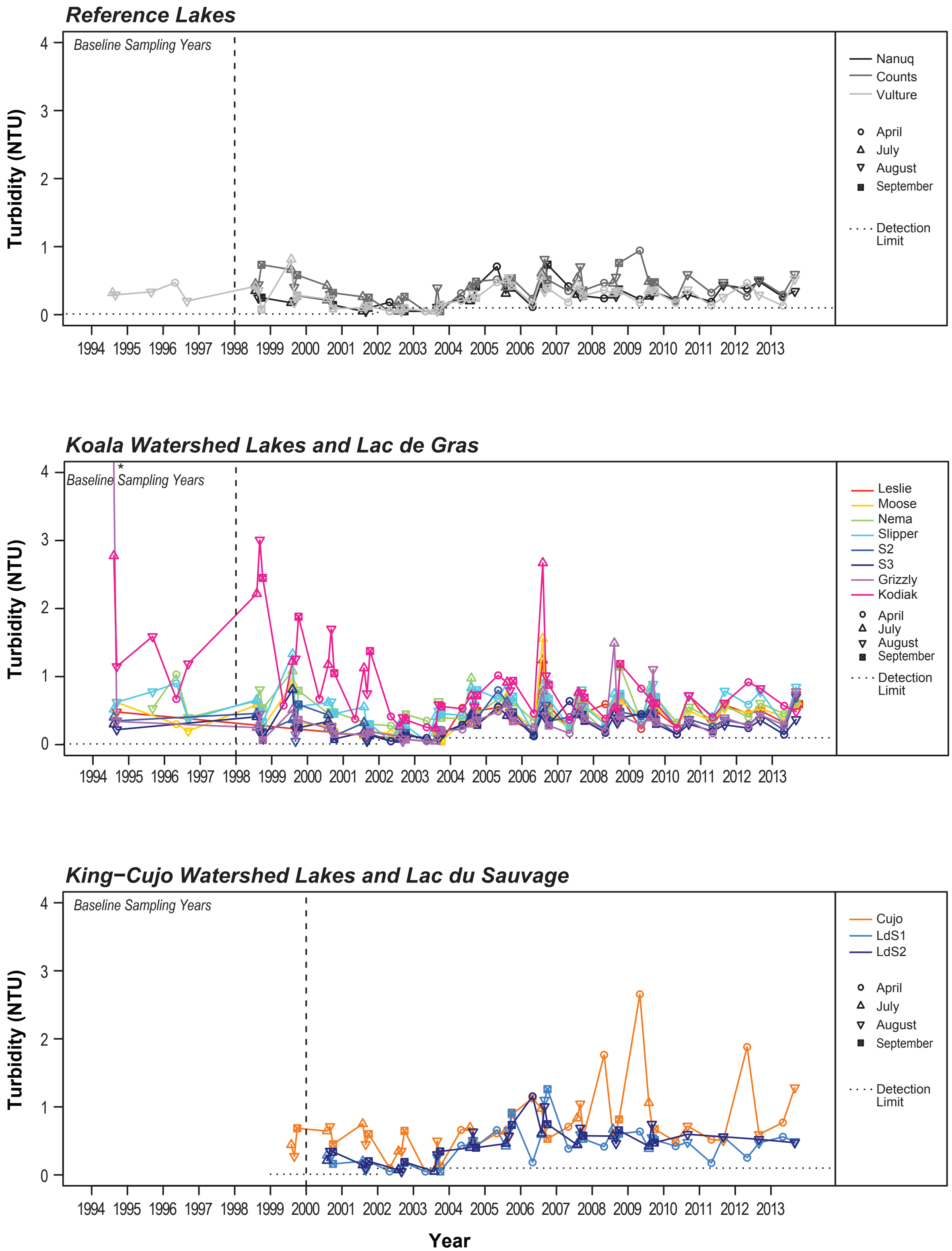
Figure 5-11

Total Suspended Solids at AEMP Lake Sites, 1994 to 2013

ERM

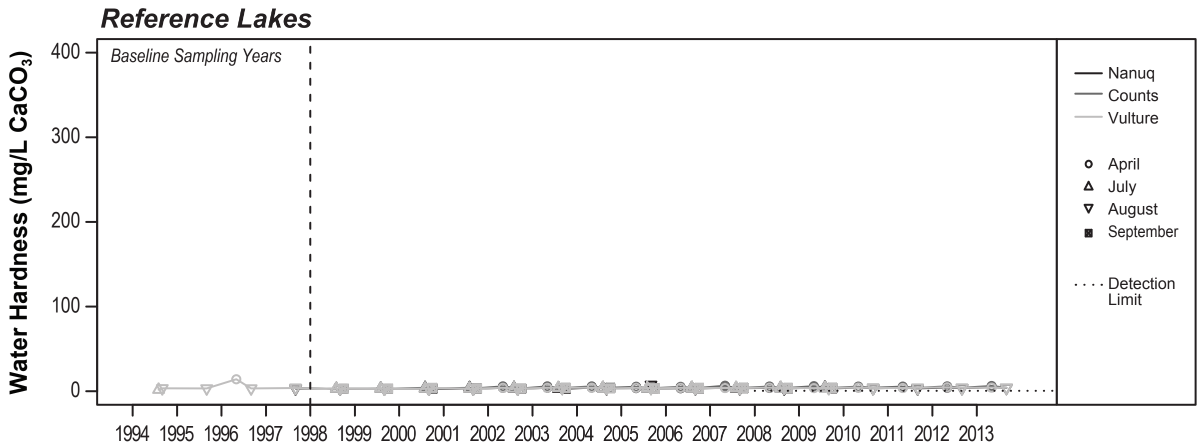


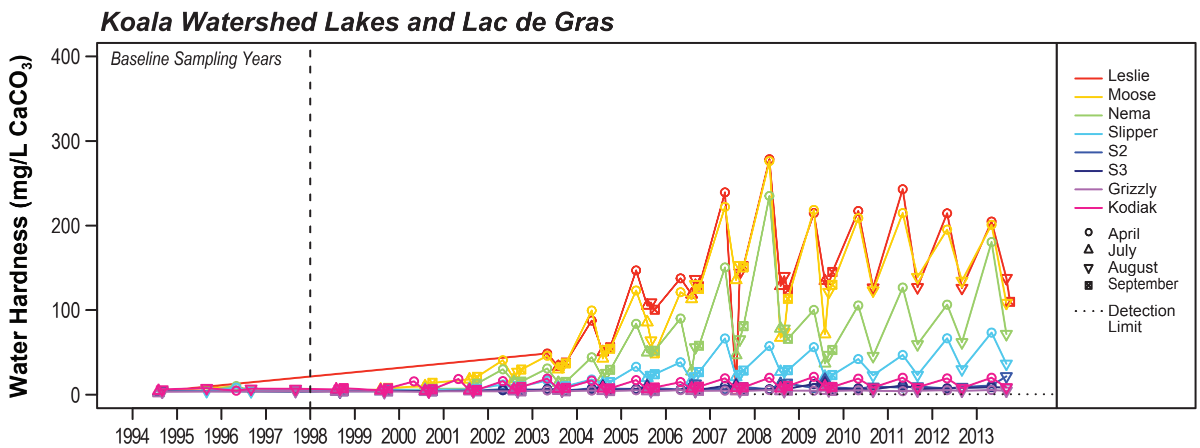
Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME Guideline = a maximum average increase of 5 mg/L from background levels (long term exposure).

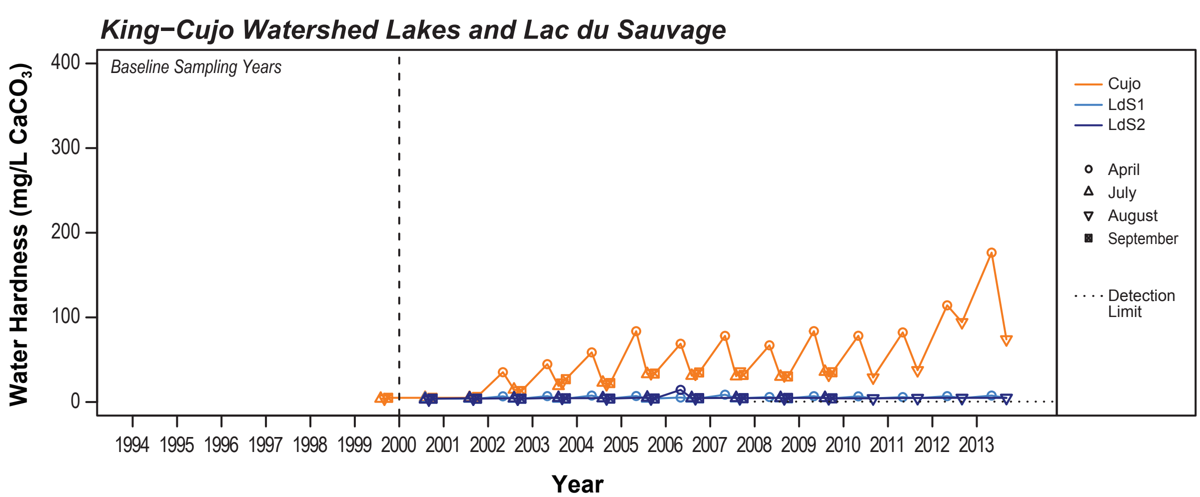


Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
** = 20.5 NTU*
CCME Guideline = a maximum average increase of 2 NTUs from background levels (long term exposure).

Figure 5-13
Water Hardness at AEMP Lake Sites, 1994 to 2013



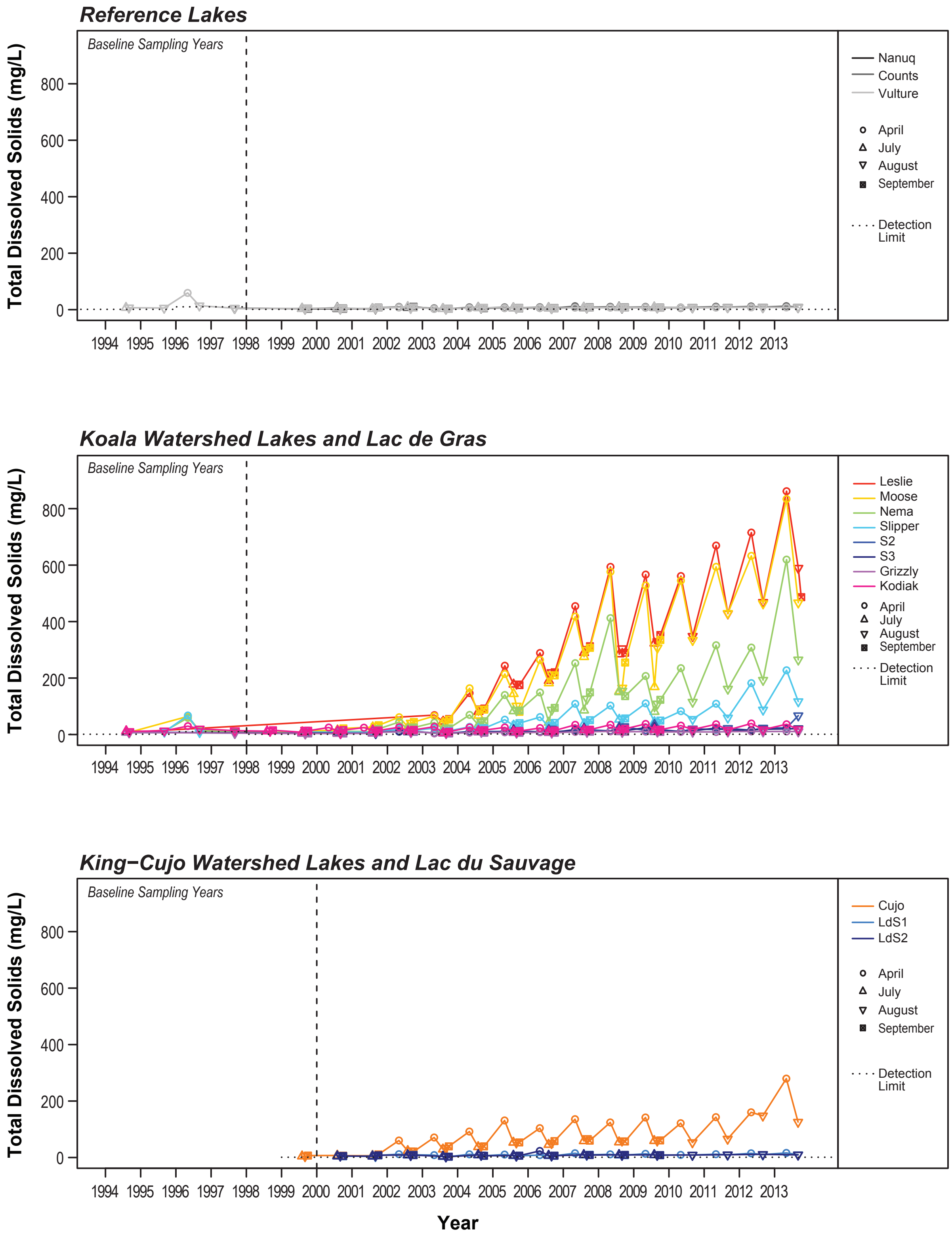




Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 5-14

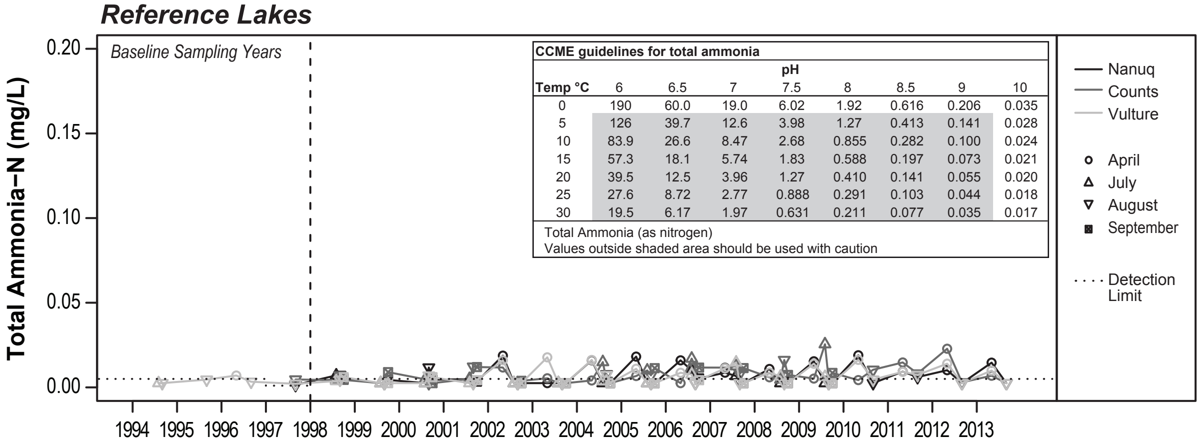
Total Dissolved Solids at AEMP Lake Sites, 1994 to 2013

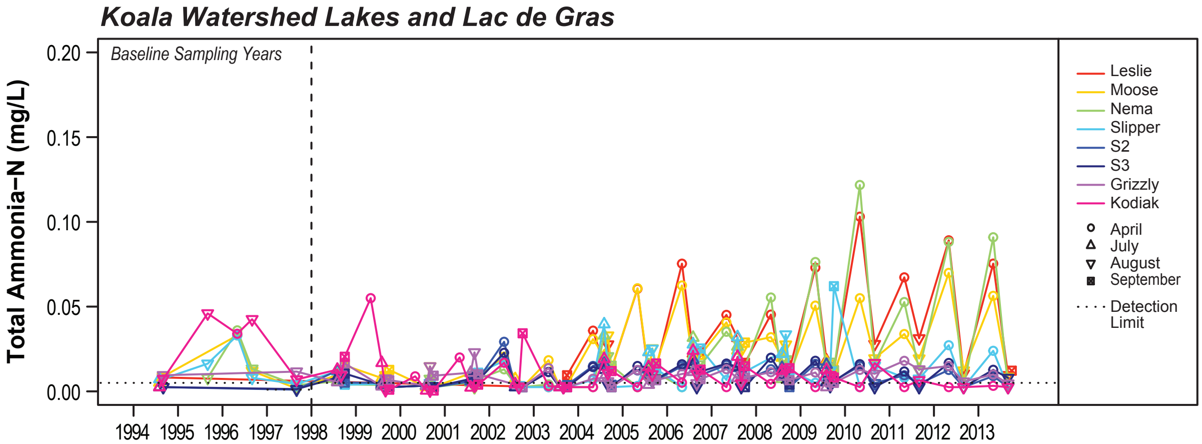


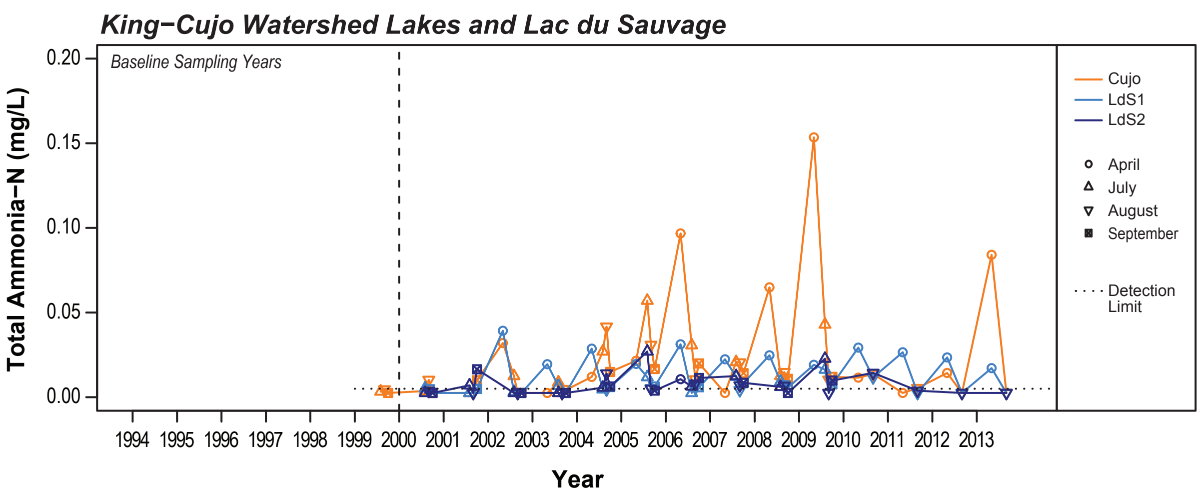
Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 5-15
Total Ammonia-N Concentrations at AEMP Lake Sites, 1994 to 2013

ERM



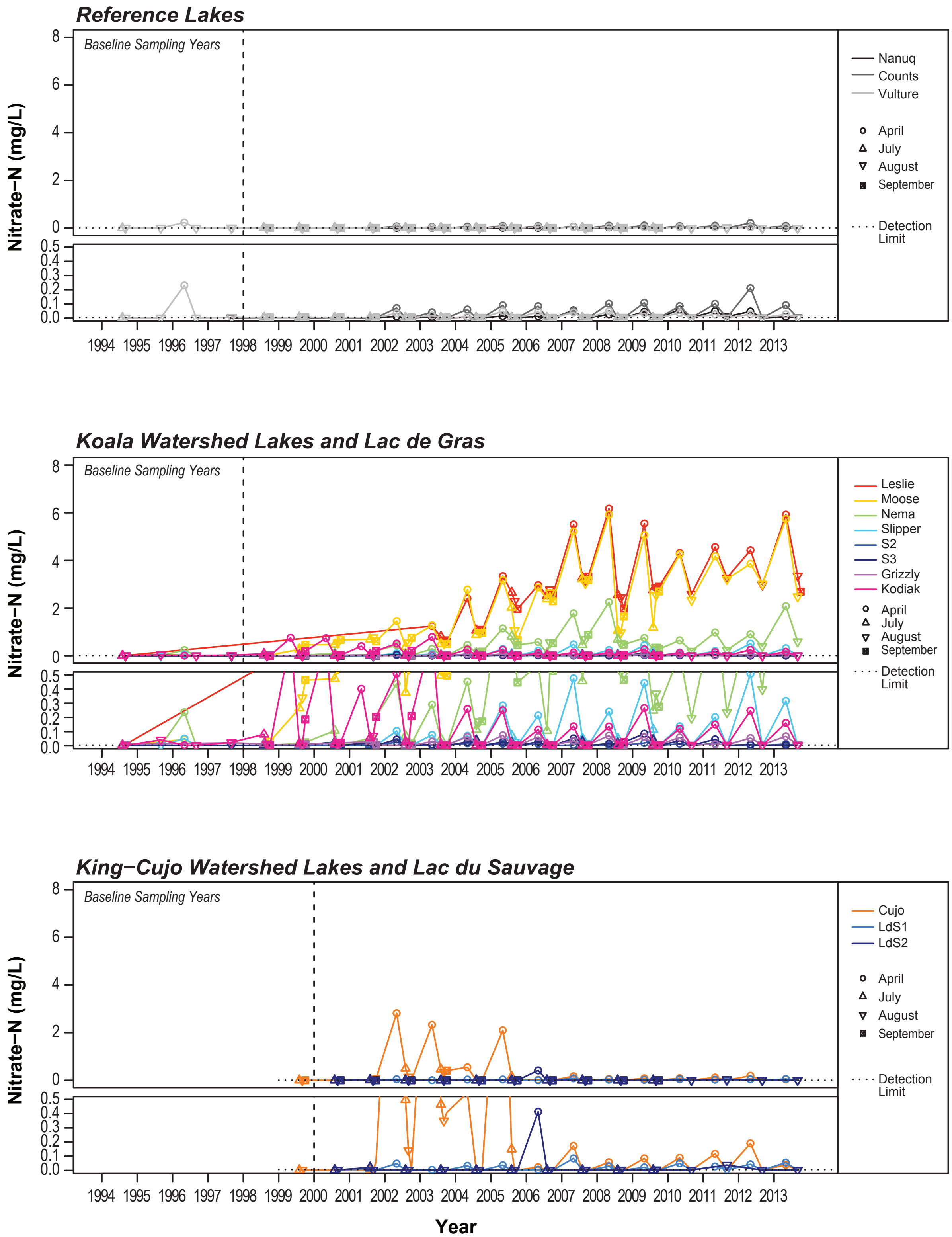




Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown. CCME guidelines are pH- and temperature-dependent (see inset).

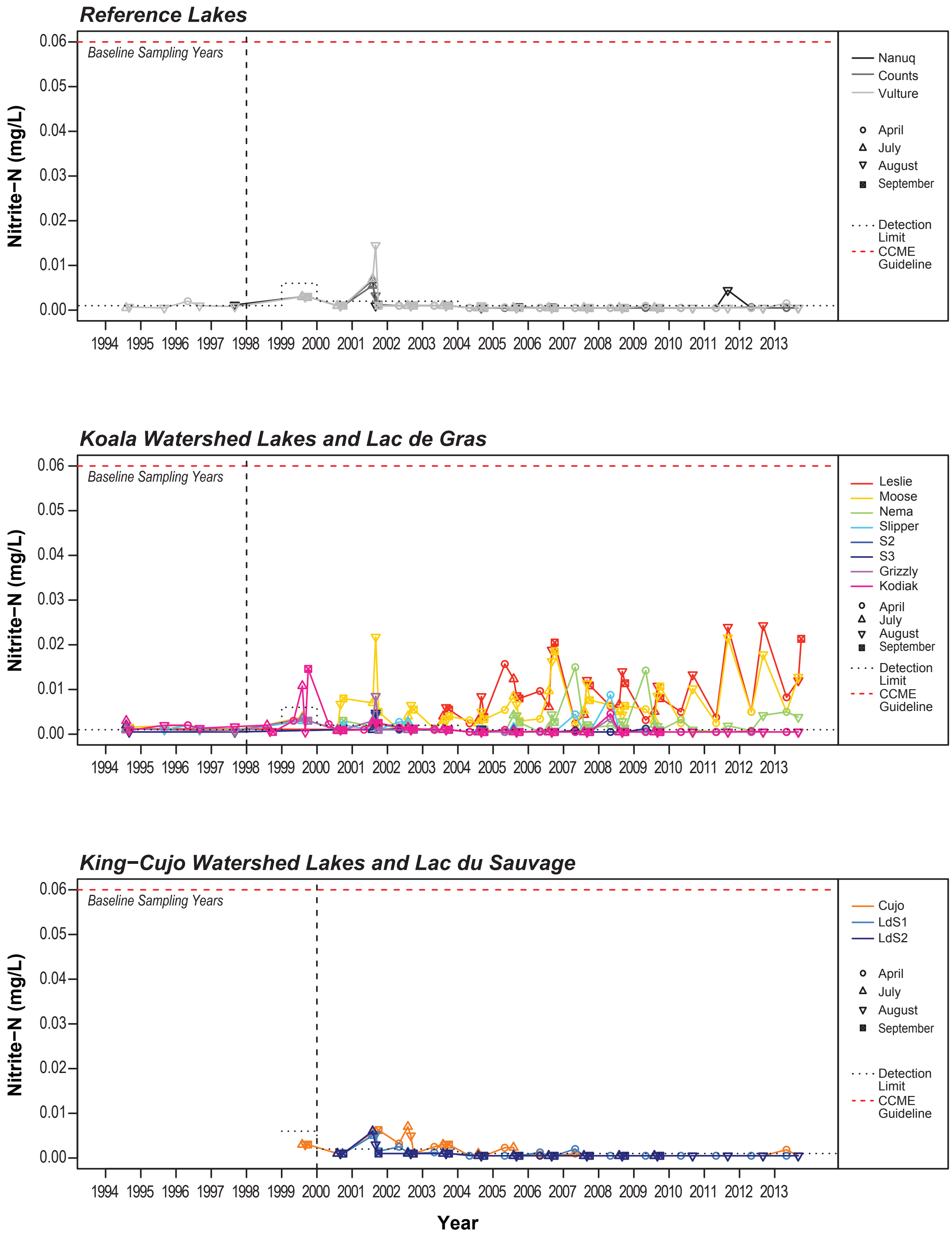
Figure 5-16

Nitrate-N Concentrations at AEMP Lake Sites, 1994 to 2013



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
SSWQO = $e^{0.9518 \times \ln(\text{Hardness}) - 2.032}$ mg/L, where hardness < 160 mg/L.

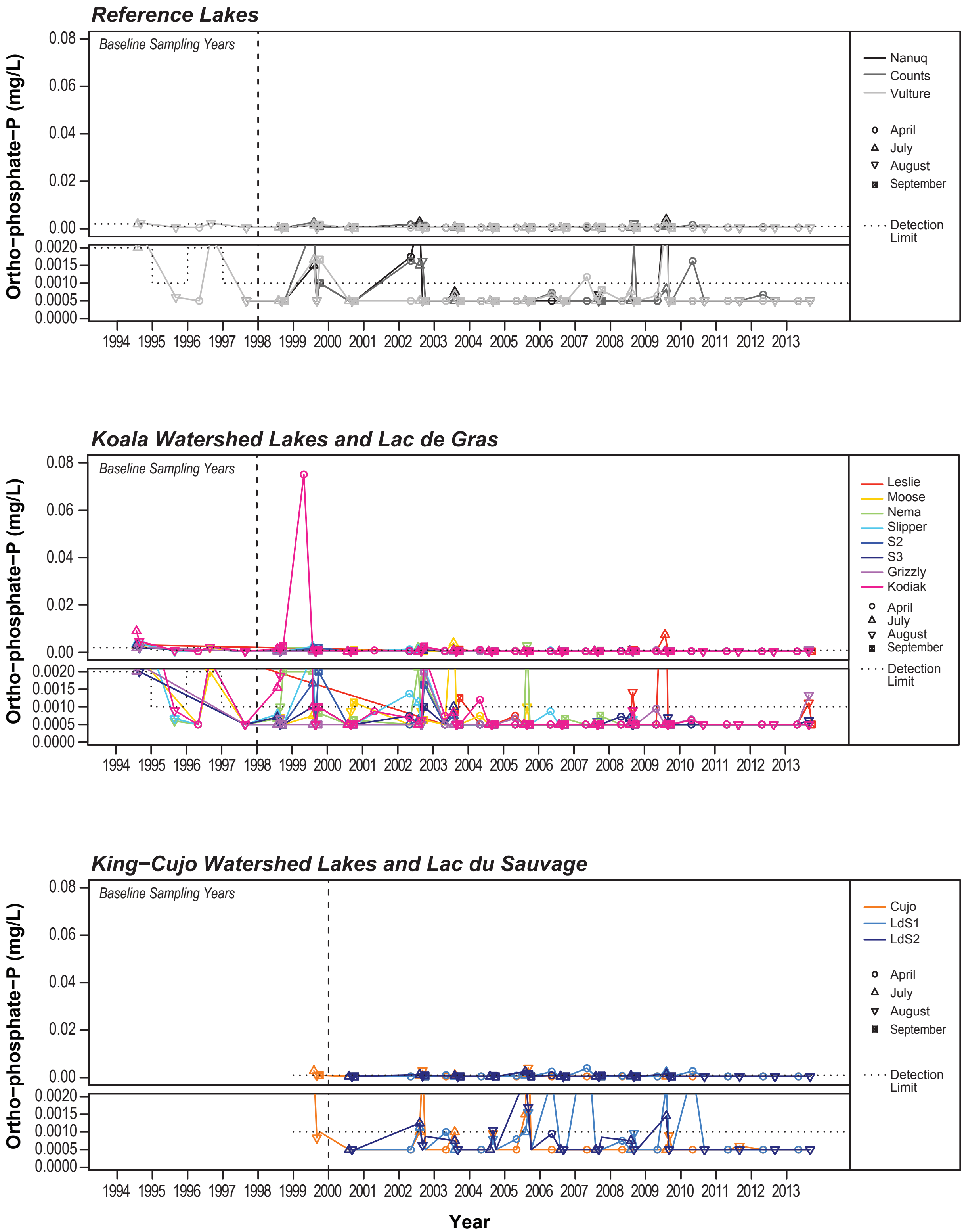
Nitrite-N Concentrations at AEMP Lake Sites, 1994 to 2013



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME Guideline = 0.06 mg/L.

Figure 5-18

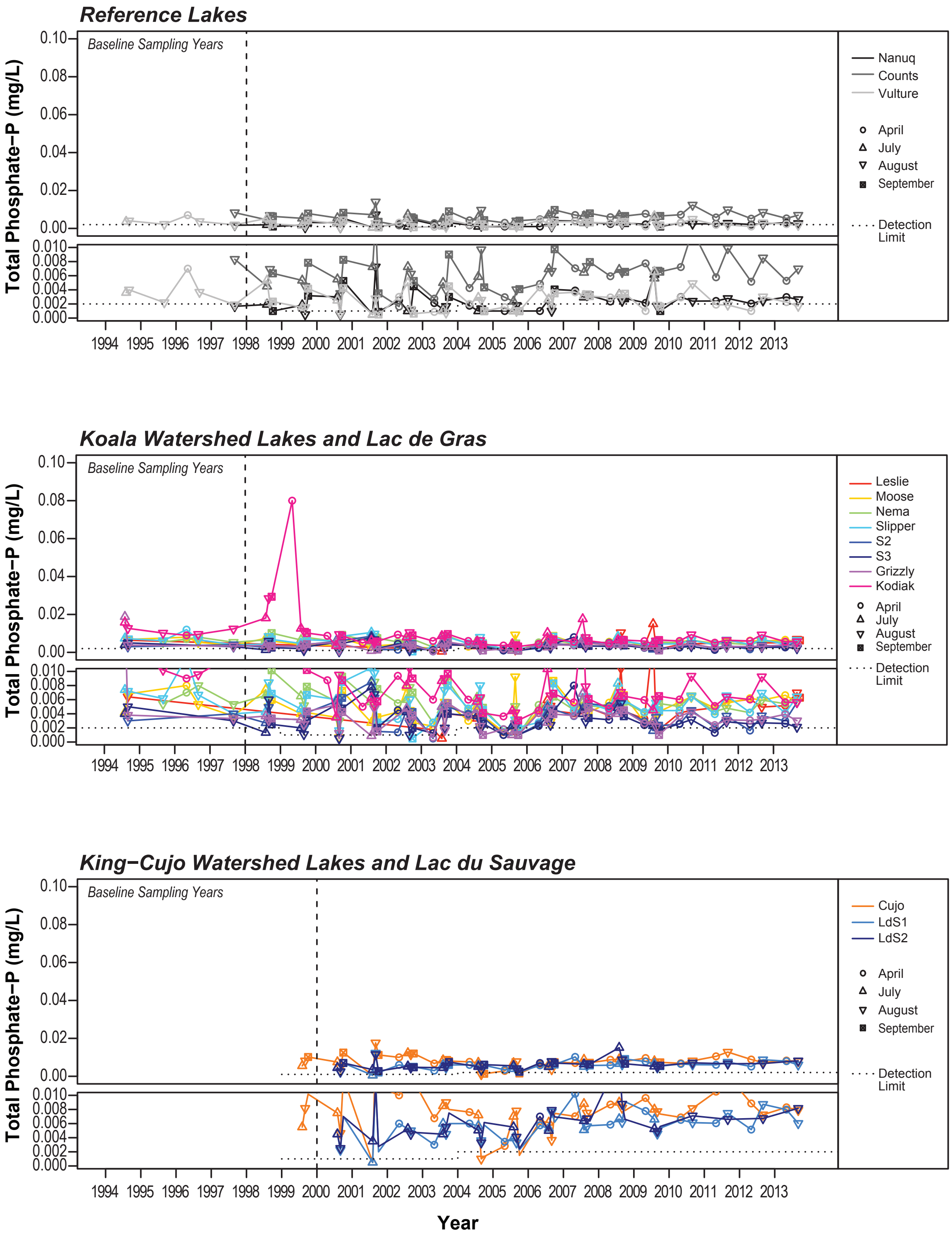
Orthophosphate-P Concentrations at AEMP Lake Sites, 1994 to 2013



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 5-19

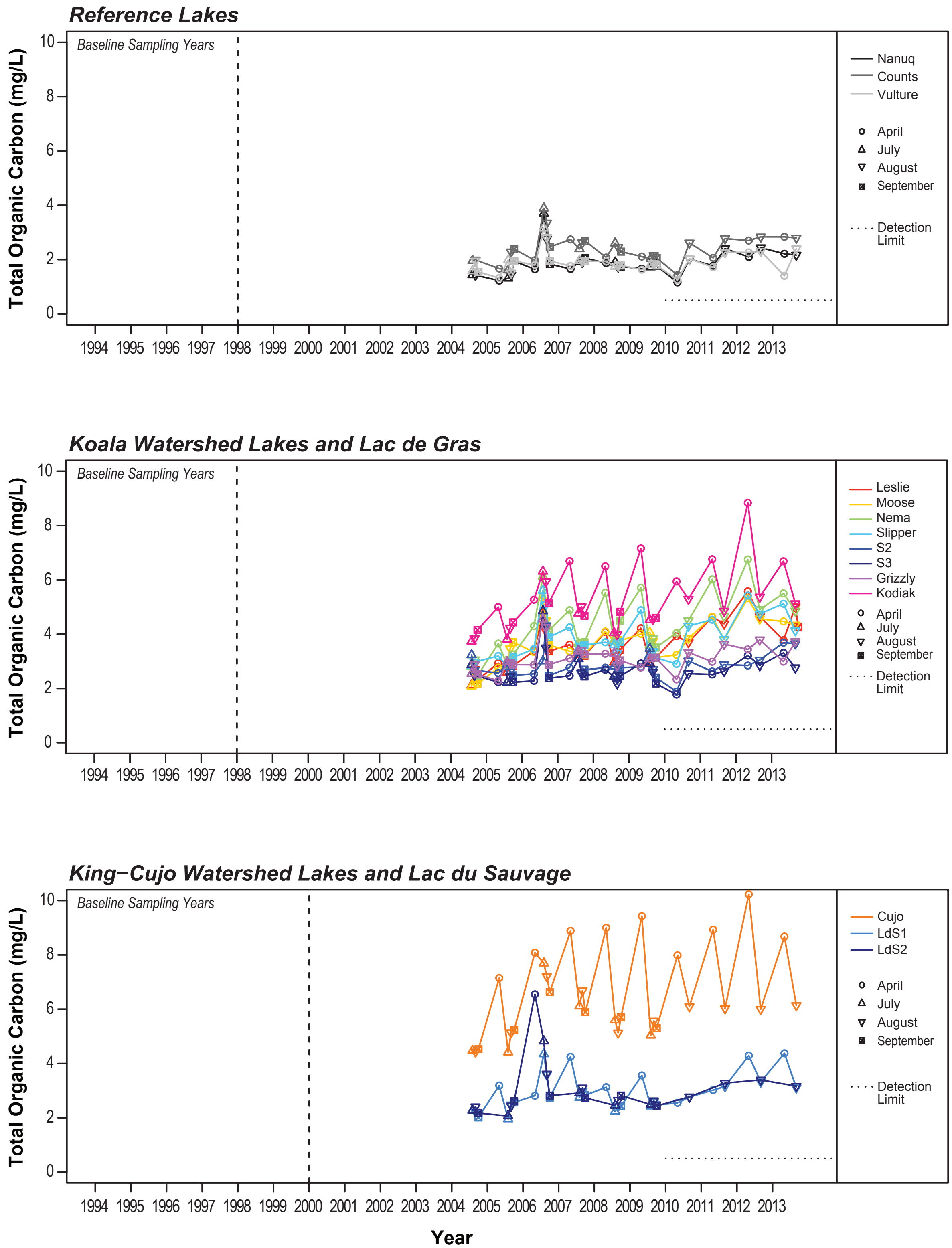
Total Phosphate-P Concentrations at AEMP Lake Sites, 1994 to 2013



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
See Tables 2.3-3. and 2.3-4. for phosphorus trigger ranges and lake-specific benchmark concentrations.

Figure 5-20

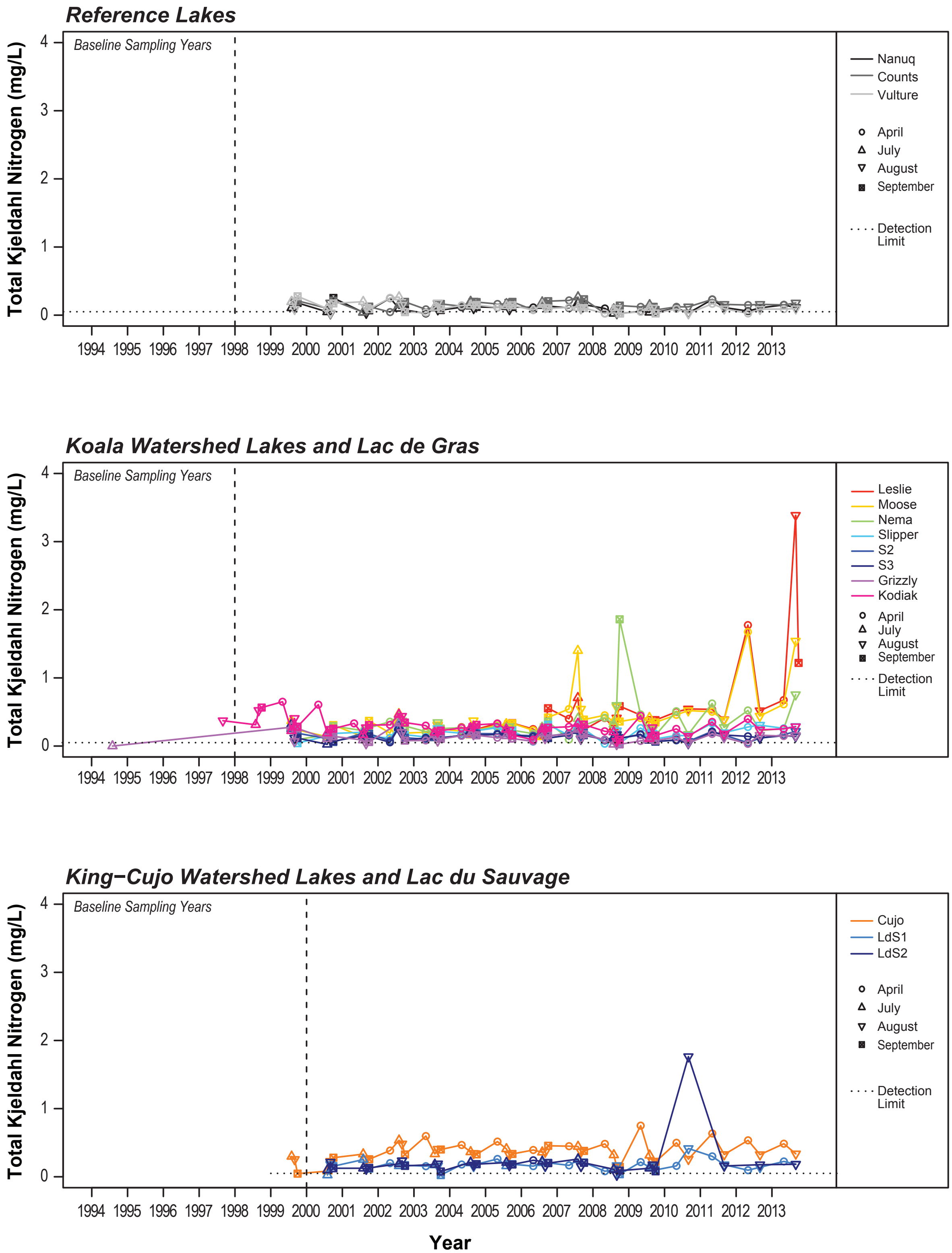
Total Organic Carbon Concentrations at AEMP Lake Sites, 1994 to 2013



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 5-21

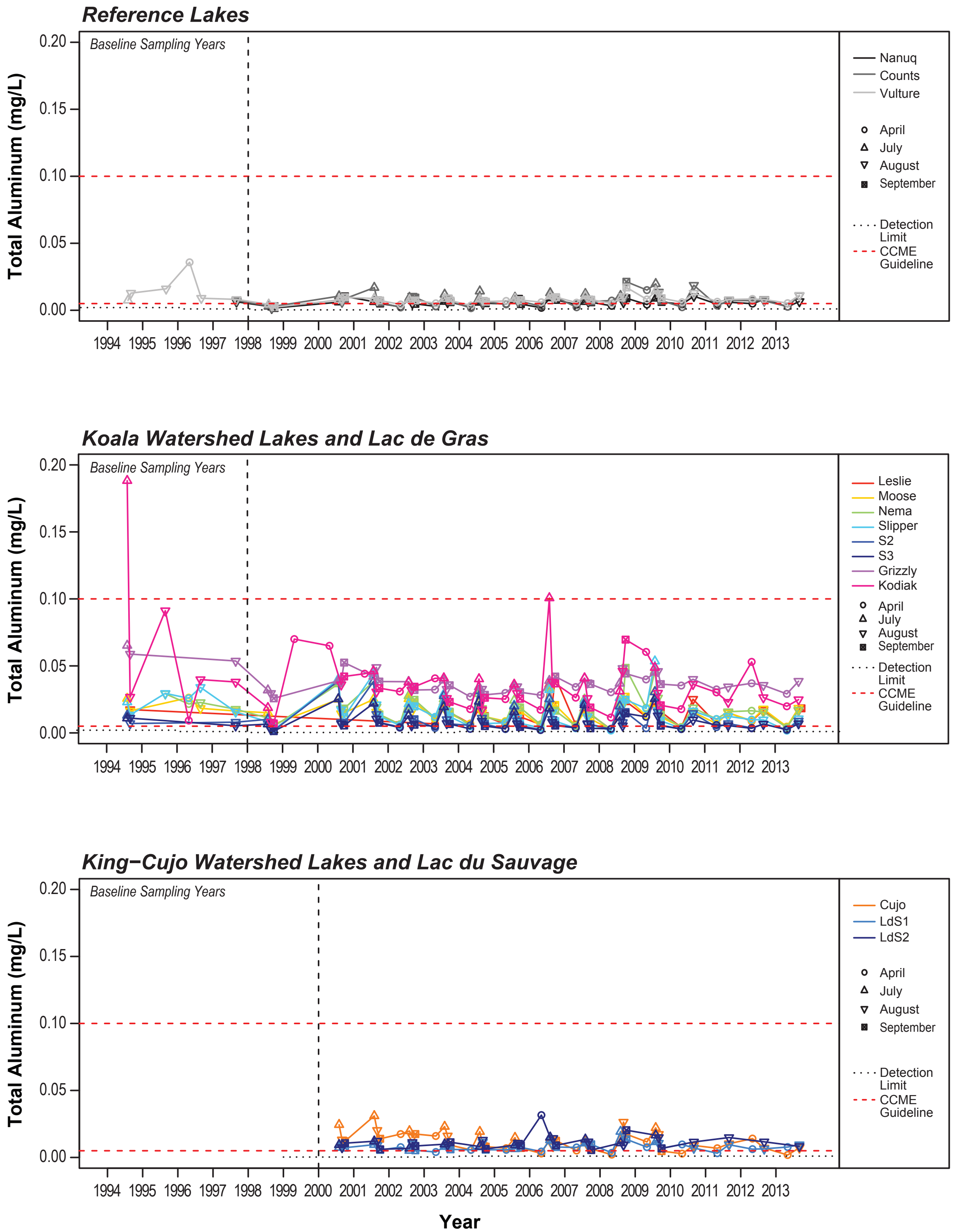
Total Kjeldhal Nitrogen Concentrations at AEMP Lake Sites, 1994 to 2013



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 5-22

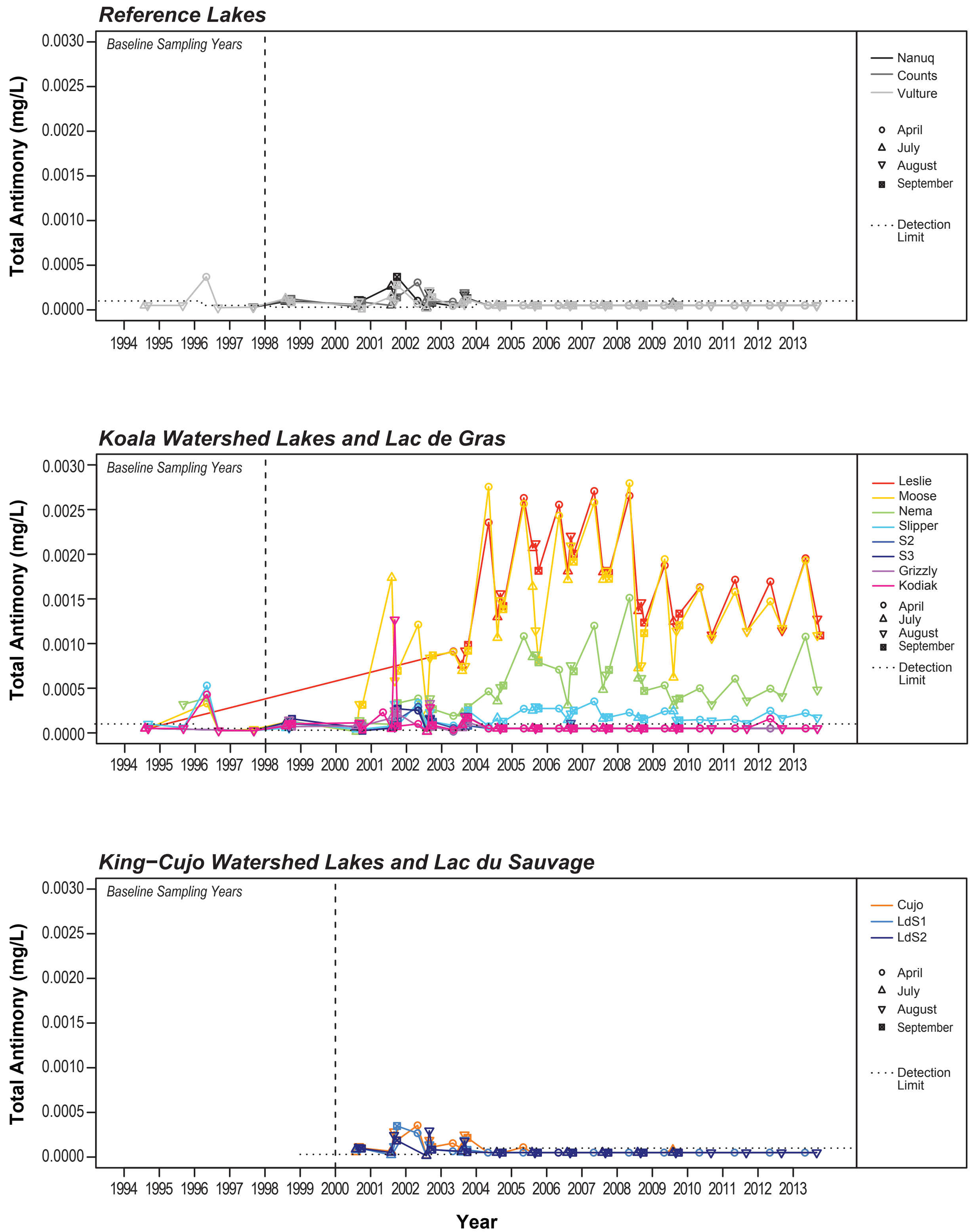
Total Aluminum Concentrations at AEMP Lake Sites, 1994 to 2013



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME Guideline = 0.005 mg/L at pH < 6.5; 0.1 mg/L at pH ≥ 6.5.

Figure 5-23

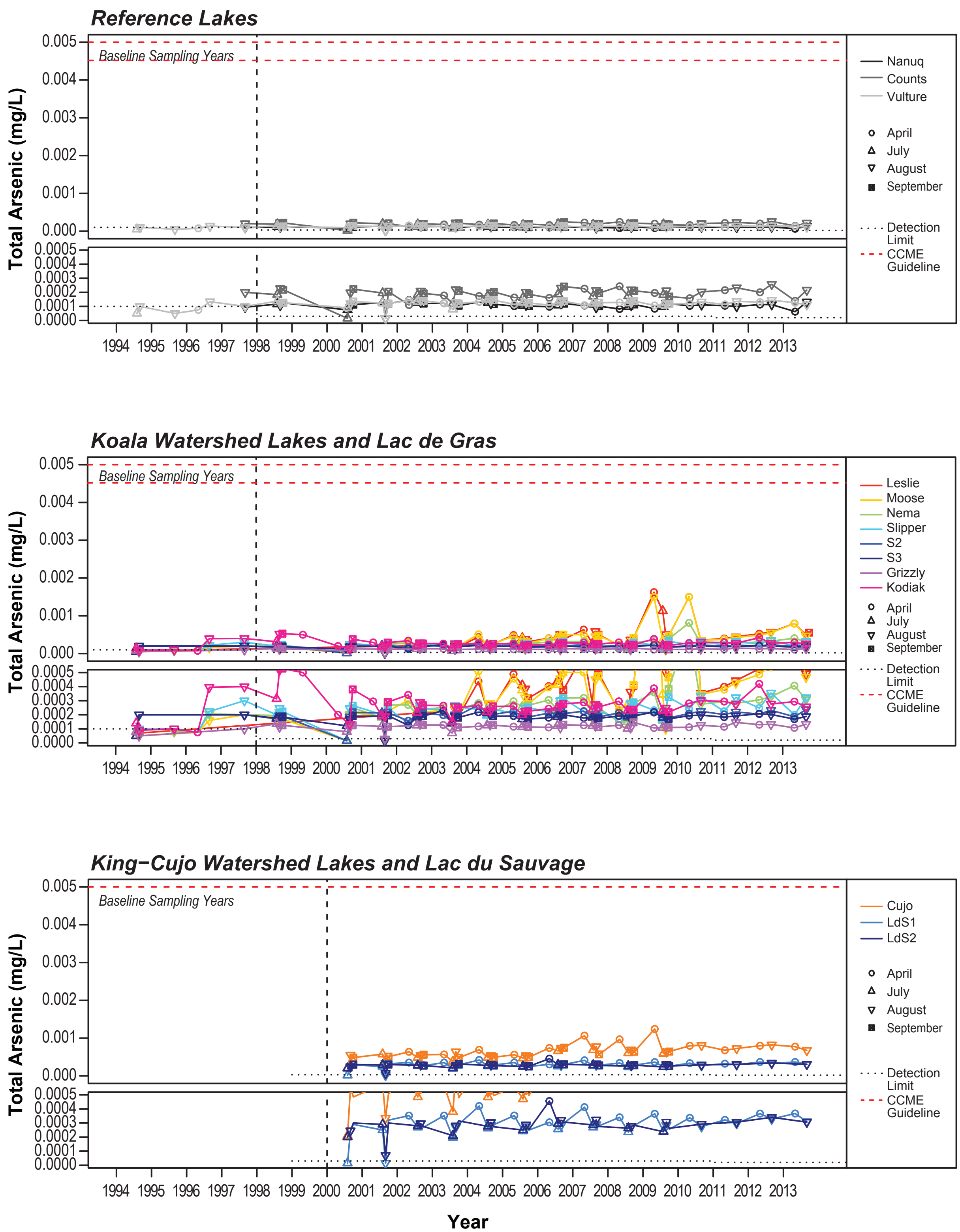
Total Antimony Concentrations at AEMP Lake Sites, 1994 to 2013



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
Water quality benchmark (Fletcher et al. 1996) = 0.02 mg/L.

Figure 5-24

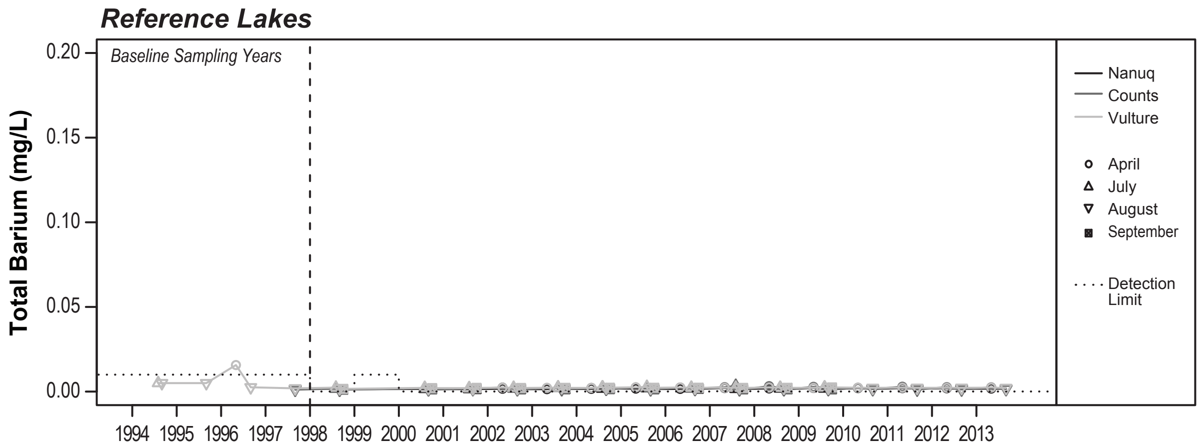
Total Arsenic Concentrations at AEMP Lake Sites, 1994 to 2013

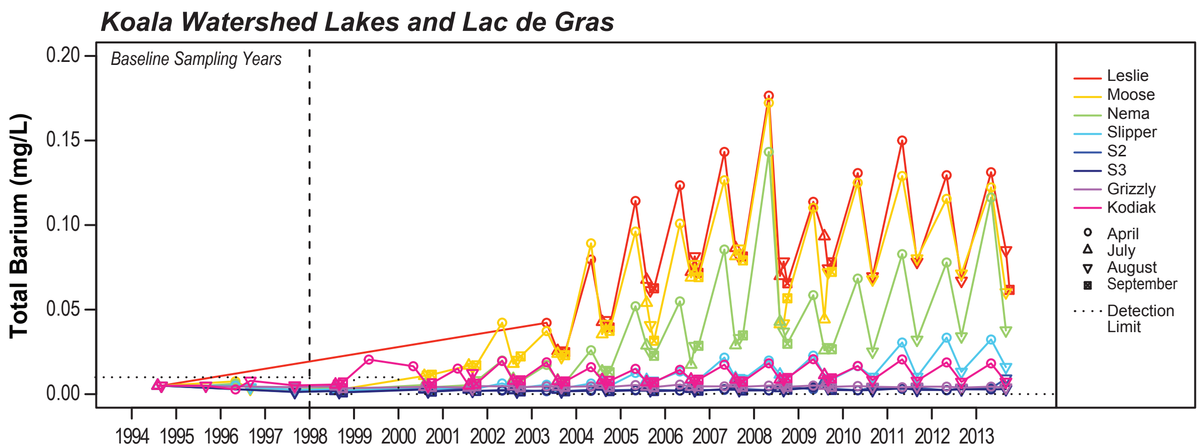


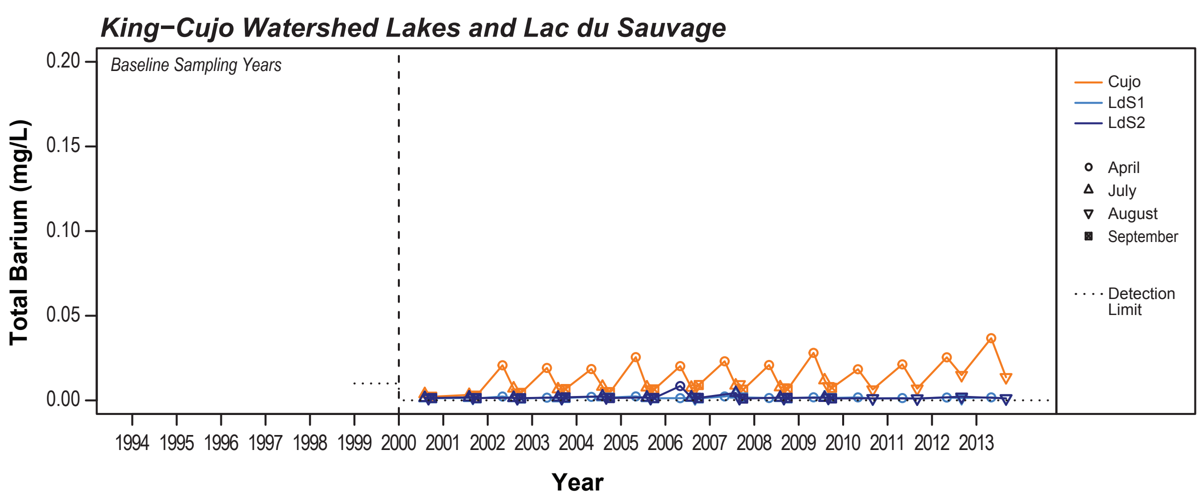
Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown. CCME Guideline = 0.005 mg/L.

Figure 5-25

Total Barium Concentrations at AEMP Lake Sites, 1994 to 2013

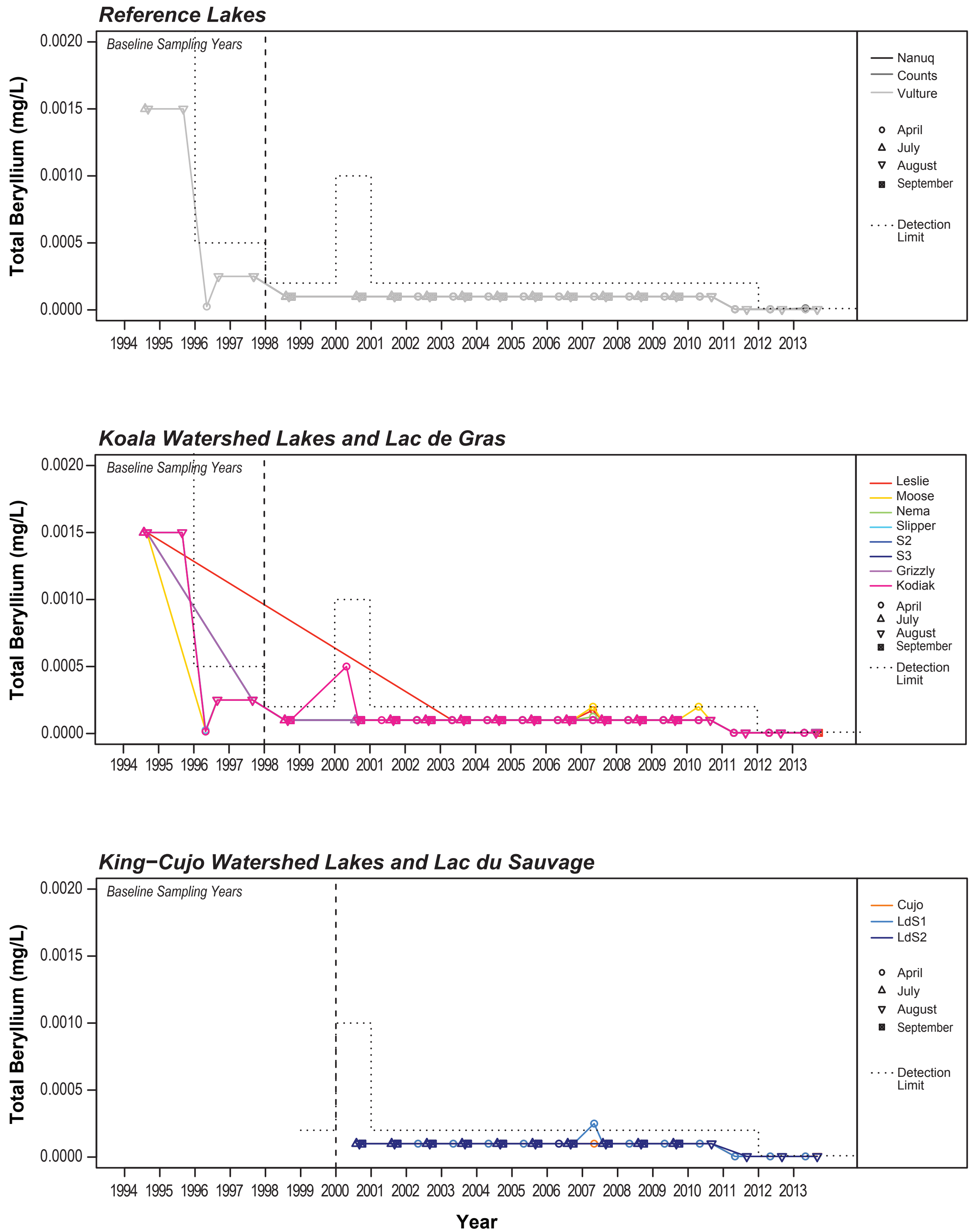






Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
Water quality benchmark (Haywood and Drinnan 1983) = 1 mg/L.

Figure 5-26
Total Beryllium Concentrations at AEMP Lake Sites, 1994 to 2013

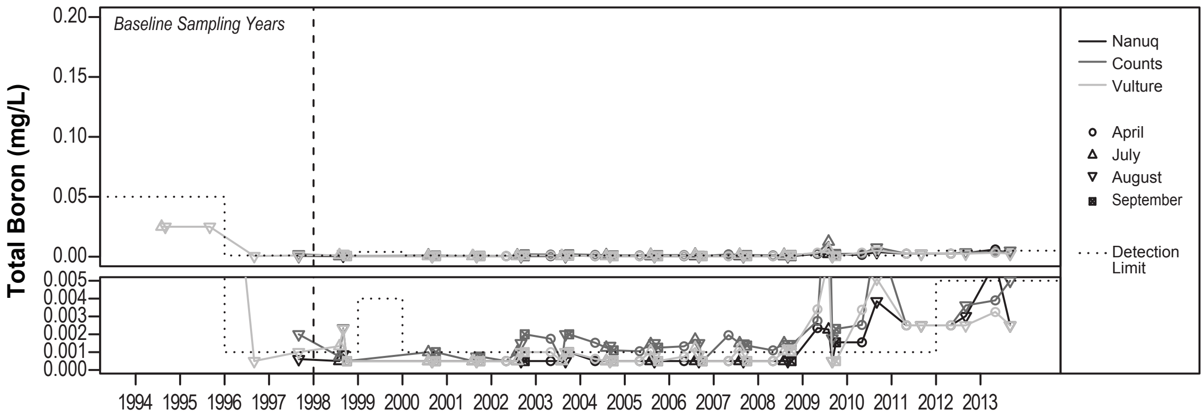


Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

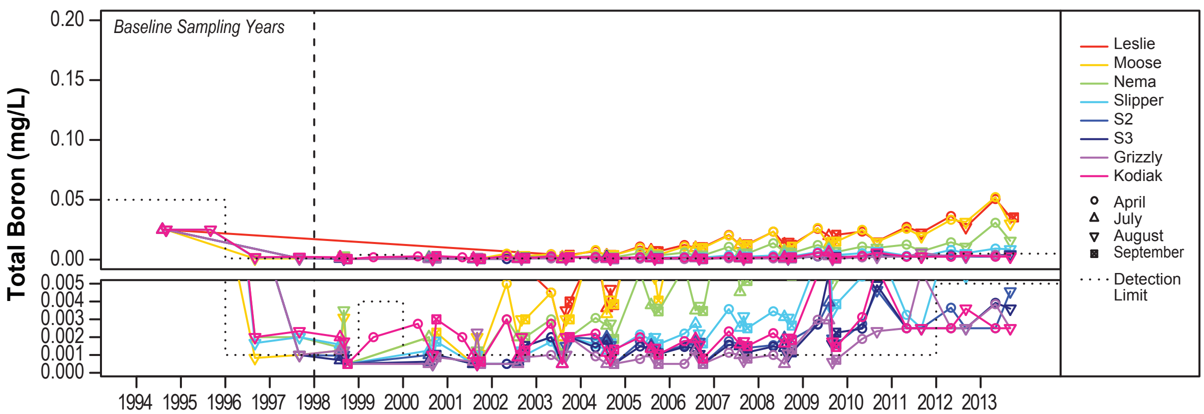
Figure 5-27

Total Boron Concentrations at AEMP Lake Sites, 1994 to 2013

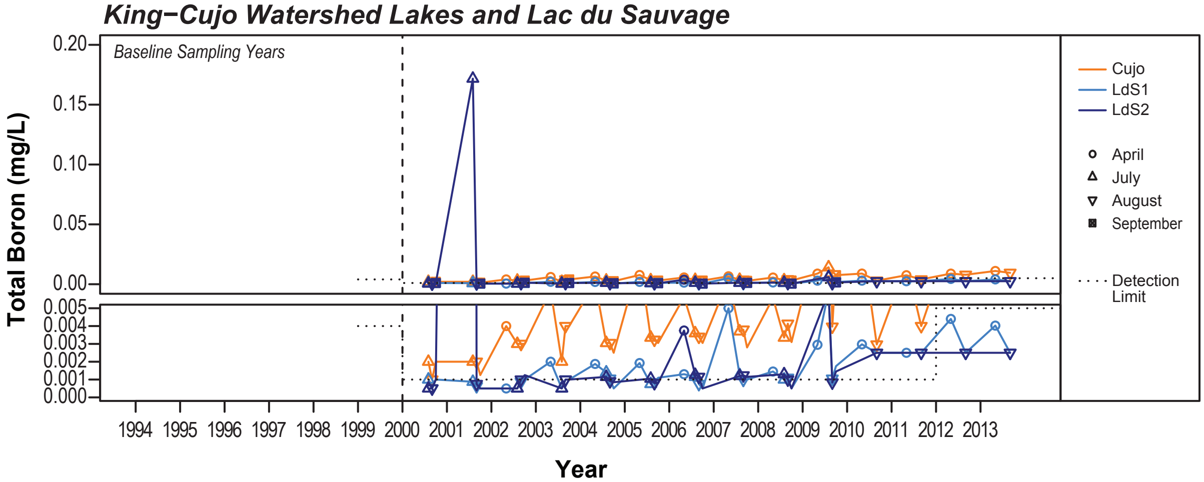
Reference Lakes



Koala Watershed Lakes and Lac de Gras



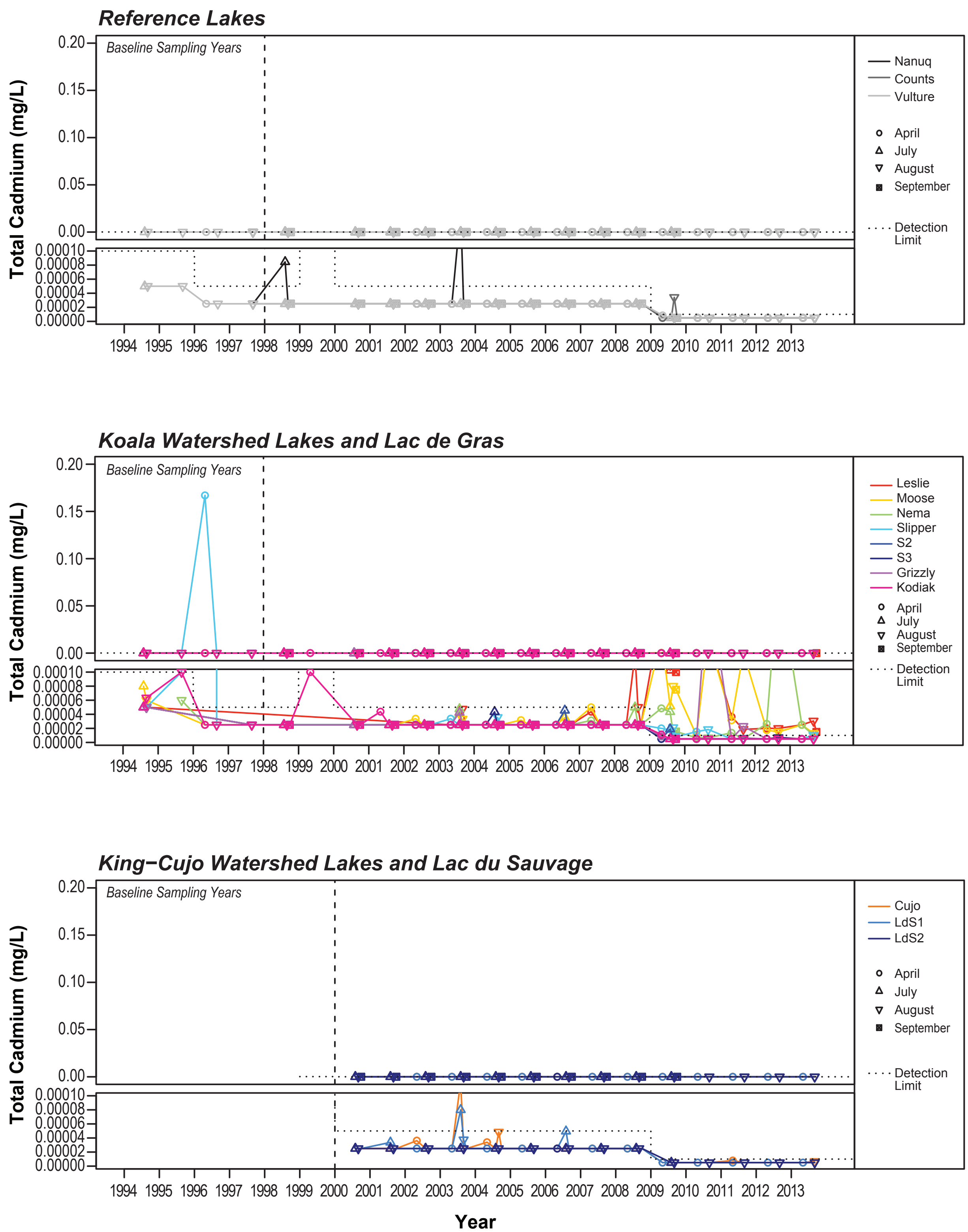
King−Cujo Watershed Lakes and Lac du Sauvage



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME Guideline = 1.5 mg/L.

Figure 5-28

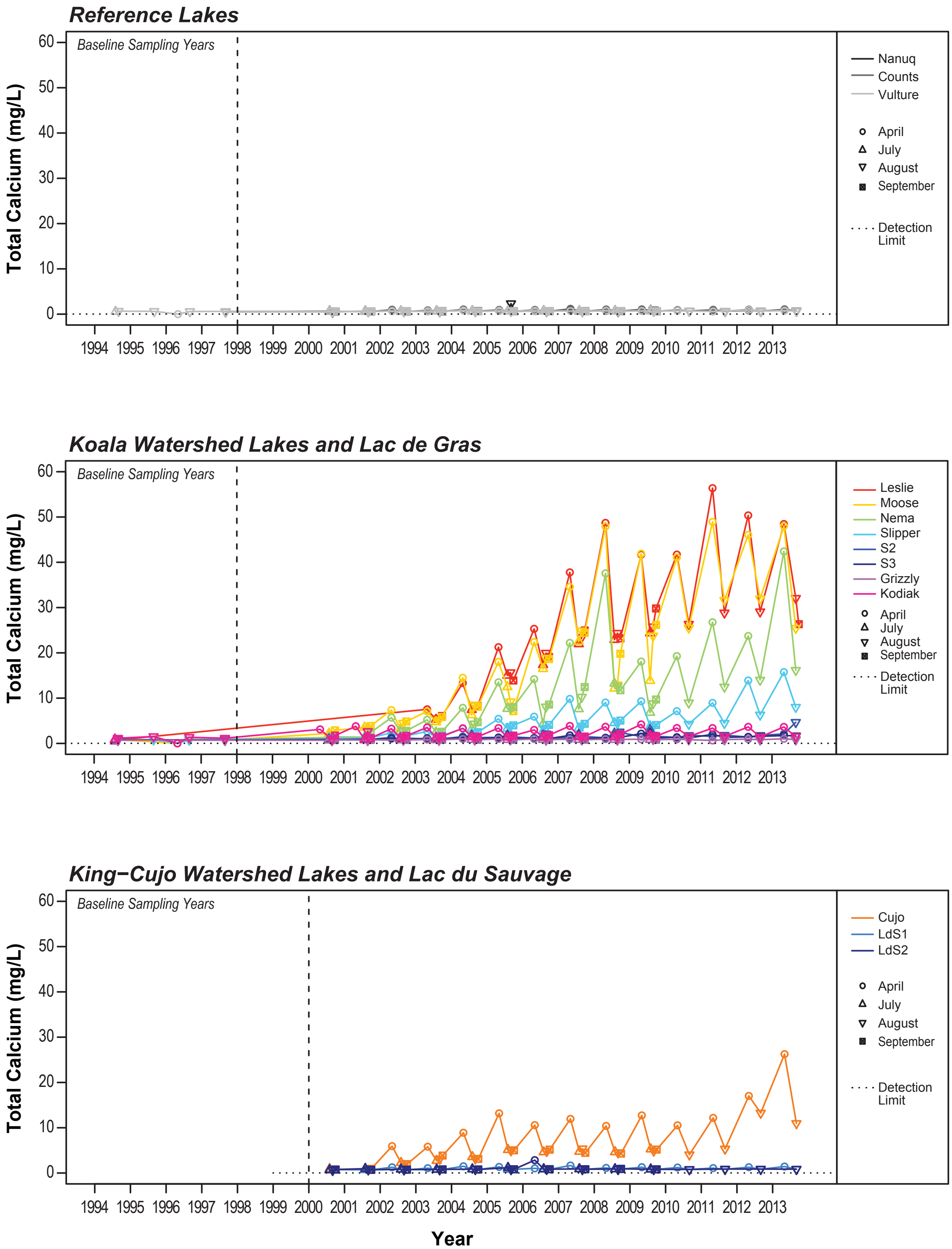
Total Cadmium Concentrations at AEMP Lake Sites, 1994 to 2013



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME Guideline = $10^{0.83 X (log10Harndess - 2.46)}$ /1,000 mg/L, with minimum = 0.00004 mg/L where hardness = 0-16 mg/L and maximum = 0.00037 mg/L where hardness > 280 mg/L.

Figure 5-29

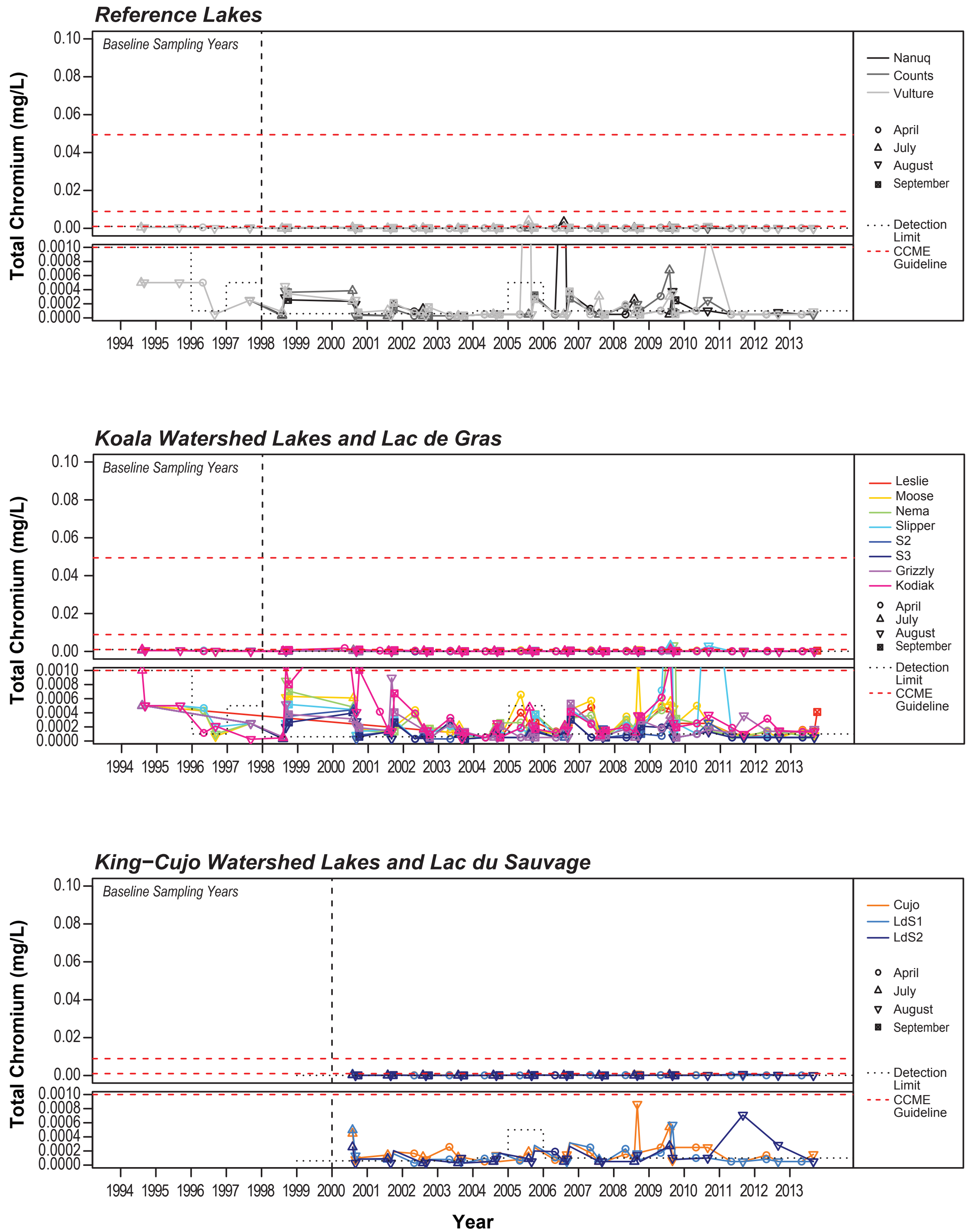
Total Calcium Concentrations at AEMP Lake Sites, 1994 to 2013



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 5-30

Total Chromium Concentrations at AEMP Lake Sites, 1994 to 2013

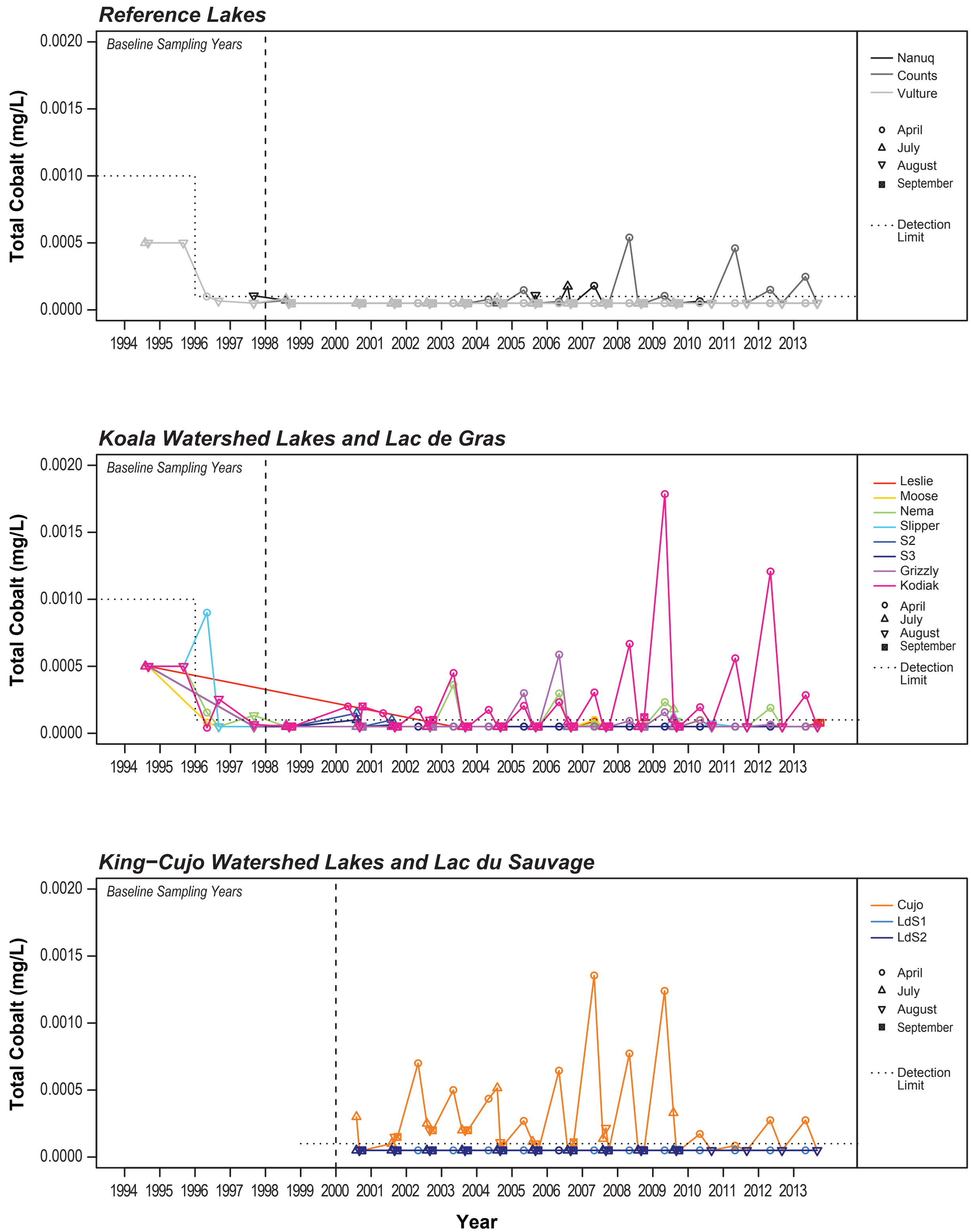


Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME Guideline = 0.001 mg/L (hexavalent CrVI); 0.0089 mg/L (trivalent CrIII).

Figure 5-31

Total Cobalt Concentrations at AEMP Lake Sites, 1994 to 2013

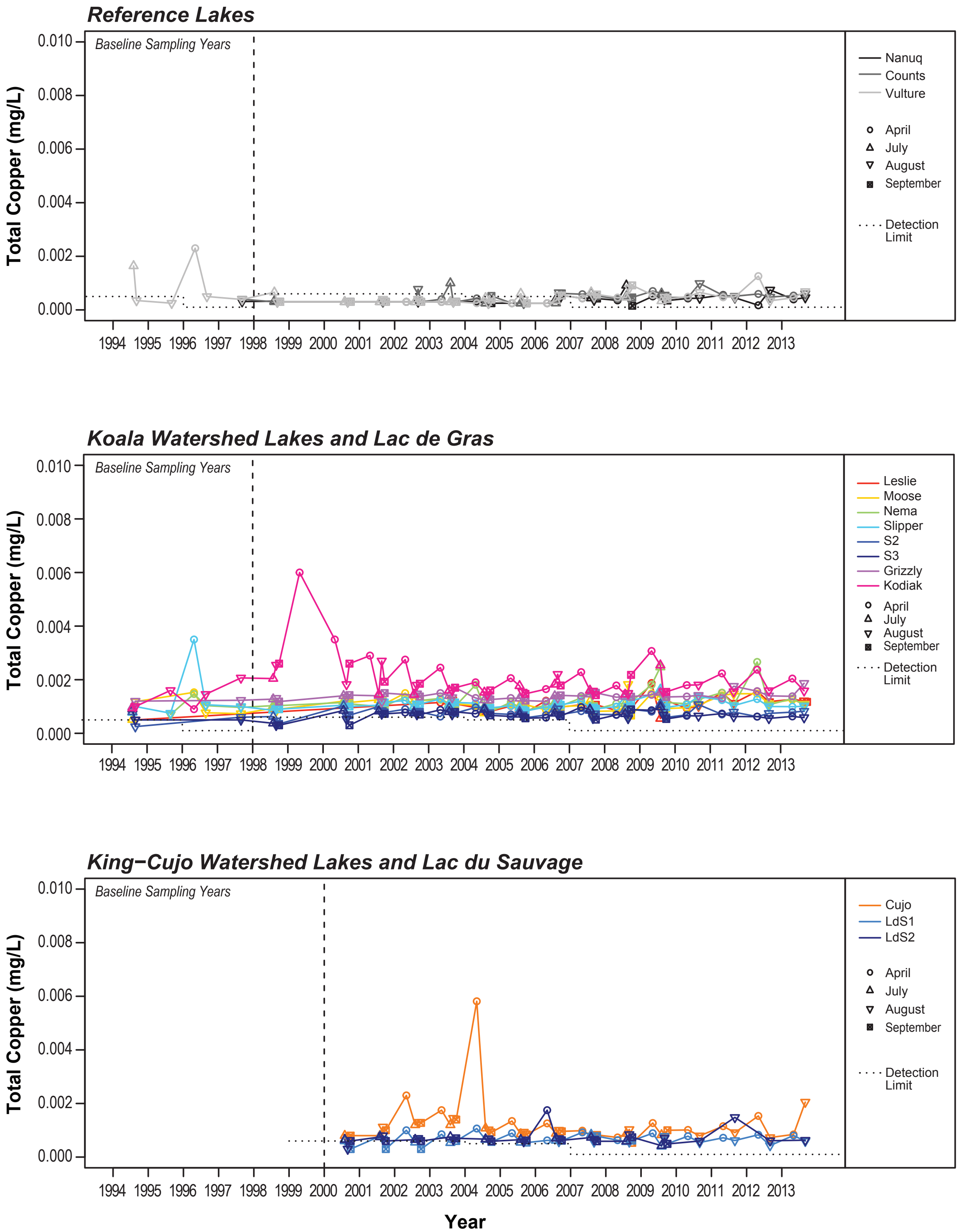
ERM



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 5-32

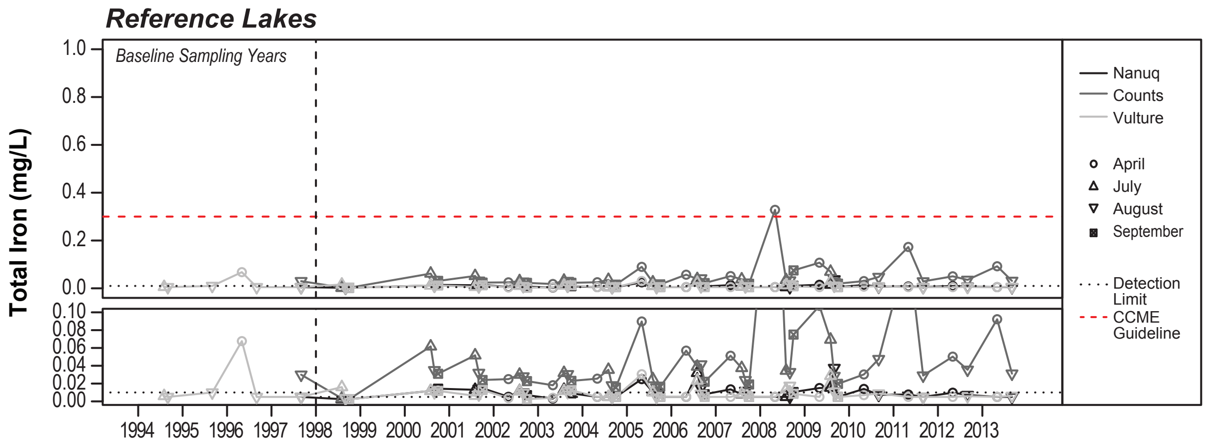
Total Copper Concentrations at AEMP Lake Sites, 1994 to 2013

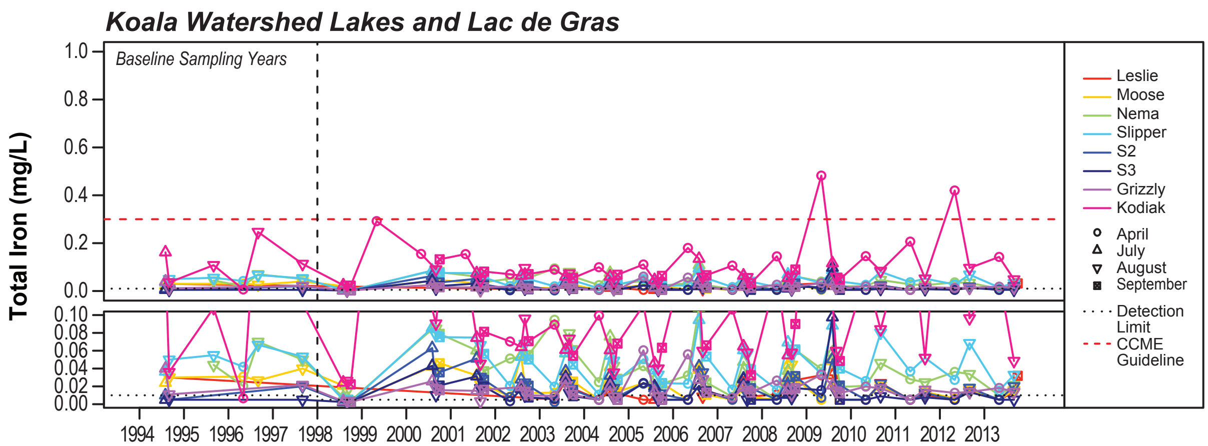


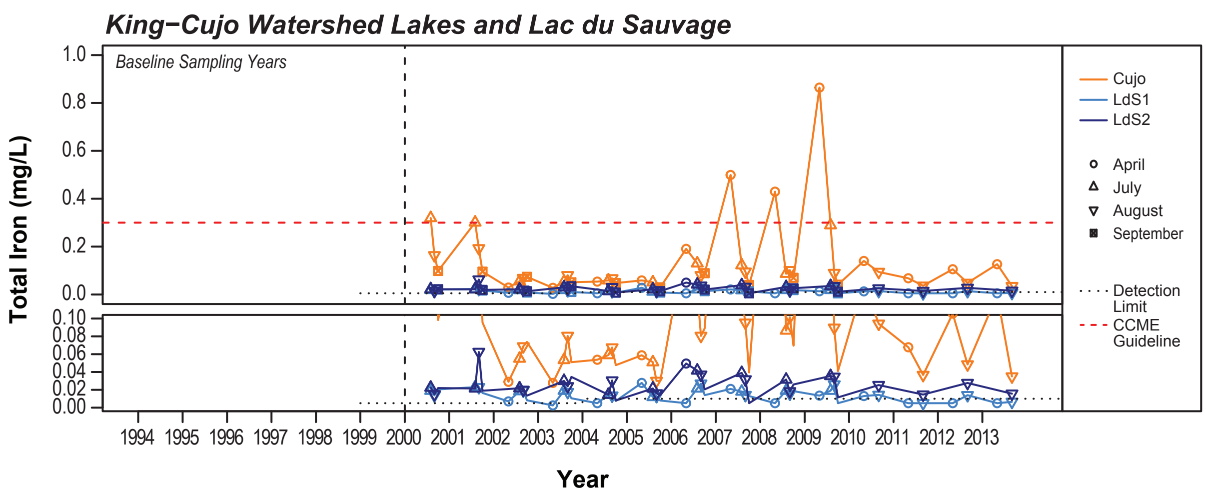
Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME Guideline = $e^{0.8545 X (\ln Hardness) - 1.465 * 0.2}/1000$ mg/L, where hardness < 180 mg/L
and 0.004 mg/L where hardness is ≥ to 180 mg/L . Minimum benchmark = 0.002 mg/L.

Figure 5-33

Total Iron Concentrations at AEMP Lake Sites, 1994 to 2013



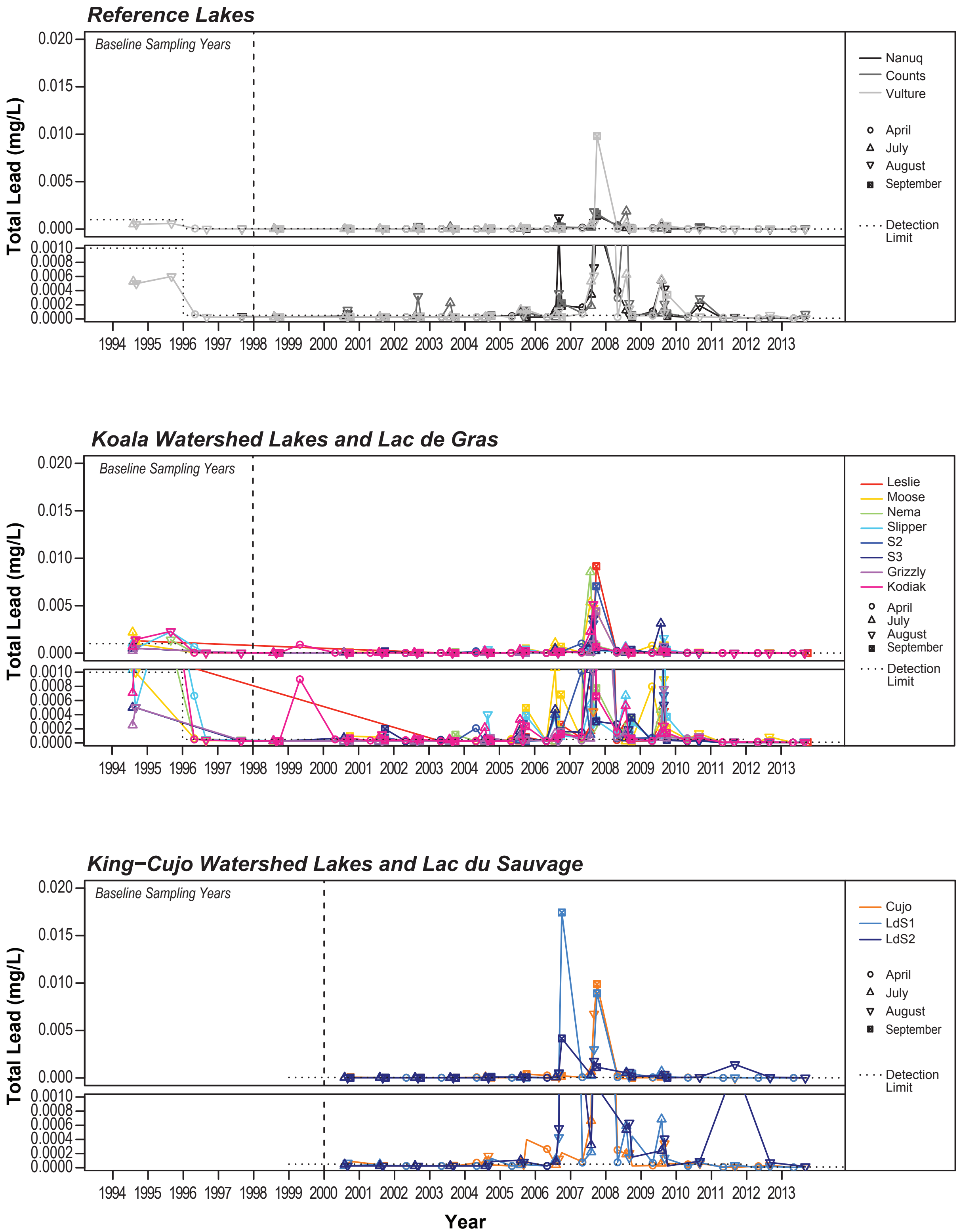




Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown. CCME Guideline = 0.3 mg/L.

Figure 5-34

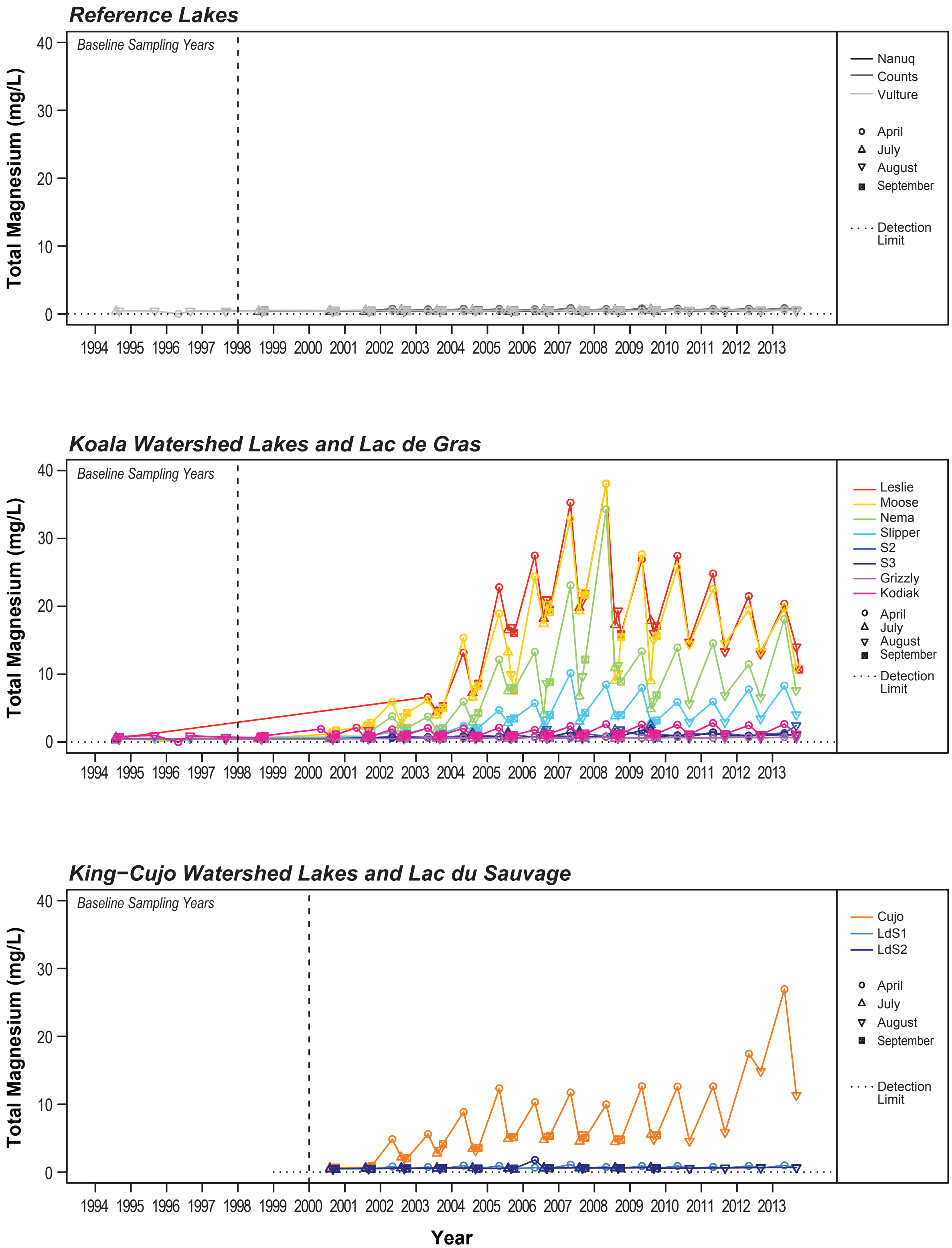
Total Lead Concentrations at AEMP Lake Sites, 1994 to 2013



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown. CCME Guideline = $e^{1.273 X (lnHardness) - 4.705}/1000$ mg/L, where hardness = 60 - 180 mg/L, 0.001 mg/L where hardness < 60 mg/L and 0.007 mg/L where hardness > 180 mg/L.

Figure 5-35

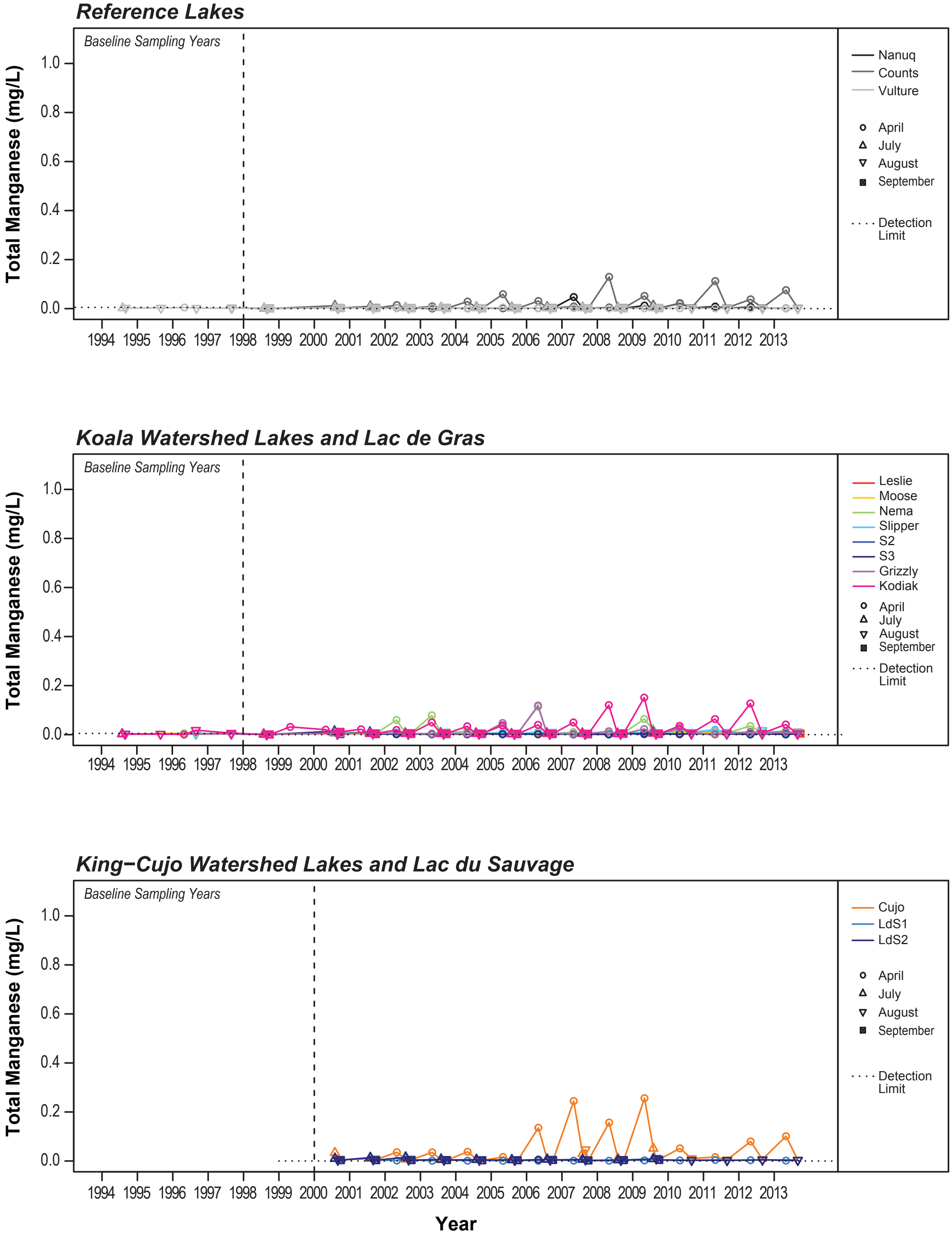
Total Magnesium Concentrations at AEMP Lake Sites, 1994 to 2013



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 5-36

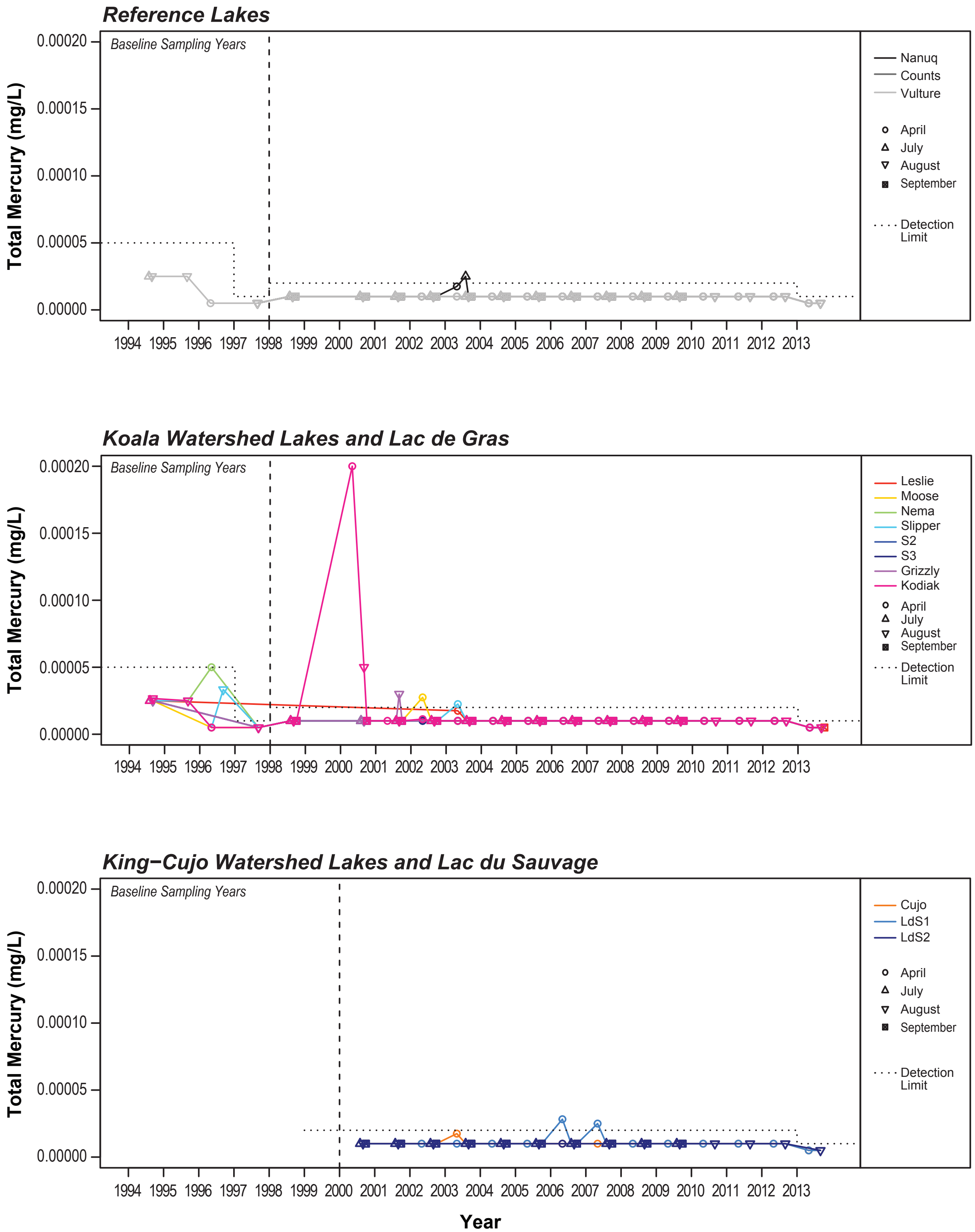
Total Manganese Concentrations at AEMP Lake Sites, 1994 to 2013



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 5-37

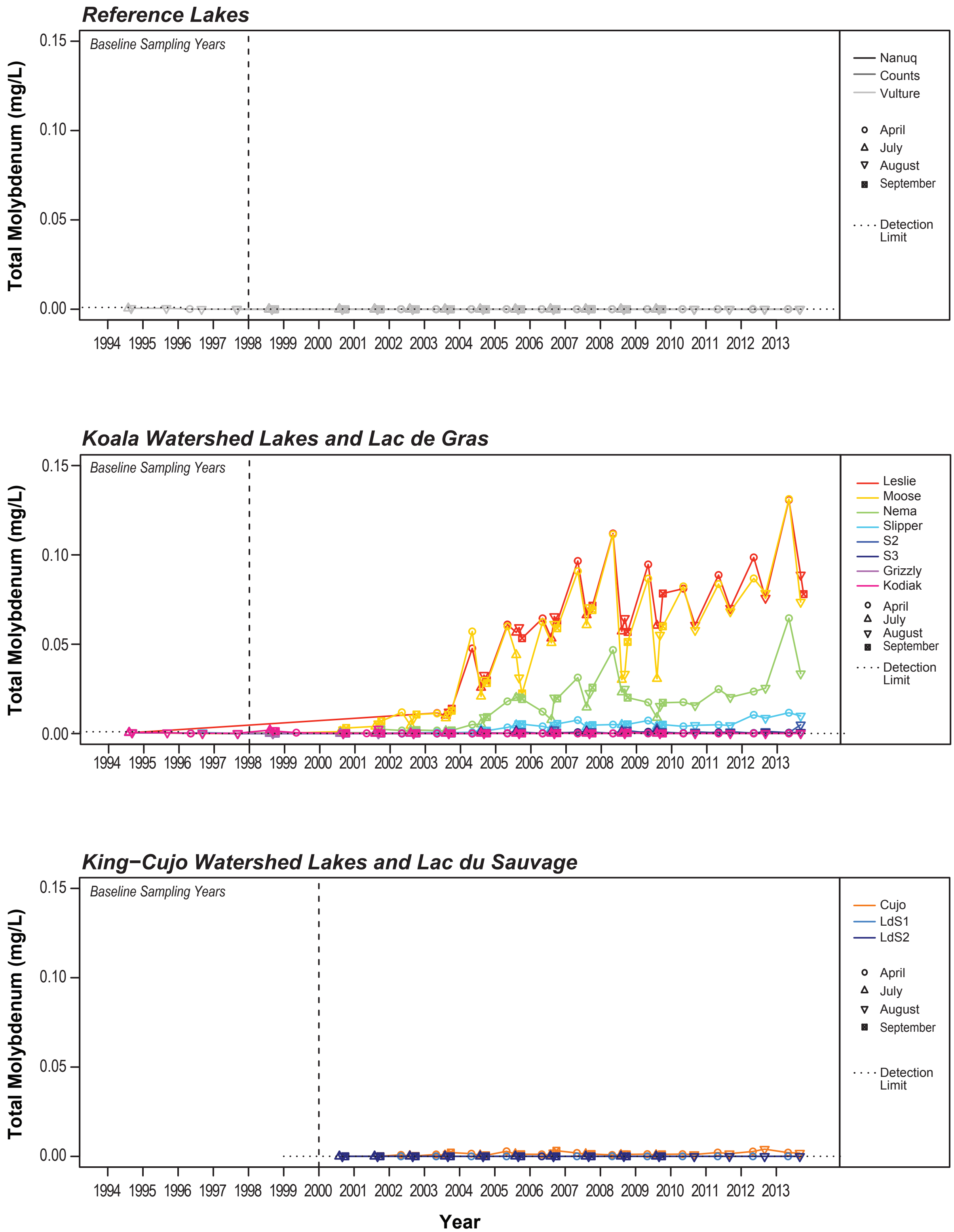
Total Mercury Concentrations at AEMP Lake Sites, 1994 to 2013



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 5-38

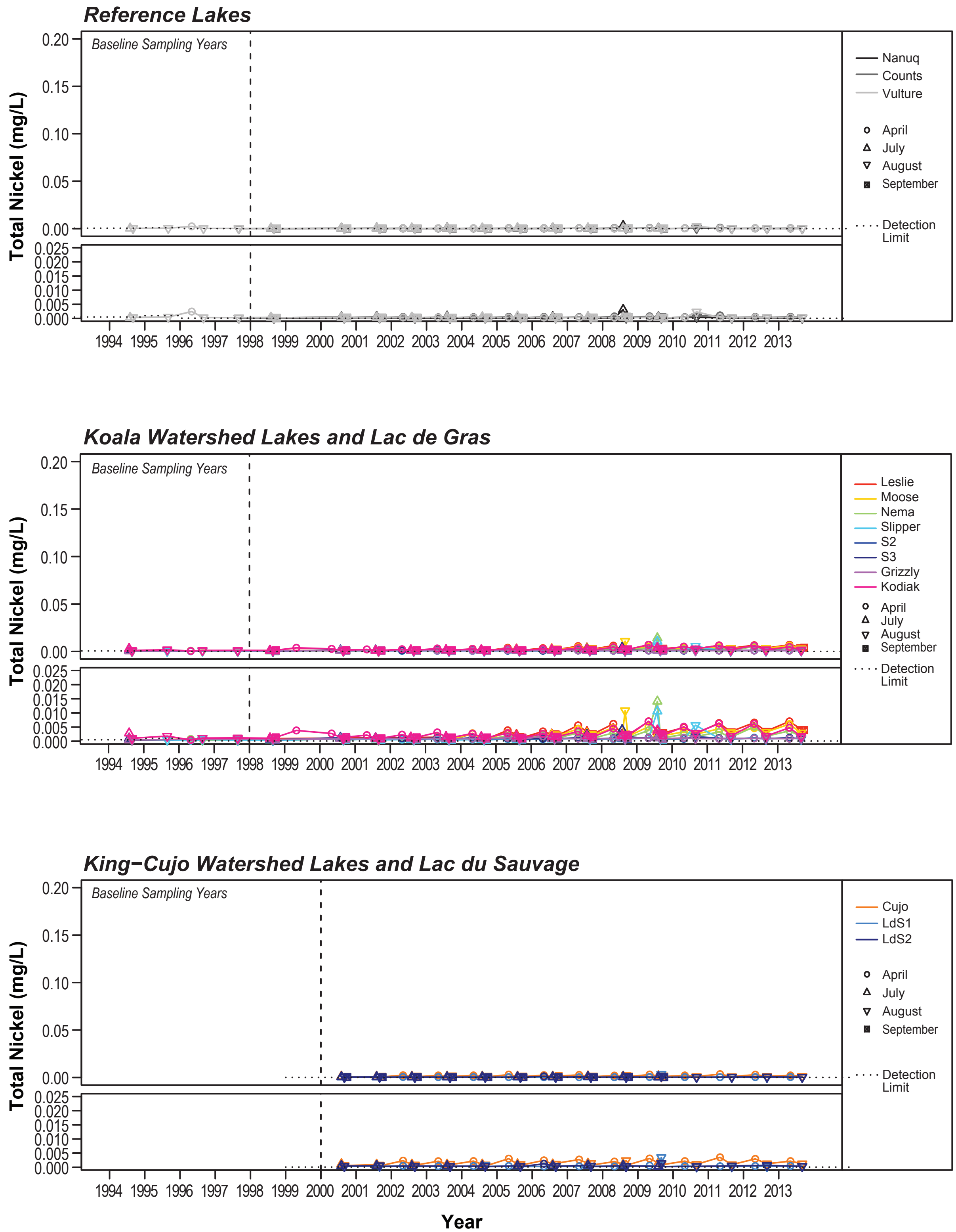
Total Molybdenum Concentrations at AEMP Lake Sites, 1994 to 2013



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
SSWQO = 19.38 mg/L.

Figure 5-39

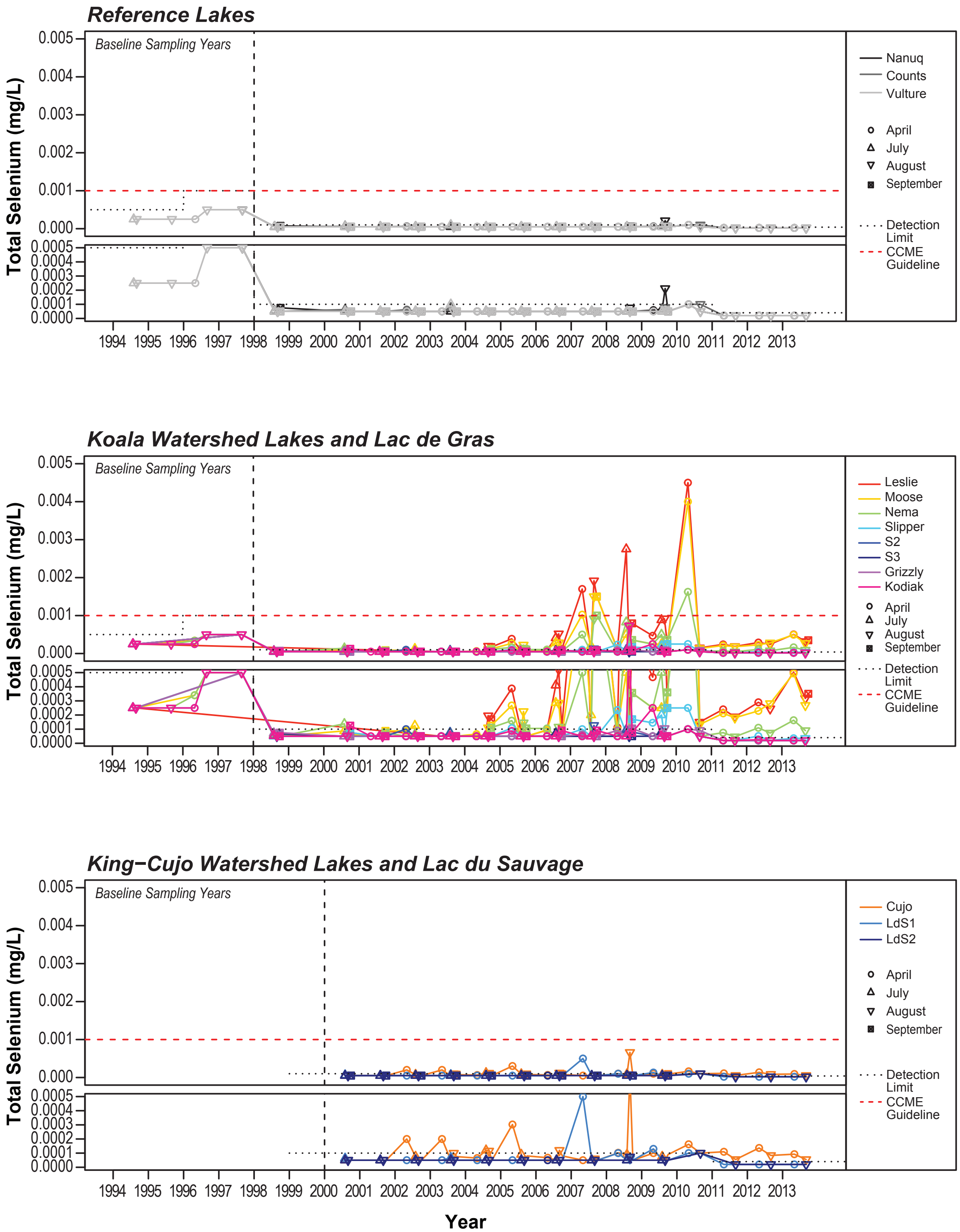
Total Nickel Concentrations at AEMP Lake Sites, 1994 to 2013



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown. CCME Guideline = $e^{0.76 X (\ln Hardness) + 1.06}/1000$ mg/L, where hardness = 60 - 180 mg/L, 0.025 mg/L where hardness < 60 mg/L, and 0.15 mg/L where hardness > 180 mg/L.

Figure 5-40

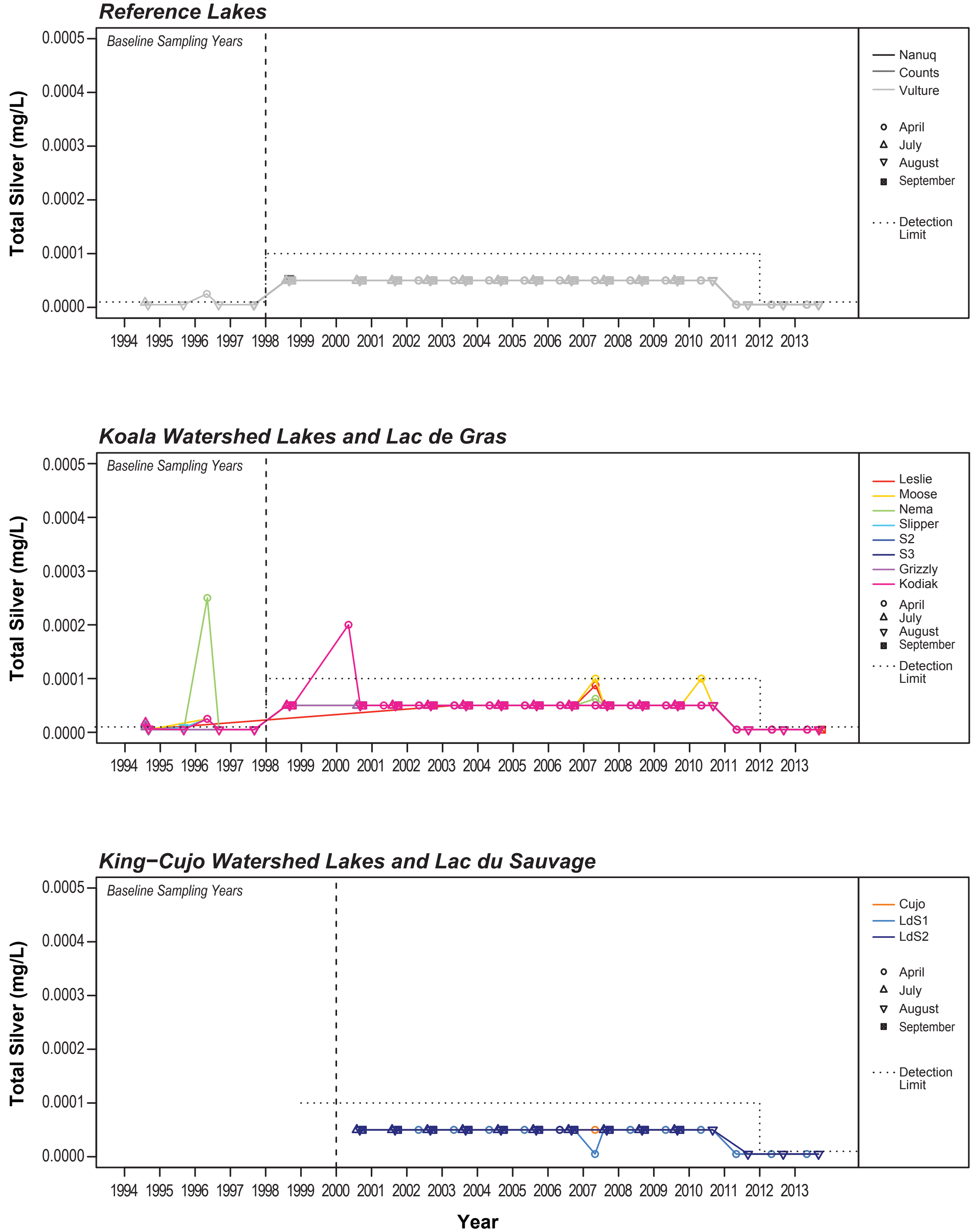
Total Selenium Concentrations at AEMP Lake Sites, 1994 to 2013



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME Guideline = 0.001 mg/L.

Figure 5-41

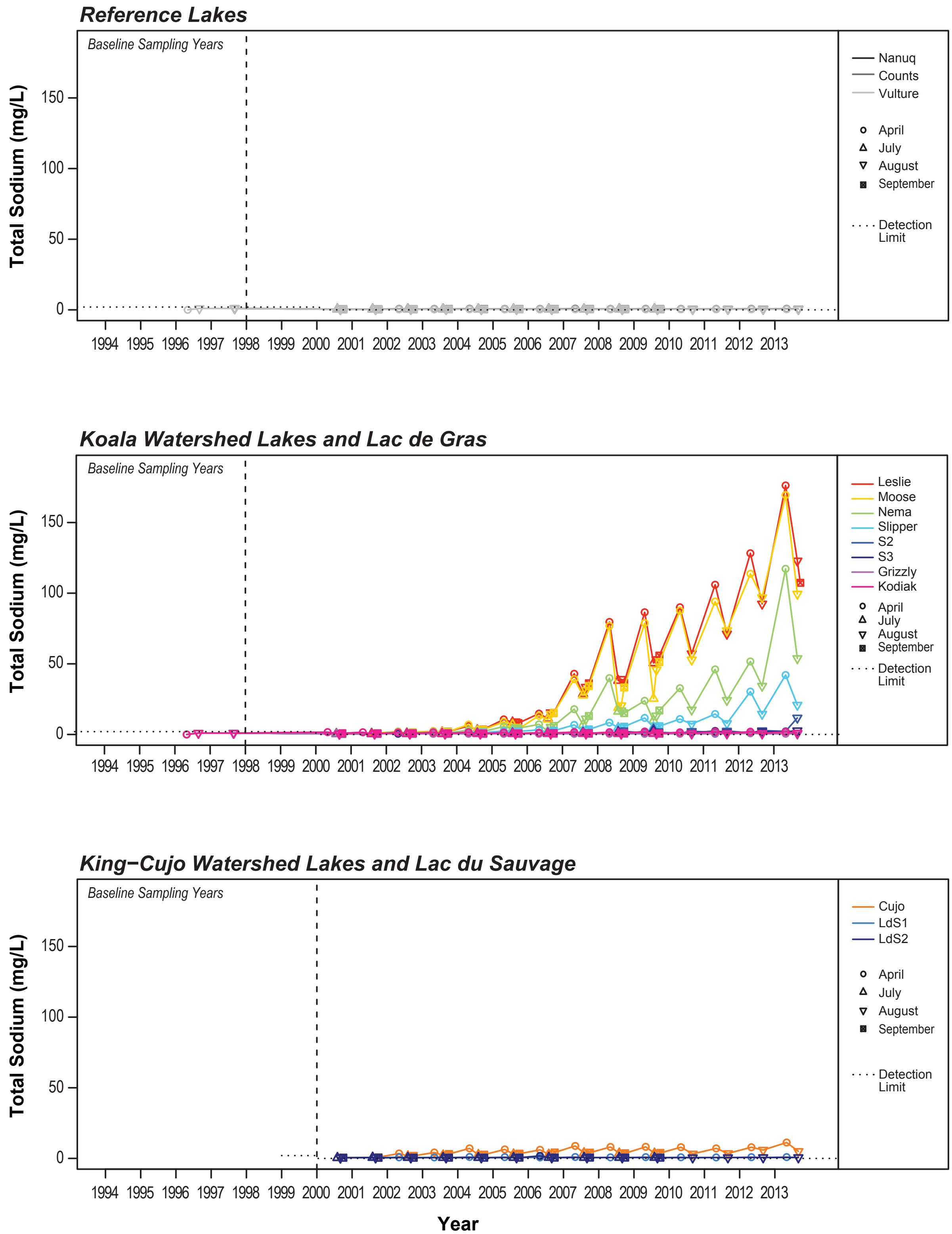
Total Silver Concentrations at AEMP Lake Sites, 1994 to 2013



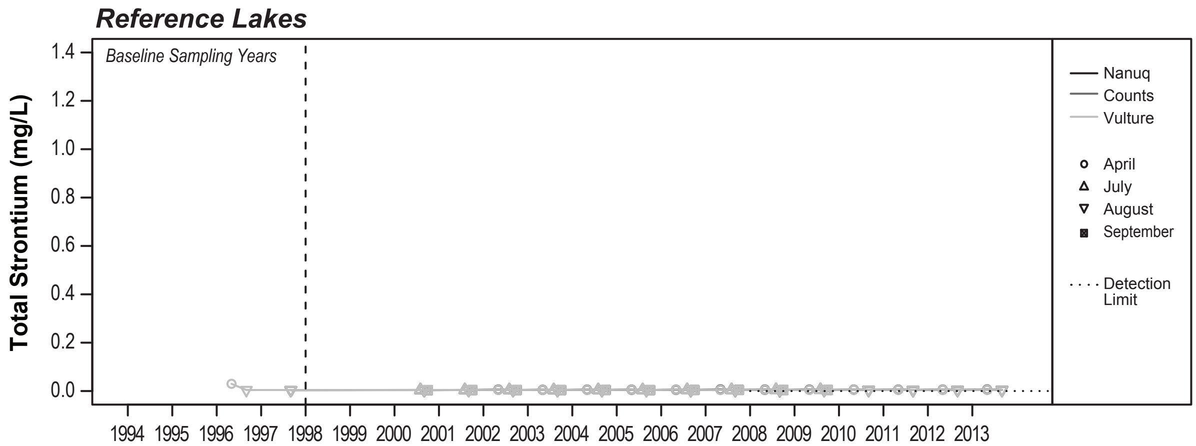
Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

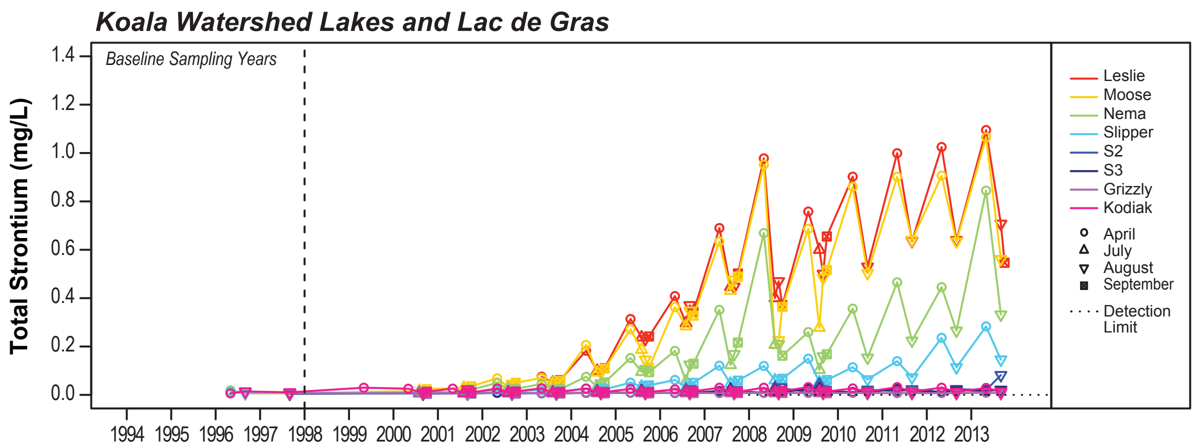
Figure 5-42

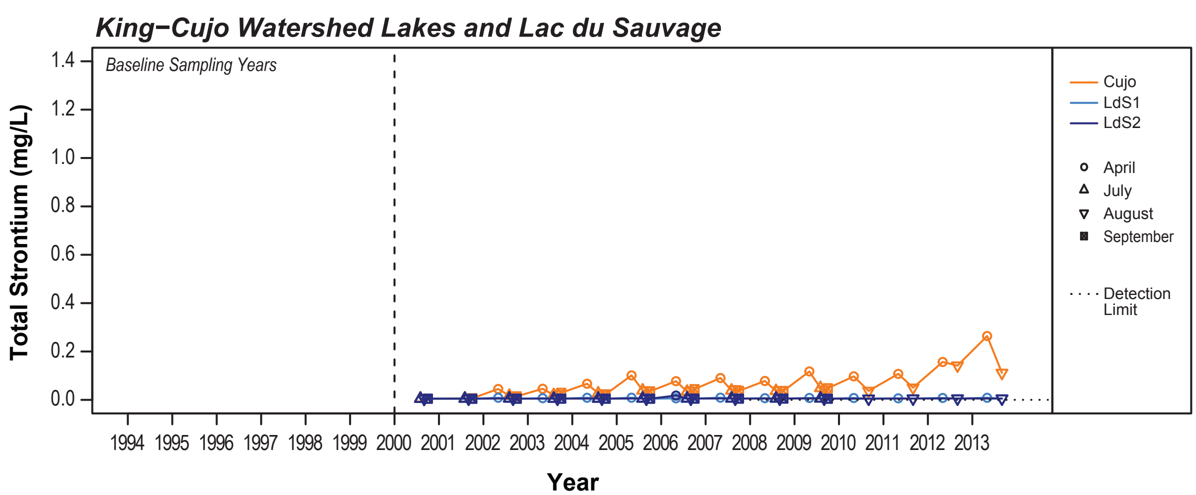
Total Sodium Concentrations at AEMP Lake Sites, 1994 to 2013



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

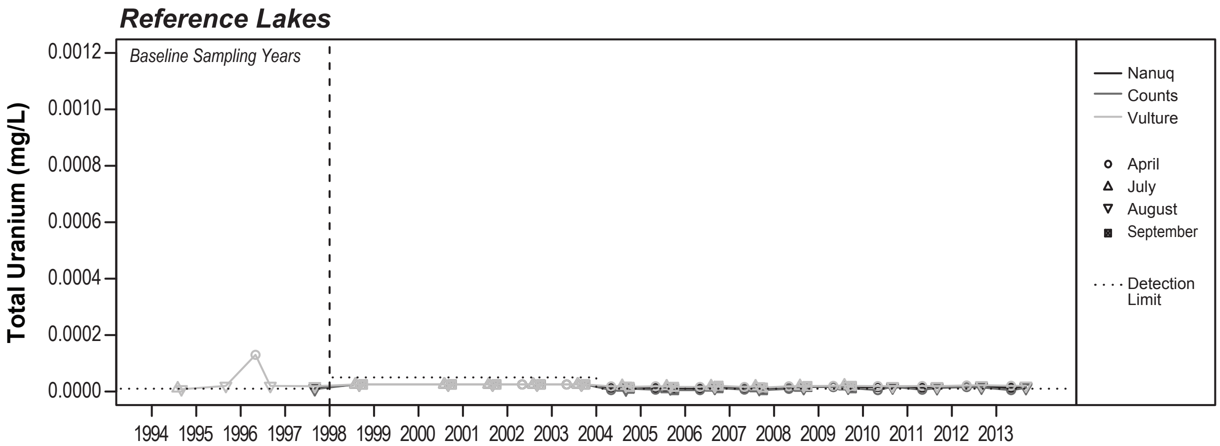


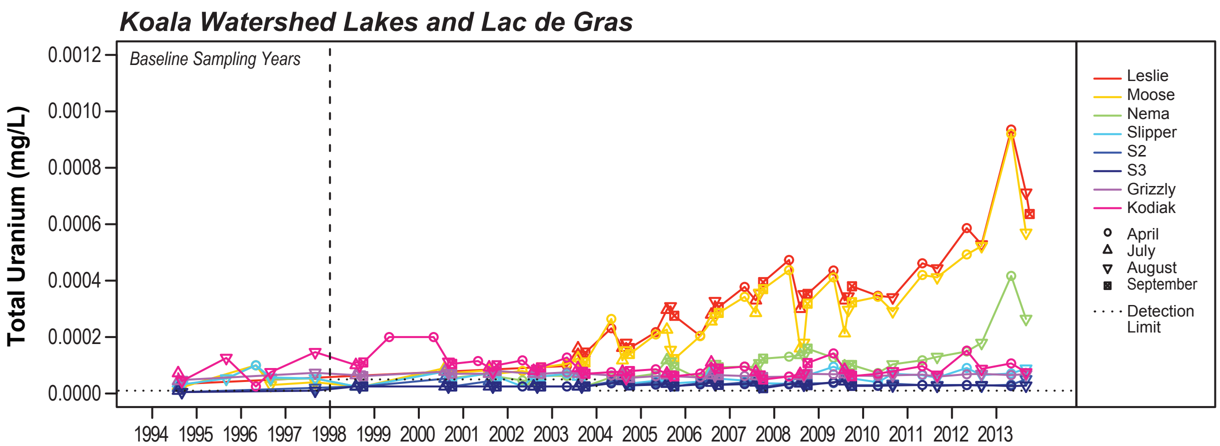


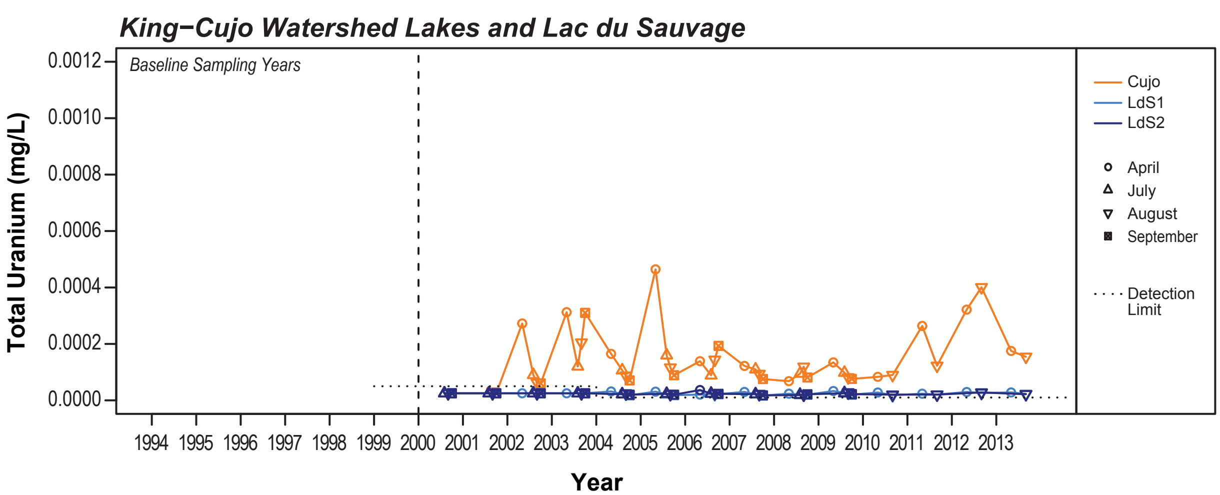


Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
Water quality benchmark (Golder 2011) = 6.242 mg/L.

Figure 5-44
Total Uranium Concentrations at AEMP Lake Sites, 1994 to 2013



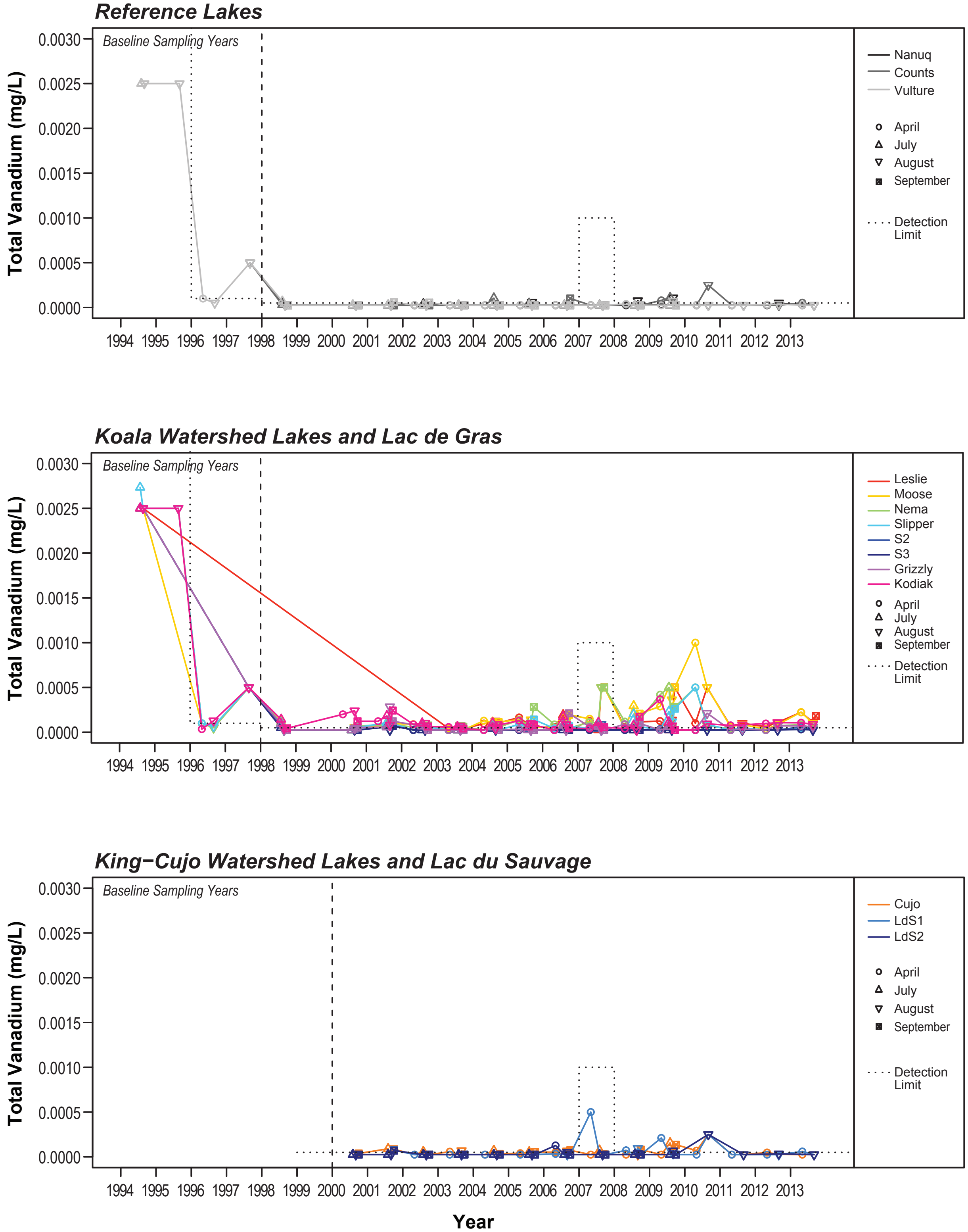




Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME Guideline = 0.015 mg/L.

Figure 5-45

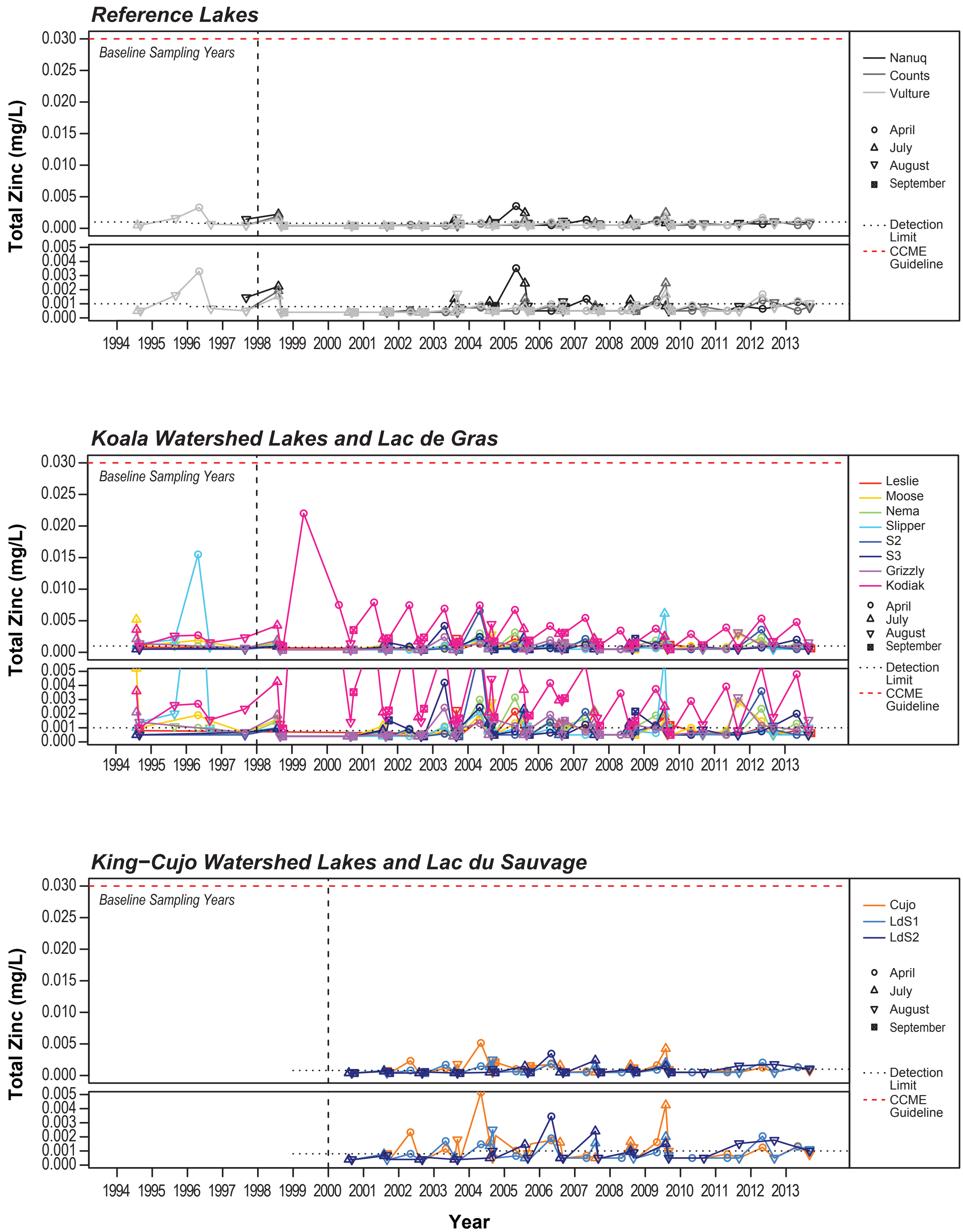
Total Vanadium Concentrations at AEMP Lake Sites, 1994 to 2013



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
SSWQO = 0.03 mg/L.

Figure 5-46

Total Zinc Concentrations at AEMP Lake Sites, 1994 to 2013



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME Guideline = 0.03 mg/L.

The 2013 values for hydrological variables (e.g., runoff depth) are presented in Part 2 - Data Report. Although the evaluation of effects does not include hydrological variables specifically, historical values of key hydrological variables are presented here. The location of each sampling station is provided in Table 5-2 and Figure 2.1-1 of this report. Hydrological variables for each AEMP monitored stream included for historical comparison were:

- minimum and maximum recorded unit yield ($L/s/km^2$; Tables 5-3 and 5-4);
- runoff Depth (mm) (Table 5-5);
- the computed Runoff Coefficients (Table 5-6); and
- comparison of 2013 daily flows with the historical record (Figures 5-47 to 5-53).

Table 5-2. AEMP Hydrometric Stations, 1994 to 2013

Station Number	WGS-14	WGS-39		WGS-02	WGS-24	WGL-46		WGS-35
Location	**Vulture-Polar**	**Lower PDC**	**Nema Outflow**	**Long Lake Outflow[1]**	**Slipper-Lac de Gras**	**Cujo Outflow**	**Christine Outflow**	**Counts Outflow**
Northing (m)	**7179565**	**7175900**	**7170646**	**7173110**	**7164913**	**7162100**	**7163840**	**7169713**
Easting (m)	**521484**	**518600**	**513921**	**514253**	**507616**	**539000**	**540025**	**535280**
Drainage Area (km^2)	**7**	**21**	**114**	**44**	**185**	**3**	**13**	**4**
1994					X			
1995				X	X			
1996				X	X			
1997	X				X			X
1998	X				X			X
1999	X	X			X	X		X
2000	X	X			X	X		X
2001	X	X			X	X		X
2002	X	X			X	X		X
2003	X	X			X	X		X
2004	X	X			X	X		X
2005	X	X			X	X		X
2006	X	X			X	X		X
2007	X	X			X	X		X
2008	X	X			X	X		X
2009	X	X			X	X		X
2010	X	X			X	X		X
2011	X	X			X	X		X
2012	X	X			X	X		X
2013	X	X	X		X	X	X	X

[1] *Flows from the Long Lake Containment Facility have been regulated since December of 1997.*

Table 5-3. Maximum Recorded Unit Yield (L/s/km^2) for AEMP Streams and Points of Regulated Discharge, 1995 to 2013

Year	Vulture-Polar		Lower PDC		Long Lake Outflow (1995, 1996) / LLCF Discharge (1998-2010)		Slipper-Lac de Gras		King Pond		Cujo Outflow		Nanuq Outflow		Counts Outflow	
1995	-		-		92	(Jun 14)	70.2	(Jun 10)	-		-		-		-	
1996	-		-		32	(Jun 16)	29.1	(Jun 14)	-		-		-		-	
1997	255.3	(Jul 3)	-		-		170.8	(Jun 3)	-		-		-		-	
1998	21.1	(May 22)	-		63.2	(Apr)	117.1	(May 23)	-		-		4.4	(May 31)	16.3	(May 22)
1999	273.2	(May 29)	166.1	(Jun 3)	29.8	(Jul)	103.4	(Jun 5)	-		135.9	(Jun 5)	13.9	(Jun 10)	55.2	(Jun 10)
2000	95	(Jun 11)	85.1	(May 27)	15.2	(Jul)	86.9	(Jun 12)	-		214.3	(Jun 11)	17.2	(Jun 22)	67.8	(Jun 22)
2001	185.5	(Jun 7)	78.7	(Jun 12)	24.3	(Jun)	371	(Jun 8)	-		77.4	(Jun 7)	18.2	(Jun 22)	120.1	(Jun 7)
2002	69.7	(Jun 5)	40.2	(Jun 8)	15.4	(Jul)	37.2	(Jun 11)	106.3	(Sep)	154	(Sep 20)	14.2	(July 4)	15.3	(Jun 26)
2003	28.8	(Jun 2)	32.4	(May 31)	-		45.7	(May 31)	-		59	(Jun 5)	-		44.3	(Jun 12)
2004	124.5	(Jun 6)	47.9	(Jun 8)	10.8	(Feb)	93.4	(Jun 10)	11.4	(Sep)	93.9	(Jun 10)	-		28.6	(Jun 20)
2005	64.1	(Jun 6)	110.6	(Jun 5)	26.1	(Jun)	107.3	(Jun 6)	15.2	(Sep)	93.1	(Jun 8)	-		44.4	(Jun 16)
2006	83.6	(May 26)	63.9	(May 16)[1]	23.6	(Jun)	79.5	(May 21)	76.8	(Jul)	50	(May 21)	-		39	(Jun 6)
2007	81.7	(Jun 1)	85.7	(Jun 2)	20.7	(Jul)	56.2	(Jun 4)	42.7	(Jul)	59	(Jun 4)	-		37.1	(Jun 16)
2008	32.7	(Jun 3)	55.8	(May 28)	2.7	(Sep)	31.4	(Jun 5)	34.5	(Jul)	22.1	(Jun 5)	-		21.8	(Jun 17)
2009	102.6	(Jun 10)	60.5	(Jun 11)	14.2	(Jul)	55.3	(Jun 13)	25.6	(Sep)	55.8	(Jun 13)	-		53.1	(Jun 20)
2010	54.1	(Jun 4)	86.7	(Jun 3)	23.7	(Jul)	33.9	(Jun 5)	74.3	(Aug)	50.5	(Aug 14)	-		35.2	(Jun 22)
2011	11.4	(Jun 5)	19.5	(Jun 3)	32	(Jul)	16.5	(Jun 4)	81.4	(Sep)	57.2	(Sep 10)	-		24	(Jun 20)
2012	48.5	(May 31)	76.2	(May 30)	26.2	(Jul)	54	(Jun 3)	119.5	(Jun)	76.2	(Jul 1)	-		54.3	(Jun 5)
2013	43.7	(Jun 3)	95.9	(Jun 1)	11.3	(Aug)	54.7	(Jun 3)	25.4	(Jul)	36.6	(May 31)	-		32.2	(Jun 8)

Table 5-4. Minimum Recorded Unit Yield (L/s/km^2) for AEMP Streams and Points of Regulated Discharge, 1995 to 2013

Year	Vulture-Polar		Lower PDC		Long Lake Outflow (1995, 1996) / LLCF Discharge (1998-2010)		Slipper-Lac de Gras		King Pond		Cujo Outflow		Nanuq Outflow		Counts Outflow	
1994	-		-		-		1.2	(Aug 15)	-		-		-		-	
1995	-		-		2.7	(Aug 6)	5.0	(Aug 27)	-		-		-		-	
1996	-		-		1	(Aug 7)	0.7	(Aug 7)	-		-		-		-	
1997	1.1	(Aug 20)	-		-		2.2	(Aug 22)	-		-		6.6	(Aug 21)	6.1	(Aug 19)
1998	0.9	(Aug 5)	-		0.0	(Jun)	2.2	(Aug 5)	-		-		2.7	(Aug 21)	0.7	(Aug 22)
1999	2.5	(Aug 16)	0.6	(Aug 16)	3.4	(Oct)	6.2	(Aug 22)	-		0.4	(Aug 17)	5.5	(Aug 25)	1.8	(Aug 22)
2000	2.0	(Aug 13)	1.9	(Aug 13)	4.3	(Sep)	5.1	(Aug 14)	-		0.1	(Aug 14)	7.1	(Aug 13)	-	
2001	2.9	(Aug 23)	0.8	(Aug 23)	6.3	(Aug)	0.7	(Aug 22)	-		1	(Aug 17)	-		9.8	(Aug 24)
2002	4.0	(Aug 9)	0.6	(Aug 9)	0	(Jun)	2.9	(Jul 3)	0.0	(Jul - Aug)	0.7	(Aug 6)	10	(Aug 22)	7.1	(Aug 9)
2003	1.0	(Aug 22)	0.2	(Aug 22)	3.9	(Jul)	1.7	(Aug 22)	0.6	(Jun)	4.2	(Jul 5)	-		4.5	(Aug 22)
2004	3.7	(Sep 3)	3.3	(Sep 3)	0.0	(Apr - Jun)	0.6	(Sep 3)	0.0	(Jan - Jun)	0.6	(Aug 25)	-		4.0	(Jun 15)
2005	4.3	(Aug 7)	1.1	(Aug 7)	0.0	(Jan - Jun)	1.6	(Aug 7)	0.0	(Jan - Aug)	0.7	(Aug 28)	-		0.2	(Aug 25)
2006	4.1	(Aug 22)	2.3	(Aug 22)	0.0	(Jan - May)	7.9	(Aug 22)	0.0	(Jan - Jul)	0.6	(Jul 16)	-		2.8	(Sep 18)
2007	0.7	(Sep 16)	0.8	(Sep 17)	0.0	(Jan - May)	3.1	(Sep 6)	0.0	(Jan - May)	0.5	(Aug 25)	-		1.2	(Sep 16)
2008	2.5	(Aug 13)	1.9	(Aug 10)	0.0	(Jan - Jul)	1.7	(Aug 13)	0.0	(Jan - May)	0.7	(Aug 13)	-		1.8	(Aug 13)
2009	1.4	(Sep 16)	1.2	(Sep 6)	0.0	(Jan - Jun)	1.3	(Sep 17)	0.0	(Jan - May)	0	(Aug 24)	-		1.9	(Sep 7)
2010	0.8	(Sep 13)	1.3	(Sep 13)	0.0	(Jan - Jun)	1.7	(Aug 22)	0.0	(Jan - Jun)	5.3	(Sep 13)	-		1.1	(Jun 10)
2011	2.9	(Aug 15)	1.2	(Aug 14)	0.0	(Jan - Jun)	3.6	(Aug 15)	0.0	(Jan - Jul)	0	(Aug 30)	-		3.0	(Aug 15)
2012	3.4	(Aug 26)	1.4	(Aug 26)	0.0	(Jan - May)	2.0	(Aug 26)	0.0	(Jan - May)	0	(Aug 3)	-		0	(Sep 10)
2013	4.1	(Sep 6)	1.5	(Sep 7)	0.0	(Jan - May)	5.2	(Sep 4)	0.0	(Jan - Jun)	0.4	(Sep 8)	-		0.0	(Jun 3)

Table 5-5. Runoff Depth (mm) for AEMP Streams Recorded from 1995 to 2013

	Vulture-Polar	Lower PDC	Long Lake Outflow	LLCF	Slipper-Lac de Gras	King Pond	Cujo Outflow	Counts Outflow
1995	-	-	*161*	-	123	-	-	-
1996	-	-	-	-	77	-	-	-
1997	182	-	-	-	195	-	-	7
1998	62	-	-	*327*	63	-	-	17
1999	337	215	-	*410*	*298*	-	141	95
2000	*170*	*169*	-	*166*	*180*	-	*160*	108
2001	169	75	-	204	194	-	137	225
2002	*135*	*59*	-	*71*	*84*[1]	414	*261*	*122*
2003	*62*	*55*	-	-	*88*	-	*198*	*138*
2004	*170*	*80*	-	-	*85*	-	*202*	*116*
2005	*147*	*171*	-	-	*113*	-	*111*	*130*
2006	*259*	*254*	-	-	*268*	-	*176*	*124*
2007	*156*	*106*	-	*228*[2]	*88*	*144*	*94*	*117*
2008	*122*	*153*	-	*55*	*130*	*9*	*73*	*92*
2009	*133*	*121*	-	*173*	*96*	*287*	*112*	*133*
2010	*97*	*137*	-	*154*	*107*	*483*	*180*	*122*
2011	*96*	*93*	-	*241*	*93*	*194*	*95*	*97*
2012	*147*	*137*	-	*215*	*138*	*455*	*172*	*145*
2013	*147*	*135*	-	*161*	*132*	*68*	*77*	*114*

Notes:
Runoff depths in italics are from the entire open water season.
[1] *The runoff depth for Slipper-LdG includes the influence of Fox Lake dewatering (~3 mm) during the 2002 hydrologic year.*
[2] *Runoff from October 2006 to September 2007.*

Table 5-6. Runoff Coefficients Computed for AEMP Streams, 1999 to 2013

Year	Total Precipitation (mm)	Vulture-Polar		Lower PDC		Slipper-Lac de Gras		Cujo Outflow		Counts Outflow	
		Runoff Depth (mm)	Runoff Coefficient	Runoff Depth (mm)	Runoff Coefficient	Runoff Depth (mm)	Runoff Coefficient	Runoff Depth (mm)	Runoff Coefficient	Runoff Depth (mm)	Runoff Coefficient
1999	458	n/a	n/a	n/a	n/a	298	0.65	n/a	n/a	n/a	n/a
2000	279	170	0.61	169	0.61	180	0.65	160	0.57	n/a	n/a
2001	336	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
2002	321	80	0.42	51	0.18	115	0.26	62	0.81	84	0.38
2003	288	62	0.21	55	0.19	88	0.3	198	0.69	138	0.48
2004	222	170	0.76	64	0.36	85	0.38	194	0.91	116	0.52
2005	248	147	0.59	171	0.69	113	0.45	111	0.45	130	0.52
2006	430	259	0.6	254	0.59	268	0.62	176	0.41	124	0.29
2007	257	156	0.61	106	0.41	88	0.34	94	0.37	117	0.46
2008	325	122	0.38	153	0.47	130	0.4	73	0.23	92	0.28
2009	251	133	0.53	121	0.48	96	0.38	112	0.45	112	0.45
2010	283	97	0.34	137	0.48	107	0.38	180	0.64	122	0.43
2011	384	96	0.25	93	0.24	93	0.24	95	0.25	97	0.25
2012	505	147	0.29	137	0.27	138	0.27	172	0.34	145	0.29
2013	370	147	0.40	135	0.37	132	0.36	77	0.21	114	0.31

Figure 5-47

Comparison of 2013 Daily Flow at Vulture-Polar with the Historical Record (1997 to 2013)

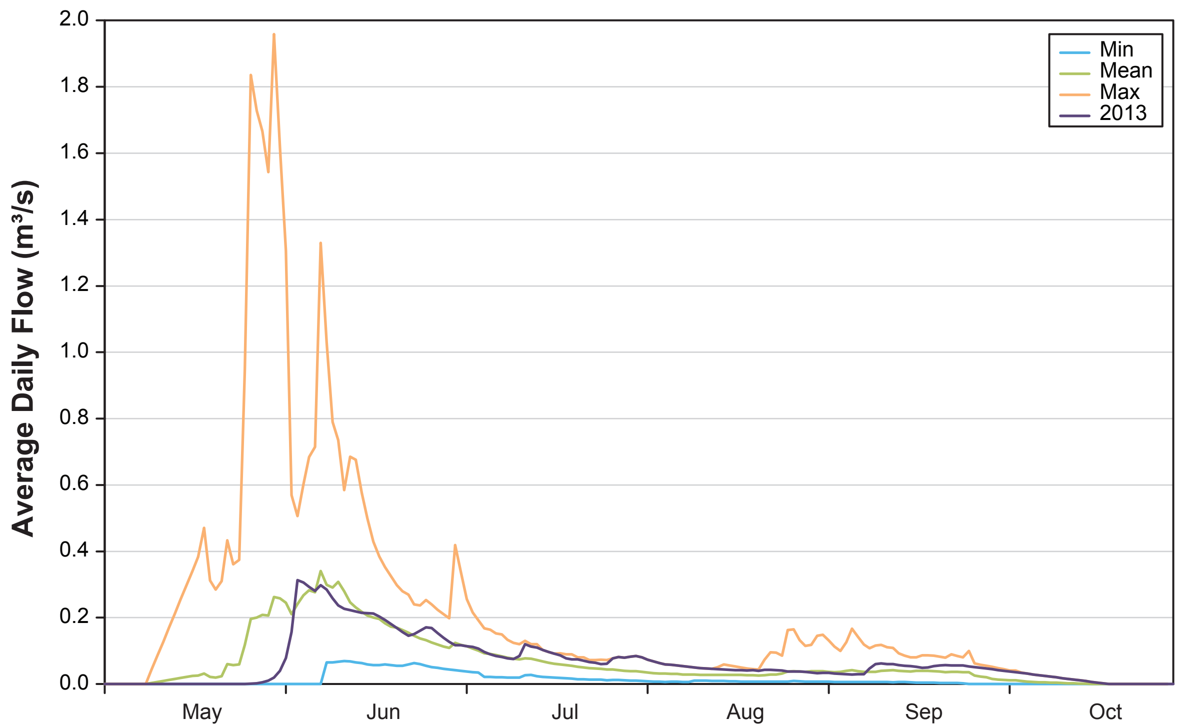


Figure 5-48

Comparison of 2013 Daily Flow at Lower PDC with the Historical Record (1999 to 2013)

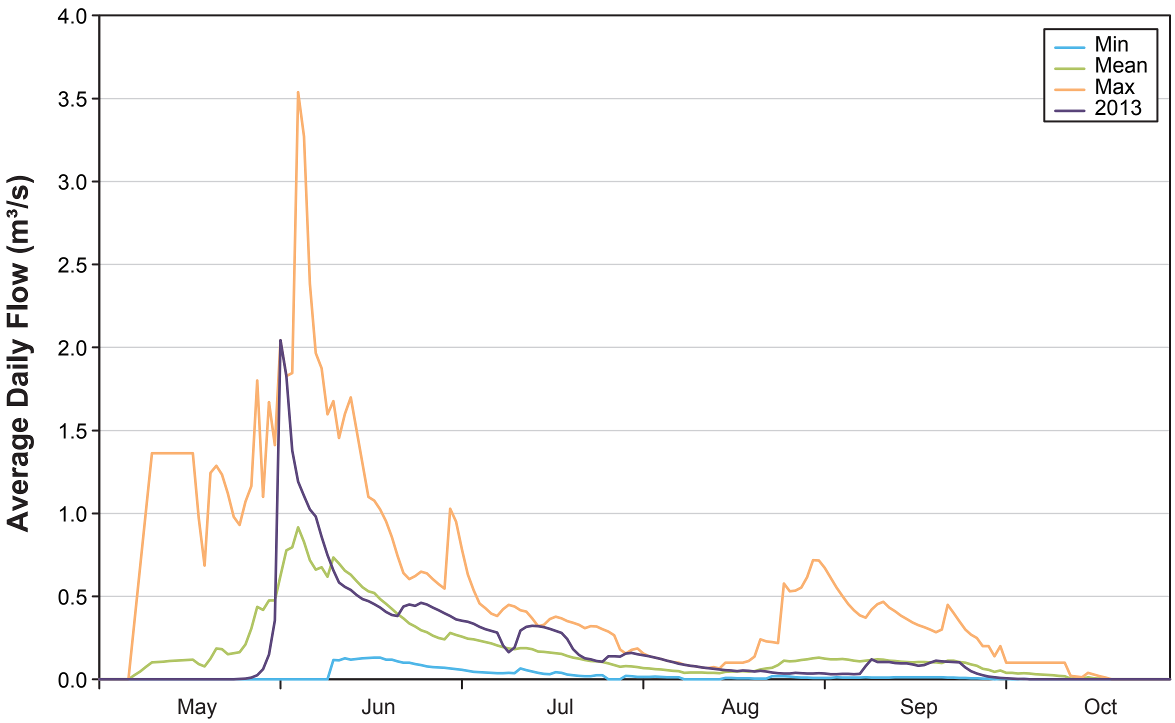


Figure 5-49

Comparison of 2012 to 2013 Daily Flow at LLCF (1616-30) with the Historical Record (2000 to 2013)

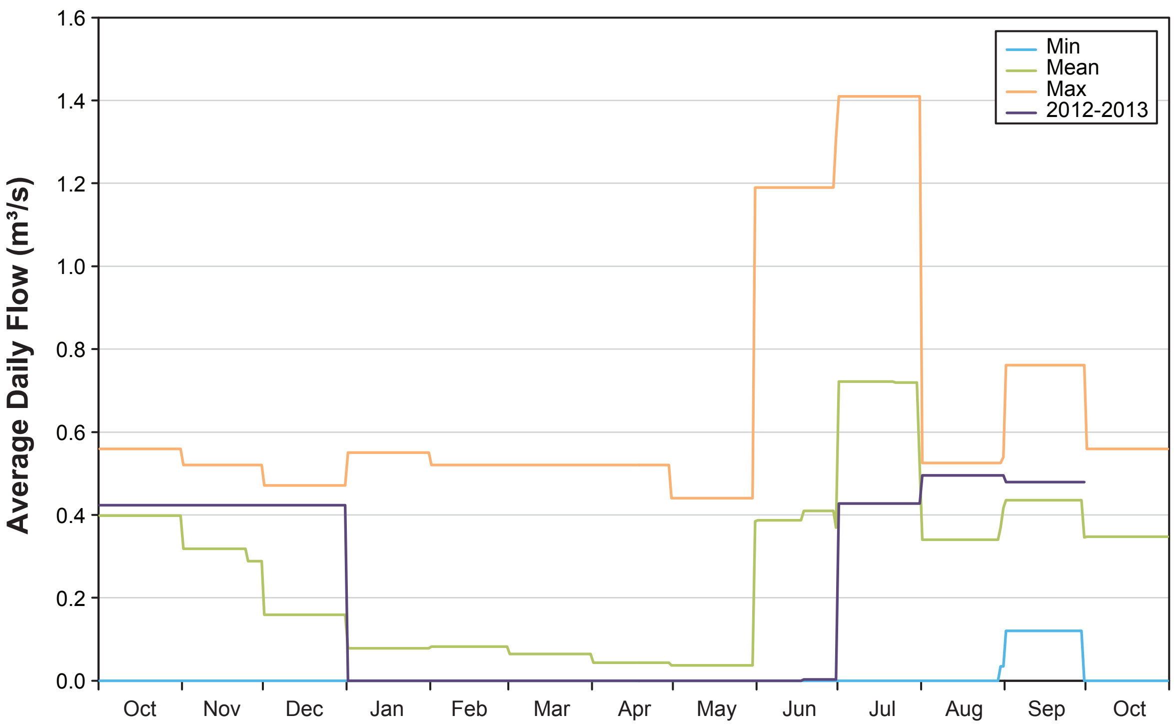


Figure 5-50

Comparison of 2013 Daily Flow at Slipper-Lac de Gras with the Historical Record (1994 to 2013)

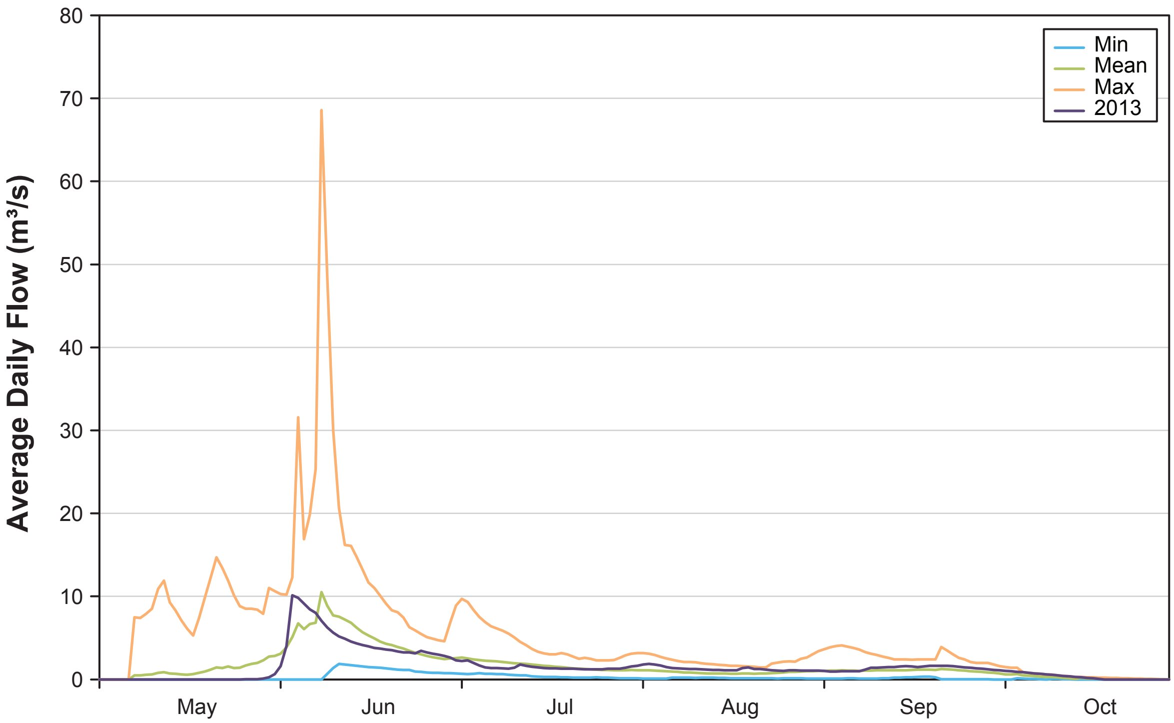


Figure 5-51

Comparison of 2012 to 2013 Daily Flow at KPSF (1616-43) with the Historical Record (2000 to 2013)

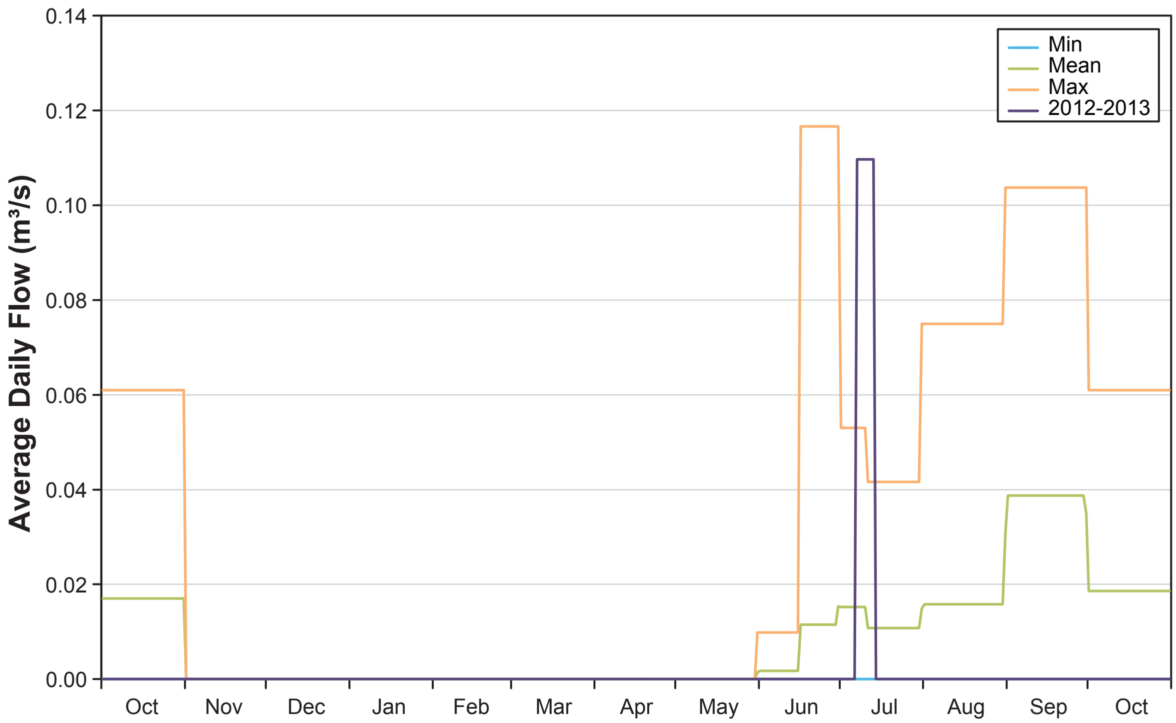


Figure 5-52

Comparison of 2013 Daily Flow at Cujo Outflow with the Historical Record (1999 to 2013)

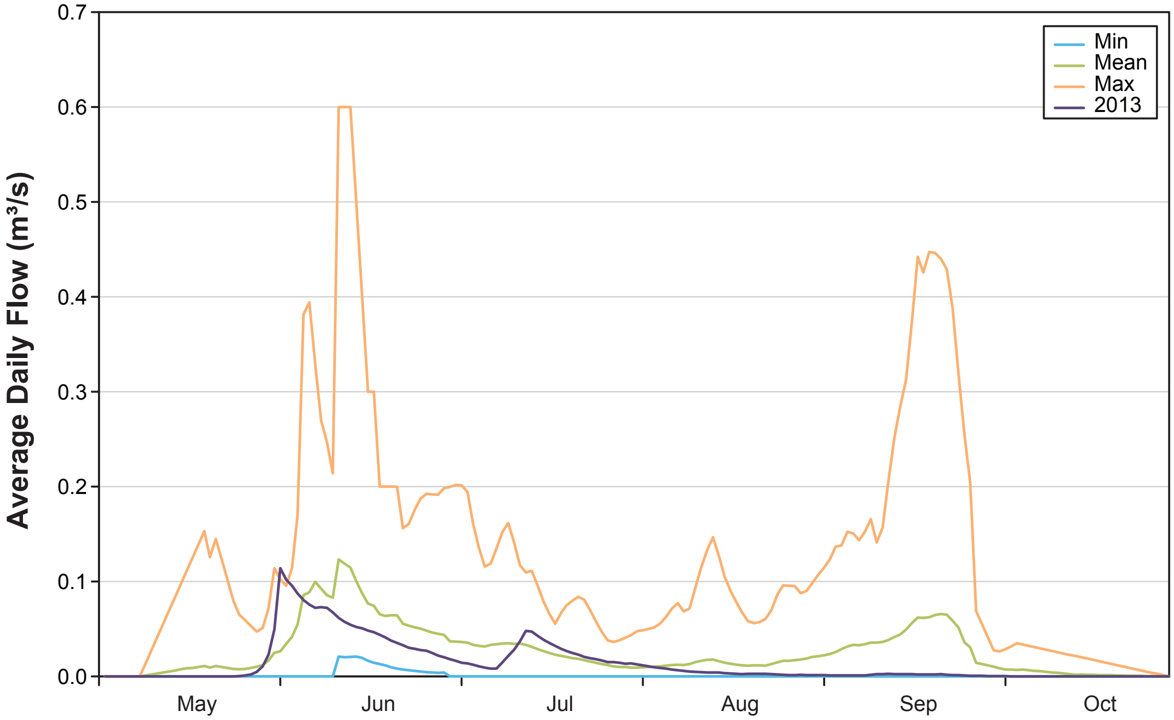
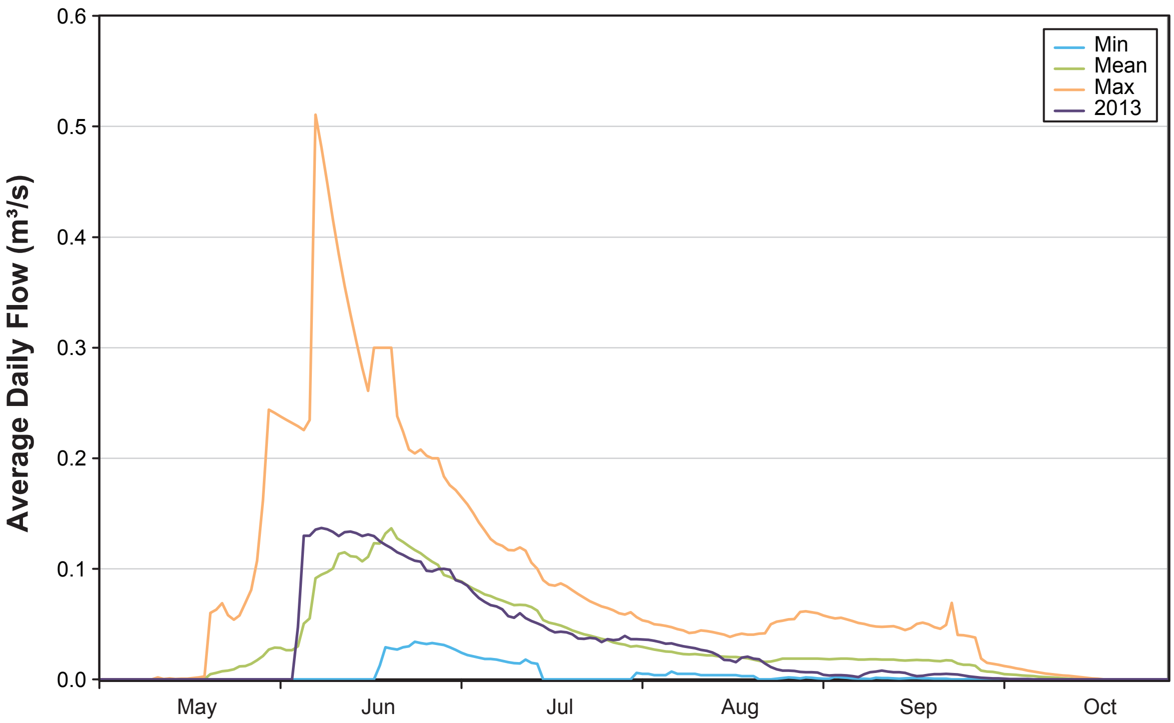


Figure 5-53

Comparison of 2013 Daily Flow at Counts Outflow with the Historical Record (1997 to 2013)



6. Lake Residence Time

6. Lake Residence Time

The residence time for a lake or water body, calculated on an annual basis, is defined as:

Annual residence time (yr) = Lake volume (m^3)/Total Inflow to Lake (m^3/yr)

Monthly or weekly residence times can be calculated by varying the time over which the inflow acts. The residence time is a theoretical estimate of the time required to replace the total volume of water in the lake with inflowing water. For studies where the water quality of the lake is of interest a low residence time (e.g., inflows would replace the total volume of water in the lake in a matter of days or weeks) would suggest that the water quality of the lake would be dominated by the quality of the inflowing water. In contrast a lake with a high residence time (e.g., inflows would need to occur for a number of months or years to replace the volume of water in the lake) would have a buffering capacity resisting changes to the lake water quality as a result of inflows.

Annual residence times are calculated for each of the lakes lying between the Long Lake Containment Facility (LLCF) and Lac de Gras for an average, 1 in 100 dry and 1 in 100 wet runoff years. Water discharged from the LLCF flows through seven lakes before entering Lac de Gras. These lakes are generally relatively small and shallow with lake volumes lower than the average annual discharge volume from the LLCF (6.4 Mm^3). If natural runoff from local catchments surrounding each lake is also added, the total annual inflow to each lake is significantly more than the lake volumes. This is illustrated in Table 6-1 for the three cases. The annual residence time for a year with average precipitation ranges from only 19 days for Moose Lake to 77 days for Nero Lake. The residence times presented are estimations based on runoff values calculated using frequency analysis of observed precipitation data for the average, (2 year return period), 100 dry and 100 year wet scenarios. Actual lake residence times will vary between years based on precipitation and flow conditions.

Table 6-1. Calculation of Annual Residence Times for Lakes Lying Downstream of the LLCF

Lake	Local Catchment flowing to water body (km^2)	Total Catchment (km^2)	Lake Area (km^2)	Annual inflow for year with average annual runoff (Mm^3)	Lake Volume (Mm^3)	Average Residence Time (days) [c]		
						Year with average annual runoff[d]	Year with 100 year dry runoff[d]	Year with 100 year wet runoff[d]
LLCF	31.7	31.7	10.7	6.4 [a]	-	-	-	-
Leslie	3.4	35.1	0.62	7.0 [b]	1.4	73	180	47
Moose	39.5	74.6	0.44	13.5 [b]	0.7	19	42	11
Nero	24.4	99	1.4	17.6 [b]	3.7	77	168	44
Nema	7.1	106.1	0.78	18.8 [b]	1.5	29	64	17
Martine	14.5	120.6	1	21.2 [b]	1.8	31	67	18
Rennie	28.2	148.8	0.94	25.9 [b]	1.5	21	45	12
Slipper	25.5	174.3	1.9	30.1 [b]	6.1	74	158	41

a: Average of observed LLCF discharge
b: Calculated as Total Catchment downstream of LLCF (km^2) x annual average runoff total (166.5 mm), added to the discharge from the LLCF. The contribution from direct precipitation on lake is balanced by evaporation from the lake surface.
c: Average residence time = Lake Volume / Annual Inflow
d: Average annual runoff = 166.5 mm, 1 in 100 dry year runoff = 81 mm, 1 in 100 wet year runoff = 310.5 mm ; values based on statistical analysis of Koala Meteorological Station precipitation data multiplied with runoff coefficient of 0.5. Dashes indicate not applicable.

The annual flow hydrograph for streams at the Ekati Diamond Mine shows a marked seasonality. Nearly all the annual flow occurs soon after snow melt, typically in June, with flows decreasing through the year (Table 6-2). The monthly flow distributions in 2013 at the Ekati Diamond Mine were very similar to the average distribution (Table 6-2). Higher than average flow occurred in June 2013, and was compensated by lower than average flows in May and July. There are zero flows in winter months as most streams at Ekati freeze to their beds in winter. As a result, an assessment of residence times in response to average monthly flows is also useful with results provided in Table 6-3. The results indicate that during June and July the total average monthly inflows to all lakes lying downstream of the LLCF are greater than lake volumes (i.e., residence times are less than 1 month). For other months monthly inflows are typically less than the lake volume.

Table 6-2. Average Monthly Flow Distributions at the Ekati Diamond Mine

	Percentage of Annual Runoff Total in Each Month					
	May	June	July	August	September	October
Average	4	55	23	8	8	1
2013	2	58	21	10	8	2

Table 6-3. Calculation of Monthly Residence Times for Lakes Lying downstream of the LLCF, Year with Average Annual Runoff

	Monthly Residence Time (days)					
Lake	May	June	July	August	September	October
Leslie	> 1 month (20% of lake volume in 1 month)	11	27	> 1 month (40% of lake volume in 1 month)	> 1 month (40% of lake volume in 1 month)	> 1 month (5% of lake volume in 1 month)
Moose	> 1 month (80% of lake volume in 1 month)	3	7	20	20	> 1 month (20% of lake volume in 1 month)
Nero	> 1 month (20% of lake volume in 1 month)	12	28	> 1 month (40% of lake volume in 1 month)	> 1 month (40% of lake volume in 1 month)	> 1 month (5% of lake volume in 1 month)
Nema	> 1 month (50% of lake volume in 1 month)	5	11	31	30	> 1 month (10% of lake volume in 1 month)
Martine	> 1 month (50% of lake volume in 1 month)	5	11	> 1 month (90% of lake volume in 1 month)	> 1 month (90% of lake volume in 1 month)	> 1 month (10% of lake volume in 1 month)
Rennie	> 1 month (70% of lake volume in 1 month)	3	8	22	22	> 1 month (20% of lake volume in 1 month)
Slipper	> 1 month (20% of lake volume in 1 month)	11	27	> 1 month (40% of lake volume in 1 month)	> 1 month (40% of lake volume in 1 month)	> 1 month (5% of lake volume in 1 month)

Care should be taken in interpreting and using the results of this assessment. The concept of residence time assumes that all inflowing water to a lake effectively displaces (pushes out) existing lake water. However, in reality mixing and flow processes within lakes are more complex. In deep lakes there is the potential for inflowing water to flow along surface under some conditions (e.g., soon after ice melt when a less dense cold (< 4 °C) layer of melt water can flow above relatively warmer and denser lake water), with limited displacement of deeper water. In broad shallow lakes there can be a preferred flow pathway through the middle of a lake with inflowing water passing through the lake with limited mixing with water in the shallows at the edge of the lake. Given the shallow lakes at the Ekati Diamond Mine the latter process is more likely.

In summary, residence times for the chain of lakes lying downstream of the LLCF are low, with lake inflow volumes during freshet (June and July) typically larger than the volume of water in the lakes at the onset of freshet. These results indicate that the lakes have limited buffering capacity with respect to the water quality of inflows to the lakes and that on an annual basis the water quality of the lakes will respond rapidly to any change in inflow water quality. The average annual discharge volume from the LLCF is a significant percentage of the total inflow to downstream lakes (i.e., it is 92% of the total annual inflow to Leslie Lake and 47% of the total annual inflow to Moose Lake), with the percentage falling to around 21% for Slipper Lake due to dilution with runoff from natural catchments draining to the lakes.

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Definitions of the acronyms and abbreviations used in this reference list can be found in the Glossary and Abbreviations section.

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