Review of Snap Lake AEMP

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Table 1: Acronym Definitions

Acronym	Definition
ANOVA	analysis of variance
EAR	environmental assessment review/report
FF	far-field
INAC	Indian and Northern Affairs Canada
ISQG	interim sediment quality guideline
MF	mid-field
MVEIRB	Mackenzie Valley Environmental Impact Review Board
MVLWB	Mackenzie Valley Land and Water Board
NF	near-field
PEL	probable effect level
SD	standard deviation
SLEMA	Snap Lake Environmental Monitoring Agency
SNP	surveillance network program
TDS	total dissolved solids
VEC	valued ecosystem component
WL	water license
WLWB	Wek'eezhii Land and Water Board

1 Introduction

Zajdlik & Associates Inc. has been contracted to review the Snap Lake Aquatic Effects Monitoring Program (AEMP) focusing on phytoplankton/zooplankton, sediment, benthic and fish health. In addition, the Snap Lake Environmental Monitoring Agency (SLEMA) has asked that the following more specific questions be addressed:

1. "Taking into account the five years of AEMP reporting, please make comment based on your opinion on whether the current AEMP program remains sufficiently robust to meaningfully monitor aquatic community health?

2. TDS level increases were predicted and have been realized in Snap Lake. Please make comment based on your opinion if these increases pose any threat to the relatively small water system such as Snap Lake.

3. The Traditional Knowledge panel has expressed a concern over the effects of blasting work under Snap Lake and the effects on fish. Please make comment based on your opinion how these effects may directly or indirectly impact the health of fish in Snap Lake.

4. DeBeers data shows that there are elevated levels of Mercury, Uranium, Rubidium, Thallium, Barium, Vanadium, Strontium, Molybdenum, Lithium, Arsenic, and Boron at the diffuser and the NF area. There are no predicted increases of any of these elements and actually many are not even listed in the list of environmental assessment review (EAR) elements. Please make comment based on your opinion of whether the continual discharge of these elements create a "hotspot" around the diffuser and what effects can be expected in Snap Lake.

5. Bring to SLEMA's attention anything that you feel may derogate the Snap Lake Aquatic community as well as anything that may be out of compliance with national or territorial standards or best practices."

1.1 Chronology

A brief chronology of relevant events provides context for the reports and data being reviewed.

1998-2001
- Baseline data collection

2002
- Additional water collection in Snap Lake

2003
- Additional water collection in Snap Lake

2004

- Begin construction
- June 2004 effluent discharge begins through temporary diffuser
- AEMP data collection begins May 2004 under an unapproved AEMP (DeBeers 2005a, pg. 12)
- Fisheries Authorization for the Snap Lake Project issued in August 2004

2005

- Phase II construction begins
- Snap Lake Working Group approves AEMP known as the July 2005 AEMP.

2006

- Permanent diffuser[1] operational May, 2006
- Water quality monitoring (but no sediment quality or benthic invertebrate community monitoring[2]) in reference lake (Northeast Lake) begins

2007

- Plume Characterization Special Study
- Sediment quality and benthic invertebrate community monitoring attempted in Northeast Lake but not completed for safety reasons.

2008

- Major construction activities (DeBeers, 2009).
- Picoplankton pilot study in Snap and Northeast Lakes.
- Sediment quality and benthic invertebrate community monitoring begins in Northeast Lake.

1.2 Review of Environmental Assessment Review Predictions

The AEMP program is designed in part to assess those changes predicted during the environmental assessment review (EAR). Thus predictions made in MVEIRB (2003) relevant to the objectives of this review (presented in section 1) are re-stated below. This section begins with definitions of levels of change because it is these definitions of change that determine whether there is an unacceptable effect of the Snap Lake Mine.

1.2.1 Quantifying Change

The adjectives "negligible", "low", "moderate" and "high" used to describe changes due to the Snap Lake Diamond mine are defined in MVEIRB (2003) and are quoted below.

- "Negligible if the water quality change would affect less than 5% of the aquatic community throughout Snap Lake or would affect more than 20% of the aquatic community in less than 1% of Snap Lake;

[1] Effluent comprised of treated minewater and domestic wastewater from sewage treatment system.
[2] Attempted but not possible due to safety and logistical challenges.

- Low if the water quality change would affect less than 10% of the aquatic community or would affect more than 10% of the aquatic community in less than 10% of Snap Lake;

- Moderate if the water quality change would affect more than 10% of the aquatic community in more than 10% of Snap Lake; and,

- High if the water quality change would affect more than 20% of the aquatic community in more than 20% of Snap Lake."

1.2.2 EAR Recommendations

The EAR conclusions and recommendations (MVEIRB, 2003) pertaining to the review of the AEMP questions posed by SLEMA in section 1 are presented below. Thus not all EAR conclusions or recommendations are included here.

1.2.2.1 The AEMP

From MVEIRB, (2003, 2.6.3.1):

"The Board concludes that the uncertainties in EA predictions for surface water quality effects require the development and implementation of a comprehensive AEMP for application through the Production Water Licence. The Board concludes that taking into consideration the mitigation measures proposed by De Beers and the uncertainties around EA predictions, there remains the potential of significant adverse impacts on surface water quality in Snap Lake. Implementation of an AEMP will allow for the identification of unforeseen adverse impacts over the life of the mine."

The ensuing recommendation (R4) is: "The Mackenzie Valley Land and Water Board shall develop the terms and conditions of an AEMP as part of the Production Water Licence for the SLDP. The AEMP shall include the commitments made by De Beers during the EA process and specific recommendations made in other parts of this report."

The implication of these conclusions and recommendations with respect to the review is that:

- the AEMP must address uncertainties in the EA prediction; and,
- allow identification of unforeseen adverse impacts.

1.2.2.2 Phosphorus and Dissolved Oxygen

From MVEIRB, (2003, 2.6.3.2):

“De Beers predicted that Snap Lake will become enriched with phosphorus as a result of water discharge from the mine. Average total phosphorus concentrations in Snap Lake will increase from 4-12 ug/L to 13-23 ug/L, with corresponding increases in algal growth.”

“Dissolved oxygen levels were predicted to decrease by 1-2.2 mg/L in deep portions of Snap Lake. These changes would reduce optimal lake trout habitat volume from 95% of the lake to 92% of the lake, and deep water benthic habitat from 98% to 96% of the lake volume.”

Following consideration of comments received MVEIRB concluded that “no significant adverse impacts to water quality or aquatic life are likely to be associated with the nutrient enrichment of Snap Lake.” This led to the following suggestion (S9):

“The AEMP which was recommended as a component of the Production Water Licence (Section 2.6.4.1) should include the requirement to verify EA predictions of changes in trophic and dissolved oxygen status of Snap Lake by the monitoring of:

- Dissolved oxygen concentrations in profiles at deep portions of Snap Lake with monitoring occurring under the ice in winter conditions and in late summer;
- Concentrations of total phosphorus, orthophosphate and organic phosphorus in connate groundwater and mine effluent on a regular basis and in Snap Lake under the ice in March and in early summer;
- Concentrations of chlorophyll *a* in Snap Lake in early summer after the loss of ice cover and in midsummer; and,
- Algal biomass and species community composition for phytoplankton and periphyton in Snap Lake in mid-summer. The monitoring should include measures of cyanobacteria biomass and species composition and cyanotoxins in the event that algal community composition shifts to favour cyanobacteria.

This review will assess the ability of the AEMP to assess changes in species community composition for phytoplankton in Snap Lake in mid-summer.

1.2.2.3 Effects of TDS in Snap Lake - Accuracy of Predictions and Effects on Aquatic Life

The primary conclusion from MVEIRB, (2003, 2.6.3.3) regarding TDS is that there is a potential for adverse effects if DeBeers predicted concentrations are exceeded and that DeBeers loadings predictions are driven by “the concentrations of TDS in connate groundwater and the proportion of connate groundwater to Snap Lake inflow from the mine water discharge.” The purpose of the AEMP (with respect to TDS and for the purposes of this review) is monitor TDS to ensure that: whole lake average TDS concentration in Snap Lake not exceed 350 mg/L at any point in the mine life”. Note that assessing the precision of the TDS loadings to Snap Lake requires reviewing the SNP effluent sampling program and not the AEMP.

Significant effects on aquatic life are not expected at the predicted concentrations MVEIRB, (2003, 2.7.3.3) however there is uncertainty regarding the predicted TDS concentrations. The uncertainties are addressed in MVEIRB recommendations R10 through R12.

In recommendation 10, a limit is placed on annual TDS loadings to Snap Lake and in recommendation 12, routine toxicity testing of effluent is suggested. Recommendation 11 states: "The AEMP shall be comprehensive enough to verify De Beers' EA predictions that changes to the aquatic community of Snap Lake will be of low magnitude. The monitoring program should incorporate the results of a statistical power analysis to guide sampling and interpretation of changes." The implementation of this latter recommendation within the AEMP is addressed in this review.

1.2.2.4 Certainty of Mixing of TDS in Snap Lake

The MVEIRB (2003, section 2.6.3.4) concluded that: "De Beers has not provided sufficient evidence to demonstrate that significant adverse impacts on the mixing regime of Snap Lake and subsequent effects to aquatic life are not likely to occur as a result of the SLDP. Without additional analysis or evidence, the Board concludes that precautionary measures must be implemented to prevent significant adverse impacts from occurring." The following recommendation (R8) pertaining to the AEMP is:

"De Beers develop a monitoring and Adaptive Management Plan to address uncertainty in lake mixing and density stratification. This shall include a monitoring program sufficient to provide early warning of persistent density stratification and a description of thresholds and mitigation measures such as tempering of the effluent stream or mechanical mixing. This recommendation could be implemented though the Production Water Licence AEMP recommended earlier in this report."

The implication with respect to this review is assessment of the monitoring program's ability to provide early warning of persistent density stratification.

1.2.2.5 Site Specific Water Quality Benchmarks

The MVEIRB (2003, section 2.6.3.5) concluded that: "the Snap Lake Diamond Project is not likely to have a significant adverse impact on aquatic life as a result of the discharge of trace metals to the lake. However, the Board is of the opinion that additional measures should be considered to address monitoring of mine discharge water and water within the mixing zone of Snap Lake to assess EA predictions of metal concentrations and ensure that any effects can be identified in a timely fashion for effective management over the life of the mine." Aside from the general monitoring recommendation, MVEIRB specifically suggests (S11):

"The AEMP should include assessment of cadmium and chromium levels in fish in Snap Lake."

1.2.2.6 Phosphorus and Dissolved Oxygen

The MVEIRB (2003, section 2.7.3.2) concluded that:

- There is some uncertainty associated with predicted DO concentrations; however effects will be reversible, of short duration and possibly occurring at later stages of the mine life. Changes in DO may fall within the natural range of DO concentrations.

- Cyanobacteria will increase but “not to levels that will pose a threat to water use by humans or wildlife”.

- “No significant adverse impacts on aquatic life or human health are likely to result from the nutrient enrichment of Snap Lake to the levels predicted by De Beers”.

These conclusions led to suggestion (S15) that total phosphorus, orthophosphate, organic phosphorus, nitrate, nitrite, ammonia, Kjeldahl nitrogen, chlorophyll *a*, algal biomass, phytoplankton community composition, monitoring of cyanotoxins if cyanobacterial species increase to 30% of the algal community biomass and DO profiles in deep areas be measured in Snap Lake to “ensure that any effects can be identified in a timely fashion and managed effectively in the future.”

2 Methods

2.1 Available Data

This section tabulates the data collected that are the focus of this review. Other data, even data of the same type may also have been collected.

Table 2: Summary of Available, Relevant Data

	Water Quality	Plankton – different in 2008
Mixing Zone	3	
NF	9	6[3]
MF	5	3[3]
FF	8	2[3]
Northwest Arm	5	4
wetlands (contingency receiving environment for treated sewage)	1	
outflow of Snap Lake, upstream of King Lake	1	
inflow to Snap Lake	1	
Northeast Lake	5	5

[3] Categorization into NF, MF and FF areas based on congruence of sample locations with water quality stations (Figure 2-1, DeBeers, 2008) categorized as such.

2.2 Selection of Reference Lake

A reference lake was selected following a two part process as described in Golder (2005 a, b). Northeast lake was selected as an "appropriate reference lake on the bases of bathymetry, water quality, sediment quality, and fish community composition". At this time, the rationalization for selecting Northeast Lake as a reference lake has not been reviewed.

The use of Northeast Lake as a reference lake began in 2006, although historical data are available from 2002, 2004, and 2005.

2.3 Utility of Baseline Data

Golder (2006) states that: "Sediment quality results from 2005 were not compared to baseline results from 1999 to 2001 because these samples were collected at different locations with substantial differences in particle size and total organic carbon content."

The baseline sediment quality data are of doubtful utility. The 1999 baseline data at least are not used for statistical analyses herein, because of limited sample size and sampling frequency.

3 Results

3.1 Ability of AEMP to Monitor Ecosystem Health

This section addresses the following question has posed by SLEMA:

"Taking into account the five years of AEMP reporting, please make comment based on your opinion on whether the current AEMP program remains sufficiently robust to meaningfully monitor aquatic community health?"

The ability of an AEMP to monitor ecosystem health is a function of many decisions including; how "health" is defined, the correct site conceptual model, the allowable uncertainty around making a decision error and a judicious choice of valued ecosystem component (VEC). Thus, a "passing grade" for an AEMP can mean many things and a (failing) grade may be meaningless.

For example chlorophyll *a* is a required measurement endpoint under the water license but DeBeers does not consider chlorophyll *a* to be a suitable surrogate measure of the phytoplankton community in Snap Lake and therefore did not statistically analyze or interpret these data within the 2007 AEMP. In the 2008 AEMP (Golder, 2009), DeBeers reiterates this statement. One might consider "failing" the AEMP due to this omission; however this decision was likely carefully considered by DeBeers. Note that the rationalization for this omission can only be inferred from the report and should be made more explicit.

In this assessment, a "passing grade" is defined as the ability to detect the change(s) of interest as stated in the EAR.

3.1.1 Benthic Macroinvertebrates

Benthic invertebrate samples collected in 2009 and earlier were collected during late winter reflecting an expected worst-case exposure due to highest effluent concentration and lowest dissolved oxygen. Sampling during worst-case exposure is consistent with guidance provided in INAC (2009), Environment Canada (2002, 2004).

In 2010, benthic macroinvertebrate samples will be collected in summer for logistical reasons. Thus 2010 will comprise a discontinuity in the temporal benthic community dataset. Since benthic populations are highly seasonally dependent it is unlikely that the pre and post 2010 benthic datasets can be defensibly reconciled.

The terms of the contract under which this review was conducted were drafted in December of 2009 prior the change in the benthic macroinvertebrate sampling program. The review of the benthic data began prior to the release of the 2009 AEMP report with the intention of adding the 2009 benthic macroinvertebrate data to the review. However due to the change in sampling season for 2010 (which will likely preclude defensible integration of pre-2010 benthic data), the 2009 benthic data were not included within this review as any comment in terms of improving AEMP design based upon statistical analyses of data collected during the winter is moot. The AEMP design-based discussion of benthic data collected in 2008 and earlier is now of limited utility and has therefore been removed to Appendix 1: Assessment of Early Benthic Macroinvertebrate Data.

3.1.2 Phytoplankton, Zooplankton and Supporting Variables

Golder (2009) states that "the objectives of the plankton component of the AEMP were to evaluate whether the Mine has affected plankton community composition and biomass in Snap Lake, and to determine whether the Mine discharge is causing nutrient enrichment in the lake."

Phytoplankton samples were collected[4] from the upper 6m at all locations (except Snap1 and Snap 31) and over all years (Golder, 2009). Equal volumes of water sampled from 0, 2, 4 and 6 m depths were composited. At Snap 1 water samples from 0, 2 and 4 m depths were composited whereas at Snap 31 water samples from 0 and 1.5 m depths were composited. The consistency in depth sampling over time is commendable. Water samples from the composite were submitted for analysis of chlorophyll *a*, microcystin-LR, total nitrogen, total phosphorous, phytoplankton, picoplankton and flagellate taxonomy. Two sub-samples were submitted for chlorophyll *a* analyses, however only one sample was analyzed; the other was held in reserve in case of loss or for checking anomalous results. The description of sample collection is clear and well presented.

[4] This review does not assess the protocols for sampling, preservation, identification and enumeration of plankton.

Zooplankton samples were collected[4] using vertical tows beginning at 1 m from the lake bottom. Two vertical tows were collected at each location. Golder (2009) incorrectly identifies these samples as replicates. Given the comparisons of interest (nearfield versus farfield, etc.) replication occurs at the scale of different locations within this area. Collecting many samples at one location measures micro-scale environmental variability. This degree of variability does not represent the variability among stations within a specific area and consequently underestimates the variance term used in hypothesis testing. The effect of this is a test with statistical power that is biased upwards.

The discussion regarding the utility of chlorophyll *a*, potential increases in cyanoabacteria and possible effects of changes in the environment and zooplankton community composition and attendant effects on phytoplankton in section 3.1.1 of Golder (2009) could be improved by linking the statements made with taxa important to the ecology of Snap Lake (ecological importance of calanoid copepods in similar lakes), demonstrable effects in similar lakes (cyanotoxin effects in oligotrophic lakes subject to eutrophication), how observed or anticipated changes in the Snap Lake planktonic community have been associated (or not) with similar changes in other comparable lakes, etc.

DeBeers (2002) classifies Snap Lake as oligo-mesotrophic versus ultraoligotrophic noting that this differs from most other lakes in the area. It is important that this classification be well established as it forms the classification against which changes are measured.

3.1.2.1 Monitoring Program

Table 3: Summary of Plankton Sampling Program

Year	Season	Frequency	Number of Locations	Bases for Comparison	Experimental Design
2004	open-water	every two weeks	12	among areas, within-lake reference area and temporal	BACI – within lake exposure and reference, temporal (1999 versus 2004)
2004-2006	open-water	every two weeks	12	within-lake reference area	
2007	open-water	every two weeks	15	within-lake reference area	
2008	open-water	three times (July 15th-16th, Aug. 12th-13th, Sept. 9th-10th)	9: 1 FF 1 MF 4 NF 3 NW Arm	appears to be only temporal	Unclear for plankton due to differences in trophic status between the Snap Lake and Northeast Lake (Golder, 2009, 3-6).
	open-water	twice (July 10th -12th and September 17th)	5	Northeast Lake	

3.1.2.2 Taxonomy and Enumeration

Golder (2009) provides details[5] of sorting, identification, biomass estimation, taxonomic keys used, and methods for estimating biomass for phytoplankton and zooplankton. Some or all of this important material was missing from DeBeers (2005b).

Table 4: Evaluation of Taxonomic Level of Effort for Plankton Enumeration

Year	Minimum Sub-Sample Size	Level of Identification	Sieve Size	Taxonomist	Source
2004	200	lowest practical level	12-inch, 153 µm	Bio-Limno Research Consulting, Inc.	DeBeers (2005a)
2008	not stated, cite Environment Canada (2002) and Gibbons et al (1993) as standard protocols	lowest practical level	not stated, cite Environment Canada (2002) and Gibbons et al (1993) as standard protocols	Bio-Limno Research Consulting, Inc.	DeBeers (2009)

3.1.2.3 Data Analyses

3.1.2.3.1 2004

DeBeers (2005a, pg. 99) states that "α and β will be set at 0.05 and power at 95% (1-β) for this (phytoplankton and zooplankton) study".

The 2004 data are classified in various ways. These include:

- spring, summer, and fall which are "determined by examining the mean whole lake phytoplankton biomass for each sampling period and grouping dates with similar mean total biomass. Zooplankton will be grouped the same way to maintain consistency." DeBeers (2005a);
- phytoplankton versus zooplankton;
- phytoplankton groups: (cyanobacteria, chlorophytes, chrysophytes, cryptophytes, dinoflagellates, and diatoms); and,
- zooplankton groups: cladocerans, calanoids, cyclopoids, and rotifers.

[5] Note that review of these details is limited to ensuring that such details are included. A review of the enumeration-related details is outside the scope of this review.

The log-transformed number of organisms within each grouping was tested for equality across open-water seasons using ANOVA.   Total biomass was also compared in a similar fashion.

Paired t-tests were used to test for changes in biomass between 1999 and 2004 for each of groups listed above and also for total biomass; likely using the groupings of spring, summer and fall.

The correlation (Pearson product moment) between plankton and water quality variables was estimated likely on a date-specific basis. Some reasonable restrictions based on habitat utilization were placed on the depth data used to estimate the mean water quality variables.

A regression of chlorophyll *a* concentrations on total phytoplankton biomass was conducted to determine the strength of the relationship.

3.1.2.3.2   2008

The 2008 chlorophyll *a*, total nitrogen, total phosphorous and microcystin-LR concentrations were not analyzed statistically. No reason is provided for this omission.

The 2008 data were divided into taxonomic groups (Phytoplanktonic groupings are: cyanoabacteria, chlorophytes, chrysophytes, cryptophytes, dinoflagellates and diatoms. Zooplanktonic groupings are: cladocerans, calanoid copepods, cyclopoid copepods and rotifers.) and the relative proportion based on biomass or abundance was estimated by station and sampling event for data collected between 2004 and 2008 inclusive.

Abundance and biomass for the taxonomic groupings listed above are compared between seasons. Note that although taxa are identified no summary metrics such as richness, diversity etc. are used to provide further insight into the data. Prior to 2008, phytoplanktonic data are subjectively grouped on the basis of one of these metrics and then compared using statistical tools. Zooplanktonic data follow the grouping established for phytoplankton. Problems associated with this approach are described in section 3.1.2.3.1. In 2008, data are grouped into one of the three open-water sampling events labelled as “spring”, “summer” and “fall”.

The means and 95th percentiles of the phytoplankton and zooplankton biomass estimates by taxonomic grouping were compared among seasons and years.

Nonmetric multidimensional scaling was used to investigate changes in biomass by season and year. Nonmetric multidimensional scaling is commonly used to investigate patterns in ecological data. Golder’s (2009) description includes both the subjective inputs and the criteria for assessing adequacy or implementing and interpreting the nonmetric multidimensional scaling results, respectively. However Golder (2009) does not specify how the number of dimensions was chosen; the description implies an automatic selection by the statistical software used. Bray-Curtis similarities were used as distances. Since both the number of dimensions and the choice of distance metric can affect the outcome, the implications of the decisions made by Golder should be discussed. Note that the results are also a function of the taxonomic groupings with the implication that changes and taxonomic groupings can produce different outcomes.

The null hypothesis that biomass does not change by year was tested using analysis of similarities. Although not stated, it is likely that the taxonomic groupings used in the nonmetric multidimensional and scaling comprise the response variables used in this procedure. The data used in the analysis is comprised of seasonal data with seasons nested within years. It is not clear from the description provided by Golder (2009) that this nesting structure was acknowledged. If the structure was unacknowledged the conclusions reached may be incorrect.

Given the concerns expressed by MVEIRB (2003) regarding eutrophication, it is not clear why the eutrophication-related variables total nitrogen, total phosphorus or microcystin-LR were not statistically analyzed.

3.1.2.4 Results and Interpretation

3.1.2.4.1 2004

DeBeers (2005a) concluded that there were significant changes in biomass during the open water season for some of the taxonomic groups examined. However the data analysis uses data grouped by dates where “seasonal groupings were determined by examining the mean whole lake phytoplankton biomass for each sampling period and the dates that had similar mean total biomass were grouped together.”

The ANOVA is an example of “data snooping”. In data snooping patterns are noticed in the data and then, a statistical analysis is conducted to test for the presence of the pattern. This type of data analysis produces levels of significance that are not usable.

The valid analysis of plankton data by season requires that data be grouped into seasons on the basis of another independent variable. This might be calendar day, degree day, start of ice free season, etc. Alternatively the effect of season can be assessed by using calendar date as a continuous variable rather than categorizing the continuous data.

The analysis used by DeBeers (2005a) also fails to use the temporal structure in the data. When mean changes in categorized continuous data are slight relative to variability in the dataset, true temporal differences can be obscured. This may be the reason that no differences among the data grouped as described were detected for some taxonomic groupings.

It is not clear how DeBeers can discuss trends in the data following ANOVA. ANOVA tests the null hypothesis: Ho: All group means are equal, against the alternative hypothesis: Ha: at least one mean differs from another. Unless contrasts or *a posteriori* comparisons are made (neither of which are mentioned) statements such as: “Chrysophyte biomass initially decreased in July, but showed an increasing trend through the remainder of the open water season” should not be included in a paragraph describing statistical analyses unless such an analysis was conducted. The implication of such statements by virtue of location within the document is that the data provided sufficient evidence to statistically make such statements. This is either incorrect to do or there has been an omission in the description of the statistical methods used.

Also, DeBeers (2005a, Table 2.3-1) incorrectly reports the number of chlorophyll *a* and zooplankton samples collected as 24. Samples collected at the same location and time (and using the same method) reflect only micro-scale variation. As interest centers upon comparing differences among locations the variability associated with the duplicate samples collected at a location is an inappropriate measure of among location variance. The duplicate samples are sub-samples of the location. The implications of this are that variances are underestimated and the amount of available information (degrees of freedom) over-represented. The net effect is that statistical hypothesis tests are artificially more sensitive than they should be declaring significant differences when in fact differences (at the specified significance level) do not exist. Also, the statistical power (if estimated) will be biased upwards.

Finally, although the requirement for normality is mentioned in the context of ANOVA and two-sample t-tests there is no evidence that this assumption was tested. DeBeers (2005a) log-transformed all data prior to analysis assuming that 1) this transformation was necessary; and 2) that the transformation was successful in achieving "the statistical requirements of normality and homogeneity of variance". Neither the requirement for the transformation or whether the expected benefit was achieved, was assessed. However, note that the assumption of homogeneity of variance was tested for the data analyses explicitly using t-tests.

3.1.2.4.2 2008

3.1.2.5 Additional Analyses

There is a lack of within-lake comparisons for phytoplankton biomass and an entire absence of phytoplankton abundance data for 2008. In keeping with the stated purpose of AEMP plankton monitoring (presented in introduction to this section) spatial changes in phytoplankton biomass and abundance are investigated for 2008, herein. This investigation is hindered by the limited sampling in the midfield and farfield (one site each).

Phytoplankton biomass and abundance data were extracted from the file "AEMP 2008 Part 2 Appendices_31Aug2009.pdf" obtained from the MVLWB registry. The raw data were modified due to the following discrepancy: *Monoraphidium irregulare* (G.M.Smith) Komarkova-Legenerova appears in two entries. This is undoubtedly due to a transcription error where only two non-zero entries are found in the first row representing this taxa (station Snap31 trip 3). Entries corresponding to this station in the second appearance of this taxon are zero. Therefore the two non-zero entries in the first occurrence of this taxa are transferred to the second instance and the first instance is deleted.

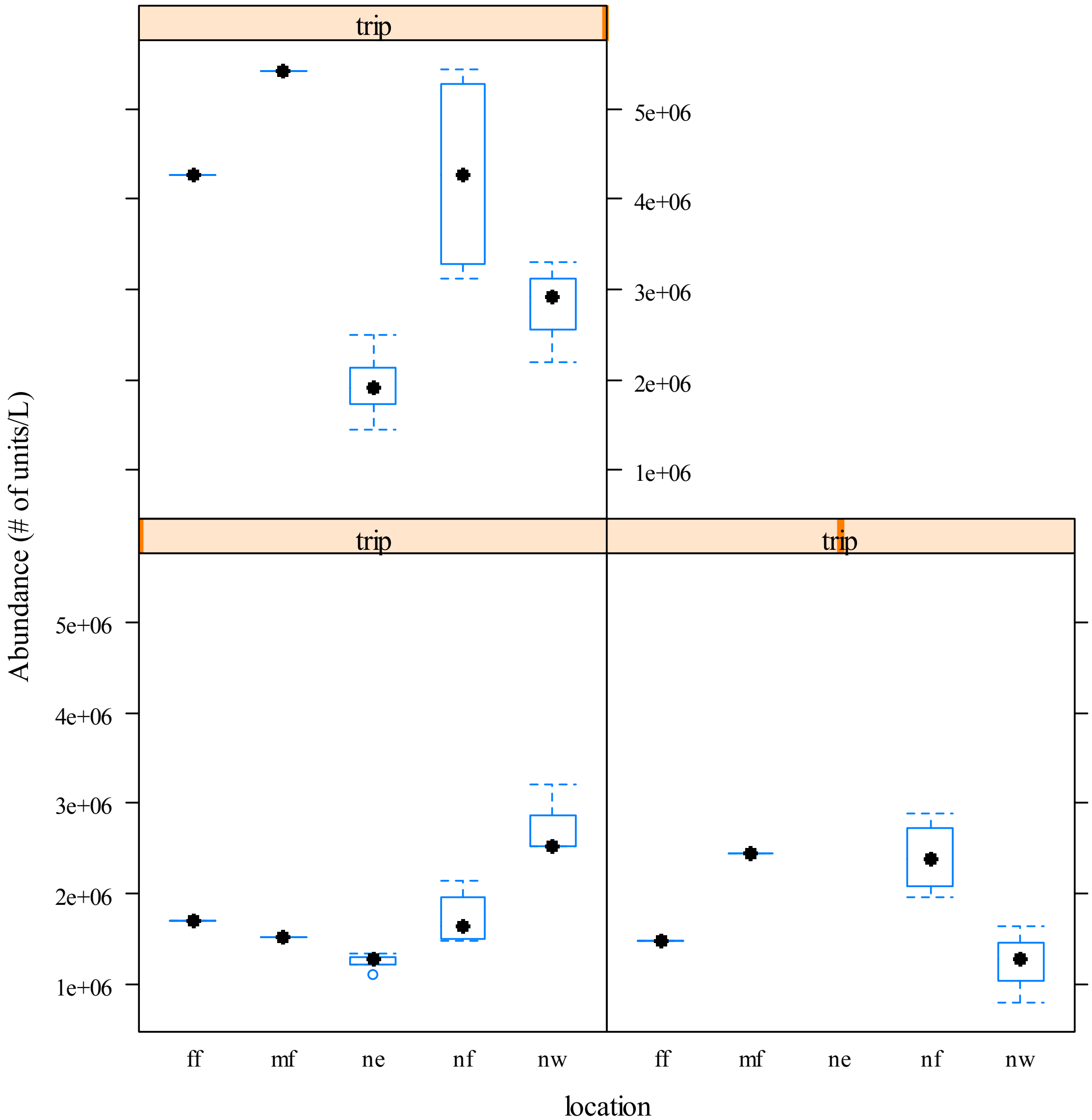

Figure 1: Phytoplankton Abundance, 2008

Figure 1 shows that phytoplankton abundance is lowest in Northeast Lake (ne) in both sampling trips. Phytoplankton abundance is highest at the nearfield stations (nf) in the latter two trips relative to the Northwest Arm (nw). In the first sampling trip phytoplankton abundance is lower than in the Northwest Arm.

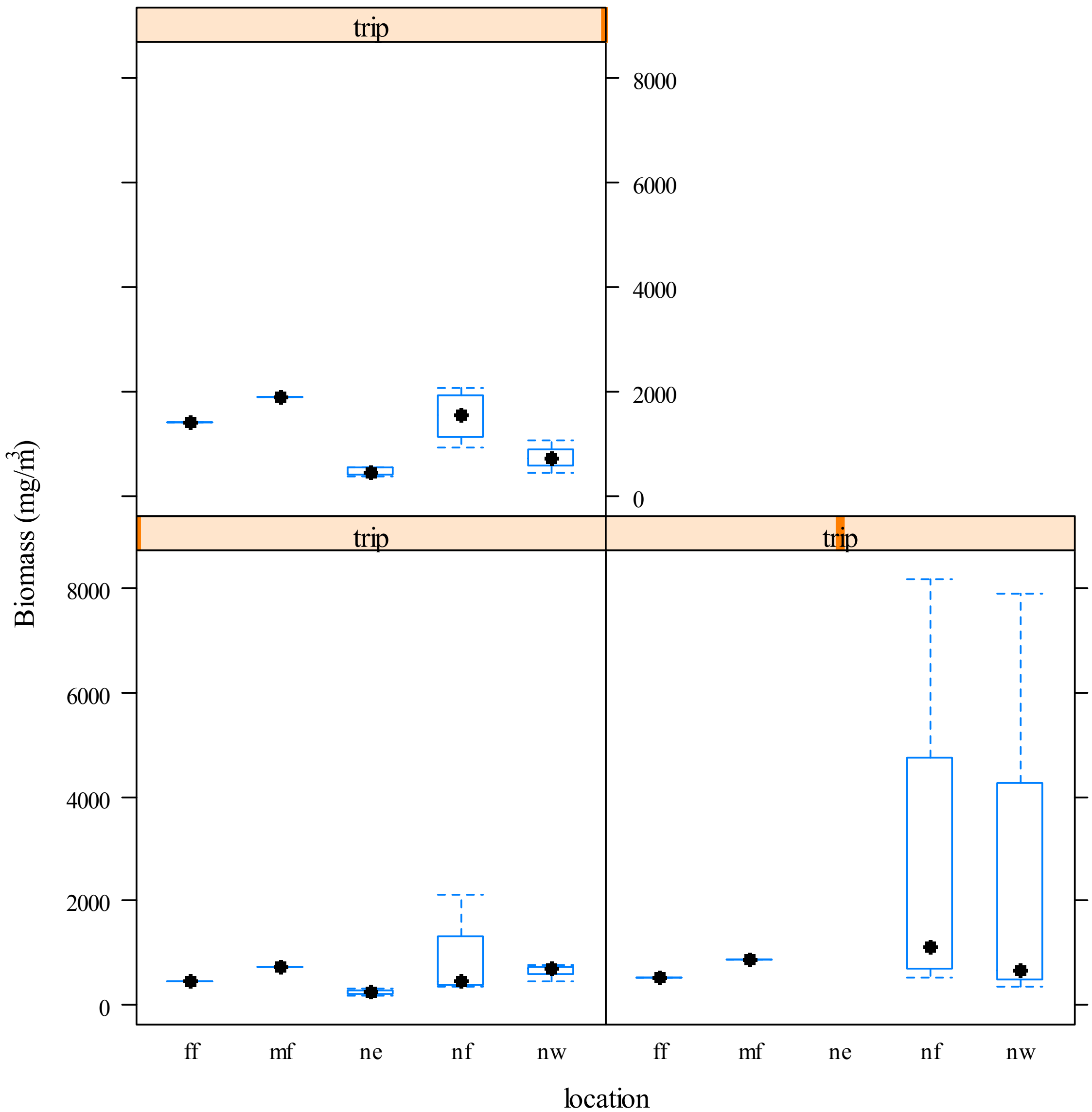

Figure 2: Phytoplankton Biomass, 2008

The results presented in Figure 2 lead to the same conclusions as those reached following interpretation of Figure 1 but with respect to biomass; and, with an increased in variability over sampling trips.

3.1.2.6 Recommendations

Recommendations are presented in bullet form below, separated by AEMP year.

3.1.2.6.1 2004

- Grouping data by similarity of biomass and then testing for differences using statistical tools invalidates the levels of significance. Data collected across seasons should be either; 1) grouped prior to data collection and then analyzed for significant differences across groups; or 2) be analyzed to include explicit model terms that incorporate the implicit ordering of data collected over time. This latter recommendation addresses not only data snooping but the comment regarding the absence of information to support changes over time.

- Using subsamples as replicates is incorrect. There should be a clear distinction between subsamples and replicates.

- The specific details regarding tests of assumptions for valid statistical hypothesis testing should be addressed.

- Section 2.3.4.3 of DeBeers (2005a) uses the phrase "mean annual biomass" but does not define how this value is estimated. This should be defined. Also; it is known that plankton biomass changes seasonally. Differential sampling across years without acknowledging seasonality will produce different mean estimates even when plankton biomass is not changing. If sampling dates and frequencies changed between 1999 and 2004 and the effect of differential sampling among seasons is not addressed comparisons of inter-annual means may not be valid. Possible effects of differential sampling among seasons should be addressed prior to comparing inter-annual means.

3.1.2.6.2 2008

- Since the number of dimensions and the choice of distance metric can affect the outcome of nonmetric multidimensional scaling, the implications of these decisions made by Golder should be discussed.

- It is not clear that the inherent structure of the data used within the ANOSIM analysis was acknowledged. This should be clarified.

- Reasons for not statistically analyzing the eutrophication-related variables total nitrogen, total phosphorus and microcystin-LR should be provided particularly given the concerns expressed by MVEIRB (2003).

3.1.2.7 Robustness of Plankton Monitoring

This section addresses the following question has posed by SLEMA in the context of plankton monitoring"

Taking into account the five years of AEMP reporting, please make comment based on your opinion on whether the current AEMP program remains sufficiently robust to meaningfully monitor aquatic community health?

Robustness is a vague term that can encompass a variety of criteria. Here robustness is used in the context of how the available information was used and whether the information is sufficient to make conclusions regarding the general question "Is the mine causing[6] changes in the planktonic community?" The sufficiency of information is in part answered through the ability of the AEMP to statistically detect changes of interest.

Golder (2009, pg. 3-27) states: "There is limited information on background year-to-year variation in phytoplankton community structure in Snap Lake. Therefore, at this time it is not possible to determine the relative contributions of natural variation and the discharge of treated effluent from the diffuser."

While the statement is correct with respect to the limited amount of background information it is possible to assess spatial effects within the lake. Gradients in planktonic metrics (both phytoplankton and zooplankton) associated with demonstrable changes in water quality variables associated with the effluent and/or distance from the effluent may be used to assess the effect of the effluent. Environment Canada (2002, 2004) and INAC (2009) advocate the use of gradient analyses for this purpose.

Such analyses are also indicated within the EAR (DeBeers, 2002 section 9.5.2.2.1) which discusses linkages between possible effects on the non-fish aquatic community and the mine. These links are through exposure via analytes released by DeBeers to the water column or following deposition, in the sediment (for benthic macroinvertebrates). As the primary release of analytes from the DeBeers facility to the receiving environment is through the effluent diffuser, an exposure gradient in analytes discharged from the mine is expected and indeed observed. An assessment of the linkage between the mine and the non-fish aquatic community should be conducted by 1) examining spatial trends in planktonic measurement endpoints; and 2) examining correlation between planktonic measurement endpoints and measures of exposure. Note that section 3.4.4.2 of Golder (2009) contains some mention of spatial differences in taxonomic composition within Snap Lake but a comprehensive analysis of planktonic spatial patterns is not present. The two types of analyses described above should be carried out by DeBeers.

DeBeers uses summary metrics such as richness, diversity and evenness to interpret benthic macroinvertebrate community data. These metrics as a group are quite useful in identifying changes in the benthic community that might be obscured by consideration of individual taxa.

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[6] Note that strictly speaking, causality cannot be inferred from an observational study.

The metrics are so useful that they are a routinely estimated and used when evaluating benthic macroinvertebrate data in programs such as the national Environmental Effects Monitoring Programs. The identification and enumeration of benthic taxa is a costly exercise, particularly relative to data analysis. Given that the phytoplankton and zooplankton collected by DeBeers are already identified, often to species and also enumerated, the additional cost to estimate and interpret summary metrics is minimal. The value of doing so is that: 1) further insights into potential effects or lack thereof are possible; 2) that concerns regarding conversion of abundance to biomass become moot; and, 3) that facultative changes in community composition masked by examining only biomass may be detected; and 4) that the data analysis results in only a small incremental cost.

DeBeers should estimate and interpret summary metrics such as taxonomic richness, diversity and evenness in addition to abundance and biomass for both phytoplankton and zooplankton. This recommendation is in keeping with the assessment of "quality" (as opposed to "quantity") of the non-fish aquatic community as discussed in section 9.5.2.1.1 (DeBeers, 2002). Note that DeBeers (2009) does investigate the high level taxonomic composition of the planktonic communities[7] which to some extent addresses the same issues that investigation of diversity, richness and evenness do.

The paragraphs above discussed how the information collected was used in order to comment on "robustness". Another element of "robustness" is the ability of the program to detect change. The ability to detect change is a function of the magnitude of the change of interest and the amount of natural variability. One way to determine the power of the AEMP is to conduct *a posteriori* power analyses to determine what power the AEMP did achieve. This is discussed in recommendation 11 of the EAR which states: "The AEMP shall be comprehensive enough to verify De Beers' EA predictions that changes to the aquatic community of Snap Lake will be of low magnitude. The monitoring program should incorporate the results of a statistical power analysis to guide sampling and interpretation of changes."

DeBeers does use the results of generic power analyses conducted by Environment Canada for designing the benthic macroinvertebrate survey. These recommendations were (commendably) confirmed for use in Snap Lake using the 2008 data. However it does not appear that power analyses were conducted for 1) water[8] or sediment quality analyte comparisons; or, 2) comparison of plankton metrics (abundance and biomass). DeBeers should conduct these calculations.

At this point in time given the absence of summary metrics for phytoplankton and zooplankton no statistical analyses (particularly spatial trends or correlation with analytes) were conduced. It follows that *a posteriori* power analyses were also not conducted[9].

[7] Golder (2009, Appendix B) presents proportional taxonomic composition abundances (Figure B-1) and biomass (Figure B-2) by location in Snap Lake to assess temporal differences within a location. Similar graphics are presented for zooplankton (Figure B-5 and B-6).

[8] Power calculations conducted herein for spatial comparisons of TDS within Snap Lake (presented Section 3.2.6) using the 2009 data, show that the AEMP is powerful enough to detect changes in 2009.

[9] Please see section 3.2 for a recommendation on this topic.

Overall it is not clear whether the plankton monitoring program is "robust" or not. Some recommendations are provided to use the information collected to better assess the linkages between mine and planktonic community discussed in the EAR. Another recommendation regarding *a posteriori* power analyses is made. If these recommendations are adopted it will be easier to determine whether the plankton monitoring program is "robust" or not.

3.1.3 Overall Recommendations – Ability to Monitor Ecosystem Health

The following general recommendations extracted from section 3.1 are provided below in no particular order. Some very specific recommendations are not restated in this summary.

- Summary metrics such as those used to interpret benthic macroinvertebrate community data should also be used to interpret the planktonic data.

- The utility of chlorophyll *a* measurements is questioned by DeBeers. The following authorities also suggest the limited utility of chlorophyll *a* as a surrogate measure of the phytoplankton community in Snap Lake.

    - Chlorophyll *a* may not be the dominant photopigment (Bowman, 2005).

    - Dolan et al (1978) found that "chlorophyll *a* concentrations were inconsistent with phytoplankton cell volume concentrations" in Saginaw Bay of Lake Michigan.

    - El-Shaarawi and Munawar (1978) found a significant but seasonally varying relationship between phytoplankton biomass and chlorophyll *a*. They found that variability in chlorophyll *a* concentration varied with taxonomic composition.

    - Jónasson et al (1992) state that 5 to 10 fold differences in the chlorophyll *a*/ biomass ratio are not uncommon.

  DeBeers should present a case for omitting chlorophyll *a* as a surrogate for the phytoplankton community in Snap Lake and instead focus on the analysis of the enumerated taxa. If chlorophyll *a* were omitted, and phytoplanktonic and zooplanktonic taxa were investigated in more detail, the costs of the current AEMP would be reduced and the information content would likely increase.

    - DeBeers should conduct power analyses for 1) water and sediment quality analyte comparisons; and, 2) comparisons of plankton metrics (abundance and biomass and possibly others) over space and with exposure measurements.

3.2 Effects of TDS in Snap Lake

This section addresses the following question has posed by SLEMA:

"TDS level increases were predicted and have been realized in Snap Lake. Please make comment based on your opinion if these increases pose any threat to the relatively small water system such as Snap Lake."

This question is addressed through a brief review of the scientific literature and examination of the TDS data provided for Snap Lake as limited by the constraints of the budget allocated for this task.

3.2.1 TDS Data Quality

Samples were submitted for TDS measurements to two laboratories. Eleven and 3 % of the TDS data were invalidated due to excessive holding times (Golder, 2009 Table A2-5) for samples submitted to ALS and Maxxam respectively. The overall percentage of invalidated samples is 3.3% ((9+3)/(340+27)). However 15.3% of TDS samples submitted ((43 + 13) /(340 + 27)) exceeded hold times and were subject to warnings. The relative percent difference in total dissolved solid measurements was ≤ 14% with most differences < 5% (Golder, 2009 Table A2-7). Two of 9 split samples (22.2%) exceeded a relative percent difference of 20%. The maximum relative percent difference was 32%; this coupled with frequency of observations with relative percent differences greater than 20% suggests a minor degree of concern for the data quality. Note that results for some analytes are quite poor Golder, (2009, Table A2-11). Golder has creditably investigated these results and identified reasons for the discrepancies. These reasons are not attributable to Golder.

Overall, the investigation of TDS quality assurance data indicates minor concerns with TDS data quality with the exception of holding times. Strong follow-up actions were taken by Golder when investigating quality assurance issues for other analytes.

3.2.2 TDS Data Quantity

Under the AEMP, TDS data are collected three times during the open-water season at 9 locations within Snap Lake and twice during the open-water season at 5 locations within Northeast Lake.

This is inconsistent with the main AEMP document (Golder, 2009) which states:

- AEMP water monitoring
  - monthly, 3 depths
    - 3 diffuser
  - quarterly, at least mid-depth
    - 7 nearfield

- 2 midfield
- 4 far field
- 4 northwest arm

Three water samples are collected if (Golder, 2009):

- Conductivity measurements throughout the water column at the site were between 0 and 60 microSiemens per centimetre (µS/cm) and the conductivity range in the water column was greater than 15 µS/cm; or

- maximum conductivity was >60 µS/cm and the range was greater than 25%.

The DFO Fisheries Authorization SC-00-196[10] states that TDS is to be monitored "in close proximity to the freshwater intake, treated minewater outlet and the artificial reef structures within Snap Lake" in 1 m vertical increments. Temporally, TDS monitoring must occur "once per month under ice and at least two times per year during open water conditions".

3.2.3 TDS Requirements and Expectations

The DFO Fisheries Authorization SC-00-196[11] states that action is required "if the monitoring results indicate that discrete depth concentrations of TDS exceed 350 mg/L at any depth in 3 consecutive sampling events at any of the sampling locations".

The purpose of the AEMP (with respect to TDS and for the purposes of this review) is to monitor TDS to ensure that: whole lake average TDS concentration in Snap Lake not exceed 350 mg/L at any point in the mine life".

3.2.4 Effects of TDS

The effects of TDS on aquatic organisms are due to the concentrations of the ions contributing to the TDS measurement (Mount et al., 1997; Weber- Scannell and Jacobs, 2001; and Weber-Scannell and Duffy, 2007). Weber-Scannell and Duffy, (2007) suggest that for the protection of salmonid species guidelines for individual ions should be used rather than for TDS. In Alaska the effects of ions contributing to TDS measurements are addressed through explicit regulation (Alaska DEC, 2009) on site-specific bases. Site-specific TDS regulations which acknowledge contributory ions include variation in TDS concentrations with Ca concentration and site–specific limits for ions contributing to TDS concentrations such as chloride, Na and sulphates associated with magnesium.

[10] Fisheries and Oceans Canada. 2004. Authorization for Works or Undertakings Affecting Fish Habitat (SC-00-196)

[11] Fisheries and Oceans Canada. 2004. Authorization for Works or Undertakings Affecting Fish Habitat (SC-00-196)

Mount et al, (1997) examined the relative toxicities of ions that commonly contribute to TDS concentrations to *Ceriodaphnia dubia*, *Daphnia magna* and *Pimephales promelas*. They found the following relative toxicities for the ions (HCO_3^-, Ca^{2+}, Cl^-, K^+, Mg^{2+}, Na^+ and SO_4^{2-}) tested: $K^+ > HCO_3^- = Mg^{2+} > Cl^- > SO_4^2$. Calcium and Na ions were not toxic at the concentrations tested.

Weber-Scannell and Duffy (2007) review the effects of TDS on aquatic organisms with emphasis on results relevant to Alaskan waters. They conclude that due to the differential effects of TDS composition and life stages, that limits vary temporally and compositionally (i.e. some analytes contributing to TDS are more toxic than others) to protect sensitive life stages. This approach affords the least number of restrictions to proponents and protects sensitive organisms / life stages. In at least one waterbody receiving mine effluents TDS limits do vary seasonally to coincide with increased sensitivity of salmonids to TDS during spawning (Scannel, 2003).

An unequivocal statement regarding whether increases in TDS poses a concern to a small lake such as Snap Lake cannot reasonably be made, given the amount of time allocated for this task. What can be said is that:

Recent literature reviews conducted for taxa quite similar to those of Snap Lake provide a range of TDS toxicity values, most of which are greater than the 350 mg/L currently set with the DFO Fisheries Authorization for Snap Lake.

3.2.5 TDS in Snap Lake

TDS is examined by year and by stations[21] (with at least four years data) in the following graphic. Colours are used to distinguish between the following areas: NF, MF, FF and NW-Arm and the lines are the result of nonparametric smoothing functions. Of interest is whether all areas and stations behave similarly over time.

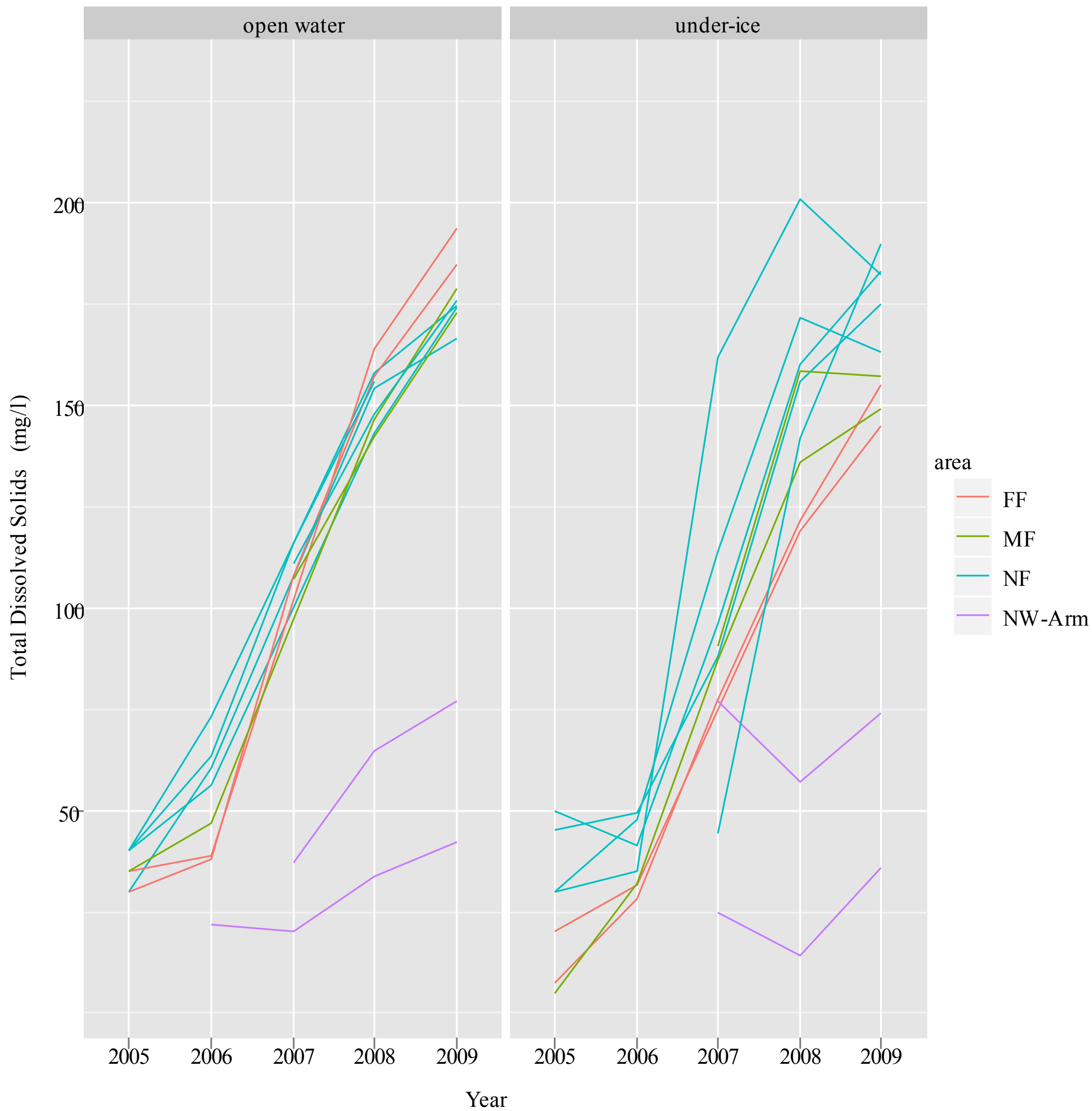

Figure 3: TDS in Snap Lake over Time and by Station

Figure 3 shows that there is an approximate four-fold increase in TDS over time at all stations in the NF, MF and FF areas. TDS concentrations are similar at all stations (except the NW-Arm) during the open water season. Under ice TDS concentrations are more variable with generally higher concentrations in the nearfield.

3.2.6 Power Calculations

The ability of the AEMP to detect changes in TDS across areas for the 2008 data was estimated using a two-way factorial[12] model with terms for "area" and "season". The following cell means estimated from the 2008 were used as inputs:

Table 5: 2009 TDS Cell Means

	FF	MF	NF	NW-Arm	Averages
open water	189.2	175.8	172.5	50.7	147.1
under-ice	150.0	153.0	180.5	47.0	132.6
Averages	169.6	164.4	176.5	48.8	

The model residual mean square error (403 units^2) was used to estimate the among-measurement standard deviation and the number of available observations for each area was estimated from the average over open water and under-ice to obtain the vector of sample sizes: {4, 4, 8, 6} for the FF, MF, NF and NW-Arm "areas"; respectively.

The statistical power of the design was estimated using Hintze (2008). Note the statistical power is approximate because the analysis assumes equal replication among the levels of at least one factor but replication was not equal between open-water and under-ice periods.

The ability to detect a change in the area means was very high (estimated as 1[13]) which is not surprising given the very low mean in the NW-Arm. If the NW-Arm data are omitted and the two-way factorial analysis re-run[14] the power of the among area comparisons drops slightly to 0.999 which is still very strong. The ability of the 2008 AEMP to detect changes in TDS among the three or four areas within Snap Lake is very high.

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[12] A "factorial" model describes the relationship between a response (TDS in this case) and two or more factors or categorical independent variables ("season" and "area" in this case) . A defining attributed of the factorial model is that each observation can be categorized as belonging to only one season and area. Once the model was fitted, the assumptions of normality and heterogeneity of variance of residuals were validated using Shapiro-Wilks and Levene's Tests respectively. The latter test was run collapsing area means over seasons given that; 1) this is the primary effect of interest; and, 2) that no seasonal effects were detected.

[13] Using a total sample size for the balanced analysis conducted in PASS 2008 similar to that actually available.

[14] Power analysis inputs were derived using the same methodology as described for the complete dataset.

3.3 Effects of Blasting in Snap Lake

This section addresses the following question has posed by SLEMA:

"The Traditional Knowledge panel has expressed a concern over the effects of blasting work under Snap Lake and the effects on fish. Please make comment based on your opinion how these effects may directly or indirectly impact the health of fish in Snap Lake."

3.3.1 Known Effects of Blasting

Changes in pressure levels due to use of explosives can cause tearing or rupture of swim bladders in fish (Wiley et al., 1981 and Govoni et al., 2003). Effects are likely greater in smaller fish (Munday et al., 1986; Young, 1991 and Keevin and Hempen, 1997). The effects of pressure waves due blasting has been assessed for some organisms without swim bladders. Keevin and Hempen (1997) following an extensive review, suggest that organisms without gas containing structures (such as most invertebrates) are likely insensitive to pressure changes associated with blasting. However they caution that: "invertebrate mortality studies have used inadequate sample sizes, lacked adequate controls, and failed to conduct pressure waveform analysis of the explosion".

Wright and Hopky (1998) quote Wright (1982) stating that "sublethal effects of explosives on shellfish and crustaceans including behavioural modifications are little known or understood". On the basis of the limited review conducted herein it does not appear that chronic effects of blasting have been well studied in freshwater aquatic species in ensuing years; hence the conclusion by Wright and Hopky (1998) likely still remains relevant.

3.3.2 Factors Affecting Blasting Effects

The effects of blasting on aquatic species are a function of the force[15] of the pressure wave. The force of a pressure wave decreases with distance from the shock source and thus affects the potential for, and severity of, effects. US Army Corps of Engineers, (2000, Figure 6) demonstrates a very strong decrease in average peak water shock pressure with distance. Within approximately 50 metres, average peak water shock pressures were reduced by approximately 95%. Goertner, (1994) and Govoni et al., (2008) also show that effects on fish diminish rapidly with distance from the explosion.

The location of blasts affects potential blasting effects. For example, the aqueous effects of blasting on land may be greatly diminished relative to the same detonation in water. US Army

[15] The "force" of a pressure wave may be measured in a variety of ways. US Army Corps of Engineers (2000) describe average and peak water shock pressure (units = pressure/area) and energy flux density (units = pressure/area). Govoni (2003) found that the integral of the pressure waveform over time (units = pressure-time) correlated with effects on juvenile and larval fish. The relationships between various force measurements of pressure waves are described in many introductory physics textbooks.

Corps of Engineers, (2000) found that the effect of blasting in rock was only 0.014 of blast in water.

Finally, Wright (1982) states that the degree of damage is related to type of explosive, size and pattern of the charge(s), method of detonation, distance from the point of detonation, water depth, and species, size and life stage of fish.

3.3.3 Mitigating Blasting Effects

There is a large reduction in the pressure wave felt at a given site if there is a slight delay between blasts comprising a blast pattern. This may be due to formation of gasses by the first blast which are compressible (whereas water is not). Thus the pressure wave generated by a simultaneous blast pattern is smaller in absolute value because pressures waves are not added but are sequential and the pressure waves generated by successive blasts are mitigated by the gasses created by previous blasts. This mitigative effect may have the most effect when blasting occurs in water.

3.3.4 Guidelines

Wright and Hopky (1998) provide guidelines for the use of explosives in or near Canadian fisheries waters. The numeric criteria presented within the guidelines are restated below:

- "No explosive is to be detonated in or near fish habitat that produces, or is likely to produce, an instantaneous pressure change (i.e., overpressure) greater than 100 kPa (14.5 psi) in the swimbladder of a fish"

- "No explosive is to be detonated that produces, or is likely to produce, a peak particle velocity greater than 13 mm•s-1 in a spawning bed during the period of egg incubation[16]."

Wright and Hopky (1998) also provide setback distances for a variety of substrates suitable for achieving the numeric criteria above.

[16] Faulkner et al. (2008) confirmed the protectiveness of this value for rainbow trout eggs.

3.3.5 Preliminary Assessment of Blasting Effects at Snap Lake

Golder VME (2001) conducted an impact assessment of "existing and proposed future blasting operations at the Snap Lake Diamond Project". The recommendations following the assessment of pre-operations data based on DFO blasting guideline limits (Wright and Hopky[17], 1998) are:

- "The maximum explosive loads for limiting peak ground vibration and underwater overpressure levels to 13 mm/s and 100 kPa, would be 111 and 1,937 kg respectively, based on a minimum distance of 115 m."

- "Based on initial production round estimates of about 230 kg per delay as provided by the Snap Lake mine planners, the set-back distances calculated for maintaining the limiting peak ground vibration and underwater overpressure levels of 13 mm/s and 100 kPa, would be approximately 150 and 60 m respectively".

The conclusion reached by Golder (VME) based upon the recommendations above is that DeBeers could likely meet DFO blasting requirements but that "monitoring of underwater overpressure and ground vibration effects during the initial stages of regular production blasting should however, be carried out to better define the attenuation characteristics developed for this site."

At this point in time it is not clear whether this monitoring was carried out. If not, it should be carried to verify the Golder predictions.

3.3.6 Conclusions

MVEIRB (2003, Appendix D. List of De Beers Commitments) states that: "A follow-up program to the July, 2001 blast monitoring program will be undertaken to allow for refinement of the equations used to calculate peak particle velocity and overpressure once mine production begins to ensure that predicted blast overpressure and ground vibrations estimates are correct (IR 3.9.12b)". Currently it is not clear that study was conducted. If the study has not been conducted the study might be conducted or more practically, peak particle velocity and overpressure might be measured under worst-case scenarios to ensure that Canadian guidelines are not exceeded. In addition the mortality of potentially affected eggs *in situ* might be assessed relative to a control location in a manner similar to that described in Faulkner et al. (2005).

This recommendation is not intended to supersede that contained in MVEIRB (2003) but only to provide a practical means to address potentially unresolved issues.

[17] Although not cited explicitly, this is DFO regulation extant at the time of reporting.

3.4 Substances not Listed in the EAR

This section addresses the following question has posed by SLEMA:

"DeBeers data shows that there are elevated levels of Mercury, Uranium, Rubidium, Thallium, Barium, Vanadium, Strontium, Molybdenum, Lithium, Arsenic, and Boron at the diffuser and the NF area. There are no predicted increases of any of these elements and actually many are not even listed in the list of environmental assessment review (EAR) elements. Please make comment based on your opinion of whether the continual discharge of these elements creates a "hotspot" around the diffuser and what effects can be expected in Snap Lake."

3.4.1 Toxicity Testing in Snap Lake

This sub-section briefly examines toxicity test results in order to provide supporting information regarding effects of the analytes flagged by SLEMA. It is important to note that effluent toxicity test results almost certainly represent a worst-case acute exposure scenario with respect to organisms in Snap Lake with respect to the effluent collected at that point in time[18]. Also any significant toxicity is attributable to the suite of analytes and any synergistic or antagonistic effects among analytes as mediated by toxicity modifying factors.

Table 6: Summary of Aquatic Toxicity Tests, Snap Lake AEMP, 2008

	Time	Organism	Response	Result
Lake water near diffuser (SNP 02-20 d, e and f)	July, September 2008	*Ceriodaphnia dubia* (water flea)	mortality, fecundity	no effect
		Pseudokirchneriella subcapitata (green algae	growth (IC25, IC50)	no effect
Water Treatment Plant (Site 02-17 / 02-17B)	April, March, July, October 2008	*Oncorhynchus mykiss* (rainbow trout)	mortality	no effect
		Daphnia magna (water flea)	mortality, fecundity	two of five tests[19] showed significant reproductive effects
		Ceriodaphnia dubia (water flea)	mortality, fecundity	no effect
		Pseudokirchneriella subcapitata (green algae	growth (IC25, IC50)	no effect

[18] In 2008, toxicity samples are collected twice at diffuser stations () in July and September. Samples are collected at the depth of maximum conductivity or mid-depth in the absence of a conductivity gradient.

[19] This section only reviews toxicity test results as supporting information. Therefore details are not provided but may be found in section A5 (Golder, 2009).

Table 6 shows that for the species used and dates tested, no mortality or reproductive effects were observed in Snap Lake in 2008. Reproductive effects were noted for two of five tests with water fleas but no effects were noted for rainbow trout or the green alga tested in water from the water treatment plant. In 2007 and 2009, reproductive effects were noted in lake water.

3.4.2 Data Modifications

The following changes were made to the datasets provided by DeBeers prior to analysis / investigation:

- Variable names corresponding to the 2008 data were (very slightly) modified to be consistent with those used in other years.

3.4.2.1 Water Quality

3.4.2.1.1 Station Selection

In order to assess the question posed by SLEMA it was originally intended to assess temporal trends in water quality at stations proximal to the diffuser; i.e. the SNP 02 20 series of stations. However, water samples were collected at the SNP 02 20 a, b, c series from 2004 to 2006 but dropped from 2007 to 2009. In 2006 monitoring began at SNP 02 20 e and f. Thus there are no SNP 02 20 data series spanning the time frame from 2004 to 2009.

The temporal availability over time for another station proximal to the diffuser, SNAP 14 is examined below.

Table 7: Historic Sampling Effort at Snap 14, All Water Quality Analytes

Label	2004	2005	2006	2007	2008	2009
SNAP14	218	30	24	0	0	0
SNAP14-bottom	0	224	187	283	194	0
SNAP14-mid	0	321	373	377	393	105
SNAP14-surface	0	227	185	282	194	0

Table 7 shows that sampling is inconsistent with respect to depth of sampling at Snap 14 from 2005 onwards. Note that Golder (2009) does state that vertical conductivity profiles are used to determine if sampling by depth is required. The lack of any bottom or surface samples in 2009 implies a lack of conductivity gradient at all sampling locations.

Thus investigation of the water quality analytes flagged by SLEMA was hampered by inconsistent sampling. A pragmatic decision to use stations with 4 or more years of data when investigating temporal changes led to the analyses presented in the remainder of this section. This restriction on data used is explicitly stated whenever invoked.

3.4.2.1.2 Analytes not Investigated by Golder

The investigation of water quality variables in Golder (2009) is limited to those variables that exhibit a correlation[20] with conductivity implying (although possibly explicitly stated in the AEMP document) that an absence of correlation means that the analyte is not effluent related. Some of the variables omitted were those identified by SLEMA as comprising a potential hotspot. Therefore these variables, omitted by Golder are explicitly investigated here. Note that at this point in time the data are used "as presented" in the data files provided by DeBeers with no adjustment for detection limits.

[20] Correlation analyses were conducted using a Pearson product moment correlation using data collected from 2004 to 2009. Table A7-2, Golder (2009).

3.4.2.1.2.1 Arsenic

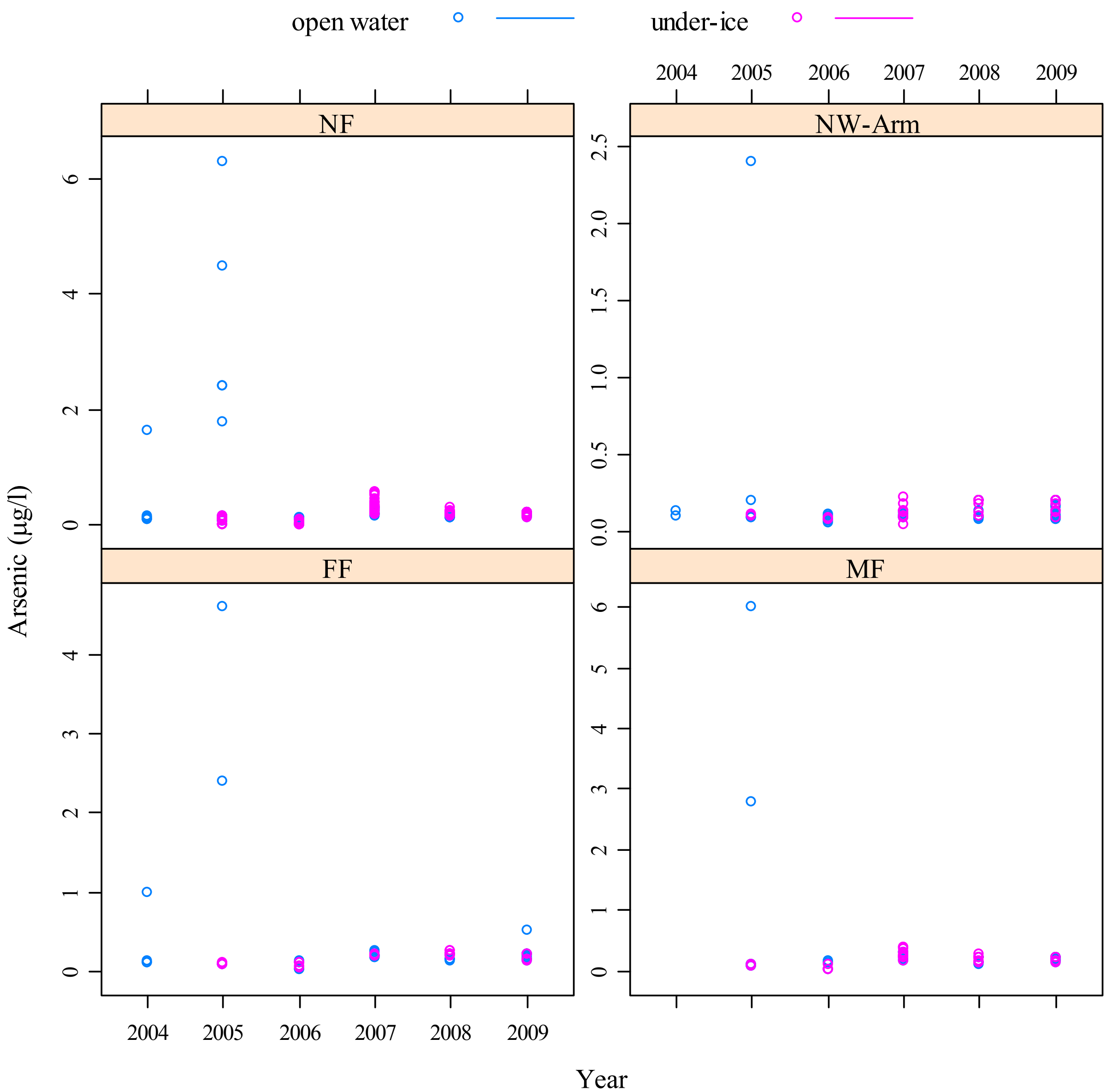

Figure 4: Total Aqueous As over Time and Space

Error! Reference source not found. shows that total As was higher in 2004 and 2005 than currently with a possible weak gradient from the mine site. Details subsequent to 2005 are obscured due to these higher values. The next graphic presents only the latter years.

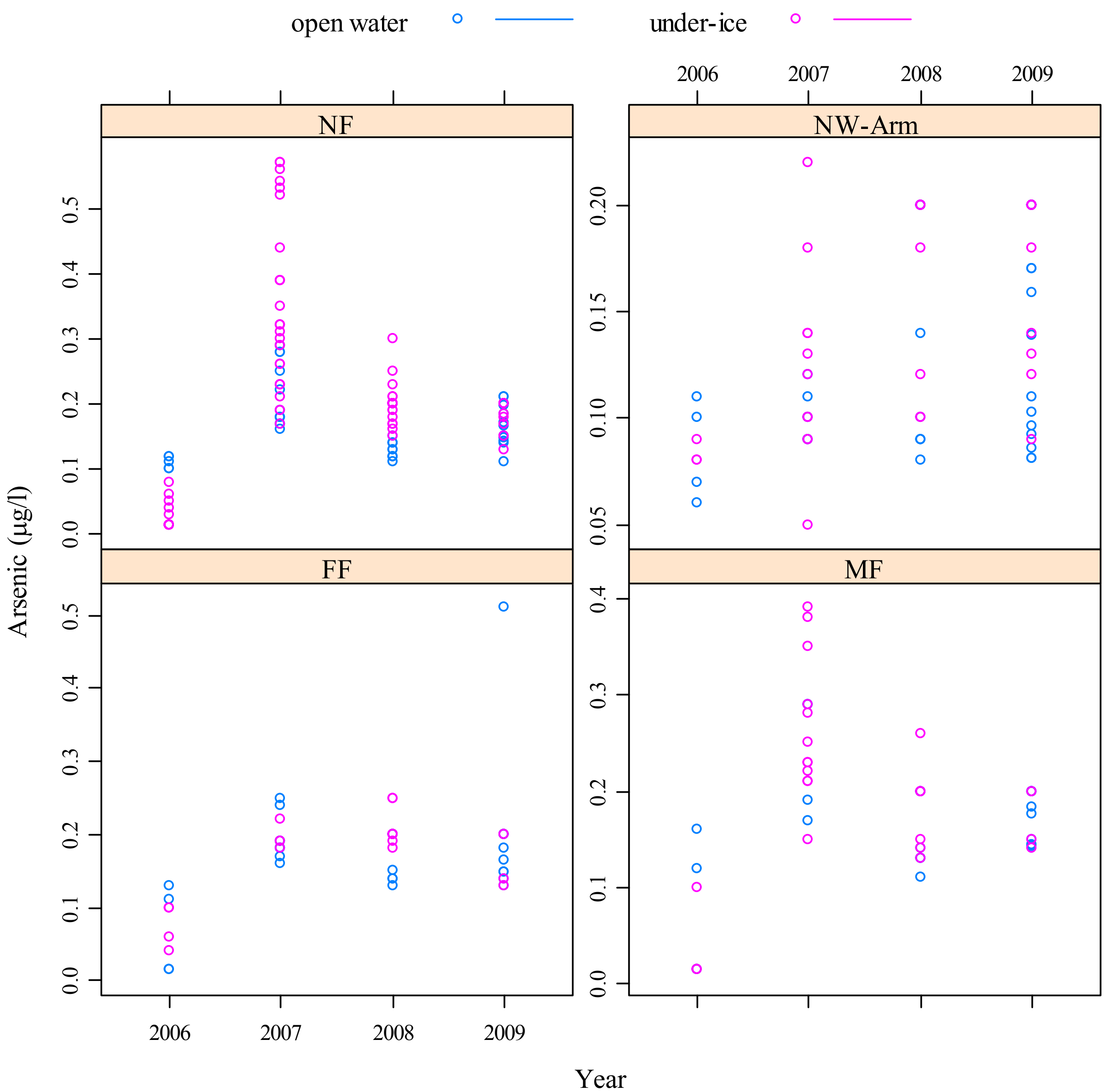

Figure 5: Total Aqueous As over Time and Space, 2006 through 2009 only

If specific stations are ignored, total As concentrations appear to be increasing over time although there is considerable variation. There may be a slight step change in concentration between NF, MF and FF stations versus the northwest arm in 2009. The following graphic presents the 2005 through 2009 data using smoothed lines for each station. The graphic differs slightly from that above because stations were selected with at least four years[21] data.

[21] These stations are: SNAP02A-mid, SNAP03-mid, SNAP05-mid, SNAP06-mid, SNAP07-mid, SNAP08-mid, SNAP09-mid, SNAP11A-mid" , SNAP14-mid, SNAP20B-mid, SNAP23-mid, and SNAP26-mid.

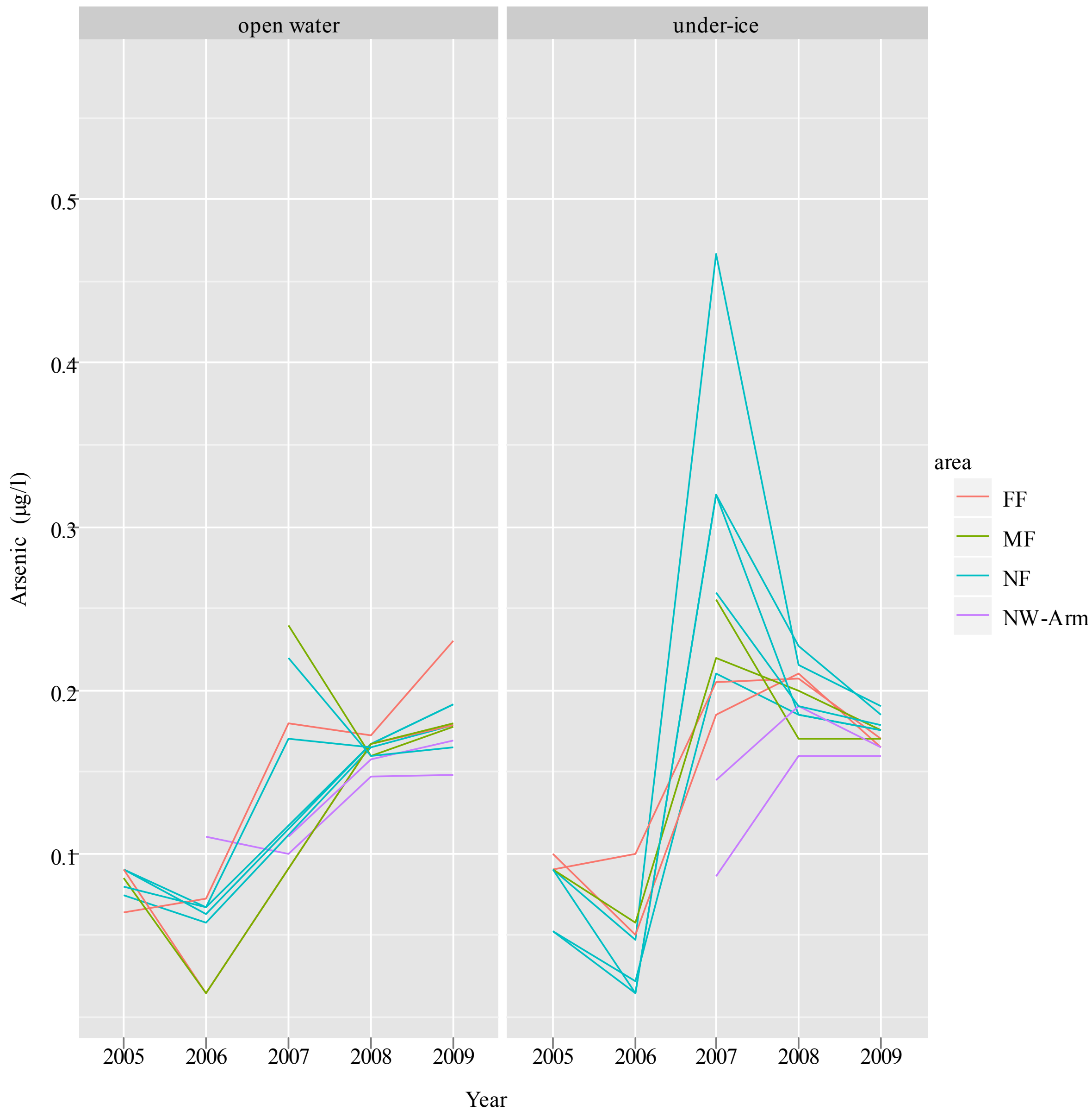

Figure 6: Total Aqueous As over Time and Space by Station

There is an increasing trend in As concentration in the open water season at most stations as shown in **Error! Reference source not found.**. Under the ice an initial increase to high values observed in the near-field at nearfield stations in 2007 are followed by a possible asymptote for all stations in subsequent years. This asymptote is approximately the same as the open water total As concentrations. These observations <u>may</u> indicate that total As concentrations are stabilizing at a concentration of approximately 0.18 µg/L throughout the lake over the year.

3.4.2.1.2.2 Mercury

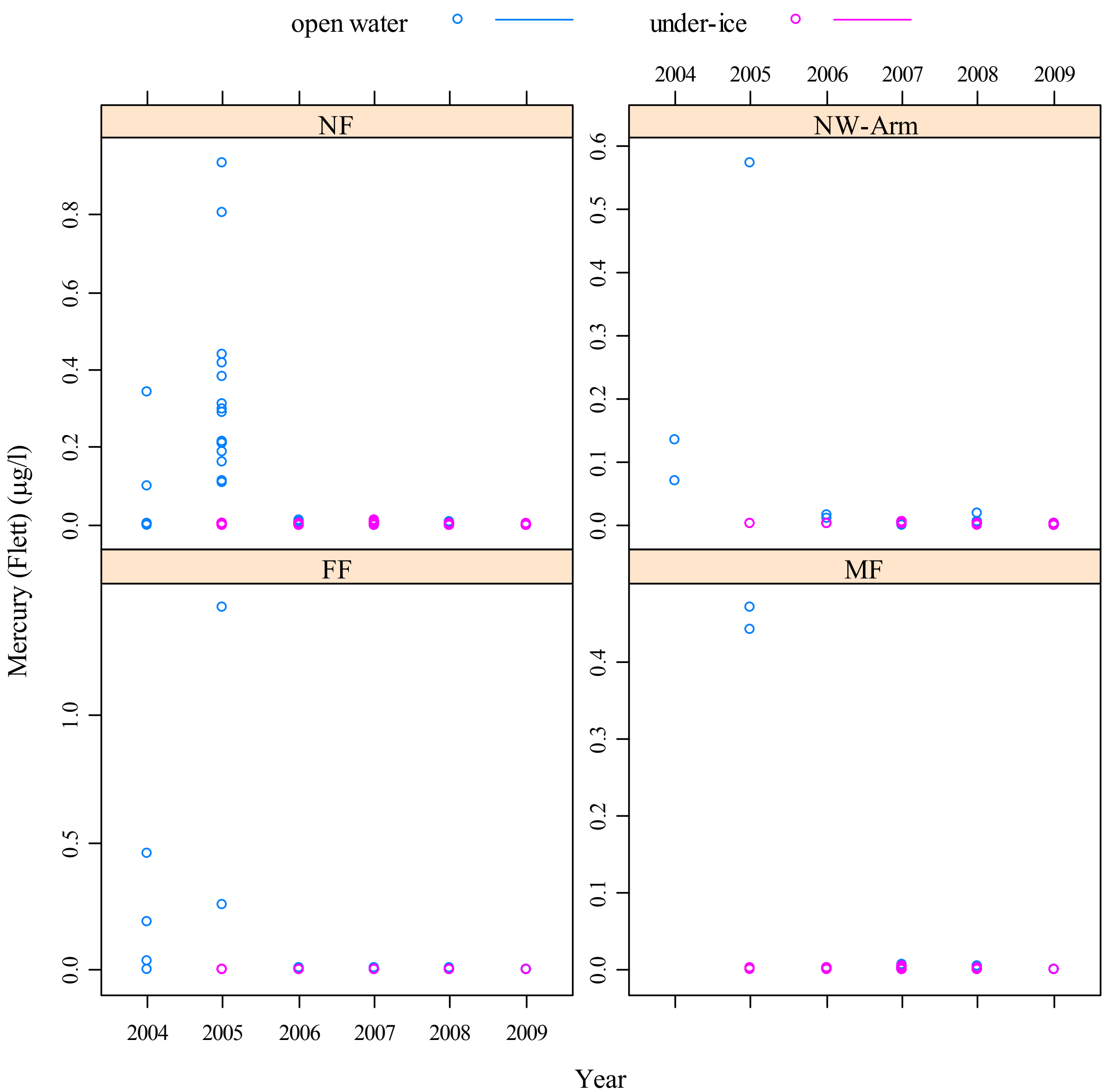

Figure 7: Total Aqueous Hg over Time and Space

Error! Reference source not found. shows that total Hg was higher in 2004 and 2005 than currently. Total Hg was highest at the FF station. These observations do not make sense if; 1) the data are valid; and, 2) effluent is the only potential Hg source because Hg is below the (low level) detection subsequent to 2004 and 2005. It is possible that the Hg detected in 2004 and 2005 represents uncontrolled run-off or sewage inputs[22].

[22] Neither of these two potential mechanisms have been investigated.

Details subsequent to 2005 are obscured due to these higher values. The next graphic presents only the latter years.

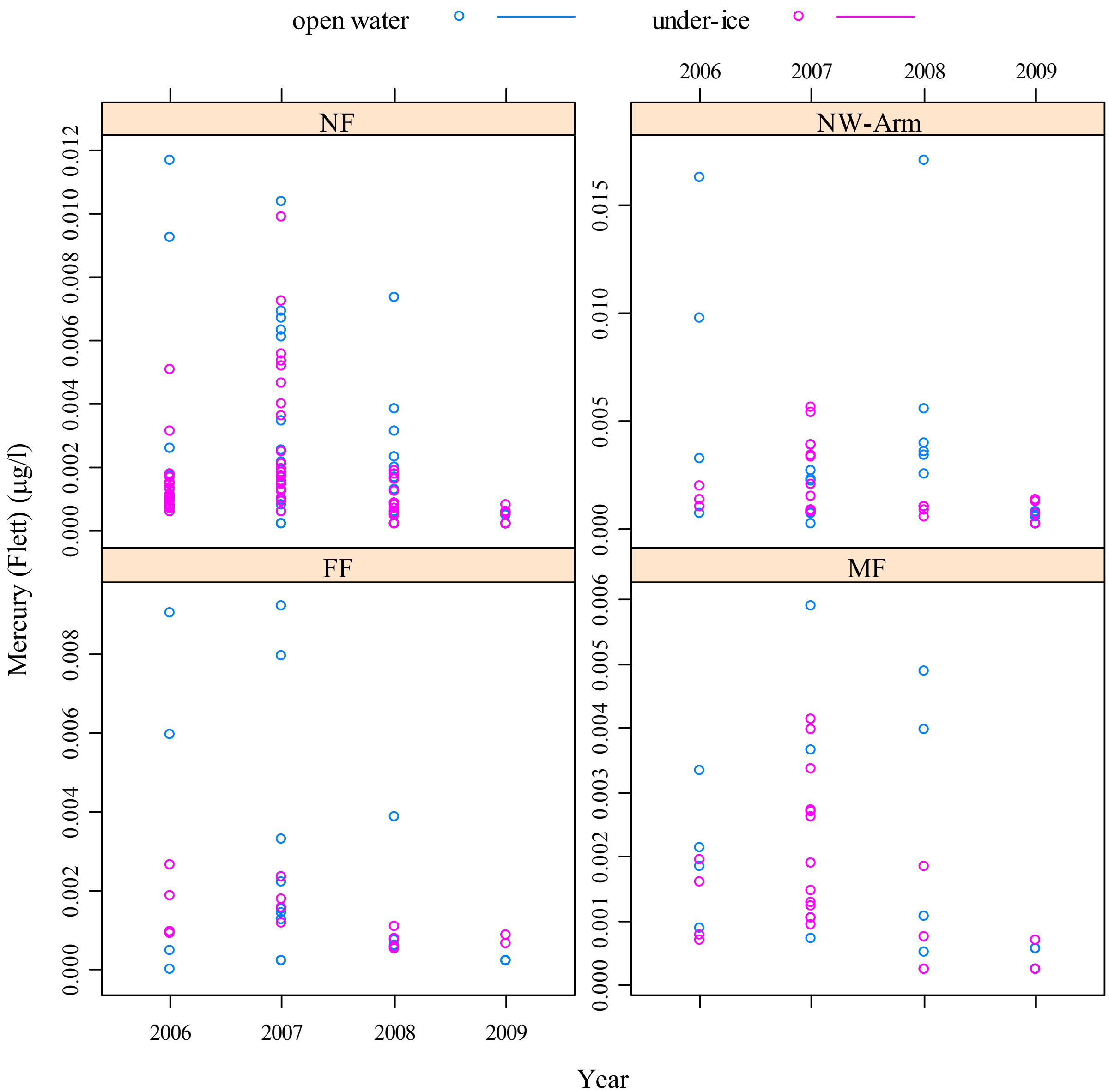

Figure 8: Total Aqueous Hg over Time and Space, 2006 through 2009 only

There is considerable variation in total Hg although the lowest and least variable concentrations are observed in 2009. This is not a consequence of sampling effort as sampling effort is similar across all years except for 2007 where sampling effort was approximately double[23]. There is no

[23] The sampling effort in 2007 is approximately double that in other years. It is possible that the raw data represent sub-samples or some other nuance of sampling. As this does not change the import of the graphics no investigation is conducted at this time.

readily apparent spatial gradient, at least based on the crude segregation by general area. Highest concentrations are observed in the northwest arm.

The following graphic presents the data above using smoothed lines for each station[21].

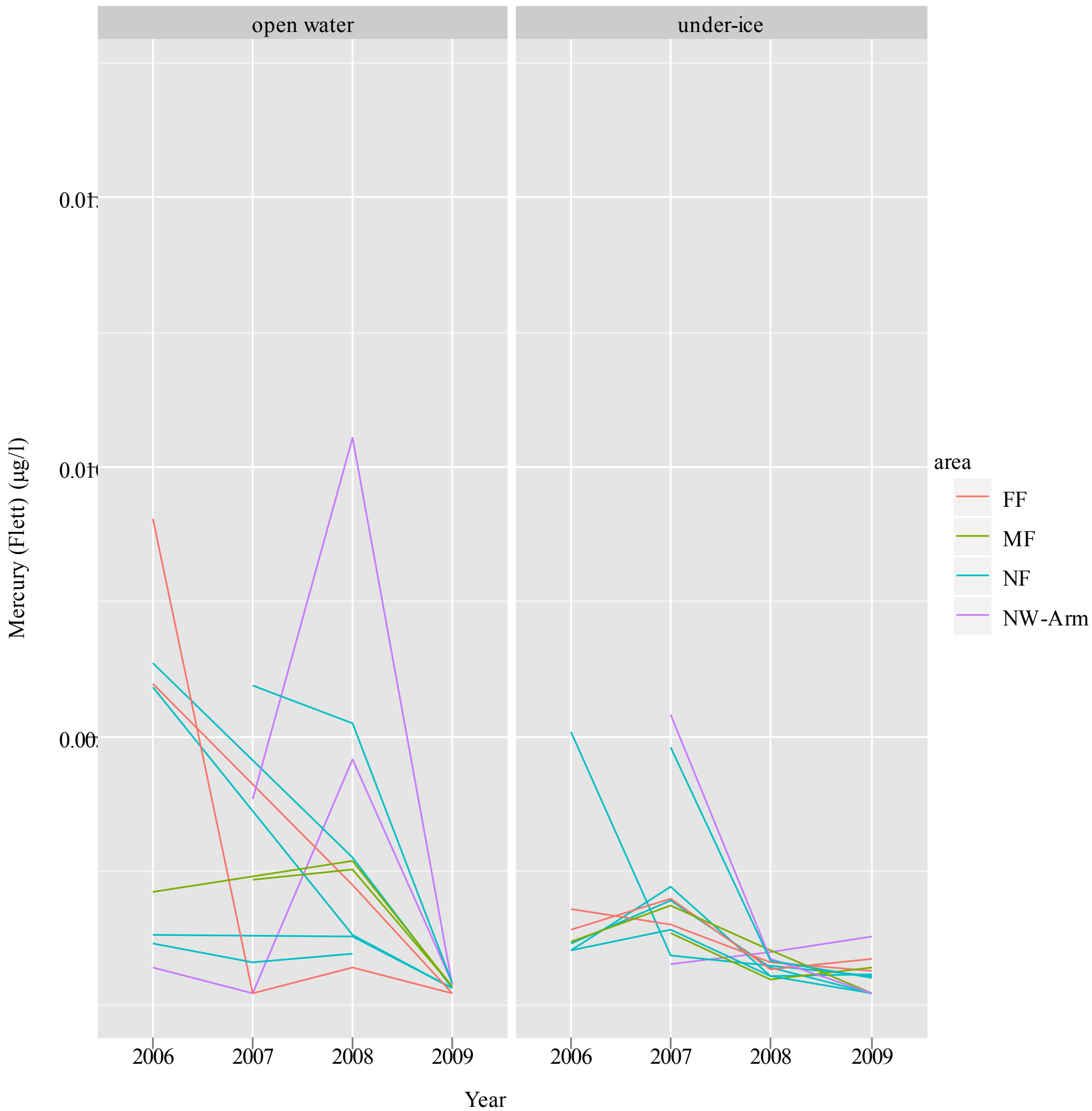

Figure 9: Total Aqueous Hg over Time and Space by Station, 2006 through 2009 only

Figure 9 shows that in general, total Hg levels drop from 2006 onward. Most concentrations, particularly under ice are less than 0.05 µg/L.

3.4.2.1.2.3 Rubidium

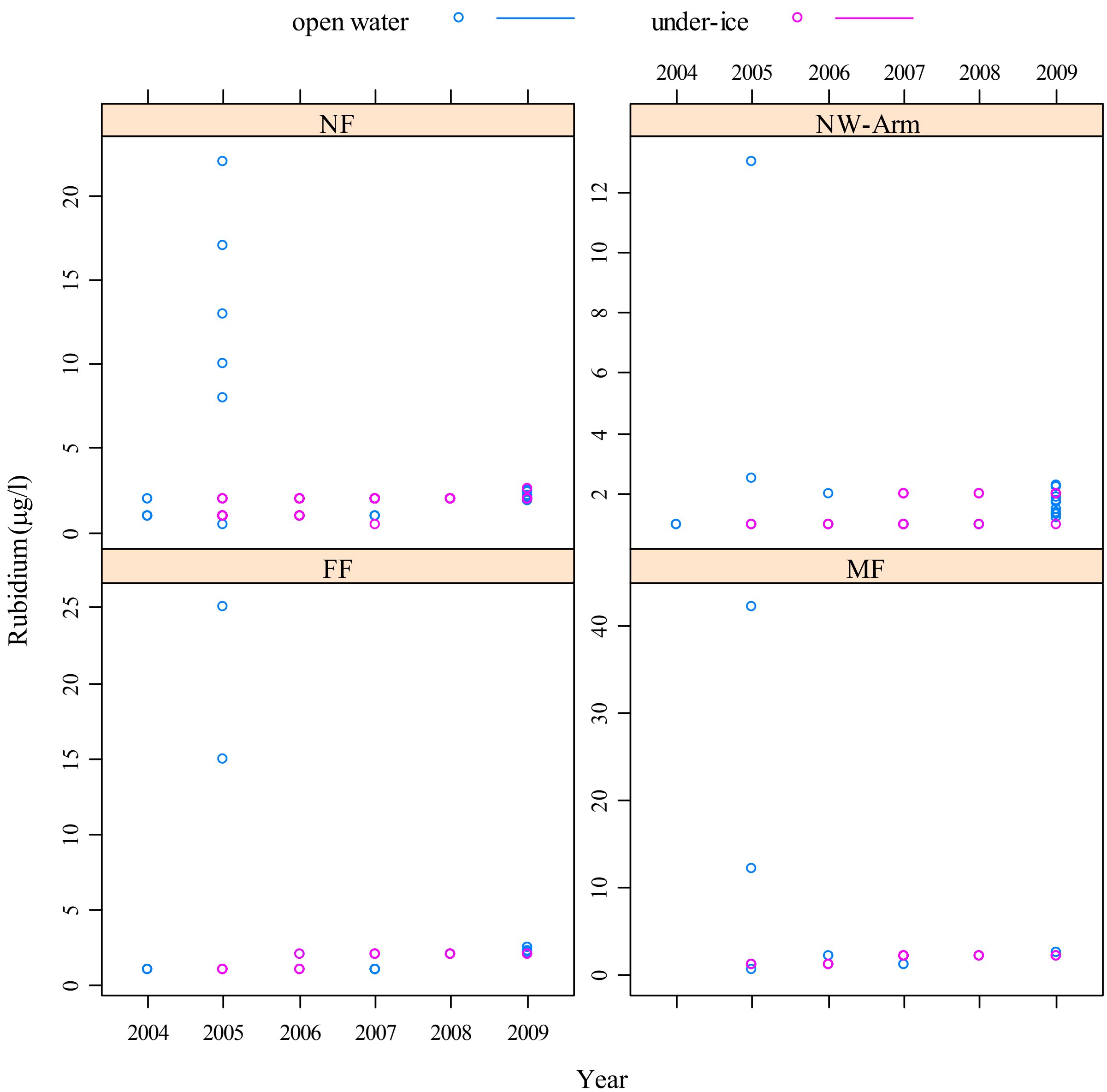

Figure 10: Total Aqueous Rb over Time and Space

Total rubidium levels are highest in the open water season in 2005 obscuring differences in Rb concentrations at other times. The 2005 and earlier data are omitted and the data replotted below.

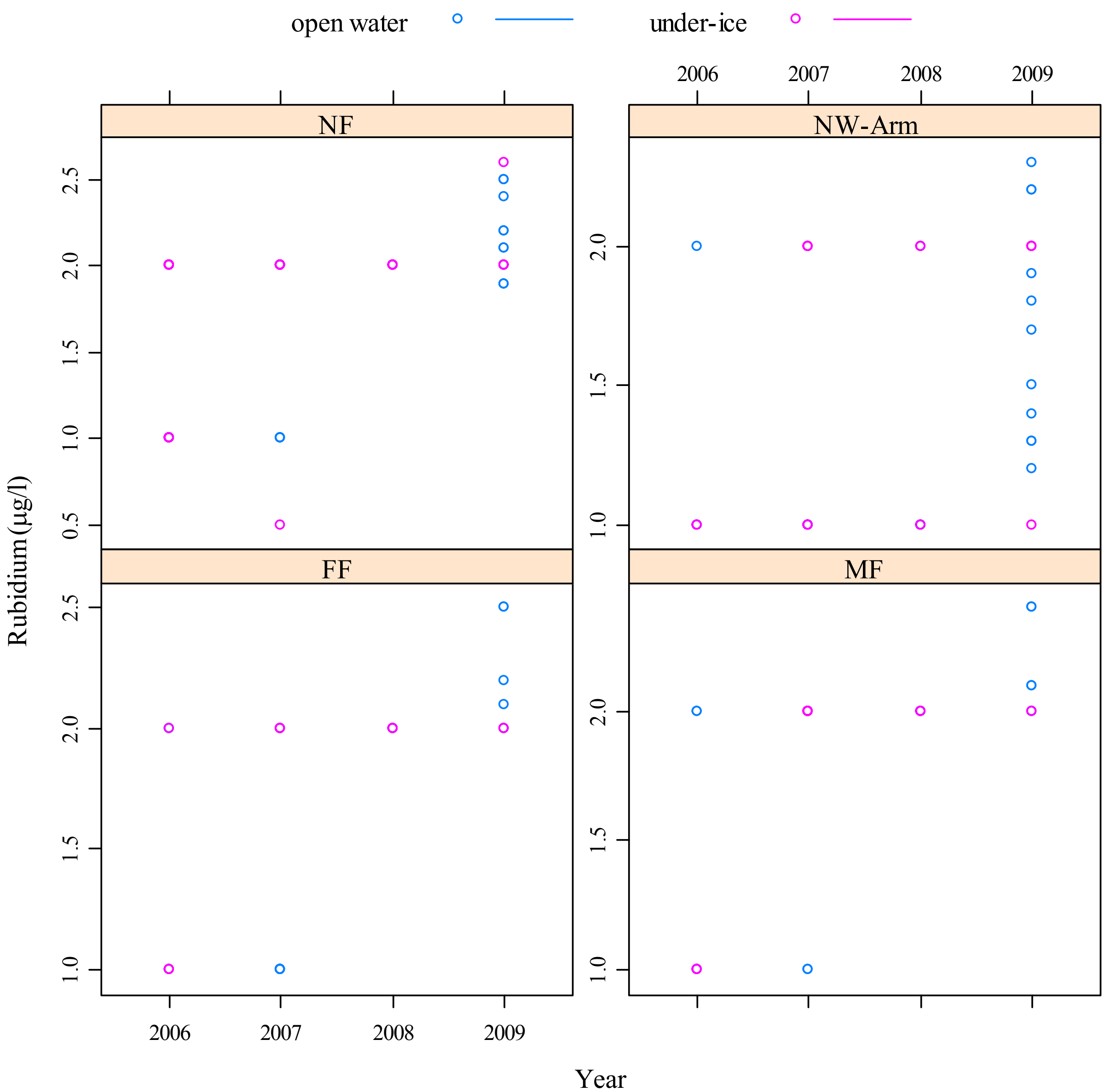

Figure 11: Total Aqueous Rb over Time and Space, 2006 through 2009 only

The data in Figure 11 appear to represent a series of detection limits at 0.5, 1.0 and 2.0 µg/L. However, examination of the "Detection Limit" and "Qualifier" fields show that the only detection limit in the extracted data set is 1.0 µg/L and that there are no data qualifiers. The data may represent a quality assurance failure that should be investigated. The following graphic presents the data above using smoothed lines for each station[21].

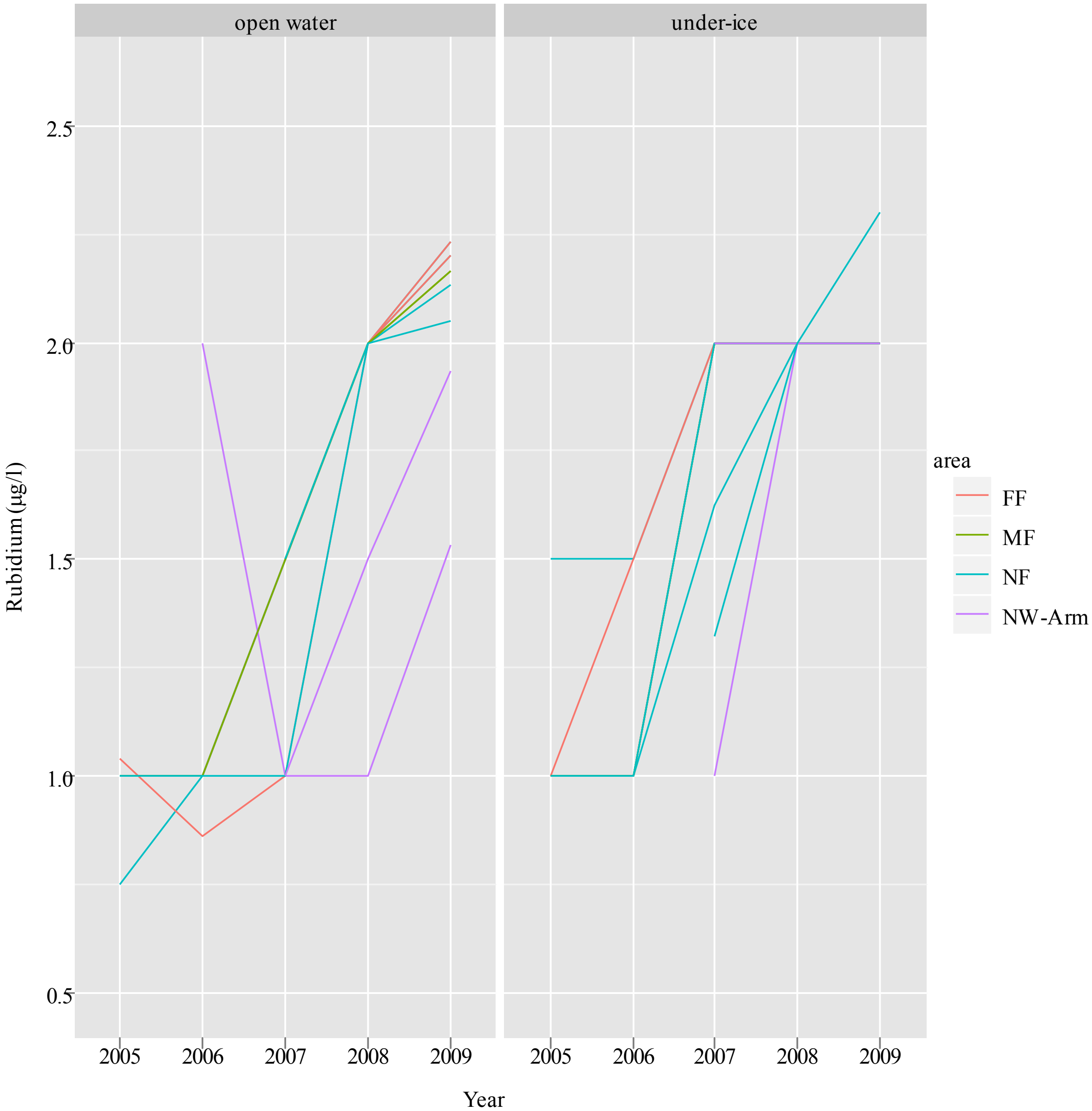

Figure 12: Total Aqueous Rb over Time and Space by Station

It is difficult to interpret Figure 12 due to the limited number of discrete data values.

3.4.2.1.2.4 Thallium

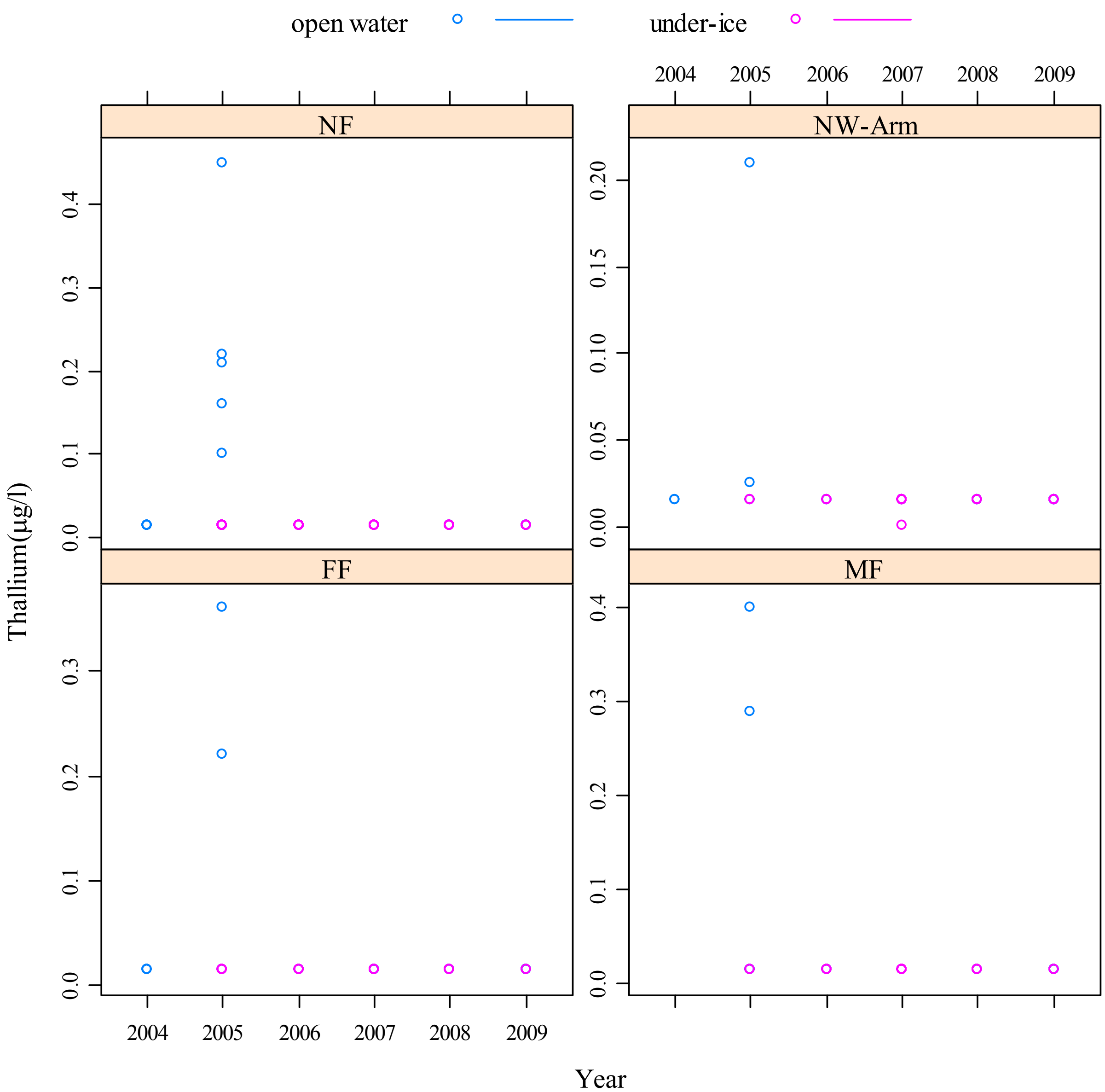

Figure 13: Total Aqueous Tl over Time and Space

Total thallium, like As and Hg shows highest concentrations during project construction. Details subsequent to 2005 are obscured due to these higher values. The next graphic presents only the latter years.

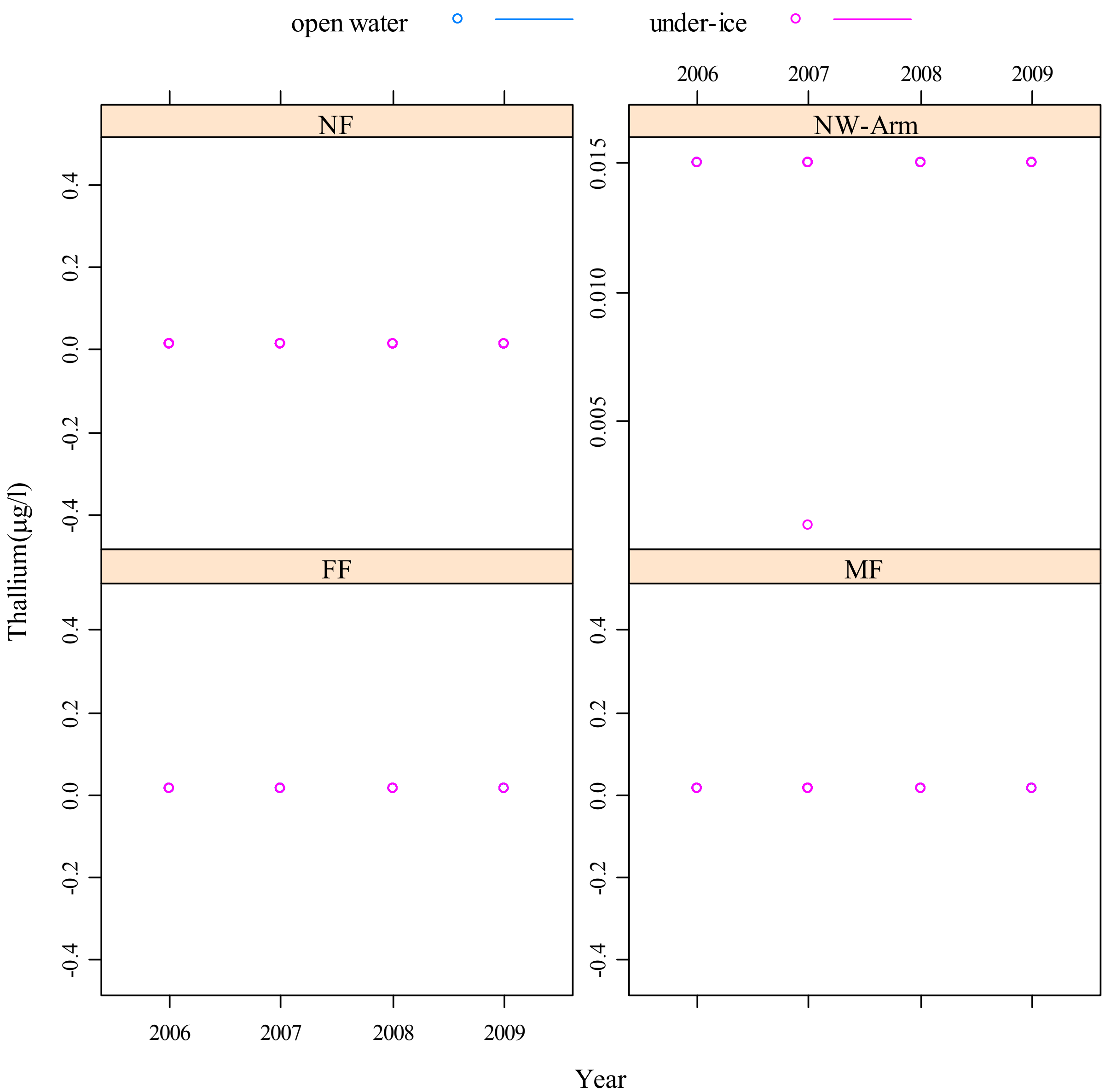

Figure 14: Total Aqueous Tl over Time and Space, 2006 through 2009 only

All but one of the observations in Figure 14represents ½ the detection limit of 0.03 µg/L. The single observation reported as 0.001 µg/L may comprises a data entry error. Given the lack of detections no further investigation is conducted.

3.4.2.1.2.5 Vanadium

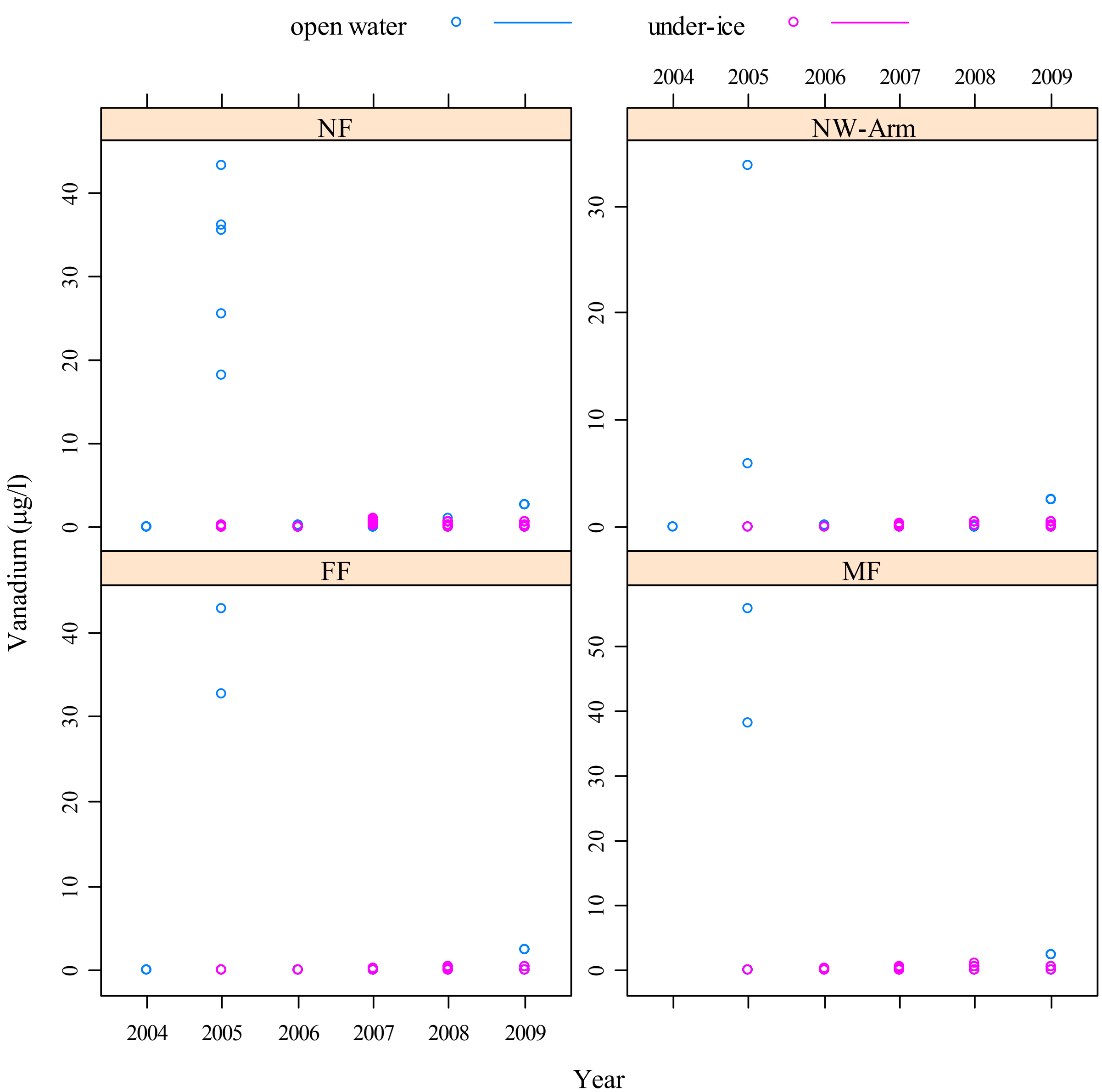

Figure 15: Total Aqueous V over Time and Space

Total vanadium, like As, Hg and Tl shows highest concentrations during project construction. Details subsequent to 2005 are obscured due to these higher values. The next graphic presents only the latter years.

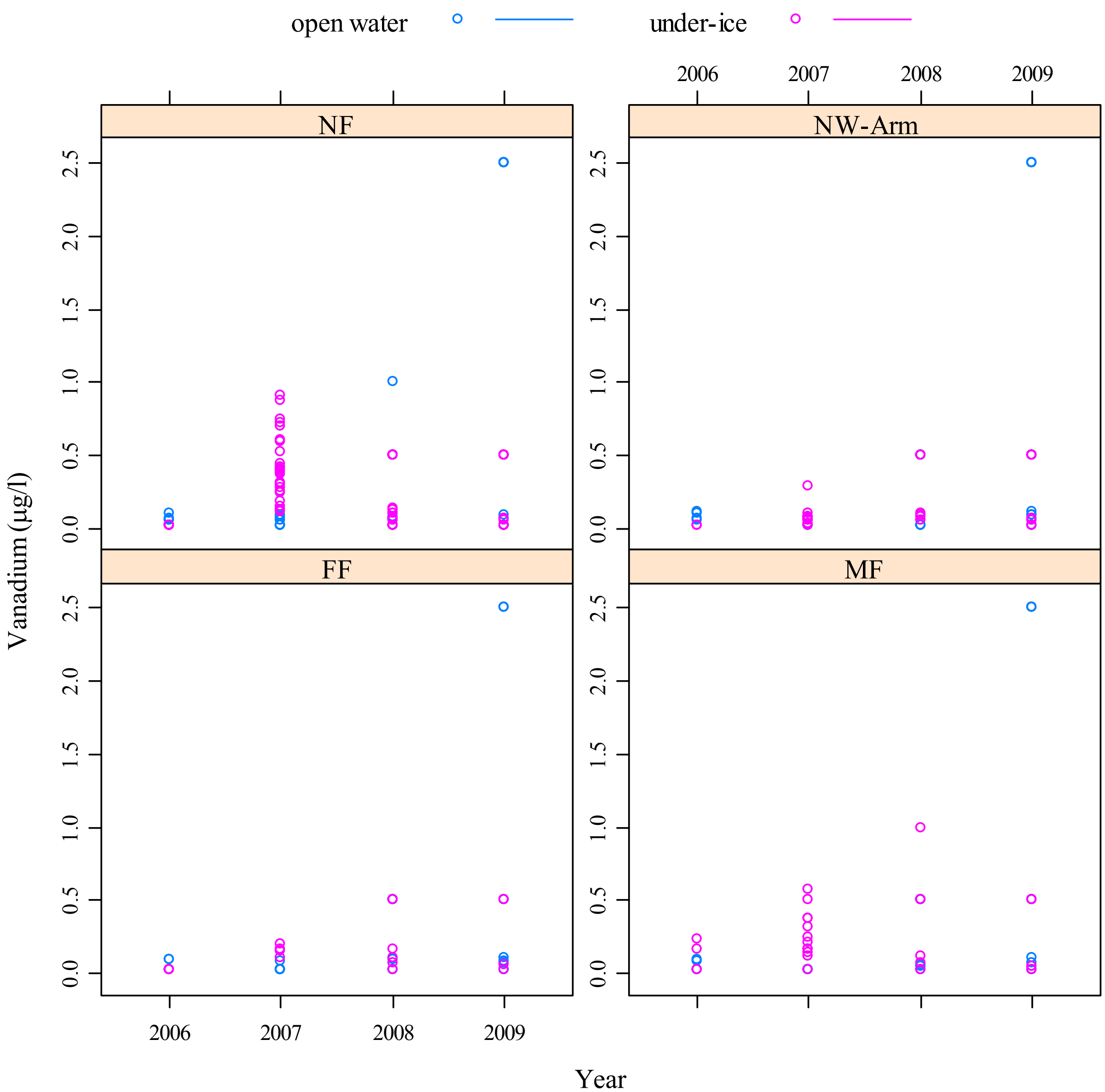

Figure 16: Total Aqueous V over Time and Space, 2006 through 2009 only

Many of the observations in Figure 16 **Error! Reference source not found.**represent ½ the observation-specific detection limits (0.05, 1 and 5 µg/L). Note that the high values observed in 2009 in the open water season represent ½ the detection limit of 5 µg/L. At this point in time this data set with a high proportion of observations less than observation-specific detection limits is not examined further due to the lack of readily apparent trends. The following graphic presents the data above using smoothed lines for each station[21].

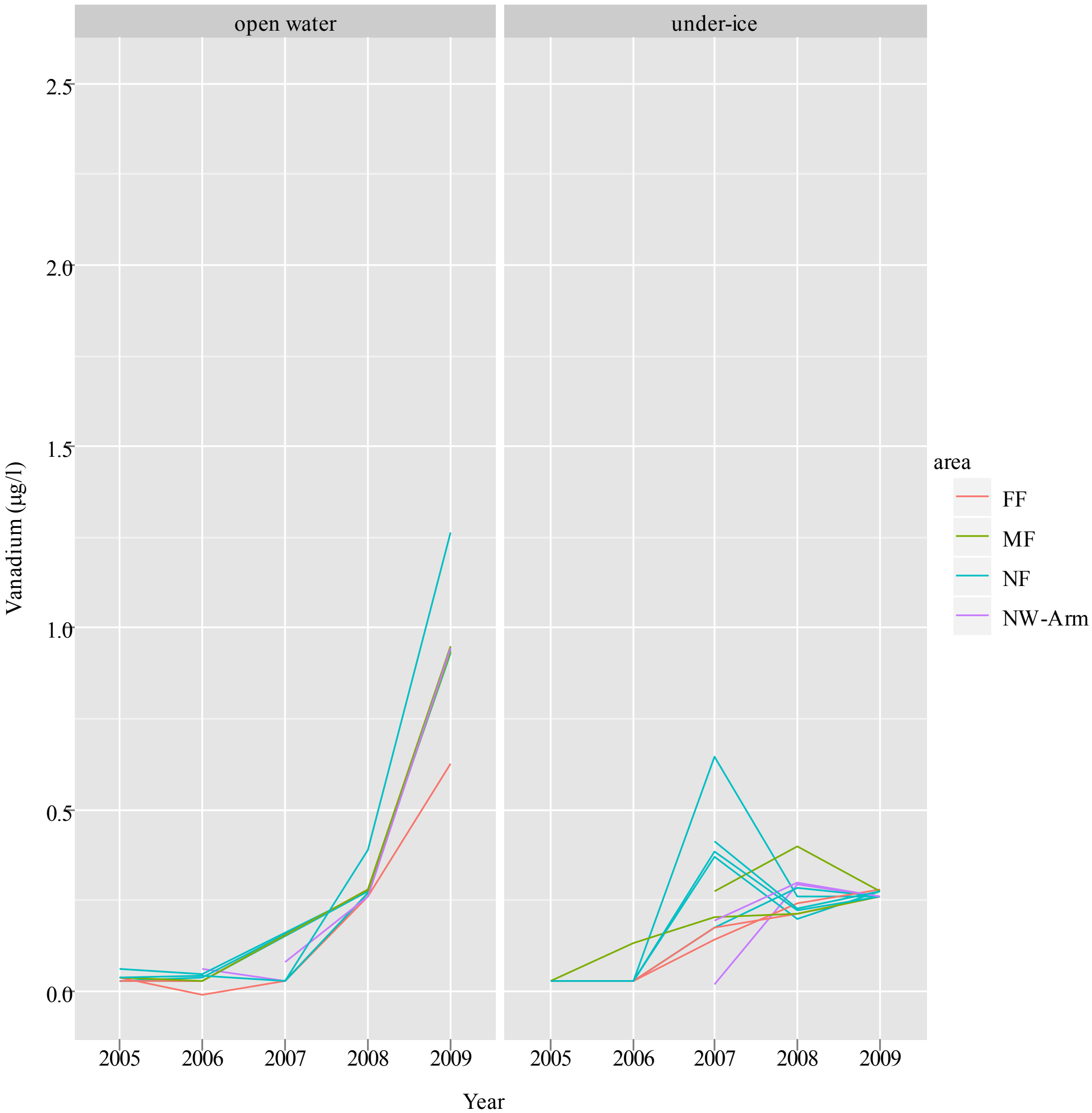

Figure 17: Total Aqueous V over Time and Space by Station

Figure 17 shows that open water total vanadium concentrations are increasing over all stations. Under the ice concentrations are generally (but not monotonically) increasing since 2005. The most variability is observed in the nearfield.

3.4.2.1.3 Summary

A summary of the investigations regarding the analytes selected by SLEMA is presented below.

Table 8 Summary of Investigations of Aquatic Analytes not Listed in the EAR

	Assessment of Spatial Trends	**Assessment of Temporal Trends, Herein**	**Assessment of Temporal Trends (Table 2-14 Golder 2009[24],[25])**					**Comparison with Guidelines**			
Analyte	Area-specific mean comparisons for 2008 data (section 2.2.2.4[24], Golder 2009)	All years, visual assessment of non-parametric regression.	NW Arm	FF	MF	NF	Diffuser	Protection of Aquatic Life, CCME (µg/L)	Drinking Water, CCME (µg/L)	Measurement Type (Table 2-5[24] Golder, 2009)	Concentration (µg/L) (Table 2-5[24] Golder, 2009)
As	no obvious trend (2005-2009 data), Figure 5, herein	Increase between 2006 and 2009 Figure 5.	no obvious trend					5	25	max	0.5
B	Y (Figure 2-45)	NA	no trend	↑	↑	↑ (mid and surface depths only)	↑			max	20 (Table 2-14, Golder 2009)
Ba	Y (Figure 2-44)	NA	↑	↑	↑	↑	↑		1000		18.5
Hg	N Figure 8 herein.	A general drop in concentration from 2006 onward. Figure 8.	N - . Figure 8 herein					0.026	1	max	0.0171
Li	Y (Figure 2-47, particularly under ice)	NA	↑	↑	↑	↑	↑				6.5

[24] Effect of outlying observations removed by Golder not assessed.
[25] Source unless otherwise stated.

	Assessment of Spatial Trends	**Assessment of Temporal Trends, Herein**	**Assessment of Temporal Trends (Table 2-14 Golder 2009[24,25])**					**Comparison with Guidelines**			
Mo	Y (Figure 2-49)	NA	no trend	↑	↑	↑ (mid and surface depths only)	↑ (mid and surface depths only)	73			0.63
Rb	? - Possible quality assurance issue.	? - Possible quality assurance issue.	no trend	no trend	no trend	no trend	no trend			max	2 (Table 2-14 Golder 2009)
Sr	Y (Figure 2-51, particularly under ice)	NA	↑	↑	↑	↑	↑			max	366 (Table 2-14, Golder 2009)
Tl	N (Figure 2-52)	NA	no trend Figure 13 herein.					0.8			<0.03
U	Detections at diffuser and NF during ice cover season, no detections otherwise.	NA	no trend	no trend	no trend	no trend	no trend		20		0.07
V	no obvious general trend (2005-2009 data), See Figure 14 herein.	A general increase. See Figure 14.	↑, Figure 14herein.								NA

The results of sections 3.4.1, 3.4.2.1.2 and summary in **Error! Reference source not found.** show that:

- There are increasing spatial trends for B, Ba, Li, Mo and Sr. Spatial gradients are not apparent for Ag, Hg, Tl, U and V;
- There are increasing temporal trends for As, B, Ba, Li, Mo, Sr and V. Hg, U and Tl do not appear to be increasing over time;
- No conclusions are made regarding either spatial or temporal trends in Rb due to possible data quality assurance issues.
- Of the substances listed in **Error! Reference source not found.** not all have environmental quality guidelines. Of those that do, no exceedances are reported. Note however that the maximum Hg concentration observer in 2009 (0.0171 µg/L) is 65% of the CCME Protection of Aquatic Life criterion.
- No effects on mortality or chronic endpoints for the aquatic toxicity test species or responses were observed on the dates tested.

As a body of evidence it is clear that many of the analytes flagged by SLEMA are increasing in concentration and there are clear gradients from the diffuser implicating the effluent as a source of these analytes. Some of the (relatively) high observations noted during the construction phase are perplexing because if they represented random variation similar results would be expected in latter years. At this point given the low concentration of these analytes relative to concentrations in Northeast Lake, magnitude relative to environmental quality guidelines and lack of repeated high concentrations further investigation is not warranted. However Hg concentrations should be watched carefully due to proximity to the CCME guideline for protection for aquatic life. Also (and although outside the scope of this review) Hg in edible fish tissues should be monitored and possibly also in small bodied fish or benthic macroinvertebrates to provide an early warning of undesirable changes.

Without an exhaustive review of the toxicological effects of the analytes flagged by SLEMA and speculation regarding synergistic, antagonistic and ameliorative effects of toxicity modifying factors it is not possible to discuss the ecological implications of the demonstrable "hot spot". Although toxicity in Snap Lake water samples was not observed in 2009, some toxicity was observed in 2007 and 2009. The toxicity data should be watched.

3.4.2.2 Sediment

In this section, the body of evidence regarding presence and effects of a hotspot attributable to the analytes flagged by SLEMA is summarized in Table 9, below.

Table 9: Summary of Investigations of Sediment Analytes not Listed in the EAR

	Assessment of Spatial Trends		Assessment of Temporal Trends		Comparison with Guidelines		
Analyte	Area-specific mean comparison for 2008 data[26].	Linear Regression[27], Top 5 cm Sediment sample 2008 data, (Table 4-10, Golder (2010))	Linear Regression, Top 5 cm Sediment sample 2004 versus 2008 slope comparison, (Table 4-10, Golder (2010))	Assess whether bulk sediment analyte medians from 1999 to 2004 are lower, higher, or equal to bulk sediment analyte medians for 2008[28]. (Table 4.5[29], Golder 2009)	Table 4-4 ISQG[30] (mg/kg)	Table 4-4 PEL[31] (mg/kg)	Exceedances top 5 cm or bulk (Table 4-8, Golder, 2009)
As	NwA > D> NF > MF > FF[32]	no trend	NA	equal	5.9	17	none
B	D≈ NF > MF ≈ FF > NwA	positive trend	NA	virtually equal	NA	NA	NA
Ba	MF ≈ NwA > NF > D > FF	positive trend	no difference	higher	NA	NA	NA
Hg	NwA > D ≈ NF ≈ MF ≈ FF	no trend	NA	virtually equal	0.17	0.49	none
Li	D ≈ NF ≈ MF > FF > NwA	positive trend	NA	virtually equal	NA	NA	NA
Mo	NwA ≈ NF > MF > D > FF	no trend	NA	lower	NA	NA	NA
Rb	MF > NF > D > FF > NwA	positive trend	NA	virtually equal	NA	NA	NA
Sr	D ≈ NF ≈ MF > FF > NwA	positive trend	higher in 2008	lower	NA	NA	NA
Tl	NwA ≈ MF > NF > D ≈ FF	positive trend	NA	lower	NA	NA	NA
U	MF > FF > NF > D > NwA	positive trend	NA	lower	NA	NA	NA
V	NwA ≈ NF ≈ MF > D > FF	no trend	NA	lower	NA	NA	NA

[26] Designation as "approximately equal" is subjective. Sample size is 1 for diffuser.
[27] The independent variable in the regression analysis is conductivity measured in March and April at the bottom of the water column. This choice of independent variable implies a belief that the single aqueous conductivity measurement is a good surrogate for `distance` from the diffuser (where distance refers to the distance the effluent travels from the diffuser) despite possible temporal and vertical changes in conductivity at a given site.
[28] Comparisons of limited use because all Snap Lake samples are compared, not just samples near diffuser.
[29] Section 4.4.4 of Golder (2009) regresses sediment analyte concentrations on bottom water conductivity to assess whether sediment concentrations are related to effluent (as measured by its surrogate, conductivity). While this may be reasonable it is important to note that water quality measurements comprise a single measurement in time whereas a sediment quality measurement represents (in a depositional environment) an accumulation over time. Thus the absence of a significant correlation should not be construed as a lack of mine effect.
[30] ISQG – interim sediment quality guideline
[31] PEL – probable effect level
[32] D – diffuser, NF – nearfield, MF – midfield, FF – farfield, NwA – northwest arm

3.4.2.3 Overall Summary

Table 9 shows that most of the analytes flagged by SLEMA show a concentration gradient commensurate with the diffuser as the analyte source. However the data as a whole do not support a temporal trend. While Golder (2009) attributes this to natural variability[33] and a short time series another possible explanation is the very low sedimentation rate in Arctic lakes[34]. Collecting deep sediments sediment samples relative to the depth of the most recently deposited sediments can dilute recent inputs to the point where no changes are observed.

In the two cases where sediment quality guidelines are available for the analytes flagged by SLEMA, no exceedances were detected in 2008. Golder (2009)[35] states that the benthic macroinvertebrate survey temporal analyses were inconsistent for nearfield and midfield sites. Spatial differences were largely restricted to differences in midges (Microtendipes) between Snap Lake and Northeast Lake.

The body of evidence regarding a `hotspot` and potential effects of the analytes flagged by SLEMA is moderate. The `dose` portion of the dose-response equation (as measured by sediment analyte concentrations) is / may be[36] weak due to the effect of dilution by historic sediment. The response portion of the equation is based on the review of the 2008 benthic data interpretation, moderate (the benthic macroinvertebrate community analyses) or non-existent (sediment toxicity tests). Five benthic macroinvertebrate community metrics were evaluated by Golder, (2009 Table 5-9). Negative changes in diversity (decreased) and evenness (decreased) in the nearfield and midfield areas relative to Northeast Lake were observed. This suggests based only on proximity to the effluent source that significant changes in the benthic community are observed. Whether these changes are solely attributable to the analytes flagged by SLEMA cannot be determined. Certainly other analytes discharged by DeBeers may be implicated.

The overriding conclusion (regardless of what analytes are implicated) is that significant adverse changes to the aquatic community are observed proximal to the discharge point. Of the metrics investigated, richness and diversity are (arguably) the two most "important". Of these two, diversity dropped and richness showed no change.

3.4.3 Recommendations

- The data investigation was challenged by changing monitoring locations over time. For example the SNP 02 20 a, b, c series were collected from 2004 to 2006. These stations

[33] Golder (2009) discusses `large` natural variability in the concentrations of sediment analytes. This may be due insufficient sampling and compositing. The sediment sampling plan might profitably be reviewed.
[34] A recommendation to modify sediment sampling depths is presented in section 5.
[35] Only the conclusions reached by Golder (2009) were used in this section as the conclusions are used as ancillary information to assess the effects of the potential `hotspot` as indentified by SLEMA. A review of the benthic macroinvertebrate survey is presented in section 3.1.1.
[36] This statement can be refuted by collecting shallower sediment samples.

were dropped in 2007. In 2006 monitoring began at SNP 02 20 e and f. Thus the SNP 02 20 data series, which is proximal to the diffuser is disjointed.

DeBeers should commit to consistent long-term monitoring using the same techniques, locations and depths.

- Cognizance of the low sedimentation rates typical of Arctic lakes led numerous reviewers of another AEMP (Diavik Diamond Mines) to suggest that shallow surficial sediment samples be collected rather than deeper samples. Golder (Section 4.1.1, 2009) also acknowledges the low sedimentation rates in Arctic lakes and uses only the top 5 cm from an Ekman grab rather than the entire sample which increases the relevance of sediment chemistry analyses because a shallower sample better represents recent inputs. However if the combination of inputs and sedimentation rates are low the mass sediment loading will also be low and concentrations will still be diluted by historic sediment. Thus reviewers of the Diavik Diamond Mine AEMP suggested that 1 cm cores be collected. This recommendation was accepted by the Wek'eezhii Land and Water Board. DeBeers should also collect 1cm sediment samples in conjunction with the samples being collected now to enable a correlation to be established. Once a suitably strong correlation is established the deeper sampling may be discontinued.

- The extent of the observed changes should be compared to the expected changes described in the EAR.

- Spatial analyses within Snap Lake could augment[37] the comparison among areas including Northeast Lake to further define the spatial extent of changes within Snap Lake. This recommendation sidesteps the issue of natural variability between Snap and Northeast Lakes noted by Golder (2009).

3.5 Congruence with Environmental Monitoring Best Practices

This section addresses the following question posed by SLEMA:

"Bring to SLEMA's attention anything that you feel may derogate the Snap Lake Aquatic community as well as anything that may be out of compliance with national or territorial standards or best practices."

3.5.1 Synoptic Sampling

When assessing the potential environmental impacts using different lines of evidence and some lines of evidence are often affected by others. For example contaminants in water or sediment may adversely affect phytoplankton, zooplankton and or benthic and/or macroinvertebrates. It is

[37] Comparisons with a reference area are still necessary as the absence of a gradient in Snap Lake could simply indicate either a lake-wide effect or lack of effect entirely.

desirable to collect samples representing different lines of evidence the same time whenever possible and certainly at the same locations. This is a known as synoptic sampling. INAC (2009) and Environment Canada (2002, 2004) recommend synoptic sampling.

In 2007, only eight of fifteen plankton monitoring stations are in the same locations as water-quality sampling locations. Of the remaining seven plankton monitoring stations, SNAP11 and SNAP02 likely correspond to their letter to counterparts i.e. SNAP11a and SNAP02a; SNAP31 is geographically close to SNAP29 but is on the other side of the connecting channel between the Northwest Arm and Snap Lake proper and therefore could represent a markedly different location. Some of the plankton sampling locations that are not matched to water sampling locations were newly added in 2007. SNAP30 and SNAP31 were added to assess "potential differences in water quality related to the treated domestic wastewater discharge to the wetlands and treated effluent from the diffuser outfall" Golder (2008). Another station (SNAP32) was added in 2007 for reasons not stated in Golder (2008).

The addition of sampling locations for plankton monitoring shows adaptation reflecting expressed concerns which is commendable. However, not all plankton sampling stations that were unmatched with water-quality sampling locations were added for this reason. Even when sampling locations are added for good reasons, care should be taken to ensure concordance with other sampling locations and times to the extent reasonably possible. This allows assessing the correlation between lines of evidence and drawing much stronger conclusions than otherwise.

The suggestion to make sampling different lines of evidence synoptic is consistent with best national practices (McDonald et al. 2009; and Environment Canada 2002, 2004) and reflects the intent of Part G, 2b. vii of the water license (MV2001-L2) which states that "The AEMP shall include, but not be limited to the following a process for measuring the Project-related effects on the communities of zooplankton and phytoplankton due to changes in water quality".

3.5.2 Constancy of Sampling

The data investigations presented in section 3.4 were challenged by changing monitoring locations over time. For example the SNP 02 20 a, b, c series were collected from 2004 to 2006. These stations were dropped in 2007. In 2006 monitoring began at SNP 02 20 e and f. Thus the SNP 02 20 data series, which is proximal to the diffuser is disjointed.

Another break in the temporal time series for benthic macroinvertebrates occurred when DeBeers switched from winter sampling to summer sampling without calibrating one set of samples against another. This decision will limit the utility of earlier benthic macroinvertebrate data.

Other, less substantive changes to sampling locations were noted during the review but not highlighted herein.

DeBeers should commit to consistent long-term monitoring using the same techniques, locations and depths. This will make the data already collected most useful. If a change is necessary, the new data should be calibrated to older data prior to ceasing to collect data in the earlier manner.

4 General Comments

In general, the 2008 AEMP report is very well laid out with the necessary information presented in the appropriate sections in a logical and clear manner. Some of the ancillary information such as ecological attributes of taxonomic subgroups and the implications of observed changes were very insightful.

Reconciling commonly used terminology with that used in other caught aquatic effects monitoring-related documents such as Environment Canada (2002) is both helpful and indicative of clarity of thought. Finally, the ability to extract text and bookmark a document as large as the AEMP certainly facilitated the review.

4.1 Utility of Reference Lake

An extensive consultation regarding selection of a suitable reference lake took place prior to sampling of Northeast Lake. The rationalizations for choosing Northeast Lake as a reference lake have not been reviewed at this time. However the use of Northeast Lake as a basis for comparison is reviewed because this comprises a review of the experimental design underpinning the AEMP.

Golder (2009, pg. 3-6) states: "detailed statistical analysis and interpretation were not completed (for zooplanktonic and phytoplanktonic metrics) because: differences in the trophic status of these two lakes under background conditions indicate that direct statistical comparisons are inappropriate". At this point in time it is not clear why a reference lake was chosen such that comparisons of variables that are associated with trophic status such as total dissolved solids, total phosphorus, total nitrogen, phytoplanktonic taxonomic composition and biomass, and zooplanktonic taxonomic composition and biomass are "inappropriate".

Also, Golder (2009, pg. 3-6) states that: "plankton monitoring will be continued in 2009 to begin tracking annual open-water variability in the plankton community as well as the year-to-year variation within Northeast Lake". This presumes that variation in the metrics being compared is independent of the trophic status differences that preclude comparison of quantiles of the metrics themselves. Golder should clarify that comparison of the variances of metrics is tenable despite the differences noted between the two lakes.

5 Additional Recommendations

Some additional recommendations that do not fall into any previous section are presented below. However, there are a large number of documents associated with aquatic effects monitoring in Snap Lake. The primary documents reviewed are the 2007 and 2008 AEMPs and while other documents were searched, the search was not exhaustive. Thus some of the recommendations provided below may have been addressed previously. If so, I apologize for making an unnecessary recommendation.

- At this point in time a detailed evaluation of the plankton monitoring locations has not been conducted. DeBeers should investigate adjustment of the plankton monitoring stations to make all plankton sampling locations consistent with water quality monitoring stations which is standard practice for monitoring the aquatic environment (INAC, 2009; Environment Canada, 2004). Currently the plankton sampling locations SNAP30 and SNAP01 are not supported by the full suite of water quality measurements collected. SNAP31 and SNAP13 are very close to water quality monitoring stations which might serve as surrogate locations.

- Water license item 2c (assessment of trace metal burdens in benthic macroinvertebrates) was removed from the AEMP as it was "shown not to be realistically achievable". If not already done so, DeBeers should provide further explanation before this useful measure and endpoint is omitted from the AEMP. This should include an estimate of the required sampling effort to measure trace metal burdens in benthic macroinvertebrates using the observed range of macroinvertebrate densities in Snap Lake as best and worst case scenarios.

- The current benthic macroinvertebrate experimental design is defined as a control/impact study and marks a change from the original gradient design. A visual assessment of the 2007 conductivity data indicates a gradient in conductivity (which connotes presence of the effluent) with increasing distance from the diffuser. If such a gradient exists, a gradient "analysis" should be used regardless of what the experimental design is called. A data analytic scheme that acknowledges known spatial relationships among treatments is more powerful than an analytical scheme that does not.

- The power achieved in the statistical within-year, among area comparisons should be reported. Only a range of achieved power is presented making it impossible to determine the ability of the program to detect differences among areas. This should be accompanied by the minimum detectable difference.

- Multivariate analyses of benthic macroinvertebrate data should be considered. Various forms of multivariate analyses were used to assess water quality and plankton data.

- Two recommendations made regarding the use of the reference lake are made in section 4.1.

- The holding times for some parameters such as hexavalent chromium were exceeded for all samples collected and warnings were issued by the analytical laboratories used. Some of the holding times were exceeded by such a large amount that the data were invalidated by the analytical laboratories. This is an issue when samples are collected in remote locations and submitted for analysis in distant laboratories. Every reasonable effort should be made to ensure that holding times are not exceeded. The results in Table A2-5 (Golder, 2009) suggest that the Maxxam laboratory not be used due to the very high percentage of samples for which minimum hold times were exceeded. Note that Maxxam laboratory was used for the analysis of samples used for interlaboratory comparisons and not the majority of samples.

- Golder (VME) (2001) concluded that that DeBeers could likely meet DFO blasting requirements but that "monitoring of underwater overpressure and ground vibration effects during the initial stages of regular production blasting should however, be carried out to better define the attenuation characteristics developed for this site." The MVEIRB (2003) recommended that: "A follow-up program to the July, 2001 blast monitoring program will be undertaken to allow for refinement of the equations used to calculate peak particle velocity and overpressure once mine production begins to ensure that predicted blast overpressure and ground vibrations estimates are correct (IR 3.9.12b)". At this point in time investigation by SLEMA (D. White. pers. comm.) failed to find evidence of this work. A letter should be written to DeBeers requesting results of this study. If the study has not been completed, the MVEIRB recommendation should be followed.

6 Citations

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Appendix 1: Assessment of Early Benthic Macroinvertebrate Data

Introduction

The terms of the contract under which this review was conducted were drafted in December of 2009 prior the change in the benthic macroinvertebrate sampling program. The review of the benthic data began prior to the release of the 2009 AEMP report with the intention of adding the 2009 benthic macroinvertebrate data to the review. However due to the change in sampling season for 2010 (which will likely preclude defensible integration of pre-2010 benthic data), the 2009 benthic data were not included within this review as any comment in terms of improving AEMP design based upon statistical analyses of data collected during the winter is moot. This earlier review is however retained within this section.

Sampling Design

The original sample design as discussed in section 5.3.2 of Golder (2008) was a gradient design. Vertical mixing of the effluent was demonstrably affected by depth and therefore standardized depths were chosen for sampling. This is an improvement to the original monitoring design as it acknowledges a known confounding effect. A change to a Control-Impact design also occurred due to the low conductivity measurements at locations other than the nearfield stations. While this was correct in the early days of the AEMP it appears to be no longer correct. See figure 5-3, DeBeers (2008), years progressing from 2005 through to 2007, where the midfield conductivity measurements increase with time.

The current experimental design is designated as control/impact, with a reference area (northwest arm of Snap Lake, n = 3), a NF exposure area (north basin of Snap Lake, n = 4[38]) and a MF exposure area (south basin of Snap Lake, n = 5). Sampling of Northeast Lake (as an additional reference area) was not completed due to logistic and weather-related issues. Also, two stations in the northwest arm (SNAP21, SNAP22) and one station in the near-field area (SNAP13) were not sampled due to logistic constraints. These sampling omissions represent a 40% loss of sampling effort in the Northwest Arm and a 20% loss in the nearfield.

If a gradient in conductivity (which connotes presence of the effluent) with increasing distance from the diffuser is noted, consideration should be given to the use of a gradient "analysis[39]" in which the spatial relationships between stations is acknowledged. This analysis is more powerful than an equivalent Control-Impact "analysis" when a gradient exists.
Golder (2008) used a critical effects size of ±2 SD, estimated from reference area data following Environment Canada, (2002). In 2004 and 2005 the Northwest Arm was designated as a reference area. It is not clear what was used as a reference area subsequent to 2005. Given that

[38] I noted that an additional nearfield station will be added in 2008.
[39] Note that the term "design" can include both the physical location of samples, the nature in which observations are collected from each sampling location and the data analytical scheme.

the Northwest Arm is now not suitable as a reference area it is critical that samples be collected from a suitable[40] reference lake as soon as possible.

Data collected from different areas within a year were compared using analysis of variance (ANOVA). Comparisons were declared statistically significant using a Type I error rate (α) of 10%. The statistical power achieved is summarized but results from individual tests are presented. The range of power achieved is reported as 0.66 (unacceptable) to 0.91 (just acceptable). It would be useful to see what proportion of the within-year comparisons achieved a Type II error rate = Type I error rate as recommended in INAC (2009) and Environment Canada (2002, 2004).

The "responses" used in the ANOVAs conducted to assess changes in the benthic macroinvertebrate community data are some of the commonly used metrics that summarize the information represented by the enumeration of taxa at a single location. Of these, my experience has been that Simpson's diversity and taxonomic richness corroborated other lines of evidence. Each of the metrics used presents a summary of the community at a location but each metric is not necessarily independent (from an information theoretic perspective) of one another. Thus, in addition to stating which metrics differed or did not, DeBeers should discuss 1) why each metric was chosen; 2) which metric or metrics should carry more weight in the interpretation; and, 3) the ecotoxicological implications of some metrics varying while others do not. Finally, there are a variety of evenness indices some of which are better than others. DeBeers should state which evenness index was used and why.

The use of ANY single summary metric has been criticized by some due to information loss. Univariate (one at a time) comparisons of summary metrics have been gradually superseded by multivariate comparisons. See for example and Resh and McElravy, (1993) and Environment Canada, (1998). Warwick and Clarke (1991) found that multivariate techniques (ordination and classification) were more sensitive than univariate (indices) or graphical methods (relative abundances or biomass of single species plotted against covariates such as site, etc) in detecting changes in benthic community structure.

DeBeers concludes that the "sensitivity of statistical tests was appropriate for variables other than total density" because the among-area differences (excluding Station SNAP23) "were within the ranges in 2 SD for richness, diversity, evenness, and dominance, as well as total density in the mid-field area." This statement presumes that 1) the reference area is appropriate and 2) the standard deviation in the reference area is well estimated.

With respect to the first of these issues it is not clear that the reference area is appropriate in the latter (2007 and possibly 2006) years of monitoring. This issue should be resolved when an appropriate40 reference lake is used as the basis for comparison. The second issue merits some discussion.

Consider an average estimated from a few observations. There will be some variability associated with that average, and, as the number of observations used to estimate the average

[40] I am aware that Northeast Lake was evaluated and accepted as a reference lake but I have not reviewed that rationalization. And

increases, the variability and will begin to stabilize. At this point there is little benefit in collecting additional operations. Now consider the standard deviation. The standard deviation, like the average is a statistic estimated from observations and therefore has an associated variability. If the variability of the standard deviation is very large there may be a little that can be meaningfully said regarding the sensitivity of the statistical analyses conducted. The sensitivity of the statistical analyses conducted is best assessed by the minimum detectable difference. If this minimum detectable difference is small relative to a well-estimated standard deviation (of the reference condition, in this case) then the statistical test is sensitive; otherwise it is not.

Confidence intervals[41] for the standard deviations for the summary metrics presented in Table 5-11 of Golder (2008) are presented below.

Variable	**Year**	**n**	**Mean**	**SD**	**95% Lower Confidence Limit**	**95% Upper Confidence Limit**
Total Density (# / m2)	2004	8	948	332	219.5098	675.7114
	2005	5	742	381	228.27	1094.824
	2006	6	762	668	416.9711	1638.349
	2007	2	366	-		
Richness [no. of taxa]	2004	8	12.6	2.7	1.78517	5.495243
	2005	5	13	3.3	1.977142	9.482729
	2006	6	12.2	3	1.872625	7.357854
	2007	2	10	-		
Simpson's diversity index	2004	8	0.82	0.07	0.046282	0.142469
	2005	5	0.82	0.06	0.035948	0.172413
	2006	6	0.82	0.1	0.062421	0.245262
	2007	2	0.72	-		
Evenness	2004	8	0.5	0.12	0.079341	0.244233
	2005	5	0.48	0.21	0.125818	0.603446
	2006	6	0.55	0.18	0.112357	0.441471
	2007	2	0.46	-		
Dominance [%]	2004	8	29.7	10.8	7.140679	21.98097
	2005	5	32.2	10.2	6.111165	29.31025
	2006	6	38.6	8.3	5.180928	20.35673
	2007	2	47.3	-		

Golder (2008) investigates temporal changes in benthos. Some aspects of the temporal data collected in the reference area are summarized below.

[41] Confidence intervals assume that the SD was estimated from a normal distribution. This was NOT tested using the raw data. Confidence intervals follow Mood et al (1974).

Table 10: Benthic Macroinvertebrate Reference Area Sampling Effort over Time

Source	Reference Area Locations (# Grabs)	Total Benthic Macroinvertebrate Abundance in Reference Area	Sampling Time
Golder (2008)	SNAP20 (6), SNAP02A(6), SNAP03(6), SNAP23(6)	297	April 8 to 16, 2007
Golder (2007)	SNAP20 (6), SNAP02A(6), SNAP23(6)	448	Apr 8 -16, 2006
Golder (2006)	SNAP01 (5), SNAP02(5)	453	April and May, 2005
DeBeers (2005b)	SNAP01 (3), SNAP02(3)	2666	May 5 -11, 2004

Table 10 shows that:

- Sampling effort varies substantively over the years with 24 grabs being collected in 2007 and only six being collected in 2005;

- There is a large difference in benthic macro invertebrate abundance in the reference the area over years. The largest number of benthic macroinvertebrates was collected in 2004. This can be explained as follows: Organisms grow rapidly in the Arctic spring. Differences in abundances over a month can be due to small organisms passing through a 500 µm mesh and then a month later being large enough to be retained by the same mesh size. Consider the abundance of the sub-family *Tanytarsini* (relatively small chironomids) in May of 2004 (Table B-2 of that year's AEMP report) and their virtual absence earlier in the year, April of 2007 (Table D-1of that year's AEMP report). This difference, although confounded by years, is likely a maturation effect. Also in 2004 (relative to 2007) there seem to be breeding bivalves.

Table 10 shows the importance of consistent sampling within the same time period to ensure comparability of results over years.