5 EUTROPHICATION INDICATORS

5.1 Introduction

Indicators of eutrophication consist of nutrients (i.e., TP, TN, total dissolved phosphorus [TDP], total dissolved nitrogen [TDN], SRP, ammonia, and N+N), chlorophyll *a,* phytoplankton biomass and zooplankton biomass (as ash-free dry mass [AFDM]). Eutrophication indicators were added in 2007 as a component of the AEMP, because the EA predicted that the discharge of nutrients in effluent from the Mine would cause an increase in productivity in part of Lac de Gras.

5.1.1 Background

DDMI has been conducting studies and monitoring programs relating to the aquatic ecosystem of Lac de Gras since 1994, with AEMP data collected under AEMP study designs *Version 1.0* (2001[6] to 2006), *Version 2.0* (2007 to 2011), *Version 3.0* (2012 to 2016), and most recently under *Version 4.0* (2017). Since there is the potential for Mine effluent to affect Lac de Gras, the Eutrophication Indicators program has been included as a component of the AEMP since 2007, and is monitored every year (i.e., in both comprehensive and interim years). The objective of the Eutrophication Indicators program is to monitor indicators of eutrophication, and to confirm the predictions set forth in the EA (DDMI 1998b). More specifically, the program has been designed and implemented to identify changes in:

- nutrient concentrations (i.e., TP, TN, TDP, TDN, SRP, ammonia, and N+N);
- chlorophyll *a*;
- phytoplankton biomass; and
- zooplankton biomass.

5.1.2 Component History

Selected nutrients were measured during the 1995 and 1996 baseline surveys (Acres and Bryant 1996; and Golder 1998, respectively) in support of the EA for the Mine, and during the *AEMP Study Design Version 1.0* (2000 to 2006), as part of the water quality component. During the *AEMP Study Design Version 2.0* (2007 to 2010), nutrient samples were collected during the ice-cover season (April/May) and during the open-water season in July, August, and September. Chlorophyll *a*, phytoplankton biomass, and zooplankton biomass (as AFDM) samples were also collected during the open-water season in July, August and September. A review of the four years of data collected during the *AEMP Study Design Version 2.0* demonstrated that any open-water month would be equally appropriate for monitoring the indicators of eutrophication (Golder 2014a). As a result, under the *AEMP Study Design Version 3.5*, frequency of monitoring for the indicators of eutrophication was reduced to a single ice-cover and a single open-water (15 August to 15 September) sampling event (Golder 2014a). Consistent with the *AEMP Study Design*

[6] One year of baseline data was also collected in 2000 under Version 1.

Version 3.5, data for samples collected between August 15th and September 15th were included in the analysis for this re-evaluation report.

Sampling methods and laboratory procedures used during the AEMP (2000 to 2016) were generally the same as those used during the baseline period (1996 to 2000), which allowed comparisons over time. However, there have been some differences in methods over the years that have resulted in compatibility issues between recent and historical data. These included differences in sampling locations, depth, and timing, as well as analytical laboratories contracted for sample analyses, DLs, and variables analyzed. These modifications to the AEMP design were introduced, as required, to allow the annual monitoring programs to meet the goals of the AEMP and are discussed in Section 5.2.

5.2 Methods

Sampling areas for eutrophication indicators were the same as those for other AEMP components, and are based on exposure to the Mine effluent (Golder 2014a). Sampling areas consisted of a NF area and three FF areas (FF1, FFA and FFB). In addition, three transects (referred to as MF areas) between the NF and FF areas were sampled. The NF-MF1-FF1 transect extends towards the FF1 area, northwest of the Mine site. The NF-MF2-FF2 transect extends to the northeast, towards the FF2 area near the Lac du Sauvage inflow (the Narrows). The NF-MF3-FFB-FFA transect extends to the south and west of the Mine site towards FFB and FFA areas and the outlet of Lac de Gras. A single station at the Lac de Gras outlet (LDG-48) was also sampled. Clusters of five replicate stations were sampled in the NF and FF areas. The number of stations in the MF areas was changed from *AEMP Study Design Version 2.0* to *Version 3.0* to better delineate the extent of effects and define gradients along each transect; the number of stations in the MF3 area were increased, and the number of stations in the MF1 and MF2 areas were decreased. In addition to samples collected in Lac de Gras, samples were collected at three stations in Lac du Sauvage (LDS-1, LDS-2 and LDS-3), near the outlet to Lac de Gras.

5.2.1 Data Sources

Indicators of eutrophication data included in the *2014 to 2016 Aquatic Effects Re-revaluation Report* were taken from the following data sources:

- baseline data collected from 1995 to 1999
- data collected during the AEMP *Study Design Version 1.0* (2000 to 2006)
- data collected during the AEMP *Study Design Version 2.0* (2007 to 2010)
- data collected during the AEMP *Study Design Version 3.2* (2011 to 2013)
- data collected during the AEMP *Study Design Version 3.5* (2014 to 2016)

Sampling locations for indicators of eutrophication from baseline to the *AEMP Study Design Version 3.5* are presented in Table 5-1. Sampling locations from baseline studies and *AEMP Study Design Version 1.0* and *Version 2.0* were assigned to current AEMP stations. Historical stations not included in the current *AEMP Study Design Version 3.5* (i.e., discontinued stations) are shaded in grey and were not included in trend plots or data analyses.

Table 5-1 Summary of Baseline and AEMP Eutrophication Indicators Data, 1996 to 2016

Waterbody	Area	Station[a]	UTM Coordinates[b]		Distance from Diffusers[c] (km)	Years Sampled										
			Easting (m)	Northing (m)		1996, 1997[d]	1998[d]	1999[d]	2000	2002-2006	2007	2008	2009 - 2011	2012, 2013	2014, 2015	2016
Lac de Gras[f]	Near-field	NF1	535740	7153854	0.4						X	X	X	X	X	X
		NF2	536095	7153784	0.5						X	X	X	X	X	X
		NF3	536369	7154092	0.9						X	X	X	X	X	X
		NF4	536512	7154240	1.1						X	X	X	X	X	X
		NF5 (WQ-06, LDG 42)	536600	7153864	1.0	X	X	X	X	X	X	X	X	X	X	X
	Mid-field 1	MF1-1	535008	7154699	1.5						X	X	X	X	X	X
		MF1-2	533682	7155356	2.9						X	X	X			
		MF1-3	532236	7156276	4.7				X	X	X	X	X	X	X	X
		MF1-4	532494	7157657	7.2						X	X	X			
		MF1-5	528432	7157066	8.5									X	X	X
	Mid-field 2	MF2-1	538033	7154371	2.4						X	X	X	X	X	X
		MF2-2	539198	7154643	3.7						X	X	X			
		MF2-3	540365	7156045	5.4						X	X	X	X	X	X
		MF2-4	540955	7157359	6.9						X	X	X			
	Far-field 2[e]	FF2-1	541500	7159522	9.3						X	X	X			
		FF2-2	541588	7158561	8.3				X	X	X	X	X	X	X	X
		FF2-3	543478	7159267	10.1						X	X	X			
		FF2-4	543752	7158945	10.2						X	X	X			
		FF2-5	544724	7158879	11.4						X	X	X	X	X	X
	Mid-field 3	MF3-1	537645	7152432	2.7						X	X	X	X	X	X
		MF3-2	536816	7151126	4.2				X	X	X	X	X	X	X	X
		MF3-3	536094	7148215	7.2						X			X	X	X
		MF3-4 (WQ-05)	532545	7147011	11.0	X		X	X	X	X	X	X	X	X	X
		MF3-5	528956	7146972	14.6									X	X	X
		MF3-6	525427	7148765	18.5					X	X	X	X	X	X	X
		MF3-7	521859	7150039	22.3									X	X	X
	Far-field 1	FF1-1	525430	7161043	13.6						X	X	X	X		X
		FF1-2	524932	7159476	12.9						X	X	X	X		X
		FF1-3	526407	7160492	12.8						X	X	X	X		X
		FF1-4	526493	7159058	11.4						X	X	X	X		X
		FF1-5	526683	7161824	12.8						X	X	X	X		X
	Far-field B	FFB-1	516831	7148207	26.4					X	X	X	X	X		X
		FFB-2	518473	7150712	25.0						X	X	X	X		X
		FFB-3	518048	7147557	25.2						X	X	X	X		X
		FFB-4	515687	7150036	27.6						X	X	X	X		X
		FFB-5	516533	7150032	26.8						X	X	X	X		X

Table 5-1 Summary of Baseline and AEMP Eutrophication Indicators Data, 1996 to 2016

Waterbody	Area	Station(a)	UTM Coordinates(b)		Distance from Diffusers(c) (km)	Years Sampled										
			Easting (m)	Northing (m)		1996, 1997(d)	1998(d)	1999(d)	2000	2002-2006	2007	2008	2009 - 2011	2012, 2013	2014, 2015	2016
Lac de Gras(f)	Far-field A	FFA-1	506453	7154021	36.8						X	X	X	X		X
		FFA-2	506315	7155271	38.3						X	X	X	X		X
		FFA-3	505207	7153887	38.7						X	X	X	X		X
		FFA-4	503703	7154081	40.2						X	X	X	X		X
		FFA-5	505216	7156657	40.0					X	X	X	X	X		X
Outlet of Lac de Gras	n/a	LDG-48	490900	7161750	55.6						X	X	X	X	X	X
Lac du Sauvage	n/a	LDS-1	546398	7161179	-								X	X		X
		LDS-2	546807	7160027	-									X		X
		LDS-3	547191	7160256	-									X		X

Note: Shading identifies stations that were discontinued, as per the *AEMP Study Design Version 3.0* (Golder 2011b). These stations are not included in the current (2014 to 2016) Aquatic Effects Re-evaluation Report.
a) Current AEMP station names are presented, and historical sampling station names are provided in parentheses.
b) UTM coordinates are reported as Zone 12, North American Datum (NAD) 83.
c) Approximate distance from the Mine effluent diffusers along the most direct path of effluent flow.
d) For 1996, 1997, information is provided only for stations located within AEMP areas.
e) From 2012 onwards, the FF2 area is considered to be part of the NF-MF2-FF2 transect.
f) SNP data are not included.
UTM = Universal Transverse Mercator coordinate system; - = not applicable.

5.2.1.1 Baseline and AEMP Version 1.0 Data (1995 to 2006)

Baseline eutrophication data were collected in 1995 (Acres and Bryant 1996) and 1997 (Golder 1998); however, data quality and comparability issues (based on analytical methods) were identified with the collected data (DDMI 2007). For example, much of the nutrient data were disqualified due to high DLs and laboratory changes. As a result, only the total ammonia data could be used from the baseline period.

Data from 2000 to 2006 included an annual set of ice-cover and open-water data collected from mid-depth in the water column. Laboratory changes and differences in field sampling methods limited the number of eutrophication indicator variables available from this period. Data considered useful from the 2000 to 2006 period were TN, total ammonia, N+N, and chlorophyll *a*; however, these variables were analyzed in mid-depth samples, as opposed to the depth-integrated samples collected during the open-water season under *AEMP Study Design Version 2.0, Version 3.1* and *Version 3.5*. Phytoplankton biomass data were also available from 2003 to 2006, and are described in Section 7.2.

5.2.1.2 AEMP Version 2.0 Data (2007 to 2011)

From 2007 to 2010, samples were collected under *AEMP Study Design Version 2.0* (DDMI 2007). From 2007 to 2010, nutrient samples from NF, MF and FF2 areas were collected from three depths (top, middle and bottom) during the ice-cover season. In the FF areas (i.e., FF1, FFA, and FFB), water samples were collected from the middle of the water column. During the open-water season, depth-integrated water samples were collected for nutrients, chlorophyll *a*, and phytoplankton biomass. Depth-integrated water samples for nutrients were collected to provide an estimate of the concentrations of nutrients in the water column to which phytoplankton are exposed. Depth-integrated samples were collected from the top 10 m of the water column.

The nutrient samples were analyzed by three analytical laboratories during 2007 to 2010. Total and dissolved forms of phosphorus, nitrogen, and SRP were analyzed by the Freshwater Institute (Fisheries and Oceans Canada [DFO]), Winnipeg, Manitoba in 2007. In 2008, nutrients were analyzed by DFO and by the Biogeochemical Analytical Laboratory at the University of Alberta (UofA), Edmonton, Alberta. In 2009, 2010 and 2011, all nutrients were analyzed by UofA. Inorganic forms of nitrogen were analyzed by ALS, Edmonton, Alberta, in 2007 and 2008 and by UofA from 2009 to 2011. Chlorophyll *a* and phytoplankton biomass were analyzed by UofA and Bio-Limno Research and Consulting Ltd., respectively, during all five years.

In 2009, there was a laboratory error in the nutrient data set, whereby particulate phosphorus values for the September samples were not corrected by the appropriate subtraction factor of 0.86 µg/L. Therefore, in the 2009 annual AEMP report, TP values were inflated by a factor of 0.86 µg/L. This error was corrected in the *2007 to 2010 AEMP Summary Report* (Golder 2011a), and the corrected data were used for analyses presented in subsequent re-evaluation reports and herein.

The 2007 zooplankton biomass (as AFDM) data were excluded from the data analysis due to subsampling errors, which prevented accurate calculation of zooplankton biomass (DDMI 2008). From 2008 to 2011, zooplankton samples were collected starting from 1 m above the sediment, extending up through the water column. During the 2008 AEMP field program, zooplankton biomass samples were mistakenly collected from a 10 m depth to the surface, rather than starting from the bottom of the water column; these samples were removed from data analysis (Table 5-2). From 2008 to 2011, zooplankton biomass (as AFDM) was analyzed by HydroQual, Calgary, Alberta (now Nautilus Environmental).

Table 5-2 Zooplankton Samples Removed from Data Analysis as a Result of Field Sampling Errors in 2008

Area	n	Date	Depth (m)
NF	7	15 Aug to 15 Sept 2008	10
MF1	4	17 Aug to 1 Sept 2008	10
MF2-FF2	4	1 Sept to 4 Sept 2008	10
MF3	2	3 Sept 2008	10
FF1	3	5 Sept to 6 Sept 2008	10
FFB	5	22 Aug to 26 Aug 2008	10
FFA	6	21 Aug to 12 Sept 2008	10

NF = near-field; MF = mid-field; FF = far-field.

5.2.1.3 AEMP Version 3.0 (2012 to 2013)

From 2011 to 2013, samples were collected under *AEMP Study Design Version 3.0* (Golder 2011b). Sampling locations and methods for nutrients, chlorophyll *a*, phytoplankton biomass, and zooplankton biomass (as AFDM) were consistent with *AEMP Study Design Version 2.0*, except that a single open-water sampling season was sampled. Samples collected during the 2012 and 2013 ice-cover seasons were sent to UofA for analysis of total and dissolved phosphorus and nitrogen, SRP, total ammonia, and N+N. Depth-integrated samples collected during the open-water season in 2012 were sent to UofA, while open-water samples collected in 2013 were sent to Maxxam Analytics Inc. (Maxxam), Burnaby, British Columbia, for analysis of the same suite of nutrient parameters. Chlorophyll *a* was analyzed by UofA, and zooplankton biomass (as AFDM) was analyzed by HydroQual (now Nautilus Environmental) during all three years. Phytoplankton biomass was analyzed by Bio-Limno Research and Consulting Ltd., in 2012, but in 2013, a change in taxonomist was made and the phytoplankton biomass samples were sent to Eco-Logic Ltd. (Eco-Logic), Vancouver, British Columbia (see Section 7.2 for details).

5.2.1.4 AEMP Version 3.5 (2014 to 2016)

From 2014 to 2016, samples were collected for the Eutrophication Indicators program under *AEMP Study Design Version 3.5* (Golder 2014a). Sampling methods were consistent with the *AEMP Study Design Version 2.0*, with the exception that a single sampling period was identified from 15 August to 15 September, rather than three open-water sampling events. During the interim sampling years (i.e., 2014 and 2015), nineteen stations located within four general areas of Lac de Gras were sampled. Sampling areas were selected based on exposure to the Mine effluent (Golder 2014a), and consisted of the NF area and three MF areas. In 2016, a comprehensive sampling program was completed and included sampling 34 stations within seven general areas of Lac de Gras, a single station at the outlet of Lac de Gras (LDG-48) and three stations in Lac du Sauvage.

Depth-integrated chlorophyll *a* samples were analyzed by UofA, and phytoplankton samples were analyzed by Eco-Logic from 2014 to 2016. Nutrient and zooplankton biomass (as AFDM) samples were analyzed by

Maxxam. Sampling for zooplankton biomass (as AFDM) in 2014, 2015 and 2016 under the *AEMP Study Design Version 3.5* (Golder 2014a) was consistent with previous years and analysis was completed by HydroQual (now Nautilus Environmental) during all three years. In 2014, the zooplankton biomass samples were accidentally disposed of by the laboratory prior to analysis (see Appendix E of Golder 2016b); therefore, no data are unavailable for 2014.

Differences in the net dimensions were noted in 2017 among sampling years, which required adjustment of the data for some years. Re-measurement of the net diameter in 2016 determined that the zooplankton net used in 2014, 2015 and 2016 had a mouth diameter of 30.0 cm; therefore, recalculation of the zooplankton community biomass data was necessary for 2015 and 2016. The re-calculation increased the overall zooplankton biomass in 2015 and 2016 by 12.9% and 6.6%, respectively, as the volume of water actually sampled was smaller than that used in the 32 cm and 31.0 cm net diameter calculation (Table 5-3). These adjustments do not affect the conclusions reported in the respective annual reports.

Table 5-3 Corrected Zooplankton Biomass, 2014 to 2016

Year	Original Net Diameter Used in Annual Report (cm)	Adjusted Net Diameter (cm)	Percent Adjustment to the Overall Dataset (%)
2014	30.0	-	-
2015	32.0	30.0	12.9
2016	31.0	30.0	6.6

= not applicable, no adjustment made.

In response to a commitment following review of the *AEMP Design Plan Version 4.0*, phytoplankton biomass data were incorporated into the Eutrophication Indicators component in 2016 and herein. It is included in the evaluation of spatial extent of effects, WOE effects ratings, Action Level assessment and temporal trend analysis. Results of the Plankton component were also considered when interpreting the results of the Eutrophication Indicators component, by including trend analysis results for selected plankton variables (total phytoplankton and total zooplankton biomass), and evaluating concordance between Plankton and Eutrophication Indicators results.

In addition, in response to the WLWB Directive and Reasons for Decision re. W2015L2-0001 Schedule 8 Update, DDMI committed to include data from LDG-48 in future eutrophication analyses. In response to this commitment, nutrient and chlorophyll *a* (open-water season only) data from LDG-48 were incorporated into the temporal trend analysis presented in Section 5.3.4

5.2.2 Data Handling

5.2.2.1 Data Screening

Initial screening of the annual nutrient, chlorophyll *a*, phytoplankton biomass (Section 7.2) and zooplankton biomass (as AFDM) datasets was completed before data analyses to identify unusually high (or low) values, and decide whether to retain or exclude anomalous data from further analysis (Appendix 5A). The anomalous data screening approach for AEMP component datasets is described in Appendix A of the Eutrophication Indicators Reports in Support of the 2014 through 2016 AEMP Annual Reports (Golder 2016a,b,c2017e), and was approved as part of the *2011 to 2013 Aquatic Effects Re-evaluation Report*

(Golder 2016a). Anomalous values that were excluded from further analysis are presented in Table 5A-1. In cases where unusual values were identified in the dataset, scatterplots were generated (using R Version 3.2.3; R Core Team 2015) to allow a visual review of anomalous data and provide transparency (Figures 5B-1).

5.2.2.2 Censored Data

For the purposes of the AEMP, censored data are concentrations reported below the analytical DL (referred to as non-detect values). Due to the location of Lac de Gras on the Canadian Shield, concentrations of many water quality and nutrient variables are low and at or below the DL. Prior to data analyses, duplicate data were averaged and non-detect values were multiplied by 0.5 to achieve a value of half the DL. Substitution with half the DL is a common approach used to deal with censored data (US EPA 2000) and is consistent with the approved methods applied in the calculation of the normal range in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b).

5.2.2.3 Quality Assurance/Quality Control

The *Quality Assurance Project Plan Version 2.0* (Golder 2017a), or QAPP, outlined the QA/QC procedures employed to support the collection of scientifically-defensible and relevant data required to meet the objectives of the *AEMP Study Design Version 3.5* (Golder 2014a). Quality assurance project plans were designed so that field sampling, laboratory analysis, data entry, data analysis, and report preparation activities that follow these plans produce technically sound and scientifically defensible results. Descriptions of the QA/QC programs and the quality control (QC) results for the eutrophication indicators datasets are provided in the annual reports.

5.2.3 Data Analysis

5.2.3.1 Normal Range

The magnitude of effects on indicators of eutrophication were evaluated by comparing nutrient concentrations, chlorophyll *a*, and phytoplankton and zooplankton biomass in the NF, MF and FF areas to background values. Background values for Lac de Gras are those that fall within the range of natural variability, referred to as the normal range. Normal ranges were calculated using data from three AEMP FF areas (FF1, FFA, and FFB) from 2007 to 2010 (with some exceptions). The normal ranges were obtained from the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b) and are summarized in Table 5-4.

The WLWB requested that as part of the *2014 to 2016 Aquatic Effects Re-evaluation Report*, an assessment of the reference conditions for the phytoplankton variables be examined (Table 1-1, Section 7). The assessment includes a comparison of the AEMP results to reference conditions as defined using the currently approved 2007 to 2010 reference area data (Golder 2017b), and the 2013 reference area data. Both normal ranges are presented herein.

Table 5-4 Normal Ranges for Eutrophication Indicators

Variable	Unit	Normal Range			
		Ice-Cover		Open-water	
		Lower Limit	Upper Limit	Lower Limit	Upper Limit
Total phosphorus	µg-P/L	2.0	5.0	2.0	5.3
Total dissolved phosphorus	µg-P/L	1.1	3.2	0	3.5
Soluble reactive phosphorus	µg-P/L	0	1.5	0	1.0
Total nitrogen	µg-N/L	137.7	172.5	121.5	152.7
Total dissolved nitrogen	µg-N/L	130	166	105	133
Total ammonia	µg-N/L	11	17	0	6
Nitrate + nitrite	µg-N/L	5	10	0	1
Chlorophyll *a*	µg/L	-	-	0.31	0.82
Phytoplankton biomass (2007 to 2010)	mg/m^3	-	-	140	352
Phytoplankton biomass (2013)	mg/m^3	-	-	79	326
Zooplankton biomass as AFDM	mg/m^3	-	-	16.4	40.5

Source: AEMP Reference Conditions Report Version 1.2 (Golder 2017b).

AFDM = ash-free dry mass; µg-P/L = micrograms phosphorus per litre; µg-N/L = micrograms nitrogen per litre; mg/m^3 = milligrams per cubic metre; - = not applicable.

5.2.3.2 Temporal Trends

To visually evaluate temporal trends in Lac de Gras, including at LDG-48, Secchi depth, nutrients, chlorophyll *a* and zooplankton biomass (as AFDM), were plotted against time (years). The time series plots included a shaded region showing the normal range obtained from the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b).

In response to the EMAB commitment #11, from the WLWB Board Directive and Reasons for Decision W2015L2-0001 Schedule 8 Update (Table 1-1), LDG-48 was included in eutrophication temporal trend plots and analyses. In addition, as directed by WLWB following EMAB commitment #5 from the *Design Plan Version 4.0* (Table 1-1), phytoplankton biomass was incorporated into the Eutrophication Indicators component, and was compared to the normal range for all years and plotted as boxplots for each area and year.

5.2.3.3 Temporal Trend Analysis

The general methods used for statistical analysis of AEMP data are described in Section 2.4.2. The information provided in the following text describes details specific to the statistical trend analysis used in the Eutrophication Indicators section.

Data below the analytical DL were considered censored. High presence of censored data can potentially bias results based on parametric analyses, due to violation of underlying assumptions. Based on USEPA guidance, variables with ≥15% of the data within a sampling area as censored were examined to determine an alternative analysis method. Analytical approach was determined separately for each variable within each season.

The majority of the Eutrophication Indicators variables had sufficient detected data to proceed with parametric analysis. For SRP and TDP (in both IC and OW seasons), logistic modelling was used, since below DL values were recorded in multiple years and areas throughout the sampling program, and it was the overall trends that were of interest. In this analysis, a logistic model predicted changes in the odds of observing data above the DL as a function of year and area. Simple, fixed-effects additive models, without interactions between year and area, random effects, weights, or autocorrelation terms were used to allow model convergence. Data were expressed as presence or absence of detected data. Only one model was constructed for each variable or season combination, where the year effect was represented as a linear trend, to prevent spurious preference of parabolic effects due to scarce data. Model outputs included parameter significance, whether linear slopes were significantly different from zero, and plots of predicted probabilities overlaying the presence or absence of detected data.

The temporal trend analysis for the phytoplankton biomass dataset differed from the nutrients, chlorophyll *a* and zooplankton biomass (as AFDM) datasets, because of the change in phytoplankton taxonomists in 2013 (Section 7.2). The analysis of the 2002 to 2012 phytoplankton dataset proceeded as outlined in Section 2.4.2. The 2013 to 2016 phytoplankton data were analyzed using a separate modeling approach. The full four years of data were available for the NF area; however, only two years of data were available for the MF and FF areas. Therefore, the general methods that used year as a continuous covariate to examine temporal trends could not be applied. Instead, the trend model described in Section 2.4.2 was simplified into a fixed effects model of only the NF data, with year as a single, discrete covariate. No model selection was performed, since there was only one candidate model for analysis (i.e., one-way analysis of variance [ANOVA]). Data transformation using Yeo-Johnson transformations were performed as described in Section 2.4.2. Residual heteroscedasticity was examined, and heteroscedasticity effects by year or predicted value were added to the model if they improved model fit (i.e., as identified using AIC scores).

5.2.3.4 Extent of Effects

The area of the lake with values greater than the normal range was estimated for each variable (i.e., TP, TN, chlorophyll a, phytoplankton biomass, and zooplankton biomass [as AFDM]), and this measure was used to estimate the extent of effects. The extent of effects estimated for each year were compared to evaluate whether effects are expanding over time. To provide the most conservative view of effluent effects, the season and depth with the greatest extent of effects was selected for this evaluation.

The extent of eutrophication effects on phytoplankton biomass was not previously calculated or presented in the annual reports before 2016. As directed by the WLWB as part of the Board Directive and Reasons for Decisions re. W2015L2-001 Schedule 8 Update, and as part of EMAB commitment #5 from the *Design Plan Version 4.0*, and commitment #49 from the *2016 AEMP Annual Report* (Table 1-1), the extent of effects on phytoplankton community biomass was estimated and presented to visually evaluate spatial trends for all years from 2007 to 2016.

To quantify the extent of effects along each transect, a linear interpolation method was used to estimate the distance between the station farthest from the diffuser with a value greater than the normal range and the adjacent station with a value below the normal range. In cases where concentrations did not decrease uniformly with distance from the diffuser, a conservative approach was taken by assuming that the effect extended to the farthest station with a concentration above the normal range, even if closer stations along the transect had concentrations below the normal range. For example, in 2015, concentrations of chlorophyll *a* at stations MF2-3 and FF2-2 along the NF-MF2-FF2 transect did not exceed the upper boundary of normal range. However, since concentrations at these stations were near the upper boundary, and chlorophyll *a* at FF2-5 was above the normal range, a conservative approach was taken by assuming that the effect extended along the full length of the NF-MF2-FF2 transect.

The one exception to this conservative approach was if there were three consecutive stations below the normal range, which showed a decline in concentration with distance from the diffuser, followed by a single station above the normal range. Under such circumstances, the elevated concentration at that distant station was assumed to be unrelated to the Mine. This situation arose in 2016 for the top-depth TP data collected during the ice-cover season. Concentrations of TP at MF2-1, MF2-3, and FF2-2 were all below the normal range, while concentrations of TP were greater than the normal range at the farthest station from the diffuser along the NF-MF2-FF2 transect (FF2-5). In this case, the NF-MF2-FF2 transect was considered unaffected by TP in 2016 because there was no evidence of a Mine-related effect based on the observed spatial trend.

5.2.3.5 Nutrient Relationships

The WLWB has requested that as part of the *2014 to 2016 Aquatic Effects Re-evaluation Report* the Eutrophication Indicators component include:

- an evaluation of the relationships between nutrients, chlorophyll *a* and phytoplankton biomass (EMAB commitment #11; WLWB Directive and Reasons for Decisions W2015L2-0001 Schedule 8 Update; and Commitment #47 from 2016 AEMP Annual Report [Table 1-1])
- an evaluation of the role of TN in explaining variation in chlorophyll *a* concentrations (EMAB commitment #61; WLWB Directive and Reasons for Decisions W2015L2-0001 Schedule 8 Update [Table 1-1])
- continue to evaluate the relationship between chlorophyll *a* concentrations and phytoplankton biomass (EMAB commitment #9 from the 2015 AEMP Annual Report [Table 1-1])
- nutrient ratios and evaluate trends over time (EMAB commitment #61; WLWB Directive and Reasons for Decisions W2015L2-0001 Schedule 8 Update; and (EMAB commitment #25 from the 2015 AEMP Annual Report [Table 1-1])
- an evaluation of the role of TN and the ratio of nitrogen:phosphorus (N:P) in explaining variation in phytoplankton biomass (Section 7) and chlorophyll *a* concentrations (EMAB commitments #61 and 94; WLWB Directive and Reasons for Decisions W2015L2-0001 Schedule 8 Update [Table 1-1])

In addition, as directed by WLWB following the *Design Plan Version 4.0*, key findings from the Plankton section were incorporated, where appropriate (Table 1-1).

Relationships between TN, TP, TDS, N:P and chlorophyll a concentrations were explored for each year between 2007 and 2016. The relationships between TN, TP, TDS, N:P and phytoplankton biomass was examined in Section 7 (Plankton).

As a result of quality control issues with the TDS data (i.e., in 2016; Golder 2017c), the relationship between specific conductivity from previous years ($y = 0.5x – 0.94$) was used to estimate TDS. Specific conductivity and TDS are strongly correlated ($r^2 = 0.94$) and both variables can be used as tracers of Mine effluent.

Molar and mass nutrient ratios were calculated for each station in Lac de Gras and were plotted to evaluate temporal changes over time. Nutrient limitation was evaluated by comparing N:P calculated for Lac de Gras to those reported by Hecky et al. 1993, and Redfield (as described in Wetzel 2001).

The trophic status of each waterbody was also evaluated by examining the nutrient concentrations, chlorophyll *a*, and water transparency (Secchi depth). The trophic status was determined using the Vollenweider (1968) trophic classification scheme for lakes (using TP, TN, chlorophyll *a*, and Secchi depth), the Canadian Council of Ministers of the Environment (CCME 2004) trophic classification scheme for Canadian lakes and streams (using TP), and the Trophic State Index (TSI) developed by Carlson (1977). A description of the classification systems and the numerical scales for each of the trophic status indices was presented in the *AEMP Study Design Version 3.0.*

The TSI is a numerical trophic state index for lakes with a scale from 0 to 100 (Carlson 1977). Each major division (e.g., 10, 20, 30…) represents a doubling in algal biomass. The index number was generated as a rounded average from the TSI values calculated from Secchi depth, chlorophyll *a*, and TP measurements, using the following equations (Carlson 1977):

$$TSI\ (TP) = 10\left(6 - \frac{\ln \frac{48}{TP}}{\ln 2}\right)$$

$$TSI\ (Chl) = 10\left(6 - \frac{2.04 - 0.68 \ln Chl}{\ln 2}\right)$$

$$TSI\ (Secchi) = 10\left(6 - \frac{\ln SD}{\ln 2}\right)$$

where:

TSI = trophic state index;
TP = total phosphorus;
ln = natural logarithm;
Chl = chlorophyll a;
SD = standard deviation; and,
Secchi = Secchi depth.

The values are multiplied by 10 to give the scale a range of 0 to 100.

An averaged TSI from the indices calculated from Secchi depth, and chlorophyll *a* and TP concentrations was used to determine a general trophic classification. An averaged TSI from the indices calculated from Secchi depth, and chlorophyll *a* and TP concentrations was used to determine a general trophic classification.

In addition, a multivariate comparison of the TSIs for TP, Secchi depth, and chlorophyll *a* was completed following the approach described in Carlson and Simpson (1996).

5.2.3.1 Nutrients in Effluent and the Mixing Zone

The quantity of nutrients contributed by effluent was evaluated graphically by plotting annual loads of nutrients. The total annual loads represent the sum of the total monthly loads for a given year from 2002 to 2016. Water was sampled at the mixing zone boundary monthly, at five depths (2, 5, 10, 15, and 20 m) at each of the three mixing zone stations (i.e., SNP 1645-19A, SNP 1645-19B, SNP 1645-19C). The results for the mixing zone were summarized by showing the 5th percentile, median, and 95th percentile concentrations in each month.

The quality of the effluent was assessed in Section 4 by comparing water chemistry results at stations SNP 1645-18 and SNP 1645-18B with the EQC defined in the Water Licence (WLWB 2015b). Results for key nutrient variables are presented in this report, including: TP, TDP, SRP, TN, total ammonia, nitrate, and nitrite. Total phosphorus has an EQC specified in terms of load, rather than concentration. The Water Licence specifies that the load of TP must not exceed a maximum of 300 kg/mo, an average annual load of 1,000 kg/yr during the life of the Mine, and a maximum load of 2,000 kg/yr in any year during the life of the Mine.

5.2.3.2 Effects from Dust Deposition and Dike Construction

Concerns have been raised regarding the potential for dust emissions to affect water quality and nutrient concentrations in Lac de Gras. To address these concerns, in 2016, an analysis of effects at stations potentially affected by dust emissions was conducted, as directed by the WLWB (Table 1-1) and as per the *AEMP Design Plan Version 4.0* (Golder 2016d). The zone of influence from dust deposition in Lac de Gras is estimated to be approximately 4 km from the geographic centre of the Mine, or approximately 1 km from the Mine boundary, extending radially from the source. These distances were estimated based on gradient analysis of dust deposition relative to distance from the Mine site, and encompass the area of the lake where potential effects would be expected to be measurable (see Figures 3-5 and 3-6 and Table 3-1 in Golder 2016d). Beyond this estimated zone, dust deposition levels are similar to background levels. The AEMP sampling stations that fall within the expected zone of influence from dust deposition include the five stations in the NF area and stations MF1-1, MF2-1, MF3-1 and MF3-2.

To assess potential effects from dust emissions, phosphorus concentrations at stations within the estimated zone of influence from dust deposition were evaluated graphically and compared to results at other nearby stations and to reference conditions for Lac de Gras (as defined in the *AEMP Reference Conditions Report Version 1.2* [Golder 2017b]). This comparison was only done on the open-water season data, because dust deposition to lake water under ice (where samples are collected) is prevented by ice cover during winter. If phosphorus concentrations at the potentially dust-affected stations (i.e., MF1-1, MF2-1, MF3-1 and MF3-2) were elevated beyond the expected range (i.e., the normal range) based on exposure to effluent alone, this may indicate a potential additional effect from dust deposition, or in 2015 and 2016, dike construction.

5.2.3.3 Weight-of-Evidence Effect Ratings

The results of the AEMP indicators of eutrophication surveys are integrated through the WOE evaluation process, which determines the strength of evidence supporting the two broad impact hypotheses for Lac de Gras (i.e., toxicological impairment and nutrient enrichment), as described in the *AEMP Study Design Version 3.5* (Golder 2014a). The WOE is not intended to determine the ecological significance or level of concern associated with a given change. The WOE effect ratings incorporate statistical comparisons of the

NF and FF areas, and comparisons of the NF area to the normal range as defined in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b). The indicators of eutrophication data from 2014 and 2015 were not assessed following the *AEMP Study Design Version 3.5* (Golder 2014a) because only NF and MF area data were collected in those years. The indicators of eutrophication data were assessed according to the WOE effect level ratings described in Section 10 and summarized in Table 5-5.

Table 5-5 Weight-of-Evidence Effect Ratings for Eutrophication Indicators

LOE Group	Measurement Endpoint Analysis	Early Warning/ Low ↑/↓	Moderate ↑↑/↓↓	High ↑↑↑/↓↓↓
Water Quality (nutrients)	Comparison to Normal Range[(a)] Total phosphorus Total nitrogen	NF area mean greater than normal range	Less than or equal to 20% of the lake area with concentrations greater than the normal range	Concentrations in more than 20% of the lake area greater than the normal range
Biological Productivity	Comparison to FF Areas and Normal Range[(a)] Chlorophyll *a*[(b)] Zooplankton biomass (AFDM) Phytoplankton biomass	Statistically significant change, NF vs FF areas	Low + NF area mean outside normal range	Moderate + values in more than 20% of the lake area either greater or lower than the normal range

Notes: Normal ranges for each LOE group and measurement endpoint are defined and provided in the *AEMP Reference Conditions Report, Version 1.2* (Golder 2017b).

a) Applied separately for each measurement endpoint.

b) Chlorophyll *a* is interpreted both as an exposure and a biological response endpoint.

LOE = Line of Evidence; NF = near-field; FF = far-field; AFDM = ash-free dry mass.

5.2.3.4 Action Levels

The severity of possible effects to an assessment endpoint was categorized according to the Response Framework described for eutrophication indicators in the *AEMP Study Design Version 3.5* (Golder 2014a). The Action Level classifications were developed to meet the goals of the draft *Guidelines for Adaptive Management – A Response Framework for Aquatic Effects Monitoring* (WLWB 2010) and Racher et al. (2011). The main goal of the Response Framework is to ensure that significant adverse effects never occur. This is accomplished by requiring proponents to take actions at predefined Action Levels, which are triggered well before significant adverse effects could occur.

Termed a Significance Threshold in the Action Levels, a significant adverse effect for TP was defined in the EA (Government of Canada 1999). The magnitude of effect for TP at the Significance Threshold level was defined as a concentration that exceeds the EA threshold by more than 20% (i.e., 116 km^2 of the surface area of Lac de Gras). The Significance Threshold for the indicators of eutrophication is a concentration of chlorophyll *a* that exceeds the Effects Threshold by more than 20% in the FFA area of Lac de Gras (Golder 2014a). In contrast to toxicological impairment responses in water chemistry (e.g., from elevated concentrations of metals), the eutrophication responses are difficult to link to nutrient concentrations. As demonstrated by years of monitoring in Lac de Gras, concentrations of TP do not predict the actual biological response to nutrient enrichment. Rather, the increase in the biomass of algae as measured by chlorophyll *a* has been a useful measure of the effects of nutrient enrichment.

Elevated concentrations of nutrients were predicted in Lac de Gras (Government of Canada 1999). Specifically, up to 20% (i.e., 116 km^2) of the surface area of Lac de Gras was expected to exceed the EA threshold for TP during peak operations during the open-water season, and up to 11% (i.e., 64 km^2) of the lake during the ice-cover season. Outside these areas, TP concentration was predicted to increase relative to baseline in parts of Lac de Gras, but concentrations would remain below the EA threshold. The “extent of effect” for the chlorophyll *a* Action Levels reflects this prediction (Table 5-6).

Table 5-6 Action Levels for Chlorophyll *a*

Action Level	Magnitude of Effect	Extent of Effect	Action/Notes
1	95th percentile of MF values greater than normal range[(a)]	MF station	Early warning.
2	NF and MF values greater than normal range[(a)]	20% of lake area or more	Establish Effects Benchmark.
3	NF and MF values greater than normal range plus 25% of Effects Benchmark[(b)]	20% of lake area or more	Confirm site-specific relevance of existing benchmark. Establish Effects Threshold.
4	NF and MF values greater than normal range plus 50% of Effects Threshold[(c)]	20% of lake area or more	Investigate mitigation options.
5	NF and MF values greater than Effects Threshold	20% of lake area or more	The WLWB to re-assess EQC for phosphorus. Implement mitigation required to meet new EQC if applicable.
6	NF and MF values greater than Effects Threshold +20%	20% of lake area or more	The WLWB to re-assess EQC for phosphorus. Implement mitigation required to meet new EQC if applicable.
7	95th percentile of MF values greater than Effects Threshold +20%	All MF stations	The WLWB to re-assess EQC for phosphorus. Implement mitigation required to meet new EQC if applicable.
8	95th percentile of FFB values greater than Effects Threshold +20%	FFB	The WLWB to re-assess EQC for phosphorus. Implement mitigation required to meet new EQC if applicable.
9[(d)]	95th percentile of FFA values greater than Effects Threshold+20%	FFA	Significance Threshold[(d)].

a) The normal range for chlorophyll *a* was obtained from the *AEMP Reference Conditions Report, Version 1.2* (Golder 2017b).

b) Indicates 25% of the difference between the Effects Benchmark and the top of the normal range.

c) Indicates 50% of the difference between the Effects Threshold and the top of the normal range.

d) Although the Significance Threshold is not an Action Level, it is shown as the greatest Action Level to demonstrate escalation of effects towards the Significance Threshold.

NF = near-field; MF = mid-field; FFA = far-field A; FFB = far-field B; WLWB = Wek'èezhìı Land and Water Board; EQC = Effluent Quality Criteria.

An Action Level 2 for chlorophyll *a* was triggered in previous years (Golder 2016b); therefore, an Effects Benchmark for chlorophyll *a* was developed. The chlorophyll *a* Effects Benchmark concentration of 4.5 µg/L (Golder 2014a) is appropriate in terms of both the aesthetic quality and food web functionality in Lac de Gras. Aesthetic qualities are likely to be preserved at chlorophyll *a* concentrations of up to 10 µg/L, while a benchmark of 4.5 µg/L maintains the trophic classification of the lake as oligotrophic (Golder 2014a).

5.3 Results

5.3.1 Summary of Effects

5.3.1.1 Extent of Effects

For the *2014 to 2016 Re-evaluation Report*, the extent of effects for phytoplankton biomass was calculated for all years. Since only open-water samples were collected for chlorophyll *a*, phytoplankton and zooplankton biomass, the open-water data were used for these variables to estimate the spatial extent of effects. For TP and TN, the season and depth representing the greatest extent of effects for each year were presented.

For all indicators of eutrophication, the spatial extent of effects (i.e., increases above the normal range) was highly variable among years (Figures 5-1 to 5-5; Table 5-7). The spatial extent of effects on TP concentrations was greatest in 2008, estimated as 112 km^2 or 19.6% of the lake area, followed by 2013 (14%) and 2009 (9.3%). In other years, the extent of effects on TP has been low (<0.6 to 6.5%), and has extended northeast towards the Lac du Sauvage inflow and northwest along the NF-MF1-FF1 transect (Figure 5-1; Table 5-7).

Overall, the greatest extent of effects was observed on TN in 2016 (484.9 km^2, or 84.7% of lake area) (Figure 5-2; Table 5-7). The extent of effects on TN increased between 2007 and 2016, and has consistently shown an affected area >20% since 2008, while the spatial extent of effects on other indicators of eutrophication (i.e., TP, phytoplankton and zooplankton biomass) has decreased. The lack of a relationship between areas where TN is greater and areas where biological effects were observed (chlorophyll *a*, phytoplankton and zooplankton biomass) is consistent with N not being the limiting nutrient in Lac de Gras. The boundary of effects on concentrations of TN generally extends to the northwest (to the end of the NF-MF1-FF1 transect) and to the northeast (towards the Lac du Sauvage inlet), with an exception in 2014 when the extent of effects appeared to be localized around the NF area. In 2015 and 2016, the extent of effects included the MF1 and MF2-FF2 areas and to the south of the Mine; the boundary extended to Station MF3-7 in 2015, and to the end of the NF-MF3-FFB-FFA transect (i.e., FFA-5) in 2016. The southernmost arm of the lake was also considered affected in 2016; however, there is uncertainty as to whether the extent of effects on TN extended beyond FFA-5, or into the northern and southern bays (Golder 2017c).

In all years with available monitoring data up to 2016, there was no indication of additional sources of N interacting with the spread of effects on N concentration from the Diavik discharge. The only other known source of N to the lake is the inflow of Ekati operation-affected water to Slipper Bay. However, the TN effect from Diavik has extended to the FFA area for the first time in 2016.

Similarly, the greatest extent of effects on chlorophyll *a* concentrations was observed in 2016 (250.4 km^2, or 43.7% of lake area), closely followed by 2014 (≥242.8 km^2, or ≥42.4% of lake area) (Figure 5-3; Table 5-7). The extent of effects on chlorophyll *a* concentrations also generally increased from 2007 to

2016, with the exception of 2012 (when only 3% of the lake was affected). The extent of effects on chlorophyll *a* concentrations did not extend as far as the effect on TN. The extent of effects on chlorophyll *a* concentrations along the NF-MF3-FFB-FFA transect did not extend beyond the MF3-7 station between 2007 and 2016, and generally extended to the northeast towards the Lac du Sauvage inlet and the northwest along the NF-MF1-FF1 transect.

The area affected for phytoplankton biomass was greater from 2008 to 2011 compared to more recent years (i.e., 2014 to 2016; Figure 5-4; Table 5-7). The greatest extent of effects on phytoplankton biomass was observed in 2009 (274.1 km^2, or 47.8% of lake area), followed by 2010 (216.9 km^2, or 37.9% of lake area). The boundary of effects on phytoplankton biomass generally extended to the northwest to station MF1-5 and into the FF1 area in 2008. The extent of effects to the northeast of the Mine often extended to the Lac du Sauvage inlet. The boundary of effects to the south of the Mine was not observed to extend beyond station MF3-2 in five of the eight years of monitoring; however, in 2009 it extended into the FFB area, in 2010 it extended past station MF3-6, and in 2011 it extended past station MF3-4.

The area affected for zooplankton biomass varied the most out of all the indicators of eutrophication (Figure 5-5; Table 5-7). The greatest extent of effects on zooplankton biomass was observed in 2013 (355 km^2, or 62.1% of lake area), followed by 2010 (22.5%), and 2011 (13.4%). Generally, the extent of effects on zooplankton biomass has been low (0 to 9%), and has extended northeast towards the Lac du Sauvage inlet, or northwest along the NF-MF1-FF1 transect. In 2013, the boundary of effects south of the Mine extended past the FFB area.

DDMI committed to the WLWB that it would discuss potential reasons for comparatively lower year-to-year spatial variability in the TN affected area compared to other indicators of eutrophication (commitment #29 from the *2015 AEMP Annual Report* [Table 1-1]). DDMI also committed to comment on the continued large spatial extent of effects on TN in comparison to the reduction of extent for other variables (i.e., TP, chlorophyll *a*, and zooplankton biomass), while discussing whether additional inputs affecting TN concentrations in Lac de Gras may be a contributing factor. These items are addressed below:

Lower year-to-year variability in the TN affected area compared to other indicators of eutrophication: Year-to-year variability in affected area was examined by calculating the coefficient of variation (CV) as a rough estimate of the variability observed in the percent affected area among years (Table 5-7, bottom row). The CVs suggest that the variation in affected area for TN (CV= 56%) is only slightly less than those in chlorophyll *a* (CV=65%) and phytoplankton biomass (CV=65%), but substantially less than those in TP (CV=103%) and zooplankton biomass (CV=144%). Spatial extent is calculated from concentration data, and higher analytical variability can be expected to contribute to greater variation in the estimated spatial extent of effects. TN is a variable incorporating all forms of N, and is calculated from concentrations of other N variables (i.e., TKN, N+N), most of which are measured in Lac de Gras at concentrations well above the analytical DLs. In contrast, TP and chlorophyll *a* are often measured at concentrations close to the DL, where analytical uncertainly is expected to be greater and contribute to variability in monitoring results. Therefore, based on this factor, lower year-to-year variability in spatial extent of effects is expected in TN compared to TP and chlorophyll *a*.

The concentration of TN is elevated in Lac de Gras, with most of the increase relative to background originating from one source (the Mine discharge), which has been contributing a similar load of TN during the last decade. This is expected to result in a gradually expanding affected area, with relatively low year-to-year variation.

Community level biological indicators (i.e., phytoplankton and zooplankton biomass) are typically subject to large spatial and temporal variability due to habitat variation and clumped spatial distribution, which may be further increased by variation introduced during field sampling and laboratory analysis.

Overall, substantially lower year-to-year variation in TN affected area was observed compared to TP and zooplankton biomass, while the variation in TN affected area was only slightly less than those for chlorophyll *a* and phytoplankton biomass. The lower year-to-year variation in TN affected area is as expected, and likely reflects low analytical variation in chemistry data, loading mostly from a single, constant source, and a lower influence of biological factors compared to other indicators.

- **Continued large spatial extent of effects on TN:** The annual TN load to Lac de Gras since 2009 (following a decline in loads relative to previous years) has ranged from approximately 25,000 to 45,000 kg/yr, compared to a TP load range of approximately 400 to 800 kg/yr during the same period. The large difference in loading rates between these nutrients explains the greater spatial extent of the effect on TN in Lac the Gras relative to TP. Additionally, P is the limiting nutrient and is utilized by plankton, which further reduces TP concentrations and the spatial extent of effects in the lake.
- **Additional inputs affecting TN concentrations in Lac de Gras:** Available information is insufficient to comment with certainty on the contributions of additional inputs affecting TN concentrations in Lac de Gras. Two Mine-related sources of TN to Lac de Gras exist: The Diavik mine and the Ekati mine, but their relative contributions, and resulting spatial patterns in TN concentration, have not been evaluated.

Table 5-7 Extent of Effects on Concentrations of Total Phosphorus, Total Nitrogen and Chlorophyll *a*, and on Phytoplankton and Zooplankton Biomass, 2007 to 2016

Year	Total Phosphorus		Total Nitrogen		Chlorophyll *a*		Phytoplankton Biomass		Zooplankton Biomass (AFDM)	
	Area (km^2)[(a)]	Lake Area (%)[(b)]	Area (km^2)[(a)]	Lake Area (%)[(b)]	Area (km^2)[(a)]	Lake Area (%)[(b)]	Area (km^2)[(a)]	Lake Area (%)[(b)]	Area (km^2)[(a)]	Lake Area (%)[(b)]
2007	29.4	5.1	-[(d)]	-[(d)]	89.0	15.5	67.1	11.7	-[(d)]	-[(d)]
2008	112[(c)]	19.6	84.8	14.8	77.1	13.5	116.3	20.3	-[(e)]	-[(e)]
2009	53.5[(c)]	9.3	180	31.5	121	21.0	274.1	47.8	0	0
2010	23.8[(c)]	4.2	132[(c)]	23.1	88.5	15.5	216.9	37.9	52.3	9.1
2011	9.2[(c)]	1.6	213[(c)]	37.2	89.3	15.6	124.5	21.7	129	22.5
2012	3.6[(c)]	0.6	118	20.7	17.0	3.0	67.4	11.8	76.7	13.4
2013	80.6[(c)]	14.1	183[(c)]	31.9	129	22.6	59.3	10.4	355	62.1
2014	3.5[(c,f)]	0.6[(f)]	≥229.6[(c,f)]	≥40.1[(f)]	≥242.8[(f)]	≥42.4[(f)]	-[(h)]	-[(h)]	-[(i)]	-[(i)]
2015	<3.5[(f,g,j)]	<0.6[(f,g,j)]	≥242.8[(c)]	≥42.4[(f)]	59.0[(f)]	10.3[(f)]	-[(h)]	-[(h)]	<3.5[(j)]	<0.6[(j)]
2016	37.1[(c)]	6.5	484.9	≥84.7[(k)]	250.4	43.7	74.6	13.0	2.9	0.5
CV (%)	-	103	-	56	-	65	-	64	-	144

a) Lake area reported is the greater of the area affected during the ice-cover and open-water seasons.

b) The lake area affected represents the percentage (%) of lake area experiencing levels greater than the normal range, and was calculated relative to the total surface area of Lac de Gras (573 km^2).

c) Lake area reported is for the ice-cover season.

d) Data not available due to field subsampling errors (Golder 2016a).

e) Data not available due to differences in sample collection procedures (Golder 2016a).

f) Percent lake area affected could not be estimated with certainty, because the FF1, FFA, and FFB areas were not sampled in 2014 and 2015.

g) Lake are reported for the open-water season.

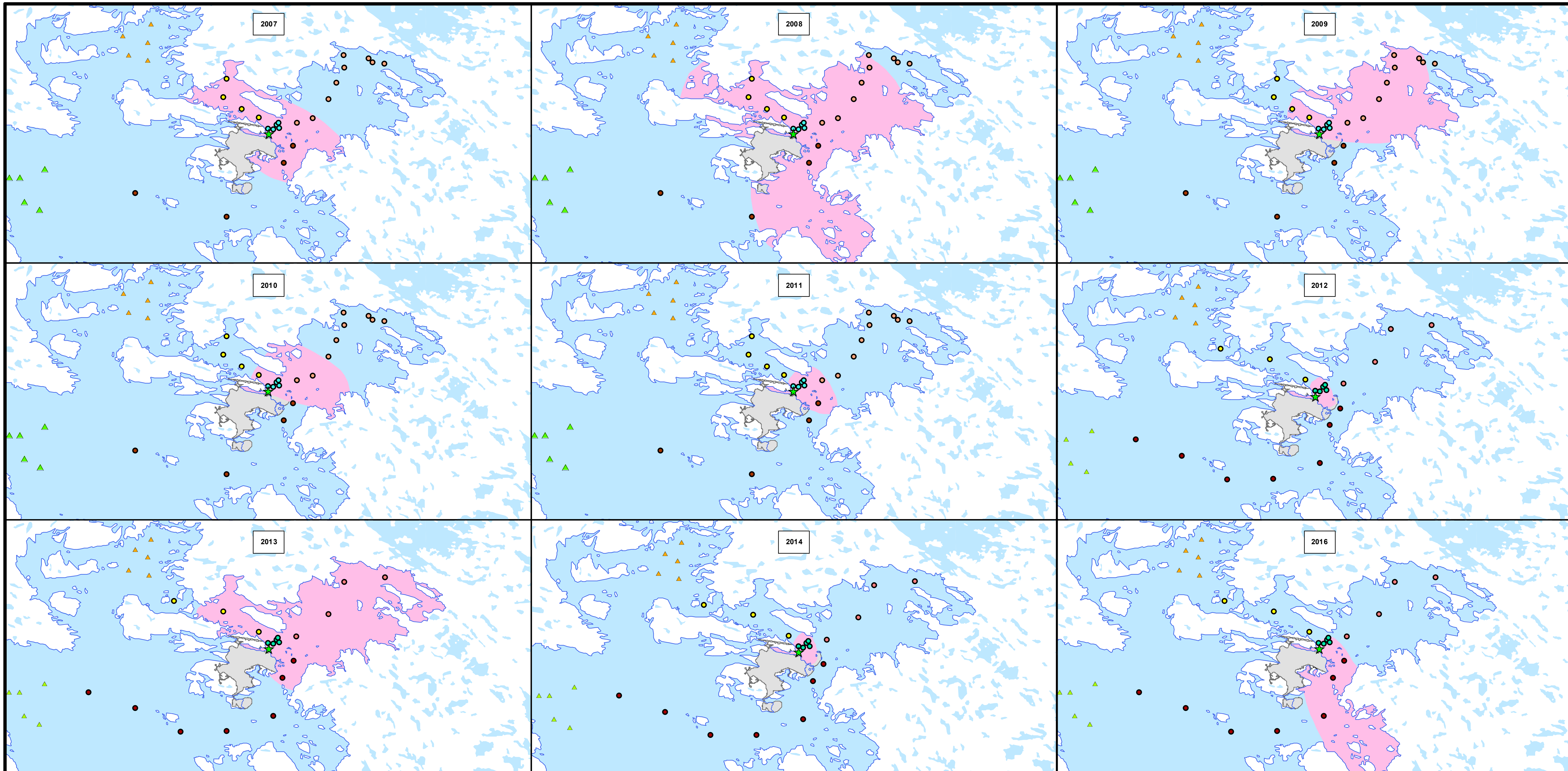
h) Only NF area sampled in 2014 and 2015; therefore, extent of effects was not calculated.

i) Data not available due to the loss of the zooplankton samples.

j) The mean of the NF area stations was within the normal range. Since only one or two NF stations exceeded the normal range, the affected area was assumed to be less than the total area of the NF area (0.6% of lake area).

k) Due to an uncertain effect boundary at the end of the NF-MF3-FFB-FFA transect, the extent of effects could have been greater than the area presented.

<= less than; ≥ = greater than or equal to; - = not determined; NF = near-field; FF1 = far-field 1; FFA = far-field A; FFB = far-field B; AFDM = ash-free dry mass; CV = coefficient of variation, calculated by setting lake areas to the numbers shown (i.e., by removing <or ≥ signs).



LEGEND

- ★ DIFFUSER

EXPOSURE

- NEAR-FIELD
- MID-FIELD 3
- MID-FIELD 1
- FAR-FIELD 2; MID-FIELD 2
- FAR-FIELD 1
- FAR-FIELD A
- FAR-FIELD B
- AFFECTED AREA
- DIAVIK FOOTPRINT
- LAC DE GRAS
- WATERBODY

REFERENCE

HYDROGRAPHY DATA OBTAINED FROM GEOGRATIS, © DEPARTMENT OF NATURAL RESOURCES CANADA.

PROJECTION: UTM ZONE 12 DATUM: NAD 83

0 10 20

1:300,000 KILOMETRES

PROJECT

DIAVIK DIAMOND MINES INC.

TITLE

TOTAL PHOSPHORUS AFFECTED AREA IN LAC DE GRAS, 2007 TO 2016

GOLDER

PROJECT		1771843.4000	FILE No.
DESIGN	KH	02 Jan. 2018	SCALE AS SHOWN / REV: 0
GIS	BP	14 Mar. 2018	
CHECK	KH	14 Mar. 2018	FIGURE: 5 - 1
REVIEW	ZK	14 Mar. 2018	

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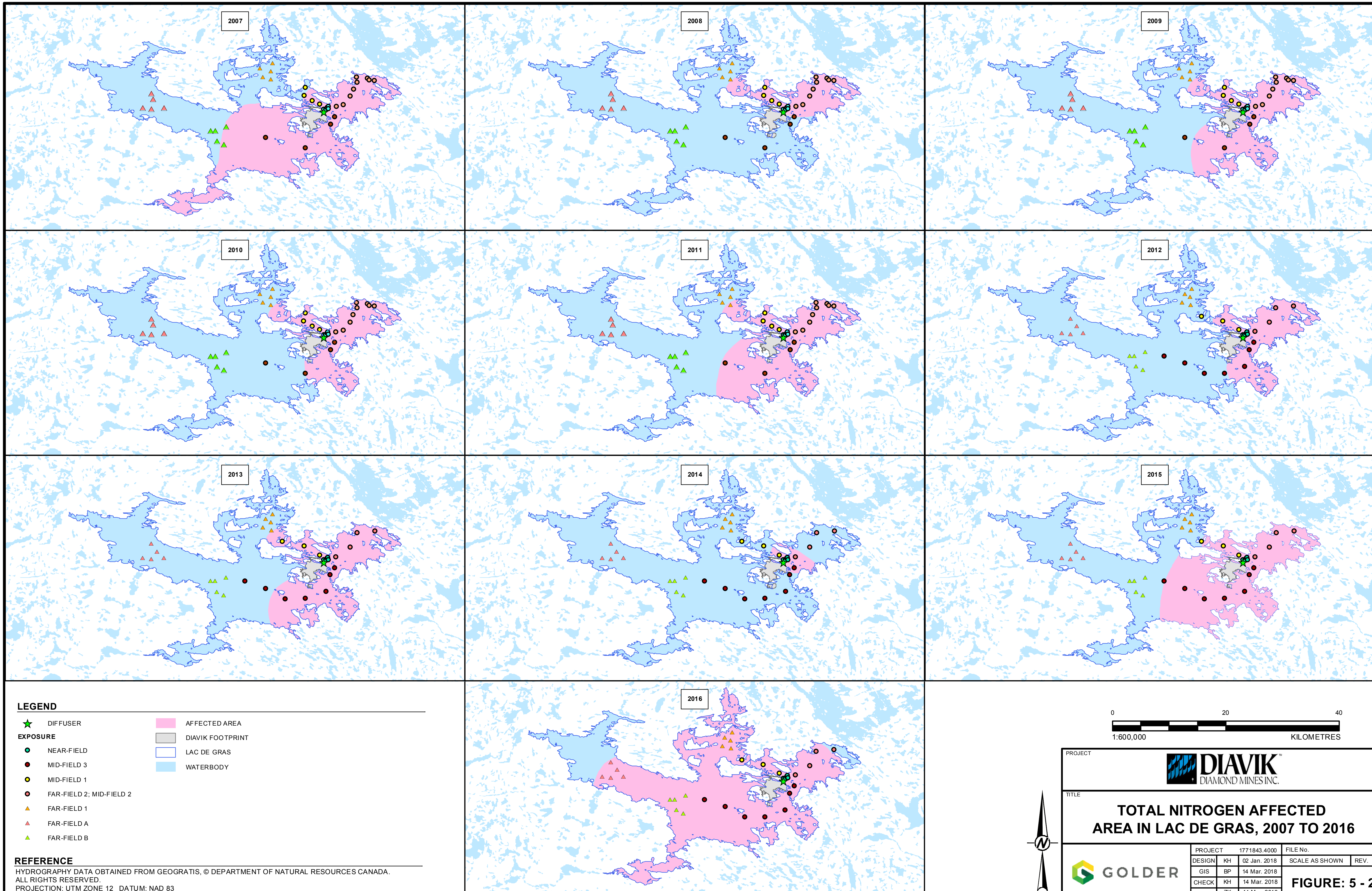

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REFERENCE

HYDROGRAPHY DATA OBTAINED FROM GEOGRATIS, © DEPARTMENT OF NATURAL RESOURCES CANADA.

PROJECTION: UTM ZONE 12 DATUM: NAD 83

PROJECT

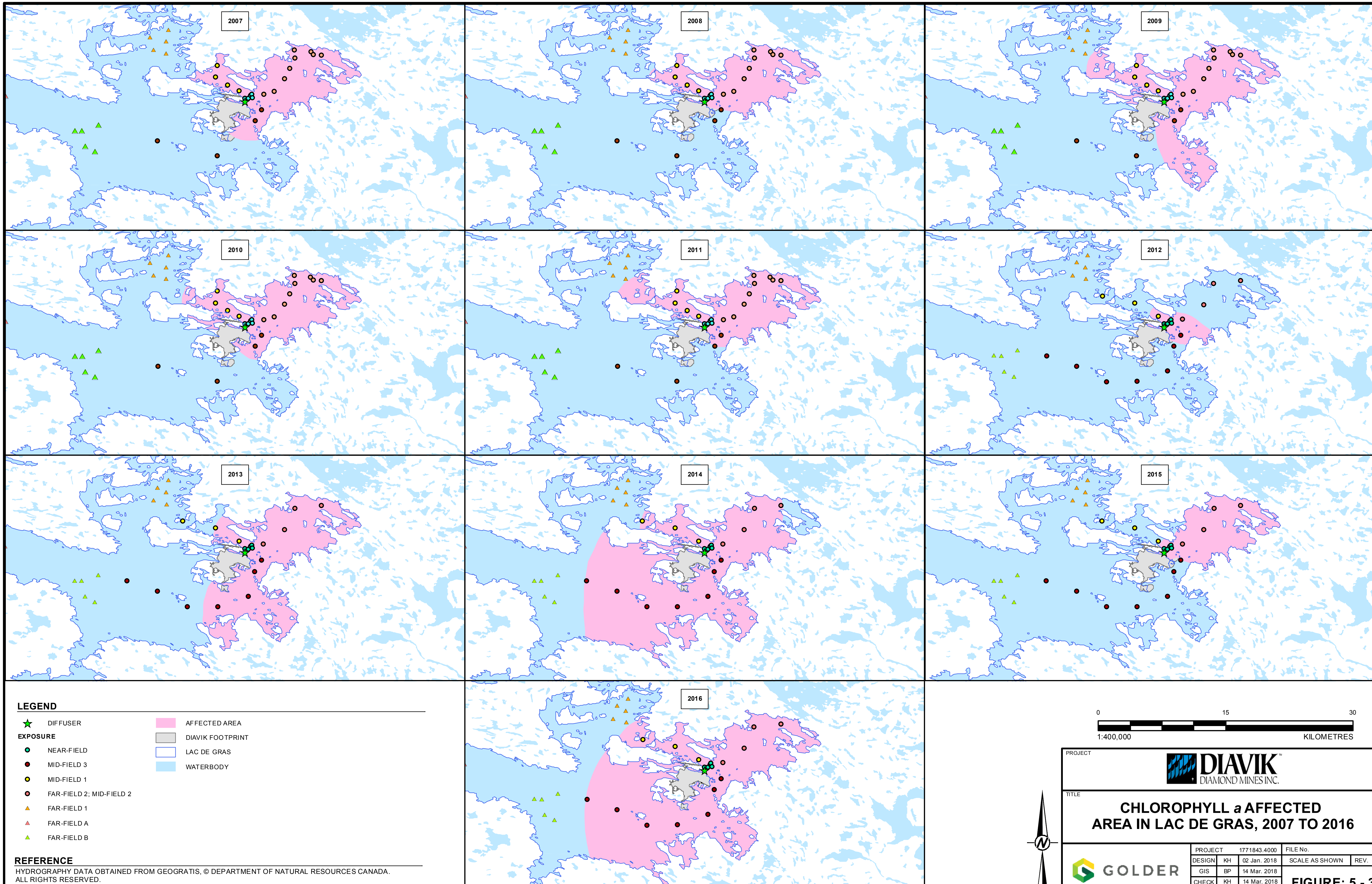
DIAVIK DIAMOND MINES INC.

TITLE

TOTAL NITROGEN AFFECTED AREA IN LAC DE GRAS, 2007 TO 2016

GOLDER

PROJECT		1771843.4000	FILE No.	
DESIGN	KH	02 Jan. 2018	SCALE AS SHOWN	REV. 0
GIS	BP	14 Mar. 2018		
CHECK	KH	14 Mar. 2018	FIGURE: 5 - 2	
REVIEW	ZK	14 Mar. 2018		



LEGEND

- ★ DIFFUSER

EXPOSURE

- NEAR-FIELD
- MID-FIELD 3
- MID-FIELD 1
- FAR-FIELD 2; MID-FIELD 2
- FAR-FIELD 1
- FAR-FIELD A
- FAR-FIELD B

- AFFECTED AREA
- DIAVIK FOOTPRINT
- LAC DE GRAS
- WATERBODY

REFERENCE

HYDROGRAPHY DATA OBTAINED FROM GEOGRATIS, © DEPARTMENT OF NATURAL RESOURCES CANADA.

PROJECTION: UTM ZONE 12 DATUM: NAD 83

0 15 30
1:400,000 KILOMETRES

PROJECT
DIAVIK DIAMOND MINES INC.

TITLE
CHLOROPHYLL *a* AFFECTED AREA IN LAC DE GRAS, 2007 TO 2016

GOLDER

PROJECT		1771843.4000	FILE No.	
DESIGN	KH	02 Jan. 2018	SCALE AS SHOWN	REV. 0
GIS	BP	14 Mar. 2018		
CHECK	KH	14 Mar. 2018	FIGURE: 5 - 3	
REVIEW	ZK	14 Mar. 2018		

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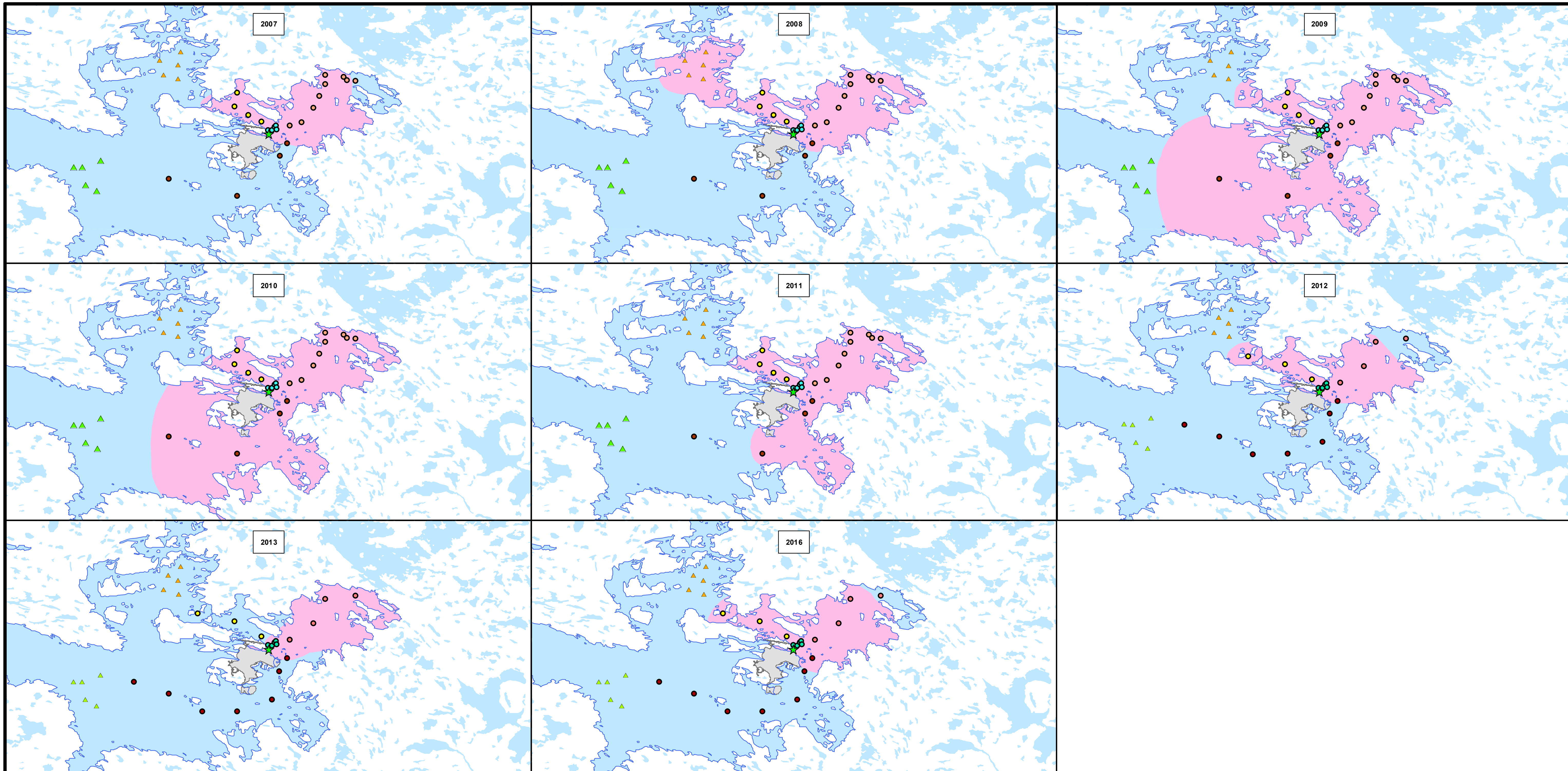

LEGEND

- DIFFUSER

EXPOSURE

- NEAR-FIELD
- MID-FIELD 3
- MID-FIELD 1
- FAR-FIELD 2; MID-FIELD 2
- FAR-FIELD 1
- FAR-FIELD A
- FAR-FIELD B
- AFFECTED AREA
- DIAVIK FOOTPRINT
- LAC DE GRAS
- WATERBODY

REFERENCE

HYDROGRAPHY DATA OBTAINED FROM GEOGRATIS, © DEPARTMENT OF NATURAL RESOURCES CANADA.

PROJECTION: UTM ZONE 12 DATUM: NAD 83

15 0 15
1:400,000 KILOMETRES

PROJECT

DIAVIK DIAMOND MINES INC.

TITLE

TOTAL PHYTOPLANKTON BIOMASS AFFECTED AREA IN LAC DE GRAS, 2007 TO 2016

GOLDER

PROJECT		1771843.4000	FILE No.
DESIGN	KH	02 Jan. 2018	SCALE AS SHOWN REV. 0
GIS	BP	14 Mar. 2018	
CHECK	KH	14 Mar. 2018	**FIGURE: 5 - 4**
REVIEW	ZK	14 Mar. 2018	

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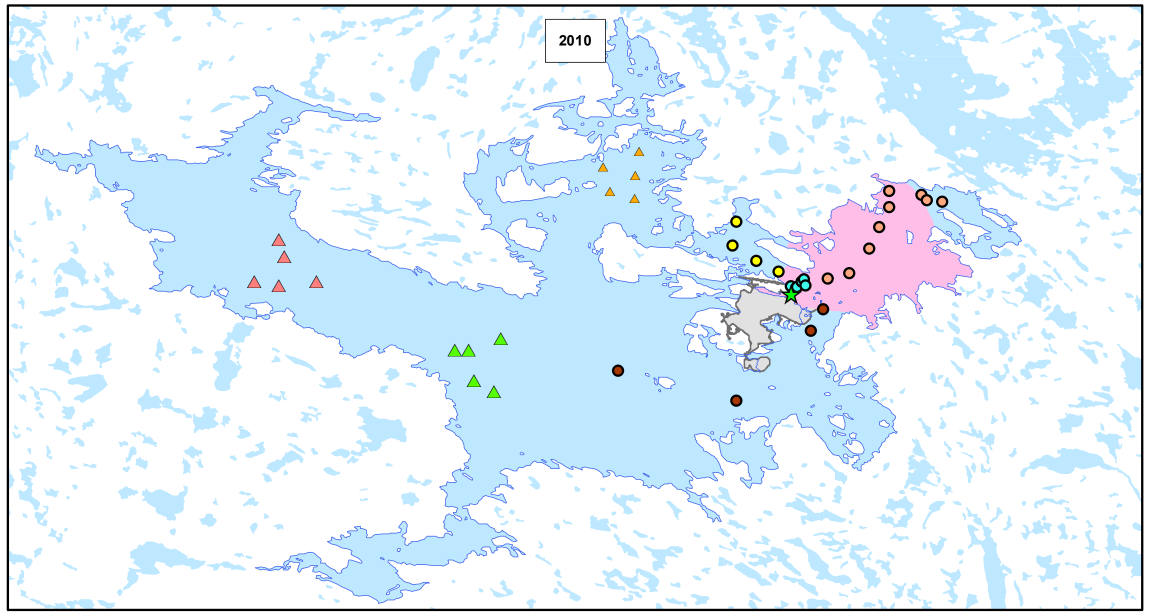

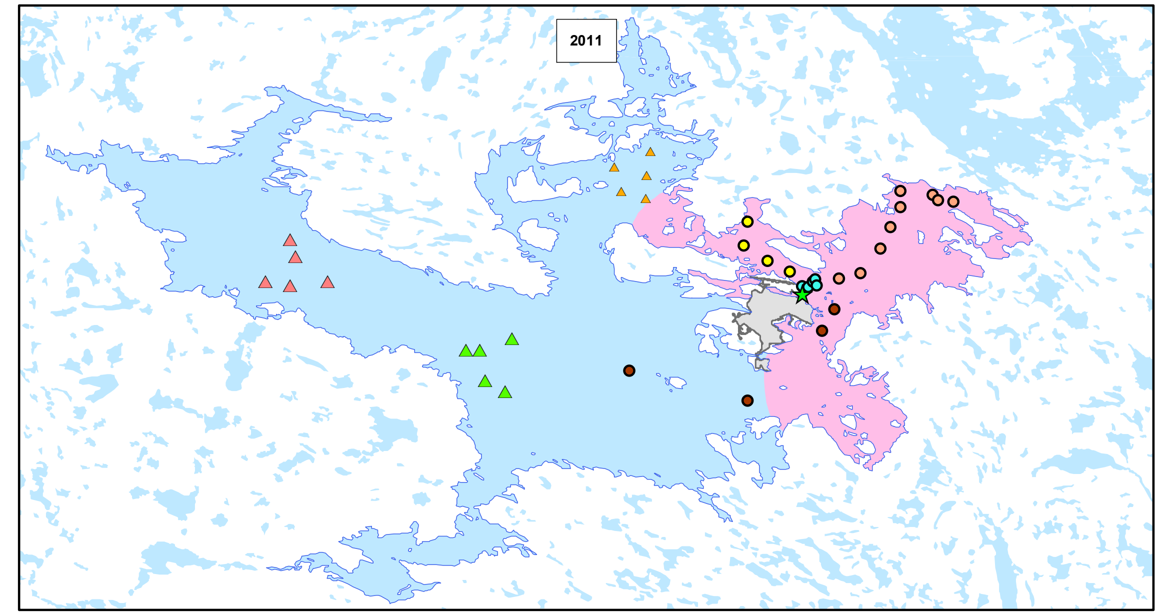

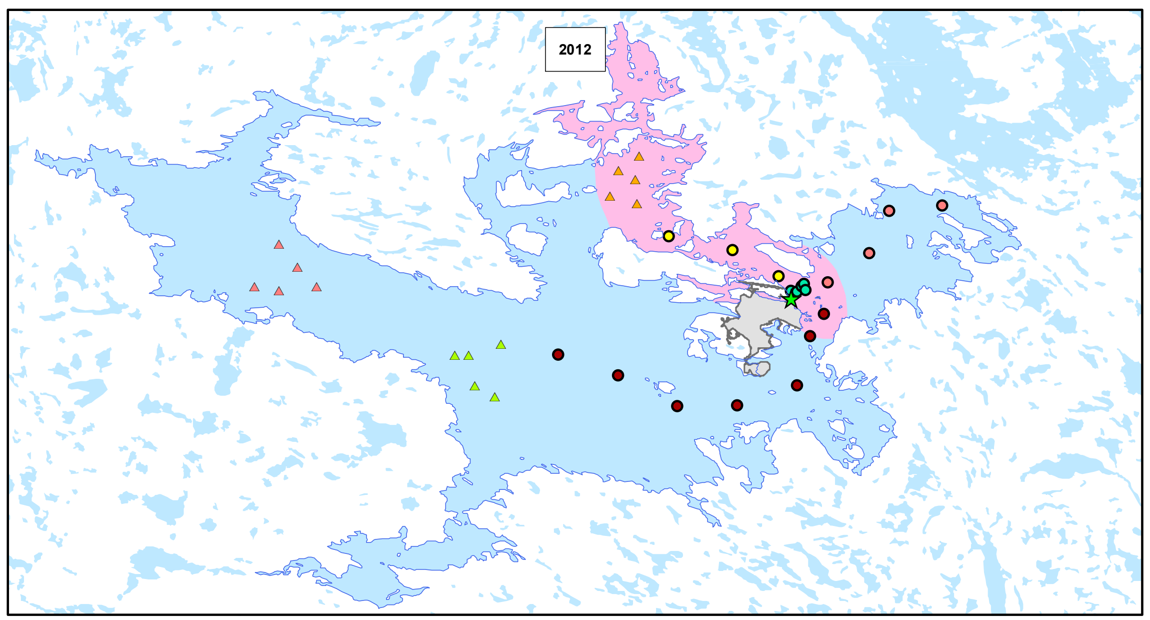

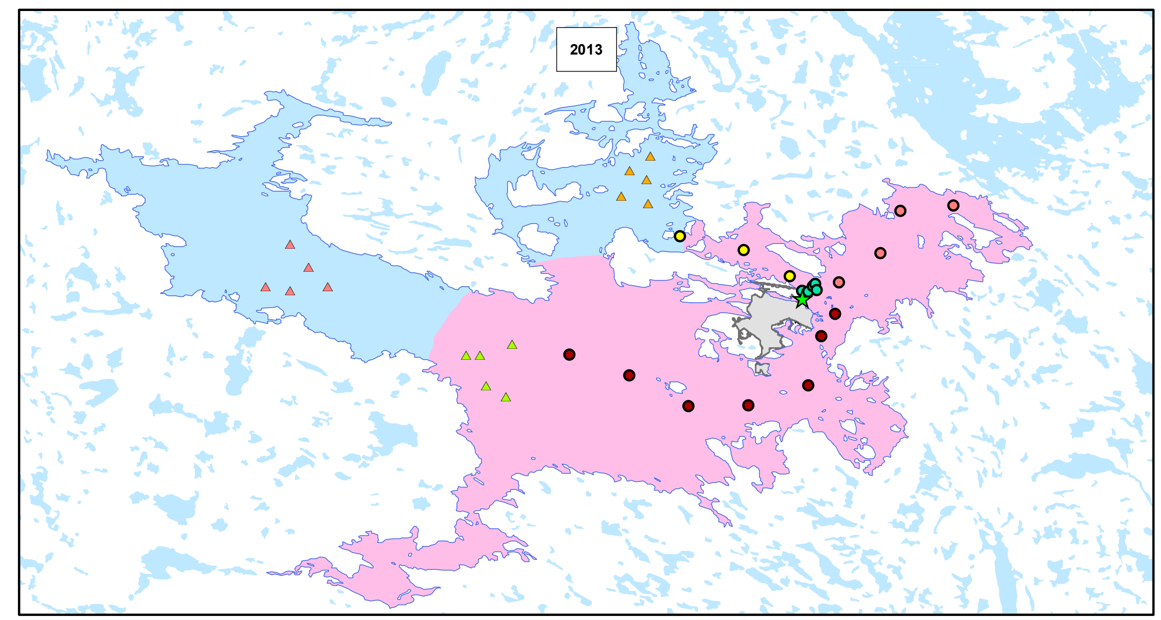

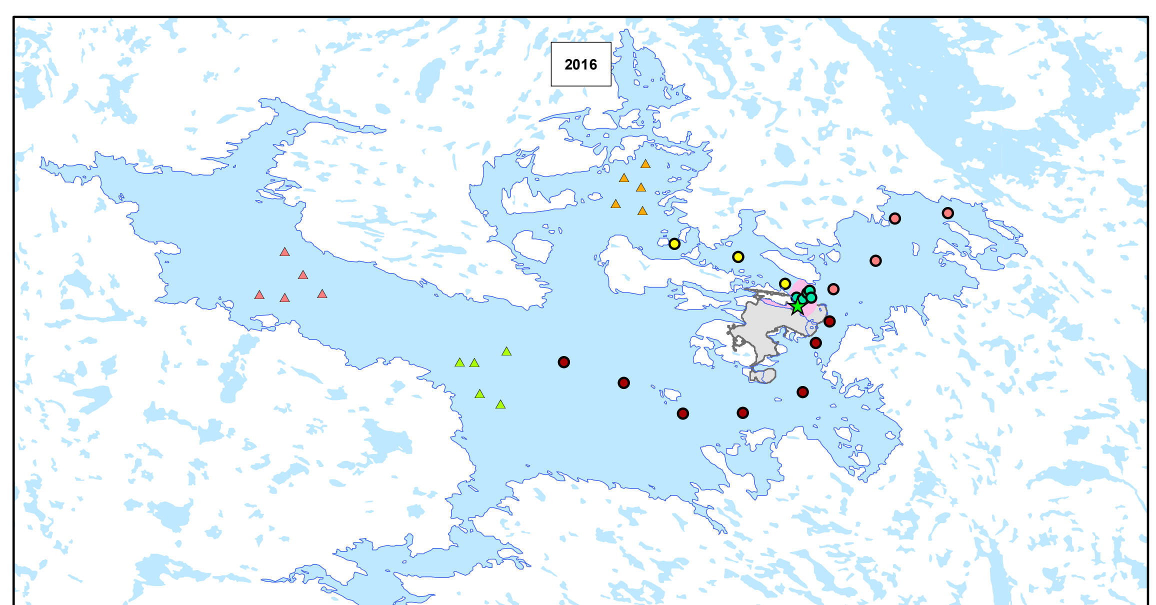

LEGEND

- DIFFUSER

EXPOSURE

- NEAR-FIELD
- MID-FIELD 3
- MID-FIELD 1
- FAR-FIELD 2; MID-FIELD 2
- FAR-FIELD 1
- FAR-FIELD A
- FAR-FIELD B

- AFFECTED AREA
- DIAVIK FOOTPRINT
- LAC DE GRAS
- WATERBODY

REFERENCE

HYDROGRAPHY DATA OBTAINED FROM GEOGRATIS, © DEPARTMENT OF NATURAL RESOURCES CANADA.

PROJECTION: UTM ZONE 12 DATUM: NAD 83

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1:450,000 KILOMETRES

PROJECT

DIAVIK DIAMOND MINES INC.

TITLE

TOTAL ZOOPLANKTON BIOMASS (AS AFDM) AFFECTED AREA IN LAC DE GRAS, 2007 TO 2016

GOLDER

PROJECT		1771843.4000	FILE No.	
DESIGN	KH	02 Jan. 2018	SCALE AS SHOWN	REV. 0
GIS	BP	14 Mar. 2018		
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REVIEW	ZK	14 Mar. 2018		

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5.3.1.2 Weight-of-Evidence Effect Ratings

Effect ratings for the eutrophication indicators have consistently demonstrated a nutrient enrichment response across years, but the extent of this response has varied, fluctuating between moderate to high from 2007 to 2016 (with the exception of low-level ratings in 2009 for zooplankton biomass [as AFDM] and phytoplankton biomass in 2013; Table 5-8). The effects ratings for TP were consistently moderate across years, with the exception of 2016, when no response was observed. The effects rating for TN was moderate in 2008, and from 2009 to 2016, TN concentrations exceeded the upper bound of the normal range in more than 20% of the lake, resulting in a high-level rating.

The effects ratings for chlorophyll *a* fluctuated between moderate and high from 2007 to 2016 (Table 5-8). Chlorophyll *a* concentrations in the NF area were above the upper bound of the normal range from 2007 to 2016, but the affected area only exceeded 20% of the lake in 2009, 2013, and 2016. For phytoplankton biomass, greater than 20% of the lake was affected in 2009 and 2011, as demonstrated by the high effects rating. The response of zooplankton biomass (as AFDM) has ranged from a low rating in 2009 to a moderate nutrient enrichment rating in 2010, 2012, and 2016, and a high rating in 2011 and 2013.

Table 5-8 Summary of Weight-of-Evidence Effect Ratings for Plankton, 2007 to 2016

Variable	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Total phosphorus	↑↑	↑↑	↑↑	↑↑	↑↑	↑↑	↑↑	n/a	n/a	0
Total nitrogen	n/a	↑↑	↑↑↑	↑↑↑	↑↑↑	↑↑↑	↑↑↑	n/a	n/a	↑↑↑
Chlorophyll *a*	↑↑	↑↑	↑↑↑	↑↑	↑↑	↑↑	↑↑↑	n/a	n/a	↑↑↑
Phytoplankton biomass (based on enumeration)	↑↑	↑↑	↑↑↑	↑↑	↑↑↑	↑↑	↑	n/a	n/a	↑↑
Zooplankton biomass (as AFDM)	n/a	n/a	↑	↑↑	↑↑↑	↑↑	↑↑↑	n/a	n/a	↑↑

Notes: The direction of the arrow indicates the direction of change for a given biological response endpoint relative to the far-field and/or normal range. 0 = no response; ↑/↓ = early warning low-level rating; ↑↑/↓↓ = moderate-level rating; ↑↑↑/↓↓↓ = high-level rating; n/a = data not available or not assessed. The direction of the sign (↑ or ↓) indicates the direction of difference relative to the FF areas. For community structure endpoint, both arrows are included (e.g., ↑/↓ or ↑↑/↓↓) to reflect that a community shift normally involves combined increases and decreases in biomass.

5.3.1.3 Action Levels

Chlorophyll *a* concentrations in the NF and MF areas were consistently greater than the upper bound of the normal range (0.82 µg/L) from 2007 to 2016 (Table 5-9). However, the total affected area differed among years (Table 5-7). In 2009, 2013, 2014, and 2016, over 20% of the lake experienced chlorophyll *a* concentrations above the normal range, resulting in Action Level 2 being triggered. A trigger of an Action Level 2 for chlorophyll *a* in 2013 resulted in the establishment of an Effects Benchmark, which was defined a 4.5 µg/L and approved by the WLWB (Golder 2014a).

The percentage of the lake with concentrations greater than the normal range plus 25% of the Effects Benchmark (i.e., 1.74 µg/L) was also calculated for each year to determine if Action Level 3 was triggered. Results revealed that <20% of Lac de Gras had concentrations greater than the normal range plus 25% of the Effects Benchmark (i.e., 1.74 µg/L). Because less than 20% of the lake area was above 1.74 µg/L, Action Level 3 was not triggered in any year.

Table 5-9 Summary of Action Level Rankings for Chlorophyll *a*, 2007 to 2016

Variable	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Chlorophyll *a*	AL1	AL1	AL2	AL1	AL1	AL1	AL2	AL2	AL1	AL2

AL1 = Action Level 1 triggered; AL2 = Action Level 2 triggered.

5.3.2 Nutrients in Effluent and Mixing Zone

This section provides information on temporal trends in nutrient loads and concentrations in effluent, and concentrations at the mixing zone boundary over the period of discharge (2002 to 2016). Trends have been assessed using time series plots of loadings to Lac de Gras from the effluent, rate of effluent discharge, concentrations in the effluent, and concentrations in the mixing zone. A statistical evaluation of trends for effluent chemistry was considered, but was determined not to be appropriate, due to the non-linear (i.e., up and down) trends and seasonal or cyclical nature of the concentrations in the effluent within years, and over the long-term (Section 4). The plots provided are considered to be sufficient to allow a visual evaluation of the variation in effluent variables over time.

Total phosphorus has an EQC specified in terms of load, rather than concentration. The Water Licence specifies that the load of TP must not exceed a maximum of 300 kg/mo, an average annual load of 1,000 kg/yr during the life of the Mine, and a maximum load of 2,000 kg/yr in any year during the life of the Mine. Monthly and annual TP loadings from 2002 to 2016 were consistently below the Water Licence TP load limit of 300 kg/mo and 1,000 kg/yr, respectively.

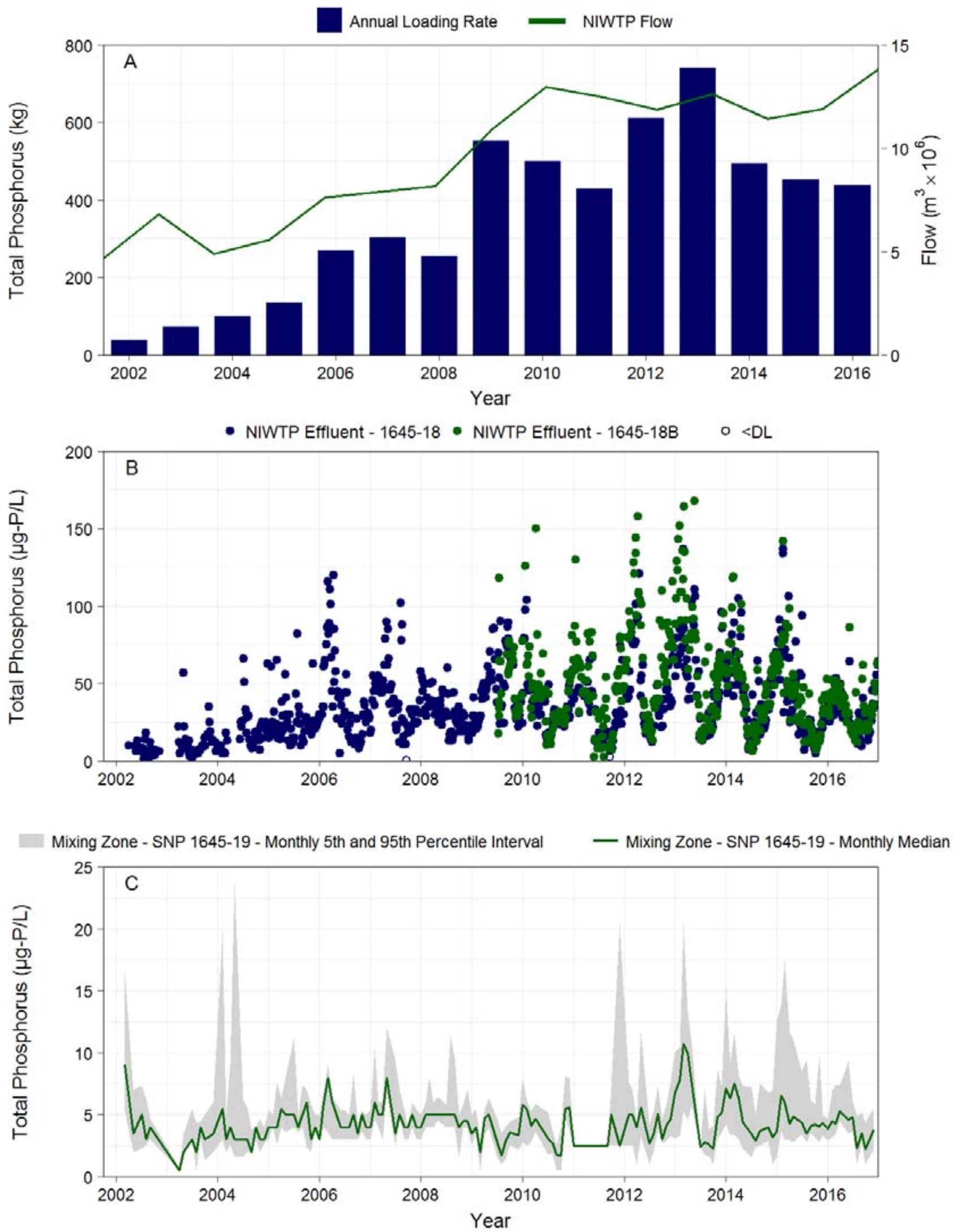
The annual loads of TP from the NIWTP have increased in a stepwise fashion during three sampling periods between 2002 and 2016 (Figure 5-6). The first increase in annual loadings occurred between 2002 and 2006, and was due to TP effluent concentrations increasing by 10 fold during a period of gradual increase in annual volume of effluent discharged over time. Annual loadings remained steady until 2009, when they sharply increased due to an increased effluent flow rate, despite relatively lower TP effluent concentrations. The last increase occurred between 2011 and 2013, which corresponded to relatively high effluent concentrations despite a steady flow rate. Annual loadings declined in 2014 and have remained approximately the same between 2014 and 2016.

Median effluent concentrations of TP at the mixing zone boundary have typically remained between 2 and 5 µg-P/L from 2002 to 2016, with a few exceptions observed in 2006, 2007, and 2013 to 2015. These peaks correspond to increased TP effluent concentrations.

The step-wise increase in annual loadings observed in TP was not observed in TDP or SRP. Annual loadings of TDP increased slowly between 2002 and 2009, corresponding to increased flow and increased TDP effluent concentrations, particularly in 2009 (Figure 5-7). The annual loading quickly increased from 2011 onwards, to a peak in 2014. This increased loading was due to increased concentrations of TDP in the effluent, along with higher flow rates. This same pattern was observed in annual loadings and effluent concentrations of SRP (Figure 5-8). Median mixing zone concentrations of TDP and SRP generally remained similar, although variable, between 2002 and 2011 (Figure 5-7 and Figure 5-8). After 2011, there was a small overall increase in concentrations, which reflect the greater effluent concentrations observed between 2011 and 2016.

Different patterns were observed for TN, total ammonia, nitrate and nitrite (Figures 5-9, 5-10, 5-11 and 5-12). Annual loadings increased from 2002 to 2006 for TN and nitrate, to 2007 for total ammonia, and to 2008 for nitrate, generally following the increase in effluent concentrations. Annual loadings declined noticeably in 2007 for TN and nitrate, and in 2008 for ammonia; decreases continued until 2010, reflecting lower effluent concentrations. After 2010, annual loadings were similar through to 2016, due to relatively similar flow rates and effluent concentrations. Median mixing zone concentrations generally followed patterns observed in effluent concentrations, with highest peaks occurring between 2002 and 2010.

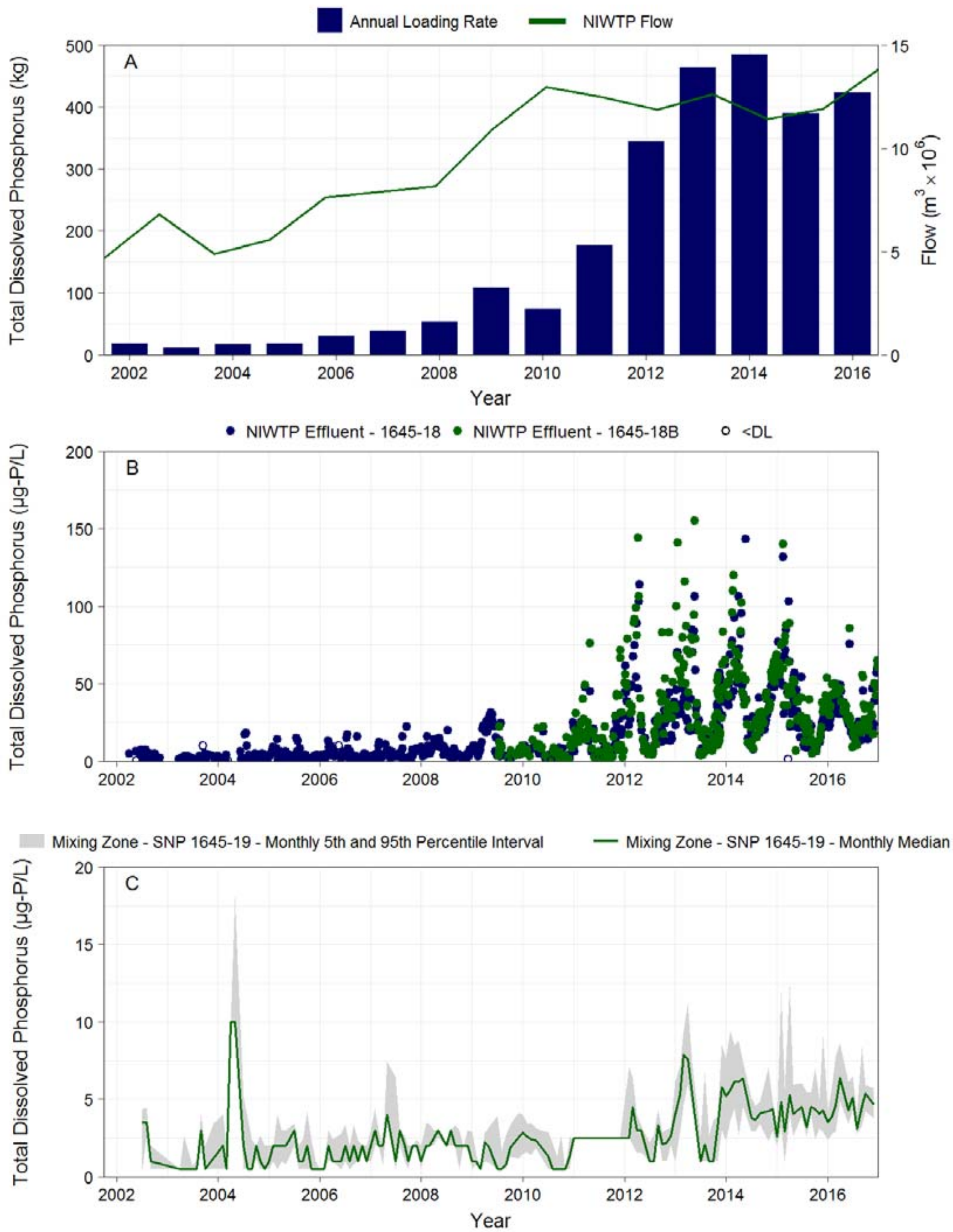
Figure 5-6 Annual Loading of Total Phosphorus from the North Inlet Water Treatment Plant (A), and Concentrations in Effluent (B), and at the Mixing Zone Boundary (C), 2002 to 2016



Note: Effluent values represent concentrations in individual samples. Mixing zone values represent the monthly median concentration and 5th and 95th percentile interval at three stations (i.e., SNP 1645-19A, SNP 1645-19B/B2, SNP 1645-19C) and five depths (i.e., 2 m, 5 m, 10 m, 15 m, and 20 m). Gaps in the mixing zone dataset reflect times when samples could not be collected due to hazardous sampling conditions (e.g., ice-on and ice-off periods).

µg-P/L = micrograms phosphorus per litre; NIWTP = North Inlet Water Treatment Plant; SNP = surveillance network program.

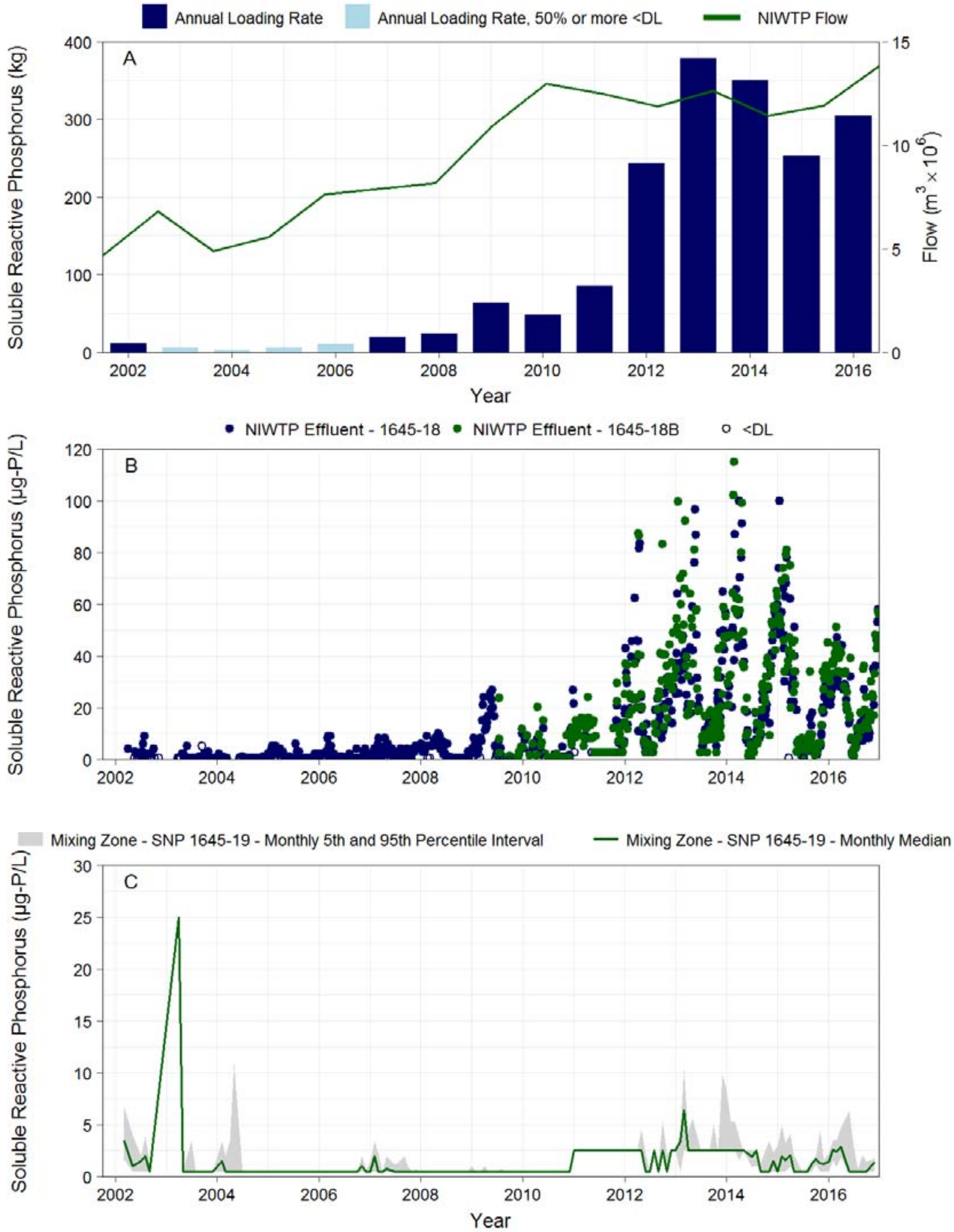
Figure 5-7 Annual Loading of Total Dissolved Phosphorus from the North Inlet Water Treatment Plant (A), and Concentrations in Effluent (B), and at the Mixing Zone Boundary (C), 2002 to 2016



Note: Effluent values represent concentrations in individual samples. Mixing zone values represent the monthly median concentration and 5th and 95th percentile interval at three stations (i.e., SNP 1645-19A, SNP 1645-19B/B2, SNP 1645-19C) and five depths (i.e., 2 m, 5 m, 10 m, 15 m, and 20 m). Gaps in the mixing zone dataset reflect times when samples could not be collected due to hazardous sampling conditions (e.g., ice-on and ice-off periods).

µg-P/L = micrograms phosphorus per litre; NIWTP = North Inlet Water Treatment Plant; SNP = surveillance network program.

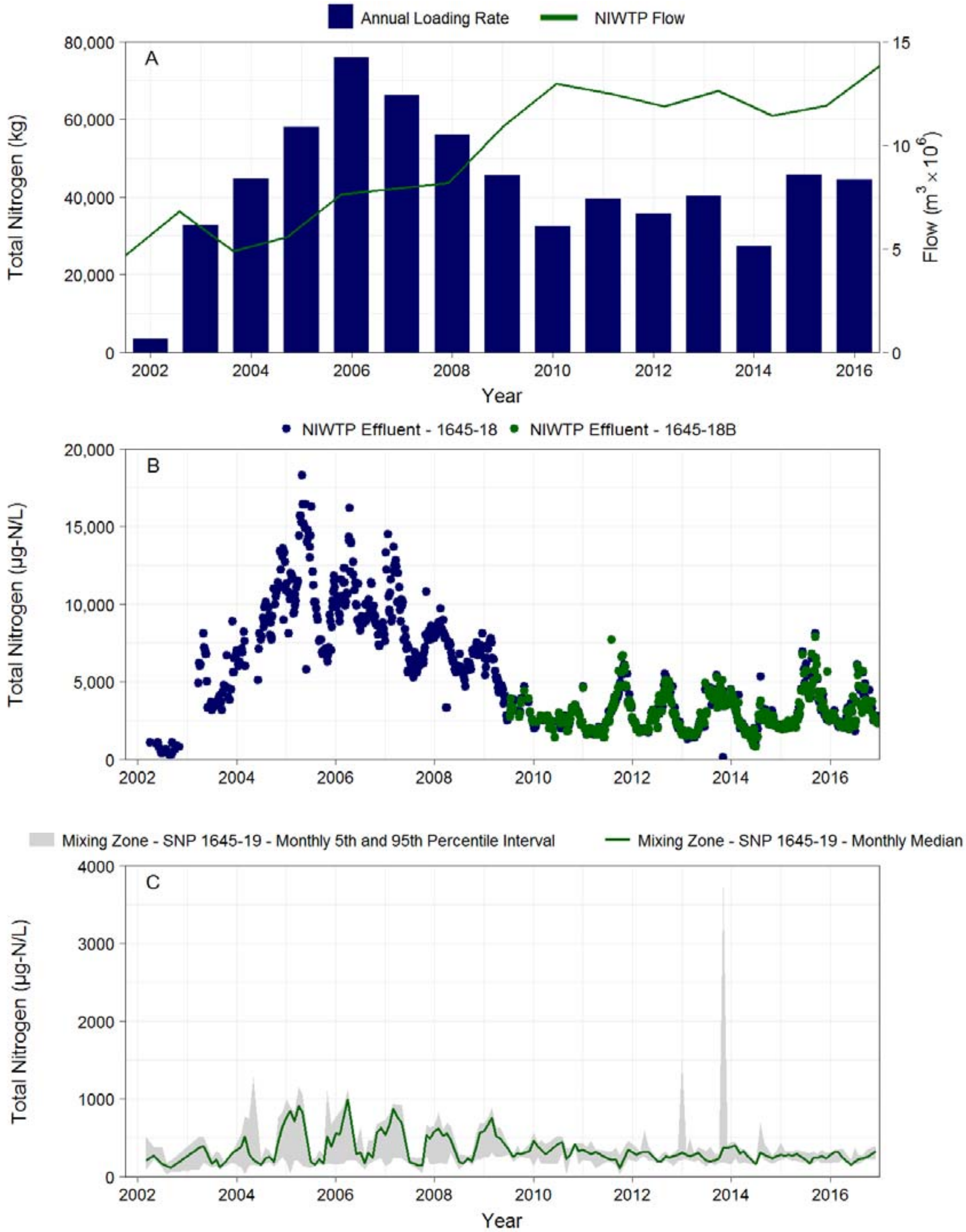
Figure 5-8 Annual Loading of Soluble Reactive Phosphorus from the North Inlet Water Treatment Plant (A), and Concentrations in Effluent (B), and at the Mixing Zone Boundary (C), 2002 to 2016



Note: Effluent values represent concentrations in individual samples. Mixing zone values represent the monthly median concentration and 5th and 95th percentile interval at three stations (i.e., SNP 1645-19A, SNP 1645-19B/B2, SNP 1645-19C) and five depths (i.e., 2 m, 5 m, 10 m, 15 m, and 20 m). Gaps in the mixing zone dataset reflect times when samples could not be collected due to hazardous sampling conditions (e.g., ice-on and ice-off periods).

µg-P/L = micrograms phosphorus per litre; NIWTP = North Inlet Water Treatment Plant; SNP = surveillance network program.

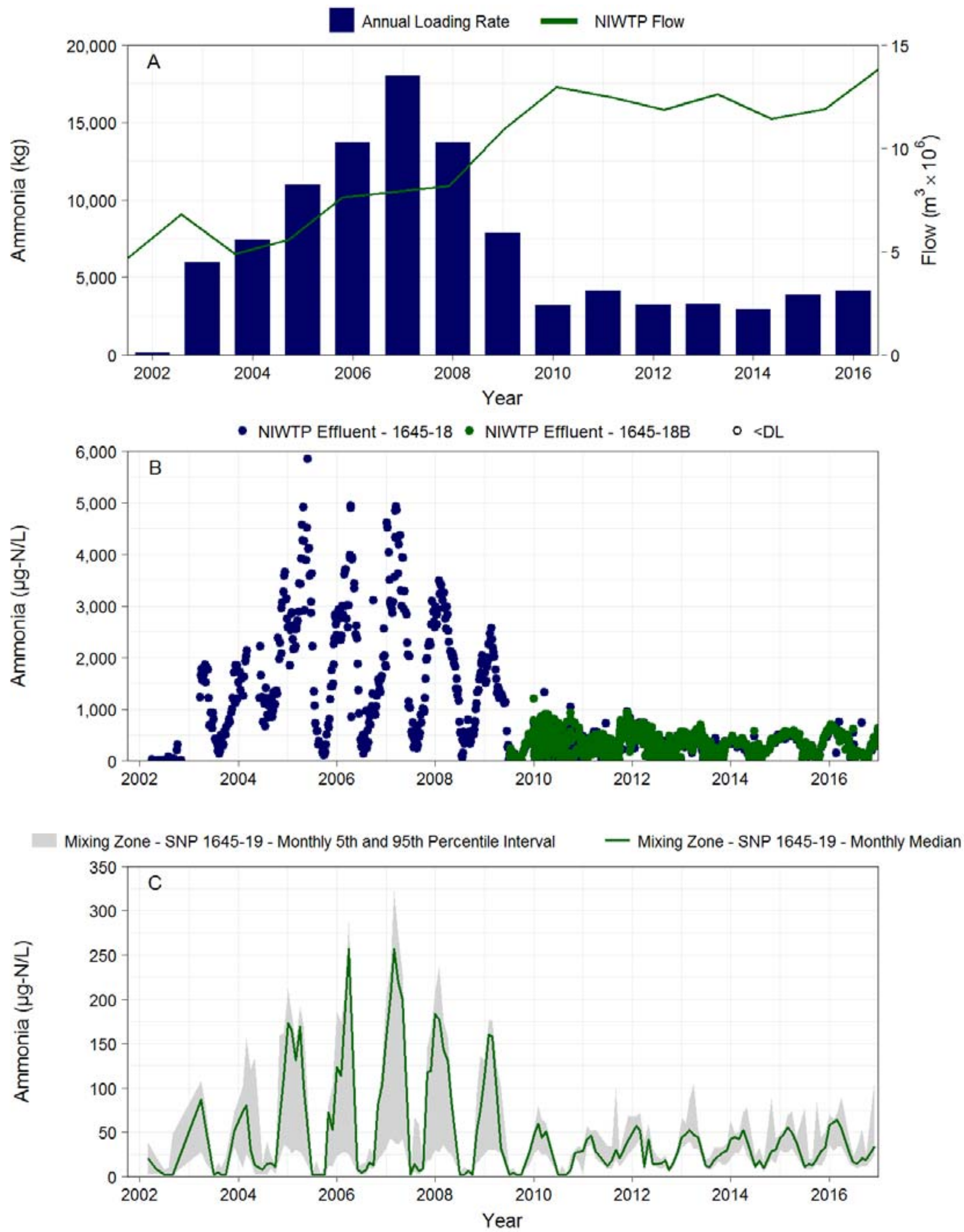
Figure 5-9 Annual Loading of Total Nitrogen from the North Inlet Water Treatment Plant (A), and Concentrations in Effluent (B), and at the Mixing Zone Boundary (C), 2007 to 2016



Note: Effluent values represent concentrations in individual samples. Mixing zone values represent the monthly median concentration and 5th and 95th percentile interval at three stations (i.e., SNP 1645-19A, SNP 1645-19B/B2, SNP 1645-19C) and five depths (i.e., 2 m, 5 m, 10 m, 15 m, and 20 m). Gaps in the mixing zone dataset reflect times when samples could not be collected due to hazardous sampling conditions (e.g., ice-on and ice-off periods).

µg-N/L = micrograms nitrogen per litre; NIWTP = North Inlet Water Treatment Plant; SNP = surveillance network program.

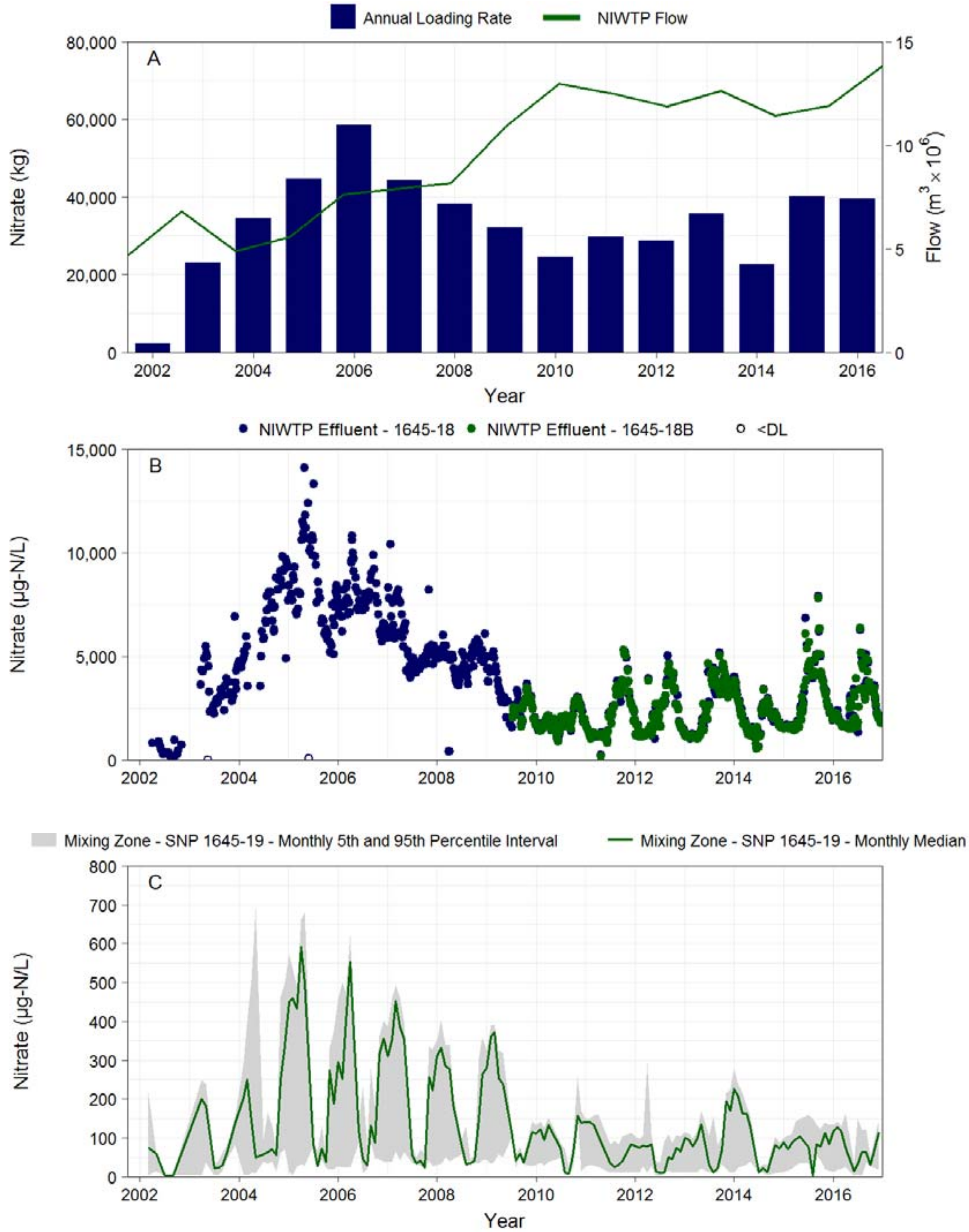
Figure 5-10 Annual Loading of Total Ammonia from the North Inlet Water Treatment Plant (A), and Concentrations in Effluent (B), and at the Mixing Zone Boundary (C), 2002 to 2016



Note: Effluent values represent concentrations in individual samples. Mixing zone values represent the monthly median concentration and 5th and 95th percentile interval at three stations (i.e., SNP 1645-19A, SNP 1645-19B/B2, SNP 1645-19C) and five depths (i.e., 2 m, 5 m, 10 m, 15 m, and 20 m). Gaps in the mixing zone dataset reflect times when samples could not be collected due to hazardous sampling conditions (e.g., ice-on and ice-off periods).

µg-N/L = micrograms nitrogen per litre; NIWTP = North Inlet Water Treatment Plant; SNP = surveillance network program.

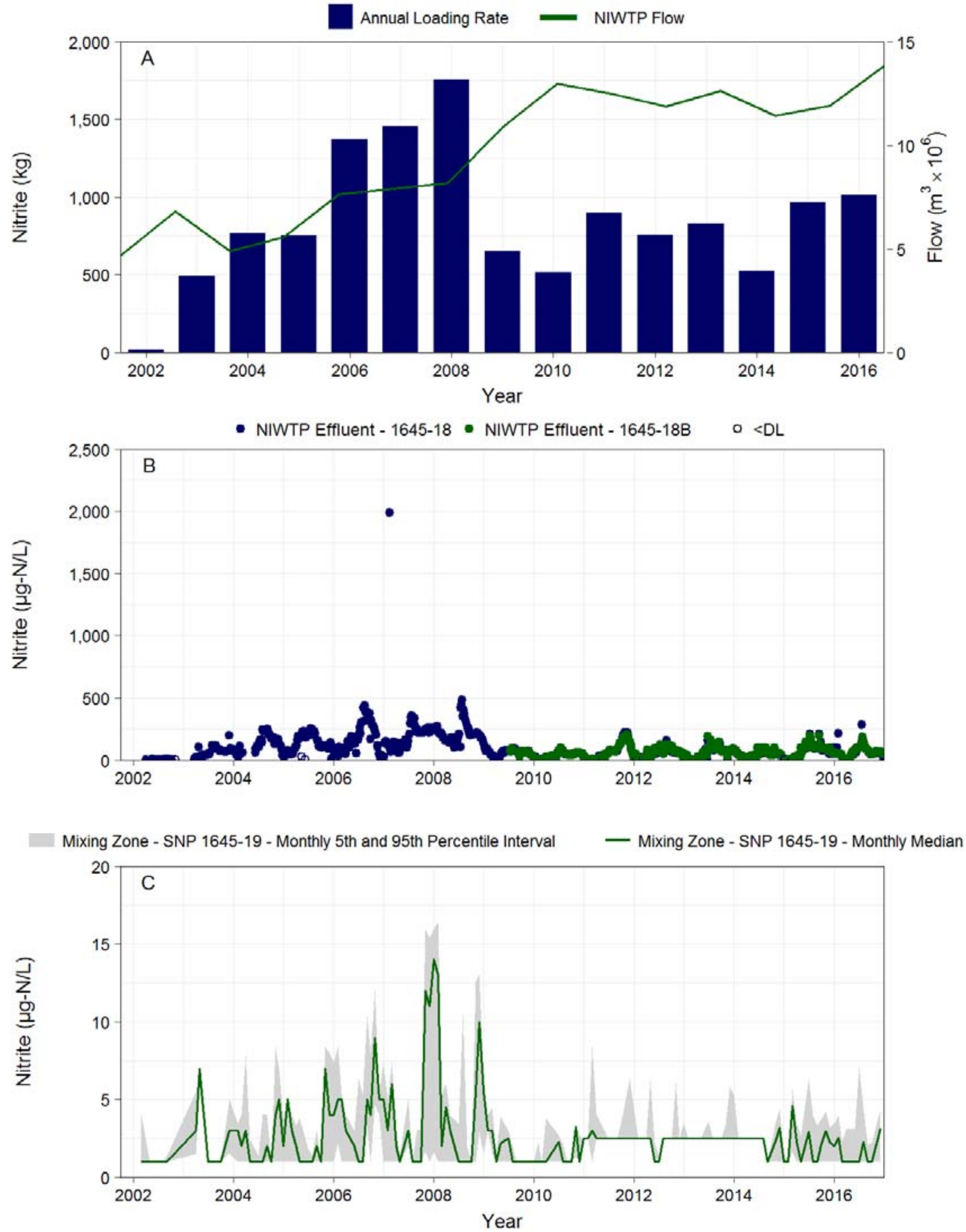
Figure 5-11 Annual Loading of Nitrate from the North Inlet Water Treatment Plant (A), and Concentrations in Effluent (B), and at the Mixing Zone Boundary (C), 2002 to 2016



Note: Effluent values represent concentrations in individual samples. Mixing zone values represent the monthly median concentration and 5th and 95th percentile interval at three stations (i.e., SNP 1645-19A, SNP 1645-19B/B2, SNP 1645-19C) and five depths (i.e., 2 m, 5 m, 10 m, 15 m, and 20 m). Gaps in the mixing zone dataset reflect times when samples could not be collected due to hazardous sampling conditions (e.g., ice-on and ice-off periods).

µg-N/L = micrograms nitrogen per litre; NIWTP = North Inlet Water Treatment Plant; SNP = surveillance network program.

Figure 5-12 Annual Loading of Nitrite from the North Inlet Water Treatment Plant (A), Concentrations in Effluent (B), and at the Mixing Zone Boundary (C), 2002 to 2016



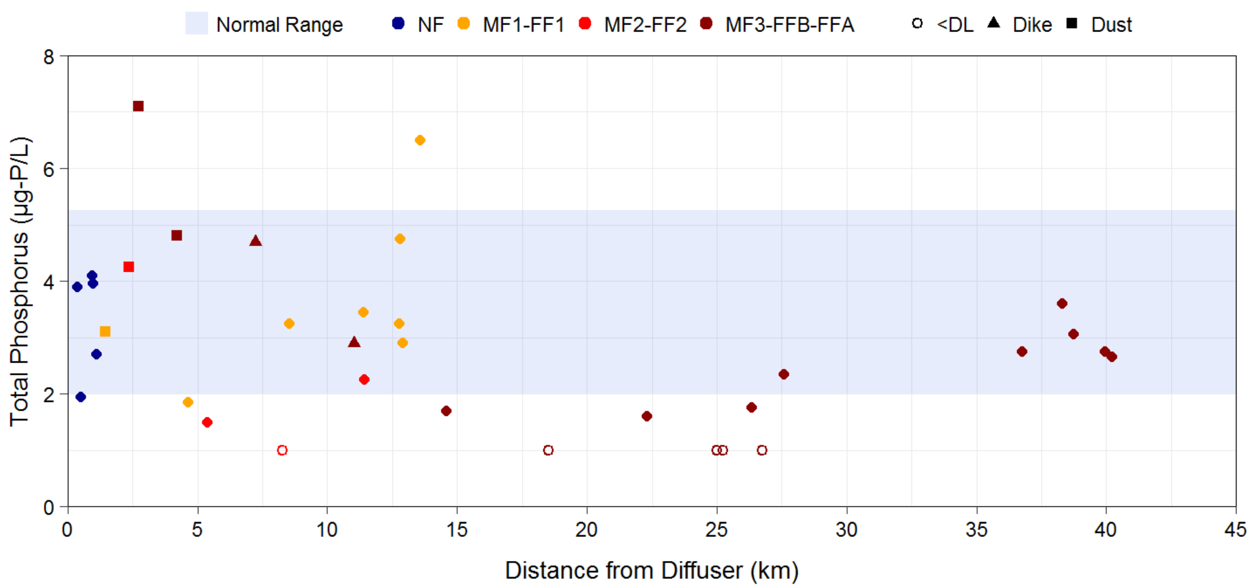
Note: Effluent values represent concentrations in individual samples. Mixing zone values represent the monthly median concentration and 5th and 95th percentile interval at three stations (i.e., SNP 1645-19A, SNP 1645-19B/B2, SNP 1645-19C) and five depths (i.e., 2 m, 5 m, 10 m, 15 m, and 20 m). Gaps in the mixing zone dataset reflect times when samples could not be collected due to hazardous sampling conditions (e.g., ice-on and ice-off periods).

µg-N/L = micrograms nitrogen per litre; NIWTP = North Inlet Water Treatment Plant; SNP = surveillance network program.

5.3.3 Effects of Dust Deposition and Dike Construction

There are nine stations in the zone of influence from dust deposition: NF1 to NF5, MF1-1, MF2-1, MF3-1, and MF3-2 (Golder 2017c) In 2016, concentrations of TP during the open-water season were above the normal range at one station, MF3-1 (Figure 5-13). Concentrations at two additional stations along the NF-MF3-FFB-FFA transect, MF3-2 and MF3-3, were at the top of the normal range. Two of these stations (i.e., MF3-1 and MF3-2) are also within the area of Lac de Gras shown to be affected by the construction of the A21 dike during the 2016 open-water season (Section 4). All four stations near the dike (i.e., MF3-1 to MF3-4) showed elevated concentrations of other particulate-related variables (i.e., TSS, aluminum, cadmium, copper, manganese, barium, bismuth, chromium, cobalt, iron, lead, lithium, nickel, silicon, thallium, tin, titanium, uranium, vanadium, zinc and zirconium; Section 4). As a result, the elevated concentrations of TP at the MF stations are most likely related to dike construction, rather than dust deposition.

Figure 5-13 Concentrations of Total Phosphorus in Lac de Gras and Lac du Sauvage in Relation to Dike Construction and Dust Deposition, 2016



5.3.4 Temporal Trends

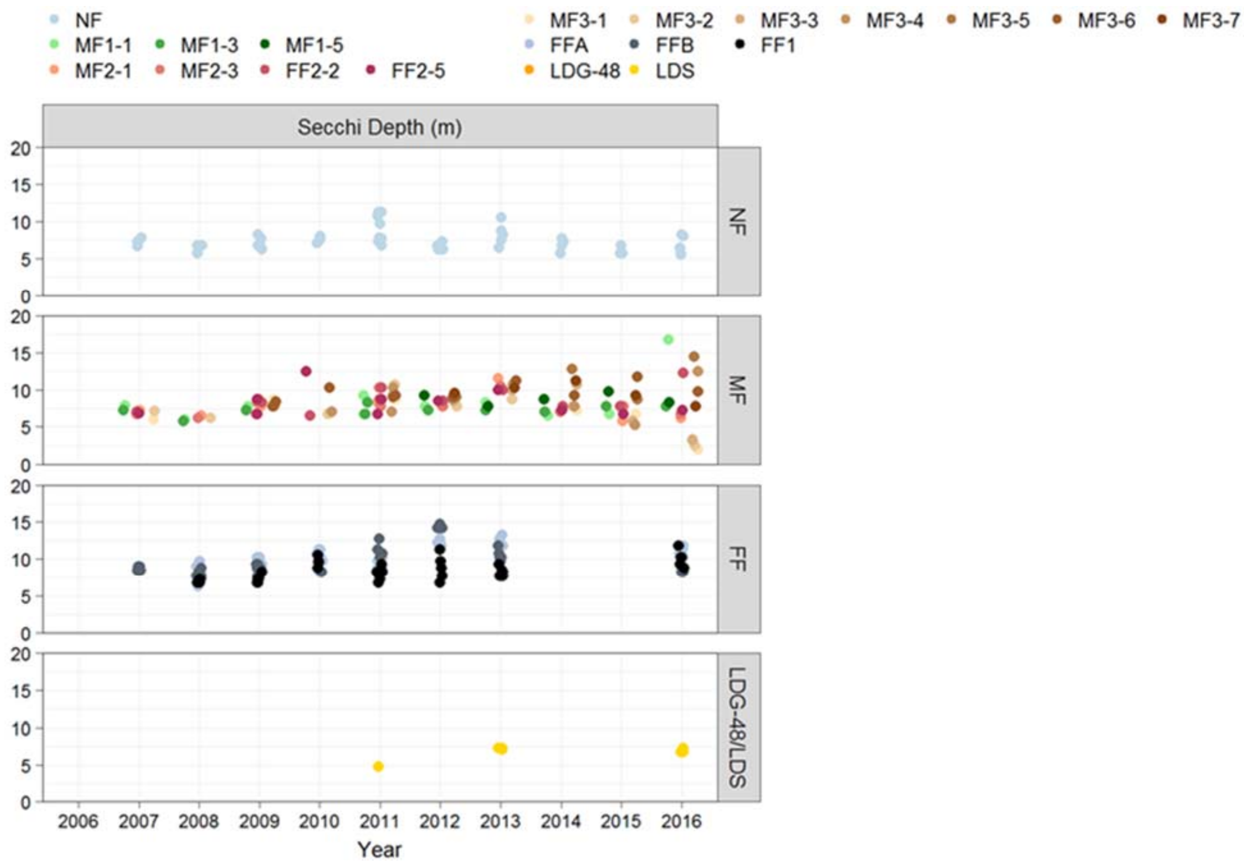
5.3.4.1 Secchi Depth

Secchi depth measurements indicate good light penetration in all areas of Lac de Gras. Secchi depths in the NF area were typically between 5 and 8 m during the open-water seasons of 2007 to 2016, with the exception of 2011 and 2013, which had maximum Secchi depths of 11.25 m and 10.5 m, respectively (Figure 5-14). Secchi depths along transects were more variable than in the NF area, particularly in recent years; between 2014 and 2016, Secchi depths ranged from 2.0 to 16.75 m. The shallow Secchi depth readings along the NF-MF3-FFB-FFA transect (i.e., stations MF3-1, MF3-1 and MF3-3) in 2015 and 2016 likely resulted from elevated concentrations of particulate material in the water column from the A21 dike construction in 2016 (Figure 5-14, Section 4). Secchi depths in the FF area were generally greater than in the NF area; for example, in 2016, Secchi depths ranged from 5.5 to 8.25 m in the NF area, but from 8.25 to 11.75 m in FF areas. Secchi depths in Lac du Sauvage were similar to those in the NF area in 2013 and 2016.

Secchi depth corresponds to the depth at which approximately 10% of the surface light remains (Dodds and Whiles 2010). The euphotic zone extends to a depth where approximately 1% of surface light remains, often estimated as twice the Secchi depth (Dodds and Whiles 2010). Typically in more productive (i.e., eutrophic) waterbodies or in waterbodies with large amounts of suspended or dissolved materials, the water contains more material to absorb or reflect light, which inhibits light transmittance to deeper depths and equates to shallower Secchi depths, as observed along the NF-MF3-FFB-FFA transect in 2016 (i.e., 2.0 m; Figure 5-14).

In less productive (i.e., oligotrophic) waterbodies with low amounts of suspended or dissolved material, light is transmitted to greater depths (Dodds and Whiles 2010), as observed in most areas of Lac de Gras throughout the time series (Figure 5-14). Secchi depth data are critical to defining the extent of the euphotic habitats occupied by phytoplankton. Given the Secchi depths measured in Lac de Gras, a large proportion of the total volume of this lake is within the euphotic zone, and can support phytoplankton growth.

Figure 5-14 Secchi Depths Measured in Lac de Gras during the Open-water Season, 2006 to 2016



Notes: A normal range for Secchi depth has not been developed. DG = Lac de Gras; LDS = Lac du Sauvage.

5.3.4.2 Nutrients

In the analysis of nutrient variables from 2000 to 2016, the temporal trends in concentrations of TN, TDN (open-water only) and N+N were parabolic, and the trends in concentrations of TP and TDN (ice-cover only) were linear (Table 5-10). Concentrations of TDP and SRP have frequently been below the DL and, as a result, the standard statistical trend analysis method could not be employed, and was replaced with a logistic regression (Section 5.2.3.3). Variables were log, inverse log, square root, or inverse square root transformed (Table 5-10), with the exception of total ammonia, TDP and SRP. Total dissolved phosphorus and SRP data were transformed as presence or absence of detectable data.

Data quality issues with analysis of low levels of total ammonia occurred from 2013 to 2016 (Eutrophication Indicators data) and from 2011 to 2016 (Water Quality component data; Section 4). In general, total ammonia concentrations in blank samples analyzed by Maxxam were at or above levels found in Lac de Gras, while concentrations reported for lake water samples were greater and more variable than values previously provided by UofA or ALS (Section 4). As a result, the data were not compared to the normal range, which is based on the UofA data, and trend analysis was not completed for total ammonia. Efforts that have taken place to address the QC issues for total ammonia are detailed in Section 4, Appendix 4B.

Table 5-10 Nutrient AIC Model Selection, 2000 to 2016

Variable	Years	Season	Yeo-Johnson transformation[a]	Parabolic AIC Score	Linear AIC Score
Total phosphorus	2007 to 2016	Ice-cover	0	46	46
		Open-water	0.5	493	492
Total nitrogen	2000 to 2016	Ice-cover	0	-59	-29
		Open-water	-1	-2643	-2596
Total dissolved nitrogen	2007 to 2016	Ice-cover	-0.5	-974	-974
		Open-water	-1	-2398	-2336
Nitrate + nitrite	2000 to 2016	Ice-cover	0	261	283
		Open-water	-0.5	-6	-7

a) See Section 2.4.2.1 for details on transformation approach.

Note: Grey shaded cells indicate the model chosen. AIC = Akaike's information criterion.

Phosphorus Variables

Temporal trends in TP were observed during the ice-cover and open-water seasons between 2007 and 2016 in Lac de Gras (Figures 5-15 and 5-16; Table 5-11). Mid-depth concentrations of TP during the ice-cover season have significantly decreased in all areas of Lac de Gras, including at LDG-48, between 2007 and 2016 (Figure 5-16; Table 5-12). No difference in trends was detected among sampling areas (Table 5-11). During the open-water season, significant declining trends were only observed in the NF and FFB areas of Lac de Gras (Figure 5-16; Table 5-12).

Concentrations of TP in the NF area were slightly greater during the ice-cover season compared to the open-water season, while in the MF and NF areas, TP concentrations were generally similar between seasons (Figure 5-15; Appendix 5B, Figures 5B-1 and 5B-2). Concentrations of TP at several stations in the MF1, MF2-FF2, and MF3 areas, and in the FF areas during the ice-cover season, exceeded the normal range between 2007 and 2013. Between 2014 and 2016, concentrations were generally within or below the lower limit of the normal range, with the exception of the concentration measured at the top depth of FF2-5 in 2016 (Appendix 5B; Figure 5B-1).

Concentrations of TP in the NF area have remained mostly within or just above the upper limit of the normal range during the open-water season from 2007 to 2016 (Figure 5-15). As observed in the previous re-evaluation report, there were four conspicuously high values in the NF, MF1-1, MF2-1, and MF3-1 areas in 2009 that did not meet the criterion for exclusion as anomalous values (Golder 2016a), but were nonetheless atypical. These elevated values do not appear to reflect an increase in effluent loads or concentrations. Exceedances of the normal range occurred at several stations in the MF areas, particularly in 2007 and 2012. Concentrations also exceeded the normal range at several FF area stations in 2010, 2012, and 2016. Concentrations at LDG-48 were within or just above the upper limit of the normal range during the open-water season, from 2007 to 2016.

Concentrations of TDP and SRP have been frequently below the DL since 2007 (Figures 5-17 and 5-19). Significant differences were identified among areas and over time (Table 5-13), and results indicated declining trends in the areas and stations analyzed (i.e., greater frequency of non-detect values in recent years; Figures 5-18 and 5-20).

Concentrations of TDP in the NF area during the ice-cover season exceeded the normal range in several samples between 2007 and 2016 (Figure 5-17;Appendix 5B Figures 5B-3 and 5B-4). Fewer exceedances of the normal range were noted in the MF areas and TDP concentrations were typically within the normal range between 2014 and 2016. In the FF areas, concentrations during the ice-cover season have been within the normal range since 2010.

During the open-water season, TDP concentrations in the NF area have remained within or at the upper limit of the normal range from 2007 to 2016, with the exception of an atypically high value in 2009 (Figure 5-17). Exceedances of the normal range occurred at some stations along the MF areas, and in the FFA and FFB areas, particularly in 2012. Concentrations of TDP at LDG-48 have generally been within normal range during both ice-cover and open-water seasons.

During the ice-cover season, SRP concentrations exceeded the normal range in the NF area at about a third of the stations (Figure 5-19; Appendix 5B Figures 5B-5 and 5B-6). Fewer exceedances of the normal range were observed at mid-depth in the MF areas. More exceedances in the MF areas were observed in the samples taken at bottom depth (Appendix 5B Figure 5B-6), compared to those taken at the top or middle depths (Figure 5-11, Appendix 5B-5). Concentrations of SRP occasionally exceeded the normal range in the FF areas and at LDG-48.

Concentrations of SRP were frequently reported at or below the DL (1 µg/L) during the open-water season from 2007 to 2016 in the NF areas (Figure 5-19). From 2014 to 2016, in the MF and FF areas, concentrations were more frequently detected at concentrations above the upper limit of the normal range than in previous years. In particular, the highest concentrations were observed in 2016 in the FF areas. Concentrations were typically at or below DL at LDG-48. Concentrations of SRP in all areas of Lac de Gras were within five times the DL in both seasons, where results are subject to greater analytical uncertainty compared to greater concentrations.

Table 5-11 Significance of Nutrient Fixed Effects Models, 2002 to 2012

Variable	Year	Season	Coefficient	Numerator DF	Denominator DF	F-value	*P*-value
Total phosphorus	2007 to 2016	Ice- cover	Area	7	16	1.66	0.189
			Year	1	165	89.44	<0.001
			Area × Year	7	165	1.67	0.120
		Open-water	Area	7	16	0.73	0.648
			Year	1	188	9.56	0.002
			Area × Year	7	188	0.73	0.646
Total nitrogen	2000 to 2016	Ice-cover	Area	7	18	4.12	0.007
			Year	1	158	0.14	0.712
			Year^2	1	158	0.14	0.713
			Area × Year	7	158	4.12	**<0.001**
			Area × Year^2	7	158	4.12	**<0.001**
		Open-water	Area	7	18	7.12	<0.001
			Year	1	194	16.94	<0.001
			Year^2	1	194	16.95	<0.001
			Area × Year	7	194	7.12	**<0.001**
			Area × Year^2	7	194	7.11	**<0.001**
Total dissolved nitrogen	2007 to 2016	Ice-cover	Area	7	16	6.139	0.001
			Year	1	146	23.88	<0.001
			Area × Year	7	146	6.11	**<0.001**
		Open-water	Area	7	16	1.28	0.319
			Year	1	167	51.71	<0.001
			Year^2	1	167	51.75	<0.001
			Area × Year	7	167	1.28	0.263
			Area × Year^2	7	167	1.28	0.263
Nitrate + nitrite	2000 to 2016	Ice-cover	Area	7	18	4.19	0.007
			Year	1	142	0.27	0.604
			Year^2	1	142	0.27	0.607
			Area × Year	7	142	4.20	**<0.001**
			Area × Year^2	7	142	4.20	**<0.001**
		Open-water	Area	7	18	2.07	0.101
			Year	1	172	0.36	0.551
			Year^2	1	172	0.36	0.550
			Area × Year	7	172	2.07	**0.049**
			Area × Year^2	7	172	2.07	**0.049**

Notes: **Bolded** values indicate significance on the interaction term at P <0.05; DF = degrees of freedom; F-value = F-statistic; *P*-value = probability value.

Table 5-12 Estimated Significance of Difference of Linear Slopes from Zero

Variable	Area							
	NF	MF1-3	FF2-2	MF3-4	FF1	FFB	FFA	LDG-48
Total phosphorus (IC)	**<0.001 ↓**	**0.016 ↓**	**0.001 ↓**	**0.004 ↓**	**0.027 ↓**	**<0.001 ↓**	**<0.001 ↓**	**<0.001 ↓**
Total phosphorus (OW)	**0.027 ↓**	0.981	0.060	0.561	0.512	**0.007 ↓**	0.927	0.400
Total dissolved nitrogen (IC)	0.971	0.217	0.304	0.061	**<0.001 ↑**	**<0.001 ↑**	**<0.001 ↑**	0.848

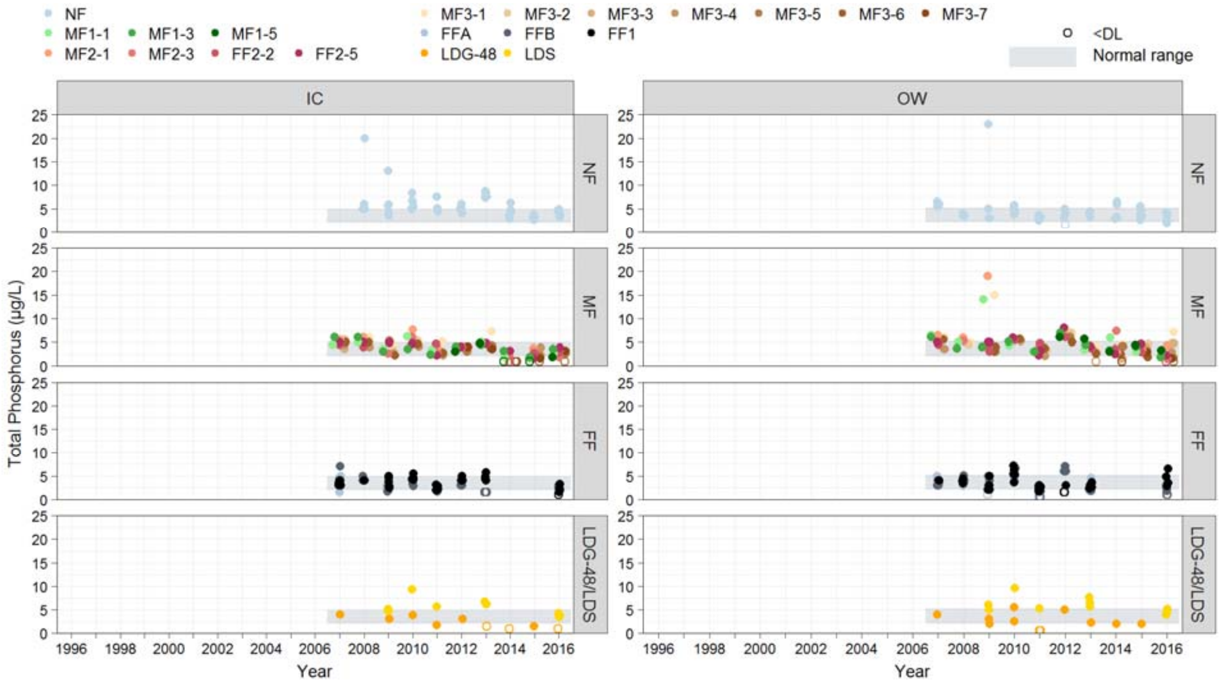
Notes: The arrow indicates direction of trend; IC = ice-cover; OW = open-water; NF= near-field; MF = mid-field; FF = far-field; LDG = Lac de Gras; <= less than.

Table 5-13 Significance of Nutrient Fixed Effects Models (Logistic Regression), 2006 to 2016

Variable	Season	Coefficient	Degrees of Freedom	Chi Squared	*P*-value
Total dissolved phosphorus	Ice-cover	Area	7	13.42	0.063
		Year	1	55.92	**<0.001**
	Open-water	Area	7	6.10	0.528
		Year	1	21.44	**<0.001**
Soluble reactive phosphorus	Ice-cover	Area	7	35.40	**<0.001**
		Year	1	29.96	**<0.001**
	Open-water	Area	7	14.08	0.050
		Year	1	13.88	**<0.001**

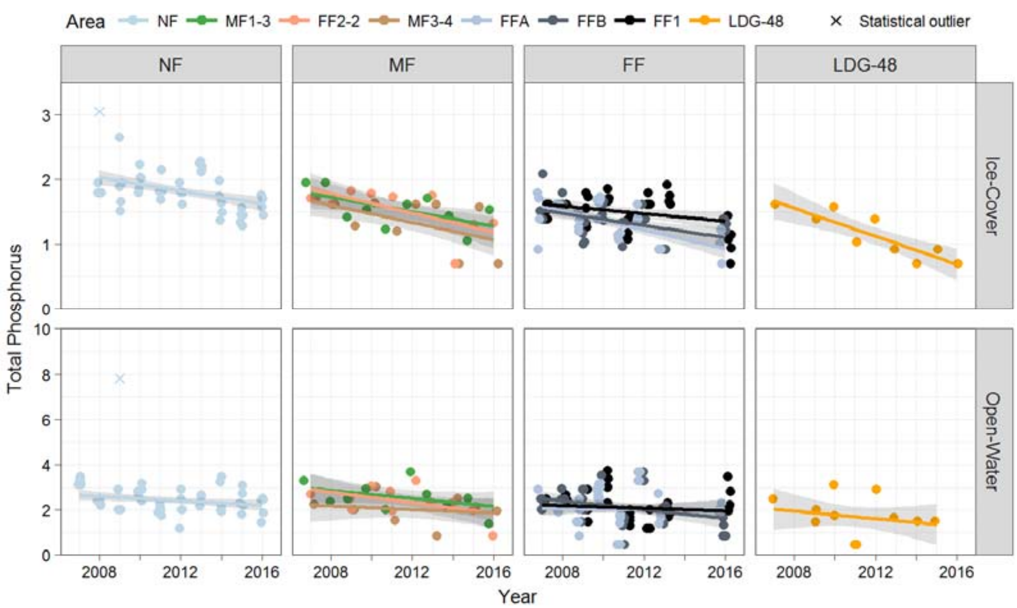
Note: **Bold text** = *P*-value significant at <0.05; <= less than.

Figure 5-15 Concentrations of Total Phosphorus in Lac de Gras during the Ice-cover (Mid-depth) and Open-water Seasons, 2007 to 2016



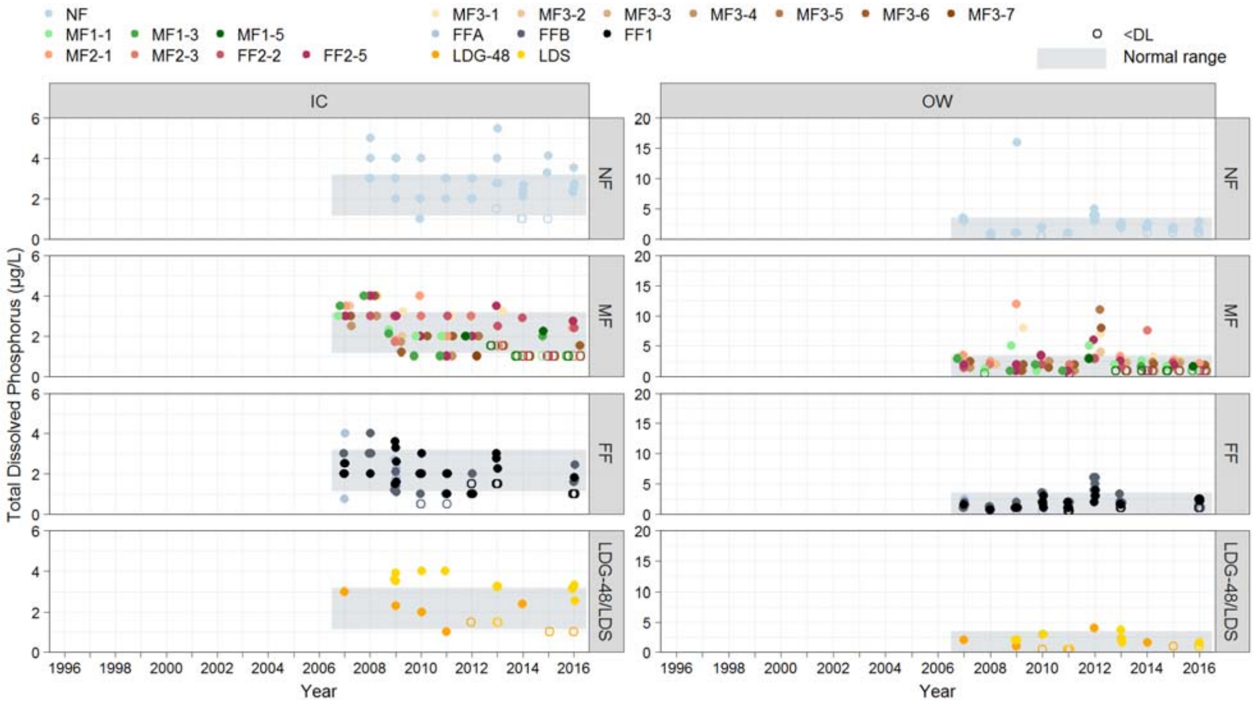
Note: NF = Near-field; MF = Mid-field; FF = Far-field; LDS = Lac du Sauvage; Lac de Gras

Figure 5-16 Trend Analysis for Total Phosphorus Concentrations, 2007 to 2016



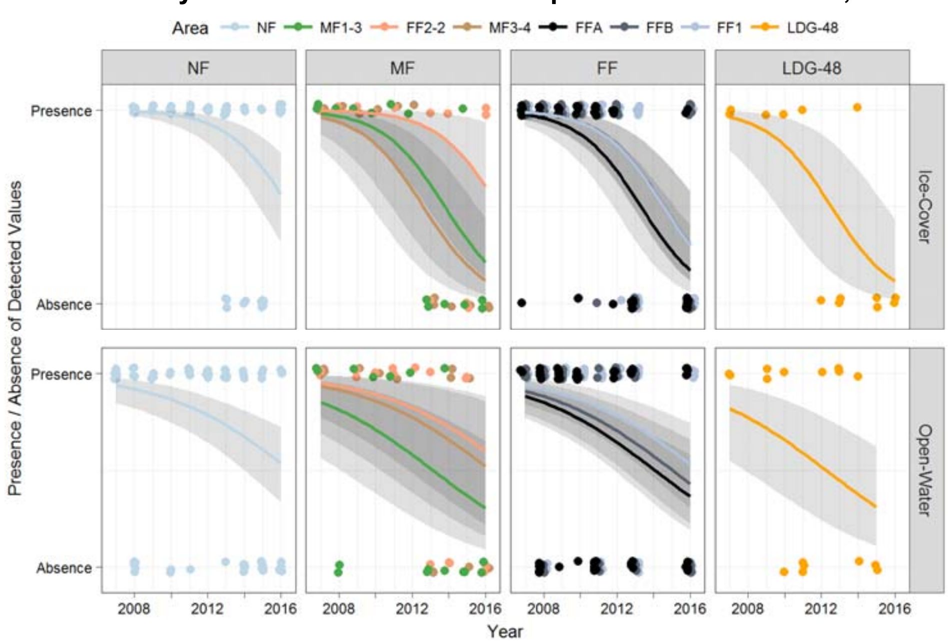
Note: NF = Near-field; MF = Mid-field; FF = Far-field; LDG = Lac de Gras

Figure 5-17 Concentrations of Total Dissolved Phosphorus in Lac de Gras during the Ice-cover (Mid-depth) and Open-water Seasons, 2007 to 2016



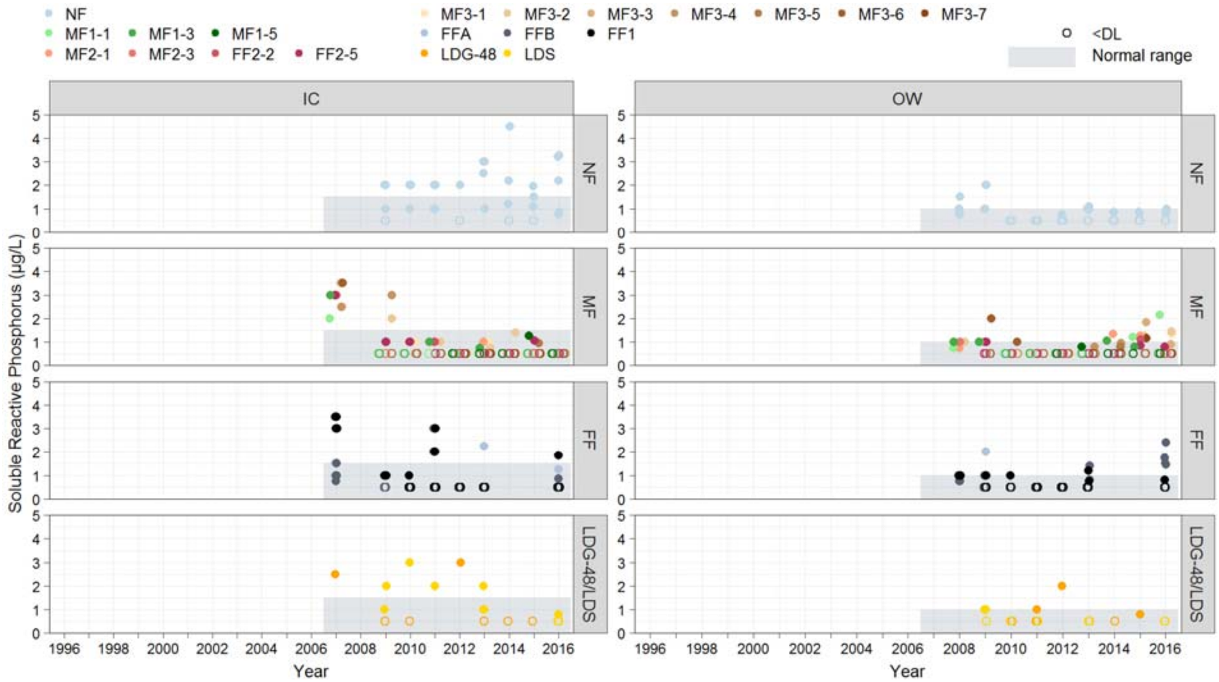
Note: NF = Near-field; MF = Mid-field; FF = Far-field; LDS = Lac du Sauvage; LDG = Lac de Gras

Figure 5-18 Trend Analysis for Total Dissolved Phosphorus Concentrations, 2007 to 2016



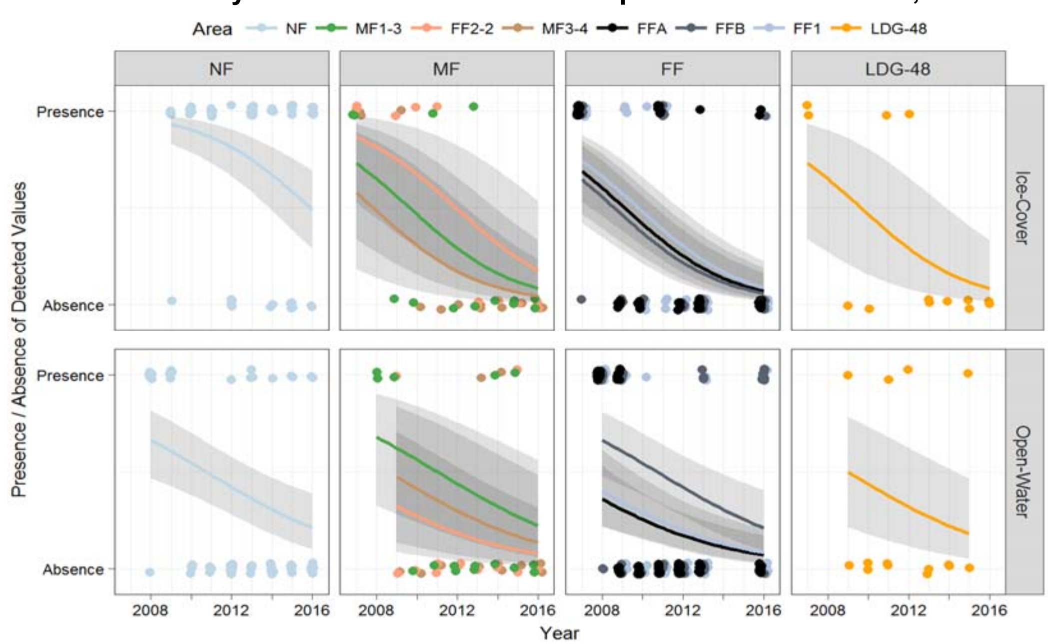
Note: NF = Near-field; MF = Mid-field; FF = Far-field; LDG = Lac de Gras

Figure 5-19 Concentrations of Soluble Reactive Phosphorus in Lac de Gras during the Ice-cover (Mid-depth) and Open-water Seasons, 2007 to 2016



Note: NF = Near-field; MF = Mid-field; FF = Far-field; LDS = Lac du Sauvage; LDG = Lac de Gras

Figure 5-20 Trend Analysis for Soluble Reactive Phosphorus Concentrations, 2007 to 2016



Note: NF = Near-field; MF = Mid-field; FF = Far-field; LDG = Lac de Gras

Nitrogen Variables

Temporal trends in TN differed among sampling areas and seasons in Lac de Gras between 2000 and 2016 (Figures 5-21 and 5-22; Table 5-12). During the ice-cover season, mid-depth TN concentrations in the NF and MF areas generally exceeded the normal range, whereas concentrations of TN in the FF areas were generally within the normal range, with the exception of 2016 (Figure 5-21; Appendix 5B Figures 5B-7 and 5B-8). Concentrations of TN at LDG-48 have remained mostly within the normal range under ice-cover. In 2010, TN concentrations in the NF area were significantly greater than concentrations measured in the MF and FF areas and from LDG-48 (Table 5-14). In 2013, TN concentrations the NF area were similar to concentrations observed at the MF1-3 and FF2-2 stations, but significantly greater than in the FF areas and at LDG-48. In 2016, the NF area was not significantly different from the other areas in Lac de Gras (Table 5-14). The parabolic models fitted to the ice-cover TN data suggest no apparent trends or shallow trends, with potential recent increases in 2016 in the FF areas only (Figure 5-22).

During the open-water season, TN concentrations typically exceeded the upper limit of the normal range in the NF and MF areas (Figure 5-21). Concentrations of TN in the FF areas were generally within the normal range during the open-water season from 2007 to 2012, although exceedances of the upper limit were observed before and after this period. The more recent elevated concentrations at several NF, MF and FF area stations may in part reflect the change in labs from UofA to Maxxam that occurred in 2013 (Golder 2016a). Concentrations of TN at LDG-48 were generally within the normal range in 2013; however, greater concentrations were observed between 2014 and 2016. In 2010 and 2013, TN concentrations in the NF area were significantly greater than the FF areas and at LDG-48, but were not different from the MF areas (Table 5-14). In 2016, the NF area was not significantly different from the other areas in Lac de Gras. The parabolic models fitted to the open-water TN data suggest no apparent trends or shallow trends, with potential recent increases in 2016 in the FF areas and LDG-48 (Figure 5-22).

Temporal trends in TDN differed among sampling areas and seasons in Lac de Gras between 2007 and 2016 (Figures 5-23 and 5-24; Table 5-12). Concentrations of TDN during the ice-cover season in the NF and MF areas generally exceeded the normal range, particularly in the MF1 and MF2 areas (Figure 5-23). In the FF areas, TDN concentrations were generally within the normal range until 2016, which may be driving the significantly increasing temporal trends observed in the FF areas between 2007 and 2016 (Figure 5-24; Table 5-12).

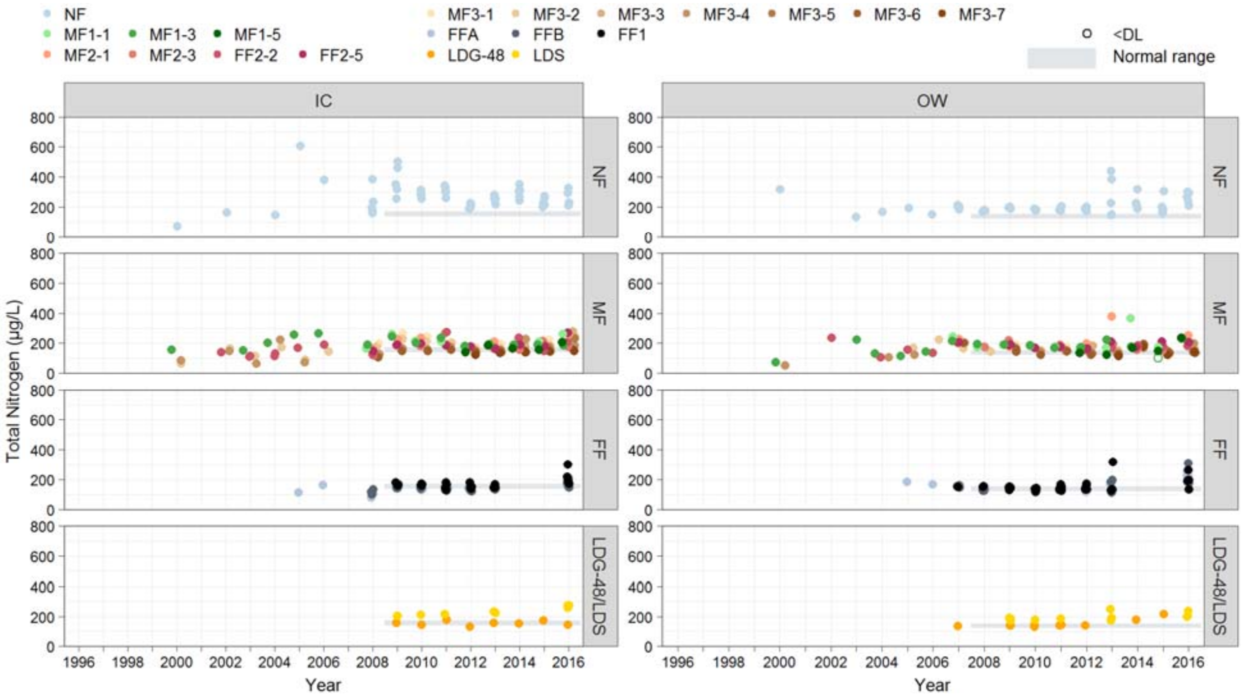
Total dissolved nitrogen concentrations during the open-water season generally followed the same patterns as those observed during the ice-cover season, with the exception of low TDN concentrations in 2013 in all areas (Figure 5-23). Concentrations generally exceeded the normal range in the NF, MF1 and MF2 areas, and were within the normal range in the FF areas, with the exception of greater concentrations observed in 2016. Concentrations of TDN at LDG-48 were generally within the normal range during the ice-cover season and during the open-water season from 2007 to 2012. From 2014 to 2016, concentrations of TDN at LDG-48 were above the upper limit of the normal range. In 2010 and 2013, TDN concentrations in the NF area were significantly greater than in the FF areas, while in 2016, TDN concentrations in the NF area did not differ from concentrations observed in the FF1 area or at LDG-48 (Figure 5-24; Table 5-14). The parabolic models fitted to the open-water TDN data suggest increasing trends in recent years in all sampling areas (Figure 5-24).

Table 5-14 Multiple Comparisons of Nutrient Trends within 2010, 2013 and 2016

Variable	Year	Season	Area							
			NF	MF1-3	FF2-2	MF3-4	FF1	FFB	FFA	LDG-48
Total nitrogen	2010	Ice-cover	d	cd	bc	abc	b	a	a	ab
	2013		c	bc	bc	b	ab	a	a	ab
	2016		a	a	a	a	a	a	a	a
	2010	Open-water	b	ab	ab	ab	a	a	a	a
	2013		c	abc	abc	abc	ab	a	ab	b
	2016		a	a	a	a	a	a	a	a
Total dissolved nitrogen	2010	Open-water	d	cd	d	bc	ab	a	a	ab
	2013		d	cd	cd	abc	ab	ab	a	bc
	2016		b	ab	ab	ab	ab	a	a	ab
Nitrate + Nitrite	2010	Ice-cover	d	cd	bc	abc	b	a	a	a
	2013		c	bc	b	b	b	a	a	a
	2016		c	abc	abc	abc	b	a	a	ab
	2010	Open-water	c	bc	bc	ab	a	a	a	a
	2013		c	ab	b	ab	a	a	a	a
	2016		b	a	a	ab	a	a	a	ab

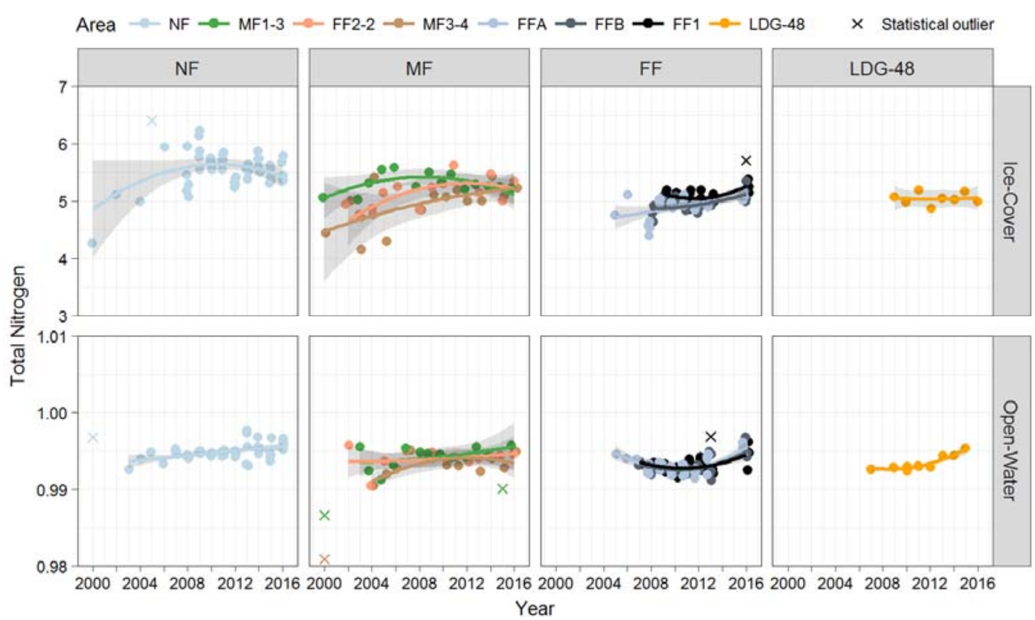
Notes: The different letters designate areas significantly different from on another at a *P*-value of 0.05; NF= near-field; MF = mid-field; FF = far-field.

Figure 5-21 Concentrations of Total Nitrogen in Lac de Gras during the Ice-cover (Mid-depth) and Open-water Seasons, 2000 to 2016



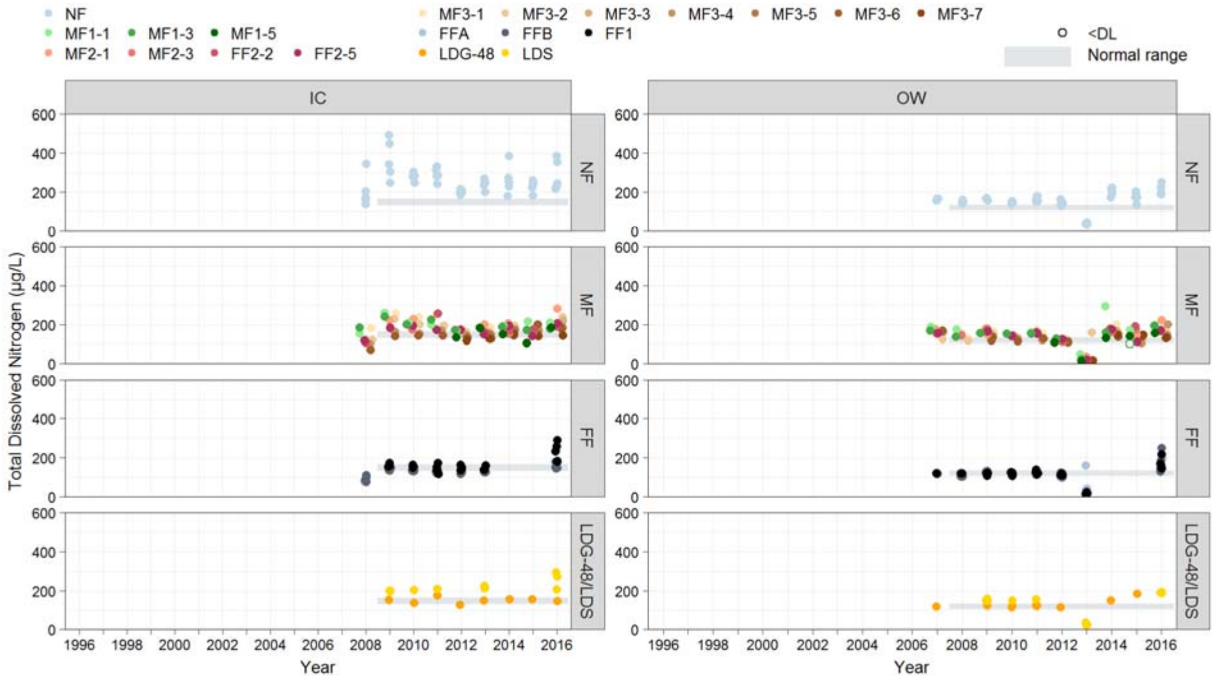
Note: NF = Near-field; MF = Mid-field; FF = Far-field; LDS = Lac du Sauvage; LDG = Lac de Gras

Figure 5-22 Trend Analysis for Total Nitrogen Concentrations, 2000 to 2016



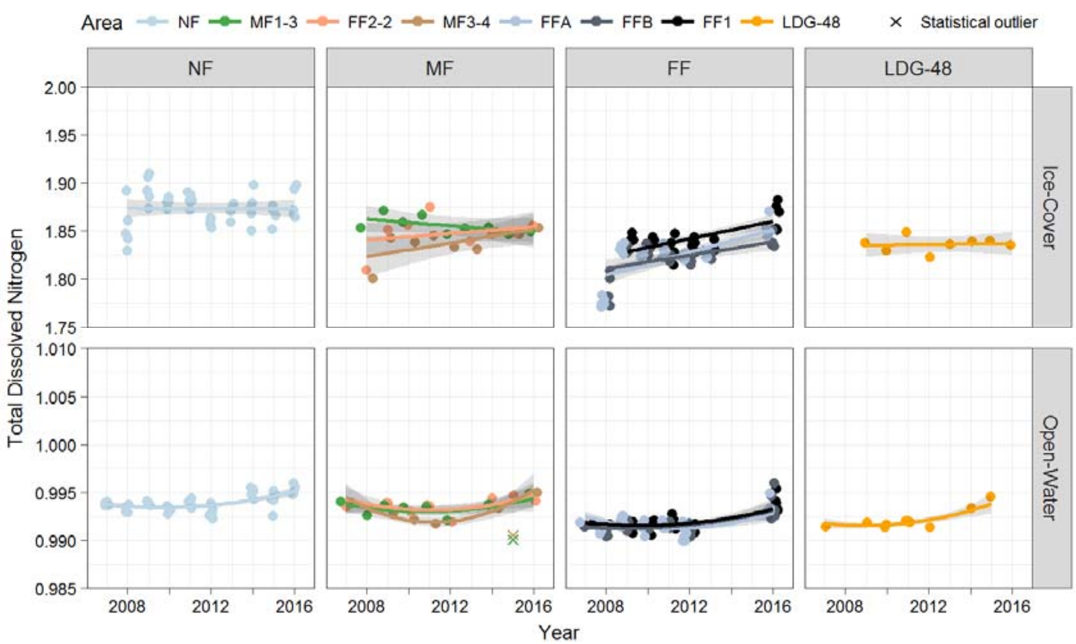
Note: NF = Near-field; MF = Mid-field; FF = Far-field; LDG = Lac de Gras

Figure 5-23 Concentrations of Total Dissolved Nitrogen in Lac de Gras during the Ice-Cover Season (Mid-depth), 2007 to 2016



Note: NF = Near-field; MF = Mid-field; FF = Far-field; LDS = Lac du Sauvage; LDG = Lac de Gras

Figure 5-24 Trend Analysis for Total Dissolved Nitrogen Concentrations, 2007 to 2016



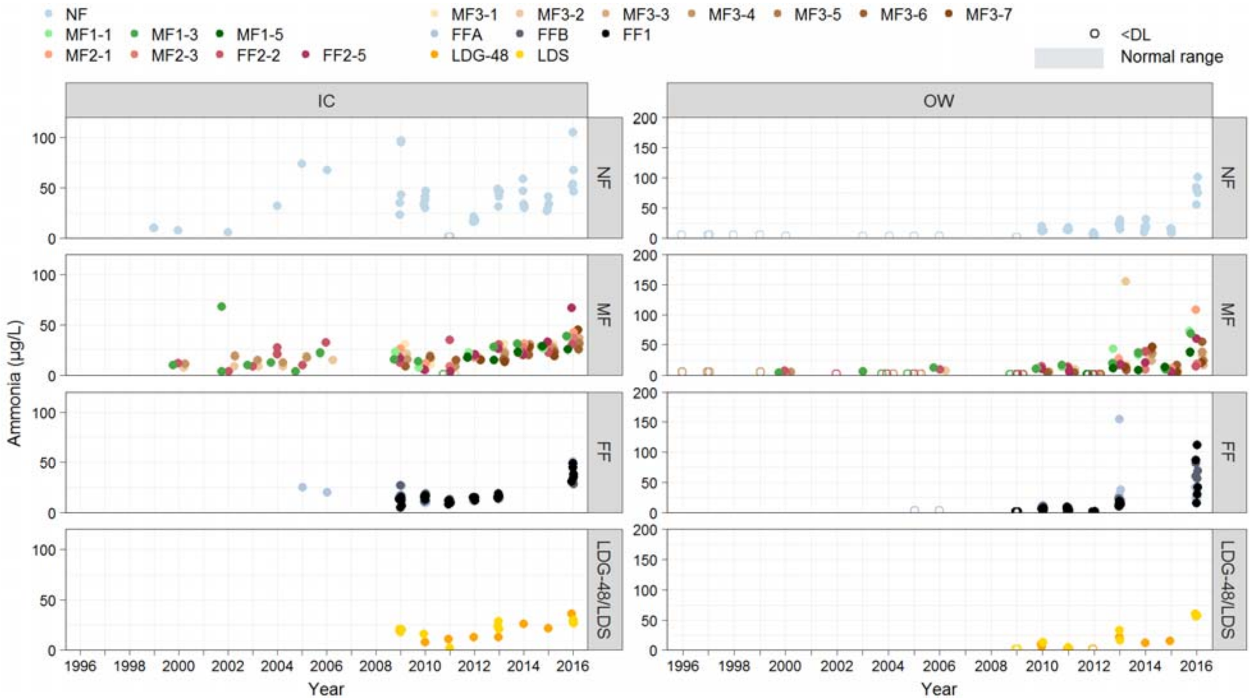
Note: NF = Near-field; MF = Mid-field; FF = Far-field; LDG = Lac de Gras

Lower TDN concentrations were reported during the open-water season in 2013 in all areas (Figure 5-23). The lower concentrations observed in most samples in 2013 can be attributed to the change in analytical laboratories in the summer of 2013 from UofA to Maxxam (Golder 2016a). Concentrations in 2014 to 2016 were also analyzed by Maxxam; however, concentrations were similar to or greater than those reported prior to 2013. Professional judgement suggests that the 2013 TDN data is anomalous; however, due to the nature of the method used for determining anomalous data (i.e., within-year comparisons) the data did not meet the criteria for exclusion. The 2013 open-water TDN data were retained on the temporal plots for visual evaluation of the data among years; however, they were excluded from the trend analysis (Figures 5-23 and 5-24, respectively).

The total ammonia data are presented in Figure 5-25 to allow visual review of the results. As discussed above, data were not compared to the normal range, or included in trend analyses. In general, slight increasing trends during the ice-cover season in the NF area between 2002, 2009, 2012, and 2016, and throughout the time series in the MF and FF areas, and LDG-48, are apparent upon visual examination of the plots. During the open-water season, a slight increasing trend was observed in all areas, driven by high total ammonia concentrations in 2016 compared to previous years. These potential trends should be interpreted with caution, given the analytical issues identified for this variable. For example, greater variation in recent years related to laboratory data quality issues result in visually apparent increasing trends, which may not be accurate representations of the variation over time in total ammonia concentration.

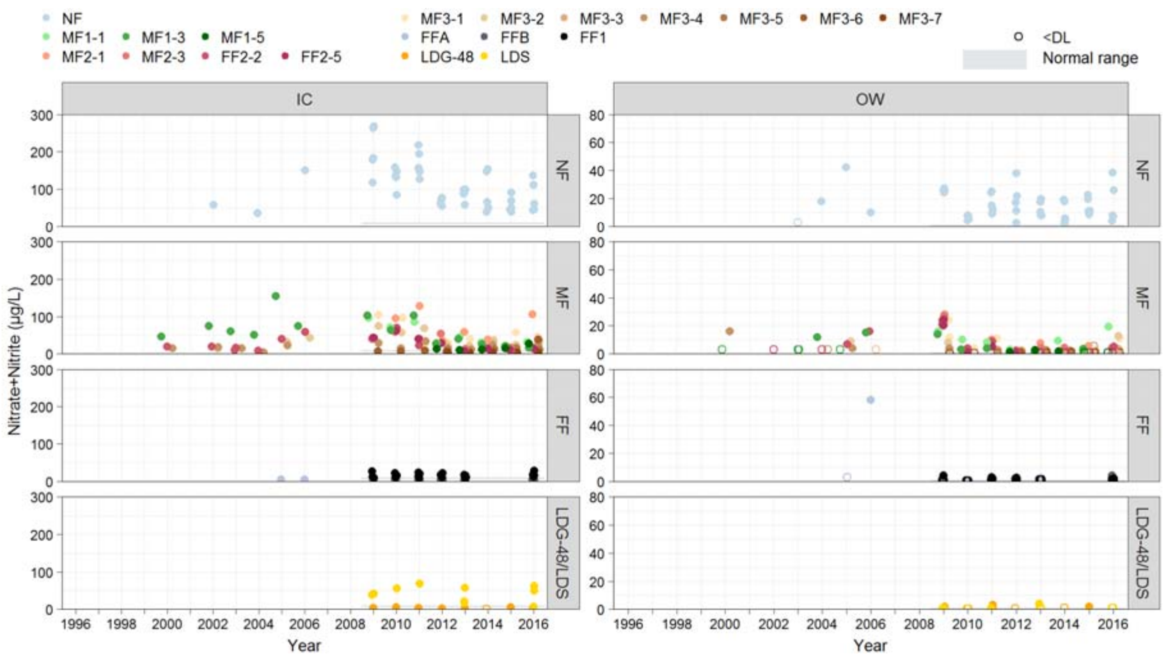
Temporal trends in N+N concentrations differed among sampling areas and seasons in Lac de Gras between 2000 and 2016 (Figures 5-26 and 5-27; Table 5-12). Concentrations of N+N in the NF area have been consistently greater than the normal range during the ice-cover and open-water seasons between 2007 and 2016 (Figures 5-26, Appendix 5B Figures 5B-13 and 5B-14). Concentrations in the FF areas also frequently exceeded the normal range during both sampling seasons, but to a much lower extent than observed in the NF area. Concentrations of N+N at LDG-48 were within or below the normal range. During the ice-cover season, concentrations of N+N in the NF area were significantly greater than concentrations in the FF areas in 2010, 2013 and 2016 (Figure 5-27; Table 5-14). During the open-water season, concentrations of N+N in the NF were significantly greater than concentrations in the FF areas and at LDG-48 in 2010, 2013 and 2016, with the exception of at LDG-48 in 2016 which had N+N concentrations similar to concentrations observed in the NF area (Figure 5-27; Table 5-14). The parabolic models fitted to the TDN data suggest no trends or declining trends in the NF, MF and FF areas, and increasing trends in recent years at LDG-48 (Figure 5-27).

Figure 5-25 Concentrations of Total Ammonia in Lac de Gras during the Ice-cover (Mid-depth) and Open-water Seasons, 2000 to 2016



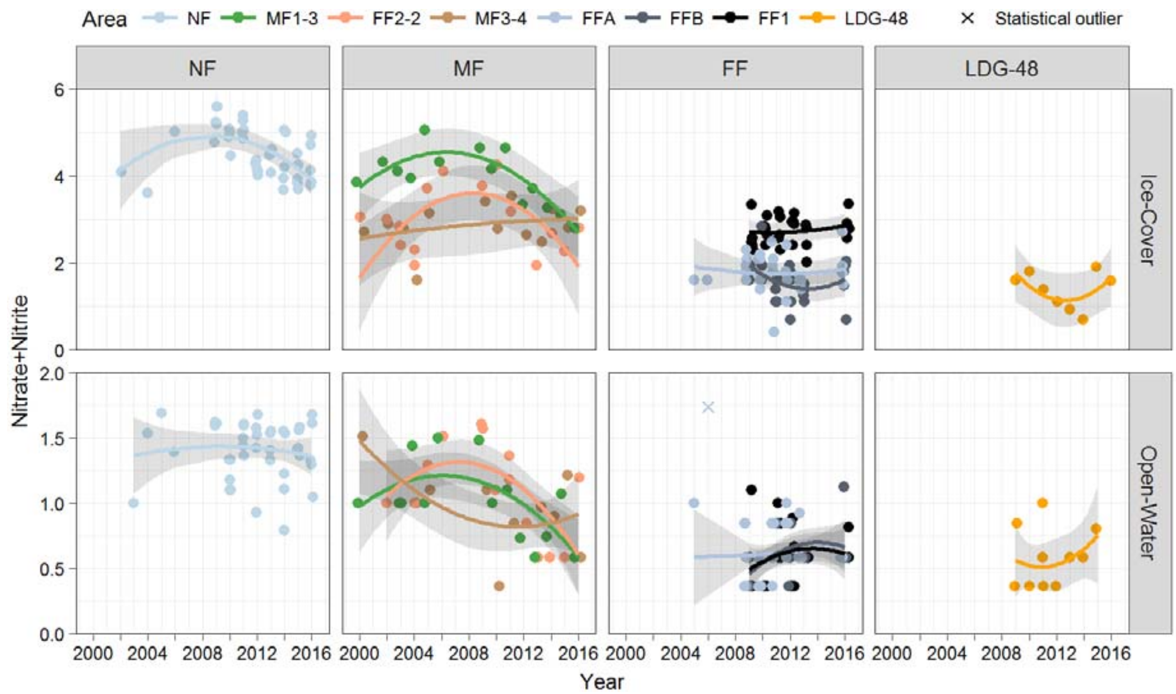
Note: NF = Near-field; MF = Mid-field; FF = Far-field; LDS = Lac du Sauvage; LDG = Lac de Gras

Figure 5-26 Concentrations of Nitrate + Nitrite in Lac de Gras during the Ice-cover (Mid-depth) and Open-water Seasons, 2000 to 2016



Note: NF = Near-field; MF = Mid-field; FF = Far-field; LDS = Lac du Sauvage; LDG = Lac de Gras

Figure 5-27 Trend Analysis for Nitrate + Nitrite Concentrations, 2000 to 2016



Note: NF = Near-field; MF = Mid-field; FF = Far-field; LDG = Lac de Gras

At the LDS stations, nutrient concentrations were generally similar to those observed in the FF areas of Lac de Gras, but less than those observed in the NF area. Concentrations of TP in Lac de Sauvage were greater than in the FF areas of Lac de Gras in 2010 and 2013, during both the ice-cover and open-water seasons (Figure 5-15). Total dissolved phosphorus concentrations were greater during the ice-cover season at the LDS stations compared to the FF areas of Lac de Gras; during the open-water season concentrations were similar among areas (Figure 5-17). Concentrations of SRP have been frequently below the DL, since 2007, particularly in the open-water season (Figure 5-19). Concentrations of TN, TDN, total ammonia and N+N were similar between the FF areas of Lac de Gras and the LDS stations (Figures 5-21 to 5-26). Similar to the FF areas of Lac de Gras, total ammonia concentrations were elevated at the LDS stations in 2016 (Figure 5-25).

Nutrient Summary

The concentration of TP has generally remained within, or just above, the upper limit of the normal range throughout Lac de Gras between 2007 and 2016. Temporal trends in TP, TDP and SRP were observed under ice-cover and during the open-water season. Mid-depth concentrations of TP during the ice-cover season have significantly decreased in all areas of Lac de Gras, including at the lake outlet (i.e., LDG-48), between 2007 and 2016. During the open-water season, significant decreases were observed in the NF and FFB areas of Lac de Gras.

Concentrations of TP in the NF area were slightly greater during the ice-cover season compared to the open-water season, while in the MF and NF areas, TP concentrations were generally similar between seasons. Concentrations of TP at several stations in the MF1, MF2-FF2 and MF3 areas, and in the FF areas during the ice-cover season exceeded the normal range between 2007 and 2013. Between 2014 and 2016, concentrations were generally within the normal range or below the lower limit.

Concentrations of TDP and SRP have been frequently below the DL. Significant differences were identified among areas and over time, and results indicated declining trends in the areas and stations analyzed (i.e., greater frequency of non-detect values in recent years).

Concentrations of TN typically exceeded the upper limit of the normal range in the NF and MF areas in both seasons. Total nitrogen concentrations in the FF areas were generally within the normal range from 2007 to 2012, but have increased above the normal range in 2013 and 2016, which may in part be caused by the change in analytical laboratory 2013. Temporal trends in TN, TDN and N+N differed among sampling areas and seasons in Lac de Gras between 2000 and 2016. During the ice-cover season, mid-depth TN concentrations in the NF and MF areas generally exceeded the normal range, whereas concentrations of TN in the FF areas were generally within normal range, with the exception of 2016.

Concentrations of TDN during the ice-cover season in the NF and MF areas, particularly those of the MF1 and MF2 areas, generally exceeded the normal range. In the FF areas, TDN concentrations were generally within the normal range, with the exception of high concentrations that were above the normal range in 2016. Concentrations of TDN during the open-water season generally followed the same patterns as those observed during the ice-cover season, with the exception of low TDN concentrations in 2013 in all areas. The lower concentrations observed in most samples in 2013 can be attributed to the change in analytical laboratories in the summer of 2013 from UofA to Maxxam (Golder 2016a).

Data quality issues with analysis of low levels of total ammonia occurred from 2013 to 2016. Therefore, observed trends in ammonia in recent years are subject to uncertainty. In general, slight increasing trends were observed during the ice-cover season in the NF area between 2002 and 2009, and 2012 and 2016. During the open-water season, a slight increasing trend was observed in all areas, driven by high total ammonia concentrations in 2016 compared to previous years.

Concentrations of N+N in the NF have been consistently greater than the normal range during the ice-cover and open-water seasons between 2007 and 2016. Concentrations in the FF areas also frequently exceeded the normal range during both sampling seasons, but to a much lower extent than observed in the NF area. During the open-water season, concentrations of N+N in the NF were significantly greater than concentrations in the FF areas and at LDG-48 in 2010, 2013 and 2016, with the exception of at LDG-48 in 2016, which had N+N concentrations similar to concentrations observed in the NF area.

A number of conclusions relevant to Mine-related effects in Lac de Gras can be drawn based on the review of temporal trends in nutrient loadings and concentrations:

- Nutrient concentrations remain low throughout Lac de Gras, within the oligotrophic ranges for both P and N.
- The large increases in annual loadings of all forms of P from the Mine effluent between 2008 and 2012 and thereafter (i.e., 2-fold for TP; 5 to 9-fold for TDP; 10 to 15-fold for SRP) did not translate to increasing trends in TP concentrations in lake water, even in the NF area.
- Loadings of N have declined by approximately 2-fold for most N-variables, since the maximum loads observed between 2006 and 2008; however, no trends or only slight increasing trends were observed in the NF and MF areas, while greater increases were apparent in the FF areas and LDG-48, where concentrations are now slightly above the normal ranges. These results suggest that the greater load of N discharged to the lake before 2008 has now reached the western portion of the lake.

- Although dike construction has affected TP concentrations in the MF3 area in 2015, and especially in 2016, the effect was small and did not noticeably influence trends on the time series plots, or the trends detected by statistical analysis.

5.3.4.3 Chlorophyll *a* Concentrations, and Phytoplankton and Zooplankton Biomass

Trend analysis of chlorophyll *a* data was done using a linear model, whereas parabolic models were selected for the trend analysis of phytoplankton and zooplankton biomass (Table 5-15).

Table 5-15 Biological Variables for Eutrophication Indicators AIC Model Selection

Variable	Years	Yeo-Johnson transformation[(a)]	Parabolic AIC Score	Linear AIC Score
Chlorophyll *a* concentrations	2007 to 2016	0	-218	-229
Total phytoplankton biomass	2002 to 2012	0	186	238
Total zooplankton biomass (as AFDM)	2009 to 2016	0	113	126

Note: Grey shaded cells indicate the model chosen. AIC = Akaike's information criterion.
a) See Section 2.4.2.1 for details on transformation approach.

Temporal trends in chlorophyll *a* concentrations differed among sampling areas in Lac de Gras between 2007 and 2016 (Figures 5-28 and 5-29; Table 5-16). Chlorophyll *a* concentrations in the NF area generally ranged between 1 and 2 µg/L and exceedances of the normal range occurred in all years (Figure 5-28). Concentrations in the MF areas were generally the same or slightly less than concentrations observed in the NF area. At MF3-4, chlorophyll *a* concentrations increased significantly between 2007 and 2016 (Figure 5-29 and Table 5-16). Concentrations in the FF area and the LDG-48 station were typically within or at the lower limit of the normal range. Chlorophyll *a* concentrations increased significantly between 2007 and 2016 in the FF1 and FFB areas, but not in the FFA area or at LDG-48, suggesting that the biological effect of nutrient enrichment from the Mine had not reached the FFA area by 2016.

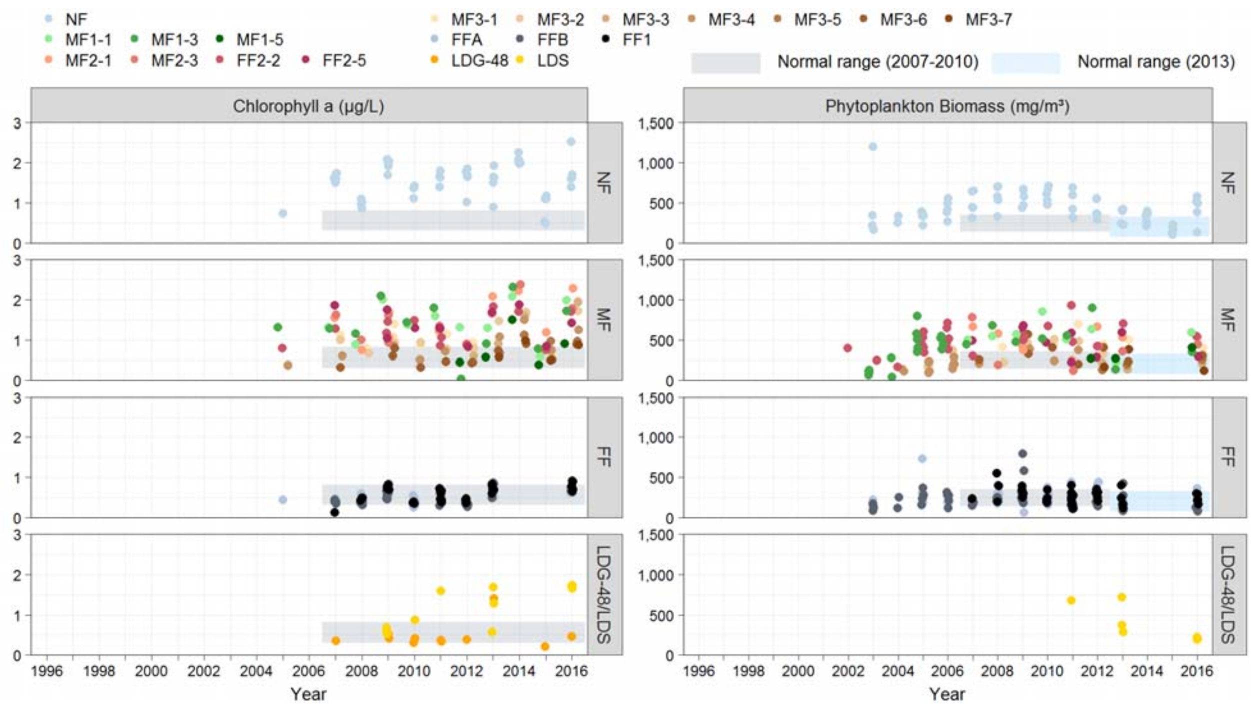
Temporal trends in total phytoplankton biomass differed among sampling areas in Lac de Gras between 2002 and 2012 (Figures 5-28, 5-30 and 5-31; Table 5-16). In response to commitments from the *Design Plan Version 4.0*, phytoplankton biomass was compared to the normal range using boxplots for each area and year (Figure 5-30), in addition to the temporal trend plots. Data comparisons provided in the box-plots agree with those shown in the temporal trend plots; therefore, interpretation of the data is based on the temporal plots.

Phytoplankton biomass increased in the NF area between 2003 and 2010, peaked in 2010, and then decreased until 2015 (Figures 5-28 and 5-30). Between 2015 and 2016, biomass increased in the NF area with significantly greater biomass in 2016 compared to 2013 (Figure 5-32 and Table 5-18). Biomass exceeded the upper limits of the 2007 to 2010 normal range at all or most stations in the NF area from 2006 to 2012. From 2011 to 2013, phytoplankton biomass decreased, and in 2013 and 2014 phytoplankton biomass was within or near the upper limit of both the 2007 to 2010 and 2013 normal ranges. In 2015, biomass at stations in the NF area was at or below the lower limit of the 2007 to 2010 normal range, but within the 2013 normal range. With the increase between 2015 and 2016, biomass at stations in the NF area was generally at or above the upper limit of both the 2007 to 2010 and 2013 normal ranges.

Biomass at the majority of stations in the MF1, MF2-FF2, and MF3 areas followed a similar pattern to that seen in the NF area (Figures 5-28, 5-30 and 5-31). Similarly, biomass in the FF areas followed comparable trends to the NF and MF areas, but stations in the FF areas were generally within the 2007 to 2010 and 2013 normal ranges (Figure 5-28). In 2004, phytoplankton biomass at stations in the FF1 and FFA areas was not significantly different than that in the NF area, but in 2008 biomass in the NF area was significantly greater than in the FF areas (Table 5-19). In 2012, biomass in the NF area was significantly greater than in the FFA and FFB areas.

At the LDS stations, phytoplankton biomass was generally similar to biomass in the FF areas of Lac de Gras with the exception of 2011, and one of three stations in 2013, which had greater biomass. Chlorophyll *a* concentrations at the LDS stations were generally greater than concentrations measured in the FF areas of Lac de Gras.

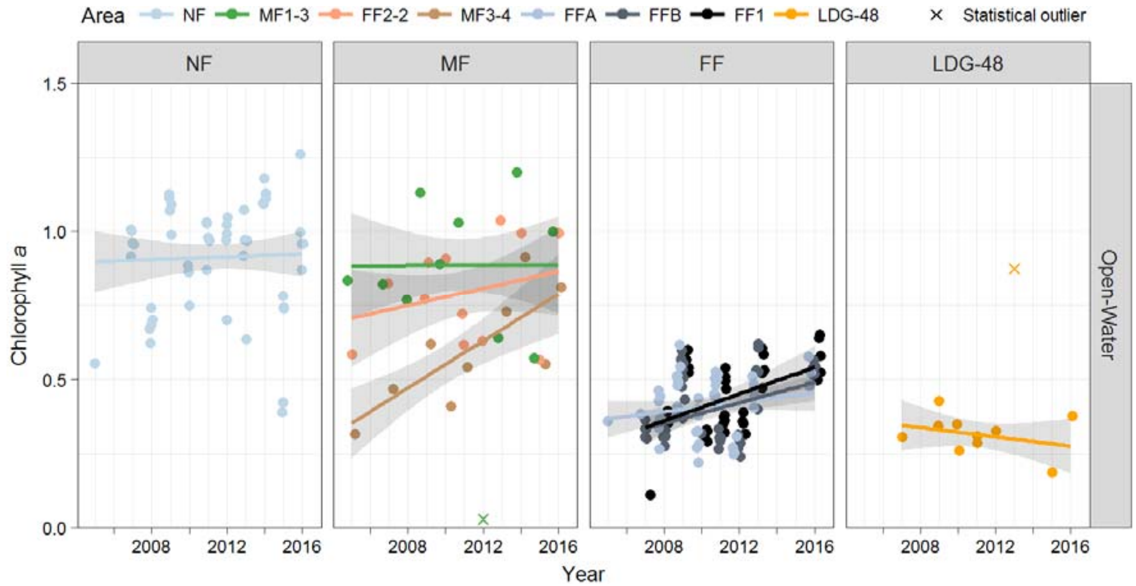
Figure 5-28 Concentration of Chlorophyll *a* and Total Phytoplankton Biomass in Lac de Gras during the Open-water Season, 2002 to 2016



Note: Values represent concentrations in individual samples.

NF = near-field; MF = mid-field; FF = far-field; LDG-48 = Lac de Gras outlet; LDS = Lac du Sauvage.

Figure 5-29 Trend Analysis for Chlorophyll *a* Concentrations, 2005 to 2016



Note: NF = Near-field; MF = Mid-field; FF = Far-field

Table 5-16 Significance of Biological Variables for Eutrophication Indicators Fixed Effects Models, 2002 to 2016

Variable	Years	Coefficient	Numerator DF	Denominator DF	F-value	*P*-value
Chlorophyll *a* concentration	2005 to 2016	Area	7	18	2.91	0.032
		Year	1	190	14.46	<0.001
		Area × Year	7	190	2.90	**0.007**
Total phytoplankton biomass	2002 to 2012	Area	6	19	2.30	0.031
		Year	1	198	10.84	0.001
		$Year^2$	1	198	10.85	0.001
		Area × Year	6	198	2.99	**0.008**
		Area × $Year^2$	6	198	2.99	**0.008**
Total zooplankton biomass (as AFDM)	2009 to 2016	Area	6	16	2.23	0.094
		Year	1	126	13.50	<0.001
		$Year^2$	1	126	13.49	<0.001
		Area × Year	6	126	2.23	**0.044**
		Area × $Year^2$	6	126	2.23	**0.044**

Notes: **Bolded** values indicate significance; AFDM = ash-free dry mass; DF = degrees of freedom; F-value = F-statistic; *P*-value = probability value.

Table 5-17 Estimated Significance of Difference of Linear slopes from zero

Variable	Area							
	NF	MF1-3	FF2-2	MF3-4	FF1	FFB	FFA	LDG-48
Chlorophyll *a* concentration	0.744	0.981	0.247	**<0.001** ↑	**0.001** ↑	**0.001** ↑	0.134	0.360

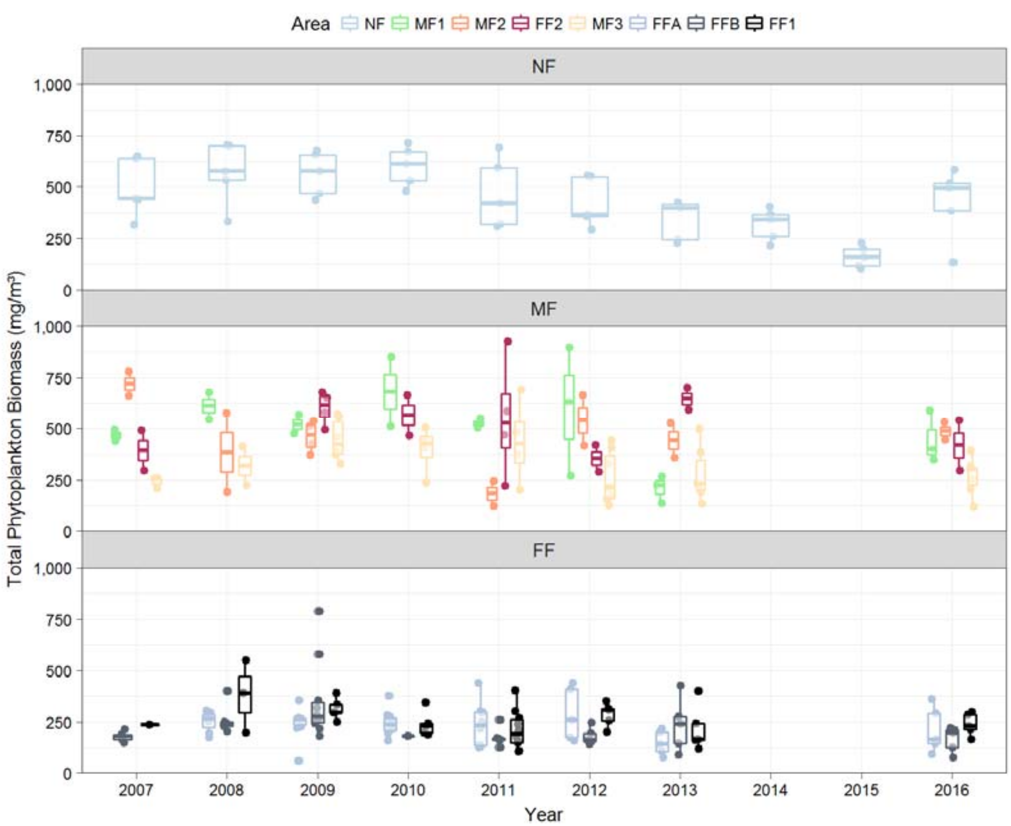
Notes: The arrow indicates direction of trend; IC = ice-cover; OW = open-water; NF= near-field; MF = mid-field; FF = far-field; LDG = Lac de Gras; <= less than.

Table 5-18 Multiple Comparisons of Parabolic Trends in the NF area within 2013 and 2016

Variable	2013	2016
Total phytoplankton biomass	b	a

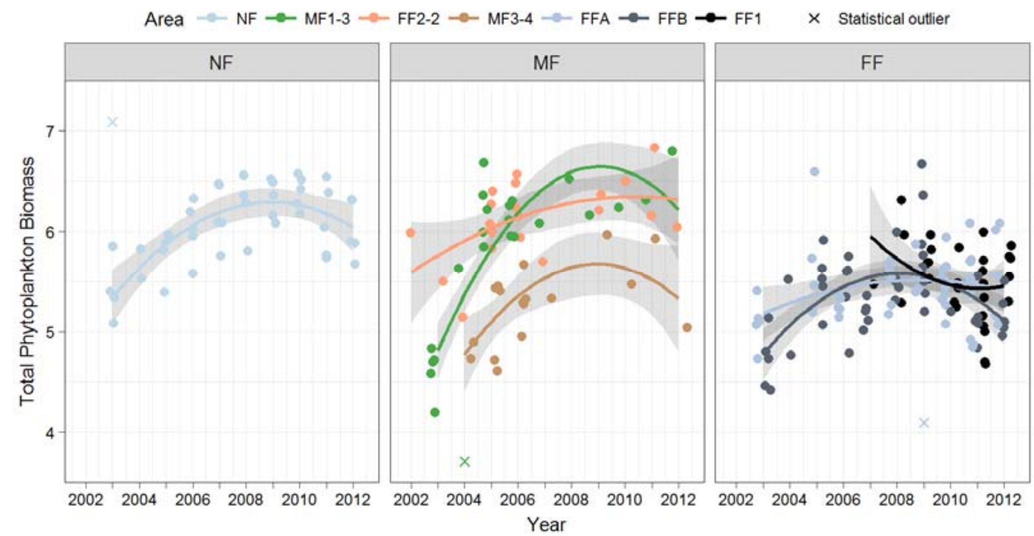
Notes: The different letters designate areas significantly different from on another at a *P*-value of 0.05; NF= near-field; MF = mid-field; FF = far-field.

Figure 5-30 Phytoplankton Biomass in Lac de Gras, 2002 to 2016



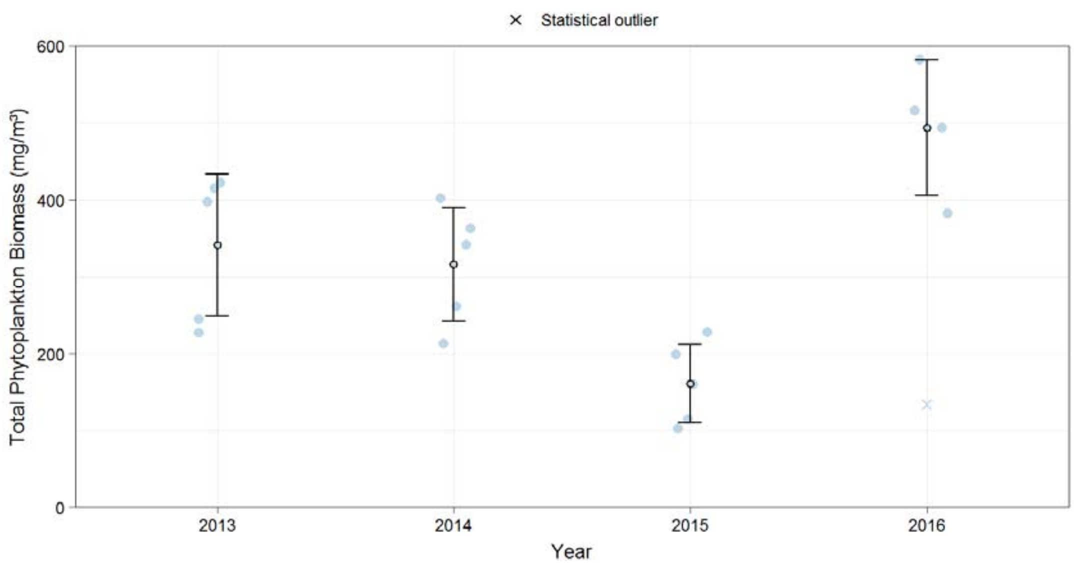
Note: NF = Near-field; MF = Mid-field; FF = Far-field

Figure 5-31 Trend Analysis for Total Phytoplankton Biomass, 2002 to 2016



Note: NF = Near-field; MF = Mid-field; FF = Far-field

Figure 5-32 Trend Analysis for Phytoplankton Variables in the NF Area, 2013 to 2016



Note: Error bars represent 95% confidence interval.

Table 5-19 Multiple Comparisons of Parabolic Trends in Total Phytoplankton and Zooplankton (as AFDM) Biomass

Variable	Year	Area						
		NF	MF1-3	FF2-2	MF3-4	FF1	FFB	FFA
Total phytoplankton biomass	2004	bc	ab	c	a	abc	a	ab
	2008	b	b	b	a	a	a	a
	2012	cd	bcd	d	abcd	abc	a	ab
Total zooplankton biomass	2010	c	abc	bc	abc	ab	ab	a
	2013	b	b	ab	ab	a	a	a
	2016	a	a	a	a	a	a	a

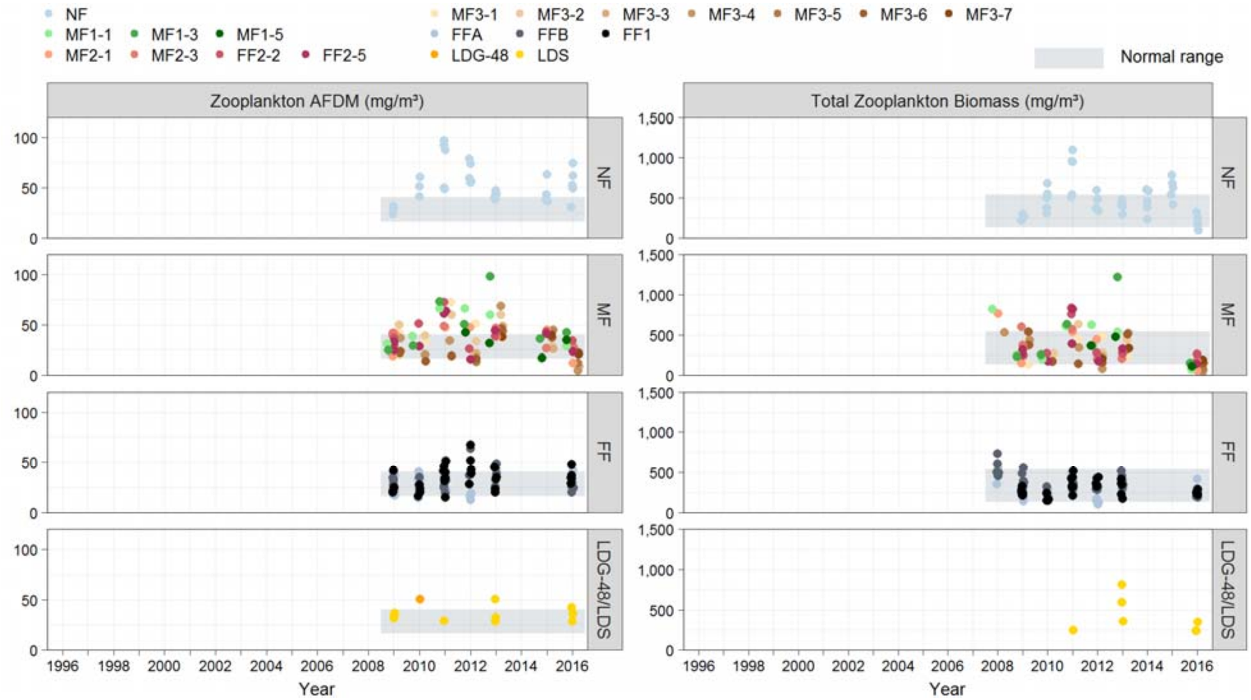
Notes: The different letters designate areas significantly different from on another at a *P*-value of 0.05; NF= near-field; MF = mid-field; FF = far-field.

Temporal trends in total zooplankton biomass (as AFDM) differed among sampling areas in Lac de Gras between 2009 and 2016 (Figures 5-33 and 5-34; Table 5-16). Zooplankton biomass in the NF area varied throughout the monitoring period, with biomass as AFDM generally exceeding the normal range from 2010 to 2016 at most stations. Zooplankton biomass increased in the NF area between 2008 and 2011, peaking in 2011 and then decreased until 2013. Between 2013 and 2015, biomass remained similar in the NF area; however, between 2015 and 2016 biomass increased. Zooplankton biomass in the MF and FF areas was generally within or above the upper limit of the normal range, with the exception of smaller zooplankton biomass in the MF3 area in 2016.

Zooplankton biomass based on enumeration increased in the NF area between 2008 and 2011, peaking in 2011 and then decreasing until 2013 (Figure 5-33). Between 2013 and 2015, biomass increased in the NF area; however, between 2015 and 2016 biomass decreased. Zooplankton biomass exceeded the upper limit of the normal range at stations in the MF2-FF2 area in 2008, 2009, and 2011, and in the MF1 area in 2008, 2011, 2012 and 2013. However, in 2016, zooplankton biomass in the MF areas was at or below the lower limit of the normal range (Figure 5-33; Section 7). In the MF3 area, stations were generally within the normal range or just below the lower limits from 2008 to 2016.

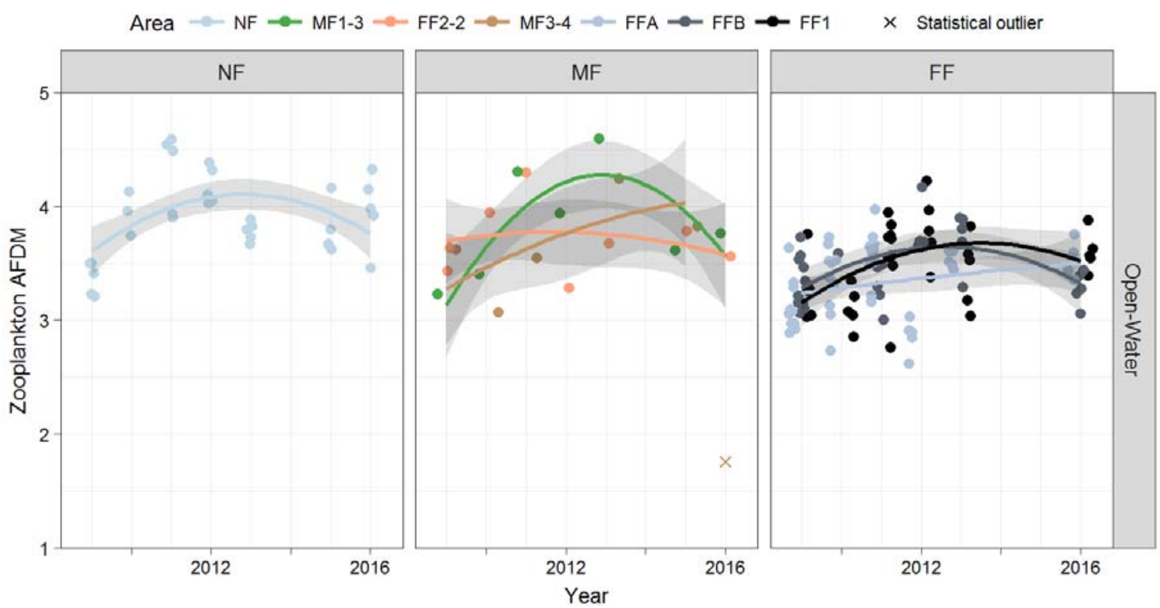
Temporal variation in zooplankton biomass measured as AFDM and based on enumeration was similar between 2008 and 2015, but diverged in 2016, with a decline apparent in all areas in the dataset based on enumeration, but only in the MF3 area based on AFDM (Figure 5-33). The reason for this divergence is unknown, as no data quality issues were apparent in either dataset.

Figure 5-33 Total Zooplankton Biomass and Zooplankton AFDM in Lac de Gras during the Open-water Season, 2009 to 2016



Note: AFDM = ash-free dry mass; mg/m^3 = milligrams per cubic metre; NF = near-field; MF = mid-field; FF = far-field; LDG-48 = Lac de Gras outlet.

Figure 5-34 Trend Analysis for Zooplankton Biomass (as AFDM), 2009 to 2016



Note: NF = Near-field; MF = Mid-field; FF = Far-field

Biological Variable Summary

Temporal trends in chlorophyll *a* concentrations and phytoplankton biomass differed among sampling areas in Lac de Gras between 2007 and 2016. Chlorophyll *a* concentrations in the NF area exceeded the normal range in all years. Concentrations in the MF areas were generally the same or slightly less than concentrations observed in the NF area; in the FF areas, concentrations were typically within or at the lower limit of the normal range. Phytoplankton biomass varied over time in the NF and MF areas of Lac de Gras from within the normal range to both above (2016) and below (2015) in recent years.

Temporal trends in total zooplankton biomass measured as AFDM differed among sampling areas in Lac de Gras between 2009 and 2016. Zooplankton biomass in the NF area has varied throughout the time series, generally exceeding the normal range from 2010 to 2016 at most stations. Biomass in the MF and FF areas was generally within or above the upper limit of the normal range, with the exception of low biomass in the MF3 area in 2016. Temporal variation in zooplankton biomass measured as AFDM and by enumeration was similar between 2008 and 2015, but diverged in 2016, with a decline apparent in all areas in the dataset based on enumeration, but only in the MF3 area based on AFDM.

A number of conclusions relevant to Mine-related effects in Lac de Gras can be drawn based on the review of temporal trends in biological indicators of nutrient enrichment:

- Biological indicators of eutrophication provide evidence of effects related to nutrient enrichment in the NF and MF areas.
- The large increases in annual loadings of all forms of P from the Mine effluent between 2008 and 2012 and thereafter did not result in similar increasing trends in biological indicators.
- Chlorophyll *a* concentrations and plankton biomass displayed high variability in recent years, especially since 2014, without an obvious explanation in terms of nutrient concentrations that would account for this observation.
- MF3 area results for 2016 indicated a potential effect of dike construction on zooplankton biomass (apparent when considering all MF3 stations; Golder 2017c).
- Although the two zooplankton biomass data sets (i.e., AFDM and enumeration) show similar temporal trends over time, there was a divergence between them in 2016, for which there is no current explanation.

5.3.5 Relationships between Biological Variables and Nutrient Variables

The WLWB has requested that as part of the *2014 to 2016 Aquatic Effects Re-evaluation Report,* an integrative assessment of nutrient, chlorophyll *a* and plankton variables be conducted. The assessment is to include:

- continued evaluation of the relationship between chlorophyll *a* concentrations and phytoplankton biomass (EMAB commitment #9 from the 2015 AEMP Annual Report [Table 1-1])
- an evaluation of the relationships between nutrients, chlorophyll *a* and phytoplankton biomass (EMAB commitment #11; WLWB Directive and Reasons for Decisions W2015L2-0001 Schedule 8 Update; and Commitment #47 from 2016 AEMP Annual Report [Table 1-1])

- an evaluation of the role of TN in explaining variation in chlorophyll *a* concentrations (EMAB commitment #61; WLWB Directive and Reasons for Decisions W2015L2-0001 Schedule 8 Update [Table 1-1])
- calculation of nutrient ratios and evaluation of trends over time (EMAB commitment #61; WLWB Directive and Reasons for Decisions W2015L2-0001 Schedule 8 Update; and (EMAB commitment #25 from the 2015 AEMP Annual Report [Table 1-1])
- an evaluation of the role of TN and the ratio of N:P in explaining variation in phytoplankton biomass (Section 7) and chlorophyll *a* concentrations (EMAB commitments #61 and 94; WLWB Directive and Reasons for Decisions W2015L2-0001 Schedule 8 Update [Table 1-1])

The assessment includes a comparison of TDS (as an indicator of exposure to Mine effluent in Lac de Gras) and nutrients (TN, TP, and the N to P molar ratio) to chlorophyll *a* concentrations.

5.3.5.1 Relationships between Biological Variables

An evaluation of the relationship between concentrations of chlorophyll *a* and phytoplankton biomass was provided in the *2007 to 2010 Summary Report* (Golder 2011a). Chlorophyll *a* concentration was plotted against phytoplankton biomass (based on biovolume) to explore the utility of chlorophyll *a* as a phytoplankton biomass indicator. These plots indicated that the relationship between chlorophyll *a* concentration and phytoplankton biomass was poor in all sampling areas (r values ranging from 0.17 to 0.33). A re-evaluation of this relationship using data updated to 2016 indicated that the overall relationship between chlorophyll *a* concentrations and phytoplankton biomass remains poor ($r = 0.52$; Figure 5-35). However, evaluation of the relationship by year (rather than by area, as was completed in the *2007 to 2010 Summary Report*; Golder 2011a) indicated moderate to strong relationships between chlorophyll *a* concentrations and phytoplankton biomass between 2007 and 2016 ($r > 0.60$; Table 5-20), with the exception of in 2011 ($r = 0.52$).

An evaluation of the relationship between zooplankton biomass measured as AFDM and based on enumeration indicated a moderately strong overall relationship ($r = 0.75$; Figure 5-36). Evaluation of the relationship within individual years also indicated moderate to strong relationships between 2009 and 2016 (r^2 values ranging from 0.55 to 0.89; Table 5-20). In 2016, a number of samples in the NF and MF1 areas showed poor agreement between zooplankton biomass measured as AFDM and based on enumeration (Figure 5-36). In these samples, biomass as AFDM was notably greater than that based on enumeration, compared to other samples collected in 2016 and the overall relationship based on all years of data. These results suggest that the samples analyzed for AFDM contained additional particulate organic material. The reasons for this discrepancy cannot be explained based on available data.

Table 5-20 Relationship between Chlorophyll *a* Concentrations and Phytoplankton Biomass (2007 to 2016), and the Relationship between Zooplankton Biomass Measured as AFDM and as Enumeration (2009 to 2016)

Year	Chlorophyll *a vs* Phytoplankton Biomass		Zooplankton Biomass: AFDM *vs* Enumeration	
	n	*r*	*n*	*r*
2007	20	0.81	-	-
2008	26	0.69	-	-
2009	46	0.61	46	0.57
2010	27	0.86	27	0.67
2011	42	0.55	40	0.84
2012	30	0.62	31	0.89
2013	33	0.69	32	0.55
2014	-	-	-	-
2015	-	-	-	-
2016	34	0.73	34	0.62

r = Pearson correlation co-efficient; *n* = sample size; vs = versus; - = no data; Chl *a* = chlorophyll *a*; AFDM = ash-free dry mass.

Figure 5-35 Relationship Between Total Phytoplankton Biomass and Concentrations of Chlorophyll *a*, 2007 to 2016

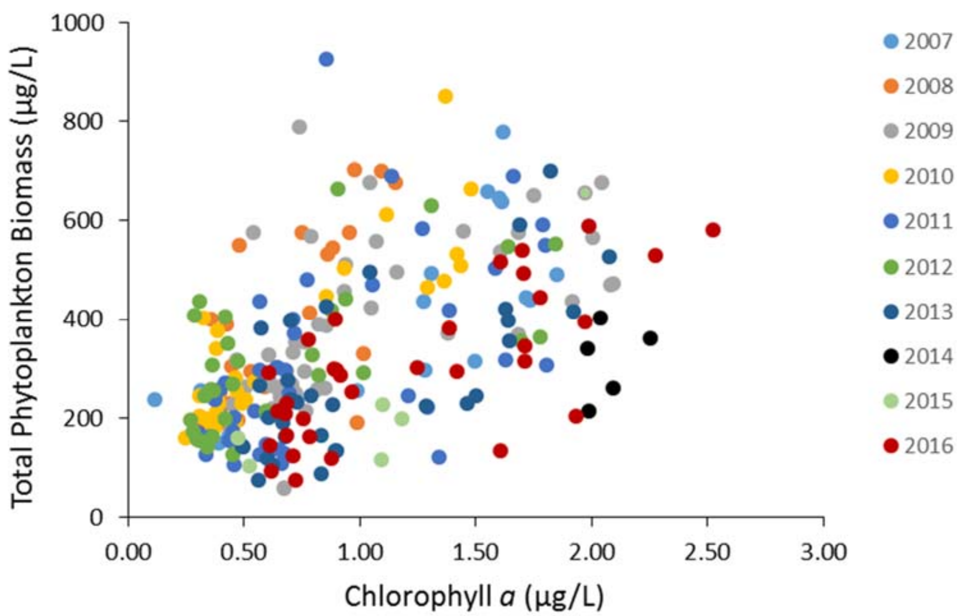
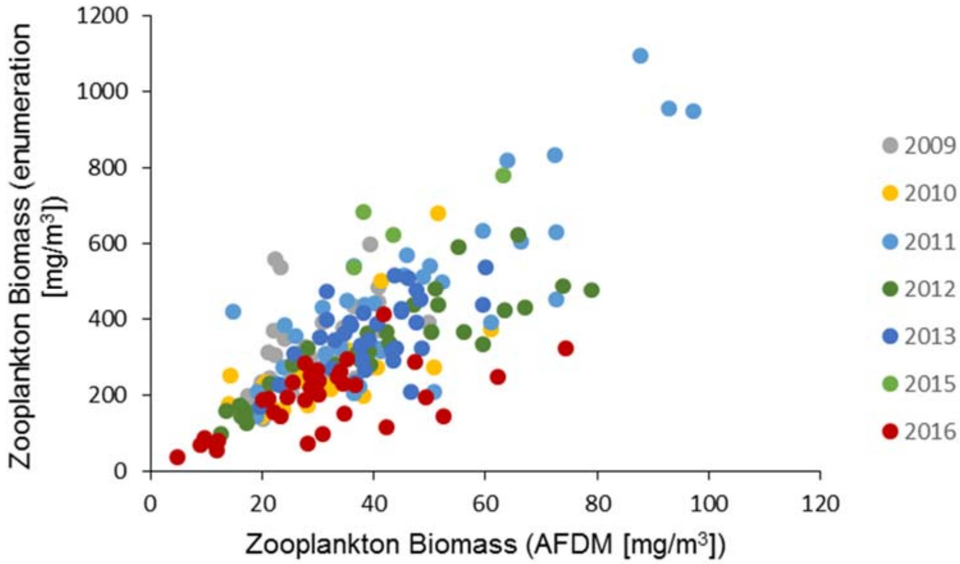


Figure 5-36 Relationship Between Zooplankton Biomass measured as AFDM and measured as Enumeration, 2009 to 2016



Note: AFDM = ash-free dry mass.

5.3.5.2 Relationship between TDS and Nutrients

Concentrations of TDS are elevated in Mine effluent and have shown a clear relationship with distance from the diffuser (Section 7); therefore, TDS can be used as an effluent tracer (i.e., an indicator of the concentration of effluent in lake water). Total phosphorus and nitrogen (Figure 5-37; Table 5-21) were plotted against TDS and correlations per year were evaluated to explore the relationship between the effluent tracer and nutrient concentrations.

Generally, a moderate to strong relationship (*r* >0.5) was observed between TN and TDS in most years (Figures 5-37; Table 5-21). In 2007, TP was strongly correlated to TDS *(r* = 0.78) but, from 2008 to 2016, TP had a poor relationship with TDS (*r* ranging from 0.12 to 0.57). Moderate to strong relationships between TN and TDS were also observed between 2007 and 2012 (*r* ranging from 0.64 to 0.93), in 2013 and 2016 the TN to TDS relationship was poor (*r* = 0.40 and *r* = 0.41, respectively).

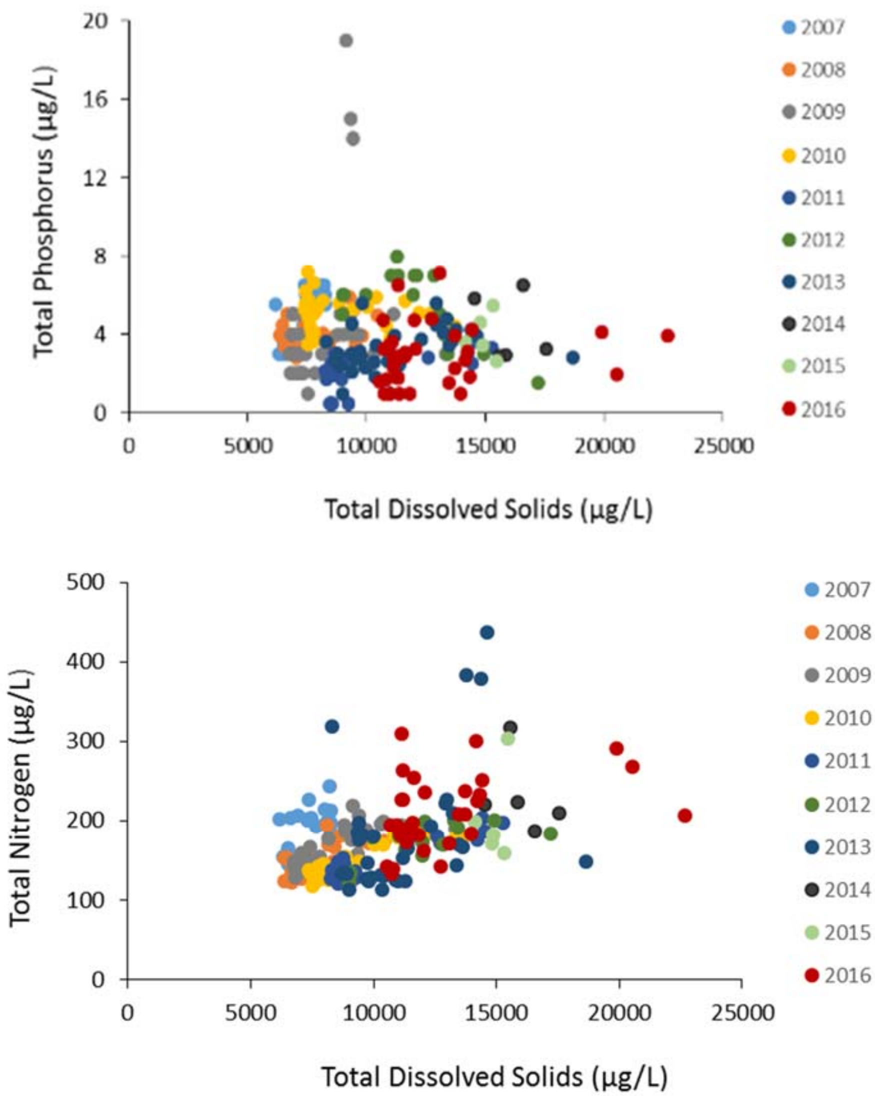
The weaker correlations with TP are as expected, since available P is expected to be utilized at a faster rate than N in a P-limited lake; therefore, a trend related to effluent concentration based on a conservative variable (TDS) may not be apparent in the concentrations of P variables. The significant correlations between TDS and TN suggest that a relatively small proportion of the N is being utilized by primary producers. In addition, the strong correlations between TDS and TN suggest that biological effects related to TN or components of TDS (i.e., micronutrients) may not be separated by a correlational approach.

Table 5-21 Relationships among Concentrations of Total Phosphorus and Nitrogen, and Total Dissolved Solids, 2007 to 2016

Year	n	TP vs TDS (*r*)	TN vs TDS (*r*)
2007	20	0.78	0.64
2008	26	0.37	0.79
2009	46	0.28	0.81
2010	27	0.12	0.91
2011	42	0.57	0.93
2012	30	0.65	0.69
2013	33	0.35	0.40
2014	-	-	-
2015	-	-	-
2016	34	0.12	0.41

Notes: *r* = Pearson correlation coefficient; n = sample size; vs = versus; TDS = total dissolved solids; TP = total phosphorus; TN = total nitrogen; - = insufficient data for analysis.

Figure 5-37 Relationships between Concentrations Total Phosphorus and Nitrogen and Total Dissolved Solids, 2007 to 2016



5.3.5.3 Relationship between Nutrients and Chlorophyll a and Concentrations

Phytoplankton biomass and the concentration of chlorophyll *a* (as a surrogate for phytoplankton biomass) are often controlled by the limiting nutrient (Schindler 1977). In Lac de Gras, the limiting nutrient is P, as determined based on nutrient ratios and TSI calculations (Section 5.3.7). If chlorophyll *a* concentration is controlled by the limiting nutrient, a moderate to strong correlation between the two variables would be excepted; however, the relationship between concentrations of chlorophyll *a* and TP was poor (r <0.5) in all years except 2007 (r <0.71) (Figure 5-38; Table 5-22; log-transformation of the data did not improve the relationship). This may be the result of a limited range in TP concentration in Lac de Gras (0.5 to 7.2 µg/L), as suggested by Shortreed and Stockner (1986) for lakes with low TP concentrations.

The relationship between concentrations of chlorophyll *a* and TN was moderate to strong, ranging between r = 0.52 and 0.92 between 2007 and 2013, while in 2016 the relationship was poor (r <0.2) (Figure 5-38; Table 5-22). The relationship between chlorophyll *a* concentrations and the N to P ratio was poor (r <0.5; Table 5-21; Figure 5-39). The linear relationships between nutrients and phytoplankton biomass were evaluated in the Plankton component (Section 7). A poor relationship between phytoplankton biomass and TP and the N to P ratio was observed, while the relationship between phytoplankton biomass and TN was moderate to strong (r = 0.55 to 0.82) between 2007 and 2012, but poor (r <0.5) from 2013 to 2016.

To further explore nutrient relationships, chlorophyll *a* concentrations were plotted against TDS and correlations were run. A moderate to strong relationship between chlorophyll a concentrations and TDS was observed within each year, similar to the relationships with TN (Figure 5-38; Table 5-21). In 2007, chlorophyll *a* had a poor relationship (r = 0.23) with TDS, but strong relationships were found from 2008 to 2016 (r ranging from 0.63 to 0.96). The strong correlation between chlorophyll *a* concentration and TDS suggests a Mine-related eutrophication effect, related to increased phytoplankton biomass responding to an increase in micronutrients associated with TDS, rather than an increase in TN concentrations. The moderate to strong relationships between TN and chlorophyll *a* concentrations, and TN and phytoplankton biomass, may be the result of the strong correlation between TN and TDS (Table 5-21).

Chlorophyll *a* concentrations are influenced by a number of factors, including nutrient concentrations (often dependent on the limiting nutrient and micronutrients), and biological (e.g., herbivory), and physical (e.g., thermal stratification or lake morphometry) interactions (Mazumder 1994). Therefore a strong linear relationship to a single variable may not be evident.

Table 5-22 Relationship between Concentrations of Total Phosphorus and Total Nitrogen and Chlorophyll *a* Concentrations, 2007 to 2016

Year	n	Chl *a* vs TP (*r*)	Chl *a* vs TN (*r*)	Chl a to TDS (r)	Chl *a* vs P:N molar ratio (*r*)
2007	20	0.71	0.70	0.23	0.46
2008	26	0.26	0.89	0.79	0.41
2009	46	0.35	0.75	0.85	0.08
2010	27	0.10	0.92	0.91	0.63
2011	42	0.41	0.87	0.96	0.18
2012	30	0.26	0.81	0.92	0.63
2013	33	0.52	0.52	0.63	0.11
2014	-	-	-	-	-
2015	-	-	-	-	-
2016	34	0.28	0.14	0.68	0.06

Notes: *r* = Pearson correlation co-efficient; n = sample size; vs = versus; Chl *a* = chlorophyll *a*; TP = total phosphorus; TN = total nitrogen; TDS = total dissolved solids; - = insufficient data; Log = Log (x+1) data transformation.

Figure 5-38 Relationships between Concentrations of Chlorophyll *a* and Total Phosphorus, Total Nitrogen and Total Dissolved Solids, 2007 to 2016

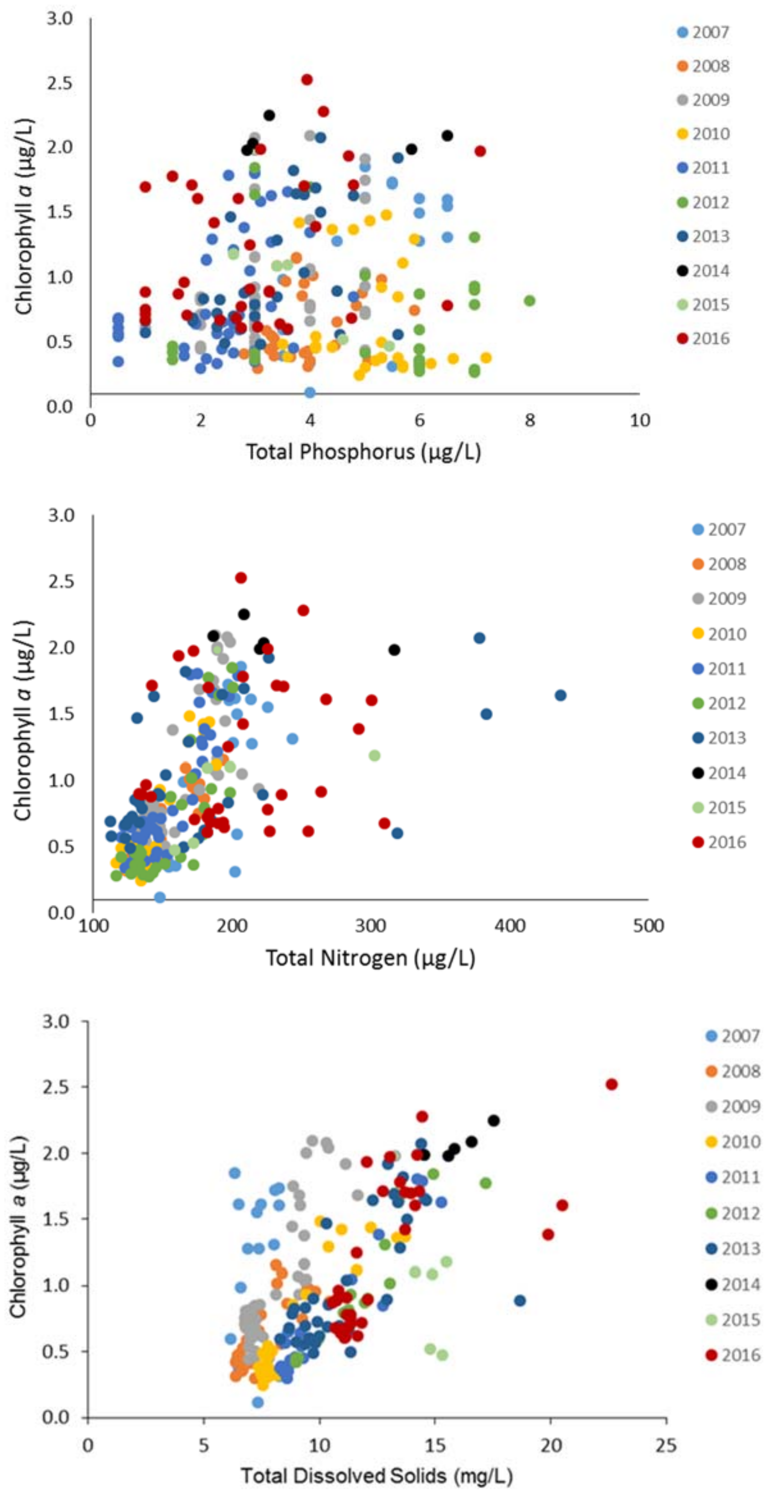
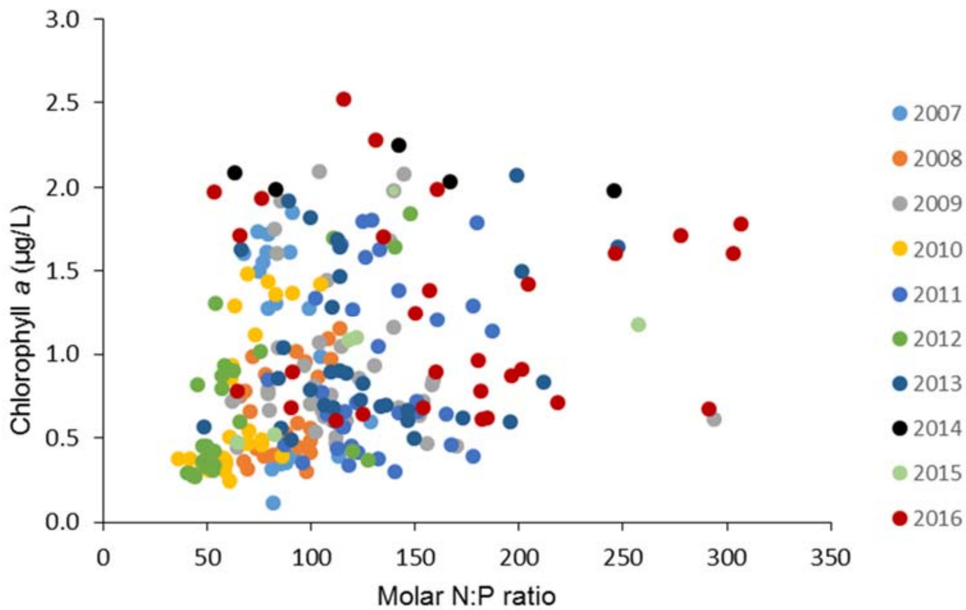


Figure 5-39 Relationship between Concentrations of Chlorophyll *a* and the N:P Molar Ratio, 2007 to 2016



N = nitrogen; P = phosphorus.

5.3.6 Concordance with Plankton Monitoring Results

The plankton community data suggest that plankton communities in Lac de Gras continue to exhibit Mine-related nutrient enrichment effects, rather than toxicity effects. Phytoplankton taxonomic richness increased in the NF between 2002 and 2012, and in 2013 and 2016, richness was at or above the upper limit of the 2007 to 2010 normal range. Zooplankton taxonomic richness has generally remained within the normal range from 2008 to 2016.

The total phytoplankton and zooplankton biomass data have already been incorporated as indicators of eutrophication. Temporal trends in the biomass of the major plankton groups were examined in Section 7. Trends in biomass of the major phytoplankton groups differed among sampling areas and among years in Lac de Gras between 2002 and 2012, with the exception of microflagellate and cyanobacteria biomass, which increased in most areas between 2002 and 2012. Between 2013 and 2015, cyanobacteria and microflagellate biomass decreased in the NF area, but in 2016, microflagellate biomass increased.

Temporal trends in the biomass of the major zooplankton groups differed among sampling areas in Lac de Gras between 2008 and 2016. A decrease over time was observed in calanoid copepod biomass in all areas. Cladoceran biomass exceeded the normal range in the NF area in most years from 2008 to 2013, but decreased from 2014 to 2016 to within the normal range. Cyclopoid copepod and rotifer biomass have generally been within or above the upper limit of the normal range between 2008 and 2016.

The plankton community MDS results indicate that changes over time in plankton community structure have been occurring in the NF area of Lac de Gras. For phytoplankton, changes have occurred in the NF area over time, but the NF and FF areas in 2013 and 2016 were more similar to one another than observed in previous years. Changes over time were also observed in zooplankton community structure, especially in the MF3 area in 2016.

The plankton component results generally support the conclusions of the eutrophication indicators analysis, and are consistent with a Mine-related nutrient enrichment effect and a divergence in community structure in the NF (phytoplankton) and MF (zooplankton) areas in recent years. Although the changes in phytoplankton community structure in the NF area are interpreted as the likely result of nutrient enrichment, other factors (e.g., dike construction) may be responsible for the changes in zooplankton community structure in the MF3 area in 2016.

5.3.7 Limiting Nutrient and Trophic Status

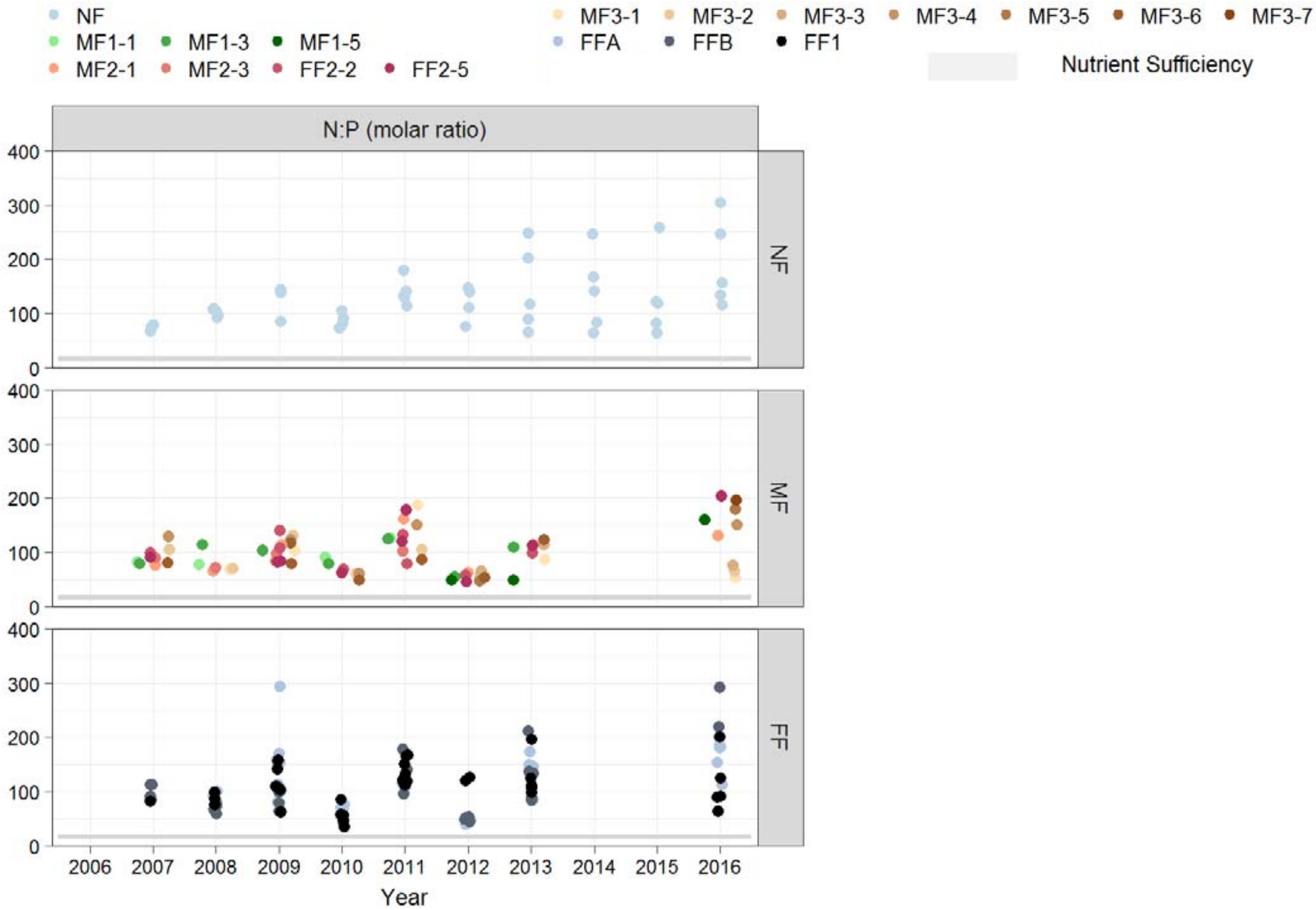
5.3.7.1 Nutrient Ratios

Phosphorus limitation in lakes is traditionally denoted by N to P ratios that exceed 16:1 based on the molar Redfield ratio (Wetzel 2001) or 22:1 based on Hecky et al. (1993). Nutrient sufficiency is achieved within N to P molar ratio boundaries of 13 to 22; N to P ratios less than 13 indicate N-limitation (Hecky et al. 1993).

Baseline N to P ratios in Lac de Gras were >35 (DDMI 1998b); this is consistent with the general N to P range for Canadian Shield lakes reported in Rühland et al. (2003), Pienitz et al. (1997), and Hecky et al. (1993). Phosphorus-limited lakes have limited available P concentrations to sustain primary productivity (Wetzel 2001), irrespective of the amount of available nitrogen that could be used by phytoplankton. Work by Schindler (1974, 2012) and Schindler et al. (2008, 2016) and others examining trophic conditions in lakes has illustrated the significance of phosphorus as the principal limiting growth factor compared to nitrogen (and carbon), especially in oligotrophic lakes such as Lac de Gras.

Nutrient ratios in Lac de Gras indicate severe P-limitation in NF area from 2007 to 2016 (N to P ratios range from 63 to 303) (Figure 5-40). Phosphorus-limitation at stations in the MF areas varied from near P-sufficiency (e.g., N to P ratio of 45 at FF2-5 in 2012) to severe P-limitation (e.g., N to P ratio of 204 at FF2-5 in 2016). In the FF areas, the N to P ratio also ranged from near P-sufficiency to severe P-limitation, in the FF1 area the N to P ratios ranged from 36 to 201 between 2007 and 2016 and in the FFA and FFB areas the N to P ratios ranged from 40 to 294. Overall, P-limitation appears to have increased in all areas of Lac de Gras between 2007 and 2016 (Figure 5-40).

Figure 5-40 Nitrogen-Phosphorus Molar Ratio in Lac de Gras, 2006 to 2016



NF = near-field; MF = mid-field; FF = far-field; N = nitrogen; P = phosphorus.

Note: Grey shading indicating nutrient sufficiency shows the range in N to P ratio where neither nutrient is limiting.

5.3.7.2 Trophic Status Classification

Trophic status was evaluated by examining the concentrations of TP, chlorophyll *a,* and water transparency (i.e., Secchi depth). The raw data ranges for each year in the NF and FF areas and corresponding TSI values are presented in Table 5-23. Based on these values, and the classification systems of Vollenweider (1968), CCME (2004), and Carlson (1977), as summarized in (DDEC 2016), Lac de Gras is classified as oligotrophic in all areas, including the NF area. This trophic status classification is consistent with observations made in Sections 5.3.1, 5.3.3, 5.3.4 and 5.3.5 for Lac de Gras, which confirm that it is a nutrient-poor, unproductive lake.

A multivariate comparison of the Carlson and Simpson (1996) trophic state index was also completed, where TSI (chlorophyll *a*) to TSI (TP) and TSI (chlorophyll *a*) to TSI (Secchi) were plotted on a single graph by year (Figure 5-41) and by area (Figure 5-42). On these plots, the majority of points fell above the zero line on the y-axis, suggesting P-limitation. Along the x-axis, points spread from -14 to 14, indicating that both small and large particles are present in Lac de Gras. Generally, large particles predominated in the NF, MF2-FF2, and MF1 areas, while smaller particles predominated (i.e., dissolved colour and clay particles) in the MF3 and FF areas (Figure 5-42). Turbidity in Lac de Gras in most areas and during most years was related to non-algal turbidity.

The points on the right side of the x-axis in Figures 5-41 and 5-42 indicate situations where transparency is greater than expected by considering chlorophyll *a* alone (i.e., points in the NF and MF2-FF2 areas). This can often occur if large phytoplankton are present, such as some forms of cyanobacteria, and transparency is affected by these larger taxa (Carlson and Simpson 1996). It may also occur if zooplankton grazing removes the smaller particles from the water column and leaves only large phytoplankton forms. The points on the left side of the x-axis are related to situations where transparency is dominated by non-algal factors such as colour or turbidity, or where very small particles predominate (i.e., MF3 stations and FF areas).

Points that fall below zero on the y-axis in Figures 5-41 and 5-42 are associated with situations where chlorophyll *a* is under-predicted by TP (i.e., situations where phosphorus may not be the limiting nutrient for chlorophyll *a*). There are a few instances of this occurring in 2008, 2009, 2010 and 2012, generally in the FF areas. Carlson (1992) reported that this zero line is related to N to P ratios greater than 33, but a better interpretation of these values falling below the line is that there is a greater probability of something other than phosphorus limiting algal growth, potentially micronutrients associated with TDS. The majority of the points are lying on the diagonal to the left of the origin, indicating that phosphorus and transparency are correlated, but chlorophyll *a* is not. The points on or near this line would be found in turbid situations, where P is bound to clay particles and, therefore, turbidity and P are related but chlorophyll *a* is not.

Table 5-23 Concentrations of Total Phosphorus and Chlorophyll *a*, Secchi Depth, and Trophic State Index in Near-field and Far-Field areas in Lac de Gras, 2007 to 2016

Area	Year	Mean TP (µg/L)	TSI (TP)	Chlorophyll *a* (µg/L)	TSI (Chl)	Secchi Depth (m)	TSI (Secchi)
Oligotrophic Lakes[(a)]		0 to 12	0 to 40	0 to 2.6	0 to 40	4 to >8	0 to 40
Near-field Mean	2007	5.9	30	1.6	35	7.3	31
	2008	3.8	23	1.0	30	6.6	33
	2009	3.5	22	1.9	37	7.2	32
	2010	4.7	27	1.3	33	7.5	31
	2011	3.0	20	1.7	36	9.6	28
	2012	3.3	20	1.6	35	6.6	33
	2013	4.3	25	1.5	34	8.3	30
	2014	4.3	24	2.1	38	6.9	32
	2015	3.9	23	0.9	28	6.0	34
	2016	3.3	21	1.8	36	6.8	33

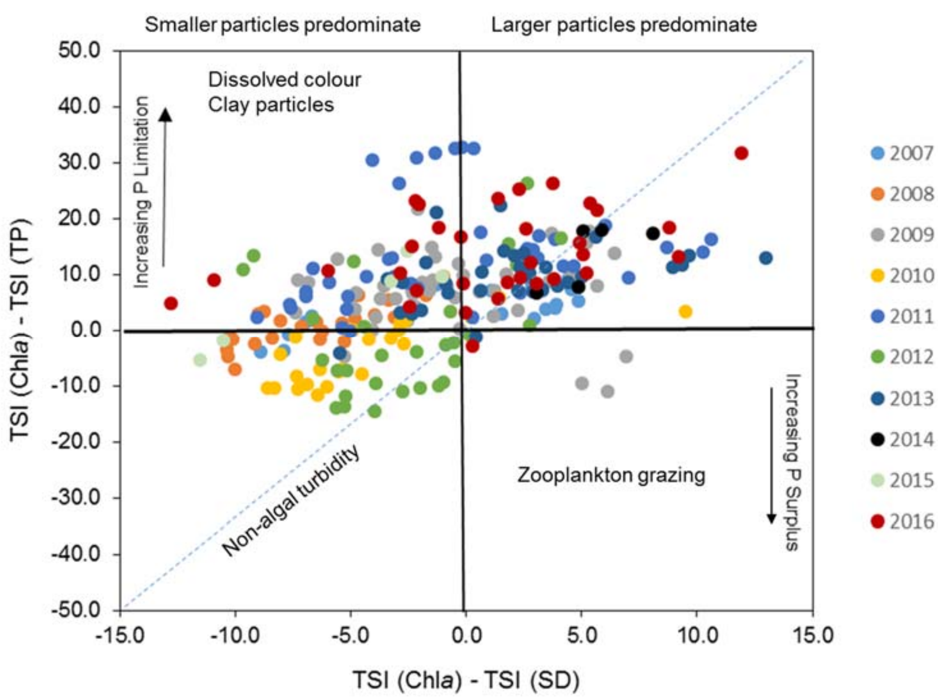
Table 5-23 Concentrations of Total Phosphorus and Chlorophyll *a*, Secchi Depth, and Trophic State Index in Near-field and Far-Field areas in Lac de Gras, 2007 to 2016

Area	Year	Mean TP (µg/L)	TSI (TP)	Chlorophyll *a* (µg/L)	TSI (Chl)	Secchi Depth (m)	TSI (Secchi)
Far-field area Mean	2007	3.5	22	0.4	21	8.7	29
	2008	3.6	23	0.4	22	7.9	30
	2009	3.0	19	0.7	26	8.3	30
	2010	5.1	27	0.4	21	10.0	27
	2011	1.8	10	0.5	24	9.3	28
	2012	4.8	25	0.3	20	11.8	25
	2013	2.8	18	0.7	27	10.4	27
	2014	-	-	-	-	-	-
	2015	-	-	-	-	-	-
	2016	2.9	17	0.7	27	9.8	27

Notes: the far-field area mean is equal to the mean of the three far-field areas (i.e., FF1, FFB, FFA); TP = total phosphorus; TSI = trophic state index; Chl = chlorophyll *a*.

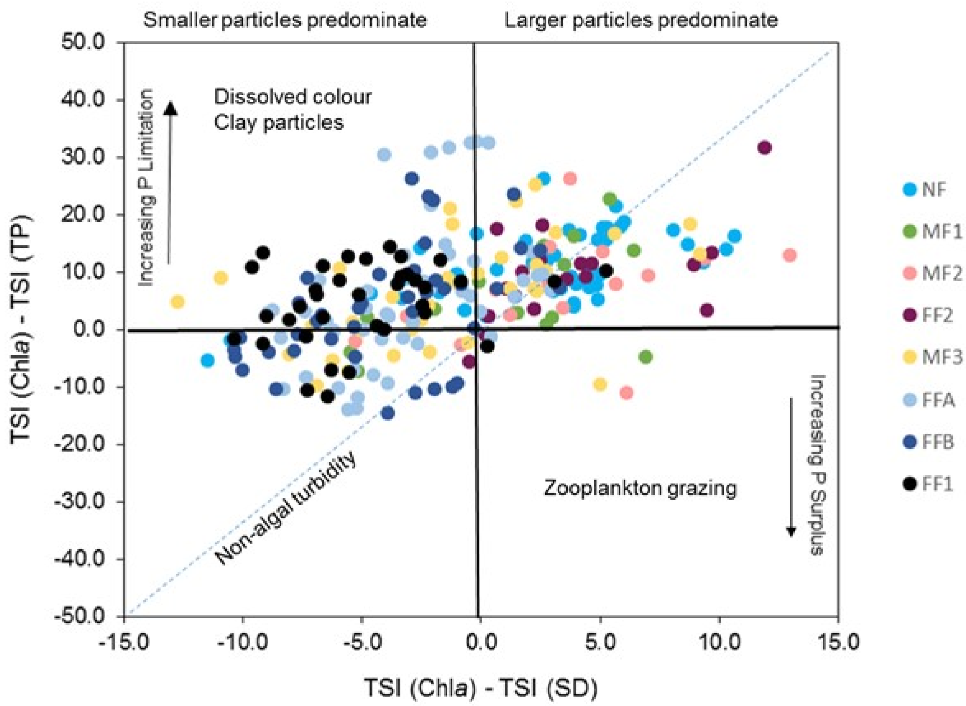
a) Based on Carlson (1977(and Carlson and Simpson (1996).

Figure 5-41 A Multivariate Comparison of Total Phosphorus, Secchi Depth, and Chlorophyll *a* using Carlson's Trophic State Index Equations (from Carlson and Simpson 1996) in Lac de Gras, 2007 to 2016



Note: TSI = trophic state index; Chl = chlorophyll *a*; TP = total phosphorus; SD = Secchi depth; P = phosphorus

Figure 5-42 A Multivariate Comparison of Total Phosphorus, Secchi Depth, and Chlorophyll *a* using Carlson's Trophic State Index Equations (from Carlson and Simpson 1996) based on Area in Lac de Gras, 2007 to 2016



Note: TSI = trophic state index; Chl = chlorophyll *a*; TP = total phosphorus; SD = Secchi depth; P = phosphorus; NF = near-field; MF = mid-field; FF = far-field.

5.3.7.3 Summary and Overall interpretation

An evaluation of the relationships between nutrients, including TDS, and biological variables, and among nutrient variables and biological variables in Lac de Gras showed the following:

- The relationship between TN and TDS concentrations was moderate to strong between 2007 and 2012, but poor in 2013 and 2016.
- The relationship between TP and TDS concentrations was generally poor.
- The relationship between concentrations of chlorophyll *a* and TP was poor in all years except 2007, likely as a result of a limited range in TP concentration in Lac de Gras (0.5 to 7.2 µg/L).
- The relationship between concentrations of chlorophyll *a* and TN was moderate to strong between 2007 and 2013; however, in 2016, the relationship was poor.
- The overall relationship between chlorophyll *a* concentration and phytoplankton biomass was poor; however, a within-year evaluation indicated moderate to strong relationships between chlorophyll *a* concentrations and phytoplankton biomass between 2007 and 2016.
- A moderate to strong relationship was observed between chlorophyll *a* and TDS concentrations within each year between 2008 and 2016.

- The relationship between zooplankton biomass measured as AFDM and based on enumeration indicated a moderately strong overall relationship.

In Lac de Gras, the limiting nutrient is P, as determined based on nutrient ratios and TSI calculations. Studies have shown that Arctic lakes may be N-limited (Levine and Whalen 2001; Keatley et al. 2007; Symons et al. 2012); however, the lakes in these studies were often small and shallow (<2 km^2 and ~2 m, respectively), unlike Lac de Gras.

The TSI classifies Lac de Gras as oligotrophic in all areas, including the NF area. A poor relationship between chlorophyll *a* concentrations and TP, but a strong relationship to TDS, suggests phytoplankton may be responding to a Mine-related eutrophication effect related to an increase in micronutrients associated with TDS. The moderate to strong relationship between TN and chlorophyll *a* concentrations may be the result of the strong correlation between TN and TDS.

5.3.8 Eutrophication Indicators Response Framework Evaluation

5.3.8.1 Relevant Directives

Directives relevant to the Eutrophication Indicators component are listed below and each directive is addressed separately in the subsections that follow.

Verification of Assumption Related to Action Level Testing for Eutrophication Indicators

In the Reasons for Decision associated with the WLWB's letter approving the *2014 AEMP Annual Report*, the WLWB directed DDMI to evaluate the assumptions of the Action Level testing for Eutrophication Indicators (Table 1-1). The WLWB noted that the sequence of Action Levels appears to assume that the magnitude of the effect will increase and trigger Action Levels at NF/MF stations prior to reaching FF stations and triggering Action Levels related to effects at those FF stations, and asked DDMI to evaluate this assumption during the *2014 to 2016 Aquatic Effects Re-evaluation Report*, particularly if effects continue to extend beyond the MF areas.

Phosphorus Management Framework and Inclusion of Phosphorus in the Response Framework

In response to concerns associated with the Eutrophication Indicators Action Levels being based exclusively on chlorophyll *a* concentrations, the WLWB has directed DDMI to evaluate the Response Framework for the Eutrophication Indicators component (Table 1-1).

DDMI has also been directed to consider the inclusion of phosphorus concentrations in the Response Framework, including a discussion of observed phosphorus concentrations and how they relate to the Canadian Council of Ministers of the Environment (CCME) *Phosphorus Management Framework* (CCME 2004; *2015 AEMP Annual Report* WLWB Directive). Since DDMI did not respond directly to this part of the information presented by the GNWT-ENR, the WLWB requires DDMI to show that the existing Response Framework based on chlorophyll *a* concentrations is adequately capturing the intent of the *Phosphorus Management Framework*, in light of observed phosphorus concentrations.

Effects Threshold for Eutrophication Indicators

In the Reasons for Decision associated with the WLWB's letter approving the *2014 AEMP Annual Report*, the WLWB noted that the GNWT has identified concern regarding the use of the Effects Threshold and Significance Threshold for Indicators of Eutrophication (GNWT-ENR Comment #11 [Table 1-1]). The WLWB summarized the concern as follows:

> "In essence, the concern appears to lie on the argument that the Effects Threshold has not been formally defined (i.e., been assigned a numerical value), and that the available definition appears to link the Effects Threshold to an unacceptable level of change, which parallels the definition of the Significance Threshold. The Significance Threshold is defined as 'a level of change that, if exceeded, would result in a significant adverse effect' and is being applied here as a concentration of chlorophyll *a* that exceeds the Effects Threshold by more than 20% in FFA. As part of the CSR, the Significance Threshold was defined as a concentration of TP that exceeds the EA Benchmark (i.e., 0.005 mg/L) by more than 20%. In keeping with the intent of the definition from the EA, DDMI uses a Significance Threshold for indicators of eutrophication as a concentration of chlorophyll *a* that exceeds the Effects Threshold by more than 20% in FFA of Lac de Gras. The outstanding question appears to be: How do you ensure that the Effects Threshold is defined so that the Significance Threshold is not reached? By the time effects are being assessed at FFA, where the Significance Threshold is evaluated through the Response Framework, the effect will have already spread to more than 20% of the lake area; thus, setting the Effects Threshold will likely be of great importance."

DDMI was directed to address this concern following the *2014 to 2016 Aquatic Effects Re-evaluation Report*. However, a discussion is included herein as it is relevant to other directives related to the Significance Threshold.

Evaluation of Suggested Significance Threshold

In addition, DDMI has committed to re-evaluate the Significance Threshold for eutrophication indicators (commitment #11 from the *2014 AEMP Annual Report* [Table 1-1]). DDMI has suggested that the Significance Threshold for this component should be based on biological response variables, rather than exposure variables; however, the WLWB has directed an evaluation of the use of a combination of three eutrophication-related measures as the basis of an updated Significance Threshold (i.e., chlorophyll *a* , TP and TN concentrations):

a) The mean of the five FFA depth integrated chlorophyll *a* concentrations does not exceed 4.5 µg/L; **or**
b) The mean of the five FFA TP concentrations does not exceed 10 µg/L; **or**
c) The mean of the five FFA TN concentrations does not exceed 700 µg/L.

5.3.8.2 Verification of Assumption Related to Action Level Testing for Eutrophication Indicators

The Mine effluent discharge is a point source input to Lac de Gras. The assumption of gradual expansion of effects on water quality and primary productivity over time, therefore, reflects the expected propagation of effects in a large lake. During the last comprehensive monitoring year (i.e., 2016), gradient responses were observed in TP (ice-cover season), TN, chlorophyll *a*, phytoplankton biomass, and to some extent, zooplankton biomass (Golder 2017c). The strongest gradient responses were observed in chlorophyll *a* and phytoplankton biomass, representing the biological variables that are expected to directly respond to nutrient addition.

The currently approved Action Level system for chlorophyll *a* is designed to track effects and trigger Action Levels based on spatial expansion of the effect on this indicator of phytoplankton biomass. Action Level 1 is triggered when the estimated maximum chlorophyll *a* concentration in the MF area exceeds the normal range (i.e., when the effect has spread beyond the NF area). From this point onwards, Action Levels 1 to 7 track increasing concentrations and spatial extent of effects through the MF areas, and Action Level 8 triggers when the effect reaches the FFB area, before triggering the Significance Threshold.

Based on monitoring data collected to date under the AEMP, the observed gradient response of biological indicators of eutrophication is well matched to the Response Framework, which is designed to track the spatial expansion of effects of increasing magnitude through Lac de Gras.

5.3.8.3 Phosphorus Management Framework

The *Phosphorus Management Framework* (the Framework) employs a tiered approach, with predefined trigger ranges that are based on the trophic status of the lake being assessed. Trophic status is based on the baseline or reference TP concentrations. The first step in the Framework is to set ecosystem goals and objectives. For Lac de Gras, the objective is to maintain a healthy, diverse and functioning aquatic ecosystem, remaining within the oligotrophic range. Under the Framework, further assessment of environmental factors and potential management options is required if TP concentrations increase above the upper value of the applicable trigger range (i.e., 10 µg/L for oligotrophic waters), or increase by “more than 50% over baseline (reference) levels”. More specifically, under the Framework, if the appropriate trigger range upper limit (10 µg/L, in this case) is not exceeded, “the risk of impact is regarded as low”. Exceeding the baseline plus 50% value indicates that “the risk of observable effects is considered to be high, and further assessment is recommended”. The Framework also permits site-specific management of phosphorus.

The comment from the GNWT-ENR explains that the chlorophyll *a* benchmark used by DDMI is based on the goal of preserving trophic status, which is analogous to the goal of the *Phosphorus Management Framework*. Specifically, the GNWT-ENR stated that: “Golder (2014b) presents a well-reasoned and conceptually identical (preservation of trophic status) argument for defining a chlorophyll *a* benchmark.” The concerns raised by the GNWT-ENR appear to be that the chlorophyll *a* benchmark is not a sensitive enough trigger, and that TP concentrations have exceeded the criterion for further assessment under the Framework.

With regards to phosphorus concentrations and the potential exceedance of the criterion for further assessment under the Framework, the Government of the Northwest Territories, Environment and Natural Resources (GNWT-ENR) presented an updated argument with evidence that has not been addressed to date by DDMI. The GNWT-ENR stated that the “…key principles presented in Environment Canada (2004) are that trophic status should be maintained and, if the increase from the baseline is greater than 50%, regardless of whether it lies within the trigger range (trigger ranges are defined as boundaries between trophic classifications), further assessment is recommended.”

The GNWT-ENR explained that because DDMI compares eutrophication indicator endpoints other than TP (i.e., chlorophyll *a*), it does not follow the principle of assessing trophic status as done under the Framework, and that an increase in 50% of the baseline P concentration does not lead to further assessment, as it should under the Framework. The GNWT-ENR states that “using baseline data the median total P concentrations are 3 and 4 µg/L respectively; for the open water and under ice seasons." If the criterion of an increase of 50% over baseline is followed, the GNWT-ENR states that "observations greater than 4.5 and 6 µg/L for the open water and under ice seasons, respectively require further assessment."

To address these comments, the Framework was applied to TP concentrations in Lac de Gras. The appropriate spatial scale for the application of the Framework is either whole-lake (for small lakes), or in large lakes "the most sensitive areas (e.g., river mouth, point sources, or the littoral zone)" (CCME 2004). In Lac de Gras, there is no reason to assume that a river mouth (e.g., Coppermine River) or the littoral zone would be more ecologically sensitive than other parts of the lake, so the NF area (affected by a point source) was used for evaluating the Framework, as representing the likely worst-case effects from increased TP concentrations. The Framework does not specify the statistic (i.e., mean, median, or maximum) upon which to base the 50% increase relative to baseline. It was conservatively assumed that the appropriate TP concentration for estimating the 50% increase is the seasonal reference median concentration, as defined in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b). The NF area median TP concentration was selected for comparison to thresholds under the Framework, as it accounts for a potentially non-normal distribution of concentrations in the NF area. Concentrations measured at individual stations may not be representative of a larger NF area sampled at multiple locations, especially given analytical uncertainty at the low TP concentrations typical of Lac de Gras.

The *AEMP Reference Conditions Report Version 1.2* indicates that the median TP concentrations during ice-cover and open-water seasons are 3.6 and 3.3 µg/L, respectively (Golder 2017b). An increase of 50% would be represented by 5.0 and 5.4 µg/L for these seasons, respectively. Following the GNWT-ENR's argument, TP concentrations greater than 5.4 and 5.0 µg/L during the ice-cover and open-water seasons, respectively, would require further assessment under the Framework.

The reference condition for TP is estimated as the normal range based on the 2007 to 2010 FF area concentrations, as defined in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b). The values representing 50% increases in TP concentration relative to the reference median concentrations are close to the upper limits of the seasonal normal ranges: 5.0 and 5.3 µg/L, for the ice-cover and open-water seasons, respectively. The suggestion by GNWT-ENR equates to assessing P effects further if the NF area median remains within the normal range for the open-water season (5.0 µg/L, compared to normal range of 2.0 to 5.3 µg/L), or slightly exceeds the normal range for the ice-cover season (5.4 µg/L, compared to normal range of 2.0 to 5.0 µg/L).

From 2007 to 2016, representing the period with data from multiple stations and sampling depths from the NF area, the NF median concentrations of TP were as follows:

- open-water: 3.0 to 4.8 µg/L, with the exception of 6.0 µg/L in 2007, resulting in one exceedance of the baseline plus 50% value of 5.0 µg/L in 2007
- ice-cover: 2007: 5.5 µg/L, 2008: 6.0 µg/L, 2009: 5.3 µg/L and 2010: 5.2 µg/L, 2011: 4.6 µg/L, 2012: 5.0 µg/L, 2013: 8.0 µg/L, 2014: 4.7 µg/L, 2015: 3.2 µg/L, 2016: 4.3 µg/L, resulting in three exceedances of the baseline plus 50% value of 5.4 µg/L in 2007, 2008 and 2013

During this same period, Action Level 1 or 2 triggers were observed for chlorophyll *a* during each year, which resulted in establishing the Effects Benchmark of 4.5 µg/L for chlorophyll *a*, representing the upper limit of chlorophyll *a* concentrations under oligotrophic conditions. The ice-cover and open-water season TP concentrations of 5.5 and 6.0 µg/L in the NF area in 2007 did not result in an effect on chlorophyll *a* that is larger than observed in other years; only Action Level 1 was triggered in 2007, whereas Action Level 2 was triggered in 2009, 2013, 2014 and 2016. The Action Level 2 trigger for chlorophyll *a* (assessed during the open-water season) for only one of these years (i.e., 2013) was preceded by an ice-cover season exceedance of the baseline plus 50% value. These results indicate poor correspondence between the conservatively applied baseline plus 50% value comparison for TP and observed effects on chlorophyll *a* in Lac de Gras, as currently assessed under the AEMP Response Framework.

To further evaluate the suitability of the baseline plus 50% increase comparison, it was applied to the FF1, FFB and FFA area TP results from 2007 to 2016. During this period, chlorophyll *a* concentrations, phytoplankton biomass and zooplankton biomass remained unaffected in these areas (Figures 5-28 to 5-33), with the possible exception of recent slight increases in chlorophyll *a* concentration. Results are as follows:

- FF1 open-water: 2007: 4.0 µg/L, 2008: 3.8 µg/L, 2009: 3.0 µg/L, 2010: 5.3 µg/L, 2011: 2.5 µg/L, 2012: 1.5 µg/L, 2013: 2.7 µg/L, 2016: 3.4 µg/L, resulting in one exceedance of the baseline plus 50% value of 5.0 µg/L in 2010
- FFB open-water: 2007: 3.5 µg/L, 2008: 4.0 µg/L, 2009: 3.0 µg/L, 2010: 6.6 µg/L, 2011: 2.0 µg/L, 2012: 6.0 µg/L, 2013: 2.7 µg/L, 2016: 1.0 µg/L, resulting in two exceedances of the baseline plus 50% value of 5.4 µg/L in 2010 and 2012
- FFA open-water: 2007: 5.0 µg/L, 2008: 3.2 µg/L, 2009: 2.5 µg/L, 2010: 4.7 µg/L, 2011: 0.5 µg/L, 2012: 6.5 µg/L, 2013: 2.5 µg/L, 2016: 2.8 µg/L, resulting in one exceedance of the baseline plus 50% value of 5.0 µg/L in 2012, and a near-exceedance in 2007
- No exceedances in TP during the ice-cover season in the FF1, FFA and FFB areas from 2007 to 2016. In the FF1 area, TP ranged from 2.1 to 4.5 µg/L; in the FFB area, TP ranged from 1.5 to 4.0 µg/L; and in the FFA area, TP ranged from 1.5 to 4.4 µg/L

This additional comparison illustrates the extreme sensitivity of the baseline plus 50% increase comparison for Lac de Gras, which suggest actions required to manage Mine-related P inputs in the absence of effects in the FF areas. It is notable that the exceedances of the baseline plus 50% values in the FF areas do not match those in the NF area, with the only exception being the FFA area during the open-water season in 2007, and there are no exceedances for 2007 in areas between the NF and FFA areas. This sensitivity arises from the small change represented by the 50% increase relative to baseline (open-water: 1.7 µg/L, ice-cover: 1.8 µg/L), which in one season remains within the normal range for TP concentration. At the level of TP concentration measured in Lac de Gras (mostly within five times the analytical DL of 1 µg/L), analytical uncertainty is greater and contributes to the variation observed among stations.

With the availability of extensive lake-wide monitoring data beginning in 2007, it is also possible to evaluate whether the previous exceedances of the baseline plus 50% value in the NF area would have resulted in actions that were warranted to manage mine-related P inputs to Lac de Gras. If the Framework was applied as attempted above, the exceedances of the baseline plus 50% values observed in 2007, 2008 and 2013 would have represented Action Level exceedances requiring further assessment, in addition to continued monitoring. However, historical results indicate that continued annual monitoring and development of the chlorophyll *a* Effects Benchmark, as dictated by the currently approved Response Framework, were sufficient to anticipate and manage effects on biological indicators of eutrophication. Biological effects in the NF area have not spiked in years with exceedances of baseline plus 50% values for TP, and increasing trends suggesting chlorophyll *a* concentrations reaching the Effects Benchmark are not apparent in any of the lake sampling areas.

Overall, this evaluation of the application of the *Phosphorus Management Framework* elements suggested by GNWT-ENR has shown that adopting the baseline plus 50% TP value in the AEMP Response Framework would not result in a useful addition to the management of P inputs to Lac de Gras from the Mine. The current approach to managing P inputs, which consists of (1) the EQC under the Water Licence, (2) monitoring and assessment of nutrient concentrations and biological response variables under the AEMP, and (3) focusing on biological effects to trigger response actions, is adequate and has functioned well to date. The Framework allows for site specific management of P, which is the preferred approach for Lac de Gras.

Inclusion of Phosphorus Concentration in the Response Framework

DDMI has been directed to consider the inclusion of P concentration in the AEMP Response Framework for eutrophication indicators. Based on results of the AEMP re-evaluation and additional analyses presented in preceding sections, inclusion of TP in the Response Framework is not recommended, for the following reasons:

- Chlorophyll *a* concentration has proven to be a simple and robust indicator of the biological response to nutrient additions to Lac de Gras. It is a more reliable indicator of trophic status than TP, which is an exposure variable rather than a response variable.
- Increases in TP in Lac de Gras have been small and sporadic, without strong spatial trends, while clear increases in chlorophyll *a* concentrations have been observed in the NF and MF areas, indicating a gradient-type response to nutrient addition.
- The Significance Threshold is recommended to continue to be based on biological response (chlorophyll *a*), rather than nutrient concentrations. The Response Framework “builds up” to the Significance Threshold and, therefore, must be based on a matching variable, which is chlorophyll *a*.

5.3.8.4 Effects Threshold for Eutrophication Indicators

Definitions of key terms used in this section are provided below, based on descriptions in the *AEMP Study Design Version 3.5* (Golder 2014a):

- The Effects Benchmark represents a value that is protective of aquatic life and is intended to be conservative. It represents a level which, if exceeded, could cause adverse effects; not a level which, if exceeded, would cause adverse effects. For chlorophyll *a*, the Effects Benchmark was set at the concentration representing the upper limit characteristic of oligotrophic waters, based on a review of the relevant scientific literature (Section 5.4.2 in Golder 2014a).
- The Effects Threshold is based on the specific conditions of Lac de Gras. An Effects Threshold for biological endpoints is described by Golder (2014a) as “the level representing an unacceptable biological effect”. In practical terms, the Effects Thresholds represents a site-specific refinement of the Effects Benchmark, and may be the same value.

- The Significance Threshold is defined as “a level of change that, if exceeded, would result in a significant adverse effect” and is applied for eutrophication indicators as the chlorophyll *a* concentration that exceeds the Effects Threshold by more than 20% in the FFA area of Lac de Gras.

For chlorophyll *a*, the Effects Benchmark currently in effect is 4.5 µg/L, as developed by Golder (2014a). If, upon triggering Action Level 3, it is found that the Effects Benchmark is a reasonable estimate of the upper limit of chlorophyll *a* concentration under oligotrophic conditions in Lac de Gras, the Effects Threshold would be set at a value equal to the Effects Benchmark.

The Effects Threshold has no spatial component; it is a chlorophyll *a* concentration to be applied according to the Action Level system, which uses this concentration to track the magnitude and spatial expansion of the biological effects of nutrient inputs from the Mine. Therefore, the Effects Threshold cannot function as a substitute for the Significance Threshold, which has a defined spatial component representing the upper end of the Action Level system for chlorophyll *a*. Given this context, there is no potential to define the Effects Threshold in a manner that would interfere with the application of the already defined Significance Threshold.

The question related to 20% of lake area being affected is not relevant to defining the Effects Threshold, which is a concentration value corresponding to a certain level of effect on chlorophyll *a*, without spatial context. The EA prediction of effects related to nutrient enrichment was a TP concentration greater than 5 µg/L in 20% of the lake area. Because the concentration of 5 µg/L is now known to be at the upper limit or within the normal range (depending on season), the effect prediction is interpreted as a change in trophic status in 20% of the lake area from oligotrophic to mesotrophic. Therefore, an increase in chlorophyll *a* concentration is expected in greater than 20% of the lake area, because the enrichment effect will develop as a gradient across the lake. However, this is a much smaller effect than that represented by the Significance Threshold, which corresponds to a change in trophic status in the entire lake, as measured by chlorophyll *a* concentration in the sampling area located farthest from the Mine effluent diffuser.

5.3.8.5 Evaluation of Suggested Significance Threshold for Eutrophication Indicators

The Significance Threshold for the AEMP was defined based on the *Comprehensive Study Report* (Government of Canada 1999), which defined a significant adverse effect as “an effect that has a high probability of a permanent or long-term effect of high magnitude, within the regional area, that cannot be technically or economically mitigated”. For the purposes of the AEMP, “regional area” is interpreted as the entire Lac de Gras, although it was originally defined in the EA as the drainage basin of the lake. High effect magnitude for water quality was defined as a concentration that exceeds a threshold by more than 20%, and was adopted for chlorophyll *a*. The significance threshold based on chlorophyll *a* concentration was developed and approved for eutrophication indicators, and is stated as: “95th percentile of FFA values greater than the Effects Threshold+20% in the FFA area” (Golder 2014a). Making the assumption that the Effects Benchmark for chlorophyll *a* (4.5 µg/L), based on protecting the oligotrophic status of the lake, is a reasonable estimate of the Effects Threshold (to be developed), the Significance Threshold is interpreted as the estimated maximum chlorophyll *a* concentration in the part of Lac de Gras farthest from the Mine discharge exceeding 5.4 µg/L (4.5 µg/L plus 20%). In simpler terms, the Significance Threshold corresponds to the trophic status of the entire lake becoming mesotrophic.

The following sections evaluate the Significance Threshold suggested by the WLWB, by evaluating each of its three components (a, b, and c) separately.

a) The mean of the five FFA depth integrated chlorophyll *a* concentrations does not exceed 4.5 µg/L

Significance Threshold component a) is based on chlorophyll *a* concentration and, similar to the currently approved Significance Threshold, is intended to protect the oligotrophic status of the lake. It is largely equivalent to the current Significance Threshold. It is based on the mean concentration (a lower value) rather than the estimated maximum concentration used in the current threshold, and is compared to the Effects Benchmark of 4.5 (a lower value) rather than the benchmark+20% used in the current threshold. However, it would be expected to be triggered at about the same level of effect as the current Significance Threshold.

The suggested threshold represents a departure from the definition of significance by the *Comprehensive Study Report* (Government of Canada 1999), which is based on a high effect magnitude as defined in the EA (i.e., benchmark+20%).

b) The mean of the five FFA TP concentrations does not exceed 10 µg/L

The suggested Significance Threshold component b) based on TP is also intended to protect trophic status of the lake and is equivalent to that under option a), with the notable exception that it is based on an exposure variable rather than a biological response variable. This is an important shortcoming, because factors other than TP (e.g., temperature, hydrology, turbidity, micronutrients) may also influence the level of primary productivity in a lake.

As demonstrated by over a decade of monitoring in Lac de Gras using an extensive sampling design, the concentration of TP does not accurately predict the biological response to nutrient enrichment (Section 5.3.5), and TP concentration alone is not sufficient to evaluate changes in lake productivity. In fact, measuring TP can only evaluate the *potential* for an increase in lake productivity. The range in TP concentrations has been low in all areas of Lac de Gras, ranging from 0.5 to 9.0 µg/L between 2003 and 2016, with a few exceptions, and an increasing trend has not been observed in the NF area; however, responses in biological variables have been observed (Section 5.3).

Biological response to nutrient enrichment can be measured directly, as currently done under the AEMP. Monitoring in Lac de Gras has shown that the concentration of chlorophyll *a* (as an indicator of phytoplankton biomass and trophic status) has been a sensitive and robust measure of biological response to nutrient inputs from the Mine (Section 5.3).

Finally, the suggested TP threshold of 10 µg/L is a generic value applicable to freshwater lakes in Canada, and may not be appropriate to evaluate the significance of effects in large subarctic lakes on the Canadian shield.

Therefore, basing the Significance Threshold on TP is not recommended, as a better measure of biological response is available, and is currently in use.

c) The mean of the five FFA TN concentrations does not exceed 700 µg/L

The suggested Significance Threshold component c) based on TN is considered inappropriate based on evidence accumulated through the AEMP and the scientific literature. Lac de Gras is highly unlikely to be N-limited, based on nutrient ratios and spatial trends in responses to nutrient inputs from the Mine (Section 5.3.5).

Despite poor relationships observed with phytoplankton biomass, TP or micronutrients associated with increased TDS (Section 5.3.5) are the most likely key drivers of phytoplankton biomass in Lac de Gras. Boundary concentrations for N are not clearly defined in trophic classification systems, particularly for northern Canadian lakes (i.e., there is a high level of uncertainty around a definitive and appropriate trophic threshold for N). The OECD (1982) found that the mean TN concentration in oligotrophic lakes is 660 µg/L but ranges from 310 to 1,630 µg/L, while Forsberg and Ryding (1980) suggested <400 µg/L for lakes with macrophytes, and Vollenweider (1968) presented a range of 400 to 800 µg/L (based on inorganic and organic forms of nitrogen). In all of these cases though, the *observed* TN concentrations in lakes of different trophic status (as evaluated by other means) were reported, rather than estimating TN concentrations through cause-effect evaluations.

Overall, TN is not suitable in Lac de Gras for evaluating a potential trophic status shift from oligotrophic to mesotrophic.

Conclusion Regarding Significance Threshold

Since chlorophyll *a* has been a reliable indicator of the biological response to nutrient enrichment in Lac de Gras, it is recommended that the Significance Threshold continue to be based on this variable for the Eutrophication Indicators component of the AEMP. The additional Significance Threshold components other than chlorophyll *a* suggested by the WLWB are based on generic guidelines or literature summaries, are not based on biological responses to nutrient input, or are inappropriate based on the limiting nutrient in Lac de Gras.

Continued use of the percentile-based Significance Threshold based on chlorophyll *a* concentration is recommended, because it is a direct biological measure of effects from nutrient enrichment, and remains consistent with the definition of significance in the *Comprehensive Study Report* (Government of Canada 1999).

5.3.8.6 Conclusion Regarding Response Framework Evaluation for Eutrophication Indicators

The overall conclusion from the evaluation of issues related to the Response Framework for eutrophication indicators is that the current framework is functioning as intended and requires no updates, for reasons summarized below:

- The assumption of spatially expanding effects related to nutrient enrichment, and associated sequential Action Level triggers, is reasonable based on AEMP monitoring data collected to date. The observed gradient response of biological indicators of eutrophication is well-matched to the Response Framework, which is designed to track the spatial expansion of effects of increasing magnitude through Lac de Gras.
- Application of the *Phosphorus Management Framework* elements suggested by GNWT-ENR has shown that adopting the baseline plus 50% TP value in the AEMP Response Framework would not result in a useful addition to the management of phosphorus inputs to Lac de Gras from the Mine.
- Inclusion of TP in the Response Framework is not recommended, because chlorophyll *a* concentration has proven to be a reliable indicator of the biological response to nutrient additions, and is a more appropriate and direct indicator of trophic status than TP. Increases in TP in Lac de Gras have been small and sporadic, without strong spatial trends, while clear increases in chlorophyll *a* concentrations have been observed, and indicate a gradient-type response to nutrient addition.
- Clarification regarding the Effect Threshold, predicted effect and Significance Threshold was provided, and indicate no conceptual misalignment in the currently approved Response Framework for eutrophication indicators.
- Adopting the expanded Significance Threshold suggested by GNWT-ENR is not recommended, because the proposed additions are based on generic guidelines or literature summaries, are not based on biological responses to nutrient input, or are inappropriate based on the limiting nutrient in Lac de Gras.

The current approach to managing P inputs to Lac de Gras, which consists of (1) the EQC under the Water Licence, (2) monitoring and assessment of nutrient concentrations and biological response variables under the AEMP, and (3) focusing on biological effects to trigger response actions, is adequate and has functioned well to date.

5.4 Comparison to EA Predictions

Elevated concentrations of nutrients were predicted in Lac de Gras (Government of Canada 1999). Specifically, up to 20% (i.e., 116 km^2) of the surface area of Lac de Gras was expected to exceed the EA threshold for phosphorus (i.e., 5 µg/L) during peak operations during the open-water season, and up to 11% (i.e., 64 km^2) of the lake during the ice-cover season. Outside these areas, TP concentration was predicted to increase relative to baseline in parts of Lac de Gras, but concentrations would remain below the EA threshold. In the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b), the normal range was re-calculated and demonstrated that the EA threshold of 5 µg/L ([DDMI 1998b) was inappropriate given that it was within the normal range for the lake. Therefore, a new threshold based on the upper limit of the normal range (i.e., 5 µg/L for ice-cover and 5.3 µg/L for open-water) has been employed and approved, which is considered conservative given that the EA predicted an increase of phosphorus over background concentrations. The “extent of effect” for the chlorophyll *a* Action Levels reflects this prediction (Table 5-9).

The EA predictions for TP at the edge-of-mixing zone have not exceeded the original or updated EA predictions (Table 5-24). The prediction for the extent of the lake area that would be subject to TP concentrations above 5 µg/L has not been exceeded in open-water conditions, but has been exceeded on two occasions in ice-cover conditions (2008 and 2013). Concentrations of TP greater than the normal range have never occurred in an area greater than 20% of the lake (Table 5-7).

Table 5-24 Comparison of Environmental Assessment (EA) Predictions and Observations for Total Phosphorus Concentrations in Lac de Gras, 2002 to 2016

Season or Location	EA Prediction(a)		Unit	Area Affected or Concentration at Edge-of-Mixing-Zone														
	TP (µg/L)	Lake Area (km^2)		2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Ice-cover	>5(b)	64	km^2	-	-	-	-	-	18.6	112	53.5	23.8	9.2	3.6	80.6	3.5	0	37.1
Open-water	>5.3(b)	116	km^2	-	-	-	-	-	29.4	0	16.2	9.2	0	-	0	<3.5	<3.5	0
Edge-of-mixing-zone	max. 11.7(c)	0.01	µg/L	9.0	4	5.5	6	8	8	5	5	6.2	5	5.6	10.7	7.5	6.3	5.6

a) DDMI 1998b.

b) In the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b), the normal range was re-calculated and demonstrated that the EA threshold of 5 µg/L ([DDMI 1998b) was inappropriate given that it was within the normal range for the lake. Therefore, comparisons are being made to the top of the normal range (which should be considered conservative given that the EA predicted an increase of phosphorus over background concentrations).

c) Based on water column average concentration. The concentration shown under each year is the maximum water column median recorded in that year. The full range over the water column and the entire year is shown in Figure 5-6.

5.5 Summary and Conclusions

Summary of Effects

- Lac de Gras remains a P-limited, clear water (based on Secchi depth), oligotrophic lake. The increases in chlorophyll *a* concentrations appear to be associated with inputs of P from the Mine discharge and increases in micronutrients associated with increased TDS concentrations in Lac de Gras.
- The spatial extent of effects on concentrations of TP, TN, chlorophyll *a* and phytoplankton and zooplankton biomass has been variable among years in Lac de Gras. The greatest extent of effects was observed on TN in 2016 (84.7% of lake area affected), followed by chlorophyll *a* in 2016 (43.7% of lake area affected). Generally, the extent of effects on TP has been low (<0.6 to 6.5%), with a few exceptions (i.e., in 2008: 19.6%, 2013: 14%, and 2009: 9.3%). The lake area affected for phytoplankton and zooplankton biomass has been more variable compared to other indicators of eutrophication, but in 2016 both variables were affected in less than 20% of the lake.
- Weight-of-evidence effect ratings for TP were at a moderate rating from 2007 to 2013; however, in 2016, the effect rating for TP was negligible. The effect rating for TN increased from moderate to high in 2009 and has since remained at that rating. Effect ratings for chlorophyll *a* concentrations and phytoplankton and zooplankton biomass have fluctuated between low, moderate and high since 2007, although in 2016, chlorophyll *a* concentration had a high rating, while both phytoplankton and zooplankton biomass had moderate ratings.
- Action Level 1 was triggered for chlorophyll *a* because concentrations in the NF and MF areas were consistently greater than the upper bound of the normal range (0.82 µg/L) between 2007 and 2016. In 2009, 2013, 2014 and 2016, chlorophyll *a* concentrations in the NF and MF areas were greater than the upper bounds of the normal range in over 20% of the lake, resulting in Action Level 2 being triggered.

Nutrients in Mixing Zone

- Monthly and annual TP loadings from 2002 to 2016 were consistently below the Water Licence TP load limit of 300 kg/mo and 1,000 kg/yr, respectively. Although below the EQC, annual loads of TP from the NIWTP have increased between 2002 and 2016. Similar increases in annual loadings were not observed in TDP or SRP.
- Median effluent concentrations of TP at the mixing zone boundary have typically remained between 2 and 5 µg/L from 2002 to 2016, with a few exceptions observed in 2006, 2007, and 2013 to 2015.
- Median mixing zone concentrations of TDP and SRP generally remained similar, although variable between 2002 and 2011.
- Annual loadings increased from 2002 to 2006 for TN and nitrate, to 2007 for total ammonia, and to 2008 for nitrate, generally following the increase in concentrations in effluent. Annual loadings declined noticeably in 2007 for TN and nitrate, and in 2008 for ammonia, decreases continued until 2010, reflecting lower concentrations in effluent.

- Median mixing zone concentrations of TN and its constituents generally followed patterns observed in the effluent, with highest peaks occurring between 2002 and 2010.

Effects of Dust

- Elevated concentrations of TP at the MF3-1 to MF3-4 stations in 2016 are most likely related to dike construction, rather than dust deposition. The effects on TP inputs from dust deposition could not be separated from those related to dike construction.

Temporal Trends

Nutrients and Secchi Depth

- Secchi depths were deeper in the NF area compared to the MF and FF areas and more variable in the MF3 area in 2015 and 2016 compared to other years and areas, as a result of elevated concentrations of particulate material in the water column from the A21 dike construction.
- Temporal trends in TP, TDP and SRP were observed during the ice-cover and open-water seasons between 2007 and 2016 in Lac de Gras. Mid-depth concentrations of TP during the ice-cover season have significantly decreased in all areas of Lac de Gras, including the lake outlet (Station LDG-48), between 2007 and 2016. During the open-water season, significant decreases were observed in the NF and FFB areas of Lac de Gras.
- Temporal trends in TN, TDN and N+N differed among sampling areas and seasons in Lac de Gras between 2000 and 2016. During the ice-cover and open-water seasons, TN, TDN and N+N concentrations in the NF and MF areas generally exceeded the normal range, whereas in the FF areas they generally remained within the normal range, with a few exceptions.
- Data quality issues with analysis of low levels of total ammonia occurred from 2013 to 2016. In general, slight increasing trends were observed during the ice-cover season in the NF area between 2002 and 2009, and 2012 and 2016. During the open-water season, a slight increasing trend was observed in all areas, driven by high total ammonia concentrations in 2016 compared to previous years.

Biological Variables

- Temporal trends in chlorophyll *a* concentrations and phytoplankton biomass differed among sampling areas in Lac de Gras between 2007 and 2016.
- Chlorophyll *a* concentrations in the NF area exceeded the normal range in all years. Concentrations in the MF areas were generally the same or slightly less than concentrations observed in the NF area; in the FF area concentrations were typically within or at the lower limit of the normal range.
- Phytoplankton biomass varied over time in the NF and MF areas of Lac de Gras from within the normal range to both above (2016) and below (2015) in recent years.
- Temporal trends in total zooplankton biomass measured as AFDM in the NF area varied throughout the time series, generally exceeding the normal range from 2010 to 2016 at most stations.
- Zooplankton biomass in the MF and FF areas was generally within or above the upper limit of the normal range, with the exception of low zooplankton biomass in the MF3 area in 2016.

Trophic Status, Nutrient Limitation and Nutrient Relationships

- The TSI classifies Lac de Gras as oligotrophic in all areas, including the NF area.
- The limiting nutrient in Lac de Gras is P, as determined based on nutrient ratios and TSI calculations.
- The relationship between TN and TDS concentrations was moderate to strong between 2007 and 2012, but poor in 2013 and 2016. The relationship between TP and TDS concentrations was generally poor.
- The relationship between concentrations of chlorophyll *a* and TP was poor in all years except 2007, likely as a result of a limited range in TP concentration in Lac de Gras (0.5 to 7.2 µg/L). The relationship between concentrations of chlorophyll *a* and TN was moderate to strong between 2007 and 2013; however, in 2016, the relationship was poor.
- The overall relationship between chlorophyll *a* concentration and phytoplankton biomass was poor; however, a within-year evaluation indicated moderate to strong relationships between chlorophyll *a* concentrations and phytoplankton biomass between 2007 and 2016.
- A moderate to strong relationship was observed between chlorophyll *a* and TDS concentrations within each year between 2008 and 2016.
- The overall relationship between zooplankton biomass measured as AFDM and based on enumeration was moderate to strong, confirming the utility of using AFDM as an indicator of zooplankton biomass.
- A poor relationship between chlorophyll *a* concentrations and TP, but a strong relationship to TDS, suggests phytoplankton may be responding to a Mine-related eutrophication effect related to an increase in micronutrients associated with TDS, in addition to P input. The moderate to strong relationship between TN and chlorophyll *a* concentrations may be the result of the strong correlation between TN and TDS.

Concordance with Plankton Monitoring Results

- The Plankton and Eutrophication Indicators sections both suggest that Lac de Gras is continuing to exhibit a mild Mine-related nutrient enrichment effect in the NF and MF areas, while still remaining oligotrophic.

Response Framework Evaluation

- The evaluation of issues and directives related to the Response Framework for eutrophication indicators concluded that the current framework requires no updates. The current approach to managing P inputs to Lac der Gras, which consists of (1) the EQC under the Water Licence, (2) monitoring and assessment of nutrient concentrations and biological response variables under the AEMP, and (3) focusing on biological effects to trigger response actions, is adequate and has functioned well to date.

Comparison to EA Predictions

- The EA predictions for TP at the edge-of-mixing zone have not been exceeded.
- The predicted extent of the lake area that would be subject to TP concentrations above 5 µg/L has not been exceeded in open-water conditions but has been exceeded on two occasions during the ice-cover season (2008 and 2013).
- Concentrations of TP greater than the normal range have not occurred in an area greater than 20% of the lake.

6 SEDIMENT QUALITY

6.1 Introduction

This chapter provides a summary of changes in the sediment chemistry of Lac de Gras over time. The objectives of this section are:

- to summarize Mine related effects observed from 2014 to 2016 and compare these to effects observed previously (i.e., from 2007 to 2013)
- to analyze temporal trends in sediment chemistry for the period extending from baseline (i.e., 1996) to 2016

6.1.1 Background

Sediment chemistry provides information regarding chemical stressors present in the sediments and may help explain effects observed on benthic invertebrates. Sediment particle size distribution is an important factor influencing benthic community structure. Total organic carbon (TOC) aids in assessing occurrence and potential bioavailability of metals in sediment and food availability to benthic invertebrates. Therefore, monitoring of sediment quality is highly relevant to provide supporting information for biological monitoring.

DDMI has been conducting studies and monitoring programs relating to the aquatic ecosystem of Lac de Gras since 1994, with AEMP data collected under AEMP study designs Version 1.0 (2001[7] to 2006), Version 2.0 (2007 to 2011), Version 3.0 (2012 to 2016), and most recently under Version 4.0 (2017). Since there is potential for Mine effluent to affect sediment quality in Lac de Gras, the Sediment Quality program has been included as a component of the AEMP since 2001. Sediment quality is monitored only during comprehensive years since 2010.

The Sediment Quality program evaluates the effects of Mine effluent on sediment quality, and provides supporting environmental information for the interpretation of results obtained in the benthic invertebrate community survey, and to confirm the predictions set forth in the EA (DDMI 1998a). More specifically, the

[7] One year of baseline data was also collected in 2000 under Version 1.

program has been designed and implemented to identify potential spatial trends in sediment quality, and to determine whether those trends explain patterns observed in the benthic invertebrate community.

6.1.2 Component History

Lac de Gras sediments were sampled in 1996 and 1997 to assess baseline sediment quality as part of the EA for the Mine. Sediments were also sampled in 1999 by the Department of Indian Affairs and Northern Development (DIAND; now Indigenous and Northern Affairs Canada, or INAC), which provides an additional year of baseline sediment quality data. Results obtained from these early studies represent the pre-development sediment quality conditions in Lac de Gras. Sediment quality has been monitored as part of the AEMP since 2001. The original AEMP (Version 1.0) included one year of sediment data collection prior to initiation of the Mine effluent discharge to Lac de Gras, which occurred in March 2002. The first AEMP sediment quality monitoring event to occur with treated effluent being discharged to Lac de Gras was in August 2002. Sediment quality monitoring under the AEMP continued annually until 2010, after which monitoring moved to a three year cycle (Golder 2014a). Sediment quality has also been monitored annually at the edge of the mixing zone in Lac de Gras since 2002, as part of the Mine's Surveillance Network Program (SNP).

6.2 Methods

6.2.1 Data Sources

The same historical dataset as that used in the previous re-evaluation report (Golder 2016a) is used herein, and the 2014 to 2016 data have been added. Sediment chemistry data included in the evaluation of temporal trends were taken from the following data sources:

- baseline data collected by DDMI in 1996 and 1997
- baseline data collected by DIAND in 1999
- data from the Mine's SNP for the mixing zone stations, which were collected annually beginning when the Mine effluent discharge was initiated in 2002, and including data up to 2016
- data collected annually during the AEMP Version 1.0 from 2001 to 2006
- data collected annually during the AEMP Version 2.0 from 2007 to 2010
- data collected during the AEMP Version 3.0 and 3.5 in 2013 and 2016

A summary of available baseline and AEMP sediment chemistry data is provided in Table 6-1.

Table 6-1 Summary of Baseline and AEMP Sediment Chemistry Data, 1996 to 2016

Waterbody	Area	Station[a]	UTM Coordinates[b]		Distance from Diffusers[c] (km)	Years Sampled									
			Easting (m)	Northing (m)		1996[d]	1997[d]	1999[d]	2001-2006	2007	2008	2009	2010	2013	2016
Lac de Gras[f]	Near-field	NF1	535740	7153854	0.4					X	X	X	X	X	X
		NF2 (N7)	536095	7153784	0.5	X				X	X	X	X	X	X
		NF3	536369	7154092	0.9					X	X	X	X	X	X
		NF4	536512	7154240	1.1					X	X	X	X	X	X
		NF5 (LDG-NF)	536600	7153864	1.0	X	X		X	X	X	X	X	X	X
	Mid-field 1	MF1-1	535008	7154699	1.5					X	X	X	X	X	X
		MF1-2	533682	7155356	2.9										
		MF1-3	532236	7156276	4.7	X		X		X	X	X	X	X	X
		MF1-4	532494	7157657	7.2										
		MF1-5	528432	7157066	8.5									X	X
	Mid-field 2	MF2-1	538033	7154371	2.4					X	X	X	X	X	X
		MF2-2 (F14, LDG-MF)	539198	7154643	3.7				X						
		MF2-3	540365	7156045	5.4					X	X	X	X	X	X
		MF2-4	540955	7157359	6.9										
	Far-field 2[e]	FF2-1	541500	7159522	9.3										
		FF2-2	541588	7158561	8.3						X	X	X	X	X
		FF2-3	543478	7159267	10.1										
		FF2-4	543752	7158945	10.2										
		FF2-5	544724	7158879	11.4			X			X	X	X	X	X
	Mid-field 3	MF3-1	537645	7152432	2.7					X	X	X	X	X	X
		MF3-2	536816	7151126	4.2	X	X			X	X	X	X	X	X
		MF3-3	536094	7148215	7.2									X	X
		MF3-4	532545	7147011	11.0			X		X	X	X	X	X	X
		MF3-5	528956	7146972	14.6									X	X
		MF3-6	525427	7148765	18.5			X		X	X	X	X	X	X
		MF3-7	521859	7150039	22.3									X	X

Table 6-1 Summary of Baseline and AEMP Sediment Chemistry Data, 1996 to 2016

Waterbody	Area	Station(a)	UTM Coordinates(b)		Distance from Diffusers(c) (km)	Years Sampled									
			Easting (m)	Northing (m)		1996(d)	1997(d)	1999(d)	2001-2006	2007	2008	2009	2010	2013	2016
Lac de Gras(f)	Far-field 1	FF1-1	525430	7161043	13.6						X	X	X	X	X
		FF1-2	524932	7159476	12.9						X	X	X	X	X
		FF1-3	526407	7160492	12.8						X	X	X	X	X
		FF1-4	526493	7159058	11.4							X	X	X	X
		FF1-5	526683	7161824	12.8						X	X	X	X	X
	Far-field B	FFB-1	516831	7148207	26.4					X	X	X	X		X
		FFB-2	518473	7150712	25.0					X	X	X	X	X	X
		FFB-3	518048	7147557	25.2					X	X	X	X	X	X
		FFB-4	515687	7150036	27.6					X	X	X	X	X	X
		FFB-5	516533	7150032	26.8					X	X	X	X	X	X
	Far-field A	FFA-1	506453	7154021	36.8						X	X	X	X	X
		FFA-2	506315	7155271	38.3							X	X	X	X
		FFA-3	505207	7153887	38.7					X	X	X	X	X	X
		FFA-4	503703	7154081	40.2					X	X	X	X	X	X
		FFA-5 (WQ14, LDG-FF)	505216	7156657	40.0		X		X		X	X	X	X	X

Note: Shading identifies stations that were discontinued, as per the AEMP Study Design, Version 3.0 (Golder 2011b). These stations are not included in the current (2014 to 2016) Aquatic Effects Re-evaluation Report.

a) Current AEMP station names are presented, and historical sampling station names are provided in parentheses.

b) UTM coordinates are reported as Zone 12, North American Datum (NAD) 83.

c) Approximate distance from the Mine effluent diffusers along the most direct path of effluent flow.

d) For 1996, 1997 and 1999, information is provided only for stations located within AEMP areas.

e) From 2012 onwards, the FF2 area is considered to be part of the MF2-FF2 area.

f) SNP data are not listed.

UTM = Universal Transverse Mercator coordinate system; - = not applicable.

Data presented herein were produced by different analytical laboratories: Enviro Test Laboratories (ETL) in Edmonton, Alberta (2001 to 2006 data); ALS in Edmonton, Alberta (2007 to 2010 data), which purchased the ETL facility in 2007; and, Maxxam Analytics (Maxxam) in Burnaby, British Columbia (2013 data) and Calgary, Alberta (2016 data). Improvements in the analytical DLs (DLs) over the 1996 to 2016 monitoring period interfered with the temporal analysis for some variables, as results obtained using older higher DLs could only be reported as less than or below the DL. Finally, the suite of variables analyzed since baseline has expanded. As a result, data for some analytes were not available for the earlier monitoring years.

Sampling methods and laboratory procedures used during the AEMP (2001 to 2016) were generally the same as those used during baseline surveys (1996 to 1999); however, there have been some differences in methods over the years that resulted in comparability issues between recent and historical data. These issues included differences in sampling locations, sample collection methods, analytical laboratories contracted for sample analyses, DLs, and variables analyzed. These modifications to the AEMP design were introduced, as required, to allow the annual monitoring programs to meet the goals of the AEMP.

The stations sampled throughout the baseline monitoring period (1996 to 1999) and during the AEMP Version 1.0 differ from the current AEMP stations, which were established in 2007 and adjusted in 2012 (Golder 2014a). The pairing of historical stations with current AEMP stations is summarized in Table 6-2. Historical sampling stations WQ14 and LDG-FF are slightly displaced in relation to current station FFA-5. This small difference is apparent in varying sampling depths among years and likely accounts for part of the variation in concentrations of some of the variables evaluated.

Historical sampling stations not located in the vicinity of current AEMP stations were not included in the analysis. In addition, AEMP Version 2.0 stations no longer sampled in AEMP Version 3.0 were excluded from the analysis (Table 6-1).

Table 6-2 List of Historical Sediment Quality Sampling Stations Included in the Temporal Assessment

Program	Year	Historical Station	Current AEMP/SNP Station or Area
Baseline	1996	WQ2	MF1 3
Baseline	1996 to 1997	WQ6	NF5
Baseline	1996 to 1997	WQ7	MF3 2
Baseline	1997	WQ14	FFA-5
DIAND	1999	HCR 11	MF1 3
DIAND	1999	HCR 4	FF2 5
DIAND	1999	HCR 6	MF3 4
DIAND	1999	HCR 7	MF3 6
AEMP	2001 to 2006	LDG MF1, LDG MF2, LDG MF3	MF2 2
AEMP	2001 to 2006	LDG NF1, LDG NF2, LDG NF3	NF5
AEMP	2001 to 2006	LDG FF1, LDG FF2, LDG FF3	FFA-5
SNP	2002 to 2016	1645 19B	1645 19B2

Sediment collection methods differed among sampling programs. Sediment samples collected during the 1996 and 1997 baseline program and the 2001 to 2006 AEMP were collected using a sediment corer, and the top 5- to 6- cm fractions were analyzed. The 1999 samples were collected by DIAND using sediment traps. The 2007 to 2016 AEMP samples were collected using a sediment gravity-corer, and the top 1 cm fraction was analyzed for nitrogen, phosphorus and metals. From 2007 to 2010, nitrogen, phosphorus and TOC were analyzed from the top 5 cm fraction of Ekman grab samples, from 2013 onwards, grain size analysis were conducted on top 10 to 15 cm samples, while TOC was analyzed in both the top 1 cm core samples and the top 10 to 15 cm Ekman grab samples. The 2002 to 2012 SNP sediment samples were collected using an Ekman grab, and the top 5 cm fraction was analyzed. From 2013 to 2016, SNP samples were analyzed from the top 1 cm fraction of sediment core samples.

6.2.2 Data Handling

6.2.2.1 Data Screening

Initial screening of the baseline, SNP and AEMP sediment quality data sets was completed before data analyses, to identify unusually high (or low) values in the datasets and decide whether to retain or exclude anomalous data from further analyses. An explanation of the objectives and approach taken to complete initial screening is provided in Section 2.5. Initial screening of the datasets was conducted using a method based on Chebyshev’s theorem (Mann 2010) combined with the visual examination of scatterplots. Thus, the data screening approach includes a numerical method to aid in the identification of outliers, removing the subjectivity of classifying values based on visual evaluation of data in scatterplots alone. Further details on this approach are provided in Section 2.6 of the *2011 to 2013 Aquatic Effects Re-evaluation Report* (Golder 2016a), when this data screening method was first adopted. This data screening approach has also been used in subsequent AEMP Annual Reports for the Mine (i.e., 2014, 2015, 2016; Golder 2016b,c, 2017f, respectively). Data flagged as anomalous values were not included in analyses in this report or previous applicable Annual Reports.

A total of ten anomalous values were identified within the sediment quality data sets; seven values occurred between 2007 and 2010, two were identified in 2013, and one in 2016. Scatter plots for these anomalous data were presented in previous reports (Golder 2016a; 2017g).

6.2.2.2 Censored Data

Prior to conducting data analyses, data from field duplicate samples were removed, and non detect data (i.e., those below the DL) were multiplied by 0.5. Substitution with half the DL is a common approach used to deal with censored data (US EPA 2000) and is consistent with the approved methods applied in the calculation of the normal range in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b).

6.2.2.3 Quality Assurance/Quality Control

The Quality Assurance Project Plan Version 3.1 (QAPP) outlines the QA/QC procedures employed to support the collection of scientifically-defensible and relevant data addressing the objectives of the AEMP (Golder 2013a). It facilitates creation of a technically-sound and scientifically-defensible report by standardizing field sampling methods, laboratory analysis methods, data entry and storage, data analysis and report preparation activities. The reader is directed to the AEMP Annual Reports and the *2011 to 2013 Aquatic Effects Re-evaluation Report* (Golder 2016a) for a detailed description of QA/QC practices applied to the sediment quality component of the AEMP and an evaluation of the QC data for each year.

6.2.3 Data Analysis

6.2.3.1 Substances of Interest

The intent of defining SOIs was to identify a meaningful set of variables that will undergo further analyses, while limiting analyses on variables that were less likely to be affected. In the AEMP annual reports, sediment chemistry variables were identified as SOIs if concentrations in the NF area were significantly elevated relative to the far-field (FF) areas. The last comprehensive monitoring program that included collection of sediment quality samples took place in 2016 (i.e., sediment sampling was not a required component of the AEMP in 2014 and 2015 [Golder 2014c]). The assessment of temporal trends, therefore, focused on the SOIs identified for sediment quality in the 2016 AEMP Annual Report (Golder 2017c): bismuth, lead, molybdenum, TN, potassium, sodium, strontium, tin and uranium. These variables were retained, along with the additional SOIs from previous AEMP reports, which were not identified as SOIs in 2016 AEMP Annual Report: aluminum, boron, calcium, chromium, magnesium, lithium, titatnium, and vanadium.

Despite not being identified as an SOI until 2016, TP was included in the trend analysis for sediment chemistry because assessment of the effects of nutrients discharged to Lac de Gras is a key objective of the AEMP (Golder 2014a). In addition, sediment physical variables (i.e., percentage of fine sediments [silt and clay] and TOC) were evaluated, because these variables can influence the concentrations of metals and nutrients in bottom sediments and, therefore, could interfere with the interpretation of temporal trends.

6.2.3.2 Weight-of-Evidence Effect Ratings

The results of the AEMP sediment quality surveys are integrated through the WOE evaluation process, which determines the strength of evidence supporting the two broad impact hypotheses for Lac de Gras (i.e., toxicological impairment and nutrient enrichment), as described in the AEMP Study Design Version 3.5 (Golder 2014a). The WOE is not intended to determine the ecological significance or level of concern associated with a given change. The WOE effect ratings incorporate statistical comparisons of the NF and FF areas, and comparisons of the NF area to the normal range as defined in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b). The Sediment Quality data were assessed according to the WOE effect level ratings described in Section 10, and are summarized in Table 6-3.

Table 6-3 Effect Level Ratings Applied for Sediment Quality

LOE Group	Measurement Endpoint Analysis	No Response 0	Early Warning/Low ↑	Moderate ↑↑	High ↑↑↑
Sediment Quality (substances of potential toxicological concern)	Comparison to FF Areas, Normal Range, and Guidelines[(a)]	<ISQG	Statistically significant increase, NF vs FF areas	Low + NF >(ISQG+PEL)/2 (or other appropriate guideline)[(b)] **AND** NF area median >normal range	MF >(ISQG+PEL)/2 (or other appropriate guideline) **AND** MF area median >normal range **OR** NF >PEL **AND** NF area median >normal range

Notes: NF = near-field; MF = mid-field; FF = far-field; LEL = lowest effect level; PEL = probable effect level, SEL = severe effect level; SOI = substance of interest; ISQG = interim sediment quality guideline; >= greater than; <= less than. Normal ranges for each LOE group and measurement endpoint are defined and provided in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b).

a) Applied separately for each variable.

b) For example, the OMOEE (1993) [LEL+SEL]/2.

In 2013, the criteria for determining a moderate and high effect ranking for sediment chemistry were modified to include a comparison of the NF area data with sediment quality guidelines (SQGs). For 2016, this approach was maintained. The ranking procedure required that sediment chemistry variable concentrations in the NF area exceed the average of the CCME Interim Sediment Quality Guideline (ISQG) and of either the Probable Effect Level guideline (PEL) or another appropriate guideline (Ontario Ministry of the Environment and Energy [OMOEE] 1993) before a moderate or high level ranking would be applied. Sediment chemistry results from 2007 to 2016 were categorized according to the revised WOE ratings. The patterns of response observed in the WOE effects ratings over the 2007 to 2016 summary period were evaluated qualitatively to identify trends over time. Sediment quality results from prior to 2007 were excluded from the WOE effects ratings because the sample collection procedures and locations of stations differed appreciably from the AEMP Versions 2.0 and 3.0. The full suite of sediment quality variables analyzed during the AEMP Versions 2.0 and 3.0 were evaluated, with the exception of particle size. Variables recently included as SOIs did not have WOE effects ratings assessed for 2007 to 2013 period.

6.2.3.3 Correlations with Physical Variables

The physical characteristics of sediments (i.e., particle size and TOC) have the potential to influence sediment chemistry. To address this potential confounding factor, correlation analysis was used to investigate relationships between these physical variables and sediment chemistry. Since the data did not meet normality assumptions of parametric correlation analysis, Spearman's coefficient of rank correlation coefficient were generated (r_s). Correlations of SOIs with physical variables were considered significant at P <0.05. All non-detect values were removed from the dataset prior to calculating the correlations as well as previously identified outliers. No grouping was performed prior to analysis; therefore, each individual concentration (representing a composite sample form a station) was used in the analysis. The correlation analysis was conducted with R software (R Core Team 2017).

6.2.3.4 Temporal Trends

Time series plots were prepared for each SOI included in the trend analysis. Unlike previous re-evaluation reports, temporal trends were evaluated using time series plots as well as a statistical trend analysis. Trend analysis methods are described in detail in Section 2.4.2. Briefly, the approach is based on fitting two different models to the time series of each SOI considered in the analysis. The best model was decided using AIC (Akaike's Information Criterion), an objective quantity used to evaluate which model best fits the data. Data normalization using fines or TOC and data transformation using Yeo-Johnson transformations were performed as described in Section 2.4.2. Residual heteroscedasticity was examined, and heteroscedasticity effects by area, year, or predicted value were added to the model, if they improved model fit (as identified using AIC scores).

The temporal trend analysis used for lithium data differed from the general trend analysis methods described in Section 2.4.2. Because only three years of data were available for analysis (i.e., 2010, 2013, and 2016), the general methods could not be applied and the trend model described in Section 2.4.2 was simplified into a fixed effect model (with year and area as discrete covariates). The full model contained effects of year, area, and their interaction. No model selection was performed, since there was only one candidate model for analysis. Analysis was only performed on NF and FF data, since the MF data only had a single data point per year per area.

Time series plots were organized according to the sampling areas of Lac de Gras. These areas consisted of:

- the NF area (all stations included)
- two MF areas (MF1: Station MF1-3; and MF3: Station MF3-4), located along transects extending away from the NF area
- a MF area (MF2) and a FF area (FF2) (Station FF2-2) , which were grouped together, because they form a single transect (hereafter referred to as MF2-FF2 area).
- three FF areas (FF1, FFB and FFA: all stations included).

Sediment quality data collected at the mixing zone boundary (i.e., SNP stations 1645-19A, 1645-19B2, and 1645-19C; collectively referred to as 1645-19 in the figures) were included in time series plots to allow comparisons with the rest of the lake. Concentrations in the three MF areas were plotted by individual station, because each MF station is subject to a different degree of effluent exposure. Data from the FF2 area were incorporated into the figures for the MF2 area, because the FF2 area stations are located at the far northeast end of the NF-MF2-FF3 transect. Trends occurring in the FF areas (i.e., FF1, FFA and FFB) may represent natural trends in sediment quality, or potentially, the presence of Mine effluent. Non-detect data were included in the time series plots and are represented by open symbols, plotted at the DL value.

Trends in sediment chemistry over time were evaluated in relation to the normal range for Lac de Gras, which was calculated based on percentiles using reference area data (Golder 2017b). The normal ranges for SOIs were calculated using reference area data collected during the *AEMP Study Design Version 2.0* (2007 to 2010; Table 6-4).

Table 6-4 Normal Ranges for Sediment Quality Variables

Variable	Year	Unit	Normal Range	
			Lower Limit	Upper Limit
Physical Variables				
Total organic carbon	2007 to 2010	% dw	0.7	4.7
Percent fine sediment	2007 to 2010	% dw	29.5	97.0
Nutrients				
Total nitrogen	2007 to 2010	% dw	0.05	0.41
Total phosphorus	2007 to 2010	mg/kg dw	681	1,650
Metals				
Aluminum	2007 to 2010	mg/kg dw	10,723	18,433
Antimony	2010	mg/kg dw	0	0.28
Arsenic	2007 to 2010	mg/kg dw	12.99	269.4
Bismuth	2010	mg/kg dw	0.31	0.59
Barium	2007 to 2010	mg/kg dw	64.1	263.9
Beryllium	2007 to 2010	mg/kg dw	0.38	0.75
Boron	2007 to 2009	mg/kg dw	2.2	7.0
Calcium	2007 to 2010	mg/kg dw	800	1,978
Cadmium	2007 to 2010	mg/kg dw	0.06	1.09
Chromium	2007 to 2010	mg/kg dw	32.5	67.4
Cobalt	2007 to 2010	mg/kg dw	26.9	258.8
Copper	2007 to 2010	mg/kg dw	36.68	91.35
Iron	2007 to 2010	mg/kg dw	20,463	100,595
Lead	2007 to 2010	mg/kg dw	4.5	9.5
Lithium[(a)]	2010	mg/kg dw	24.9	54.2
Magnesium	2007 to 2010	mg/kg dw	4,180	9,127
Manganese	2007 to 2010	mg/kg dw	684.9	57,533
Mercury	2007 to 2010	mg/kg dw	0	0.05
Molybdenum	2007 to 2010	mg/kg dw	1.85	7.63
Nickel	2007 to 2010	mg/kg dw	46.96	268.6
Potassium	2007 to 2010	mg/kg dw	1,969	4,644
Selenium	2007 to 2010	mg/kg dw	0	1.69
Silver	2007 to 2010	mg/kg dw	0	0.2
Sodium	2007 to 2010	mg/kg dw	100	259
Strontium	2007 to 2010	mg/kg dw	6.0	20.8
Thallium	2007 to 2010	mg/kg dw	0	0.951
Tin	2007 to 2010	mg/kg dw	0	2
Titanium	2007 to 2010	mg/kg dw	366	1,066
Uranium	2007 to 2010	mg/kg dw	3.0	5.4
Vanadium	2007 to 2010	mg/kg dw	27.3	51.8
Zinc	2007 to 2010	mg/kg dw	58.1	151.4

Source: Golder 2017b.

a) Antimony and lithium were not analyzed from 2007 to 2009.

dw = dry weight.

6.3 Results

6.3.1 Summary of Effects

A total of seventeen variables analyzed from 2007 to 2013 satisfied the requirement for a low effect rating in at least one year of monitoring, because mean concentrations in the NF area were significantly greater than in the FF areas (Table 6-5). The number of variables having a low effect ranking from 2007 to 2016 varied among years, ranging from 6 variables in 2010 to 14 variables in 2008, but has not increased over time.

Since the first low effect rating for molybdenum, strontium and vanadium occurred in 2016, these metals were not evaluated in relation to the normal range for Lac de Gras in previous years (Golder 2016a). The other fourteen variables (i.e., aluminum, bismuth, boron, calcium, chromium, lead, lithium, magnesium, potassium, sodium, tin, titanium, TN, and uranium) were greater than the normal range at one or more NF stations in at least one year from 2007 to 2016. Of these fourteen variables, four (bismuth, lead, uranium and calcium to a lesser extent) were consistently elevated above the normal range in most years, while the remaining variables exceeded the normal range by a relatively small margin in one or two years. Few of the variables had concentrations above SQGs. Considerations regarding SQGs included the following:

- SQGs do not currently exist for bismuth, titanium and tin, and information is not available regarding toxicity of these metals in aquatic sediments. Results of the 2010 dike monitoring study (DDMI 2011a), and current and past AEMPs have detected no clear toxicity-related effect on the benthic invertebrate or fish communities in areas of Lac de Gras with bismuth concentrations above the background range, suggesting no sediment toxicity due to bismuth.
- SQGs do not exist for calcium, magnesium, sodium and potassium, which are common ions in freshwater.
- SQGs do not exist for aluminum and molybdenum, which are metals of ubiquitous distribution.
- SQGs do not exist for uranium. The primary route of exposure of aquatic organisms to uranium is likely from the water, rather than through food or sediment; ingestion of sediment is a possible route of exposure, though likely minimal (CCME 2011). Sheppard et al. (2005) report a predicted no-effect level for freshwater benthos to uranium of 100 mg/kg dw. Uranium, at a mean concentration as high as 17.9 mg/kg dw (maximum as high as 37.7 mg/kg dw) in the NF area in 2016 is, therefore, considered unlikely to pose a toxicological risk to aquatic biota, particularly as uranium bioavailability is reduced by complexation with humic substances and inorganic ligands found in sediments (Lenhart et al. 2000; Markich 2002; Trenfield et al. 2011a,b, 2012).
- Lead concentrations in all samples were below the CCME's ISQG and OMOEE's lowest effects level; therefore, toxicological risk to aquatic biota is not expected.
- Chromium exceeded SQGs (i.e., lowest effects level and ISQG) frequently since 2007, while the PEL was exceeded once in 2008, and the severe effect level was never exceeded. Nevertheless, chromium exceeds guidelines in both NF and FF areas, therefore, these concentrations are likely to reflect elevated background levels for Lac de Gras.
- Total nitrogen exceeded the severe effect level only in samples collected during 2013, in both NF and FF areas.

Table 6-5 Summary of Weight-of-Evidence Effect Ratings for Sediment Chemistry Substances of Interest, 2007 to 2016

Substance of Interest	2007	2008	2009	2010	2013	2016
Total nitrogen	n/a	n/a	n/a	n/a	n/a	↑
Total phosphorus	n/a	n/a	n/a	n/a	n/a	n/a
Aluminum	0	↑	0	0	↑	0
Bismuth	↑	↑	↑	↑	↑	↑
Boron	↑	↑	↑	n/a	↑	n/a
Calcium	↑	↑	↑	↑	↑	0
Chromium	↑	↑	0	0	↑	0
Lead	↑	↑	↑	↑	↑	↑
Lithium	n/a	n/a	n/a	0	↑	0
Magnesium	↑	↑	↑	0	↑	0
Molybdenum	n/a	n/a	n/a	n/a	n/a	↑
Potassium	↑	↑	↑	↑	↑	↑
Sodium	0	↑	0	↑	↑	↑
Strontium	0	↑	0	0	0	↑
Tin	0	↑	0	0	↑	↑
Titanium	↑	↑	↑	0	↑	0
Uranium	↑	↑	↑	↑	↑	↑
Vanadium	↑	↑	0	0	0	0

Notes: Effects ratings for exposure endpoints only consider increases in concentrations in the near-field area relative to far-field areas. A decrease in concentration for a given variable is considered a "no response". 0 = no effect; ↑ = low effect ranking (increase); n/a = not analyzed. Weight-of-evidence effects ratings are not reported in 2011, 2012, 2014 and 2015 because sediment quality sampling was not a required component of the AEMP in those years.

6.3.2 Correlations with Physical Variables

Concentrations of SOIs were correlated with physical variables (i.e., percent fine sediments and TOC) as summarized in Table 6-6. Correlation coefficients for all variables with TOC and percent fine sediments are available in Appendix 6A (Table 6A-1). High levels of significance were found for correlations with some metals due to the large number of samples included in the analysis (i.e., greater than 200 for some variables). However, these results were not necessarily accompanied by high correlation coefficients. Stronger correlations ($r_s > |0.40|$) with fine sediments were detected for magnesium, sodium, potassium and titanium, while strontium and TN had stronger correlations with TOC.

Percent fine sediments were significantly correlated to 12 SOIs in Lac de Gras sediments. All correlations were positive with the exception of molybdenum. Therefore, the influence of sediment grain size could have interfered with detecting temporal trends for several SOIs. Time series plots for fine sediments, however, demonstrated that particle size distribution was generally similar among years. These results indicated that although particle size influenced sediment chemistry, it was unlikely to interfere with the assessment of temporal trends.

Metals with pronounced Mine-related spatial and temporal trends in Lac de Gras (i.e., bismuth, lead and uranium) were only weakly correlated with percent fine sediments which suggests the spatial distribution of these metals in Lac de Gras sediments more likely reflects inputs from the Mine effluent, than the influence of physical factors.

Spearman rank correlations between sediment chemistry variables and TOC were significant for 11 SOIs (Table 6-6). As expected, the highest positive correlation with TOC was observed for TN, because both variables are related to organic matter content of sediments. Other SOIs demonstrated significant positive and negative correlations with TOC (Table 6-6). Since metals tend to bind to organic matter, positive correlations were expected. Negative correlations may be spurious, resulting from low and variable sediment TOC, combined with large sample sizes.

Table 6-6 Results of Spearman Rank Correlations between Sediment Quality Variables and Percent Fine Sediment and Total Organic Carbon

Variable	Fine Sediment (%)		Total Organic Carbon (%)	
	n	r_s	***n***	r_s
Total Nitrogen	203	-0.082	204	**0.743********
Total Phosphorus	240	0.105	262	**0.124***
Aluminum	242	**0.344********	302	0.042
Bismuth	174	**0.176***	221	**-0.397*******
Boron	174	0.106	231	-0.061
Calcium	232	**0.242*******	282	0.116
Chromium	242	**0.396********	302	-0.078
Lead	242	0.028	302	**0.147***
Lithium	106	**0.253******	117	-0.177
Magnesium	242	**0.406********	291	**-0.182******
Molybdenum	241	**-0.267********	302	**0.333********
Potassium	233	**0.522********	282	**-0.314********
Sodium	227	**0.429********	276	**-0.212*******
Strontium	242	-0.044	302	**0.462********
Tin[(a)]	109	0.044	125	**-0.189***
Titanium	238	**0.511********	298	**-0.365********
Uranium	238	**0.211******	298	-0.028
Vanadium	242	**0.349********	302	-0.006

Notes: Notes: Probability of type one error: * = <0.05, ** = <0.01, *** <0.001, ****<0.0001.

Bolded values indicate significant correlations between sediment chemistry variables and percent fines or TOC. Percent fine substrate is calculated as the sum of percent clay and silt in a sediment sample.

a) Correlations for tin were based on a reduced sample size because approximately half the dataset for tin consisted of non-detect values which were removed from the analysis.

n = number of samples; r_s = Spearman rank correlation coefficient.

6.3.3 Temporal Trends

Temporal trends in sediment quality variables were analyzed in two steps:

- Concentrations were compared to normal ranges and patterns in the data were assessed visually for all variables (subsections 6.3.3.1 to 6.3.3.3).
- Statistical trend analysis was completed for each SOI (Section 2.5).

In the analysis of 18 sediment variables, temporal trends for 5 variables were parabolic and 12 variables were were linear, and one (lithium) was not included in the AIC model selection (Table 6-7). All variables, except lithium, had data ranging from at least 2001 to 2016. Trend analyses were performed following normalization of the data by TOC or percent fines, where applicable, and transformation of data using Box-Cox transformations.

Table 6-7 Sediment Variable AIC Model Selection, 2001 to 2016

Variable	Years	Yeo-Johnson transformation(a)	Parabolic AIC Score	Linear AIC Score	Normalizing variable
Total Nitrogen	2001 to 2016	-0.5	-3,265	-3,237	TOC
Total Phosphorus	2001 to 2016	-1	-3,226	-3,176	TOC
Aluminum	2001 to 2016	-2	-6,309	-6,325	Fines
Bismuth	2001 to 2016	-0.5	-224	-215	TOC
Boron	2001 to 2013	0.5	311	301	Fines
Calcium	2001 to 2016	-0.5	-1,085	-1,095	Fines
Chromium	2001 to 2016	-1	-1,293	-1,310	Fines
Lead	2001 to 2016	0	-24	-18	-
Lithium	2010 to 2016	-0.5	-	-	Fines
Magnesium	2001 to 2016	-0.5	-2,842	-2,859	Fines
Molybdenum	2001 to 2016	-0.5	-139	-153	Fines
Potassium	2001 to 2016	-0.5	-1,315	-1333	Fines
Sodium	2001 to 2016	0.5	993	978	Fines
Strontium	2001 to 2016	0	167	159	TOC
Tin	2001 to 2016	0	315	373	Fines
Titanium	2001 to 2016	0	1,112	1,105	Fines
Uranium	2001 to 2016	-1	-612	-622	Fines
Vanadium	2001 to 2016	-1	-1,247	-1,263	Fines

a) See Section 2.4.2.1 for details on transformation approach.

Notes: Lithium was excluded from the AIC model selection, because only three years of data were available for analysis.

Shaded cells indicate the chosen model, typically the lower AIC score; whether the AIC is positive or negative, does not influence the analysis – it is the value relative to the AIC scores of other candidate models on the same dataset that needs to be considered. Log transformation was performed using Yeo-Johnson transformation, which uses an offset

= not applicable; AIC = Akaike's information criterion.

Linear mixed model results, including significance of each coefficient for all SOIs analyzed, are presented in Table 6-8. Lithium was the only variable analyzed with a fixed effects model; results for this variable are summarized in Table 6-9.

For variables with linear trends, Table 6-10 summarizes multiple-comparison tests of parabolic trends for 2010, 2013 and 2016 data separately for 17 SOIs, while Table 6-11 summarizes multiple comparisons for lithium. Table 6-12 summarizes the significance of differences of linear slopes from zero.

Table 6-8 Significance of Sediment Mixed Effects Models, 2001-2016

Variable	Coefficient	Numerator DF	Denominator DF	F-value	*P*-value
Total Nitrogen	Area	6	16	5.708	0.002
	Year	1	107	28.181	<0.001
	Year²	1	107	28.161	<0.001
	Area × Year	6	107	5.705	**<0.001**
	Area × Year²	6	107	5.702	**<0.001**
Total Phosphorus	Area	6	16	4.200	0.010
	Year	1	123	9.494	0.003
	Year²	1	123	9.535	0.002
	Area × Year	6	123	4.199	**<0.001**
	Area × Year²	6	123	4.198	**<0.001**
Aluminum	Area	6	16	1.148	0.380
	Year	1	129	0.450	0.503
	Area × Year	6	129	1.146	0.340
Bismuth	Area	6	16	0.377	0.883
	Year	1	122	4.308	0.040
	Year²	1	122	4.316	0.040
	Area × Year	6	122	0.378	0.892
	Area × Year ²	6	122	0.378	0.892
Boron	Area	6	16	2.734	0.050
	Year	1	84	2.659	0.107
	Area × Year	6	84	2.740	**0.018**
Calcium	Area	6	16	1.853	0.152
	Year	1	129	5.699	0.018
	Area × Year	6	129	1.853	0.094
Chromium	Area	6	16	1.492	0.243
	Year	1	129	0.749	0.388
	Area × Year	6	129	1.493	0.185
Lead	Area	6	16	2.085	0.113
	Year	1	123	2.792	0.097
	Year²	1	123	2.792	0.097
	Area × Year	6	123	2.084	0.060
	Area × Year²	6	123	2.083	0.060
Magnesium	Area	6	16	2.568	0.062
	Year	1	128	0.008	0.928
	Area × Year	6	128	2.562	**0.022**

Table 6-8 Significance of Sediment Mixed Effects Models, 2001-2016

Variable	Coefficient	Numerator DF	Denominator DF	F-value	*P*-value
Molybdenum	Area	6	16	1.639	0.200
	Year	1	128	15.474	<0.001
	Area × Year	6	128	1.640	0.141
Potassium	Area	6	16	1.000	0.458
	Year	1	128	0.269	0.605
	Area × Year	6	128	1.000	0.428
Sodium	Area	6	16	2.512	0.066
	Year	1	129	19.730	<0.001
	Area × Year	6	129	2.518	**0.024**
Strontium	Area	6	16	1.454	0.255
	Year	1	130	0.032	0.859
	Area × Year	6	130	1.452	0.200
Tin	Area	6	16	0.863	0.542
	Year	1	122	2.099	0.150
	$Year^2$	1	122	2.110	0.149
	Area × Year	6	122	0.863	0.524
	Area × $Year^2$	6	122	0.863	0.525
Titanium	Area	6	16	3.053	0.035
	Year	1	129	29.845	<0.001
	Area × Year	6	129	3.052	**0.008**
Uranium	Area	6	16	1.754	0.173
	Year	1	129	3.327	0.070
	Area × Year	6	129	1.776	0.109
Vanadium	Area	6	16	2.293	0.087
	Year	1	127	0.457	0.500
	Area × Year	6	127	2.296	**0.039**

DF = degrees of freedom. Significance of the interaction terms was evaluated at a *P*-value of <0.05.

Table 6-9 Significance of Fixed Effects Models for Lithium, 2001 to 2016

Variable	Coefficient	Numerator DF	Denominator DF	F-value	*P*-value
Lithium	Area	3	16	2.45	0.101
	Year[(a)]	2	31	28.73	<0.001
	Area × Year[(a)]	6	31	4.41	**0.002**

a) Year was considered as a discrete covariate.
DF = degrees of freedom. Bolded values are significant at *P*<0.05

Table 6-10 Multiple Comparisons of Parabolic Trends in 2010, 2013 and 2016

Variable	Year	Area / Station						
		NF	MF1-3	FF2-2	MF3-4	FF1	FFB	FFA
Total Nitrogen	2010	a	a	a	a	a	a	a
	2013	ab	ab	ab	ab	b	ab	a
	2016	a	a	a	a	a	a	a
Total Phosphorus	2010	a	a	a	a	a	a	a
	2013	b	ab	ab	ab	a	ab	ab
	2016	c	abc	bc	abc	a	ab	ab
Bismuth	2010	c	bc	bc	ab	ab	ab	a
	2013	d	cd	bcd	abc	abc	ab	a
	2016	d	bcd	cd	abc	abc	a	ab
Lead	2010	b	ab	ab	ab	a	a	a
	2013	b	ab	ab	ab	a	a	a
	2016	b	ab	ab	ab	ab	ab	a
Tin	2010	a	a	a	a	a	a	a
	2013	a	a	a	a	a	a	a
	2016	a	a	a	a	a	a	a

Note: Different letters designate areas/stations that are significantly different at the $P<0.05$ level.

Table 6-11 Multiple Comparisons for Lithium in 2010, 2013 and 2016

Variable	Year	Area			
		NF	FF1	FFB	FFA
Lithium	2010	a	ab	ab	b
	2013	b	ab	a	ab
	2016	a	a	a	a

Note: Different letters designate areas/stations that are significantly different at the $P<0.05$ level.

Table 6-12 Estimated Significance of Differences of Linear Slopes From Zero

Variable	Area / Station						
	NF	MF1-3	FF2-2	MF3-4	FF1	FFB	FFA
Aluminum	0.468	0.699	0.160	0.198	0.544	0.110	0.806
Boron	0.460	0.795	0.860	**0.035** ↓	0.837	**<0.001** ↓	**0.012** ↓
Calcium	**0.008** ↑	0.904	**0.038** ↑	0.256	0.297	0.148	0.123
Chromium	0.216	0.647	**0.040** ↑	0.602	0.854	0.100	0.626
Magnesium	0.819	0.093	**0.028** ↑	**0.023** ↑	0.854	0.127	0.867
Molybdenum	**<0.001** ↑	**0.046** ↑	0.675	0.791	0.071	0.167	**<0.001** ↑
Potassium	0.158	0.940	**0.009** ↑	0.591	0.743	0.433	0.827
Sodium	**0.003** ↓	0.058	0.794	0.288	0.308	**0.002** ↓	**<0.001** ↓
Strontium	**0.005** ↑	0.828	0.819	0.865	0.236	0.410	**0.013** ↑
Titanium	**<0.001** ↓	**0.001** ↓	0.482	**0.033** ↓	0.103	**<0.001** ↓	0.336
Uranium	**<0.001** ↑	0.452	0.481	0.774	0.264	0.519	0.508
Vanadium	0.835	0.890	**0.001** ↑	0.473	0.695	0.090	0.970

Note: **Bold** text = *P*-value significant at <0.05. Arrows after significant *P*-values indicate direction of trend.

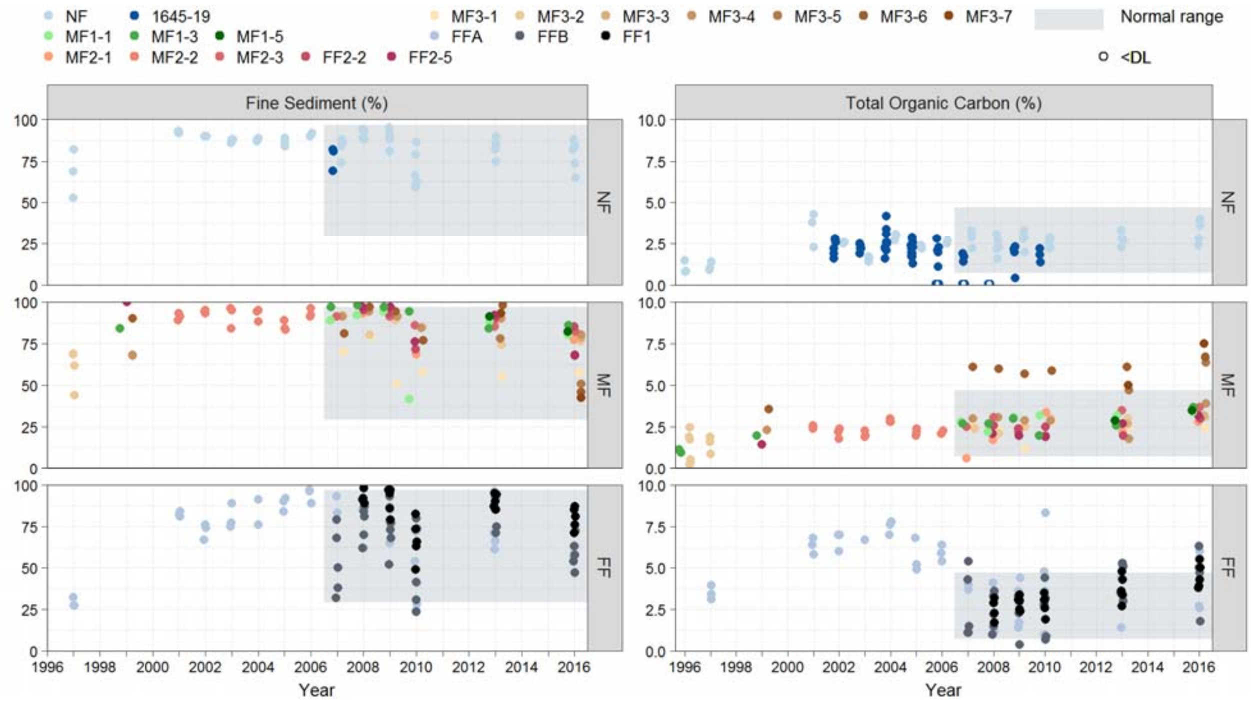
6.3.3.1 Physical Characteristics of Sediment

Physical characteristics of aquatic sediments are important because they influence concentrations of chemistry variables. Percent fines (i.e., silt plus clay) was high in most sediment samples from Lac de Gras (i.e., up to 100%; Figure 6-1). Percent fines fluctuated over time in each sampling area, and the pattern in these temporal fluctuations was similar across all areas, (although more pronounced in the FF areas). Lower values (i.e., <50%) were occasionally measured in the western end of the MF3 area and in the FFA and FFB areas, which suggests that sediments in areas in the western portion of the lake were generally coarser. Median values for percent fines in FFB were the lowest (70%), while the MF2-FF2 area were the highest (92.5%), with percent fines values typically 10% to 20% less than in other areas of Lac de Gras. In addition, stations in those areas generally had greater variability in percent fines than those in sampling areas located closer to the diffusers (i.e., NF, MF1, and MF2).

Sampling methods used to collect sediments for analysis of TOC were modified in 2013 to be consistent with collection procedures used for total metals and nutrients. From 1996 to 2010, TOC was analyzed from the top 5 cm of core samples (1996 to 2006) or top 5 to 6 cm portion of Ekman grab samples (2007 to 2010) (Section 6.2.1). However from 2013 onwards, the top 1 cm portion of the core samples was analyzed as well as the top 10 to 15 cm from Ekman grabs. The top 1 cm core sample results are reported here to allow a comparison of more consistent data across years. Given that the earlier TOC data were collected from a deeper sediment layer, TOC concentrations reported from 1996 to 2010 are less representative of recent depositional conditions than samples collected in 2013 and 2016. Comparison of the 2013 and 2016 TOC data with results from 1996 to 2010, however, indicated that values were generally within the same ranges in all years (Figure 6-1).

Concentrations of TOC has remained within a similar range over time at most sampling areas of Lac de Gras, specifically at NF and MF, although results above the normal range became more frequent after 2007 (i.e., with the addition of more stations at the west end of the MF3 area). TOC concentrations decreased notably at the FFA area between 2001 and 2006 to 2007, and have been generally increasing since (Figure 6-1). Lower TOC concentrations after 2006 reflect a change in sampling location (i.e., closer to the centre of the lake) from the previous sampling location closer to the north shore (which was alos deeper). A similar pattern in TOC concentration was observed for NF and MF areas, and most results remained within the normal range. Spatial differences in TOC concentration were typically small over the years. While prior to 2007, the FFA area median TOC was at least two times greater than the NF, from 2007 to 2010, median TOC concentrations were similar among areas, and still slightly greater in the FF area.

Figure 6-1 Percent Fine Sediments and Total Organic Carbon in Sediments, 1996 to 2016



6.3.3.2 Nutrients

Concentrations of phosphorus and nitrogen have been monitored in Lac de Gras sediments since 1996 and 2001, respectively. Prior to 2013, samples for analysis of TP and TN were collected from a deeper sediment layer (top 5 cm) using the methods described in Section 6.2.1 for TOC. Since 2013, sample collection methods were modified to target more recent sediment deposits, and only the top 1 cm portion of core samples were retained for chemical analysis.

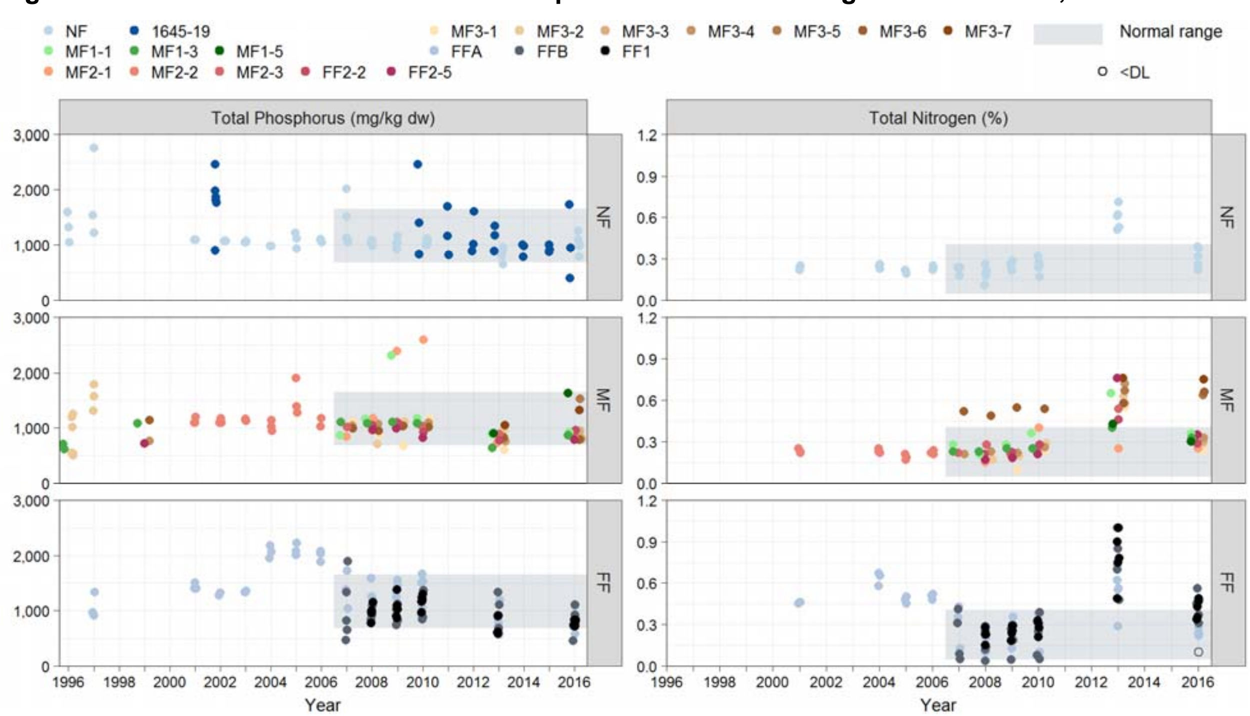
Concentrations of TP were highly variable over the years in all areas, but despite this variability, the general patterns suggest stability within the normal range (Figure 6-2). Samples collected in 2013 and 2016 had slightly lower TP concentrations in Lac de Gras sediments, compared to values reported historically for 5 cm core or Ekman grab samples. Spatial variability in TP is not evident, with NF and MF areas similar to FF areas in most years. Concentrations of TP at the mixing zone boundary were generally greater than at the NF stations in most years when data were available for both areas.

Data for TOC-normalized TP was fit to a parabolic model (Table 6-7). Both shallow and steep parabolas were estimated, depending on stations and areas. However, in general, the fitted models indicated decreases in TP concentration for recent years, with peaks between 2007 and 2010 (Figure 6-3). Areas with long-term data available (i.e., NF and FFA) had very distinct predictions, with a shallower parabola estimated for the NF area. First and second order interactions for area and year were significant, which was expected due to clear differences in trends among areas (Figure 6-3). Although 2010 TOC-normalized TP concentrations were not significantly different among areas, in 2013 and 2016, the NF area was significantly different from the FF1 area (Table 6-10).

The temporal pattern in TN concentration was similar in all areas after 2006, excluding the MF3 area, where concentrations were greater at Station MF3-5 (Figure 6-2), consistent with greater TOC content of sediments (Figure 6-1). Excluding the 2013 dataset (which was generated using different lab methods from other years' datasets), TN concentrations were within the normal range for Lac de Gras. Although results from samples collected in 2013 suggested that there were clear differences in the concentration of TN between sediment-core sampling depths, 2016 data did not corroborate this. Concentrations of TN were considered atypical in 2013, and inconsistent with other years' results, most likely due to a difference in laboratory methods in 2013 relative to other years (Golder 2017c). Median values reported in top 5 cm samples from 2001 to 2010 (AEMP 1.0 and AEMP 2.0) were similar or slightly less than top 1 cm concentrations in the 2016 samples. The increase in concentrations observed in 2013 samples, however, was similar between all areas of Lac de Gras, also suggesting differences in laboratory methods may be the cause.

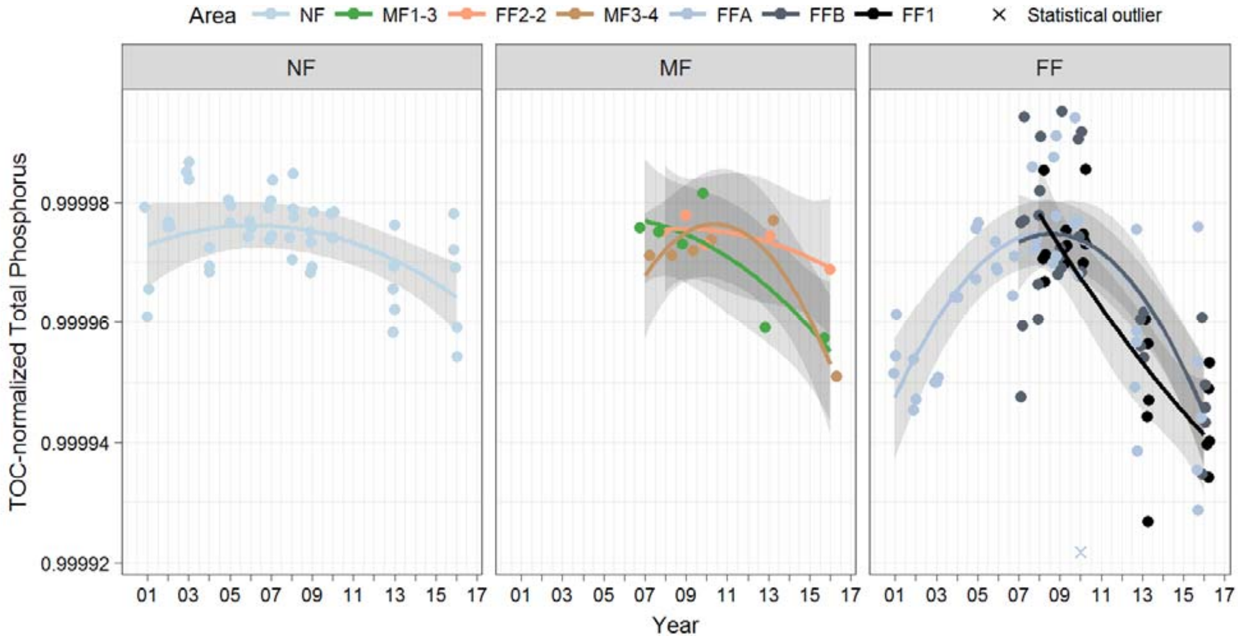
A parabolic model was selected for the trend analysis of TOC-normalized TN (Table 6-7). In both FFA and NF areas, where long-term data were available, the model predicted a shallow parabolic increase since 2001; in the FFA area, the parabola was estimated to reach its peak and start decreasing in 2013 to 2016 (Figure 6-4). In areas where data were only available between 2007 and 2016 (i.e., FF1, FFB, and the three MF areas), estimated trends were steep parabolas, increasing to a peak in 2013 and rapidly decreasing to fit the data collected in 2016. These trends should be interpreted with caution, due to the uncertainty in the 2013 TN data noted above. Temporal trends among areas were significantly different, as indicated by significant interaction terms in Table 6-8. Multiple comparisons of areas in 2010, 2013, and 2016 only identified a single significant difference, between FF1 and FFA in 2013 (Table 6-11), which also may have resulted from the unusual TN results for 2013.

Figure 6-2 Concentrations of Total Phosphorus and Total Nitrogen in sediments, 1996 to 2016



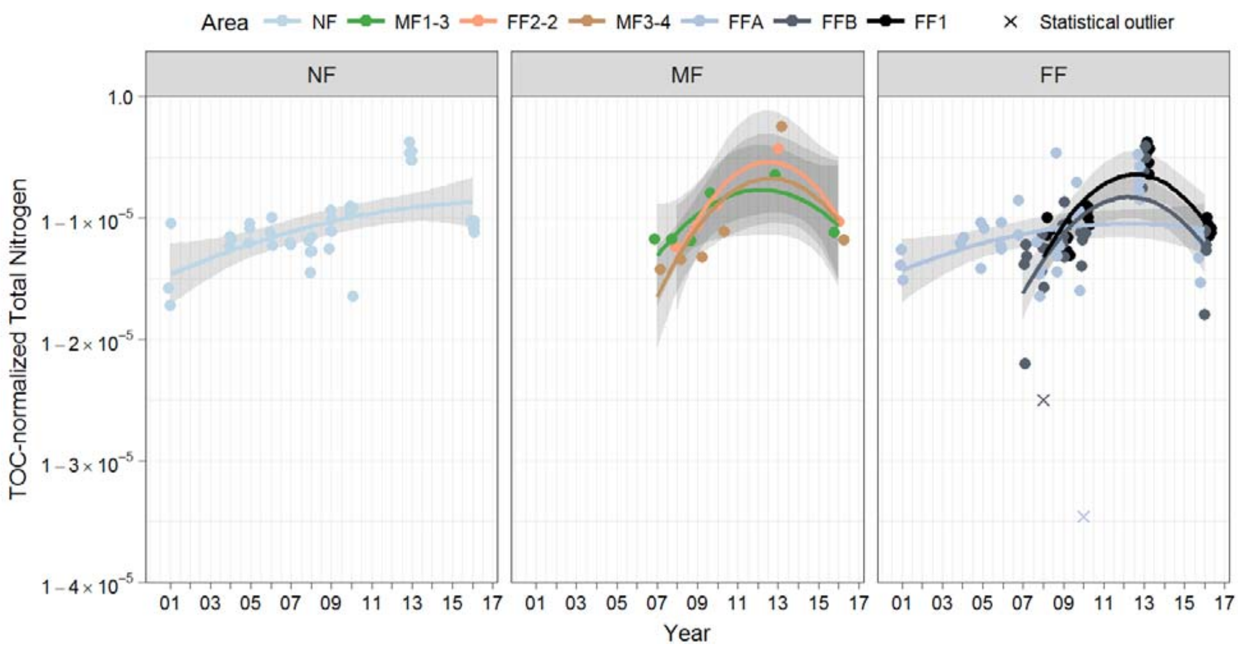
Note: TN results for 2013 are atypical due methodological issues during laboratory analysis.

Figure 6-3 Temporal Trend Analysis of TOC-Normalized Total Phosphorus



Note: Grey ribbons are 95% confidence intervals.

Figure 6-4 Temporal Trend Analysis of TOC-Normalized Total Nitrogen



Note: Grey ribbons are 95% confidence intervals. TN results for 2013 are atypical due methodological issues in laboratory analysis.

6.3.3.3 Metals and Metalloids

Time series plots and trend analysis plots for metals and metalloids are presented in Figures 6-5 to 6-36.

Concentrations of most metals in Lac de Gras have remained within a similar range over the 1996 to 2016 monitoring period. The concentration of lithium, which was only analyzed during the baseline program (1996 and 1997) and during the two most recent AEMP surveys (2010, 2013 and 2016), has not varied substantially since the baseline monitoring period (Figure 6-18). Out of 31 metals analyzed, only 3 (i.e., bismuth, lead and uranium) were beyond their respective normal ranges at most sampling locations and during most years. Occasional exceedances of the normal range were noted primarily during the baseline period (1996 to 1999) and AEMP Version 1.0 years (2001 to 2006), which were typically more variable than the 2007 to 2016 data. Bismuth, lead and uranium have had NF area concentrations consistently greater than the normal range throughout the AEMP monitoring period. Time series plots for bismuth, lead and uranium indicate that the concentrations of these metals increased at the mixing zone boundary and in the NF area in 2002, following Mine start-up (Figure 6-8, Figure 6-16 and Figure 6-33). During the past two sampling events (i.e. 2013 and 2016) somewhat higher variability was observed for bismuth and uranium. In the MF areas, increasing concentrations of uranium were also observed in 2002, while increasing MF concentrations of bismuth and lead were first observed in 2006 or 2007.

Uranium is a water quality SOI, and lead is regularly detected in the effluent; therefore, effluent is a likely source of these two metals. At the standard DL of 0.0002 µg/L, bismuth is typically not detected in the effluent or at AEMP water quality sampling stations. Hence, there is potentially another source of these metals to the sediments. The response patterns identified in this report for bismuth, lead and uranium in bottom sediments are consistent with the results of the dike monitoring studies (DDMI 2011a), which identified greater concentrations of these metals in the vicinity of the Mine effluent diffusers, as well as near the A154 and A418 dikes. Results of the most recent dike monitoring study indicated that bismuth, lead and uranium concentrations were greatest along the two transects closest to the diffusers, and concentrations decreased with distance along each of these transects. Concentrations at transects farther away from the effluent discharge were lower, but they still demonstrated gradual decreases with distance away from the dikes. These results indicate that, in addition to Mine effluent, other factors related to dike construction may have contributed to the increases in concentrations observed in this area.

Arsenic, cobalt and iron showed similar patterns of increasing concentrations after 2003 in the FFA, FFB and FF1 areas (Figures 6-9, 6-15 and 6-16), which may in part reflect a change in the range of concentrations due to a greater number of samples being collected in the FF areas in the mid-2000s. A similar pattern was not evident in the NF and MF areas; therefore, it is not possible to link these changes to potential Mine-related inputs. Other metals such as barium, cadmium, manganese, nickel, thallium and zinc have had greater variability in results since 2007, which is likely also due to the increase in number of sampling stations. Changes in sediment sampling methods did not appear to have an effect on metal concentrations, as measured concentrations since 2013 are within the same ranges as results from prior to 2013 (i.e., 5 cm core or Ekman grab samples).

Trend analysis was conducted for metals identified as SOIs. For most of these metals, a first order linear model represented the best fit to the data according to AIC analysis (Table 6-7). Exceptions were TOC-normalized bismuth, lead and TOC-normalized tin, which exhibited best fit to parabolic models.

Trend analyses for bismuth and lead (Figures 6-9 and 6-17) indicate parabolic increases peaking between 2010 and 2013 for the NF and FFA areas. TOC-normalized bismuth concentrations in the MF areas and the other two FF areas decreased slightly since 2007. Lead concentrations also declined since 2007, reaching lowest predicted values in 2013, and then increasing again in 2016 at FF2 and MF3 stations, and the FF1 and FFB areas. These variables had no significant Area × Year interactions (Table 6-8), indicating that trends were generally similar among the areas and stations included in the analysis. TOC-normalized bismuth results from the NF area were significantly different from FF1, FFA, FFB and MF3-4 areas in 2010, 2013 and 2016, but not from FF2-2 and MF1-3, which are closer to the NF area. For lead, NF results were significantly different from FF1, FFB and FFA in 2010 and 2013, but only from FFA in 2016.

Tin concentrations in sediments were mostly within the normal range in all sampling areas included in the trend analysis (Figure 6-30), and Mine effects were not apparent. The parabolic trends in the NF and FFA areas (both with long-term datasets) were nearly identical, while trends from other areas and stations analyzed were similar to the declining arm of the NF and FFA parabolas (Figure 6-31). Mixed effects model analysis for tin did not return significant results for interactions (Table 6-8). Multiple comparisons of parabolic trends in 2010, 2013 and 2016 did not identify any significant differences among areas (Table 6-10). If data collected prior to 2007 were removed from the analysis, a linear model would be expected for all areas based on Figure 6-31.

Linear models were selected for analyses of 12 sediment variables (Table 6-7), seven of which did not have significant Area × Year interactions: aluminum, calcium, chromium, molybdenum, potassium, strontium and uranium. Some of the slopes of the models fit to these variables were significantly different from zero (all seven except for aluminum had at least one significant slope) (Table 6-12). All significant trends for these variables were increasing over time. In all cases when multiple areas had slopes significantly different from zero, the direction of the trends were the same, consistent with the lack of significant Area × Year interactions. Linear trends in the NF area were significantly increasing for calcium, molybdenum, strontium and uranium (Figures 6-13, 6-23, 6-29 and 6-34).

Five variables (boron, magnesium, sodium, titanium and vanadium; all fines-normalized) had significant interactions between area and year, indicating that temporal trends were significantly different among areas (Table 6-8). However, all trends with significant slopes were in the same direction for each of these variables (Table 6-12). Increasing trends were only detected for magnesium (MF3-4 and FF2-2) and vanadium (FF2-2). In the NF area, sodium (Figure 6-27) and titanium (Figure 6-32) had significantly declining trends.

The lithium fixed effects model had significant interaction between year and area, indicating that temporal trends among areas were significantly different (Table 6-9). Multiple comparisons demonstrated that for 2010, lithium concentration in FFA was significantly greater compared to the NF area (Table 6-11). In 2013, the NF lithium concentration was significantly greater than that in FFB, while in 2016, there were no differences among groups (Figure 6-19).

In summary, in addition to bismuth, lead and uranium (which have been identified previously as variables that demonstrate a Mine-related increase in concentration in lake sediments), calcium, molybdenum and strontium were found to have significant increasing trends at NF area (Table 6-12). Calcium concentrations was also significantly increasing at FF2-2, and similar, but non-significant increases were also apparent in the two FF areas (i.e., FFB and FF1; Figure 6-13). Similarly, molybdenum concentrations were increasing in sediments in all areas of Lac de Gras (Figure 6-23), but only the steeper trends in the NF and FFA areas and Station MF1-3 were statistically significant. Strontium concentrations were significantly increasing in the NF and FFA areas, but not in the MF areas (Figure 6-29). Therefore, the increasing trends observed for these variables in sediments in the NF area are not unique relative to other areas of Lac de Gras distant from the discharge (as observed for bismuth, lead and uranium), and are unlikely to represent Mine-related effects on sediment quality.

Figure 6-5 Concentrations of Aluminum and Antimony in Sediments, 1996 to 2016

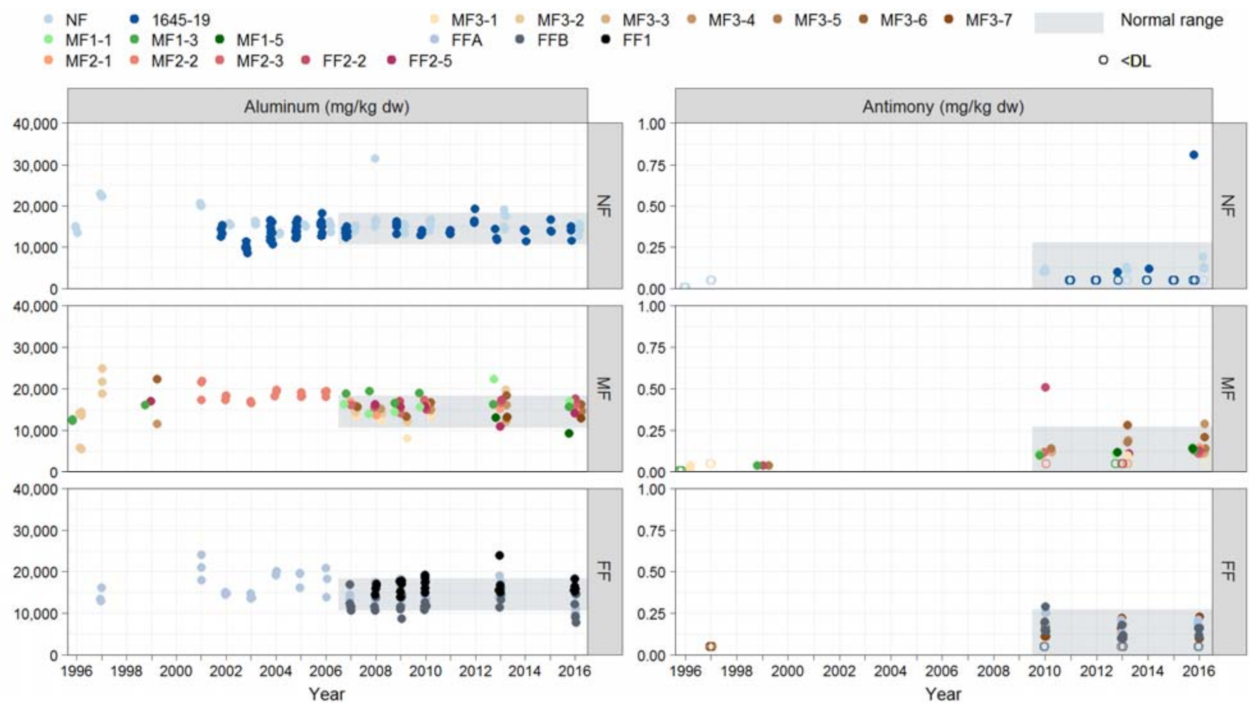
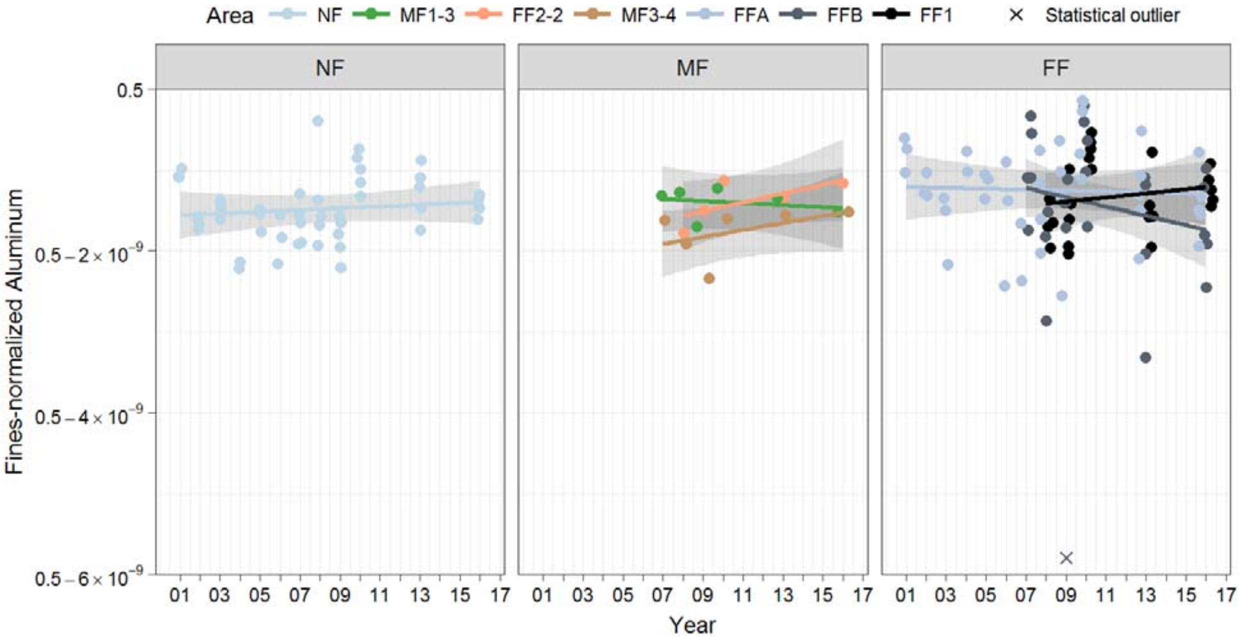


Figure 6-6 Temporal Trend Analysis of Fines-Normalized Aluminum



Note: Grey ribbons are 95% confidence intervals.

Figure 6-7 Concentrations of Arsenic and Barium in Sediments, 1996 to 2016

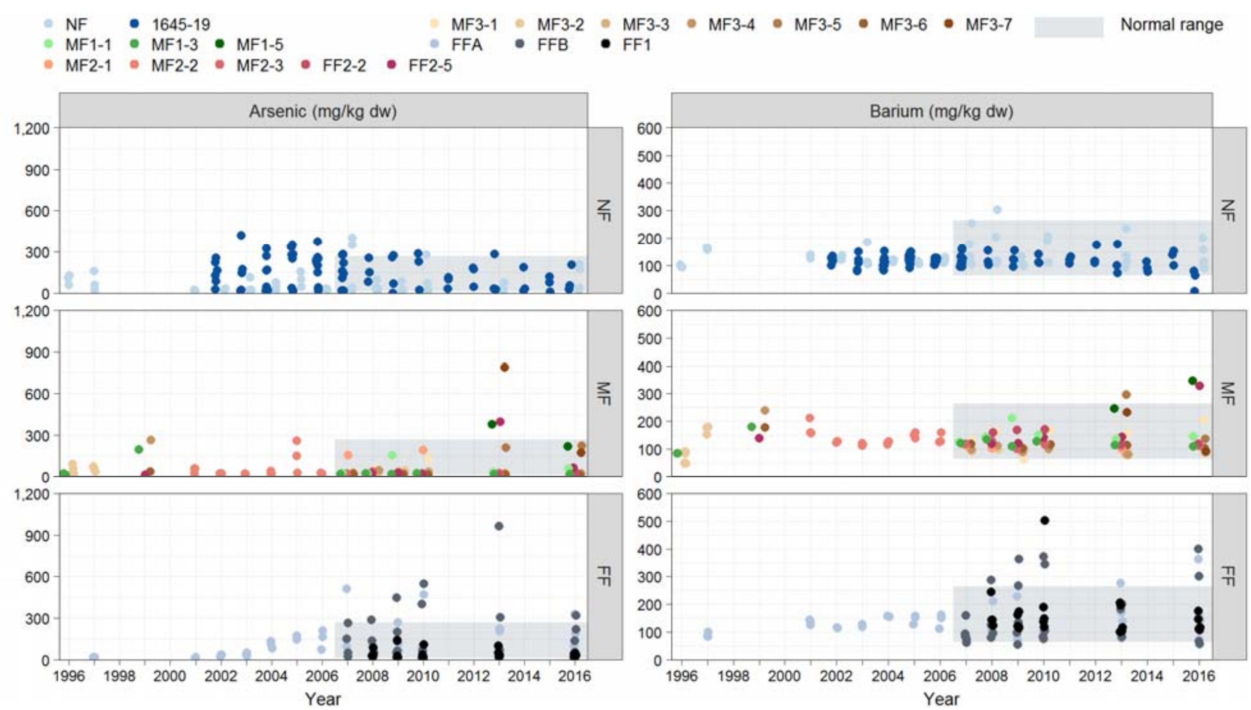
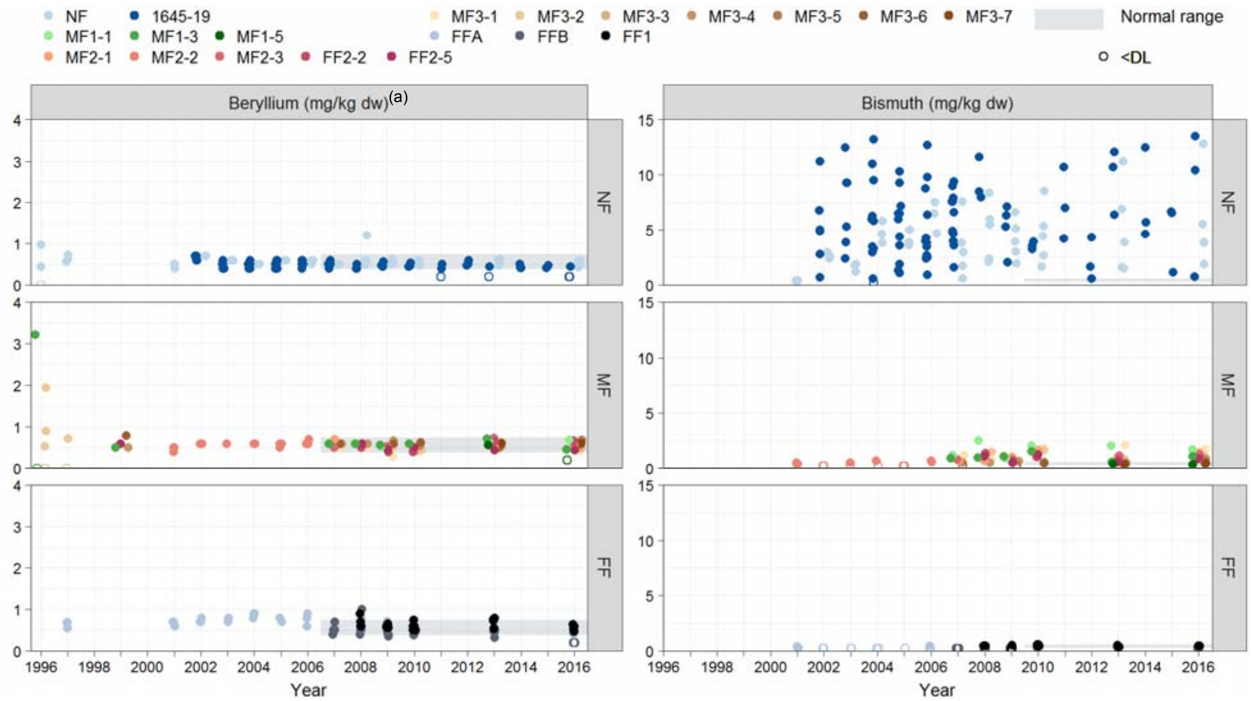
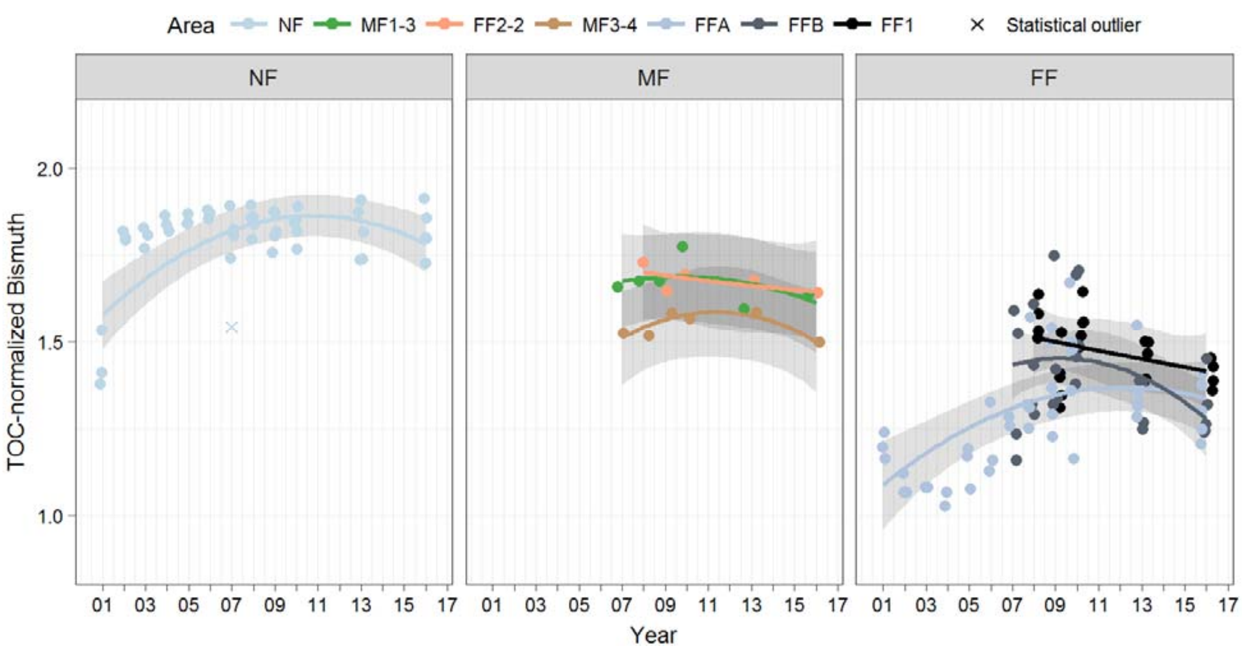


Figure 6-8 Concentrations of Beryllium and Bismuth in Sediments, 1996 to 2016



(a) Anomalous value of 86.8 mg/kg omitted from graph for improved visualization of trends.

Figure 6-9 Temporal Trend Analysis of TOC-Normalized Bismuth



Note: Grey ribbons are 95% confidence intervals.

Figure 6-10 Concentrations of Boron and Cadmium in Sediments, 1996 to 2016

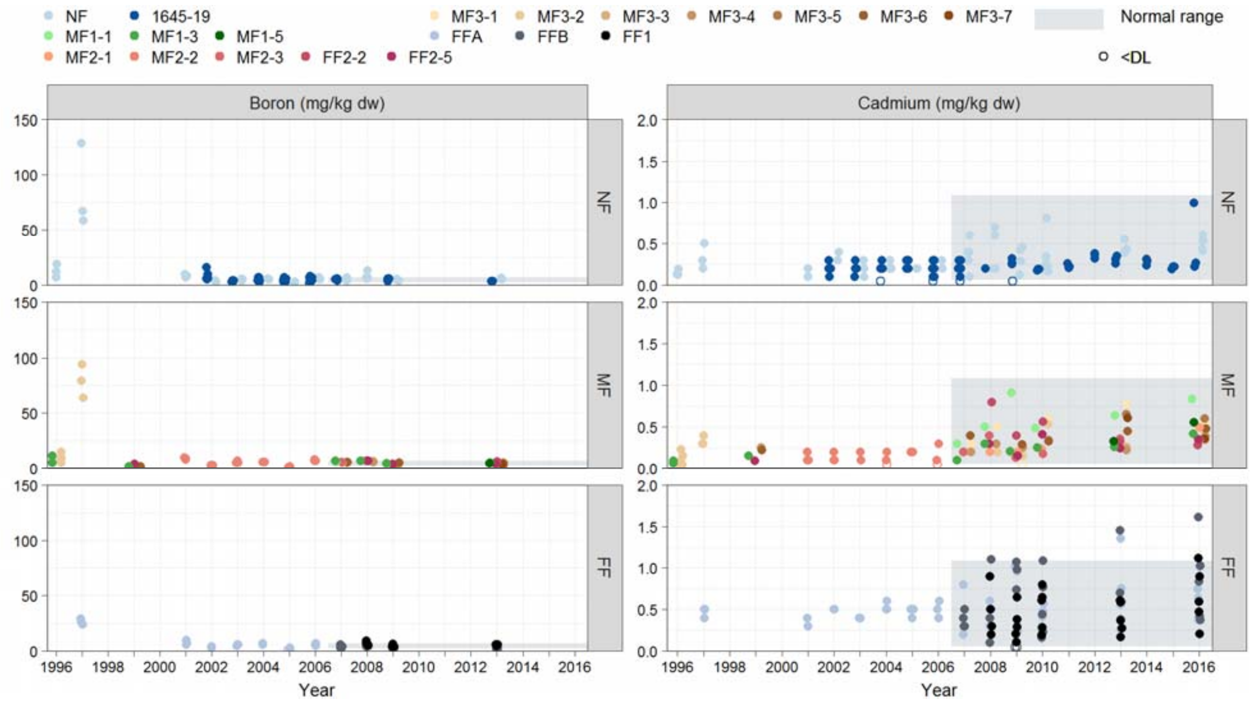
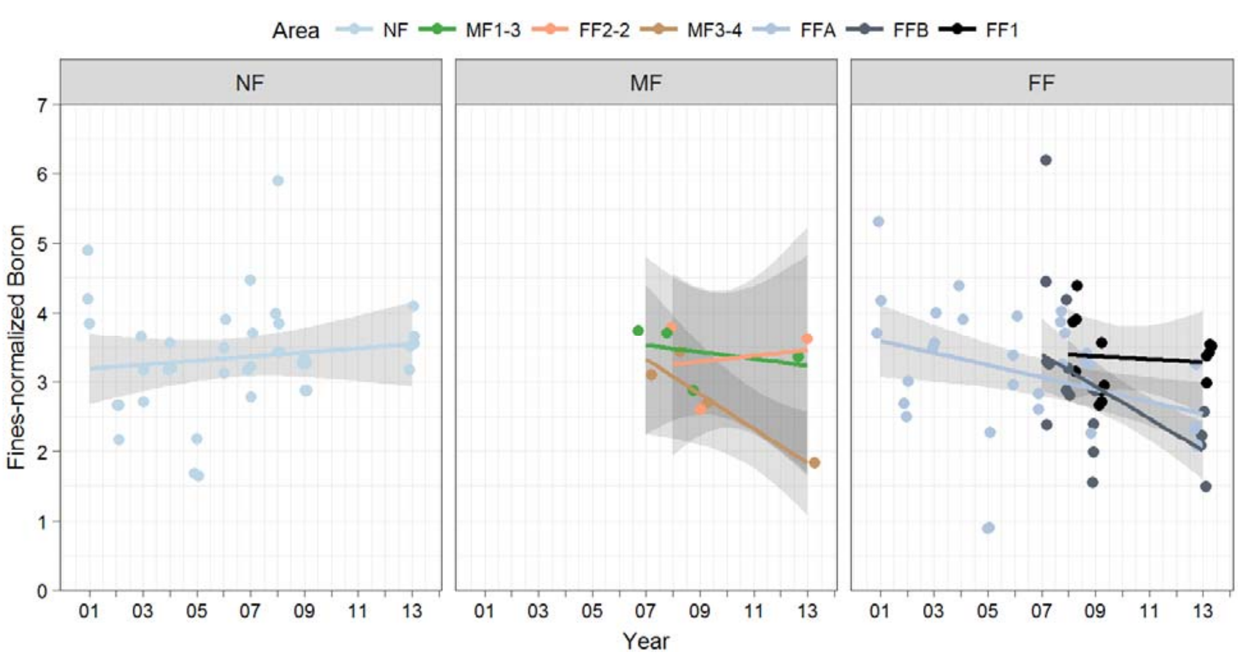
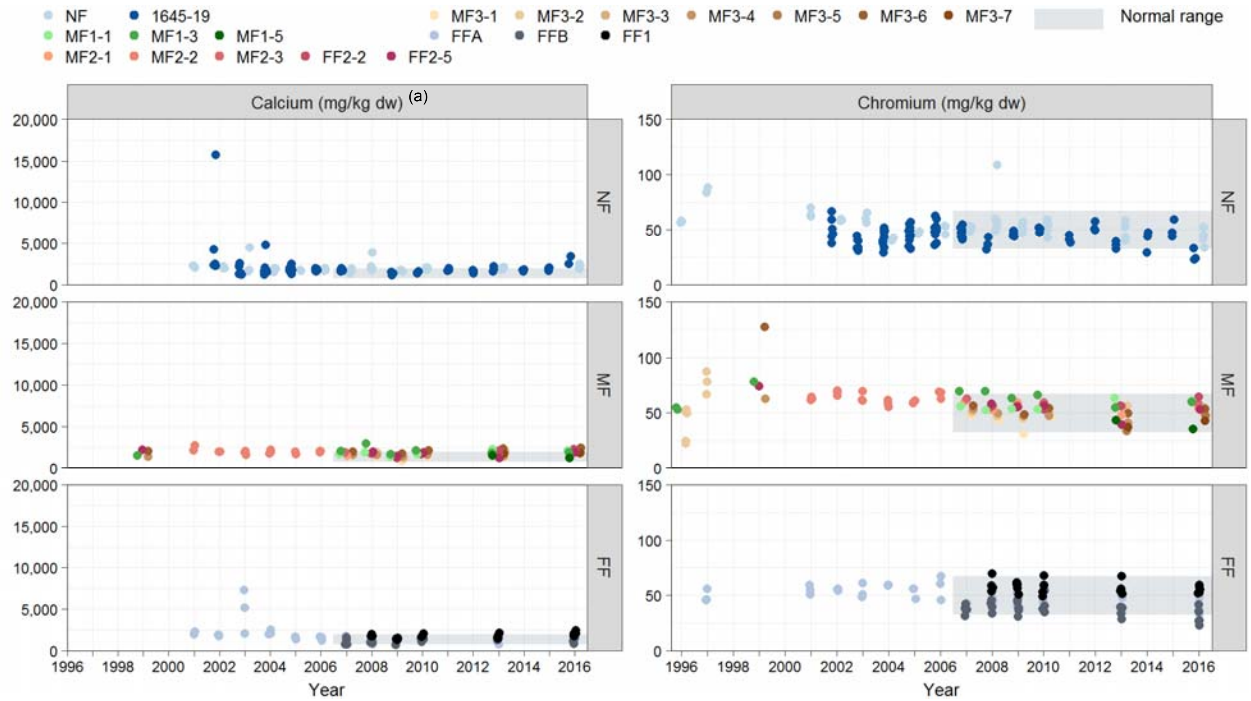


Figure 6-11 Temporal Trend Analysis of Fines-Normalized Boron



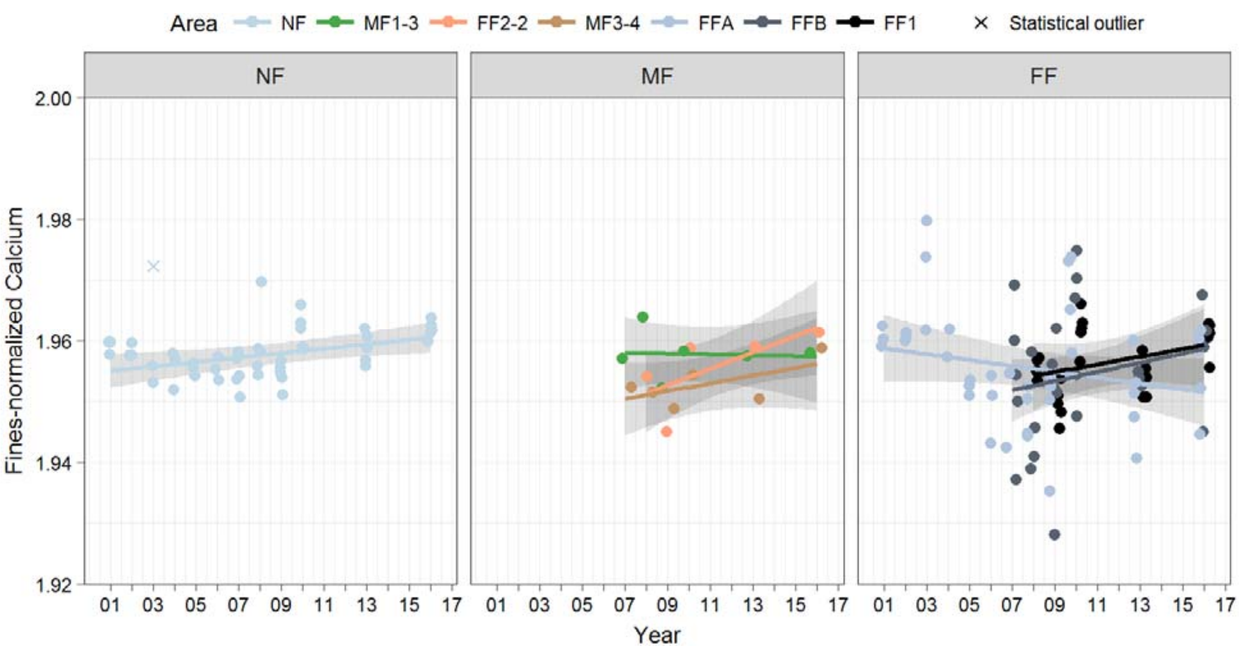
Note: Grey ribbons are 95% confidence intervals.

Figure 6-12 Concentrations of Calcium and Chromium in Sediments, 1996 to 2016



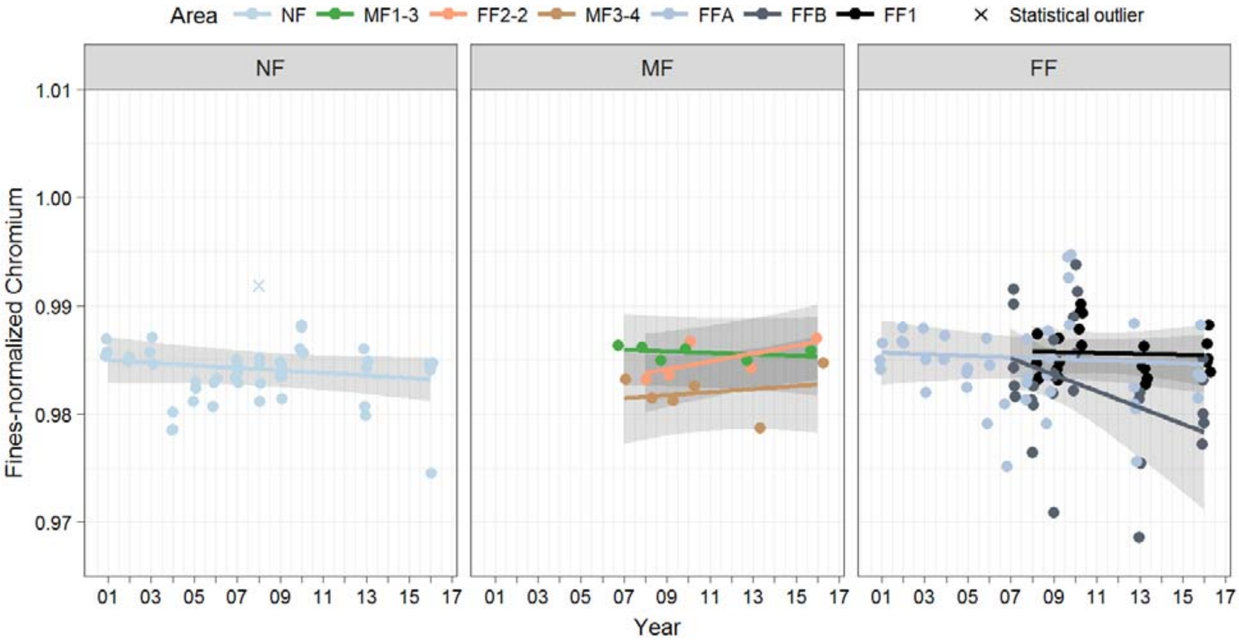
(a) Anomalous value of 76,700 mg/kg omitted from graph for improved visualization.

Figure 6-13 Temporal Trend Analysis of Fines-Normalized Calcium



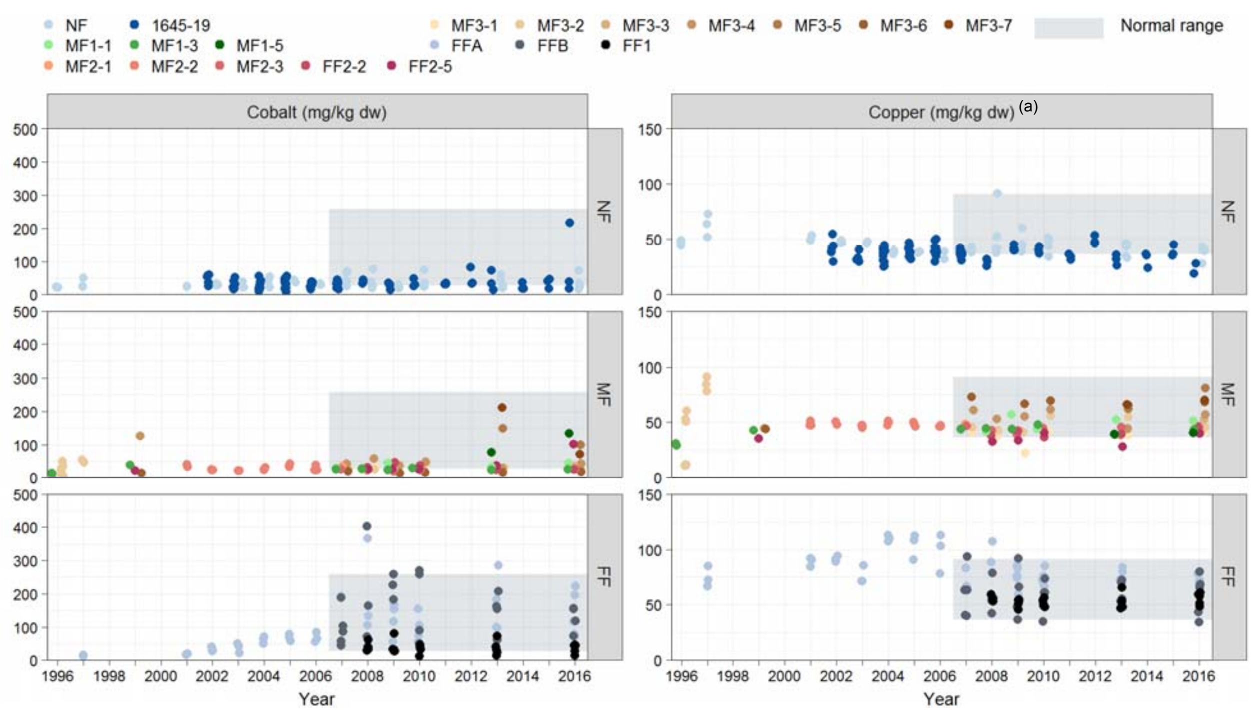
Note: Grey ribbons are 95% confidence intervals.

Figure 6-14 Temporal Trend Analysis of Fines-Normalized Chromium



Note: Grey ribbons are 95% confidence intervals.

Figure 6-15 Concentrations of Cobalt and Copper in Sediments, 1996 to 2016



(a) Anomalous value of 2,300 mg/kg omitted from graph for improved visualization.

Figure 6-16 Concentrations of Iron and Lead in Sediments, 1996 to 2016

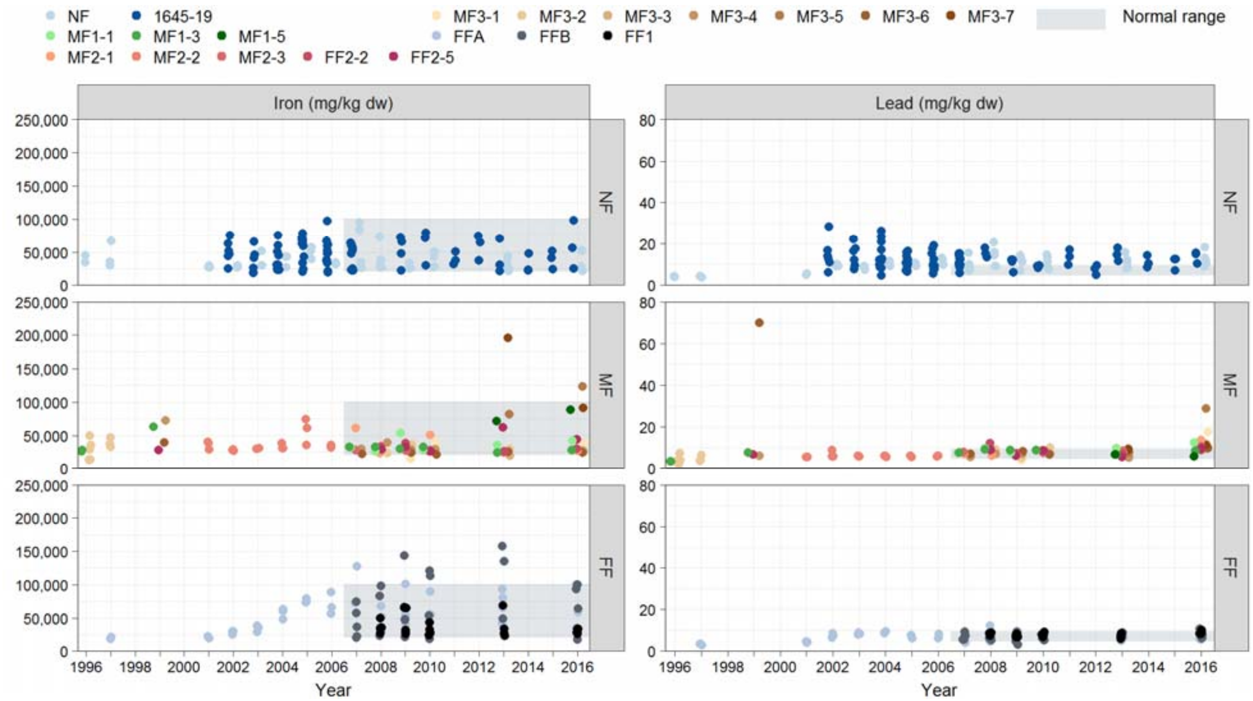
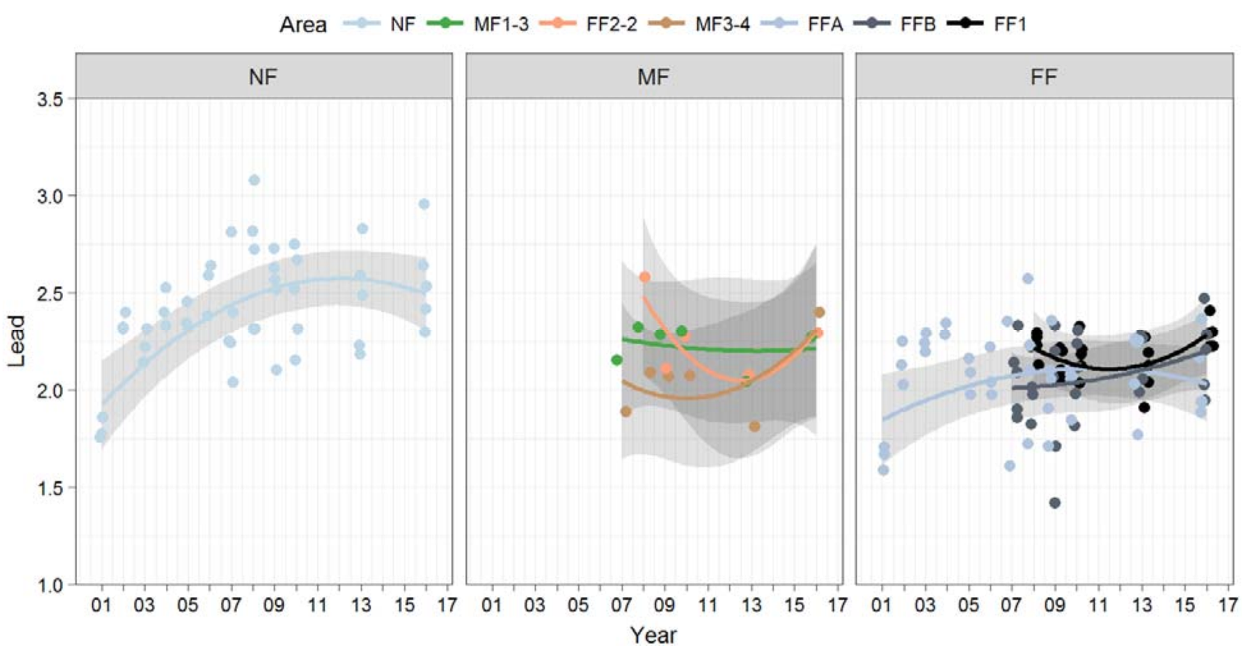


Figure 6-17 Temporal Trend Analysis of Lead



Note: Grey ribbons are 95% confidence intervals.

Figure 6-18 Concentrations of Lithium and Magnesium in Sediments, 1996 to 2016

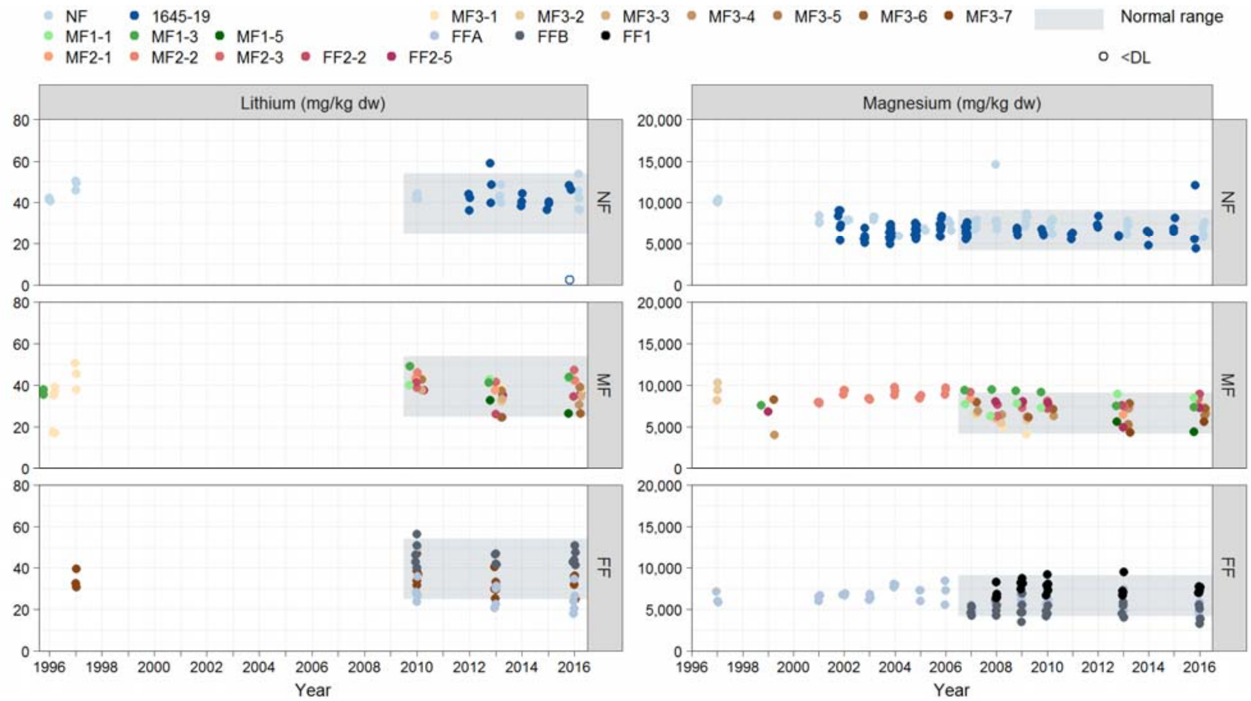
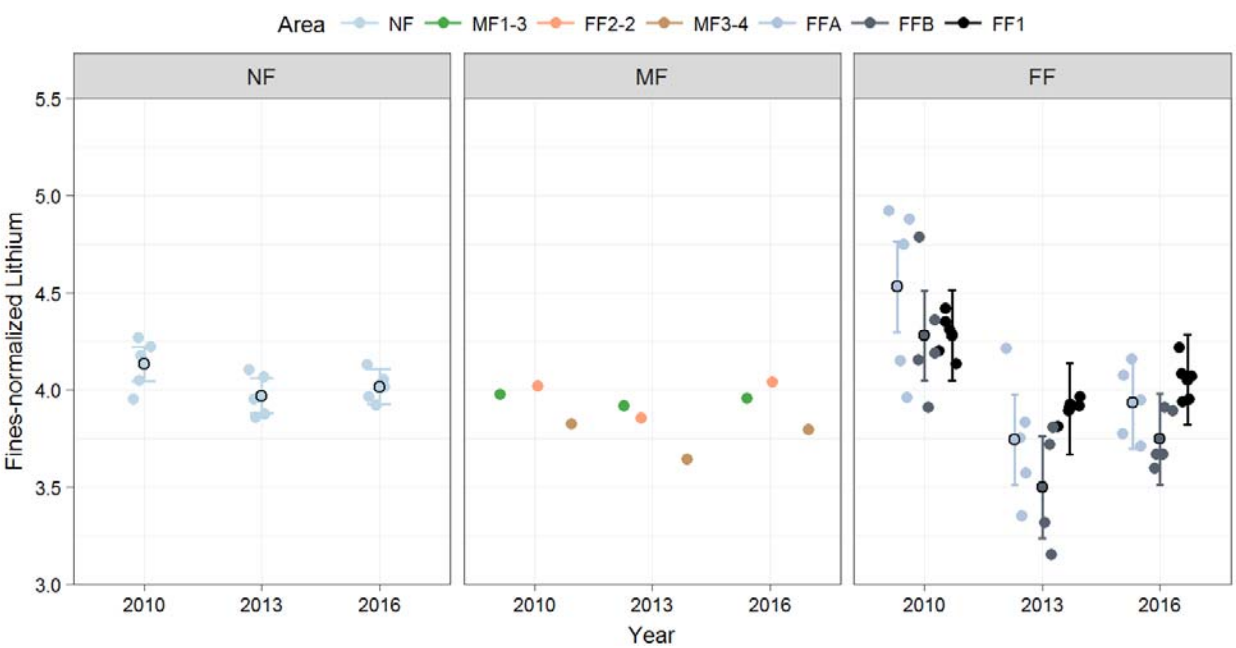
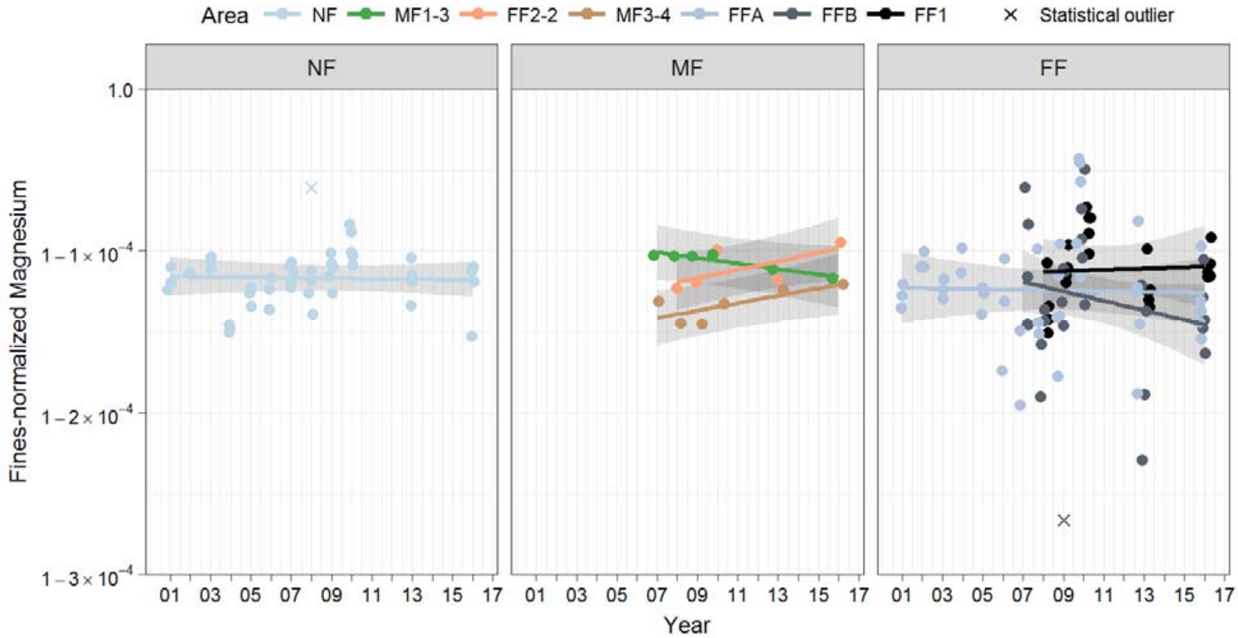


Figure 6-19 Temporal Trend Analysis of Fines-Normalized Lithium



Note: Outlined symbols are means; error-bars are 95% confidence intervals.

Figure 6-20 Temporal Trend analysis of Fines-Normalized Magnesium



Note: Grey ribbons are 95% confidence intervals.

Figure 6-21 Concentrations of Manganese and Mercury in Sediments, 1996 to 2016

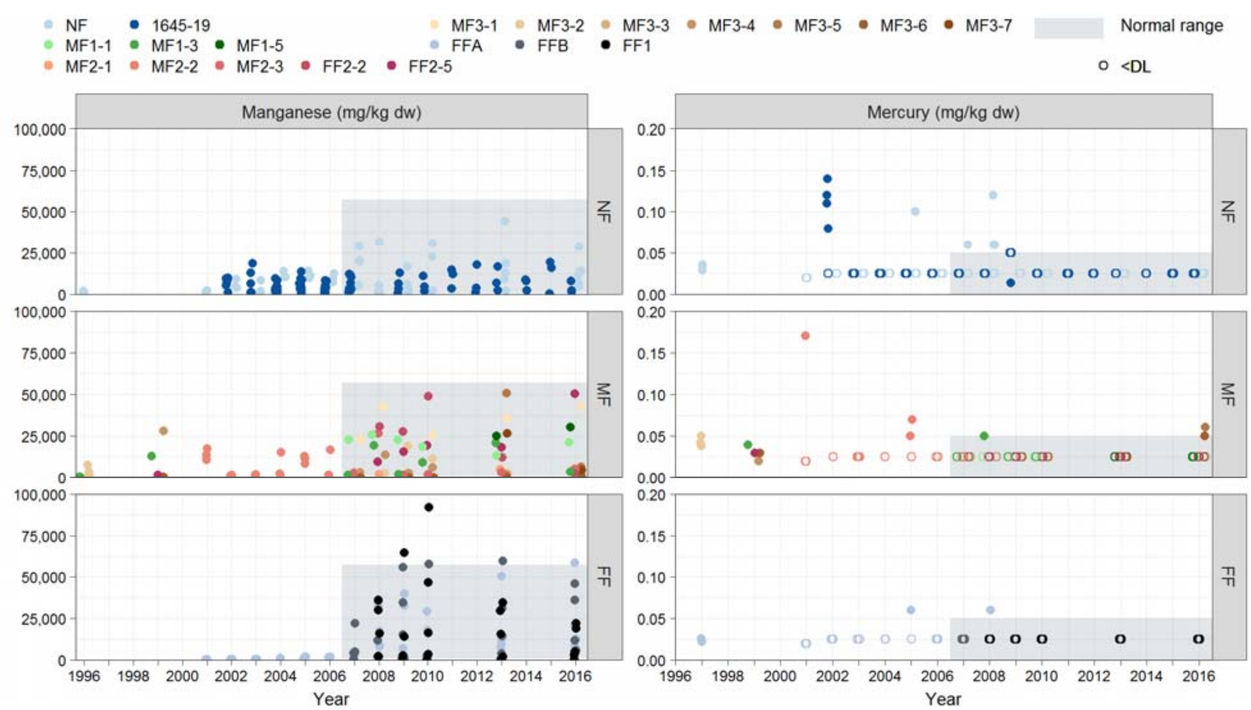


Figure 6-22 Concentrations of Molybdenum and Nickel in Sediments, 1996 to 2016

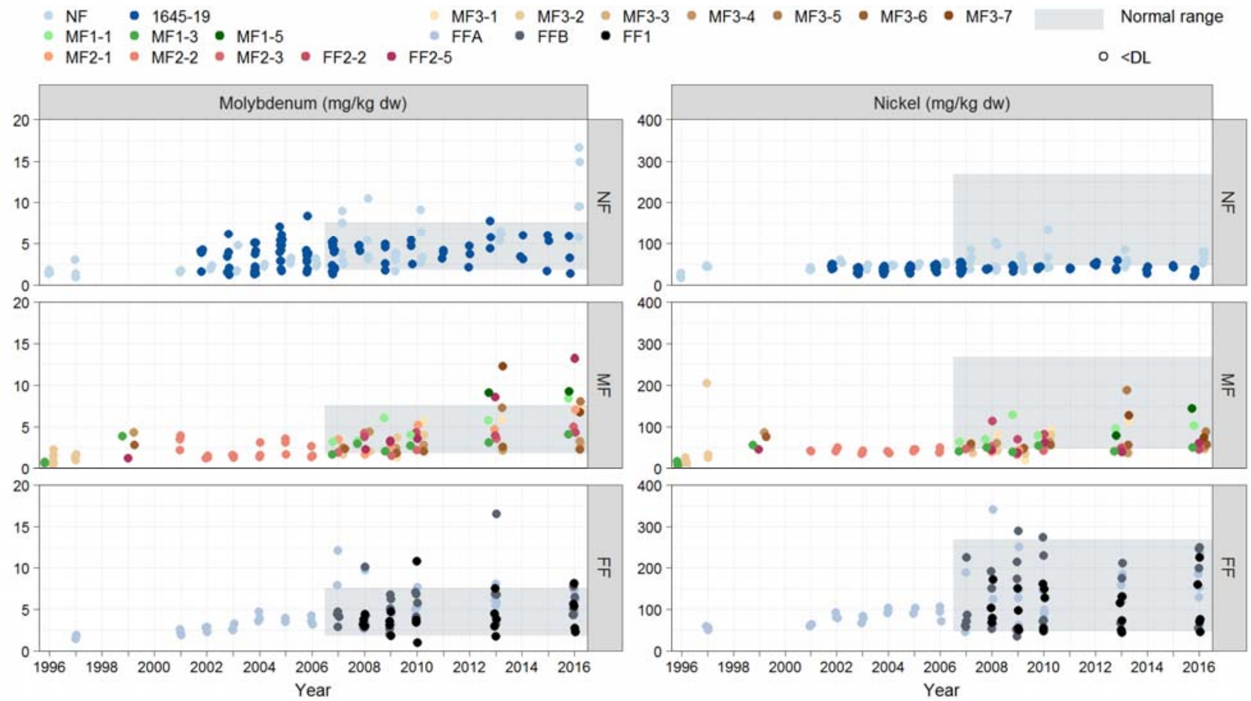
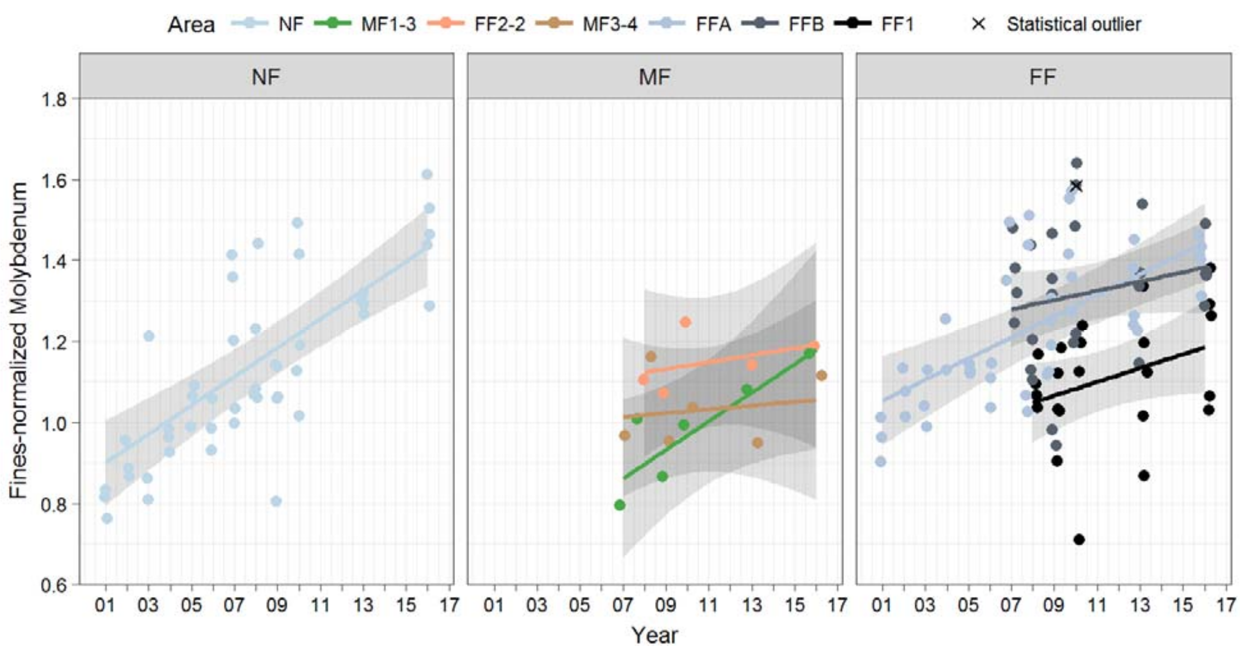


Figure 6-23 Temporal Trend Analysis of Fines-Normalized Molybdenum



Note: Grey ribbons are 95% confidence intervals.

Figure 6-24 Concentrations of Potassium and Selenium in Sediments, 1996 to 2016

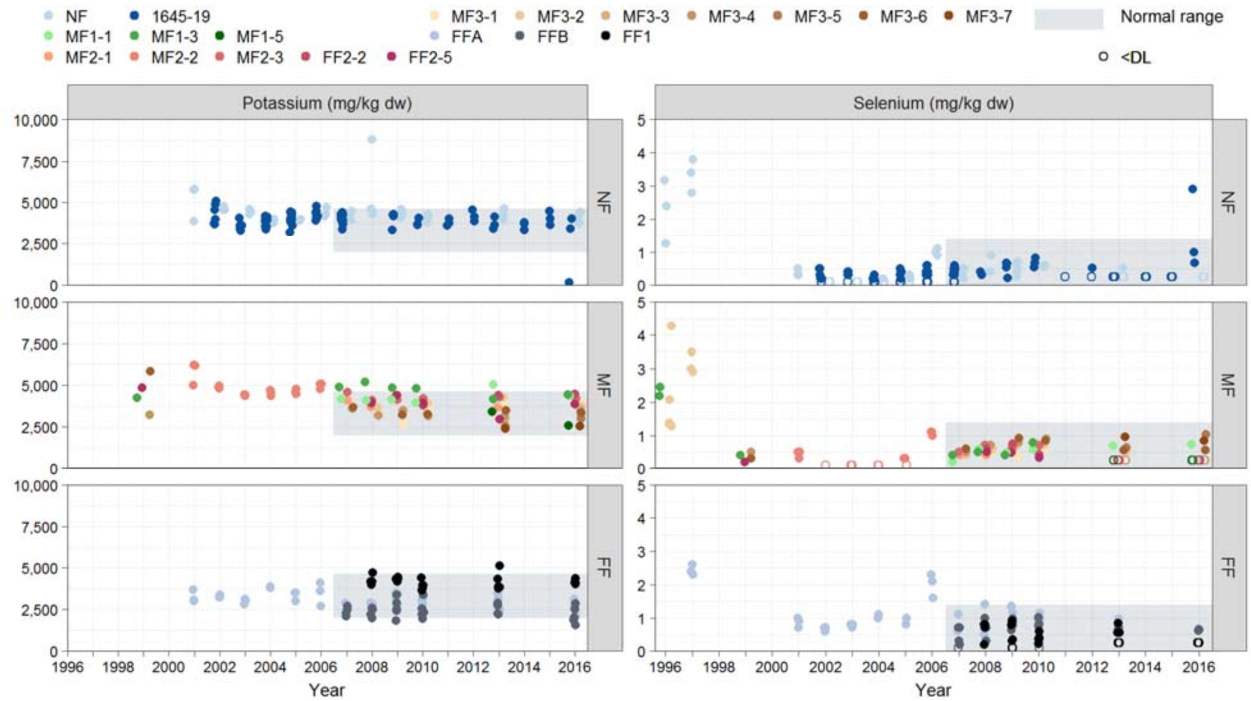
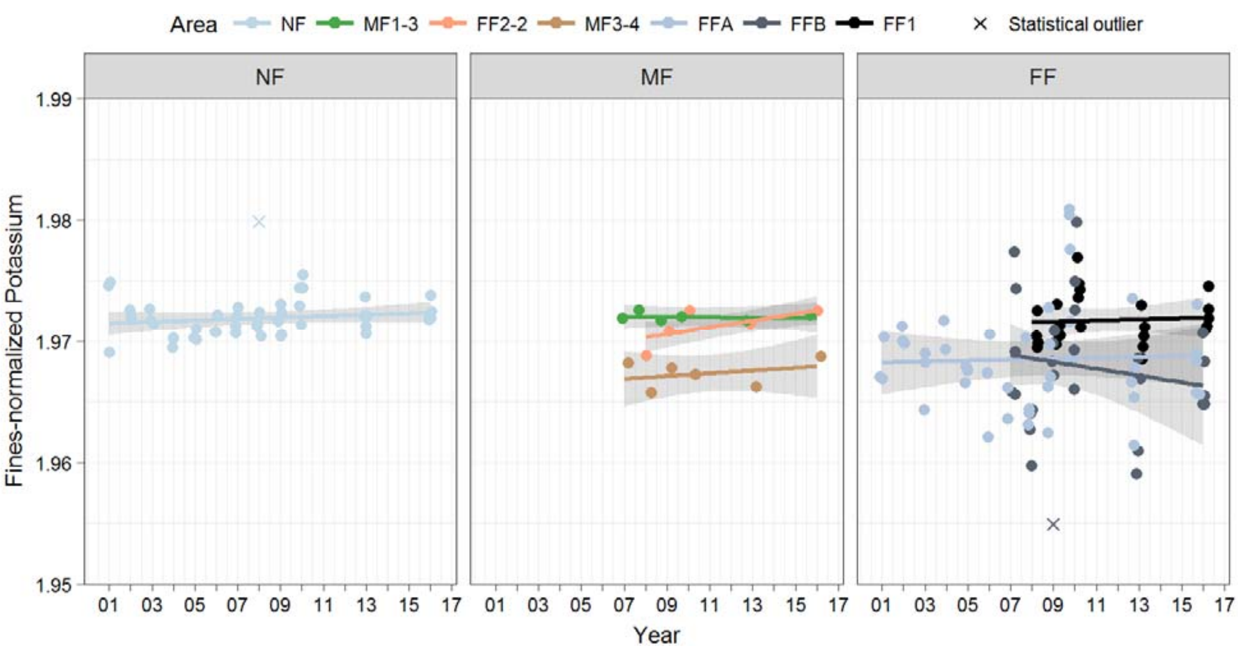


Figure 6-25 Temporal Trend Analysis of Fines-Normalized Potassium



Note: Grey ribbons are 95% confidence intervals.

Figure 6-26 Concentrations of Silver and Sodium in Sediments, 1996 to 2016

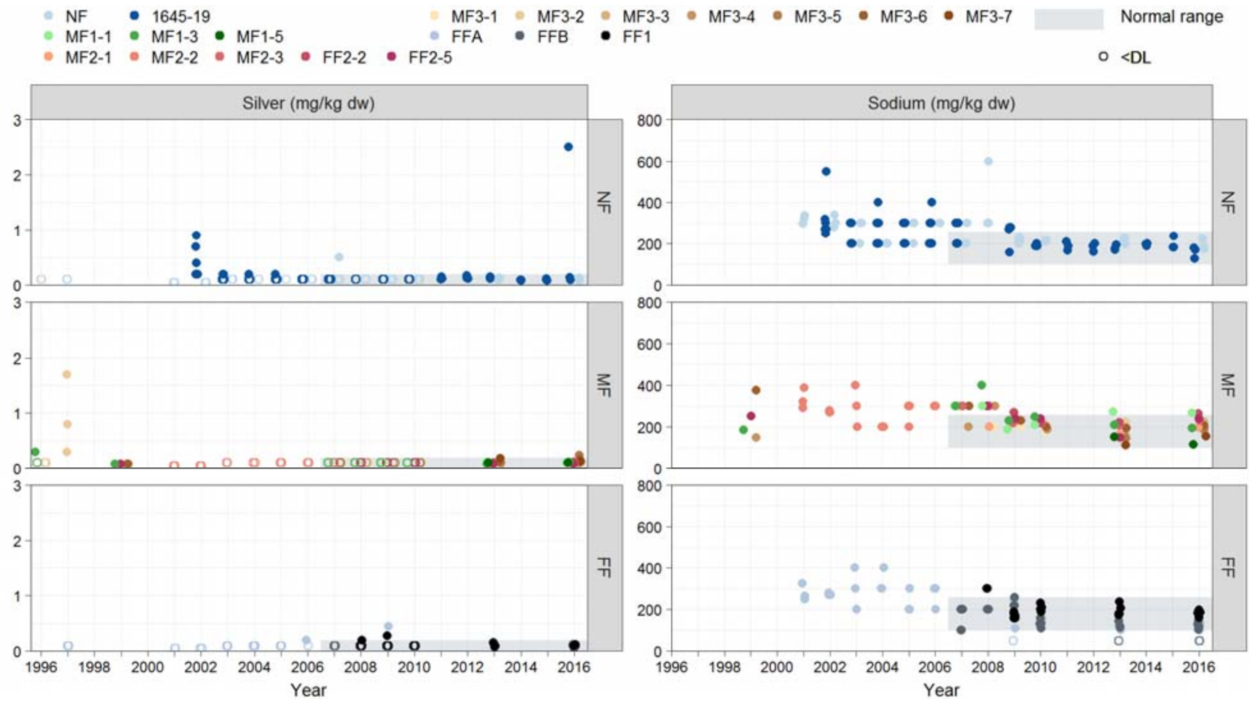
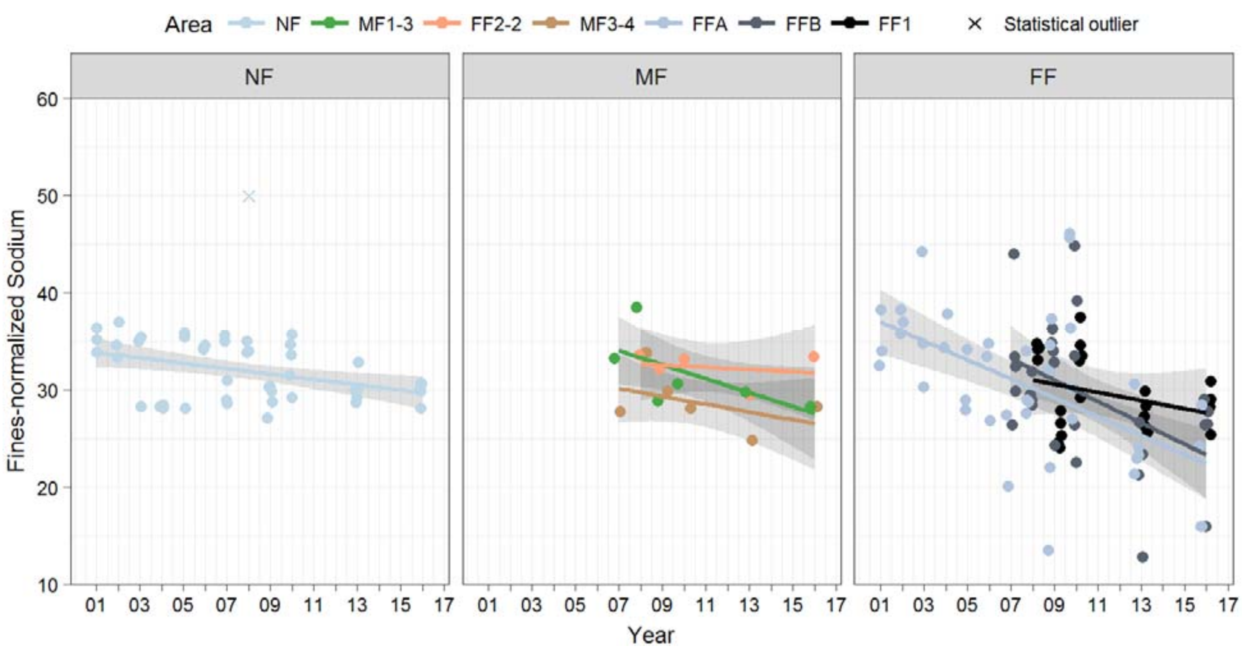
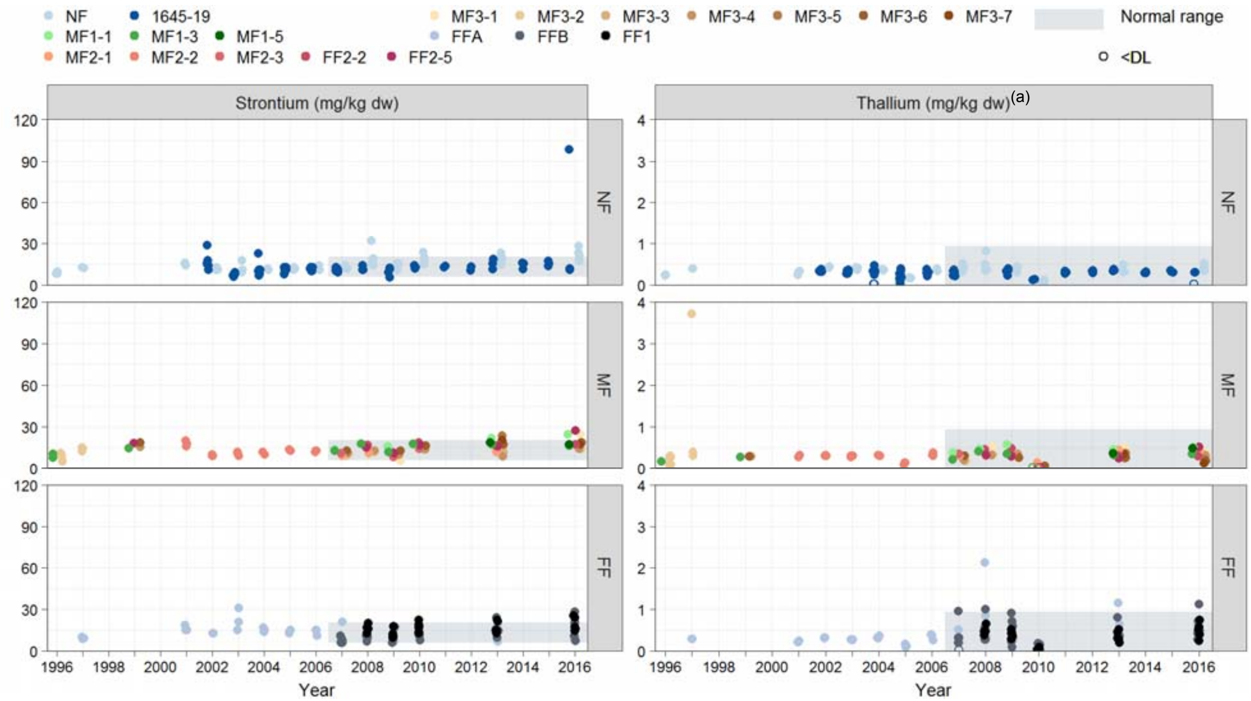


Figure 6-27 Temporal Trend Analysis of Fines-Normalized Sodium



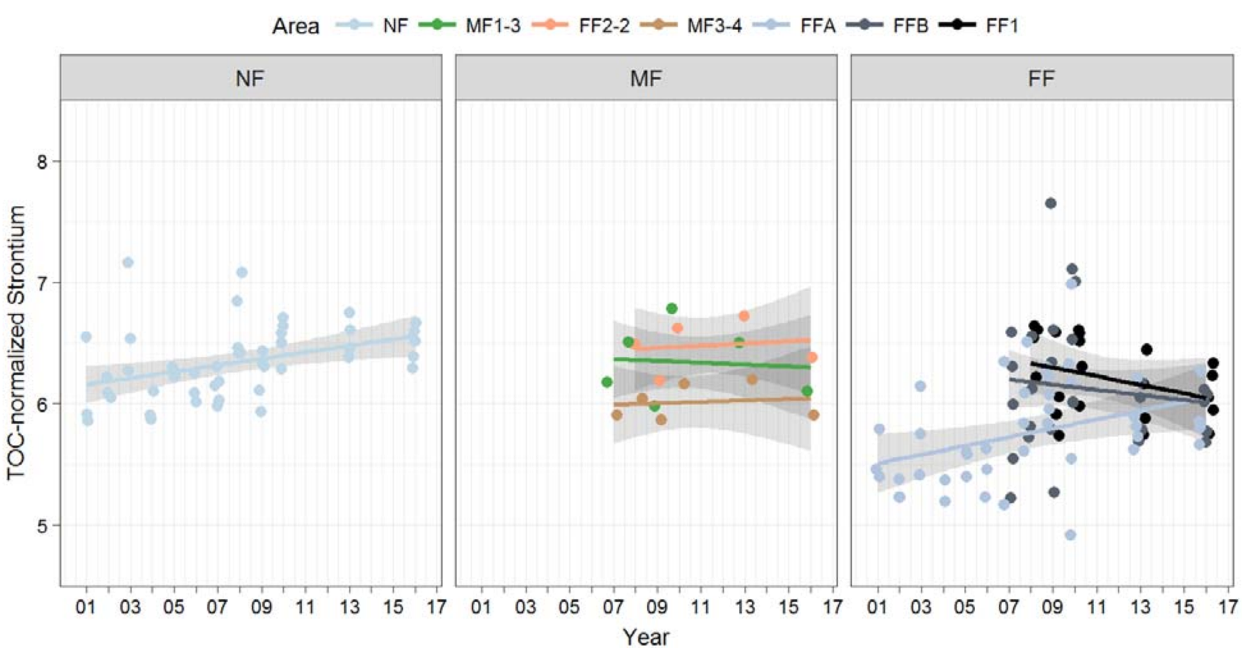
Note: Grey ribbons are 95% confidence intervals.

Figure 6-28 Concentrations of Strontium and Thallium in Sediments, 1996 to 2016



(a) Anomalous values of 1,040 and 19 mg/kg omitted from graph for improved visualization.

Figure 6-29 Temporal Trend Analysis of TOC-Normalized Strontium



Note: Grey ribbons are 95% confidence intervals.

Figure 6-30 Concentrations of Tin and Titanium in Sediments, 1996 to 2016

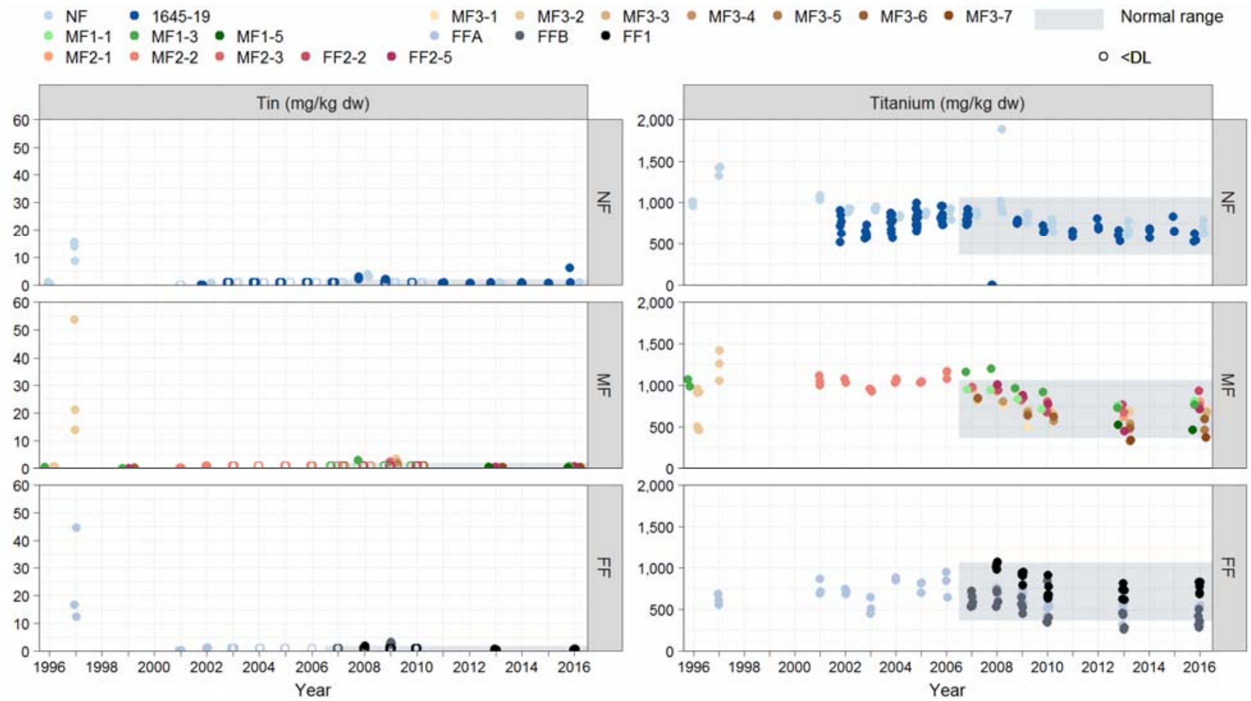
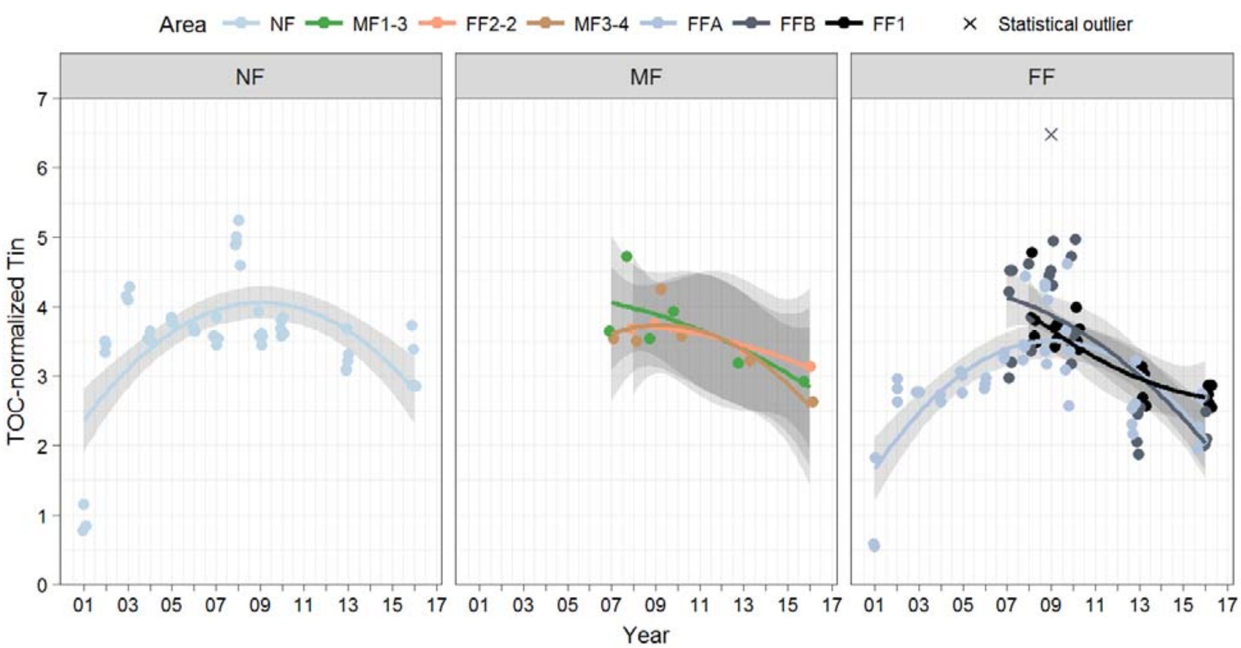
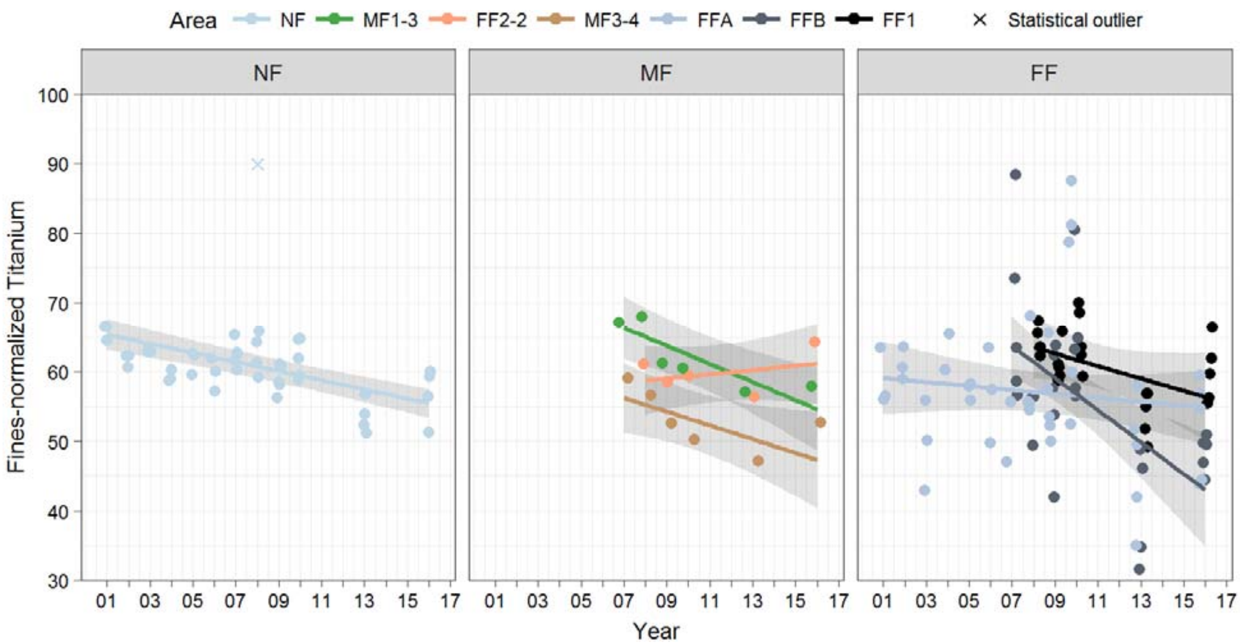


Figure 6-31 Temporal Trend Analysis of TOC-Normalized Tin



Note: Grey ribbons are 95% confidence intervals.

Figure 6-32 Temporal Trend analysis of Fines-Normalized Titanium



Note: Grey ribbons are 95% confidence intervals.

Figure 6-33 Concentrations of Uranium and Vanadium in Sediments, 1996 to 2016

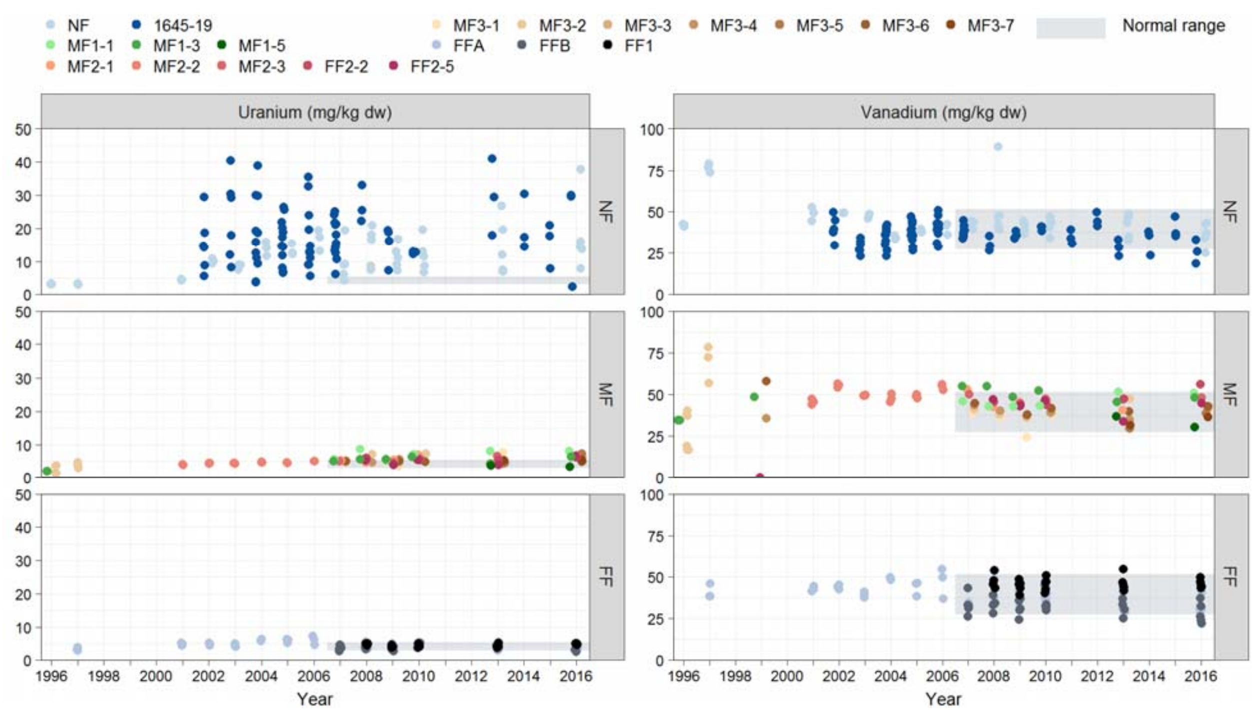
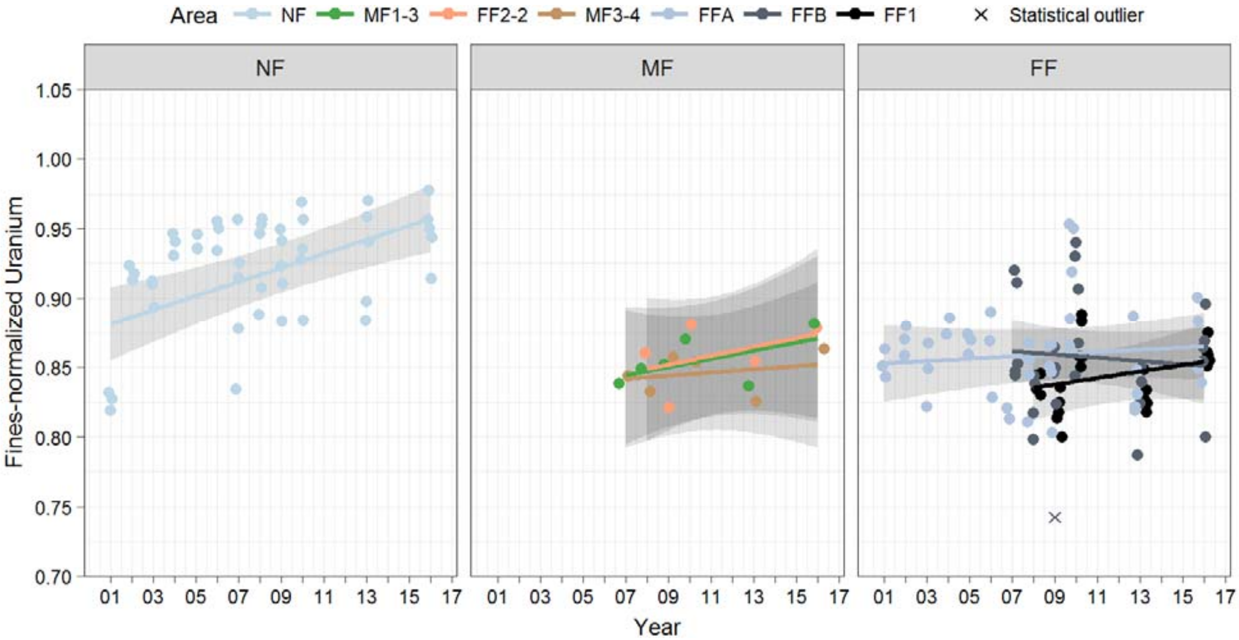
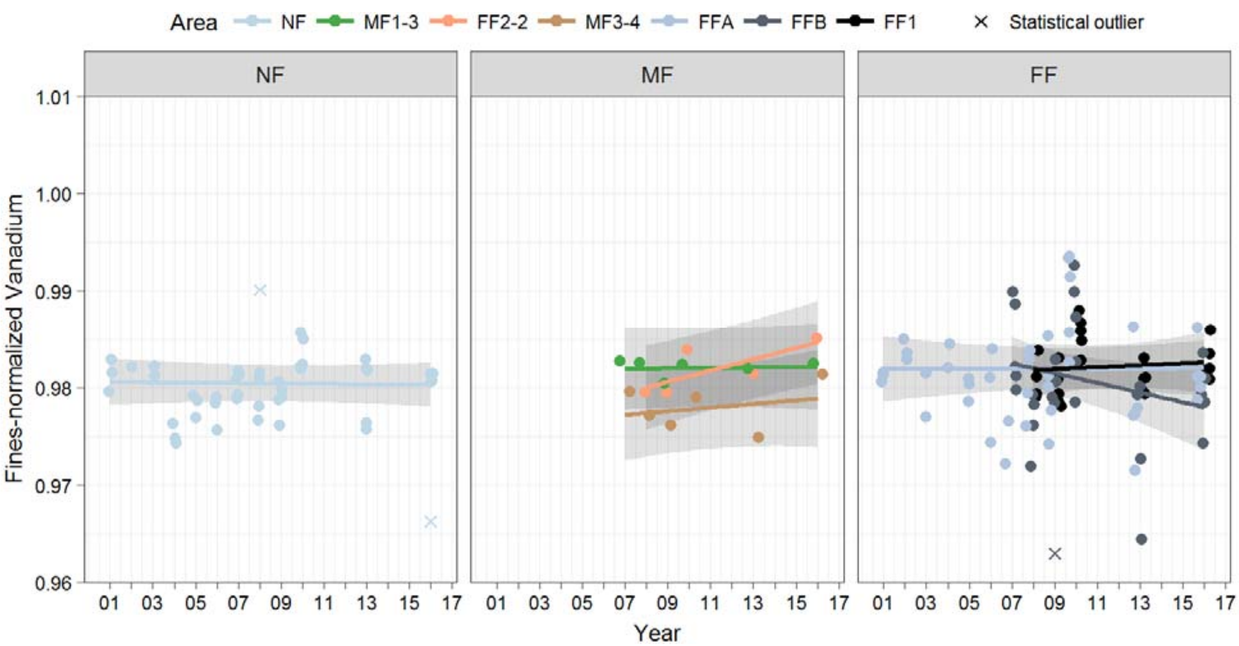


Figure 6-34 Temporal Trend Analysis of Fines-Normalized Uranium



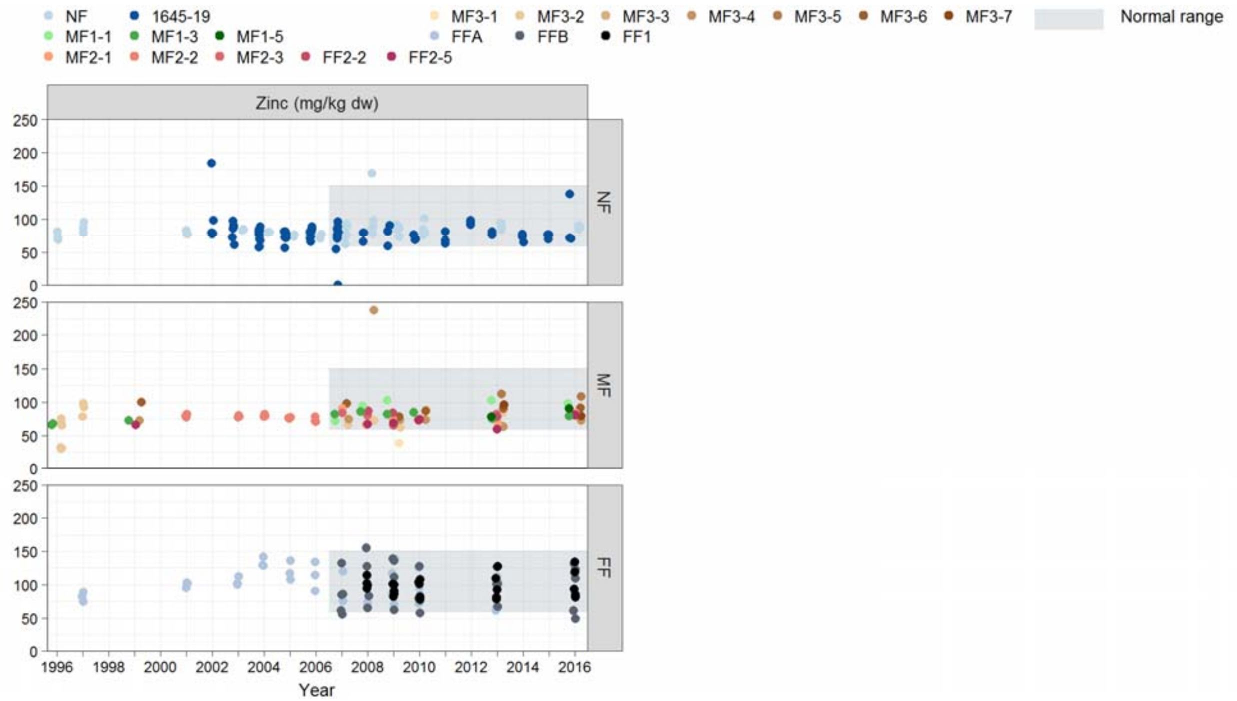
Note: Grey ribbons are 95% confidence intervals.

Figure 6-35 Temporal Trend Analysis of Fines-Normalized Vanadium



Note: Grey ribbons are 95% confidence intervals.

Figure 6-36 Concentrations of Zinc in Sediments, 1996 to 2016



6.4 Comparison to EA Predictions

No predictions were made in the EA regarding sediment quality.

6.5 Summary and Conclusions

Seventeen variables had NF area median concentrations that were statistically greater than median concentrations in FF areas in at least one year of monitoring: TN, aluminum, bismuth, boron, calcium, chromium, lead, lithium, magnesium, molybdenum, potassium, sodium, strontium, tin, titanium, uranium and vanadium. Nine of these variables were observed at greater than FF area concentrations in 2016, and all were retained as SOIs for this re-evaluation. Total phosphorus was added to assess a potential effect of nutrient discharges to Lac de Gras in sediments, as evaluating nutrient-related effects is a key objective of the AEMP.

A summary of the 2014 to 2016 AEMP re-evaluation results is provided below for sediment quality:

- The current approach of collecting single composite samples at each station is considered adequate to detect Mine-related effects on sediment quality variables.
- Physical variables (TOC and percent fine sediments) explained some of the variability in the concentrations of metals and nutrients in sediment that had no clear temporal patterns, and did not interfere with detecting Mine-related effects.

- Concentrations of certain metals, such as arsenic and cadmium, in sediments throughout Lac de Gras were above SQGs. These variables generally reflected patterns in TOC content of bottom sediments or background variation in sediment quality, and had no clear spatial trends related to the Mine.
- Table 6-13 summarizes the trend analysis results for the 18 sediment quality SOIs. Increasing trends in the NF area were found for five SOIs (i.e., calcium, lead [uncertain], molybdenum, strontium and uranium). Similar increasing trends were also detected for three of these variables (lead, molybdenum and strontium) in at least one FF area. Bismuth, boron, TP, sodium, tin, and titanium had decreasing trends at three of more sampling areas or stations. No trends or only potential weak trends were apparent for TN, aluminum and lithium.
- Three SOIs (i.e., bismuth, lead, and uranium) had NF area median concentrations that were greater than their respective normal ranges. The NF area median concentrations of the remaining SOIs were within their respective normal ranges. Results of the most recent dike monitoring study reported similar elevations of bismuth, lead and uranium in the vicinity of the A154 and 418 dikes. Sediment results indicated that effluent discharge is likely the primary source of these metals in the NF area, although other factors, such as dike construction, may also contribute to the observed patterns.
- The toxicological risks associated with elevated bismuth concentrations in the NF area sediments are unknown, because no guidelines exist and no sediment toxicity data were available in the primary literature; however, given the low aqueous concentrations, stable results since 2006, and expected low aquatic toxicity of bismuth, bismuth is unlikely a constituent of concern in sediment. Lead and uranium concentrations are unlikely to pose a toxicological risk to biota based on comparisons to SQGs and information from the primary literature.
- Of the 18 SOI variables, chromium and lead are the only SOI with applicable SQGs; lead concentrations in Lac de Gras did not exceed either of the CCME or OMOEE guidelines, and while chromium exceeded the ISQG, it did not meet the WOE criteria for moderate effect level. Chromium guideline exceedances were observed in both the NF and FF areas; therefore, existing concentrations likely reflect elevated background levels for Lac de Gras.

Table 6-13 Summary of Trends in Concentrations of Sediment Quality SOI Variables in Lac de Gras, 2014 to 2016

Substance of Interest	Area / Station						
	NF	MF1-3	FF2-2	MF3-4	FF1	FFB	FFA
Total Nitrogen	—	↓ —	—	—	—	—	—
Total Phosphorus	↓	↓	↓	↓	↓	↓	↓
Aluminum	—	—	—	—	—	—	—
Bismuth	—	↓ —	↓ —	↓ —	↓	↓	—
Boron	—	—	—	↓	—	↓	↓
Calcium	↑	—	↑	—	—	—	—
Chromium	—	—	↑	—	—	—	—
Lead	↑ —	—	—	↑ —	↑ —	↑ —	—
Lithium	—	—	—	—	—	—	—
Magnesium	—	—	↑	↑	—	—	—

Table 6-13 Summary of Trends in Concentrations of Sediment Quality SOI Variables in Lac de Gras, 2014 to 2016

Substance of Interest	Area / Station						
	NF	MF1-3	FF2-2	MF3-4	FF1	FFB	FFA
Molybdenum	↑	↑	—	—	—	—	↑
Potassium	—	—	↑	—	—	—	—
Sodium	↓	—	—	—	—	↓	↓
Strontium	↑	—	—	—	—	—	↑
Tin	↓	↓	↓	↓	↓	↓	↓
Titanium	↓	↓	—	↓	—	↓	—
Uranium	↑	—	—	—	—	—	—
Vanadium	—	—	↑	—	—	—	—

Notes: Linear trends with slopes that are significantly different from a slope of zero are presented by arrows. Parabolic trends have been summarized based on visual inspections of the trend plots for recent years, particularly those past the vertex of the parabola, where applicable. ↑ = increasing trend; ↓ = decreasing trend; — = no trend or no consistent trend; ↑ — or ↓ — = potential slightly increasing or slightly decreasing trend.

7 PLANKTON

7.1 Introduction

The term “plankton” refers to small, usually microscopic organisms that live suspended in lakes and ponds. For the purpose of this study, the term “phytoplankton” refers to the algal component of plankton and includes the following five major ecological groupings:

- cyanobacteria
- chlorophytes (Chlorophyceae, Prasinophyceae, Euglenophyceae, Trebouxiophyceae, Pedinophyceae Nephroselmidophyceae, Conjugatophyceae, and Klebsormidiophyceae)
- microflagellates (Chrysophyceae, Cryptophyceae, Coccolithophyceae, and Haptophyceae)
- dinoflagellates (Dinophyceae)
- diatoms (Bacillariophyceae
- others (Xanthophyceae, Euglenophyceae, and other unidentified algae).

The term “zooplankton” refers to small animals, ranging from microscopic to visible with the naked eye, and includes crustaceans (i.e., Cladocera [cladocerans], Cyclopoida [cyclopoids], Calanoida [calanoids]) and Rotifera (rotifers).

The plankton section provides a summary of changes in the plankton community of Lac de Gras over time. The objectives of this section are:

- to summarize Mine-related effects observed from 2014 to 2016 and compare these to effects observed previously (i.e., from 2007 to 2013); and
- to analyze temporal trends in phytoplankton and zooplankton for the period extending from baseline (i.e., 1996) to 2016.

7.1.1 Background

DDMI has been conducting studies and monitoring programs relating to the aquatic ecosystem of Lac de Gras since 1994, with AEMP data collected under AEMP study designs *Version 1.0* (2001[8] to 2006), *Version 2.0* (2007 to 2011), *Version 3.0* (2012 to 2016), and most recently under *Version 4.0* (2017). Since there is potential for Mine effluent to affect the aquatic ecosystem in Lac de Gras, the Plankton program has been included as a component of the AEMP since 2002, initially as a special effects study, and is monitored every year, (i.e., during both comprehensive and interim years). The objective of the Plankton program is to determine if effluent from the Mine is having an effect on the plankton community of Lac de Gras, and to confirm the predictions set forth in the EA (DDMI 1998). More specifically, the program has been designed and implemented to assess Mine-related changes in the plankton community of Lac de Gras.

7.1.2 Component History

Baseline plankton sampling in Lac de Gras began in 1995, by estimating abundance of the plankton community (Acres and Bryant 1996) and continued in 1997, when both abundance and biomass were estimated (Golder 1998). Plankton community sampling began in 2002, as part of a special effects study (SES) under the *AEMP Study Design Version 1.0*, during the start of effluent discharge into Lac de Gras. The plankton community data collected from 2003 to 2006 during the *AEMP Study Design Version 1.0* was archived and not analyzed until 2008. The plankton SES continued over the course of the *AEMP Study Design Version 2.0* (DDMI 2007). The main objective of the SES was to determine the feasibility and utility of using plankton community composition and biomass as indicators of biological effects of the Mine. A secondary objective was to determine if a single open-water sampling event could be used to collect data that were adequate to describe community metrics and detect Mine-related effects.

A review of the four years of data collected during the AEMP Version 2.0 demonstrated that plankton could indeed be a useful and sensitive monitoring component (Golder 2011a). It also indicated that, based on the seasonal variation observed during the plankton SES, any open-water period would be equally appropriate for plankton monitoring. Accordingly, plankton was added as a regular component of the AEMP in 2011 under the *AEMP Study Design Version 3.0* (Golder 2011b). Under the *AEMP Study Design Version 3.5*, which applies to this re-evaluation, plankton monitoring occurred during a single open-water monitoring season (from 15 August to 15 September) in conjunction with the other AEMP components (Golder 2014a). The sampling frequency for plankton in the far-field areas (FF1, FFA and FFB) is once every three years,

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[8] One year of baseline data was also collected in 2000 under Version 1.

to be consistent with the other AEMP components sampled during comprehensive monitoring years. Although samples were collected monthly during the open-water season during the *AEMP Study Design Version 1.0* (2002 to 2006) and *2.0* (2007 to 2010), only data for samples collected from 15 August to 15 September were included in the analysis for this report, to limit the effect of seasonal variation on the re-evaluation results.

7.2 Methods

Plankton sampling was completed at stations sampled by other AEMP components, and were selected based on exposure to the Mine effluent (Golder 2014a). Sampling areas consisted of a NF area and three FF areas (i.e., FF1, FFA, and FFB). In addition, three transect lines (referred to as MF areas) between the NF and FF areas were sampled. The NF-MF1-FF1 transect was sampled towards the FF1 area, northwest of the Mine site. The NF-MF2-FF2 transect was sampled to the northeast, towards the FF2 area near the Lac du Sauvage inlet. The NF-MF3-FFB-FFA transect was sampled south of the Mine site towards FFB and FFA far-field areas and the outlet of Lac de Gras. Within each sampling area, clusters of replicate stations were sampled. Five stations were sampled in the NF area and in each of the three FF areas. To better delineate the extent of effects and define gradients along each transect, the number of stations in the MF3 area was increased from AEMP Version 2.0 to Version 3.0, and the number of stations in the MF1 and MF2 areas was reduced.

7.2.1 Data Sources

7.2.1.1 Phytoplankton

Sampling locations for phytoplankton biomass from baseline to *AEMP Study Design Version 3.5* are presented in Table 7-1. Sampling locations from baseline, *AEMP Study Design Versions 1.0 and 2.0* were paired with current AEMP sampling areas. Historical stations not included in the current *AEMP Study Design Version 3.5* (i.e., discontinued stations) have been shaded in grey and were not included in trend plots or data analyses.

Table 7-1 Summary of Baseline and AEMP Phytoplantkon Biomass Data, 1997 to 2016

Waterbody	Area	Station[a]	UTM Coordinates[b]		Distance from Diffusers[c] (km)												
			Easting (m)	Northing (m)		1997[d]	2002 -2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Lac de Gras[f]	Near-field	NF1 (WQ-06; LDG-42)	535740	7153854	0.4		X	X	X	X	X	X	X	X	X	X	X
		NF2 (WQ-06; LDG-42)	536095	7153784	0.5		X	X	X	X	X	X	X	X	X	X	X
		NF3 (WQ-06; LDG-42)	536369	7154092	0.9		X	X	X	X	X	X	X	X	X	X	X
		NF4 (WQ-06; LDG-42)	536512	7154240	1.1		X	X	X	X	X	X	X	X	X	X	X
		NF5 (WQ-06; LDG-42)	536600	7153864	1.0		X	X	X	X	X	X	X	X	X	X	X
	Mid-field 1	MF1-1	535008	7154699	1.5			X	X	X	X	X	X	X	X	X	X
		MF1-2	533682	7155356	2.9			X	X	X	X	X	X	X	X	X	X
		MF1-3 (LDG-40)	532236	7156276	4.7		X	X	X	X	X	X	X	X	X	X	X
		MF1-4	532494	7157657	7.2			X	X	X	X	X					
		MF1-5	528432	7157066	8.5								X	X	X	X	X
	Mid-field 2	MF2-1	538033	7154371	2.4			X	X	X	X	X	X	X	X	X	X
		MF2-2	539198	7154643	3.7			X	X	X	X	X					
		MF2-3	540365	7156045	5.4			X	X	X	X	X	X	X	X	X	X
		MF2-4	540955	7157359	6.9			X	X	X	X	X					
	Far-field 2[e]	FF2-1	541500	7159522	9.3			X	X	X	X	X					
		FF2-2 (LDG-45)	541588	7158561	8.3		X	X	X	X	X	X	X	X	X	X	X
		FF2-3	543478	7159267	10.1			X	X	X	X	X					
		FF2-4	543752	7158945	10.2			X	X	X	X	X					
		FF2-5	544724	7158879	11.4			X	X	X	X	X	X	X	X	X	X
	Mid-field 3	MF3-1	537645	7152432	2.7				X	X	X	X	X	X	X	X	X
		MF3-2 (LDG-43)	536816	7151126	4.2		X	X	X	X	X	X	X	X	X	X	X
		MF3-3	536094	7148215	7.2								X	X	X	X	X
		MF3-4 (WQ-05; LDG-41)	532545	7147011	11.0	X	X	X	X	X	X	X	X	X	X	X	X
		MF3-5	528956	7146972	14.6								X	X	X	X	X
		MF3-6	525427	7148765	18.5			X	X	X	X	X	X	X	X	X	X
		MF3-7	521859	7150039	22.3								X	X	X	X	X
	Far-field 1	FF1-1	525430	7161043	13.6				X	X	X	X	X	X			X
		FF1-2	524932	7159476	12.9				X	X	X	X	X	X			X
		FF1-3	526407	7160492	12.8				X	X	X	X	X	X			X
		FF1-4	526493	7159058	11.4				X	X	X	X	X	X			X
		FF1-5	526683	7161824	12.8			X	X	X	X	X	X	X			X
	Far-field B	FFB-1	516831	7148207	26.4			X	X	X	X	X	X	X			X
		FFB-2	518473	7150712	25.0			X	X	X	X	X	X	X			X
		FFB-3 (LDG-50)	518048	7147557	25.2		X	X	X	X	X	X	X	X			X
		FFB-4	515687	7150036	27.6			X	X	X	X	X	X	X			X
		FFB-5	516533	7150032	26.8			X	X	X	X	X	X	X			X
	Far-field A	FFA-1	506453	7154021	36.8		X	X	X	X	X	X	X	X			X
		FFA-2	506315	7155271	38.3		X	X	X	X	X	X	X	X			X
		FFA-3	505207	7153887	38.7		X	X	X	X	X	X	X	X			X
		FFA-4	503703	7154081	40.2		X	X	X	X	X	X	X	X			X
		FFA-5 (WQ-14; LDG-46)	505216	7156657	40.0	X	X	X	X	X	X	X	X	X			X

Table 7-1 Summary of Baseline and AEMP Phytoplantkon Biomass Data, 1997 to 2016

Waterbody	Area	Station[a]	UTM Coordinates[b]		Distance from Diffusers[c] (km)												
			Easting (m)	Northing (m)		1997[d]	2002 -2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Outlet of Lac de Gras	n/a	LDG-48	490900	7161750	55.6												
Lac du Sauvage	n/a	LDS-1	546398	7161179	-							X		X			X
		LDS-2	546807	7160027	-							X		X			X
		LDS-3	547191	7160256	-							X		X			X

Note: Shading identifies stations that were discontinued, as per the AEMP Study Design, Version 3.0 (Golder 2011b). These stations are not included in the current (2014 to 2016) Aquatic Effects Re-evaluation Report.
a) Current AEMP station names are presented, and historical sampling station names are provided in parentheses.
b) UTM coordinates are reported as Zone 12, North American Datum (NAD) 83.
c) Approximate distance from the Mine effluent diffusers along the most direct path of effluent flow.
d) For 1997, information is provided only for stations located within AEMP areas.
e) From 2012 onwards, the FF2 area is considered to be part of the MF2-FF2 area.
f) SNP data are not included.
UTM = Universal Transverse Mercator coordinate system; - = not applicable.

7.2.1.1.1 Baseline and AEMP Version 1.0 Data (1995 to 2006)

Baseline phytoplankton community data were collected in 1995 (Acres and Bryant 1996) and 1997 (Golder 1998). Phytoplankton samples were collected from the top 10 m of the water column. Only abundance estimates were provided in 1995 and 1997; however, biomass estimates were also provided in 1997. Taxonomy was performed at a high taxonomic level (i.e., major group), the results of which are not directly comparable to the more recently collected data.

The phytoplankton community data from the *AEMP Study Design Version 1.0* (2002 to 2006) and from the first year of the *AEMP Study Design Version 2.0* (2007) were first compiled in the *2008 Plankton SES Report* (DDMI 2008). The historical data were obtained from three sources:

- archived phytoplankton samples from 2003 to 2006 AEMP surveys, which were analyzed in 2008 (150 samples);
- phytoplankton samples collected and analyzed as part of the 2002 AEMP program (15 samples); and,
- phytoplankton samples collected and analyzed as part of the 2007 AEMP program (45 samples).

Sampling locations from baseline and from 2003 to 2006 were paired with current AEMP sampling areas (Table 7-1). The 2002 to 2006 samples were collected from the top 10 m of the water column. If the water depth was less than 10 m, 80% of the water column was sampled. Taxonomic analyses of the 2002 to 2006 samples were completed by Bio Limno Research and Consulting, Inc. (Bio Limno), Halifax, Nova Scotia.

7.2.1.1.2 AEMP Version 2.0 Data (2007 to 2011)

Phytoplankton community sampling was completed over the course of the AEMP Version 2.0 as part of a Plankton SES. Sampling for the *AEMP Study Design Version 2.0 Plankton SES* continued to follow the procedures outlined in the *AEMP Study Design Version 1.0*; however, during revisions to the DDMI specific operating procedures for the summer sampling program (SOP-ENV-AQU 08), the phytoplankton sampling procedure was inadvertently changed to use the Secchi depth to determine the sampling depth (DDMI 2007). Since the 2007 AEMP plankton program used Secchi depth to determine sampling depth, instead of the top 10 m of the water column, sampling depths were approximately 2 m shallower than those between 2003 and 2006. From 2008 to 2016, the methods reverted back to the original sampling protocol of sampling the top 10 m of water column.

Secchi depths in 2007 were approximately 8 m; phytoplankton are found within the euphotic zone (estimated as two times the Secchi depth); therefore, it is likely that the 2007 samples are comparable to the 2008 to 2016 samples. A comparison of samples collected from the two different depths found that there was no significant difference for the chlorophyll *a* values from the two depths (Golder 2011a); consequently, the 2007 phytoplankton data were included in the evaluation of temporal trends.

Similar to previous years, taxonomic analyses of the 2007 to 2011 samples were completed by Bio Limno.

7.2.1.1.3 AEMP Version 3.0 Data (2012 to 2013)

The phytoplankton community was sampled in 2012 and 2013 as part of an AEMP monitoring component under the *AEMP Study Design Version 3.0* (Golder 2011b). Sampling methods were consistent with the *AEMP Study Design Version 2.0*, except that a single sampling period was identified from 15 August to 15

September, rather than three open-water sampling periods. In 2012, taxonomic analyses were performed by Bio Limno. However, in 2012 a number of issues with data quality were observed; therefore, in 2013, the samples were sent to a new taxonomist, Eco Logic Ltd., Vancouver, British Columbia. Differences between taxonomists exist and are unavoidable; therefore, an examination of temporal plots must be interpreted with the different taxonomic analyses in mind.

In addition to a change in taxonomists, the level of taxonomic detail needed for evaluating effects was also investigated. It was determined that species level data may be prone to errors, while genus level taxonomic identification can provide the necessary data needed for the monitoring program. All species level data from previous years were converted to the genus level, and taxonomic richness calculations were performed at the genus level, for comparison to the 2013 data (Golder 2014c).

7.2.1.1.4 AEMP Version 3.5 Data (2014 to 2016)

The phytoplankton community was sampled in 2014, 2015 and 2016 as part of an AEMP monitoring component under the *AEMP Study Design Version 3.5* (Golder 2014a). Sampling methods were consistent with the *AEMP Study Design Version 2.0*, with the exceptions that a single sampling period was identified from 15 August to 15 September, rather than three open-water sampling periods; and, during the interim sampling years (i.e., 2014 and 2015) only the NF area was sampled, as required for evaluating Action Level exceedances. A number of updates outlined in the *AEMP Design Plan Version 4.0* (Golder 2016d) and in the Wek'èezhìı Land and Water Board (WLWB) directives (28 July 2015, 26 May 2016, 14 November 2016, and 2 March 2017 Decision Packages) were incorporated into the 2016 comprehensive plankton report.

All taxonomic identifications were carried out at the genus-level because genus-level taxonomic identifications provide the necessary data needed for these monitoring programs (Golder 2014c). Magnitude of effect was assessed by comparing plankton variables in the NF area to the reference condition. Reference conditions for Lac de Gras were defined in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b). Values that were beyond the reference condition were those values that exceeded what would be considered natural variation in Lac de Gras. The magnitude of effects observed on plankton variables was determined according to the Action Level classification defined in the *AEMP Study Design Version 3.5* (Golder 2014a).

7.2.1.2 Zooplankton

Zooplankton biomass sampling locations from *AEMP Study Design Version 2.0* (2008) to *AEMP study Design Version 3.5* are presented in Table 7-2. Zooplankton data collected prior to 2008 were excluded from the data analysis due to laboratory subsampling errors and field sampling errors (Sections 7.2.1.2.1 and 7.2.1.2.2).

Table 7-2 Summary of Baseline and AEMP Zooplankton Biomass Data, 2008 to 2016

Waterbody	Area	Station[a]	UTM Coordinates[b]		Distance from Diffusers[c] (km)	Years Sampled								
			Easting (m)	Northing (m)		2008	2009	2010	2011	2012	2013	2014	2015	2016
Lac de Gras[e]	Near-field	NF1	535740	7153854	0.4		X	X	X	X	X	X	X	X
		NF2	536095	7153784	0.5		X	X	X	X	X	X	X	X
		NF3	536369	7154092	0.9		X	X	X	X	X	X	X	X
		NF4	536512	7154240	1.1		X	X	X	X	X	X	X	X
		NF5	536600	7153864	1.0		X	X	X	X	X	X	X	X
	Mid-field 1	MF1-1	535008	7154699	1.5	X	X	X	X	X	X	X	X	X
		MF1-2	533682	7155356	2.9		X	X	X					
		MF1-3	532236	7156276	4.7		X	X	X	X	X			X
		MF1-4	532494	7157657	7.2		X	X	X					
		MF1-5	528432	7157066	8.5					X	X			X
	Mid-field 2	MF2-1	538033	7154371	2.4	X	X	X	X	X	X			X
		MF2-2	539198	7154643	3.7		X	X	X					
		MF2-3	540365	7156045	5.4	X	X	X	X	X	X			X
		MF2-4	540955	7157359	6.9		X	X	X					
	Far-field 2[d]	FF2-1	541500	7159522	9.3		X	X	X					
		FF2-2	541588	7158561	8.3		X	X	X	X	X			X
		FF2-3	543478	7159267	10.1		X	X	X					
		FF2-4	543752	7158945	10.2		X	X	X					
		FF2-5	544724	7158879	11.4		X	X	X	X	X			X
	Mid-field 3	MF3-1	537645	7152432	2.7		X	X	X	X	X			X
		MF3-2	536816	7151126	4.2		X	X	X	X	X			X
		MF3-3	536094	7148215	7.2					X	X			X
		MF3-4	532545	7147011	11.0	X	X	X	X	X	X			X
		MF3-5	528956	7146972	14.6					X	X			X
		MF3-6	525427	7148765	18.5		X	X	X	X	X			X
		MF3-7	521859	7150039	22.3					X	X			X
	Far-field 1	FF1-1	525430	7161043	13.6		X	X	X	X	X			X
		FF1-2	524932	7159476	12.9		X	X	X	X	X			X
		FF1-3	526407	7160492	12.8		X	X	X	X	X			X
		FF1-4	526493	7159058	11.4		X	X	X	X	X			X
		FF1-5	526683	7161824	12.8		X	X	X	X	X			X
	Far-field B	FFB-1	516831	7148207	26.4	X	X	X	X	X	X			X
		FFB-2	518473	7150712	25.0	X	X	X	X	X	X			X
		FFB-3	518048	7147557	25.2	X	X	X	X	X	X			X
		FFB-4	515687	7150036	27.6	X	X	X	X	X	X			X
		FFB-5	516533	7150032	26.8	X	X	X	X	X	X			X
	Far-field A	FFA-1	506453	7154021	36.8		X	X	X	X	X			X
		FFA-2	506315	7155271	38.3	X	X	X	X	X	X			X
		FFA-3	505207	7153887	38.7	X	X	X	X	X	X			X
		FFA-4	503703	7154081	40.2	X	X	X	X	X	X			X
		FFA-5	505216	7156657	40.0		X	X	X	X	X			X

Table 7-2 Summary of Baseline and AEMP Zooplankton Biomass Data, 2008 to 2016

Waterbody	Area	Station[(a)]	UTM Coordinates[(b)] Easting (m)	UTM Coordinates[(b)] Northing (m)	Distance from Diffusers[(c)] (km)	Years Sampled 2008	2009	2010	2011	2012	2013	2014	2015	2016
Outlet of Lac de Gras	n/a	LDG-48	490900	7161750	55.6									
Lac du Sauvage	n/a	LDS-1	546398	7161179	-				X		X			X
		LDS-2	546807	7160027	-				X		X			X
		LDS-3	547191	7160256	-				X		X			X

Note: Shading identifies stations that were discontinued, as per the AEMP Study Design, Version 3.0 (Golder 2011b). These stations are not included in the current (2014 to 2016) Aquatic Effects Re-evaluation Report.
a) Current AEMP station names are presented, and historical sampling station names are provided in parentheses.
b) UTM coordinates are reported as Zone 12, North American Datum (NAD) 83.
c) Approximate distance from the Mine effluent diffusers along the most direct path of effluent flow.
d) From 2012 onwards, the FF2 area is considered to be part of the MF2-FF2 area.
e) SNP data are not included.
UTM = Universal Transverse Mercator coordinate system; - = not applicable.

7.2.1.2.1 Baseline and AEMP Version 1.0 Data (1995 to 2006)

Baseline data for the zooplankton community was collected during the open-water seasons in 1995 (Acres and Bryant 1996) and 1997 (Golder 1998). The 1995 and 1997 baseline surveys collected zooplankton samples from 5 m and 10 m depths to the surface, respectively. The current sampling procedure consists of starting from 1 m above the sediment and pulling the plankton net up through the entire water column, which is not comparable to the 1995 and 1997 data; therefore, the 1995 and 1997 data are not included in this report. Archived zooplankton samples from 2000 to 2006 were not submitted for analysis due to laboratory subsampling errors discovered in 2008 (DDMI 2008).

7.2.1.2.2 AEMP Version 2.0 Data (2007 to 2011)

The 2007 data were excluded from the data analysis due to subsampling errors, which prevented accurate calculation of zooplankton biomass (DDMI 2008). From 2008 to 2011, zooplankton samples were collected through the entire water column, starting at 1 m from the bottom (referred to as bottom). A number of samples from the 2008 AEMP program were mistakenly collected from a 10 m depth, rather than from the bottom of the water column; these samples were removed from data analysis (Table 7-3). Samples collected from 2008 to 2011 were analyzed by Salki Consultants Inc., Winnipeg, Manitoba.

Table 7-3 Zooplankton Samples Removed from Data Analysis as a Result of Field Sampling Errors in 2008

Area	Station	Date	Depth[a] (m)
Near-field	NF1	15-Aug-2008	10
	NF2	16-Aug-2008	10
	NF3	15-Aug-2008	10
	NF1	10-Sept-2008	10
	NF2	10-Sept-2008	10
Mid-field 2	MF2-3	02-Sept-2008	10
Mid-field 3	MF3-1	03-Sept-2008	10
	MF3-2	03-Sept-2008	10
Far-field 1	FF1-1	05-Sept-2008	11.25
	FF1-2	06-Sept-2008	11.25
	FF1-3	06-Sept-2008	11.25
Far-field A	FFA-1	21-Aug-2008	10
	FFA-1	12-Sept-2008	11.25
	FFA-5	15-Sept-2008	10

a) Starting depth during sampling.

7.2.1.2.3 AEMP Version 3.0 Data (2012 to 2013)

The zooplankton community was sampled in 2012 and 2013 as part of an AEMP monitoring component under the *AEMP Study Design Version 3.0* (Golder 2011b). Sampling methods were consistent with those under the *AEMP Study Design Version 2.0*, except that there was only a single sampling period (from 15 August to 15 September), rather than the three open-water periods sampled under the *AEMP Study Design Version 2.0*. Each sample consisted of a composite of three vertical hauls of the entire water column (from a depth of 1 m above the sediment). Samples were analyzed by Salki Consultants Inc., Winnipeg, Manitoba.

7.2.1.2.4 AEMP Version 3.5 Data (2014 to 2016)

The zooplankton community was sampled in 2014, 2015 and 2016 under the *AEMP Study Design Version 3.5* (Golder 2014a). Sampling methods were consistent with the *AEMP Study Design Version 2.0*, with the exceptions that (1) a single sampling period was identified from 15 August to 15 September, rather than three open-water sampling periods, and (2) during the interim sampling years (i.e., 2014 and 2015), only the NF area was sampled, as required for evaluating Action Level exceedances. A number of updates outlined in the *AEMP Design Plan Version 4.0* (Golder 2016d) and in the Wek'èezhìı Land and Water Board (WLWB) directives (28 July 2015, 26 May 2016, 14 November 2016, and 2 March 2017 Decision Packages) were incorporated into the 2016 comprehensive plankton report.

Magnitude of effect was assessed by comparing plankton variables in the NF area to the reference condition. Reference conditions for Lac de Gras were defined in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b). Values that were beyond the reference condition were those values that exceeded what would be considered natural variation in Lac de Gras. The magnitude of effects observed on plankton variables was determined according to the Action Level classification defined in the *AEMP Study Design Version 3.5* (Golder 2014a).

Sampling for the zooplankton community in 2014, 2015 and 2016 under the *AEMP Study Design Version 3.5* (Golder 2014a) was consistent with previous years; however, differences in the net dimensions were noted among years (Table 7-4). Re-measurement of the net diameter determined that the zooplankton net used in all three years was 30.0 cm, and did not vary between 2014 and 2016; therefore, recalculation of the zooplankton community biomass data was necessary for 2014 and 2016. The recalculation increased the overall zooplankton biomass in 2014 and 2016 by 3.4% and 6.8%, respectively, as the volume of water actually sampled was smaller than that used in the 30.5 cm and 31.0 cm net diameter calculation. This increase did not affect the Action Level classification in these years, as the Action Levels were assessed against the current years FF data.

Table 7-4 Adjusted Zooplankton Biomass, 2014 to 2016

Year	Original Net Diameter Used in Annual Report (cm)	Adjusted Net Diameter (cm)	Percent Adjustment to the Overall Dataset (%)
2014	30.5	30.0	3.4
2015	30.0	30.0	0
2016	31.0	30.0	6.8

7.2.2 Data Handling

7.2.2.1 Data Screening

Initial screening of the plankton data was completed prior to data analyses during each annual report to identify anomalous values and to decide whether to retain or exclude anomalous data from further analysis. The anomalous data screening approach for AEMP component datasets was approved as part of the *2011 to 2013 Aquatic Effects Re-evaluation Report* (Golder 2016a), this approach was applied to the historical dataset to screen for anomalous data herein.

7.2.2.2 Data Preparation and Variable Selection

Plankton data were prepared for the trend evaluation by averaging zooplankton replicates within stations on a yearly basis. Consistency in taxonomic level of identification was evaluated among years, and adjustments were made where appropriate. For example, certain taxa at some stations were re-classified to a higher level to match the taxonomic resolution of the remainder of the data.

The following plankton community variables were selected for the analysis of the phytoplankton and zooplankton data:

- total biomass
- taxonomic richness (richness)
- biomass of major groups
- relative biomass of major groups
- community structure based on multidimensional scaling (MDS)

7.2.2.3 Quality Assurance/Quality Control

The *Quality Assurance Project Plan Version 3.1* (Golder 2017a), or QAPP, outlined the QA/QC procedures employed to support the collection of scientifically-defensible and relevant data required to meet the objectives of the *AEMP Study Design Version 3.5* (Golder 2014a). The QAPP was designed so that field sampling, laboratory analysis, data entry, data analysis, and report preparation activities that follow the QAPP produced technically sound and scientifically defensible results. Descriptions of the QA/QC programs and the quality control (QC) results for the phytoplankton and zooplankton datasets are provided in the annual reports.

7.2.3 Data Analysis

7.2.3.1 Normal Ranges

The magnitude of effect on plankton communities was evaluated by comparing plankton variables (i.e., total biomass, richness, and the total and relative biomass of each major ecological group) in the NF, MF and FF areas to background values. Background values for Lac de Gras are those that fall within the range of natural variability, referred to as the normal range. Normal ranges were calculated using data from the three

AEMP FF areas (i.e., FF1, FFA, and FFB) from 2007 to 2010 (with some exceptions). The 2007 to 2010 normal ranges were obtained from the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b) and are summarized in Table 7-5.

The WLWB has requested that as part of the *2014 to 2016 Aquatic Effects Re-evaluation Report* an assessment of the reference conditions for the phytoplankton variables be examined (W2015L2-0001) because of the change in phytoplankton taxonomist that occurred in 2013. The assessment includes a comparison of the AEMP results from 2014 to 2016 to reference conditions as defined using the currently approved 2007 to 2010 reference area data (Golder 2017b) and to the 2013 FF area data. The 2013 normal range was calculated following the same approach as the 2007 to 2010 normal range, i.e., 2.5 and 97.5 percentiles; anomalous data were replaced by surrogate values calculated as the average of all other samples collected in that year at that station (Golder 2017b). In addition, an "adjusted" 2013 normal range was also calculated to incorporate year-to-year variation into the 2013 normal range, the year-to-year variation is based on the original 2007 to 2010 reference area data and assumes that the within-year variation between taxonomists is similar. The following steps were taken to calculate the "adjusted" normal range:

1) The annual normal ranges and the widths (i.e., the upper minus lower boundaries) of the normal ranges for 2007, 2008, 2009, 2010 and 2013 were calculated, and the combined 2007 to 2010 normal range and width was calculated.
2) The annual variability in width of the normal range was estimated as a proportion by dividing the width of each year from 2007 to 2010, by the combined 2007 to 2010 normal range width.
3) The median annual variability in normal range width for each variable was then calculated based on results of Step 2.
4) The "adjusted" 2013 normal range width was calculated as the 2013 "raw" width divided by the median annual variability in the normal range.
5) The estimated change in 2013 width was calculated as the original "raw" 2013 width subtracted from the "adjusted" width; if the lower boundary of the original 2013 normal range was zero, than the entire change in width was added to the upper limit. If the lower boundary of the original 2013 normal range was greater than zero, then half of the change was subtracted from the lower boundary and half was added to the upper boundary. If the lower boundary was negative it was adjusted to zero.

The 2007 to 2010, 2013 and "adjusted" 2013 normal ranges are summarized in Table 7-5.

Table 7-5 Normal Ranges for Plankton

Variable	Unit	2007 to 2010 Normal Range [a]		2013 Normal Range		Ratio of 2007 to 2010 Range / 2013 Range	Adjusted 2013 Normal Range	
		Lower Bound	Upper Bound	Lower Bound	Upper Bound		Lower Bound	Upper Bound
Phytoplankton								
Phytoplankton taxonomic richness	no. taxa	12	25	21	34	1.0	19	36
Total phytoplankton biomass	mg/m^3	140	352	75	319	0.9	19	385
Diatom Biomass	mg/m^3	5.2	66	0	9.5	6.8	0	13
Microflagellate Biomass	mg/m^3	1.2	119	20	65	2.6	13	72
Cyanobacteria Biomass	mg/m^3	4.9	134	4	43	3.3	0	48
Dinoflagellate Biomass	mg/m^3	0	19	0	20	1.0	0	40
Chlorophyte Biomass	mg/m^3	26	175	25	252	0.7	0	309
Relative diatom biomass	%	2.3	25	0	3	5.7	0	4
Relative microflagellate biomass	%	0.5	49	15	56	1.2	5	64
Relative cyanobacteria biomass	%	3.4	48	4	30	1.8	0	37
Relative dinoflagellate biomass	%	0	5.1	0	12	0.4	0	29
Relative chlorophyte biomass	%	17	58	26	71	0.9	17	78
Zooplankton								
Zooplankton taxonomic richness	no. taxa	11	17	n/a	n/a	n/a	n/a	n/a
Total zooplankton biomass	mg/m^3	131.5	539.8	n/a	n/a	n/a	n/a	n/a
Cladocera Biomass	mg/m^3	8.2	126.6	n/a	n/a	n/a	n/a	n/a
Calanoida Biomass	mg/m^3	60.9	359.1	n/a	n/a	n/a	n/a	n/a
Cyclopoida Biomass	mg/m^3	13.2	105.1	n/a	n/a	n/a	n/a	n/a
Rotifera Biomass	mg/m^3	1.6	7.3	n/a	n/a	n/a	n/a	n/a
Relative cladocera biomass	%	3.8	38.2	n/a	n/a	n/a	n/a	n/a
Relative calanoida biomass	%	39.8	72.2	n/a	n/a	n/a	n/a	n/a
Relative cyclopoida biomass	%	7.0	38.8	n/a	n/a	n/a	n/a	n/a
Relative rotifera biomass	%	0.5	2.2	n/a	n/a	n/a	n/a	n/a

a) Source: *AEMP Reference Conditions Report Version 1.2* (Golder 2017b).

n/a = not applicable.

7.2.3.2 Temporal Trends

To visually evaluate temporal trends, total phytoplankton and zooplankton biomass, taxonomic richness, and the total biomass and relative biomass of major ecological groups were plotted against time (years). The time series plots included a shaded region showing the normal range obtained from the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b).

7.2.3.2.1 Temporal Trend Analysis

The analysis of the 2002 to 2012 phytoplankton dataset and of the full 2008 to 2016 zooplankton dataset proceeded as outlined in Section 2.4.2, with the exception of the removal of dinoflagellates and the others group from the trend analysis. These two groups contained a large proportion of zero values in the dataset, which led to highly non-normal residuals. Because these groups contribute a small proportion of overall biomass (generally <10%), they were not included in the trend analysis. Due to the change in taxonomist used for phytoplankton analysis in 2013, the 2013 to 2016 dataset was not fully comparable to the 2002 to 2012 dataset. Therefore, the 2013 to 2016 phytoplankton data were analyzed using a separate modelling approach.

The temporal trend analysis of the 2013 to 2016 phytoplankton data differed from the general trend analysis methods described in Section 2.4.2. The full four years of data were available for the NF area; however, only two years of data were available for the MF and FF areas. Therefore, the general methods that used year as a continuous covariate to examine temporal trends could not be applied. Instead, the trend model described in Section 2.4.2 was simplified into a fixed effects model of only the NF data, with year as a single, discrete covariate. No model selection was performed, since there was only one candidate model for analysis (i.e., one-way ANOVA). Data transformation using Yeo-Johnson transformations were performed as described in Section 2.4.2. Residual heteroscedasticity was examined and heteroscedasticity effects by year or predicted value were added to the model if they improved model fit (i.e., as identified using AIC scores).

7.2.3.2.2 Extent of Effects

The area of the lake with total biomass less than the normal range was estimated for total phytoplankton and zooplankton biomass, and this measure was used to estimate the extent of potential toxicological effects. The extent of effects on phytoplankton and zooplankton community biomass were not previously calculated or presented in the annual reports; therefore, they were calculated and presented for all years from 2007 to 2016, if applicable. Extent of effects was compared among years to evaluate whether effects were expanding further into the lake over time.

To quantify the extent of effects on phytoplankton and zooplankton biomass along each transect, a linear interpolation method was used to estimate the distance between the station farthest from the diffuser with a value less than the normal range, and the adjacent station with a value above the normal range. In cases where biomass did not decrease uniformly with distance from the diffuser, a conservative approach was taken by assuming that the effect extended to the farthest station with biomass below the normal range, even if closer stations along the transect had concentrations within the normal range. The one exception to this conservative approach occurred when there were three consecutive stations above or within the normal range, which showed an increase in biomass with distance from the diffuser, followed by a single station below the normal range. Under such circumstances, the low biomass at that distant station was assumed to be unrelated to the Mine.

7.2.3.3 Multivariate Statistical Analysis

7.2.3.3.1 Community Analysis

Phytoplankton and zooplankton community structure were summarized using the non-parametric ordination method of multidimensional scaling (MDS; Clarke 1993). The MDS data were scaled in Primer Version 7 (Clarke and Gorley 2016). Genus level phytoplankton data and lowest level (genus or species) zooplankton data were log (x+1) transformed, to improve the separation of the data among stations on the MDS plots and to reduce weighting of the analysis by the most abundant taxa. A Bray-Curtis resemblance matrix was generated, and the MDS procedure was applied to this matrix. Using rank order information, MDS determined the relative positions of stations in two dimensions based on community composition. Goodness-of-fit was determined by examining the Shepard diagrams as well as the stress values, which were calculated from the deviations in the Shepard diagrams. Lower stress values (i.e., less than 0.10) indicate less deviation and a greater goodness-of-fit, higher stress values (i.e., greater than 0.20) must be interpreted with caution and often higher dimensions (i.e., 3-D) are needed to described the data (Clarke 1993). Points that fall close together on the MDS ordination plot represent samples with similar community composition; points that are far apart from each other represent samples with dissimilar community composition. Where stress was low metric MDS (mMDS) was employed; however, if stress values were higher, a non-metric MDS (nMDS) was used.

A similarity profile (SIMPROF) test was also carried out on the ordination data to identify meaningful clusters of important taxa (i.e., those taxa that behave in a coherent manner across areas) and to prevent over-interpretation of the MDS plots (Clarke et al. 2014). These SIMPROF clusters were superimposed on the MDS plots.

Station groupings were consistent with the *AEMP Study Design Version 3.5* (Golder 2014a). Area means were determined for NF and FF (FF1, FFB, and FFA) areas for each year. Community changes over time were assessed between the NF and FF areas. A trajectory line showing movement in the NF community over time was superimposed on these nMDS plots. For the zooplankton community data, all data from 2008 to 2016 were plotted together. For the phytoplankton community data, the baseline to 2012 data were plotted and analyzed separately from the 2013 to 2016 data because of the differences caused by the change in taxonomist in 2013 (Golder 2014c). In addition to the NF to FF area comparisons, temporal MDS plots for each transect were created: the NF-MF1-FF1, the NF-MF2-FF2 transect and the NF-MF3-FFB-FFA transect. The area means for the NF and FF areas (within which all stations are subject to a similar level of effluent exposure) were used in the ordinations, while for the MF data (where exposure to effluent varied among stations), single stations along the transects were shown. A trajectory line showing movement in the NF community over time was also superimposed on the phytoplankton community MDS plots. On the zooplankton community MDS plots, the trajectory line highlighted the 2016 data in each transect. The 2016 sampling year was chosen, because it showed the greatest separation in the data among stations, among all years sampled.

An overall one-way analysis of similarities (ANOSIM) test was carried out on the Bray-Curtis resemblance matrix to confirm interpretation of the separation of the points on the MDS ordination plot, and to investigate whether differences in community composition observed in the nMDS or mMDS ordination plots were significant. In addition, an area-level ANOSIM was carried out on NF to FF area Bray-Curtis resemblance matrix, to test differences among areas contrasted with differences among replicates within each area for the NF and FF areas (Clarke et al. 2014). An R-statistic ranging from 0 to 1 is produced by the analysis; a strong difference in community composition is indicated by R = 1 and no difference between communities is indicated by R = 0. Significance of the R statistic was achieved when a *P*-value of >0.05 was observed, based on the five replicate stations per area.

7.2.3.4 Integrative Environmental Assessment

The WLWB has requested that as part of the *2014 to 2016 Aquatic Effects Re-evaluation Report,* an integrative assessment of plankton variables be conducted. The assessment includes a comparison of TDS, as an indicator of exposure to Mine effluent in Lac de Gras, and nutrients, TN, TP and the N to P molar ratio (described in Section 5), to phytoplankton biomass and community structure. The relationship between zooplankton biomass and community data and TDS concentrations was evaluated. The assessment also includes a detailed evaluation of MDS ordination results to assess the influence of environmental gradients. The phytoplankton and zooplankton community data structure were summarized using the MDS procedure described in Section 7.2.3.1 (Clarke 1993). The NF and FF area MDS plot was used for these investigations, as the greatest differences in nutrient and TDS concentrations would be expected between the NF and FF areas.

To visually evaluate spatial trends relative to the Mine discharge, total phytoplankton and zooplankton biomass were plotted against TDS. When quality control issues arose with the TDS data (i.e., in 2016; Golder 2017c), the relationship between specific conductivity from previous years ($y = 0.5x – 0.94$) was used to estimate TDS, because specific conductivity and TDS are strongly correlated ($r = 0.97$) and both can be uses as tracers of Mine effluent. Concentrations of TDS at each station relative to the phytoplankton and zooplankton community structure on the MDS plots were examined using bubble plots superimposed on the MDS community plots. The size of the bubble corresponds to concentrations of TDS at each station. The bubble is superimposed on the community MDS plot, providing a visual relationship between TDS concentrations and community structure, and shows how the community separates among areas in multidimensional space. If TDS is driving the community separation it should be visible on the MDS plot as a gradient in bubble size across the plot. The relationship between phytoplankton community structure and the ratio of N to P (described in Section 5) was examined in a similar manner to TDS, with bubble plots superimposed on the MDS community plot.

The relationship (based on the Pearson correlation co-efficient [*r*]) between TN, TP, the ratio of N to P for phytoplankton biomass, and between TDS and phytoplankton and zooplankton biomass were examined for each year with available data from 2007 to 2016.

A LINKTREE test (i.e., Linkage tree) was also carried out on the phytoplankton community data. LINKTREE tests take a subset of the abiotic variables, in this case TN, TP and TDS, and use them to describe how the phytoplankton community samples are optimally split into groups on the MDS plot. The LINKTREE interprets the data by consecutively splitting the data based on abiotic variables (i.e., Group 1 communities have abiotic variable <x but Group 2 variables are >x, with no samples between these thresholds; Clarke et al. 2014). Groups 1 and 2 are then each divided into two by a different threshold on the same abiotic variable, or more likely a different abiotic variable. The result is divisive clustering of the samples (stations in this case) and an environmental interpretation. The procedure is non-metric on both the community resemblance matrix and on the abiotic variables. The way the abiotic variables are combined in the partitioning of the biotic samples is non-linear and non-additive. LINKTREE attempts local explanations rather than holistic ones; a disadvantage of local explanations is that many abiotic inequalities will explain the same assemblage divisions, unless the environmental variable set is initially drastically pruned, which was attempted in this case by only including three variables. An advantage of LINKTREE is that it is geared towards prediction and not just interpretation.

To evaluate the potential role of nutrient limitation and subsequent effects on atmospheric nitrogen (N_2)-fixing cyanobacteria, the total biomass of N_2-fixing cyanobacteria per year and sampling area (i.e., NF and FF areas) was calculated and plotted. Since, heterocyst formation (i.e., a differentiated cyanobacterial cell that carries out nitrogen fixation) is an important aspect in nitrogen fixation; taxa capable of heterocyst formation were included as N_2-fixing cyanobacteria. These taxa include species of the genera *Anabaena sp., Aphanizomenon sp., Cylindrospermopsis sp., Nodularia sp., and Nostoc sp*. Nitrogen-fixing ability, however, is also observed in several non-heterocystous unicellular forms (*Gleocapsa sp., Aphanothece sp., Gleothece sp.*) and filamentous forms (*Oscillatoria sp., Plectonema sp*.); therefore, these taxa were also included in the list of N_2-fixing cyanobacteria presented in Table 7-6. The linear relationship between TN, TP, the ratio of N to P and N_2-fixing cyanobacteria biomass was examined for each year with available data from 2007 to 2016, to determine whether changes in nutrient concentrations in the NF and FF areas of Lac de Gras are causing changes in the biomass of cyanobacteria capable of atmospheric nitrogen fixation.

Table 7-6 Cyanobacteria Taxa Present in the NF and FF areas, 2007 to 2016

Cyanobacteria Taxa Present in Lac de Gras	Capable of nitrogen-fixation
Anabaena sp.(a)	Y
Aphanizomenon sp.(b)	Y
Aphanocapsa sp.	-
Aphanothece sp.[(b)]	Y
Chroococcus sp.[(b)]	Y
Cylindrospermum sp.	-
Dactylococcopsis sp.	-
Gomphosphaeria sp.	-
Limnothrix sp.	-
Lyngbya sp.[(c)]	Y
Merismopedia sp.[(b)]	Y
Microcystis sp.[(b)]	Y
Oscillatoria sp.[(b)]	Y
Phormidium sp.[(c)]	Y
Planktolyngbya sp.	-
Planktothrix sp.	-
Pseudanabaena sp.[(c)]	Y
Rhabdoderma sp.	-
Rhaphidiopsis sp.	-
Snowella sp.	-
Synechococcus sp.	-
Synechocystis sp.[(c)]	Y

Y = yes
a) Stewart 1973
b) Issa et al. 2014
c) Bergman et al. 1997

7.2.3.5 Weight-of-Evidence Effect Ratings

The results of the AEMP plankton community surveys are integrated through the WOE evaluation process, which determines the strength of evidence supporting the two broad impact hypotheses for Lac de Gras: toxicological impairment and nutrient enrichment (Golder 2014a). The WOE is not intended to determine the ecological significance or level of concern associated with Mine-related effects. The WOE effect ratings incorporate statistical comparisons of the NF and FF areas, and comparisons of the NF area to the normal range as defined in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b). The plankton data were assessed according to the WOE effect level ratings described in Section 10 and summarized in Table 7-7.

Table 7-7 Weight-of-Evidence Effect Ratings for Plankton

Measurement Endpoint	Early Warning/Low ↑ / ↓	Moderate ↑↑ / ↓↓	High ↑↑↑ / ↓↓↓
Comparison to FF Areas and Normal Range:[a] Phytoplankton Biomass Zooplankton Biomass	Statistically significant change in the NF vs FF areas	Low rank AND NF area mean outside normal range	Moderate AND values in >20% of the lake area is either higher or lower than the normal range
Community Structure:[a] Phytoplankton community composition Zooplankton community composition	Divergent community structure at the species or genus level in the NF vs FF areas	A shift in community structure at the ecological grouping[b] level between the NF and FF areas	Moderate AND a statistically significant change in taxonomic richness >FF area mean ± 2 SD

Notes: Normal ranges are defined and provided in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b).

a) Applied separately for each measurement endpoint.

b) Ecological groupings for phytoplankton are cyanobacteria, chlorophytes, microflagellates, dinoflagellates, and diatoms; and for zooplankton are cladocerans, cyclopoids, calanoids, and rotifers.

NF= near-field; FF = far-field; >= greater than; SD = standard deviation; vs = versus.

7.2.3.6 Action Levels

The magnitude of effects on phytoplankton and zooplankton was categorized according to the Action Levels in the Response Framework presented in the *AEMP Study Design Version 3.5* (Golder 2014a). The main goal of the Response Framework is to ensure that significant adverse effects never occur. This is accomplished by requiring proponents to take actions at defined Action Levels, which are triggered well before significant adverse effects could occur. A significant adverse effect, as it pertains to aquatic biota, was defined in the EA for the Mine as a change in fish population(s) that is greater than 20% (Government of Canada 1999). The effect must have a high probability of being permanent or long-term and must occur throughout Lac de Gras. The Significance Thresholds for all aquatic biota, including plankton, are therefore related to impacts that could result in a change in fish population(s) that is greater than 20%.

Action Levels for plankton address the toxicological impairment hypothesis, while the nutrient enrichment hypothesis is addressed in the Eutrophication Indicators component. Conditions required to trigger Action Levels 1 to 3 for plankton are defined in Table 7-8. Conditions for Action Level 4 will be defined if Action

Level 3 is triggered. Defining higher Action Levels after initial effects are encountered is consistent with the draft guidelines for preparing a Response Framework in AEMPs (WLWB 2010; Racher et al. 2011).

Phytoplankton and zooplankton biomass and taxonomic richness are assessed annually, during both interim and comprehensive year sampling programs. This involves statistically comparing plankton biomass and richness between the NF area (and potentially MF areas) and the FF areas, and comparing NF area results to the reference condition (Table 7-8). Since toxicological impairment is expected to result in declines in most plankton variables relative to the reference condition, Action Level 1 will be triggered if the mean value in the NF area is significantly lower compared to the FF areas, as per the AEMP Study Design Version 3.5; Golder 2014a). Action Level 2 is triggered when the effect observed in the NF area expands to the nearest MF stations (i.e., MF1-1, MF2-1, MF3-1), and Action Level 3 is triggered when NF area results fall below the normal range as defined in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b). Under the updated AEMP Study Design Plan Version 4.1 (Golder 2017d), the statistical comparison to evaluate Action Level 1 is made between the NF area and the FF area data used to calculate normal ranges (i.e., 2007 to 2010).

Table 7-8 Action Levels for Plankton Effects

Action Level	Criteria	Extent	Action
1	Mean biomass or richness significantly less than reference area means	Near-field	Confirm effect
2	Mean biomass or richness significantly less than reference area[(a)] means	Nearest Mid-field station	Investigate cause
3	Mean richness less than normal range[(b)]	Near-field	Examine ecological significance Set Action Level 4 Identify mitigation options
4	TBD[(c)]	TBD[(c)]	Define conditions required for the Significance Threshold
5	Decline in biomass or richness likely to cause a >20% change in fish population(s)	Far-field A	Significance Threshold[(d)]

a) Action Levels were assessed by comparing NF and MF areas to the FF1, FFA and FFB areas, which formerly served as reference areas.

b) Normal ranges were obtained from the AEMP Reference Conditions Report Version 1.2 (Golder 2017b).

c) To be determined if Action Level 3 is triggered.

d) Although the Significance Threshold is not an Action Level, it is shown as the highest Action Level to demonstrate escalation of effects towards the Significance Threshold.

7.3 Results

7.3.1 Summary of Effects

7.3.1.1 Weight-of-Evidence Effect Ratings

Effect ratings for phytoplankton biomass have consistently demonstrated a nutrient enrichment response since 2007, but the extent of this response has varied, resulting in fluctuating WOE ratings between moderate to high from 2007 to 2016, with the exception of a low-level rating in 2013. (Table 7-9). Greater than 20% of the lake was affected in 2009 and 2011, as demonstrated by the high effects rating. A low effect rating on phytoplankton community structure was observed in 2008 and from 2010 to 2013, but a moderate effect rating was observed in 2016. The changes in community structure could be associated with either nutrient enrichment or a toxicological response.

The response of zooplankton biomass has been inconsistent over the years, ranging from no response in 2009 and 2013, to a moderate nutrient enrichment rating in 2011, and a low toxicological response rating in 2016 (Table 7-9). As with the phytoplankton community, the zooplankton community has demonstrated inconsistent responses over the years, and the changes observed could be categorized as either a nutrient enrichment or toxicological response. Low effect ratings were observed in 2010 and 2011, and moderate effect ratings were observed in 2013 and 2016.

Table 7-9 Summary of Weight-of-Evidence Effect Ratings for Plankton, 2007 to 2016

Variable	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Phytoplankton biomass (based on enumeration)	↑↑	↑↑	↑↑↑	↑↑↑ (a)	↑↑↑	↑↑	↑	n/a	n/a	↑↑
Phytoplankton community structure	0	↑/↓	0	↑/↓	↑/↓	↑/↓	↑/↓	n/a	n/a	↑↑/↓↓
Zooplankton biomass (based on enumeration)	n/a	↑↑	0	↑	↑↑	↑	0	n/a	n/a	↓
Zooplankton community structure	n/a	0	0	↑/↓	↑/↓	0	↑↑/↓↓	n/a	n/a	↑↑/↓↓

Notes: a) re-calculation of the area of effected in 2017 determined that a high-level rating was more appropriate because >20% of the lake was affected. The direction of the arrow indicates the direction of change for a given biological response endpoint relative to the far-field and/or normal range. 0 = no response; ↑/↓ = early warning low-level rating; ↑↑/↓↓ = moderate-level rating; ↑↑↑/↓↓↓ = high-level rating; n/a = data not available or not assessed. The direction of the sign (↑ or ↓) indicates the direction of difference relative to the FF areas. For community structure endpoint, both arrows are included (e.g., ↑/↓ or ↑↑/↓↓) to reflect that a community shift normally involves combined increases and decreases in biomass.

7.3.1.2 Action Levels

The Action Levels for plankton effects address the toxicological impairment hypothesis. The response of phytoplankton and zooplankton biomass and taxonomic richness indicate that an Action Level 1 was not reached between 2007 and 2014 (Table 7-10). An Action Level 1 was triggered in 2015 when significantly lower phytoplankton biomass was observed in the NF area compared to the FF areas. In 2015, cyanobacterial biomass was also assessed against the plankton Action Levels, and was reported to trigger Action Level 1. However, this variable is not included in the Response Framework for plankton and, therefore, the reported Action Level 1 trigger for cyanobacterial biomass was not considered in this re-evaluation. An Action Level 2 was triggered in 2016 when significantly lower zooplankton biomass was observed in the NF area and in the nearest MF stations compared to the FF areas.

The Action Level 1 and Action Level 2 exceedances for phytoplankton biomass in 2015 and zooplankton biomass in 2016 provide an early-warning indication of potential Mine-related changes, and are not of immediate concern regarding toxicity to the plankton community in Lac de Gras, for the following reasons:

- Phytoplankton biomass only triggered an Action Level 1 in 2015 and did not trigger an Action Level again in 2016.
- The difference between mean total zooplankton biomass in 2016 in the area subject to greatest effluent exposure (NF area) and the FF areas was 17%, which is low compared to the typical degree of variation observed among areas within a large lake (Golder 2017c).
- Mean total zooplankton biomass in the NF area was within the normal range (Golder 2017c).
- A similar response was not observed in another, more directly measured indicator of total zooplankton biomass (i.e.,AFDM), which was found to be significantly greater in the NF area compared to the FF areas by the eutrophication indicators component of the AEMP (Golder 2017c).
- The water quality component of the AEMP did not document effluent toxicity to invertebrates or algae, and concentrations of all variables in Lac de Gras were below AEMP Effects Benchmarks (Golder 2016b,c,2017c).

Table 7-10 Summary of Action Level Rankings for Plankton, 2007 to 2016

Variable	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Phytoplankton biomass (based on enumeration)	-	-	-	-	-	-	-	-	AL1	-
Phytoplankton taxonomic richness	-	-	-	-	-	-	-	-	-	-
Zooplankton biomass (based on enumeration)	-	-	-	-	-	-	-	-	-	AL2
Zooplankton taxonomic richness	-	-	-	-	-	-	-	-	-	-

- = did not trigger an Action Level; AL1 = Action Level 1 triggered; AL2 = Action Level 2 triggered; AL3 = Action Level 3 triggered.

7.3.2 Temporal Trends

7.3.2.1 Phytoplankton Community

In the analysis of phytoplankton variables from 2002 to 2012, the temporal trends for phytoplankton taxonomic richness, total phytoplankton and chlorophyte biomass were parabolic and the trends for microflagellate, diatom, and cyanobacteria biomass were linear (Table 7-11). Phytoplankton taxonomic richness data were not transformed; however, the remaining variables were log or square root transformed (Table 7-11).

Table 7-11 Phytoplankton AIC Model Selection, 2002 to 2012

Variable	Yeo-Johnson transformation[(a)]	Parabolic AIC Score	Linear AIC Score
Phytoplankton taxonomic richness	none	1339	1343
Total phytoplankton biomass	0	186	238
Total diatom biomass	0	679	677
Total microflagellate biomass	0.5	1461	1463
Total cyanobacteria biomass	0.5	1583	1579
Total chlorophyte biomass	0	562	602

Note: Grey shaded cells indicate the model chosen; – = no data transformation; AIC = Akaike's information criterion.

a) See Section 2.4.2.1 for details on transformation approach.

7.3.2.1.1 Phytoplankton Taxonomic Richness and Biomass

Temporal trends in phytoplankton taxonomic richness differed among sampling areas in Lac de Gras between 2002 and 2012 (Figures 7-1 and 7-2; Table 7-12). In the NF area, a linear increase between 2002 and 2012 was observed; however, between 2013 and 2016 no trend was observed (Figure 7-3). Taxonomic richness at stations in the NF area generally remained within the 2007 to 2010 normal range between 2007 and 2011, with individual station excursions outside the upper limit of the normal range occurring throughout (Figure 7-1). In 2012, stations in the NF area were within or just above the upper limit of the normal range. From 2013 to 2016, taxonomic richness at stations in the NF area was within or above the upper limit of the 2007 to 2010 normal range, but within the 2013 and "adjusted" 2013 normal ranges.

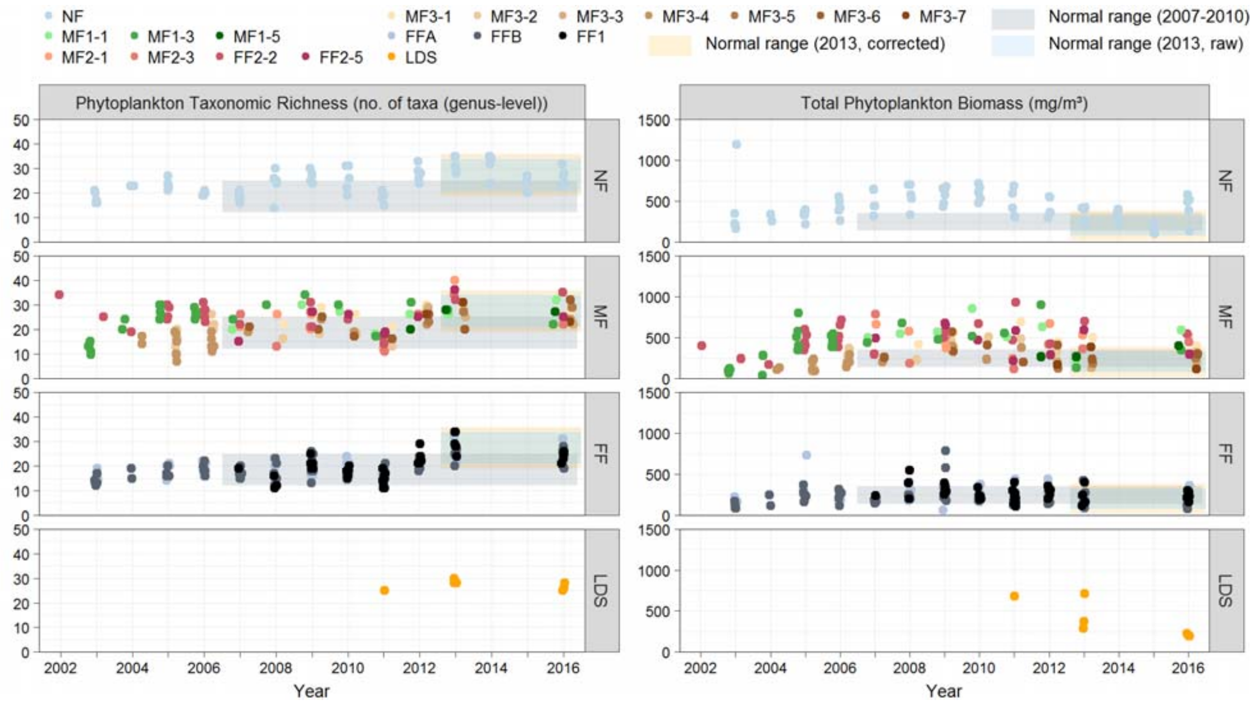
Between 2002 and 2012, taxonomic richness at stations in the MF2-FF2, MF1 and MF3 areas was often within the 2007 to 2010 normal range (Figure 7-1). Generally, stations in the MF1 and MF3 areas showed an increase, until 2009 in the MF3 area and 2012 in the MF1 area, when richness at most stations decreased. Stations in the MF2-FF2 area showed a general decreasing trend (Figure 7-2). In all areas, multiple stations in the three MF areas exceeded the upper limit of the normal range in one or more years after 2009. In 2011, taxonomic richness at FF2-2 and FF2-5 in the MF2-FF2 area were at or just below the lower limit of the 2007 to 2010 normal range. From 2013 to 2016, stations in the MF areas were above the upper limit of the 2007 to 2010 normal range but generally within the 2013 and "adjusted" 2013 normal ranges.

Taxonomic richness followed a similar temporal trend in the FF areas as observed in the MF areas (Figure 7-1) Taxonomic richness at stations in the FF areas generally fell within the 2007 to 2010 normal range from 2007 to 2012. In 2013 and 2016, taxonomic richness in the FF areas was at or above the upper limit of the 2007 to 2010 normal range, but within the 2013 normal ranges. No significant increases or decreases in richness were observed in the FF areas (Figure 7-2). Higher taxonomic richness was observed at most stations in the MF3 area and at most stations in the FF areas in 2013 (Figure 7-1). In 2004, taxonomic richness at stations in the FF1 and FFA areas was not significantly different than that in the NF area, but in 2008 taxonomic richness in the NF area was significantly greater than in the FF areas (Table 7-13). In 2012, only the NF area was significantly greater than the FFA area (Table 7-13). Taxonomic richness did not differ between 2013 and 2016 in the NF area (Table 7-14).

Temporal trends in total phytoplankton biomass differed among sampling areas in Lac de Gras between 2002 and 2012 (Figures 7-1 and 7-4; Table 7-12). Phytoplankton biomass increased in the NF area between 2003 and 2010, peaking in 2010 and then decreasing until 2015 (Figure 7-1). Between 2015 and 2016, biomass increased in the NF area with significantly greater biomass in 2016 compared to 2013 (Figure 7-3 and Table 7-14). Biomass exceeded the upper limits of the 2007 to 2010 normal range at all or most stations in the NF area from 2006 to 2012. From 2011 to 2013, phytoplankton biomass decreased, and in 2013 and 2014 phytoplankton biomass was within or near the upper limit of the 2007 to 2010, 2013 and "adjusted" 2013 normal ranges. In 2015, stations in the NF area were at or below the lower limit of the 2007 to 2010 normal range but within the 2013 and "adjusted" 2013 normal ranges. With the increase between 2015 and 2016, biomass at stations in the NF area was generally above the upper limit of the normal ranges, with the exception of one station.

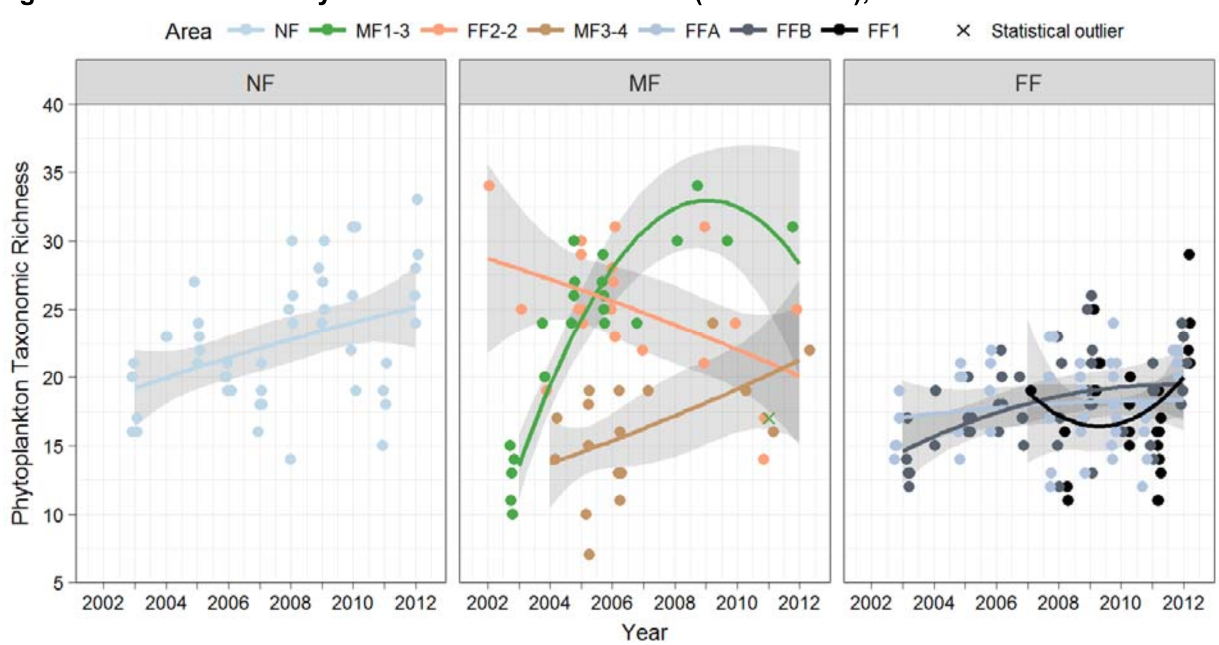
Biomass at the majority of stations in the NF1, MF2-FF2, and MF3 areas followed a similar pattern to that seen in the NF area (Figures 7-1 and 7-4). Similarly, biomass in the FF areas followed comparable trends to the NF and MF areas, but stations in the FF areas were generally within the 2007 to 2010, 2013 and "adjusted" 2013 normal ranges (Figure 7-1). In 2004, phytoplankton biomass at stations in the FF1 and FFA areas was not significantly different than that in the NF area, but in 2008 biomass in the NF area was significantly greater than in the FF areas. In 2012, the NF area biomass was significantly greater than biomass in the FFA and FFB areas (Table 7-13).

Figure 7-1 Total Phytoplankton Biomass and Taxonomic Richness (Genus-level), 2002 to 2016



Note: NF = near-field; MF = mid-field; FF = far-field; LDS = Lac du Sauvage.

Figure 7-2 Trend Analysis for Taxonomic Richness (Genus-level), 2002 to 2012



Note: NF = near-field; MF = mid-field; FF = far-field; grey ribbons are 95% confidence intervals.

Figure 7-3 Trend Analysis for Phytoplankton Variables in the Near-Field Area, 2013 to 2016

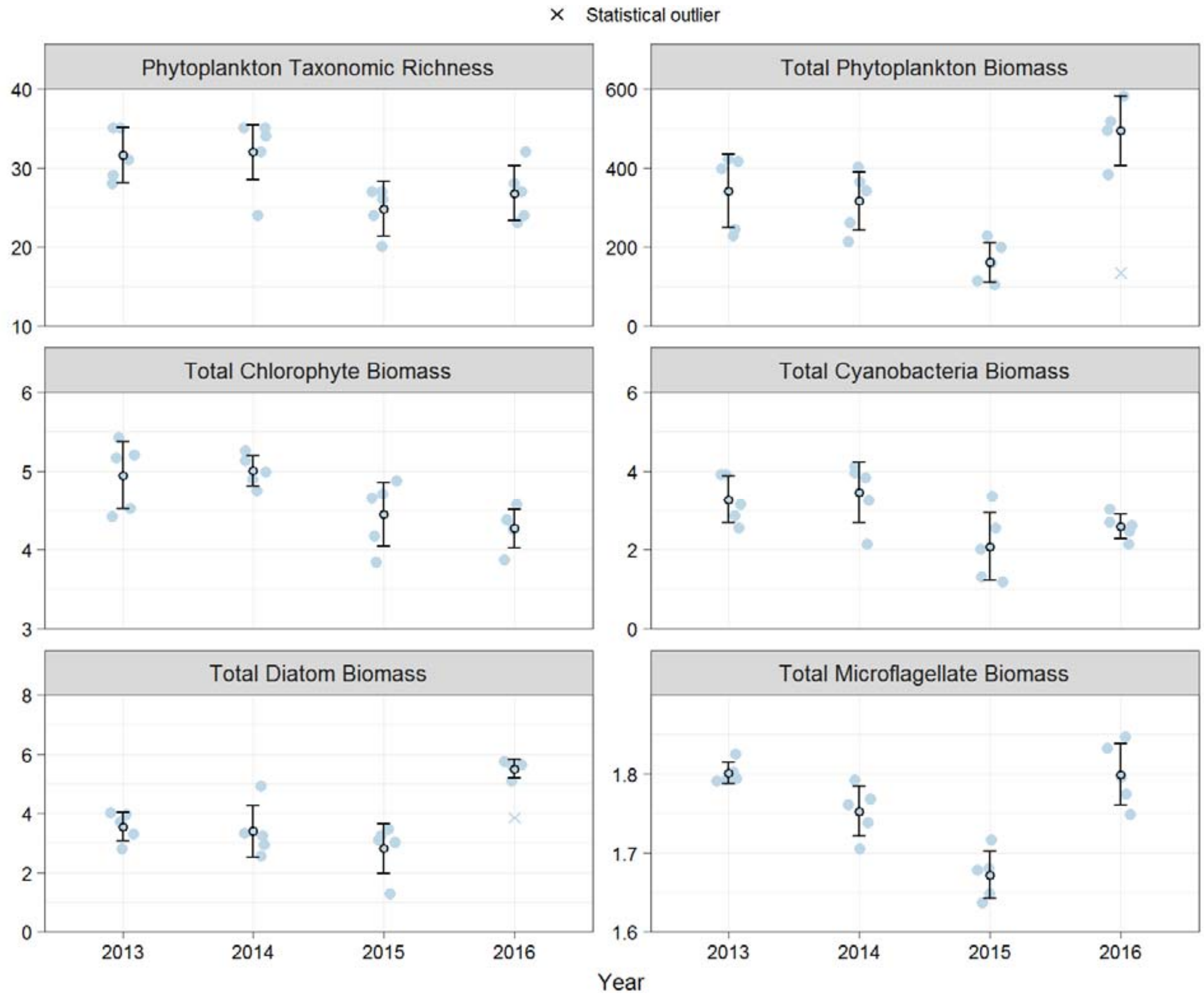


Table 7-12 Significance of Phytoplankton Fixed Effects Models, 2002 to 2012

Variable	Coefficient	Numerator DF	Denominator DF	F-value	*P*-value
Phytoplankton taxonomic richness	Area	6	19	2.90	0.035
	Year	1	200	0.10	0.750
	$Year^2$	1	200	0.10	0.750
	Area × Year	6	200	2.89	**0.010**
	Area × $Year^2$	6	200	2.89	**0.010**
Total phytoplankton biomass	Area	6	19	2.30	0.031
	Year	1	198	10.84	0.001
	$Year^2$	1	198	10.85	0.001
	Area × Year	6	198	2.99	**0.008**
	Area × $Year^2$	6	198	2.99	**0.008**
Total microflagellate biomass	Area	6	19	1.62	0.197
	Year	1	208	32.26	<0.001
	Area × Year	6	208	1.62	0.144
Total diatom biomass	Area	6	19	3.27	0.022
	Year	1	206	0.53	0.468
	Area × Year	6	206	3.27	**0.004**
Total cyanobacteria biomass	Area	6	19	4.38	0.006
	Year	1	208	41.41	<0.001
	Area × Year	6	208	4.38	**<0.001**
Total chlorophyte biomass	Area	6	19	2.42	0.066
	Year	1	199	16.38	<0.001
	$Year^2$	1	199	16.42	<0.001
	Area × Year	6	199	2.41	**0.028**
	Area × $Year^2$	6	199	2.41	**0.028**

Notes: **Bolded** values indicate significance on the interaction term at P <0.05; DF = degrees of freedom; F-value = F-statistic; *P*-value = probability value.

Table 7-13 Multiple Comparisons of Areas with Parabolic Trends in 2004, 2008 and 2012

Variable	Year	Area						
		NF	MF1-3	FF2-2	MF3-4	FF1	FFB	FFA
Phytoplankton taxonomic richness	2004	B	ab	c	a	abc	a	ab
	2008	B	c	b	ab	a	a	a
	2012	B	ab	ab	ab	ab	ab	a
Total phytoplankton biomass	2004	Bc	ab	c	a	abc	a	ab
	2008	B	b	b	a	a	a	a
	2012	Cd	bcd	d	abcd	abc	a	ab
Total chlorophyte biomass	2004	Ab	a	b	a	ab	ab	ab
	2008	Ab	b	ab	ab	ab	a	a
	2012	A	a	a	a	a	a	a

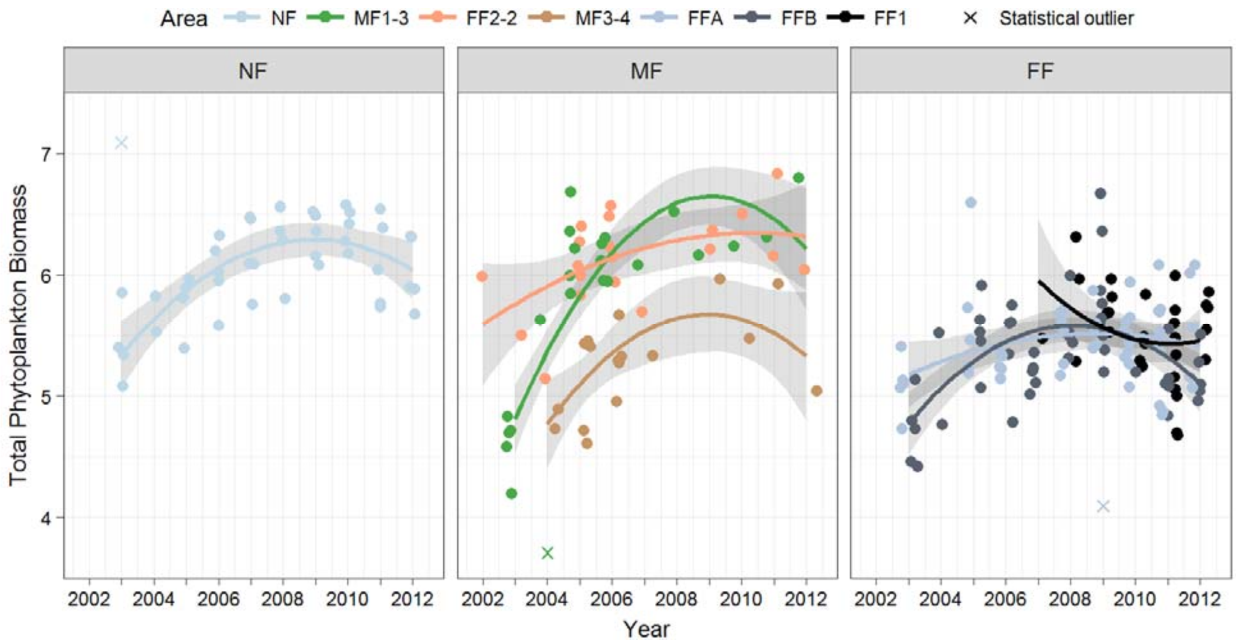
Notes: The different letters designate areas significantly different from on another at a *P*-value of 0.05; NF= near-field; MF = mid-field; FF = far-field.

Table 7-14 Multiple Comparisons in the NF Area between 2013 and 2016

Variable	2013	2016
Phytoplankton taxonomic richness	a	a
Total phytoplankton biomass	a	b
Total microflagellate biomass	a	a
Total diatom biomass	a	b
Total cyanobacteria biomass	b	a
Total chlorophyte biomass	b	a

Notes: The different letters designate areas significantly different from on another at a *P*-value of 0.05; NF= near-field; MF = mid-field; FF = far-field.

Figure 7-4 Trend Analysis for Total Phytoplankton Biomass, 2002 to 2012



Note: NF = near-field; MF = mid-field; FF = far-field; grey ribbons are 95% confidence intervals.

7.3.2.1.2 Biomass of Major Phytoplankton Groups

Temporal trends in the biomass of the major phytoplankton groups differed among sampling areas in Lac de Gras between 2002 and 2012, with the exception of microflagellate biomass, which increased in most areas between 2002 and 2012 (Figures 7-5 and 7-6; Tables 7-13 and 7-15). Microflagellates and cyanobacteria comprised a large proportion of the relative biomass in the NF, MF and FF areas between 2007 and 2016 and were often the dominant groups (Figures 7-5 and 7-7).

Microflagellate biomass demonstrated a peak in the NF area in 2007, and in 2011 along in the MF2-FF2 transect area; no peak was observed in the FF areas (Figure 7-5). The NF area has seen a decline in microflagellate biomass since 2010, with the majority of stations falling within the 2007 to 2010 normal range in 2013, and remaining within that normal range from 2013 to 2016. From 2013 to 2016, microflagellate biomass was at or above the upper limit of the 2013 and “adjusted” 2013 normal ranges, with the exception of in 2015, when microflagellate biomass was at or below the lower limit of the 2013 normal range but within the “adjusted” 2013 normal range.

Microflagellate biomass in the NF area did not differ between 2013 and 2016 (Table 7-14). Overall, microflagellate biomass has increased significantly from 2002 to 2012 (Table 7 15). Along In the MF1, MF2-FF2 and MF3 transect areas, stations were generally near the upper limit or above the normal ranges from 2007 to 2011. In 2012, 2013 and 2016, stations along the MF transect areas were generally within the 2007 to 2010 normal range, but generally above the 2013 and “adjusted” 2013 normal ranges. Despite an increasing trend in the FF areas and a significant increase in the FFA area (Table 7-14), biomass generally remained within the 2007 to 2010, 2013 and “adjusted” 2013 normal ranges between 2002 and 2016, but was above the 2013 normal range (Figures 7-5 and 7-6). Relative microflagellate biomass was generally within or below the lower limits of the 2013 normal range in all areas, but within the 2007 to 2010 and “adjusted” 2013 normal ranges, with a few exceptions (Figure 7-5).

Cyanobacteria biomass increased significantly in the NF, FFA and FFB areas, and at the MF1-3, FF2-2 and MF3-4 stations between 2002 and 2012 (Figure 7-8 and Table 7-13), but only a slight, non-significant decline was observed in the NF area between 2013 and 2016 (Figure 7-4 and Table 7-14). Cyanobacteria biomass peaked in the NF area in 2010 and declined thereafter (Figure 7-7). Some MF stations have seen an increase in cyanobacteria biomass over time, with many stations above the 2007 to 2010 normal range from 2010 to 2013; however, in 2016 all stations were within both normal ranges (Figure 7-7). Relative cyanobacteria biomass fluctuated within the normal range in all areas between 2007 and 2012 (Figure 7-9), but a general decrease in the relative proportion of cyanobacteria was observed between 2011 and 2016. From 2013 to 2016, relative cyanobacteria biomass was below or within the lower limits of both of the 2007 to 2010 and 2013 normal ranges, but within the “adjusted” 2013 normal range.

Temporal trends during 2002 to 2012 in diatom biomass differed among sampling areas (Table 7-12). In the NF and FF areas, diatom biomass was generally within the 2007 to 2010 normal range between 2007 and 2015, but generally above the considerably narrower 2013 and “adjusted” 2013 normal ranges (Figure 7-9). In 2016, diatom biomass increased to above both the normal ranges at four out of five stations in the NF area (Figure 7-9). A significant difference in biomass between 2013 and 2016 was observed in the NF area (Table 7 13). A significant decreasing trend in diatom biomass was observed at the FF2-2 station, and a significant increasing trend was observed at the MF1-3 station between 2002 and 2012 (Figure 7-10; Table 7-15). Generally, stations along the MF1 and MF3 transect areas from 2007 to 2013 were within the 2007 to 2010 normal range, but above the 2013 and “adjusted” 2013 normal ranges. In

2016, most stations in the MF1 and MF2-FF2 areas were above or within the upper limits of the 2007 to 2010 normal range, while stations in the MF3 area were within the 2007 to 2010 normal range. The relative biomass of diatoms in the NF, MF and FF areas, has generally been within the 2007 to 2010 normal range, with the exception of relative diatom biomass in 2016, which was above the 2007 to 2010 normal range at most stations in the NF area (Figure 7-9). Between 2013 and 2016, diatom biomass has generally been above the 2013 and "adjusted" 2013 normal range in all areas.

Temporal trends during 2002 to 2012 in chlorophyte biomass differed among sampling areas (Table 7-12). Chlorophyte biomass was generally within the normal range in the NF, MF and FF areas between 2002 and 2012 (Figure 7-11). The parabolic models fitted to the 2002 and 2012 data show declines in chlorophyte biomass in all sampling areas (Figure 7-12). Biomass was similar in the NF and FF areas in 2004, 2008 and 2012 (Table 7-13); however, there were significantly fewer chlorophytes in 2016 compared to 2013 in the NF area (Table 7-14), although chlorophyte biomass at most stations between 2013 and 2016 fell within the 2013 and "adjusted" 2013 normal ranges (Figure 7-11).

The biomass of dinoflagellates has generally remained within the 2007 to 2010, 2013 and "adjusted" 2013 normal ranges, with excursions above the normal range in 2007, 2011, 2012 and in 2016, for the 2007 to 2010 and 2013 normal ranges but not the "adjusted" 2013 normal range (Figure 7-13). Trend analyses were not conducted for this group because of the high percentage of zero values in the dataset.

Overall, the relative biomass of diatoms, chlorophytes, and dinoflagellates demonstrated few patterns between 2002 and 2012. The time series plots suggest that the community composition (based on major groups) is not undergoing a change with time in Lac de Gras. The relative biomass of chlorophytes and dinoflagellates has remained within the 2007 to 2010, 2013, and "adjusted" 2013 normal ranges in the NF area, with a few exceptions between 2007 and 2016.

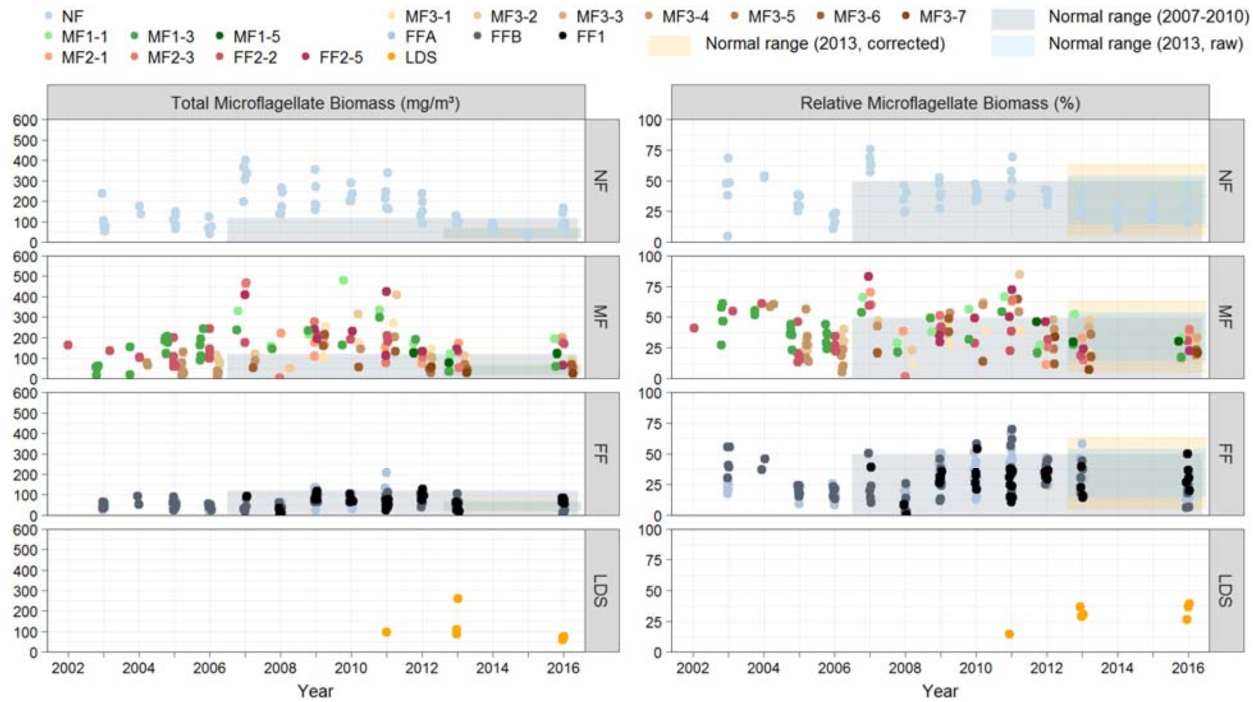
In Lac du Sauvage, phytoplankton taxonomic richness was similar among stations and years from 2011 to 2016 (Figure 7-1). Total phytoplankton biomass has shown a decreasing trend at the stations sampled. No apparent changes in microflagellate, chlorophyte or diatom biomass were observed among years, but the relative proportions of these groups has varied. Microflagellate and chlorophyte relative biomass appears to have increased (Figures 7-5 and 7-11), while the relative proportion of diatoms has not changed over time. Cyanobacteria and dinoflagellate biomass, and the relative proportions of these groups appeared to have decreased since 2011 (Figures 7-7 and 7-13).

Table 7-15 Estimated Significance of Difference of Linear slopes from zero

Variable	Area						
	NF	MF1-3	FF2-2	MF3-4	FF1	FFB	FFA
Total microflagellate biomass	**0.038** ↑	**0.002** ↑	0.230	**0.050** ↑	0.129	0.097	**<0.001** ↑
Total cyanobacteria biomass	**<0.001** ↑	**<0.001** ↑	**<0.001** ↑	**0.018** ↑	0.630	**0.014** ↑	**0.008** ↑
Total diatom biomass	0.260	**0.002** ↑	**0.033** ↓	0.482	0.341	0.450	0.087

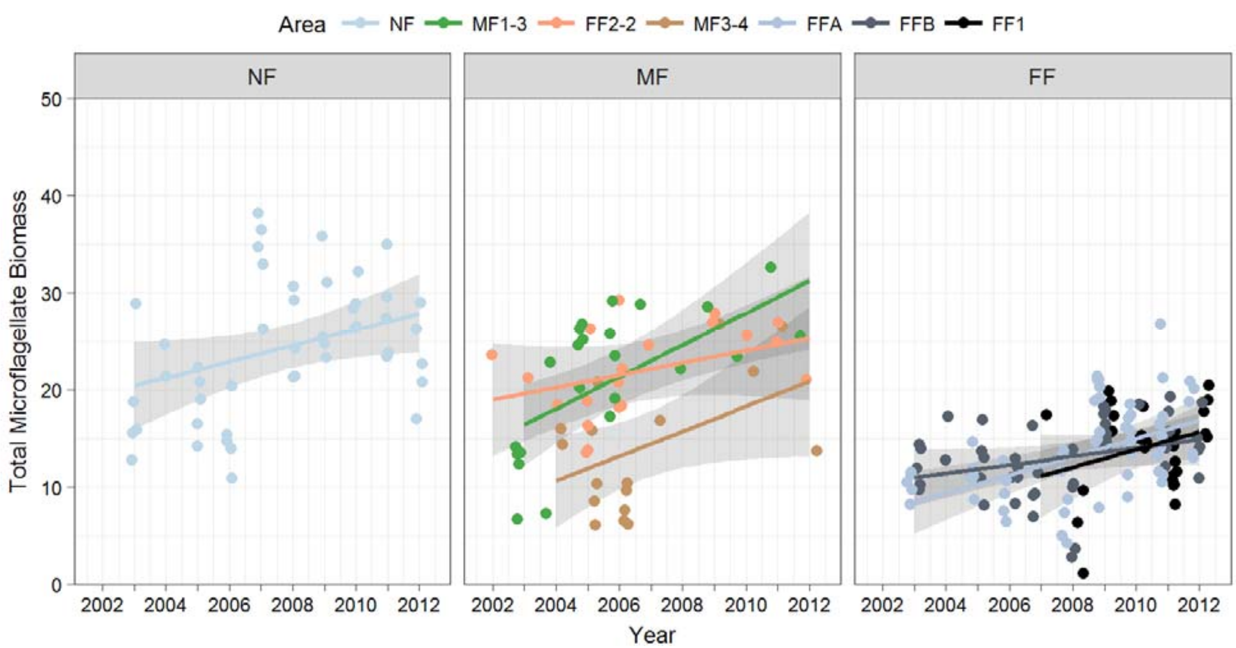
Notes: **Bolded** values indicate significance. The arrow indicates direction of trend.
NF= near-field; MF = mid-field ; FF = far-field; <= less than.

Figure 7-5 Total Microflagellate Biomass and Relative Microflagellate Biomass, 2002 to 2016



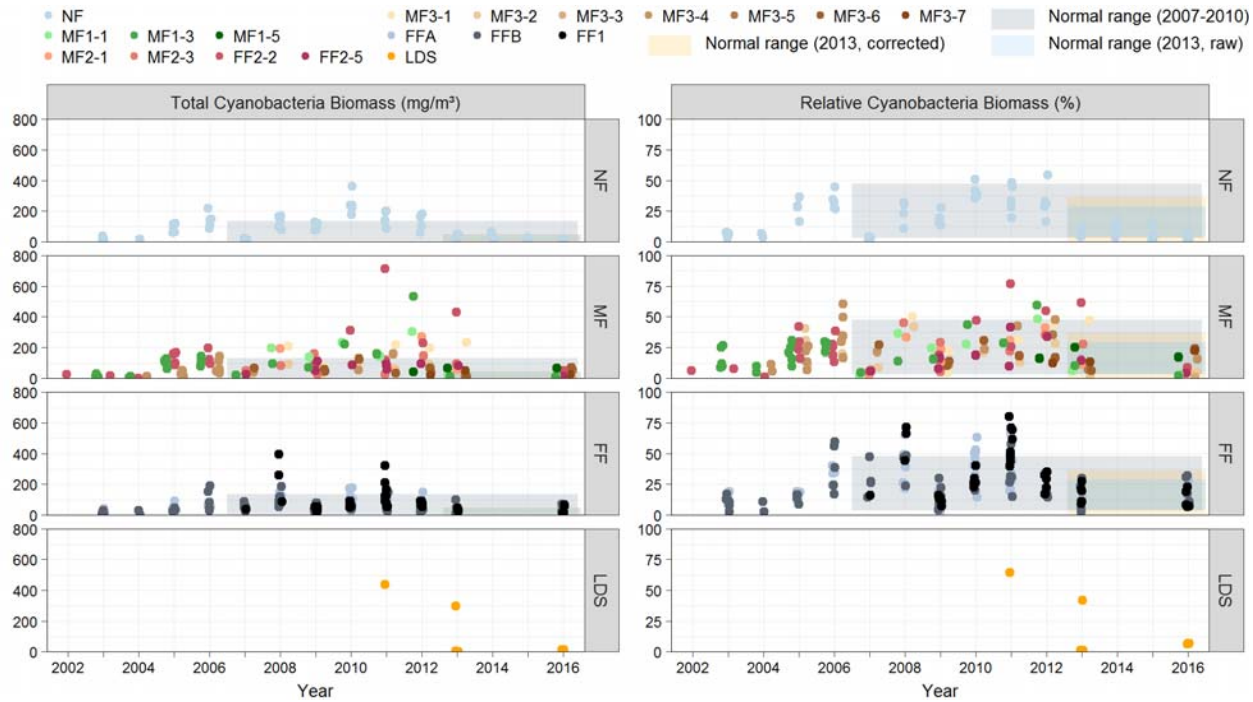
NF = near-field; MF = mid-field; FF = far-field; LDS = Lac du Sauvage.

Figure 7-6 Trend Analysis for Total Microflagellate Biomass, 2002 to 2012



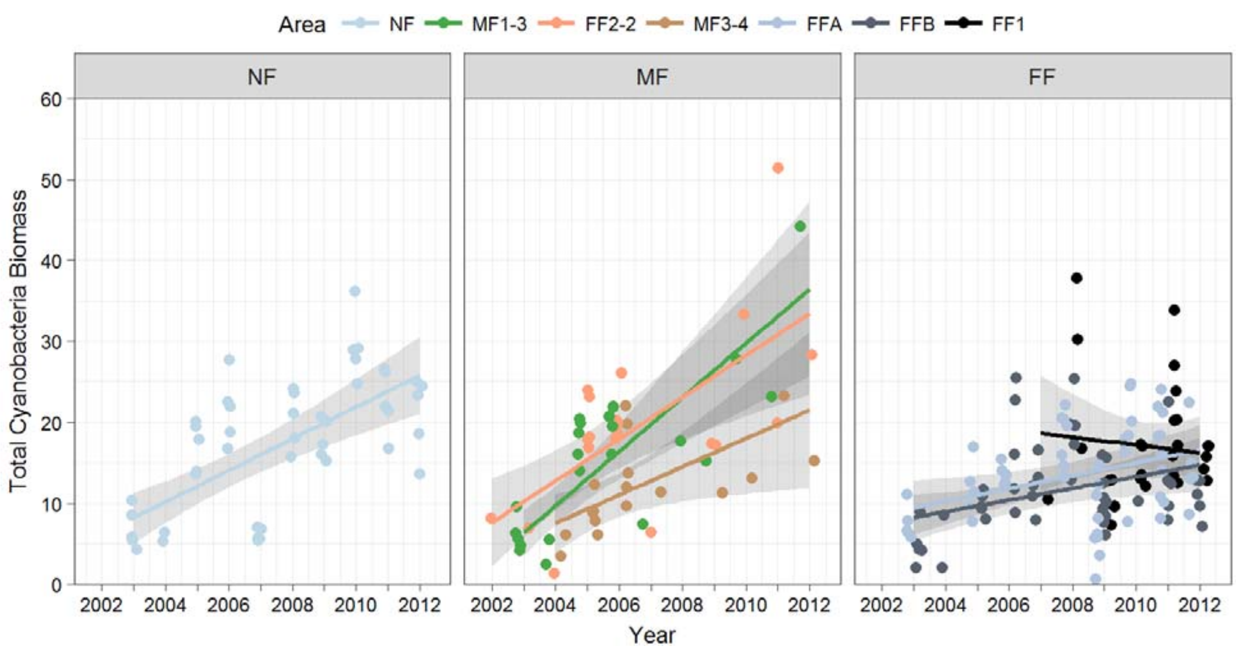
NF = near-field; MF = mid-field; FF = far-field; grey ribbons are 95% confidence intervals.

Figure 7-7 Total Cyanobacteria Biomass and Relative Cyanobacteria Biomass, 2002 to 2012



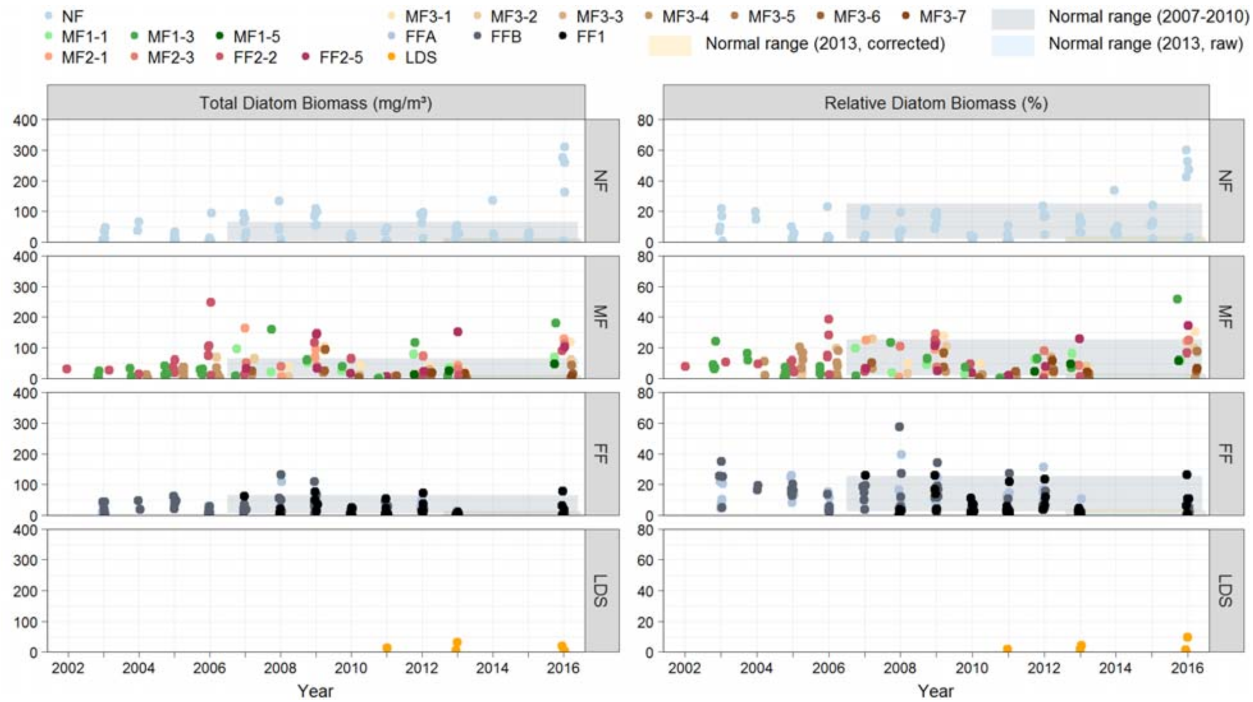
NF = near-field; MF = mid-field; FF = far-field; LDS = Lac du Sauvage.

Figure 7-8 Trend Analysis for Total Cyanobacteria Biomass, 2002 to 2012



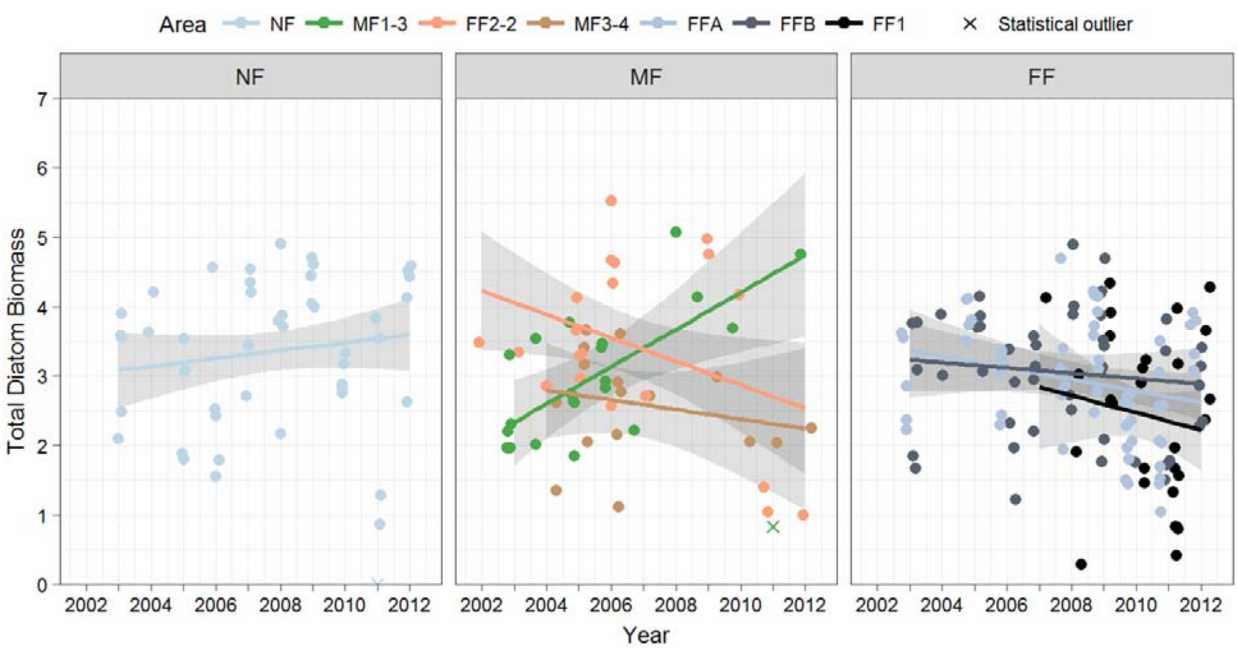
NF = near-field; MF = mid-field; FF = far-field; grey ribbons are 95% confidence intervals.

Figure 7-9 Total Diatom Biomass and Relative Diatom Biomass, 2002 to 2016



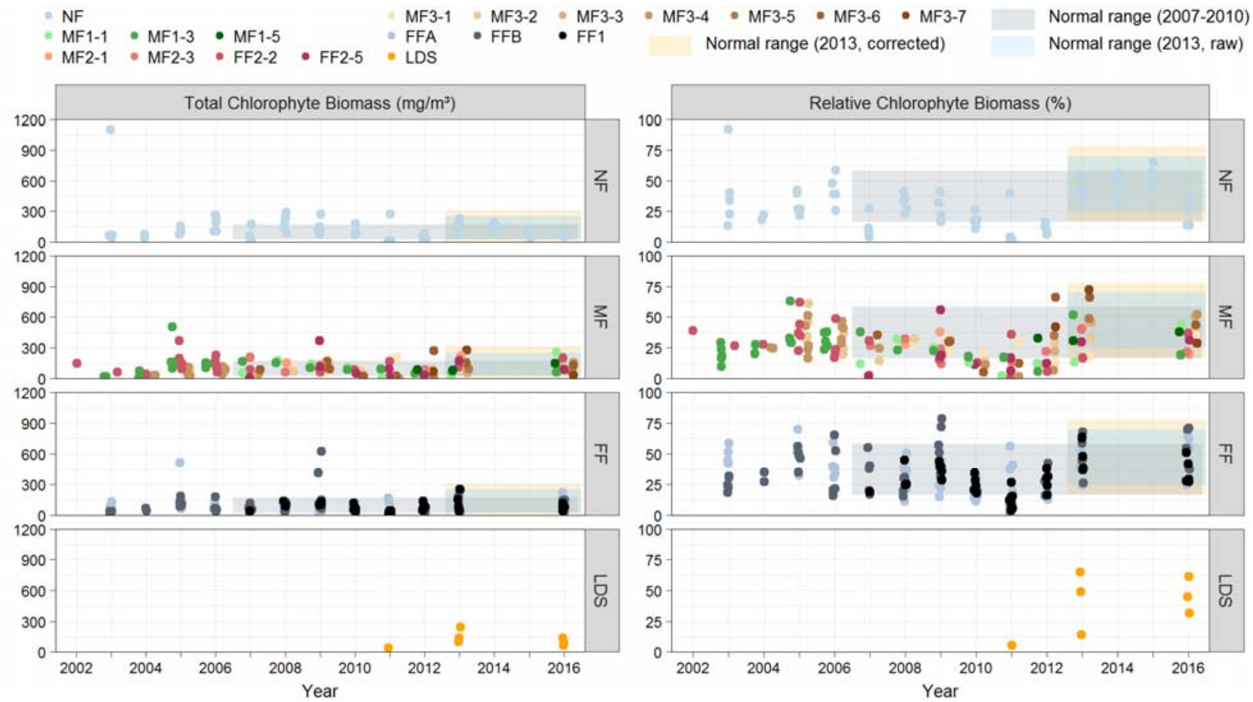
NF = near-field; MF = mid-field; FF = far-field; LDS = Lac du Sauvage.

Figure 7-10 Trend Analysis for Total Diatom Biomass, 2002 to 2012



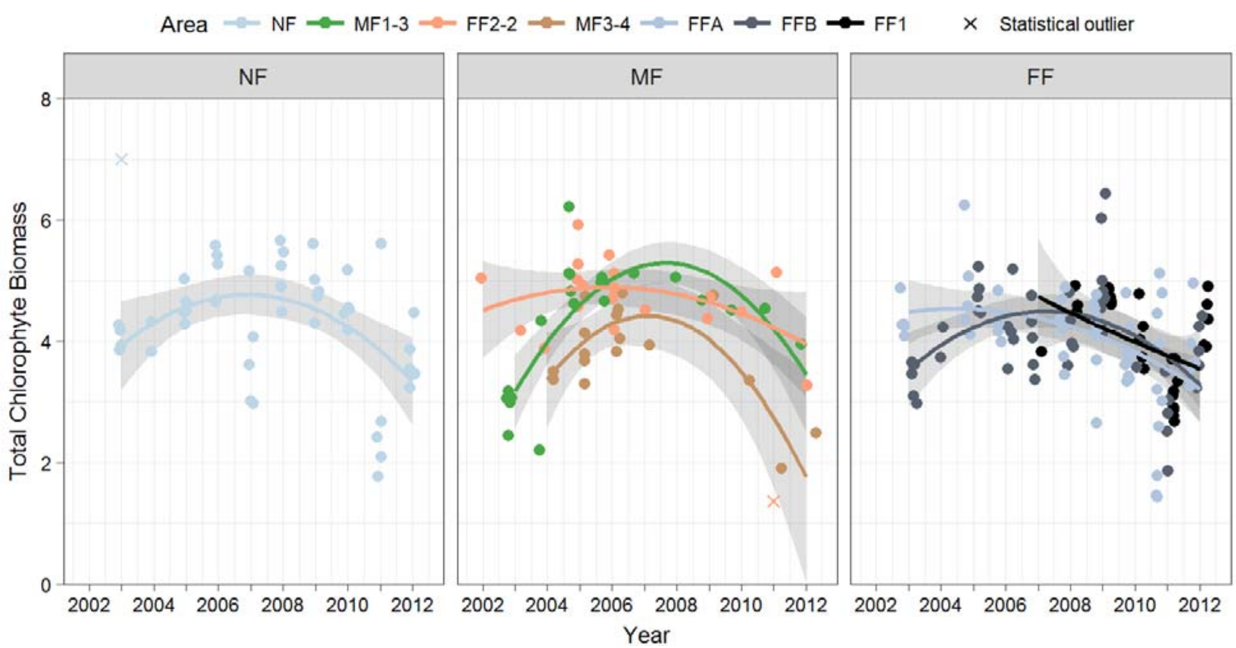
NF = near-field; MF = mid-field; FF = far-field; grey ribbons are 95% confidence intervals.

Figure 7-11 Total Chlorophyte Biomass and Relative Chlorophyte, 2002 to 2016



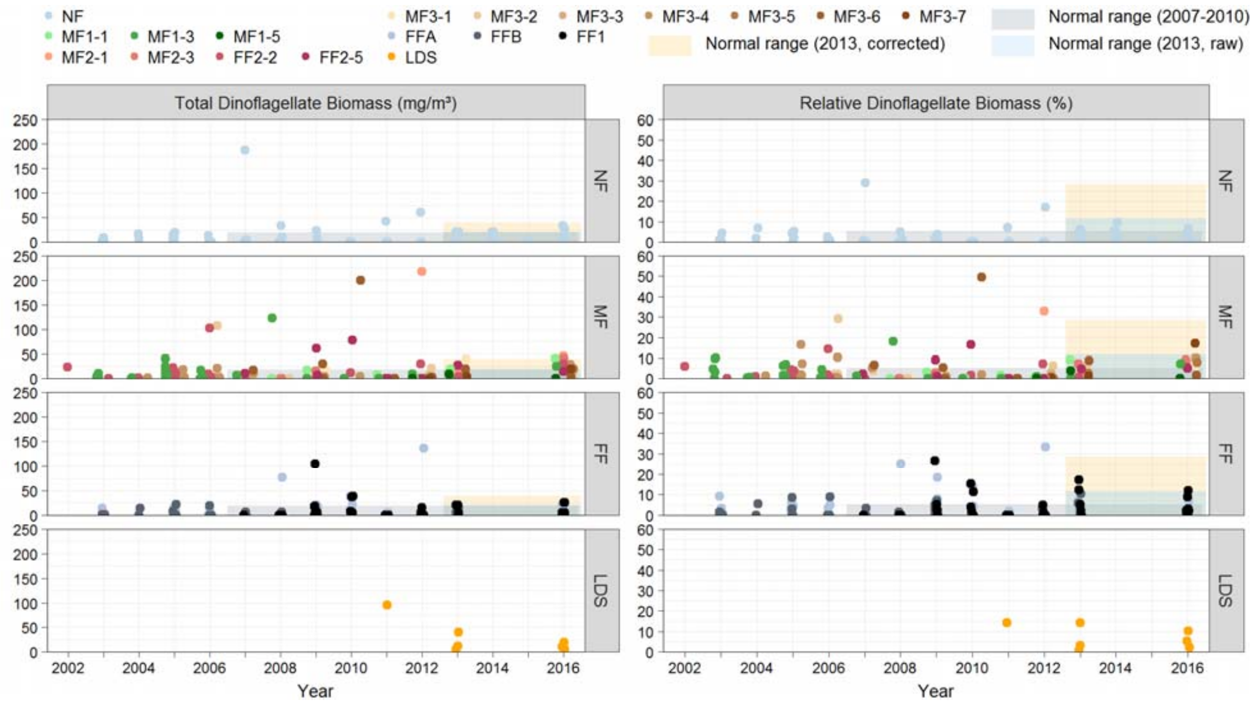
NF = near-field; MF = mid-field; FF = far-field; LDS = Lac du Sauvage.

Figure 7-12 Trend Analysis for Total Chlorophyte Biomass, 2002 to 212



NF = near-field; MF = mid-field; FF = far-field; grey ribbons are 95% confidence intervals.

Figure 7-13 Total Dinoflagellate Biomass and Relative Dinoflagellate, 2002 to 2016



NF = near-field; MF = mid-field; FF = far-field; LDS = Lac du Sauvage.

7.3.2.1.3 Phytoplankton Normal Range Evaluation

In 2013, a number of data quality issues were observed with previous taxonomy data. Therefore, a switch in taxonomists occurred, from Bio Limno to Eco-Logic. Differences in the measures of abundance and biomass from the two taxonomists, based on an quality control evaluation of duplicate samples sent to both taxonomist in 2013, were noted (Golder 2016b). These differences were large enough to warrant the use of the 2013 FF data to estimate new normal ranges for phytoplankton. In general, a comparison of the normal ranges showed that abundance and biomass estimates from the previous taxonomist were greater than those from the new taxonomist, in part because the previous normal ranges were estimated based on four years of data, whereas the 2013 normal ranges were based on a single year of data (Section 7.2.3.1).

In the *2011 to 2013 Aquatic Effects Re-evaluation Report* (Golder 2016a), a 2013 normal range was used for the 2013 data comparisons to the normal range. This approach raised concerns during a public review of the *AEMP Reference Conditions Report Version 1.1* (Golder 2015a). The concern was that the data collected in 2013 may have already been influenced by mine activity and should not be used to simulate reference conditions. However, neither nutrient-related nor toxicological effects were observed in 2013 in the FF areas across biological monitoring components, which implies that 2013 data could be used to estimate normal ranges without impairing the ability of the AEMP to detect Mine-related effects. It was also noted, during the review, that it is unlikely that a change in taxonomist would significantly affect biomass calculations of the major groups of phytoplankton. However, clear group-level differences have been observed between the 2007 to 2012 and 2013 and 2016 datasets, most notably in the abundance and biomass of cyanobacteria, microflagellates and diatoms. To estimate the appropriate normal range moving forward, the WLWB has requested that as part of the *2014 to 2016 Aquatic Effects Re-evaluation Report*, an assessment of the reference conditions for the phytoplankton variables occur.

In general, the 2007 to 2010 normal range is wider for most phytoplankton variables, while the 2013 normal range provides a more narrow and conservative normal range, in part because only one year of data was used. The “width” of the normal ranges (i.e., upper limit minus lower limit) varies considerably between the 2007 to 2010 and 2013 normal ranges (Table 7.5). This is as expected, because year-to-year variation is incorporated into the 2007 to 2010 normal ranges, but not in the 2013 normal ranges. Five of the 12 variables (i.e., diatom, microflagellate and cyanobacteria biomass, and relative diatom and cyanobacteria biomass) have 2013 normal ranges that are approximately two to seven times narrower than the 2007 to 2010 normal ranges. This implies that the 2013 normal ranges for these five variables are unlikely to be realistic, and required adjustment to account for year-to-year variability (i.e., the “adjusted” 2013 normal range). The “adjusted” 2013 normal range was still narrow and continues to provide a conservative estimate of the normal range; however, adjustment for year-to-year variability has increased the “width” of the normal range to provide more suitable reference condition boundaries based on the appropriate taxonomist data.

Most phytoplankton variables generally fell within the 2007 to 2010, 2013 and “adjusted” 2013 normal ranges in the FF areas in 2013 and 2016, with the exception of taxonomic richness (Figures 7-1 to 7-13). For taxonomic richness, the 2007 to 2010 normal range is inappropriate for 2013 to 2016 comparisons, as most data fall outside the 2007 to 2010 normal range, or within the upper limits of that normal range. For phytoplankton total biomass, the normal ranges are similar, although the 2013 and “adjusted” 2013 normal ranges have lower boundary values than the lower limit of the 2007 to 2010 normal range. The 2007 to 2010, 2013 and “adjusted” 2013 normal ranges were similar for total and relative biomass of chlorophytes and dinoflagellates, while the “adjusted” 2013 normal ranges were “wider”. For microflagellates, cyanobacteria and diatoms, which are the more dominant groups, the normal ranges differed. For each of the three groups, the 2013 normal ranges were narrower. For diatom total and relative biomass, the 2013 and “adjusted” 2013 normal ranges were especially narrow (0 to 9 mg/m^3 and 0% to 4%, respectively) as a result of low diatom biomass in the FF areas in 2013. Most stations in all areas were outside these normal ranges for diatoms biomass between 2013 and 2016.

Overall, based on the clear differences in the data sets produced by the two different taxonomists, the “adjusted” 2013 normal range (refered to going forward as the “ 2013 normal range”) is recommended for comparisons from 2013 onwards.

7.3.2.1.4 Phytoplankton Community Structure

The two dimensional nMDS and mMDS configurations for phytoplankton biomass in the NF and FF areas from 2003 to 2012, and from 2013 to 2016 had stress values of 0.14 and 0.16, respectively, indicating a “fair” fit to the original dataset (Figures 7-14 and 7-15). The SIMPROF test (P <0.05) indicated that the level of interpretation of the 40%, 50%, 60%, 70% and 80% clusters is acceptable and the global ANOSIM (R = 0.614, P = 0.001 for 2003 to 2012, and R = 0.31, P = 0.001 for 2013 to 2016) tests indicate that statistical interpretation of the MDS structure is permitted. The 2003 to 2012 ordination plot indicates separation between the NF and FF areas and among-year groupings in terms of phytoplankton community composition. The phytoplankton communities from the NF areas in 2003 to 2007 generally differed from the communities in the FF areas. However, NF area communities between 2008 and 2012 and FF areas between 2009 to 2012 were generally 50% similar (Figure 7-14). Overall, the NF and FF area communities were 40% similar between 2003 and 2012. The 2013 to 2016 ordination plot indicates separation between years, and the NF and FF areas grouped together in 2013 at 70% similarity, separate from the NF and FF areas from 2016, which also grouped together at 70% similarity (Figure 7-15). The 2014 NF area community grouped together with the 2013 cluster at 60% similarity, and the 2015 NF area community grouped with the 2016 cluster at 60% similarity. Overall, all areas were 50% similar. The pair-wise ANOSIM for NF versus FF areas for each year indicated significant differences in 2007, 2011, and 2012 between the NF area and the FF areas ($R > 0.5$, $P < 0.05$), the greatest difference was observed in 2012 ($R = 0.73$; Table 7-16).

The two dimensional nMDS configurations for phytoplankton biomass along each transect from 2003 to 2012 had stress values of 0.19 for the NF-MF1-FF1 transect and 0.18 for the NF-MF2-FF2 and NF-MF3-FFB-FFA transects, indicating a reasonable level of fit to the original datasets (Figures 7-16 to 7-18). From 2013 to 2016 the nMDS configurations for phytoplankton biomass had stress values of 0.11 for the NF-MF1-FF1 transect, 0.09 for the NF-MF2-FF2 transect and 0.17 for the NF-MF3-FFB-FFA transect, indicating a reasonable level of fit to the original datasets (Figures 7-19 to 7-20).

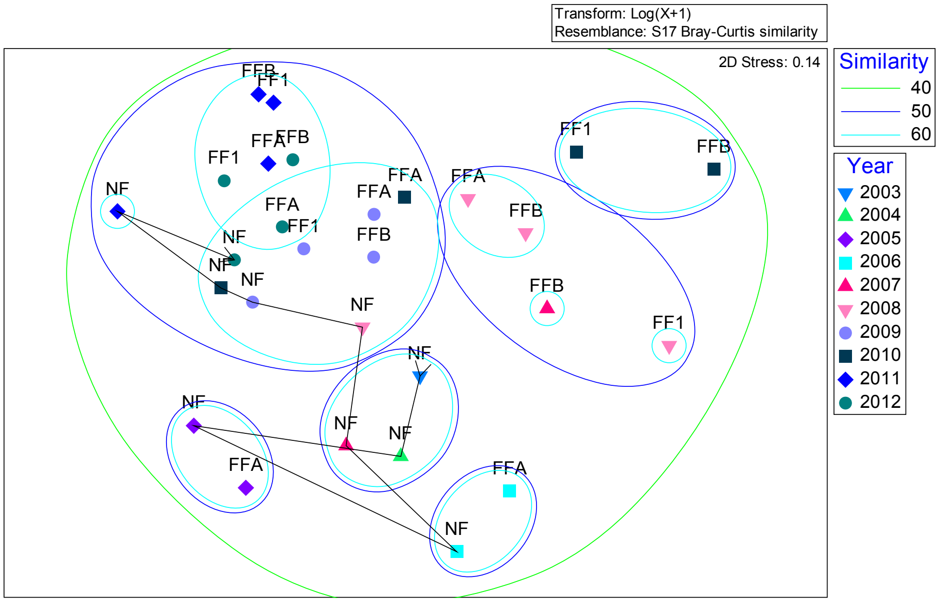
Along the NF-MF1-FF1 transect from 2003 to 2012, all stations and areas grouped together within a 40% similarity ellipse (Figure 7-16). Generally the NF and FF areas and the MF stations from 2009 to 2012 grouped together at 50% similarity. From 2013 to 2016, the MF stations from 2013 grouped together and the MF stations from 2016 grouped together at 60% similarity. The NF areas in all years except 2016 were separate from the 2013 and 2016 clusters, and NF and FF1 communities were similar in 2013 and 2016 (Figure 7-19).

Stations along the NF-MF2-FF2 transect from 2003 to 2012 grouped together within a 20% similarity ellipse, clear area-year or station-year groupings were not observed (Figure 7-17). From 2013 to 2016, the MF2-FF2 and NF stations were grouped by year in each of 2013 and 2016, at 60% similarity (Figure 7-20). This result is consistent with previous observations indicating similar communities throughout the north-eastern part of Lac de Gras.

Stations along the NF-MF3-FFB-FFA transect, which represents the longest gradient in Lac de Gras, were 40% similar, with the exception of MF3-1 and MF3-2 in 2008 which grouped together at 40% similarity. Otherwise, few clear area-year or station-year groupings were observed (Figure 7-19). On the 2013 to 2016 ordination plot, the MF3-FFB-FFA stations grouped along with the NF area in both 2013 and 2016, at 60% similarity (Figure 7-21), indicating limited variation among communities in different areas of the lake within these two years.

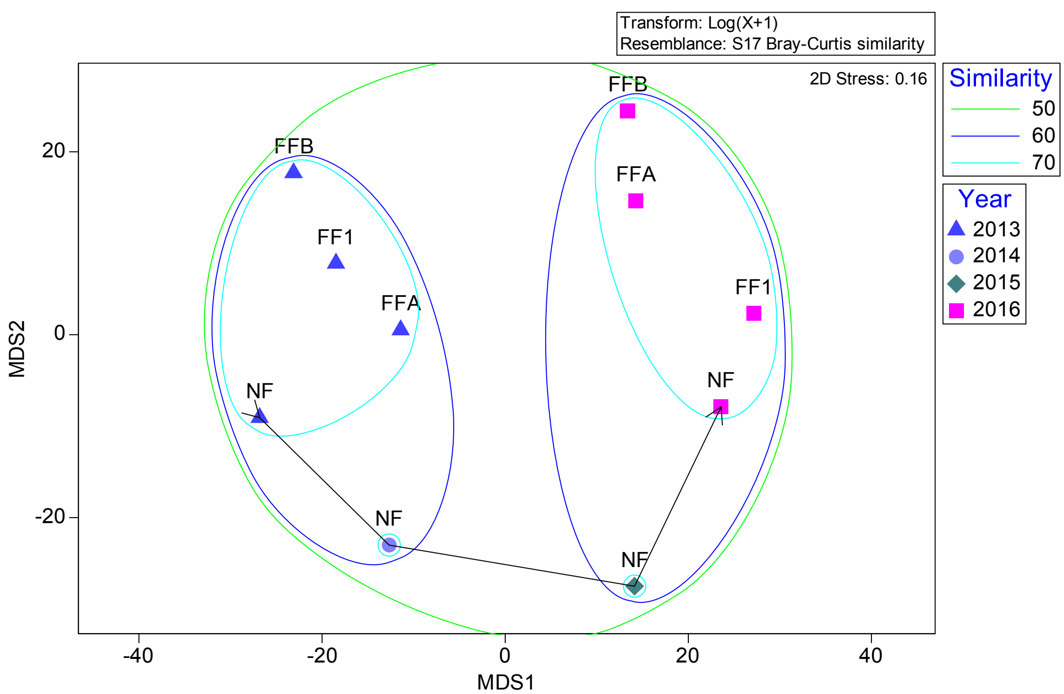
The MDS results indicate that changes over time in phytoplankton community structure have been occurring in the NF area of Lac de Gras, especially between 2003 to 2007, and that conditions in 2016 differed from conditions in 2013 throughout the lake, but the NF and FF areas in these two years were more similar than observed in previous years.

Figure 7-14 Non-metric Multidimensional Scaling of Phytoplankton Biomass in the Near-field and Far-field Areas, 2003 to 2012



NF = near-field; MF = mid-field; FF = far-field.

Figure 7-15 Metric Multidimensional Scaling of Phytoplankton in the Near-field and Far-field Areas, 2013 to 2016



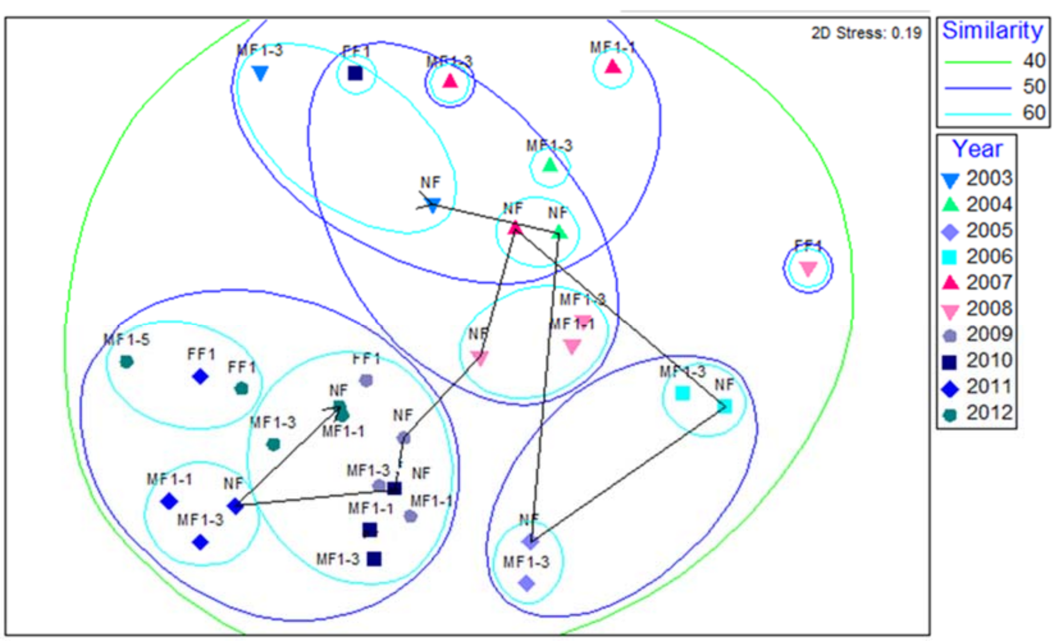
Note: MDS = metric multidimensional scaling axes 1 and 2; NF = near-field; MF = mid-field; FF = far-field.

Table 7-16 Analysis of Similarities (ANOSIM) Tests Between Near-field and Far-field Phytoplankton Communities in Lac de Gras, 2009 to 2016

Year	R statistic	*P-value*
2007	0.55	**0.008**
2008	0.49	**0.001**
2009	0.11	0.196
2010	0.36	**0.001**
2011	0.55	**0.001**
2012	0.73	**0.001**
2013	0.14	0.142
2014	n/a	n/a
2015	n/a	n/a
2016	0.40	**0.001**

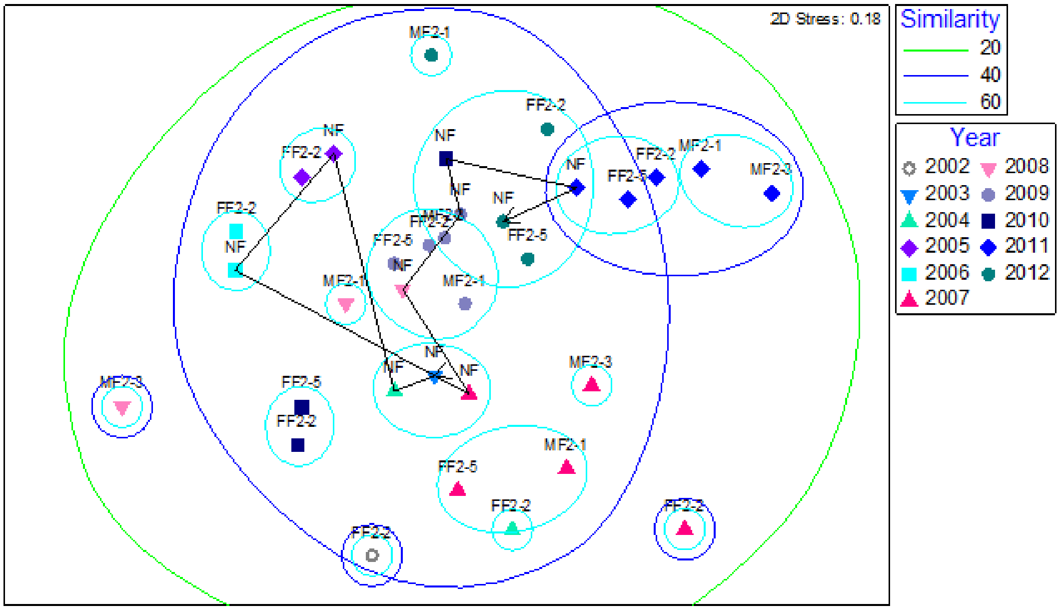
Note: NF vs. FF area comparisons significant with *P*-values <0.05. Bolded values indicate significance.
n/a = not applicable; *P* = probability.

Figure 7-16 Non-metric Multidimensional Scaling of Phytoplankton Biomass along the NF-MF1-FF1 Transect, 2003 to 2012



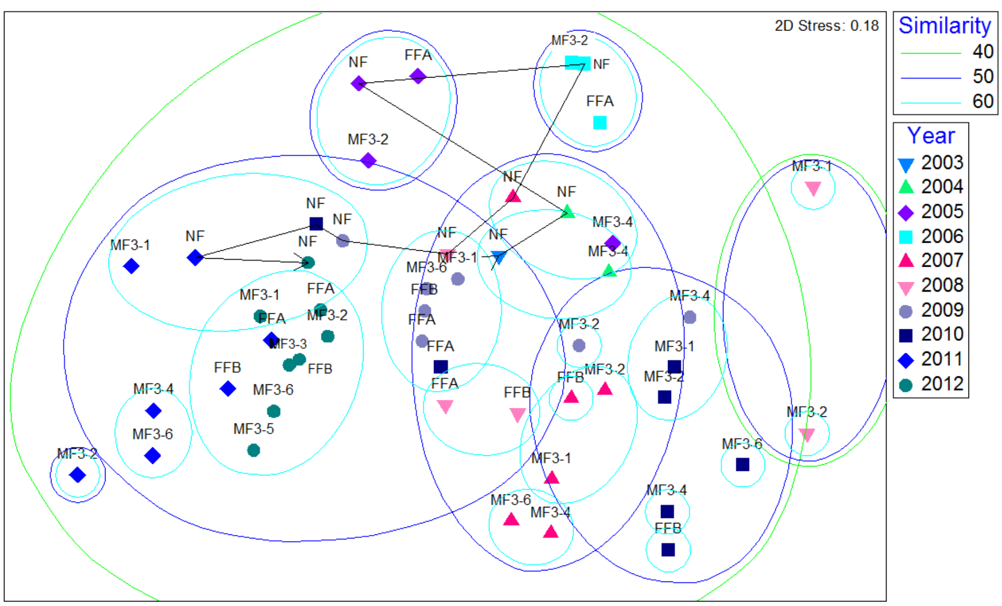
NF = near-field; MF = mid-field; FF = far-field.

Figure 7-17 Non-metric Multidimensional Scaling of Phytoplankton Biomass along the NF-MF2-FF2 Transect, 2002 to 2012



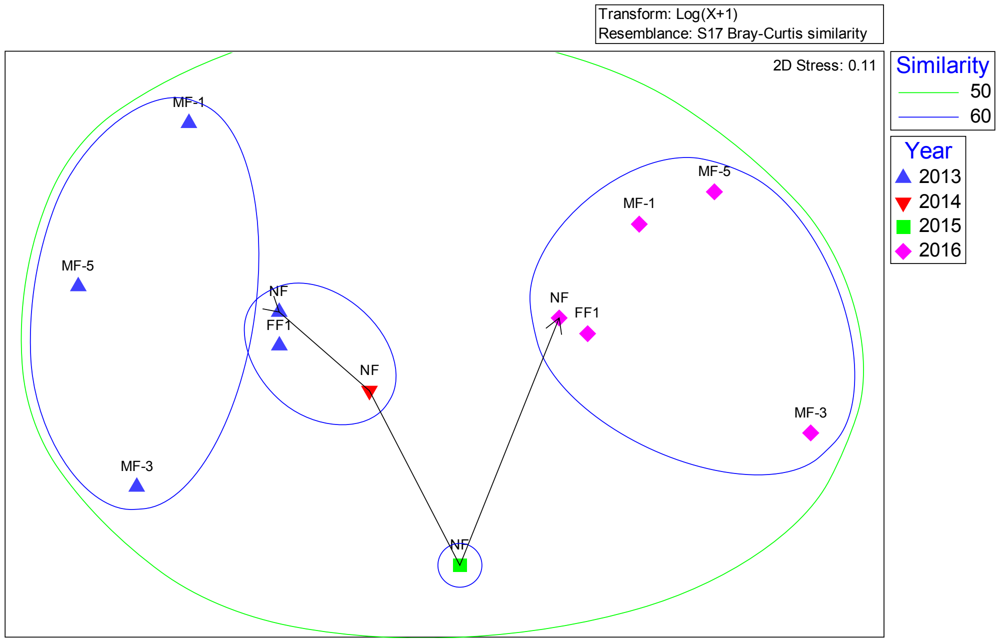
NF = near-field; MF = mid-field; FF = far-field.

Figure 7-18 Non-metric Multidimensional Scaling of Phytoplankton Biomass along the NF-MF3-FFB-FFA Transect, 2003 to 2012



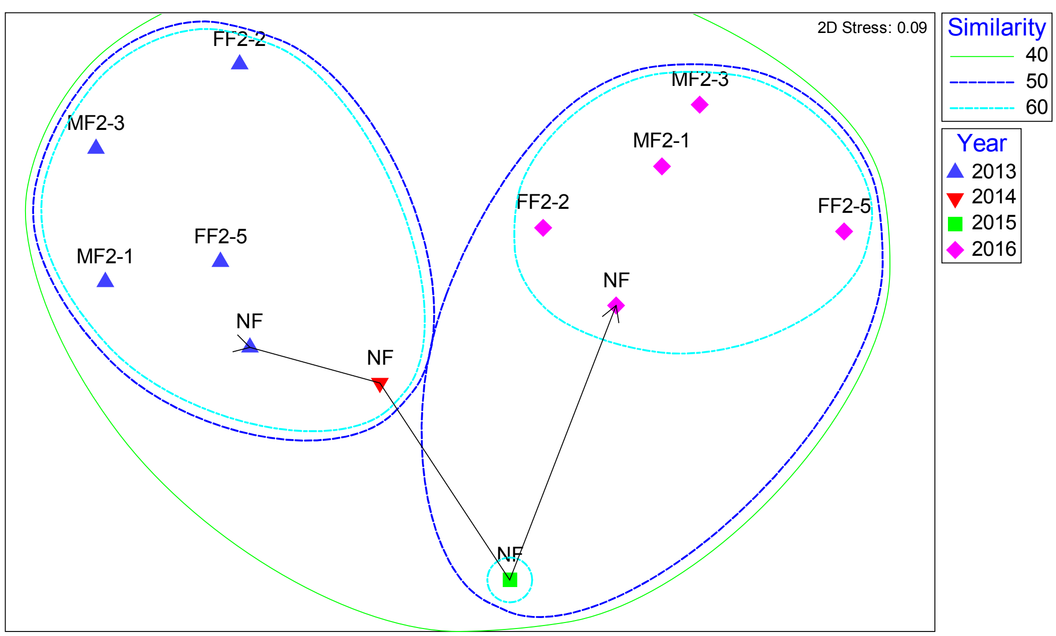
NF = near-field; MF = mid-field; FF = far-field.

Figure 7-19 Non-metric Multidimensional Scaling of Phytoplankton Biomass along the NF-MF1-FF1 Transect in Lac de Gras from 2013 to 2016



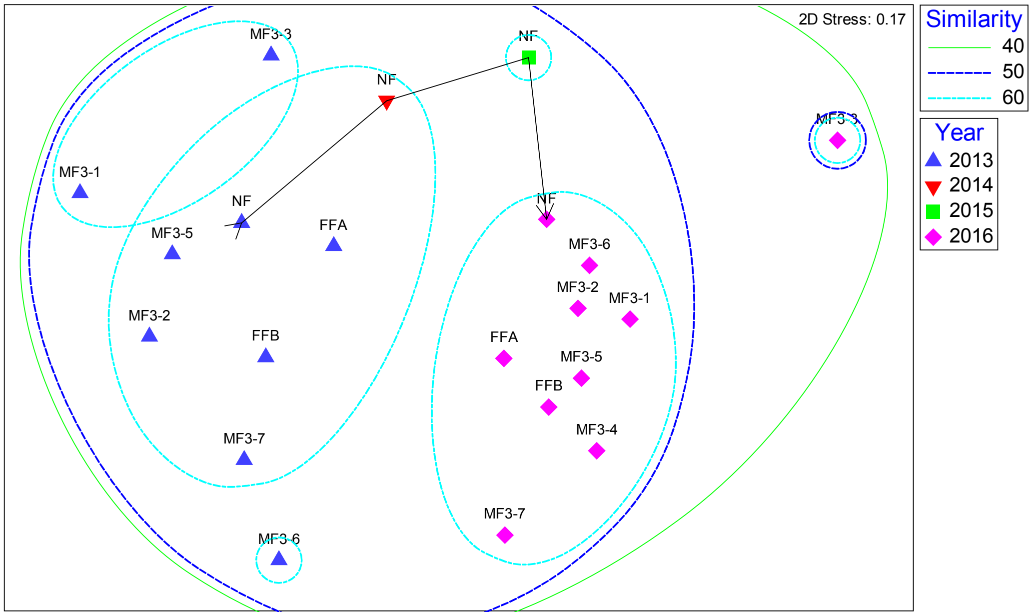
NF = near-field; MF = mid-field; FF = far-field.

Figure 7-20 Non-metric Multidimensional Scaling of Phytoplankton Biomass along the NF-MF2-FF2 Transect in Lac de Gras from 2013 to 2016



NF = near-field; MF = mid-field; FF = far-field.

Figure 7-21 Non-metric Multidimensional Scaling of Phytoplankton Biomass along the NF-MF3-FFB-FFA Transect, 2013 to 2016



NF = near-field; MF = mid-field; FF = far-field.

7.3.2.1.5 Phytoplankton Summary

A linear increase in taxonomic richness was observed in the NF area between 2002 and 2012, but no trends were apparent between 2013 and 2016. From 2002 to 2012, biomass at stations in the NF area was within or just above the upper limit of the normal range, with excursions outside the normal range occurring at individual stations throughout the time series. From 2013 to 2016, richness was within or above the upper limit of the 2007 to 2010 normal range, but within the 2013 normal range. Taxonomic richness at stations in the FF area generally fell within the 2007 to 2010 normal range from 2007 to 2012. In 2013 and 2016, taxonomic richness in the FF areas was at or above the upper limit of the 2007 to 2010 normal range but within the 2013 normal range. No substantial increases or decreases in taxonomic richness were observed in the FF areas.

Phytoplankton biomass in the NF area fluctuated from within the normal ranges (2013 and 2014) to both above (2007 to 2012 and 2016) and below (2015) the normal ranges throughout the time series. Biomass in the FF areas followed comparable trends to the NF and MF areas, but stations in the FF areas were generally within the normal ranges throughout the time series. Temporal trends in the biomass of the major phytoplankton groups differed among sampling areas and among years in Lac de Gras between 2002 and 2012, with the exception of microflagellate and cyanobacteria biomass which increased in most areas between 2002 and 2012. Between 2013 and 2015, cyanobacteria and microflagellate biomass followed a decreasing trend in the NF area; in 2016 microflagellate biomass increased. Microflagellates and cyanobacteria contributed a large proportion of relative biomass in the NF, MF and FF areas between 2007 and 2016, and were often the dominant groups.

The phytoplankton MDS results indicate that changes over time in phytoplankton community structure have been occurring in the NF area of Lac de Gras, especially between 2003 to 2007, and that conditions in 2016 differed from conditions in 2013 throughout the lake, but the NF and FF areas in these two years were more similar than observed in previous years.

7.3.2.2 Zooplankton Community

In the analysis of zooplankton variables from 2008 to 2016, the temporal trends for zooplankton taxonomic richness, and zooplankton, cladoceran, cyclopoid and rotifer biomass were parabolic, while temporal trends for calanoid biomass were linear (Table 7-17). All zooplankton variables were log transformed, with the exception of rotifer biomass, which was inverse square root transformed (Table 7-17).

Table 7-17 Zooplankton AIC Model Selection, 2008 to 2016

Variable	Yeo-Johnson transformation[a]	Parabolic AIC Score	Linear AIC Score
Zooplankton taxonomic richness	0	-381	-378
Total zooplankton biomass	0	203	232
Total Cladocera biomass	0	373	453
Total Calanoida biomass	0	276	261
Total Cyclopoida biomass	0	251	286
Total Rotifera biomass	-0.5	-104	-78

a) See Section 2.4.2.1 for details on transformation approach.

Note: Grey shaded cells indicate the model chosen. AIC = Akaike's information criterion.

7.3.2.2.1 Total Zooplankton Biomass (Based on Enumeration) and

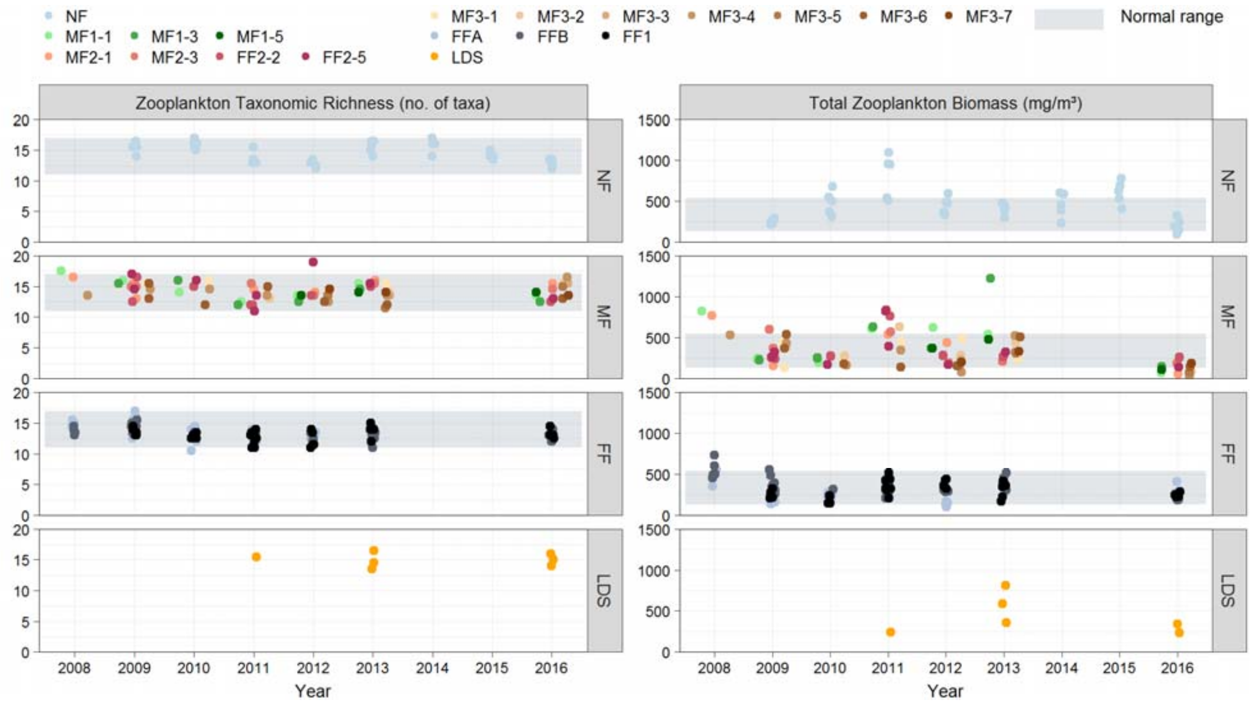
Zooplankton taxonomic richness generally remained within the normal range in the NF and FF areas of Lac de Gras from 2008 to 2016 (Figure 7-22). Trends over time did not differ among the areas or stations included in the analysis (Table 7-18).The majority of the stations in the MF1, MF2-FF2, and MF3 areas remained within the normal range with the exception of FF2-5 in 2012 and MF1-1 in 2007. Declining trends were observed in taxonomic richness in all areas of Lac de Gras (Figure 7-23; Table 7-18). In 2010 and 2013, the NF areas were significantly greater than the FF areas; however in 2016, the NF and FF areas did not differ (Table 7-19).

Temporal trends in total zooplankton biomass differed among sampling areas in Lac de Gras between 2008 and 2016 (Figures 7-22 and 7-24; Table 7-18). Zooplankton biomass has been more variable in the NF and MF areas compared to the FF areas, but generally has remained within the normal range. Excursions in the NF area above the normal range occurred in 2008 and 2011 and below the normal range in 2016 (Figure 7-22). Zooplankton biomass increased in the NF area between 2008 and 2011, peaking in 2011 and then decreasing until 2013. Between 2013 and 2015, biomass increased in the NF area; however, between 2015 and 2016 biomass decreased. Based on the parabolic models fitted to the long-term dataset, zooplankton biomass has declined since 2013 in a number of areas and stations included in the trend analysis, although biomass has been highly variable over time and the trends in the fitted lines are subject to uncertainty.

Zooplankton biomass exceeded the upper limits of the normal range at stations in the MF2-FF2 area in 2008, 2011, 2012 and 2013 and in the MF1 area in 2008, 2009 and 2011; however, in 2016 zooplankton biomass was at or below the lower limit of the normal range (Figure 7-22). In the MF3 area, stations were generally within the normal range, or just below the lower limit from 2008 to 2016. A clear decreasing temporal trend in zooplankton biomass, driven by low biomass in 2016, was observed at the MF3-4 station (Figure 7-23), which may reflect a short-term effect of dike construction in 2015 and 2016.

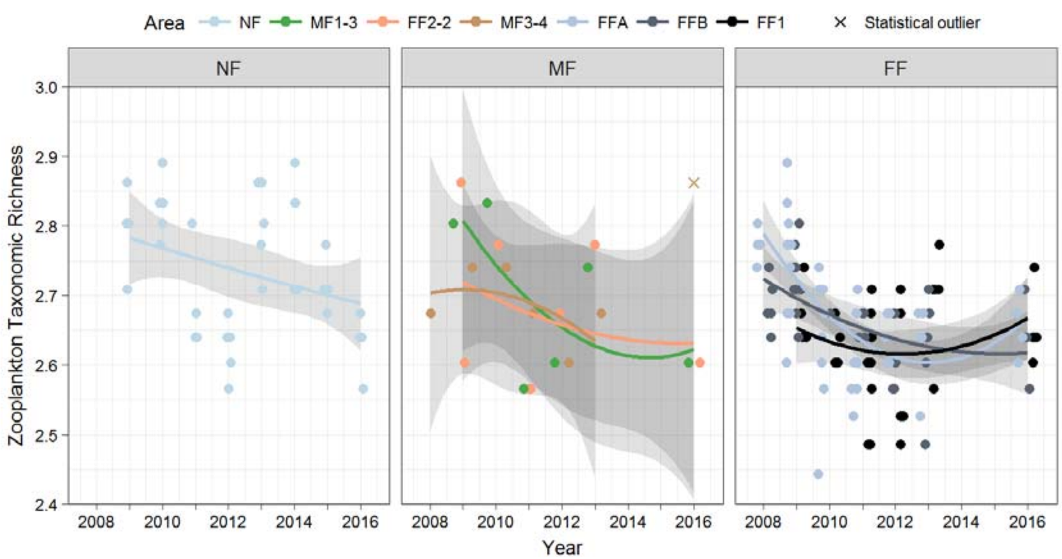
Zooplankton biomass in the FF area generally remained within the normal range, showing no clear trends over time, other than a potential slight decline in the FFB area (Figure 7-22 and 7-24). In 2010, zooplankton biomass at stations in the FF1 and FFA areas was significantly different than that in the NF area, while the FFB area was not. In 2013, the NF area differed significantly from the three FF areas; in 2016, no significant difference was detected among sampling areas in Lac de Gras (Table 7-18).

Figure 7-22 Total Zooplankton Biomass and Taxonomic Richness, 2008 to 2016



NF = near-field; MF = mid-field; FF = far-field; LDS = Lac du Sauvage.

Figure 7-23 Trend Analysis for Zooplankton Taxonomic Richness, 2008 to 2016



NF = near-field; MF = mid-field; FF = far-field; grey ribbons are 95% confidence intervals.

Table 7-18 Significance of Zooplankton Fixed Effects Models, 2008 to 2016

Variable	Coefficient	Numerator DF	Denominator DF	F-value	*P*-value
Zooplankton taxonomic richness	Area	6	16	1.16	0.374
	Year	1	142	0.59	0.443
	Year^2	1	142	0.59	0.444
	Area × Year	6	142	1.16	0.330
	Area × Year^2	6	142	1.16	0.331
Total zooplankton biomass	Area	6	16	7.35	<0.001
	Year	1	143	11.07	0.001
	Year^2	1	143	11.07	0.001
	Area × Year	6	143	7.35	**<0.001**
	Area × Year^2	6	143	7.35	**<0.001**
Total Cladocera biomass	Area	6	16	13.56	<0.001
	Year	1	143	71.38	<0.001
	Year^2	1	143	71.37	<0.001
	Area × Year	6	143	13.56	**<0.001**
	Area × Year^2	6	143	13.56	**<0.001**
Total Calanoida biomass	Area	6	16	6.72	0.001
	Year	1	150	24.12	<0.001
	Area × Year	6	150	6.72	**<0.001**
Total Cyclopoida biomass	Area	6	16	10.55	<0.001
	Year	1	143	5.25	0.023
	Year^2	1	143	5.25	0.023
	Area × Year	6	143	10.55	**<0.001**
	Area × Year^2	6	143	10.56	**<0.001**
Total Rotifera biomass	Area	6	16	6.33	0.001
	Year	1	142	4.00	0.047
	Year^2	1	142	4.01	0.047
	Area × Year	6	142	6.33	**<0.001**
	Area × Year^2	6	142	6.33	**<0.001**

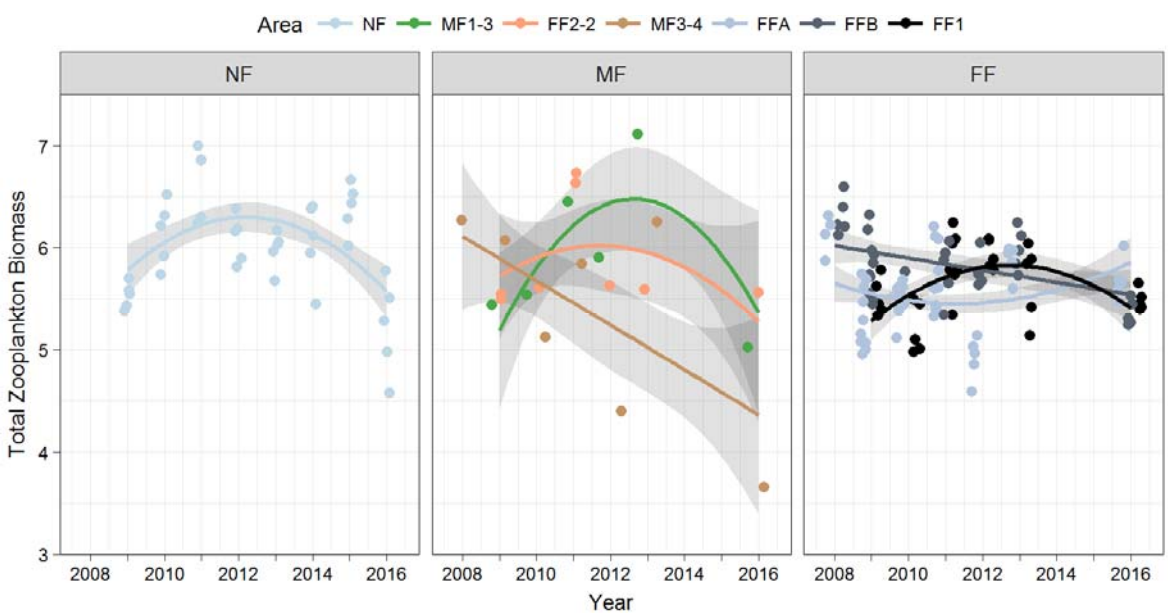
Notes: **Bolded** values indicate significance on the interaction term at P <0.05; DF = degrees of freedom; *F*-value = F-statistic; *P*-value = probability value.

Table 7-19 Multiple Comparisons of Parabolic Trends within 2010, 2013 and 2016

Variable	Year	Area						
		NF	MF1-3	FF2-2	MF3-4	FF1	FFB	FFA
Zooplankton taxonomic richness	2010	b	ab	ab	ab	a	a	a
	2013	b	ab	ab	ab	a	a	a
	2016	a	a	a	a	a	a	a
Total zooplankton biomass	2010	b	ab	ab	ab	a	b	a
	2013	c	bc	abc	a	b	ab	a
	2016	a	a	a	a	a	a	a
Total Cladocera biomass	2010	bc	abc	bc	abc	a	c	ab
	2013	c	bc	abc	a	ab	ab	a
	2016	ab	abcd	abc	a	bcd	cd	d
Total Cyclopoida biomass	2010	c	bc	abc	bc	b	b	a
	2013	c	bc	ab	ab	b	b	a
	2016	b	ab	ab	ab	a	b	b
Total Rotifera biomass	2010	c	bc	abc	a	a	ab	ab
	2013	d	cd	abcd	ab	bc	ab	a
	2016	b	ab	ab	ab	a	ab	a

Notes: The different letters designate areas significantly different from on another at a *P*-value of 0.05; NF= near-field; MF = mid-field; FF = far-field.

Figure 7-24 Trend Analysis for Total Zooplankton Biomass, 2008 to 2016



NF = near-field; MF = mid-field; FF = far-field; grey ribbons are 95% confidence intervals.

7.3.2.3 Biomass of Major Zooplankton Groups (Based on Enumeration)

Temporal trends in the biomass of the major zooplankton groups differed among sampling areas in Lac de Gras (Table 7-18). In the NF area, cladoceran biomass was above or near the upper limit of the normal range in all years from 2010 to 2015; in 2016 cladoceran biomass was within the normal range (Figure 7-25). An increasing trend was observed from 2008 to 2011 in the NF area, with a peak in 2011, and from 2012 to 2016, a decrease in cladoceran biomass was observed (Figure 7-26). Similar trends were observed in the MF areas. In the FF areas, cladoceran biomass was generally within or just above the upper limit of the normal range from 2008 to 2016. There was a significant difference in cladoceran biomass between the NF and FF1 area in 2010; in 2013 and 2016, the NF area differed from FFA and FFB areas but in 2016, the NF area did not differ from the FF1 area (Table 7-19).

Calanoid copepod biomass had decreased from 2010 to 2013 in all areas to near or below the lower limit of the normal range (Figure 7-27). In the NF area in 2014 and 2015, calanoid biomass increased to within the normal range; however, low biomass was observed again in all areas in 2016. Calanoid biomass significantly decreased in the NF area, at the MF3-4 station, and in the FFB and FFA areas (Figure 7-28; Table 7-20).

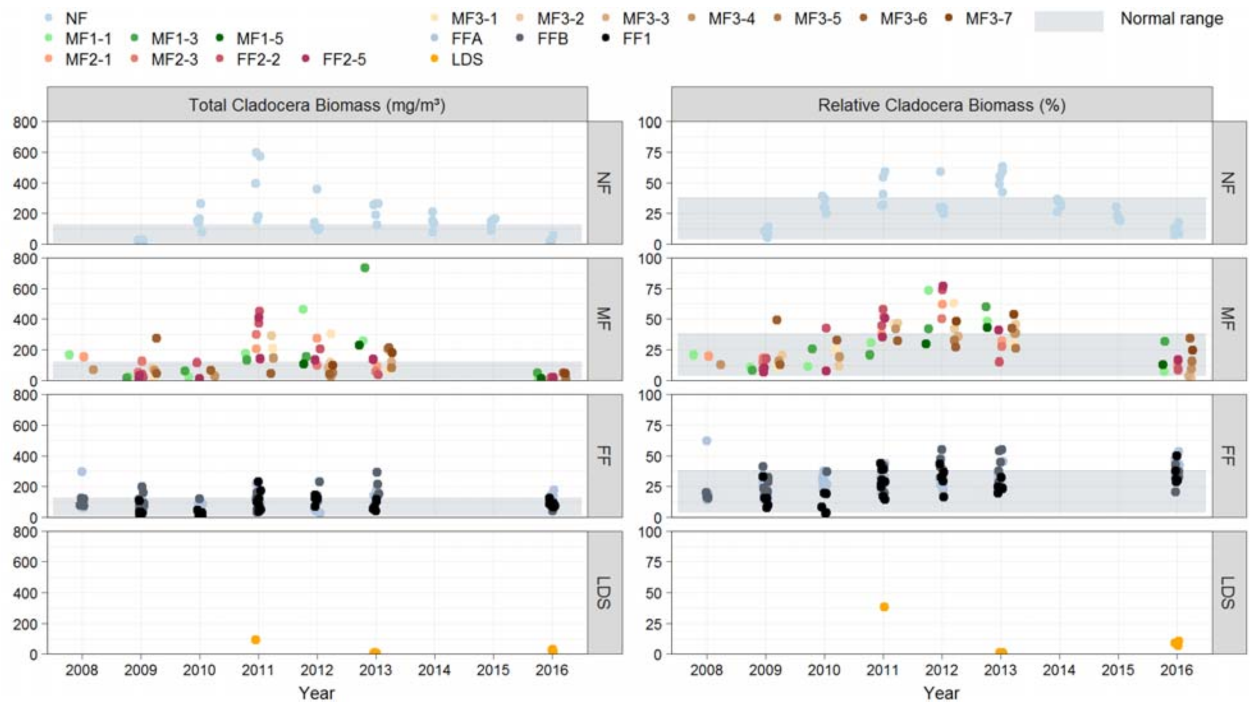
In all sampling areas in Lac de Gras, cyclopoid biomass has generally been within or above the upper limit of the normal range between 2008 and 2016 (Figure 7-29). The fitted parabolic models indicated inconsistent trends among areas, suggesting recent declines in the NF and FF1 areas, and stations MF1-3 and MF3-4, and increases at other areas and stations (Figure 7-30). Cyclopoid biomass in the NF area in 2010 and 2013 was significantly greater than in the FF areas. In 2016, biomass in the NF area was significantly greater than the FF1 area, but not the FFB and FFA areas (Table 7-19).

Rotifer biomass has increased in the NF area since 2009 (Figures 7-31 and 7-32). Peaks in rotifer biomass above the normal range were observed in 2013 and 2016 in the NF area. In the MF areas, rotifer biomass was generally within the normal range with excursions in the MF1 area at one or more stations in 2007, 2013, 2016; in the MF2-FF2 area in 2013; and in the MF3 area in 2013 and 2016. In the FF area, rotifer biomass was within the normal range or just above the upper limit from 2008 to 2016. Rotifer biomass in the NF area in 2010 and 2013 was significantly greater than biomass in the FF areas; however, in 2016 rotifer biomass in the NF area was greater than in the FF1 and FFA areas, but not the FFB area (Figure 7-32; Table 7-19).

The relative biomass of the major zooplankton groups showed generally similar trends as biomass. Relative cladoceran biomass increased over time in the NF, MF and FF areas until 2012 or 2013, but decreased in the NF area and at most MF stations between 2013 and 2016 (Figure 7-25). In contrast, relative calanoid biomass decreased over time in the NF, MF, and FF areas until 2013, but increased in the NF area and at most MF stations between 2013 and 2016. The majority of the NF and MF stations, and some of the FF area stations fell below the lower limits of the normal range (Figure 7-27). No clear treads were observed in relative cyclopoid biomass, and biomass was generally within or just above the upper limits of the normal range in the NF, MF and FF areas from 2008 to 2016 (Figure 7-29). Relative rotifer biomass has remained within or near the upper limit of the normal range from 2008 to 2013, with excursions occurring in most areas for the first time in 2013, with relative rotifer biomass above the normal range at most stations in the NF and MF areas in 2016 (Figure 7-31).

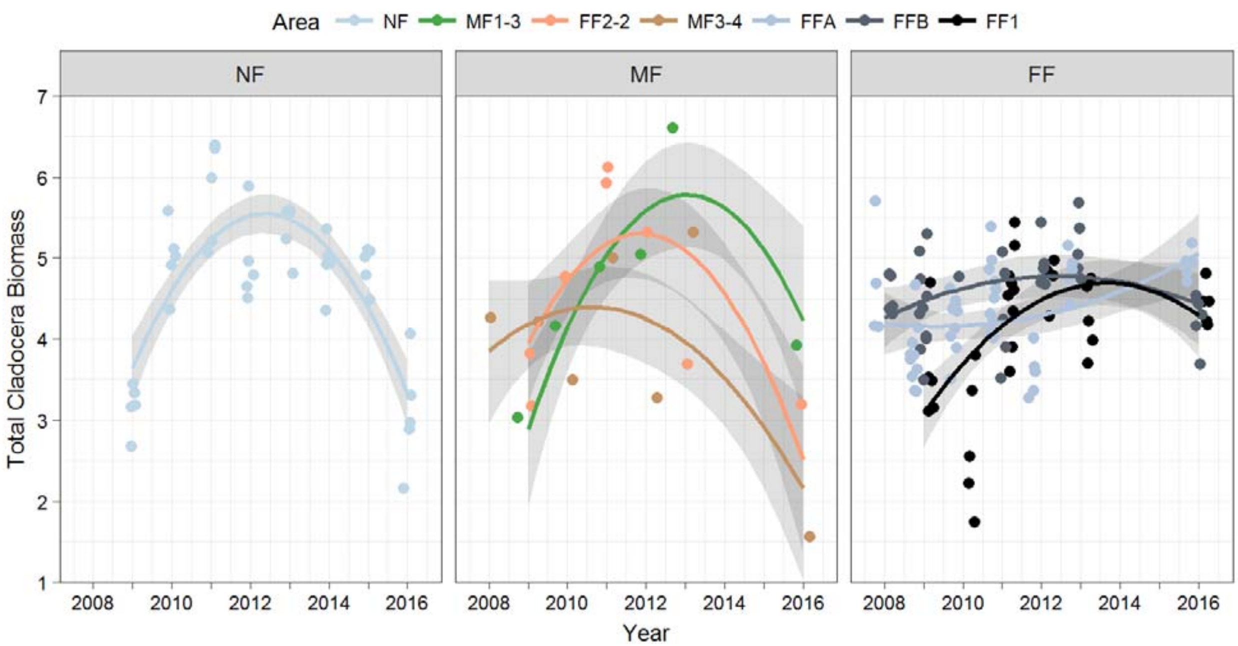
In Lac du Sauvage, zooplankton taxonomic richness was similar among stations and years from 2011 to 2016 (Figure 7-23). Total zooplankton biomass was greater in 2013 compared to 2011 and 2016, which were similar; no clear trend in biomass was observed. Total Cladocera biomass and the relative proportions of cladocerans were greater in 2011 compared to 2013 and 2016 (Figure 7-25). There was no apparent change among years and stations in calanoid biomass, but the relative proportions of calanoids decreased from 2011 to 2016 (Figure 7-27). Cyclopoid biomass and the relative proportions of cyclopoids followed the trend observed in total biomass, i.e., greater biomass in 2013 compared to 2011 and 2016; however, cyclopoid biomass in 2016 was greater than biomass measured in 2011, unlike total biomass (Figure 7-29). In contrast, total rotifer biomass and the proportions of rotifers were less in 2016 compared to 2011 and 2013 (Figure 7-31).

Figure 7-25 Total Cladocera Biomass and Relative Cladocera Biomass, 2008 to 2016



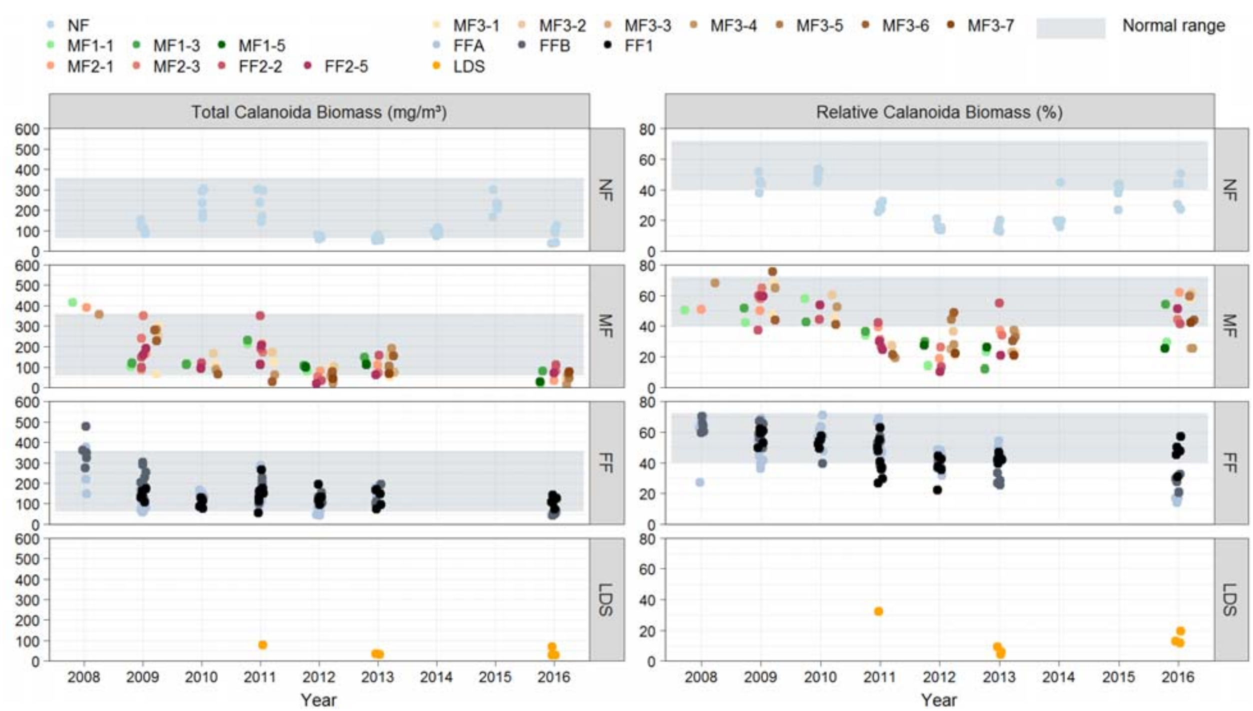
NF = near-field; MF = mid-field; FF = far-field; LDS = Lac du Sauvage.

Figure 7-26 Trend Analysis for Total Cladocera Biomass, 2008 to 2016



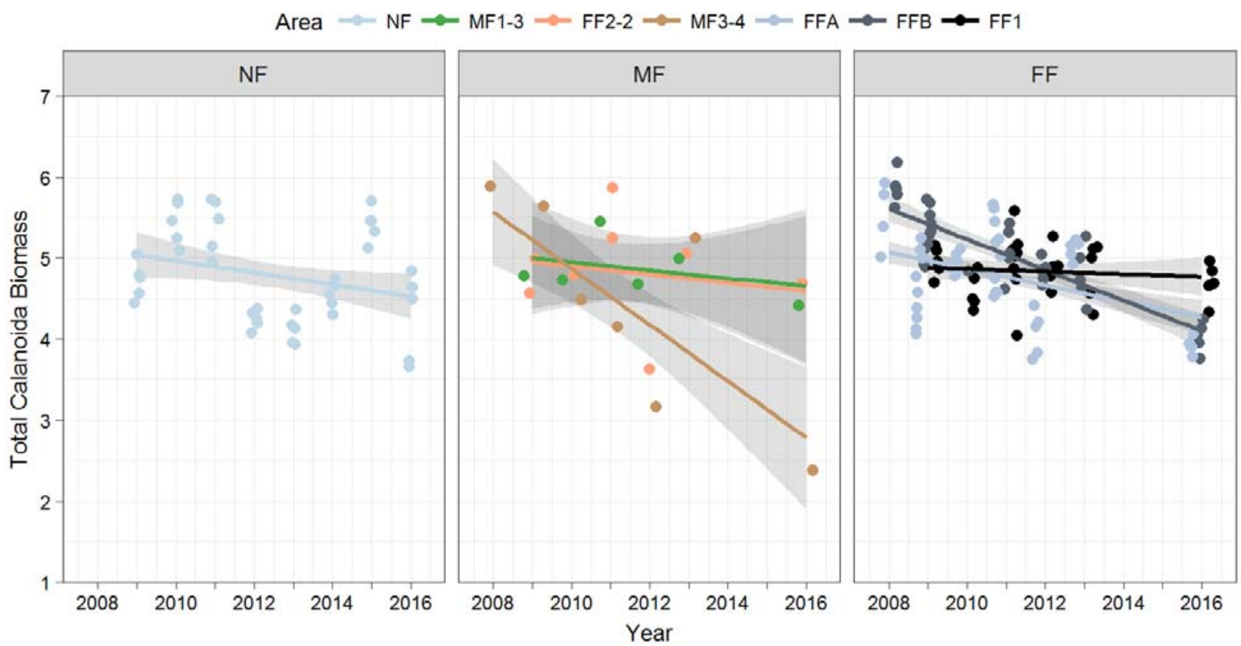
NF = near-field; MF = mid-field; FF = far-field; grey ribbons are 95% confidence intervals.

Figure 7-27 Total Calanoida Biomass and Relative Calanoida Biomass, 2008 to 2016



NF = near-field; MF = mid-field; FF = far-field; LDS = Lac du Sauvage.

Figure 7-28 Trend Analysis for Total Calanoida Biomass, 2008 to 2016



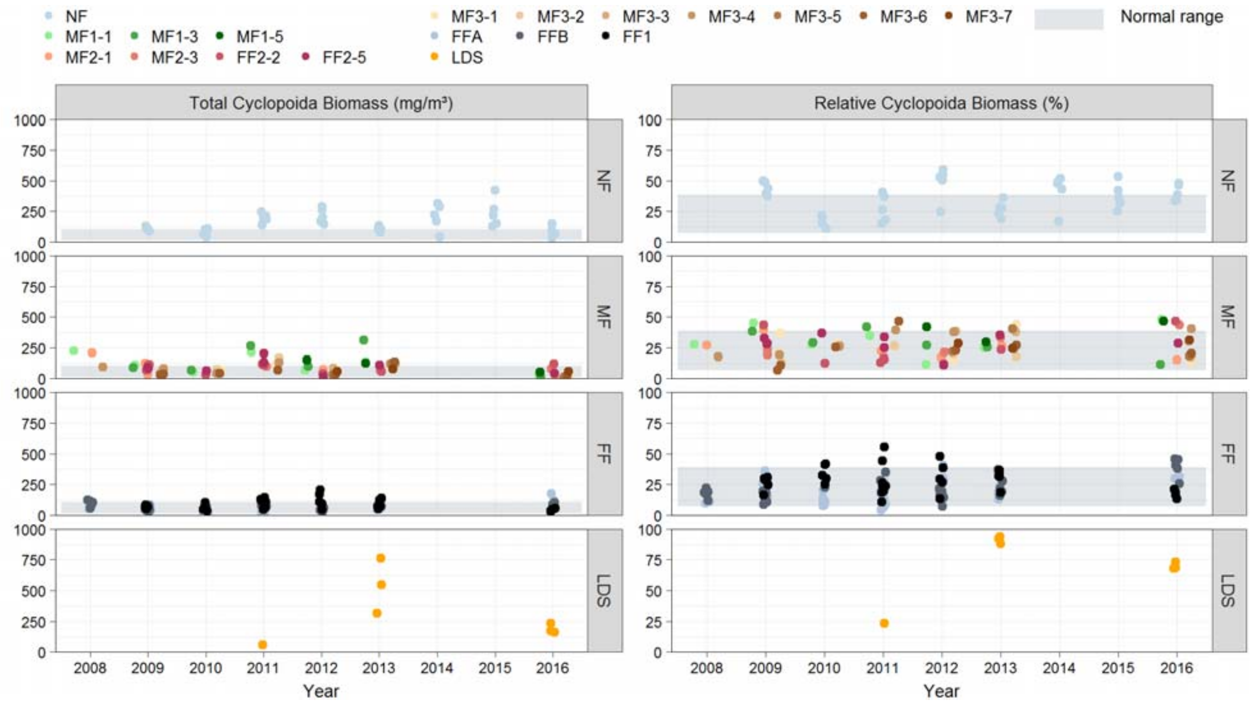
NF = near-field; MF = mid-field; FF = far-field; grey ribbons are 95% confidence intervals.

Table 7-20 Estimated Significance of Difference of Linear Slopes from zero

Variable	Area						
	NF	**MF1-3**	**FF2-2**	**MF3-4**	**FF1**	**FFB**	**FFA**
Total Calanoida biomass	**0.033** ↓	0.629	0.589	**<0.001** ↓	0.524	**<0.001** ↓	**<0.001** ↓

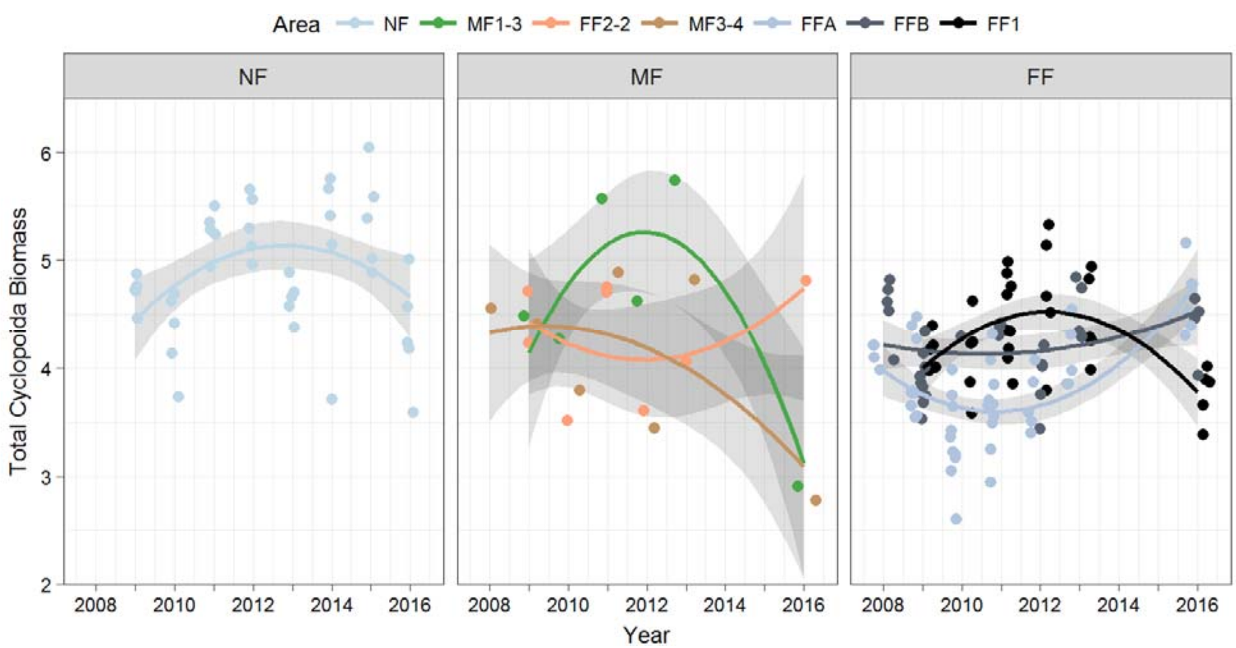
Notes: **Bolded** values indicate significance. The arrow indicates direction of trend; NF= near-field; MF = mid-field ; FF = far-field.

Figure 7-29 Total Cyclopoida Biomass and Relative Cyclopoida Biomass, 2008 to 2016



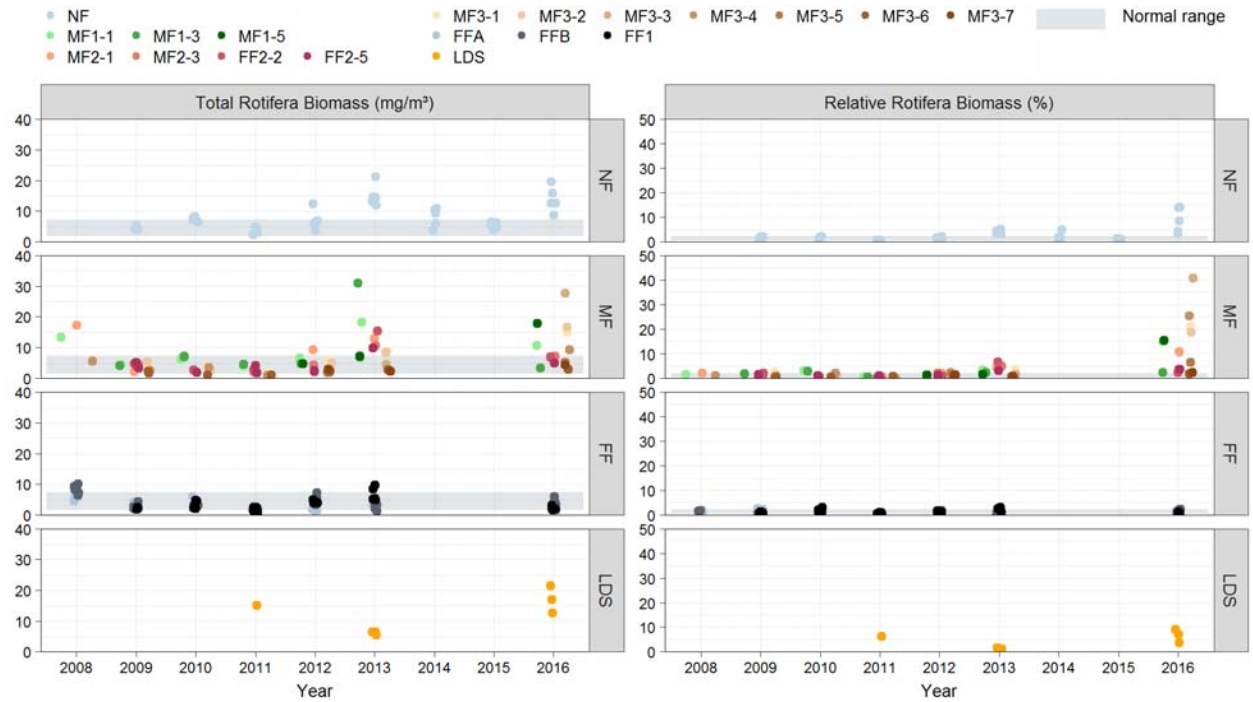
NF = near-field; MF = mid-field; FF = far-field; LDS = Lac du Sauvage.

Figure 7-30 Trend Analysis for Total Cyclopoida Biomass, 2008 to 2016



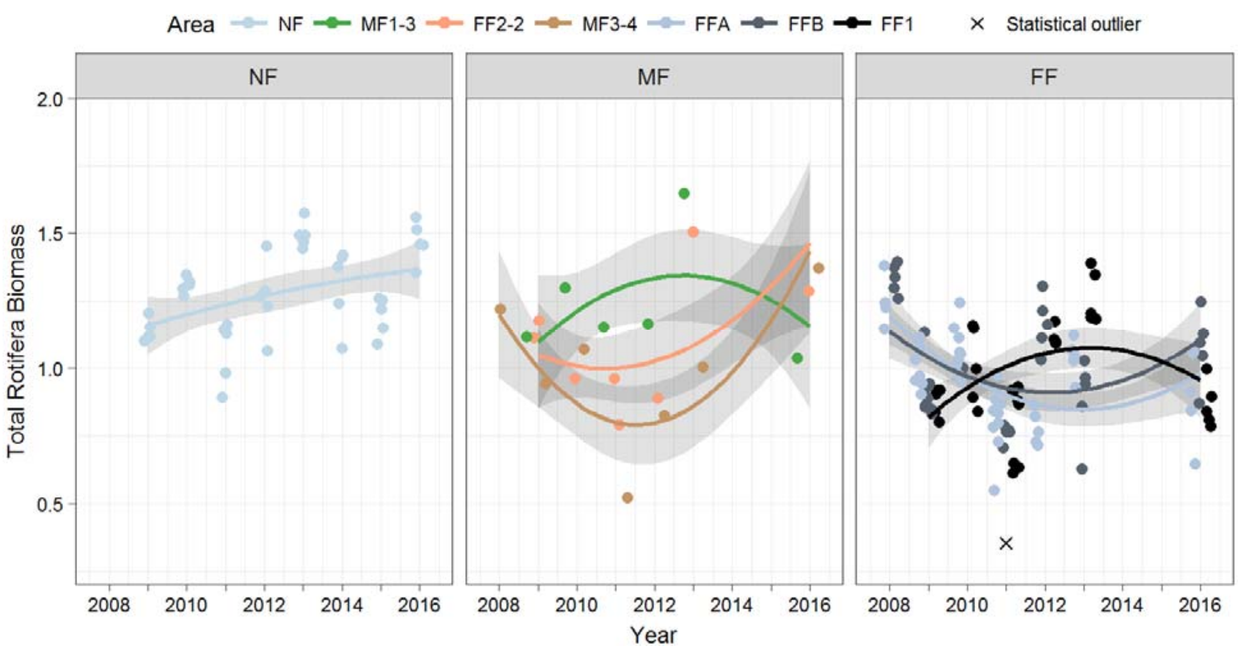
NF = near-field; MF = mid-field; FF = far-field; grey ribbons are 95% confidence intervals.

Figure 7-31 Total Rotifera Biomass and Relative Rotifera Biomass, 2008 to 2016



NF = near-field; MF = mid-field; FF = far-field; LDS = Lac du Sauvage.

Figure 7-32 Trend Analysis for Total Rotifera Biomass, 2008 to 2016



NF = near-field; MF = mid-field; FF = far-field; grey ribbons are 95% confidence intervals.

7.3.2.4 Zooplankton Community Structure

The two dimensional nMDS configurations for zooplankton biomass in the NF and FF areas from 2008 to 2016 had a stress value of 0.13, indicating a reasonable level of fit to the original dataset (Figure 7-33). The SIMPROF test (P<0.05) indicated that the level of interpretation of the 40%, 60% and 80% clusters was acceptable and the global ANOSIM (R = 0.361, P = 0.001) test indicated that statistical interpretation of the MDS structure was permitted. The ordination plot indicated separation between the area-year groupings between the NF and FF areas in terms of zooplankton community composition, i.e., the zooplankton community in the NF areas differed among years from 2008 to 2016, while the FF areas generally grouped together at 80% similarity during these years. The NF area communities from 2009 and 2012 grouped with the FF areas, while in 2010 and 2011, the NF area communities diverged and grouped together at 80% similarity. The 2013, 2014 and 2015 NF area communities also grouped together at 80% similarity, and diverged from other areas. The 2016 NF community differed from those in all other years and sampling areas. Overall, all areas and years clustered together within a 40% similarity ellipse. The pair-wise ANOSIM for NF versus FF areas for each year indicated significant differences in each year between the NF area and the FF areas (R >0.5, P = 0.001), the greatest difference observed in 2016 (R = 0.99, P = 0.001; Table 7-21).

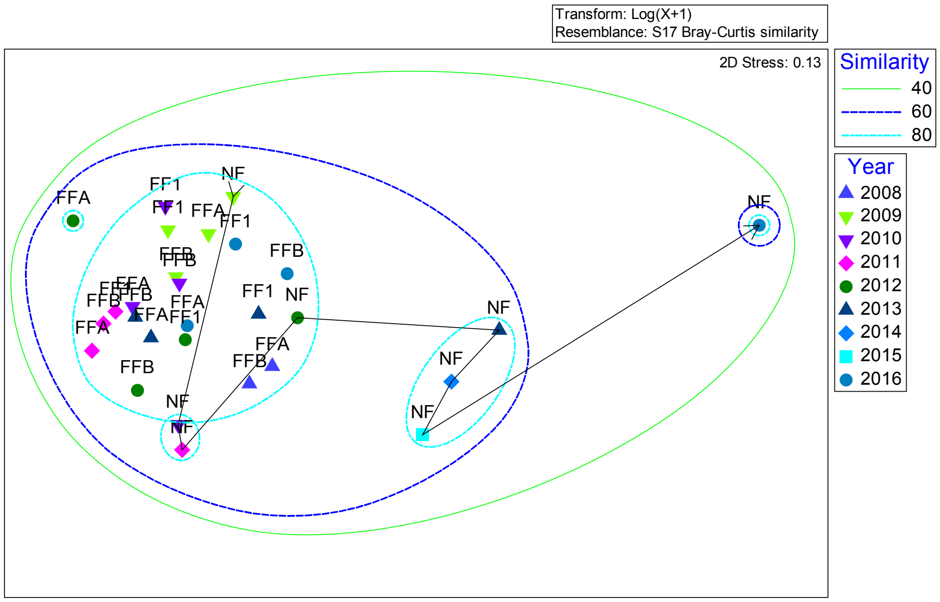
The two dimensional nMDS configurations for zooplankton biomass along each transect from 2008 to 2016 had stress values of 0.11 for the NF-MF1-FF1 and NF-MF2-FF2 transect analyses, and 0.09 for the NF-MF3-FFB-FFA transect analysis, indicating a good fit to the original datasets (Figures 7-34 to 7-36). Along each of the transects, the zooplankton community in 2016 differed from the communities sampled between 2008 and 2015 (Figures 7-34 to 7-36). Along the NF-MF1-FF1 transect, all stations and areas grouped together from 2008 to 2015 within a 60% similarity ellipse, while the NF area, and MF1-3 and MF1-1 stations in 2016 were separate from this grouping within their own 60% similarity ellipse. Overall, the stations along the NF-MF1-FF1 transect were 40% similar (Figure 7-34).

Stations along the NF-MF2-FF2 transect grouped together within a 60% similarity ellipse, with the exception of station MF2-1, which was separated from the remainder of the stations (Figure 7-35). The 2016 data from the NF area, and the remainder of the stations along the NF-MF2-FF2 transect grouped together at 80% similarity within the overall 60% similarity ellipse.

Stations along the NF-MF3-FFB-FFA transect were 40% similar; however, stations along the transect sampled in 2016 generally grouped separately at 50% similarity from the remainder of the stations from 2008 to 2015 (Figure 7-36). The FFB and FFA areas and stations located farther along the transect (i.e., MF3-7 and MF3-6) grouped with the samples collected between 2008 and 2015.

The nMDS results indicate that changes over time in zooplankton community structure have been occurring in the NF area of Lac de Gras, and that conditions in 2016 differed from previous years. In 2016, zooplankton community structure was different in the NF and MF areas of Lac de Gras relative to the FF areas. Specifically, zooplankton community structure along the NF-MF3-FFB-FFA transect in 2016 also differed from all other areas of Lac de Gras, potentially as a result changes in water quality associated with dike construction.

Figure 7-33 Non-metric Multidimensional Scaling of Zooplankton Biomass in the Near-field and Far-field Areas, 2008 to 2016



NF = near-field; FF = far-field.

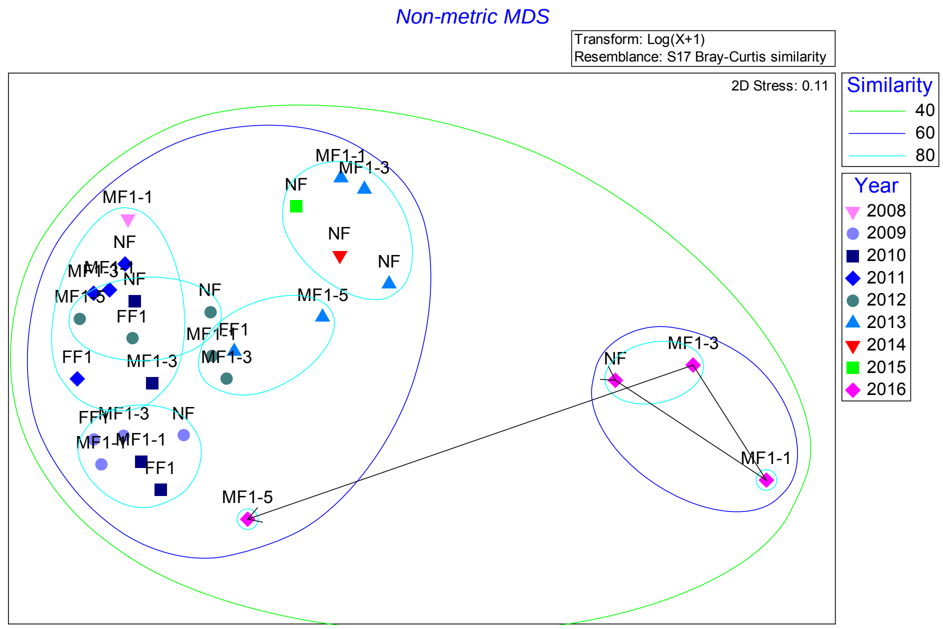
Table 7-21 Analysis of Similarities (ANOSIM) Tests Between Near-field and Far-field Zooplankton Communities in Lac de Gras, 2009 to 2016

Year	R statistic	*P-value*
2009	0.57	**0.001**
2010	0.62	**0.001**
2011	0.69	**0.001**
2012	0.52	**0.001**
2013	0.88	**0.001**
2014	n/a	n/a
2015	n/a	n/a
2016	0.99	**0.001**

Note: NF vs. FF area comparisons significant with *P-values* <0.05. Bolded values indicate significance.

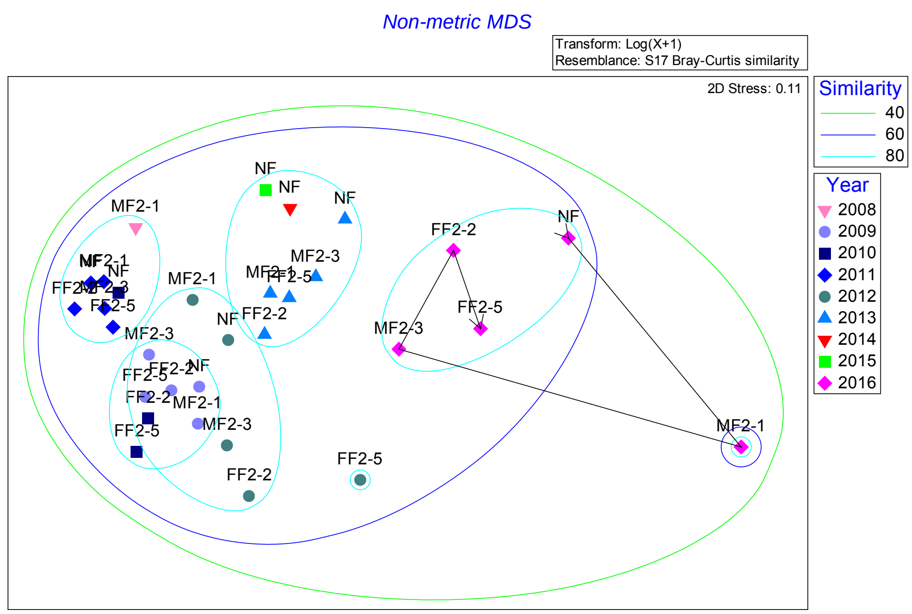
n/a = not applicable; *P* = probability.

Figure 7-34 Non-metric Multidimensional Scaling of Zooplankton Biomass along the NF-MF1-FF1 Transect, 2008 to 2016



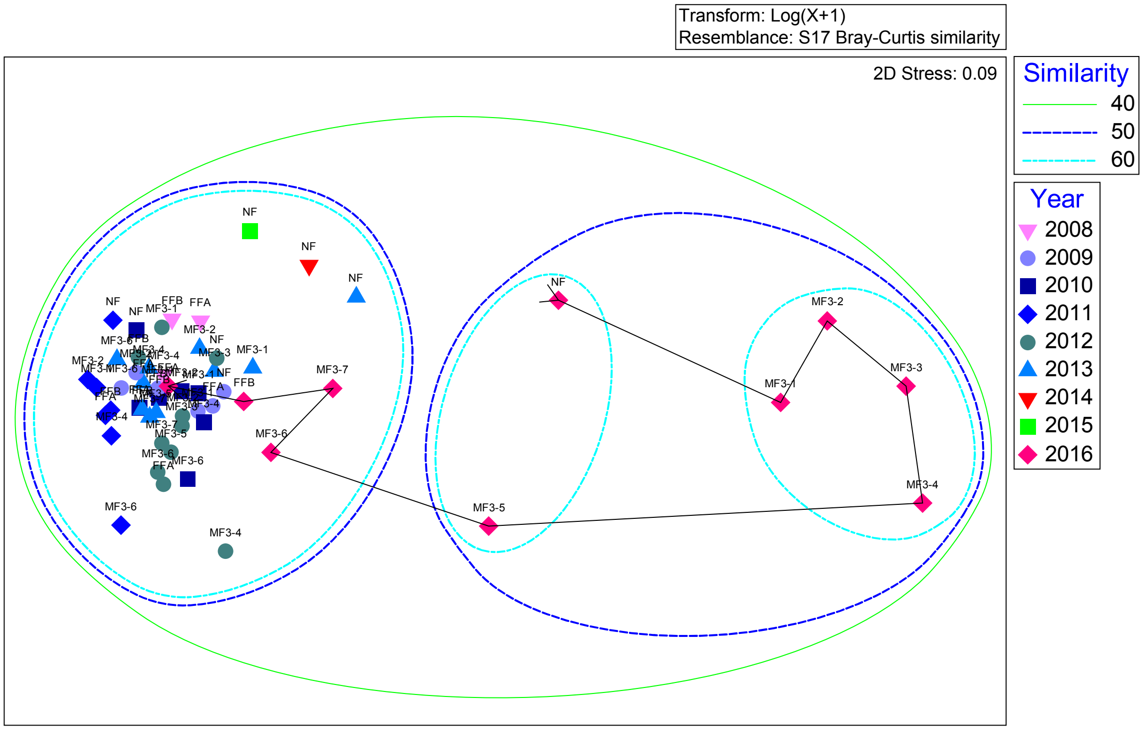
NF = near-field; MF = mid-field; FF = far-field.

Figure 7-35 Non-metric Multidimensional Scaling of Zooplankton Biomass along the NF-MF2-FF2 Transect, 2008 to 2016



NF = near-field; mF = Mid-field; FF = far-field.

Figure 7-36 Non-metric Multidimensional Scaling of Zooplankton Biomass along the NF-MF3-FFB-FFA Transect, 2008 to 2016



NF = near-field; MF = mid-field; FF = far-field.

7.3.2.4.1 Zooplankton Summary

Temporal trends in zooplankton taxonomic richness were not observed, and richness generally remained within the normal range in the NF and FF areas of Lac de Gras from 2008 to 2016. Zooplankton biomass has been more variable in the NF and MF areas compared to the FF areas, but generally has remained within the normal range. Excursions in the NF area above the normal range occurred in 2008 and 2011, and below the normal range in 2016. A decreasing temporal trend in zooplankton biomass was observed at the MF3-4 station.

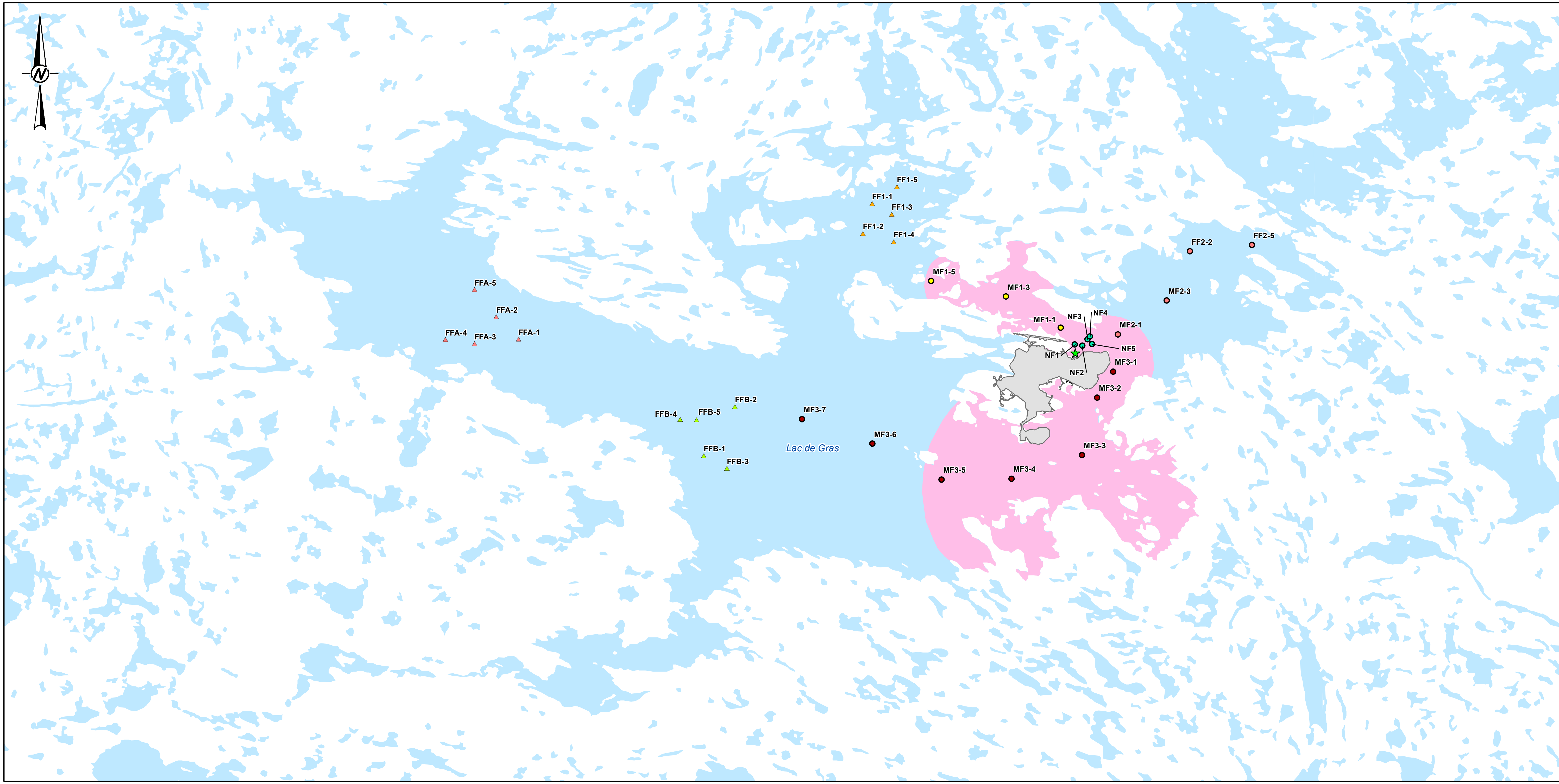
Temporal trends in the biomass of the major zooplankton groups differed among sampling areas in Lac de Gras between 2008 and 2016. Cladoceran biomass exceeded the normal range in the NF area in most years from 2008 to 2013, but from 2014 to 2016 it was within the normal range; relative biomass of this group increased until 2013, and then decreased to 2016. A decrease over time was observed in calanoid copepod biomass and relative biomass in all areas, calanoid biomass had decreased from 2009 to 2016 in all areas, with the exception of FF1, to near or below the lower limit of the normal range. In the NF area in 2014 and 2015, calanoid biomass increased to within the normal range; however, low biomass was observed again in all areas in 2016. Cyclopoid copepod and rotifer biomass have generally been within or above the upper limit of the normal range between 2008 and 2016.

The zooplankton nMDS ordination plot indicates that changes over time in zooplankton community structure have been occurring in the NF area of Lac de Gras, and that conditions in 2016 differed from previous years. In 2016, zooplankton community structure was different in the NF and MF areas of Lac de Gras relative to the FF areas, with the greatest differences in the MF3 area.

7.3.3 Extent of Effects

The extent of effects on plankton biomass was not previously calculated or presented in the annual reports. The area of Lac de Gras with total zooplankton biomass below the normal range was delineated for 2016 (Figure 7-37), which is the first year this type of effect was observed. No declines below the normal range were observed for total phytoplankton biomass since monitoring began under the AEMP. Zooplankton biomass in 2016 triggered an Action Level 2, as a result of the spatial extent of the effects; however, the difference between mean total zooplankton biomass in 2016 in the area subject to greatest effluent exposure (i.e., the NF area) and the FF areas was 17%, which is low compared to the typical degree of variation observed among areas within a large lake (Golder 2017c). The mean total zooplankton biomass in the NF area remained within the normal range, and a similar response was not observed in zooplankton biomass measured as AFDM.

Effects on zooplankton biomass based on enumeration were observed primarily along the NF-MF1-FF1 and NF-MF3-FFB-FFA transects. The boundary of effects on zooplankton biomass to the northwest extended to station MF1-5 (Figure 7-37). The extent of effects to the northeast of the Mine extended to station MF2-1. The boundary of effects south of the Mine extended to station MF3-5. The extent of the effect on zooplankton biomass, based on the affected stations, was calculated to be 103 km^2. Compared to the total surface area of the lake (573 km^2), the area demonstrating effects on zooplankton biomass represents 18% of the lake area.



LEGEND

DIFFUSER

EXPOSURE

- NEAR-FIELD
- MID-FIELD 3
- MID-FIELD 1
- FAR-FIELD 2; MID-FIELD 2
- FAR-FIELD 1
- FAR-FIELD A
- FAR-FIELD B

DIAVIK FOOTPRINT

WATERBODY

ZOOPLANKTON AFFECTED AREA (BASED ON ENUMERATION)

REFERENCE

HYDROGRAPHY DATA OBTAINED FROM GEOGRATIS, © DEPARTMENT OF NATURAL RESOURCES CANADA.

PROJECTION: UTM ZONE 12 DATUM: NAD 83

7 0 7

1:200,000 KILOMETRES

PROJECT

DIAVIK DIAMOND MINES INC.

TITLE

SPATIAL EXTENT OF EFFECTS ON ZOOPLANKTON BIOMASS (BASED ON ENUMERATION) ALONG THE NF-MF3 FFB FFA TRANSECT IN LAC DE GRAS FROM 2008 TO 2016

GOLDER

PROJECT		1771843.4000	FILE No.	
DESIGN	KH	02 Jan. 2018	SCALE AS SHOWN	REV. 0
GIS	BP	14 Mar. 2018		
CHECK	KH	14 Mar. 2018	FIGURE: 7 - 37	
REVIEW	ZK	14 Mar. 2018		

I:\CLIENTS\DIAVIK\1771843\Mapping\MXD\Aquatics\2017AEMP_ReEvaluation_Report\2007_2016AEMP\REV0\Fig7_37_Zooplankton_Enumeration_Affected_Area_2016_Rev0.mxd

7.3.4 Integrative Environmental Assessment

7.3.4.1 Approach

The WLWB has requested that as part of the *2014 to 2016 Aquatic Effects Re-evaluation Report,* an integrative assessment of plankton variables be conducted, which is to include:

- an evaluation of nMDS results to identify environmental gradients
- an evaluation of the role of TN and the ratio of N:P in explaining variation in phytoplankton biomass
- a more in-depth and integrative assessment of the plankton data

The assessment includes a comparison of TDS, as an indicator of exposure to Mine effluent in Lac de Gras, and nutrients, TN, TP and the N to P molar ratio, to phytoplankton biomass and community structure. The relationship between zooplankton biomass and community data and TDS concentrations is also evaluated. A detailed evaluation of MDS ordination results to assess the influence of environmental gradients on phytoplankton and zooplankton community structure are summarized using the MDS ordinations.

7.3.4.2 Physical Limnology, Water Quality and Nutrient Concentrations

Conditions in Lac de Gras are suitable for the persistence of phytoplankton and zooplankton communities typical of large subarctic lakes. Lac de Gras remains a clear, soft water lake, with generally neutral to acidic pH within the top and middle of the water column (Section 4). Secchi depth measurements (typically 5 to 15 m) indicate good light penetration in all areas of Lac de Gras. The water column has remained well-oxygenated throughout the lake during the period monitored. Between 2007 and 2008, water temperature was similar throughout the water column, although in 2008, water temperatures decreased in the NF area with increasing depth within the euphotic zone. This pattern was not observed again until 2013 in the FF area.

Specific conductivity and TDS have continued to increase since 2007, although TDS is still considered low in absolute terms (<50 µS/cm) in all areas of Lac de Gras (Section 4). Ions, such as sodium, sulphate, magnesium, calcium and chloride are also showing similar increasing trends, but remain within ranges suitable for healthy aquatic communities. Conductivity profiles have shown that the water column is well mixed vertically in the FF areas, but the effluent plume is sometimes detectable in the NF area, and occasionally at MF stations, at the middle and bottom depths. Metals, such as strontium and molybdenum, are increasing in a similar manner to TDS, while barium appears to have increased and is more recently decreasing. Metal concentrations are well below ranges where effects on aquatic health would be expected. Some of the metals with increasing temporal trends supply micronutrients to aquatic organisms (Wetzel 2001).

Nutrient concentrations remain low in Lac de Gras, and both TP and TN are within the oligotrophic range (Section 5). Between 2007 and 2016, concentrations of TP and TDP have generally remained within or just above the upper limit of the normal range in all areas of Lac de Gras during the open-water season. Concentrations of SRP, the most bioavailable form of phosphorus, were frequently reported at or below the DL (1 µg/L) in the NF area during the open-water season. In the MF and FF areas, concentrations were more frequently detected at concentrations above the upper limit of the normal range in recent years (2014 to 2016); however, concentrations remained below 3 µg/L. Total and dissolved nitrogen concentrations have exceeded the upper limit of the normal range in the NF and MF areas since 2009 (Section 5).

Concentrations of TN in the former reference areas were generally within the normal range during the open-water season from 2007 to 2012, but have increased above the upper limit of the normal range in 2013 and 2016; this may be partly an artifact caused by a change in analytical laboratory. Dissolved nitrogen concentrations in the FF areas were within the normal range with the exception of greater concentrations in 2016. Total ammonia and N+N concentrations exceeded the normal range in the majority of samples in the NF and MF areas since 2010 (Section 5). Exceedances of the normal range were also noted in the FF areas; however, quality control issues identified with the open-water season total ammonia data analyzed by Maxxam in 2013 suggest that these data are likely biased high.

7.3.4.3 Relationships with Effluent Exposure

Concentration of TDS (calculated) is elevated in the effluent and has shown a clear relationship with distance from the diffuser each year after 2006 (Figure 7-38); therefore, TDS was used as an effluent tracer to explore relationships between effluent exposure and selected plankton community variables. Total phytoplankton and zooplankton biomass were plotted against TDS and Pearson correlations were run for each year of monitoring to explore relationships between plankton biomass and concentrations of TDS and nutrients. Generally, moderate to strong relationships (r >0.5) were detected between phytoplankton and zooplankton biomass and TDS concentration (Figures 7-39 and 7-40; Table 7-22). In 2010, 2011 and 2016, strong relationships were observed between phytoplankton biomass and TDS concentration (r = 0.86, r = 0.67, r = 0.63, respectively), and in 2011 and 2012, a strong relationship was observed between zooplankton biomass and TDS concentration (r = 0.83, r = 0.62, respectively). Poor relationships (r <0.5) were observed between TDS and phytoplankton biomass in 2007, 2012 and 2013, and in zooplankton biomass in 2009, 2013 and 2016. Greater total phytoplankton biomass was observed in areas with greater TDS concentrations in the majority of years between 2007 and 2016, which suggests a Mine-related enrichment effect (discussed further in Section 5). Lower biomass in areas with greater TDS would have suggested a Mine-related toxicity effect on the plankton community; this was not observed in any year throughout the AEMP monitoring period, consistent with water quality results indicating a lack of ions or metals at concentrations that could result in adverse effects on aquatic life.

Phytoplankton and zooplankton community structure at each station were also examined relative to TDS concentration on the MDS plots. A relationship between TDS concentrations and phytoplankton community structure is apparent between 2007 and 2012, with the NF areas (greater TDS) being separated from the FF areas (less TDS; Figure 7-41). A similar relationship between TDS concentrations and phytoplankton community structure was observed in a separate analysis for 2013 to 2016, with the NF areas also being separated from the FF areas in 2013 and 2016 (Figure 7-42). A weaker relationship is apparent between TDS concentrations and zooplankton community structure during 2008 to 2016 (Figure 7-43), which also shows that since 2013, the NF area has become more widely separated from the FF areas compared to previous years of monitoring.

Figure 7-38 Open-water, Top and Mid-depth Total Dissolved Solids Concentrations in relation to Distance from the Diffusers, 2000 to 2016

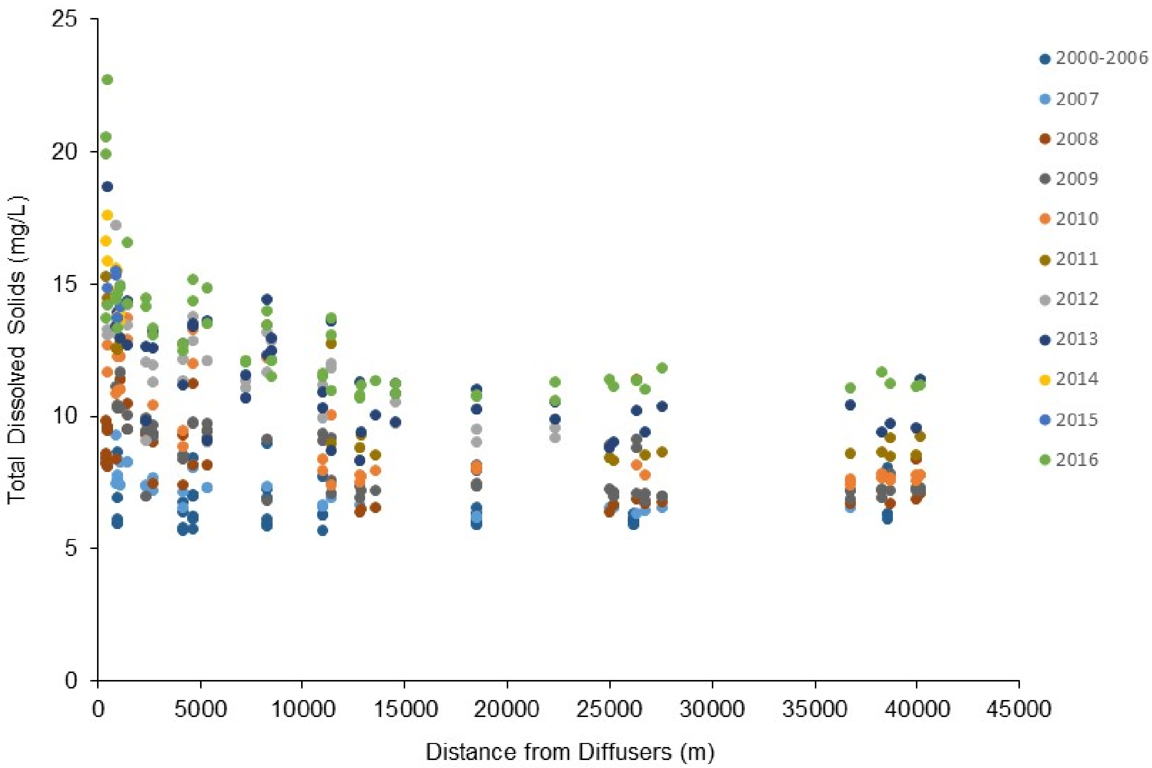


Table 7-22 Relationship between Total Dissolved Solids and Phytoplankton and Zooplankton Biomass, 2007 to 2016

Year	TDS vs Total Phytoplankton Biomass (*r*)	TDS vs Total Zooplankton Biomass (*r*)
2007	0.26	-
2008	0.57	-
2009	0.61	0.10
2010	0.86	0.55
2011	0.67	0.83
2012	0.34	0.62
2013	0.45	0.06
2014	-	-
2015	-	-
2016	0.63	0.24

Notes: *r* = Pearson correlation co-efficient; n = 20 to 46; TDS = total dissolved solids; - = insufficient data; only the NF area was monitored for plankton (2014, 2015), or data were not available due to lab or field error.

Figure 7-39 Total Phytoplankton Biomass relative to Total Dissolved Solids, 2007 to 2016

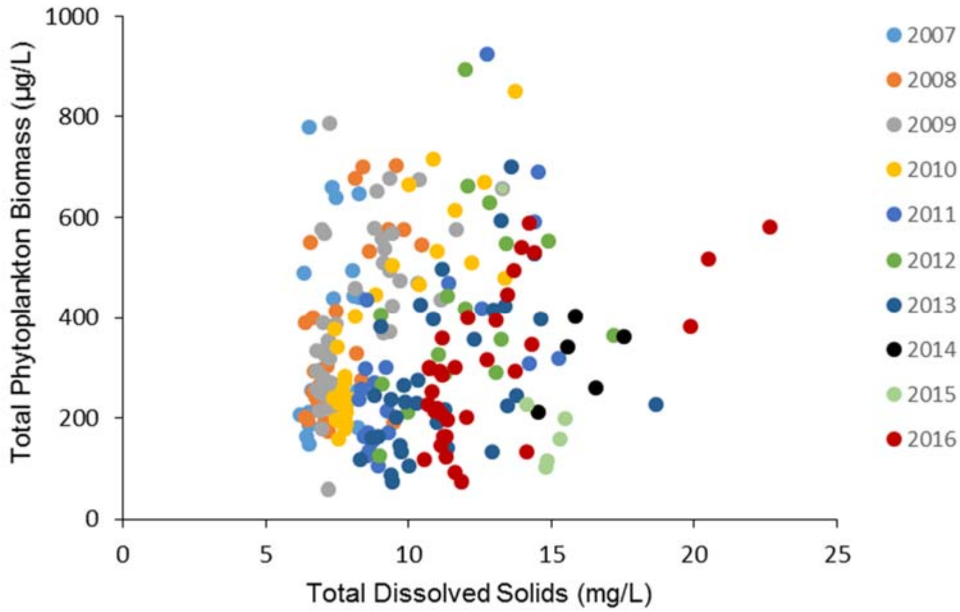


Figure 7-40 Total Zooplankton Biomass relative to Total Dissolved Solids, 2008 to 2016

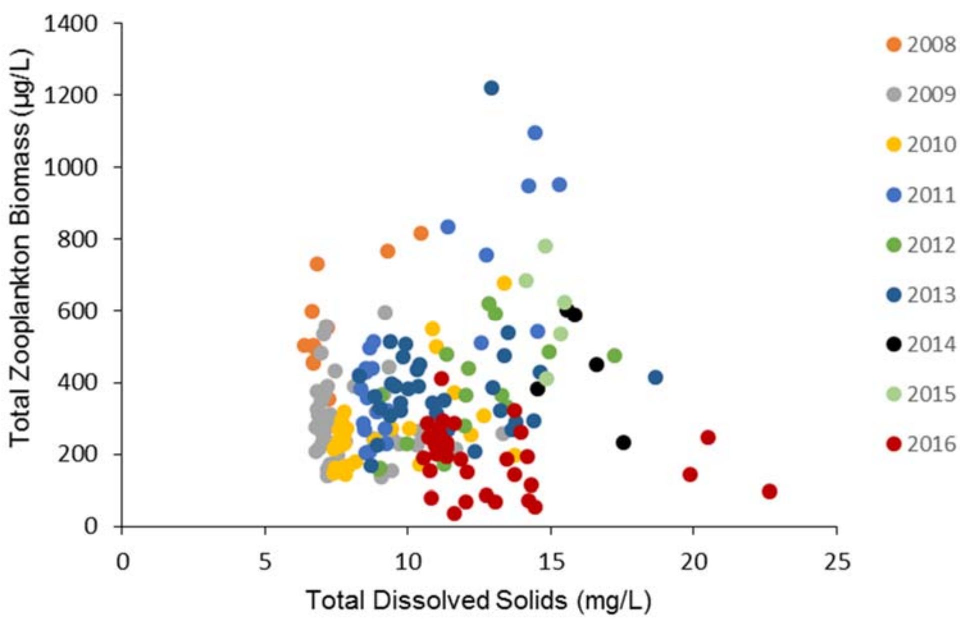
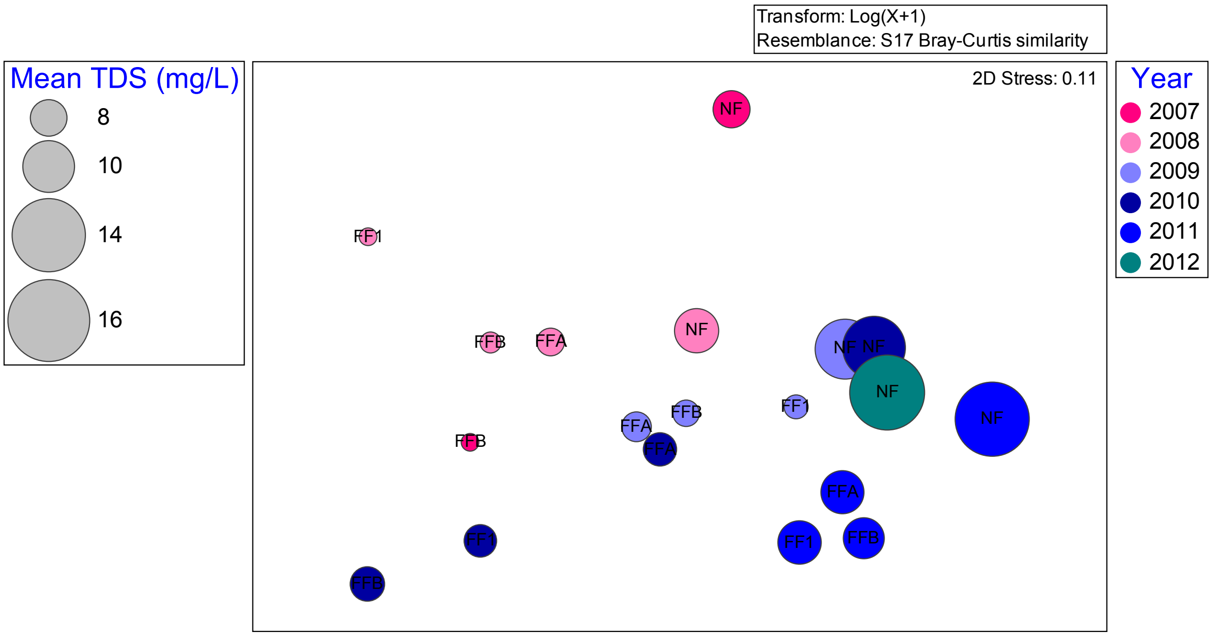
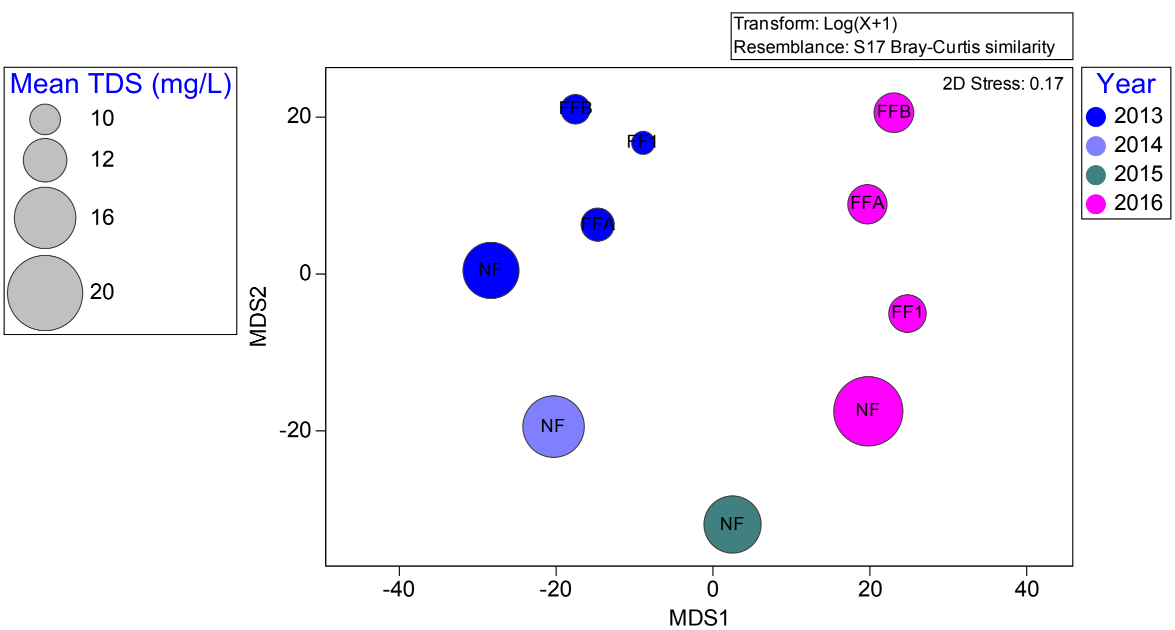


Figure 7-41 Ordination of the Phytoplankton Community in the NF and FF areas, with symbols scaled to TDS Concentrations, 2007 to 2012



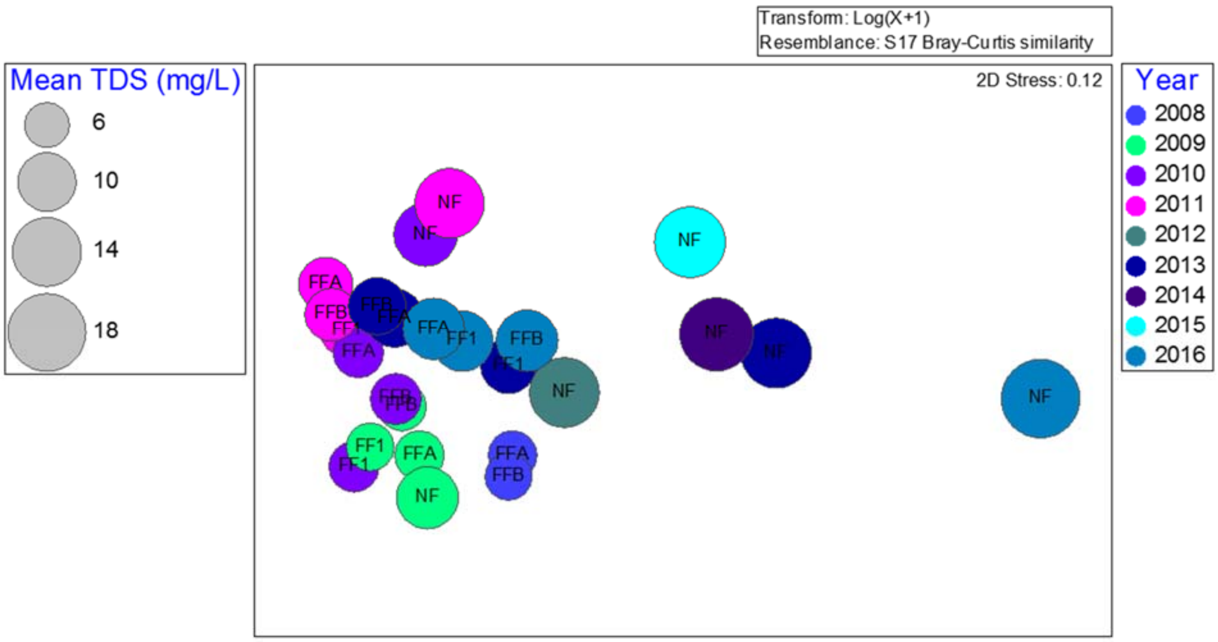
Notes: TDS = total dissolved solids; NF = near-field area; FF = far-field areas.

Figure 7-42 Ordination of the Phytoplankton Community in the NF and FF areas, with symbols scaled to TDS concentrations, 2013 to 2016



Notes: MDS = metric multidimensional scaling axes 1 and 2; TDS = total dissolved solids; NF = near-field area; FF = far-field areas.

Figure 7-43 Ordination the Zooplankton Community in the NF and FF areas, with symbols scaled to TDS concentrations, 2008 to 2016



TDS = total dissolved solids; NF = near-field area; FF = far-field areas.

7.3.4.4 Nutrient Relationships

Phytoplankton community composition and biomass are controlled to a large extent by concentrations of limiting nutrients. The nutrient ratio of N to P indicates severe P-limitation in the NF area of Lac de Gras from 2007 to 2016 (Section 5). Phosphorus-limitation at MF and FF stations in Lac de Gras varied from near P-sufficiency to severe P-limitation throughout the time series. The degree of P-limitation appears to have increased in all areas of Lac de Gras between 2007 and 2016, as a result of clear N-enrichment from the Mine discharge accompanied by only minor increases in P concentrations, which resulted in a shift in the N to P ratio over time.

The linear relationships between total phytoplankton biomass and TP, TN and the ratio of N to P were examined for all years from 2007 to 2016 (Table 7-23; Figure 7-44 to 7-46). The relationship between phytoplankton biomass and TP was poor (r <0.5) in all years, with the exception of in 2007 (r = 0.76), likely as a result of the small range in TP concentrations (0.5 to 7.2 µg/L in the NF and FF areas). The relationship between phytoplankton biomass and TN was moderate to strong (r = 0.55 to 0.82) between 2007 and 2012, but from 2013 to 2016, the relationship was poor (r <0.5). It is notable that in monitoring years before 2013, TN varied over a relatively narrow range (100 to 250 µg/L), and increases in concentrations at a number of stations since 2012 resulted in no apparent increase in phytoplankton biomass. The interpretation of these results is not clear, however, it is possible that (1) N inputs during the earlier years of monitoring resulted in increased phytoplankton biomass, which is now controlled by other factors, or (2) another factor, such as micronutrients supplied by the discharge (as TDS and TN are correlated, r ranging from 0.64 to 0.93 between 2007 and 2012 [Section 5]), is responsible for the initial increases in phytoplankton biomass, and TN has negligible influence on phytoplankton biomass.

The relationship between phytoplankton biomass and the ratio of N to P was generally poor (r <0.5; Figure 7-46). Visual evaluation of the pattern on the plot suggests that the higher ratios in 2011 and 2016, at a number of stations, which indicated a greater degree of N enrichment than in other stations and years, did not result in greater phytoplankton biomass. The ratio of N to P at each station was also examined relative to phytoplankton community structure on the MDS plots. No relationships were observed on these plots (Figures 7-47 and 7-48).

A LINKTREE analysis was also carried out to visually interpret the influence of TDS, TN, and TP on the MDS plots (Figures 7-49 and 7-50). Based on the LINKTREE interpretation, between 2007 and 2016, the phytoplankton community differences were largely driven by differences in TDS and TN, which tended to be intercorrelated (Section 5). Between 2008 and 2012, the NF areas grouped together and had TDS concentrations >8.9 mg/L and TN concentrations >173 but <200 µg/L, while the majority of the FF areas grouped together at <152 µg/L TN. The NF areas in 2014, 2015 and 2016 grouped together and had TDS concentrations >14.9 mg/L and TN concentrations >200 µg/L. The FF areas in 2013 grouped together and had TDS concentrations >11.0 and <14.9 mg/L and TN concentrations >195 and <215 µg/L, and the FF areas in 2016 grouped together and had TDS concentrations <11.0 mg/L and TN concentrations <169 µg/L.

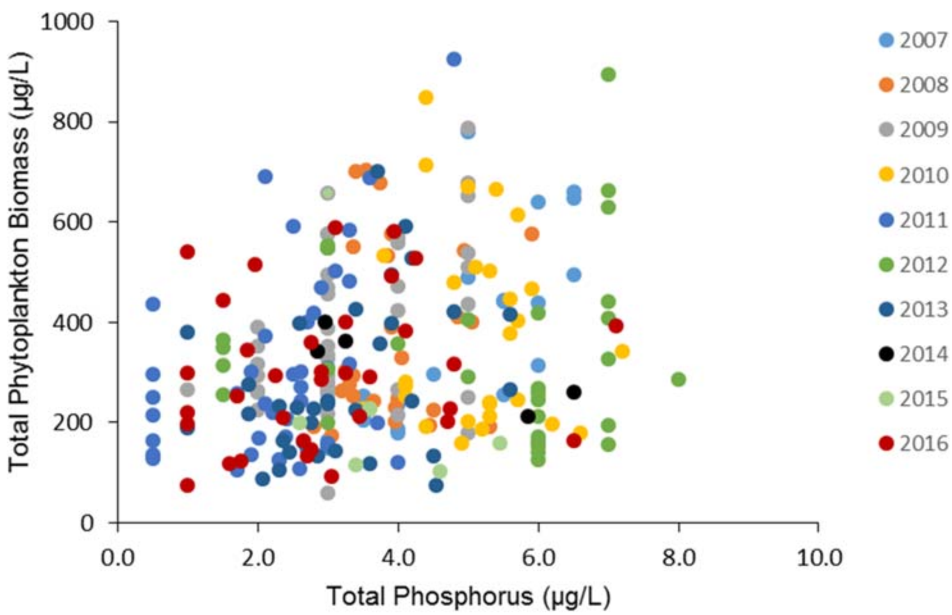
The results of the LINKTREE analysis are based on divisive clustering of the environmental data to provide an environmental interpretation of the community data. The clustering of the environmental dataset is compared to the community dataset and the assumption is made that if both datasets are clustering in a similar manner, then those environmental variables are likely driving the community changes. However, this may not be the case if a key driver for community structure was not included in the analysis. In addition, strong inter-correlations between variables (in this case between TN and TDS) may create uncertainty in the interpretation, because the effects of the inter-correlated variables cannot be separated. LINKTREE attempts local explanations rather than holistic ones, which can create divisions in the environmental data that may not fully translate into differences in the community data. A disadvantage of local explanations is that many environmental inequalities, in this case TDS and TN, can explain the same assemblage divisions. Overall, the LINKTREE analysis should be interpreted with caution.

Table 7-23 Relationship between Concentrations of Total Phosphorus and Total Nitrogen and Phytoplankton Biomass, 2007 to 2016

Year	Total phytoplankton biomass vs TP (*r*)	Total phytoplankton biomass vs TN (*r*)	Total phytoplankton biomass vs N to P (molar) ratio (*r*)
2007	0.76	0.69	0.55
2008	0.14	0.75	0.46
2009	0.45	0.63	0.19
2010	0.04	0.82	0.54
2011	0.41	0.60	<0.001
2012	0.03	0.55	0.25
2013	0.33	0.14	0.17
2014	-	-	-
2015	-	-	-
2016	0.14	0.17	<0.01

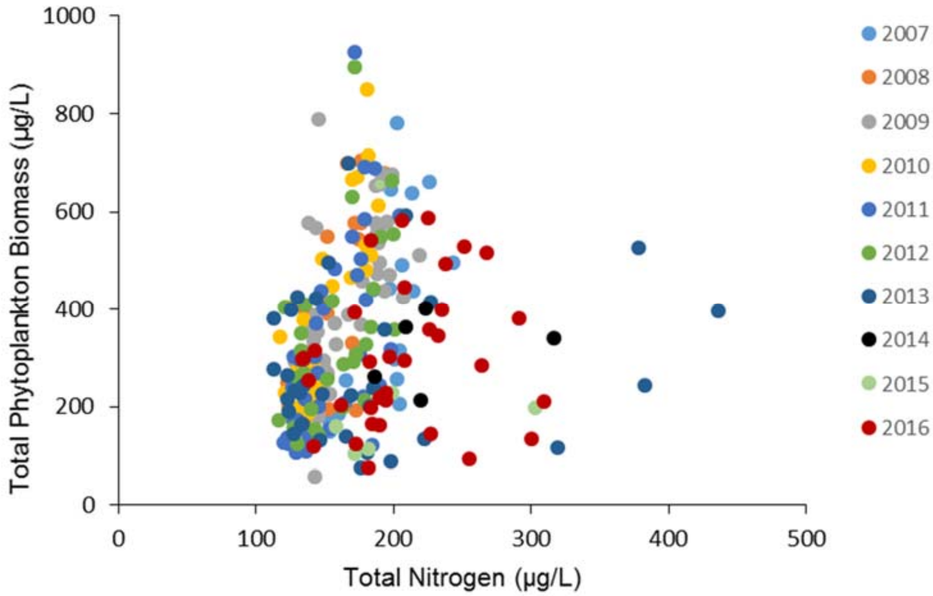
r = Pearson correlation co-efficient; TP = total phosphorus; TN = total nitrogen; N = nitrogen; P = phosphorus; - = insufficient data.

Figure 7-44 Total Phosphorus Concentration versus Total Phytoplankton Biomass, 2007 to 2016



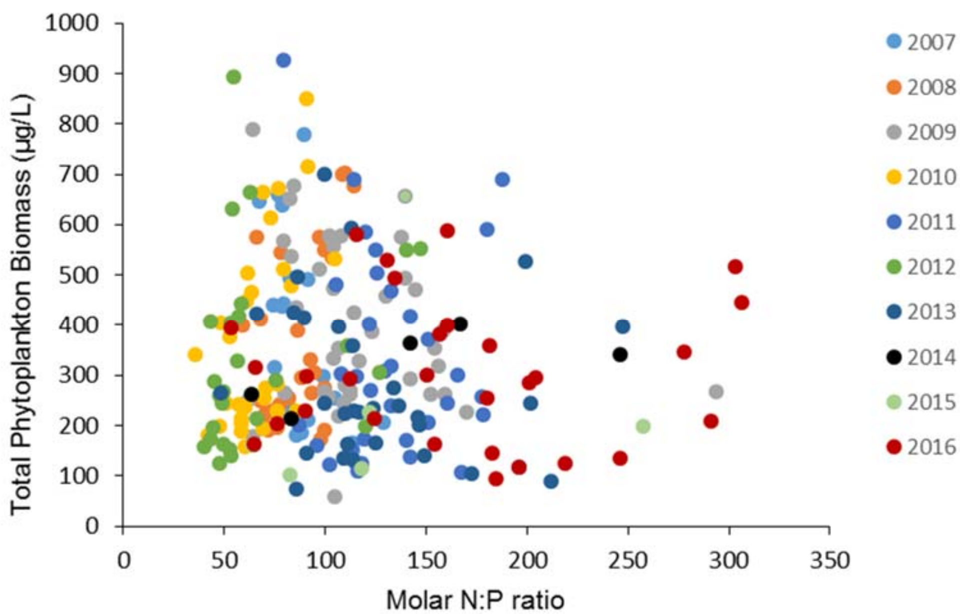
TP = total phosphorus.

Figure 7-45 Total Nitrogen Concentrations versus Total Phytoplankton Biomass, 2007 to 2016



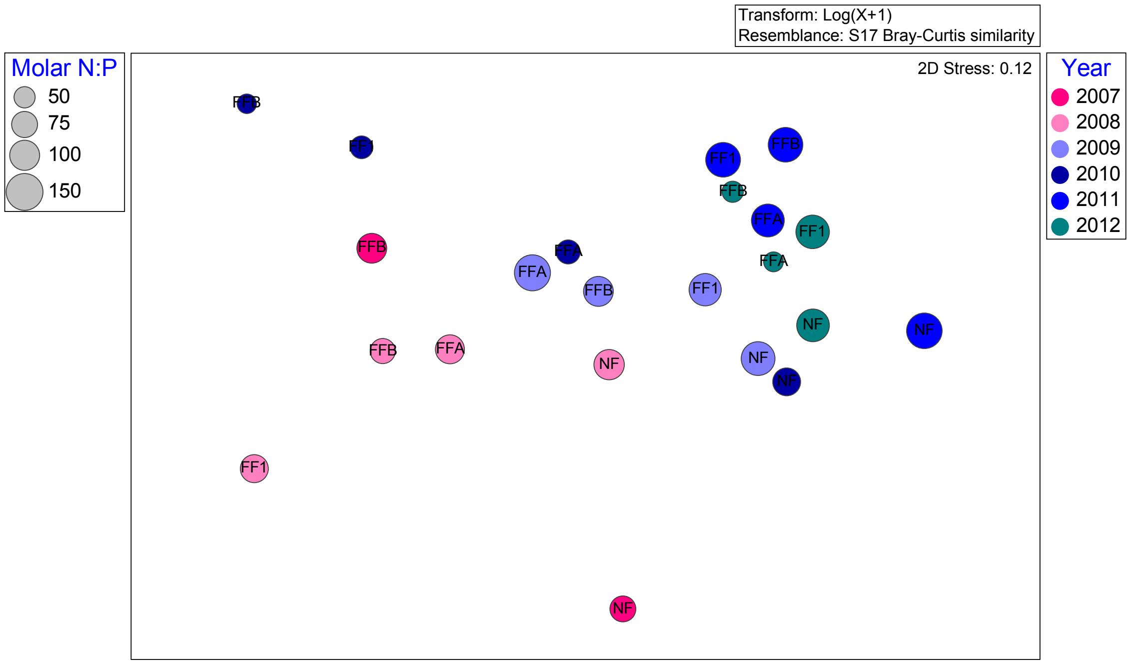
TN = total nitrogen.

Figure 7-46 Molar ratios of N to P relative to Total Phytoplankton Biomass, 2007 to 2016



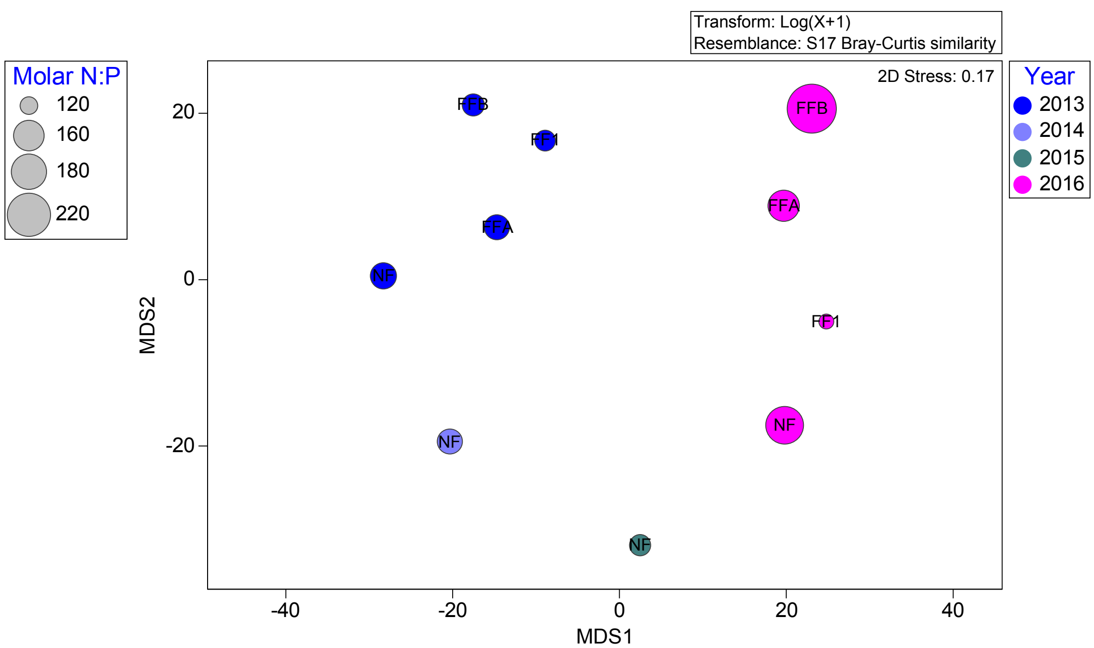
N = nitrogen; P = phosphorus.

Figure 7-47 Ordination the Phytoplankton Community in the NF and FF areas, with symbols scaled to the N to P (molar) ratio, 2007 to 2012



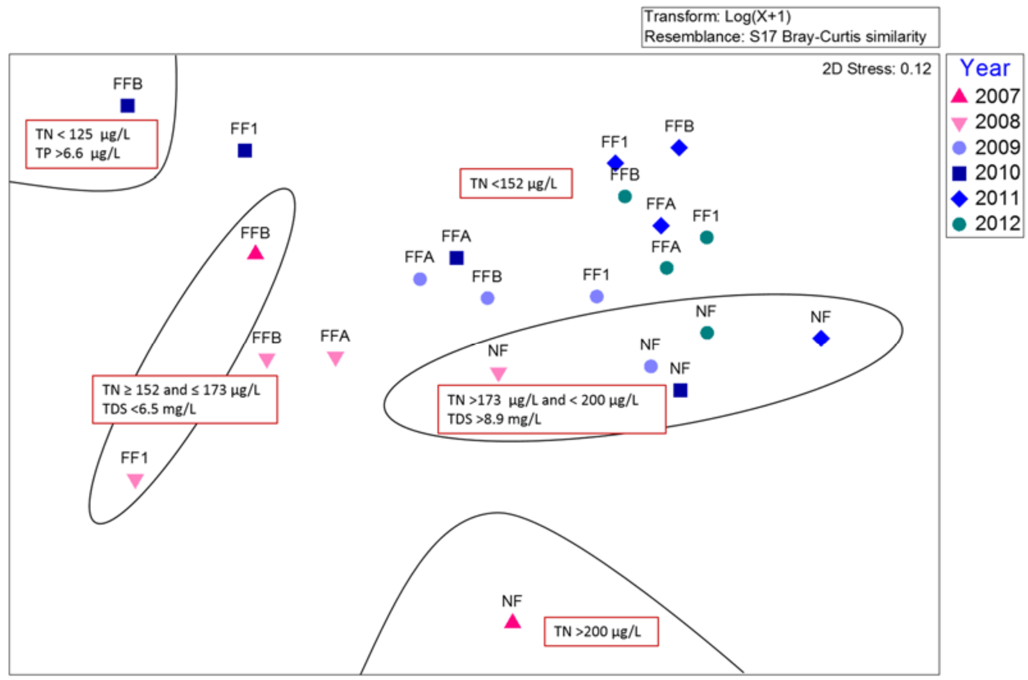
N = nitrogen; P = phosphorus; NF = near-field area; FF = far-field areas.

Figure 7-48 Ordination the Phytoplankton Community in the NF and FF areas, with symbols scaled to the N to P (molar) ratio, 2013 to 2016



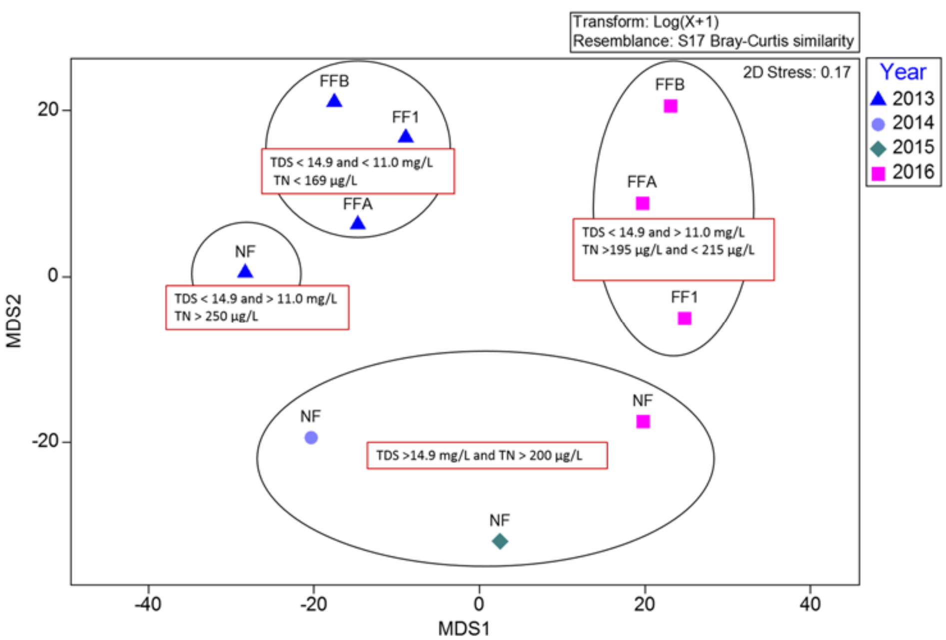
Notes: MDS = metric multidimensional scaling axes 1 and 2; N = nitrogen; P = phosphorus; NF = near-field area; FF = far-field areas.

Figure 7-49 LINKTREE Results for Phytoplankton Community Biomass, 2007 to 2012



TDS = total dissolved solids; TN = total nitrogen; NF = near-field area; FF = far-field areas; mg/L = milligrams per litre; <= less than; > = greater than.

Figure 7-50 LINKTREE Results for Phytoplankton Community Biomass, 2013 to 2016



MDS = metric multidimensional scaling axes 1 and 2; TDS = total dissolved solids; TN = total nitrogen; NF = near-field area; FF = far-field areas; mg/L = milligrams per litre; <less than; >greater than.

7.3.4.5 Overall Interpretation

Overall, water clarity and temperature regimes in Lac de Gras are suitable to support healthy phytoplankton and zooplankton communities. Water chemistry in Lac de Gras generally remains below aquatic life benchmarks and toxic effects are not being observed. The concentrations of TDS and associated ions have increased over time, and therefore, more micronutrients are available to the plankton communities. There has also been a shift in the nutrient balance in Lac de Gras, towards greater N to P ratios.

The increase in micronutrients and the shift in the N to P nutrient balance have resulted in a shift in phytoplankton community composition in Lac de Gras. Phytoplankton biomass and chlorophyll *a* concentrations (Section 5) in the NF area have fluctuated from within the normal ranges to both above and below the normal ranges throughout the time series; in the FF areas, they were generally within the normal ranges. The phytoplankton MDS results confirm that changes over time in phytoplankton community structure have been occurring in the NF area of Lac de Gras, especially between 2003 to 2007, and that conditions in 2016 differed from conditions in 2013 throughout the lake, but the NF and FF areas in these two years were more similar than observed in previous years. Overall, the phytoplankton community in Lac de Gras continued to be exhibit a Mine-related nutrient enrichment effect in the NF area.

Zooplankton biomass has been more variable in the NF and MF areas compared to the FF areas, but generally has remained within the normal range. Excursions in the NF area above the normal range occurred in 2008 and 2011, and below the normal range in 2016. In addition, a decreasing temporal trend in zooplankton biomass was observed at the MF3-4 station. The zooplankton nMDS ordination plot indicated that changes over time in zooplankton community structure have been occurring in the NF area of Lac de Gras, and that conditions in 2016 differed from previous years.

In 2016, zooplankton community structure was different in the NF and MF areas of Lac de Gras relative to the FF areas, with the greatest differences in the MF3 area. The lower biomass in 2016, especially in the MF3 area, triggered an Action Level 2; however, the Action Level 2 exceedance is an early-warning indication of potential Mine-related changes, and is not of immediate concern regarding toxicity to the plankton community in Lac de Gras. The difference between mean total zooplankton biomass in 2016 in the area subject to greatest effluent exposure (NF area) and the FF areas was 17%, which is low compared to the typical degree of variation observed among areas within a large lake (Golder 2017c) and the mean total zooplankton biomass in the NF area was within the normal range (Golder 2017c). A similar response was not observed in total zooplankton biomass measured as AFDM, which was found to be significantly greater in the NF area compared to the FF areas by the eutrophication indicators component of the AEMP (Section 5). The water quality component of the AEMP did not document effluent toxicity to invertebrates or algae. Although subject to uncertainty, the decrease in zooplankton biomass observed in 2016 may have been caused by increased turbidity associated with dike construction during the open-water season (i.e., in the MF3 area). This effect may be a one-time occurrence (to be verified as part of the 2017 AEMP Annual Report).

7.3.5 Nitrogen Fixing Cyanobacteria

The WLWB has requested that as part of the *2014 to 2016 Aquatic Effects Re-evaluation Report,* a review of nutrient ratios to evaluate potential nutrient limitation and effects on N_2-fixing cyanobacteria be carried out (2016 AEMP Annual Report Commitment #61). To evaluate the potential role of nutrient limitation and subsequent effects on N_2-fixing cyanobacteria, the total biomass of N_2-fixing cyanobacteria per year and sampling area (i.e., NF and FF areas) was calculated and plotted (Section 7.2.3.4). The linear relationship between TN, TP, the ratio of N to P and N_2-fixing cyanobacteria biomass was also examined for each year, to determine whether changes in nutrient concentrations in the NF and FF areas of Lac de Gras are causing changes in the biomass of cyanobacteria capable of atmospheric nitrogen fixation.

Many cyanobacterial species are capable of fixing atmospheric nitrogen, because of this ability, they usually dominate in lakes with low N to P ratios (Huisman and Hulot 2005). According to Schindler (1977), the critical N to P molar ratios at which cyanobacteria, both N_2-fixing and non-fixing, tend to dominate in lakes are found in the range of 11 to 22. Blooms related to species capable of fixation of atmospheric N can be linked to periodic N limitation. However, numerous lakes with clear N limitation are not always dominated by N_2-fixing cyanobacteria, and lakes with high N concentrations will still have N_2-fixing cyanobacteria present in their phytoplankton communities (Noges et al. 2008).

Howarth et al. (1999) suggested that planktonic N_2-fixation can occur if the N to P molar ratio is equal to or lower than the Redfield ratio of 15, while Smith et al. (1995) and Noges et al. (2008), suggested higher N to P ratios. Smith et al. (1995) showed that an N to P molar ratio of 48 provides a distinct boundary between lakes dominated by N_2-fixing cyanobacteria and lakes where few N_2-fixing cyanobacteria occur, and Noges et al. (2008) found that N_2-fixing cyanobacteria achieved high biomass at N to P molar ratios of 66 and 88 in large, shallow, temperate Northeast European lakes.

The mean N to P molar ratio in the NF and FF areas of Lac de Gras have exceeded the lower range of these ratios since 2007 :

- NF area: 75 to 191
- FF1 area: 57 to 172
- FFB area: 42 to 348
- FFA area: 44 to 419

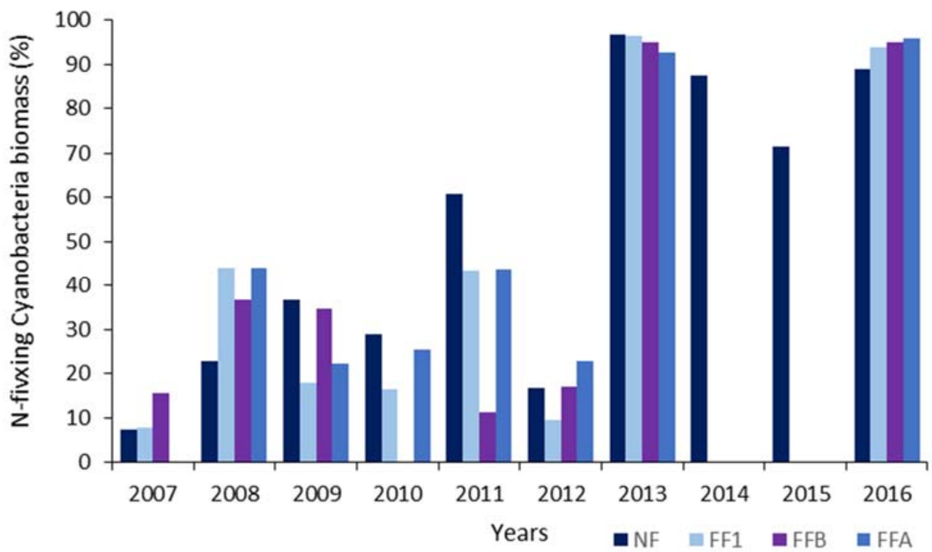
Ratios near the lower ends of these ranges fell within the suggested optimal N to P mass ratios for N_2-fixing cyanobacteria suggested by Smith et al. (1995) and Noges et al. (2008); therefore, the relationship between the nutrients and N_2-fixing cyanobacteria biomass was explored further.

The relative proportion of N_2-fixing cyanobacteria biomass (expressed as the percentage of total cyanobacteria biomass), appeared to have increased in the NF and FF areas of Lac de Gras (Figure 7-52). Greater relative biomass was observed after 2013; however, this increase in biomass of N_2-fixing cyanobacteria may be the result of the change in taxonomist in 2013, rather than an increase in N_2-fixing cyanobacteria biomass.

The relationships between N_2-fixing cyanobacteria biomass and TP, TN and the molar ratio of N to P were examined for all years from 2007 to 2016 in the NF and FF areas (Figures 7-53 to 7-55). No clear relationships (r <0.32) between nutrient concentrations or ratios (i.e., TN, TP, and the ratio of N to P) and N_2-fixing cyanobacteria biomass were observed in any year.

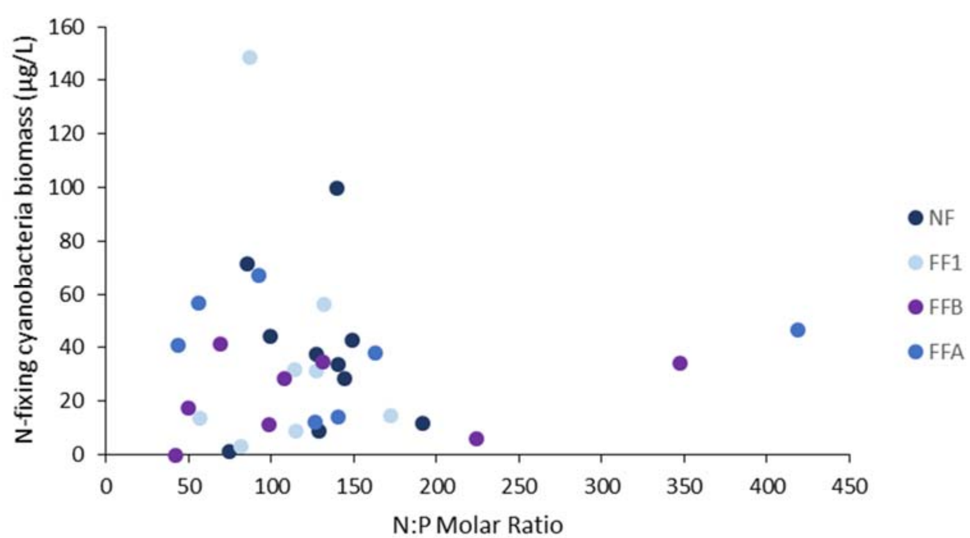
In summary the results regarding N_2-fixing cyanobacteria biomass suggest that this analysis is likely confounded by the change in taxonomist in 2013; therefore, although relationships between N_2-fixing cyanobacteria biomass and nutrient concentrations or nutrient ratios were briefly evaluated, the results are unlikely to be reliable or useful in interpreting trends in cyanobacteria biomass.

Figure 7-51 Relative Biomass of Nitrogen-fixing Cyanobacteria in the NF and FF areas, 2007 to 2016



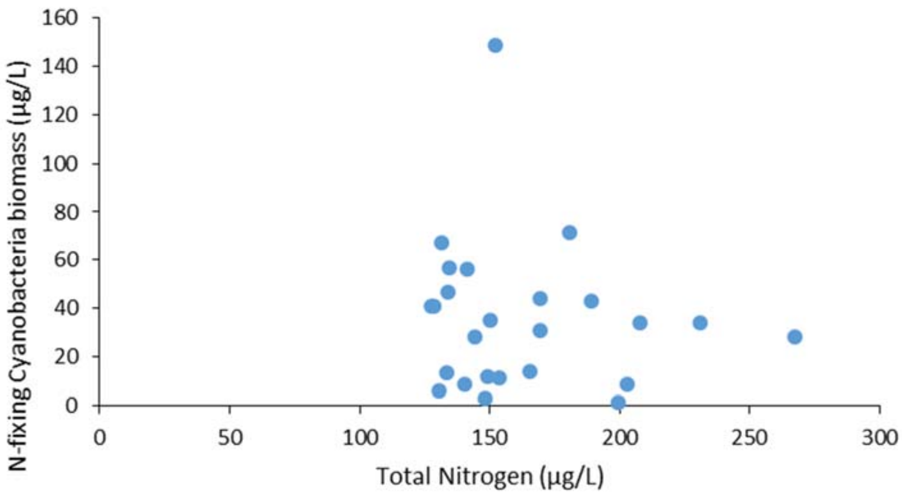
Notes: change in taxonomist in 2013; NF = near-field; FF = far-field; N = nitrogen.

Figure 7-525 Relationship between the mean N to P (molar) ratio and mean Nitrogen-fixing Cyanobacteria biomass in the NF and FF areas, 2007 to 2016



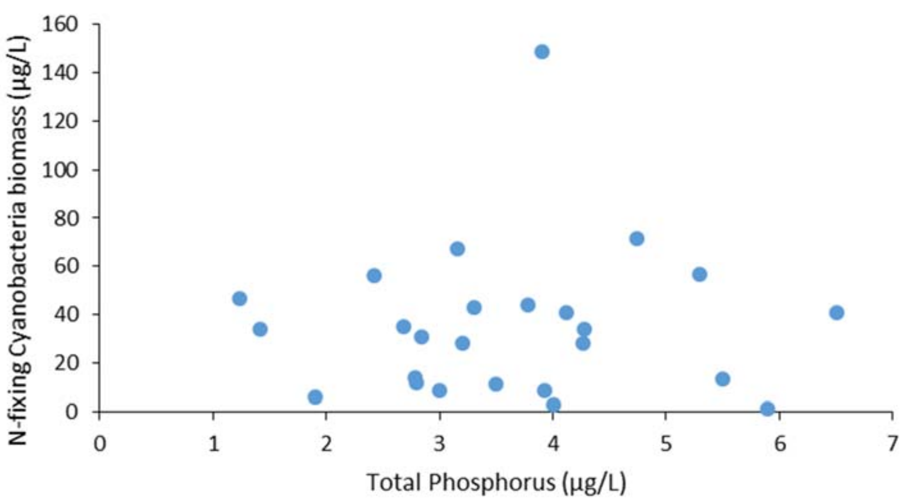
Notes: NF = near-field; FF = far-field; µg/L = micrograms per litre; N = nitrogen; P = phosphorus.

Figure 7-53 Relationship between mean Total Nitrogen Concentration and mean Nitrogen-fixing Cyanobacteria Biomass in the NF and FF areas, 2007 to 2016



Notes: NF = near-field; FF = far-field; µg/L = micrograms per litre; N = nitrogen.

Figure 7-543 Relationship between mean Total Phosphorus Concentration and mean Nitrogen-fixing Cyanobacteria Biomass in the NF and FF areas, 2007 to 2016



Notes: NF = near-field; FF = far-field; µg/L = micrograms per litre; N = nitrogen.

7.3.6 Comparison to EA Predictions

No specific predictions were made in the EA regarding plankton communities, however, an increase in primary productivity in part of Lac de Gras was predicted, resulting from the input of nutrients (particularly phosphorus) from the Mine effluent discharge. This increase can be expected to also result in increased secondary productivity (i.e., zooplankton and benthic invertebrates). Increased phytoplankton biomass was observed in NF and MF areas of Lac de Gras, above the normal range in 2009 to 2013 and 2016, which is consistent with EA predictions regarding primary productivity. Occasional peaks in zooplankton biomass in NF and MF areas (in 2009 and 2011) were also consistent with the increased phytoplankton productivity. A more detailed evaluation of EA predictions related to nutrient enrichment, where the affected area is estimated by year, is provided in Section 6 (Eutrophication Indicators).

7.4 Summary and Conclusions

Overall Conclusions

- Conditions in Lac de Gras are suitable for persistence of phytoplankton and zooplankton communities typical of large subarctic lakes. Lac de Gras remains a clear, soft water lake, with generally neutral to acidic pH within the top and middle of the water column. In all areas of the lake, the water column has remained well-oxygenated throughout the time series. Nutrient concentrations remain low, within the oligotrophic range, while concentrations of TDS continue to increase.
- Effect ratings for the WOE assessment have remained within the low to moderate level from 2007 to 2016 for zooplankton biomass and zooplankton and phytoplankton community structure. For phytoplankton biomass, effect ratings have been consistently moderate to high, with the exception of 2013, which yielded a low rating.
- The assessment of effects according to Action Levels addresses the toxicological impairment hypothesis. From 2007 to 2014, neither phytoplankton nor zooplankton community variables triggered an Action Level in Lac de Gras. However, Action Level 1 was triggered for phytoplankton biomass in 2015, and Action Level 2 was triggered for zooplankton biomass in 2016. The Action Level exceedances for phytoplankton biomass and zooplankton biomass provide an early-warning indication of potential Mine-related changes, and are not of immediate concern regarding toxicity to the plankton community in Lac de Gras.
- The extent of reduced zooplankton biomass in 2016, based on the affected stations, was calculated to be 103 km^2. Compared to the total surface area of the lake (573 km^2), the area demonstrating effects on zooplankton biomass represents 18% of the lake area.
- The plankton communities in Lac de Gras continue to exhibit a Mine-related nutrient enrichment effect in the NF and MF areas.

Phytoplankton Community

- An increase in taxonomic richness in the NF between 2002 and 2012, within or above the normal range was observed; between 2013 and 2016 no trend was observed. Richness at stations in the FF area generally fell within the 2007 to 2010 normal range from 2007 to 2012, but in 2013 and 2016, richness was at or above the upper limit of the 2007 to 2010 normal range, and within the 2013 normal range. No significant increases or decreases in taxonomic richness were observed in the FF areas.
- Phytoplankton biomass in the NF area fluctuated from within the normal range (2013 and 2014) to both above (2007 to 2012 and 2016) and below (2015) the normal ranges throughout the period monitored. Biomass in the FF areas followed comparable trends to the NF and MF areas, but stations in the FF areas were generally within the normal ranges throughout the time series.
- Temporal trends in biomass of the major phytoplankton groups differed among sampling areas and among years in Lac de Gras between 2002 and 2012, with the exception of microflagellate and cyanobacteria biomass, which increased in most areas between 2002 and 2012. Between 2013 and 2015, cyanobacteria and microflagellate biomass showed a decreasing trend in the NF area; in 2016 microflagellate biomass increased.
- Microflagellates and cyanobacteria contributed a large proportion of relative biomass in the NF, MF and FF areas between 2007 and 2016 and were often the dominant groups.
- The phytoplankton MDS results indicate that changes over time in phytoplankton community structure have been occurring in the NF area of Lac de Gras, especially between 2003 and 2007, and that conditions in 2016 differed from conditions in 2013 throughout the lake, but the NF and FF areas in these two years were more similar than observed in previous years.

Zooplankton Community

- Temporal trends in zooplankton taxonomic richness were not observed and richness generally remained within the normal range in the NF and FF areas of Lac de Gras from 2008 to 2016.
- Zooplankton biomass has been more variable in the NF and MF areas compared to the FF areas, but has generally remained within the normal range. Excursions in the NF area above the normal range occurred in 2008 and 2011, and below the normal range in 2016. A clear decreasing temporal trend in zooplankton biomass was observed at the MF3-4 station.
- Temporal trends in the biomass of the major zooplankton groups differed among sampling areas in Lac de Gras between 2008 and 2016.
- Cladoceran biomass exceeded the normal range in the NF area in most years from 2008 to 2013, but from 2014 to 2016 it has been within the normal range; the relative biomass of this group increased until 2013 and then decreased in 2016.
- A decrease over time was observed in calanoid copepod biomass and relative biomass in all areas. Calanoid biomass had decreased from 2009 to 2016 in all areas, with the exception of FF1, to near or below the lower limit of the normal range. In the NF area in 2014 and 2015, calanoid biomass increased to within the normal range; however, low biomass was observed again in all areas in 2016.

- Cyclopoid copepod and rotifer biomass have generally been within or above the upper limit of the normal range between 2008 and 2016.
- The zooplankton nMDS ordination plot indicates that changes over time in zooplankton community structure have been occurring in the NF area of Lac de Gras, and that conditions in 2016 differed from previous years. In 2016, zooplankton community structure was different relative to the FF areas in the NF and MF areas of Lac de Gras, with the greatest differences in the MF3 area.

Nutrient Relationships

- Generally, a moderate to strong relationship was observed between phytoplankton and zooplankton biomass and TDS concentrations (an indicator of effluent exposure) during each year of monitoring. Results of multivariate analysis suggest that TDS and TN are the main drivers influencing phytoplankton community structure. This analysis is based on the assumption that if both datasets are clustering in a similar manner, then the environmental variables are likely driving the community changes. However, this may not be the case, as a key driver for community structure may not have been included in the analysis. In addition, strong inter-correlations between variables (in this case between TN and TDS) may create uncertainty in the interpretation, because the effects of the inter-correlated variables cannot be separated. Overall, the multivariate analysis results should be interpreted with caution.
- Generally, relationships between phytoplankton biomass and TP, and the ratio of N to P were poor ($r<0.5$). The relationship between phytoplankton biomass and TN was moderate to strong ($r>0.5$) between 2007 and 2012, but from 2013 to 2016, the relationship was poor ($r<0.5$). The ratios of TN to TP at each station were also examined relative to phytoplankton community structure on the MDS plots, but no relationships were observed.
- No clear relationship was observed between N_2-fixing cyanobacteria biomass and concentrations of nutrients (i.e., TN, TP) or the N to P ratio. This analysis was hindered by the difference in cyanobacteria abundance resulting from the switch in taxonomist in 2011.

8 BENTHIC INVERTEBRATES

8.1 Introduction

"Benthic invertebrates" is a general term referring to aquatic invertebrates (i.e., crustaceans, snails, clams, insects and worms) that inhabit the bottoms of lakes and streams. Benthic invertebrates play an important role as secondary produces and as a primary food source for fish in lake ecosystems. Common benthic invertebrates found in Lac de Gras are:

- insects: midges of the family Chironomidae
- mollusks: fingernail clams of the family Pisidiidae
- oligochaetes: aquatic worms of the families Naididae and Lumbriculidae

These three groups contribute 98% of the total invertebrate density in Lac de Gras in all years and sampling areas combined (1996 to 2016). Chironomidae alone account for 74% of total invertebrate density, with Pisidiidae being the second most abundant group, contributing 21% of total invertebrate density.

8.1.1 Background

DDMI has been conducting studies and monitoring programs relating to the aquatic ecosystem of Lac de Gras since 1994, with AEMP data collected under AEMP study designs Version 1.0 (2001[9] to 2006), Version 2.0 (2007 to 2011), Version 3.0 (2012 to 2016), and most recently under Version 4.0 (2017). Since there is potential for Mine effluent to affect Lac de Gras, the benthic invertebrate program has been included as a component of the AEMP since Version 1, and is monitored only during comprehensive years (i.e., is not monitored in interim years).

The objective of the benthic invertebrate community program is to evaluate whether the benthic invertebrate community of Lac de Gras is affected by effluent discharged from the Mine and, if so, to classify and evaluate the type of effect. Confirmation of the predictions set forth in the EA (DDMI 1998a) is an additional objective. More specifically, the program was been designed and implemented to identify effects on benthic invertebrate communities by comparing community indices (i.e., density, richness, etc.) between the NF and FF areas, and along the effluent exposure gradient represented by the NF, MF, and FF areas combined, using statistical analyses.

8.1.2 Component History

Benthic invertebrates have been monitored in Lac de Gras since the 1996 and 1997 baseline surveys in support of the EA (Table 8-1). Previous AEMP sampling events were annual from 2001 to 2011, and included one sampling event prior to effluent discharge to Lac de Gras, which began in April 2002. Following 2011, benthos were sampled during comprehensive AEMP years, in 2013 and 2016. Samples were collected during the late open-water season in August, although in some years, sampling extended from early August to late September. Benthic invertebrate communities were sampled once over the past three years, in 2016, under AEMP Version 3.5.

8.2 Methods

Benthic invertebrate sampling areas are sampled in conjunction with other AEMP components, and were selected based on exposure to the Mine effluent (Golder 2011b). Sampling areas consist of the NF area, three MF areas (MF1, MF2-FF2, and MF3) and three FF areas (FF1, FFA, and FFB) (Figure 2-5; Table 8-1). The FF areas were formerly designated as reference areas, but are now exposed to Mine effluent, as indicated by elevated concentrations of TDS and associated ions contributed by the Mine discharge. Three transect lines, referred to as MF areas, are sampled between the NF and FF areas. The NF-MF1-FF1 transect extends from the NF area towards the FF1 area to northwest of the east island. The NF-MF2-FF2 transect extends from the NF area to the northeast, towards the Lac du Sauvage inlet. The NF-MF3-FFB-FFA transect extends from the NF area to the south and then turns west, towards FFB and

[9] One year of baseline data was also collected in 2000 under Version 1. Data were not summarized from this sampling event, due to data quality issues associated with taxonomic identifications.

FFA areas. Within each NF and FF sampling areas, clusters of five replicate stations are sampled at similar levels of exposure to the effluent, whereas stations in the MF areas are exposed to varying levels of effluent and thus do not serve as replicates.

Station selection was constrained by water depth, which was kept as close to 20 metres (m) as possible to prevent confounding the study by variability in depth, which has been previously demonstrated to strongly influence the benthic community in Lac de Gras (Golder 1997). Six subsamples, each consisting of a single Ekman grab with a sampling area of 0.023 m^2, are collected at each station and are combined to form a composite sample, which is sieved through a 500 μm mesh Nitex screen. Although taxonomists have varied through the early part of the monitoring period, the same taxonomist (J. Zloty, PhD, independent consultant) analyzed samples collected in 2007 and thereafter.

Table 8-1 Summary of Baseline and AEMP Benthic Invertebrate Community Data, 1996 to 2016

Waterbody	Area	Station[a]	UTM Coordinates[b] Easting (m)	UTM Coordinates[b] Northing (m)	Distance from Diffusers[c] (km)	Year 1996[d]	1997[d]	1999	2001-2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Lac de Gras[f]	Near-field	NF1	535740	7153854	0.4					X	X	X	X	X		X			X
		NF2 (N7)	536095	7153784	0.5	X				X	X	X	X	X		X			X
		NF3	536369	7154092	0.9					X	X	X	X	X		X			X
		NF4	536512	7154240	1.1					X	X	X	X	X		X			X
		NF5 (LDG-NF)	536600	7153864	1.0				X	X	X	X	X	X		X			X
	Mid-field 1	MF1-1	535008	7154699	1.5					X	X	X	X	X		X			X
		MF1-2	533682	7155356	2.9					X	X	X	X	X					
		MF1-3	532236	7156276	4.7					X	X	X	X	X		X			X
		MF1-4	532494	7157657	7.2					X	X	X	X	X					
		MF1-5	528432	7157066	8.5											X			X
	Mid-field 2	MF2-1	538033	7154371	2.4					X	X	X	X	X		X			X
		MF2-2 (F14, LDG-MF)	539198	7154643	3.7	X	X		X	X	X	X	X	X					
		MF2-3	540365	7156045	5.4					X	X	X	X	X		X			X
		MF2-4	540955	7157359	6.9					X	X	X	X	X					
	Far-field 2[e]	FF2-1	541500	7159522	9.3					X	X	X	X	X					
		FF2-2	541588	7158561	8.3					X	X	X	X	X		X			X
		FF2-3	543478	7159267	10.1					X	X	X	X	X					
		FF2-4	543752	7158945	10.2					X	X	X	X	X					
		FF2-5	544724	7158879	11.4					X	X	X	X	X		X			X
	Mid-field 3	MF3-1	537645	7152432	2.7					X	X	X	X	X		X			X
		MF3-2	536816	7151126	4.2					X	X	X	X	X		X			X
		MF3-3	536094	7148215	7.2											X			X
		MF3-4	532545	7147011	11.0					X	X	X	X	X		X			X
		MF3-5	528956	7146972	14.6											X			X
		MF3-6	525427	7148765	18.5					X	X	X	X	X		X			X
		MF3-7	521859	7150039	22.3											X			X
	Far-field 1	FF1-1	525430	7161043	13.6						X	X	X	X		X			X
		FF1-2	524932	7159476	12.9						X	X	X	X		X			X
		FF1-3	526407	7160492	12.8						X	X	X	X		X			X
		FF1-4	526493	7159058	11.4							X	X	X		X			X
		FF1-5	526683	7161824	12.8					X	X	X	X	X		X			X

Table 8-1 Summary of Baseline and AEMP Benthic Invertebrate Community Data, 1996 to 2016

Waterbody	Area	Station[(a)]	UTM Coordinates[(b)]		Distance from Diffusers[(c)] (km)	Year													
			Easting (m)	Northing (m)		1996[(d)]	1997[(d)]	1999	2001-2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Lac de Gras(	Far-field B	FFB-1	516831	7148207	26.4					X	X	X	X	X		X			X
		FFB-2	518473	7150712	25.0					X	X	X	X	X		X			X
		FFB-3	518048	7147557	25.2					X	X	X	X	X		X			X
		FFB-4	515687	7150036	27.6					X	X	X	X	X		X			X
		FFB-5	516533	7150032	26.8					X	X	X	X	X		X			X
	Far-field A	FFA-1	506453	7154021	36.8						X	X	X	X		X			X
		FFA-2	506315	7155271	38.3							X	X	X		X			X
		FFA-3	505207	7153887	38.7					X	X	X	X	X		X			X
		FFA-4	503703	7154081	40.2					X	X	X	X	X		X			X
		FFA-5 (LDG-FF)	505216	7156657	40.0				X		X	X	X	X		X			X

Note: Shading identifies stations that were discontinued, as per the AEMP Study Design, Version 3.0 (Golder 2011b). These stations are not included in the current (2014 to 2016) Aquatic Effects Re-evaluation Report, with the exception of MF2-2; MF2-2 and MF2-3 station data were pooled for trend analysis.
a) Current AEMP station names are presented, and historical sampling station names are provided in parentheses.
b) UTM coordinates are reported as Zone 12, NAD 83.
c) Approximate distance from the Mine effluent diffusers along the most direct path of effluent flow.
d) For 1996 and 1997, information is provided only for stations located within AEMP areas.
e) From 2012 onwards, the FF2 area is considered to be part of the NF-MF2-FF2 transect.
UTM = Universal Transverse Mercator coordinate system; NAD = North American Datum; X = benthic invertebrate samples were collected.

8.2.1 Data Sources

8.2.1.1 Baseline Programs (1996 and 1997)

Baseline benthic invertebrate surveys using quantitative methods in areas of Lac de Gras that were sampled in subsequent years were completed in 1996 and 1997. These surveys used a 6 x 6" (15.2 x 15.2 cm) Ekman grab to sample benthic invertebrates in deep-water areas of Lac de Gras. Three grabs per station were taken from three stations in 1996, and six grabs per station were collected from four closely-spaced stations in 1997. Two of the baseline sampling locations were close to present day AEMP sampling stations; these were N7 (1996; near the current NF-2 station) and F14 (1996 and 1997); near the current MF2-1 station).

8.2.1.2 AEMP Version 1.0 Data (1996 to 2006)

Using field methods similar to those used in the baseline studies, the 2001 to 2006 AEMP programs sampled benthic invertebrates at three closely spaced stations in the NF area (LDG-NF; near the current NF-5 station), MF2 area (LDG-MF; near the current MF2-1 station) and just north of the FFA area (LDG-FF; near the current FFA-5 station).

Although field methods were similar among sampling programs and were also similar to those used during the most recent AEMP surveys, three factors limit comparability of data among programs from the baseline and AEMP Version 1.0 data:

- During the 1996 and 1997 baseline surveys, a 250 µm mesh sieve was used to screen samples in the field and during sample processing in the laboratory, whereas 500 µm mesh screens were used from 2001 to 2016. Although results of an analysis of differences in data sets collected with different mesh sizes indicated that effects can be reliably detected using either mesh size (DDMI 2011b), abundances in the 1996 and 1997 data sets were expected to be greater than those reported during subsequent surveys.
- Large numbers of midges (Chironomidae) in the 1996 and 1997 baseline samples were only identified to subfamily/tribe or family taxonomic levels, whereas taxonomists processing samples from 2001 to 2013 identified midges to genus, with very few organisms left at higher taxonomic levels. Although the 1996 and 1997 data were included in the calculation of summary variables for the temporal trend analysis, results for these years need to be interpreted with caution.
- Sampling designs varied among programs, which may affect representativeness, spatial coverage and temporal variation of the summarized data. For example, the NF area mean for richness (i.e., the number of invertebrate taxa present) since 2007 is based on the richness from the five NF stations, with six subsamples collected from each station. In contrast, the NF area richness for 1996 and 1997 is based on three or four closely-spaced stations, with three and six subsamples per station, respectively. Consequently, the baseline datasets are expected to exhibit lower richness and more variation over time compared to area means calculated from five, more widely-spaced stations.

8.2.1.3 AEMP Version 2.0 Data (2007 to 2011)

Previous program changes are listed below from the *2011 to 2013 Aquatic Effects Re-evaluation Report* (Golder 2016a), and have been updated for historical continuity and clarity:

- Values for the Bray-Curtis index (BCI), a distance measure, have historically been calculated by comparing the benthic invertebrate community of each station to the median reference area community (using pooled reference area data). This method was consistent with the Metal Mining Technical Guidance for Environmental Effects Monitoring (MMER TGD; Environment Canada 2012). Since 2011, BCI values have been calculated using the "all pair-wise comparisons" method, as described in the 2011 AEMP Benthic Invertebrate Community Report (Golder 2012). Huebert et al. (2011) pointed out that using the reference median value as the basis for calculating Bray Curtis distance values would result in frequently finding effects where none exist, referred to as a Type I error. To correctly calculate BCI, Huebert et al. (2011) recommended that pairwise, among-area comparisons of individual reference and exposure stations be conducted to generate BCI values for statistical comparisons. To allow comparisons of results from the 2011, 2013 and 2016 AEMPs to earlier AEMP results, BCI values for 2007 to 2010 were recalculated using the pairwise method. Baseline data and 2001 to 2006 AEMP data were not used to calculate BCI values, because of taxonomist differences, and because the earlier sampling designs were inconsistent with the spatially extensive design implemented in 2007 (and thus did not allow comparisons of NF and MF area stations to stations in multiple FF areas, or FF areas to FF areas to quantify background variation).
- Gradient analyses have been previously done to attempt to link changes in the benthos to Mine effluent. These analyses were conducted with a linear regression of benthic community variables against the concentration of barium (an indicator of effluent concentration) along each transect (i.e., each MF area, and the NF and FF areas at the ends of the transects). Given that barium was no longer a useful effluent tracer, distance from the diffuser or TDS was used as an indicator of effluent exposure.
- Based on the results of the *2011 to 2013 Aquatic Effects Re-evaluation Report* (Golder 2016a) a change was made in the moderate level criterion for the WOE effect ratings. Before 2011, a significant regression combined with a coefficient of determination (r^2) >0.5 indicated a moderate effect ranking. A moderate effect rating now requires a statistical difference in slopes among years, and that this difference be indicative of a more pronounced effect.

8.2.1.4 AEMP Version 3.0 Data (2012 to 2013)

The benthic invertebrate community was sampled in 2013 under the *AEMP Study Design Version 3.0* (Golder 2011b). The number of stations in the MF areas were changed from AEMP Version 2.0 to Version 3.0 to better delineate the extent of effects and define gradients along each transect. The number of stations in the MF3 area was increased, and the number of stations in the MF1 and MF2-FF2 areas were decreased. There have been no changes to the AEMP stations for the benthic invertebrate community survey since Version 3.0.

8.2.1.5 AEMP Version 3.5 Data (2014 to 2016)

The benthic invertebrate community was sampled in 2016 as part of an AEMP monitoring component under the *AEMP Study Design Version 3.5* (Golder 2014a). Sampling methods were consistent with the *AEMP Study Design Version 2.0* (Golder 2013a). A number of updates outlined in the *AEMP Design Plan Version 4.0* (Golder 2016d) and in the WLWB directives (28 July 2015, 26 May 2016, 14 November 2016, and 2 March 2017 Decision Packages) were incorporated into the 2016 comprehensive Benthic Invertebrate Community Report (Golder 2017c).

Magnitude of effect was assessed by comparing benthic invertebrate variables in the NF area to the reference condition represented by the normal range for each variable. Normal ranges for Lac de Gras were defined in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b). The importance of effects observed on benthic invertebrate variables was determined according to the Action Level classification defined in the *AEMP Study Design Version 3.5* (Golder 2014a).

Information included to address WLWB directives or review comments (Table 1-1) specific to benthic invertebrates consists of the following:

- Decreases in Pisidiidae densities were reported in the NF area in 2016 and were attributed to nutrient enrichment (Golder 2017c). In their Reasons for Decision, the WLWB encouraged DDMI to incorporate literature findings providing support for nutrient enrichment as the main reason for the observed changes in Pisidiidae density, to address EMAB comment 93. A discussion regarding the likely causes of the observed changes in Pisidiidae density is provided in in Section 8.3.4.1.
- In response to submission of *AEMP Design Plan Version 4.0* (Golder 2016d), the WLWB provided Directive 4F, related to calculation of the BCI: “DDMI is to include a consideration of the recommendation regarding the taxonomic resolution issue for Bray-Curtis distance measures for benthic invertebrate data as part of the 2014 to 2016 Aquatic Effects Re-evaluation Report.” This directive is addressed in Section 8.3.4.2.

8.2.2 Data Handling

8.2.2.1 Data Screening

Initial screening of the benthic invertebrate community data was completed as part of the *2011 to 2013 Aquatic Effects Re-evaluation Report* (Golder 2016a) before data analysis, to identify unusually high (or low) values and decide whether to retain or exclude anomalous data from further analysis. The anomalous data screening approach was approved as part of the *AEMP Re-evaluation Report Version 3.2* (Golder 2016a). Data collected before 2016 were not re-screened for the current re-evaluation. Initial data screening of the 2016 benthic invertebrate community dataset did not identify potential anomalous data (Golder 2017c).

8.2.2.2 Data Preparation and Variable Selection

Data were prepared for trend analysis as per *AEMP Study Design Version 3.5* (Golder 2014a). Non-benthic invertebrates (i.e., Copepoda, Cladocera, and pupae) and benthic meiofauna (i.e., Nematoda), which were not quantitatively sampled, and terrestrial invertebrates were removed from the dataset. In addition, abundances per sample were converted to densities (i.e., number of organisms per square metre [no./m^2]) based on the bottom area of the sampling device and the number of subsamples collected.

Consistency in taxonomic level of identification was evaluated in previous years, and adjustments were made where appropriate. For example, certain taxa at some stations were reclassified to a higher level to match the taxonomic resolution of the remainder of the data. Historical data (1996 and 1997; 2001 to 2006) were excluded from BCI calculations due to different sampling designs and taxonomic resolutions between those periods and the more recent years of monitoring. Taxonomic identifications have been completed by the same taxonomist since 2007.

The following variables were included in the statistical analysis:

- total invertebrate density
- richness (total taxa per station at the lowest level of identification)
- dominance (percentage of the dominant taxon at a station)
- Simpson's diversity index (SDI)
- Simpson's evenness index (SEI)
- BCI (the distance measure, based on pair-wise comparisons of NF and FF stations)
- Chironomidae relative density
- densities of common taxa (consistent with those evaluated during previous AEMP benthic invertebrate community surveys and to provide continuity for the Aquatic Effects Re-evaluation Report):
  - Pisidiidae
  - *Procladius*
  - *Heterotrissocladius*
  - *Micropsectra*

Additional aspects of the benthic community structure were examined visually, and included community comparisons based on relative densities of major invertebrate groups and multivariate analysis.

8.2.2.3 Quality Assurance/Quality Control

The *Quality Assurance Project Plan Version 2.0* (Golder 2013a), or QAPP, outlined the QA/QC procedures employed to support the collection of scientifically-defensible and relevant data required to meet the objectives of the *AEMP Study Design Version 3.5* (Golder 2014a). The QAPP was designed so that field sampling, laboratory analysis, data entry, data analysis, and report preparation activities that follow the plan produce technically sound and scientifically defensible results. Descriptions of the QA/QC programs and the QC results for the benthic invertebrate community datasets are provided in the respective annual reports.

8.2.3 Data Analysis

8.2.3.1 Normal Ranges

The magnitude of effect on benthic invertebrate communities was evaluated by comparing benthic invertebrate community variables in the NF area to background values. Background values for Lac de Gras are those that fall within the range of natural variability, referred to as the normal range. Normal ranges were calculated using data from three AEMP FF areas (FF1, FFA, and FFB) from 2007 to 2010 (with some exceptions, as directed by the WLWB). Normal ranges were obtained from the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b) and are summarized in Table 8-2.

Table 8-2 Normal Ranges for Benthic Invertebrate Community Variables

Variable	Unit	Normal Range	
		Lower Limit	Upper Limit
Total Density	no./m^2	110	998
Richness	number of taxa	4.3	15.0
Dominance	%	21.7	57.3
Simpson's Diversity Index	-	0.60	0.86
Simpson's Evenness Index	-	0.23	0.76
Bray-Curtis Distance	-	0.45	0.81
Percent Chironomidae	%	46.9	91.3
Pisidiidae Density	no./m^2	0	206
Procladius Density	no./m^2	0	150
Heterotrissocladius Density	no./m^2	0	203
Micropsectra Density	no./m^2	0	172

Source: AEMP Reference Conditions Report Version 1.2 (Golder 2017b).
Note: Limits of normal ranges for numbers greater than 100 were rounded to whole numbers.
no./m^2 = number of organisms per square metre.

8.2.3.2 Temporal Trends

8.2.3.2.1 Time Series Plots

To visually evaluate temporal trends, benthic invertebrate community variables were plotted against time (year). The time series plots included a shaded region showing the normal range obtained from the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b). Temporal trends of community composition were also plotted as relative abundances of major invertebrate groups for each year in a stacked bar graph format.

8.2.3.2.2 Temporal Trend Analysis

Methods used for statistical trend analysis of AEMP data are described in Section 2.4.2. Data were pooled from the MF2-2 and MF2-3 stations for the benthic invertebrate analysis to retain the long-term dataset available for MF2-2, which was discontinued after 2011. The MF2-2 and MF2-3 stations are within 2 km of one another and are subject to a similar level of exposure to the Mine effluent.

8.2.3.3 Multivariate Analysis

Benthic invertebrate community structure was summarized using the non-parametric ordination method of multidimensional scaling (MDS; Clarke 1993). The MDS data were scaled in Primer, Version 7 for Windows (Clarke and Gorley 2016). Lowest taxonomic level (genus) benthic invertebrate data were square root transformed to improve the separation of the data among stations on the MDS plots and to reduce weighting of the analysis by the most abundant taxa. A non-metric MDS (nMDS) was run using a Bray-Curtis resemblance matrix generated from the transformed density data. Using rank order information, MDS determined the relative positions of stations in two dimensions based on community composition. Goodness-of-fit was determined by examining the Shepard diagrams as well as the stress values, which were calculated from the deviations in the Shepard diagrams. Lower stress values (i.e., less than 0.10) indicate less deviation and a greater goodness-of-fit (Clarke 1993). Points that fall close together on the MDS ordination plot represent samples with similar community composition; points that are far apart from each other represent samples with dissimilar community composition.

A similarity profile (SIMPROF) test was also carried out on the ordination data to identify meaningful clusters of important taxa (i.e., those taxa that behave in a coherent manner across areas) and to prevent over-interpretation of the MDS plots (Clarke et al. 2014). These SIMPROF clusters were superimposed on the MDS plots.

Station groups were consistent with the *AEMP Study Design Version 3.5* (Golder 2014a). Area means were determined for NF and FF (i.e., FF1, FFB, and FFA) areas for each year. The temporal community changes were assessed between the NF and FF areas. A trajectory line showing movement in the NF community over time was superimposed on these nMDS plots. Temporal MDS plots for each transect were also created: the NF-MF1-FF1, the NF-MF2-FF2 transect and the NF-MF3-FFB-FFA transect. The area means for the NF and FF areas were used in the ordinations, while for the MF data, data for single stations along each transect were used. A trajectory line showing movement in the NF community over time was also superimposed on the MDS plots.

An overall one-way analysis of similarities (ANOSIM) test was carried out on the Bray-Curtis resemblance matrix to confirm interpretation of the separation of points on the MDS ordinations and to confirm differences in community composition observed in the nMDS ordination plots were significant. In addition, an area-level ANOSIM was carried out on NF vs FF area Bray-Curtis resemblance matrix to test differences between areas contrasted with differences among replicates within each area for the NF and FF areas (Clarke et al. 2014). An R-statistic ranging from 0 to 1 is produced by the analysis; a strong difference in community composition is indicated by R = 1 and no difference between communities is indicated by R = 0. Significance of the R statistic was achieved when P >0.05 was observed, based on the five replicate stations per area.

To evaluate the effect of effluent exposure (using TDS [calculated] as an indicator of exposure) and habitat variables (i.e., particle size, as percent fines, and TOC) on the benthic invertebrate communities in Lac de Gras, bubble plots of each variable were superimposed on the MDS community plots. On these plots, the size of each bubble corresponds to the magnitude of the selected environmental variable at the equivalent station. The bubbles were then superimposed on the community MDS plot, providing an indication of whether communities separate in multidimensional space and along environmental gradients. If a particular environmental variable is driving the separation of communities among sampling areas, it should be visible on the nMDS plot as area/station groups having different sized bubbles. Bottom TDS data were used in this analysis in the NF and MF areas (where multiple depths were sampled for water quality), because these data best represent the exposure of the benthic community to the Mine effluent.

8.2.3.4 Weight-of-Evidence Effect Ratings

The results of the AEMP benthic invertebrate community surveys are integrated through the WOE evaluation process, which determines the strength of evidence supporting the two broad AEMP impact hypotheses (i.e., toxicological impariment and nutrient enrichment), as described in the *AEMP Study Design Version 3.5* (Golder 2014a). The WOE is not intended to determine the ecological significance or level of concern associated with a given change. The WOE effect ratings incorporate statistical comparisons of the NF and FF areas, and comparisons of the NF area to the normal range as defined in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b). The benthic invertebrate community data were assessed according to the WOE effect level ratings described in Section 10, and are summarized in Table 8-3.

Table 8-3 Weight-of-Evidence Effect Ratings for the Benthic Invertebrate Community

Measurement Endpoint	Early Warning/Low ↑ / ↓	Moderate ↑↑ / ↓↓	High ↑↑↑ / ↓↓↓
Comparison to FF Areas[(a)] Total invertebrate density Density of dominant Invertebrates (multiple endpoints) Richness Simpson's Diversity Index Dominance Simpson's Evenness Index Bray-Curtis Index	Statistically-significant change, NF vs FF areas	Low + NF area mean outside normal range	Moderate rating extending beyond NF

Table 8-3 Weight-of-Evidence Effect Ratings for the Benthic Invertebrate Community

Measurement Endpoint	Early Warning/Low ↑ / ↓	Moderate ↑↑ / ↓↓	High ↑↑↑ / ↓↓↓
Community Structure[a] Relative abundance of dominant Taxa	Difference in relative abundances fo dominant taxa in NF compared to FF areas	Difference in relative abundances of dominant taxa in NF and first MF station	Difference in relative abundances of major taxa extending further into MF or loss of major taxon from community in NF area
Gradient Analysis[a] Total invertebrate density Density of dominant invertebrates (multiple endpoints) Richness Simpson's Diversity Index Dominance Simpson's Evenness Index Bray-Curtis Index	Significant regression between endpoint and distance from the diffuser (an indicator of effluent exposure)	Low level effect AND a significant change in the slope from previous years' data that is indicative of a more pronounced effect	Not defined

Notes: Normal ranges for each measurement endpoint are defined and provided in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b). Percent Chironomidae is not listed as a separate variable, because it is included in the community structure analysis as one of the relative abundances of dominant taxa.

a) Applied separately for each measurement endpoint.

NF = near-field; MF = mid-field; FF = far-field.

8.2.3.5 Action Levels

The magnitude of effects to the benthic invertebrate community were categorized according to Action Levels described in the Response Framework presented in the *AEMP Study Design Version 3.5* (Golder 2014a). The main goal of the Response Framework is to ensure that significant adverse effects never occur. This is accomplished by requiring proponents to take actions at defined Action Levels, which are triggered well before significant adverse effects could occur. A significant adverse effect, as it pertains to aquatic biota, was defined in the EA for the Mine as a change in fish population(s) that is greater than 20% (Government of Canada 1999). The effect must have a high probability of being permanent or long-term and must occur throughout Lac de Gras. The Significance Thresholds for all aquatic biota, including plankton, are therefore related to impacts that could result in a change in fish population(s) that is greater than 20%.

Annual AEMP reports address the two impact hypotheses for Lac de Gras: toxicological impairment and nutrient enrichment. Biological Action Levels, and therefore the benthic invertebrate component, only address the toxicological impairment hypothesis. The nutrient enrichment hypothesis is assessed in the Eutrophication Indicators report (Section 5). Conditions required to trigger Action Levels 1 to 3 for the benthic invertebrate community are provided in Table 8-4. Conditions for Action Level 4 will be defined if

Action Level 3 is triggered. Defining higher Action Levels after initial effects are encountered is consistent with the draft guidelines for preparing a Response Framework in AEMPs (WLWB 2010; Racher et al. 2011).

Table 8-4 Action Levels for Benthic Invertebrate Community Effects

Action Level	Benthic Invertebrates	Extent	Action
1	The mean of a community index[(a)] significantly less than reference area[(b)] means	NF	Confirm effect
2	The mean of a community index[(a)] significantly less than reference area[(b)] means	Nearest MF station	Investigate cause
3	The mean of any measurement endpoint[(a)] less than normal range[(c)]	NF	Examine ecological significance Set Action Level 4 Identify mitigation options
4	To be determined[(d)]	-	Define conditions required for the Significance Threshold
5	Decline of community indices[(a)] likely to cause a >20% change in fish population(s)	FFA	Significance Threshold[(e)]

a) Refers to benthic invertebrate variables (total density, richness, Simpson's diversity index, Simpson's evenness index, Bray-Curtis index and densities of dominant taxa). The criterion for the Bray-Curtis index is a significantly higher mean value compared to the FF1, FFA and FFB areas.

b) Action Levels were assessed by comparing NF and MF areas to the FF1, FFA and FFB areas, which formerly served as reference areas.

c) Normal ranges were obtained from the AEMP *Reference Conditions Report Version 1.2* (Golder 2017b).

d) To be determined if an Action Level 3 effect is triggered.

e) Although the Significance Threshold is not an Action Level, it is shown as the highest Action Level to demonstrate escalation of effects towards the Significance Threshold.

To evaluate Action Levels 1 and 2, comparisons of the NF area to the FF1, FFA and FFB areas were completed. To evaluate Action Level 3 triggers, comparisons were made to the normal range, following Action Level 1 and 2 triggers. AEMP results indicate that the three former FF reference areas have now become exposed to low levels of Mine effluent and, therefore, can no longer be considered valid reference areas in a control-impact comparison. Evaluation of annual water quality results has shown that the magnitudes of increases in concentrations of Mine-affected variables (i.e., TDS, associated ions, and other Mine-related variables; e.g., molybdenum and strontium) in the FF areas have been small and concentrations remain well below AEMP benchmarks (Golder 2017c). These results indicate negligible potential for toxicity to aquatic life in the FF areas, despite the effects observed on water quality. Therefore, from the perspective of evaluating the potential for aquatic toxicity in the NF and MF areas, the FF areas were considered suitable as "minimally affected" sampling areas in the statistical comparisons for evaluating Action Level exceedances.

8.3 Results

8.3.1 Summary of Effects

8.3.1.1 Weight-of-Evidence Effect Ratings

Mine-related effects in the NF area were identified to varying degrees during annual monitoring from 2007 to 2016 (Table 8-5). Total density exhibited at least low effect ratings in most years, peaking at a high effect rating in 2010 and returning to a low level effect rating in the most recent years (2013 and 2016). Richness was rated low in the majority of years, although the direction of the effect alternated from a negative to a positive position in 2010, and returned to a negative position in 2011. In 2013 and 2016 no effect was reported on richness. No effect has been observed in dominance, SDI and BCI since the last AEMP re-evaluation report (i.e., 2011 to 2013). SEI prior to 2016 showed no effect rating, and in 2016 a low level negative effect was observed, when the NF area mean richness was statistically lower relative to the FF area mean, while remaining within the normal range.

Pisidiidae densities in 2008 to 2010, exhibited moderate to high effect ratings in a positive direction, and no effects were detected in 2011 and 2013. In 2016, lower mean density was observed in the NF area relative to the FF areas, but density remained within the normal range, resulting in a low-level effect rating in a negative direction. No change has been observed in *Procladius* effect ratings since 2007; the rating remained high in all years. *Heterotrissocladius* effect ratings indicated no effect in 2016, whereas previously, low to moderate effect ratings were observed since 2009. No effect was observed in *Micropsectra* density during the last re-evaluation period (2011 to 2013); in 2016, a low level positive effect rating was assigned, as determined by a statistically greater mean density in the NF area compared to the FF areas.

A high-level effect rating was applied to the relative abundance of dominant taxa in 2016, as the communities showed a relatively greater abundance of Chironomidae and lower relative densities of Pisidiidae in the NF area compared to the FF areas.

Table 8-5 Summary of Weight-of-Evidence Effect Ratings for the Benthic Invertebrate Community, 2007 to 2016

Variable	2007	2008	2009	2010	2011	2013	2016
Total Density	0	↑	↑↑	↑↑↑	↑↑	↑	↑
Richness	0	↓	↓	↑	↓	0	0
Dominance	0	0	↑	0	0	0	0
Simpson's Diversity Index	0	0	↓	0	0	0	0
Simpson’s Evenness Index	0	0	0	0	0	0	↓
Bray-Curtis Index	0	0	0	0	0	0	0
Pisidiidae Density	0	↑↑	↑↑	↑↑↑	0	0	↓
Procladius sp. Density	↑↑↑	↑↑↑	↑↑↑	↑↑↑	↑↑↑	↑↑↑	↑↑↑
Heterotrissocladius sp. Density	0	↓	↑↑	↑↑	↑↑	↑	0
Micropsectra sp. Density	n/a	0	↑	↑↑	0	0	↑
Relative Abundance of Major Taxa	0	0	0	0	↑(C)	↑(C)/ ↓(P)	↑↑↑ (C)/ ↓↓↓(P)

Notes: The direction of the arrow indicates the direction of change or relationship, where a positive direction indicates potential eutrophication and a negative direction indicates a potential toxic effect.

0 = no response; ↑/↓ = early warning low-level rating; ↑↑/↓↓ = moderate-level rating; ↑↑↑/↓↓↓ = high-level rating; n/a = not applicable; (C) = Chironomidae; (P) = Pisidiidae; arrows in last row indicate direction of >10% difference in NF area relative to range in reference condition dataset.

Based on gradient analysis results, low level effects occurred in total density and *Procladius* sp. density in the direction of nutrient enrichment during all years, with greater densities in the NF area and a decline with distance from the diffuser (i.e., negative slopes indicated by down arrows in Table 8-6). Similarly, low level effects were observed in other benthic invertebrate community groups, but the results were more variable and were not repeated each year. Richness showed no effect, except for 2009, when a low-level effect occurred indicating lower richness in the NF area compared to the FF areas.

Community indices were variable among years. Dominance, BCI and percent Chironomidae indicated the potential for low level effects in a negative direction (i.e., higher values in the NF area), whereas both SDI and SEI indicated low level effects in the opposite direction (Table 8-6). These responses did not occur consistently every year, nor did the various responses occur at the same time.

Low level effects occurred in densities of dominant invertebrates in the direction of nutrient enrichment in up to four years, with one moderate rating for *Heterotrissocladius* in 2011, indicating greater densities in the NF area and a decline with distance from the diffuser (Table 8-6).

The results for nearly all variables support the nutrient enrichment hypothesis. Invertebrate densities were higher in the NF area compared to the FF areas. Only a single response in the opposite direction was observed (i.e., richness in 2009, which was lower in the NF area compared to the FF areas). Community indices, which are not specific to the type of effect, showed responses consistent with nutrient enrichment: greater dominance and BCI values, and lower SDI and SEI, in the NF area compared to the FF areas.

Table 8-6 Summary of Weight-of-Evidence Effect Ratings for the Benthic Invertebrate Community (Based on Gradient Analysis)

Variable	2007	2008	2009	2010	2011	2013	2016
Total density	↓	↓	↓	↓	↓	↓	↓
Richness	0	0	↑	0	0	0	0
Dominance	↓	0	↓	0	↓	0	↓
Simpson's Diversity Index	↑	0	↑	0	↑	0	0
Simpson's Evenness Index	↑	0	0	↑	0	↑	↑
Bray-Curtis Index	0	0	↓	0	0	0	↓
Pisidiidae density	0	↓	↓	↓	0	0	0[(a)]
Procladius sp. density	↓	↓	↓	↓	↓	↓	↓
Heterotrissocladius sp. density	0	0	↓	0	↓↓	0	0
Micropsectra sp. density	↓	↓	0	0	↓	0	↓

Note: 0 = No effect; ↑ = Low effect ranking, positive slope; ↑↑ = Moderate effect rating with a greater positive slope compared to slopes from reference condition dataset (2007 to 2010); ↓ = Low effect ranking, negative slope; ↓↓ = Moderate effect rating with a greater negative slope compared to slopes from reference condition dataset.

a) Pisidiidae densities differed compared to 2007 to 2010, but slope of regression line did not differ significantly from zero.

8.3.1.2 Action Levels

Action Level triggers for toxicological impairment have varied over time and among variables since 2007 (Table 8-7), generally indicating no consistent response, and more likely indicating background year-to-year variation in the benthic invertebrate community, combined with high sensitivity of Action Level criteria. Action Level 1 was triggered for three variables in 2008 (i.e., richness, percent Chironomidae and *Heterotrissocladius* density), and two variables in 2009 (i.e., richness and SDI). In 2011, richness also met the criterion for an Action Level 1. These responses were not repeated in subsequent years. In 2016, SEI triggered Action Level 2 and Pisidiidae density triggered Action Level 1. No Action Levels were triggered in 2007, 2010 and 2013.

Table 8-7 Summary of Action Level Rankings for the Benthic Invertebrate Community, 2007 to 2016

Variable	2007	2008	2009	2010	2011	2013	2016
Total Density	-	-	-	-	-	-	-
Richness	-	AL1	AL1	-	AL1	-	-
Dominance	-	-	-	-	-	-	-
Simpson's Diversity Index	-	-	AL1	-	-	-	-
Simpson's Evenness Index	-	-	-	-	-	-	AL2
Bray-Curtis Index	-(a)	-(a)	-(a)	-(a)	-	-	-
Percent Chironomidae	-	AL1	-	-	-	-	-
Pisidiidae Density	-	-	-	-	-	-	AL1
Procladius sp. Density	-	-	-	-	-	-	-
Heterotrissocladius sp. Density	-	AL1	-	-	-	-	-
Micropsectra sp. Density	-	-	-	-	-	-	-

- = did not trigger an Action Level; AL1 = Action Level 1 triggered; AL2 = Action Level 2 triggered.

a) BCI was calculated based on reference median method (inconsistent with the current method or pairwise comparisons) and was therefore not used in the Action Level evaluation.

8.3.2 Assessment of Trends

Linear models were selected for trend analyses of all benthic invertebrate community variables (Table 8-8). Results reported here are based on transformation of data using the Box-Cox transformation, as outlined in Section 2.4.2. Although parabolic trends were a better fit for specific areas and variables (e.g., SEI in the NF area), the selected trend model was the overall dominant model that accounted best for all areas.

Results of significance testing of the Area × Year interaction term for each variable, which evaluates whether trends differ among areas/stations included in the analysis, are provided in Table 8-9. Results indicate that only richness and percent chironomidae had significantly different trends among areas/stations included in the analysis. Significance of the slopes of linear trends from zero are presented for each variable and area/station in Table 8-10.

Table 8-8 Benthic Invertebrate Community AIC Model Selection, 1996 to 2016

Variable	Yeo-Johnson transformation (a)	Parabolic AIC Score	Linear AIC Score
Total Density	0	384	368
Richness	0.5	553	542
Dominance	Logit	299	287
Simpson's Diversity Index	Logit	260	247
Simpson's Evenness Index	Logit	322	315
Bray-Curtis Index(b)	Logit	192	182
Percent Chironomidae	Logit	429	410
Pisidiidae Density	0.5	1,359	1,355
Procladius Density	0.5	1,317	1,300
Heterotrissocladius Density	0	715	702
Micropsectra Density	0	748	744

a) See Section 2.4.2.1 for details on transformation approach.
b) Bray-Curtis Index is presented using data from 2007 to 2016.
Note: Shaded cells represent the chosen model for each variable. AIC = Akaike's information criterion.

Table 8-9 Significance of Benthic Invertebrate Community Fixed Effects Models,1996 to 2016

Variable	Coefficient	Numerator DF	Denominator DF	F-value	*P*-value
Total Density	Area	7	17	1.45	0.250
	Year	1	152	3.3	0.071
	Area × Year	7	152	1.45	0.189
Richness	Area	7	17	2.46	0.061
	Year	1	152	1.89	0.172
	Area × Year	7	152	2.47	**0.020**
Dominance	Area	7	17	1.98	0.118
	Year	1	150	0.06	0.808
	Area × Year	7	150	1.99	0.060
Simpson's Diversity Index	Area	7	17	1.84	0.145
	Year	1	149	0.07	0.793
	Area × Year	7	149	1.84	0.084
Simpson’s Evenness Index	Area	7	17	1.75	0.163
	Year	1	150	0.34	0.562
	Area × Year	7	150	1.75	0.102
Bray-Curtis Index	Area	7	17	1.25	0.329
	Year	1	131	1.04	0.311
	Area × Year	7	131	1.25	0.279
Percent Chironomidae	Intercept	1	152	9.89	0.002
	Area	7	17	2.19	0.088
	Year	1	152	9.73	0.002
	Area × Year	7	152	2.2	**0.037**
Pisidiidae Density	Area	7	17	2	0.115
	Year	1	151	0.12	0.733
	Area × Year	7	151	2	0.058
Procladius Density	Area	7	17	1.54	0.219
	Year	1	150	0.02	0.883
	Area × Year	7	150	1.53	0.162
Heterotrissocladius Density	Area	7	17	1.79	0.156
	Year	1	150	7.79	0.006
	Area × Year	7	150	1.79	0.094
Micropsectra Density	Area	7	17	1.45	0.252
	Year	1	152	15.62	<0.001
	Area × Year	7	152	1.45	0.190

Note: **Bolded** values represent results considered significant at $P<0.05$.
DF = degrees of freedom; F-value = F statistic; *P*-value = probability.

Table 8-10 Estimated Significance of Differences of Linear Slopes from Zero

Variable	Area / Station							
	NF	**MF1-3**	**MF2-2/MF2-3**	**FF2-2**	**MF3-4**	**FF1**	**FFB**	**FFA**
Total Density	**0.039** ↓	0.378	**<0.001** ↓	0.786	0.976	**0.037** ↓	0.466	0.092
Richness	0.055	0.255	**<0.001** ↓	0.942	0.778	0.679	0.945	**0.010** ↓
Dominance	0.643	0.265	0.086	**0.006** ↓	0.373	0.305	0.941	0.871
Simpson's Diversity Index	0.717	0.421	0.058	**0.021** ↑	0.241	0.205	0.533	0.178
Simpson's Evenness Index	0.392	0.074	**0.015** ↑	0.074	0.311	0.273	0.837	0.215
Bray-Curtis Index	0.593	**0.026** ↑	0.142	0.456	0.865	0.573	0.576	0.993
Percent Chironomidae	0.294	0.689	0.491	**0.025** ↓	0.307	**0.023** ↓	**0.001** ↓	0.172
Pisidiidae Density	0.235	0.704	**0.012** ↓	0.150	0.539	0.637	0.052	0.971
Procladius Density	0.422	0.175	0.889	0.182	0.254	0.081	0.276	0.227
Heterotrissocladius Density	0.144	0.615	**<0.001** ↓	0.465	0.550	**0.007** ↓	**0.001** ↓	0.451
Micropsectra Density	0.212	0.531	**0.018** ↓	0.478	0.089	**<0.001** ↓	0.073	0.109

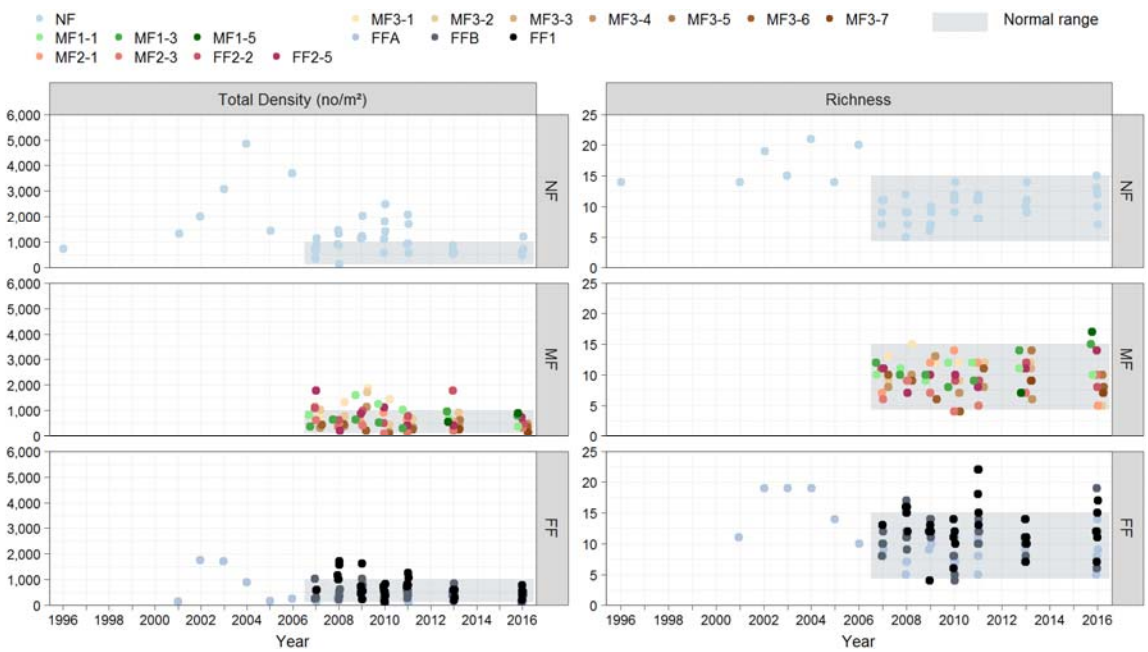
Note: **Bolded** values represent results considered significant at $P<0.05$; ↑ = increasing trend over time; ↓= decreasing trend over time.
NF = near-field; MF = mid-field; FF = far-field.

8.3.2.1 Total Invertebrate Density and Richness

Total invertebrate density increased from 2001 to a maximum in 2003, followed by fluctuating densities between 2004 and 2007 (Figure 8-2). Densities were less variable following 2006, potentially reflecting more consistent sampling depth among stations. Between 2007 and 2010, total density was within or above the normal range, and in 2013 and 2016, total densities at the majority of stations were within the normal range. In general, less variation was observed in 2013 and 2016 compared to previous years. Trend analysis results indicated declines in benthic invertebrate density over time in all areas of the lake (Figure 8-3), except for stations MF1-3 and MF3-4. The declining trends in Figure 8-3 had significantly different slopes from zero in the NF and FF1 areas and the MF2-2/MF2-3 stations (Table 8-10).

Richness values before 2007 were at the upper limit or above the normal range (Figure 8-2). Since 2007, richness was within or slightly above the normal range in all areas, with a few exceptions in 2016 (MF2 and FF areas). The difference in richness between 1996 to 2006 and 2007 to 2016 is most likely a reflection of the difference in taxonomists between these periods. Trend analysis results show a decline in richness in the NF, MF2 and FF areas; however significant slopes were only detected at the MF2-2/MF2-3 stations and the FFA area (Table 8-10). A significant Area × Year interaction was detected for richness, indicating that temporal trends differed among sampling areas/stations (Table 8-9).

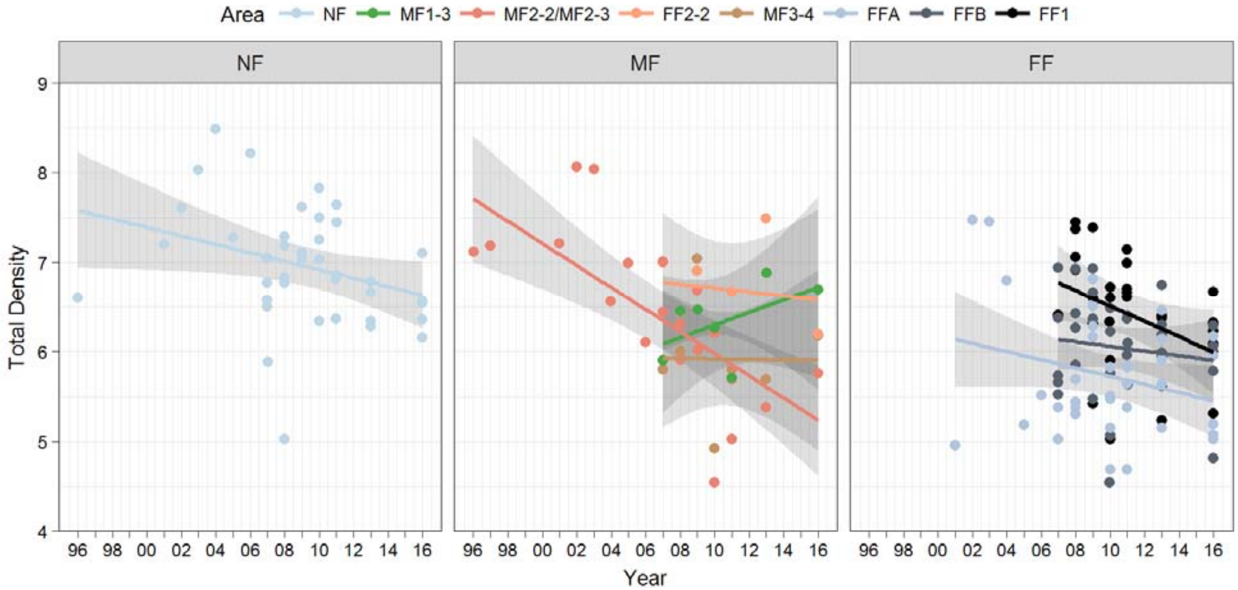
Figure 8-2 Benthic Invertebrate Total Density and Richness, 1996 to 2016.



Note: Shaded area represents the normal range (Golder 2017b).

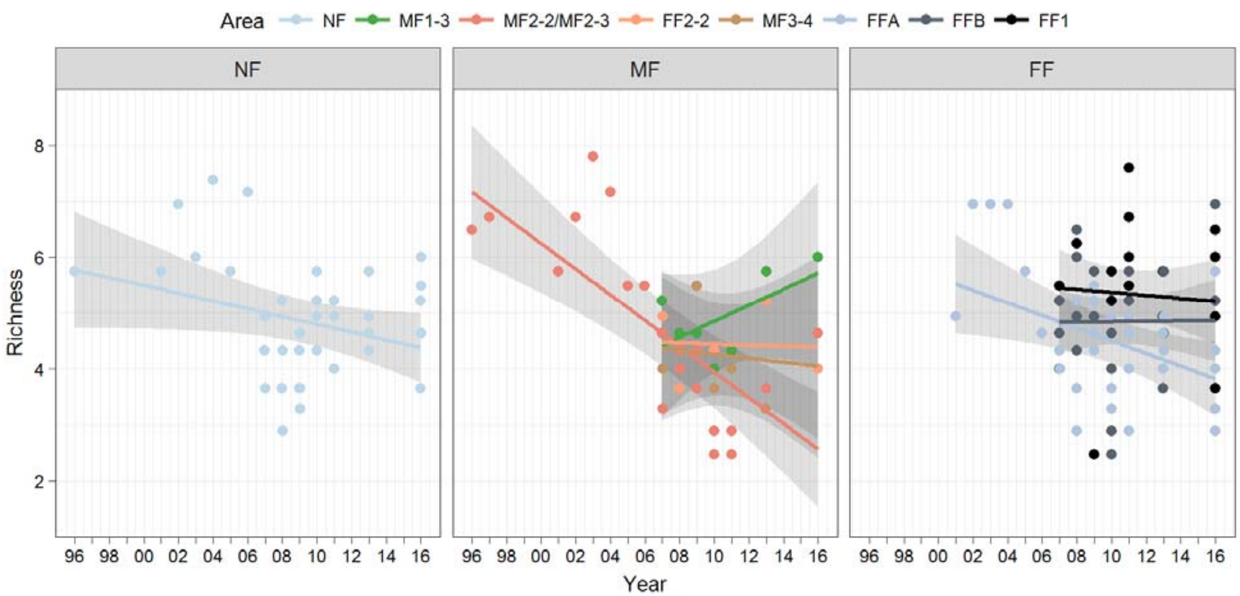
NF = near-field; MF = mid-field; FF = far-field; no/m^2 = number per metre squared.

Figure 8-3 Trend Analysis of Total Benthic Invertebrate Density, 1996 to 2016.



Note: Shaded area represents the 95% confidence interval.
NF = near-field; MF = mid-field; FF = far-field.

Figure 8-4 Trend Analysis of Benthic Invertebrate Richness, 1996 to 2016.



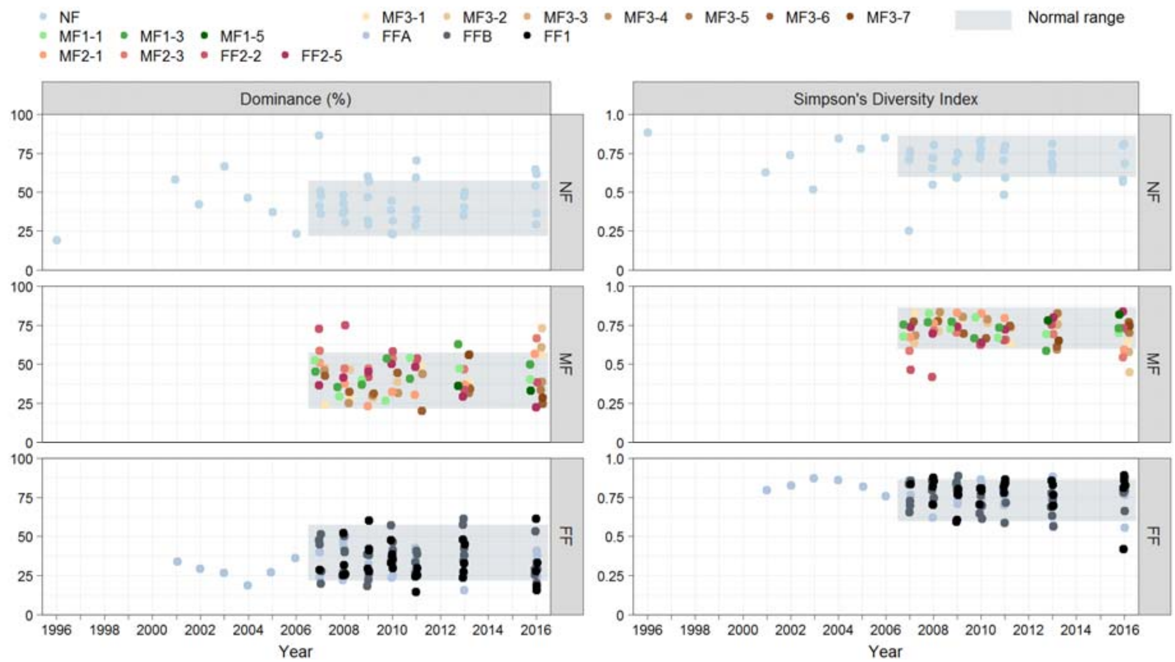
Note: Shaded area represents the 95% confidence interval.
NF = near-field; MF = mid-field; FF = far-field.

8.3.2.2 Benthic Community Indices

Dominance (i.e., relative density of the dominant invertebrates) fluctuated between 2001 and 2006, after which it was mostly within the normal range in all sampling areas, with excursions beyond the upper limit more commonly seen in the NF and MF areas compared to the FF areas (Figure 8-5). Dominance showed slight, non-significant increasing trends in NF area and most MF areas, and no trends in the FF areas (Figure 8-6, Table 8-10). Significantly declining dominance was found at the FF2-2 station.

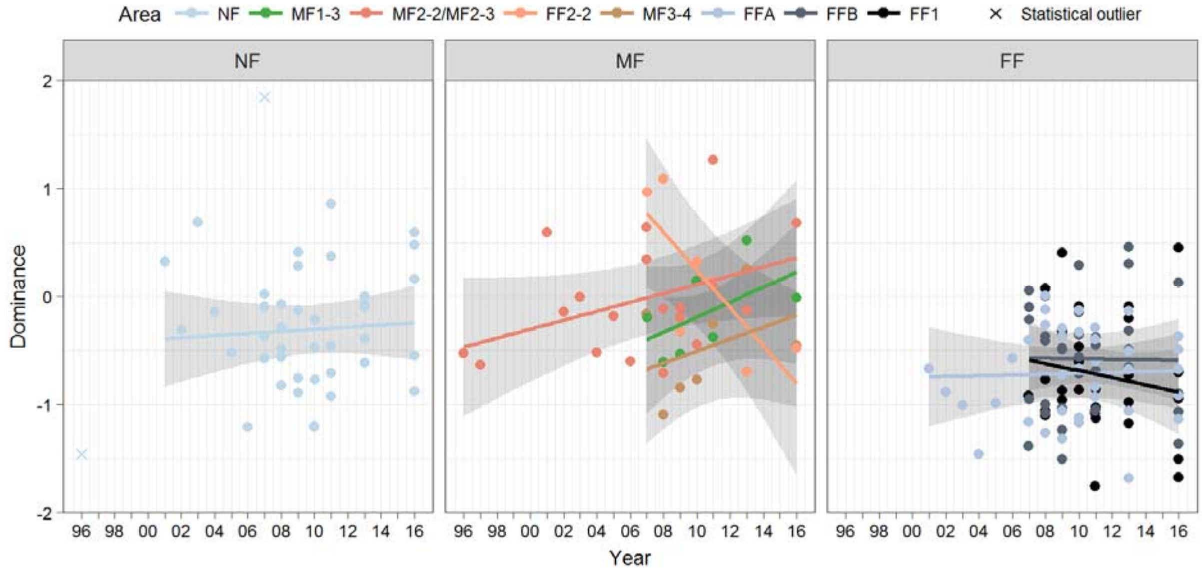
The SDI was within or occasionally below the normal range in all areas of Lac de Gras since 1996 (Figure 8-5). Trend analysis results indicate mostly stable communities in the NF, FFA and FFB areas, and slight increases over time in the MF areas, with one significant increasing trend at the FF2-2 station (Figure 8-7, Table 8-10).

Figure 8-5 Dominance and Simpson's Diversity Index, 1996 to 2016.



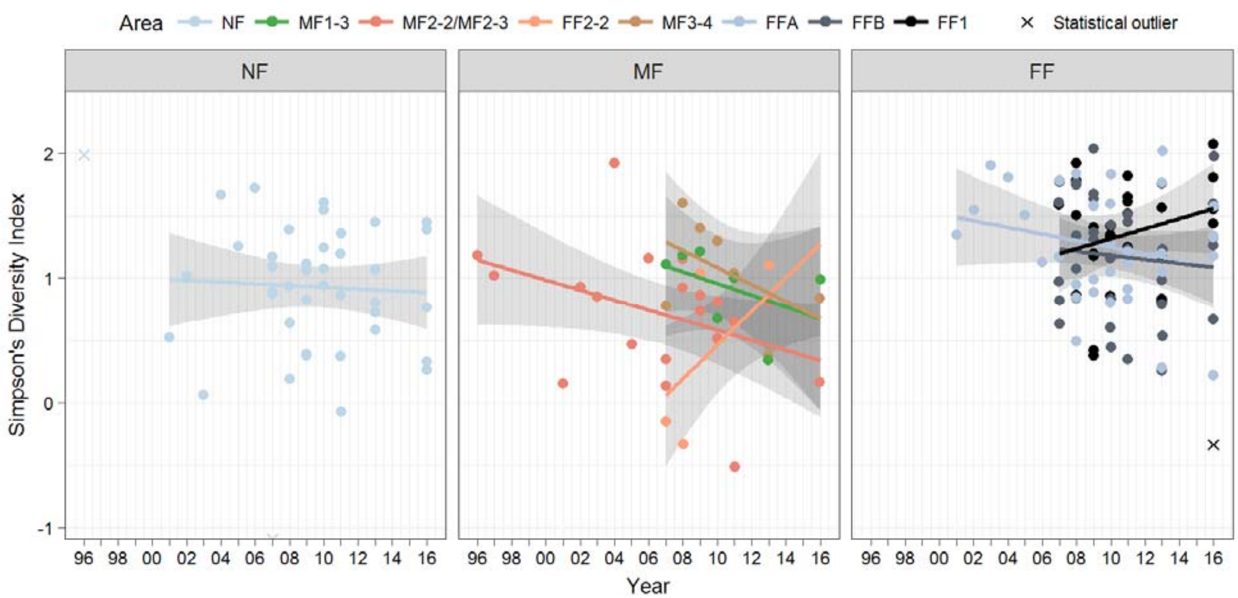
Note: Shaded area represents the normal range (Golder 2017b).
NF = near-field; MF = mid-field; FF = far-field.

Figure 8-6 Trend Analysis of Dominance, 1996 to 2016.



Note: Shaded area represents the 95% confidence interval.
NF = near-field; MF = mid-field; FF = far-field.

Figure 8-7 Trend Analysis of Simpson's Diversity Index, 1996 to 2016.

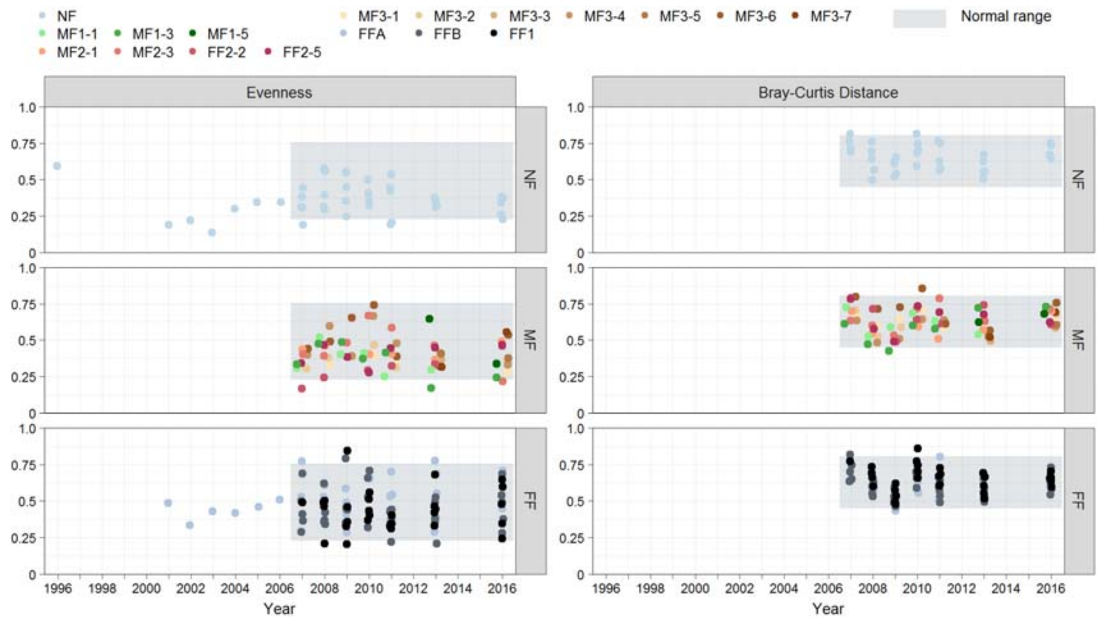


Note: Shaded area represents the 95% confidence interval.
NF = near-field; MF = mid-field; FF = far-field.

The SEI increased in the NF area from 2001 and peaked in 2007 (Figure 8-8). Since 2007, it has declined in the NF area and moved towards the lower limit of the normal range by 2016. The SEI has also remained mostly within the normal range in other sampling areas, with occasional excursions below the lower limit in multiple years in all areas. Trend analysis shows no obvious trends in SEI in the NF and FF areas, other than potential slight increases over time, and variable trends in the MF areas, with a significant increasing trend at the MF2-2/MF2-3 stations (Figure 8-9, Table 8-10). The trend in the NF area was closer to parabolic, with a recent decline in SEI, which was not captured by the trend analysis.

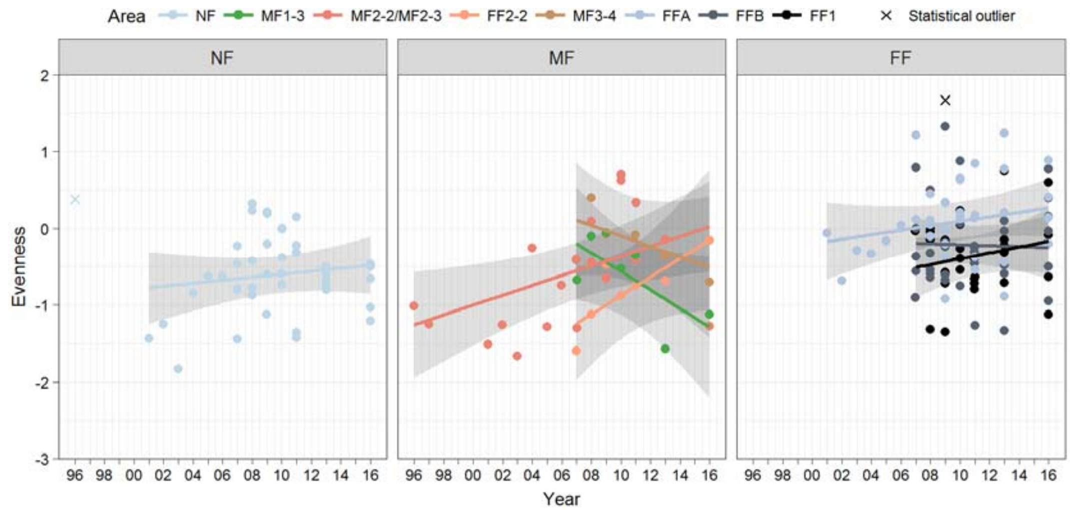
BCI has been calculated using consistent methods since 2007, and has generally remained within the normal range in all sampling areas, with only minor excursions beyond the upper limit of the normal range (Figure 8-8). Trend analysis revealed very slight, non-significant declines in the NF and FF areas, and variable trends in the MF areas (Figure 8-10). A significant slope was detected at the MF1-3 station in an increasing direction (Table 8-10).

Figure 8-8 Simpson's Evenness Index and Bray-Curtis Index, 1996 to 2016.



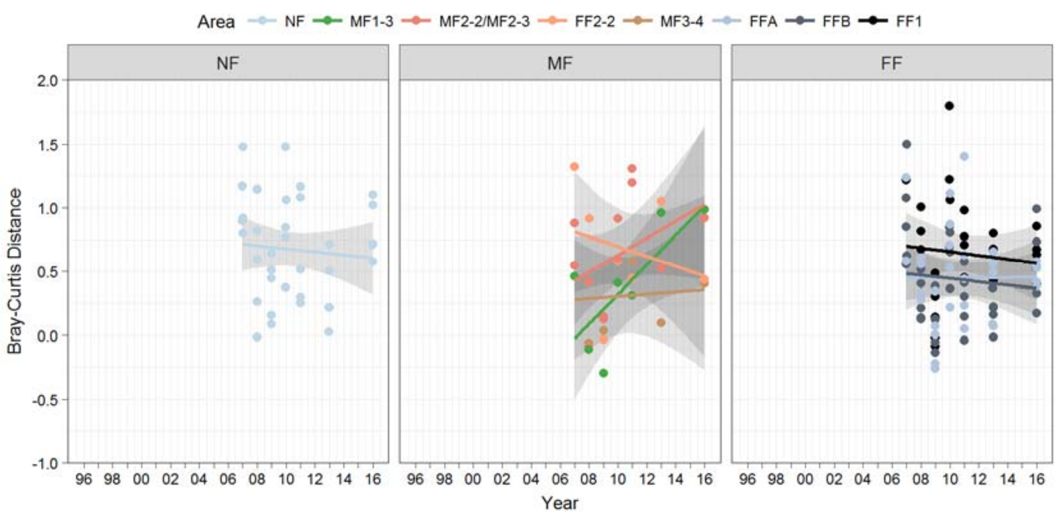
Note: Shaded area represents the normal range (Golder 2017b). Bray-Curtis index was plotted from 2007 to 2016.
NF = near-field; MF = mid-field; FF = far-field; no/m^2 = number per metre squared.

Figure 8-9 Trend Analysis of Simpson's Evenness Index, 1996 to 2016.



Note: Shaded area represents the 95% confidence interval.
NF = near-field; MF = mid-field; FF = far-field.

Figure 8-10 Trend Analysis of Bray-Curtis Index, 2007 to 2016.



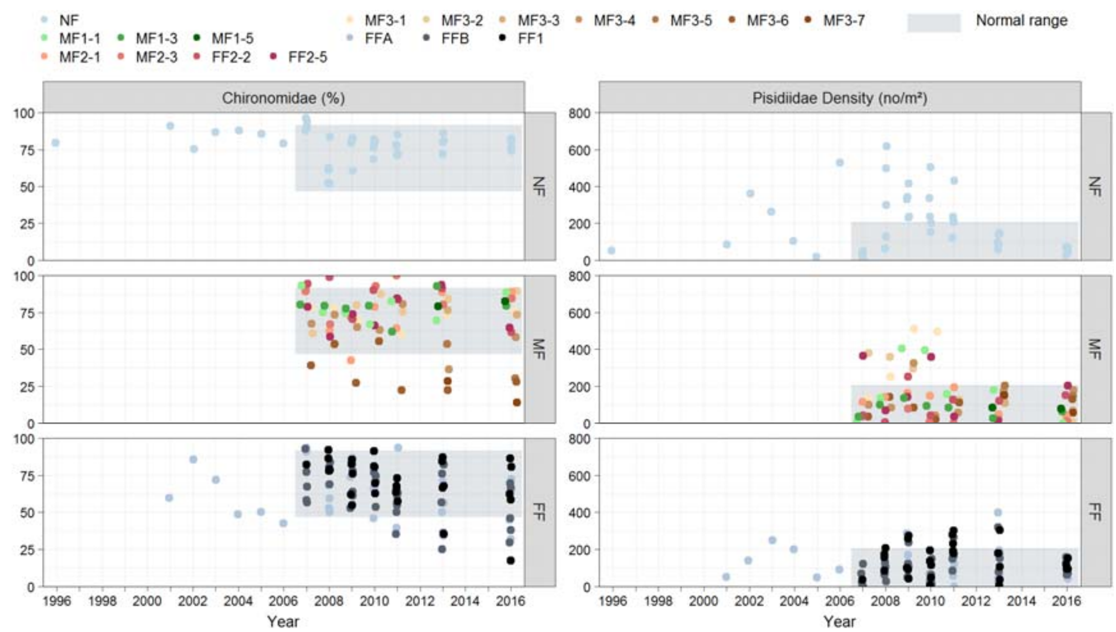
Note: Shaded area represents the 95% confidence interval.
NF = near-field; MF = mid-field; FF = far-field.

8.3.2.3 Densities of Common Invertebrates

The relative density of Chironomidae has undergone substantial variation over time in Lac de Gras (Figure 8-11). In the NF area, percent Chironomidae was above the normal range in 2007, but has consistently remained within the normal range since 2007, and has become less variable among stations. The MF areas show a greater degree of variation, depending on transect, and had frequent values above and below the normal range, with values below the normal range mostly in the MF3 area. A similar pattern was observed in the FF areas, where the majority of stations fell within or below the normal range in 2016. Trend analysis detected no significant trend in the NF area, but declining trends were apparent at the FF2-2 station, and the FF1 and FFB areas (Figure 8-12, Table 8-10).

Pisidiidae density has shown a moderate amount of variation over time, whereby densities were observed approaching zero in some years (2005 and 2007), but were also above the normal range in other years (2008 and 2011) (Figure 8-11). Recently, Pisidiidae density has been within the normal range in the NF and MF areas, with values near the lower limit in the NF area in 2016. The FF area values have shown similar trends, and were within the normal range in 2016. Pisidiidae density shows a recent, non-significant decline in the NF area, varying trends in the MF areas, and no trends in the FF areas (Figure 8-13). A significant declining trend was detected at the MF2-2/MF2-3 stations (Table 8-10).

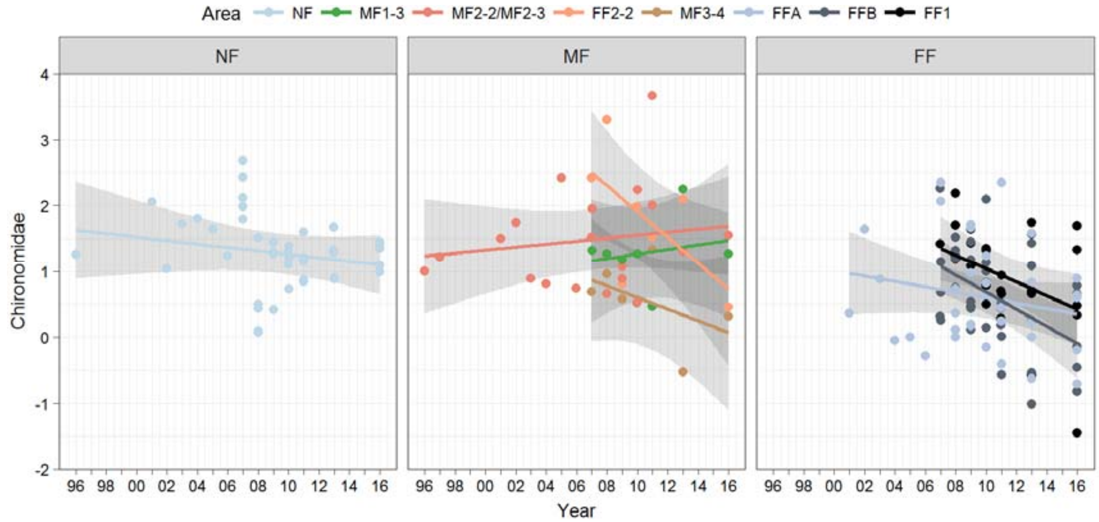
Figure 8-11 Relative Chironomidae Density and Pisidiidae Density, 1996 to 2016.



Note: Shaded area represents the normal range (Golder 2017b).

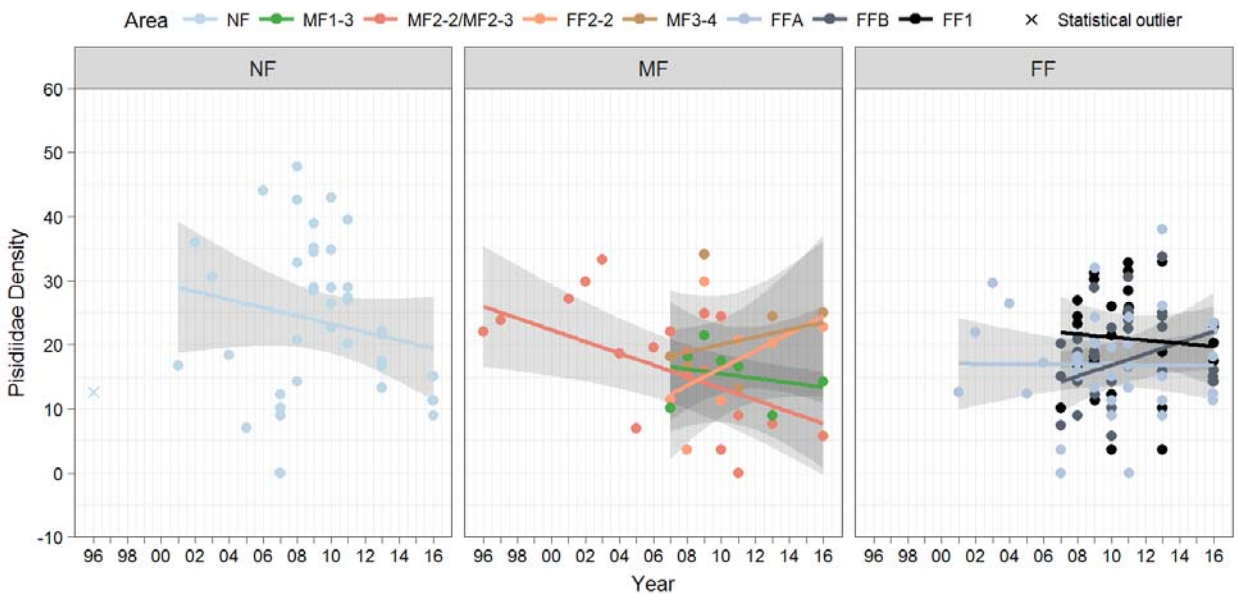
NF = near-field; MF = mid-field; FF = far-field; no/m^2 = number per metre squared.

Figure 8-12 Trend Analysis of Relative Chironomidae Density, 1996 to 2016.



Note: Shaded area represents the 95% confidence interval.
NF = near-field; MF = mid-field; FF = far-field.

Figure 8-13 Trend Analysis of Pisidiidae Density, 1996 to 2016.

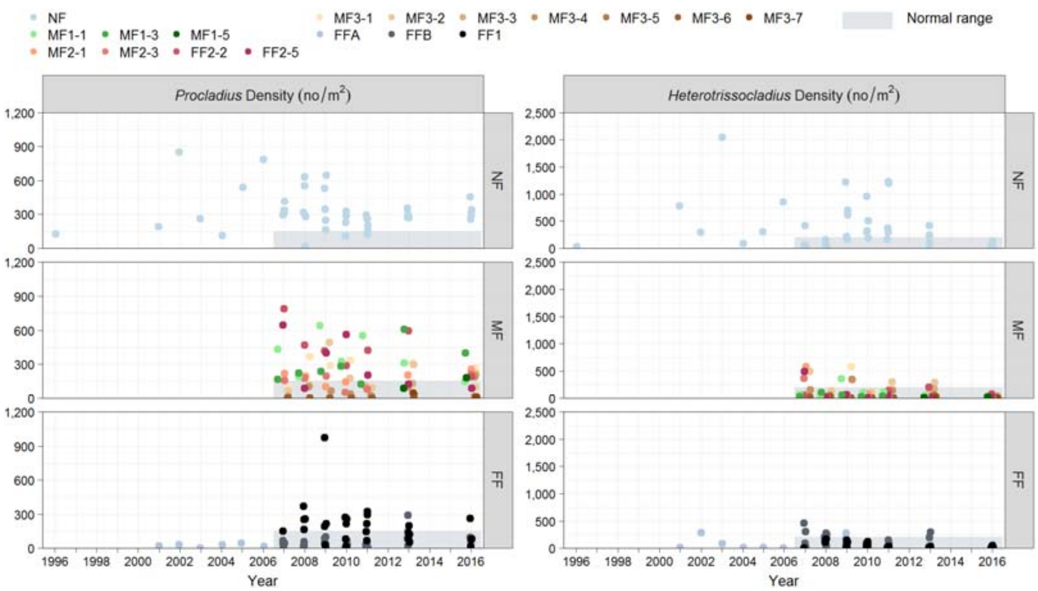


Note: Shaded area represents the 95% confidence interval.
NF = near-field; MF = mid-field; FF = far-field.

Procladius density has shown large variation in all sampling areas throughout the monitoring period (Figure 8-14). It has generally remained above the normal range in the NF area, and within or above the normal range in the MF and FF areas. No long-term trends were detected in *Procladius* density (Figure 8-15, Table 8-10).

The density of *Heterotrissocladius* in the NF area has been highly variable prior to 2013 (Figure 8-14). Density peaked between 2008 and 2011 and began declining in 2013. In 2016, *Heterotrissocladius* density was within the normal range in the NF area. Less variation over time was observed in the MF and FF areas. Trend analysis shows generally declining trends in all areas, with significant slopes at the MF2-2/MF2-3 stations, and in the FF1 and FFB areas (Figure 8-16, Table 8-10).

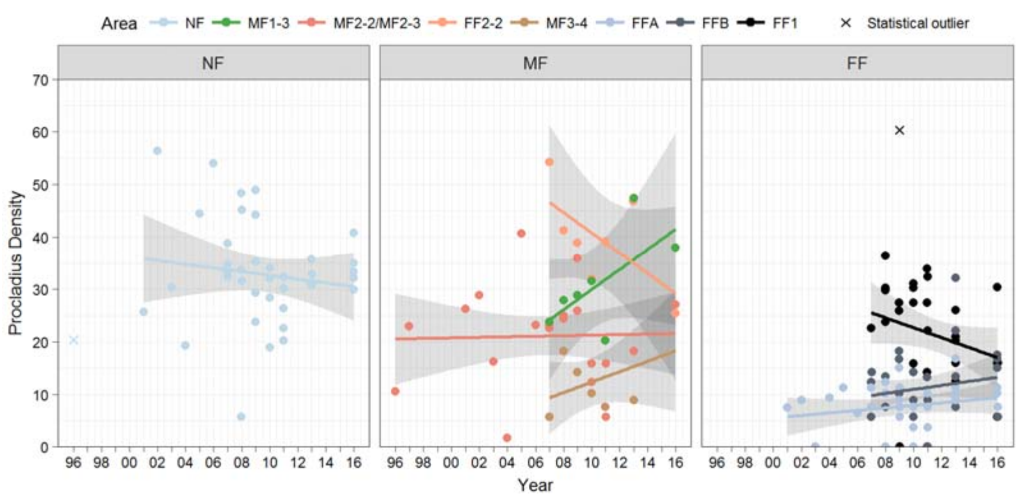
Figure 8-14 *Procladius* and *Heterotrissocladius* Density, 1996 to 2016.



Note: Shaded area represents the normal range (Golder 2017b).

NF = near-field; MF = mid-field; FF = far-field; no/m^2 = number per metre squared.

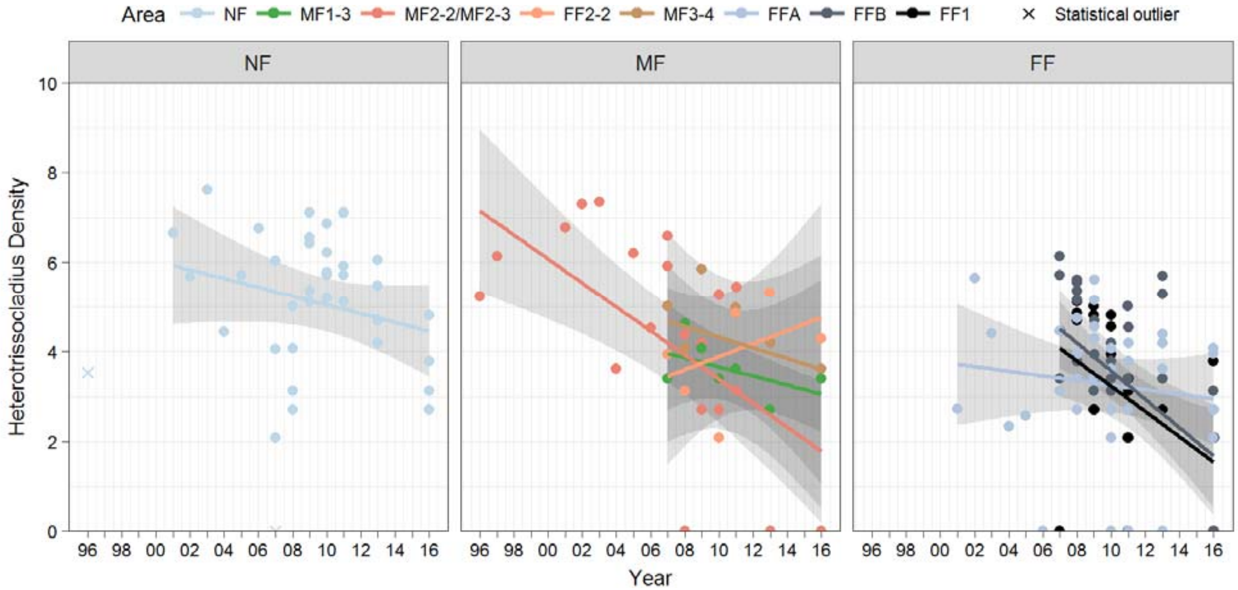
Figure 8-15 Trend Analysis of *Procladius* Density, 1996 to 2016.



Note: Shaded area represents the 95% confidence interval.

NF = near-field; MF = mid-field; FF = far-field.

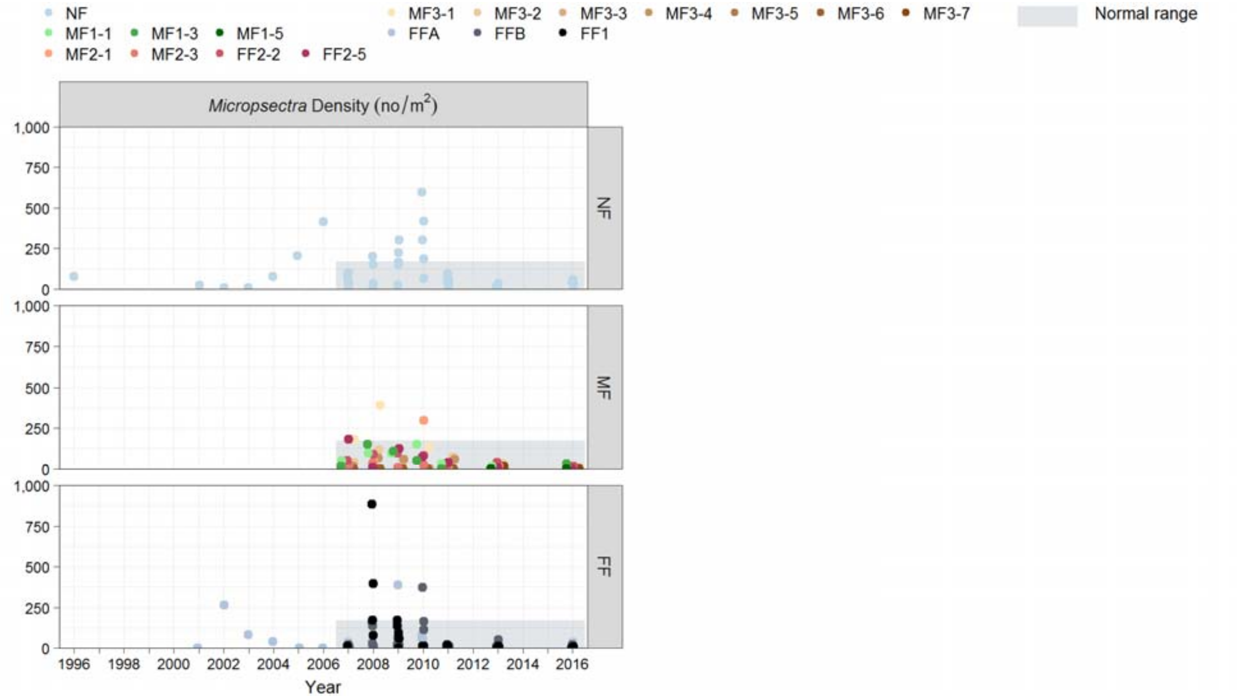
Figure 8-16 Trend Analysis of *Heterotrissocladius* Density, 1996 to 2016.



Note: Shaded area represents the 95% confidence interval.
NF = near-field; MF = mid-field; FF = far-field.

Micropsectra density was highly variable in all sampling areas until 2010 (Figure 8-17). After 2010, densities declined to within the normal range in all sampling areas. In 2013 and 2016, *Micropsectra* density was near the lower limit of the normal range in all areas of Lac de Gras. All sampling areas show similar negative trends in density over time (Figure 8-18). Significant declines were detected at the MF2-2/MF2-3 stations and the FF1 area (Table 8-10).

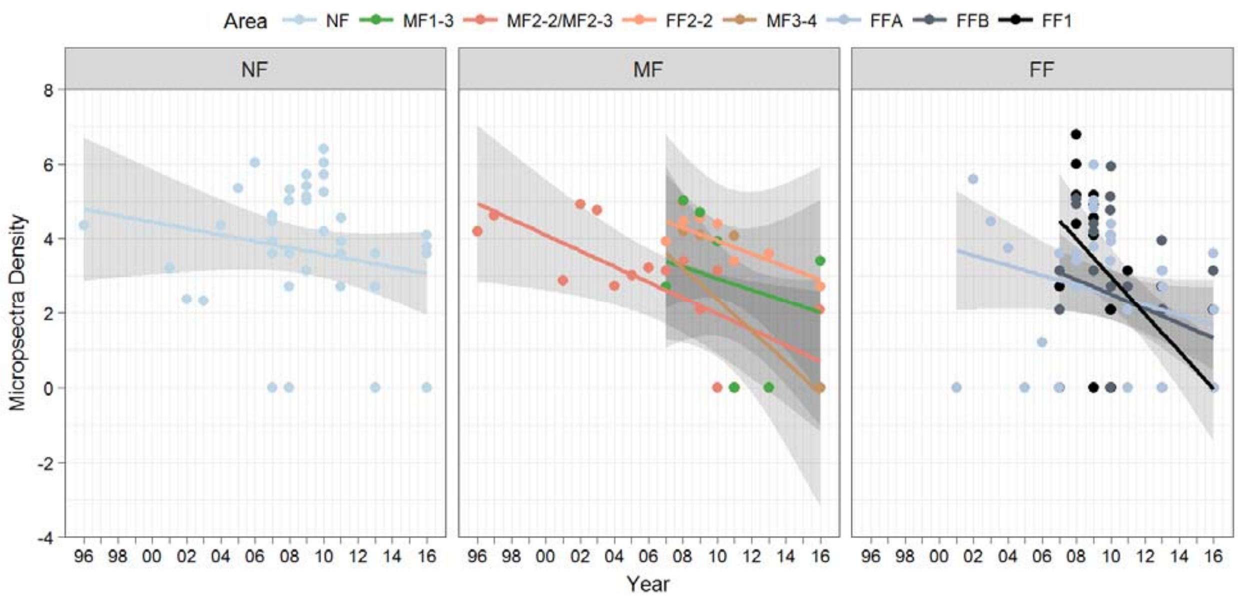
Figure 8-17 *Micropsectra* Density, 1996 to 2016.



Note: Shaded area represents the normal range (Golder 2017b).

NF = near-field; MF = mid-field; FF = far-field; no/m^2 = number per metre squared.

Figure 8-18 Trend Analysis of *Micropsectra* Density, 1996 to 2016.



Note: Shaded area represents the 95% confidence interval.

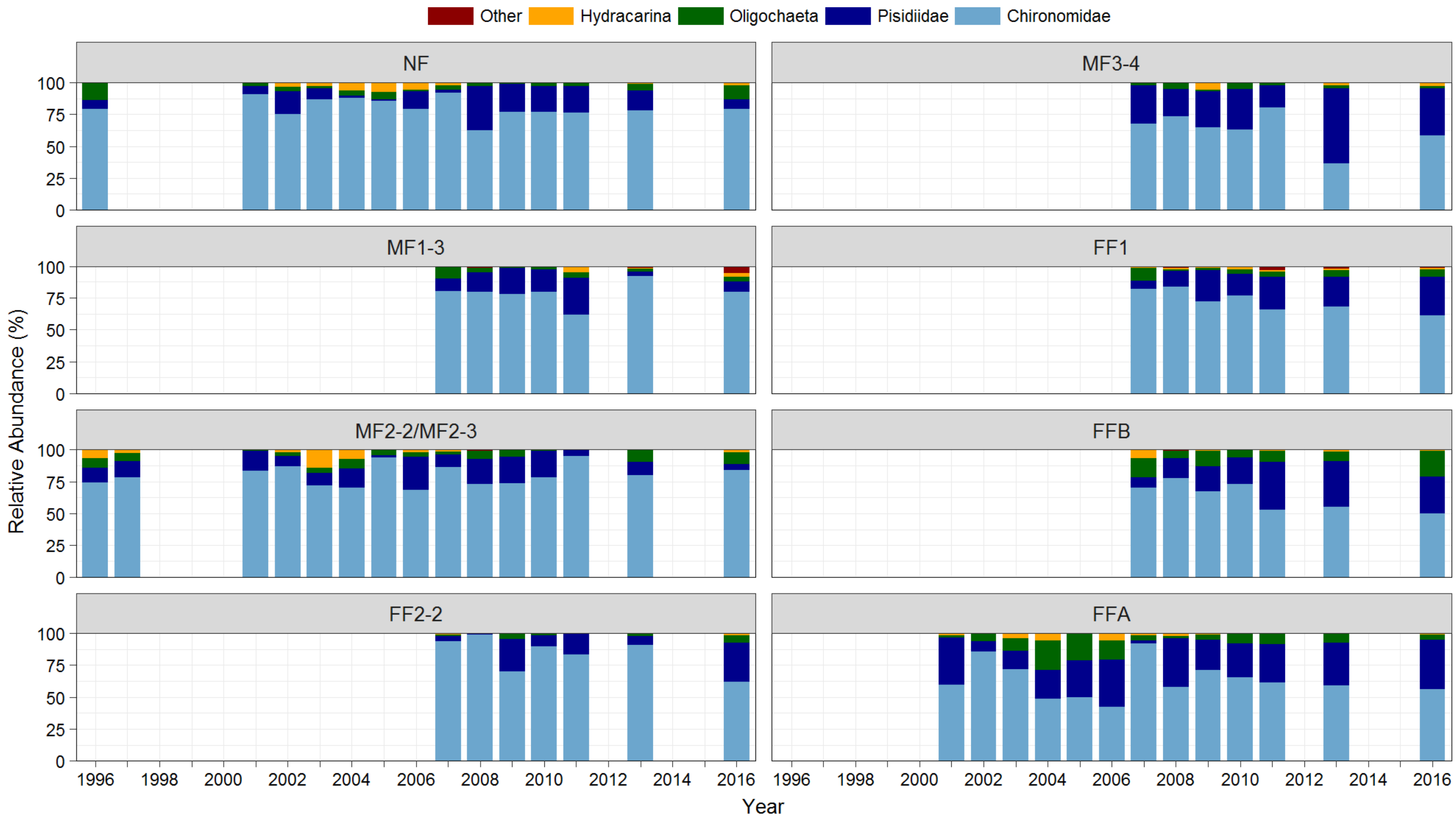
NF = near-field; MF = mid-field; FF = far-field.

8.3.2.4 Community Composition by Major Group

Chironomidae midges have dominated the Lac de Gras benthic invertebrate communities in all sampling areas during baseline years and each year of monitoring under the AEMP (Figure 8-19). In most years, the combined densities of Chironomidae and Pisidiidae accounted for over 90% of the total benthic invertebrate density in all sampling areas. Less abundant but commonly occurring groups include aquatic worms (Oligochaeta) and mites (Hydracarina). The most common Chironomidae genera in Lac de Gras include *Procladius, Heterotrissocladius* and *Micropsectra*. Numerous other Chironomidae genera are also present in the lake, although at low and variable densities.

The benthic community of the NF area was strongly chironomid-dominated in all years of monitoring, with low and varying relative densities of non-chironomid groups (Figure 8-19). Pisidiidae were the second most abundant group in most years, but rarely accounted for more than 20% of total density. At this coarse level of examination, the NF community in 2016 closely resembled the baseline community sampled in 1996. MF and FF area communities were similar to the NF area community, but included a greater relative abundance of Pisidiidae at the MF3-4 station, and in the FFB and FFA areas. A divergence in community composition is apparent between the NF and FF areas in recent years. The relative density of Pisidiidae has been low (<10%) in the NF area and at most MF stations, but remained greater (>20%) in the FF areas. Composition of the MF area communities was variable in 2016, with stations located closer to the NF area (MF1-3, MF2-2/MF2-3) having communities resembling the NF area community, and others having communities more similar to those in the FF areas (FF2-2, MF3-4).

Figure 8-19 Relative Densities of Major Invertebrate Groups in the NF and FF Areas, and Selected MF Stations, 1996 to 2016.



NF = near-field; MF = mid-field; FF = far-field.

8.3.3 Multivariate Analysis

8.3.3.1 Community Analysis

The two-dimensional configuration produced by the nMDS for the benthic invertebrate community dataset along the various transects in all years, generally showed low stress values (i.e., <0.2), according to the stress categories provided by Clarke (1993), except for the NF-MF2-FF2 transect (0.22). The ANOSIM results showed significant differences between the NF and FF communities in 2007, 2009 and 2016 (Table 8-11; overall ANOSIM R = 0.499; P = 0.0001).

The nMDS ordination results indicated a separation of the NF area from one of more of the FF areas during some years of monitoring. The 2007 to 2016 NF area communities were located near the top of the ordination and were all within the 60% similarly group compared to at least one FF area, except in 2007, when only one FF area (FFB) was sampled (Figure 8-20). The FF1 area communities were within the 60% similarity region with the NF area during each year of monitoring, while both the FFA and FFB areas fell within this region in 2009 and 2013. These results indicate that the NF and FF1 area communities are more similar to one-another, compared to the similarity between the NF area and the FFA and FFB areas. This result is as expected based on expected habitat similarity, and both the NF and FF1 areas are located in “narrow” areas of Lac de Gras that are closer to shore than the mid-lake FFA and FFB areas.

The majority of stations along the NF-MF1-FF1 transect fell within one similarity group, including the 2016 NF and MF1-3 stations (in close proximity to one another; Figure 8-21). The majority of FF1 stations were grouped separately from the NF and MF1 stations in a separate cluster. The NF-MF1-FF1 transect nMDS results also indicated a high degree of similarity between the MF1-1 (2016) and FF1 (2007) in a separate group from the other sampling areas/yrs. Overall, differences along the NF-MF1-FF1 transect mostly indicated a difference between communities of the FF1 area and other stations along the transect, and showed more pronounced differences among stations and areas in 2016 compared to previous years.

The NF-MF2-FF2 transect ordination plot showed numerous overlapping groups, although most of the 2016 stations were grouped together at the centre of the plot (Figure 8-22). The 2016 NF and FF2-2 station were closest to one another within the group, and at similar positions to the 2013 NF and MF2-1 stations. Earlier AEMP years (i.e., 2007 to 2010) generally grouped together at 60% similarity, with few exceptions and with minor overlap with the centre group containing most of the 2016 stations, demonstrating a separation of the 2016 communities from previous years. The NF-MF2-FF2 transect ordination had a relatively high stress value of 0.22, potentially indicating poor fit to the input data, and was rerun in three-dimensions, after which the stress value was marginally reduced, however, results were similar to the two-dimensional configuration in terms of grouping of areas and stations.

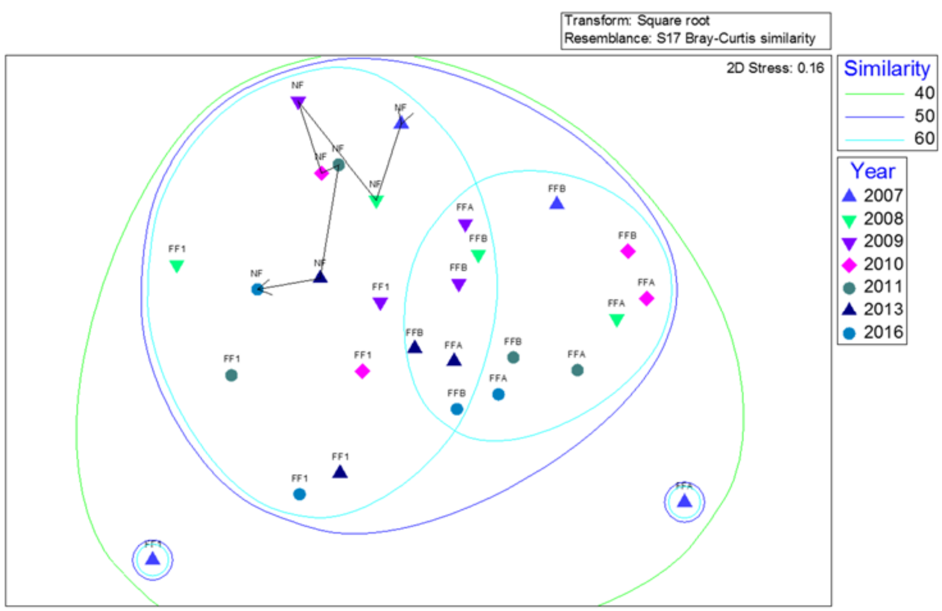
On the ordination plot of the NF-MF3-FFB-FFA transect, most stations/yrs fell within the same group at 40% similarity, with the exception of the MF3-6 and MF3-7 stations in nearly all monitoring years, which formed a separate cluster (Figure 8-23). These results highlight a potentially habitat-related difference in benthic community composition at these stations relative to all other stations.

Table 8-11 Analysis of Similarities (ANOSIM) Test between the Benthic Invertebrate Communities in the NF and FF areas, 2007 to 2016

Year	R statistic	*P-value*
2007	0.37	**0.006**
2008	0.12	0.151
2009	0.37	**0.004**
2010	0.06	0.300
2011	0.19	0.070
2013	0.03	0.400
2016	0.33	**0.009**

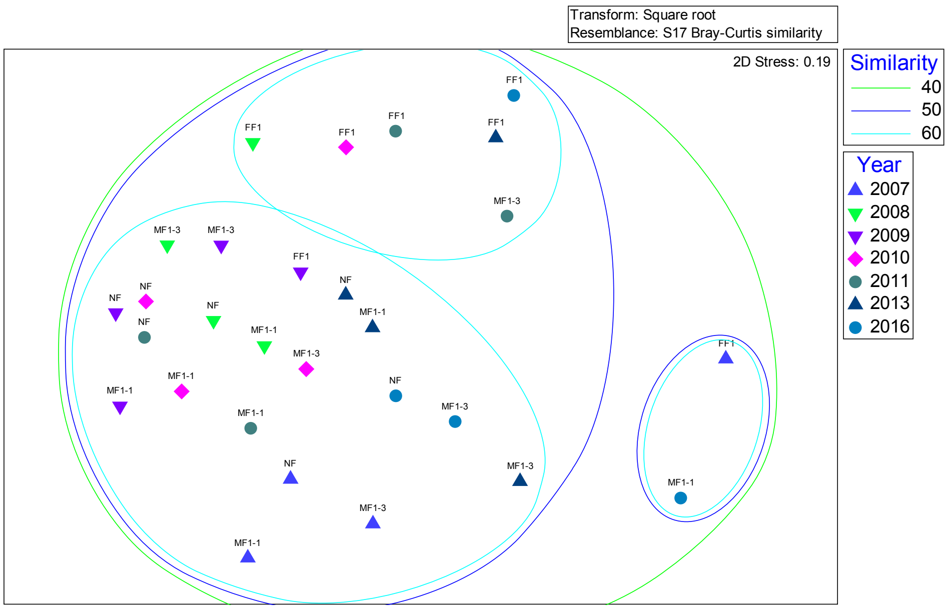
Note: NF vs. FF area comparisons significant with *P*-values <0.05. Significance indicated by **bolded** values.
n/a = not applicable; *P* = probability.

Figure 8-20 Ordination Plots of Benthic Invertebrate Density in the NF and FF areas, 2007 to 2016



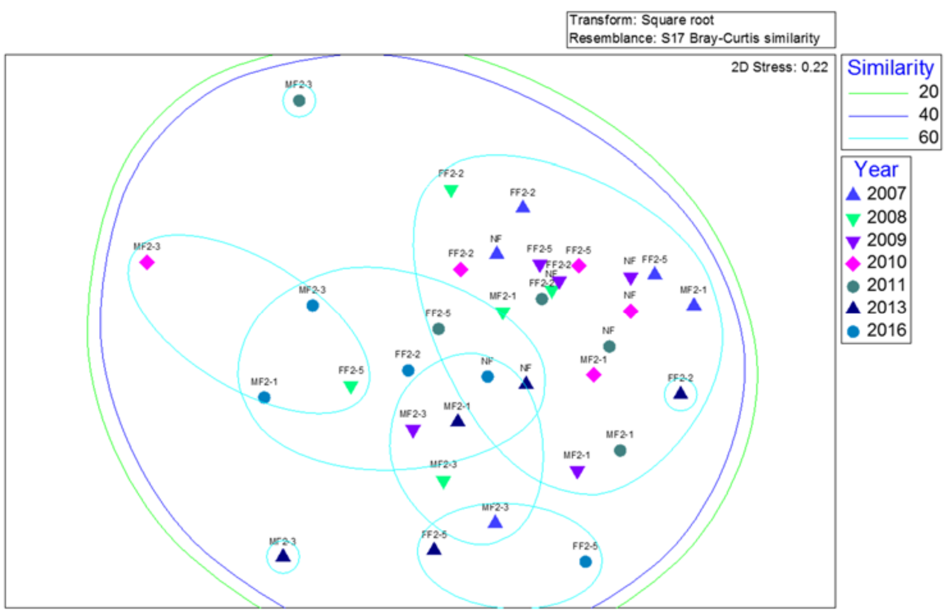
NF = near-field; MF = mid-field; FF = far-field.

Figure 8-21 Ordination Plots of Benthic Invertebrate Density along the NF-MF1-FF1 Transect, 2007 to 2016



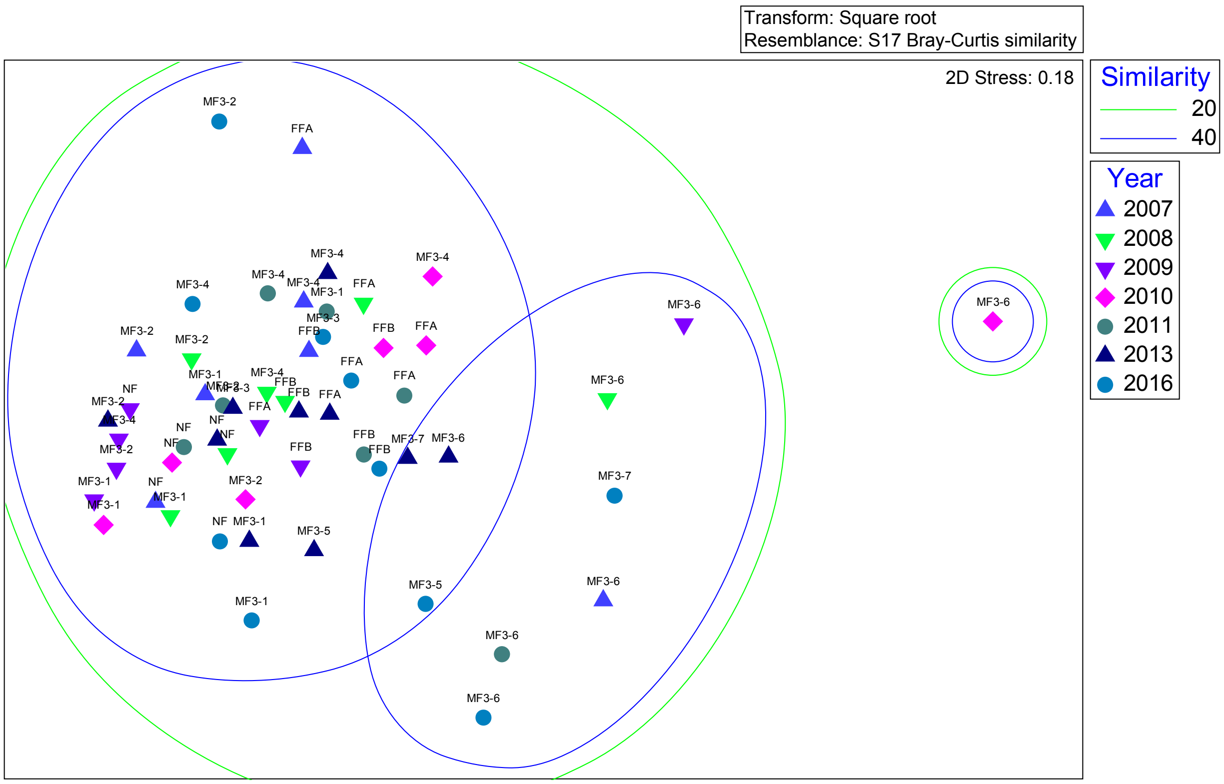
NF = near-field; MF = mid-field; FF = far-field.

Figure 8-22 Ordination Plots of Benthic Invertebrate Density along the NF-MF2-FF2 Transect, 2007 to 2016



NF = near-field; MF = mid-field; FF = far-field.

Figure 8-23 Ordination Plots of Benthic Invertebrate Density along the NF-MF3-FFB-FFA Transect, 2007 to 2016



NF = near-field; MF = mid-field; FF = far-field.

8.3.3.2 Relationships Between the Benthic Community and Environmental Variables

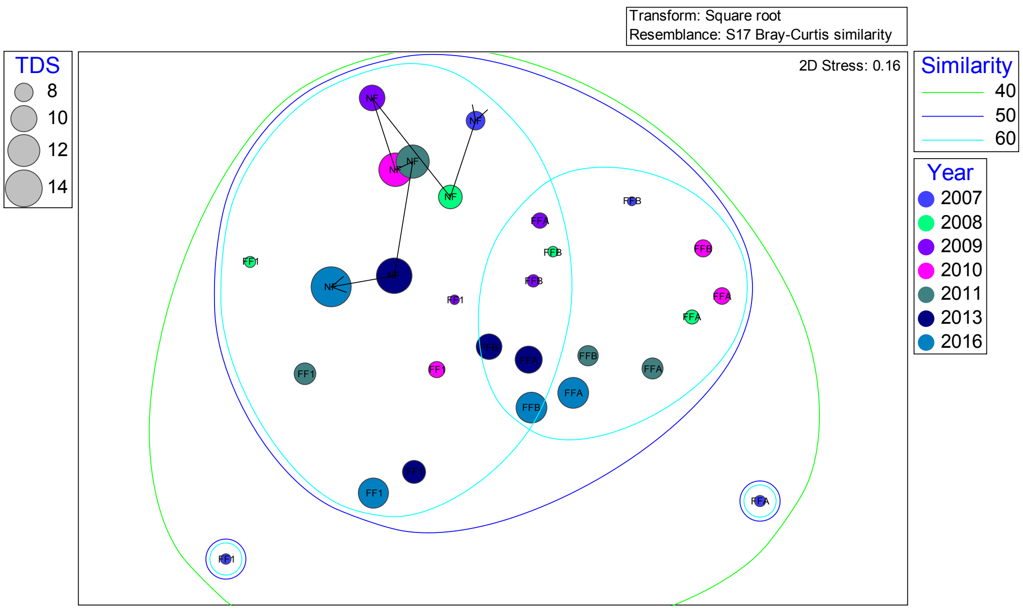
The concentrations of the exposure variable (i.e., bottom TDS, calculated) and selected habitat variables (i.e., sediment TOC and percent fines) were superimposed on the nMDS ordination plots as scaled bubbles for the NF and FF areas, and for each transect. The intent was to allow a visual evaluation of whether the effluent exposure or habitat variation accounted for the variation in benthic invertebrate community structure over time and among sampling areas.

Near-field and Far-field Areas

The concentration of TDS has increased over time in the both the NF and FF areas, with the most notable increases in the FF areas after 2011 (Figure 8-24). If effluent exposure, as indicated by TDS, was an important driver of community composition, then a convergence of the NF and FF areas would be expected over time. This was not apparent on the ordination plot, which showed an approximately parallel movement of NF and FF area symbols over time. Each of the two groups of areas/stations based on the biological data, as delineated by the 60% similarity boundary, included close to the full range of variation in TDS.

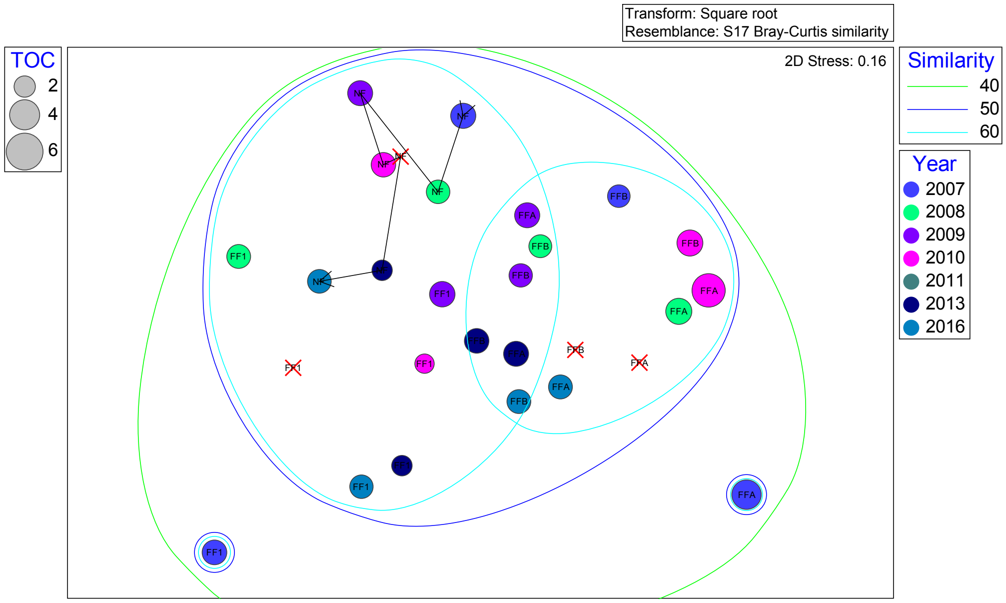
Little variation was observed in TOC and percent fine sediments among areas and over time (Figure 8-25 and 8-26) and reflects the successful application of the study design objectives (e.g., to limit habitat variability among stations). Neither of these variables clearly account for the differences between NF and FF area communities indicated by the ordination results.

Figure 8-24 Ordination Plots of Benthic Invertebrate Density in the NF and FF areas, with Symbols Scaled to TDS Concentration, 2007 to 2016



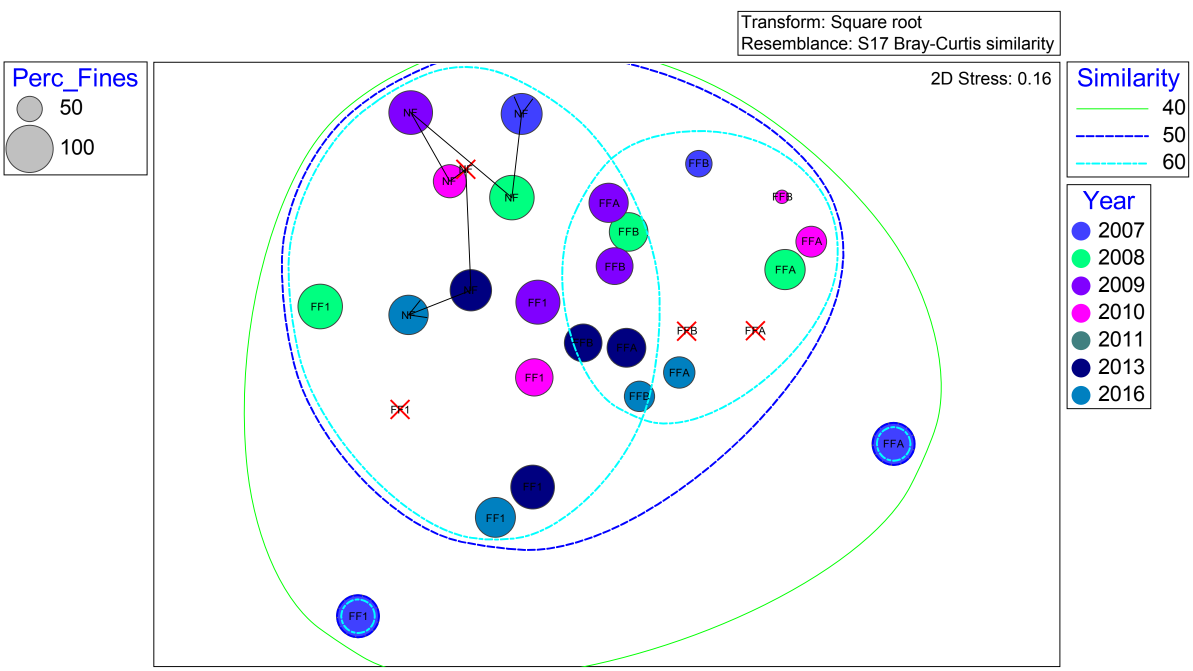
NF = near-field; FF = far-field; TDS = total dissolved solids.

Figure 8-25 Ordination Plots of Benthic Invertebrate Density in the NF and FF areas, with Symbols Scaled to Sediment TOC, 2007 to 2016



NF = near-field; FF = far-field; TOC = total organic carbon. Sediment quality data are not available for 2011, and is indicated by crossed out symbols.

Figure 8-26 Ordination Plots of Benthic Invertebrate Density in the NF and FF areas, with Symbols Scaled to Particle Size, 2007 to 2016

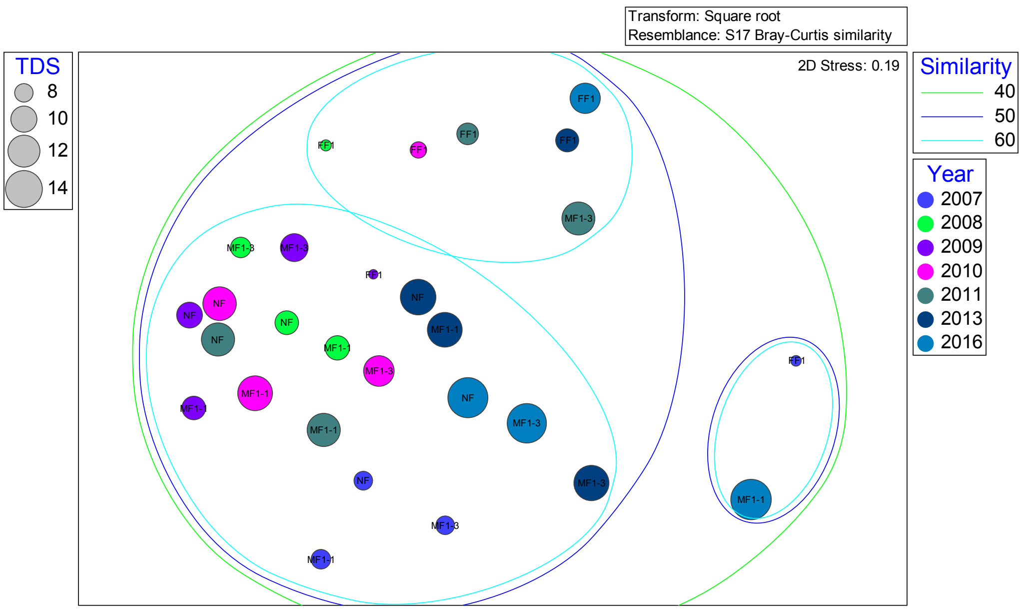


NF = near-field; FF = far-field; Perc_Fines = percent fines. Sediment quality data are not available for 2011, and is indicated by crossed out symbols.

NF-MF1-FF1 Transect

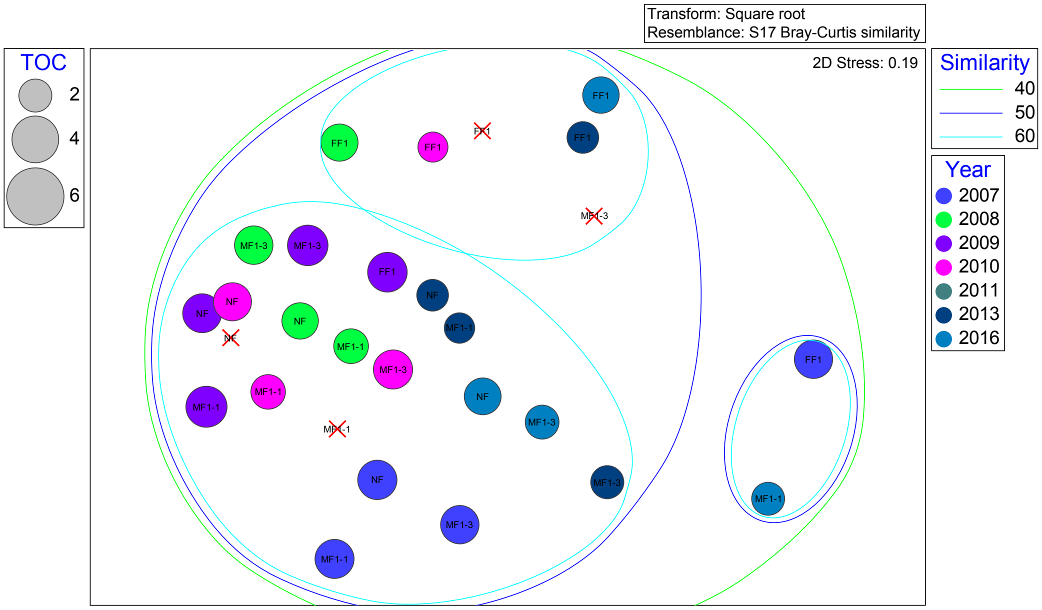
The nMDS ordination plot showing variation in TDS among areas and stations along the NF-MF1-FF1 transect showed no indication that the degree of effluent exposure (TDS concentration) was a key driver of benthic invertebrate community structure. Each group of areas/stations based on biological similarity included close to the full range of variation in TDS concentration (Figure 8-27). Sediment TOC and particle size varied little among years, sampling areas and area/station groups delineated based on the ordination (Figures 8-28 and 8-29). None of the three variables appeared to account for the variation in benthic community structure observed among areas and over time.

Figure 8-27 Ordination Plots of Benthic Invertebrate Density along the NF-MF1-FF1 Transect, with Symbols Scaled to TDS Concentration, 2007 to 2016



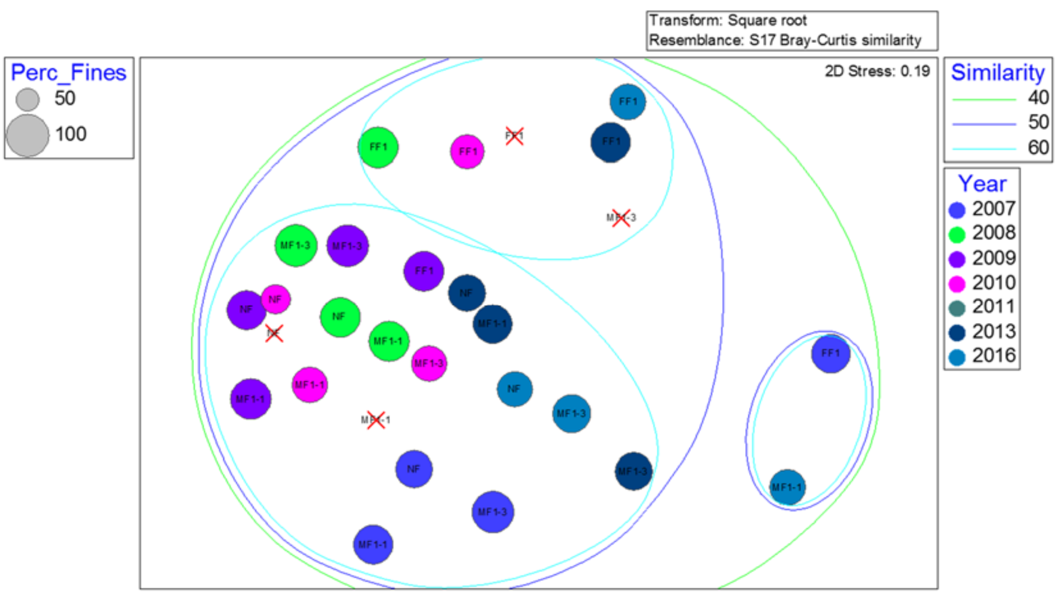
NF = near-field; MF = mid-field; FF = far-field; TDS = total dissolved solids.

Figure 8-28 Ordination Plots of Benthic Invertebrate Density along the NF-MF1-FF1 Transect, with Symbols Scaled to Sediment TOC, 2007 to 2016



NF = near-field; MF = mid-field; FF = far-field; TOC = total organic carbon. Sediment quality data are not available for 2011, and is indicated by crossed out symbols.

Figure 8-29 Ordination Plots of Benthic Invertebrate Density along the NF-MF1-FF1 Transect, with Symbols Scaled to Sediment Particle Size, 2007 to 2016

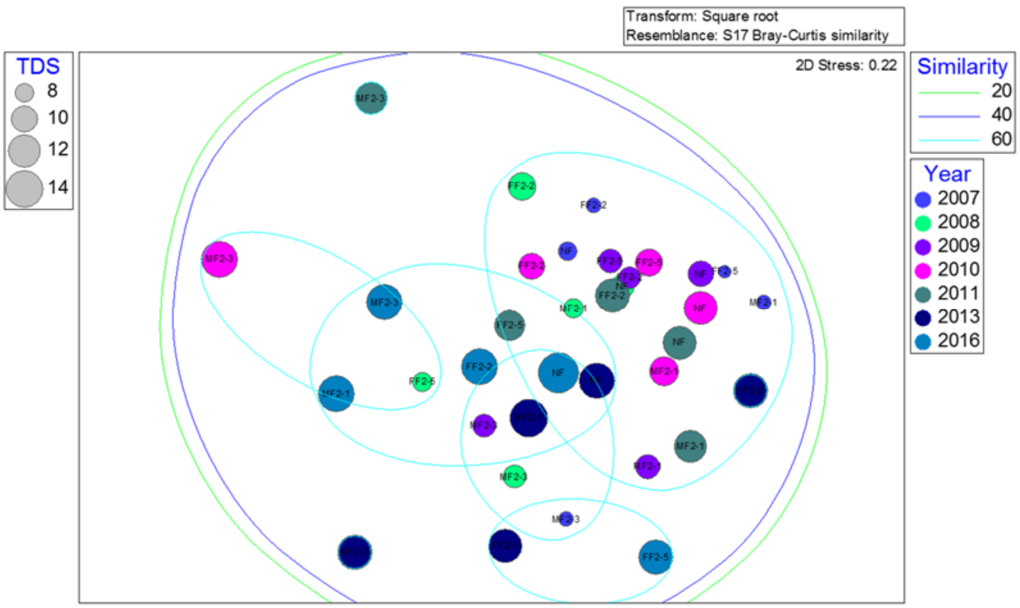


NF = near-field; MF = mid-field; FF = far-field; Perc_Fines = percent fines. Sediment quality data are not available for 2011, and is indicated by crossed out symbols.

NF-MF2-FF2 Transect

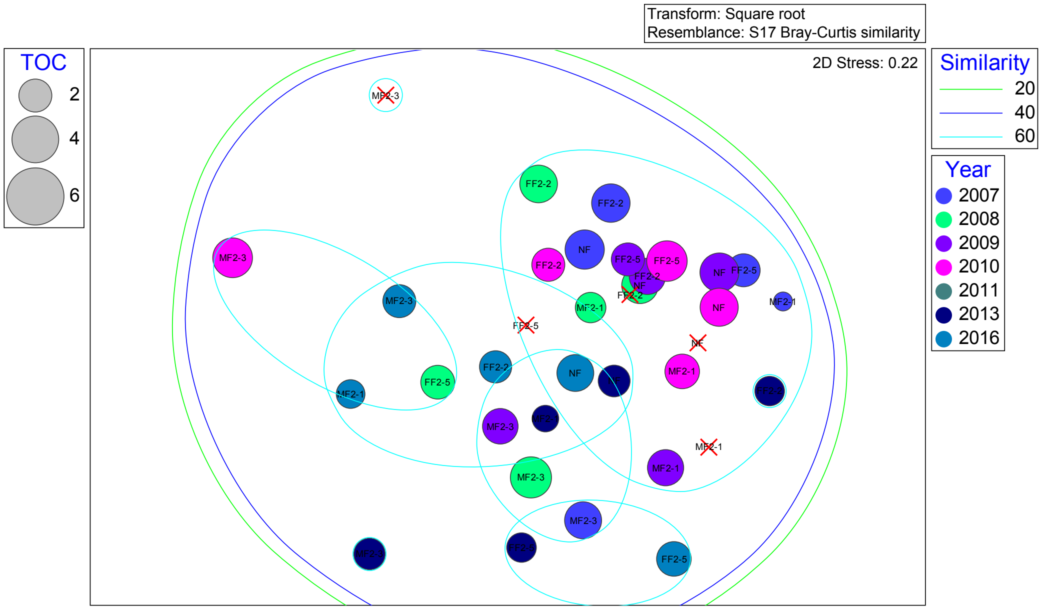
Similar to the NF-MF1-FF1 transect results, the nMDS ordination plot showing variation in TDS among areas and stations along the NF-MF2-FF2 transect showed no indication that the degree of effluent exposure (TDS concentration) was a key driver of benthic invertebrate community structure (Figure 8-30). Each group of areas/stations based on biological similarity included a wide range of variation in TDS concentration. Sediment TOC and particle size varied little among years, sampling areas and clusters delineated by the ordination (Figures 8-31 and 8-32). Effluent exposure and the selected habitat variables did not appear to account for the variation in benthic community structure among areas and over time.

Figure 8-30 Ordination Plots of Benthic Invertebrate Density along the NF-MF2-FF2 Transect, with Symbols Scaled to TDS Concentration, 2007 to 2016



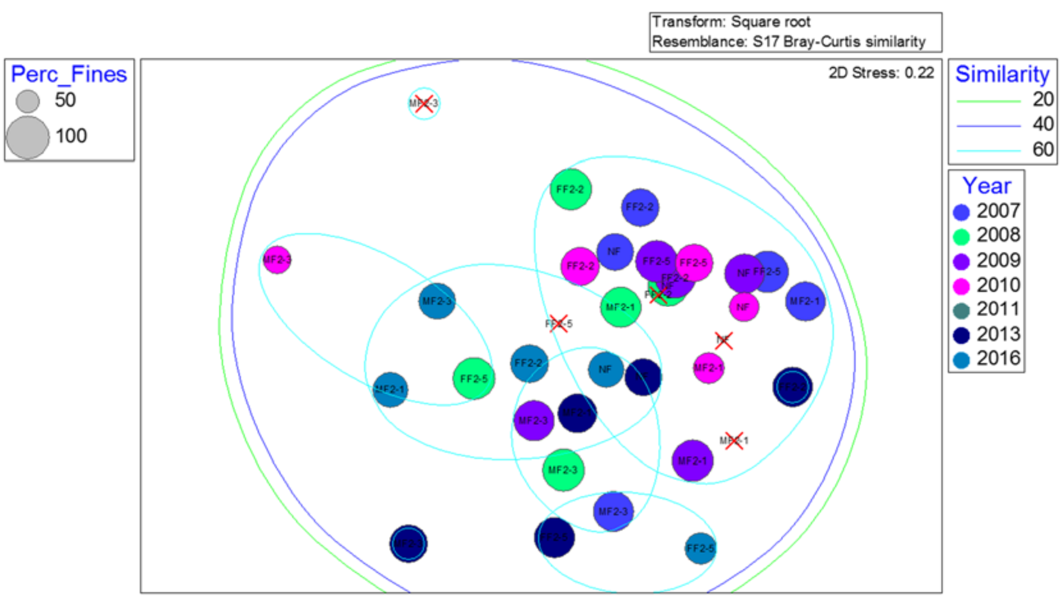
NF = near-field; MF = mid-field; FF = far-field; TDS = total dissolved solids.

Figure 8-31 Ordination Plots of Benthic Invertebrate Density along the NF-MF2-FF2 Transect, with Symbols Scaled to Sediment TOC, 2007 to 2016



NF = near-field; MF = mid-field; FF = far-field; TOC = total organic carbon. Sediment quality data are not available for 2011, and is indicated by crossed out symbols.

Figure 8-32 Ordination Plots of Benthic Invertebrate Density along the NF-MF2-FF2 Transect, with Symbols Scaled to Sediment Particle Size, 2007 to 2016



NF = near-field; MF = mid-field; FF = far-field; Perc_Fines = percent fines. Sediment quality data are not available for 2011, and is indicated by crossed out symbols.

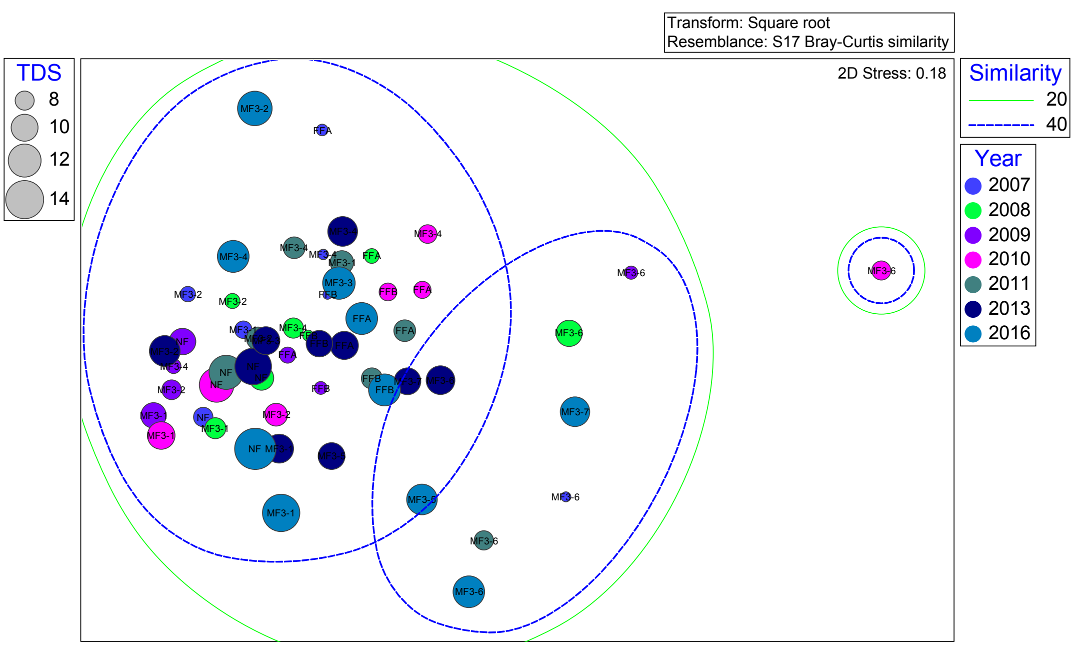
NF-MF3-FFB-FFA Transect

The NF-MF3-FFB-FFA transect represents the longest spatial and effluent exposure gradient sampled in Lac de Gras. Similar to the other two transect results, the nMDS ordination plot showing variation in TDS among areas and stations along this transect showed no indication that the degree of effluent exposure (TDS concentration) was a key driver of benthic invertebrate community structure (Figure 8-33). Each of the two groups of areas/stations based on biological similarity included a wide range of variation in TDS concentration.

Sediment TOC and particle size varied moderately among years, sampling areas and stations, and to a greater extent along the NF-MF3-FFB-FFA transect than observed along other transects (Figures 8-34 and 8-35). Greater variation was observed in percent fines in 2010 and 2016 compared to other monitoring years, but each of the two groups of areas/stations based on biological data included close to the full range of variation in percent fines. Sediment TOC was noticeably lower, as indicated by smaller bubbles, in the group of areas/stations on the left side of Figure 8-34, compared to those in the group of stations on the right side of the figure. This pattern suggests that greater sediment TOC at stations MF3-6 and MF3-7 (median TOC = 4.9%) was an important driver of community differences between these stations and other stations along the transect (median TOC = 2.3%).

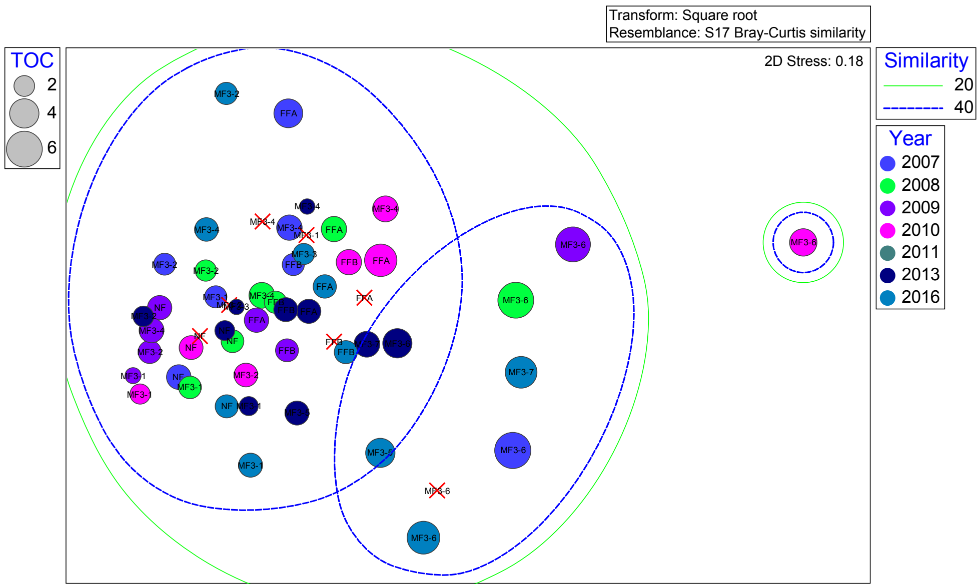
In summary, along the NF-MF3-FFB-FFA transect, effluent exposure and percent fine sediments did not appear to account for the variation in benthic community structure among areas and over time. However, sediment TOC was identified as a driver of benthic community variation along this transect.

Figure 8-33 Ordination Plots of Benthic Invertebrate Density along the NF-MF3-FFB-FFA Transect, with Symbols Scaled to TDS Concentration, 2007 to 2016



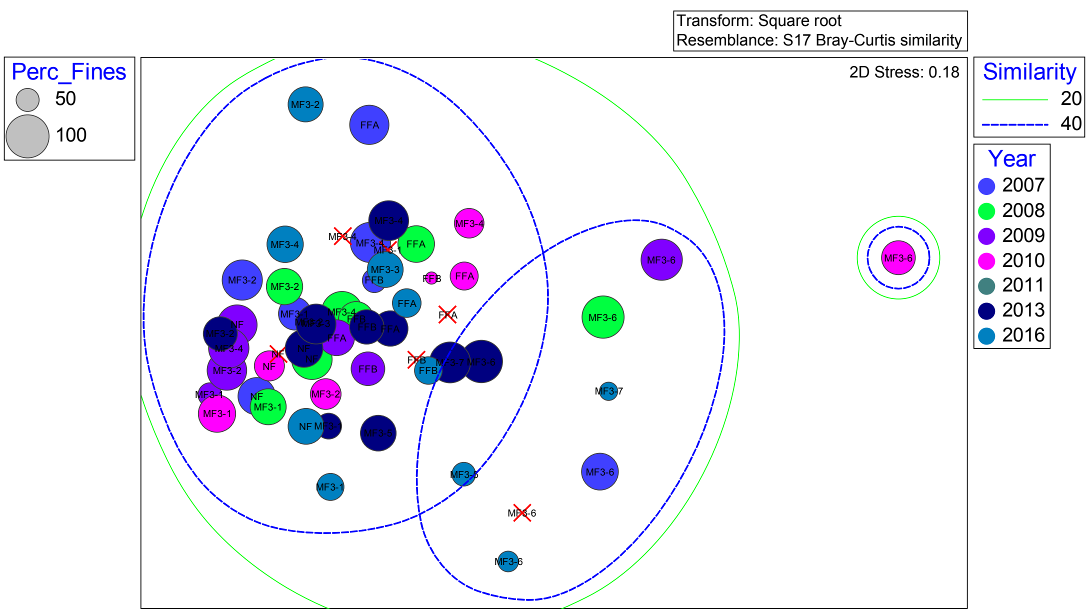
NF = near-field; MF = mid-field; FF = far-field; TDS = total dissolved solids.

Figure 8-34 Ordination Plots of Benthic Invertebrate Density along the NF-MF3-FFB-FFA Transect, with Symbols Scaled to Sediment TOC, 2007 to 2016



NF = near-field; MF = mid-field; FF = far-field; TOC = total organic carbon. Sediment quality data are not available for 2011, and is indicated by crossed out symbols.

Figure 8-35 Ordination Plots of Benthic Invertebrate Density along the NF-MF3-FFB-FFA Transect, with Symbols Scaled to Sediment Particle Size, 2007 to 2016



NF = near-field; MF = mid-field; FF = far-field; Perc_Fines = percent fines. Sediment quality data are not available for 2011, and is indicated by crossed out symbols.

8.3.3.3 Summary of Multivariate Analysis Results

The nMDS ordination results indicated a separation of the NF area communities from one of more of the FF areas during some years of monitoring. The benthic community of the NF area was found to be more similar to the FF1 area community, compared to the FFA and FFB areas. This result was as expected based on likely habitat similarity, because the NF area is closer to the FF1 area than to FFA and FFB, and both the NF and FF1 areas are located in “narrow” areas of Lac de Gras, and closer to shore than the mid-lake FFA and FFB areas.

Variation in benthic community composition along the NF-MF1-FF1 transect mostly indicated a difference between communities of the FF1 area and other stations along the transect, and showed more pronounced differences among stations and areas in 2016 compared to previous years. The NF-MF2-FF2 transect ordination had a relatively high stress value of 0.22, potentially indicating poor fit to the input data, and showed numerous overlapping groups with no indication of a Mine or habitat-related effect. On the ordination plot of the NF-MF3-FFB-FFA transect, most stations and years fell within the same group, with the exception of the MF3-6 and MF3-7 stations in nearly all monitoring years, which formed a separate group.

A visual evaluation of relationships between area/station groups delineated on ordination plots and three environmental variables (i.e., TDS as the indicator of effluent exposure, sediment TOC, and percent fines) detected no patterns related to TDS or percent fine sediments. Groups of areas/stations delineated based on benthic community composition included nearly the full range of TDS and percent fines. Habitat-related separation of biological area/station groups was only observed along the NF-MF3-FFB-FFA transect, where sediment TOC was identified as a driver of benthic community variation.

Although results of multivariate analysis showed some separation of the NF and FF area communities, which implies an effect of the Mine discharge on benthic community structure, the relative importance of habitat variation versus exposure to the Mine effluent could not be evaluated.

8.3.4 Directives and Information Requests Related to Benthic Invertebrate Monitoring

8.3.4.1 Decreases in Pisidiidae Densities

In 2016, comparisons of Pisidiidae relative density across AEMP sampling areas showed that Pisidiidae contributed a relatively large proportion of the total invertebrate density at most of stations (19% on average), with the highest relative density observed at Station FF1-4 (66%). In general, stations closer to the diffuser had lower relative densities of Pisidiidae, whereas greater relative densities were observed in the FF areas. In addition, significantly lower Pisidiidae density was detected in the NF area compared to the FF areas during the 2016 statistical analysis (Golder 2017c). Based on this result, Pisidiidae density in 2016 triggered an Action Level 1 for toxicological impairment.

For additional context to the result of lower Pisidiidae density in the NF area in 2016, it is relevant to note that Pisidiidae density in 2016 was not unusual for this sampling area, or for Lac de Gras in general. Pisidiidae density has been variable in all sampling areas in Lac de Gras (Figure 8-12), with a normal range of 0 to 206 organisms/m^2. Mean Pisidiidae density was 52 organisms/m^2 in the NF area in 2016, similar to the mean value observed in this area during the baseline period in 1996 (53 organisms/m^2) and greater than mean NF area values observed in 2005 and 2007 (19 and 23 organisms/m^2, respectively). In addition, although mean Pisidiidae density was significantly lower in the NF area compared to the FF areas in 2016, it remained within the normal range.

The evaluation of linear trends in Section 8.3.2.3 showed a non-significant decline in Pisidiidae density in the NF area, varying trends in the MF areas, and no trends in the FF areas. Closer examination of the time series plots for different lake areas show a similar pattern in Pisidiidae density throughout Lac de Gras, but with greater variability in the NF area (Figure 8-12). In all sampling areas, Pisidiidae densities were similar and low in 2007, were greater from 2008 to 2011 or 2013 (i.e., above the normal range at some stations in most years), and then declined in 2016 to within the normal range. These results suggest lake-wide temporal variation in Pisidiidae density, which may be further influenced by the Mine effluent discharge and habitat variation.

The reasons for the observed decline in Pisidiidae density in the NF area in 2016 may reflect a variety of factors. An effect due to water or sediment toxicity is unlikely. Water quality constituents remained at concentrations below AEMP benchmarks in the NF area in 2016 (Golder 2017c) and sediment quality substances of interest were below levels of concern (Golder 2017c). In addition, no potential toxic effects were observed on other benthic invertebrate or plankton variables (Golder 2017c).

The simplest explanation for the 2016 results is that the enrichment effect originating from the diffuser resulted in a greater Chironomid midge dominance in areas closer to the diffuser, with a corresponding reduction in the density of Pisidiidae. The types and directions of effects on other benthic invertebrate variables, also suggested that the decline in Pisidiidae density reflected the effects of community changes caused by nutrient enrichment.

Findings of some previous studies suggest that the effect of nutrient enrichment on fingernail clam densities may be in either direction. Nutrient enrichment has been shown to increase the dominance of a few species and decrease evenness of the community (Hillebrand 2003). Fertilization experiments have been shown to dramatically increase filter feeder densities as a result of increased primary productivity (Namayandeh and Quinlan 2011). Pisidiidae were initially favoured by increased nutrient concentrations, but in deeper locations (i.e., 11 to 17 m) decreases in densities were eventually observed with increased enrichment (Holopainen and Jonasson 1989). In contrast, an Arctic lake fertilization experiment found that density of *Pisidium* clams differed little between the exposure and reference areas over time (Hershey 1992). Consistent with the findings of Holopainen and Jonasson (1989), a pattern of an initial increase in Pisidiidae density, followed by a decline in recent years is apparent in the NF area in Lac de Gras, as shown in Figure 8-11.

It also remains possible that the lower Pisidiidae density in 2016 at least partly reflects habitat variation among sampling areas. Some habitat variation was observed among sampling areas in 2016, with the NF area having a greater mean proportion of fine sediments (79%) than two of the FF areas (FF1: 80%; FFA: 61%; FFB: 59%). The NF area was similar to the FF areas in terms of water depth and sediment TOC content. A weak negative correlation was detected between Pisidiidae density and percent fine sediments, which suggests the lower Pisidiidae density in the NF area is consistent with the response of this invertebrate family to variation in sediment particle size distribution.

Overall, the lower Pisidiidae density in the NF area in 2016 is most likely linked to a combination of nutrient enrichment and habitat variation. Other AEMP components that sampled the same stations have demonstrated responses to nutrient enrichment, and the 2016 analyses implicated variation in sediment physical characteristics as a potential influence on Pisidiidae density. The mechanism of effect remains uncertain, and may incorporate biological interactions. The lower Pisidiidae density in the NF area may reflect the greater degree of midge dominance in areas closer to the diffuser in response to nutrient enrichment, through competition for resources (i.e., space and food). Invertebrate densities in Lac de Gras, however, remain relatively low, consistent with the oligotrophic status of the lake. In the absence of lower near-bottom DO concentrations in the NF area, the effect on Pisidiidae in 2016 is unlikely to have resulted from enrichment-related DO stress.

In light of this evaluation, and the observation that Pisidiidae density in the NF area remained within the normal range, the appropriate action to follow up on the observed effect on Pisidiidae density is to confirm the effect during the next comprehensive AEMP sampling year.

8.3.4.2 Bray-Curtis Index Taxonomic Resolution Issue

The BCI is a measure of ecological distance between two communities, and is used during comprehensive AEMP years to evaluate whether the ecological distance between FF areas and areas closer to the Mine discharge (NF and MF areas) differs from those representing background variation in community structure among the three FF areas. The three FF areas included in this analysis are FF1, FFA and FFB areas. These areas were considered representative of background conditions in Lac de Gras from 2007 to 2010, and as minimally-affected FF areas in subsequent years. The BCI is also useful for evaluating temporal trends, which provide an indication of whether FF area communities remain similar to one another, and whether NF and FF area communities are diverging over time from the FF area communities. BCI values from 2007 onwards were calculated using the “all pair-wise comparisons” method, as described in the 2011 AEMP Benthic Invertebrate Report (Golder 2012) and as prescribed by the *AEMP Design Plan 4.1* (Golder 2017d).

Upon reviewing the *2011 to 2013 Aquatic Effects Re-evaluation Report (Golder 2016a)*, EMAB expressed a concern that historical benthic invertebrate data (i.e., 1996, 1997 and 2001 to 2006) were excluded from calculating BCI due to variation in taxonomic resolution between the historical period and more recent monitoring years (i.e., 2007 onwards). EMAB also questioned whether BCI could be calculated for historical data using the subfamily/tribe taxonomic level (for Chironomidae). The intent of EMAB’s comment was to encourage use of other methods to avoid excluding data from the analysis of long-term trends.

As part of the data summary provided herein, the historical data were reviewed to assess whether BCI could be calculated for the historical data for long-term trend analysis. The main criterion for this evaluation was the ability to have a long-term record of BCI that would allow an evaluation of variation over time in community structure (1) among FF areas and (2) between FF areas and the NF and MF areas, based on pairwise comparisons of stations.

This assessment revealed that there are insufficient historical data for calculating BCI among the FF areas to characterize background variation in years prior to 2007, for the following reasons:

- In 1996, a single station (F-16) was sampled in the current FF1 area, which doesn’t allow FF area to FF area comparisons using the BCI.
- In 1997 and 2001 to 2006, four and three closely-spaced stations, respectively, were sampled near the current AEMP station FFA-5; no other FF areas were sampled. Therefore, FF area to FF area BCI comparisons are also not possible for these years.

Similarly, calculation of BCI indices for NF and MF areas, comparable to those calculated for the 2007 to 2016 period is not possible, for the following reason:

- As indicated above, only one to four FF stations, located in a single FF area, are available for BCI calculation for 1996, 1997 and 2001 to 2006. Under current AEMP methods, BCI values calculated for each NF and MF station were the means of pairwise BCIs comparing the station to each FF station (15 stations in three FF areas). Therefore, although BCI values could be calculated for each NF or MF station during this period, they would not be comparable to those calculated from 2007 onwards, because they would be based on substantially less data.

Furthermore, in 1996 and 1997, the samples were screened using a 250 µm mesh sieve, which is inconsistent with methods used during subsequent monitoring, when a 500 µm sieve was used to process samples. In addition, different taxonomists were used between 1996 to 1997, 2001 to 2006, and 2007 onwards. Taxonomic resolution was slightly different among the three taxonomists in terms of the number of chironomid midges identified only to subfamily/tribe. Most critically, sampling design varied between the pre- and post-2007 sampling programs, as described above.

Therefore, BCI was not calculated for pre-2007 data, and the 2007 to 2016 dataset was used for the trend analysis and time series plots for this variable.

8.3.5 Comparison to EA Predictions

No specific predictions were made in the EA regarding the benthic invertebrate community. An increase in primary productivity (i.e., phytoplankton) in part of Lac de Gras, resulting from nutrient inputs (particularly phosphorus) from the Mine discharge was predicted for the plankton community (Section 7). This increase can be expected to also result in increased secondary productivity (i.e., zooplankton and benthic invertebrates).

The results of the benthic invertebrate analysis are consistent with the prediction of increased primary productivity in Lac de Gras originating from the Mine discharge. Several benthic invertebrate variables (i.e., total density, percent Chironomidae, SEI, BCI, Pisidiidae density, *Procladius* density, *Micropsectra* density) exhibited trends across the gradient of exposure to Mine effluent, with the directions of the trends indicating greater effects near the Mine effluent discharge in 2016, and in directions consistent with nutrient enrichment resulting in increased primary productivity. A more detailed evaluation of EA predictions related to nutrient enrichment are provided in Section 5 (Eutrophication Indicators).

8.4 Summary and Conclusions

Results of this review of the benthic invertebrate community data collected under the AEMP are summarized below.

Temporal Trends

- Total benthic invertebrate density declined in all areas of Lac de Gras, with statistically significant declines observed in NF and FF1 areas and the MF2-2/MF2-3 stations. In 2016, density at most stations was within the normal range.

- Richness values before 2007 were at the upper limit or above the normal range, but since 2007, richness was within or slightly above the normal range in all areas. Significant declines were detected at the MF2-2/MF2-3 stations and the FFA area.
- Dominance fluctuated between 2001 and 2006, after which it was mostly within the normal range in all sampling areas, with occasional values above the upper limit more commonly seen in the NF and MF areas. Dominance showed slight, non-significant increasing trends in NF area and most MF areas, and no trends in the FF areas.
- SDI was within or occasionally below the normal range in all areas of Lac de Gras since 1996. Trend analysis results indicate mostly stable communities in the NF, FFA and FFB areas, and slight increases over time in the MF areas.
- The SEI increased in the NF area from 2001 to 2007 and then declined in the NF area to near the lower limit of the normal range by 2016. The SEI also remained mostly within the normal range in other sampling areas, with occasional excursions below the lower limit in multiple years in all areas.
- BCI has been calculated since 2007, and has generally remained within the normal range in all sampling areas, with only minor excursions above the normal range.
- The relative density of Chironomidae has undergone substantial variation over time in Lac de Gras. In the NF area, percent Chironomidae has consistently remained within the normal range since 2007. The MF and FF areas showed a greater degree of variation. Trend analysis detected no significant trend in the NF area, but declining trends were apparent at the FF2-2 station, and the FF1 and FFB areas.
- Pisidiidae density has shown a moderate amount of variation over time, with frequent exceedances of the normal range. Recently, Pisidiidae density declined to within the normal range in the NF and MF areas, with values near the lower limit in the NF area in 2016. The FF area values have shown similar trends, and were within the normal range in 2016.
- *Procladius* density has shown large variation in all sampling areas throughout the monitoring period. It has generally remained above the normal range in the NF area, and within or above the normal range in the MF and FF areas. No long-term trends were detected in *Procladius* density.
- The density of *Heterotrissocladius* in the NF area has been highly variable prior to 2013, but was within the normal range in the NF area in 2016. Less variation over time was observed in the MF and FF areas. Trend analysis showed generally declining trends in all areas.
- *Micropsectra* density was highly variable in all sampling areas until 2010, and subsequently declined to within the normal range in all sampling areas. All sampling areas showed similar negative trends in density over time.

Community Composition by Major Group

- Chironomidae midges have dominated the Lac de Gras benthic invertebrate communities in all sampling areas during baseline years and each year of monitoring under the AEMP. Pisidiidae were the second most abundant group in most years, but rarely accounted for more than 20% of total density.
- The NF area community in 2016 closely resembled the baseline community sampled in 1996. A divergence in community composition was apparent between the NF and FF areas in recent years. The

relative density of Pisidiidae has been low (<10%) in the NF area and at most MF stations, but remained greater (>20%) in the FF areas. Composition of the MF area communities was intermediate between the NF and FF area communities.

Multivariate Analysis

- The nMDS ordination results indicated a separation of the NF area communities from one of more of the FF areas during some years of monitoring. The benthic community of the NF area was found to be more similar to the FF1 area community than to the FFA and FFB areas, which likely reflects the degree of habitat similarity among sampling areas.
- Variation in benthic community composition along the NF-MF1-FF1 transect mostly indicated a difference between communities of the FF1 area and other stations along the transect. The NF-MF2-FF2 transect ordination results were subject to uncertainty, and showed numerous overlapping groups. Along the NF-MF3-FFB-FFA transect, most stations and years fell within the same group, with the exception of the MF3-6 and MF3-7 stations in nearly all monitoring years, which formed a separate group.
- A visual evaluation of relationships between area/station groups delineated on ordination plots and three environmental variables (i.e., TDS, an indicator of effluent exposure, sediment TOC, and percent fines), detected no patterns related to TDS or percent fine sediments. Habitat-related separation of areas/stations was only observed along the NF-MF3-FFB-FFA transect, due to variation in TOC. The relative importance of habitat variation versus degree of exposure to the Mine effluent could not be evaluated.

Directives

- EMAB expressed a concern that historical benthic invertebrate data were excluded from calculating BCI due to variation in taxonomic resolution between the historical period and more recent monitoring years. The results of this review indicate that historical data should not be included in BCI calculations, due to differences in sampling design between the pre- and post-2007 programs, which precludes the inclusion of among-FF area community differences as the basis of evaluating the importance of FF area to NF/MF area community differences. Differences in sampling mesh size and taxonomic resolution also limit the use of the BCI for the pre-2007 data, but to a lesser extent. Therefore, BCI was not calculated for pre-2007 data.
- In response to a request from the WLWB, the declines in Pisidiidae density observed in 2016 in the NF and MF areas were further evaluated. It was concluded that the lower Pisidiidae density in the NF area in 2016 was most likely linked to a combination of nutrient enrichment and habitat variation, rather than a toxicological effect. The mechanism of effect remains uncertain, and may incorporate biological interactions.

Overall Conclusions

- Benthic invertebrate monitoring in Lac de Gras continues to provide evidence of mild nutrient enrichment in areas close to the Mine effluent discharge, as indicated by total density, richness, and dominance within or above normal ranges, and reduced evenness, greater densities of a number of common invertebrates, and stronger dominance of chironomid midges, relative to the FF areas.
- Trend analysis and visual evaluation of trends found mostly declining trends in the NF area during recent years in invertebrate densities (except *Procladius*) and evenness, and no changes in richness, diversity and BCI. Most variables that were above normal ranges before 2013 have returned to within normal ranges, suggesting a recent reduction in the magnitude of Mine-related effect on the benthic community in Lac de Gras.
- Analyses incorporating the entire communities showed differences in community composition between the NF and FF areas, but the relative importance of habitat variation versus exposure to the Mine effluent accounting for this separation could not be evaluated.

9 FISH

9.1 Introduction

The fish chapter provides a summary of changes observed in fish health and fish tissue chemistry of Slimy Sculpin (*Cottus cognatus*) in Lac de Gras over time. It also provides an update on the monitoring of mercury concentration in Lake Trout (*Salvelinus namaycush*). These fish have been monitored in Lac de Gras as a component of the AEMP since 2007. The objectives of this section are as follows:

- summarize Mine-related effects observed from 2014 to 2016 and compare these to effects observed previously (i.e., from 2007 to 2013); and
- analyze temporal trends in fish health and fish tissue for the period extending from baseline (where possible) to 2016.

9.1.1 Background

DDMI has been conducting studies and monitoring programs relating to the aquatic ecosystem of Lac de Gras since 1994, with AEMP data collected under AEMP study designs Version 1.0 (2001 to 2006), Version 2.0 (2007 to 2011), Versions 3.0 (2012 to 2016), and most recently under Versions 4.0 (2017). A fish program was included as a component of the AEMP since 2007 as there is potential for Mine effluent to affect Lac de Gras. There is a requirement under the *Fisheries Act* Authorization to conduct monitoring of fish populations and indices of fish health. Small-bodied fish health and tissue chemistry are monitored during comprehensive years but are not included in interim years. Additional fish studies with large-bodied fish (i.e., Lake Trout) have been performed in some interim years. The objective of the fish program is to monitor fish health and tissue chemistry in Lac de Gras, and to confirm if the predictions from the EA (DDMI 1998a) were met. The fish program uses Slimy Sculpin as a sentinel fish species, and has been designed and implemented to determine whether effluent discharged from the Mine is having an effect on the Lac de Gras Slimy Sculpin fish population.

9.1.2 Component History

The Fish program consists of two main components, fish health and fish tissue chemistry. The fish health and fish tissue surveys are completed with Slimy Sculpin in Lac de Gras. Additional fish tissue mercury data are collected from Lake Trout from Lac de Gras and Lac du Sauvage.

Four AEMP fish health and tissue chemistry surveys have been completed with Slimy Sculpin in Lac de Gras (i.e., in 2007, 2010, 2013, and 2016). In addition to the four surveys conducted under the AEMP, one other study on Slimy Sculpin in Lac de Gras was undertaken. A fish heath and tissue chemistry survey was performed in 2004 (CRI 2006) where Slimy Sculpin were collected from East Island on Lac de Gras and assessed for length, weight, condition, population structure, and tissue chemistry.

During the 2007 AEMP, elevated mercury concentrations were detected in Slimy Sculpin captured from the NF area compared to the FF areas. In response, DDMI initiated a monitoring program for mercury in Lake Trout that was subsequently added as a component to the AEMP in the *AEMP Study Design Version 3.5* (Golder 2014a). Historically, Lake Trout tissue chemistry was measured in muscle, liver and kidney tissue collected in Lac de Gras in 1996, 2002, 2003, 2004, 2005 and 2008, and in Lac du Sauvage in 1996, 2008. Mercury concentrations were measured in muscle in 2011 and 2014 in both Lac de Gras and Lac du Sauvage as part of the AEMP. Mercury in Lake Trout muscle was also collected as part of the palatability studies in 2002, 2003, 2004, 2012 and 2015; the palatability data are presented herein for comparison purposes.

Although *AEMP Study Design Version 3.5* (Golder 2014a) was the approved Version of the AEMP design at the time of the surveys, a number of updates outlined in the proposed *AEMP Design Plan Version 4.1* (Golder 2017d) and in WLWB directives (28 July 2015, 26 May 2016, 14 November 2016 and 2 March 2017 Decision Packages) were incorporated into the 2016 Fish program, where relevant.

9.2 Fish Health

9.2.1 Methods

9.2.1.1 Data Sources

Five fish health surveys have been carried out with Slimy Sculpin in Lac de Gras (Table 9-1):

- 2004 survey (CRI 2006);
- 2007 AEMP (August 22 to September 2) (Golder 2008);
- 2010 AEMP (June 28 to July 20) (Golder 2011c);
- 2013 AEMP (August 27 to September 10) (Golder 2014c) and;

2016 AEMP (August 31 to September 14) (Golder 2017c).

As noted above, Slimy Sculpin were collected from East Island on Lac de Gras as part of the 2004 fish survey (CRI 2006). These data are not included in the current review because the methods used between this study and the AEMP design were not the same (e.g., stage detemination, data handling method of parasitized fish).

The 2007 survey was conducted in late summer. The 2010 fish survey was moved to spring to try to improve field methods, capturing fish prior to spawning when gonad size is generally larger and it is easier to distinguish adult fish from young fish. The 2010 fish survey began on 28 June, which was the earliest date at which sufficient ice had melted to allow access to sampling areas by boat. Unfortunately, fish had already spawned and the conclusion was that Slimy Sculpin at this latitude likely spawn under ice-covered conditions. An updated review of recommended sampling times was available later in 2010 (Barrett et al. 2010) and recommended Slimy Sculpin be sampled in late fall. The subsequent fish survey in 2013, and again in 2016, returned to late summer sampling (i.e., when gonads were expected to exhibit early stages of development in preparation for the next spawning period, and gonads were of sufficient size to measure in the field). Given the above noted inconsistencies, the 2010 spring data were omitted from this review. It is not valid to compare a fish who has completed spawning with a fish who has begun gonad development for the subsequent year, as their energy allocation and, therefore, growth and organ weights are expected to be different. Further, differences in gonad and body size due to time of year may not be distinguishable from potential differences from Mine effluent. The 2010 Slimy Sculpin fish health results are not provided in trends sections (i.e., comparisons to normal range and temporal trends), due to the incompatibility in sampling seasons. The 2010 comparison to normal ranges were summarized in the *2011 to 2013 Aquatic Effects Re-evaluation Report* (Golder 2016a) and are not repeated here. The 2010 results are included in this report for comparison of catch rates and incidence of tapeworm infection.

Table 9-1 Review and Summary of AEMP Fish Health Data Collected (up to and including 2016)

Waterbody	Area	Years Sampled												
		2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Lac de Gras	Near-field (NF)				✓			X			✓			✓
	Far-field 2 (FF2)				✓			X			✓			✓
	Mid-field 3 (MF3)	X									✓			✓
	Far-field 1 (FF1)				✓			X			✓			✓
	Far-field A (FFA)				✓			X			✓			✓

✓ = fish health data collected and included in analysis

X = fish health data collected but not included in comparison to normal ranges and temporal trends due to differences in sampling seasons.

9.2.1.2 Data Handling

9.2.1.2.1 Data Screening

The combined 2007, 2013, and 2016 fish health datasets were screened to remove fish infected with adult tapeworms, following the methods described in previous AEMP reports (Golder 2014c, 2017c) and the *AEMP Study Design Version 3.5* (Golder 2014a). Inclusion of infected fish in data analyses could increase the variability of the data; therefore, this source of variability was removed by excluding fish infected with tapeworms during field sampling using a visual, external assessment.

Following the removal of tapeworm infected fish, the dataset was examined for accuracy of gonad weight (which was used to determine stage, as detailed in Section 9.2.1.2.3) in the following cases:

- fish with total length of 30 mm or less (i.e., fish that were expected to be YOY) but with a recorded GSI of 0.5 or greater
- fish with a recorded GSI of 4 or greater

For both data subsets, the field photographs of separated gonads, the field notes, and recorded field crew comments were used to identify potential transcription errors, scale accuracy concerns, or potential inclusion of non-gonad material in gonad weight. This examination resulted in the removal of two adult fish (one from FF1, sampled in 2007, and one from FFA, sampled in 2010), and four young of the year fish (three from FF1 and one from FF2, all four sampled in 2010).

Statistical outliers were identified by the analysis of studentized residuals (SR), where observations that were more than 3.5 SR from the mean were considered to be statistical outliers (Section 9.2.1.3.5). The outlier screening approach was approved as part of the *2011 to 2013 Aquatic Effects Re-evaluation Report* (Golder 2016a) to handle outliers in the AEMP datasets. Statistical outliers identified during modeling were removed from analyses, but included on the plots describing analysis predictions for full transparency.

9.2.1.2.2 Descriptive Statistics

Summary statistics (i.e., sample size, arithmetic mean, and SD) were calculated for each sampling area and each biological variable for age-1+, adult males, and adult females separately. Common fish indices, as described in the MMER TGD (Environment Canada 2012), describing relationships between body metrics (i.e., Fulton's condition factor [K], liversomatic index [LSI] and gonadosomatic index [GSI]) were calculated as follows:

$$\text{Fulton's Condition Factor (Age-1+) } K = \frac{total\ body\ weight}{fork\ length^3} \times 100{,}000$$

$$\text{Fulton's Condition Factor (adults) } K = \frac{carcass\ weight}{fork\ length^3} \times 100{,}000$$

$$\text{Liversomatic Index } LSI = \frac{liver\ weight}{carcass\ weight} \times 100\%$$

$$\text{Gonadosomatic Index } GSI = \frac{gonad\ weight}{carcass\ weight} \times 100\%$$

Carcass (i.e., eviscerated) weight was used in the calculations of GSI and LSI to eliminate confounding effects of including organ weight in body weight; gonad development can differ among individual fish in the fall. However, condition factor for age-1+ was calculated using total body weight as the more appropriate metric for this calculation (Arciszewski et al. 2010). The age-1+ fish gonads are not developed in the fall, so there is no need to eliminate this as a confounding effect. In addition to these indices, total weight adjusted to the mean total length, liver weight adjusted for size (carcass weight), and gonad weight adjusted for size (carcass weight) were also provided as summary statistics. Since the ages of individual fish could not be determined with sufficient accuracy (CRI 2014), mean age as a variable of interest was not evaluated. Stomach content data are presented as a presence/absence table only, with no further analysis.

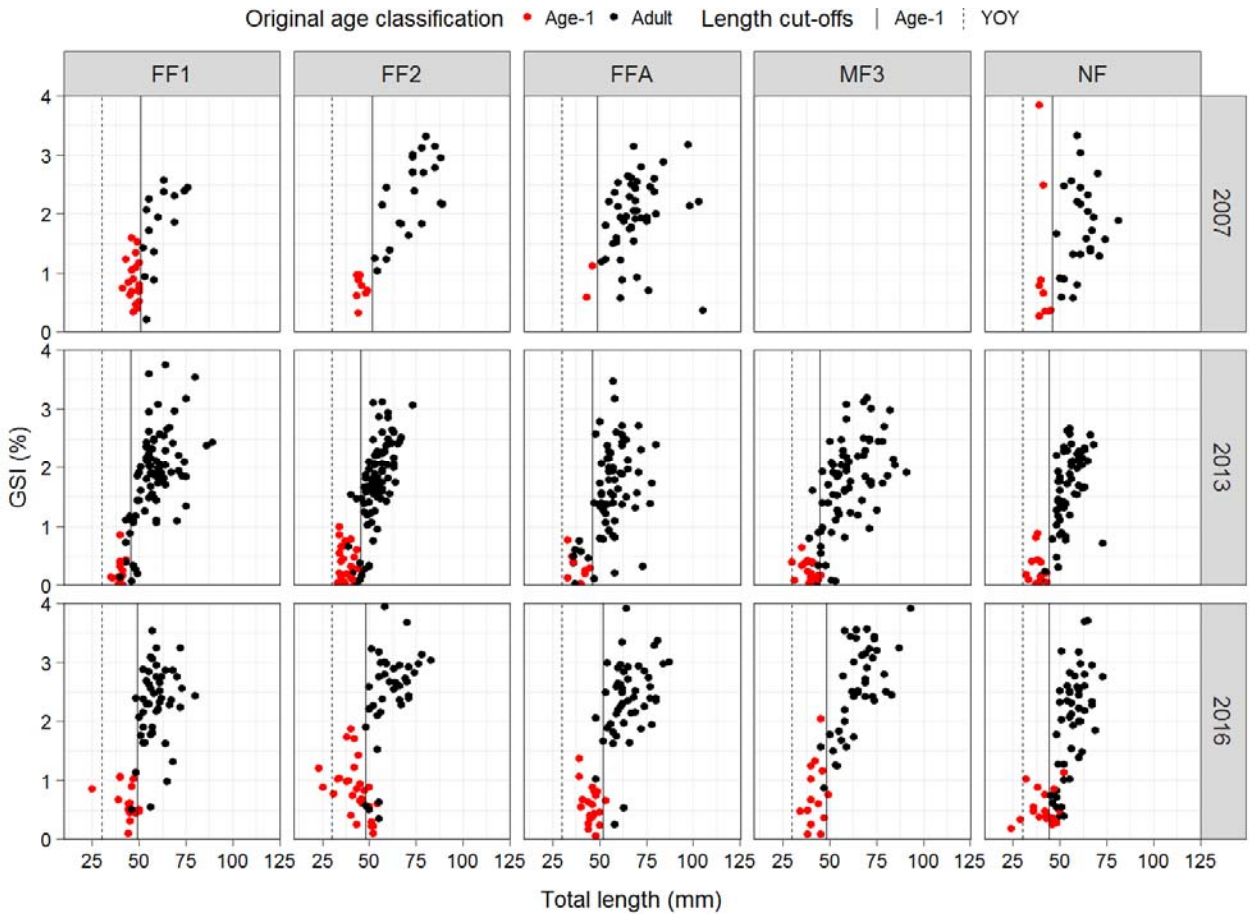
9.2.1.2.3 Stage Re-classification

Slimy Sculpin were grouped according to maturity (i.e., age-1+ or adult) and sex, and then analyzed as separate groups. Maturity and sex determination were based on field observations and confirmed by lab histology data, where available. Fish infected with tapeworms were removed from analyses, except for the analysis of incidence of parasitism.

Stage classification was estimated for the fall-sampled fish (2007, 2013, and 2016 collections) using the approach taken in the 2016 AEMP (Golder 2017c). Length-frequency histograms were used to differentiate young-of-the-year (YOY) sculpin from older fish. Fish less than 30 mm total length and without a GSI value were considered to be YOY and were removed from analysis. Previous studies suggest the general use of 1% GSI as a cut-off value between immature and mature fish (Environment Canada 2010). However, as suggested by the data distribution (Figure 9-1), a GSI cut-off of 1% would include many fish that are likely age-1+; therefore, a GSI value of 1.2% was used. In addition, maturity curves, constructed to describe fish maturity (age-1+/adult) as a function of total length, were used to calculate the total length at which 50% of the Slimy Sculpin were expected to be mature, by sampling area and year. This size at maturity value was specific to each sampling area and year and was applied to scatter plots of fish GSI (see section 9.2.1.2.2) versus total length. Using these metrics, fish smaller than the size at 50% maturity, specific to each area and year, with a GSI value less than 1.2% were considered to be age-1+ fish. Slimy Sculpin that were larger than the area-specific size at maturity, or had GSI greater than 1.2%, were assigned to the age-2+ group. Apart from YOY, fish with no known GSI were not assigned an age. The separation of YOY, age-1+, and adult fish was important because the different energetic requirements associated with reproduction result in differences in the rate of growth and body weight gain. In the 2016 AEMP, maturity codes were used to further filter results – only fish with maturity codes of “unknown”, “immature”, or “early development” were included in the age-1+ group. In addition, fish whose gonads were classified as “immature”, “resting”, “spent”, “reabsorbing”, and “unknown” were removed from the age-2+ group. However, the 2007 dataset did not include detailed maturity codes, and this approach could, therefore, not be applied to the entire dataset. The 2010 data were not reclassified and are not presented in these analyses, since the GSI classification approach cannot be applied to fish that recently spawned (i.e., fish captured in the late spring).

The inclusion of GSI when assigning stage to fish from the uninfected dataset increased the confidence in assigned age and the corresponding analyses. However, fish that were assessed non-lethally could not be used in the analysis, since their gonad weight was unknown; this reduced the sample size of fish available for statistical analysis.

Figure 9-1 Scatter Plot of Slimy Sculpin GSI versus Total Length



The reclassification of fish ages resulted in stage assignment change of 10 fish from the 2007 dataset, 31 fish from the 2013 dataset, and 9 fish from the 2016 dataset (Table 9-2). In addition, the stage classification of 193 fish, mostly age-1+ fish from 2007 and 2013 datasets, was removed (denoted as "None" in Table 9-2) due to missing GSI information.

Table 9-2 Reclassification of Slimy Sculpin Stages

Year	Original Stage	New Stage Assigned			
		YOY	Age-1+	Adult	None
2007	YOY	0	0	0	0
	Age-1+	2	31	5	99
	Adult	0	0	99	3
2013	YOY	0	0	0	0
	Age-1+	4	74	1	91
	Adult	0	26	307	0

Table 9-2 Reclassification of Slimy Sculpin Stages

Year	Original Stage	New Stage Assigned			
		YOY	Age-1+	Adult	None
2016	YOY	18	6	0	0
	Age-1+	0	71	0	0
	Adult	0	3	252	0

Note: The stage re-classification method was based on having gonad size information. Fish that did not have GSI information (e.g., non-lethal assessment), were previously classified based on weight, but could not be classified using this current revised approach given there was no gonad size to incorporate. These fish were removed from the data interpretation and are presented here in the "none" column.

9.2.1.3 Data Analysis

9.2.1.3.1 Habitat

Seasonal water temperature data at each sampling area were available from 2013 and 2016. Two temperature loggers were deployed in the spring under ice and retrieved in the fall, so that the sampling period encompassed the temperature range of the principal period of growth for fish in the area. Mean daily water temperature values were calculated for each individual logger and then a daily average within each sampling area was calculated across the two loggers. Water temperature trends were assessed visually for 2013 and 2016. Differences in temperatures between the NF, MF and FF areas were assessed by plotting the difference in mean daily temperature between NF, FF2, and MF3 and the pooled FF1 and FFA mean daily temperature.

Available substrate data from 2007, 2013, and 2016 were used to tabulate the dominant substrate type in each sampling area for each year. This was used to examine high-level differences in substrate between areas.

9.2.1.3.2 Catch-Per-Unit-Effort

The spring fish collection period in 2010 was considered acceptable for comparison of catch-per-unit-effort (CPUE) to the other years' fall collections (i.e., 2007, 2013 and 2016); therefore, 2010 data were included in the CPUE analysis. The number of Slimy Sculpin captured at each area for each year was standardized as CPUE, defined as the number of fish caught per 100 seconds of electrofishing effort. CPUE provides an estimate of relative abundance among sampling areas for a given year by standardizing the catch data according to the fishing effort.

9.2.1.3.3 Parasitism

The spring fish collection period in 2010 was considered acceptable for comparison of parasite presence to the other years' fall collections (i.e., 2007, 2013 and 2016); therefore, 2010 data were included in the parasite analysis. Parasitism was, however, reported diferently for 2007 and 2010 relative to 2013 and 2016. In 2007 and 2010, both external assessments of adult tapeworm presence and internal confirmation of tapeworm presence were recorded, only the internal confirmation results were reported in the annual report (Golder 2008, Golder 2011c). For the current assessment, the 2007 to 2016 data were compiled, and external tapeworm assessment data were summarized as proportion of total fish with and without suspected tapeworm infection. The data were then analyzed using a logistic model, where the response variable was expressed as a binomial variable with total counts of presence and absence of parasitism, and the explanatory variables included area, year (as factor), and their interaction. The model was used to estimate mean probability of tapeworm infection and the associated 95% confidence interval for each area and year, and to perform multiple comparisons between years within area. *P*-values were corrected using the Tukey method (Tukey1977). All analyses were performed in R v. 3.4.2 (R 2017).

9.2.1.3.4 Comparison to Normal Ranges

Differences in fish health variables were evaluated by comparing results in the NF, FF2 and MF areas to background values. Background values for Lac de Gras are those that fall within the range of natural variability, referred to as the normal range. The late fall normal ranges for fish health were revised as part of this re-evaluation report to account for the stage re-classification (Section 9.2.1.2.3). Normal ranges were calculated using data from two AEMP FF areas (FF1 and FFA), incorporating data from 2007 and 2013. In both 2007 and 2013, sampling was performed in late summer which corresponds with the sampling season for the 2016 data collections. Normal range calculations are detailed in Appendix A, and are summarized in Table 9-3.

Table 9-3 Late Fall Normal Ranges for Fish Health Indicators

Variable	Unit	Age-1+		Adult male		Adult female	
		Lower limit	Upper limit	Lower limit	Upper limit	Lower limit	Upper limit
Total length	mm	33.2	50.0	46.4	76.6	48.4	82.4
Weight	g	0.3	1.0	0.7	3.4	0.8	4.2
Carcass weight	g	0.2	0.8	0.6	2.9	0.7	3.6
Condition factor	-	0.6	0.9	0.5	0.8	0.6	0.8
LSI	%	0.9	3.6	1.2	3.7	1.9	5.7
GSI	%	-	-	0.2	2.7	1.0	3.2

Note: Fish health normal ranges are described in Appendix 9A.

9.2.1.3.5 Temporal Trends

Time series plots were generated for each variable by life stage/sex using data from 2007, 2013 and 2016 for the NF, FF2, and MF3 areas, and the FF areas (FF1 and FFA) of Lac de Gras. No 2007 data were available for the MF3 area. The 2010 dataset, collected in the spring, was omitted from the overall temporal trend figures, as described in Section 9.2.1.2.

The temporal trend analysis used for fish health data differed from the general trend analysis methods described in Section 2.4.2. Since only three years of data were available for analysis (2007, 2013, and 2016), and the general methods used year as a continuous covariate to examine temporal trends, this method could not be applied. In addition, fish health data do not have stations nested within NF/FF areas as do the other AEMP components (represented in the general methods as a random factor). Therefore, the trend model described in Section 2.4.2 was simplified into a fixed effect model, with year and area as discrete covariates for analysis of total length, fresh weight, and carcass weight. The full model contained effects of year, area, and their interaction. No model selection was performed, since there was only one candidate model for analysis. Data transformation using Yeo-Johnson transformations were performed as described in Section 2.4.2.

For the analysis of condition, relative liver weight and relative gonad weight, ANCOVA models were constructed, with weight (total weight for age-1+ fish, and carcass weight for adult fish), liver weight, and gonad weight as the response variables. The covariates for the three analyses were log-transformed: total length for the analysis of condition factor, and raw carcass weight for the analysis of relative gonad weight and relative liver weight. The full model contained effects of year, area, the covariate, and all possible interactions. For the analysis of condition, the response variable (weight) was log-transformed to linearize the relationship between the response variable and the covariate. Residual heteroscedasticity was examined and heteroscedasticity effects by area, year, or predicted value were added to the model, if they improved model fit (assessed using AIC scores). Analysis outputs included tables of parameter significance, Tukey-adjusted multiple comparisons (Tukey 1977) between years within area, and plots of fitted estimates and 95% confidence intervals, overlaying the fish health data.

9.2.1.3.6 Action Level Evalulation

The importance of effects to a fish health endpoint has been categorized according to Action Levels. The Action Level classifications were developed to meet the goals of the draft *Guidelines for Adaptive Management – A Response Framework for Aquatic Effects Monitoring* (WLWB 2010; Racher et al. 2011), and are listed in the *AEMP Study Design Version 3.5* (Golder 2014a). The goal of the AEMP Response Framework is to make sure that significant adverse effects never occur. A significant adverse effect, as it pertains to aquatic biota, was defined in the EA as a change in fish population(s) that is greater than 20% (Government of Canada 1999). This effect must have a high probability of being permanent or long-term in nature and must occur throughout Lac de Gras. The Significance Thresholds for all aquatic biota are, therefore, related to impacts that could result in a change in fish population(s) that is greater than 20%.

Although the AEMP addresses two broad impact hypotheses for Lac de Gras, the toxicological impairment hypothesis and the nutrient enrichment hypothesis, the Action Levels for fish health address only the toxicological impairment hypothesis.

Fish health responses are assessed every three years to evaluate effects as described in the *AEMP Study Design Version 3.5* (Golder 2014a). This involves measuring responses in the NF area against those in the two FF areas (i.e., FF1 and FFA). The occurrence of an Action Level 1 is determined by finding significant differences in fish health responses in the NF area compared to those in the FF1 and FFA areas that are indicative of a toxicological effect. The approved Action Levels from *AEMP Study Design Version 3.5* (Golder 2014a) are provided in Table 9-4.

Table 9-4 Action Levels for Fish Health

Action Level	Fish Health	Extent	Action
1	Statistical difference from reference[(a)] indicative of a toxicological response[(b)]	NF	Confirm effect
2	Statistically significant difference from reference[(a)] indicative of a toxicological response[(b)]	MF area	Investigate cause
3	A measurement endpoint beyond the normal range[(c)]	NF	Examine ecological significance Set Action Level 4 Identify mitigation options
4	To be determined[(d)]	To be determined[(d)]	Define conditions required for the Significance Threshold
5[(e)]	Indications of severely impaired reproduction or unhealthy fish likely to cause a >20% change in fish population(s)	FFA	Significance Threshold

a) Action Levels were assessed by comparing NF and MF areas to the FF1 and FFA areas.

b) Such a response could include a decrease in recruitment (fewer young fish), smaller gonads, reduced fecundity, changes to liver size, changes in condition, increased incidence of pathology, reduced growth, and reduced survival.

c) Normal range as defined and approved in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b).

d) To be determined if Action Level 3 is reached.

e) Although the Significance Threshold is not an Action Level, it is shown as the highest Action Level to demonstrate escalation of effects towards the Significance Threshold.

>= greater than; NF = near-field; MF = mid-field; FF = far-field.

9.2.1.3.7 Magnitude of Difference (Critical Effect Size)

As per the MMER TGD (Environment Canada 2012), a Critical Effect Size (CES) is defined as “a threshold above which an effect may be indicative of a higher risk to the environment” (Environment Canada 2012). CES are defined for fish weight, relative liver size and relative gonad size as 25% of the reference area mean, and for condition as 10% of the reference area mean (Environment Canada 2012). The variables that triggered Action Level exceedances in 2016 were compared to the CES.

9.2.1.3.8 Weight-of-Evidence Effect Ratings

The results of the AEMP fish health surveys are integrated through the WOE evaluation process, which determines the strength of evidence supporting the two broad impact hypotheses for Lac de Gras (i.e., toxicological impairment and nutrient enrichment), as described in the *AEMP Study Design Version 3.5* (Golder 2014a). The WOE is not intended to determine the ecological significance or level of concern associated with any given changes. The WOE effect ratings incorporate statistical comparisons of the NF and FF areas, and comparisons of the NF area to the normal range as defined in Appendix A and Table 9-3. The fish health data were assessed according to the WOE effect ratings described in Section 10 and Table 9-5. Normal range comparisons were performed using the revised normal ranges presented in Appendix A and Table 9-3

Table 9-5 Weight-of-Evidence Effect Ratings for Fish Health

Endpoint	No Response 0	Early Warning/Low ↑/↓	Moderate ↑↑/↓↓	High ↑↑↑/↓↓↓
Comparison to FF Areas and Normal Ranges[a] Growth – Body Size Energy Stores - K Energy Stores – LSI Relative Reproductive Success – Age-1+ Abundance Relative Reproductive Success – GSI Taperworm Parasitism - Occurrence	No difference	Statistically-significant change, NF vs FF areas	Low + NF area mean outside normal range	Moderate rating extending beyond NF

Notes: Normal ranges for each LOE group and measurement endpoint are defined in Appendix A and Table 9-4.
a) Applied separately for each measurement endpoint.
K = condition factor; LSI = liver-somatic index; GSI – gonadosomatic index; NF = near-field; FF = far-field.

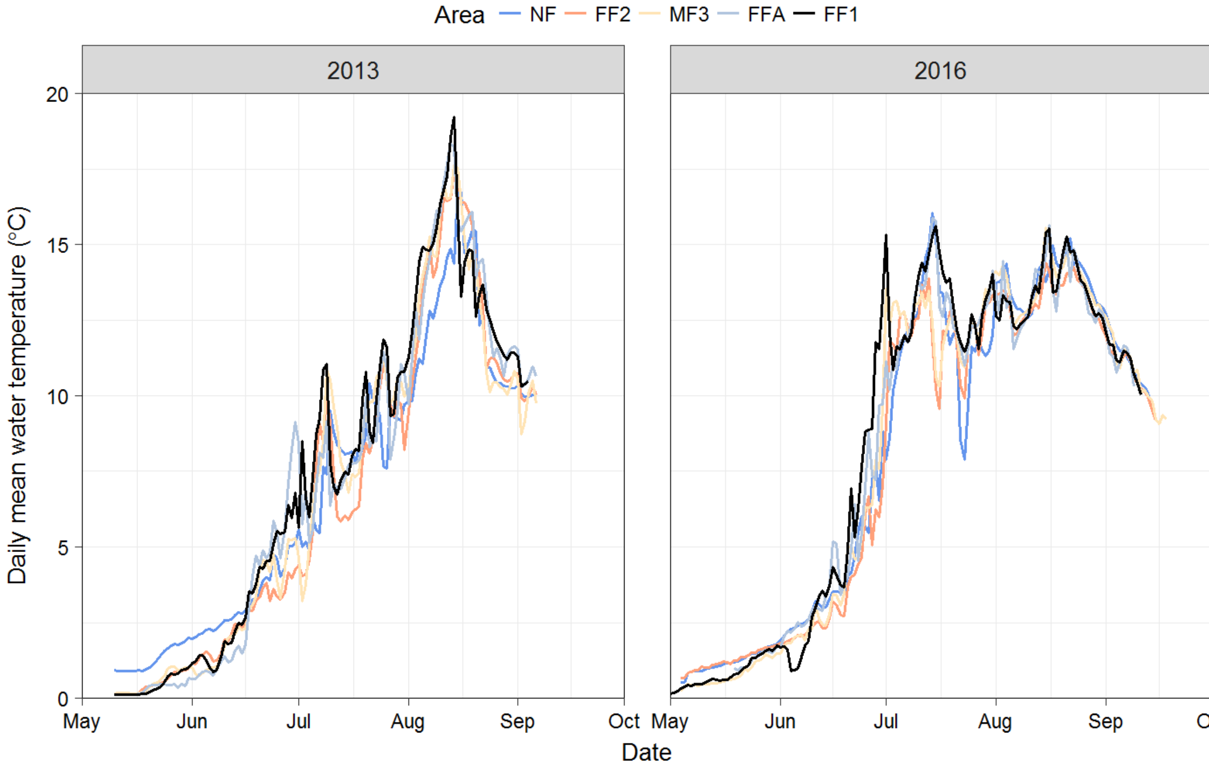
9.2.2 Results

9.2.2.1 Habitat

Measured daily water temperatures varied among the five sampling areas throughout the 2013 and 2016 sampling periods (Figure 9-2). While temperatures were similar throughout May and mid August-September, there were larger temperature fluctuations in June and July in both years, as well as in early August in 2013. In both years, water temperature was generally higher at FF1 than at the other sampling areas. For example, in late June and early July of 2016, water temperatures at FF1 and MF3 were up to 4ºC warmer than at NF and FFA. In 2013, throughout early August, NF temperatures were 2 to 4ºC less those recorded at FF1.

Note that temperature logger deployment depths differed among areas in 2016, ranging between 0.05 m (at FF1) and 2.3 m (at NF). The differences in depth of logger deployment may have accounted for some of the among-area variability in temperature measurements, as deeper loggers are more likely to record lower water temperatures. However, water temperatures recorded by loggers at the deeper deployment depths were similar to temperatures recorded by hand-held YSI units near the surface during the 2016 AEMP fish sampling (Golder 2017c). In addition, no consistent bias was observed in the temperature records (i.e., areas of deeper logger deployment did not consistently have colder water temperatures.

Figure 9-2 Mean Daily Water Temperature in Lac de Gras, 2013 and 2016



The comparison of 2007, 2013, and 2016 substrate data indicated little difference in the dominant substrate type among areas (Table 9-6). The main difference between areas was that cobbles were recorded as dominant or semi-dominant substrate in all areas except for NF in either 2013 or 2016, whereas NF substrate was recorded as boulders in all three years. Substrate data available prior to 2016 were limited in the extent of detail, therefore, it was not possible to conduct a more in-depth analysis of substrate composition and how it relates to sculpin habitat requirements.

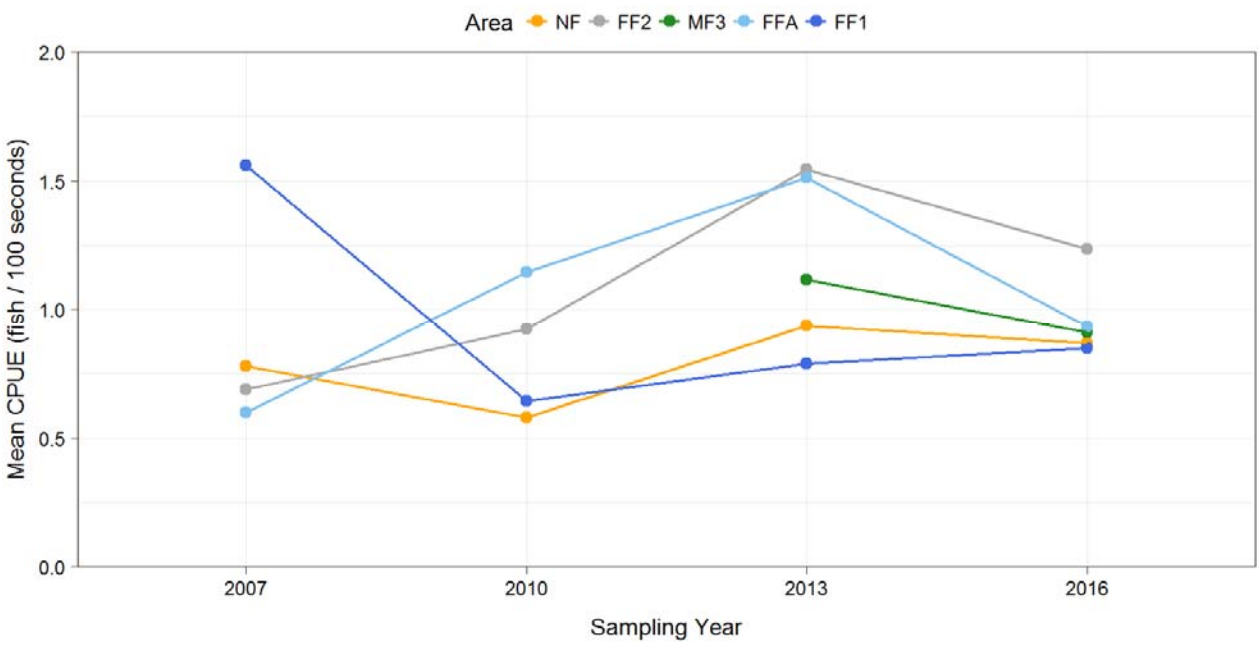
Table 9-6 Dominant Substrate Type at Slimy Sculpin Sampling Areas in Lac de Gras, 2007 to 2016

Year	NF	FF2	MF3	FF1	FFA
2007	Boulder	Boulder	-	Boulder	Boulder
2013	Boulder	Cobble	Cobble	Cobble	Boulder/Cobble
2016	Boulder	Boulder	Boulder/Cobble	Cobble	Boulder/Cobble

9.2.2.2 CPUE

The temporal trends of mean CPUE values differed between areas (Figure 9-3). At FF1, CPUE was highest in 2007, decreased in 2010, and gradually increased throughout 2013 and 2016. At FFA, CPUE increased between 2007, 2010, and 2013, but decreased in 2016. At NF, the 2007 CPUE value was lower than that in FF1, but CPUE values throughout 2010, 2013, and 2016 were similar to FF1. Mean CPUE values at MF3 and FF2 were highest of the five sampling areas in 2010, 2013, and 2016. Based on the temporal evaluation of CPUE, there does not appear to be a change in CPUE that can be attributed to the Mine effluent.

Figure 9-3 Mean CPUE Values Recorded at Each Area and Sampling Year

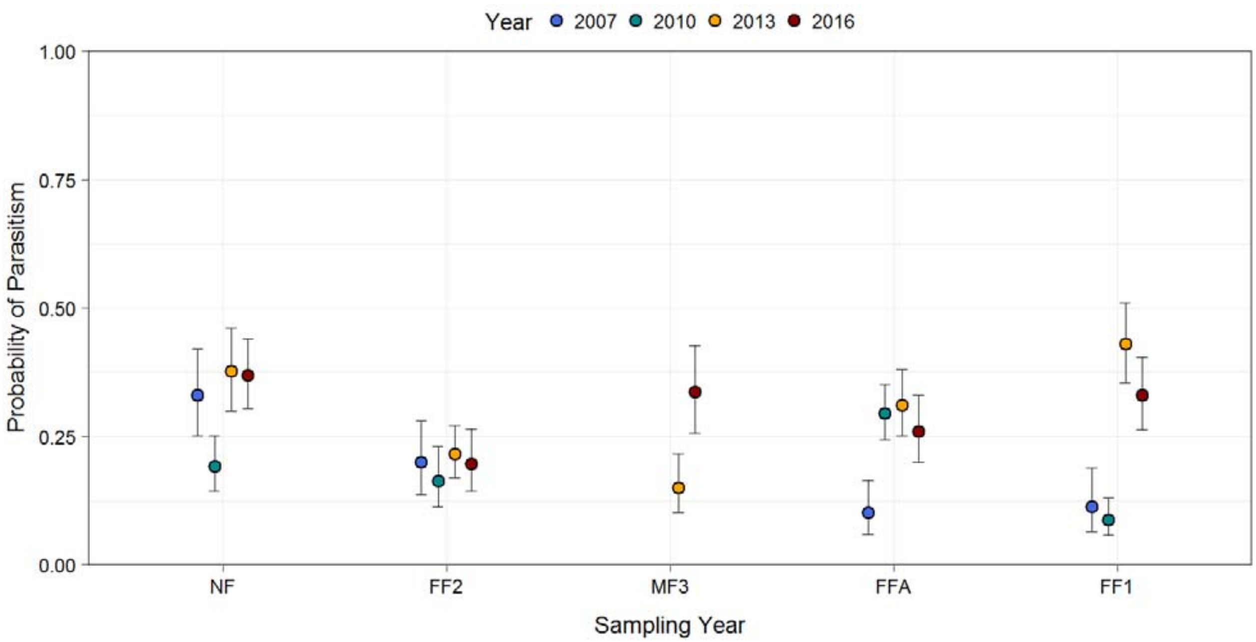


9.2.2.3 Parasitism

The logistic regression of parasitism occurrence in Slimy Sculpin had a significant Area × Year interaction (*P* <0.001), indicating differences in temporal trends among areas. The incidence of recorded adult tapeworm presence, based on external examination, decreased significantly at the NF between 2007 and 2010, then increased in 2013, and remained stable in 2016 (Figure 9-4; Table 9-7). At FF2, parasitism remained relatively stable throughout 2007 to 2016, whereas parasitism at MF3 increased significantly from 2013 to 2016. Temporal trends of parasite presence at FF1 and FFA differed greatly among years; while FF1 was similar between 2007 and 2010, parasite presence increased in 2013, then decreased in 2016 (Figure 9-4), while at FFA, parasite presence increased between 2007 and 2013, then decreased slightly in 2016. All sites except FF2 had increased parasite presence in 2013 and 2016 relative to 2007. Overall, parasite assessment did not support a pattern of spatial and temporal gradient of response consistent with a Mine-related effect.

There is uncertainty in the interpretation of the parasitism occurance findings, because it is impossible to tease apart whether observed differences in parasitism are linked to the Mine, or are caused by other factors (e.g., bird populations, host-parasite interactions, water temperature fluctuations). The intent of documenting tapeworm presence/absence in the field is to aid in interpretation of fish health results (i.e., could differences in internal organs be caused by parasitism as opposed to Mine effluent), not to distinguish whether or not the Mine is having an effect on the rate of parasitism.

Figure 9-4 Estimated Probabilities of Parasitism in Slimy Sculpin



Note: Errorbars are 95% confidence intervals.

Table 9-7 Post-hoc Multiple Comparisons for Parasitism Occurrence

Area	2007	2010	2013	2016
NF	b	a	b	b
FF2	a	a	a	a
MF3	---	---	a	b
FFA	a	b	b	b
FF1	a	a	b	b

Note: Grey shading denotes significant decreasing trend

9.2.2.4 Comparison to Normal Ranges

Normal range comparsions were conducted on all data collected in the late summer (i.e., 2007, 2013 and 2016), using the updated normal range values presented in Table 9-3. Mean within-area values exceeded the upper limit of the normal range for age-1+ condition in 2007, due to presence of two age-1+ fish with high condition (Figure 9-8). No other variables had mean values outside of the normal range in any of the three sampling years (Figures 9-5 to 9-10).

Figure 9-5 Temporal Trend Plot of Total Length in Slimy Sculpin

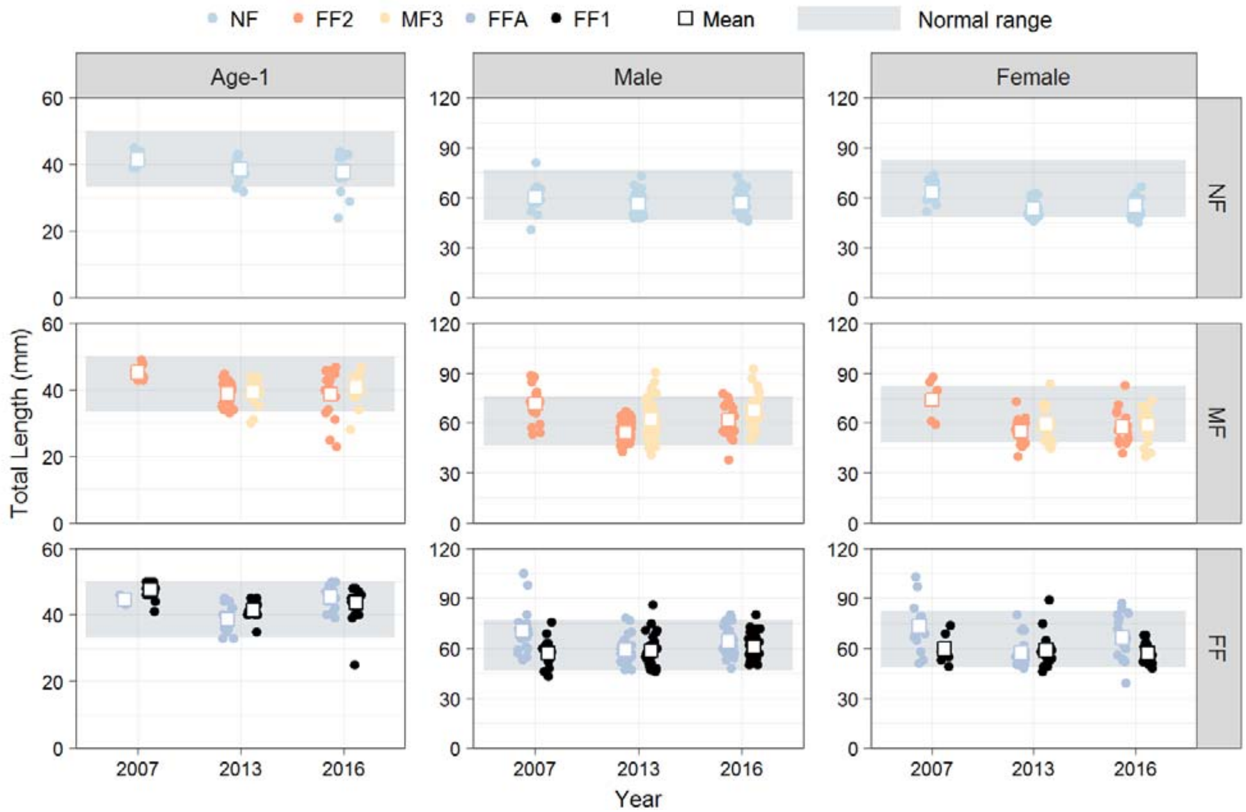


Figure 9-6 Temporal Trend Plot of Fresh Weight in Slimy Sculpin

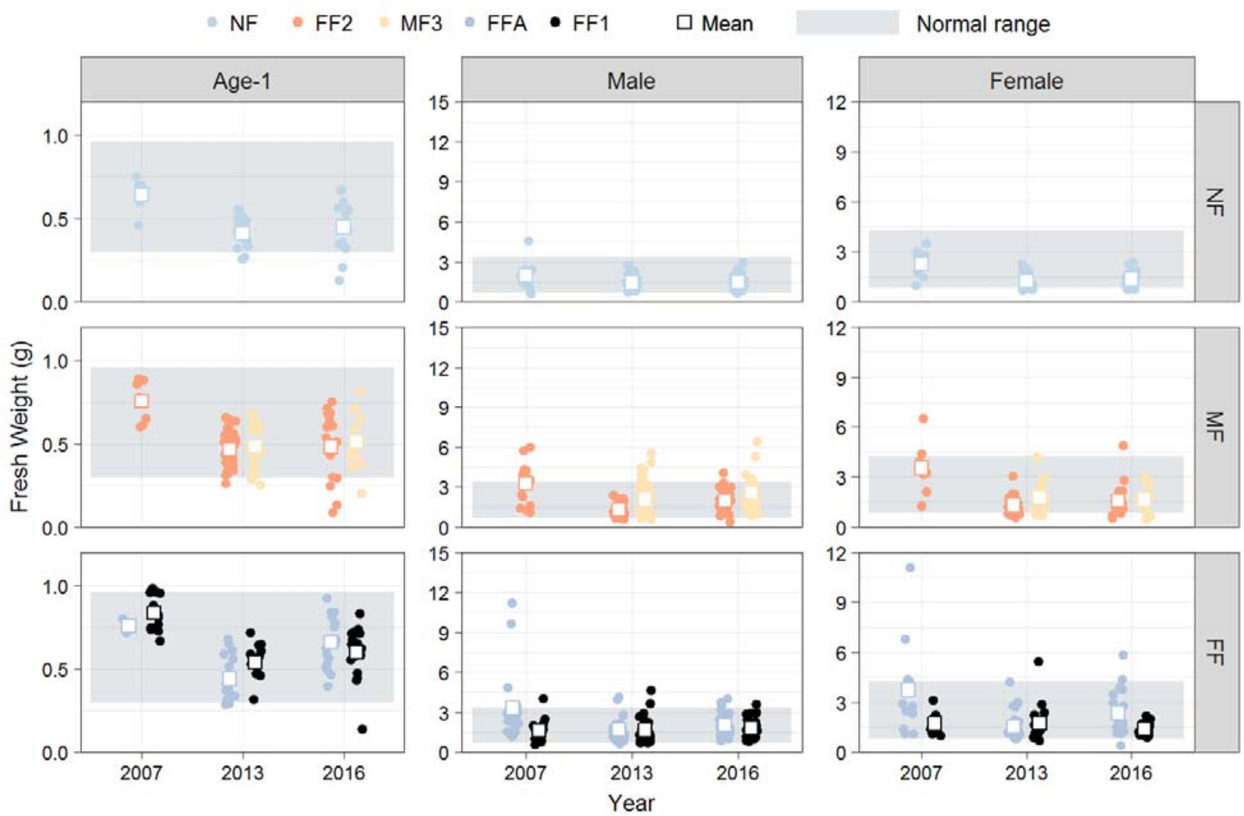


Figure 9-7 Temporal Trend Plot of Carcass Weight in Slimy Sculpin

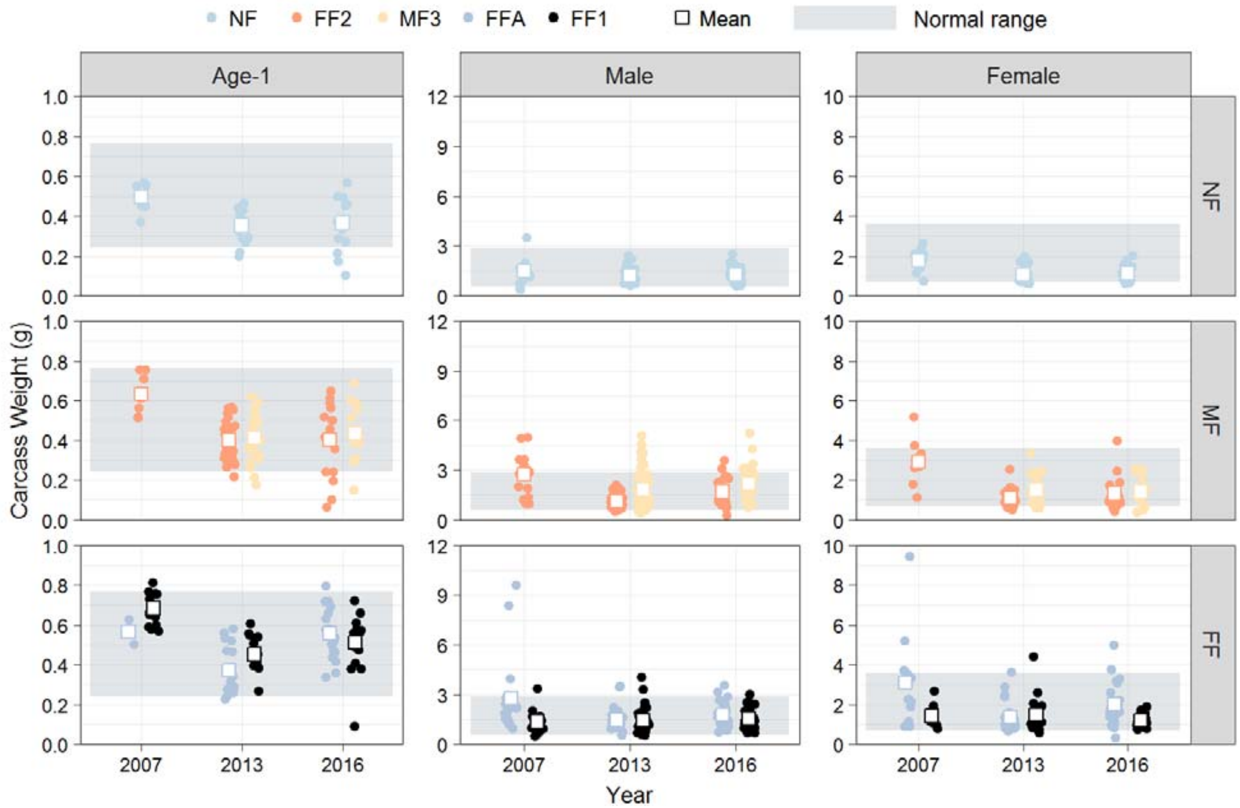


Figure 9-8 Temporal Trend Plot of Condition Factor in Slimy Sculpin

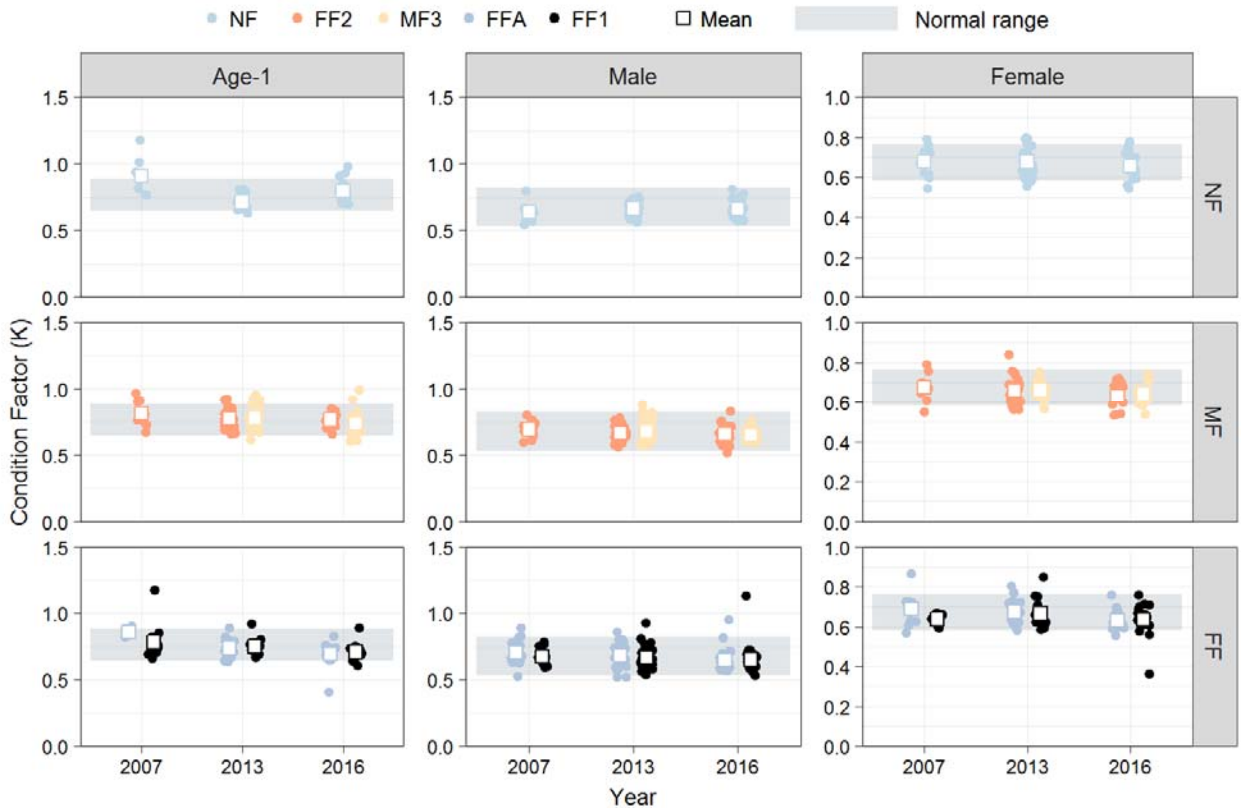


Figure 9-9 Temporal Trend Plot of LSI in Slimy Sculpin

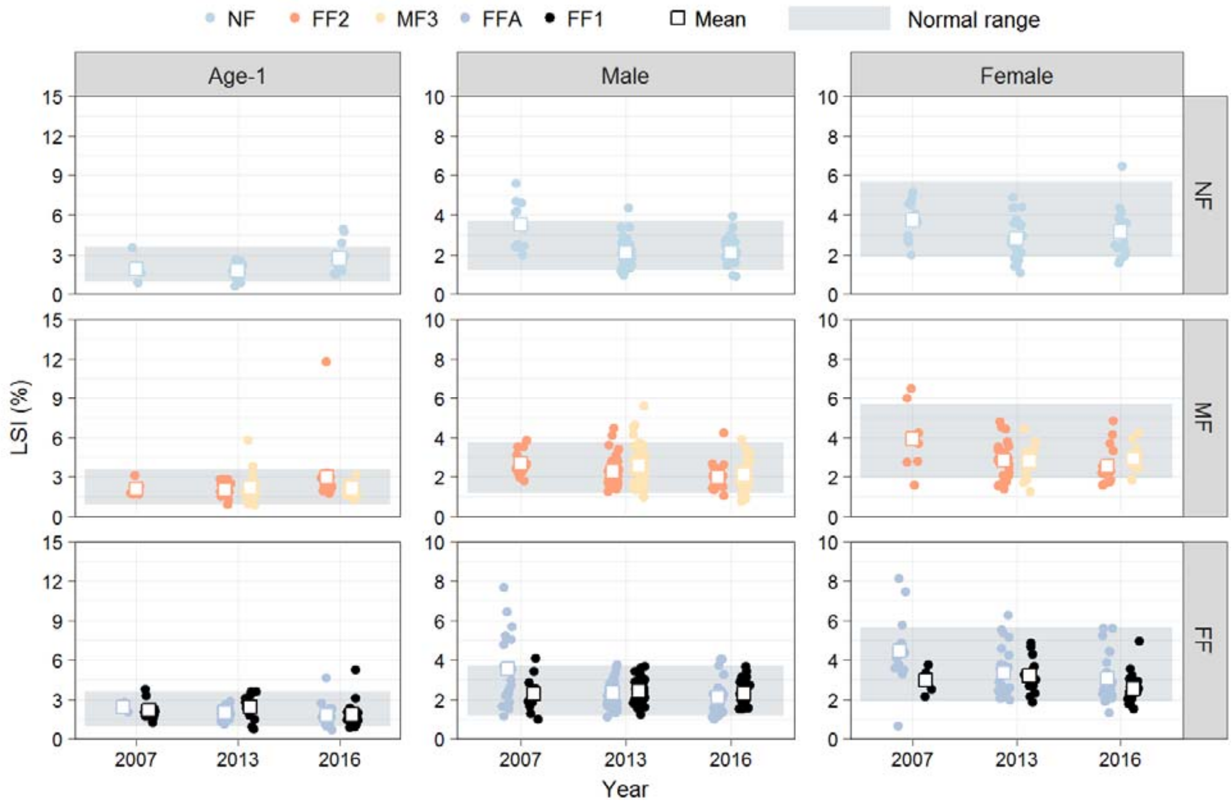
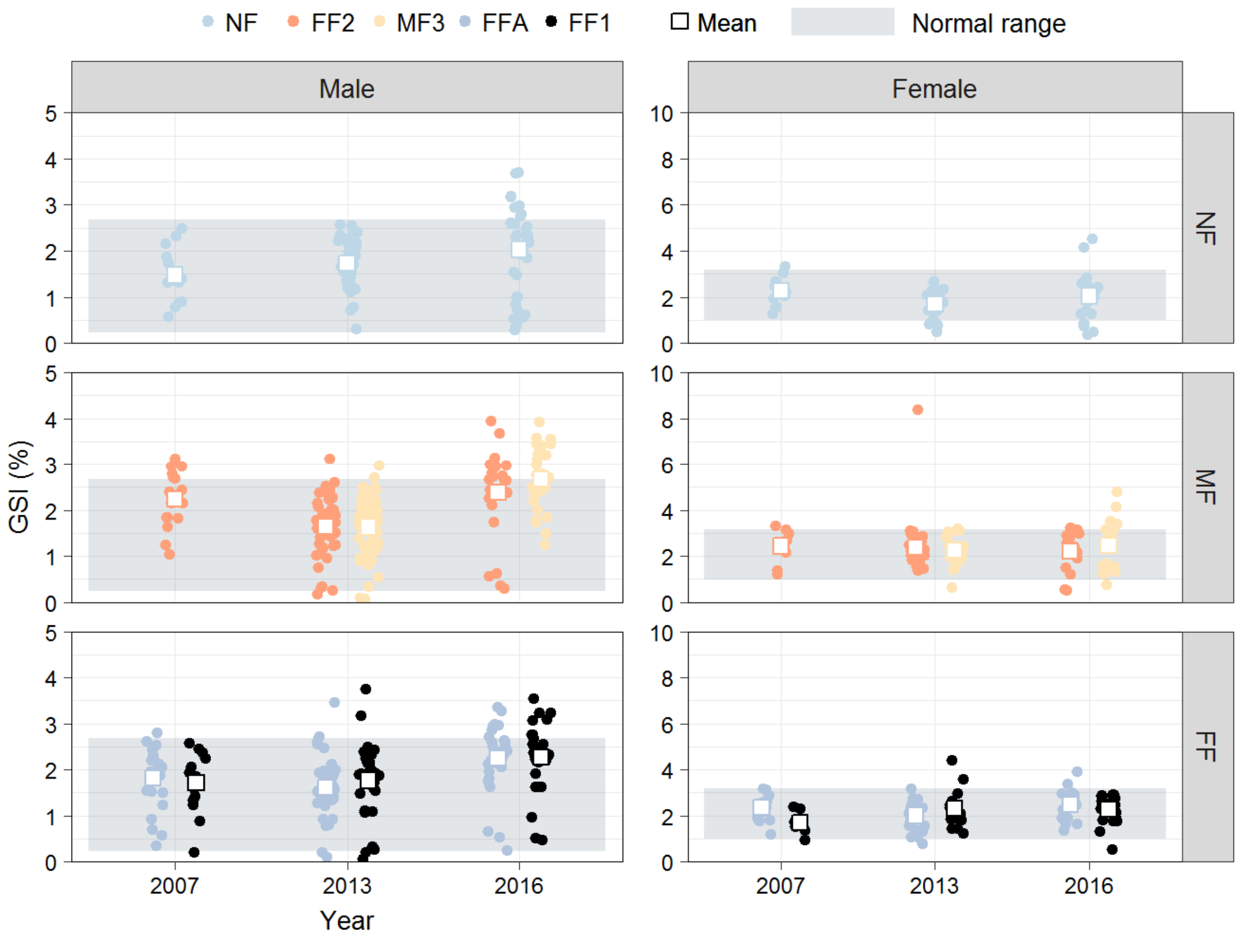


Figure 9-10 Temporal Trend Plot of GSI in Slimy Sculpin



9.2.2.5 Temporal Trends

Total length, fresh weight, and carcass weight had a significant interaction between year and area for age-1+, male, and female fish (Table 9-8). Some of the variables only had significant interactions or year effects due to temporal changes at the FF areas (Table 9-9); these variables include age-1+ total length, fresh weight, and carcass weight, and are not considered further. Significant, increasing trends were estimated for male fish at FF2 between 2007 and 2013 for total length, fresh weight, and carcass weight (Table 9-9), and at MF3 between 2013 and 2016 for total length, as well as at FFA between 2013 and 2016 for total length and fresh weight. Significant, increasing trends were estimated for female fish at FFA between 2013 and 2016 for total length, fresh weight, and carcass weight. Significant, decreasing trends were estimated for both male and female fish at FF2 and FFA between 2007 and 2013 for total length, fresh weight, and carcass weight. Mean total length, fresh weight, and carcass weight estimates were within the normal range limits for age-1+, male, and female fish from all areas and years.

The analyses of condition, LSI, and GSI as ANCOVAs of weight, liver weight, and gonad weight, respectively, all had at least one significant interaction between either year of area effect and the covariate, with the exception of LSI analysis in age-1+ fish, which only had an interaction between area and year (Table 9-10). The analysis of condition only had a single significant multiple comparison for small fish for all stages/sexes – an increasing trend between 2013 and 2016 at FF2 (Table 9-9). For large fish, there were no significant trends in age-1+ fish, while in both male and female fish, a significant decrease was estimated at FF2 between 2007 and 2013, and a significant decrease was estimated at FFA between 2007 and 2016 for male fish.

The results of analysis of relative liver weight differed by stage/sex. In age-1+ fish, a significant increase in liver weight was estimated at NF between 2013 and 2016 for small fish, and a significant decrease was estimated at FF1 between the same years. For large age-1+ fish, no significant differences were estimated between years within areas. In small males, there were no significant differences in liver weight between years. In large males, on the other hand, significant decreases in liver weight were found between 2007 and 2013 in NF, between 2007 and 2016 at FF2, and between 2013 and 2016 at MF3. In small female fish, a significant increase in liver weight was estimated between 2007 and 2013 at FF2, while in large female fish, a significant decrease in liver weight was estimated between 2013 and 2016 at FF2 and between 2007 and 2013 at FFA.

The results of analysis of relative gonad weight differed by fish sex. In small males, there were no significant differences in gonad weights between years within areas. In large males, on the other hand, there were significant increases in gonad weight between 2007, 2013, and 2016 in NF, and between 2013 and 2016 in MF3 and FFA. In small females, a significant reduction in gonad weight was estimated between 2007 and 2013 at NF, and a significant increase in gonad weight was estimated between 2007 and 2013 at FF2. In large females, gonad weight increased significantly between 2007 and 2013 at NF and between 2013 and 2016 at MF3.

Figure 9-11 Temporal Trend Analysis of Total Length, Fresh Weight, and Carcass Weight in Age-1+ Slimy Sculpin

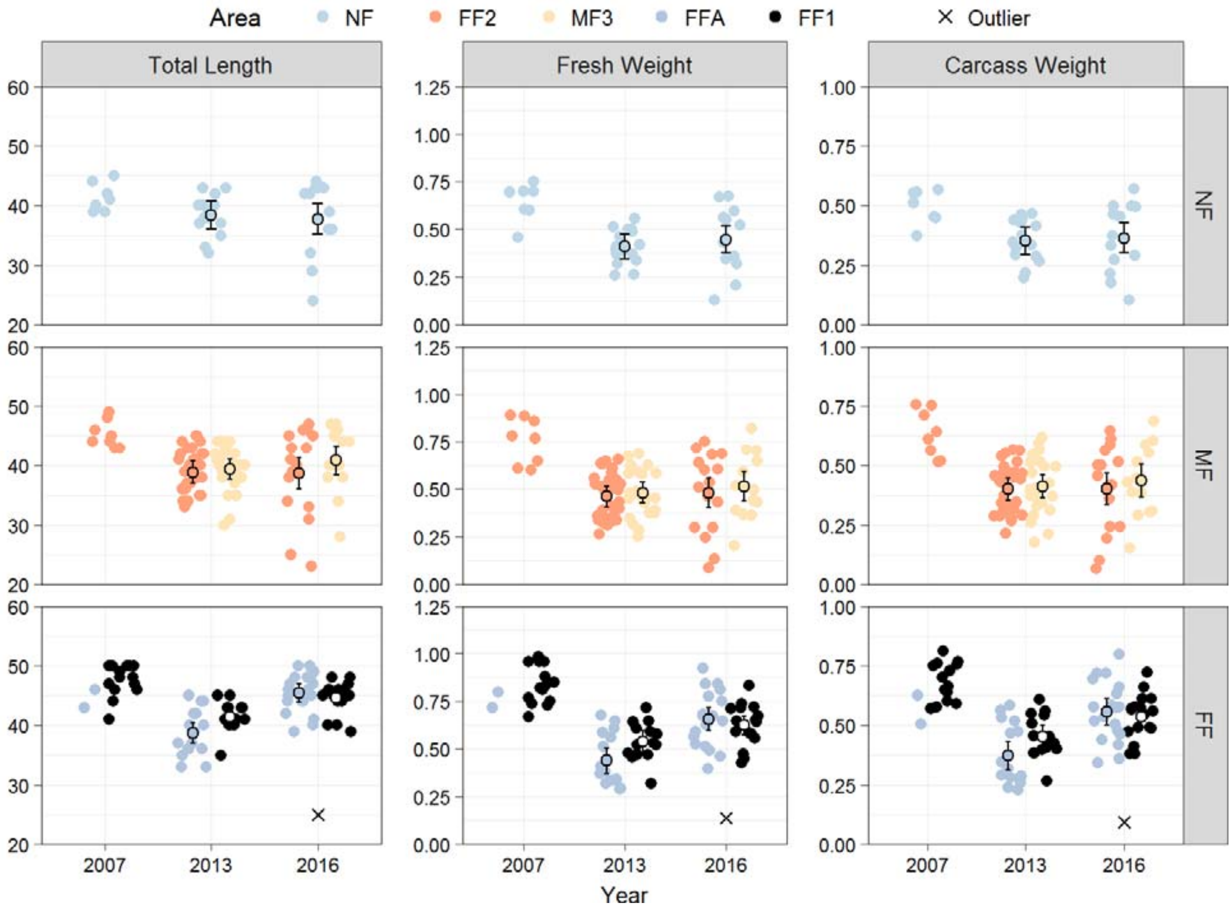


Figure 9-12 Temporal Trend Analysis of Inverse-Transformed Total Length in Adult Slimy Sculpin

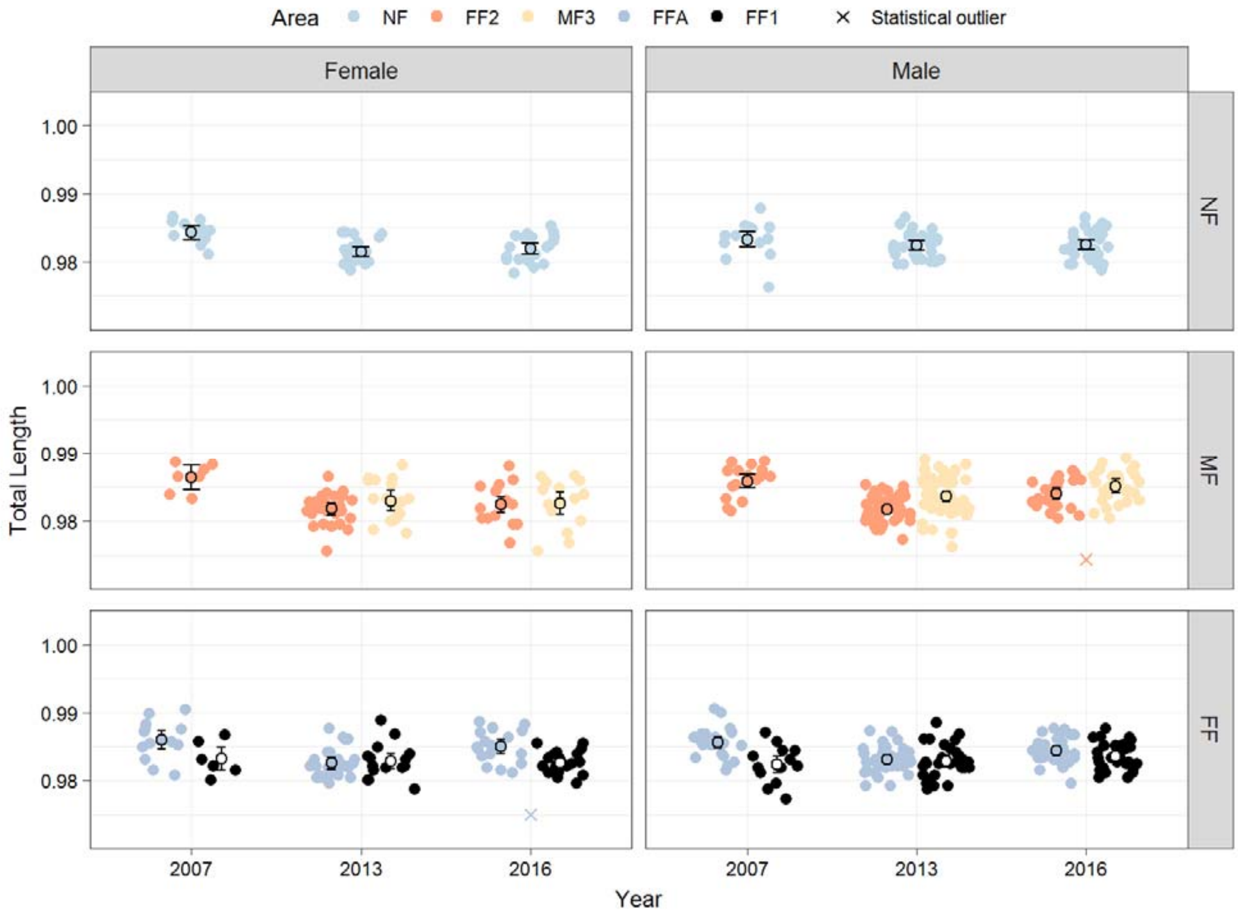


Figure 9-13 Temporal Trend Analysis of Inverse-Transformed Fresh Weight in Adult Slimy Sculpin

Figure 9-14 Temporal Trend Analysis of Inverse-Transformed Carcass Weight in Adult Slimy Sculpin

Table 9-8 Results of Trend Analysis of Total Length, Fresh Weight, and Carcass Weight in Slimy Sculpin

Sex/Stage	Variable	Yeo-Johnson transformation[(a)]	Coefficient	Df	F value	*P* value
Age-1+	Total Length	none	Area	4	8.9	<0.001
			Year	1	26.2	<0.001
			Area × Year	4	4.9	**0.001**
	Fresh Weight	none	Area	4	7.4	<0.001
			Year	1	15.9	<0.001
			Area × Year	4	3.1	**0.017**
	Carcass Weight	none	Area	4	7.3	<0.001
			Year	1	13.0	<0.001
			Area × Year	4	3.4	**0.011**

Table 9-8 Results of Trend Analysis of Total Length, Fresh Weight, and Carcass Weight in Slimy Sculpin

Sex/Stage	Variable	Yeo-Johnson transformation[a]	Coefficient	Df	F value	*P* value
Male	Total Length	-1	Area	4	9.7	<0.001
			Year	2	28.8	<0.001
			Area × Year	7	4.5	**<0.001**
	Fresh WeighT	-1	Area	4	8.9	<0.001
			Year	2	33.4	<0.001
			Area × Year	7	3.9	**<0.001**
	Carcass Weight	-1	Area	4	9.7	<0.001
			Year	2	25.4	<0.001
			Area × Year	7	4.1	**<0.001**
Female	Total Length	-1	Area	4	5.7	<0.001
			Year	2	23.2	<0.001
			Area × Year	7	2.4	**0.024**
	Fresh Weight	-1	Area	4	4.3	0.002
			Year	2	25.3	<0.001
			Area × Year	7	2.5	**0.019**
	Carcass Weight	-1	Area	4	4.7	0.001
			Year	2	21.0	<0.001
			Area × Year	7	2.4	**0.021**

a) See Section 2.4.2.1 for details on transformation approach.

Note: *P*-values of significant Area × Year interactions and significant main effects (where no significant interactions were found) are shown in bold.

Table 9-9 Post-hoc Multiple Comparisons for Fish Health, 2007 to 2016

Sex / Stage	Variable	Covariate	NF			FF2			MF3			FFA			FF1		
			2007	2013	2016	2007	2013	2016	2007	2013	2016	2007	2013	2016	2007	2013	2016
Age-1+	Total Length	---	---	a	a	---	a	a	---	a	a	---	a	b	---	a	b
	Fresh Weight	---	---	a	a	---	a	a	---	a	a	---	a	b	---	a	b
	Carcass Weight	---	---	a	a	---	a	a	---	a	a	---	a	b	---	a	b
	Condition	min	---	a	a	---	a	b	---	a	a	---	a	a	---	a	a
		max	---	a	a	---	a	a	---	a	a	---	a	a	---	a	a
	LSI	min	---	a	b	---	a	a	---	a	a	---	a	a	---	b	a
		max	---	a	a	---	a	a	---	a	a	---	a	a	---	a	a
Male	Total Length	---	a	a	a	c	a	b	---	a	b	b	a	b	a	a	a
	Fresh Weight	---	a	a	a	c	a	b	---	a	a	c	a	b	a	a	a
	Carcass Weight	---	a	a	a	c	a	b	---	a	a	b	a	a	a	a	a
	Condition	min	a	a	a	a	b	b	---	a	a	a	a	a	a	a	a
		max	a	a	a	b	a	a	---	a	a	b	ab	a	a	a	a
	GSI	min	a	a	a	a	a	a	---	a	a	a	a	a	a	a	a
		max	a	b	c	a	a	a	---	a	b	a	a	b	a	a	a
	LSI	min	a	a	a	a	a	a	---	a	a	a	a	a	a	a	a
		max	b	a	a	b	ab	a	---	b	a	a	a	a	a	a	a
Female	Total Length	---	b	a	a	b	a	a	---	a	a	b	a	b	a	a	a
	Fresh Weight	---	b	a	a	b	a	a	---	a	a	b	a	b	a	a	a
	Carcass Weight	---	b	a	a	b	a	a	---	a	a	b	a	b	a	a	a
	Condition	min	a	a	a	a	b	ab	---	a	a	a	a	a	a	a	a
		max	a	a	a	b	a	a	---	a	a	a	a	a	a	a	a
	GSI	min	b	a	a	a	b	b	---	a	a	a	a	a	a	a	a
		max	a	b	b	a	a	a	---	a	b	a	a	a	a	a	a
	LSI	min	a	a	a	a	b	b	---	a	a	a	a	a	a	a	a
		max	a	a	a	ab	b	a	---	a	a	b	a	a	a	a	a

Note: "-" means that data from that year/area were not included in the analysis of the variable. Different letters signify statistically significant differences (at the 0.05 level) of years within areas following post-hoc multiple comparisons. Grey shading denotes significant decreasing trend, underline denotes significant increasing trend. For variables analyzed using ANCOVAs (condition, LSI, and GSI), multiple comparisons were performed at two levels of the covariate – minimum and maximum values that were not statistical outliers.

Table 9-10 Results of Trend Analysis of Condition, LSI, and GSI in Slimy Sculpin

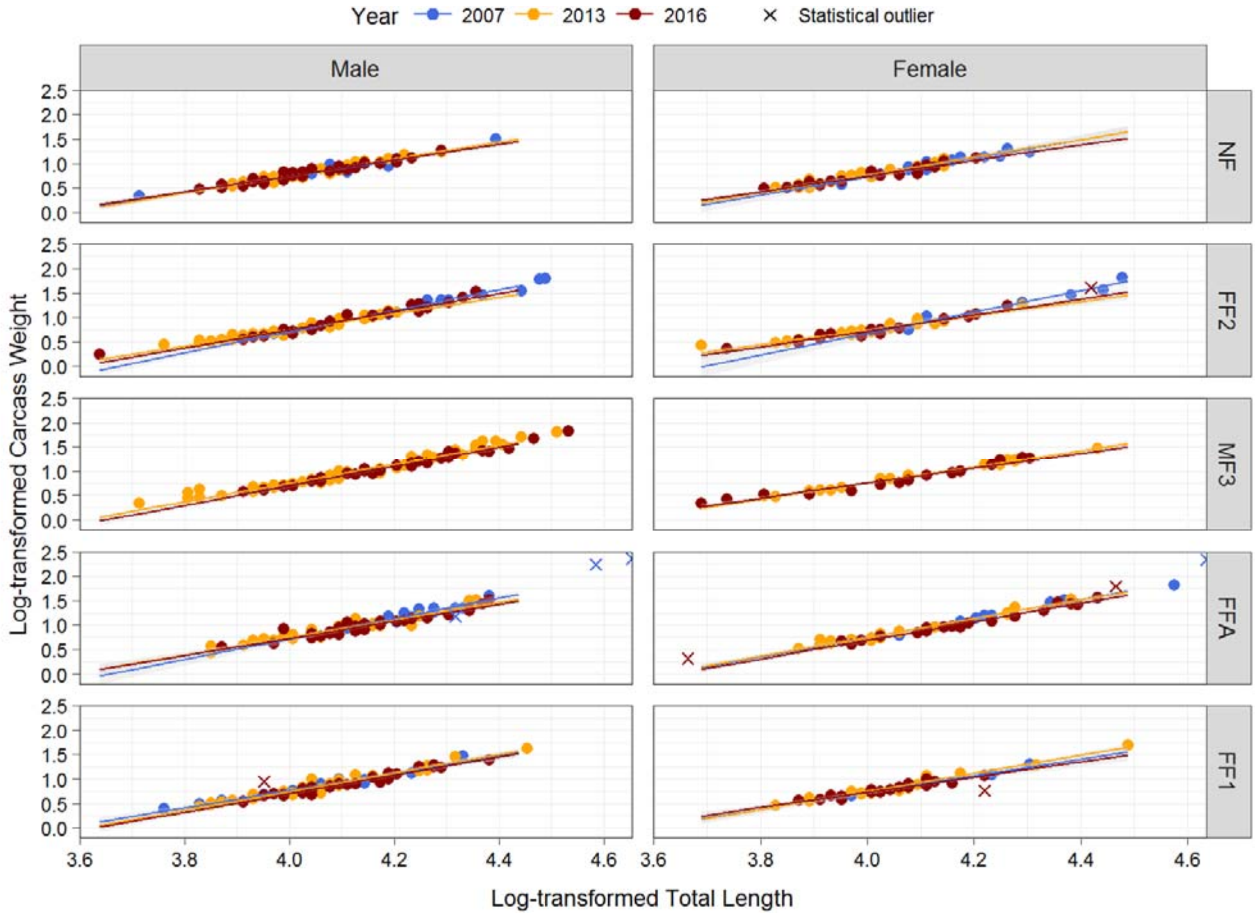
Sex / Stage	Variable	Transformation	Coefficient	Df	F value	*P*-value
Age-1+	Condition	Natural log-transformed both response (fresh weight) and covariate (total length)	Area	4	1.8	0.130
			Year	1	4.6	0.034
			Covariate (Total Length)	1	1481.2	<0.001
			Area × Year	4	3.1	0.018
			Area × Total Length	4	5.6	**<0.001**
			Year × Total Length	1	1.5	0.218
			Area × Year × Total Length	4	1.2	0.328
	LSI	None for both response variable (liver weight and covariate (carcass weight)	Area	4	0.5	0.755
			Year	1	1.0	0.328
			Covariate (Carcass Weight)	1	190.2	<0.001
			Area × Year	4	6.6	**<0.001**
			Area × Carcass Weight	4	1.1	0.371
			Year × Carcass Weight	1	0.7	0.400
			Area × Year × Carcass Weight	4	1.3	0.273
Male	Condition	Natural log-transformed both response (fresh weight) and covariate (total length)	Area	4	2.3	0.056
			Year	2	8.2	<0.001
			Covariate (Total Length)	1	7538.6	<0.001
			Area × Year	7	1.9	0.065
			Area × Total Length	4	6.8	**<0.001**
			Year × Total Length	2	3.3	**0.036**
			Area × Year × Total Length	7	3.2	**0.003**
	GSI	None for both response variable (gonad weight and covariate (carcass weight)	Area	4	2.6	0.038
			Year	2	25.0	<0.001
			Covariate (Carcass Weight)	1	1542.9	<0.001
			Area × Year	7	1.2	0.318
			Area × Carcass Weight	4	1.6	0.164
			Year × Carcass Weight	2	18.0	**<0.001**
			Area × Year × Carcass Weight	7	2.1	**0.048**
	LSI	None for both response variable (liver weight and covariate (carcass weight)	Area	4	0.6	0.637
			Year	2	11.9	<0.001
			Covariate (Carcass Weight)	1	726.2	<0.001
			Area × Year	7	1.4	0.203
			Area × Carcass Weight	4	2.5	**0.039**
			Year × Carcass Weight	2	5.5	**0.004**
			Area × Year × Carcass Weight	7	2.3	**0.027**

Table 9-10 Results of Trend Analysis of Condition, LSI, and GSI in Slimy Sculpin

Sex / Stage	Variable	Transformation	Coefficient	Df	F value	*P*-value
Female	Condition Length	Natural log-transformedboth response (fresh weight) and covariate (total length)	Area	4	0.7	0.583
			Year	2	10.9	<0.001
			Covariate (Total Length)	1	5391.9	<0.001
			Area × Year	7	1.3	0.261
			Area × Total Length	4	5.7	**<0.001**
			Year × Total Length	2	3.2	**0.044**
			Area × Year × Total Length	7	2.4	**0.020**
	GSI	None for both response variable (gonad weight and covariate (carcass weight)	Area	4	3.0	0.019
			Year	2	3.6	0.030
			Covariate (Carcass Weight)	1	1209.5	<0.001
			Area × Year	7	1.3	0.264
			Area × Carcass Weight	4	4.3	**0.003**
			Year × Carcass Weight	2	2.6	0.078
			Area × Year × Carcass Weight	7	3.9	**<0.001**
	LSI	None for both response variable (liver weight and covariate (carcass weight)	Area	4	4.3	0.002
			Year	2	0.7	0.482
			Covariate (Carcass Weight)	1	619.6	<0.001
			Area × Year	7	3.5	**0.002**
			Area × Carcass Weight	4	0.7	0.566
			Year × Carcass Weight	2	3.1	**0.046**
			Area × Year × Carcass Weight	7	2.3	**0.026**

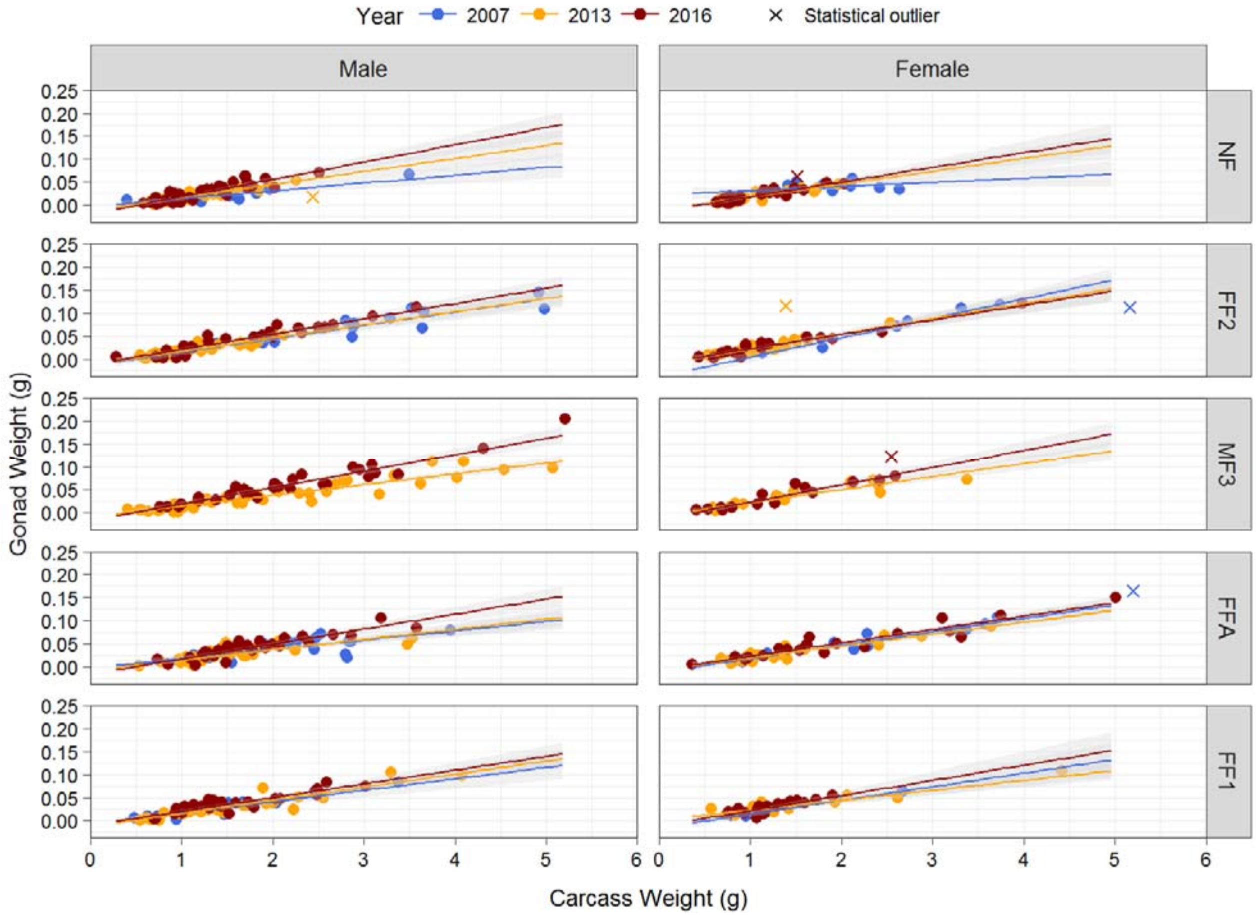
Notes: For analysis of condition factor, the response variables (total weight for age-1+ fish and carcass weight for adult fish) were transformed using Yeo-Johnson log-transformation, which uses an offset. *P*-values of significant interactions and significant main effects (where no significant interactions were found) are shown in bold.

Figure 9-15 Temporal Trend Analysis of Condition in Adult Slimy Sculpin



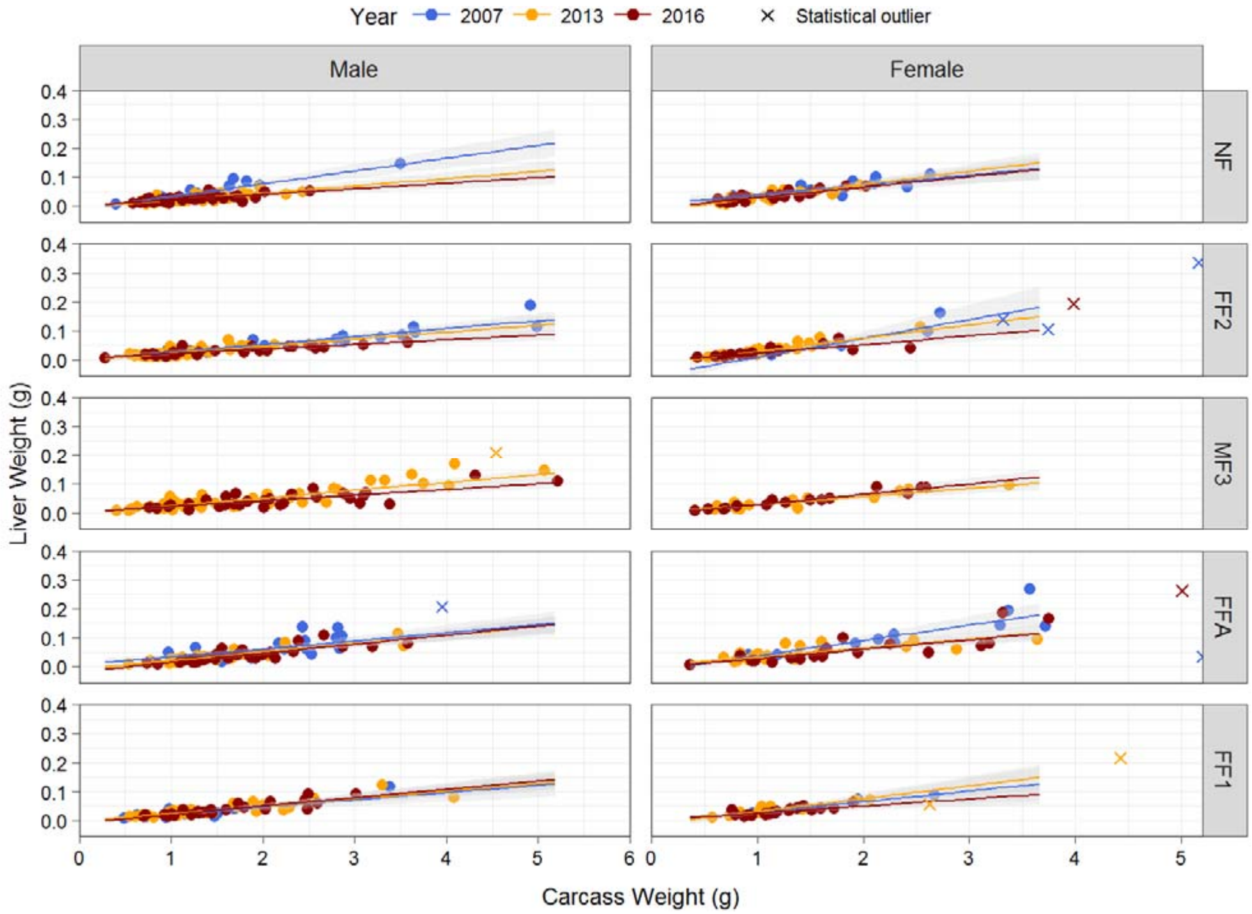
Notes: Response variable was transformed using Yeo-Johnson log-transformation, which uses an offset.

Figure 9-16 Temporal Trend Analysis of Gonad Weight in Adult Slimy Sculpin



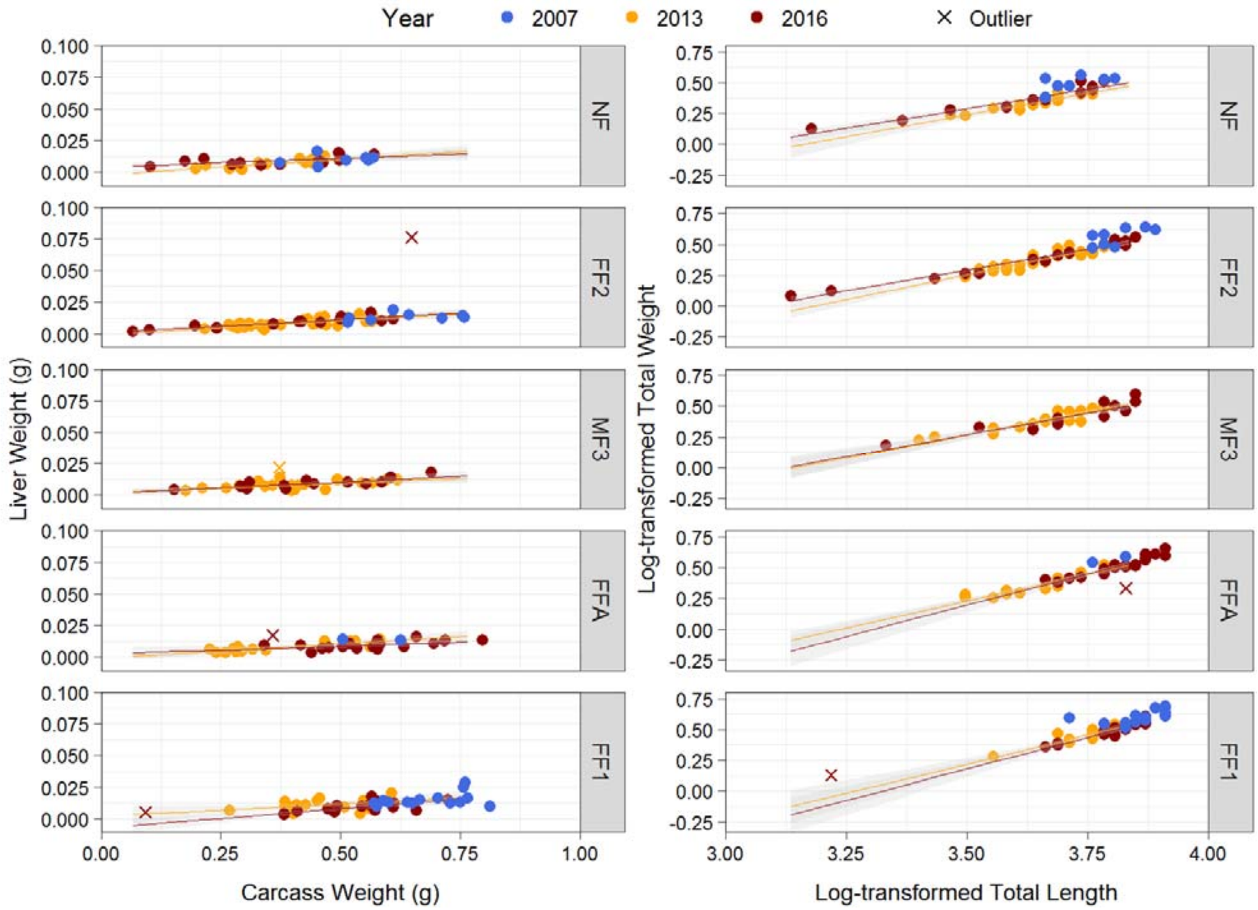
Notes: Three statistical outliers not shown on plots for legibility, all from 2007 FFA samples a female fish, with a carcass weight of 9.5 g and gonad weight of 0.21 g, and two male fish, with carcass weights of 9.6 g and 8.4 g, and gonad weights of 0.04 g and 0.18 g, respectively.

Figure 9-17 Temporal Trend Analysis of Liver Weight in Adult Slimy Sculpin



Notes: Three statistical outliers not shown on plots for legibility, all from 2007 FFA samples: a female fish, with a carcass weight of 9.5 g and a liver weight of 0.77 g and two male fish, with carcass weights of 9.6 g and 8.4 g and liver weights of 0.62 g and 0.65 g.

Figure 9-18 Temporal Trend Analysis of Liver Weight and Condition in Age-1+ Slimy Sculpin



Notes: For analysis of condition factor (right panel), the response variable (total weight) was transformed using Yeo-Johnson log-transformation, which uses an offset.

9.2.2.6 Weight-of-Evidence

Overall, the WOE ratings varied during each fish survey. No consistent effect was observed in 2007. In 2010, a low-level enrichment effect was observed, while in 2013 and 2016 a low-level toxicological effect was documented. No additional visible pattern was evident upon further re-evaluation of the historical WOE assessment for fish health.

Table 9-11 Summary of Weight-of-Evidence Effect Ratings for Slimy Sculpin Fish Health, 2007 to 2016

Endpoint	2007				2010				2013				2016			
	Age-1+	Male	Female	Overall	Age-1+	Male	Female	Overall	Age-1+	Male	Female	Overall	Age-1+	Male	Female	Overall
Length-frequency – Survival	↑/↓			0	↑/↓			0	↑/↓			0	n/a	n/a	n/a	
Growth – Body Size	↓	0	0	0	0	↑↑	0	↑	↓	↓	↓	↓	↓	↓	↓	↓
Energy Stores – Condition Factor	↑	↓	0	0	0	↑	↑	↑	↓	0	0	0	0	0	0	0
Energy Stores – LSI	0	↑	0	0	0	0	↑	↑	↓	↓	↓	↓	0	0	0	0
Relative Reproductive Success – Age-1+ Abundance	↓	n/a	n/a	0	0	n/a	n/a	0	↓	n/a	n/a	0	0	n/a	n/a	0
Relative Reproductive Success - GSI	n/a	0	0	0	n/a	n/a	n/a	n/a	n/a	0	↓	↓	n/a	0	0	0
Tapeworm Parasitism - Occurence	0			0	↑			↑	0			0	↑			↑

Notes: The direction of the arrow, up or down, indicates the direction of change for a given biological response endpoint relative to the far-field areas and/or normal range, or the direction of the relationship between a given endpoint and distance from the diffuser (i.e., for gradient analysis). For biological community structure endpoints, both arrows are included (e.g. ↑/↓) to reflect that a community shift normally involves combined increases and decreases in abundance and diversity.

0 = no response; ↑ = increase ↓ = decrease ↑/↓ change not associated with a direction; n/a = analysis not conducted on sex/stage-class. The number of these charcters represents effect rating (one = early warning/low effect rating; two = moderate level effect rating’ three = high level effect rating; LSI = liversomatic index; GSI = gonadosomatic index.

9.2.2.7 Action Levels

A summary of previous Action Level assessments is provided in Table 9-12. The Action Levels were approved as part of the AEMP Study Design Version 3.2 (Golder 2016a), and the first year of implementation for the fish program was 2013. The Action Levels for fish health address the toxicological impairment hypothesis. Table 9-4 outlines the approved Action Levels from the *AEMP Study Design Version 3.5* (Golder 2014a). Results from the 2013 and 2016 Slimy Sculpin monitoring showed statistically significant decreases for select fish health endpoints in the NF (2013) and MF (2016) areas; however, all endpoints were within the normal ranges. Therefore, according to the approved Action Levels (Golder 2014a), Action Level 1 was triggered in 2013, and Action Level 2 was triggered in 2016.

As part of the stage re-classification, normal ranges were revised for the fish health endpoints using the updated stage groupings (Appendix A). According to the approved Action Levels, an Action Level 3 is triggered when a measurement endpoint is beyond the normal range. As such, a re-assessement of whether or not Action Level 3 was triggered when compared to the updated normal ranges was required. Given that there were no endpoints outside of the normal ranges in either 2013 or 2016 (Section 9.2.1.3.4), the Action Level assessments previously conducted in 2013 and 2016 were considered valid, and an Action Level 3 was not triggered.

9.2.2.8 Magnitude of Difference (Critical Effect Size)

Of the variables that triggered Action Levels in 2016 (i.e., fish weight, fish length, and relative liver size), CES are defined for fish weight, relative liver size and relative gonad size as 25% of the reference area mean, and for fish condition as 10% of the reference area mean. Where fish weight is discussed herein, total body weight and carcass weight are both considered.

Age-1+ total body weight, carcass weight, and adult female total weight and carcass weight were the only Action Level triggers that were also at or above the CES of 25% (Environment Canada 2012; Table 9-12). In Age-1+ fish, carcass weight at NF exceeded the CES, while there was little difference in carcass weights between the two FF areas. In adult females, the largest exceedances of the CES were observed in the FF areas (i.e., the FF areas were most different from each other, at 59% and 58% for total weight and carcass weight, respectively). The observed effects in adult females did not follow the expected spatial gradient of a Mine-related effect (i.e., NF effect was not greater than FF2/MF effect, which was also not greater than FF1 and FFA effect).

No CES is defined for fish length (Environment Canada 2012). In 2016, the magnitude of the difference in fish length ranged from -7.9% (Age-1+ fish at MF3) to -12.5% (Age-1+ fish at NF) for those groups where significant differences were detected (i.e., Age-1+, adult male, and adult female fish; Table 9-12). While no direct comparison to a CES was possible for total length, the magnitude of the difference was below or similar to the effect sizes considered to be ecologically meaningful for the other fish health endpoints (i.e., 25% for weight, relative liver size, and relative gonad size, and 10% for condition [Environment Canada 2012]).

No magnitude of difference could be calculated for LSI due to a significant interaction between area and covariate (i.e., carcass weight); however, this variable was conservatively carried forward for further review.

Table 9-12 Percent Difference of Variables that Triggered Action Levels in Slimy Sculpin from Lac de Gras, 2016

Sex/Stage	Variables	2016			
		NF/ MF vs FF(a)			FF(b)
		NF	FF2	MF3	FF1 vs FFA
Age-1+	Total length (mm)	**-12.5**	**-10.5**	**-7.9**	-1.9
	Total weight (g)	**-25.0**	**-19.4**	**-19.7**	-5.0
	Carcass weight (g)	**-28.8**	**-21.2**	**-20.1**	-3.8
Adult male	Total length (mm)	**-9.2**	0.9	5.6	-4.4
	Total weight (g)	**-23.3**	6.0	19.2	-11.8
	Carcass weight (g)	**-24.3**	5.8	17.3	-11.7
	LSI (g) (c)	*Area-specific interpretation not possible due to significant interaction*			
Adult female	Total length (mm)	**-10.1**	-7.7	-5.8	**-17.8**
	Total weight (g)	-23.7	**-30.8**	-18.4	**-58.9**
	Carcass weight (g)	**-26.5**	**-31.9**	-18.8	**-58.4**

Notes: Values for variables shown to be significantly different from FF area means are shown in bold. Variables that exceed the Environment Canada (2012) CES are highlighted grey. - = Action Level not triggered; LSI = liversomatic index; NF = near-field; MF = mid-field; FF = far-field.

a) Percent difference between NF, FF2 or MF area means and the pooled FF area mean (i.e., magnitude difference as defined for the AEMP).

b) Relative percent difference (RPD) between the FF area means, calculated as (predicted mean for FF1 – predicted mean for FFA)/mean of predicted means for FFA + FF1.

c) LSI was analyzed using an ANCOVA with carcass weight as a covariate.

9.3 Fish Tissue

9.3.1 Methods

9.3.1.1 Data Sources

Fish tissue chemistry data are available for both Slimy Sculpin and Lake Trout.

9.3.1.1.1 Slimy Sculpin

Five fish tissue chemistry surveys have been carried out with Slimy Sculpin in Lac de Gras (Table 9-1):

- 2004 survey (CRI 2006);
- 2007 AEMP (August 22 to September 2) (Golder 2008);
- 2010 AEMP (June 28 to July 20) (Golder 2011c);
- 2013 AEMP (August 27 to September 10) (Golder 2014c) and;
- 2016 AEMP (August 31 to September 14) (Golder 2017c).

As noted above, as part of the 2004 fish survey (CRI 2006) Slimy Sculpin were collected from East Island on Lac de Gras. These data are not included herein, because the metals anaylsis was conducted on homogenized body carcass tissue (i.e., carcass excluding heads, gonads, stomachs, livers, and gas bladders), which is not consistent with the fish bodies submitted as part of the AEMP program (i.e. carcass excluding gonads, otoliths, and stomachs). The 2010 fish tissue chemistry results are not included in the current review (as per Section 9.2.1.1).

Table 9-13 Review and Summary of AEMP Slimy Sculpin Fish Tissue Chemistry Data Collected (up to and including 2016)

Waterbody	Area	Years Sampled												
		2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Lac de Gras	Near-field (NF)				✓			X			✓			✓
	Far-field 2 (FF2)				✓			X			✓			✓
	Mid-field 3 (MF3)	X									✓			✓
	Far-field 1 (FF1)				✓			X			✓			✓
	Far-field A (FFA)				✓			X			✓			✓

✓ = fish tissue collected and included in analysis

X = fish tissue data collected but not included because different fish tissues were included in the analyses (2004) or captured at different time of year (2010).

Eight composite samples of Slimy Sculpin captured at each of the four study areas (i.e., NF, FF2, FF1, FFA) were submitted for the analysis of metals in 2007, 2010, 2013, and 2016. Eight composite samples of Slimy Sculpin collected from the MF3 area were also analyzed in 2013 and 2016. In 2007, 2010 and 2013 the samples consisted of fish bodies (i.e., carcass excluding gonads, otoliths, and stomachs). Gonads, otoliths and stomachs were not included, as they were required for separate analyses as part of the fish health assessment. In 2016, the samples consisted of carcass only. The livers were excluded from the samples in 2016 due to a field error. In all years, the fish making up a composite sample were of the same sex and size class; four male and four female composite samples from each area were submitted for analysis. Samples were composited to meet minimum sample weight requirements of 5 to 7 g wet weight (wwt) from each area. Samples were analyzed by ALS, Edmonton, Alberta in 2007, and ALS, Burnaby, British Columbia in 2010 and 2013 and 2016 for metals listed in Table 9-14.

Table 9-14 Variables Analyzed and Method Detection Limits for Slimy Sculpin Tissue Samples from Lac de Gras, 2007 to 2016

Variable	Detection Limit (μg/g wwt)		
	2007	2013	2016
% Moisture Content	0.1	0.1	0.5[(b)]
% ww Lipid Content	nt	nt	0.5
Aluminum	2	0.4	0.4-1
Antimony	0.05	0.002	0.002
Arsenic	0.05	0.004	0.004-0.006

Table 9-14 Variables Analyzed and Method Detection Limits for Slimy Sculpin Tissue Samples from Lac de Gras, 2007 to 2016

Variable	Detection Limit (µg/g wwt)		
	2007	2013	2016
Barium	0.1	0.01	0.01
Beryllium	0.2	0.002	0.002
Bismuth	0.2	0.002	0.002
Boron	2	0.2	0.2
Cadmium	0.01	0.002	0.001-0.02
Calcium	20	5.0[(a)]	4
Cesium	0.05	0.001	0.001
Chromium	0.1	0.01	0.01
Cobalt	0.1	0.004	0.004
Copper	0.05	0.01	0.02-0.04
Gallium	nt	0.004	nt
Iron	5	0.2	0.6-1
Lead	0.02	0.004	0.004-0.01
Lithium	nt	0.02	0.1
Magnesium	5	10[(a)]	0.4
Manganese	0.5	0.004	0.01
Mercury	0.01	0.001	0.001
Molybdenum	0.05	0.004	0.004-0.008
Nickel	0.02	0.01	0.04
Phosphorus	20	50[(a)]	2
Potassium	20	200[(a)]	4
Rhenium	nt	0.002	nt
Rubidium	0.05	0.01	0.01
Selenium	0.05	0.02	0.01-0.02
Silver	0.05	0.001	0.001
Sodium	20	200	4
Strontium	0.05	0.01	0.01-0.02
Tellurium	0.5	0.004	0.004
Thallium	0.05	0.0004	0.0004
Thorium	nt	0.002	nt
Tin	0.1	0.02	0.02
Titanium	0.2	0.01	0.02-0.1
Uranium	0.05	0.0004	0.0004
Vanadium	0.1	0.020[(a)]	0.02
Yttrium	nt	0.002	nt

Table 9-14 Variables Analyzed and Method Detection Limits for Slimy Sculpin Tissue Samples from Lac de Gras, 2007 to 2016

Variable	Detection Limit (µg/g wwt)		
	2007	2013	2016
Zinc	0.5	0.1	0.1-0.2
Zirconium	nt	0.04	0.04

a) Laboratory detection limit differed from that originally provided by the lab and listed in the *AEMP Study Design Version 3.5* (Golder 2014a).

b) Laboratory detection limit differed from that originally provided by the lab and listed in the *AEMP Study Design Version 4.1* (Golder 2017d). In 2016 the AEMP Design Plan Version 4.1 detection limits were reported to reflect updated analytical procedures.

c) One sample had a detection limit of 0.01 µg/g wwt.

nt = variable not tested.

9.3.1.1.2 Lake Trout

Mercury concentrations were measured in muscle, liver and kidney tissue from Lake Trout collected in Lac de Gras in 1996, 2002, 2003, 2004, 2005 and 2008, and Lac du Sauvage in 1996 and 2008. Mercury concentrations were also measured in muscle in 2011 and 2014 in both Lac de Gras and Lac du Sauvage. Additional mercury in muscle tissue data were collected as part of the Mine palatability studies in 2002, 2003, 2004, 2012, and 2015, and these data were incorporated into the analyses, where appropriate. These data were not previously included in the AEMP Annual Report or the previous re-evaluation reports because these programs had low sample sizes and different field methods than the AEMP. For this re-evaluation, however, statistical analyses were revised to try and maximize use of the full fish tissue dataset. It is noted, however, that comparison of all tissue chemistry up until 2010 was provided in the *AEMP Version 2.0 Summary Report* (Golder 2011a). As the only new additional data collected since 2010 was mercury in muscle tissue, only a comparison of mercury in muscle tissue is provided herein.

The comparison of Lake Trout chemistry data across the years had the following constraints:

- Concentrations in fish in 1996 were measured in composite samples, not individual fish. Because mercury bioaccumulates and biomagnifies in fish tissue and differences in mercury concentrations can be confounded by differences in fish body size, the 1996 data are not appropriate for use. Temporal and spatial comparisons were not conducted with this data.
- Tissue mercury samples in 2005 and 2008 were analyzed by ALS with a DL of 0.01 µg/g wwt. The 2008 mercury samples were also analyzed by Flett (Flett Research Ltd.), Winnipeg, Manitoba, with a DL 0.0004 µg/g wwt. The 2011 and 2014 mercury samples were analyzed by Flett. The palatability study samples (i.e., 2002, 2003, 2004, 2012 and 2015) were analysed by ALS with a DL of 0.01 µg/g wwt.

9.3.1.2 Data Handling

9.3.1.2.1 2016 Correction Factor for Omission of Livers

As part of the *2014 to 2016 AEMP Response Plan Fish – Supplemental Report* (Golder 2017e), the effect of liver exclusion on the results of molybdenum, strontium, and uranium concentrations in Slimy Sculpin tissues in 2016 was examined by comparing liver and muscle metal concentrations in other fish species in the Northwest Territories, Canada (i.e., data from other projects' reference lakes were used, with permission). This approch was presented in the *2014 to 2016 AEMP Response Plan Fish – Supplemental Report* (Golder 2017e), which was approved by the WLWB (WLWB 2018).The same approach has been implemented herein to include the full metal suite analysed as part of the Slimy Sculpin fish tissue chemistry program.

In the De Beers Snap Lake AEMP, Lake Trout and Round Whitefish captured in Northeast Lake, Lake 13, Mackay Lake, and Reference Lakes between 1999 and 2016 were analysed for tissue metal concentrations in muscle and liver (De Beers 2002, 2005, 2010, 2014, 2016a, and 2017). For this report, the ratio between individual fish liver and muscle metal concentrations was calculated for each Round Whitefish and Lake Trout. The means and 95% confidence intervals of liver:muscle concentration ratios were calculated for each of the two species and each of the metals. The higher of the two species' concentration ratios for each metal was used as a conservative scenario to apply to Slimy Sculpin.

Slimy Sculpin mean liver weight was calculated for each composite sample sent for tissue chemistry analysis using the fish health data collected for the individual fish that constituted the composite sample. The absolute amount of metals in Slimy Sculpin livers was calculated by multiplying carcass concentrations of metals by each composite mean liver weight; this value was then corrected by multiplying by the liver:muscle concentration ratio described above. Total body burden was then re-calculated as carcass burden (i.e., original metal concentration multiplied by carcass weight), summed with the corrected liver burden. The resulting value was divided by the sum of carcass weight and liver weight, and compared to the original concentrations of metals. In cases where the corrected concentrations were less than the original concentrations (due to lower accumulation of the metal in liver relative to muscle of Round Whitefish or Lake Trout), the original values were retained.

Estimated changes in carcass concentrations of metals due to inclusion of livers in tissue chemistry analyses differed by variable (Table 9-15). The exclusion of liver tissue from the 2016 fish tissue chemistry analysis may have under-estimated concentrations of some metals, particularly those that preferentially accumulate in the liver. The percent changes in corrected tissue concentrations were calculated based on relationships between liver and muscle tissue in reference lakes in the NWT in large-bodied fish (Round Whitefish and Lake Trout), and these lakes had reported lower concentrations of select metals in the water (e.g., molybdenum is 0.06 to 0.14 µg/L at Northeast Lake and Lake 13, respectively, in 2015 [De Beers 2016b]). This may mean fish in Lac de Gras are bioconcentrating some metals less in their tissues than in the De Beers reference lakes. These estimates, therefore, may over-estimate the contribution of Slimy Sculpin livers in the corrected tissue concentrations for some metals like cadmium and silver, but have been conservatively retained.

The corrected 2016 Slimy Sculpin fish tissue concentrations were used in all subsequent data analyses and interpretation presented herein.

Table 9-15 Correction Factors for Omission of Livers in 2016 Slimy Sculpin Fish Tissue Chemistry Data

Variable	Correction ratio (liver concentration [µg/g wwt] : muscle concentration [µg/g wwt])	Percent change in corrected tissue concentration (%)		
		NF	MF	FF
Aluminum	4.5 (2.9 - 6.1)	9 (5 - 13)	8 (4 - 12)	9 (5 - 13)
Antimony	1.1 (1 - 1.3)	0 (0 - 1)	0 (0 - 1)	0 (0 - 1)
Arsenic	3.5 (2.7 - 4.3)	6 (4 - 8)	6 (4 - 8)	6 (4 - 8)
Barium	1.7 (1 - 2.7)	2 (0 - 4)	2 (0 - 4)	2 (0 - 4)
Beryllium	1 (1 - 1.1)	0 (0 - 0)	0 (0 - 0)	0 (0 - 0)
Bismuth	6 (1 - 11.8)	12 (0 - 27)	11 (0 - 24)	12 (0 - 27)
Boron	1 (1 - 1.1)	0 (0 - 0)	0 (0 - 0)	0 (0 - 0)
Cadmium	157.9 (116.1 - 199.8)	394 (289 - 499)	355 (260 - 449)	388 (285 - 492)
Calcium	1.3 (1 - 1.6)	1 (0 - 2)	1 (0 - 1)	1 (0 - 2)
Cesium	1 (1 - 1)	0 (0 - 0)	0 (0 - 0)	0 (0 - 0)
Chromium	2.7 (2 - 3.5)	4 (2 - 6)	4 (2 - 6)	4 (2 - 6)
Cobalt	26.1 (18.5 - 33.7)	63 (44 - 82)	57 (39 - 74)	62 (43 - 81)
Copper	56.5 (46.4 - 66.5)	139 (114 - 165)	125 (103 - 148)	137 (112 - 162)
Iron	75.5 (57.5 - 93.5)	187 (142 - 232)	168 (128 - 209)	184 (140 - 229)
Lead	2.8 (1.7 - 3.9)	5 (2 - 7)	4 (2 - 7)	4 (2 - 7)
Lithium	1 (1 - 1.1)	0 (0 - 0)	0 (0 - 0)	0 (0 - 0)
Magnesium	1 (1 - 1)	0 (0 - 0)	0 (0 - 0)	0 (0 - 0)
Manganese	15.6 (13.7 - 17.6)	37 (32 - 42)	33 (29 - 38)	36 (31 - 41)
Mercury	1.4 (1.3 - 1.5)	1 (1 - 1)	1 (1 - 1)	1 (1 - 1)
Molybdenum	69.7 (58 - 81.5)	173 (143 - 202)	155 (129 - 182)	170 (141 - 199)
Nickel	3.5 (2.7 - 4.3)	6 (4 - 8)	6 (4 - 7)	6 (4 - 8)
Phosphorus	1.4 (1.4 - 1.5)	1 (1 - 1)	1 (1 - 1)	1 (1 - 1)
Potassium	1 (1 - 1)	0 (0 - 0)	0 (0 - 0)	0 (0 - 0)
Rubidium	1.2 (1.1 - 1.3)	0 (0 - 1)	0 (0 - 1)	0 (0 - 1)
Selenium	7.2 (6.3 - 8)	15 (13 - 18)	14 (12 - 16)	15 (13 - 17)
Silver	111.3 (68.4 - 154.2)	277 (169 - 385)	249 (152 - 346)	273 (167 - 379)
Sodium	5.2 (4.8 - 5.5)	10 (10 - 11)	9 (9 - 10)	10 (9 - 11)
Strontium	2.2 (1.7 - 2.8)	3 (2 - 5)	3 (2 - 4)	3 (2 - 4)
Tellurium	1 (1 - 1)	0 (0 - 0)	0 (0 - 0)	0 (0 - 0)
Thallium	8.3 (7.3 - 9.2)	18 (16 - 21)	16 (14 - 19)	18 (16 - 20)
Tin	7 (4 - 10.1)	15 (8 - 23)	14 (7 - 20)	15 (7 - 22)
Titanium	3.9 (2.3 - 5.5)	7 (3 - 11)	7 (3 - 10)	7 (3 - 11)
Uranium	11.8 (8.5 - 15.2)	27 (19 - 36)	25 (17 - 32)	27 (19 - 35)
Vanadium	1.3 (1.1 - 1.4)	1 (0 - 1)	1 (0 - 1)	1 (0 - 1)
Zinc	9.2 (8.6 - 9.8)	21 (19 - 22)	18 (17 - 20)	20 (19 - 22)
Zirconium	1 (1 - 1)	0 (0 - 0)	0 (0 - 0)	0 (0 - 0)

Notes: Values presented are means with 95% confidence intervals in parentheses (for correction ratio and estimated tissue concentrations).

9.3.1.2.2 Data Screening

Initial screening of the AEMP fish tissue chemistry datasets was completed before data analyses to identify unusually high (or low) values in the datasets, and decide whether to retain or exclude anomalous data from further analysis. The anomalous data screening approach for AEMP component datasets was approved as part of the *2011 to 2013 Aquatic Effects Re-evaluation Report* (Golder 2016a). The screening of the datasets was conducted using a method based on Chebyshev's theorem (Mann 2010) combined with the visual examination of scatterplots. Data flagged as potentially anomalous points following the application of Chebyshev's theorem were visually examined to assess whether data were flagged simply due to high percentage of <DL values. Thus, the data screening approach includes a numerical method to aid in the identification of outliers, removing the subjectivity of classifying values based on visual evaluation of data in scatterplots alone.

Chebyshev's theorem (combined with the visual examination of scatterplots)

This method allows for detection of multiple outliers at one time and assumes that the data being screened contain a relatively small percentage of outliers (Amidan et al. 2005). The theorem states that at least $1–1/k^2$ proportion of the data of any distribution (i.e., no assumption of normality) lies within k SDs of the mean. Setting $1–1/k^2 = 0.95$ and solving for k results in 4.47 SD, indicating that 95% of the data, regardless of distribution, will be within approximately 4.5 SD of the mean. In the case of a normal distribution, 95% of the data is expected to be within 2 SD, suggesting that the method based on Chebyshev's inequality is conservative (i.e., identifies values that are far removed from the mean). The method is applied by first identifying data that lie outside the 4.47 SD on a scatterplot of annual data, and then visually verifying the anomalous values based on potential spatial trends. If data were visually anomalous, the value was investigated to see if it was reported in error, or if it was consistent with associated variables (e.g., TDS and major ion concentrations) and data collected in previous years. No data were identified as anomalous based on visual evaluation alone.

In cases where the above numerical screening identified an elevated value in the NF area as anomalous, the identified value was conservatively retained in the dataset used for analysis if the SD distance from the mean was less than two times the 4.5 SD criterion discussed above. Hence, only very extreme values, which were greater than approximately 9 SD from the mean, were removed from further analysis of NF area data, upon visual confirmation of screening results. In cases where the annual datasets contained a large proportion of non-detect data (i.e., censored values), only values that were greater than or equal to 5 times the DL were considered anomalous and were removed from the analysis if visual screening confirmed the numerical screening results.

Full details on this data screening approach are provided in the *Quality Assurance Project Plan Version 3.1* (Golder 2017a), and Section 2.6 of the *2011 to 2013 Aquatic Effects Re-evaluation Report* (Golder 2016a), when this data screening method was first adopted.

9.3.1.2.3 Censored Data

Prior to data analyses, non-detect values (i.e., less than DL) were multiplied by 0.5 to achieve a value of half the DL. Substitution with half the DL is a common approach used to handle censored data (US EPA 2000) and is consistent with the approved methods applied in the calculation of the normal range in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b). While other components (e.g., effluent and water chemistry) generally refrained from analysis of data with ≥15% non-detect values, this approach was not possible for fish tissue data for two reasons:

1) Temporal trends in fish tissue chemistry were analyzed with year as a discrete variable (see Section 9.3.1.3.2). This meant that the proportion of less than DL data within each year (rather than across years within an area) had to be considered to avoid cases where some years had a high proportion of less than DL data, which affected residual normality and homoscedasticity. The same reason prevented the use of the 15% cutoff within year across area.
2) Only eight fish tissue samples were available for each year and area; therefore, if a 15% cutoff was applied within each year and area, the presence of a single less than DL value would prevent analysis of the variable.

Therefore, the cutoff used for fish tissue analysis was 60% within at least one area in any given year, following the methods used in the 2016 Annual AEMP (Golder 2017c). In these cases, modified analysis approaches were used when possible.

For cesium, molybdenum, thallium, and tin, where 2007 data had higher DLs than subsequent years, the analysis was performed on 2013 and 2016 data only. For uranium, where a temporal trend was visually identified between 2007, 2013, and 2016, and all 2007 FF data were below the DL, the analysis was performed using the full dataset for 2013 and 2016 and only NF data in 2007, to preserve the three years in the trend analysis.

For lead analysis, logistic modeling was used, since 60% or more below DL values were recorded in multiple years and areas throughout the sampling program, but the overall trends were of interest. In this analysis, a logistic model predicted changes in the likelihood of observing data above the DL as function of year and area.

Temporal trend analyses were not possible for antimony, beryllium, bismuth, boron, chromium, silver, tellurium, lithium, and zirconium, due to a very high proportion of less than DL data across sampling years. For example, bismuth data were not analyzed, since less than DL values accounted for 100%, 70%, and 78% of the data in 2007, 2013, and 2016 respectively. The distribution of the presence/absence of detected data did not allow for the use of logistic regression, since all data from FF areas were less than DL across all years and since the temporal gradient of presence/absence of detected data in NF and MF was not apparent enough to support logistic regression. Silver data were not analyzed, because while in 2010 all data were detected, less than DL data accounted for 100%, 40%, and 80% of the data in 2007, 2013, and 2016, with an insufficient temporal gradient in the change between the presence and absence of detected data.

9.3.1.3 Data Analysis

9.3.1.3.1 Comparison to Normal Ranges

Slimy Sculpin

Differences in Slimy Sculpin fish tissue variables were evaluated by comparing results in the NF, FF2 and MF areas to background values. Background values for Lac de Gras were those that fell within the range of natural variability, referred to as the normal range. Normal ranges were calculated using data from two

AEMP FF areas (i.e., FF1 and FFA), incorporating data from 2007 and 2013. In 2007, 2013, and 2016, sampling was performed in late summer. As the 2010 samples were collected in early spring and could not be directly compared to all other years of data, the normal range comparison for 2010 was excluded from this analysis. The 2010 normal range comparison was previously presented in the *2011 to 2013 Aquatic Effects Re-evaluation Report* (Golder 2016a). The normal ranges for fish tissue chemistry were obtained from the *AEMP Reference Conditions Report, Version 1.2* (Golder 2017b) and are summarized in Table 9-16.

Time series plots for variables with detected concentrations were generated using data from 2007, 2013, and 2016 for the NF, FF2, and MF3 (2013 and 2016 only) areas, and the FF areas (i.e., FF1 and FFA) of Lac de Gras. The 2010 time series plots were presented in the previous re-evaluation report (Golder 2016a) and are not reproduced as part of this re-evaluation. The mean values were plotted for each variable and were compared to the normal range (Table 9-16).

Table 9-16 Late Summer Normal Ranges for Fish Tissue Chemistry

Variable	Late Summer Normal Range (µg/g wwt)	
	Lower limit	Upper limit
Aluminum	14.8	30.0
Antimony	0	0.002
Arsenic	0.120	0.150
Barium	3.73	4.95
Beryllium	0	0.002
Bismuth	0	0.002
Boron	0	2.0
Cadmium	0.020	0.030
Calcium	7,503	10,575
Cesium	0	0.095
Chromium	0.65	2.00
Cobalt	0.125	0.300
Copper	0.93	1.113
Gallium	0	0.004
Iron	30	43
Lead	0	0.02
Lithium	0.031	0.056
Magnesium	349	426
Manganese	9.23	12.60
Mercury	0.033	0.085
Molybdenum	0	0.05
Nickel	0.913	1.420
Phosphorus	5,723	7,338
Potassium	3,260	3,365
Rhenium	0	0.002
Rubidium	5.82	6.83
Selenium	0.403	0.453

Table 9-16 Late Summer Normal Ranges for Fish Tissue Chemistry

Variable	Late Summer Normal Range (µg/g wwt)	
	Lower limit	Upper limit
Silver	0	0.001
Sodium	1,083	1,198
Strontium	26.4	34.9
Tellurium	0	0.004
Thallium	0.004	0.005
Thorium	0	0.00255
Tin	0.038	0.049
Titanium	0	0.4
Uranium	0.009	0.0167
Vanadium	0.20	0.20
Yittrium	0	0.003
Zinc	25.23	29.48
Zirconium	0	0.04

Source: *AEMP Reference Conditions Report Version 1.2* (Golder 2017b).

µg/g wwt = microgram per gram wet weight.

9.3.1.3.2 Temporal Trends

Slimy Sculpin

The temporal trend analysis used for fish tissue data differed from the general trend analysis methods described in Section 2.4.2. Since only three years of data were available for analysis (i.e., 2007, 2013, and 2016), the general methods could not be applied because year was not a continuous covariate to examine temporal trends. In addition, fish data do not have stations nested within NF/FF areas (represented in the general methods as a random factor). Therefore, the trend model described in Section 2.4.2 was simplified into a fixed effect model, with year and area as discrete covariates. The full model contained effects of year, area, and their interaction. No model selection was performed, since there was only one candidate model for analysis. Data transformations using Yeo-Johnson transformations were performed as described in Section 2.4.2. Residual heteroscedasticity was examined and heteroscedasticity effects by area, year, or predicted value were added to the model, if they improved model fit (as assessed using AIC scores). Analysis outputs included tables of parameter significance, Tukey-adjusted multiple comparisons (Tukey 1977) between years within area, and plots of fitted estimates and 95% confidence intervals, overlaying the fish data.

For variables that were used in logistic regressions due to high prevalece of less than DL data, values were expressed as presence/absence of detected data. Model outputs included parameter significance and plots of predicted probabilities overlaying the presence/absence of detected data.

Lake Trout

The distributions of mercury concentrations in Lake Trout muscle tissue by year and area (i.e., Lac de Gras or Lac du Sauvage) were plotted using boxplots for 1996 to 2015. The boxes were defined with the 25th percentile, the median, and the 75th percentile. The whiskers were defined as 1.5 times the interquartile distance, and concentrations beyond the whiskers were plotted as individual points. A temporal and spatial analysis for mercury concentrations was conducted using linear regression. A set of five models was constructed, with the following predictors:

1) Lake, year, and fork length, with an interaction between year and lake
2) Lake, year, and fork length
3) Lake and fork length (no year effect)
4) Year, and fork length (no lake effect)
5) Fork length only (no year, no lake effects)

In all the models, year was represented as a polynomial of the 2nd degree, to account for the parabolic pattern of mercury with time. Once all five models were fit, AIC, corrected for small sample size (AICc) was used for model selection. The model with the lowest AICc score among a set of candidate models was interpreted to have the strongest support, given the set of examined models and the collected data (Burnham and Anderson 2002), and was selected for interpretation. Models with AICc scores within 2 units of each other were considered to have similar levels of support; therefore, the simplest model with the fewest parameters that was within 2 AICc units from the best-supported model was selected for interpretation (Arnold 2010). The model selected for analysis was used to plot prediction curves with 95% confidence intervals. Since mercury concentration is length-dependent, to compare mercury between lakes, it was required to account for differences in Lake Trout lengths between lakes. Therefore, the model was used to conduct pairwise comparisons for year and lake, for a mean Lake Trout fork length of 620 mm to assess spatial and temporal differences in mercury concentrations.

In addition to the overall set of models, the difference between Lac du Sauvage and Lac de Gras mercury concentrations was evaluated using a three-way analysis of covariance (ANCOVA) with year, lake, and fork length effects and an interaction between year and lake. The model parameters were evaluated for significance, and pairwise comparisons between the two lakes were performed if the interaction was significant.

9.3.1.3.3 Guideline Comparison

Human Health

Mercury concentrations in Lake Trout muscle were compared to available national guidelines for human health. The Canadian Food Inspection Agency (CFIA) and Health Canada guideline state that fish collected for commercial use may contain a maximum of 0.5 μg/g wwt mercury to be approved for human health consumption (CFIA 2015).

Fish Health

Excessive levels of mercury can adversely affect fish health. Mercury concentrations in Lake Trout muscle were compared to a potential effects benchmark/tissue residue guideline of 1.0 µg/g wwt. Above this concentration, adverse effects to fish health may be observed (Jarvinen and Ankley 1998). This is likely a very conservative benchmark; Environment and Climate Change Canada recently conducted a review of mercury in the Canadian environment and noted that the lowest adverse effect concentrations would be 0.5 to 1 mg/g wet weight in fish species such as Northern Pike and Walleye (ECCC 2016), many times higher than using the generic benchmark.

9.3.1.3.4 Weight-of-Evidence Effect Ratings

The results of the fish tissue chemistry surveys were integrated through the WOE evaluation process, which determines the strength of evidence supporting the two broad impact hypotheses for Lac de Gras (i.e., toxicological impairment and nutrient enrichment), as described in the *AEMP Study Design Version 3.5* (Golder 2014a). The WOE is not intended to determine the ecological significance or level of concern associated with given changes. The WOE effect ratings incorporate statistical comparisons of the NF and FF areas, and comparisons of the NF area to the normal range as defined in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b). The fish tissue chemistry data were assessed according to the WOE effects ratings described in Section 10 and described in Table 9-17. The 2010 WOE assessment is included herein for completeness.

Table 9-17 Weight-of-Evidence Effect Ratings for Fish Tissue Chemistry

LOE Group	Measurement Endpoint Analysis	No Response 0	Early Warning/Low ↑	Moderate ↑↑	High ↑↑↑
Slimy Sculpin Tissue Chemistry	**Comparison to FF Areas and Normal Range[(a)]** Substances of potential toxicological concern	No difference	Statistically-significant increase, NF vs FF areas	Low + NF area mean >normal range	Moderate + MF area mean >normal range

Notes: Normal ranges for each LOE group and measurement endpoint are defined and provided in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b).

a) Applied separately for each measurement endpoint.

NF = near-field; MF = mid-field, FF = far-field; >= greater than.

9.3.2 Results

9.3.2.1 Comparison to Normal Ranges

Slimy Sculpin

Normal range comparsions were conducted on data collected in the late summer (i.e., 2007, 2013 and 2016). Mean within-area values exceeded the upper limit of normal range for several metals (Figures 9-19 to 9-26), as follows:

- **Near field**: variables that exceeded the normal range in the NF area only were calcium, mercury, phosphorus, and zinc in 2007, and thallium in 2013.
- **Near-field and Mid-field**: Variables that exceeded the normal range in NF and at least one MF area were barium, lead, magnesium, sodium, strontium, titanium, and vanadium in 2007; bismuth, calcium, potassium, silver, sodium, strontium, tin, uranium, and zinc in 2013; and, calcium, molybdenum, phosphorus, silver, strontium, tin, uranium, and zinc in 2016.
- **Far-field:** Variables that exceeded the normal range in at least one FF area were titanium, and vanadium in 2007; calcium, potassium, silver, sodium, strontium, tin, and zinc in 2013; and, calcium, molybdenum, phosphorus, silver, sodium, strontium, tin, and zinc in 2016.

Strontium was the only variable with normal range exceedances at the NF area in all years. Uranium normal range could not be determined for 2007 due to changes in DLs from the lab (Golder 2017b); however, uranium exceeded the normal range at the NF and MF area for all subsequent years. These two variables were evaluated further as part of the *2014 to 2016 AEMP Response Plan Fish – Supplemental Report* (Golder 2017c).

Gallium, rhenium, thorium, and yttrium were presented in this re-evaluation for the sake of completeness. These metals are, however, no longer provided within the fish tissue metals package as analyzed by the laboratory, and as such they were discontinued in the *AEMP Study Design Version 4.1* (Golder 2017d) and will not be presented in future reports.

Figure 9-19 Temporal Trend Plot of Aluminum, Antimony, Arsenic, Barium, and Beryllium Concentrations in Slimy Sculpin Tissue

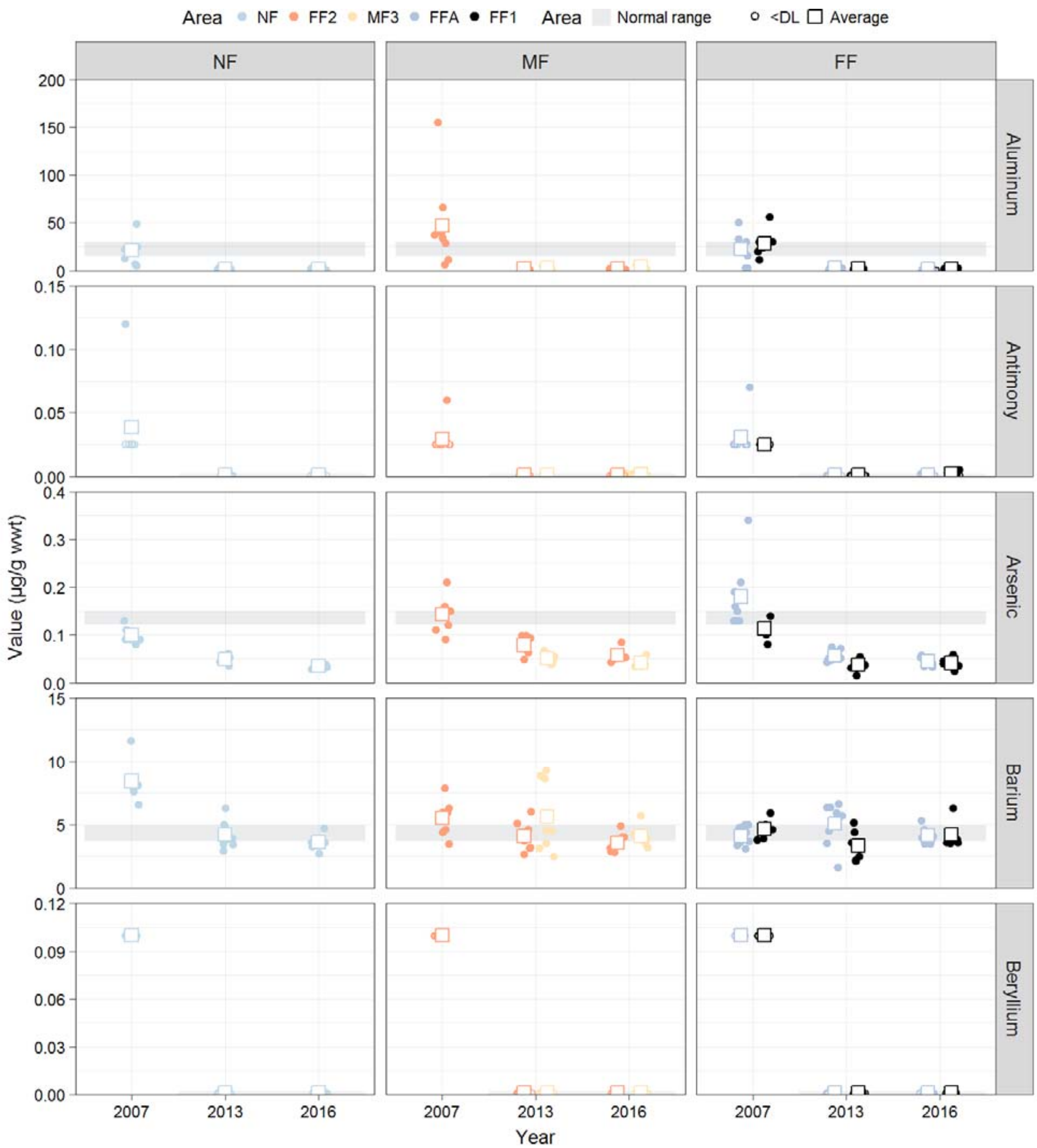
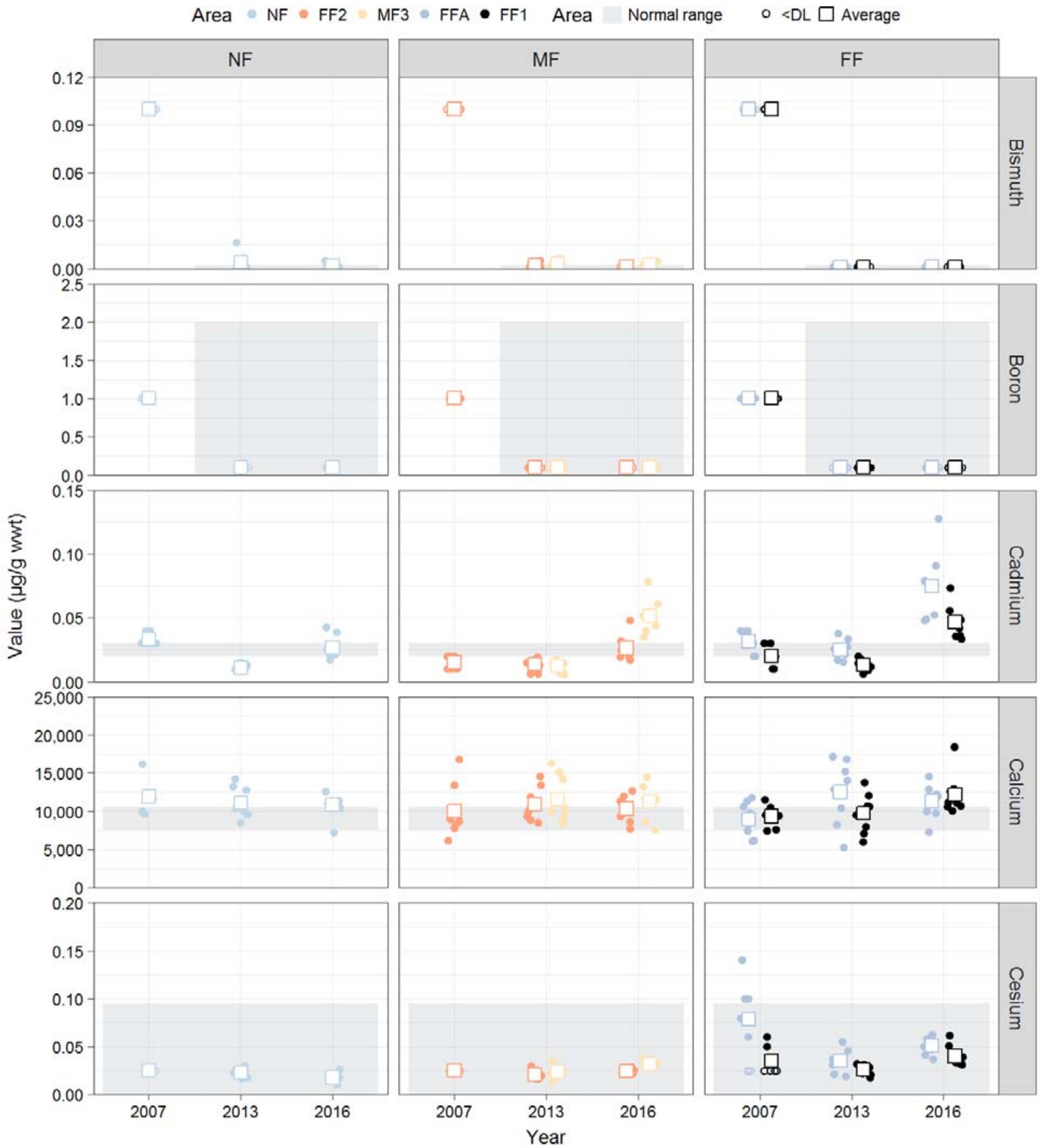


Figure 9-20 Temporal Trend Plot of Bismuth, Boron, Cadmium, Calcium, and Cesium Concentrations in Slimy Sculpin Tissue



Note: For bismuth, the normal range is 0 to 0.002 µg/g wwt and not visible on the scale of the plot.

Figure 9-21 Temporal Trend Plot of Chromium, Cobalt, Copper, Gallium, and Iron Concentrations in Slimy Sculpin Tissue

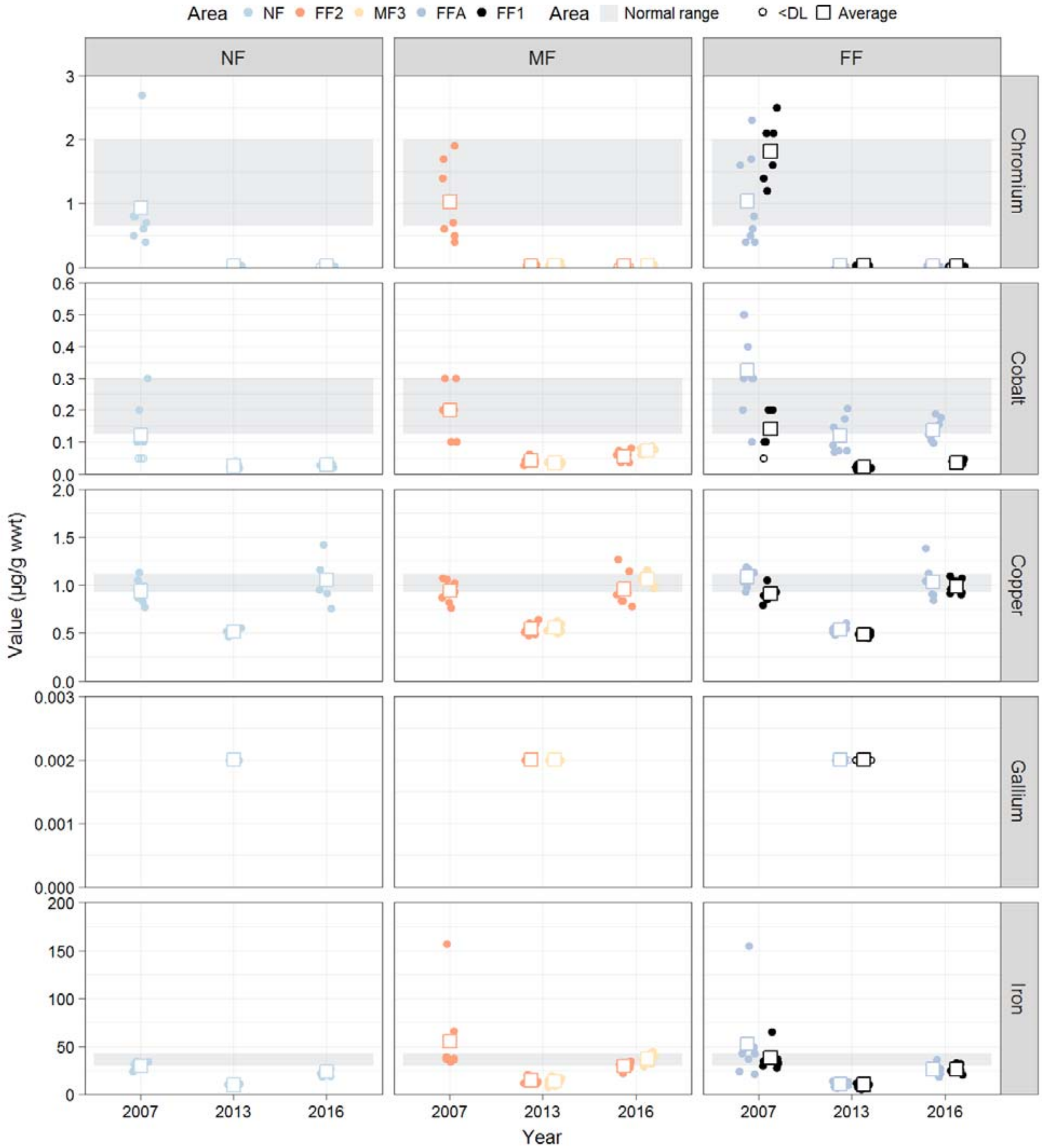


Figure 9-22 Temporal Trend Plot of Lead, Lithium, Magnesium, Manganese, and Mercury Concentrations in Slimy Sculpin Tissue

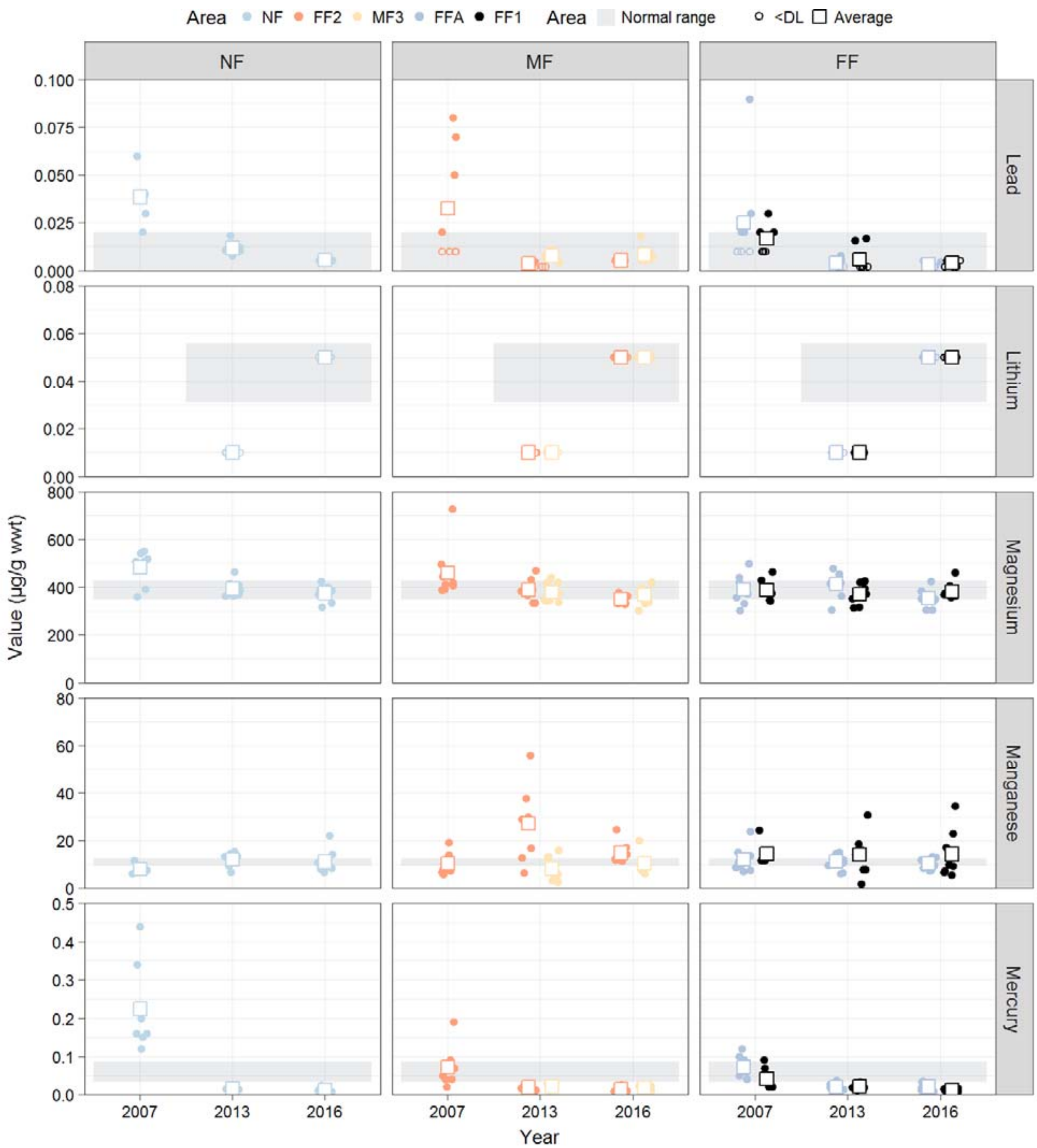


Figure 9-23 Temporal Trend Plot of Molybdenum, Nickel, Phosphorus, Potassium, and Rhenium Concentrations in Slimy Sculpin Tissue

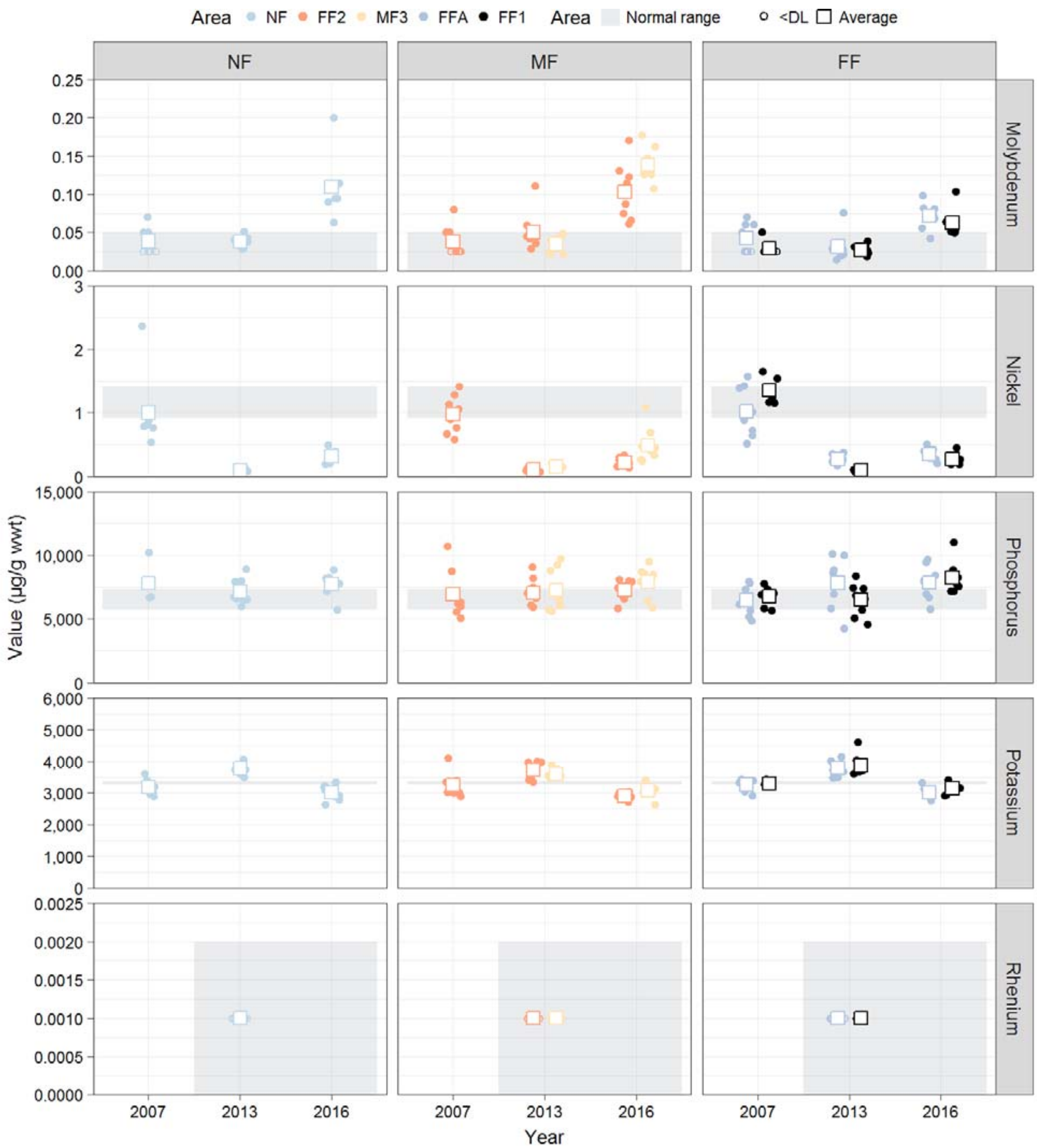


Figure 9-24 Temporal Trend Plot of Rubidium, Selenium, Silver, Sodium, and Strontium Concentrations in Slimy Sculpin Tissue

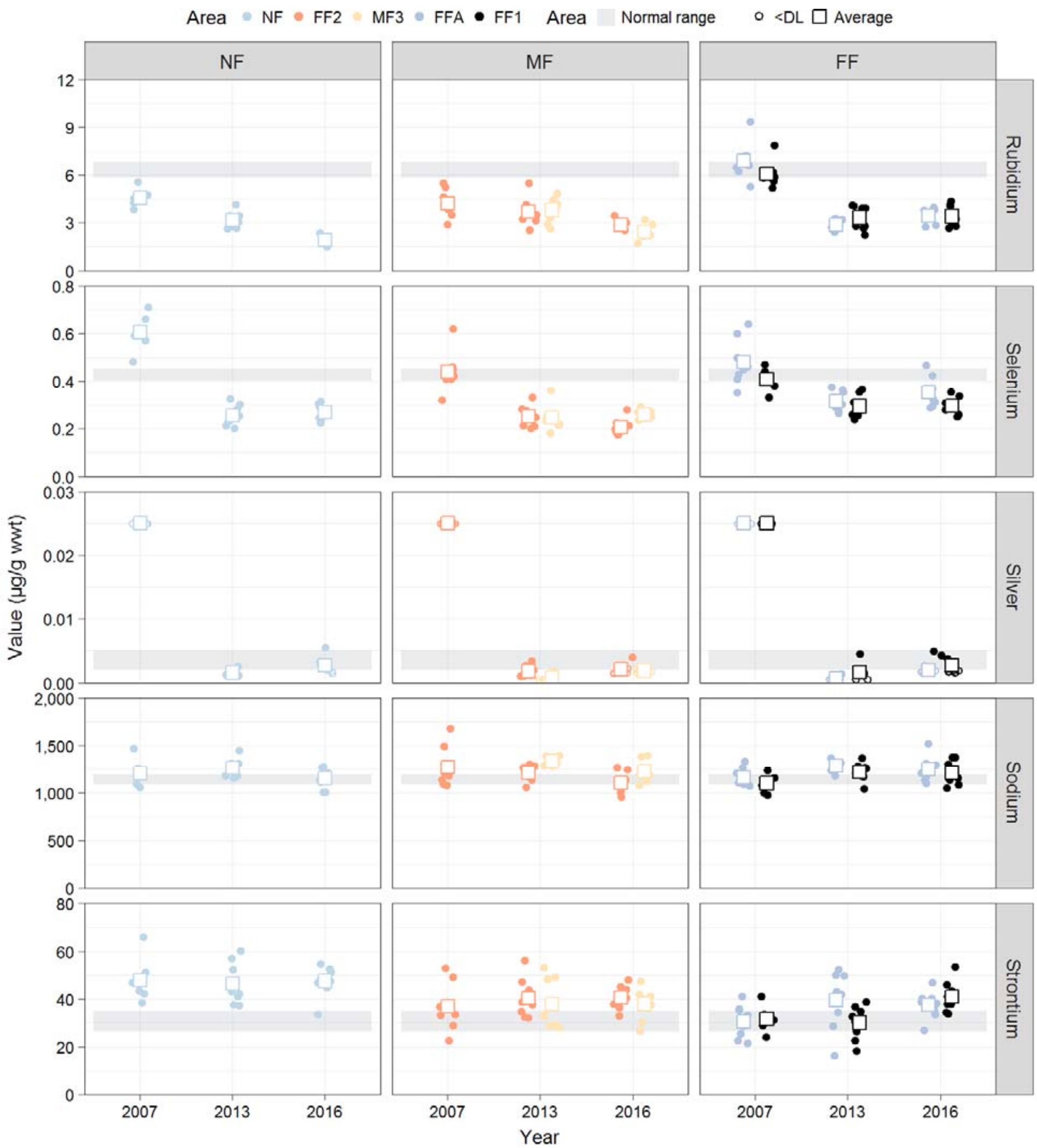


Figure 9-25 Temporal Trend Plot of Tellurium, Thallium, Thorium, Tin, and Titanium Concentrations in Slimy Sculpin Tissue

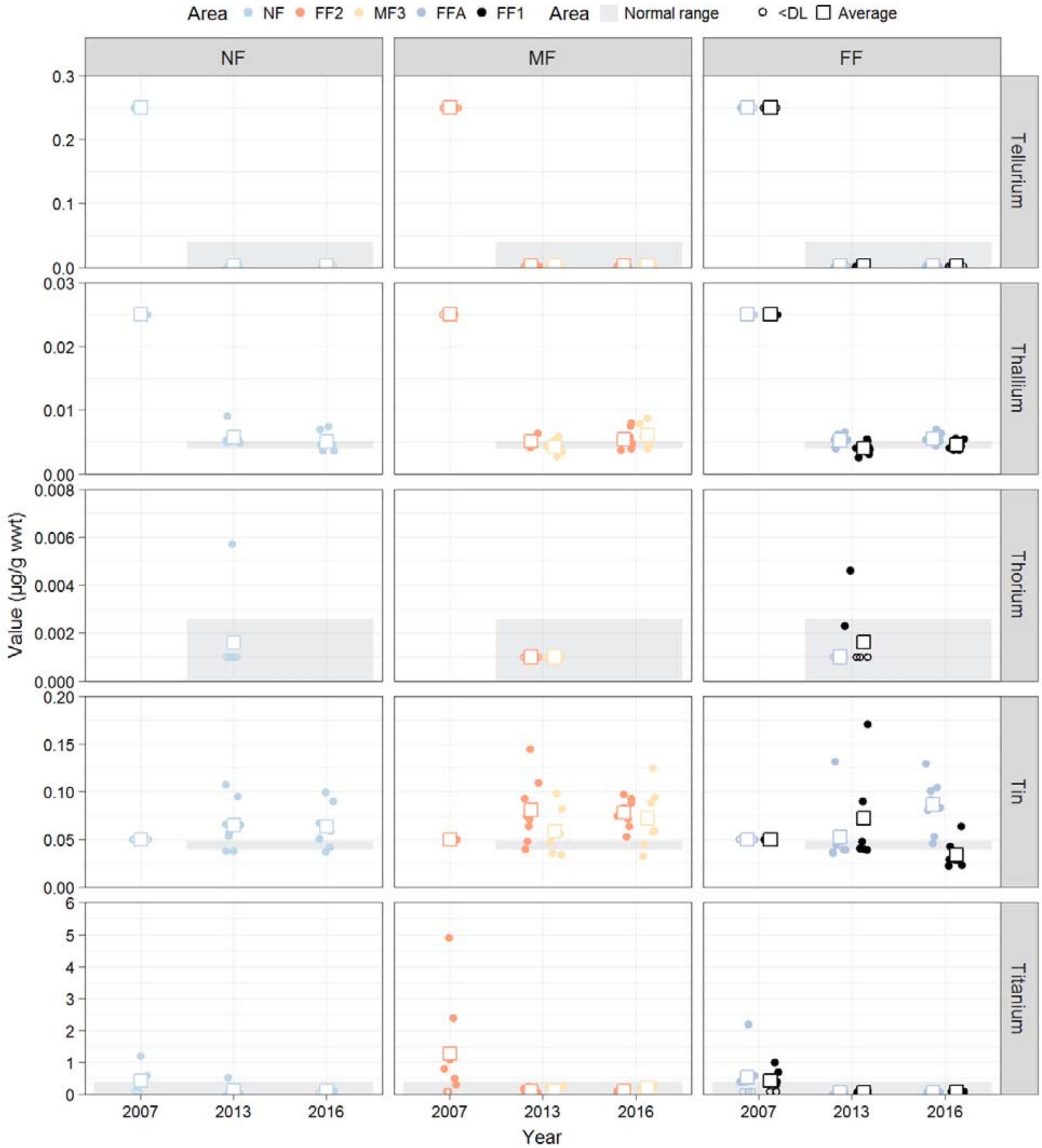
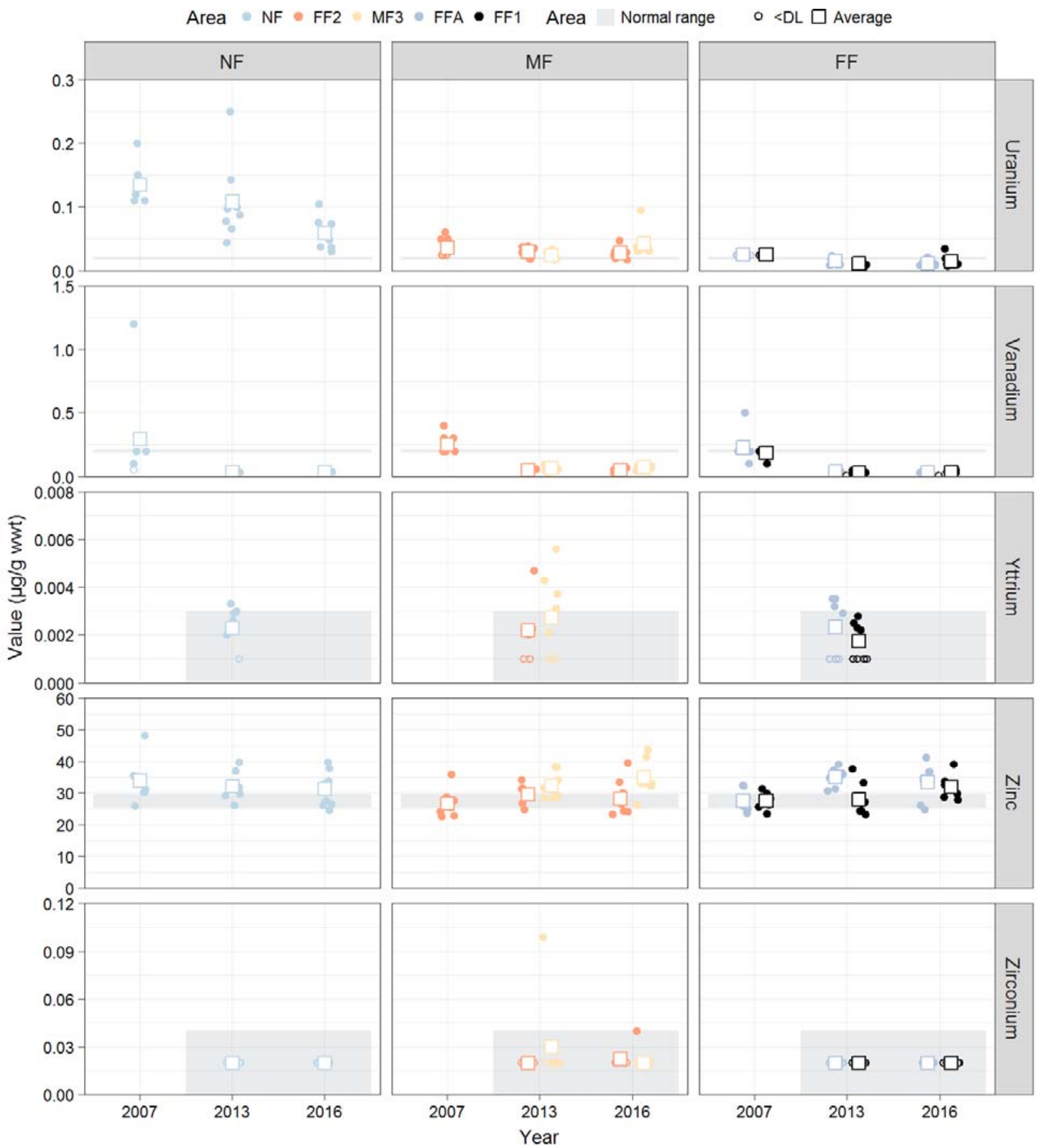


Figure 9-26 Temporal Trend Plot of Uranium, Vanadium, Yttrium, Zinc, and Zirconium Concentrations in Slimy Sculpin Tissue



9.3.2.2 Temporal Trends

Slimy Sculpin

The majority of tested variables had an interaction between year and area (Table 9-18). Of the variables that did not have a significant interaction, all had a significant effect of year, with the exception of calcium. Because calcium had no significant effects of either area or year, it is not discussed further. Many of the variables that had either a significant interaction or a significant year effect decreased over time and are, therefore, not considered further. These variables include arsenic, barium, magnesium, mercury, selenium, sodium, titanium, vanadium, and zinc (Table 9-19). Phosphorus and rubidium had significant, increasing trends in the FF area only (between 2013 and 2016).

Significant, increasing trends at NF and FF2 were estimated for cadmium, copper, iron, molybdenum, and nickel between 2013 and 2016, and for manganese and potassium between 2007 and 2013 (Table 9-19). Significant, increasing trends at MF3 were estimated for aluminum, cadmium, cesium, cobalt, copper, iron, molybdenum, nickel, and thallium between 2013 and 2016. Note that for cesium, molybdenum, thallium and tin, where 2007 data had higher DLs than subsequent years, the analysis was performed on 2013 and 2016 data only. In the FF, significant, increasing trends were estimated for cadmium, cesium, cobalt, copper, iron, molybdenum, nickel, phosphorus, potassium, rubidium, strontium. Of the variables with significant increasing trends, thallium exceeded the normal range at the NF area in 2013 and molybdenum exceeded the normal range at the NF and MF areas in 2016.

The significant increasing trend observed for cadmium, copper and iron from 2013 to 2016 may be due to an over-estimate as a result of the liver correction (see Table 9-15). However, given the uncertainty in the concentrations as a result of liver omission in 2016, these increases have been flagged herein, and will be confirmed during the next fish tissue chemistry program should they still be elevated relative to 2013 concentrations.

While strontium was flagged as being above the normal range at both the NF and MF areas for all years, a significant temporal increase was not observed. Strontium has remained at a similar concentration in the NF and MF areas since 2007. Uranium, the other variable flagged as being consistently above the normal range, decreased at the NF area, and increased at the MF area from 2013 to 2016. In all years, uranium followed the expected spatial gradient, with the greatest mean concentration at the NF area, and least at the FF areas; however, the temporal gradient appears to be showing a decreasing trend at the NF area over time. A detailed review of strontium and uranium potential effects was presented in the *2014 to 2016 AEMP Response Plan Fish – Supplemental Report* (Golder 2017e).

Figure 9-27 Temporal Trend Analysis of Transformed Aluminum, Arsenic, Barium, Cadmium, and Calcium Concentrations in Slimy Sculpin Tissue

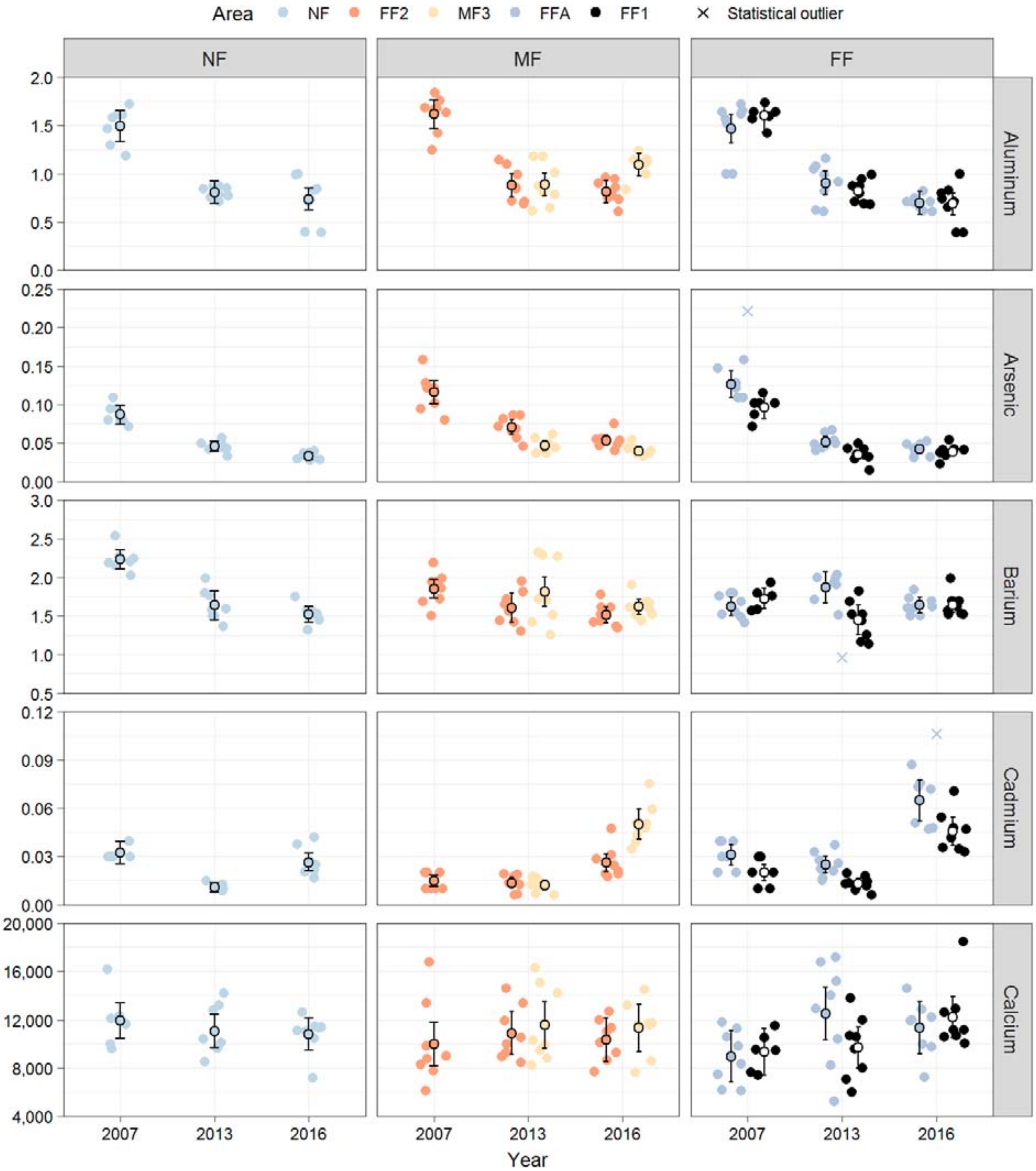


Figure 9-28 Temporal Trend Analysis of Transformed Cesium, Cobalt, Copper, Iron, and Magnesium Concentrations in Slimy Sculpin Tissue

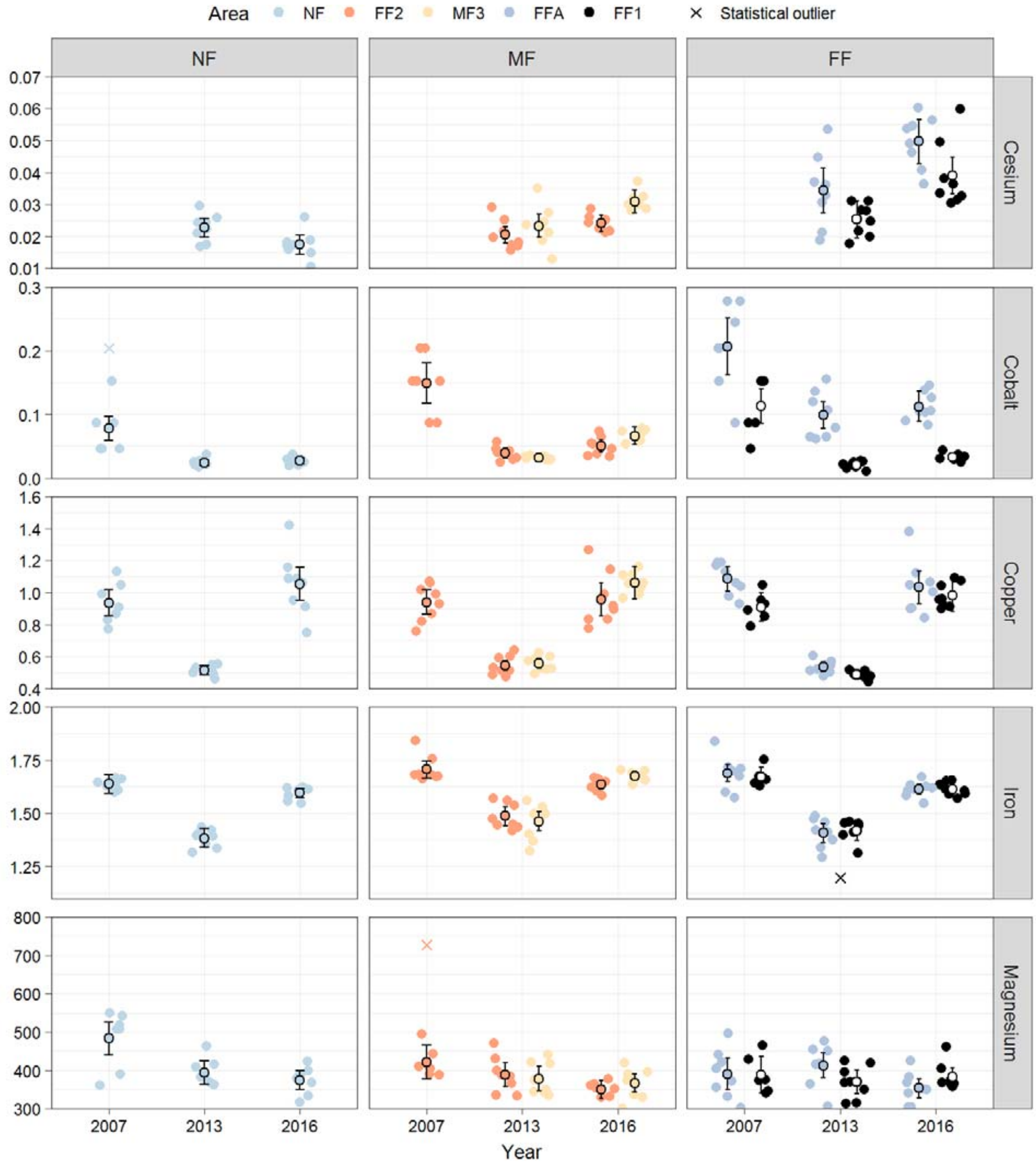


Figure 9-29 Temporal Trend Analysis of Transformed Manganese, Mercury, Molybdenum, Nickel, and Phosphorus Concentrations in Slimy Sculpin Tissue

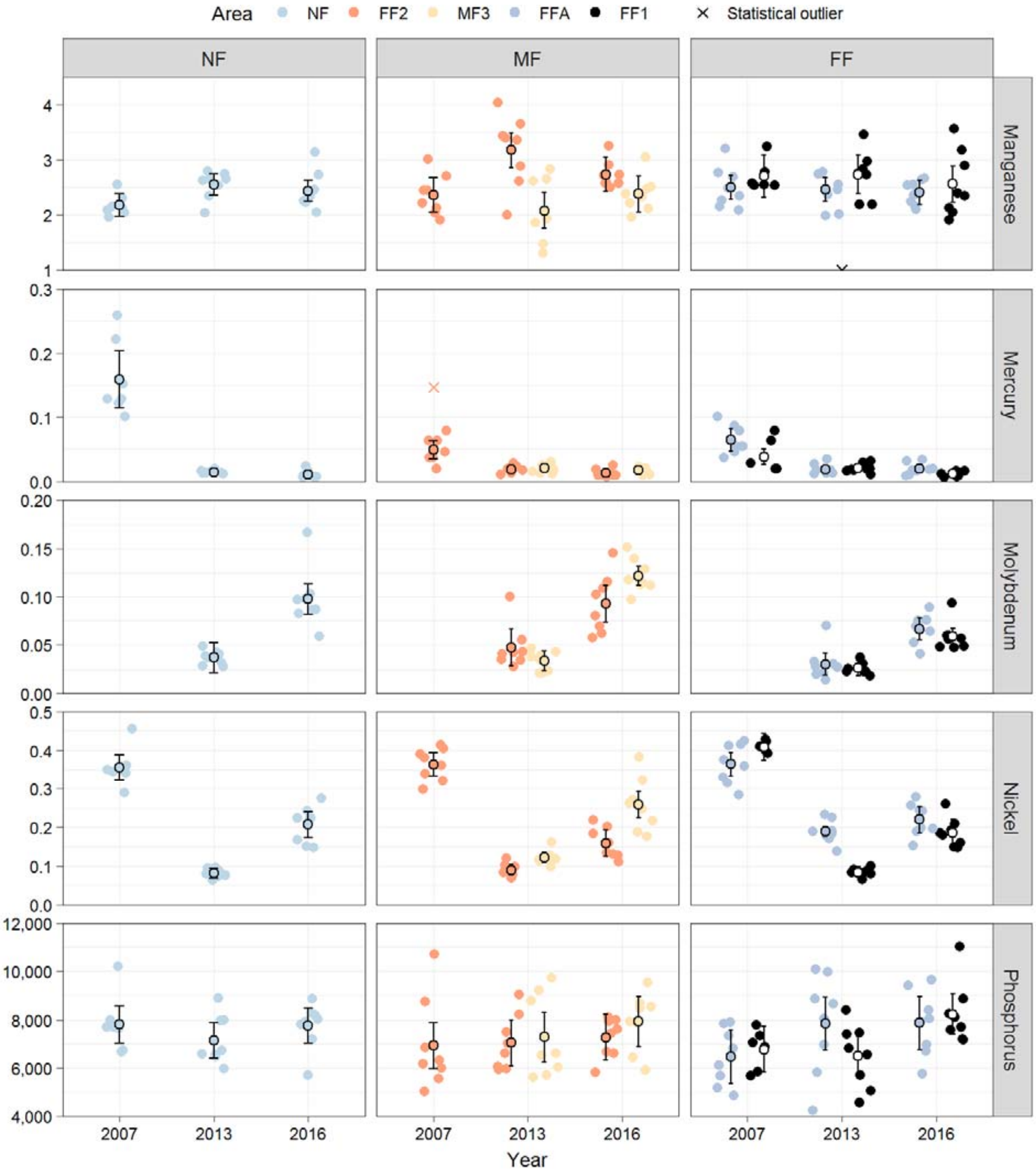


Figure 9-30 Temporal Trend Analysis of Transformed Potassium, Rubidium, Selenium, Sodium, and Strontium Concentrations in Slimy Sculpin Tissue

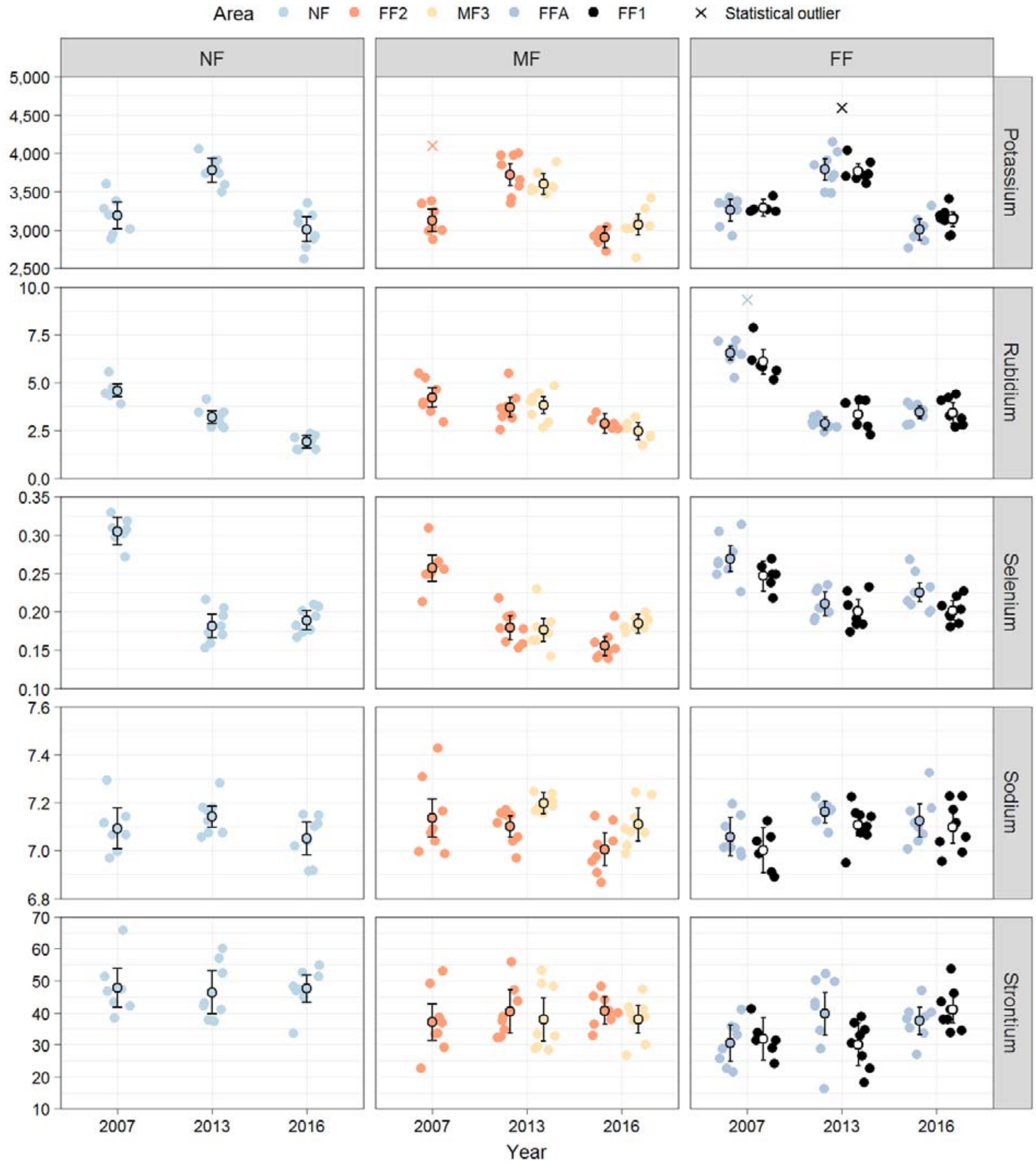


Figure 9-31 Temporal Trend Analysis of Transformed Thallium, Tin, Titanium, Uranium, and Vanadium Concentrations in Slimy Sculpin Tissue

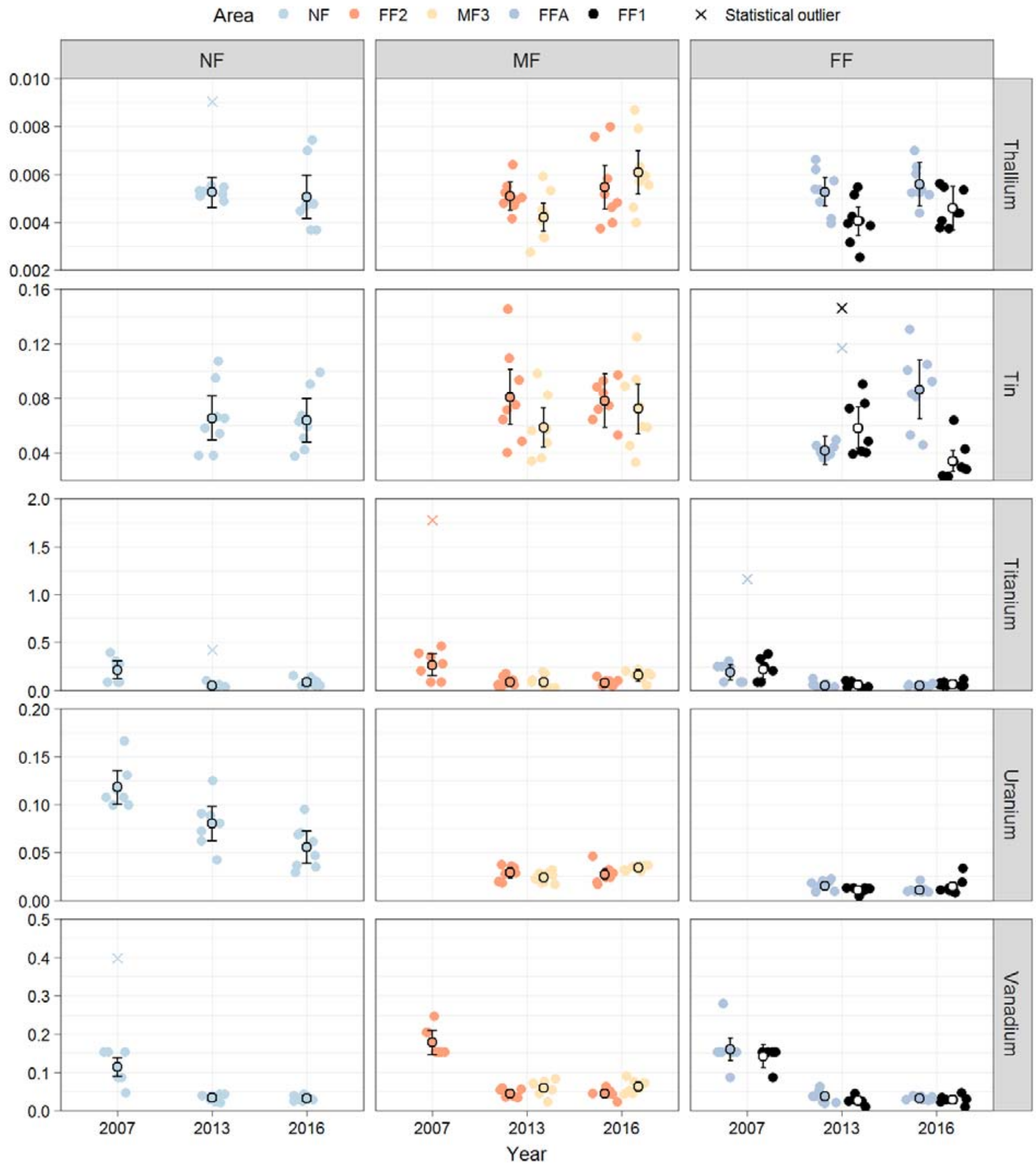


Figure 9-32 Temporal Trend Analysis of Transformed Zinc Concentrations in Slimy Sculpin Tissue

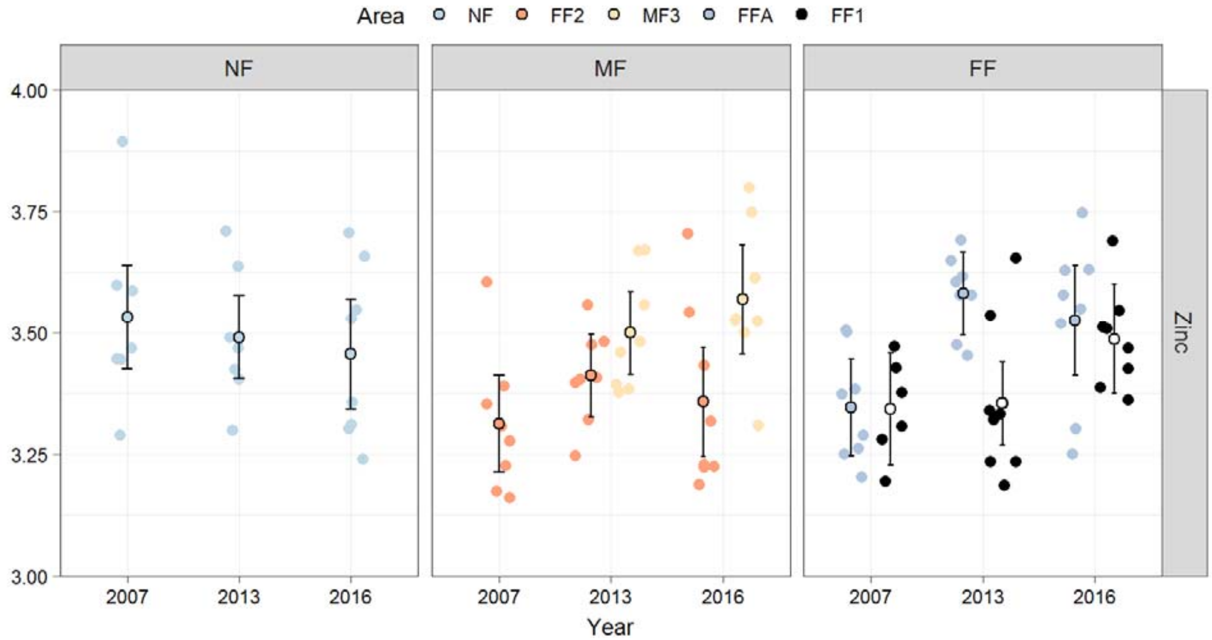


Table 9-18 Results of Trend Analysis of Slimy Sculpin Tissue Chemistry

Variable	Yeo-Johnson transformation[(a)]	Coefficient	Df	F value	*P* value
Aluminum	-0.5	Area	4	5.6	<0.001
		Year	2	141.5	<0.001
		Area × Year	7	2.3	**0.030**
Arsenic	-2	Area	4	16.9	<0.001
		Year	2	126.4	<0.001
		Area × Year	7	3.1	**0.005**
Barium	0	Area	4	2.6	0.040
		Year	2	25.8	<0.001
		Area × Year	7	8.9	**<0.001**
Cadmium	0	Area	4	14.3	<0.001
		Year	2	75.2	<0.001
		Area × Year	7	8.1	**<0.001**
Calcium	none	Area	4	0.6	0.668
		Year	2	0.7	0.504
		Area × Year	7	1.8	0.105

Table 9-18 Results of Trend Analysis of Slimy Sculpin Tissue Chemistry

Variable	Yeo-Johnson transformation[(a)]	Coefficient	Df	F value	*P* value
Cesium	0	Area	4	22.8	<0.001
		Year	1	8.7	0.004
		Area × Year	4	7.7	**<0.001**
Cobalt	-2	Area	4	42.0	<0.001
		Year	2	71.7	<0.001
		Area × Year	7	3.9	**0.001**
Copper	none	Area	4	4.4	0.003
		Year	2	394.4	<0.001
		Area × Year	7	1.6	0.147
Iron	-0.5	Area	4	11.6	**<0.001**
		Year	2	181.4	**<0.001**
		Area × Year	7	0.8	0.574
Magnesium	none	Area	4	1.2	0.330
		Year	2	10.9	<0.001
		Area × Year	7	2.4	**0.027**
Manganese	0	Area	4	5.3	0.001
		Year	2	3.7	0.030
		Area × Year	7	2.2	**0.039**
Mercury	-2	Area	4	2.8	0.030
		Year	2	42.0	<0.001
		Area × Year	7	4.7	**<0.001**
Molybdenum	-1	Area	4	17.7	<0.001
		Year	1	205.6	<0.001
		Area × Year	4	10.4	**<0.001**
Nickel	-2	Area	4	38.8	<0.001
		Year	2	480.4	<0.001
		Area × Year	7	8.1	**<0.001**
Phosphorus	none	Area	4	0.6	0.670
		Year	2	3.9	**0.023**
		Area × Year	7	1.3	0.259
Potassium	none	Area	4	2.5	**0.047**
		Year	2	147.1	**<0.001**
		Area × Year	7	1.0	0.441
Rubidium	none	Station/Area	4	16.2	<0.001
		Year	2	184.1	<0.001
		Area × Year	7	12.5	**<0.001**

Table 9-18 Results of Trend Analysis of Slimy Sculpin Tissue Chemistry

Variable	Yeo-Johnson transformation[(a)]	Coefficient	Df	F value	*P* value
Selenium	-2	Area	4	15.1	<0.001
		Year	2	115.3	<0.001
		Area × Year	7	6.3	**<0.001**
Sodium	0	Area	4	3.4	**0.012**
		Year	2	7.2	**0.001**
		Area × Year	7	1.7	0.110
Strontium	None	Area	4	8.9	**<0.001**
		Year	2	3.4	**0.038**
		Area × Year	7	1.4	0.233
Thallium	0	Area	4	3.52	0.011
		Year	1	5.88	**0.018**
		Area × Year	4	2.02	0.101
Tin	-1	Area	4	9.43	<0.001
		Year	1	0.05	0.819
		Area × Year	4	5.75	**<0.001**
Titanium	0	Area	4	4.6	**0.002**
		Year	2	21.6	**<0.001**
		Area × Year	7	0.9	0.519
Uranium	-1	Area	4	43.32	<0.001
		Year	2	12.36	<0.001
		Area × Year	4	6.01	**<0.001**
Vanadium	-2	Area	4	15.1	**<0.001**
		Year	2	113.8	**<0.001**
		Area × Year	7	1.4	0.204
Zinc	0	Area	4	5.0	0.001
		Year	2	3.2	0.045
		Area × Year	7	2.2	**0.044**

Notes: Log transformation was performed using Yeo-Johnson transformation, which uses an offset. *P*-values of significant Area × Year interactions and significant main effects (where no significant interactions were found) are shown in **bold**.

a) See Section 2.4.2.1 for details on transformation approach.

Table 9-19 Post-hoc Multiple Comparisons for Slimy Sculpin Tissue Chemistry, 2007 to 2016

Variable	NF			FF2			MF3			FFA			FF1		
	2007	2013	2016	2007	2013	2016	2007	2013	2016	2007	2013	2016	2007	2013	2016
Aluminum	b	a	a	b	a	a		a	b	c	b	a	b	a	a
Arsenic	c	b	a	c	b	a		a	a	b	a	a	b	a	a
Barium	b	a	a	b	ab	a		a	a	a	a	a	a	a	a
Cadmium	b	a	b	a	a	b		a	b	a	a	b	a	a	b
Cesium		b	a		a	a		a	b		a	b		a	b
Cobalt	b	a	a	b	a	a		a	b	b	a	a	c	a	b
Copper	b	a	b	b	a	b		a	b	b	a	b	b	a	b
Iron	b	a	b	c	a	b		a	b	c	a	b	b	a	b
Magnesium	b	a	a	b	ab	a		a	a	ab	b	a	a	a	a
Manganese	a	b	ab	a	b	ab		a	a	a	a	a	a	a	a
Mercury	b	a	a	b	a	a		a	a	b	a	a	c	b	a
Molybdenum		a	b		a	b		a	b		a	b		a	b
Nickel	c	a	b	c	a	b		a	b	b	a	a	c	a	b
Phosphorus	a	a	a	a	a	a		a	a	a	a	a	ab	a	b
Potassium	a	b	a	a	b	a		b	a	b	c	a	a	b	a
Rubidium	c	b	a	b	ab	a		b	a	c	a	b	b	a	a
Selenium	b	a	a	c	b	a		a	a	b	a	a	b	a	a
Sodium	a	a	a	b	ab	a		a	a	a	a	a	a	a	a
Strontium	a	a	a	a	a	a		a	a	a	a	a	ab	a	b
Thallium	-	a	a	-	a	a		a	b	-	a	a	-	a	a
Tin	-	a	a	-	a	a		a	a	-	a	b	-	b	a
Titanium	b	a	a	b	a	a		a	a	b	a	a	b	a	a
Uranium	-	b	a	-	a	a		a	b	-	a	a	-	a	a
Vanadium	b	a	a	b	a	a		a	a	b	a	a	b	a	a
Zinc	a	a	a	a	a	a		a	a	a	b	ab	a	a	a

Note: "-" means that data from that year/area were not included in the analysis of the variable. Different letters signify statistically significant differences (at the 0.05 level) of years within areas following post-hoc multiple comparisons. Grey shading denotes significant increasing trend.

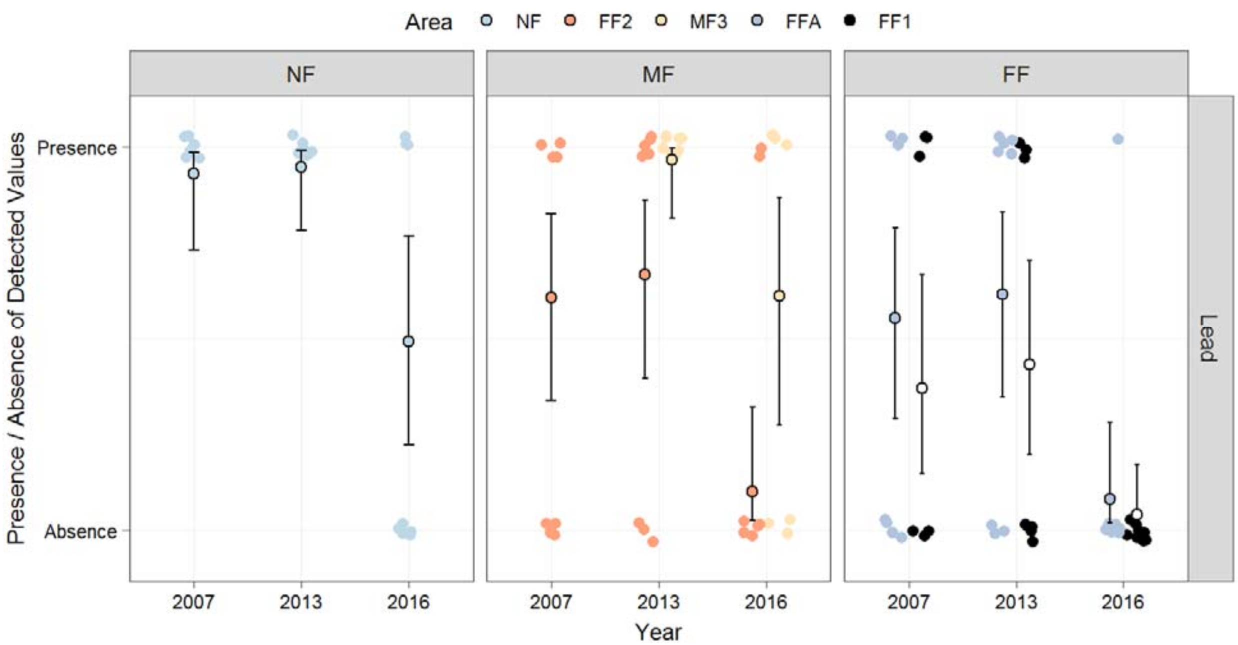
In the logistic regression of lead, the interaction between year and area was removed from the model due to convergence issues. The remaining main effects of year and area were both significant (Table 9-20). The probability of detected data was highest in the NF in 2007 and 2013 and in the MF in 2013 (Figure 9-33). The probability of detected data decreased between 2013 and 2016 in each area, leading to a significantly different estimate for 2016 than for either 2007 or 2013, and suggesting an overall trend of decreasing concentrations. This interpretation of the result is supported by the measured lead concentrations (Figure 9-22), which showed a decline between 2007 and 2013.

Table 9-20 Results of Logistic Trend Analysis of Lead in Slimy Sculpin

Variable	Coefficient	Degrees of Freedom	Chi Squared	*P*-value
Lead	Area	4	26.2	**<0.001**
	Year	2	26.8	**<0.001**

Note: **Bold text** = *P*-value significant.

Figure 9-33 Presence/Absence of Detected Lead Concentrations and Predicted Probabilities in Lac de Gras Slimy Sculpin

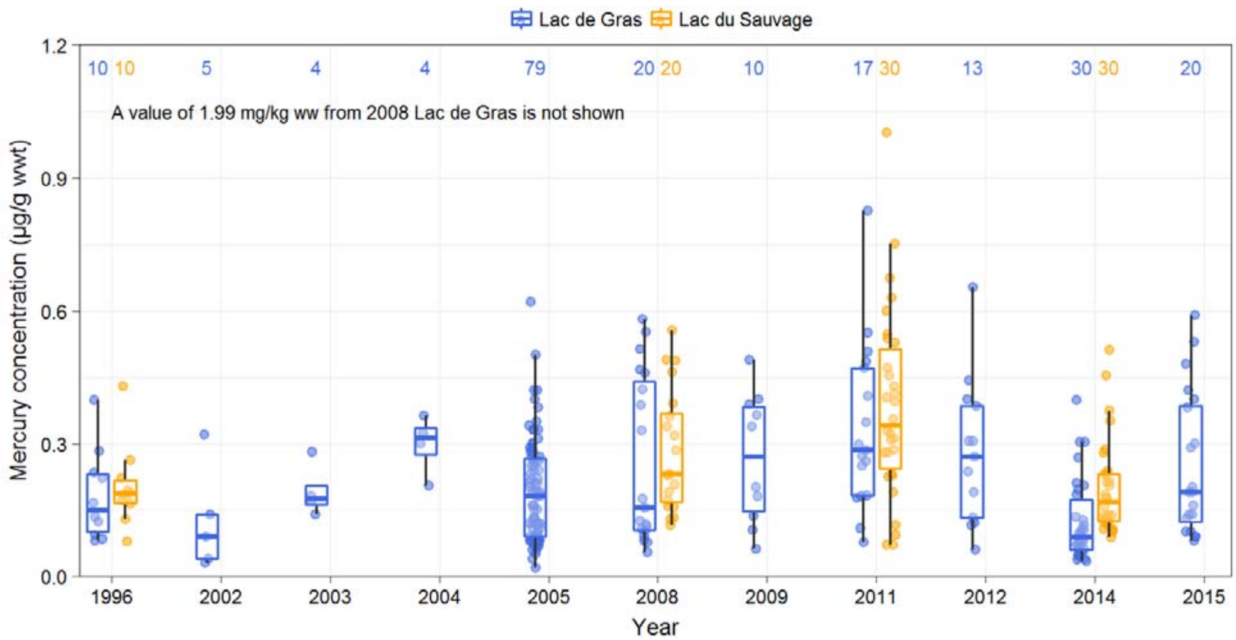


Note: Points with error bars represent predicted probabilities and 95% confidence intervals.

Lake Trout

On the basis of the Lake Trout data from 1996, 2005, 2008, and 2011, concentrations of mercury in fish muscle showed an increasing pattern from 2008 to 2011, with the increase being more pronounced in Lac du Sauvage (Figure 9-34). In 2014, mercury was detected at near or below baseline concentrations in both Lac de Gras and Lac du Sauvage. Because of differences in field and analytical methods, and the differing body sizes of fish within the dataset, these results were considered further.

Figure 9-34 Mercury Concentrations in Lake Trout Tissue from Lac de Gras and Lac du Sauvage between 1996 and 2015



Note: As outlined in the methods, fish were collected by different field and analytical methods and of different body sizes but the pooled data is shown here for the sake of completeness. Sample sizes for each year are provided at the top of the figure.

Of the five examined models, the model that included lake, year, fork length, and an interaction between lake and year had the best support (Table 9-21). All model coefficients were significant (Table 9-22). Raw mercury concentrations in samples collected from Lac de Gras in 2015 were overall greater than those in samples collected in 2014 (Figure 9-34). However, when fish length was accounted for in the analysis, mercury concentrations were generally similar between the two years (see 2014 and 2015 values in Figure 9-35).

Table 9-21 Model Selection for Analysis of Mercury Concentrations in Lake Trout

Model #	Predictors	AICc	ΔAICc
1	Lake + Year + Fork Length + Lake×Year	-179	0
2	Lake + Year + Fork Length	-176	3
4	Year	-159	20
3	Lake+ Fork Length	-130	50
5	Fork Length	-92	88

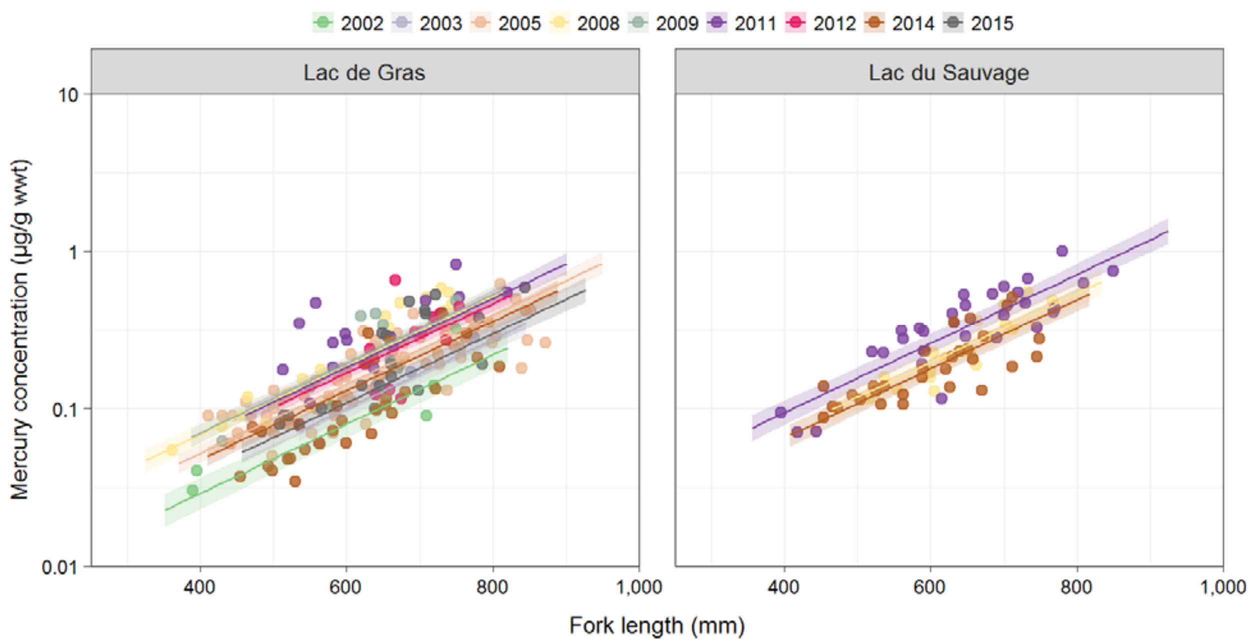
AIC = Akaike's information criterion.

Table 9-22 ANOVA Table of Model 3 that was Selected for Interpretation of Lake Trout Mercury Concentrations

Coefficient	Df	F-value	*P*-value
Lake	1	19.5	**<0.001**
Year	2	27.7	**<0.001**
Fork length	1	475.6	**<0.001**
Lake×Year	2	3.8	0.023

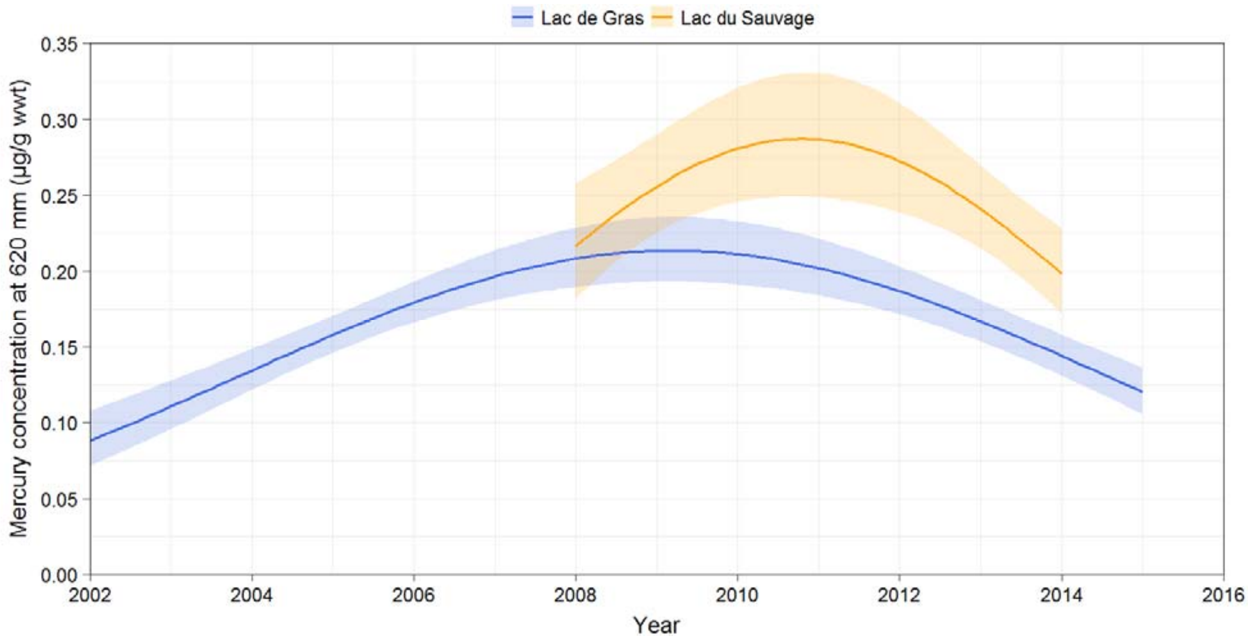
Note: **Bold text =** *P*-value significant.

Figure 9-35 Lake Trout Mercury Concentrations Over Time: Predicted Means and 95% Confidence Intervals



Similar to the general pattern shown in Figure 9-34, the overall temporal trend in Lake Trout mercury concentration reflected the increase in adjusted mercury between 2002 and 2009/2011, and the decrease in mercury following 2011 (Figure 9-36). Lac du Sauvage Lake Trout were estimated to have greater mean adjusted mercury concentrations in 2011 and 2014 compared to Lac de Gras.

Figure 9-36 Predicted Mercury Concentrations for a Lake Trout of 620 mm



Note: Ribbons are 95% confidence intervals.

In 2014, it was determined that Lake Trout were crossing the narrows between Lac de Gras and Lac du Sauvage (Golder 2015b), indicating mixing between the Lake Trout populations of the two lakes. As a result, there was no reason to account for lake effect in the analyses. However, the modeling results indicated there was a significant difference between the two lakes, implying that despite any fish movement that may be occurring between the two lakes, there was a lake effect. As such, the Lac de Gras and Lac du Sauvage data were evaluated separately.

The model assessing differences in mercury concentrations between Lac du Sauvage and Lac de Gras in 2008, 2011, and 2014 had a significant Year × Lake interaction (Table 9-23), suggesting significant differences in length-adjusted mercury concencentrations between the two lakes in at least one year. Pairwise comparisons between the two lakes in each of the three sampling years indicated no significant differences in 2008 and 2011 (*P*-values of 0.5 for both years) and a significant difference between the two lakes in 2014 (*P*-value <0.001). The reason for the significant difference between Lac du Sauvage and Lac de Gras has not been explored.

Table 9-23 ANOVA table of the Model Assessing Lac du Sauvage vs Lac de Gras Mercury Concentration in Lake Trout (2008, 2011, and 2014 only)

Coefficient	Degrees of Freedom	*F*-value	*P*-value
Lake	1	22.9	**<0.001**
Fork length	1	313.9	**<0.001**
Year	2	44.8	**<0.001**
Year × Lake	2	15.6	**<0.001**

Note: **Bold text** = *P*-value significant.

9.3.2.3 Guideline Comparison

The concentration of mercury in Lake Trout was above the CFIA guideline of 0.5 µg/g wwt in longer, older fish in both Lac de Gras and Lac de Sauvage (see Figure 9-34). The concentration of mercury in Lake Trout muscle was below a conservative effect threshold/tissue guideline (1.0 µg/g wwt) in all captured fish, such that Lake Trout health is unlikely to be affected; there was one exception in 2008, where one fish had a concentration of 1.99 µg/g wwt in Lac de Gras. On the basis of the most recent mercury concentrations in Lake Trout from each lake (2014), no concerns to human health or fish health are expected.

9.3.2.4 Weight-of-Evidence Ratings

The WOE ratings for fish tissue chemistry results (including 2010) are presented in Table 9-24. Three changes were made to the WOE summary table from those listed in the annual reports: two for corrections and one for the liver re-classification outlined in this report. More specifically, where there were corrections identified in the annual AEMP reports in the WOE tables through the WLWB review process, the corrected ratings were used herein.

- 2016: In the WOE Appendix XV of the 2016 Annual AEMP (Golder 2017c), metals that showed a statistically significant decrease in the NF area were classified as having a low effect rating (arsenic, cadmium, cesium, cobalt, rubidium and selenium). This was corrected because only a statistically significant increase receives a low effect rating, and as such these should have been classified as 'no response'.
- 2010: In 2010, silver and thorium were observed to be increasing but were assigned a low level effect rating as there was no indication that these two variables were linked to the Mine. This was corrected because linkage to the Mine is not currently a specific component of the WOE effect ratings. These two variables were re-assigned to a 'moderate level effect rating' in Table 9-11.
- 2016: When the liver correction factor was applied to molybdenum (See Section 9.3.1.2.1), the corrected NF and MF area means exceeded the normal range. Given this, the previous low effect level rating was re-assigned to a high level effect rating.

Overall, mean molybdenum, strontium and uranium concentrations in Slimy Sculpin collected from the NF area in 2016 were statistically greater than those in the FF areas. All three of these variables also exceeded the normal range at the MF area. As such, molybdenum, strontium and uranium were assigned a high level effect for 2016. High level effects were also assigned to strontium in 2007, strontium and uranium in 2013, and a moderate level effect was assigned to molybdenum in 2010. These three metals were further evaluated as part of the *2014 to 2016 AEMP Response Plan Fish – Supplemental Report* (Golder 2017e).

Other variables had moderate or high level effect ratings in 2007 or 2010, but not in 2013 and 2016: magnesium, mercury and selenium in 2007; aluminum, lithium, silver, thorium titanium and yttrium in 2010; and barium and lead in both 2007 and 2010. No response was detected in lead in 2016, despite a high level effect in 2007, a moderate level effect in 2010, and a low level effect in 2013. Thallium showed a moderate level effect in 2013, but no response in 2016. Bismuth showed a low level effect in 2010 and a high level effect in 2013; however, no response was observed in 2016.

Table 9-24 Weight-of-Evidence Effect Ratings for Fish Tissue Chemistry

Variable	2007	2010	2013	2016
Aluminum	0	↑↑	0	0
Antimony	n/d	n/d	n/d	n/d
Arsenic	0	0	0	0
Barium	↑↑	↑↑	0	0
Beryllium	n/d	n/d	n/d	n/d
Bismuth	n/d	↑[(a)]	↑↑ [(b)]	n/d
Boron	n/d	n/d	n/d	n/d
Cadmium	0	0	0	0
Calcium	0	0	0	0
Cesium	0	0	0	0
Chromium	0	0	0	n/d
Cobalt	0	0	0	0
Copper	0	0	0	0
Gallium	n/a	n/d	0	n/a
Iron	0	0	0	0
Lead	↑↑	↑↑	↑	n/d
Lithium	n/a	↑↑	n/d	n/d
Magnesium	↑↑	0	0	0
Manganese	0	0	0	0
Mercury	↑↑	0	0	0
Molybdenum	n/d	↑↑	0	↑↑↑
Nickel	0	0	0	0
Phosphorus	0	0	0	0
Potassium	0	0	0	0
Rhenium	n/a	n/d	0	n/a
Rubidium	0	0	0	0
Selenium	↑↑	0	0	0
Silver	n/d	↑↑	0	n/d
Sodium	0	0	0	0
Strontium	↑↑	↑↑	↑↑	↑↑↑
Tellurium	n/d	n/d	n/d	n/d
Thallium	n/d	0	↑↑	0
Thorium	n/a	↑↑	n/a	n/a
Tin	n/d	0	0	0
Titanium	0	↑↑	0	0
Uranium	↑[(a)]	↑↑	↑↑	↑↑↑
Vanadium	0	0	0	0
Yttrium	n/a	↑↑	0	n/a
Zinc	0	0	0	0
Zirconium	n/a	n/d	n/d	n/d

Notes: Effects ratings for exposure endpoints only consider increases in concentrations in the near-field area relative to far-field areas. A decrease in concentration for a given variable is considered a "no response". Statistical analyses are presented in Golder 2008, 2011c, and 2017f.

0 = no response; ↑ = early warning low-level rating; ↑↑ = moderate-level rating; ↑↑↑ = high-level rating; n/a = not applicable as variable was not measured; n/d = not determined, at least one area with >60% values <DL.

a) Normal range not determined in the Reference Conditions Report Version 1.2 (Golder 2017b).

b) Statistical comparisons could not be completed as >60% of values were <DL in the FF areas; however, based on a visual assessment, a substantial difference between the NF/MF and FF areas were assumed.

9.4 Comparison to EA Predictions

The EA had one prediction regarding fish quality which relates to fish tissue. The EA predicted that mercury concentration in sport and subsistence fisheries would remain below a mean of 0.2 µg/g wwt (unadjusted for length). The prediction was exceeded in Lac de Gras in 2004, from 2008 to 2012 and again in 2015 (Table 9-25).

Table 9-25 Unadjusted Lake Trout Mercury Concentration in Lac de Gras, 1996 to 2015

Year	Mean and Range of Unadjusted Mercury Concentration (µg/g wwt)
1996	0.18 (0.08 - 0.4)
2002	0.12 (0.03 - 0.32)
2003	0.19 (0.14 - 0.28)
2004	0.3 (0.2 - 0.36)
2005	0.19 (0.02 - 0.62)
2008	0.34 (0.05 - 1.99)
2009	0.27 (0.06 - 0.49)
2011	0.33 (0.08 - 0.83)
2012	0.28 (0.06 - 0.65)
2014	0.12 (0.03 - 0.4)
2015	0.25 (0.08 – 0.59)

Notes: Values presented are means with minimum and maximum values in parentheses.

9.5 Summary and Conclusions

9.5.1 Fish Health

Habitat

Temporal variability in recorded water temperatures confounded interpretation of the influence of water temperature on Slimy Sculpin growth. For example, while FF1 and FFA temperatures were generally greater than temperatures at other sampling areas between the middle of June and late July in both 2013 and 2016, they were comparable to water temperatures in other areas throughout August and September. It is likely that variability in water temperatures among sampling areas influenced fish growth, and cooler temperatures at the NF in mid-July relative to the other sampling areas could have reduced growth in these fish and contributed to the differences observed in fish size among study areas. However, the possibility that the inconsistent depth of data loggers also attributed to the differences in recorded temperatures cannot be ruled out.

The high level comparison of 2007, 2013, and 2016 substrate data indicated little difference in the dominant substrate type among areas. The low resolution of these data from the early years of the fish program (i.e.,

2007) precluded any meaningful interpretation of these data and whether or not substrate differences were contributing to the differences observed among areas. Beginning in 2013, a more detailed method of determining substrate type was implemented, and as more years of data are collected using the improved method, the influence of substrate type will be teased apart further to help distinguish Mine-related versus habitat effects.

Parasitism

Parasitism levels did not follow an expected spatial gradient of response consistent with a Mine influence, as FF areas had the greatest proportion of parasites in two of the four sampling years (i.e., 2010 and 2013). While parasitism likely affects fish growth, it appears unlikely that parasitism is primarily linked to Mine effects.

Fish Health Effects

Patterns in fish health variables from 2007 to 2016 are summarized in Table 9-26, which examinies if (1) a significant trend was present over time, (2) if a variable was within the normal range, (3) whether the magnitude of difference between the NF and FF areas was greater than the CES as defined in the MMER TGD (Environment Canada 2012), and (4) whether or not a spatial gradient was observed (i.e., NF>MF>FF) suggestive of an effect of the Mine.

In Age-1+ fish, body size (i.e., total length, total weight, and carcass weight) in the NF and sometimes MF areas was smaller than those in the FF areas in recent years (Table 9-26). The magnitude of effect was greater than the CES for fresh weight and carcass weight; however, all three variables remained within normal ranges. Analysis of 2007, 2013, and 2016 data across sampling areas indicated no significant decrease between NF and FF areas over time, but there was a significant increase at the FF areas from 2013 to 2016. Therefore, we conclude that while fish size was demonstrating a recent decrease in body size spatially (i.e., NF fish are smaller than FF fish), there did not appear to be a decrease in body size over time at the NF area. The difference in fish size between the NF and FF area may actually have been due to the recent increase in body size at the FF areas (e.g., an increasing tendency in Age-1+ body size at the FF area versus a relatively stable total length at NF).

Condition and relative liver size for Age-1+ fish did not show any spatial or temporal patterns. However, there was an exceedance of the normal range for condition in 2007 due to the presence of two age-1 fish with high condition.

In adult males, body size (i.e., total length, fresh weight, carcass weight) at the NF was smaller than at the FF in recent years (Table 9-26).The magnitudes of effect were below the CES, and all body size variables remained within the normal ranges. Analysis of the 2007, 2013 and 2016 data across sampling areas indicated a significant decrease in body size at both the MF area (FF2) and FF area (FFA) from 2007 to 2013, and a significant increase from 2013 to 2016. The increase at the FF area may in part be driving the recent differences between the NF and FF areas for body size.

For adult males in 2016, a comparison to the liver CES could not be made (Golder 2017c). However, based on a visual assessment of spatial pattern, it appears there was a decrease in liver size in 2016. The analysis of 2007, 2013 and 2016 data across sampling years suggested no temporal gradients. Condition and relative gonad weight did not exhibit any spatial or temporal gradients, were all within the respective normal range, and were below the CES.

In adult females, body size (i.e., total length, total weight, and carcass weight) in the NF and sometimes MF areas was smaller than in the FF areas in recent years (Table 9-26). The magnitudes of effect were above the CES for total and carcass weights in 2016; however, all three variables were within their respective normal ranges. Analysis of the 2007, 2013 and 2016 data across sampling areas suggested a temporal increase in body size in the FF areas in recent years, but no spatial gradient was observed. Condition, relative liver size and relative gonad size for adult females did not exhibit any spatial or temporal gradients, were all within the respective normal range, and were below the CES.

Action Level Assessment

In 2016, Action Level 2 was triggered for fish weight, fish length, and relative liver size. An Action Level 2 is reached when a statistical difference between the MF and FF areas is reported and is indicative of changes that could be a toxicological response. Prior to the stage re-classification of fish, it was reported that an Action Level 1 was triggered in 2013. The cause of the observed toxicological effects for fish health was investigated as part of the *2014 to 2016 AEMP Response Plan Fish - Supplemental Report* (Golder 2017e), and this Response Plan was recently approved (WLWB 2018).

Table 9-26 Fish Health Summary, 2007 to 2013

Sex/Stage	Effect	Total Length	Fresh Weight	Carcass Weight	Condition	LSI	GSI
Age-1+	What is the pattern across the gradient? (NF → MF → FF)	2007: ↓ 2013: 0 2016: ↓	2007: ↓ 2013: ↓ 2016: ↓	2007: ↓ 2013: ↑ 2016: ↓	2007: ↑ 2013: ↓ 2016: ↑	2007: 0 2013: 0 2016: ↑	---
	Within Normal Range?	Yes	Yes	Yes	No (NF – 2007)	Yes	---
	Exceedance of CES in 2016?	No[a]	Yes (NF)	Yes (NF)	No	No	---
	What is the pattern over time?	• NF stable • MF stable • FF increasing tendency	• NF stable • MF stable • FF increasing tendency	• NF stable • MF stable • FF increasing tendency	• NF stable • MF stable • FF stable	• NF stable • MF stable • FF stable	---
Adult male	What is the pattern across the gradient? (NF → MF → FF)	2007: 0 2013: 0 2016: ↓	2007: ↓ 2013: 0 2016: ↓	2007: ↓ 2013: 0 2016: ↓	2007: ↓ 2013: 0 2016: 0	2007: 0 2013: 0 2016: 0	2007: ↓ 2013: 0 2016: ↓
	Within Normal Range?	Yes	Yes	Yes	Yes	Yes	Yes
	Exceedance of CES in 2016?	No[a]	No	No	No	Area specific interpretation not possible due to significant interaction	No
	What is the pattern over time and space?	• NF stable • MF increasing tendency • FF variable, recent increasing tendency	• NF stable • MF variable, recent increasing tendency • FF variable, recent increasing tendency	• NF stable • MF variable, recent increasing tendency • FF variable, recent increasing tendency	• All areas stable	• All areas stable	• All areas stable

Table 9-26 Fish Health Summary, 2007 to 2013

Sex/Stage	Effect	Total Length	Fresh Weight	Carcass Weight	Condition	LSI	GSI
Adult female	What is the pattern across the gradient? (NF → MF → FF)	2007: 0 2013: ↓ 2016:↓	2007: 0 2013: ↓ 2016: ↓	2007: ↓ 2013: ↓ 2016:↓	2007: 0 2013: 0 2016: 0	2007: 0 2013: ↓ 2016: 0	2007: 0 2013: ↓ 2016: 0
	Within Normal Range?	Yes	Yes	Yes	Yes	Yes	Yes
	Exceedance of CES in 2016?	No[a]	Yes (MF), however larger exceedance observed at FF	Yes (NF, MF); however, larger exceedance observed at FF	No	No	No
	What is the pattern over time and space?	• NF recently stable • MF recently stable • FF1 stable, recent increasing tendency at FFA	• NF recently stable • MF recently stable • FF1 stable, recent increasing tendency at FFA	• NF recently stable • MF recently stable • FF1 stable, recent increasing tendency at FFA	• All areas stable	• All areas stable	• All areas stable

a) No CES is defined for length in the MMER TGD (Environment Canada 2012); therefore, CES for weight (25%) was applied.

↓↑ = visual estimate of spatial gradient, ↓ indicates NF<MF<FF; ↑ indicates NF>MF>FF, arrow indicates direction of difference; ; 0 = no spatial gradient; NF = near-field; MF = mid-field; FF = far-field; CES = critical effect size; n/d = not determined.

9.5.2 Fish Tissue

9.5.2.1 Slimy Sculpin

Fish tissue monitoring in Slimy Sculpin is sensitive to changes in concentrations in fish tissue. The sample size of eight composite fish with similar size and sex is based on the recommendations of the MMER TGD (Environment Canada 2012) and appears successful at detecting change between areas. Patterns in fish tissue from 2007 to 2016 are summarized in Table 9-27, examining if (1) a significant upward trend was present over time, (2) what the direction was relative to the normal range (above, below or within), and (3) whether or not a gradient in space was observed (i.e., NF>MF>FF) suggestive of an effect of the Mine.

The conclusions of the re-evaluation of the Slimy Sculpin fish tissue are:

- Patterns in fish tissue concentrations were highly variable (i.e., increasing, decreasing, or stable).
- Few variables were increasing in the NF area and MF/FF areas, were outside of normal range, and showed a trend from high to low from the NF area to the FF area (i.e., demonstrated a gradient response). Molybdenum and uranium were exceptions, where a gradient suggestive of Mine influence (i.e., NF>MF>FF) was observed.
- A few variables appeared to be increasing at MF3 that were not increasing in the NF area, suggesting an alternate influence (e.g., dike construction) rather than effluent may be contributing to changes in fish tissue chemistry.
- The FF areas exhibited some variability not seen in the NF or MF areas.
- There was no statistically significant increasing trend in mercury observed in Slimy Sculpin between 2013 and 2016, and mercury was not outside of the normal range.
- Continued monitoring of fish tissue to support the understanding of the exposure of small-bodied fish to potential Mine-effects is warranted.

Table 9-27 Summary of Slimy Sculpin Fish Tissue Key Results from 2007 to 2016

Variable	Slimy Sculpin Tissue		
	Statistically Significant Upward Trend over Time	2016 Mean Relative to Normal Range	Gradient in space in 2016 (NF >MF >FF)
Aluminum	No except for MF3 from 2013 to 2016	↓	No
Antimony	-	In for all except ↑ for FF1	No
Arsenic	No	↓	No
Barium	No	In or below at NF and MF	No
Beryllium	-	In for all	No
Bismuth	-	In for all except ↑ for MF3	No
Boron	-	In	No
Cadmium	Yes, all areas from 2013 to 2016	In (NF), in or ↑ at MF, ↑ at FF	No
Calcium	-	↑ at all except FF2	No
Cesium	Yes in MF and FF, but not NF	In	No

Table 9-27 Summary of Slimy Sculpin Fish Tissue Key Results from 2007 to 2016

Variable	Slimy Sculpin Tissue		
	Statistically Significant Upward Trend over Time	2016 Mean Relative to Normal Range	Gradient in space in 2016 (NF >MF >FF)
Chromium	-	↓	No
Cobalt	Yes in MF3 and FF1 but not NF	↓ at all except FFA	No
Copper	Yes, all areas from 2013 to 2016	In	No
Iron	Yes, all areas from 2013 to 2016	↓ NF, FF, in at MF	No
Lead	No	In	Yes
Magnesium	No	In	No
Manganese	Yes in NF and FF2 from 2007 to 2013	In (NF), in or ↑ at MF, in or ↑ at FF	No
Mercury	No	↓	No
Molybdenum	Yes, all areas from 2013 to 2016	↑	Yes
Nickel	Yes, all areas from 2013 to 2016, except for FFA	↓	No
Phosphorus	No, except for FF1 from 2013 from 2016	↑ at all except FF2	No
Potassium	Yes in all areas from 2007 to 2013, but declined in 2016	↓	No
Rubidium	No, except FFA from 2013 to 2016	↓	No
Selenium	No	↓	No
Silver	-	In all except ↓ at MF and FFA	No
Sodium	No	In all except ↑ at MF3 and FFA	No
Strontium	No, except for FF1 from 2013 to 2016	↑	No
Tellurium	-	In	No
Thallium	No except for MF3 from 2013 to 2016	↑ in all except in for FF1	No
Tin	No except for FFA from 2013 to 2016	↑ except ↓ at FF1	No
Titanium	No	In	No
Uranium	No except MF3 from 2013 to 2016	↑ at NF and MF, ↓ at FF	Yes
Vanadium	No	↓	No
Zinc	No	↑ except in at MF3	No
Zirconium	-	In	No

Note: Grey shading denotes statistically significant upward temporal trend, outside of the normal range in 2016 and gradient observed.
↑ = above the normal range;. - = not determined as >60% <DL within at least one area in any given year.

9.5.2.2 Mercury in Lake Trout

Based on a re-evaluation of the data from 1996 to 2015 and data adjusted for length of Lake Trout, the conclusions are:

- The general pattern in the raw (unadjusted) mercury fish tissue data suggests tissue mercury concentrations increased over time, then declined in both Lac de Gras and Lac de Sauvage.
- This pattern was also evident in length-adjusted mercury data where statistical differences over time were observed.
- Lac de Gras and Lac de Sauvage exhibited differing concentrations of mercury in Lake Trout for fish of similar sizes. Based on the small sample size, Lac du Sauvage fish have greater adjusted mercury concentrations in 2014 than fish from Lac de Gras.
- The mean unadjusted mercury concentration exceeded the EA prediction of 0.2 µg/g wwt in six of 11 years of Lake Trout fish tissue sampling.
- As noted above, there was no statistically significant increasing trend in mercury observed in Slimy Sculpin between 2013 and 2016.

9.6 Overall Conclusion

Fish body size is trending towards the capture of smaller sized fish at the NF and MF areas. Based on analysis of spatial and temporal gradients, this is inconsistent with a Mine effect. Local habitat variation among the areas may be contributing to this finding. Parasites and metal concentrations appear unlikely to have contributed to differences observed in Slimy Sculpin body size at the NF and MF areas. The magnitudes of the differences between areas is limited given that most fish body sizes were within normal range, and there were only limited exceedances of the CES; this confirms that changes in fish health were small and almost all were within the range of natural variability. Slimy Sculpin are present and healthy in each area and are capable of growth, reproduction, and energy storage.

A Mine-related effect in fish tissue was detected based on analysis of spatial and temporal gradients; this was an increase in molydeum and uranium concentrations in small-bodied fish (Slimy Sculpin), and was consistent with the findings of the *2014 to 2016 AEMP Response Plan Fish – Supplemental Repor*t (Golder 2017e) which was approved by the WLWB. No further actions are suggested herein. No statistically significant increasing trend in mercury was observed in Slimy Sculpin between 2013 and 2016. Overall, there is some uncertainty in the tissue chemistry conclusions given a correction factor was applied due to missing liver tissue in the 2016 samples; it is possible some variable concentrations have been adjusted too high; however, the analysis remains conservative. Fish tissue analysis will be completed in 2019 as part of the Fish Program, and concentrations will be reviewed at that time.

On the basis of the mean and range of mercury concentrations observed in Lake Trout over time, fish are safe to eat. Older (longer) fish have concentrations that occassionally exceed the CFIA guideline of 0.5 µg/g wwt, similar to other lakes in the north (GNWT 2018).

10 WEIGHT OF EVIDENCE

10.1 Introduction

The goal of the AEMP is to assess and monitor the effects of Mine-related stressors (primarily metals and nutrients) that are released to Lac de Gras. Related to these stressors, the AEMP has identified two broad impact hypotheses for Lac de Gras:

- **Toxicological Impairment Hypothesis**: Toxicity to aquatic organisms could occur due to chemical contaminants (primarily metals) released to Lac de Gras.
- **Nutrient Enrichment Hypothesis**: Eutrophication could occur due to the release of nutrients (i.e., phosphorus and nitrogen) to Lac de Gras**.**

The WOE analysis is structured to distinguish between these two hypotheses, and provides a systematic and transparent method for integrating complex environmental data. The objectives of the WOE analysis are two-fold:

- to apply a standardized process to evaluate strength of evidence for potential toxicological impairment and nutrient enrichment effects in the aquatic ecosystem of Lac de Gras; and
- to summarize the AEMP findings in a semi-quantitative manner that provides broad AEMP conclusions, to inform decision-making for ongoing environmental stewardship of Lac de Gras.

The WOE integrates the following field components: water quality; sediment quality; fish tissue chemistry; lake productivity (i.e., nutrients, chlorophyll *a*, and zooplankton biomass); plankton communities; benthic invertebrates; and, fish population health. As the WOE requires the results of all endpoints for exposure and effects (i.e., biological responses) the WOE is conducted every three years, following the comprehensive sampling program, when all components and all locations are sampled.

The products of the WOE analysis are estimates of the Evidence of Impact (EOI) in support of each hypothesis for the following valued ecosystem components (VECs): lake productivity; benthic invertebrate community; and, fish population health. A higher rank represents a higher strength of support for a particular hypothesis. The term "Impact" is used in this report in a generic sense to indicate a change (positive or negative) in Lac de Gras related to the Mine or Mine activities. It is not intended to reflect the ecological significance or level of concern associated with a given change, nor is it intended to indicate that "pollution" of Lac de Gras has occurred.

As described in the *AEMP Study Design Version 3.5* (Golder 2014a), ecological significance and the severity of possible effects to an assessment endpoint are categorized in the AEMP according to Action Levels. These classifications were developed to meet the goals of guidelines for developing Response Frameworks drafted by the WLWB (Racher et al. 2011). The goal of the Response Framework is to ensure that significant adverse effects never occur. When Action Levels are triggered for a particular component of the AEMP, the findings of the WOE analysis serve to inform response planning and environmental stewardship. For example, if the plankton community structure were to shift to a degree that management responses and/or mitigation were considered necessary based on the Response Framework, then the WOE findings would inform whether adaptive management should focus on the mitigation of nutrient releases or toxicant releases.

10.1.1 Background

DDMI has been conducting baseline studies and monitoring programs related to the aquatic ecosystem of Lac de Gras since 1994, with AEMP data collected under AEMP study designs Version 1.0 (2001[10] to 2006), Version 2.0 (2007 to 2011), Version 3.0 (2012 to 2016), and most recently under Version 4.0 (2017; to be reported in the *2017 AEMP Annual Report*). Since there is potential for Mine effluent to affect Lac de Gras, water quality and aquatic biology hve been monitored annually since 2001 as part of the AEMP. The WOE component, which integrates the results of all other AEMP monitoring components, was added in 2007, and is currently completed during each comprehensive monitoring year.

10.1.2 Component History

WOE analyses were conducted every year between 2007 and 2011 per the WOE integration framework developed according to the *AEMP Design Document Version 2.0* (DDMI 2007). Contaminant variables in water quality, sediment quality, and fish tissue were not monitored in 2012, precluding a WOE analysis that year. In 2011, the third Version of the AEMP was submitted as *AEMP Study Design Version 3.0* (Golder 2011b), which underwent five rounds of revisions. Under the updated AEMP design and beginning with the 2013 AEMP, the WOE analysis is conducted every three years, in conjunction with the comprehensive sampling program, when all components and all locations are sampled. Thus, subsequent to the 2011 AEMP, a WOE analysis was conducted in 2013 and 2016. The 2007 to 2016 WOE analyses are described in the individual AEMP reports for each year. The 2013 AEMP was carried out according to *AEMP Study Design Version 3.3* (Golder 2014d), whereas the 2016 AEMP was carried out according to *AEMP Study Design Version 3.5* (Golder 2014a). Although *AEMP Study Design Version 3.5* (Golder 2014a) was the approved Version of the AEMP design at the time the 2016 AEMP report was written, where relevant a number of updates outlined in the proposed *AEMP Design Plan Version 4.1* (Golder 2017d) and in WLWB directives (28 July 2015, 26 May 2016, 14 November 2016 and 2 March 2017 Decision Packages) were incorporated into the 2016 analysis. These updates were also included in this report as discussed in Section 10.2.

10.2 Approach

The WOE analysis for Lac de Gras is structured to distinguish between the impact hypotheses of toxicological impairment and nutrient enrichment. The products of the WOE analysis are estimates of the EOI Rankings for the three VECs in support of each hypothesis. For each VEC, the EOI Ranking indicates the strength of support for each hypothesis determined by the pattern of exposure and biological response in Lac de Gras, identifying the more likely cause of biological responses as enrichment versus toxicity. The EOI Ranking is not intended to convey the severity of observed impacts, or their ecological significance.

The measurement endpoints, Lines of Evidence (LOEs), LOE groups, and ecosystem components included in the WOE framework for each impact hypothesis are summarized in Tables 10-1 and 10-2. Within each

[10] One year of baseline data were also collected in 2000 under the AEMP Version 1.

ecosystem component, two distinct LOE groups are assessed to integrate information about exposure and effects:

- **Exposure group**: measures of the potential exposure of receptors of potential concern to Mine-related SOIs, including surface water, sediment, and fish tissue chemistry
- **Biological Response group**: observationally-based measures of potential ecological changes, including measures of primary productivity, zooplankton biomass, benthic invertebrate community structure, and fish population health

For many of the LOE groups, multiple endpoints have been measured in Lac de Gras providing a "battery" approach for assessing the degree of effect associated with each LOE. In evaluating multiple endpoints for each LOE, a wide variety of possible changes are considered in the overall analysis. The two LOE groups bring distinct types of information to the WOE framework. For example, sediment chemistry (which is an exposure endpoint) provides information on contamination relevant to benthic invertebrates and demersal fish, but not on biological effects. The diversity of the benthic invertebrate community present in Lac de Gras (which is a biological response endpoint) provides evidence of substance-related effects in the environment, although observed alterations may be due to biological (e.g., predation, seasonal population dynamics, competition) and/or physical effects (e.g., habitat alteration) unrelated to either contaminants or nutrient enrichment. Results that demonstrate a high degree of concordance between the two LOE groups provide stronger evidence regarding potential Mine-related ecological effects than reliance on one type of LOE in isolation. *A posteriori* weighting factors are applied in the WOE analysis to characterize the degree of concordance between endpoints in the exposure and biological response LOE groups.

Because the WOE framework is primarily a scoring and weighting logical system, it is not well suited to statistical analysis of trends over time. Thus, the 2007 to 2016 summary of the WOE findings followed a different approach to the preceding sections, and includes:

- a summary of WOE conclusions from 2007 to 2016 (Section 10.3)
- examination of patterns of response and key "driver" endpoints across AEMP years (Section 10.4)
- comparison of WOE findings to EA predictions (Section 10.5)

For the current re-evaluation report, there were several revisions to the WOE analysis that were incorporated in response to WLWB directives (26 May 2016 and 28 August 2017 Decision Packages Table 1-1). For consistency, WOE analyses were re-run for all AEMP years to incorporporate the revisions. In some cases, the revisions resulted in changes to the EOI Rankings relative to the previous re-evaluation report and the WOE reports of individual AEMPs. However, it was considered preferable to examine patterns and trends according to the revised WOE approach, rather than attempting to explain differences in conclusions between differing WOE approaches. The revisions were:

- A corrected direction-weighting factor was incorporated for tapeworm parasitism per Directive 3D of 28 August 2017 Decision Package (Table 1-1). A direction-weighting factor of 0.5 (i.e., neutral) was applied for an observed increase in the pathology (tapeworm parasitism) occurrence endpoint in the re-evaluation, in recognition that an increase in pathology occurrence could be associated with either toxicological impairment or nutrient enrichment.

- A corrected *a priori* weighting factor for chlorophyll *a* was incorporated per Directive 3G of 28 August 2017 Decision Package (Table 1-1). The *a priori* Representativeness weighting factor for chlorophyll *a* was changed from 1.5 to 1.0 (i.e., poor), because similar to water quality, it is an imperfect representation of spatial and temporal variability, providing a "snapshot" estimate of phytoplankton biomass.
- Per Directive 2D of 26 May 2016 Decision Package (Table 1-1), sediment chemistry was added as an exposure endpoint to the LOE group of the Fish Population Health ecosystem component for the toxicological impairment WOE.
- Changes were made to previous years' WOE effect ratings for some individual fish tissue chemistry variables. Rationale for these changes are described in Section 9.3.2.1.

There was a potential for change to *a posteriori* weighting factors due to some of the revisions listed above. For example, for tapeworm parastism applying an equal direction-weighting factor to both impact hypotheses could result in a change to the *a posteriori* weighting that was previously applied due to a change in professional judgement with respect to the strength of linkage and/or coherence of response for the LOE.

The updated WOE analyses are provided in Appendix 10A. Because field sampling of Lac de Gras fish populations was only included in the 2007, 2010, 2013, and 2016 AEMPs, the WOE summary for fish population health is based on these four years only. The WOE effects ratings from 2007 to 2016 are tabulated in the preceding component chapters. These effects ratings were then incorporated into the WOE analyses provided in Appendix 10A, where they were scored, weighted, and compared to a calibrated scale to determine the EOI Ranking.

The weighting factors characterize professional judgement and are explained in detail in the annual WOE component reports (e.g., Golder 2017k). WLWB (2017e) noted in their *Reasons for Decision* that the *a priori* weighting factors for the fish tissue exposure endpoint (Sculpin Tissue Chemistry - Comparison to FF) and one of the biological response endpoints for fish population health (Growth – Size at Age) were omitted from the 2016 Weight-of-Evidence Report (see Table 2-5 of Golder 2017k). Per Directive 3H of 28 August 2017 Decision Package (Table 1-1), rationale for the *a priori* weighting factors for these endpoints are provided below:

- The *a priori* weighting for the Sculpin Tissue Chemistry - Comparison to FF exposure endpoint was assigned an overall weighting factor of 14.1, which is the product of scores assigned to the following four *a priori* factors (each factor was assigned a score ranging from 1 to 3, with a higher score indicating a stronger contribution of an endpoint to the WOE analysis):
  - Representativeness = 2.5 (tissue chemistry concentrations integrate temporal and spatial exposure, as well as bioavailability; however, the presence of metals in tissue does not mean that effects are occurring)
  - Methodological Robustness = 2.5 (methodologies are well established, with accepted QA/QC measures, and data analysis techniques)
  - Clarity of Interpretation = 1.5 (moderate uncertainty in the linkage between internal chemical concentrations and ecological effects, especially when toxicity benchmarks are not considered)

- Permanence of Effects = 1.5 (although tissue residues reflect long-term exposure, the persistence of metals in Slimy Sculpin tissues would be limited by excretion processes for many metals, and the relatively short lifespan of this species, relative to large top predator species such as Lake Trout)

- The overall *a priori* weighting factor for Body Size was 25, which is the product of scores assigned to the following four *a priori* factors:
  - Representativeness = 2.5 (there is some natural variability in forage fish communities; energy expenditure/stores and reproductive investment can vary seasonally and interannually)
  - Methodological Robustness = 2.5 (methodologies are well established, with accepted QA/QC measures and data analysis methods; visual inspection for pathology may be somewhat subjective relative to other metrics)
  - Clarity of Interpretation = 2.0 (community-level measures in the field have clear relevance to ecological effects)
  - Permanence of Effects = 2.0 (impacts to population structure could take generations to recover; energy expenditure/stores and reproductive investment affect long-term productivity and stability of population)

Table 10-1 Endpoints and Lines of Evidence for Each Component – Toxicological Impairment Hypothesis

Ecosystem Component	Line of Evidence Group	Line of Evidence	Endpoints
Lake Productivity	Exposure	Contaminant Exposure	Water Chemistry
	Biological Response	Biological Productivity	Chlorophyll *a*
			Phytoplankton Biomass
			Zooplankton Biomass
			Phytoplankton Community Composition
			Zooplankton Community Composition
Benthic Invertebrate Community	Exposure	Contaminant Exposure	Water Chemistry
			Sediment Chemistry
	Biological Response	Benthic Invertebrate Community	Total Invertebrate Density
			Dominant Taxa Density
			Richness
			Simpson's Diversity Index (SDI)
			Evenness
			Dominance
			Bray-Curtis Distance
			Relative Abundances of Dominant Taxa
Fish Community	Exposure	Contaminant Exposure	Water Chemistry
			Sediment Chemistry
			Fish Tissue Chemistry
	Biological Response	Fish Population Health	Population Structure – Survival
			Population Structure – Size
			Growth – Size at Age
			Energy Stores – Condition (K)
			Energy Stores – Liver Somatic Index (LSI)
			Reproductive Investment – Age 1 Abundance
			Reproductive Investment – Gonadosomatic Index (GSI)
			Pathology – Occurrence

Table 10-2 Endpoints and Lines of Evidence for Each Component – Nutrient Enrichment Hypothesis

Ecosystem Component	Line of Evidence Group	Line of Evidence	Endpoints
Lake Productivity	Exposure	Nutrient Exposure	Water Chemistry – Total Nitrogen and Total Phosphorus
	Biological Response	Biological Productivity	Chlorophyll *a*
			Phytoplankton Biomass
			Zooplankton Biomass
			Phytoplankton Community Composition
			Zooplankton Community Composition
Benthic Invertebrate Community	Exposure	Nutrient Exposure	Water Chemistry – Total Nitrogen and Total Phosphorus
			Sediment Chemistry – Total Organic Carbon (TOC)
		Primary Productivity	Chlorophyll *a*
	Biological Response	Benthic Invertebrate Community	Total Invertebrate Density
			Dominant Taxa Density
			Richness
			Simpson's Diversity Index
			Evenness
			Dominance
			Bray-Curtis Distance
			Relative Abundances of Dominant Taxa
Fish Community	Exposure	Nutrient Exposure	Water Chemistry – Total Nitrogen and Total Phosphorus
			Sediment Chemistry – TOC
		Primary Productivity	Chlorophyll *a*
	Biological Response	Fish Population Health	Population Structure – Survival
			Population Structure – Size
			Growth – Size at Age
			Energy Stores – Condition (K)
			Energy Stores – Liver Somatic Index (LSI)
			Reproductive Investment – Age 1 Abundance
			Reproductive Investment – Gonadosomatic Index (GSI)
			Pathology – Occurrence

10.3 2007 to 2016 Summary

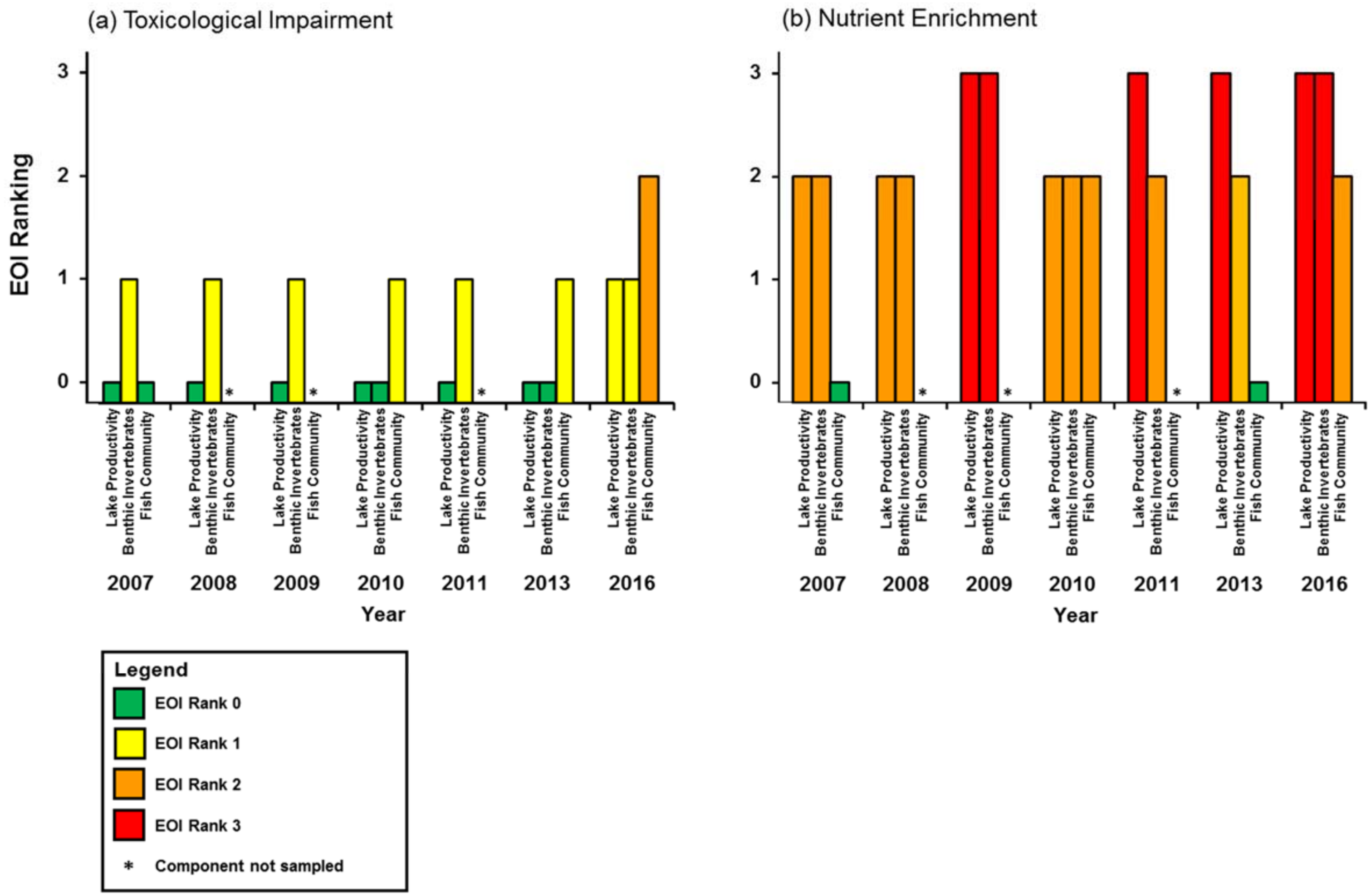
Figure 10-1 provides a year-by-year summary of the EOI Rankings for the toxicological impairment and nutrient enrichment WOE analyses.

For the toxicological impairment hypothesis, the EOI Rankings for lake productivity and benthic invertebrate community have remained weak or low from 2007 to 2016. The lake productivity component has had an EOI Rank of 0 each year up until 2016 when it increased to an EOI Rank of 1. The low-level EOI ranking for lake productivity in 2016 was due to zooplankton biomass based on AFDM, which was statistically less than in the NF area compared to the FF areas; however, zooplankton biomass based on enumeration was statistically greater in the NF area compared to FF areas, and was greater than the normal range. The observed biological response indicating possible impairment wss not consistent with multiple other responses indicating enrichment. The benthic invertebrate community component had an EOI Rank of 0 or 1 from 2007 to 2016. The fish population health component had an EOI Rank of 0 in 2007, an EOI Rank of 1 in 2010 and 2013, and an EOI Rank of 2 in 2016. In 2010, the highest weighted response for fish health was increased pathology (due to presence of tapeworms), which was attributed to enrichment rather than toxicity, but could not be excluded from the rating and weighting process. For consistency in application of the WOE framework, the EOI Rank of 1 was retained for fish population health in 2010; however, it likely overestimates support for the toxicological impairment hypothesis.

For the nutrient enrichment hypothesis, the EOI Rankings for lake productivity and benthic invertebrate community were moderate to strong from 2007 to 2016. The lake productivity component has varied between an EOI Rank of 2 or 3 over time, but was consistent at an EOI Rank of 3 in the last three comprehensive monitoring years (i.e., 2011, 2013, and 2016). The benthic invertebrate community component had an EOI Rank of 2 for all years except 2009 and 2016 when the EOI Rank was 3. The fish population health component increased from an EOI Rank of 0 in 2007 to an EOI Rank of 2 in 2010, then decreased back to an EOI Rank of 0 in 2013. In 2016, the fish population health component had an EOI Rank of 2.

These EOI Ranking results are generally indicative of stronger evidence for an enrichment impact on each ecosystem component in Lac de Gras, relative to the evidence for toxicological impairment. In 2013 and 2016, there was equal or stronger evidence for toxicological impairment in the fish community compared to nutrient enrichment. Key driver endpoints in the WOE analysis that contributed to the EOI Rankings are discussed further in the following sections.

Figure 10-1 Summary of 2007 to 2016 Evidence of Impact Rankings



EOI = Evidence of Impact

10.4 Trend Analysis

10.4.1 Patterns of Response

Figures 10-2 and 10-3 provide summaries of the key driver endpoints for the toxicological impairment and nutrient enrichment WOE analyses from 2007 to 2016 (refer to Appendix 10A). The key driver endpoints were those with the highest weighted scores for each ecosystem component within each line of evidence group (i.e., exposure and response groups). The weighted scores for these key driver endpoints resulted in the final WOE scores for each ecosystem component, leading to the EOI Rankings summarized in Figure 10-1. Note that fish population health and tissue chemistry were only measured in the 2007, 2010, 2013, and 2016 AEMPs and, therefore, Figures 10-2 and 10-3 only include a 4-year summary for these endpoints.

10.4.1.1 Toxicological Impairment

The key driver endpoint for the contaminant exposure LOE group for the lake productivity ecosystem component was water chemistry (i.e., the only measurement endpoint). Water chemistry was consistently determined to have a low effect level rating each year (Figure 10-2a). Between 2007 and 2013, the key driver endpoints for the biological response group of the lake productivity component varied between phytoplankton and zooplankton community structure/richness. However, these endpoints do not have a high degree of specificity with respect to impact type, and based on the responses of other endpoints (e.g., increased chlorophyll *a* and biomass of plankton), the observed changes were more likely related to enrichment rather than toxicological impairment. This was represented in the WOE analyses by the lower *a posteriori* weighting of these endpoints for toxicological impairment relative to those for nutrient enrichment (refer to Appendix 10A). Accordingly, when plankton community structure was the key endpoint driver, the resulting EOI Rank was 0. In 2016, lower zooplankton biomass in the NF area compared to FF areas was the key driver endpoint, resulting in an EOI Rank of 1 (Figure 10-2a).

The key driver endpoint for the contaminant exposure LOE group of the benthic invertebrate community ecosystem component was consistently sediment chemistry, whereas the key driver endpoints for the biological response LOE group varied (Figure 10-2b). Lower richness in the NF area compared to FF areas was the key driver endpoint in three of the five years resulting in an EOI Rank of 1. Relative abundance of dominant taxa was the key driver endpoint in 2007 and 2016, rated as high effect level and resulting in an EOI Rank of 1 in both years. These endpoint responses indicate a potential shift in community structure as a result of proximity to the Mine or exposure to Mine effluent. Observed changes in relative abundance of dominant taxa were likely due to nutrient enrichment, based on responses of other endpoints (e.g., increased chlorophyll *a* and biomass of plankton) and were, therefore, down-weighted *a posteriori*.

The key driver endpoint for the contaminant exposure LOE group for the fish population health ecosystem component has been tissue chemistry, which had a moderate effect level rating in 2007, 2010, and 2013, and a high effect level rating in 2016 (Figure 10-2c). In 2007, biological responses associated with the moderate effect level rating in tissue chemistry were not observed, while in 2010 the key driver endpoint was an increase in pathology occurrence (i.e.,parasites), which was likely due to an enrichment effect (i.e., enrichment increasing parasite abundance) consistent with other observed responses of fish population health enpoints in 2010 (i.e., an increase in body size, condition factor, and LSI). As discussed in Section 10.3, the EOI Rank of 1 for 2010 likely overemphasizes the toxicological impairment hypothesis.

For 2013, the key driver endpoints for biological responses were lower LSI and GSI in the NF area compared to FF areas, resulting in low effect level ratings. In 2016, the key driver endpoint was size at age, which was lower in the NF area compared to FF areas and resulted in a low effect level rating. The observed low effect level ratings for biological responses in 2016 combined with high effect level rating for the exposure LOE group resulted in an EOI Rank of 2.

Figure 10-2 Key Driver Endpoints for the Toxicological Impairment Weight-of-Evidence

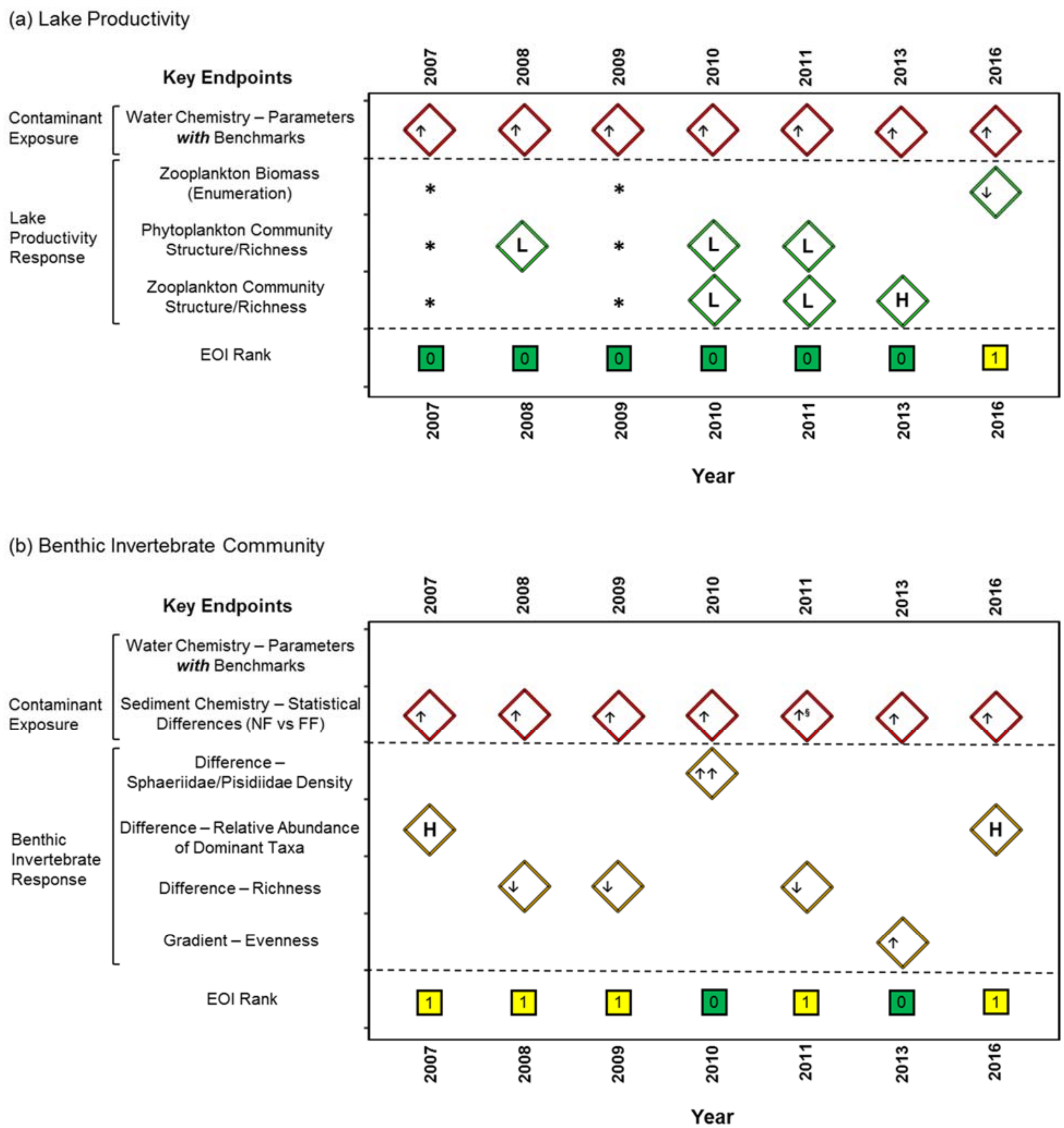
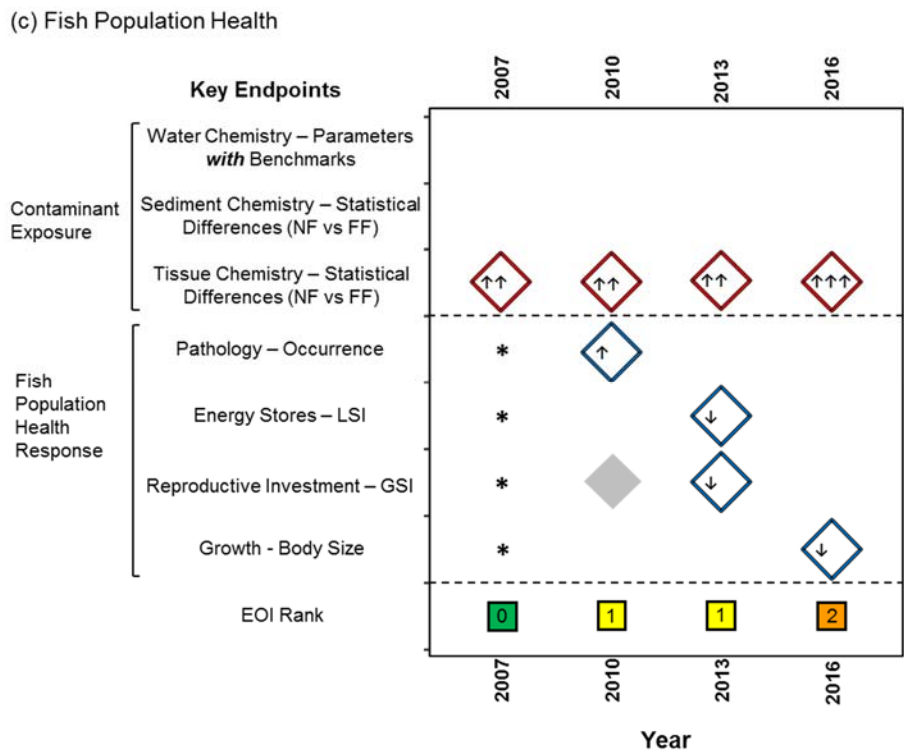


Figure 10-2 Key Driver Endpoints for the Toxicological Impairment Weight-of-Evidence (continued)



Notes:
Red diamond = exposure endpoints; green diamonds = lake productivity response endpoints; brown diamonds = benthic invertebrate community response endpoints; blue diamonds = fish population health endpoints; grey diamonds = endpoint not measured that year.
↑ or ↓ = low effect level rating; ↑↑ or ↓↓ = moderate effect level rating; ↑↑↑ or ↓↓↓ = high effect level rating; L = ↑/↓ (i.e., high effect level rating for non-specific endpoint); H = ↑↑↑/↓↓↓ (i.e., high effect level rating for non-specific endpoint); § = effect rating for 2011 sediment endpoint taken from results of 2010; * = response direction does not indicate toxicity.
EOI = Evidence of Impact; EOI Rank 0 = negligible evidence of impact; EOI Rank 1 = low evidence of impact; EOI Rank 2 = moderate evidence of impact; EOI Rank 3 = strong evidence of impact.

10.4.1.2 Nutrient Enrichment

The key driver endpoint for the nutrient enrichment exposure LOE group of the lake productivity ecosystem component in 2007 and 2008 was TP (moderate effect level rating), and from 2009 to 2016 it was TN (high effect level rating; Figure 10-3a). The key driver endpoints for the biological response LOE group of lake productivity have varied, but phytoplankton and zooplankton biomass featured prominently between 2007 and 2013, and chlorophyll *a* was the key driver in 2016. The moderate to high effect level ratings for both the exposure and biological response LOE groups have resulted in an EOI Rank of 2 or 3 for nutrient enrichment of lake productivity.

The key driver endpoint for the nutrient enrichment exposure LOE group of the benthic invertebrate community ecosystem component has generally been chlorophyll *a* from 2007 to 2016, at an effect level rating of moderate or high (Figure 10-3b). The exception was 2010 and 2011 when TN at a high effect level rating was the key driver endpoint. In 2007 and 2008, TP was also a key driver endpoint along with chlorophyll *a*. For the benthic invertebrate community biological response LOE group, the key driver endpoints varied from 2007 to 2016, but were generally total invertebrate density (gradient or statistical differences) or gradients/differences in the densities of dominant taxa (e.g., Pisidiidae, *Heterotrissocladius*, *Procladius*). The effect level rating of the key driver endpoints were generally either moderate or high for the biological response LOE group. An EOI Rank of 2 was determined for the benthic invertebrate community component in all years except 2009 and 2016, which were at an EOI Rank of 3 due to the high effect level rating in the exposure LOE group (for chlorophyll *a*), combined with supporting nutrient-enrichment-related biological responses (*Procladius* density).

The key driver endpoint for the nutrient enrichment exposure LOE group of the fish population health ecosystem component was chlorophyll *a* and TP (moderate effect level ratings) in 2007, TN (high effecl level rating) in 2010, and chlorophyll *a* (high effect level rating) in 2013 and 2016 (Figure 10-3c). In 2007 there were no responses observed for any of the fish health biological response endpoints, while in 2013 the direction of response for lower body size, LSI, and GSI was not consistent with a nutrient enrichment effect. The lack of, or inconsistent, biological responses observed in 2007 and 2013 resulted in an EOI Rank of 0 for these years. The key driver endpoint for fish health was condition factor in 2010 and pathology occurrence (i.e., parasites) in 2016, which were both rated as low level effects. Combined with a high effect level rating for the nutrient enrichment exposure LOE group during these assessment years resulted in an EOI Rank of 2.

Figure 10-3 Key Driver Endpoints for the Nutrient Enrichment Weight-of-Evidence

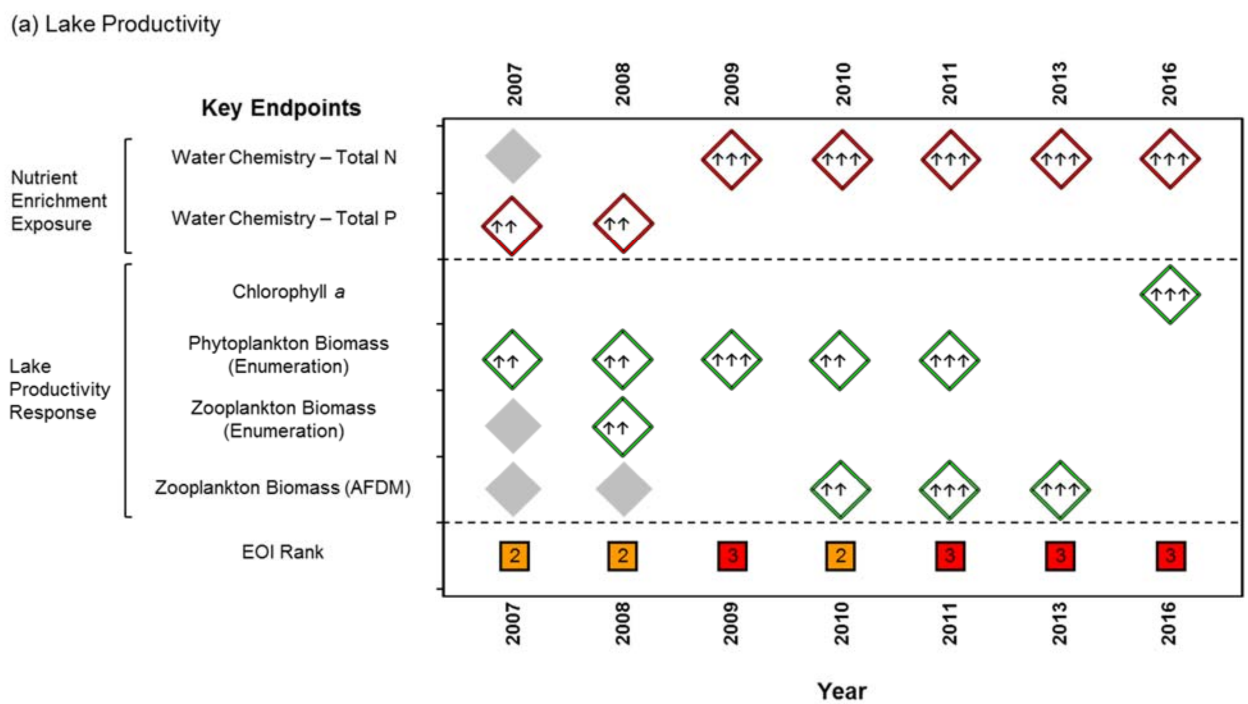


Figure 10-3 Key Driver Endpoints for the Nutrient Enrichment Weight-of-Evidence (continued)

(b) Benthic Invertebrate Community

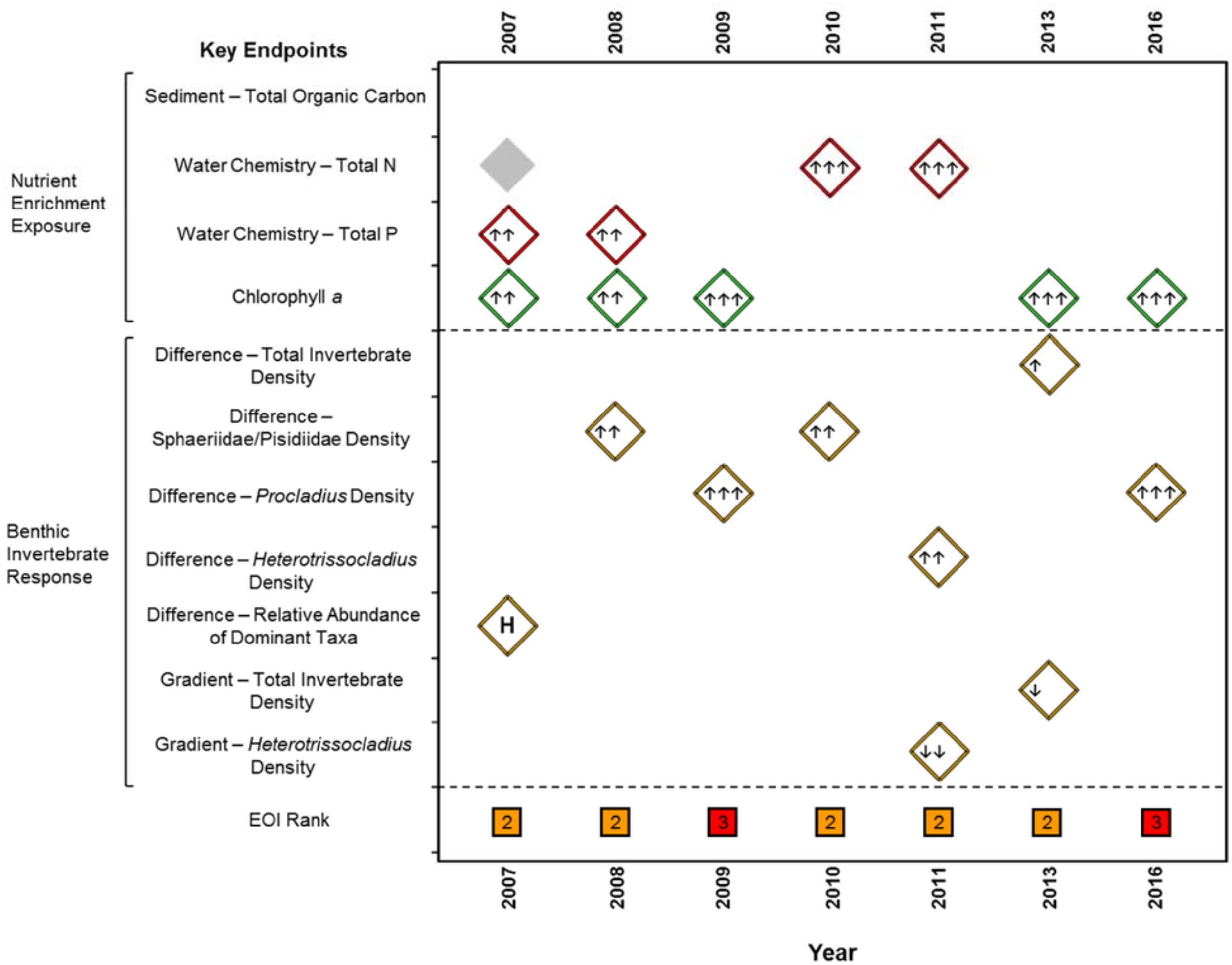
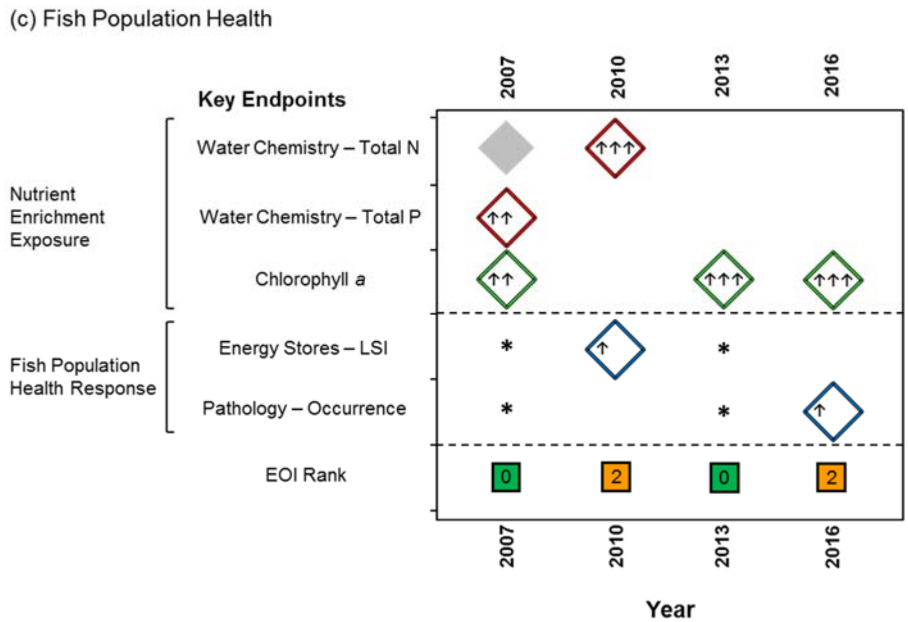


Figure 10-3 Key Driver Endpoints for the Nutrient Enrichment Weight-of-Evidence (continued)



Notes:
Red diamond = nutrient enrichment exposure endpoints; green diamonds = chlorophyll *a* exposure endpoint and lake productivity response endpoints; brown diamonds = benthic invertebrate community response endpoints; blue diamonds = fish population health endpoints; greay diamonds = endpoint not measured that year.

↑ or ↓ = low effect level rating; ↑↑ or ↓↓ = moderate effect level rating; ↑↑↑ or ↓↓↓ = high effect level rating; H = ↑↑↑/↓↓↓ (i.e., high effect level rating for non-specific endpoint); * = response direction does not indicate toxicity.

EOI = Evidence of Impact; EOI Rank 0 = negligible evidence of impact; EOI Rank 1 = low evidence of impact; EOI Rank 2 = moderate evidence of impact; EOI Rank 3 = strong evidence of impact.

10.5 Comparison to EA Predictions

The EA for the Diavik Project (DDMI 1998a) predicted that overall, Lac de Gras water would remain at a high quality for use as drinking water and by aquatic life. The main impact was expected to be the introduction of nutrients, particularly phosphorus, with a concomitant increase in primary productivity over a portion of the lake. It was predicted that in up to 20% of the surface area of Lac de Gras, TP concentration would increase above the EA threshold of 5 µg/L, without a lake-wide increase in trophic status. Based on the predicted effects of phosphorus, the overall effect of the Mine water discharge on water quality for the protection of aquatic life was predicted to be a mid-term, moderate magnitude, regional effect. The overall trophic status in most of Lac de Gras was not expected to change, and was expected to remain ultra-oligotrophic (now considered oligotrophic under baseline conditions, based on additional phosphorus data collected in the FF areas after the EA).

The magnitude and type of response that has been observed in Lac de Gras appears to be mild nutrient enrichment, resulting in increased lake productivity. The area of effect for TN exceeds 20% (i.e., a high level rating) in most years, and for some years, this magnitude of effect is also observed for chlorophyll *a*. In 2016, concentrations of chlorophyll *a* in the NF and MF areas were greater than the upper limit of the normal range in an area representing greater than 20% of the lake surface, and consequently the magnitude of the eutrophication effect is equivalent to Action Level 2 in the Response Framework. However, the area affected by TP (the limiting nutrient in Lac de Gras) has generally been low (<0.5 to 6.5%), with the greatest spatial extent observed in 2008, estimated as 19.6% of the lake area. Although there are statistically significant changes to indicators of enrichment in the NF area (and in some cases MF areas), the severity with respect to the ecological integrity of Lac de Gras associated with these changes is considered to be low. On the whole, AEMP monitoring results do not indicate that the trophic status of the lake is changing.

In contrast to results for nutrient enrichment, there is weak evidence of impairment to lake productivity, benthic invertebrate community, and fish community as a result of contaminant exposure, with the only moderate EOI Ranking observed for the fish community in 2016. The low to moderate EOI Rankings have relatively high uncertainty, because the link to contaminant exposure is generally not strong and the responses indicating possible impairment are not consistent with multiple other responses indicating enrichment. Although results of the fish population health survey in 2016 triggered an Action Level 2, Golder (2017j) concluded that the the fish health endpoint responses were inconsistent with a Mine effect, because the magnitude of the responses were small and within the bounds of expected natural variability. The findings with respect to toxicological impairment are likely due to an inability to definitively rule out the possibility that toxic impacts might be occurring in Lac de Gras, as opposed to definitive evidence that toxic impacts are occurring.

10.6 Conclusions

The EOI in Lac de Gras has remained relatively constant since 2008. For the toxicological impairment hypothesis, the EOI Ranking has remained low (EOI Rank of 0 or 1) for lake productivity and benthic invertebrate community components from 2007 to 2016, and low to moderate (EOI Rank of 1 to 2) for the fish population health component. For the nutrient enrichment hypothesis, the EOI Ranks for the ecosystem components have generally varied between 2 and 3 from 2007 to 2016, with the exception of the EOI Rank for the fish population health component in 2007 and 2013 (EOI Rank of 0).

The general pattern of response was one of nutrient enrichment in an area representing greater than 20% of Lac de Gras, with a concurrent response in the plankton and benthic invertebrate communities. Although nutrient exposure was occurring for fish populations in this area, the response in this trophic level was at a lower degree than in the plankton and benthic invertebrate communities. The type of impact being observed in Lac de Gras was consistent with that of mild nutrient enrichment. Increased exposure to contaminants was also occurring for plankton, benthic invertebrate community, and fish, but biological responses that might indicate toxicity were weak and could be attributed to natural variability and ecological factors.

Overall, the WOE assessment is performing as intended. The assessment is structured to distinguish between the two Mine-related impact hypotheses (i.e., to answer the question “if nutrient encrichment or toxicological impairment are the only hypotheses that explain responses observed in biological endpoints of VECs, what is the strength of evidence for each?”); however, any observed changes may also be due to biological (e.g., predation, seasonal abundance, competition) and/or physical (e.g., habitat alteration) effects unrelated to contaminants or nutrient enrichment. The strength of evidence for toxicological impairment and nutrient enrichment associated with observed changes are evaluated using an array of exposure and biological response measurement endpoints specific to the WOE assessment. Results that demonstrate a high degree of concordance between the exposure and biological response LOE groups provide stronger evidence for potential Mine-related ecological effects than reliance on one type of LOE in isolation.

The WOE assessment is designed to err on the side of conservatism, identifying evidence of impact regardless of whether the apparent changes are expected to have ecological significance. The Response Framework sets criteria for changes in exposure and biological response in Lac de Gras that would warrant adaptive management responses and/or mitigation, and the findings of the WOE assessment serve to inform reponse planning and environmental stewardship.

11 TRADITIONAL ECOLOGICAL KNOWLEDGE

11.1 Introduction

This chapter presents a summary of the AEMP Traditional Knowledge (TK) Study. In partnership with the Participation Agreement community organizations, DDMI coordinated an annual TK fish tasting program beginning in 2002 and continuing to 2009. At that time, communities identified an opportunity to strengthen the TK component of the program while building upon the TK acquired in previous years of monitoring, as well as to integrate TK and western scientific information related to the aquatic environment. The development of a methodology by which TK was incorporated into the AEMP was initiated at community meetings that took place in the winter of 2011 and spring of 2012. The AEMP Version 3.0 included an expanded role of TK in aquatic monitoring, with the aim of identifying TK indicators that could be used to evaluate the overall performance of the Mine and determine if there was an effect on the condition and health of the fish and water. The AEMP TK Study is now conducted every 3 years.

For the 2015 AEMP TK Study, work was conducted from 6-10 August at a camp located near the Mine on the south side of Lac de Gras. Participants included Elders and youth from the five Aboriginal parties to Diavik's Environmental Agreement: Kitikmeot Inuit Association; Łutsel K'e Dene First Nation; North Slave Métis Alliance; Tłı̨chǫ Government; and Yellowknives Dene First Nation.

11.2 Objectives

TK plays an important role in both the fish and water quality components of the AEMP. The primary objective of the 2012 and 2015 AEMP TK studies were to facilitate the two-way flow of information, resources, and knowledge between TK holders and scientists regarding the health of fish and water in Lac de Gras. The approach employed to realize this objective included the following:

1) provide a forum for information exchange between community members, scientists and DDMI (particularly around mining and environmental issues)

2) engagement and capacity building opportunities for communities

3) community involvement in, and understanding of, DDMI's monitoring programs

4) record relevant Aboriginal language references, customs, and stories

5) facilitate Elder-youth and cross-cultural exchange

6) collect, record, discuss and verify shared TK

7) collect data in a standardized way, such that data sets are comparable across years

8) include TK interviews relating to both water and fish

9) identify and monitor indicators of water quality and fish health from a TK perspective

11.3 Methods

DDMI, with the assistance of Thorpe Consulting Services, held multiple workshops with communities to jointly develop a mutually agreeable study approach for a TK program to support the AEMP. Further, agreements around information sharing between knowledge holders and DDMI were discussed during workshops, such that community decisions were made prior to data collection. DDMI continues to work with communities to refine the program and report information in ways that are respectful and mutually agreeable.

In 2012 and 2015, TK interview sessions were conducted with a focus on both documenting and communicating TK in ways that respect intellectual property rights and are in keeping with standard and accepted protocols specific to the five Aboriginal organizations that assert ties to the Lac de Gras region (Golder 2013b). Community members collected samples, observed the environment near the mine, and used traditional methods and indicators to evaluate fish condition, water quality, and the health of the land in the area of the Mine.

The TK fish palatability and texture studies, and the water quality study were conducted from 6-10 August 2015. From 8-10 June 2015, a preparatory workshop was held in Yellowknife with camp participants and some representatives from the community organizations to review the program methodology, conduct training, discuss details related to timing of the camp and confirm community member attendees. The key activities agreed to by participants for the 2015 AEMP TK Study were as follows:

- water quality and fish health indicators were identified and observed from a TK perspective
- Elders, youth, and scientists collaborated on location and methods to set nets and inspect overall fish health
- Elders tasted a total of four fish for a palatability and texture analysis - these fish were baked, boiled, fried, and grilled - and youth recorded the Elder’s observations
- cultural practices and ceremonies related to water and fish were shared and recorded
- leadership and practical lessons related to fish, water and the land were shared
- cross-cultural fish inspections, sharing and learning about fish health with scientists and community members, occurred
- traditional methods for water quality testing were conducted from samples taken in the lake; water was observed, tasted and used for tea; youth recorded the Elder’s observations
- scientific data collection of fish tissue and water quality samples occurred
- observations on TK indicators for fish and water were recorded
- a video documentary was recorded

- interviews to capture stories and traditions important to each community and camp participants occurred and were recorded
- cultural and social activities (e.g., berry picking, hiking, story-telling, language lessons, healing ceremony, feeding the fire/paying the water) occurred
- health and safety protocols were followed

11.4 Results

The TK program brought together results from TK and scientific knowledge. The results were originally reported in the 2012 and 2015 Annual AEMP Reports (Golder 2013b, 2016c) and the companion video-documentaries entitled “5 Days, 2 Ways, 1 Camp” (Roaming Pictures, 2013) and “We fish today, for fish tomorrow” (aRTLeSS Collective 2016) which were filmed and produced through a partnership of participating youth and a production crew.

Across both years of the study, camp participants commented that the present overall status of the fish in Lac de Gras adjacent to the Mine was good and better than they expected given the proximity to industrial activity. Elders inspected and recorded their observations on the external and internal parts of the fish. In general, Elders raised minimal concerns about the state of the fish. With the exception of several instances of cysts and worms, the fish selected for inspection and consumption were considered healthy. During 2012, two fish were identified as being of poorer condition, noting that the fish were skinny and, in the case of one, had a larger head. Another fish was also observed as having some intestinal worms and being of poorer condition. Participants noted that this tends to occur in all fish populations and that the fish are not eaten. During the 2015 fish inspections, two Elders noted that one fish was considered inedible because the heart was smaller than usual, the eggs were dark red and there were cysts on the liver, although this was not considered to be directly linked to mining activities. Thematic analysis of the comments indicates overall satisfaction with the health, taste, texture, and colour of the fish. In both years, those fish tasted as part of the palatability study resulted in scores of 1 (excellent for eating, looks better than fish usually caught) or 2 (good for eating, looks similar to fish usually caught) from all participants. People appreciated experiencing the current state of the environment personally and evaluating fish and water “with their own eyes.”

Similarly, camp participants used indicators grounded in TK to evaluate water quality. From this holistic, interconnected perspective, camp participants deduced that water quality was good by virtue of observing the following:

- the shoreline, and health of vegetation surrounding the lake, or submerged in the lake
- water clarity, movement, temperature, and taste
- presence/absence of surface foam and/or vegetation
- birds, wildlife, and fish

A “tea test” was carried out whereby water samples were taken from Lac de Gras, boiled and then made into tea to evaluate the taste. In all cases, the taste of the water was said to be good. Elders also expressed the desire to have their TK observations confirmed and/or supported by lab results and scientific analysis.

There was general agreement that lab analysis can provide information that traditional ways of knowing may not, particularly with respect to concentrations of potential contaminants. Water quality results from scientific results and TK support the same general conclusion that the water is still good in Lac de Gras.

11.5 Comparison to EA Predictions

The EA had one prediction regarding fish quality relevant to TK. The EA predicted that no adverse effect on fish quality (i.e., texture, taste) in Lac de Gras would occur as a result of tainting through the introduction of chemicals or fuels at the Mine site. Results of the fish palatability and texture studies have indicated that overall fish quality has not changed since baseline studies.

The EA also had a prediction regarding fish quality which relates to fish tissue chemistry. The EA predicted that mercury concentration in sport and subsistence fisheries would remain below a mean of 0.2 µg/g wwt (unadjusted for length). The prediction was exceeded in Lac de Gras in 2004, from 2008 to 2012 and again in 2015 (see Section 9, Table 9-25).

11.6 Conclusion

In conclusion, results from the fish taste and texture and water tests corroborate observations made by TK holders that there are presently no concerns about fish or water quality in Lac de Gras. Ongoing monitoring using both ways of knowing will be critical to preserving the future health of the aquatic environment.

Participants acknowledged that the fish and water are healthy and indicated that the revamped AEMP TK Study embodied a new relationship model for industry and communities. The high level of verification and “checking back” through meeting minutes, reporting and the video production empowered participants to take ownership of the process. They appreciated the opportunity to make sure they were recorded correctly.

12 AEMP SUMMARY OF EFFECTS

While the annual AEMP reports focus on within-year spatial trends, periodic evaluation of trends over time is also required by DDMI’s Water Licence. The preceding technical sections summarize temporal trends from the baseline period (where data were available) through 2016, and describe the development of Mine-related effects and changes relative to what is considered normal for Lac de Gras.

The WOE evaluation provides an integrated assessment of the effects of the Mine on the aquatic environment in Lac de Gras, which is focused on two impact hypotheses: nutrient enrichment and toxicological impairment. Every year between 2007 and 2011, WOE analyses were conducted according to *AEMP Design Document Version 2.0* (DDMI 2007). Beginning in 2013, WOE analyses were completed every three years as part of the comprehensive report, per the WOE integration framework developed according to the *AEMP Study Design Version 3.3* (in 2013; Golder 2014d) and *Version 3.5* (in 2016; Golder 2014a).

A summary of effects documented by the AEMP to date is provided in the following sections for each impact hypothesis, based on individual AEMP component results and WOE evaluation results.

12.1 Nutrient Enrichment

The EA predicted that the discharge of treated water from the Mine would introduce greater concentrations of nutrients, particularly phosphorus from natural groundwater, to Lac de Gras. It predicted that water quality in Lac de Gras would remain acceptable for aquatic life. The only variable that was predicted to be above its aquatic life threshold was TP (defined in the EA at 5 μg/L) at the mixing zone boundary during operations; TP concentration in up to 20% of the surface area of Lac de Gras was expected to exceed this threshold during operations, while the remainder of the lake would remain below this threshold. Effects associated with the discharge of phosphorus could not be predicted with certainty in the EA, but the EA indicated that effects could include an increase in algal growth, increases in fish growth rates, improvements in fish health, increases in the abundance of some aquatic species, and declines in the abundance of others (DDMI 1998b).

Lac de Gras remains a phosphorus-limited, clear water, oligotrophic lake. The nutrients with EQCs in the Water Licence (i.e., ammonia and nitrite) have remained within their EQCs to date, and TP load in effluent has remained below load limits specified in the Water Licence. Aerial deposition of phosphorus contained in fugitive dust has been considered, and when compared to the mass load of phosphorus from the watershed, the loading from dust would not be expected to change the background concentration of phosphorus in Lac de Gras. Concentrations of TP in the NF area have remained at similar levels and within the normal range during the open-water season since 2008. During the ice-cover season, TP concentrations in the NF area increased from within the normal range in 2010 to exceeding the normal range in 2013. Between 2014 and 2016, TP concentrations during the ice-cover season were generally within or below the lower limit of the normal range. Mid-depth concentrations of TP during the ice-cover season have significantly decreased in all areas of Lac de Gras between 2007 and 2016. During the open-water season, significant decreases were observed in the NF and FFB areas of Lac de Gras. The spatial extent of the effects on TP has ranged from less than 0.6% to 19.6% of the lake area.

Chlorophyll *a* concentrations in the NF and MF areas have been consistently greater than the upper limit of the normal range (0.89 μg/L) between 2007 and 2016. In 2009, 2013, 2014, and 2016 chlorophyll *a* concentrations in the NF and MF areas were greater than the upper limit of the normal range in over 20% of the lake area, but in all areas remained below the Effects Benchmark (4.5 μg/L) representing the upper limit of oligotrophic status. Chlorophyll *a* concentrations were above the normal range in the NF area in all years of monitoring, while in the MF areas, concentrations were the same or slightly less than those observed in the NF area; FF area concentrations were typically within or near the lower limit of the normal range.

Plankton monitoring results indicate that phytoplankton biomass in the NF area fluctuated from within the normal range (2013 and 2014) to both above (2007 to 2012, and 2016) and below (2015), throughout the period monitored. Temporal trends in the biomass of the major phytoplankton groups differed among sampling areas and among years in Lac de Gras. The phytoplankton analysis results indicate that changes over time in phytoplankton community structure have been occurring in the NF area of Lac de Gras, especially during the early years of operation (i.e., between 2003 and 2007), as a result of the community responding to nutrient enrichment. Conditions in 2016 differed from conditions in 2013 throughout the lake, with the NF and FF areas in these two years being more similar to one another than observed in previous years. Effects on zooplankton have been less pronounced over the years, as evidenced by the lack of temporal trends in total zooplankton biomass, which generally remained within the normal range from 2008 to 2016. Temporal trends in the biomass of the major zooplankton groups differed among sampling areas in Lac de Gras. Large zooplankton (i.e., cladocerans) increased with the initial increase in nutrients from 2008 to 2013, followed by a decrease until 2016, while calanoid copepods decreased between 2008 and 2013 and increased from 2013 to 2016. There continues to be a difference in the zooplankton community in the NF area, associated with the effluent discharge, compared to the FF areas in Lac de Gras. The zooplankton community in the NF and MF areas in 2016 differed more than the FF areas compared to previous years, likely as a result of combined effects of dike construction and effluent discharge.

The effects of nutrient enrichment have also been observed on the benthic invertebrate community, but recent data indicate a decline in effect magnitude. Total invertebrate density and densities of most major benthic invertebrate groups were greater in the NF and MF areas compared to the FF areas before 2013. In 2013 and 2016, declines to generally within the normal ranges were observed in the NF and MF areas for total density, richness and densities of dominant invertebrates, except the midge *Procladius*, which remained more abundant. Community indices have remained within their normal ranges throughout the monitoring period, although evenness in the NF area and at some MF stations has declined in 2016 to near the lower limit of the normal range. Community composition varied between the NF and FF areas in most years, with a divergence observed in 2016, exhibited as the reduced relative densities of Pisidiidae and greater relative densities of Chironomidae in the NF area compared to the FF areas.

The WOE EOI Rankings in Lac de Gras have remained relatively constant since 2008. For the nutrient enrichment hypothesis, the EOI Ranks for the ecosystem components have generally varied between 2 and 3 from 2007 to 2016, with the exception of the EOI Rank for the fish population health component in 2007 and 2013 (EOI Rank of 0). Although nutrient exposure is occurring for fish populations in this area, the response of this trophic level is at a lower degree than in the plankton and benthic invertebrate communities.

The type of impact being observed in Lac de Gras is consistent with that of mild nutrient enrichment in an area representing greater than 20% of the lake surface. The changes being observed are also consistent with those the EA described as being possible under phosphorus loading from the effluent.

12.2 Toxicological Impairment

Characterization of the Mine effluent provides no indication that a toxic response would occur in Lac de Gras. Effluent tested from 2002 to 2016 was generally non-toxic to aquatic test organisms, as shown in over 430 acute toxicity tests and over 290 sub-lethal toxicity tests. Mine effluent continues to meet EQC specified in the Water Licence. Moreover, although effluent loads and/or concentrations of some metals identified as SOIs have increased over time (e.g., molybdenum and strontium), others have either decreased (e.g., antimony) or remained at relatively similar levels over time (e.g., aluminum, uranium).

Trends in the concentrations of SOIs at the mixing zone boundary generally reflected the temporal patterns described in the annual loading rates. The magnitude of the variation observed at the mixing zone, however, was often less pronounced than that in the effluent. Some general observations of trends in AEMP water quality data were:

- concentrations of TDS (calculated), chloride, fluoride, calcium, potassium, sodium, and sulphate in Lac de Gras were greater than the normal ranges in both the ice-cover and open-water seasons, and are generally increasing over time

- molybdenum and strontium were consistently detected in Lac de Gras at concentrations above the normal range, particularly in the NF and MF areas, and are generally increasing over time
- nitrate and uranium generally have decreasing or no trends in Lac de Gras over time, with the exception of increasing trends in nitrate in the FF1 area in the ice-cover season, and uranium at the MF3-4 station in the open-water season

The majority of SOIs, except for those added to assess dike/dust effects, triggered either Action Level 1 or Action Level 2 during the period of evaluation (2007 to 2016). As a result of the Action Level 2 triggers, DDMI has developed Effects Benchmarks for turbidity, dissolved sodium, total aluminum, total antimony, total silicon, and total tin.

In sediments, some metals (e.g., arsenic and cadmium) were present in concentrations above sediment quality guidelines throughout Lac de Gras. These variables generally reflected patterns in TOC content of bottom sediments or background variation in sediment quality, and had no clear spatial trends related to the Mine. Increasing trends were identified in sediment for five SOIs (i.e., calcium, lead, molybdenum, strontium and uranium). Three SOIs (i.e., bismuth, lead, and uranium) had NF area median concentrations that were greater than normal range, and while effluent discharge is likely the primary source of these metals in the NF area, dike construction may also have contributed to the observed patterns.

Of the 18 sediment SOIs, chromium and lead were the only two with applicable sediment quality guidelines. Lead did not exceed CCME or OMOEE guidelines, and while chromium exceeded ISQG, it did not meet the WOE criteria for a moderate effect level. Further, chromium also exceeded guidelines in the FF area, suggesting existing chromium concentrations are likely reflective of elevated background concentrations in Lac de Gras rather than a Mine effect. There are no Action Levels for sediment quality.

From 2007 to 2014, neither phytoplankton nor zooplankton community variables triggered an Action Level in Lac de Gras related to the toxicological impairment hypothesis. However, Action Level 1 was triggered for phytoplankton biomass in 2015, and Action Level 2 was triggered for zooplankton biomass in 2016. The Action Level exceedances for phytoplankton biomass and zooplankton biomass provide an early-warning indication of potential Mine-related changes, and are not of immediate concern regarding toxicity to the plankton community in Lac de Gras. Benthic invertebrate monitoring provided evidence of nutrient enrichment, and effects on community composition variables that were non-specific with respect to effect type and were attributed to changes associated with nutrient enrichment, rather than toxicity. In 2016, Pisidiidae was found to be present at lower densities in the NF area compared to FF areas, and triggered Action Level 1, and evenness was found to be reduced in the NF and MF areas, thereby triggering Action Level 2. These effects were most likely the result of community shifts resulting from nutrient enrichment.

The responses of fish health endpoints have not always been consistent among life stage and sex, nor among years. In Age-1+ fish, body size (i.e., total length, total weight, and carcass weight) in the NF and sometimes MF areas was smaller than that in the FF areas in recent years, although fish health endpoints in the NF and MF areas were all within the normal ranges for Lac de Gras (with the exception of condition in the NF area in 2007). It is likely that variability in water temperature among sampling areas influenced fish growth, and cooler temperatures in the NF area in the middle of June and late July in 2013 and 2016 relative to the other sampling areas could have reduced growth in these fish. While fish size demonstrated a recent decrease in body size spatially (i.e., NF fish are smaller than FF fish), there did not appear to be a decrease in body size over time in the NF area. Parasitism levels did not follow an expected spatial gradient of response consistent with a Mine influence; the FF areas had the greatest proportion of parasites in 2010 and 2013. An Action Level 2 was triggered for fish health in 2016, and was indicative of changes that could be a toxicological response. Overall, responses in fish health were not consistent with those of the other biological components of the AEMP, and were attributed to possible early warning changes associated with a toxicological response.

In terms of fish tissue chemistry, few variables were (1) increasing in the NF area and MF/FF areas, (2) outside of normal range, and (3) showing a trend from high to low from the NF area to the FF area (i.e., demonstrating a gradient response). Molybdenum and uranium were exceptions, where a gradient suggestive of Mine influence (i.e., NF>MF>FF) was observed. There was no statistically significant increasing trend in mercury concentration in Slimy Sculpin between 2013 and 2016, and mercury was not outside of the normal range for small-bodied fish. In large-bodied fish, the mean unadjusted mercury concentration in Lake Trout exceeded the EA prediction of 0.2 µg/g wwt in six years of Lake Trout fish tissue sampling, out of a total of 11 years.

For the Toxicological Impairment Hypothesis, the EOI Ranking has remained low (EOI Rank of 0 or 1) for lake productivity and benthic invertebrate community components from 2007 to 2016, and low to moderate (EOI Rank of 1 to 2) for the fish population health component. Based on the biological monitoring results discussed above, the WOE results with respect to toxicological impairment (e.g., EOI Rank of 1) more likely reflect an inability to definitively rule out the possibility that toxic effects may be occurring in Lac de Gras, as opposed to providing definitive evidence that toxic effects are occurring.

12.3 Ecological Significance

As predicted by the EA, Lac de Gras is showing evidence of mild nutrient enrichment, as indicated by water quality and non-fish biological monitoring components, while fish component results may be showing early warning changes consistent with toxicological impairment. The strength of evidence for nutrient enrichment is moderate to strong, while it remains low (non-fish biological communities) to low to moderate (fish health) for toxicological impairment.

Severity of effect is not evaluated by the WOE analysis. For nutrient enrichment, the plankton and benthic invertebrate community monitoring results were consistent with the early stages of nutrient enrichment, which were reflected in increased density and biomass at lower trophic levels, and small community shifts, as organisms that can take advantage of the increased food supply increase in abundance. Lac de Gras remains oligotrophic, and no trends towards a change in trophic status were apparent based on the trend analyses completed. The ecological significance of the observed mild nutrient enrichment effect can be qualitatively described as low.

The potential toxicological effect observed in fish health included slightly ligher and shorter fish, and fish with smaller livers, at the NF and sometimes the MF, compared to the FF areas. These results were inconsistent with a Mine effect, based on analysis of spatial and temporal gradients. Localized habitat variation among the study areas likely contributes to the observed differences in fish health endpoints, while parasites and metal concentrations appear unlikely to have contributed to differences observed in Slimy Sculpin length, weight and relative liver size in the NF and MF study areas. The magnitudes of effect (i.e., within normal range, and limited CES exceedances) confirm the potential toxicological changes in fish health endpoints are small, and within the bounds expected of natural variability. As such, the observed mild toxicological effect to fish is considered to be of low ecological significance.

13 UPDATES TO EFFECT PREDICTIONS

No recommendations are provided for updates to predictions of Mine-related aquatic effects based on AEMP results to date. With few exceptions, observed Mine-related effects were consistent with EA predictions (DDMI 1998b), as summarized below by monitoring component:

- **Dust:** Deposition rates of dust measured since 2001 have exceeded those predicted by the air quality modelling in the EA. Air quality model predictions focussed on conservatively estimating effects to air quality (rather than deposition) during the operating phase of the Mine. Predictions did not account for construction activities, which occurred in parallel with mining.
- **Water quality:** Results of the AEMP and SNP are generally consistent with EA predictions related to water quality at the mixing zone boundary, TDS concentration in the FF areas of Lac de Gras, and effects from dust deposition. The majority of variables were consistently below Effects Benchmarks at the mixing zone boundary during the AEMP monitoring period from 2002 to 2016. The vast majority of TDS concentrations are less than those predicted by the modelling, and the slope of the increasing trend in the FF areas is notably shallower than predicted in the EA. Elevated concentrations of TSS observed at MF3 stations in 2016 were below the EA prediction of a 3 mg/L increase from dust deposition, and more likely reflect the effect of dike construction.
- **Eutrophication indicators:** The EA prediction for TP at the mixing zone boundary has not been exceeded. The predicted extent of the lake area that would be subject to TP concentrations above 5 µg/L (i.e., up to 20% of lake area) has not been exceeded in open-water conditions, but has been exceeded on two occasions during the ice-cover season (2008 and 2013). Concentrations of TP greater than the normal range have not occurred in an area greater than 20% of the lake. An increase in chlorophyll *a* concentration, indicating a shift in trophic status, has not been observed in any part of Lac de Gras at any time during AEMP monitoring.
- **Sediment quality:** No predictions were made in the EA regarding sediment quality.
- **Plankton:** No specific predictions were made in the EA regarding plankton communities, other than increased primary productivity resulting from the input of nutrients from the Mine effluent discharge. Increased phytoplankton biomass was observed in NF and MF areas of Lac de Gras, above the normal range in 2009 to 2013 and 2016, which is consistent with EA predictions regarding primary productivity. Occasional peaks in zooplankton biomass in 2009 and 2011 were consistent with increased phytoplankton productivity.
- **Benthic invertebrates:** The results of the benthic invertebrate analysis are consistent with the EA prediction of increased primary productivity in part of Lac de Gras resulting from nutrient inputs from the Mine discharge.
- **Fish health and tissue chemistry:** The EA provided no predictions for effects on fish health, and predicted that mercury concentration in sport and subsistence fisheries would remain below a mean of 0.2 µg/g wwt (unadjusted for length). The prediction was exceeded in Lac de Gras in 2004, from 2008 to 2012 and again in 2015. The general pattern in the raw (unadjusted) mercury fish tissue data suggests tissue mercury concentrations increased over time, then declined in both Lac de Gras and Lac de Sauvage.
- **Weight-of-evidence:** The EA predicted that overall, Lac de Gras water would remain at a high quality for use as drinking water and by aquatic life. The main impact was expected to be the introduction of

greater concentrations of nutrients, particularly phosphorus, with a concomitant increase in primary productivity over a portion of the lake. The WOE evaluation results are consistent with this prediction.

- **Traditional ecological knowledge:** The EA predicted that no adverse effect on fish quality (i.e., texture, taste) in Lac de Gras would occur as a result of tainting through the introduction of chemicals or fuels at the Mine site. Results of the fish palatability and texture studies have indicated that overall fish quality has not changed since baseline studies.

The above summary suggests that after 15 years of monitoring under the AEMP, the primary effects predicted by the EA are either being observed as predicted, or observed effects tend to be of lower magnitude than predicted. This is as expected, because EA predictions tend to be conservative. Occasional observations above EA predictions have occurred and potentially represent year-to-year variation in measurement endpoints. Updates to the EA predictions of Mine-related effects are not warranted at this time. In the case of the one AEMP component where predictions have been exceeded (Dust), observed effects in the receiving environment are not a magnitude that would warrant the modelling effort required to update predictions.

14 PROPOSED UPDATES TO AEMP DESIGN PLAN

The *AEMP Design Plan 4.1* (Golder 2017d) is currently in effect and has been implemented during the 2017 annual monitoring program. The re-evaluation presented herein has identified a number of aspects of the AEMP design that require updating. Overall recommendations for the AEMP sampling design and recommended changes to individual monitoring components are provided in the following sections.

14.1 Sampling Design

The AEMP was originally designed as a “hybrid” sampling program that allowed both control-impact and gradient analysis of effects (DDMI 2007). Recent AEMP annual monitoring results and re-evaluation results presented herein indicate that Mine effluent-related changes in water quality are apparent throughout Lac de Gras, and therefore, a control-impact analysis is no longer appropriate in annual analyses of Mine effects. Before-after analyses between new data and the 2007 to 2010 FF reference area data remain appropriate, based on the assumption that the FF1, FFA and FFB areas were similar to other lake areas under background conditions in terms of water quality and biological characteristics.

The overall recommendation regarding sampling design is that the AEMP should be converted to a gradient design, with corresponding changes to sampling locations and data analysis methods. This recommendation is consistent with that provided by ECCC in its review of the *2016 AEMP Annual Report*. This recommendation entails the following changes to be made in the AEMP Design Plan 5.0:

- Adjust sampling locations to (1) eliminate redundant replicate stations in the FF1, FFA and FFB areas, while retaining those stations that are already appropriately positioned along the effluent exposure gradient, and (2) establish new stations to fill spatial gaps between FF areas. A sufficient number of stations (i.e., minimum of three) need to be retained in the FFA area to allow an evaluation of escalation of effects towards the Significance Threshold, which is based on FFA area data. This reallocation of stations is likely to result in a reduction in the number of AEMP stations in Lac de Gras from the current 34 stations to 29 stations.

- Develop data analysis methods for evaluating Mine-related effects using the gradient design. Gradient analysis methods (e.g., linear regression) have already been employed by the benthic invertebrate component during previous annual data analyses to provide input to WOE analysis, and are being used to analyze 2017 AEMP data. Formalization of the analysis methods is required in the AEMP Design Plan 5.0 in a form that will be applicable to all AEMP components that sample according to the adjusted sampling design.

14.2 AEMP Component Details

14.2.1 Dust

The WLWB requested a critical review of the present sampling plan for dust, and requested DDMI consider revisions to the existing program based on the findings presented herein. Potential revisions requested by the WLWB (Table 1-1) included the following:

- Consider the implementation of additional dustfall and/or snow water chemistry monitoring sites (W2015L2-0001 update, Commitment A)
- Review the location and number of duplicate and blank samples for the dustfall and the snow water chemistry program (W2015L2-0001 update, Commitment B)

The current number and location of the dustfall and snow water monitoring locations is sufficient to evaluate both the spatial and the temporal trends of dust deposition (e.g., Figure 3-10 and 3-17); nutrient deposition (Figure 3-11 to 3-14 and 3-18); and metal deposition (Figures 3-15, 3-16 and 3-19) in the vicinity of the Mine. Consequently, no additional monitoring locations are recommended at this time.

The relative percent differences among sample duplicates are occasionally high for the snow water chemistry data. This is commonly observed for variables related to dust due to the episodic nature of dust deposition, and the discrete but stochastic nature of particle deposition [11]. Geometric averaging of samples plus their duplicates in the 2014 to 2016 re-analysis effectively accounted for occasionally high RPD values observed between duplicate snow chemistry samples collected at the same location. Potential outliers were effectively screened using the Z-score approach, and consequently no additional sample duplicates are proposed.

14.2.2 Fish

The following recommendations are made for the Fish Program based on the re-evaluation or directives from the WLWB.

[11] Consider, for example, that the deposition of approximately 1000, 1-µm dust particles is required to equal the deposition of a single, 10-µm dust particle. If the composition of the more numerous 1-mm particles in a sample duplicate differ from that of a single 10-mm particle found in the original sample, then large differences in the chemistry can occur.

14.2.2.1 Non-lethal Slimy Sculpin Survey

The current AEMP design is focussed on lethal sampling of Slimy Sculpin in the fall. By necessity, this program is targeted and non-random, with field effort aimed at trying to capture unparasitized juvenile and adult fish (i.e., larger fish). However, to better understand possible effects of the Mine effluent on fish abundance (i.e., CPUE) and overall reproductive success, additional data from random sampling of the population in each study area is required. Reproductive success is typically measured by direct measures of the gonads such as egg size and fecundity immediately prior to spawning, which is not possible for fall-sampled Slimy Sculpin, when gonads are undergoing development for spring spawning (Barrett et al 2010). Indirect measures such as abundance of young/small fish or presence/absence of young/small fish may be used as a surrogate (Environment Canada 2012).

For Lac de Gras, this would require a small addition to the field program, whereby the first fish sampling done in a given year would be a random field sampling effort at each of the study areas documenting each fish captured, before then moving to a targeted lethal program capturing adults and juveniles. Length, weight and body condition would be measured non-lethally from each fish captured. As noted above, two additional endpoints would be obtained from this additional non-lethal effort: CPUE and length-frequency histograms for each area. These two metrics would be based on random sampling, which is more appropriate than attempting to use the data from targeted (i.e., biased) sampling for adults.

There are limitations inherent in the use of any type of fish gear, however, and for electrofishing small fish in boulder-cobble lake habitat, it is likely the smallest size of fish (i.e., YOY and short juveniles) will still be under-represented by this sampling method. Given this, it is recommended that the smaller size classes in the length-frequency histogram be examined for presence/absence but not be considered as part of the WOE.

14.2.2.2 Study Frequency

Consistent with the previous study design, the Slimy Sculpin survey should continue at a frequency of once every three years to balance the lethal effects of the program on the local population against the AEMP sampling requirements. It is recommended, however, if two consecutive sampling events demonstrate that toxicological effects are not observed (i.e., Action Level 3 has not been triggered [Table 9-4]), then the next Slimy Sculpin survey would take place in six years. This schedule is consistent with the federal environmental effects monitoring (EEM) program for metal mines (Environment Canada 2012). If fish health assessment endpoints demonstrate effects equivalent to Action Level 3 (Table 5.2-4), a Lake Trout survey would be conducted, if appropriate. The specific timing of a Lake Trout fish health survey, however, would be defined in an AEMP Response Plan, which would be implemented as and when approved by the WLWB.

14.2.2.3 Updates to WOE Endpoints for Fish

Parasitism

Parasitism should be removed from the WOE as an endpoint for the fish population health LOE. While parasitism should continue to be reported and discussed as part of the fish program, it is not a valid LOE, as measured, to directly determine if there is a toxicological or nutrient-related effect.

Growth

Total length, fresh weight, and/or carcass weight should be added as endpoints to the WOE analysis instead of size at age, which cannot be determined because age data are not available for Slimy Sculpin (due to the difficulties of interpreting ages using otoliths with this species). Total length, fresh weight, and/or carcass weight are valid endpoints for growth as per the MMER TGD (Environment Canada 2012). Population Structure – Size should also be removed as an endpoint, because length and weight are more appropriate to use as growth and energy use metrics.

Population Structure

Survival, and Reproductive Investment – Age-1+ Abundance should both be removed from the WOE endpoints for fish population health. Neither of these endpoints can be accurately determined due to difficulties in capturing the smallest size classes (e.g., fishing gear bias), and while length-frequency and presence/absence analysis for the smallest size class may be considered as a surrogate for Age-1+ abundance, they should not be included as part of the WOE.

14.2.3 Water Quality

Effects Benchmarks adopted for the AEMP are based on CWQG for the protection of aquatic life (CCME 1999a + updates), the Canadian Drinking Water Qualty Guidelines (Health Canada 1996, 2006), guidelines from other jurisdictions (e.g., provincial and state guidelines), adaptations of general guidelines to site-specific conditions in Lac de Gras (Appendix IV.1 in DDMI 2007), or values from the scientific literature. The CWQG included in the Effects Benchmarks were updated between AEMP Study Design Versions 3.5 and 4.1.

Given WLWB Directive 4. related to the 2016 AEMP Response Plans to include relevant updates to CWQG as part of the proposed changes to the AEMP Design (Table 1-1), the Effects Benchmarks based on CWQG were reviewed (CCME 1999c). One update was identified: the Effect Benchmark for total silver will change from 0.1 µg/L to 0.25 µg/L.

14.2.4 Sediment Quality

In 2016, the WLWB provided a directive related to the sediment sampling approach in the AEMP Design Plan Version 4.1. The directive pertained to a previous WLWB directive for the AEMP Design Plan, which recommended, based on guidance in the MMER TGD (Environment Canada 2012), that DDMI collect five replicate sediment samples (each being a composite) at each AEMP station (Table 1-1). However, there was a misunderstanding, and in the AEMP Design Plan Version 4.1, DDMI increased the number of grabs included in the composite sample collected at each station (from three to five), instead of increasing number of composite samples collected at a station from one to five.

The recommendation from ECCC was considered further. The current approach of collecting single composite samples, and increasing the number of grabs per sample to five is considered adequate to address the objectives of the AEMP sediment quality component, and is expected to improve representativeness of composite samples collected at each station. A nearly five-fold increase in sediment sampling and analytical effort, as recommended by WLWB, is not justified based on AEMP results to date. The sediment quality dataset summarized herein demonstrates the ability to detect change associated with

Mine-related inputs to Lac de Gras. In addition, data for the additional recommended samples would not increase the sample sizes used in statistical comparisons, because data would be averaged per station for among-area comparisons and trend analysis. The five samples from a station would not be considered replicates; the AEMP sampling design considers stations as replicates. Up to this point, the current approach has been sufficient to detect and describe Mine-related effects on concentrations of sediment quality variables. Effects to date have been limited mostly to three variables (i.e., bismuth, uranium and lead) and the current sampling approach provides data that allow identification of effects both statistically and upon visual evaluation of spatial trends. Therefore, it is recommended the sediment sampling approach described in the AEMP Design Plan 4.1 remain unaltered.

14.2.5 Eutrophication Indicators

DDMI committed as part of the 2015 AEMP Annual Report review process to evaluate variation over time in eutrophication indicators and, if applicable, make recommendations regarding other information that could be collected in future years to explain variation in extent of effects in Lac de Gras (Commitment #29; Table 1-1). Variation over time in eutrophication indicators was evaluated in Section 5.3.1.1 and no additional information was identified that would assist with explaining variation in extent of effects. Therefore, no recommendation related to this request is provided at this time.

The WLWB requested DDMI consider (1) additional data collection to help explain chlorophyll *a* effects, and (2) more explicit analysis of the role of nitrogen in explaining variation and the spatial extent of chlorophyll *a* effects (Directive F re. W2015L2-0001 update; Table 1-1). One recommendation is provided regarding additional data collection to help explain chlorophyll *a* effects (see addition of soluble reactive silica as a variable below); data on concentrations of other important nutrients and supporting variables related to chlorophyll *a* are already being collected by the AEMP. The role of nitrogen in explaining variation and the spatial extent of chlorophyll *a* effects was considered in Sections 5.3.5 and 5.3.7.

Three recommendations are made for the Eutrophication Indicators component based on data interpretation completed as part of this re-evaluation:

- It is recommended that bicarbonate and pH be discontinued as variables reported in the Eutrophication Indicators annual report These variables are unlikely to explain variation in nutrients and biological indicators of productivity, and will continue to be reported by the Water Quality component.
- Consistent with EMAB comment 84 on the 2016 AEMP Annual Report (Table 1-1), it is recommended that soluble reactive silica be added as a new variable for the Eutrophication Indicators component. Results presented herein indicate that variation in concentrations of micronutrients over time and across Lac de Gras may influence primary productivity. Addition of this variable may assist with explaining variation in biological indicators of productivity among AEMP sampling areas and over time.
- Sampling of Eutrophication Indicators variables other than field parameters, nutrients and chlorophyll *a* at LDS-4 (the Narrows) is recommended to be discontinued, as this location is shallow with flowing water, and not appropriate for plankton sampling.

14.2.6 Plankton

The WLWB requested that DDMI make a recommendation regarding sampling depth for phytoplankton (W2015L2-001 update, Reviewer Comments Applicable to Other Submissions #2b; Table 1-1). It is recommended that the sampling depth for phytoplankton remain the same to allow continued temporal comparisons. The depth-integrated samples collected within the top 10 m of the water column should capture a large proportion of the community, as Secchi depths generally range from approximately 5 to 15 m in Lac de Gras (Section 5).

As part of the WLWB approval of the 2016 AEMP Annual report (Directive 3C re. W2015L2-0001 update), DDMI was directed to continue to collect phytoplankton samples at the MF stations during the interim sampling years, as initiated in 2017. It is recommended that the AEMP Design 5.0 reflect this change.

Use of the 2013 normal range for phytoplankton variables is recommended for comparisons with data collected from 2013 onwards, thereby limiting comparisons to data generated by the same taxonomist. However, an adjustment to the 2013 normal range to account for year-to-year variability is required prior to implementing this recommendation. The 2013 normal range may be adjusted by one or a combination of the following procedures: (1) expanding the percentiles used for the normal range calculation to account for year-to-year variation; (2) applying the among-year variability observed between 2007 and 2010; or (3) including the 2016 FF area data in the 2013 normal range calculation. The adjustment procedure will be documented in the 2017 AEMP Annual Report.

As described in Section 14.2.5, it is recommended that plankton sampling at LDS-4 (the Narrows) should be discontinued, as this location is a shallow with flowing water, and not appropriate for plankton sampling.

14.2.7 Benthic Invertebrates

There are no recommended changes to the AEMP design for the benthic invertebrate community sampling and data analysis, beyond the shift to a gradient sampling design (Section 14.1).

One recommendation is provided for variables included as WOE endpoints for benthic invertebrates. Non-directional benthic invertebrate variables (i.e., dominance, SDI, SEI, BCI) should be removed from the list of variables with WOE ratings. These four variables do not add useful information to the analysis beyond indicating a community change, which would continue to be considered as supporting information in the WOE analysis.

14.2.8 Weight-of-Evidence

Updates to the endpoints and LOE presented in the Design Plan Version 4.1 (Golder 2017d) may be required for those components that are recommending additional endpoints (e.g., the Eutrophication Indicators component is recommending analysis of soluble reactive silica) or removal of endpoints (e.g., the Fish Health component is recommending removal of several endpoints due to difficulty in measuring, or because of questionable relevance to the WOE assessment, and the benthic invertebrate component is recommending removal of non-directional variables with WOE ratings). In addition, several components are recommending changes to the statistical evaluation of Mine-related effects in Lac de Gras (e.g., from statistical comparisons between NF and FF areas to a gradient analysis).

The transition to a gradient design will have implications to the WOE Effect Ratings, which are not explicitly defined in the AEMP design document. The necessary changes to the Effect Ratings will be developed during the next comprehensive report (i.e., 2019) and implemented at that time.

Although the recommended changes by other components may result in some inconsistency in WOE assessment methods among versions of the AEMP, continued retroactive WOE evaluation (as completed herein) is not recommended in future re-evaluation reports. Changes to AEMP design and methods reflect the evolution of effects in Lac de Gras over time and/or best professional judgement, necessitating updated field approaches. However, the WOE methods for a given AEMP Version are considered appropriate for the effects being observed, aside from the minor adjustments to ratings and weightings made to date. For example, control-impact analysis has been appropriate for evaluating early effects of the Mine, which have not spread across the lake, whereas a gradient analysis is considered appropriate going forward, because effects on water quality are observed throughout the lake. This should not result in gradient re-analysis of all previous years of monitoring data in a WOE assessment.

In their Reasons for Decision, WLWB (2017e; Section 3.12, Part 2e) recommends that benthic macroinvertebrate density be added as a nutrient enrichment exposure endpoint of the fish population health ecosystem component, or that rationale be provided for the omission (Table 1-1). Chlorophyll *a* is currently included as a nutrient enrichment exposure endpoint for the fish population health ecosystem, which is intended to be indicative of food supply. It is assumed that an increase in the biomass of algae as measured by chlorophyll *a* provides an early indication of an enrichment-related increase in zooplankton and/or benthic invertebrate food supply for fish. Adding benthic invertebrate abundance or density would be redundant considering there is already a conservative measurement of enrichment-related food supply in the WOE analysis. Moreover, the benthic invertebrate community samples are collected from deep-water stations and as such the abundance or density from these samples are not representative of food supply for shallow-water, shoreline-dwelling Slimy Sculpin.

14.2.9 Traditional Ecological Knowledge

The WLWB directed DDMI to provide an explanation for if, and why, the comparison of tissue metal concentrations to baseline concentrations should be considered differently in the future (Table 1-1). No changes are recommended at this time, as current practice as presented herein is appropriate to meet the needs of the fish tissue chemistry analyses (Section 9) and the palatability study as part of the TK component (Section 11).

14.3 Changes to Action Levels

14.3.1 Water Quality

On 24 January 2018, the WLWB issued a Directive requiring DDMI to address the following items relating to water quality (WLWB 2018; Table 1-1):

1) DDMI is to address GNWT-ENR recommendations 10 and 11, with regards to changes to Action Levels for effluent and water chemistry, as part of the *2014 to 2016 Aquatic Effects Re-evaluation Report*

2) DDMI is to include relevant updates to Canadian Water Quality Guidelines as part of the proposed changes to the AEMP Design to be submitted with the *2014 to 2016 Aquatic Effects Re-evaluation Report*

Item (1) is addressed below, while Item (2) is addressed in Section 14.2.2.

14.3.1.1 Action Level 1

The GNWT-ENR comment 10 references the WLWB decision requiring DDMI to include a spatial-temporal assessment of all analyzed variables herein, and further recommends that an increasing temporal trend in the mixing zone, NF area, and the MF1 and MF2 areas should be incorporated into Action Level 1 (WLWB 2018). Results of the spatial-temporal assessment have been included in Section 4.3.2.1.2 for SOIs and Appendix 4C for other variables (i.e., non-SOIs). The following text addresses the inclusion of temporal trends into Action Level 1.

In the AEMP Response Framework for the Mine, a water quality variable triggers Action Level 1 if the median concentration in the NF area is greater than two times the median reference concentration (as defined in the *AEMP Reference Conditions Report Version 1.2* [Golder 2017b]). In addition, the increase in concentration in the NF area has to be linked to the Mine to trigger Action Level 1. The intent of Action Level 1 is to act as an early warning indicator and identify a short-list of SOIs that should be evaluated in greater detail within the annual AEMP report. No specific action is required upon an Action Level 1 trigger for water quality.

The current Action Level definitions are relevant to toxicological impairment and are designed to first identify increasing concentrations in the NF area. Once nearly all NF area concentrations are above the normal range (as indicated by the 5th percentile in the Action Level 2 definition), the Action Levels switch to the edge of the mixing zone, and incorporate comparisons to the Effects Benchmarks. The intent of this approach is to allow for some increase in concentrations the NF area and beyond, but ensure that concentrations approaching the Effects Benchmark do not reach beyond the mixing zone. This approach focusses on a small spatial scale, because it is considered unacceptable to have toxic effects in the receiving water.

The approach proposed by GNWT-ENR, to set Action Levels according to increasing temporal trend, is inappropriate for this purpose for three key reasons:

1. It ignores the magnitude of change in concentration in the receiving waters. Increasing trends may be detected within the normal range, may be of low magnitude in terms of concentration increase, and may represent background year-to-rear variation. Additionally, based on the composition of the Mine effluent, increasing trends in the receiving waters are expected, and should only be considered to be of concern if concentrations reach a magnitude that is of concern based on a pre-defined criterion.
2. The annual Action Level assessment needs to be based on the actual magnitude of measured concentrations in the year being evaluated, rather than on trends that are influenced by previous years' data. Given a long-term dataset, it is possible to have a concentration increase in one year without a significant trend being detected.
3. The proposed Action Level approach would result in a list of SOIs that would not be substantially different from those identified using the current method (see information provided below), but would unreasonably increase the time and effort necessary to complete the response planning analyses. Given the number of variables analyzed, trend analysis is not a trivial exercise and is currently required as part of aquatic effects re-evaluation at three-year intervals.

Of the 49 variables in Table 14-1, 17 SOIs were identified for 2016 based on triggering Action Level 1 in the NF area (Table 14-1, columns 4 and 5; also see Section 4.2.2), while 18 variables were identified as having increasing trends within the NF, MF1 and/or MF2 areas (Table 14-1). Of the 18 variables identified with increasing trends, 10 triggered Action Level 1, indicating that the current Action Level 1 is already inclusive of about half of the variables with increasing trends over time. Of the remaining eight variables that did not trigger Action Level 1 but had increasing trends, fluoride, cobalt and iron were identified as SOIs, because they triggered other SOI selection criteria in 2016 (see Section 4.2.2). While dissolved calcium did not trigger Action Level 1, total calcium did; therefore, calcium was included as an SOI. Dissolved and total magnesium and potassium were the remaining variables that had increasing trends, but did not trigger Action Level 1 in 2016. Five variables (i.e., turbidity, nitrate, aluminum, copper, uranium) were flagged as SOIs because they triggered the Action Level 1 in 2016, but they did not have increasing trends. If Action Level 1 had been based on increasing trends over time, these variables would not have been identified as SOIs, indicating that the current Action Level is, in fact, more inclusive in selecting SOIs.

Given the above reasons and comparisons between the current Action Level 1 and one based on temporal trends, as suggested by GNWT-ENR, no changes to the current Action Level 1 for water quality are recommended. The existing Action Level 1 definition is appropriate for identifying a short-list of variables (i.e., SOIs) for further analysis, and should be retained as is for use in future AEMP data evaluation.

Table 14-1 Evaluation of Variables that Have Temporal Trends and Trigger Action Level 1 and Action Level 2 (Current and as Proposed by GNWT-ENR), 2016

Variable	Unit	Temporal Trend[(a)] (Yes/No)	Current Action Level 1 Triggered (Yes/No)		Current Action Level 2 (5th Percentile) Triggered (Yes/No)		Proposed Action Level 2 (75th Percentile) Triggered (Yes/No)	
			Ice-cover	Open-water	Ice-cover	Open-water	Ice-cover	Open-water
Conventional Parameters								
Total dissolved solids, calculated	mg/L	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
Total alkalinity	mg/L	No	No	No	No	No	**Yes**	**Yes**
Total suspended solids	mg/L	No	No	No	No	No	No	No
Total organic carbon	mg/L	No	No	No	No	No	No	No
Turbidity – lab	NTU	No	**Yes**	No	No	No	No	**Yes**
Major Ions								
Calcium (dissolved)	mg/L	**Yes**	No	No	No	No	**Yes**	**Yes**
Chloride	mg/L	**Yes**	**Yes**	**Yes**	**Yes**	No	**Yes**	**Yes**
Fluoride	mg/L	**Yes**	No	No	No	No	**Yes**	**Yes**
Magnesium (dissolved)	mg/L	**Yes**	No	No	No	No	**Yes**	**Yes**
Potassium (dissolved)	mg/L	**Yes**	No	No	No	No	**Yes**	**Yes**
Sodium (dissolved)	mg/L	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
Sulphate	mg/L	**Yes**	**Yes**	**Yes**	No	No	**Yes**	**Yes**
Nutrients								
Ammonia	µg-N/L	n/a[(b)]	**Yes**[(b)]	**Yes**[(b)]	**Yes**[(b)]	**Yes**[(b)]	**Yes**[(b)]	**Yes**[(b)]
Nitrate	µg-N/L	No	**Yes**	**Yes**	**Yes**	No	**Yes**	**Yes**
Nitrite	µg-N/L	n/a	No	No	No	No	**Yes**	No
Total Metals								
Aluminum	µg/L	No	**Yes**	**Yes**	No	No	**Yes**	**Yes**
Antimony	µg/L	No	No	No	No	No	**Yes**	No
Arsenic	µg/L	No	No	No	No	No	**Yes**	**Yes**
Barium	µg/L	No	No	No	No	No	**Yes**	**Yes**
Beryllium	µg/L	n/a	No	No	No	No	No	No
Bismuth	µg/L	n/a	No	No	No	No	No	No
Boron	µg/L	n/a	No	No	No	No	No	No
Cadmium	µg/L	n/a	No	No	No	No	No	No
Calcium	mg/L	**Yes**	**Yes**	No	No	No	**Yes**	**Yes**
Chromium	µg/L	n/a	No	No	No	No	**Yes**	**Yes**
Cobalt	µg/L	**Yes**	No	No	No	No	**Yes**	No
Copper	µg/L	No	**Yes**	**Yes**	No	No	No	**Yes**
Iron	µg/L	**Yes**	No	No	No	No	No	**Yes**
Lead	µg/L	**Yes**	No	**Yes**	No	No	No	**Yes**
Lithium	µg/L	No	No	No	No	No	**Yes**	**Yes**
Magnesium	mg/L	**Yes**	No	No	No	No	**Yes**	**Yes**
Manganese	µg/L	**Yes**	**Yes**	No	No	No	**Yes**	**Yes**
Mercury	µg/L	n/a	No	No	No	No	No	No
Molybdenum	µg/L	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
Nickel	µg/L	No	No	No	No	No	No	No
Potassium	mg/L	**Yes**	No	No	No	No	**Yes**	**Yes**
Selenium	µg/L	n/a	No	No	No	No	No	No
Silicon	µg/L	n/a	**Yes**	No	No	No	**Yes**	No
Silver	µg/L	n/a	No	No	No	No	No	No
Sodium	mg/L	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
Strontium	µg/L	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
Sulphur	mg/L	No	No	No	No	No	**Yes**	**Yes**
Thallium	µg/L	n/a	No	No	No	No	No	No
Tin	µg/L	No	No	No	No	No	No	No
Titanium	µg/L	n/a	No	No	No	No	No	No
Uranium	µg/L	No	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
Vanadium	µg/L	n/a	No	No	No	No	No	No
Zinc	µg/L	No	No	No	No	No	No	No
Zirconium	µg/L	n/a	No	No	No	No	No	No

a) Presence of temporal trends was evaluated based on statistical trends in SOIs as described in Section 4.3.2.1.2 and visual inspection of mid-depth plots presented in Appendix 4C for non-SOI variables.

b) Results for ammonia are uncertain due to laboratory quality control issues.

NTU = nephelometric turbidity unit; n/a = not applicable; µg-N/L = micrograms nitrogen per litre.

14.3.1.2 Action Level 2

In the *2013 AEMP Annual Report* (Golder 2014c), Action Level 2 for water quality was triggered when the 75th percentile of NF values was greater than the normal range in Lac de Gras. Action Level 2 was subsequently revised, because it was being triggered prior to Action Level 1 (WLWB 2015c). The revised Action Level 2 is triggered if the 5th percentile concentration in the NF area is greater than two times the median concentration in reference datasets and is greater than the normal range for Lac de Gras. The revised Action Level 2 was applied successfully as part of the analyses completed for the *2011 to 2013 Aquatic Effects Re-evaluation Report* (Golder 2016a) and the 2014, 2015, and 2016 Annual Reports (Golder 2016b,c, 2017c), and resulted in sequential triggering of Action Levels. Under the Response Framework, when a water quality variable triggers Action Level 2, the required management action is to establish an AEMP Effects Benchmark for that variable, if one does not already exist.

The revisions to Action Level 2 were proposed as an update in the *AEMP Design Plan Version 4.0* (Golder 2016d). Formal review of the revised Action Level 2 occurred as part of the approval process for the *AEMP Design Plan Version 4.0* (Golder 2016d). The revisions to Action Level 2 were approved by the WLWB on 22 December 2015 (WLWB 2015c).

As outlined in the January 2018 directives, during the public review, GNWT-ENR comment 11 recommended that Action Level 2 should return to what was previously used and that a change to Action Level 1 be considered instead (WLWB 2018). As illustrated in Table 14-1 for data from 2016, the previous Action Level 2 is extremely conservative. When Action Level 2 is based on the 75th percentile of the NF values being greater than the normal range, 29 of the 49 variables (i.e., 60%) trigger the Action Level, and often before Action Level 1 is triggered. The reason for this conservatism is that the 75th percentile-based Action Level 2 triggers when most (75%) of the measured concentrations are still within the normal range. Given this level of conservatism, designating an Action Level 1 would be problematic, given the limited concentration “room” available between the reference data set and the 75th percentile-based Action Level 2. The action defined when reaching Action Level 2 is to develop an Effects Benchmark for the affected variable, if one does not exist. Reverting to the 75th percentile-based Action Level 2 would result in developing Effects Benchmarks for a number of additional variables, despite there being little concern related to toxicological effects from the Mine effluent discharge for those variables, now or in the future.

As presented in 14.3.1.1, the current Action Level 1 is effective at identifying a short-list of SOIs where more detailed analysis can be focused, and the SOI selection process is robust in identifying variables of potential concern. No change to the current Action Level 2 is recommended at this time.

14.3.2 Biological Action Levels

14.3.2.1 Lower Trophic Levels

The approved Action Levels for plankton and benthic invertebrates from the *AEMP Study Design Version 3.5* (Golder 2014a) are provided in Tables 7-8 and 8-4, respectively. Currently, biological Action Levels 1 and 2 include a comparisons to the FF areas, which is no longer a suitable comparison, given that the FF areas have become exposed to Mine effluent in recent years (Golder 2016a,b,c). As per WLWB (2017a), an update to the statistical comparisons detailed in biological Action Levels 1 and 2 are required as part of this re-evaluation report (Table 1-1). Changes are also proposed to Action Level 3 for these components, to better track escalation of effects and enhance consistency among biological monitoring components. The proposed updates to Action Levels 1, 2 and 3 are provided in Table 14-2, and will be further described in the AEMP Study Design Version 5.0.

14.3.2.2 Fish

The approved Action Levels from *AEMP Study Design Version 3.5* (Golder 2014a) are defined in Table 9-4. As per WLWB (2017a), an update to the statistical comparisons detailed in biological Action Levels 1 and 2 are required as part of this re-evaluation report. Changes are also proposed to Action Level 3 for fish to better track escalation of effects, enhance consistency among biological monitoring components, and better align with EEM requirements. The proposed updates to Action Levels 1, 2 and 3 are provided in Table 14-2, and will be further described in the AEMP Study Design Version 5.0.

Table 14-2 Biological Action Levels for Toxicological Impairment

Action Level	Plankton	Benthic Invertebrates	Fish[(a)]	Extent	Action
1	Mean phytoplankton or zooplankton biomass statistically significantly less than mean of reference dataset	Total invertebrate density, richness, or densities of dominant invertebrates statistically significantly less than mean of reference dataset	Effect indicators[(a)] statistically significantly different from mean of reference dataset	NF	No action
2	Mean phytoplankton or zooplankton biomass statistically significantly less than mean of reference dataset in two consecutive years	Total invertebrate density, richness, or densities of dominant invertebrates statistically significantly less than mean of reference dataset, with an effect size equal to or above the critical effect size defined by EEM	Effect indicators[(a)] statistically significantly different from mean of reference dataset, with an effect size equal to or above the critical effect size defined by EEM, that is indicative of an impairment to fish health and can be linked to the Mine.	NF	No action, confirm effect
3	Mean phytoplankton or zooplankton biomass statistically significantly less than mean of reference dataset AND Observed in three consecutive sampling events AND Below the normal range[(b)]	Total invertebrate density, richness, or densities of dominant invertebrates statistically significantly less than mean of reference dataset, with an effect size equal to or above the critical effect size defined by EEM AND Observed in two consecutive sampling events AND Below the normal range[(b)]	Effect indicators[(a)] statistically significantly different from mean of reference dataset, with an effect size equal to or above the critical effect size defined by EEM, that is indicative of an impairment to fish health and can be linked to the Mine. AND Observed in two consecutive sampling events AND Beyond the normal range[(b)]	NF	Investigation of Cause/Response Plan Initiate large-bodied fish survey, if appropriate Set Action Level 4 Examine Ecological significance, including an assessment of plankton edibility Identify mitigation options
4	TBD[(c)]	TBD[(c)]	TBD[(c)]	TBD[(c)]	TBD[(c)]
5	TBD[(c)]	TBD[(c)]	TBD[(c)]	TBD[(c)]	TBD[(c)]
Significance Threshold	Decline in biomass likely to cause a >20% change in fish population(s)	Decline in invertebrate density likely to cause a >20% change in fish population(s)	Indications of severely impaired reproduction or unhealthy fish likely to cause a >20% change in fish population(s)	Far-field A (FFA)	

a) Effect indicators are modified from the EEM program (i.e., length, weight, condition, relative liver size and relative gonad size).
b) Normal range as defined and approved in the *AEMP Reference Conditions Report Version 1.2* (Golder 2017b).
c) To be determined (TBD) if Action Level 3 is triggered.

15 ASSESSMENT OF RESPONSE FRAMEWORK PERFORMANCE

The objective of the AEMP Response Framework is to ensure that significant adverse effects never occur. This is accomplished by requiring proponents to take actions at defined Action Levels, which are triggered well before significant adverse effects could occur in the receiving environment. The AEMP has been effective at identifying Mine-related changes in chemistry and biological endpoints, and application of the Response Framework has resulted in numerous Action Level triggers. As a result, five Response Plans were prepared during the last re-evaluation period: one for each of water quality, eutrophication indicators, plankton, benthic invertebrate community, and fish.

Effectiveness of the Response Framework was evaluated based on whether Action Level triggers occurred at magnitudes of effect appropriate for the intent of the given Action Level (e.g., early-warning change for Action Level 1; change warranting further investigation for Action Level 2), direction of change (i.e., was a toxicological impairment Action Level triggered by a nutrient enrichment response?), and whether the response plans prepared were warranted by the changes resulting in Action Level triggers. In addition, an overall qualitative evaluation is provided regarding the effectiveness of the Response Framework to prevent the occurrence of significant adverse effects.

Effectiveness of the Action Levels are evaluated below for each relevant AEMP component.

Water Quality

Water quality Action Levels 1 and 2 for toxicological impairment were triggered during each year of monitoring, reflecting increases in concentrations above reference conditions for Lac de Gras. A single Action Level 3 trigger occurred in 2007, for total chromium; this occurred before the Response Framework was in effect, and did not occur again.

The frequency and severity of Action Level exceedances for water quality have increased over time, reflecting gradual changes in water quality in the NF area (Table 15-1). Both the number and percentage of total variables with Action Level triggers nearly doubled between 2007 and 2016. This indicates that the Action Levels for water quality are responsive to changing water quality conditions in the receiving environment.

Table 15-1 Frequency of Action Level Triggers for Water Quality Since 2007

Quantity	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Total variables	47	47	48	47	49	51	52	52	52	52
Number of Action Level 1 triggers	5	4	5	5	6	4	9	10	8	8
Number of Action Level 2 triggers	3	4	6	7	7	8	8	8	9	7
Number of Action Level 3 triggers	1	0	0	0	0	0	0	0	0	0
Total Action Level triggers	9	8	11	12	13	12	17	18	17	15
Percentage of variables with Action Level triggers	19%	17%	23%	26%	27%	24%	33%	35%	33%	29%

Note: Information in this table is based on Table 4-22.

Concentrations in Lac de Gras have been well below Effects Benchmarks, suggesting that triggering of Action Levels 1 and 2, representing early-warning and low level effects was appropriate. Concentrations in the most exposed areas of the lake (i.e., mixing zone boundary and NF area) also remain well below Effects Benchmarks. Since 2008, none of the variables triggered Action Level 3 (which is defined as an increase at the mixing zone that is greater than 25% of the distance between the top of the normal range and the Effects Benchmark). Based on these results, the number and level of Action Level triggers for water quality based on the currently approved Action Levels are consistent with the environmental quality of Lac de Gras and currently observed Mine effects.

Sediment Quality

Action Levels for sediment quality were approved as part of the *AEMP Design Plan 4.1*; therefore, Action Level triggers will first be evaluated during the next comprehensive AEMP year, in 2019.

Eutrophication Indicators

Chlorophyll *a* concentration triggered Action Level 1 or 2 for nutrient enrichment during each year of monitoring under the AEMP. Given the magnitude and spatial extent of the observed effect, these triggers were appropriate and resulted in an appropriate action (i.e., development of an Effects Benchmark).

Plankton

Action Level 1 for toxicological impairment was triggered by phytoplankton biomass in 2015 and Action Level 2 was triggered by zooplankton biomass (based on enumeration) in 2016. These triggers were appropriate, with the exception that the zooplankton biomass trigger was at a higher Action Level than reasonably appropriate, because it was a first trigger for this variable, and was not corroborated by another measure of biomass (AFDM). Recommended changes to the plankton Action Levels (Section 14.3.2.1) would result in at most Action Level 1 triggers for both of these variables based on data collected to date.

Benthic Invertebrates

One to three benthic invertebrate variables triggered Action Level 1 for toxicological impairment in 2008, 2009 and 2011. SEI triggered Action Level 2 and Pisidiidae density triggered Acton Level 1 in 2016. Richness triggers in each of 2008, 2009 and 2011, and the *Heterotrissocladius* density trigger in 2008 were appropriate early-warning signals for an Action Level 1. However, triggers by percent Chironomidae (2008) and SDI (2009) were more likely related to nutrient enrichment. The 2016 Response Plan concluded that both triggers in that year most likely originated from nutrient enrichment; however, given the direction of the trigger for Pisidiidae, the Action Level 1 trigger was warranted and required further evaluation.

Recommended changes to the benthic invertebrate Action Levels (Section 14.3.2.1) would retain Action Level 1 triggers for richness, *Heterotrissocladius* density, and Pisidiidae density, and eliminate the triggers for percent Chironomidae, SDI and SEI, which more likely reflect the effect of nutrient enrichment than a toxicological effect.

Fish

The first year that Action Levels for toxicological impairment were implemented for the fish health program was 2013. Action Level 1 was triggered in 2013, and Action Level 2 was triggered in 2016. The differences detected between the NF or MF area and the FF areas were consistent with early-warning changes and thus appropriate to trigger these Action Levels. The conclusions of the *2014 to 2016 AEMP Response Plan Fish – Supplemental Report* further supported the Action Level 2 early warning trigger. The conclusion of the Reponse Plan indicated that continued monitoring under the exisiting AEMP schedule (i.e., every three years) was recommended with no further action required. The results did not warrant initiation of a large-bodied fish survey.

Recommended changes to the fish Action Levels would result in similar Action Level triggers in both 2013 and 2016 as previously presented. Action Level 3 would not have been triggered in either 2013 or 2016, given that there were no endpoints outside of the normal ranges in either 2013 or 2016 (Section 9.2.1.3.4).

Overall Evaluation and Conclusion

This qualitative evaluation of the performance of the AEMP Response Framework indicates that it is functioning as intended, and is appropriate to support the management of Mine-related effects in Lac de Gras.

Since implementation of the Response Framework in 2013, and based on retroactive analysis for monitoring components with long-term data, low Action Levels (1 and 2) for both effect types have been triggered by early-warning or low magnitude changes, indicating an appropriate level of sensitivity to observed changes in Lac de Gras. The number and type of Action Level triggers for water quality are realistically tracking the escalation of effects in Lac de Gras from the Mine discharge. Most biological Action Level triggers have appropriately identified changes that warrant further investigation. However, some of the triggers of toxicological impairment Action Levels were attributed to nutrient enrichment, and resulted in recommendations to change those Action Levels. Recommended changes to Action Levels provided herein would eliminate most of the spurious Action Level triggers, and in most cases would result in triggers similar to those assigned based on the original Action Level system. Therefore, as per the WLWB Directives to re-examine Action Levels for water quality (Section 13.3.1) and biological variables (Section 13.3.2), the proposed updates to Action Levels are expected to maintain or improve the effectiveness of the Response Framework to detect Mine-related changes in Lac de Gras in the future.

Action Level triggers have resulted in Response Plans submitted to and approved by the WLWB, and the prescribed actions were completed by DDMI. As dictated by these Response Plans, effects were investigated further, or new Effects Benchmarks were developed. Literature-based investigations of the likely causes of Action Level triggers have contributed to a better understanding of the nature of effects of the Mine discharge in Lac de Gras (e.g., fish and benthic invertebrate Response Plans). The new Effects Benchmarks have been useful to further assess the magnitudes of observed effects. For example, the chlorophyll *a* benchmark developed after an Action Level 2 trigger is a useful tool to track escalation of the Mine-related nutrient enrichment effect in Lac de Gras.

Overall, the AEMP Response Framework has met its objectives to date. Since its implementation, most of its key elements have been successfully applied, even at the relatively low lever effects observed to date.

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