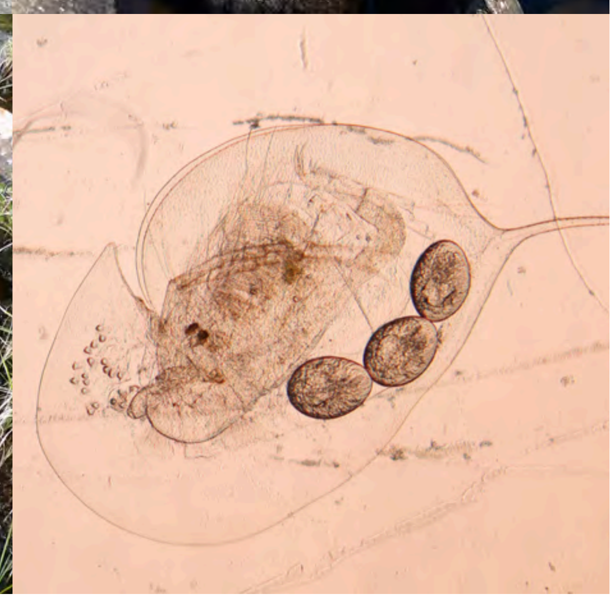
Ekati Diamond Mine

2014 Aquatic Effects Monitoring Program
Part 1 - Evaluation of Effects





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Dominion Diamond Ekati Corporation

EKATI DIAMOND MINE

2014 Aquatic Effects Monitoring Program Part 1 – Evaluation of Effects

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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.
AIC	Akaike Information Criterion
ALS	ALS Environmental Services
BACI	Before After Control Impact
Benthic	Pertaining to the bottom region of a water body, on or near bottom sediments or rocks.
Benthos	Benthos communities are a group of organisms that live associated with the bottom of lakes or streams. These communities contain a diverse assortment of organisms that have different mechanisms of feeding. The term benthos is used interchangeably with benthic invertebrates in this report. Benthos are an important food source for fish.
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
CCMS	Collision Cell Mass Spectrometry
CCREM	Canadian Council of Resource and Environment Ministers
Chlorophyll	Chlorophyll is a molecule contained in photosynthetic organisms which is required to carry out photosynthesis. Chlorophyll *a* is used as an indicator of phytoplankton biomass in this report.
CPK	Coarse Processed Kimberlite
CPOM	Coarse Particulate Organic Matter
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 a taxonomic order of insects. Dipterans are the true flies, and their larval stages 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. Diptera include the familiar mosquito and black-fly, and their larvae are an important food source for fish. 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 in terms of genera a community of organisms is. In general, a healthy ecosystem will support a variety of species and have a high diversity index.
DL	Detection Limit
DO	Dissolved Oxygen
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.
EPT	Ephemeroptera, Plecoptera and Trichoptera
EQC	Effluent Quality Criteria
EROD	Ethoxyresorufin-O-deethylase
ERM	Environmental Resources Management
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
FPOM	Fine Particulate Organic Matter

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 a few weeks.
GCL	Geosynthetic Clay Liner
HDPE	High Density Polyethylene
Hydrology	The study of the properties of water and its movement in relation to land.
IEMA	Independent Environmental Monitoring Agency
Invertebrates	Collective term for all animals without a backbone or spinal column.
ICPMS	Inductively Coupled Plasma Mass Spectrometry
ISQG	Interim Sediment Quality Guideline
K-B	Kajak-Brinkhurst
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 in the King-Cujo Watershed used to store mine water at the Ekati Diamond Mine.
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) and mine water in Long Lake at the Ekati Diamond Mine.
LME	Linear Mixed Effects
MDD	Minimum Detectable Difference

NRP	Nitrogen Response Plan
PDC	Panda Diversion Channel. An engineered channel used to divert water from North Panda Lake to Kodiak Lake.
PEL	Probable Effects Level
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.
PSD	Pigeon Stream Diversion. An engineered diversion constructed to allow flows from the headwater reaches of the Yamba/Exeter Watershed to enter Fay Bay channel unaltered and to circumvent Pigeon Pit.
Pupa	The stage between larva and adult in insects with complete metamorphosis.
Q-Q	Quantile-quantile
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.
SD	Standard Deviation
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: $H = -\sum pi \times \ln(pi)$, where pi is the proportion of the ith species or genera at a sampling station and $\sum$ indicates that the $pi \times \ln(pi)$ 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 $\sum$ indicates that the $(p_i)2$ is summed over all species or genera.
SNP	Surveillance Network Program.
SPE	Solid Phase Extraction
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
TSS	Total Suspended Solids
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.
Waste Rock	Barren rock or rock too low in grade to be mined or processed economically.
WLWB	Wek'èezhìi Land and Water Board

WRSA	Waste Rock Storage Area
Zooplankton	Zooplankton are small animals that live in the water column. They are secondary producers and feed mainly on phytoplankton.

Units of Measurement and Symbols

Centimetre	cm	**Metres above sea level**	masl
Cubic metre	m^3	**Micrometre (micron)**	µ
Degree	º	**Microsiemens**	µS
Degrees Celsius	ºC	**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**	±
Metre	m		

1. INTRODUCTION

1.1 BACKGROUND

The Aquatic Effects Monitoring Program (AEMP) at the Ekati Diamond Mine is a requirement specified in Dominion Diamond Ekati Corporation's (DDEC's) Class A Water Licence (W2012L2-0001). Sampling conducted for the 2014 AEMP was permitted through the Aurora Research Institute Scientific Research Licence (15382) issued for the Ekati Diamond Mine for the collection of samples between January 1 and December 31, 2014.

The AEMP is designed to detect changes in the aquatic ecosystem that may be caused by mine activities. The 2014 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 2012c). 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'èezhìi 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 AEMP Re-evaluation (Rescan 2012a) and two additional requests made by the WLWB. A summary of the WLWB approved changes made to the Evaluation of Effects following the 2012 AEMP Re-evaluation is provided in Section 1.4.

As completed in the past, the 2014 AEMP report includes a Summary Report which provides an overall summary of the Evaluation of Effects. The main 2014 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 2014, 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 Ekati Diamond Mine activities. To that end, the following components of the aquatic ecosystem were monitored in 2014:

- hydrology (October 2013 to September 2014);
- under-ice physical limnology (April/May 2014);
- open water season physical limnology (August 2014);
- ice-covered season lake water quality (April/May 2014);
- open water season lake water quality (August 2014);
- open water season stream water quality (June, July, August, and September 2014);
- lake sediment quality (August 2014);
- phytoplankton (August 2014);
- zooplankton (August 2014);
- lake benthos (August 2014); and
- stream benthos (August to September 2014).

Lake water quality and physical limnology were also monitored in July and September in the Pigeon-Fay and Upper Exeter Watershed.

Meteorological data are collected year round at the Ekati Diamond Mine between October 2013 and September 2014 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 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, 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 a decrease in 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 2011e, 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 eight main sections:

1. Introduction: includes the background and objectives of the 2014 AEMP program and provides a summary of the major mining, construction, and water management activities that occurred at the mine during the 2014 AEMP year;

2. Methods: includes a brief overview of the AEMP sampling design and a description of the statistics and interpretation methods used in evaluating the data;
3. Evaluation of Effects in the Koala Watershed and Lac de Gras: includes a summary of statistical results and discussion of each evaluated variable including identification of mine-related effects and impacts;
4. Evaluation of Effects in the King-Cujo Watershed 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;
5. Evaluation of Effects in the Pigeon-Fay and Upper Exeter Watershed: includes a summary of the statistical results and a discussion for each evaluated variable, including identification of mine-related effects and impacts;
6. Historical Lake Water Quality and Stream Hydrology: includes historical averages (by month) of all measured water variables for each of the AEMP lakes in the Koala Watershed, the King-Cujo Watershed, and the Pigeon-Fay and Upper Exeter Watershed for each baseline and monitoring year. Historical values of key hydrological variables are also included;
7. Historical Sediment Quality: includes historical averages (by month) of all measured sediment variables for each of the AEMP lakes in the Koala Watershed, the King-Cujo Watershed, and the Pigeon-Fay and Upper Exeter Watershed for each baseline and monitoring year; and
8. Lake 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.

Figure 1.2-1

Schematic of Aquatic Monitoring Programs at the Ekati Diamond Mine

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.
- Three major compliance points are discharge from Cell E of the Long Lake Containment Facility (LLCF), discharge from the King Pond Settling Facility (KPSF), and discharge from Desperation Pond.
- If water from the three 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 17 years.

Special Effects Monitoring Programs and Studies
(Annual Report or more frequent)
e.g., Pigeon Stream Diversion 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.

In 2014, the following four studies were undertaken as part of the special effects studies and monitoring programs:

1. Lac de Gras Water Quality Monitoring Station – monitoring at sampling at sites S5 and S6 in the north arm of Lac de Gras, beyond the current extent of the AEMP, was continued to determine if a new water quality monitoring station is required beyond the current site, S3. Sampling at sites S5 and S6 began in 2013 and the necessity of the addition of one or both stations to the annual AEMP program will be assessed as part of the 2015 AEMP Re-evaluation.
2. Grizzly Lake Biological Communities – phytoplankton and zooplankton were sampled in August to assess if communities have been altered following observed changes in the under-ice temperature profiles from 2011 to 2013. Results from biological monitoring in 2013 indicated that the taxonomic composition of the zooplankton assemblage may have changed through time; however, the lack of data from 2004 to 2012 made it difficult to determine whether these changes represented a trend through time or natural variability (ERM Rescan 2014b). Thus, an additional year of phytoplankton and zooplankton monitoring was recommended in 2014 (ERM Rescan 2014b)
3. 2014 Pigeon Stream Diversion (PSD) Monitoring Program – the PSD was designed and constructed as compensation for the loss of stream habitat during the development of Pigeon Pit at the Ekati Diamond Mine in accordance with *Fisheries Authorization* #SC99037. Under the authorization a monitoring program was established to assess the effectiveness of the PSD in providing productive fish habitat. The 2014 PSD Monitoring Report describes the results of the first post-construction year of the monitoring program of the PSD (ERM 2015). Physical components including stream flow, water temperature, stream habitat, water quality and soil and sediment quality were assessed. Sampled biological components included vegetation monitoring, coarse and fine particulate organic matter (CPOM/FPOM), CPOM retention, organic matter processing (leaf packs), periphyton/epilithon, benthic invertebrates, and the number, biological characteristics and migration patterns of all species and life stages of fish, although fish monitoring efforts focused primarily on Arctic grayling.
4. Hydrocarbon Exposure to Fish – a follow-up study to the results of the 2012 EROD (ethoxyresorufin-O-deethylase) activity analyses (completed as part of the 2012 AEMP; Rescan 2013d) which indicated evidence of hydrocarbon exposure in slimy sculpin and round whitefish that may have been related to mine activities.

The results of the two first studies are presented in Part 2 – Data Report. The results of the PSD monitoring are presented in a separate report (ERM 2015). The results from the hydrocarbon exposure to fish study were submitted on December 18, 2014 to the WLWB in a separate report entitled "Characterization of Hydrocarbons found in the Arctic Aquatic Environment near the Ekati Diamond Mine" (DDEC 2014).

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 those data are presented separately from the AEMP report.

1.3 OVERVIEW OF THE EKATI DIAMOND MINE ACTIVITIES

1.3.1 Koala Watershed

The Koala Watershed contains the majority of the Ekati Diamond Mine infrastructure including the main camp, the process plant, the LLCF, and the airstrip, as well as the Panda, Koala, Koala North, Fox, and Beartooth pits with associated waste rock storage areas (WRSAs). The following major activities took place in the Koala Watershed during the 2014 AEMP period (October 1, 2013 to September 30, 2014):

- Main camp housed an average of 16,050 people per month (535 people per night);
- Construction:
  - Old Camp South Pond Reclamation: Commencing in July 2014, excavation of the Phase 1 Processed Kimberlite South Pond was undertaken to begin reclamation activities. Work in the area began by discharging surface water in the South Pond to Larry Lake after it was tested and confirmed to meet water licence discharge criteria. Pumping activities were stopped when the remaining water began to be contaminated with sediment, and this water was instead trucked to the LLCF for disposal. The existing Old Camp Road was improved to allow safe usage by 777 haul trucks. The approach lights within the center of the pond were deactivated, and a temporary bypass installed around the pond. Reclamation activities involved the removal of both the processed kimberlite and the pond liner system. Crews separated the high-density polyethylene (HDPE) and geosynthetic clay liners (GCL) and disposed of them within the Ekati Diamond Mine landfill. The processed kimberlite was loaded into trucks and disposed in the designated coarse processed kimberlite (CPK) disposal facility. The entire excavated surface was topped with clean esker sand and graded to facilitate surface water management. The only remaining activities are to complete the construction of a runoff channel through the reclaimed area and subsequent water quality monitoring, as well as minor grading and housekeeping of liner debris.
  - Misery Power Supply Phase 1: Commencing in August 2014, the Misery Power Supply project will allow power generated at the main Ekati Diamond Mine power plant to be provided to the Misery Camp. The generating system at the Misery Camp will be shut-down when the new power distribution system is operational and only utilized for emergency back-up purposes. Project activities for Phase 1 include the installation of utility poles along the East side of the Misery Road and related conductor, communication, and protection systems. As of writing, the construction of access push-outs along the Misery Road had begun. Drilling of pole holes and installation of framed utility poles is underway.
  - Construction of Pigeon WRSA: Placement of clean granite material for the Pigeon WRSA and road access development began on May 13, 2014, and movement of the Pigeon till dump from the previously mined Pigeon bulk sample pit started on July 16, 2014. In the period up to September 30, 2014, 1,240,570 tonnes of granite and 227,610 tonnes of till were moved. Construction of an access road around the location of the final pit extent was also completed during this time, with the construction of a water diversion berm still in progress.

- Panda Diversion Channel (PDC) Phase 3: Commencing in January 2014, the final phase of the Panda Diversion Channel stabilization project was started. This last phase of work involved the benching of the North East portion of the canyon section to provide long-term stability (approximately last remaining quarter of the project). Construction started with the installation of a protective ice pad in the bottom of the PDC. Crews utilized drill and blast techniques to create the final designed bench and then load and haul techniques to remove and dispose of the blasted material. Where required, a rock fillet was installed to prevent future permafrost degradation. A geotextile-lined berm was installed on the crest of the new bench to provide sediment control. Final clean-up activities included scraping of the ice pad surface to remove sediment material, and excavation of a trench to allow freshet water flow. Environmental staff performed regular water quality testing to ensure water licence standards were being met;
- Grizzly Road Realignment: Commencing in July 2014, a short section of the Grizzly Road was re-aligned, due to the close proximity of the new catch bench installed in the North East section of the PDC. Construction activities included realignment of a short section of road, installation of safety berms, and realignment of the freshwater pipeline.
- LLCF Reclamation Vegetation Trial: Commencing in winter 2014, an experimental placement of rock cover was completed in Cell B over the area seeded in fall 2013. Construction activities included the haulage of rock material with various material specifications to the North end of Cell B, and the placement of this material over the seeded area by 730 haul truck and excavator (according to design requirements).

- Mining activities:
  - Fox Pit:
    - Kimberlite ore was transported to the process plant;
    - Waste rock was transported to the Fox WRSA; and
    - Kimberlite coarse ore rejects were placed in the coarse kimberlite rejects area of the Panda/Koala WRSA.
  - 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;
    - Waste rock from the underground was transported to the Panda/Koala WRSA; and
    - Kimberlite coarse ore rejects were placed in the coarse rejects area of the Panda/ Koala WRSA.
  - Koala Pit:
    - Kimberlite ore from underground was transported to the process plant;
    - Waste rock from the underground was transported to the Panda/Koala WRSA; and
    - Kimberlite coarse ore rejects were placed in the coarse rejects area of the Panda/ Koala WRSA.

- Dewatering and discharge:
  - Surface sump water and treated effluent from the sewage treatment plant continued to be deposited into the LLCF. Fine processed kimberlite was deposited to both the LLCF and Beartooth pit;
  - Fine processed kimberlite and underground minewater were pumped to Beartooth Pit (total volume = 2,147,165 m^3 and 170,886 m^3, respectively);
  - Grizzly Lake drawdown for use at main camp continued (volume = 82,311 m^3). Total volume of water drawn from Grizzly Lake (including for use at Main Camp and for PSD and PDC ice pad construction) was 91,135 m^3;
  - Water was pumped from Bearclaw Lake to North Panda Lake from August 7 to 14, 2014 (total volume = 132,623 m^3);
  - Water was pumped from Beartooth Pit to Cell C of the LLCF from June 25 to September 30, 2014 (ongoing). The total volume pumped from June 25 to September 30, 2014 was 971,312 m^3;
  - Water from Cell E of the LLCF was discharged into Leslie Lake from October 1 to November 19, 2013 (ongoing from September 2013; total volume = 1,043,235 m^3) and from July 28 to August 6, 2014 (total volume = 315,849 m^3);
  - Water was pumped from the Pigeon Test Pit to Cell B of the LLCF from June 26 to July 12, 2014. The total volume pumped was approximately 115,200 m^3; and
  - All water discharged from Cell E to the receiving environment met Effluent Quality Criteria (EQC) defined in Water Licence W2012L2-0001.

1.3.2 King-Cujo Watershed

The King-Cujo Watershed contains the KPSF, as well as a portion of the Misery Camp and Misery WRSA. The following major activities took place in the King-Cujo Watershed during the 2014 AEMP period (October 1, 2013 to September 30, 2014):

- Misery camp housed an average of 2,991 people per month (99 people per night).
- Construction: No construction took place within the King-Cujo Watershed
- Mining activities:
  - Misery Pit:
    - Kimberlite was stored on Ore Storage Pads at Misery Camp before transport to and processing at the Main Camp Process Plant; and
    - Waste rock was hauled to the Misery WRSA.
- Dewatering and discharge:
  - No water was pumped from the Waste Rock Dam into the KPSF in 2014;
  - No water was pumped from Misery Pit into the KPSF in 2014; and
  - No water was pumped from the KPSF to Cujo Lake in 2014.
  - Water was discharged from Desperation Pond to KPSF between July 4 to 7, 2014 (total volume = 20,763 m^3).

1.3.3 Carrie Pond Watershed

The Carrie Pond Watershed contains a portion of the Misery Pit, the associated WRSA, and Desperation Pond. The following major activities took place in the Carrie Pond Watershed during the 2014 AEMP period (October 1, 2013 to September 30, 2014):

- Mining activities:
- Dewatering and discharge:
  - No water was pumped from Desperation Pond into Carrie Pond in 2014.

1.3.4 Pigeon-Fay and Upper Exeter Watershed

The Pigeon-Fay and Upper Exeter Watershed contains the Pigeon test pit and the PSD. The following major activities took place in the Pigeon-Fay and Upper Exeter Watershed during the 2014 AEMP period (October 1, 2013 to September 30, 2014):

- Construction:
  - Pigeon Stream Diversion (PSD): Commencing in January 2014, the final portion of construction activities was completed at the PSD. Construction activities included the installation of a protective ice road to allow access to the area by construction equipment, and completion of the inlet and outlet sections. Construction crews worked to close-up the inlet to the existing Pigeon Stream, extend and complete the inlet berm to design specifications, and install a fish barrier at the outlet of the existing Pigeon Stream. During freshet of 2014, the PSD was the main route for water flow through the area, as the original Pigeon Stream was hydrologically isolated from upstream water flow via the water diversion berm at the inlet section of the PSD;
  - Pigeon infrastructure: New infrastructure for the Pigeon Pit development included two explosives magazines, a refuge trailer, and a power generating system. In the spring of 2014, two laydown pads were installed at the re-aligned section of the Sable Road to locate the new explosives magazines. The two explosives magazines were installed and commissioned according to the regulatory requirements. A constructed pad was established along Sable Road to allow installation of the refuge station, generators, and an equipment parking area. A lined berm was installed to provide secondary containment for the fuel storage tank (double-walled) and generator enclosure. The fuel storage and generating system was installed and commissioned according to regulatory and manufacturer requirements. The refuge station was installed and provides washroom facilities to workers and a safe refuge during winter storms; and
  - Pigeon ring road and water management berms: Commencing in August 2014, an access ring road was constructed around the perimeter of the planned Pigeon Pit. This ring road also provided access for construction equipment to install the water management berms that will help to prevent surface water from entering the pit. Construction activities for the water management berms include the excavation of a key trench and placement of till in designated lifts to a higher elevation than the existing height of ground. These water management berms will deflect water around the pit when completed. In 2014, it is

planned to complete installation of the North water management berms and in 2015 to complete installation of the South water management berms.

- Mining activities:
  - No mining activities took place in 2014.

The year 2014 was the 17th consecutive year of post-baseline monitoring within the Koala Watershed and Lac de Gras, the 14th consecutive year of post-baseline monitoring within the King-Cujo Watershed and Lac du Sauvage, the 2nd year of monitoring in the Carrie Pond Watershed, and the 1st year of post-baseline monitoring within the Pigeon-Fay and Upper Exeter Watershed.

1.4 CHANGES TO EVALUATION OF EFFECTS FOLLOWING THE 2012 AEMP 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 five years of data available, water quality data collected from Leslie-Moose Stream were 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 2014.
3. 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 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. The Akaike 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.
5. In the event that both transformed and untransformed data satisfied parametric assumptions, the AIC was used to determine which transformation provides the best fit. 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.

6. 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-squared values indicated that model fit was weak ($r^2 < 0.5$) or poor ($r^2 < 0.2$) and that results of statistical analyses should 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 for each watershed (Sections 3.3, 4.3, and 5.3 of this report).

In 2014, the final portion of construction activities were completed at the PSD and it was connected to the natural Pigeon Stream. During freshet of 2014, the PSD was the main route for water flow through the Pigeon Pit area. Thus, the Pigeon AEMP was implemented in the winter of 2014. The Pigeon AEMP involved the monitoring of two lake sites and two stream sites in the Pigeon-Fay and Upper Exeter Watershed. Details on sampling locations and the sampling program undertaken in the Pigeon-Fay and Upper Exeter Watershed are provided in Sections 2.1 and 2.2.

2. METHODS

2.1 SAMPLE COLLECTION

2.1.1 2014 Field Methodology

Field methodologies are typically consistent for all AEMP sampling periods. If deemed appropriate, minor changes may be implemented by DDEC following new information or conditions, or stakeholder recommendation. A complete description of field methodologies is provided in Part 2 - Data Report, which includes a summary of modifications made to the field and laboratory methods that occurred following the 2012 AEMP Re-evaluation.

DDEC personnel conducted all of the ice-covered season sampling and the majority of stream flow measurements. ERM scientists conducted the open water season lake and stream sampling with the assistance of DDEC personnel. The only exception was the open water season sampling of Pigeon-Fay and Upper Exeter Watershed lakes and streams, which was done by DDEC in July and September.

2.1.2 2014 Sampling Locations

The 2014 AEMP lake and stream sampling sites are provided in Table 2.1-1 and shown in Figure 2.1-1. A surface water flow diagram through the AEMP sampling area is provided in Figure 2.1-2. Bathymetric maps depicting the aquatic sampling locations within each lake are provided in Figures 2.1-3 through 2.1-16 of Part 2 - Data Report.

Table 2.1-1. 2014 AEMP Sampling Locations

Lake Sites	Stream Sites
Reference Watersheds	
Nanuq Lake	Nanuq Outflow
Counts Lake	Counts Outflow
Koala Watershed and Lac de Gras	
Vulture Lake (reference)	Vulture-Polar (reference)
Kodiak Lake	Lower PDC[1]
Grizzly Lake[4]	Kodiak-Little[3]
1616-30 (LLCF)[5]	1616-30 (LLCF)[5]
Leslie Lake	Leslie-Moose[2]
Moose Lake	Nema-Martine
Nema Lake	Slipper-Lac de Gras
Slipper Lake	
Lac de Gras: S2, S3[6]	

(continued)

Table 2.1-1. 2014 AEMP Sampling Locations (completed)

Lake Sites	Stream Sites
King-Cujo Watershed and Lac du Sauvage	
1616-43 (KPSF)[5]	1616-43 (KPSF)[5]
Cujo Lake	Cujo Outflow
Lac du Sauvage: LdS1, LdS2[4]	Christine-Lac du Sauvage[1]
Carrie Pond Watershed	
	Mossing Outflow[2]
Pigeon-Fay and Upper Exeter Watershed	
Fay Bay[7]	Pigeon Stream Reach 7 (reference)[2]
Upper Exeter Lake[7]	Pigeon Reach 1[2]

[1] *Water quality and hydrology only.*
[2] *Water quality only.*
[3] *Water quality and stream benthos only*
[4] *Water quality and physical limnology only. Some biological variables monitored in Grizzly Lake as part of a special study presented in Part 2 – Data Report.*
[5] *Water quality and pumping data only.*
[6] *No sediment quality or benthos sampled*
[7] *No zooplankton or benthos sampled*

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 PDC are also located upstream of the LLCF, but are in close proximity to 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 potential 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 WRSAs.

All AEMP sampling stations in the King-Cujo Watershed are located downstream of the KPSF (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 Carrie Pond Watershed includes one AEMP sampling station (i.e., Mossing Outflow). The main influence to Mossing Outflow is Desperation Pond, which is located upstream (Figure 2.1-1).

Figure 2.1-1
AEMP Lake and Stream Sampling Locations, 2014

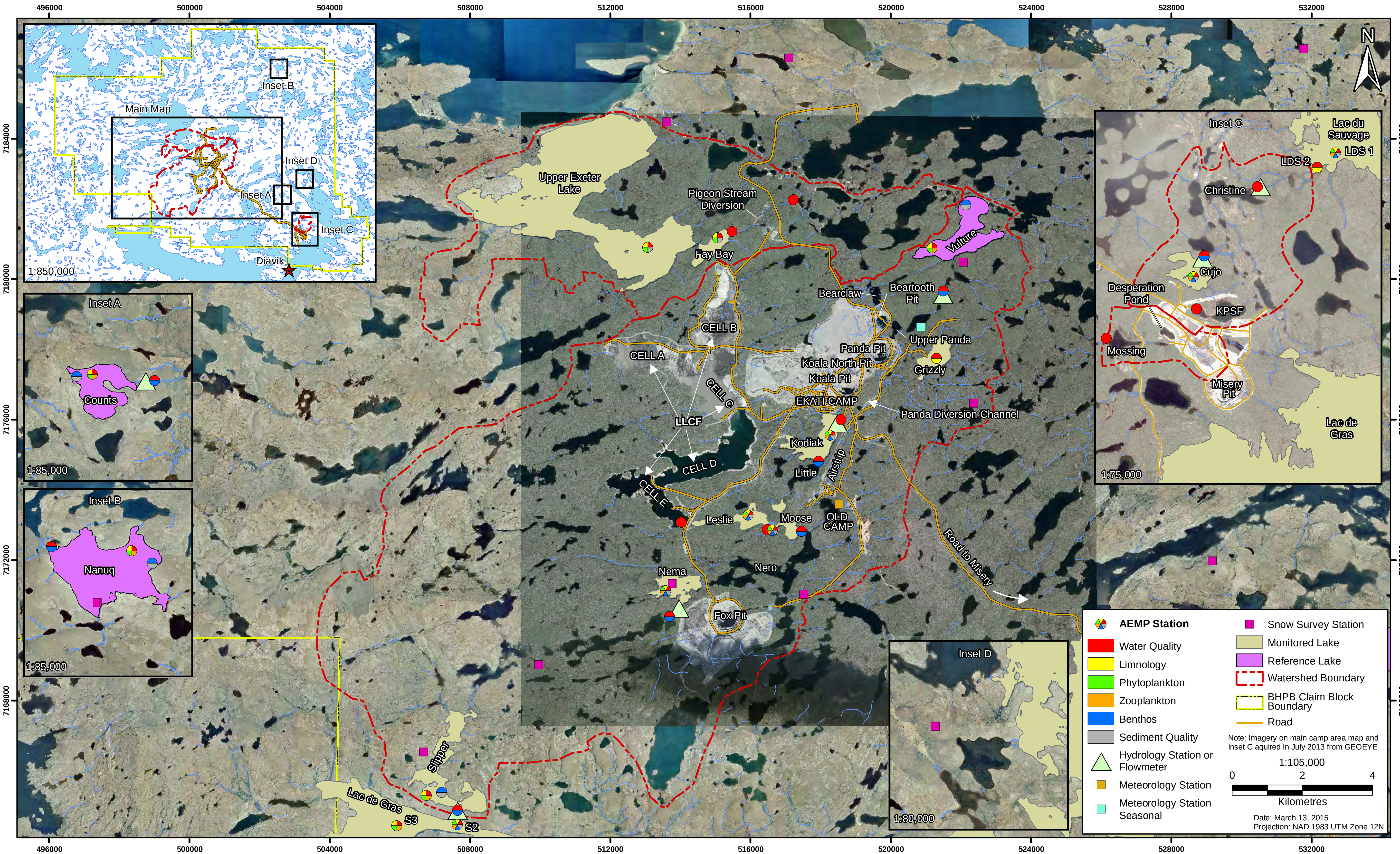
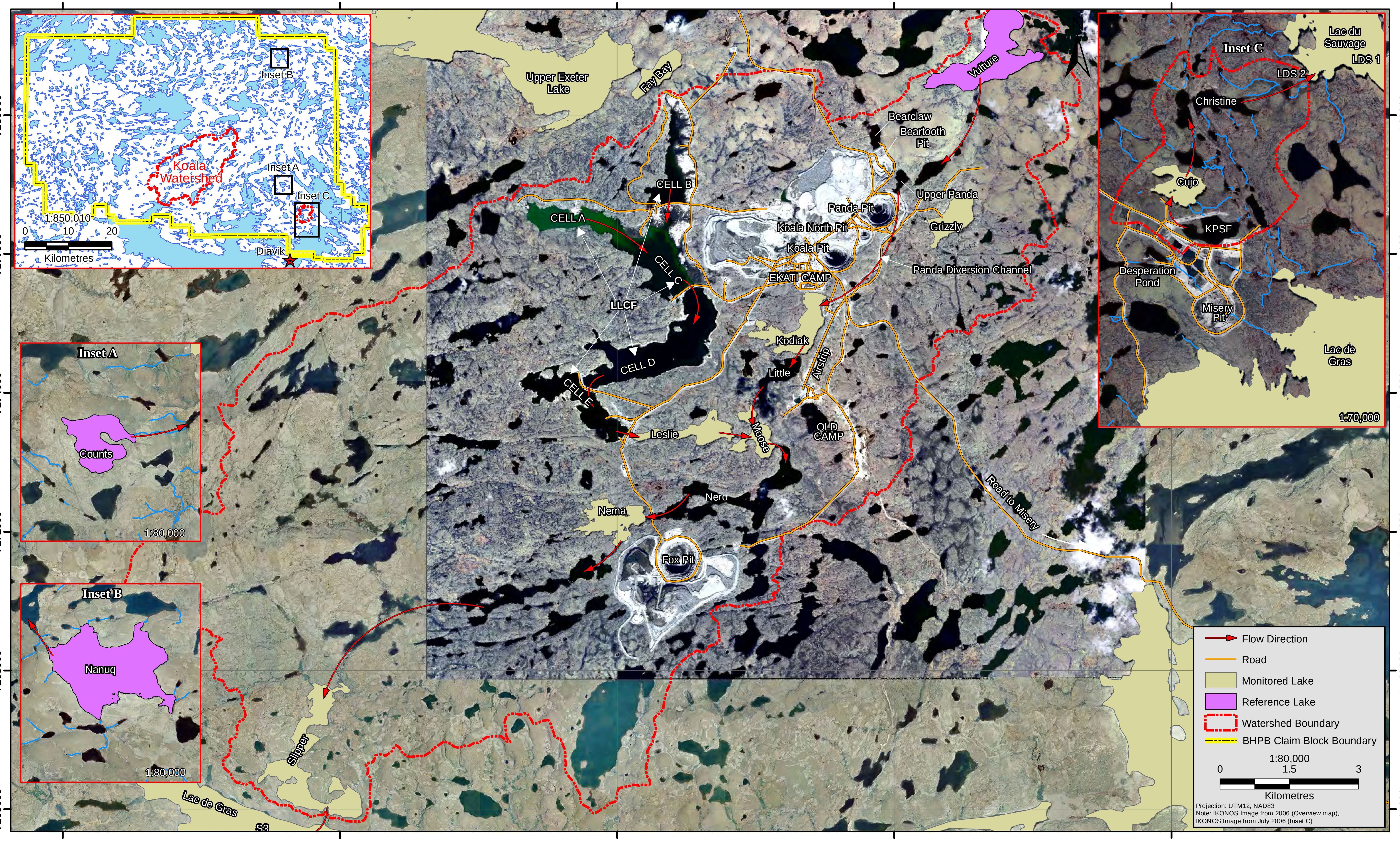


Figure 2.1-2
Surface Water Flow Through the AEMP Sampling Area



All but one of the AEMP sampling stations in the Pigeon-Fay and Upper Exeter Watershed are located downstream of the PSD. The one exception is Pigeon Reach 7, which is an internal reference site located upstream of the PSD. The Pigeon-Fay and Upper Exeter Watershed does not receive any discharge from Pigeon Pit, as all minewater and drainage from the WRSA is directed into the LLCF. Thus, potential effects at these sites stem from the construction and operation of the PSD and from fugitive dust deposition from Pigeon Pit activities.

The external reference lakes and streams (Nanuq and Counts lakes and their respective outflows) are located well away from any mine activities (Figure 2.1-1). Nanuq Lake is located in the northeast corner of the DDEC 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.

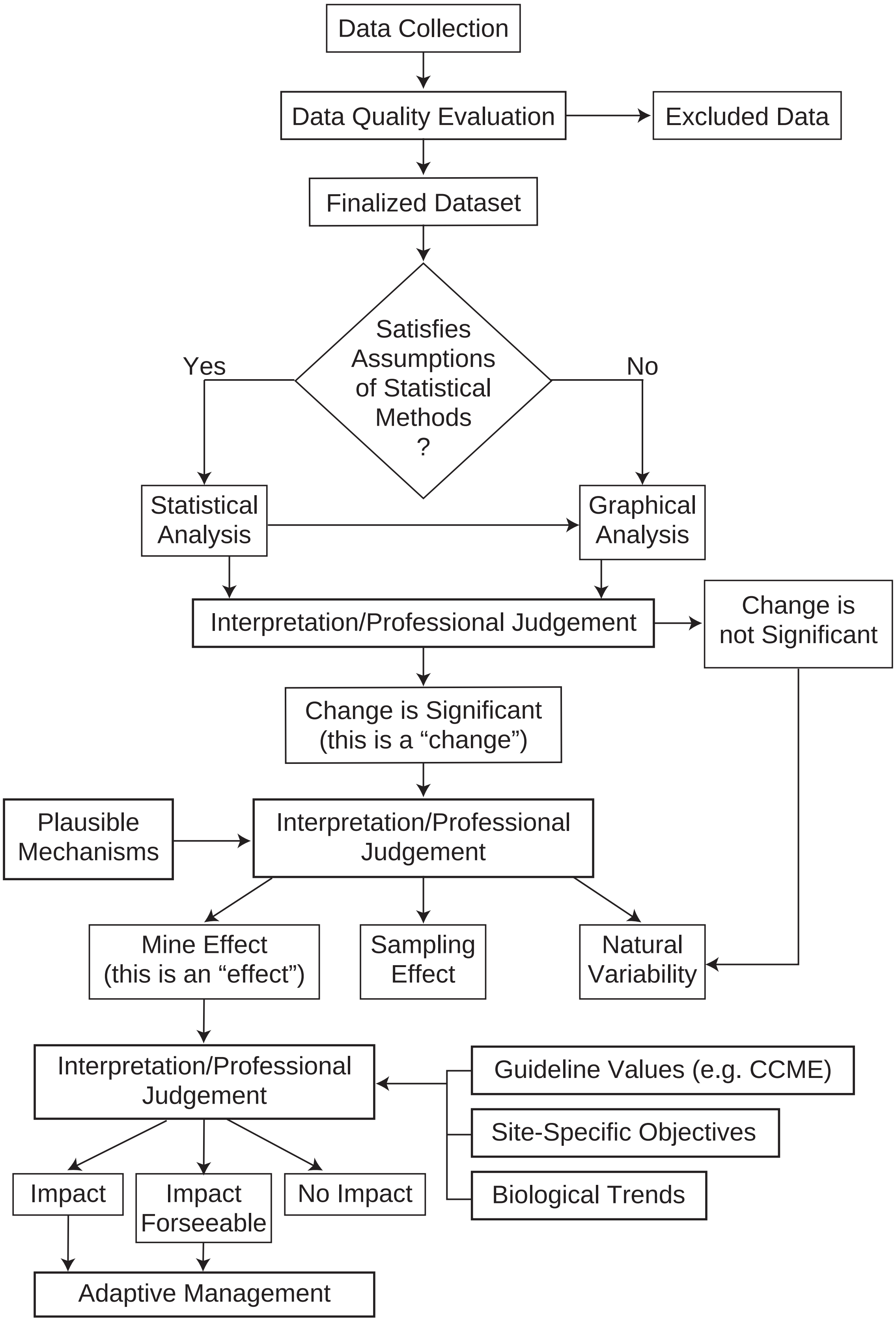
2.2 EVALUATION METHODS

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 2014 AEMP sampling program are reported in Part 2 – Data Report of the 2014 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, are presented in Part 2 – Data Report of the 2014 AEMP report.

The finalized dataset was graphically and statistically analysed to detect possible mine effects. For sites in the Koala Watershed and Lac de Gras, and sites in the King-Cujo Watershed and Lac du Sauvage, 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). The Mossing Outflow site in the Carrie Pond Watershed was not evaluated for effects in 2014 as only two years of data were available for that location. The inclusion of Mossing Outflow in the Evaluation of Effects will be assessed as part of the 2015 AEMP Re-evaluation. For sites in the Pigeon-Fay and Upper Exeter Watershed, Before-After/Control-Impact (BACI) analysis was used to detect changes in the aquatic environment as a result of Pigeon development activities. The BACI analysis compared before-after trends apparent at monitored sites with that of a corresponding reference site to determine if the trends were parallel and thus attributable to a natural process. If statistical analyses were not possible because assumptions or data requirements were not satisfied, variables were subjected to graphical analysis (see Section 2.2.6). In such cases, data were examined for historical trends and spatial gradients. In all cases, analyses were combined with best professional judgement when evaluating effects for all variables (see Section 2.2.7).


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 2014 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 2014 AEMP sampling program.

Table 2.2-1. Summary of the 2014 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=2 @ 1 m below surface n=2 @ mid water column depth n=2 @ 2m from the bottom (Leslie Lake only)
	July and September	Pigeon-Fay and Upper Exeter Watershed Only n=2 @ 1 m below surface n=2 @ mid water column depth
Physical Limnology	April[1]	n=1 profile over deepest part of lake, or at lake station
	early August	n=1 profile over deepest part of lake, or at lake station, Secchi depth
	July and September	Pigeon-Fay and Upper Exeter Watershed Only n=1 profile over deepest part of lake, or at lake station
Phytoplankton	early August	n=3 @ 1 m
Zooplankton[2]	early August	n=3 vertical hauls from 1 m above bottom to surface, with flowmeter
Benthos[2]	early August	n=3 @ 5–10 m depth (mid)
Sediment quality	early August	n=3 @ 5-10 m depth (mid) Ekman grabs and cores
Streams		
Water quality	June (freshet), early July, early/mid-August, September (fall high flows)	n=2
	Biweekly during open water season	Pigeon Reach 1 Only n=2
Benthos	early August to early September	n=5
Automated station installation	installation prior to freshet, maintenance as necessary	n=1
Hydrology manual flow measurements[3]	late-May to late August	bi-weekly, 2 to 3 times during freshet

(continued)

Table 2.2-1. Summary of the 2014 AEMP Sampling Program (completed)

Monitoring	Seasonal Frequency	Replication and Depths at each Lake/ Stream per Sampling Event
Streams (*cont'd*)		
Hydrometric levelling surveys	early May to late August	at time of installation, then bi-weekly to monthly
Hydraulic Geometry Survey	August	during low flow

n = number of samples or measurements

[1] Dissolved oxygen and temperature profiles were collected several times throughout the ice-covered season in Cujo Lake and Kodiak Lake.

[2] Reference lakes and lakes of the Koala and King-Cujo watersheds only.

[3] Reference streams and streams of the Koala and King-Cujo watersheds only.

2.2.3 Variables Evaluated in 2014

The variables evaluated in the 2014 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 for Koala and King-Cujo Watersheds

Regression models were used to compare data from each of the monitored lakes to reference lake data over the monitoring period, between 1998 and 2014. 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 Sections 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 (Section 2.2.4.5). Details of the statistical results for each variable are presented in Part 3 – Statistical Report of the 2014 AEMP report.

The statistical methodology outlined above has been used to assess patterns in 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, 2012c). The minor modifications introduced in 2013 are described in Section 1.4 (see bullet points #3 to #6).

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).

Table 2.2-2. Aquatic Variables Evaluated in 2014

Physical Limnology - Lakes	Water Quality – Lakes and Streams	Sediment Quality – Lakes	Aquatic Ecology
• Under-ice dissolved oxygen • Secchi depth • Open water dissolved oxygen[1] • Hydrology[1,2]	Physical/Ions • pH • Total alkalinity • Water hardness • Chloride • Potassium • Sulphate • Total suspended solids[3] Nutrients • Total ammonia-N • Nitrite-N • Nitrate-N • Total phosphate-P • Total organic carbon Metals • Total antimony • Total arsenic • Total barium • Total boron • Total cadmium • Total copper[4] • Total molybdenum • Total nickel • Total selenium • Total strontium • Total uranium • Total vanadium	Nutrients • Available Phosphorus • Total Nitrogen • Total Organic Carbon Metals • Antimony • Arsenic • Copper[4] • Cadmium • Molybdenum • Nickel • Phosphorus • Selenium • Strontium	Phytoplankton • Chlorophyll *a* concentrations • Phytoplankton density • Phytoplankton diversity • Relative densities of major phytoplankton taxa Zooplankton[5] • Zooplankton biomass • Zooplankton density • Zooplankton diversity • Relative densities of major zooplankton taxa Lake Benthos[5] • Lake benthos density • Lake benthos dipteran diversity • Relative densities of major dipteran taxa Stream Benthos[5] • Stream benthos density • Stream benthos dipteran diversity • Relative densities of major dipteran taxa • Stream benthos EPT diversity • Relative densities of EPT taxa

[1] *Open water season DO and 2014 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 6.*
[3] *Pigeon-Fay and Upper Exeter Watershed only.*
[4] *King-Cujo Watershed only.*
[5] *Koala and King-Cujo watersheds only.*

2.2.4.1 *Linear Mixed Effects (LME) Regression*

Model Form

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:

(1) $$y = Lake + Year + Year^2 + Lake*Year + Lake*Year^2,$$

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:

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

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:

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

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:

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

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 quantile-quantile plot (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:

(5) *y = Lake + Year + Year*^{2} *+ Lake*Year + Lake*Year*^{2} *+ Year-R + Error-R,*

or mathematically:

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

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 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. 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 $x1,\ldots,xp$ and a single error term $\varepsilon \sim N\left(0,\sigma^2\right)$ is:

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

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:

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

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:

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

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 $j \neq k$.

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).

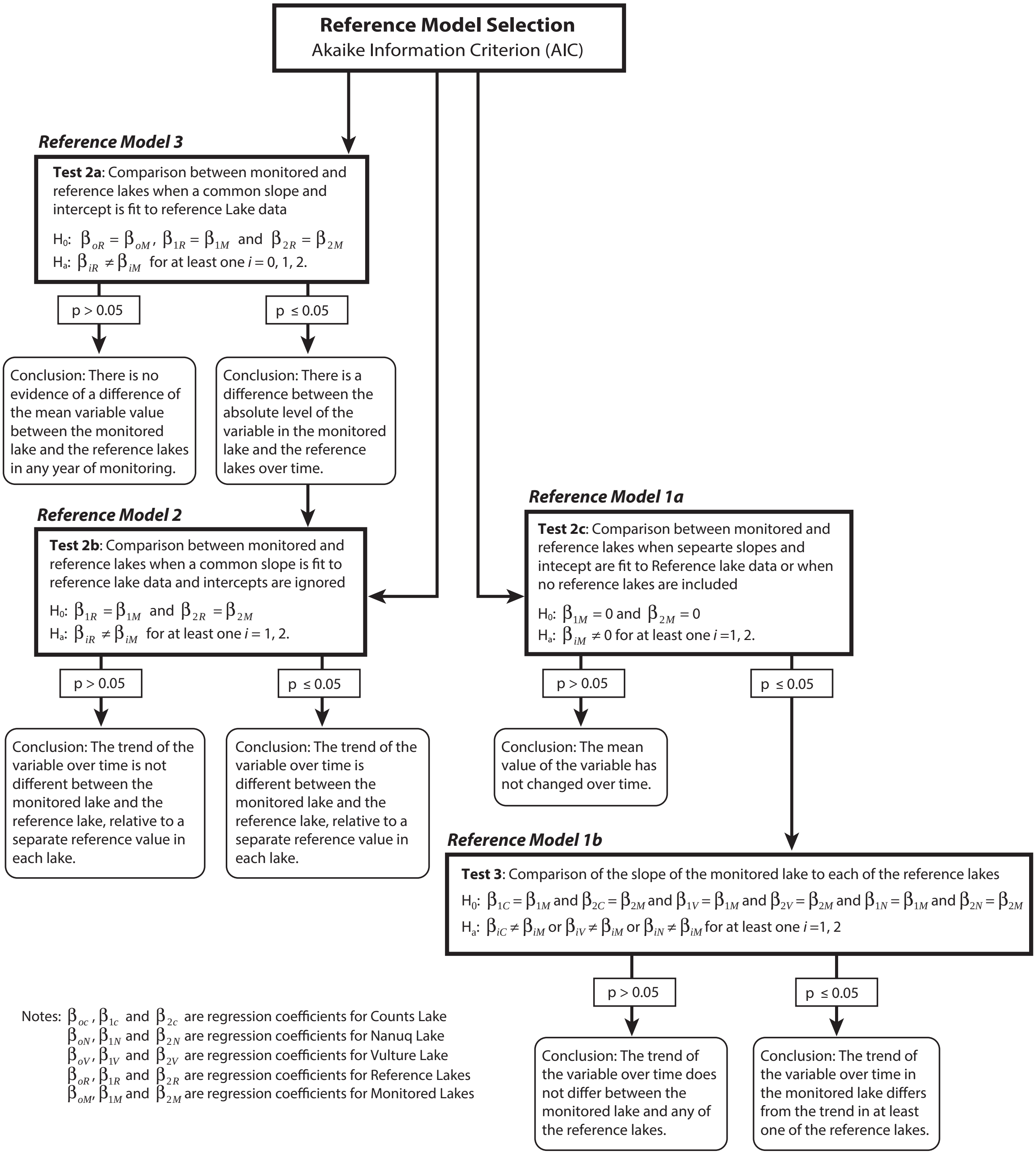
Figure 2.2-2

Hypothesis Testing Procedure for Evaluation of Effects in the Koala and King-Cujo Watersheds

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

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:

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

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

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:

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

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 streams 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 (national 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.

Minimum Detectable Difference

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 2014 as fitted means were generally clearly greater than, or less than water quality benchmark values. In cases where only the 95% confidence interval around the fitted mean exceeded a benchmark value in monitored lakes, similar patterns were observed in reference lakes.

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 2014 AEMP Report.

2.2.5 Statistical Analysis for the Pigeon-Fay and Upper Exeter Watershed

Data from monitored and reference sites in the Pigeon-Fay and Upper Exeter Watershed were analyzed for potential mine effects on water quality, sediment quality, and phytoplankton variables. BACI models were used to compare data from each monitored site to reference data. The analysis followed the following steps:

- consideration of censored data due to analytical detection limits;
- identification and filtering of outliers;
- data transformations for normality and homogeneity of variances;
- optimization of statistical models with and without a term for random, inter-annual variation;
- testing and validation of statistical hypotheses on patterns in monitoring data; and
- presentation of observed and modelled results.

The statistical methodology was used to assess patterns in water quality, sediment quality, and phytoplankton communities at the monitored sites in 2014, with the interval of 1994 to 2007 serving as the baseline (i.e., before) period.

Observations at or below analytical detection limits were considered censored. Censored data can potentially bias statistical analyses because of violation of underlying mathematical assumptions. Data were excluded from the analysis if greater than 60% of observations from a site in a sampling year were censored. If more than 10% of observations from a site were censored, data were flagged to caution interpretation of results. If censored data were included in the analyses, the data were assumed to be equal to half the analytical detection limit.

In addition to the BACI analysis, 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).

2.2.5.1 Before-After Control-Impact Analysis

<u>Model Form</u>

Linear models were constructed for each monitored location based on a BACI design. A model was constructed for each monitored site. The models follow the general form given in Eq. 11.

(11) $$y = site + period + site{:}period$$

This model identifies variation associated with different components, where:

- *site* was the term describing the fixed differences between the sites;
- *period* was the term describing the fixed differences between the before and after periods across all sites (reference and monitored); and
- *site:period* was the interaction term describing site-specific differences between periods (the BACI term).

The *site:period* term is the key statistical term that describes changes to the monitored site during the period of potential mine effects.

The model can be written mathematically as:

(12) $$E(y_{sp}) = \beta_0 + \beta_s + \beta_p + \beta_{s:p} + \varepsilon$$

where β_0 is the intercept, β_s is the expect value for site s, β_p is the expected value for period p, $\beta_{s:p}$ is the expected value for site s in period p, and ε is the residual variation (error). Fitted values were calculated from the estimates of these modelled parameters to assess model performance and aid in interpretation (see below and in Sections 2.2.5.2 and 2.2.5.3).

Assessing Model Fit

The linear modeling approach used in the analysis makes the following assumptions:

- observations are normally distributed; and
- variances are homoscedastic (i.e., error variances are not correlated with the values of predictors).

The general approach in this analysis was to compare the normalized residuals and overall model performance for a simple version of the linear model using both untransformed and $\log_{10}$-transformed data. Plots of standardized residuals and fitted values, normal Q-Q plots, and model performance statistics (e.g., R^2 and residual standard error) were examined to establish the most appropriate choice of transformation. A data transformation was conducted if the simple model showed a more uniform random distribution of residuals, a closer distribution along the 1:1 reference line on the Q-Q plot, and equivalent-or-improved overall model performance.

Pseudoreplication

Observations from the same site within the same season were pooled for the purposes of the statistical analysis. Although this approach does confound within-site variation between sampling depths and sampling times, the pooling was necessary to provide sufficient degrees of freedom for model fitting and hypothesis testing.

Random Variation

An additional term was included in the model to control for natural inter-annual variation (*year*). This variation was modeled as a random effect with a mean of zero and a variance of σ_{year}. This variation was modelled to affect all sites because a site-specific random effect would have the potential of incorrectly assigning variation to the random *year* term rather than the *site:period* BACI term. The model can be written mathematically as:

(13) $$E(y_{spy}) = \beta_0 + \beta_s + \beta_p + \beta_{s:p} + \varepsilon_y + \varepsilon$$

where β_0 is the intercept, β_s is the expect value for site s, β_p is the expected value for period p, $\beta_{s:p}$ is the expected value for site s in period p, ε_y is the term for inter-annual variation, and ε is the residual variation not described by the other terms.

The inclusion of the random *year* term was assessed using nested models and a statistical hypothesis test. A nested model omitting the *year* term (Equation 12) was compared to the full model (Equation 13) using a likelihood ratio test. The more complex model was retained only if the likelihood ratio test showed a significant (a = 0.05) improvement overall model performance.

Baseline Data

Baseline data collection began in 1994 at some of the reference sites and was expanded to include sites in the Pigeon-Fay and Upper Exeter Watershed in 2001. From 2008 to 2010, data from Fay Bay and Upper Exeter Lake were collected as part of a monitoring program in response to an unplanned release of fine processed kimberlite (FPK) in May of 2008 (Rescan 2011b). Given the potential for confounding effects as a result of the unplanned release of FPK, water quality data collected from 2008 to 2013 were not used in the statistical analyses. Thus, baseline data encompasses data from 1994 to 2007 and is referred to as the "before" period in the BACI analysis. Monitoring data includes all data collected in 2014 and is referred to as the "after" period in the BACI analysis. Although stream water quality sites in the Pigeon-Fay and Upper Exeter Watershed would not have been affected by the unplanned release of FPK, the same "before" and "after" periods were used for the BACI analysis of stream and lake data in order to facilitate comparisons between stream and lake sites. This comparison is an essential component of the evaluation of effects in the Pigeon-Fay and Upper Exeter Watershed because Pigeon Stream represents the main source of potential inputs from the PSD to Fay Bay. The temporal coverage of observations varied across sites and years due to the availability of data and the extent of censoring due to analytical detection limits.

2.2.5.2 Hypothesis Testing

A mine-related effect would be expected to result in a significant difference between the monitored site and the reference sites in the after period. The *site* and *period* terms in the BACI model control for natural variability between sites as well as regional-scale variation. The *site:period* term describes the site-specific variability that was observed in the after period. For each monitored site, the *site:period* term was assessed in a hypothesis test using Wald-type chi-square tests based on normal approximation for maximum likelihood estimation. If the p-value for this *site:period* hypothesis test was less than a = 0.05, then it was concluded that a significant site-specific difference between the before and after periods was observed.

Confidence Intervals for Contrast Terms

However, the crucial site-specific differences would be the difference(s) between the monitored site and the reference sites. The Wald hypothesis test does not identify which site-specific differences were significantly different. The modelling estimates values for each site in each period in the $\beta_{s:p}$ term of the mathematical formulation of the model (Equation 12). Contrasts were calculated to compare the difference in the $\beta_{s:p}$ term between the monitored site and each reference site. In this approach, any contrast substantially different from zero would represent a difference between the monitored site and the reference site currently being contrasted.

These contrasts were estimated in the modelling and are therefore subject to uncertainty. Confidence intervals for the contrasts were calculated to support the interpretation of the contrasts and, in turn, support the identification of significant site-specific differences. Parametric bootstrapping was conducted to generate 999 independent contrast estimates. From these independent bootstrap simulations, 95% interquantile ranges were calculated to provide a confidence interval for contrast estimates. If the confidence interval of a contrast estimate did not include zero, then it was concluded that a significant site-specific difference between the *before* and *after* periods was observed between the monitored site and that particular reference site.

2.2.5.3 Plots of Observed and Modelled Values

Plots of the observed and fitted values for each variable were created to visualize the variation between sites and the overall performance of the statistical model. On these plots, the observed mean values of the variable for each site in each year are shown as symbols. Sites located downstream of potential mine effects were assigned colours from a heat colour palette corresponding to the distance downstream. Red-coloured sites are closer to the source of potential mine effects and blue-coloured sites are further downstream. Observations below analytical detection limits were substituted with half the analytical detection limit for the calculation of annual means.

Fitted values from the models were plotted as the vertices of the lines labelled with each site. The error bars show the 95% interquantile range of bootstrapped fitted values from 999 simulations. These bootstrapped fitted values provide an estimate of the performance of the model for describing the observed values.

2.2.5.4 Assumptions and Interpretation of Results

Conclusions on the effects of the mine on the Pigeon-Fay and Upper Exeter Watershed were drawn from the results of the hypothesis testing, the contrast analysis, and the graphical presentation of the observations and modelled data. The conclusions are based on the change in the monitored sites between the before and after periods relative to the changes observed in the reference sites. The fundamental assumption in this approach is that any changes between the before and after periods would be similar at all sites. Any changes specific to the monitored sites would therefore be assigned to mine effects. However, to control for natural variability, site-specific differences were interpreted as potential mine effects only if a monitored site differed significantly from at least two reference sites, on the basis of the contrast terms. Furthermore, the identification of a potential mine effect on the environment included an assessment of observed variables with respect to benchmarks and biological trends (Section 2.2.2).

2.2.5.5 Computing

All steps of the analysis were performed using statistical computing package R 3.1.2 (R Development Core Team 2014). The following versions of packages were used in the analyses:

- plyr (1.8.1);
- AICcmodavg (2.0-1);
- boot (1.3-13)
- lme4 (1.1-7);

- Rcpp (0.11.3);
- Matrix (1.1-4);
- nlme (3.1-118);
- lattice (0.20-29);
- reshape2 (1.4.1);
- grid (3.1.2);
- MASS (7.3-35);
- minqa (1.2.4);
- nloptr (1.0.4);
- reshape (0.8.5);
- splines (3.1.2);
- stats4 (3.1.2);
- stringr (0.6.2);
- tools (3.1.2);
- unmarked (0.10-4); and
- VGAM (0.9-6).

Results from the statistical tests are provided in Part 3 – Statistical Report.

2.2.6 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.

2.2.6.1 Visual Gradient

The two main point sources for potential water quality effects in the receiving environment at the Ekati Diamond Mine are discharge from the LLCF into Leslie Lake and discharge from the KPSF into Cujo Lake. In 2014, monitoring in the Pigeon-Fay and Upper Exeter Watershed began, with outflow from the PSD through the Pigeon Stream and into Fay Bay being the primary potential source of changes in water quality. Historical data are therefore presented by location within a given watershed, in order to identify and assess how these three sources may be affecting downstream lakes and streams. Lakes and streams located downstream of discharge sites (i.e., the LLCF or KPSF) or outflow sites (i.e., the PSD) are assigned colors from a red to blue heat palette that correspond to distance from the discharge or outflow site, with red representing close proximity to the LLCF, KPSF, or PSD and blue representing sites that are furthest downstream of the LLCF, KPSF, or PSD. This enables the identification of concentration gradients from the point sources and allows the overall downstream distance of effects to be determined. All evaluated variables are visually analyzed for spatial gradient trends.

2.2.6.2 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 nitrate-N concentrations in Kodiak Lake and the trend in nitrate-N concentrations in reference lakes in April of 2013. Visual analysis of the historical trend in nitrate-N concentrations indicated that nitrate-N concentrations in Kodiak Lake had declined from initially high levels to stabilise at current levels. Thus, the statistical difference in the trend in Kodiak Lake was attributed to this decline, which was not observed in reference lakes.

2.2.6.3 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 2014 had to appear different from all of the data collected in baseline years to be considered an effect. For example, if values at monitored sites in 2014 appeared different from those in 1996 but not from 1994, no mine effects were indicated. However, if values from 2014 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 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.

2.2.7 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 DDEC 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 2014c) and site specific water quality objectives (SSWQOs; Table 2.3-1).

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 CCME guideline values: pH, total suspended solids (TSS), 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 CCME guidelines for cadmium, copper, and nickel are hardness-dependent with minimum values of 0.0004 mg/L for cadmium, 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 2.3-1. The Ekati Diamond Mine Water Quality Benchmarks Used for the AEMP Evaluation of Effects

Variable	Source	Benchmark Value	Notes
Physical/Ion			
Dissolved Oxygen (DO)	CCME (1999d)	6.5 mg/L[1]	
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 2012e)	$e^{(0.9116 \times \ln(\text{hardness}) + 1.712)}$ (where hardness < 160)	Hardness as mg/L $CaCO_3$
Potassium	SSWQO (Rescan 2012f)	41	
Total Suspended Solids (TSS)	CCME (1999e)	Maximum average increase of 5 mg/L from background levels (long term exposure)[2]	
Nutrients/Organics			
Total Ammonia-N	CCME (2001b)	Dependent on pH and temperature (see Table 2.3-3)	
Nitrate-N	SSWQO (Rescan 2012d)	$e^{(0.9518 [\ln(\text{hardness})] - 2.032)}$ (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 (1999c)	0.005	
Barium	Haywood and Drinnan (1983)	1	
Boron	CCME (2009)	1.5	
Cadmium	CCME (2014a)	$10^{(0.83 \times \log10(\text{hardness}) - 2.46)}/1000$ (with minimum = 0.0004 where hardness = 0-16 and maximum = 0.00037 where hardness > 280)	Hardness as mg/L $CaCO_3$

(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*)			
Copper	CCREM (1987)	$e^{(0.8545 \times \ln(hardness) - 1.465)} * 0.2/1000$ (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)	$e^{(0.76 \times \ln(hardness) + 1.06)}/1000$ (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 2012g)	0.03	

Note: Units are mg/L unless otherwise specified.
[1] *Benchmark value is for non-early life stages. Where baseline concentrations < 110% of the guideline, the benchmark concentration is 90% of the baseline concentration.*
[2] *Benchmark value is for freshwater systems with clear flow. Maximum average increase for high flow systems is 25 mg/L when background levels are from 25 – 250 mg/L and ≤ 10% of background levels when background ≥ 250 mg/L.*

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

Temperature (°C)	pH							
	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

Notes: 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. In such cases, other environmental factors should be considered and further assessment may include the recommendation of remediation or restoration (CCME 2004). The 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 considered benchmarks for the management of phosphorus at the Ekati Diamond Mine in lakes downstream of the LLCF, KPSF, and PSD (Table 2.3-4).

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)	0.0069
Fay Bay	0.0093
Upper Exeter	0.0053

Site specific water quality objectives used in the 2014 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, 2012d, 2012e, 2012f, 2012g). 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 2012e). 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 2012d). The SSWQO for potassium, molybdenum, and vanadium are not hardness-dependent.

2.4 CCME Sediment Quality Guidelines

The CCME sediment quality guidelines for the protection of aquatic life (CCME 2014b) are compared to AEMP sediment data to aid in the evaluation of effects and determine whether potential changes in sediment quality may result in mine impacts. Currently arsenic, cadmium, and copper have CCME sediment quality guidelines (Table 2.4-1). CCME guidelines include the Interim Sediment Quality Guidelines (ISQG) and the Probable Effect Levels (PEL), which provide an interpretive tool for evaluating the toxicological significance of sediment chemistry data (CCME 2014b). Sediment chemical concentrations below the ISQG are not expected to adversely affect biotic communities; however, concentrations above the PEL are expected to be associated with adverse effects on biotic communities. There are no site specific sediment quality objectives.

Table 2.4-1. CCME Sediment Quality Guidelines Used for the AEMP Evaluation of Effects

Variable	Source	CCME Guideline (ISQG)	CCME Guideline (PEL)
Arsenic	CCME (2001a)	5.9	17
Cadmium	CCME (1999a)	0.6	3.5
Copper	CCME (1999b)	35.7	197

Notes: Units are mg/kg.
ISQG = Interim Sediment Quality Guideline
PEL=Probable Effects Level

3. EVALUATION OF EFFECTS: KOALA WATERSHED AND LAC DE GRAS

3.1 PHYSICAL LIMNOLOGY

3.1.1 Variables

Two physical limnology variables were evaluated for potential effects caused by mine activities: under-ice 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 2014c). The guideline of 6.5 mg/L is more applicable to the biota living in the water column under-ice in lakes. The guideline of 9.5 mg/L was established to protect salmonid larvae in redds and is therefore more applicable to DO concentrations at the lake bottom (CCME 1999d). Although temperature is not an evaluated variable, it was measured in conjunction with under-ice DO. Water temperature is an important factor in understanding changes in DO, as the solubility of oxygen decreases with increasing water temperatures.

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).

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	-	-

(continued)

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 (completed)

Year	Nanuq	Counts	Vulture	Grizzly	Kodiak	Leslie	Moose	Nema	Slipper	S2	S3
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
2014	Apr-7	Mar-31	Apr-8	Apr-5	Apr-4	Apr-13	Apr-13	Apr-12	Apr-9	Apr-9	Apr-9

Note: 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
2009	Jul-30	Aug-1	Jul-30	Aug-2	Aug-8	Aug-5	Jul-30	Jul-30	Aug-3	Jul-31	Jul-31

(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
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
2014	Aug-5	Aug-9	Aug-3	Aug-10	Jul-29	Jul-31	Jul-31	Aug-2	Aug-4	Jul-30	Jul-30

Note: Dashes indicate no data were available.

3.1.3 Results and Discussion

3.1.3.1 Under-ice Dissolved Oxygen and Temperature

Summary: Under-ice temperature profiles have changed through time in Kodiak, Grizzly, Leslie, Nema, and possibly Moose and Slipper lakes. In Kodiak Lake, the observed changes in the under-ice temperature profiles likely stem from attempts to increase under-ice DO using aerators from 1997 to 2006. The causes of the temporal changes in under-ice temperature profiles downstream of the LLCF and in Grizzly Lake are unclear at this time. Though the patterns are not as pronounced, there is some evidence of a cooling trend in reference lakes in recent years, suggesting that changes in thermal profiles may reflect natural climactic variability. With the exception of decreased under-ice DO concentrations observed in Kodiak Lake over time (likely a 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.

No statistical analyses could be performed on under-ice DO 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.

DO concentrations measured in early April of 2014 were generally within the historical ranges observed in each lake (Figures 3.1-1 and 3.1-2). DO concentrations were typically greatest just below the surface ice and declined with depth and as temperature increased. Under-ice DO concentrations in 2014 were slightly greater than those observed in recent years in Leslie, Moose, Nema, and Slipper lakes, which coincided with a cooling trend observed in these lakes in recent years. Under-ice DO concentrations were greater than the CCME guideline value of 6.5 mg/L throughout the water column in all monitored lakes downstream of the LLCF, at sites S2 and S3 in Lac de Gras, and in Grizzly Lake (CCME 2014c). 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). While DO has generally remained above CCME guidelines throughout the top three to five meters of the water column, DO concentrations were slightly lower than historical levels at shallower depths and higher than historical levels at deeper depths (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 2014 (Figure 3.1-2b). However, 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 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 greater availability of oxygen in the upper portions of the water column.

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 3.1-2e). Under-ice temperature profiles have also shown some indication of a cooling trend in recent years in Moose and Slipper lakes (Figure 3.1-2d, and 3.1-2f); however, temperature profiles have been more variable in Moose Lake than in other lakes, likely owing to its comparatively small volume. There is also some evidence of a general cooling trend, at all depths, in two of the reference lakes (i.e., Nanuq and Counts lakes) in recent years (Figures 3.1-1a and 3.1-1b), though the patterns are more variable and less pronounced. Overall, the trends in reference lakes suggest that shifts in temperature profiles in monitored lakes may reflect natural climatic variability. Temperature profiles in 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 3.1-2g and 3.1-2h).

In Grizzly Lake, under-ice temperature profiles from 2011to 2013 differed from previous years, with surface waters being cooler than in previous years (Figure 3.1-2a). In addition, under-ice temperatures showed a pattern of increase with increasing depth from 2010 to 2013, rather than remaining homogeneous throughout the water column as in previous years. In 2014, surface waters in Grizzly Lake were warmer than those observed from 2010 to 2013, but the pattern of increasing temperature with increasing depth remained present (Figure 3.1-2a). There were no associated changes in the DO profiles in Grizzly Lake from 2010 to 2014 (Figure 3.1-2a). It is unclear why the observed thermal profile in Grizzly Lake may have changed in recent years. 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.

3.1.3.2 Secchi Depth

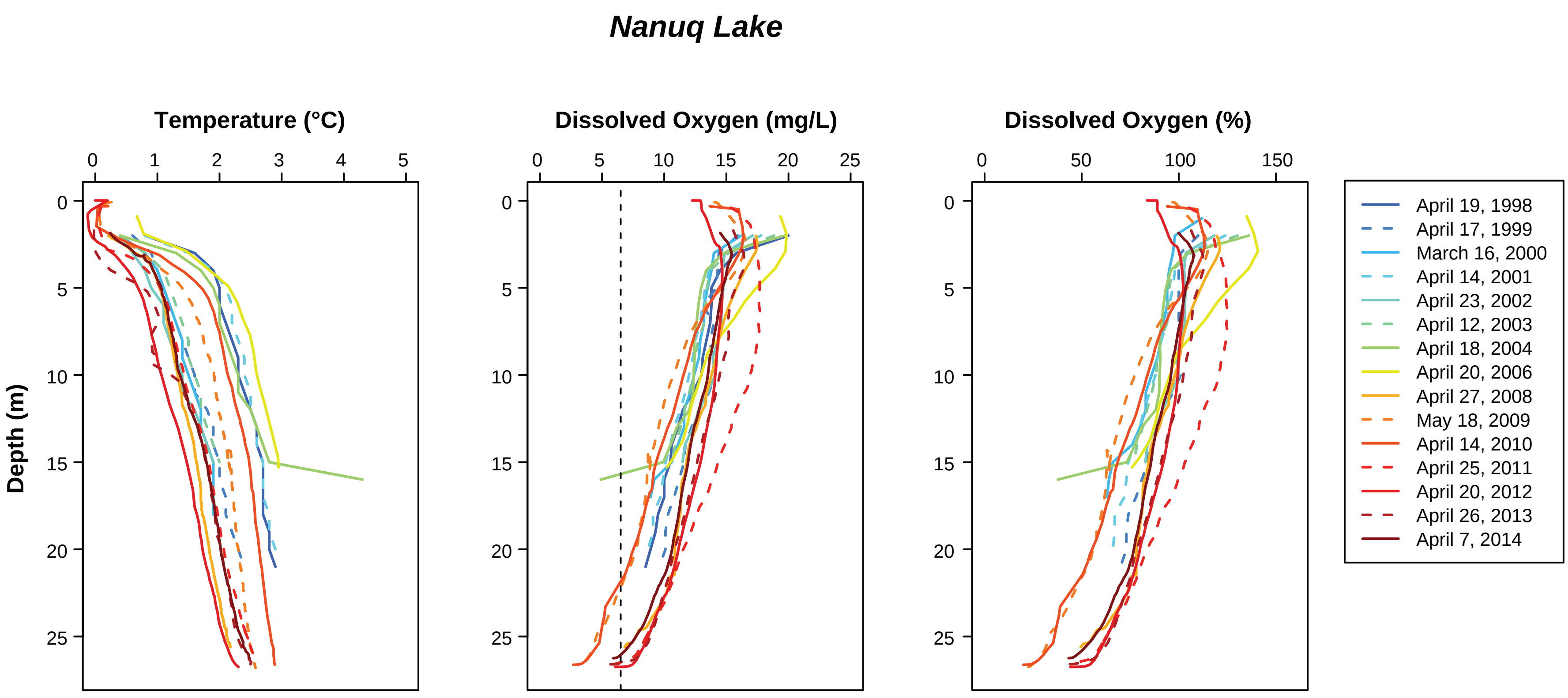
Secchi depth is an indicator of underwater light conditions in lakes. It can be used as an indicator of changes in water quality or plankton density. Graphical analysis and best professional judgment were used to evaluate if a significant change in Secchi depth occurred in monitored lakes of the Koala 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 estimated error for each year, observed August 2014 Secchi depths were similar to those observed in baseline years in all monitored lakes, except for Moose and Slipper lakes in which 2014 Secchi depths were shallower than in baseline years (Figure 3.1-3). However, a similar pattern was observed in one of the reference lakes (i.e., Nanuq Lake). Thus, no mine effects were detected with respect to Secchi depth in monitored lakes of the Koala Watershed or Lac de Gras.

Figure 3.1-1a

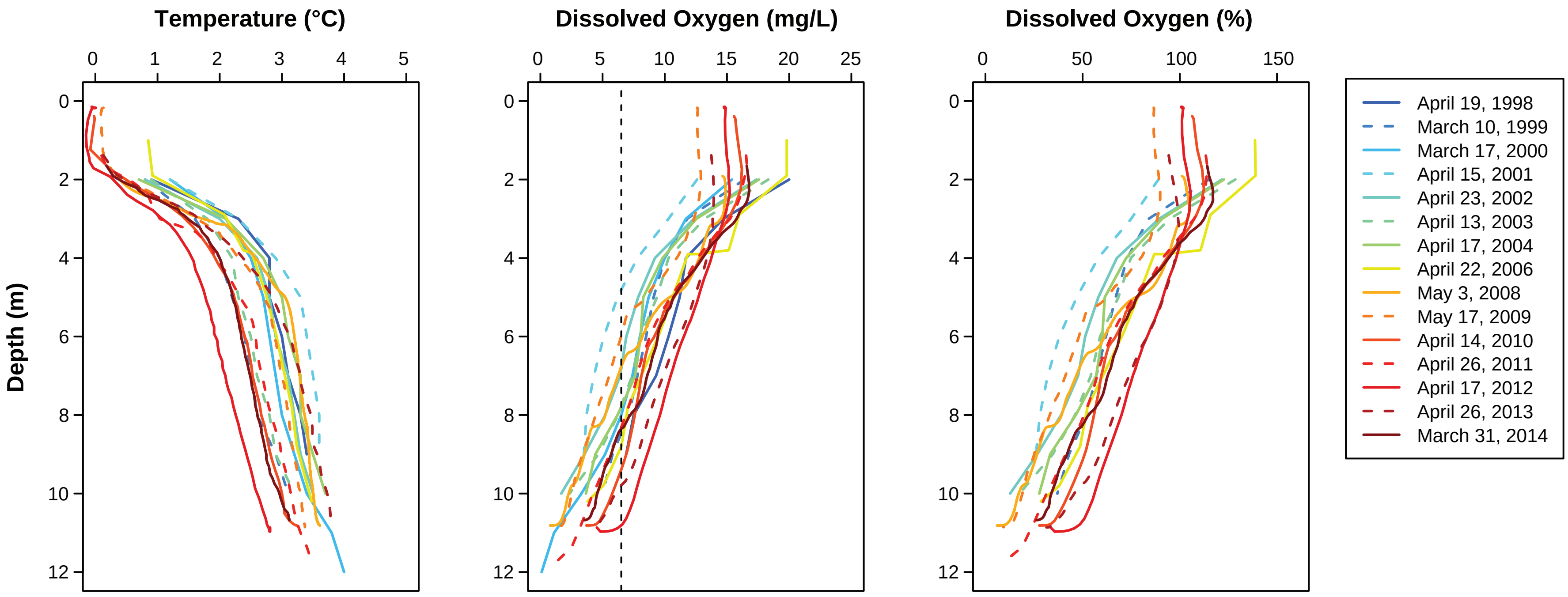
Under-ice Dissolved Oxygen and Temperature Profiles for AEMP Reference Lakes, 1998 to 2014

ERM



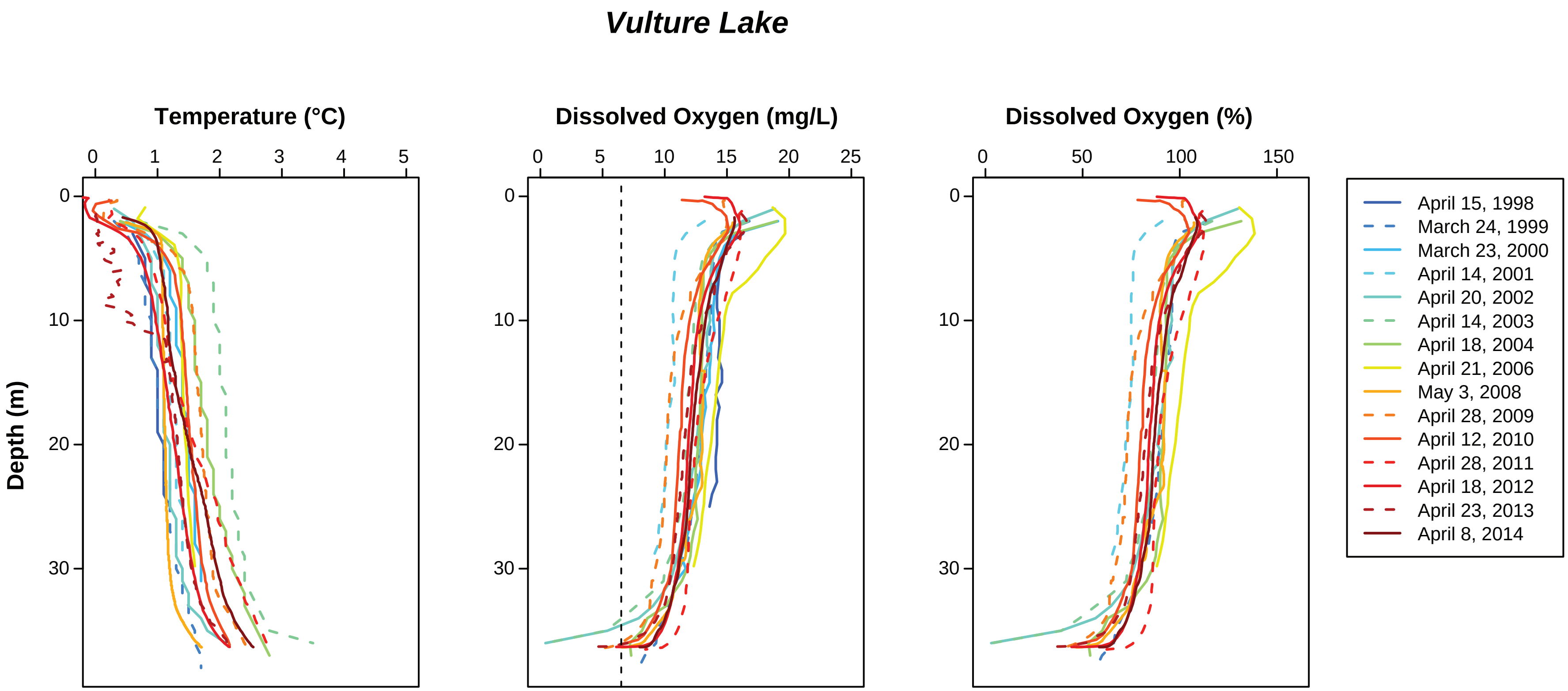
Note: Vertical dashed line represents the CCME guideline for non-early life stages (6.5 mg/L).
Data collected and supplied by DDEC.

Counts Lake



Note: *Vertical dashed line represents the CCME guideline for non-early life stages (6.5 mg/L).*
Data collected and supplied by DDEC.

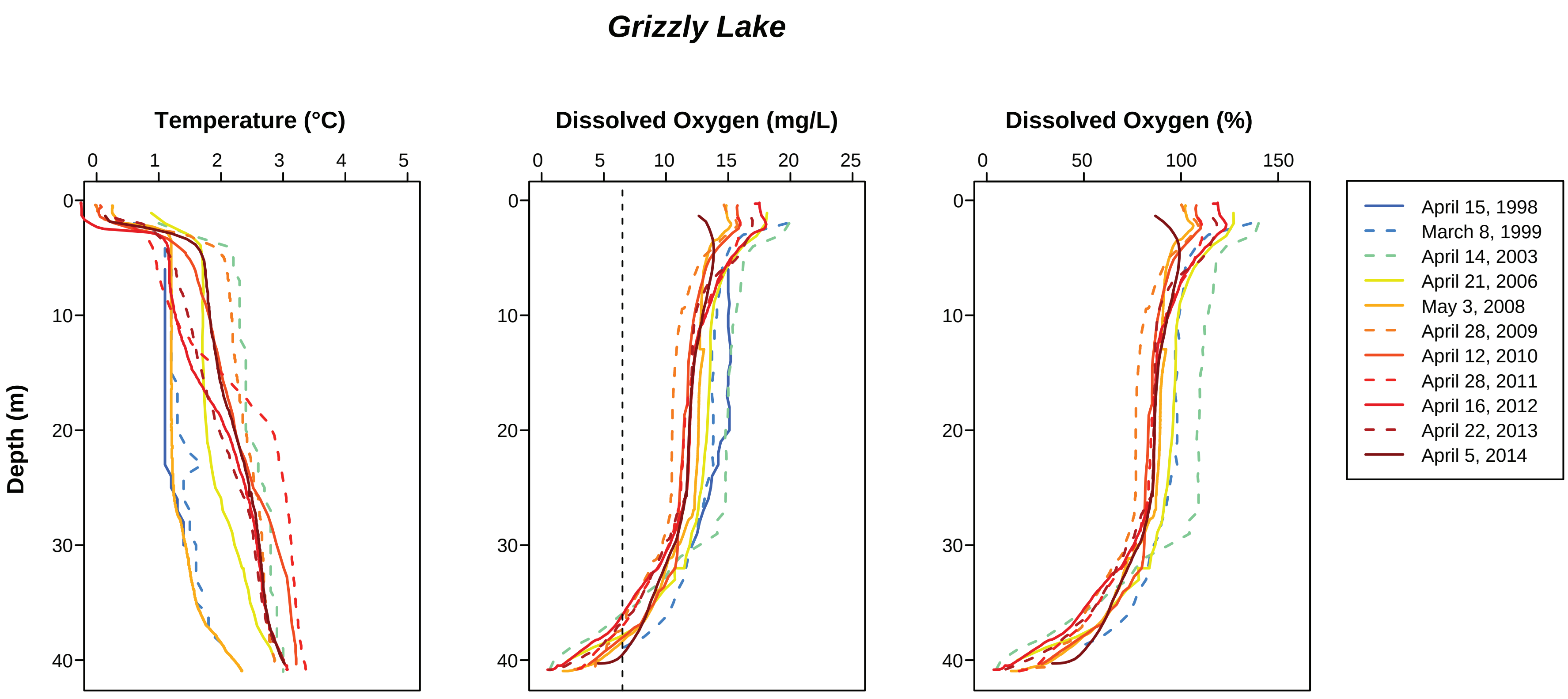
Figure 3.1-1c
Under-ice Dissolved Oxygen and Temperature Profiles for AEMP Reference Lakes, 1998 to 2014



Note: Vertical dashed line represents the CCME guideline for non-early life stages (6.5 mg/L).
Data collected and supplied by DDEC.

Under-ice Dissolved Oxygen and Temperature Profiles for Koala Watershed Lakes and Lac de Gras, 1994 to 2014

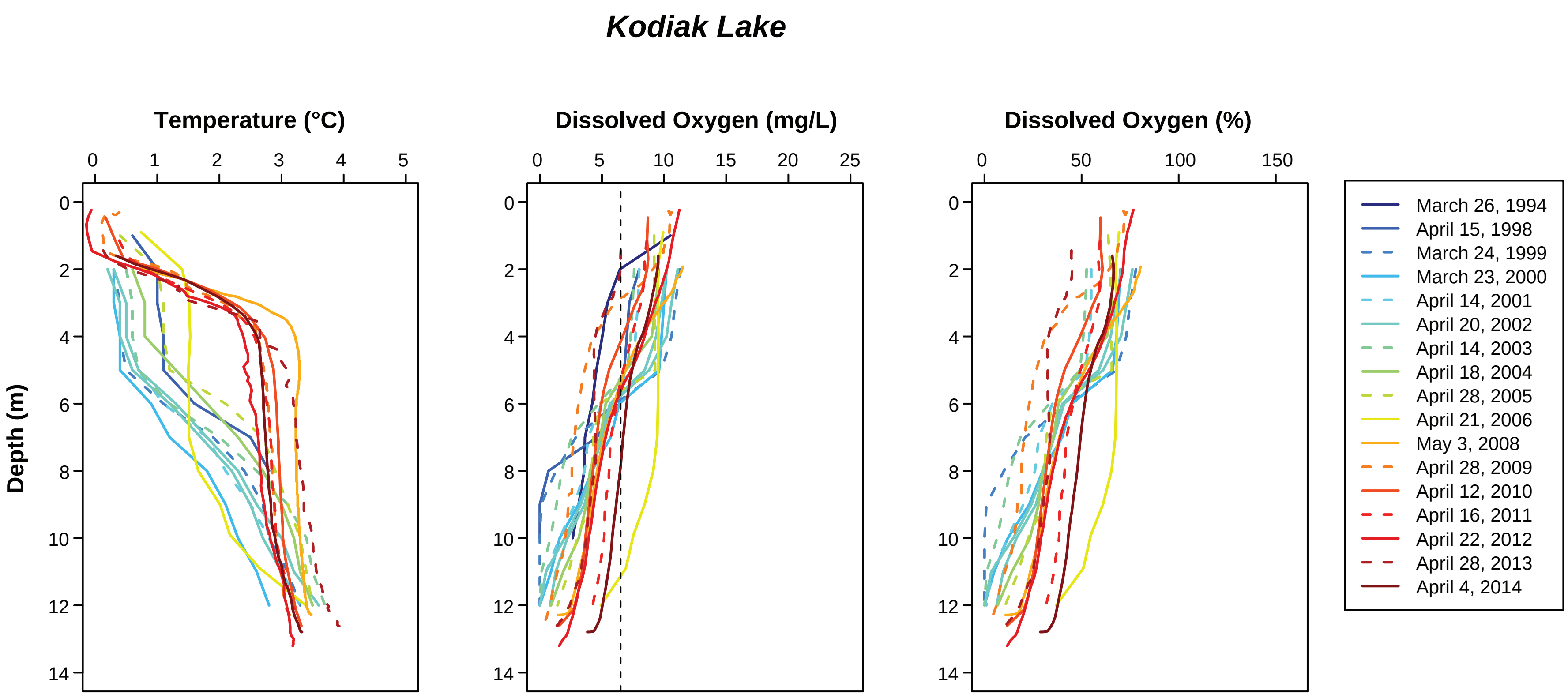
ERM



Note: Vertical dashed line represents the CCME guideline for non-early life stages (6.5 mg/L).
Data collected and supplied by DDEC.

Figure 3.1-2b

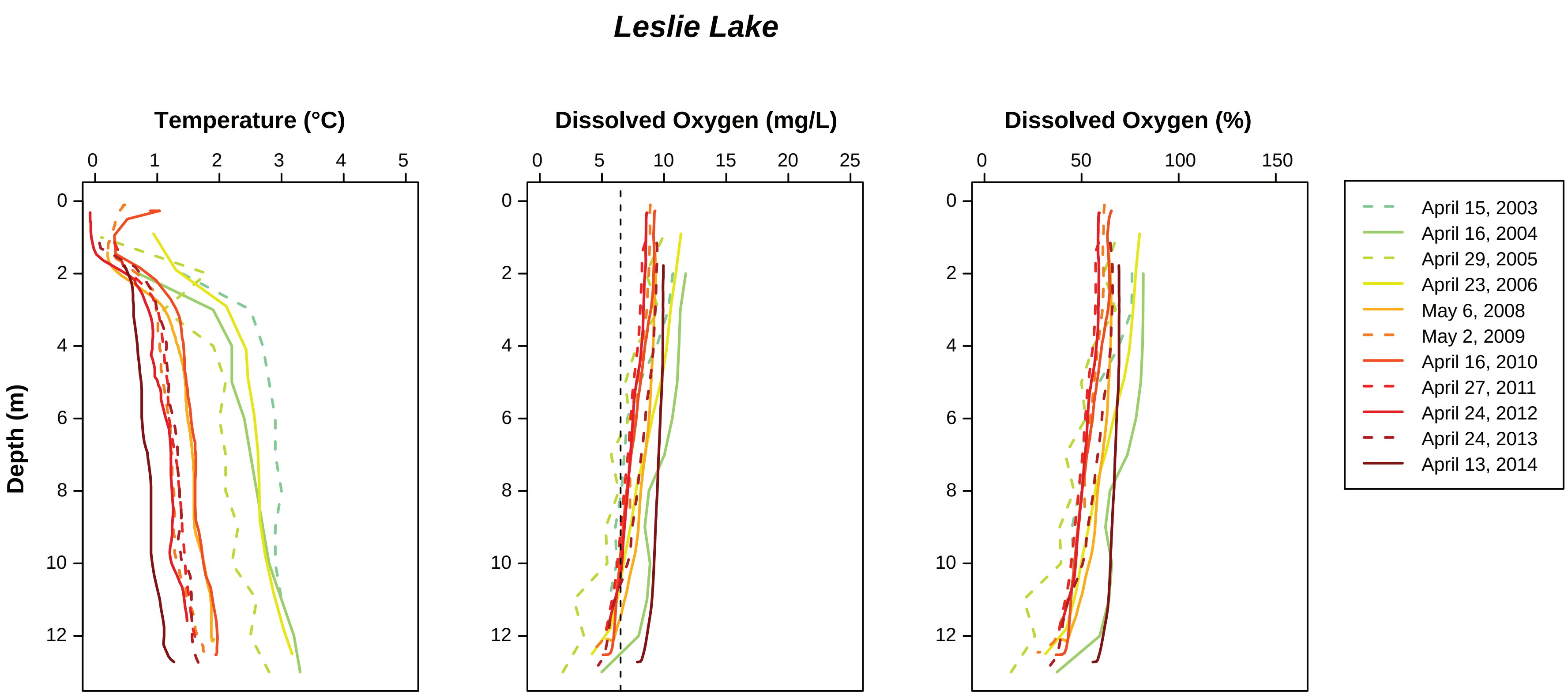
Under-ice Dissolved Oxygen and Temperature Profiles for Koala Watershed Lakes and Lac de Gras, 1994 to 2014



Note: Vertical dashed line represents the CCME guideline for non-early life stages (6.5 mg/L).
Data collected and supplied by DDEC.

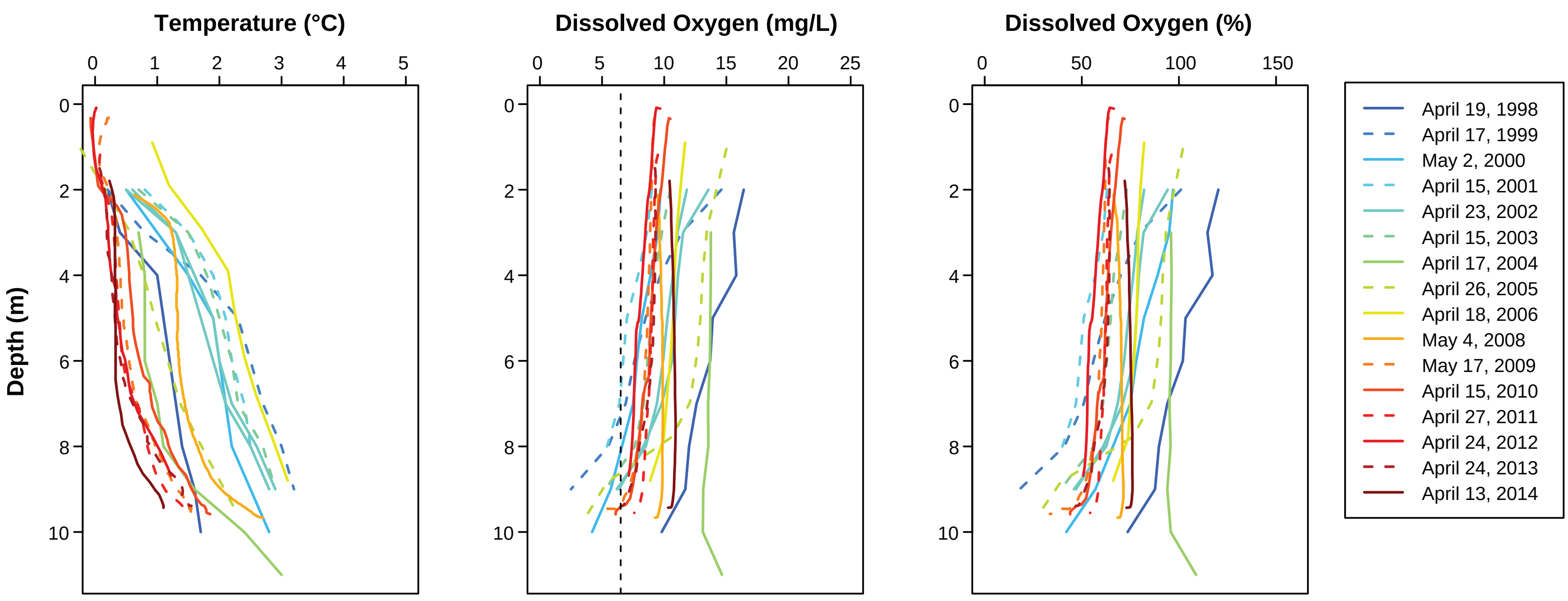
Figure 3.1-2c

Under-ice Dissolved Oxygen and Temperature Profiles for Koala Watershed Lakes and Lac de Gras, 1994 to 2014

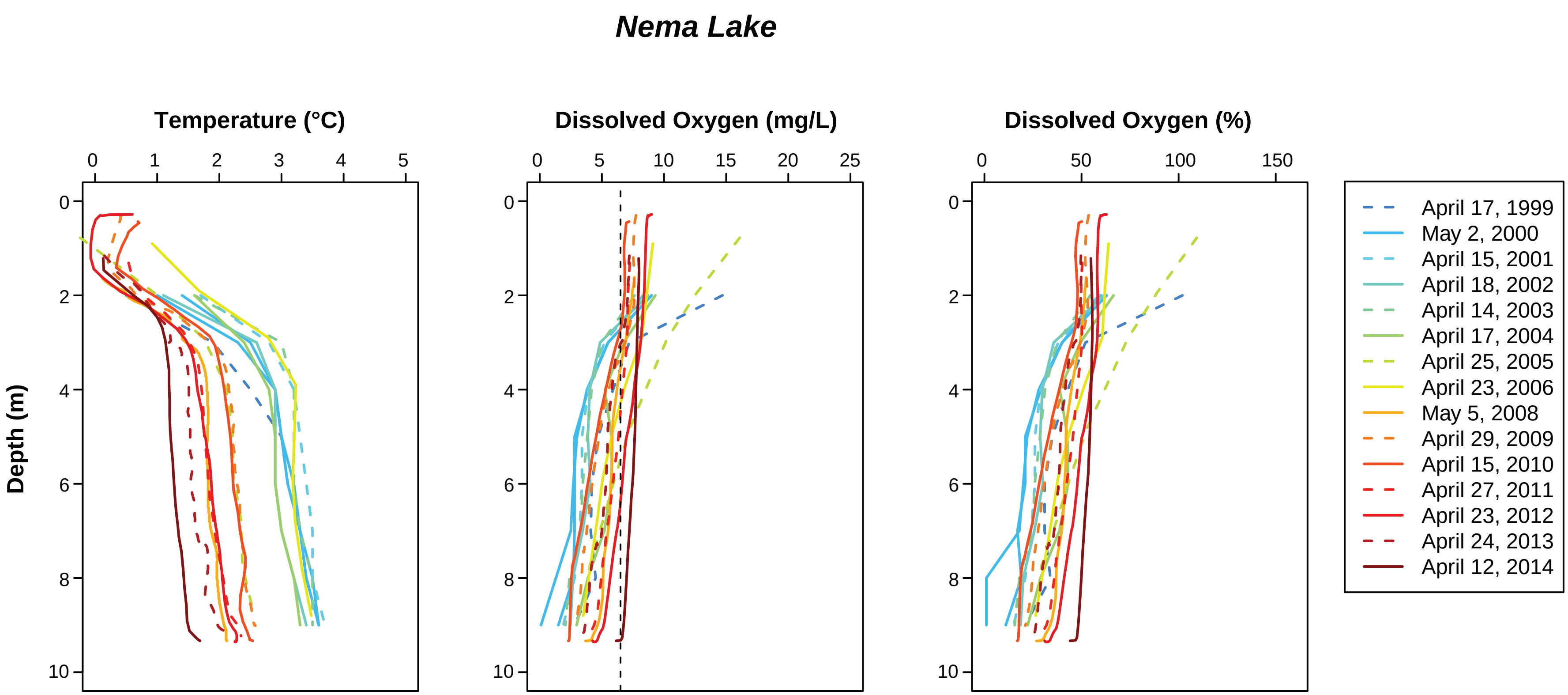


Note: Vertical dashed line represents the CCME guideline for non-early life stages (6.5 mg/L). Data collected and supplied by DDEC.

Moose Lake



Note: Vertical dashed line represents the CCME guideline for non-early life stages (6.5 mg/L).
Data collected and supplied by DDEC.

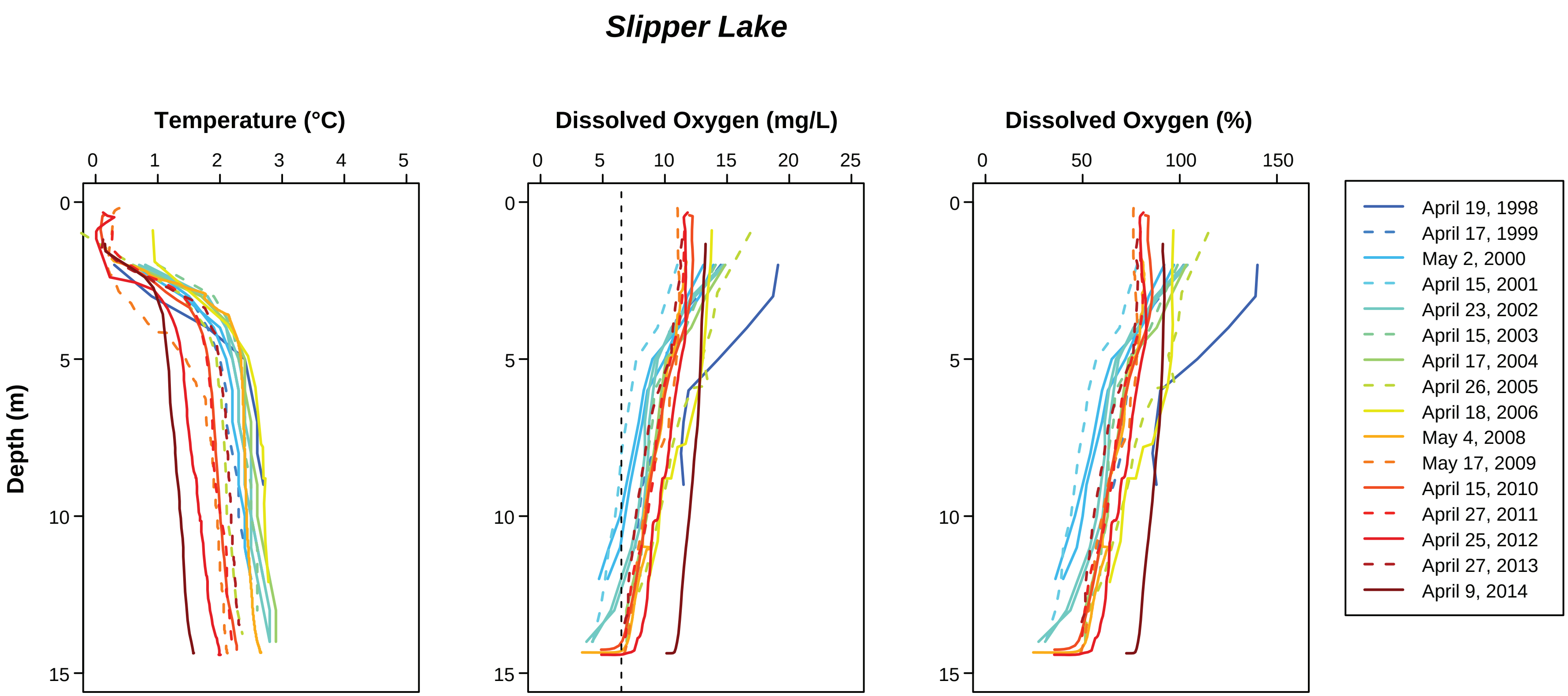


Note: Vertical dashed line represents the CCME guideline for non-early life stages (6.5 mg/L).
Data collected and supplied by DDEC.

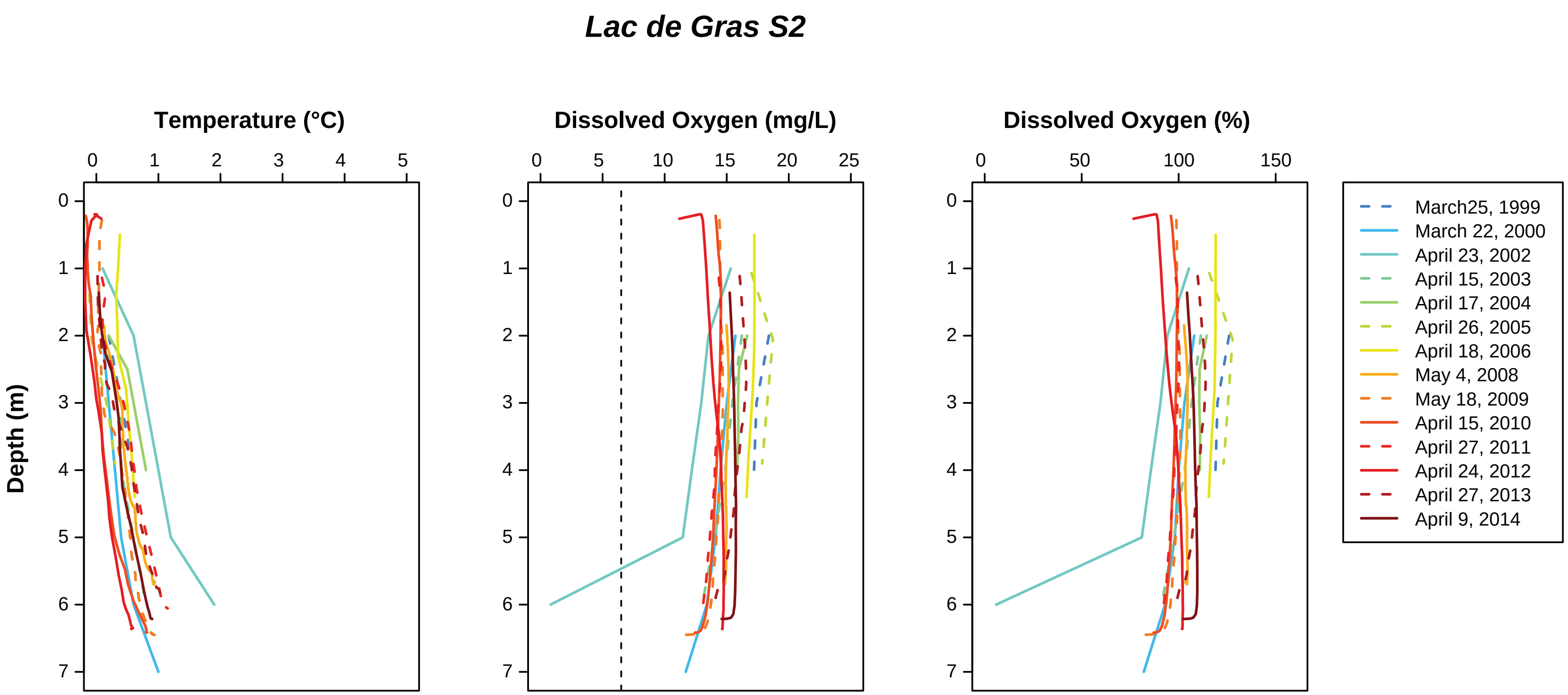
Figure 3.1-2f

Under-ice Dissolved Oxygen and Temperature Profiles for Koala Watershed Lakes and Lac de Gras, 1994 to 2014

ERM

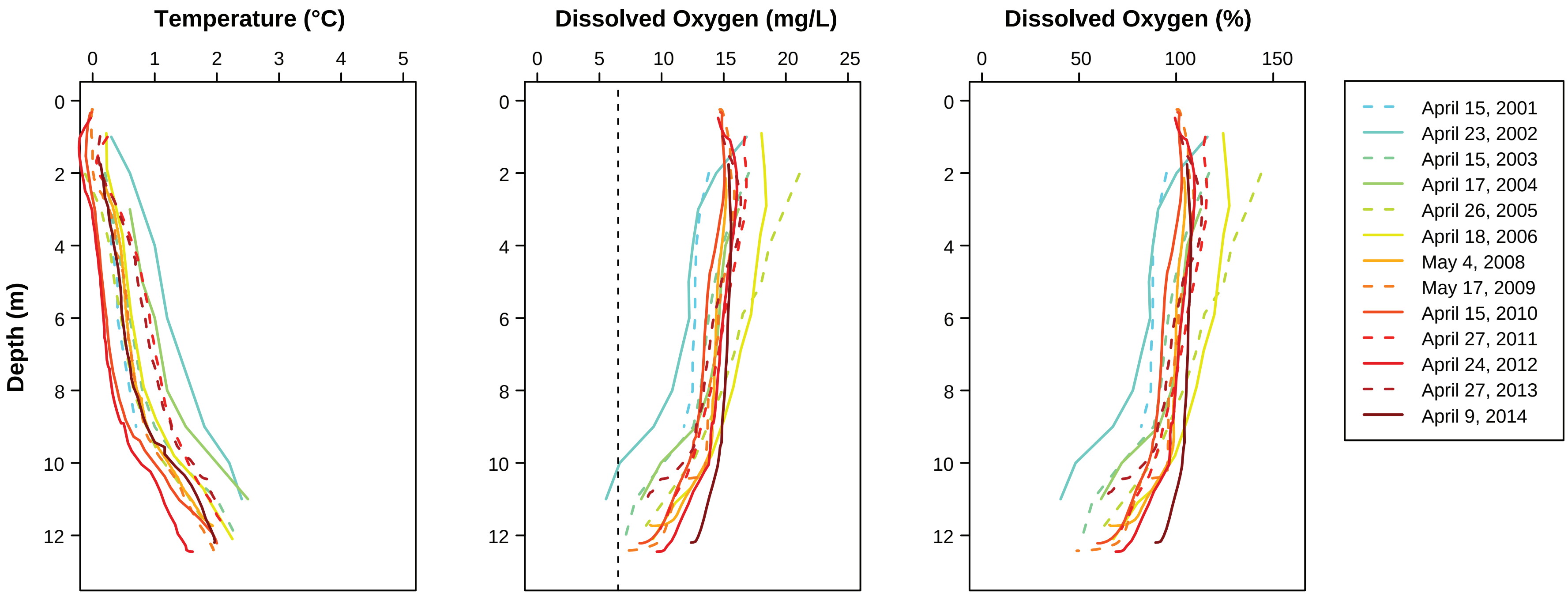


Note: Vertical dashed line represents the CCME guideline for non-early life stages (6.5 mg/L).
Data collected and supplied by DDEC.



Note: Vertical dashed line represents the CCME guideline for non-early life stages (6.5 mg/L).
Data collected and supplied by DDEC.

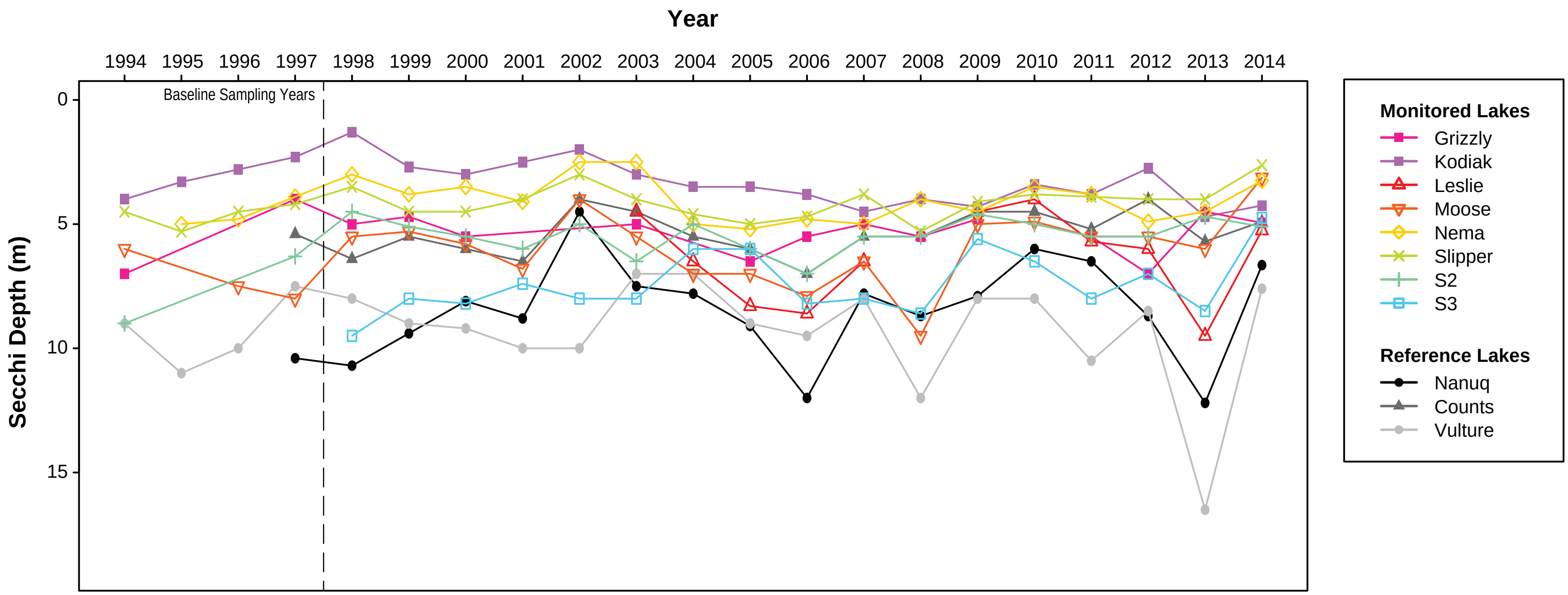
Lac de Gras S3



*Note: Vertical dashed line represents the CCME guideline for non-early life stages (6.5 mg/L).
Data collected and supplied by DDEC.*

Figure 3.1-3

August Secchi Depths for Koala Watershed Lakes and Lac de Gras, 1994 to 2014



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, King-Cujo, and Pigeon-Fay and Upper Exeter 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). A 23rd variable was also evaluated in the King-Cujo Watershed (total copper) and in the Pigeon-Fay and Upper Exeter Watershed (total suspended solids; TSS).

CCME guidelines for the protection of aquatic life exist for 12 of the evaluated water quality variables, including pH, TSS, 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 2014c). 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 include provincial guidelines or ones taken from the published literature (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 2014c). 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 and Ekati Diamond Mine SSWQOs 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).

Total Suspended Solids (TSS)

TSS is the concentration of suspended matter in water and includes all particles suspended in water that will not pass through a filter. The size of suspended particles is often influenced by the velocity or wave action of water. Common sources of TSS include soil erosion and runoff, disturbed bottom sediments, and algal blooms. High concentrations of TSS can lead to increases in water temperature as particles absorb heat from solar radiation. Such increases in water temperature can then result in lower concentrations of dissolved oxygen and reduced water clarity (i.e., increases euphotic depth). Nutrients and metals can also bind to suspended particles, leading to increases in these water quality variables.

Nutrients

Nutrients – especially macronutrients such as carbon, nitrogen, and phosphorus - 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 phosphorus 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 DO 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, molybdenum, etc.), which are 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 *Ceriodaphnia 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, phosphorus 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 phosphorus 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).

Total Phosphate-P

Total phosphate is a combined measure of the inorganic and organic forms of phosphorus. Excess quantities of phosphorus 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 TOC, 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 DO concentrations in Cujo Lake. Consequently, baseline conditions are not defined.

Metals

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

3.2.2.1 Lakes

For each of the sampling years between 1998 and 2014, lake water quality data were collected for the evaluation of effects between mid-April and mid-May during the ice-covered season (Table 3.2-1) and between late July and mid-August during the open water season (Table 3.2-2). Baseline water quality data, collected from 1994 to 1997, are included in the data summary tables (Tables 3.2-1 and 3.2-2) and illustrated in Figures 3.2-1 to 3.2-22 for visual comparison, but were not included in the statistical evaluation of effects. Station 1616-30 (LLCF) is not sampled during the ice-covered season as part of the AEMP and was not included in the April (ice-covered) regression analysis. Water from Cell E of the LLCF was discharged into Leslie Lake from October 1 to November 19, 2013 and from July 28 to August 6, 2014. Therefore, August sampling is representative of post-discharge water quality in the 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 AEMP Re-evaluation (Rescan 2010c). Historical lake water quality data – including all sampling events – is presented graphically in Section 6 of this report. Summaries of the 2014 April and August lake water quality data are provided in Part 2 of the AEMP (Data Report).

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, and the timing of sample collection and analytical laboratory have been 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 two 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 2000, 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 2006 AEMP Re-evaluation (Rescan 2006), triplicate samples were collected from each depth in order to provide sufficient data for August-only sampling from 2007 to 2013. In 2014, the Wek'èezhìi Land and Water Board approved a reduction from triplicate to duplicate sampling for August lake water quality sampling after it was shown that a reduction in sampling replication would not significantly affect the ability to detect potential mine related effects (WLWB 2014).

Table 3.2-1. Dataset Used for Evaluation of Effects on the April (Ice-covered) Water Quality Koala Watershed Lakes and Lac de Gras

Year	Nanuq	Counts	Vulture	Grizzly	Kodiak	Leslie	Moose	Nema	Slipper	S2	S3
1994*	-	-	-	-	-	-	-	-	-	-	-
1995*	-	-	-	-	-	-	-	-	-	-	-
1996*	-	-	Apr-18 (1)	-	Apr-18 (1)	-	Apr-17 (1)	Apr-17 (1)	Apr-17 (1)	-	-
1997*	-	-	-	-	-	-	-	-	-	-	-
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)
2014	Apr-7 (4)	Mar-31 (4)	Apr-8 (4)	Apr-5 (4)	Apr-4 (4)	Apr-13 (4)	Apr-13 (4)	Apr-12 (4)	Apr-9 (4)	Apr-9 (4)	Apr-9 (4)

Notes: Dashes indicate no data were available
Number of samples 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*

Table 3.2-2. Dataset Used for Evaluation of Effects on the August (Open Water) Water Quality of 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 (6), Aug-14 (6)	Jul-27 (6), Aug-10 (6)	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)

(continued)

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

Year	Nanuq	Counts	Vulture	Grizzly	Kodiak	1616-30	Leslie	Moose	Nema	Slipper	S2	S3
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)
2014	Aug-5 (4)	Aug-9 (4)	Aug-3 (4)	Aug-10 (4)	Jul-29 (4)	Jul-28 (1), Aug-2 (2), Aug-4 (1), Aug-6 (2), Aug-25 (2)	Jul-31 (6)[2]	Jul-31 (4)	Aug-2 (4)	Aug-4 (4)	Jul-30 (4)	Jul-30 (4)

Notes: Dashes indicate no data were available.
Number of replicates is indicated in brackets.
[1] Three additional bottom depth samples were collected.
[2] Two 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; however, for some variables at some sites (e.g., nitrite-N in Leslie Lake), analytical detection limits have increased owing to required dilutions as conductivity in water quality samples has increased. Analytical detection limits for water quality variables are indicated as black dotted lines in Figures 3.2-1 to 3.2-22 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. Owing to potential changes in water column structure in Leslie Lake, additional water quality samples were also collected from the lower strata in Leslie Lake during August AEMP sampling beginning in 2013, to provide a more accurate depiction of open water-season water quality in the lake (Rescan 2013d). The lower strata water quality data was pooled with upper and middle strata water quality data collected from Leslie Lake in the evaluation of effects.

Over the years, data were removed from the dataset prior to analysis and interpretation as a result of contamination (Table 3.2-3).

3.2.2.2 Streams

Stream water quality data has been collected in June, August, and September of each year between 1998 and 2014. July stream water quality sampling was added to the AEMP program in 2010. In 2014, August stream sampling was representative of post-discharge water quality as water from Cell E of the LLCF was discharged into Leslie Lake from October 1 to November 19, 2013 and from July 28, 2014 to August 6, 2014. Thus, August 2014 samples were used for the evaluation of effects (Table 3.2-4). 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 2014. Baseline water quality data, collected from 1994 to 1997, are included in the data summary tables (Tables 3.2-2 and 3.2-4) and illustrated in Figures 3.2-1 to 3.2-22 for visual comparison, but were not used in the statistical evaluation of effects.

Table 3.2-3. 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
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
2014	August 2	Nema (shallow rep 2)	All	Unexplained contamination

Table 3.2-4. 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)

(continued)

Table 3.2-4. Dataset Used for Evaluation of Effects on the August (Open Water) Water Quality in Koala Watershed Streams and Lac de Gras (completed)

Year	Nanuq Outflow	Counts Outflow	Vulture-Polar	Lower PDC	Kodiak-Little	1616-30	Leslie-Moose	Moose-Nero	Nema-Martine	Slipper-Lac de Gras
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)
2014	Aug-1 (2)	Aug-4 (2)	Aug-1 (2)	Aug-2 (2)	Aug-4 (2)	Jul-28 (1), Aug-2 (2), Aug-4 (1), Aug-6 (2), Aug-25 (2)	Aug-4 (2)	Aug-2 (2)	Aug-4 (2)	Aug-4 (2)

Notes: 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 a different monitoring program.

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 3.2-4). 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 a different 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., five years) available for Leslie-Moose Stream in 2014 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 2014.

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 as a result of sample contamination (Table 3.2-3).

3.2.3 Statistical Description of Results

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 tables also indicate data, if any, that were excluded from the analysis. 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:

- <u>Model fit = 1a</u> 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.
- <u>Model fit = 1b</u> 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.
- <u>Model fit = 2</u> 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.

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 lower confidence interval of the fitted mean for Grizzly Lake was below the lower CCREM guideline value during the ice-covered season. However, the lower confidence intervals were also less than the lower CCREM guideline value in all reference lakes during the ice-covered season.

Statistical and graphical analyses indicate that pH has changed through time, relative to reference sites, at 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 3.2-5; Figure 3.2-1). Only five years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends unlikely; 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).

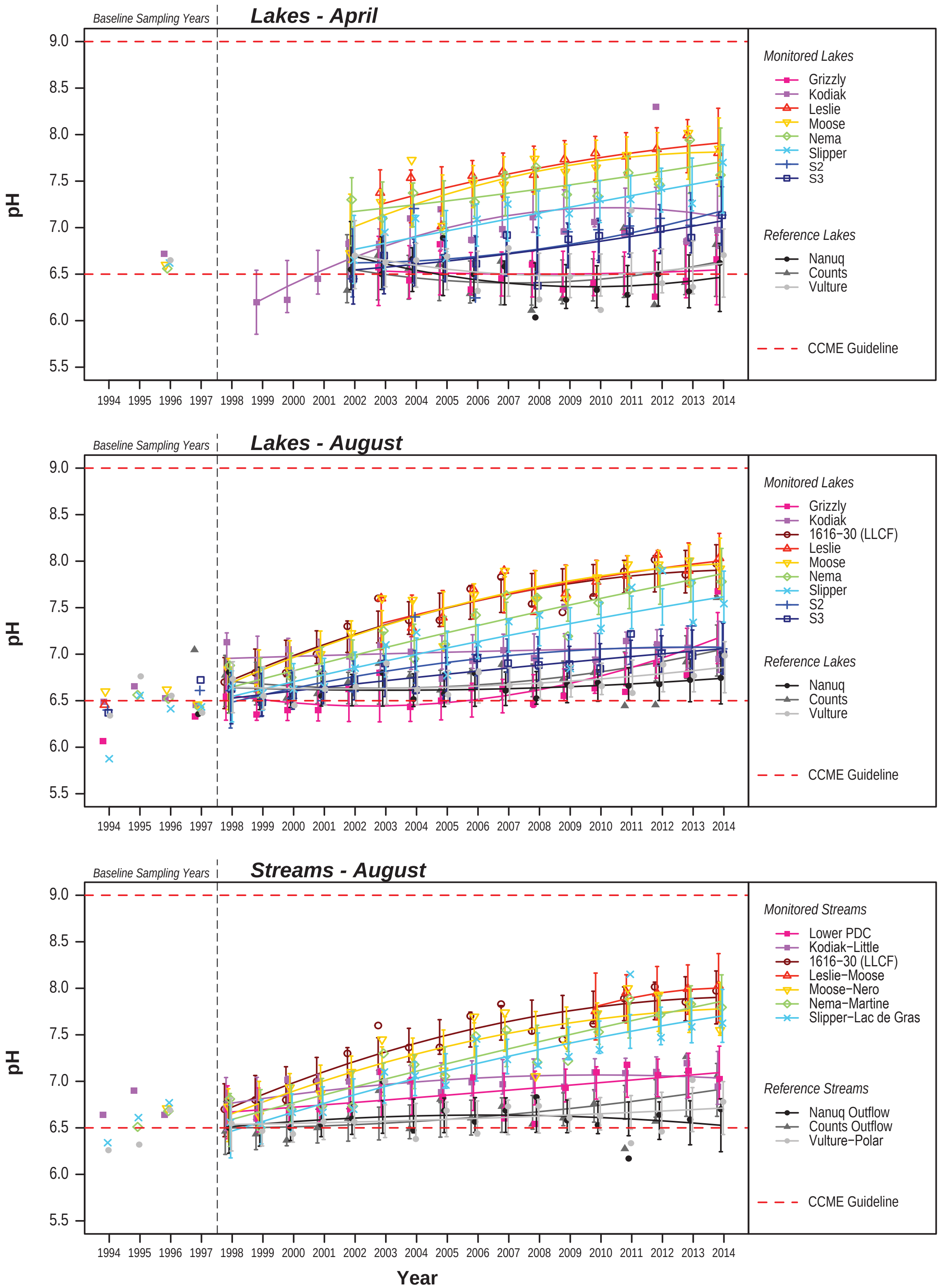
Table 3.2-5. Statistical Results of pH in Lakes and Streams in the Koala Watershed and Lac de Gras

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, Slipper	-	1-1
Aug	Lake	-	LME	3	Grizzly, Kodiak, 1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2, S3	Grizzly, 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

Note: Dashes indicate not applicable.

Figure 3.2-1

Observed and Fitted Means for pH in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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.

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 Grizzly Lake and the Lower PDC during the open water season (Table 3.2-5). Graphical analysis of Grizzly Lake during the open water season shows that although pH levels overlap with values observed in reference lakes, they have increased above baseline levels in recent years (Figure 3.2-1). The source of the increase in pH in Grizzly Lake is unclear at this time, but unlikely related to mine operations. 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 relatively stable over time and overlap with values observed in reference streams (Figure 3.2-1). Given that pH levels in Kodiak Lake, Kodiak-Little, and the Lower PDC have been stable through time during the open water season (Table 3.2-5; Figure 3.2-1), no mine effects were detected at these sites.

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 lower confidence interval for Grizzly Lake was below the lower guideline value during the ice-covered season. However, the lower confidence intervals were also less than the lower CCREM guideline value in all reference lakes during the ice-covered season.

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 3.2-6). 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).

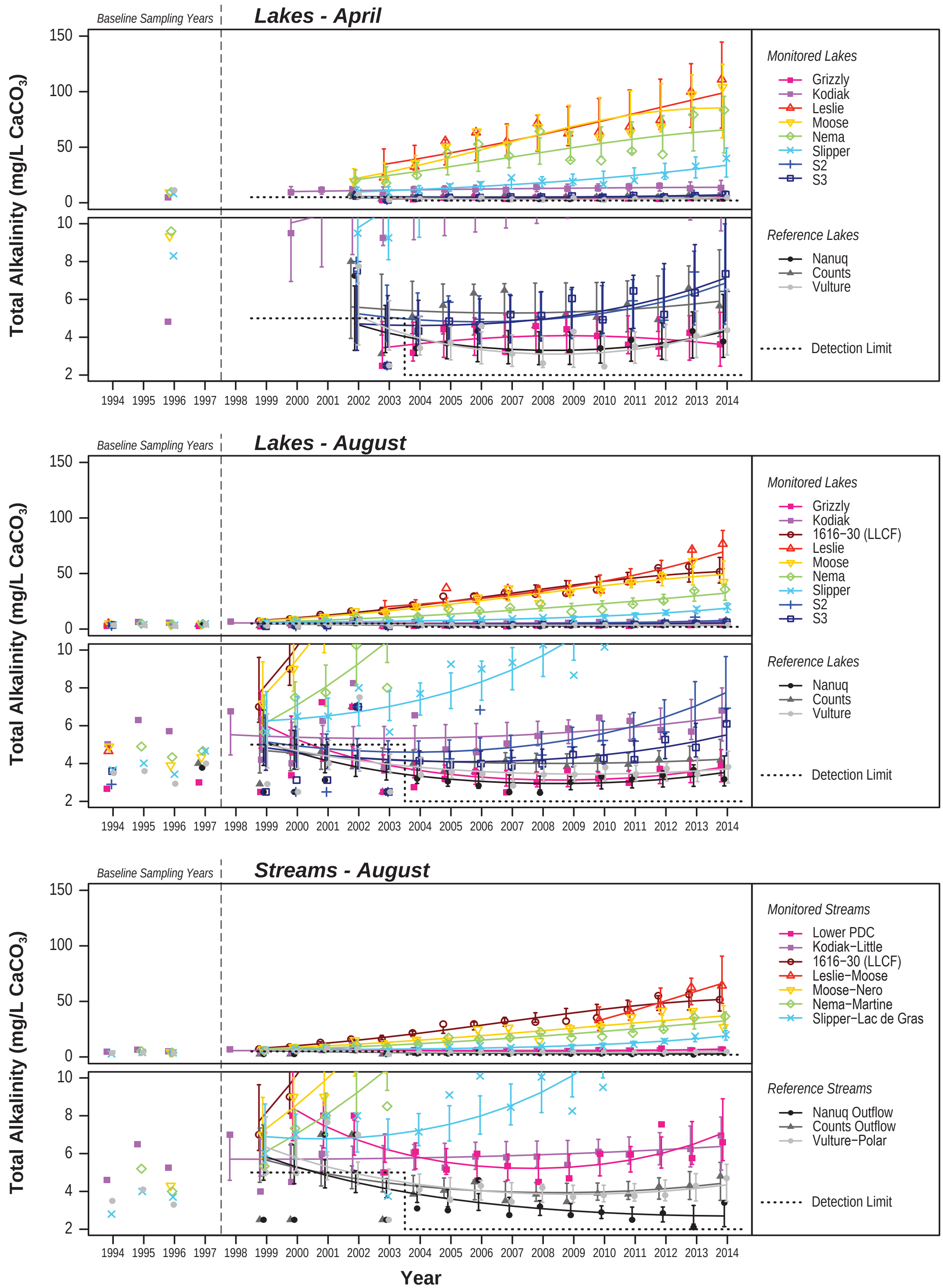
Table 3.2-6. Statistical Results of Total Alkalinity in Lakes and Streams in the Koala Watershed and Lac de Gras

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

Note: Dashes indicate not applicable.

Figure 3.2-2

Observed and Fitted Means for Total Alkalinity in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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

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 3.2-6). 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 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 lakes and interconnecting streams downstream of the LLCF as far as site S3 in Lac de Gras, with the exception of site S2 during the ice-covered season and Leslie-Moose Stream during the open water season (Table 3.2-7). Only five years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends unlikely; however, graphical analysis shows that hardness values in Leslie-Moose Stream were similar to those in the LLCF in all years during which Leslie-Moose Stream was monitored. Graphical analysis suggests that water hardness has increased downstream of the LLCF as far as site S3 since monitoring began, but has stabilised as far as Slipper Lake 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).

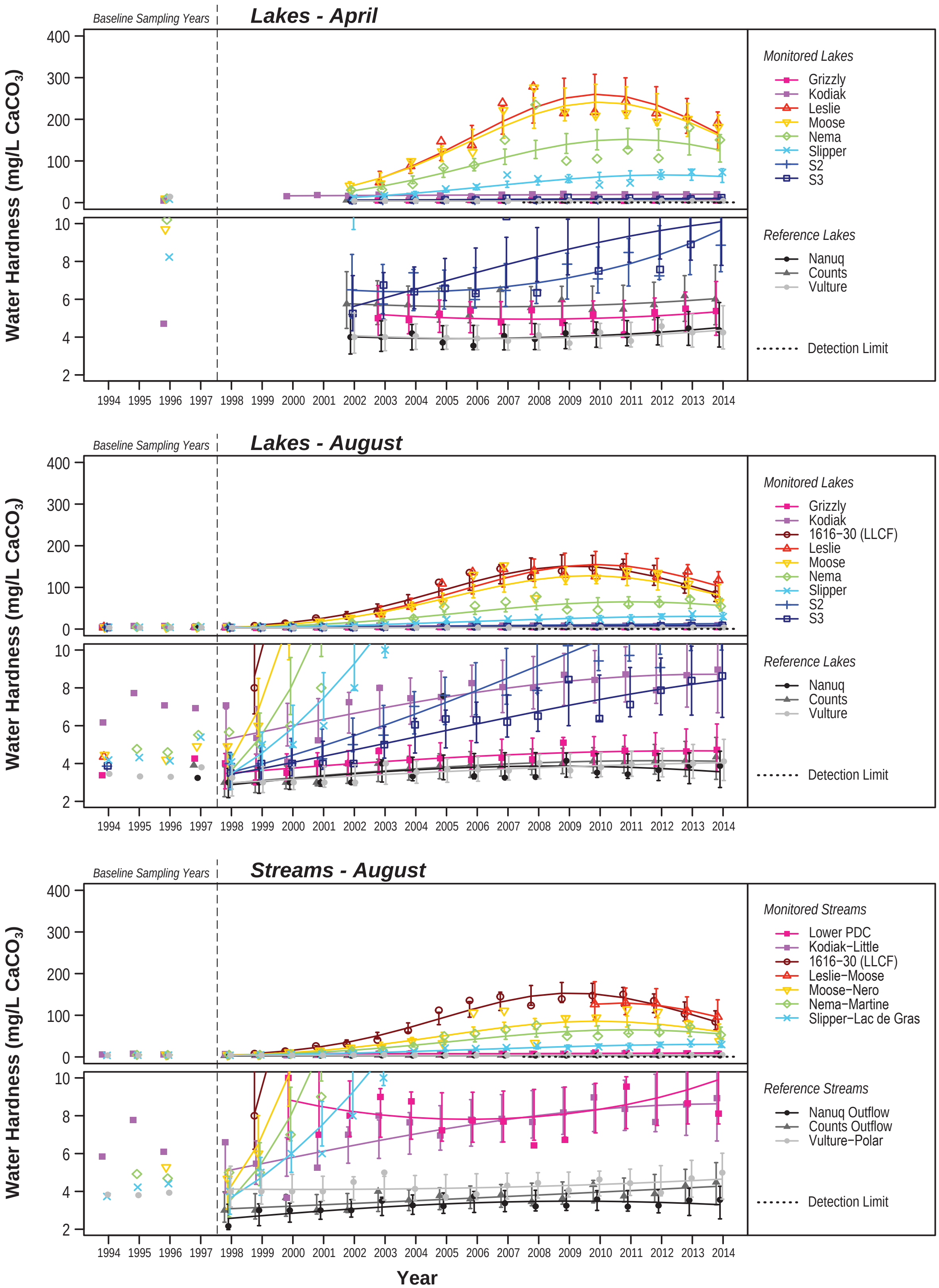
Table 3.2-7. Statistical Results of Water Hardness in Lakes and Streams in the Koala Watershed and Lac de Gras

Month	Lake/ Stream	Lakes/Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	2	-	Leslie, Moose, Nema, Slipper, S3	-	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

Note: Dashes indicate not applicable.

Figure 3.2-3

Observed and Fitted Means for Water Hardness in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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

At sites that are not downstream of the LLCF, statistical analyses suggest that water hardness has changed through time, relative to reference streams, in Kodiak-Little (Table 3.2-7). 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 3.2-7). 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 3.2-7). No mine effects were detected at sites that are not downstream of the LLCF.

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. 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 2014.

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 3.2-8; Figure 3.2-4). Only five years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends unlikely; 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 also suggests chloride concentrations decrease with downstream distance from the LLCF (Figure 3.2-4).

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

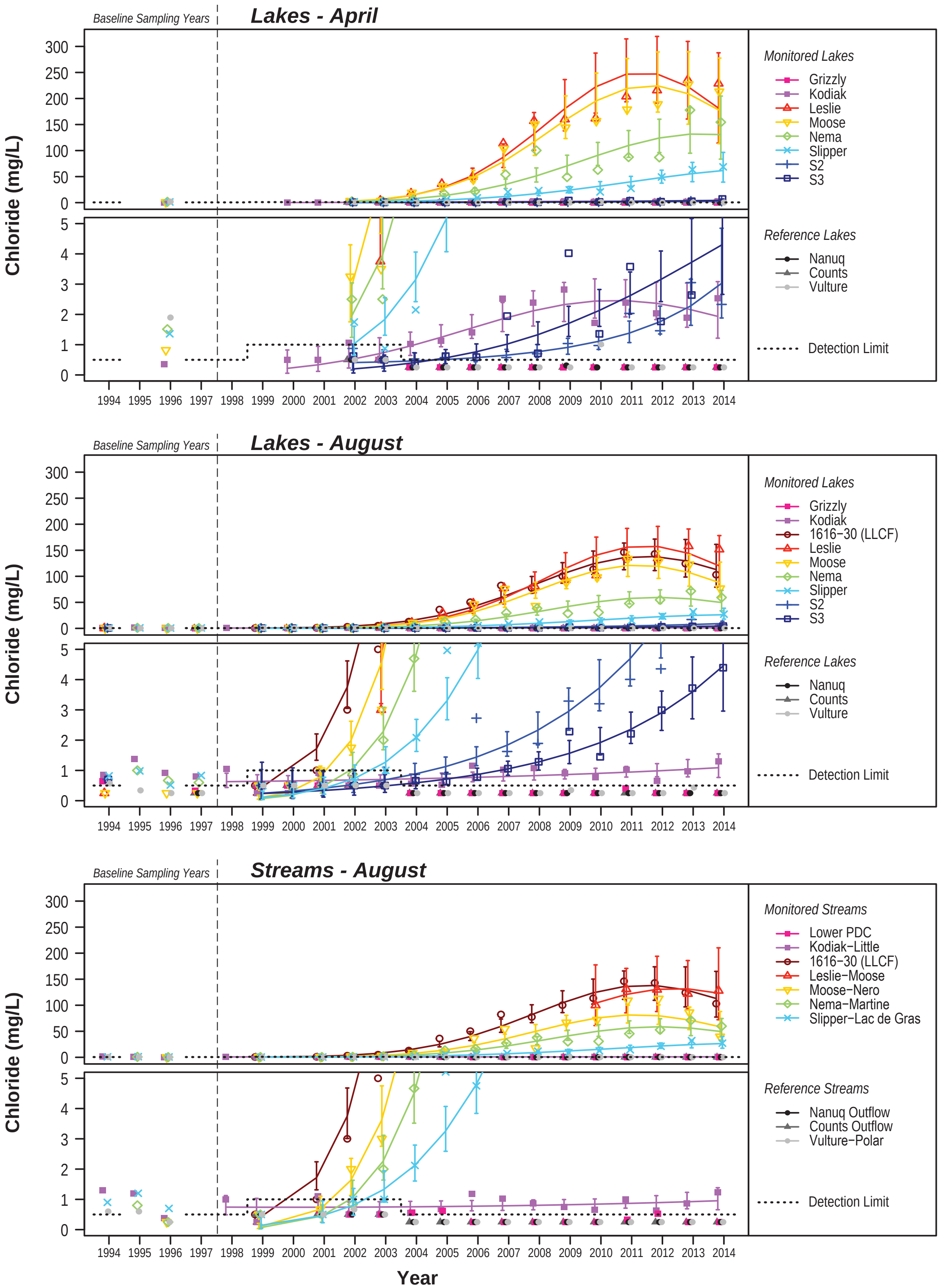
Month	Lake/ Stream	Lakes/Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	Grizzly, Nanuq, Counts, Vulture	Tobit	1a	-	-	Kodiak, Leslie, Moose, Nema, Slipper, S2, S3	1-55
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

Note: Dashes indicate not applicable.

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 3.2-8; Figure 3.2-4).

Figure 3.2-4

Observed and Fitted Means for Chloride Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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.

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 2014 (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 2014 (see Part 2 – Data Report).

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 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 2014.

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 3.2-9; Figure 3.2-5). Only five years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends unlikely; 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. Sulphate concentrations had shown signs of stabilizing in recent years downstream of the LLCF as far as Nema-Martine Stream; however, sulphate concentrations in 2014 have continued to increase at all sites during the ice-covered and open water seasons (Figure 3.2-5). Sulphate concentrations decreased with downstream distance from the LLCF (Figure 3.2-5).

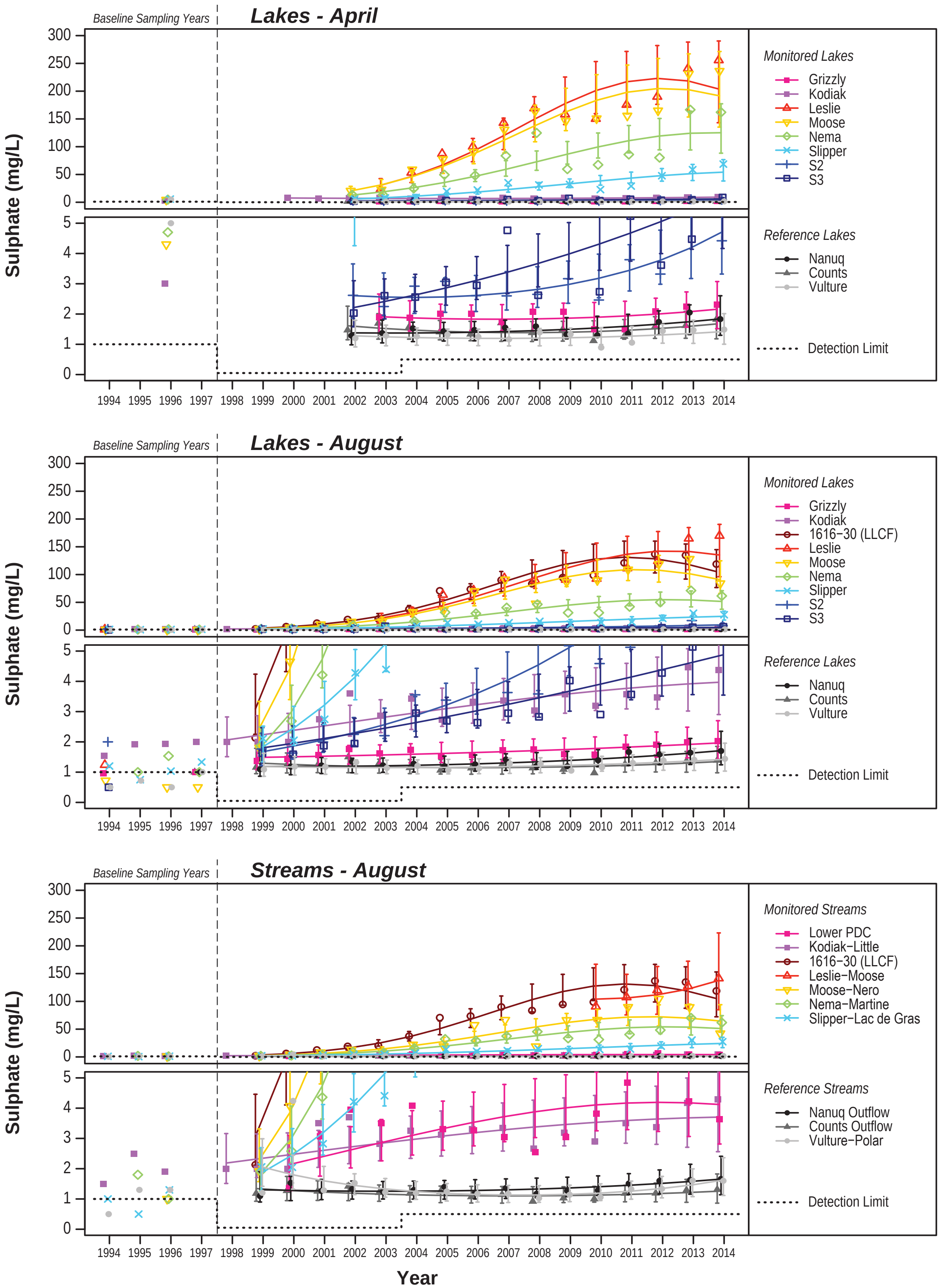
Table 3.2-9. Statistical Results of Sulphate Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

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

Note: Dashes indicate not applicable.

Figure 3.2-5

Observed and Fitted Means for Sulphate Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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.

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 3.2-9). 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 Kodiak Lake during the open water season (Figure 3.2-5).

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 2014 (Rescan 2012e). Sulphate concentrations were also less than the hardness-dependent sulphate SSWQO in all monitored streams in June, July, August, and September in 2014 (see Part 2 – Data Report; Rescan 2012e).

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 and in the deep water samples of Leslie Lake during the open water season. No mine effects were detected at sites that are not downstream of the LLCF.

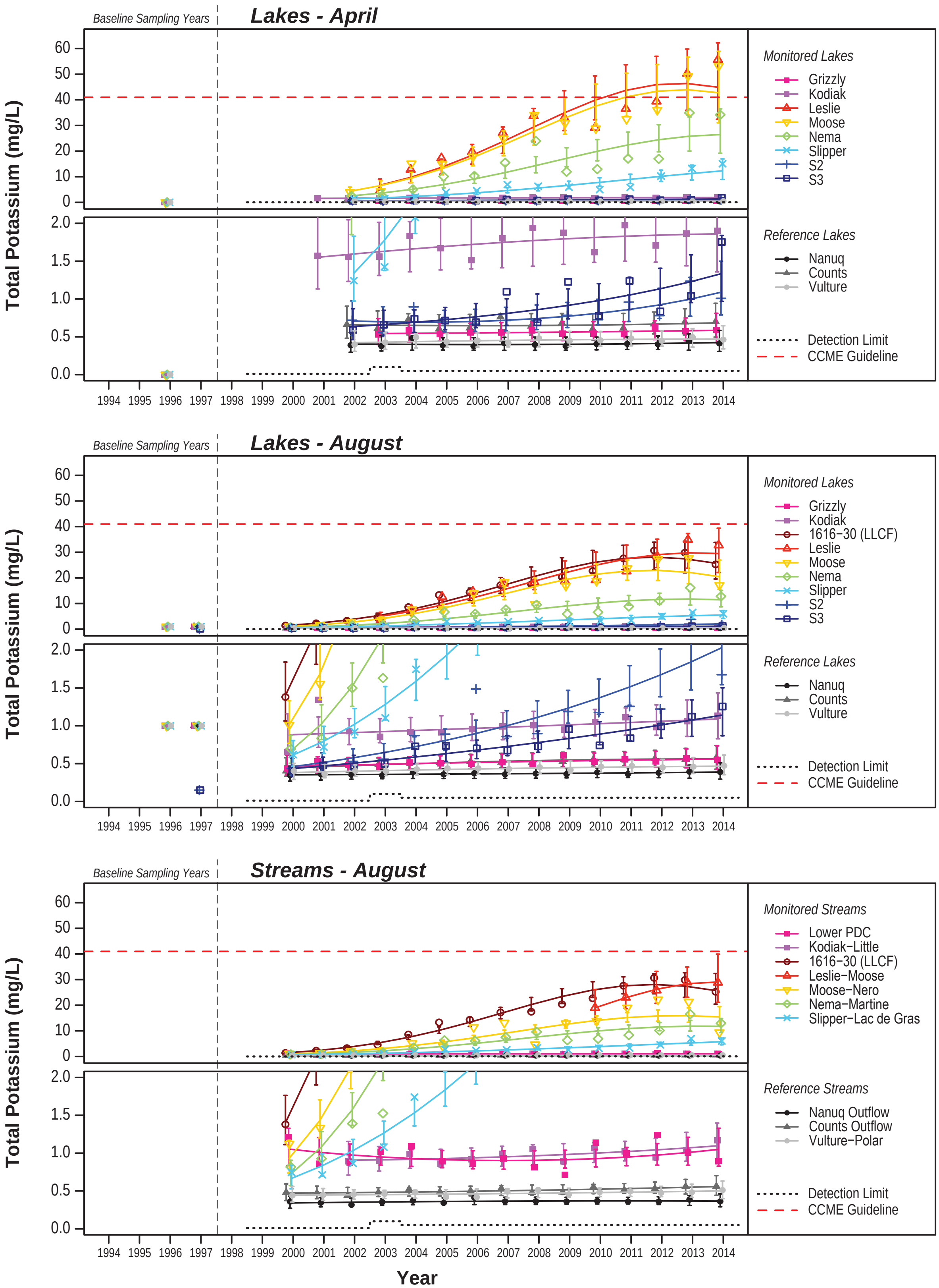
Statistical analyses indicate that temporal trends in potassium concentrations differ from those observed at reference sites at all monitored sites downstream of the LLCF as far as Slipper Lake during the ice-covered season and as far as site S3 in Lac de Gras during the open water season, with the exception of Leslie-Moose Stream (Table 3.2-10). Only five years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends unlikely; 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 3.2-10; 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 2012f). During the open water season, potassium concentrations in both samples taken from two meters above the sediment-water interface (i.e., deep water samples) in Leslie Lake were greater than the SSWQO; however, the mean of water samples taken across all three depths in Leslie Lake was less than the SSWQO (Figure 3.2-6; see Part 3 – Statistical Report; Rescan 2012f). Observed potassium concentrations were less than the long-term SSWQO in all monitored streams in June, July, August, and September in 2014 (see Part 2 – Data Report; Rescan 2012f).

Figure 3.2-6

Observed and Fitted Means for Potassium Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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 3.2-10. 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	Kodiak, 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

Note: Dashes indicate not applicable.

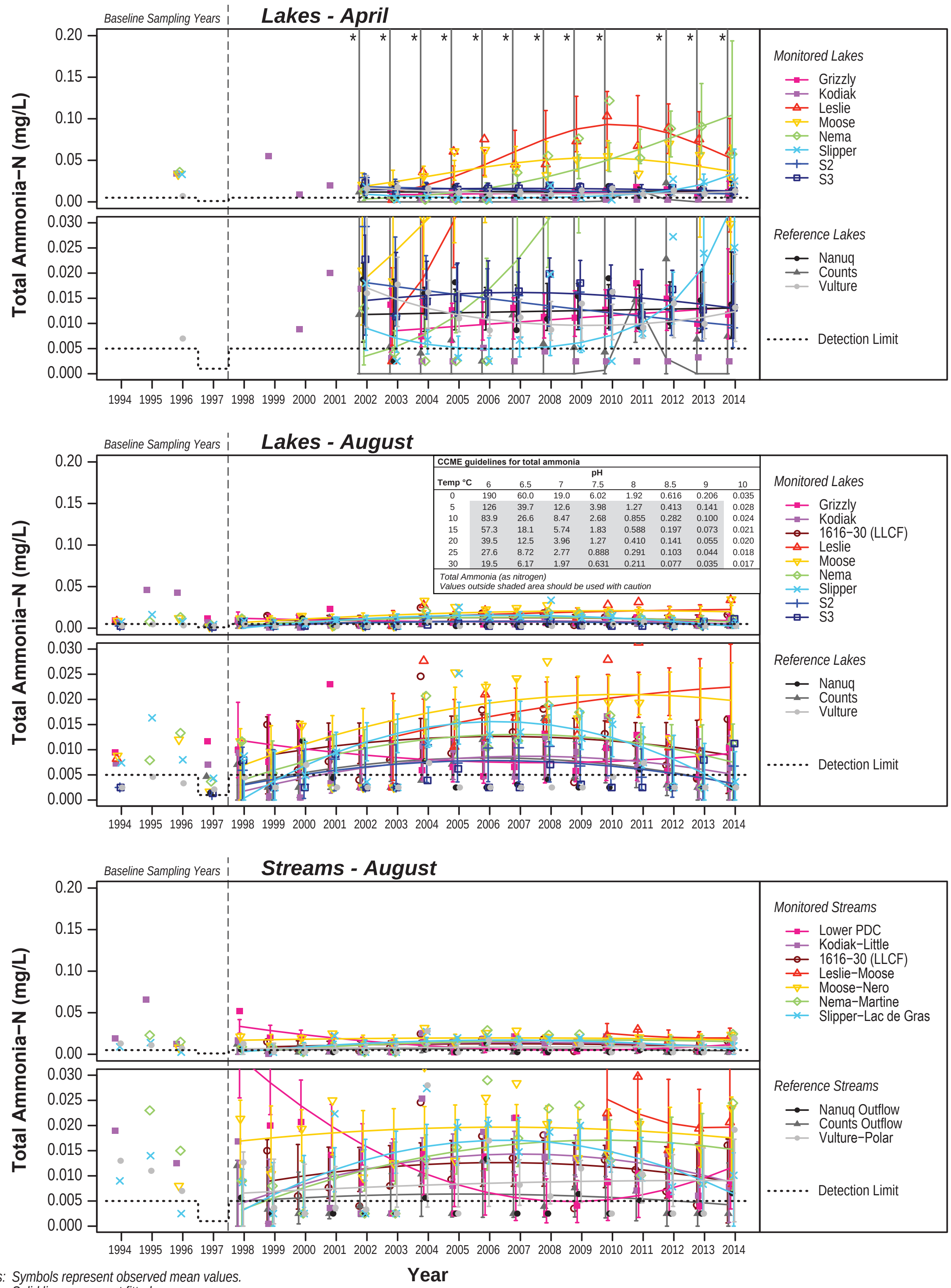
3.2.4.7 Total Ammonia-N

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 2014. Observed total ammonia-N concentrations were less than pH- and temperature-dependent CCME guidelines at all monitored sites in 2014. 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 3.2-11). Graphical analysis suggests that total ammonia-N has increased in Leslie, Moose, Nema, and Slipper lakes during the ice-covered and in Leslie and Moose lakes during the open water seasons (Figure 3.2-7); however, 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).

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 2014



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.
CCME Guideline is pH and temperature dependent (see inset table).
** Upper 95% Confidence Interval on the fitted mean of Counts Lake in April 2002 = 1.12×10^{270}, 2003 = 4.10×10^{259}, 2004 = 1.71×10^{244}, 2005 = 8.12×10^{223} mg/L, 2006 = 4.40×10^{198} mg/L, 2007 = 2.72×10^{168}, mg/L, 2008 = 1.92×10^{133} mg/L, 2009 = 1.54×10^{93} mg/L, 2010 = 1.41×10^{48} mg/L, 2012 = 5.21×10^{51} mg/L, 2013 = 1.53×10^{106} mg/L, and 2014 = 3.74×10^{161} mg/L.*

Table 3.2-11. 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	Nanuq, Vulture, S3	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

Note: Dashes indicate not applicable.

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, relative to reference sites, except the Lower PDC during the open water season (Table 3.2-11). 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).

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 2014 (CCME 2001b). Observed total ammonia-N concentrations were less than pH- and temperature-dependent CCME guidelines at all monitored lakes and stream sites in 2014 (see Part 2 - Data Report; CCME 2001b).

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 2014.

Nitrite-N concentrations were less than detection limits at all reference sites in 2014 (Table 3.2-12; 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 Moose Lake and Leslie-Moose Stream (Table 3.2-12; Figure 3.2-8). Only five 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.

Table 3.2-12. 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, Nema, 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	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

Note: Dashes indicate not applicable.

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 have decreased over the last two years, 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 3.2-12; Figure 3.2-8).

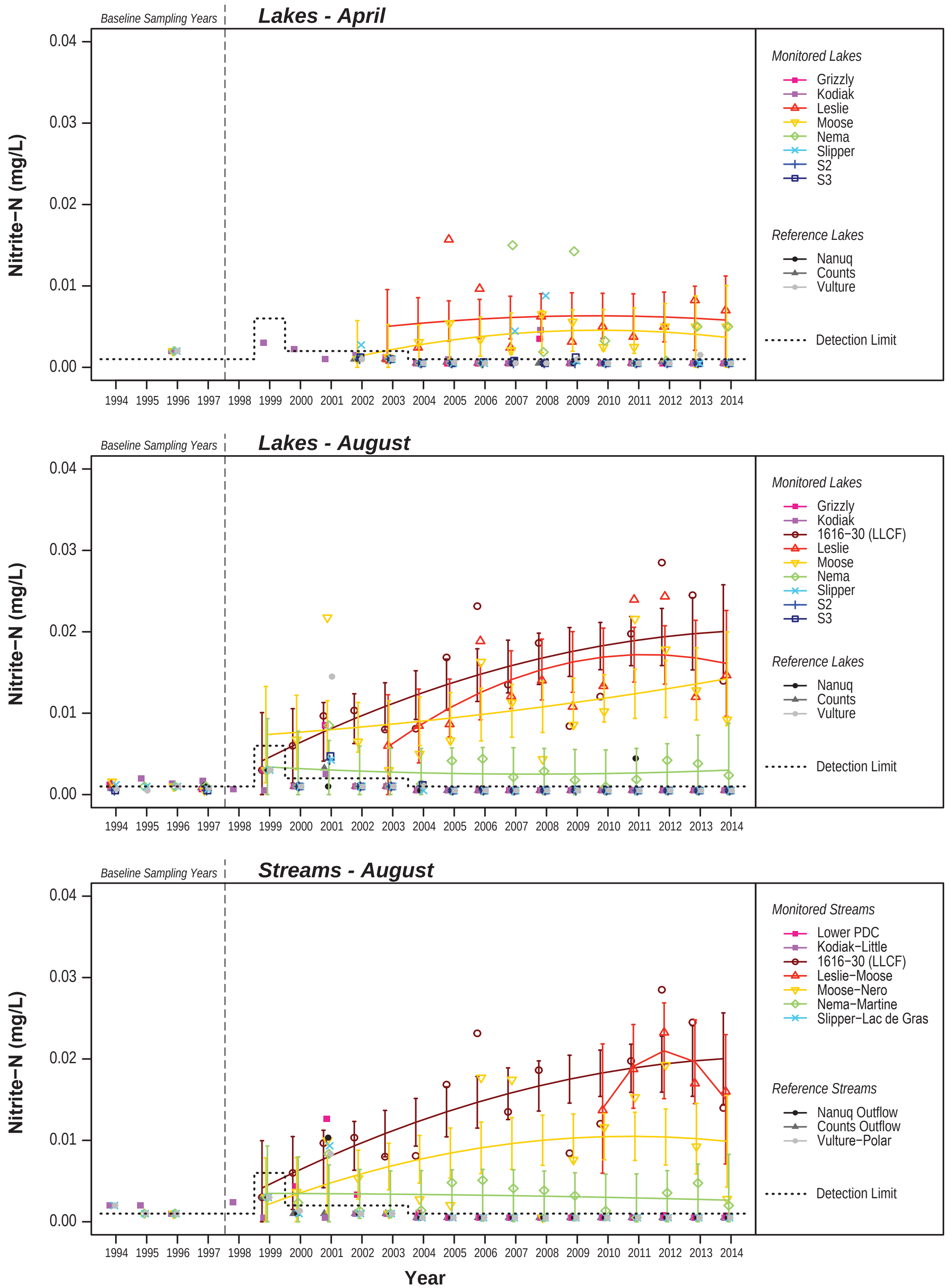
The 95% confidence intervals of the fitted mean and the observed mean nitrite-N concentrations were less than the 0.06 mg/L CCREM guideline value for nitrite-N in all reference and monitored lakes in April and August 2014 (CCREM 1987). Nitrite-N concentrations were also less than the CCREM guideline value for nitrite-N in all reference and monitored streams in June, July, August, and September in 2014 (see Part 2 - Data Report; CCREM 1987).

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 2014.

Figure 3.2-8

Observed and Fitted Means for Nitrite-N Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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.

Statistical analyses indicate that nitrate-N concentrations have changed through time, relative to reference lakes, in all monitored lakes 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 the exception of Leslie-Moose Stream (Table 3.2-13). Only five years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends improbable; however, graphical analysis shows that nitrate-N concentrations in Leslie-Moose Stream were similar to those in the LLCF in recent years. 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.

Table 3.2-13. 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	1-138
Aug	Lake	Kodiak, 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

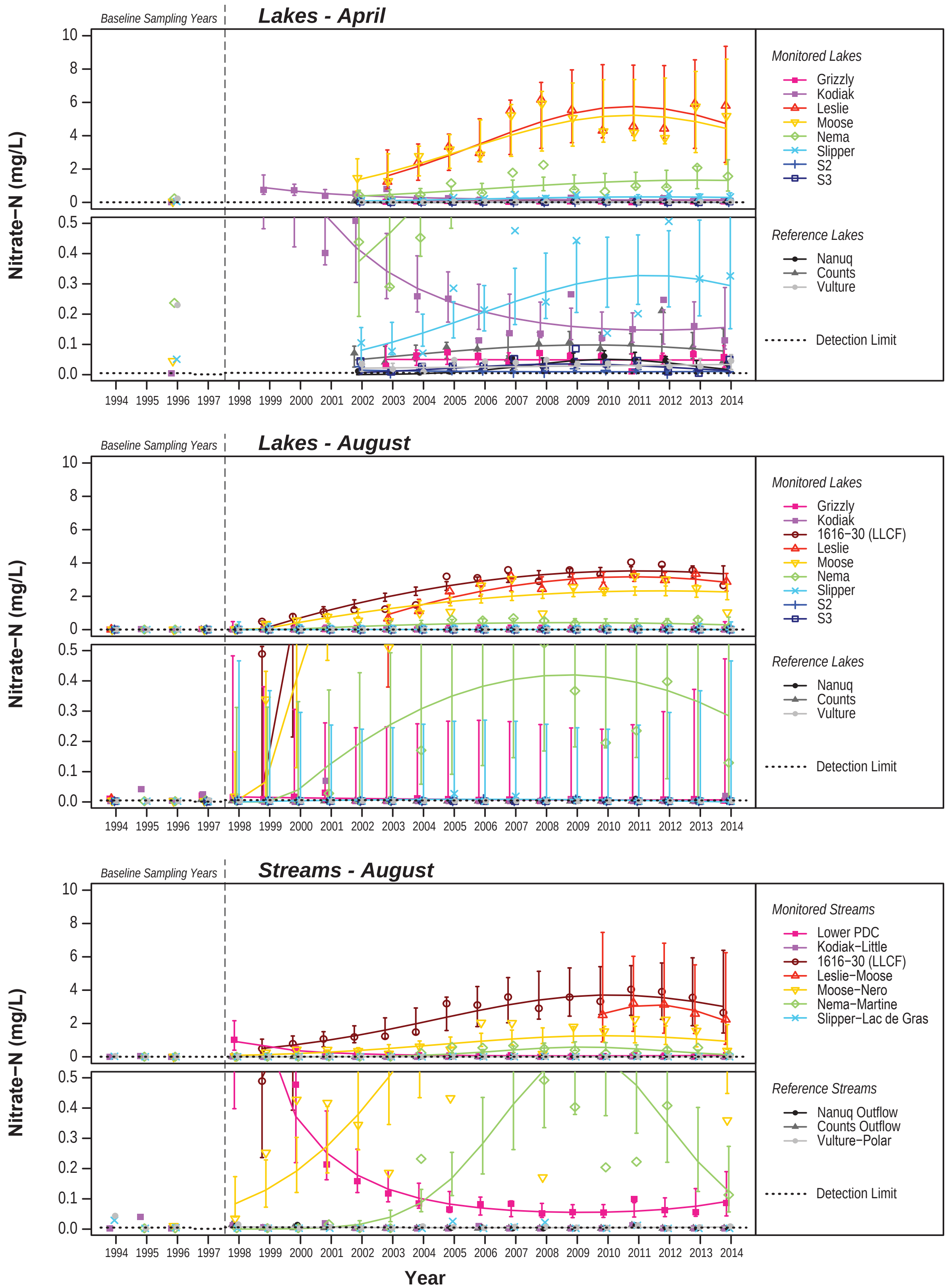
Note: Dashes indicate not applicable.

At sites not downstream of the LLCF 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 3.2-13). 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.

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 2014 (Rescan 2012d). 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; Rescan 2012d).

Figure 3.2-9

Observed and Fitted Means for Nitrate-N Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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.

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 during the ice-covered season. No mine effects were detected at sites that are not downstream of the LLCF. In several cases, the observed and fitted mean or the upper 95% confidence interval of 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 two of the reference lakes.

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 3.2-14). Total phosphate-P concentrations have shown no signs of change during the open water season in either Leslie or Moose lakes (Table 3.2-14; 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 3.2-14; 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.

Table 3.2-14. Statistical Results of Total Phosphate-P Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

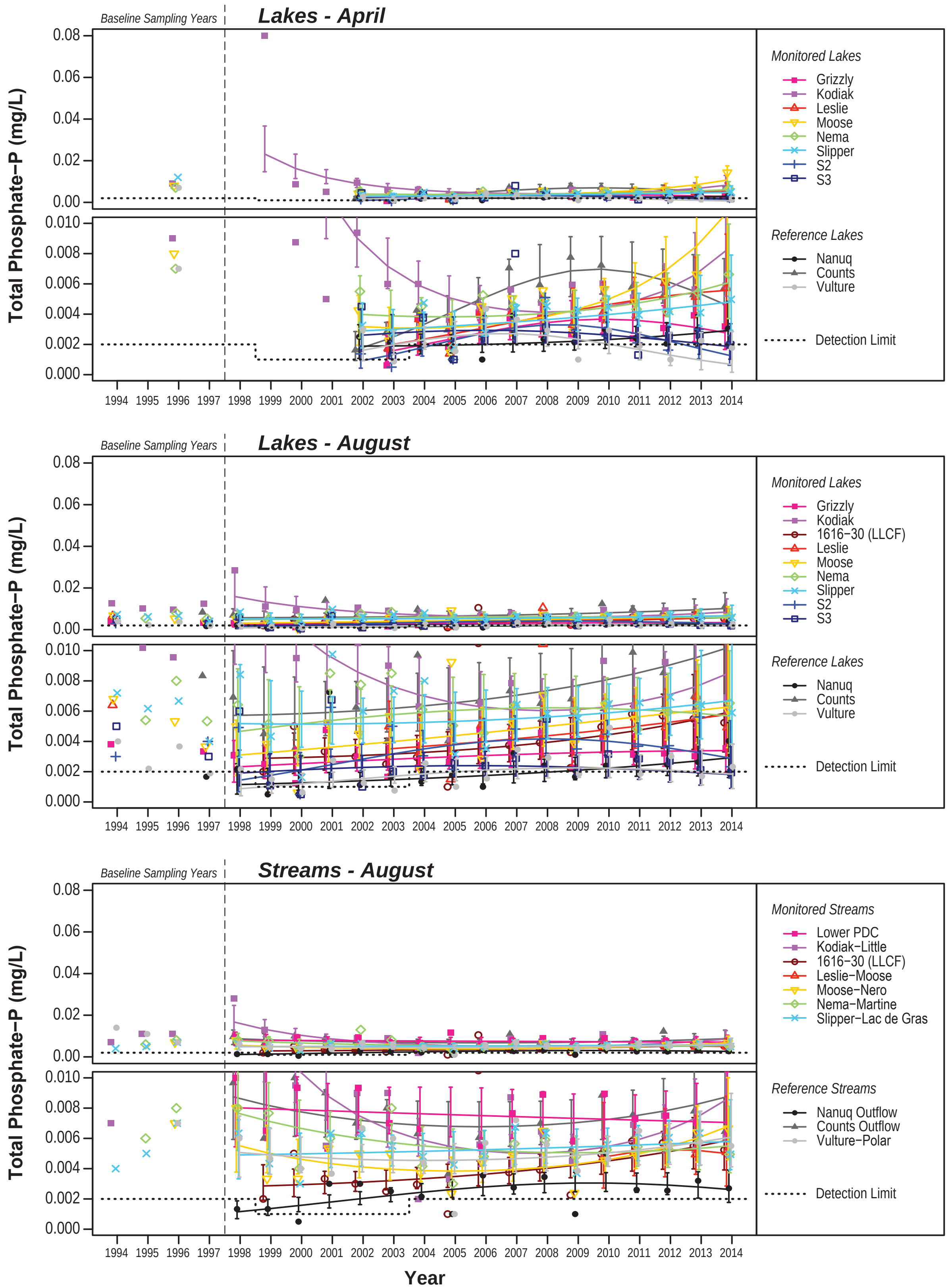
Month	Lake/ Stream	Lakes/Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	Tobit	1b	-	-	Kodiak, Leslie, Moose	1-154
Aug	Lake	-	Tobit	2	-	Kodiak	-	1-160
Aug	Stream	-	Tobit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

Note: Dashes indicate not applicable.

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 3.2-14). 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 (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 at sites not downstream of the LLCF.

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 2014



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

The 95% confidence interval of the fitted mean total phosphate-P concentrations was 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 Lake during the open water season in 2014, while the observed and fitted mean for Counts Lake was greater than the 0.01mg/L trigger during the open water season (Figure 3.2-10; CCME 2004). The 95% confidence intervals of fitted mean total phosphate-P concentrations were also greater than the benchmark trigger of mean baseline concentration + 50% (CCME 2004), in Leslie, Moose, Nema, and Nanuq during the open water season, while the observed and fitted mean was greater than the benchmark trigger in Moose and Nanuq lakes during the ice-covered season in 2014. Overall, similar patterns in exceedance of the relevant guideline values were observed in both 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 Slipper Lake, no clear downstream spatial gradient was present and it was concluded that no effects were detected. No mine effects were detected at sites that are not downstream of the LLCF.

Statistical analyses indicate that TOC concentrations have changed through time, relative to reference lakes, in all monitored lakes downstream of the LLCF as far as Slipper Lake during both the ice-covered and open water seasons, except for Nema Lake during the ice-covered season (Table 3.2-15). 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 has increased through time in lakes as far downstream as Slipper 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.

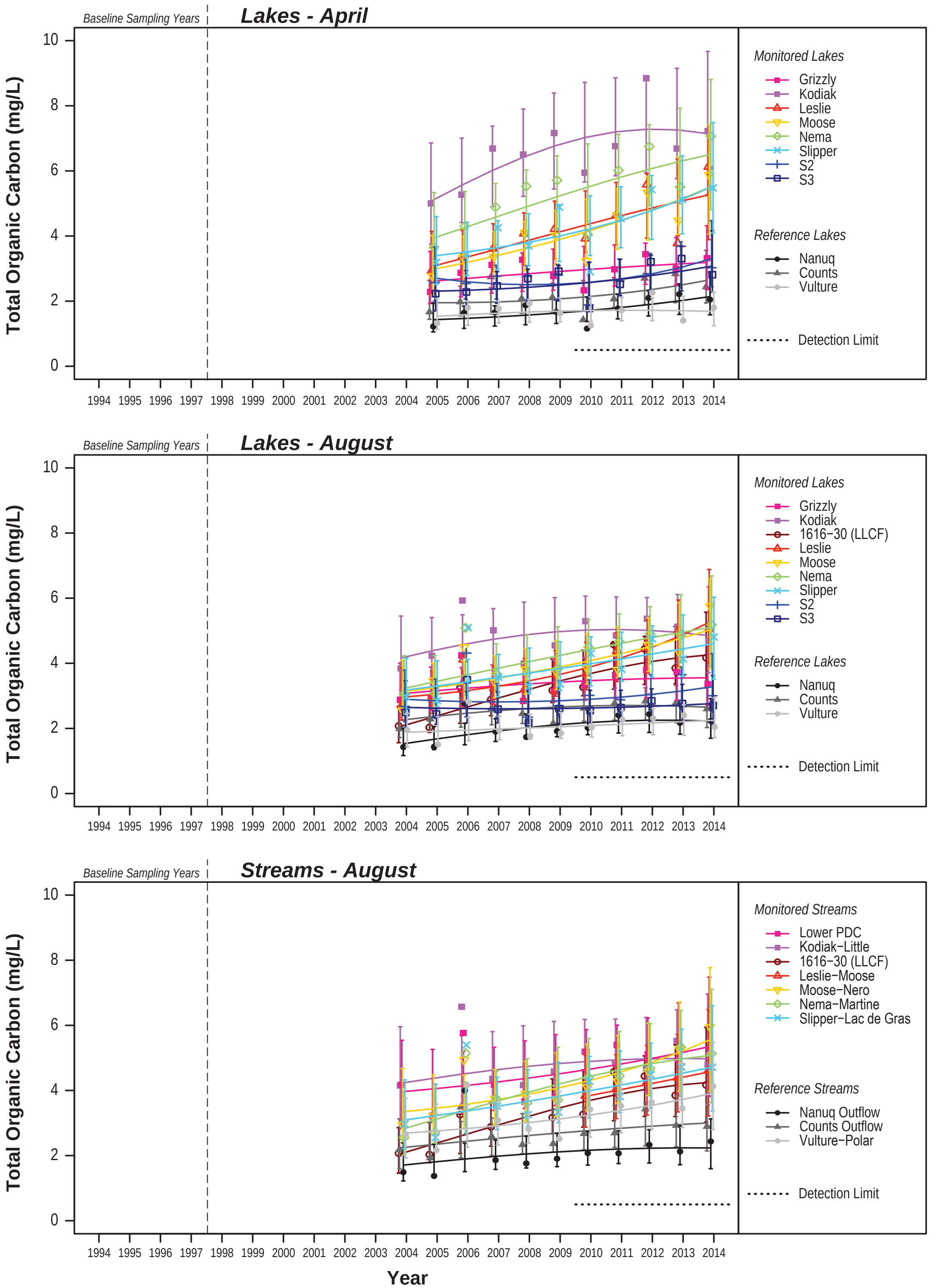
Table 3.2-15. Statistical Results of Total Organic Carbon in Lakes and Streams in the Koala Watershed and Lac de Gras

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	-	-	Leslie, Moose, Slipper	1-172
Aug	Lake	-	LME	1b	-	-	1616-30 (LLCF), Leslie, Moose, Nema, Slipper	1-178
Aug	Stream	-	LME	2	-	none	-	1-184

Note: Dashes indicate not applicable.

Figure 3.2-11

Observed and Fitted Means for Total Organic Carbon in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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

At sites that are not downstream of the LLCF, statistical and graphical analyses indicate that TOC concentrations have remained stable through time during the ice-covered and open water seasons (Table 3.2-15; 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; Rescan 2002). 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 Nema-Martine Stream. 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 2014.

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 3.2-16). Only five years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends improbable; however, graphical analysis shows that total antimony concentrations in Leslie-Moose Stream were similar to those in the LLCF in recent years. Graphical analysis suggests that total antimony concentrations increased to peak concentrations around 2006 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 from the LLCF as far as Slipper-Lac de Gras Stream. However, results from the 2013 AEMP Evaluation of Effects (ERM Rescan 2014a), along with results from 2014, 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 Nema-Martine Stream remain above baseline and reference lake concentrations, with concentrations decreasing with downstream distance of the LLCF (Figure 3.2-12).

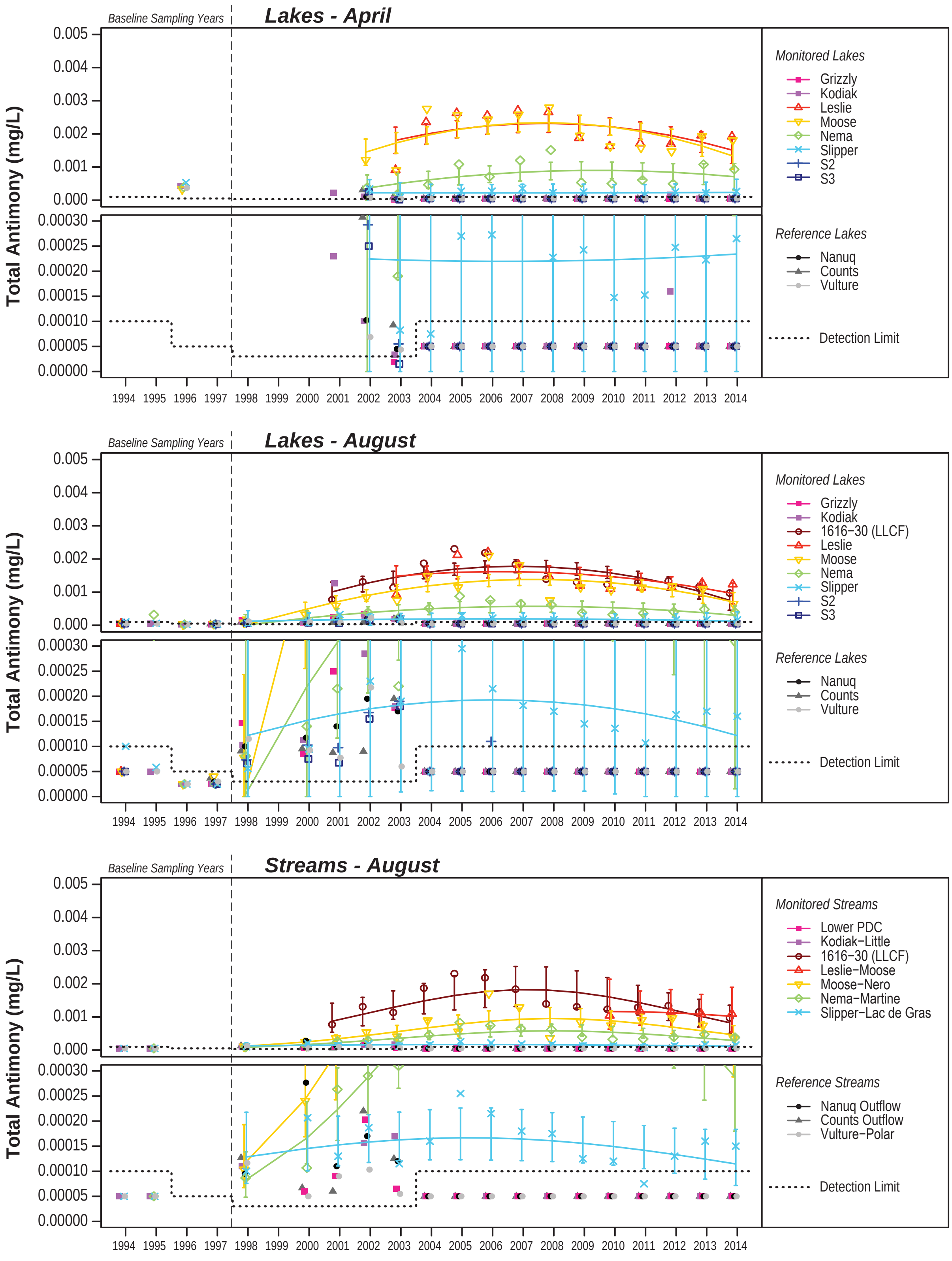
Table 3.2-16. Statistical Results of Total Antimony Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

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, Counts, Outflow, Vulture-Polar	LME	1a	-	-	1616-30 (LLCF), Moose-Nero, Nema-Martine	1-200

Note: Dashes indicate not applicable.

Figure 3.2-12

Observed and Fitted Means for Total Antimony Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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 (Fletcher et al. 1996) = 0.02 mg/L.

At sites that are not downstream of the LLCF, no temporal trends were observed. Concentrations at these sites have generally been less than analytical detection limits since monitoring began (Table 3.2-16; 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 2014 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 2014 (see Part 2 - Data Report; Fletcher et al. 1996).

3.2.4.13 Total Arsenic

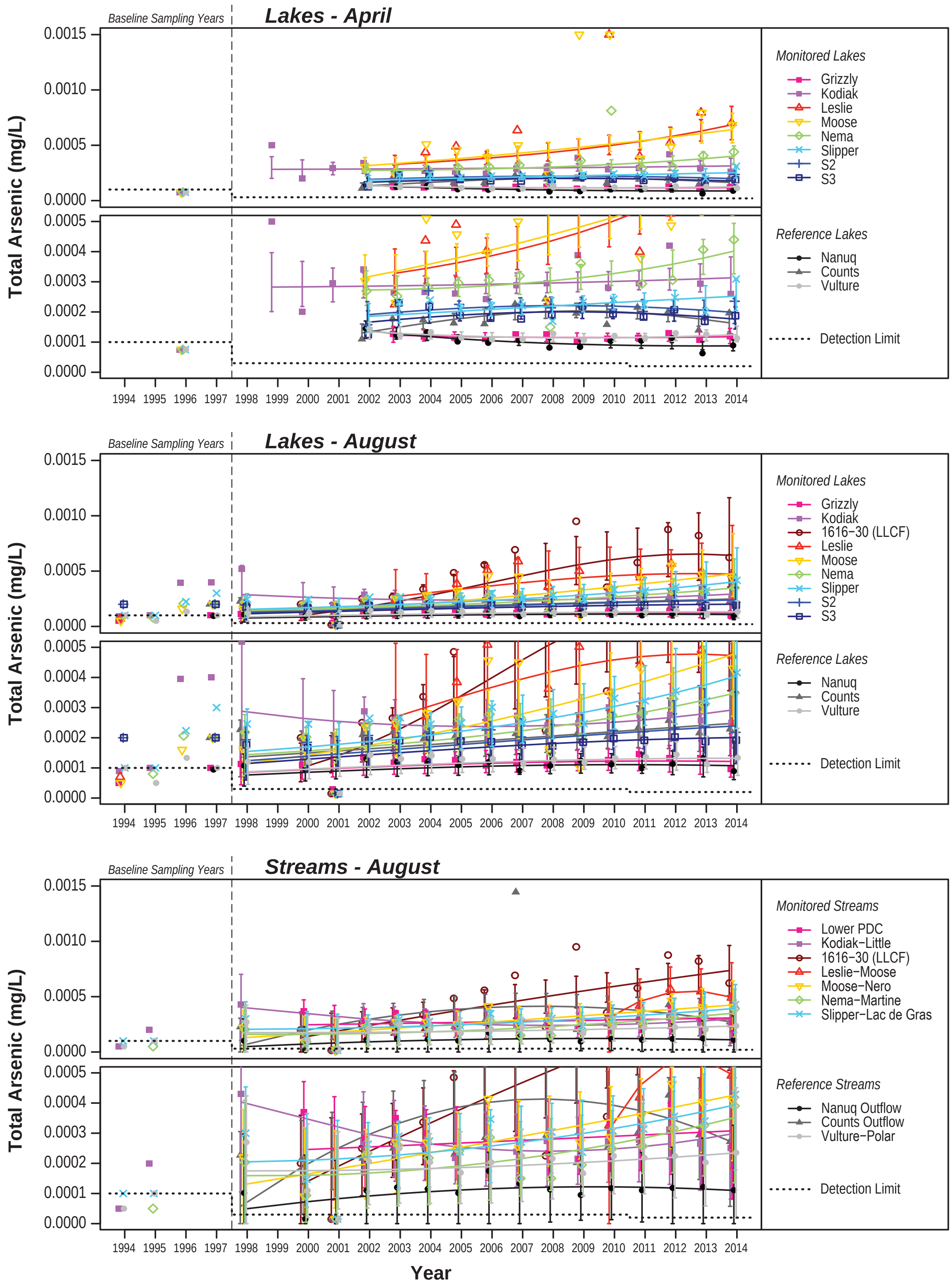
> ***Summary:*** Together, statistical and graphical analyses suggest that total arsenic concentrations have increased downstream of the LLCF as far as Nema Lake during the ice-covered season as a result of mine operations. No mine effects were detected at sites that are not downstream of the LLCF. Observed and fitted mean concentrations were less than the arsenic CCME guideline at all sites in 2014.

Statistical and graphical analyses indicate that total arsenic concentrations have changed through time, relative to reference lakes, in Leslie, Moose, and Nema lakes during the ice-covered season (Table 3.2-17; 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 3.2-17; Figure 3.2-13). The observed trend in Leslie, Moose, and Nema 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). With the exception of some stream samples in June, the target detection limit was also achieved for ice-covered and open water lake and stream samples in 2014.

No temporal changes in total arsenic concentrations were detected at sites that are not downstream of the LLCF (Table 3.2-17). Graphical analysis also suggests that total arsenic concentrations have been stable through time in these lakes and streams (Figure 3.2-13). Thus, no mine effects were detected at sites that are not downstream of the 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 2014



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 = 0.005 mg/L.

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

					Significant Monitored Contrasts			
Month	Lake/ Stream	Lakes/Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Model Fit = 3	Model Fit = 2	Model Fit = 1	Statistical Report Page No.
Apr	Lake	-	Tobit	1b	-	-	Leslie, Moose, Nema	1-205
Aug	Lake	-	Tobit	2	-	1616-30 (LLCF)	-	1-211
Aug	Stream	-	Tobit	2	-	1616-30 (LLCF)	-	1-217

Note: Dashes indicate not applicable.

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 2014 (see Part 2 - Data Report; CCME 1999c). Total arsenic concentrations did not exceed CCME guidelines in any of the monitored streams in June, July, August, or September 2014 (see Part 2 - Data Report; CCME 1999c).

3.2.4.14 Total Barium

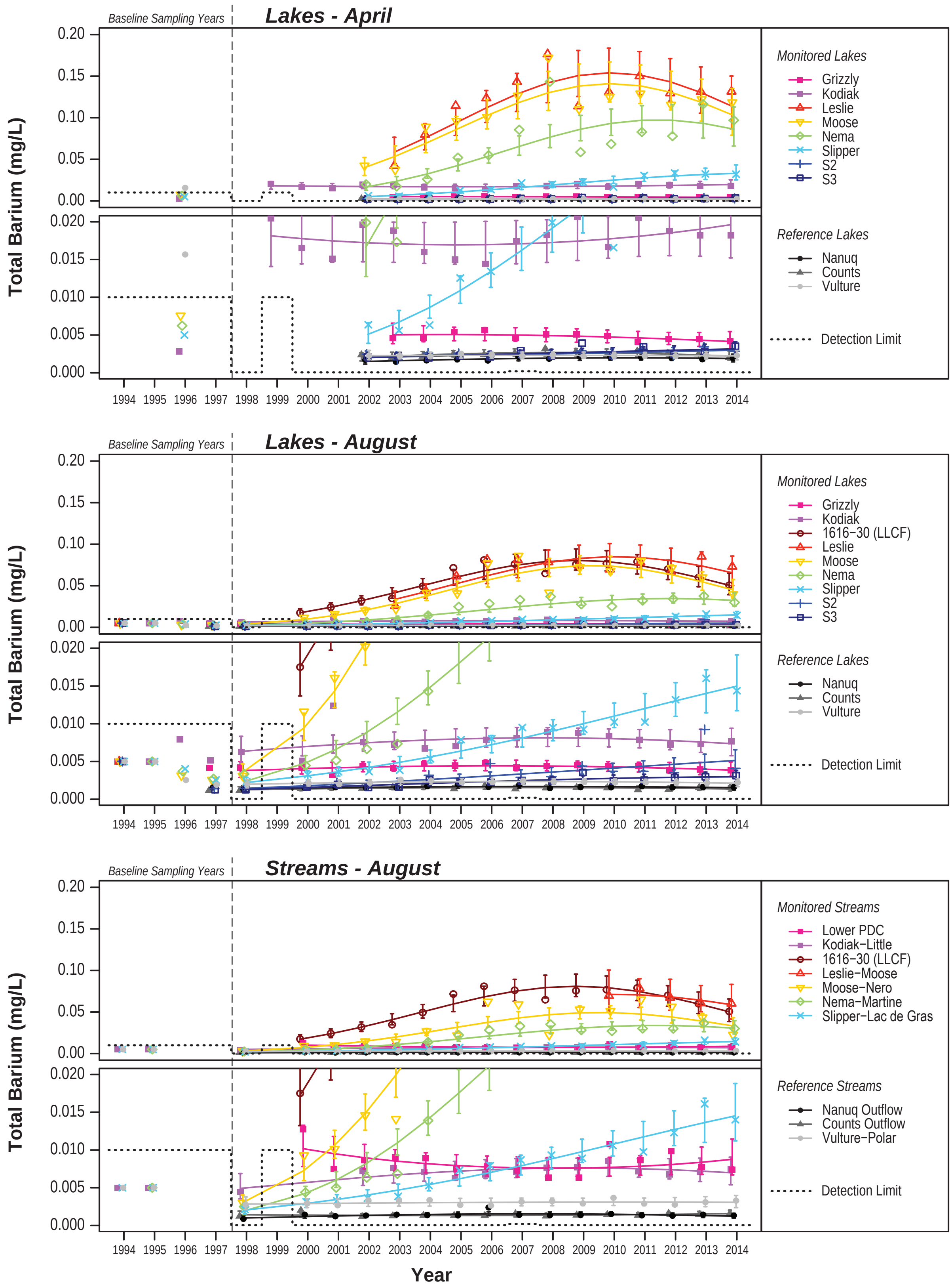
Summary: Statistical and graphical analyses suggest that total barium concentrations have increased at all monitored sites that are downstream of the LLCF as far as Slipper-Lac de Gras as a result of mine operations. However, barium concentrations have stabilised at levels greater than observed historical 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 2014.

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 Leslie-Moose Stream during the open water season (Table 3.2-18). Only five years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends improbable; however, graphical analysis shows that total barium concentrations in Leslie-Moose Stream were similar to those in the LLCF in recent years. Graphical analysis suggests that the concentration of total barium has increased in all monitored lakes and streams downstream of the LLCF as far as Slipper-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 3.2-18; Figure 3.2-14). Thus, no mine effects were detected at sites that are not downstream of the LLCF.

Figure 3.2-14

Observed and Fitted Means for Total Barium Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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 3.2-18. 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		1-223
Aug	Lake	-	LME	2	-	1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2, S3	-	1-229
Aug	Stream	-	LME	1b	-	-	1616-30 (LLCF), Moose-Nero, Nema-Martine, Slipper-Lac de Gras	1-235

Note: Dashes indicate not applicable.

The 95% confidence intervals of the 2014 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 2014 (see Part 2 - Data Report; Haywood and Drinnan 1983).

3.2.4.15 Total Boron

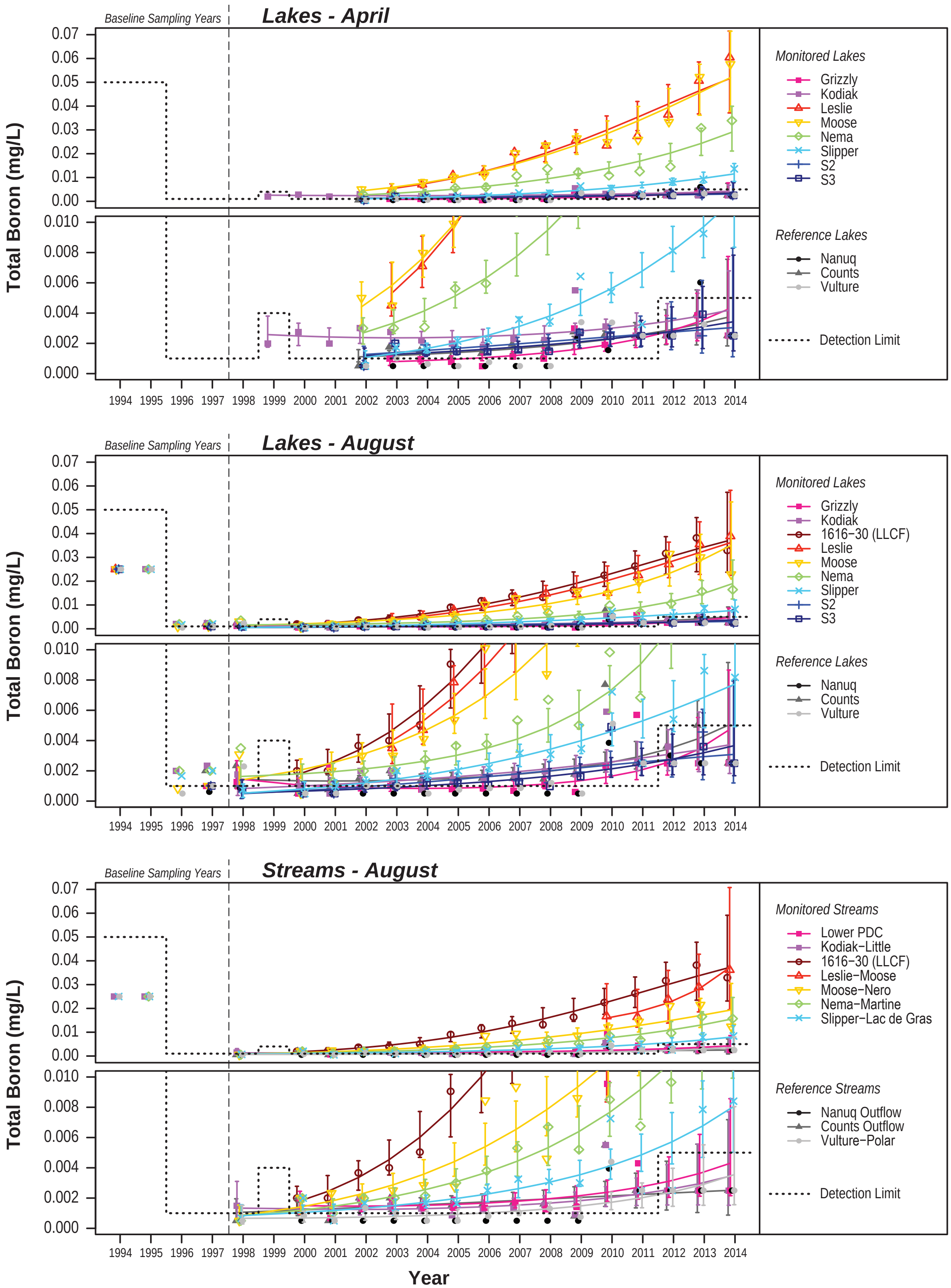
Summary: Statistical and graphical analyses suggest that total boron concentrations have increased at all sites downstream of the LLCF as far as Slipper–Lac de Gras Stream as a result of mine operations, with total boron concentrations decreasing with increasing downstream distance from the LLCF. No 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 2014.

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 3.2-19). Only five years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends unlikely; 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-Lac de Gras Stream, 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 Grizzly Lake during the open water 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 3.2-19; Figure 3.2-15).

Figure 3.2-15

Observed and Fitted Means for Total Boron Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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.

Table 3.2-19. 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, Vulture	Tobit	1b	-	-	Leslie, Moose, Nema, Slipper	1-241
Aug	Lake	Nanuq, Vulture	Tobit	1b	-	-	Grizzly, 1616-30 (LLCF), Leslie, Moose, Nema, Slipper	1-247
Aug	Stream	Nanuq Outflow	Tobit	2	-	1616-30 (LLCF), Moose-Nero, Nema-Martine	-	1-253

Note: Dashes indicate not applicable.

The 95% confidence intervals of the 2014 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 2014 (see Part 2 - Data Report; CCME 2009).

3.2.4.16 Total Cadmium

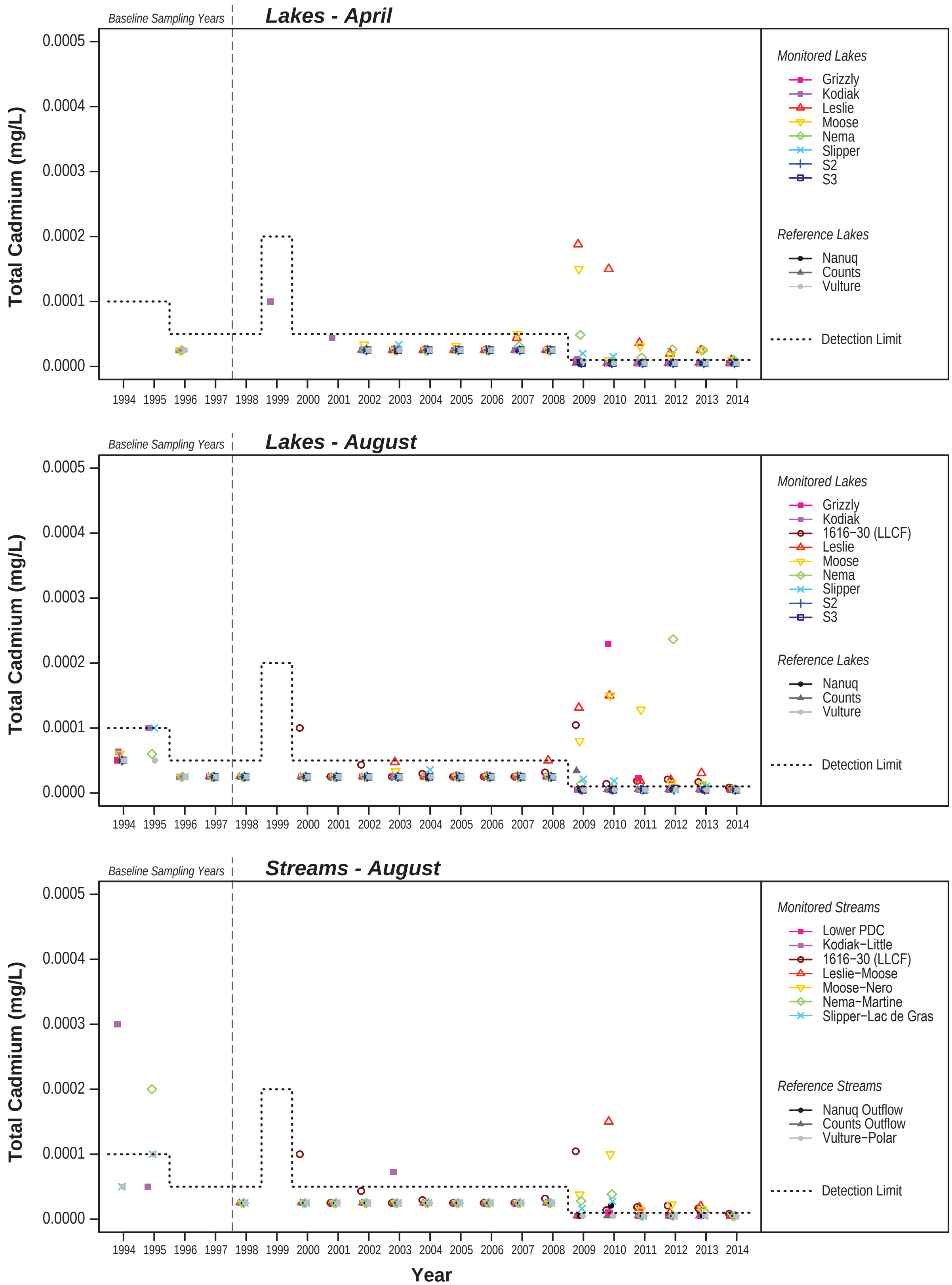
Summary: Concentrations of total cadmium have generally been below detection limits in all reference and monitored lakes and streams since monitoring began. The concentration in one sample from Moose-Nero Stream in June was greater than the hardness-dependent cadmium CCME guideline. No mine effects were 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 3.2-20).

Cadmium concentrations at some sites appeared elevated in 2009 and 2010 (Figure 3.2-16); however, these elevated concentrations reflect increased analytical detection limits as a result of matrix interference. Matrix interference can be caused by elevated concentrations of ions and, in the case of cadmium, by elevated concentrations of molybdenum. At elevated molybdenum concentrations, ICPMS methods cannot distinguish concentrations of cadmium from molybdenum and samples must be diluted prior to analysis. However, when samples are diluted, detection limits are increased accordingly. A new analytical approach was introduced in 2011 (i.e., collision cell mass spectrometry; CCMS), which improved the sensitivity for cadmium. However, samples from sites closest to the LLCF (i.e., Leslie, Moose, and Nema lakes) have still occasionally required dilution under CCMS methods, resulting in elevated detection limits. To improve results for these samples, an alternative method (i.e., solid phase extraction; SPE) was implemented in 2014. The target detection limit of 0.00002 mg/L for SPE is greater that the target detection limit of 0.0001 mg/L for regular CCMS. Thus, SPE was only implemented in cases where detection limits under CCMS were elevated above 0.0002 mg/L.

Figure 3.2-16

Observed and Fitted Means for Total Cadmium Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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 = $10^{0.83 X (log10Hardness - 2.46)}$/1000 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.

Table 3.2-20. 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	-	-	-	-	-	1-259
Aug	Lake	ALL	-	-	-	-	-	1-262
Aug	Stream	ALL	-	-	-	-	-	1-265

Note: Dashes indicate not applicable.

The detection limit for total cadmium was below the cadmium hardness-dependent CCME guideline for all observations in 2014. Observed cadmium concentration in one sample from Moose-Nero Stream in June was greater than the cadmium hardness-dependent CCME guideline (CCME 2014a). It was concluded that no mine effects were detected for total cadmium in lakes and streams of the Koala Watershed and Lac de Gras.

3.2.4.17 *Total Molybdenum*

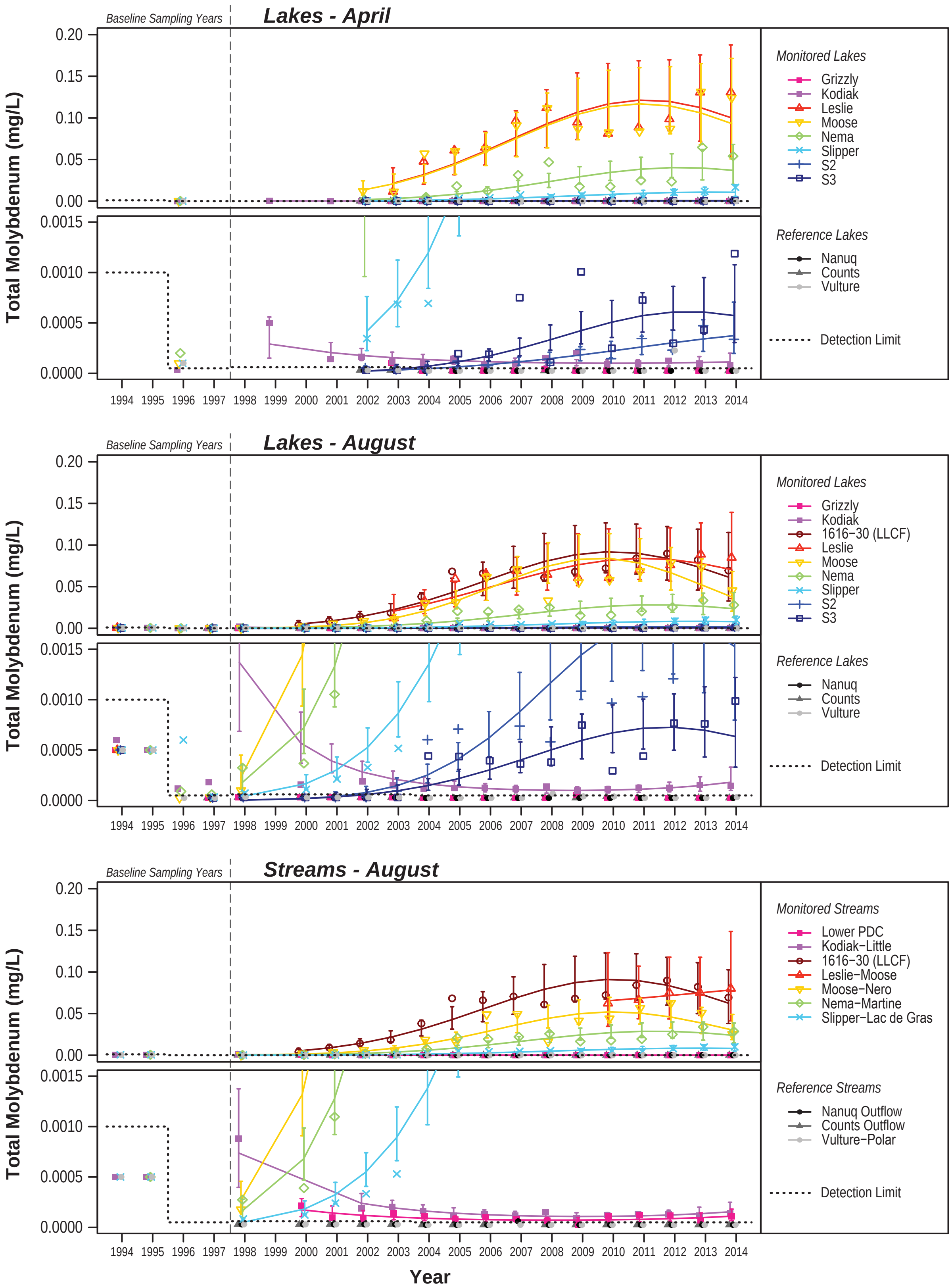
Summary: Concentrations of molybdenum have increased in monitored lakes and streams downstream of the LLCF as far as site S3 in Lac de Gras. However, molybdenum concentrations have stabilised at levels greater than observed historical and reference lake concentrations at sites as far downstream as Nema 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 2014. 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 3.2-21; Figure 3.2-17). Only five years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends unlikely; 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 sites S2 and S3 in Lac de Gras during the ice-covered 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.

At sites that are not downstream of the LLCF, statistical analyses indicate that total molybdenum concentrations have changed through time in Kodiak Lake, the Lower PDC, and Kodiak-Little Stream (Table 3.2-21). However, graphical analysis suggests that concentrations have decreased slightly through time to stabilise at current levels shortly after monitoring began at all of these sites (Figure 3.2-17). Thus, no mine effects were detected at sites that are not downstream of the LLCF.

Figure 3.2-17

Observed and Fitted Means for Total Molybdenum Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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 3.2-21. 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	1-268
Aug	Lake	Grizzly, Nanuq, Counts, Vulture	Tobit	1a	-	-	Kodiak, 1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2, S3	1-273
Aug	Stream	Nanuq Outflow, Counts Outflow, Vulture-Polar	LME	1a	-	-	Lower PDC, Kodiak-Little, 1616-30 (LLCF), Moose-Nero, Nema-Martine, Slipper-Lac de Gras	1-278

Note: Dashes indicate not applicable.

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 2014 during both the ice-covered and open water seasons (see Part 2 - Data Report; Rescan 2012a).

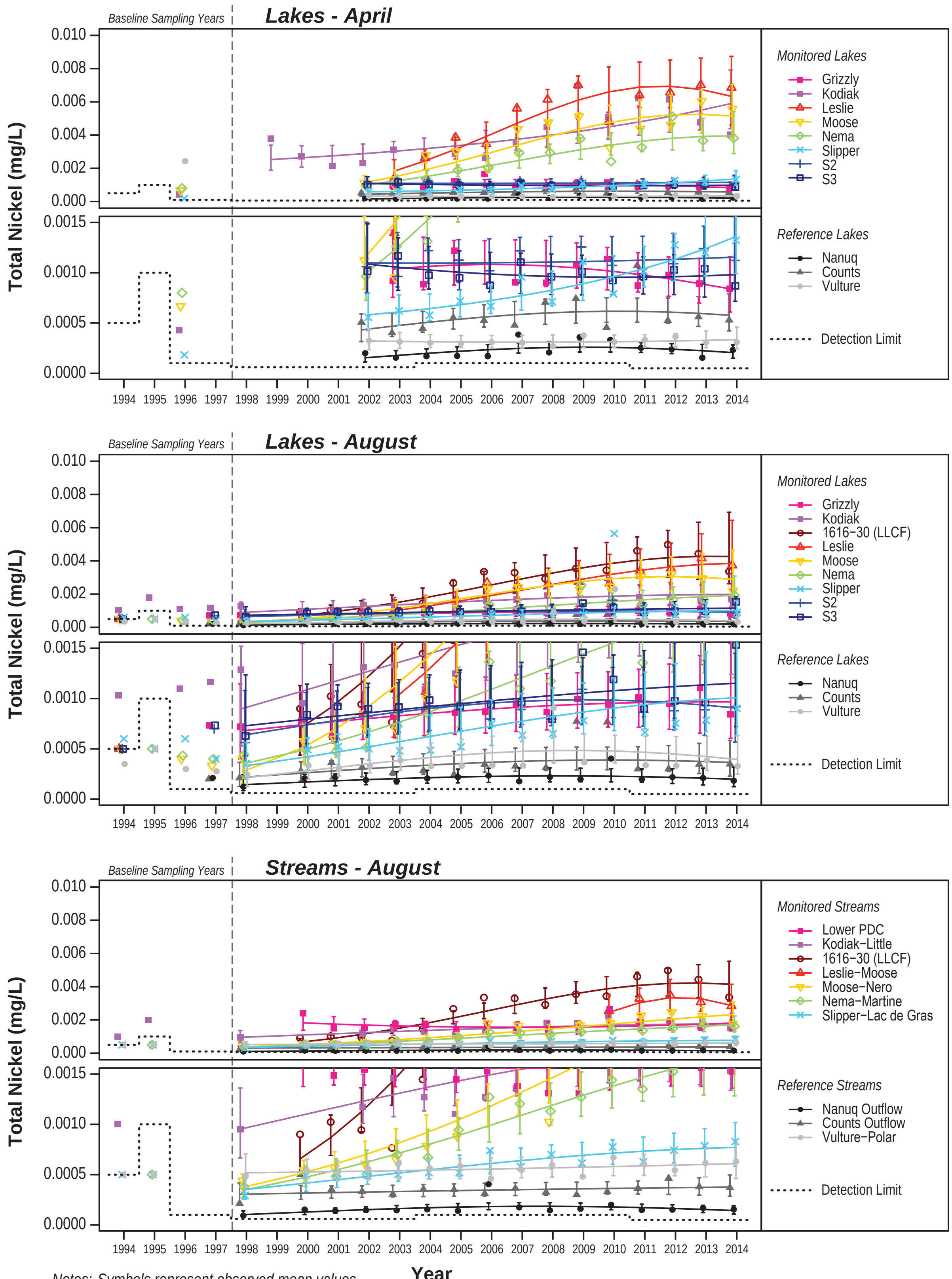
3.2.4.18 Total Nickel

> ***Summary:*** Statistical and graphical analyses suggest that total nickel concentrations have increased in all lakes and streams downstream of the LLCF as far as Slipper-Lac de Gras Stream as a result of mine operations. However, nickel 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, 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 2014.

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 3.2-22). Only five years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends improbable; however, graphical analysis shows that total nickel concentrations in Leslie-Moose Stream were similar to those in the LLCF in recent years. Graphical analyses suggest that total nickel concentrations have increased through time in all lakes and streams downstream of the LLCF as far as Slipper-Lac de Gras Stream, but that concentrations have stabilised in Leslie, Moose, and Nema lakes 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 Slipper-Lac de Gras 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 Slipper-Lac de Gras 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 2014



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 = $e^{0.76 X (\ln Hardness) + 1.06}/1000$ 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.

Table 3.2-22. 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			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	2	-	Leslie, Moose, Nema		1-283
Aug	Lake	-	LME	2	-	1616-30 (LLCF), Leslie, Moose, Nema		1-289
Aug	Stream	-	LME	1b	-	-	Kodiak-Little, 1616-30 (LLCF), Moose-Nero, Nema-Martine, Slipper-Lac de Gras	1-295

Note: Dashes indicate not applicable.

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 3.2-22). 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 the last two years (Table 3.2-22; 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 2014 (CCREM 1987). Total nickel concentrations were less than the hardness-dependent CCREM guideline for all monitored streams in June, July, August, and September 2014 (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. However, graphical analysis suggests that total selenium concentrations have increased downstream of the LLCF as far as Moose Lake as a result of mine operations. Observed and fitted mean concentrations were less than the selenium CCREM guideline at all sites in 2014. No mine effects were detected.

With the exception of a few values observed in 2006, and from 2009 to 2014, 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, most lakes and streams were removed from the statistical analyses, especially during the open water season (Table 3.2-23). Although historically variable detection limits resulting from matrix interference due to elevated chloride concentrations during ICPMS analysis have made it somewhat difficult to discern clear temporal trends, 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) and some streams in June 2014. Graphical analysis suggests that selenium concentrations have increased at sites downstream of the LLCF as far as Moose Lake (Figure 3.2-19).

Table 3.2-23. 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, S2, S3, Nanuq, Counts, Vulture	Tobit	1a	-	-	None	1-301
Aug	Lake	Grizzly, Kodiak, Nema, Slipper, S2, S3, Nanuq, Counts, Vulture	Tobit	1a	-	-	1616-30 (LLCF)	1-306
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)	1-311

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 2014 (CCREM 1987).

3.2.4.20 Total Strontium

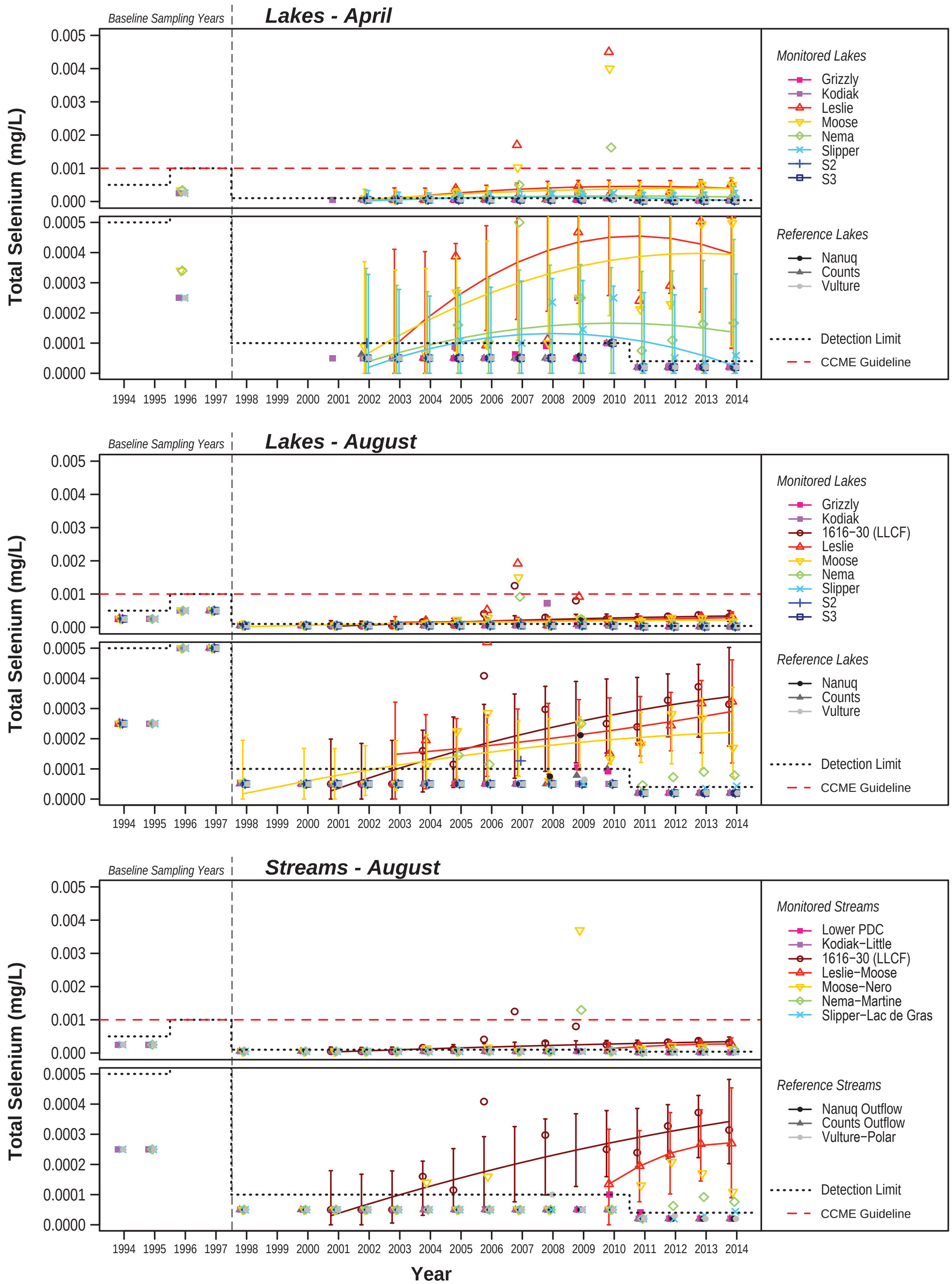
Summary: Statistical and graphical analyses suggest that total strontium concentrations have increased at all sites downstream from the LLCF as far as site S3 in Lac de Gras. No mine effects were detected at sites that are not downstream of the LLCF. Observed and fitted mean concentrations were less than the strontium water quality benchmark (6.242 mg/L) at all sites in 2014.

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 3.2-24; 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 3.2-24; 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 2014 (see Part 2 - Data Report; Golder 2011).

Figure 3.2-19

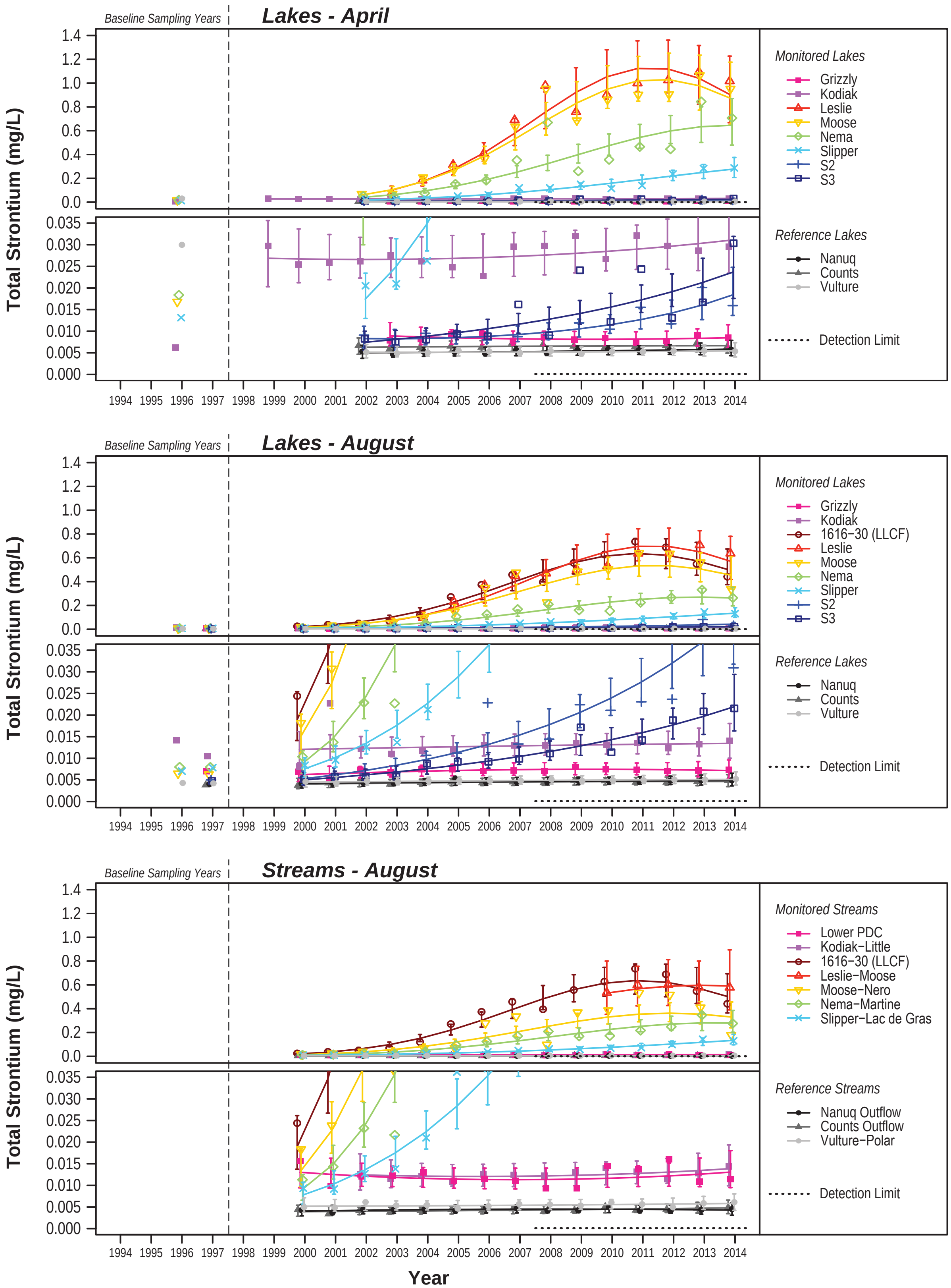
Observed and Fitted Means for Total Selenium Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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

Figure 3.2-20

Observed and Fitted Means for Total Strontium Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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 3.2-24. 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	-	1-316
Aug	Lake	-	LME	3	Grizzly, Kodiak, 1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2, S3	1616-30 (LLCF), Leslie, Moose, Nema, Slipper, S2, S3	-	1-322
Aug	Stream	-	LME	1b	-	-	1616-30 (LLCF), Moose-Nero, Nema-Martine, Slipper-Lac de Gras	1-328

Note: Dashes indicate not applicable.

3.2.4.21 Total Uranium

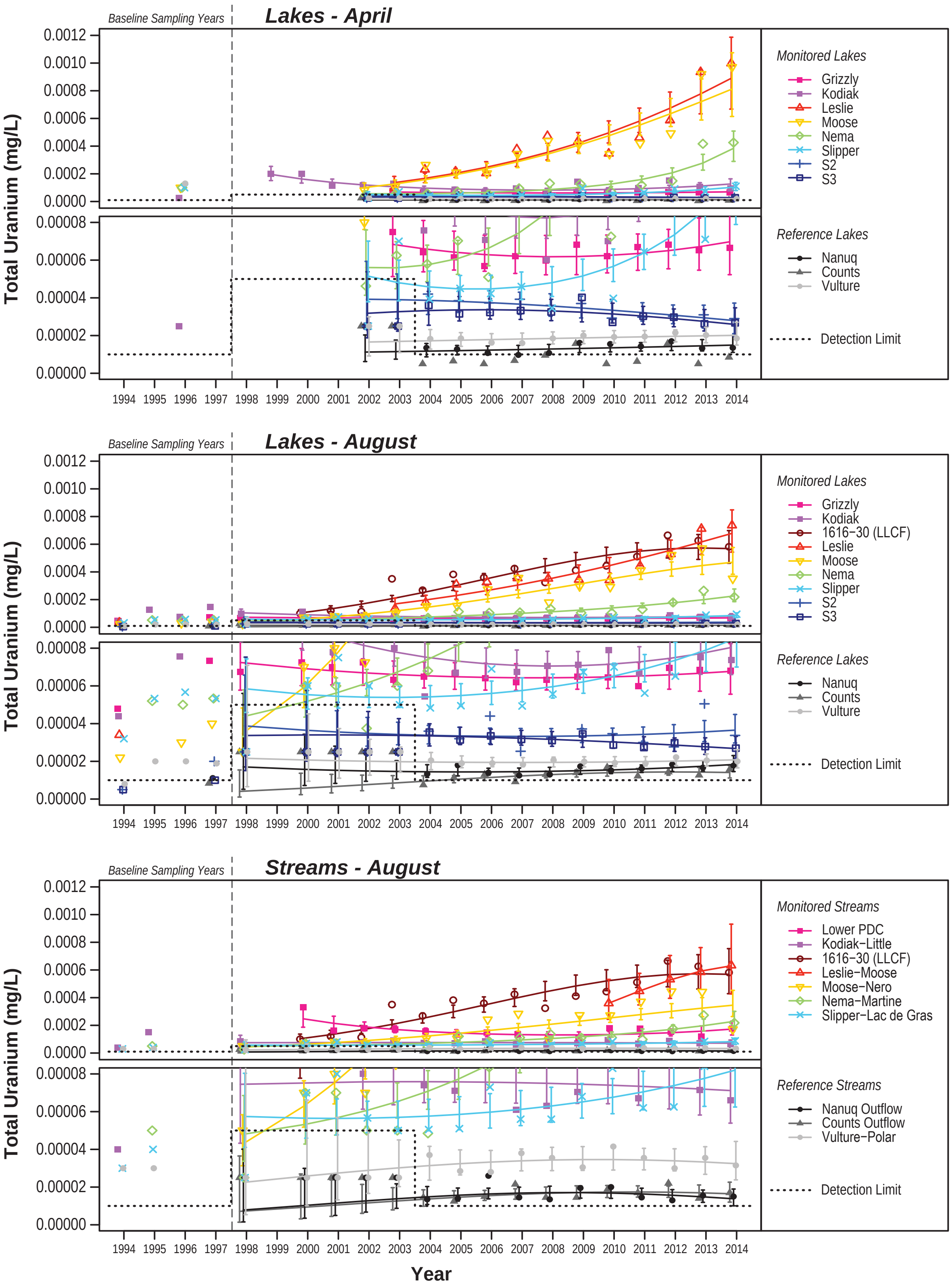
> ***Summary:*** Statistical and graphical analyses suggest that total uranium concentrations have increased in all lakes and streams downstream from the LLCF as far as Slipper Lake as a result of mine operations. No mine effects were detected at sites that are not downstream of the LLCF. Observed and fitted mean concentrations were less than the uranium CCME guideline at all sites in 2014.

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 Slipper Lake during the ice-covered season, and as far as Nema-Martine Stream during the open water season, except at Leslie-Moose Stream (Table 3.2-25; Figure 3.2-21). Only five years of data have been collected from Leslie-Moose Stream, rendering the statistical detection of trends unlikely; 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).

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 3.2-25). 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.

Figure 3.2-21

Observed and Fitted Means for Total Uranium Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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.

Table 3.2-25. 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, Slipper	-	1-334
Aug	Lake	-	Tobit	2	-	1616-30 (LLCF), Leslie, Moose, Nema	-	1-340
Aug	Stream	-	Tobit	2	-	Lower PDC, 1616-30 (LLCF), Moose-Nero, Nema-Martine	-	1-346

Note: Dashes indicate not applicable.

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 2014 (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 2014. No mine effects were detected.

Statistical 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, expect for Moose Lake (Table 3.2-26; Figure 3.2-22). However, graphical analysis suggests that total vanadium concentrations in Moose Lake have been stable through time (Table 3.2-26; Figure 3.2-22). Thus, no mine effects were detected in any monitored lakes or streams of the Koala Watershed and Lac de Gras.

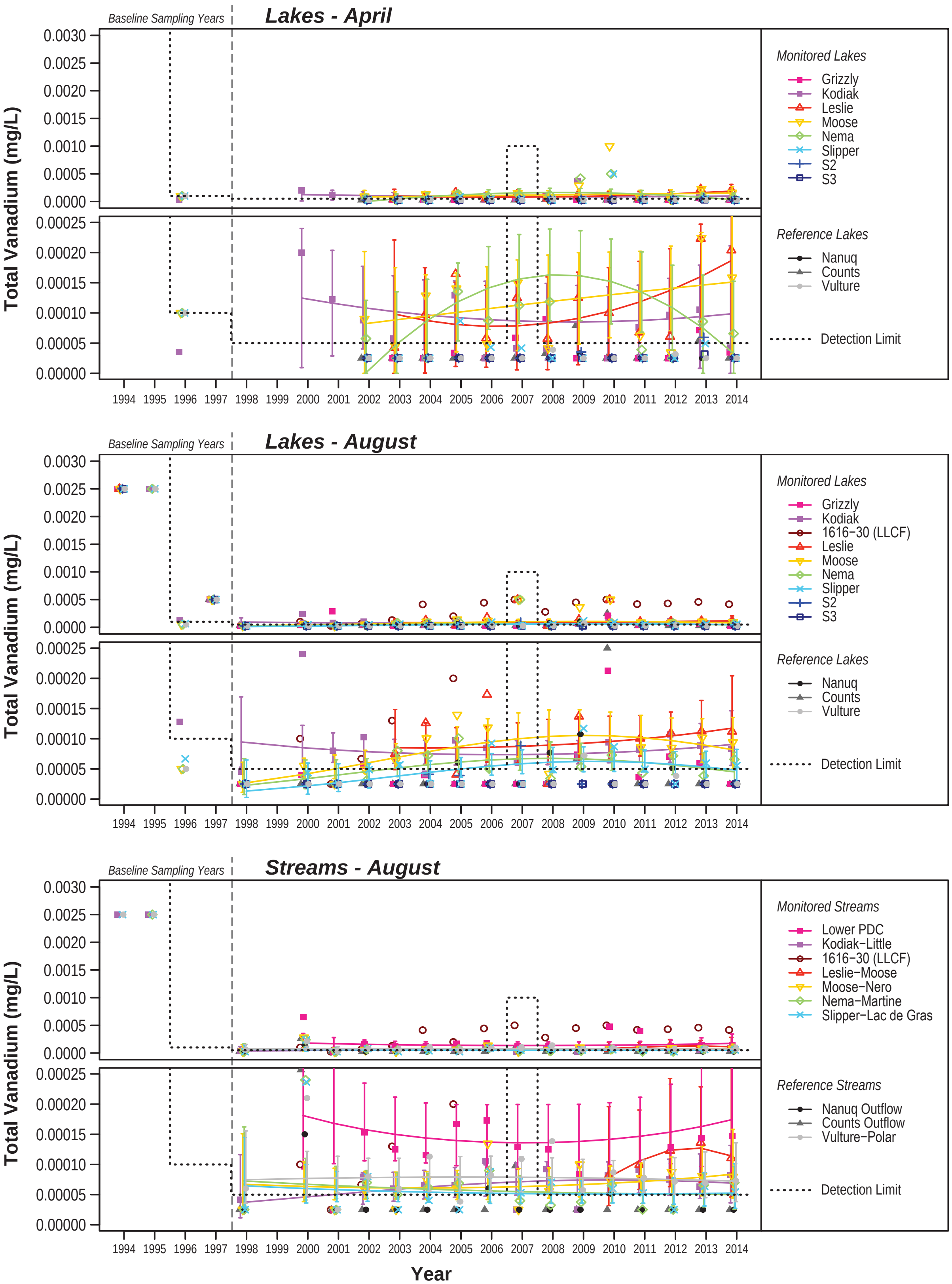
Table 3.2-26. Statistical Results of Total Vanadium Concentrations in Lakes and Streams in the Koala Watershed and Lac de Gras

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	1-352
Aug	Lake	Grizzly, 1616-30 (LLCF), S2, S3, Counts, Nanuq, Vulture	Tobit	1a	-	-	Moose	1-357
Aug	Stream	1616-30 (LLCF), Counts Outflow, Nanuq Outflow	Tobit	1b	-	-	None	1-362

Note: Dashes indicate not applicable.

Figure 3.2-22

Observed and Fitted Means for Total Vanadium Concentrations in Koala Watershed Lakes and Streams and Lac de Gras, 1994 to 2014



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.

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 2014 (see Part 2 - Data Report; Rescan 2012g).

3.3 LAKE SEDIMENT QUALITY

3.3.1 Variables

Eleven lake sediment quality variables were evaluated for potential effects caused by mine activities in the Koala, King-Cujo, and Pigeon-Fay and Upper Exeter watersheds. These included nutrients (TOC, available phosphorus, total nitrogen) and metals (antimony, arsenic, cadmium, molybdenum, nickel, phosphorus, selenium, and strontium). A 12th variable was also evaluated in the King-Cujo Watershed (total copper). CCME guidelines for the protection of aquatic life exist for three of the evaluated sediment quality variables, including arsenic, cadmium, and copper (see Section 2.4; CCME 2014b).

3.3.2 Dataset

The lake sediment quality data used in the 2014 evaluation of effects were collected from late July to mid-August every third year from 1998 to 2014 (Tables 3.3-1 and 3.3-2). Baseline sediment quality data collected from 1994 to 1997 were not used in the statistical evaluation of effects but are included in Tables 3.3-1 and 3.3-2 and shown in Figures 3.3-1 to 3.3-11 for visual comparison. In 1994, sediment quality data were collected in both early July and mid-August, but sediment quality did not differ significantly between these sampling periods (Rescan 2009). Thus, data from early July 1994 are included in Tables 3.3-1 and 3.3-2 and shown in Figures 3.3-1 to 3.3-11. Subsequent sampling occurred once per open water season, in early August.

Between 1998 and 2001, Kodiak Lake sediment quality data was collected as part of the Kodiak Lake Sewage Effects Study (KLSES; Rescan 2002). Sediment quality data collected from Kodiak Lake as part of the KLSES were screened and selected to correspond to AEMP sampling dates. Kodiak Lake sediment quality monitoring has been part of the AEMP since 2002. In Leslie Lake, sediment quality has only been monitored since 2005 and only four years of data exist.

Sediment sampling methods have been consistent since monitoring began in 1994. Samples were collected using a standard Ekman grab and the top 2 cm were collected for analysis. In 2011 and 2014, sediment samples at all AEMP lake sites were also collected using a Kajak-Brinkhurst (K-B) corer. Samples from 2011 were collected as part of a study to determine whether core sampling might provide a better measure of potential changes in sediment chemistry (Rescan 2012c). Results from that study will be further examined along with results from 2014 as part of the 2015 AEMP Re-evaluation. ALS has been analyzing the AEMP sediment samples since 1994. Analytical detection limits for sediment quality variables are illustrated as black dashed lines (Figures 3.3-1 to 3.3-11).

Analyses are conducted on sediments collected from one depth strata: mid (5.1 - 10 m). Shallow samples (<5 m) of lake sediment and benthos were eliminated from the AEMP in 2007 because the physical and biological characteristics of the shallow benthic areas of EKATI lakes are too variable to reasonably discriminate potential mine effects from natural variability (Rescan 2006). Following the 2012 AEMP Re-evaluation, deep depth (>10 m) sediment and benthos samples were removed from the AEMP since it was determined that deep depth sampling represented a duplication of effort (Rescan 2012c).

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

Year	Nanuq	Counts	Vulture	Kodiak	Leslie	Moose	Nema	Slipper	S2
1994*	-	-	Jul-1 (1), Aug-13 (1)	Aug-19 (2)	-	Jul-6 (1), Aug-22 (2)	-	Jul-7 (1), Aug-15 (1)	Jul-8 (2), Aug-14 (1)
1997*	Aug-4 (1)	Aug-4 (1)	Aug-4 (1)	Aug-8 (1)	-	Aug-4 (1)	Aug-4 (1)	Aug-4 (1)	Aug-4 (1)
1998	Aug-4 (3)	Aug-4 (2)	Aug-7 (3)	Jul-29 (3)	-	Aug-8 (3)	Aug-7 (5)	Aug-6 (1)	Aug-5 (3)
1999	Jul-30 (3)	Jul-30 (3)	Jul-29 (3)	Aug-6 (3)	-	Aug-2 (3)	Aug-2 (3)	Aug-1 (3)	Aug-1 (3)
2001	-	-	-	Jul-28 (3)	-	-	-	-	-
2002	Aug-3 (3)	Aug-7 (3)	Aug-3 (3)	Aug-2 (3)	-	Aug-5 (3)	Aug-4 (3)	Aug-6 (3)	Aug-4 (3)
2005	Aug-1 (3)	Aug-7 (3)	Jul-31 (3)	Aug-3 (3)	Aug- 4 (3)	Aug-9 (3)	Aug-9 (3)	Aug-5 (3)	Aug-5 (3)
2008	Aug-8 (3)	Jul-31 (3)	Aug-5 (3)	Jul-27 (3)	Jul- 31 (3)	Jul-29 (3)	Jul-29 (3)	Jul-29(3)	Aug-7 (3)
2011	Aug-13 (3)	Aug-11 (1), Aug-14 (1)	Aug-13 (3)	Aug-17 (3)	Aug- 20 (3)	Aug-10 (3)	Aug-17 (3)	Aug-12 (3)	Aug-12 (3)
2014	Aug-5 (3)	Aug-9 (3)	Aug-3 (3)	Jul-29 (3)	Jul-31 (3)	Jul-31 (3)	Aug-2 (3)	Aug-4 (3)	Aug-3 (3)

Notes: Number of replicates is indicated in brackets.
Dashes indicate no data were available.
** Indicates 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 year in which sediment data was collected, averages for the mid depth strata were calculated by pooling data from replicates collected within that strata. Several data points were not included in the analyses either for statistical reasons, as a result of analytical error or contamination. These include two statistical outliers (the arsenic concentration from mid depth replicate 3 from Vulture Lake collected on July 31, 2005 and replicate 3 of the sediment sample collected from site S3 in Lac de Gras on August 5, 2005) and the nitrogen data from Kodiak Lake in 2008, which was excluded as a consequence of logistical and analytical error.

3.3.3 Results and Discussion

3.3.3.1 TOC

Summary: Statistical and graphical analyses suggest that mine operations have had no effect on TOC at any of the monitored sediment sites of the Koala Watershed or Lac de Gras.

Statistical analyses indicate that TOC percentages in sediments have changed over time, relative to reference lakes, in Leslie Lake and at site S2 in Lac de Gras (Table 3.3-2). However, graphical analysis suggests that TOC percentages at all monitored sites have been stable over time and similar to that in reference lakes and during baseline years (Figure 3.3-1). Thus, no mine effects were detected.

Table 3.3-2. Statistical Results for Total Organic Carbon in Sediments in Koala Watershed Lakes 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	
Total Organic Carbon	-	LME	1b	-	-	Leslie, S2	1-367

Note: Dashes indicate not applicable.

3.3.3.2 Available Phosphorus

Summary: Statistical and graphical analyses suggest that mine operations have had no effect on available phosphorus concentrations at any of the monitored sediment sites of the Koala Watershed or Lac de Gras.

Statistical analyses indicate that available phosphorus concentrations in sediments have changed through time, relative to reference lakes, at site S2 in Lac de Gras (Table 3.3-3). However, graphical analysis suggests that available phosphorus concentrations at site S2 have decreased over time (Figure 3.3-2). Thus, no mine effects were detected.

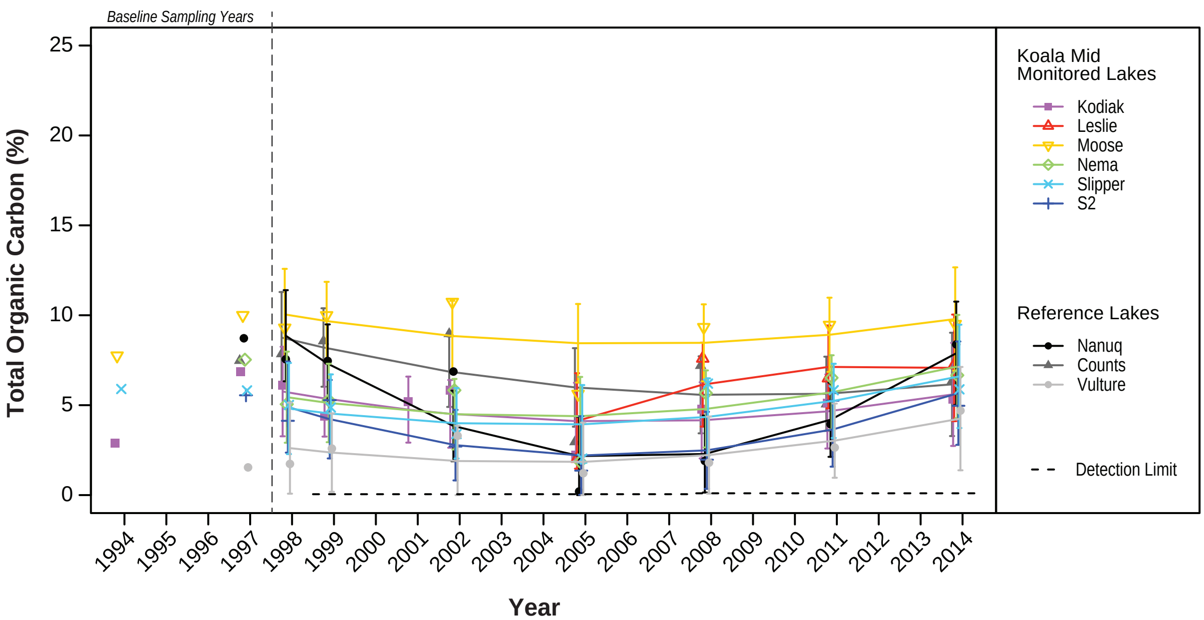
Table 3.3-3. Statistical Results for Available Phosphorus Concentrations in Sediments in Koala Watershed Lakes and Lac de Gras

	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	
Available Phosphorus	-	Tobit	2	-	S2	-	1-373

Note: Dashes indicate not applicable.

Figure 3.3-1

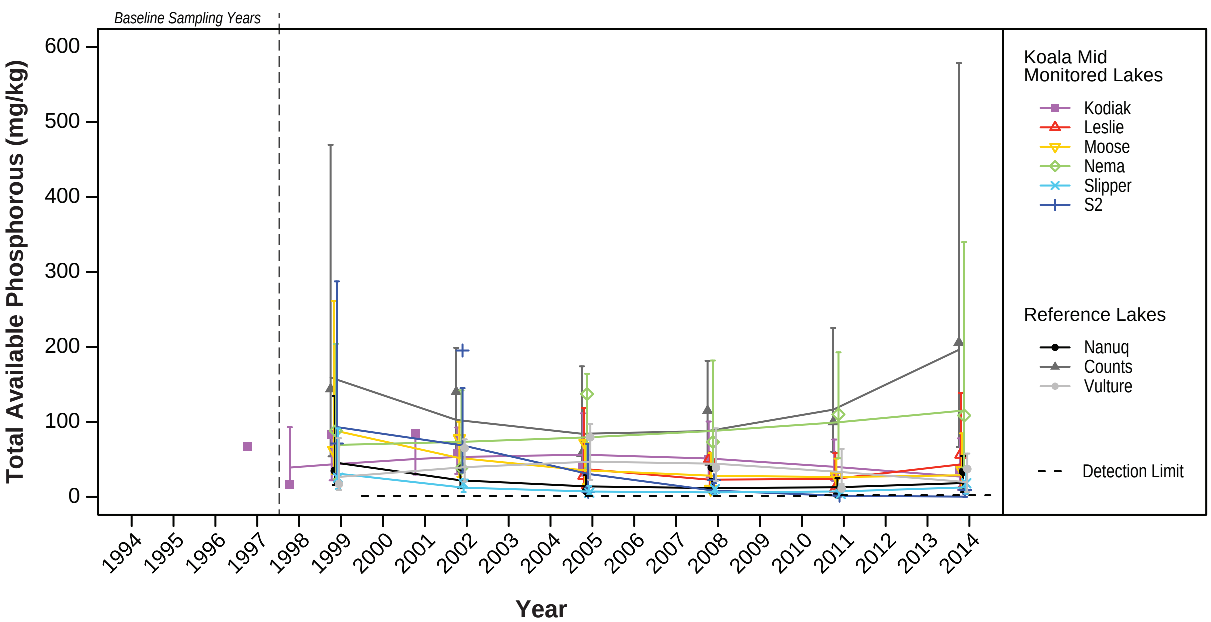
Observed and Fitted Means for Total Organic Carbon Percentages in Sediments in Koala Watershed Lakes and Lac de Gras, 1994 to 2014



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 3.3-2

Observed and Fitted Means for Available Phosphorus Concentrations in Sediments in Koala Watershed Lakes and Lac de Gras, 1994 to 2014



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

3.3.3.3 Total Nitrogen

Summary: Statistical and graphical analyses suggest that mine operations have had no effect on total nitrogen at any of the monitored sediment sites of the Koala Watershed or Lac de Gras.

Statistical analyses indicate that total nitrogen percentages in sediments have changed over time, relative to reference lakes, in Slipper Lake and at site S2 in Lac de Gras (Table 3.3-4). Graphical analysis suggests that total nitrogen percentages in sediments of monitored lakes have generally been within the range observed for reference lakes since monitoring began (Figure 3.3-3). Graphical analysis also suggests that there have been recent increases in total nitrogen in Nema and Slipper lakes and at site S2 in Lac de Gras; however, a similar pattern was observed in one of the three reference lakes (i.e., Nanuq Lake; Figure 3.3-3). Thus, no mine effects were detected at this time.

Table 3.3-4. Statistical Results for Total Nitrogen Concentrations in Sediments in Koala Watershed Lakes 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	
Total Nitrogen	-	LME	1b	-	-	Slipper, S2	1-379

Note: Dashes indicate not applicable.

3.3.3.4 Total Antimony

Summary: No statistical analyses were possible at this time; however, graphical analysis suggests that antimony concentrations are greater than those observed in reference lakes in sediments of all monitored sites downstream of the LLCF, as far as Slipper Lake. The pattern of decreasing concentration with increasing distance from the LLCF suggests a mine effect.

Antimony concentrations in sediments have only been analyzed for three years (i.e., 2008, 2011, and 2014). Thus, all lakes were excluded from the statistical analyses and no tests were performed (Table 3.3-5). Graphical analysis and best professional judgment were the primary methods used in the evaluation of effects. Graphical analysis suggests that observed antimony concentrations in sediments of lakes downstream from the LLCF as far as Slipper Lake were greater than those observed in reference lakes, with concentrations decreasing with downstream distance from the LLCF (Figure 3.3-4). At sites that are not downstream of the LLCF, graphical analysis suggests that antimony concentrations have been comparable to those observed in reference lakes (Figure 3.3-4).

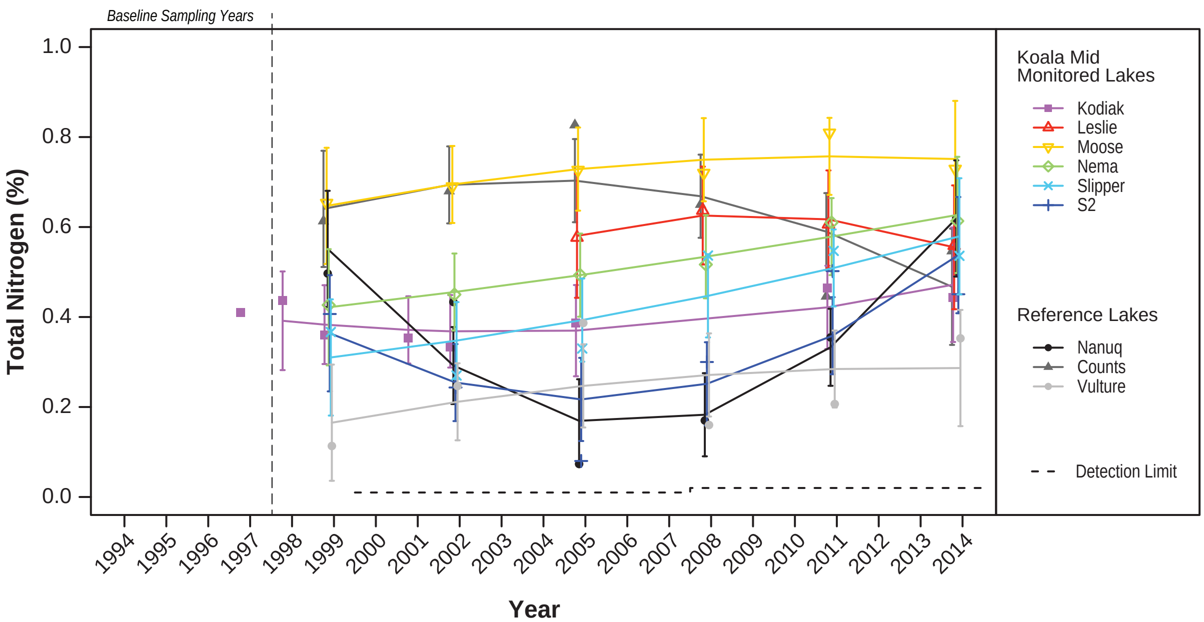
Table 3.3-5. Statistical Results for Total Antimony Concentrations in Sediments in Koala Watershed Lakes and Lac de Gras

	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	
Total Antimony	ALL	-	-	-	-	-	1-385

Note: Dashes indicate not applicable.

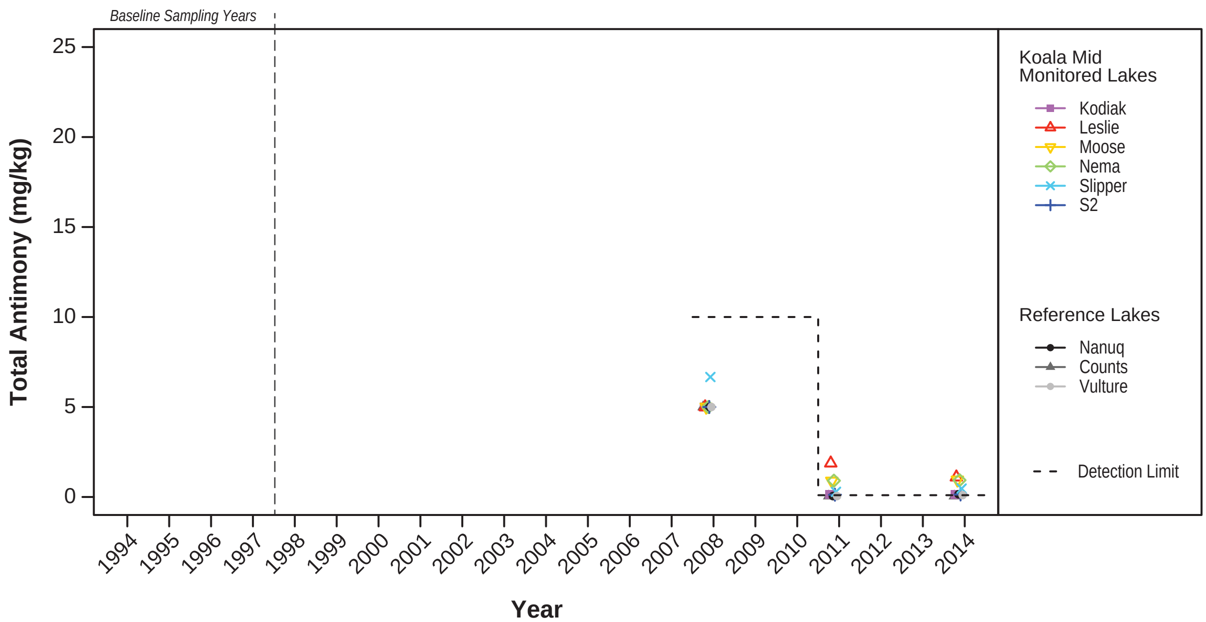
Figure 3.3-3

Observed and Fitted Means for Total Nitrogen Percentages in Sediments in Koala Watershed Lakes and Lac de Gras, 1994 to 2014



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

Observed and Fitted Means for Total Antimony Concentrations in Sediments in Koala Watershed Lakes and Lac de Gras, 1994 to 2014



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

Starting in 2011, once antimony concentrations in sediments were above detection limits, antimony concentrations in sediments follow a similar pattern to that observed for total antimony concentrations in water quality samples (i.e., decreasing concentration with increasing distance from the LLCF; see Section 3.2.4.12); therefore, elevated antimony concentrations in sediments may stem from antimony contained in LLCF discharge.

3.3.3.5 Total Arsenic

Summary: Statistical and graphical analyses suggest that mine operations have had no effect on arsenic concentrations at any of the monitored sediment sites of the Koala Watershed or Lac de Gras. The observed mean exceeded the CCME ISQG in all monitored sites and the CCME PEL in Slipper Lake and at site S2 in Lac de Gras. The 95% confidence intervals around the fitted mean arsenic concentration exceeded the CCME PEL in all monitored sites, except Nema Lake. However, similar exceedance patterns were observed in all three reference lakes.

Statistical analyses indicate that arsenic concentrations in sediments have changed through time, relative to reference lakes, at site S2 in Lac de Gras (Table 3.3-6). Graphical analysis suggests that arsenic concentrations at site S2 have increased in recent years; however, recent concentrations are less than those observed during baseline years (Figure 3.3-5).

Table 3.3-6. Statistical Results for Total Arsenic Concentrations in Sediments in Koala Watershed Lakes 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	
Total Arsenic	-	LME	1b	-	-	S2	1-387

Note: Dashes indicate not applicable.

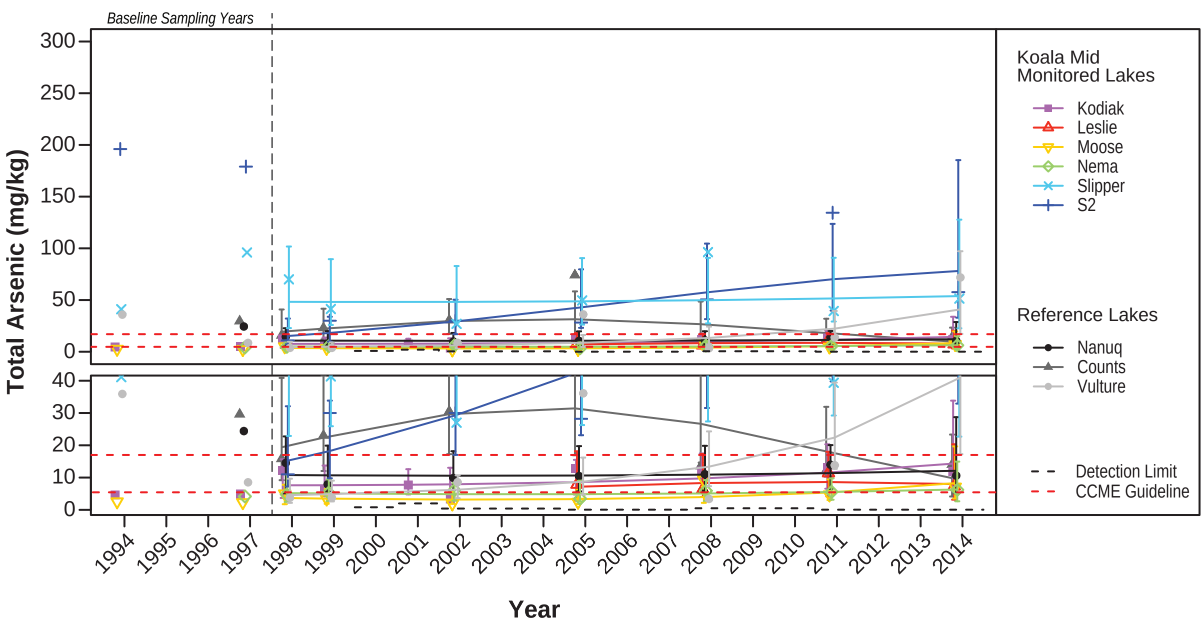
The observed mean arsenic concentration in 2014 exceeded the CCME ISQG of 5.9 mg/kg in all monitored and reference sites and exceeded the CCME PEL of 17 mg/kg in Slipper Lake and at site S2 in Lac de Gras (CCME 2001a). The 95% confidence intervals around the fitted mean in 2014 also exceeded the CCME PEL in all monitored and reference lakes, except Nema Lake. Similar patterns were observed in all three reference lakes and arsenic concentrations remained within the range of baseline concentrations observed in each lake through time (Figure 3.3-5). Thus, no mine effects were detected at any of the monitored sediment sites in the Koala Watershed or Lac de Gras.

3.3.3.6 Total Cadmium

Summary: Statistical and graphical analyses suggest that mine operations have had no effect on cadmium concentrations at any of the monitored sediment sites of the Koala Watershed or Lac de Gras. The 95% confidence intervals around the fitted mean in 2014 exceeded the CCME ISQG in Slipper Lake and at site S2 in Lac de Gras; however, a similar pattern was observed in one reference lake. Cadmium concentrations in sediments were less than the CCME PEL value in all monitored sites.

Figure 3.3-5

Observed and Fitted Means for Total Arsenic Concentrations in Sediments in Koala Watershed Lakes and Lac de Gras, 1994 to 2014



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 guidelines: ISQG = 5.9 mg/kg; PEL = 17 mg/kg.

Statistical analyses indicate that cadmium concentrations in sediments have changed through time, relative to reference lakes, at site S2 in Lac de Gras (Table 3.3-7). Graphical analysis suggests that cadmium concentrations at site S2 increased from 2005 to 2011; however, current concentrations are less than those observed during baseline years (Figure 3.3-6). The 95% confidence intervals around the fitted mean in 2014 exceeded the CCME ISQG of 0.6 mg/kg in Slipper Lake and at site S2 in Lac de Gras; however, a similar pattern was observed in one reference lake (i.e., Nanuq Lake; Figure 3.3-6; CCME 1999a). The 95% confidence intervals of the fitted mean and the observed mean cadmium concentrations were less than the CCME PEL of 3.5 mg/kg at all monitored sediment sites in 2014 (CCME 1999a). Thus, no mine effects were detected at any of the monitored sediment sites in the Koala Watershed or Lac de Gras.

Table 3.3-7. Statistical Results for Total Cadmium Concentrations in Sediments in Koala Watershed Lakes 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	
Total Cadmium	Leslie, Moose	Tobit	1b	-	-	S2	1-393

Note: Dashes indicate not applicable.

3.3.3.7 *Total Molybdenum*

Summary: Statistical and graphical analyses suggest that molybdenum concentrations in sediments have increased at all monitored sites that are downstream of the LLCF as far as Slipper Lake as a result of mine operations. No mine effects were detected at sites that are not downstream of the LLCF.

Statistical analyses indicate that molybdenum concentrations in sediments have changed through time at all sites downstream of the LLCF, as far as Slipper Lake, with the exception of Leslie Lake (Table 3.3-8). Only four years of data have been collected from Leslie Lake, rendering the statistical detection of trends less likely. Graphical analysis suggests that molybdenum concentrations in sediments have increased at all sites downstream of the LLCF, as far as Slipper Lake, with concentrations decreasing with downstream distance from the LLCF (Figure 3.3-7). Molybdenum concentrations in sediments follow the same pattern observed for total molybdenum concentrations in water quality samples (see Section 3.2.4.17); therefore, increased molybdenum concentrations in sediments likely stem from molybdenum contained in LLCF discharge.

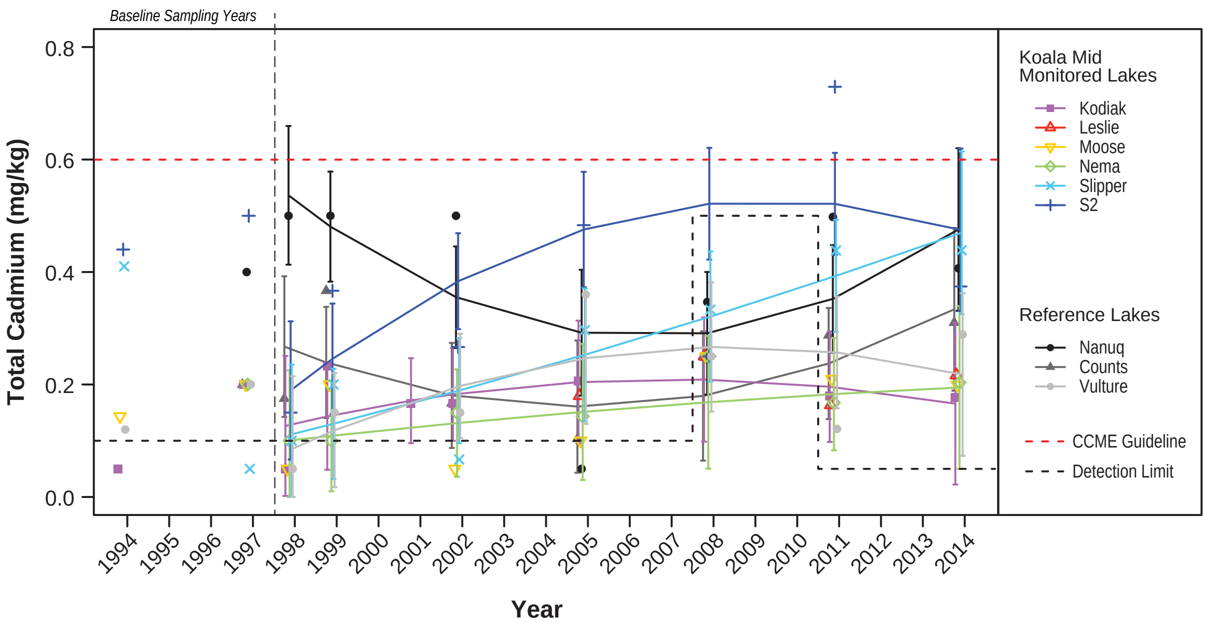
Table 3.3-8. Statistical Results for Total Molybdenum Concentrations in Sediments in Koala Watershed Lakes and Lac de Gras

	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	Nanuq, Counts, Vulture, Kodiak, S2	Tobit	1a	-	-	Moose, Nema, Slipper	1-399

Note: Dashes indicate not applicable.

Figure 3.3-6

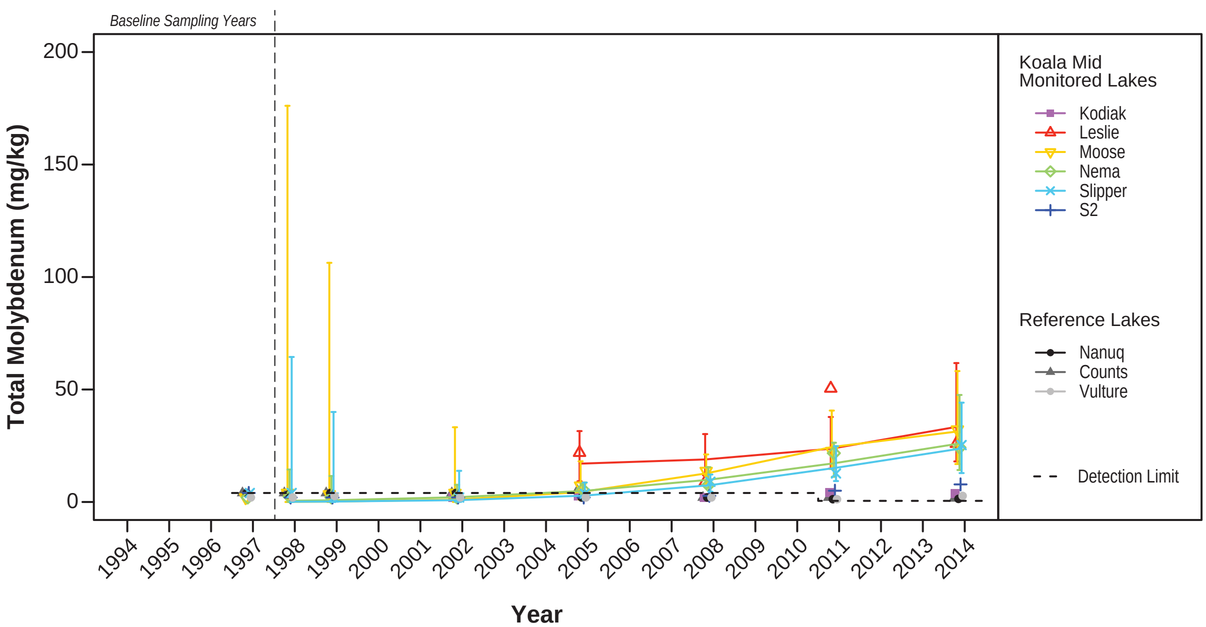
Observed and Fitted Means for Total Cadmium Concentrations in Sediments in Koala Watershed Lakes and Lac de Gras, 1994 to 2014



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 guidelines: ISQG = 0.6 mg/kg; PEL = 3.5 mg/kg (not shown).

Figure 3.3-7

Observed and Fitted Means for Total Molybdenum Concentrations in Sediments in Koala Watershed Lakes and Lac de Gras, 1994 to 2014



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

3.3.3.8 Total Nickel

Summary: Statistical and graphical analyses suggest that mine operations have had no effect on nickel concentrations at any of the monitored sediment sites of the Koala Watershed or Lac de Gras.

Statistical and graphical analyses indicate that nickel concentrations in sediments have remained stable through time, relative to reference lakes, at all monitored sites in the Koala Watershed and Lac de Gras (Table 3.3-9; Figure 3.3-8). Thus, no mine effects were detected.

Table 3.3-9. Statistical Results for Total Nickel Concentrations in Sediments in Koala Watershed Lakes 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	
Total Nickel	-	LME	2	-	None	-	1-404

Note: Dashes indicate not applicable.

3.3.3.9 Total Phosphorus

Summary: Statistical and graphical analyses suggest that mine operations have had no effect on total phosphorus concentrations at any of the monitored sediment sites of the Koala Watershed or Lac de Gras.

Statistical and graphical analyses indicate that total phosphorus concentrations in sediments have remained stable through time, relative to reference lakes, at all monitored sites in the Koala Watershed and Lac de Gras (Table 3.3-10; Figure 3.3-9). Thus, no mine effects were detected.

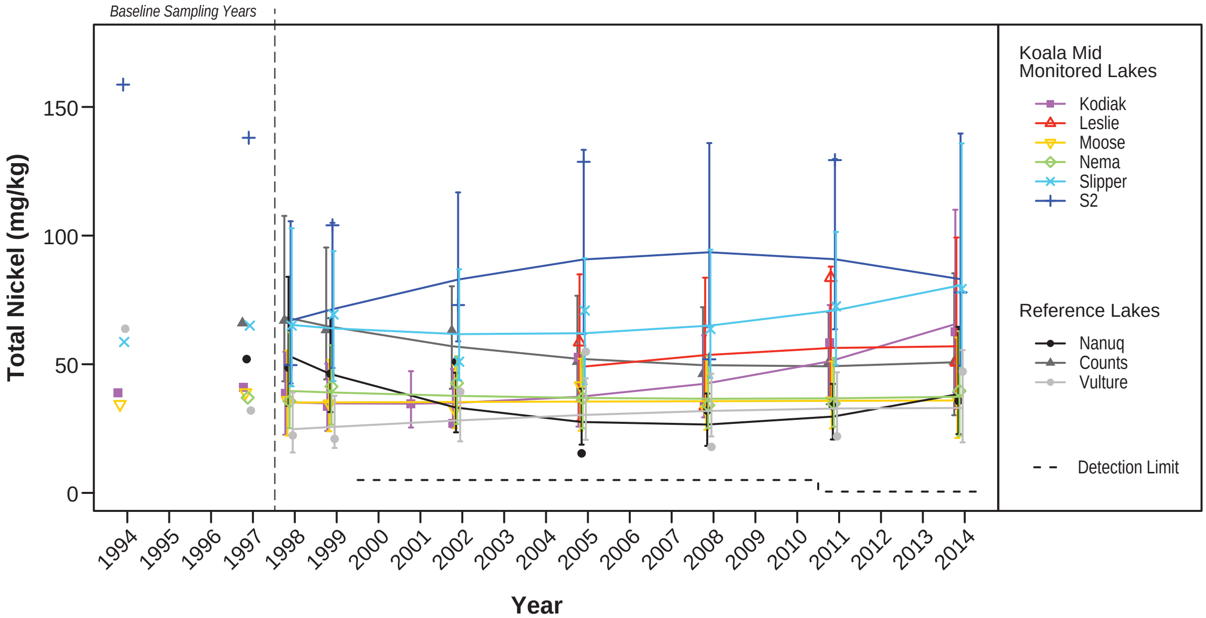
Table 3.3-10. Statistical Results for Total Phosphorus Concentrations in Sediments in Koala Watershed Lakes and Lac de Gras

	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	
Total Phosphorus	-	LME	1b	-	-	None	1-410

Note: Dashes indicate not applicable.

Figure 3.3-8

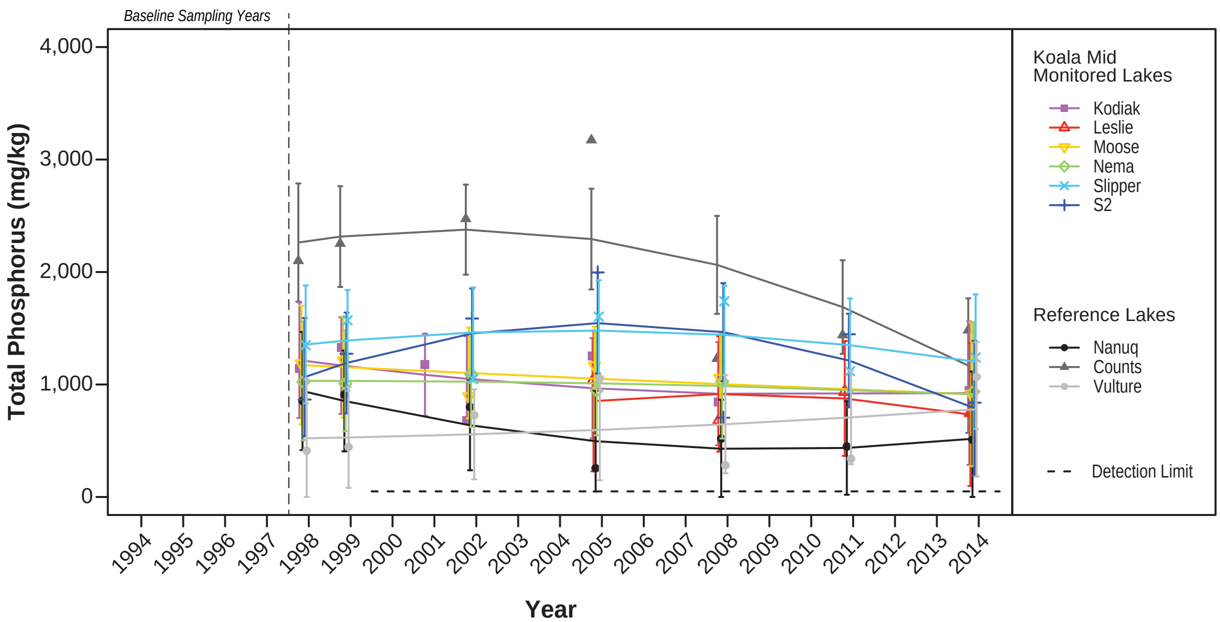
Observed and Fitted Means for Total Nickel Concentrations in Sediments in Koala Watershed Lakes and Lac de Gras, 1994 to 2014



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 3.3-9

Observed and Fitted Means for Total Phosphorus Concentrations in Sediments in Koala Watershed Lakes and Lac de Gras, 1994 to 2014



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

3.3.3.10 Selenium

Summary: Statistical and graphical analyses suggest that mine operations have had no effect on selenium concentrations at any of the monitored sediment sites in the Koala Watershed or Lac de Gras.

Statistical and graphical analyses indicate that selenium concentrations in sediments have remained stable through time, relative to reference lakes, at all monitored sites in the Koala Watershed and Lac de Gras (Table 3.3-11; Figure 3.3-10). Thus, no mine effects were detected.

Table 3.3-11. Statistical Results for Total Selenium Concentrations in Sediments in Koala Watershed Lakes 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	
Total Selenium	-	Tobit	1b	-	-	None-	1-415

Note: Dashes indicate not applicable.

3.3.3.11 Strontium

Summary: No statistical analyses were possible at this time; however, graphical analysis suggests that strontium concentrations are greater than those observed in reference lakes in sediments of all monitored sites downstream of the LLCF, as far as site S2 in Lac de Gras and in sediments of Kodiak Lake. The pattern of decreasing concentration with increasing distance from the LLCF suggests the possibility of a mine effect.

Strontium concentrations in sediments have only been analyzed for three years (i.e., 2008, 2011, and 2014). Thus, all lakes were excluded from the statistical analyses and no tests were performed (Table 3.3-12). Graphical analysis suggests that observed strontium concentrations in sediments were greater than those observed in reference lakes at all monitored sites downstream from the LLCF as far as site S2 in Lac de Gras, with concentrations decreasing with downstream distance from the LLCF (Figure 3.3-11). At sites that are not downstream of the LLCF, graphical analysis suggests that strontium concentrations were greater than observed in reference lakes (Figure 3.3-11). Strontium concentrations in sediments follow a similar pattern to that observed for total strontium concentrations in water quality samples (see Section 3.2.4.20); therefore, elevated strontium concentrations in sediments likely stem from strontium contained in LLCF discharge.

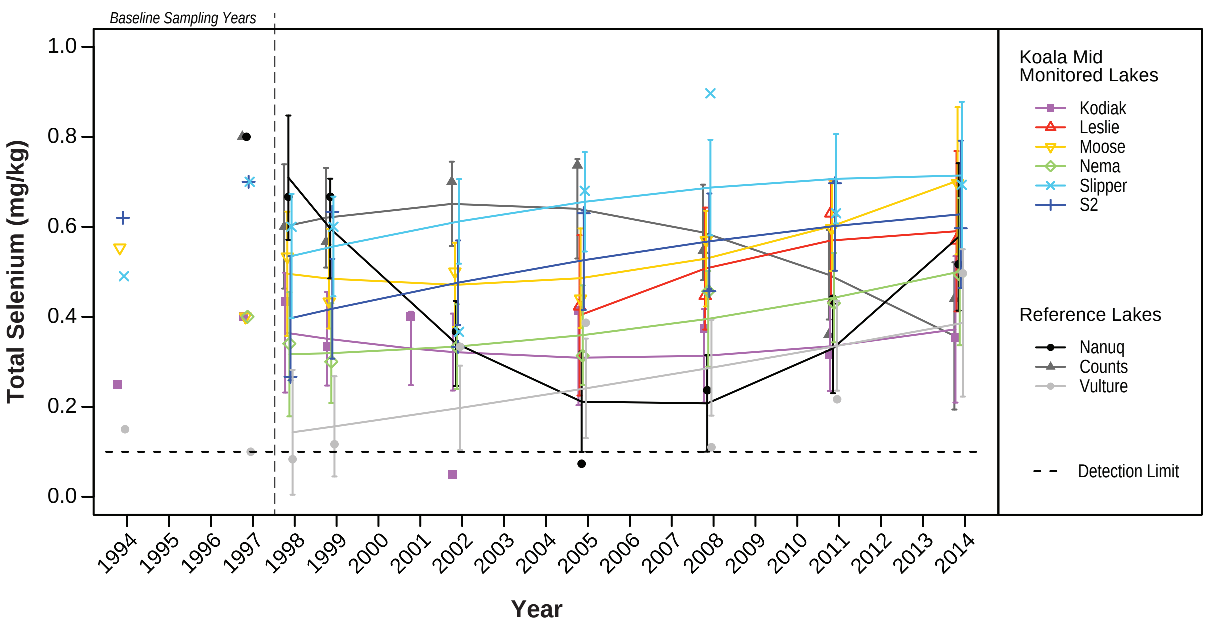
Table 3.3-12. Statistical Results for Total Strontium Concentrations in Sediments in Koala Watershed Lakes and Lac de Gras

	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	
Total Strontium	-	-		-	-	-	1-420

Note: Dashes indicate not applicable.

Figure 3.3-10

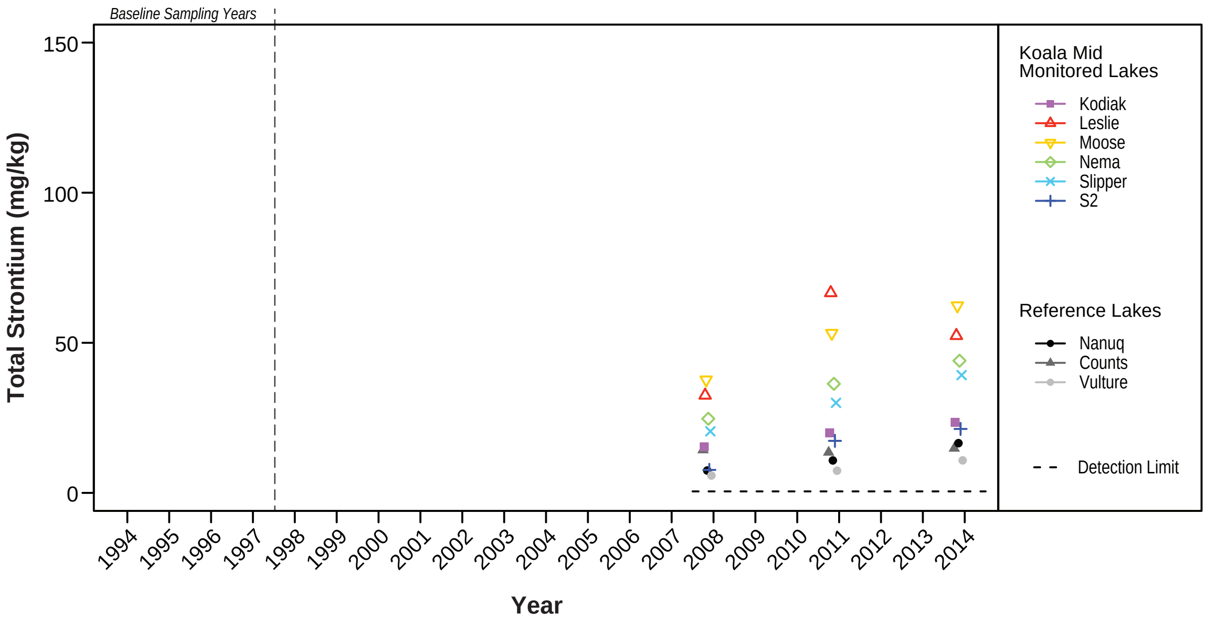
Observed and Fitted Means for Total Selenium Concentrations in Sediments in Koala Watershed Lakes and Lac de Gras, 1994 to 2014



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 3.3-11

Observed and Fitted Means for Total Strontium Concentrations in Sediments in Koala Watershed Lakes and Lac de Gras, 1994 to 2014



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

3.4 AQUATIC BIOLOGY

The extent to which changes in water and sediment 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). Benchmarks and CCME guidelines for the protection of aquatic life exist for some water and sediment quality variables (see Sections 2.3 and 2.4). These guidelines and benchmarks provide an important interpretive tool for evaluating the toxicological significance of water and sediment chemistry data. 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 19 evaluated water quality variables have increased downstream of the LLCF as a result of mine activities (see Section 3.2.4). A 20th variable (i.e., TOC) also showed evidence of an increase through time; however, no clear downstream spatial gradient was present suggesting that observed patterns may represent natural regimes. In general, the 95% confidence intervals around the fitted mean and the observed mean concentrations for these 20 water quality variables were below their respective CCME guideline value, SSWQO, or relevant benchmark value (see Section 3.2.4). Exceptions included pH, total-phosphate-P, total cadmium, 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 exceedances are not related to mine activities. For total cadmium, the concentration was greater than the hardness-dependent cadmium CCME guideline in only one sample from Moose-Nero Stream in June. Since concentrations of total cadmium have generally been below detection limits in all reference and monitored sites since monitoring began, this exceedance is unlikely related to mine activities. In contrast, potassium exceedances 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 2012c). With the exception of potassium, concentrations of all the water quality variables in the Koala Watershed and Lac de Gras in 2014 remained below the lowest identified chronic effect level for the most sensitive species (Rescan 2012f). The observed mean potassium concentrations in Leslie and Moose lakes during the ice-covered season, and the two samples collected from two meters above the sediment-water interface (i.e., deep water samples) in Leslie Lake during the open water season, exceeded the potassium SSWQO (41 mg/L; see Part 2 - Data Report; Rescan 2012f). In Leslie and Moose lakes, 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 phosphorus (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 2012c). As the trends in the evaluated water quality variables in 2014 are consistent with those observed in the 2011, 2012, and 2013 AEMP (Rescan 2012b, 2013b; ERM Rescan 2014a), 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 2014. 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 2012f).

Results from sediment quality analyses in the Koala Watershed and Lac de Gras also suggest that changes might be expected in biological communities downstream of the LLCF, because the concentration of one evaluated sediment quality variable (i.e., molybdenum) has increased as far as Slipper Lake and elevated concentrations of two other evaluated sediment quality variables (i.e., antimony and strontium) have been detected downstream of the LLCF (see Section 3.3.3). However, no CCME guidelines or relevant benchmark values currently exist for these three sediment quality variables, suggesting that no toxic effects are expected.

3.4.1 Phytoplankton

3.4.1.1 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) and community composition were evaluated to determine if mine activities have affected phytoplankton communities.

3.4.1.2 Dataset

Phytoplankton data have been collected between late July and early August of each year for the evaluation of effects (Table 3.4-1). Baseline data, which was collected from 1994 to 1997, are included in graphical analysis but not in the statistical evaluation of effects.

Table 3.4-1. 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
2014	Aug-5	Aug-9	Aug-3	Jul-29	Jul-31	Jul-31	Aug-2	Aug-4	Jul-30	Jul-30

Notes: 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 2014 for taxonomic analysis.
** Indicates data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison.*

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. Phytoplankton taxonomy samples were analyzed by Fraser Environmental Services in Surrey, BC from 1996 to 2012 and by EcoAnalysts in Moscow, Idaho, USA from 2013 to present. The methods differed slightly between the two taxonomists: The methods of Fraser Environmental Services included a "rare species scan", which is not part of the general protocol employed by EcoAnalysts. In 2013, EcoAnalysts was requested to complete a "rare species scan" in addition to their regular protocols in order to make the data comparable among years. The "rare species scan" encompassed a very small fraction of the total phytoplankton abundance and represents species that were not detected in the original subsample used for taxonomic identification, thus resulting in abundance measurements that are recoded as less than detection limit for that taxa. In 2014, the decision was made to discontinue the scan, which required the removal of rare species from the historical dataset. Although no general differences in trends were expected as a result of this change, small variations when comparing to previous AEMP reports may be present.

3.4.1.3 Results and Discussion

Chlorophyll *a*

Statistical and graphical analyses indicate that chlorophyll *a* concentrations have been stable through time in all monitored lakes (Table 3.4-2; Figure 3.4-1). Compared to mean baseline concentrations ± 2 SD, mean chlorophyll *a* concentrations in 2014 were greater in Moose, Nema, and Slipper lakes and sites S2 and S3 in Lac de Gras (Table 3.4-3). However, a similar pattern was observed in at least one reference lake (i.e., Vulture Lake; Table 3.4-3). Although concentrations in 2014 were elevated relative to baseline, there was no evidence of an increase in chlorophyll *a* concentration through time. Thus, no mine effects were detected with respect to chlorophyll *a*.

Table 3.4-2. 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	-	1-422

Note: Dashes indicate not applicable.

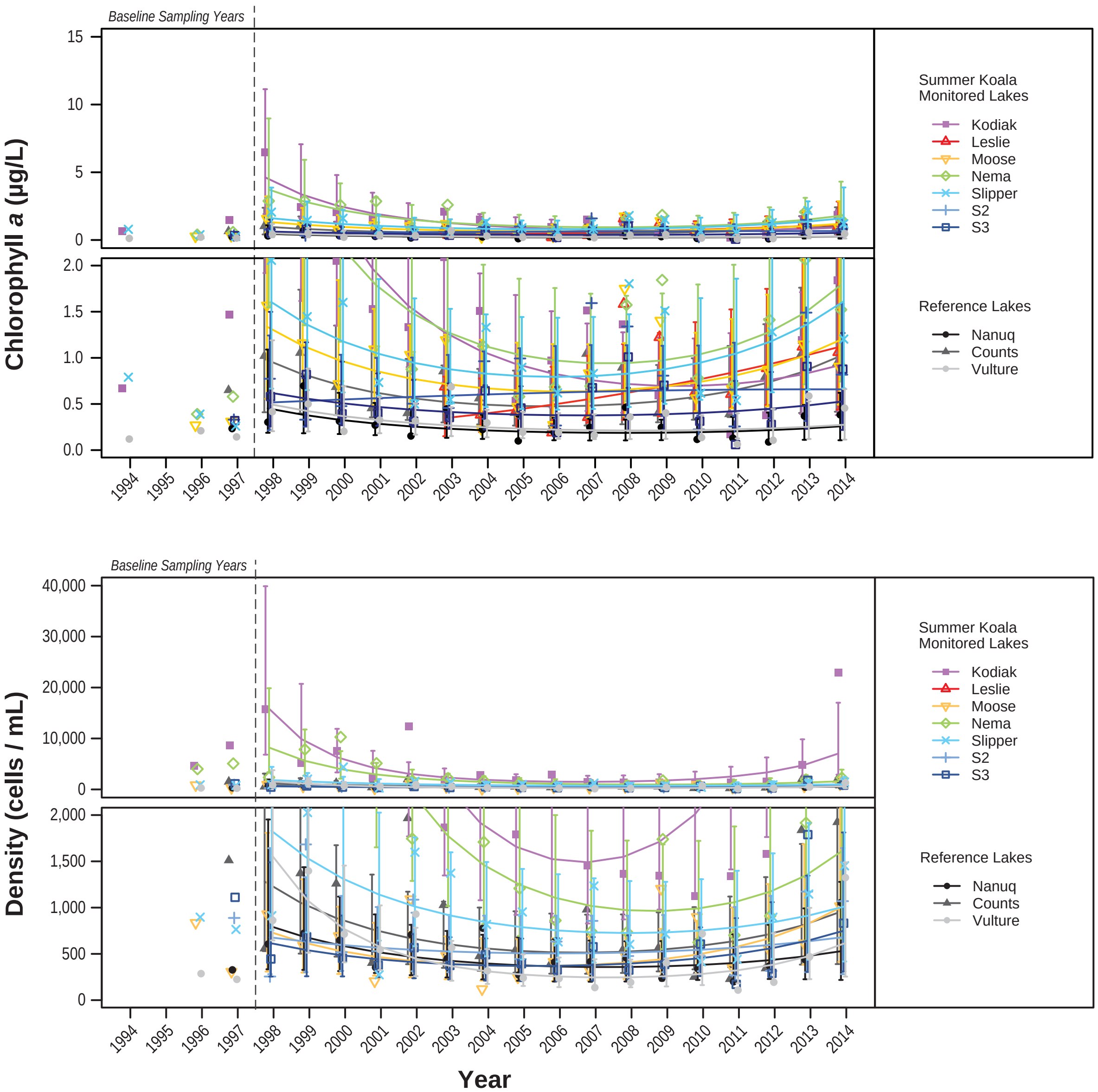
Table 3.4-3. Mean ± 2 Standard Deviations (SD) Baseline Concentrations of Chlorophyll *a* in Each of the Koala Watershed Lakes and Lac de Gras

Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD
Nanuq	0.23 (1)	0- 0.51	0.38 ± 0.37
Counts	0.65 (1)	0 – 1.45	0.93 ± 0.35
Vulture	0.15 (3)	0.08- 0.23	0.46 ± 0.24
Kodiak	1.24 (3)	0.46- 2.01	1.84 ± 0.59
Leslie	-	-	1.06 ± 0.10
Moose	0.30 (2)	0 - 0.74	0.91 ± 0.36
Nema	0.53 (2)	0.21 - 0.85	1.52 ± 1.32
Slipper	0.39 (3)	0 - 0.88	1.20 ± 1.50
S2	0.33 (1)	0.13 - 0.53	0.82 ± 0.64
S3	0.32 (1)	0.14 - 0.50	0.88 ± 0.04

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

Figure 3.4-1

Observed and Fitted Means for Chlorophyll *a* Concentrations and Phytoplankton Density in Koala Watershed Lakes and Lac de Gras, 1994 to 2014



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 phytoplankton densities have been stable through time, relative to trends observed in reference lakes, in all monitored lakes except in Leslie Lake (Table 3.4-4). Although no significant difference was found when comparing the trend in Leslie Lake to Reference Model 3, the test for Reference Model 2 revealed that the trend in Leslie Lake differed from the common slope of reference lakes (Table 3.4-4). This is likely because the result for Leslie Lake in the test for Reference Model 3 was marginal ($p = 0.07$; see Part 3 – Statistical Report) and the test for Reference Model 2 has greater power to detect differences owing to fewer parameters in the model (i.e., no intercept). In contrast, graphical analysis suggests that phytoplankton densities have been relatively stable through time in Leslie Lake (Figure 3.4-1).

Table 3.4-4. 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, Nema	Leslie	-	1-428

Note: Dashes indicate not applicable.

Compared to mean baseline densities ± 2 SD, mean phytoplankton density in 2014 was greater in Kodiak Lake (Table 3.4-5). However, a similar pattern was observed in two reference lakes (i.e., Nanuq and Vulture lakes; Table 3.4-5). Thus, no mine effects were detected with respect to phytoplankton density.

Table 3.4-5. 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	2014 Mean ± 1 SD
Nanuq	327 (1)	51 - 603	1,055 ± 346
Counts	1,511 (1)	109 - 2,913	1,924 ± 755
Vulture	256 (2)	53 - 459	1,324 ± 398
Kodiak	6,634 (3)	1,949 - 11,319	22,912 ± 7,270
Leslie	-	-	1,212 ± 117
Moose	571 (2)	0 - 1,469	1,016 ± 258
Nema	4,537 (2)	432 - 8,643	2,285 ± 770
Slipper	830 (2)	100 - 1,560	1,452 ± 499
S2	888 (1)	289 - 1,488	1,070 ± 64
S3	1,111 (1)	0 - 2,293	832 ± 143

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

Diversity and Community Composition

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 3.4-2) 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 3.4-3 to 3.4-8). Note that following recent advances in taxonomic classification, the names of two phytoplankton groups were 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 3.4-2). While the variability makes it somewhat difficult to discern temporal trends, diversity in Leslie Lake decreased between 2006 and 2011; however, diversity in Leslie Lake has increased in recent years and has stabilised at values similar to those observed historically in Leslie Lake and to those currently observed in reference lakes (Figure 3.4-2b). Diversity in Kodiak Lake has generally been stable through time, but has decreased in recent years and was lower than values observed in all three reference lakes (Figure 3.4-2a). Diversity at site S2 in Lac de Gras has also been generally stable though time, but was lower in 2014 when compared to baseline years (Figure 3.4-2c). In all other lakes, diversity appears to have remained relatively stable over time.

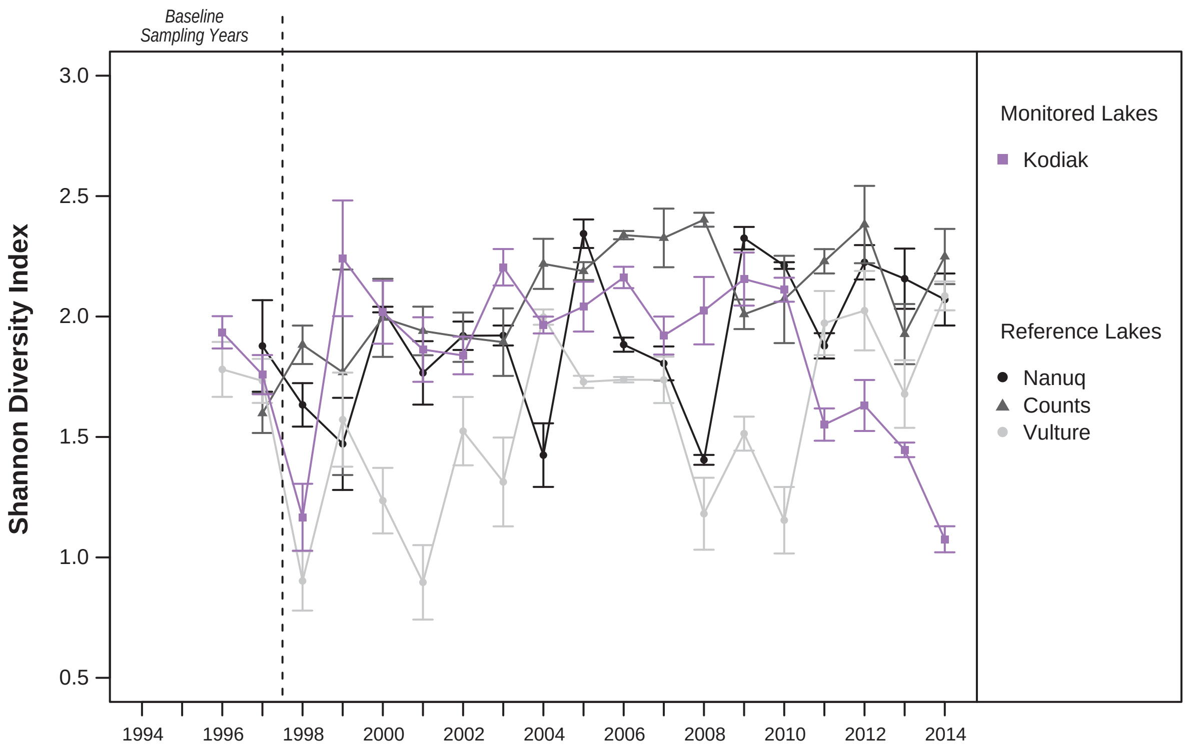
Comparisons between mean diversity ± 2 SD in baseline years and mean diversity in 2014 indicate differences between baseline and 2014 values for both Shannon and Simpson's diversity in Kodiak Lake and at sites S2 and S3 in Lac de Gras (Table 3.4-6). At site S3, mean diversity indices in 2014 were greater than those observed in baseline years; however, a similar pattern was observed in one reference lake (i.e., Counts Lake; Table 3.4-6). In Kodiak Lake and at site S2 in Lac de Gras, mean diversity indices in 2014 were lower than those observed during baseline years (Table 3.4-6). In addition, mean Shannon diversity in 2014 in Nema Lake was greater than the mean Shannon diversity ± 2 SD in baseline years, but a similar pattern was observed in one reference lake (i.e., Vulture Lake; Table 3.4-6). In all other lakes, diversity indices remained within two SD of baseline values in 2014 and there was no decreasing trend in diversity with downstream distance from the mine (Table 3.4-6).

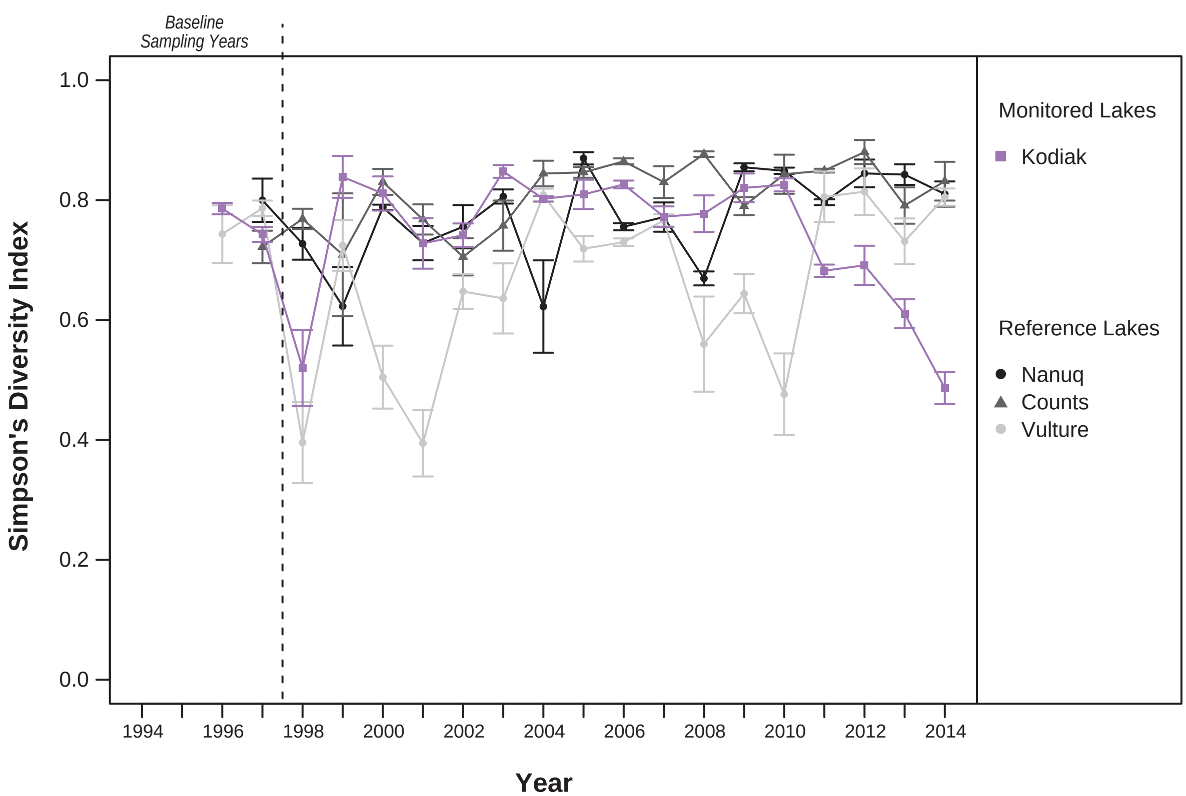
Together, the evidence suggests that phytoplankton diversity has been stable through time in all monitored lakes of the Koala Watershed and Lac de Gras, except Leslie and Kodiak lakes. Although diversity was lower at site S2 in Lac de Gras in 2014, there is currently no evidence of a decreasing trend through time. Phytoplankton diversity in Leslie Lake 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 3.4-6 to 3.4-8). 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 2014 and increases in Crytophyceae in Moose Lake from 2001 to 2004; Figure 3.4-7a). 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 3.4-4 to 3.4-5).

Figure 3.4-2a

Average Diversity Indices for Phytoplankton in Koala Watershed Lakes and Lac de Gras, 1996 to 2014

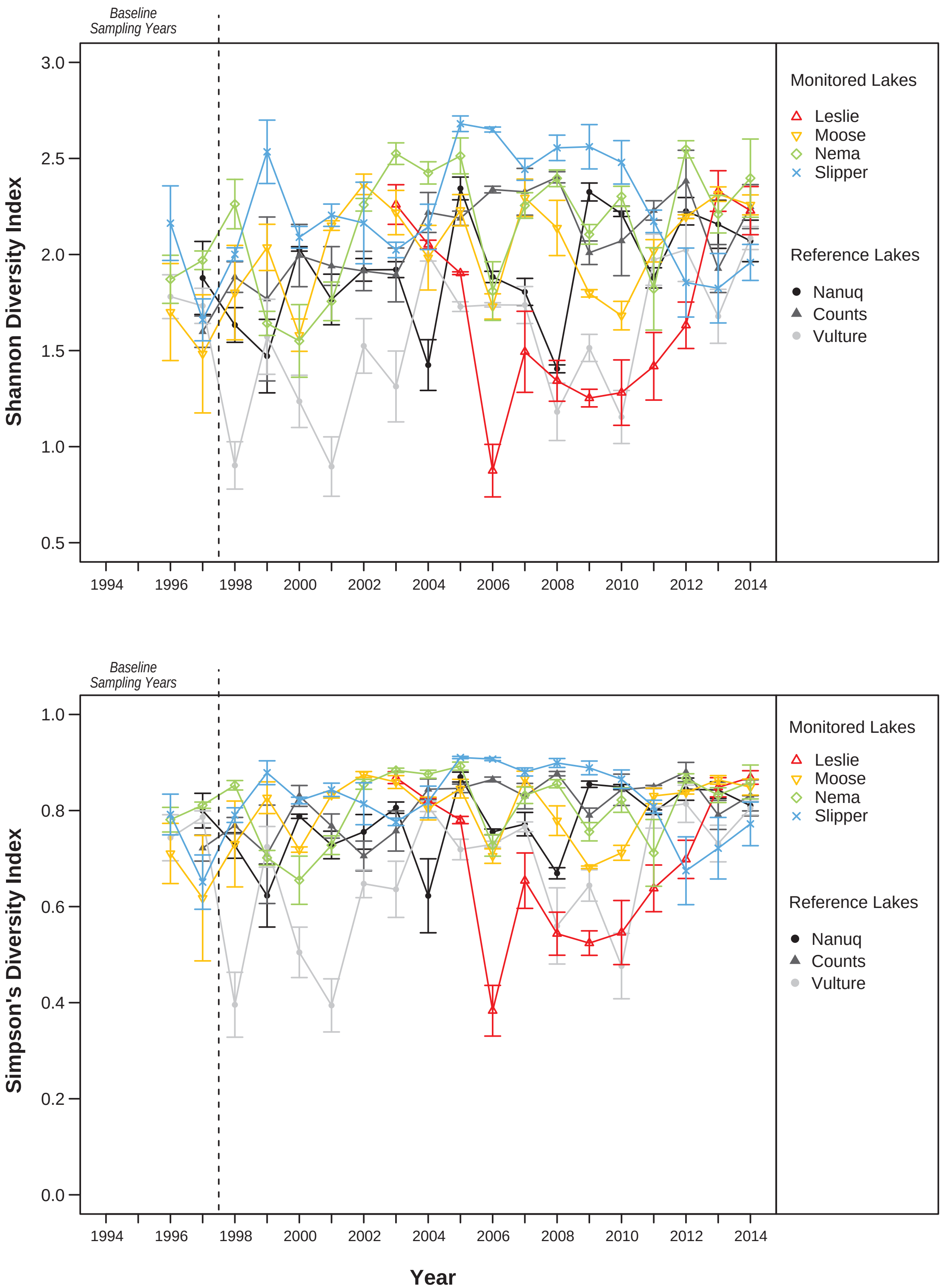




Notes: Symbols represent observed mean values.
Error bars indicate standard error of the observed means.

Figure 3.4-2b

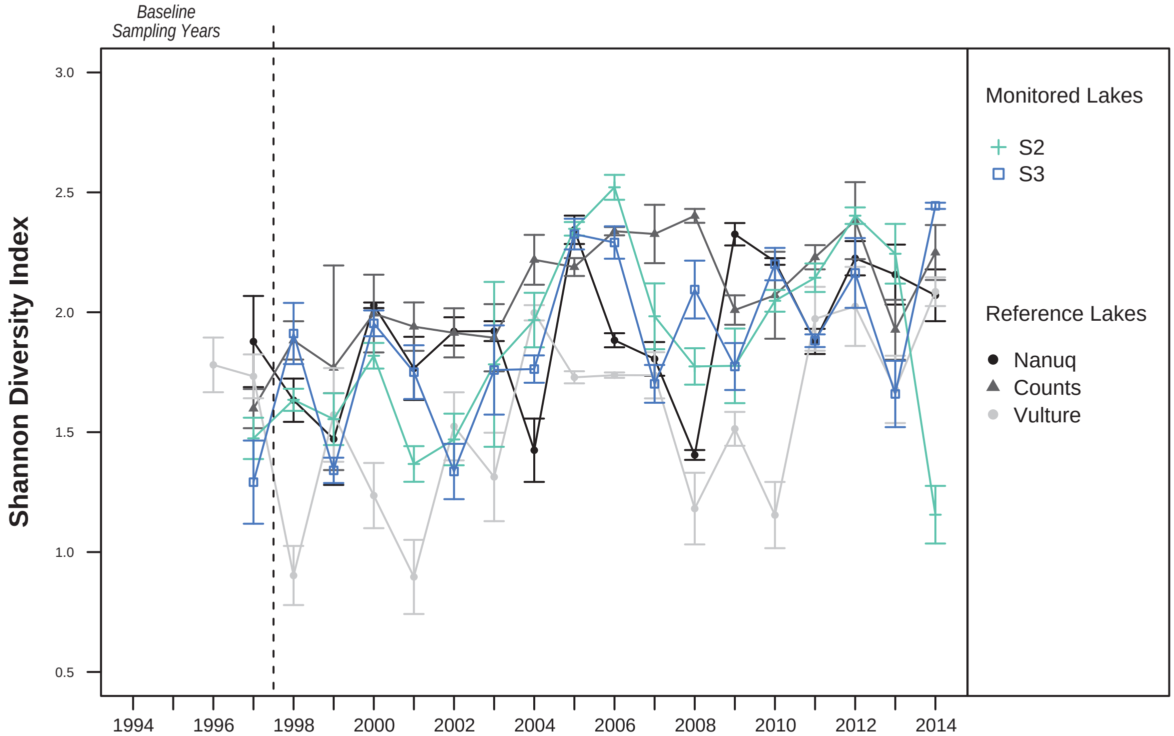
Average Diversity Indices for Phytoplankton in Koala Watershed Lakes and Lac de Gras, 1996 to 2014

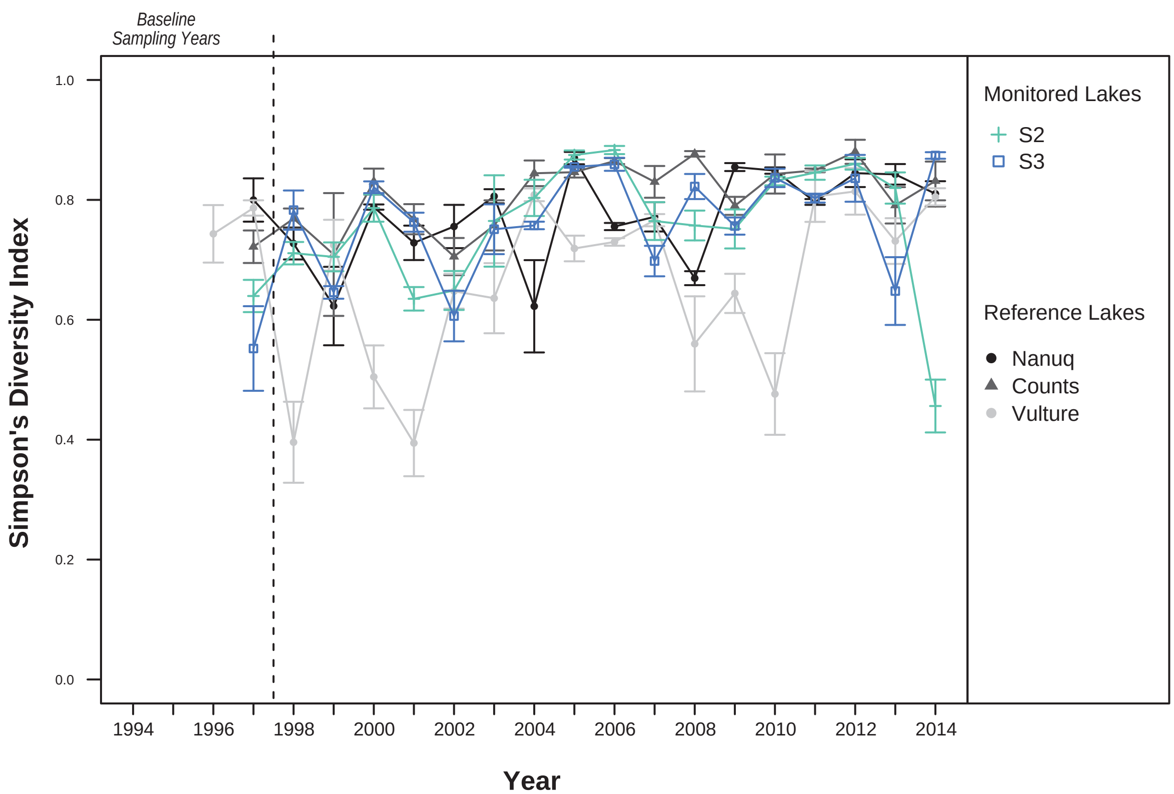


Notes: Symbols represent observed mean values.
Error bars indicate standard error of the observed means.

Figure 3.4-2c

Average Diversity Indices for Phytoplankton in Koala Watershed Lakes and Lac de Gras, 1996 to 2014





Notes: Symbols represent observed mean values.
Error bars indicate standard error of the observed means.

Figure 3.4-3

Average Phytoplankton Density by Taxonomic Group for AEMP Reference Lakes, 1996 to 2014

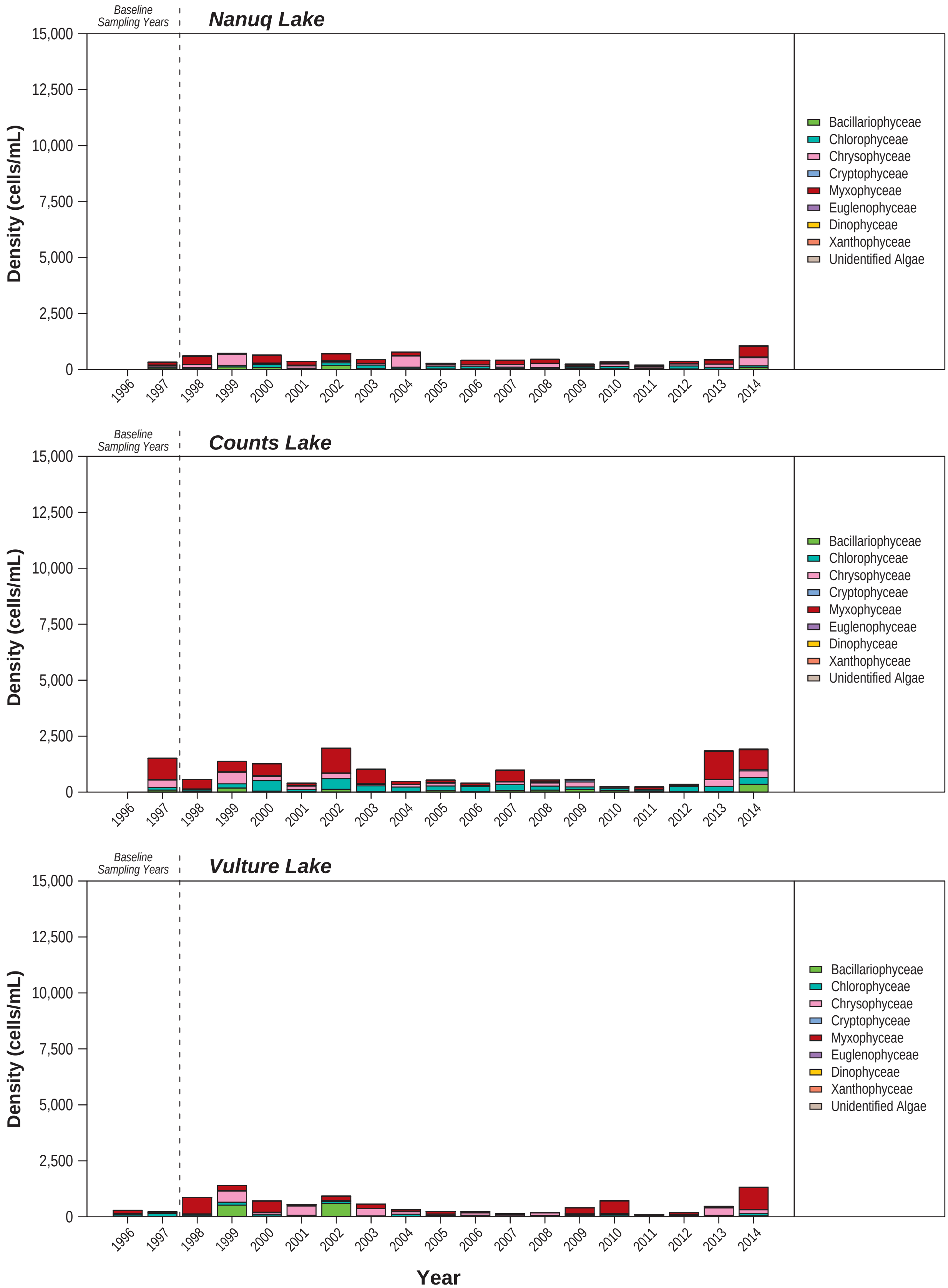
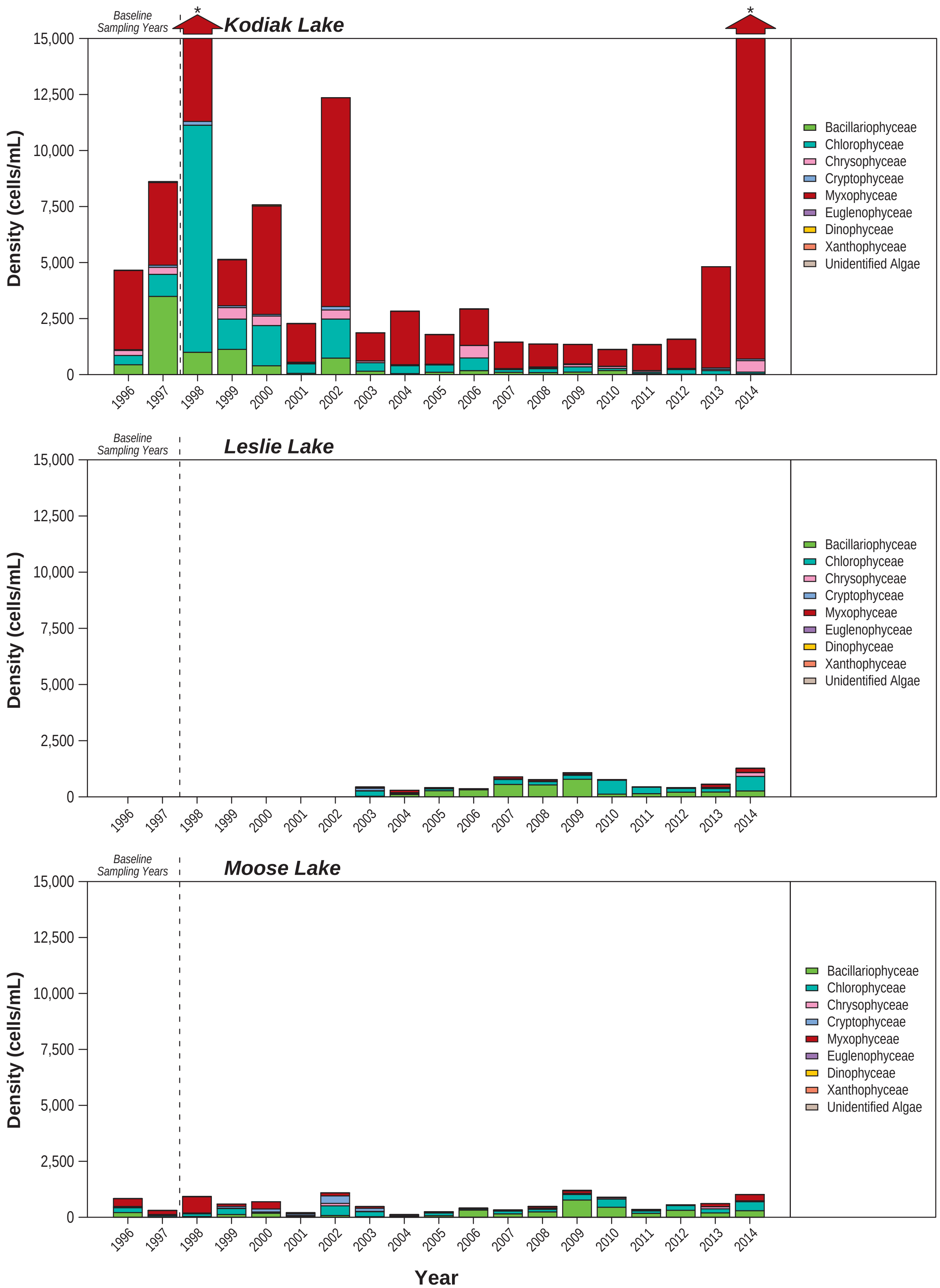


Figure 3.4-4a

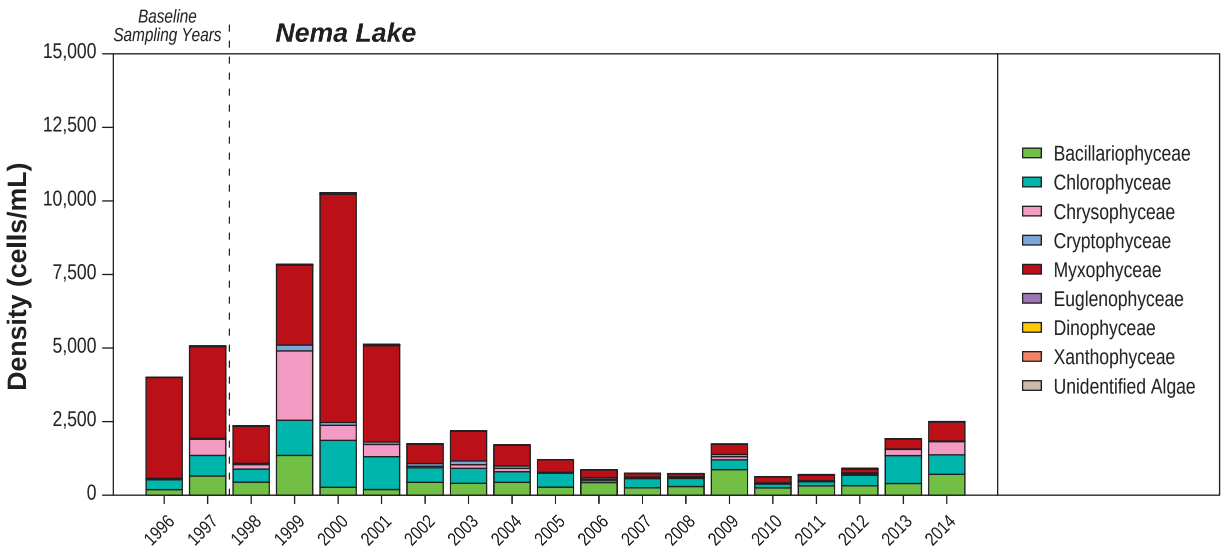
Average Phytoplankton Density by Taxonomic Group for Lakes of the Koala Watershed, 1996 to 2014



*Note: *Total density in 1998 = 15,705: Myxophyceae = 4,409; Total density in 2014 = 22,912: Myxophyceae = 22,197, Euglenophyceae = 4, Dinophyceae = 15.*

Figure 3.4-4b

Average Phytoplankton Density by Taxonomic Group for Lakes of the Koala Watershed, 1996 to 2014



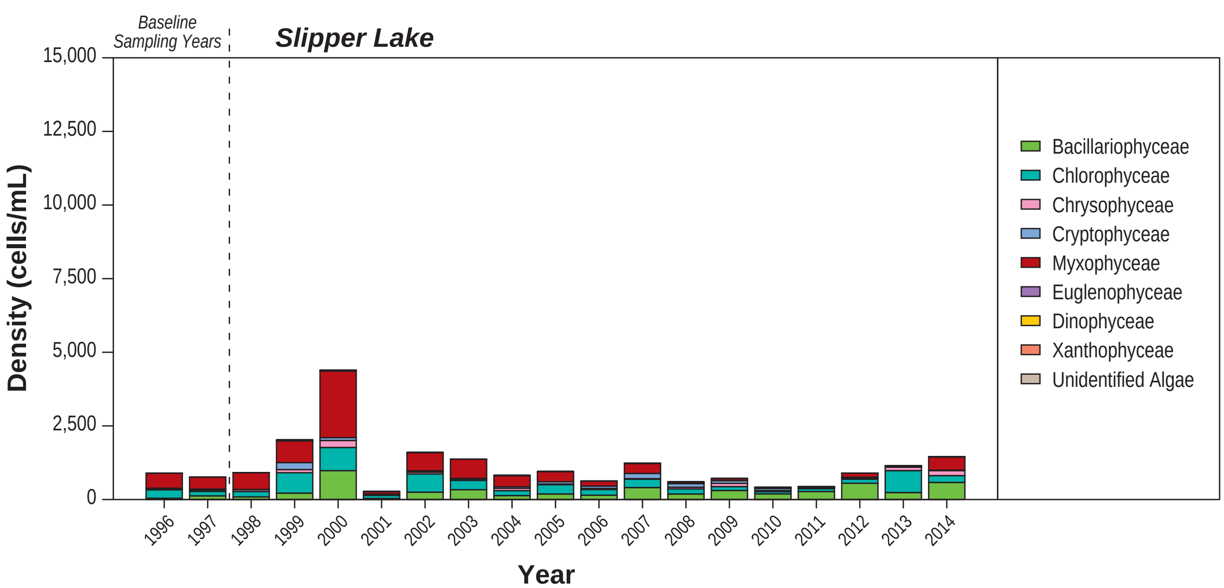
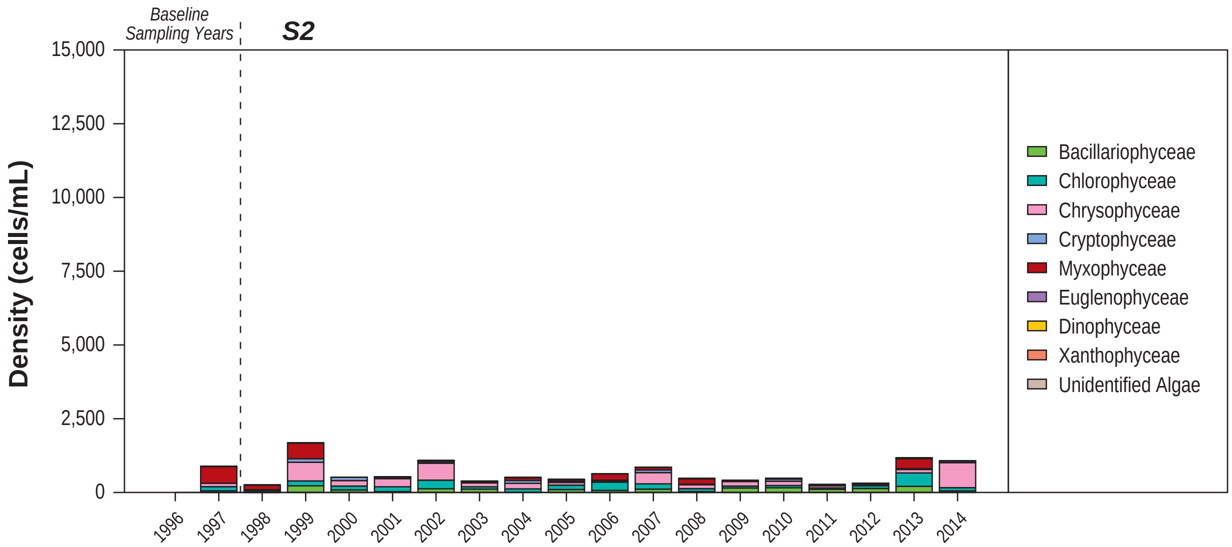


Figure 3.4-5

Average Phytoplankton Density by Taxonomic Group for Lac de Gras, 1996 to 2014



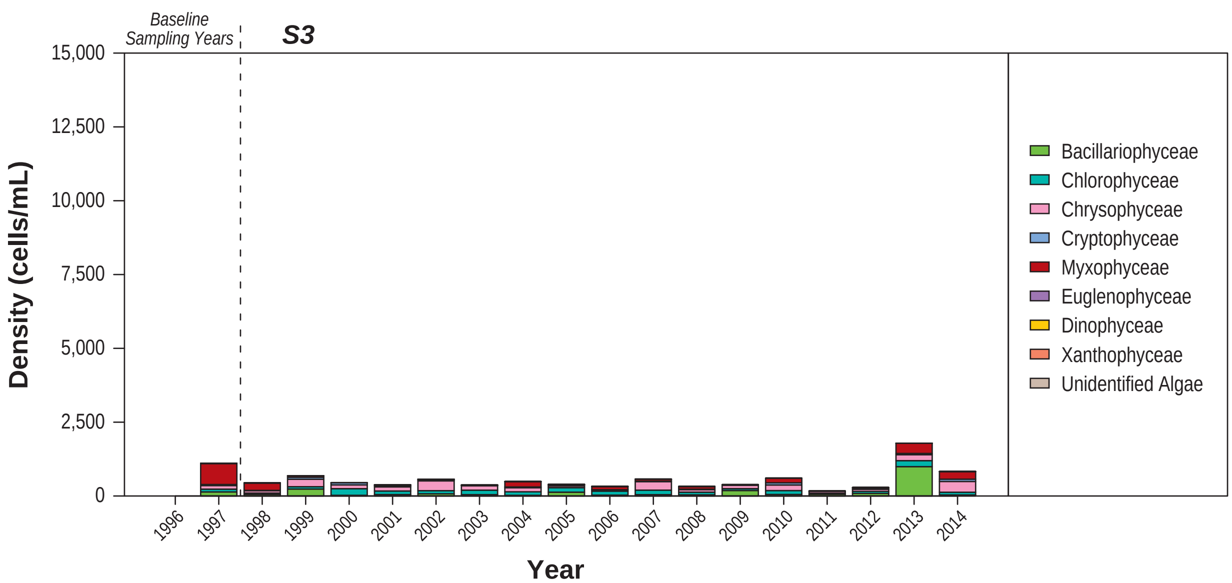


Figure 3.4-6

Relative Densities of Phytoplankton Taxa in AEMP Reference Lakes, 1996 to 2014

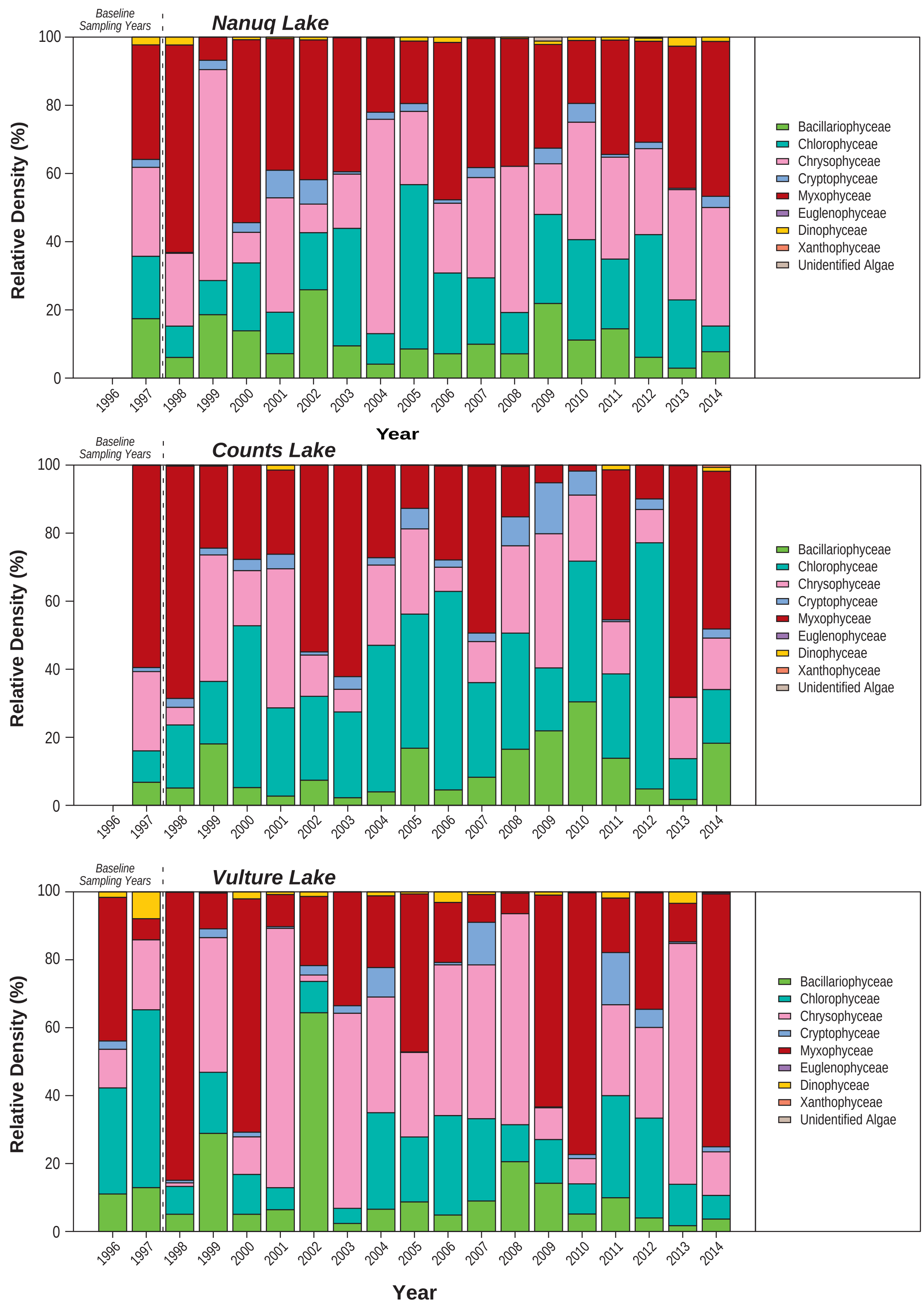


Figure 3.4-7a

Relative Densities of Phytoplankton Taxa in Lakes of the Koala Watershed, 1996 to 2014

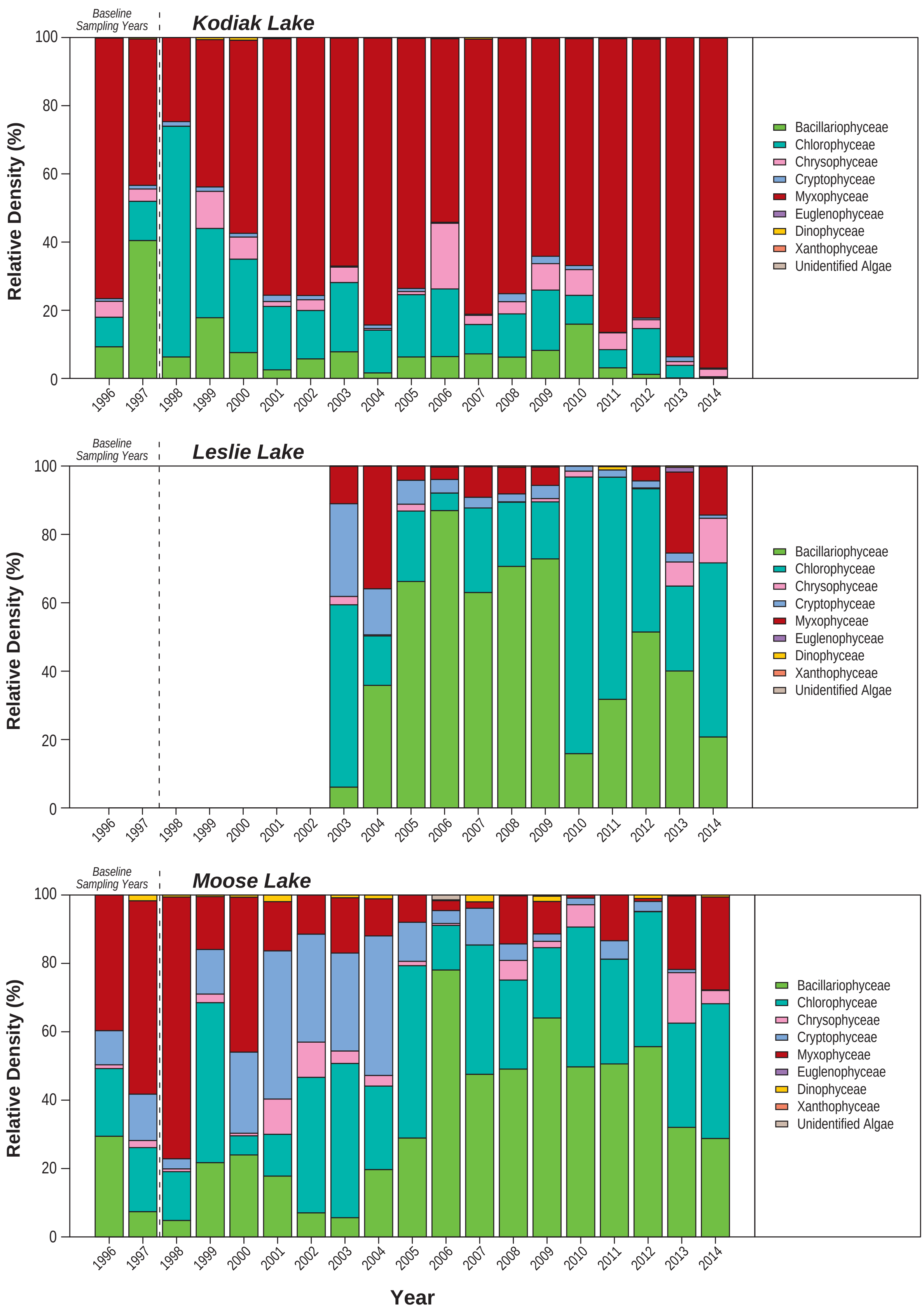


Figure 3.4-7b

Relative Densities of Phytoplankton Taxa in Lakes of the Koala Watershed, 1996 to 2014

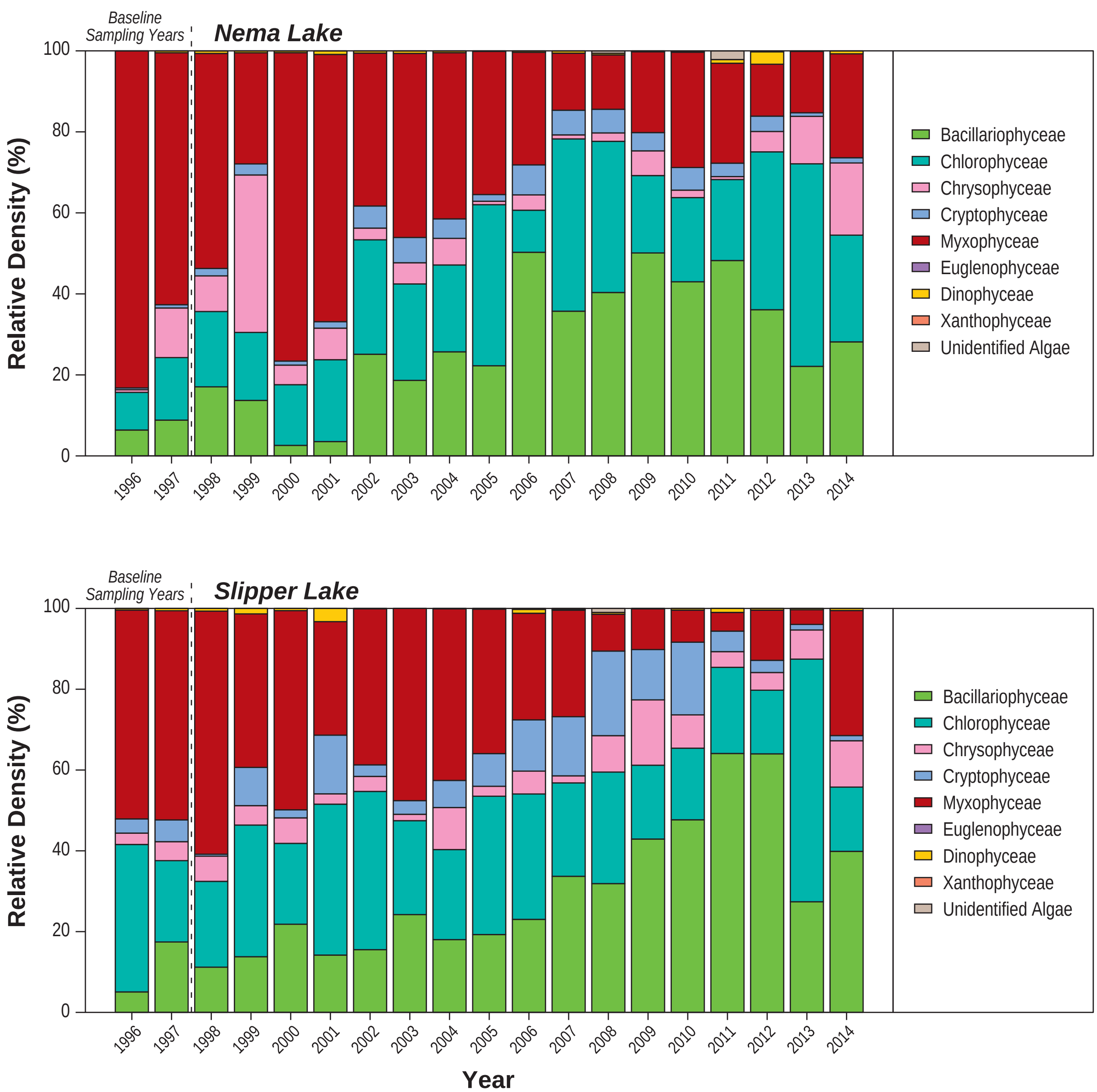
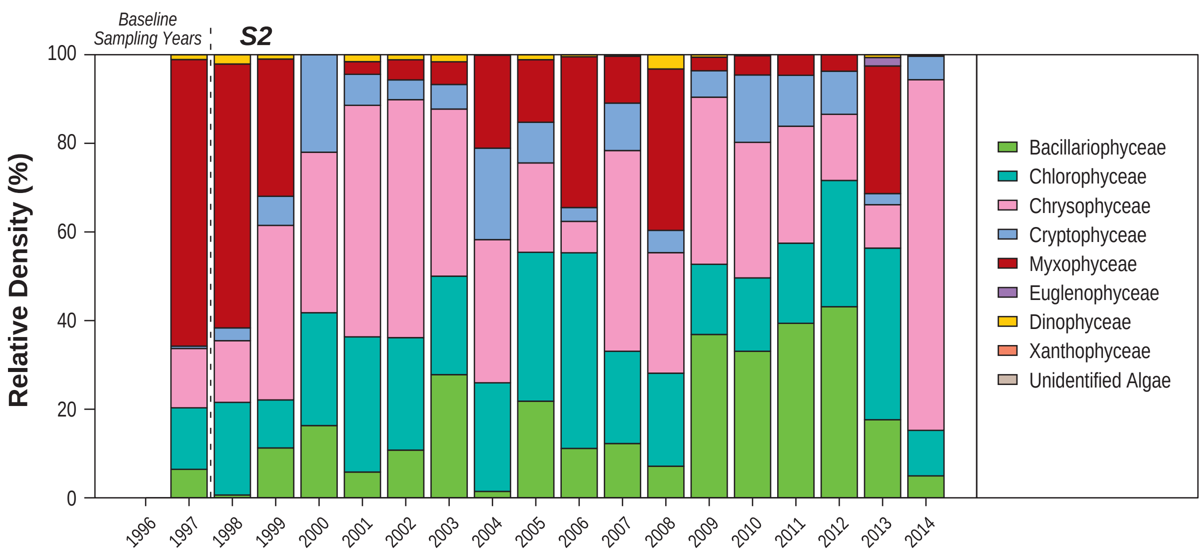


Figure 3.4-8

Relative Densities of Phytoplankton Taxa in Lac de Gras, 1996 to 2014



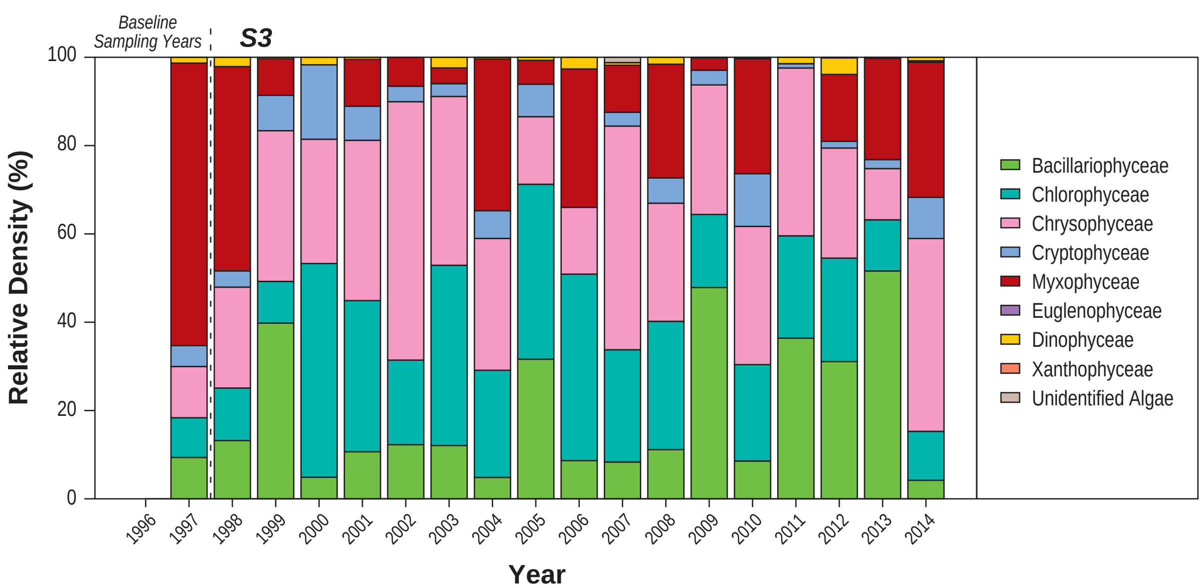


Table 3.4-6. Mean ± 2 Standard Deviations (SD) Baseline Phytoplankton Diversity in each of the Koala Watershed Lakes and Lac de Gras

	Shannon Diversity			Simpson's Diversity		
Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD
Nanuq	1.88 (1)	1.22 – 2.54	2.07 ± 0.19	0.80 (1)	0.67 – 0.92	0.81 ± 0.04
Counts	1.60 (1)	1.32 - 1.88	2.25 ± 0.20	0.72 (1)	0.63 – 0.82	0.83 ± 0.06
Vulture	1.76 (2)	1.43 – 2.08	2.09 ± 0.10	0.76 (2)	0.64 – 0.88	0.81 ± 0.02
Kodiak	1.85 (2)	1.55 – 2.15	1.08 ± 0.09	0.76 (2)	0.71 – 0.82	0.49 ± 0.05
Leslie	-	-	2.23 ± 0.22	-	-	0.87 ± 0.02
Moose	1.59 (2)	0.69 – 2.50	2.26 ± 0.09	0.66 (2)	0.33 – 1.0	0.85 ± 0.03
Nema	1.92 (2)	1.61 – 2.23	2.40 ± 0.35	0.80 (2)	0.73 – 0.86	0.86 ± 0.06
Slipper	1.91 (2)	1.18 – 2.65	1.96 ± 0.16	0.72 (2)	0.50 – 0.94	0.77 ± 0.08
S2	1.47 (1)	1.18 – 1.77	1.16 ± 0.21	0.64 (1)	0.55 – 0.73	0.46 ± 0.08
S3	1.29 (1)	0.70 – 1.89	2.44 ± 0.02	0.55 (1)	0.31 – 0.80	0.87 ± 0.01

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

The changes in phytoplankton community composition observed downstream of the LLCF 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 2010 to 2012 (Figure 3.4-4a, and 3.4-7a). In 2013 and 2014, 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.

Increases in the density of Myxophyceae also corresponded to a decrease in the relative density of Bacillariophyceae in Leslie Lake in 2013 and 2014. Similar patterns were observed in Moose Lake (Figures 3.4-4a and 3.4-7a). These patterns in community composition are more comparable to community composition in baseline years than those observed more recently. Whether these shifts indicate the onset of recovery in phytoplankton communities or represent an anomaly in recent trends is unclear at this time. In 2013, another potentially important shift was observed in Nema and Slipper lakes: The absolute density of Chlorophyceae increased, corresponding to a decrease in the relative density of Bacillariophyceae, resembling patterns observed in Leslie Lake from 2010 to 2012 (Figure 3.4-4 and 3.4-7). Although Chlorophyceae densities in Leslie Lake remain elevated in 2014, densities in Nema and Slipper lakes have decreased (Figures 3.4-4 and 3.4-7).

Although the changes in phytoplankton community composition observed downstream of the LLCF have only appeared to affect diversity indices in Leslie Lake, the diversity at site S2 in Lac de Gras decreased in 2014 and was lower than that observed during baseline years. The absolute and relative densities of Chrysophyceae at site S2 increased in 2014, with a corresponding decrease in Bacillariophyceae and Chlorophyceae (Figures 3.4-5 and 3.4-8). The low diversity observed at site S2 in 2014 likely reflects the uneven distribution across species as a result of increased density of Chrysophyceae (Figure 3.4-2c).

At sites that are not downstream of the LLCF, graphical analyses of taxonomic composition suggest that the relative density of phytoplankton groups in Kodiak Lake has changed in recent years. The absolute and relative densities of Myxophyceae in Kodiak Lake increased in 2013 and 2014, with a corresponding decrease in Bacillariophyceae and Chlorophyceae (Figures 3.4-4a and 3.4-7a). The low diversity observed in Kodiak Lake in 2014 likely reflects the uneven distribution across species as a result of increased density of Myxophyceae (Figure 3.4-2a).

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 (Figures 3.4-3 and 3.4-6). 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, and possibly in Kodiak Lake. Hypotheses regarding potential underlying causes of these changes are summarized in the Aquatic Biology Summary below (Section 3.4.5).

Overall, the main change 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 diatom genus *Cyclotella* or the golden algae genus *Ochromonas* (see Table 3.5-2 in Part 2 – Data Report). The subsequent shift from diatoms to chlorophytes in Leslie Lake observed from 2010 to 2014, may also affect higher trophic levels. Chlorophytes are usually rare in sub-Arctic freshwater systems in the Northwest Territories (Moore 1978). Of the chlorophytes, the edible *Tetrastrum komarekii* predominates in Leslie Lake (see Table 3.5-2 in Part 2 – Data Report).

3.4.2 Zooplankton

3.4.2.1 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) and community composition were monitored to detect potential mine effects.

3.4.2.2 *Dataset*

Zooplankton data have been collected between late July and early August each year from 1994 to 2014 (Table 3.4-7). 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 3.4-7 and shown graphically in Figures 3.4-9 to 3.4-16, but are not included in the statistical evaluation of effects.

Table 3.4-7. 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
2014	Aug-5	Aug-9	Aug-8	Jul-29	Jul-31	Jul-31	Aug-2	Aug-4	Jul-30	Jul-30

Notes: Dashes indicate no data were available.
Triplicate samples were collected yearly for biomass analysis from 1998 to 2014.
Triplicate samples were collected yearly for taxonomic analysis from 1995 to 2014.
**Indicates data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison.*

3.4.2.3 Results and Discussion

Biomass

Statistical and graphical analyses suggest that zooplankton biomass has remained relatively stable through time in all monitored and reference lakes (Table 3.4-8; Figure 3.4-9). It was not possible to compare mean zooplankton biomass in 2014 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 3.4-8. 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	-	1-434

Note: Dashes indicate not applicable.

Density

Statistical analyses indicate that zooplankton densities have been stable through time in all monitored and reference lakes (Table 3.4-9). Graphical analysis also suggests that zooplankton densities have been relatively stable in all monitored and reference lakes since monitoring began, though observed zooplankton densities were elevated, relative to baseline years, in Kodiak Lake in 2011 and 2012 (Figure 3.4-9).

Table 3.4-9. Statistical Results of Zooplankton Density in Lakes in the Koala Watershed and Lac de Gras

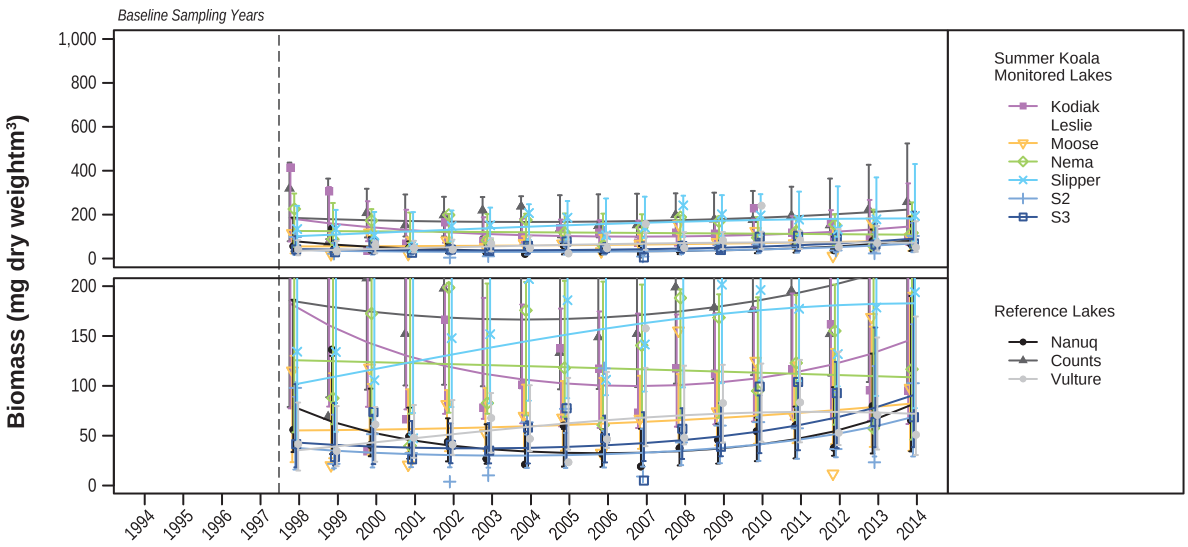
	Lakes/Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
				Model Fit = 3	Model Fit = 2	Model Fit = 1	
Zooplankton density	-	LME	3	None	None	-	1-440

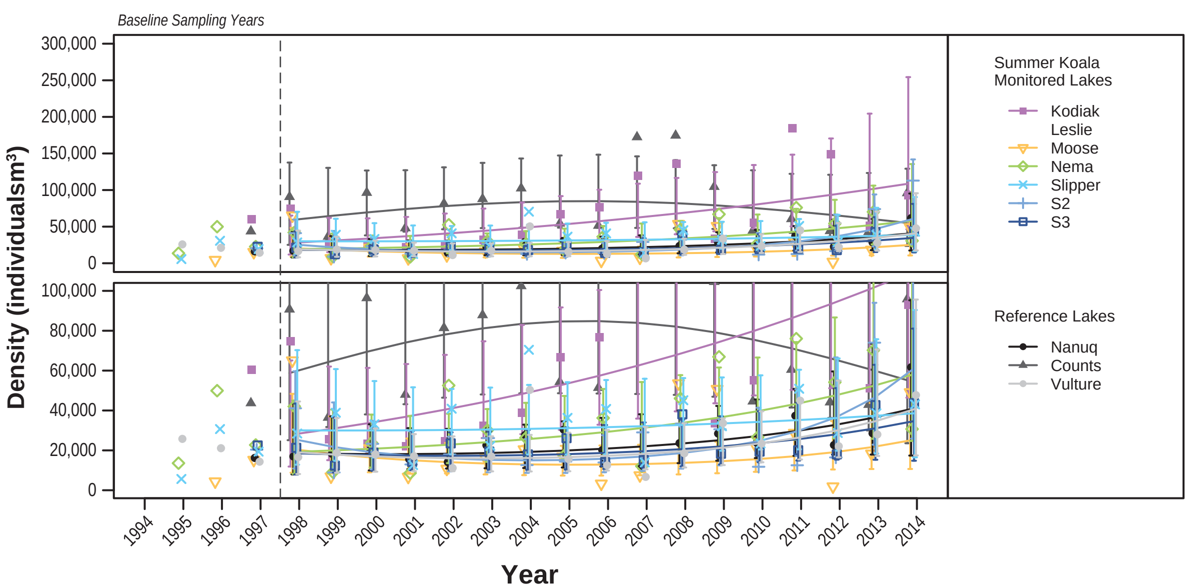
Note: Dashes indicate not applicable.

Compared to mean baseline densities ± 2 SD, mean zooplankton densities in 2014 were greater in Moose and Slipper lakes and at sites S2 and S3 in Lac de Gras (Table 3.4-10). However, a similar pattern was observed in all three reference lakes (Table 3.4-10). Thus, no mine effects were detected with respect to zooplankton density.

Figure 3.4-9

Observed and Fitted Means for Zooplankton Biomass and Density in Koala Watershed Lakes and Lac de Gras, 1994 to 2014





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 3.4-10. 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	2014 Mean ± 1 SD
Nanuq	16,209 (1)	13,053 - 19,365	61,696 ± 8,959
Counts	43,710 (1)	33,027 - 54,392	95,872 ± 2,666
Vulture	20,384 (3)	9,704 - 31,064	47,709 ± 8,213
Kodiak	113,472 (3)	0 - 286,939	93,123 ± 17,438
Leslie	-	-	47,918 ± 8,511
Moose	9,782 (2)	0 - 21,900	49,110 ± 10,781
Nema	28,744 (3)	0 - 65,016	30,683 ± 1,232
Slipper	18,562 (3)	0 - 41,464	43,051 ± 8,759
S2	20,360 (1)	15,280 - 25,441	112,912 ± 36,687
S3	22,451 (1)	14,665 - 30,238	42,680 ± 11,349

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

Diversity and Community Composition

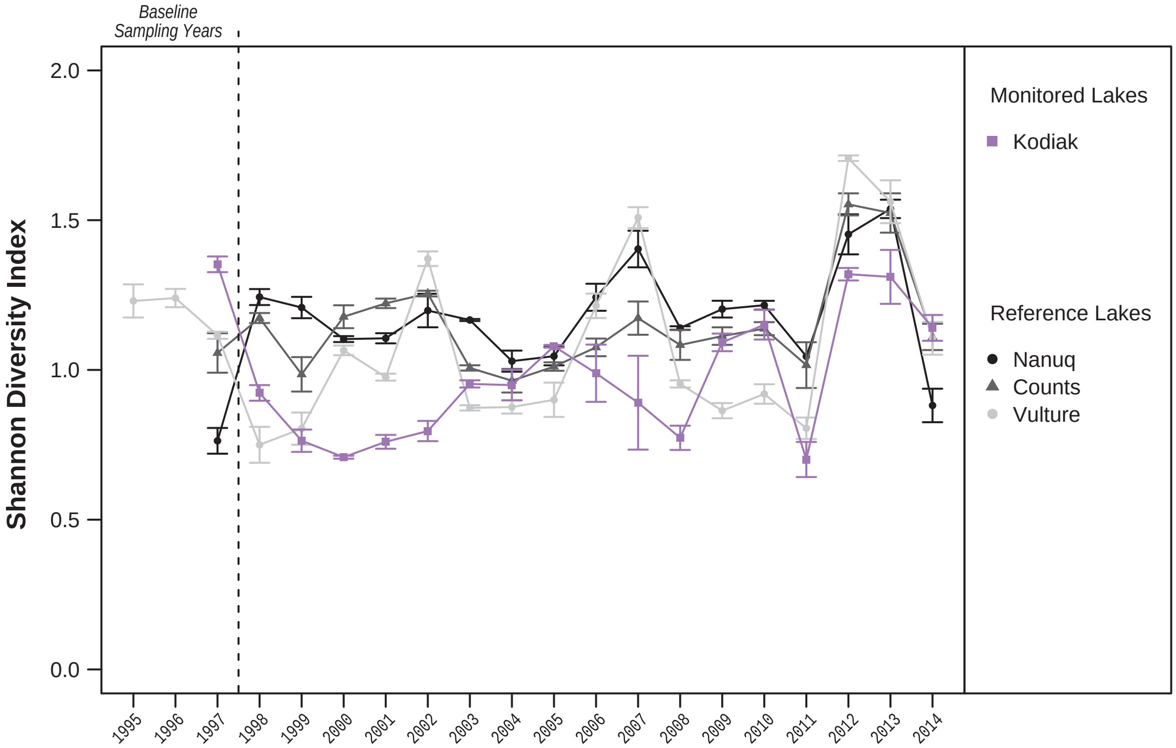
Statistical analyses were not performed on the diversity datasets because the calculation of indices can result in data abnormalities that prevent statistical analysis. Consequently, graphical analyses of temporal trends in diversity indices (Figure 3.4-10) and best professional judgment were the primary 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 3.4-11 to 3.4-16).

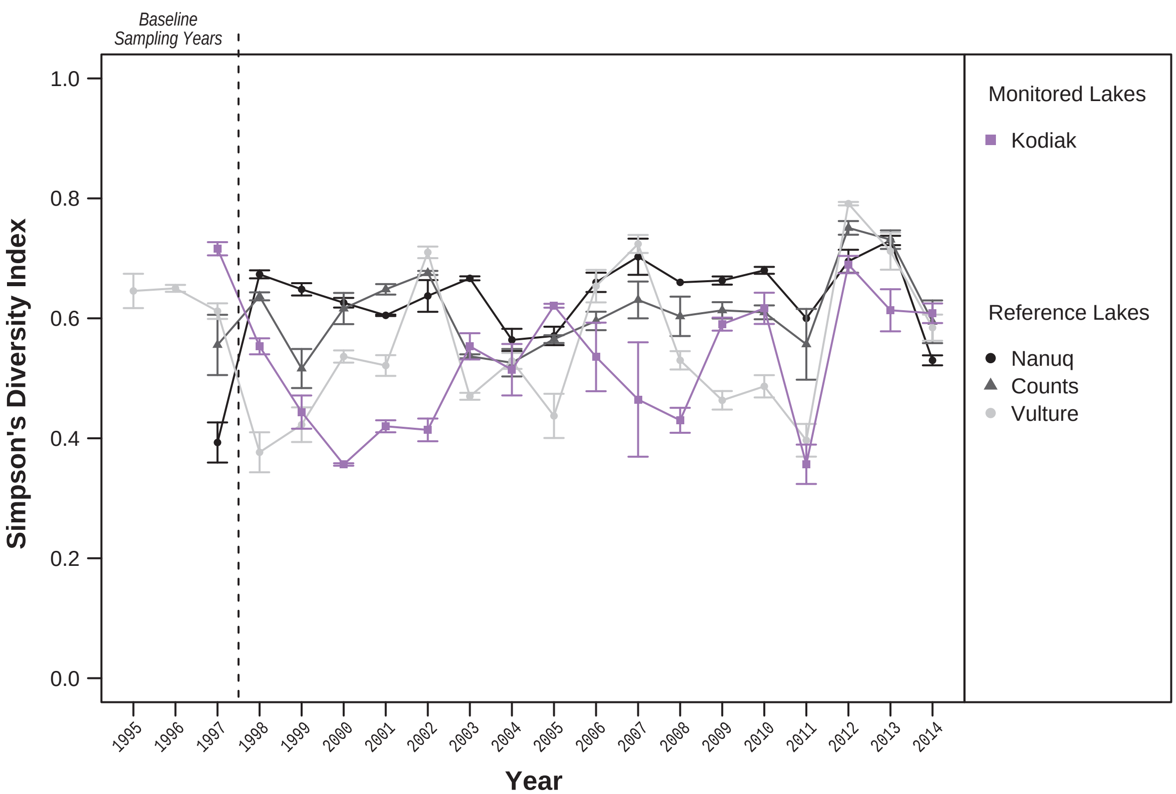
Both Shannon and Simpson's diversity indices have varied considerably through time in both monitored and reference lakes (Figure 3.4-10). While the variability makes it somewhat difficult to discern temporal trends, both Shannon and Simpson's diversity indices have generally declined through time in Leslie and Moose lakes since monitoring began (Figure 3.4-10b). However, in both cases, diversity has increased in recent years and has stabilised at values equal to or greater than those observed in reference lakes or during baseline years. Although diversity in Nema Lake decreased in 2012 and 2013, it increased in 2014 (Figure 3.4-10b). At sites S2 and S3 in Lac de Gras, diversity decreased in 2013 and 2014 to values less than those observed during baseline years; however, a general decline was observed in all three reference lakes in 2014. In addition, diversity had fluctuated down to these values in site S2 and S3 in previous years (i.e., 2007 and 2008).

Compared to mean diversity ± 2 SD in baseline years, mean Shannon and Simpson's diversity indices were lower at sites S2 and S3 in Lac de Gras in 2014 and mean Simpson's diversity was lower in Kodiak Lake (Table 3.4-11). A similar pattern of lower Simpson's diversity was observed in one reference lake (i.e., Nanuq Lake; Tale 3.4-11). Zooplankton diversity in 2014 was greater than in baseline years in Nema Lake (Table 3.4-11).

Figure 3.4-10a

Average Diversity Indices for Zooplankton in Koala Watershed Lakes and Lac de Gras, 1995 to 2014

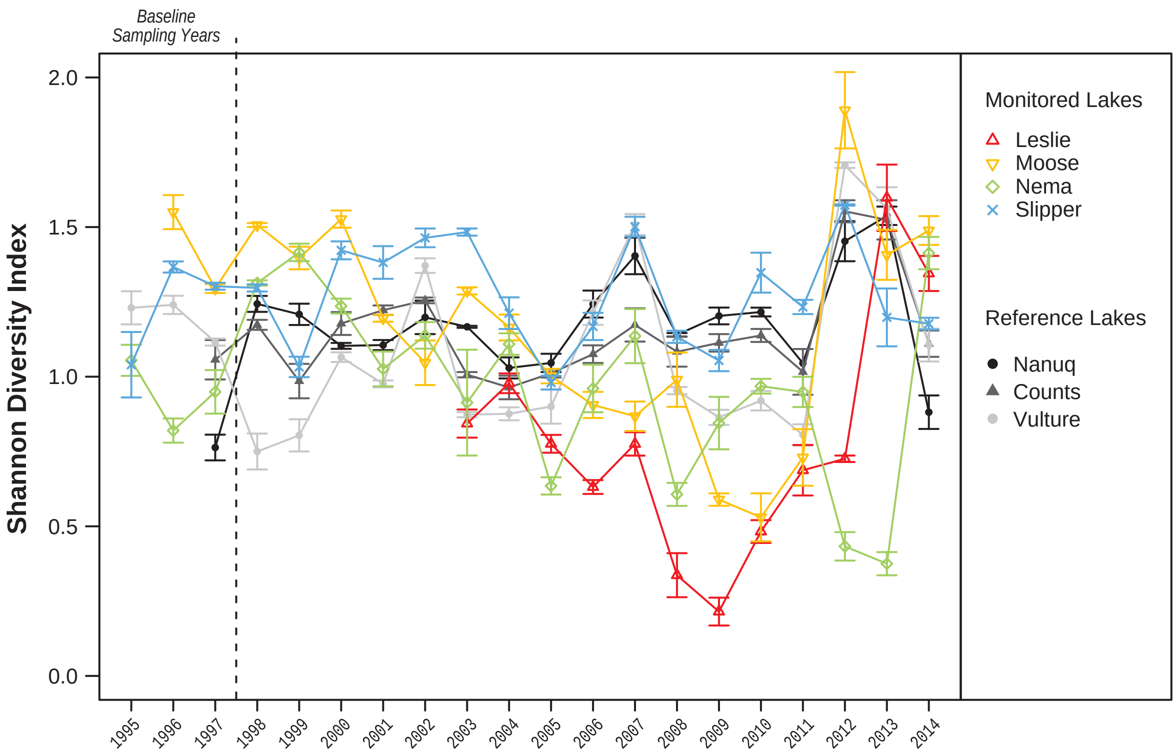


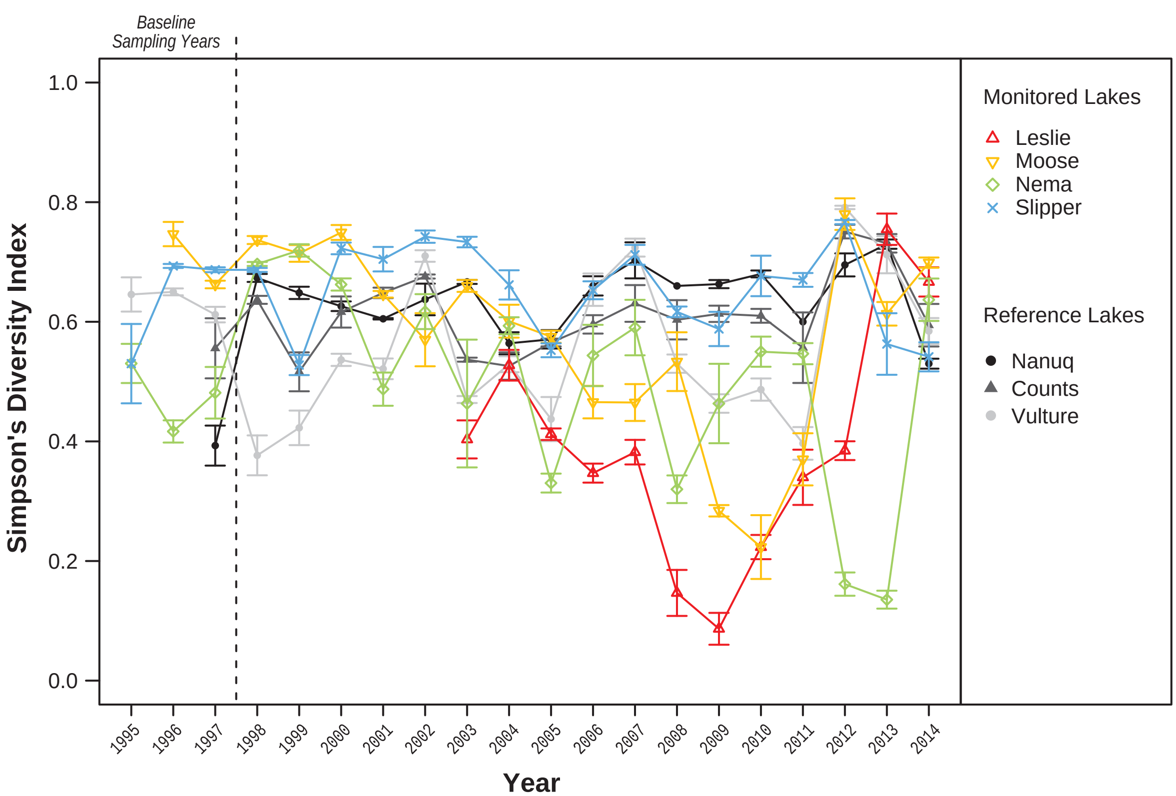


Notes: Symbols represent observed mean values.
Error bars indicate standard error of the observed means.

Figure 3.4-10b

Average Diversity Indices for Zooplankton in Koala Watershed Lakes and Lac de Gras, 1995 to 2014

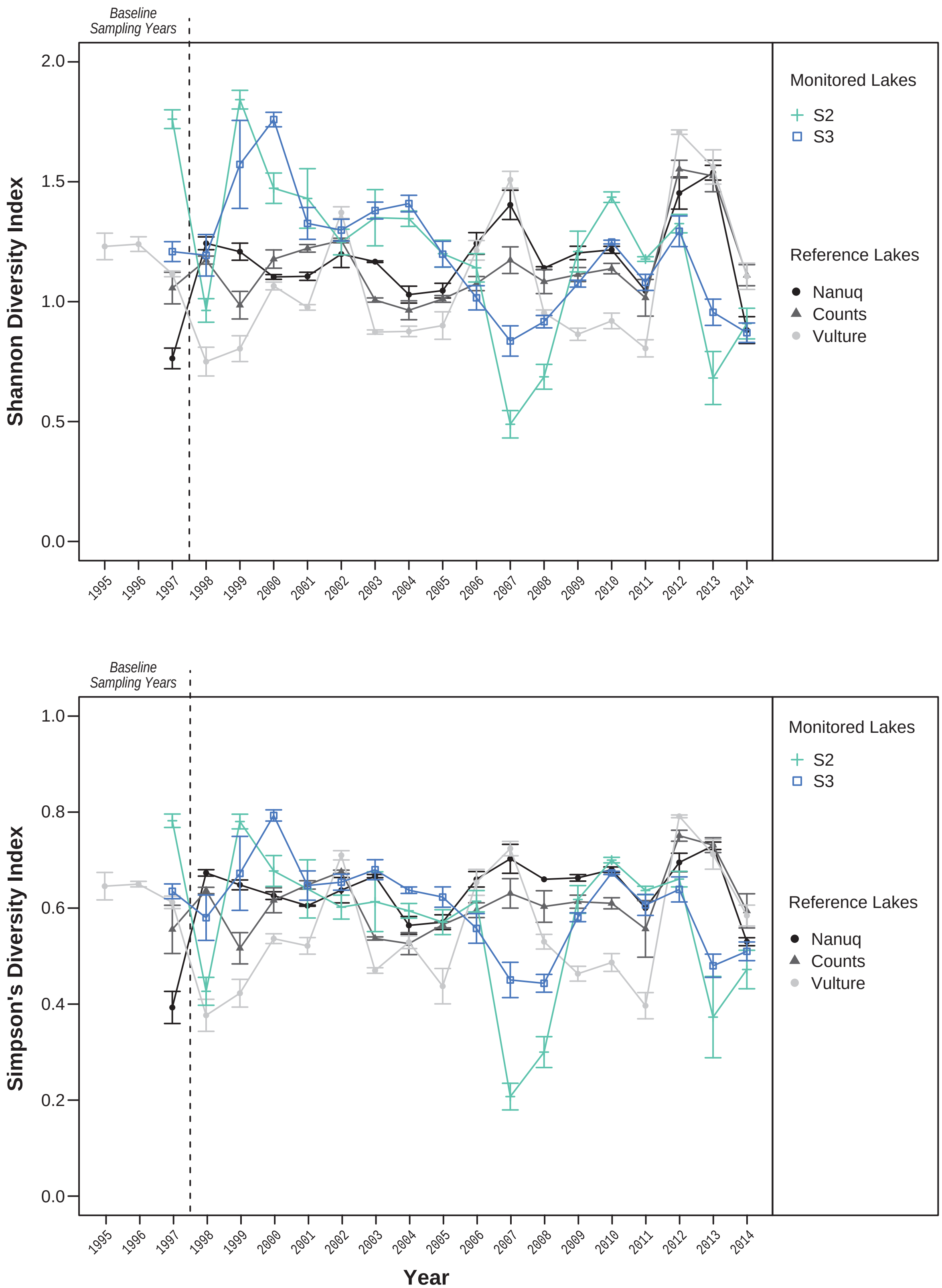




Notes: Symbols represent observed mean values.
Error bars indicate standard error of the observed means.

Figure 3.4-10c

Average Diversity Indices for Zooplankton in Koala Watershed Lakes and Lac de Gras, 1995 to 2014



Notes: Symbols represent observed mean values.
Error bars indicate standard error of the observed means.

Figure 3.4-11

Average Zooplankton Density by Taxonomic Group for AEMP Reference Lakes, 1995 to 2014

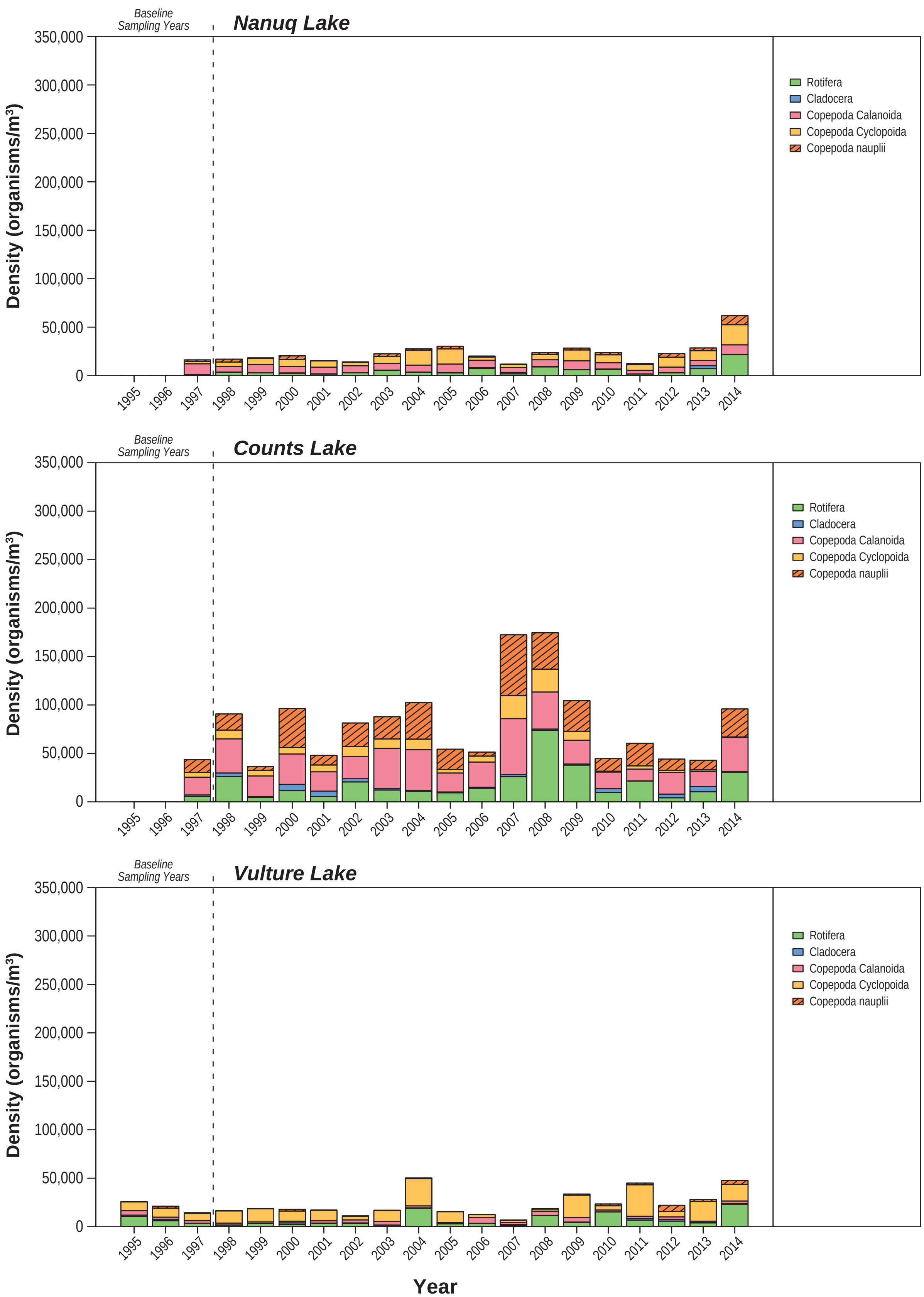


Figure 3.4-12a

Average Zooplankton Density by Taxonomic Group for Lakes of the Koala Watershed, 1995 to 2014

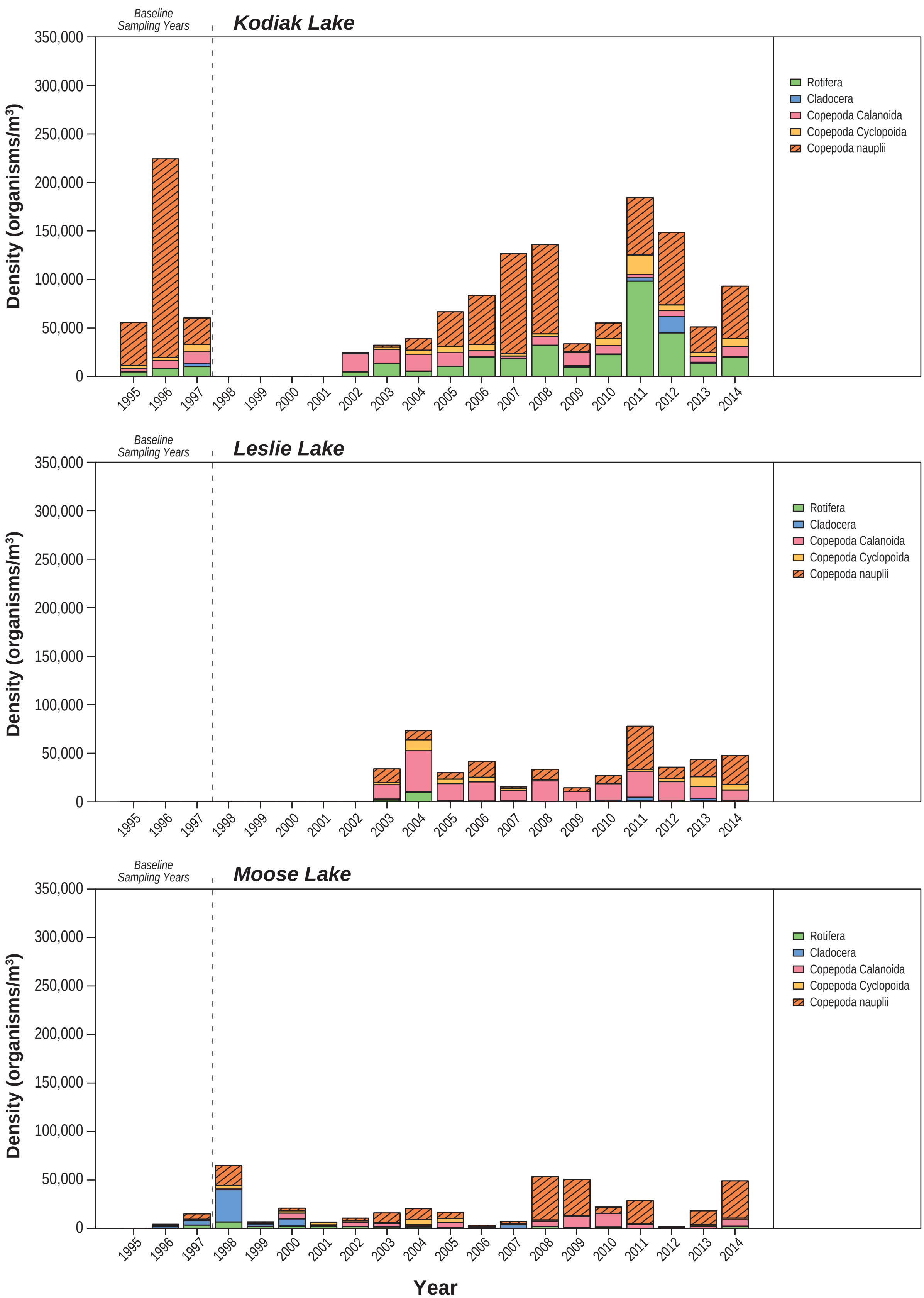
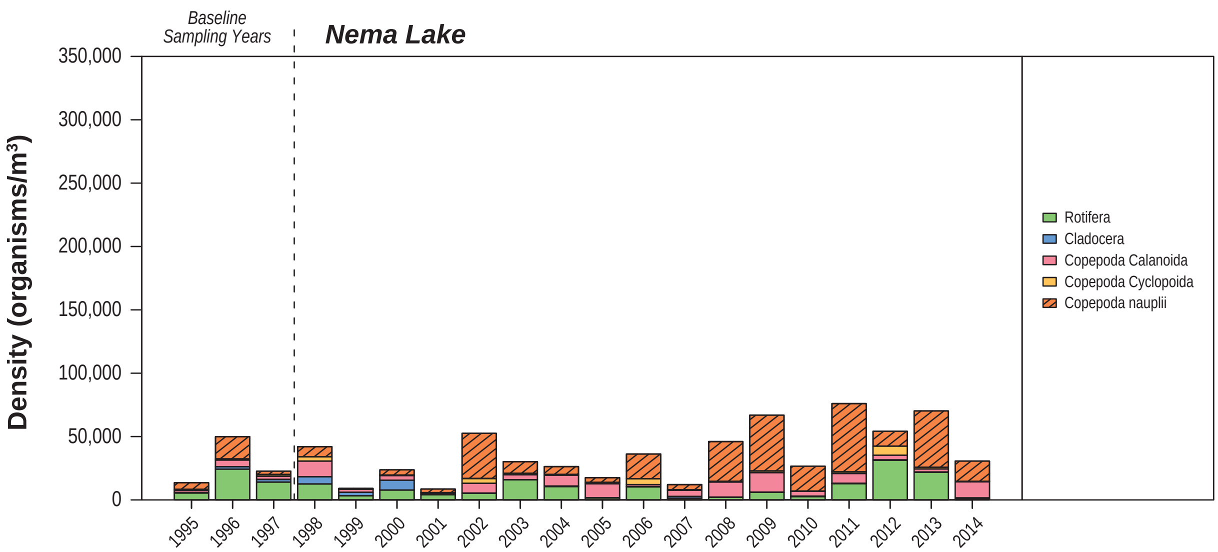


Figure 3.4-12b

Average Zooplankton Density by Taxonomic Group for Lakes of the Koala Watershed, 1995 to 2014



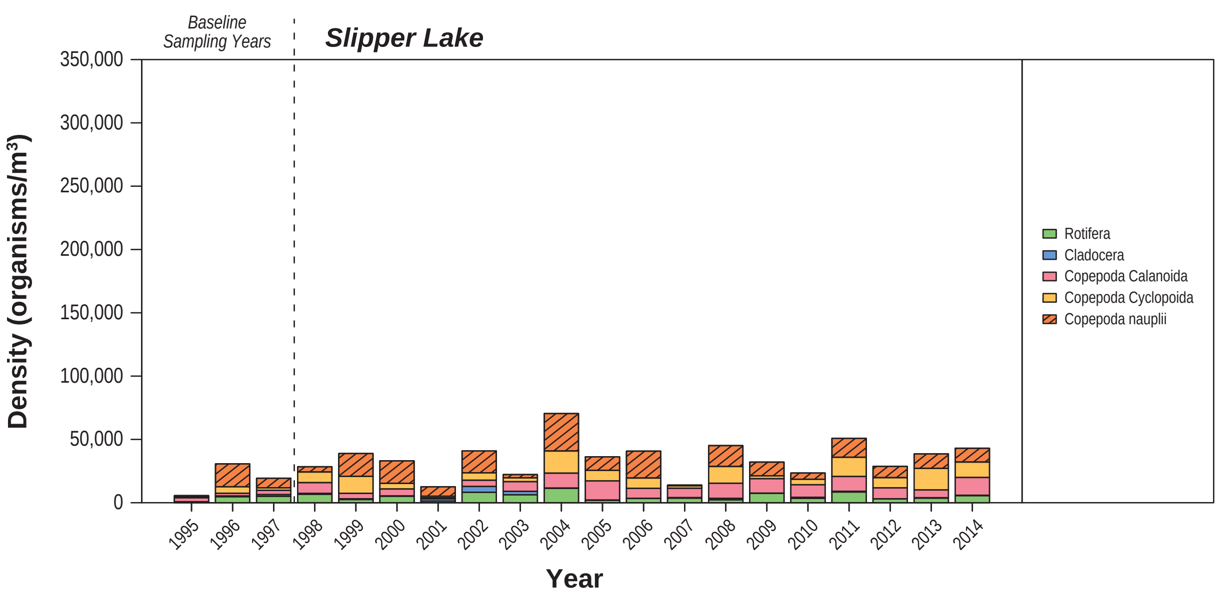


Figure 3.4-13

Average Zooplankton Density by Taxonomic Group for Lac de Gras, 1995 to 2014

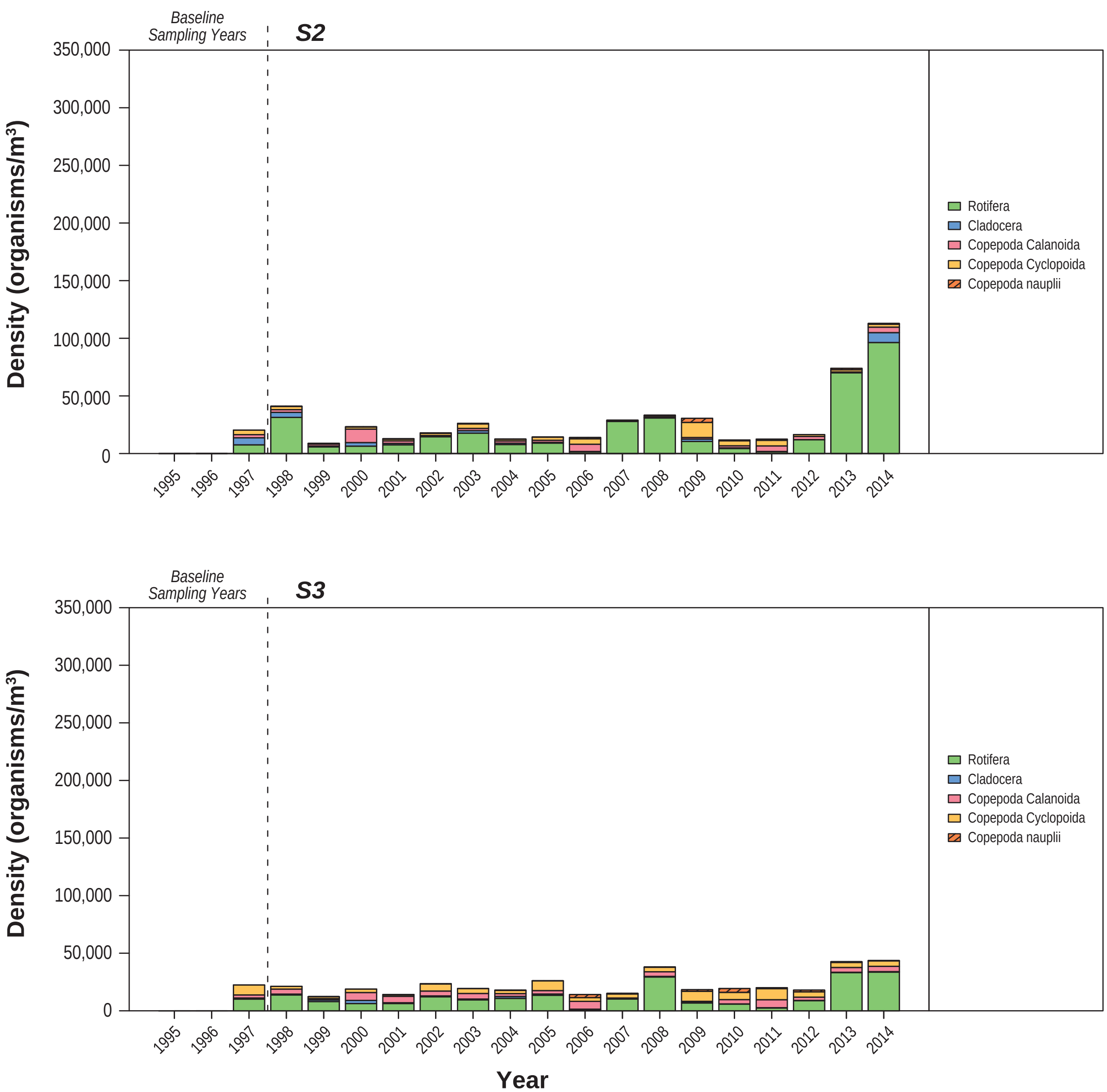


Figure 3.4-14

Relative Densities of Zooplankton Taxa in AEMP Reference Lakes, 1995 to 2014

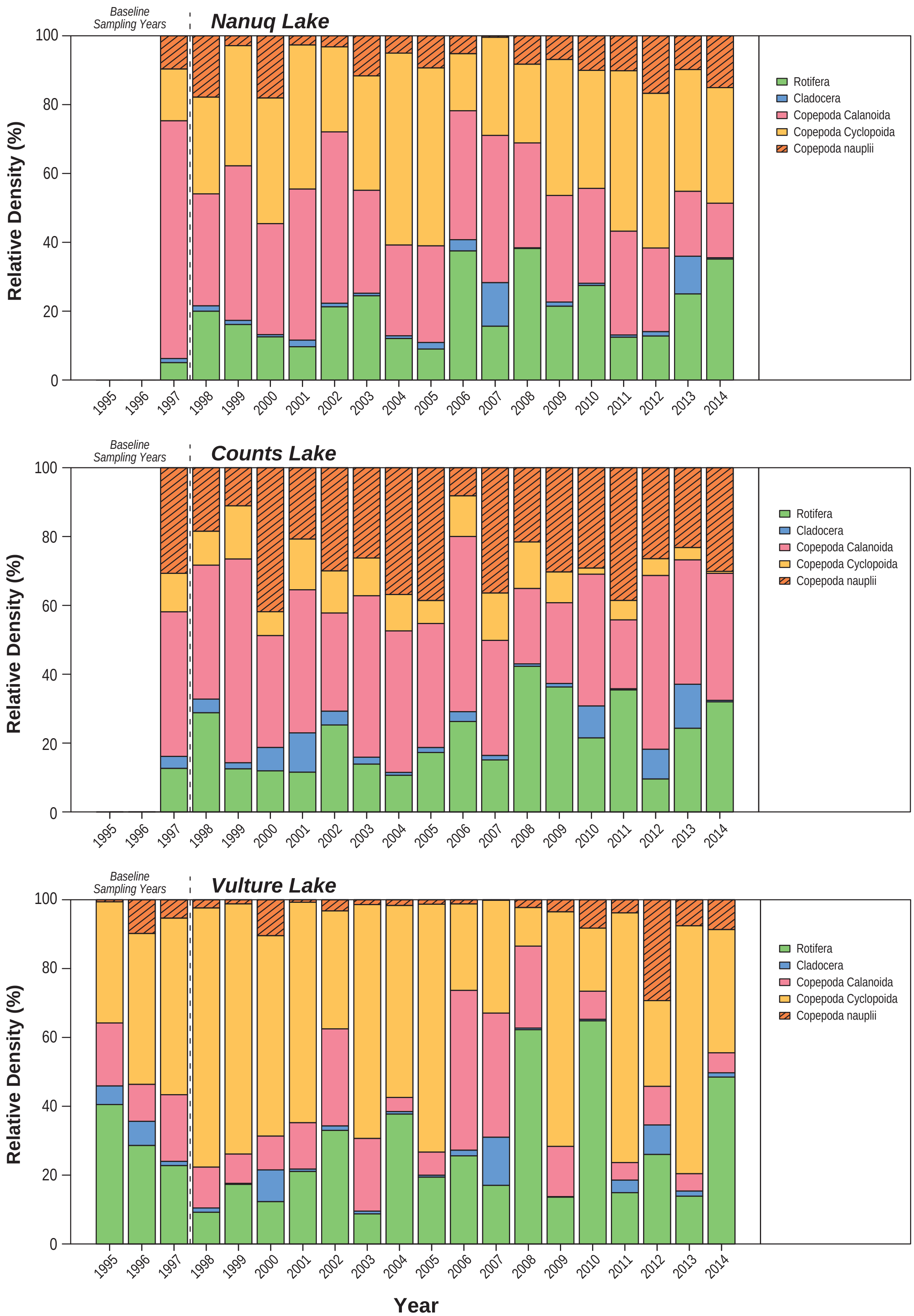


Figure 3.4-15a

Relative Densities of Zooplankton Taxa in Lakes of the Koala Watershed, 1995 to 2014

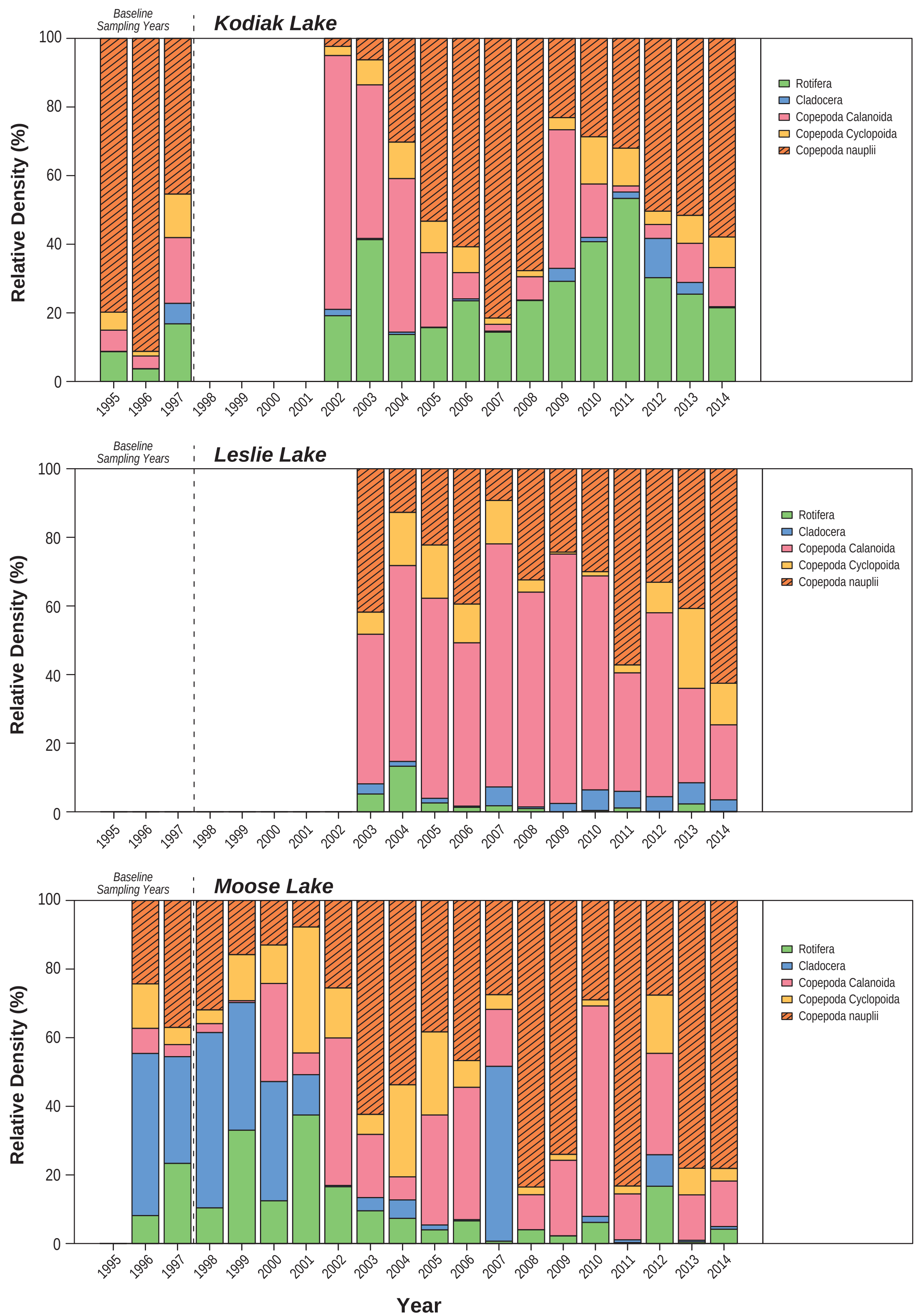


Figure 3.4-15b

Relative Densities of Zooplankton Taxa in Lakes of the Koala Watershed, 1995 to 2014

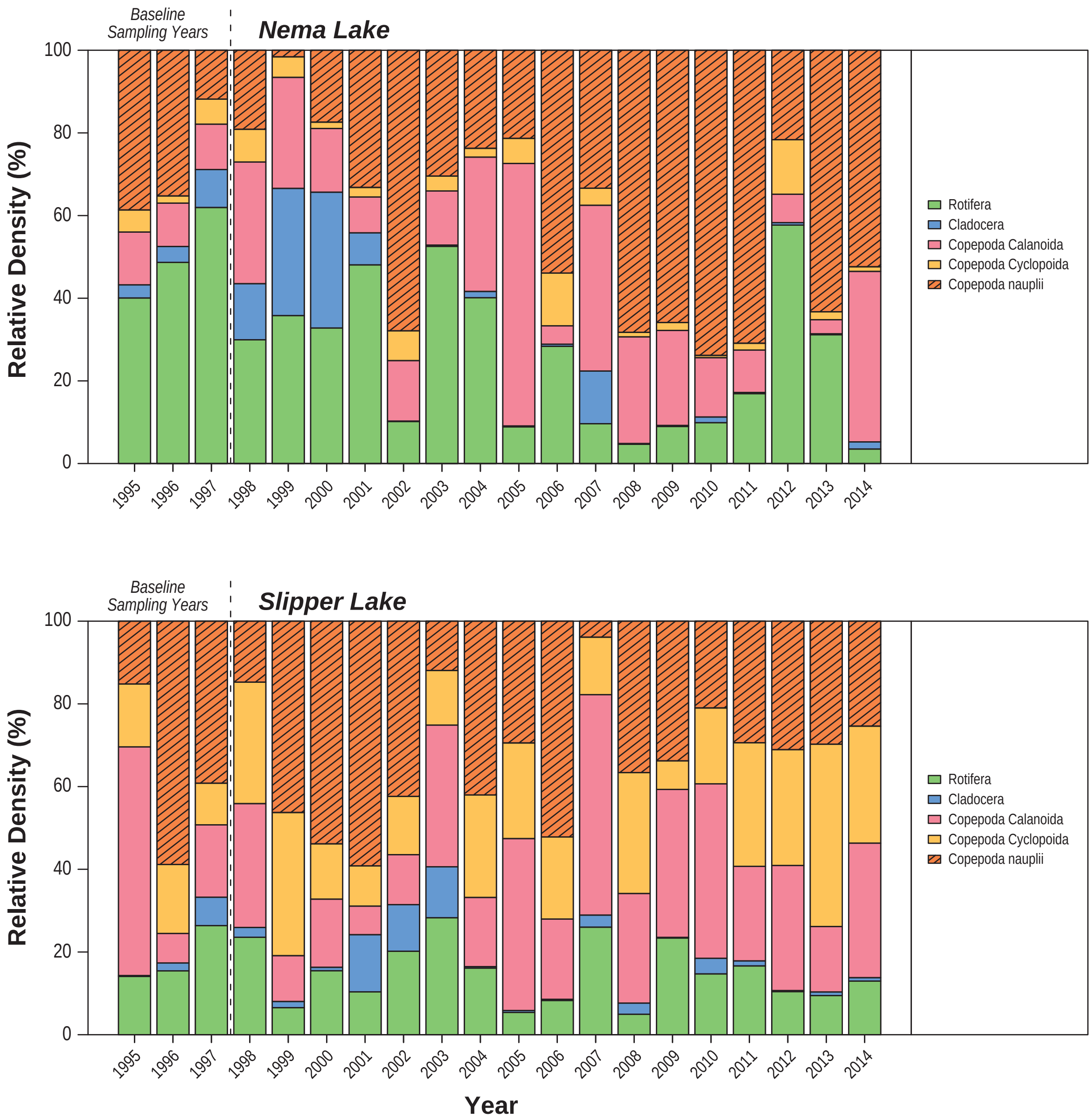


Figure 3.4-16

Relative Densities of Zooplankton Taxa in Lac de Gras, 1995 to 2014

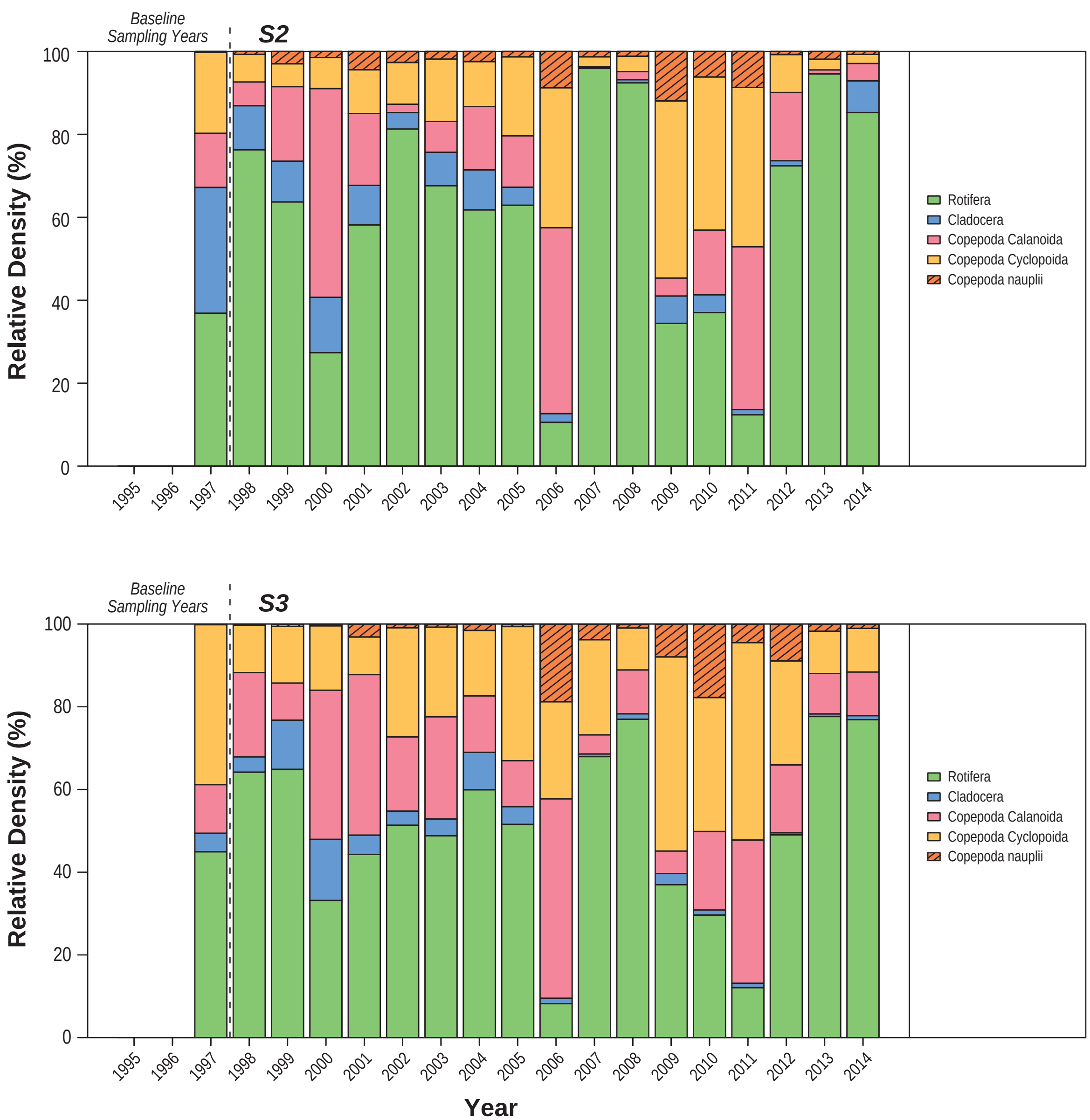


Table 3.4-11. Mean ± 2 Standard Deviations (SD) Baseline Zooplankton Diversity in Each of the Koala Watershed Lakes and Lac de Gras

Lake	Shannon Diversity			Simpson's Diversity		
	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD
Nanuq	0.76 (1)	0.61 – 0.91	0.88 ± 0.10	0.39 (1)	0.28 – 0.51	0.53 ± 0.01
Counts	1.06 (1)	0.83 – 1.29	1.11 ± 0.08	0.56 (1)	0.38 – 0.73	0.59 ± 0.06
Vulture	1.19 (3)	1.03 – 1.36	1.11 ± 0.09	0.64 (3)	0.57 – 0.70	0.58 ± 0.04
Kodiak	1.35 (1)	1.26 – 1.44	1.14 ± 0.07	0.72 (1)	0.68 – 0.75	0.61 ± 0.03
Leslie	-	-	1.35 ± 0.10	-	-	0.67 ± 0.04
Moose	1.44 (2)	1.11 – 1.73	1.49 ± 0.08	0.70 (2)	0.60 – 0.81	0.70 ± 0.01
Nema	0.94 (3)	0.68 – 1.21	1.41 ± 0.09	0.48 (3)	0.34 – 0.62	0.64 ± 0.06
Slipper	1.24 (3)	0.88 – 1.59	1.18 ± 0.03	0.64 (3)	0.44 – 0.83	0.54 ± 0.04
S2	1.76 (1)	1.63 – 1.90	0.91 ± 0.11	0.78 (1)	0.73 – 0.83	0.47 ± 0.07
S3	1.21 (1)	1.06 – 1.35	0.87 ± 0.07	0.63 (1)	0.58 – 0.69	0.51 ± 0.03

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

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 3.4-11 and 3.4-14). 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 3.4-11 and 3.4-14; see Part 2 – Data Report).

In contrast to reference lakes, zooplankton community compositions have been more variable in Moose and Nema lakes (Figures 3.4-12 and 3.4-15). 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 3.4-12a). 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 3.4-12b). Overall, rotifer densities appear to have declined over time in Moose and Nema lakes (Figures 3.4-12 and 3.4-15), a trend that is consistent with results of the 2012 AEMP Re-evaluation (Rescan 2012c). Although rotifer populations in Nema Lake showed a trend towards recovering in recent years (2011-2013), densities in 2014 were low and comprised a very small fraction (< 4%) of the total zooplankton density (Figures 3.4-12b and 3.4-15b). In Leslie Lake, zooplankton compositions have been similar since monitoring began (Figures 3.4-12a and 3.4-15a). 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 (Figures 3.4-12 and 3.4-15). Further downstream from the LLCF, community compositions have been more consistent through time, if somewhat more variable than in reference lakes (Figures 3.4-12 to 3.4-16). 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 3.4-13 and 3.4-16). The densities of rotifers at site S2 and S3 increased in 2013 and 2014, with a corresponding decrease in copepods at site S2 (Figures 3.4-13 and 3.4-16). The low diversity observed at site S2 and S3 in 2014 likely reflects the uneven distribution across species as a result of increased rotifers (Figure 3.4-10c).

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 2008 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 3.2.4-6 , the observed mean potassium concentrations in Leslie and Moose lakes exceeded the potassium SSWQO of 41 mg/L (Rescan 2012f). The observed potassium concentration in Leslie and Moose lakes also 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, but has shown some signs of possible recovery in recent years. *K. longispina* density in Moose Lake in 2014 was two orders of magnitude greater than in the last four years and was comparable to densities observed during baseline years. Although *K. longispina* density was low in Nema Lake in 2014, densities from 2011 to 2013 have been comparable to or greater than baseline values.

In contrast, densities of rotifers at sites S2 and S3 in Lac de Gras have increased in 2013 and 2014. This recent increase appears to be mostly driven by an increase in the density of *Conochilus* at both sites, and an increase in the density of *Keratella* sp. at site S3. Sites S2 and S3 in Lac de Gras have generally been dominated by rotifers and this pattern of increasing rotifer densities contrasts the changes in community composition observed at more upstream sites, closer to the LLCF.

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. 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 3.4.5).

3.4.3 Lake Benthos

3.4.3.1 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) and community composition were evaluated for potential mine effects.

3.4.3.2 Dataset

Lake benthos samples have been collected in triplicate replicates in late July or early August of each year since 1994 (Table 3.4-12). 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 3.4-12 and shown graphically in Figures 3.4-17 to 3.4-22 for visual comparison.

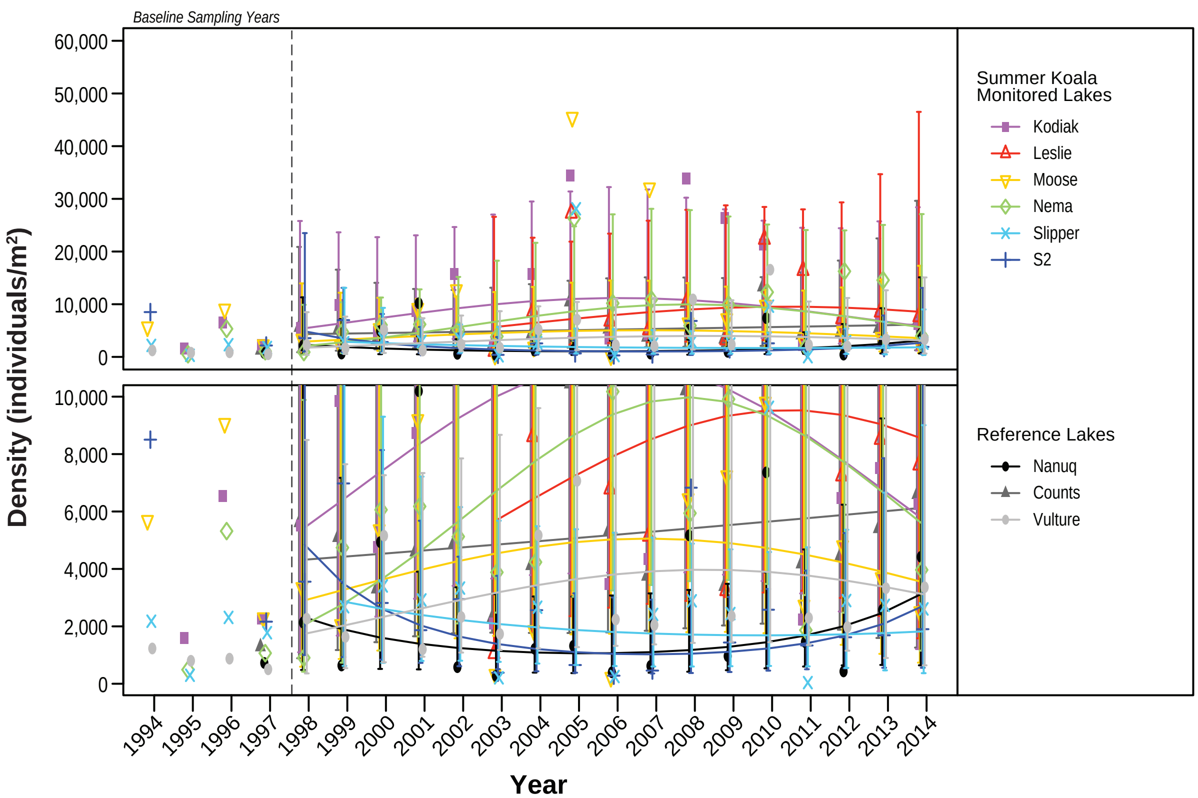
Table 3.4-12. Dataset Used for Evaluation of Effects on the Benthos in Koala Watershed Lakes and Lac de Gras

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
2014	Aug-5	Aug-9	Aug-3	Jul-29	Jul-31	Jul-31	Aug-2	Aug-4	Aug-3

Note: Dashes indicate no data were available.
** Indicates data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data ford visual comparison.*

Figure 3.4-17

Observed and Fitted Means for Benthos Densities in Koala Watershed Lakes and Lac de Gras, 1994 to 2014



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

Benthos Density

Statistical and graphical analyses indicate that the density of benthos has been stable through time in all monitored and reference lakes (Table 3.4-13; Figure 3.4-17). Mean benthos densities in monitored lakes in 2014 remained within ± 2 SD of the mean benthos densities observed during baseline years (Table 3.4-14). Thus, no mine effects were detected with respect to lake benthos density.

3.4.3.3 Results and Discussion

Table 3.4-13. 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	
Benthos Density	-	LME	2	-	None	-	1-446

Note: Dashes indicate not applicable.

Table 3.4-14. Mean ± 2 Standard Deviations (SD) Baseline Lake Benthos Density in Each of the Koala Watershed Lakes and Lac de Gras

Lake	Baseline Mean (N)	Mean Baseline Range , ± 2 SD	2014 Mean ± 1 SD
Nanuq	726 (1)	325 – 1,126	4,420 ± 1,198
Counts	1,289 (1)	0 – 3,212	6,594 ± 838
Vulture	852 (4)	0 – 1,960	3,363 ± 168
Kodiak	3,471 (3)	0 – 10,430	6,270 ± 838
Leslie	-	-	7,620 ± 2,023
Moose	5,683 (3)	0 – 17,195	2,509 ± 522
Nema	2,291 (3)	0 – 8,005	3,965 ± 1,565
Slipper	1,641 (4)	0 – 4,218	2,607 ± 869
S2	5,333 (2)	0 – 12,536	1,901 ± 1,177

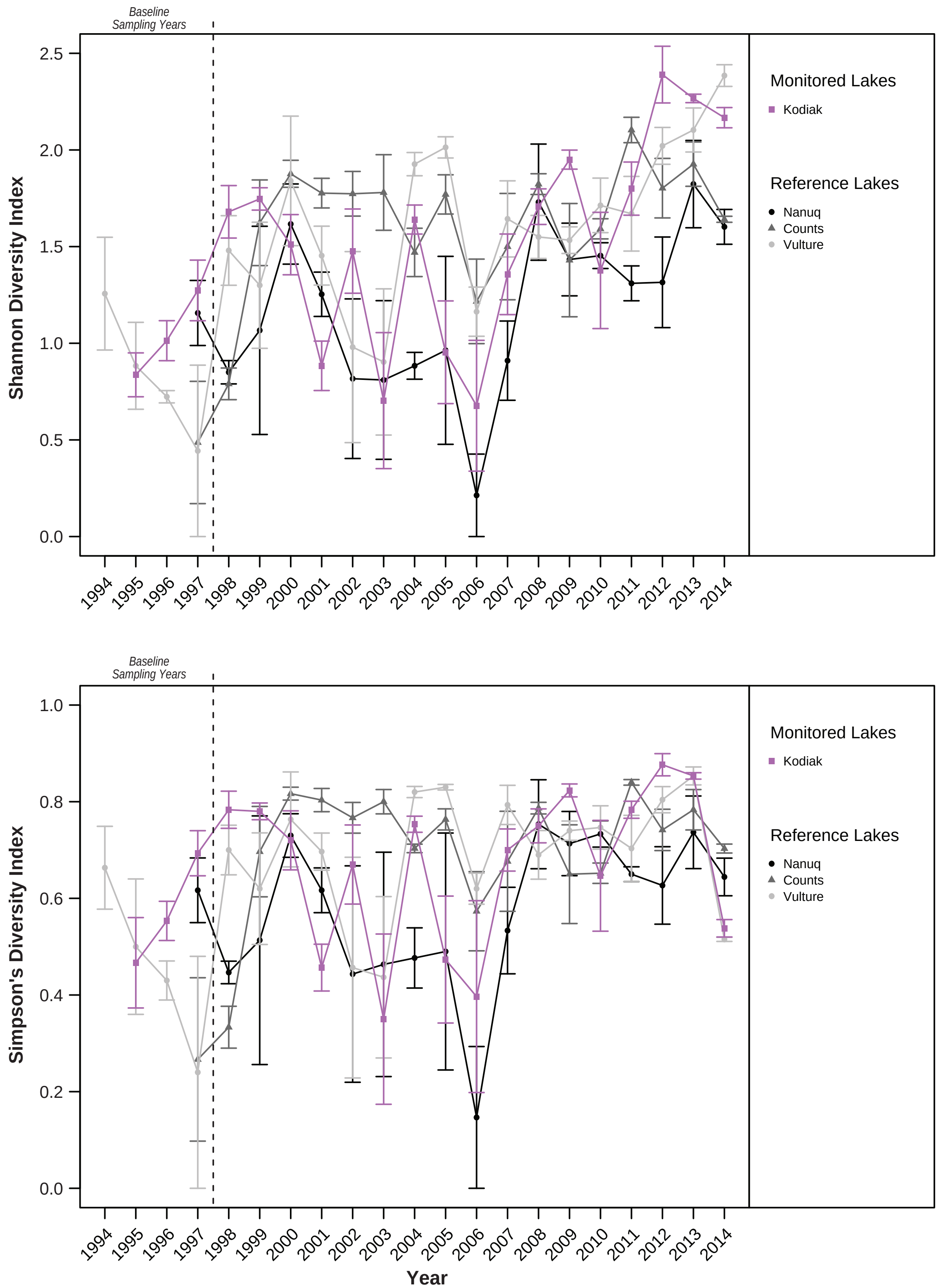
Notes: Units are organisms/m^2.
Negative values were replaced with zeros.
N = number of years data were collected.
Dashes indicate data not available.

Dipteran Diversity and Community Composition

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 3.4-18) 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 3.4-19 to 3.4-22).

Figure 3.4-18a

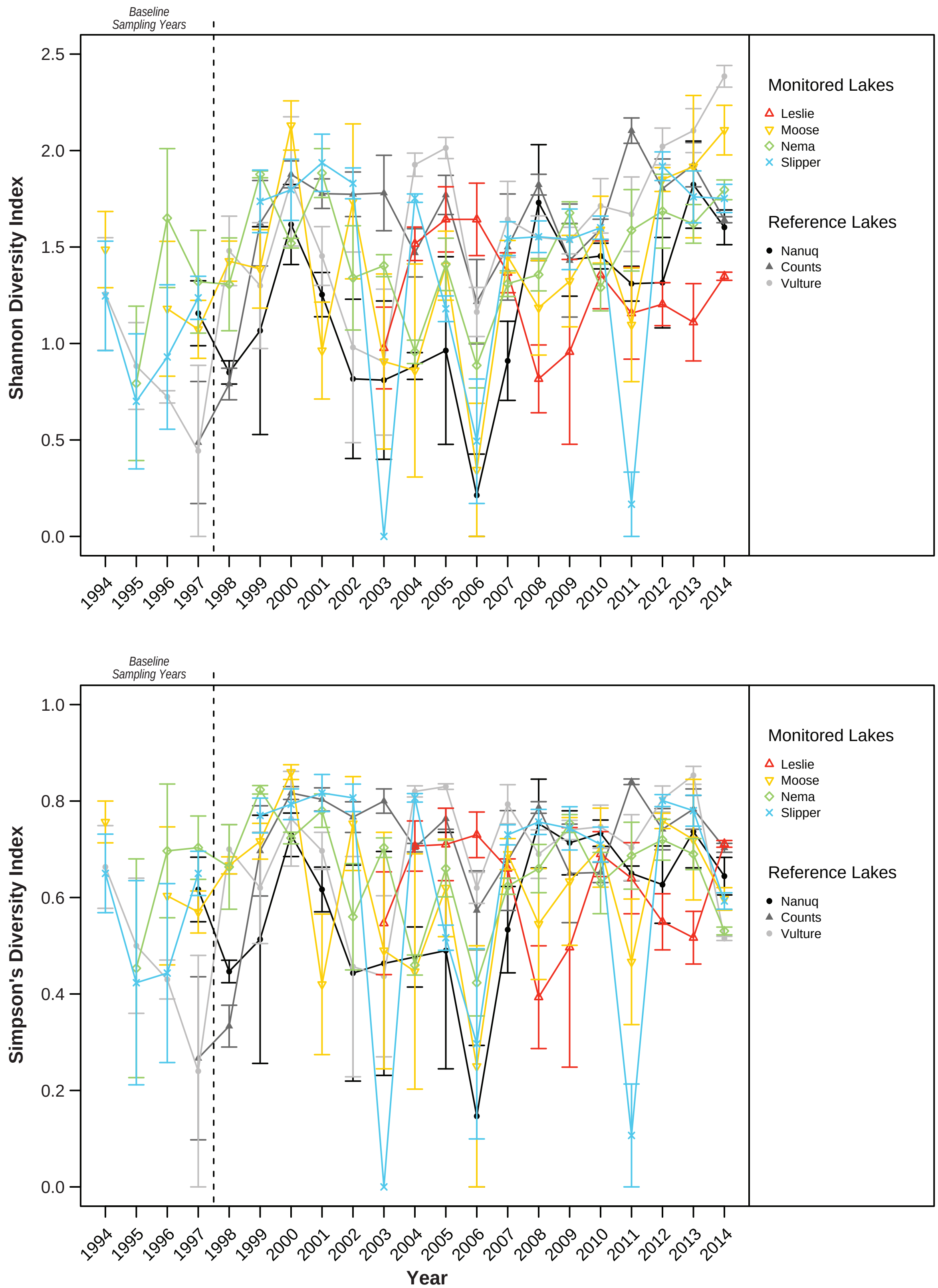
Average Diversity Indices for Benthic Dipterans in Koala Watershed Lakes and Lac de Gras, 1994 to 2014



Notes: Symbols represent observed mean values.
Error bars indicate standard error of the observed means.

Figure 3.4-18b

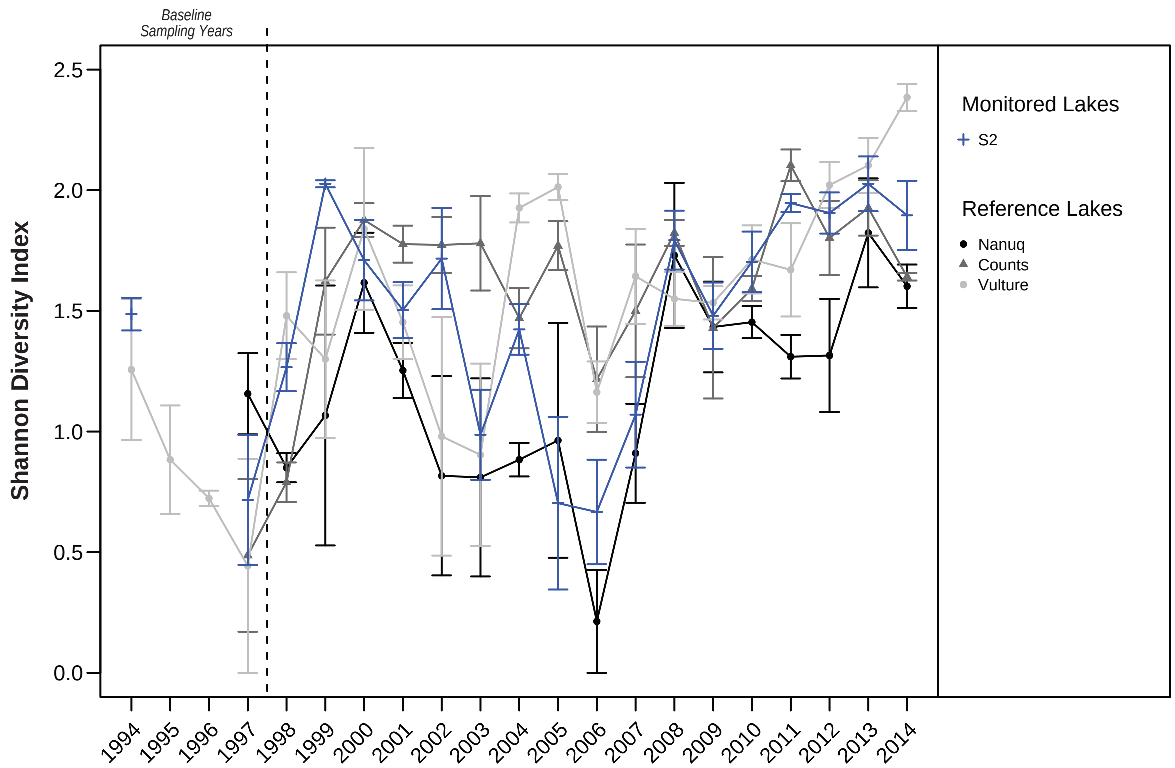
Average Diversity Indices for Benthic Dipterans in Koala Watershed Lakes and Lac de Gras, 1994 to 2014

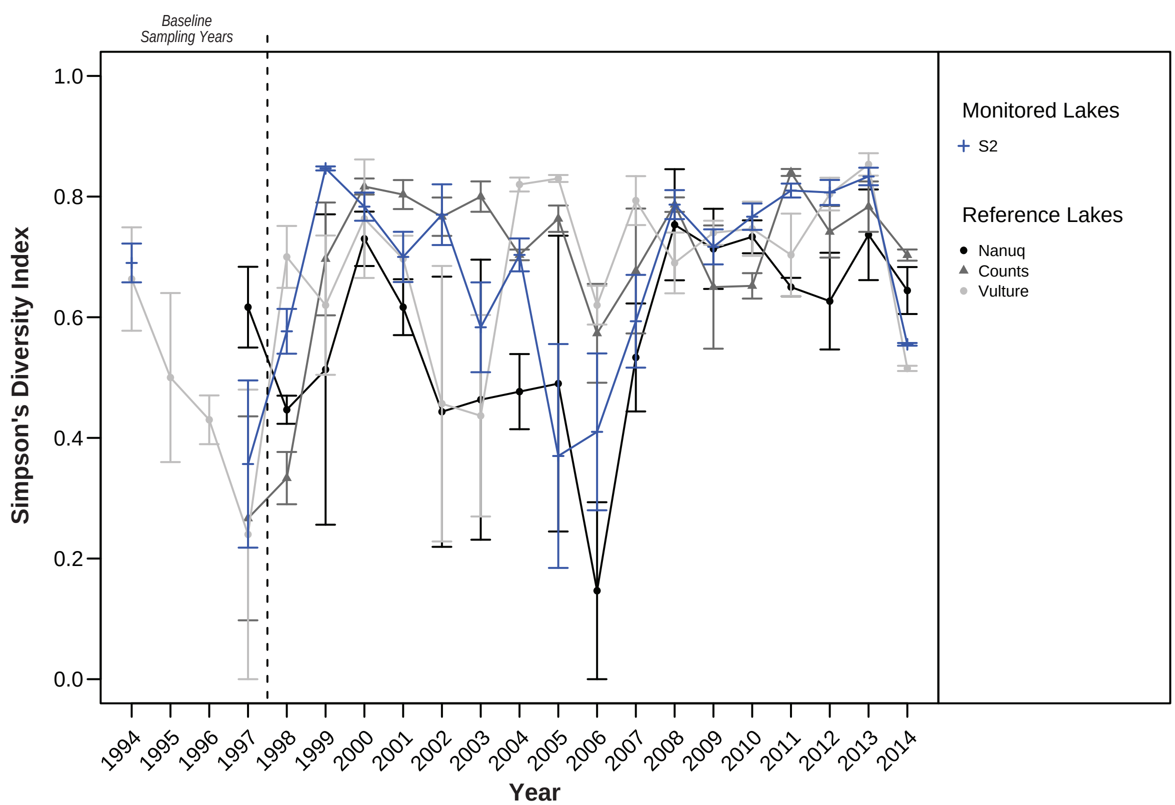


Notes: Symbols represent observed mean values.
Error bars indicate standard error of the observed means.

Figure 3.4-18c

Average Diversity Indices for Benthic Dipterans in Koala Watershed Lakes and Lac de Gras, 1994 to 2014





Notes: Symbols represent observed mean values.
Error bars indicate standard error of the observed means.

Figure 3.4-19

Average Diptera Density by Taxonomic Group for AEMP Reference Lakes, 1994 to 2014

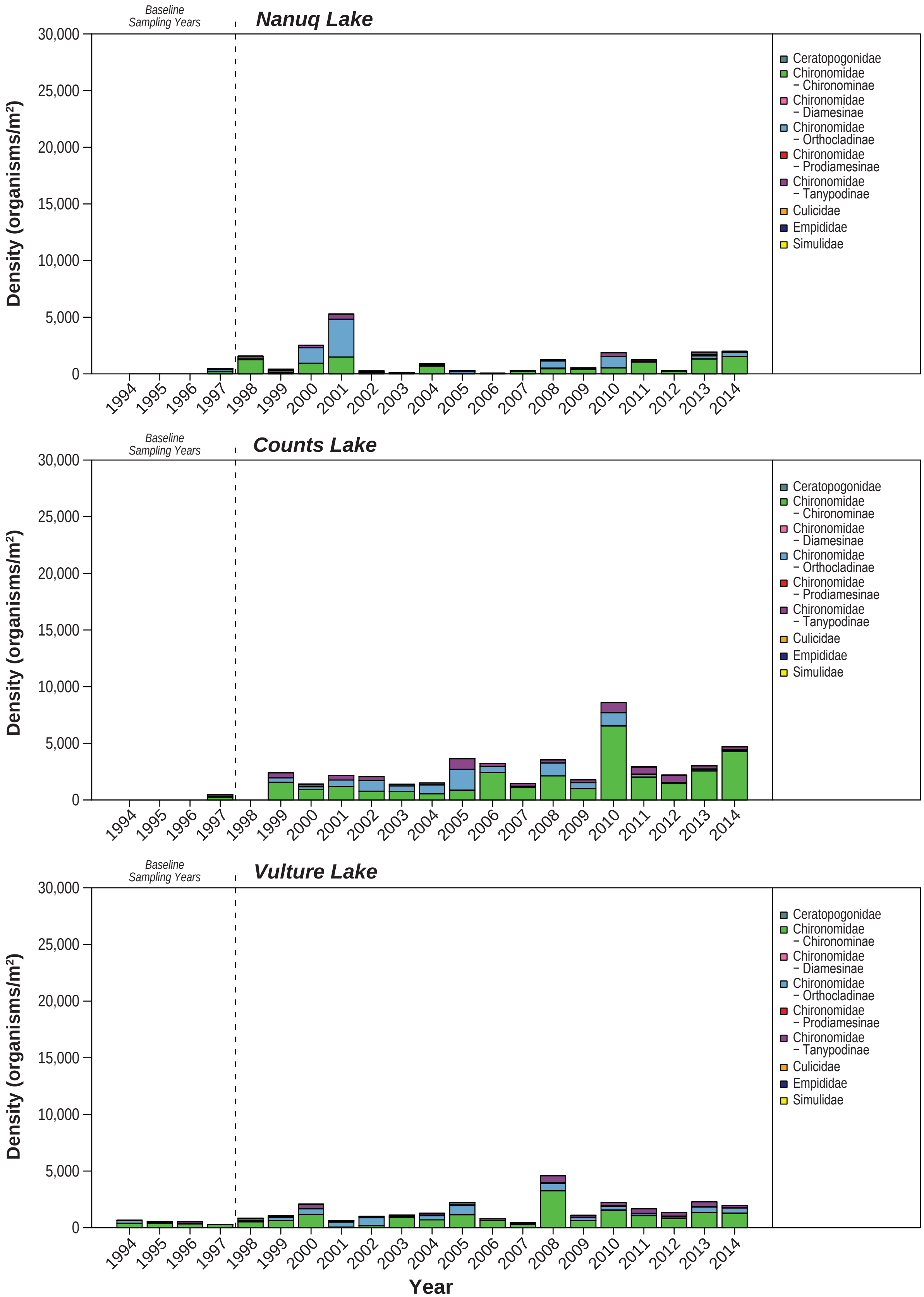


Figure 3.4-20a

Average Diptera Density by Taxonomic Group for Lakes of the Koala Watershed and Lac de Gras, 1994 to 2014

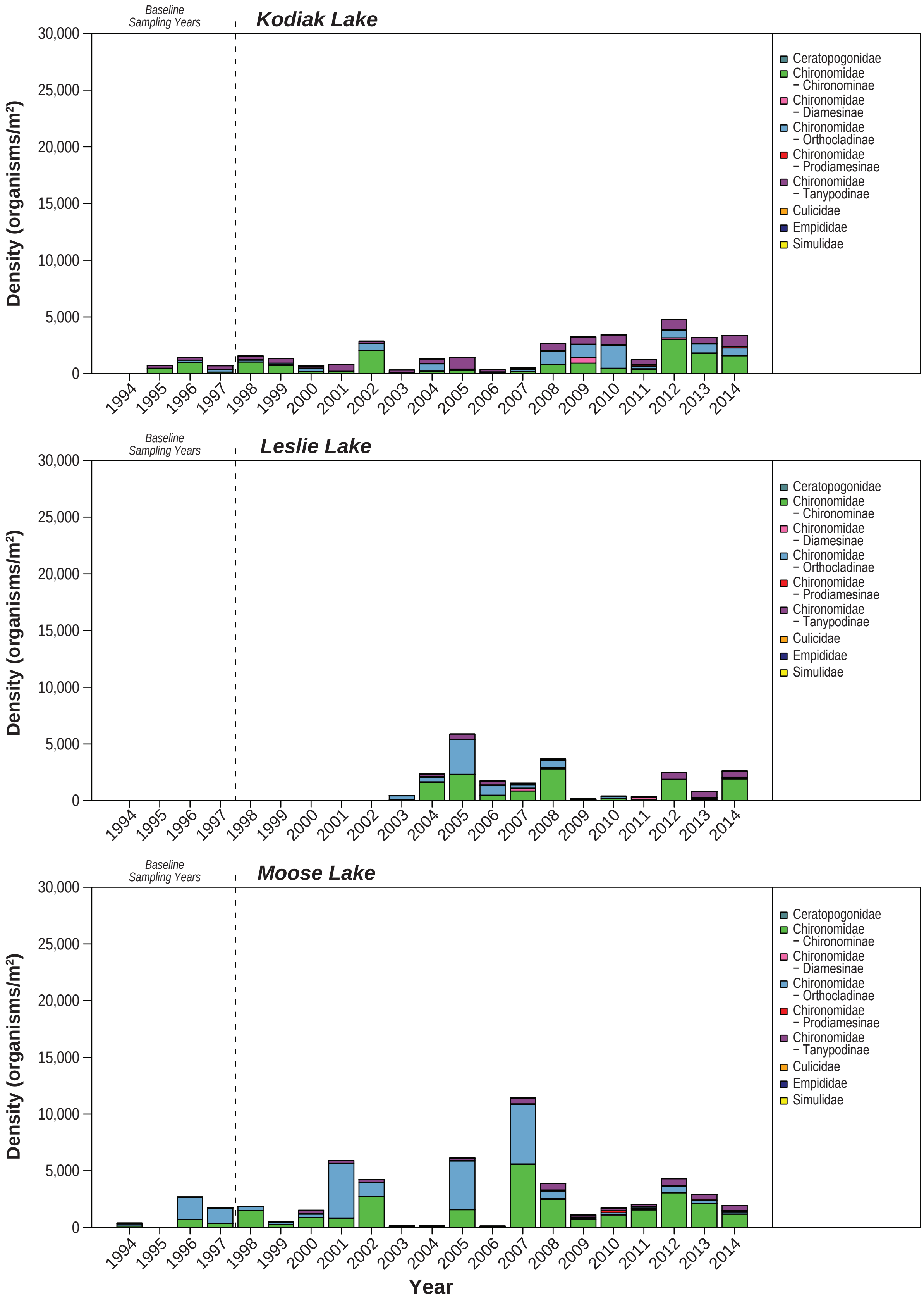


Figure 3.4-20b

Average Diptera Density by Taxonomic Group for Lakes of the Koala Watershed and Lac de Gras, 1994 to 2014

ERM

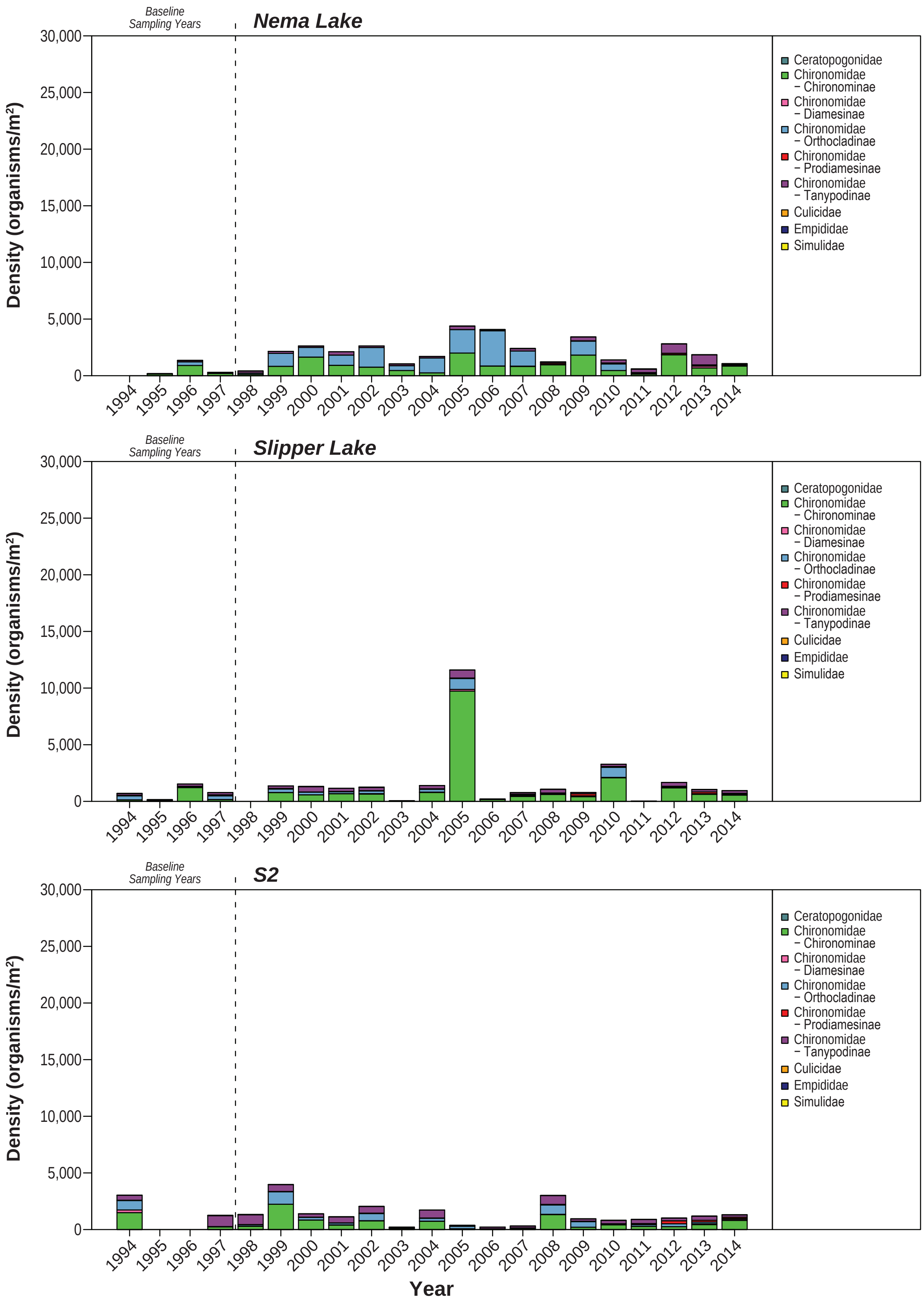


Figure 3.4-21

Relative Densities of Diptera Taxa in AEMP Reference Lakes, 1994 to 2014

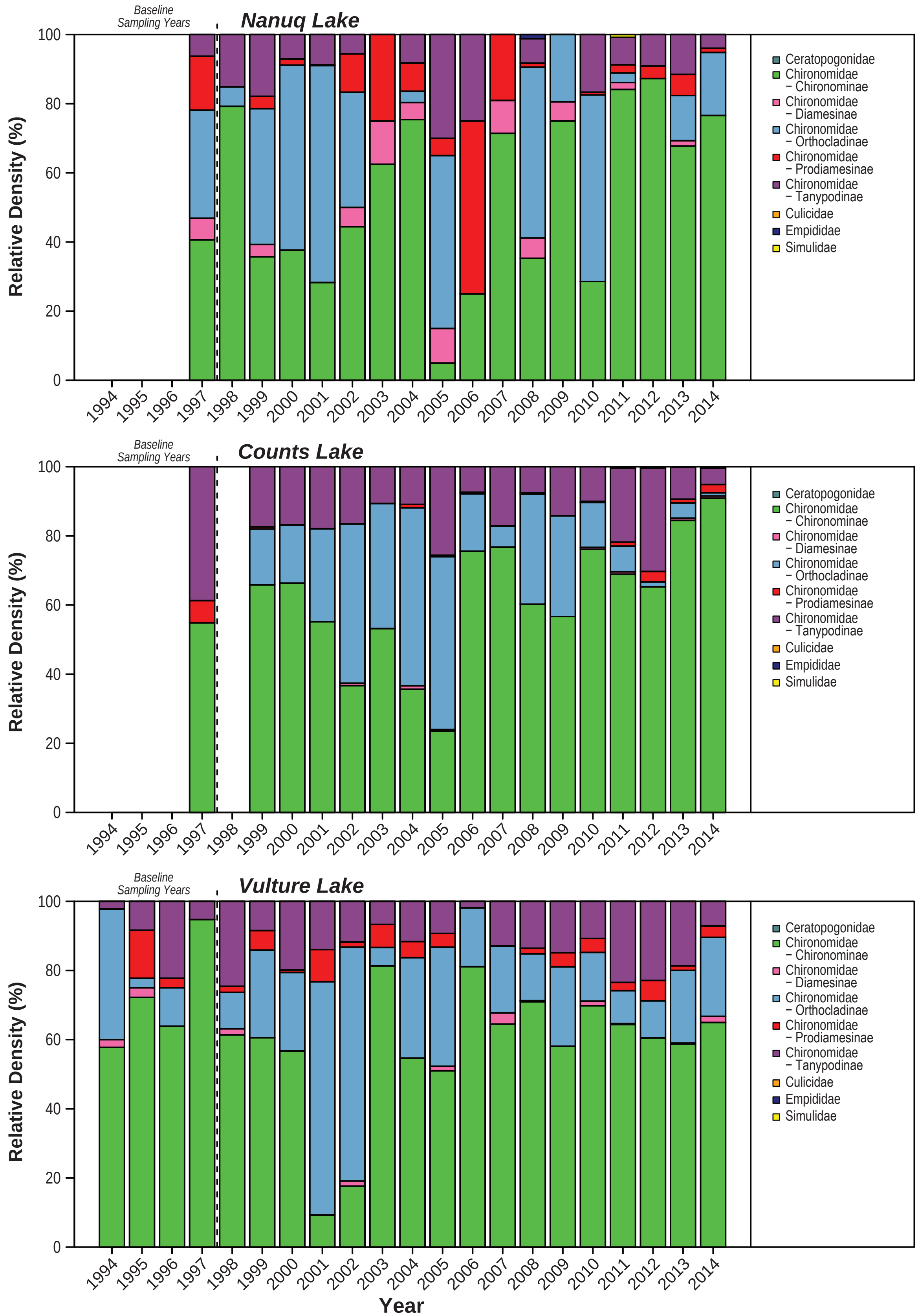


Figure 3.4-22a

Relative Densities of Diptera Taxa in Lakes of the Koala Watershed and Lac de Gras, 1994 to 2014

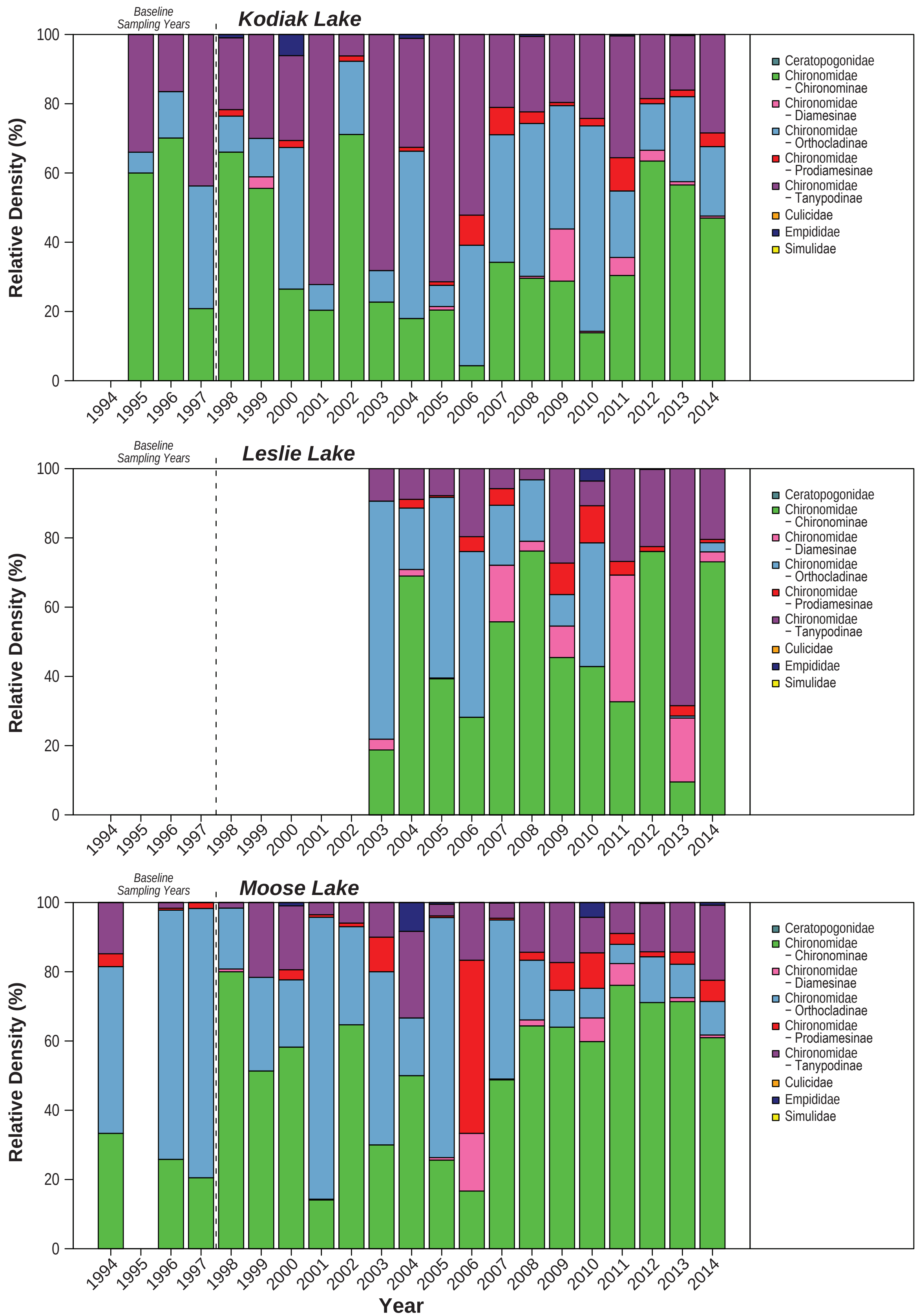
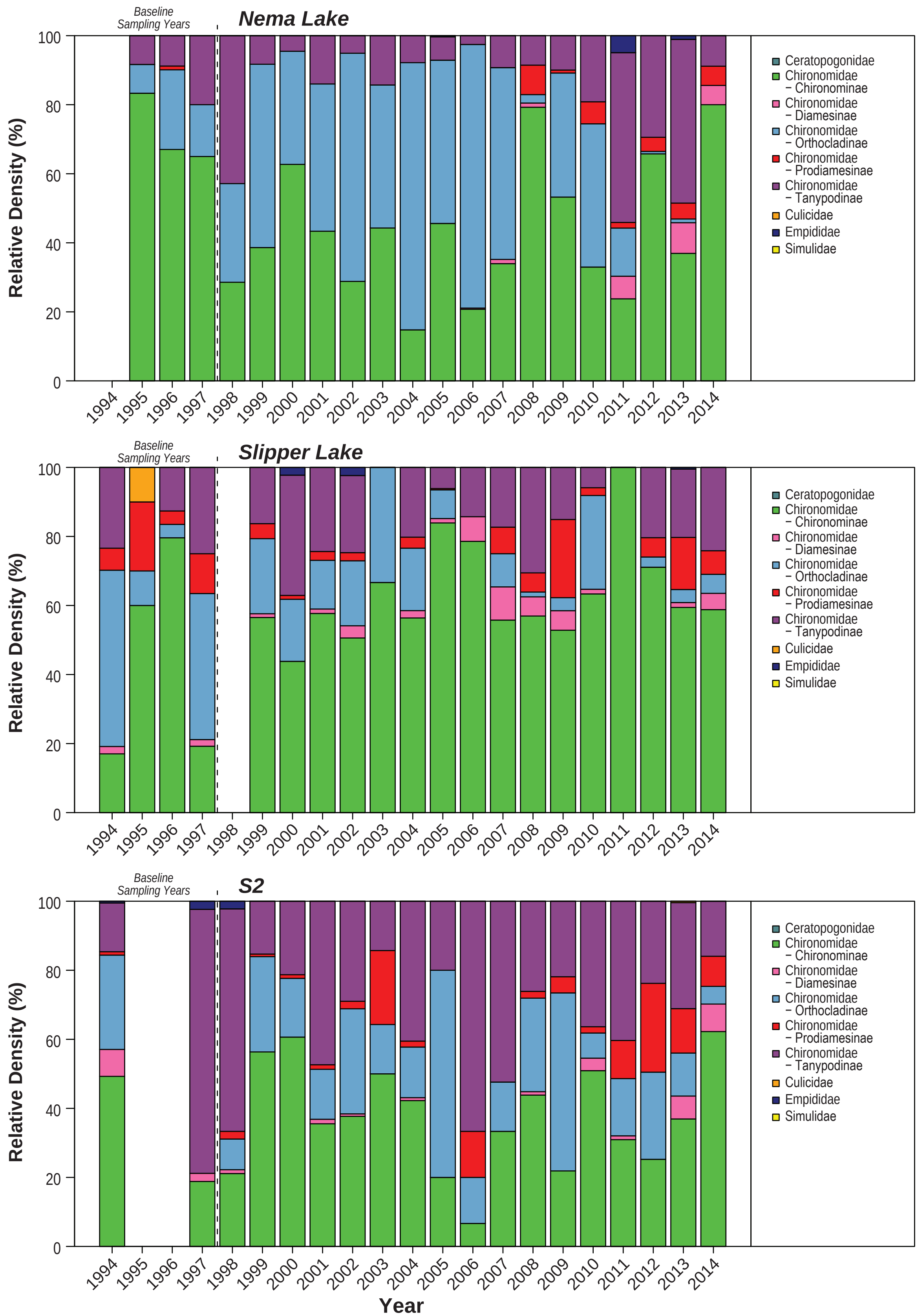


Figure 3.4-22b

Relative Densities of Diptera Taxa in Lakes of the Koala Watershed and Lac de Gras, 1994 to 2014



Both Shannon and Simpson's diversity indices have varied considerably through time in both monitored and reference lakes since monitoring began (Figure 3.4-18). While the variability makes it difficult to discern temporal trends, it also suggests that diversity tends to fluctuate consistently through time in all lakes, with patterns in monitored lakes generally similar to ones observed in reference lakes (Figure 3.4-18). For example, mean Shannon diversity was greater than mean baseline diversity ± 2 SD in Kodiak and Moose lakes, and at site S2 in Lac de Gras; however, a similar pattern was observed in two of the reference lakes (i.e., Counts and Vulture lakes; Table 3.4-15). Therefore, no mine effects were detected with respect to dipteran diversity in lakes of the Koala Watershed or Lac de Gras.

Table 3.4-15. 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	2014 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD
Nanuq	1.16 (1)	0.57 – 1.75	1.60 ± 0.16	0.39 (1)	0.28 – 0.51	0.64 ± 0.07
Counts	0.49 (1)	0 – 1.59	1.64 ± 0.03	0.27 (1)	0 – 0.86	0.70 ± 0.02
Vulture	0.44 (4)	0 – 1.49	2.38 ± 0.10	0.24 (4)	0.61 – 0.78	0.52 ± 0.01
Kodiak	1.27 (3)	0.73 – 1.81	2.17 ± 0.09	0.69 (3)	0.42 – 0.97	0.54 ± 0.03
Leslie	-	-	1.35 ± 0.04	-	-	0.71 ± 0.01
Moose	1.07 (3)	0.24 – 1.90	2.11 ± 0.22	0.57 (3)	0.25 – 0.89	0.60 ± 0.04
Nema	1.32 (3)	0.04 – 2.60	1.80 ± 0.09	0.70 (3)	0.17 – 1.0	0.53 ± 0.01
Slipper	1.24 (4)	0.23 – 2.24	1.75 ± 0.13	0.65 (4)	0 – 1.0	0.59 ± 0.03
S2	0.72 (2)	0 – 1.76	1.90 ± 0.22	0.36 (2)	0 – 0.83	0.56 ± 0.004

Notes: 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 3.4-19 to 3.4-22). 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 densities and relative densities of dipteran taxonomic groups have changed through time in Leslie and Moose lakes (Figures 3.4-20a and 3.4-22a). Specifically, the densities of Orthocladiinae have decreased, while densities of Chironominae, Prodiamesinae, Diamesinae, and/or Tanypodinae have increased through time (Figures 3.4-20a and 3.4-22a). These patterns are consistent with those that were first identified through the multivariate analyses conducted as part of the 2012 AEMP Re-evaluation (Rescan 2012c). In addition, graphical analyses suggest that relative densities of Orthocladiinae in Nema Lake have also decreased, with a coincidental increase in densities of Tanypodinae, Diamesinae, and Prodiamesinae (Figures 3.4-20b and 3.4-22b) and that densities of Prodiamesinae have recently increased at site S2 in Lac de Gras

(Figures 3.4-20b and 3.4-22b). Although these patterns were not generally observed in reference lakes, graphical analyses revealed a recent trend of decreasing densities of Orthocladiinae with increasing densities of Chironominae in Counts Lake (Figures 3.4-19 and 3.4-21).

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, similarly to results from the 2013 AEMP (ERM Rescan 2014a), some of the trends that were described in the 2012 AEMP (Rescan 2013) were less apparent in recent years. 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 many genera 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 genera *Heterotanytarsus* and *Rheocricotopus* in Moose Lake and *Psectrocladius* in Leslie Lake. However, organisms from *Rheocricotopus* and *Psectrocladius* have only been previously observed on two occasions in Moose Lake and Leslie Lake, respectively. 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. In addition, organisms from the genus *Zalutschia* were abundant in 1998 and 1999 and organisms from the genus *Heteroctrissocladius* were present from 2001 to 2006. Both genera have been absent since 2007. In Counts Lake, the decrease in Orthocladiinae is less clearly linked to any particular genus, but may be related to recent declines in *Heteroctrissocladius* and *Psectrocladius*;
- The increase in Chironominae in Moose Lake may be due to recent increases in *Cladotanytarsus*, *Paratanytarsus* and *Stempellinella*. In Counts Lake, the increase in Chironominae seems more likely related to recent increases in *Corynocera*, *Paratanytarsus*, and *Stictochironomus*;
- The increase in Prodiamesinae in Leslie, Moose, and Nema 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, Moose and Nema lakes appears to be related to increases in the density of organisms from the genus *Protanypus*; and
- The increase in Tanypodinae in Leslie and Nema lakes 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 or sediment chemistry (see Section 3.3.5; Rescan 2012c).

3.4.4 Stream Benthos

3.4.4.1 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) and community composition were evaluated for potential mine effects.

3.4.4.2 Dataset

Stream benthos samples have been collected over a one month period from early August to early September of each year since 1995 (Table 3.4-16). Five replicates were collected from each stream in 1995 and between 1999 and 2014, except in Kodiak-Little in 1999 when only three replicates were collected. In 1997 and 1998, triplicate samples were collected from each stream. Baseline data, which were collected between 1994 and 1997, were not used in the statistical evaluation of effects but are included in Table 3.4-16 and shown graphically in Figures 3.4-23 to 3.4-33 for visual comparison.

Although stream benthos samples were collected in 2010, they were not included in the evaluation of effects as a result of laboratory error. In 2011, 2012, and 2013, benthos samples were processed through sieves in the laboratory with incorrect mesh sizes. Historically, AEMP stream benthos samples have been processed through a 180 µm mesh sieve, but a 250 µm mesh was used in 2011 and 2012, and a 500 µm mesh sieve was used in 2013. A laboratory study was conducted in 2014 to determine the number of individuals, and the taxonomic identity of individuals, that passed through 250 µm and 500 µm mesh sieves. It was determined that dipterans from various taxonomic groups were small enough to pass through the 250 µm mesh, while EPT taxa were never found to pass through the 250 um mesh, but some EPT individuals were able to pass through the 500 µm mesh. As a result, total benthos density, and dipteran diversity, density and relative density from 2011 to 2013 were excluded from the evaluation of effects; EPT diversity, density and relative density were only excluded in 2013. However, all data collected from 2011 to 2013 is included in Table 3.4-16 and shown graphically for visual comparison.

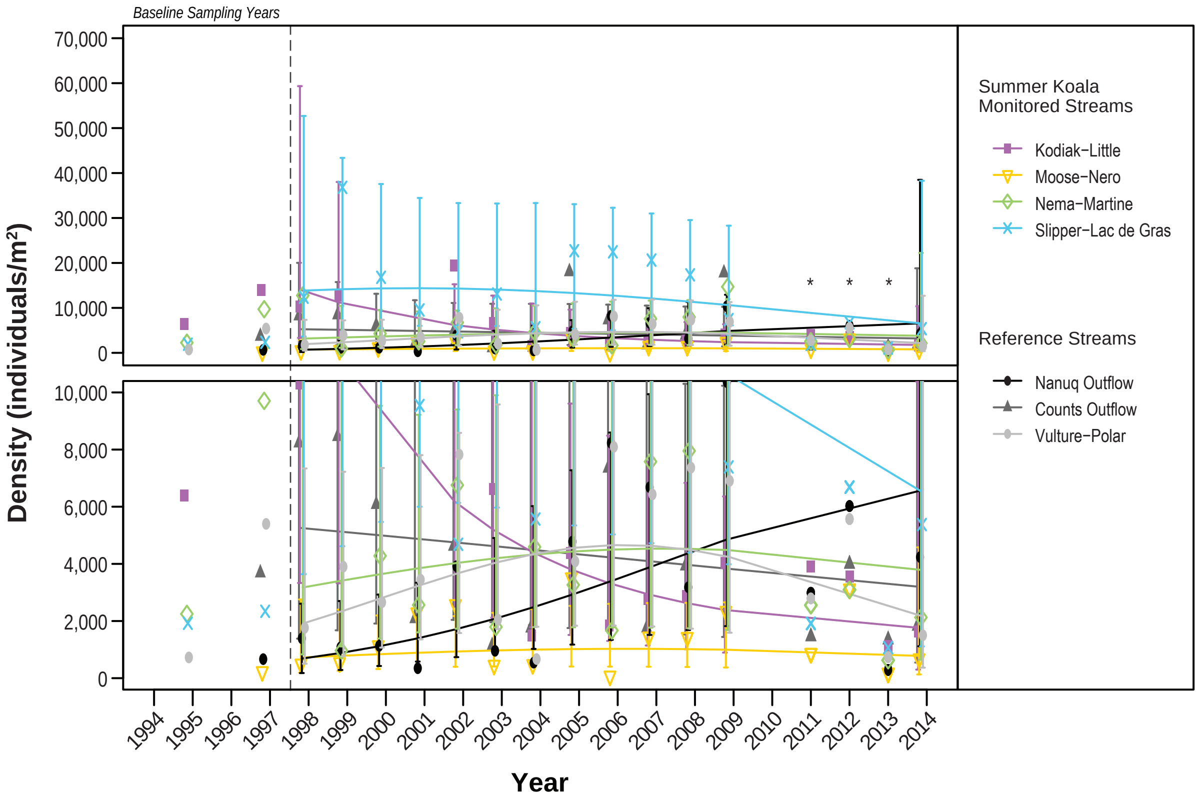
3.4.4.3 Results and Discussion

<u>Benthos Density</u>

At sites downstream from the LLCF, statistical and graphical analyses indicate that stream benthos density has remained stable over time, relative to reference streams (Table 3.4-17; Figure 3.4-23). Although mean stream benthos density in 2014 was greater than the mean baseline density ±2 SD in Moose-Nero Stream, a similar pattern was observed in one of the reference streams (i.e., Nanuq Outflow; Table 3.4-18). Thus, no mine effects were detected with respect to stream benthos density at sites downstream from the LLCF.

Figure 3.4-23

Observed and Fitted Means for Benthos Densities in Koala Watershed Streams, 1995 to 2014



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 values in 2011, 2012, and 2013 were not included in the statistical analysis. Observed means are plotted here for reference only.*

Table 3.4-16. 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
2014	Aug 1 – Sept 2	Aug 4 – Sept 4	Aug 1 – Sept 2	Aug 4 – Sept 5	Aug 2 – Sept 3	Aug 4 – Sept 5	Aug 4 – Sept 3

Notes: 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.
**Indicates 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.
§ Data were collected, but processed through the incorrect mesh size in the laboratory. Only EPT taxonomy data from 2011 and 2012 were used in the evaluation of effects.

Table 3.4-17. 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	
Benthos Density	-	LME	2	-	Kodiak-Little	-	1-452

Note: Dashes indicate not applicable.

Table 3.4-18. Mean ± 2 Standard Deviations (SD) Baseline Stream Benthos Density in Each of the Koala Watershed Lakes and Lac de Gras

Lake	Baseline Mean (N)	Mean Baseline Range , ± 2 SD	2014 Mean ± 1 SD
Nanuq Outflow	667 (1)	166 - 1,168	4,238 ± 5,753
Counts Outflow	3,685 (1)	0 – 8,242	1,820 ± 597
Vulture-Polar	2,479 (2)	0 – 8,551	1,520 ± 989
Kodiak-Little	9,240 (2)	0 – 23,851	1,629 ± 413
Moose-Nero	230 (1)	61 – 399	678 ± 203
Nema-Martine	5,044 (2)	0 – 13,208	2,127 ± 855
Slipper-Lac de Gras	2,079 (2)	0 – 5,623	5,375 ± 6,345

Notes: Units are organisms/m².
Negative values were replaced with zeros.
N = number of years data were collected.

Statistical and graphical analyses indicate that stream benthos density has decreased over time, relative to reference streams, in Kodiak-Little Stream (Table 3.4-17; Figure 3.4-23). The cause of the decline observed in Kodiak-Little Stream is unclear at this time, but may reflect historical effects as graphical analysis indicates that benthos density in Kodiak-Little Stream has declined from initially high levels in 1997 to 1999 (Figure 3.4-23).

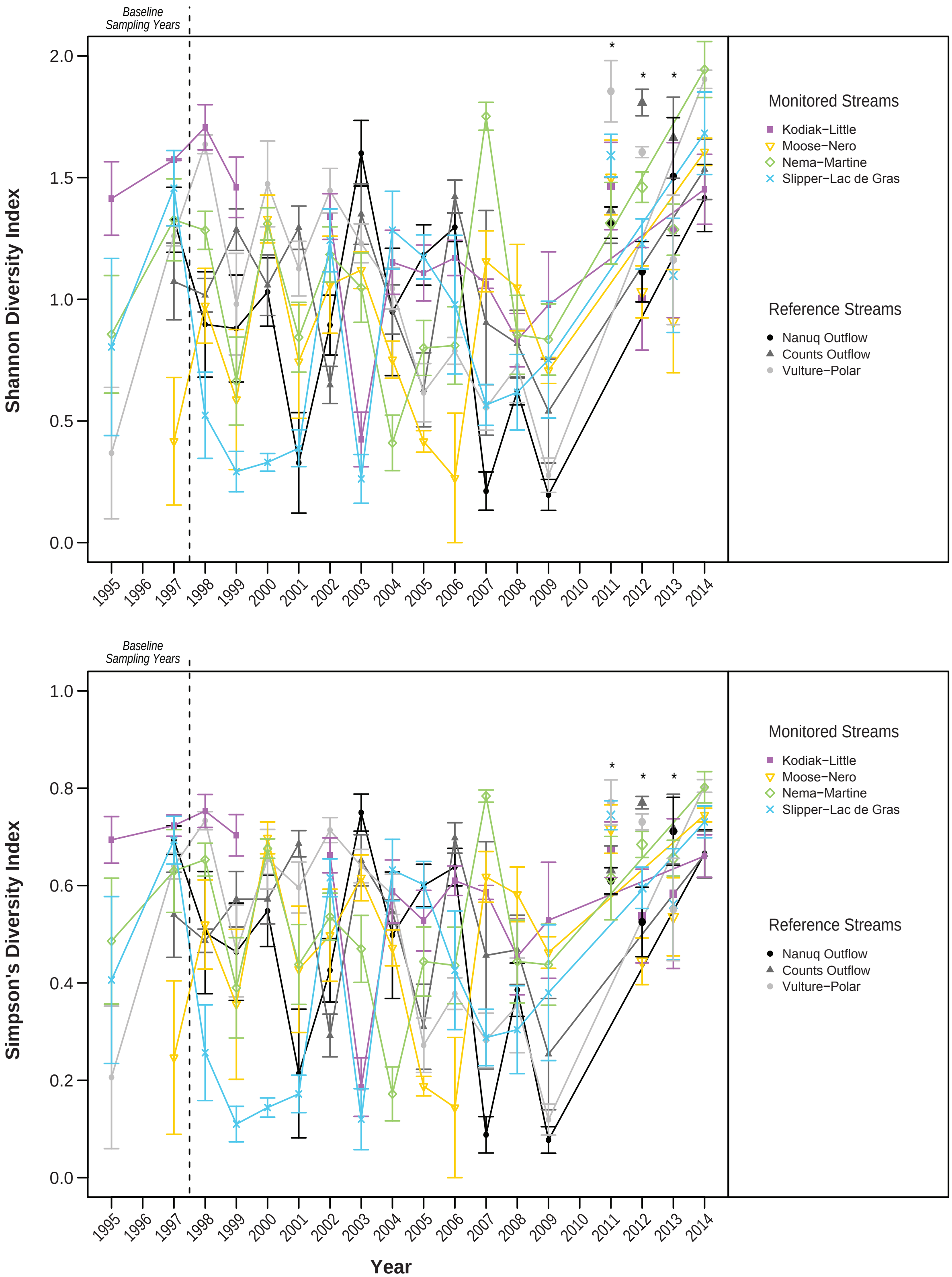
Dipteran Diversity and Community Composition

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 3.4-24) 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 3.4-25 to 3.4-28).

Both Shannon and Simpson's dipteran diversity indices have varied considerably through time in both monitored and reference streams since monitoring began (Figure 3.4-24). 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 3.4-24). In 2014, mean Shannon and Simpson's dipteran diversities were within ± 2 SD of mean baseline diversities in all cases except Moose-Nero Stream, in which Shannon diversity was greater (Table 3.4-19). The increased diversity observed in Moose-Nero Stream in 2014 is likely related to the greater density of organisms from the sub-family Tanypodinae (Figure 3.4-26a).

Figure 3.4-24

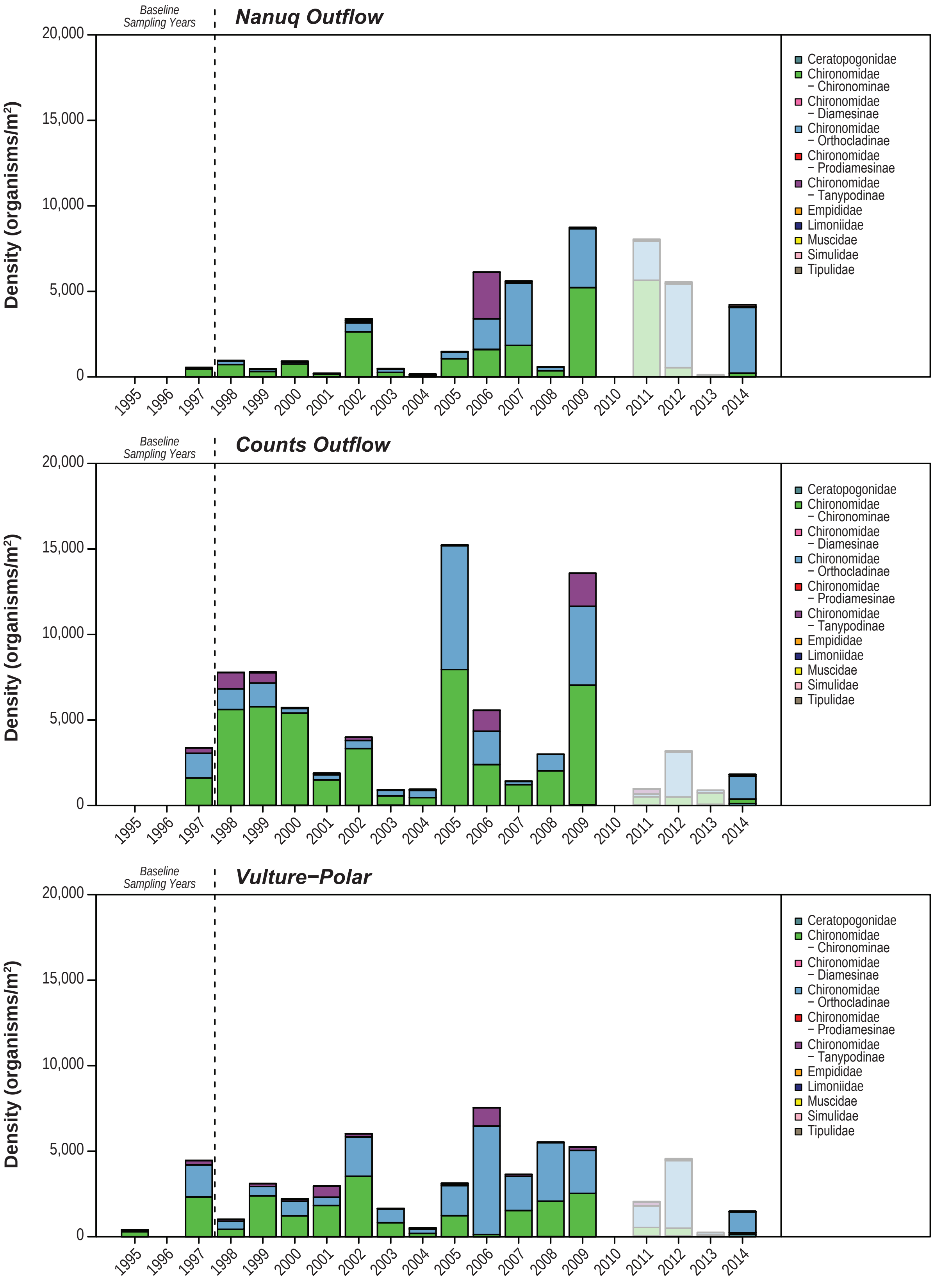
Average Diversity Indices for Benthic Dipterans in Koala Watershed Streams, 1995 to 2014



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 values in 2011, 2012, and 2013 were not included in the statistical analysis. Observed means are plotted here for reference only.*

Figure 3.4-25

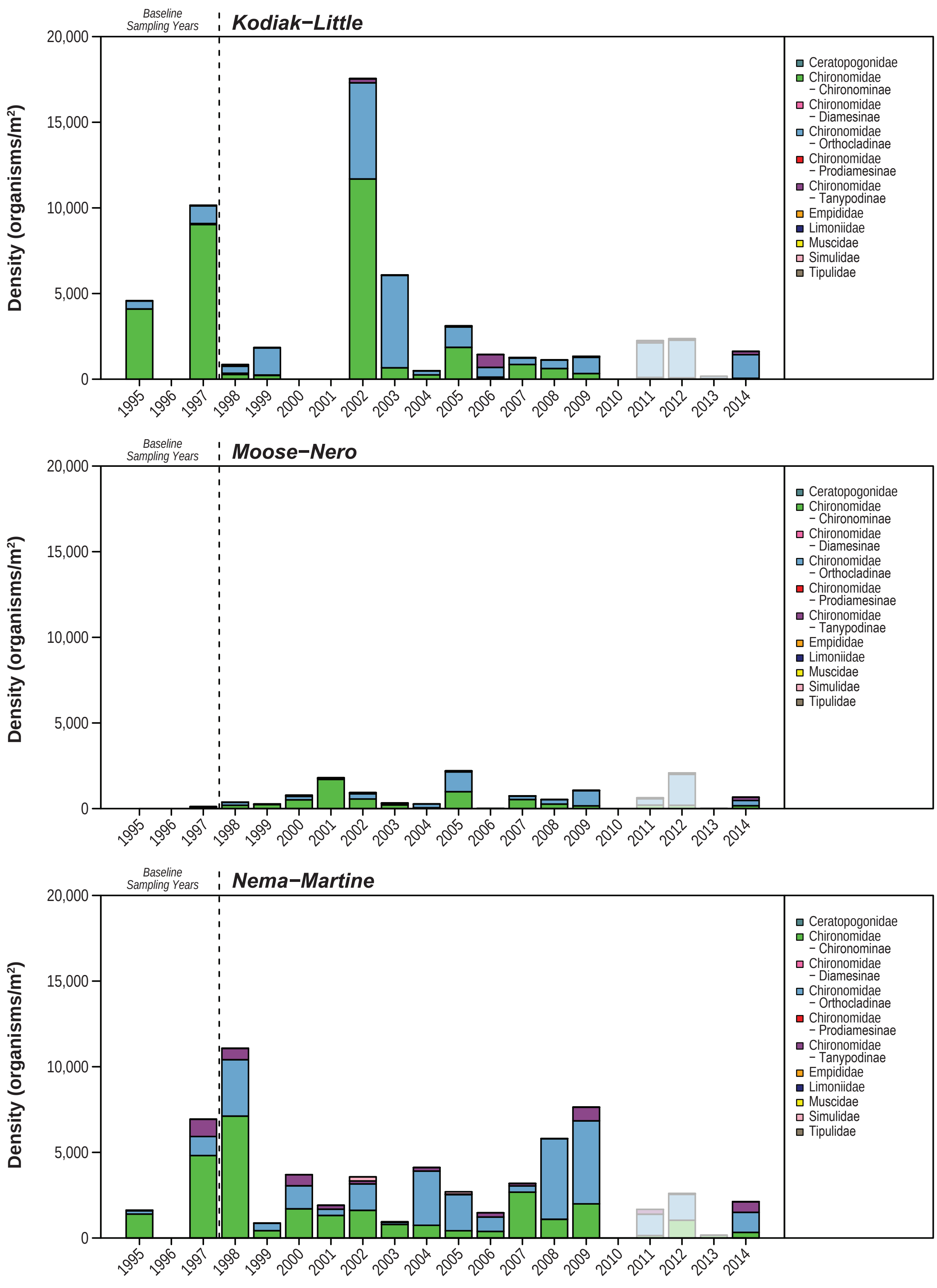
Average Benthic Dipteran Density by Taxonomic Group for AEMP Reference Streams, 1995 to 2014



Note: Density values in 2011, 2012, and 2013 were not considered in the evaluation of effects. Values are plotted here for reference only.

Figure 3.4-26a

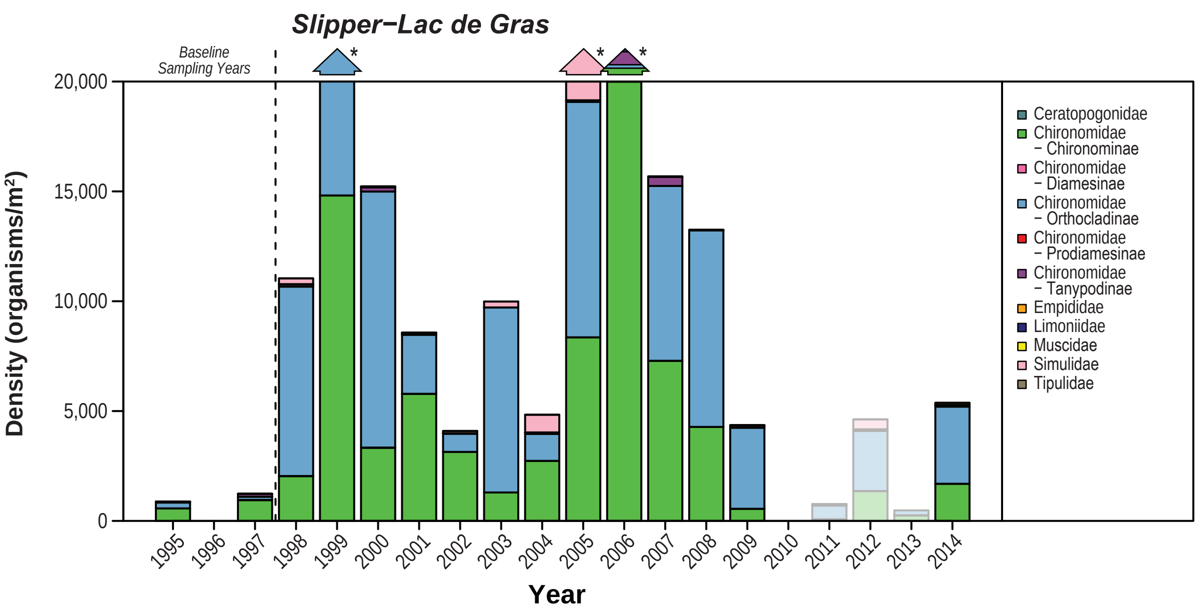
Average Benthic Dipteran Density by Taxonomic Group for Streams of the Koala Watershed, 1995 to 2014



Note: Density values in 2011, 2012, and 2013 were not considered in the evaluation of effects. Values are plotted here for reference only.

Figure 3.4-26b

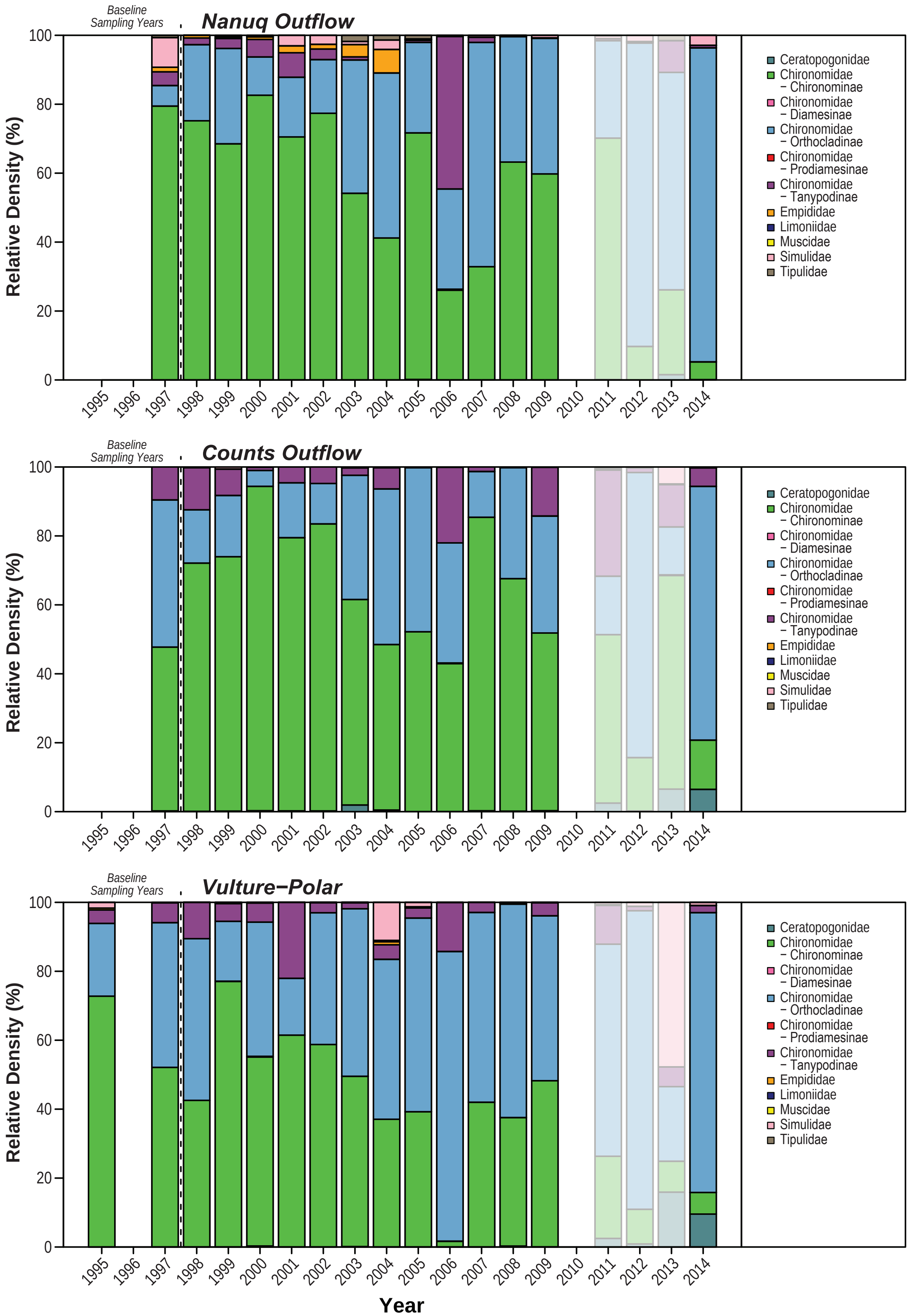
Average Benthic Dipteran Density by Taxonomic Group for Streams of the Koala Watershed, 1995 to 2014



*Notes: *1999 total density = 34,948, Orthocladiinae = 20,008;*
2005 total density = 21,524, Simulidae = 2,378;
2006 total density = 21,396, Chironiminae = 20, 438, Orthocladiinae = 180, Tanypodinae = 669, Limoniidae = 67, Tiplulidae = 35
Density values in 2011, 2012, and 2013 were not considered in the evaluation of effects. Values are plotted here for reference only.

Figure 3.4-27

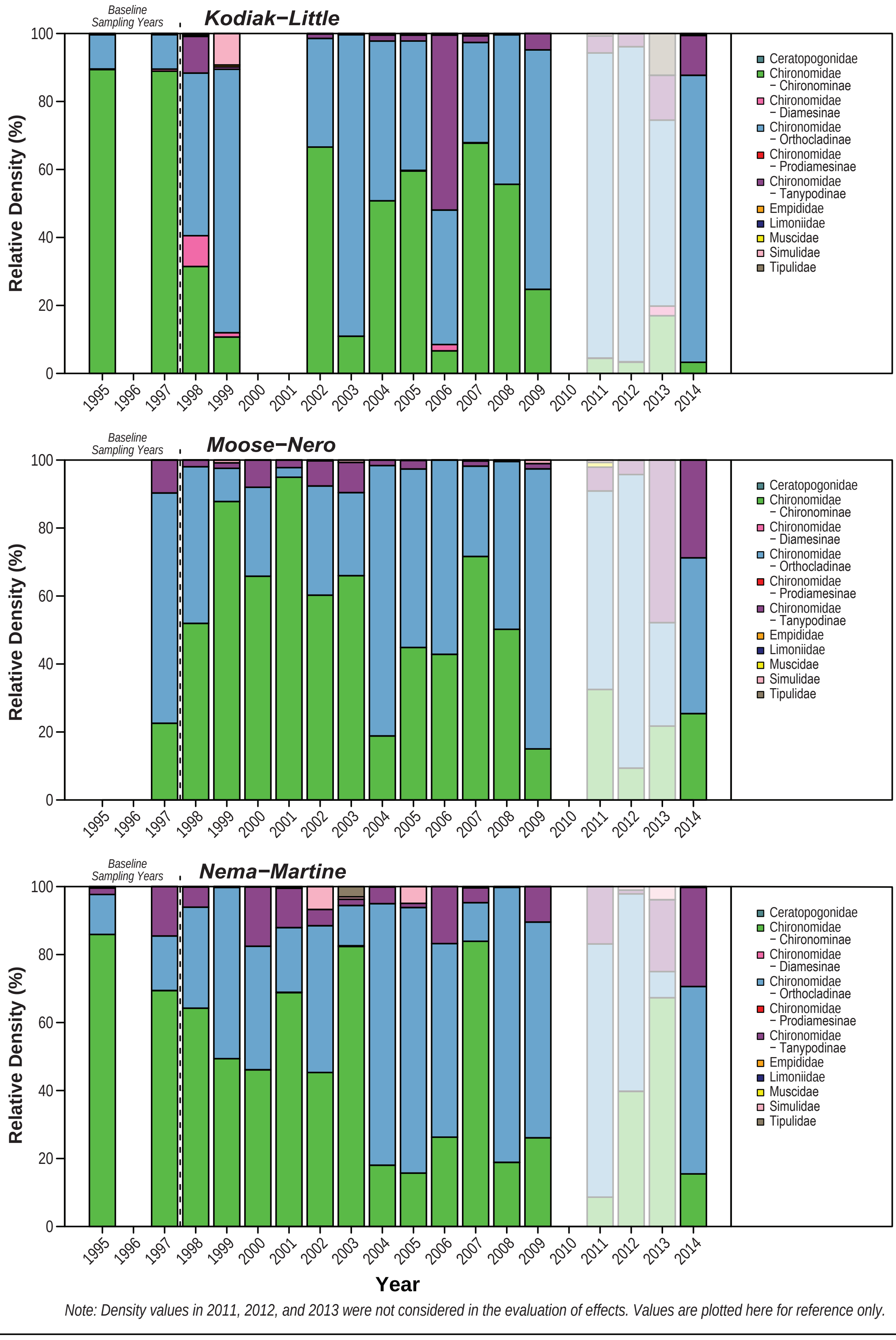
Relative Densities of Benthic Dipteran Taxa in AEMP Reference Streams, 1995 to 2014



Note: Density values in 2011, 2012, and 2013 were not considered in the evaluation of effects. Values are plotted here for reference only.

Figure 3.4-28a

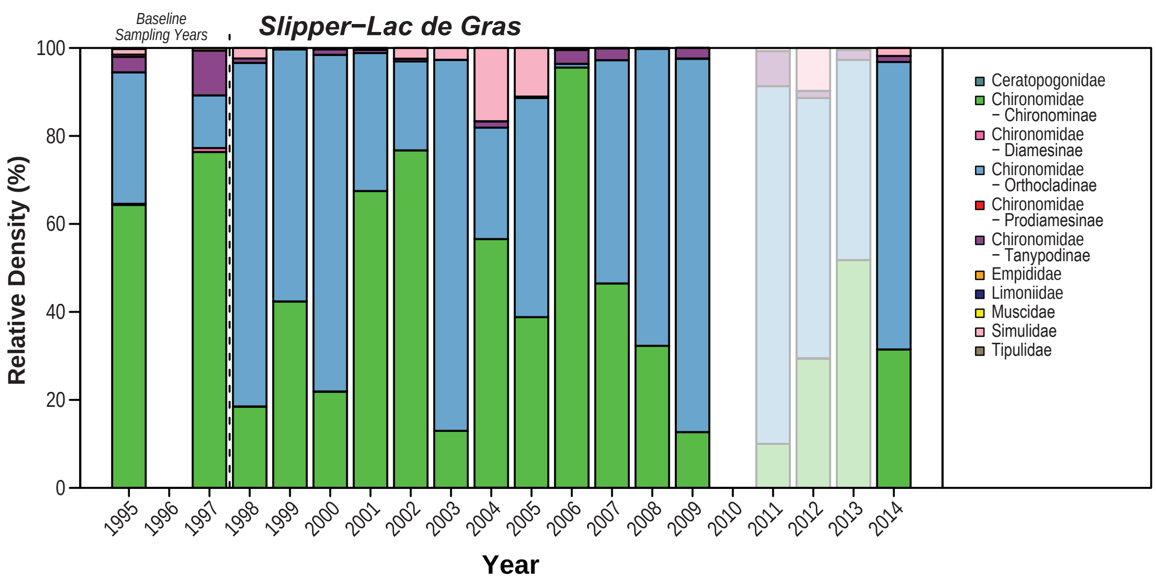
Relative Densities of Benthic Dipteran Taxa in Streams of the Koala Watershed, 1995 to 2014



Note: Density values in 2011, 2012, and 2013 were not considered in the evaluation of effects. Values are plotted here for reference only.

Figure 3.4-28b

Relative Densities of Benthic Dipteran Taxa in Streams of the Koala Watershed, 1995 to 2014



Note: Density values in 2011, 2012, and 2013 were not considered in the evaluation of effects. Values are plotted here for reference only.

Table 3.4-19. Mean ± 2 Standard Deviations (SD) Baseline Dipteran Diversity in Each of the Koala Watershed Streams

	Shannon Diversity			Simpson's Diversity		
Stream	Baseline Mean (N)	Mean Baseline Range ± 2 SD	2014 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range ± 2 SD	2014 Mean ± 1 SD
Nanuq Outflow	1.33 (1)	0.68 - 1.79	1.42 ± 0.31	0.69 (1)	0.59 – 0.79	0.67 ± 0.11
Counts Outflow	1.07 (1)	0.53 – 1.62	1.53 ± 0.28	0.54 (1)	0.24 – 0.84	0.66 ± 0.11
Vulture-Polar	0.70 (2)	0 – 2.00	1.90± 0.08	0.37 (2)	0 - 1	0.80 ± 0.03
Kodiak-Little	1.47 (2)	0.94 – 2.01	1.45 ± 0.32	0.71 (2)	0.54 – 0.87	0.66 ± 0.10
Moose-Nero	0.42 (1)	0 – 1.32	1.61 ± 0.13	0.25 (1)	0 – 0.79	0.74 ± 0.03
Nema-Martine	1.03 (2)	0.03 – 2.03	1.94 ± 0.26	0.54 (2)	0.05 – 1	0.80 ± 0.07
Slipper-Lac de Gras	1.05 (2)	0 – 2.48	1.68 ± 0.38	0.51 (2)	0 – 1	0.73 ± 0.07

Notes: 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.

The relative densities of dipteran taxonomic groups have been fairly consistent through time in all monitored and reference streams (Figures 3.4-25 to 3.4-28). However, there was some evidence of a trend toward greater densities of organisms from the sub-family Orthocladiinae and lesser densities of organisms from the sub-family Chironominae through time (Figures 3.4-25 to 3.4-28). 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 3.4-25 to 3.4-28).

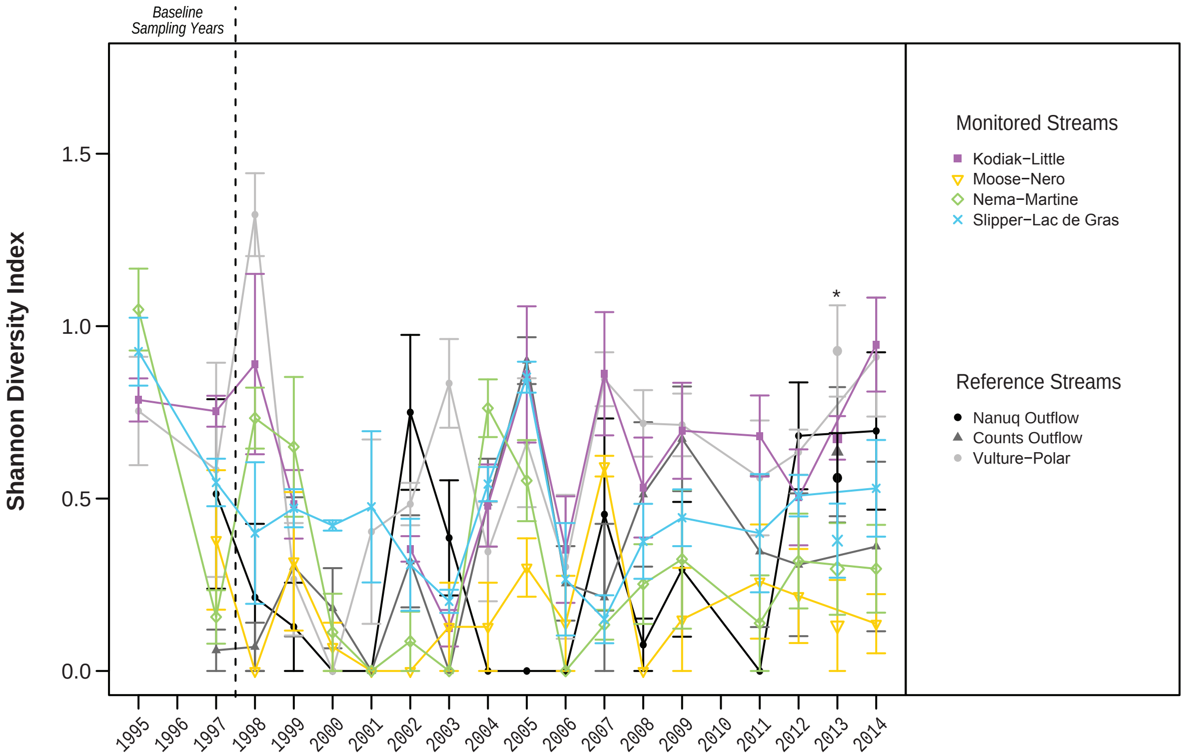
In general, most of the dipteran taxa present belong to the family Chironomidae. Chironomidae are often found in large numbers in freshwater systems as they have a variety of adaptations that allow them to live in a wide variety of environments. The subfamily Chironominae is a particularly diverse and 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.

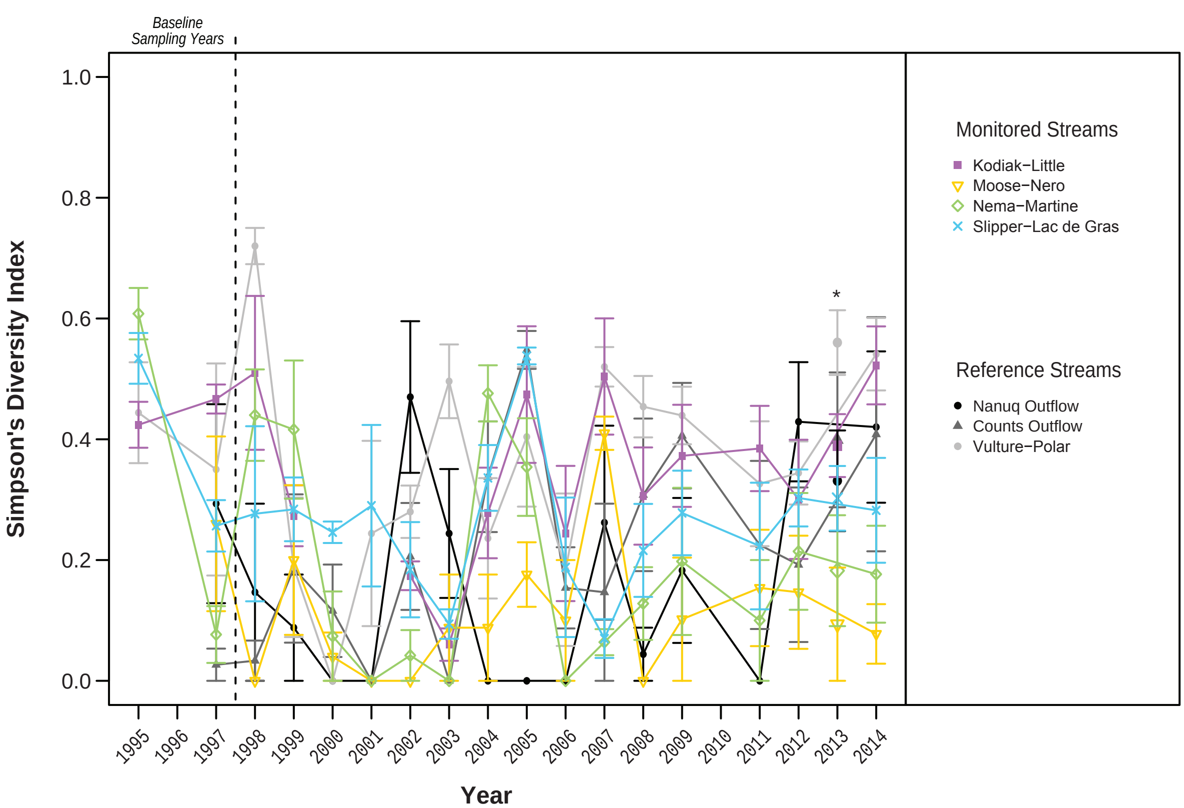
EPT Diversity and Community Composition

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 3.4-29) 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 3.4-30 to 3.4-33).

Figure 3.4-29

Average Diversity Indices for Benthic EPT Taxa in Koala Watershed Streams, 1995 to 2014

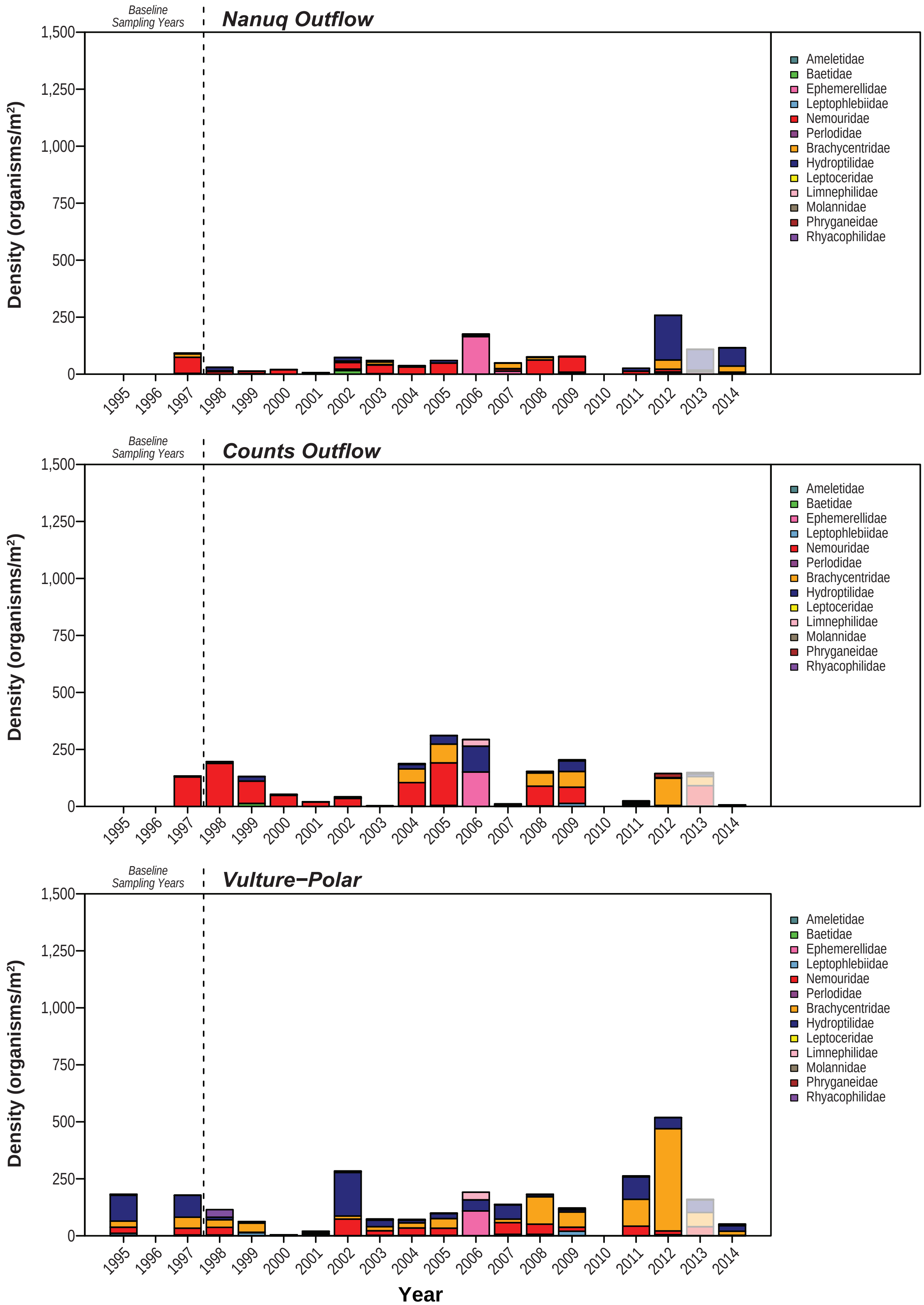




Notes: Symbols represent observed mean values. Error bars indicate standard error of the observed means.
**Diversity indices in 2013 were not included in the evaluation of effects. Means and standard errors are plotted here for reference only.*

Figure 3.4-30

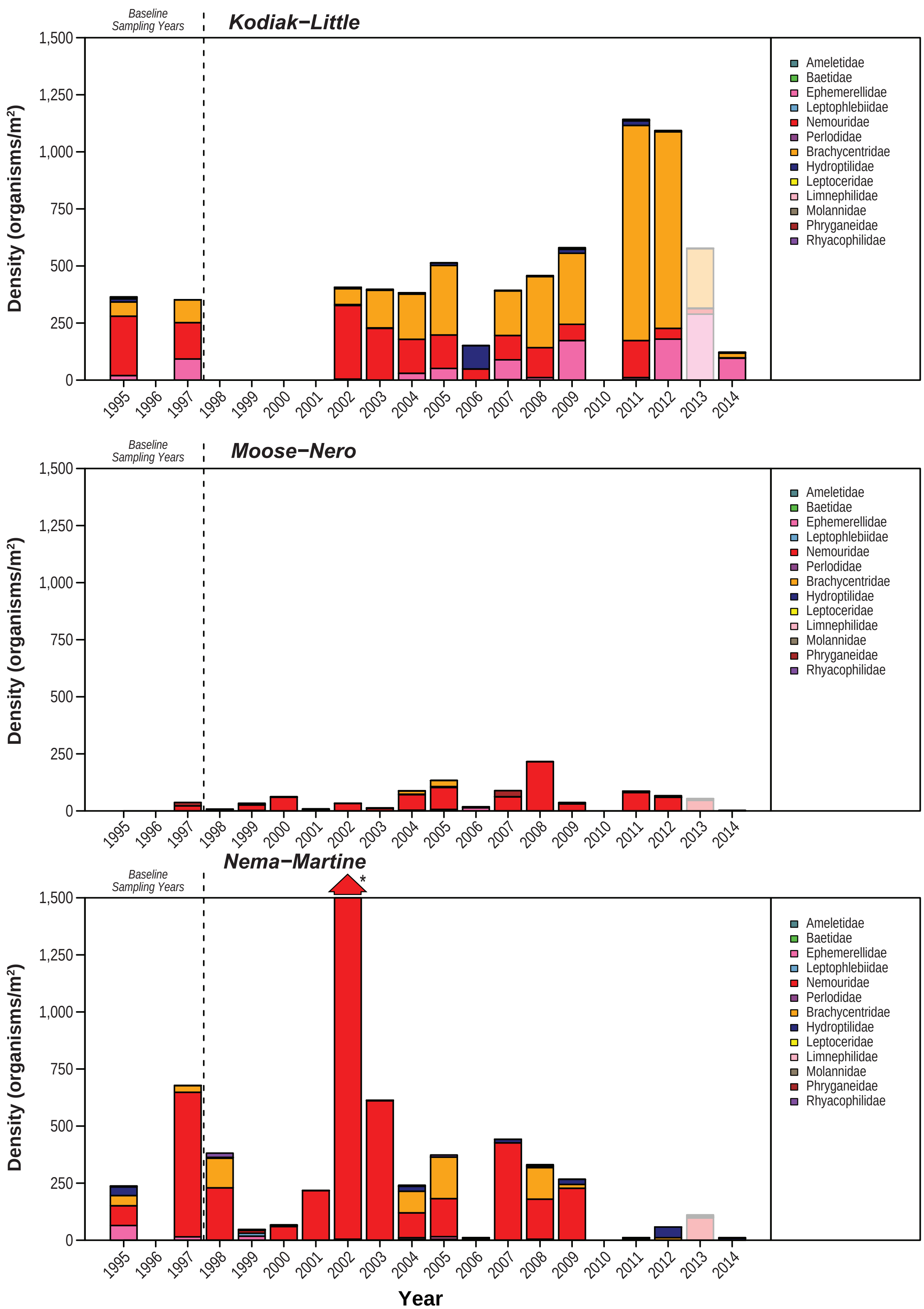
Average Benthic EPT Density by Taxonomic Group for AEMP Reference Streams, 1995 to 2014



Note: Density values in 2013 were not considered in the evaluation of effects. Values are plotted here for reference only.

Figure 3.4-31a

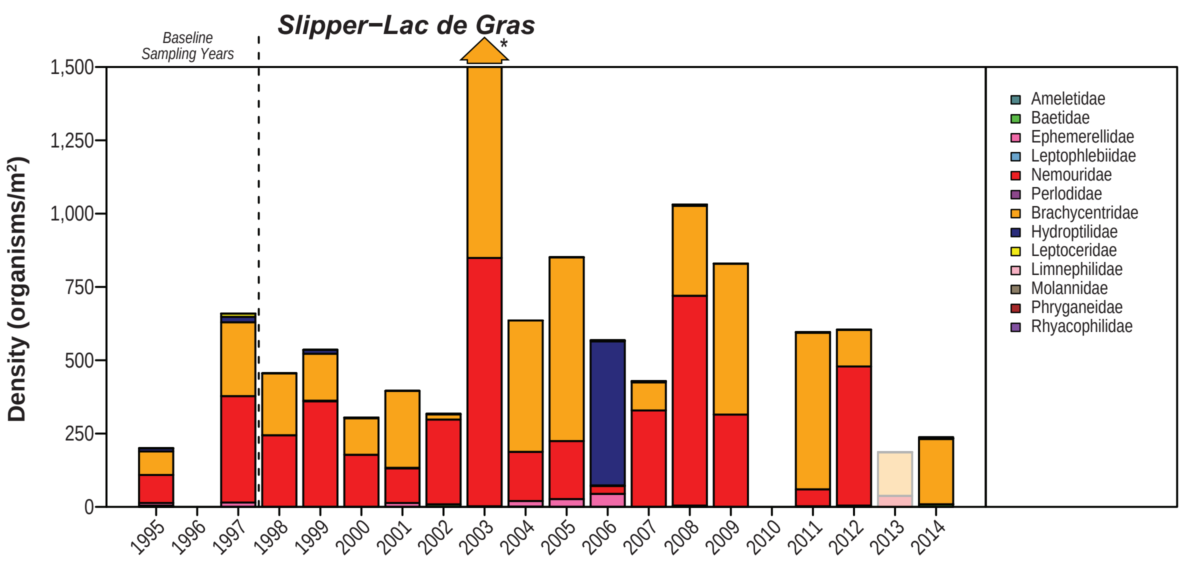
Average Benthic EPT Density by Taxonomic Group for Streams of the Koala Watershed, 1995 to 2014



Notes: 2002 total density = 1,956, Nemouridae = 1,938, Brachycentridae = 7, Hydroptilidae = 7.
Density values in 2013 were not considered in the evaluation of effects. Values are plotted here for reference only.

Figure 3.4-31b

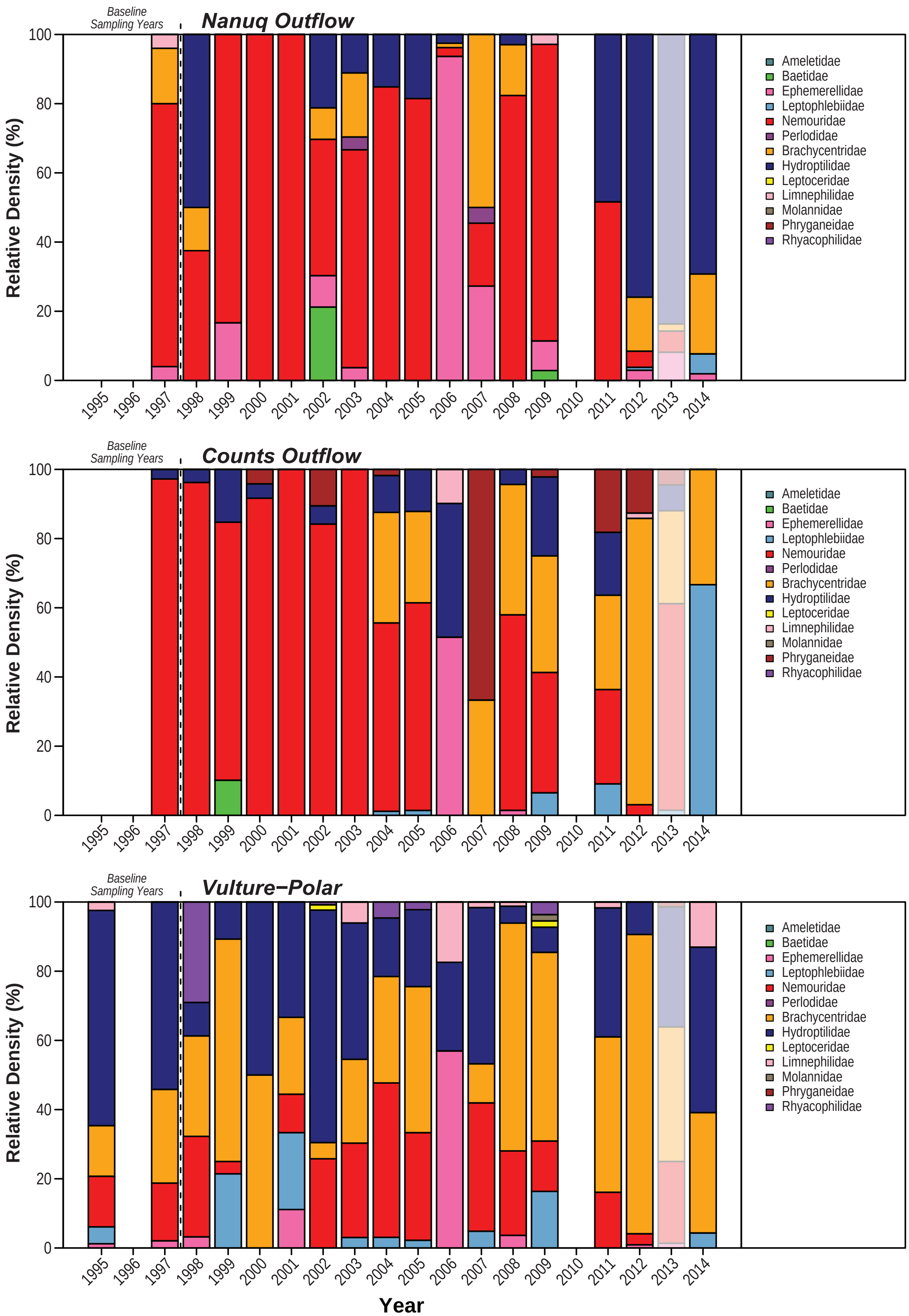
Average Benthic EPT Density by Taxonomic Group for Streams of the Koala Watershed, 1995 to 2014



Notes: 2003 total density = 2,980, Brachycentridae = 2,129, Rhyacophilidae = 2.
Density values in 2013 were not considered in the evaluation of effects. Values are plotted here for reference only.

Figure 3.4-32

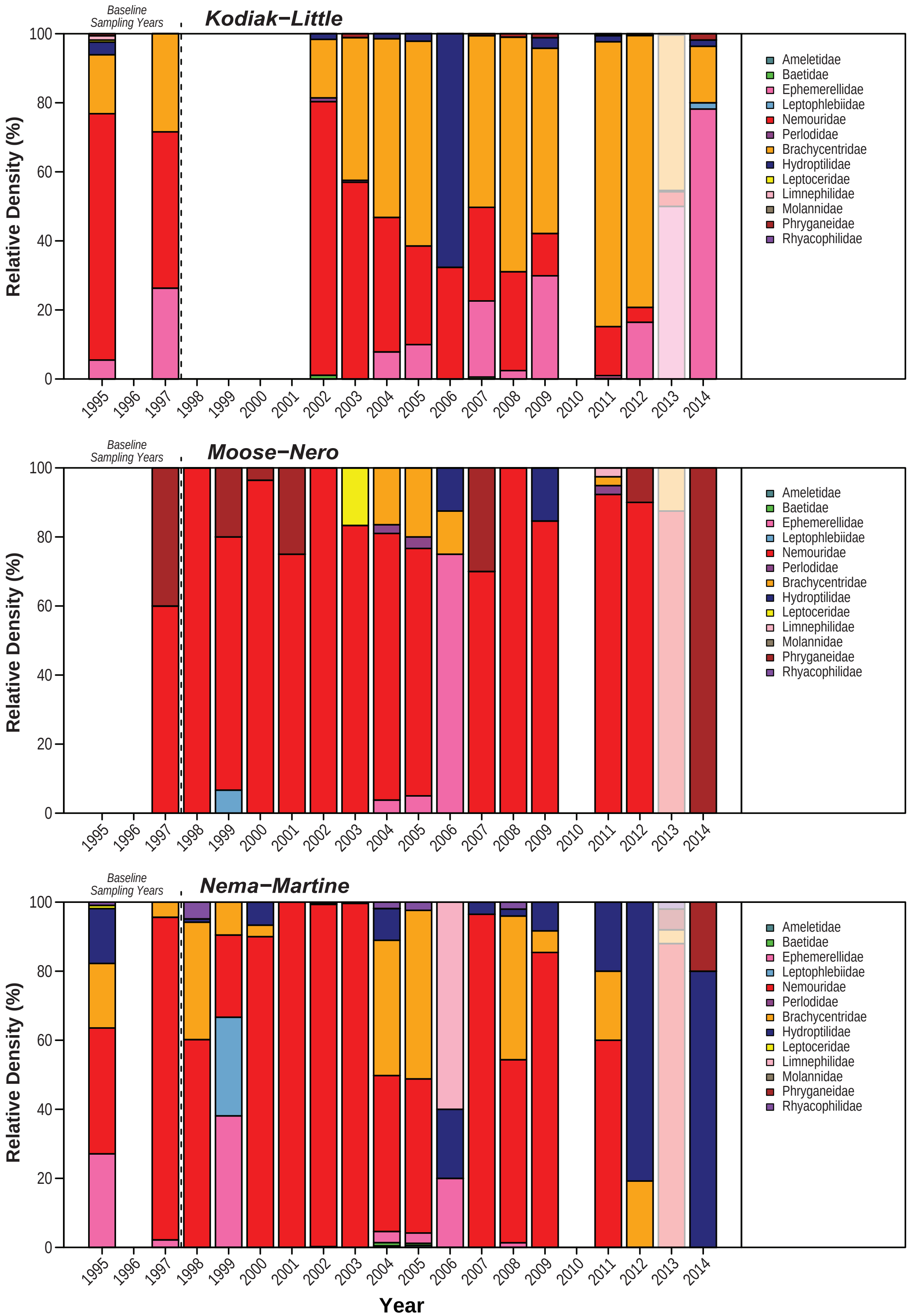
Relative Densities of Benthic EPT Taxa in AEMP Reference Streams, 1995 to 2014



Note: Density values in 2013 were not considered in the evaluation of effects. Values are plotted here for reference only.

Figure 3.4-33a

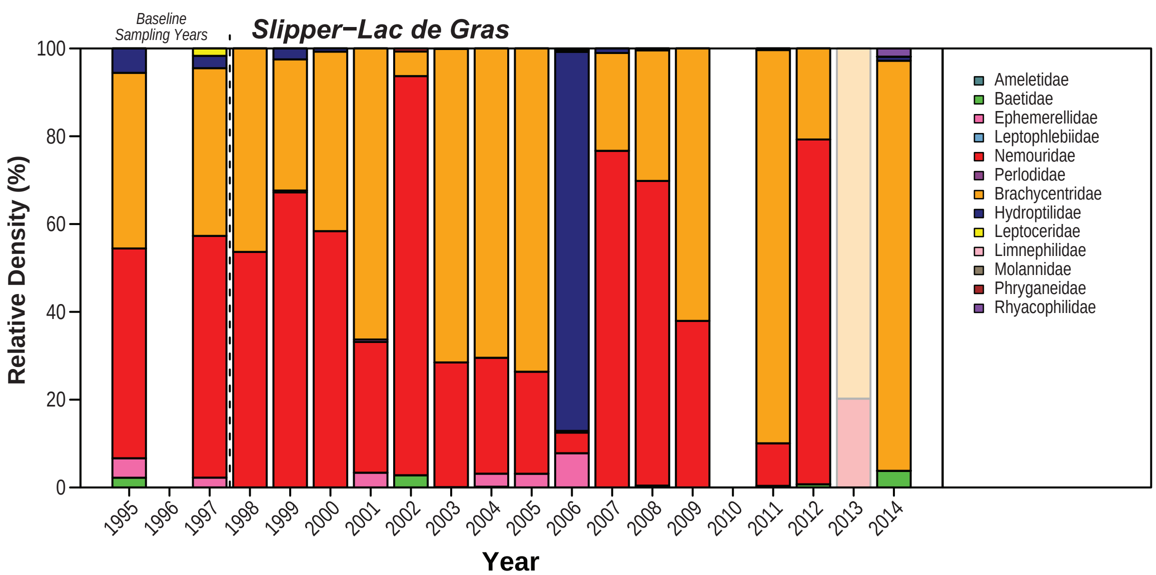
Relative Densities of Benthic EPT Taxa in Streams of the Koala Watershed, 1995 to 2014



Note: Density values in 2013 were not considered in the evaluation of effects. Values are plotted here for reference only.

Figure 3.4-33b

Relative Densities of Benthic EPT Taxa in Streams of the Koala Watershed, 1995 to 2014



Note: Density values in 2013 were not considered in the evaluation of effects. Values are plotted here for reference only.

Both Shannon and Simpson's EPT diversity indices have varied considerably through time in both monitored and reference streams since monitoring began (Figure 3.4-29). Still, graphical analysis suggests that both Shannon and Simpson's EPT diversity were within the range of values observed in baseline years in all monitored streams in 2014 (Figure 3.4-29). Mean EPT diversities in 2014 were also within ± 2 SD of baseline means in all monitored streams (Table 3.4-20). 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 3.4-30 to 3.4-33). The one possible exception is Kodiak-Little, in which there is some evidence that Nemouridae have been replaced by Brachycentridae through time (Figures 3.4-31a and 3.4-33a). However, a similar pattern was observed in Counts Outflow (Figures 3.4-30 and 3.4-32). Thus, no mine effects were detected with respect to EPT diversity or taxonomic composition.

Table 3.4-20. 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	2014 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD
Nanuq Outflow	0.51 (1)	0 – 1.47	0.70 ± 0.51	0.29 (1)	0 – 0.86	0.42 ± 0.28
Counts Outflow	0.06 (1)	0 – 0.27	0.36 ± 0.55	0.03 (1)	0 – 0.12	0.41 ± 0.43
Vulture-Polar	0.69 (2)	0 – 1.49	0.91 ± 0.39	0.41 (2)	0 – 0.85	0.54 ± 0.13
Kodiak-Little	0.77 (2)	0.54 – 1.00	0.95 ± 0.30	0.44 (2)	0.30 – 0.58	0.52 ± 0.14
Moose-Nero	0.38 (1)	0 – 1.08	0.14 ± 0.19	0.26 (1)	0 – 0.76	0.08 ± 0.11
Nema-Martine	0.71 (2)	0 – 1.73	0.30 ± 0.28	0.41 (2)	0 – 0.98	0.18 ± 0.18
Slipper-Lac de Gras	0.78 (2)	0.25 – 1.31	0.53 ± 0.31	0.43 (2)	0.10 – 0.76	0.28 ± 0.19

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

3.4.5 Aquatic Biology Summary

Six changes in biological variables were observed in 2014:

- Altered phytoplankton genera diversity in Leslie and Kodiak lakes, though Leslie Lake diversity returned to historical levels in 2013;
- Altered taxonomic composition of phytoplankton assemblages in lakes downstream of the LLCF as far as site S2 in Lac de Gras and in Kodiak Lake;
- Decreased zooplankton diversity in lakes downstream of the LLCF as far as Nema Lake, though diversity has increased in recent years;
- Altered taxonomic composition of zooplankton assemblages in Leslie, Moose, and Nema lakes;
- 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; and
- Decreased benthos density in Kodiak-Little Stream.

Phytoplankton diversity has been stable through time in all monitored lakes of the Koala Watershed and Lac de Gras, except Leslie and Kodiak lakes. Phytoplankton diversity in Leslie Lake decreased from 2006 to 2011, but returned to historical levels by 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 increases in nitrate-N concentrations following the onset and subsequent expansion of underground mining operations in 2002; these changes in nitrate-N concentration also show a spatial gradient with downstream distance from the LLCF (see Section 3.2.4.9). In lakes that are not downstream of the LLCF, a recent trend of potentially decreasing diversity has been observed in Kodiak Lake. In contrast to lakes downstream of the LLCF, Kodiak Lake has shown a recent increase in the densities of blue-green algae with a corresponding decrease in diatoms and Chlorophyceae (green algae). This shift from diatoms and green algae to blue-green algae may reflect the decreasing trend in nitrate-N concentrations observed in Kodiak Lake (see Section 3.2.4.9).

Examination of species tolerances with respect to current water quality and sediment 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 2012c). Accumulating research suggests that the ratio of available elements, especially macronutrients like carbon (C), nitrogen (N), and phosphorus (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 downstream from the LLCF. 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 phosphorus 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). In fulfilment of Part J, Item 11 of W2012L2-0001, DDEC's Nitrogen Response Plan v1.1 was approved by the WLWB on August 11, 2014. This plan was designed to describe current nitrogen sources and management practices, assess current blasting practices at the Ekati Diamond Mine via an audit conducted by appropriate experts and address recommendations from the audit report. Key findings from the audit indicate that DDEC has many positive practices in place to contain, handle, use and dispose of explosives. Moreover, many of the recommendations made in a 2008 blast audit have been incorporated into standard operating procedures on site. The report concludes that the most significant area of potential for minimizing the availability of nitrogen for dissolution into minewater, and subsequent release to the receiving environment, is through improved usage practices in the open pits. In 2014, DDEC worked actively towards fulfilling the commitments made in the Implementation plan. A full update on the Nitrogen Response Plan (NRP) is available in Section 8 of the Summary Report. Although water quality modelling predicts that nitrate concentrations will continue to increase in the LLCF and Koala Watershed lakes downstream of the LLCF (Rescan 2012h), results suggest that nitrogen concentrations have remained stable in 2014 (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 3.4-4 and 3.4-7). Although Chlorophyceae densities in 2014 have decreased in Nema and Slipper lakes, they remain elevated in Leslie Lake (Figures 3.4-4 and 3.4-7). This second shift in primary producer community composition may be explained by the addition of phosphorus to Cell D of the LLCF from 2009 to 2011 as an adaptive management response to increased nitrate concentrations (Rescan 2011d). The increase in Chlorophyceae observed further downstream, in Nema and Slipper lakes, in 2013, may reflect a spatiotemporal lag in the effect of phosphorus 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 2012c, 2012f). However, there was no evidence that elevated potassium concentrations have led to declines in the density of the most sensitive species (*Daphnia magna*; see Section 3.4.2). 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, Moose and Nema lakes in recent years, the zooplankton communities remain dominated by copepods. 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 2012c). In these lakes, the relative densities of organisms from the Chironomidae subfamily Orthocladiinae (likely from the genera *Heterotanytarsus, Rheocricotopus* and *Psectrocladius*) have decreased, while densities of Diamesinae (most likely organisms from the genus *Protanypus*), Prodiamesinae (most likely organisms from the genus *Monodiamesa*), Chironominae (most likely organisms from the genera *Cladotanytarsus*, *Paratanytarsus* and *Stempellinella*) and/or Tanypodinae (most likely organisms from the genera *Procladius* and *Ablabesmyia*) 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, Zalutschia, and Heteroctrissocladius*) 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.

At sites downstream from the LLCF, no mine effects were detected with respect to stream benthos density, dipteran diversity, EPT diversity, or dipteran community composition. A decrease in benthos density was observed in Kodiak-Little Stream. The cause of the decline observed in Kodiak-Little Stream is unclear at this time, but may reflect historical effects as graphical analysis indicates that benthos density in Kodiak-Little Stream has declined from initially high levels. The only water quality parameter that has changed through time in Kodiak-Little Stream is total nickel; however, concentrations have remained below the hardness-dependent nickel CCREM guideline value (CCREM 1987).

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 strong mine effects on monitored fish populations in the Koala Watershed (Rescan 2012c). 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.

3.5 SUMMARY

Table 3.5-1 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.

Table 3.5-1. **Summary of Evaluation of Effects for the Koala Watershed and Lac de Gras**

Variable	Change Downstream of LLCF?	Change Upstream of LLCF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Physical Limnology							
Under-ice Temperature Profiles	Yes	Yes, Historical	Leslie, Moose, Nema, and Slipper lakes; Grizzly Lake; Kodiak Lake	Cooling in Leslie, Moose; Nema, and Slipper; surface cooling in Grizzly; surface warming in Kodiak Lake	Unknown downstream of LLCF and Grizzly Lake; aerators in Kodiak Lake	No downstream of LLCF, Historical in Kodiak Lake	Cooling trends also observed in some reference lakes during the ice-covered season; In Kodiak Lake, current temperature profiles likely represent undisturbed conditions
Under-ice DO Profiles	No	Yes, Historical	Kodiak Lake	Decreased concentrations at shallower depths	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 in Kodiak Lake likely represent undisturbed conditions.
August Secchi Depths	No	No	-		-	No	-
Lake and Stream Water Quality							
pH	Yes	Yes	Downstream to site S3; Grizzly	Increase	LLCF; unknown in Grizzly	Yes downstream of LLCF; Unlikely in Grizzly	The lower confidence interval for Grizzly Lake was below the lower CCREM guideline value during the ice-covered season; similar pattern observed in reference lakes.
Total Alkalinity	Yes	No	Downstream to site S2	Increase	LLCF	Yes	-
Hardness	Yes	No	Downstream to site S3	Increase	LLCF	Yes	Concentrations have stabilised as far as Slipper Lake since 2006.
Chloride	Yes	No	Downstream to site S3	Increase	LLCF	Yes	All 2014 concentrations less than the SSWQO.

(continued)

Table 3.5-1. Summary of Evaluation of Effects for the Koala Watershed and Lac de Gras (continued)

Variable	Change Downstream of LLCF?	Change Upstream of LLCF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Lake and Stream Water Quality (*cont'd*)							
Sulphate	Yes	Yes	Downstream to site S3; Kodiak Lake	Increase	LLCF; Unknown in Kodiak	Yes downstream of LLCF; Unlikely in Kodiak	All 2014 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; Concentrations in both deep water samples from Leslie Lake during the open water season exceeded the SSWQO.
Total Ammonia-N	Yes	No	In lakes downstream to Slipper Lake	Increase	LLCF	Yes	Trend is less clear in streams than in lakes; 95% confidence interval around the 2014 fitted mean exceeded the CCME guideline value in Counts Lake during the ice-covered season. Concentrations have stabilised in recent years as far as Nema Lake.
Nitrite-N	Yes	No	Downstream to Moose-Nero Stream	Increase	LLCF	Yes	All 2014 concentrations less than the CCREM guideline.
Nitrate-N	Yes	No	Downstream to Slipper Lake	Increase	LLCF	Yes	All 2014 concentrations less than the SSWQO. Concentrations have stabilised in recent years as far as Slipper Lake.

(continued)

Table 3.5-1. Summary of Evaluation of Effects for the Koala Watershed and Lac de Gras (continued)

Variable	Change Downstream of LLCF?	Change Upstream of LLCF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Lake and Stream Water Quality (*cont'd*)							
Total Phosphate-P	Yes	No	Downstream to Moose Lake	Increase	LLCF	Yes	The 2014 upper 95% confidence interval around fitted mean in Leslie, Moose, Nema, Slipper, and Nanuq during the open water season exceeded the CCME trigger range or benchmark values; The observed and fitted mean in Moose and Nanuq lakes during the ice-covered season and in Counts Lake during the open water season exceeded the benchmark value.
Total Organic Carbon	Yes	No	Downstream to Slipper Lake	Increase	Unknown	No	No clear spatial gradient and no baseline data for comparison
Total Antimony	Yes	No	Downstream to Nema-Martine Stream	Increase	LLCF	Yes	All 2014 concentrations less than the water quality benchmark. Concentrations in Leslie and Moose lakes have stabilised since about 2006.
Total Arsenic	Yes	No	Downstream to Nema Lake	Increase	LLCF	Yes	All 2014 concentrations less than the CCME guideline.
Total Barium	Yes	No	Downstream to Slipper-Lac de Gras	Increase	LLCF	Yes	All 2014 concentrations less than the water quality benchmark. Concentrations as far as Nema-Martine Stream have stabilised since 2007.
Total Boron	Yes	No	Downstream to Slipper-Lac de Gras	Increase	LLCF	Yes	All 2014 concentrations less than the CCME guideline.
Total Cadmium	No	No	-	-	-	No	The concentration in one sample from Moose-Nero Stream in June was greater than the hardness-dependent CCME guideline.
Total Molybdenum	Yes	No	Downstream to site S3	Increase	LLCF	Yes	All 2014 concentrations less than the SSWQO. Concentrations as far as Nema Lake have stabilised in recent years.

(continued)

Table 3.5-1. Summary of Evaluation of Effects for the Koala Watershed and Lac de Gras (continued)

Variable	Change Downstream of LLCF?	Change Upstream of LLCF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Lake and Stream Water Quality (*cont'd*)							
Total Nickel	Yes	Yes	Downstream to Slipper-Lac de Gras; Kodiak Lake and Kodiak-Little Stream	Increase	LLCF; unknown in Kodiak Lake and Kodiak-Little Stream	Yes downstream of LLCF; Unlikely in Kodiak Lake and Kodiak-Little Stream	All 2014 concentrations less than the CCME guideline. Concentrations as far as Nema Lake have stabilised in recent years.
Total Selenium	Yes	No	Downstream to Moose Lake	Increase	LLCF	Yes	All 2014 concentrations less than the CCME guideline.
Total Strontium	Yes	No	Downstream to site S3	Increase	LLCF	Yes	All 2014 concentrations less than the water quality benchmark.
Total Uranium	Yes	No	Downstream to Slipper Lake	Increase	LLCF	Yes	All 2014 concentrations less than the CCME guideline.
Total Vanadium	No	No	-	-	-	-	All 2014 concentrations less than the SSWQO.
Sediment Quality							
TOC	No	No	-	-	-	No	-
Available Phosphorus	No	No	-	-	-	No	-
Total Nitrogen	No	No	-	-	-	No	-
Total Antimony	Possibly	No	Downstream to Slipper Lake	Possible increase	LLCF	Possibly	No statistical analyses possible at this time. Observed means at monitored sites downstream of the LLCF, as far as Slipper Lake, were greater than observed means at reference sites, with a pattern of decreasing concentration with increasing distance from the LLCF.

(continued)

Table 3.5-1. **Summary of Evaluation of Effects for the Koala Watershed and Lac de Gras (continued)**

Variable	Change Downstream of LLCF?	Change Upstream of LLCF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Sediment Quality (*cont'd*)							
Total Arsenic	No	No	-	-	-	No	The observed mean exceeded the CCME ISQG in all monitored sites and the observed mean or the 95% confidence interval around the fitted mean exceeded the CCME PEL in all monitored sites, except Nema Lake, in 2014. Similar pattern observed in all three reference lakes.
Total Cadmium	No	No	-	-	-	No	The 95% confidence intervals around the fitted mean exceeded the CCME ISQG in Slipper Lake and at site S2 in 2014; however, a similar pattern was observed in one reference lake. All 2014 concentrations less than the CCME PEL.
Total Molybdenum	Yes	No	Downstream to Slipper Lake	Increase	LLCF	Yes	-
Total Nickel	No	No	-	-	-	No	-
Total Phosphorus	No	No	-	-	-	No	-
Total Selenium	No	No	-	-	-	No	-
Total Strontium	Possibly	Possibly	Downstream to site S2; Kodiak Lake	Possible increase	LLCF; Unknown in Kodiak Lake	Possibly	No statistical analyses possible at this time. Observed means at monitored sites downstream of the LLCF, as far as site S2, and in Kodiak Lake were greater than observed means at reference sites, with a pattern of decreasing concentration with increasing distance from the LLCF.
Phytoplankton							
Chlorophyll *a*	No	No	-	-	-	No	-
Density	No	No	-	-	-	No	-

(continued)

Table 3.5-1. Summary of Evaluation of Effects for the Koala Watershed and Lac de Gras (continued)

Variable	Change Downstream of LLCF?	Change Upstream of LLCF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Phytoplankton (*cont'd*)							
Diversity	Yes	Yes	Leslie Lake; Kodiak Lake	Decrease	LLCF for Leslie Lake, Possibly related to decrease in nitrate-N in Kodiak Lake	Yes in Leslie Lake; Possibly in Kodiak Lake	Diversity in Leslie Lake returned to historical levels in 2013.
Relative Densities of Major Taxa	Yes	Yes	Downstream to site S2; Kodiak Lake	(see Notes column)	LLCF downstream to site S2; Possibly related to decrease in nitrate-N in Kodiak Lake	Yes; Possibly in Kodiak Lake	Decline in relative abundance of blue-green algae and increase in diatoms downstream of the LLCF. Replacement of diatoms with green algae in Leslie Lake between 2010 and 2014. Replacement of diatoms with blue-green algae in Kodiak Lake between 2013 and 2014.
Zooplankton							
Biomass	No	No	-	-	-	No	-
Density	No	No	-	-	-	No	-
Diversity	Yes	No	Leslie, Moose, and Nema lakes; Possibly sites S2 and S3	Decrease	LLCF downstream to Nema Lake; Unknown in sites S2 and S3	Yes in lakes downstream of the LLCF as far as Nema Lake	Generally declining over time in Leslie and Moose lakes, but has increased to historical or reference lake levels since 2013. Low values recorded in Nema Lake in 2012 and 2013, but has increased to historical or reference lake levels in 2014. Decreased values observed in 2013 and 2014 in sites S2 and S3; however, within the range of historical values in Lac de Gras and similar pattern observed in reference lakes in 2014

(continued)

Table 3.5-1. Summary of Evaluation of Effects for the Koala Watershed and Lac de Gras (continued)

Variable	Change Downstream of LLCF?	Change Upstream of LLCF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Zooplankton (*cont'd*)							
Relative Densities of Major Taxa	Yes	No	Downstream to Nema Lake; Sites S2 and S3	Decrease in cladocerans and rotifers downstream to Nema Lake; Increase in rotifers at sites S2 and S3	LLCF downstream to Nema Lake; Unknown in sites S2 and S3	Yes in lakes downstream of the LLCF as far as Nema Lake	Decrease in proportion of cladocerans and rotifers downstream to Nema Lake possibly as a result of the LLCF discharge; Increased density of rotifers (*Conochilus* and *Keratella*) in recent years at sites S2 and S3.
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 Orthocladiinae in Leslie, Moose and Nema lakes with an increase in Diamesinae, Prodiamesinae, Chironominae, and/or Tanypodinae; Increase in Prodiamesinae at site S2	-	Yes	Changes in community composition may be related to decreases in some genera (*Heterotanytarsus*, *Rheocricotopus*, *Psectrocladius*) and increases in others (*Monodiamesa*, *Protanypus*, *Cladotanytarsus*, *Paratanytarsus*, *Stempellinella*, *Procladius*, *Ablabesmyia*). A similar pattern of decreased Orthocladiinae with increasing Chironominae (*Corynocera, Paratanytarsus*, *Stictochironomus*) observed in recent years in one reference lake (i.e., Counts Lake).
Stream Benthos							
Density	No	Yes, Historical	Kodiak-Little Stream	Decrease	Unknown	Historical	Densities have decreased from initially high levels.

(continued).

Table 3.5-1. Summary of Evaluation of Effects for the Koala Watershed and Lac de Gras (completed)

Variable	Change Downstream of LLCF?	Change Upstream of LLCF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Stream Benthos (*cont'd*)							
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 enumeration/identification observed.
EPT Diversity	No	No	-	-	-	No	-
EPT Relative Density	No	No	-	-	-	No	-

Notes: Dashes indicate not applicable.
Comparisons to CCME guidelines and other relevant benchmarks are for 2014 data only.
DO = dissolved oxygen
CCME = Canadian Council of Ministers of the Environment
SSWQO = Site-specific Water Quality Objective

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 and increasing temperatures with increasing depth. Although surface temperatures in Grizzly Lake were warmer in 2014, the pattern of increasing temperature with increasing depth was still present. The cause of the change in Grizzly Lake is unclear. 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 more recent DO profiles likely represent undisturbed conditions in Kodiak Lake: aerators would cause mixing of the water column which would result in homogeneity of temperature throughout the water column and greater availability of oxygen in the upper portions of the water column.

Open water season temperature and dissolved oxygen profiles are not evaluated as part of the AEMP; however, recent trends of decreasing temperature and increasing dissolved oxygen with increasing depth are becoming apparent in some lakes downstream of the LLCF (i.e., Leslie, Moose and Slipper lakes) and in Kodiak Lake (data not shown). In contrast, Secchi depths were similar to those observed in previous years in all monitored lakes.

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 recent change in the shape of the under-ice 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 and zooplankton) were assessed in Grizzly Lake in 2014 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 3.10.2 of Part 2 – Data Report.

Twenty-two water quality variables were evaluated in the 2014 AEMP for the Koala Watershed and Lac de Gras. Of these, concentrations of 19 variables have changed in lakes or streams in the Koala Watershed or Lac de Gras (Table 3.5-1). Although concentrations of seven water quality variables have stabilised at some sites in recent years, concentrations remain elevated above baseline or reference concentrations in all 19 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 the 19 water quality variables identified in Table 3.5-1 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. A 20th variable (i.e., TOC) also showed evidence of an increase through time; however, no clear downstream spatial gradient was present suggesting that observed patterns may represent natural regimes. In monitored lakes and streams that are not downstream of the LLCF (i.e., Grizzly Lake, Kodiak Lake and associated streams), only three water quality variables have increased through time: pH has increased in Grizzly Lake, 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 2014c). 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 in Section 2.3). 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 (CCME 2004). Other water quality benchmark values include provincial guidelines or ones taken from the published literature (see Table 2.3-1 in Section 2.3; antimony, barium, and strontium). 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, potassium, and total cadmium (see Table 3.5-1). For pH and total phosphate-P, levels and concentrations in reference lakes or streams also exceeded CCME guidelines, suggesting that exceedances are not related to mine activities. For total cadmium, the concentration was greater than the CCME guideline in only one sample from one stream in June. Since total cadmium concentrations have generally been below detection limits in all reference and monitored sites since monitoring began, this exceedance is unlikely related to mine activities. In contrast, potassium exceedances were unique to the two most upstream monitored lakes and are thus likely related to mine activities.

Eleven sediment quality variables were evaluated in the 2014 AEMP for the Koala Watershed and Lac de Gras. Of these, the concentration of one variable (i.e., total molybdenum) has changed through time and two other variables (i.e., total antimony and total strontium) showed signs of potential increases or mine effects (Table 3.5-1). Total molybdenum concentrations in sediments have increased in lakes downstream of the LLCF as far as Slipper Lake. In monitored lakes downstream from the LLCF, total antimony concentrations as far as Slipper Lake and total strontium concentrations as far as site S2 were higher than those observed in reference lakes. In both cases, concentrations decreased with increasing distance from the LLCF. Concentrations of molybdenum, antimony, and strontium follow the same pattern observed for concentrations in water quality samples; suggesting that increased concentrations in sediments are likely mine effects that stem from LLCF discharge.

CCME guidelines for the protection of aquatic life exist for two of the evaluated sediment quality variables, including arsenic and cadmium (see Table 2.4-1 in Section 2.4; CCME 2014b). For arsenic, the observed mean exceeded the CCME ISQG in all monitored sites and the CCME PEL in Slipper Lake and at site S2 in Lac de Gras (Table 3.5-1). The 95% confidence intervals around the fitted mean arsenic concentration exceeded the CCME PEL in all monitored sites, except Nema Lake. However, similar exceedance patterns were observed in all three reference lakes (Table 3.5-1). For cadmium, the 95% confidence intervals around the fitted mean in 2014 exceeded the CCME ISQG in Slipper Lake and at site S2 in Lac de Gras; however, a similar pattern was observed in one reference lake (Table 3.5-1). Cadmium concentrations in sediments were less than the CCME PEL value in all monitored sites (Table 3.5-1).

Despite increases in 19 evaluated water quality variables and three evaluated sediment quality variables downstream of the LLCF, observed concentrations were generally below 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 3.4.2; Biesinger and Christensen 1972). Thus, observed changes in biological community composition in the Koala Watershed 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 2012c). 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 toward increased densities of green algae was observed in Leslie Lake from 2010 to 2012, and in Nema and Slipper lakes in 2013. Although Chlorophyceae densities in 2014 have decreased in Nema and Slipper lakes, they remain elevated in Leslie Lake. 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 Nema lakes, in which relative densities of Orthocladiinae have decreased, while densities of Diamesinae, Prodiamesinae, Chironominae and/or Tanypodinae 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 2012c).

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 DO concentrations and open water season Secchi depths (see Section 3.1.1).

4.1.2 Dataset

Under-ice DO 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

(continued)

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 (completed)

Year	Nanuq	Counts	Vulture	Cujo	LdS1
2013	Apr-26	Apr-26	Apr-23	May-11	Apr-26
2014	Apr-7	Mar-31	Apr-8	Apr-6	Apr-6

Note: 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
2014	Aug-5	Aug-9	Aug-3	Aug-7	Aug-8

Note: Dashes indicate no data were available.

4.1.3 Results and Discussion

4.1.3.1 Under-ice Dissolved Oxygen and Temperature

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 bottom half of the water column in mid-February and mid-March 2014. 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 March to early April of 2014 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 uniform throughout the water column, while temperature increased with depth (Figure 4.1-1). DO concentrations in Cujo Lake have been variable through time, but were slightly greater than the CCME guideline throughout most of the water column during April 2014 AEMP sampling (Figure 4.1-1; CCME 2014c). As in previous years, DO concentrations at site LdS1 in Lac du Sauvage were greater than the CCME guideline of 6.5 mg/L at all depths (Figure 4.1-1; CCME 2014c).

Additional over-winter monitoring conducted in Cujo Lake revealed variability in under-ice DO concentrations between January and April 2014 (Figure 4.1-2). In general, the concentration of DO in Cujo Lake was greater than the 6.5 mg/L CCME guideline in the upper portion of the water column throughout the winter (Figure 4.1-3; CCME 2014c). DO concentrations were less than the CCME guideline throughout the bottom half of the water column in mid-February and mid-March, but began to increase in April (Figure 4.1-3), which is consistent with expected patterns in under-ice DO in ice-covered lakes (discussed in Section 3.1.3.1). Similar to past years, a portion of the surface ice on Cujo Lake was cleared of snow in late winter (April 21, 2014) to allow for increased light penetration and thus increased DO production through photosynthesis. 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 3.1-1b). 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 2014.

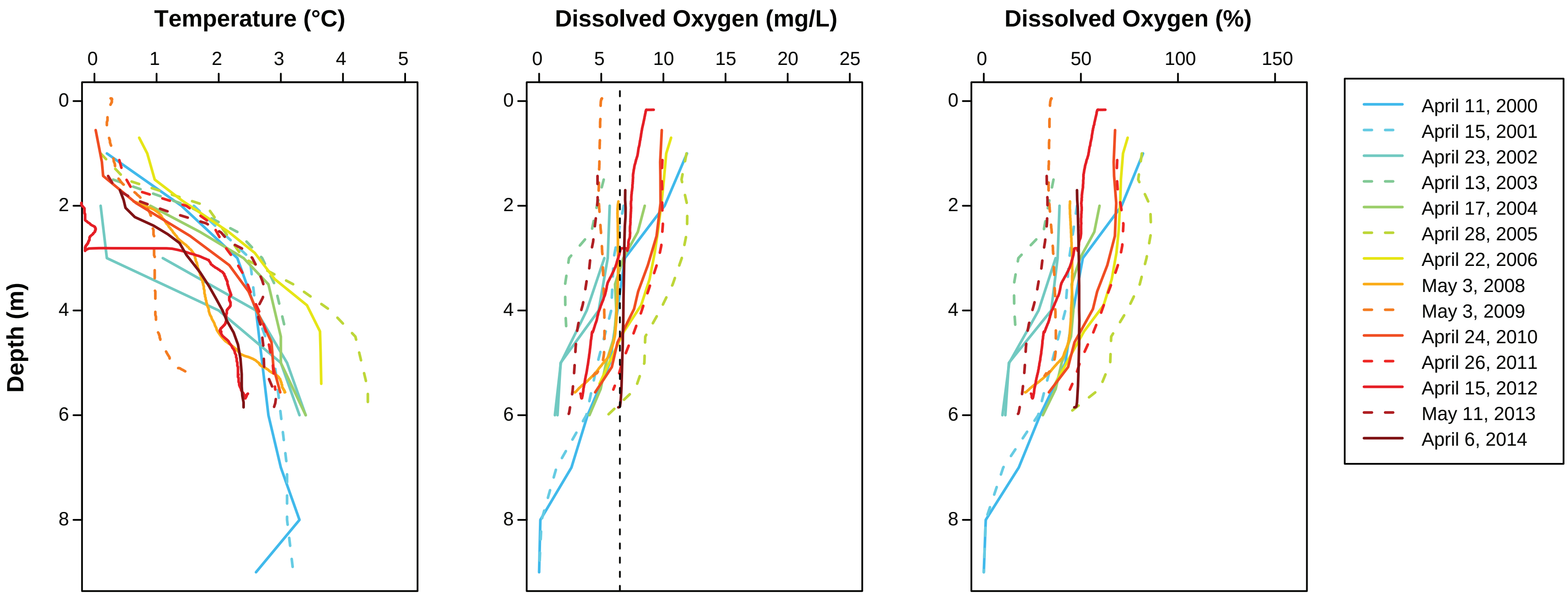
Although under-ice temperature profiles in Cujo Lake have been variable through time, 2014 temperature profiles in Cujo Lake and at site LdS1 in Lac du Sauvage were similar to those observed in previous years, with water temperature warming from the surface to the bottom of the lakes (Figure 4.1-1). Trends in monitored lakes were also similar to those observed in reference lakes (Figure 3.1-1).

4.1.3.2 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 King-Cujo Watershed and Lac du Sauvage (Figure 4.1-3). A value of ± 0.5 m was used as an estimate of error due to sampler bias for interpreting graphical results.

Figure 4.1-1a
Under-ice Dissolved Oxygen and Temperature Profiles for King-Cujo Watershed Lakes and Lac du Sauvage, 2000 to 2014

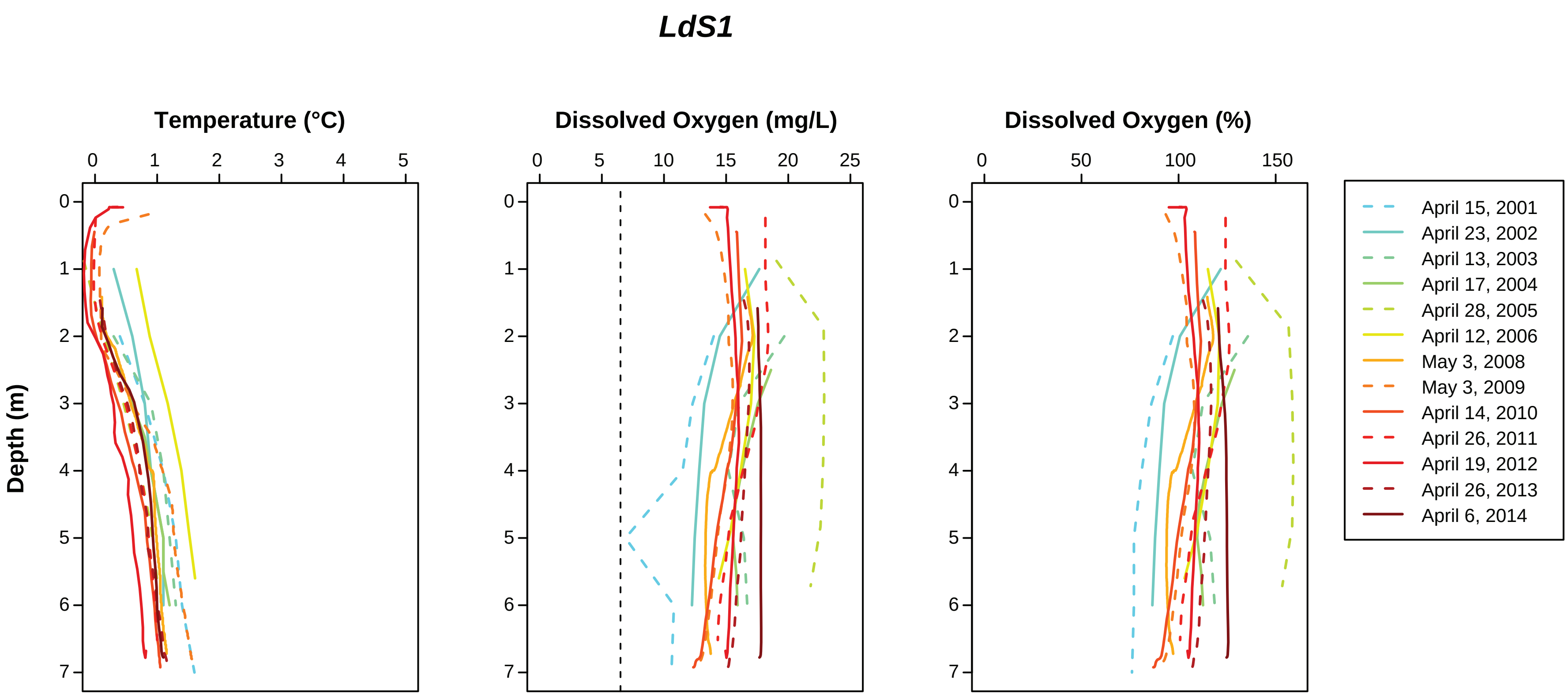
Cujo Lake



*Note: Vertical dashed line represents the CCME guideline for non-early life stages (6.5 mg/L).
Data collected and supplied by DDEC.*

Figure 4.1-1b

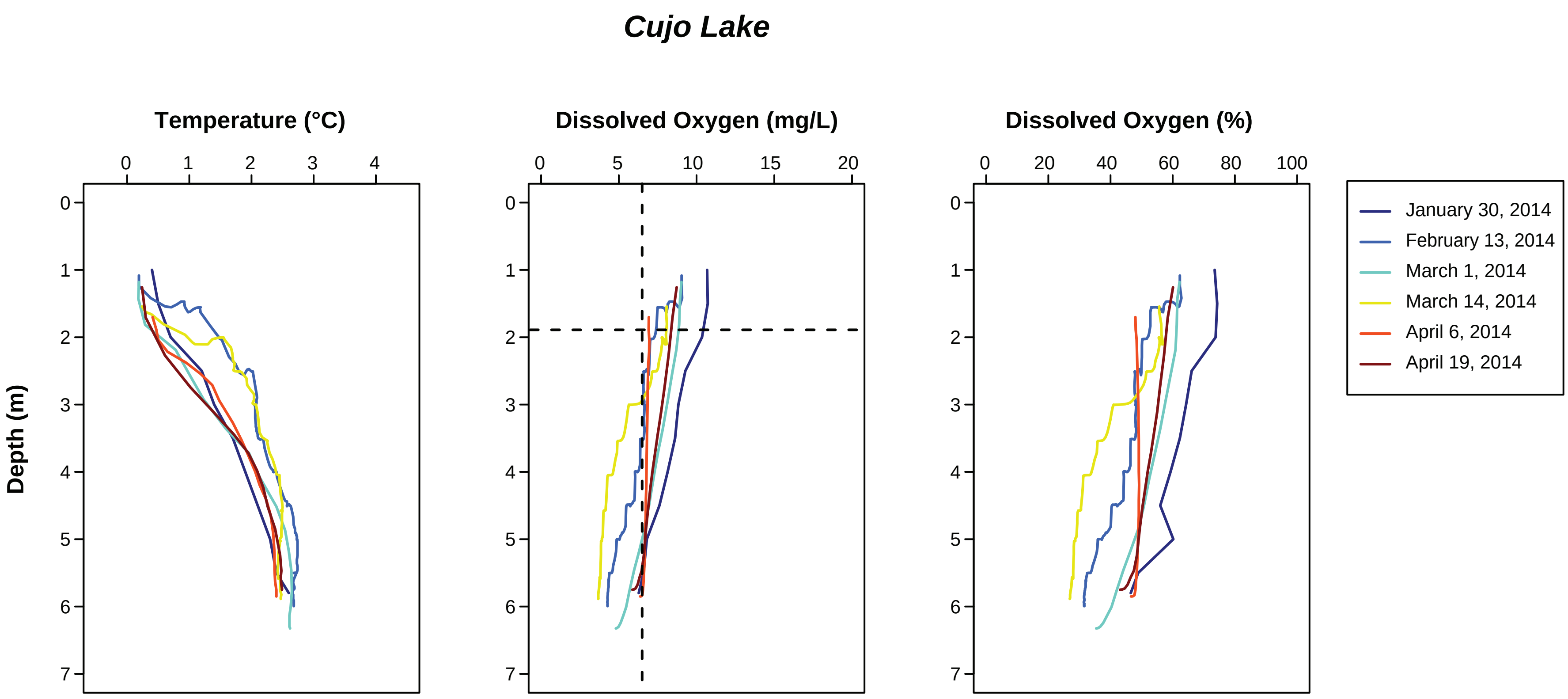
Under-ice Dissolved Oxygen and Temperature Profiles for King-Cujo Watershed Lakes and Lac du Sauvage, 2000 to 2014



Note: Vertical dashed line represents the CCME guideline for non-early life stages (6.5 mg/L).
Data collected and supplied by DDEC.

Figure 4.1-2

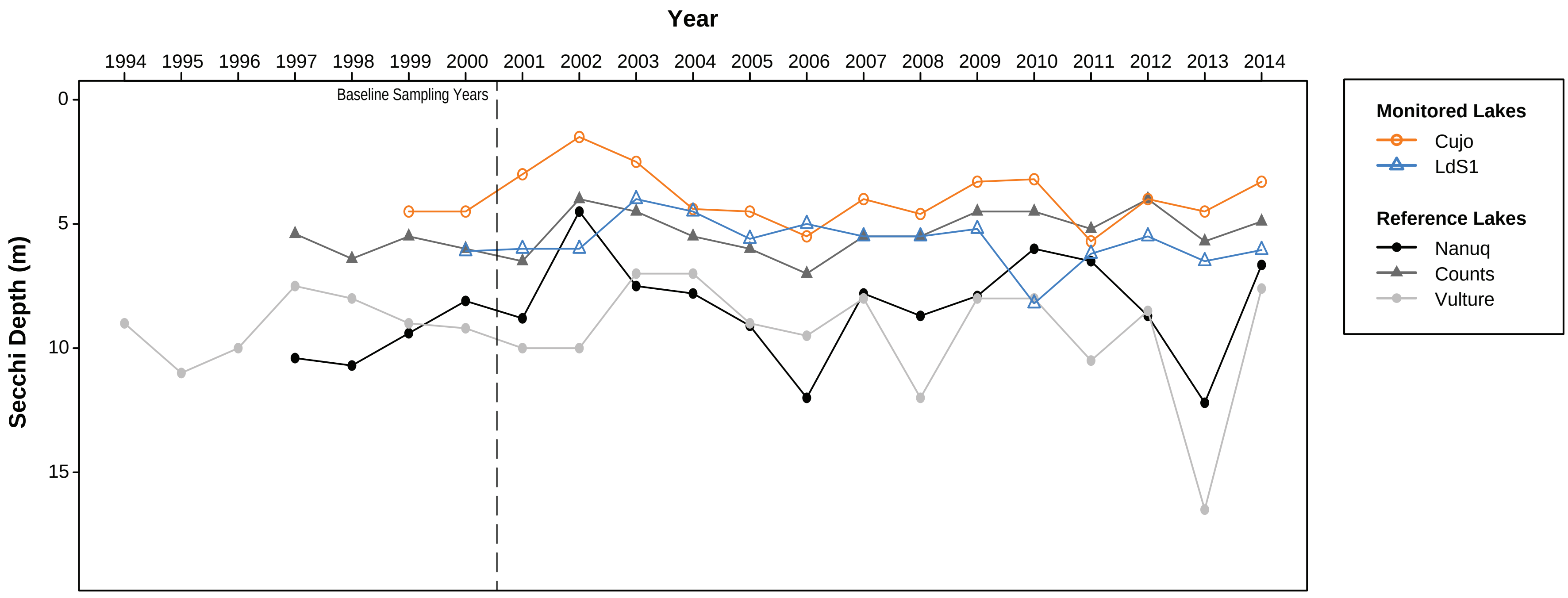
Dissolved Oxygen and Temperature Profiles for Cujo Lake, Ice-covered Season 2014



Note: Horizontal dashed line represents maximum ice thickness.
Vertical dashed line represents the CCME guideline for non-early life stages (6.5 mg/L).
Data collected and supplied by DDEC.

Figure 4.1-3

August Secchi Depths for King-Cujo Watershed Lakes and Lac du Sauvage, 1994 to 2014



Taking into account estimated error for each year, observed August 2014 Secchi depths were similar to those observed in baseline years at site LdS1 in Lac du Sauvage (Figure 4.1-3). In Cujo Lake, the 2014 Secchi depth was shallower than in baseline years, but a similar pattern was observed in in one of the reference lakes (i.e., Nanuq Lake; Figure 4.1-3). Thus, no mine effects were detected with respect to Secchi depth in monitored lakes of 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 2014c). 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 include provincial guidelines or ones taken from the published literature (see Table 2.3-1 in Section 2.3).

4.2.2 Dataset

4.2.2.1 Lakes

Lake water quality data were 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 2014 (Tables 4.2-1 and 4.2-2). No water was pumped from the KPSF into Cujo Lake during the 2014 AEMP year. 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	-	-	-	-	-

(continued)

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 (completed)

Year	Nanuq	Counts	Vulture	Cujo	LdS1
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)
2014	Apr-7 (4)	Mar-31 (4)	Apr-8 (4)	Apr-6 (4)	Apr-6 (4)

Notes: 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.*

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 (6), Aug-14 (6)	Jul-27 (6), Aug-10 (6)	-	-	-	-
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)

(continued)

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 (completed)

Year	Nanuq	Counts	Vulture	1616-43	Cujo	LdS2	LdS1
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)
2014	Aug-5 (4)	Aug-9 (4)	Aug-3 (4)	Aug-1 (2)	Aug-7 (4)	Aug-8 (2)	Aug-8 (4)

Notes: 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 from 1994 to 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 in Figures 4.2-1 to 4.2-23 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.

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. Over the years, data were removed from the dataset prior to analysis and interpretation as a result of contamination or laboratory difficulties in sample analysis (Table 4.2-3).

4.2.2.2 Streams

Stream water quality was collected from late July to mid-August of each year from 1998 to 2014 (Table 4.2-4). Baseline water quality data from reference streams, collected between 1994 and 1997, were not used in the statistical evaluation of effects, but are included in Table 4.2-3 and shown graphically for visual comparison in Figures 4.2-1 to 4.2-23 below.

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
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 outlier

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

Year	Nanuq Outflow	Counts Outflow	Vulture-Polar	KPSF 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)
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)

(continued)

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

Year	Nanuq Outflow	Counts Outflow	Vulture-Polar	KPSF 1616-43	Cujo Outflow	Christine-Lac du Sauvage
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)
2014	Aug-1 (2)	Aug-4 (2)	Aug-1 (2)	Aug-1 (2)	Aug-4 (2)	Aug-4 (2)

Notes: 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 2014, two replicate samples were collected at each site.

Stream water quality samples were analyzed as described in Section 3.2.2.1. Some data have 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

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 tables also indicate data, if any, that were excluded from the analysis. 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:

- <u>Model fit = 1a</u> 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.

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 stabilised in recent years. Observed and fitted mean pH values were within the CCREM guideline range (pH 6.5 to 9) at all monitored sites during both the ice-covered and open water season.

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 4.2-5). 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 during the ice-covered season and more recently during the open water season (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 continued until July 2013. The 95% confidence intervals of the fitted mean and the observed pH values were within the CCREM guideline range (pH 6.5 to 9) in all monitored lakes and streams in 2014 (CCREM 1987).

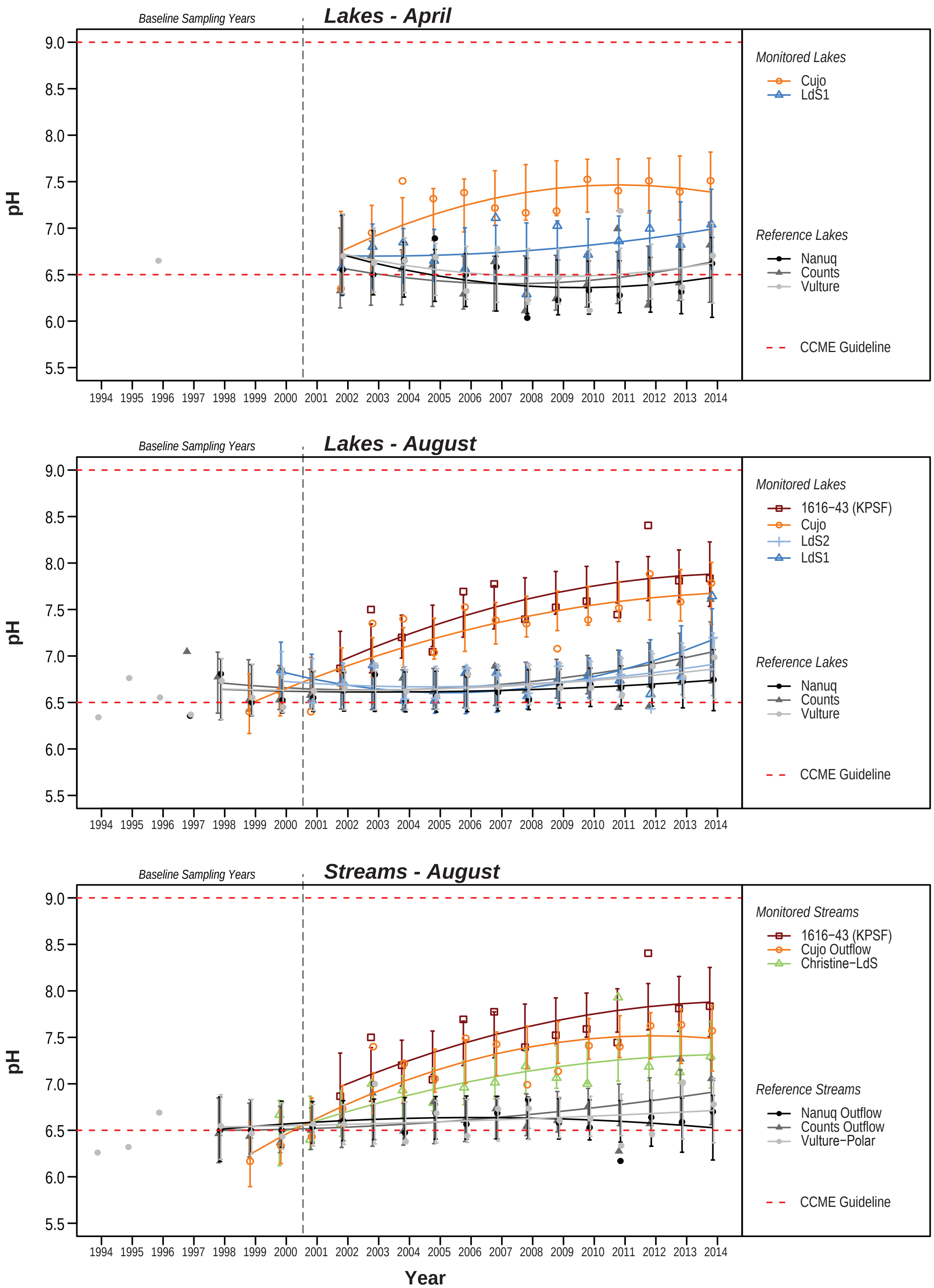
Table 4.2-5. Statistical Results of pH in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

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	Cujo Outflow, Christine- Lac du Sauvage	Cujo Outflow, Christine- Lac du Sauvage		2-12

Note: 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 2014



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

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.

Statistical and graphical analyses indicate that total alkalinity has increased 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 4.2-6; Figure 4.2-2) as a result of mine operations.

Table 4.2-6. 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	-	2-24
Aug	Stream	-	Tobit	2	-	1616-43 (KPSF), Cujo Outflow, Christine-Lac du Sauvage	-	2-30

Note: Dashes indicate not applicable.

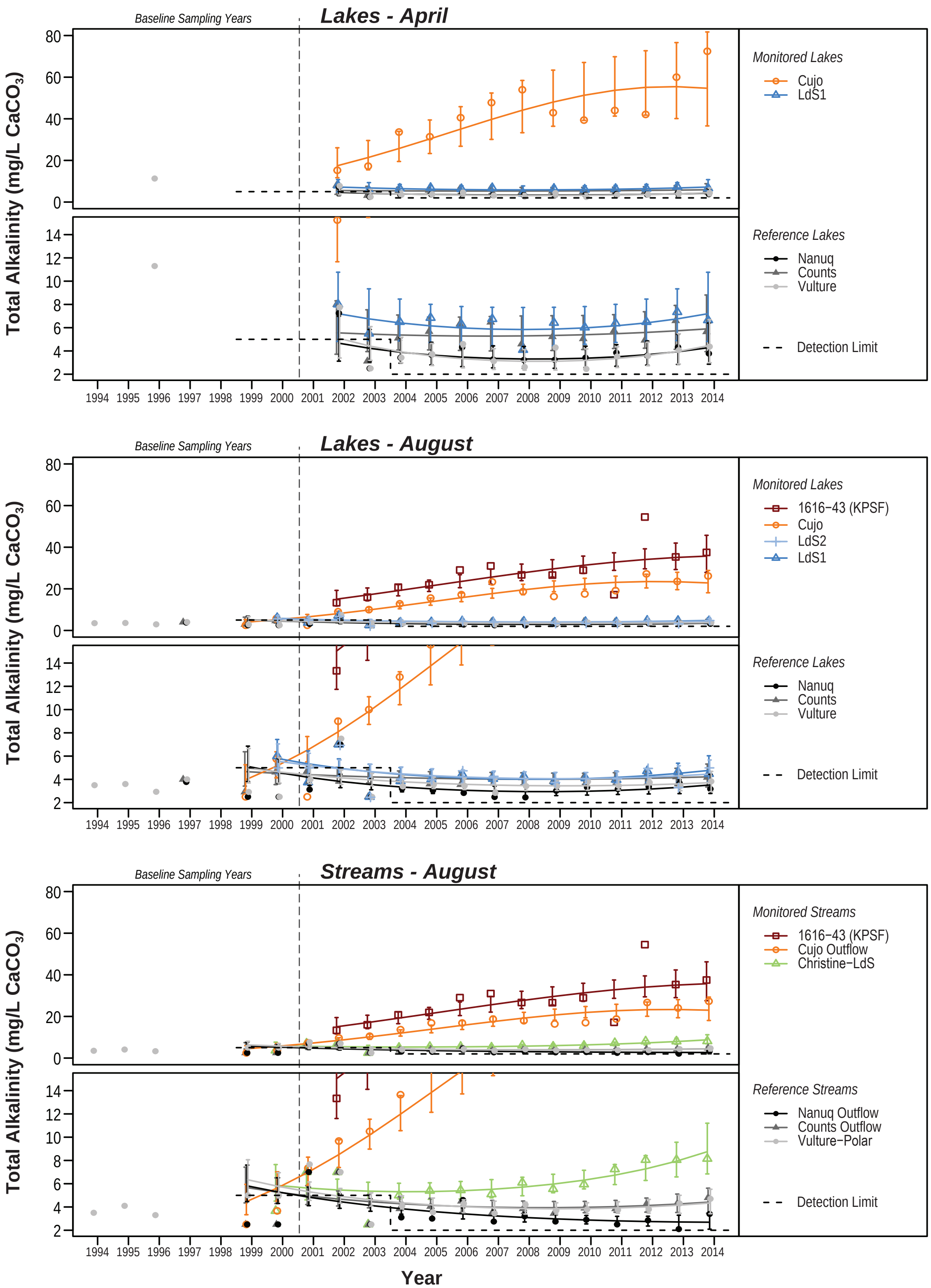
4.2.4.3 *Water Hardness*

> ***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 4.2-7). 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 continued until July 2013.

Figure 4.2-2

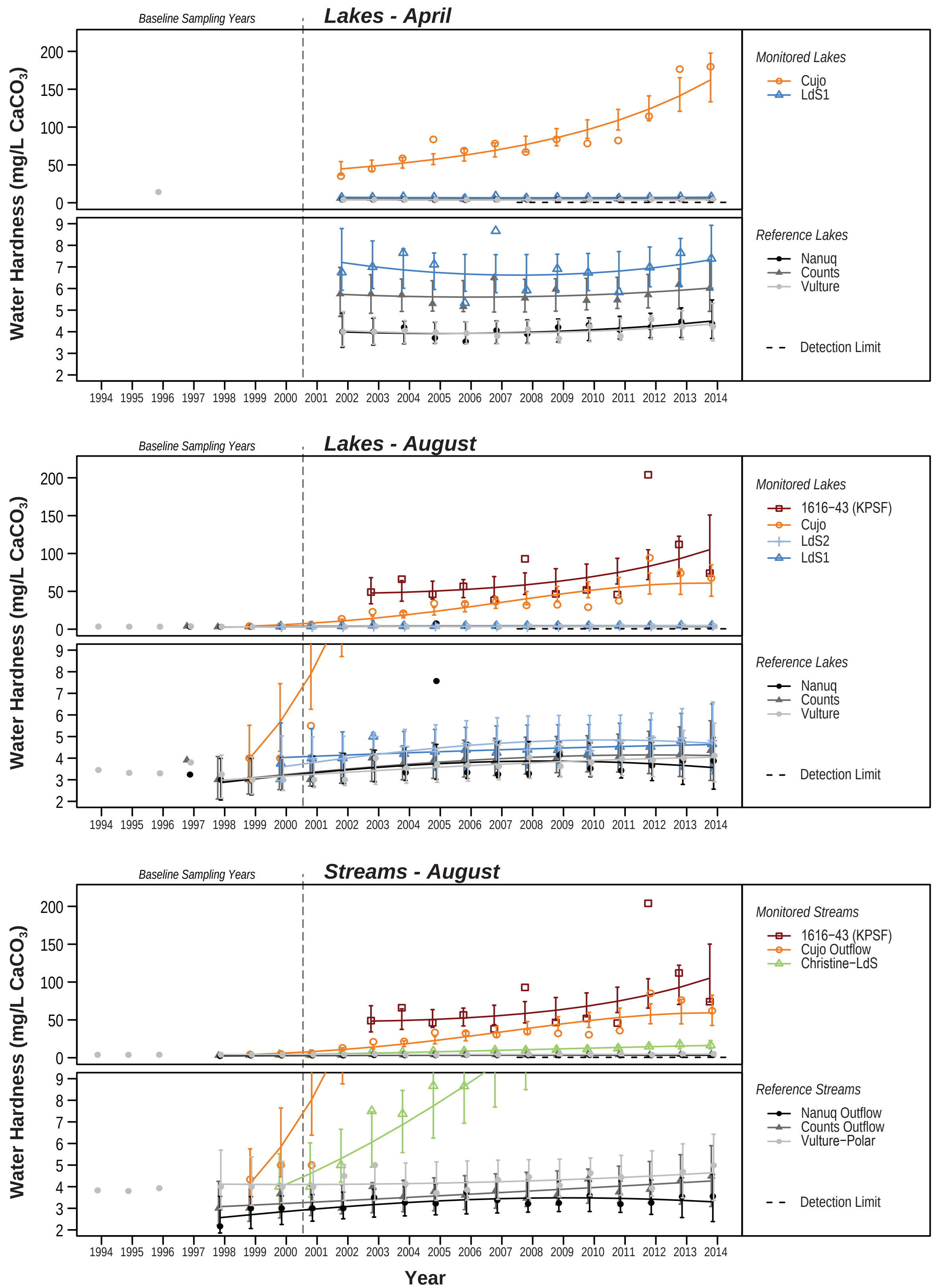
Observed and Fitted Means for Total Alkalinity in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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 2014



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 4.2-7. 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		2-36
Aug	Lake	-	LME	3	1616-43 (KPSF), Cujo, LdS1, LdS2	Cujo	-	2-42
Aug	Stream	-	LME	2	-	Cujo Outflow, Christine-Lac du Sauvage		2-48

Note: 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 2014.

Statistical analyses indicate that chloride concentrations have changed through time at all monitored sites downstream of the KPSF, as far as Cujo Outflow (Table 4.2-8). Graphical analysis suggests that chloride concentrations in Cujo Lake and Cujo Outflow have increased through time (Figure 4.2-4). In contrast, graphical analysis suggests that chloride concentrations have decreased through time at site LdS1 in Lac du Sauvage, likely as a result of a reduction in detection limits rather than actual decreases in chloride concentrations (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 4.2-8. Statistical Results of Chloride Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

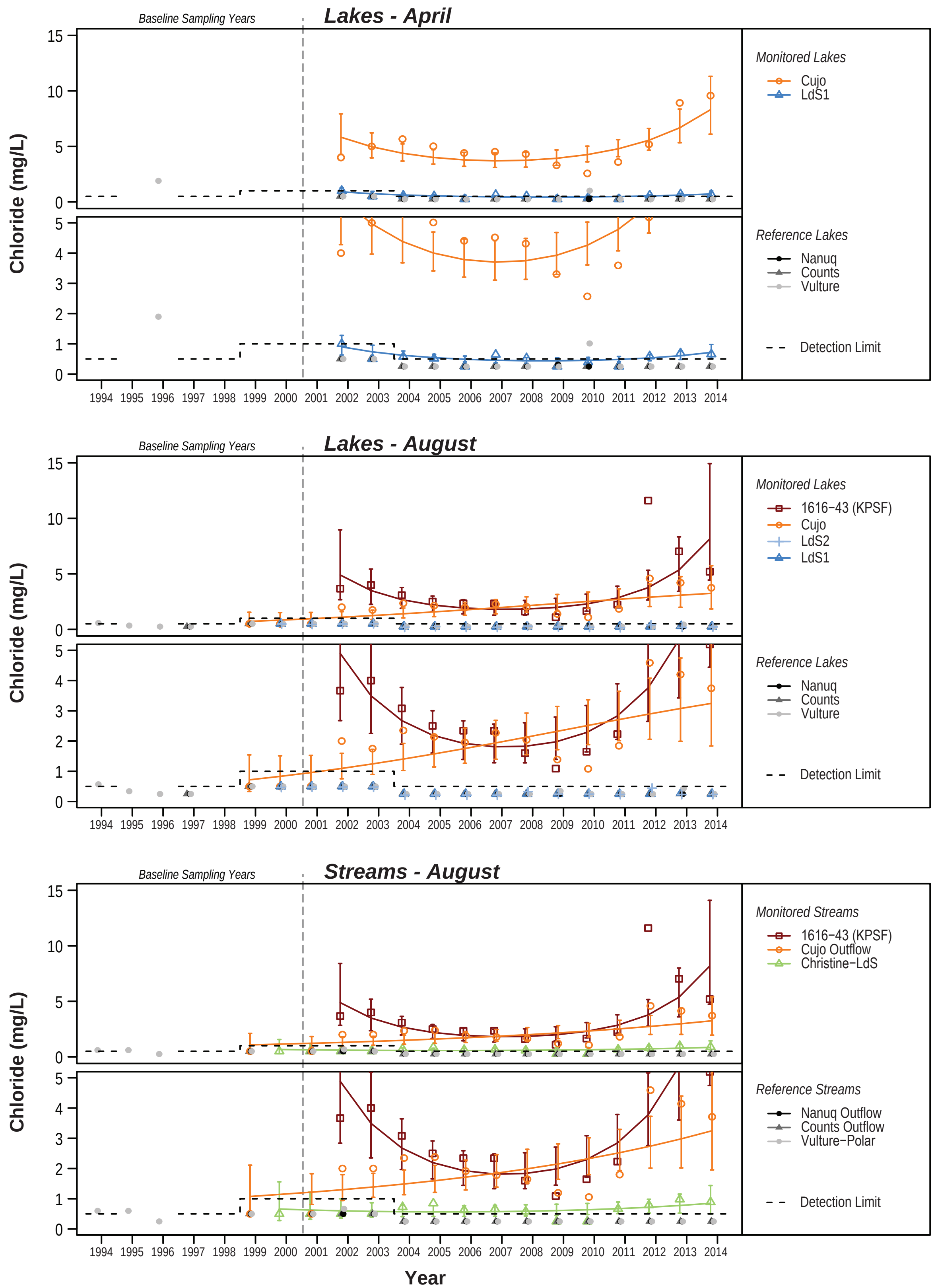
Month	Lake/ Stream	Lakes/Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	Counts, Nanuq, Vulture	Tobit	1a	-	-	Cujo, LdS1	2-54
Aug	Lake	LdS1, LdS2, Counts, Nanuq, Vulture	Tobit	1a	-	-	1616-43 (KPSF), Cujo	2-58
Aug	Stream	Counts Outflow, Nanuq Outflow, Vulture Outflow	Tobit	1a	-	-	1616-43 (KPSF), Cujo Outflow	2-62

Note: Dashes indicate not applicable.

Figure 4.2-4

Observed and Fitted Means for Chloride in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014

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.
SSWQO = 116.6 X ln(Hardness) - 204.1, where hardness = 10 - 160 mg/L.

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 2014 (Elphick, Bergh, and Bailey 2011). Chloride concentrations were also less than the SSWQO in all monitored streams in June, July, August, and September 2014 (see Part 2 – Data Report).

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 2014.

Statistical and graphical analyses indicate that sulphate concentrations have increased through time, relative to reference lakes and streams, downstream from the KPSF as far as Christine-Lac du Sauvage Stream (Table 4.2-9). Increased sulphate concentrations downstream of the KPSF likely reflect increased concentrations within the KPSF in 2012 and 2013 (Figure 4.2-5).

Table 4.2-9. 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	-	2-66
Aug	Lake	-	LME	3	1616-43 (KPSF), Cujo	Cujo	-	2-72
Aug	Stream	-	LME	3	1616-43 (KPSF), Cujo Outflow, Christine-Lac du Sauvage	Cujo Outflow, Christine-Lac du Sauvage	-	2-78

Note: Dashes indicate not applicable.

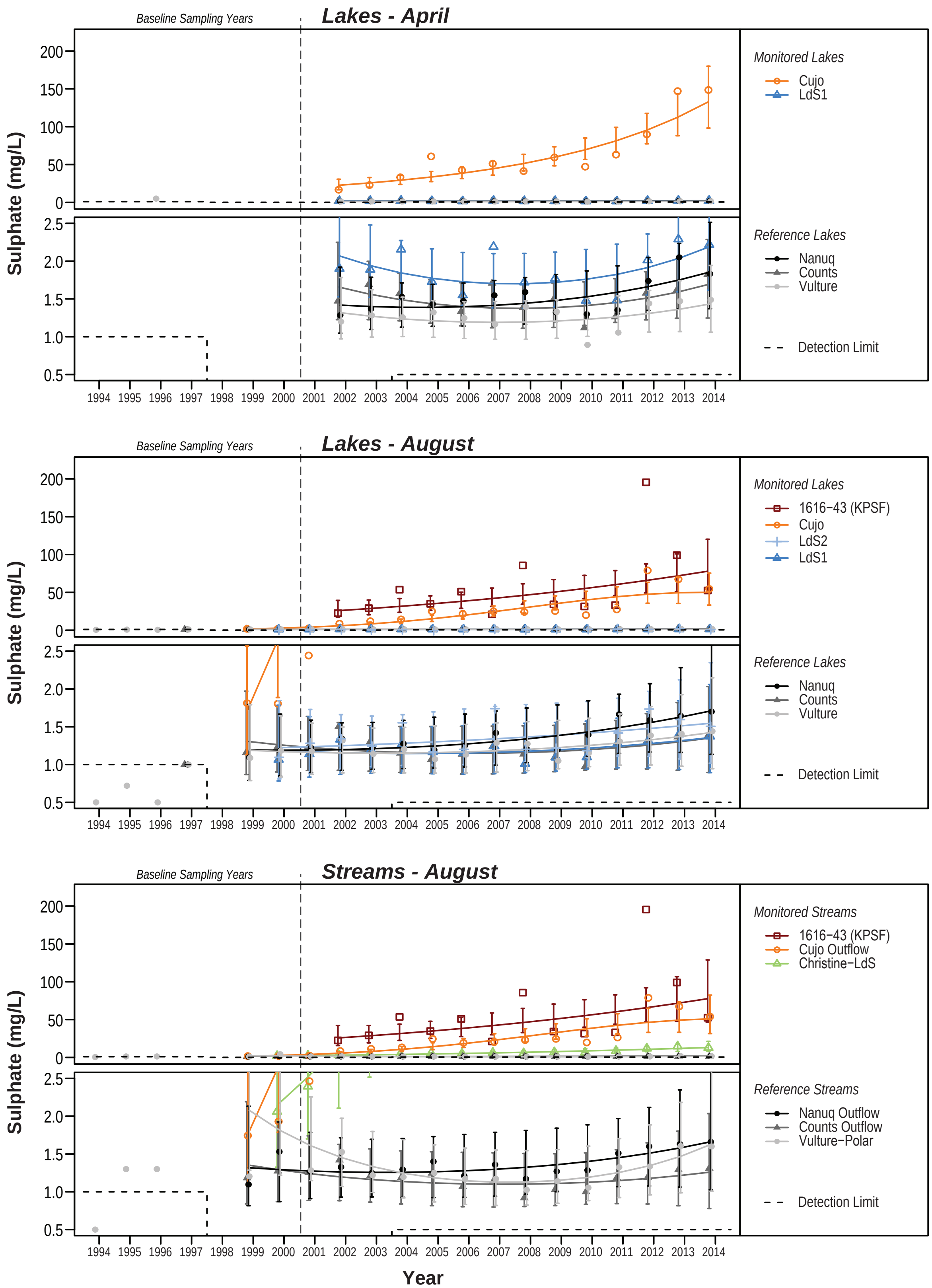
The 95% confidence intervals around the fitted mean and the observed mean sulphate concentrations were less than the hardness-dependent sulphate SSWQO in all reference and monitored lakes and streams in 2014 (Rescan 2012e). Sulphate concentrations were also less than the SSWQO in all monitored streams in June, July, August, and September 2014 (see Part 2 - Data Report; Rescan 2012e).

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 2014.

Figure 4.2-5

Observed and Fitted Means for Sulphate in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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.

Statistical and graphical analyses suggest that potassium concentrations have increased through time, relative to reference sites, at all sites downstream of the KPSF as far as Christine-Lac du Sauvage Stream (Table 4.2-10). Graphical analysis also suggests that potassium concentrations 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.

Table 4.2-10. 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	2	-	Cujo	-	2-84
Aug	Lake	-	LME	2	-	1616-43 (KPSF), Cujo	-	2-89
Aug	Stream	-	LME	1b	-	-	1616-43 (KPSF), Cujo Outflow, Christine-Lac du Sauvage	2-95

Note: 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 in 2014 (Rescan 2012f). Potassium concentrations in all monitored streams in June, July, August, and September 2014 were also less than the long-term potassium SSWQO (see Part 2 - Data Report; Rescan 2012f).

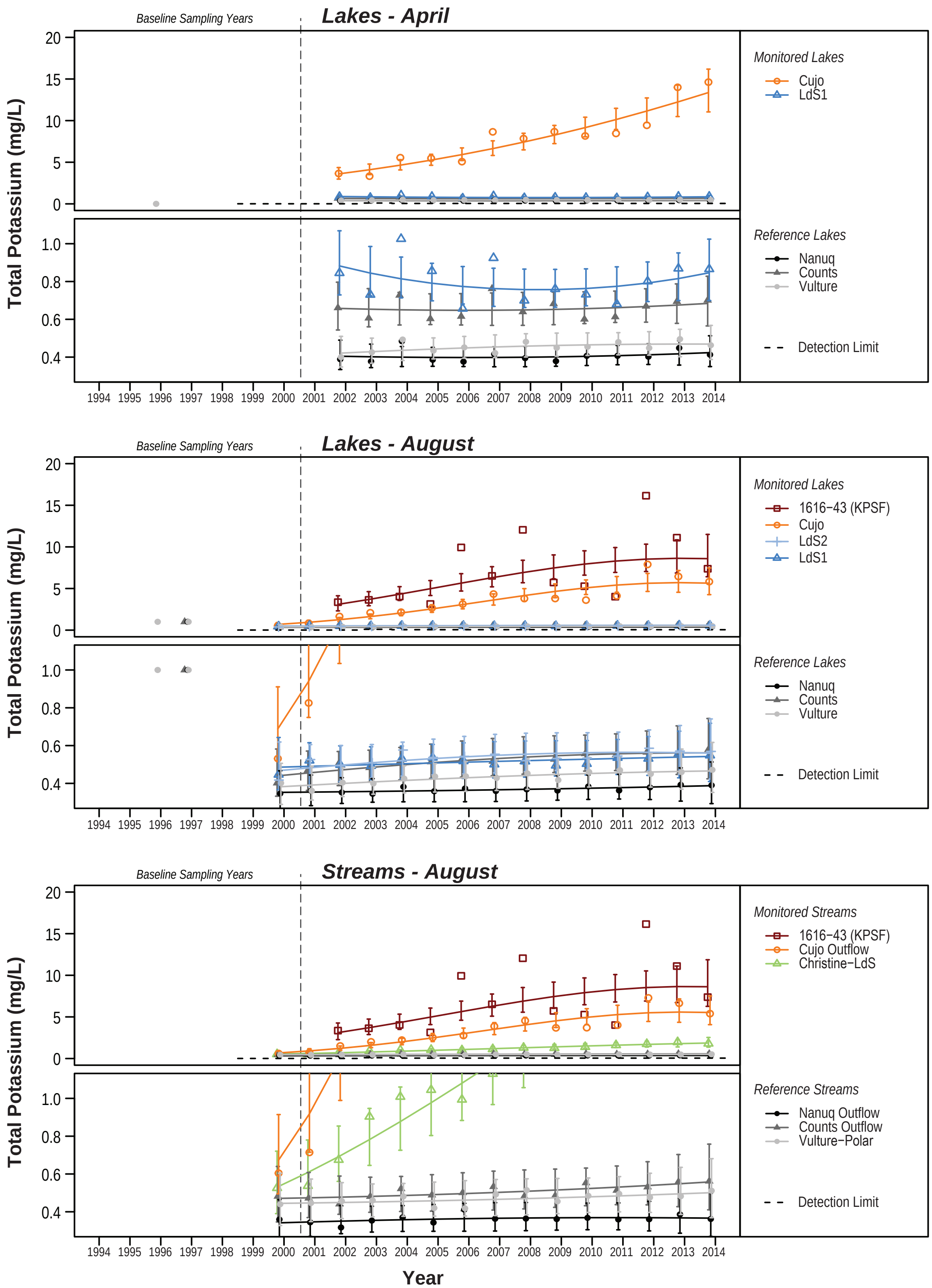
4.2.4.7 Total Ammonia-N

> ***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 2014.

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 4.2-11). 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 continued until July 2013. 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.

Figure 4.2-6

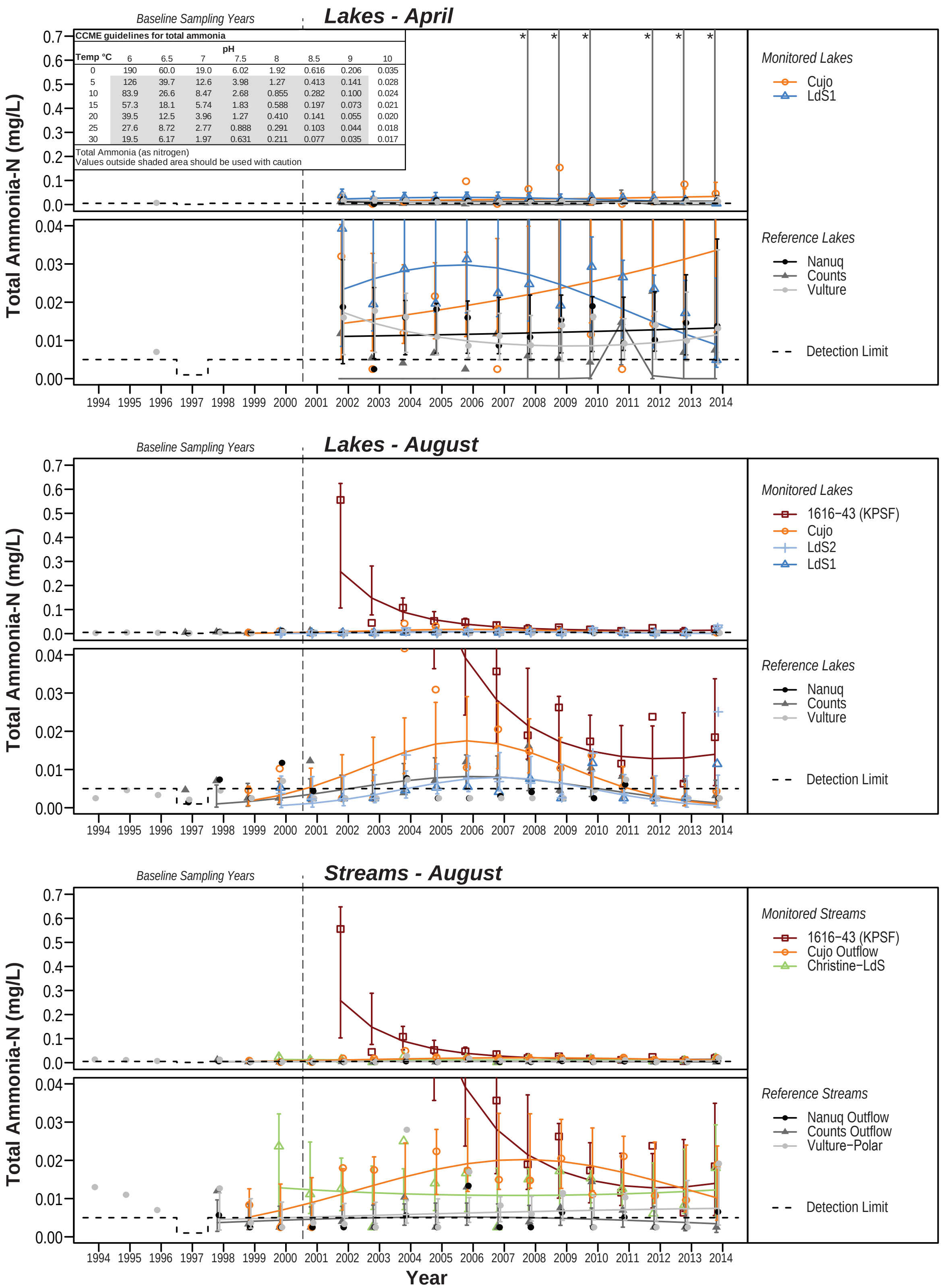
Observed and Fitted Means for Potassium in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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.

Figure 4.2-7

Observed and Fitted Means for Total Ammonia-N in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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.
CCME Guideline is pH and temperature dependent (see inset table).
** Upper 95% Confidence Interval on the fitted mean of Counts Lake in April 2008 = 3.25x10^{246} mg/L, 2009 = 1.76x10^{172} mg/L, 2010 = 2.92x10^{89} mg/L, 2012 = 2.66x10^{95} mg/L, 2013 = 6.58x10^{194} mg/L, and 2014 = 2.24x10^{296}*

Table 4.2-11. 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	2-101
Aug	Lake	LdS1, Nanuq, Vulture	Tobit	1b	-	-	1616-43 (KPSF)	2-105
Aug	Stream	Nanuq Outflow	Tobit	3	1616-43 (KPSF), Cujo Outflow, Christine-Lac du Sauvage	1616-43 (KPSF)	-	2-110

Note: Dashes indicate not applicable.

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 2014 (CCME 2001b). Total concentrations in all monitored streams in June, July, August, and September 2014 were also less than pH- and temperature-dependent ammonia CCME guideline (see Part 2 - Data Report; CCME 2001b).

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 2014. 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 and Lac du Sauvage since monitoring began (Table 4.2-12; 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 and Lac du Sauvage, since nitrite is primarily formed through the oxidisation of ammonia (see Figure 4.2-9). Moreover, nitrite is a relatively transient form of nitrogen, which quickly oxidises to produce nitrate.

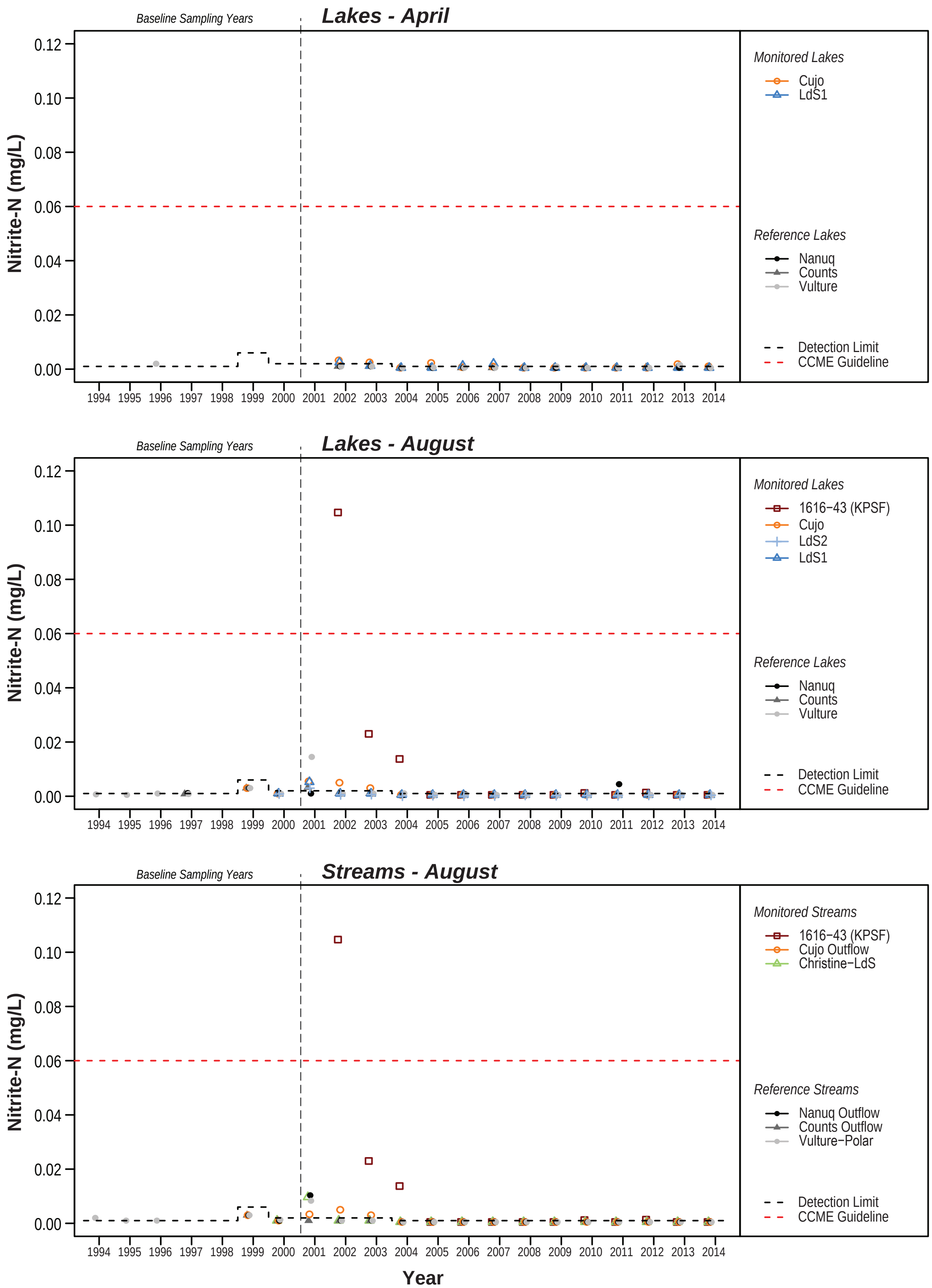
Table 4.2-12. Statistical Results of Nitrite-N Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

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	-	-	-	-	-	2-116
Aug	Lake	ALL	-	-	-	-	-	2-118
Aug	Stream	ALL	-	-	-	-	-	2-120

Note: Dashes indicate not applicable.

Figure 4.2-8

Observed and Fitted Means for Nitrite-N in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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.

All 2014 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). Nitrite-N concentrations in all monitored streams in June, July, August, and September 2014 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 2014. No mine effects were detected.

Statistical analyses indicate that nitrate-N concentrations have changed through time, relative to reference lakes, in Cujo Lake during the ice-covered season (Table 4.2-13). 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 continued until July 2013. 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 4.2-13. Statistical Results of Nitrate-N Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

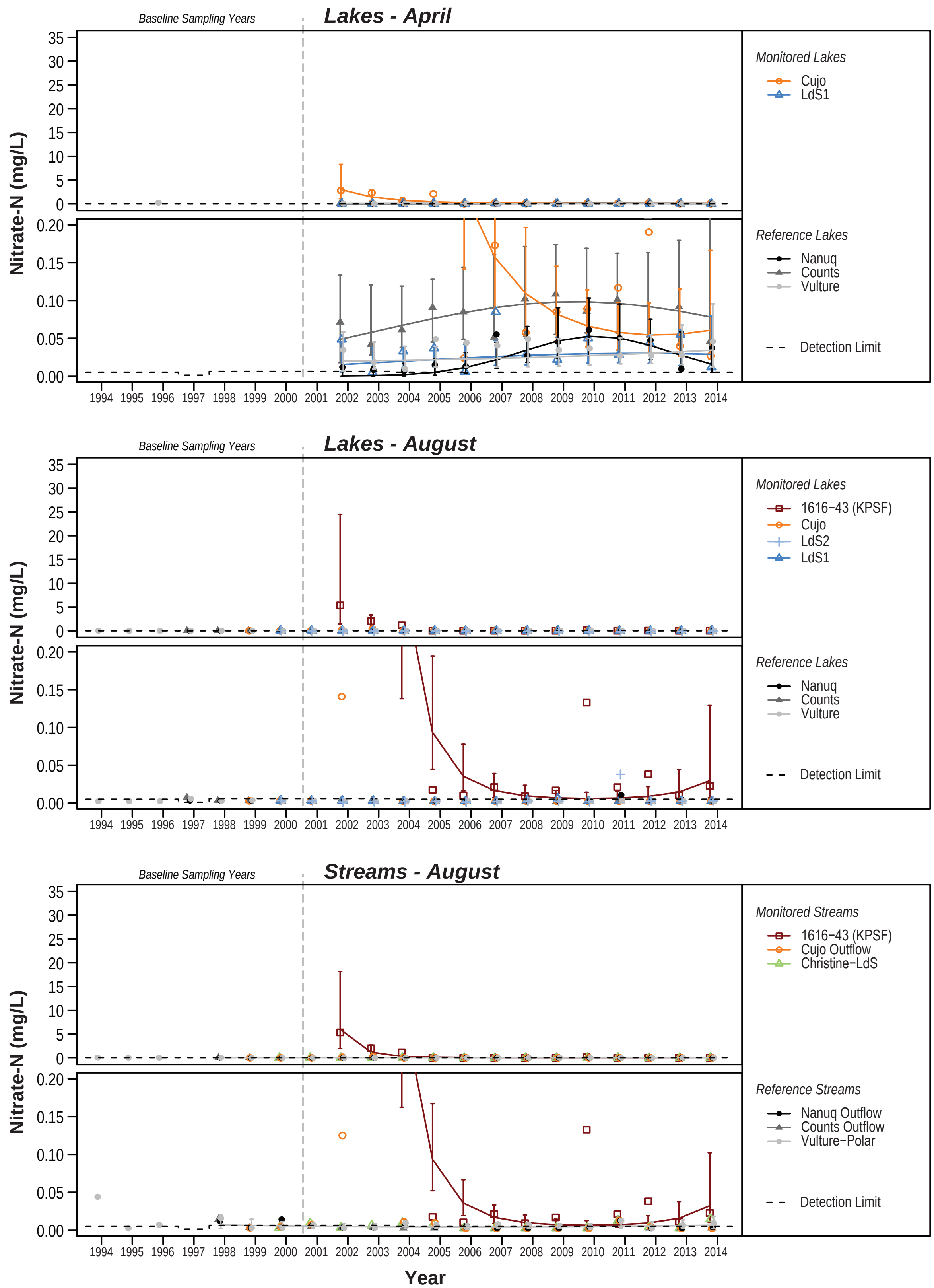
Month	Lake/ Stream	Lakes/Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	Tobit	2	-	Cujo	-	2-122
Aug	Lake	Cujo, LdS1, LdS2, Counts, Nanuq, Vulture	Tobit	1a	-	-	1616-43 (KPSF)	2-128
Aug	Stream	Cujo Outflow, Christine-Lac du Sauvage, Counts Outflow, Nanuq Outflow	Tobit	1b	-	-	1616-43 (KPSF)	2-132

Note: 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 (Rescan 2012d). Nitrate-N concentrations in all monitored streams in June, July, August, and September 2014 were also less than nitrate-N SSWQO (see Part 2 - Data Report; Rescan 2012d).

Figure 4.2-9

Observed and Fitted Means for Nitrate-N in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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.

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 observed and fitted mean total-phosphate P concentrations or the upper 95% confidence intervals around the fitted mean total-phosphate P concentration 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 and graphical analyses indicate that total phosphate-P concentrations have been stable through time, relative to reference sites, at all monitored lakes and streams in the King-Cujo Watershed and Lac du Sauvage during the ice-covered and open water seasons (Table 4.2-14; Figure 4.2-10). No mine effects were detected.

Table 4.2-14. 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	-	-	None	2-137
Aug	Lake	-	Tobit	2	-	None	-	2-142
Aug	Stream	-	Tobit	1b	-	-	None	2-147

Note: Dashes indicate not applicable.

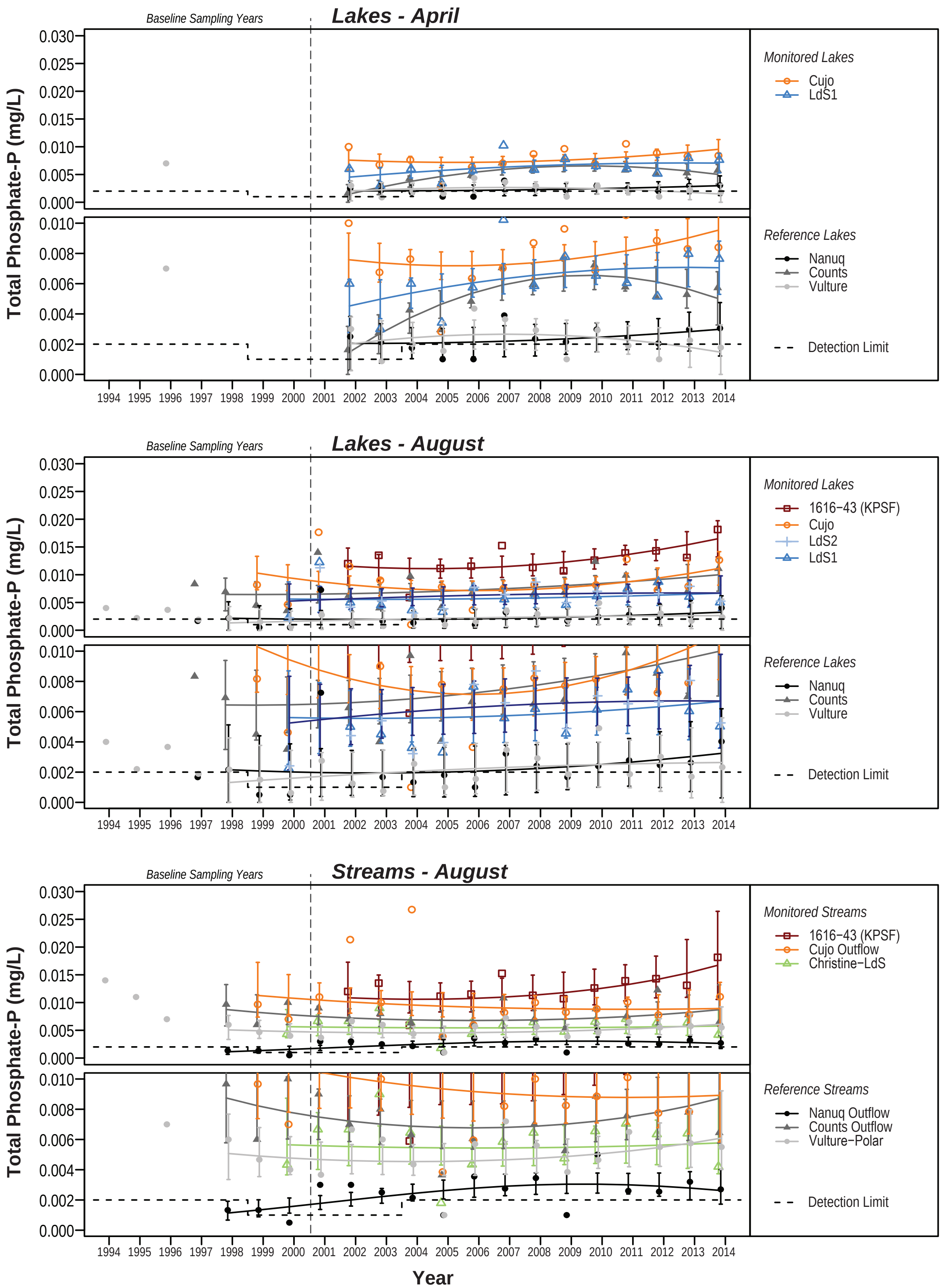
In Cujo Lake, the observed and fitted mean total phosphate-P concentrations during the open water season and the upper 95% confidence intervals around the fitted mean total phosphate-P concentration during the ice-covered season were greater than the 0.01 mg/L trigger set out for oligotrophic lakes in the Canadian Guidance Framework for the management of Phosphorus in Freshwater Systems (CCME 2004; Environment Canada 2004). Total phosphate-P concentrations were also greater than the recommended benchmark trigger of mean baseline concentration + 50% (CCME 2004) at site LdS1 during the ice-covered season; however, a similar pattern was also observed in one reference lake (i.e., Counts Lake; see Part 2 – Data Report). The upper 95% confidence intervals around the fitted mean total phosphate-P concentration also exceeded the 50% triggers at site LdS1 and LdS2 during the open water season; however, a similar pattern was observed in two of the reference lakes (i.e., Counts and Nanuq lakes).

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.

Figure 4.2-10

Observed and Fitted Means for Total Phosphate-P in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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

Statistical and graphical analyses indicate that TOC concentrations have been stable through time, relative to reference sites, at all monitored lakes and streams in the King-Cujo Watershed and Lac du Sauvage during the ice-covered and open water seasons (Table 4.2-15; Figure 4.2-11). However, graphical analysis also 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.

Table 4.2-15. 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	-	2-152
Aug	Lake	-	LME	1b	-	-	None	2-158
Aug	Stream	-	LME	2	-	None	-	2-163

Note: Dashes indicate not applicable.

4.2.4.12 Total Antimony

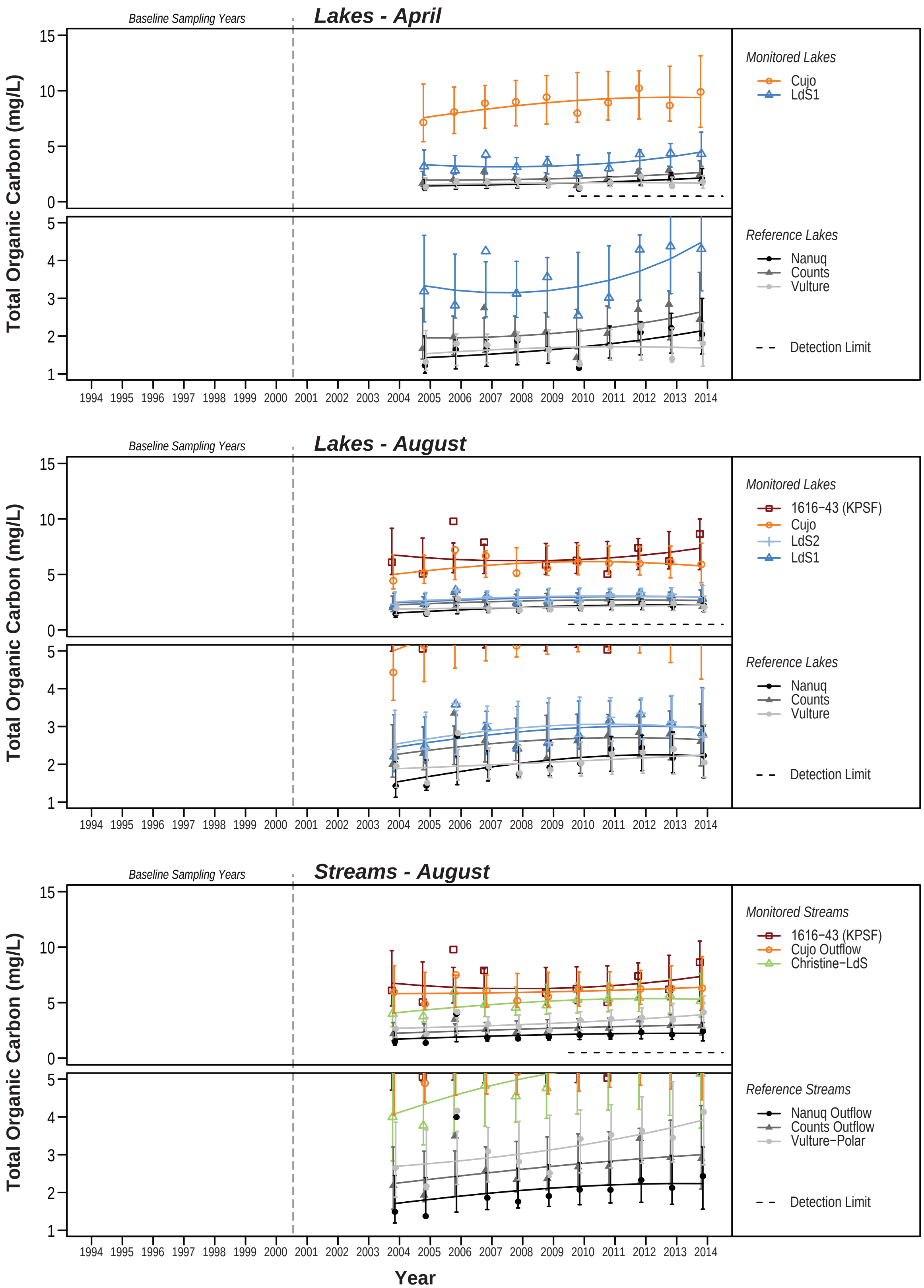
Summary: Total antimony concentrations have generally been below detection limits in all monitored lakes and streams since monitoring began. Observed and fitted mean concentrations were less than the antimony water quality benchmark (0.02 mg/L) at all sites in 2014. No mine effects were detected.

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 4.2-16; Figure 4.2-12). No mine effects were detected.

The 95% confidence intervals around the fitted mean and observed mean total antimony concentrations were below the antimony water quality benchmark of 0.02 mg/L in all reference and monitored lakes and streams in 2014 (Fletcher et al. 1996). Total antimony concentrations were also less than the benchmark concentration in monitored streams in June, July, August, and September 2014 (see Part 2 - Data Report; Fletcher et al. 1996).

Figure 4.2-11

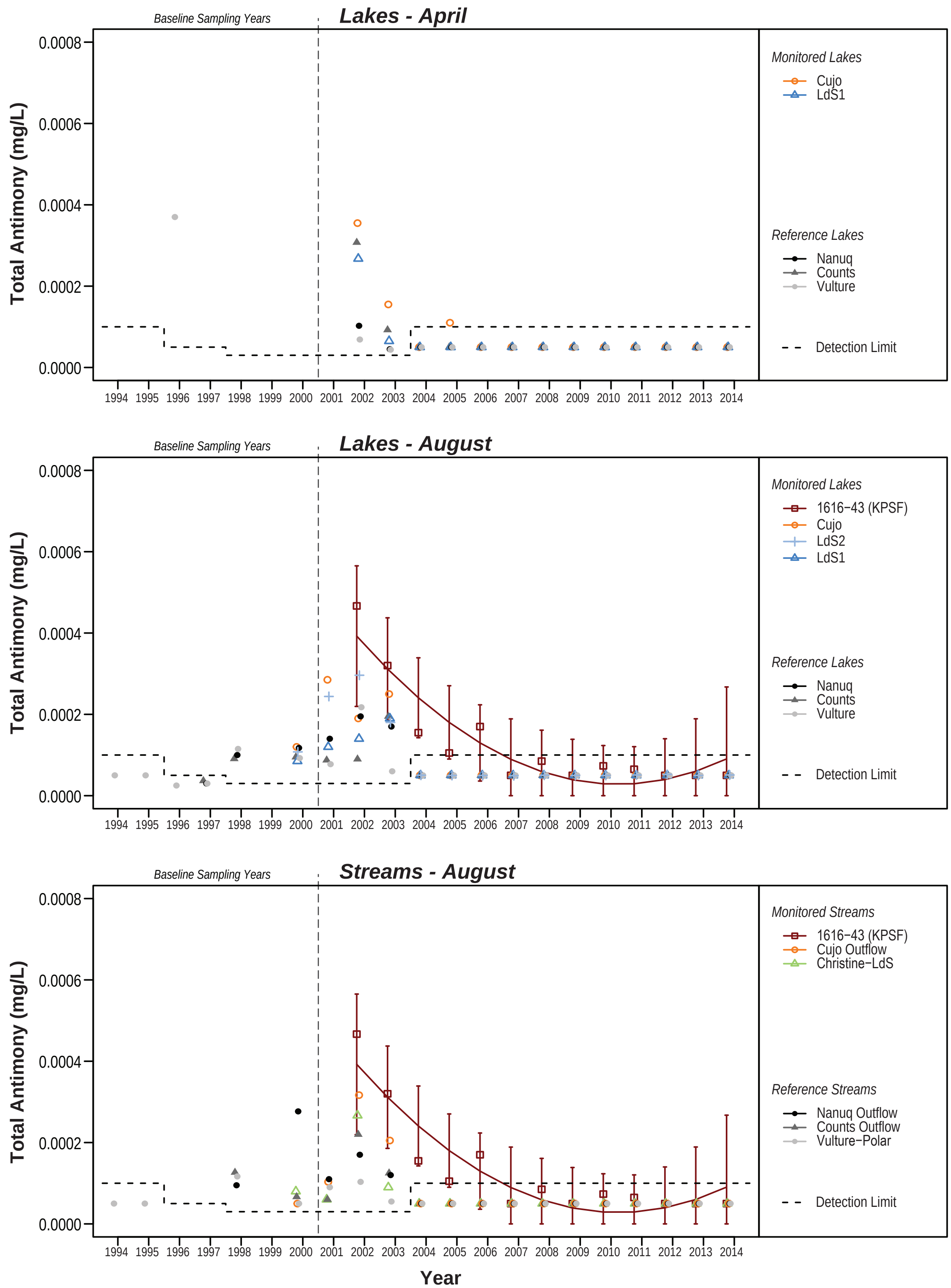
Observed and Fitted Means for Total Organic Carbon in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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 4.2-12

Observed and Fitted Means for Total Antimony in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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 (Fletcher et al. 1996) = 0.02 mg/L.

Table 4.2-16. 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	-	-	-	-	-	2-169
Aug	Lake	Cujo, LdS1, LdS2, Counts, Nanuq, Vulture	Tobit	1a	-	-	1616-43 (KPSF)	2-171
Aug	Stream	Cujo Outflow, Christine-Lac du Sauvage, Counts Outflow, Nanuq Outflow, Vulture-Polar	Tobit	1a	-	-	1616-43 (KPSF)	2-175

Note: Dashes indicate not applicable.

4.2.4.13 Total Arsenic

Summary: Together, 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 2014. Thus, no mine effects were detected.

Statistical analyses indicate that total arsenic concentrations have been stable through time, relative to reference sites, at all monitored lakes and streams in the King-Cujo Watershed and Lac du Sauvage, except in Cujo Lake during the ice-covered season (Table 4.2-17). Graphical analysis suggests that total arsenic concentrations have been elevated, relative to reference sites, in Cujo Lake and Cujo Outflow, but have remained stable over time (Figure 4.2-13). Thus, it was concluded that no mine effects related to arsenic were detected.

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

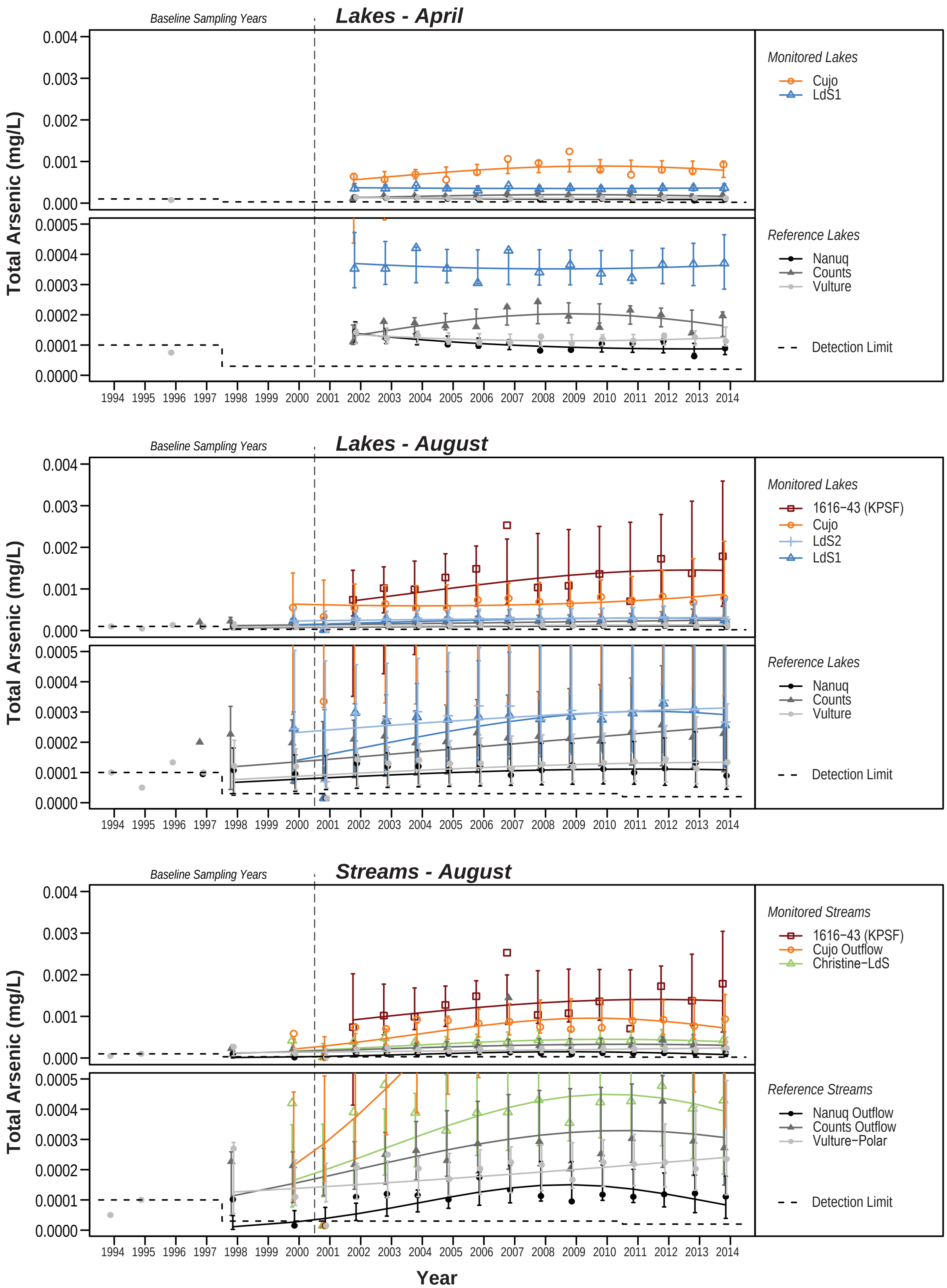
Month	Lake/ Stream	Lakes/Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	-	LME	1b	-	-	Cujo	2-179
Aug	Lake	-	LME	2	-	None	-	2-184
Aug	Stream	-	Tobit	2	-	None	-	2-190

Note: 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 2014 were less than the arsenic CCME guideline value (0.005 mg/L) (CCME 1999c). Total arsenic concentrations were also less than the CCME guideline values in monitored streams in June, July, August, and September 2014 (see Part 2 - Data Report; Fletcher et al. 1996).

Figure 4.2-13

Observed and Fitted Means for Total Arsenic in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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 = 0.005 mg/L.

4.2.4.14 Total Barium

Summary: Statistical and graphical analyses suggest that total barium concentrations have increased in Cujo Lake and Cujo Outflow 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 2014.

Statistical and graphical analyses indicate that total barium concentrations have increased through time, relative to reference sites, in Cujo Lake and Cujo Outflow (Table 4.2-18; Figure 4.2-14). Graphical analysis also indicates elevated, though stable, barium concentrations in the KPSF (Figure 4.2-14). Thus, increases are likely related to mine operations.

Table 4.2-18. 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	-	Cujo	-	2-196
Aug	Lake	-	LME	2	-	Cujo	-	2-201
Aug	Stream	-	LME	2	-	Cujo Outflow	-	2-207

Note: Dashes indicate not applicable.

The 95% confidence intervals around the fitted mean and observed mean total barium concentrations at all monitored sites in 2014 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 2014 were also less than the barium water quality benchmark (see Part 2 - Data Report; Haywood and Drinnan 1983).

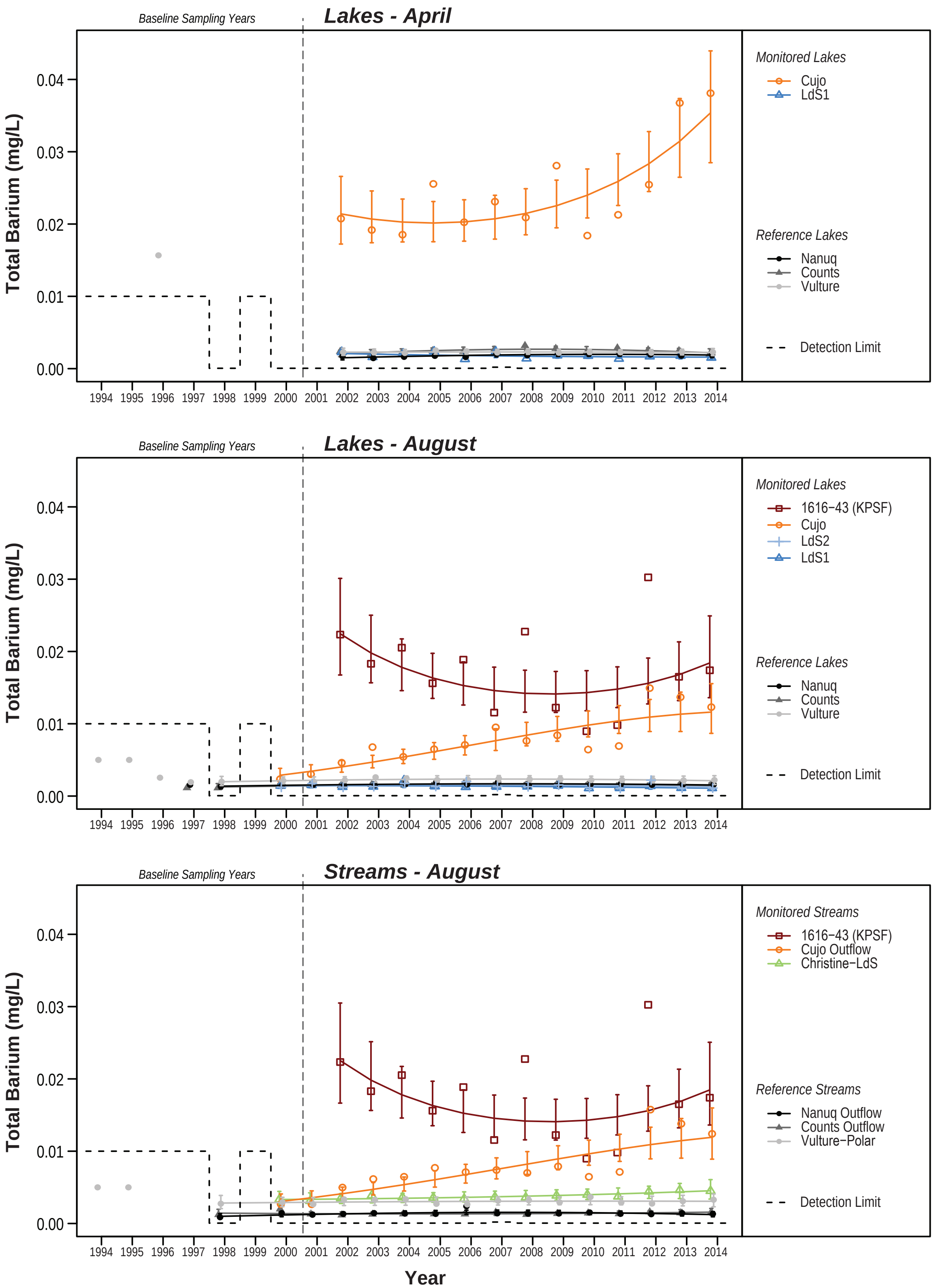
4.2.4.15 Total Boron

Summary: Together, statistical and graphical analyses suggest that total boron concentrations have increased through time as sites downstream of the KPSF as far as Cujo Outflow. All concentrations were less than the CCME guideline in 2014.

Statistical and graphical analyses indicate that total boron concentrations have increased through time, relative to reference sites, in Cujo Lake since monitoring began (Table 4.2-19; Figure 4.2-15). Graphical analysis also suggests that total boron concentrations have increased through time downstream from the KPSF as far as Cujo Outflow, with total boron concentrations decreasing with downstream distance from the KPSF (Figure 4.2-15). Although there appears to be a small increasing trend in total boron concentrations in reference lakes and streams, observed mean concentrations in reference streams are likely artificially elevated in recent years owing to increased detection limits in 2012, 2013 and 2014; more than 80% of all observations in reference streams were below detection limits in 2012, 2013, and 2014.

Figure 4.2-14

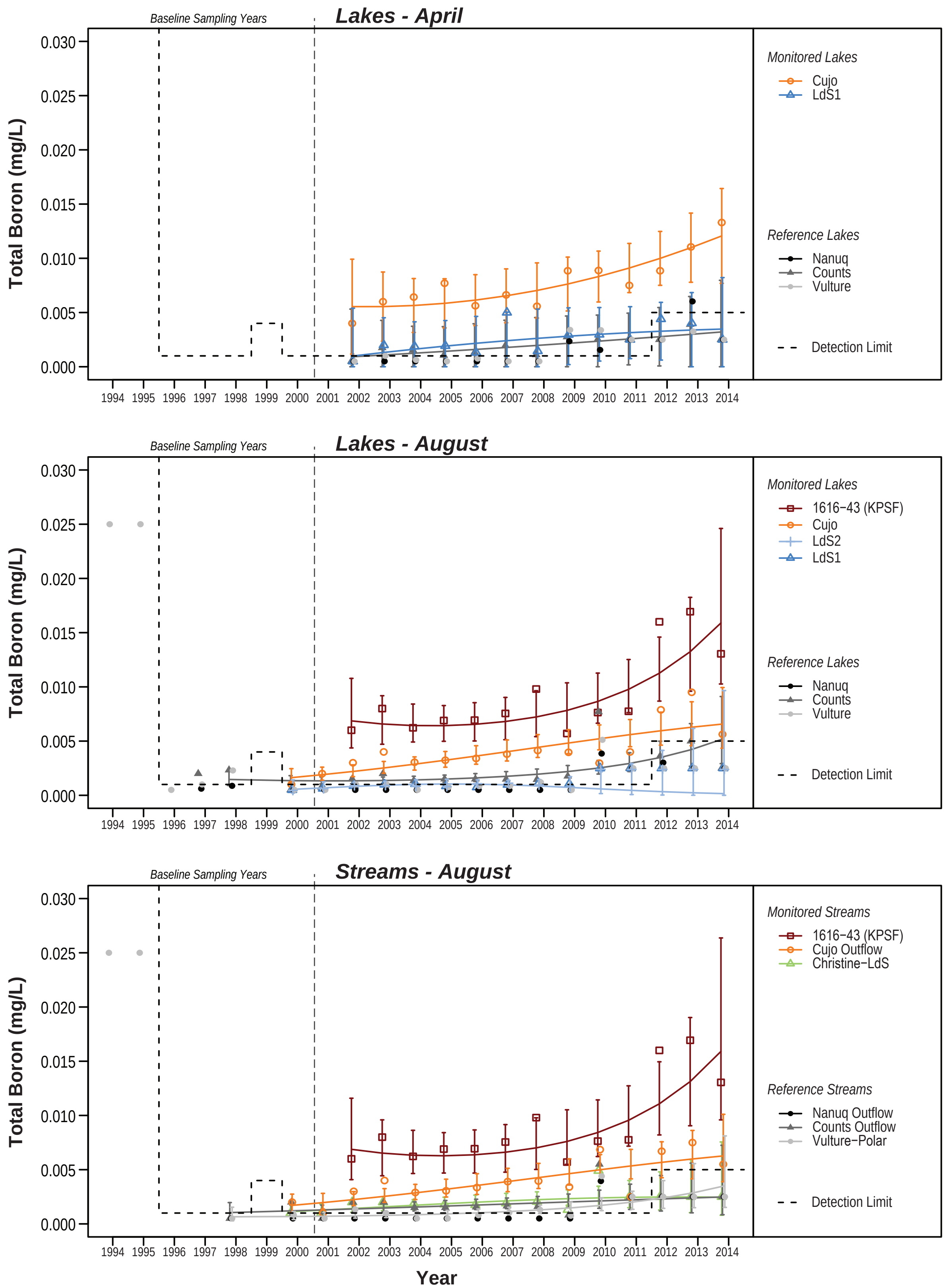
Observed and Fitted Means for Total Barium in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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.

Figure 4.2-15

Observed and Fitted Means for Total Boron in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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.

Table 4.2-19. 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	1b	-	-	Cujo	2-212
Aug	Lake	Nanuq, Vulture, LdS1	Tobit	1b	-	-	1616-43 (KPSF), Cujo	2-216
Aug	Stream	Nanuq Outflow	Tobit	2	-	None	-	2-221

Note: Dashes indicate not applicable.

The 95% confidence intervals around the fitted mean and observed mean total boron concentrations in all monitored lakes and streams in 2014 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 2014 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 detection limits for cadmium in 2014 were less than the hardness-dependent CCME guidelines. No mine effects were detected.

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 4.2-20). 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 2014 (CCME 2014a). Thus it was concluded that no mine effects were detected.

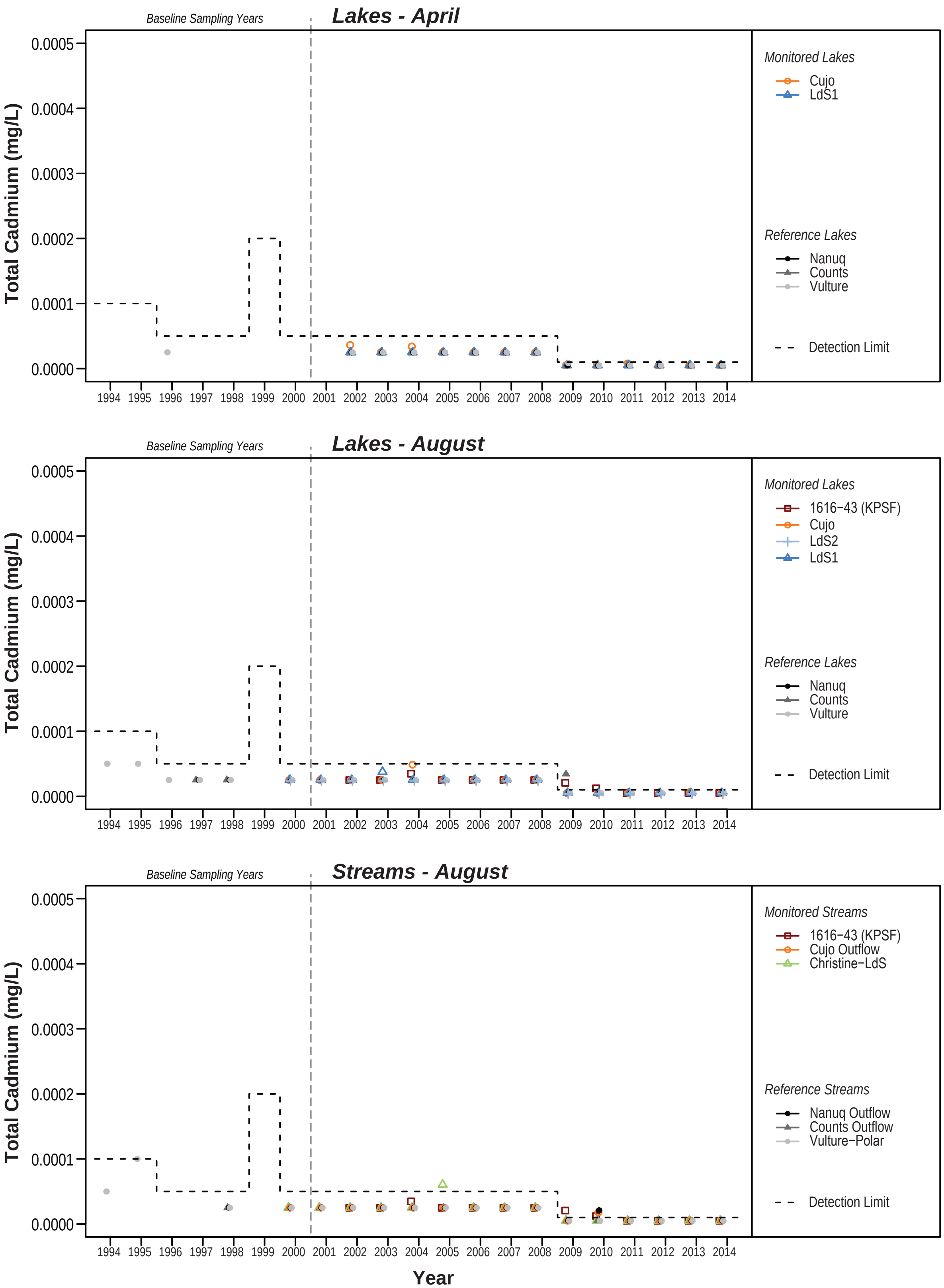
Table 4.2-20. Statistical Results of Total Cadmium Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

Month	Lake/ Stream	Lakes/Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	ALL	-	-	-	-	-	2-227
Aug	Lake	ALL	-	-	-	-	-	2-229
Aug	Stream	ALL	-	-	-	-	-	2-231

Note: Dashes indicate not applicable.

Figure 4.2-16

Observed and Fitted Means for Total Cadmium in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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 = $10^{0.83 X (\log10Hardness - 2.46)}$/1000 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.

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. Total copper concentrations were less than the CCME guideline at all monitored sites in 2014. 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 2014.

Statistical analyses indicate that copper concentrations have changed through time, relative to reference sites, in Cujo Lake and site LdS1 in Lac du Sauvage during the ice-covered season (Table 4.2-21). Graphical analyses indicate that total copper concentrations during the ice-covered season have declined through time in Cujo Lake and have remained stable over time at site LdS1 (Table 4.2-21; 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, 2013, and 2014 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 2014.

Table 4.2-21. 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	2	-	Cujo, LdS1	-	2-233
Aug	Lake	-	Tobit	2	-	1616-43 (KPSF)	-	2-238
Aug	Stream	-	Tobit	2	-	1616-43 (KPSF)	-	2-243

Note: Dashes indicate not applicable.

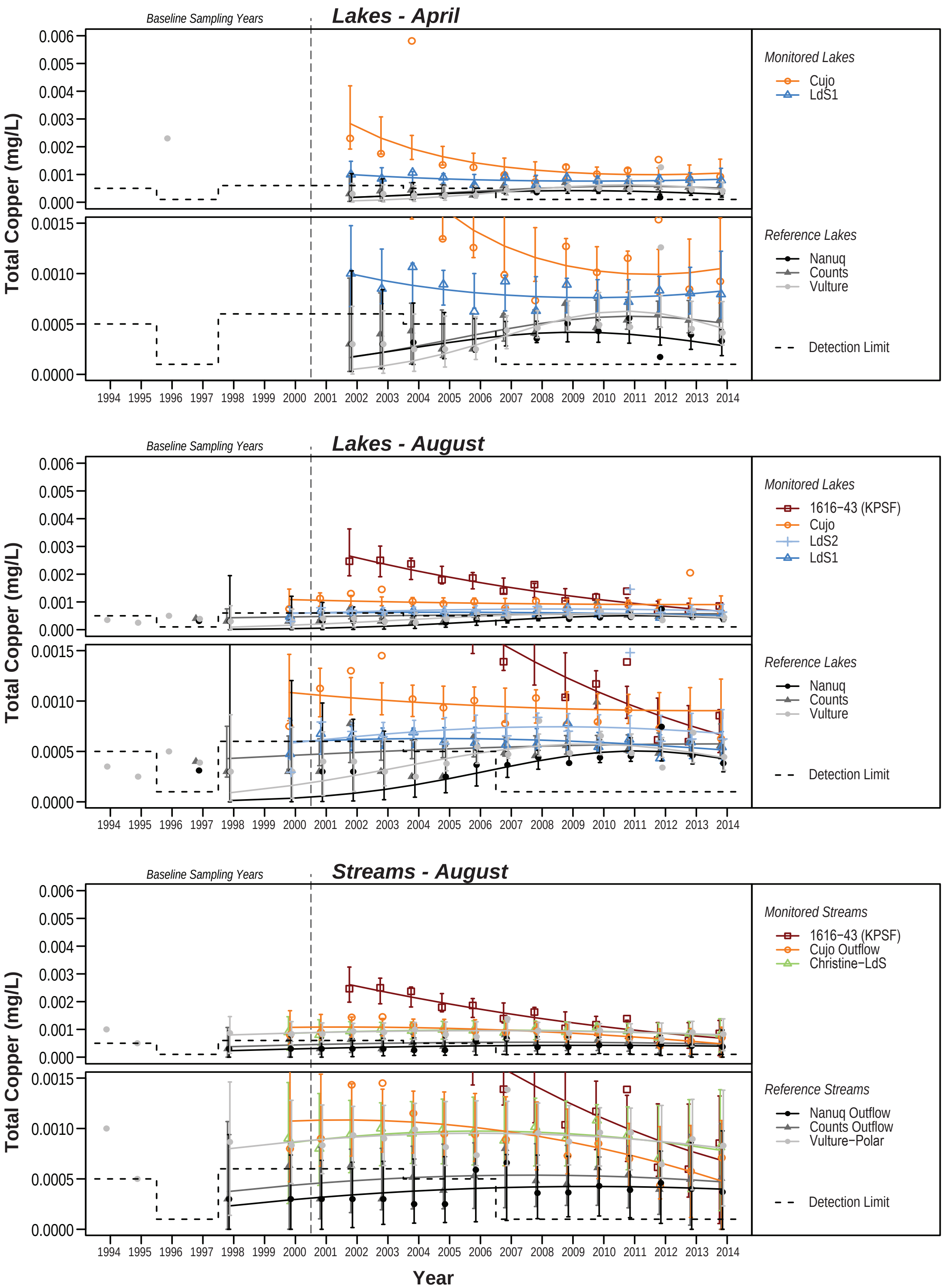
The 95% confidence intervals around the fitted mean and observed mean total copper concentrations in all reference and monitored lakes during both the ice-covered and open water seasons in 2014 were less than hardness-dependent copper CCREM guideline value (see Part 2 - Data Report; CCREM 1987).

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 2014.

Figure 4.2-17

Observed and Fitted Means for Total Copper in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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.

Statistical analyses indicate that total molybdenum concentrations have changed through time in Cujo Lake and Cujo Outflow during the open water season (Table 4.2-22). Graphical analysis suggests that total molybdenum concentrations have been elevated, but stable, in Cujo Lake during the ice-covered season, and that open water season concentrations in Cujo Lake and Cujo Outflow reached a peak around 2008 but have remained above baseline concentrations through 2014 (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 2010, 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 4.2-22; Figure 4.2-18).

Table 4.2-22. 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	Tobit	1a	-	-	None	2-249
Aug	Lake	LdS1, LdS2, Counts, Nanuq, Vulture	Tobit	1a	-	-	Cujo	2-253
Aug	Stream	Counts Outflow, Nanuq Outflow, Vulture Outflow	Tobit	1a	-	-	Cujo Outflow	2-257

Note: Dashes indicate not applicable.

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 2014 were less than the molybdenum SSWQO (19.38 mg/L) (Rescan 2012a). Total molybdenum concentrations in all monitored streams in June, July, August, and September 2014 were also less than the molybdenum SSWQO (see Part 2 - Data Report; Rescan 2012a).

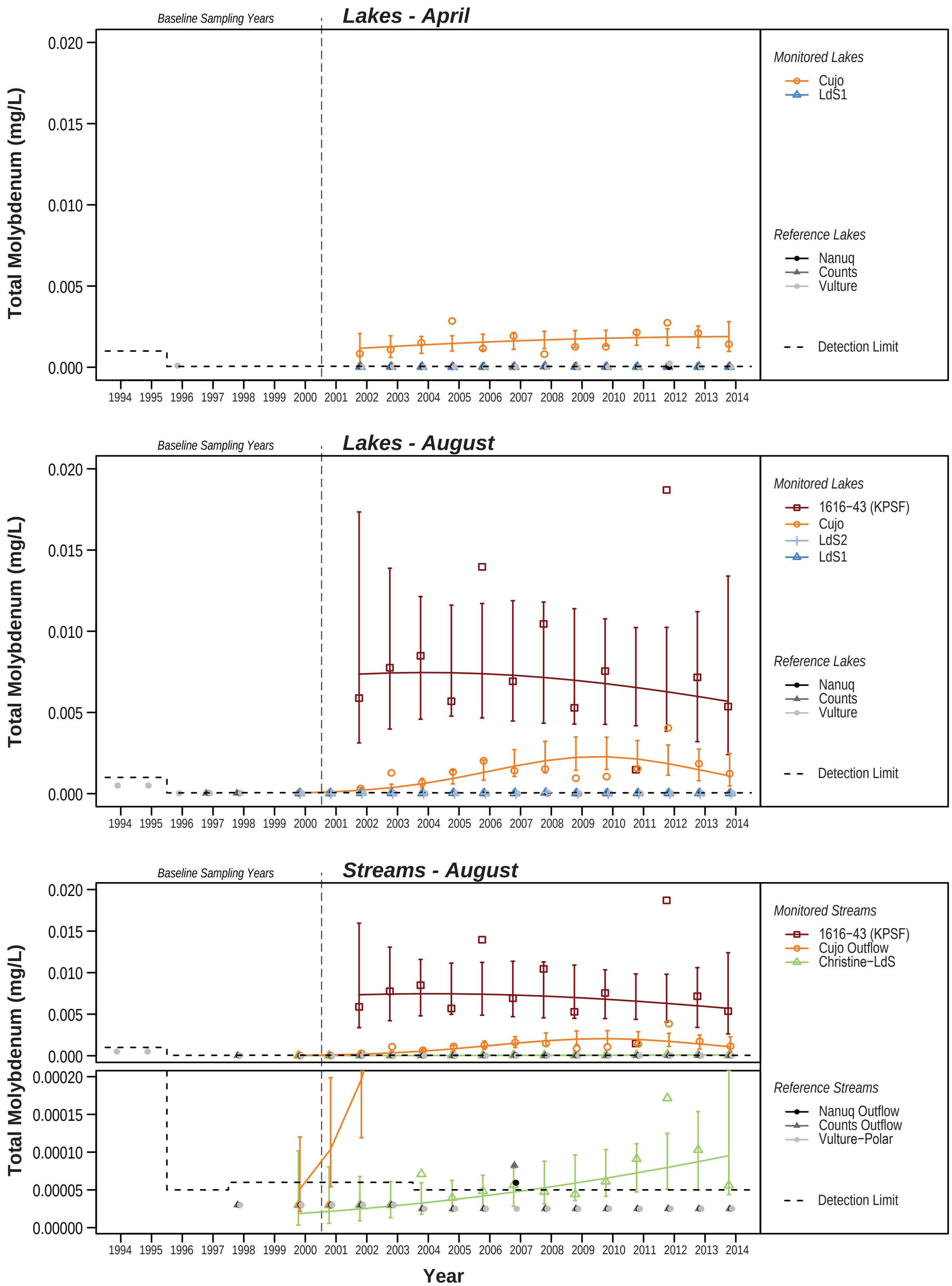
4.2.4.19 Total Nickel

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 2014. Thus, it was concluded that no mine effects were detected.

Statistical analyses indicate that total nickel concentrations have been stable through time, relative to reference sites, in all monitored lakes and streams in the King-Cujo Watershed and Lac du Sauvage (Table 4.2-23; Figure 4.2-19). Graphical analysis indicates that although total nickel concentrations in Cujo Lake, Cujo Outflow, and Christine-Lac du Sauvage have been elevated since monitoring began, concentrations have remained stable and within the range of observed values during baseline years (Figure 4.2-19). Thus, no mine effects were detected.

Figure 4.2-18

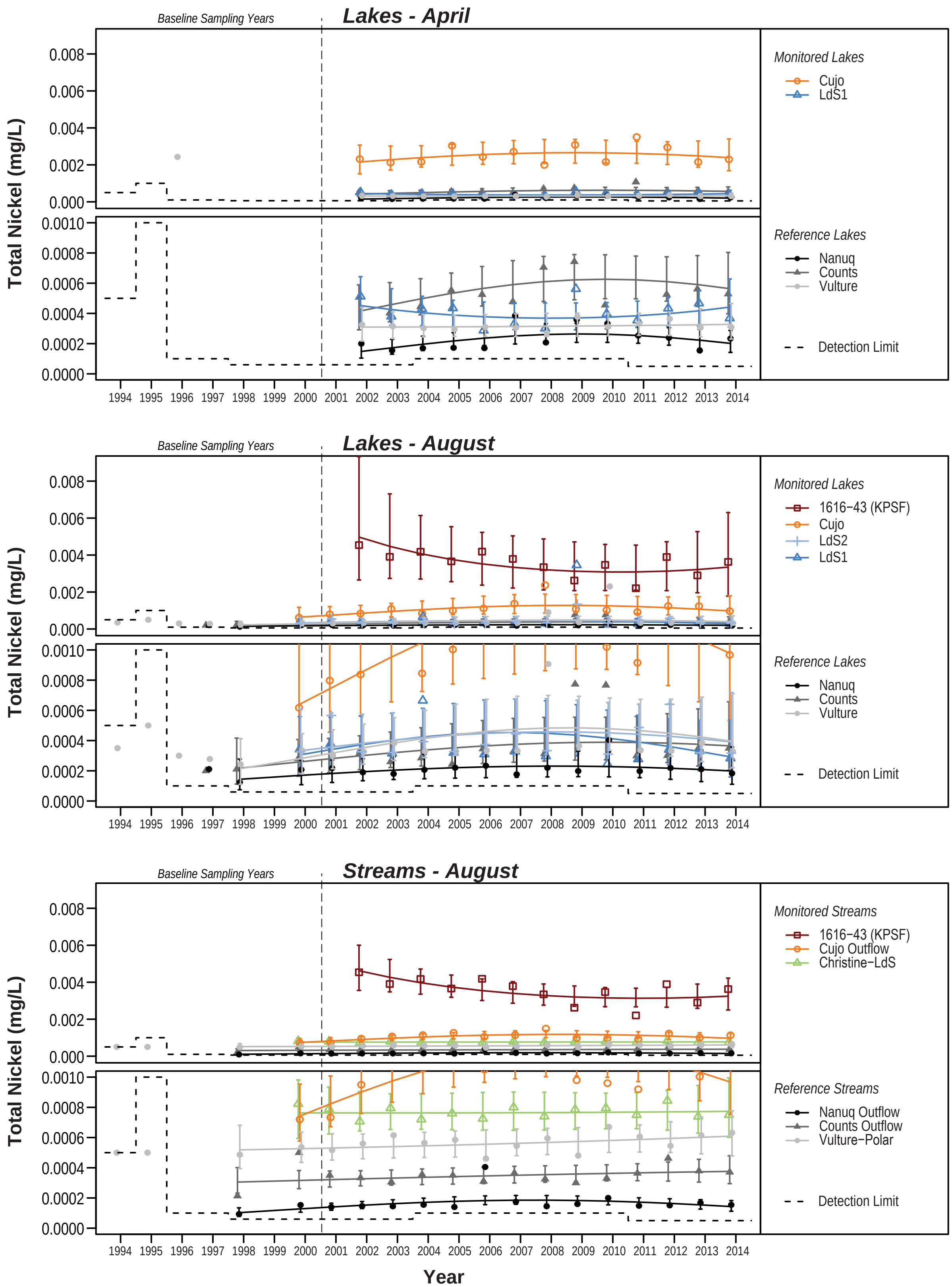
Observed and Fitted Means for Total Molybdenum in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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.

Figure 4.2-19

Observed and Fitted Means for Total Nickel in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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 = $e^{0.76 \times (\ln Hardness) + 1.06}$/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.

Table 4.2-23. 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	-	2-262
Aug	Lake	-	LME	2	-	None	-	2-268
Aug	Stream	-	LME	1b	-	-	1616-43 (KPSF	2-274

Note: Dashes indicate not applicable.

The 95% confidence intervals around the fitted mean and observed mean total nickel concentrations in 2014 were less than the hardness-dependent nickel CCREM guideline value (CCREM 1987). Total nickel concentrations in all streams in June, July, August, and September 2014 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 2014. No mine effects were detected.

Statistical and graphical analyses indicate that, with the exception of Cujo Lake and Cujo Outflow, total selenium concentrations have generally been stable and less than the analytical detection limit through time in monitored and reference lakes and streams (Table 4.2-24; Figure 4.2-20). Graphical analysis indicates that total selenium concentrations in Cujo Lake and Cujo Outflow have remained stable through time (Figure 4.2-20). The 95% confidence intervals around the fitted mean and observed mean total selenium concentrations were less than the selenium CCREM guideline (0.001 mg/L) in all monitored and reference sites in 2014 (see Part 2 - Data Report; CCREM 1987). Thus it was concluded that no mine effects were detected.

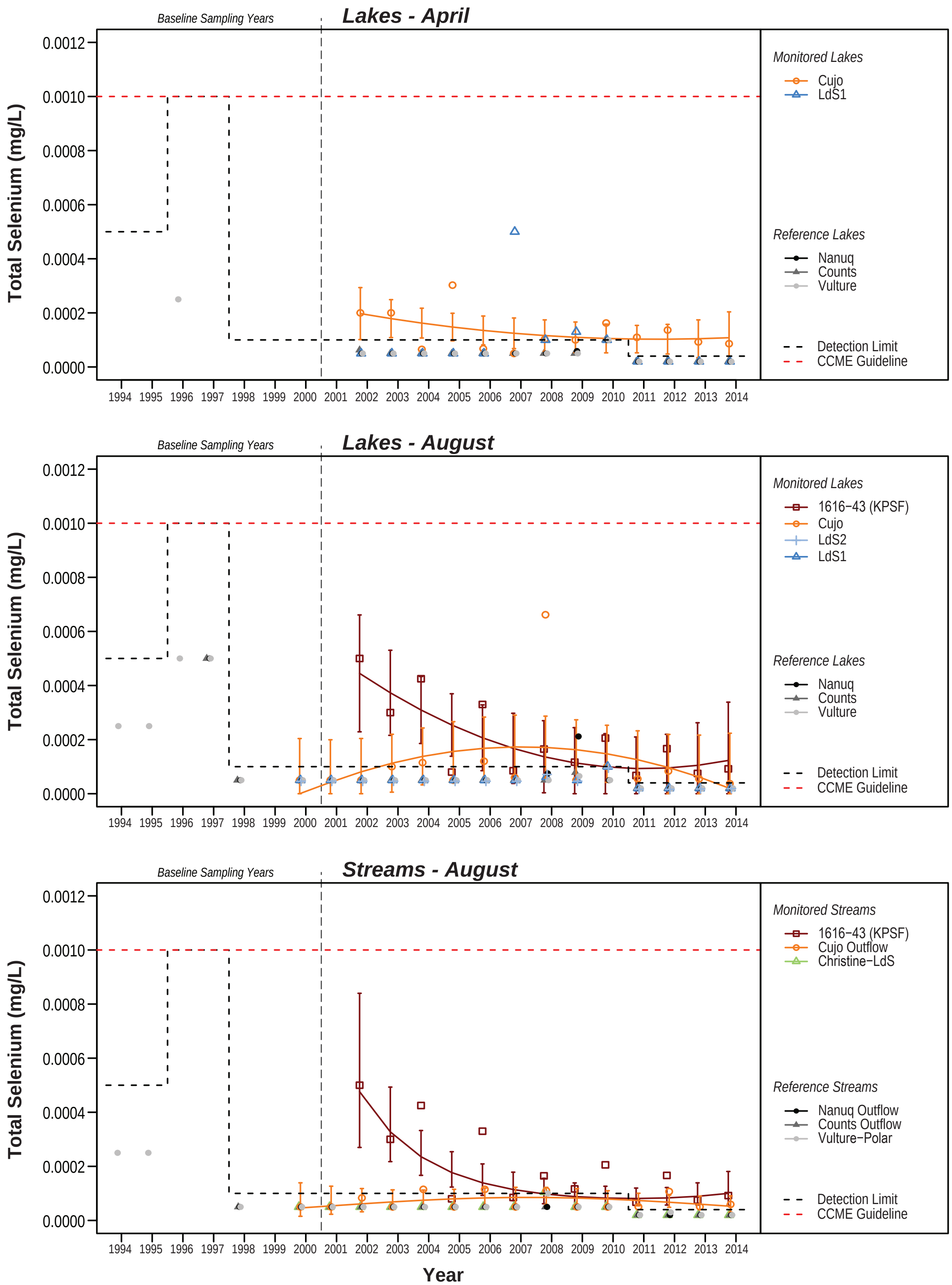
Table 4.2-24. Statistical Results of Total Selenium Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

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	2-280
Aug	Lake	LdS1, LdS2, Counts, Nanuq, Vulture	Tobit	1a	-	-	1616-43 (KPSF)	2-284
Aug	Stream	Christine-Lac du Sauvage, Counts Outflow, Nanuq Outflow, Vulture-Polar	Tobit	1a	-	-	1616-43 (KPSF)	2-288

Note: Dashes indicate not applicable.

Figure 4.2-20

Observed and Fitted Means for Total Selenium in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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 Christine-Lac du Sauvage 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 2014.

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 Christine-Lac du Sauvage (Table 4.2-25; 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 4.2-25. 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	-	2-292
Aug	Lake	-	LME	3	1616-43 (KPSF), Cujo	1616-43 (KPSF), Cujo	-	2-297
Aug	Stream	-	LME	1b	-	-	1616-43 (KPSF), Cujo Outflow, Christine-Lac du Sauvage	2-303

Note: 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 2014 were less than the strontium water quality benchmark (6.242 mg/L; Golder 2011). Total strontium concentrations in all streams in June, July, August, and September 2014 were also less than the strontium water quality benchmark (see Part 2 - Data Report; Golder 2011).

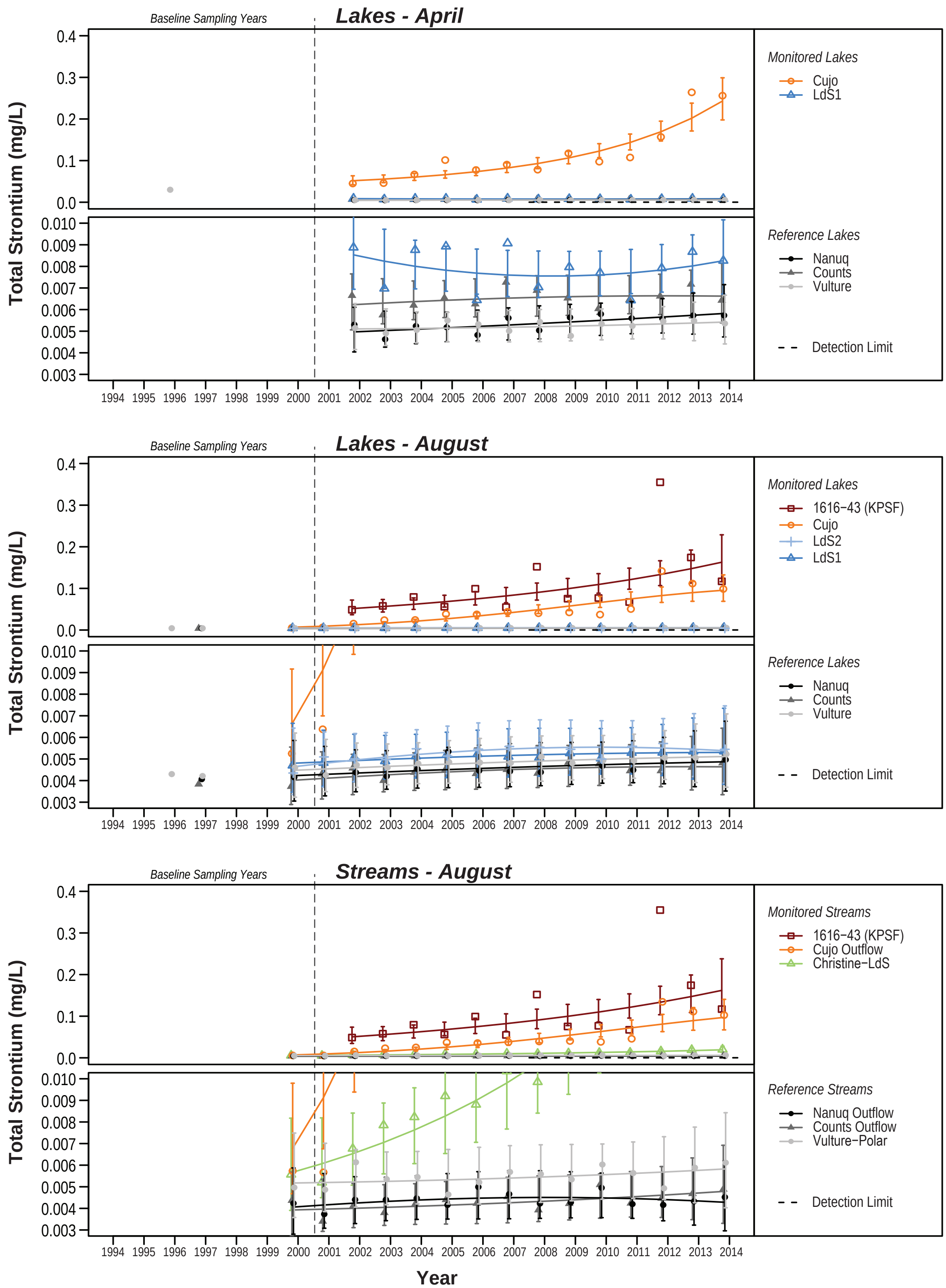
4.2.4.22 *Total Uranium*

> ***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 2014.

Statistical analyses suggest that total uranium concentrations have changed through time, relative to reference sites, in Cujo Lake during the ice-covered season (Table 4.2-26). Graphical analyses suggest that total uranium concentrations have been stable through time, relative to reference sites, in all lakes and streams in the King-Cujo Watershed and Lac du Sauvage (Table 4.2-26; Figure 4.2-22). 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 2014.

Figure 4.2-21

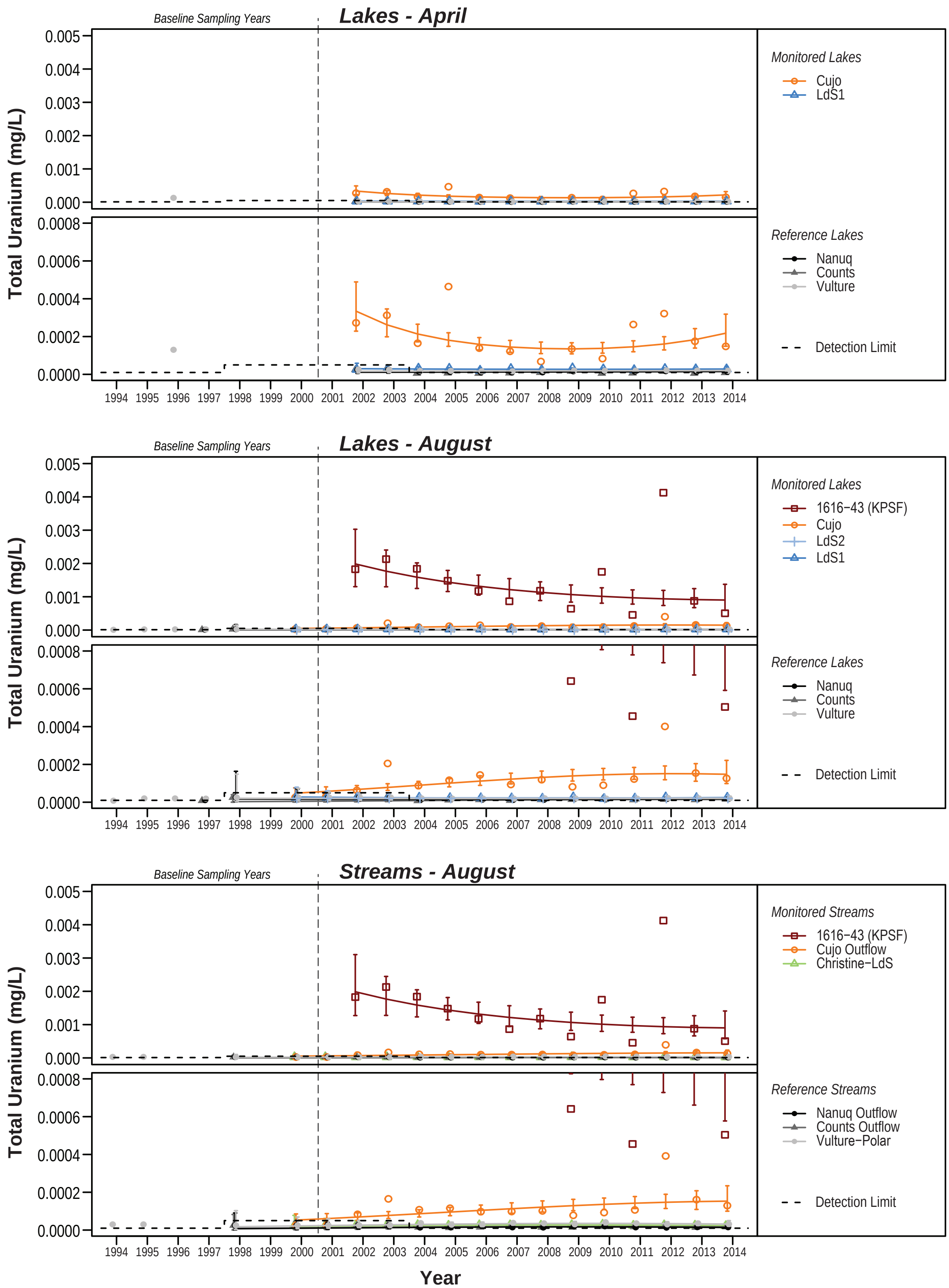
Observed and Fitted Means for Total Strontium in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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 in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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.

Table 4.2-26. 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	-	Cujo	-	2-309
Aug	Lake	-	Tobit	2	-	1616-43 (KPSF)	-	2-314
Aug	Stream	-	Tobit	2	-	1616-43 (KPSF)	-	2-320

Note: Dashes indicate not applicable.

The 95% confidence interval around the fitted mean and the observed mean total uranium concentrations in all lakes and streams during the ice-covered and open water seasons in 2014 were less the CCME uranium guideline (0.015 mg/L; CCME 2011). Total uranium concentrations in all streams in June, July, August, and September 2014 were also less than the CCME uranium guideline (see Part 2 - Data Report; CCME 2011).

4.2.4.23 Total Vanadium

Summary: Total vanadium concentrations have generally been less than detection limits through time. At sites where total vanadium concentrations have been above detection limits, statistical and graphical analyses suggest that concentrations have been stable through time. Total vanadium concentrations were less than SSWQO at all sites in 2014. No mine effects were detected.

Statistical and graphical analyses indicate that total vanadium concentrations have been stable and generally less than the analytical detection limit through time in all monitored and reference lakes and streams (Table 4.2-27; Figure 4.2-23).

Table 4.2-27. Statistical Results of Total Vanadium Concentrations in Lakes and Streams in the King-Cujo Watershed and Lac du Sauvage

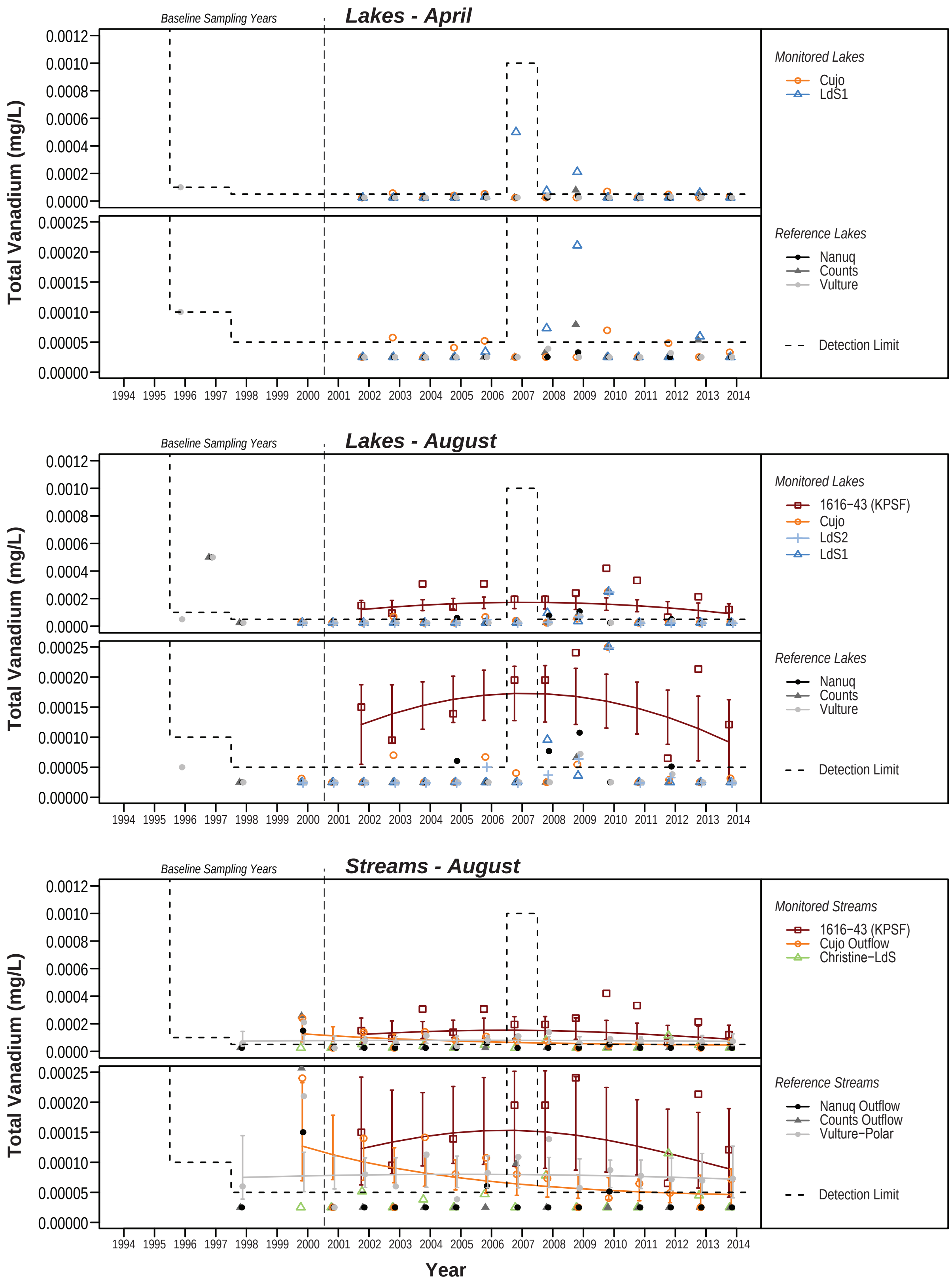
Month	Lake/ Stream	Lakes/Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
					Model Fit = 3	Model Fit = 2	Model Fit = 1	
Apr	Lake	ALL	-	-	-	-	-	2-325
Aug	Lake	Cujo, LdS1, LdS2, Counts, Nanuq, Vulture	Tobit	1a	-	-	None	2-327
Aug	Stream	Counts Outflow, Nanuq Outflow, Christine-Lac du Sauvage	Tobit	1b	-	-	None	2-331

Dashes indicate not applicable.

The 95% confidence interval around the fitted mean and the observed mean total vanadium concentrations in all lakes and streams during the ice-covered and open water seasons in 2014 were less the vanadium SSWQO (0.003 mg/L; Rescan 2012g). Total vanadium concentrations in all streams in June, July, August, and September 2014 were also less than the vanadium SSWQO (see Part 2 - Data Report; Rescan 2012g). No mine effects were detected.

Figure 4.2-23

Observed and Fitted Means for Total Vanadium in King-Cujo Watershed Lakes and Streams and Lac du Sauvage, 1994 to 2014



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.

4.3 LAKE SEDIMENT QUALITY

4.3.1 Variables

Twelve lake sediment quality variables were evaluated for potential effects caused by mine activities in the King-Cujo Watershed and Lac du Sauvage (see Section 3.3.1). CCME guidelines for the protection of aquatic life exist for three of the evaluated sediment quality variables, including arsenic, cadmium, and copper (see Section 2.4; CCME 2014b).

4.3.2 Dataset

The lake sediment quality data used in the 2014 evaluation of effects were collected from late July to mid-August every third year from 1998 to 2014 (Table 4.3.1). Baseline sediment quality data collected from 1994 to 1997 were not used in the statistical evaluation of effects but are included in Table 4.3-1 and shown in Figures 4.3-1 to 4.3-12 for visual comparison. In 1994, sediment quality data were collected in both early July and mid-August, but sediment quality did not differ significantly between these sampling periods. Thus data from early July 1994 to 1999 is included in Table 4.3-1 and shown in Figures 4.3-1 to 4.3-12. Subsequent sampling occurred every three years during the open water season, in early August.

In 2002, the mid depth sediment sampling site in Cujo Lake was moved in order to better capture any potential effects from discharge from the KPSF, though additional spatial variability was introduced into the dataset as a result.

Table 4.3-1. Dataset Used for Evaluation of Effects on Sediment Quality in King-Cujo Watershed Lakes and Lac du Sauvage

Year	Nanuq	Counts	Vulture	Cujo	LdS1
1994*	-	-	Jul-1 (1), Aug-13 (1)	-	-
1997*	Aug-4 (1)	Aug-4 (1)	Aug-4 (1)	-	-
1998	Aug-4 (3)	Aug-4 (2)	Aug-7 (3)	-	-
1999	Jul-30 (3)	Jul-30 (3)	Jul-29 (3)	Jul-31 (3)	-
2000	-	-	-	-	Aug-2 (3)
2001	-	-	-	-	Jul-31 (3)
2002	Aug-3 (3)	Aug-7 (3)	Aug-3 (3)	Aug-7 (3)	Aug-5 (3)
2005	Aug-1 (3)	Aug-7 (3)	Jul-31 (3)	Aug-9 (3)	Aug-9 (3)
2008	Aug-8 (3)	Jul-31 (3)	Aug-5 (3)	Jul-26 (3)	Jul-31 (3)
2011	Aug-13 (3)	Aug-11 (1), Aug-14 (1)	Aug-13 (3)	Aug-25 (3)	Aug-14 (3)
2014	Aug-5 (3)	Aug-9 (3)	Aug-3 (3)	Aug-7 (3)	Aug-8 (3)

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

Sediment sampling methods have been consistent since monitoring began in 1994. Samples were collected using a standard Ekman grab and the top 2 cm were collected for analysis. In 2011 and

2014, sediment samples at all AEMP lake sites were also collected using a K-B corer. Samples from 2011 were collected as part of a study to determine whether core sampling might provide a better measure of potential changes in sediment chemistry (Rescan 2012c). Results from that study will be further examined along with results from 2014 as part of the 2015 AEMP Re-evaluation. ALS has been analyzing the AEMP sediment samples since 1994. Analytical detection limits for sediment quality variables are illustrated as black dashed lines (Figures 4.3-1 to 4.3-12).

Analyses are conducted on sediments collected from one depth strata: mid (5.1 - 10 m). Shallow samples (<5 m) of lake sediment and benthos were eliminated from the AEMP in 2007 because the physical and biological characteristics of the shallow benthic areas of EKATI lakes were too variable to reasonably discriminate potential mine effects from natural variability (Rescan 2006).

For each year in which sediment data were collected, averages were calculated by pooling data from replicates collected within the mid depth strata. Several data points were not included in the analyses either for statistical reasons or as a result of analytical error. These include two statistical outliers (the arsenic concentration from mid depth replicate 3 from Vulture Lake collected on July 31, 2005 and the available phosphorus and total nitrogen concentration from replicate 2 from Cujo Lake collected on August 9, 2005) and the nitrogen data from Cujo Lake in 2008, which was excluded as a result of logistical and analytical error.

4.3.3 Results and Discussion

4.3.3.1 TOC

Summary: Statistical and graphical analyses suggest that mine operations have had no effect on TOC at any of the monitored sediment sites of the King-Cujo Watershed or Lac du Sauvage.

Statistical analyses indicate that TOC percentages in sediments have changed over time, relative to reference lakes, in Cujo Lake (Table 4.3-2). Graphical analysis suggests that TOC percentages have increased in Cujo Lake in recent years; however, a similar pattern was observed in two of the reference lakes (i.e., Nanuq and Vulture lakes). In addition, TOC percentages in Cujo Lake have remained within the range of those observed in reference lakes (Figure 4.3-1). Thus, no mine effects were detected.

Table 4.3-2. Statistical Results for TOC in Sediments 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	
TOC	-	LME	1b	-	-	Cujo	2-336

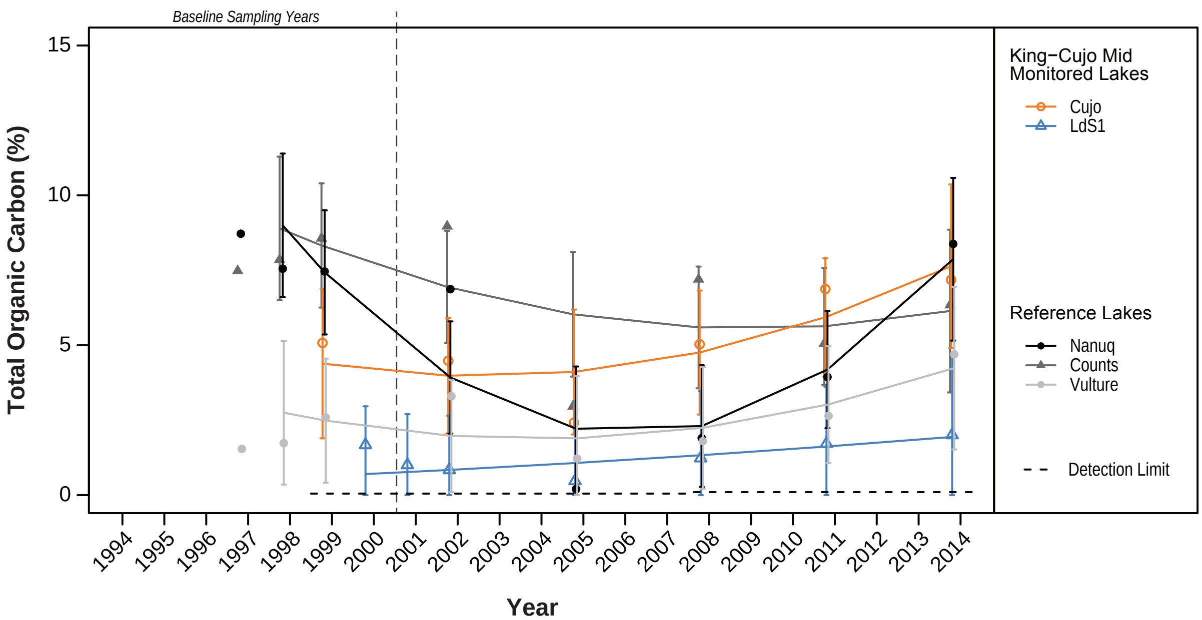
Note: Dashes indicate not applicable.

4.3.3.2 Available Phosphorus

Summary: Statistical and graphical analyses suggest that mine operations have had no effect on available phosphorus concentrations at any of the monitored sediment sites in the King-Cujo Watershed or Lac du Sauvage.

Figure 4.3-1

Observed and Fitted Means for Total Organic Carbon Percentages in Sediments in King-Cujo Watershed Lakes and Lac du Sauvage, 1994 to 2014



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

Statistical and graphical analyses indicate that available phosphorus concentrations in sediments have remained stable through time, relative to reference lakes, at all monitored sites in the King-Cujo Watershed and Lac du Sauvage (Table 4.3-3; Figure 4.3-2). Thus, no mine effects were detected.

Table 4.3-3. Statistical Results for Available Phosphorus Concentrations in Sediments 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	
Available Phosphorus	-	LME	2	-	None	-	2-342

Note: Dashes indicate not applicable.

4.3.3.3 Total Nitrogen

Summary: Statistical and graphical analyses suggest that total nitrogen in sediments of Cujo Lake have increased relative to reference lakes. It is unclear at this time if this change represents natural variability or is the result of mine activities.

Statistical and graphical analyses indicate that total nitrogen percentages have increased through time, relative to reference lakes, in Cujo Lake (Table 4.3-4; Figure 4.3-3). Graphical analysis suggests that total nitrogen percentages have recently increased in one of the reference lakes (i.e., Nanuq Lake); however, current percentages in Nanuq Lake are similar to those observed during baseline years (Figure 4.3-3). Graphical analysis also indicates that total nitrogen percentages in Cujo Lake were within the range of those observed at one of the reference lakes (i.e., Counts Lake), thus current percentages in Cujo Lake may be within the range of natural variability for the area. The cause of the observed increase in percent total nitrogen in sediments of Cujo Lake is unclear at this time as percentages of nitrate-N have been decreasing through time in water quality samples of the KPSF and Cujo Lake.

Table 4.3-4. Statistical Results for Total Nitrogen in Sediments in King-Cujo Watershed Lakes and Lac du Sauvage

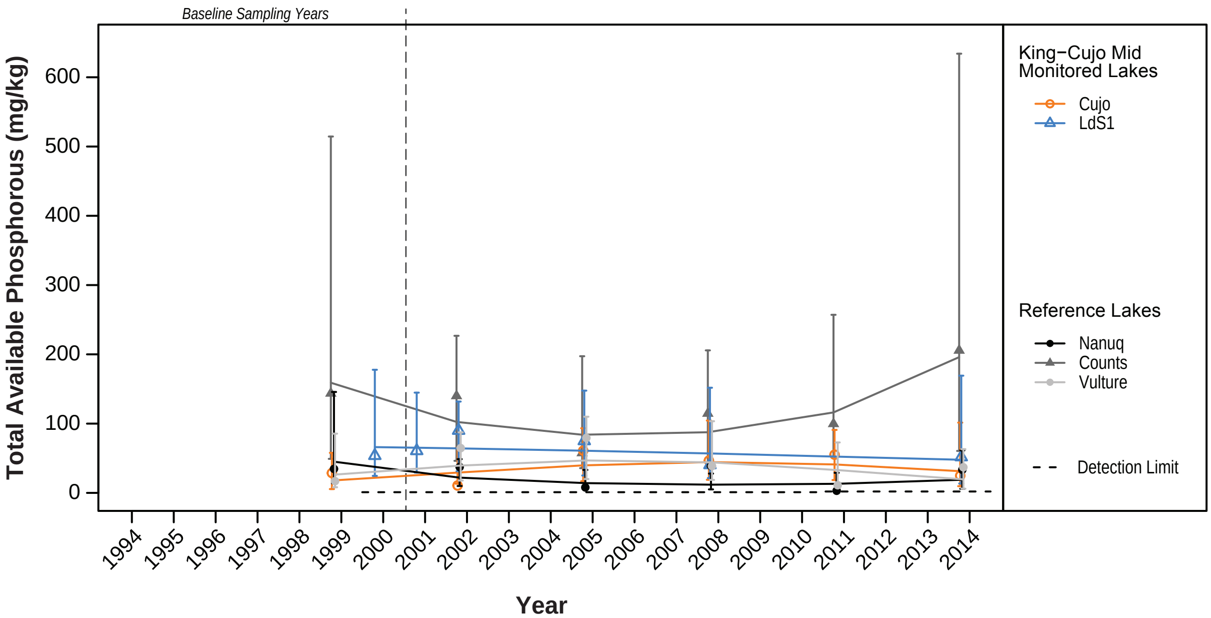
	Lakes/Streams Removed from Analysis	Model Type (LME/Tobit)	Model Fit	Significant Monitored Contrasts			Statistical Report Page No.
				Model Fit = 3	Model Fit = 2	Model Fit = 1	
Total Nitrogen	-	LME	1b	-	-	Cujo	2-348

Note: Dashes indicate not applicable.

4.3.3.4 Total Antimony

Summary: Concentrations of antimony in sediments have been at or below detection limits since monitoring began in 2008. No mine effects were detected.

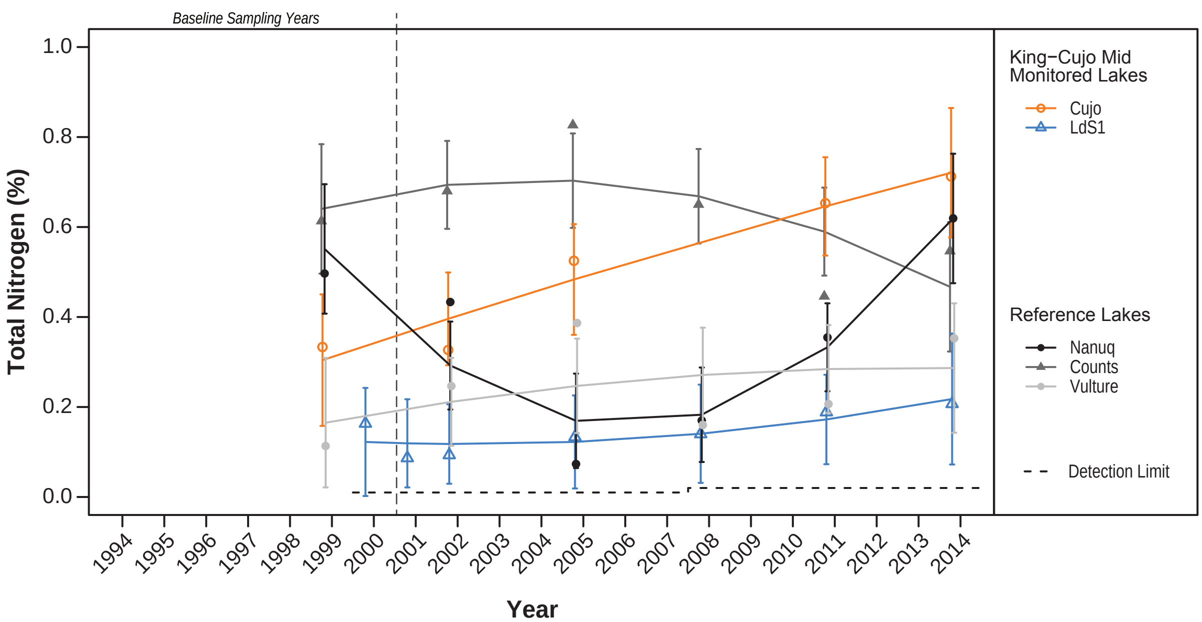
Observed and Fitted Means for Available Phosphorus Concentrations in Sediments in King-Cujo Watershed Lakes and Lac du Sauvage, 1994 to 2014



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 4.3-3

Observed and Fitted Means for Total Nitrogen Percentages in Sediments in King-Cujo Watershed Lakes and Lac du Sauvage, 1994 to 2014



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

Antimony concentrations in sediments have only been analyzed for three years (i.e., 2008, 2011, and 2014). Thus, all lakes were excluded from the statistical analyses and no tests were performed (Table 4.3-5). Graphical analysis and best professional judgment were the primary methods used in the evaluation of effects. Graphical analysis indicates that antimony concentrations in sediments of all monitored and reference lakes have been at or below detection limits since monitoring began in 2008 (Figure 4.3-4). Thus, no mine effects were detected.

Table 4.3-5. Statistical Results for Total Antimony Concentrations in Sediments 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	
Total Antimony	ALL	-	-	-	-	-	2-354

Note: Dashes indicate not applicable.

4.3.3.5 Total Arsenic

***Summary*:** Statistical and graphical analyses suggest that mine operations have had no effect on arsenic concentrations at any of the monitored sediment sites of the King-Cujo Watershed or Lac du Sauvage. The observed mean arsenic concentration exceeded the CCME ISQG and PEL in all monitored sites; however, similar patterns were observed during baseline years and in reference lakes.

Statistical and graphical analyses indicate that arsenic concentrations in sediments have remained stable through time, relative to reference lakes, at all monitored sites in the King-Cujo Watershed and Lac du Sauvage (Table 4.3-6; Figure 4.3-5). The observed means exceeded the CCME ISQG of 5.9 mg/kg in all monitored and reference sites in 2014(CCME 2002). Observed means also exceeded the CCME PEL of 17 mg/kg in all monitored lakes in 2014; however, the observed mean in one reference lake (i.e., Vulture Lake) and the upper 95% confidence interval in two reference lakes (i.e., Counts and Nanuq lakes) also exceeded the CCME PEL in 2014 (CCME 2002). Furthermore, arsenic concentrations in sediments of monitored and reference lakes were greater that the CCME guideline during baseline years. Thus, no mine effects were detected.

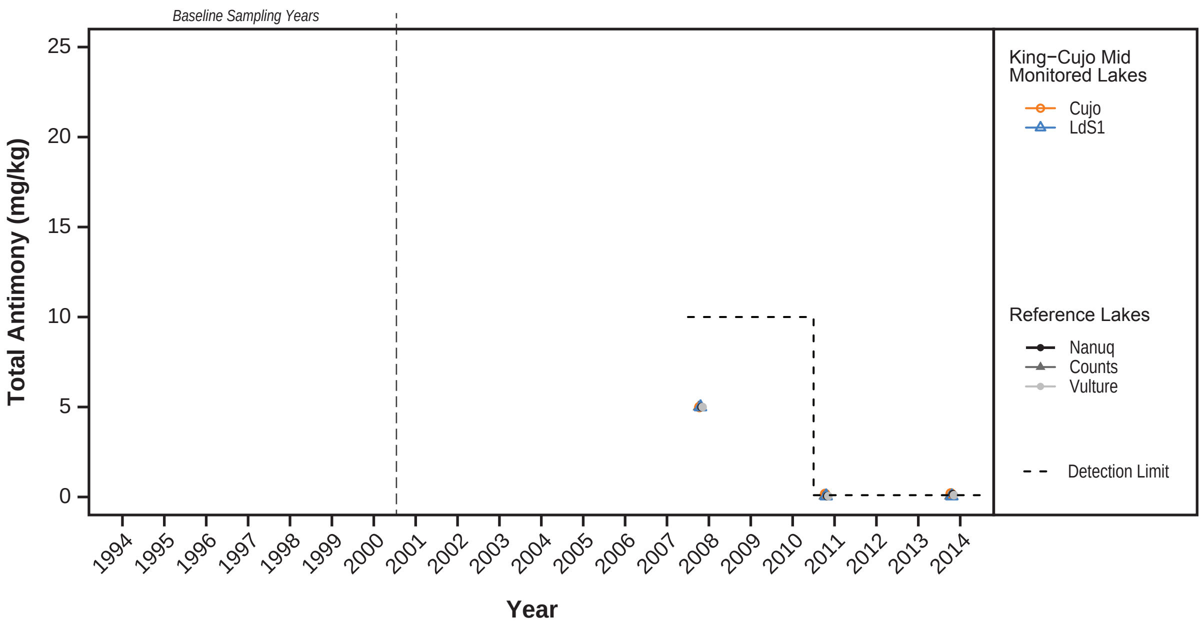
Table 4.3-6. Statistical Results for Total Arsenic Concentrations in Sediments in King-Cujo Watershed Lakes and Lac du Sauvage

	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	
Total Arsenic	-	LME	1b	-	-	None	2-356

Note: Dashes indicate not applicable.

Figure 4.3-4

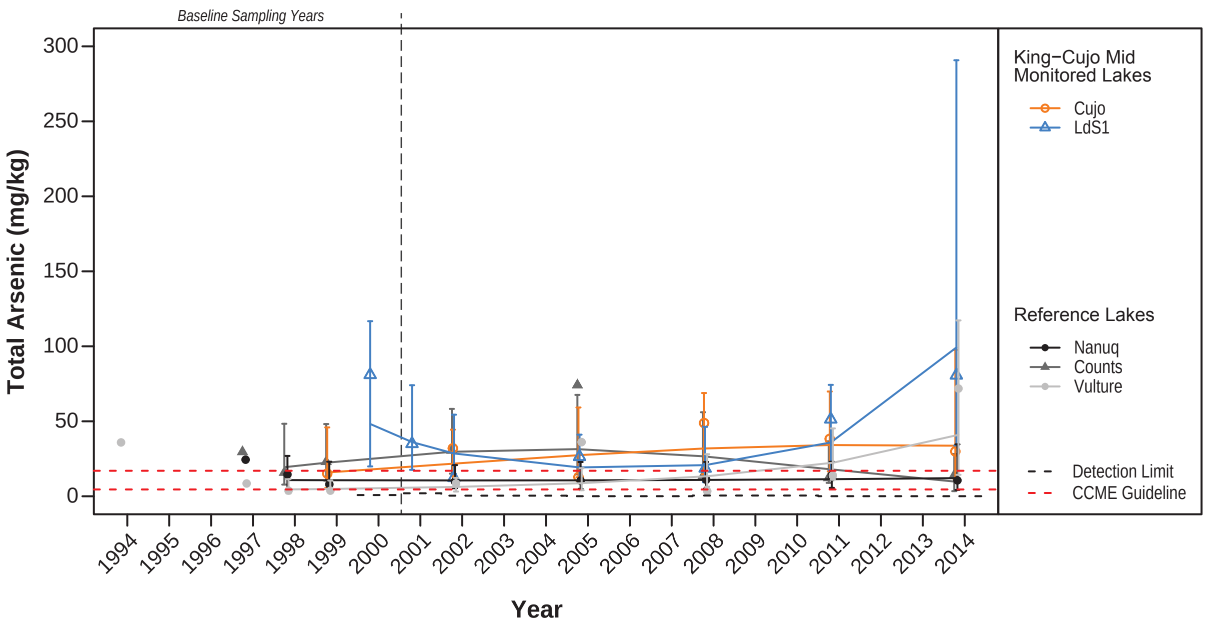
Observed and Fitted Means for Total Antimony Concentrations in Sediments in King-Cujo Watershed Lakes and Lac du Sauvage, 1994 to 2014



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 4.3-5

Observed and Fitted Means for Total Arsenic Concentrations in Sediments in King-Cujo Watershed Lakes and Lac du Sauvage, 1994 to 2014



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 guidelines: ISQG = 5.9 mg/kg; PEL = 17 mg/kg.

4.3.3.6 Total Cadmium

Summary**:** Concentrations of total cadmium in sediments have generally been less than the detection limit in all monitored lakes since monitoring began. Observed mean cadmium concentrations were less than the CCME ISQG and PEL in all monitored and reference sites. No mine effects were detected.

Concentrations of total cadmium in sediments have generally been less than the detection limit in all monitored lakes since monitoring began (Figure 4.3-6). Consequently, all lakes were removed from the statistical analyses (Table 4.3-7). Graphical analysis suggests that observed cadmium concentrations in sediments of monitored lakes may be increasing, but have generally been less than the concentrations observed in reference lakes (Figure 4.3-6). Observed concentrations were less than the CCME ISQG of 0.6 mg/kg and the CCME PEL of 3.5 mg/kg at all monitored sediment sites in 2014 (CCME 1999a). Thus it was concluded that no mine effects were detected.

Table 4.3-7. Statistical Results for Total Cadmium Concentrations in Sediments 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	
Total Cadmium	Cujo, LdS1	-	-	-	-	-	2-361

Note: Dashes indicate not applicable.

4.3.3.7 Total Copper

Summary**:** Statistical and graphical analyses suggest that mine operations have had no effect on total copper concentrations at any of the monitored sediment sites in the King-Cujo Watershed or Lac du Sauvage. The observed mean concentration in Cujo Lake and the 95% confidence interval around the fitted mean for site LdS1 exceeded the CCME ISQG in 2014; however, similar patterns were observed in all reference lakes. Total copper concentrations were less than the CCME PEL at all monitored sites in 2014.

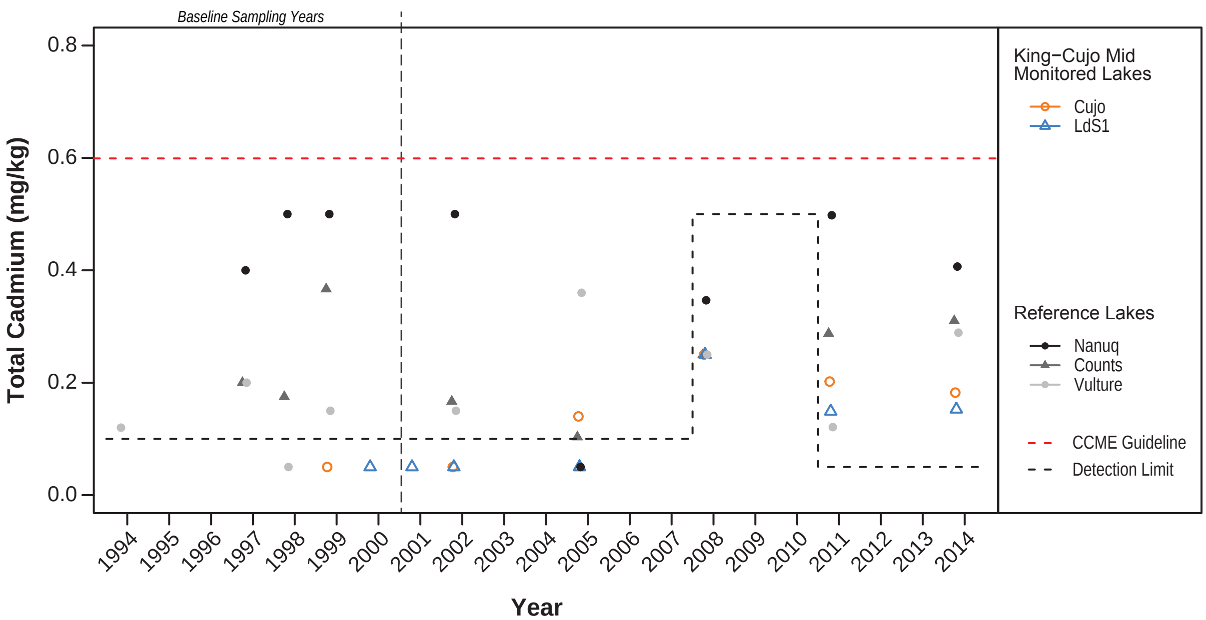
Statistical and graphical analyses suggest that copper concentrations in sediments have been stable through time, relative to reference lakes, at all monitored sites (Table 4.3-8; Figure 4.3-7). The observed mean concentration in Cujo Lake and the 95% confidence interval around the fitted mean for site LdS1 exceeded the CCME ISQG of 35.7 mg/kg in 2014; however, observed means exceeded the ISQG in all reference lakes in 2014 (CCME 2002). The 95% confidence intervals around the fitted means were less than the 197 mg/kg CCME PEL value in all monitored and reference lakes in 2014 (CCME 2002). No mine effects were detected.

Table 4.3-8. Statistical Results for Copper Concentrations in Sediments 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	
Total Copper	-	LME	1b	-	-	None	2-363

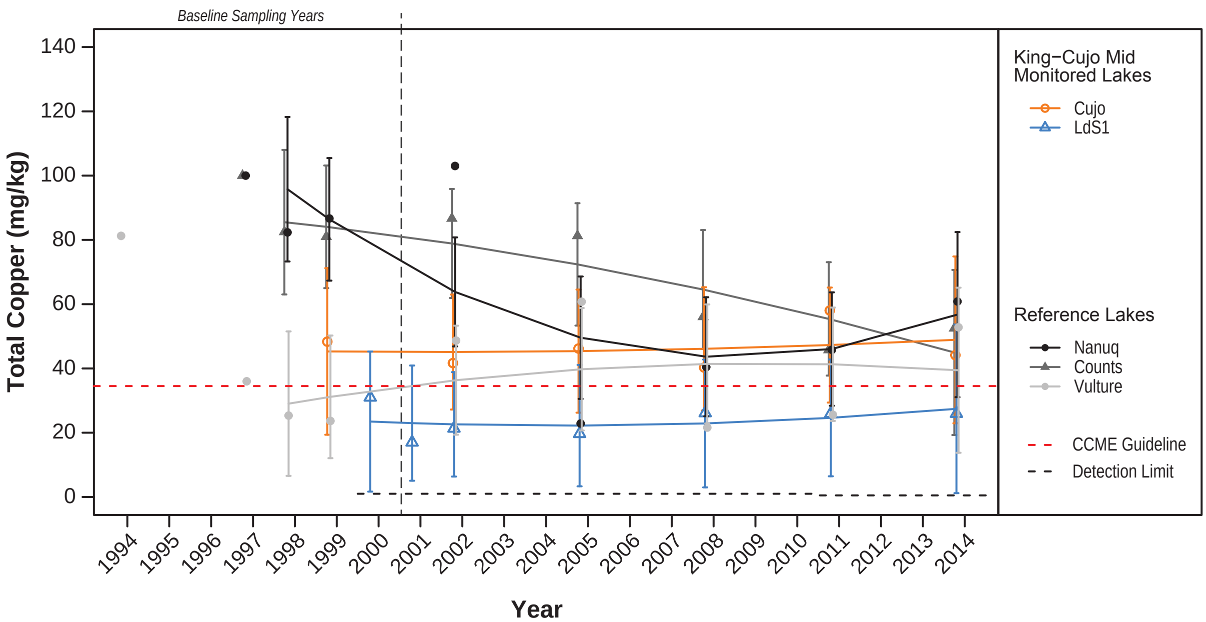
Note: Dashes indicate not applicable.

Observed and Fitted Means for Total Cadmium Concentrations in Sediments in King-Cujo Watershed Lakes and Lac du Sauvage, 1994 to 2014



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 guidelines: ISQG = 0.6 mg/kg; PEL = 3.5 mg/kg (not shown).

Observed and Fitted Means for Total Copper Concentrations in Sediments in King-Cujo Watershed Lakes and Lac du Sauvage, 1994 to 2014



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 guidelines: ISQG = 35.7 mg/kg; PEL = 197 mg/kg (not shown).

4.3.3.8 Total Molybdenum

***Summary*:** No statistical analyses were possible at this time; however, graphical analyses suggest that molybdenum concentrations have increased in sediments of Cujo Lake as a result of mine operations.

Concentrations of total molybdenum in sediments have generally been less than the detection limit in all monitored and reference lakes since monitoring began, with the exception of Cujo Lake (Figure 4.3-8). Although observations in Cujo Lake were often above detection limits, only three years of data with observations greater than the detection limit were available. Consequently, all lakes were removed from the statistical analyses (Table 4.3-9). Graphical analysis and best professional judgment were the primary methods used in the evaluation of effects. Graphical analysis suggests that molybdenum concentrations have been increasing in recent years in sediments of Cujo Lake, with observed concentrations being greater than those observed in reference lakes (Figure 4.3-8). Molybdenum concentrations in sediments followed a similar pattern as observed for total molybdenum concentrations in water quality samples (see Section 4.2.4.18); therefore, increased molybdenum concentrations in sediments likely stem from molybdenum contained in KPSF discharge.

Table 4.3-9. Statistical Results for Total Molybdenum Concentrations in Sediments 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	
Total Molybdenum	ALL	-		-	-	-	2-368

Note: Dashes indicate not applicable.

4.3.3.9 Total Nickel

***Summary*:** Statistical and graphical analyses suggest that mine operations have had no effect on total nickel concentrations at any of the monitored sediment sites in the King-Cujo Watershed or Lac du Sauvage.

Statistical and graphical analyses indicate that total nickel concentrations in sediments have remained stable through time, relative to reference lakes, at all monitored sites in the King-Cujo Watershed and Lac du Sauvage (Table 4.3-10; Figure 4.3-9). Thus, no mine effects were detected.

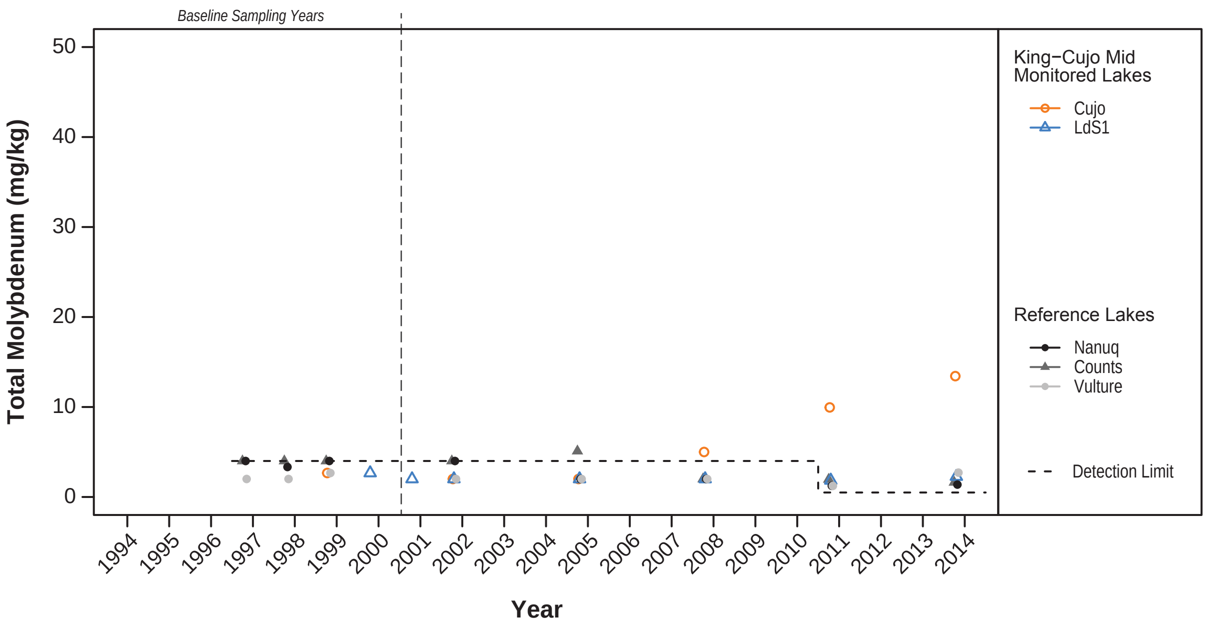
Table 4.3-10. Statistical Results for Total Nickel Concentrations in Sediments in King-Cujo Watershed Lakes and Lac du Sauvage

	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	
Total Nickel	-	LME	2	-	None	-	2-370

Note: Dashes indicate not applicable.

Figure 4.3-8

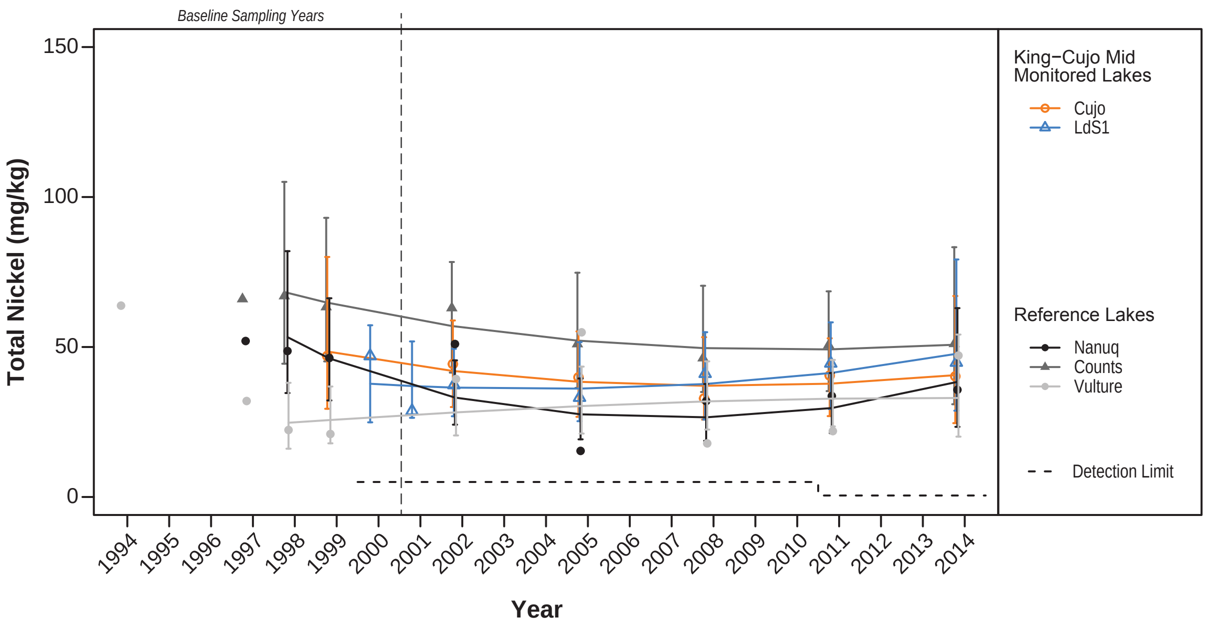
Observed and Fitted Means for Total Molybdenum Concentrations in Sediments in King-Cujo Watershed Lakes and Lac du Sauvage, 1994 to 2014



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 4.3-9

Observed and Fitted Means for Total Nickel Concentrations in Sediments in King-Cujo Watershed Lakes and Lac du Sauvage, 1994 to 2014



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

4.3.3.10 *Total Phosphorus*

> ***Summary*****:** Statistical and graphical analyses suggest that mine operations have had no effect on total phosphorus concentrations at any of the monitored sediment sites in the King-Cujo Watershed or Lac du Sauvage.

Statistical and graphical analyses indicate that total phosphorus concentrations in sediments have remained stable through time, relative to reference lakes, at all monitored sites in the King-Cujo Watershed and Lac du Sauvage (Table 4.3-11; Figure 3.3-10). Thus, no mine effects were detected.

Table 4.3-11. Statistical Results for Total Phosphorus Concentrations in Sediments 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	
Total Nitrogen	-	LME	2	-	None	-	2-376

Note: Dashes indicate not applicable.

4.3.3.11 *Total Selenium*

> ***Summary*****:** Statistical and graphical analyses suggest that mine operations have had no effect on total selenium concentrations at any of the monitored sediment sites in the King-Cujo Watershed or Lac du Sauvage.

Statistical analyses indicate that selenium concentrations have changed through time, relative to reference lakes, in sediments in Cujo Lake (Table 4.3-12). Graphical analysis suggests that selenium concentrations in sediments of Cujo Lake have increased in recent years; however, a similar pattern was observed in two of the reference lakes (i.e., Nanuq and Vulture lakes; Figure 4.3-11).Thus, no mine effects were detected at this time.

Table 4.3-12. Statistical Results for Total Selenium Concentrations in Sediments in 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	
Total Selenium	-	Tobit	1b	-	-	Cujo	2-382

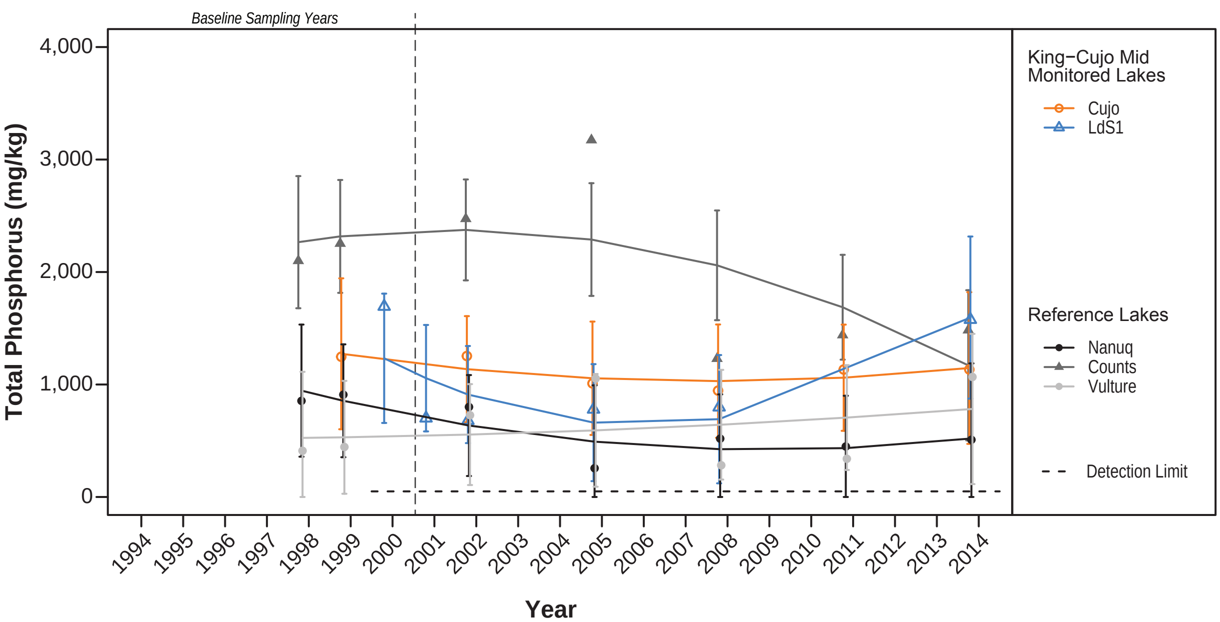
Note: Dashes indicate not applicable.

4.3.3.12 *Total Strontium*

> ***Summary*****:** No statistical analyses were possible at this time; however, graphical analyses suggest that strontium concentrations in Cujo Lake are greater than those observed in reference lakes and may be increasing as a result of mine operations.

Figure 4.3-10

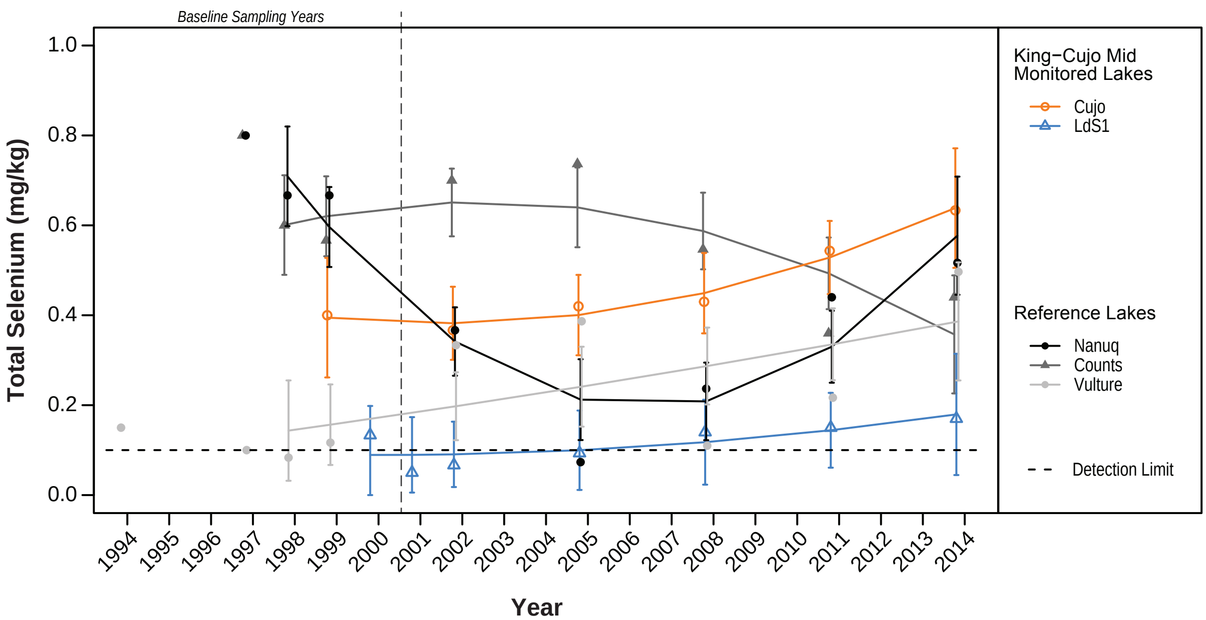
Observed and Fitted Means for Total Phosphorus Concentrations in Sediments in King-Cujo Watershed Lakes and Lac du Sauvage, 1994 to 2014



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 4.3-11

Observed and Fitted Means for Total Selenium Concentrations in Sediments in King-Cujo Watershed Lakes and Lac du Sauvage, 1994 to 2014



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

Strontium concentrations in sediments have only been analyzed for three years (i.e., 2008, 2011, and 2014). Thus, all lakes were excluded from the statistical analyses and no tests were performed (Table 4.3-13). Graphical analysis and best professional judgment were the primary methods used in the evaluation of effects. Graphical analysis suggests that strontium concentrations may be increasing in Cujo Lake sediments since monitoring began in 2008, with observed concentrations being greater than those observed in reference lakes (Figure 4.3-12). Strontium concentrations in sediments follow a similar pattern as observed for total strontium concentrations in water quality samples (see Section 4.2.4.21); therefore, increased strontium concentrations in sediments likely stem from strontium contained in KPSF discharge.

Table 4.3-13. Statistical Results for Total Strontium Concentrations in Sediments 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	
Total Strontium	ALL	-	-	-	-	-	2-388

Note: Dashes indicate not applicable.

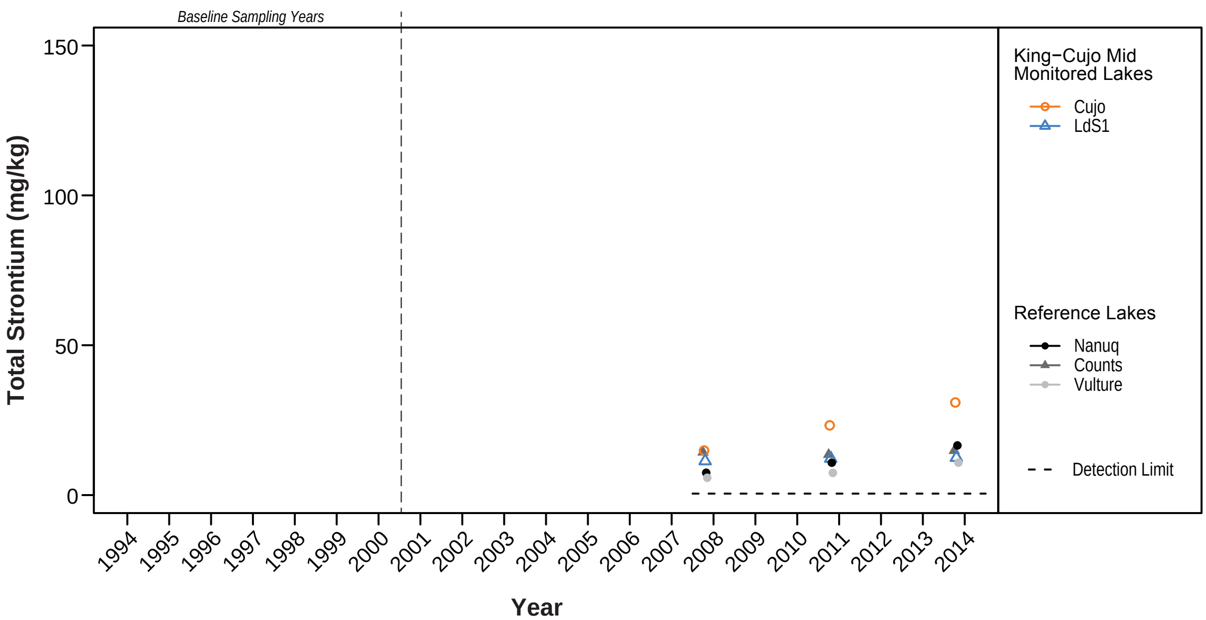
4.4 AQUATIC BIOLOGY

The extent to which changes in water and sediment 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). Benchmarks and CCME guidelines for the protection of aquatic life exist for some water and sediment quality variables (see Sections 2.3 and 2.4). These guidelines and benchmarks provide an important interpretive tool for evaluating the toxicological significance of water and sediment chemistry data. 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 4.2.4). However, with the exception of total phosphate-P, 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 4.2.4). The observed and fitted mean total phosphate-P concentration, and/or the upper 95% confidence intervals around the fitted mean total phosphate-P concentration, exceeded benchmark values in Cujo Lake and sites LdS1 and LdS2 in Lac du Sauvage. However, concentrations in reference lakes also exceeded the applicable benchmark values, suggesting that exceedances are not related to mine activities.

Figure 4.3-12

Observed and Fitted Means for Total Strontium Concentrations in Sediments in King-Cujo Watershed Lakes and Lac du Sauvage, 1994 to 2014



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

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 2012c). As in previous years, concentrations of all the water quality variables in the King-Cujo Watershed and Lac du Sauvage in 2014 remained below the lowest identified chronic effect level for the most sensitive species (Rescan 2012c). 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 King-Cujo Watershed and Lac du Sauvage.

Results from sediment quality analyses in the King-Cujo Watershed and Lac du Sauvage also suggest that changes might be expected in biological communities downstream of the KPSF, because the concentrations of two evaluated sediment quality variable (i.e., total nitrogen and molybdenum) have increased in Cujo Lake and elevated concentrations of one other evaluated sediment quality variables (i.e., strontium) have been detected in Cujo Lake (see Section 3.3.3). However, no CCME guidelines or other relevant benchmark values currently exist for these three sediment quality variables, suggesting that no toxic effects are expected.

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 2012c). Results from the 2013 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 and at site LdS1 in Lac du Sauvage (ERM Rescan 2014a). As the trends in evaluated water quality variables in 2014 were consistent with those observed in 2011 and 2012 (Rescan 2012b, 2013b), there is little reason to expect adverse biological effects in 2014. However, it is expected that the relative availability of macronutrients could continue to be an important driver of change in biological community composition.

4.4.1 Phytoplankton

4.4.1.1 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) and community composition were evaluated to determine whether mine activities have affected phytoplankton communities.

4.4.1.2 Dataset

Phytoplankton have been collected for analysis between late July and early August of each year for the evaluation of effects (Table 4.4-1). Baseline data, which was collected between 1994 and 1997, are included in graphical analysis but not in the statistical evaluation of effects.

Table 4.4-1. 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	-	-
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
2014	Aug-5	Aug-9	Aug-3	Aug-7	Aug-8

Notes: 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 2014 for density and diversity analysis.
** Indicates data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison.*

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.

Phytoplankton taxonomy samples were analyzed by Fraser Environmental Services in Surrey, BC. from 1996 to 2012 and by EcoAnalysts in Moscow, Idaho, U.S.A. from 2013 to present. The methods differed slightly between the two taxonomists: The methods of Fraser Environmental Services included a “rare species scan”, which is not part of the general protocol employed by EcoAnalysts. In 2013, EcoAnalysts was requested to complete a “rare species scan” in addition to their regular protocols in order to make the data comparable among years. The “rare species scan” encompassed a very small fraction of the total phytoplankton abundance and represents species that were not detected in the original subsample used for taxonomic identification, thus resulting in abundance measurements that are recoded as less than detection limit for that taxa. In 2014, the decision was made to discontinue the scan, which required the removal of rare species from the historical dataset. Although no general differences in trends were expected as a result of this change, small variations when comparing to previous AEMP reports may be present.

4.4.1.3 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 4.4-2; Figure 4.4-1). Mean chlorophyll *a* concentrations were less than the mean baseline concentrations ± 2 SD at site LdS1 in 2014 (Table 4.4-3); however, there was no evidence of a decrease in chlorophyll *a* concentration through time. Together, statistical and graphical results suggest that no mine effects were detected.

Table 4.4-2. 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	3	None	-	-	1-390

Note: Dashes indicate not applicable.

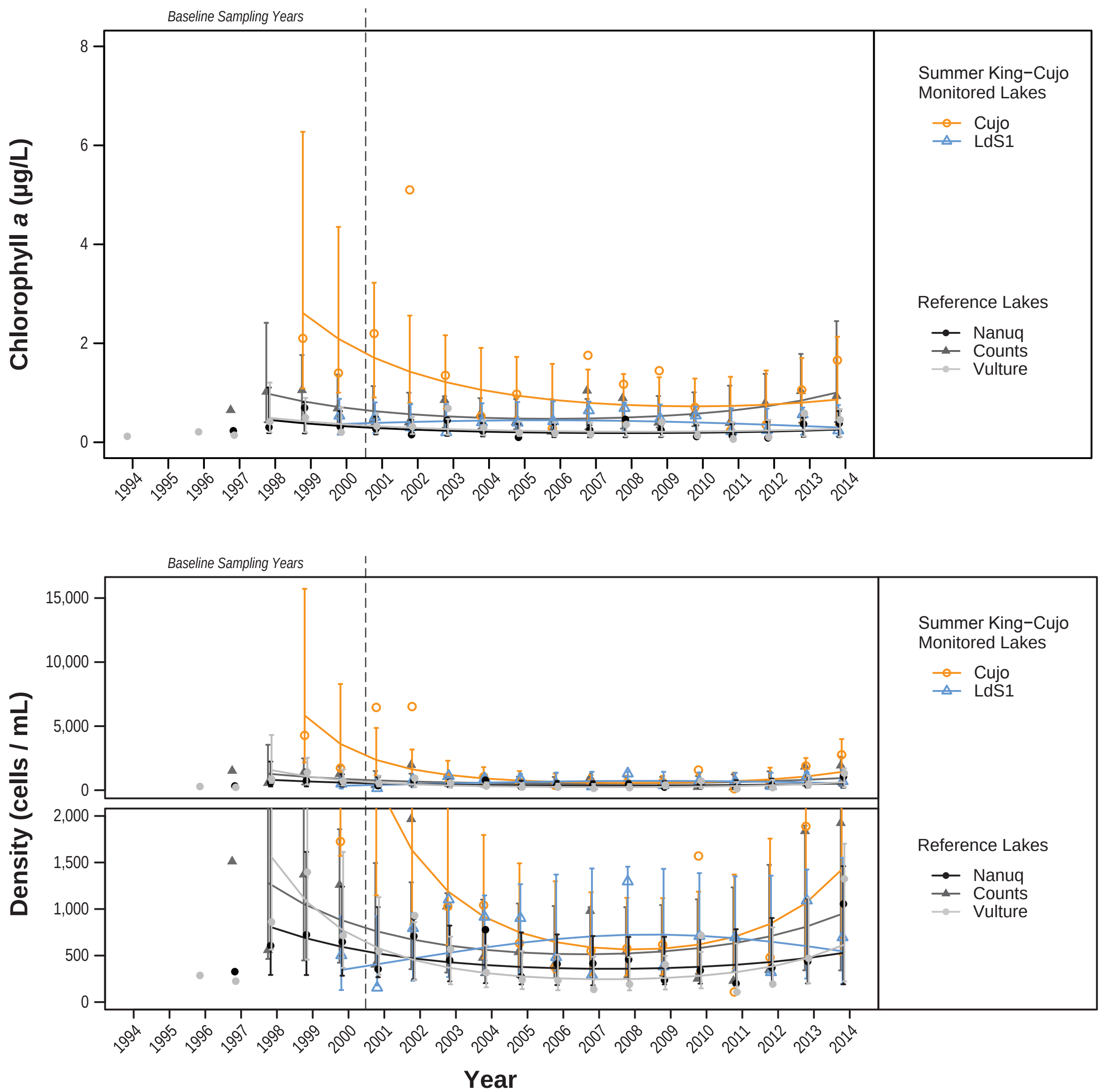
Table 4.4-3. Mean ± 2 Standard Deviations (SD) Baseline Concentrations of Chlorophyll *a* in Each of the King-Cujo Watershed Lakes and Lac du Sauvage

Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD
Nanuq	0.39 (4)	0 - 0.81	0.38 ± 0.37
Counts	0.85 (4)	0.28 - 1.42	0.93 ± 0.35
Vulture	0.29 (6)	0 - 0.62	0.46 ± 0.24
Cujo	1.75 (2)	0.93 - 2.57	1.66 ± 1.11
LdS1	0.54 (1)	0.39 - 0.68	0.24 ± 0.03

Notes: Units are µg/L.
Negative values were replaced with zeros.
N = number of years data were collected.

Figure 4.4-1

Observed and Fitted Means for Chlorophyll *a* Concentrations and Phytoplankton Density in King-Cujo Watershed Lakes and Lac du Sauvage, 1994 to 2014



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 phytoplankton densities have been stable through time, relative to reference lakes, in Cujo Lake but not at site LdS1 in Lac du Sauvage (Table 4.4-4). However, graphical analysis suggests that phytoplankton densities have been relatively stable through time at site LdS1 (Figure 4.4-1). Graphical analysis also suggests that phytoplankton density in Cujo Lake decreased from initially high values starting around 2003, but that current densities are within the range of those observed during baseline years. Moreover, phytoplankton densities in Cujo Lake and at site LdS1 in 2014 remained within ± 2 SD of the mean observed phytoplankton densities in baseline years (Table 4.4-5). Thus, no mine effects were detected with respect to phytoplankton density.

Table 4.4-4. 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	Cujo, LdS1	LdS1	-	2-395

Note: Dashes indicate not applicable.

Table 4.4-5. 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	2014 Mean ± 1 SD
Nanuq	576 (4)	0 – 1,332	1,055 ± 346
Counts	1,175 (4)	0 – 2,743	1,924 ± 755
Vulture	697 (5)	0 – 2,067	1,324 ± 398
Cujo	3,002 (2)	0 – 8,669	2,765 ± 1,678
LdS1	503 (1)	0 – 1,263	696 ± 100

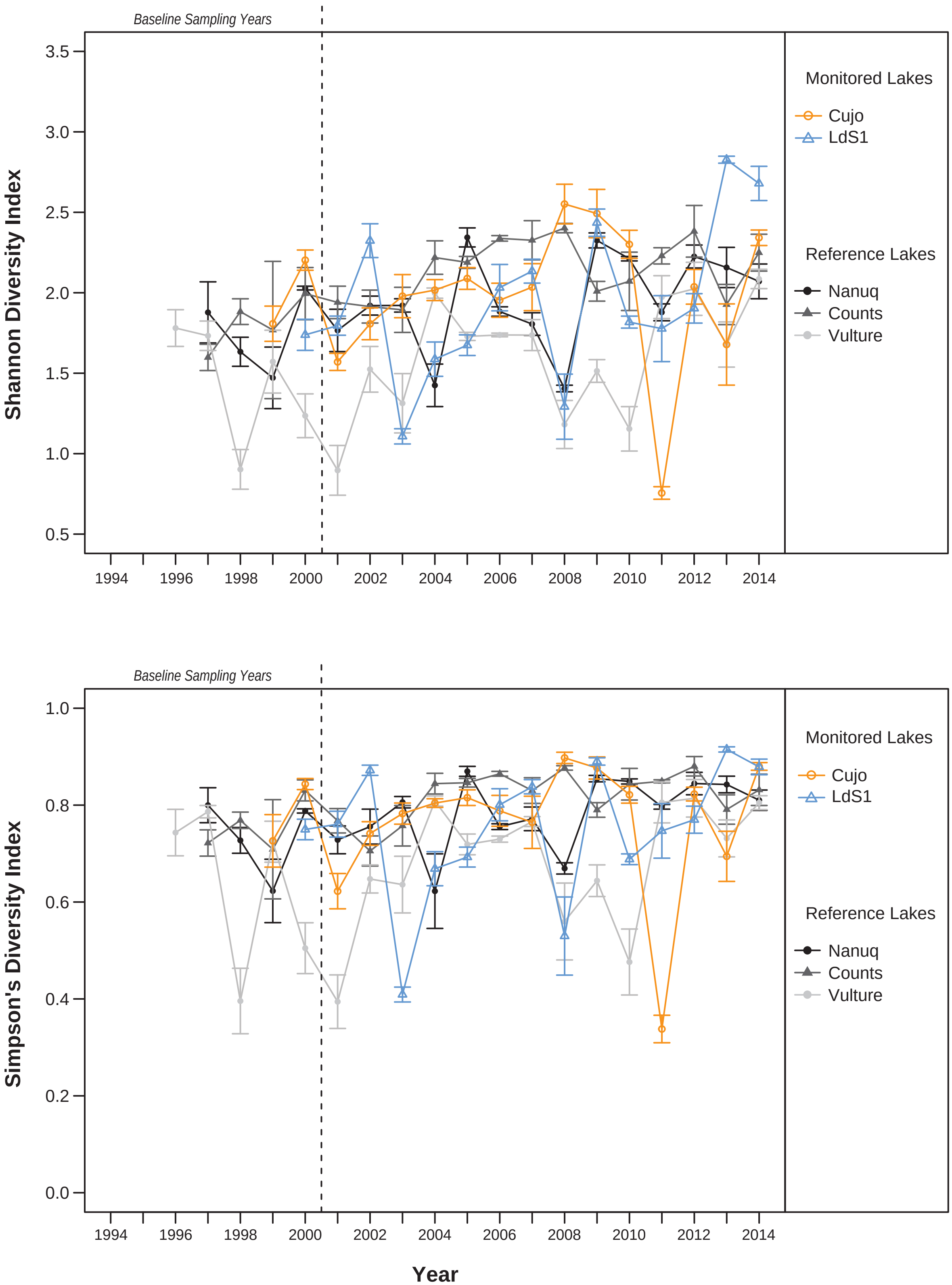
Notes: Units are cells/mL.
Negative values were replaced with zeros.
N = number of years data were collected.

Diversity and Community Composition

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 4.4-2) 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 4.4-3 to 4.4-4). Following recent advances in taxonomic classification, the names of two phytoplankton groups have been updated since 2012: the Cyanophyta are now recognized as the class Myxophyceae and the Pyrrophyta are now recognized as the class Dinophyceae.

Figure 4.4-2

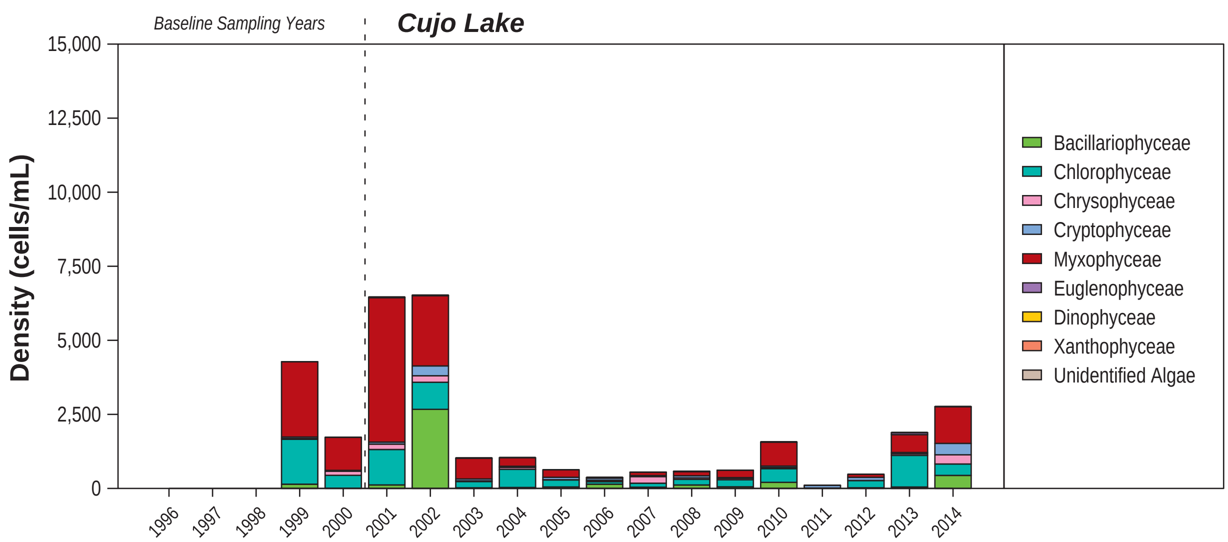
Average Diversity Indices for Phytoplankton in King-Cujo Watershed Lakes and Lac du Sauvage, 1996 to 2014



Notes: *Symbols represent observed mean values.*
Error bars indicate standard error of the observed means.

Figure 4.4-3

Average Phytoplankton Density by Taxonomic Group for Lakes of the King-Cujo Watershed and Lac du Sauvage, 1996 to 2014



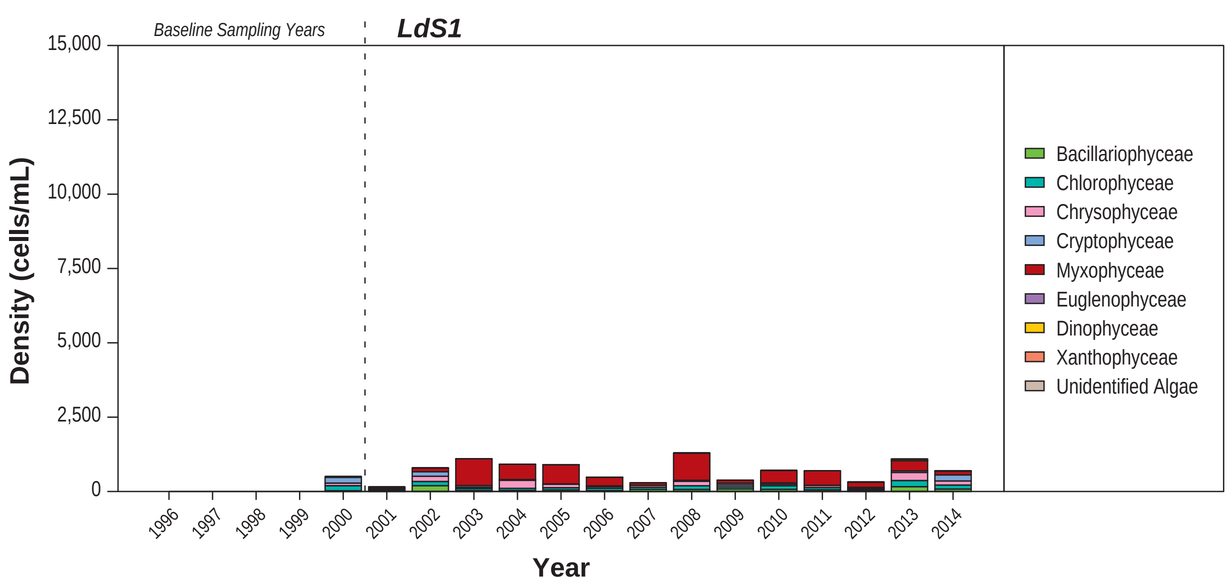
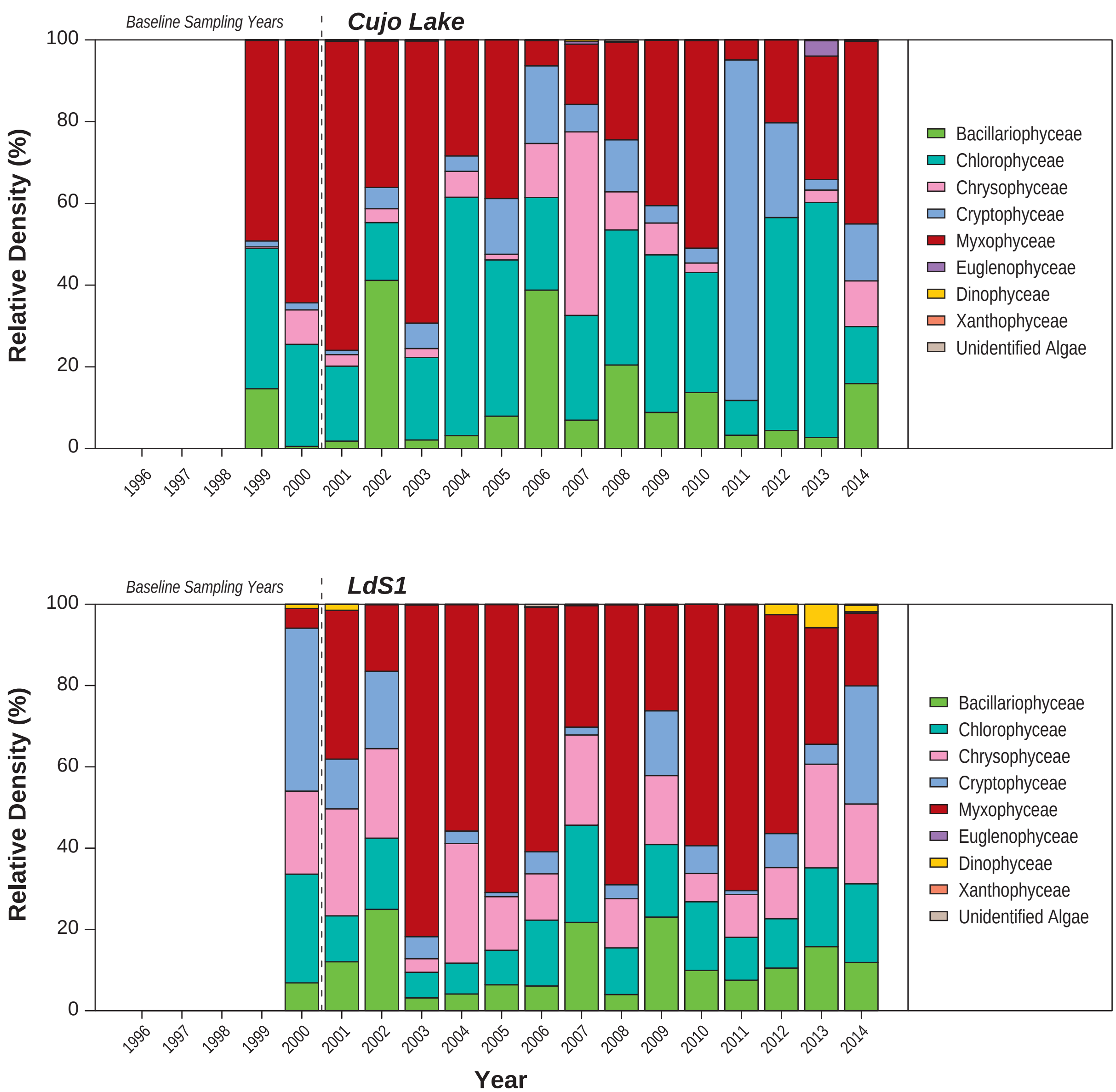


Figure 4.4-4

Relative Densities of Phytoplankton Taxa in Lakes of the King-Cujo Watershed and Lac du Sauvage, 1996 to 2014



Both Shannon and Simpson's diversity indices have varied considerably through time in monitored and reference lakes since monitoring began (Figure 4.4-2). While the variability makes it somewhat difficult to discern temporal trends, mean Shannon diversity has increased in recent years at site LdS1 (Figure 4.4-2). When compared to mean baseline diversity ± 2 SD, mean Shannon and Simpson's diversity were greater at site LdS1 in 2014 (Table 4.4-6). The increase in diversity observed at LdS1 likely reflects an increase in the absolute densities of Bacillariophyceae, Chlorophyceae, Chrysophyceae, and Cryptophyceae, corresponding to a more even distribution in abundance among the phytoplankton groups in 2013 and 2014 (Figures 4.4-3 and 4.4-4). Graphical analysis indicates that 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 as of 2014 (Figures 4.4-3 and 4.4-4). 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 4.4-6. 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	2014 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD
Nanuq	1.75 (4)	1.14 – 2.37	2.07 ± 0.19	0.73 (4)	0.55 – 0.92	0.81 ± 0.04
Counts	1.81 (4)	1.05 - 2.57	2.25 ± 0.20	0.76 (4)	0.57 – 0.95	0.83 ± 0.06
Vulture	1.44 (5)	0.65 – 2.24	2.09 ± 0.10	0.63 (5)	0.28 – 0.98	0.81 ± 0.02
Cujo	2.01 (2)	1.49 – 2.52	2.05 ± 0.49	0.78 (2)	0.61– 0.96	0.73 ± 0.09
LdS1	1.74 (1)	1.40 – 2.07	3.12 ± 0.10	0.75 (1)	0.68 - 0.82	0.93± 0.02

Note: 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).

4.4.2 Zooplankton

4.4.2.1 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) and community composition were monitored to detect potential mine effects.

4.4.2.2 Dataset

Zooplankton data have been collected during late July or August each year from 1995 to 2014 (Table 4.4-7). 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 4.4-7 and depicted graphically in Figures 4.4-5 to 4.4-8, but are not included in the statistical evaluation of effects.

Table 4.4-7. 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
2014	Aug-5	Aug-9	Aug-8	Aug-7	Aug-8

Notes: Dashes indicate no data were available
Biomass data available for 1998-2014 only.
** Indicates data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison.*

4.4.2.3 Results and Discussion

<u>Biomass</u>

Statistical analyses suggest that zooplankton biomass has changed through time at site LdS1 in Lac du Sauvage (Table 4.4-8). Graphical analysis suggests that zooplankton biomass has been highly variable through time in monitored lakes downstream from the KPSF and in one reference lake (i.e., Counts Lake); the wide and overlapping confidence intervals around the fitted means suggest that biomass has been similar through time in these three lakes (Figure 4.4-5). Zooplankton biomass in 2014 was greater than the range of ± 2 SD of mean biomass in baseline years in Cujo Lake and site LdS1 in Lac du Sauvage (Table 4.4-9). Although biomass in 2014 was elevated relative to baseline in both Cujo Lake and site LdS1, graphical analysis indicates that biomass has consistently fluctuated through time in both monitored lakes (Figure 4.4-5). Thus, no mine effects were detected with respect to zooplankton biomass.

Table 4.4-8. 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	-	-	LdS1	2-401

Note: Dashes indicate not applicable.

Table 4.4-9. 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	2014 Mean ± 1 SD
Nanuq	78 (3)	0 - 168	65.5 ± 4.6
Counts	198 (3)	0 - 418	258.1 ± 29.0
Vulture	46 (3)	21 - 71	50.9 ± 5.1
Cujo	110 (2)	70 - 150	228.1 ± 2.5
LdS1	88 (1)	76 – 100	304.4 ± 67.5

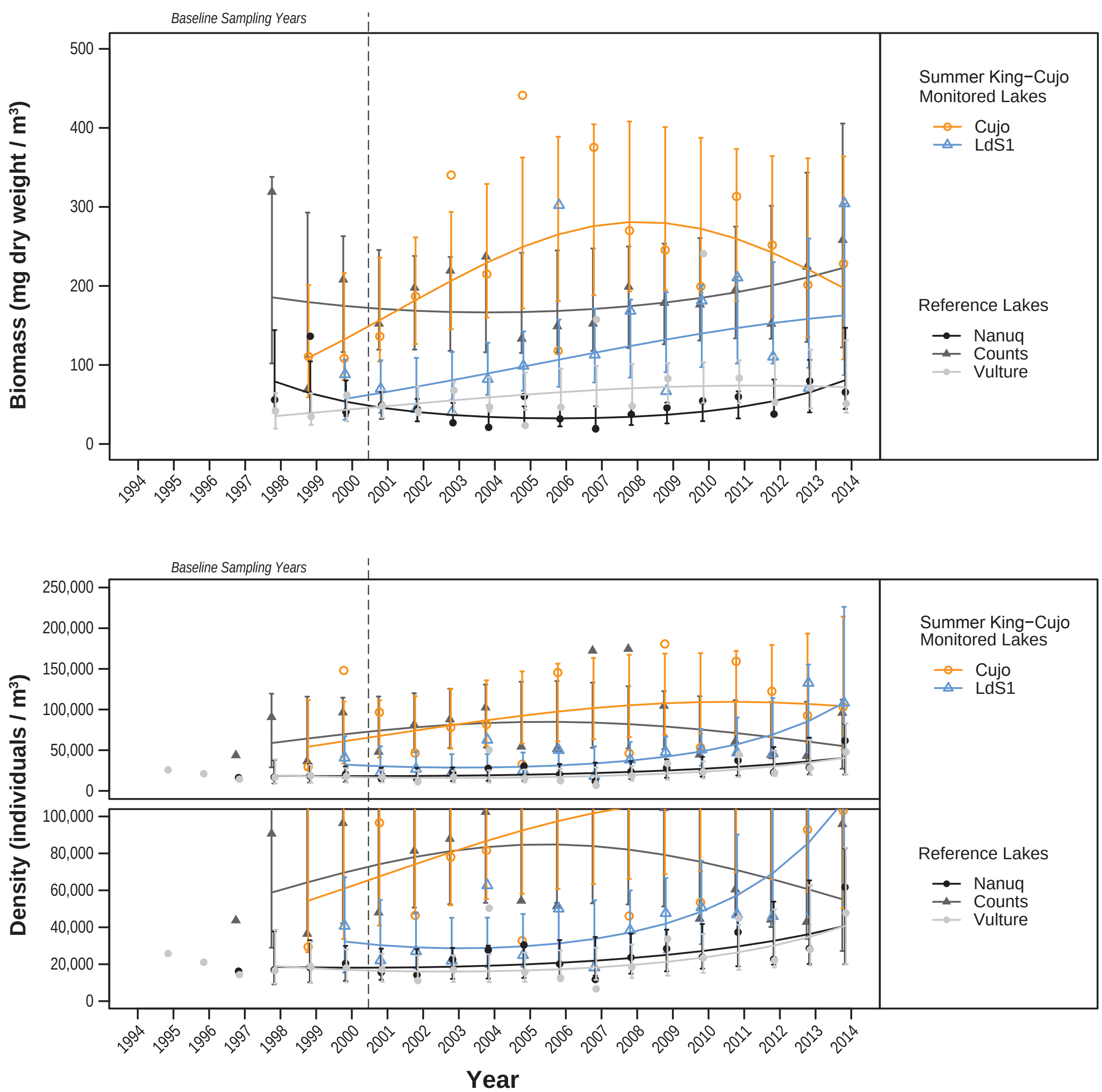
Notes: Units are g/m³.
Negative values were replaced with zeros.
N = number of years data were collected.

Density

Statistical analyses of zooplankton density suggest that densities have been stable through time, relative to reference lakes, in all monitored lakes (Table 4.4-10). Graphical analysis suggests that zooplankton density has increased in recent years at site LdS1; however, zooplankton density may also be increasing in two of the reference lakes (i.e., Nanuq and Vulture lakes), though to a lesser extent (Figure 4.4-5). Mean zooplankton densities in 2014 were greater than mean baseline densities ± 2 SD at site LdS1 in Lac du Sauvage (Table 4.4-11). However, 2014 mean zooplankton densities were also greater than baseline densities ± 2 SD in two of the reference lakes (i.e., Nanuq and Vulture lakes; Table 4.4-11). Thus, no mine effects were detected with respect to zooplankton density in lakes of the King-Cujo Watershed or Lac du Sauvage.

Figure 4.4-5

Observed and Fitted Means for Zooplankton Biomass and Density in King-Cujo Watershed Lakes and Lac du Sauvage, 1995 to 2014



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 4.4-10. 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	-	-	2-406

Note: Dashes indicate not applicable.

Table 4.4-11. 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	2014 Mean ± 1 SD
Nanuq	17,954 (4)	13,719 - 22,190	61,696 ± 8,959
Counts	66,813 (4)	10,216 – 123,410	95,872 ± 2,666
Vulture	19,081 (6)	10,978 - 27,184	47,709 ± 8,213
Cujo	88,687 (2)	0 – 219,779	103,139 ± 14,972
LdS1	40,734 (1)	32,781 – 48,688	108,251 ± 14,009

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

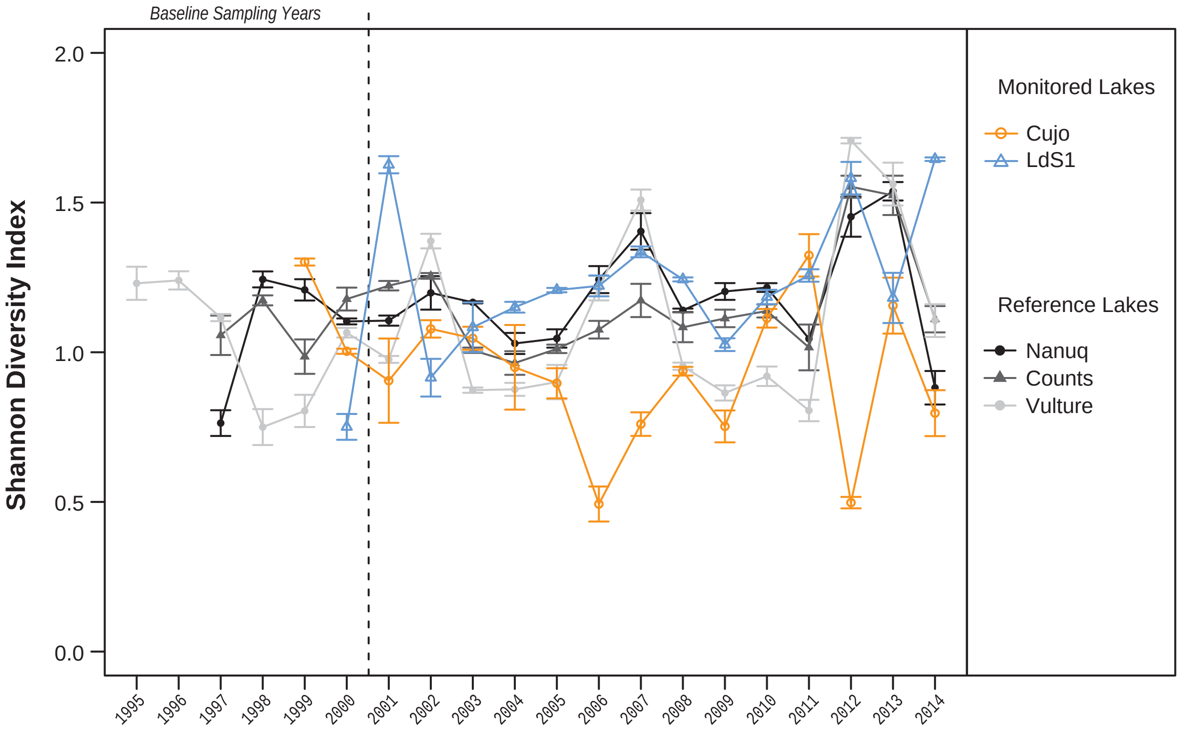
Diversity and Community Composition

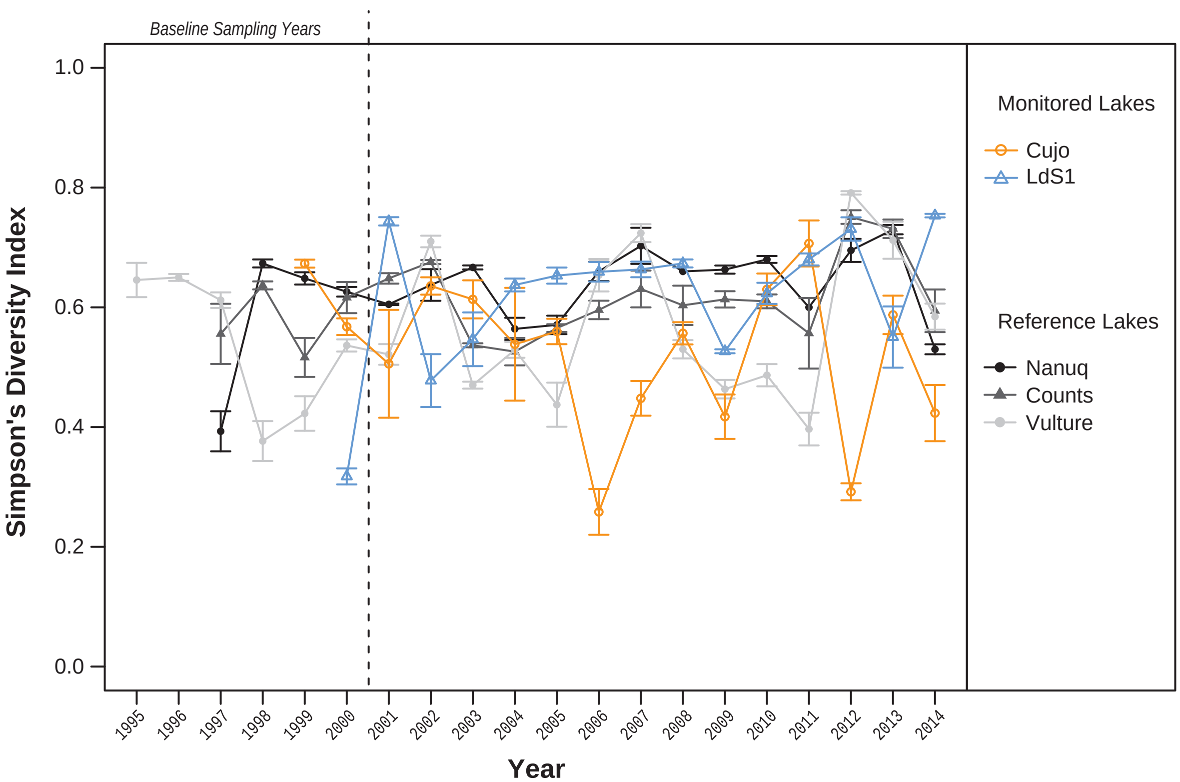
Statistical analyses were not performed on the diversity datasets because the calculation of indices can result in data abnormalities that prevent statistical analysis. Thus, graphical analyses of temporal trends in diversity indices (Figure 4.4-6) and best professional judgment were the primary methods 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 4.4-7 and 4.4-8).

Both Shannon and Simpson's diversity indices have varied considerably through time in both monitored and reference lakes (Figure 4.4-6). 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 4.4-6). Compared to mean baseline diversity ± 2 SD, mean Shannon and Simpson's diversity in 2014 was lower in Cujo Lake and greater at site LdS1 in Lac du Sauvage (Table 4.4-12). The relative densities of different zooplankton taxonomic groups have 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 than in any of the reference lakes (Figures 3.4-11 and 4.4-9 to 4.4-10). Zooplankton community composition in Cujo Lake has remained relatively similar through time, while community composition at site LdS1 in Lac du Sauvage has differed from those observed in previous years (Figures 4.4-7 and 4.4-8). 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 with respect to zooplankton diversity.

Figure 4.4-6

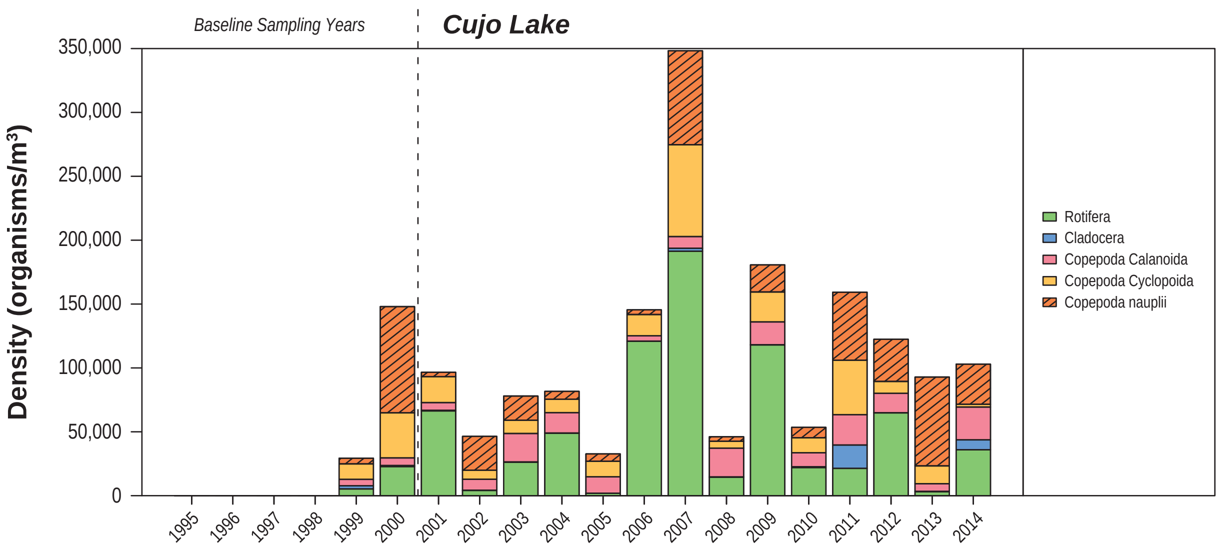
Average Diversity Indices for Zooplankton in King-Cujo Watershed Lakes and Lac du Sauvage, 1995 to 2014

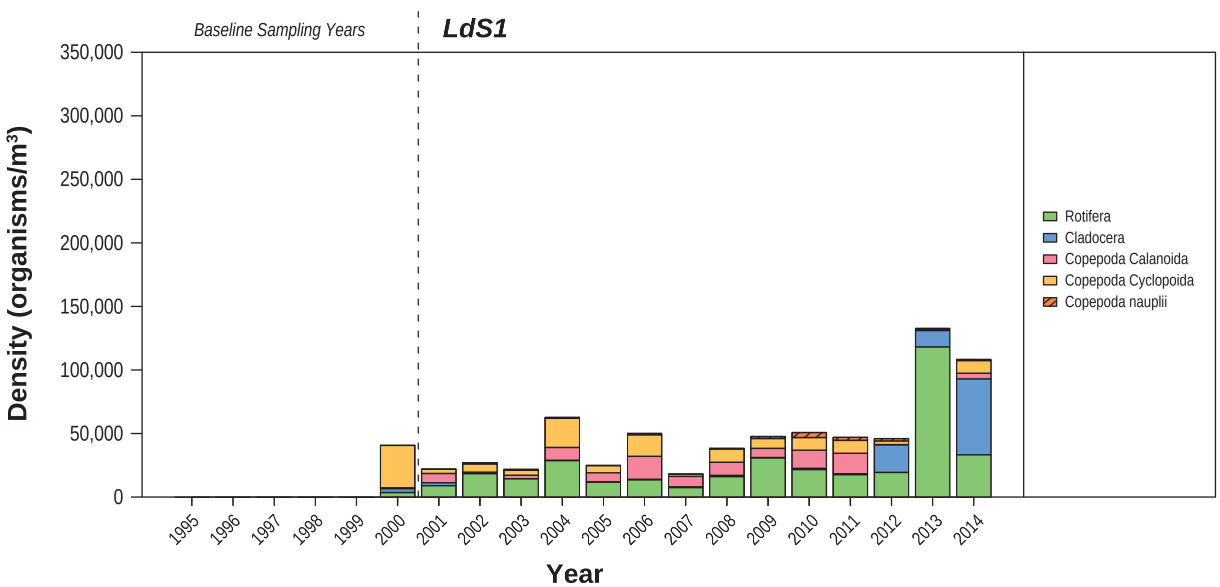




Notes: Symbols represent observed mean values.
Error bars indicate standard error of the observed means.

Average Zooplankton Density by Taxonomic Group for Lakes of the King-Cujo Watershed and Lac du Sauvage, 1995 to 2014





Relative Densities of Zooplankton Taxa in King-Cujo Watershed Lakes and Lac du Sauvage, 1995 to 2014

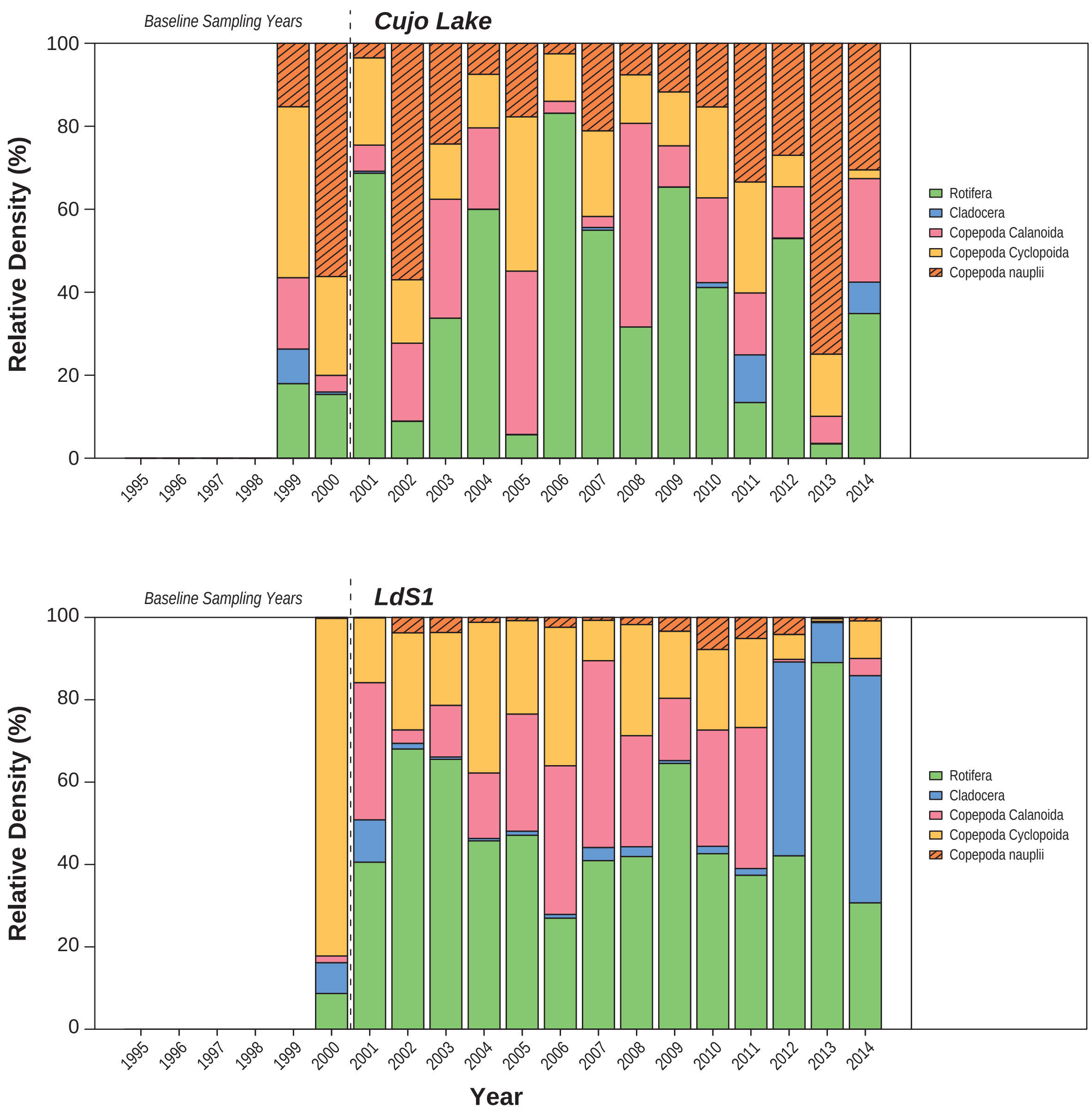
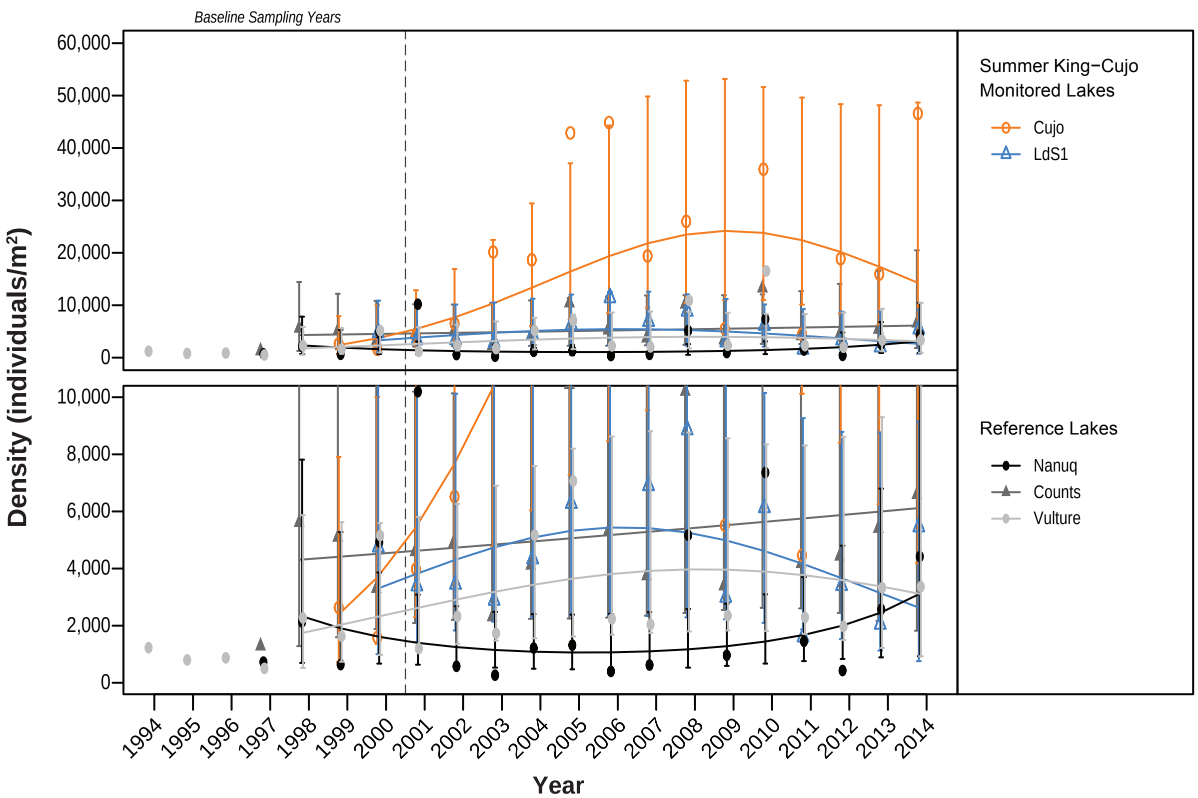


Figure 4.4-9

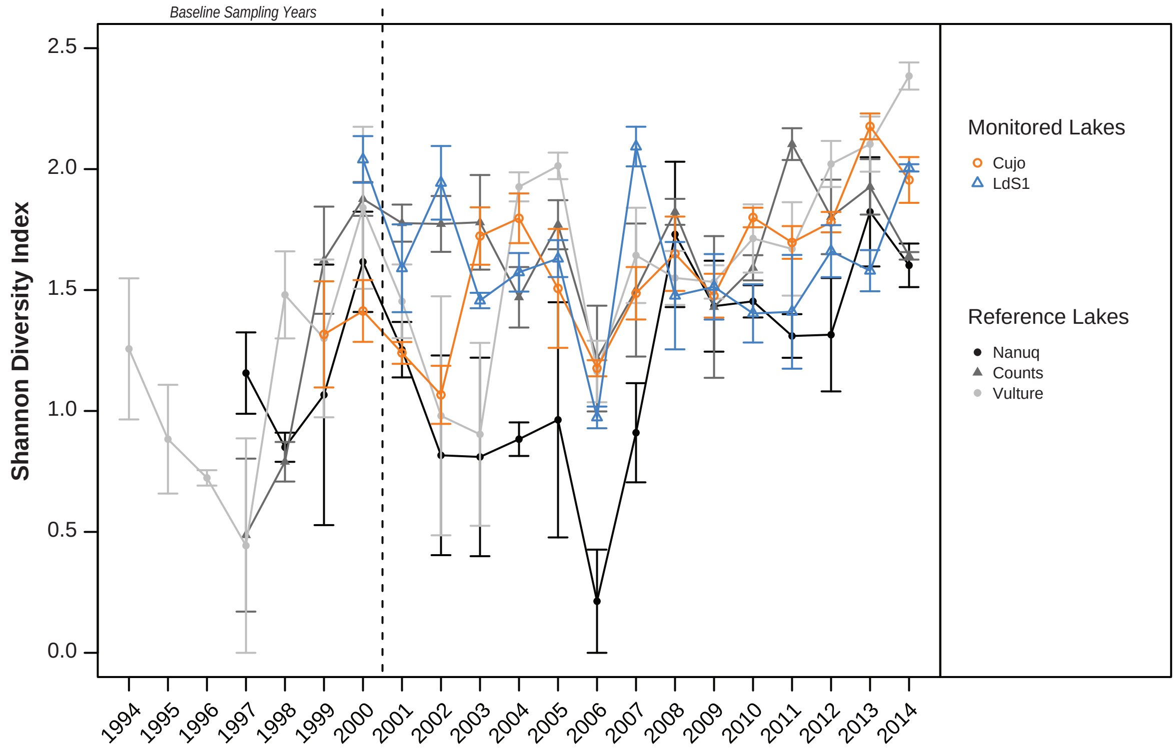
Observed and Fitted Means for Benthos Densities in King-Cujo Watershed Lakes and Lac du Sauvage, 1994 to 2014

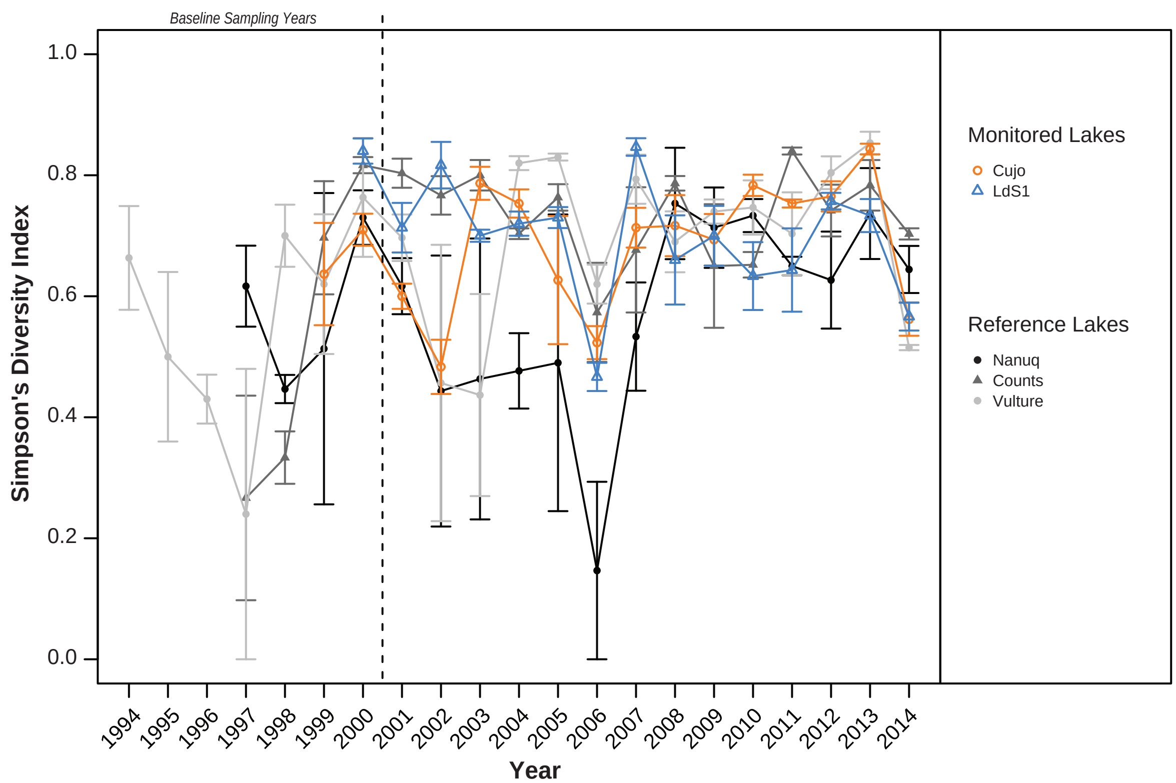


Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.

Figure 4.4-10

Average Diversity Indices for Benthic Dipterans in King-Cujo Watershed Lakes and Lac du Sauvage, 1994 to 2014





Notes: Symbols represent observed mean values.
Error bars indicate standard error of the observed means.

Table 4.4-12. 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	2014 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD
Nanuq	1.08 (4)	0.67 – 1.49	0.88 ± 0.10	0.59 (4)	0.34 – 0.83	0.53 ± 0.01
Counts	1.10 (4)	0.88 – 1.32	1.11 ± 0.08	0.58 (4)	0.44 – 0.72	0.59 ± 0.06
Vulture	1.03 (6)	0.62 – 1.45	1.11 ± 0.09	0.54 (6)	0.31 – 0.77	0.58 ± 0.04
Cujo	1.15 (2)	0.82 - 1.48	0.80 ± 0.13	0.62 (2)	0.50 – 0.74	0.42 ± 0.08
LdS1	0.75 (1)	0.60 – 0.90	1.65 ± 0.01	0.32 (1)	0.27 – 0.36	0.75 ± 0.01

Note: 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 3.4.2.3). Hypotheses regarding potential underlying causes of changes in zooplankton communities and their potential effects on higher trophic levels are included in the Aquatic Biology Summary (see Section 4.4.5).

4.4.3 Lake Benthos

4.4.3.1 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) and dipteran community composition were evaluated for potential mine effects.

4.4.3.2 Dataset

Benthos samples have been collected in triplicate replicates in late July or early August of each year since 1994 (Table 4.4-13). 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 4.4-13 and shown graphically in Figures 4.4-9 to 4.4-12 for visual comparison.

Table 4.4-13. 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	-	-

(continued)

Table 4.4-13. Dataset Used for Evaluation of Effects on Benthos in King-Cujo Watershed Lakes and Lac du Sauvage (completed)

Year	Nanuq	Counts	Vulture	Cujo	LdS1
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
2014	Aug-5	Aug-9	Aug-3	Aug-7	Aug-8

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

4.4.3.3 Results and Discussion

Benthos Density

Statistical analysis indicates that benthos density in Cujo Lake has changed over time, relative to reference lakes (Table 4.4-14). Graphical analysis indicates that benthos density initially increased in Cujo Lake, but has stabilised since about 2003 (Figure 4.4-9). Observed mean benthos density in 2014 was also greater than the range of ± 2 SD observed in baseline years in Cujo Lake (Table 4.4-15) and has generally been greater that baseline densities since around 2003 (Figure 4.4-9). Thus, it was concluded that benthos density in Cujo Lake has increased as a result of mine operations.

Table 4.4-14. 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	
Benthos Density	-	LME	2	-	Cujo	-	2-411

Note: Dashes indicate not applicable.

Table 4.4-15. 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	2014 Mean, ± 1 SD
Nanuq	2,107 (4)	0 – 6,148	4,420 ± 1,198
Counts	3,826 (4)	0 – 8,293	6,594 ± 838
Vulture	1,780 (6)	0 – 5,900	3,363 ± 168
Cujo	2,111 (2)	0 – 4,379	46,594 ± 58,140
LdS1	4,756 (1)	3,021 – 6,491	5,460 ± 1,070

Notes: Units are organisms/m².
Negative values were replaced with zeros.
N = number of years data were collected.

Dipteran Diversity and Community Composition

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 4.4-10) 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 4.4-11 and 4.4-12).

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 4.4-10). While the variability makes it somewhat difficult to discern temporal trends, diversity in Cujo Lake appears to have increased through time, though a similar pattern was observed in at least two reference lakes (i.e., Counts and Vulture lakes; Figure 4.4-10). Shannon diversity in 2014 was greater than the range of ± 2 SD of the mean in baseline years in Cujo Lake; however, a similar pattern was observed in one of the reference lakes (i.e., Vulture Lake; Table 4.4-16). Simpson's diversity in 2014 was below the range of baseline values at site LdS1 in Lac du Sauvage (Table 4.4-16); however, there is currently no clear temporal trend in either Shannon or Simpson's diversity at site LdS1 (Figure 4.4-10). Thus, it was concluded that no mine effects were detected with respect to dipteran diversity.

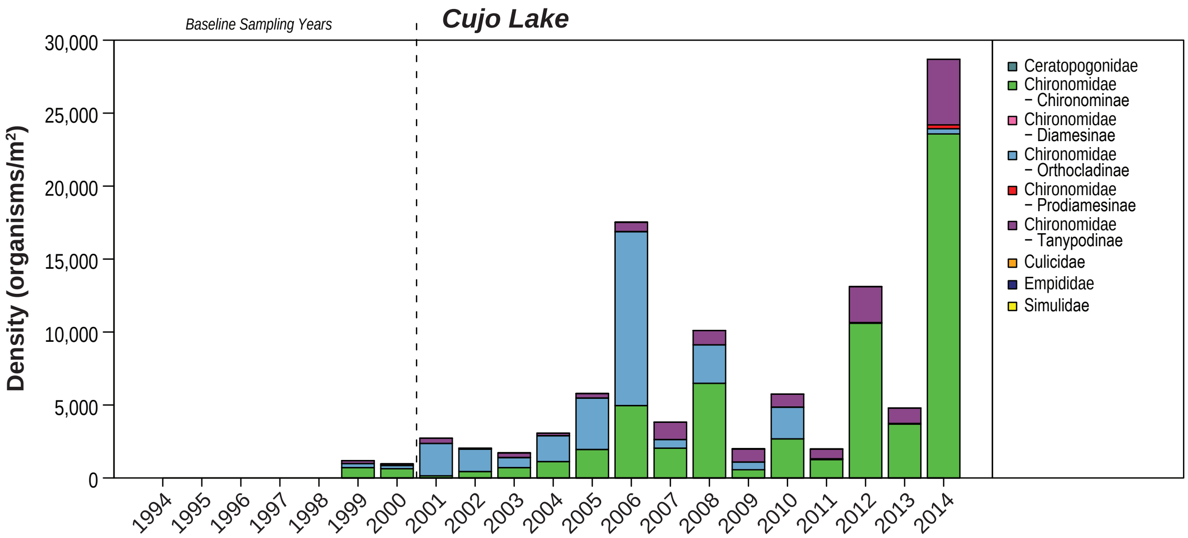
Table 4.4-16. Mean ± 2 Standard Deviations (SD) Baseline Dipteran Diversity in Each of the King-Cujo Watershed Lakes and Lac du Sauvage

	Shannon Diversity			Simpson's Diversity		
Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD
Nanuq	1.17 (4)	0.11 – 2.24	1.60 ± 0.16	0.58 (4)	0.12 – 1.0	0.64 ± 0.07
Counts	1.19 (4)	0 – 2.53	1.64 ± 0.03	0.53 (4)	0 – 1.0	0.70 ± 0.02
Vulture	1.13 (6)	0 – 2.37	2.38 ± 0.10	0.56 (6)	0.06 – 1.0	0.52 ± 0.01
Cujo	1.37 (2)	0.80 – 1.93	1.96 ± 0.16	0.67 (2)	0.46 – 0.88	0.56 ± 0.05
LdS1	2.04 (1)	1.71 – 2.37	2.01 ± 0.03	0.84 (1)	0.77 – 0.91	0.57 ± 0.04

Notes: 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 4.4-11

Average Density of Diptera Taxa for Lakes of the King-Cujo Watershed and Lac du Sauvage, 1994 to 2014



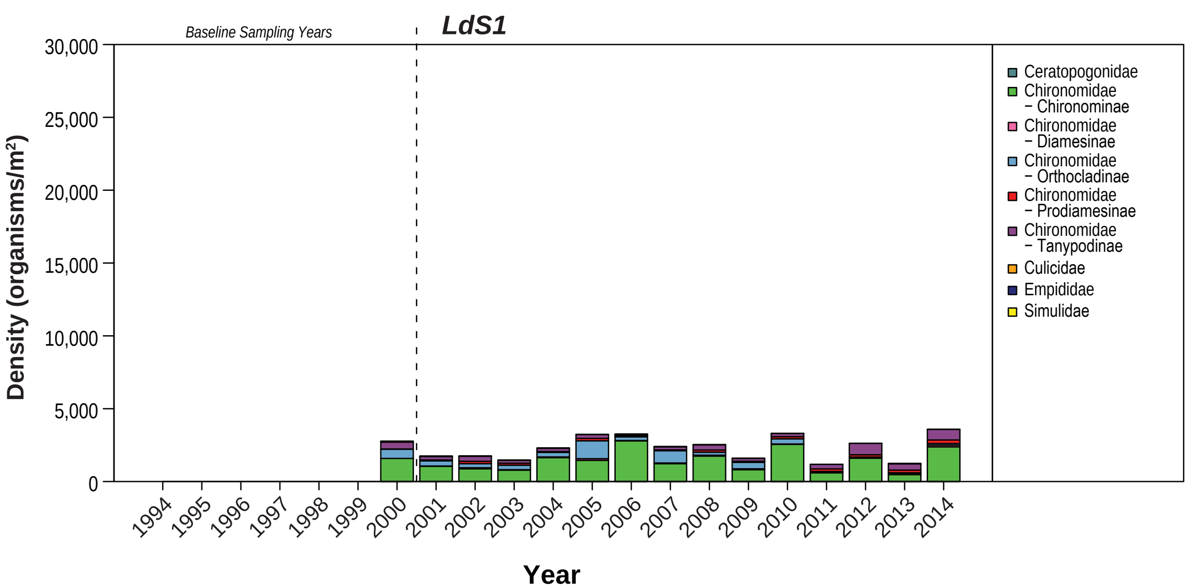
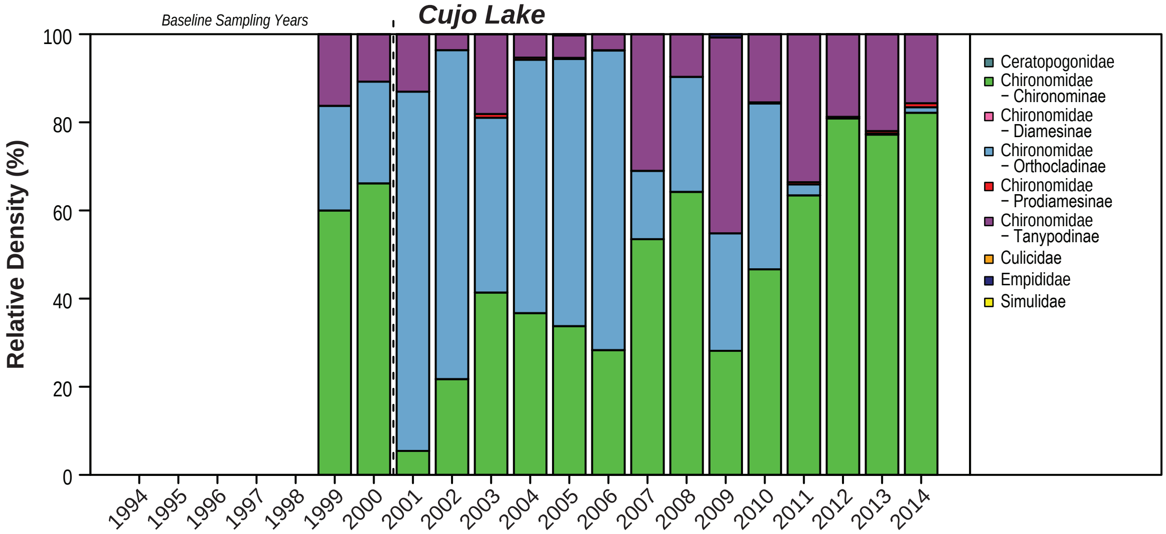
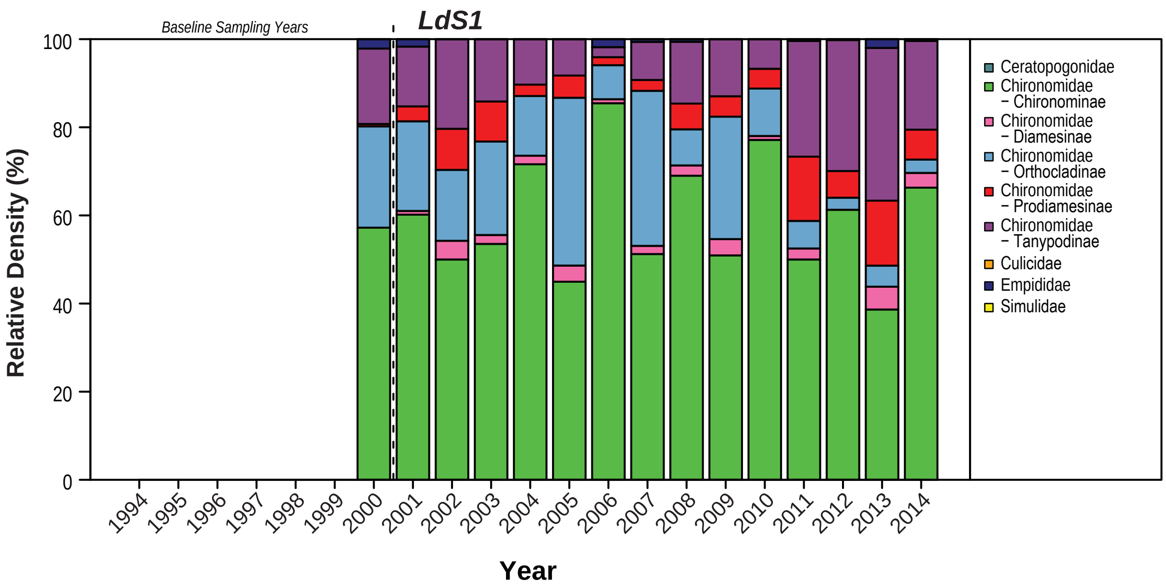


Figure 4.4-12

Average Density of Diptera Taxa for Lakes of the King-Cujo Watershed and Lac du Sauvage, 1994 to 2014





Graphical analyses suggest that densities and relative densities of dipteran taxonomic groups have changed through time in Cujo Lake (Figures 4.4-11 and 4.4-12). Specifically, densities and relative densities of Orthocladiinae have decreased, while densities of Chironominae have increased (Figures 4.4-11 and 4.4-12) and densities of Tanypodinae and Prodiamesinae have increased (Figure 4.4-11). These patterns are consistent with those that were first identified through the multivariate analyses conducted as part of the 2012 AEMP Re-evaluation (Rescan 2012c). In addition, graphical analyses suggest that densities of Orthocladiinae at site LdS1 in Lac du Sauvage have decreased through time, with a smaller coincidental increase in densities of Tanypodinae and Prodiamesinae (Figures 4.4-11 and 4.4-12). Although these patterns were generally not observed in reference lakes, graphical analyses in 2014 reveal a more recent trend of decreasing Orthocladiinae densities with coincidental increases in Chironominae densities in Counts Lake (Figures 3.4-19 and 3.4-22).

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. 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. Similar to results from the 2013 AEMP (ERM Rescan 2014a), some of the trends that were described in the 2012 AEMP (Rescan 2013) were less apparent in recent years. 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 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*, *Microtendipes*, and *Stictochironomus*. In Counts Lake, the increase in Chironominae seems more likely to be related to recent increases in *Corynocera*, *Paratanytarsus*, 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 2012c). In 2014, sediment quality analyses found an increase in total nitrogen in sediments of Cujo Lake (see Section 4.3.3.3).

4.4.4 Stream Benthos

4.4.4.1 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) and community composition were evaluated for potential mine effects.

4.4.4.2 Dataset

Stream benthos samples have been collected over a one month period from early August to early September of each year since 1995 (Table 4.4-17). Five replicates were collected from each stream in 1995 and between 1999 and 2014. In 1997 and 1998, triplicate samples were collected from each stream. Baseline data, which were collected between 1994 and 1997, were not used in the statistical evaluation of effects but are included in Table 4.4-17 and are depicted graphically in Figures 4.4-13 to 4.4-19 for visual comparison.

Table 4.4-17. 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†	-	-	-	-

(continued)

Table 4.4-17. Dataset Used for Evaluation of Effects on the Benthos in King-Cujo Watershed Streams (completed)

Year	Nanuq Outflow	Counts Outflow	Vulture-Polar	Cujo Outflow
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
2014	Aug 1 – Sept 2	Aug 4 – Sept 4	Aug 1 – Se[t 2	Aug 4 – Sept 4

Notes: 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.
** Indicates 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.
§ Data were collected, but processed through the incorrect mesh size in the laboratory. Only EPT taxonomy data from 2011 and 2012 were used in the evaluation of effects.

Although stream benthos samples were collected in 2010, they were not included in the evaluation of effects as a result of laboratory error. In 2011, 2012, and 2013, benthos samples were processed through sieves in the laboratory with incorrect mesh sizes. Historically, AEMP stream benthos samples have been processed through a 180 µm mesh sieve, but a 250 µm mesh was used in 2011 and 2012, and a 500 µm mesh sieve was used in 2013. A laboratory study was conducted in 2014 to determine the number of individuals, and the taxonomic identity of individuals, that passed through 250 µm and 500 µm mesh sieves. It was determined that dipterans from various taxonomic groups were small enough to pass through the 250 µm mesh, while EPT taxa were never found to pass through the 250 um mesh, but some EPT individuals were able to pass through the 500 µm mesh. As a result, total benthos density, and dipteran diversity, density and relative density from 2011 to 2013 were excluded from the evaluation of effects; EPT diversity, density and relative density were only excluded in 2013. However, all data collected from 2011 to 2013 is included in Table 3.4-16 and shown graphically for visual comparison.

4.4.4.3 Results and Discussion

<u>Benthos Density</u>

Statistical and graphical analyses suggest that the density of stream benthos has been stable through time in all monitored and reference streams (Table 4.4-18; Figure 4.4-13). Mean benthos density in 2014 was within the range of ± 2 SD of the baseline mean (Table 4.4-19). Thus, no mine effects were detected with respect to stream benthos density.

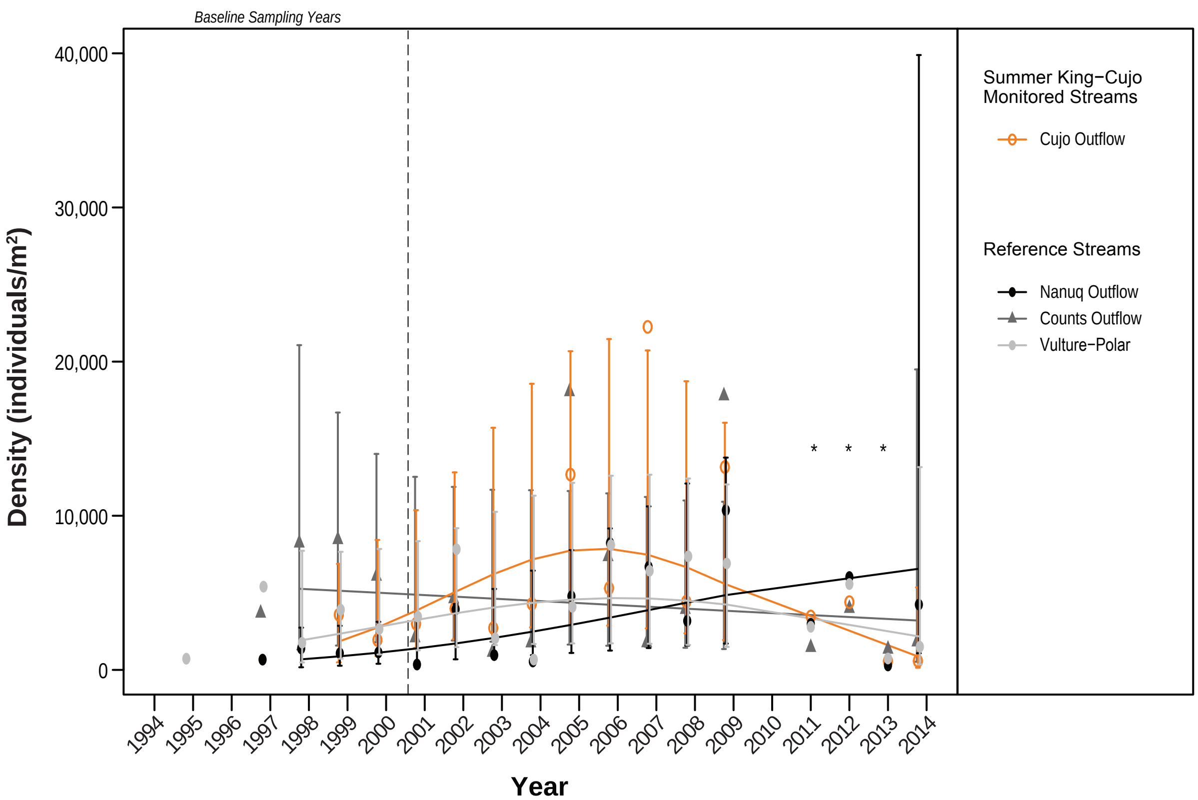
Table 4.4-18. Statistical Results of Benthos Density in Streams in the King-Cujo Watershed

	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	
Benthos Density	-	LME	1b	-	-	None	2-417

Note: Dashes indicate not applicable.

Figure 4.4-13

Observed and Fitted Means for Benthos Densities in King-Cujo Watershed Streams, 1995 to 2014



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 values in 2011, 2012, and 2013 were not included in the statistical analysis. Observed means are plotted here for reference only.*

Table 4.4-19. 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	2014 Mean, ± 1 SD
Nanuq Outflow	1,076 (4)	336 - 1,816	4,238 ± 5,753
Counts Outflow	6,778 (4)	0 – 13,660	1,820 ± 597
Vulture-Polar	2,758 (5)	0 – 6,980	1,502 ± 989
Cujo Outflow	2,758 (2)	360 – 5,155	580 ± 95

Notes: Units are organisms/m².
Negative values were replaced with zeros.
N = number of years data were collected.

Dipteran Diversity and Community Composition

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 4.4-14) 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 4.4-15 to 4.4-16).

Both Shannon and Simpson's stream dipteran diversity indices have varied considerably through time in both monitored and reference streams since monitoring began (Figure 4.4-14). While the variability makes it somewhat difficult to discern temporal trends, diversity appears to have generally decreased from 2002 to 2009 in Cujo Outflow, although a similar pattern was observed in two of the reference streams (i.e., Counts Outflow and Vulture-Polar; Figure 4.4-14). In all cases, diversity in 2014 has increased and is within the range of mean values ± 2 SD observed during baseline years (Figure 4.4-14; Table 4.4-20).

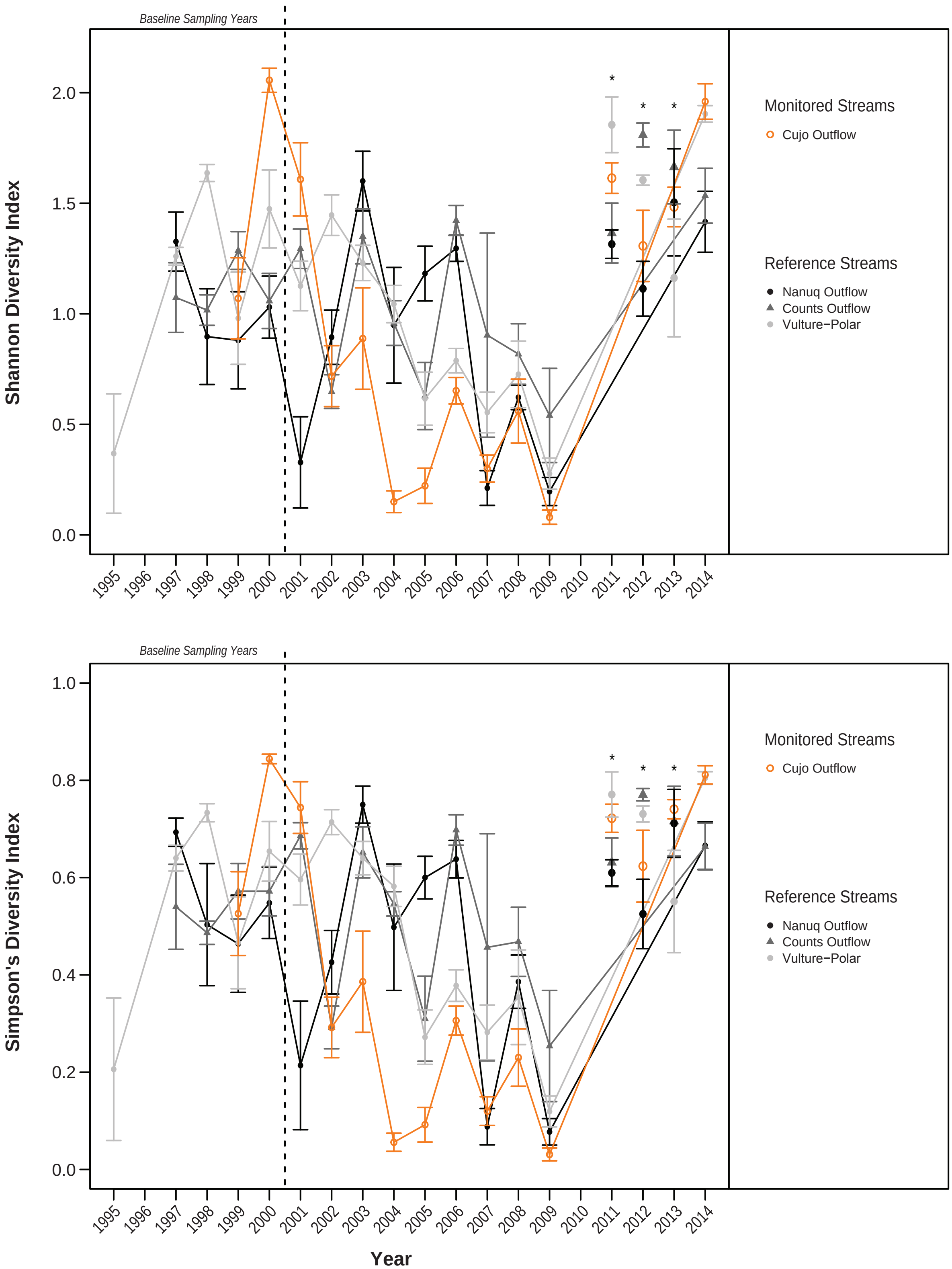
Table 4.4-20. Mean ± 2 Standard Deviations (SD) Baseline Dipteran Diversity in Each of the King-Cujo Watershed Streams

	Shannon Diversity			Simpson's Diversity		
Stream	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD
Nanuq Outflow	1.01 (4)	0.25 - 1.78	1.42 ± 0.31	0.54 (4)	0.17 – 0.91	0.67 ± 0.11
Counts Outflow	1.12 (4)	0.65 – 1.59	1.53 ± 0.28	0.55 (4)	0.33 – 0.77	0.66 ± 0.11
Vulture-Polar	1.09 (5)	0 – 2.30	1.90 ± 0.08	0.51 (5)	0 - 1	0.80 ± 0.03
Cujo Outflow	1.56 (2)	0.38 – 2.75	1.96 ± 0.14	0.69 (2)	0.26 – 1	0.81 ± 0.03

Notes: 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 4.4-14

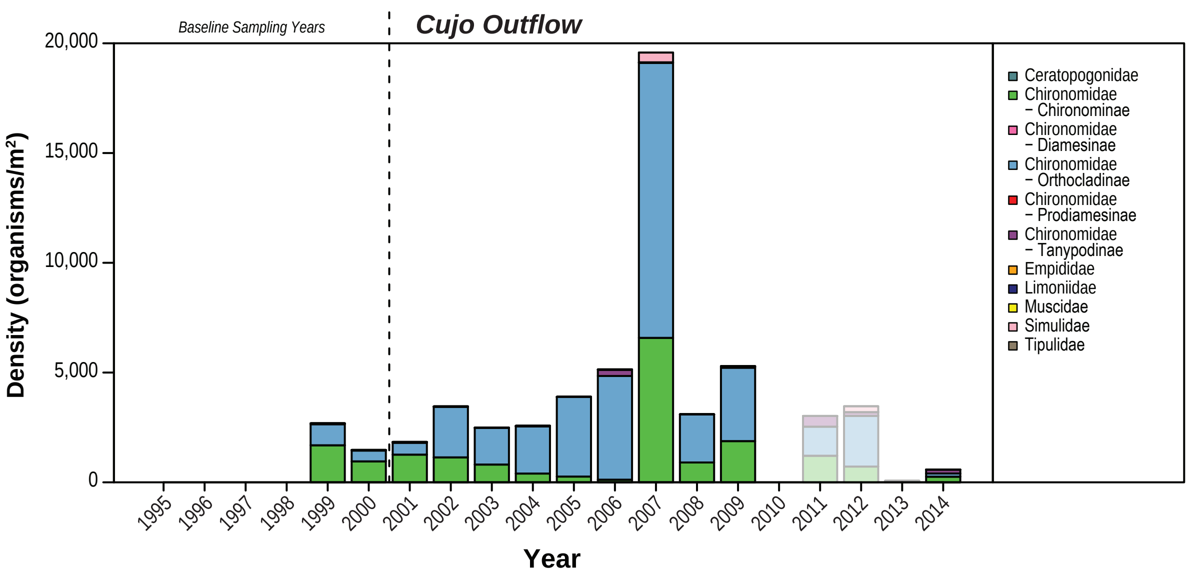
Average Diversity Indices for Benthic Dipterans in King-Cujo Watershed Streams, 1995 to 2014



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 values in 2011, 2012, and 2013 were not included in the statistical analysis. Observed means are plotted here for reference only.*

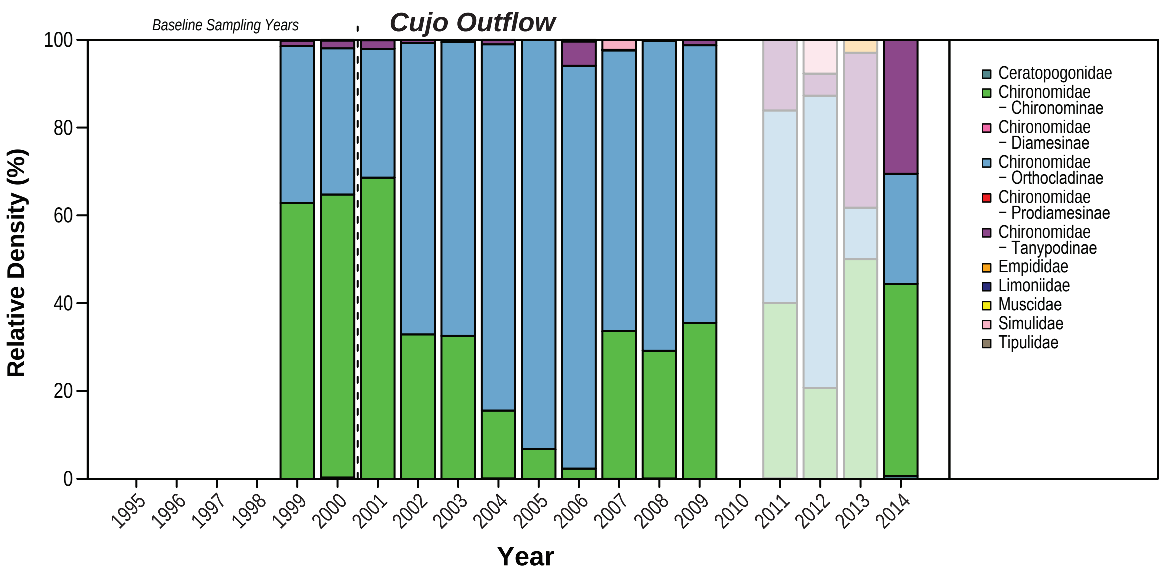
Figure 4.4-15

Average Benthic Dipteran Density by Taxonomic Group in Streams of the King-Cujo Watershed, 1995 to 2014



Note: Density values in 2011, 2012, and 2013 were not considered in the evaluation of effects. Values are plotted here for reference only.

Relative Densities of Benthic Dipteran Taxa in Streams of the King-Cujo Watershed, 1995 to 2014



Note: Density values in 2011, 2012, and 2013 were not considered in the evaluation of effects. Values are plotted here for reference only.

The relative densities of dipteran taxonomic groups have been consistent through time in all monitored and reference streams (Figures 4.4-15 to 4.4-16). 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 3.4-27, 3.4-30, 4.4-15, and 4.4-16). Observations in 2014 indicate that this trend may be reversing in Cujo Outflow (Figures 4.4-15 and 4.4-16). There is also some evidence that densities of organisms from the sub-family Tanypodinae may be increasing in Cujo Outflow (Figures 4.4-15 and 4.4-16). 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.

EPT Diversity and Community Composition

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 4.4-17) 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 4.4-18 and 4.4-19).

Both Shannon and Simpson's EPT diversity indices have varied through time in both monitored and reference streams since monitoring began (Figure 4.4-17). While the variability makes it somewhat difficult to discern temporal trends, both Shannon and Simpson's EPT diversity have generally remained within the range of baseline and historical values observed in Cujo Outflow and all reference streams through time (Figure 4.4-17). Mean EPT diversity in 2014 was within the range of mean baseline ± 2 SD in Cujo Outflow (Table 4.4-21). Densities and relative densities of EPT taxa have been variable through time in all monitored and reference streams, but show no signs of directed change in the monitored stream (Figures 3.4-34, 3.4-37, 4.4-18, and 4.4-19). Thus no mine effects were detected with respect to EPT diversity or taxonomic composition.

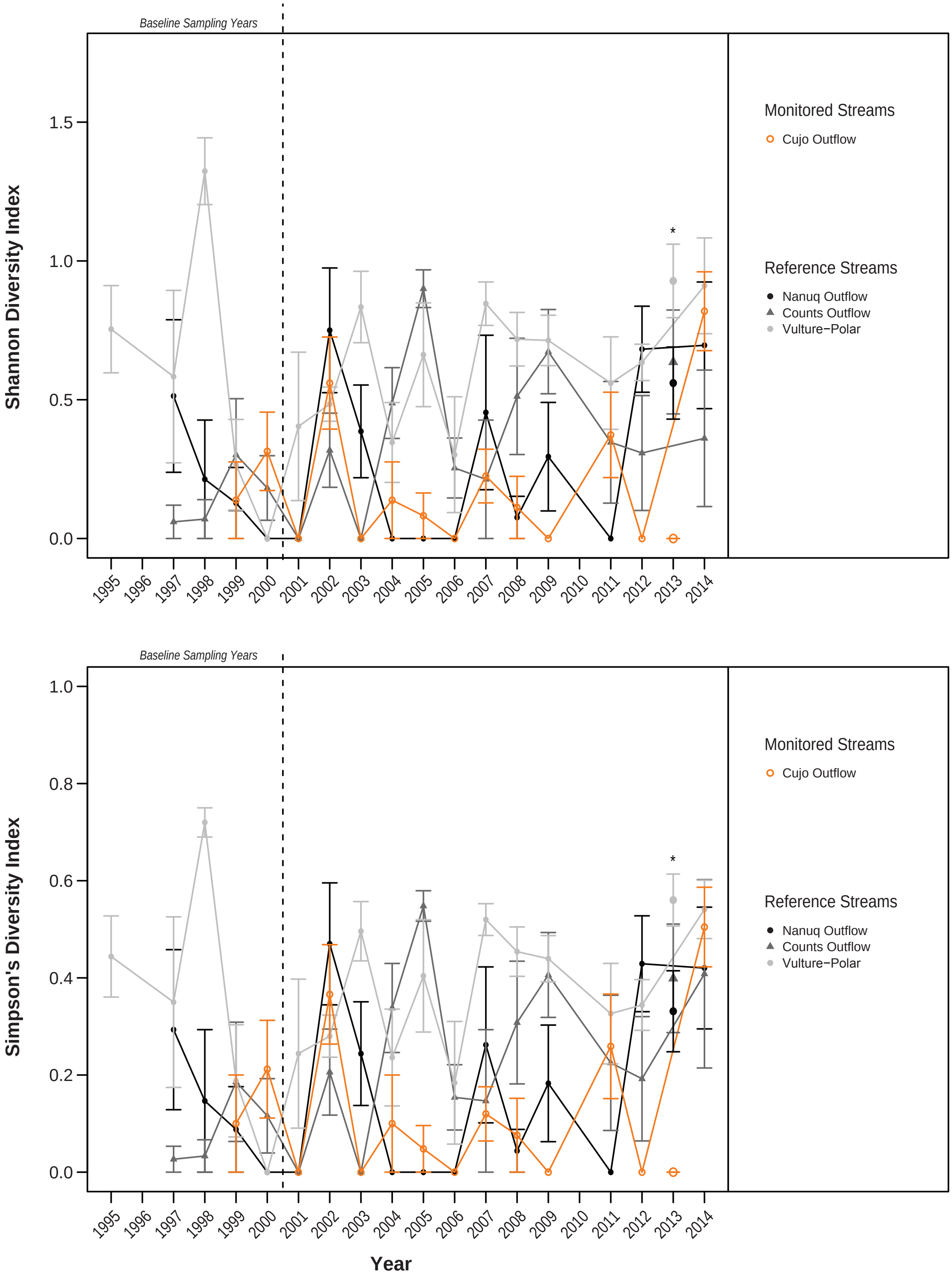
Table 4.4-21. Mean ± 2 Standard Deviations (SD) Baseline EPT Diversity in Each of the King-Cujo Watershed Streams

	Shannon Diversity			Simpson's Diversity		
Stream	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD
Nanuq Outflow	0.18 (4)	0 – 0.82	0.70 ± 0.51	0.11 (4)	0 – 0.51	0.42 ± 0.28
Counts Outflow	0.18 (4)	0 – 0.76	0.36 ± 0.55	0.11 (4)	0 – 0.47	0.41 ± 0.43
Vulture-Polar	0.52 (5)	0 – 1.57	0.91 ± 0.39	0.30 (5)	0 – 0.89	0.54 ± 0.13
Cujo Outflow	0.23 (2)	0 – 0.84	0.82 ± 0.25	0.16 (2)	0 – 0.59	0.50 ± 0.14

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

Figure 4.4-17

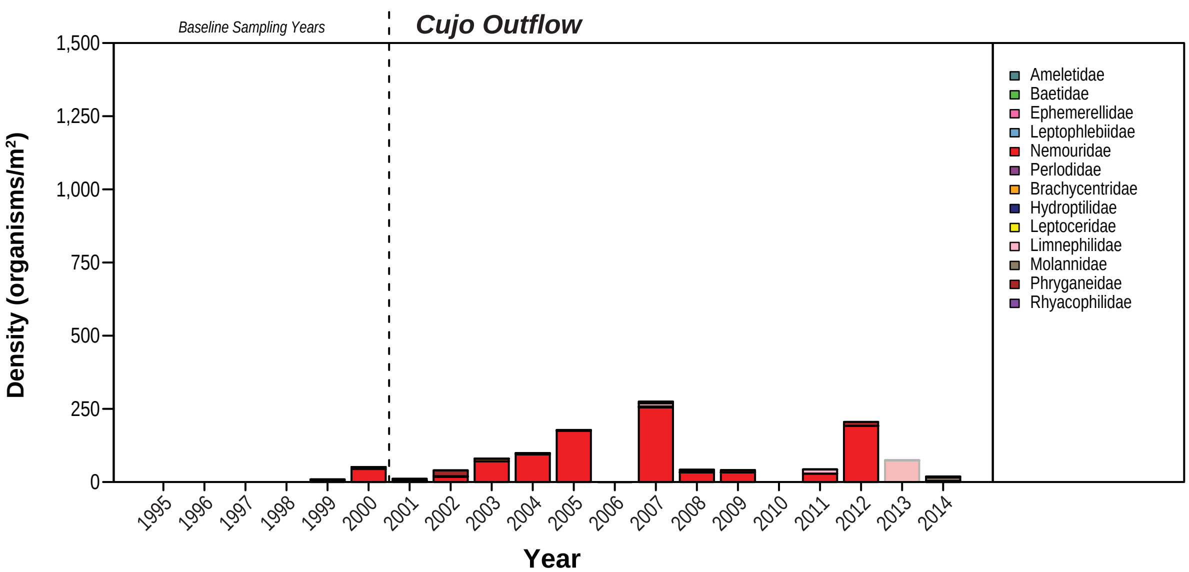
Average Diversity Indices for Benthic EPT Taxa in King-Cujo Watershed Streams, 1995 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars indicate upper and lower 95% confidence intervals of the fitted means.
** Diversity indices in 2013 were not included in the evaluation of effects. Means and standard errors are plotted here for reference only.*

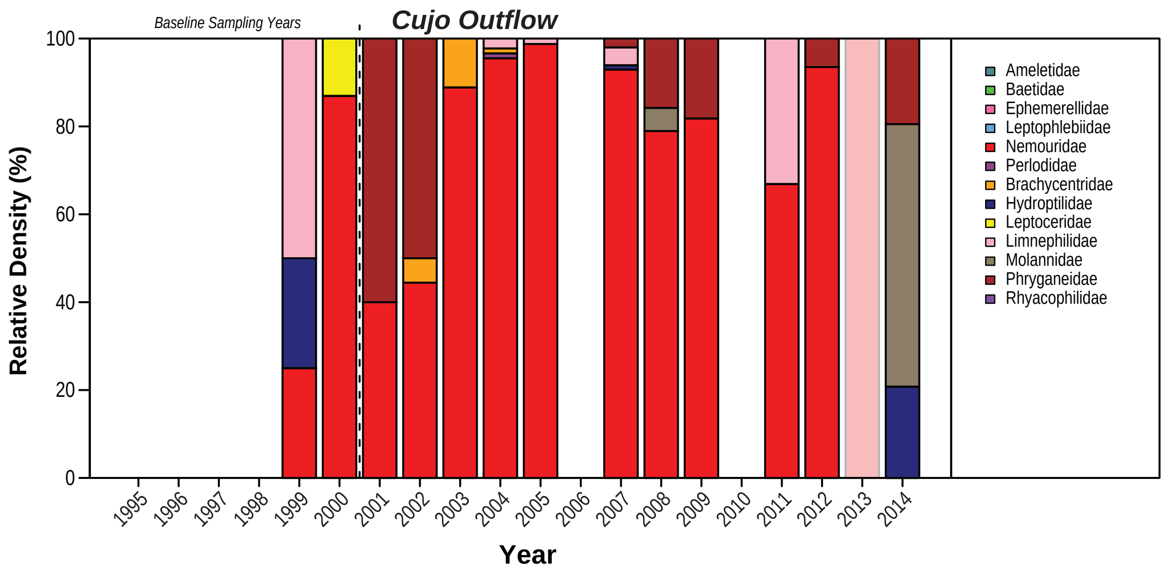
Figure 4.4-18

Average Benthic EPT Density by Taxonomic Group in Streams of the King-Cujo Watershed, 1995 to 2014



Note: Density values in 2013 were not considered in the evaluation of effects. Values are plotted here for reference only.

Relative Densities of Benthic EPT Taxa in Streams of the King-Cujo Watershed, 1995 to 2014



Note: Density values in 2013 were not considered in the evaluation of effects. Values are plotted here for reference only.

4.4.5 Aquatic Biology Summary

Two changes in biological variables were observed in 2014:

- an increase in lake benthos density in Cujo Lake; and
- a 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, 2013, and 2014. The reason for the change in composition of cladoceran genera remains unclear.

Lake benthos density has increased through time in Cujo Lake, but appears to have remained stable, though elevated, since around 2003. 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*, *Microtendipes,* 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 2012c). 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 associated with changes in macronutrient availability, rather than toxic effects 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 strong mine effects on monitored fish populations in the King-Cujo Watershed (Rescan 2012c). 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.

4.5 SUMMARY

Table 4.5-1 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 2014 (Table 4.5-1). 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 1999d). In Cujo Lake, DO measurements were less than CCME guidelines throughout the bottom half of the water column in mid-February and mid-March but had begun to increase by April coincident, with the longer photo-period. 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 and 3.1-1b). 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. However, similar to past years, a portion of the surface ice on Cujo Lake was cleared of snow in late winter (April 21, 2014) to allow for increased light penetration and thus increased DO production through photosynthesis.

A total of 23 water quality variables were evaluated for lakes and streams in the King-Cujo Watershed and Lac du Sauvage in the 2014 AEMP. Of these, concentrations of 13 variables have changed through time in monitored sites downstream of the KPSF (Table 4.5-1). 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 4.5-1). 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.

Table 4.5-1. **Summary of Evaluation of Effects for the King-Cujo Watershed and Lac du Sauvage**

Variable	Change Downstream of KPSF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Physical Limnology						
Under-ice Temperature Profiles	No	-	-	-	No	-
Under-ice DO Profiles	No	-	-	-	No	-
August Secchi Depths	No	-	-	-	No	-
Lake and Stream Water Quality						
pH	Yes	Downstream to Christine-Lac du Sauvage Stream	Increase	KPSF	Yes	All 2014 pH values within the range of CCME guideline values. Concentrations as far as Christine-Lac du Sauvage Stream have stabilised in recent years.
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	Concentrations have stabilised during the open water season.
Chloride	Yes	Downstream to Cujo Outflow	Increase	KPSF	Yes	All 2014 concentrations less than the SSWQO.
Sulphate	Yes	Downstream to Christine-Lac du Sauvage Stream	Increase	KPSF	Yes	All 2014 concentrations less than the SSWQO.
Potassium	Yes	Downstream to Christine-Lac du Sauvage Stream	Increase	KPSF	Yes	All 2014 concentrations less than the SSWQO.

(continued)

Table 4.5-1. Summary of Evaluation of Effects for the King-Cujo Watershed and Lac du Sauvage (continued)

Variable	Change Downstream of KPSF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Lake and Stream Water Quality (*cont'd*)						
Total Ammonia-N	Yes, Historical	Downstream to Cujo Lake	Increase	KPSF	Historical	Concentrations in Cujo Lake have returned to baseline and reference concentrations in recent years. All 2014 concentrations less than the CCME guideline.
Nitrite-N	No	-	-	-	No	All 2014 concentrations less than the CCME guideline.
Nitrate-N	No	-	-	-	No	All 2014 concentrations less than the SSWQO.
Total Phosphate-P	No	-	-	-	No	Observed and fitted means, and/or 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	Possible	Baseline concentrations not sampled, but downstream spatial gradient present.
Total Antimony	No	-	-	-	No	All 2014 concentrations less than the water quality benchmark.
Total Arsenic	No	-	-	-	No	All 2014 concentrations less than the CCME guideline.
Total Barium	Yes	Downstream to Cujo Outflow	Increase	KPSF	Yes	All 2014 concentrations less than the water quality benchmark.
Total Boron	Yes	Downstream to Cujo Outflow	Increase	KPSF	Yes	All 2014 concentrations less than the CCME guideline.
Total Cadmium	No	-	-	-	No	All 2014 concentrations less than the CCME guideline.
Total Copper	Yes, Historical	Downstream to Cujo Outflow	Increase	KPSF	Historical	All 2014 concentrations less than the CCME guideline. Concentrations in Cujo Lake and Cujo Outflow have declined in recent years.

(continued)

Table 4.5-1. Summary of Evaluation of Effects for the King-Cujo Watershed and Lac du Sauvage (continued)

Variable	Change Downstream of KPSF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Lake and Stream Water Quality (*cont'd*)						
Total Molybdenum	Yes	Downstream to Christine-Lac du Sauvage Stream	Increase	KPSF	Yes	All 2014 concentrations less than the SSWQO.
Total Nickel	No	-	-	-	No	All 2014 concentrations less than the CCME guideline.
Total Selenium	No	-	-	-	No	All 2014 concentrations less than the CCME guideline.
Total Strontium	Yes	Downstream to Christine-Lac du Sauvage Stream	Increase	KPSF	Yes	All 2014 concentrations less than the strontium water quality benchmark-
Total Uranium	No	-	-	-	No	All 2014 concentrations less than the CCME guideline.
Total Vanadium	No	-	-	-	No	All 2014 concentrations less than the SSWQO.
Sediment Quality						
TOC	No	-	-	-	No	-
Available Phosphorus	No	-	-	-	No	-
Total Nitrogen	Yes	Cujo Lake	Increase	Unknown	Possible	Cause of the observed increase is unclear at this time and may represent natural variability.
Total Antimony	No	-	-	-	No	-
Total Arsenic	No	-	-	-	No	The observed mean exceeded the CCME ISQG and PEL in all monitored lakes in 2014. Similar pattern observed in reference lakes.
Total Cadmium	No	-	-	-	No	All 2014 concentrations less than the CCME ISQG and PEL.

(continued)

Table 4.5-1. Summary of Evaluation of Effects for the King-Cujo Watershed and Lac du Sauvage (continued)

Variable	Change Downstream of KPSF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Sediment Quality (*cont'd*)						
Total Copper	No	-	-	-	No	The observed mean in Cujo Lake and the 95% confidence interval around the fitted mean for site LdS1 exceeded the CCME ISQG in 2014; however, similar patterns were observed in reference lakes. Concentrations were less than the CCME PEL at all monitored sites in 2014.
Total Molybdenum	Yes	Cujo Lake	Increase	KPSF	Yes	-
Total Nickel	No	-	-	-	No	-
Total Phosphorus	No	-	-	-	No	-
Total Selenium	No	-	-	-	No	-
Total Strontium	Possible	Cujo Lake	Possible increase	KPSF	Possible	No statistical analyses possible at this time. Observed means in Cujo Lake were greater than observed means at reference sites.
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	-.
Diversity	No	-	-	-	No	-
Relative Densities of Major Taxa	No	-	-	-	No	The rotifer Conochilus sp. and the cladoceran Holopedium gibberum have been largely absent from Cujo Lake since 2002.

(continued)

Table 4.5-1. Summary of Evaluation of Effects for the King-Cujo Watershed and Lac du Sauvage (completed)

Variable	Change Downstream of KPSF?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Lake Benthos						
Density	Yes	Cujo	Increase	KPSF	Yes	Benthos density in Cujo Lake has stabilised since about 2003.
Dipteran Diversity	No	-	-	-	No	-
Dipteran Relative Density	Yes	Cujo Lake and site LdS1 in Lac du Sauvage	Decrease in Orthocladiinae; 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, Microtendipes, Stictochironomus, Procladius, Ablabesmyia). A similar pattern of decreased Orthocladiinae with increasing Chironominae (Corynocera, Paratanytarsus, Stictochironomus) observed in recent years in one reference lake (i.e., 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	-

Notes: Dashes indicate not applicable.
Comparisons to CCME guidelines and other relevant benchmarks are for 2014 data only.
DO = dissolved oxygen
CCME = Canadian Council of Ministers of the Environment
SSWQO = Site-specific Water Quality Objective

TOC is a measure of the amount of live and decomposing organic matter in the water column. Elevated 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, elevated 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.

No changes through time in phytoplankton or zooplankton have been observed in the King-Cujo Watershed or Lac du Sauvage. However, an increase in lake benthos density was observed in Cujo Lake (see Section 4.4). Elevated TOC may also be associated with reductions in DO because bacteria consume oxygen as they decompose organic matter. Under-ice DO 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 DO concentrations. However, DO 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.

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 4.5-1 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 2014c). 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 in Section 2.3). 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(CCME 2004). Other water quality benchmark values include provincial guidelines or ones taken from the published literature (see Table 2.3-1 in Section 2.3; antimony, barium, and strontium). With the exception of total phosphate-P, 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. However, for total phosphate-P, concentrations in reference lakes also exceeded CCME guidelines, suggesting that exceedances are not related to mine activities.

Twelve sediment quality variables were evaluated in the 2014 AEMP for the King-Cujo Watershed and Lac du Sauvage. Of these, the concentrations of two variables (i.e., total nitrogen, total molybdenum) have changed through time and one other variable (i.e., total strontium) showed signs of a potential increase or mine effect (Table 4.5-1). Total nitrogen and total molybdenum concentrations have increased in sediments of Cujo Lake, however the cause of the increase in total nitrogen is unclear at this time and may represent natural variability. Total strontium concentrations in Cujo Lake were higher than those observed in reference lakes. CCME guidelines for the protection of aquatic life exist for three of the evaluated sediment quality variables, including arsenic, cadmium, and copper (see Table 2.4-1 in Section 2.4; CCME 2014b). The observed mean exceeded the CCME ISQG and PEL for arsenic in all monitored sites; however, similar exceedances were observed in all three reference lakes (Table 4.5-1). For cadmium, the 95% confidence intervals around the fitted mean and the observed mean concentrations were below the CCME ISQG and PEL at all monitored sites (Table 4.5-1). For copper, the observed mean in Cujo Lake and the 95% confidence interval around the fitted mean for site LdS1 exceeded the CCME ISQG in 2014; however, similar patterns were observed in reference lakes; copper concentrations were less than the CCME PEL at all monitored sites in 2014 (Table 4.5-1). Exceedances for arsenic and copper were similar in both monitored and reference lakes, suggesting that exceedances are not related to mine activities

Despite increases in 13 evaluated water quality variables and two sediment quality variables downstream of the KPSF, observed concentrations were generally below benchmark values. This suggests that concentrations of water and sediment quality variables remain less than the concentrations at which toxic effects might be expected. Thus, observed changes in biological community composition in the King-Cujo Watershed 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 2012c).

An increase in lake benthos density was observed in Cujo Lake, and 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 increased lake benthos density observed in Cujo Lake is unclear at this time, but may reflect an increase in productivity as evidenced by elevated TOC concentrations (see Section 4.2.4.11) or an increase in available nutrients as evidenced by increasing nitrogen concentrations in sediments (see Section 4.3.3.3). The underlying cause of the shift in community composition was attributed to changes in the relative availability of macronutrients (see Sections 4.4.3.3 and 4.4.5). Although changes in phytoplankton, lake benthos density, and the 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 2012c).

5. EVALUATION OF EFFECTS: PIGEON-FAY AND UPPER EXETER WATERSHED

5.1 PHYSICAL LIMNOLOGY

5.1.1 Variables

One physical limnology variable was evaluated for potential effects caused by mine activities in the Pigeon-Fay and Upper Exeter Watershed: open water season Secchi depths (see Section 3.1.1) Under-ice temperature and DO concentrations were not evaluated because the winter of 2014 predates the opening of the PSD and its connection with the natural Pigeon Stream.

5.1.2 Dataset

Secchi depths were measured during August sampling surveys (Table 5.1-1).

Table 5.1-1. Dataset Used for Evaluation of Effects on Secchi Depths in Pigeon-Fay and Upper Exeter Watershed Lakes

Year	Nanuq	Counts	Vulture	Fay Bay	Upper Exeter
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	-	-
2000	Aug-4	Aug-1	Aug-4	-	-
2001	Aug-1	Jul-30	Aug-2	Aug-5	-
2002	Aug-1	Aug-7	Aug-3	Aug-8	Aug-8
2003	Aug-9	Aug-7	Aug-4	-	-
2004	Aug-10	Aug-13	Aug-9	-	-
2005	Aug-1	Aug-7	Jul-31	-	Aug-19
2006	Aug-2	Aug-4	Aug-2	-	Aug-6
2007	Aug-11	Aug-6	Aug-12	-	-
2008	Aug-8	Jul-31	Jul-29	Aug-3	-
2009	Jul-30	Aug-1	Jul-30	Aug-4	-
2010	Aug-5	Aug-7	Aug-5	-	-
2011	Aug-2	Aug-5	Aug-5	-	-
2012	Aug-1	Aug-8	Aug-12	-	-
2013	Aug-3	Aug-1	Aug-1	-	-
2014	Aug-5	Aug-9	Aug-3	Aug-1	Aug-10

Note: Dashes indicate no data were available.

5.1.3 Results and Discussion

5.1.3.1 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 whether a significant change in Secchi depth occurred in monitored lakes of the Pigeon-Fay and Upper Exeter Lake Watershed (Figure 5.1-1). A value of ± 0.5 m was used as an estimate of error due to sampler bias for interpreting graphical results.

Secchi depths in 2014 were within the range of Secchi depths observed during the baseline sampling period in all monitored lakes (Figure 5.1-1). Thus, no mine effects were detected with respect to Secchi depths in the Pigeon-Fay and Upper Exeter Lake Watershed.

5.2 LAKE AND STREAM WATER QUALITY

5.2.1 Variables

Twenty-three water quality variables were evaluated for potential mine effects in the Pigeon-Fay and Upper Exeter Watershed (see Section 3.2.1). CCME guidelines for the protection of aquatic life exist for 11 of the evaluated water quality variables, including pH, TSS, total ammonia-N, nitrite-N, total phosphate-P, total arsenic, total boron, total cadmium, total nickel, total selenium, and total uranium (see Table 2.3-1 in Section 2.3; CCME 2014c). 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 include provincial guidelines or ones taken from the published literature (see Table 2.3-1 in Section 2.3).

5.2.2 Dataset

5.2.2.1 Lakes

Lake water quality data were collected during the ice-covered season from mid-April to mid-May and/or during the open water season from early July to mid-September of each year from 1994 to 2014 (Tables 5.2-1 and 5.2-2). Baseline water quality data for the two monitored lakes were collected from 2001 to 2006, while baseline water quality data for the three reference lakes were collected from 1994 to 2007 (Tables 5.2-1 and 5.2-2). Water quality data collected from 2008 to 2013 were not used in the statistical analyses, but are included in Tables 5.2-1 and 5.2-2 and shown in Figures 5.2-1 to 5.2-23 for visual comparison. Data collected from 2008 to 2010 in Fay Bay and Upper Exeter Lake were part of a monitoring program initiated in response to an unplanned release of fine processed kimberlite (FPK) in May of 2008 (Rescan 2011b). Thus, data from the "before" period in the BACI analysis encompasses data from 1994 to 2007 and the "after" period includes all data collected in 2014.

Figure 5.1-1

August Secchi Depths for Pigeon-Fay and Upper Exeter Watershed Lakes, 1994 to 2014

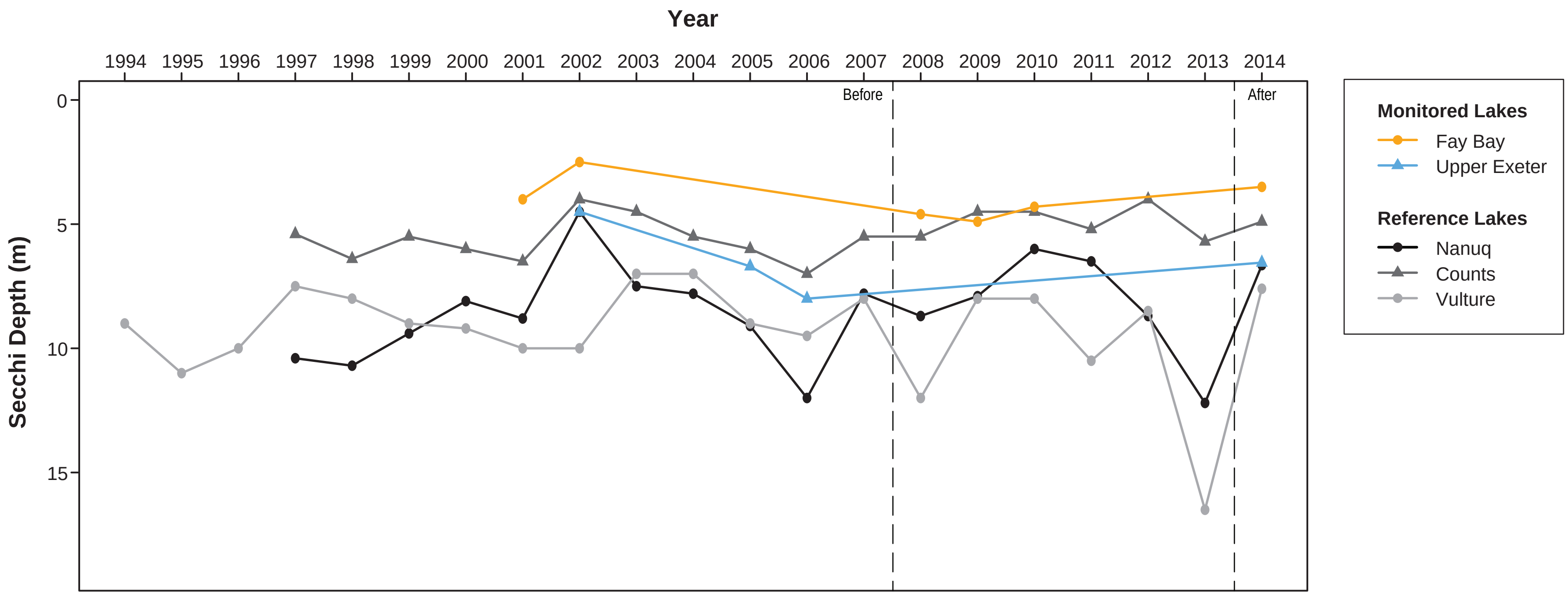


Table 5.2-1. Dataset Used for Evaluation of Effects on the Winter (Ice-covered) Water Quality in Pigeon-Fay and Upper Exeter Watershed Lakes

Year	Nanuq	Counts	Vulture	Fay Bay	Upper Exeter
1994	-	-	-	-	-
1995	-	-	-	-	-
1996	-	-	Apr-18 (1)	-	-
1997	-	-	-	-	-
1998	-	-	-	-	-
1999	-	-	-	-	-
2000	-	-	-	-	-
2001	-	-	-	-	-
2002	Apr-19 (4)	Apr-23 (4)	Apr-20 (4)	Apr-20 (3)	-
2003	Apr-12 (4)	Apr-13 (4)	Apr-14 (4)	-	-
2004	Apr-18 (4)	Apr-17 (4)	Apr-18 (4)	-	-
2005	Apr-24 (4)	Apr-24 (4)	Apr-24 (4)	Apr-22 (4)	Apr-22 (2)
2006	Apr-20 (4)	Apr-22 (4)	Apr-21 (4)	-	Apr-23 (2)
2007	Apr-21 (4)	Apr-24 (4)	Apr-22 (4)	-	-
2008*	Apr-27 (4)	May-3 (4)	May-3 (4)	-	-
2009*	May-11 (4), May-18 (4)	May-17 (4)	Apr-28 (4)	Jan-29 (4), Feb-6 (4), Apr-1 (4)	Jan-25 (4), Feb-6 (4), Apr-1 (4)
2010*	Apr-14 (4)	Apr-14 (4)	Apr-12 (4)	Feb-3 (4), Apr-19 (4)	Feb-3 (4), Apr-19 (4)
2011*	Apr 25 (4)	Apr 26 (4)	Apr 28 (4)	-	-
2012*	Apr-20 (4)	Apr-17 (4)	Apr-18 (4)	-	-
2013*	Apr-26 (4)	Apr-26 (4)	Apr-23 (4)	-	-
2014	Apr-7 (4)	Mar-31 (4)	Apr-8 (4)	Apr-2 (4)	Apr-1 (4)

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

Table 5.2-2. Dataset Used for Evaluation of Effects on the Summer (Open Water) Water Quality in Pigeon-Fay and Upper Exeter Watershed Lakes

Year	Nanuq	Counts	Vulture	Fay Bay	Upper Exeter
1994	-	-	Jul-1 (5), Aug-13 (5)	-	-
1995	-	-	Aug-9 (5)	-	-
1996	-	-	Jul-26 (3)	-	-
1997	Aug-4 (9)	Aug-14 (3)	Aug-5 (9)	-	-

(continued)

Table 5.2-2. Dataset Used for Evaluation of Effects on the Summer (Open Water) Water Quality in Pigeon-Fay and Upper Exeter Watershed Lakes (continued)

Year	Nanuq	Counts	Vulture	Fay Bay	Upper Exeter
1998	Jun-30 (6), Jul-15 (6), Jul-29 (6), Aug-11 (6), Sep-9 (6)	Jun-30 (6), Jul-18 (6), Jul-29 (6), Aug-14 (6), Sep-10 (6)	Jun-29 (6), Jul-14 (6), Jul-27 (6), Aug-10 (6), Sep-2 (6)	-	-
1999	Jul-9 (6), Aug-7 (6), Sep-4 (6)	Jul-9 (6), Aug-8 (6), Sep-4 (6)	Jul-8 (6), Aug-6 (6), Sep-3 (6)	-	-
2000	Jul-3 (4), Aug-4 (4), Sep-5 (4)	Jun-30 (4), Aug-1 (4), Sep-4 (4)	Jul-3 (4), Aug-4 (4), Sep-5 (4)	-	-
2001	Jul-7 (4), Aug-1 (4), Sep-1 (4)	Jul-5 (4), Jul-30 (4), Sep-2 (4)	Jul-7 (4), Aug-2 (4), Sep-1 (4)	Jul-8 (4), Aug-5 (4), Sep-7 (4)	
2002	Jul-10 (4), Aug-1 (4), Sep-1 (4)	Jul-10 (4), Aug-7 (4), Sep-6 (4)	Jul-4 (4), Aug-3 (4), Sep-9 (4)	Jul-13 (4), Aug-8 (4), Sep-4 (4)	Jul-13 (4), Aug-8 (4), Sep-4 (4)
2003	Jul-6 (2), Aug-9 (3), Sep-7 (2)	Jul-2 (2), Aug-7 (2), Sep-10 (2)	Jul-5 (3), Aug-4 (2), Sep-7 (2)	-	-
2004	Jul-13 (2), Aug-10 (3), Sep-12 (2)	Jul-9 (2), Aug-12 (2), Sep-11 (2)	Jul-11 (3), Aug-9 (2), Sep-7 (2)	-	-
2005	Jul-14 (2), Aug-1 (2), Sep-1 (3)	Jul-14 (2), Aug-7 (3), Sep-1 (2)	Jul-10 (2), Jul-31 (2), Sep-1 (2)	-	Aug-19 (3), Aug-31 (2)
2006	Jun-29 (2), Aug-2 (3), Sep-5 (2)	Jun-30 (2), Aug-4 (2), Sep-7 (2)	Jun-30 (3), Aug-2 (2), Sep-5 (2)	-	Aug-6 (2)
2007	Jul-15 (2), Aug-11 (6), Sep-11 (2)	Jul-13 (2), Aug-6 (6), Sep-12 (3)	Jul-10 (2), Aug-12 (6), Sep-10 (2)	-	
2008*	Jul-7 (2), Aug-8 (6), Sep-9 (2)	Jul-8 (2), Jul-31 (6), Sep-5 (2)	Jul-13 (2), Jul-29 (6), Sep-9 (2)	Jul-12 (4), Aug-3 (4), Sep-11 (3)	Jul-14 (4), Aug-5 (4), Sep-11 (2)
2009*	Jul-13 (3), Jul-30 (6), Sep-8 (2)	Jul-6 (3), Aug-1 (6), Sep-7 (3)	Jul-15 (3), Jul-30 (6), Sep-6 (2)	Jul-13 (3), Aug-4 (2), Sep-8 (2)	-
2010*	Aug-5 (6)	Aug-7 (6)	Aug-5 (6)	Jul-5 (4), Aug-13 (4), Sep-3 (4)	-
2011*	Aug-2 (6)	Aug-5 (6)	Aug-5 (6)	-	-

(continued)

Table 5.2-2. Dataset Used for Evaluation of Effects on the Summer (Open Water) Water Quality in Pigeon-Fay and Upper Exeter Watershed Lakes (completed)

Year	Nanuq	Counts	Vulture	Fay Bay	Upper Exeter
2012*	Aug-1 (6)	Aug-8 (6)	Aug-7 (6)	-	-
2013*	Aug-3 (6)	Aug-1 (6)	Aug-1 (6)	-	-
2014	Aug-5 (4)	Aug-14 (4)	Aug-3 (4)	Jul-3 (4), Aug-1 (4), Aug-31 (4)	Jul-3 (4), Aug-10 (4), Aug-31 (4)

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

The timing and number of sampling events during the open water season have 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 AEMP Re-evaluation (Rescan 2010c). As part of the Pigeon AEMP, monitored lakes in the Pigeon-Fay and Upper Exeter Watershed were sampled three times throughout the open water season (July, August, and September) in order to capture potential short-term changes in water quality that may have occurred following the opening of the Pigeon Stream Diversion prior to the freshet of 2014. Thus, all samples collected throughout the open water season in previous years from reference lakes were included in the analysis (Table 5.2-1).

For each variable, average values were calculated for ice-covered and open water seasons of each year by pooling data from all of the depths that were sampled during that season. 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 (Table 5.2-3). Water quality data from the ice-covered season were included in the evaluation of effects, even though the winter sampling period of 2014 predates the opening of the PSD and its connection with the natural Pigeon Stream. Results from the 2010 Fay Bay Monitoring Program showed that although concentrations of water quality variables were returning to background levels, some variables were still elevated (Rescan 2011b). Therefore, 2014 winter water quality data were included in order to help clarify the cause of any current potential effects.

Table 5.2-3. Data Removed from the Historical Lake and Stream Water Quality Dataset for the Pigeon-Fay and Upper Exeter Watershed

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

5.2.2.2 *Streams*

The timing and number of stream sampling events has varied through time as refinements have been made to the sampling protocol. Sampling in the three AEMP reference streams occurred in August in 1994, in July, August, and September in 1995, in July in 1996, in July and/or September in 1997, and in July, August, and September in 1998. Starting in 1999, water quality data from the AEMP reference streams has been collected in June, August, and September of each year, with July stream water quality sampling added to the AEMP program in 2010 (Table 5.2-4).

Table 5.2-4. Dataset Used for Evaluation of Effects on the Summer Water Quality in Pigeon-Fay and Upper Exeter Watershed Streams

Year	Nanuq Outflow	Counts Outflow	Vulture-Polar	Pigeon Reach 7	Pigeon Reach 1
1994	-	-	Aug-4 (1)	-	-
1995	-	-	Jul-6 (1), Aug-10 (1), Sep-14 (1)	-	-
1996	-	-	Jul-27 (1)	-	-
1997	Sep-7 (1)	Sep-7 (1)	Jul-2 (1), Sep-6 (1)	-	-
1998	Jul-22 (3), Aug-18 (3), Sep-9 (3)	Jul-22 (3), Aug-18 (3), Sep-10 (3)	Jul-20 (3), Aug-16 (3), Sep-15 (3)	-	-
1999	Aug-6 (3), Sep-14 (3)	Aug-7 (3), Sep-14 (3)	Aug-8 (3), Sep-16 (3)	-	-
2000	Jul-30 (3), Sep-8 (3)	Jul-30 (3), Sep-7 (3)	Jul-30 (3), Sep-8 (3)	-	-
2001	Aug-7 (3), Sep-8 (3)	Aug-7 (3), Sep-8 (3)	Aug-7 (3), Sep-8 (3)	-	Aug-4 (3), Sep-8 (3)
2002	Aug-6 (3), Sep-11 (3)	Aug-6 (3), Sep-11 (3)	Aug-6 (3), Sep-11 (3)	-	Aug-6 (3), Sep-11 (3)

(continued)

Table 5.2-4. Dataset Used for Evaluation of Effects on the Summer Water Quality in Pigeon-Fay and Upper Exeter Watershed Streams (completed)

Year	Nanuq Outflow	Counts Outflow	Vulture-Polar	Pigeon Reach 7	Pigeon Reach 1
2003	Aug-2 (2), Sep-4 (2)	Aug-2 (2), Sep-4 (2)	Aug-2 (2), Sep-4 (2)	-	Sep-26 (2)
2004	Aug-11 (2), Sep-9 (2)	Aug-11 (2), Sep-9 (2)	Aug-11 (2), Sep-9 (2)	Jul-11 (3), Jul-17 (1)	Jul-14 (1), Jul-17 (1), Jul-28 (2), Aug-13 (2), Aug-25 (2), Sep-11 (1), Aug-25 (2)
2005	Aug-2 (2), Sep-5 (2)	Aug-2 (2), Sep-5 (2)	Aug-2 (2), Sep-5 (2)	Aug-2 (2), Aug-29 (1), Sep-5 (2)	Jul-7 (1), Jul-26 (1), Aug-2 (3), Aug-18 (1), Aug-29 (2), Sep-5 (2), Sep-14 (2), Sep-27 (1)
2006	Jul-27 (2) Aug-31 (1), Sep-11 (1)	Jul-27 (2) Aug-31 (1), Sep-11 (1)	Jul-27 (2) Sep-3 (1), Sep-11 (1)	Jul-29 (2), Jul-16 (2), Aug-22 (2), Sep-4 (1)	Jul-5 (2), Jul-17 (1), Aug-3 (1), Aug-16 (1), Aug-30 (1), Sep-12 (1), Sep-26 (2)
2007	Aug-3 (2), Sep 6 (2)	Aug-3 (2), Sep 5 (2)	Aug-3 (2), Sep 5 (2)	Jul-10 (1)	Jul-3 (2), Jul-10 (1), Jul-16 (2), Aug-2 (2), Aug-16 (2), Sep-2 (2), Sep-16 (2)
2008	Aug-2 (2), Sep-4 (2)	Aug-1 (2), Sep-4 (2)	Aug-2 (2), Sep-5 (2)	-	Jul-4 (1), Jul-15 (1), Jul-29 (2), Aug-16 (1), Aug-26 (2), Sep-9 (1), Sep-23 (2)
2009	Aug-3 (2), Sep-4 (2)	Aug-3 (2), Sep-4 (2)	Aug-4 (2), Sep-4 (2)	Jul-5 (2), Jul-19 (1)	Jul-5 (1), Jul-19 (2), Aug-2 (2),
2010	Jul-3 (2), Aug-1 (2), Sep-8 (2)	Jul-4 (2), Aug-1 (2), Sep-8 (2)	Jul-5 (2), Aug-1 (2), Sep-8 (2)	-	Jul-10 (2), Jul-24 (2), Aug-7 (2), Aug-21 (2), Sep-4 (2), Sep-18 (2)
2011	Jul-3 (2), Jul-30 (2), Aug-30 (2)	Jul-3 (2), Jul-30 (2), Aug-30 (2)	Jul-3 (2), Jul-31 (2), Aug-31 (2)	Jul-24 (1), Sep-1 (1)	Jul-16 (1), Jul-24 (1), Aug-6 (1), Sep-1 (1),
2012	Jul-4 (2), Aug-4 (2), Sep-1 (2)	Jun-30 (2), Aug-5 (2), Aug-31 (2)	Jul-1 (2), Aug-5 (2), Sep-1 (2)	Jul-9 (2)	Jul-9 (1), Jul-23 (1), Aug-6 (1), Aug-19 (1), Sep-6 (2), Sep-16 (1), Sep-30 (1)
2013	Jul-2 (2), Aug-4 (2), Sep-3 (2)	Jul-2 (2), Aug-4 (2), Sep-3 (2)	Jul-2 (2), Aug-4 (2), Sep-3 (2)	Jul-8 (1)	Jul-8 (1), Jul-14 (1), Jul-28 (2), Aug-11 (1)
2014	Jul-5 (2), Aug-1 (2), Sep-2 (2)	Jul-5 (2), Aug-4 (2), Sep-4 (2)	Jul-5 (2), Aug-1 (2), Sep-2 (2)	Jul-3 (2), Aug-3 (2), Sep-4 (2)	Jul-3 (2), Jul-18 (2), Aug-3 (2), Aug-14 (2), Sep-4 (2), Sep-18 (2)

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

In Pigeon Stream, the timing and frequency of sampling has varied at the monitored site (i.e., Pigeon Reach 1) and the internal reference site (i.e., Pigeon Reach 7), with sampling occurring as part of the SNP program, baseline sampling, and the PSD Monitoring Program (Rescan 2011b). In 2014, open water stream sampling (i.e., July, August, and September) was representative of stream water quality inflow to Fay Bay. Thus, all open water season samples were used for the evaluation of effects in the Pigeon-Fay and Upper Exeter Watershed (Table 5.2-4).

Baseline water quality data for the monitored stream site (i.e., Pigeon Reach 1) were collected from 2001 to 2007, while baseline water quality data for three of the reference streams were collected from 1994 to 2007 (Table 5.2-4). Baseline data from a fourth internal reference stream site (i.e., Pigeon Reach 7) were also collected from 2004 to 2007 (Table 5.2-4). Water quality data collected from 2008 to 2013 were not used in the statistical analyses, but are included in Table 5.2-4 and shown in Figures 5.2-1 to 5.2-23 for visual comparison. Monitoring in the Pigeon-Fay and Upper Exeter Watershed from 2008 to 2010 was part of a monitoring program initiated in response to an unplanned release of FPK in May of 2008 (Rescan 2011b). Thus, data from the "before" period in the BACI analysis encompasses data from 1994 to 2007 and the "after" period includes all data collected in 2014. Although stream water quality sites in the Pigeon-Fay and Upper Exeter Watershed would not have been affected by the unplanned release of FPK, the same "before" and "after" periods were used for the BACI analysis of stream and lake data in order to facilitate comparisons between stream and lake sites.

For each variable, average values were calculated for each year by pooling data from all samples collected during that year. 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 (Table 5.2-3).

5.2.3 Statistical Description of Results

The statistical analyses assessed the evidence for effects on water quality, sediment quality, and phytoplankton variables (see Section 2.2.3). Complete statistical reports for all assessed variables are provided in Part 3 – Statistical Report. Section 5.2.4 summarizes the analyses for each variable, which considers both the results of the statistical and graphical analyses along with best professional judgement. In each section, the conclusions of the statistical analyses are presented along with contextual information. As discussed in Section 2.2.5.2, statistical analyses combined hypothesis tests on a BACI interaction term (site:period) with a contrast term analysis to identify significant differences. The BACI interaction term tested for significant site-specific differences between the before and after periods among the monitored and reference sites. The contrast terms were then used to compare the monitored site to each reference site. Mine effects were identified based on both the hypothesis test and the contrast terms—both represent aspects of the statistical model relevant to the evaluation of effects.

5.2.4 Results and Discussion

5.2.4.1 pH

Summary: Together, statistical and graphical analyses suggest that pH has not increased in the Pigeon-Fay and Upper Exeter Watershed as a result of mine operations. Observed pH was within the CCREM guideline values at all monitored sites in 2014. No mine effects were detected.

Statistically significant changes in pH were detected between the two monitored lakes (i.e., Fay Bay and Upper Exeter) and reference lakes during the open water season, when comparing the before and after periods (Table 5.2-5). Graphical analysis shows that pH increased in all monitored lakes and streams of the Pigeon-Fay and Upper Exeter Watershed during the 2014 open water season when compared to the before period, but a similar trend was observed at all reference sites (Figure 5.2-1). Graphical analysis also indicates that although pH in monitored lakes during the ice-covered season was elevated relative to reference lakes, pH values remained similar between the before and after periods (Figure 5.2-1). Thus, no mine effects were detected. The observed mean pH values were within the range of the CCREM guideline values (pH 6.5 to 9) in all monitored lakes and streams in 2014 (CCREM 1987).

Table 5.2-5. Statistical Results of pH in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	-	$p = 0.22$	-	3-1
	Upper Exeter	-	$p = 0.28$	-	
Summer	Fay Bay	-	$p < 0.0001$	Nanuq, Counts, Vulture	3-10
	Upper Exeter	-	$p < 0.0001$	Nanuq, Counts	
Summer	Pigeon Reach 1	-	$p = 0.62$	-	3-19

Note: Dashes indicate not applicable.

5.2.4.2 *Total Alkalinity*

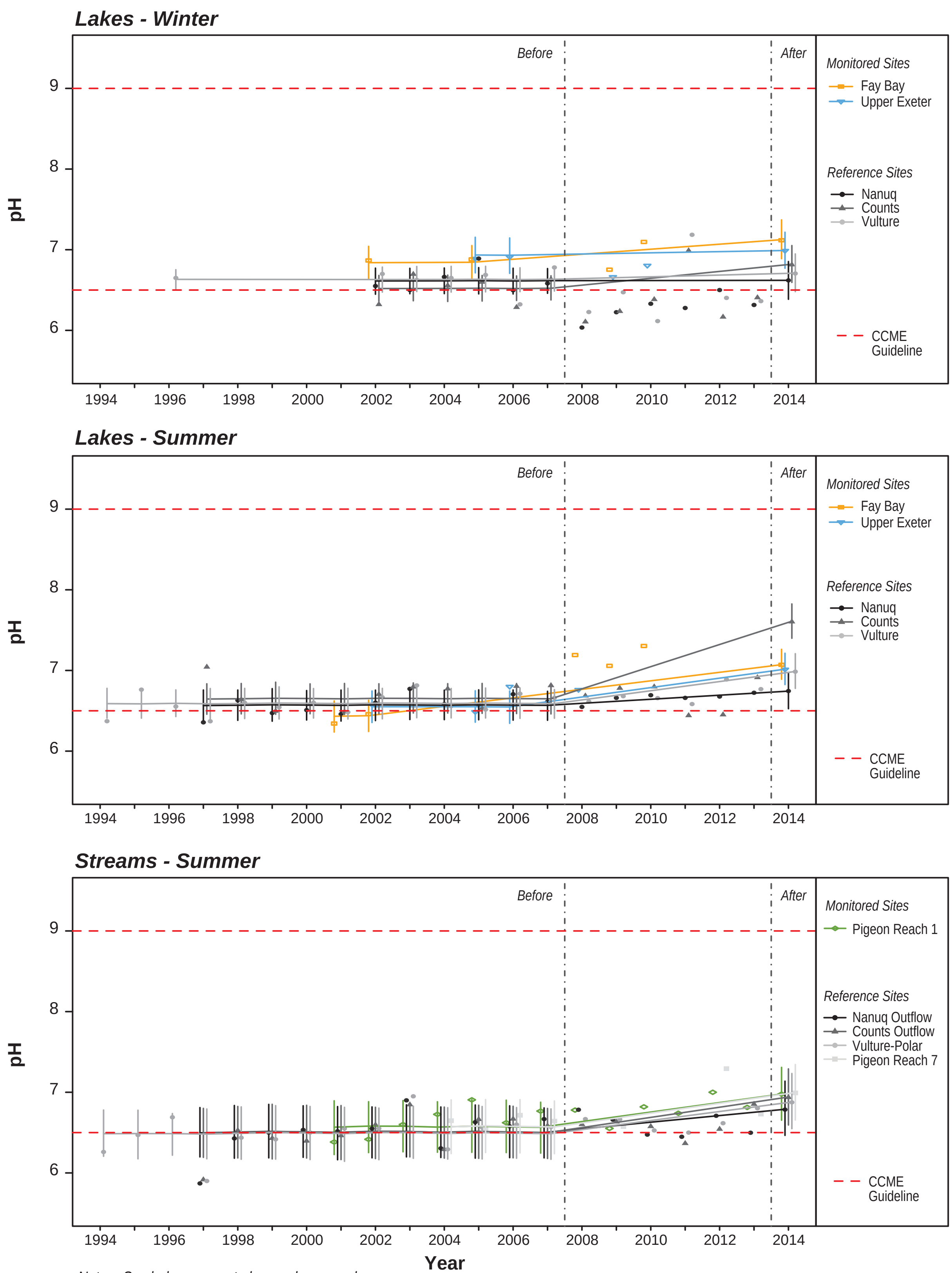
Summary: Statistical and graphical analyses suggest that total alkalinity has increased in Fay Bay during the ice-covered season. These changes may be related to the unplanned release of FPK in May of 2008. Alkalinity during the open water season remained within the range of values observed during the before period.

Statistically significant changes in total alkalinity were detected between Fay Bay and reference lakes, and between Pigeon Reach 1 and reference streams, when comparing the before and after periods (Table 5.2-6). Graphical analysis shows that total alkalinity in 2014 increased in Fay Bay during the ice-covered season when compared to the before period (Figure 5.2-2). Graphical analysis also indicates that open water season alkalinity in Fay Bay and Pigeon Reach 1 appears to have increased in 2014; however, concentrations were within the range of values observed during the before period (Figure 5.2-2). Total alkalinity in Upper Exeter Lake has remained similar between the before and after periods during the ice-covered and open water season (Table 5.2-6; Figure 5.2-2).

The source of the observed increase in Fay Bay during the ice-covered season is unclear at this time, but may be related to the unplanned release of FPK in May of 2008 (Rescan 2011b). Increases are unlikely related to the PSD since increases were observed during the ice-covered season, prior to the connection of the PSD to the natural Pigeon Stream.

Figure 5.2-1

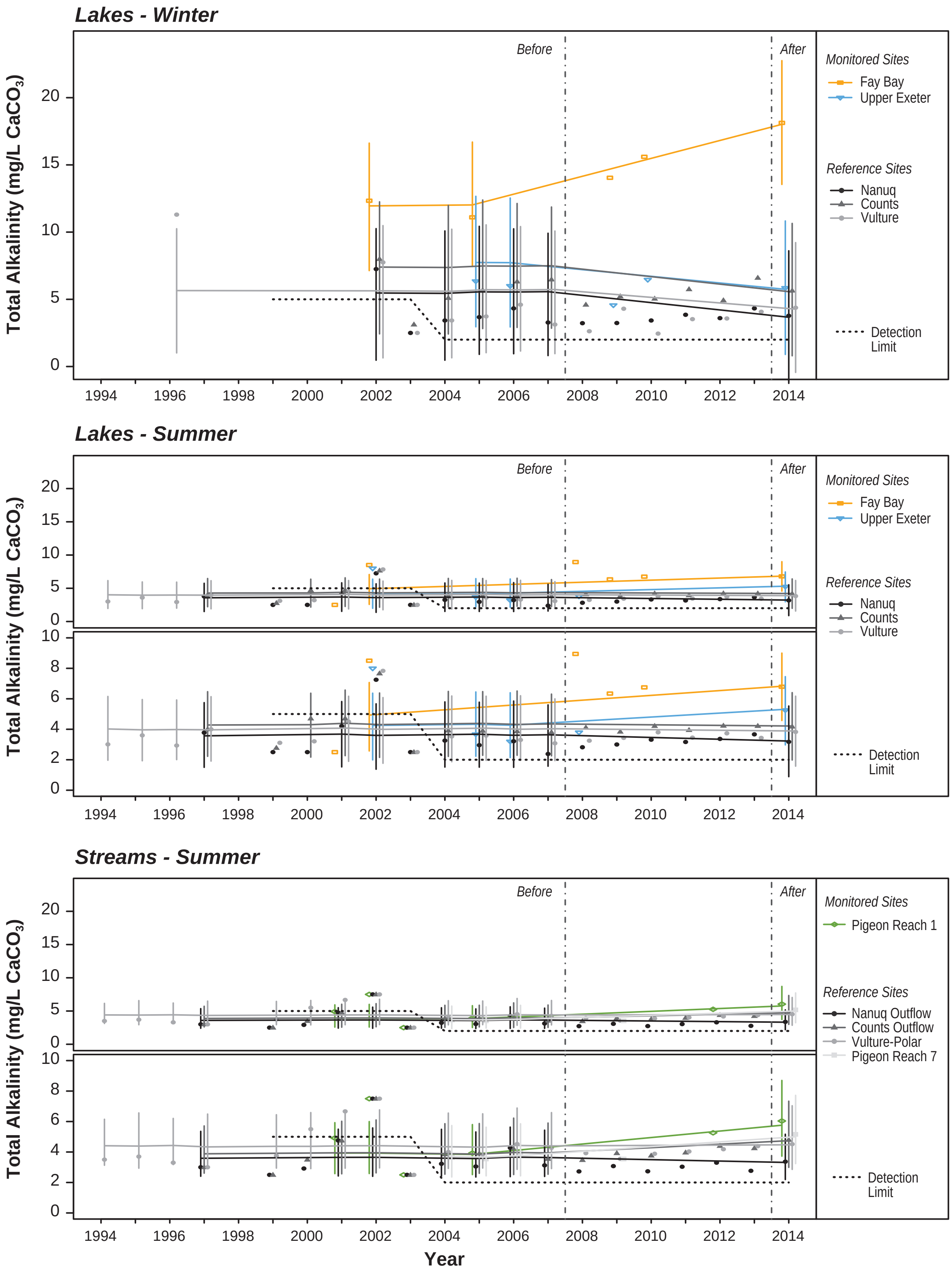
Observed and Fitted Means for pH in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
CCME guideline = 6.5 - 9.0.

Figure 5.2-2

Observed and Fitted Means for Total Alkalinity in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.

Table 5.2-6. Statistical Results of Total Alkalinity in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	-	$p < 0.0001$	Nanuq, Counts, Vulture	3-26
	Upper Exeter	-	$p = 0.69$	-	
Summer	Fay Bay	-	$p = 0.002$	Nanuq, Counts, Vulture	3-35
	Upper Exeter	-	$p = 0.06$	-	
Summer	Pigeon Reach 1	-	$p = 0.01$	Nanuq Outflow, Vulture-Polar	3-46

Note: Dashes indicate not applicable.

5.2.4.3 *Water Hardness*

Summary: Statistical and graphical analyses suggest that water hardness has increased in Fay Bay during both the ice-covered and open water season and in Upper Exeter Lake during the open water season. These changes may be related to the unplanned release of FPK in May of 2008.

Statistically significant changes in water hardness were detected between all monitored and reference lakes when comparing the before and after periods, with the exception of Upper Exeter Lake during the ice-covered season (Table 5.2-7). Graphical analysis shows that water hardness in Fay Bay and Upper Exeter Lake is naturally elevated, relative to reference sites, but was also elevated in 2014, when compared to the before period, in Fay Bay during the ice-covered season and in both monitored lakes during the open water season (Figure 5.2-3).

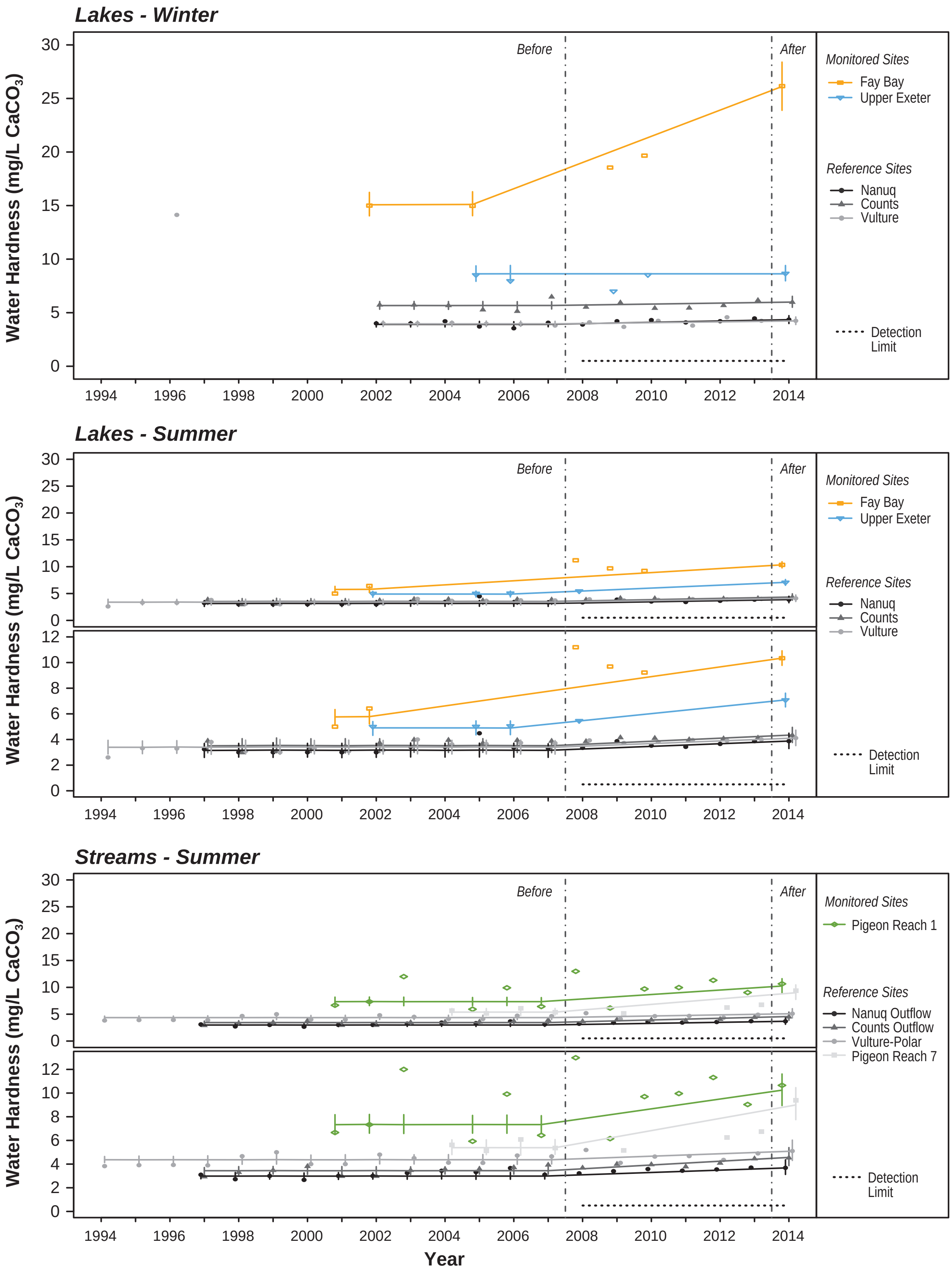
Table 5.2-7. Statistical Results of Water Hardness in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	-	$p < 0.0001$	Nanuq, Counts, Vulture	3-54
	Upper Exeter	-	$p = 0.36$	-	
Summer	Fay Bay	-	$p < 0.0001$	Nanuq, Counts, Vulture	3-63
	Upper Exeter	-	$p < 0.0001$	Nanuq, Counts, Vulture	
Summer	Pigeon Reach 1	-	$p = 0.04$	None	3-72

Note: Dashes indicate not applicable.

Figure 5.2-3

Observed and Fitted Means for Water Hardness in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.

Results from the BACI analyses also indicate that changes in water hardness were detected between Pigeon Reach 1 and reference streams when comparing the before and after periods (Table 5.2-7). However, results from the BACI contrasts indicate that the trend in Pigeon Reach 1 was similar to that observed at all four reference stream sites (Table 5.2-7). Furthermore, graphical analysis indicates that water hardness in 2014 was within the range of values observed during the before period for Pigeon Reach 1 (Figure 5.2-3).

Together, the results indicate that water hardness was naturally elevated in the Pigeon-Fay and Upper Exeter Watershed, but that hardness has increased relative to reference sites in Fay Bay during both the ice-covered and open water seasons and in Upper Exeter Lake during the open water season. The observed increase in Fay Bay may be related to the unplanned release of FPK in May of 2008 (Rescan 2011b). Increases in Fay Bay are unlikely related to the PSD since increases were observed during the ice-covered season, prior to the connection of the PSD to the natural Pigeon Stream.

5.2.4.4 Chloride

> ***Summary:*** Together, statistical and graphical analyses suggest that chloride concentrations have increased in Fay Bay. These changes may be related to the unplanned release of FPK in May of 2008. Observed concentrations were less than the hardness-dependent chloride SSWQO at all sites in 2014.

Chloride concentrations in reference lakes have generally been below the detection limit since monitoring began. Thus, no statistical analyses were possible for lake chloride concentrations (Table 5.2-8). However, graphical analysis indicates that chloride concentrations have likely increased in Fay Bay as concentrations in 2014 are greater than those observed during the before period (Figure 5.2-4).

Table 5.2-8. Statistical Results of Chloride in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

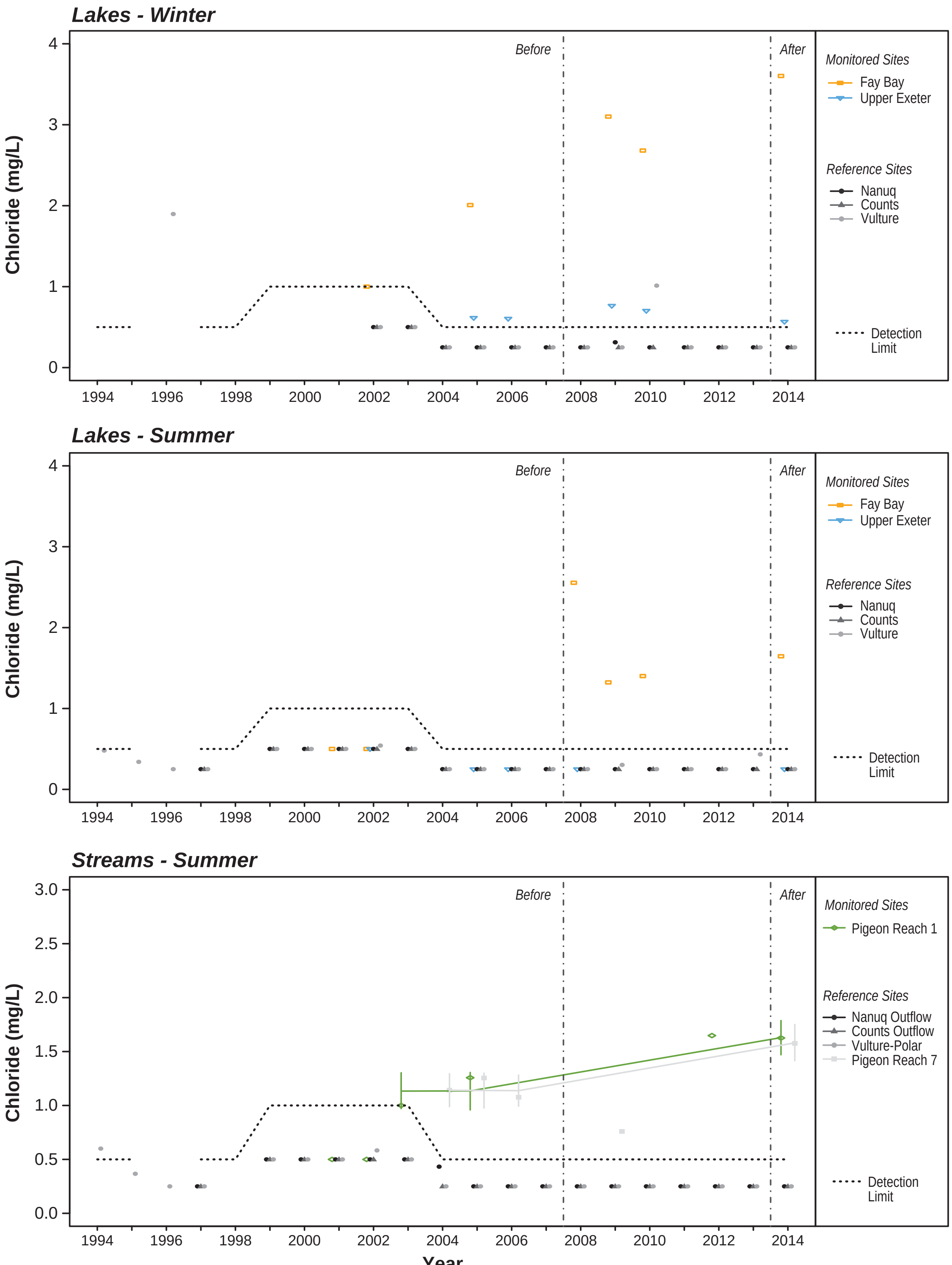
Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	Nanuq, Counts, Vulture	-	-	3-79
	Upper Exeter	Nanuq, Counts, Vulture	-	-	
Summer	Fay Bay	ALL	-	-	3-82
	Upper Exeter	ALL	-	-	
Summer	Pigeon Reach 1	Nanuq Outflow, Counts Outflow, Vulture-Polar	$p = 0.52$	-	3-85

Note: Dashes indicate not applicable.

Chloride concentrations have also generally been below detection limits in the three external AEMP reference streams, but were detected in both Pigeon Stream sites. Statistical analyses indicate that no changes in chloride concentrations were detected between the monitored and reference site in Pigeon Stream, when comparing the before and after periods (Table 5.2-8). However, graphical analysis indicates that chloride concentration in 2014 increased in both the monitored and reference sites in Pigeon Stream, when compared to the before period (Figure 5.2-4).

Figure 5.2-4

Observed and Fitted Means for Chloride Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
SSWQO = 116.6 X ln(Hardness) - 204.1, where hardness = 10 - 160 mg/L.

Together, the results indicate that chloride concentrations are naturally elevated in the Pigeon-Fay and Upper Exeter Watershed, but that increases in chloride concentrations have been observed in Fay Bay and Pigeon Stream. The increase observed in Fay Bay may be related to the unplanned release of FPK in May of 2008 (Rescan 2011b). The increase in Fay Bay is unlikely related to the PSD since increases were observed during the ice-covered season, prior to the connection of the PSD to the natural Pigeon Stream. The increase observed in Pigeon Reach 1 likely reflects natural influences as chloride concentrations also increased at the internal Pigeon Stream reference site (Figure 5.2-4).

Observed mean chloride concentrations were less than the hardness-dependent chloride SSWQO in all monitored lakes and streams in 2014 (see Part 2 - Data Report; Elphick, Bergh, and Bailey 2011).

5.2.4.5 Sulphate

> ***Summary:*** Statistical and graphical analyses suggest that sulphate concentrations have increased in Fay Bay during the ice-covered season. However, the source of the observed increase is unclear at this time and may reflect naturally variability in sulphate concentrations in Fay Bay. Observed concentrations were less than the hardness-dependent sulphate SSWQO at all sites in 2014.

Statistically significant changes in sulphate concentrations were detected between all monitored and reference lakes when comparing the before and after periods (Table 5.2-9). Graphical analysis shows that although sulphate concentrations in Fay Bay and Upper Exeter Lake are naturally elevated relative to reference sites, they were also elevated in 2014, when compared to the before period, in Fay Bay during the ice-covered season and in both monitored lakes during the open water season (Figure 5.2-5). However, the increase observed during the open water season is minimal and concentrations observed in Fay Bay during the open water season of 2014 were within the range of values observed within the Pigeon Stream (Figure 5.2-5).

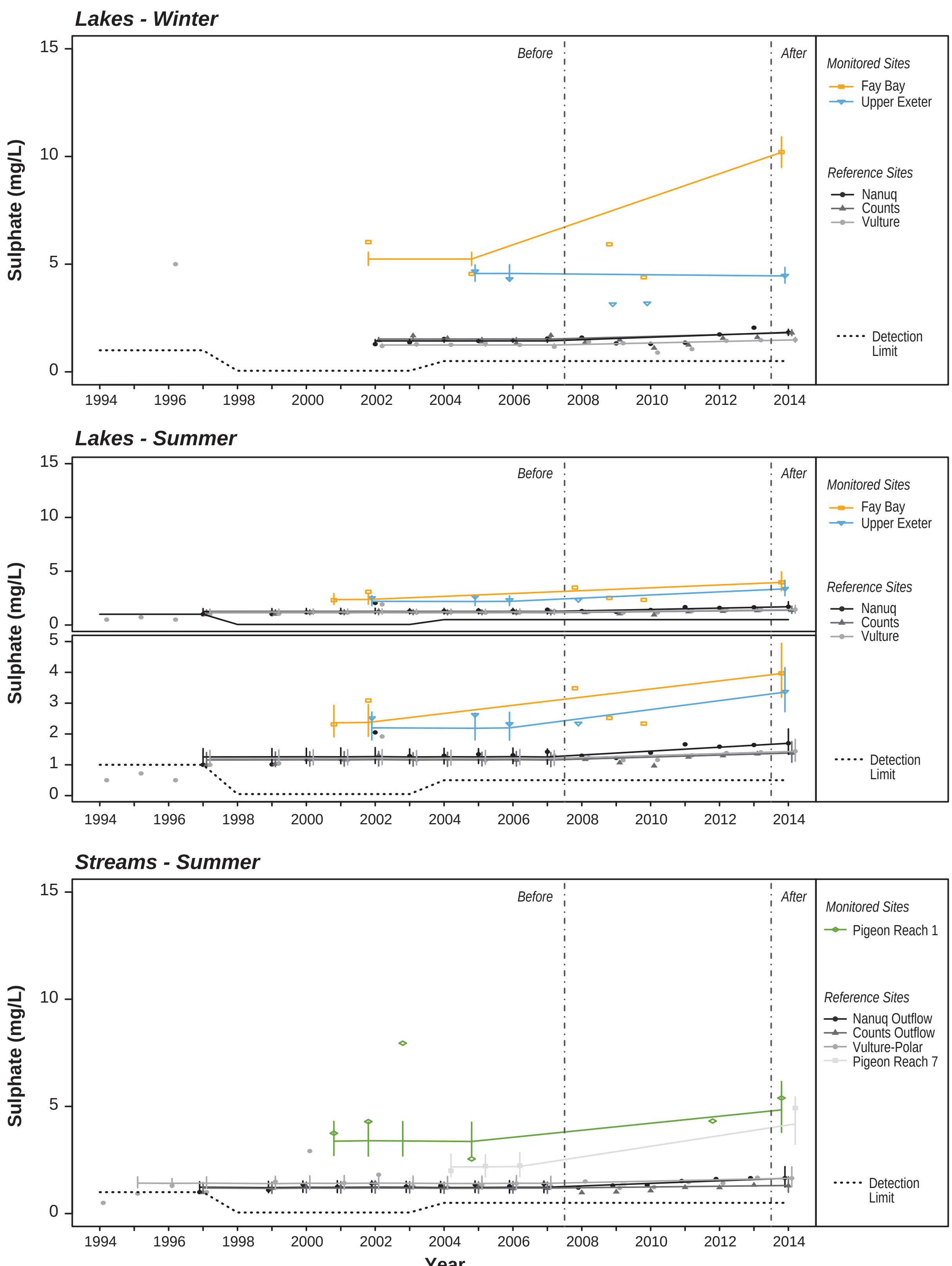
Table 5.2-9. Statistical Results of Sulphate in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results: BACI Interaction Term	BACI Results: Significant BACI Contrasts	Statistical Report Page No.
Winter	Fay Bay	-	$p < 0.0001$	Counts, Nanuq, Vulture	3-95
	Upper Exeter	-	$p = 0.001$	Counts, Nanuq, Vulture	
Summer	Fay Bay	-	$p = 0.002$	Counts, Nanuq, Vulture	3-104
	Upper Exeter	-	$p = 0.04$	Counts, Vulture	
Summer	Pigeon Reach 1	-	$p = 0.01$	None	3-113

Note: Dashes indicate not applicable.

Figure 5.2-5

Observed and Fitted Means for Sulphate Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
SSWQO = $e^{(0.9116 \times \ln(\text{Hardness}) + 1.712)}$ mg/L, where hardness < 160 mg/L.

Results from the BACI analyses also indicate that changes in sulphate concentrations were detected between Pigeon Reach 1 and reference streams when comparing the before and after periods (Table 5.2-9). However, results from the BACI contrasts indicate that the trend in Pigeon Reach 1 was similar to that observed at all four reference stream sites (Table 5.2-9). Furthermore, graphical analysis indicates that the mean sulphate concentration in 2014 is within the range of values observed during the before period for Pigeon Reach 1 (Figure 5.2-5).

Together, results indicate that sulphate concentrations are naturally elevated in the Pigeon-Fay and Upper Exeter Watershed, but that sulphate concentrations have increased, relative to reference sites, in Fay Bay during the ice-covered season. Sulphate concentrations did not increase in Fay Bay and Upper Exeter following the unplanned release of FPK in May of 2008 (Figure 5.2-5; Rescan 2011b). Furthermore, effects from the connection of the PSD to the natural Pigeon Stream would not have been possible during the winter of 2014. Thus, the source of the increase observed in Fay Bay during the ice-covered season is unclear at this time and may reflect naturally variability in sulphate concentrations in Fay Bay.

Observed mean sulphate concentrations were less than the hardness-dependent sulphate SSWQO in all reference and monitored lakes and streams in 2014 (see Part 2 - Data Report; Rescan 2012e).

5.2.4.6 Potassium

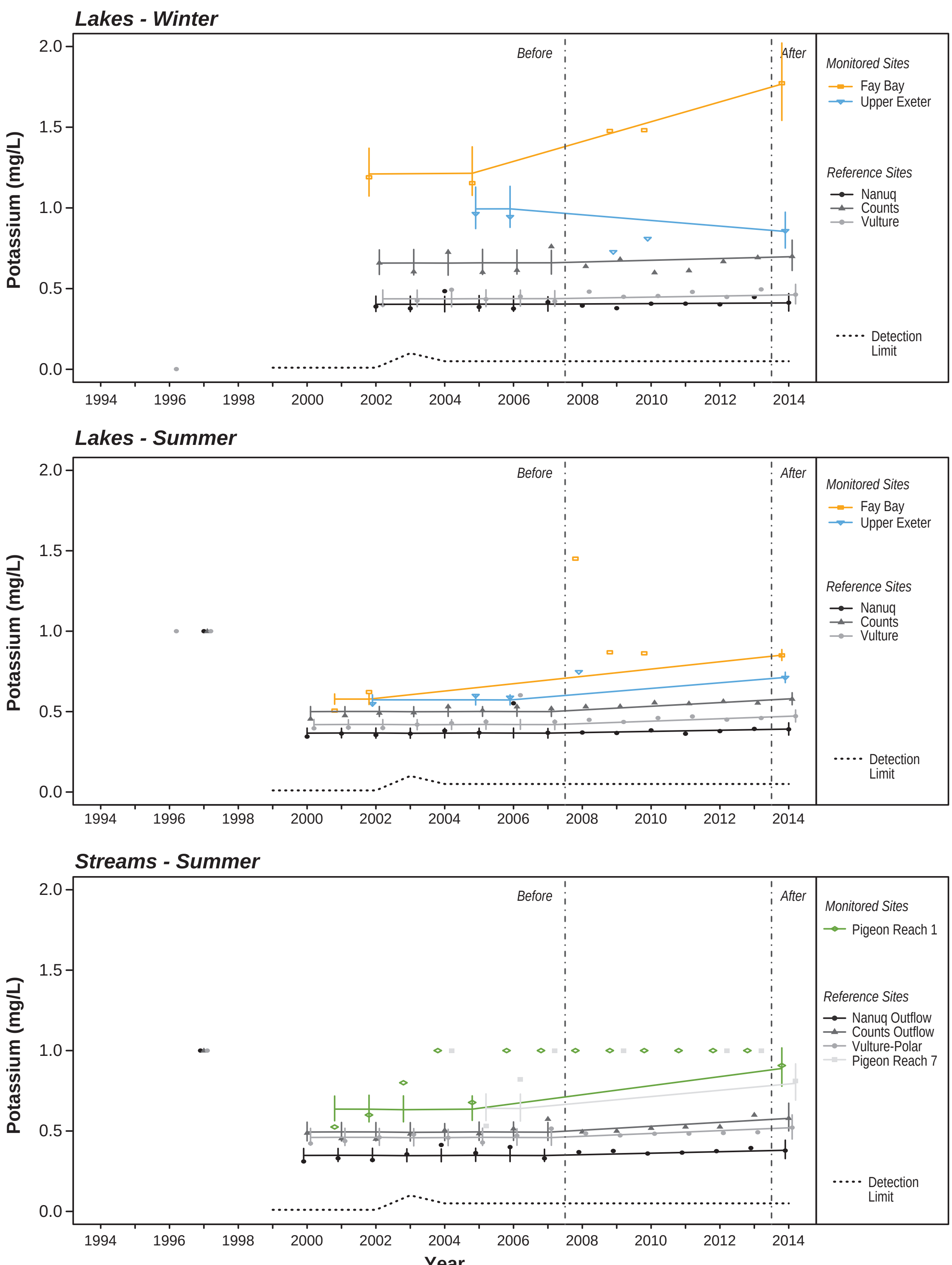
> ***Summary:*** Statistical and graphical analyses suggest that potassium concentrations have increased in Fay Bay during the ice-covered season. These changes may be related to the unplanned release of FPK in May of 2008. Observed concentrations were less than the potassium SSWQO at all monitored sites in 2014.

Statistically significant changes in potassium concentrations were detected between all monitored and reference lakes when comparing the before and after periods (Table 5.2-10). Graphical analysis shows that although potassium concentrations in Fay Bay and Upper Exeter Lake are naturally elevated relative to reference sites, they were also elevated in 2014, when compared to the before period, in Fay Bay during the ice-covered season and in both monitored lakes during the open water season (Figure 5.2-6). However, the increase observed during the open water season is minimal and concentrations observed during the open water season in Fay Bay were within the range of values observed within the Pigeon Stream (Figure 5.2-6). Graphical analysis also indicates that potassium concentrations have decreased slightly during the ice-covered season in Upper Exeter Lake in 2014, when compared to the before period (Figure 5.2-6).

Statistically significant changes in potassium concentrations were detected between Pigeon Reach 1 and reference streams when comparing the before and after periods (Table 5.2-10). However, results from the BACI contrasts indicate that the trend in Pigeon Reach 1 differed from all reference stream sites, with the exception of Pigeon Reach 7 (Table 5.2-10). Graphical analysis also indicates that potassium concentrations increased at both sites within the Pigeon Stream in 2014, when compared to the before period (Figure 5.2-6). Pigeon Reach 7 is an internal reference site for the Pigeon-Fay and Upper Exeter Watershed. The similarity in the trend observed at both sites within Pigeon Stream suggests that potassium concentrations may be naturally elevated and increasing within the watershed.

Figure 5.2-6

Observed and Fitted Means for Potassium Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
SSWQO = 41 mg/L.

Table 5.2-10. Statistical Results of Potassium in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	-	p<0.0001	Counts, Nanuq, Vulture	3-120
	Upper Exeter	-	p=0.006	Counts, Nanuq, Vulture	
Summer	Fay Bay	-	p<0.0001	Counts, Nanuq, Vulture	3-129
	Upper Exeter	-	p<0.0001	Counts, Nanuq, Vulture	
Summer	Pigeon Reach 1	-	p=0.02	Counts Outflow, Nanuq Outflow, Vulture-Polar	3-138

Note: Dashes indicate not applicable.

Together, results indicate that potassium concentrations are naturally elevated in the Pigeon-Fay and Upper Exeter Watershed, but that potassium concentrations have increased, relative to reference sites, in Fay Bay during the ice-covered season. The relative increase observed in Fay Bay during the ice-covered season may be related to the unplanned release of FPK in May of 2008 (Rescan 2011b). Increases are unlikely related to the PSD since increases were observed during the ice-covered season, prior to the connection of the PSD to the natural Pigeon Stream.

Observed mean potassium concentrations were less than the long-term potassium SSWQO (41 mg/L) in all monitored lakes and streams in 2014 (see Part 2 - Data Report; Rescan 2012f).

5.2.4.7 Total Suspended Solids

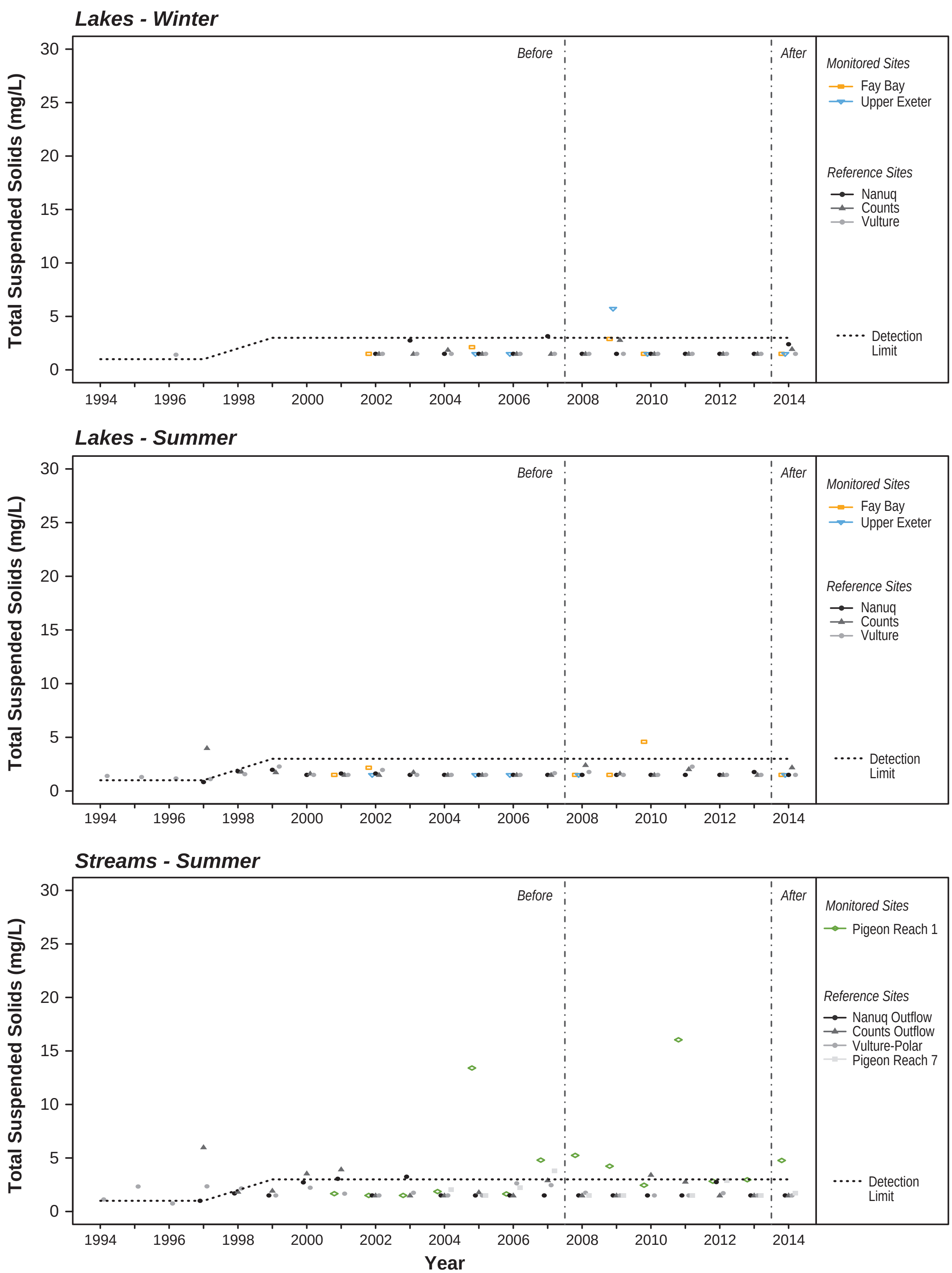
> ***Summary:*** Concentrations of TSS have generally been below detection limits since monitoring began. All concentrations in 2014 were less than the CCME guideline. No mine effects were detected

TSS concentrations in virtually all monitored and reference lakes and streams have been below the detection limit since monitoring began. Thus, no statistical analyses were possible for TSS (Table 5.2-11). TSS was detected in Pigeon Reach 1 on some sampling occasions, but graphical analysis indicates that concentrations in 2014 are within the range of concentrations observed during the before period (Figure 5.2-7). Thus, no mine effects were detected.

Since all observations in 2014 were below detection limits or less than 5 mg/L, no observations were greater than the CCME guideline in any lakes or streams of the Pigeon-Fay and Upper Exeter Watershed (CCME 1999e).

Figure 5.2-7

Observed and Fitted Means for Total Suspended Solids Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: *Symbols represent observed mean values.*
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.

Table 5.2-11. Statistical Results of TSS in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	ALL	-	-	3-145
	Upper Exeter	ALL	-	-	
Summer	Fay Bay	ALL	-	-	3-148
	Upper Exeter	ALL	-	-	
Summer	Pigeon Reach 1	ALL	-	-	3-151

Note: Dashes indicate not applicable.

5.2.4.8 Total Ammonia-N

Summary: Statistical and graphical analyses suggest that mine operations have had no effect on total ammonia-N concentrations in any of the lakes or streams in the Pigeon-Fay and Upper Exeter Watershed. 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 2014.

Statistically significant changes in total ammonia-N concentrations were detected between monitored and reference lakes during the ice-covered season when comparing the before and after periods (Table 5.2-12). Graphical analysis indicates that total ammonia-N concentrations during the ice-covered season in 2014 were within the range of values observed during the before period in Fay Bay and decreased in Upper Exeter Lake from the before to the after period (Figure 5.2-8). Most observations during the open water season were at or below analytical detection limits, thus no statistical analyses were possible (Table 5.2-12). Total ammonia-N concentrations were detected in Fay Bay occasionally during the before period and from 2008 to 2010 during the open water season, but were below the detection limit in 2014 (Figure 5.2-8).

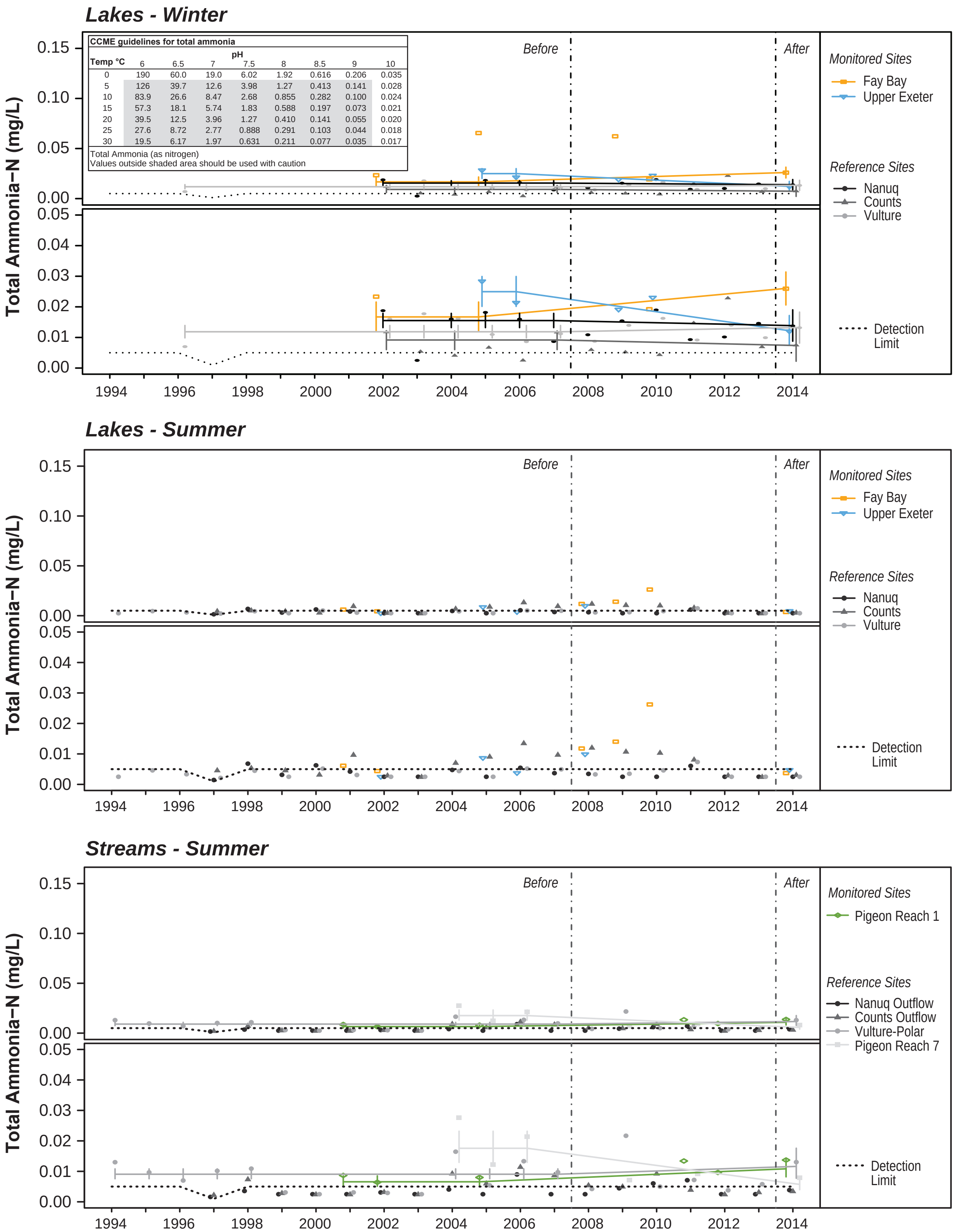
Table 5.2-12. Statistical Results of Total Ammonia-N in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	-	$p = 0.21$	-	3-154
	Upper Exeter	-	$p = 0.06$	-	
Summer	Fay Bay	ALL	-	-	3-163
	Upper Exeter	ALL	-	-	
Summer	Pigeon Reach 1	Nanuq Outflow, Counts Outflow	$p = 0.0001$	Pigeon Reach 7	3-166

Note: Dashes indicate not applicable.

Figure 5.2-8

Observed and Fitted Means for Total Ammonia-N Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



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 represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
CCME Guideline is pH and temperature dependent (see inset table).

Statistical and graphical analyses indicate that total ammonia-N concentrations appear to have increased at Pigeon Reach 1, which is in contrast to the decrease observed at the upstream reference site (i.e., Pigeon Reach 7; Table 5.2-12; Figure 5.2-8). However, results also indicate that a similar increasing trend in total ammonia-N concentrations was observed at one of the AEMP reference streams (i.e., Vulture-Polar; Table 5.2-12; Figure 5.2-8). Since elevated concentrations of total ammonia-N were observed in Pigeon Reach 1 prior to the opening of the PSD (i.e., in 2011 and 2012), and a similar trend was observed in Vulture-Polar, it was concluded that the observed increase in total ammonia-N was unlikely related to mine activities.

Observed mean total ammonia-N concentrations were less than the pH- and temperature-dependent ammonia CCME guideline in all monitored lakes and streams in 2014 (see Part 2 - Data Report; CCME 2001b).

5.2.4.9 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 2014. No mine effects were detected.

Nitrite-N concentrations in all monitored and reference lakes and streams have generally been below the detection limit since monitoring began. Thus, no statistical analyses were possible for nitrite-N concentrations (Table 5.2-13). Graphical analysis indicates that all observations in 2014 remained at or below the detection limit (Figure 5.2-9). Low concentrations of nitrite-N are likely related to low concentrations of total ammonia-N in the Pigeon-Fay and Upper Exeter Watershed, since nitrite is primarily formed through the oxidisation of ammonia (Figure 5.2-8). Moreover, nitrite is a relatively transient form of nitrogen, which quickly oxidises to produce nitrate. All 2014 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).

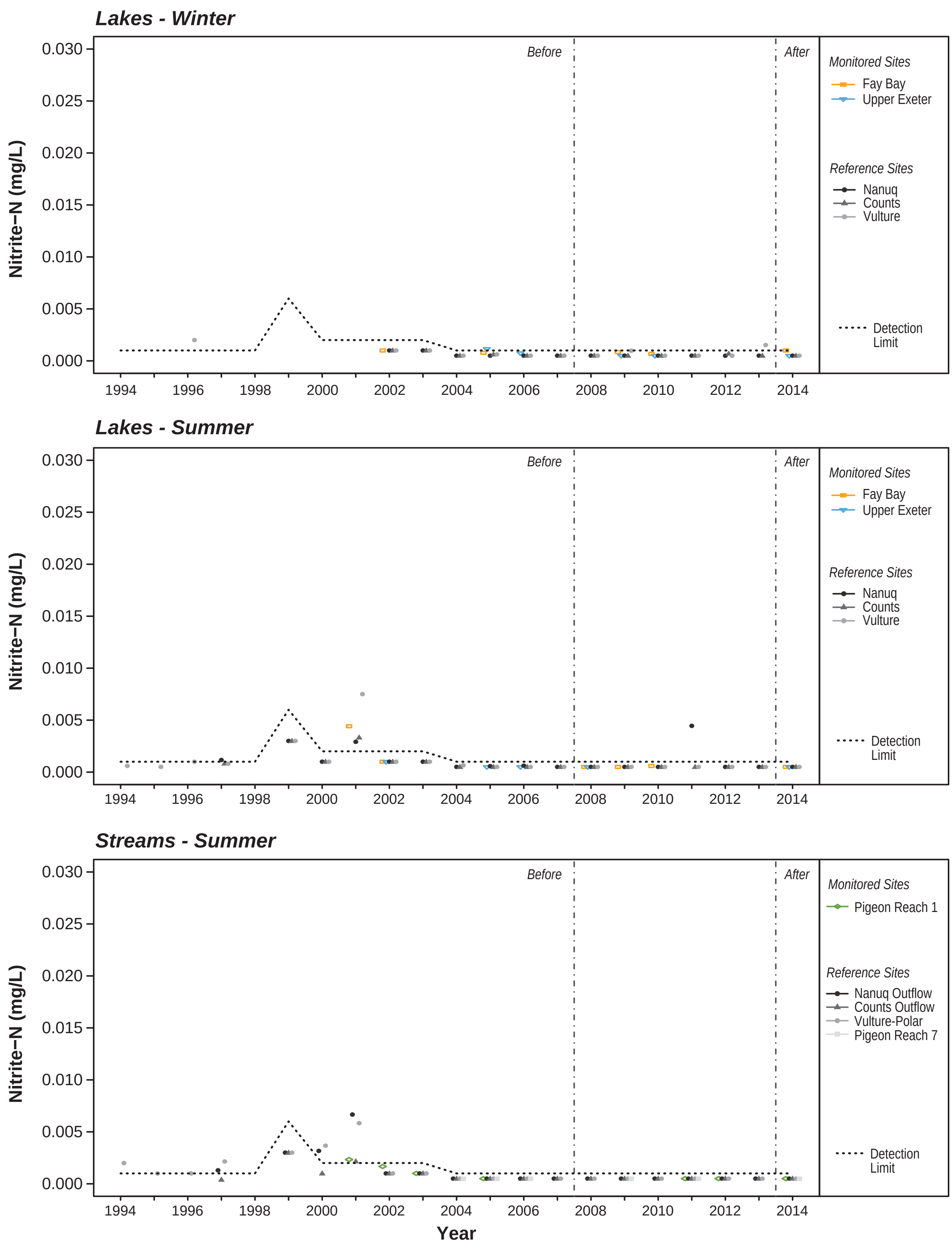
Table 5.2-13. Statistical Results of Nitrite-N in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	ALL	-	-	3-176
	Upper Exeter	ALL	-	-	
Summer	Fay Bay	ALL	-	-	3-179
	Upper Exeter	ALL	-	-	
Summer	Pigeon Reach 1	ALL	-	-	3-182

Note: Dashes indicate not applicable.

Figure 5.2-9

Observed and Fitted Means for Nitrite-N Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: Symbols represent observed mean values; solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
CCME Guideline = 0.06 mg/L.

5.2.4.10 Nitrate-N

> ***Summary:*** Statistical and graphical analyses suggest that mine operations have had no effect on total nitrate-N concentrations in any of the lakes or streams in the Pigeon-Fay and Upper Exeter Watershed. Concentrations were less than the hardness-dependent nitrate-N SSWQO at all sites in 2014.

No statistically significant changes in nitrate-N concentrations were detected between monitored and reference lakes during the ice-covered season when comparing the before and after periods (Table 5.2-14). Graphical analysis confirms that nitrate-N concentrations during the ice-covered season remained similar between the before and after periods (Figure 5.2-10). During the open water season, most observations were at or below analytical detection limits, thus no statistical analyses were possible (Table 5.2-14; Figure 5.2-10).

Table 5.2-14. Statistical Results of Nitrate-N in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	-	$p = 0.28$	-	3-185
	Upper Exeter	-	$p = 0.24$	-	
Summer	Fay Bay	ALL	-	-	3-194
	Upper Exeter	ALL	-	-	
Summer	Pigeon Reach 1	Counts Outflow, Pigeon Reach 7	$p = 0.15$	-	3-197

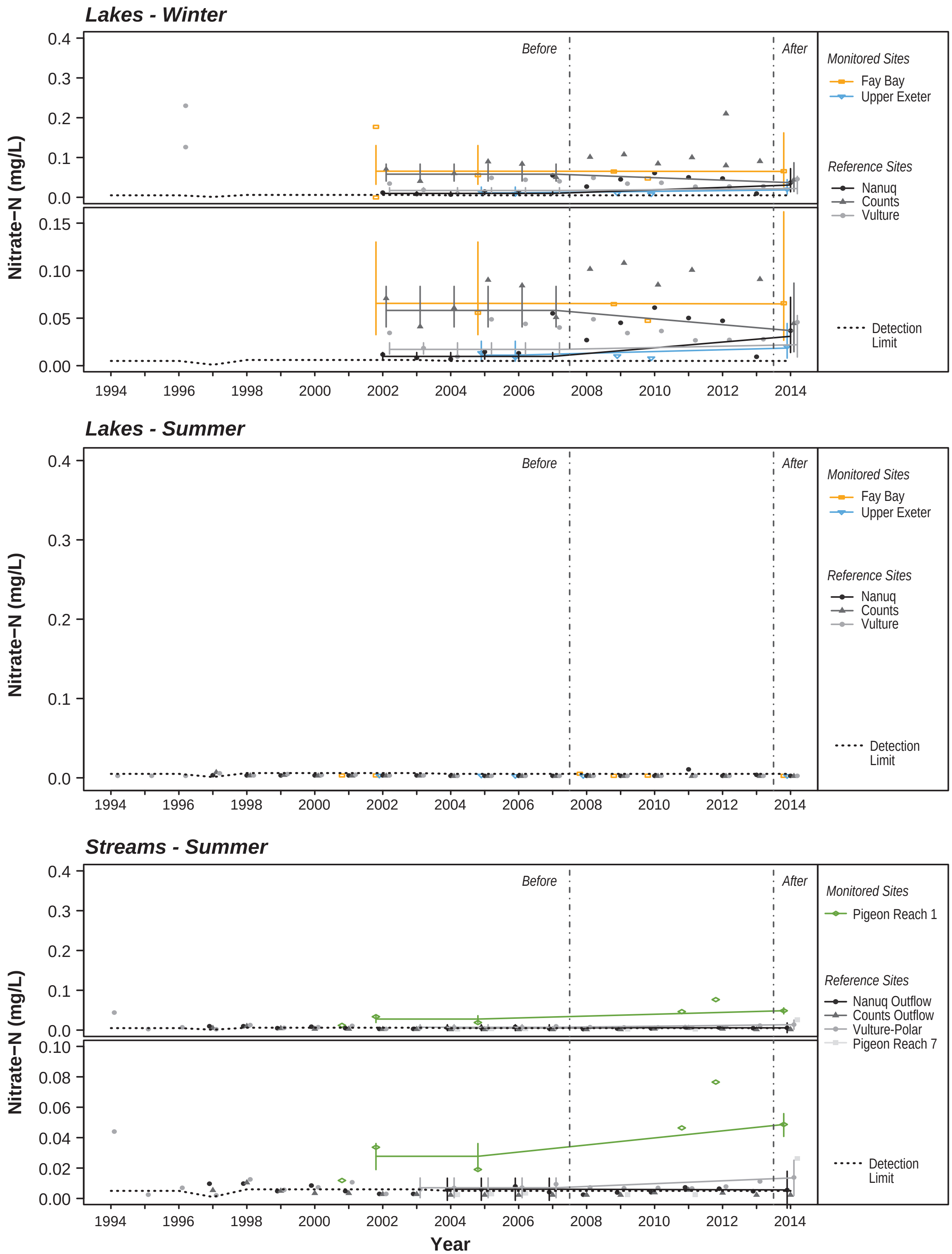
Note: Dashes indicate not applicable.

No statistically significant changes in nitrate-N concentrations were detected between monitored and reference streams when comparing the before and after periods (Table 5.2-14). Graphical analysis indicates that nitrate-N concentrations in Pigeon Steam Reach 1 appear to have increased in 2014, when compared to the before period (Figure 5.2-10). However, results also indicate that a similar increasing trend in nitrate-N concentrations was observed at one of the AEMP reference streams (i.e., Vulture-Polar; Table 5.2-14: Figure 5.2-10). Since elevated concentrations of total nitrate-N were observed in Pigeon Reach 1 prior to the opening of the PSD (i.e., in 2011 and 2012), and a similar trend was observed in Vulture-Polar, it was concluded that the observed increase in total nitrate-N was unlikely to be related to mine activities.

Observed mean nitrate-N concentrations were less than the hardness-dependent nitrate-N SSWQO in all reference and monitored lakes and streams in 2014 (see Part 2 - Data Report; Rescan 2012d).

Figure 5.2-10

Observed and Fitted Means for Nitrate-N Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: Symbols represent observed mean values; solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
SSWQO = $e^{0.9518 \times \ln(\text{Hardness}) - 2.032}$ mg/L, where hardness < 160mg/L.

5.2.4.11 *Total Phosphate-P*

Summary: Statistical and graphical analyses suggest that mine operations have had no effect on total phosphate-P concentrations in any of the lakes or streams in the Pigeon-Fay and Upper Exeter Watershed. Concentrations were less than the benchmarks at all sites in 2014.

Statistically significant changes in total phosphate-P concentrations were detected between all monitored lakes when comparing the before and after periods, with the exception of Fay Bay during the ice-covered season (Table 5.2-15). However, results from the BACI contrasts indicate that the trends observed in Fay Bay and Upper Exeter Lake were similar to those observed in at least two reference lakes (Table 5.2-15). Furthermore, graphical analysis indicates that 2014 total phosphate-P concentrations were within the range of values observed during the before period for both monitored lakes (Figure 5.2-11).

Table 5.2-15. Statistical Results of Total Phosphate-P in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	-	p = 0.08	-	3-207
	Upper Exeter	-	p = 0.05	None	
Summer	Fay Bay	-	p = 0.003	Counts	3-216
	Upper Exeter	-	p = 0.0003	Counts	
Summer	Pigeon Reach 1	-	p = 0.61	-	3-226

Note: Dashes indicate not applicable.

No statistically significant changes in total phosphate-P concentrations were detected between Pigeon Reach 1 and reference streams when comparing the before and after periods (Table 5.2-15). Graphical analysis indicates that total phosphate-P concentrations in Pigeon Reach 1 remained similar in the before and after periods (Figure 5.2-11). Total phosphate-P concentrations in both Pigeon stream sites appear to have been elevated at times between 2004 and 2010; however, detection limits for many of these samples were also elevated and concentrations in Pigeon Stream sites were below detection limits in those samples.

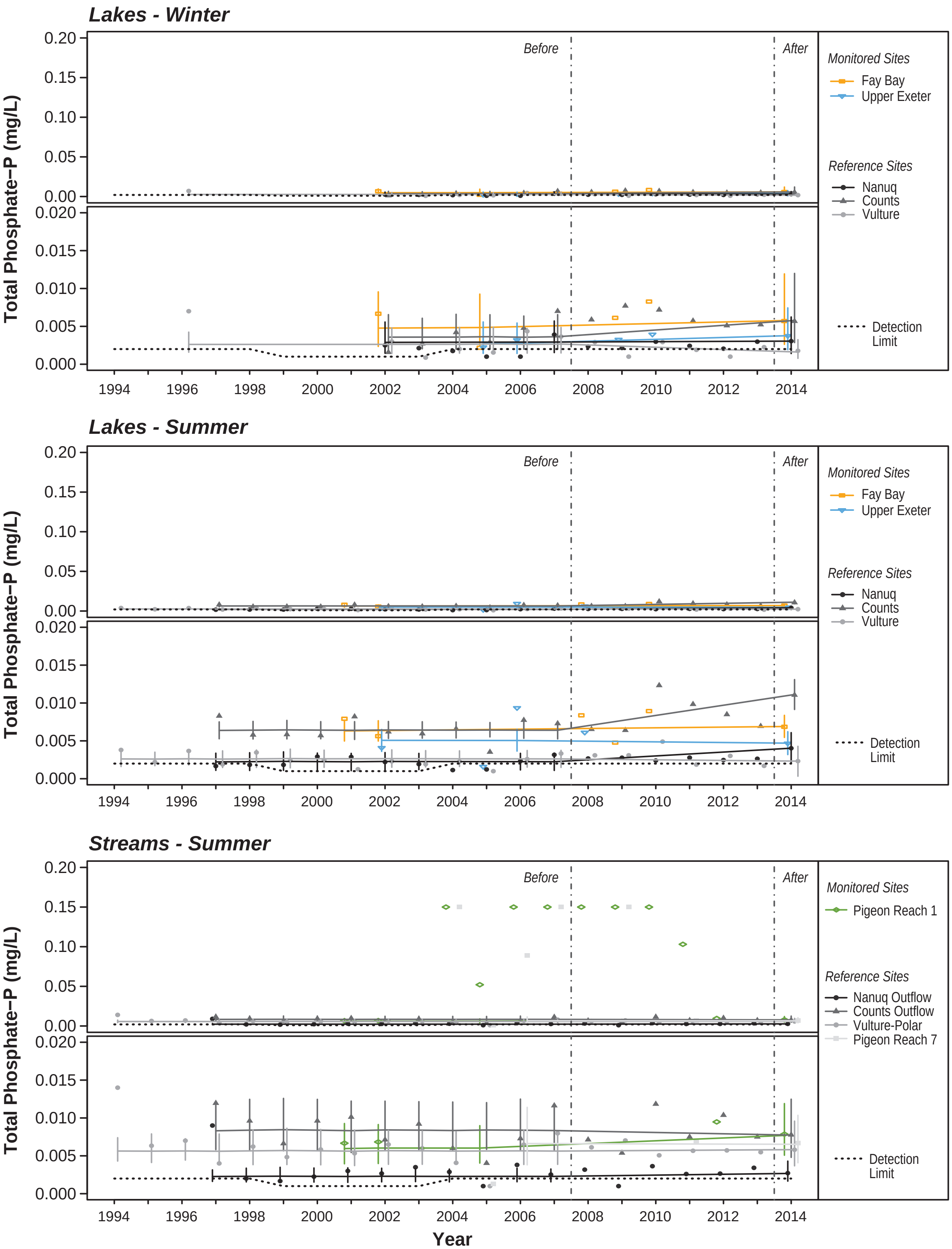
Observed mean total-phosphate P concentrations in 2014 were less than the benchmark concentrations for Fay Bay and Upper Exeter Lake (see Section 2.3) during the ice-covered and open water seasons (see Part 2 – Data Report).

5.2.4.12 *TOC*

Summary: Statistical and graphical analyses suggest that TOC concentrations have increased in Fay Bay. However, the source of the observed increase is unclear at this time and may reflect naturally variability in TOC concentrations in Fay Bay.

Figure 5.2-11

Observed and Fitted Means for Total Phosphate-P Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: *Symbols represent observed mean values; solid lines represent fitted curves.*
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.

Statically significant changes in TOC concentrations were detected between monitored and reference lakes when comparing the before and after periods during the ice-covered season (Table 5.2-16). Graphical analysis shows that although TOC concentrations in Fay Bay and Upper Exeter Lake are naturally elevated, relative to reference sites, TOC concentrations in 2014 have also increased in Fay Bay during the ice-covered season, when compared to the before period (Figure 5-2-12). Although statistical analyses were not possible for Fay Bay during the open water season, graphical analysis suggests that TOC concentrations have also increased during this season (Table 5.2-16; Figure 5.2-12). In contrast, graphical analysis indicates that TOC concentrations in Upper Exeter Lake have remained similar or decreased in 2014, when compared to the before period (Figure 5.2-12).

Table 5.2-16. Statistical Results of TOC in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	-	$p<0.0001$	Nanuq, Counts, Vulture	3-234
	Upper Exeter	-	$p=0.01$	Nanuq, Counts	
Summer	Fay Bay	Fay Bay	-	-	3-243
	Upper Exeter	-	$p=0.64$	-	
Summer	Pigeon Reach 1	-	$p=0.002$	Nanuq Outflow, Counts Outflow, Vulture-Polar	3-250

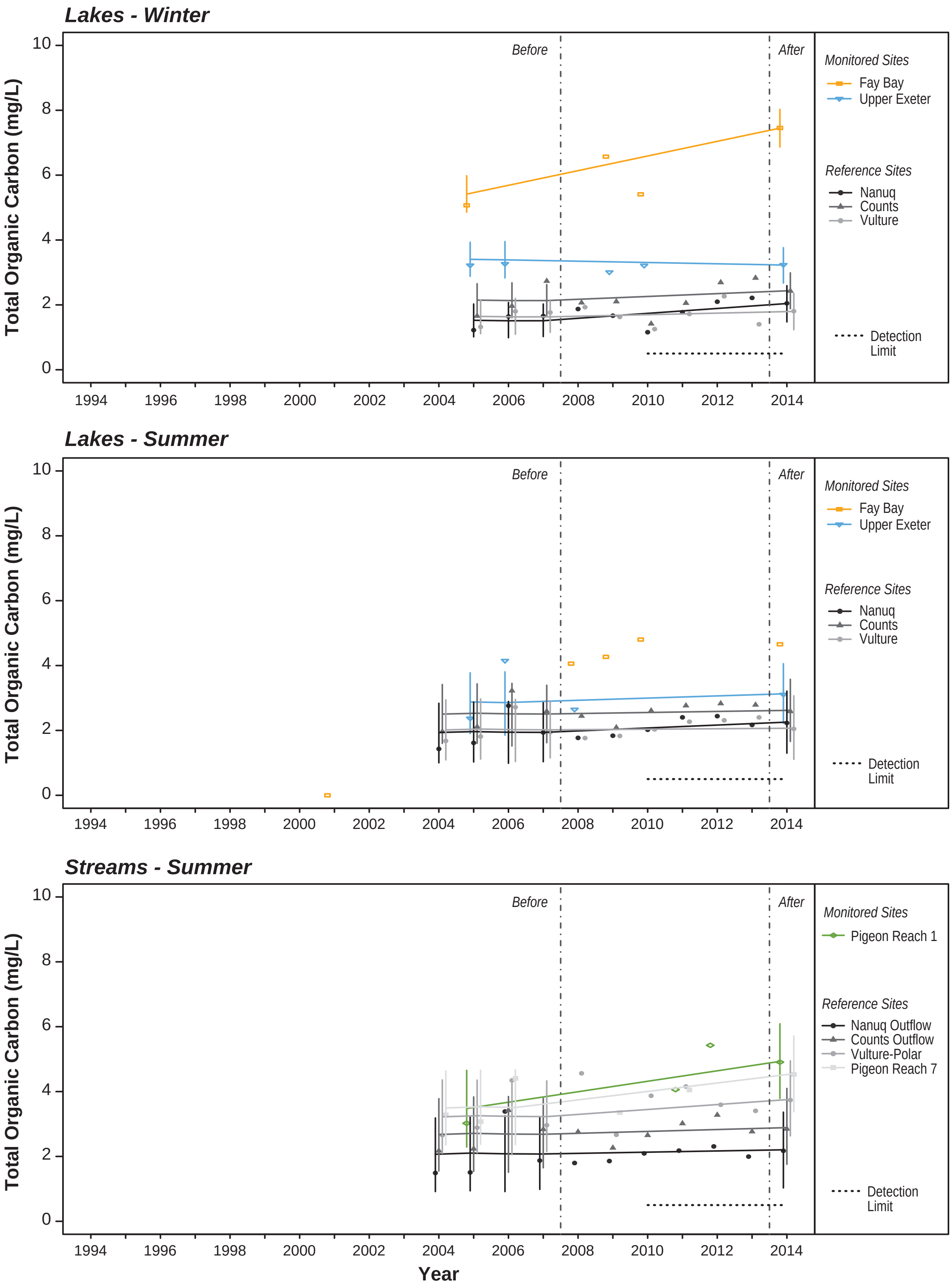
Note: Dashes indicate not applicable.

Statistically significant changes in TOC concentrations were detected between Pigeon Reach 1 and reference streams when comparing the before and after periods (Table 5.2-16). The results from the BACI contrasts indicate that the trend in Pigeon Reach 1 differed from all reference stream sites, with the exception of Pigeon Reach 7 (Table 5.2-16). Graphical analysis also indicates that 2014 TOC concentrations increased at both sites within the Pigeon Stream, when compared to the before period (Figure 5.2-12). Pigeon Reach 7 is an internal reference site for the Pigeon-Fay and Upper Exeter Watershed. The similarity in the trend observed at both sites within the Pigeon Stream suggests that TOC concentrations are naturally elevated and increasing within the Pigeon-Fay and Upper Exeter Watershed.

Together, the results indicate that TOC concentrations are naturally elevated in the Pigeon-Fay and Upper Exeter Watershed, but that TOC concentrations have increased, relative to reference sites, in Fay Bay, particularly during the ice-covered season. TOC concentrations did not increase in Fay Bay and Upper Exeter Lake following the unplanned release of FPK in May of 2008 (Figure 5.2-12; Rescan 2011b). Furthermore, effects from the connection of the PSD to the natural Pigeon Stream would not have been possible during the winter of 2014. Thus, the source of the increase observed in Fay Bay during the ice-covered season is unclear at this time and may reflect naturally variability in TOC concentrations.

Figure 5.2-12

Observed and Fitted Means for Total Organic Carbon Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: Symbols represent observed mean values; solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.

5.2.4.13 Total Antimony

Summary: Total antimony concentrations have generally been below detection limits in all monitored lakes and streams, especially during the after period. All concentrations were below the benchmark in 2014. No mine effects were detected.

Statistical and graphical analyses indicate that total antimony concentrations have generally been below detection limits, especially during the after period, in all monitored and reference lakes and streams (Table 5.2-17; Figure 5.2-13). Observed mean concentrations were less than the water quality benchmark of 0.02 mg/L (see Part 2 - Data Report; Fletcher et al. 1996) in all lakes and streams in 2014. Thus, no mine effects were detected.

Table 5.2-17. Statistical Results of Total Antimony in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	ALL	-	-	3-257
	Upper Exeter	ALL	-	-	
Summer	Fay Bay	ALL	-	-	3-260
	Upper Exeter	ALL	-	-	
Summer	Pigeon Reach 1	ALL	-	-	3-263

Note: Dashes indicate not applicable.

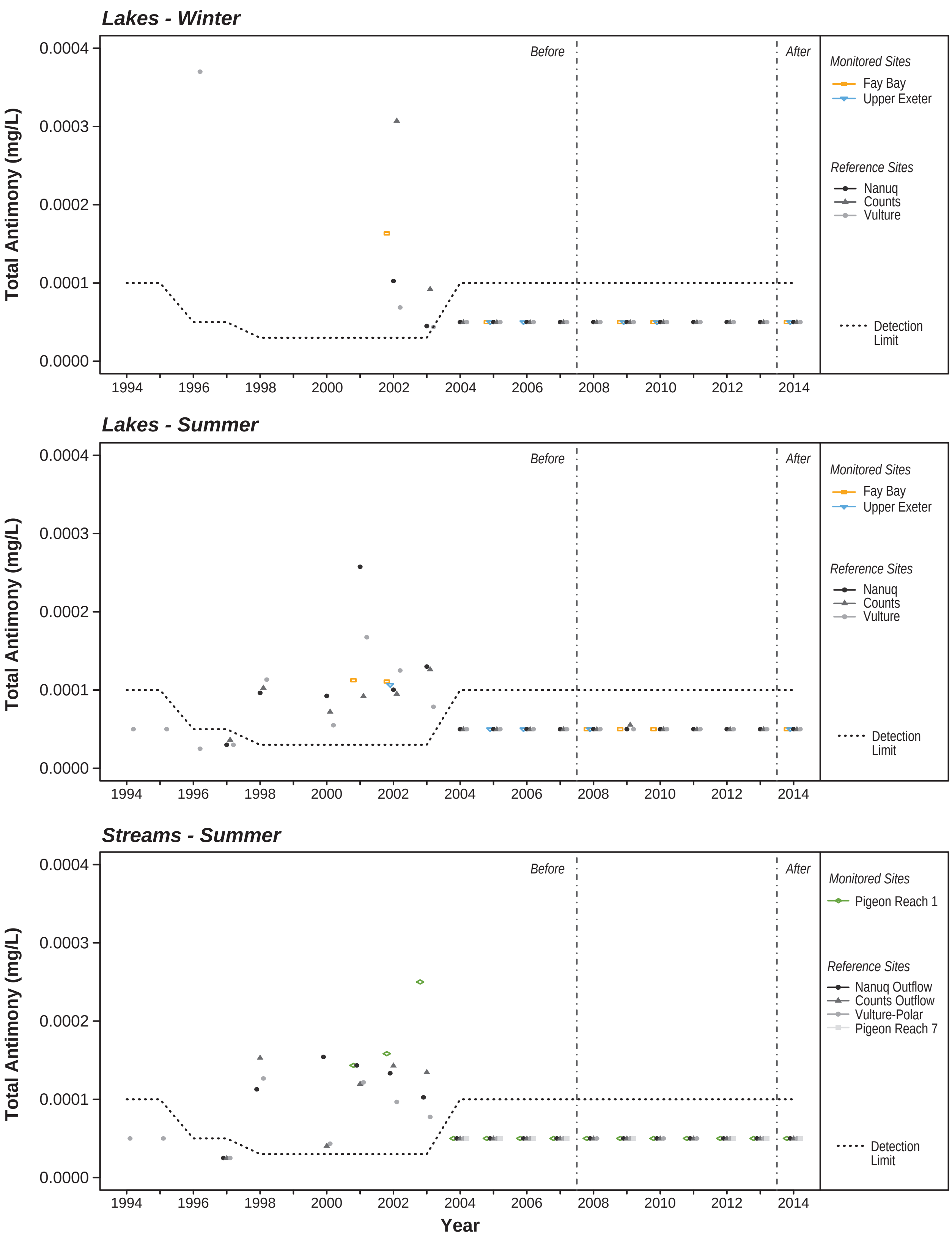
5.2.4.14 Total Arsenic

Summary: Together, statistical and graphical analyses suggest that total arsenic concentrations have not increased in the Pigeon-Fay and Upper Exeter Watershed as a result of mine operations. Observed and fitted concentrations were less than the CCME guideline value at all monitored sites in 2014.

Statistically significant changes in total arsenic concentrations were detected between all monitored and reference lakes when comparing the before and after periods, with the exception of Upper Exeter Lake during the open water season (Table 5.2-18). However, results from the BACI contrasts indicate that trends in monitored lakes were similar to those observed in at least two reference lakes during both the ice-covered and open water seasons (Table 5.2-18). Furthermore, graphical analysis shows that although arsenic concentrations were elevated in Fay Bay, relative to reference lakes, they remained similar between the before and after periods (Figure 5.2-14). Graphical analysis also indicates that concentrations in Upper Exeter Lake have remained similar in the before and after periods (Figure 5.2-14).

Figure 5.2-13

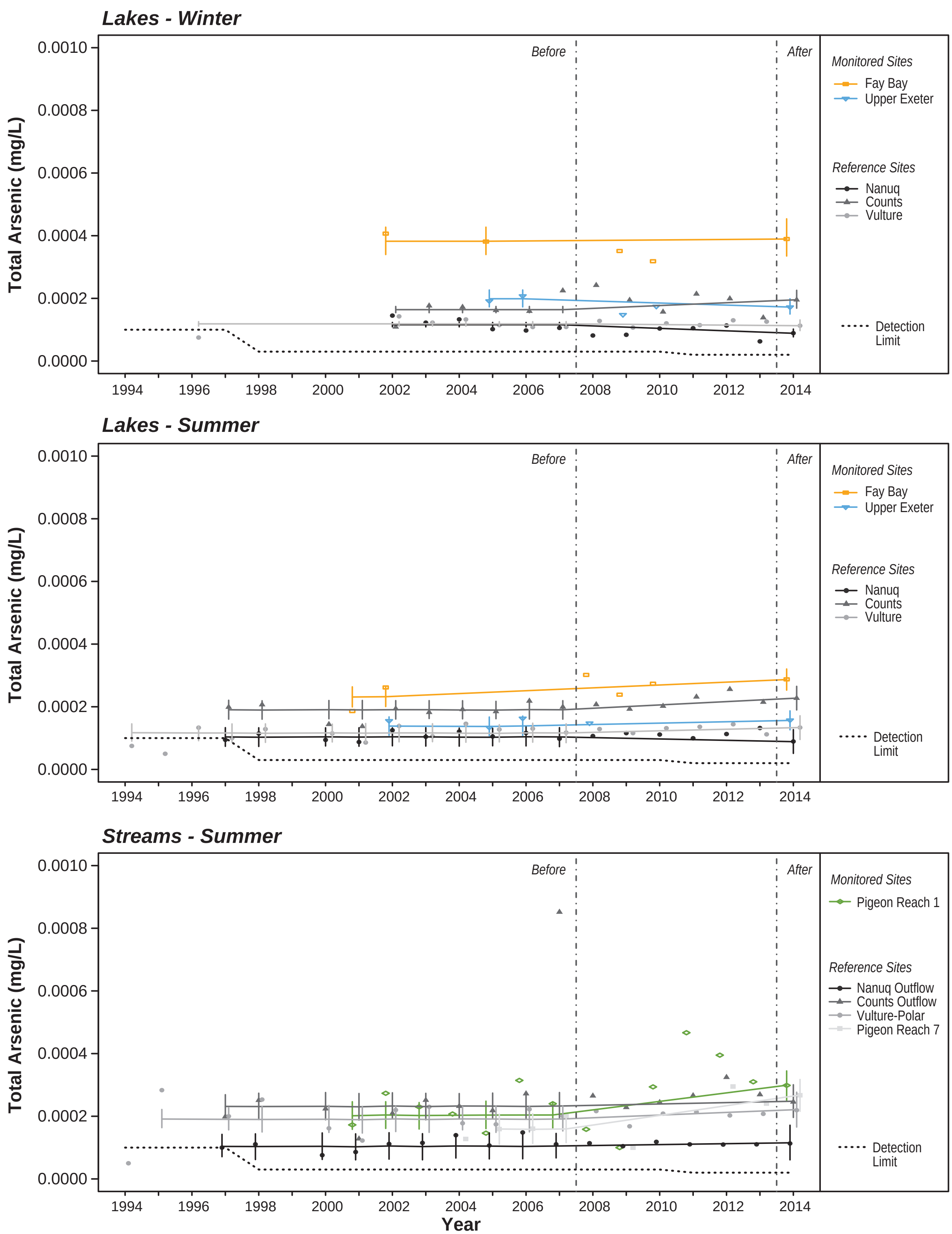
Observed and Fitted Means for Total Antimony Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: *Symbols represent observed mean values; solid lines represent fitted curves.*
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
Water quality benchmark (Fletcher et al. 1996) = 0.02 mg/L.

Figure 5.2-14

Observed and Fitted Means for Total Arsenic Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: Symbols represent observed mean values; solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
CCME Guideline = 0.005 mg/L.

Table 5.2-18. Statistical Results of Total Arsenic in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results: BACI Interaction Term	BACI Results: Significant BACI Contrasts	Statistical Report Page No.
Winter	Fay Bay	-	$p = 0.02$	None	3-266
	Upper Exeter	-	$p = 0.01$	Counts	
Summer	Fay Bay	-	$p = 0.01$	Nanuq	3-275
	Upper Exeter	-	$p = 0.07$	-	
Summer	Pigeon Reach 1	-	$p = 0.0017$	Nanuq Outflow, Counts Outflow, Vulture-Polar	3-284

Note: Dashes indicate not applicable.

Statistically significant changes in total arsenic concentrations were detected between Pigeon Reach 1 and reference streams when comparing the before and after periods (Table 5.2-18). However, results from the BACI contrasts indicate that the trend in Pigeon Reach 1 differed from all reference stream sites, with the exception of Pigeon Reach 7 (Table 5.2-18). Graphical analysis also indicates that total arsenic concentrations in 2014 increased at both sites within the Pigeon Stream, when compared to the before period (Figure 5.2-14). Pigeon Reach 7 is an internal reference site for the Pigeon-Fay and Upper Exeter Watershed. The similarity in the trend observed at both sites within the Pigeon Stream suggests that arsenic concentrations are naturally elevated within the watershed and is consistent with elevated concentrations observed in Fay Bay. Thus, no mine effects were detected with respect to total arsenic concentrations in the Pigeon-Fay and Upper Exeter Watershed.

Observed mean concentrations in 2014 were less than the arsenic CCME guideline value of 0.005 mg/L at all monitored sites (see Part 2 - Data Report; CCME 1999c).

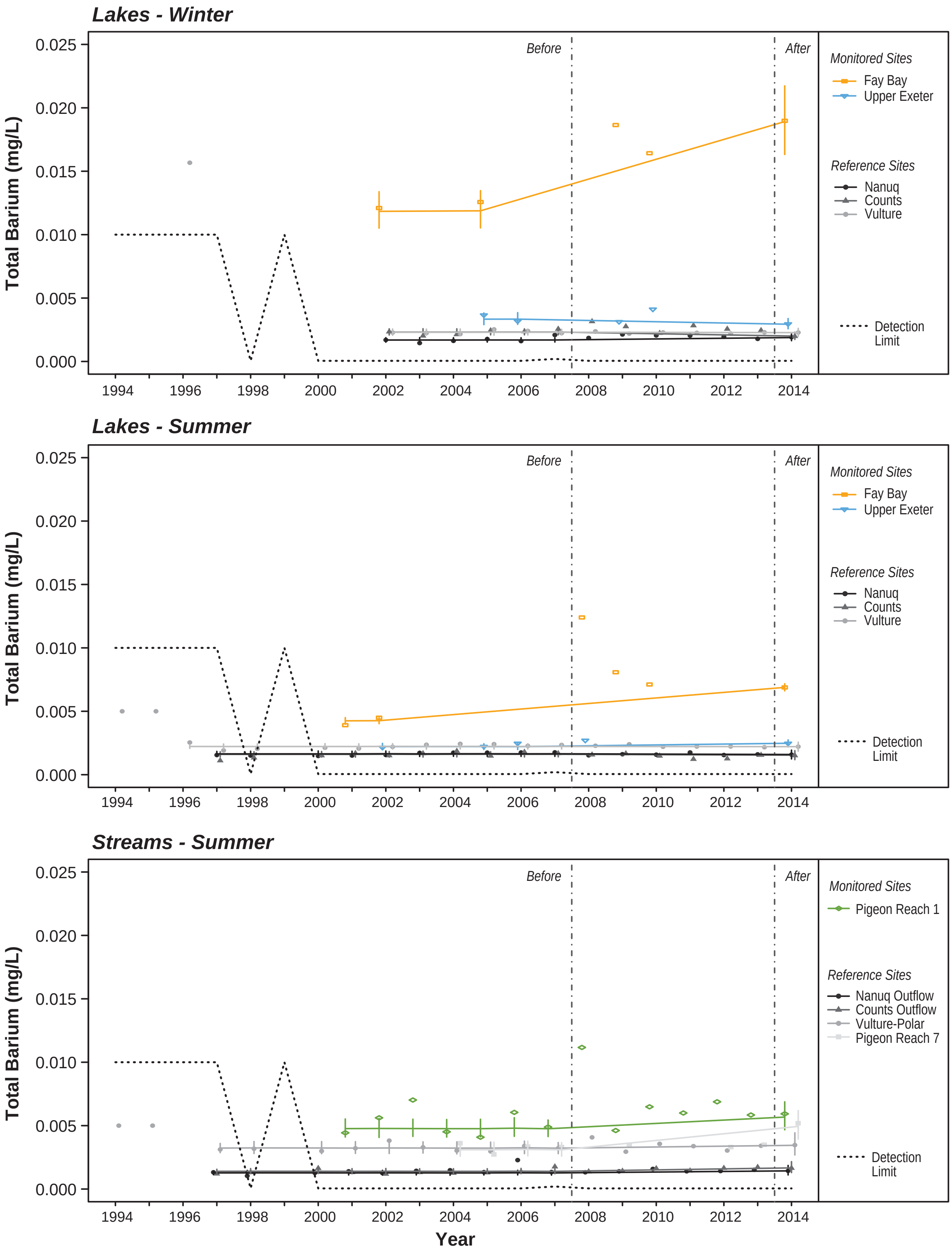
5.2.4.15 Total Barium

Summary: Statistical and graphical analyses suggest that total barium concentrations have increased in Fay Bay. These changes may be related to the unplanned release of FPK in May of 2008. Observed concentrations were less than the benchmark at all sites in 2014.

Statistically significant changes in total barium concentrations were detected between all monitored and reference lakes when comparing the before and after periods, with the exception of Upper Exeter Lake during the open water season (Table 5.2-19). However, results from the BACI contrasts indicate that the trend in Upper Exeter Lake was similar to those observed in at least two reference lakes during the ice-covered season (Table 5.2-19). Graphical analysis shows that total barium concentrations increased in Fay Bay in 2014 when compared to the before period (Figure 5.2-15). Graphical analysis also indicates that barium concentrations in Upper Exeter Lake have remained similar during the before and after periods (Figure 5.2-15).

Figure 5.2-15

Observed and Fitted Means for Total Barium Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: *Symbols represent observed mean values; solid lines represent fitted curves.*
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
Water quality benchmark (Haywood and Drinnan 1983) = 1 mg/L.

Table 5.2-19. Statistical Results of Total Barium in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	-	$p < 0.0001$	Nanuq, Counts, Vulture	3-291
	Upper Exeter	-	$p = 0.01$	Nanuq	
Summer	Fay Bay	-	$p < 0.0001$	Nanuq, Counts, Vulture	3-300
	Upper Exeter	-	$p = 0.26$	-	
Summer	Pigeon Reach 1	-	$p = 0.30$	-	3-309

Note: Dashes indicate not applicable.

No statistically significant changes in total barium concentrations were detected between Pigeon Reach 1 and reference streams when comparing the before and after periods (Table 5.2-19). Graphical analysis indicates that total barium concentrations in Pigeon Reach 1 remained similar in the before and after periods (Figure 5.2-15).

Together, results indicate that total barium concentrations have increased in Fay Bay during the ice-covered and open water seasons. The source of the increase is unclear at this time, but may be related to the unplanned release of FPK in May of 2008 (Rescan 2011b). Graphical analysis shows that concentrations were elevated, but decreasing, in Fay Bay from 2008 to 2010 during both the ice-covered and open water seasons (Figure 5.2-15). Increases are unlikely related to the PSD since increases were observed during the ice-covered season, prior to the connection of the PSD to the natural Pigeon Stream.

Observed mean concentrations at all monitored sites in 2014 were below the barium water quality benchmark of 1 mg/L (see Part 2 - Data Report; Haywood and Drinnan 1983).

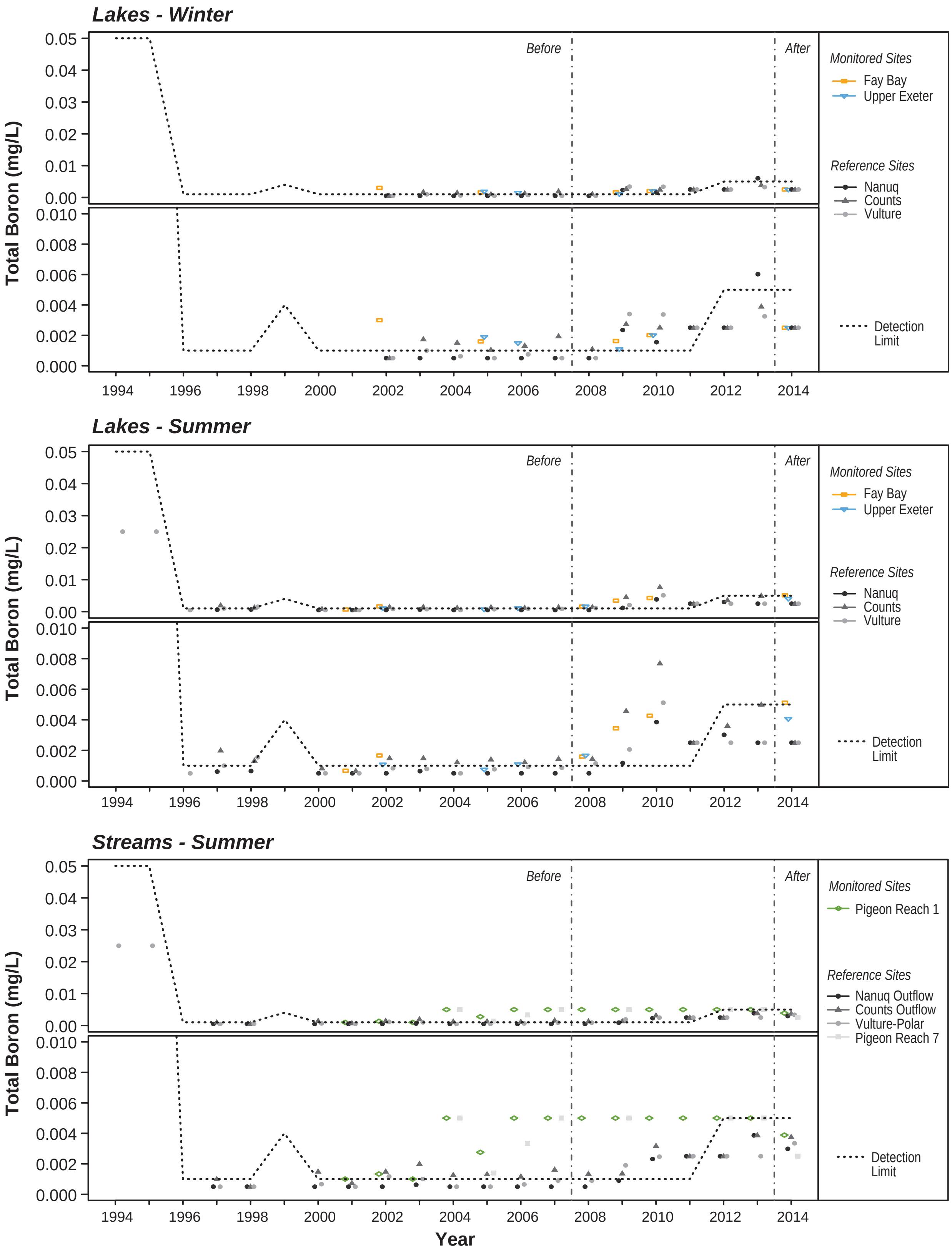
5.2.4.16 Total Boron

> ***Summary:*** Total boron concentrations have generally been below detection limits in all monitored lakes and streams since monitoring began. All concentrations were below the benchmark in 2014. No mine effects were detected.

Statistical and graphical analyses indicate that total boron concentrations have generally been below detection limits in all monitored and reference lakes and streams since monitoring began (Table 5.2-20; Figure 5.2-16). Observed mean concentrations were less than the boron CCME guideline of 1.5 mg/L (see Part 2 - Data Report; CCME 2009) in all lakes and streams in 2014. Thus, no mine effects were detected.

Figure 5.2-16

Observed and Fitted Means for Total Boron Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: Symbols represent observed mean values; solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
CCME Guideline = 1.5 mg/L.

Table 5.2-20. Statistical Results of Total Boron in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	ALL	-	-	3-316
	Upper Exeter	ALL	-	-	
Summer	Fay Bay	ALL	-	-	3-319
	Upper Exeter	ALL	-	-	
Summer	Pigeon Reach 1	ALL	-	-	3-322

Note: Dashes indicate not applicable.

5.2.4.17 Total Cadmium

> ***Summary:*** Total cadmium concentrations have generally been below detection limits in all monitored lakes and streams since monitoring began. All concentrations were below the CCME guideline in 2014. No mine effects were detected.

Statistical and graphical analyses indicate that total cadmium concentrations have generally been below detection limits in all monitored and reference sites since monitoring began (Table 5.2-21; Figure 5.2-17). The detection limit for total cadmium was less than the hardness-dependent CCME guideline at all sites in 2014 (see Part 2 - Data Report; CCME 2014a). Thus, no mine effects detected.

Table 5.2-21. Statistical Results of Total Cadmium in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	ALL	-	-	3-325
	Upper Exeter	ALL	-	-	
Summer	Fay Bay	ALL	-	-	3-328
	Upper Exeter	ALL	-	-	
Summer	Pigeon Reach 1	ALL	-	-	3-331

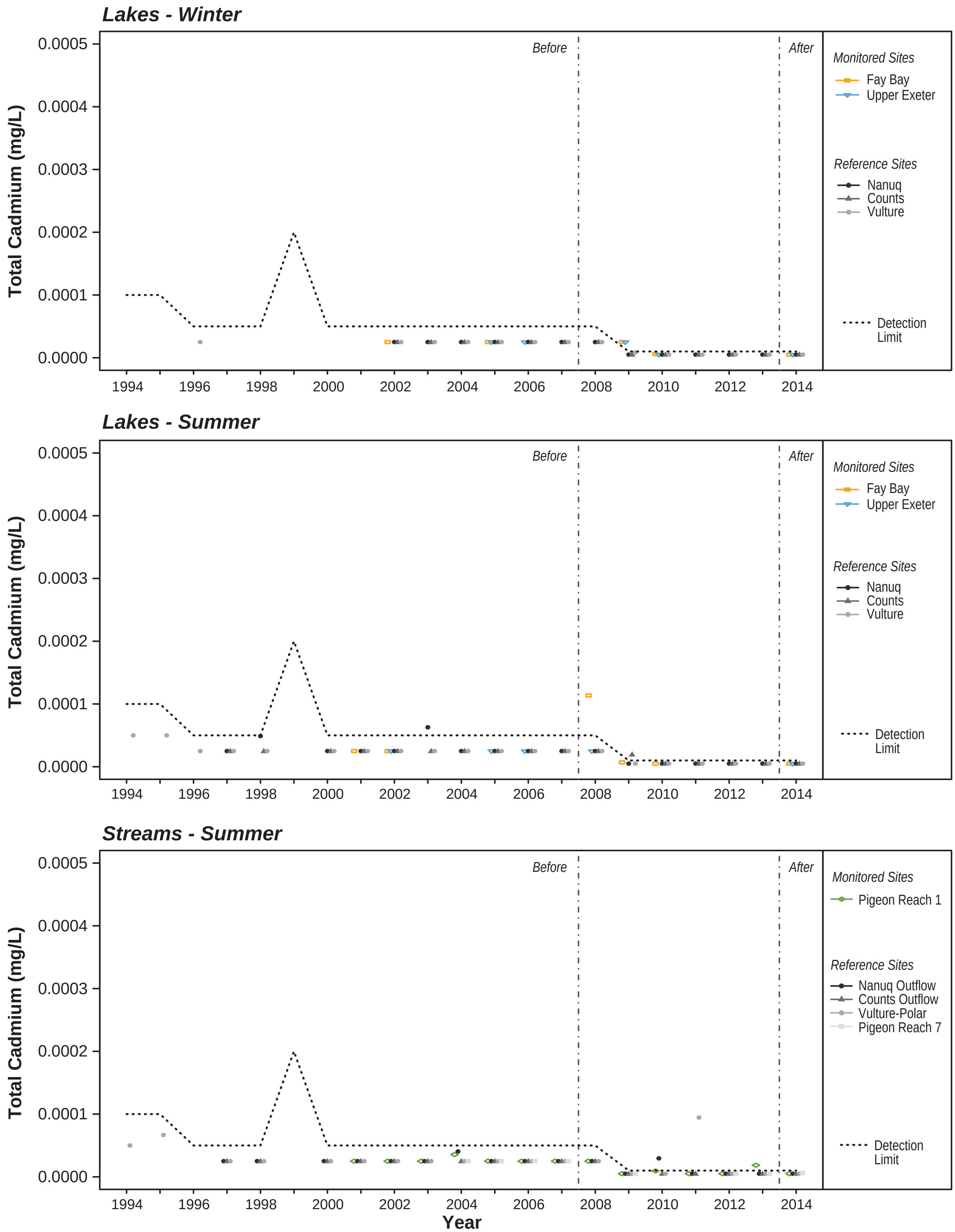
Note: Dashes indicate not applicable.

5.2.4.18 Total Molybdenum

> ***Summary:*** Total molybdenum concentrations have generally been below detection limits in all monitored lakes and streams since monitoring began. Observed concentrations were less than the molybdenum SSWQO at all monitored sites in 2014. No mine effects were detected.

Figure 5.2-17

Observed and Fitted Means for Total Cadmium Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: *Symbols represent observed mean values; solid lines represent fitted curves.*
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
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 = 0.00037 mg/L where hardness > 280 mg/L.

Statistical and graphical analyses indicate that total molybdenum concentrations have generally been below detection limits in all monitored and reference lakes and streams since monitoring began (Table 5.2-22; Figure 5.2-18). Graphical analysis indicates that total molybdenum has, at times, been detected in Fay Bay between 2008 and 2014 (Figure 5.2-18). The detection of molybdenum in 2014 likely represents natural occurrences as molybdenum was also detected at both Pigeon Stream sites in 2014. Observed mean concentrations were less than the molybdenum SSWQO (19.38 mg/L) in all monitored lakes and streams in 2014 (see Part 2 - Data Report; Rescan 2012a). Thus, no mine effects were detected.

Table 5.2-22. Statistical Results of Total Molybdenum in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	ALL	-	-	3-334
	Upper Exeter	ALL	-	-	
Summer	Fay Bay	ALL	-	-	3-337
	Upper Exeter	ALL	-	-	
Summer	Pigeon Reach 1	ALL	-	-	3-340

Note: Dashes indicate not applicable.

5.2.4.19 Total Nickel

Summary: Statistical and graphical analyses suggest that total nickel concentrations have increased in Fay Bay. These changes may be related to the unplanned release of FPK in 2008. Observed concentrations were less than the hardness-dependent nickel CCREM guideline at all sites in 2014.

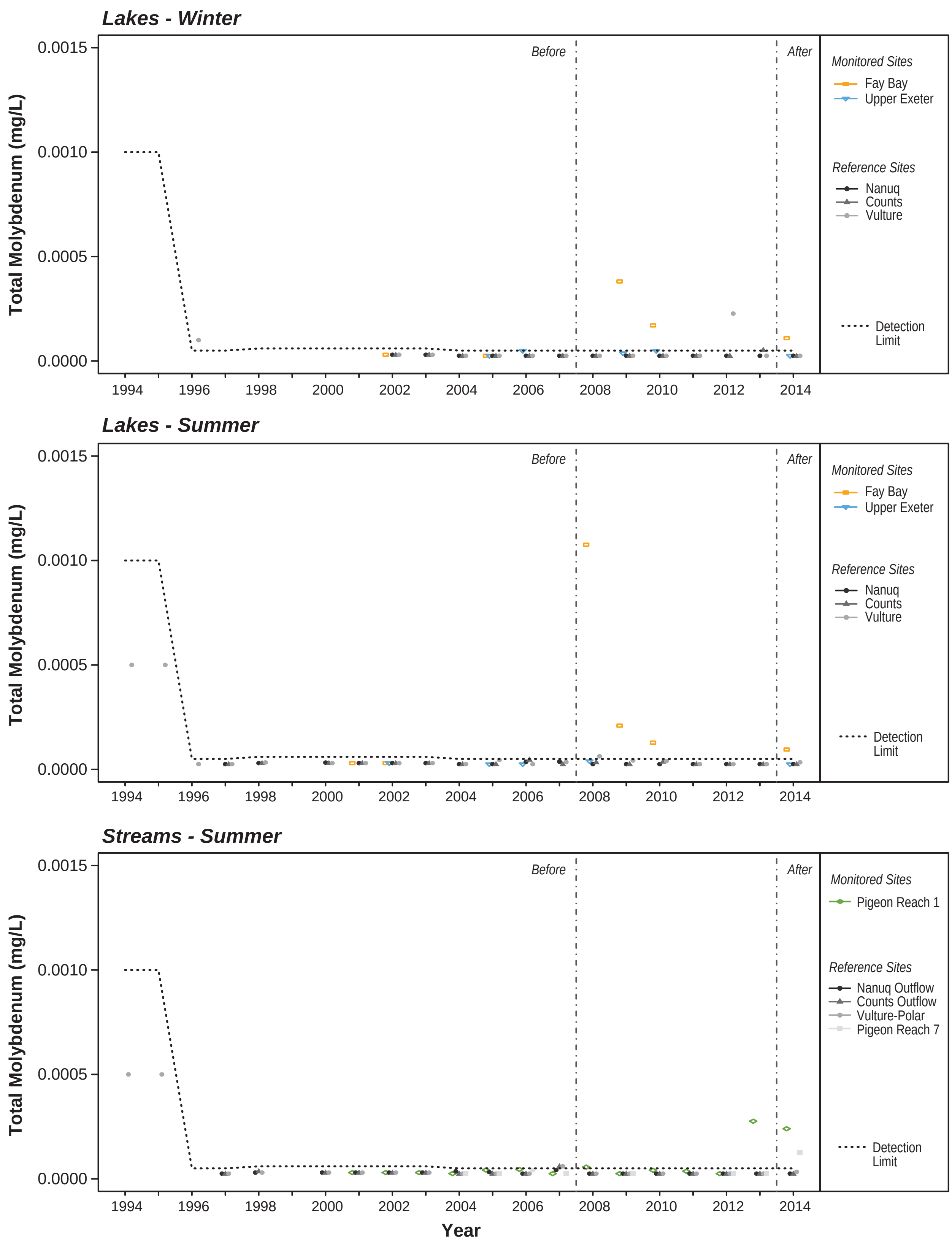
Statistically significant changes in total nickel concentrations were detected between Fay Bay and reference lakes when comparing the before and after periods (Table 5.2-23). Graphical analysis shows that total nickel concentrations increased in Fay Bay in 2014 when compared to the before period, and that the trend was more pronounced during the ice-covered season (Figure 5.2-19). In contrast, graphical analysis indicates that nickel concentrations in Upper Exeter Lake have remained similar during the before and after periods (Figure 5.2-19).

No statistically significant changes in total nickel concentrations were detected between Pigeon Reach 1 and reference streams when comparing the before and after periods (Table 5.2-23). Graphical analysis indicates that total nickel concentrations in Pigeon Reach 1 remained similar in the before and after periods (Figure 5.2-19).

The increase in nickel concentrations observed in Fay Bay, particularly during the ice-covered season, may be related to the unplanned release of FPK in May of 2008 (Rescan 2011b). Nickel concentrations increased in Fay Bay following the event, but have been decreasing towards baseline concentrations since 2008. Furthermore, effects from the connection of the PSD to the natural Pigeon Stream would not have been possible during the winter of 2014.

Figure 5.2-18

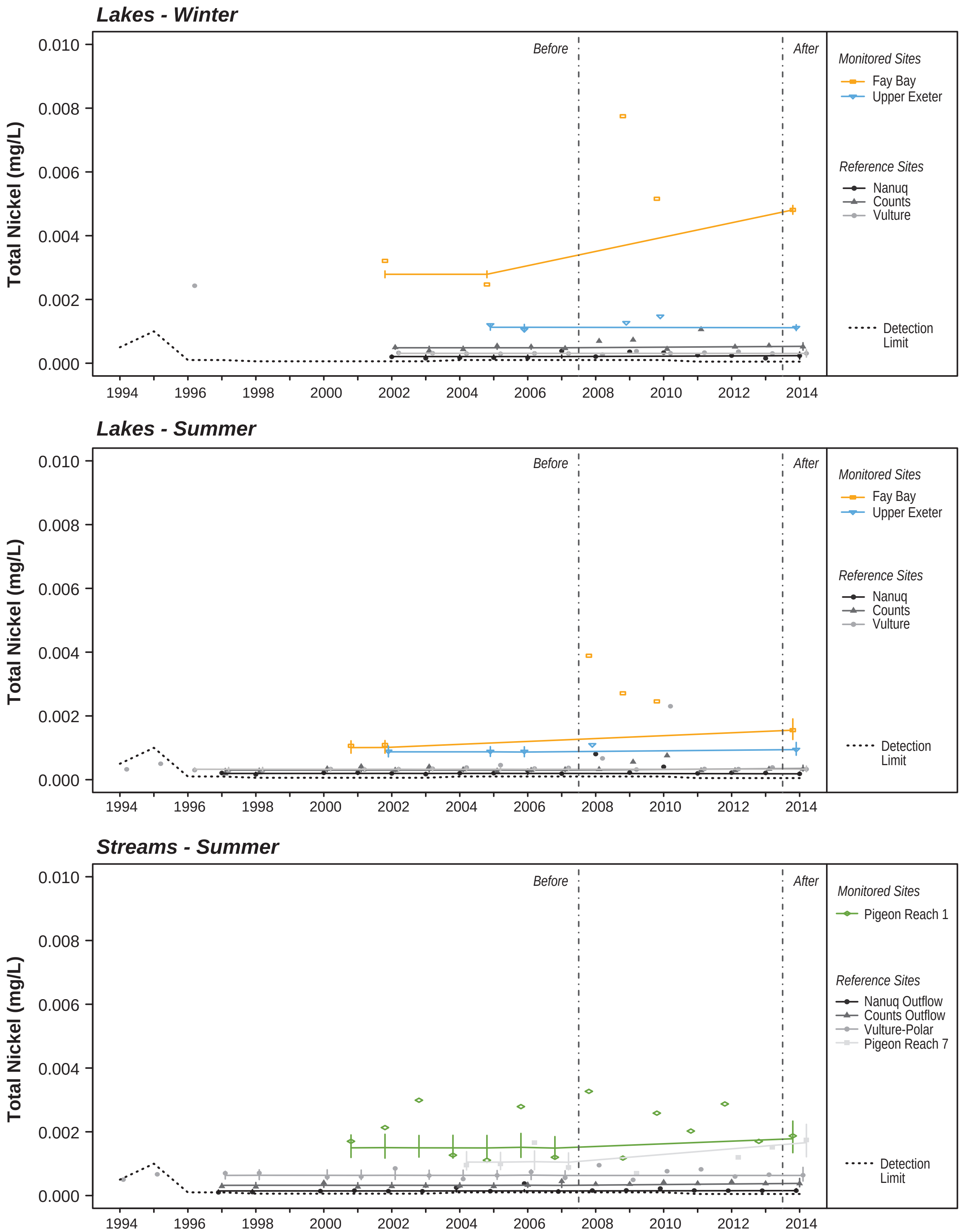
Observed and Fitted Means for Total Molybdenum Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: Symbols represent observed mean values; solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
SSWQO = 19.38 mg/L.

Figure 5.2-19

Observed and Fitted Means for Total Nickel Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: *Symbols represent observed mean values; solid lines represent fitted curves.*
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
CCME Guideline = $e^{0.76 \times (\ln Hardness) + 1.06}$*/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.

Table 5.2-23. Statistical Results of Total Nickel in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	-	$p < 0.0001$	Nanuq, Counts, Vulture	3-343
	Upper Exeter	-	$p = 0.93$	-	
Summer	Fay Bay	-	$p = 0.002$	Nanuq, Counts, Vulture	3-352
	Upper Exeter	-	$p = 0.61$	-	
Summer	Pigeon Reach 1	-	$p = 0.30$	-	3-361

Note: Dashes indicate not applicable.

Observed 2014 mean concentrations were less than the hardness-dependent nickel CCREM guideline value at all monitored sites (see Part 2 - Data Report; CCREM 1987).

5.2.4.20 Total Selenium

Summary: Total selenium concentrations have generally been below detection limits in all monitored lakes and streams since monitoring began. All concentrations were below the CCREM guideline in 2014. No mine effects were detected.

Statistical and graphical analyses indicate that total selenium concentrations have generally been below detection limits in all monitored and reference lakes and streams since monitoring began (Table 5.2-24; Figure 5.2-20). Observed mean concentrations were less than the selenium CCREM guideline of 0.001 mg/L in all monitored and reference sites in 2014 (see Part 2 - Data Report; CCREM 1987). Thus, no mine effects were detected.

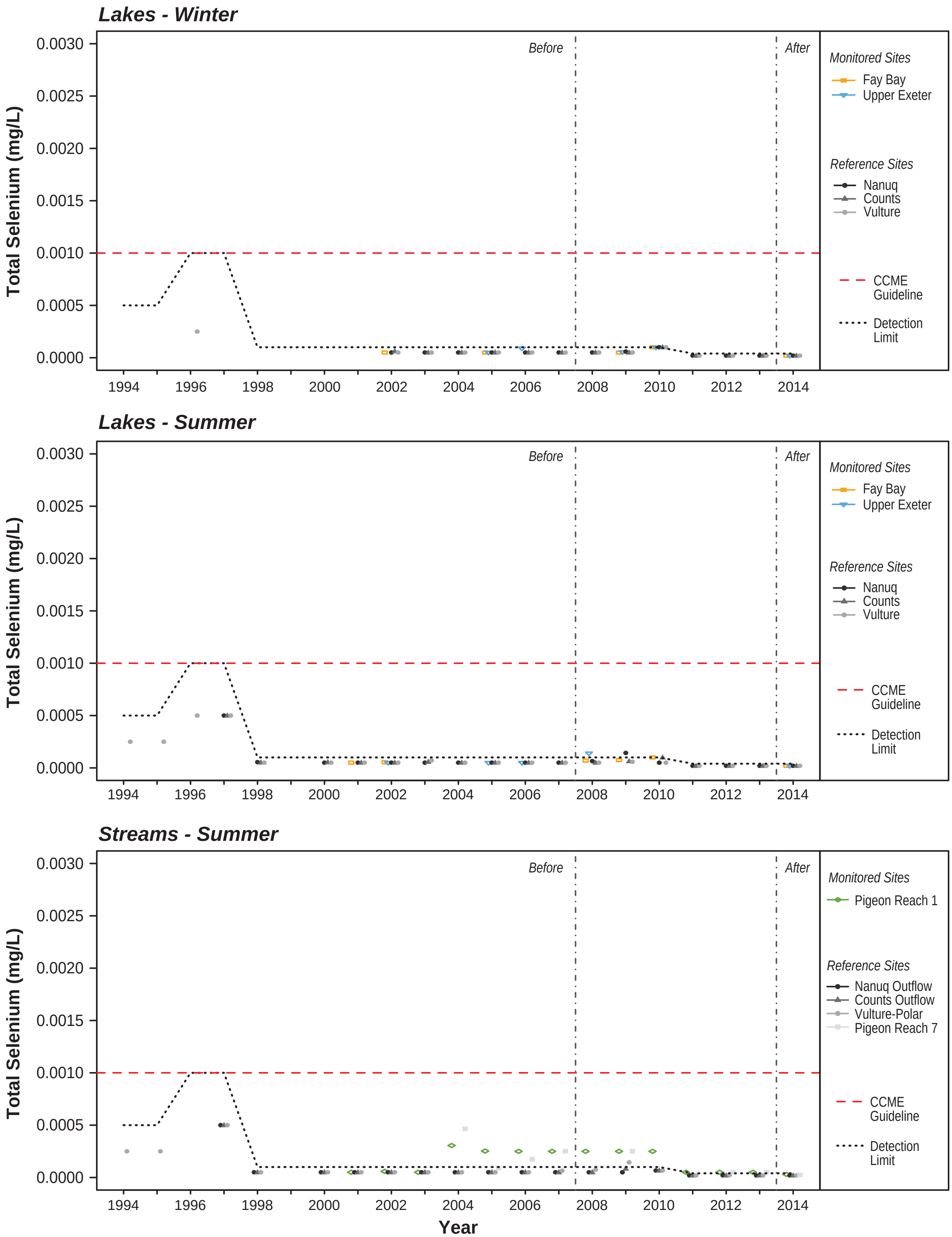
Table 5.2-24. Statistical Results of Total Selenium in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	ALL	-	-	3-368
	Upper Exeter	ALL	-	-	
Summer	Fay Bay	ALL	-	-	3-371
	Upper Exeter	ALL	-	-	
Summer	Pigeon Reach 1	ALL	-	-	3-374

Note: Dashes indicate not applicable.

Figure 5.2-20

Observed and Fitted Means for Total Selenium Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: *Symbols represent observed mean values; solid lines represent fitted curves.*
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
CCME Guideline = 0.001 mg/L.

5.2.4.21 Total Strontium

Summary: Statistical and graphical analyses suggest that total strontium concentrations have increased in Fay Bay. These changes may be related to the unplanned release of FPK in 2008. Observed and fitted concentrations were less than the benchmark value at all monitored sites in 2014.

Statistical and graphical results indicate that total strontium concentrations are naturally elevated in the monitored lakes, relative to the reference lakes, and have increased in Fay Bay, relative to reference lakes, when comparing the before and after periods (Table 5.2-25; Figure 5.2-21). Statistically significant increases were also detected in Upper Exeter Lake, but only during the open water season (Table 5.2-25). However, graphical analysis suggests that strontium concentrations were similar between the before and after periods during the open water season in Upper Exeter Lake (Figure 5.2-21).

Table 5.2-25. Statistical Results of Total Strontium in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	-	$p < 0.0001$	Nanuq, Counts, Vulture	3-377
	Upper Exeter	-	$p = 0.15$	-	
Summer	Fay Bay	-	$p < 0.0004$	Nanuq, Counts, Vulture	3-386
	Upper Exeter	-	$p < 0.0001$	Nanuq, Counts, Vulture	
Summer	Pigeon Reach 1	-	$p = 0.0001$	Nanuq Outflow, Vulture-Polar	3-395

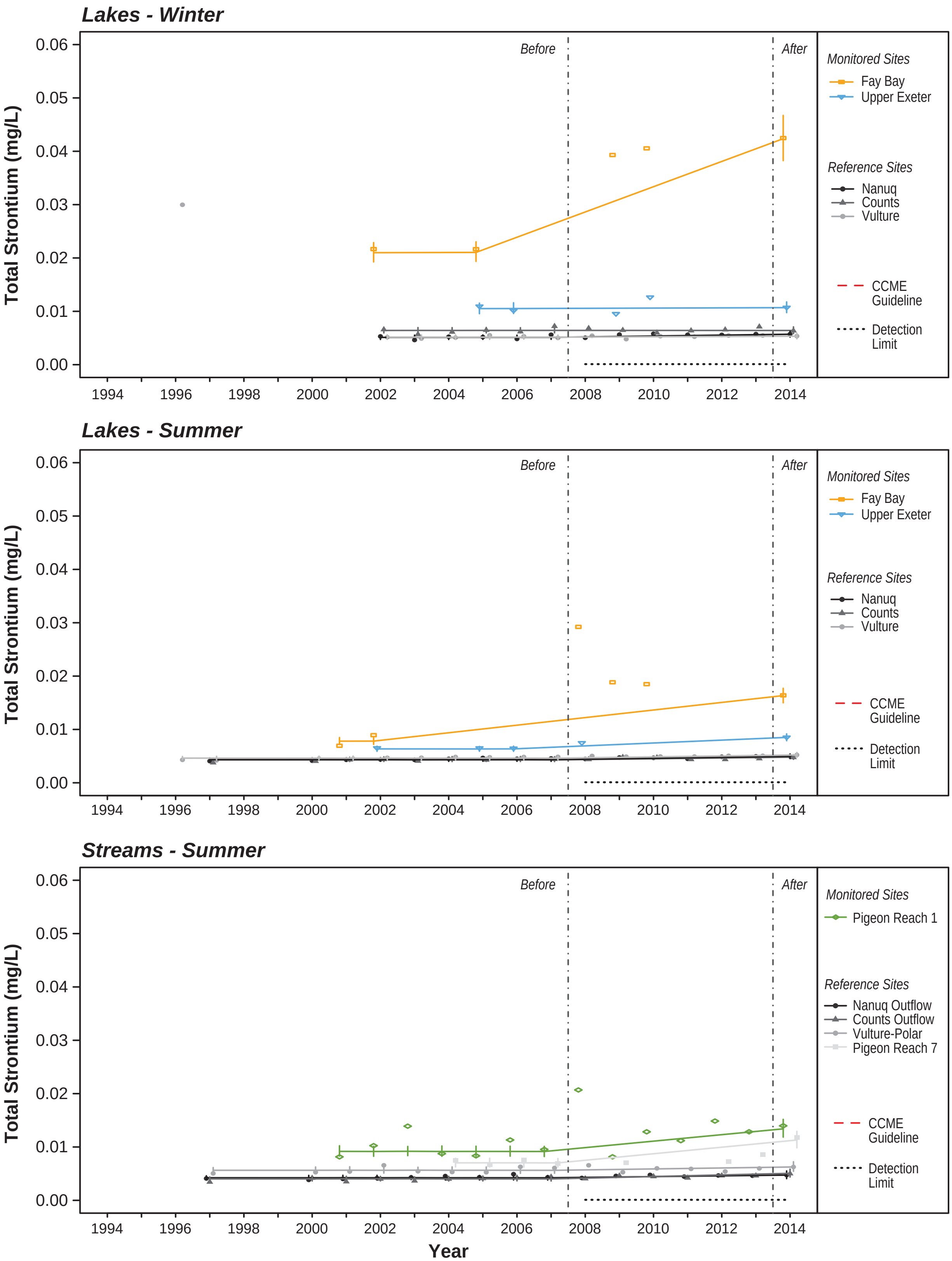
Note: Dashes indicate not applicable.

Statistically significant changes in total strontium concentrations were detected between Pigeon Reach 1 and reference streams when comparing the before and after periods (Table 5.2-25). However, results from the BACI contrasts indicate that the trend in Pigeon Reach 1 did not differ from two of the reference stream sites, including the internal Pigeon Reach 7 reference site (Table 5.2-25). Graphical analysis also indicates that total strontium concentrations increased at both sites within the Pigeon Stream in 2014, when compared to the before period (Figure 5.2-21). The similarity in trends observed at both sites within the Pigeon Stream suggests that strontium concentrations may be naturally elevated and increasing within the Pigeon-Fay and Upper Exeter Watershed.

Together, results indicate that strontium concentrations are naturally elevated in the Pigeon-Fay and Upper Exeter Watershed, but that concentrations have increased, relative to reference sites in Fay Bay, particularly during the ice-covered season. The observed increase in concentrations in Fay Bay may be related to the unplanned release of FPK in May of 2008 (Rescan 2011b). Increases are unlikely related to the PSD since increases were observed during the ice-covered season, prior to the connection of the PSD to the natural Pigeon Stream.

Figure 5.2-21

Observed and Fitted Means for Total Strontium Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: Symbols represent observed mean values; Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
Water quality benchmark (Golder 2011) = 6.242 mg/L.

Observed mean concentrations in all lakes and streams in 2014 were less than the strontium water quality benchmark of 6.242 mg/L at all monitored sites (see Part 2 - Data Report; Golder 2011).

5.2.4.22 Total Uranium

Summary: Together, statistical and graphical analyses suggest that total uranium concentrations have not increased in the Pigeon-Fay and Upper Exeter Watershed as a result of mine operations. Observed and fitted concentrations were less than the CCME guideline value at all monitored sites in 2014.

Statistical and graphical analyses indicate that total uranium concentrations during the ice-covered season increased in Fay Bay, relative to reference lakes, when comparing the before and after periods (Table 5.2-26; Figure 5.2-22). Most observations during the open water season were at or below analytical detection limits in lakes during the before period, thus no statistical analyses were possible (Table 5.2-26). Owing to elevated detection limits during the before period in the ice-covered and open water season, it is not possible to clearly determine whether uranium concentrations have changed in 2014 (Figure 5.2-22). Also, graphical analysis indicates that elevated concentrations observed in Fay Bay are within the range of concentrations observed in the Pigeon Stream during the before and after periods (Figure 5.2-22).

Table 5.2-26. Statistical Results of Total Uranium in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	Counts	$p < 0.0001$	Nanuq, Vulture	3-402
	Upper Exeter	Counts	$p = 0.67$	-	
Summer	Fay Bay	ALL	-	-	3-412
	Upper Exeter	ALL	-	-	
Summer	Pigeon Reach 1	-	$p = 0.03$	Nanuq Outflow, Vulture-Polar	3-415

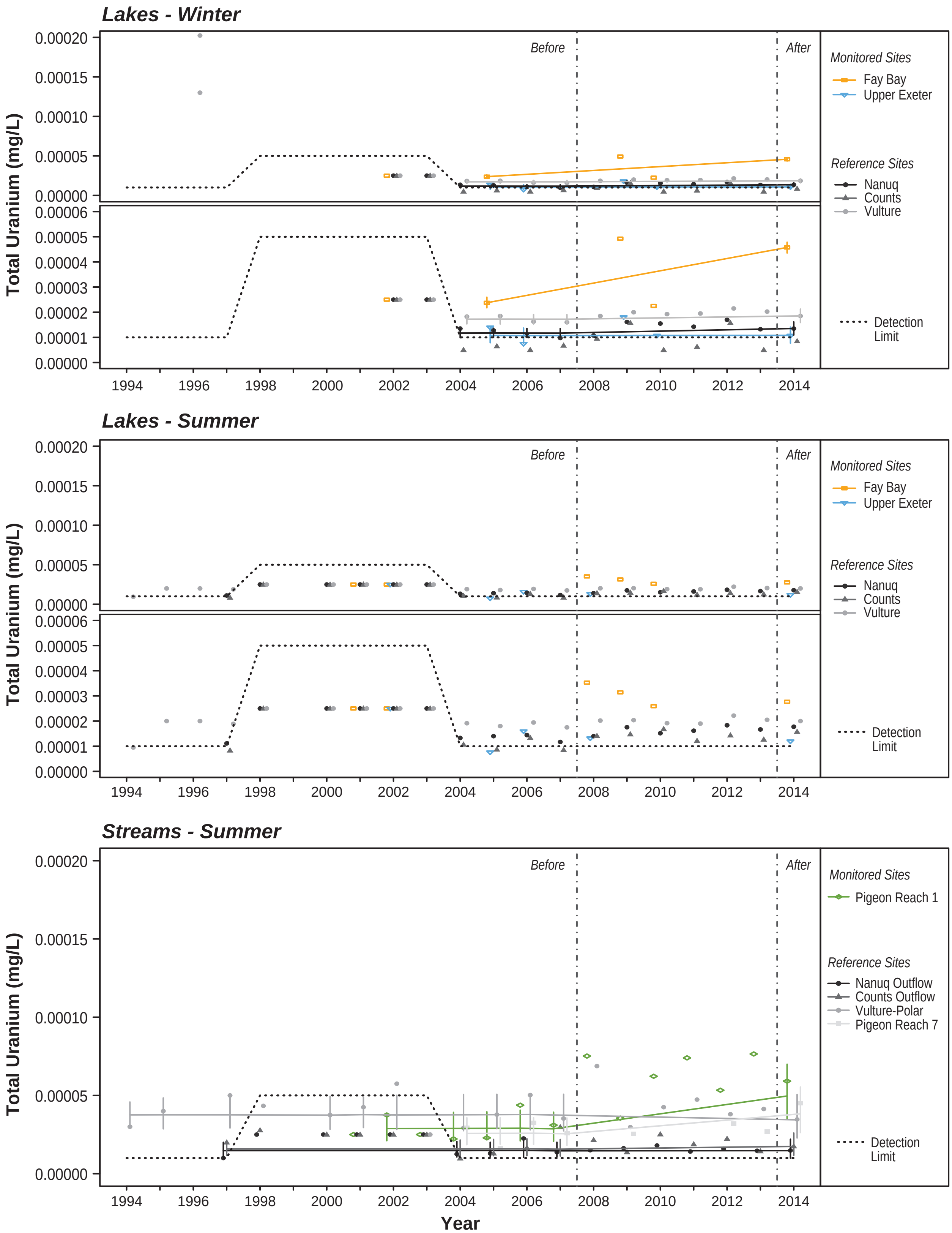
Note: Dashes indicate not applicable.

Statistical and graphical analyses indicate that total uranium concentrations appear to have increased at Pigeon Reach 1, relative to reference streams (Table 5.2-26; Figure 5.2-22). However, results also indicate that a similar increasing trend in total uranium was observed at the internal reference stream site (i.e., Pigeon Reach 7; Table 5.2-26; Figure 5.2-22). Graphical analysis also indicates that concentrations in Pigeon Reach 1 were elevated from 2008 to 2014. Since elevated concentrations of total uranium were observed in Pigeon Reach 1 prior to the opening of the PSD and an increasing trend was observed in Pigeon Reach 7, the observed increase in total uranium was unlikely related to mine activities.

Observed mean concentrations in all lakes and streams in 2014 were less than the CCME uranium guideline (0.015 mg/L; CCME 2011).

Figure 5.2-22

Observed and Fitted Means for Total Uranium Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: Symbols represent observed mean values; Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
CCME Guideline = 0.015 mg/L.

5.2.4.23 Total Vanadium

> ***Summary:*** Statistical and graphical analyses suggest that mine operations have had no effect on total vanadium concentrations in any of the lakes or streams in the Pigeon-Fay and Upper Exeter Watershed. Observed concentrations were less than the SSWQO at all sites in 2014.

Total vanadium concentrations in monitored and reference lakes have generally been below the detection limit since monitoring began. Thus, no statistical analyses were possible for lake vanadium concentrations (Table 5.2-27). Graphical analysis indicates that vanadium concentrations have at times been detected in Fay Bay and Upper Exeter Lake, but there is no evidence of an increase in concentration in 2014, when compared to the before period (Figure 5.2-23). Vanadium concentrations have also generally been below detection limits in reference streams, but were detected, at times in both Vulture-Polar Stream and at the monitored site in Pigeon Stream. Statistical and graphical analyses indicate that no changes in vanadium concentrations were detected between the monitored and reference stream site, when comparing the before and after periods (Table 5.2-27; Figure 5.2-23). Observed and fitted mean vanadium concentrations were less than the vanadium SSWQO (0.03 mg/L) in all lakes and streams in 2014 (see Part 2 - Data Report; Rescan 2012g).

Table 5.2-27. Statistical Results of Total Vanadium in Lakes and Streams in the Pigeon-Fay and Upper Exeter Watershed

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Winter	Fay Bay	ALL	-	-	3-423
	Upper Exeter	ALL	-	-	
Summer	Fay Bay	ALL	-	-	3-426
	Upper Exeter	ALL	-	-	
Summer	Pigeon Reach 1	Nanuq Outflow, Counts Outflow, Pigeon Reach 7	$p = 0.32$	-	3-429

Note: Dashes indicate not applicable.

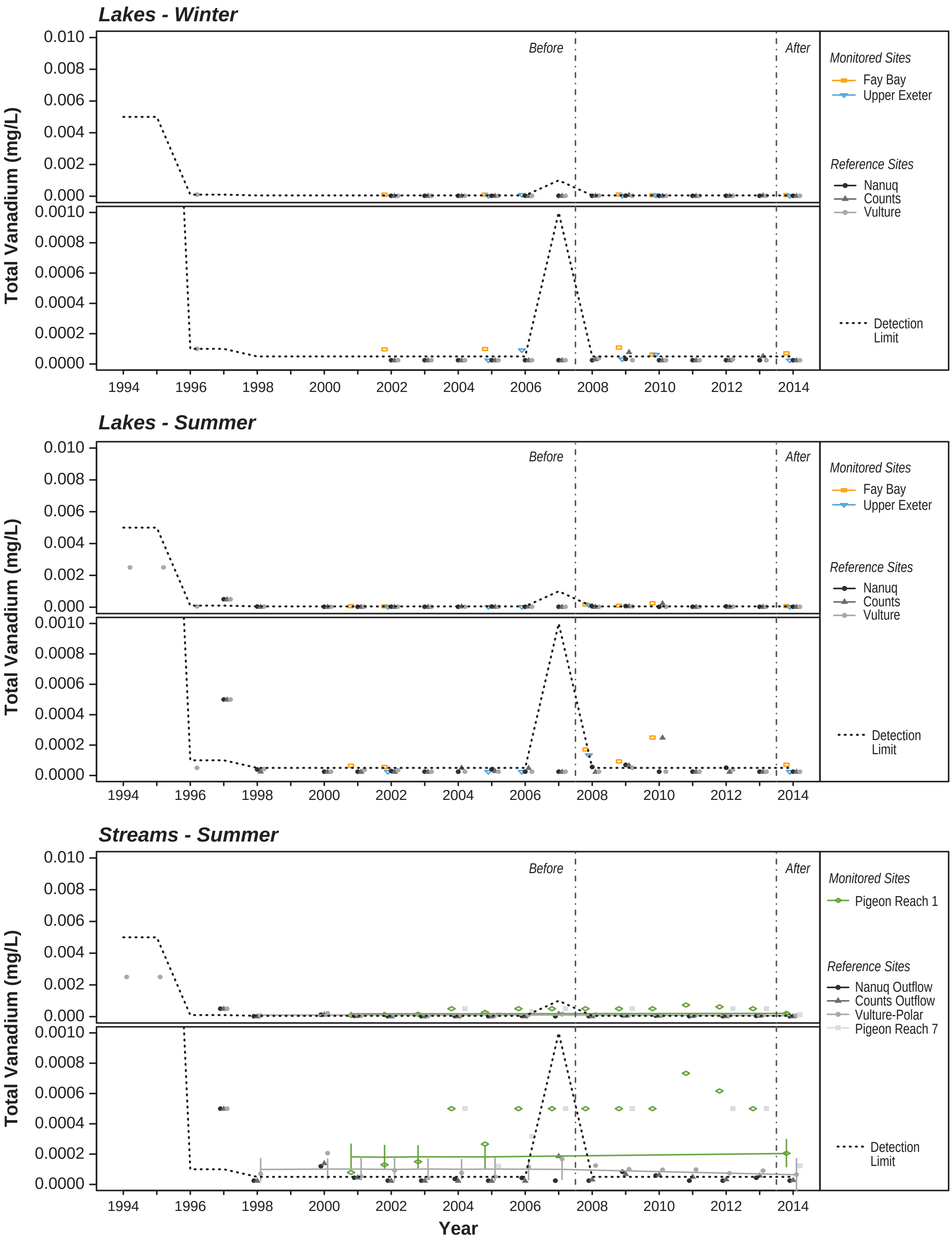
5.3 LAKE SEDIMENT QUALITY

5.3.1 Variables

Eleven lake sediment quality variables were evaluated for potential effects related to mine activities. These included total organic carbon (TOC), available phosphorus, total phosphorus, total nitrogen, antimony, arsenic, cadmium, molybdenum, nickel, selenium, and strontium.

Figure 5.2-23

Observed and Fitted Means for Total Vanadium Concentrations in Pigeon-Fay and Upper Exeter Watershed Lakes and Streams, 1994 to 2014



Notes: Symbols represent observed mean values; Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
SSWQO = 0.03 mg/L.

5.3.2 Dataset

The lake sediment quality data used in the 2014 evaluation of effects was collected from late July to mid-August, generally once every three years (Table 5.3-1). Baseline sediment quality data for the two monitored lakes were collected from 2001 to 2005, while baseline sediment quality data for the three reference lakes were collected from 1994 to 2005 (Table 5.3-1). In 1994, sediment quality data were collected in both early July and mid-August, but sediment quality did not differ significantly between these sampling periods. Thus data from early July 1994 to 1999 was included in Table 5.3-1 and shown in Figures 5.3-1 to 5.3-11. Sediment quality data collected from 2008 from 2011 were not used in the statistical analyses, but are included in Table 5.3-1 and shown in Figures 5.3-1 to 5.3-11 for visual comparison. Data collected from 2008 to 2010 in Fay Bay and Upper Exeter Lake were part of a monitoring program initiated in response to an unplanned release of fine processed kimberlite (FPK) in May of 2008 (Rescan 2011b). Thus, data from the "before" period in the BACI analysis encompasses data from 1994 to 2005 and the "after" period includes all data collected in 2014.

Table 5.3-1. Dataset Used for Evaluation of Effects on Sediment Quality in the Pigeon-Fay and Upper Exeter Watershed

Year	Nanuq	Counts	Vulture	Fay Bay	Upper Exeter
1994	-	-	Jul-1 (1), Aug-13 (1)	-	-
1995	-	-	-	-	-
1996	-	-	-	-	-
1997	Aug-4 (1)	Aug-4 (1)	Aug-4 (1)	-	-
1998	Aug-4 (3)	Aug-4 (2)	Aug-7 (3)	-	-
1999	Jul-30 (3)	Jul-30 (3)	Jul-29 (3)	-	-
2000	-	-	-	-	-
2001	-	-	-	Aug–5 (3)	-
2002	Aug-3 (3)	Aug-7 (3)	Aug-3 (3)	Aug–8 (3)	Aug-8 (3)
2003	-	-	-	-	-
2004	-	-	-	-	-
2005	Aug-1 (3)	Aug-7 (3)	Jul-31 (3)	-	Aug-19 (3)
2006	-	-	-	-	-
2007	-	-	-	-	-
2008*	Aug-8 (3)	Jul-31 (3)	Aug-5 (3)	Aug-3 (2)	-
2009*	-	-	-	Aug-4 (3)	-
2010*	-	-	-	-	-
2011*	Aug-13 (3)	Aug-11 (1), Aug-14 (1)	Aug-13 (3)	-	-
2012*					
2013*					
2014	Aug-5 (3)	Aug-9 (3)	Aug-3 (3)	Aug-3 (3)	Aug-10 (3)

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

Sediment sampling methods have been consistent since monitoring began in 1994. Samples were collected using a standard Ekman grab and the top 2 cm were collected for analysis. In 2014, sediment samples at all AEMP lake sites were also collected using a K-B corer. Core samples were collected from other watershed AEMP lakes in 2011 as part of a study to determine whether core sampling might provide a better measure of potential changes in sediment chemistry (Rescan 2012c). Results from that study will be further examined along with results from 2014 as part of the 2015 AEMP Re-evaluation. ALS has been analyzing the AEMP sediment samples since 1994. Analytical detection limits for sediment quality variables are illustrated as black dashed lines (Figures 5.3-1 to 5.3-11).

Analyses are conducted on sediments collected from the mid depth strata (5.1 – 10 m) in the three reference lakes and in Fay Bay. In Upper Exeter Lake, analyses are conducted on sediments collected from the deep depth strata (< 10 m), since baseline data was only available for the deep depth strata. A review of the sediment sampling program during the 2012 AEMP Re-evaluation indicated that mid and deep depth sediment quality have consistently yielded similar results in the AEMP Evaluation of Effects (Rescan 2012c).

For each year in which sediment data were collected, averages were calculated by pooling data from replicates collected. The arsenic concentration from replicate 3 in Vulture Lake collected on July 31, 2005 was deemed a statistical outlier and was not included in statistical analyses.

5.3.3 Results and Discussion

5.3.3.1 TOC

> ***Summary:*** Statistical and graphical analyses suggest that mine operations have had no effect on TOC percentages at any of the monitored sediment sites of the Pigeon-Fay and Upper Exeter Watershed.

Statistically significant changes in mean sediment TOC percentages were detected between monitored and reference sites when comparing the before and after periods (Table 5.3-2). However, results from the BACI contrasts indicate that the trend in monitored lakes was similar to that observed in at least two of the reference lakes (Table 5.3-2). Graphical analysis shows that TOC percentages in 2014 may have increased in Fay Bay sediments when compared to the before period, but a similar trend was observed in two of the reference lakes (i.e., Nanuq and Vulture lakes; Figure 5.3-1). Percentages in Upper Exeter Lake remained similar in the before and after periods (Figure 5.3-1). Thus, no mine effects were detected.

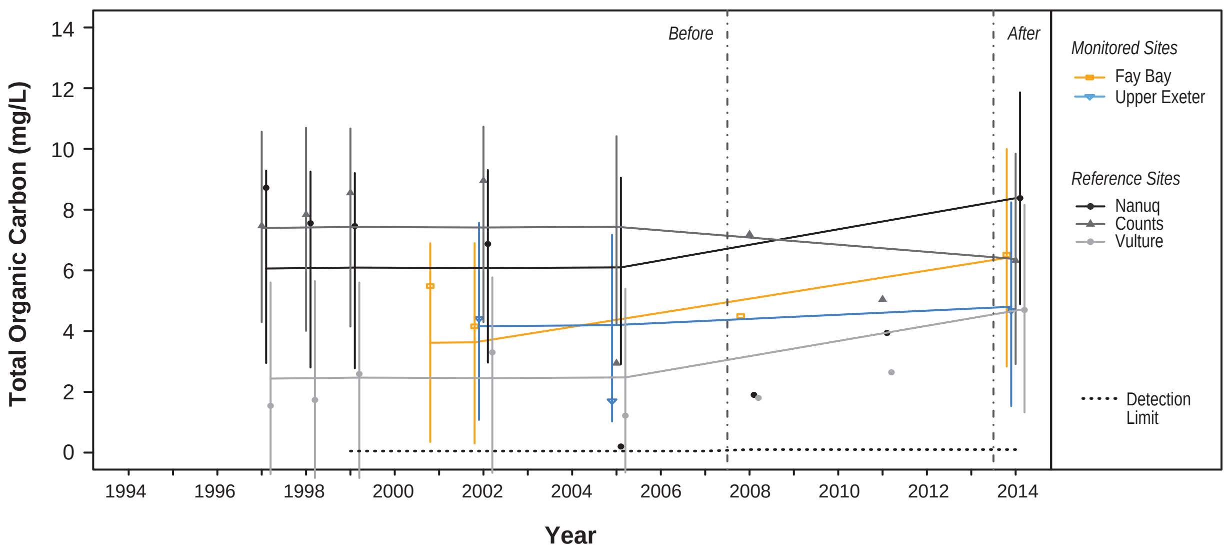
Table 5.3-2. Statistical Results of TOC in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results: BACI Interaction Term	BACI Results: Significant BACI Contrasts	Statistical Report Page No.
Summer	Fay Bay	-	p = 0.01	Counts	3-439
	Upper Exeter	-	p = 0.01	None	

Note: Dashes indicate not applicable.

Figure 5.3-1

Observed and Fitted Means for Total Organic Carbon Percentages in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.

5.3.3.2 *Available Phosphorus*

Summary: Statistical and graphical analyses suggest that mine operations have had no effect on available phosphorus concentrations at any of the monitored sediment sites in the Pigeon-Fay and Upper Exeter Watershed.

No statistically significant change in mean sediment available phosphorus concentration was detected between Fay Bay and reference sites when comparing the before and after periods (Table 5.3-3). A significant change in available phosphorus was detected in Upper Exeter Lake (Table 5.3-3); however, graphical analysis shows that available phosphorus concentrations in 2014 decreased in Upper Exeter Lake when compared to the before period (Figure 5.3-2). Thus, no mine effects were detected.

Table 5.3-3. Statistical Results of Available Phosphorus in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Summer	Fay Bay	-	p = 0.06	-	3-448
	Upper Exeter	-	p = 0.0002	Counts, Nanuq, Vulture	

Note: Dashes indicate not applicable.

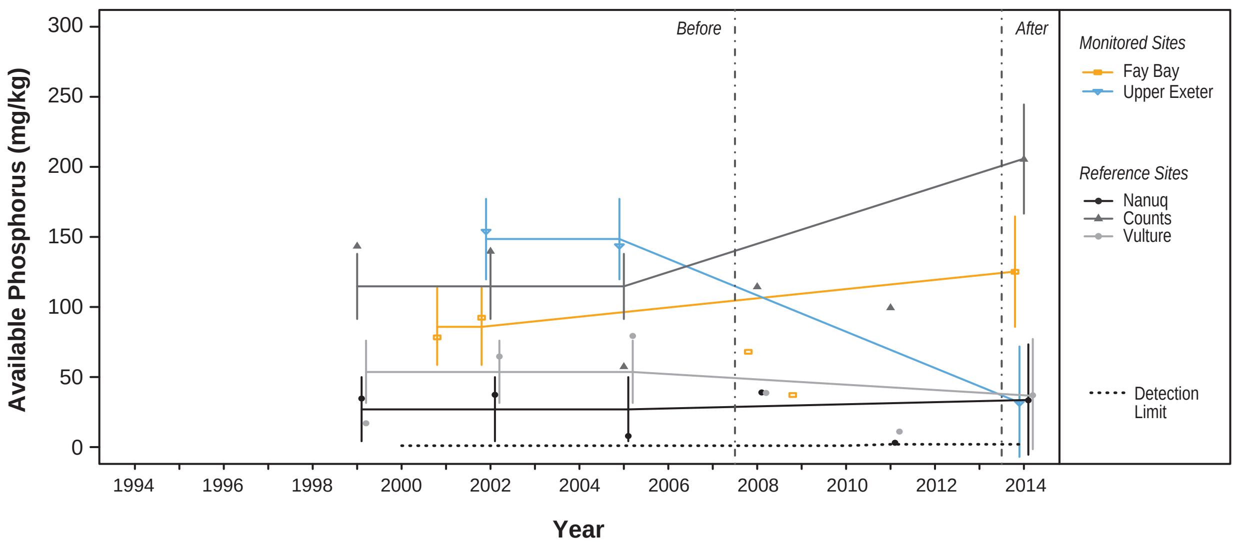
5.3.3.3 *Total Nitrogen*

Summary: Statistical and graphical analyses suggest that mine operations have had no effect on total nitrogen percentages at any of the monitored sediment sites of the Pigeon-Fay and Upper Exeter Watershed.

Statistically significant changes in mean sediment nitrogen percentages were detected between monitored and reference sites when comparing the before and after periods (Table 5.3-4). However, results from the BACI contrasts indicate that the trend in monitored lakes was similar to that observed in at least two of the reference lakes (Table 5.3-4). Graphical analysis shows that total nitrogen percentages in 2014 increased in Fay Bay sediments when compared to the before period, but a similar trend was observed in two of the reference lakes (i.e., Nanuq and Vulture lakes; Figure 5.3-3). Percentages in Upper Exeter Lake remained similar in the before and after periods (Figure 5.3-3). Thus, no mine effects were detected.

Figure 5.3-2

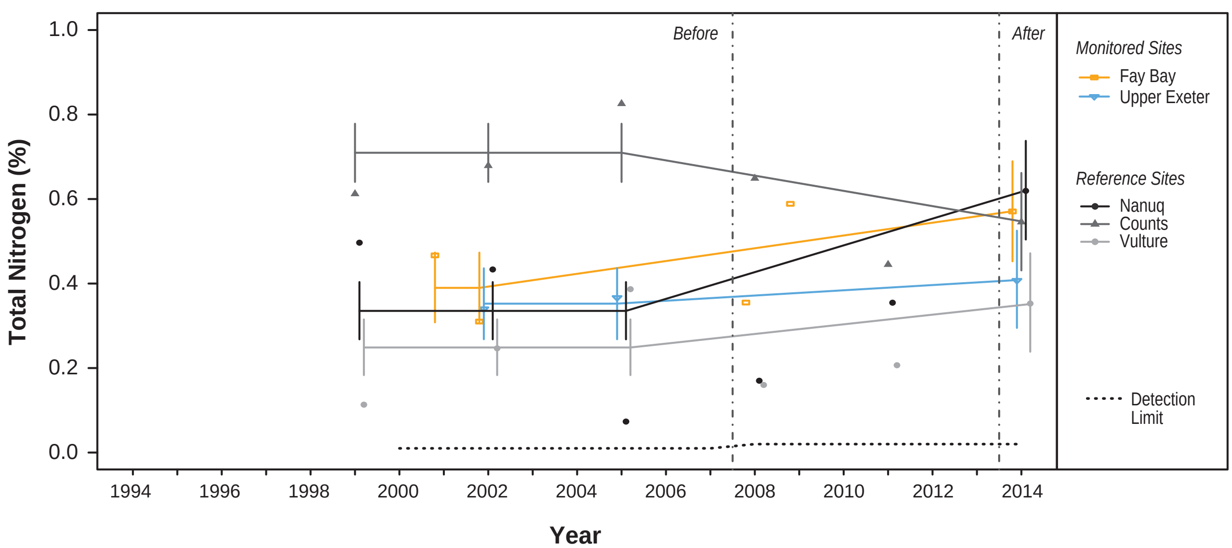
Observed and Fitted Means for Available Phosphorus Concentrations in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.

Figure 5.3-3

Observed and Fitted Means for Total Nitrogen Percentages in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.

Table 5.3-4. Statistical Results of Total Nitrogen in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Summer	Fay Bay	-	$p = 0.01$	Counts	3-457
	Upper Exeter	-	$p = 0.01$	None	

Note: Dashes indicate not applicable.

5.3.3.4 Total Antimony

Summary: Concentrations of antimony in sediments have been at or below detection limits since monitoring began. No mine effects were detected.

Antimony concentrations have only been analyzed in Upper Exeter Lake since 2005, and no before data were available for Fay Bay. For Upper Exeter Lake, all observations in 2005 were below the detection limit. Thus, all monitored lakes were excluded from the statistical analyses and no tests were performed (Table 5.3-5). Consequently, graphical analysis and best professional judgment were the primary methods used in the evaluation of effects. Graphical analysis indicates that antimony concentrations in sediments of all monitored and reference lakes have been at or below detection limits since monitoring began (Figure 5.3-4). Thus, no mine effects were detected.

Table 5.3-5. Statistical Results of Total Antimony in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Summer	Fay Bay	ALL	-	-	3-466
	Upper Exeter	ALL	-	-	

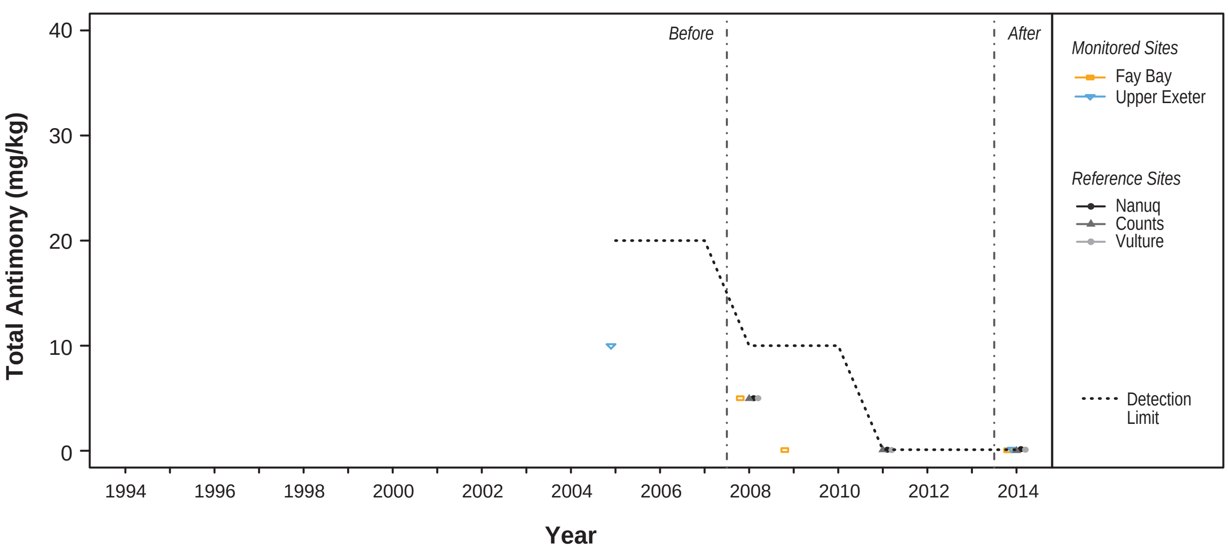
Note: Dashes indicate not applicable.

5.3.3.5 Total Arsenic

Summary: Statistical and graphical analyses suggest that mine operations have had no effect on total arsenic concentrations at any of the monitored sediment sites of the Pigeon-Fay and Upper Exeter Watershed. The 95% confidence intervals of the fitted mean in Fay Bay and the observed mean arsenic concentration in Upper Exeter Lake exceeded the CCME ISQG and PEL; however, similar patterns were observed in reference lakes.

Figure 5.3-4

Observed and Fitted Means for Total Antimony Concentrations in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.

Statistically significant changes in mean sediment arsenic concentrations were detected between monitored and reference sites when comparing the before and after periods (Table 5.3-6). However, results from the BACI contrasts indicate that the trend in Fay Bay was similar to that observed in two of the reference lakes and graphical analysis shows that arsenic concentrations were similar in the before and after periods (Table 5.3-6; Figure 5.3-5). For Upper Exeter Lake, graphical analysis shows that arsenic concentrations in 2014 increased in Upper Exeter Lake when compared to the before period (Table 5.3-6; Figure 5.3-5). However, a larger increase was found in one of the reference lakes (i.e., Vulture Lake; Figure 5.3-6).

Table 5.3-6. Statistical Results of Total Arsenic in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Summer	Fay Bay	-	$p = 0.0001$	Vulture	3-469
	Upper Exeter	-	$p < 0.001$	Counts, Nanuq	

Note: Dashes indicate not applicable.

The 95% confidence interval of the fitted mean in Fay Bay and the observed mean in Upper Exeter exceeded the CCME ISQG of 5.9 mg/kg and the CCME PEL of 17 mg/kg in 2014 (CCME 2002). However, the observed mean in all three reference lakes also exceeded the CCME ISQG, and the observed mean in one reference lake (i.e., Vulture Lake) and the 95% confidence intervals in the two other reference lakes exceeded the CCME PEL in 2014. Furthermore, arsenic concentrations in reference lake sediments have frequently been greater than the CCME guidelines since monitoring began. Thus, no mine effects were detected.

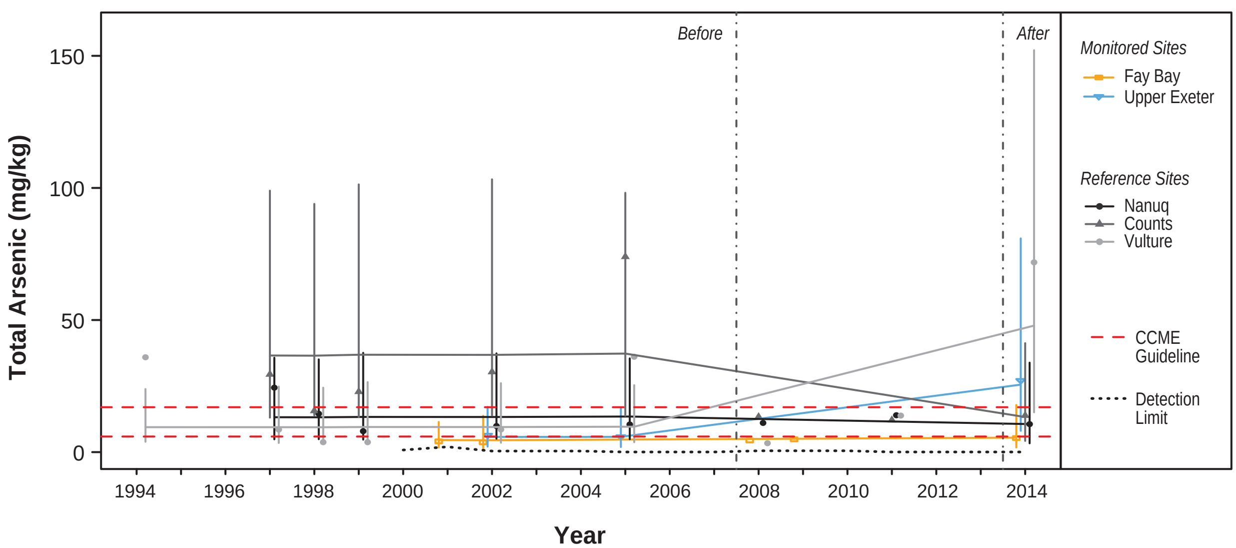
5.3.3.6 *Total Cadmium*

Summary: Statistical and graphical analyses suggest that mine operations have had no effect on total cadmium concentrations at any of the monitored sediment sites in the Pigeon-Fay and Upper Exeter Watershed. Total cadmium concentrations were less than the CCME ISQG and PEL in all monitored and reference sites.

Cadmium concentrations in Fay Bay sediments during the before period were below detection limits, thus no statistical analysis was possible for this lake (Table 5.3-7). There was no significant change in mean sediment cadmium concentration between Upper Exeter Lake and reference sites when comparing the before and after periods (Table 5.3-7). Graphical analysis shows that cadmium concentrations may have increased in sediments from Fay Bay in 2014 when compared to the before period, but a similar trend was observed in two of the reference lakes (i.e., Counts and Vulture lakes; Figure 5.3-6). All observed and fitted concentrations were less than the CCME ISQG of 0.6 mg/kg and the CCME PEL of 3.5 mg/kg at all monitored sediment sites in 2014 (CCME 1999a). Thus, no mine effects were detected.

Figure 5.3-5

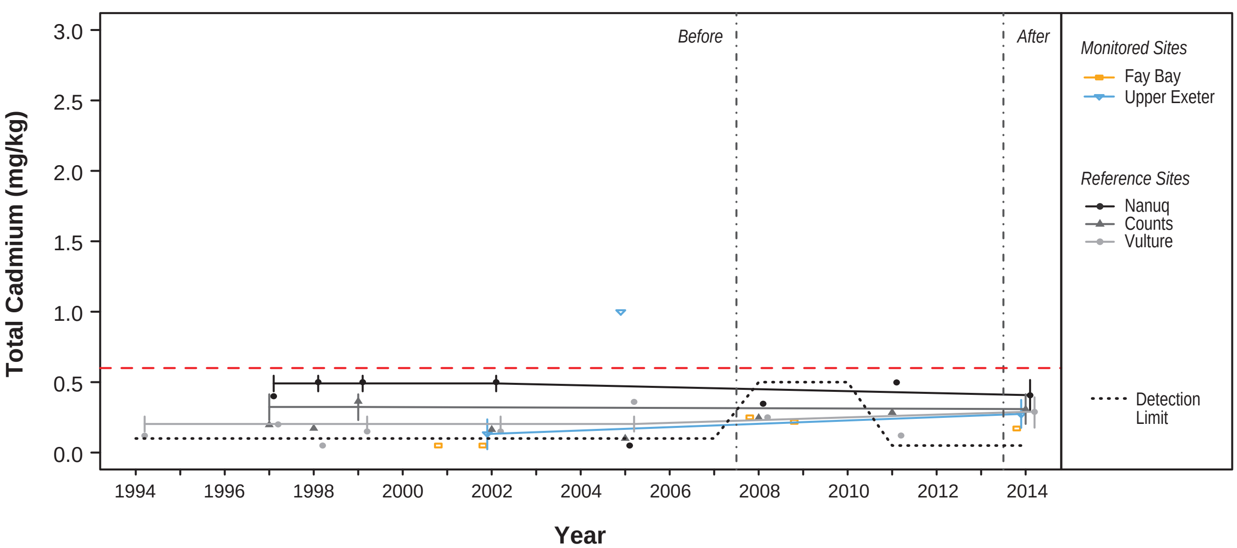
Observed and Fitted Means for Total Arsenic Concentrations in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
CCME guidelines: ISQG = 5.9 mg/kg; PEL = 17 mg/kg.

Figure 5.3-6

Observed and Fitted Means for Total Cadmium Concentrations in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.
CCME guidelines: ISQG = 0.6 mg/kg; PEL = 3.5 mg/kg (not shown).

Table 5.3-7. Statistical Results of Total Cadmium in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Summer	Fay Bay	Fay Bay	-	-	3-478
	Upper Exeter	-	p = 0.23	-	

Note: Dashes indicate not applicable.

5.3.3.7 Total Molybdenum

Summary: Concentrations of molybdenum in sediments have generally been at or below detection limits since monitoring began. No mine effects were detected.

Molybdenum concentrations in Fay Bay and Upper Exeter Lake sediments during the before period were below detection limits. Thus, all monitored lakes were excluded from the statistical analyses and no tests were performed (Table 5.3-8). Consequently, graphical analysis and best professional judgment were the primary methods used in the evaluation of effects. Graphical analysis indicates that molybdenum concentrations in sediments of all monitored and reference lakes have generally been at or below detection limits since monitoring began (Figure 5.3-7). In 2014, all observations were greater than detection limits, but there was no evidence of greater concentrations in monitored lakes when compared to reference lakes. Thus, no mine effects were detected.

Table 5.3-8. Statistical Results of Total Molybdenum in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Summer	Fay Bay	ALL	-	-	3-485
	Upper Exeter	ALL	-	-	

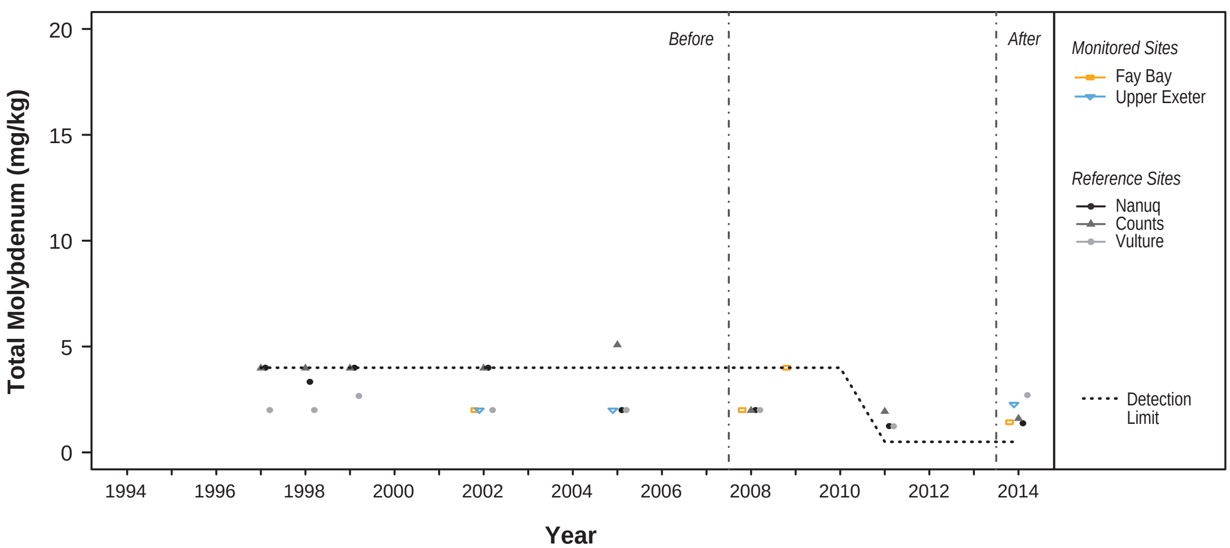
Note: Dashes indicate not applicable.

5.3.3.8 Total Nickel

Summary: Statistical and graphical analyses suggest that nickel concentrations have increased in Upper Exeter Lake. Since nickel concentrations in Upper Exeter Lake water quality have remained similar in the before and after period, the source of the increase in sediments is unclear at this time and unlikely related to mine activities.

Figure 5.3-7

Observed and Fitted Means for Total Molybdenum Concentrations in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.

No statistically significant changes in mean sediment nickel concentrations were detected between Fay Bay and reference sites when comparing the before and after periods (Table 5.3-9). Statistically significant changes in mean sediment nickel concentrations were detected between Upper Exeter Lake and reference lakes (Table 5.3-9). Graphical analysis suggests that nickel concentrations in 2014 increased in Fay Bay and Upper Exeter Lake sediments when compared to the before period, but a similar trend was observed in one of the reference lakes (i.e., Vulture Lake; Figure 5.3-8). However, the magnitude of increase observed in Upper Exeter Lake is greater than that observed in Vulture Lake. Thus, results suggest that sediment nickel concentrations have increased in Upper Exeter Lake. Nickel concentrations in Upper Exeter Lake water quality have remained similar in the before and after period, thus the source of the increase in sediments is unclear at this time and unlikely related to mine activities.

Table 5.3-9. Statistical Results of Total Nickel in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Summer	Fay Bay	-	p = 0.18	-	3-488
	Upper Exeter	-	p = 0.02	Counts, Nanuq	

Note: Dashes indicate not applicable.

5.3.3.9 Total Phosphorus

Summary: Statistical and graphical analyses suggest that mine operations have had no effect on total phosphorus concentrations at any of the monitored sediment sites in the Pigeon-Fay and Upper Exeter Watershed.

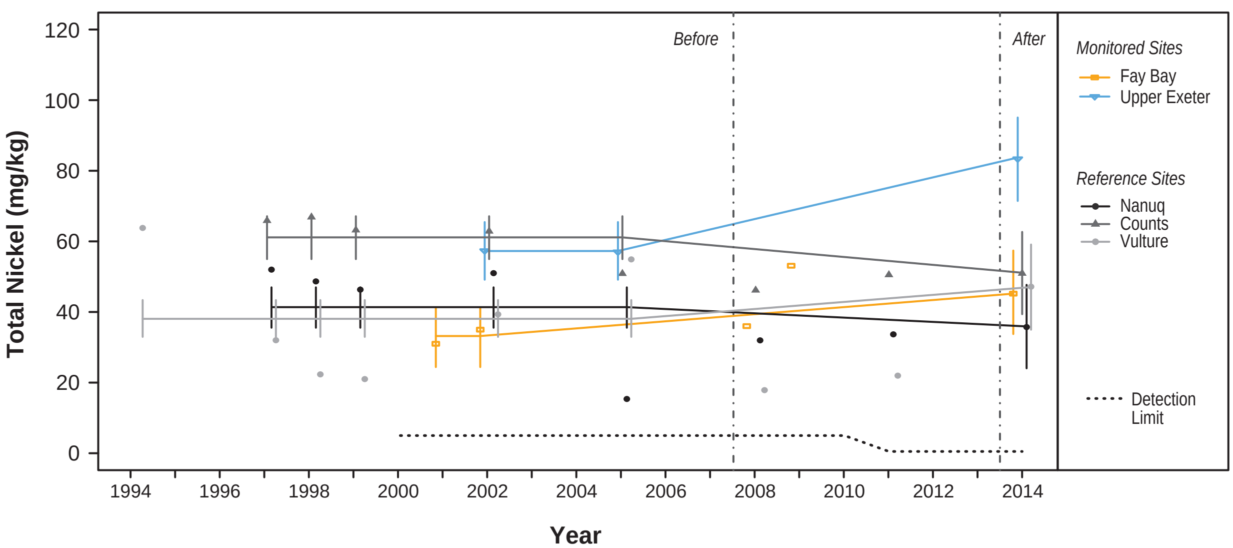
Statistically significant changes in mean sediment phosphorus concentrations were detected between monitored and reference sites when comparing the before and after periods (Table 5.3-10). However, results from the BACI contrasts indicate that the trend in monitored lakes was similar to that observed in at least two of the reference lakes (Table 5.3-10). Graphical analysis shows that total phosphorus concentrations in Fay Bay and Upper Lake were similar between the before and after periods (Figure 5.3-9). Thus, no mine effects were detected.

Table 5.3-10. Statistical Results of Total Phosphorus in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Summer	Fay Bay	-	p = 0.003	Counts	3-497
	Upper Exeter	-	p = 0.001	Counts	

Note: Dashes indicate not applicable.

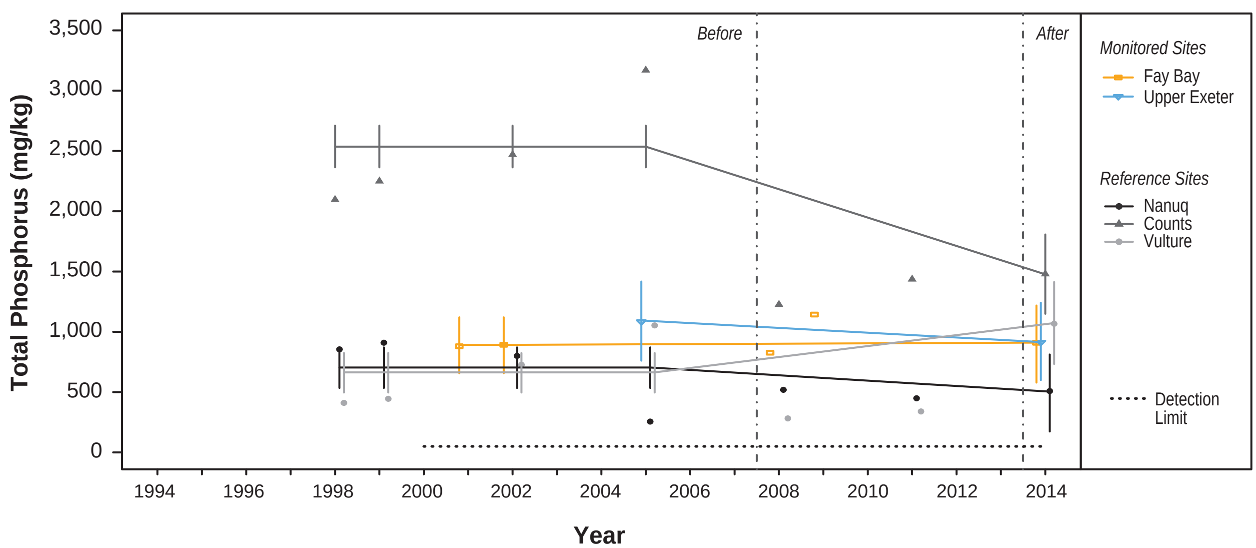
Observed and Fitted Means for Total Nickel Concentrations in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.

Figure 5.3-9

Observed and Fitted Means for Total Phosphorus Concentrations in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.

5.3.3.10 Total Selenium

> ***Summary:*** Statistical and graphical analyses suggest that mine operations have had no effect on total selenium concentrations at any of the monitored sediment sites of the Pigeon-Fay and Upper Exeter Watershed.

Statistically significant changes in mean sediment selenium concentrations were detected between monitored and reference sites when comparing the before and after periods (Table 5.3-11). However, results from the BACI contrasts indicate that the trend in monitored lakes was similar to that observed in at least two of the reference lakes (Table 5.3-11). Graphical analysis indicates that selenium concentrations in all monitored and reference lake sediments were similar in the before and after periods (Figure 5.3-10). Thus, no mine effects were detected.

Table 5.3-11. Statistical Results of Total Selenium in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Summer	Fay Bay	-	p = 0.01	Counts	3-506
	Upper Exeter	-	p = 0.02	Counts	

Note: Dashes indicate not applicable.

5.3.3.11 Total Strontium

> ***Summary:*** No statistical analyses were possible at this time; however, graphical analysis suggests that mine operations have had no effect on total strontium concentrations at any of the monitored sediment sites of the Pigeon-Fay and Upper Exeter Watershed.

Strontium concentrations have only been analyzed since 2005 in Upper Exeter Lake and no before data were available for Fay Bay or any of the reference lakes. Thus, no statistical tests were performed (Table 5.3-12). Consequently, graphical analysis and best professional judgment were the primary methods used in the evaluation of effects. Graphical analysis indicates that mean strontium concentrations in sediments of Fay Bay and Upper Exeter have been variable through time and that concentrations in reference lakes may have increased from 2008 to 2014 (Figure 5.3-11). Thus, no mine effects were detected.

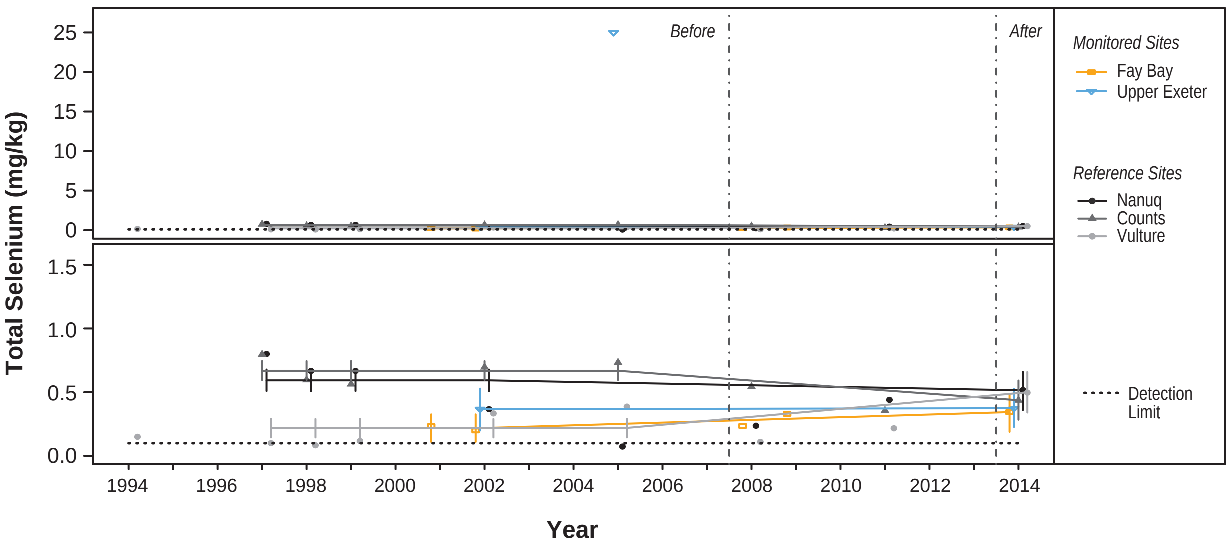
Table 5.3-12. Statistical Results of Total Strontium in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes

Season	Lake/Stream	Lakes/Streams Removed from Analysis	BACI Results		Statistical Report Page No.
			BACI Interaction Term	Significant BACI Contrasts	
Summer	Fay Bay	ALL	-	-	3-515
	Upper Exeter	ALL	-	-	

Note: Dashes indicate not applicable.

Figure 5.3-10

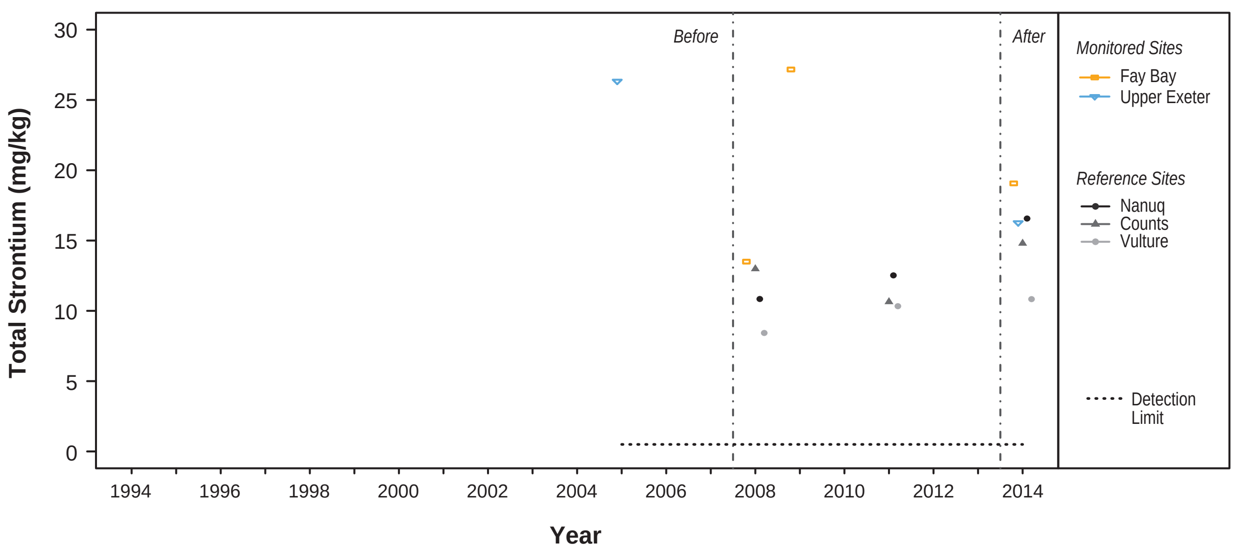
Observed and Fitted Means for Total Selenium Concentrations in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.

Figure 5.3-11

Observed and Fitted Means for Total Strontium Concentrations in Sediments in Pigeon-Fay and Upper Exeter Watershed Lakes, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.

5.4 AQUATIC BIOLOGY

The extent to which changes in water quality variables might result in changes in phytoplankton 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). Benchmarks and CCME guidelines for the protection of aquatic life exist for some water quality variables (see Sections 2.3). These guidelines and benchmarks provide an important interpretive tool for evaluating the toxicological significance of water chemistry data.

Results from water quality analyses in the Pigeon-Fay and Upper Exeter Watershed suggest that changes might be expected in phytoplankton communities in Fay Bay, as concentrations of nine evaluated water quality variables have increased in 2014 (see Section 5.2.4). Increases in water quality variables may be related to the unplanned release of FPK in 2008 (Rescan 2011b). In all cases, observed mean concentrations for these variables were below their respective benchmark values (see Section 5.2.4), and in general, increases in magnitude were relatively small.

Concentrations of water quality variables that have increased in monitored lakes of the Koala and King-Cujo watersheds 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 2012c). Concentrations of all the water quality variables in the Pigeon-Fay and Upper Exeter Watershed in 2014 were below the lowest identified chronic effect level for the most sensitive species (Rescan 2012c). 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 Fay Bay and Upper Exeter Lake.

The overall results of the 2012 AEMP Re-evaluation suggested that observed changes in biological community composition in the Koala and King-Cujo watersheds 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 2012c). In contrast to those two watersheds, no major changes in nutrients were observed in the Pigeon-Fay and Upper Exeter Watershed.

As the concentrations of all water quality variables in the Pigeon-Fay and Upper Exeter Watershed have remained below all guideline values with no toxic effects expected, and no major changes in nutrient availability have been observed, there is no reason to expect adverse biological effects in 2014.

5.4.1 Phytoplankton

5.4.1.1 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) and community composition were evaluated to determine whether mine activities have affected phytoplankton communities.

5.4.1.2 Dataset

Phytoplankton have been collected for analysis between late July and early August of each year for the evaluation of effects (Table 5.4-1). Baseline data, which was collected between 1994 and 1997, are included in graphical analysis but not in the statistical evaluation of effects.

Table 5.4-1. Dataset Used for Evaluation of Effects on the Phytoplankton in Pigeon-Fay and Upper Exeter Watershed Lakes

Year	Nanuq	Counts	Vulture	Fay Bay	Upper Exeter
1994	-	-	Aug-13	-	-
1995	-	-	-	-	-
1996	-	-	Jul-28	-	-
1997	Aug-4	Aug-14	Aug-5	-	-
1998	Aug-3	Aug-3	Aug-6	-	-
1999	Aug-7	Aug-8	Aug-6	-	-
2000	Aug-4	Aug-1	Aug-4	-	-
2001	Aug-1	Jul-30	Aug-2	Aug-5	-
2002	Aug-1	Aug-7	Aug-3	Jul-13	Jul-13
2003	Aug-9	Aug-7	Aug-4	-	-
2004	Aug-10	Aug-13	Aug-9	-	-
2005	Aug-1	Aug-7	Jul-31	-	Aug-21
2006	Aug-2	Aug-4	Aug-2	-	Aug-6
2007	Aug-11	Aug-6	Aug-12	-	-
2008*	Aug-8	Jul-31	Jul-29	Jul-12	Jul-14
2009*	Jul-30	Aug-1	Jul-30	-	-
2010*	Aug-6	Aug-7	Aug-5	Aug-13	-
2011*	Aug-2	Aug-5	Aug-5	-	-
2012*	Aug-1	Aug-8	Aug-7	-	-
2013*	Aug-3	Aug-1	Aug-1	-	-
2014	Aug-5	Aug-9	Aug-3	Aug-1	Aug-10

Notes: Dashes indicate no data were available.
Single samples were collected yearly for biomass analysis from 1994 to 1996.
Triplicate samples were collected yearly from 1996 to 2014 for density and diversity analysis.
** Indicates data were not included in the statistical evaluation of effects but were included in plots of observed and fitted data for visual comparison.*

5.4.1.3 Results and Discussion

Chlorophyll *a*

No statistically significant changes in chlorophyll *a* concentrations were detected between monitored and reference sites when comparing the before and after periods (Table 5.4-2). However, graphical analysis suggests chlorophyll *a* concentrations in 2014 have increased in Fay Bay when compared to the before period (Figure 5.4-1). Mean chlorophyll *a* concentrations in Fay Bay in 2014 were greater than the mean ± 2 SD chlorophyll *a* concentration during baseline years (Table 5.4-3). The increase in chlorophyll *a* concentration observed in Fay Bay is driven by an overall increase in the density of phytoplankton in 2014 (see Density section below).

Table 5.4-2. Statistical Results of Chlorophyll *a* in Pigeon-Fay and Upper Exeter Watershed Lakes

		BACI Results		Statistical Report Page No.
Season	Lake/Stream	BACI Interaction Term	Significant BACI Contrasts	
Summer	Fay Bay	p = 0.1	-	3-518
	Upper Exeter	p = 0.77	-	

Note: Dashes indicate not applicable.

Table 5.4-3. Mean ± 2 Standard Deviations (SD) Baseline Concentrations of Chlorophyll *a* in each of the Pigeon-Fay and Upper Exeter Watershed Lakes

Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD
Nanuq	0.29 (11)	0 - 0.65	0.38 ± 0.37
Counts	0.67 (11)	0.03 - 1.30	0.93 ± 0.35
Vulture	0.30 (13)	0 - 0.64	0.46 ± 0.24
Fay Bay	0.53 (2)	0 – 1.23	2.07 ± 1.34
Upper Exeter	0.54 (2)	0 - 0.42	0.59 ± 0.42

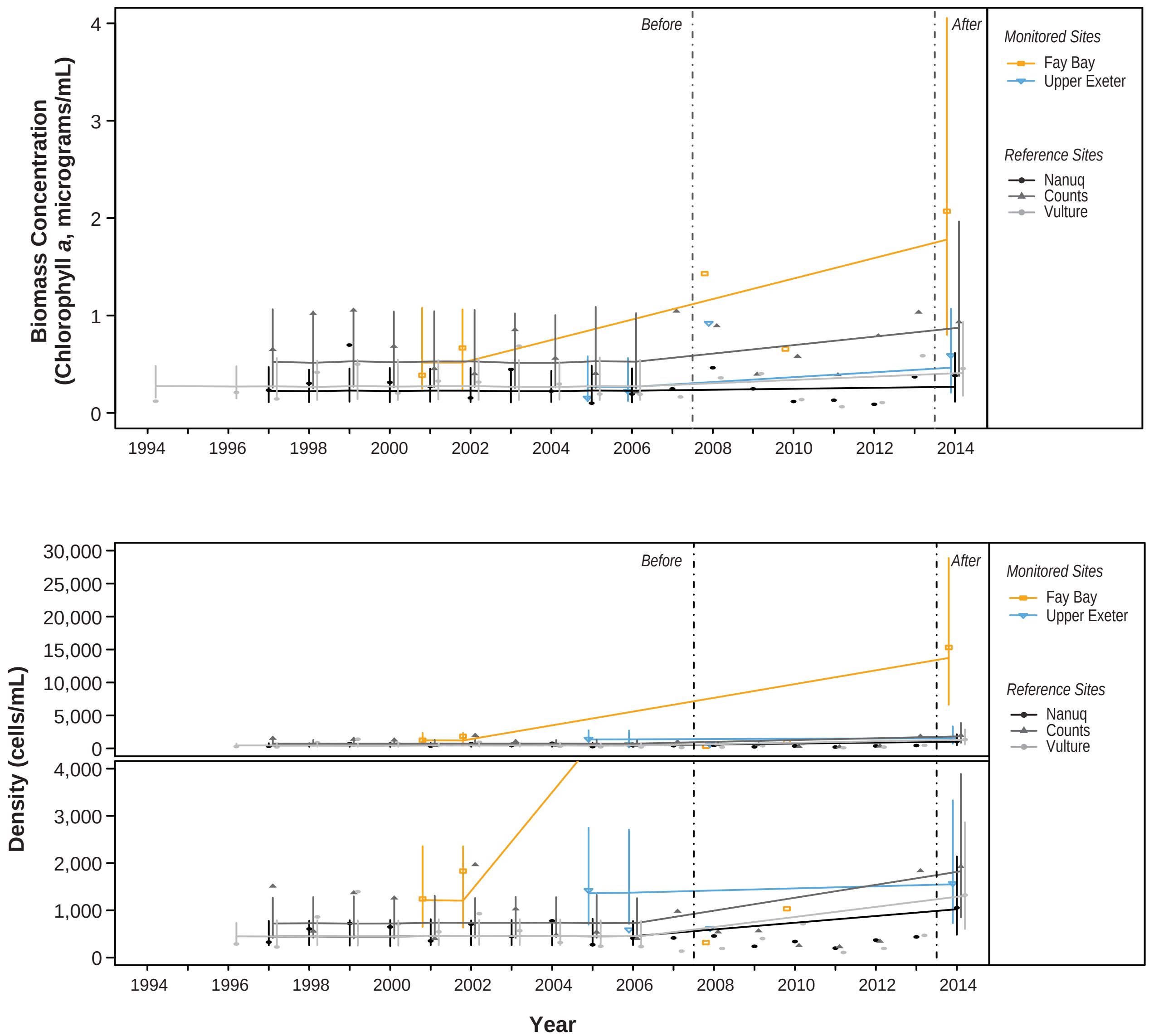
Notes: Units are µg/L.
Negative values were replaced with zeros.
N = number of years data were collected.

Density

No statistically significant changes in mean phytoplankton density were detected between Upper Exeter Lake and reference sites when comparing the before and after periods (Table 5.4-4). A change in mean phytoplankton density was detected for Fay Bay, and graphical analysis shows that phytoplankton density in 2014 increased in Fay Bay when compared to the before period (Table 5.4-4; Figure 5.4-1). Results from the BACI contrasts indicate that phytoplankton density in Fay Bay differed from all three reference lakes (Table 5.4-4). The observed mean phytoplankton density in Fay Bay in 2014 was greater than the mean baseline density ± 2 SD (Table 5.4-5). The increase in density observed in Fay Bay is driven by a large abundance of Myxophyceae (i.e., blue-green algae) in 2014 (see Diversity section below).

Figure 5.4-1

Observed and Fitted Means for Chlorophyll *a* Concentrations and Phytoplankton Density in Pigeon-Fay and Upper Exeter Watershed Lakes, 1994 to 2014



Notes: Symbols represent observed mean values.
Solid lines represent fitted curves.
Error bars represent the 95% inter-quantile range of bootstrapped fitted values based on the model.
Censored data and outliers are excluded from the model and the fitted values.
The positions of data along the x-axis have been adjusted for legibility.

Table 5.4-4. Statistical Results of Phytoplankton Density in Pigeon-Fay and Upper Exeter Watershed Lakes

		BACI Results		Statistical Report Page No.
Season	Lake/Stream	BACI Interaction Term	Significant BACI Contrasts	
Summer	Fay Bay	p = 0.001	Nanuq, Counts, Vulture	3-527
	Upper Exeter	p = 0.26	-	

Note: Dashes indicate not applicable.

Table 5.4-5. Mean ± 2 Standard Deviations (SD) Baseline Phytoplankton Density in each of the Pigeon-Fay and Upper Exeter Watershed Lakes

Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD
Nanuq	517 (11)	0 – 1,080	1,055 ± 346
Counts	953 (11)	0 - 2,337	1,924 ± 755
Vulture	538 (12)	0 – 1,542	1,324 ± 398
Fay Bay	1,537 (2)	0 – 3,169	15,307 ± 7,805
Upper Exeter	1,006 (2)	37 – 1,976	1,569 ± 455

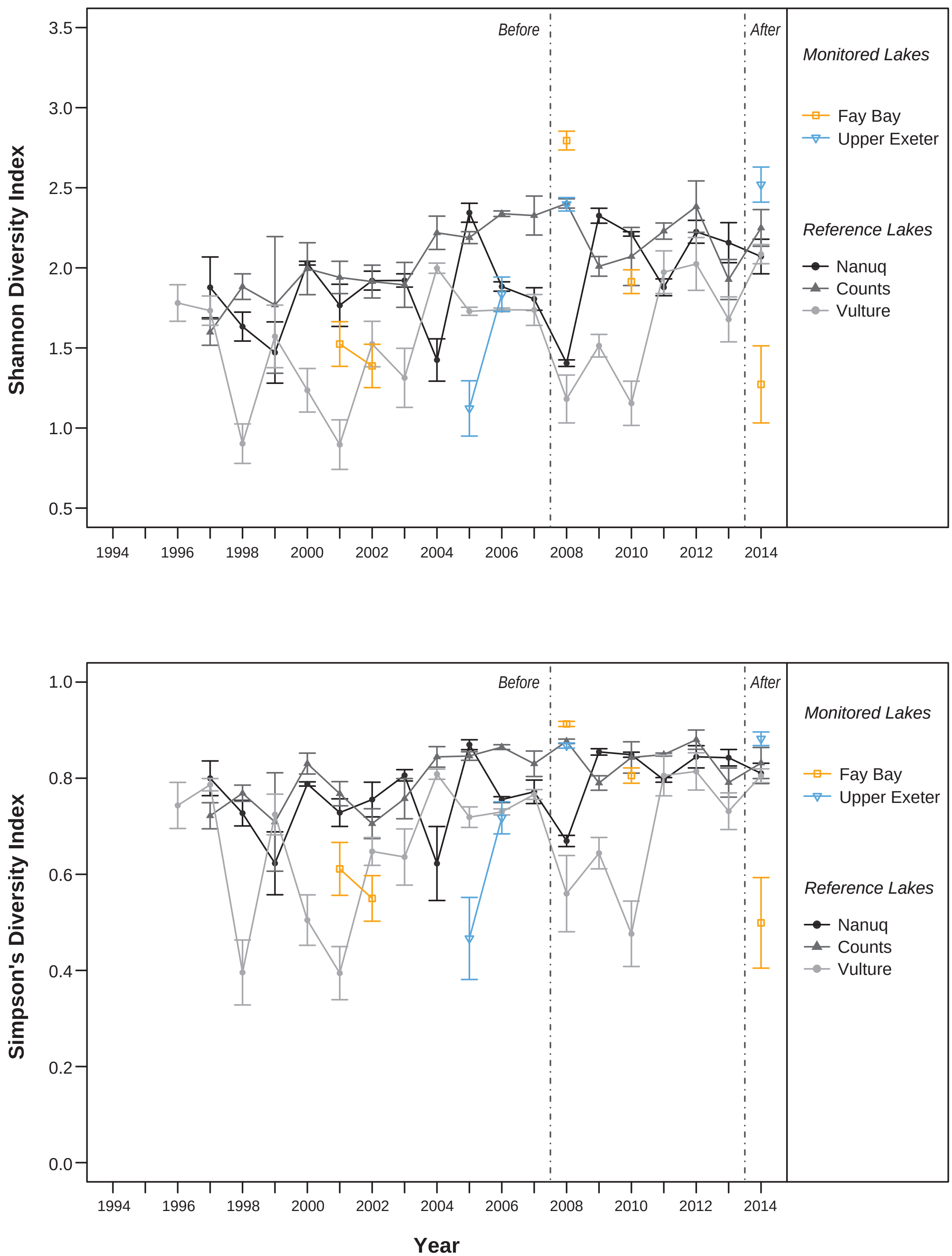
Notes: Units are cells/mL.
Negative values were replaced with zeros.
N = number of years data were collected.

Diversity and Community Composition

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 5.4-2) 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 5.4-3 and 5.4-4). Following recent advances in taxonomic classification, the names of two phytoplankton groups have been updated from previous reports: the Cyanophyta are now recognized as the class Myxophyceae and the Pyrrophyta are now recognized as the class Dinophyceae.

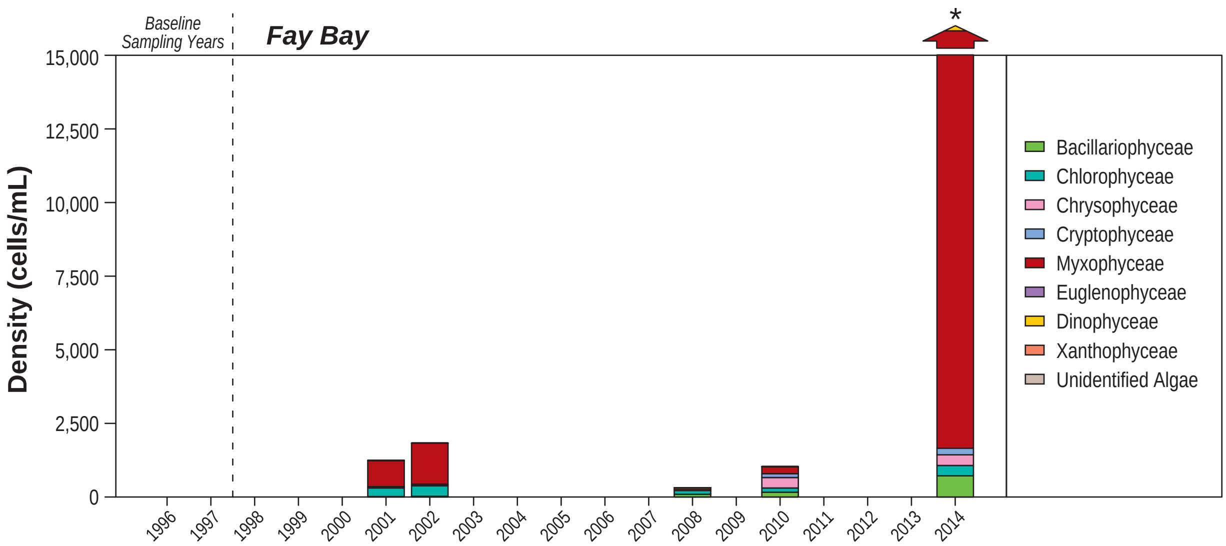
Phytoplankton diversity has been assessed sporadically through time in Fay Bay and Upper Exeter Lake, while both Shannon and Simpson's diversity indices have varied considerably through time in reference lakes since monitoring began (Figure 5.4-2). While the limited availability of data in monitored lakes and the variability observed in reference lakes makes it somewhat difficult to discern temporal trends, both Shannon and Simpson's diversity in Upper Exeter Lake appear to have increased in 2014 when compared to the before time period (Figure 5.4-2). Mean Shannon diversity was greater than ± 2 SD of mean baseline Shannon diversity in Upper Exeter Lake in 2014 (Table 5.4-6). The increase in diversity observed in Upper Exeter Lake likely reflects an increase in the absolute densities of Chlorophyceae and Chrysophyceae, corresponding to a more even distribution in abundance among the phytoplankton groups in 2014 (Figures 5.4-3 and 5.4-4).

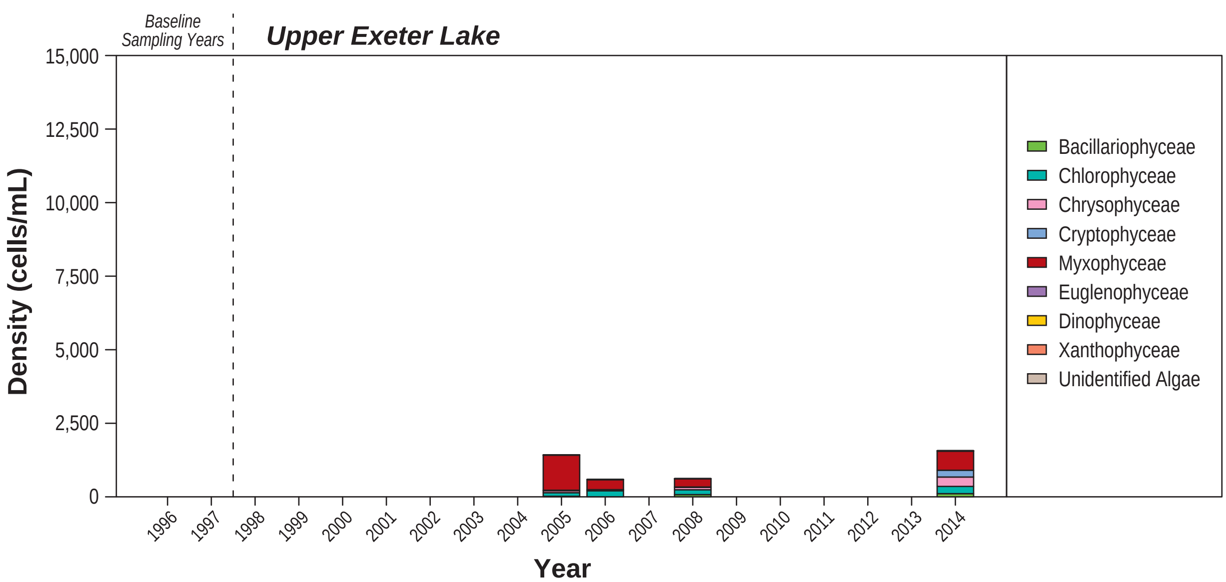
Average Diversity Indices for Phytoplankton in Pigeon-Fay and Upper Exeter Watershed Lakes, 1996 to 2014



Notes: Symbols represent observed mean values.
Error bars indicate standard error of the observed means.

Average Phytoplankton Density by Taxonomic Group for Lakes of the Pigeon-Fay and Upper Exeter Watershed, 1996 to 2014





*Note: *Total density in 2014 = 15,307: Myxophyceae = 13,627; Dinophyceae = 22.*

Figure 5.4-4

Relative Densities of Phytoplankton Taxa in Lakes of the Pigeon-Fay and Upper Exeter Watershed, 1996 to 2014

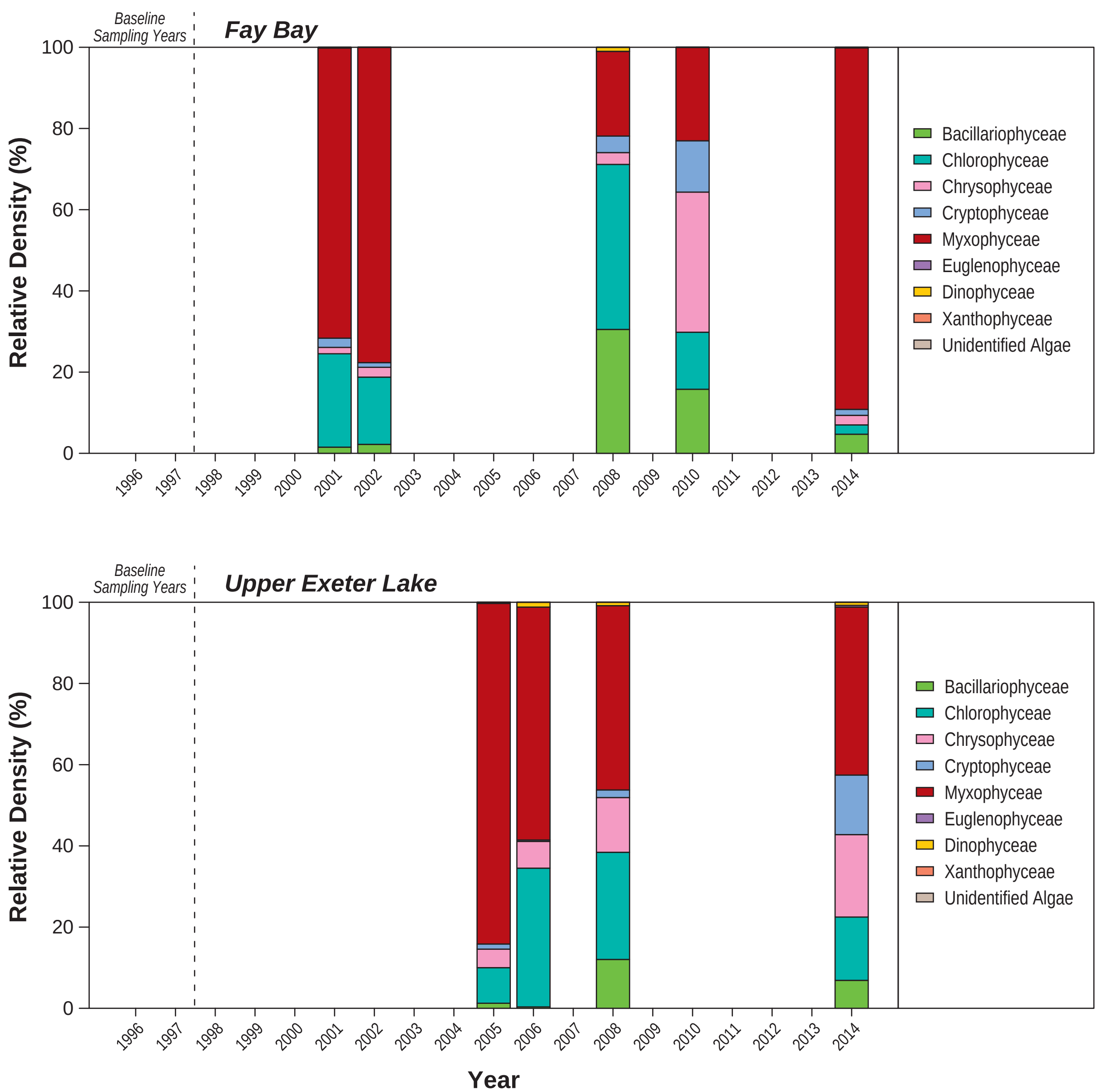


Table 5.4-6. Mean ± 2 Standard Deviations (SD) Baseline Phytoplankton Diversity in Each of the Pigeon-Fay and Upper Exeter Watershed Lakes

	Shannon Diversity			Simpson's Diversity		
Lake	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD	Baseline Mean (N)	Mean Baseline Range, ± 2 SD	2014 Mean ± 1 SD
Nanuq	1.83 (11)	1.24 – 2.41	2.07 ± 0.19	0.59 (11)	0.25 – 0.93	0.81 ± 0.04
Counts	2.01 (11)	1.35 - 2.66	2.25 ± 0.20	0.79 (11)	0.63 – 0.95	0.83 ± 0.06
Vulture	1.51 (12)	0.74 – 2.29	2.09 ± 0.10	0.65 (12)	0.35 – 0.96	0.81 ± 0.02
Fay Bay	1.46 (2)	1.00 – 1.91	1.27 ± 0.42	0.58 (2)	0.41 – 0.75	0.50 ± 0.16
Upper Exeter	1.48 (2)	0.58 – 2.38	2.52 ± 0.19	0.59 (2)	0.25 - 0.93	0.88± 0.02

Note: N = number of years data were collected.

A similar pattern of increasing densities of Chlorophyceae and Chrysophyceae was observed at the upstream Fay Bay site (Figure 5.4-3). However, there was also a large increase in the density of Myxophyceae observed at Fay Bay in 2014 (Figure 5.4-4). As a result of the large relative abundance of Myxophyceae (blue-green algae) observed at Fay Bay, no increase in diversity was observed (Figure 5.4-2). The cause of the large density of Myxophyceae observed at Fay Bay in 2014 is unclear at this time. Since blue-green algae are capable of fixing nitrogen, large densities are often linked to low nitrogen environments (Tillman et al. 1986), but no changes in nutrient availability were observed in Fay Bay between the before and after periods. The lack of data from 2011 to 2013 makes it difficult to determine whether these changes in blue-green algae density represent a trend through time and a mine effect or natural variability. If densities remain similar or increase in upcoming years, a mine effect may be indicated.

At this time, no mine effects were detected with respect to phytoplankton biomass, density, diversity, or community composition in the Pigeon-Fay and Upper Exeter Watershed.

5.4.2 Aquatic Biology Summary

No mine-related effects in chlorophyll *a* concentrations were detected in either Fay Bay or Upper Exeter Lake. Chlorophyll *a* concentration and phytoplankton density increased in 2014 in Fay Bay compared to the before period, and this increase was driven by a large abundance of Myxophyceae (i.e., blue-green algae). The cause of the large density of Myxophyceae observed at Fay Bay in 2014 is unclear at this time and may represent natural variability. Phytoplankton diversity in Upper Exeter Lake appears to have increased in 2014 compared to the before period, likely as a result of increases in the density of Chlorophyceae and Chrysophyceae, corresponding to a more even distribution in abundance among the phytoplankton groups. Phytoplankton density, diversity, and community composition have been assessed sporadically through time making it difficult to discern temporal trends. In particular, the lack of data from 2011 to 2013 makes it difficult to determine whether observed changes in blue-green algae density in Fay Bay and increased diversity in Upper Exeter Lake represent a trend through time and a mine effect, rather than representing natural variability. At this time, no mine effects were detected with respect to phytoplankton biomass, density, diversity, or community composition in the Pigeon-Fay and Upper Exeter Watershed.

5.5 SUMMARY

Table 5.5-1 summarizes the evaluation of effects for the Pigeon-Fay and Upper Exeter Watershed. 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 between the before and after periods, rather than for the direction of change.

No mine effects were detected with respect to Secchi depth in monitored lakes during the open water season in 2014 (Table 5.5-1).

A total of 23 water quality variables were evaluated for lakes and streams in the Pigeon-Fay and Upper Exeter Watershed in the 2014 AEMP. Of these, concentrations of nine variables have increased in Fay Bay in 2014, when compared to the before period (Table 5.5-1). In all cases, the source of the increase in Fay Bay is unclear at this time, but may be related to the unplanned release of FPK in May of 2008 (Rescan 2011b). CCME guidelines for the protection of aquatic life exist for ten of the evaluated water quality variables, including pH, TSS, total ammonia-N, nitrite-N, total arsenic, total boron, total cadmium, total nickel, total selenium, and total uranium (CCME 2014c). 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 in Section 2.3). 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 (CCME 2004). Other water quality benchmark values include provincial guidelines or ones taken from the published literature (see Table 2.3-1 in Section 2.3); antimony, barium, and strontium). All observed mean concentrations of the evaluated water quality variables were below their respective CCME guideline value, SSWQO, or relevant benchmark value in 2014 (Table 5.5-1), and in general, increases in magnitude were relatively small (see Section 5.2.4).

Eleven sediment quality variables were evaluated in the 2014 AEMP for the Pigeon-Fay and Upper Exeter Watershed. Of these, the concentrations of one variable (i.e., total nickel) showed signs of an increase, but the cause was unlikely related to mine activities (Table 5.5-1). CCME guidelines for the protection of aquatic life exist for two of the evaluated sediment quality variables, including arsenic and cadmium (see Table 2.4-1 in Section 2.4; CCME 2014b). For arsenic, the 95% confidence intervals of the fitted mean in Fay Bay and the observed mean in Upper Exeter Lake exceeded the CCME ISQG and PEL in 2014 (Table 5.5-1); however, similar patterns were observed in reference lakes. For cadmium, observed mean concentrations in 2014 were below the CCME ISQG and PEL guideline value in 2014 (Table 5.5-1).

Despite increases in nine evaluated water quality variables in the Pigeon-Fay and Upper Exeter Watershed, concentrations remained below all benchmark values with no toxic effects expected, and no major changes in nutrient availability were observed. Thus, there was no reason to expect adverse biological effects in 2014. An increase in chlorophyll *a* concentration and phytoplankton density in Fay Bay, and an increase in phytoplankton diversity in Upper Exeter Lake was observed in 2014. However, phytoplankton variables have been assessed sporadically through time making it difficult to discern temporal trends. In particular, the lack of data from 2011 to 2013 makes it difficult to determine whether observed changes represent a trend through time and a mine effect, rather than representing natural variability. At this time, no mine effects were detected with respect to phytoplankton biomass, density, diversity, or community composition in the Pigeon-Fay and Upper Exeter Watershed.

Table 5.5-1. Summary of Evaluation of Effects for the Pigeon-Fay and Upper Exeter Watershed

Variable	Change Downstream of Upper Pigeon?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Physical Limnology						
August Secchi Depths	No	-	-	-	No	-
Water Quality						
pH	No	-	-	-	No	All 2014 values within the CCME guideline range.
Total Alkalinity	Yes	Fay Bay (only ice-covered season)	Increase	May be related to unplanned release of FPK	Possible	-
Water Hardness	Yes	Fay Bay; Upper Exeter (only open water season)	Increase	May be related to unplanned release of FPK	Possible	
Chloride	Yes	Fay Bay	Increase	May be related to unplanned release of FPK	Possible	All 2014 concentrations less than the SSWQO.
Sulphate	Yes	Fay Bay (only ice-covered season)	Increase	Unclear	Possible	All 2014 concentrations less than the SSWQO.
Potassium	Yes	Fay Bay (only ice-covered season)	Increase	May be related to unplanned release of FPK	Possible	All 2014 concentrations less than the SSWQO.
Total Suspended Solids	No	-	-	-	No	All 2014 concentrations less than the CCME guideline.
Total Ammonia-N	No	-	-	-	No	All 2014 concentrations less than the CCME guideline.
Nitrite-N	No	-	-	-	No	All 2014 concentrations less than the CCME guideline.
Nitrate-N	No	-	-	-	No	All 2014 concentrations less than the SSWQO.
Total Phosphate-P	No	-	-	-	No	All 2014 concentrations less than the benchmark.
TOC	Yes	Fay Bay	Increase	Unclear	Possible	-

(continued)

Table 5.5-1. Summary of Evaluation of Effects for the Pigeon-Fay and Upper Exeter Watershed (continued)

Variable	Change Downstream of Upper Pigeon?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Water Quality (*cont'd*)						
Total Antimony	No	-	-	-	No	All 2014 concentrations less than the benchmark.
Total Arsenic	No	-	-	-	No	All 2014 concentrations less than the CCME guideline.
Total Barium	Yes	Fay Bay	Increase	May be related to unplanned release of FPK	Possible	All 2014 concentrations less than the benchmark.
Total Boron	No	-	-	-	No	All 2014 concentrations less than the benchmark.
Total Cadmium	No	-	-	-	No	All 2014 concentrations less than the CCME guideline.
Total Molybdenum	No	-	-	-	No	All 2014 concentrations less than the SSWQO.
Total Nickel	Yes	Fay Bay	Increase	May be related to unplanned release of FPK	Possible	All 2014 concentrations less than the CCME guideline.
Total Selenium	No	-	-	-	No	All 2014 concentrations less than the CCME guideline.
Total Strontium	Yes	Fay Bay	Increase	May be related to unplanned release of FPK	Possible	All 2014 concentrations less than the benchmark.
Total Uranium	No	-	-	-	No	All 2014 concentrations less than the CCME guideline.
Total Vanadium	No	-	-	-	No	All 2014 concentrations less than the SSWQO.
Sediment Quality						
TOC	No	-	-	-	No	-
Available Phosphorus	No	-	-	-	No	-
Total Nitrogen	No	-	-	-	No	-
Total Antimony	No	-	-	-	No	-

(continued)

Table 5.5-1. Summary of Evaluation of Effects for the Pigeon-Fay and Upper Exeter Watershed (completed)

Variable	Change Downstream of Upper Pigeon?	Locations Change Detected	Direction of Change	Source of Change	Mine Effect?	Notes
Sediment Quality (*cont'd*)						
Total Arsenic	No	-	-	-	No	The 95% confidence intervals of the fitted mean in Fay Bay and the observed mean in Upper Exeter Lake exceeded the CCME ISQG and PEL in 2014; however, similar patterns were observed in reference lakes.
Total Cadmium	No	-	-	-	No	All 2014 concentrations less than the CCME ISQG and PEL.
Total Molybdenum	No	-	-	-	No	-
Total Nickel	Yes	Upper Exeter Lake	Increase	Unclear	Unlikely	-
Total Phosphorus	No	-	-	-	No	-
Total Selenium	No	-	-	-	No	-
Total Strontium	No	-	-	-	No	-
Phytoplankton						
Chlorophyll *a*	Yes	Fay Bay	Increase	Unknown	Unlikely	-
Density	Yes	Fay Bay	Increase	Unknown	Unlikely	-
Diversity	Possible	Upper Exeter	Increase	Unknown	Unlikely	No clear trend through time, but mean Shannon diversity in Upper Exeter Lake was greater than the ± 2 SD of mean baseline diversity
Relative Densities of Major Taxa	Yes	Fay Bay, Upper Exeter	(see Notes column)	Unknown	Unlikely	Increase in relative abundance of Myxophyceae in Fay Bay; Increase in relative abundance of Chlorophyceae and Chrysophyceae in Upper Exeter.

Notes: Dashes indicate not applicable.
Comparisons to CCME guidelines are for 2014 data only.
DO = dissolved oxygen
CCME = Canadian Council of Ministers of the Environment
SSWQO = Site Specific Water Quality Objective

6. 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 Sauvage, using samples collected in April and August of each year (see Sections 2.2, 3.2, and 4.2). However, lake water quality samples are collected and screened for 47 water quality variables in the laboratory (Table 6-1). Also, prior to 2010 lake water quality was sampled in July and September in addition to the April and August sampling. Historical averages for 46 of the water quality variables (excludes ion balance) for the three reference lakes (Nanuq, Counts, and Vulture) and each of the monitored lakes in the Koala and King-Cujo Watersheds as well as Lac de Gras and Lac du Sauvage are presented below (Figures 6-1 to 6-46).

Table 6-1. AEMP Water Quality Variables in the Koala and King-Cujo Watersheds

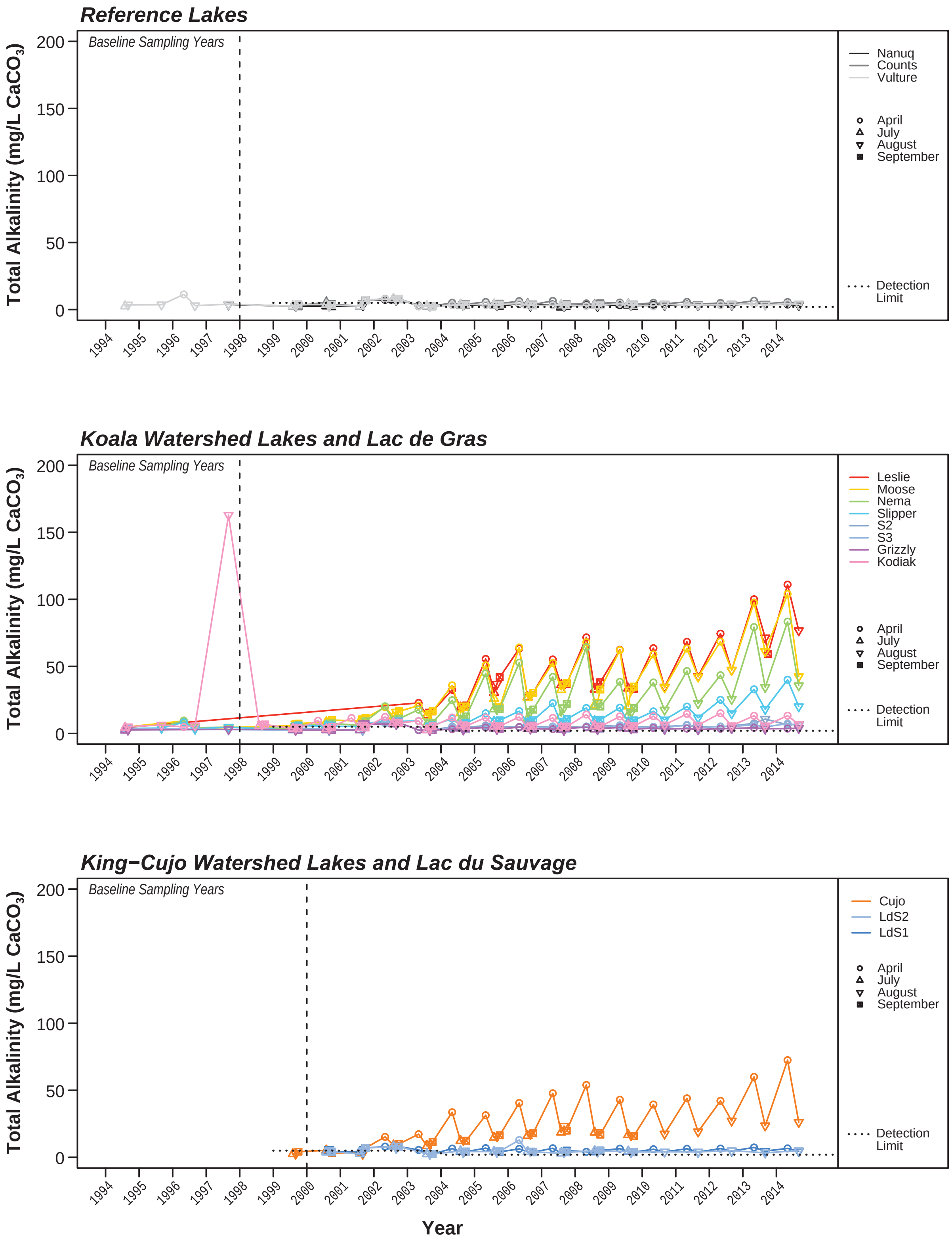
Variable	Figure Number	Variable	Figure Number
Physical/Ion		**Total Metals**	
Total Alkalinity	6-1	Aluminum	6-22
Bicarbonate	6-2	Antimony	6-23
Carbonate	6-3	Arsenic	6-24
Conductivity	6-4	Barium	6-25
Hydroxide	6-5	Beryllium	6-26
pH	6-6	Boron	6-27
Chloride	6-7	Cadmium	6-28
Potassium	6-8	Calcium	6-29
Total Silicon	6-9	Chromium	6-30
Sulphate	6-10	Cobalt	6-31
Total Suspended Solids	6-11	Copper	6-32
Turbidity	6-12	Iron	6-33
Hardness	6-13	Lead	6-34
Ion Balance	Not Shown	Magnesium	6-35
Total Dissolved Solids	6-14	Manganese	6-36
Nutrients/Organics		Mercury	6-37
Total Ammonia-N	6-15	Molybdenum	6-38
Nitrate-N	6-16	Nickel	6-39
Nitrite-N	6-17	Selenium	6-40
Orthophosphate	6-18	Silver	6-41
Total Phosphate-P	6-19	Sodium	6-42
Total Organic Carbon	6-20	Strontium	6-43
Total Kjeldahl Nitrogen	6-21	Uranium	6-44
		Vanadium	6-45
		Zinc	6-46

Starting in 2014, the AEMP Evaluation of Effects also focused on detecting changes in 23 lake water quality variables in the Pigeon-Fay and Upper Exeter Watershed, using samples collected throughout the ice-covered and open water seasons (see Sections 2.2 and 5.2). However, lake water quality samples are collected and screened for 47 water quality variables in the laboratory (Table 6-2). Also, prior to 2010 lake water quality was sampled in July and September in addition to the April and August sampling. Historical averages for 46 of the water quality variables (excludes ion balance) for the three reference lakes (Nanuq, Counts, and Vulture) and each of the monitored lakes in the Koala and King-Cujo Watersheds are presented below (Figures 6-47 to 6-92).

Table 6-2. AEMP Water Quality Variables for the Pigeon-Fay and Upper Exeter Watershed

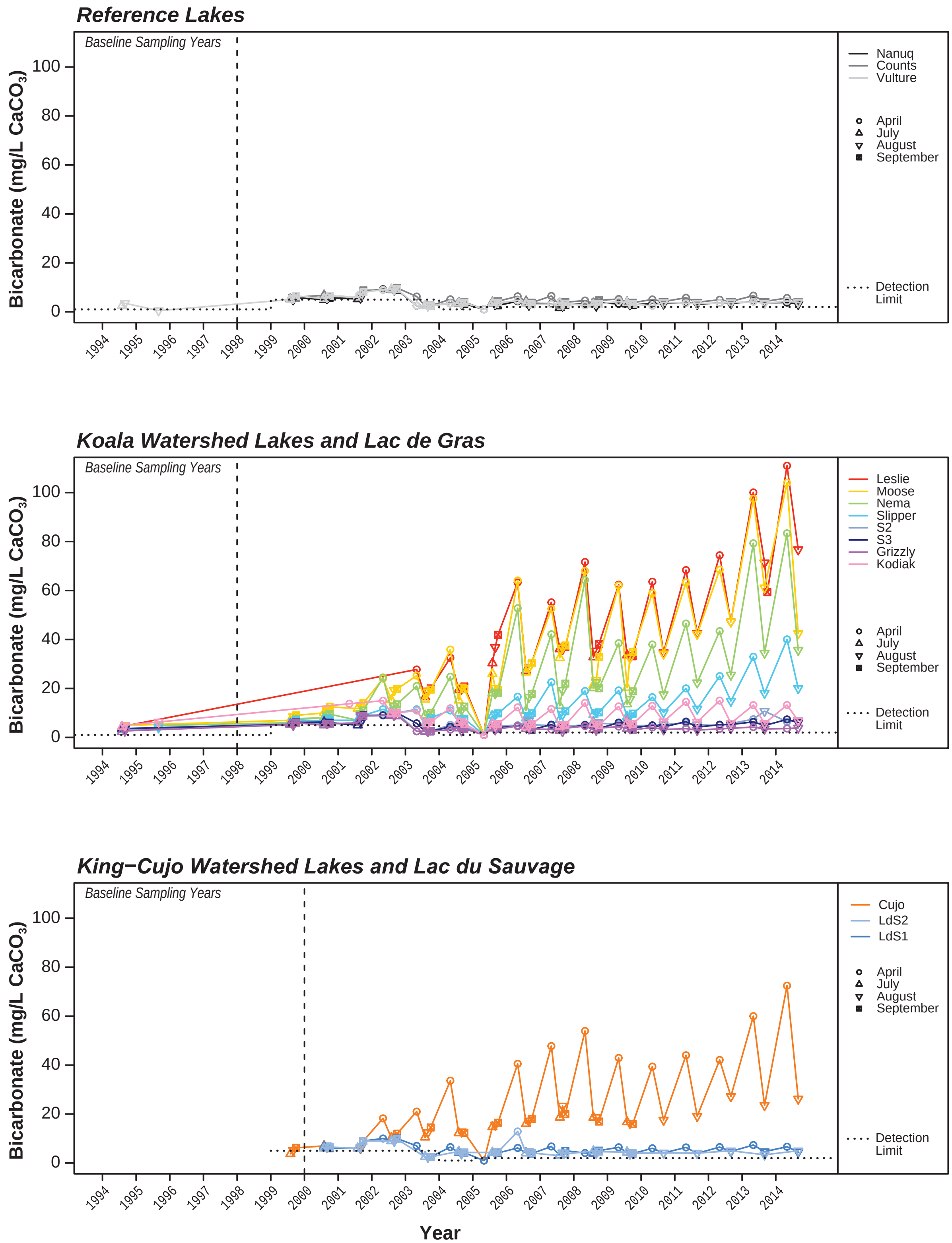
Variable	Figure Number	Variable	Figure Number
Physical/Ion		**Total Metals**	
Total Alkalinity	6-47	Aluminum	6-68
Bicarbonate	6-48	Antimony	6-69
Carbonate	6-49	Arsenic	6-70
Conductivity	6-50	Barium	6-71
Hydroxide	6-51	Beryllium	6-72
pH	6-52	Boron	6-73
Chloride	6-53	Cadmium	6-74
Potassium	6-54	Calcium	6-75
Total Silicon	6-55	Chromium	6-76
Sulphate	6-56	Cobalt	6-77
Total Suspended Solids	6-57	Copper	6-78
Turbidity	6-58	Iron	6-79
Hardness	6-59	Lead	6-80
Ion Balance	Not Shown	Magnesium	6-81
Total Dissolved Solids	6-60	Manganese	6-82
Nutrients/Organics		Mercury	6-83
Total Ammonia-N	6-61	Molybdenum	6-84
Nitrate-N	6-62	Nickel	6-85
Nitrite-N	6-63	Selenium	6-86
Orthophosphate	6-64	Silver	6-87
Total Phosphate-P	6-65	Sodium	6-88
Total Organic Carbon	6-66	Strontium	6-89
Total Kjeldahl Nitrogen	6-67	Uranium	6-90
		Vanadium	6-91
		Zinc	6-92

CCME water quality guidelines for the protection of aquatic life are provided where applicable (CCME 2014c). 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 (see Section 2.3). Analytical detection limits are also included on the figures, with the lowest detection limit presented in cases where detection limits varied between lakes and months within the same year.



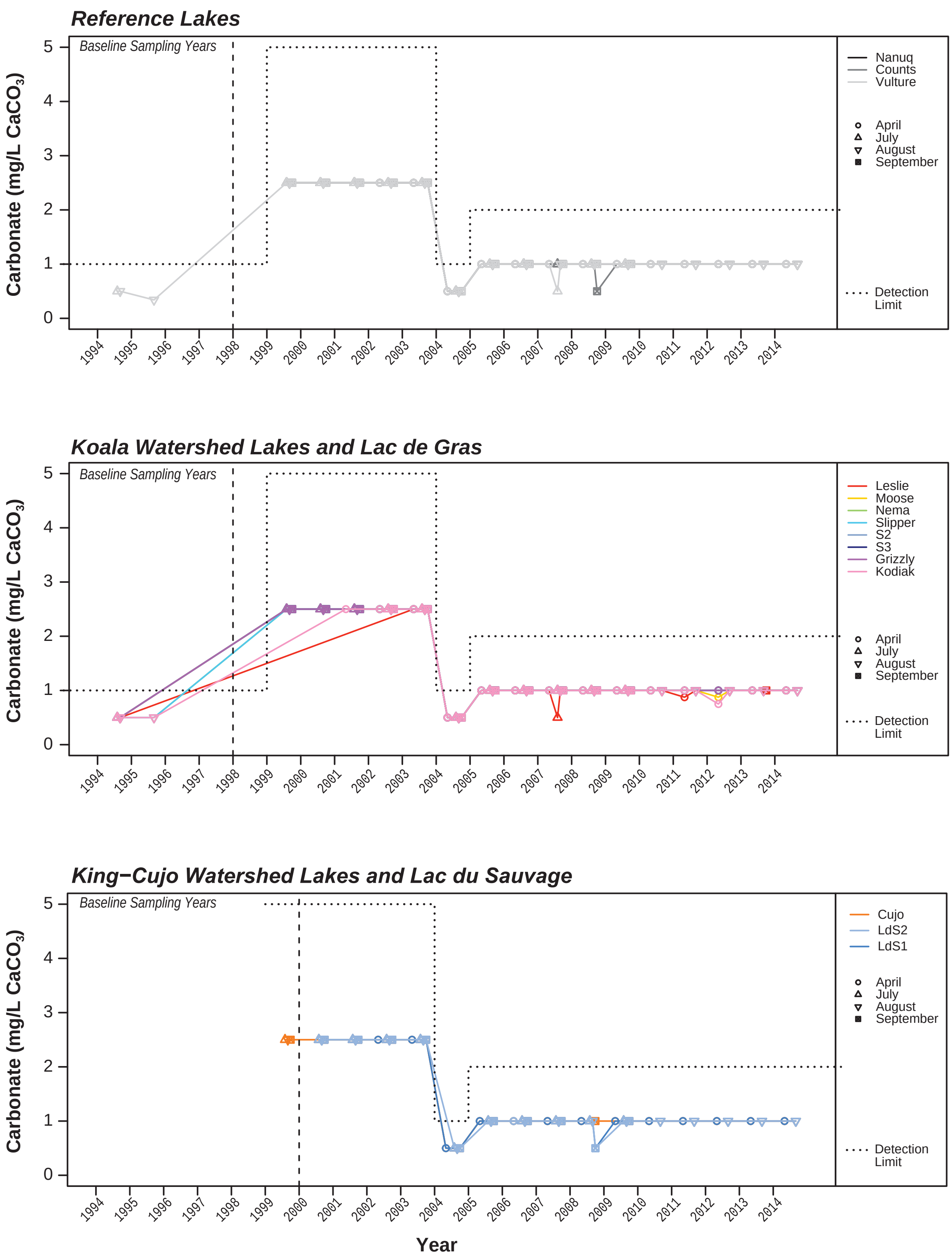
Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Bicarbonate Concentrations at AEMP Lake Sites, 1994 to 2014



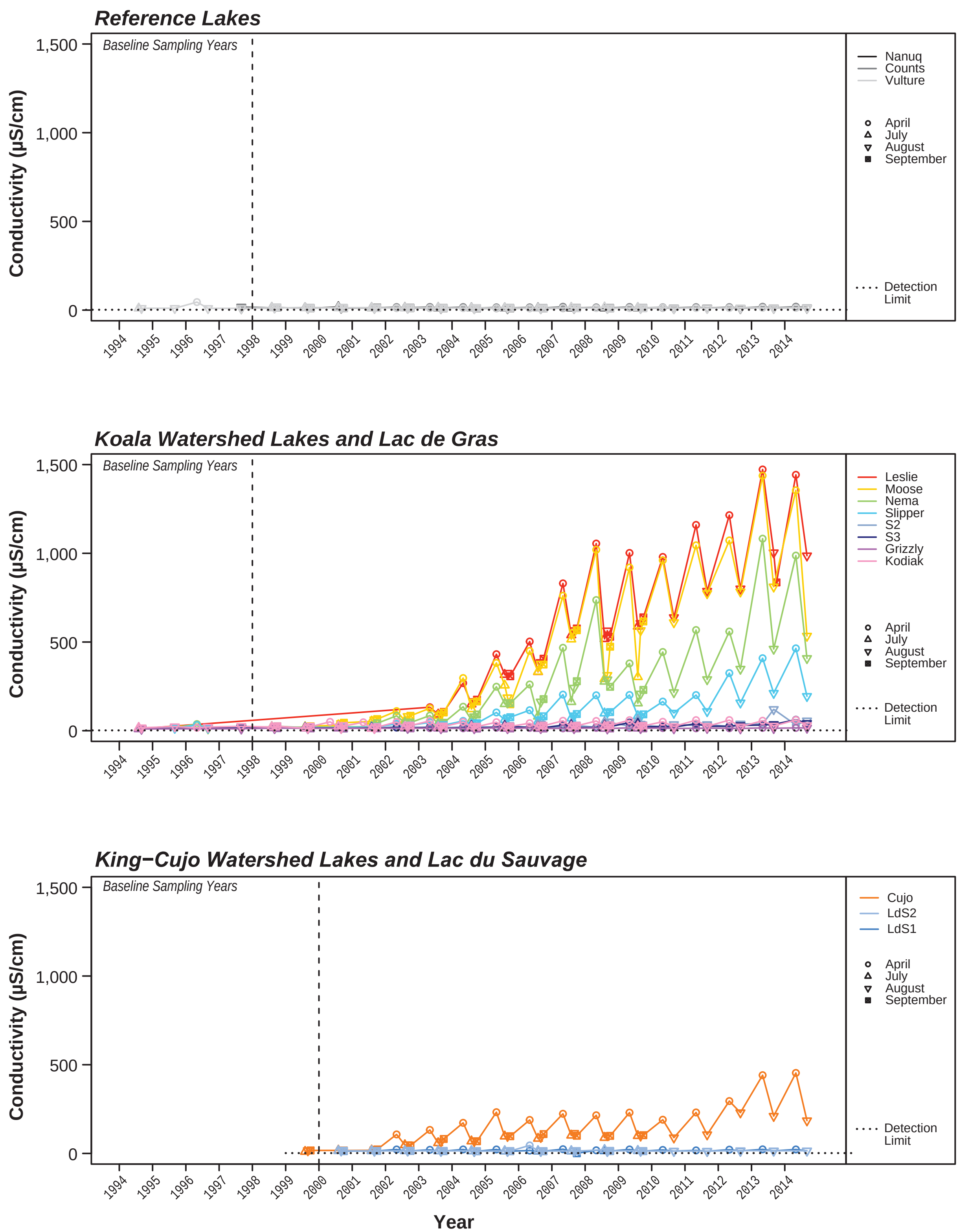
Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 6-3
Carbonate Concentrations at AEMP Lake Sites, 1994 to 2014



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

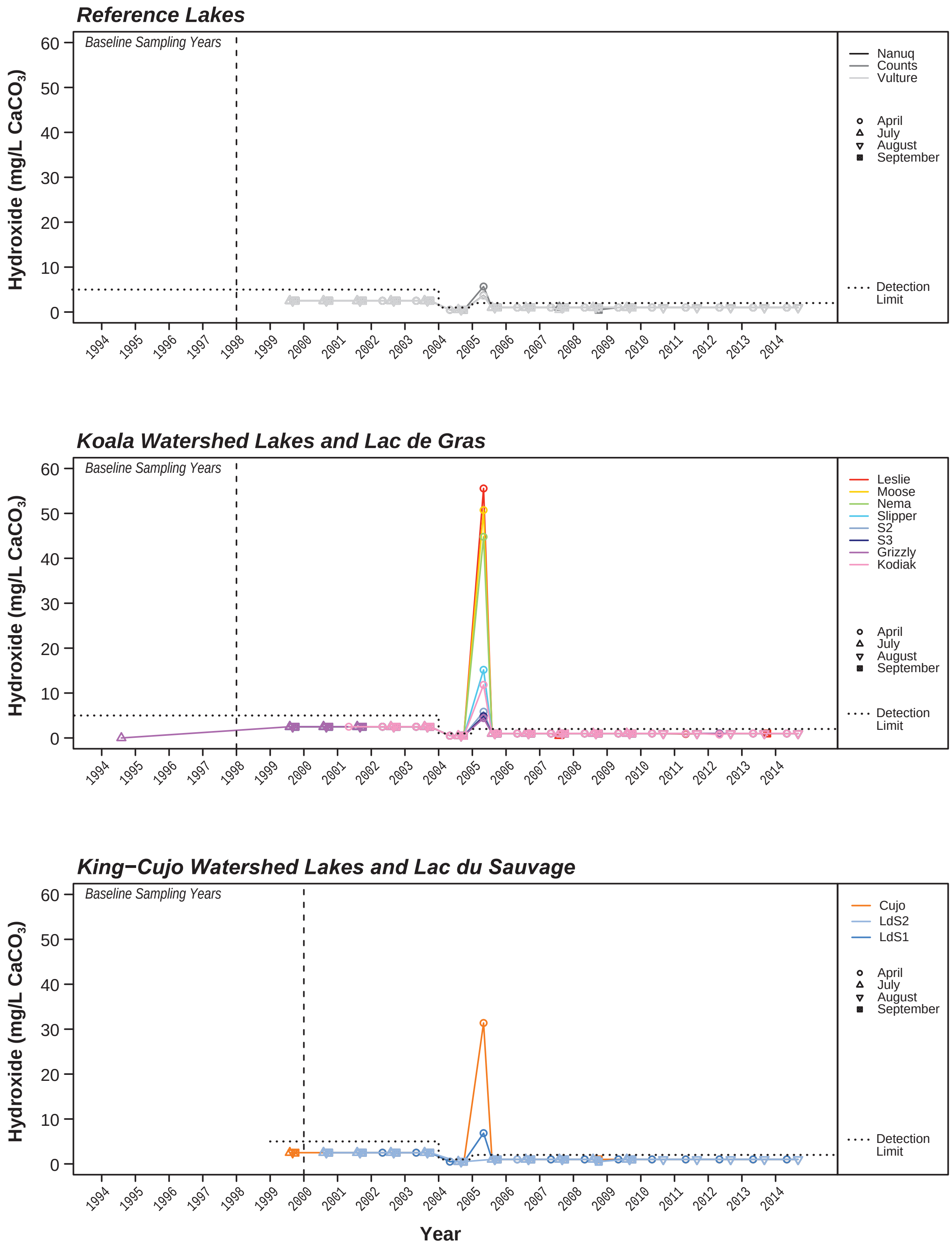
Figure 6-4
Conductivity at AEMP Lake Sites, 1994 to 2014



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

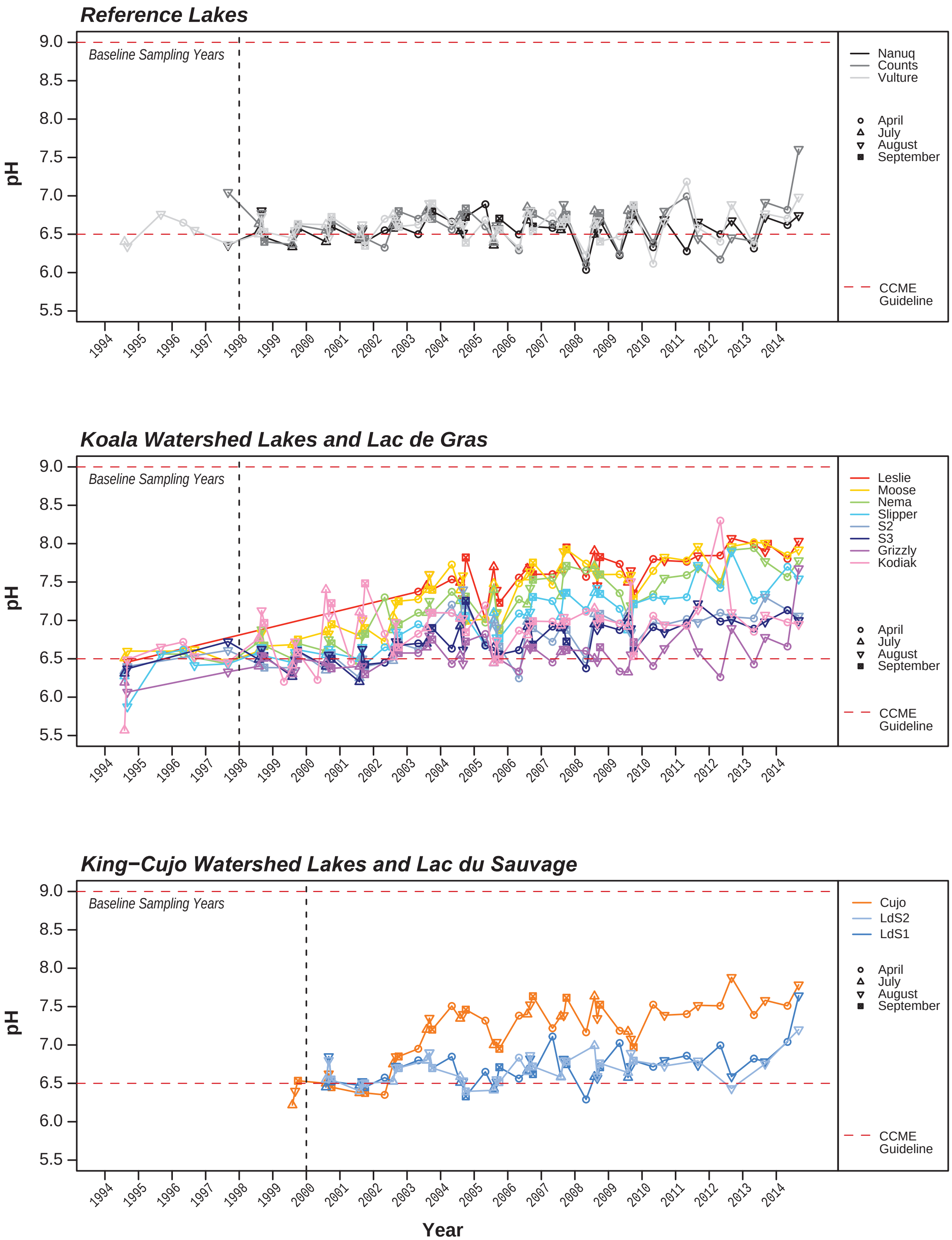
Figure 6-5

Hydroxide Concentrations at AEMP Lake Sites, 1994 to 2014



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 6-6
pH at AEMP Lake Sites, 1994 to 2014

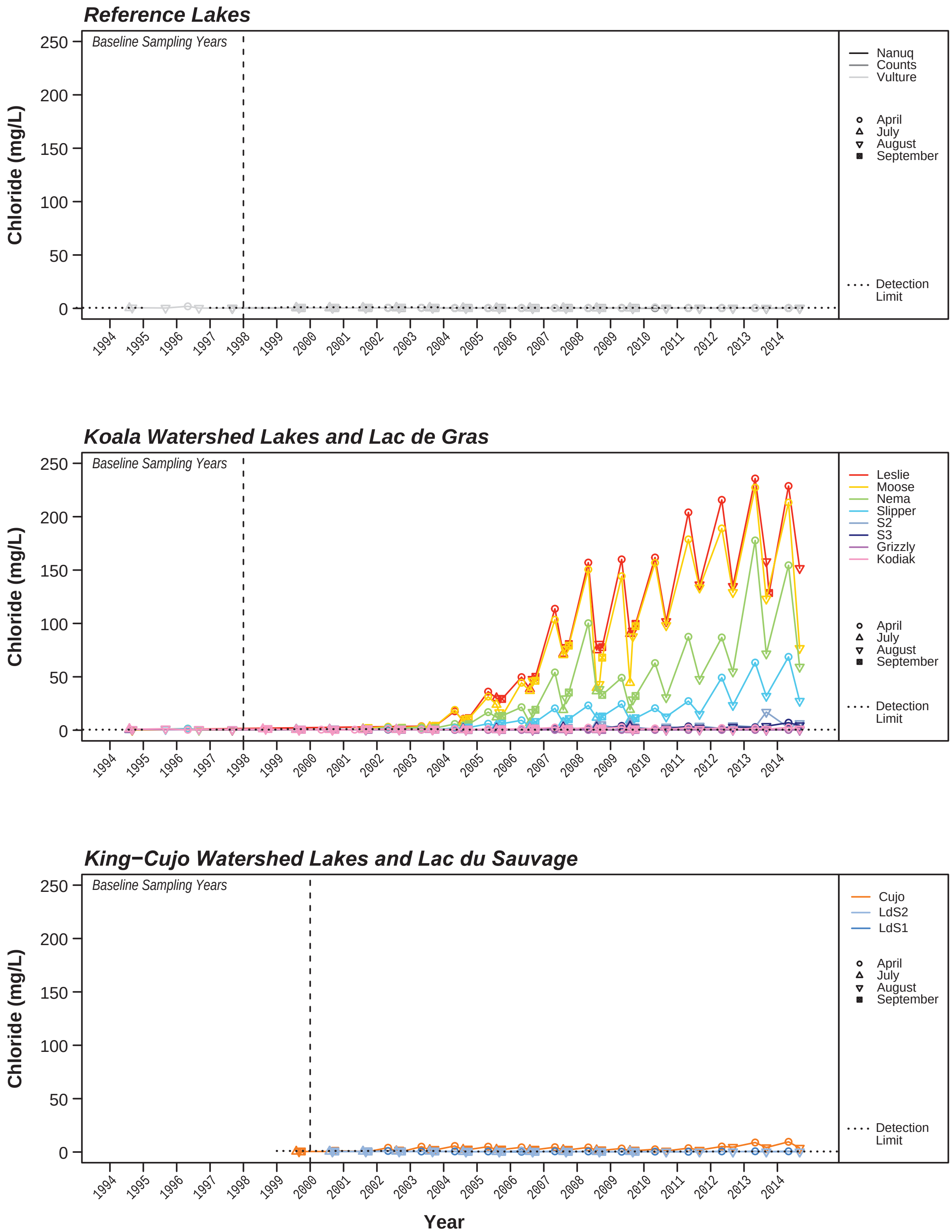


Note: CCME Guidelines are 6.5 to 9.0.

Figure 6-7

Chloride Concentrations at AEMP Lake Sites, 1994 to 2014

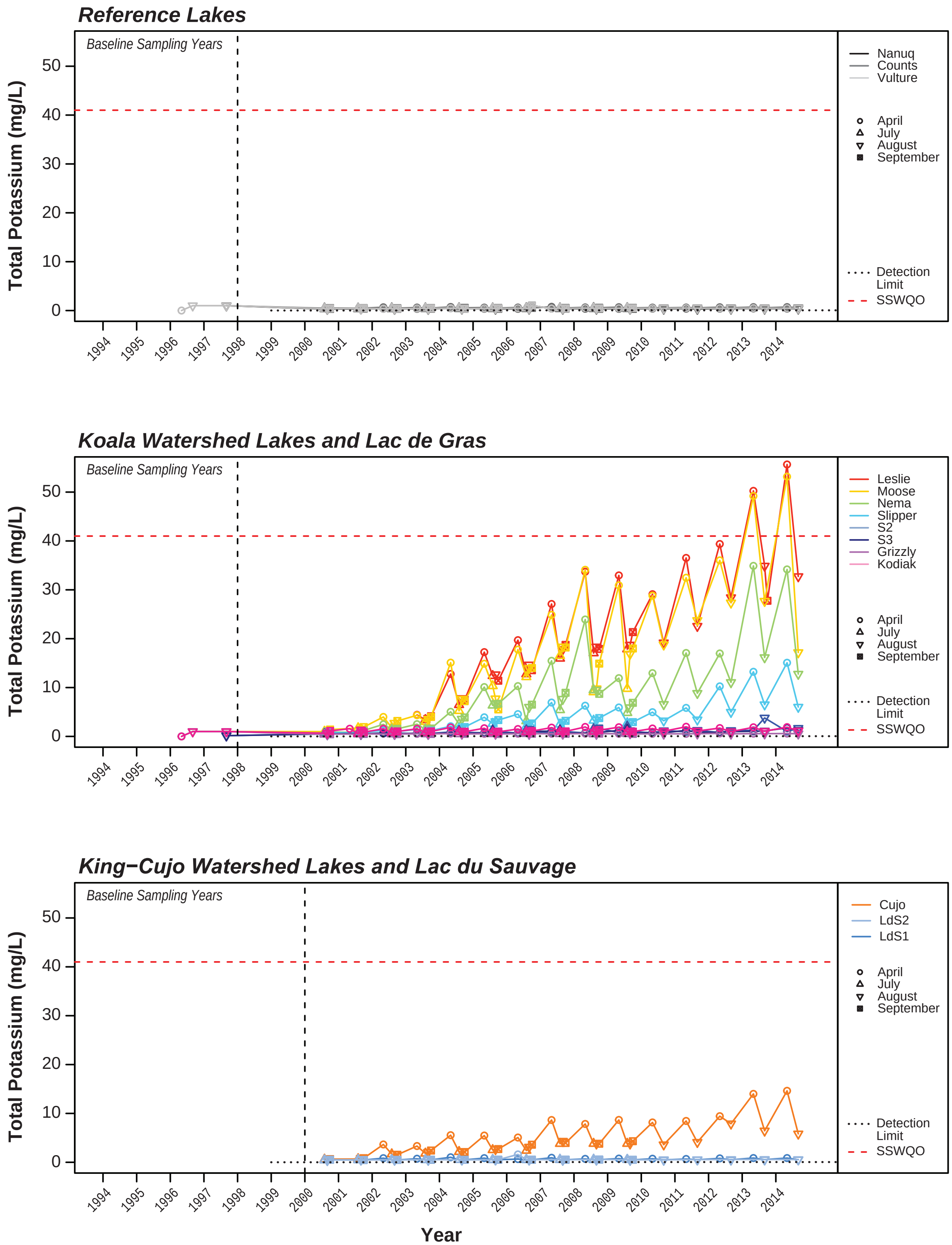
ERM



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 6-8

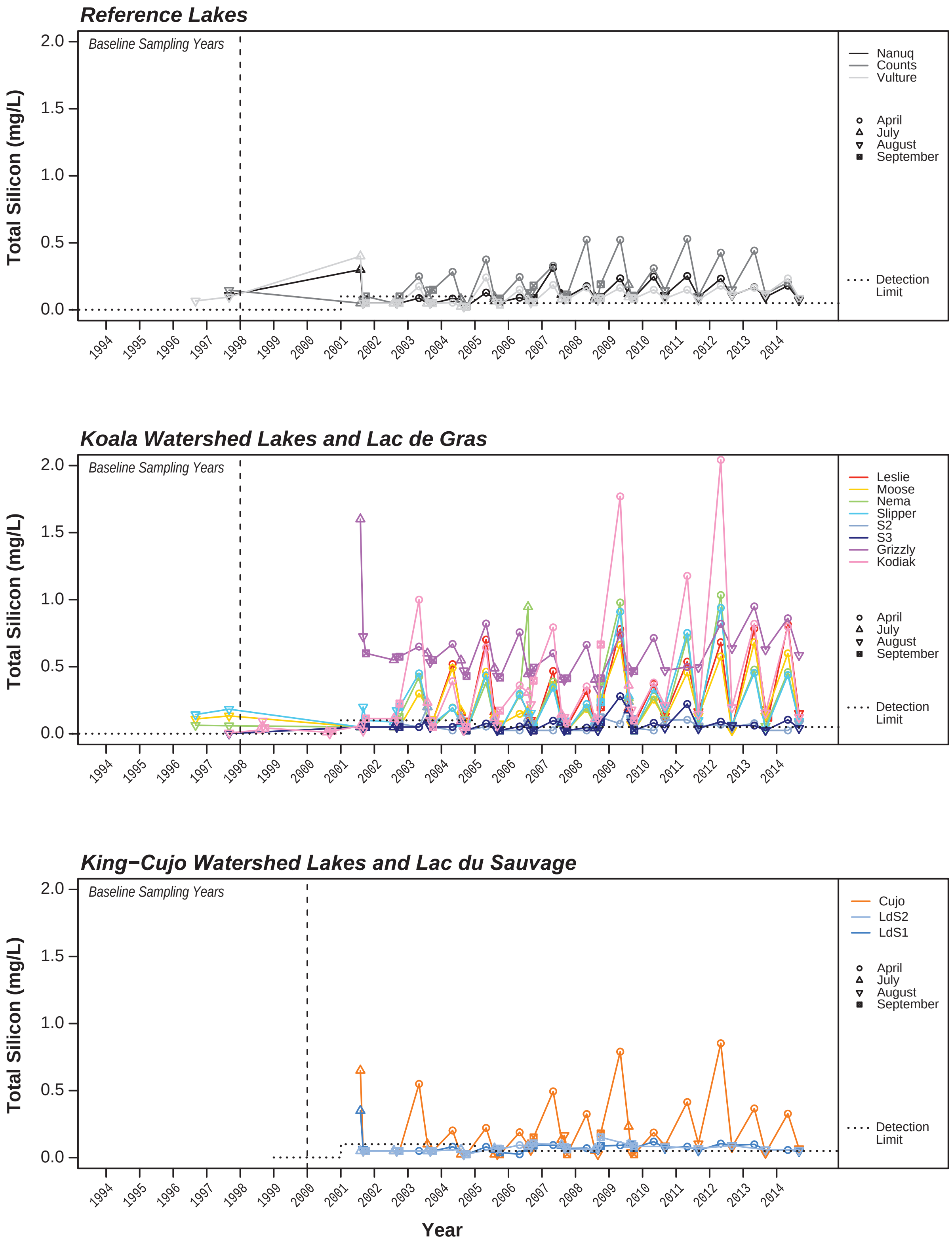
Total Potassium Concentrations at AEMP Lake Sites, 1994 to 2014



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown. SSWQO = 41 mg/L.

Figure 6-9

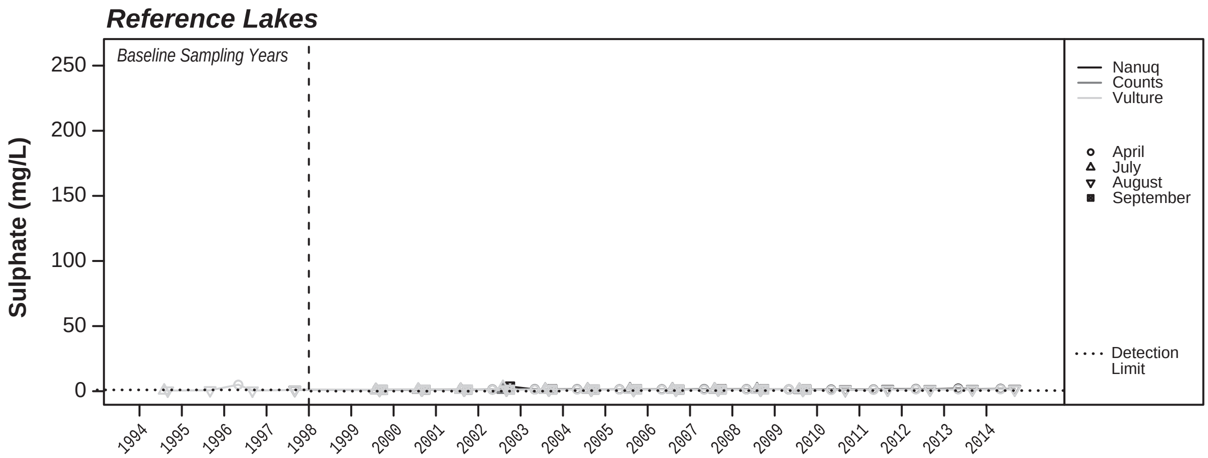
Total Silicon Concentrations at AEMP Lake Sites, 1994 to 2014

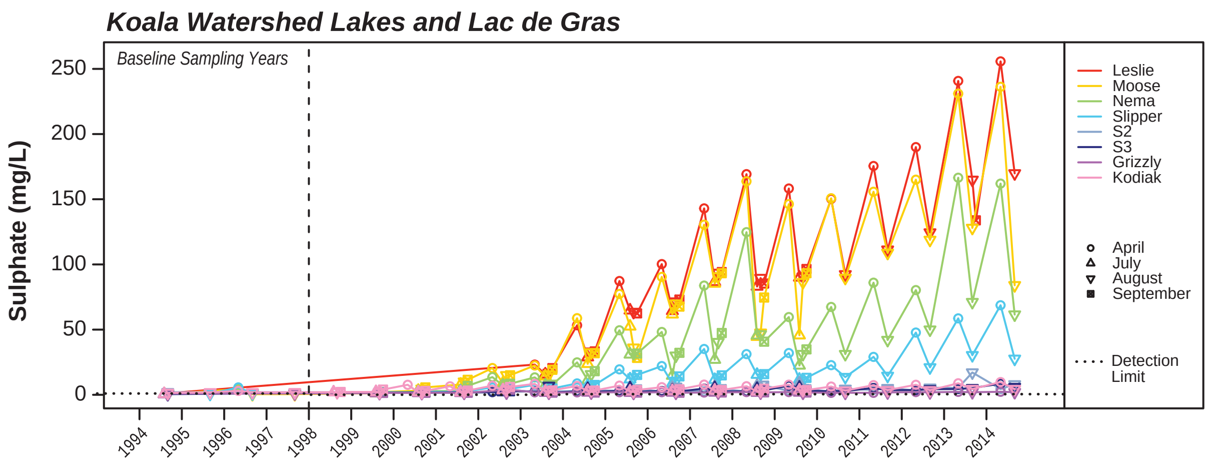


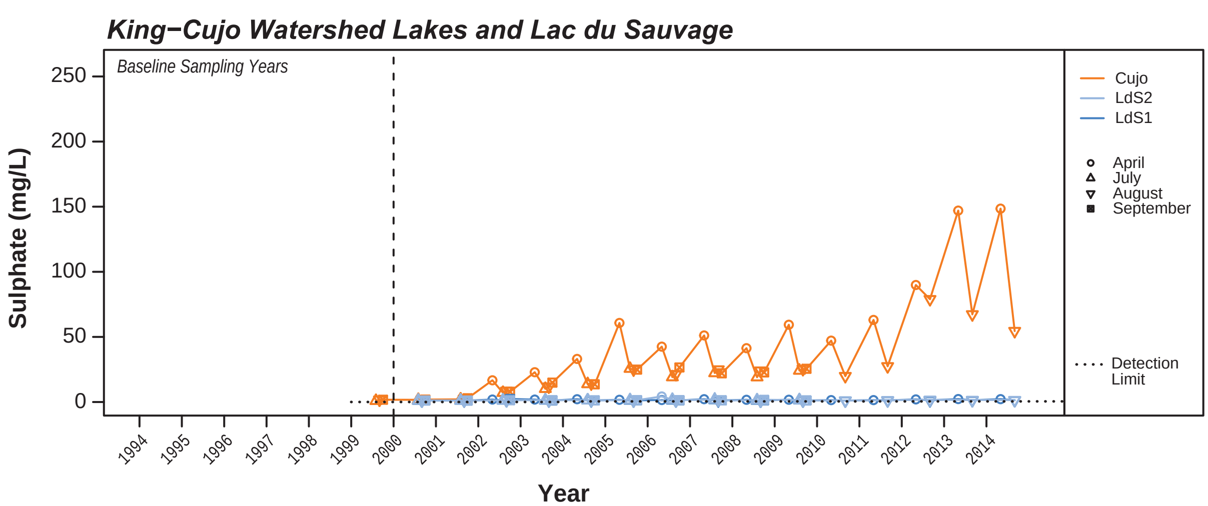
Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 6-10

Sulphate Concentrations at AEMP Lake Sites, 1994 to 2014



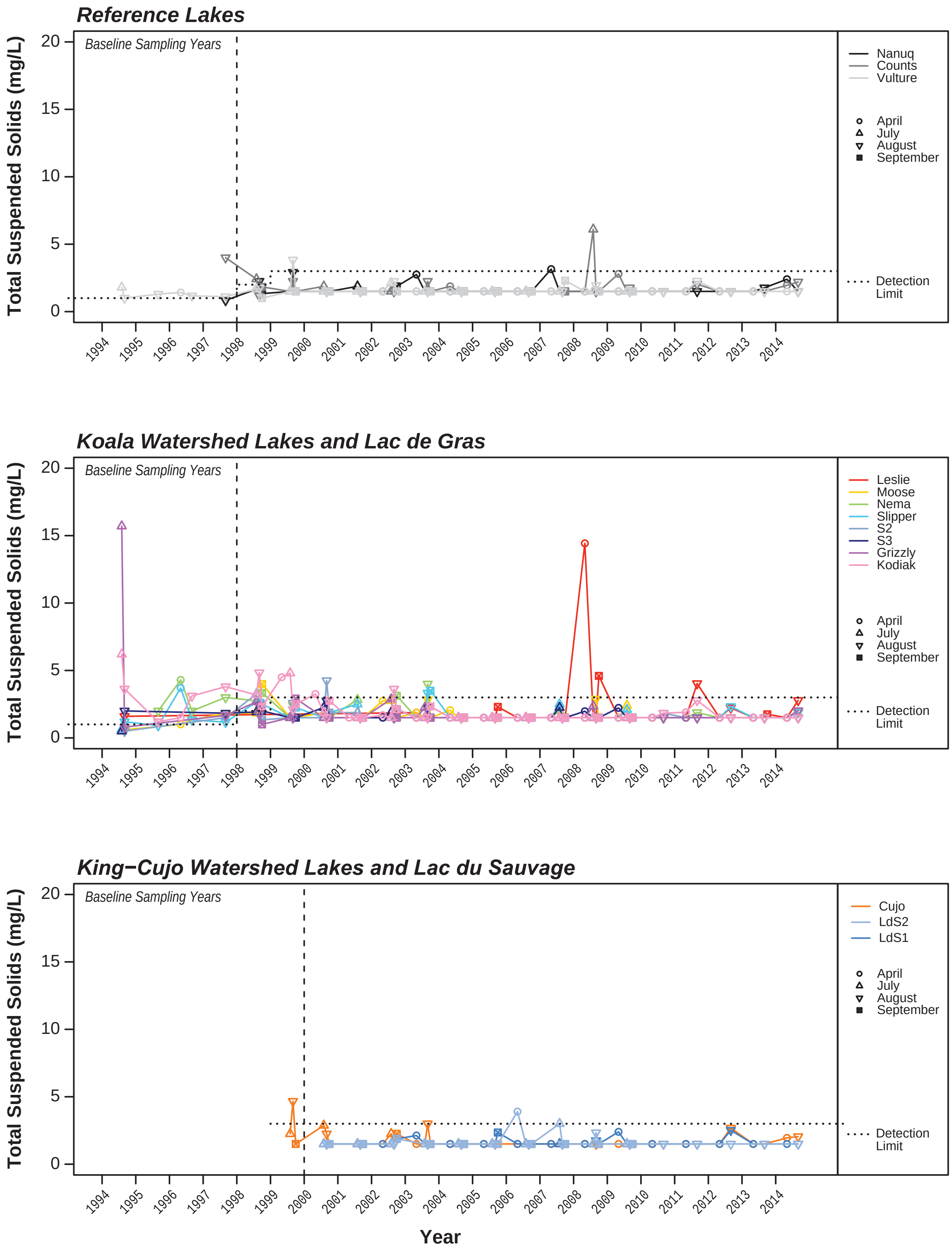




Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
SSWQO = $e^{(0.9116 \times \ln(\text{Hardness}) + 1.712)}$ mg/L, where hardness < 160 mg/L.

Figure 6-11

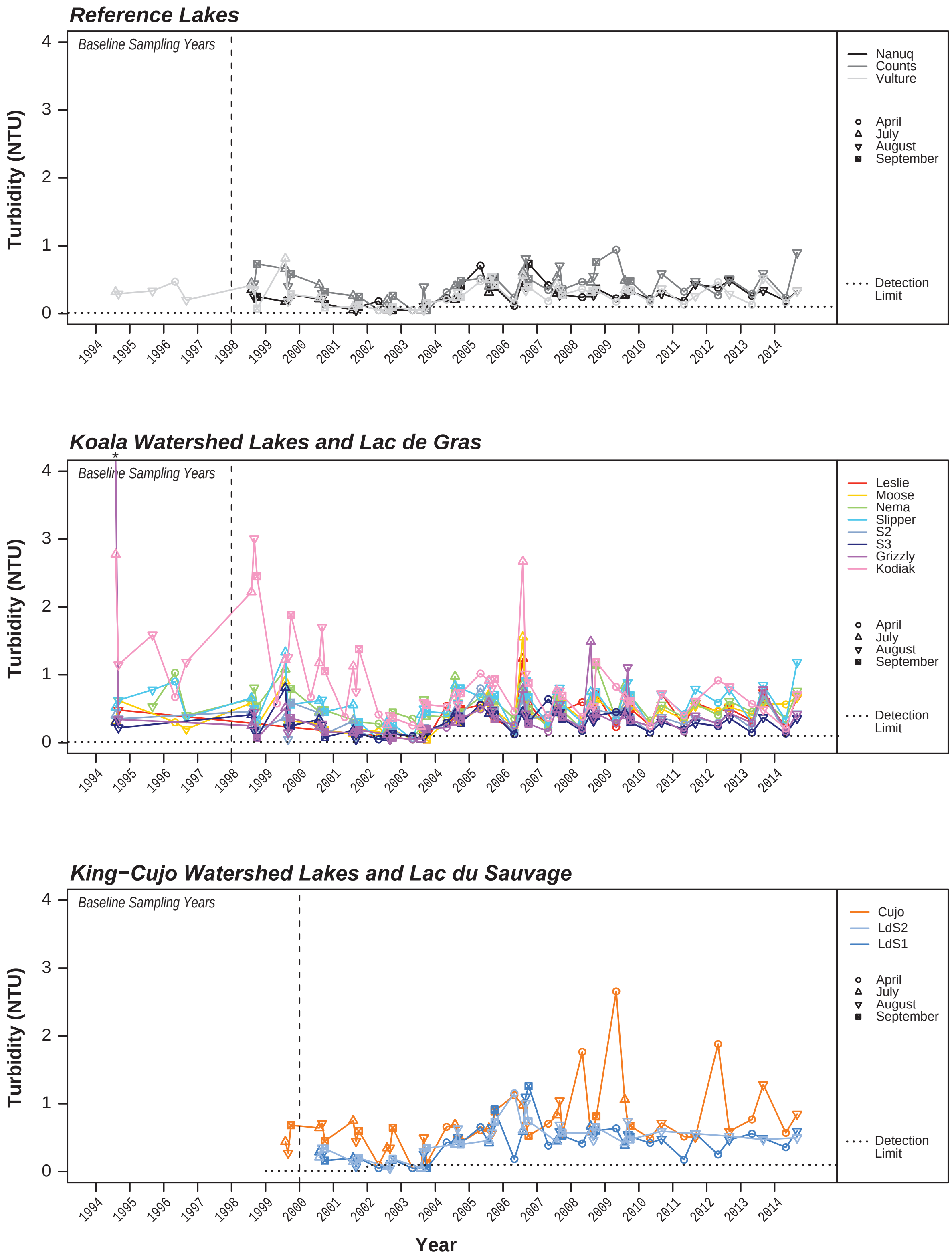
Total Suspended Solids at AEMP Sites, 1994 to 2014



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).

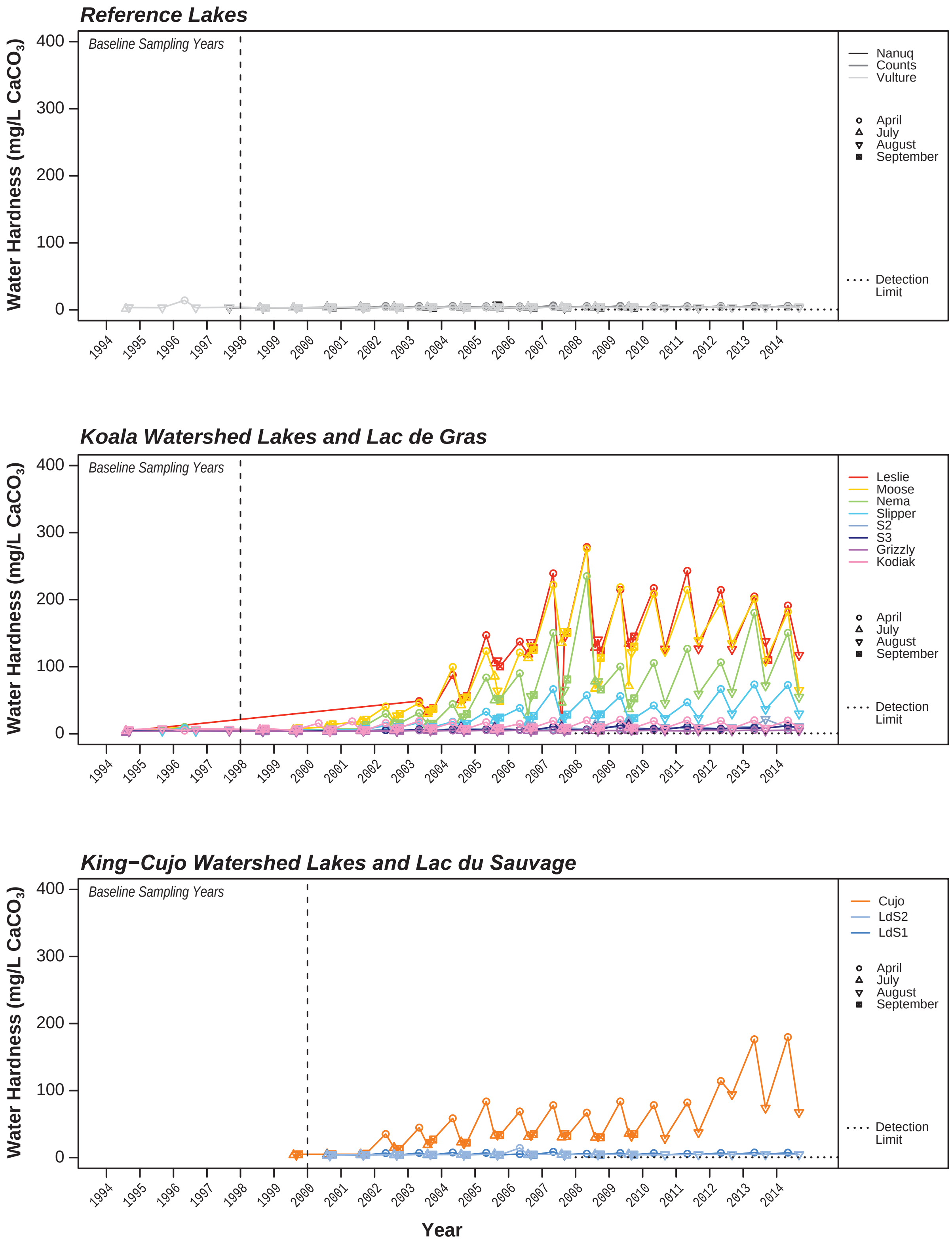
Figure 6-12

Turbidity at AEMP Lake Sites, 1994 to 2014



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).

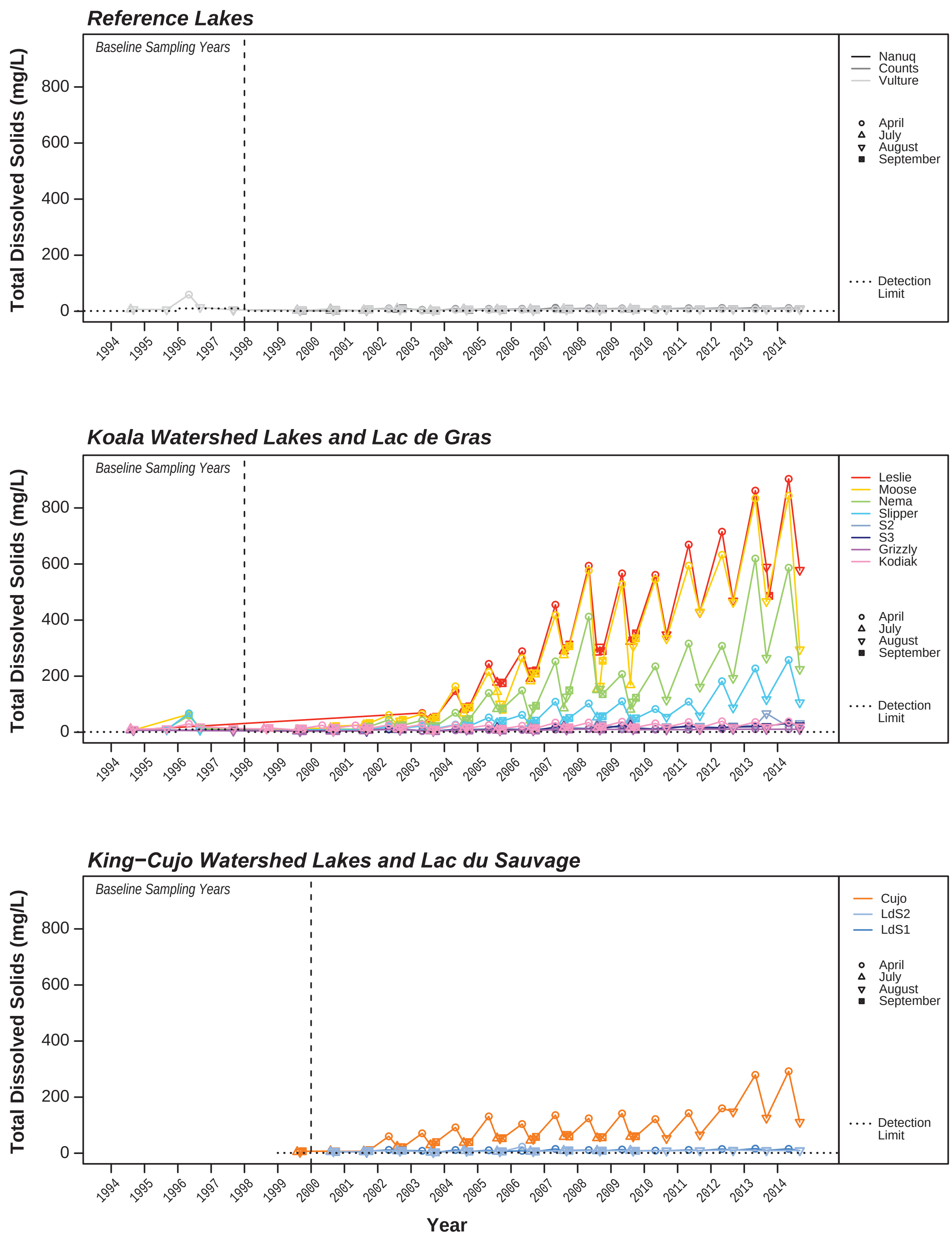
Figure 6-13
Water Hardness at AEMP Lake Sites, 1994 to 2014



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 6-14

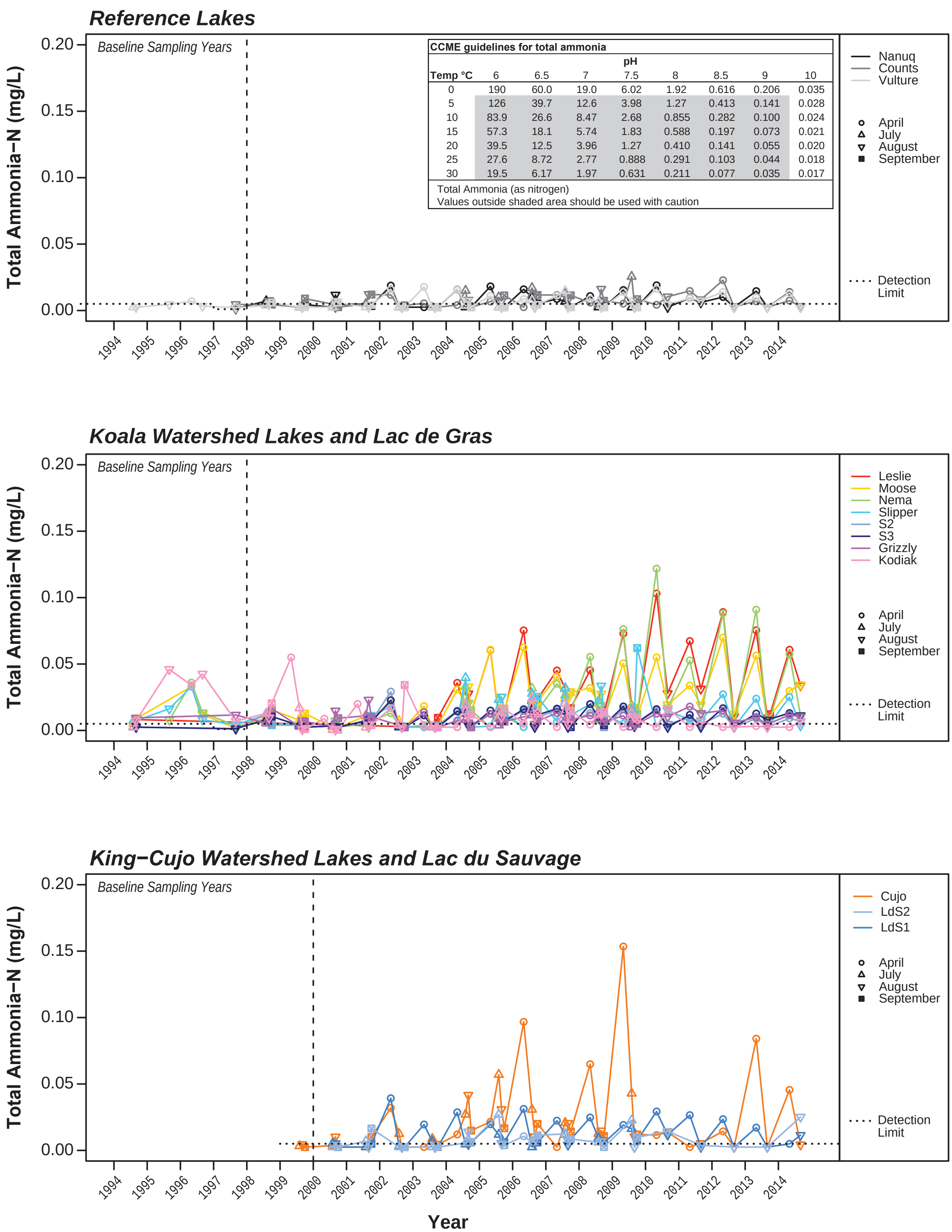
Total Dissolved Solids at AEMP Lake Sites, 1994 to 2014



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

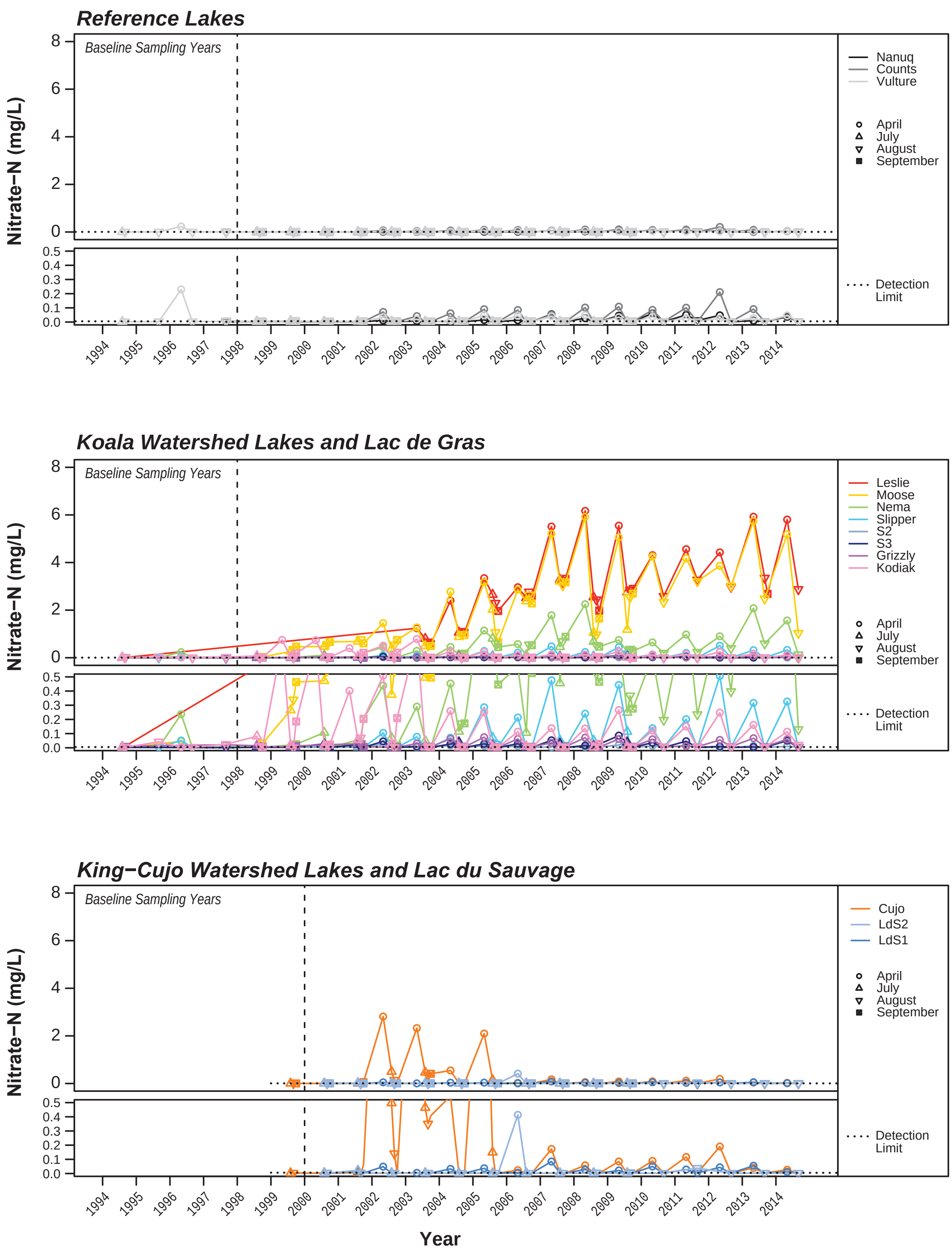
Figure 6-15

Total Ammonia-N Concentrations at AEMP Lake Sites, 1994 to 2014



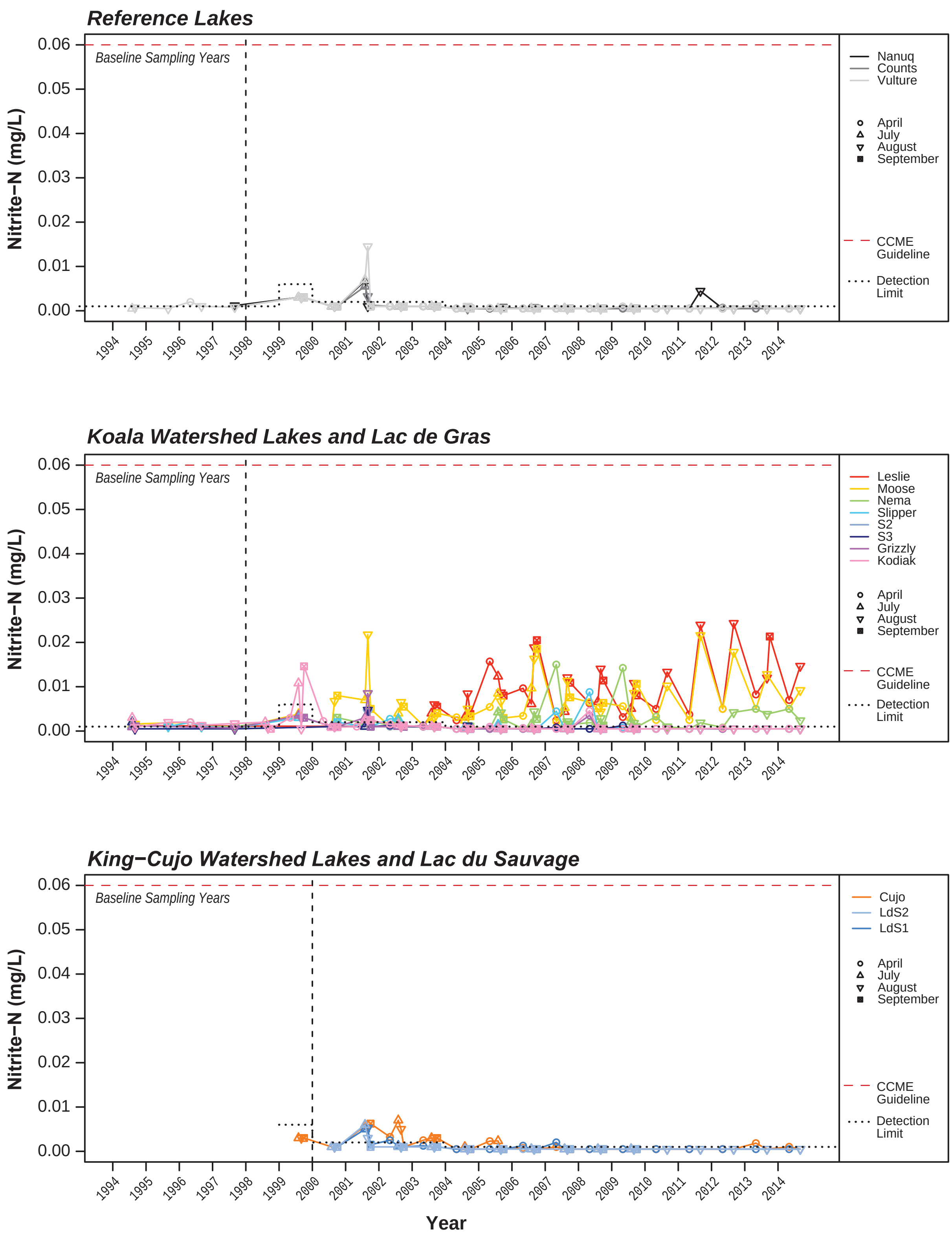
Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown. CCME guidelines are pH- and temperature-dependent (see inset).

Figure 6-16
Nitrate-N Concentrations at AEMP Lake Sites, 1994 to 2014



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
$SSWQO = e^{0.9518 \times \ln(Hardness) - 2.032}$ mg/L, where hardness < 160mg/L.

Figure 6-17
Nitrite-N Concentrations at AEMP Lake Sites, 1994 to 2014

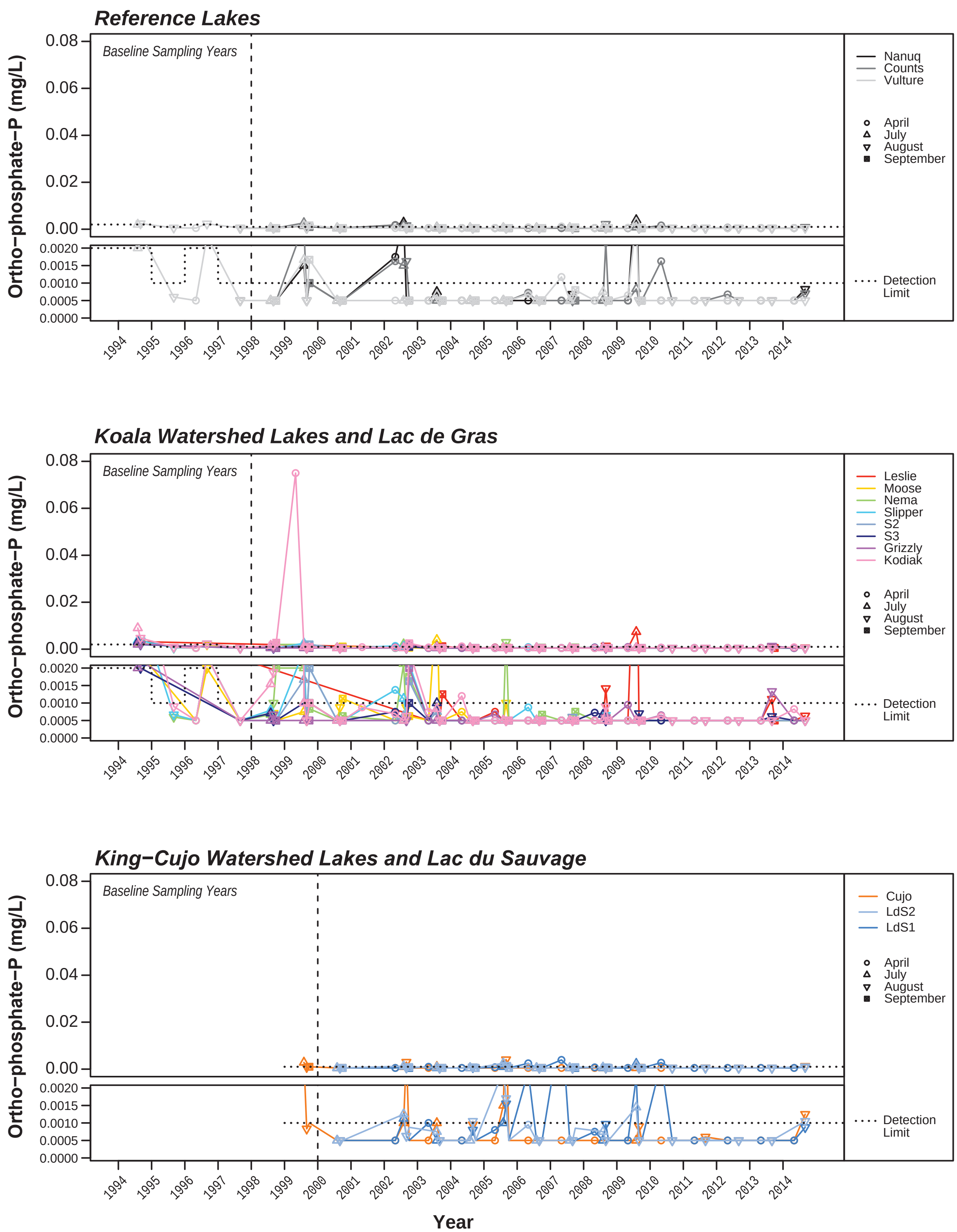


Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME Guideline = 0.06 mg/L.

Figure 6-18

Orthophosphate-P Concentrations at AEMP Lake Sites, 1994 to 2014

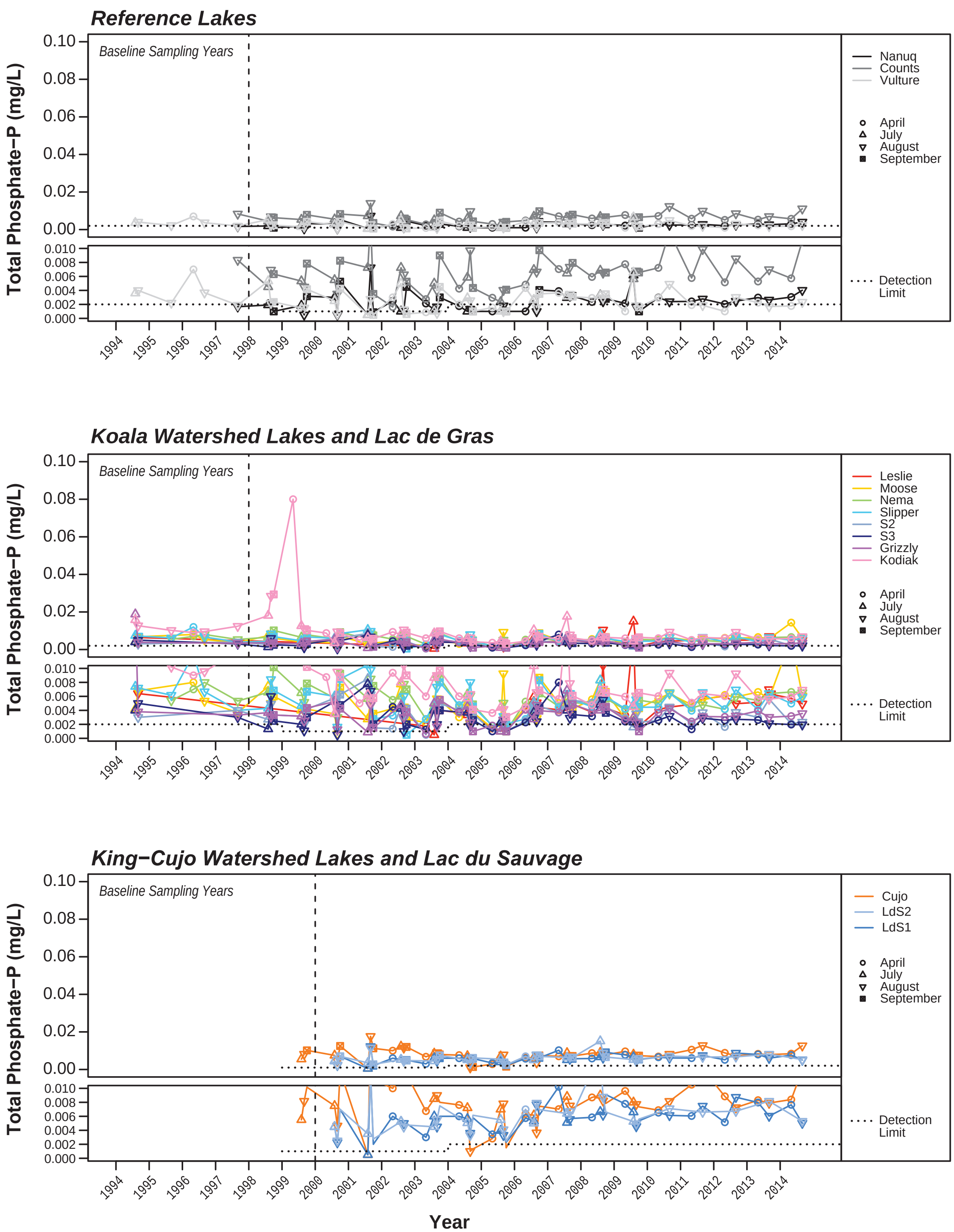
ERM



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 6-19

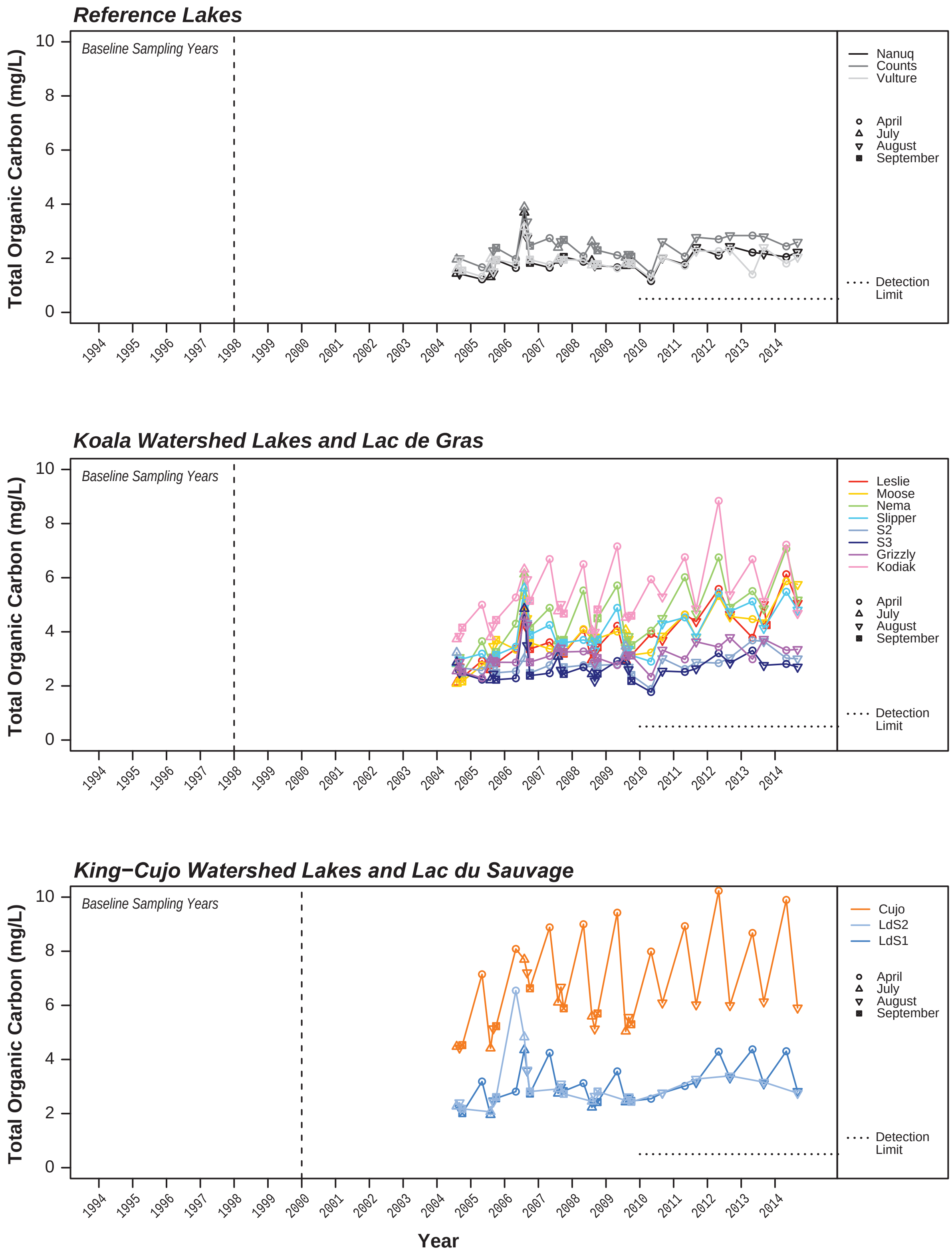
Total Phosphate-P Concentrations at AEMP Lake Sites, 1994 to 2014



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 6-20

Total Organic Carbon Concentrations at AEMP Lake Sites, 1994 to 2014

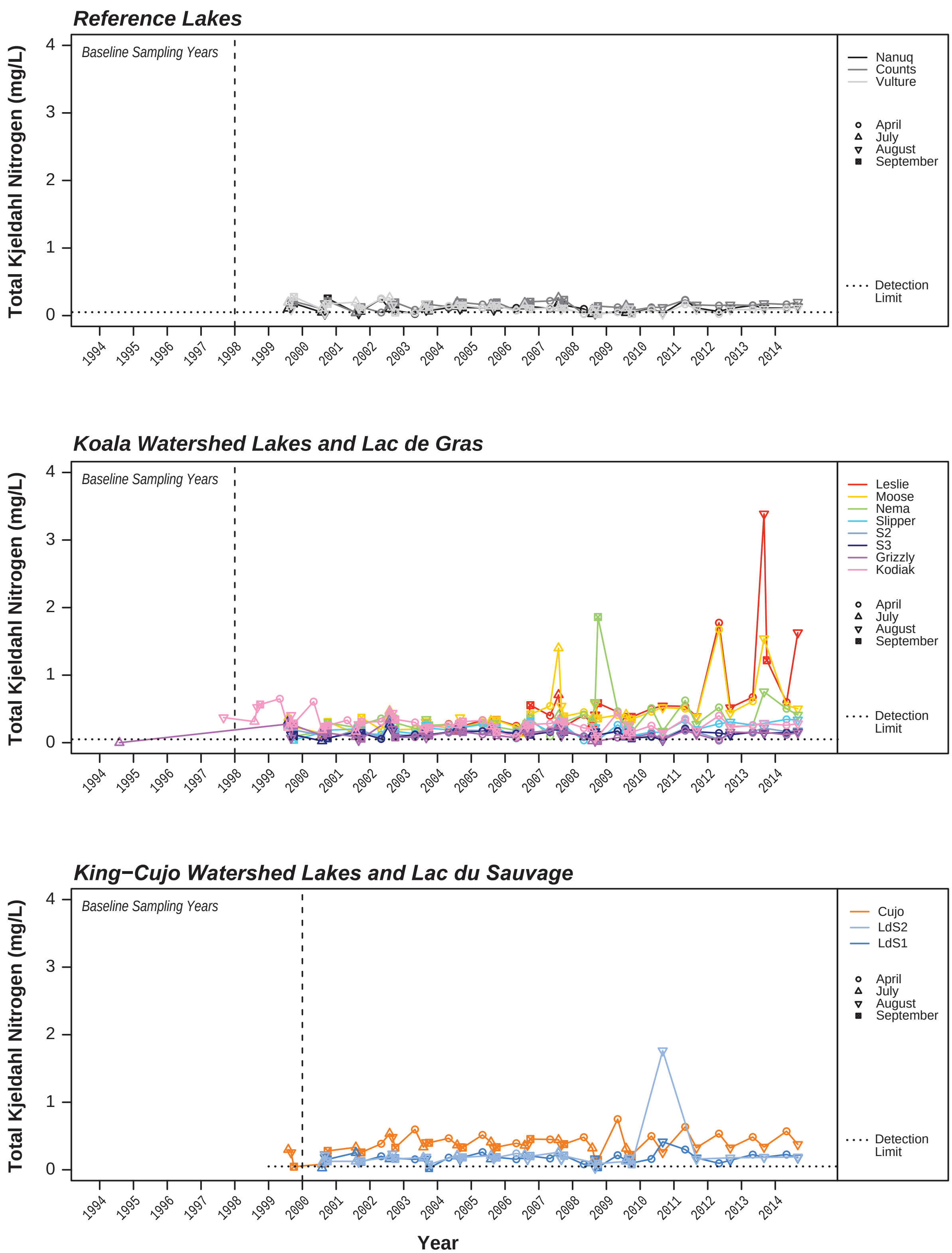


Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 6-21

Total Kjeldhal Nitrogen Concentrations at AEMP Lake Sites, 1994 to 2014

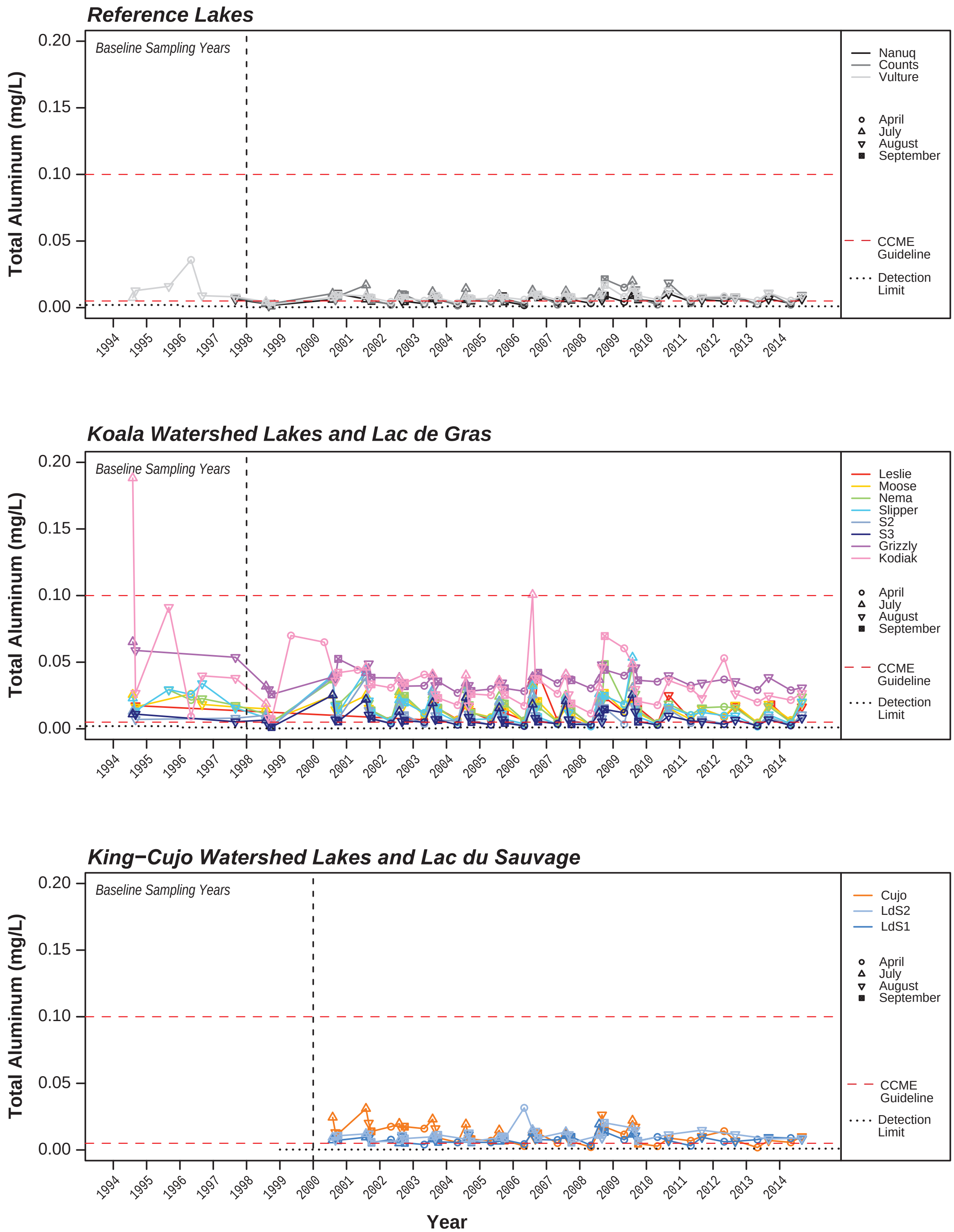
ERM



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 6-22

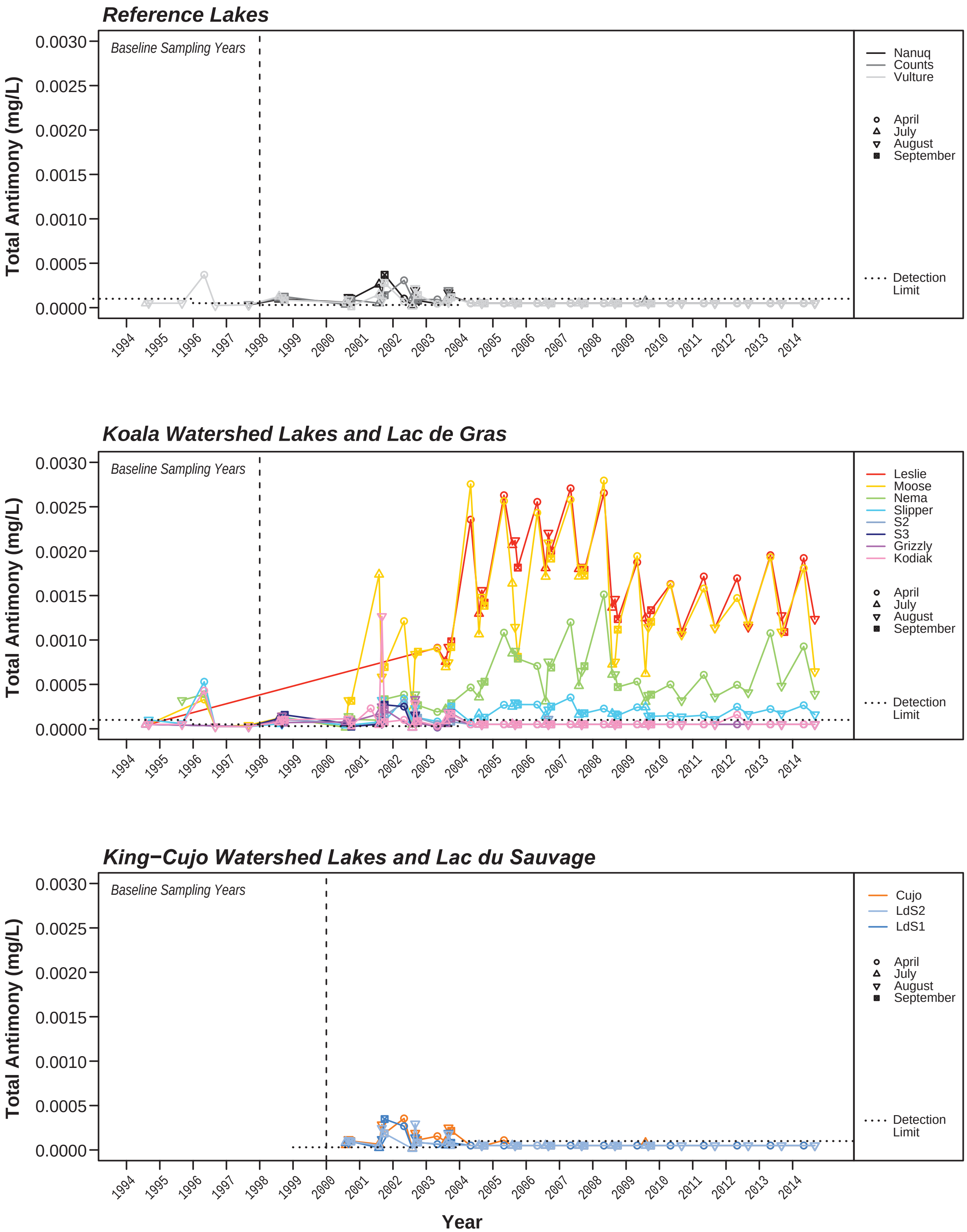
Total Aluminum Concentrations at AEMP Lake Sites, 1994 to 2014



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 6-23

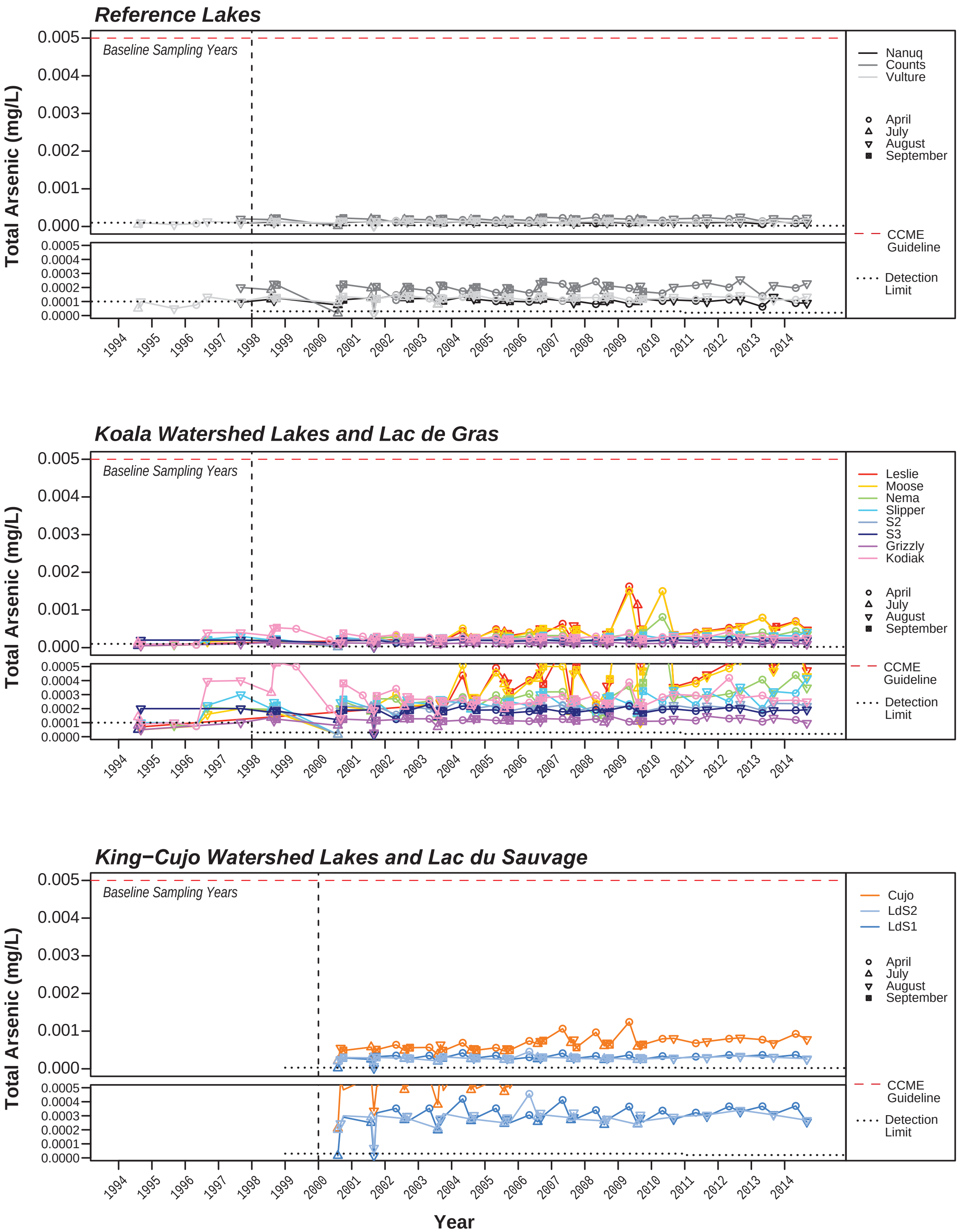
Total Antimony Concentrations at AEMP Lake Sites, 1994 to 2014



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 6-24

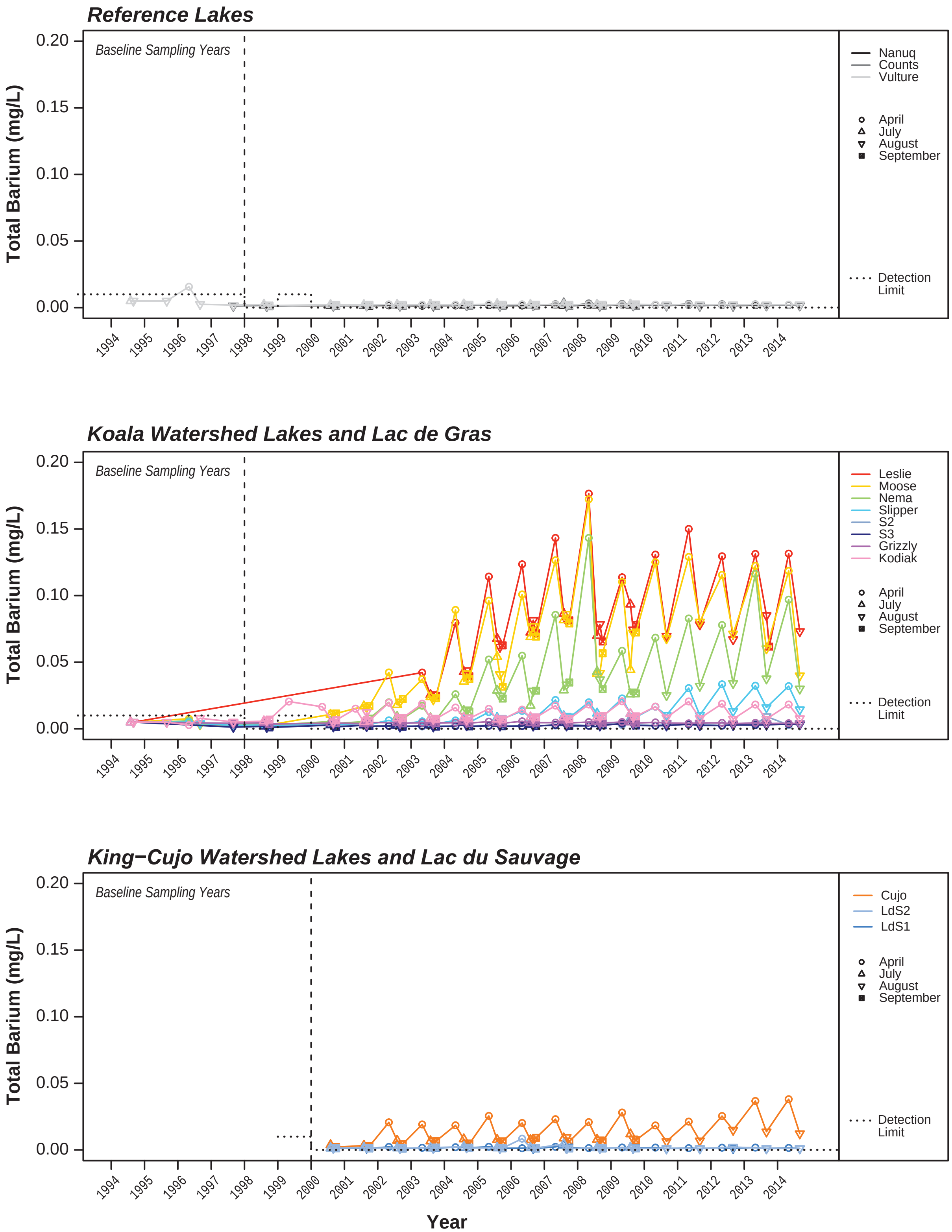
Total Arsenic Concentrations at AEMP Lake Sites, 1994 to 2014



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME Guideline = 0.005 mg/L.

Figure 6-25

Total Barium Concentrations at AEMP Lake Sites, 1994 to 2014

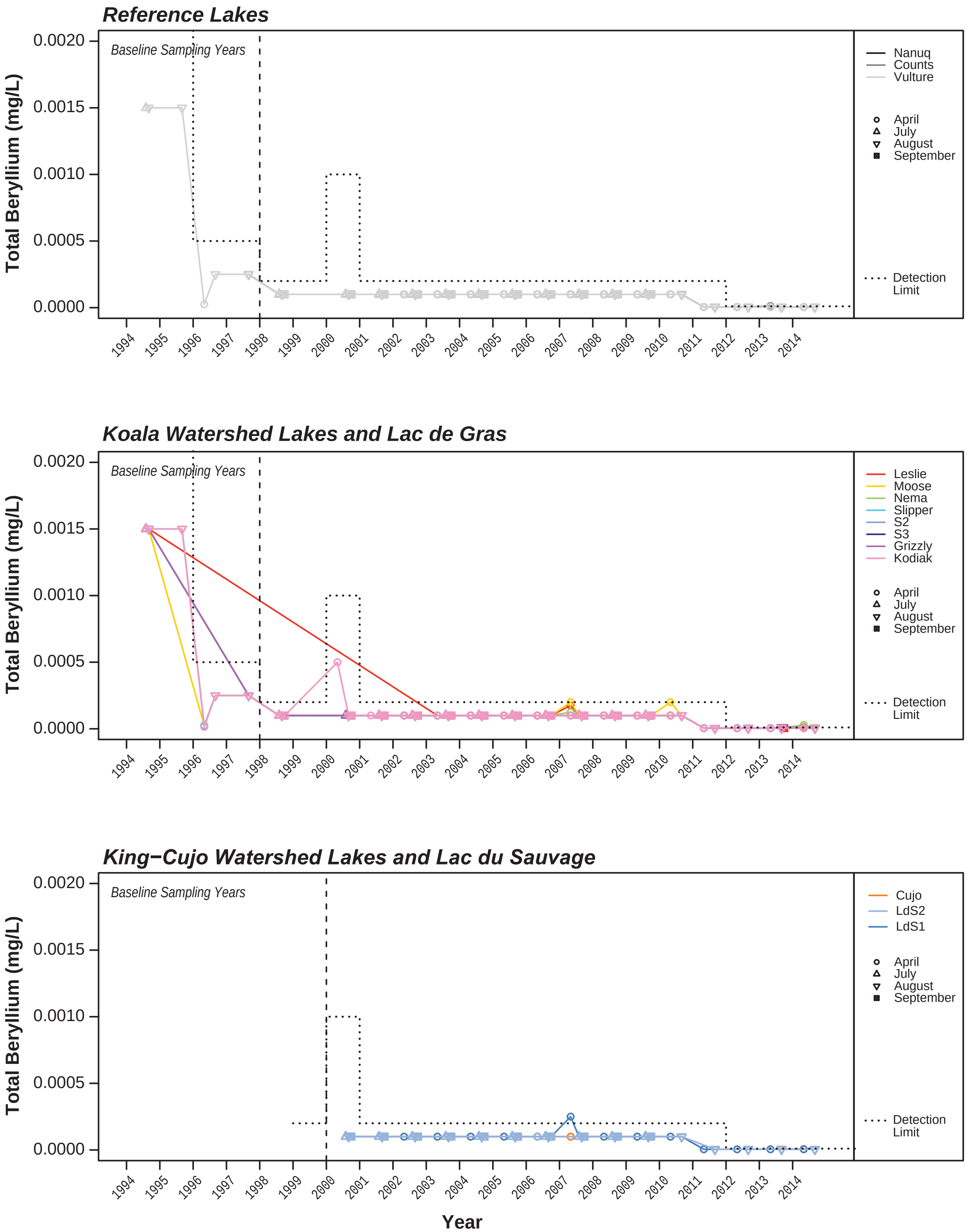


Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
Water quality benchmark (Haywood and Drinnan 1983) = 1 mg/L.

Figure 6-26

Total Beryllium Concentrations at AEMP Lake Sites, 1994 to 2014

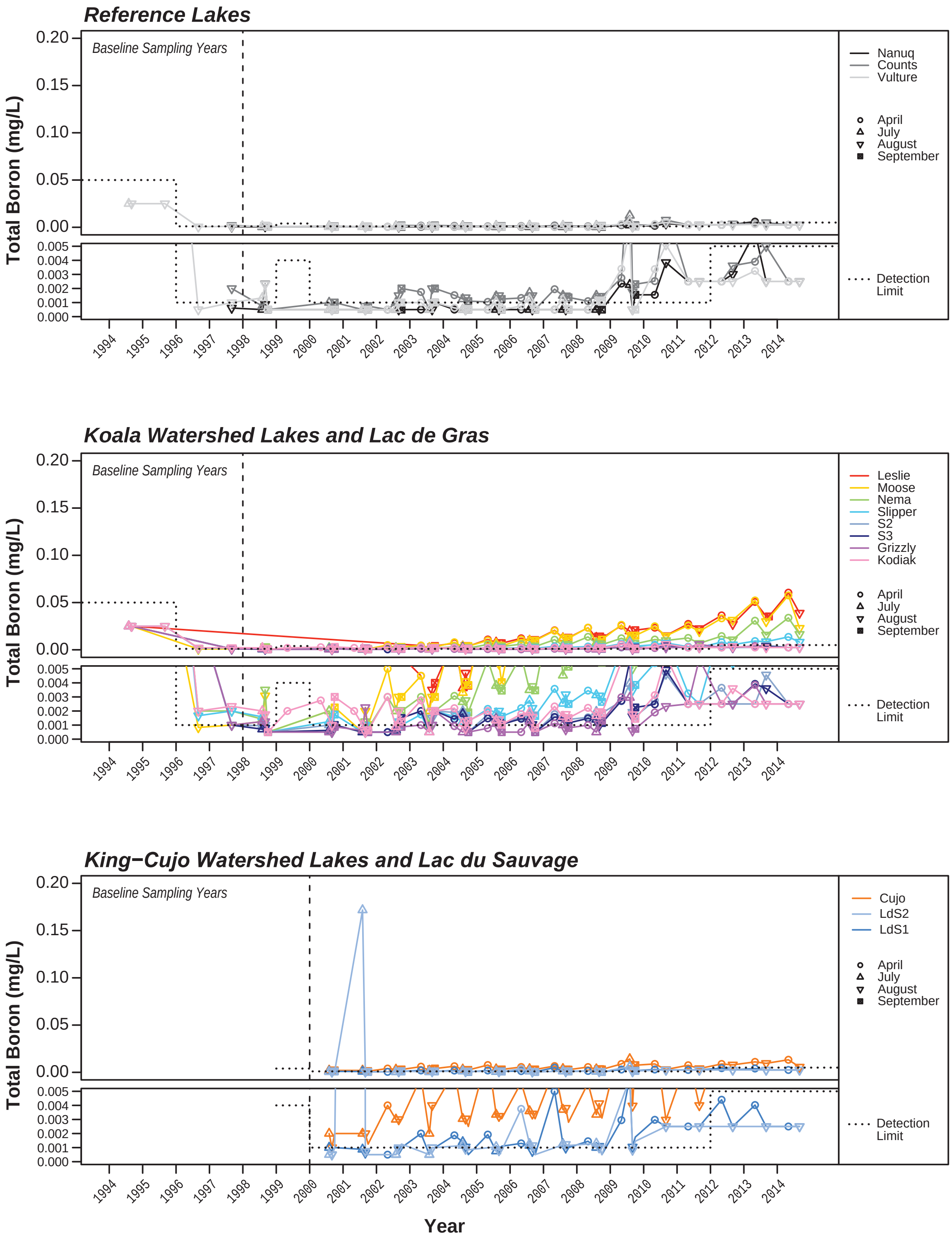
ERM



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 6-27

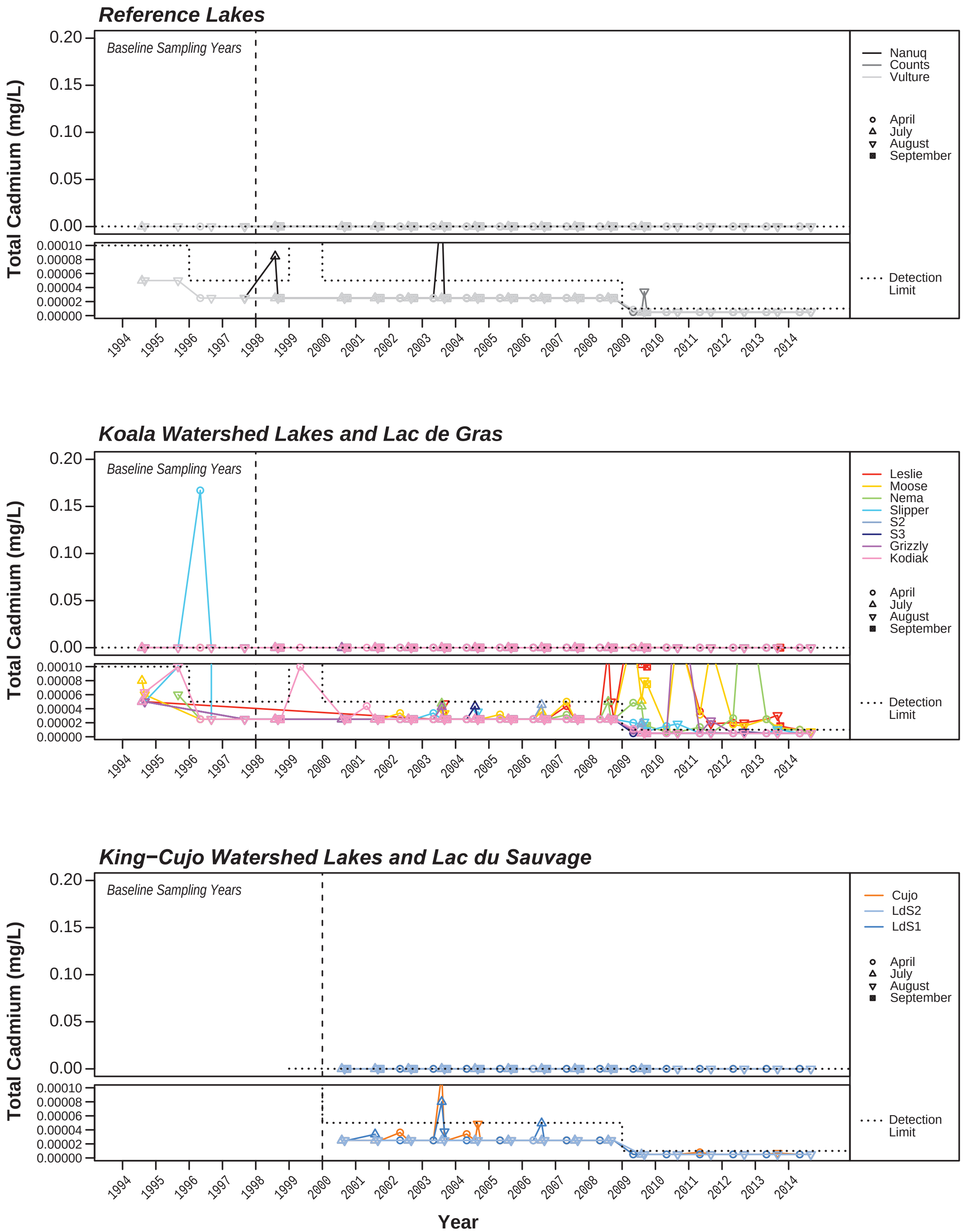
Total Boron Concentrations at AEMP Lake Sites, 1994 to 2014



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown. CCME Guideline = 1.5 mg/L.

Figure 6-28

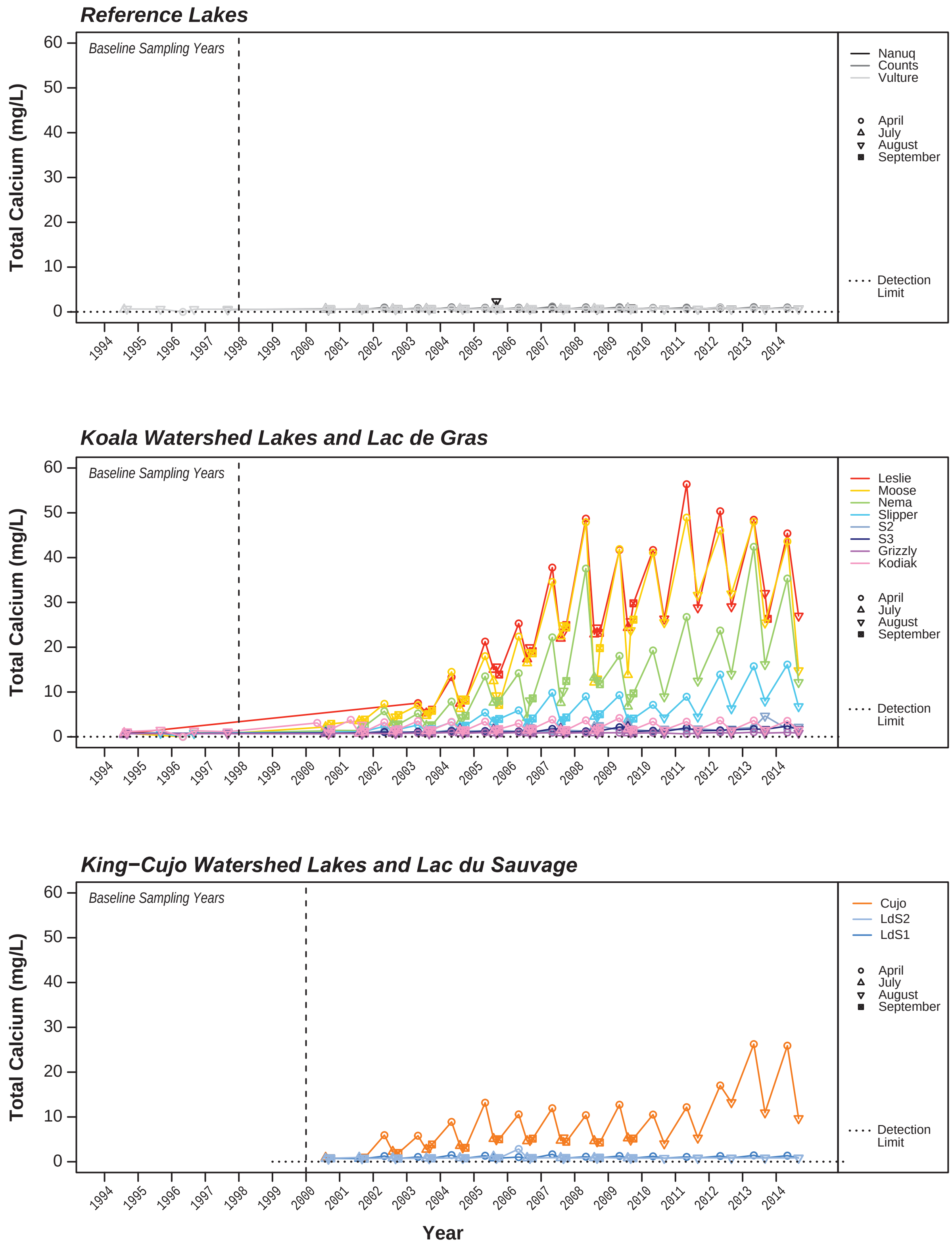
Total Cadmium Concentrations at AEMP Lake Sites, 1994 to 2014



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
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 = 0.00037 mg/L where hardness > 280 mg/L.

Figure 6-29

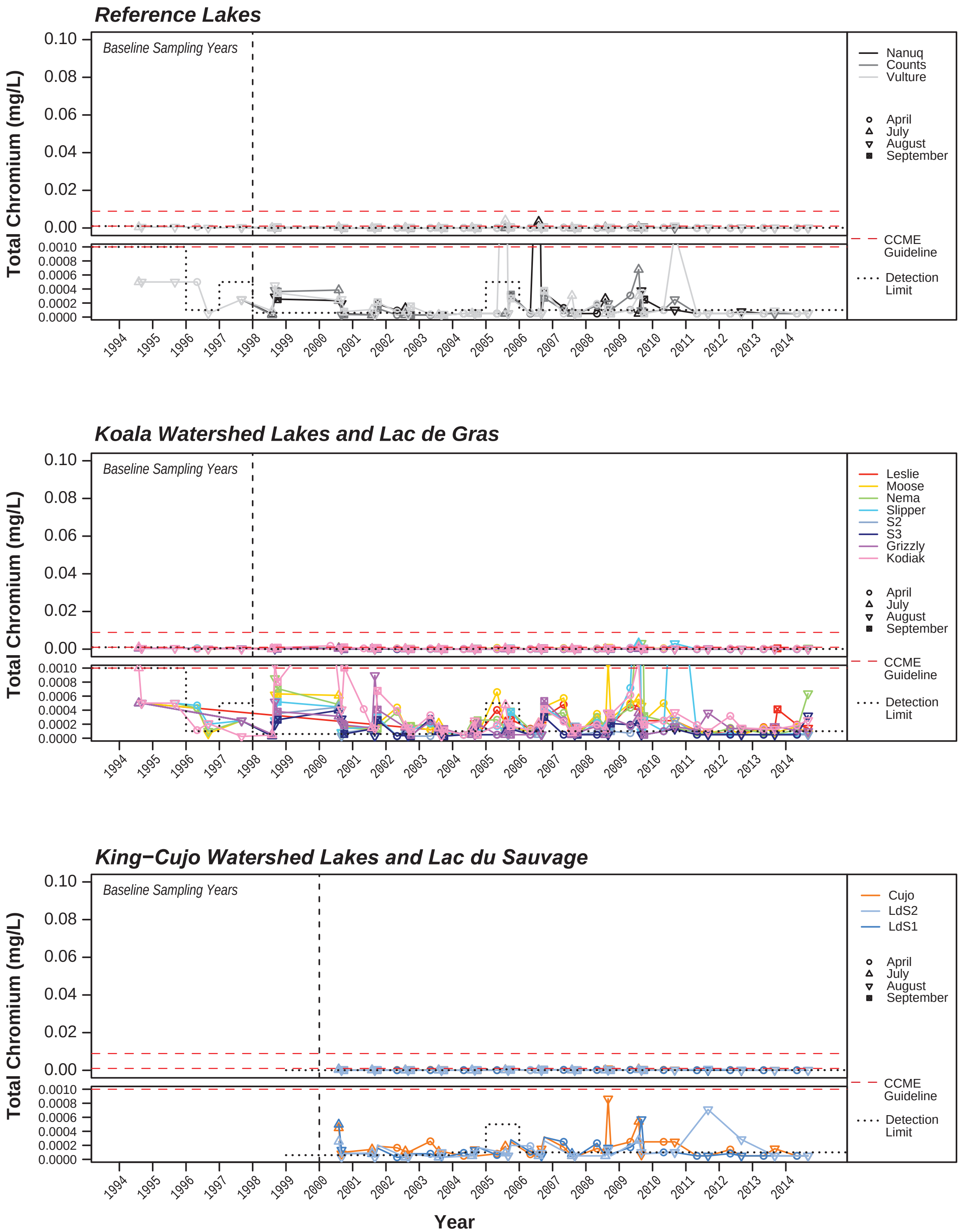
Total Calcium Concentrations at AEMP Lake Sites, 1994 to 2014



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

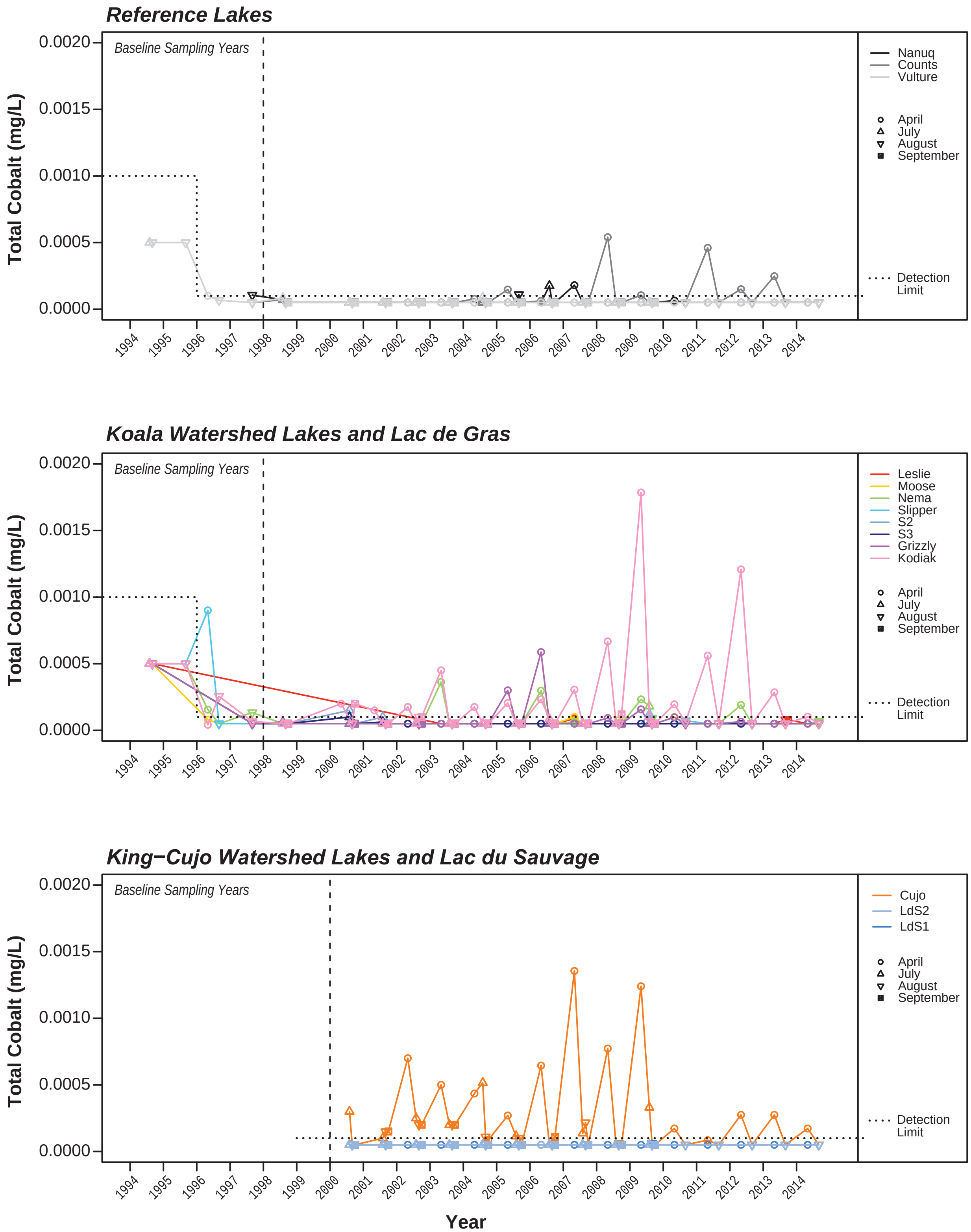
Figure 6-30

Total Chromium Concentrations at AEMP Lake Sites, 1994 to 2014



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).

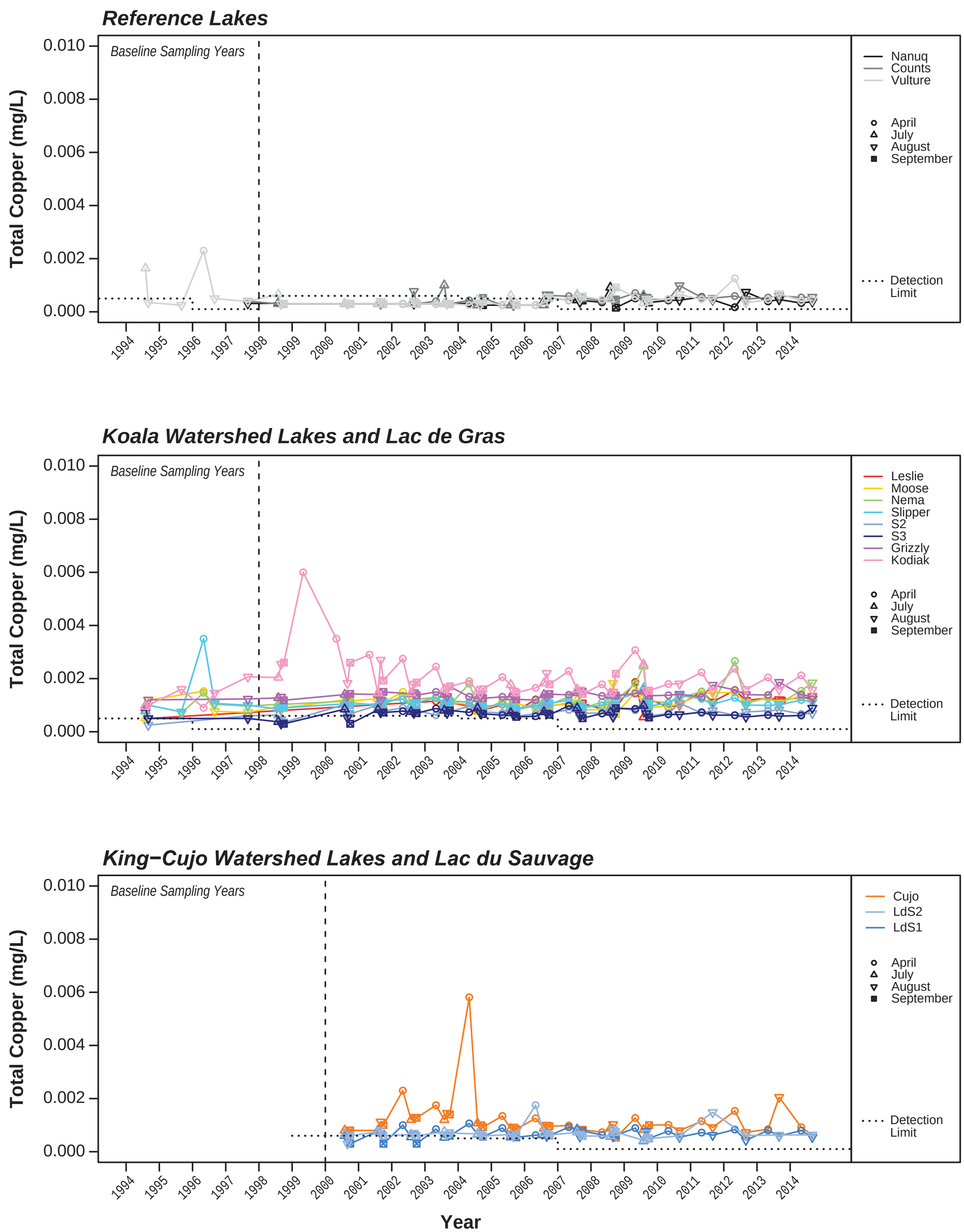
Figure 6-31
Total Cobalt Concentrations at AEMP Lake Sites, 1994 to 2014



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 6-32

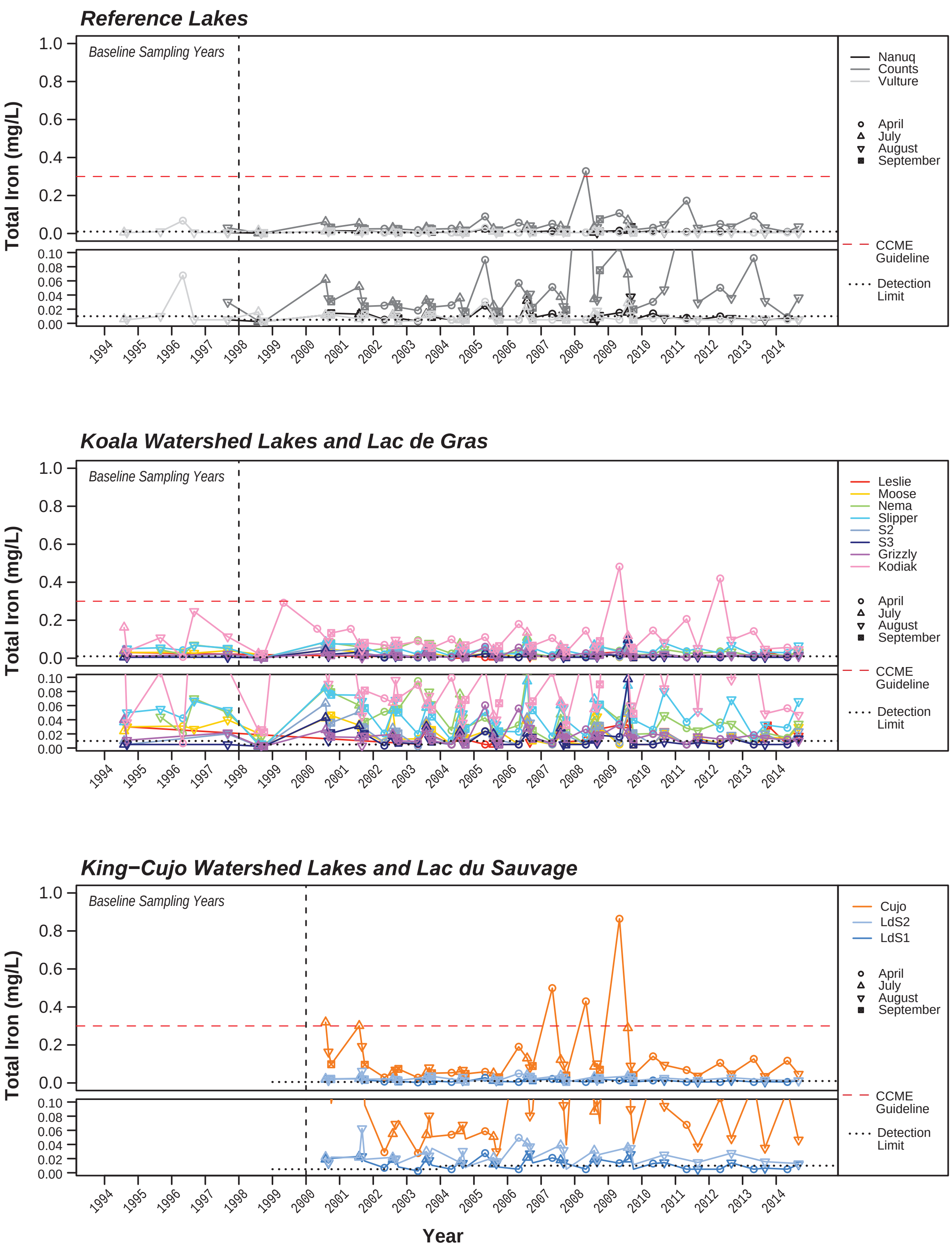
Total Copper Concentrations at AEMP Lake Sites, 1994 to 2014



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
*CCME Guideline = $e^{0.8545 \times (\ln Hardness) - 1.465}$ * 0.2/1000 mg/L, where hardness < 180 mg/L and 0.004 mg/L where hardness is ≧ to 180 mg/L . Minimum guideline = 0.002 mg/L.*

Figure 6-33

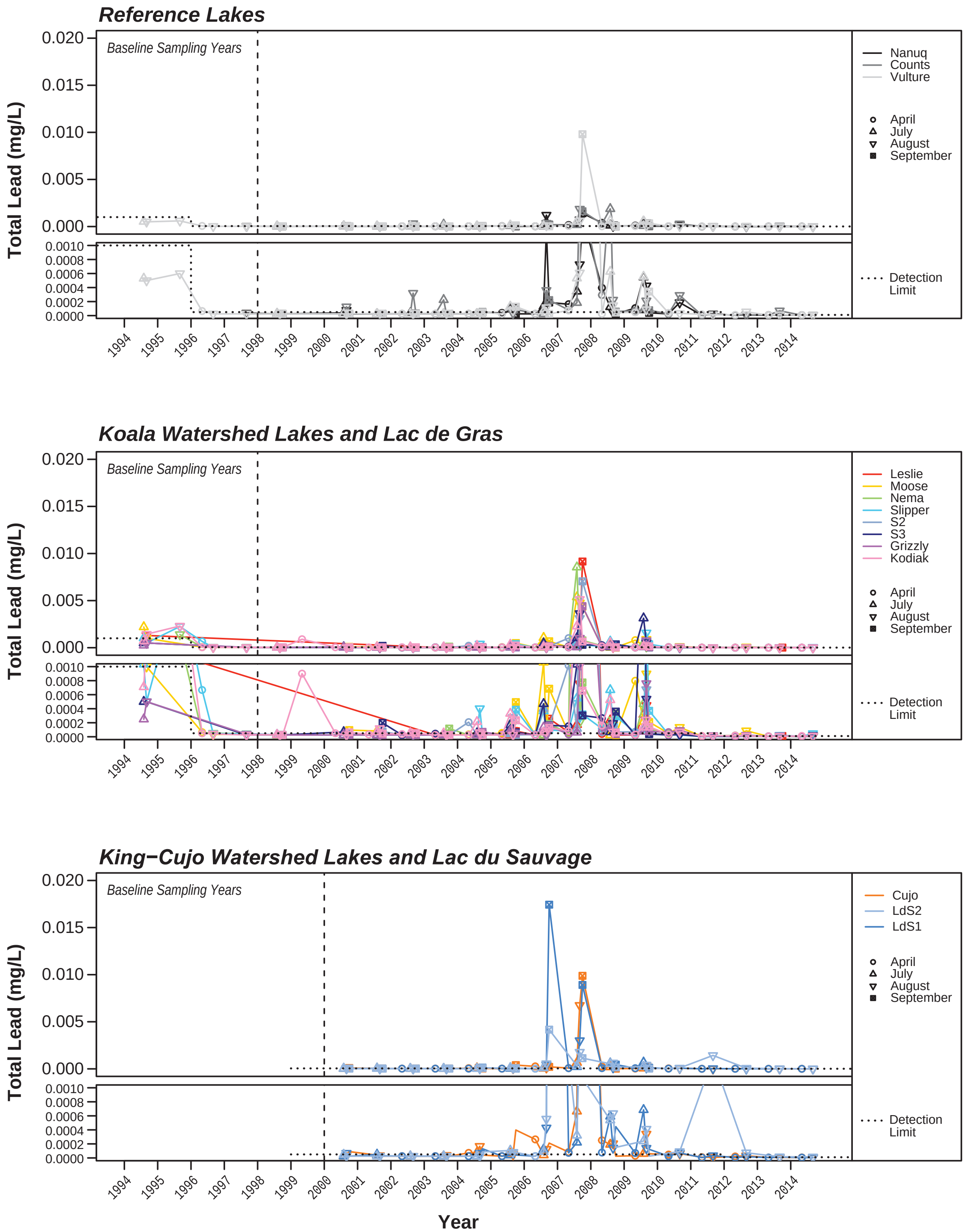
Total Iron Concentrations at AEMP Lake Sites, 1994 to 2014



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME Guideline = 0.3 mg/L.

Figure 6-34

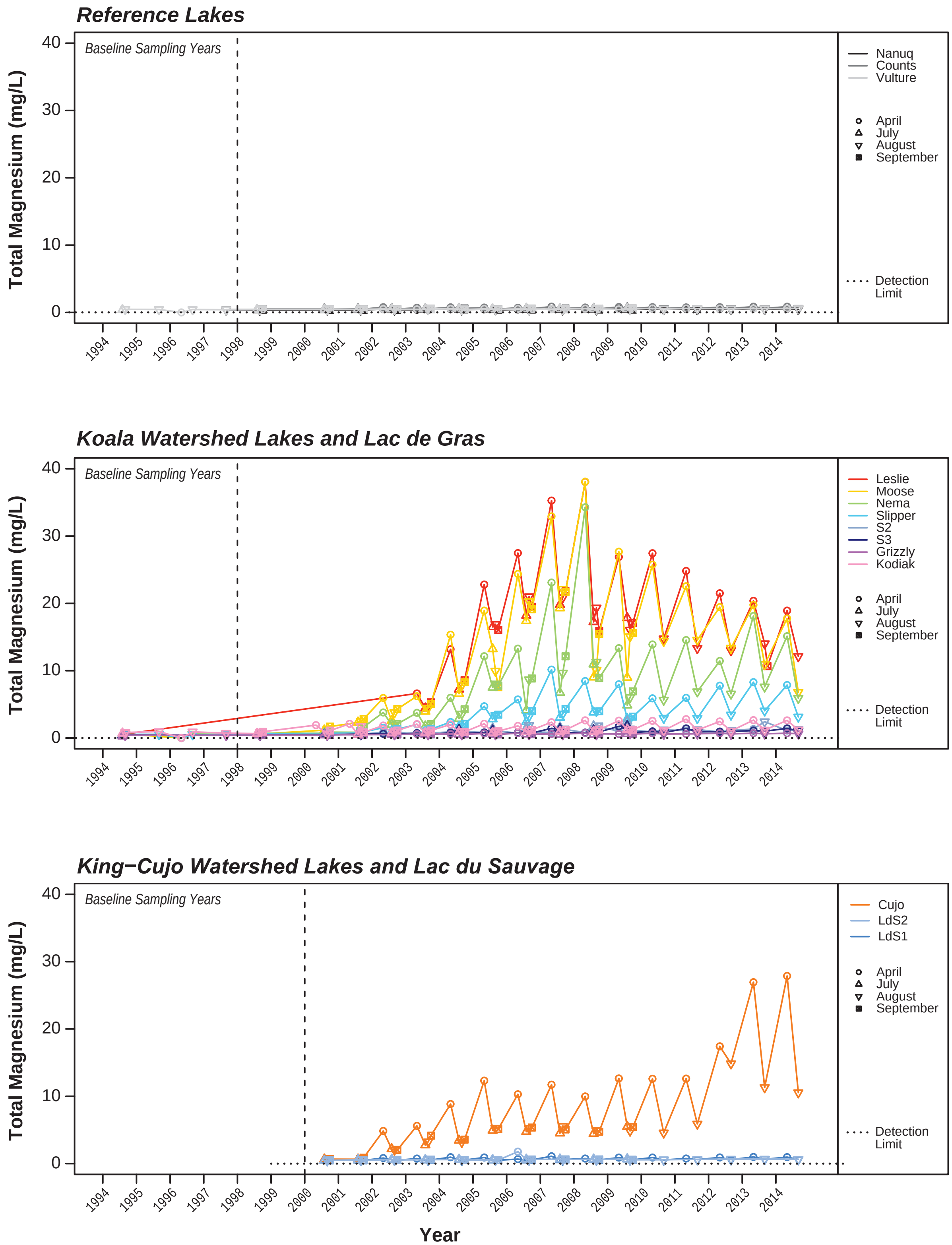
Total Lead Concentrations at AEMP Lake Sites, 1994 to 2014



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 6-35

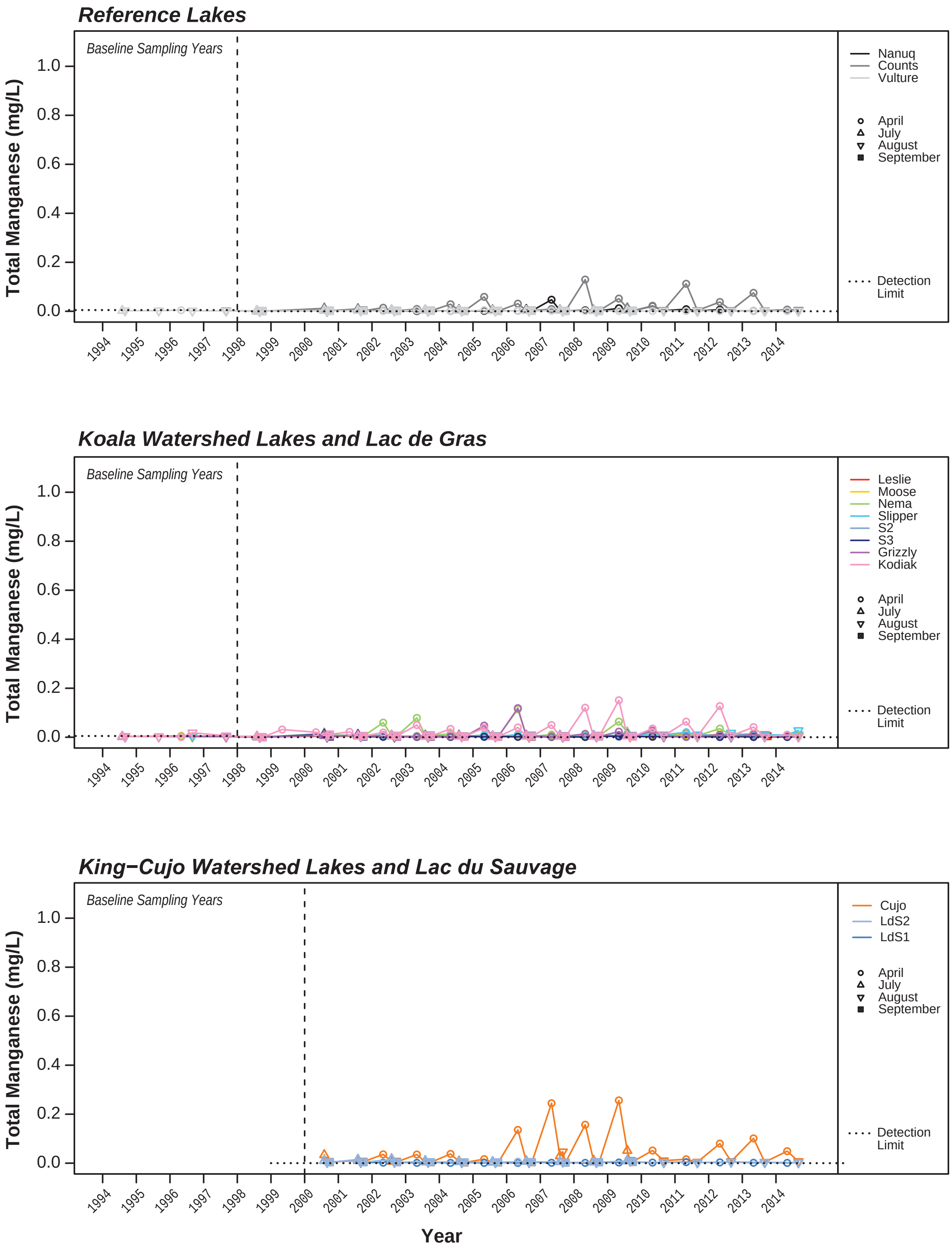
Total Magnesium Concentrations at AEMP Lake Sites, 1994 to 2014



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 6-36

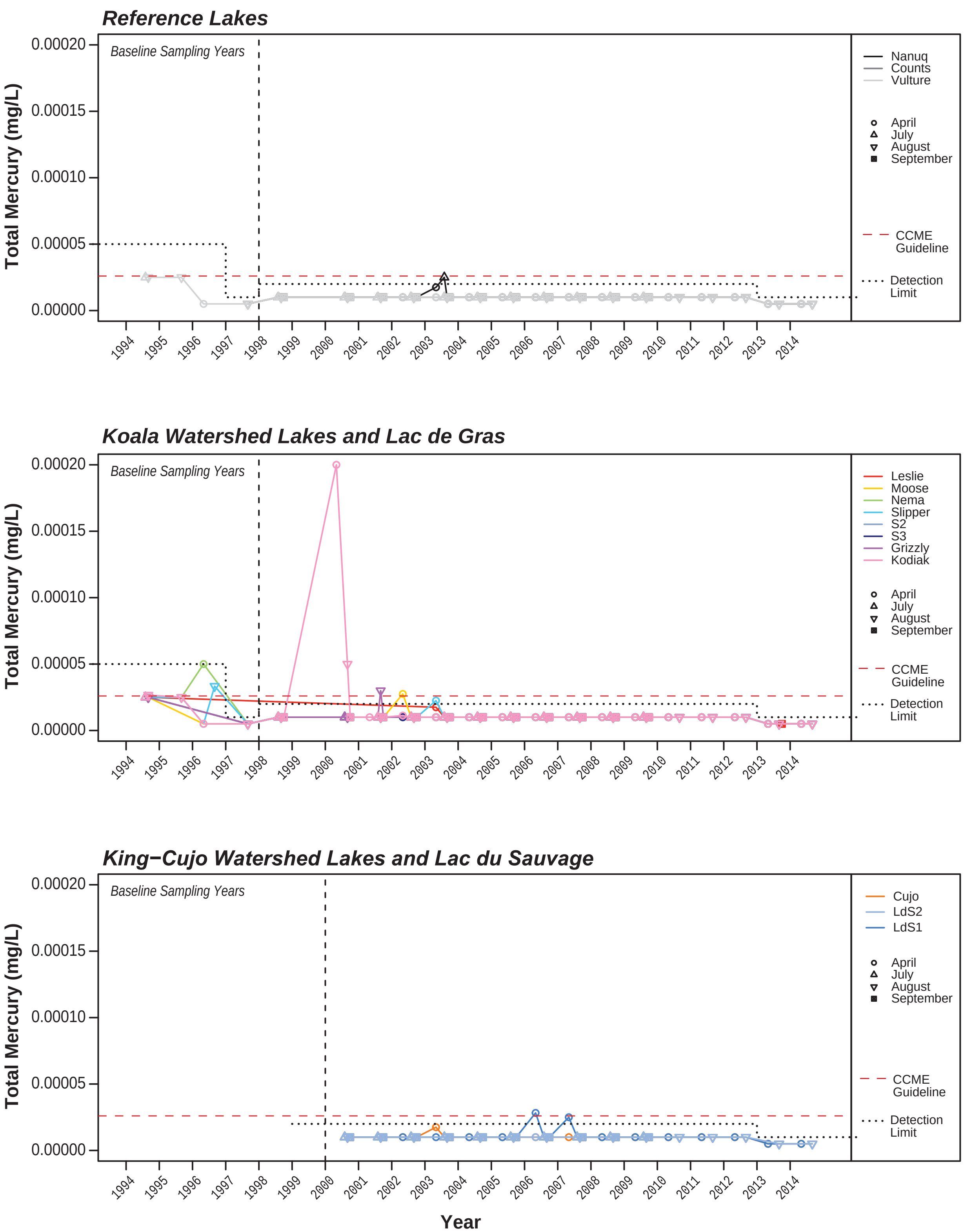
Total Manganese Concentrations at AEMP Lake Sites, 1994 to 2014



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 6-37

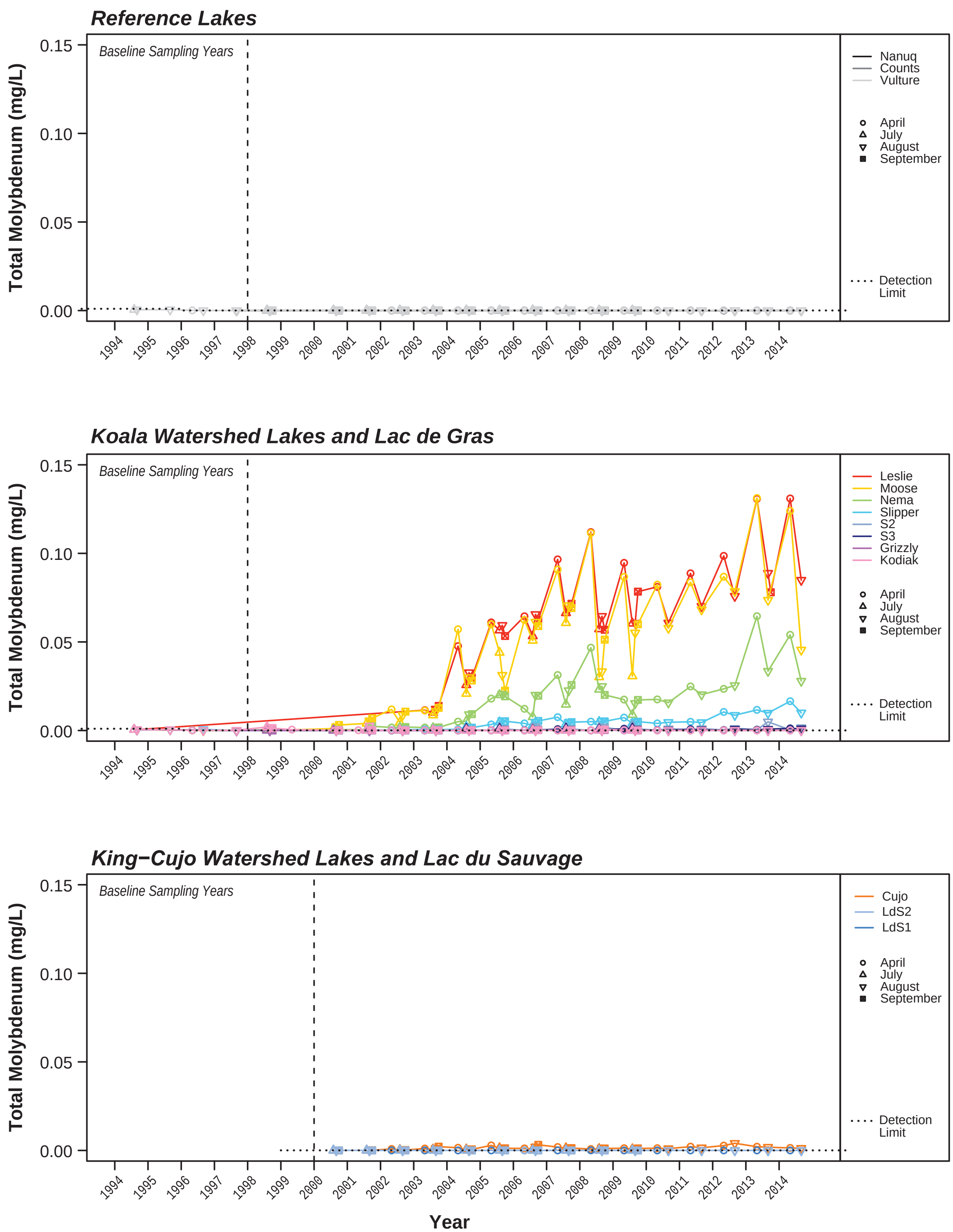
Total Mercury Concentrations at AEMP Lake Sites, 1994 to 2014



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME guideline = 0.0000026 mg/L.

Figure 6-38

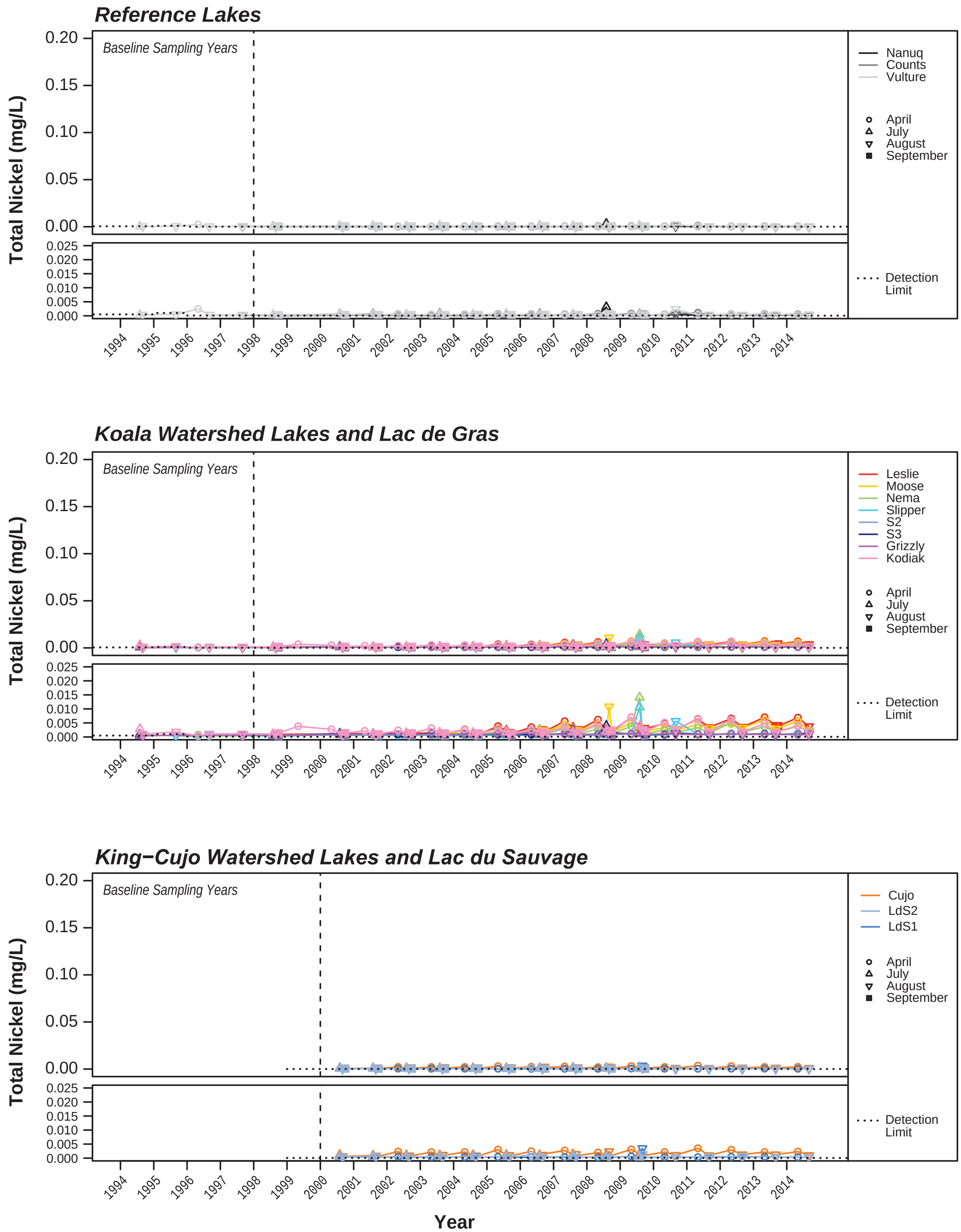
Total Molybdenum Concentrations at AEMP Lake Sites, 1994 to 2014



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown. SSWQO = 19.38 mg/L.

Figure 6-39

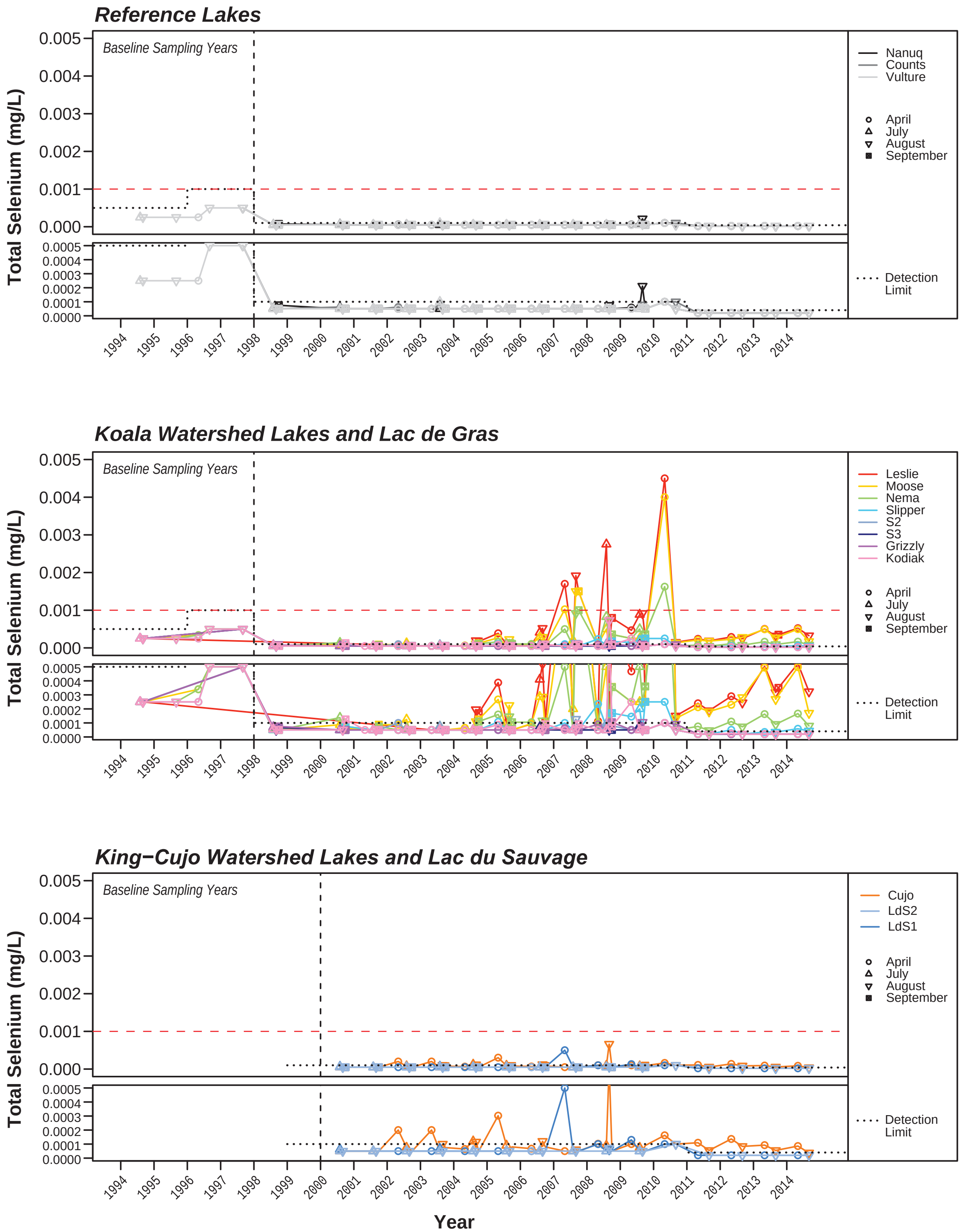
Total Nickel Concentrations at AEMP Lake Sites, 1994 to 2014



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
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.

Figure 6-40

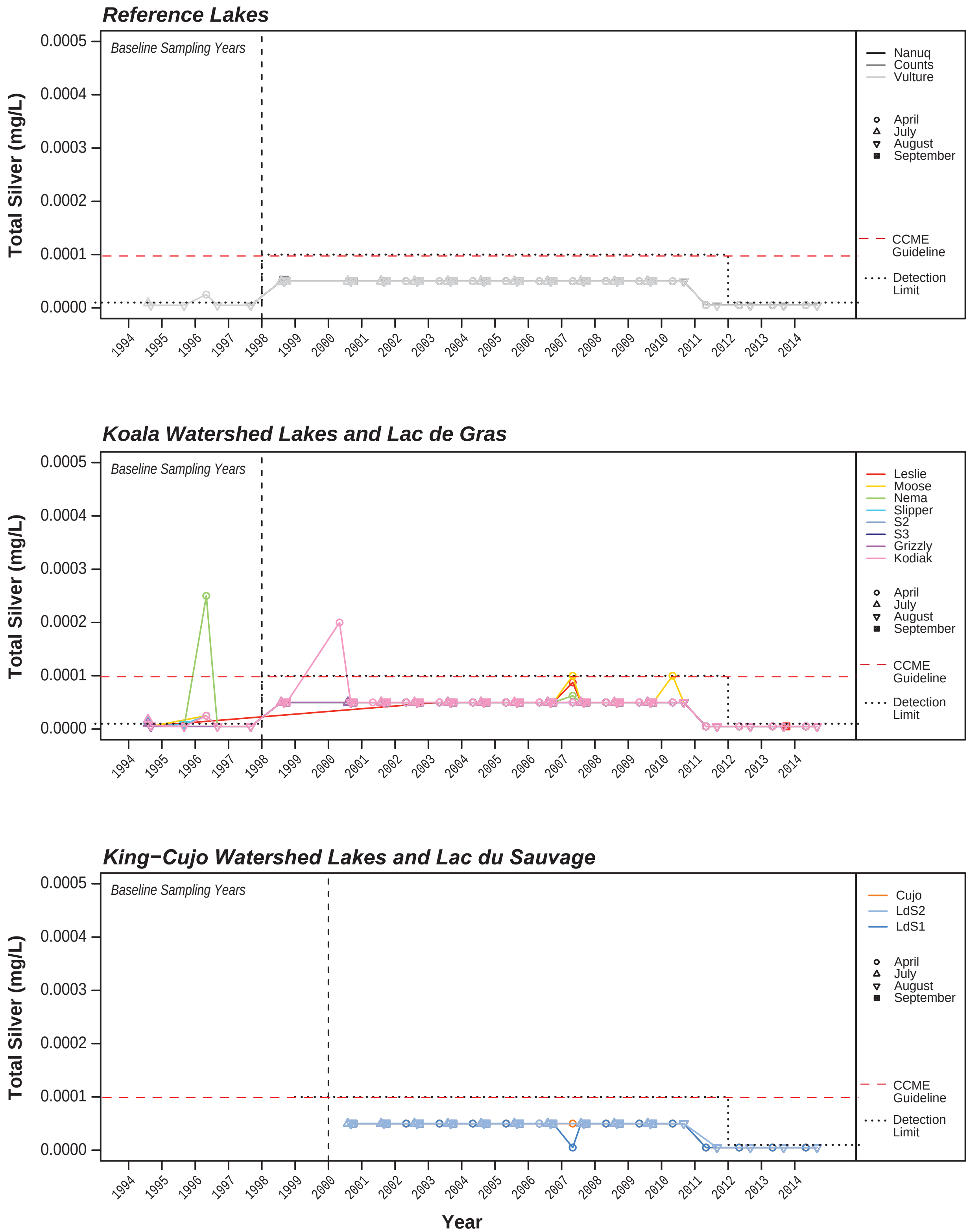
Total Selenium Concentrations at AEMP Lake Sites, 1994 to 2014



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME Guideline = 0.001 mg/L.

Figure 6-41

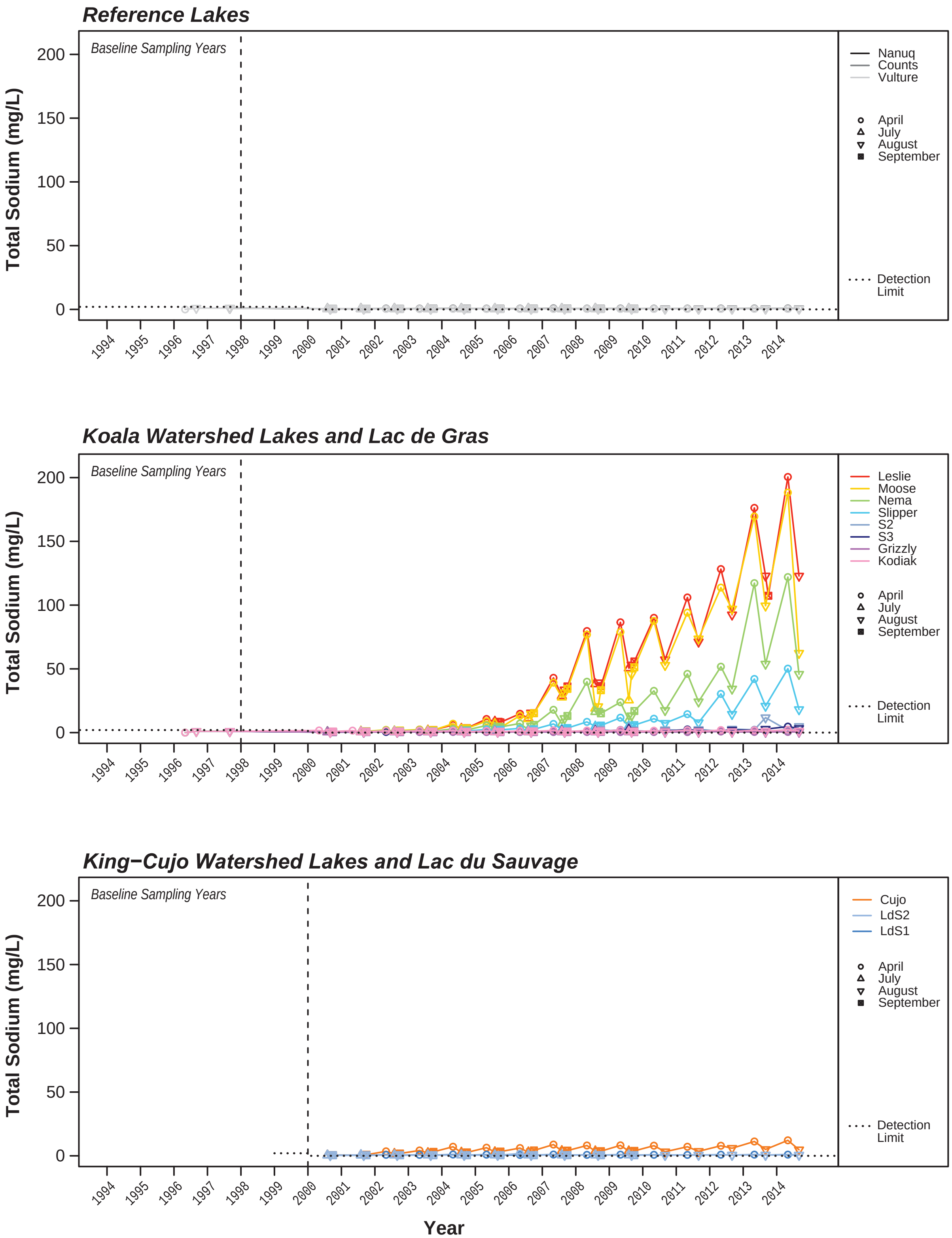
Total Silver Concentrations at AEMP Lake Sites, 1994 to 2014



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME guideline = 0.0001 mg/L.

Figure 6-42

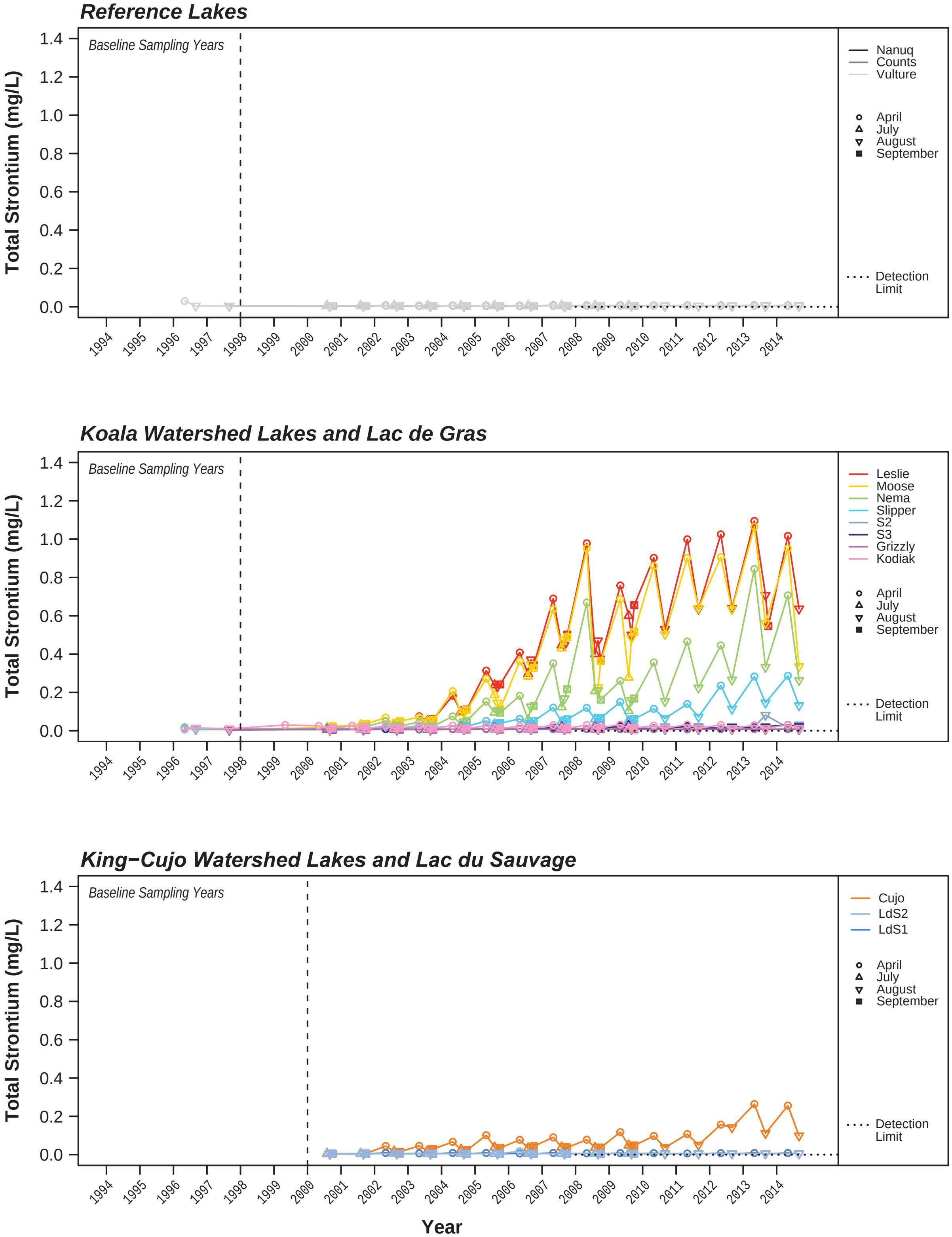
Total Sodium Concentrations at AEMP Lake Sites, 1994 to 2014



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

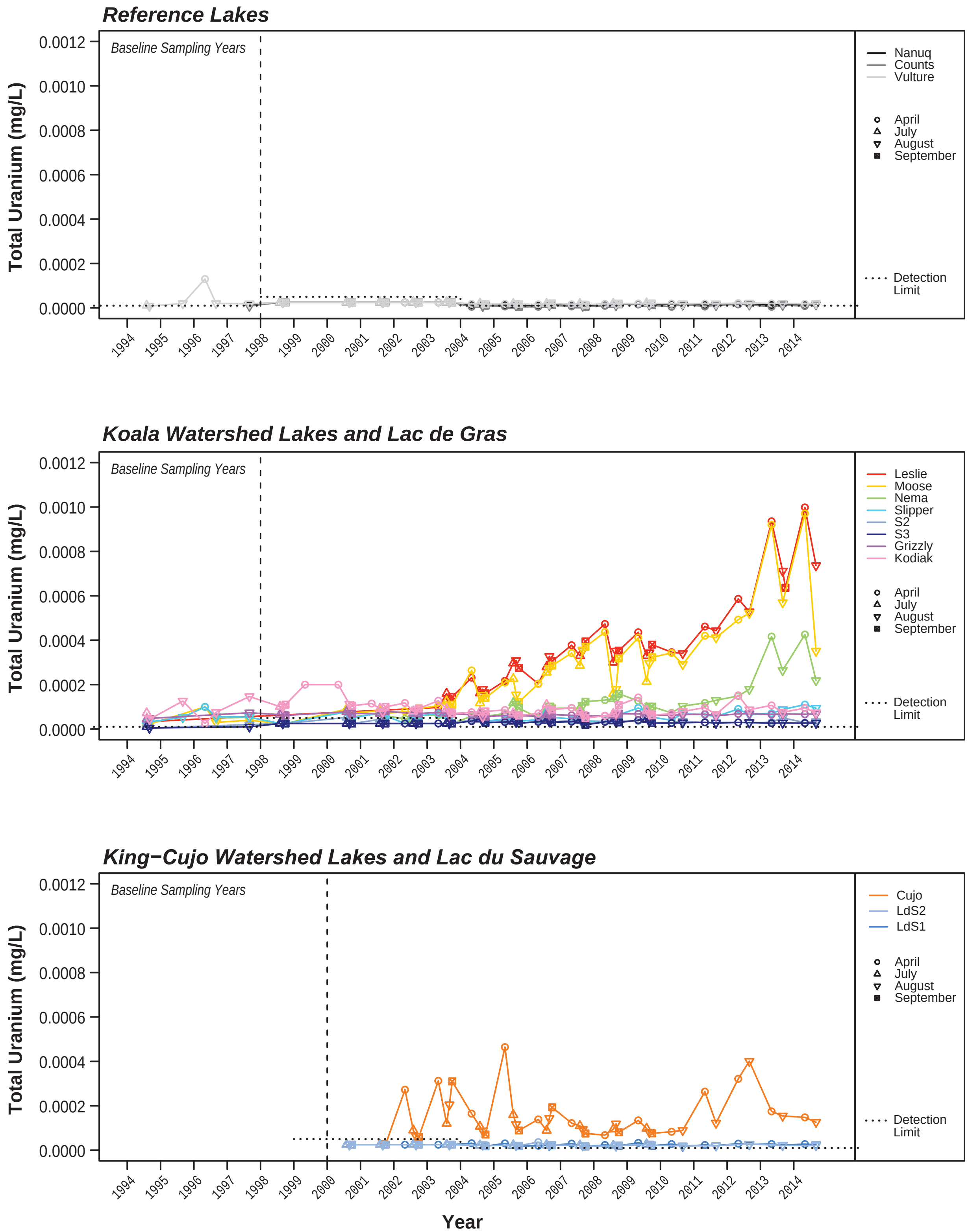
Figure 6-43

Total Strontium Concentrations at AEMP Lake Sites, 1994 to 2014



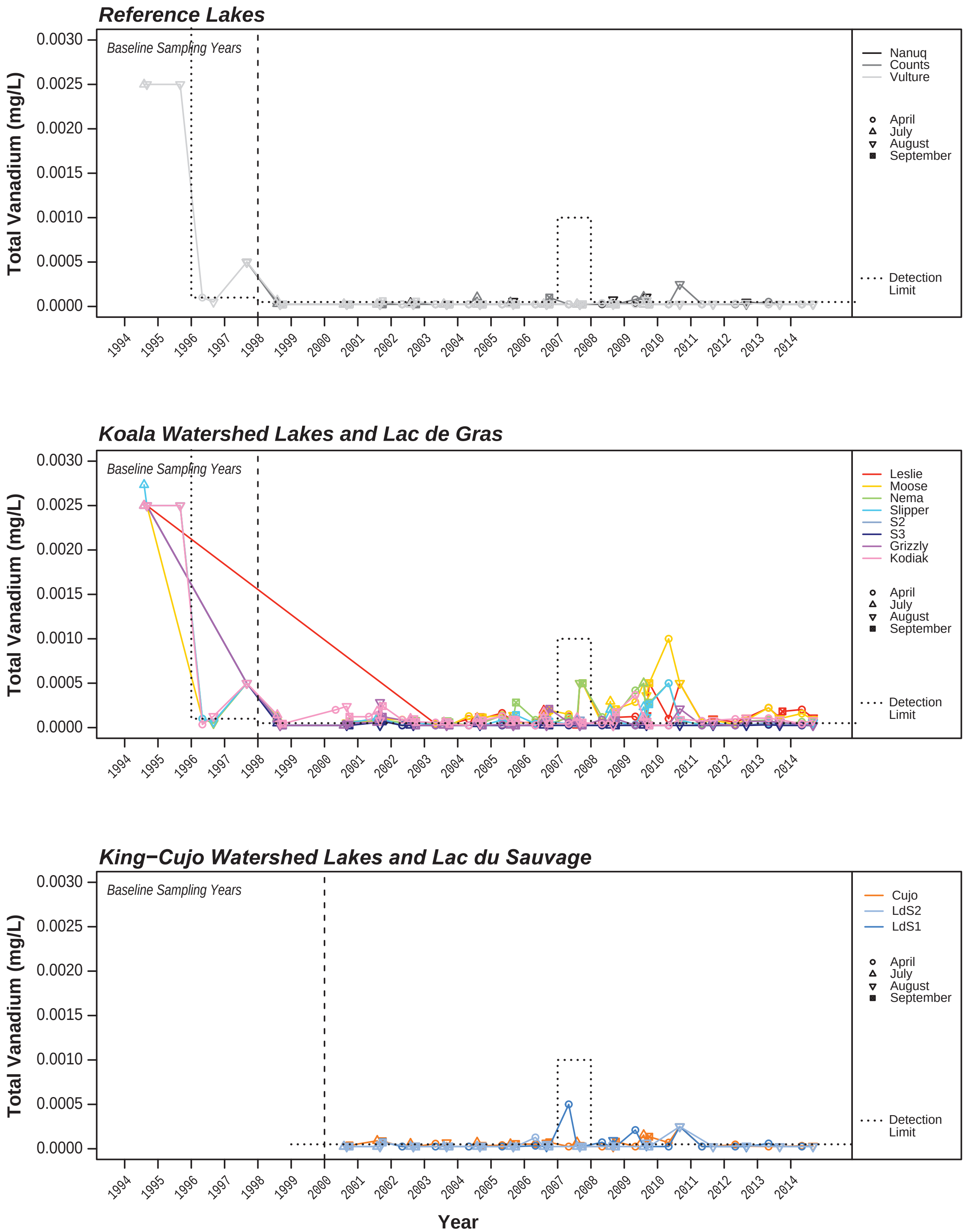
Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
Water quality benchmark (Golder 2011) = 6.242 mg/L.

Figure 6-44
Total Uranium Concentrations at AEMP Lake Sites, 1994 to 2014



Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME Guideline = 0.015 mg/L.

Figure 6-45
Total Vanadium Concentrations at AEMP Lake Sites, 1994 to 2014

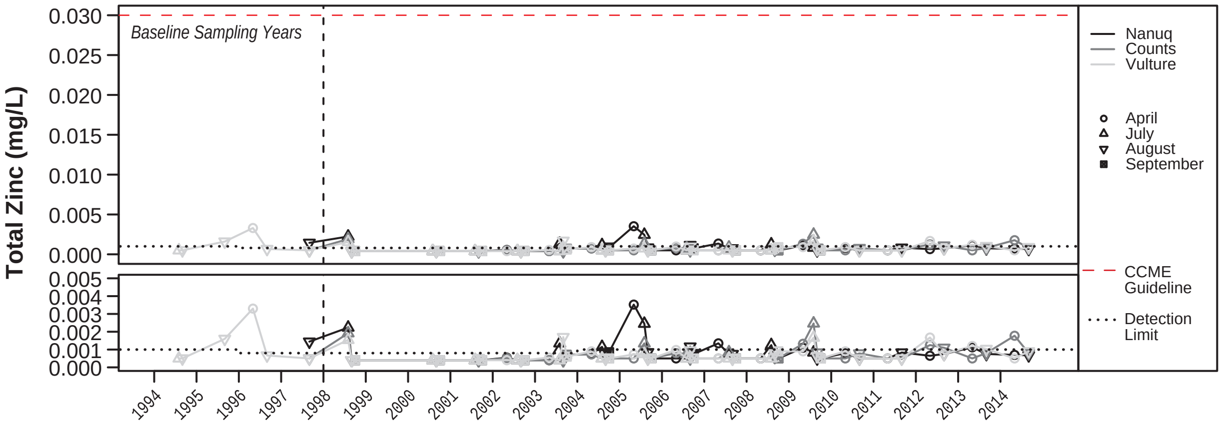


Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
SSWQO = 0.03 mg/L.

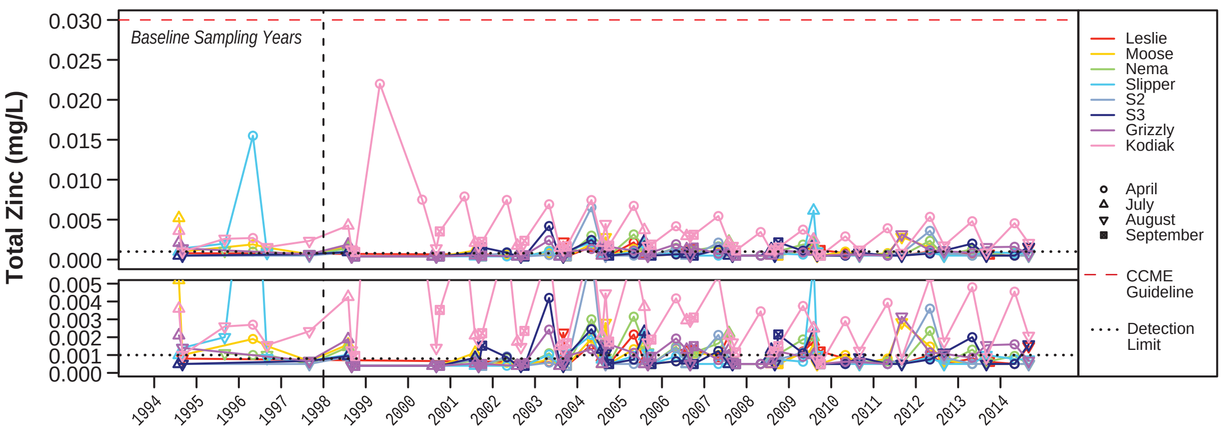
Figure 6-46

Total Zinc Concentrations at AEMP Lake Sites, 1994 to 2014

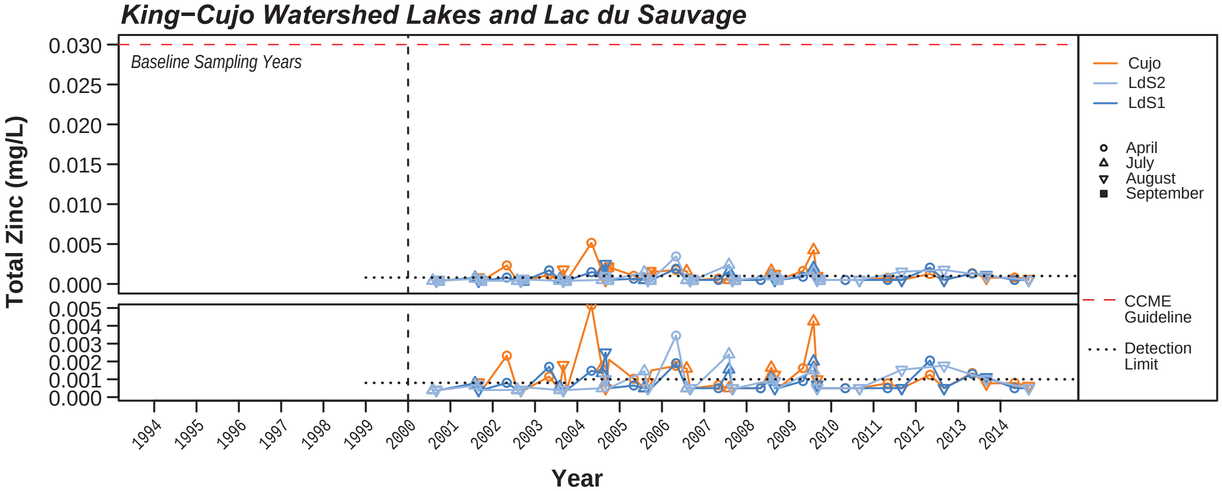
Reference Lakes



Koala Watershed Lakes and Lac de Gras



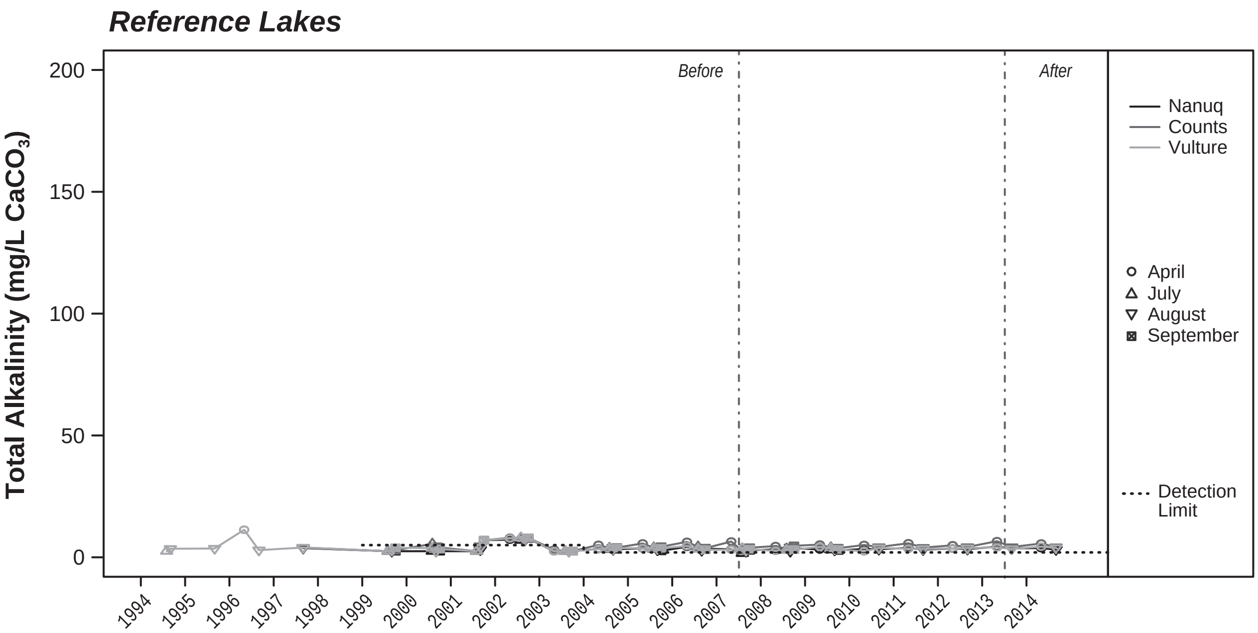
King−Cujo Watershed Lakes and Lac du Sauvage

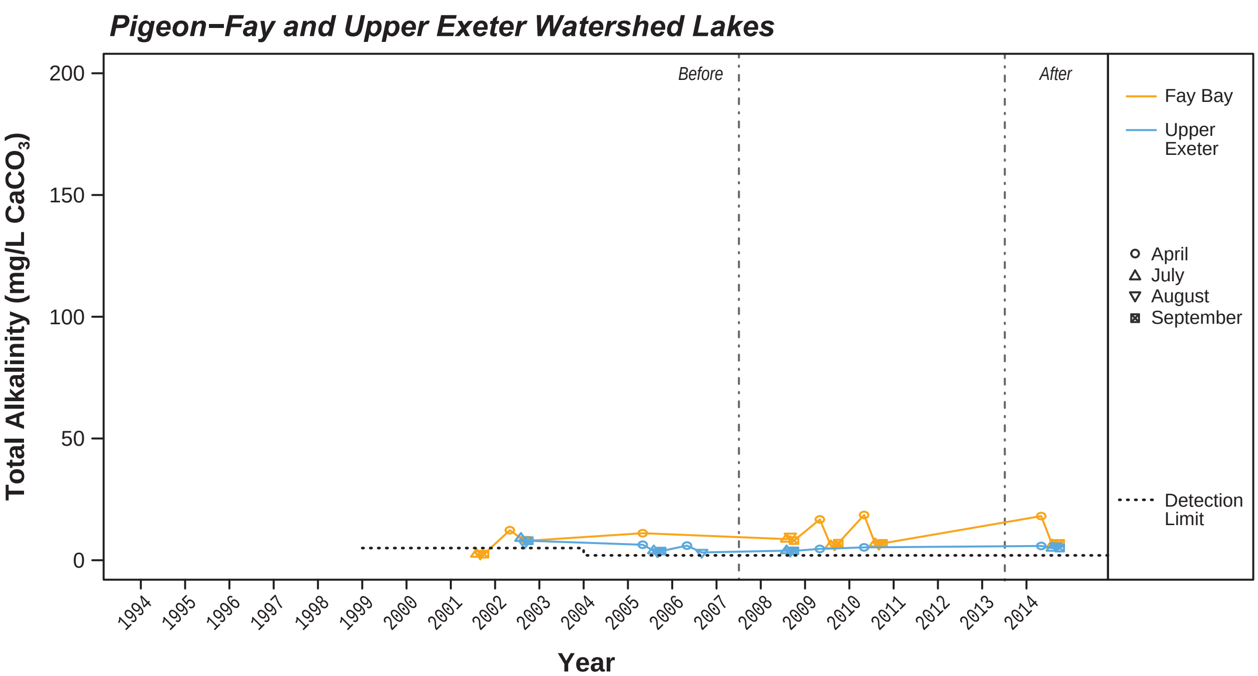


Notes: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.
CCME Guideline = 0.03 mg/L.

Figure 6-47

Total Alkalinity at Pigeon AEMP Lake Sites, 1994 to 2014

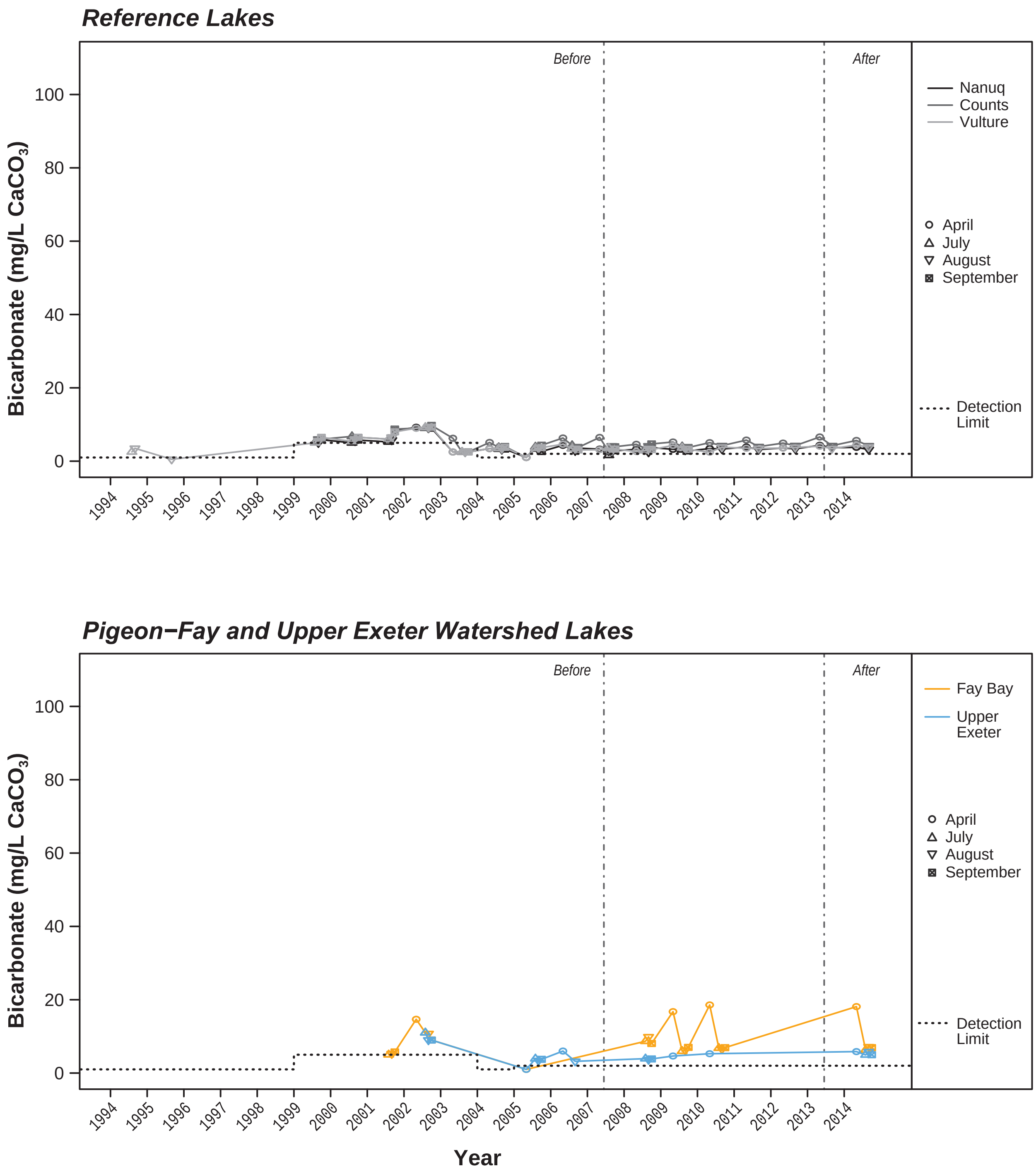




Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 6-48

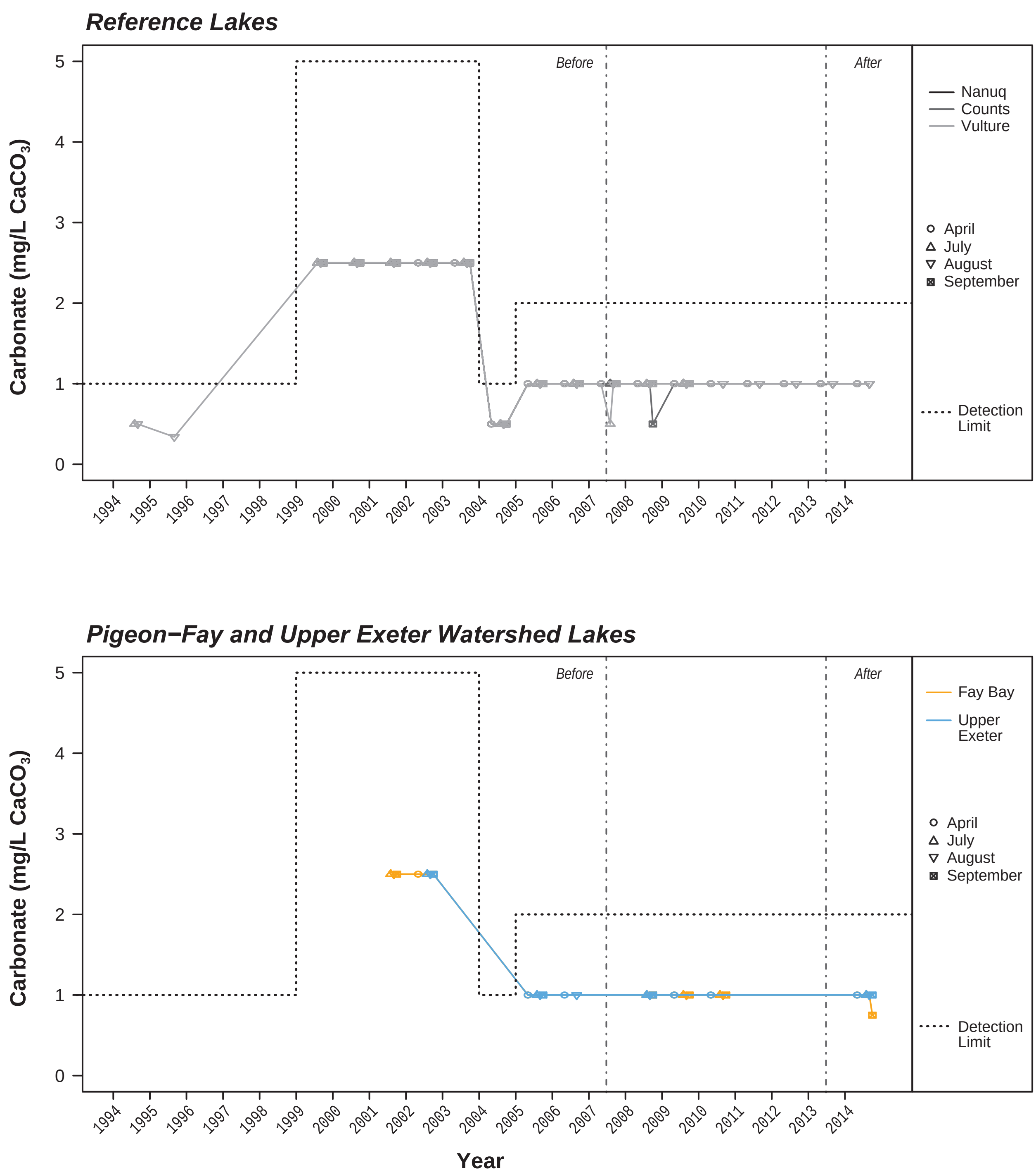
Bicarbonate Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014



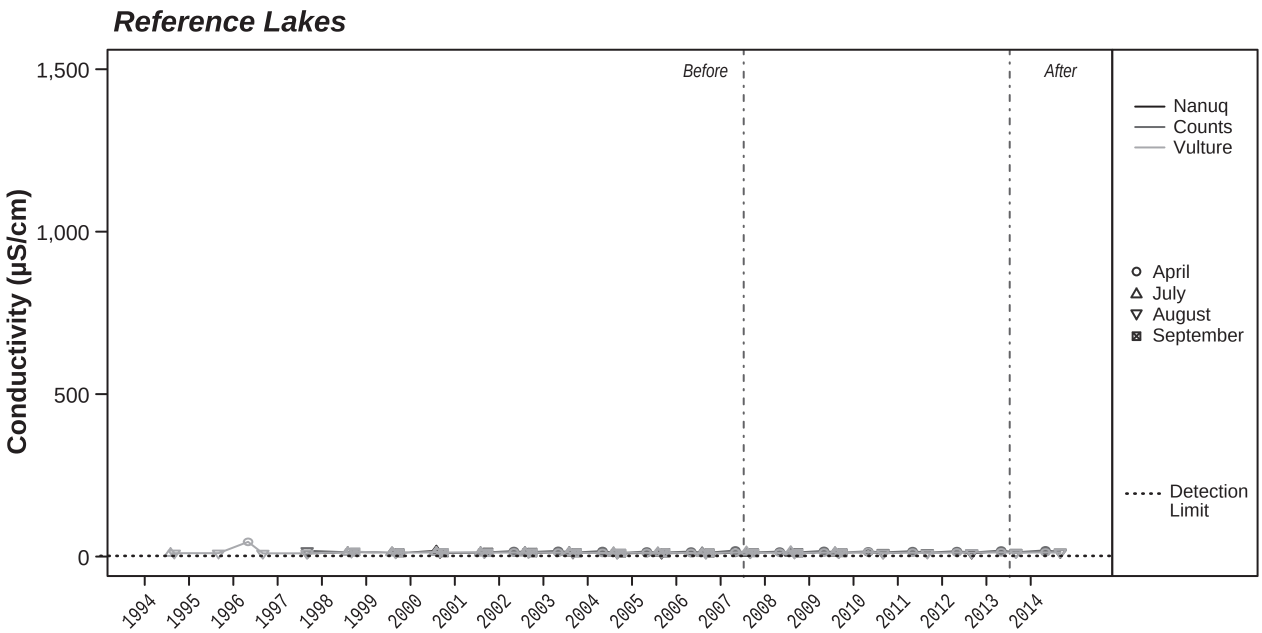
Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

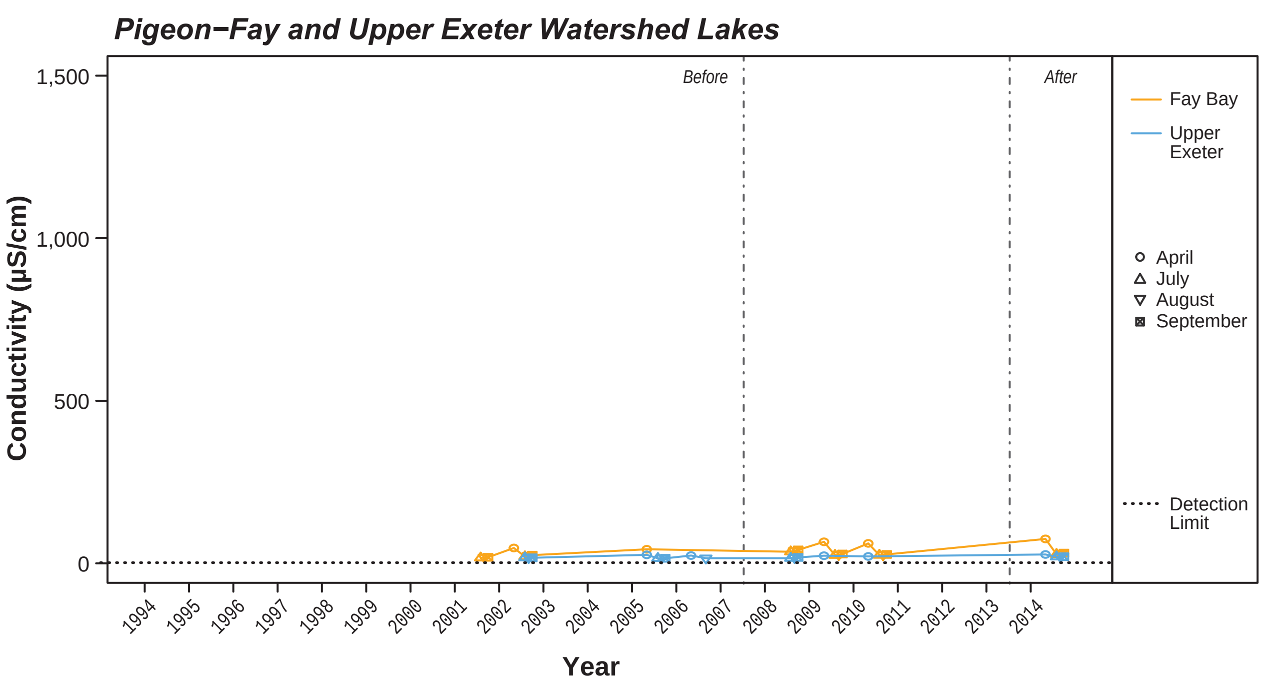
Figure 6-49

Carbonate Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014



Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

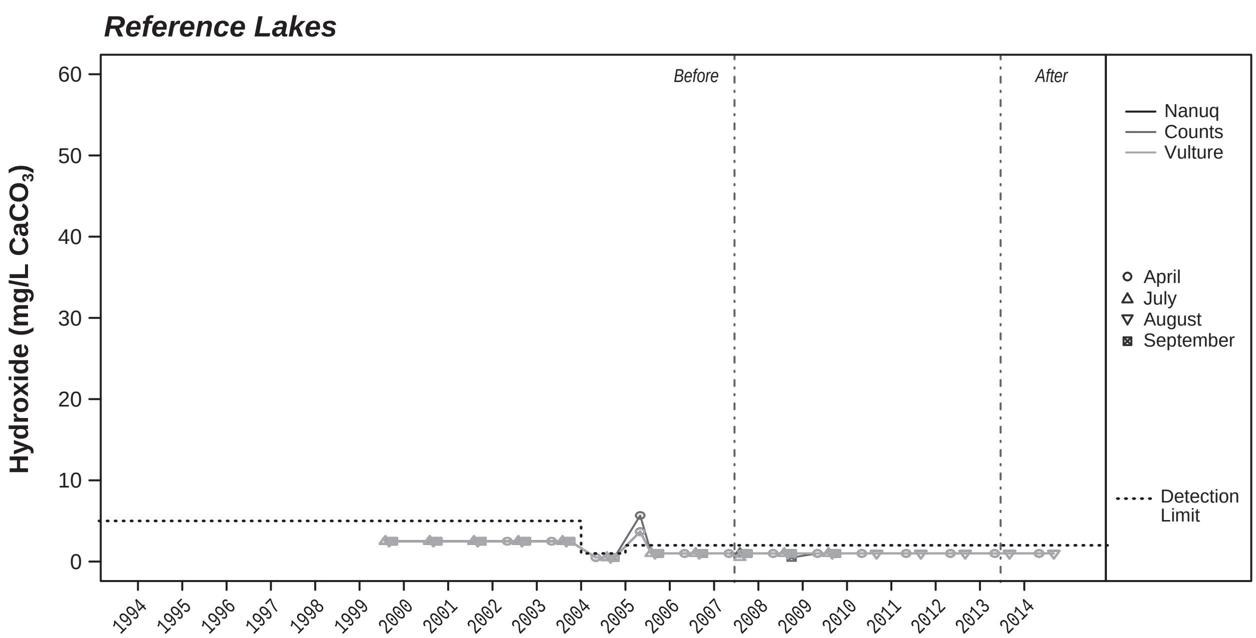


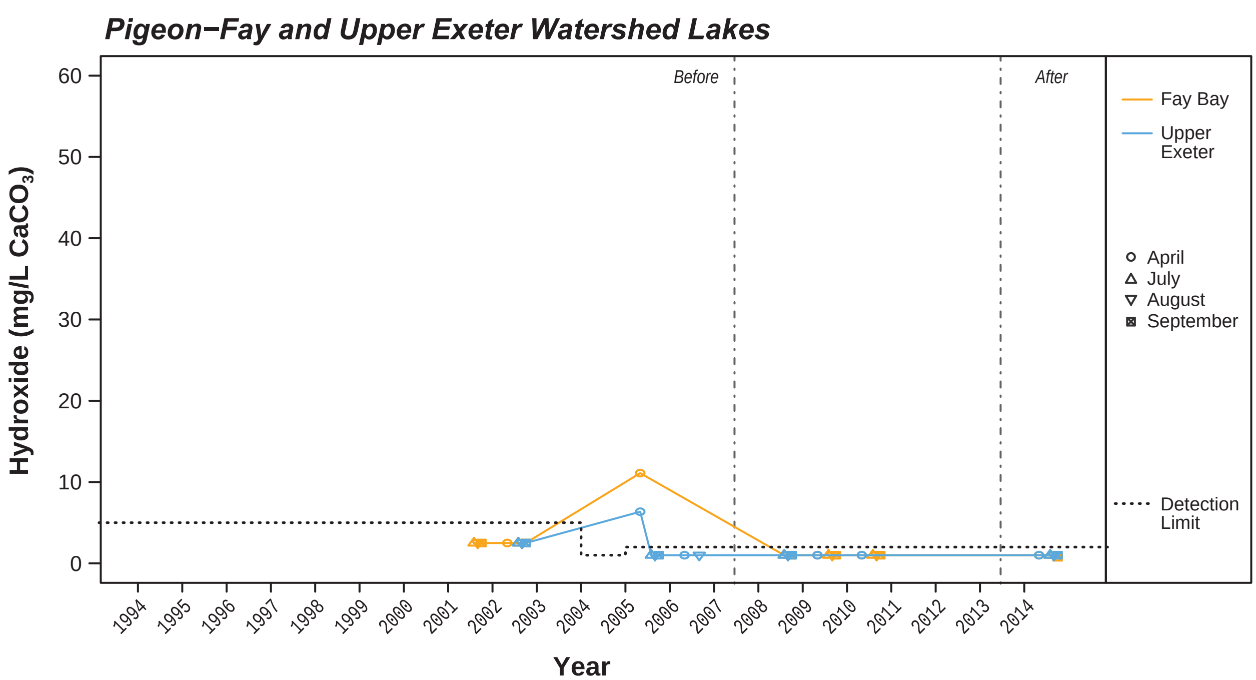


Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 6-51

Hydroxide Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

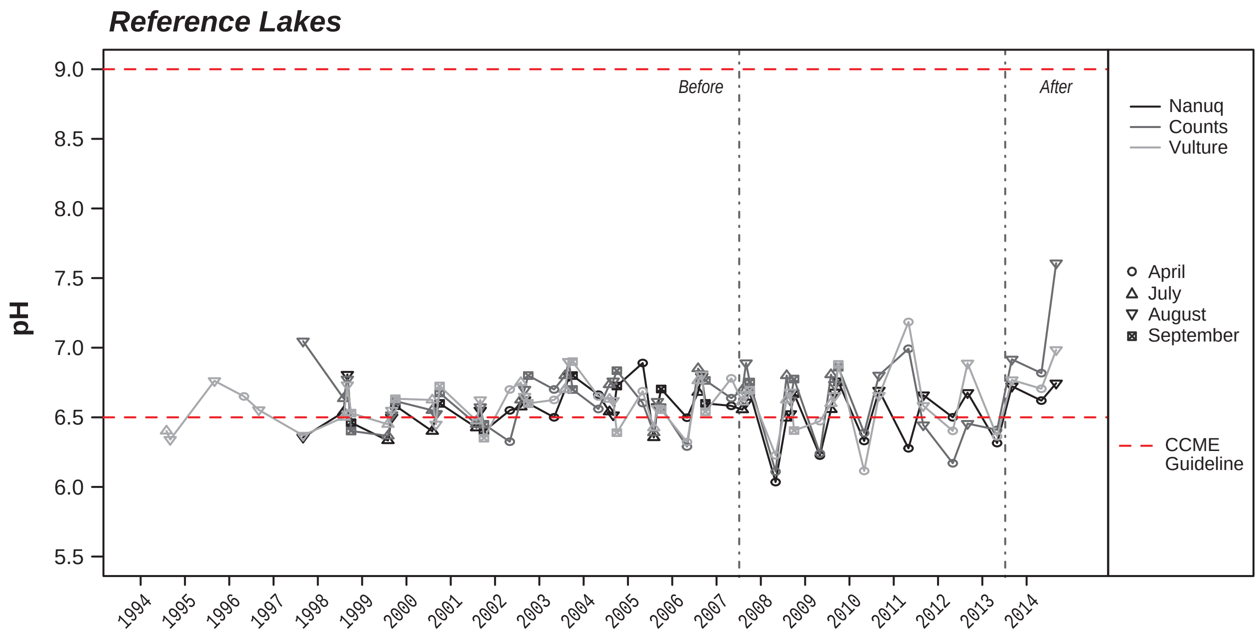


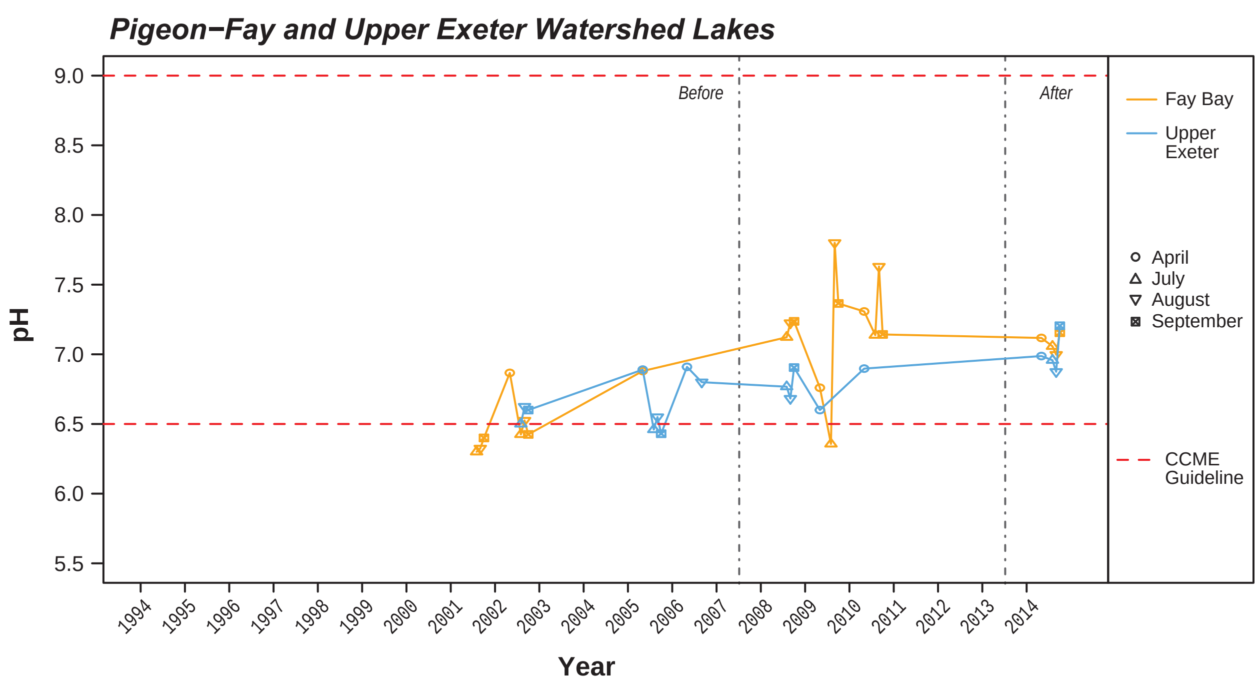


Note: For cases where detection limits varied between lakes and months, the lowest detection limit is shown.

Figure 6-52

pH at Pigeon
AEMP Lake Sites, 1994 to 2014



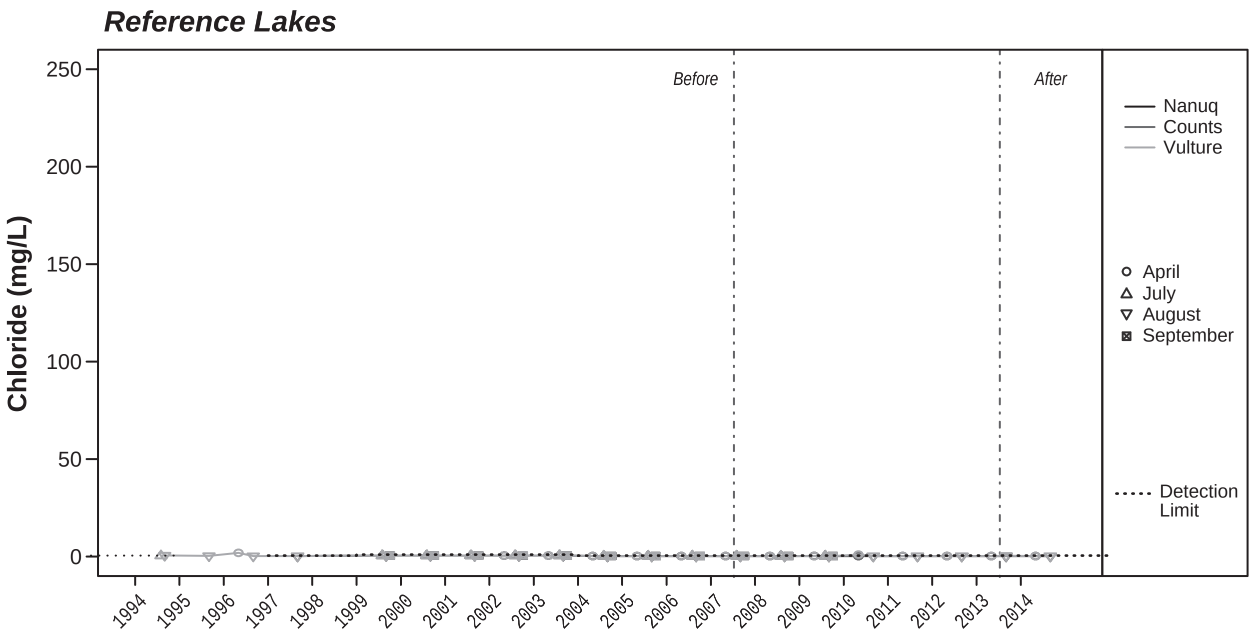


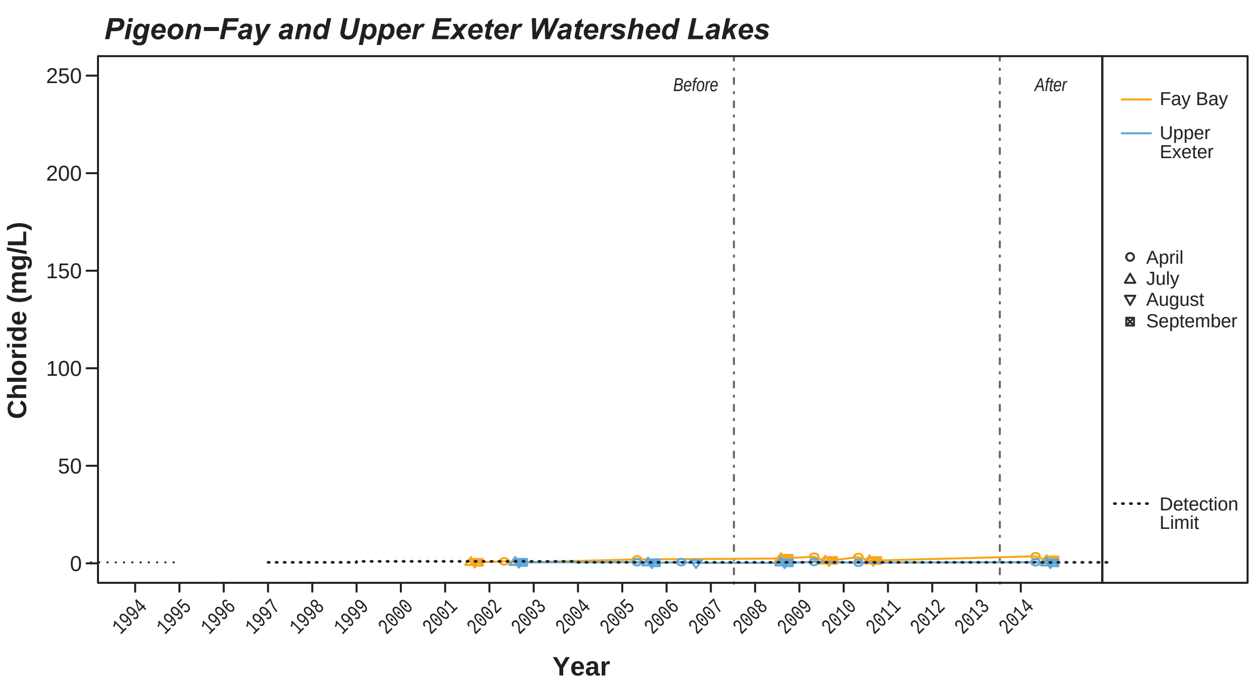
Note: CCME Guidelines are 6.5 - 9.0.

Figure 6-53

Chloride Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

ERM

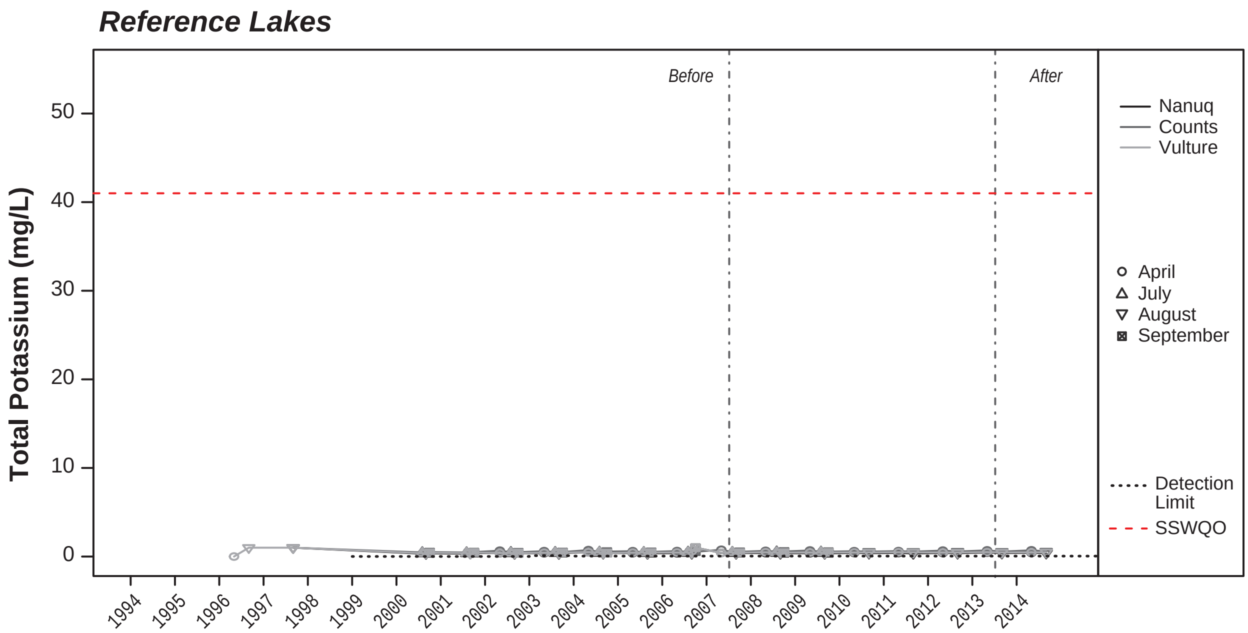


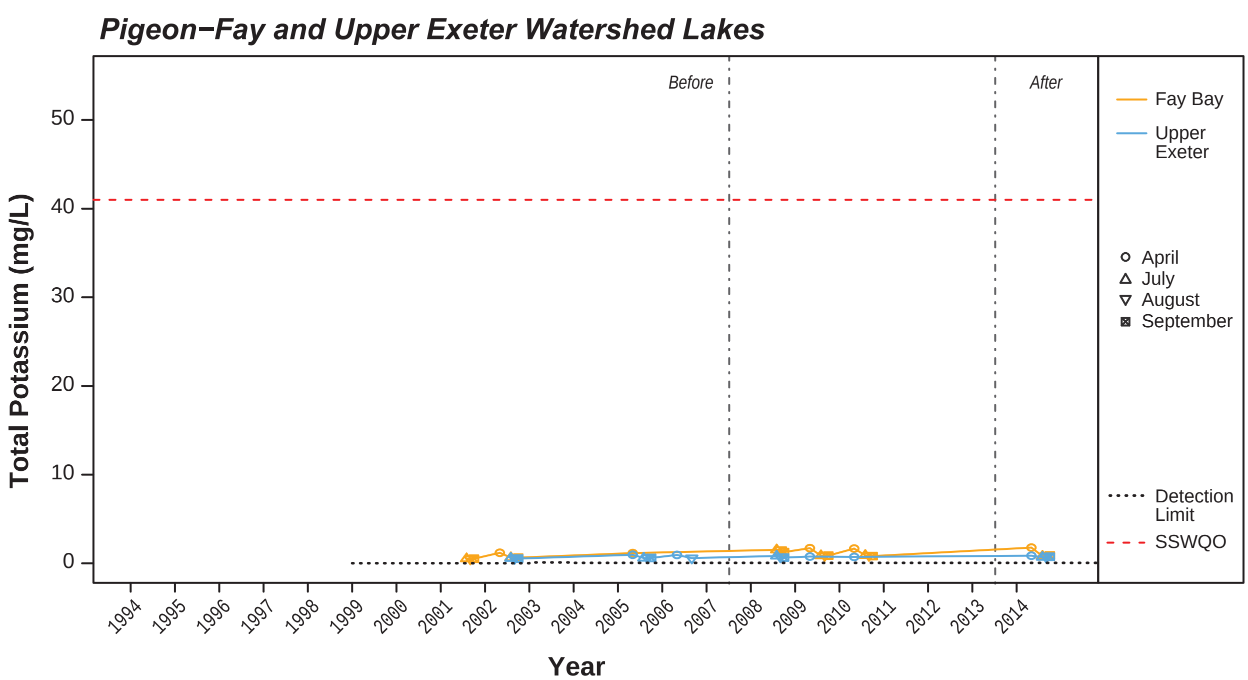


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
SSWQO = 116.6 X ln(Hardness) - 204.1 mg/L, where hardness = 10 - 160 mg/L.

Figure 6-54

Total Potassium Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

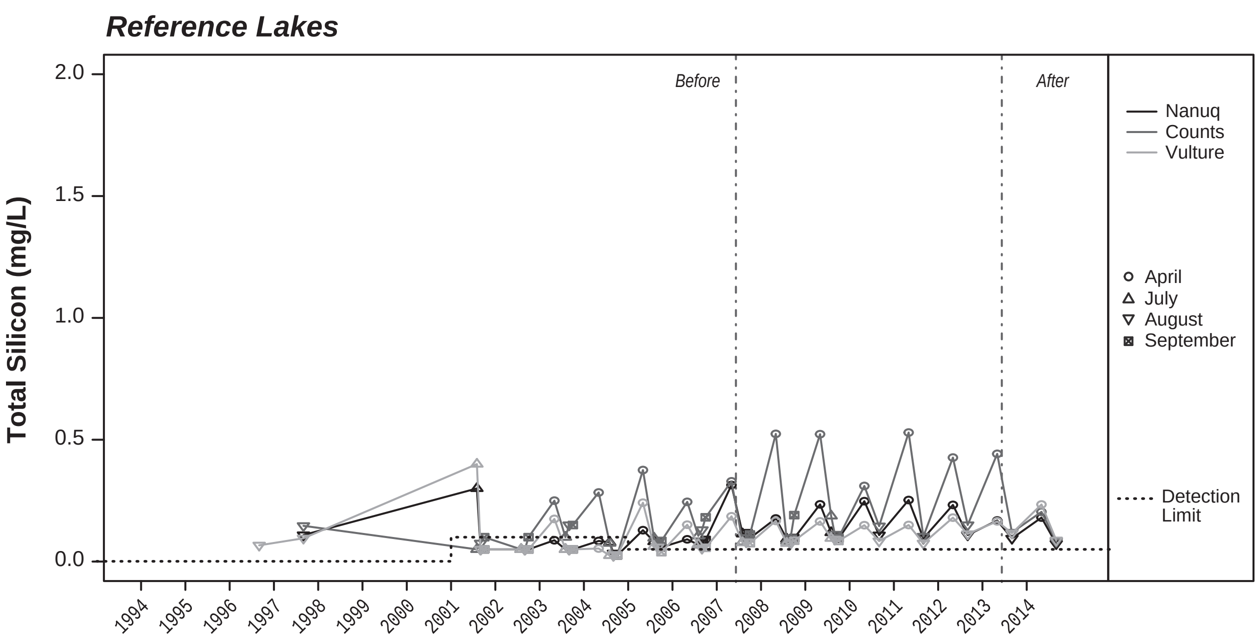


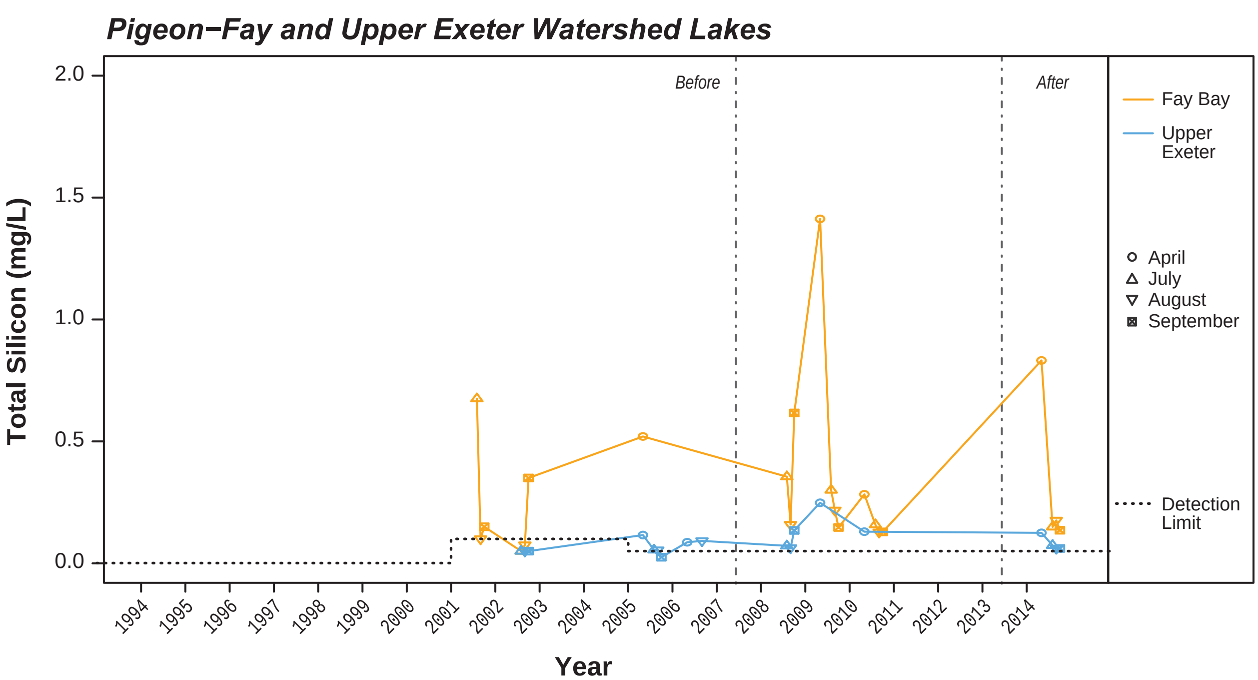


*Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
SSWQO = 41 mg/L.*

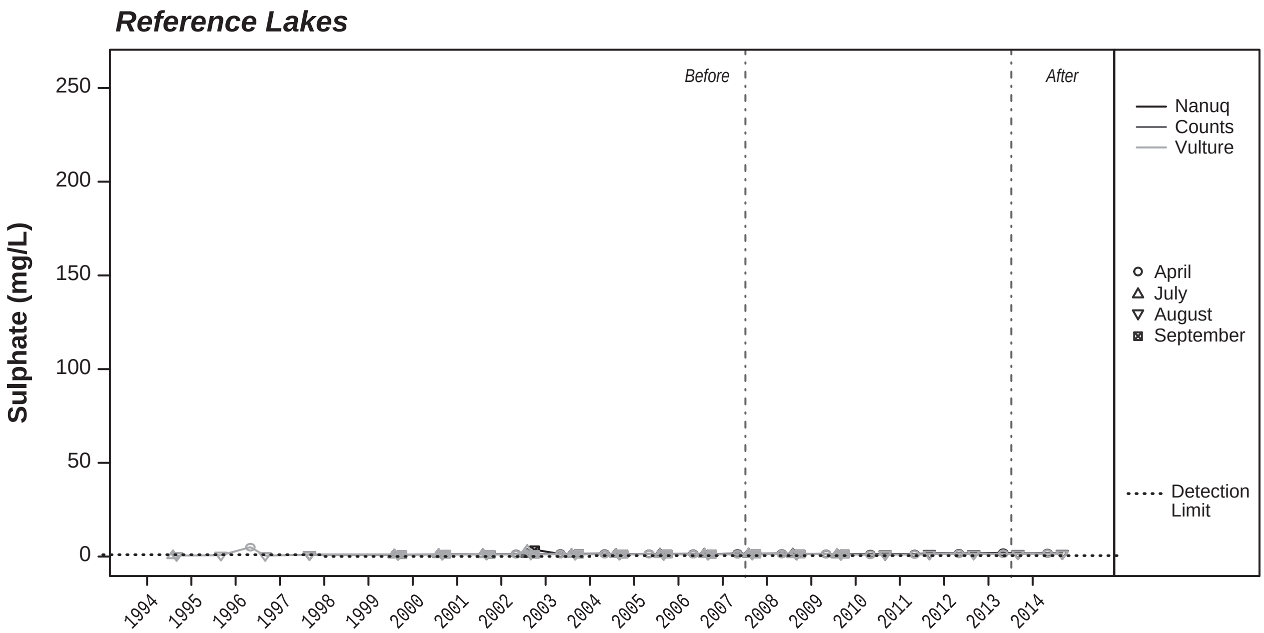
Figure 6-55

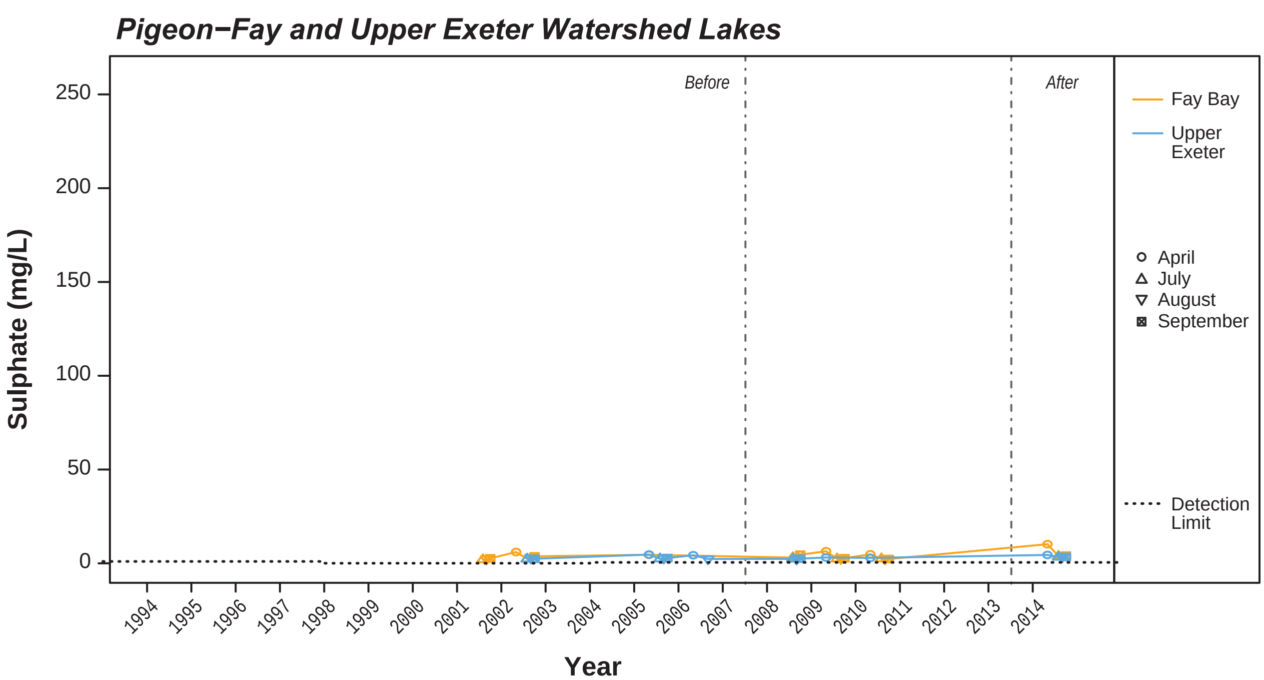
Total Silicon Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014





Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

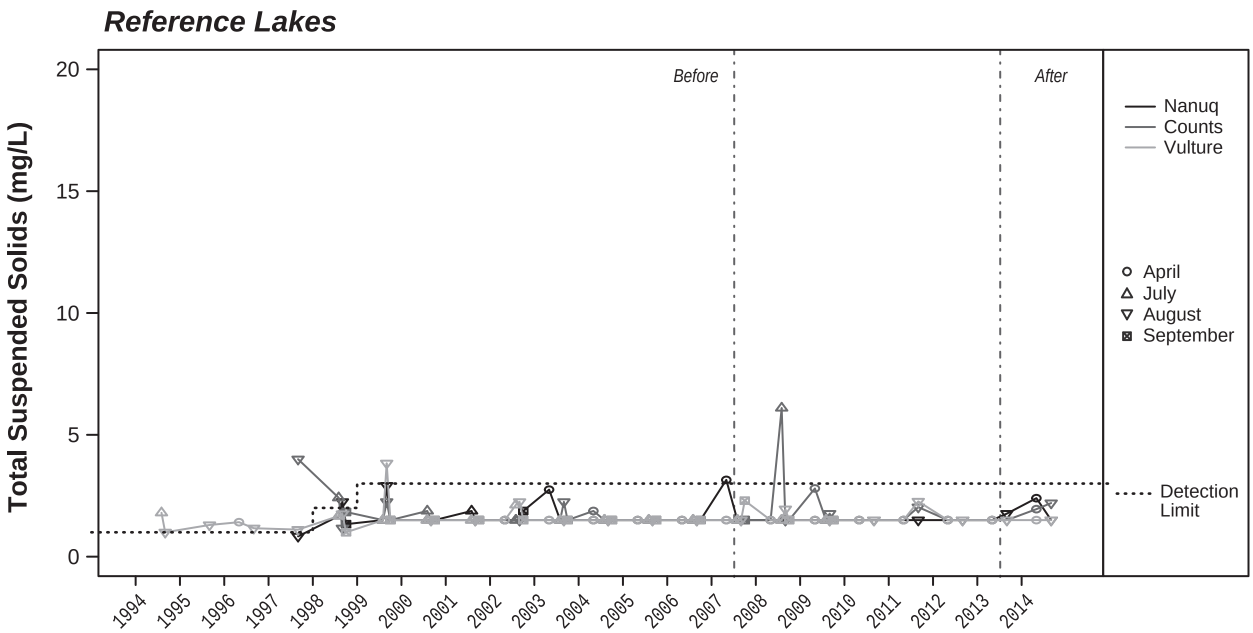


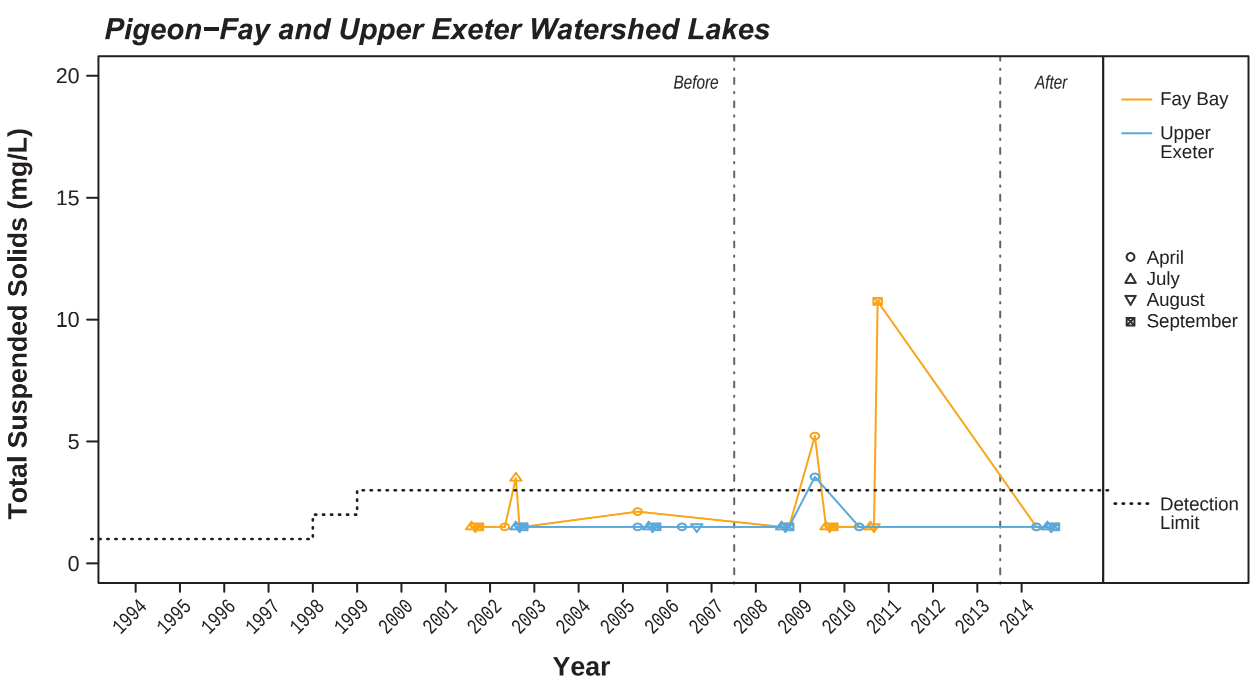


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
SSWQO = $e^{(0.9116 \times \ln(\text{Hardness}) + 1.712)}$ *mg/L, where hardness < 160 mg/L.*

Figure 6-57

Total Suspended Solids at Pigeon AEMP Lake Sites, 1994 to 2014

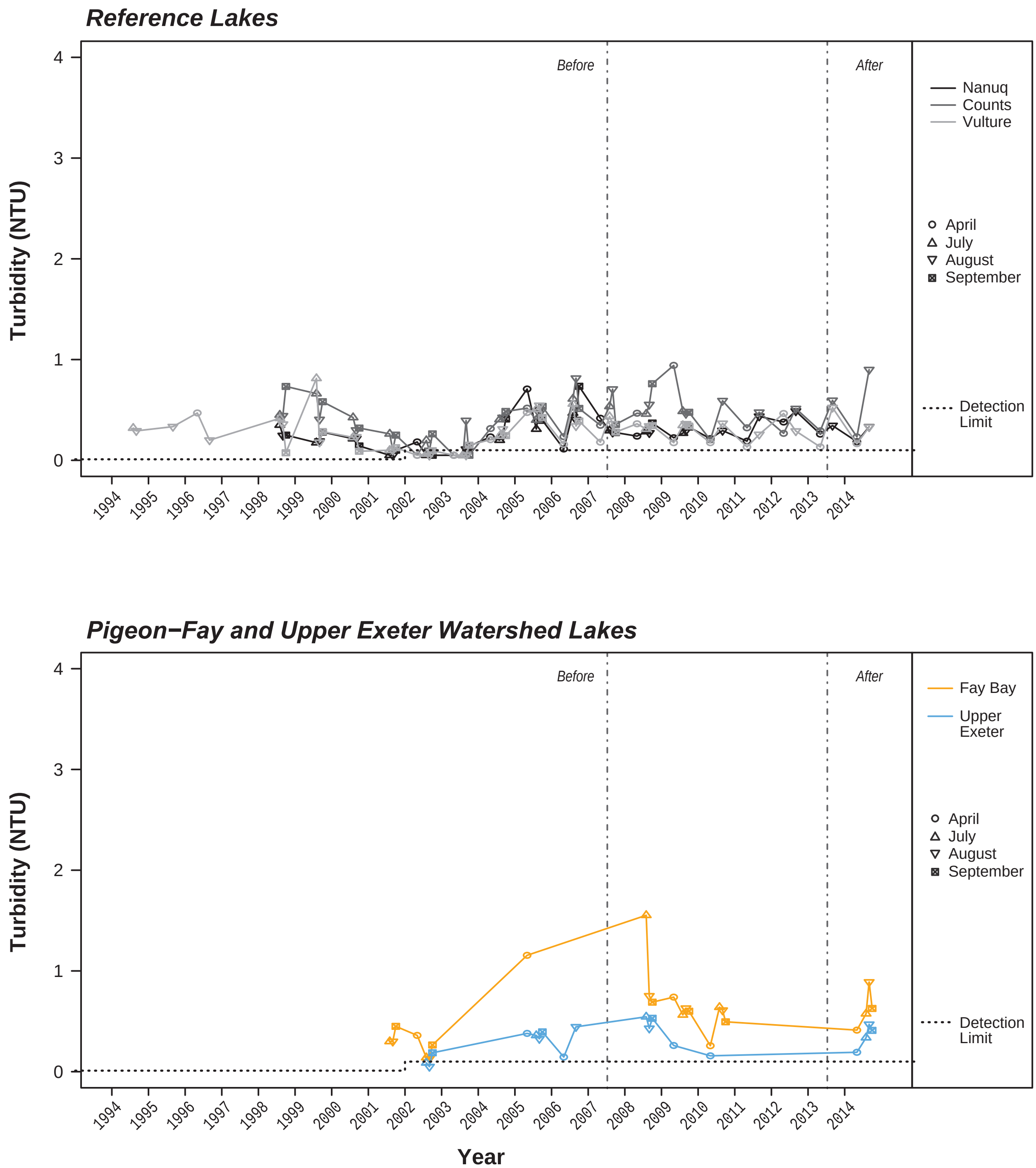




Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME Guideline = a maximum average increase of 5 mg/L from background levels (long term exposure).

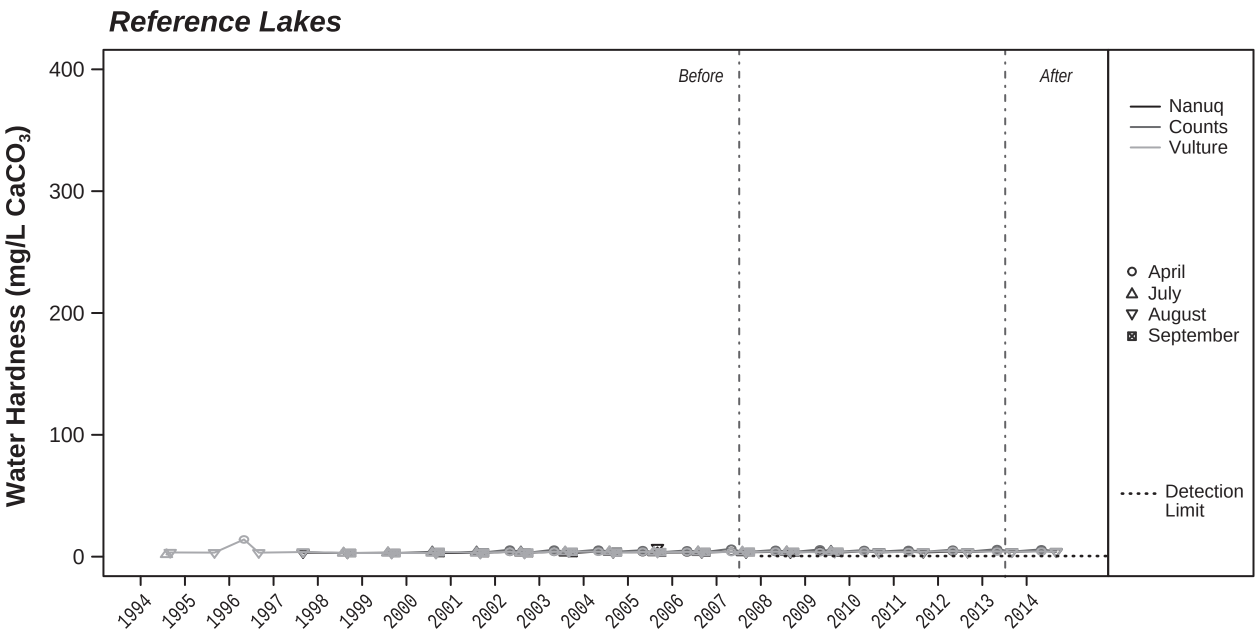
Figure 6-58

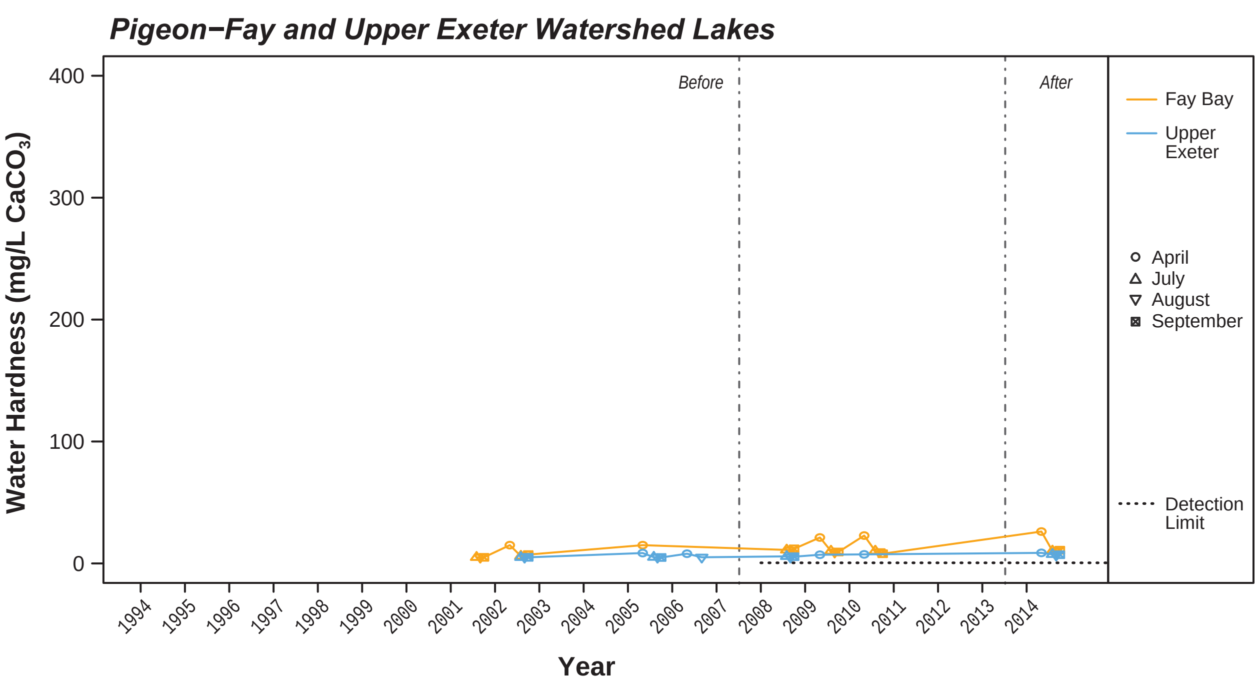
Turbidity at Pigeon
AEMP Lake Sites, 1994 to 2014



Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME Guideline = a maximum average increase of 2 NTUs from background levels (long term exposure).

Water Hardness at Pigeon AEMP Lake Sites, 1994 to 2014

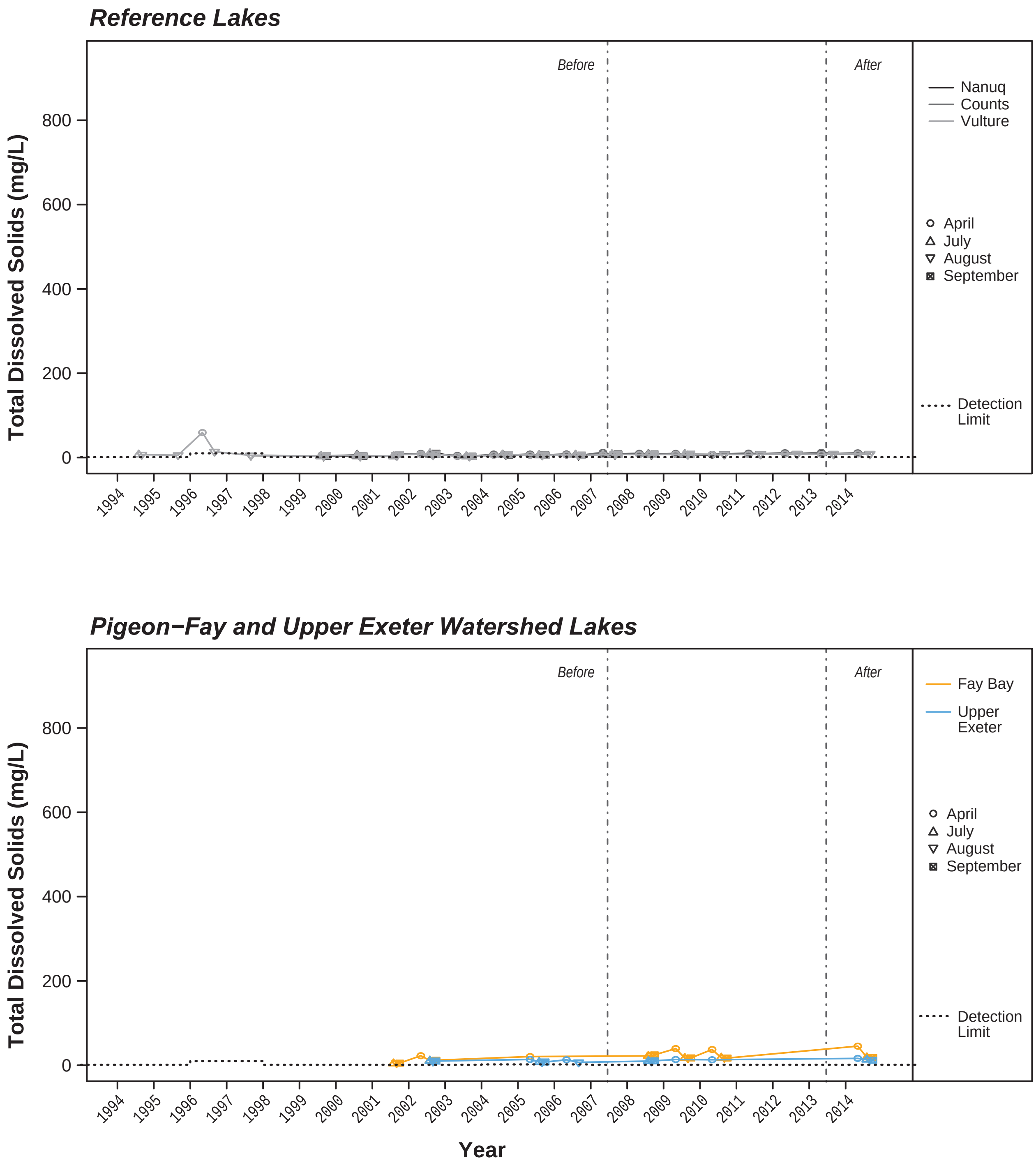




Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 6-60

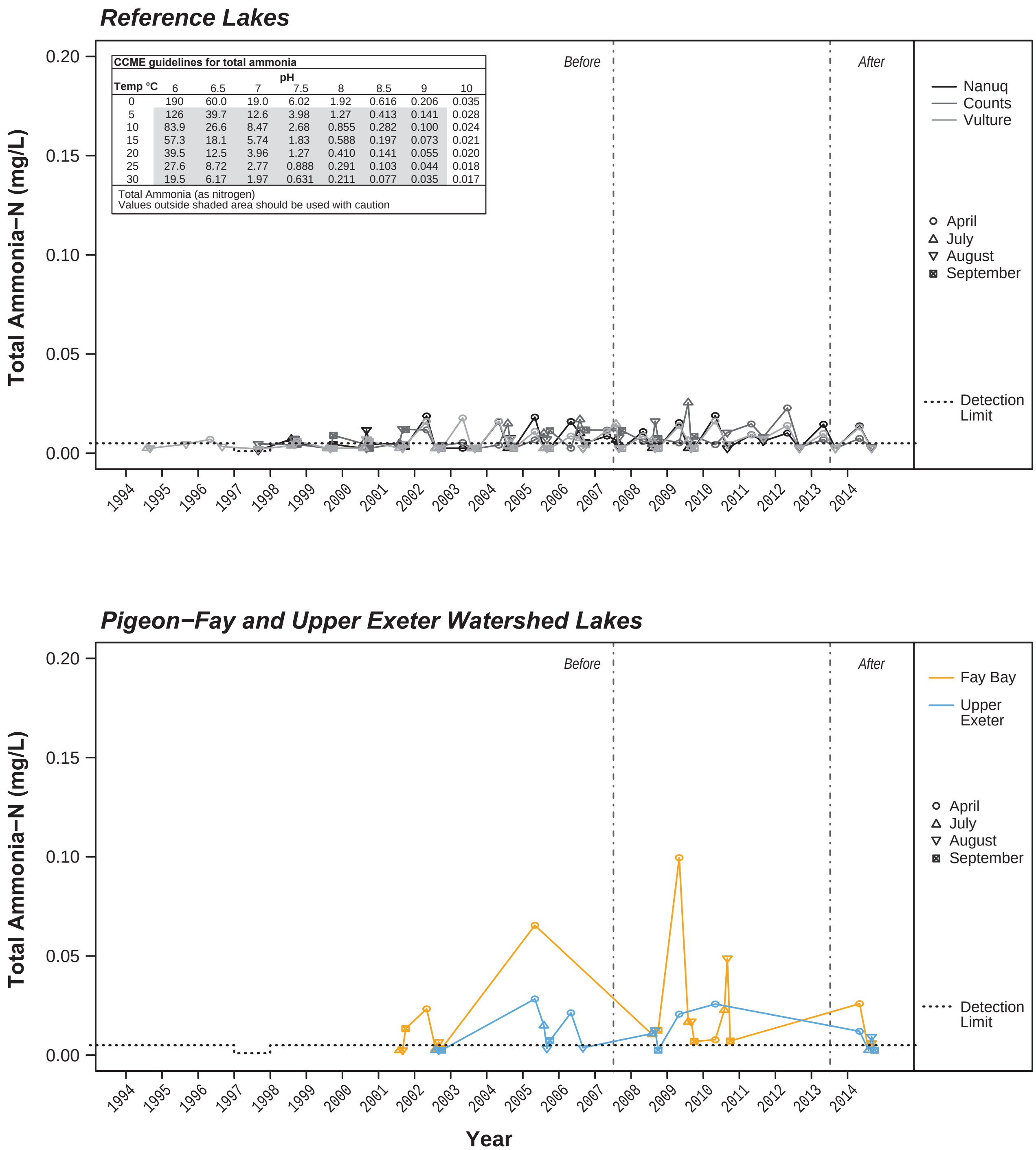
Total Dissolved Solids at Pigeon AEMP Lake Sites, 1994 to 2014



Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 6-61

Total Ammonia-N Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014



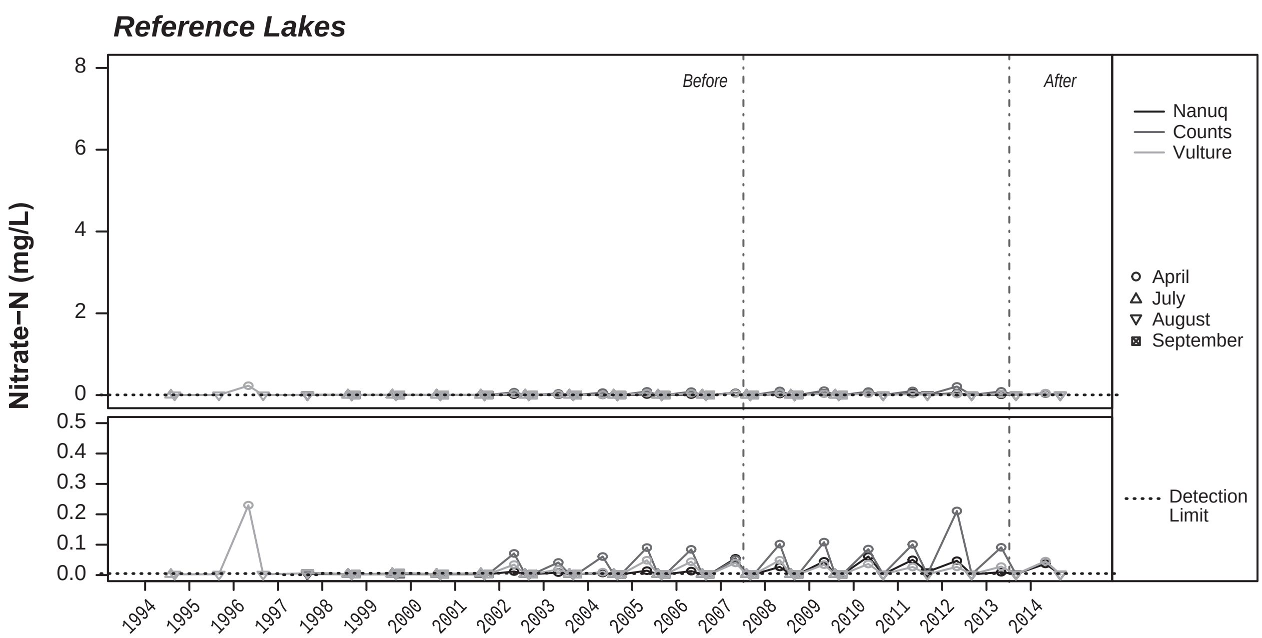
CCME guidelines for total ammonia

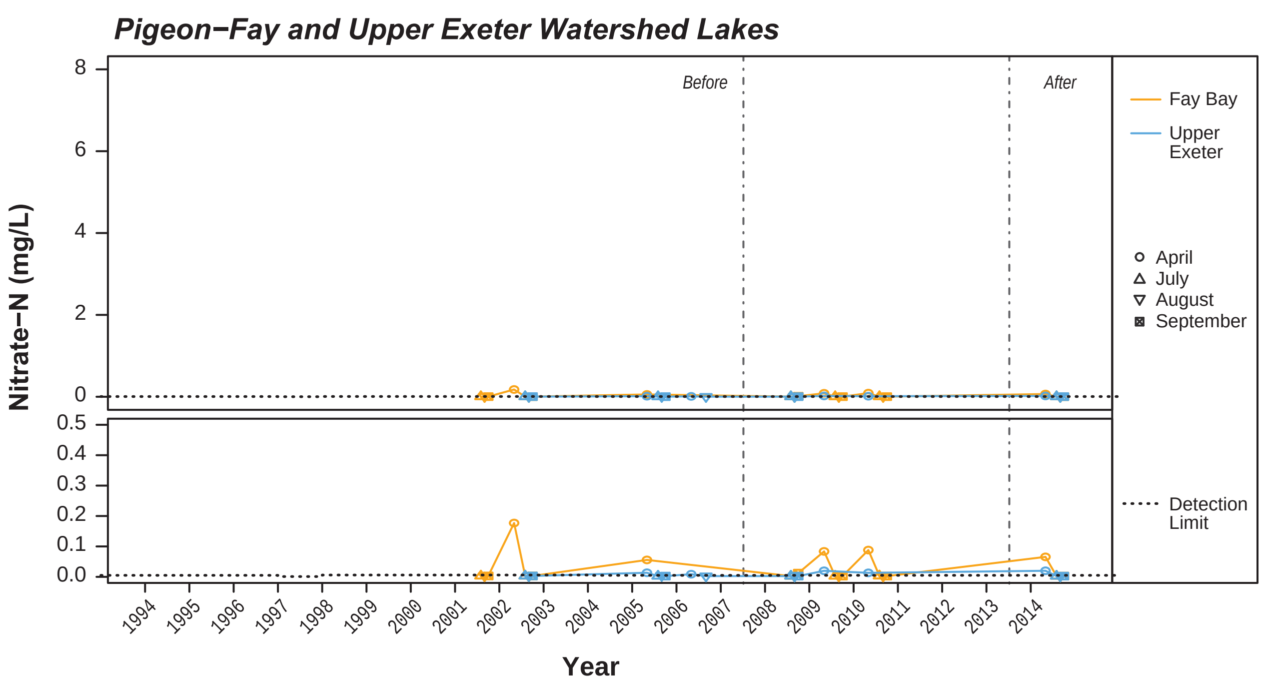
Temp °C	pH 6	6.5	7	7.5	8	8.5	9	10
0	190	60.0	19.0	6.02	1.92	0.616	0.206	0.035
5	126	39.7	12.6	3.98	1.27	0.413	0.141	0.028
10	83.9	26.6	8.47	2.68	0.855	0.282	0.100	0.024
15	57.3	18.1	5.74	1.83	0.588	0.197	0.073	0.021
20	39.5	12.5	3.96	1.27	0.410	0.141	0.055	0.020
25	27.6	8.72	2.77	0.888	0.291	0.103	0.044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

Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown. CCME guidelines are pH- and temperature-dependent (see inset).

Figure 6-62
Nitrate-N Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

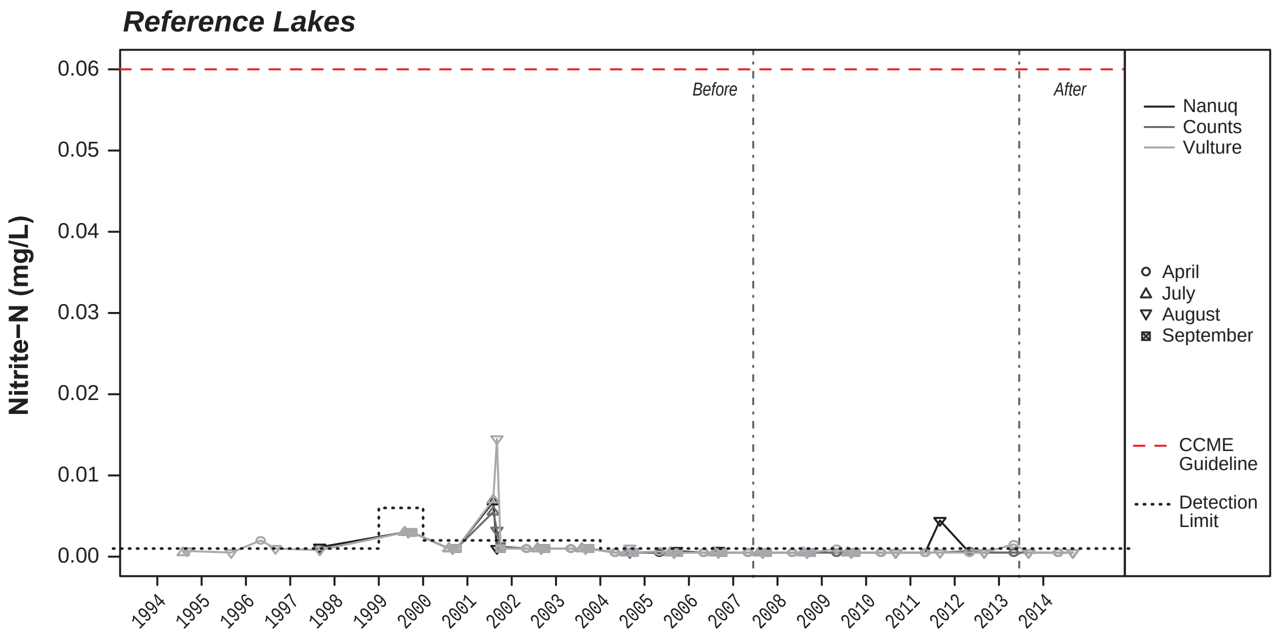


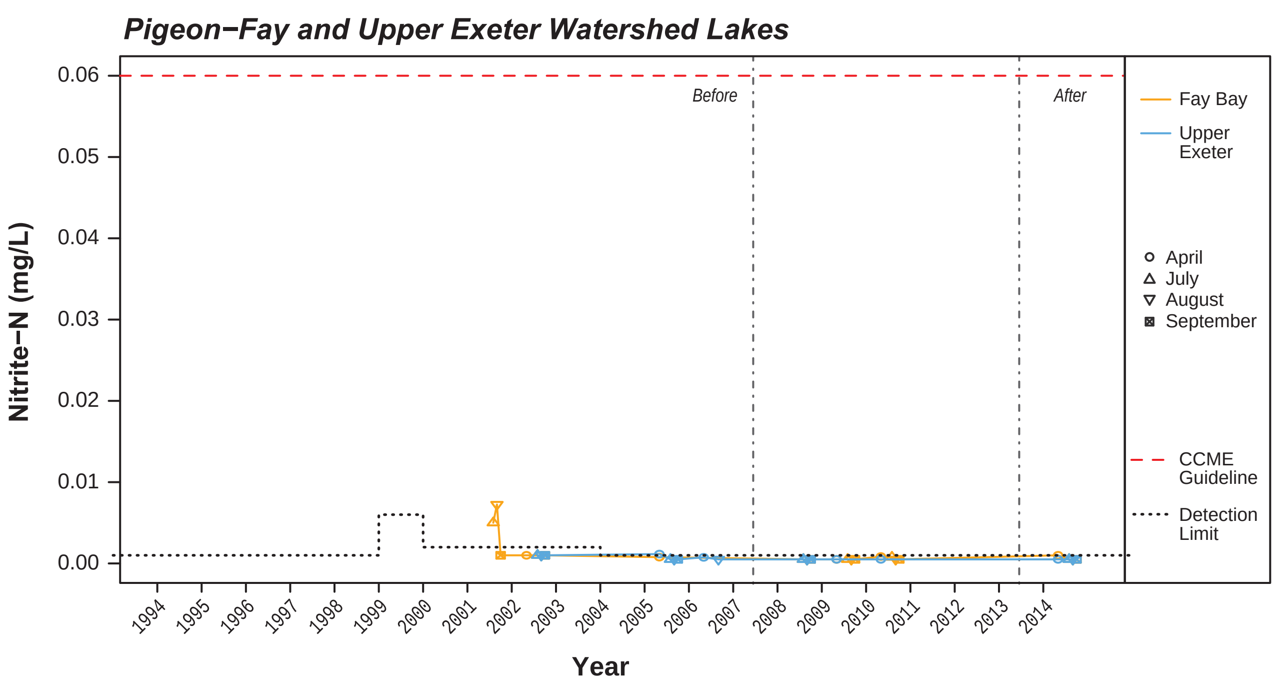


Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
SSWQO = $e^{0.9518 \times \ln(Hardness) - 2.032}$ mg/L, where hardness < 160mg/L.

Figure 6-63

Nitrite-N Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

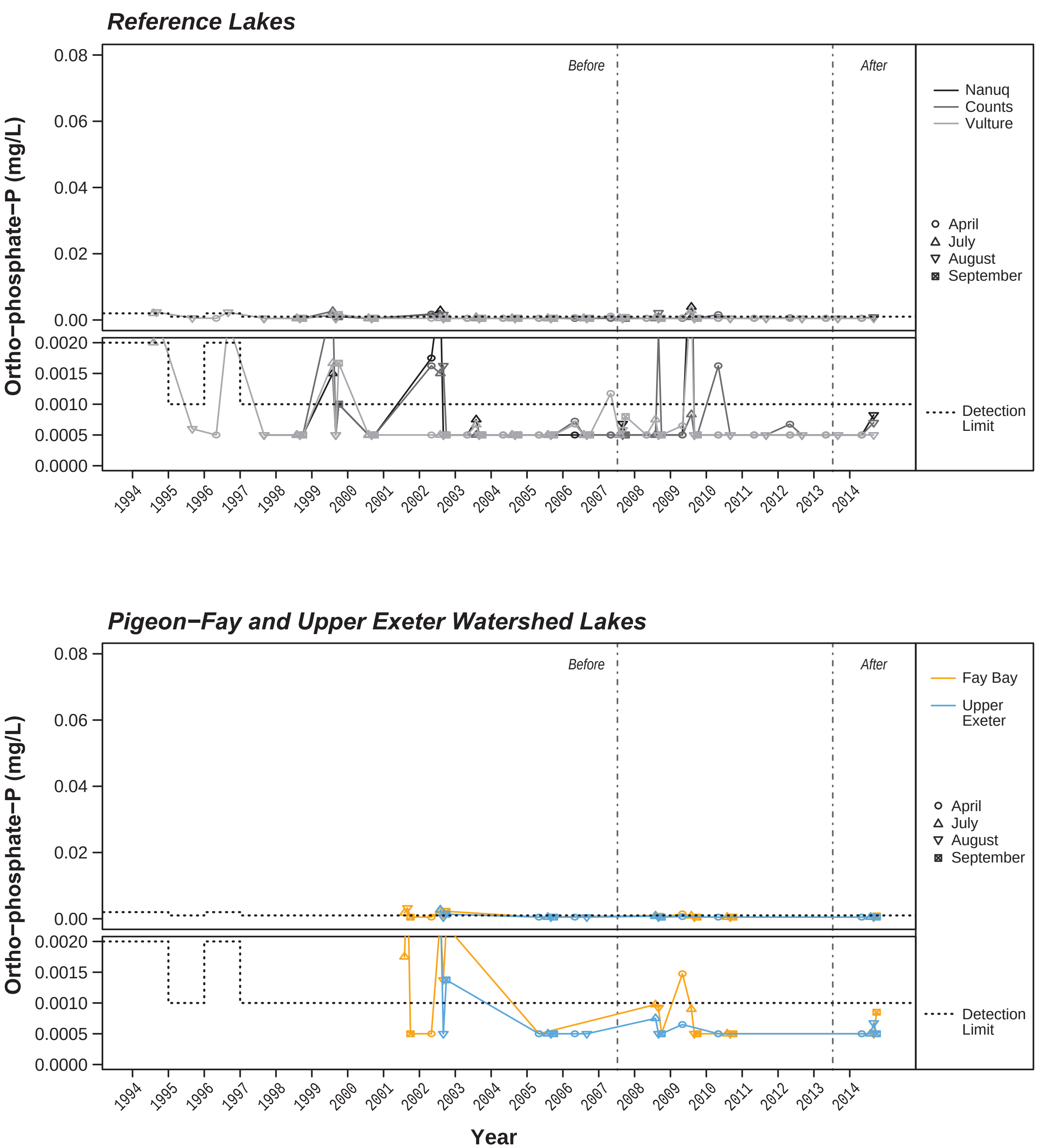




Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME Guideline = 0.06 mg/L.

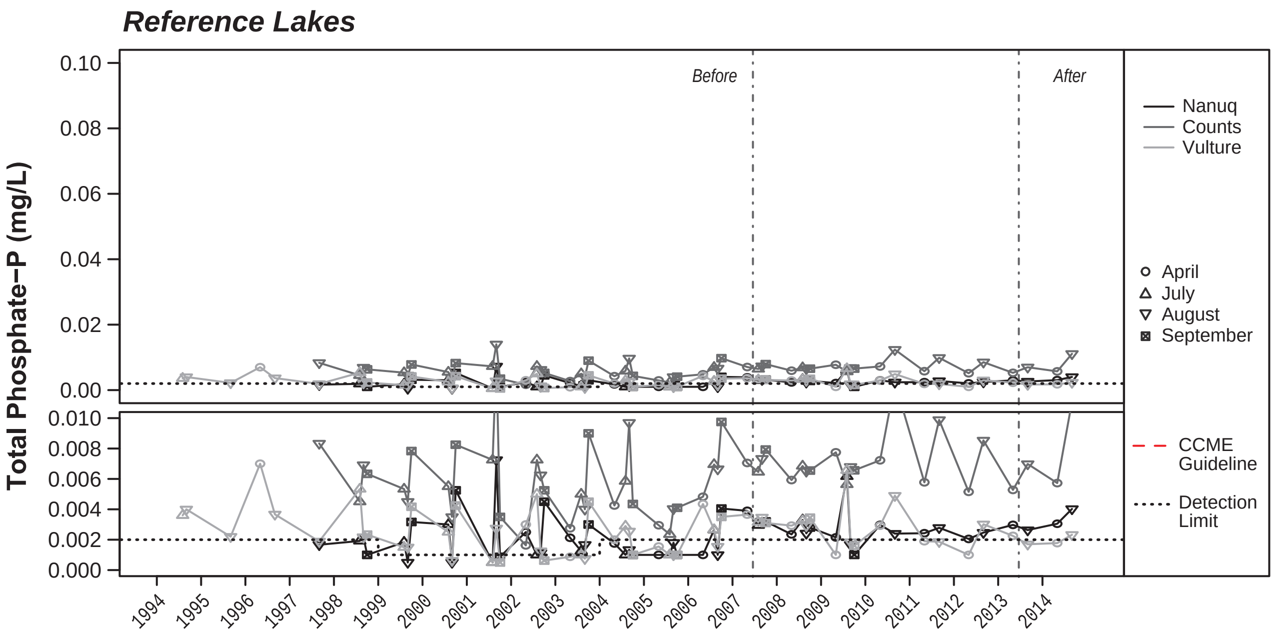
Figure 6-64

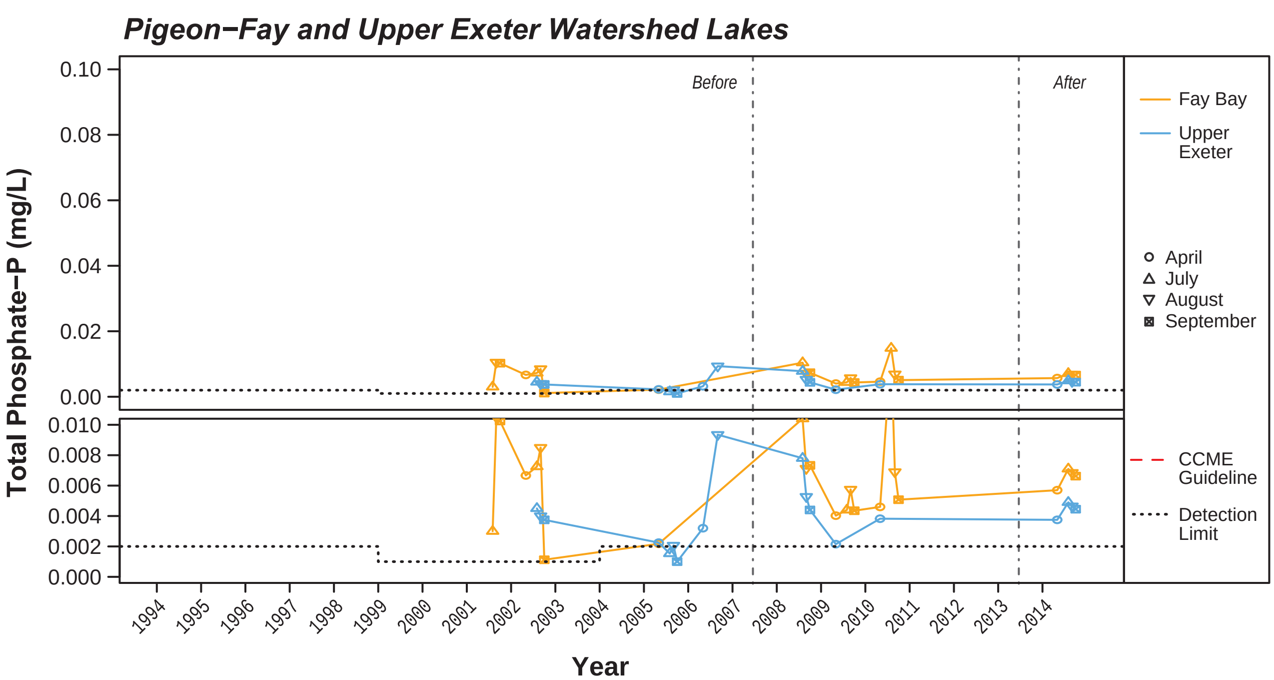
Orthophosphate-P Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014



Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Total Phosphate-P Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

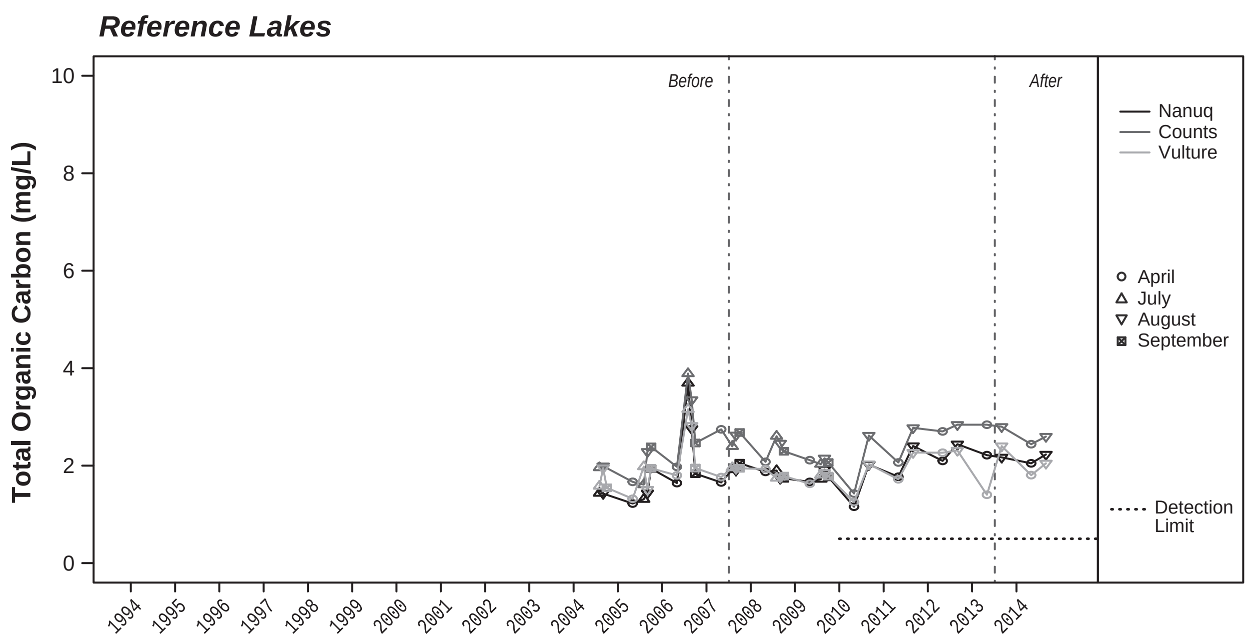


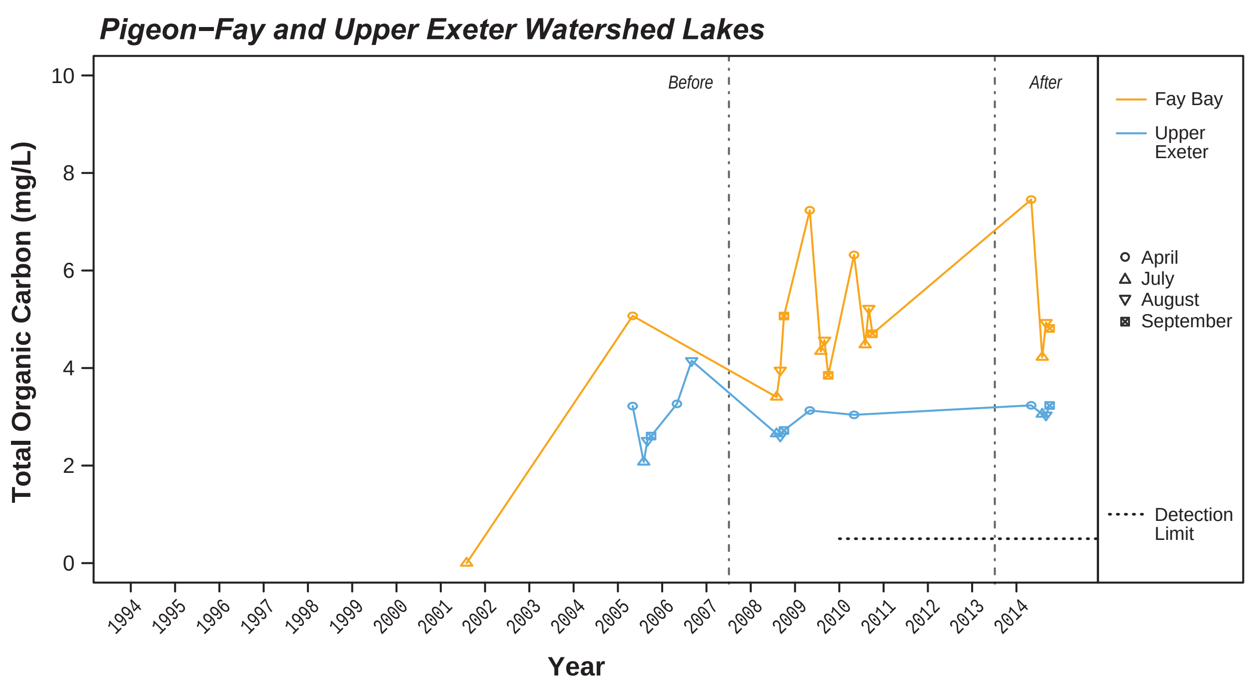


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
See Tables 2.3-3. and 2.3-4. for phosphorus trigger ranges and lake-specific benchmark concentrations.

Figure 6-66

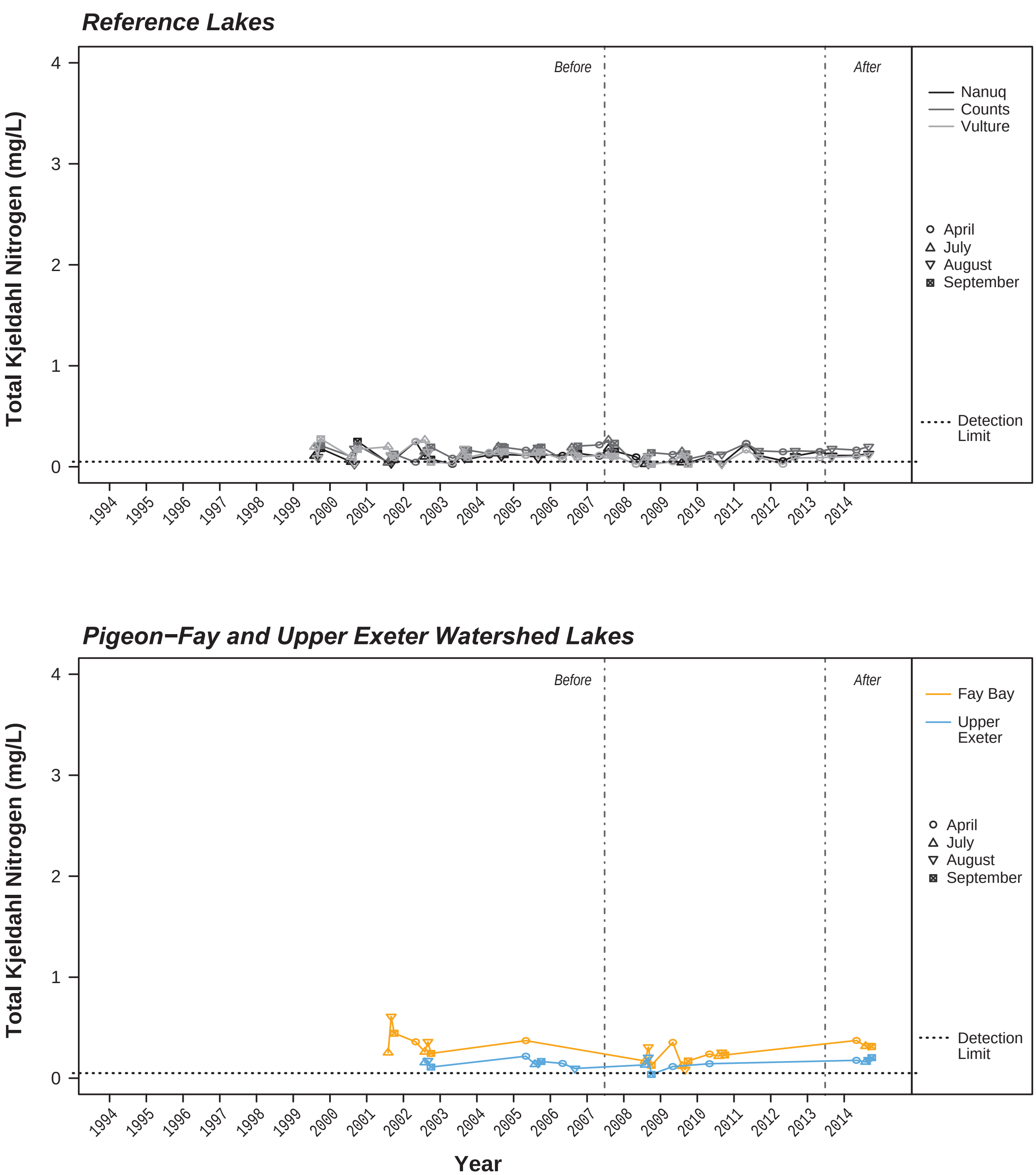
Total Organic Carbon Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014





Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

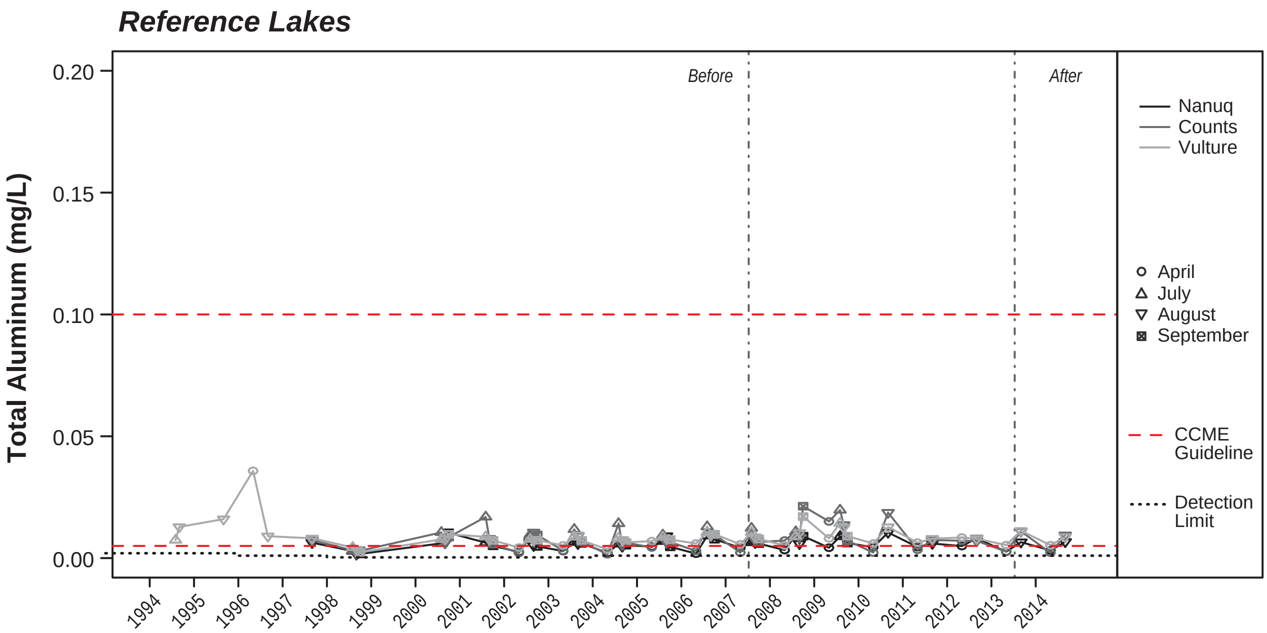
Figure 6-67
Total Kjeldhal Nitrogen Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

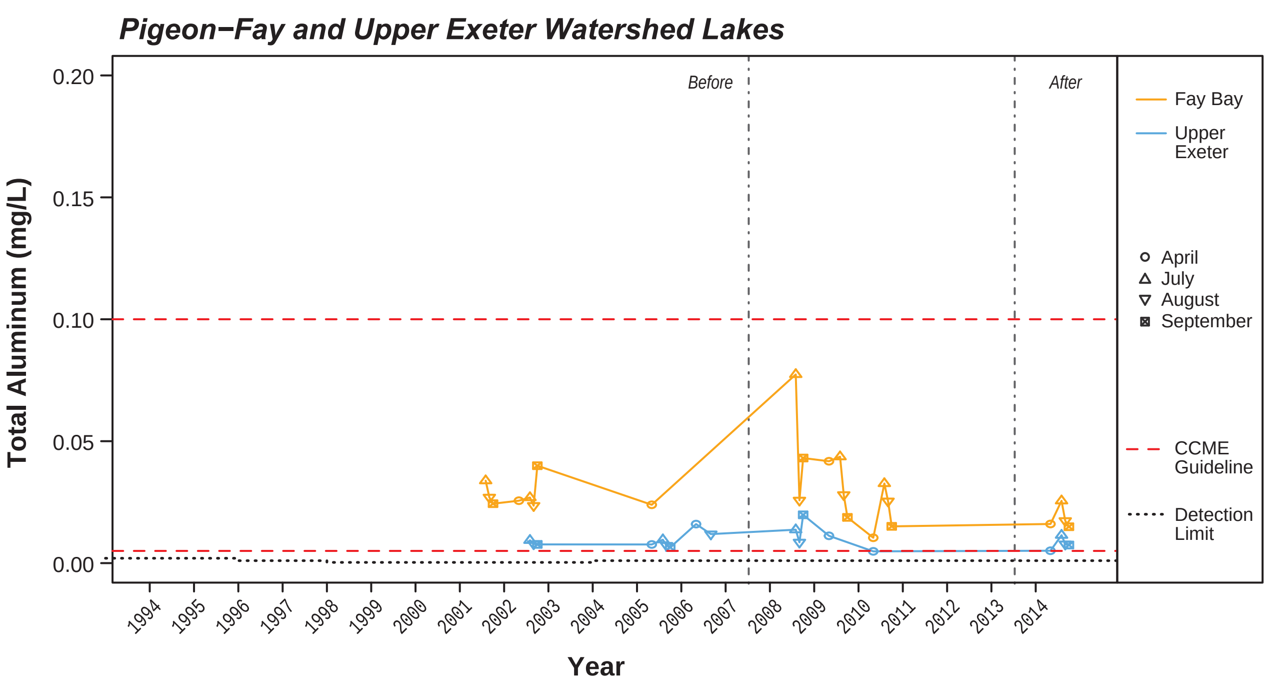


Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 6-68

Total Aluminum Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

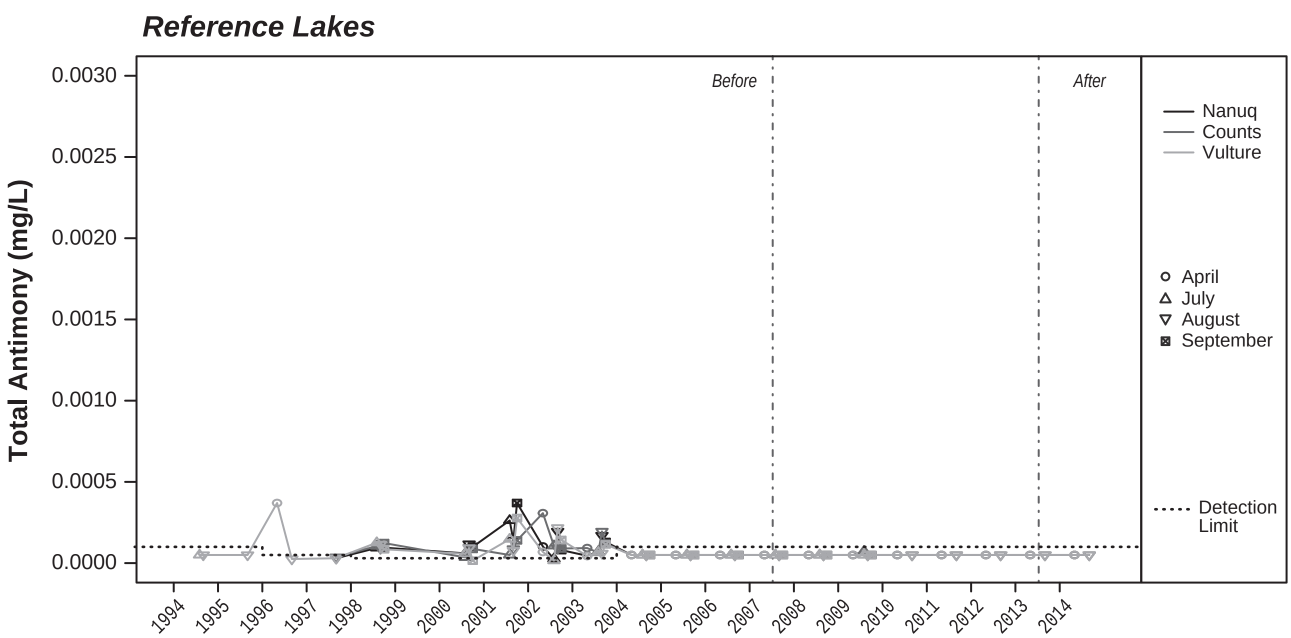


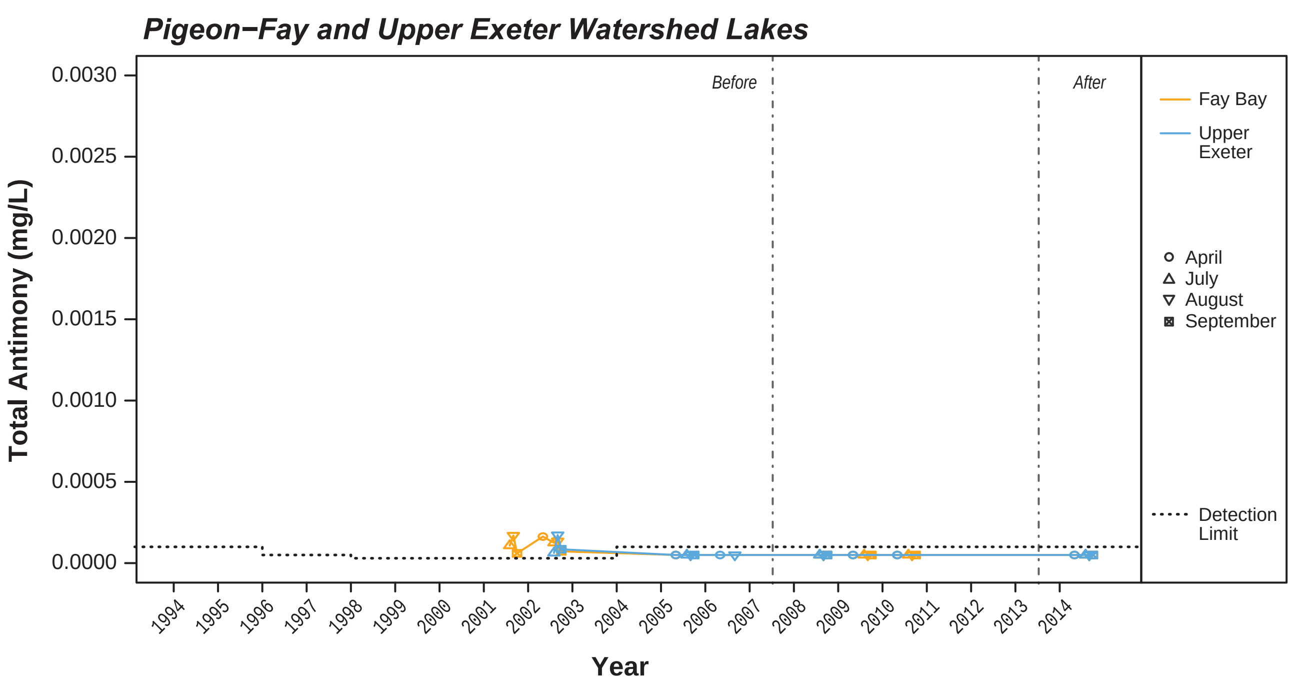


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME Guideline = 0.005 mg/L at pH < 6.5; 0.1 mg/L at pH ≥ 6.5.

Figure 6-69

Total Antimony Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

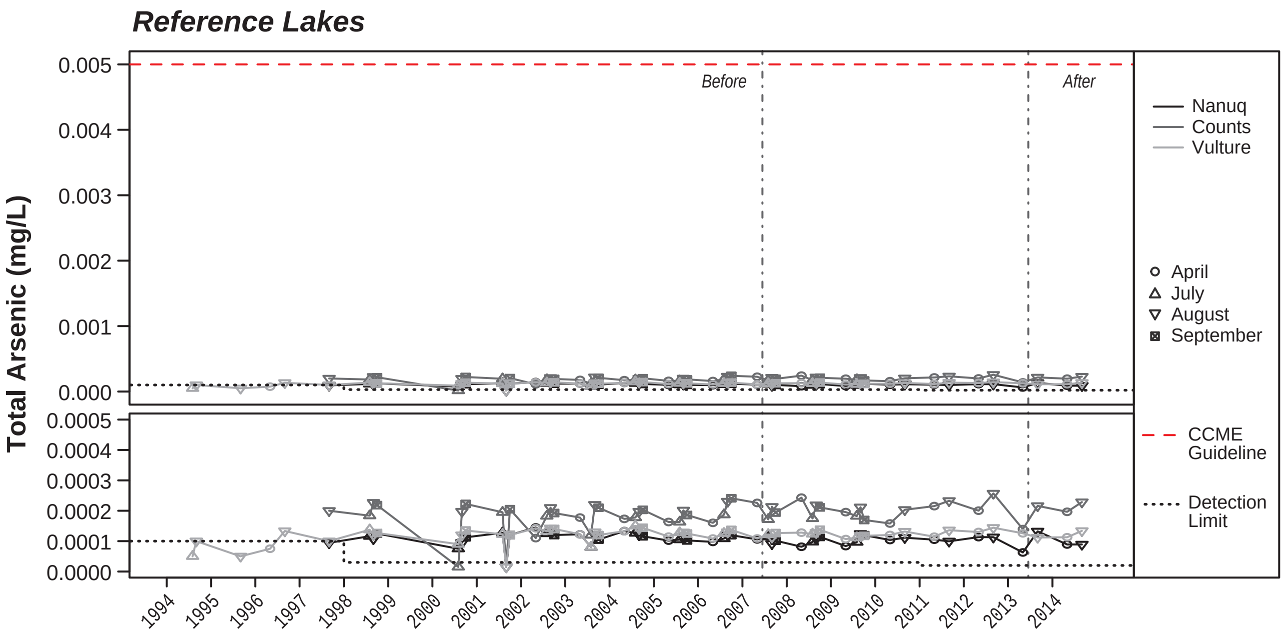


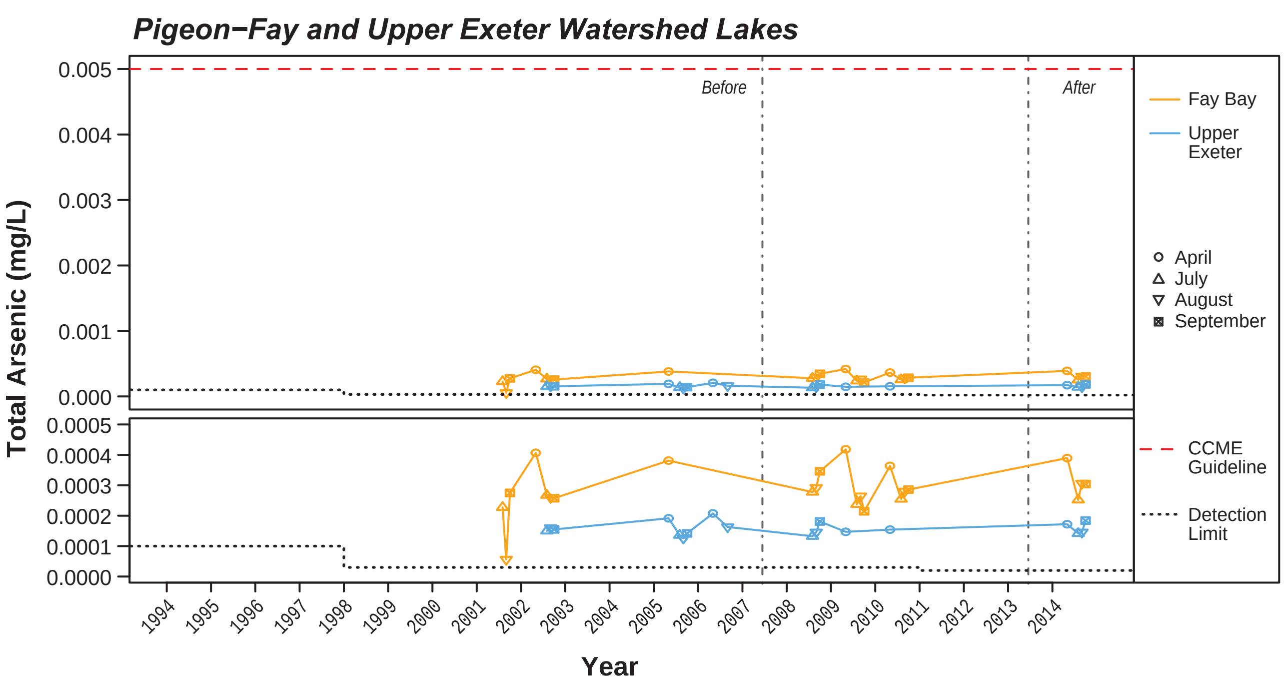


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
Water quality benchmark (Fletcher et al. 1996) = 0.02 mg/L.

Figure 6-70

Total Arsenic Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014



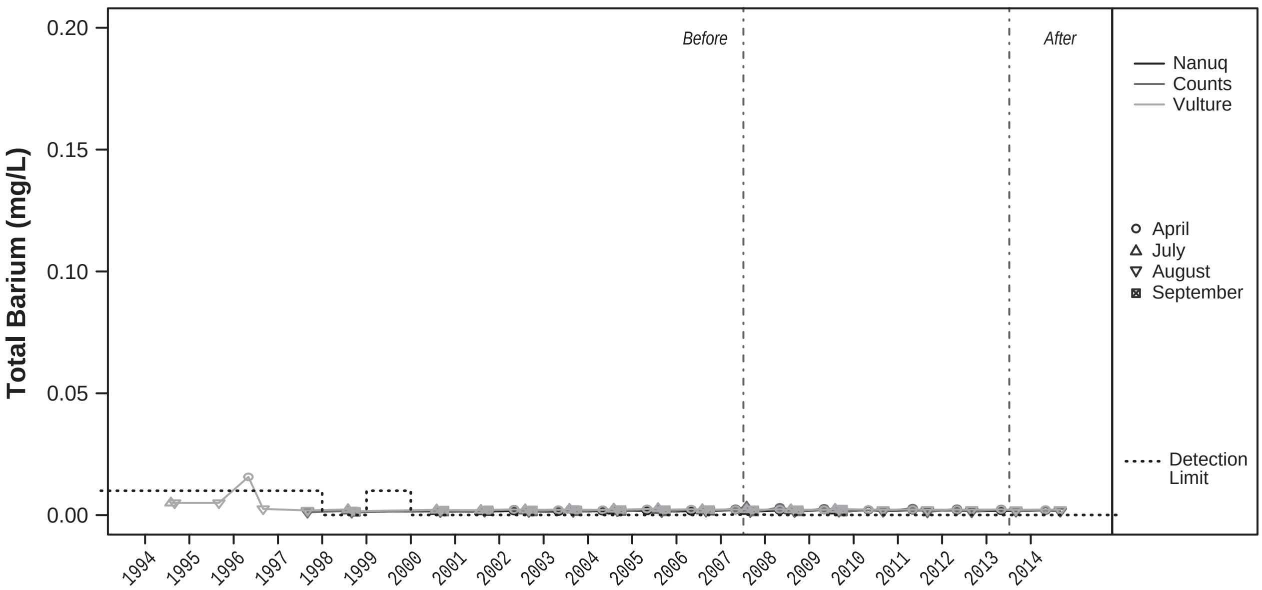


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME Guideline = 0.005 mg/L.

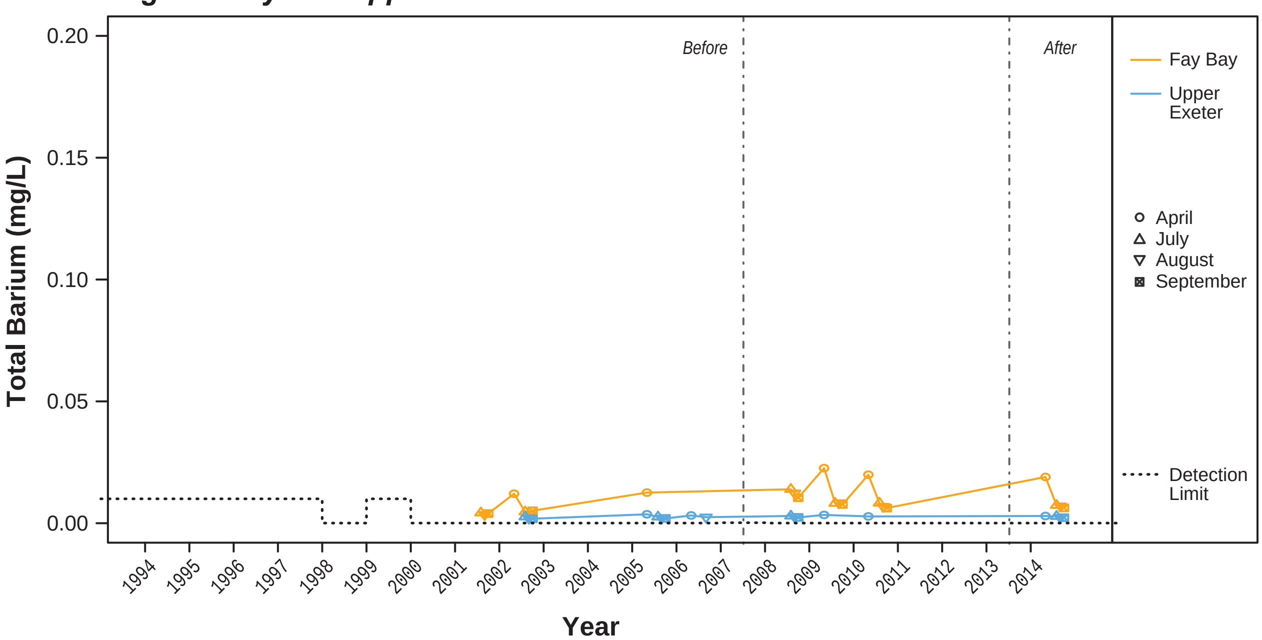
Figure 6-71

Total Barium Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

Reference Lakes



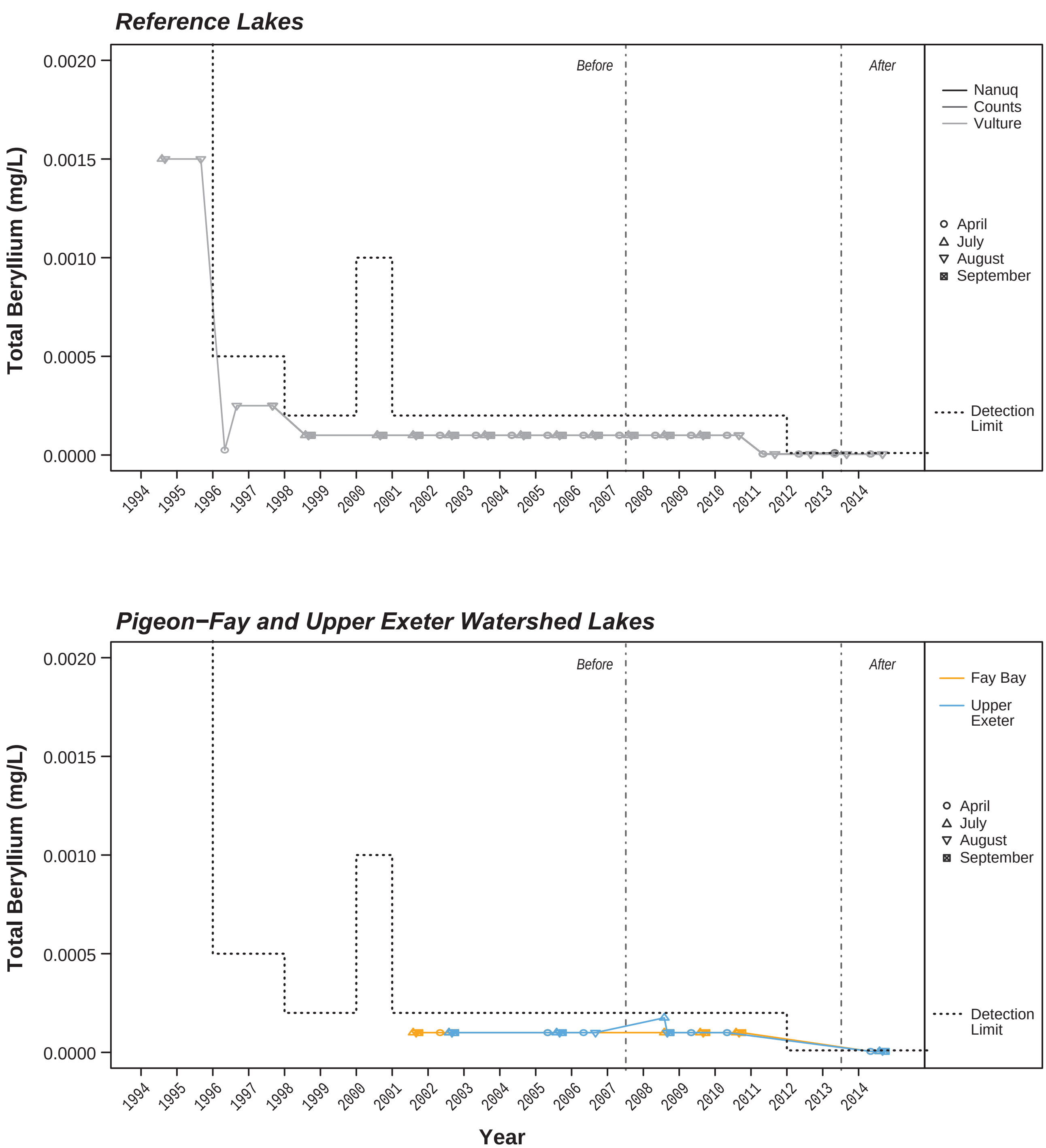
Pigeon−Fay and Upper Exeter Watershed Lakes



Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
Water quality benchmark (Haywood and Drinnan 1983) = 1 mg/L.

Figure 6-72

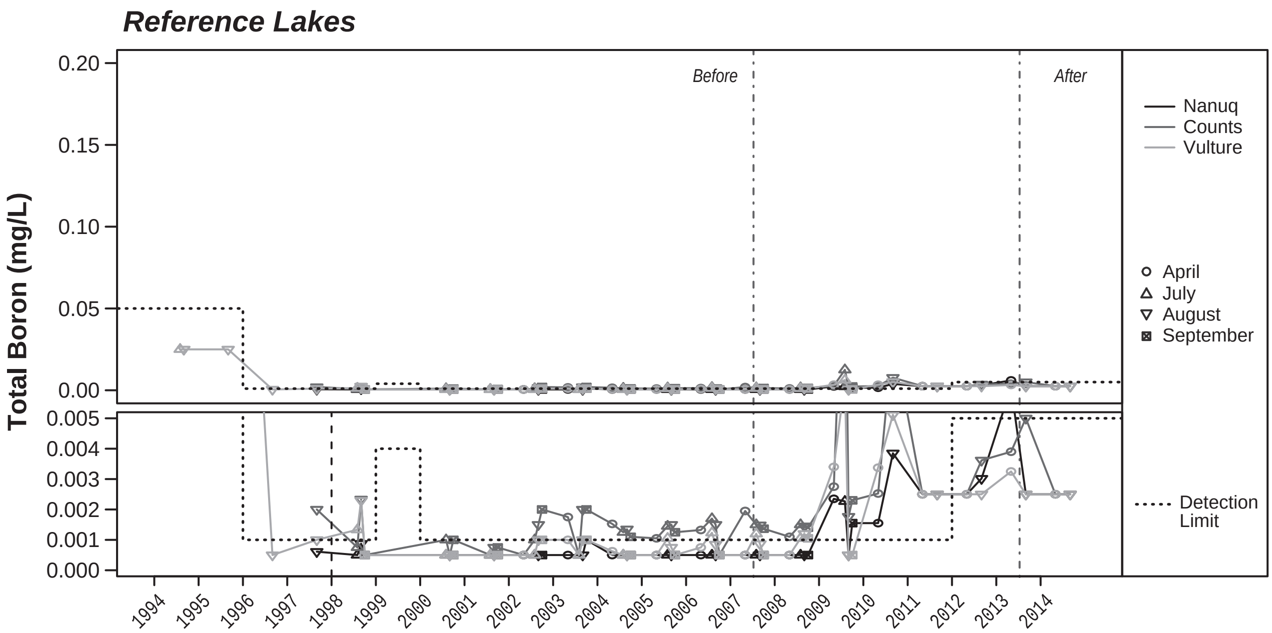
Total Beryllium Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

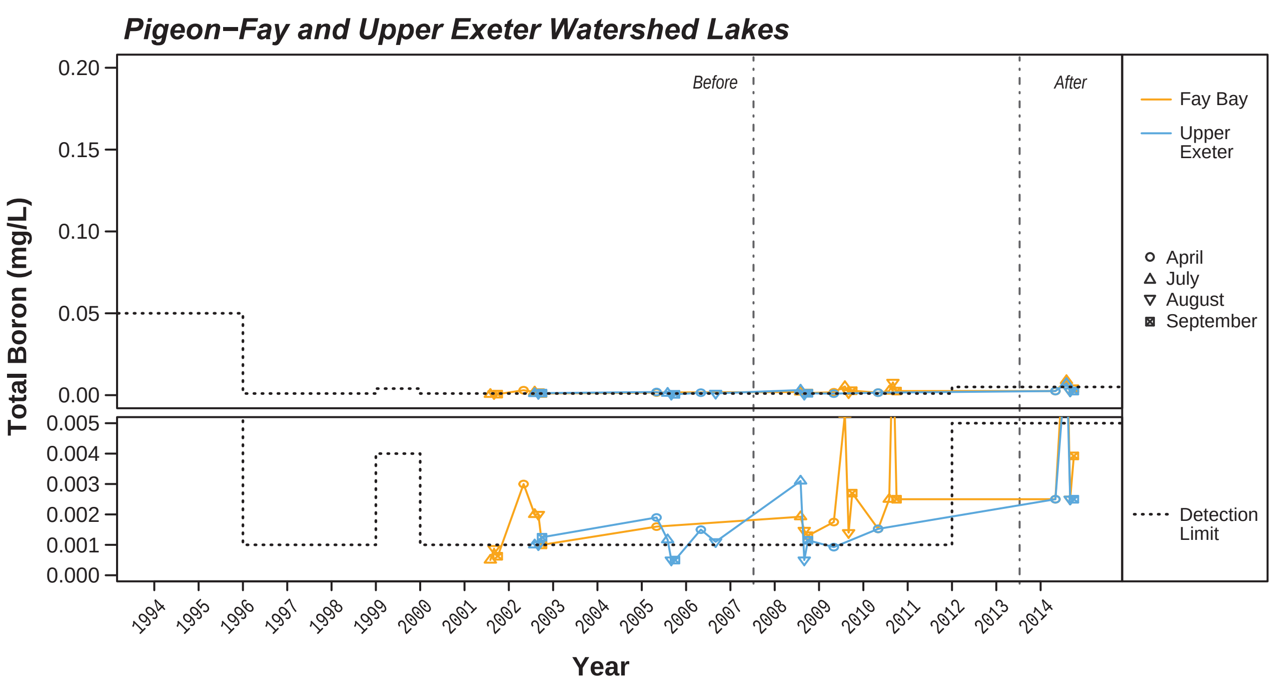


Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 6-73

Total Boron Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

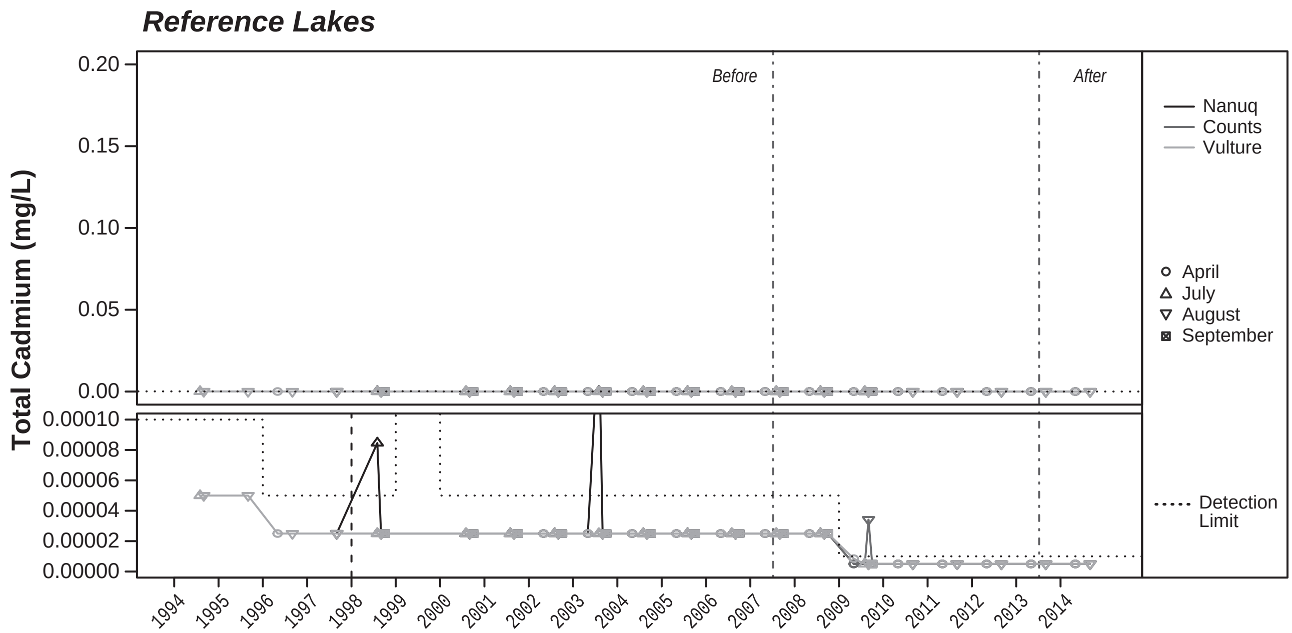


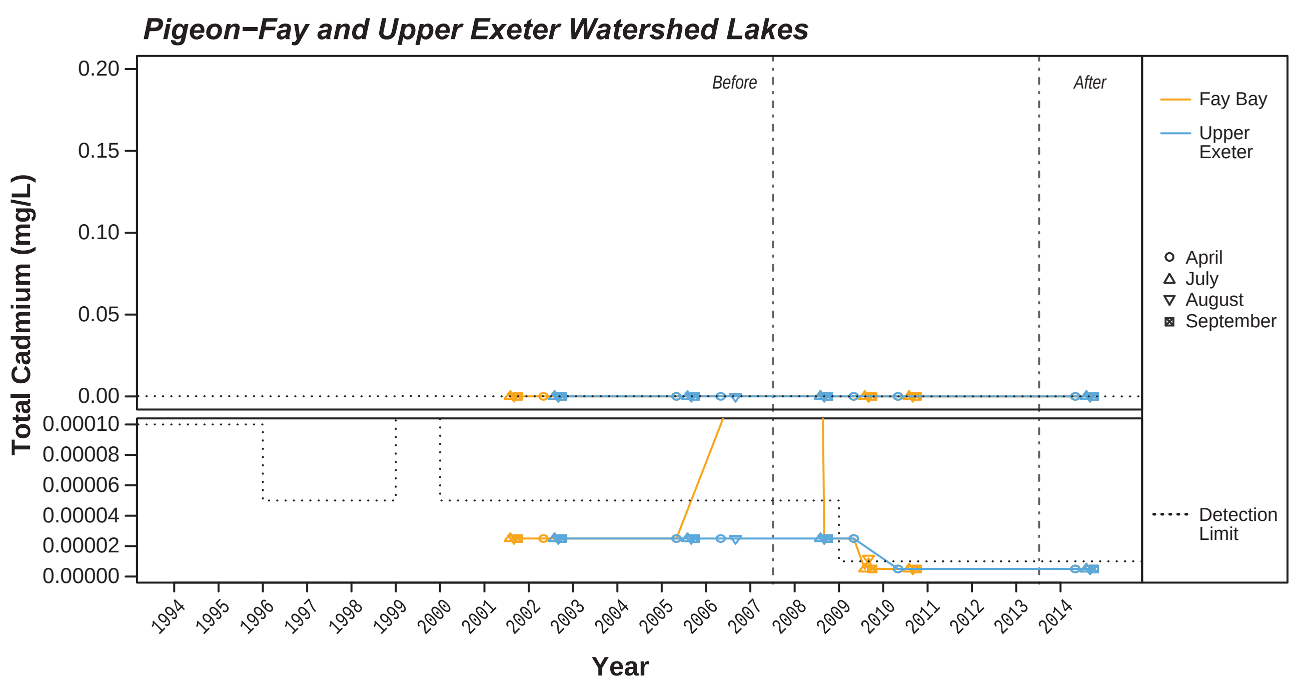


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME Guideline = 1.5 mg/L.

Figure 6-74

Total Cadmium Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

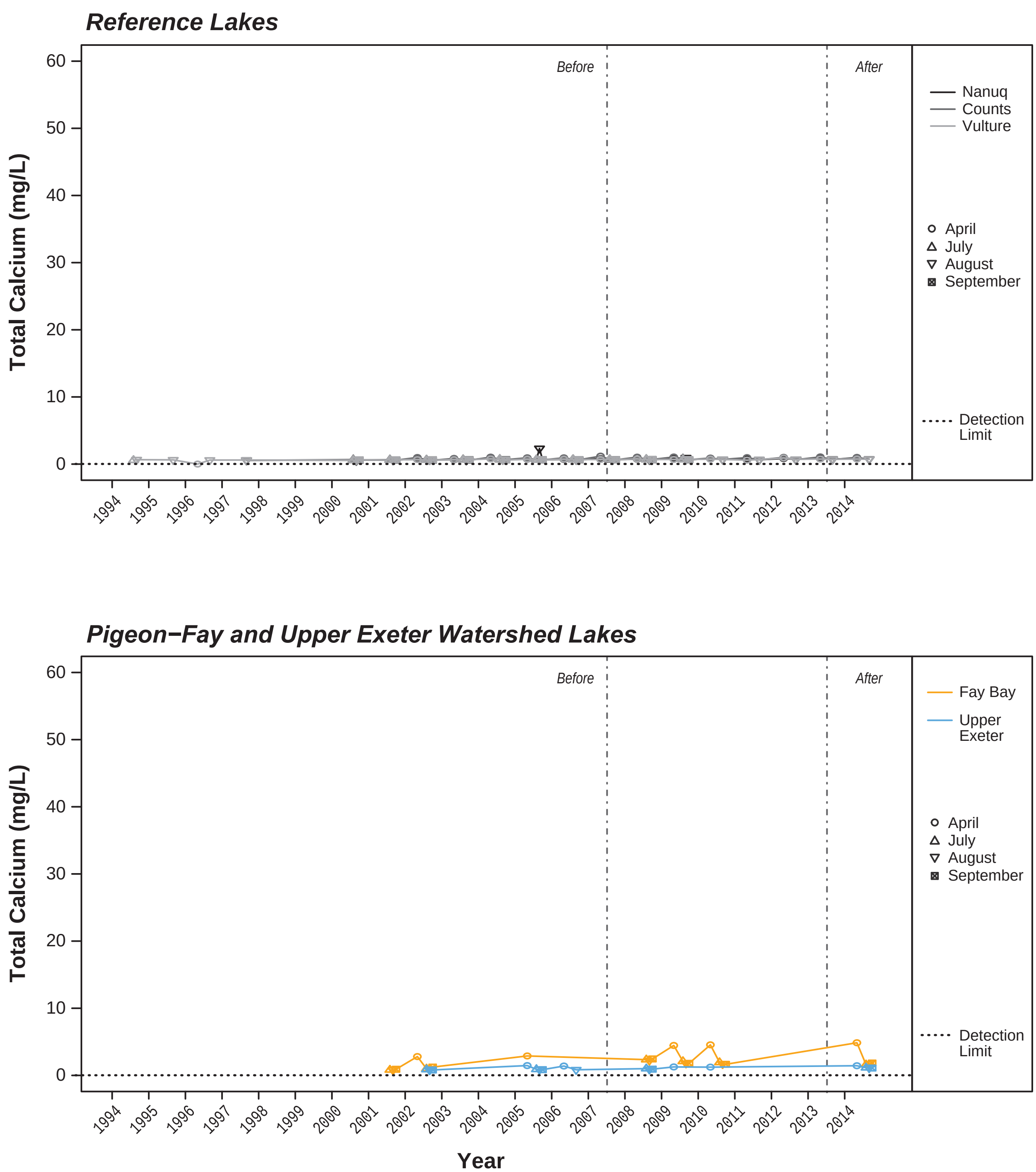




Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME Guideline = $10^{0.83 \times (\log10Hardness - 2.46)}$ /1000 mg/L, with minimum = 0.00004 mg/L where hardness = 0-16 mg/L and maximum = 0.00037 mg/L where hardness > 280 mg/L.

Figure 6-75

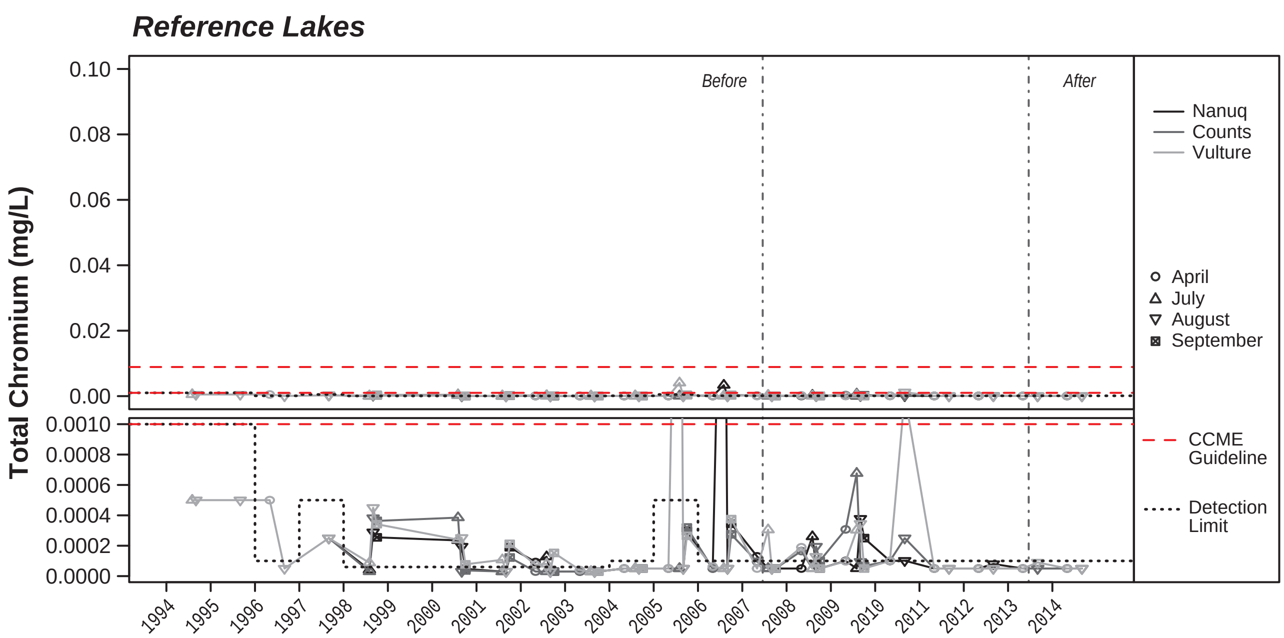
Total Calcium Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

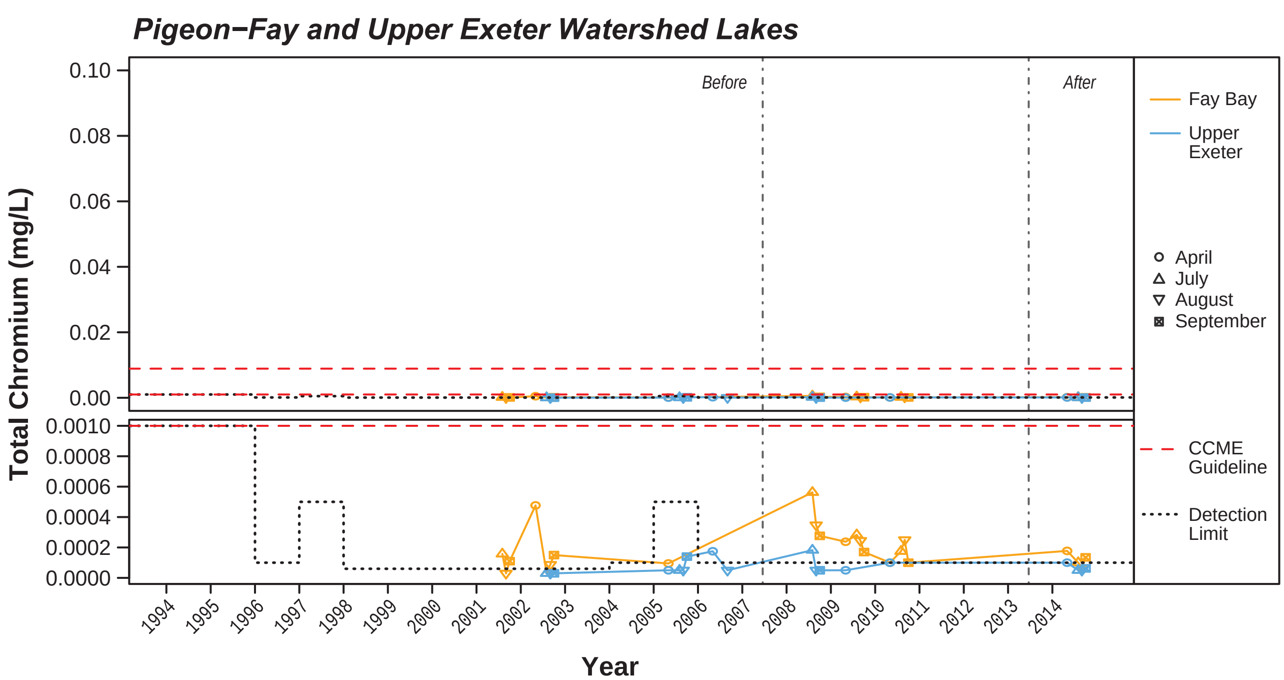


Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

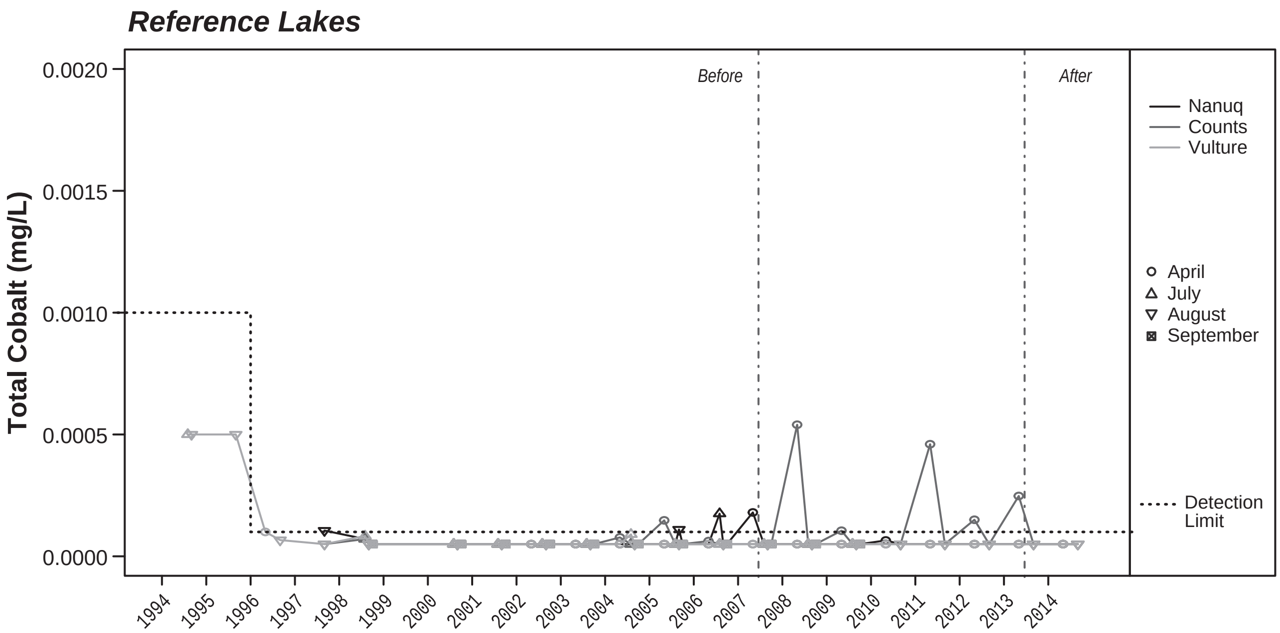
Figure 6-76

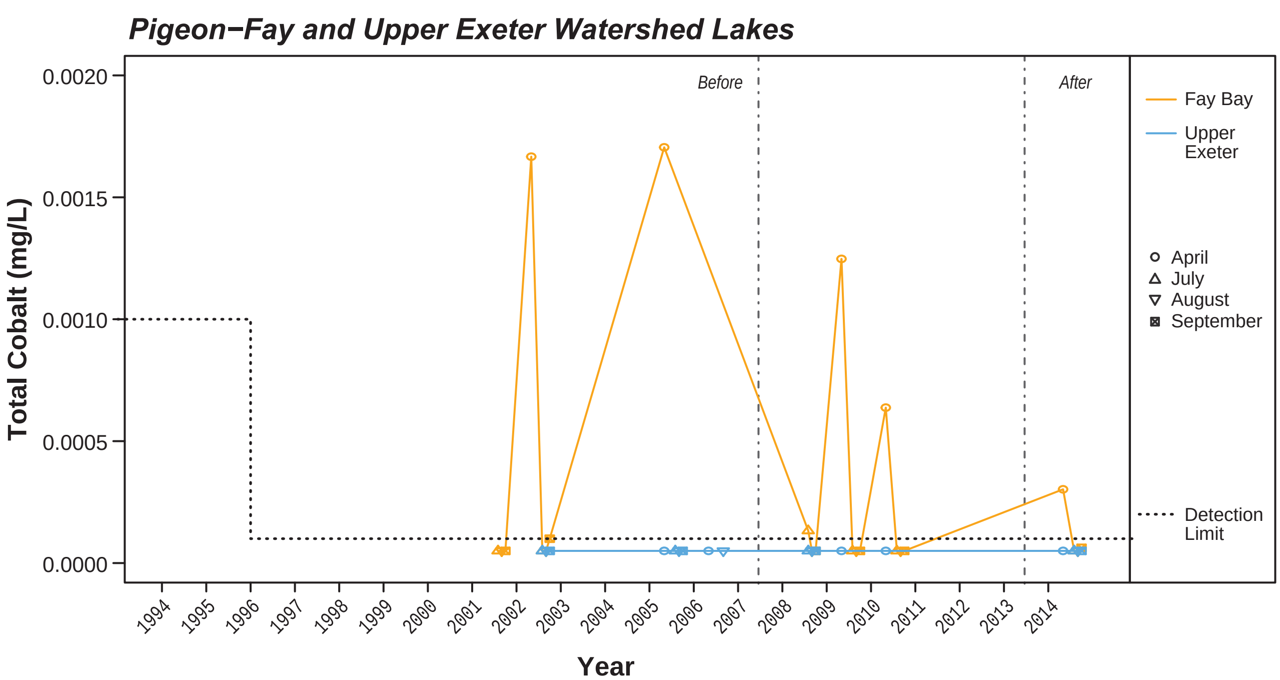
Total Chromium Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014





Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME Guideline = 0.001 mg/L (hexavalent CrVI); 0.0089 mg/L (trivalent CrIII).



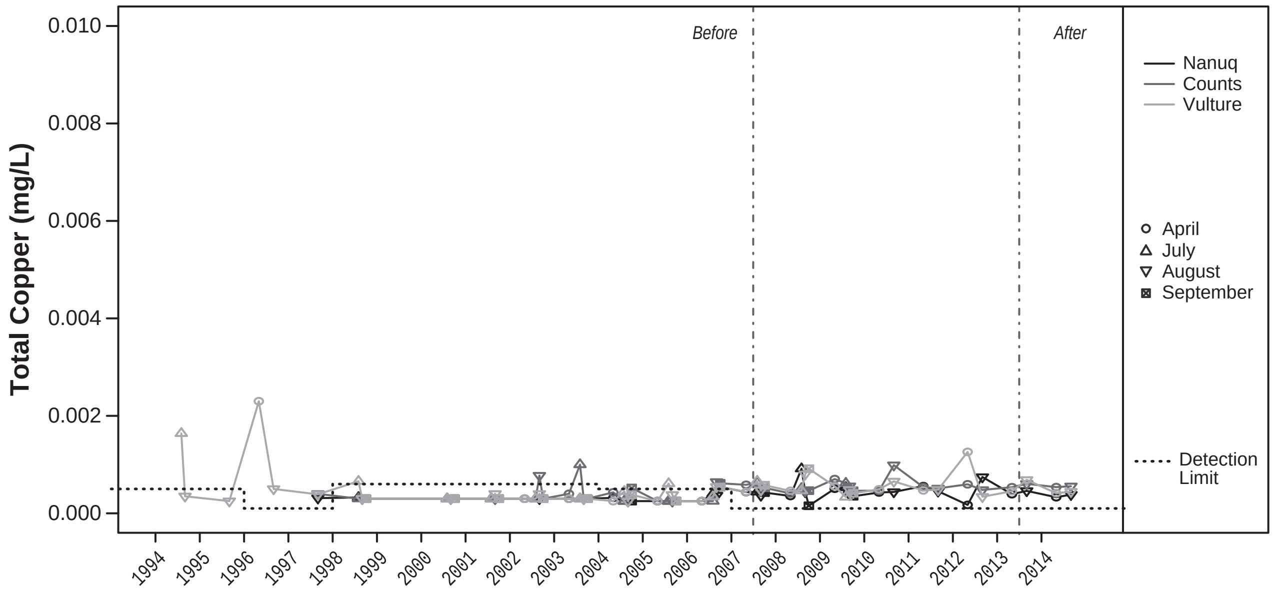


Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

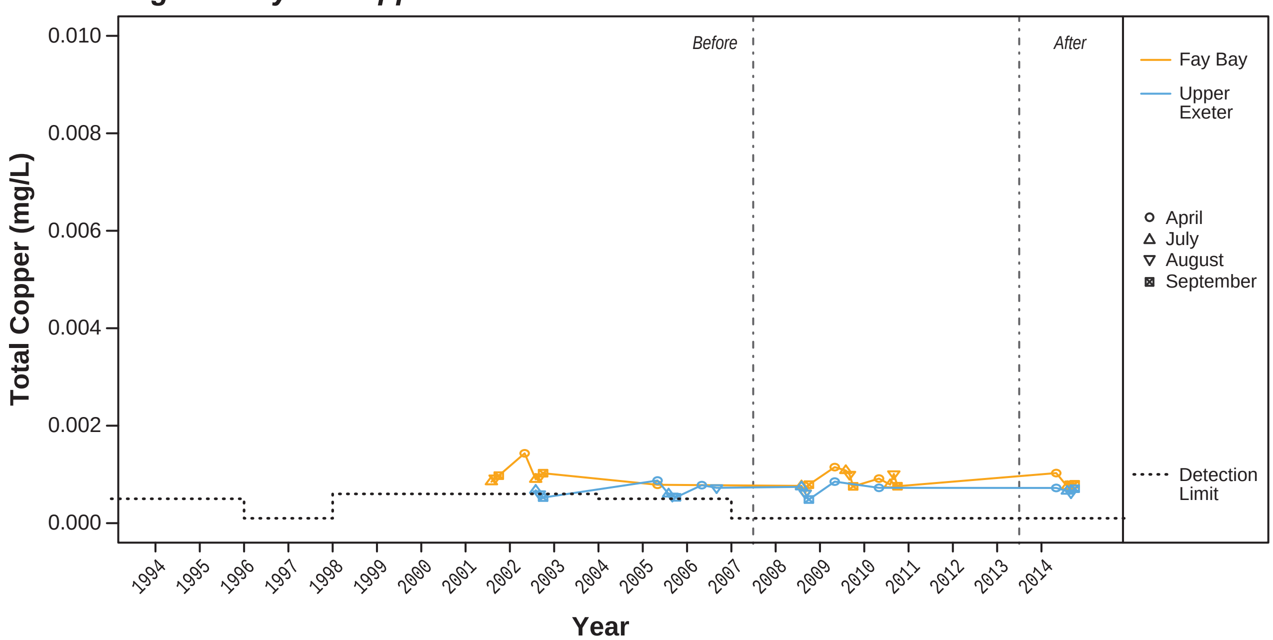
Figure 6-78

Total Copper Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

Reference Lakes



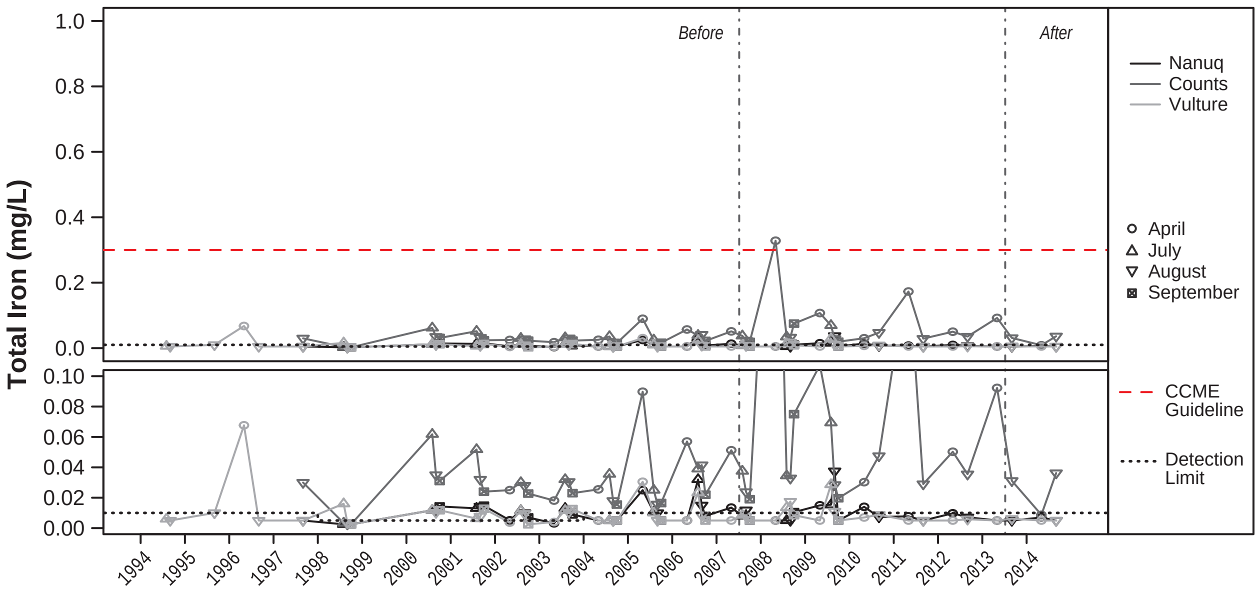
Pigeon−Fay and Upper Exeter Watershed Lakes

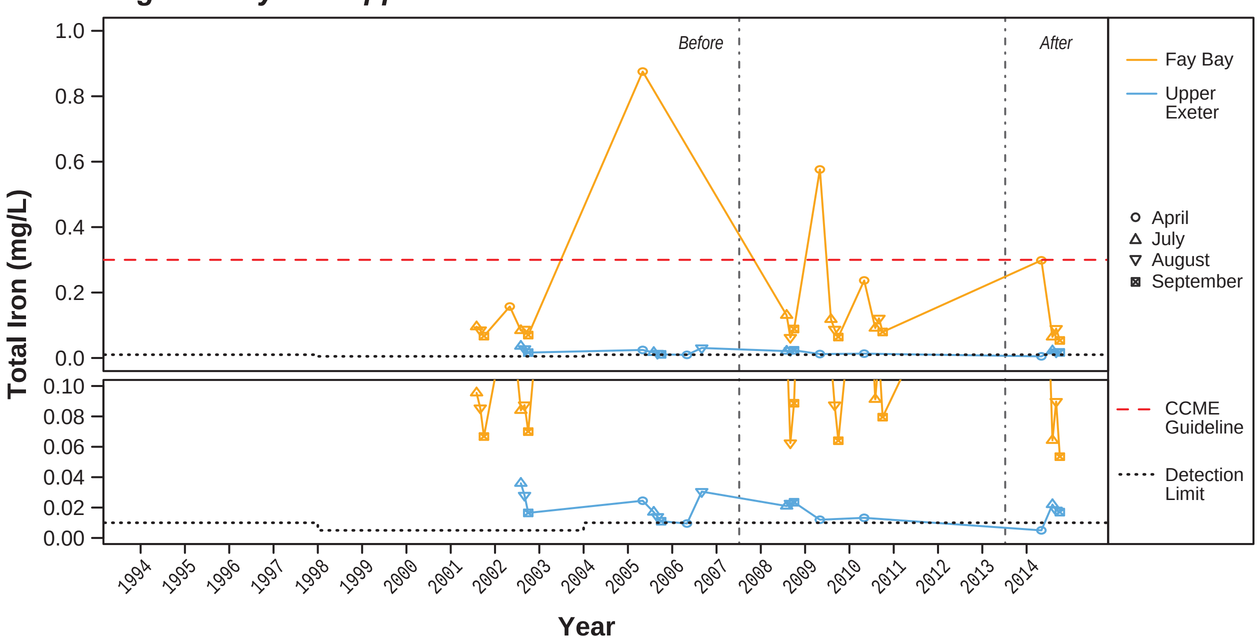


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME Guideline = $e^{0.8545 X (lnHardness) - 1.465 * 0.2}$*/1000 mg/L, where hardness < 180 mg/L and 0.004 mg/L where hardness is ≧ 180 mg/L .*
Minimum guideline = 0.002 mg/L.

Figure 6-79

Total Iron Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014



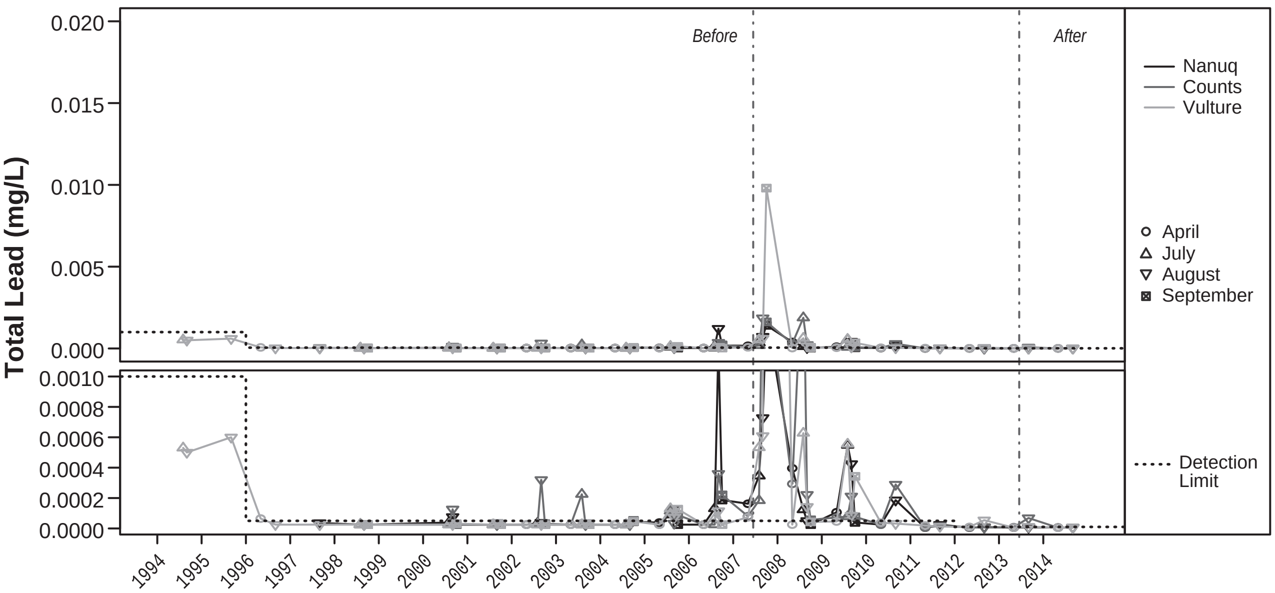


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME Guideline = 0.3 mg/L.

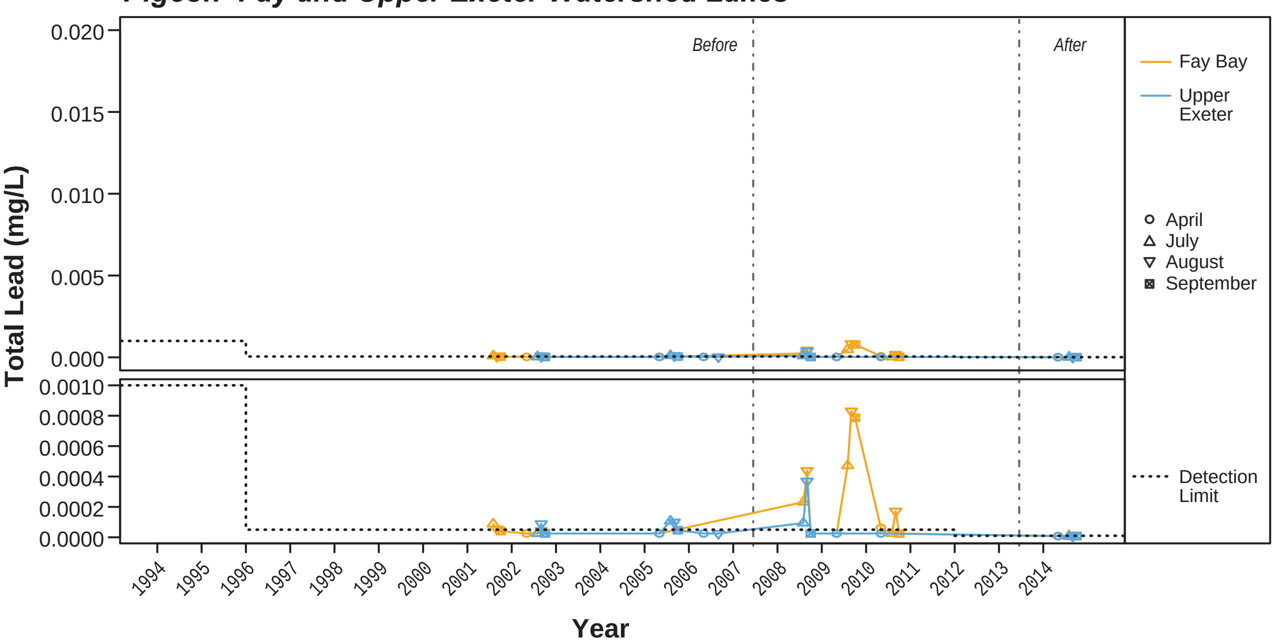
Figure 6-80

Total Lead Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

Reference Lakes



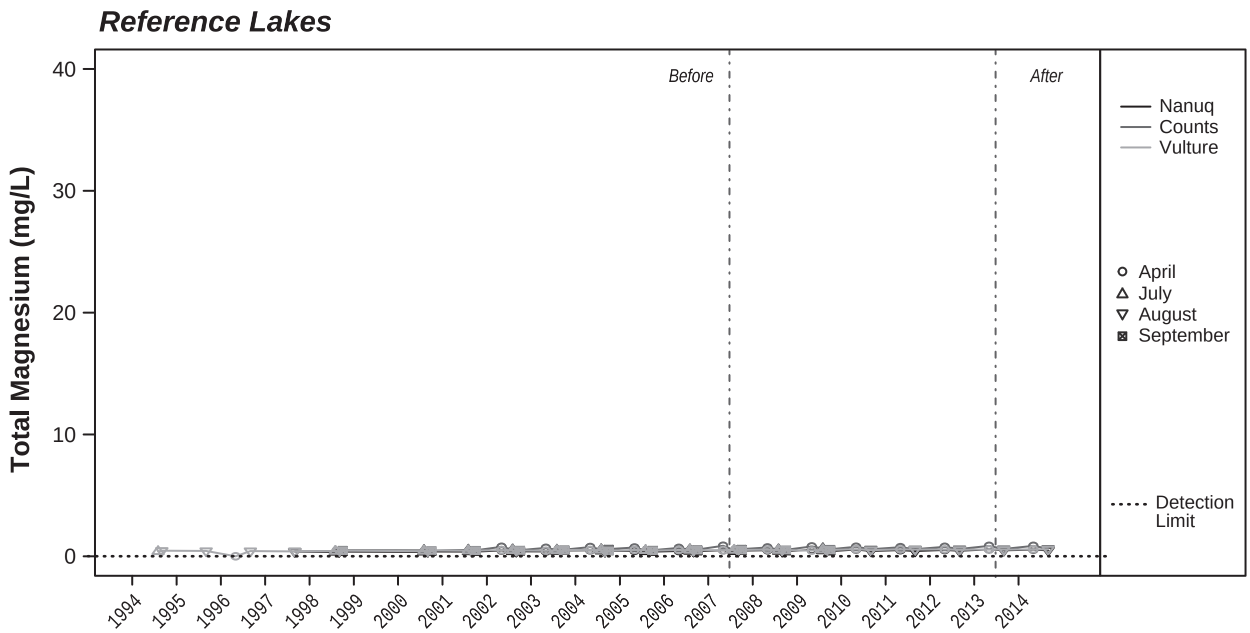
Pigeon−Fay and Upper Exeter Watershed Lakes

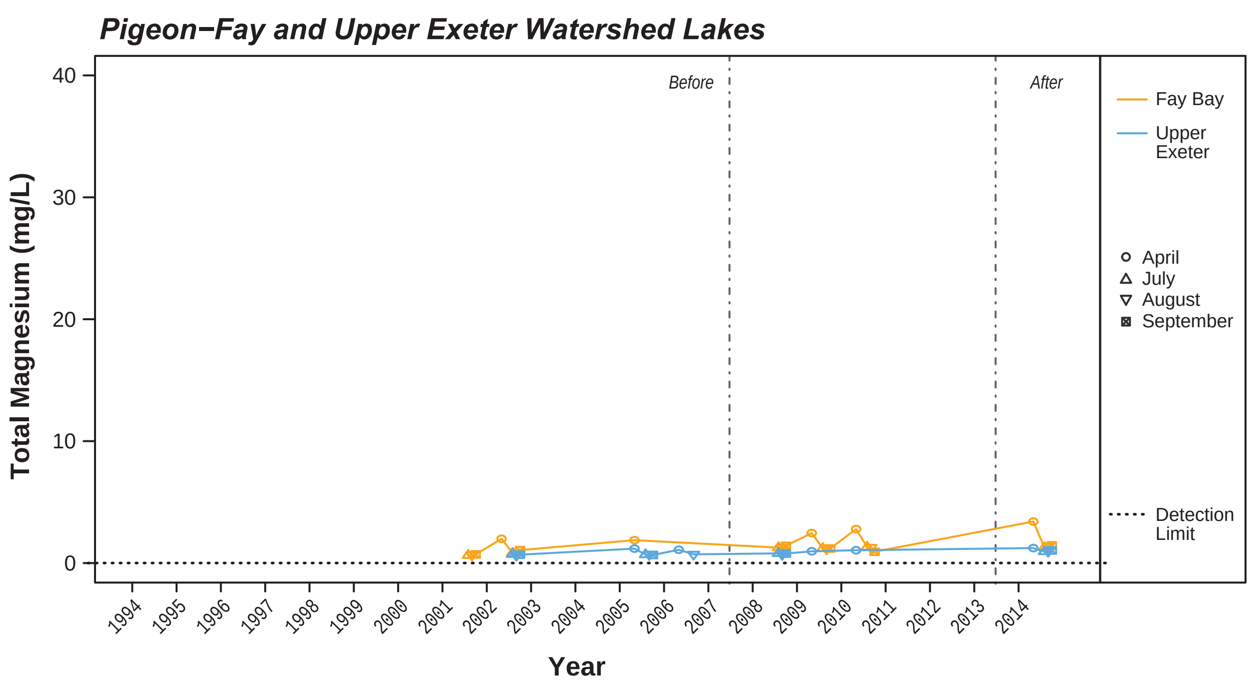


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME Guideline = $e^{1.273 X (lnHardness) - 4.705}$/1000 mg/L, where hardness = 60 - 180 mg/L,
0.001 mg/L where hardness < 60 mg/L and 0.007 mg/L where hardness > 180 mg/L.

Figure 6-81

Total Magnesium Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

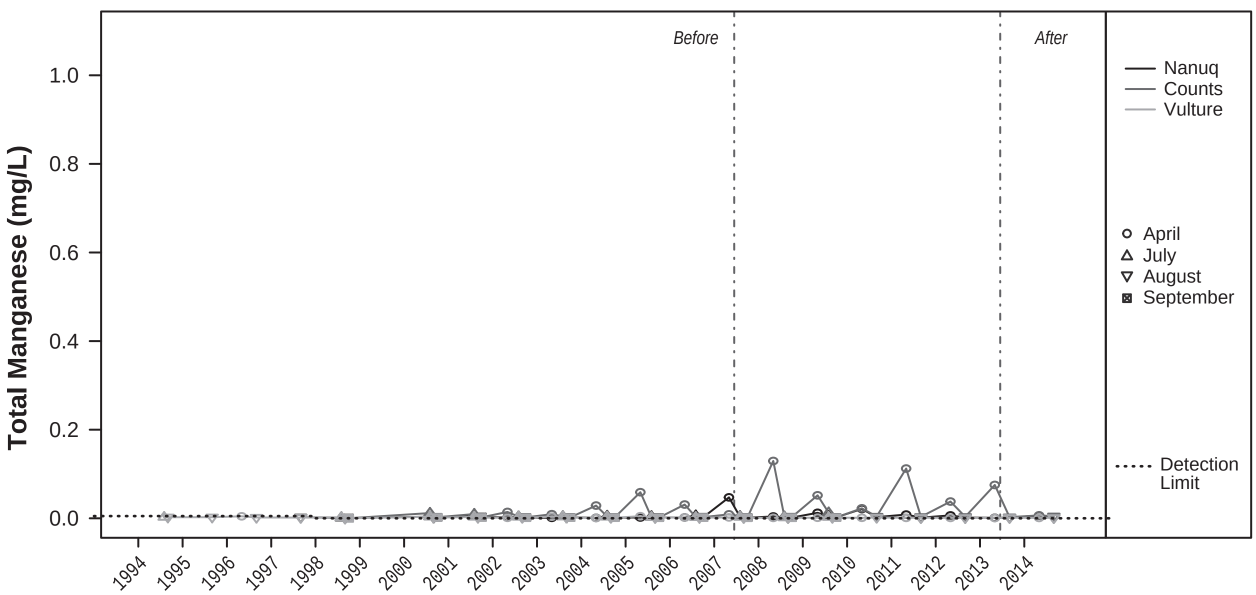


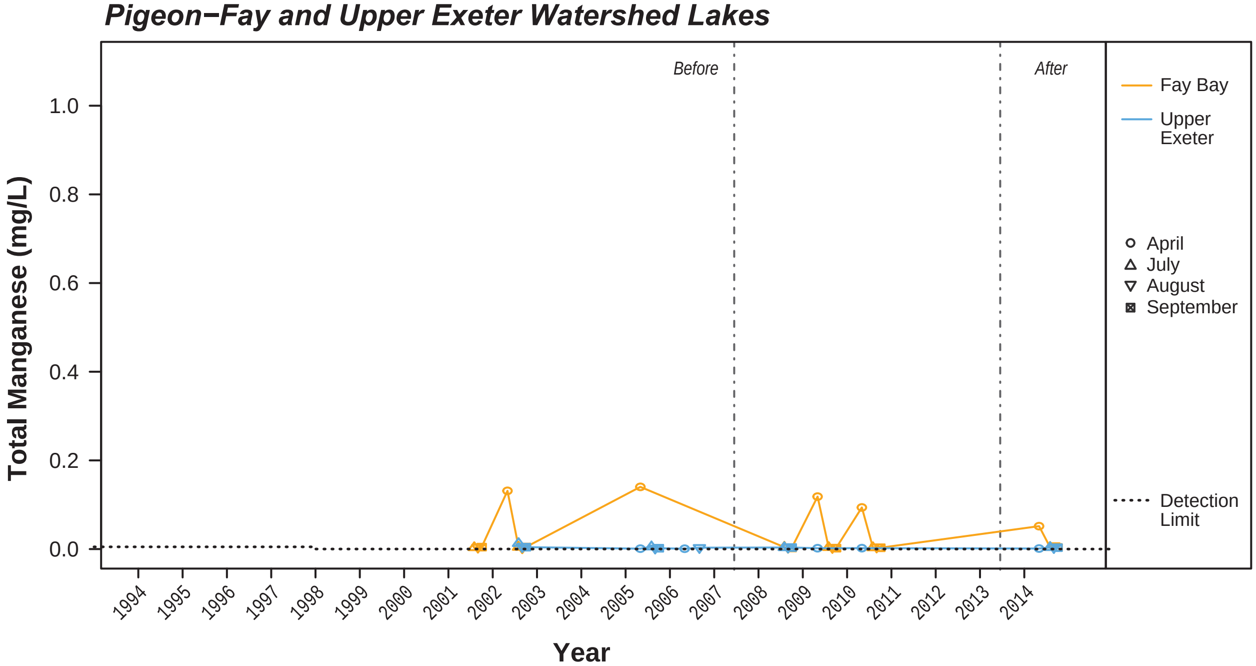


Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 6-82

Total Manganese Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

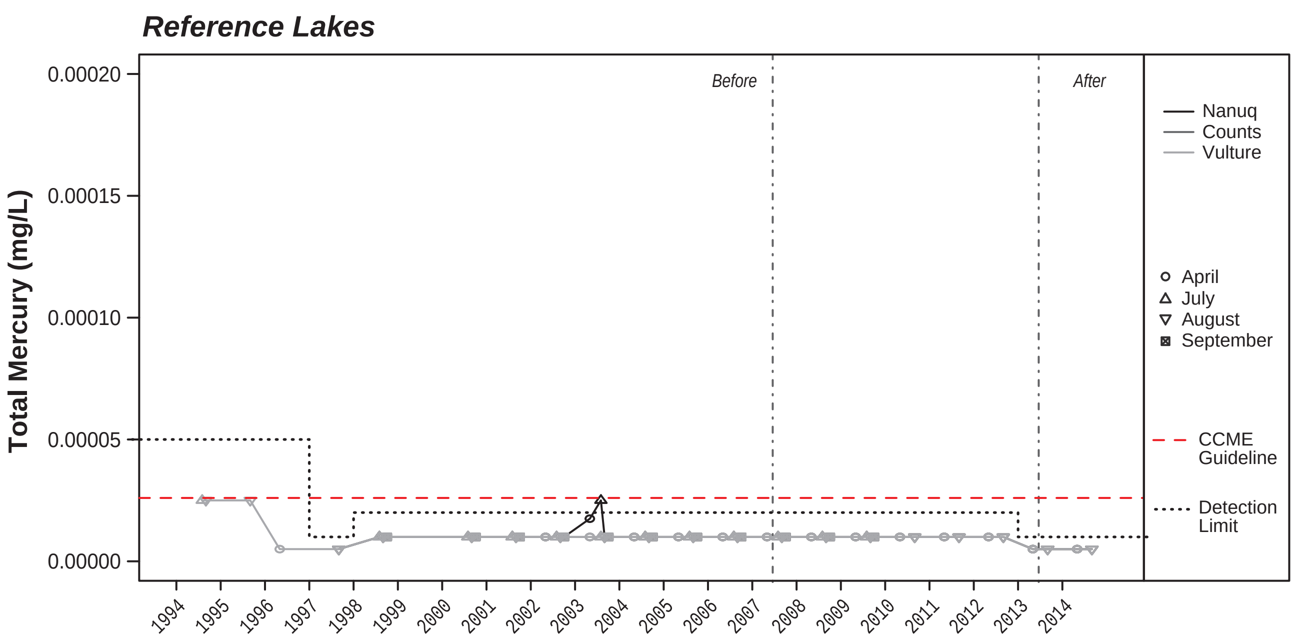


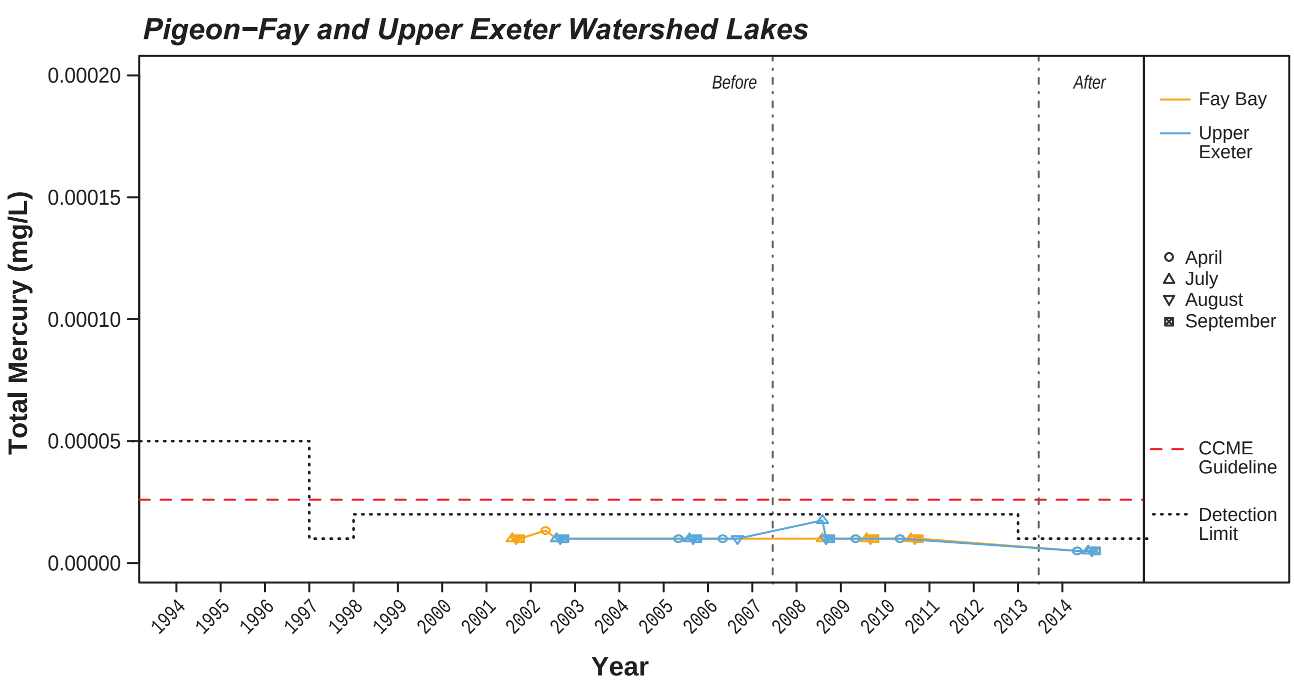


Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 6-83

Total Mercury Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

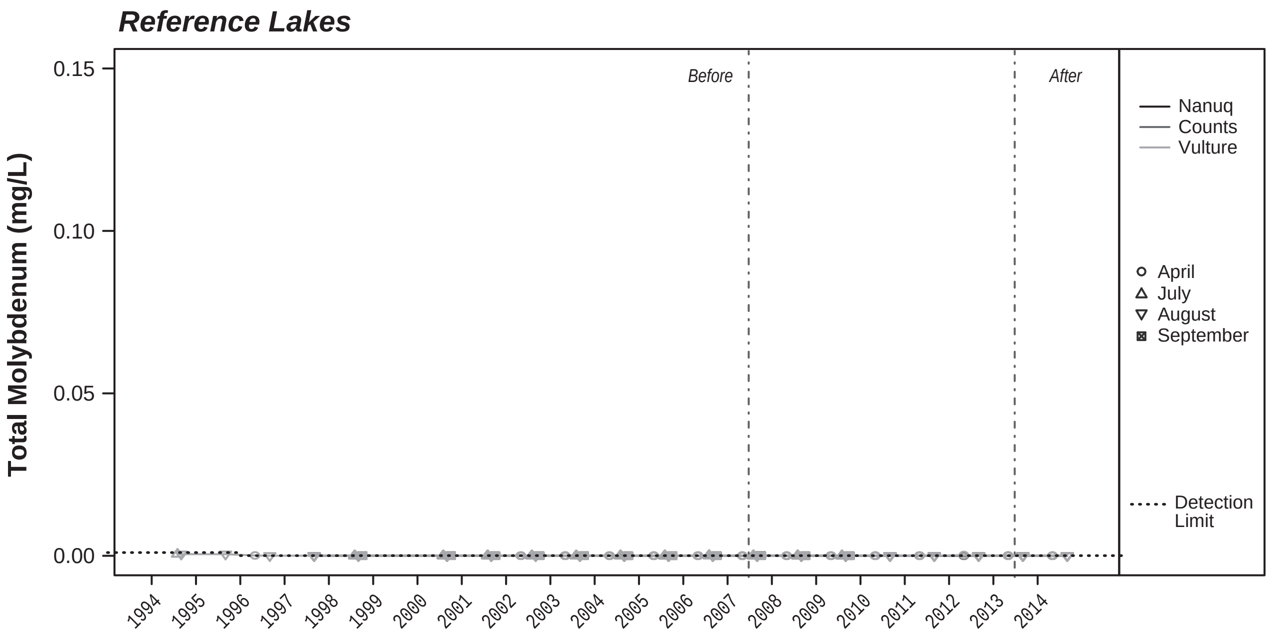


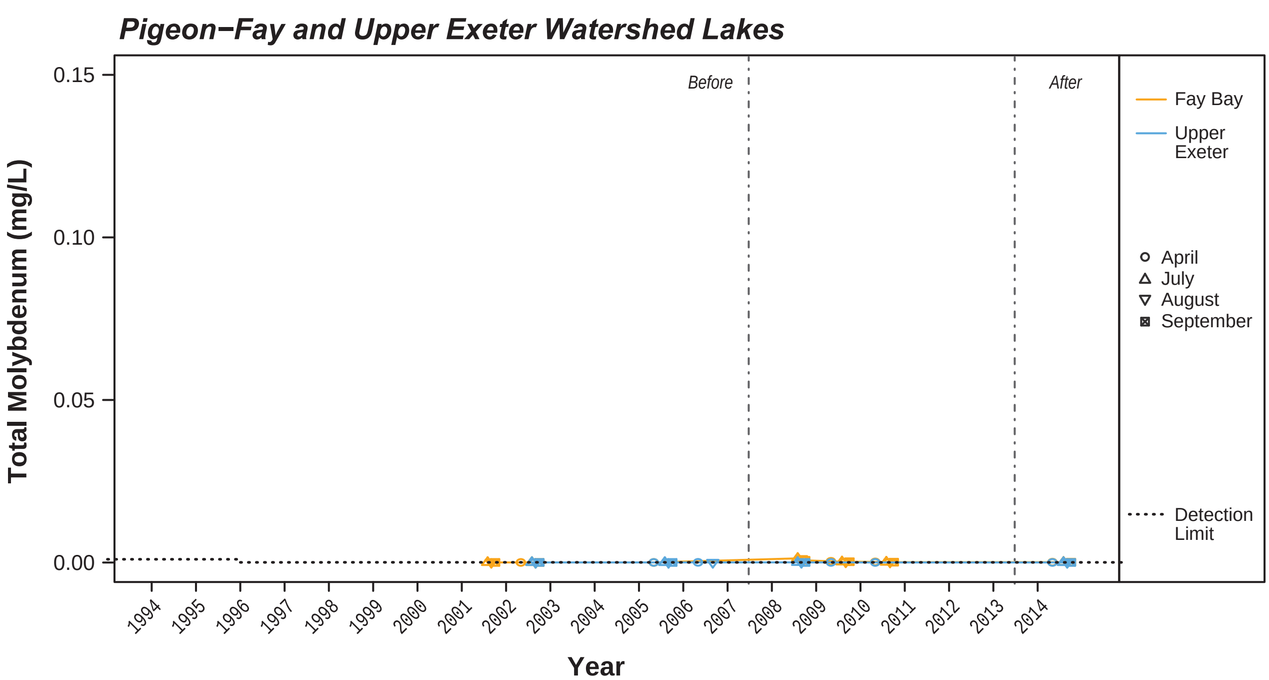


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME guideline = 0.0000026 mg/L.

Figure 6-84

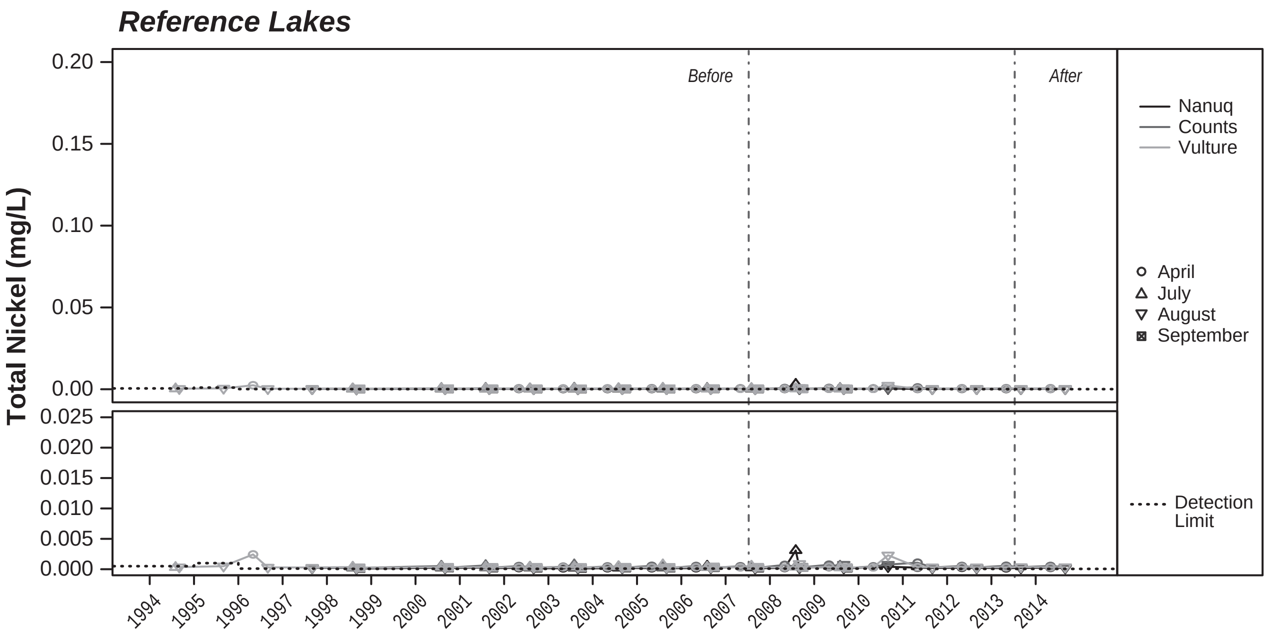
Total Molybdenum Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

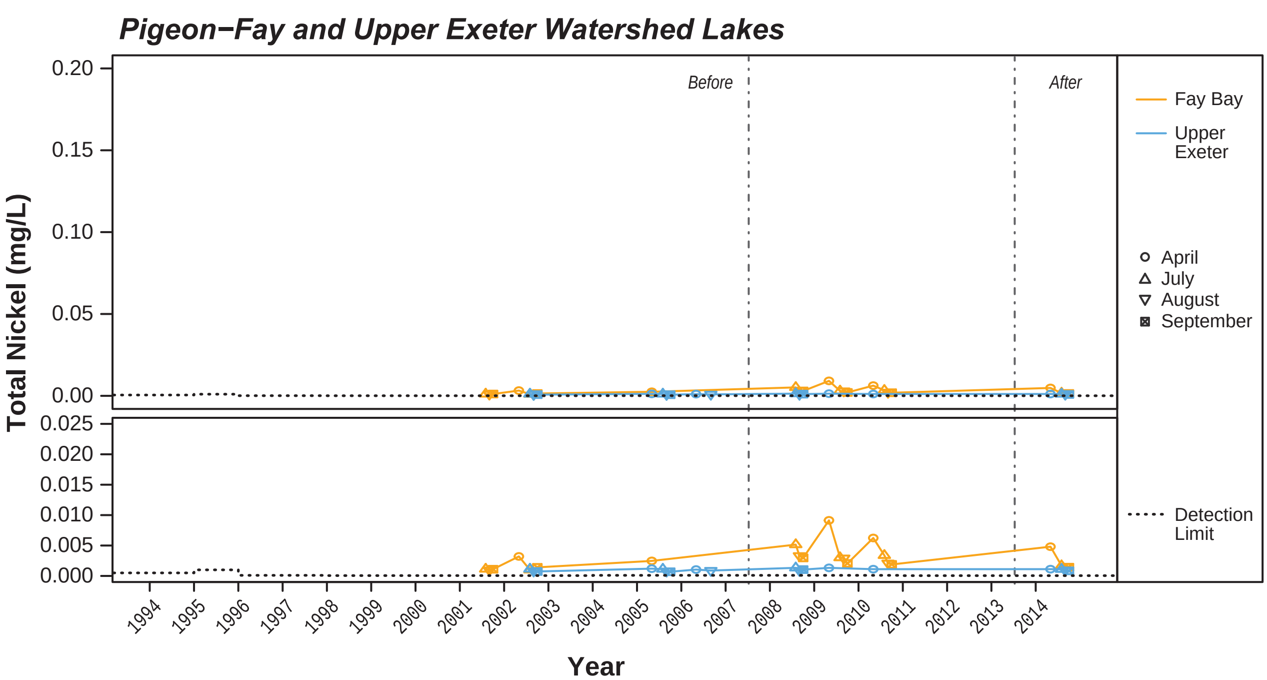




Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
SSWQO = 19.38 mg/L.

Figure 6-85
Total Nickel Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014



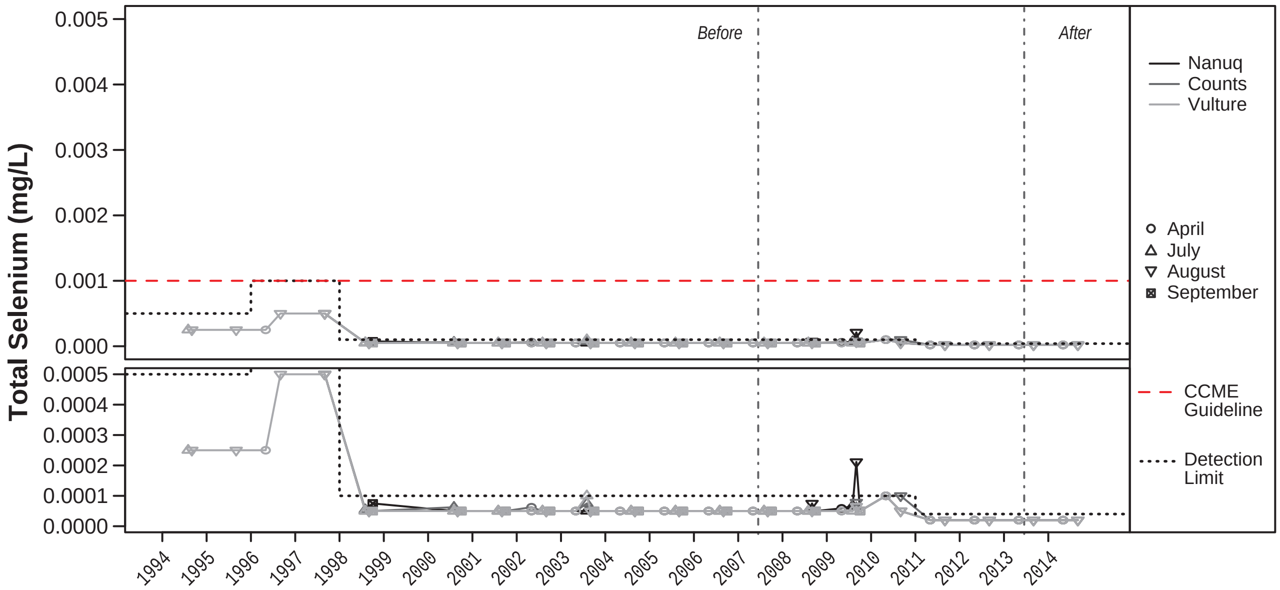


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME Guideline = $e^{0.76 X (lnHardness) + 1.06}$/1000 mg/L, where hardness = 60 - 180 mg/L,
0.025 mg/L where hardness < 60 mg/L, and 0.15 mg/L where hardness > 180 mg/L.

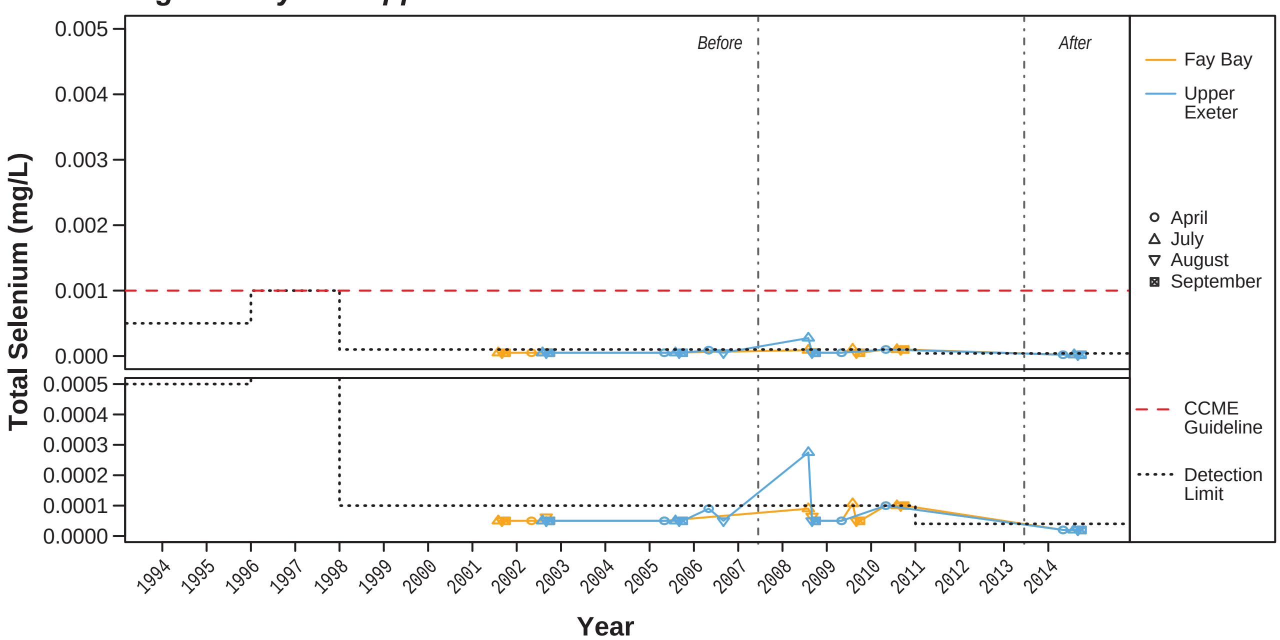
Figure 6-86

Total Selenium Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

Reference Lakes



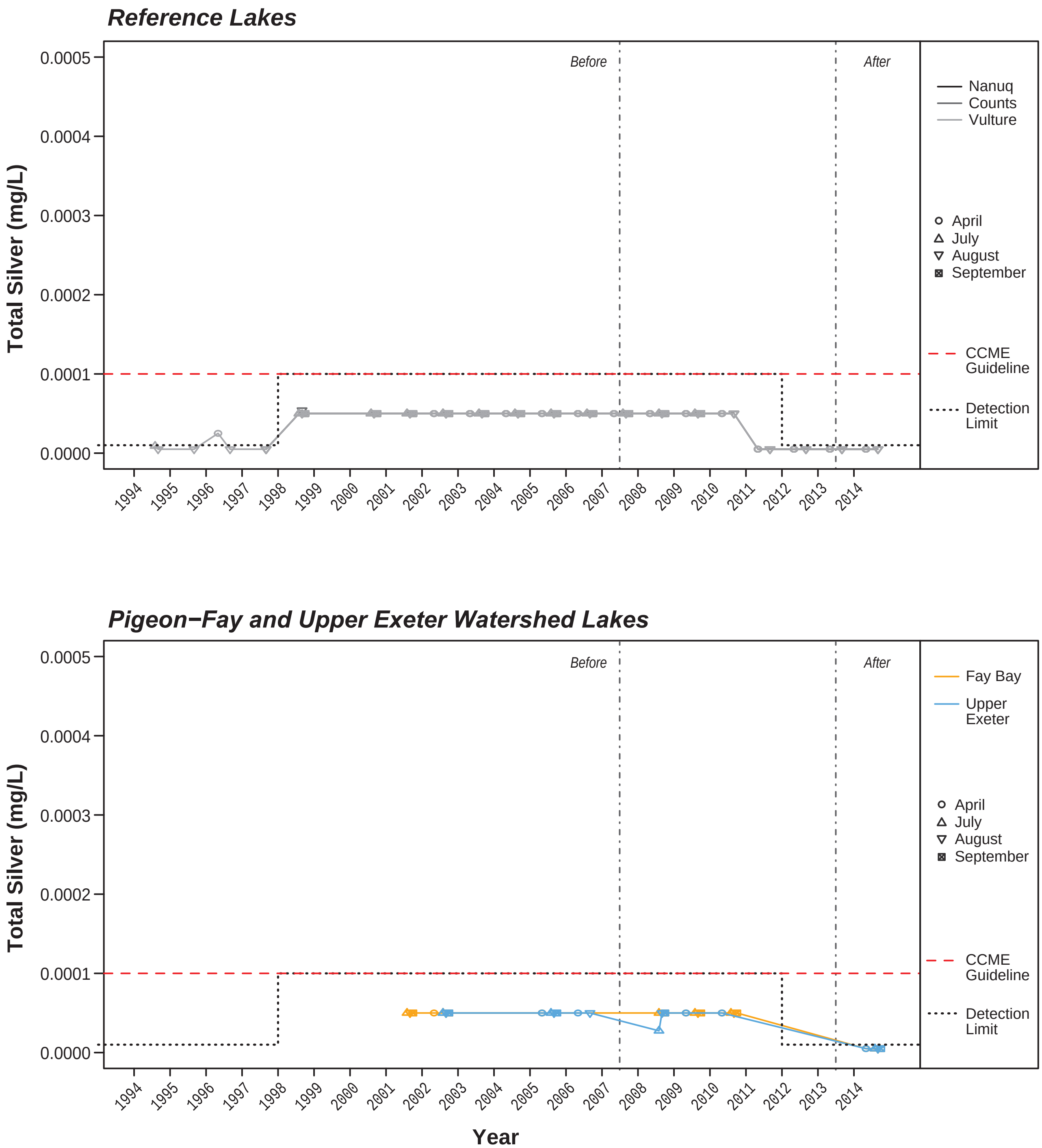
Pigeon−Fay and Upper Exeter Watershed Lakes



Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME Guideline = 0.001 mg/L.

Figure 6-87

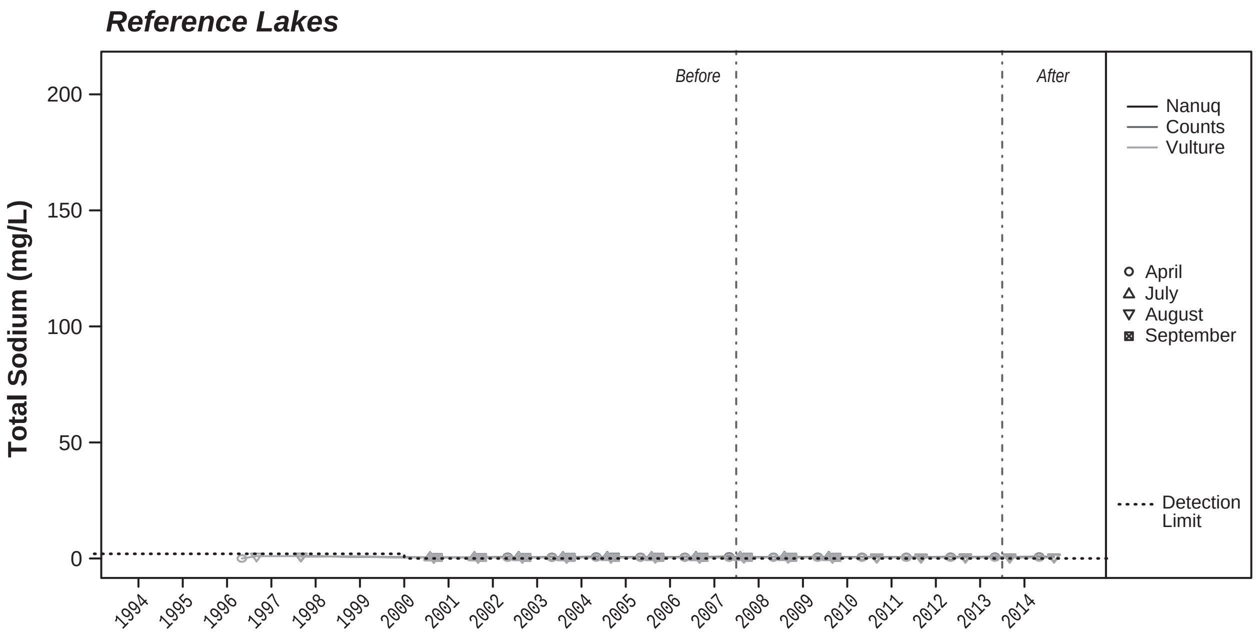
Total Silver Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

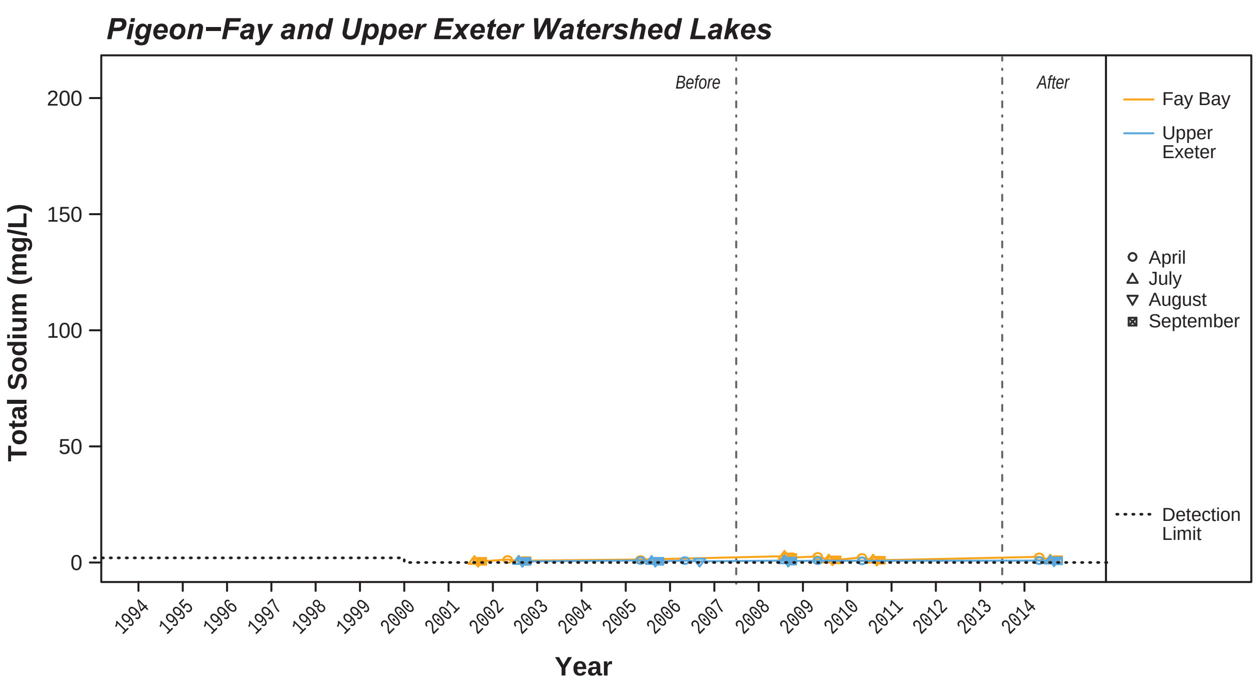


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME Guideline = 0.0001 mg/L.

Figure 6-88

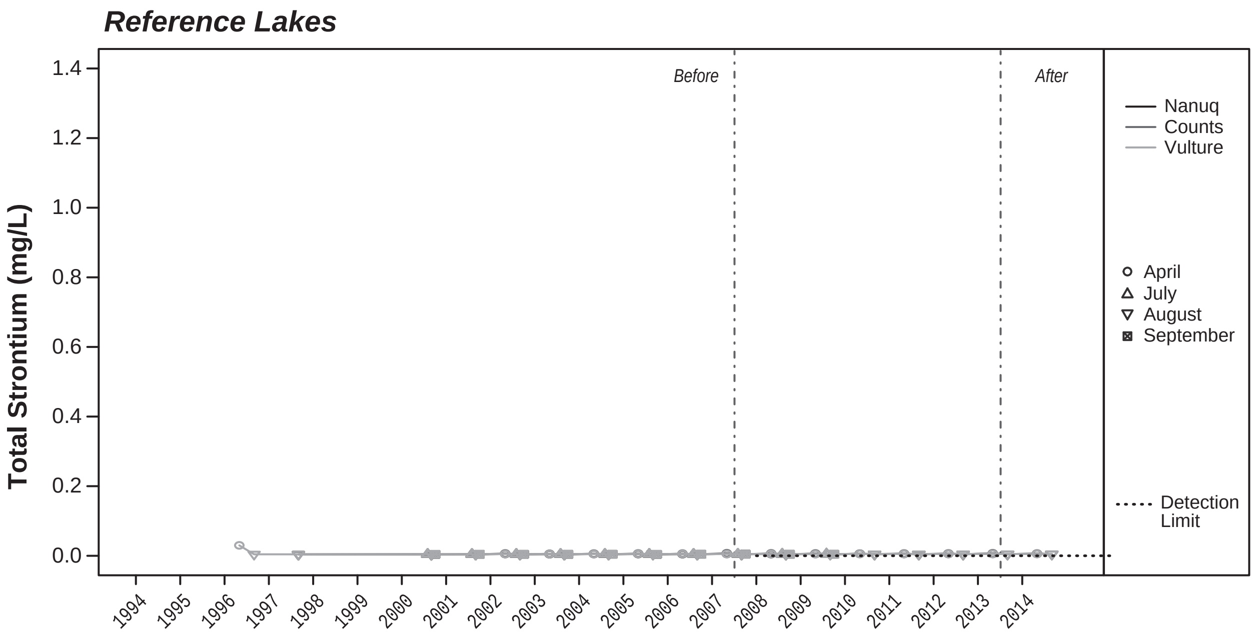
Total Sodium Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

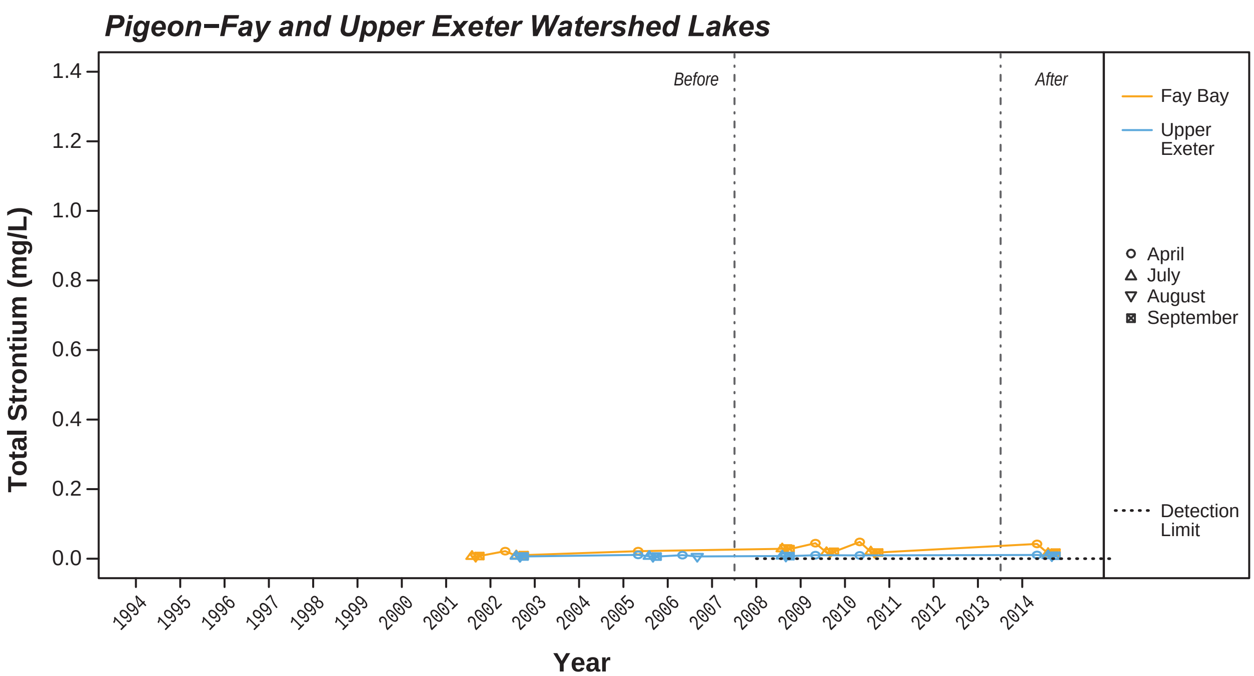




Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Total Strontium Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

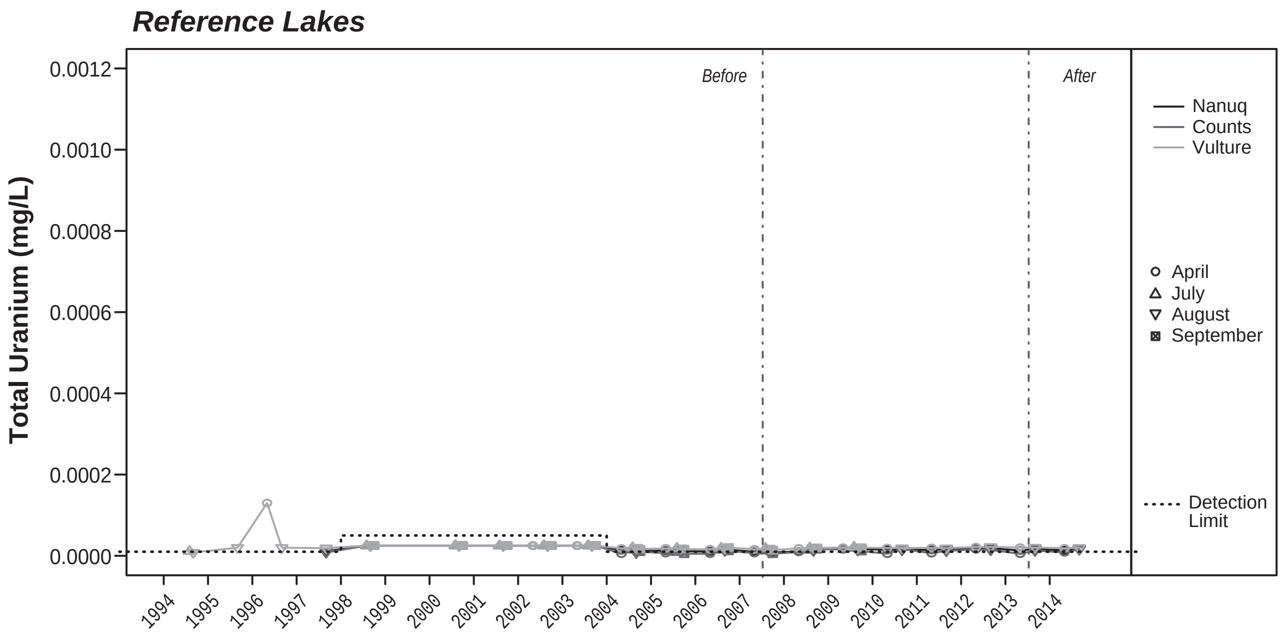


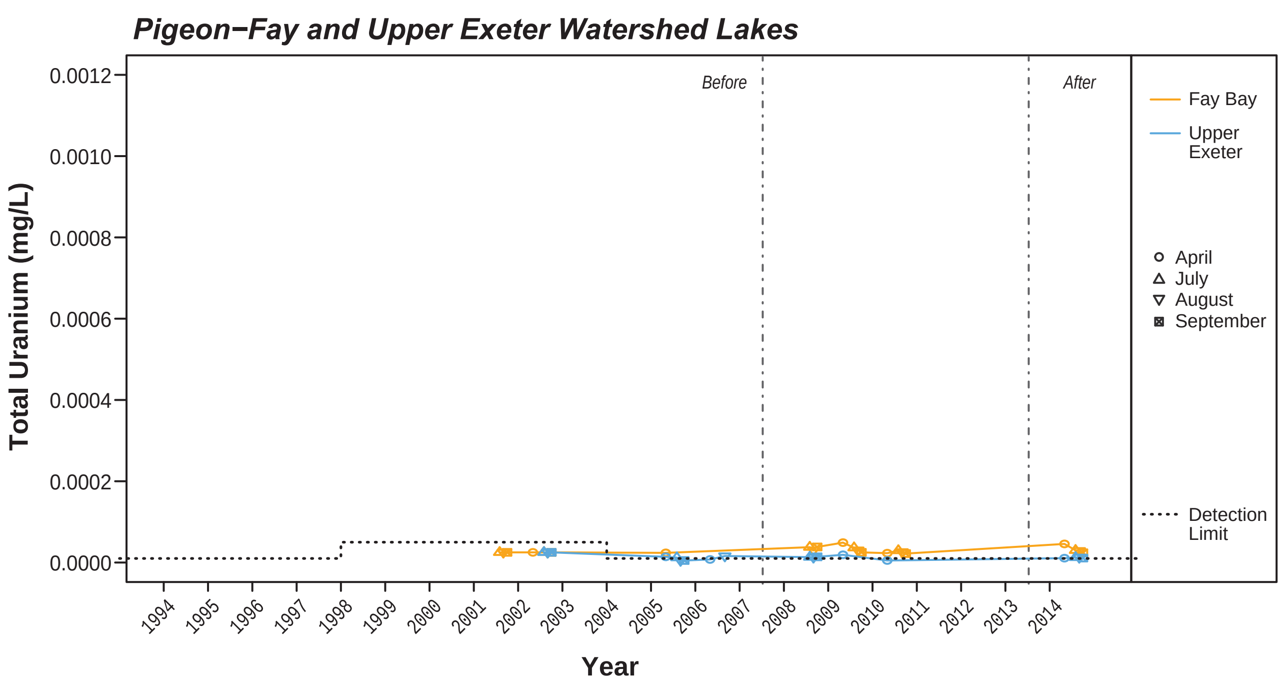


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
Water quality benchmark (Golder 2011) = 6.242 mg/L.

Figure 6-90

Total Uranium Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

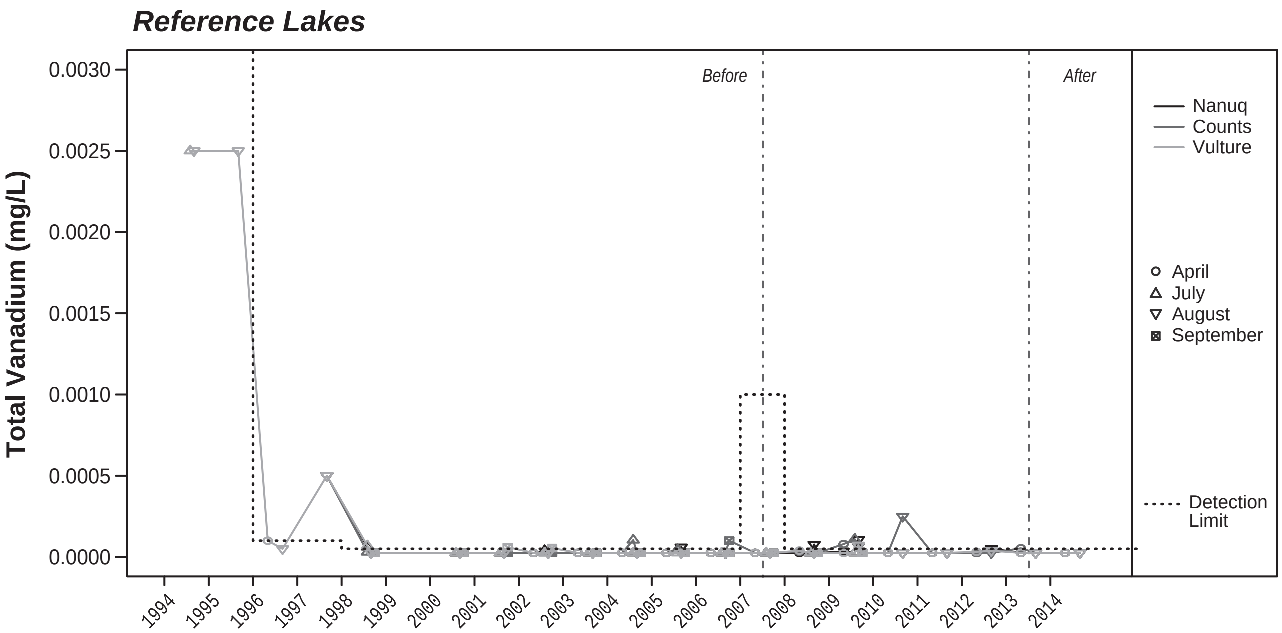


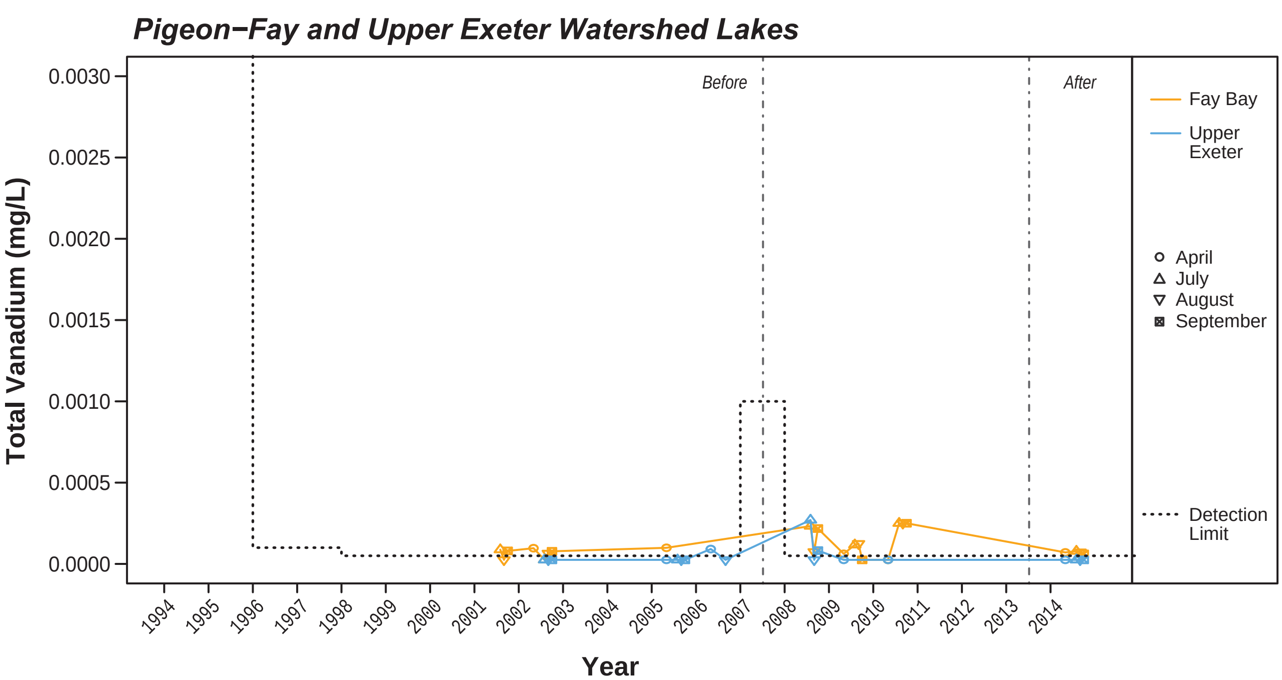


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME Guideline = 0.015 mg/L.

Figure 6-91

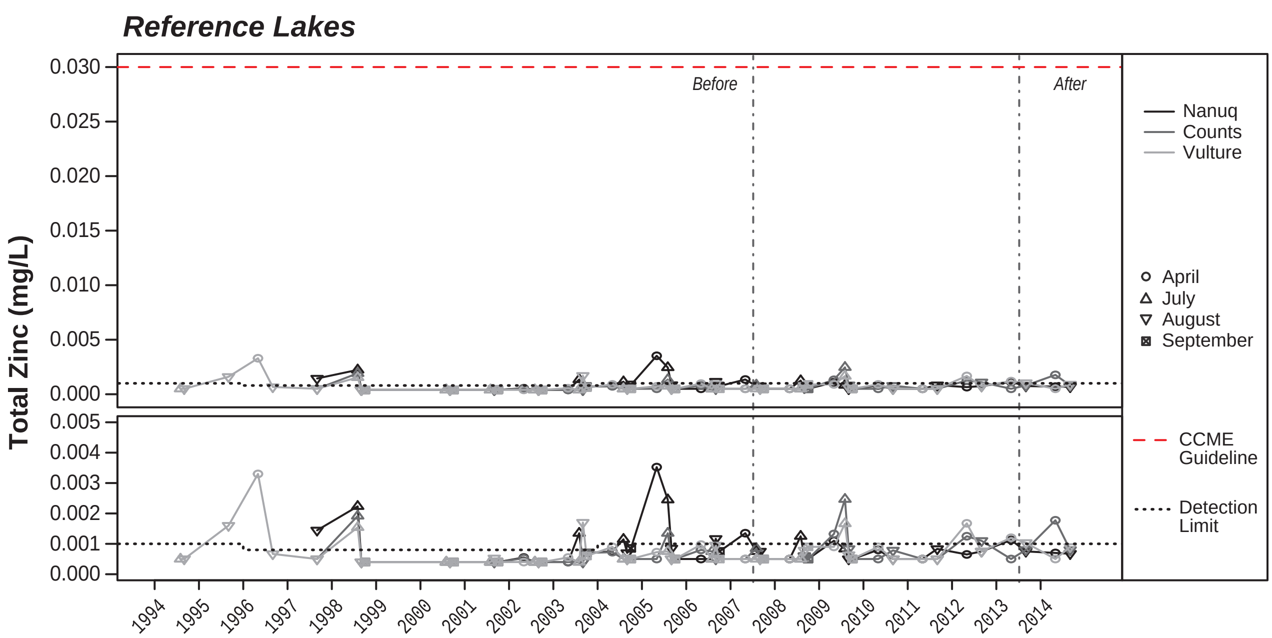
Total Vanadium Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014

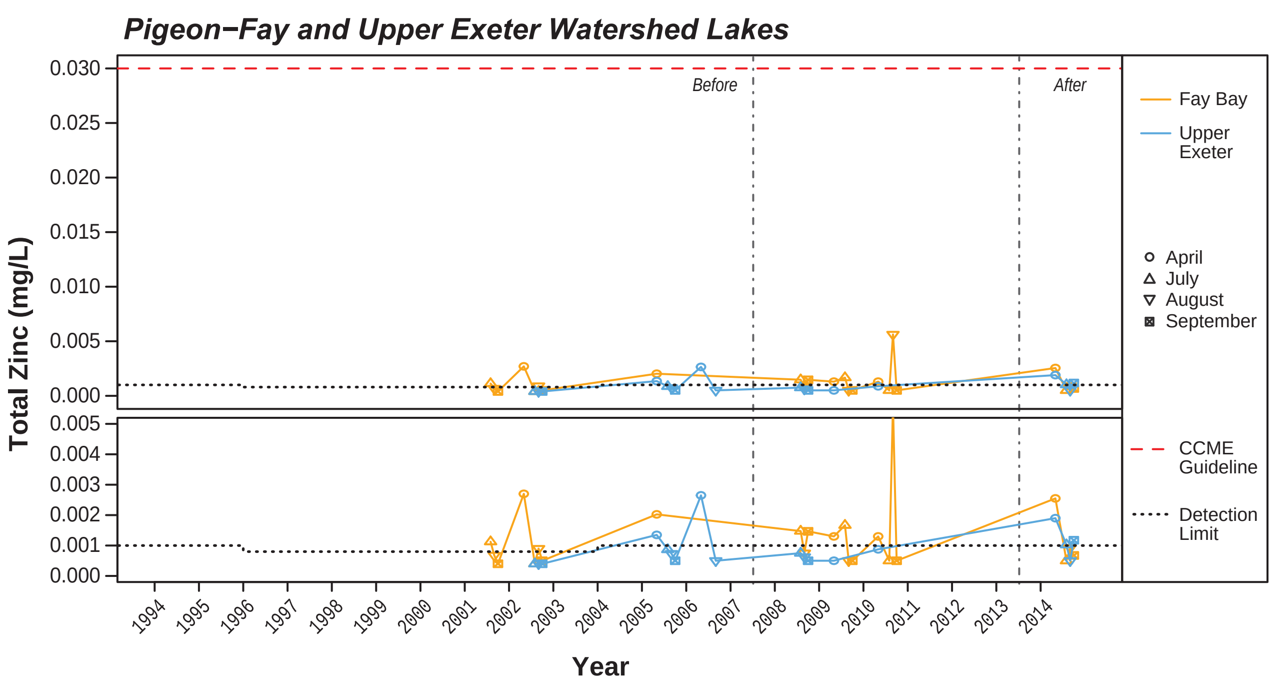




Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
SSWQO = 0.03 mg/L.

Total Zinc Concentrations at Pigeon AEMP Lake Sites, 1994 to 2014





Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME Guideline = 0.03 mg/L.

The 2014 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 6-3 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 6-4 and 6-5);
- runoff depth (mm; Table 6-6);
- the computed runoff coefficients (Table 6-7); and
- comparison of 2014 daily flows with the historical record (Figures 6-93 to 6-100).

Table 6-3. AEMP Hydrometric Stations, 1994 to 2014

Station Number	WGS-14	WGS-39		WGS-02	WGS-24	WGL-46		WGS-35
Location	Vulture–Polar	Lower PDC	Nema-Martine	Long Lake Outflow[1]	Slipper–Lac de Gras	Cujo Outflow	Christine-Lac du Sauvage	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
2014	X	X	X	-	X	X	X	X

Note: X indicates station in operation.

[1] *Flows from the Long Lake Containment Facility have been regulated since December of 1997.*

Table 6-4. Maximum Recorded Unit Yield (L/s/km²) for AEMP Streams and Points of Regulated Discharge, 1995 to 2014

Year	Nanuq Outflow		Vulture-Polar		Lower PDC		Long Lake Outflow (1995, 1996) LLCF Discharge (1998-2014)		Nema-Martine		Slipper-Lac de Gras		King Pond		Cujo Outflow		Christine-Lac du Sauvage		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	4.4	(May 31)	21.1	(May 22)	-		63.2	(Apr)	-		117.1	(May 23)	-		-		-		16.3	(May 22)
1999	13.9	(Jun 10)	273.2	(May 29)	166.1	(Jun 3)	29.8	(Jul)	-		103.4	(Jun 5)	-		135.9	(Jun 5)	-		55.2	(Jun 10)
2000	17.2	(Jun 22)	95	(Jun 11)	85.1	(May 27)	15.2	(Jul)	-		86.9	(Jun 12)	-		214.3	(Jun 11)	-		67.8	(Jun 22)
2001	18.2	(Jun 22)	185.5	(Jun 7)	78.7	(Jun 12)	24.3	(Jun)	-		371	(Jun 8)	-		77.4	(Jun 7)	-		120.1	(Jun 7)
2002	14.2	(July 4)	69.7	(Jun 5)	40.2	(Jun 8)	15.4	(Jul)	-		37.2	(Jun 11)	106.3	(Sep)	154	(Sep 20)	-		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)	29.4	(Jun 4)	54.7	(Jun 3)	25.4	(Jul)	36.6	(May 31)	74.2	(Jun 2)	32.2	(Jun 8)
2014	-		31.1	(Jun 2)	44.3	(May 30)	11.3	(Oct)	17.0	(Jun 2)	39.9	(Jun 1)	-		20.3	(Jun 2)	29.2	(Jun 4)	22.5	(May 28)

Table 6-5. Minimum Recorded Unit Yield (L/s/km²) for AEMP Streams and Points of Regulated Discharge, 1995 to 2014

Year	Nanuq Outflow		Vulture-Polar		Lower PDC		Long Lake Outflow (1995, 1996) LLCF Discharge (1998-2014)		Nema-Martine		Slipper-Lac de Gras		King Pond		Cujo Outflow		Christine-Lac du Sauvage		Counts Outflow	
1994	-		-		-		-		-		1.2	(Aug 15)	-		-		-		-	
1995	-		-		-		2.7	(Aug 6)	-		5	(Aug 27)	-		-		-		-	
1996	-		-		-		1	(Aug 7)	-		0.7	(Aug 7)	-		-		-		-	
1997	6.6	(Aug 21)	1.1	(Aug 20)	-		-		-		2.2	(Aug 22)	-		-		-		6.1	(Aug 19)
1998	2.7	(Aug 21)	0.9	(Aug 5)	-		0	(Jun)	-		2.2	(Aug 5)	-		-		-		0.7	(Aug 22)
1999	5.5	(Aug 25)	2.5	(Aug 16)	0.6	(Aug 16)	3.4	(Oct)	-		6.2	-	-		0.4	(Aug 17)	-		1.8	(Aug 22)
2000	7.1	(Aug 13)	2	(Aug 13)	1.9	(Aug 13)	4.3	(Sep)	-		5.1	-	-		0.1	(Aug 14)	-		-	
2001	-		2.9	(Aug 23)	0.8	(Aug 23)	6.3	(Aug)	-		0.7	-	-		1	(Aug 17)	-		9.8	(Aug 24)
2002	10	(Aug 22)	4	(Aug 9)	0.6	(Aug 9)	0	(Jun)	-		2.9	-	0	(Jul – Aug)	0.7	(Aug 6)	-		7.1	(Aug 9)
2003	-		1	(Aug 22)	0.2	(Aug 22)	3.9	(Jul)	-		1.7	-	0.6	(Jun)	4.2	(Jul 5)	-		4.5	(Aug 22)
2004	-		3.7	(Sep 3)	3.3	(Sep 3)	0	(Apr – Jun)	-		0.6	-	0	(Jan – Jun)	0.6	(Aug 25)	-		4	(Jun 15)
2005	-		4.3	(Aug 7)	1.1	(Aug 7)	0	(Jan – Jun)	-		1.6	-	0	(Jan – Aug)	0.7	(Aug 28)	-		0.2	(Aug 25)
2006	-		4.1	(Aug 22)	2.3	(Aug 22)	0	(Jan - May)	-		7.9	-	0	(Jan – Jul)	0.6	(Jul 16)	-		2.8	(Sep 18)
2007	-		0.7	(Sep 16)	0.8	(Sep 17)	0	(Jan - May)	-		3.1	-	0	(Jan – May)	0.5	(Aug 25)	-		1.2	(Sep 16)
2008	-		2.5	(Aug 13)	1.9	(Aug 10)	0	(Jan - Jul)	-		1.7	-	0	(Jan – May)	0.7	(Aug 13)	-		1.8	(Aug 13)
2009	-		1.4	(Sep 16)	1.2	(Sep 6)	0	(Jan - Jun)	-		1.3	-	0	(Jan – May)	0	(Aug 24)	-		1.9	(Sep 7)
2010	-		0.8	(Sep 13)	1.3	(Sep 13)	0	(Jan - Jun)	-		1.7	-	0	(Jan – Jun)	5.3	(Sep 13)	-		1.1	(Jun 10)
2011	-		2.9	(Aug 15)	1.2	(Aug 14)	0	(Jan - Jun)	-		3.6	-	0	(Jan – Jul)	0	(Aug 30)	-		3	(Aug 15)
2012	-		3.4	(Aug 26)	1.4	(Aug 26)	0	(Jan - May)	-		2	-	0	(Jan - May)	0	(Aug 3)	-		0	(Sep 10)
2013	-		4.1	(Sep 6)	1.5	(Sep 7)	0	(Jan - May)	3.9	(Jul 19)	5.2	(Sep 4)	0	(Jan - Jun)	0.4	(Sep 8)	0.2	(Sep 8)	0	(Jun 3)
2014	-		0.4	(Sep 11)	0.2	(Sep 12)	0	(Oct – Jun, Sep)	1.3	(Sep 11)	1.1	(Sep 7)	0	(Sep – Oct)	0	(Aug 21)	0.1	(Sep 11)	0.7	(Sep 11)

Table 6-6. Runoff Depth (mm) for AEMP Streams Recorded from 1995 to 2014

Year	Vulture-Polar	Lower PDC	Long Lake Outflow	LLCF	Nema-Martine	Slipper-Lac de Gras	King Pond	Cujo Outflow	Christine-Lac du Sauvage	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*	-	*841*[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*	*90*	*132*	*68*	*77*	*87*	*114*
2014	*54*	*61*	-	*62*	*33*	*77*	-	*28*	*43*	*100*

Note: Runoff depths in italics are from the entire open water season.

[1] The runoff depth for Slipper-Lac de Gras includes the influence of Fox Lake dewatering (~3 mm) during the 2002 hydrologic year.

[2] Runoff from October 2006 to September 2007.

Table 6-7. Runoff Coefficients Computed for AEMP Streams, 1999 to 2014

		Vulture-Polar		Lower PDC		Nema-Martine		Slipper-Lac de Gras		Cujo Outflow		Christine-LDS		Counts Outflow	
Year	Total Precipitation (mm)	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	Runoff Depth (mm)	Runoff Coefficient	Runoff Depth (mm)	Runoff Coefficient
1999	458	n/a	n/a	n/a	n/a	n/a	n/a	298	0.65	n/a	n/a	n/a	n/a	n/a	n/a
2000	279	170	0.61	169	0.61	n/a	n/a	180	0.65	160	0.57	n/a	n/a	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	n/a	n/a	n/a	n/a
2002	321	80	0.25	51	0.16	n/a	n/a	115	0.36	62	0.19	n/a	n/a	84	0.26
2003	288	62	0.22	55	0.19	n/a	n/a	88	0.31	198	0.69	n/a	n/a	138	0.48
2004	222	170	0.77	64	0.29	n/a	n/a	85	0.38	194	0.87	n/a	n/a	116	0.52
2005	248	147	0.59	171	0.69	n/a	n/a	113	0.46	111	0.45	n/a	n/a	130	0.52
2006	430	259	0.60	254	0.59	n/a	n/a	268	0.62	176	0.41	n/a	n/a	124	0.29
2007	257	156	0.61	106	0.41	n/a	n/a	88	0.34	94	0.37	n/a	n/a	117	0.46
2008	325	122	0.38	153	0.47	n/a	n/a	130	0.40	73	0.22	n/a	n/a	92	0.28
2009	251	133	0.53	121	0.48	n/a	n/a	96	0.38	112	0.45	n/a	n/a	112	0.45
2010	283	97	0.34	137	0.48	n/a	n/a	107	0.38	180	0.64	n/a	n/a	122	0.43
2011	384	96	0.25	93	0.24	n/a	n/a	93	0.24	95	0.25	n/a	n/a	97	0.25
2012	505	147	0.29	137	0.27	n/a	n/a	138	0.27	172	0.34	n/a	n/a	145	0.29
2013	370	147	0.40	135	0.36	90	0.24	132	0.36	77	0.21	87	0.23	114	0.31
2014	263	54	0.21	61	0.23	33	0.13	77	0.29	28	0.11	43	0.16	100	0.38

Figure 6-93

Comparison of 2014 Daily Flow at Vulture-Polar with the Historical Record, 1997 to 2014

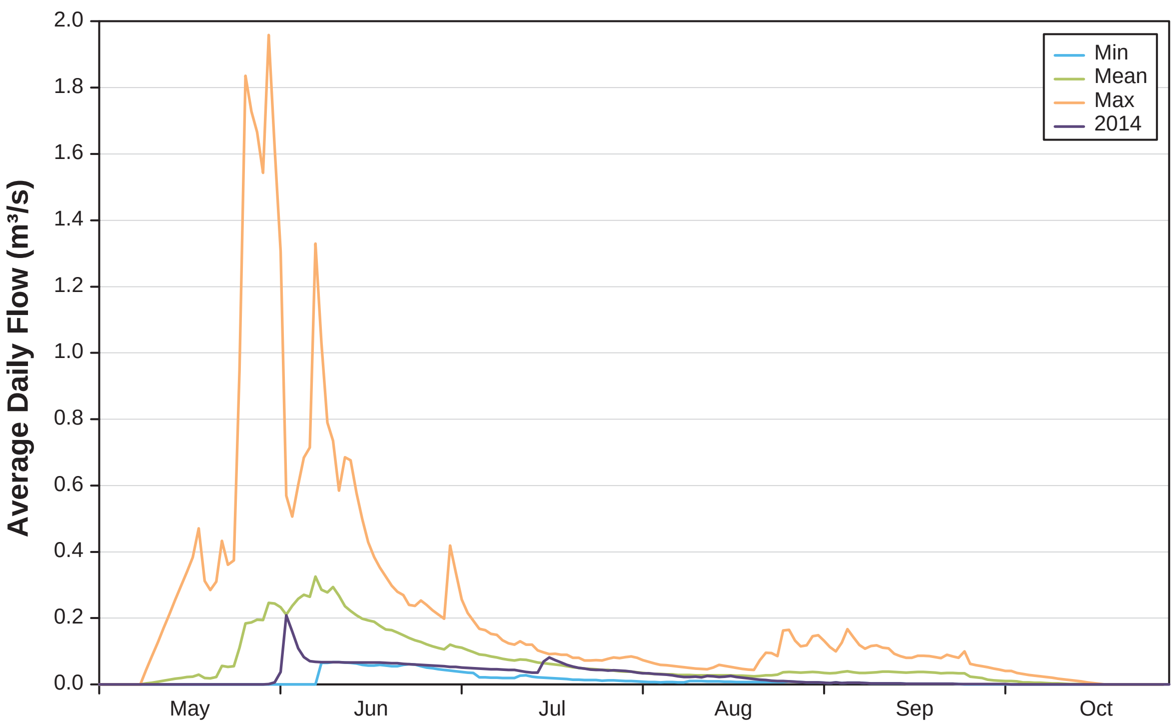
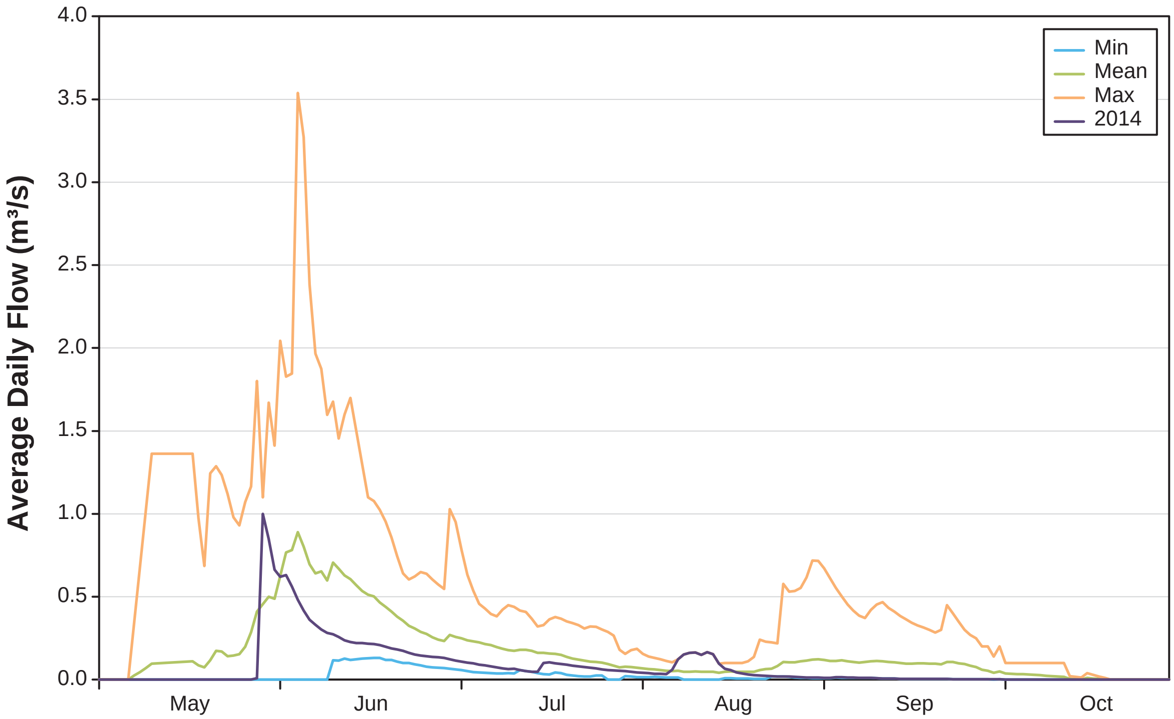


Figure 6-94

Comparison of 2014 Daily Flow at Lower PDC with the Historical Record, 1999 to 2014



Comparison of 2013-2014 Daily Flow at LLCF (1616-30) with Historical Record, 2000 to 2014

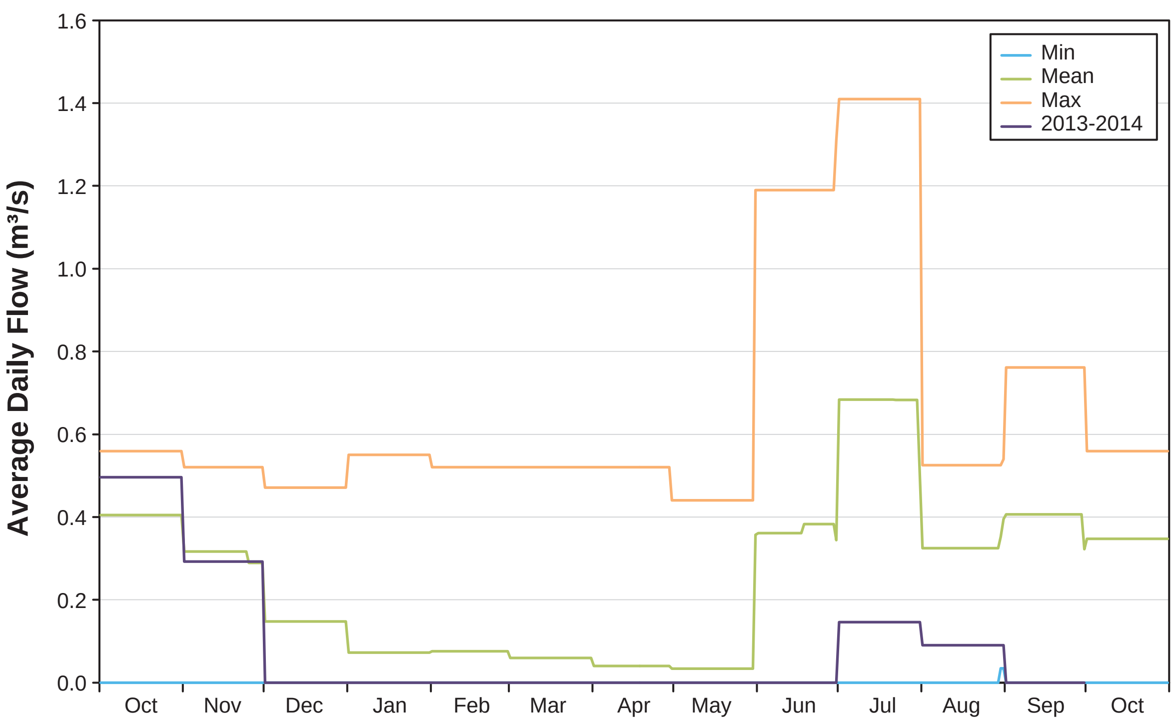


Figure 6-96

Comparison of 2014 Daily Flow at Slipper Lac de Gras with the Historical Record, 1994 to 2014

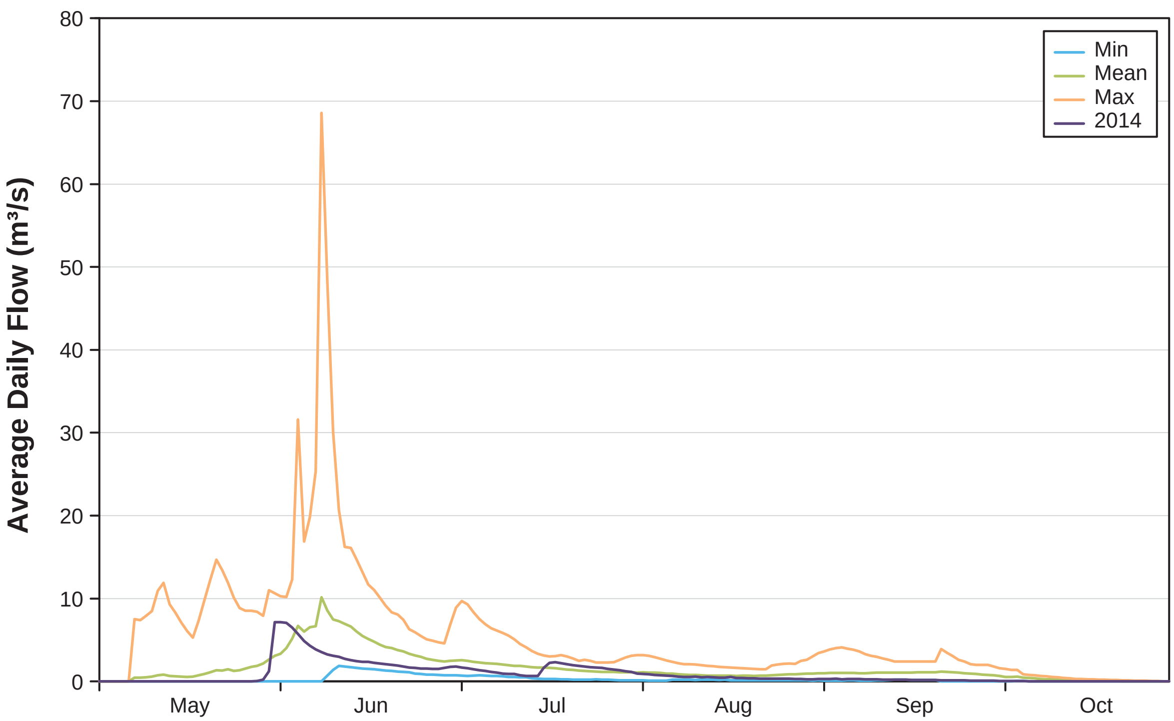


Figure 6-97

Comparison of 2013-2014 Daily Flow at KPSF (1616-43) with Historical Record, 2000 to 2014

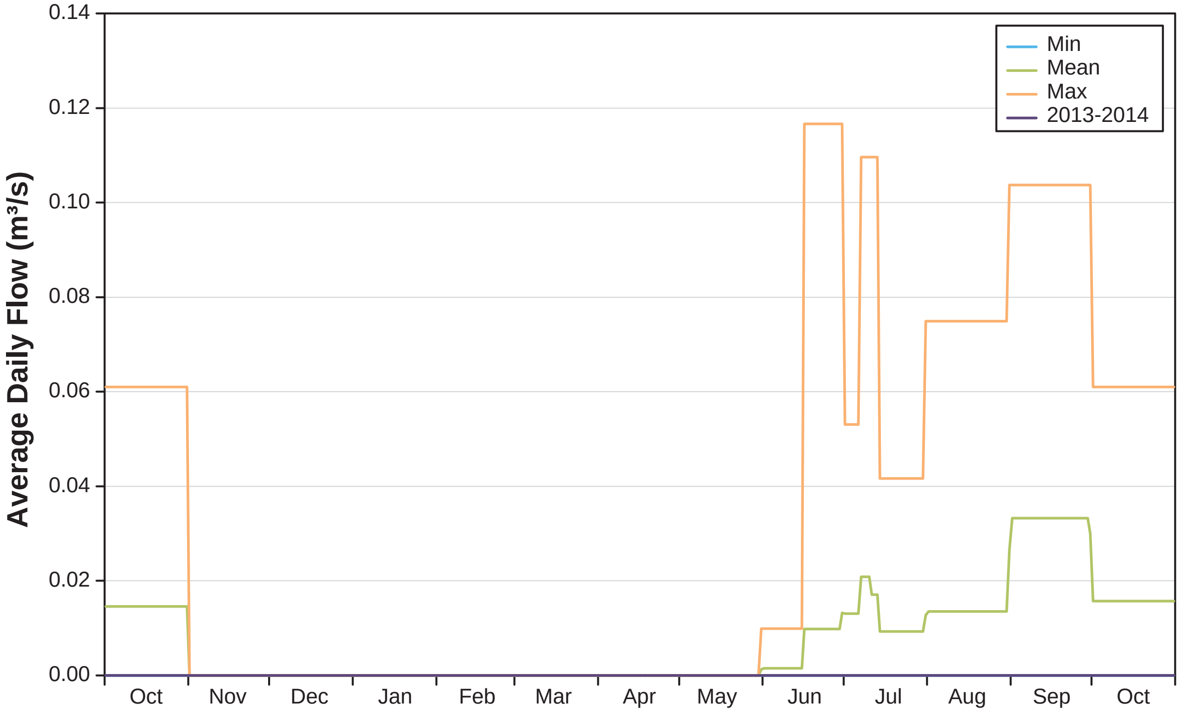


Figure 6-98

Comparison of 2014 Daily Flow at Cujo Outflow with the Historical Record, 1999 to 2014

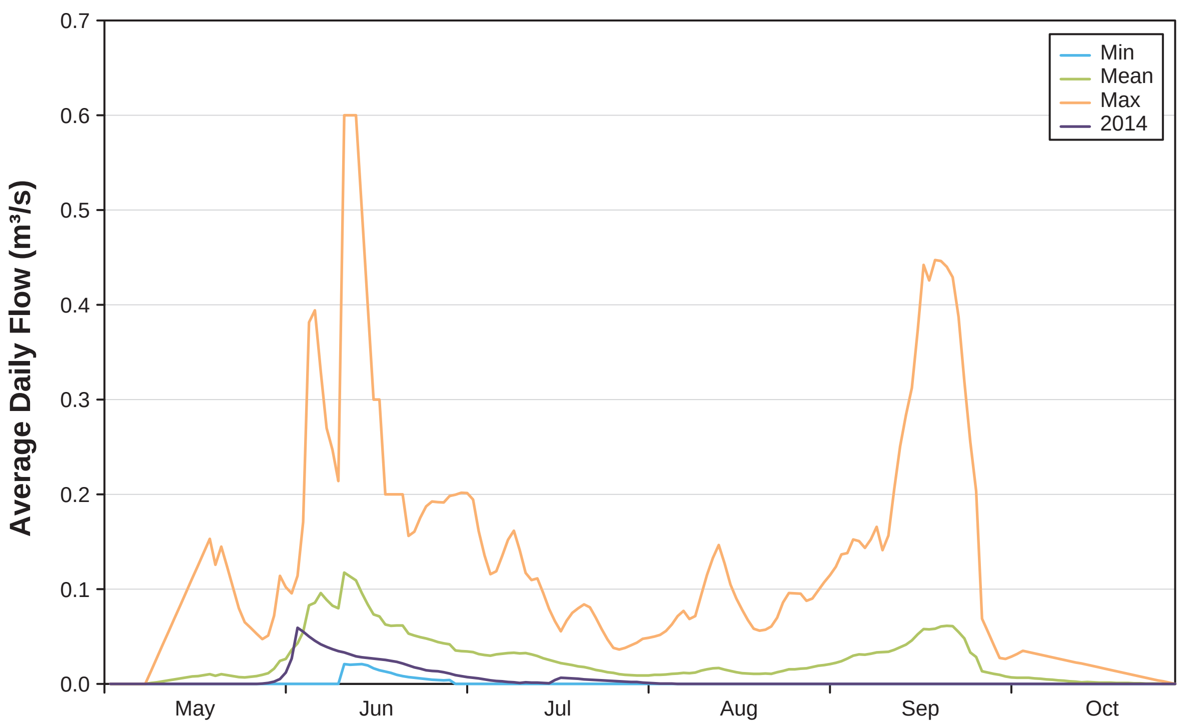


Figure 6-99

Comparison of 2014 Daily Flow at Counts Outflow with the Historical Record, 1997 to 2014

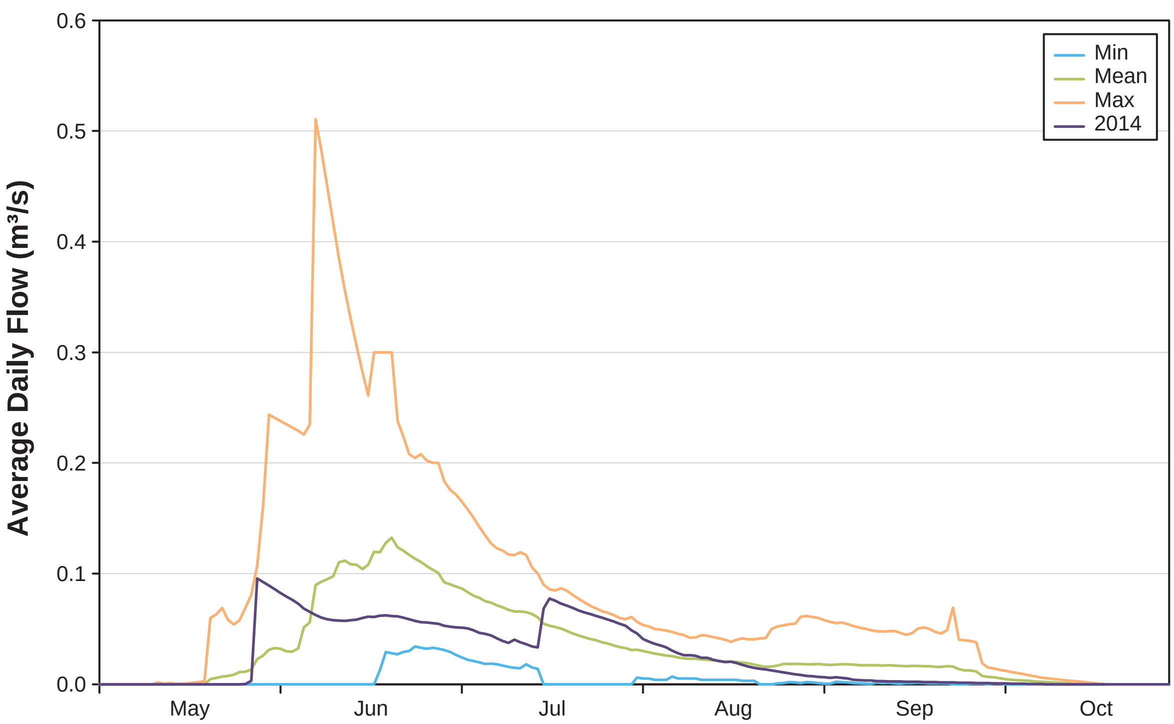
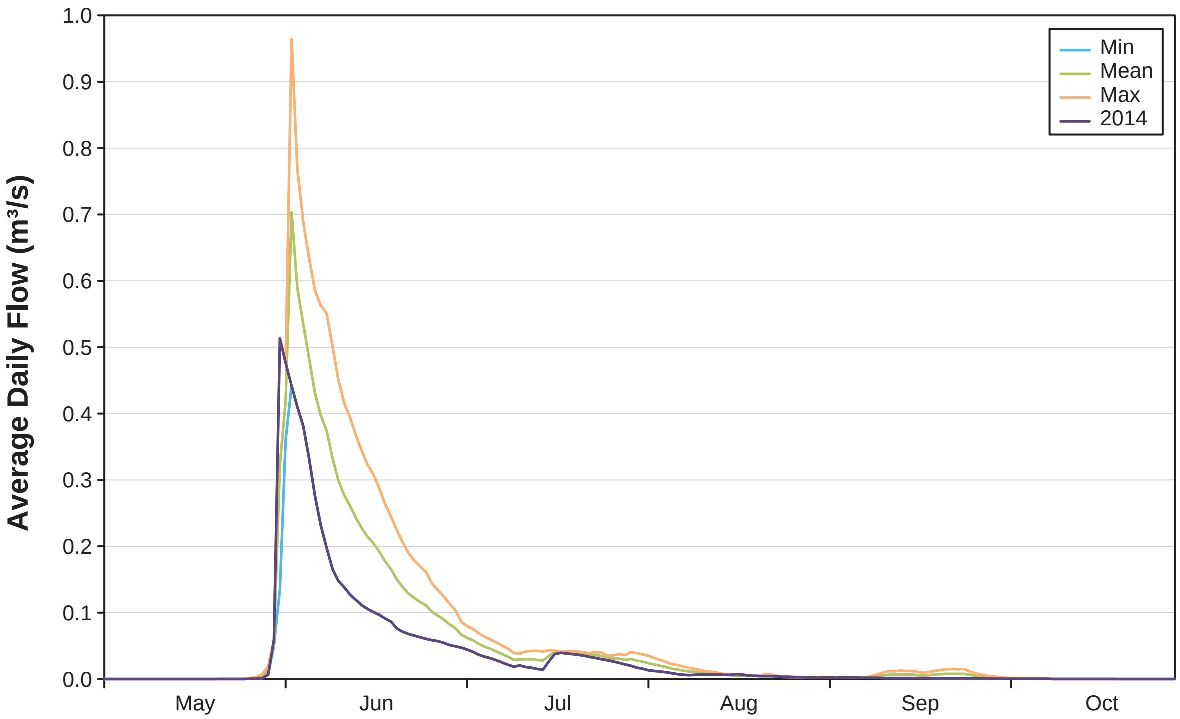


Figure 6-100

Comparison of 2014 Daily Flow at Christine-Lac du Sauvage with the Historical Record, 2013 to 2014



7. HISTORICAL SEDIMENT QUALITY

The AEMP Evaluation of Effects focuses on detecting changes in 11 lake sediment quality variables in the Koala Watershed and Lac de Gras and 12 lake sediment quality variables in the King-Cujo Watershed and Lac du Sauvage using samples collected once every three years (see Sections 2.2, 3.3, 4.3). However, each sediment sample is screened for 30 sediment quality variables in the laboratory (Table 7-1). Sediment samples have also been collected from Ekati Diamond Mine lakes at other times as part of the sampling program for other projects (e.g., Kodiak Lake Sewage Effects Study; Rescan 2000). Historical averages for 29 of the sediment quality variables (excludes moisture content) for the three reference lakes (Nanuq, Counts, and Vulture) and each of the lakes that is monitored for sediment quality in the Koala and King-Cujo Watersheds as well as Lac de Gras and Lac du Sauvage are presented below (Figures 7-1 to 7-29). Data presented in these figures only includes data collected using Ekman grabs (see Sections 3.3.2 and 4.3.2).

Table 7-1. AEMP Sediment Quality Variables in the Koala and King-Cujo Watersheds

Variable	Figure Number	Variable	Figure Number
Physical Tests		**Metals (*cont'd*)**	
% Moisture	Not Shown	Chromium	7-14
Particle Size		Cobalt	7-15
% Gravel	7-1	Copper	7-16
% Sand	7-2	Iron	7-17
% Silt	7-3	Lead	7-18
% Clay	7-4	Manganese	7-19
Nutrients/Organics		Mercury	7-20
Total Organic Carbon	7-5	Molybdenum	7-21
Available Phosphorus	7-6	Nickel	7-22
Total Nitrogen	7-7	Phosphorus	7-23
Metals		Selenium	7-24
Aluminum	7-8	Silver	7-25
Antimony	7-9	Strontium	7-26
Arsenic	7-10	Uranium	7-27
Barium	7-11	Vanadium	7-28
Boron	7-12	Zinc	7-29
Cadmium	7-13		

Starting in 2014, the AEMP Evaluation of Effects also focused on detecting changes in 11 lake sediment quality variables in the Pigeon-Fay and Upper Exeter Watershed, using samples collected in August (see Sections 2.2, and 5.3). However, each sediment sample is screened for 30 sediment quality variables in the laboratory (Table 7-2). Historical averages for 29 of the sediment quality variables (excludes moisture content) for the three reference lakes (Nanuq, Counts, and Vulture) and each of the lakes that is monitored for sediment quality in the Pigeon-Fay and Upper Exeter Watershed are presented below (Figures 7-30 to 7-58). Data presented in these figures only includes data collected using Ekman grabs (see Section 5.3.2 and 4.3.2).

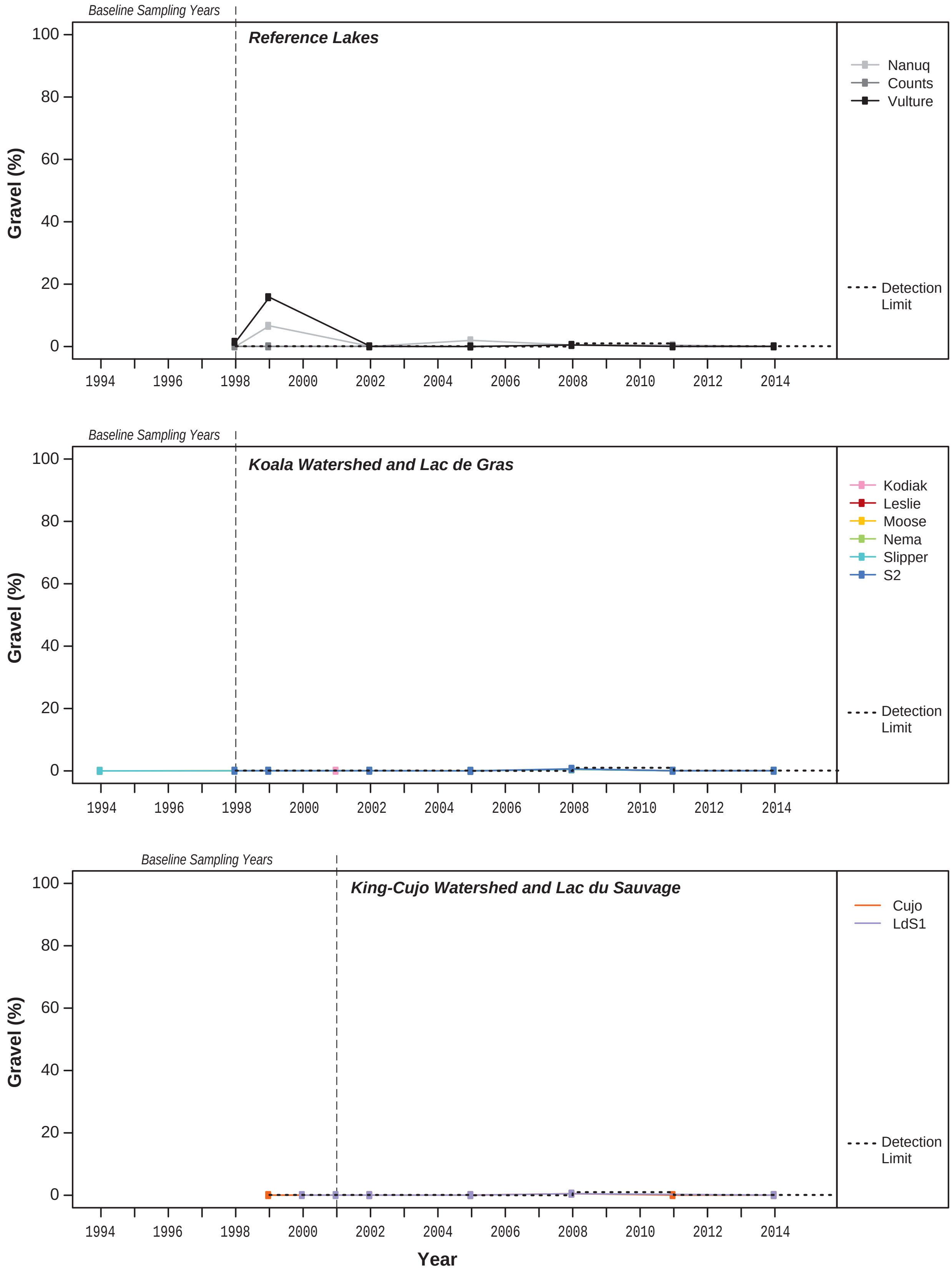
Table 7-2. AEMP Sediment Quality Variables in the Pigeon-Fay and Upper Exeter Watershed

Variable	Figure Number	Variable	Figure Number
Physical Tests		**Metals (*cont'd*)**	
% Moisture	Not Shown	Chromium	7-43
Particle Size		Cobalt	7-44
% Gravel	7-30	Copper	7-45
% Sand	7-31	Iron	7-46
% Silt	7-32	Lead	7-47
% Clay	7-33	Manganese	7-48
Nutrients/Organics		Mercury	7-49
Total Organic Carbon	7-34	Molybdenum	7-50
Available Phosphorus	7-35	Nickel	7-51
Total Nitrogen	7-36	Phosphorus	7-52
Metals	7-37	Selenium	7-53
Aluminum	7-37	Silver	7-54
Antimony	7-38	Strontium	7-55
Arsenic	7-39	Uranium	7-56
Barium	7-40	Vanadium	7-57
Boron	7-41	Zinc	7-58
Cadmium	7-42		

CCME sediment quality guidelines for the protection of aquatic life are provided where applicable (CCME 2014b). 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.

Figure 7-1

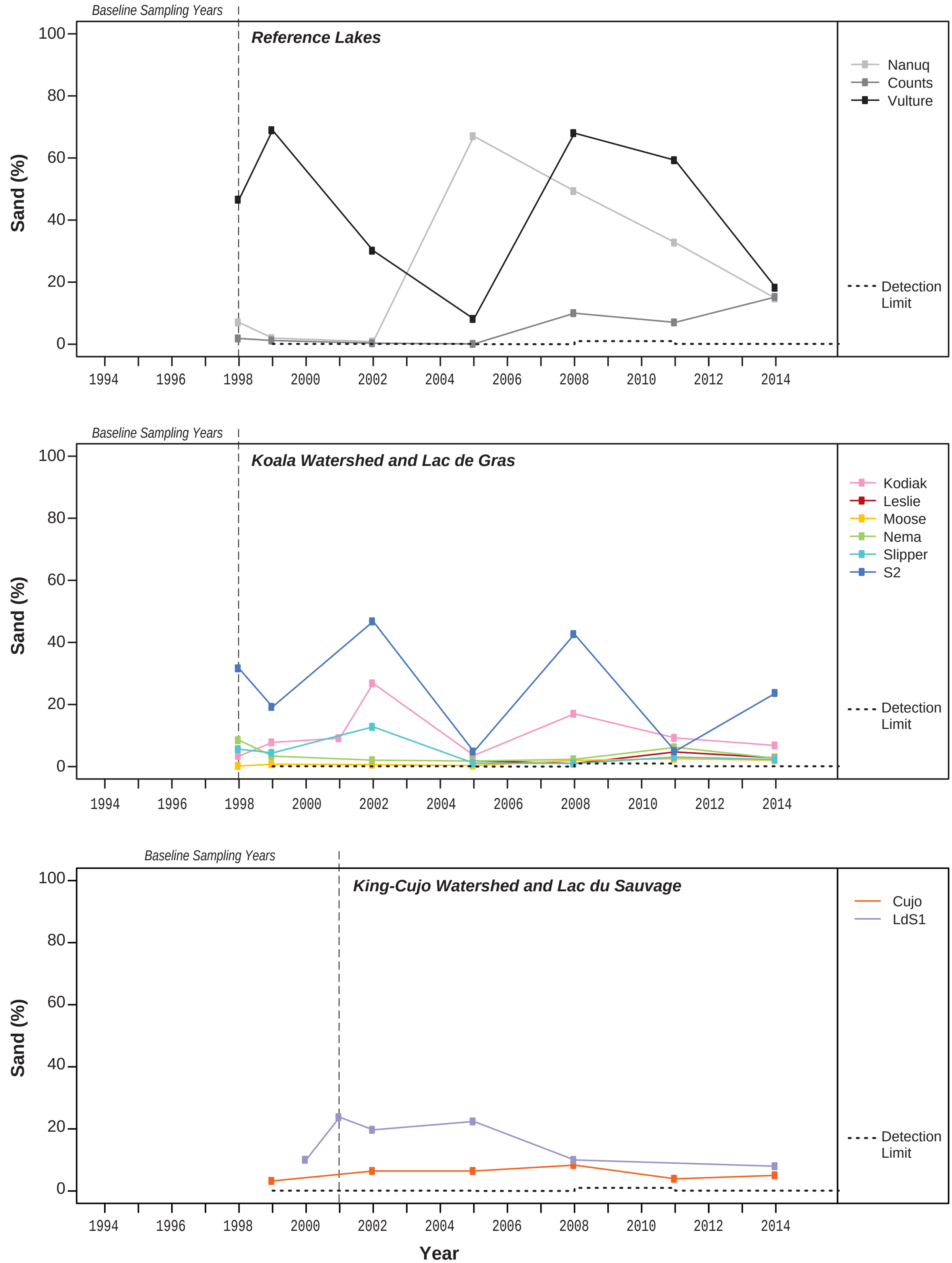
Percent Gravel in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-2

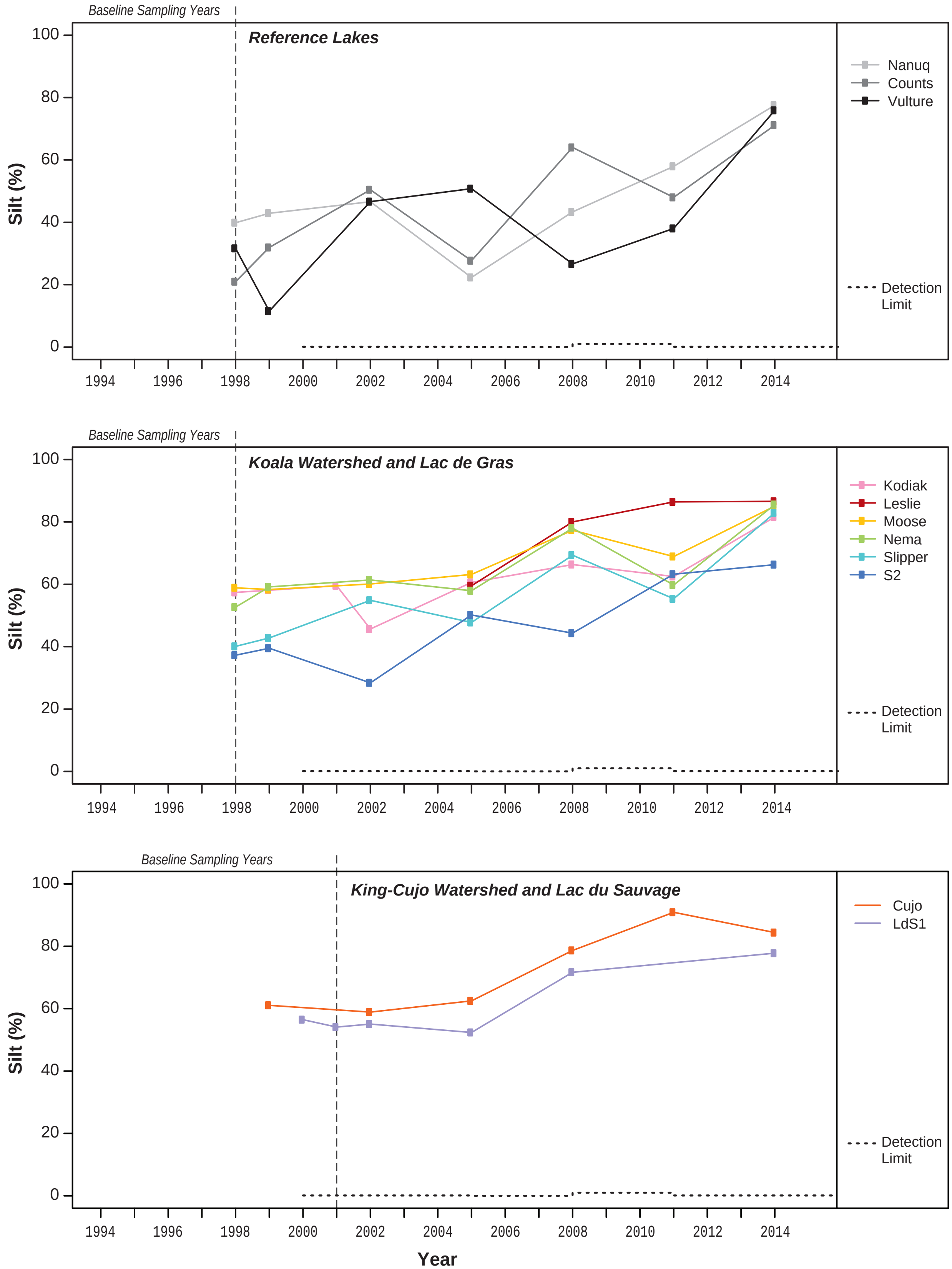
Percent Sand in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-3

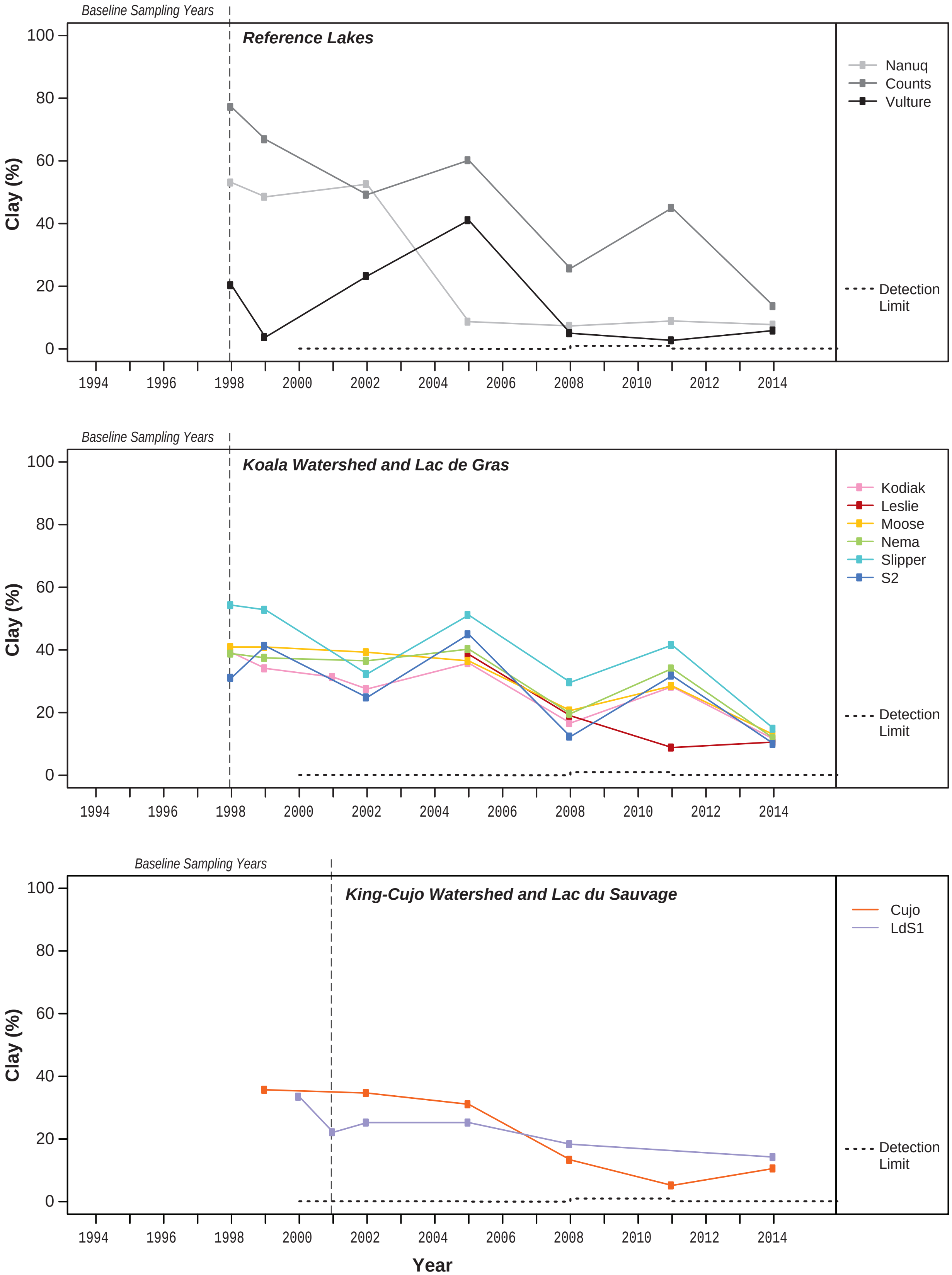
Percent Silt in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-4

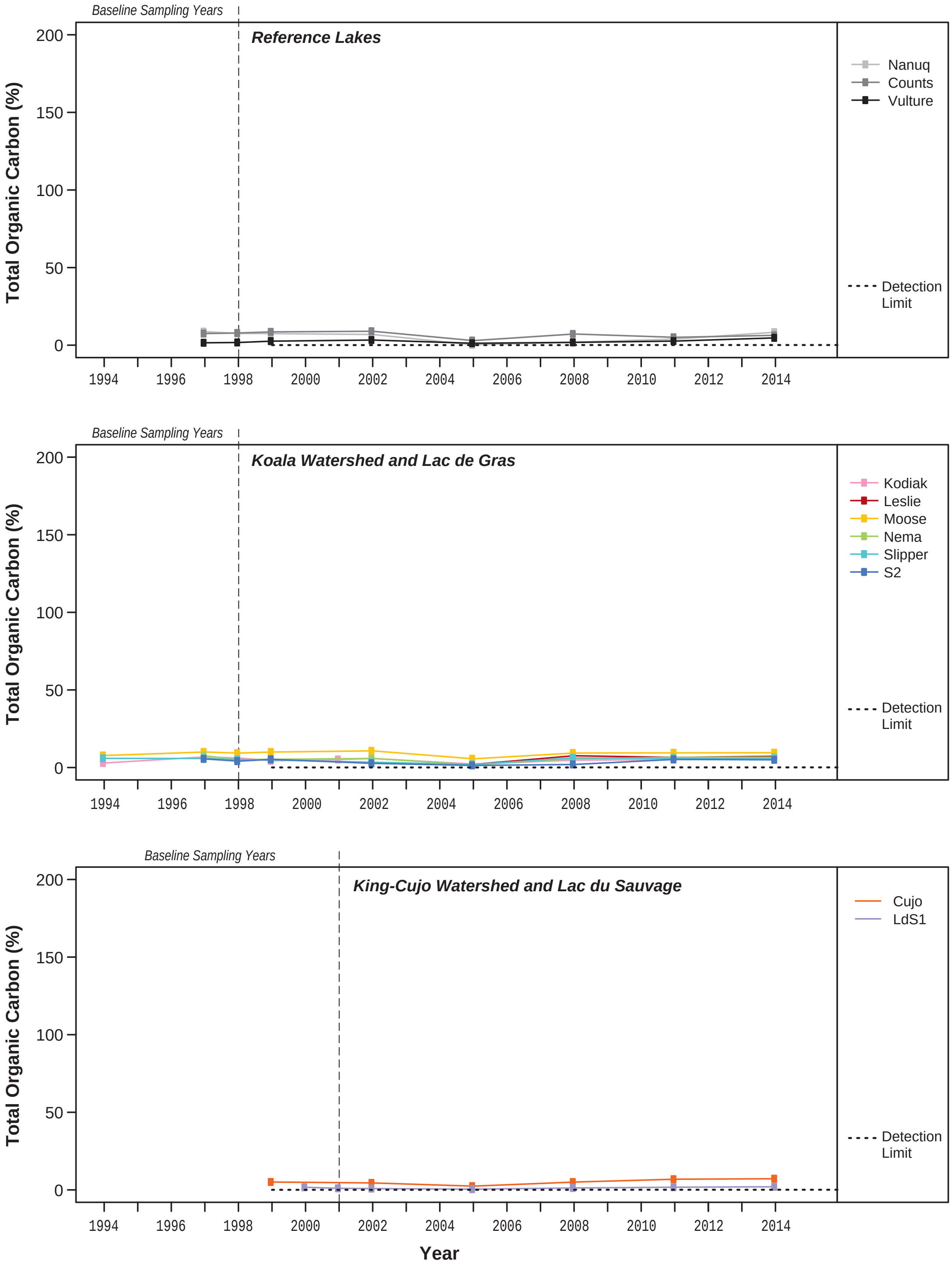
Percent Clay
in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

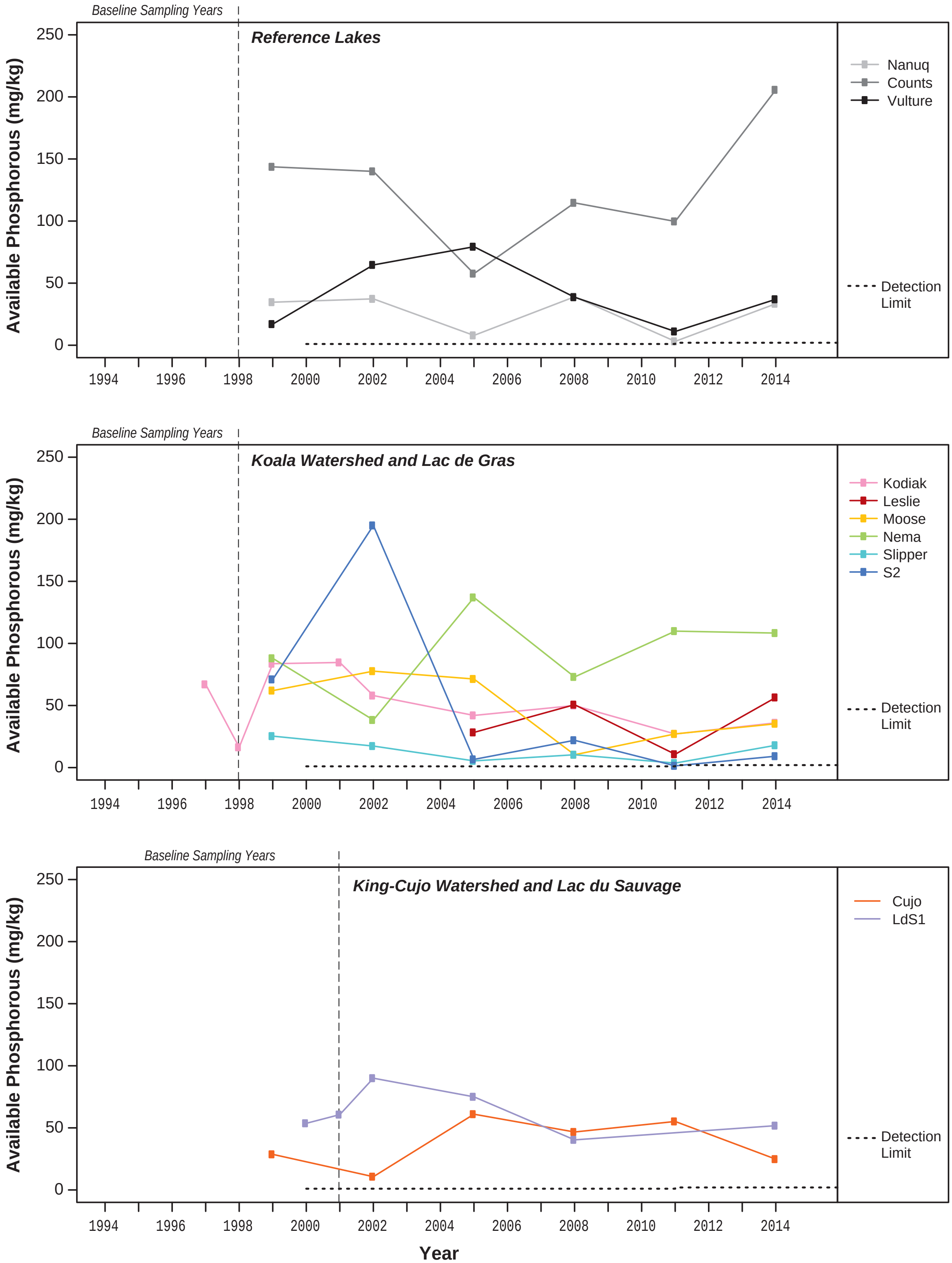
Figure 7-5

Total Organic Carbon in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

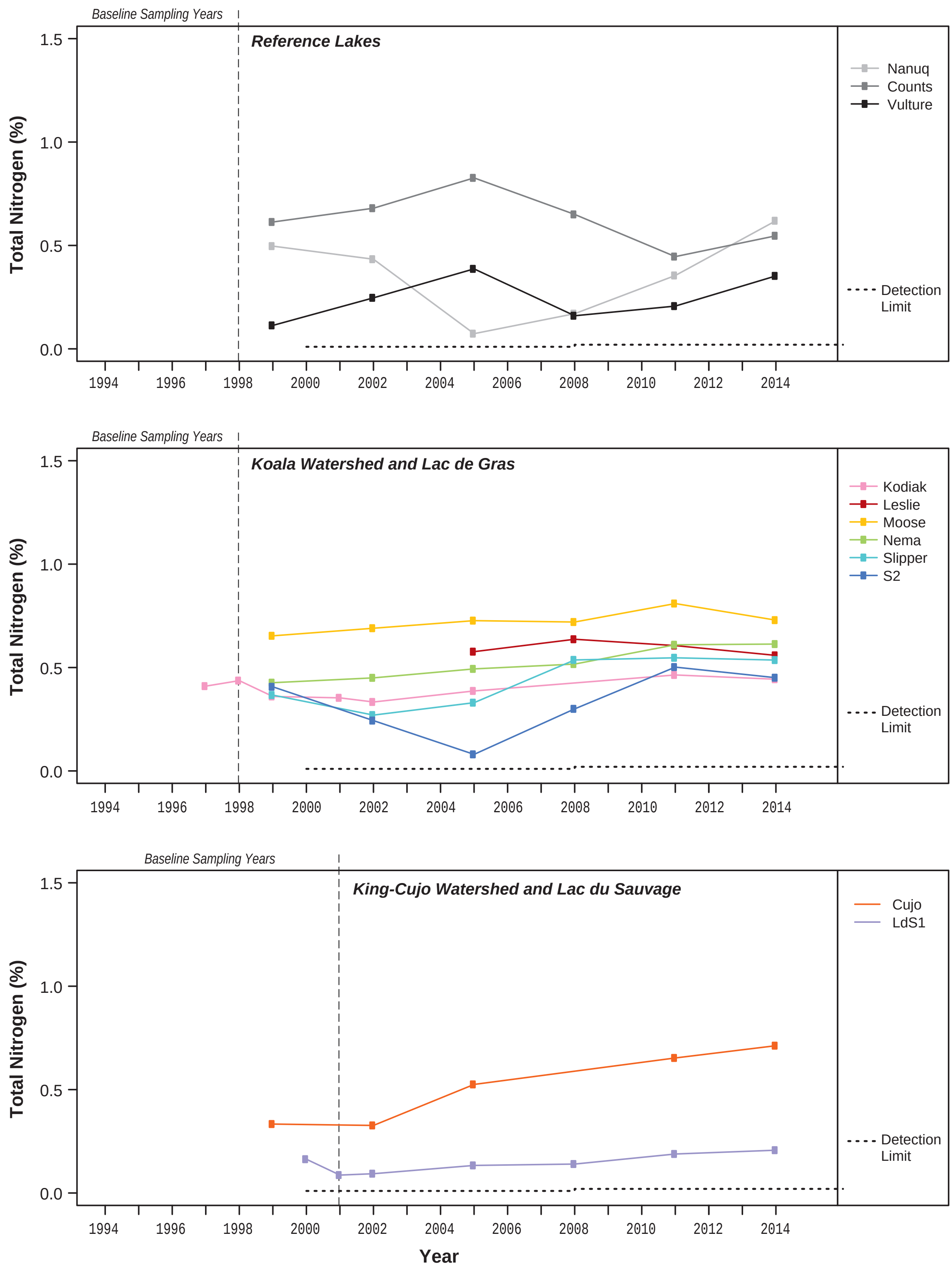
Figure 7-6
Available Phosphorus in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-7

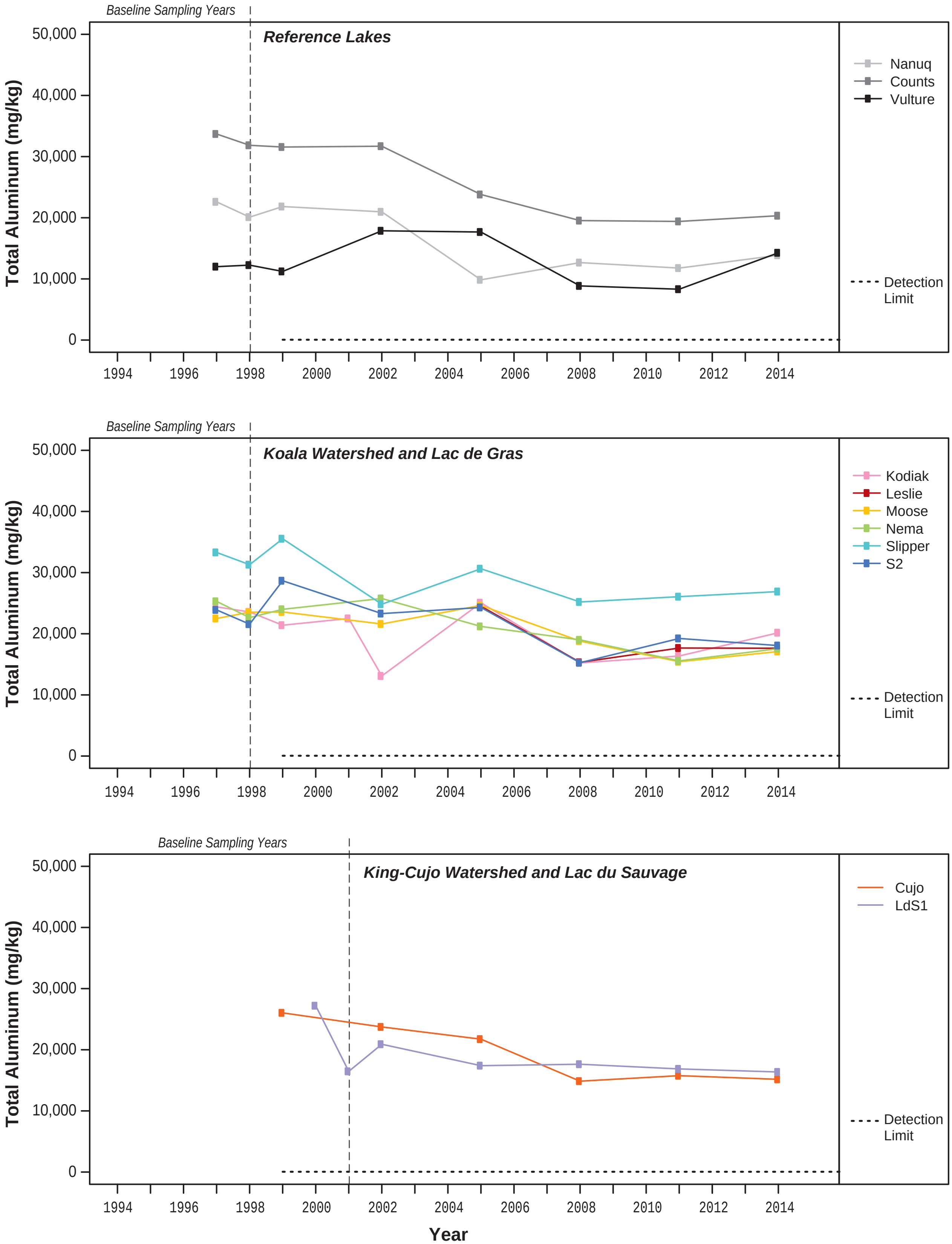
Total Nitrogen
in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-8

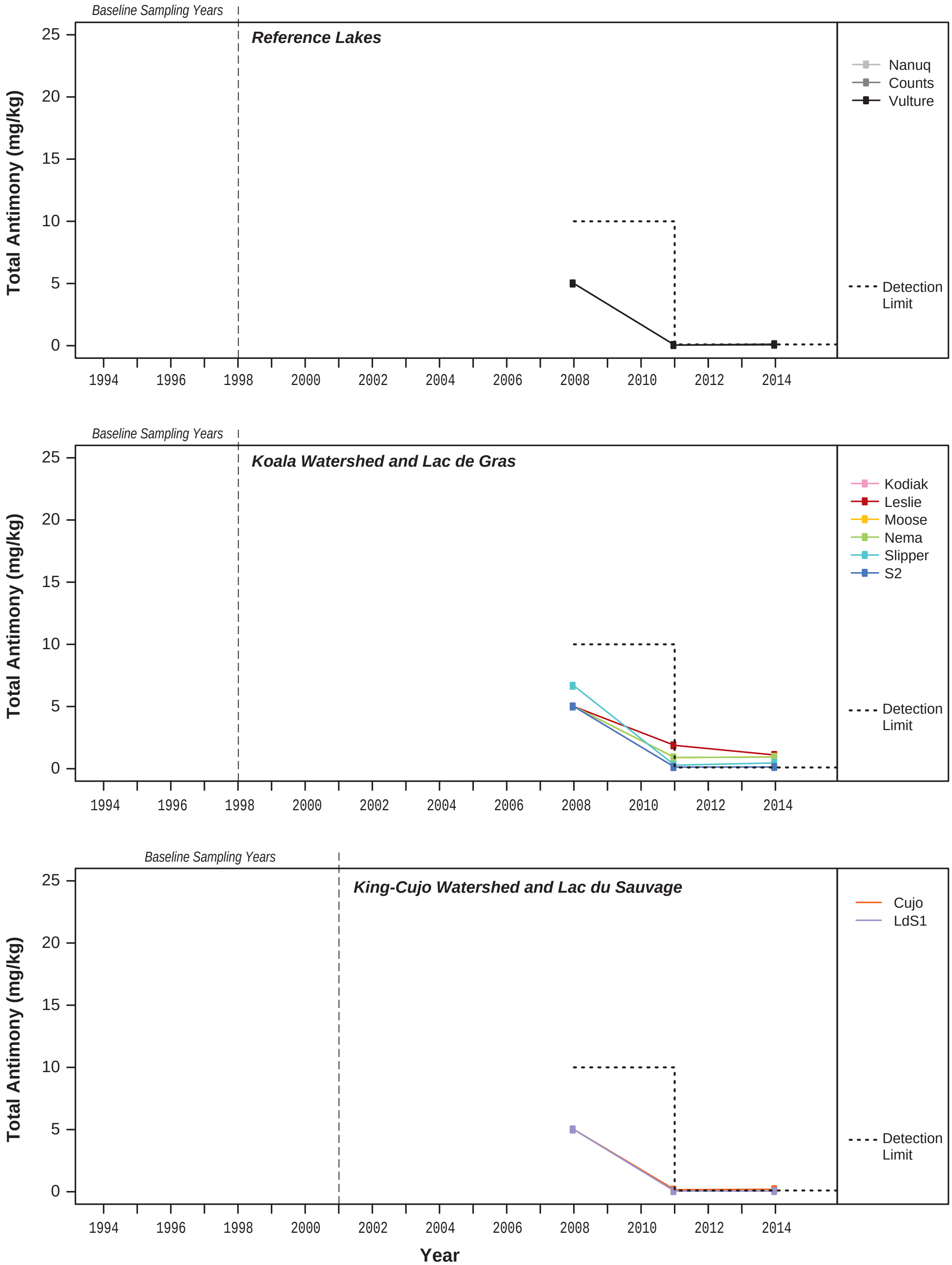
Total Aluminum in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-9

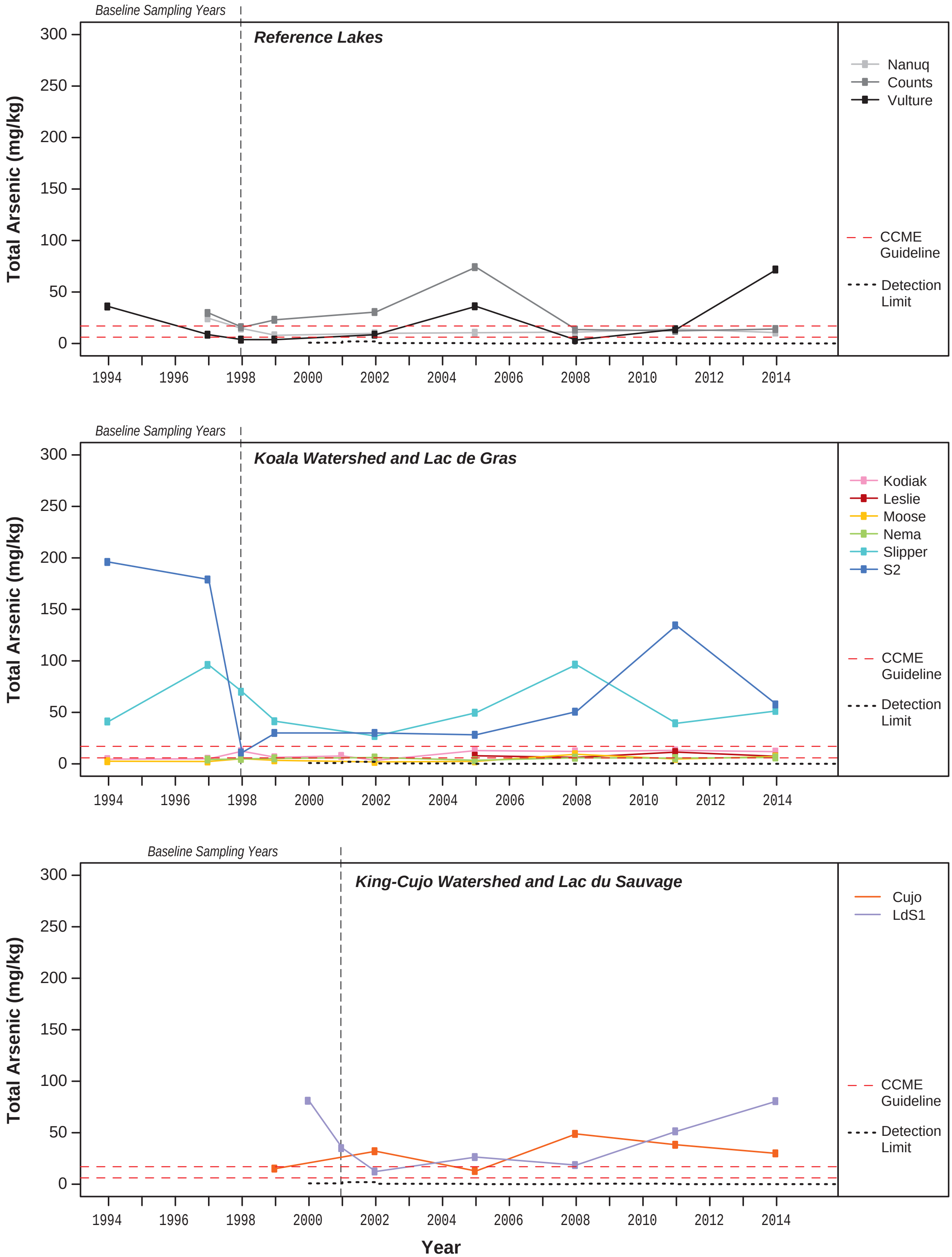
Total Antimony in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-10

Total Arsenic in AEMP Lake Sediments, 1994 to 2014

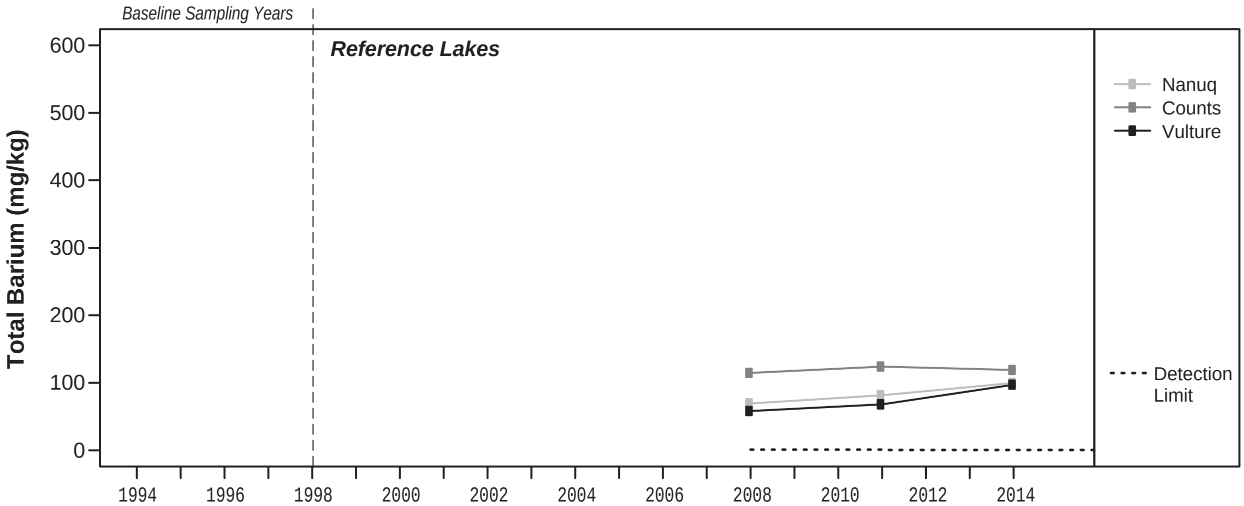


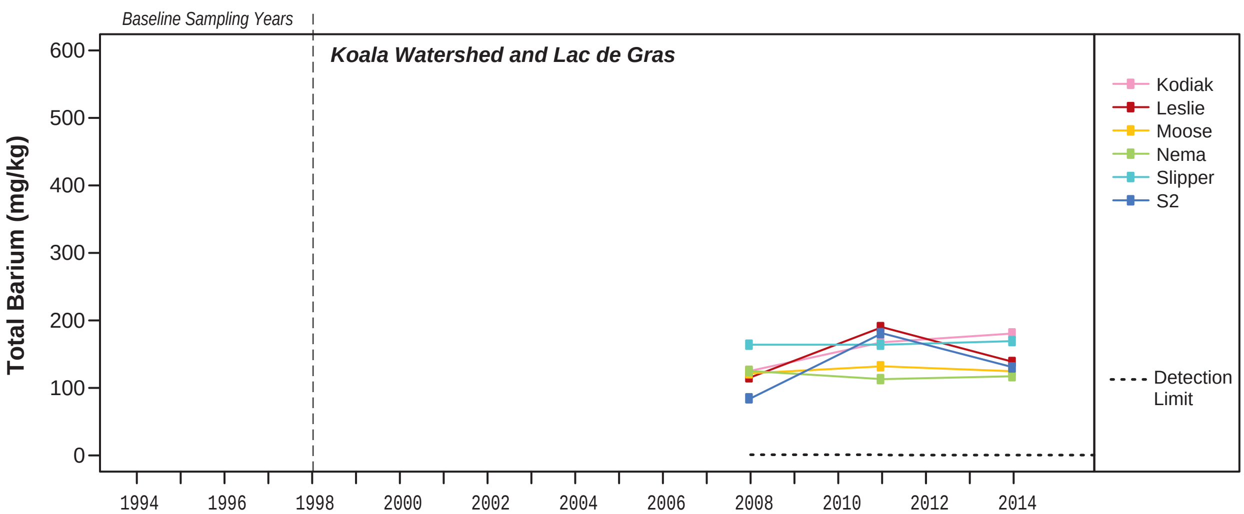
Notes: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.
CCME guidelines: ISQG = 5.9 mg/kg; PEL = 17 mg/kg.

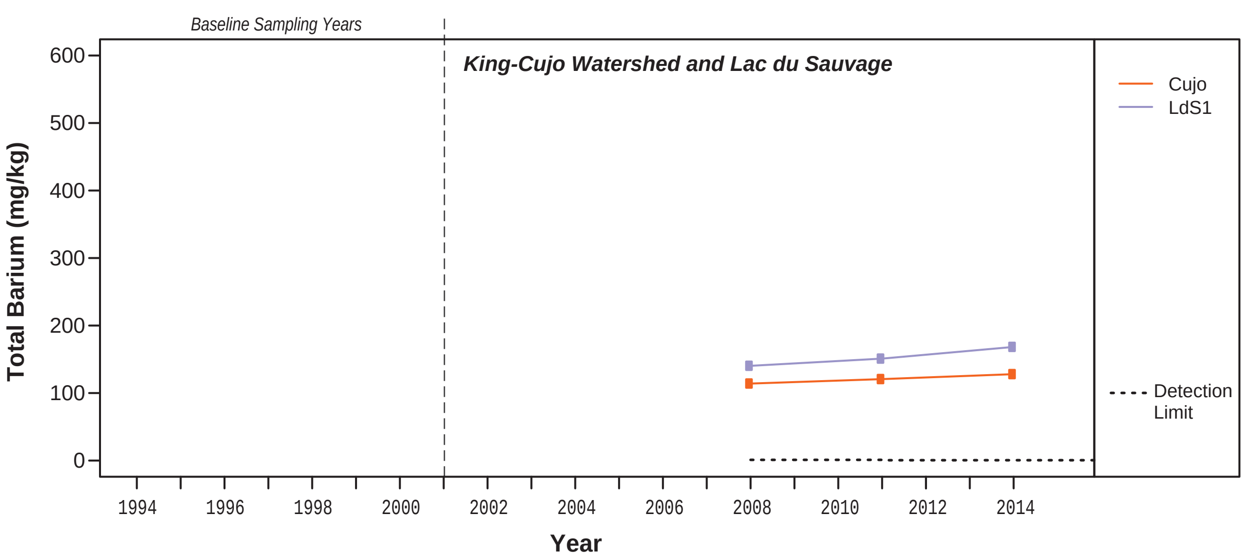
Figure 7-11

Total Barium in AEMP Lake Sediments, 1994 to 2014

ERM



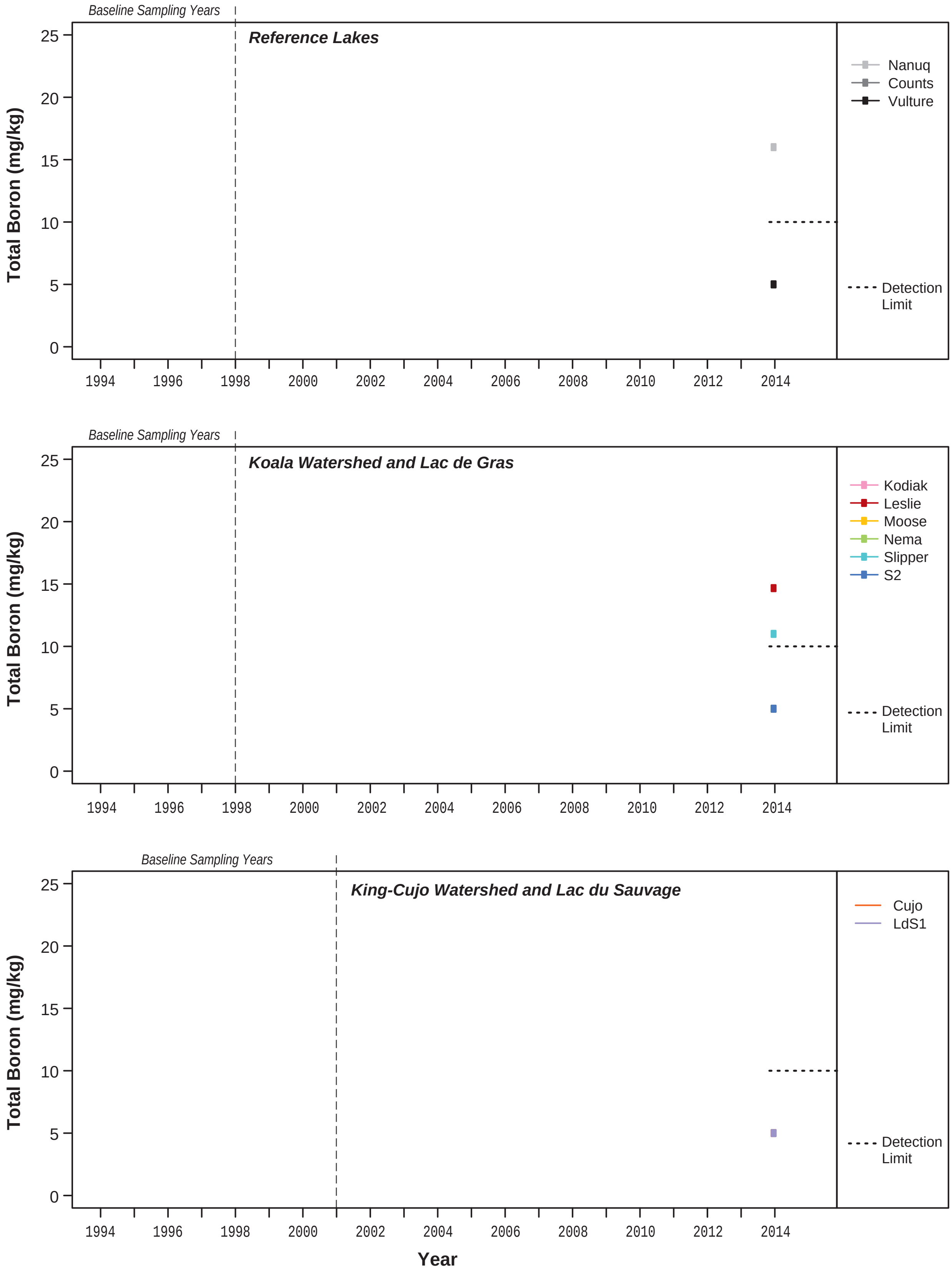




Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-12

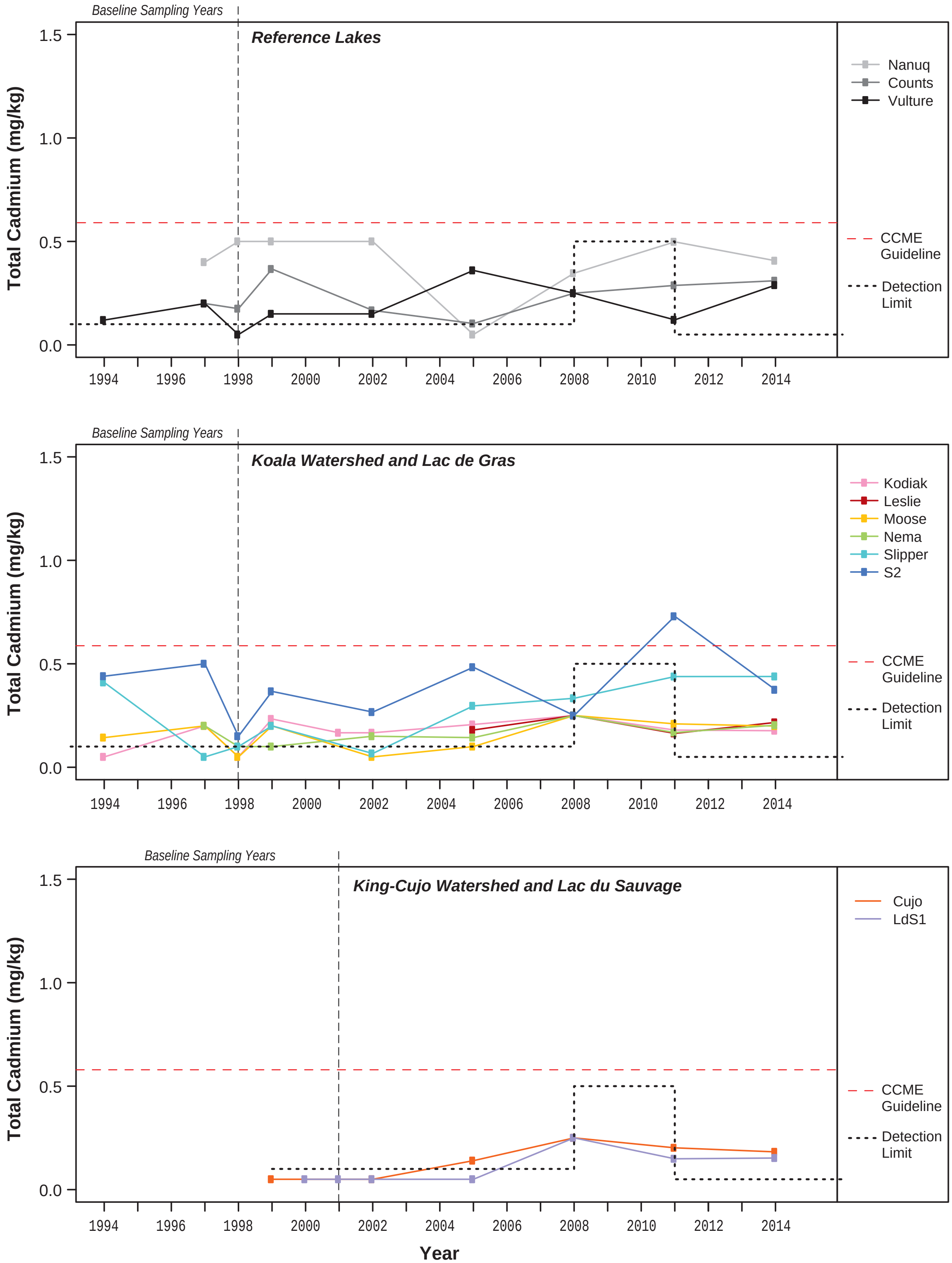
Total Boron in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-13

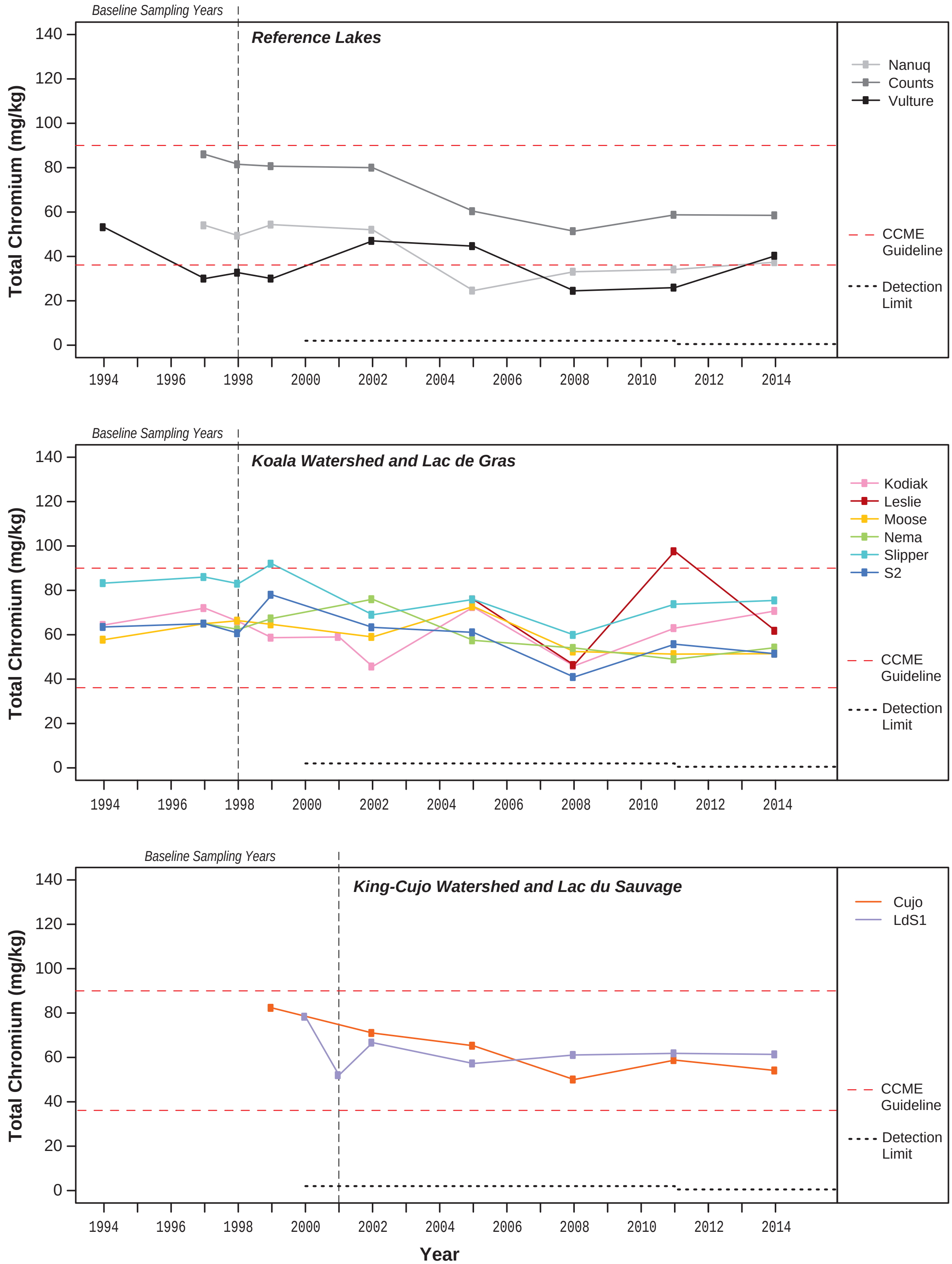
Total Cadmiuim in AEMP Lake Sediments, 1994 to 2014



Notes: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.
CCME guidelines: ISQG = 0.6 mg/kg; PEL = 3.5 mg/kg (not shown).

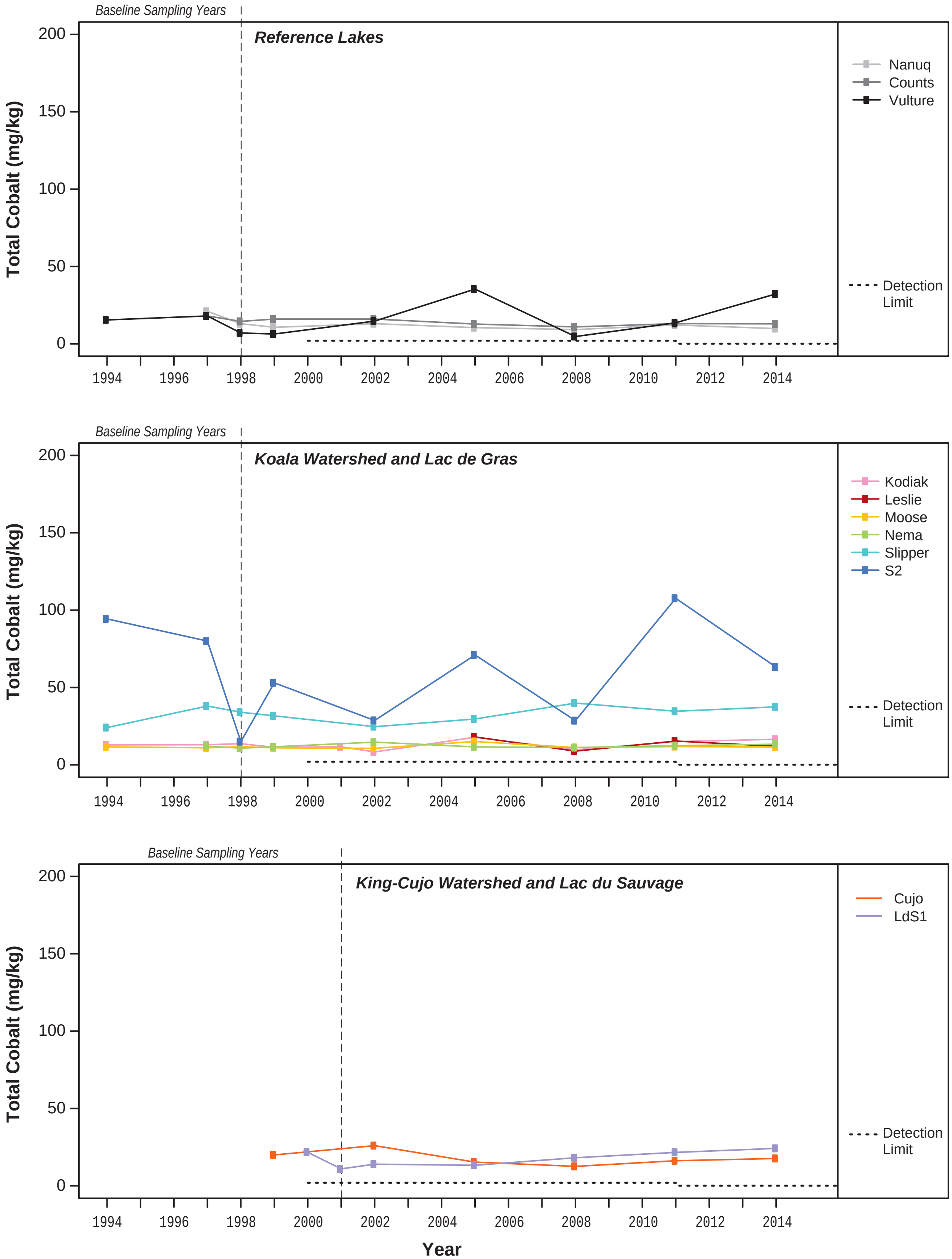
Figure 7-14

Total Chromium in AEMP Lake Sediments, 1994 to 2014



Notes: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.
CCME guidelines: ISQG = 37.3 mg/kg; PEL = 90 mg/kg.

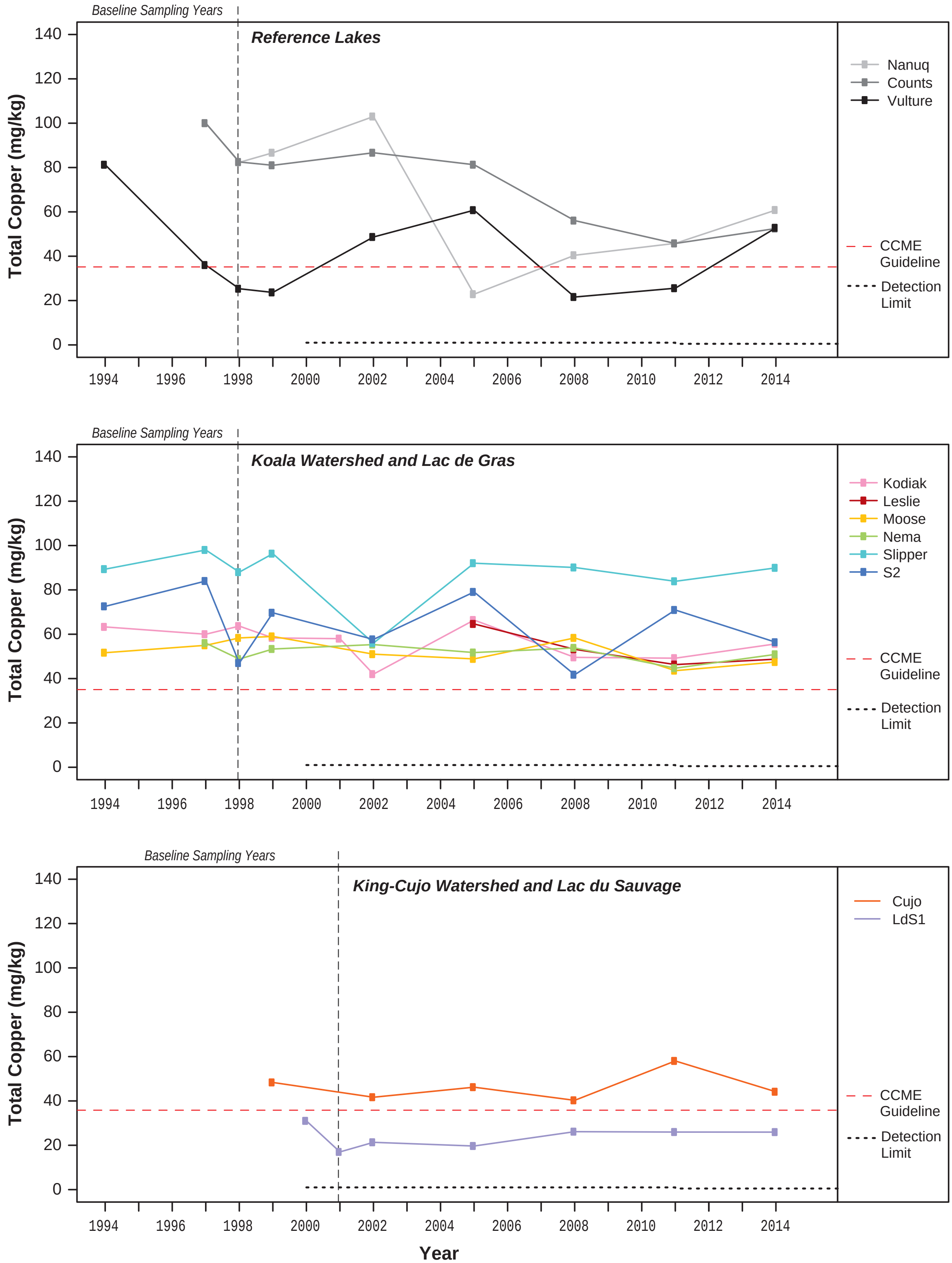
Figure 7-15
Total Cobalt in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

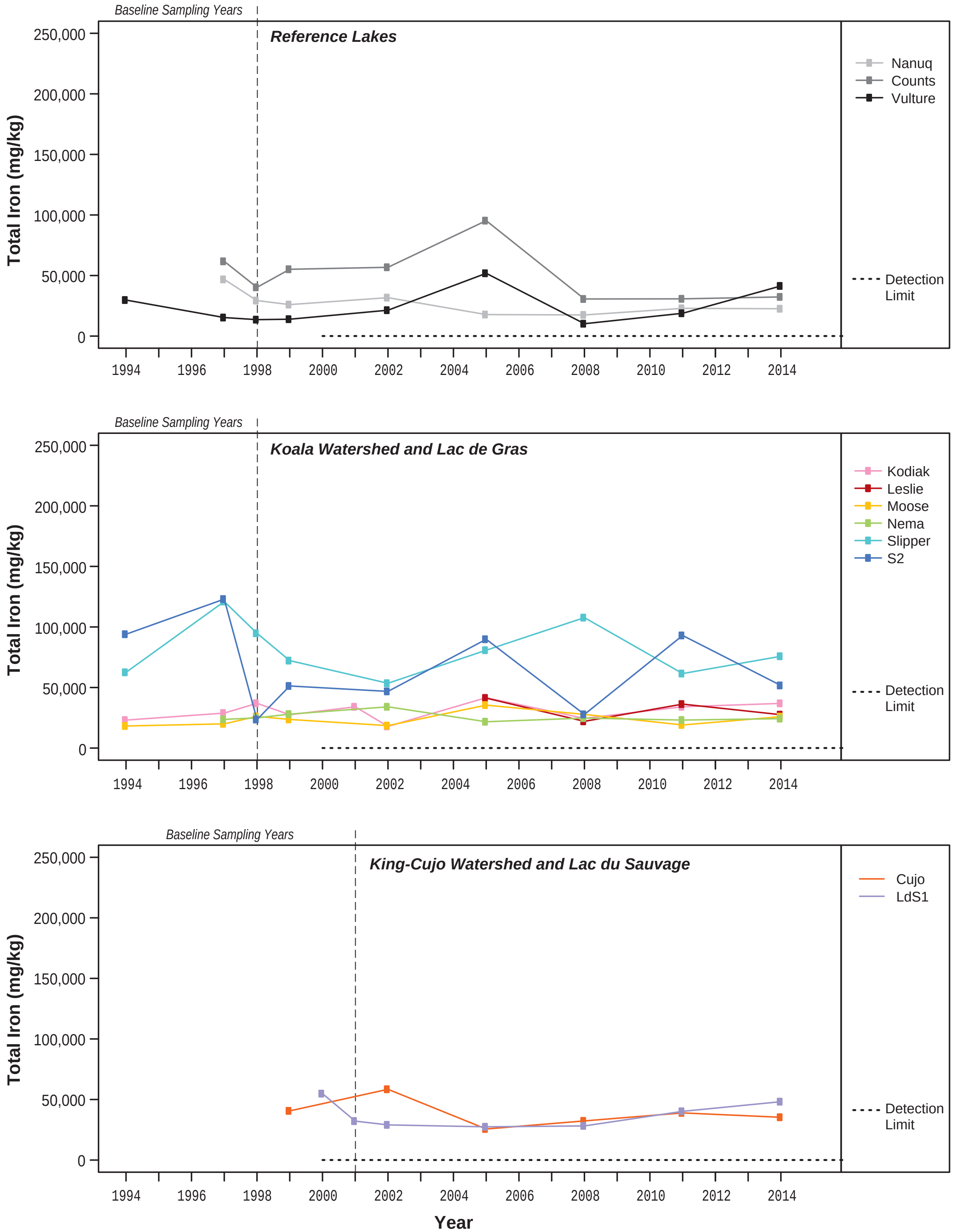
Figure 7-16

Total Copper in AEMP Lake Sediments, 1994 to 2014



Notes: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.
CCME guidelines: ISQG = 35.7 mg/kg; PEL = 197 mg/kg (not shown).

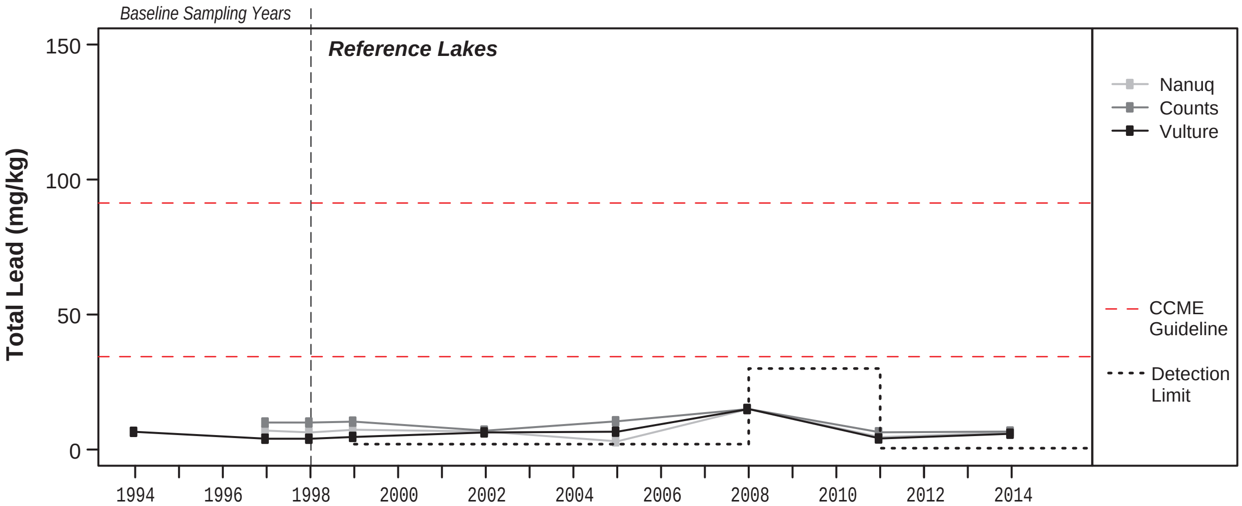
Figure 7-17
Total Iron
in AEMP Lake Sediments, 1994 to 2014

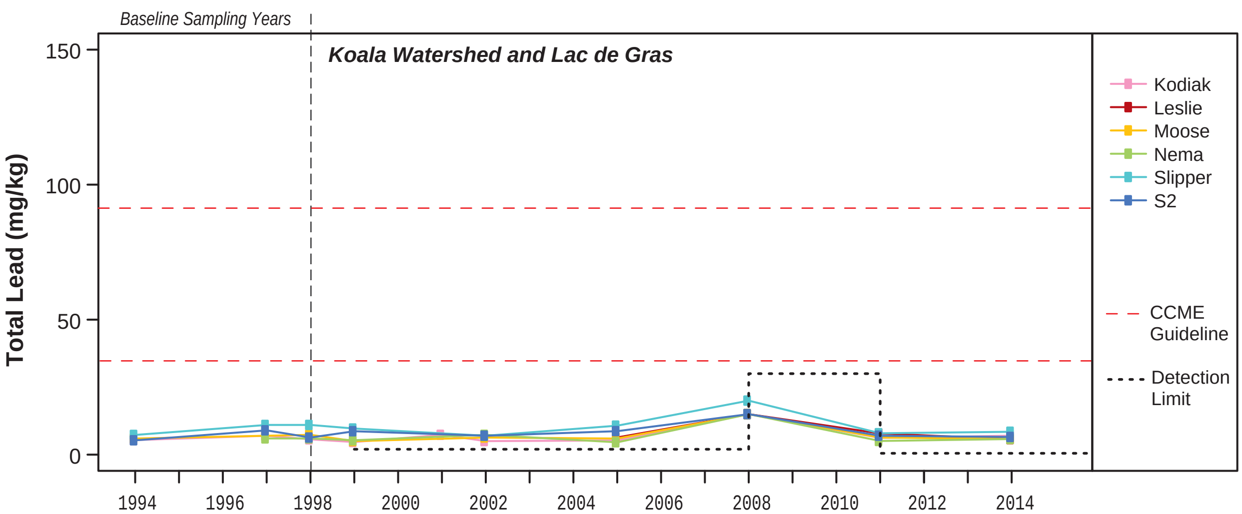


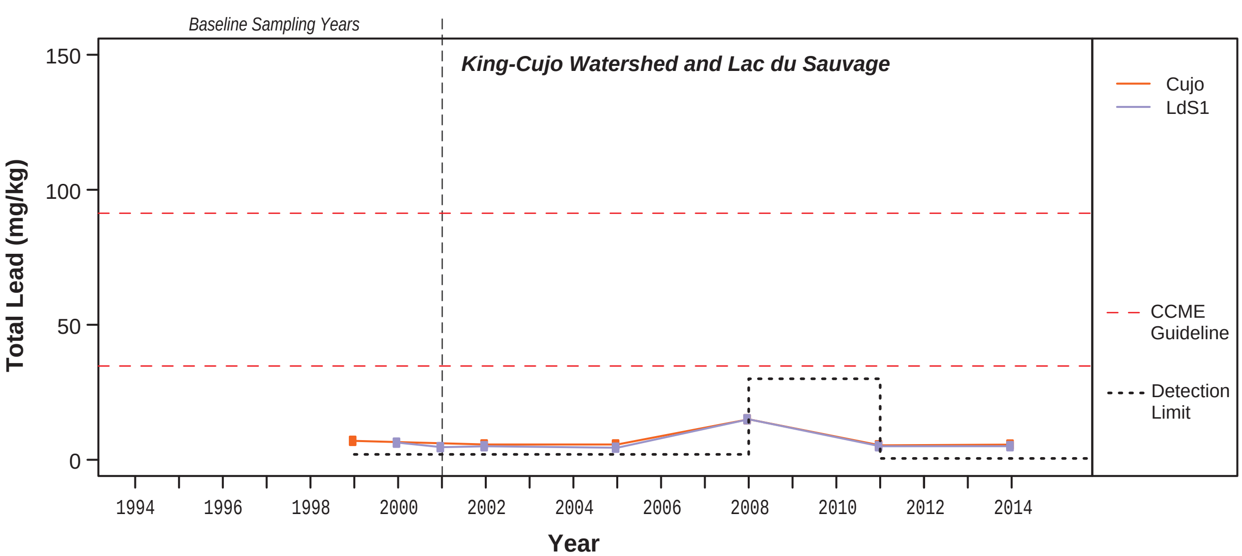
Notes: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-18

Total Lead in AEMP Lake Sediments, 1994 to 2014



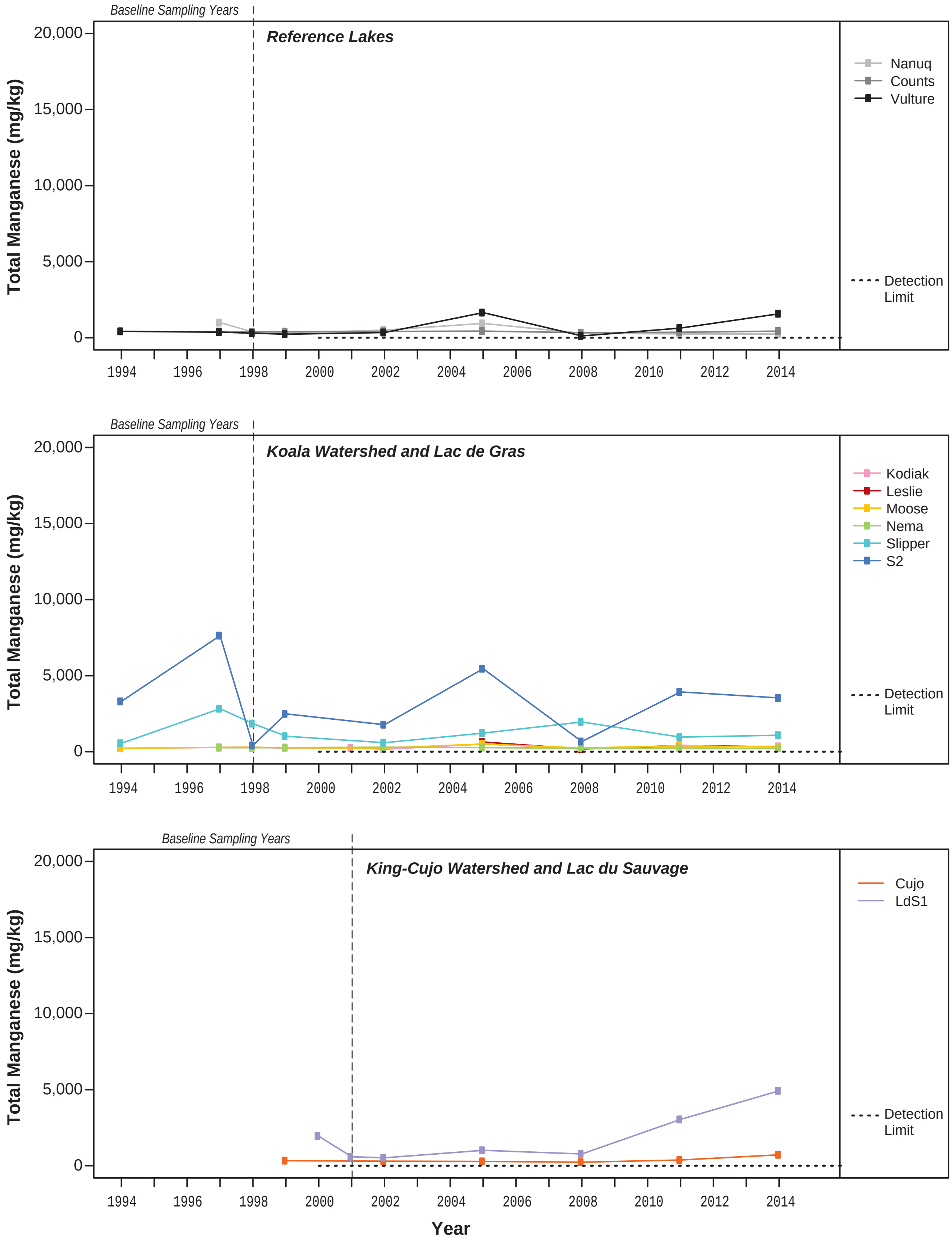




Notes: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.
CCME guidelines: ISQG = 35 mg/kg; PEL = 91.3 mg/kg.

Figure 7-19

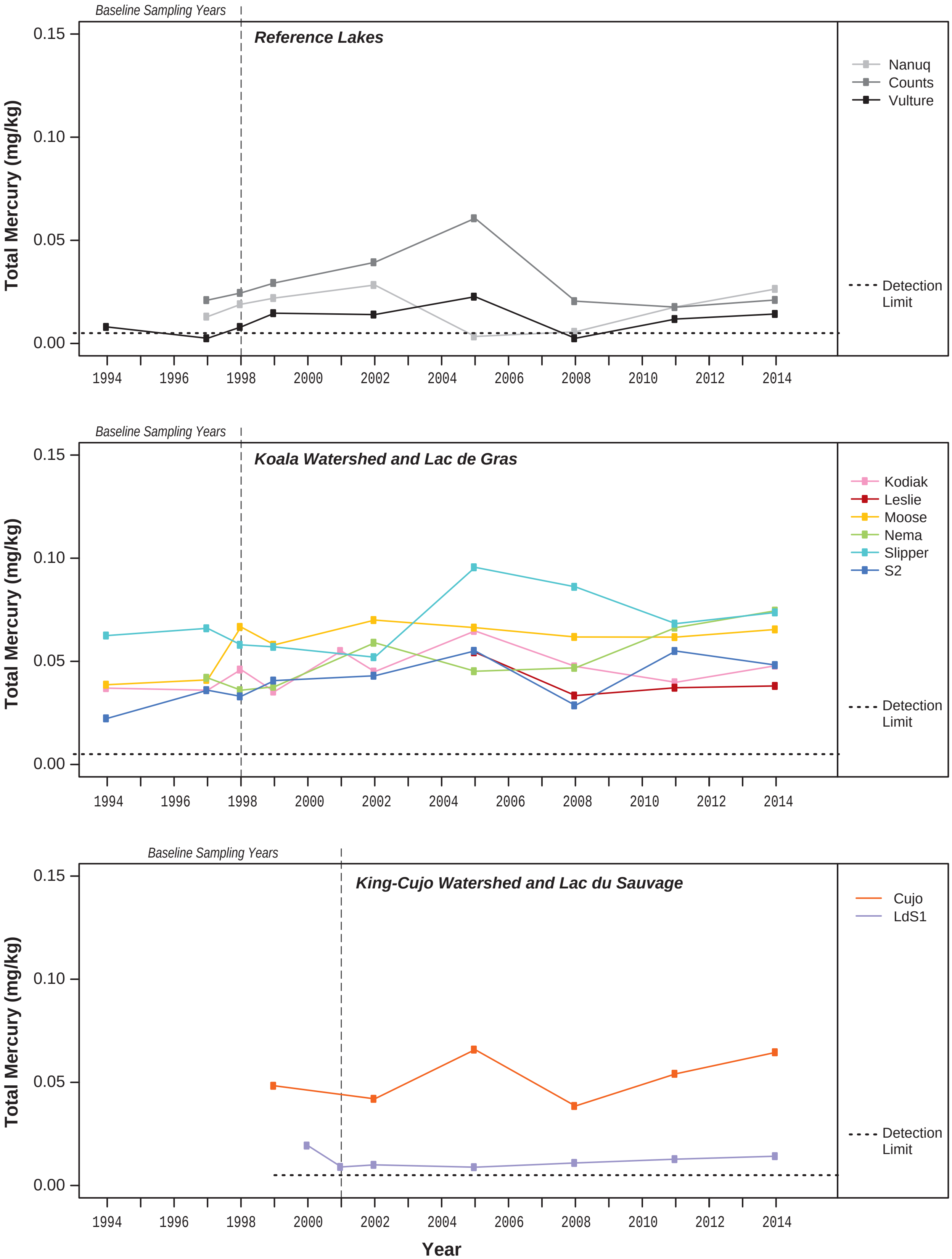
Total Manganese in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-20

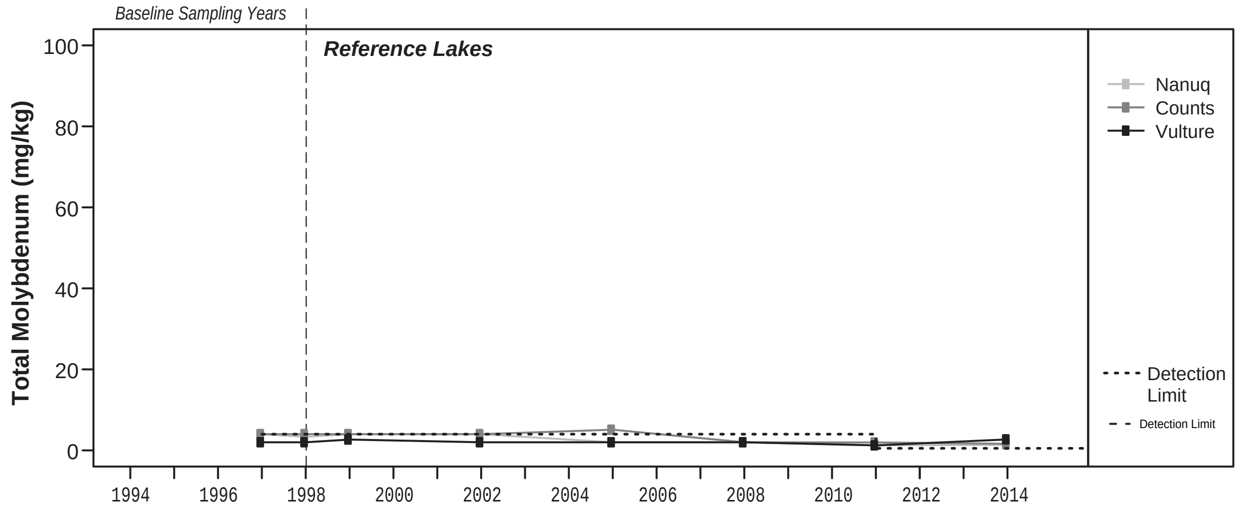
Total Mercury in AEMP Lake Sediments, 1994 to 2014

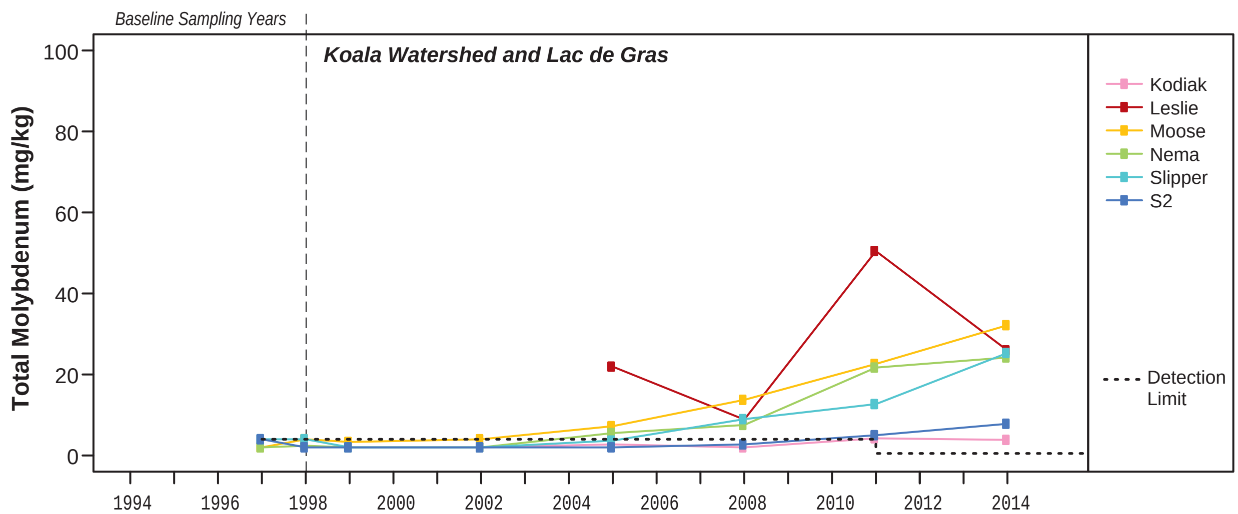


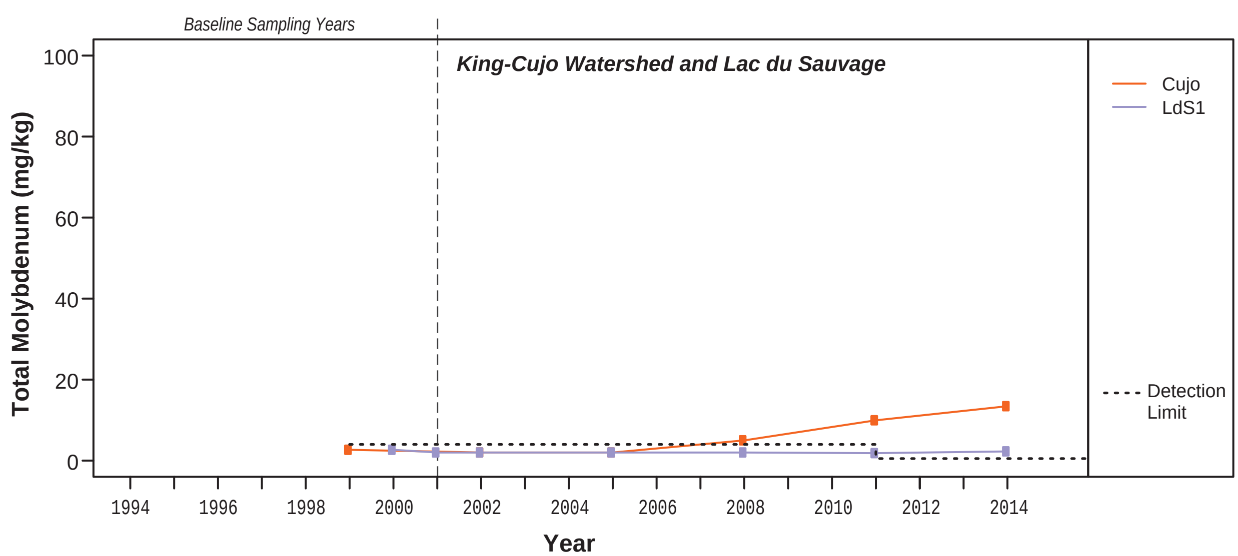
Notes: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.
CCME guidelines: ISQG = 0.17 mg/kg (not shown); PEL = 0.486 mg/kg (not shown).

Figure 7-21

Total Molybdenum in AEMP Lake Sediments, 1994 to 2014



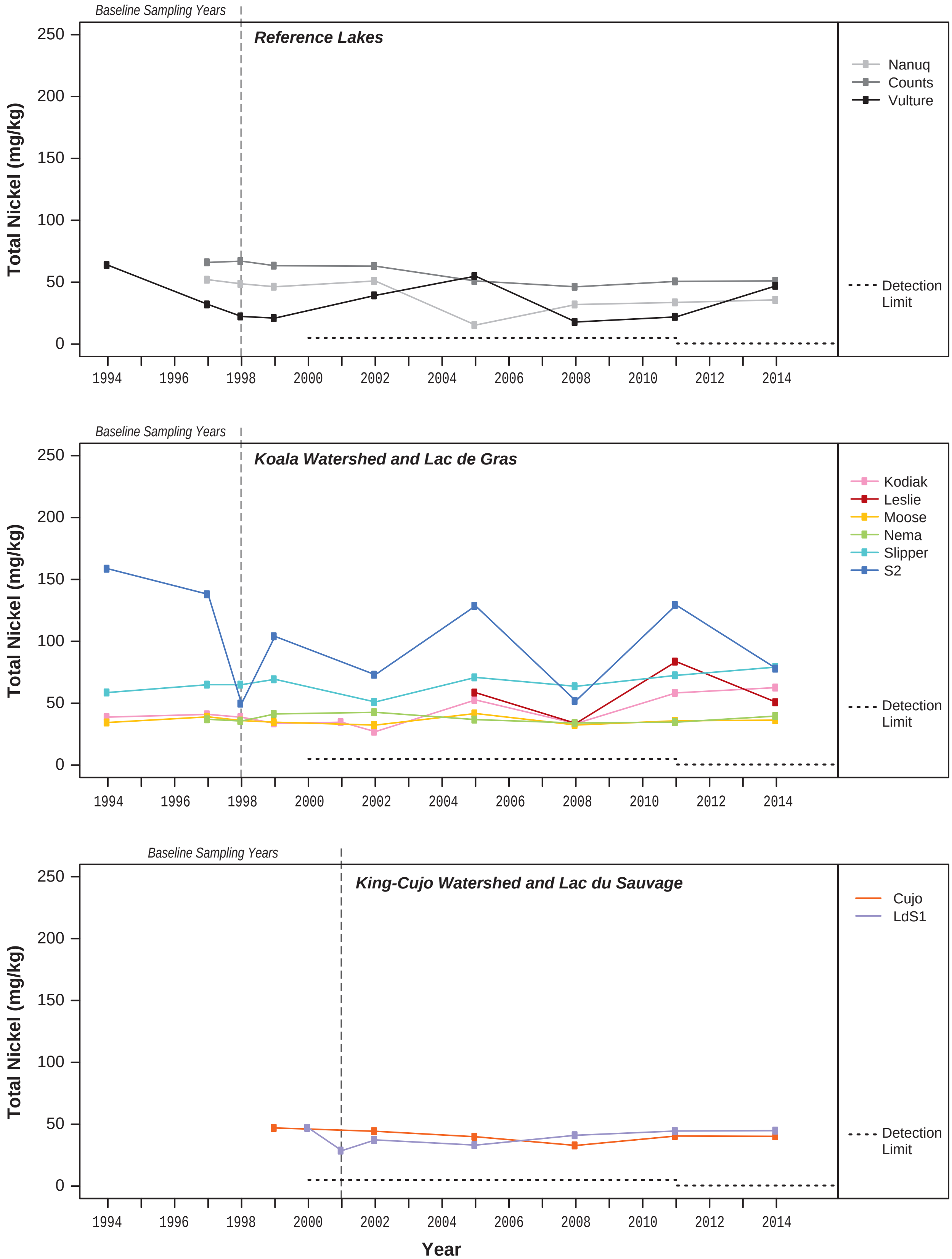




Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-22

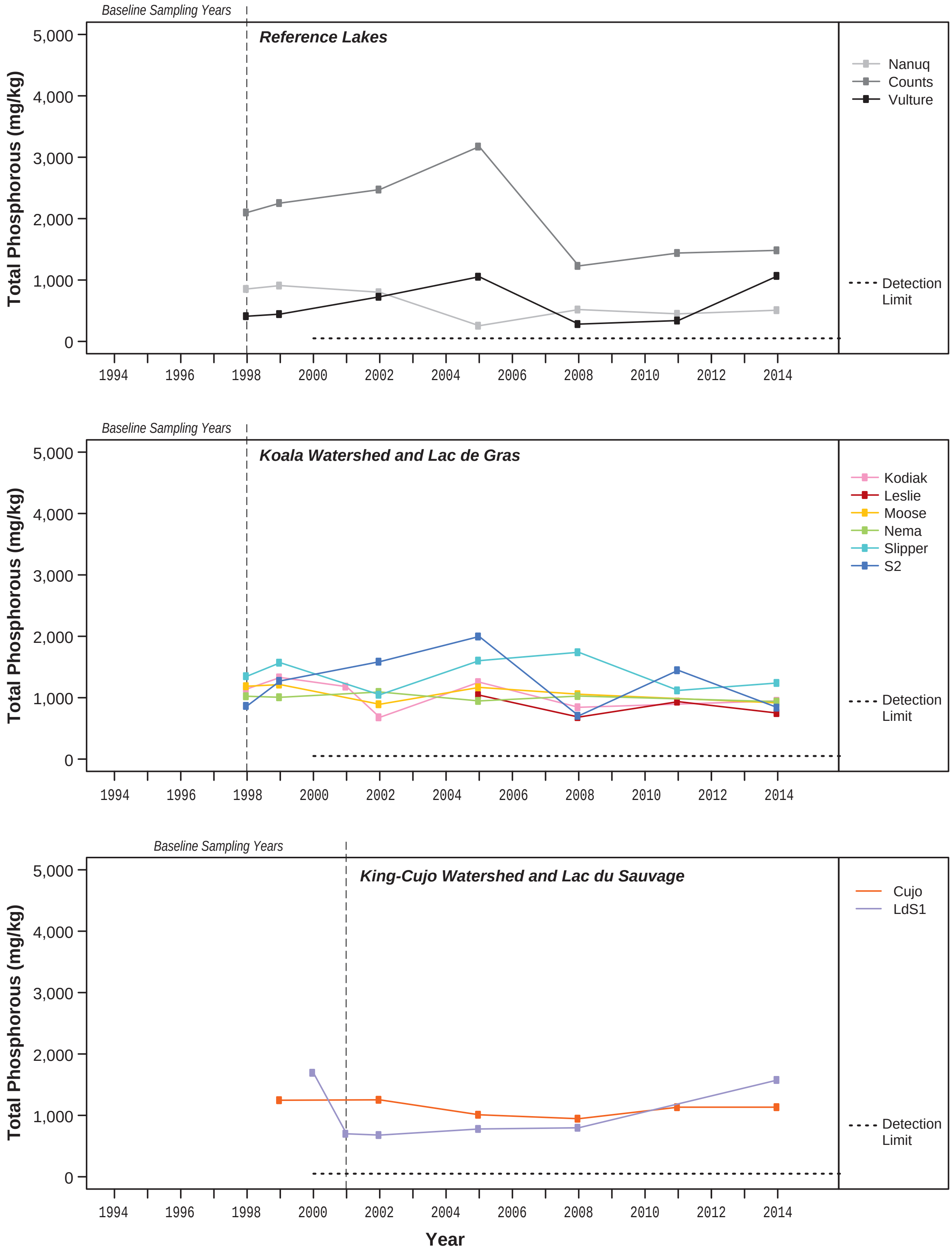
Total Nickel
in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-23

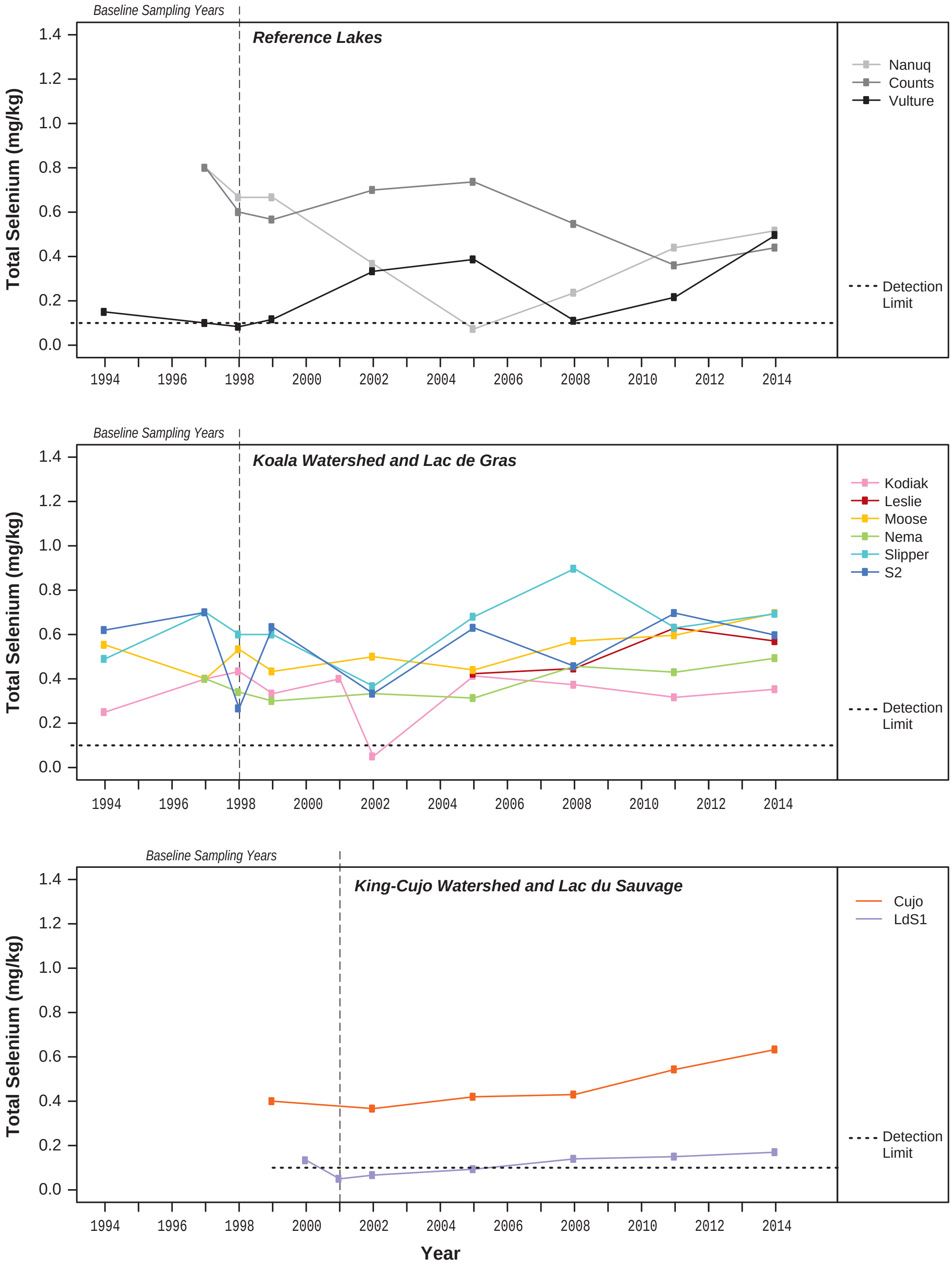
Total Phosphorus in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-24

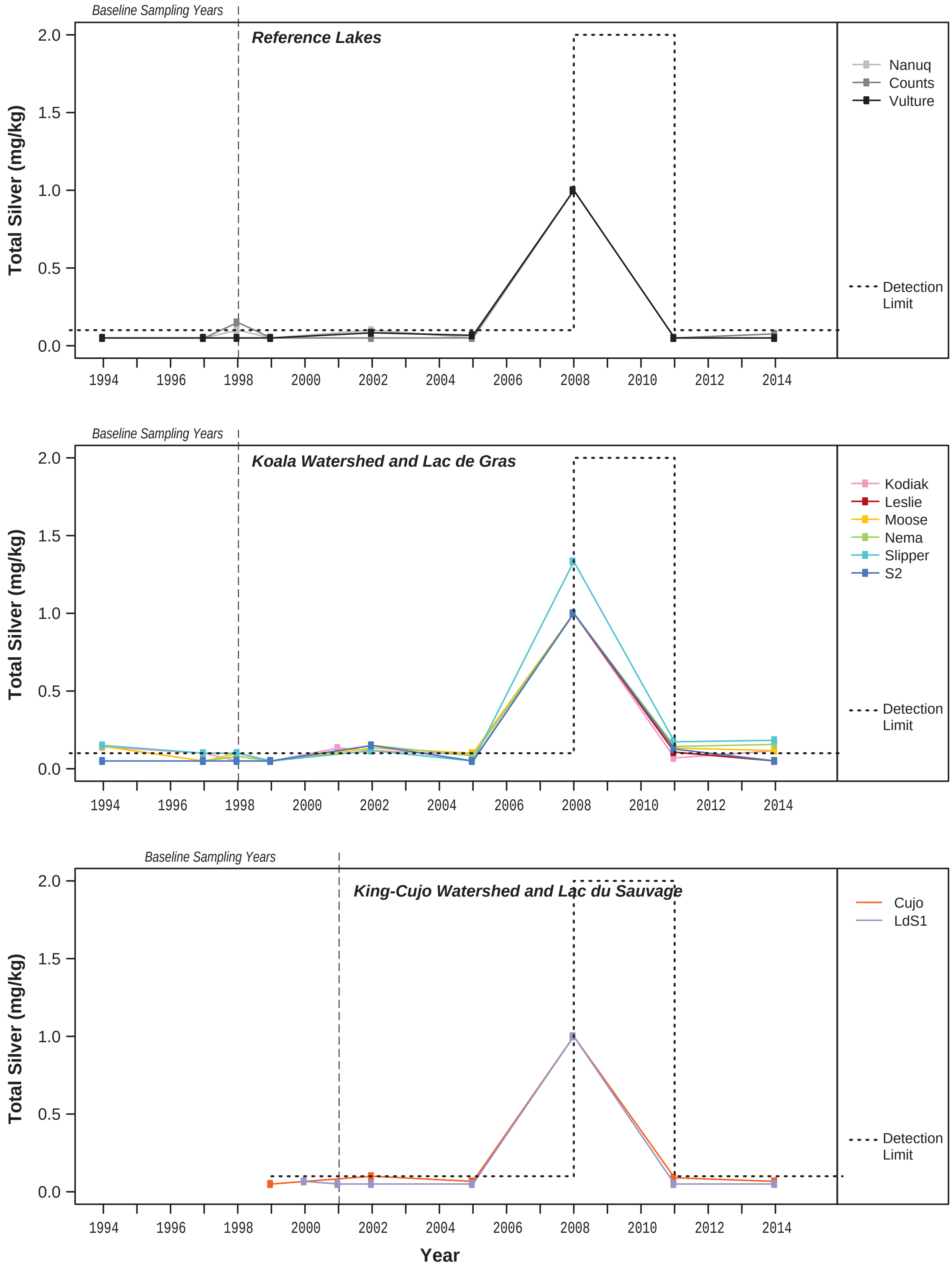
Total Selenium in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-25

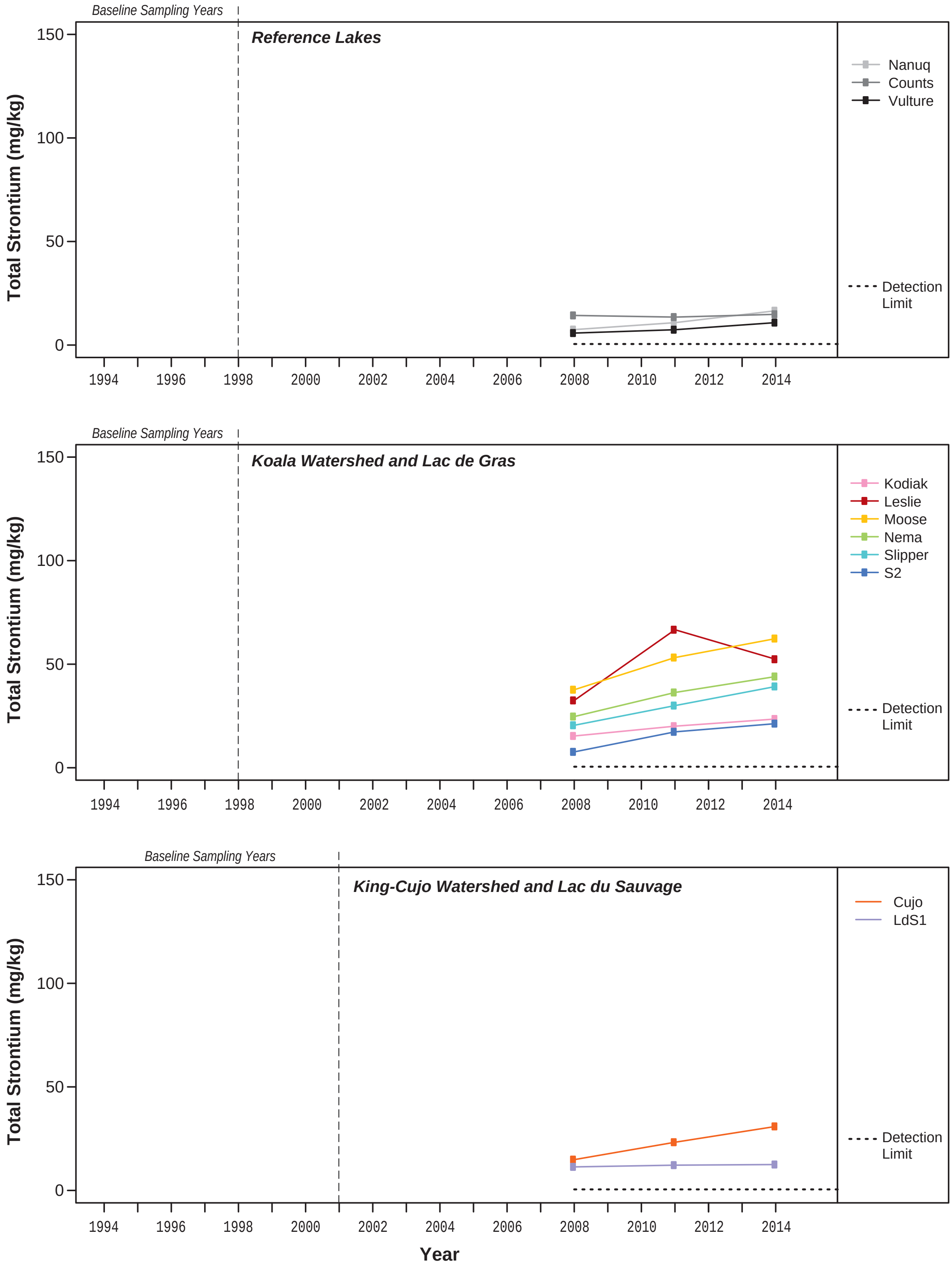
Total Silver
in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-26

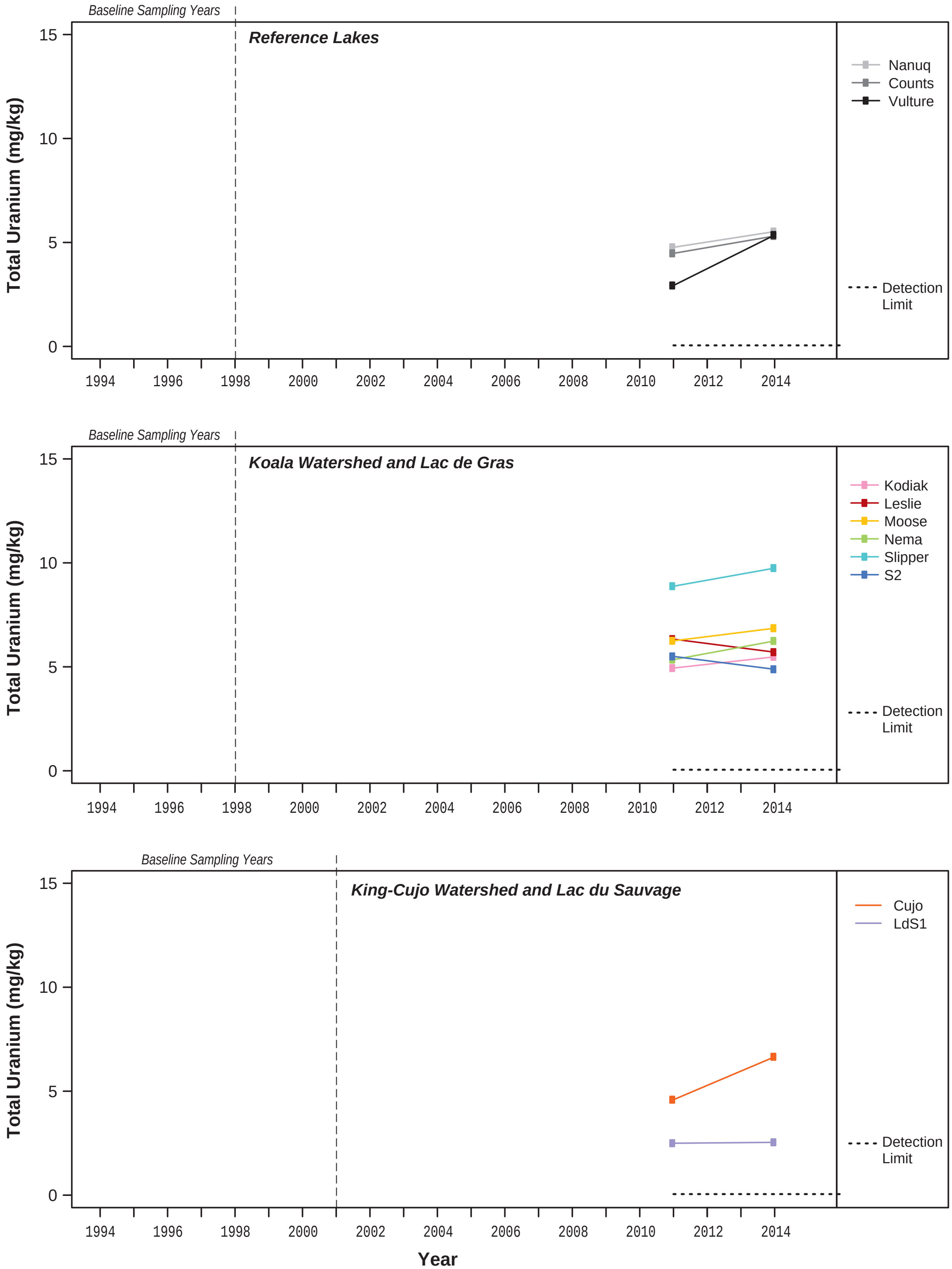
Total Strontium in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-27

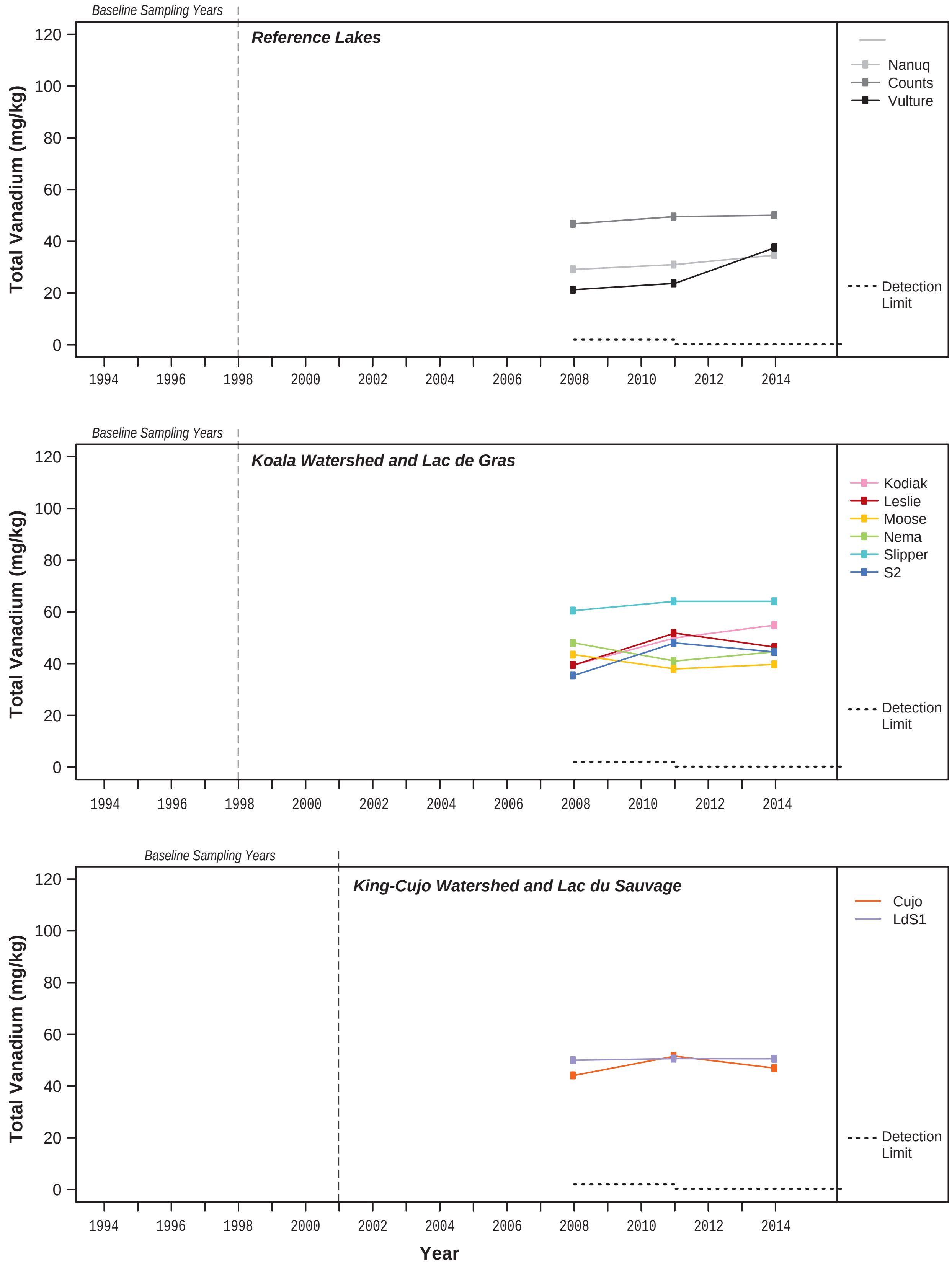
Total Uranium in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-28

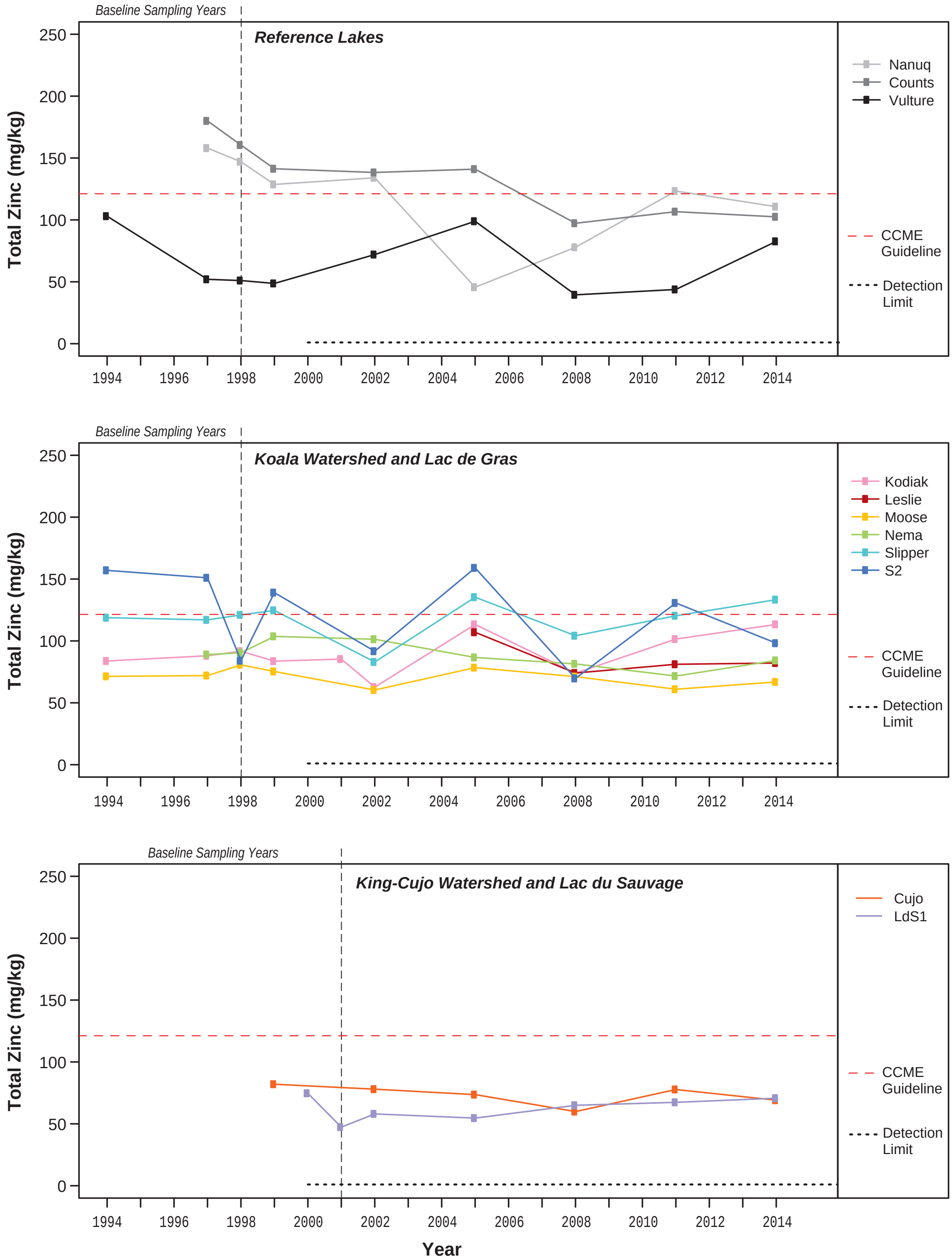
Total Vanadium in AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.

Figure 7-29

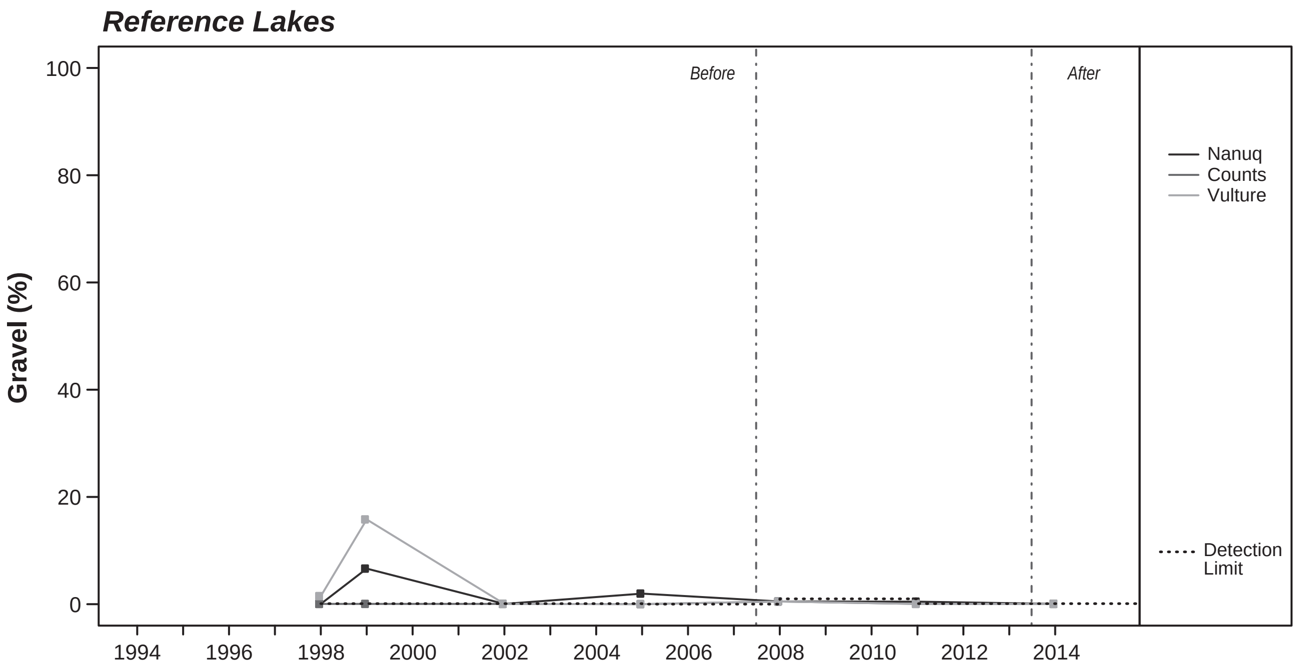
Total Zinc in AEMP Lake Sediments, 1994 to 2014

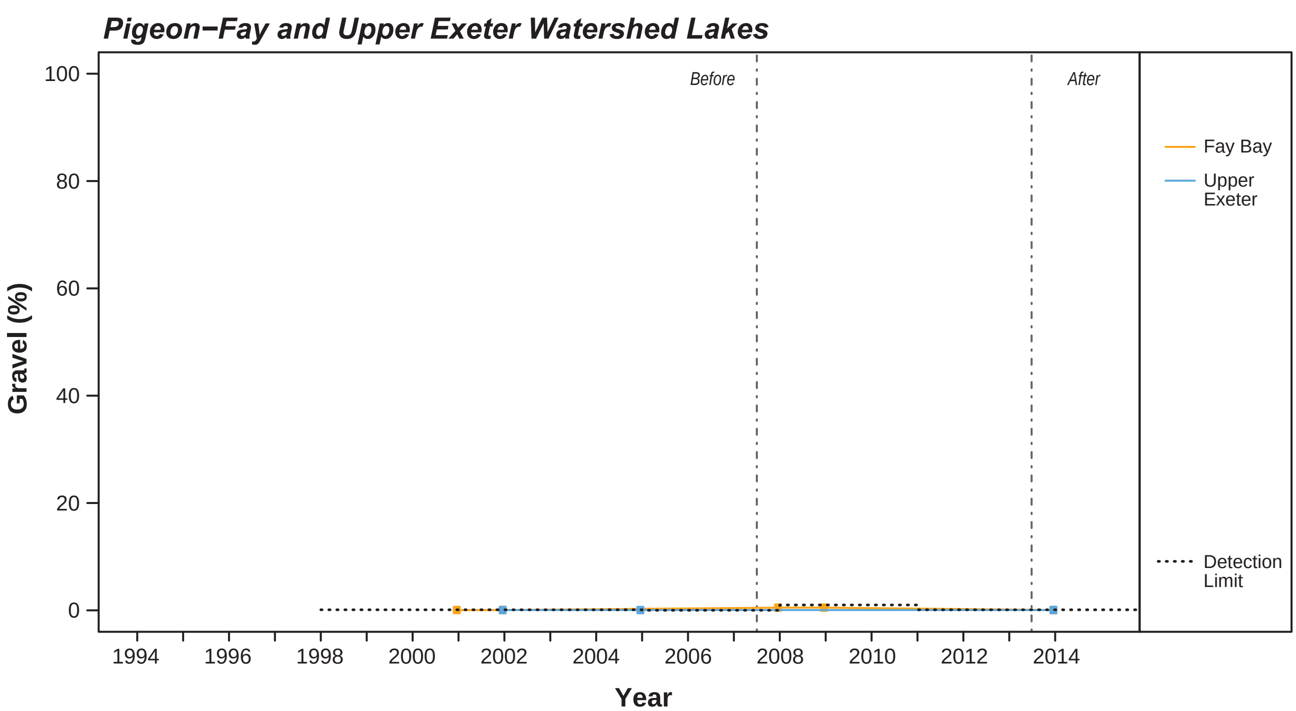


Notes: For cases where detection limits varied between lakes and month, the lowest detection limit is shown.
CCME guidelines: ISQG = 123 mg/kg; PEL = 315 mg/kg (not shown).

Figure 7-30

Percent Gravel in Pigeon AEMP Lake Sediments, 1994 to 2014

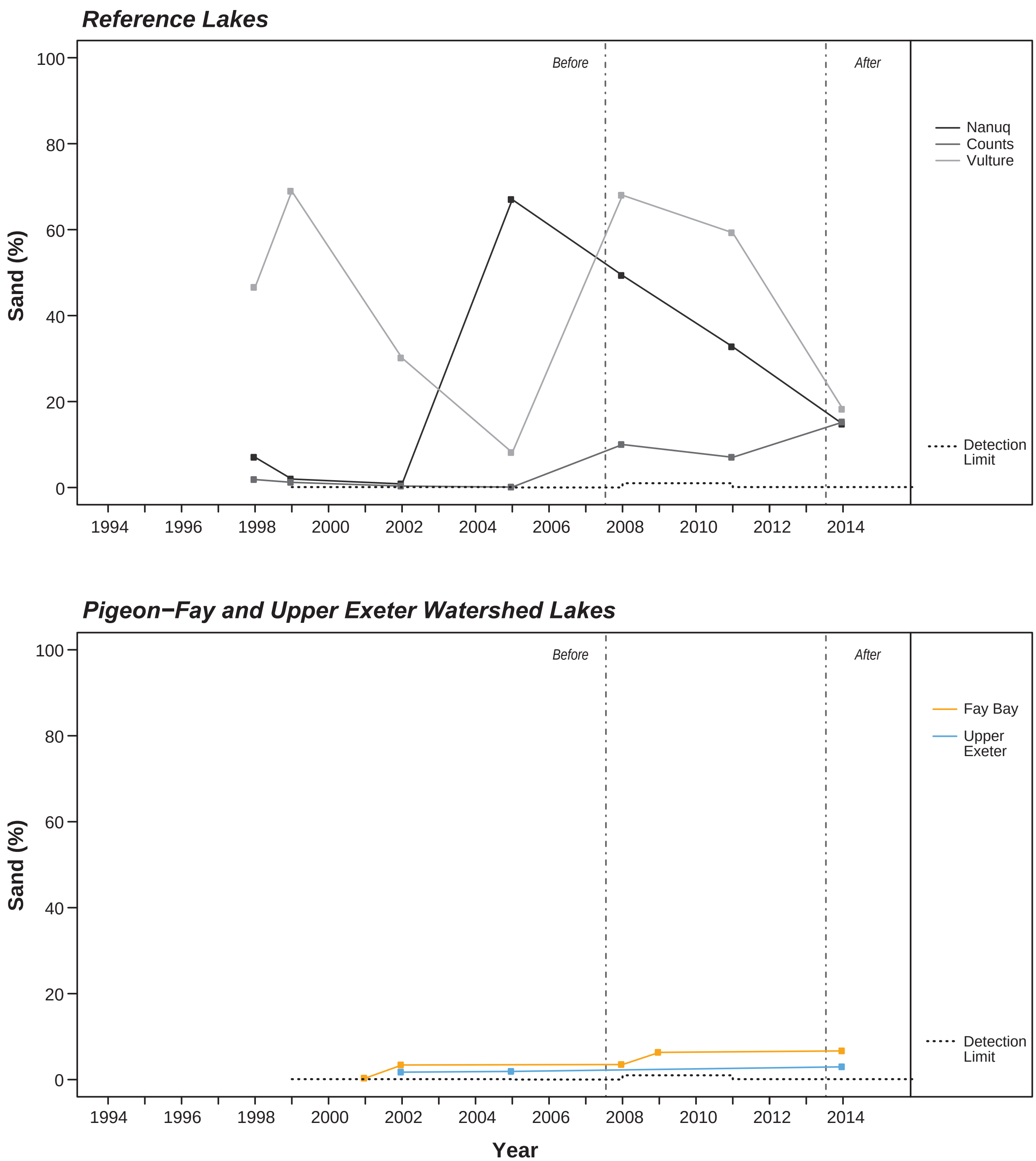




Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 7-31

Percent Sand in Pigeon AEMP Lake Sediments, 1994 to 2014

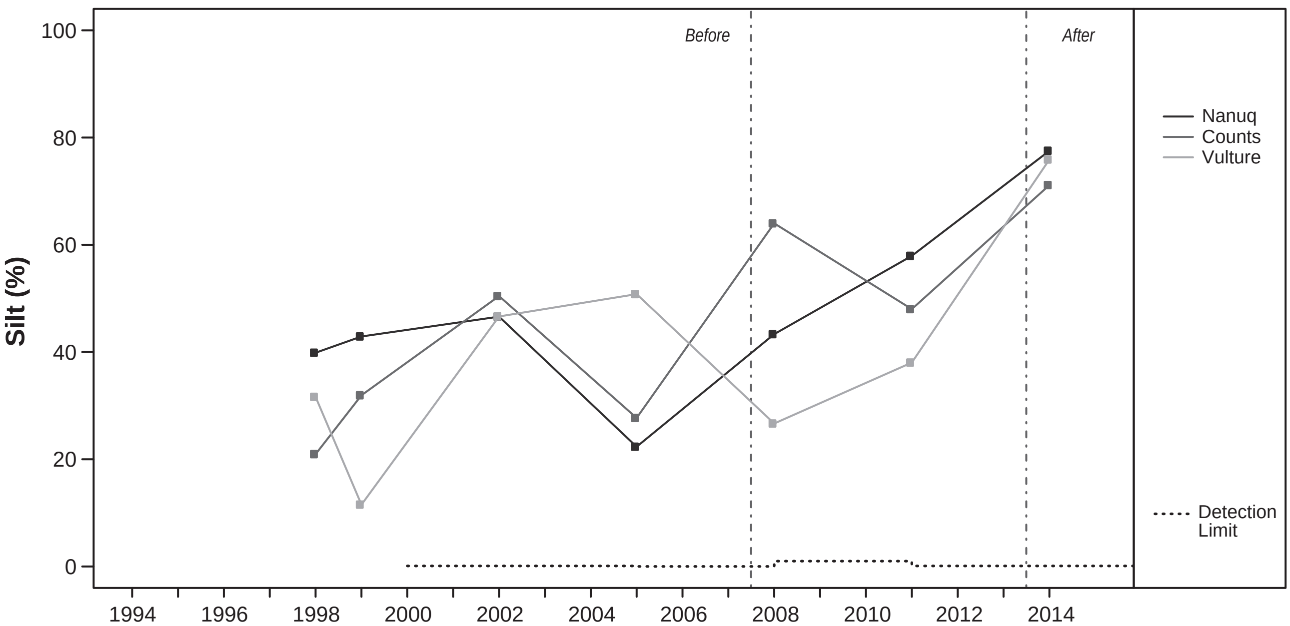


Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

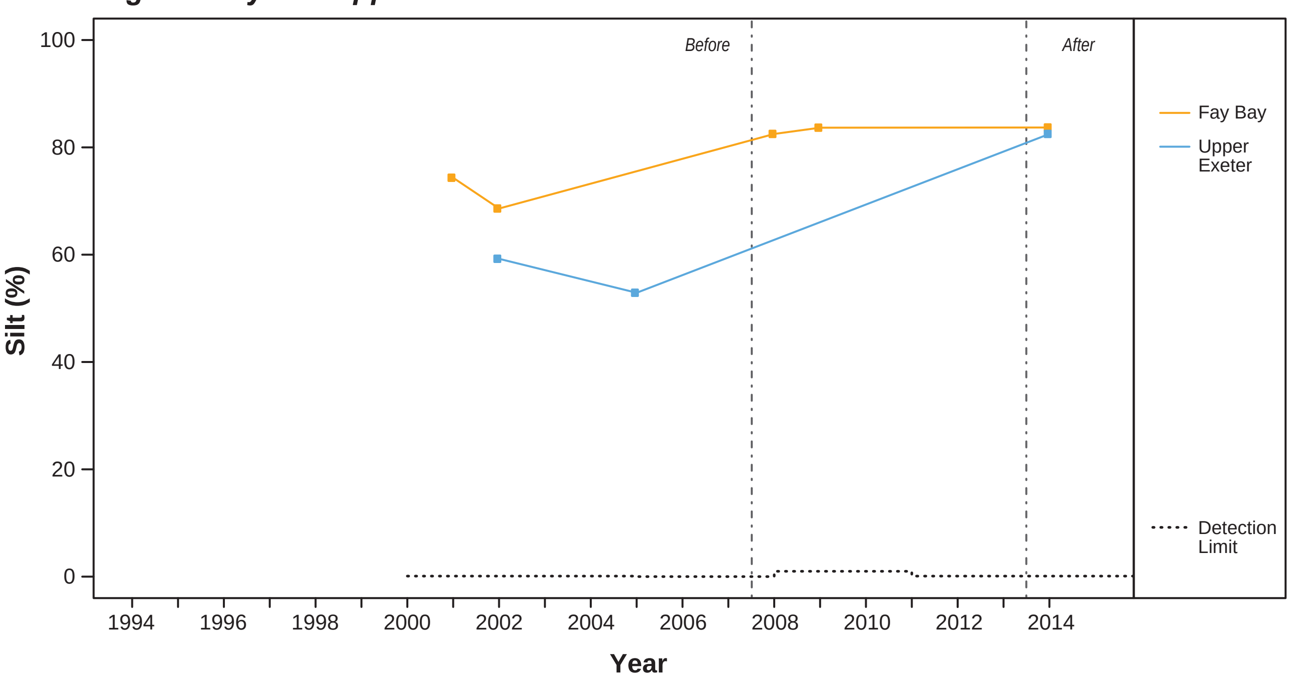
Figure 7-32

Percent Silt in Pigeon AEMP Lake Sediments, 1994 to 2014

Reference Lakes



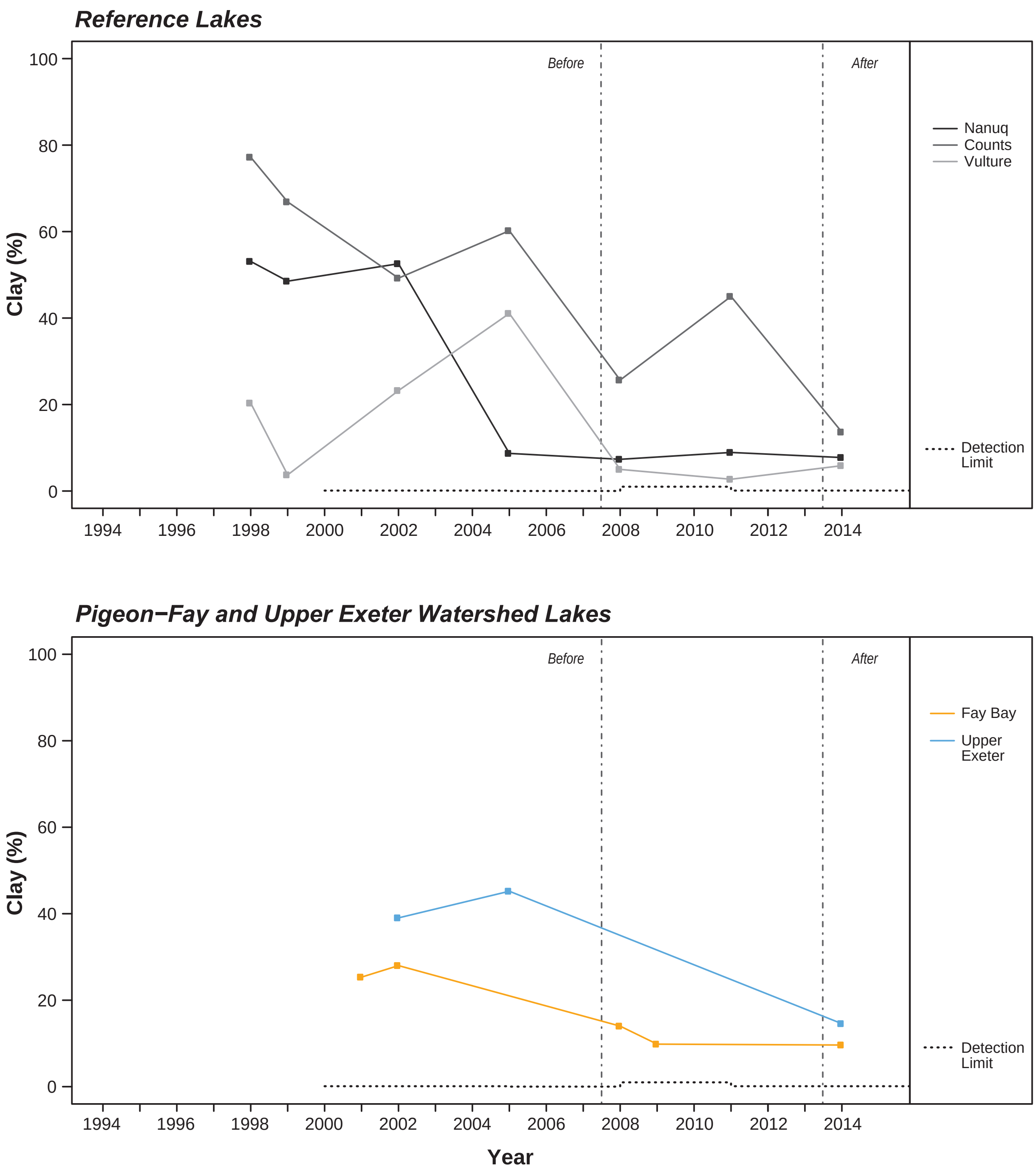
Pigeon−Fay and Upper Exeter Watershed Lakes



Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

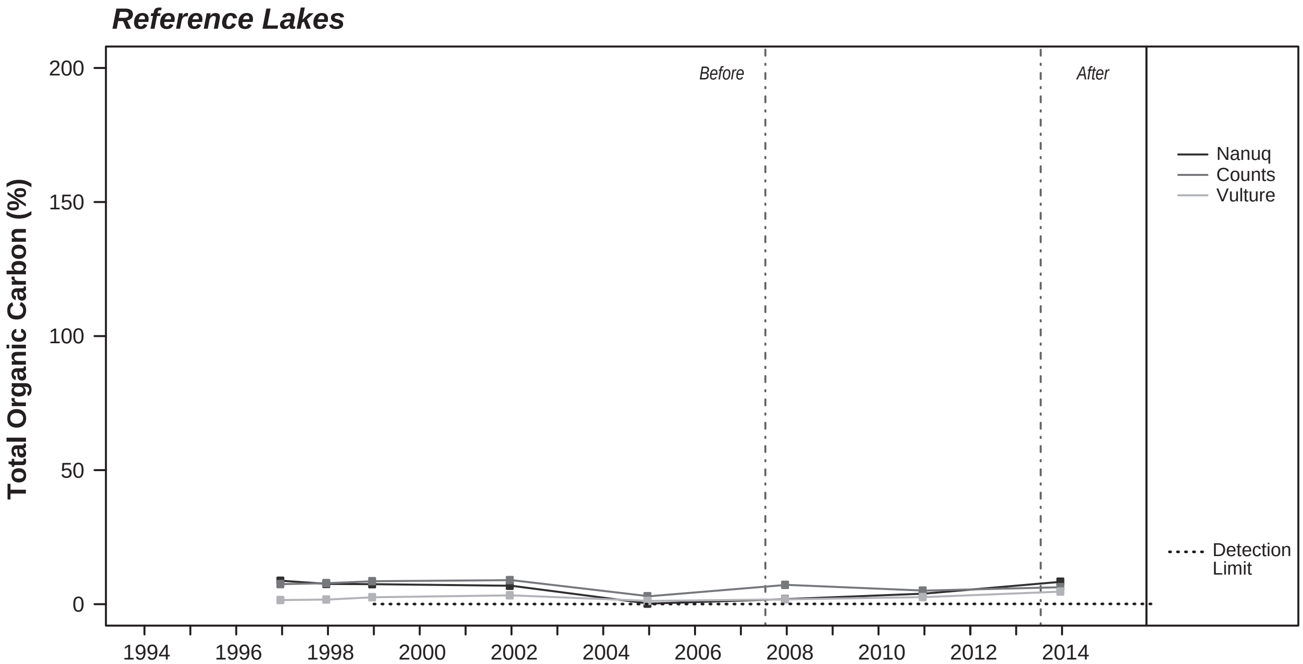
Figure 7-33

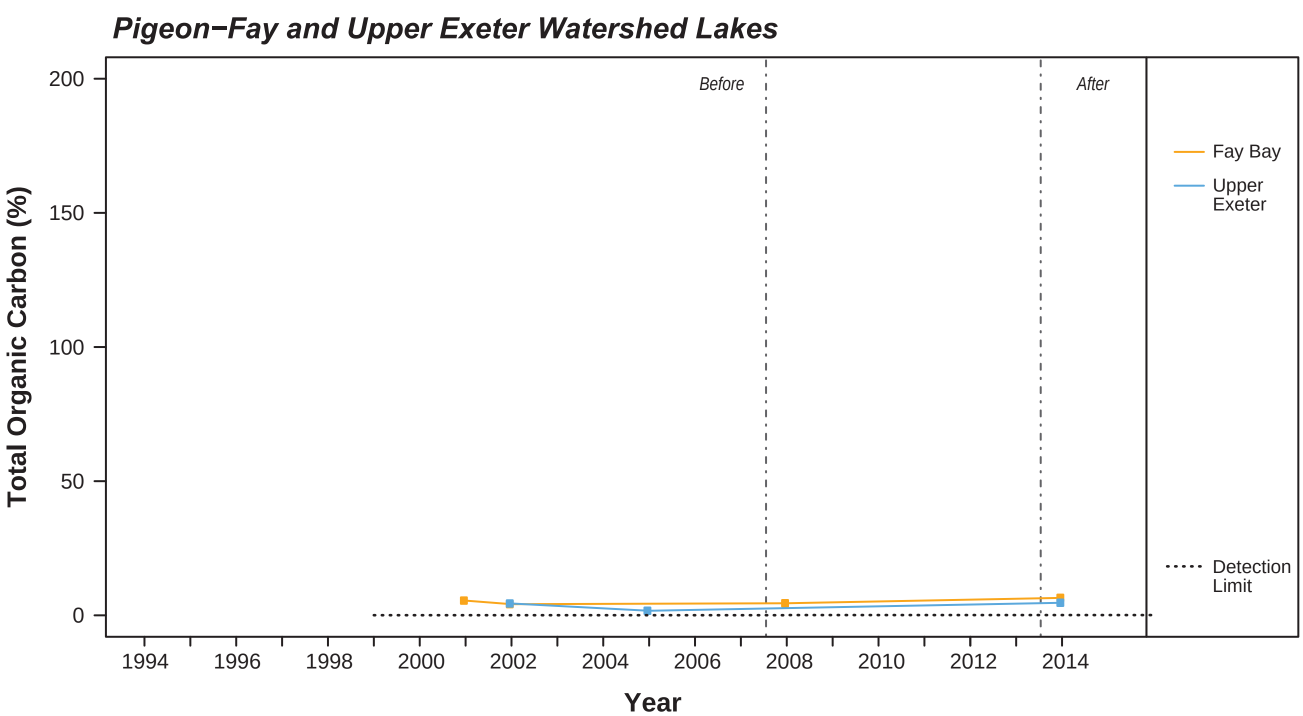
Percent Clay in Pigeon AEMP Lake Sediments, 1994 to 2014



Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Total Organic Carbon in Pigeon AEMP Lake Sediments, 1994 to 2014

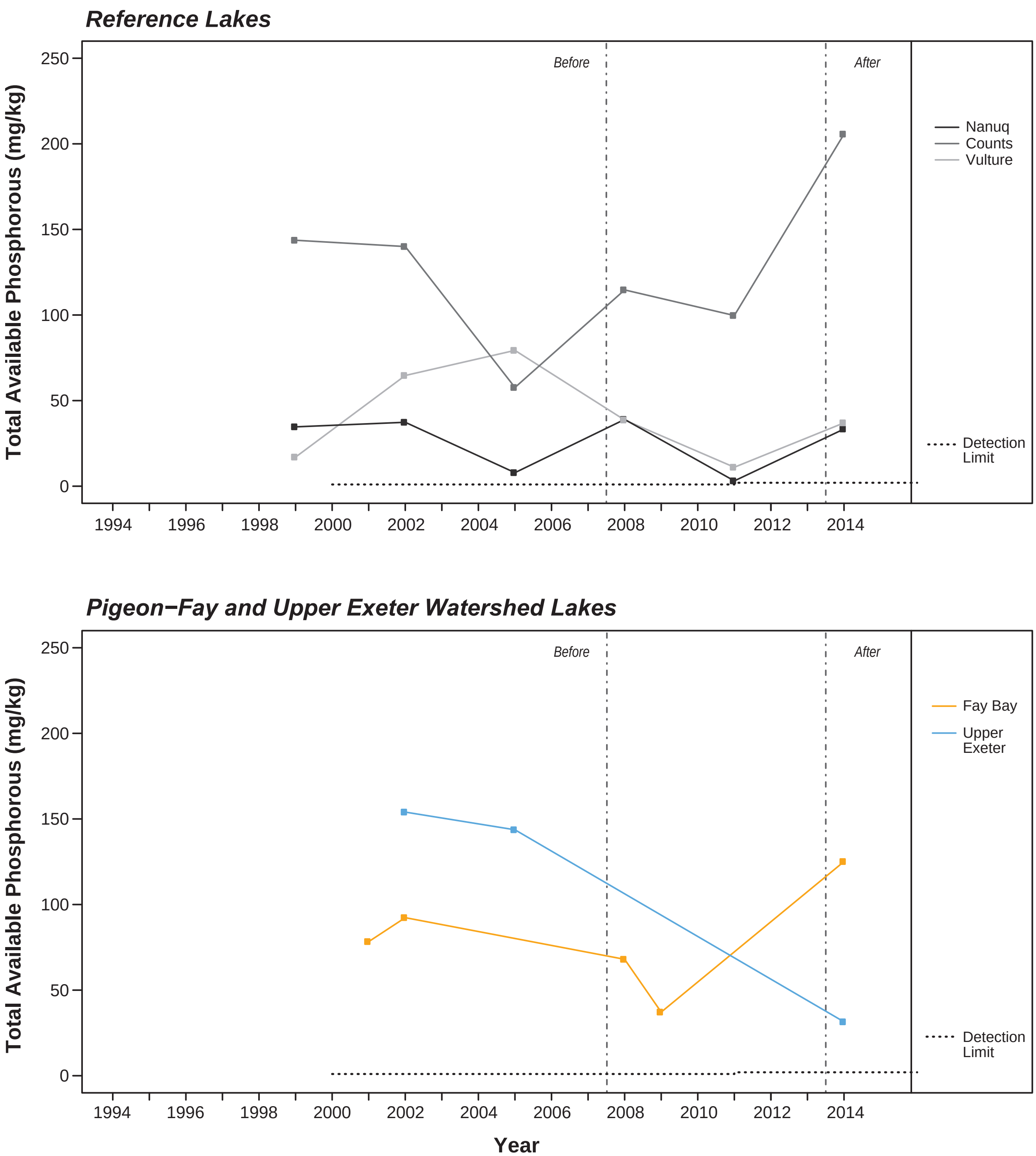




Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 7-35

Available Phosphorus in Pigeon AEMP Lake Sediments, 1994 to 2014

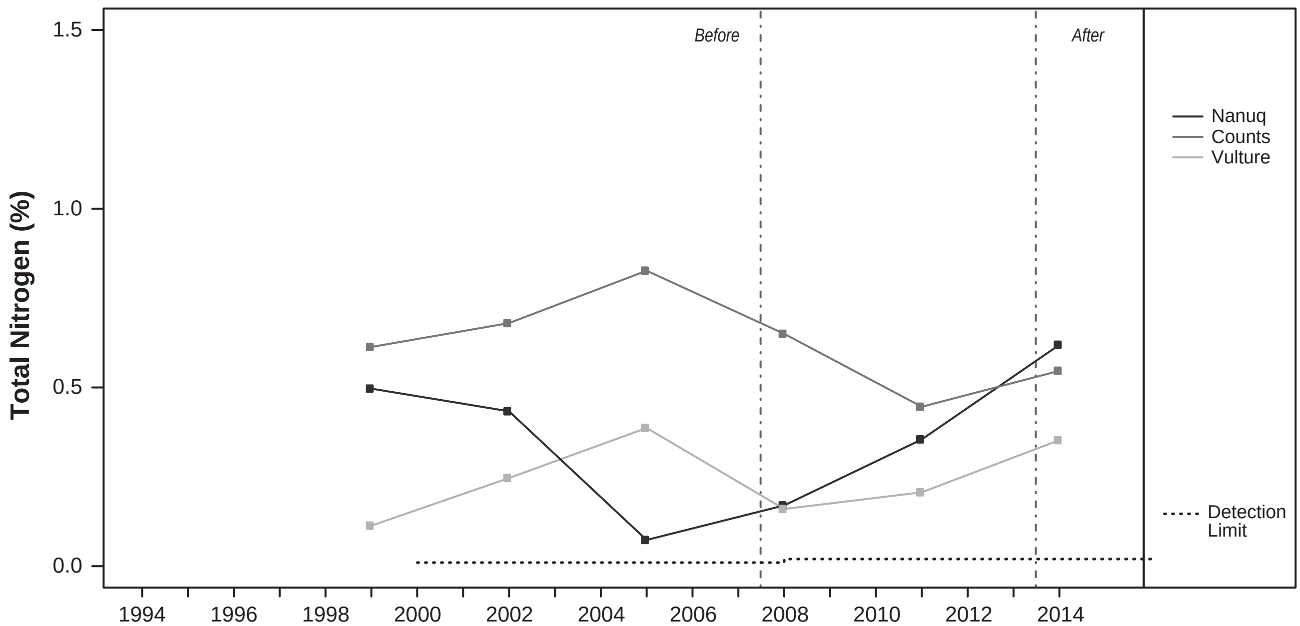


Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

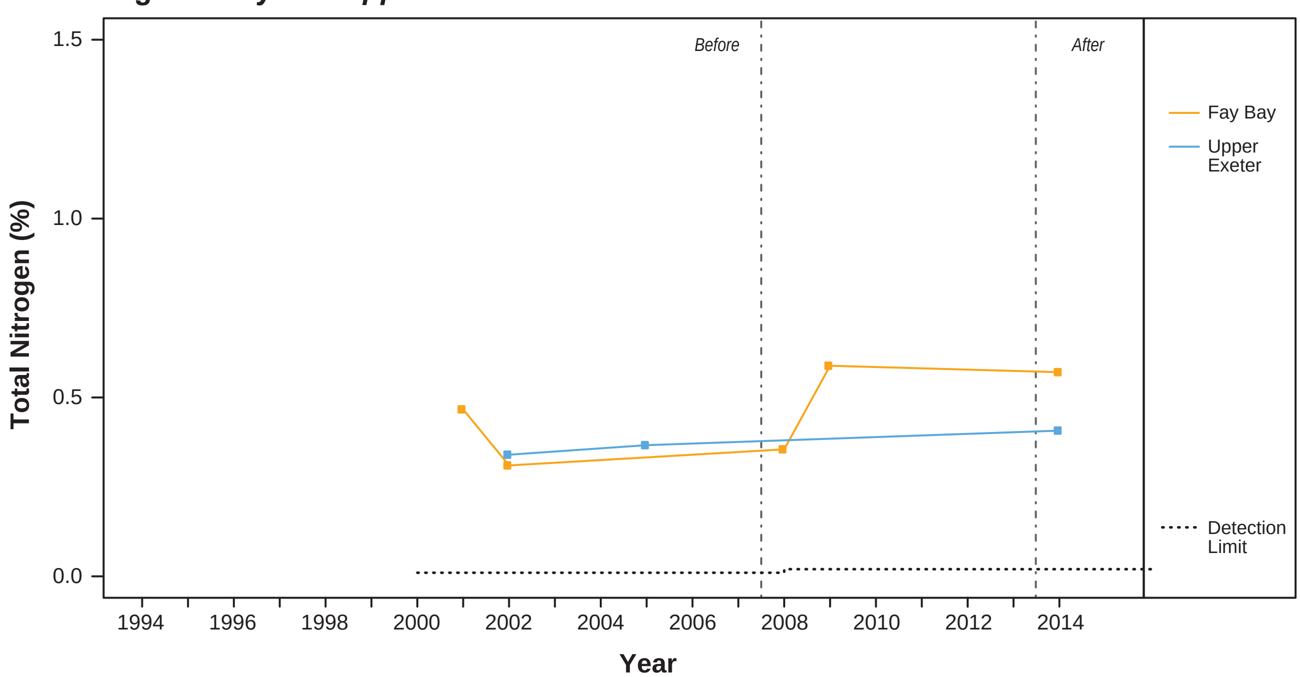
Figure 7-36

Total Nitrogen in Pigeon AEMP Lake Sediments, 1994 to 2014

Reference Lakes

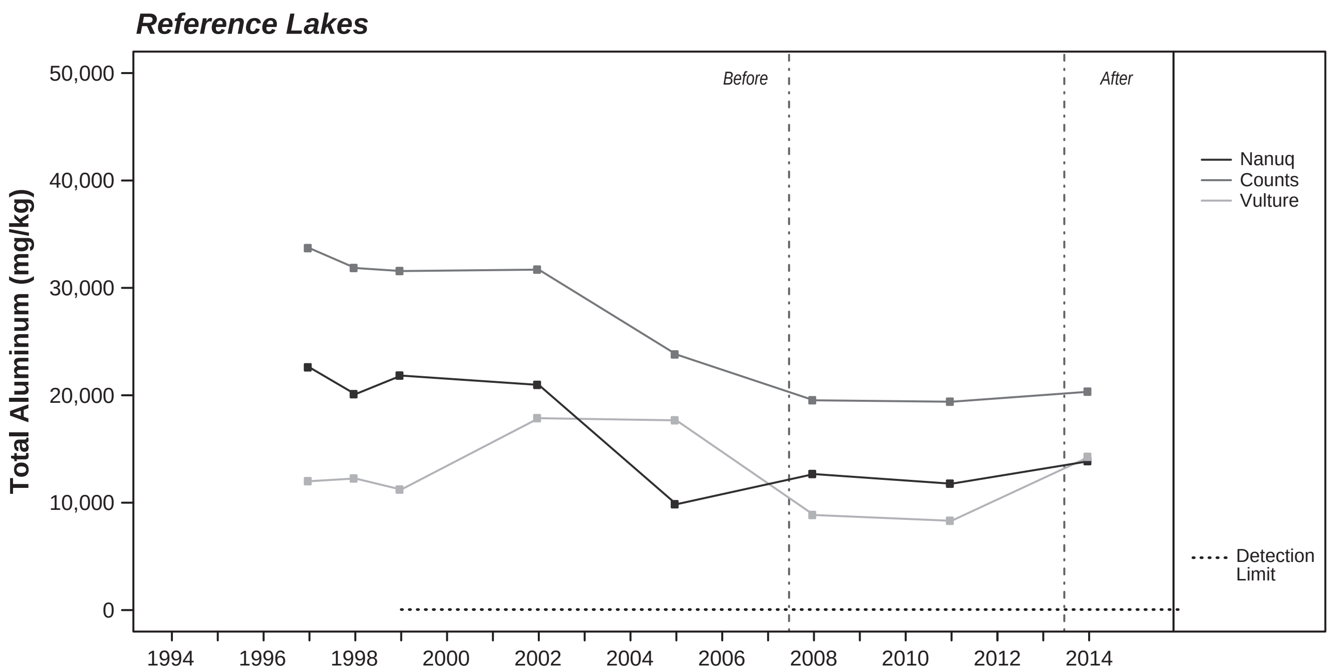


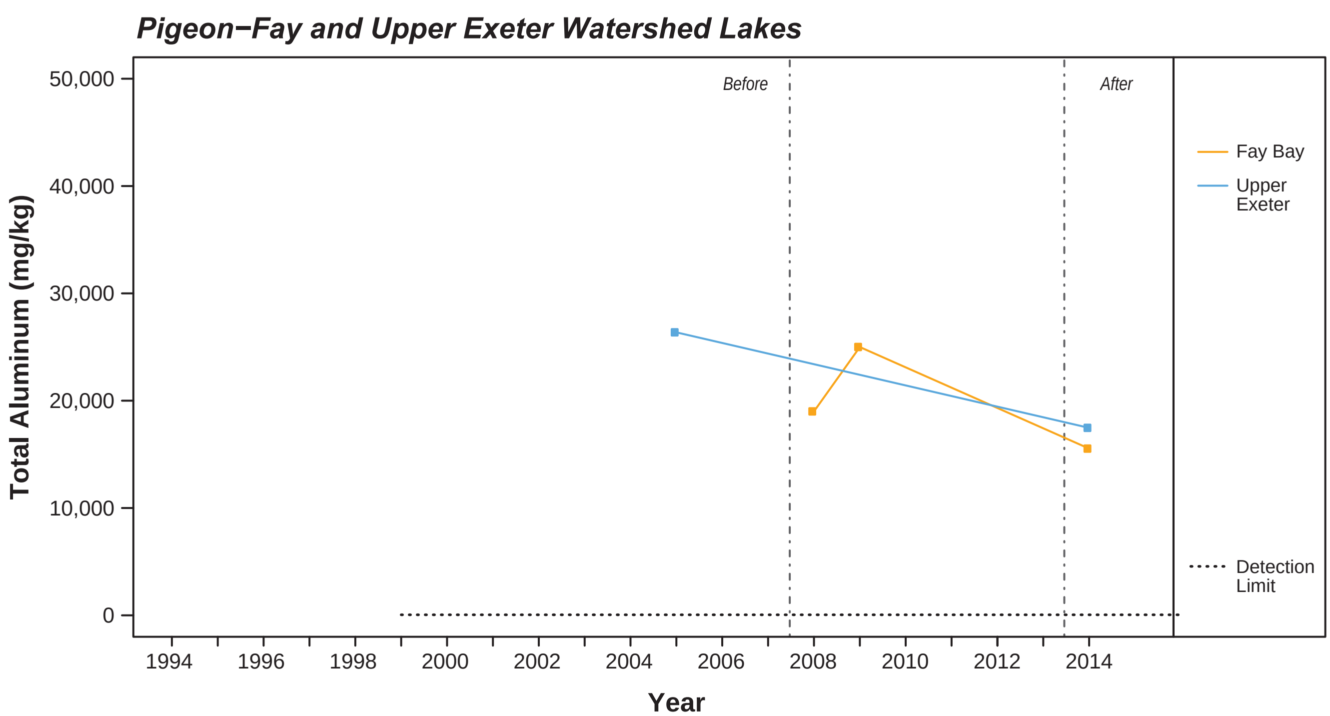
Pigeon−Fay and Upper Exeter Watershed Lakes



Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Total Aluminum in Pigeon AEMP Lake Sediments, 1994 to 2014



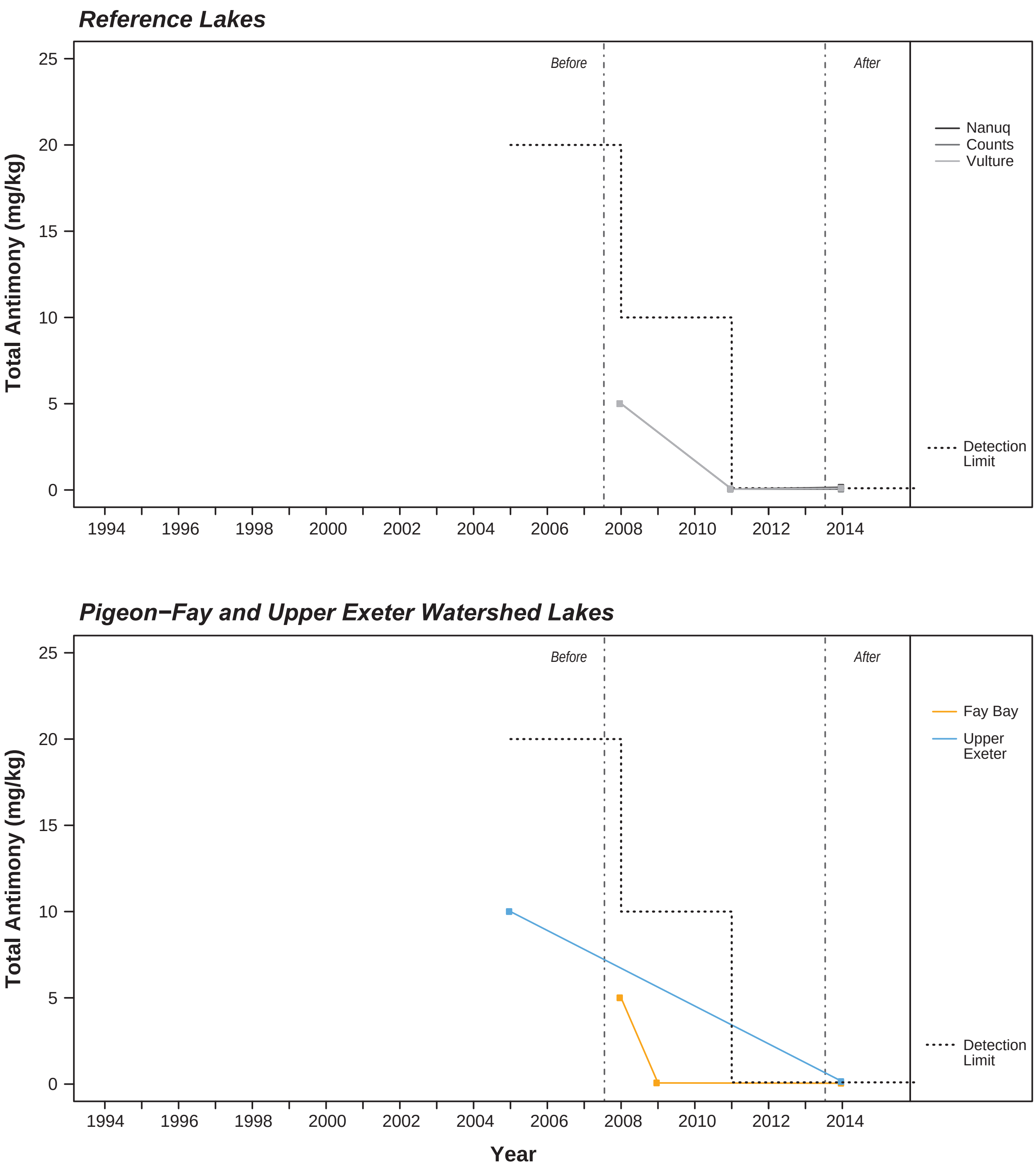


Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 7-38

Total Antimony in Pigeon AEMP Lake Sediments, 1994 to 2014

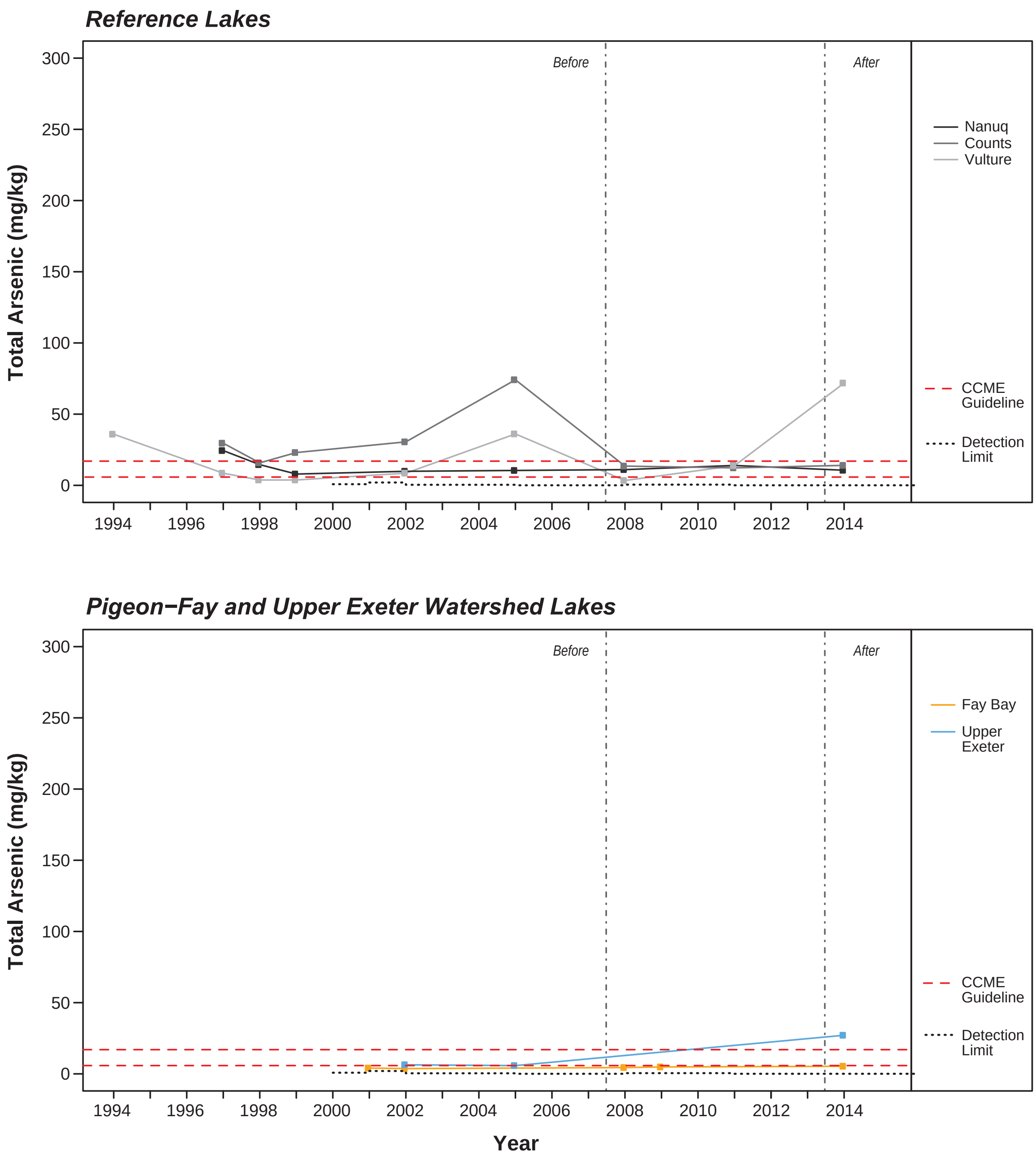
ERM



Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

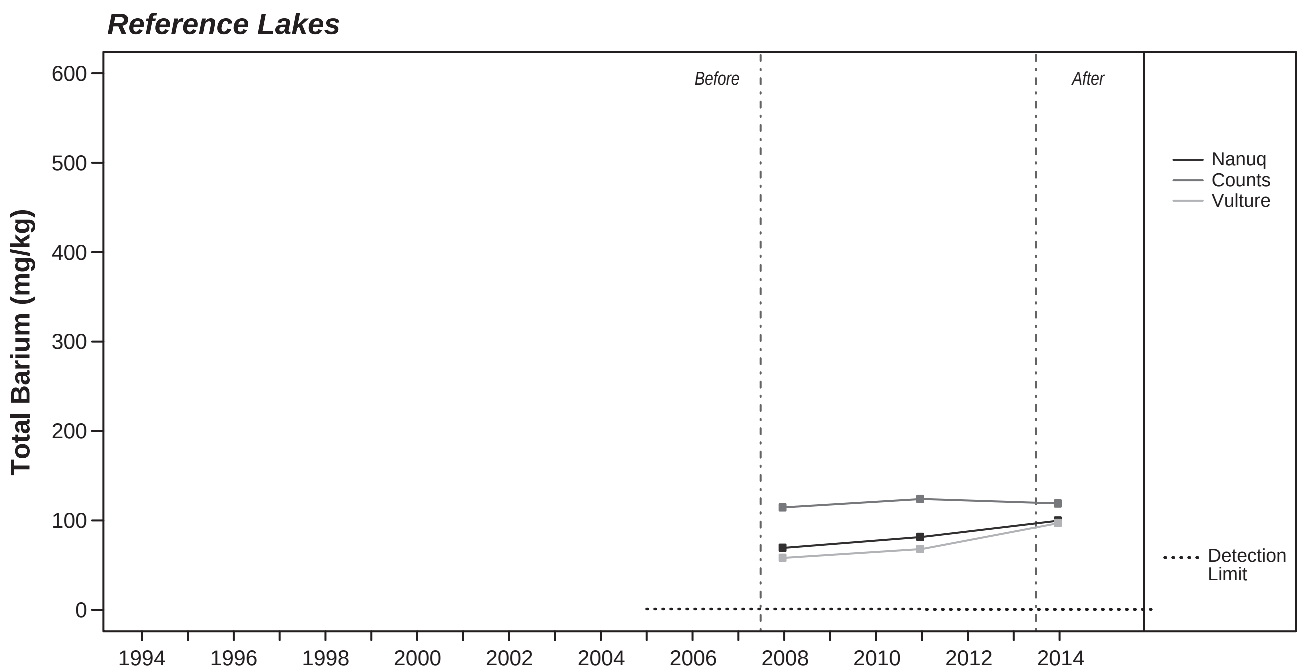
Figure 7-39

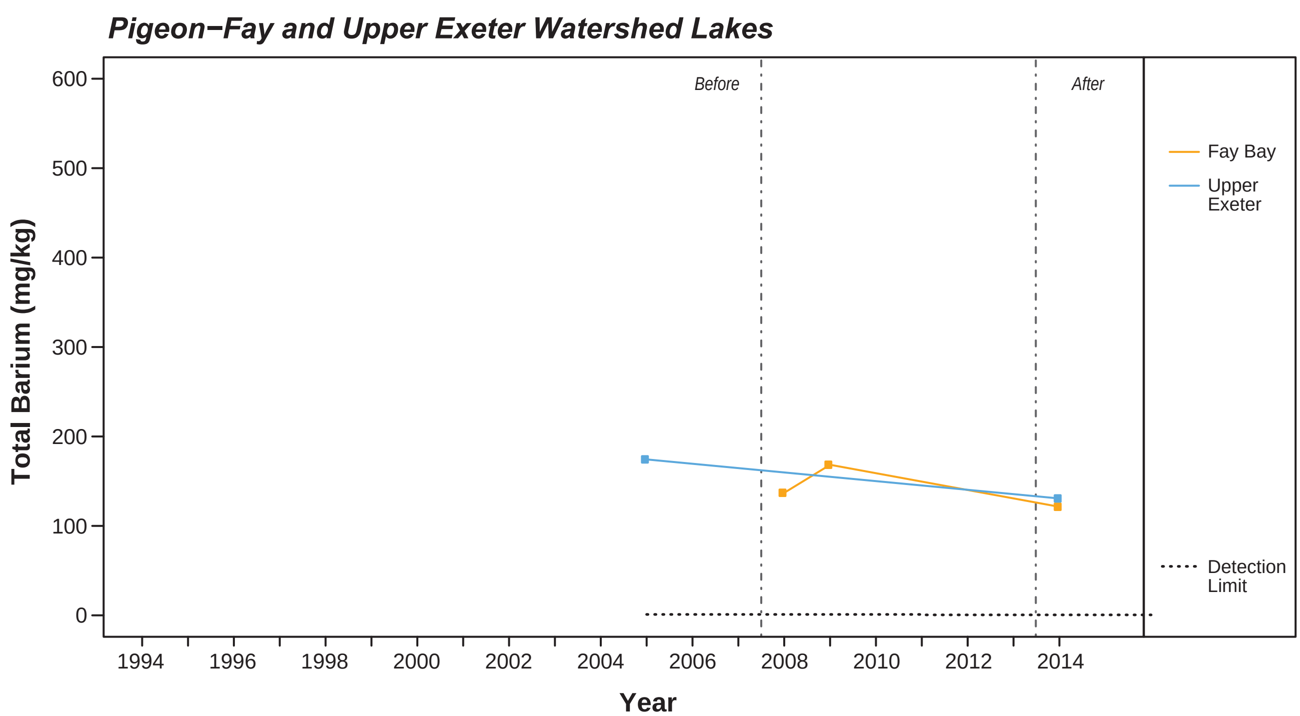
Total Arsenic in Pigeon AEMP Lake Sediments, 1994 to 2014



Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME guidelines: ISQG = 5.9 mg/kg; PEL = 17 mg/kg.

Figure 7-40
Total Barium in Pigeon AEMP Lake Sediments, 1994 to 2014

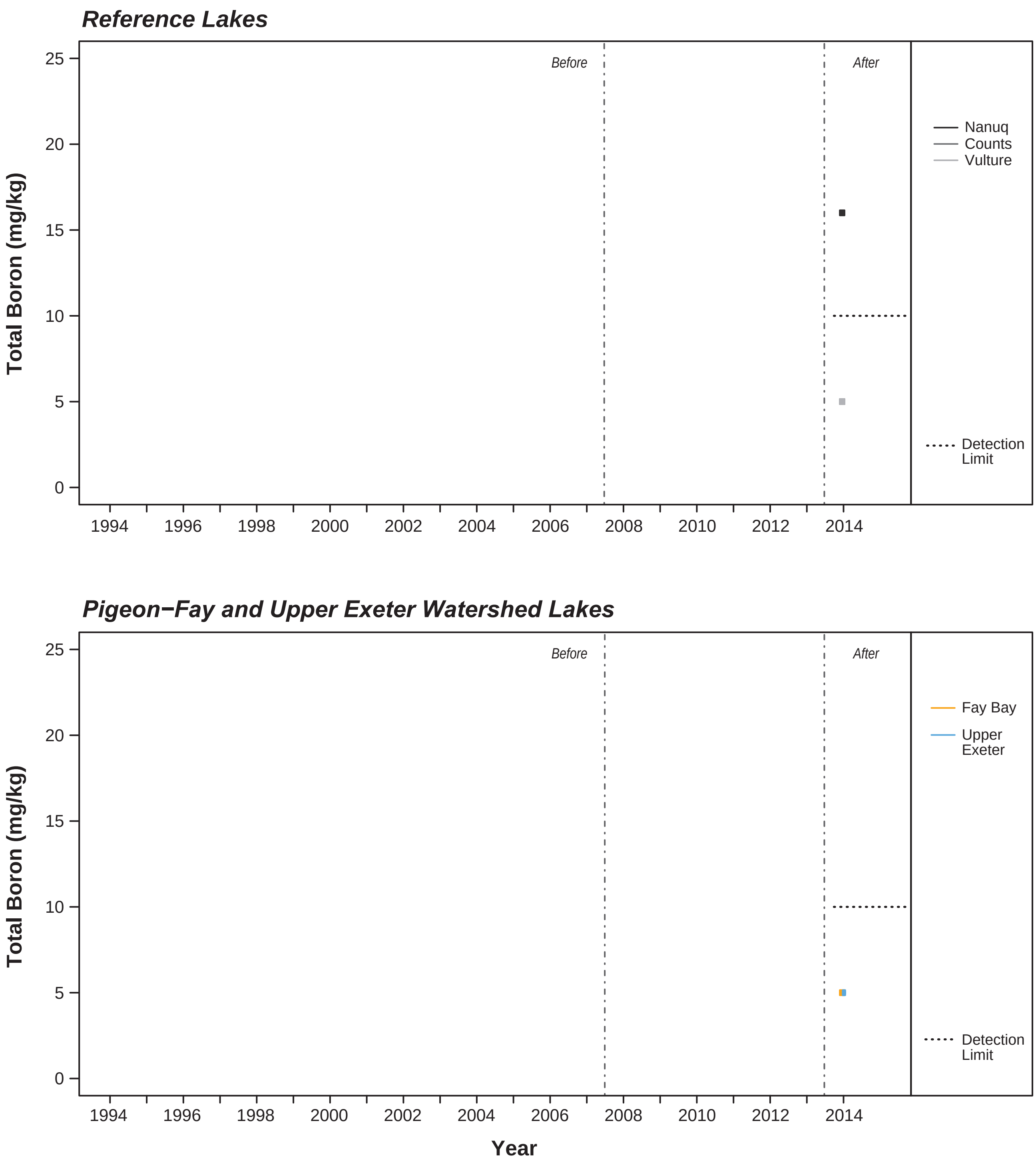




Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

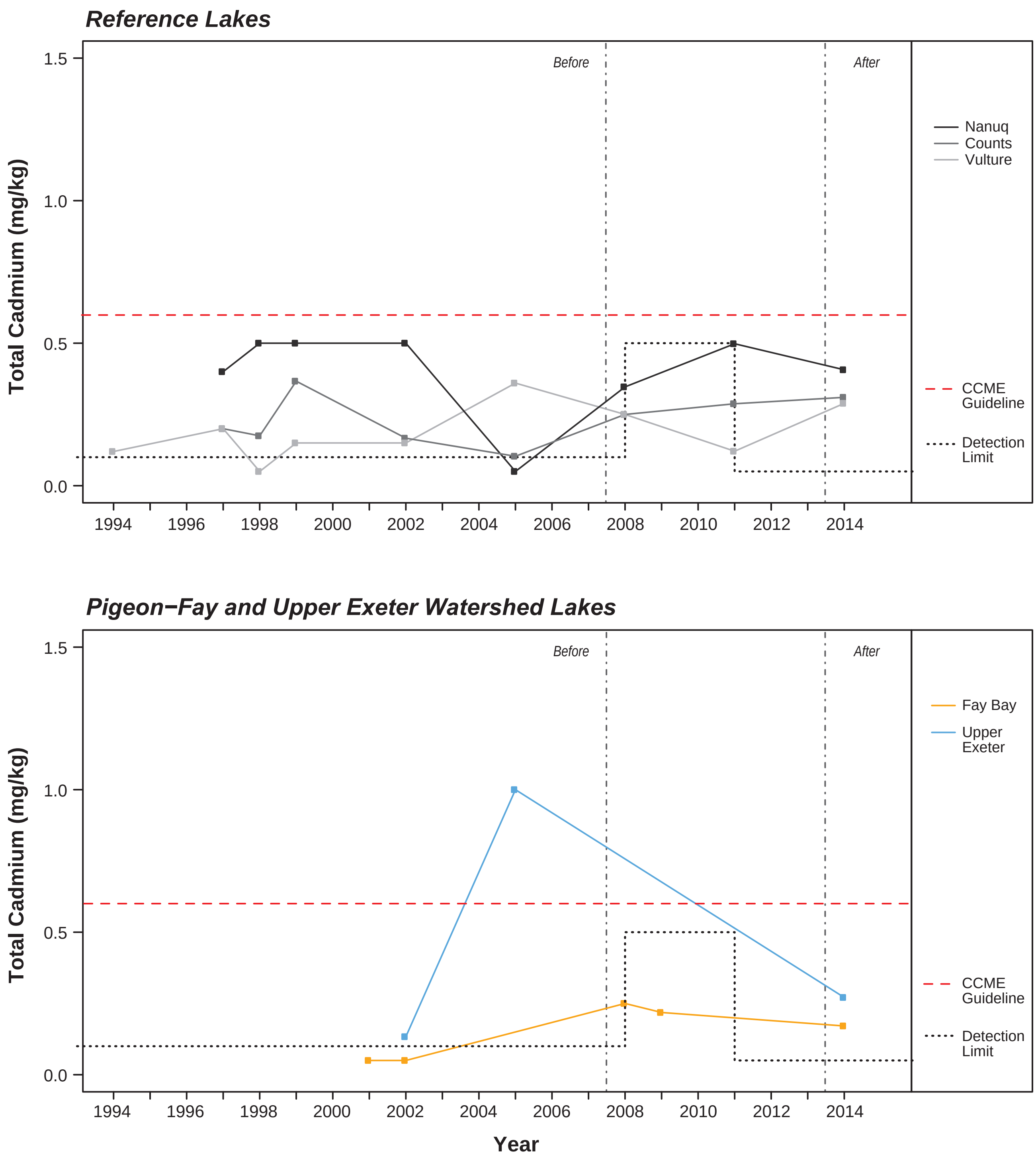
Figure 7-41

Total Boron in Pigeon AEMP Lake Sediments, 1994 to 2014



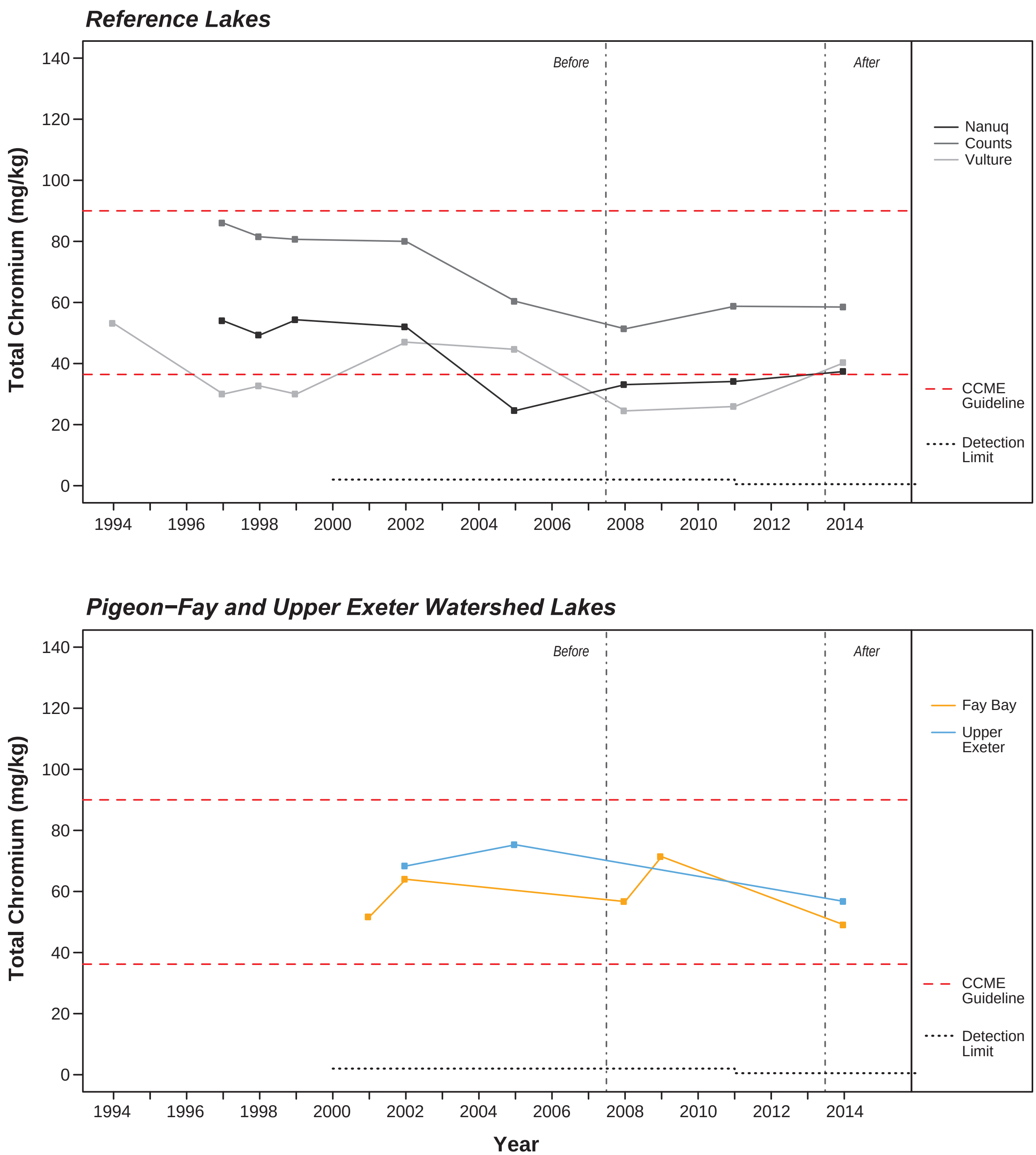
Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 7-42
Total Cadmium in Pigeon AEMP Lake Sediments, 1994 to 2014



Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME guidelines: ISQG = 0.6 mg/kg; PEL = 3.5 mg/kg (not shown).

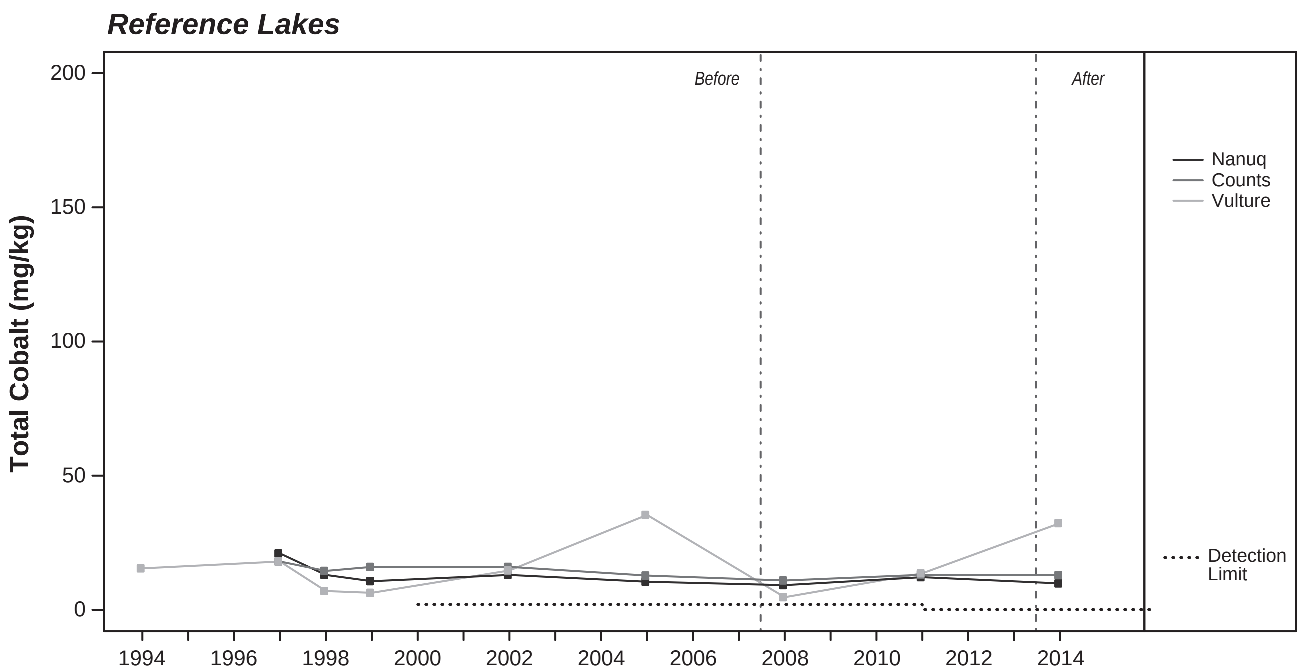
Total Chromium in Pigeon AEMP Lake Sediments, 1994 to 2014

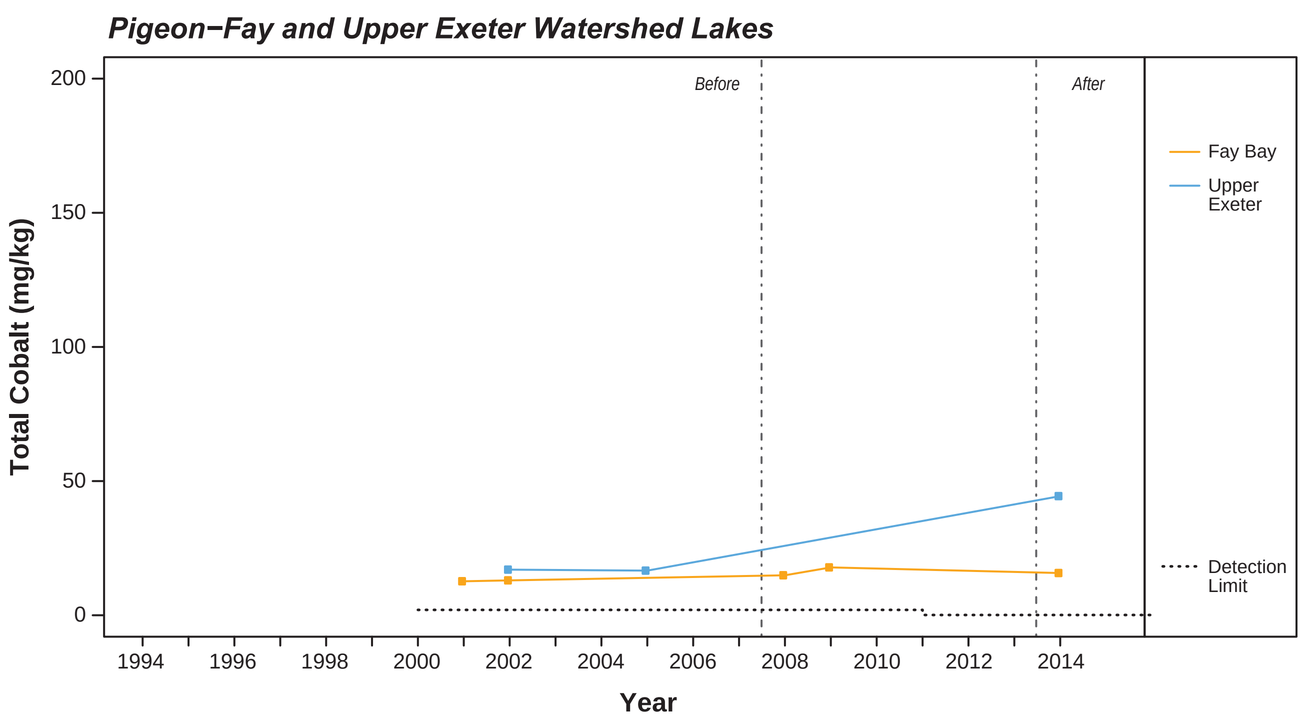


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME guidelines: ISQG = 37.3 mg/kg; PEL = 90 mg/kg.

Figure 7-44

Total Cobalt in Pigeon AEMP Lake Sediments, 1994 to 2014

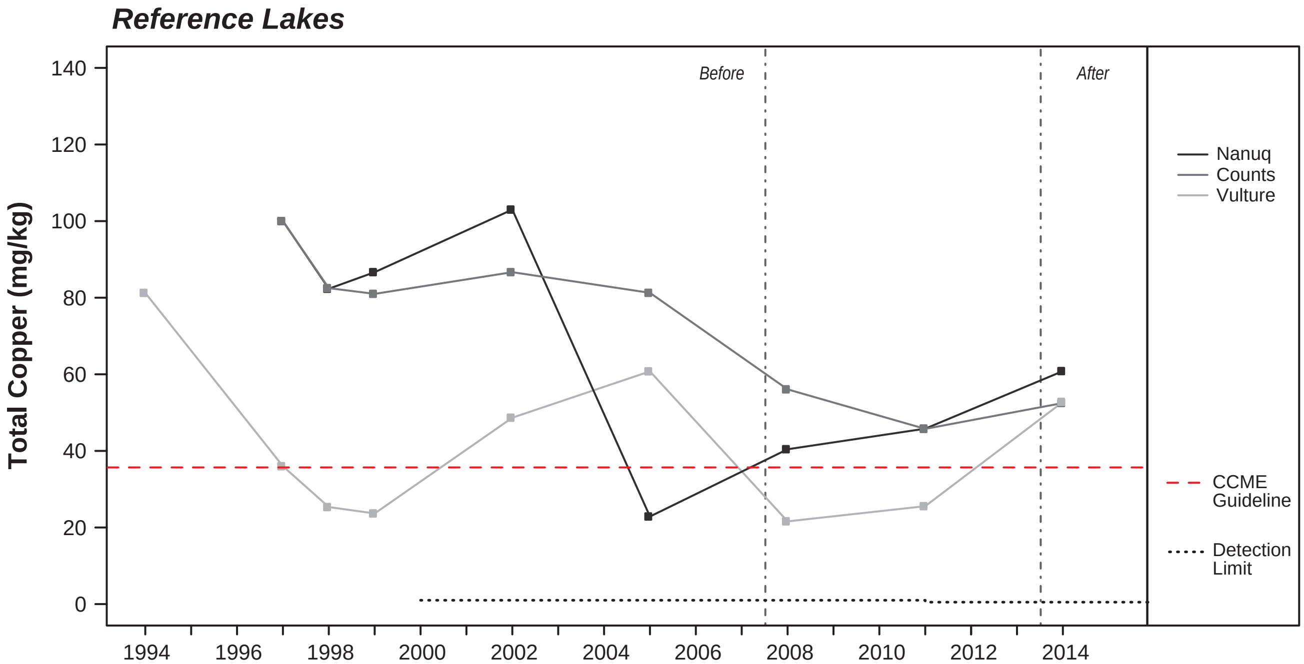


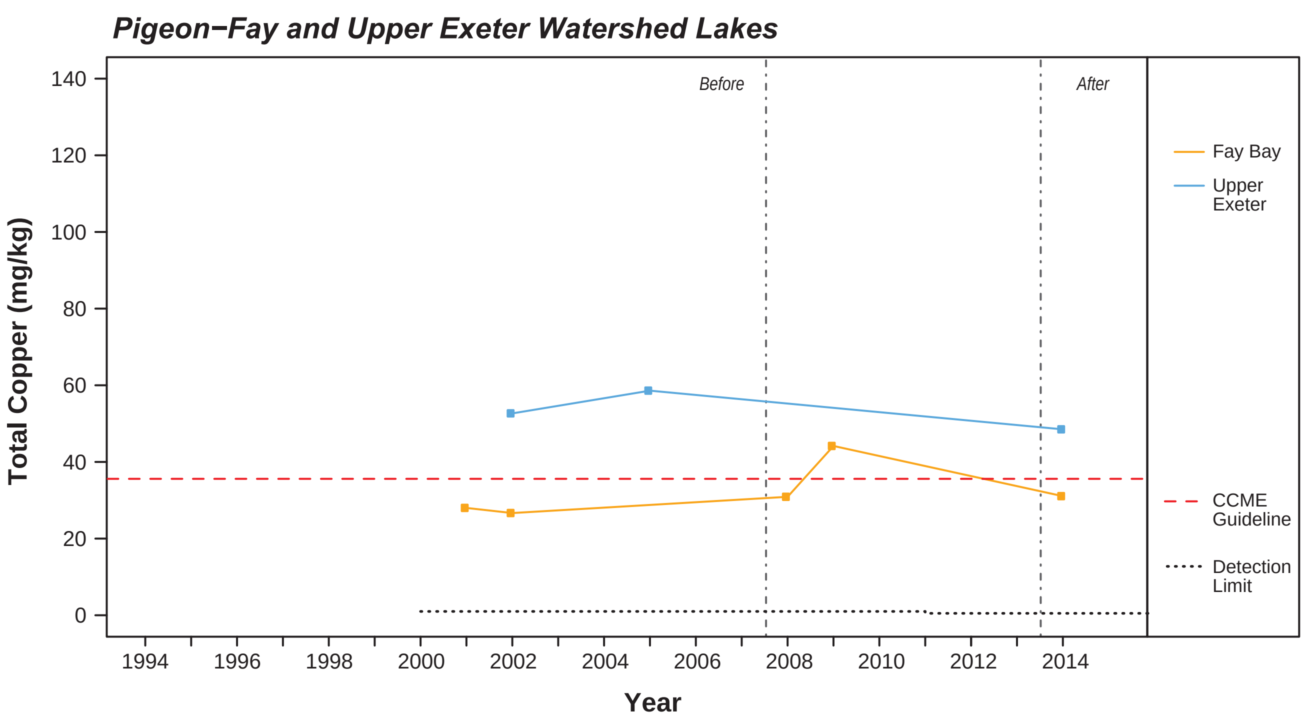


Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 7-45

Total Copper in Pigeon AEMP Lake Sediments, 1994 to 2014





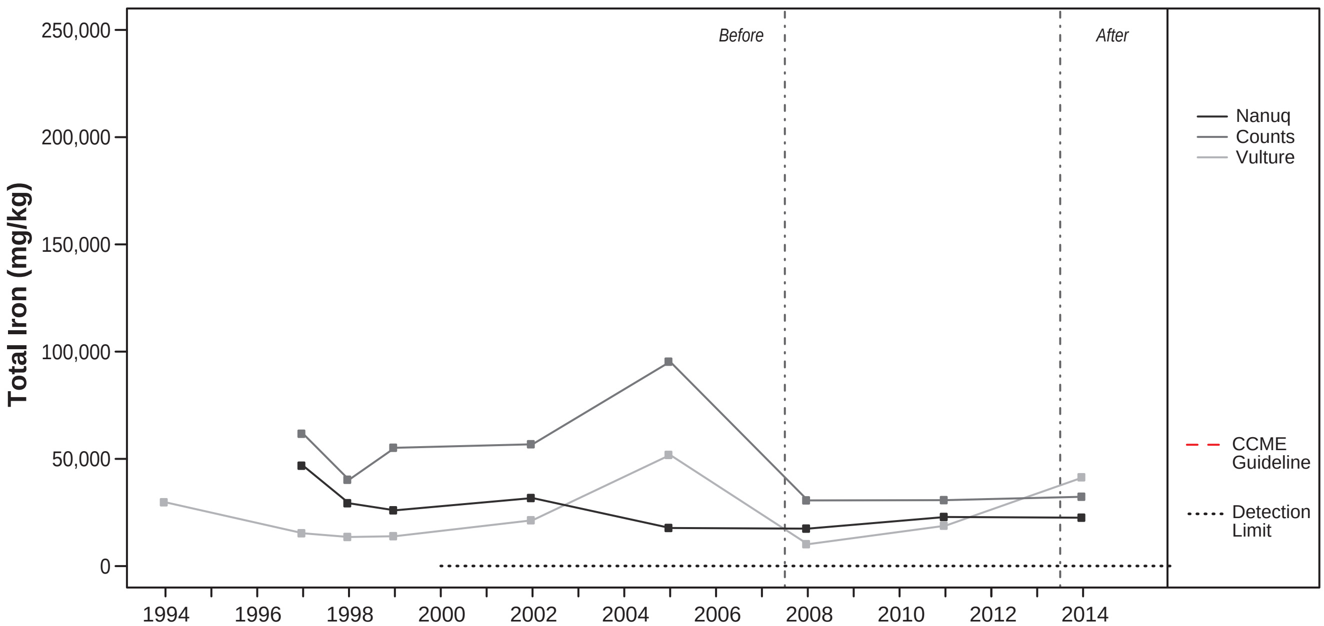
Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME guidelines: ISQG = 35.7 mg/kg; PEL = 197 mg/kg (not shown).

Figure 7-46

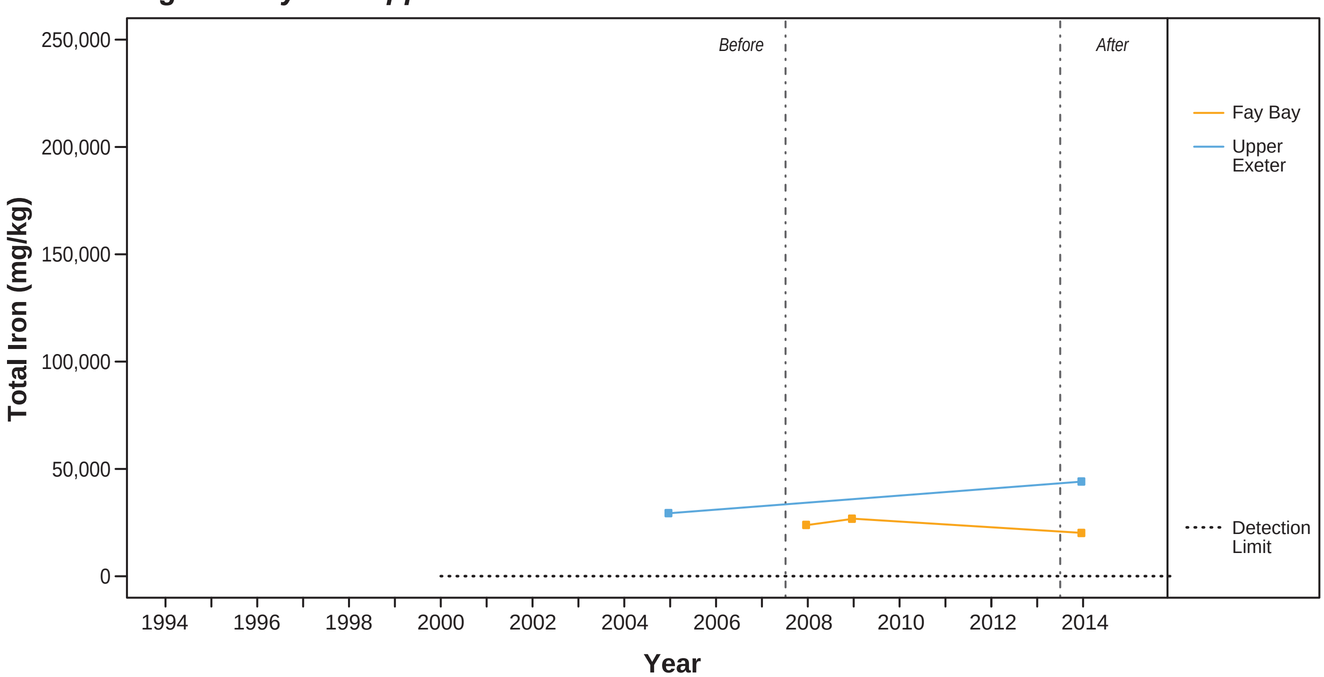
Total Iron in Pigeon AEMP Lake Sediments, 1994 to 2014

ERM

Reference Lakes



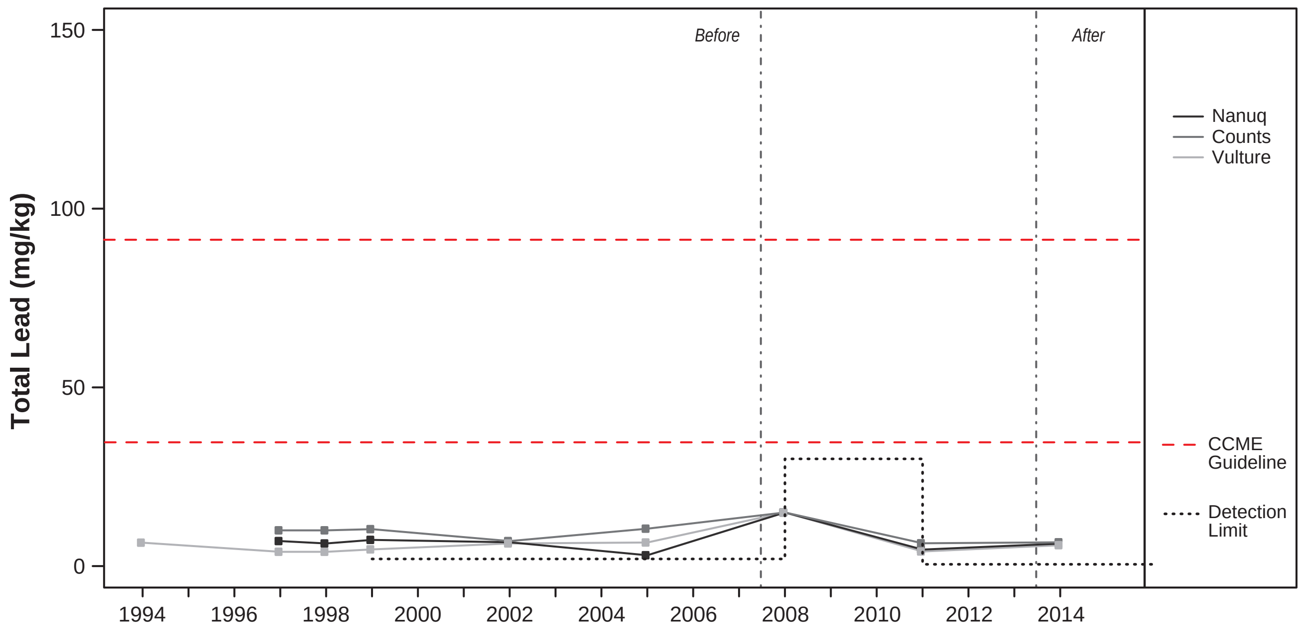
Pigeon−Fay and Upper Exeter Watershed Lakes



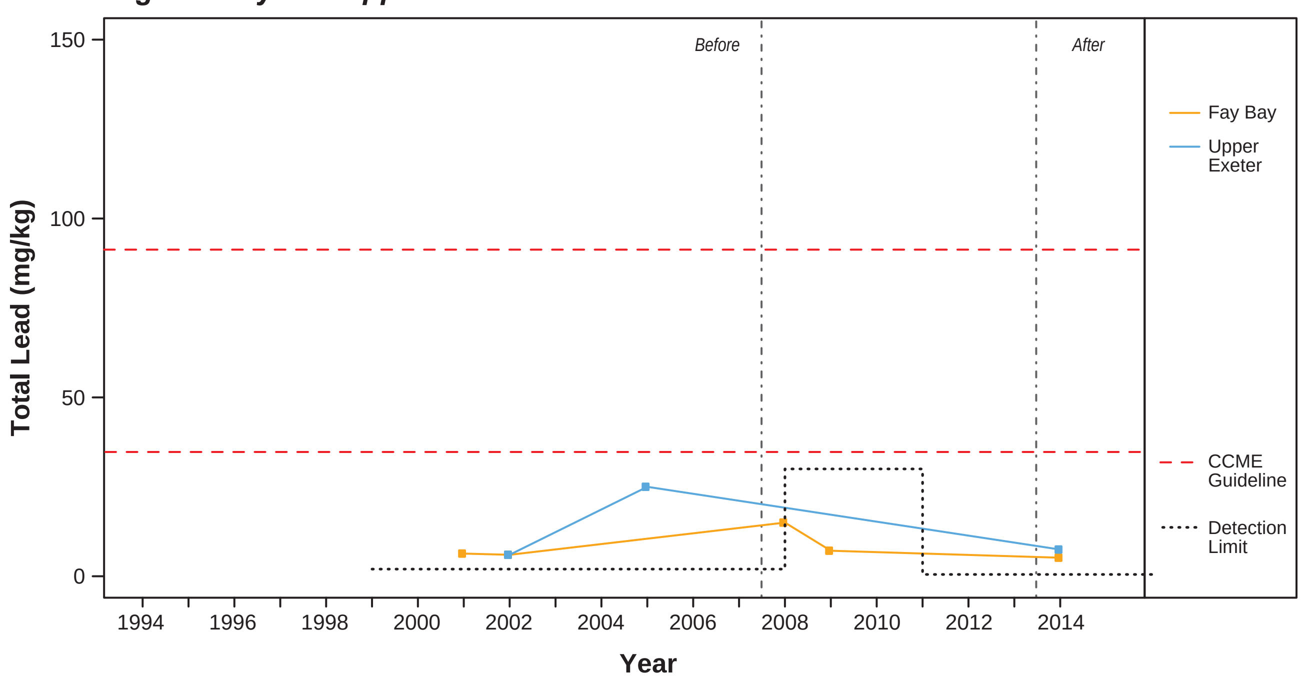
Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 7-47
Total Lead in Pigeon AEMP Lake Sediments, 1994 to 2014

Reference Lakes



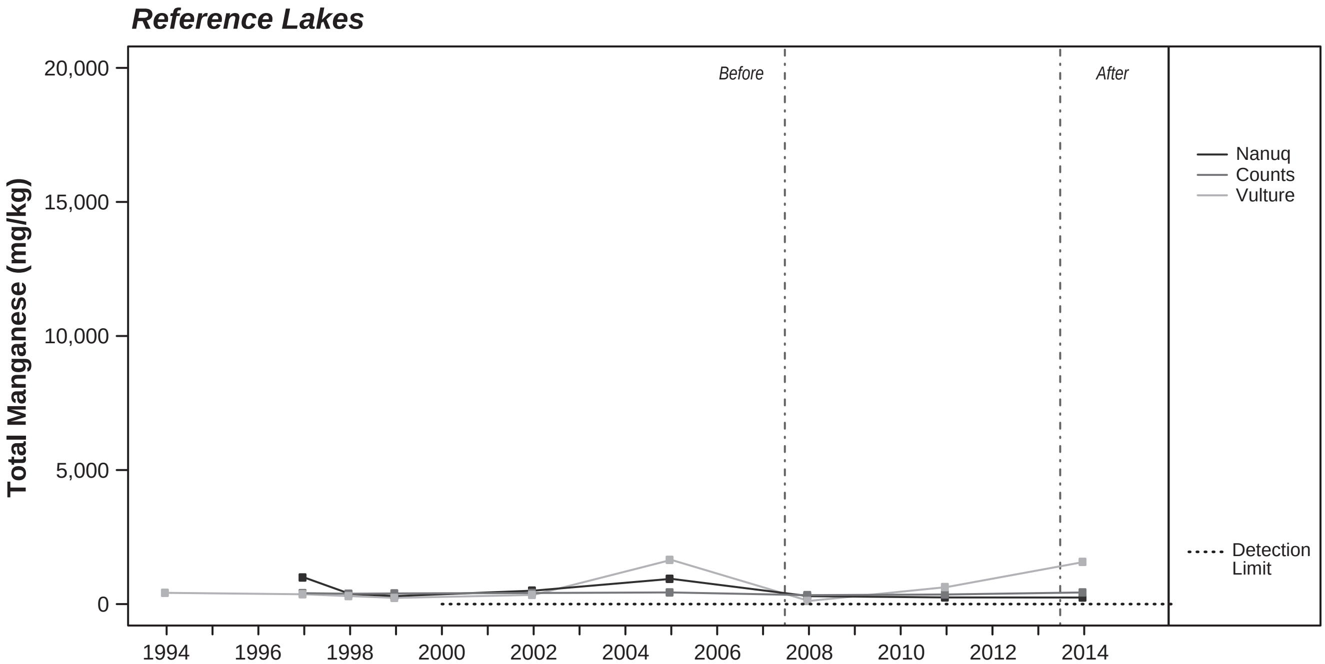
Pigeon−Fay and Upper Exeter Watershed Lakes

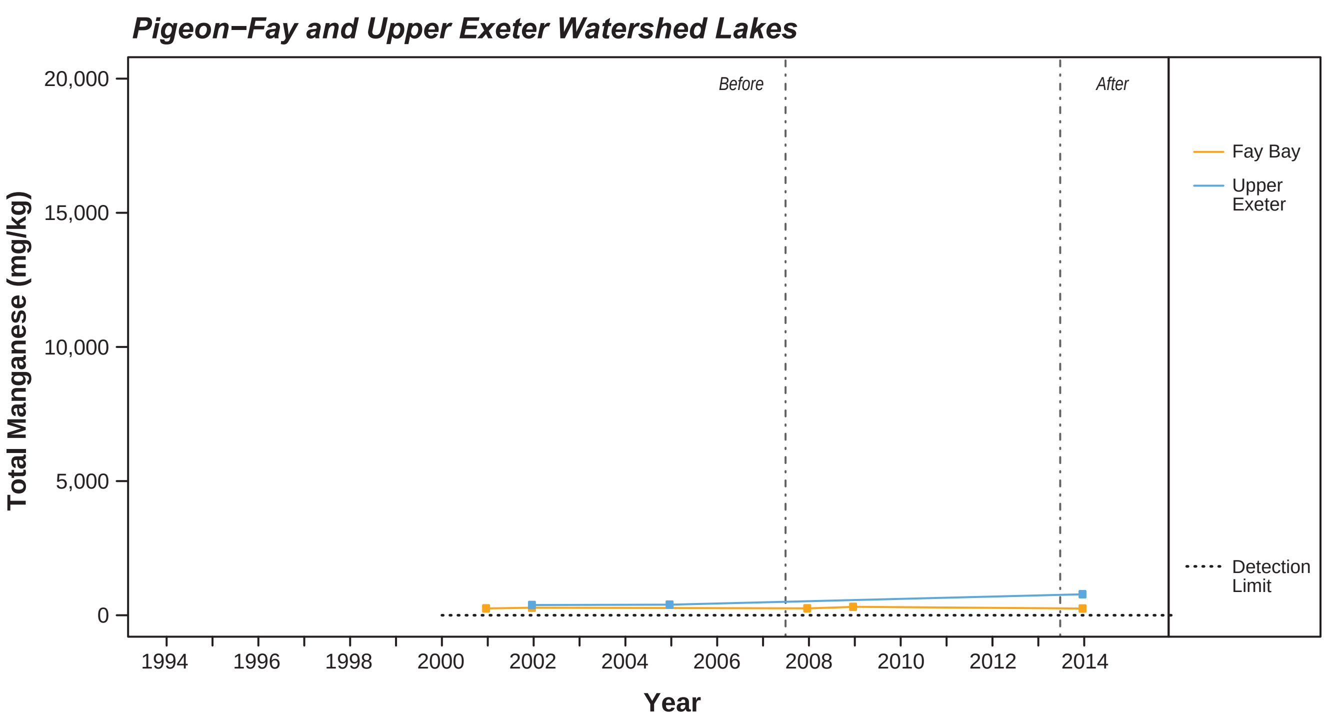


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME guidelines: ISQG = 35 mg/kg; PEL = 91.3 mg/kg.

Figure 7-48

Total Manganese in Pigeon
AEMP Lake Sediments, 1994 to 2014

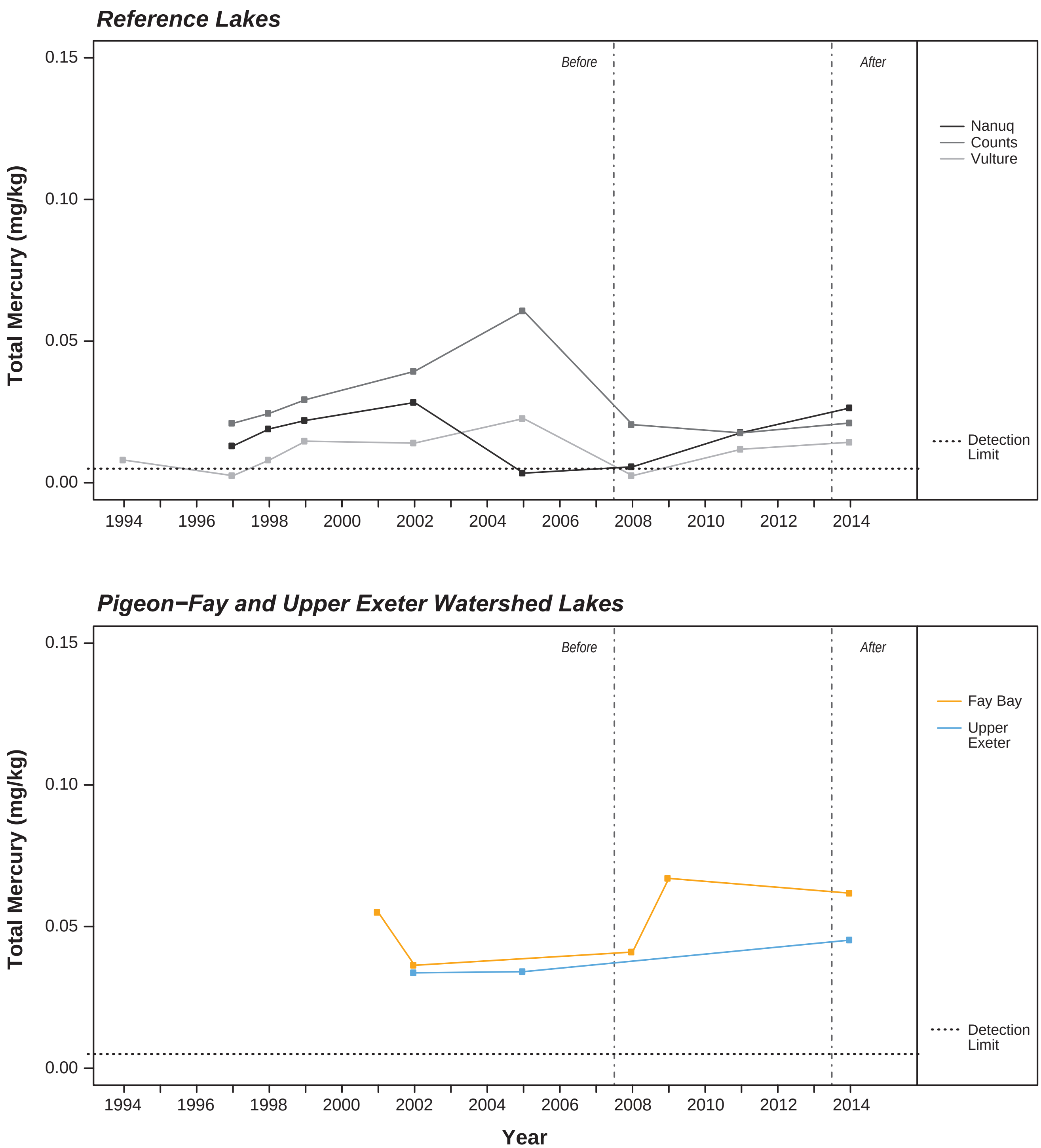




Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 7-49

Total Mercury in Pigeon AEMP Lake Sediments, 1994 to 2014

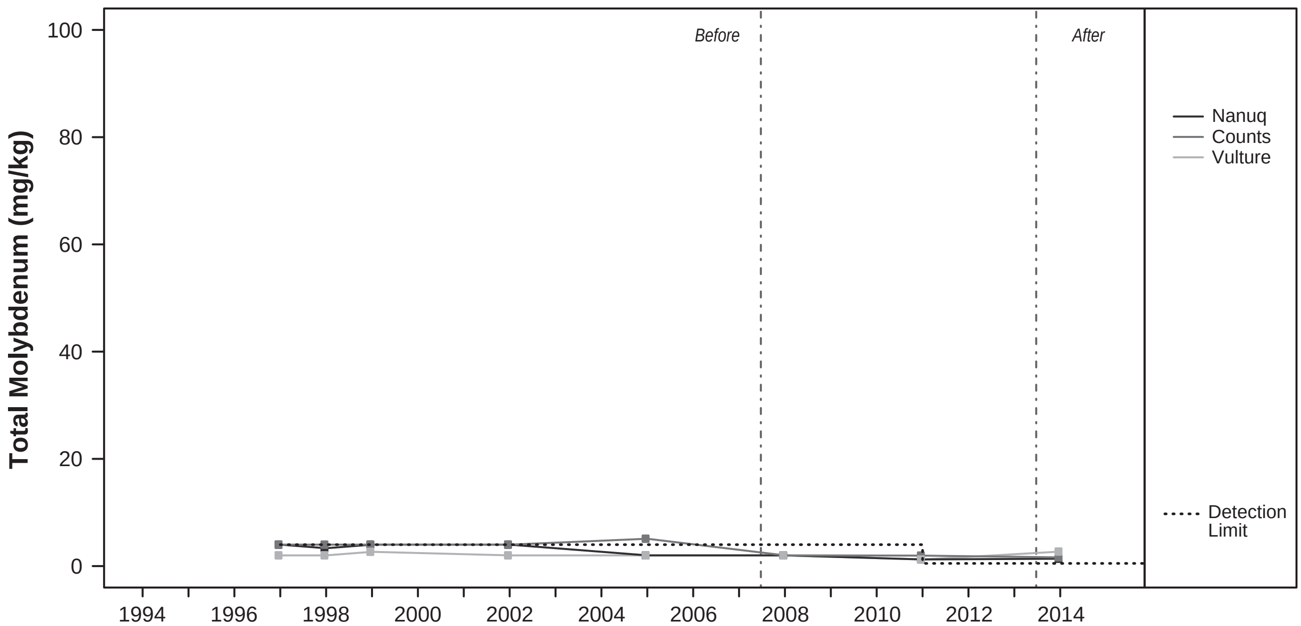


Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME guidelines: ISQG = 0.17 mg/kg (not shown); PEL = 0.486 mg/kg (not shown).

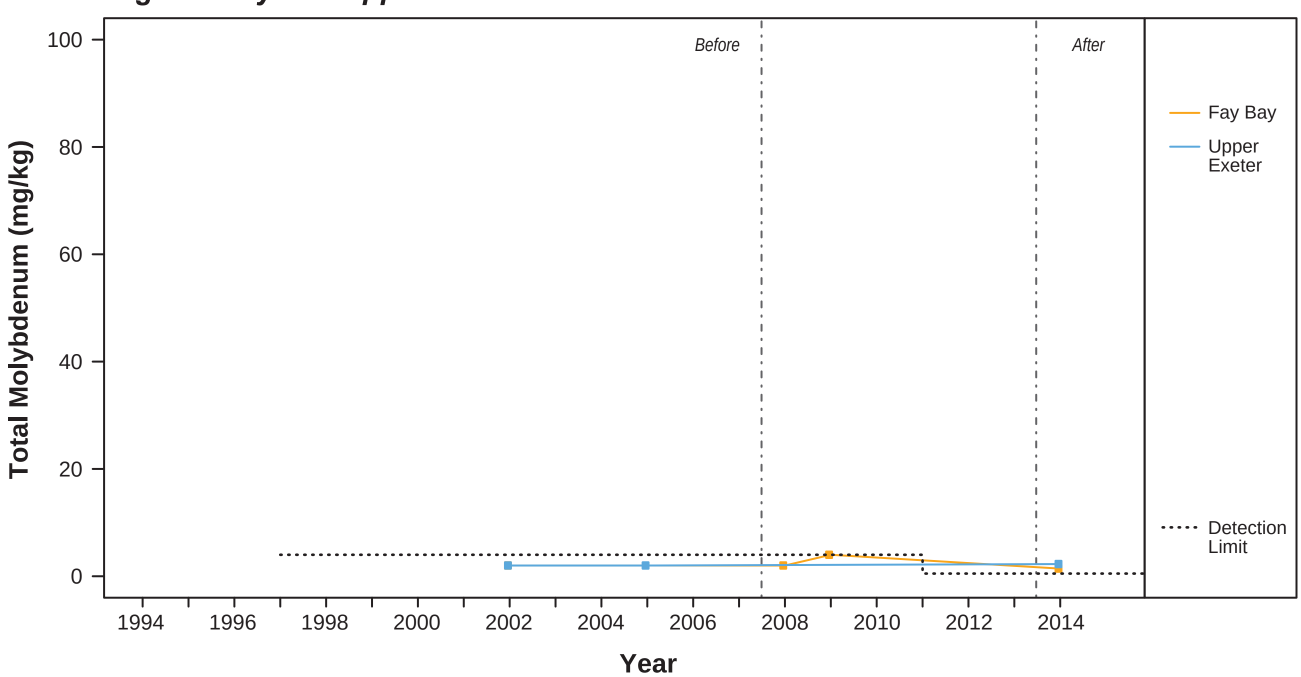
Figure 7-50

Total Molybdenum in Pigeon AEMP Lake Sediments, 1994 to 2014

Reference Lakes



Pigeon−Fay and Upper Exeter Watershed Lakes

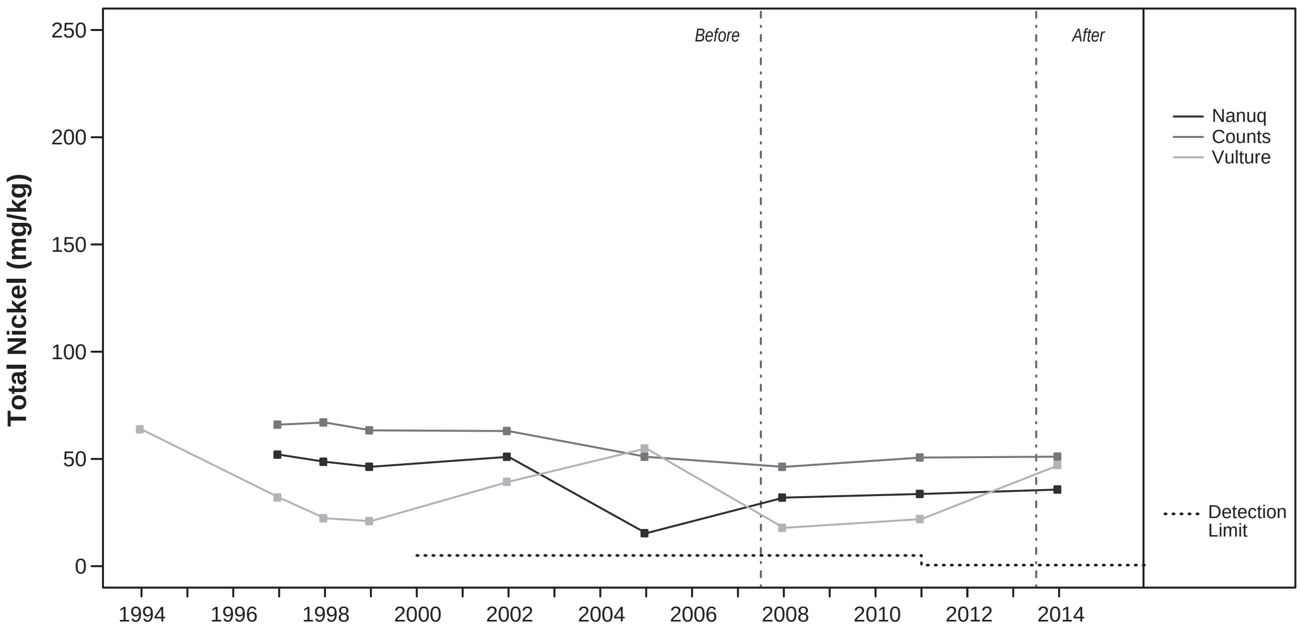


Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

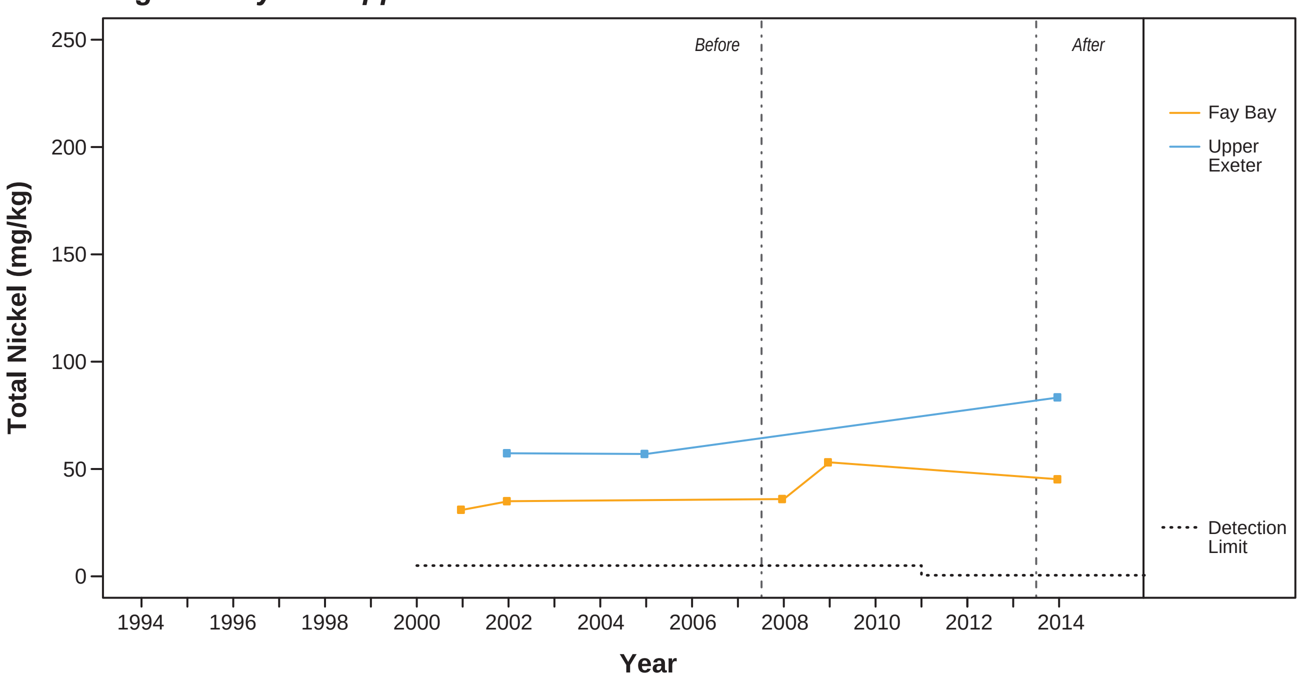
Figure 7-51

Total Nickel in Pigeon AEMP Lake Sediments, 1994 to 2014

Reference Lakes



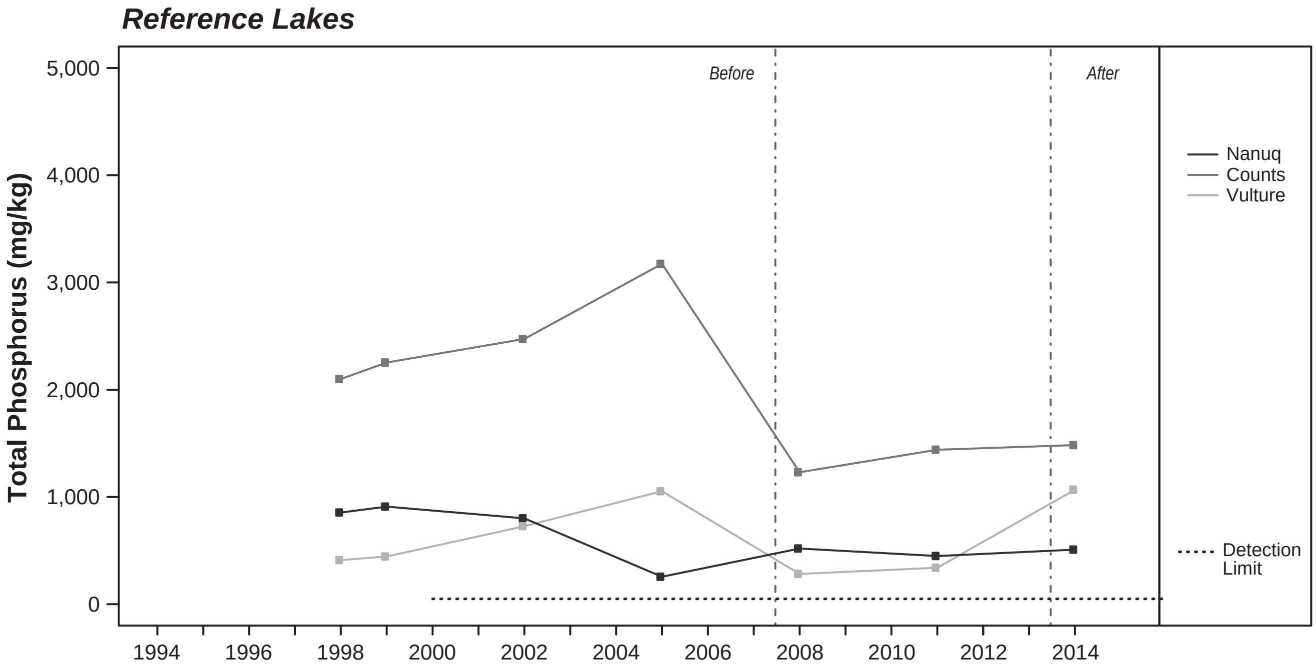
Pigeon−Fay and Upper Exeter Watershed Lakes

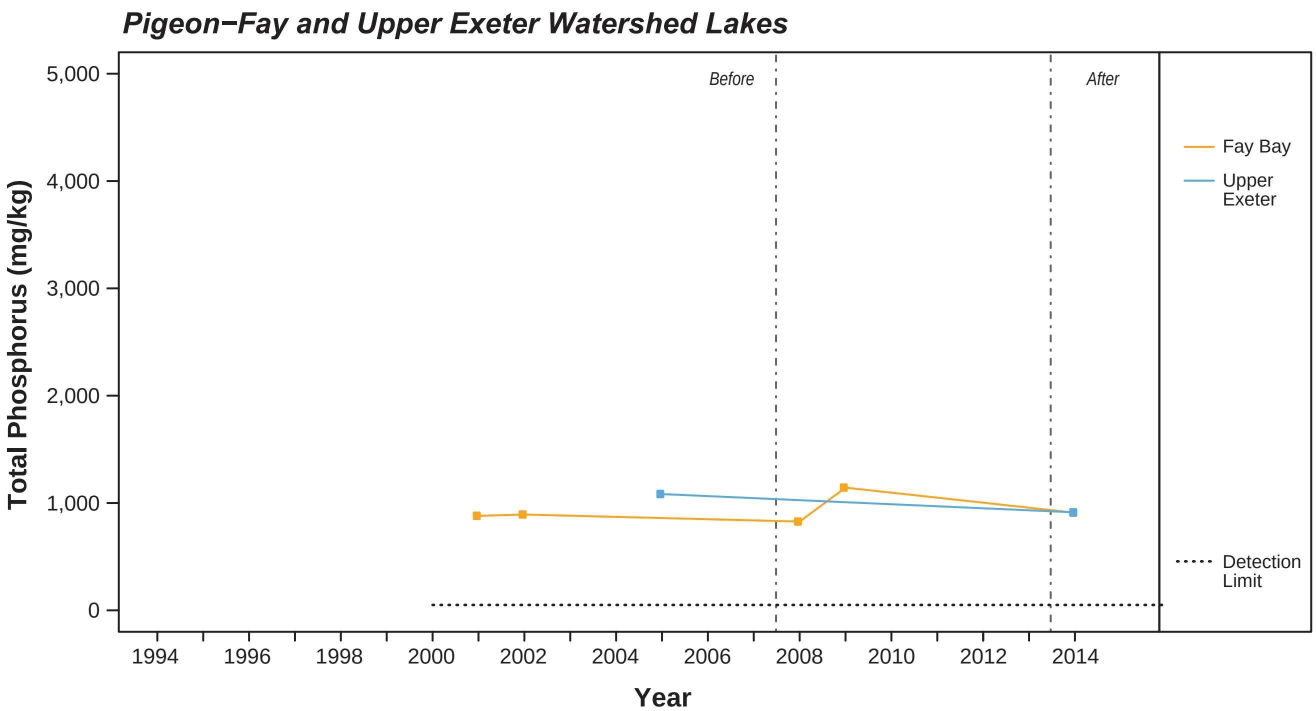


Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 7-52

Total Phosphorus in Pigeon AEMP Lake Sediments, 1994 to 2014

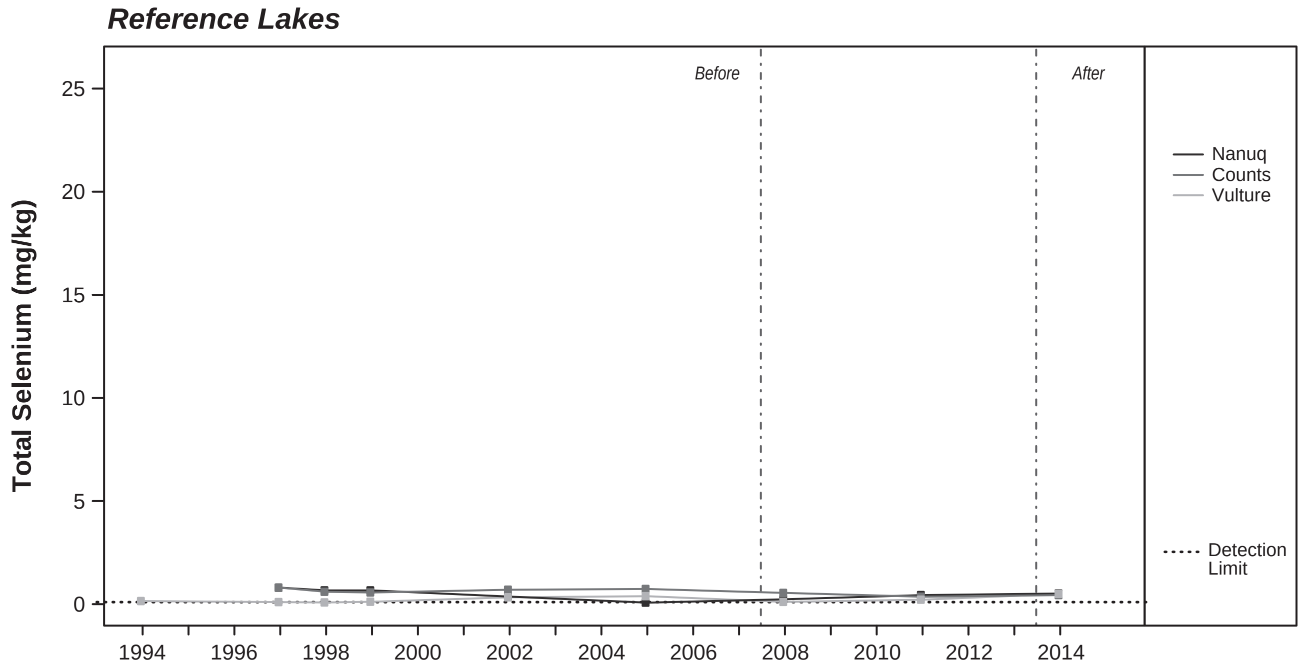


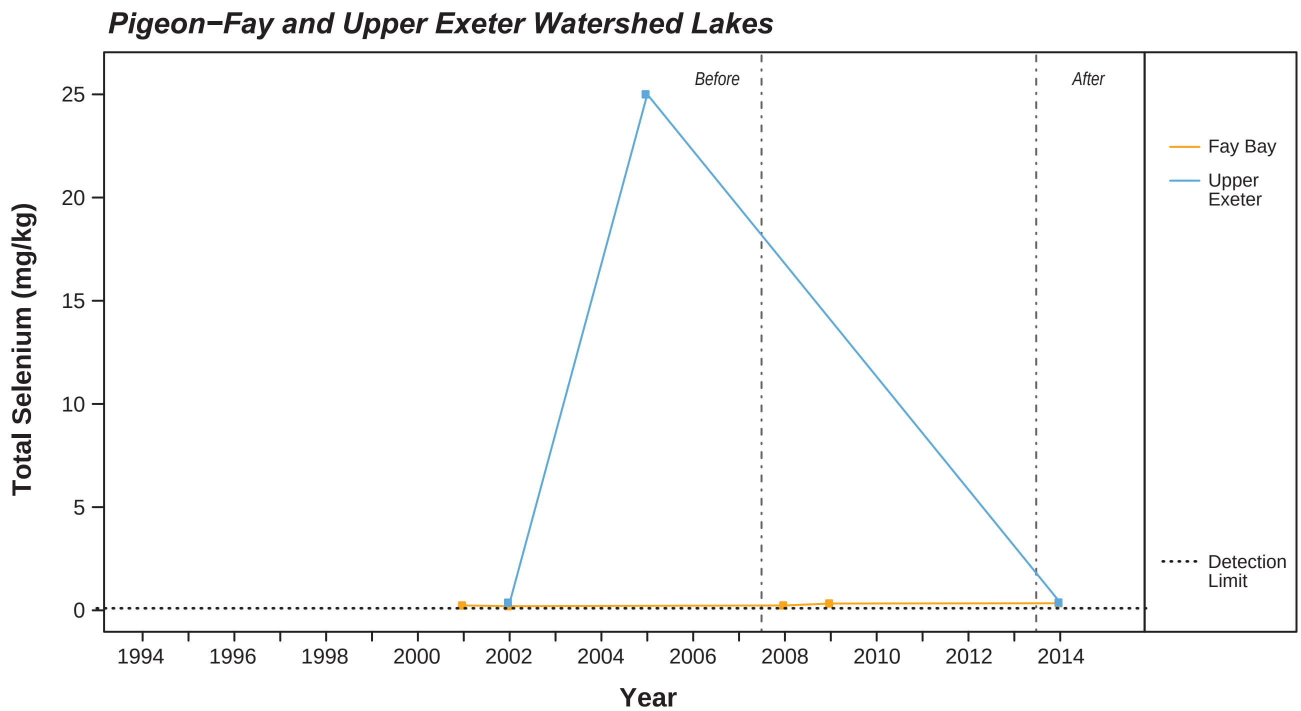


Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 7-53

Total Selenium in Pigeon AEMP Lake Sediments, 1994 to 2014

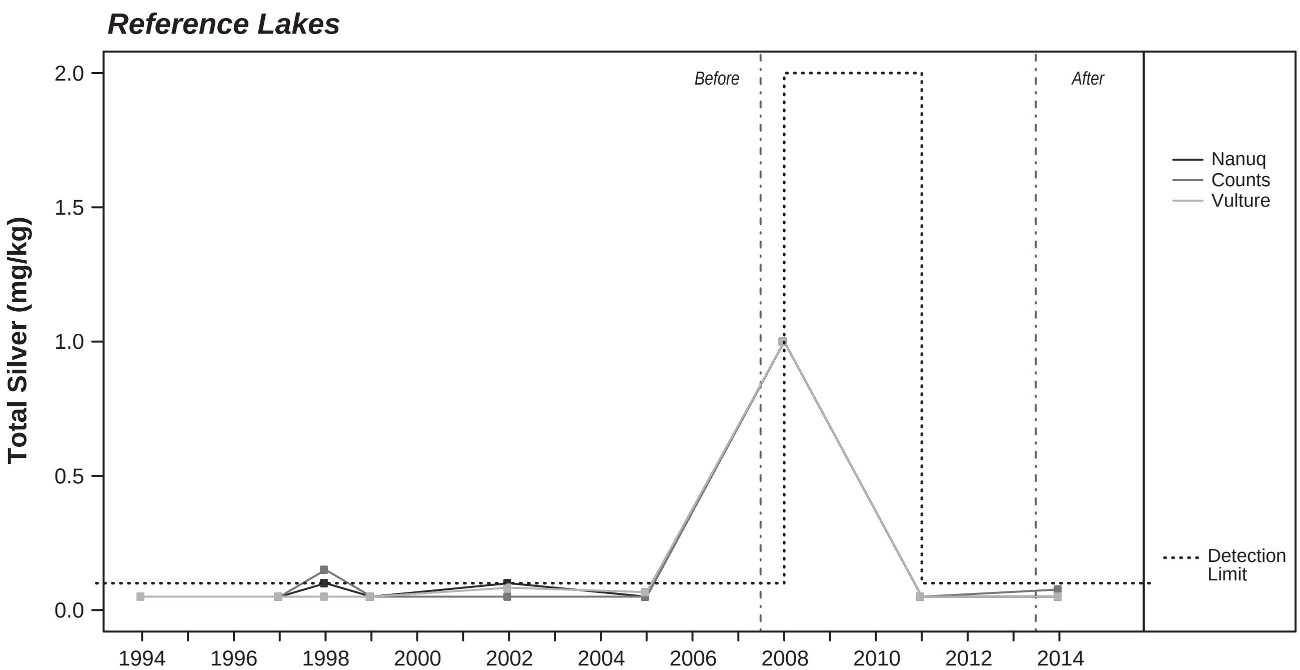


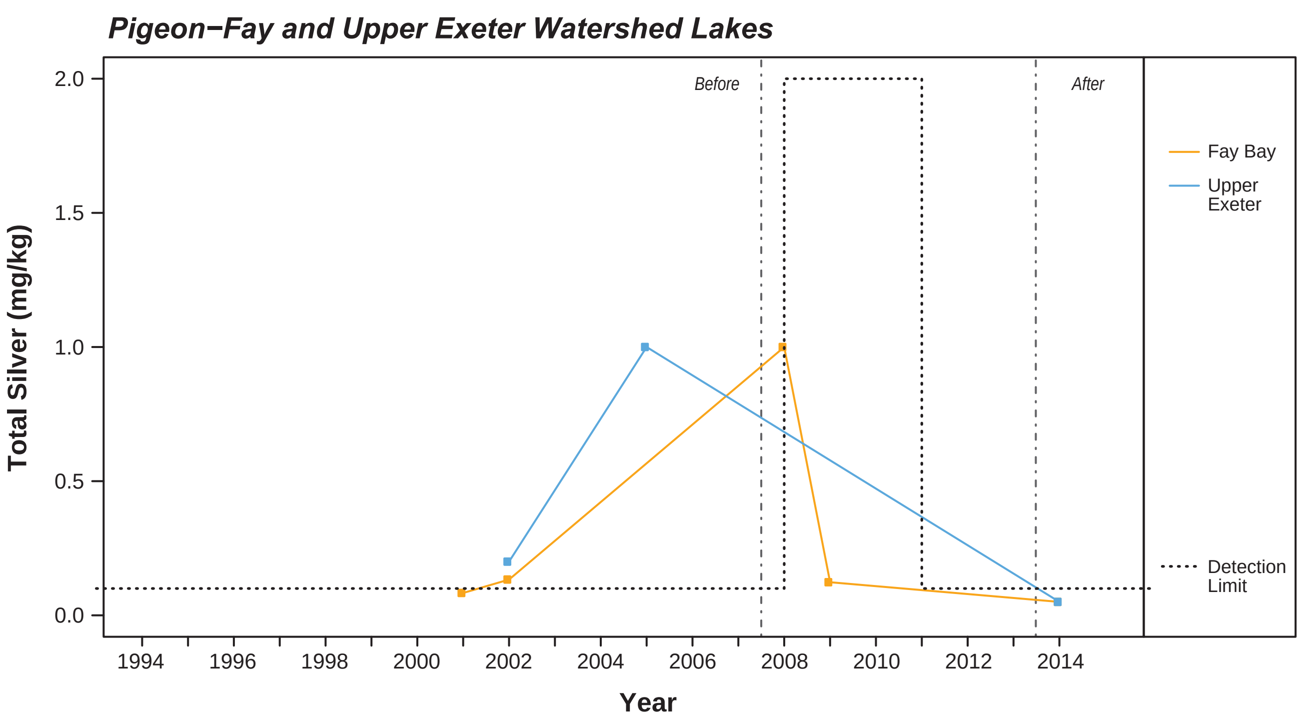


Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 7-54

Total Silver in Pigeon AEMP Lake Sediments, 1994 to 2014

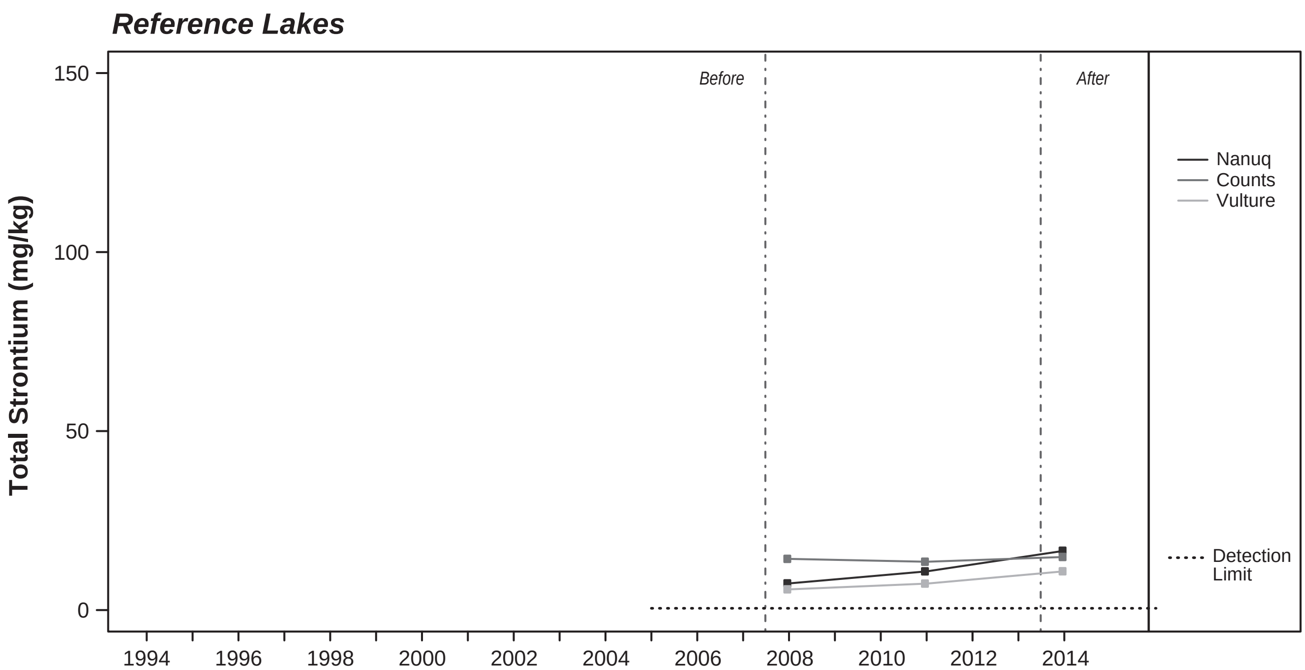


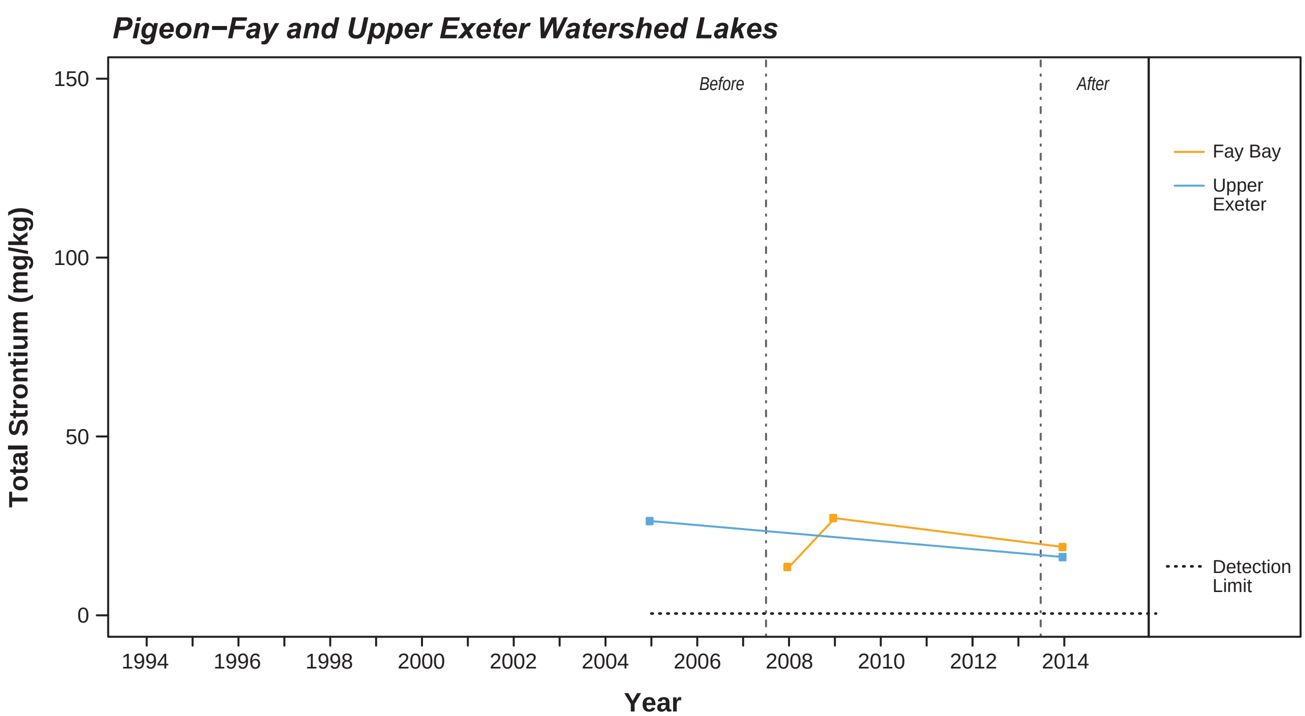


Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 7-55

Total Strontium in Pigeon AEMP Lake Sediments, 1994 to 2014

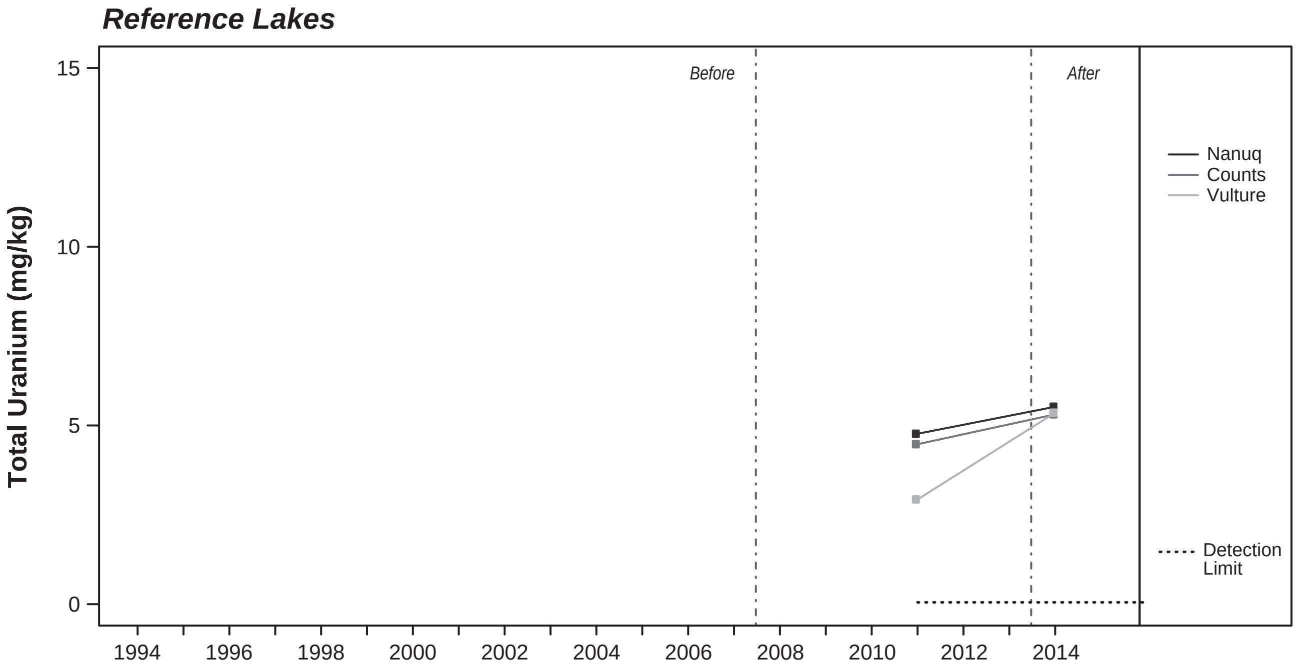


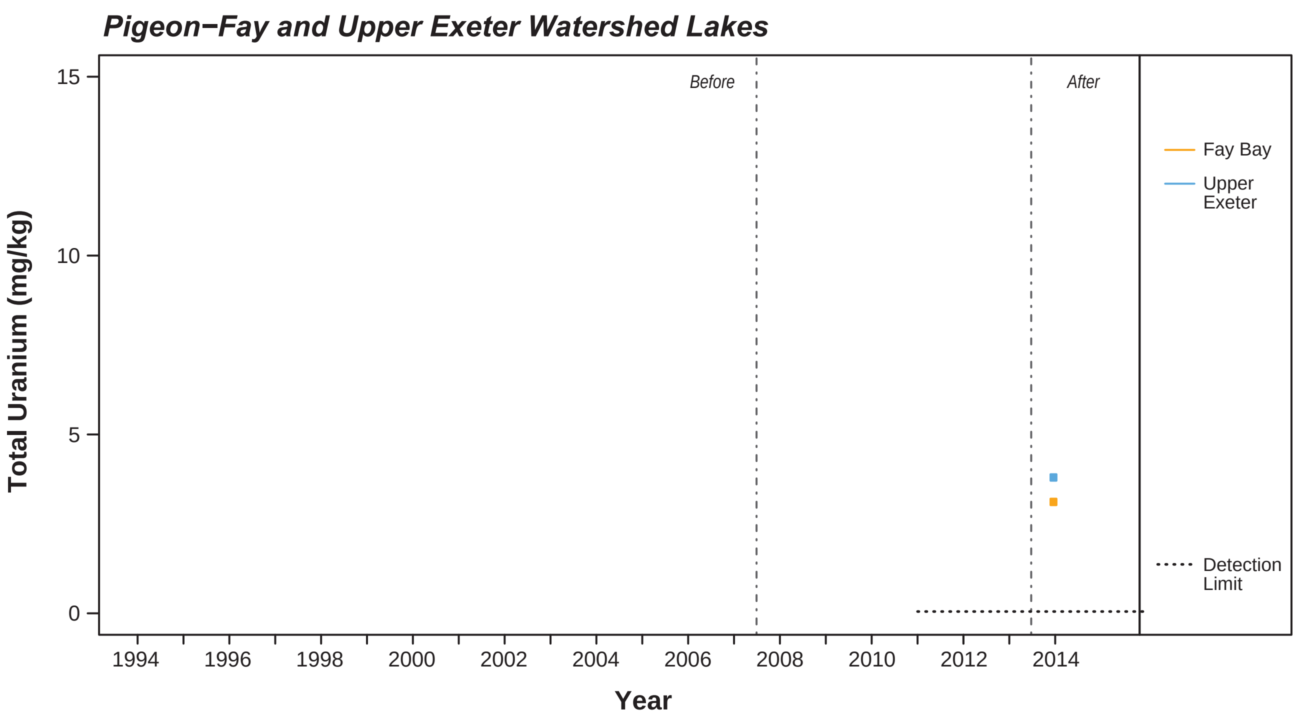


Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 7-56

Total Uranium in Pigeon AEMP Lake Sediments, 1994 to 2014

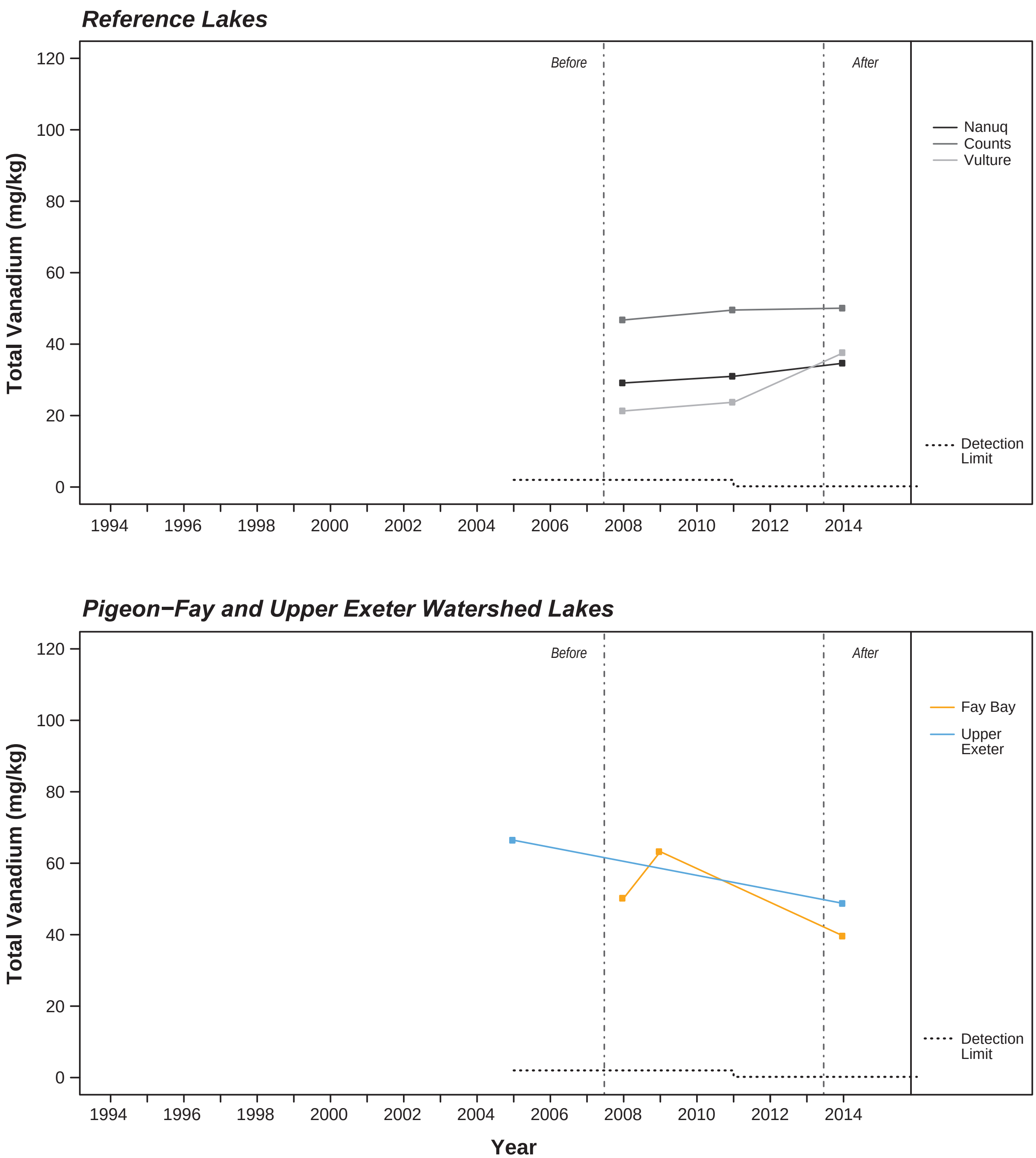




Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 7-57

Total Vanadium in Pigeon AEMP Lake Sediments, 1994 to 2014

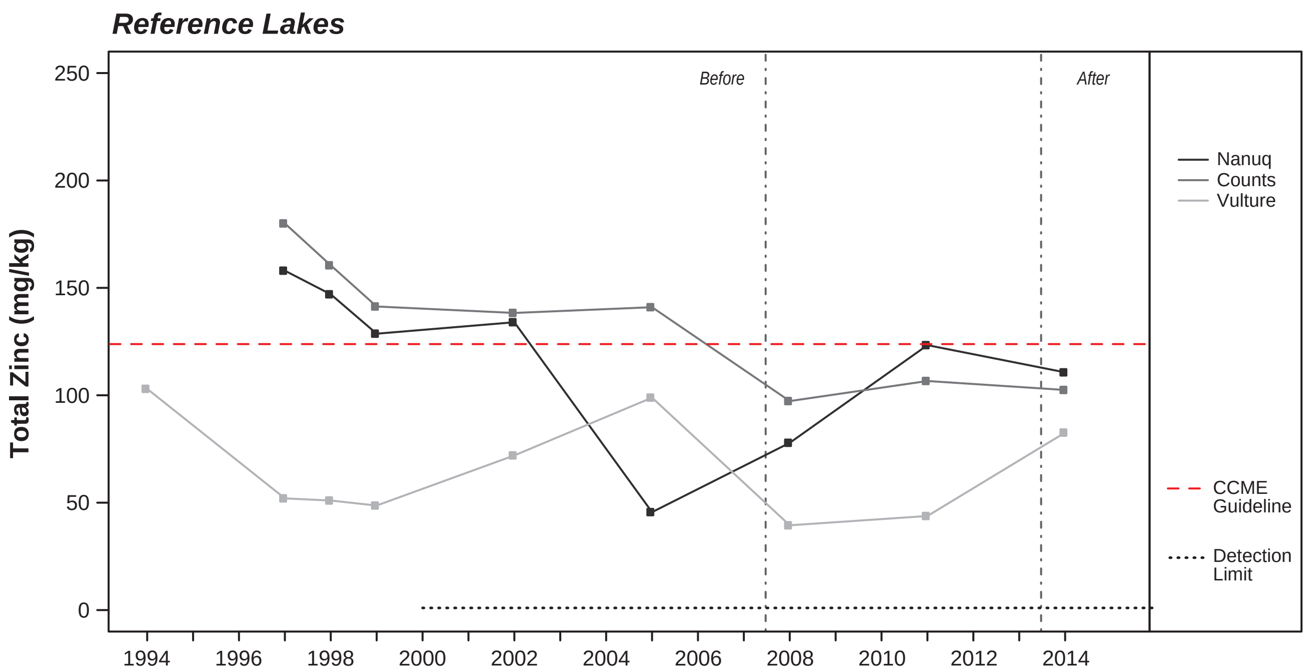


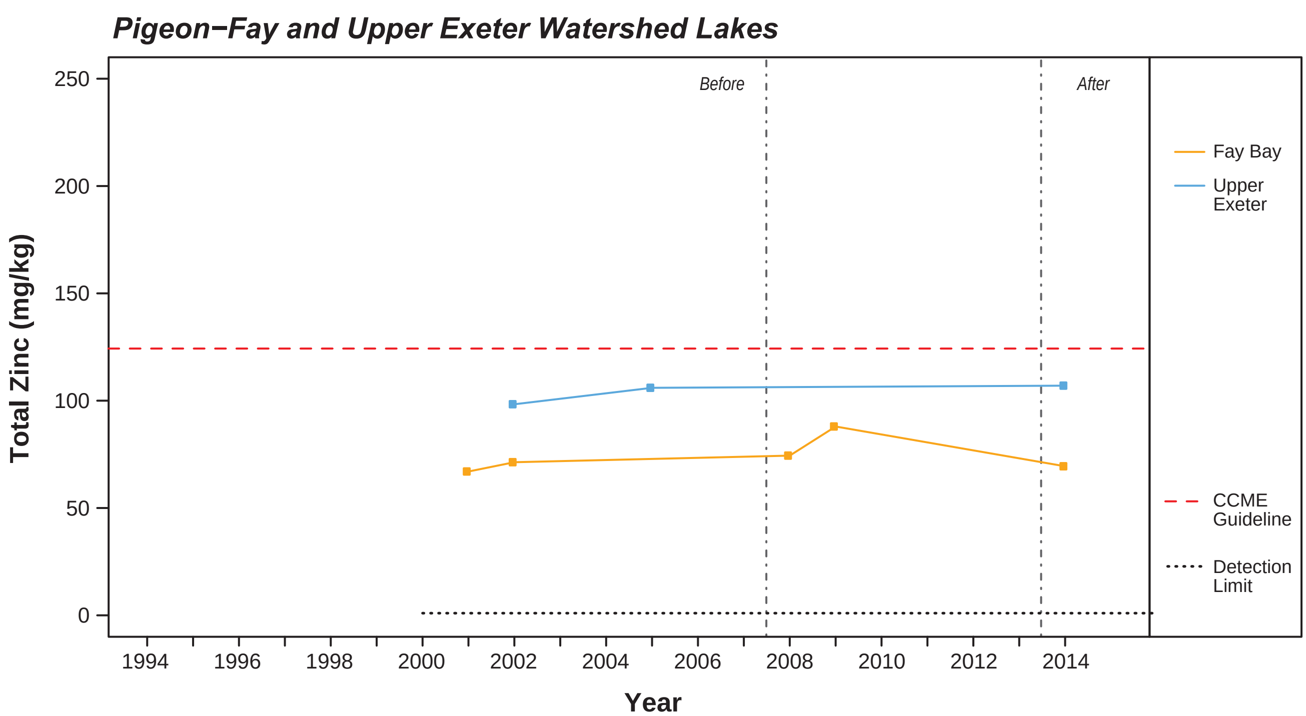
Note: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.

Figure 7-58

Total Zinc in Pigeon AEMP Lake Sediments, 1994 to 2014

ERM





Notes: For cases where detection limits varied between lakes and months, the lowest detecton limit is shown.
CCME guidelines: ISQG = 123 mg/kg; PEL = 315 mg/kg (not shown).

8. LAKE RESIDENCE TIME

The residence time for a lake or waterbody, 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 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 8-1 for the three scenarios. 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.

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 8-2). The monthly flow distributions in 2014 at the Ekati Diamond Mine were very similar to the average distribution (Table 8-2). Higher than average flow occurred in June 2014, and was compensated by lower than average flows in September and October. There are zero flows in winter months as most streams at the Ekati Diamond Mine freeze to their beds in winter. An assessment of residence times in response to average monthly flows is provided in Table 8-3. 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 8-1. 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

Note: Dashes indicate not applicable.
[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 the 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.*

Table 8-2. 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
2014	6	59	22	10	3	0

Table 8-3. Calculation of Monthly Residence Times for Lakes Lying Downstream of the LLCF, Year with Average Annual Runoff

Lake	Monthly Residence Time (days)					
	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)

(continued)

Table 8-3. Calculation of Monthly Residence Times for Lakes Lying downstream of the LLCF, Year with Average Annual Runoff (completed)

Lake	Monthly Residence Time (days)					
	May	June	July	August	September	October
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.

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 (e.g., 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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