An assessment of the ecological integrity of streams in the South Nahanni Watershed: development and application of a reference condition model

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Detailed Synopsis

Objective

We developed and applied a reference condition approach model using benthic macroinvertebrate community structure to assess possible impairment of rivers and streams in the South Nahanni Watershed, Northwest Territories, Canada. The reference condition is a powerful alternative assessment approach because large numbers of samples from many sites rather than collections of multiple samples from a few sites define reference conditions. Additionally, the full range of variability in reference conditions of biological communities is reduced into smaller sub-groups that provide more appropriate assessments of exposed, potentially degraded sites. Once developed the condition model provides a cost-effective means of reporting on the ecological condition at both small and large scales for many types stressor types.

Approach and Application

Using recognized statistical methods, we developed a reference condition approach model for medium- sized rivers and streams in the South Nahanni Watershed using data from 78 sites consisting of 53 and 25 sites sampled in 2008 and 2009, respectively. Clustering of benthic macroinvertebrate communities from these 78 sites revealed three different assemblage types consisting of 22 to 30 sites, with a fourth assemblage consisting of only two sites. We applied a discriminant function analysis using 37 potential variables to predict group membership. This analysis showed that group membership (i.e., benthic macroinvertebrate assemblage type) was strongly predictable using six variables (i.e., latitude, stream order, percent forest cover in the watershed, percent ice cover in the watershed, maximum June air temperature, average water velocity). The overall classification success of the discriminate function model was high (69%) and ranged from 63.3% to (assemblage 2) to 79.2% (assemblage 3).

We applied the reference model to assess the condition of potentially exposed sites located: i) adjacent and immediately downstream of the complex of tailings ponds at the Cantung Mine and ii) downstream of the mining effluent discharge of Harrison Creek to the receiving waters of Prairie Creek. We assessed the condition of test sites sampled in 2008 by comparing the extent that its benthic macroinvertebrate community deviated from reference conditions using probability ellipses that identify four bands of similarity between a test site and reference sites: Band 1 - ≤ 90% (equivalent to reference sites [unimpaired]), Band 2 - 90 to 99% (possibly different from reference sites) and Band 3 > 99% [impaired], Band 4 > 99.9% highly impaired). We also assessed the condition of eight duplicate samples and 18 samples collected using a fine mesh net (250 µm) to assess comparability with samples collected using the routine 400 µm mesh net.

Main findings of site assessments

Our assessment of exposed test sites showed that site condition (i.e., health) varied between years and river systems. In Prairie Creek, all sites located immediately downstream of the mine in 2008 were in reference condition, i.e., not impaired. By contrast, in 2009 sites 43, 46 and 45 located 0.31, 0.54, and 1.69 km downstream of the mine were possibly impaired. In the Flat River, sites 38 and 2, located 0.01 and 2.45 km downstream of the mine in 2008 were possibly impaired, whereas in 2009 sites 38 and 39 located 0.01 and 1.42 km downstream of Tailings Pond 3 were possibly impaired. Interestingly, the reference model showed that Site 1 located within an area of the floodplain that had historically received mine tailings was unimpaired. This result suggests that historical deposition of tailings to the Flat River floodplain is not accompanied with detectable declines in stream health at this site.

In 2008 and 2009, we evaluated error rates, i.e., differences in site assessments, by collecting duplicate samples from each of eight sites. We controlled for potential differences in sample composition by ensuring that a single CABIN-certified field crewmember collected both the original sample and the site duplicate. Our analyses showed that only 75% (i.e., 6 of 8) of duplicate samples were located in the same assessment band, suggesting moderately high levels of sample comparability.

By contrast, our comparisons showed low levels of concordance in site assessments when samples were taken with the kick net fitted with a standard 400 µm mesh net versus that fitted with a fine 250 µm mesh net. In fact, ordinations showed that the two methods produced concordant site assessments in only 22% of comparisons (i.e., 4 of 18 comparisons). This latter result suggests that reference condition approach model derived for the South Nahanni watershed using the standard 400 µm mesh net may not be an appropriate model to determine site condition when samples are taken with a kick net fitted with 250 µm mesh.

Application of the reference condition approach model

Our data suggest that the reference condition approach is a viable alternative to traditional biological assessment approaches and can be used to: i) assess the potential effects of mining on the health of the Flat River and in Prairie Creek, and ii) describe possible changes in stream health through time, including evaluations of potential effects of climate changes and long-range deposition of airborne contaminants. The existing reference condition approach model is well suited to supporting both: i) broad-scale assessment and reporting needs such as that related to "State of the Park" reporting, and ii) fine-scale assessments, such as that associated with determining impairment downstream of industrial activities.

Future applications and cost estimates

Identifying the number of streams that need to be sampled and the frequency of sampling required to satisfy broad-scale "State of the Park" reporting and finer-scale assessments depends on the specific questions being asked and the labour and cash resources available to complete the work. Satisfying "State of the Park" reporting would require, at a minimum, one large scale sampling event every 4 to 5 years and comprise sampling of 30 to 40 river sites, including rivers whose watersheds include a high percentage of glaciers. Assuming that field crews used mine sites (i.e., Cantung Mine, Prairie Creek) as operating bases, a trained and well organized two-person field crew should be able to collect samples from 30 to 40 sites in 4 to 5 days. Assuming that the return-flight cost to transport the field crew from Fort Simpson to the mine site is $3,000 and costs to access the majority of sites with helicopter is $40,000 (i.e., 4 days @$10,000 per day) travel costs would total $43,000. Costs to identify benthic macroinvertebrates (35 samples @350.00 per sample = $12,250) and analyze water samples for a suite of metals and some fractions of nutrients, including total phosphorus and total nitrogen (35 sample @170 = $5,950) would require an additional $18,200 of project funds. When these two cost categories are combined total project costs, excluding staff time, would be $61,200. When expressed over a 5-year amortization period (i.e., the need to complete a "State of the Park" report every 5 years), annual costs to satisfy "State of the Park" by reporting on stream health reporting would be about $12,240. Because information gained through site assessments using the RCA could also support each of the mines' environmental programs, substantial cost-sharing opportunities exist. Securing external funds (e.g., Cumulative Impact Monitoring Program [Aboriginal Affairs and Northern Development Canada]) would also reduce funding contributions by Parks Canada.

Model Verification and Recalibration

The current reference condition model for the South Nahanni Watershed was constructed using biological data collected in 2008 and 2009. An important consideration in the use of reference condition models is to verify that the structure of biological communities at reference sites is not changing dramatically and systematically through time. Appreciable changes in the structure of biological communities at reference sites would result in appreciable Type I and Type II errors and would compromise assessments of site condition. Confirming that biological communities at reference sites have not shifted appreciably could be conducted in 3 to 5 years (2015 to 2018) as one component efforts to complete a State of the Park report for Nahanni National Park Reserve. This would require re-sampling and assessments of about 10 to 15 reference sites that had been previously sampled in 2008 and 2009. The RCA model would be verified if the majority of these samples were located within the 90% reference ellipse, i.e., are unimpaired, and would indicate that the existing model is still valid and does not require re-calibration.

The reference condition approach model for the South Nahanni Watershed would however require re-calibration under at least three scenarios. First, if: i) biological communities from watersheds that included appreciable (>40% glaciers) amounts were not strongly predicted to an existing community type and ii) Park Managers wanted to report on the biological condition of these glacier-dominated streams. Second, if the majority of streams sampled in 2015 to 2018 were located outside of the 90% reference ellipse, i.e., were possibly impaired, impaired, or highly impaired. Third, if assessments using the existing model derived using data collected in 2008 and 2009 differed from that using a hybrid model comprised of data collected in 2008-2009 combined with data from reference that could be collected in 2015 to 2018. Re-calibration of the RCA model could be completed by Parks Canada staff or by a qualified consultant. A consultant would likely charge $6,000 to $12,000 to re-calibrate the RCA model for the South Nahanni Watershed.

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Contents

List of Figures

List of Tables

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1.0 INTRODUCTION

Maintaining healthy ecosystems amidst increasing levels of industrial development is a central challenge to the management of protected areas globally. In Canada's north, increasing levels of exploration and extraction of metals and oil and gas reserves have raised serious concerns about conservation of protected areas and the extent that they can co-exist with industrial development. Threats to the health of surface waters from industrial activities can be compounded where industrial activities beyond the borders of protected areas result in effluent discharges to streams that subsequently flow into protected areas.

Environmental assessments of the ecological effects of mining activities typically involve comparisons of the biological communities at non-exposed sites with those at sites exposed to mining effluents. In Canada, assessments of the effects of mining operations on receiving waters are required under the Metal Mining Effluent Regulations and typically include comparisons of benthic macroinvertebrate community structure using either upstream-downstream (control-impact) or reference condition approach designs. While both approaches are capable of identifying impairment, upstream-downstream designs typically describe reference conditions at only a few sites, and over short spatial scales. These aspects of upstream-downstream designs can compromise assessments of impairment due to spatial autocorrelations and because the reference condition may reflect a narrow range of natural variability.

Nahanni National Park Reserve located in the North West Territories of Canada protects 30,000 km^2 of the Mackenzie Mountains Natural Region within the South Nahanni Watershed and is one of Canada's most remote national park reserves. The reserve was inscribed as a World Heritage Site by the United Nations Educational, Scientific and Cultural Organization in 1978 reflecting its special cultural and physical significance. The South Nahanni Watershed also includes geologic formations that include rich and commercially viable deposits of tungsten, lead, zinc, silver, and gold. The watershed includes an operational tungsten mine and an advanced exploration lead, silver and zinc mine, which will likely become operational in 2-5 years. Both mining operations discharge effluents to upper reaches of the Flat River and Prairie

Creek that eventually flow into areas within Nahanni National Park Reserve. The extent that mining operations influence the ecological health of streams in the South Nahanni Park Reserve is unknown.

The objectives of the present study were four-fold.

1) Using a data set of 78 reference sites, we developed a reference condition approach model of the structure of benthic macroinvertebrate communities for the South Nahanni Watershed.

2) We used the reference condition approach model as the first step to assess the ecological health of sites located immediately adjacent and downstream of each mine. These data were also used delineate potential zones of influence at each mine site. We compared results from our study with two published studies of Spencer et al. (2008) and Bowman et al. (2010). These studies used an upstream and downstream design and a reference condition approach respectively, to assess potential biological impairment in Prairie Creek and the Flat River.

3) We assessed the ecological condition of: a) a suite of duplicate samples to quantify potential error rates and b) samples collected using a fine mesh net (250 μm mesh) to describe comparability of samples taken with the standard 400 μm mesh net.

4) Lastly, we discuss several operational considerations related to: i) our general conclusion on the application of the reference model for the South Nahanni Watershed, ii) current and future applications of the reference condition approach model, and iii) model verification and re-calibration.

2.0 STUDY AREA

Fieldwork was completed in the South Nahanni Watershed located in south-west portion of the Northwest Territories (Figure 1). The low subarctic climate in the Nahanni Plateau is characterized by cool summers (mean temperature is 9°C) and cold winters (mean temperature is -19.5°C; Environment Canada 1991, Halliwell and Catto 1998). Most areas are less than 1372 m above sea level but mountain ranges reach about 2770 m. The terrain of the study areas is underlain by Palaeozoic carbonates, and is incised by deep and narrow valleys (Halliwell and Catto 1998). Vegetation is sparse at higher elevations but open stands of black spruce with an understory of dwarf birch, Labrador tea, lichen, and moss occur in valleys and at lower elevations (Environment Canada 1991). Due to its large area and variance in geology, altitude and latitude, and vegetation, the South Nahanni Watershed supports a diversity of stream types, including small tributaries to the very large 8^{th} order main stem of the South Nahanni River.

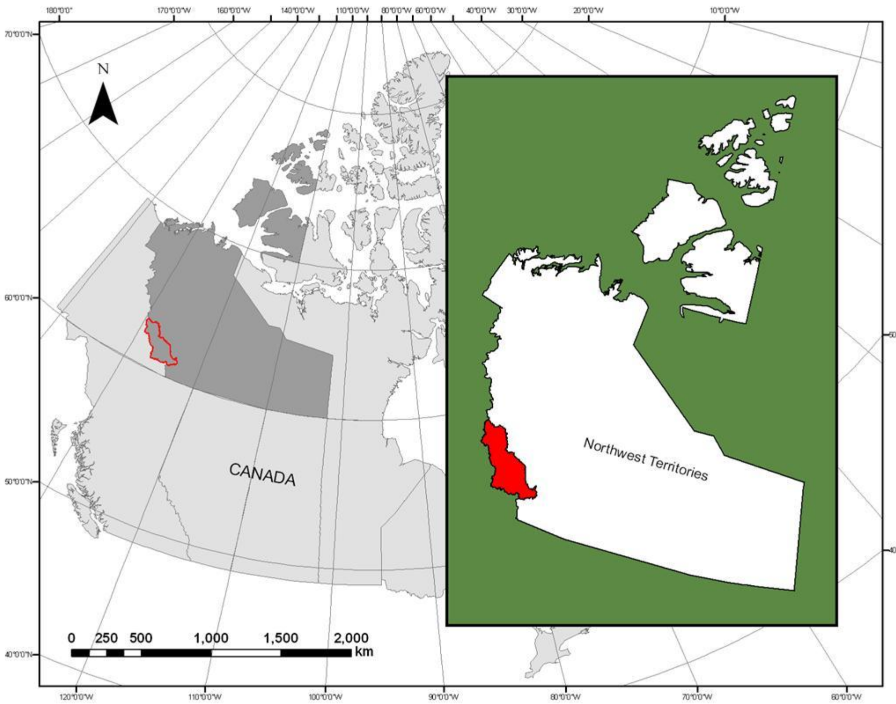

Figure 1. Location of the South Nahanni Watershed (red inset) in the Northwest Territories, Canada.

3.0 METHODS

3.1 Sampling design and selection of reference sites

The primary objective of our study was to develop and apply a reference condition model capable of: i) detecting potential biological impairment of sites on the Flat River and Prairie Creek, and ii) quantifying baseline biological community types of select streams throughout the watershed. When incorporated into a future monitoring program deviations from baseline conditions can be used to detect environmental change, such as that arising from aerial deposition of airborne contaminants, climate change, and new industrial activities.

To achieve these objectives, we selected sample sites by applying a nested, double stratified random sampling design with stream order and percent ice (typically glaciers) in the sub-watershed as the strata. These strata increase the probability that sampling sites used to define reference conditions closely approximate those in the Flat River and Prairie Creek. The first stratification of stream order ensured that sample sites were located on streams of similar size (i.e., discharge) to that immediately adjacent and downstream of the mining activities in the Flat River and in Prairie Creek. This stratum eliminated the selection of sampling sites on small to medium sized 1^{st} to 2^{nd} order creeks and large 7^{th} and 8^{th} order rivers. The second stratum of percent ice in the watershed is a proxy for the percent of glaciers in the sub-watershed. Glacier dominated streams are physically and chemically different from many other stream types (Flory and Milner 1999, Milner et al. 2000) and it is highly likely that invertebrate communities in these streams would not approximate those in the Flat River and in Prairie Creek (Spencer et al. 2008, Bowman et al. 2010). Application of this strata eliminated the selection of sampling sites on highly glacial-dominated sub-watersheds where glaciers accounted for >40% of the watershed area (i.e., highly turbid and cold water streams).

After applying these strata we randomly identified 140 candidate (i.e., potential) sampling sites located throughout the entire South Nahanni Watershed. Aerial reconnaissance showed that the majority of these sites were suitable for sampling. However, because the majority of sample sites were accessed using helicopter, some candidate sites were not sampled as they did not provide safe landing conditions, whereas other candidate sites were not samples as they supported appreciably less flow than that expected based on their our assessments of their size using the Strahler stream order classification system. Finally, other candidate sites were not sampled a wildfires preclude flight access to them. In total, we collected samples from 78 reference sites in 2008 and 2009 (Figure 2, Table 1).

Candidate reference sites were located either upstream of mining activities in Flat River and Prairie Creek watersheds, or in sub-watersheds outside of the Flat River and Prairie Creek watersheds. We ensured that reference sites located upstream of the Prairie Creek and Cantung mines were unimpaired by ground-truthing each site to ensure that there was no evidence of industrial activity combined with queries of geo-spatial data bases that describe human use.

Table 1. Summary of benthic macroinvertebrate samples collected from reference and test sites in the South Nahanni Watershed in 2008 and 2009. [1] Sites sampled in 2009 but not in 2008, [2]Reference sites sampled in both 2008 and 2009, [3] test sites sampled in 2009 were also sampled in 2008.

Year	Sample type		Location and type	Year
2008	Reference		Throughout the South Nahanni Watershed	78
	Test		Adjacent and downstream of the Cantung and Prairie Creek mine sites	20
2009	Reference	1	New reference [1]	7
		2	Repeat reference [2]	18
	Test sites	1	Adjacent and downstream of the Cantung and Prairie Creek mine sites [3]	18
		2	Duplicate samples	8
		3	Small mesh samples	18

3.2 Selection of test sites and test samples

The primary objective of sampling test sites is to determine their ecological condition by determining the extent that the benthic macroinvertebrate communities at test sites approximate those expected under reference conditions. To understand potential effects of mining activities on the health of Prairie Creek and the Flat River, we collected samples from 20 test sites in 2008 and an additional 18 test sites in 2009. These sites were located adjacent, and at increasing distances downstream of the Cantung Mine on the Flat River and the Prairie Creek Mine on Prairie Creek (Figure 2 and 3; Tables 2 and 3).

We also collected: i) duplicate benthic macroinvertebrate samples from eight sites to better understand error rates in site classification using the RCA model, and ii) benthic macroinvertebrate samples collected with a kick net fitted a non-standard fine mesh net (250 µm) to determine if mesh size (i.e., 400 µm versus 250 µm mesh) influenced assessments of site condition. We collected samples using a fine mesh net from 18 sites, and assessed site condition using these samples using standard methods (See section 3.5 Assessments of test sites using the reference condition model). A summary of benthic macroinvertebrate samples collected in 2008 and 2009 are provided in Table 1.

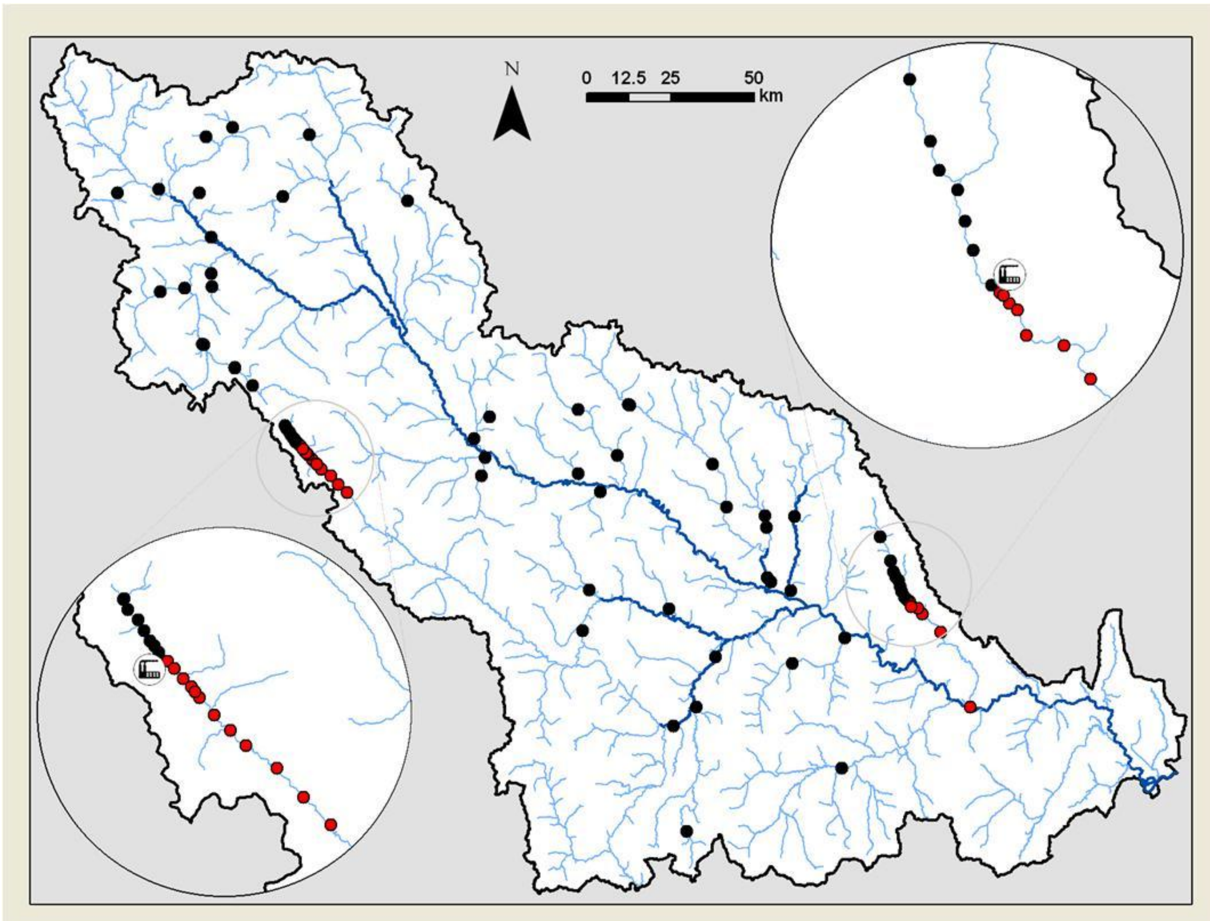

Figure 2. Locations of sampling sites in the South Nahanni Watershed in the Northwest Territories, Canada. The North American Tungsten Cantung Mine and the Canadian Zinc Prairie Creek mine are located in the northwest and southeast regions of the watershed, respectively. Black symbols represent reference sites (i.e., sites not exposed to mine effluent), whereas red symbols represent sites potentially exposed to mining effluents (i.e., located adjacent and downstream of the mining operations).

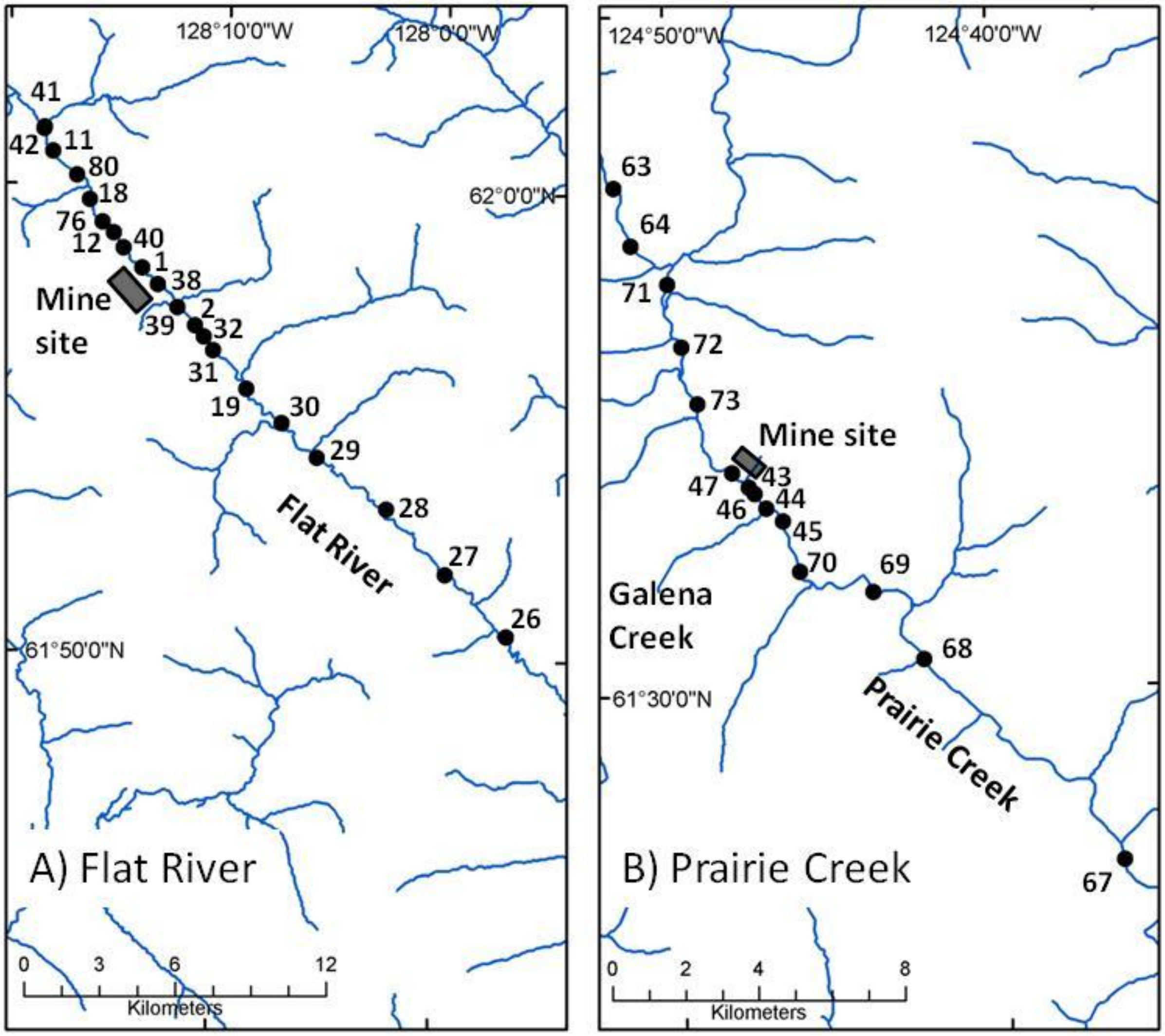

Figure 3 Location of sampling sites located upstream, adjacent and downstream of North American Tungsten Cantung mine on the Flat River (A) and Canadian Zinc's Prairie Creek mine on Prairie Creek (B) in the South Nahanni Watershed, Northwest Territories. The grey filled box represents the approximate location of the mine site.

Table 2. Distances between upstream reference sites and downstream potentially exposed sites to the discharge of mining effluent to Prairie Creek, South Nahanni Watershed, Northwest Territories. Effluent from the mine site is discharged to Harrison Creek that flows into Prairie Creek. Distance zero and the arrow denote the confluence of the Harrison Creek and Prairie Creek. The majority of sites were sampled in both 2008 and 2009.

Upstream reference sites		Downstream exposed sites	
Site number	Distance upstream of the mine effluent discharge to Prairie confluence (km)	Site number	Distance downstream of the mine effluent discharge to Prairie confluence (km)
47	0.33	43	0.31
73	2.90	46	0.54
72	4.68	44	1.08
71	6.72	45	1.69
64	8.72	70	3.17
63	10.52	69	6.03
62	14.29	68	9.22
61	23.05	67	18.04
		66	52.05

Table 3. Locations of sampling sites upstream, adjacent, and downstream of mining operations in the Flat River, South Nahanni Watershed, Northwest Territories. Effluent from the mine is currently deposited into a large tailings pond. A portion of the exfiltrate from the pond enters the Flat River. Site 1 is located upstream of the tailings pond but is considered to be a potentially exposed sites at metal-rich mine tailings had historically been deposited to the floodplain at this site. The majority of sites were sampled in both 2008 and 2009.

Upstream reference sites		Downstream potentially exposed sites	
Site number	Distance upstream of the tailings pond (km)	Site number	Distance downstream of the tailings pond
1	1.08	38	0.01
40	2.30	39	1.42
12	3.11	2	2.45
76	3.74	32	3.10
18	4.89	31	3.83
80	6.22	19	6.12
11	7.69	30	8.64
42	8.82	29	10.98
41	8.88	28	15.32
		27	19.51
		26	23.67
		33	158.86

3.3 Field collections

3.3.1 Environmental variables

We measured a suite of stream and riparian habitat variables to describe the physical characteristics of streams and riparian habitats and for potential use as predictors of benthic macroinvertebrate, community groups (Table 4). Detailed descriptions of sampling methods are available at: **http://www.ec.gc.ca/rcba-cabin/**. The full suite of potential environmental predictors is provided in Section 3.5.2.

Table 4. Summary of stream and riparian habitat descriptors recorded from all study sites in 2008 and 2009. Values in brackets are units of measurement.

Variable	Description
Bank full width	Measured with a tape measure (m)
Bank full depth	Measured with a tape measure (m)
Bank full-wetted height	Vertical distance from the bank full height to the water surface measured with a tape measure (cm)
Wetted width	Mean width of the stream water surface based on 3 to 5 measures using a tape measure (m)
Mean water velocity	Mean of five to 10 measures of water velocity taken with Marsh McBirney flowmate meter (m/s)
Mean water depth	Mean of five to 10 measures of water depth taken with Marsh McBirney flowmate meter (m)
Maximum water velocity in the reach	Measured with a Marsh McBirney flowmate meter (m/s)
Presence of stream habitats types cascade	Visual assessment of the presence of stream habitat types: i) riffle, ii) rapid, iii) straight run, iv) and pool/back eddy
Percent composition of substratum size fractions in the river bed in the study reach	Visual estimation of size fractions of: i) organic matter cover, ii) coarse sand, silt and clay, iii) very coarse sand, iv) gravel, v) pebble, vi) cobble, vii) boulder, and viii) bedrock
Substratum size composition	Measurements (cm) of 100 pieces of riverbed materials (cm) used to describe percent composition and median and geometric mean descriptors of substratum size.
Percent macrophyte coverage	Visual estimation of the percent cover of macrophytes within the stream channel (%)

Table 4. - continued

Variable	Measurement method and notes
Percent canopy coverage	Visual estimation of percent canopy coverage (%) into one of five categories of coverage
Dominant riparian vegetation	Visual assessment of the dominant riparian vegetation
Presence of vegetation types in riparian zone (i.e., grass, confer vegetation types	Visual assessment of the presence of vegetation types in the riparian zone: i) ferns/grasses, ii) shrubs, iii) deciduous trees, iii) coniferous trees)
Dominant riparian vegetation	Visual assessment of the dominant vegetation type in the riparian zone (i.e., i) ferns/grasses, ii) shrubs, iii) deciduous trees, iii) coniferous trees)
Periphyton coverage	Visual assessment of periphyton coverage based on five abundance categories (low [1] to high [5])
Presence of surrounding land use	Visual assessment of the presence of: i) forest, ii) field/pasture, iii) agriculture, iv) residential/urban, v) logging, vi) mining commercial/industrial, and vii) other
Dominant surrounding land use	Visual assessment of the dominant land use surrounding the study reach of: i) forest, ii) field/pasture, iii) agriculture, iv) residential/urban, v) logging, vi) mining commercial/industrial, or vii) other

3.3.2 Water physico-chemistry

We characterized the physical and chemical characteristics of water at each site by measuring a broad suite of chemical descriptors, measured either with water quality meters at the site (in-situ measures), or through laboratory analyses of water samples collected at each site (Table 5). Conductivity, dissolved oxygen and pH were measured at the site using a YSI model 650 meter, while turbidity was measured using a LaMotte model 2020e turbidity meter. Water samples for chemical analyses from the midstream of flow at approximately 30-cm depth. Samples were stored in the dark in a cooler during transport to a laboratory for processing.

Table 5. Summary of physical and chemical descriptors of water measured at sample sites in 2008 and 2009.   NTU = Nephelometric turbidity units.

In situ measures using meters	Laboratory analyses - Metals	
pH	Aluminum	Magnesium
Dissolved oxygen	Antimony	Nickel
Percent dissolved oxygen saturation	Arsenic	Phosphorus
Turbidity (NTU)	Barium	Potassium
Specific conductance	Beryllium	Rubidium
Temperature (^{o}C)	Bismuth	Selenium
	Boron	Silver
	Cadmium	Sodium
Laboratory analyses - Nutrients	Calcium	Strontium
	Cesium	Thallium
Dissolved organic carbon	Chromium	Tin
Total inorganic carbon	Cobalt	Titanium
Total phosphorus	Copper	Tungsten
Total nitrogen	Iron	Uranium
Nitrate+nitrite	Lead	Vanadium
Ammonia+ammonium	Lithium	Zinc

In the laboratory, water samples were sub-sampled and subsequently analyzed for nutrients and metals. Samples for dissolved inorganic carbon (DIC) and dissolved organic carbon (DOC) determination were analyzed using an UV-persulfate TOC analyzer. Samples for ammonia and ammonium (NH_3+NH_4), nitrite and nitrate (NO_2+NO_3), and total nitrogen (TN) and total phosphorus (TP) measurements were analyzed using an automated continuous segmented flow analyzer. Concentrations of metals were analyzed using inductively coupled plasma mass spectrometry. All samples were analyzed for nutrients and metals at Environment Canada's National Laboratory for Environmental Testing, Burlington, Ontario, following standard methods (Environment Canada 1994).

3.3.3 Benthic macroinvertebrates

We collected benthic macroinvertebrates from riffle habitats using the rapid bioassssment travelling kick and sweep net method developed by the Canadian Aquatic Biomonitoring Network (CABIN) (**http://www.ec.gc.ca/rcba-cabin/).** Briefly, we collected benthic macroinvertebrates by walking upstream through the study reach for three minutes and during this period we disturbed the top 5 to 10 cm of the riverbed using a foot-shuffle motion. Benthic macroinvertebrates dislodged from the riverbed were captured in a triangular metal frame net fitted a 400 µm that is placed on the river bottom. The mesh net collects benthic macroinvertebrates and organic (e.g., fine and coarse particulate organic matter) and inorganic materials dominated by gravels and sand and silt. We used a swirl method to separate macroinvertebrates from the majority of organic and inorganic materials. The swirled materials that consist predominantly of benthic macroinvertebrates and silt and fine organic matter were placed into 300 or 500 mL plastic jars and preserved in 10% formalin or 70% ethanol.

In the laboratory, samples were washed, sub-sampled, sorted and identified by Cordillera Consulting (Summerland, British Columbia) following protocols described by CABIN (**http://www.ec.gc.ca/rcba-cabin/).** Briefly, samples were washed by placing them into a bucket and swirled to remove the majority of fine sediments. Large pieces of organic matter were removed by hand, and washed to remove macroinvertebrates before being discarded. The remaining materials in the bucket were placed into a Marchant Box (width x length x height = 27 x 27 x 15 cm) comprised of 100 individual cells. Sub-sampling of randomly selected cells from the box typically involved removal and identification of the contents of 10 to 30 cells to ensure that at least 300 individuals are identified. Once initiated, all individuals within an individual cell are identified to allow for calculation of total abundance from the sub-sampled portion. In rare cases where total numbers of macroinvertebrates collected are low, individuals within the entire sample were identified. Detailed descriptions of the field sampling methods and laboratory sub-sampling methods are available at **http://www.ec.gc.ca/rcba-cabin/**

We also collected two additional sets of benthic macroinvertebrate samples to: i) quantify error from duplicate samples, and ii) assess the comparability of samples taken with a less commonly used fine mesh net (250 µm). For each comparison, we controlled for potential differences in sample composition by ensuring that a single CABIN-certified field crew member collected the original sample, the site duplicate, and the sample with the fine mesh net (250 µm).

Duplicate samples were taken from each of eight sites in 2008 and 2009 using standard methods described above. We collected two samples from each site, and then randomly assigned one sample to be the routine sample with the second being assigned as the site duplicate. We collected samples from sites where riffles areas were extensive to ensure that the collection of the first sample did not result in depletion or contamination of the second sample. Site duplicates were taken from sites located upstream and downstream of mining activities.

We collected benthic macroinvertebrate samples from 18 sites in 2009 using a kick net fitted with a non-standard 250 µm mesh in addition to the routine sample collected using a kick net fitted with 400 µm mesh. As for the collection of site duplicates, we collected the routine (400 µm mesh net) and the fine mesh (250 µm mesh) samples from sites where riffles areas were extensive to ensure that the collection of the first sample did not result in depletion or contamination of the second sample. Samples were collected using the same sampling methods described above.

We quantified comparability (i.e., concordance in site assessments) of the site duplicate and samples collected with the fine mesh net by quantifying the percentage of times that eight duplicate samples and the 18 fine mesh samples were located in the same ordination band as routine samples taken from the same sites.

3.4 Longitudinal patterns in water physico-chemistry

We quantified longitudinal patterns in physical and chemical characteristics of water at sites located immediately upstream, adjacent to, and downstream of each of the mines on Prairie Creek and in the Flat River in 2008 and 2009. Establishing the presence and strength of longitudinal patterns in water physico-chemistry provides insights on the extent that mining activities combined with local mineralization may contribute to, and explain differences in benthic macroinvertebrates community structure at each of the two mine sites. Thus, quantified longitudinal patterns in water physico-chemistry by collecting water samples in 2008 and 2009 from seven sites located upstream (0.33 to 14.3 km upstream) and downstream (0.31 to 9.2 km downstream) of the confluence of Harrison Creek and Prairie Creek. In the Flat River we sampled six sites located between 1.1 to 8.8 km upstream of the tailings pond, and six sites located between 0.01 to 8.6 km downstream of the tailings pond (Tables 1 and 2).

3.5 Development of the reference condition model

3.5.1 Model development

The reference condition approach (RCA) measures a test site's condition by assigning it to a biological community group based on environmental predictor variables. Once assigned the model compares its biological community structure with the structure expected with those from the reference group to which it has been assigned by the model (Reynoldson and others 1997; Bailey and others 2004; Bowman and Somers 2005, Scrimgeour et al. 2008). The development of the reference condition model typically involves three sets of statistical analyses of clustering, discriminant function, and ordination. Application of the reference model to assess site condition requires application of the discriminant function model and ordination to an appropriate reference group.

The RCA begins with the identification of reference conditions which represent sites that are least or minimally impaired. There is no accepted method to identify least impacted reference sites. We identified least impacted reference sites as those located upstream of mining activities or beyond historical mining activities, and at distances greater than 100 m from permanent roads. For the purpose of our study, reference sites represent least and minimally impacted sites.

We identified benthic macroinvertebrate assemblages (i.e., groups) using clustering (Bray-Curtis association measure) and the unweighted pair-group method using arithmetic averages (UPGMA) (McCune and Mefford 1999) of invertebrate taxa counts. We identified the number of assemblages by examining similarity between sites combined with an a priori criterion to ensure that assemblages contained meaningful numbers of sites (15 sites per assemblage). Analyses were completed using Primer (Clarke 1993, Clarke and Gorley 2006). Discriminant function (Tabachnik and Fidell 2001) analyses were applied to reference sites to identify variables that best discriminated (i.e., separated) sites into assemblages identified by the classification of the fish community data. Analyses were completed using Primer.

3.5.2 Data Assembly and Preparation

Data describing benthic macroinvertebrate community structure and instream and riparian habitats collected in 2008 and 2009 were uploaded to the CABIN website in the Parks Canada – South Nahanni Study. This included landscape-scale GIS data (basin morphology, climate, land cover, bedrock geology) which were derived from queries of geo-spatial databases (digital elevation model [www.geobase.ca], stream network [www.geobase.ca], climate [http://sis.agr.gc.ca/cansis/nsdb/ecostrat/district/climate.html]
bedrock geology [http://www.lib/uwo.ca/madgic/geospatial/can_geo1860_1997_data.htm].

For the RCA model-building exercise, the macroinvertebrate (at family level) and habitat data were downloaded from CABIN for sites identified as being in reference condition. Fifty-three of these potential reference sites were sampled in 2008, while 25 were sampled in 2009. Bailey et al. (2004) suggest a minimum of 25 reference sites be used for a pilot project and Reynoldson and Wright (2000) more conservatively suggest 60-70 sites be used for a preliminary RCA model. It is impossible to define an absolute minimum number of sites as it is dependent on the variability of the target system and of course on the resources available. However, based on previous studies (Reynoldson et al. 2000, 2001) and the above recommendations we consider the 78 reference sites as sufficient for initial model development.

We examined the initial habitat data set and the full set of candidate predictor variables and subsequently identified 37 candidate predictors of benthic macroinvertebrate groups. All candidate variables were considered to be unaffected by the stressors of interest (i.e., mining activities and road networks) and are shown in Table 6. Water chemistry variables that may potentially be influenced by mining activities are not suitable candidate predictor variables and were excluded from the list of candidate variables.

Table 6. Candidate environmental variables used to predict membership of benthic macroinvertebrate assemblages from the South Nahanni Watershed, Northwest Territories. Numbers in brackets reflect the number of candidate variables within each variable type.

Variable type	Description and units
Drainage area (seven variables)	Latitude (Hours, Minutes, Seconds)
	Longitude (Hours Minutes Second
	Altitude (m)
	Stream order (Strahler)
	Drainage Area (km^2)
	Perimeter – upstream drainage (km)
	Stream density (m stream/km^2rainage area)
Land cover and bedrock geology (four variables)	Land cover – Coniferous forest (%)
	Land cover – Ice (%)
	Bedrock Geology – Intrusive (%)
	Bedrock Geology – Sedimentary (%)
Climate (15 variables)	Total precipitation in January (mm)
	Snowfall in January (mm)
	Rainfall in January (mm)
	Total precipitation in June (mm)
	Snowfall in June (mm)
	Rainfall in June (mm)
	Total annual precipitation (mm)
	Total annual snowfall (mm)
	Total annual rainfall (mm)
	Mean daily air temperature in January (°C)
	Minimum daily air temperature in January (°C)
	Maximum daily air temperature in January (°C)
	Mean daily air temperature in June (°C)
	Minimum daily air temperature in June (°C)
	Maximum daily air temperature in June (°C)
Channel and site (11 variables)	Mean channel depth (cm)
	Mean bank full-wetted depth (cm)
	Mean water velocity (m/s)
	Slope (m/m)
	Sinuosity (m of steam within a 2 km linear distance of stream)
	Channel bank full width (m)
	Channel wetted width (m)
	Thickness of periphyton layer (Categorical; 1-5)
	Dominant substrate size of river bed materials (Categorical; 1-5)
	Second dominant substrate size of river bed materials (Categorical; 1-5)
	Substrate embeddedness (Categorical; 1-5)

The invertebrate data set downloaded from CABIN for the 78 potential reference sites contained 46 families. Unverified taxa from the CABIN download were merged into the data set, as we are confident in the taxonomy at the Family level. Hydridae were removed as these are not normally included as members of the benthic macroinvertebrate community. After several iterations, five Families were used to identify reference community assemblages.

3.6 Assessments of sites using the reference condition model

Assessments of test sites were completed by performing ordinations of reference sites that belonged to an individual assemblage, along with appropriate test sites. We completed these analyses using the analytical tools module within CABIN (**http://cabin.cciw.ca/BEAST/Tools/Project.aspx**) . Ordinations, typically in three dimensions, were used to develop bivariate plots and associated probability (i.e., similarity) ellipses of site scores for reference and test sites. We used three probability ellipses to identify four three bands of similarity between a test site and reference sites: band 1 - ≤90% (equivalent to reference sites [unimpaired]), band 2 - 90 to 99% (possibly different from reference sites [possibly impaired]) and band 3 - 99% to 99.9% (impaired), band 4 - >99.9% highly (highly impaired) (Reynoldson et al. 2001). We also used the analytical tools module within CABIN and the four assessment bands to evaluate comparability of duplicates samples and samples collected using the fine mesh net.

4.0 RESULTS AND INTERPRETATION

4.1 Longitudiunal patterns in water physico-chemistry and metals in Prairie Creek

4.1.1. General physico-chemistry

We evaluated longitudinal patterns in water physico-chemistry along a 23.5 km length of Prairie Creek extending from a site 62 located 14.3 km upstream of the confluence of Harrison Creek with Prairie Creek) to Site 68 located 9.2 km downstream of the mine.

Longitudinal patterns in water physico-chemistry varied appreciably depending on the variable of interest and whether samples were collected in 2008 or 2009 (Figure 4). Overall, concentrations of total phosphorus, nitrite+nitrate nitrogen, total nitrogen and benthic Chlorophyll *a* displayed only minor to modest increases along the 23.5 km reach (Figure 4). In 2008 and 2009 mean concentrations of total phosphorus at sites downstream of the mine (2008: mean = 0.0021 mg/L, range = 0.0019 to 0.0024 mg/L; 2009: mean = 0.0023, range = 0.001 to-0.0038 mg/L) were about 29% and 35% higher relative to sites upstream of the mine (2008: mean = 0.0015; range = 0.001 to 0.0022 mg/L, 2009: mean 0.0015, range = 0.005 to 0.0027 mg/L), respectively. A spike in total phosphorus concentrations immediately downstream of the Harrison Creek-Prairie Creek confluence was observed in 2009, but not in 2008. Concentrations of total phosphorus at sites 43, 44 and 46 located between 0.31 to 1.08 km downstream of the mine discharge exceeded or equalled (range of concentrations: 0.0027 to 0.0038 mg/L) that at Site 47 located 0.33 km upstream of the mine discharge to Prairie Creek (total phosphorus = 0.0027 mg/L). However, the concentration of total phosphorus at Site 45 located 1.69 km downstream of the Harrison Creek-Prairie Creek confluence (0.0021 mg/L), was lower than that at Site 47 located immediately upstream of the mine (0.0027 mg/L). These results suggest that the zone of potential phosphorus enrichment in Prairie Creek associated with the mine site is temporarily variable, and extends between 0.31 to 1.69 km (Figure 4). Spencer et al. (2008) reported that concentrations of total phosphorus did not exceed analytical detectable concentrations at the upstream reference site or the near- and far field sites downstream of the mine.

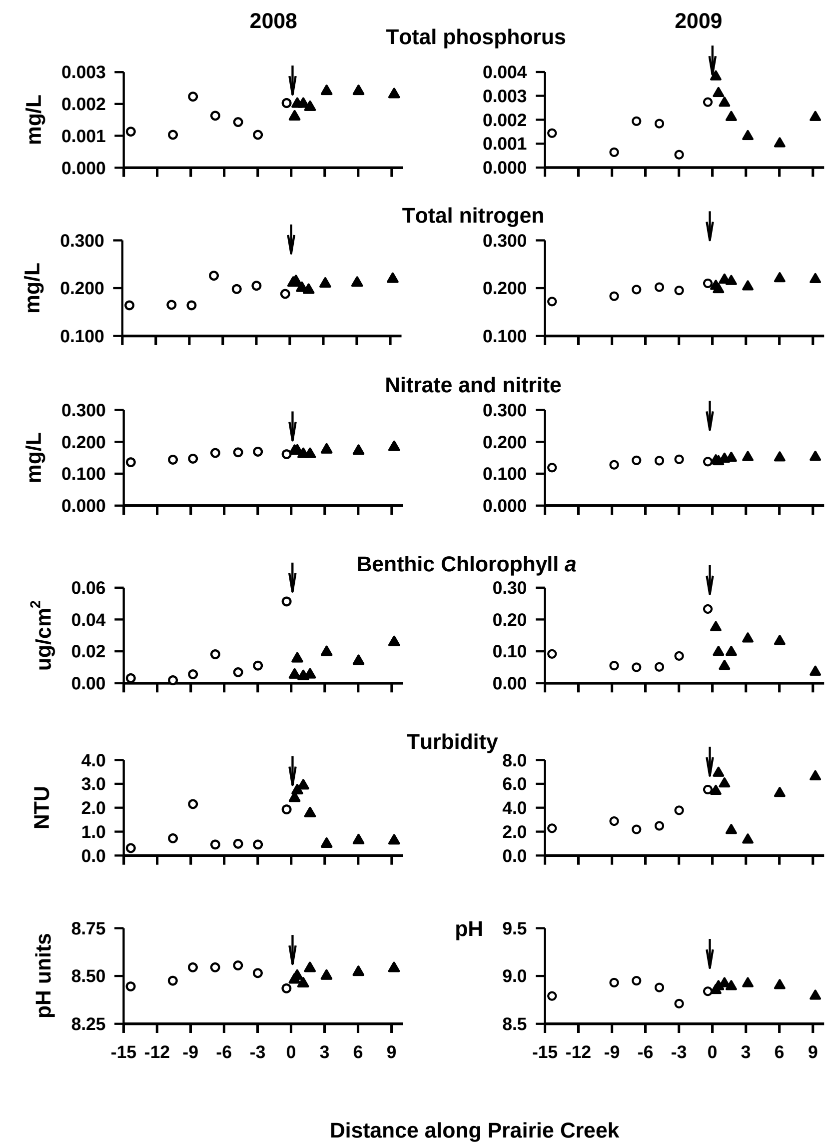

Figure 4. Longitudinal patterns in the physico-chemistry of water from sites upstream and downstream of the Canadian Zinc mine in the Prairie Creek Watershed, Northwest Territories. Effluent from the mine is discharged to Harrison Creek that enters Prairie Creek immediately adjacent to the mine. Sites denoted with negative distance are located upstream of the discharge of mine effluent to Prairie Creek (i.e., upstream of the confluence of Harrison Creek and Prairie Creek) whereas distances with positive values are located downstream of the mining discharge. Distance zero and the vertical arrow represent the confluence of Harrison Creek and Prairie Creek.

Concentrations of total nitrogen, and nitrite and nitrate nitrogen did not increase appreciably along the 23.5 km length of Prairie Creek in either 2008 or 2009, and did not display marked increases immediately downstream of the mine (Figure 4). In fact, the mean concentration of total nitrogen at sites downstream of mine in 2008 (mean = 0.208 mg/L) was only about 11% higher than that relative to sites located upstream of the mine (mean = 0.185 mg/L) and only 10% higher than at sites located downstream of the mine (mean = 0.210 mg/L) relative to sites upstream of the mine in 2009 (mean = 0.191 mg/L).

Concentrations of nitrite+nitrate nitrogen were also only moderately higher at sites downstream of the mine compared to that upstream. Concentrations of nitrate+nitrite nitrogen were about 11% an 10% higher at sites located downstream of the mine (2008: mean = 0.170 mg/L, 2009: mean = 0.147 mg/L) compared to upstream sites (2008: mean = 0.152 mg/L, 2009: 0.133 mg/L) (Figure 4). Interestingly, concentrations of total nitrogen and nitrite+nitrate nitrogen increased along the 14 km length of Prairie Creek upstream of the mine site (i.e., between sites 47 and 62) suggesting that modest increases in concentrations of nitrogen downstream of the mine site may be due to two factors comprising: i) a natural longitudinal increase in nitrogen concentrations from upstream to downstream sites , and ii) enrichment of nitrogen from the mine site. Spencer et al. (2008) also reported only modest differences in nitrate (NO_3) between the upstream reference site (0.09 mg/L) relative to the near-field site located 0.1 km downstream of the confluence of Harrison Creek and Prairie Creek (NO_3 = 0.10 mg/L) and the far-field site located about 2.0 km downstream of near-field site (NO_3 = 0.08 mg/L) (Spencer et al. 2008).

Algal biomass, and likely overall levels of primary production in Prairie Creek is low reflecting low availability of both phosphorus and nitrogen (Figure 4). Mean concentrations of benthic Chlorophyll *a* on upper stone surfaces along the 23.5 km length of Prairie Creek ranged from 0.001 to 0.05 µg/cm^2 (Figure 4). Mean Chl *a* concentrations at sites located downstream of the mine in 2008 (mean = 0.012 µg/cm^2) were similar to those at sites upstream of the mine site (mean = 0.013 µg/cm^2). Differences in benthic Chl *a* between upstream and downstream sites were also minimal in 2009 (mean of upstream sites = 0.093 µg/cm^2, mean of downstream sites = 0.104 µg/cm^2). In both years, highest levels of benthic Chl a were recorded at Site 47 located 0.33 km upstream of the confluence of Harrison and Prairie creeks (Figure 4). Spencer et al.

(2008) did not detect statistically differences in benthic Chlorophyll *a* among the upstream reference site, and the near and far-field sites located downstream of the mine.

Water clarity in Prairie Creek is typically very high and water turbidity during periods of low precipitation and stable flow typically do not exceed 10 nephelometric turbidity units (NTU) (Scrimgeour unpubl. data). Water turbidity along the 23.5 km length of Prairie Creek ranged from 0.4 to 2.9 NTU in 2008 and 1.3 to 6.9 NTU in 2009 (Figure 4). In terms of biological significance, water turbidity did not vary appreciably between sites located downstream (2008 mean = 1.7 NTU, 2009 mean = 4.8 NTU) versus that at sites upstream of the mine site (2008 mean = 0.9 NTU, 2009 mean = 3.1 NTU). Spencer et al. (2008) also reported exceedingly low concentrations of total suspended solids (< 3 mg/L) and that concentrations did not exceed level analytical detection at the upstream reference site, and the near-field and far-field sites located downstream of the mine.

Water pH in Prairie Creek is slightly alkaline reflecting the presence of limestone and dolostone deposits in the watershed (Halliwell and Catto 2003). Mean water pH did not vary appreciably between sites located downstream versus that at site located upstream of the mine site in 2008 (mean pH upstream = 8.50, mean pH downstream = 8.51) and in 2009 (mean pH upstream = 8.84, mean pH downstream = 8.88).

4.1.2 Metals

Our preliminary analyses showed that concentrations of the majority of metals from the Site 73 located 2.9 km upstream of the Prairie Creek mine greatly exceeded that from all other sites, including sites located immediately downstream of the metal rich discharge to Prairie Creek. Although high levels of metals could reflect high, local-scale mineralization, it could also reflect contamination of the sample. The absence of a sample duplicate makes it impossible to determine the cause of these high values, which is some instances reflected 10 to 50- fold differences among sites. We question the validity of concentrations of metals from Site 73 measured in 2009, and we excluded it from analyses.

In 2008, the mean concentration of aluminum at downstream sites (mean = 7.01 µg/L, range = 3.0 to 15.6 µg/L) was about 60% higher than that at sites upstream of the mine (mean = 4.37 µg/L, range = 3.7 to 6.1 µg/L). Concentrations of Al at sites downstream of the mine in 2009 (mean = 12.5 µg/L, range = 7.9 to 18.1 µg/L) also exceeded that at upstream sites (mean = 8.7 µg/L, range = 3.9 to 18.2 µg/L). Concentrations of aluminum at upstream sites in 2009, but not 2008, increased from Site 62 (4.0 µg/L) located 14.3 km upstream of the mine to Site 47 (6.1 µg/L) located 0.33 km upstream of the mine (Figure 5). This suggests that elevated concentrations of Al downstream of the mine may be due to: i) a natural longitudinal increase in concentrations of Al from upstream to downstream sites, and ii) enrichment of Al from the mine site. Spencer et al. (2008) reported a 50% difference in the concentration of aluminum at the upstream reference site (7.9 µg/L) relative to near-field site (12.6 µg/L), but that levels at the far-field site (7.3 µg/L) approximated that at the upstream site.

Mean concentrations of cadmium (Cd) at sites downstream of the mine in 2008 (mean = 0.044 µg/L, range = 0.029 to 0.084 µg/L) and 2009 (mean = 0.035 µg/L, range = 0.028 to 0.046 µg/L) were about 60% and 35% higher compared to sites located upstream of the mine (2008: mean = 0.029 µg/L, range = 0.023 to 0.034 µg/L, 2009: mean = 0.026 µg/L, range = 0.023 to 0.031 µg/L), respectively. In 2009, concentrations of Cd located immediately downstream of the mine discharge (i.e., at sites 43 and 46 located within 0.5 km downstream) to Prairie Creek were about 3-fold higher (means: Site 43 = 0.072, Site 46 0.084 µg/L) than the mean concentration at upstream sites (mean = 0.026 µg/L). By contrast, concentrations of Cd 1.08 km downstream of the mine at site (i.e., Site 44, concentration = 0.029 µg/L) was more similar to the overall mean concentrations at upstream sites (overall mean = 0.026 µg/L) (Figure 5). Spencer et al. (2008) reported only minor differences in the concentrations of cadmium at the upstream reference site (0.05 µg/L) relative to near-field (0.06 µg/L) and far-field sites (0.05 µg/L).

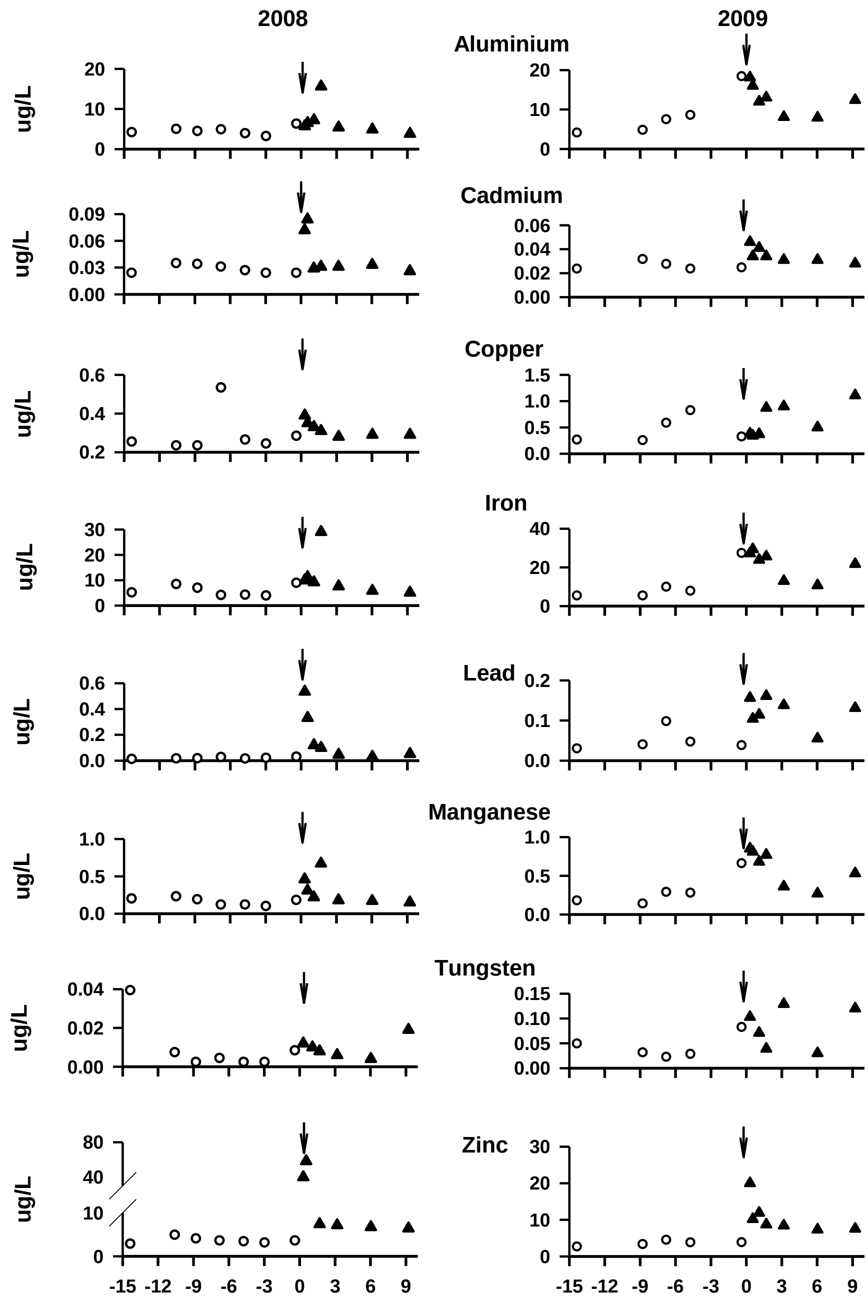

Figure 5. Longitudinal patterns in concentrations of select metals in water from sites located immediately upstream and downstream of the Canadian Zinc mine in the Prairie Creek Watershed, Northwest Territories. Effluent from the mine is discharged to Harrison Creek that enters Prairie Creek immediately adjacent to the mine. Sites denoted with negative distances (km) are located upstream of the discharge of mine effluent to Prairie Creek (i.e., upstream of the confluence of Harrison Creek and Prairie Creek) whereas distances with positive values are located downstream of the mining discharge. Distance zero represents the confluence of Harrison Creek and Prairie Creek.

Mean concentrations of copper (Cu) were appreciably higher at sites downstream of the mine in 2009 but not in 2008 (Figure 5). In 2008, mean concentrations of Cu at sites downstream of the mine (mean = 0.32 µg/L, range = 0.28 to 0.39 µg/L) were about 10% higher relative to sites located upstream of the mine (2008: mean = 0.29 µg/L, range = 0.23 to 0.53 µg/L). Concentrations of Cu downstream of the mine in 2009 (mean = 0.64 µg/L, range = 0.34 to 1.11 µg/L) were about 45% higher relative to sites located upstream of the mine (2008: mean = 0.44 µg/L, range = 0.24 to 0.81 µg/L) (Figure 5). Spencer et al. (2008) reported a 2.5 and 1.6 fold differences in the concentration of copper at the upstream reference site (0.3 µg/L) relative to near-field (0.8 µg/L) and the far-field sites (0.5 µg/L).

Mean concentrations of iron, lead, manganese, tungsten, and zinc were also appreciably higher at sites downstream of the mine compared to that upstream in both 2008 and 2009 (Iron: 2008 upstream = 5.6 µg/L = downstream = 11.1 µg/L, 2009: upstream = 10.8 µg/L = downstream = 21.6 µg/L; lead: 2008 upstream = 0.013 µg/L, downstream = 0.17 µg/L, 2009: upstream = 0.05 µg/L, downstream = 0.13 µg/L; manganese: 2008 upstream = 0.15 µg/L, downstream = 0.31 µg/L, 2009: upstream = 0.30 µg/L, downstream = 0.61 µg/L; tungsten: 2008 upstream = 0.009 µg/L, downstream = 0.018 µg/L, 2009: upstream = 0.04 µg/L, downstream = 0.11 µg/L; zinc: 2008 upstream = 3.5 µg/L, downstream = 19.5 µg/L, 2009: upstream = 3.4 µg/L, downstream = 1.6 µg/L). Mean concentrations of iron, lead, and manganese, tungsten and zinc were typically about 2 to 3-fold higher at sites downstream of the mine compared to sites located upstream of the mine (Figure 5). Increased concentrations of metals downstream of the mines were often accompanied with detectable spikes in concentrations at sites located immediately downstream of the confluence of Harrison Creek and Prairie Creek (Figure 5).

Spencer et al. (2008) reported marked differences in concentrations of zinc, but not all iron, lead or manganese, between the upstream reference site, and the near and far-field sites. In fact, they reported a 400% increase in zinc concentrations from the upstream reference site (2.4 µg/L) relative to the near field site (12.3 µg/L) located 0.1 km downstream of the confluence of Harrison Creek and Prairie Creek, and a 280% increase between the upstream reference site (2.4 µg/L) and the far-field downstream site (6.8 µg/L). By contrast, Spencer et al. (2008) reported equivalent or relatively minor differences in concentrations of iron, lead and manganese between the upstream reference site (REF), and the near (NF), and far-field (FF) downstream sites in Prairie Creek (iron: REF = 89 µg/L, NF = 89 µg/L, FF = 91 µg/L; lead: REF = 0.1 µg/L, NF = 0.2 µg/L, FF = 0.1 µg/L; manganese: REF = 0.2 µg/L, NF = 0.4 µg/L, FF = 0.3 µg/L) Spencer et al. (2008).

4.2 Longitudinal patterns in water physico-chemistry and metals in the Flat River

4.2.1 General physico-chemistry

We evaluated longitudinal patterns in water physico-chemistry along a 17.4 km length of the Flat River extending from Site 42 located 8.8 km upstream of the tailings pond to Site 30 located 8.6 km downstream of the tailings pond. Longitudinal patterns in water physico-chemistry varied appreciably depending on the variable of interest and whether samples were collected in 2008 or 2009 (Figure 6). Overall, concentrations of total phosphorus, nitrite+nitrate nitrogen, total nitrogen and benthic Chlorophyll *a* displayed only minor to modest increases along the 23.5 km reach (Figure 6).

Mean concentration of total phopshorus (TP) did not vary appreciably between sites located downtream (2008: mean = 0.0034 mg/L, range = 0.0027 to 0.0039 mg/L, 2009: 0.0113 mg/L, range = 0.0091 to 0.0138 mg/L) relative to those upstream of the tailings pond (2008: mean = 0.0033 mg/L, range = 0.0024 to 0.004 mg/L, 2009: mean = 0.0120 mg/L, range = 0.0099 to 0.0161 mg/L) (Figure 6).

Mean concentrations of total nitrogen (TN) were about 30% higher at sites located downtream (2008: mean = 0.080, range = 0.061 to 0.158 mg/L, 2009: 0.077, range = 0.066 to 0.091 mg/L) relative to those upstream of the tailings pond (2008: mean = 0.066, range = 0.049 to 0.084 mg/L, 2009: mean = 0.062 mg/L, range = 0.051 to 0.072 mg/L) (Figure 6). Total nitrogen concentration at Site 38 (2008 = 0.158 mg/L, 2009 = 0.091) located immediately downstream of the tailings pond was about was about was 6 times 0.3 times higher than that the overall average of upstream sites in 2008 and 2009, respectively (Figure 6).

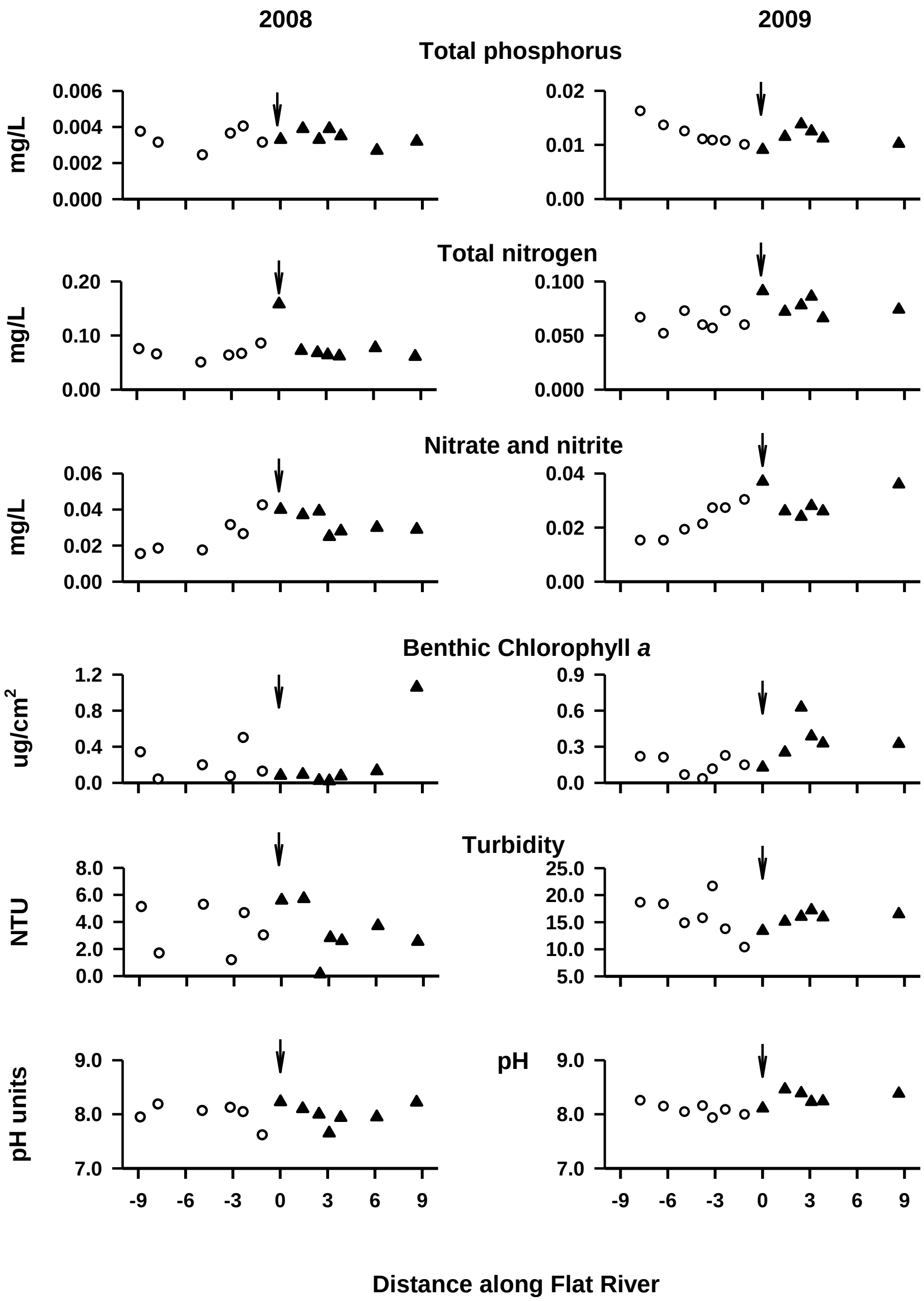

Figure 6. Longitudinal patterns in the physico-chemistry of water from sites located immediately upstream and downstream of the Cantung Mine in the Flat River Watershed, Northwest Territories. Effluent from the mine is discharged to an exfiltration tailings pond. A portion of this exfiltrate enters the Flat River. Distances (km) denoted with negative values are located upstream of the tailings pond whereas distances with positive values are located downstream of the tailings pond. Distance zero represents the location of Site 38 which is situated immediately downstream of the tailings pond.

Mean concentration of nitrate+nitrite were about 20% higher at sites located downstream (2008: mean = 0.032, range = 0.025 to 0.04 mg/L, 2009: 0.030, range = 0.024 to 0.037 mg/L) relative to those located upstream of the tailings pond (2008: mean = 0.025, range = 0.015 to 0.04 mg/L, 2009: mean = 0.022 mg/L, range = 0.015 to 0.03 mg/L) (Figure 6). Concentrations of nitrate+nitrite showed a longitudinal increase with levels increasing about 3 (2008) and 2-fold (2009) from the most upstream reference site (Site 42: 2008 = 0.015 mg/L, 2009 = 0.029 mg/L) to the most downstream reference site (Site 1: 2008 = 0.042 mg/L, 2009 0.04 mg/L). By contrast, concentrations of nitrate+nitrite either declined (2008) or were relatively unaffected (2009) with by downstream of the mines' tailings pond. This suggests that instream processes downstream of the mine may either result in adsorption or utilization of nitrate+nitrite (Figure 6). Spencer et al. (2008) also reported only moderate differences in nitrate (NO_3) between the upstream reference site (NO_3 = 0.02 mg/L) relative to the near-field site located 0.4 km downstream of the tailings pond (NO_3 = 0.03 mg/L) and the far-field site located about 2.0 km downstream of near-field site (NO_3 = 0.03 mg/L).

In 2008, mean benthic Chlorophyll *a* did not differ appreciably between sites located upstream of the tailing ponds (upstream sites = 0.20 µg/cm^2, range = 0.03 to 0.49 µg/cm^2,) relative to sites downstream of the tailings pond (mean = 0.21 µg/cm^2, range = 0.02 to 1.06 µg/cm^2). In contrast, in 2009 mean benthic Chlorophyll *a* was about 2.5-fold higher at sites located downstream of the tailing ponds (mean = 0.34 µg/cm^2, range = 0.12 to 0.63 µg/cm^2) relative to upstream sites (mean = 0.14 µg/cm^2, range = 0.03 to 0.22 µg/cm^2) (Figure 6). Interestingly, levels of benthic Chlorophyll a in 2009 were highest at sites located 2.5 to 38 km below the tailings pond relative to sites located within 1.5 km of the tailings pond (Figure 6). By contrast, Spencer et al. (2008) reported marked and statistically significant differences in benthic Chlorophyll *a* among the upstream reference site (0.46 µg/cm^2), and a near field (1.72 µg/cm^2) and far-field site (4.58 µg/cm^2) located about 2.4 km downstream of the tailings pond.

Mean water turbidity did not differ appreciably between sites located upstream relative to those downstream of the of the tailings pond in either 2008 (upstream sites: mean = 3.4 nephelometric turbidity units (NTU), range = 1.1 to 5.2 NTU; downstream sites mean = 3.3 NTU, range = 0.1 to 5.7 NTU) or in 2009 (upstream sites: mean = 16.0 NTU, range = 10.2 to 21.5 NTU;

downstream sites mean = 15.7 NTU, range = 13.4 to 17.2 NTU). Higher water turbidity in 2009 relative to that in 2008 coincided with higher discharge in 2009 than in 2008. Spencer et al. (2008) also reported low concentrations of total suspended solids in the Flat River and only modest differences among the upstream reference site (4 mg/L), and the near-field (6 mg/L), and far-field sites (<3 mg/L) located downstream of the tailings pond.

Water pH was slightly alkaline and did not differ appreciably between sites located upstream relative to those downstream of the tailings pond in 2008 (upstream sites: mean = 8.0, range = 7.6 to 8.2; downstream sites: mean = 8.0, range = 7.7 to 8.2) and in 2009 (upstream sites: mean = 8.1, range = 7.9 to 8.2; downstream sites mean = 8.3 NTU, range = 8.1 to 8.5) (Figure 6).

4.2.2 Metals

In 2008, the mean concentration of aluminum (Al) at downstream sites (mean = 71 µg/L, range = 43 to 111µg/L) was about 70% higher relative to sites upstream of the mine (mean = 42, range = 22 to 68 µg/L) (Figure 7). Concentrations of Al at sites downstream of mine in 2009 (mean = 326, range = 187 to 424 µg/L) also exceeded (difference = 28%) that at sites upstream of the mine (mean = 255, range = 216 to 340 µg/L). On average, concentrations of aluminum in 2009 were about 5 times higher relative to that in 2009, and may have reflected increased contact of shallow ground water and river with aluminum-rich deposits due to increased discharge. The concentration of aluminum at Site 30, located 8.6 km downstream of the tailings pond, resembled that at upstream reference sites (Figure 7). Spencer et al. (2008) also reported only moderate differences in aluminum between the upstream reference site (35 µg/L) relative to the near-field site located 0.4 km downstream of the tailings pond (42 µg/L) and the far-field site located about 2.0 km downstream of near-field site (53 µg/L).

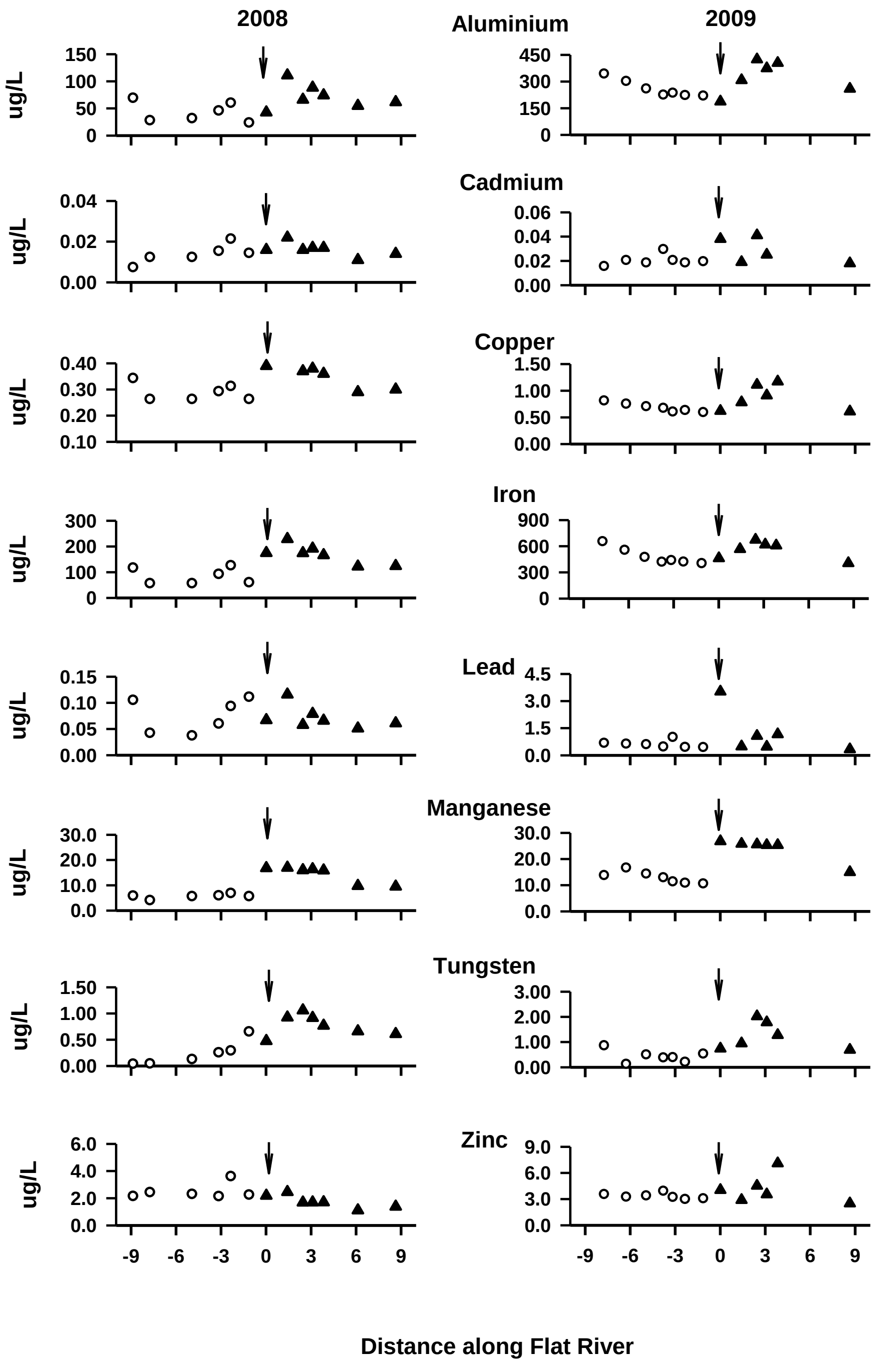

Figure 7. Longitudinal patterns in concentrations of select metals in water from sites located immediately upstream and downstream of the Cantung Mine located in the Flat River Watershed, Northwest Territories. Effluent from the mine is discharged to an exfiltration tailings pond. A portion of this exfiltrate enters the Flat River. Distances denoted with negative values are located upstream of the tailings pond whereas distances with positive values are located downstream of the tailings pond. Distance zero represents the location of Site 38 which is situated immediately downstream of the Tailings pond 3.

Mean concentrations of cadmium at sites downstream of the tailings pond in 2008 (mean = 0.016 µg/L, range = 0.011 to 0.022 µg/L) and 2009 (mean = 0.038 µg/L, range = 0.018 to 0.084 µg/L) were about 14% and 27% higher relative to sites located upstream of the mine (2008: mean = 0.014 µg/L, range = 0.007 to 0.021 µg/L, 2009: mean = 0.020 µg/L, range = 0.015 to 0.029 µg/L), respectively (Figure 7). Concentrations of cadmium at Site 32 located 3.10 km downstream of the tailings pond resembled that at upstream reference sites (Figure 7). By contrast, Spencer et al. (2008) reported concentrations of cadmium of 0.1 µg/L at the reference site, and at the near- and far field sites in the Flat River.

Mean concentrations of copper at sites downstream of the tailings pond in 2008 (mean = 0.36 µg/L, range = 0.29 to 0.44 µg/L) and 2009 (mean = 0.87 µg/L, range = 0.61 to 1.17 µg/L) were about 24% and 26% higher relative to sites located upstream of the mine (2008: mean = 0.29 µg/L, range = 0.26 to 0.34 µg/L, 2009: mean = 0.69 µg/L, range = 0.58 to 0.80 µg/L), respectively (Figure 7). The concentration of copper at Site 30 located 8.6 km downstream of the tailings pond resembled that at upstream reference sites (Figure 7). Spencer et al. (2008) reported a 10% increase in concentrations of copper between the upstream reference site (1.0 µg/L) relative to the near-field site located 0.4 km downstream of the tailings pond (1.1 µg/L) but a 200% increase between the reference site and the far-field site (2.2 µg/L).

In 2008, mean concentration of iron at sites downstream of the tailings pond (mean = 169 µg/L, range = 122 to 229 µg/L) was about two-fold higher relative to sites located upstream of the mine (2008: mean = 81 µg/L, range = 53 to 123 µg/L). Differences in concentrations of iron between sites located downstream versus upstream of the tailings were less apparent in 2009 when concentrations at downstream sites (mean = 555 µg/L, range = 406 to 673 µg/L) were about 17% higher relative to upstream sites (mean = 473 µg/L, range = 396 to 647 µg/L) (Figure 7). The concentration of iron at Site 30 located 8.6 km downstream of the tailings pond resembled that at upstream reference sites (Figure 7). Similarly, Spencer et al. (2008) reported an approximate doubling of concentrations of iron between the upstream reference site (143 µg/L) relative to the near-field site (347 µg/L) and the far-field site (326 µg/L).

In 2008, mean concentration of lead did not differ appreciably between sites located downstream of the tailings pond (mean = 0.071 µg/L, range = 0.051 to 0.12 µg/L) relative to sites upstream of the tailings pond (2008: mean = 0.074 µg/L, range = 0.036 to 0.11 µg/L). In 2009, however, concentrations of lead iron at sites located downstream of the tailings pond (mean = 1.17 µg/L, range = 0.33 to 3.52 µg/L) were about two-fold higher relative to upstream sites (mean = 0.57 µg/L, range = 0.41 to 0.97 µg/L). The marked difference in mean concentrations of lead between upstream and downstream sites reflected a high concentration of lead at Site 38 (3.52 µg/L) located about 0.01 km downstream of the tailings pond (Figure 7). The concentration of lead at Site 39 located 1.4 km of tailings pond resembled that at upstream reference sites (Figure 7). Spencer et al. (2008) reported concentrations of 2.3 µg/L and 0.1 µg/L at the upstream reference site and the near-field site, respectively. Concentration of lead at the far-field site (67.6 µg/L) was 29 times higher than that at the reference site (Spencer et al. 2008).

Mean concentrations of manganese at sites downstream of the tailings pond in 2008 (mean = 14.5 µg/L, range = 9.5 to 17.0 µg/L) and 2009 (mean = 24.0 µg/L, range = 15.0 to 26.8 µg/L) were about three and two-fold higher relative to sites located upstream of the mine (2008: mean = 5.4 µg/L, range = 3.8 to 6.6 µg/L, 2009: mean = 12.7 µg/L, range = 10.3 to 16.4 µg/L), respectively (Figure 7). The concentration of manganese at Site 30 located 8.6 km downstream of the tailings pond exceeded that at upstream reference sites in both 2008 (9.5 µg/L) and 2009 (15 µg/L) (Figure 7). Spencer et al. (2008) also reported 2.9 and 2.8-fold differences in concentrations of manganese between the upstream reference site (8.5 µg/L) relative to the near-field site located 0.4 km downstream of the tailings pond (24.9 µg/L) and the far-field site located about 2.0 km downstream of near-field site (23.4 µg/L), respectively.

Mean concentrations of tungsten at sites downstream of the tailings pond in 2008 (mean = 0.77 µg/L, range = 0.48 to 1.1 µg/L) and 2009 (mean = 1.24 µg/L, range = 0.70 to 2.0 µg/L) were about three-fold higher relative to sites located upstream of the mine (2008: mean = 0.22 µg/L, range = 0.03 to 0.64 µg/L, 2009: mean = 0.42 µg/L, range = 0.10 to 0.84 µg/L), respectively (Figure 7). The concentration of tungsten at Site 32 located 3.10 km downstream of the tailings pond resembled that at upstream reference sites (Figure 7). Spencer et al. (2008) reported 3.5 and 4.5 fold differences in concentrations of tungsten between the upstream reference site (0.8 µg/L) relative to the near-field site (2.8 µg/L) and the far-field site located about 2.0 km downstream of near-field site (3.6 µg/L), respectively.

Mean concentrations of zinc at sites downstream of the tailings pond did not differ appreciably relative to that at upstream sites (Figure 7). In fact in 2008, mean concentration at upstream sites (mean = 2.4 µg/L, range = 2.1 to 3.6 µg/L) actually exceeded that relative so downstream sites (mean = 1.8 µg/L, range = 1.1 to 2.5 µg/L). In 2009, mean concentration of zinc at sites located downstream of the tailings pond (mean = 4.1 µg/L, range = 2.5 to 7.1 µg/L) was about 24% higher relative to sites located downstream of the tailings pond (mean = 3.3 µg/L, range = 2.9 to 3.9 µg/L) (Figure 7). The concentration of tungsten at Site 30 located 8.6 km downstream of the tailings pond (2.5 µg/L) resembled that at upstream reference sites (Figure 7). Similarly, Spencer et al. (2008) reported minor differences in concentrations of zinc between the upstream reference site (4.51 µg/L) relative to the near-field site (3.56 µg/L) and the far-field site (5.0 µg/L), respectively.

Increased concentrations of many water quality parameters were not consistently highest at Site 38 located 0.01 km downstream of the tailings pond. Rather, levels of the majority of water quality descriptors were highest between 1.4 to 3.83 km downtream of the tailings pond. The underlying cause of this observation is not well understood, but could arise if exfiltrate from the tailings pond enters a large and poorly sorted alluvium that comprises the flood plain south of tailings pond and that exfiltrate migrates to the Flat River over an extended distance. Secondly, concentrations of most water chemical parameters at sites of modest distance downstream of the tailings ponds could reflect additional point source and diffuse inputs of nutrients and metal-rich water from geo-thermally enriched shallow ground water located downtream of the tailings pond.

4.3 Development of the reference condition approach model

4.3.1 Cluster analysis and ordination

Cluster analysis identified four assemblages (Figure 8). Assemblage 1 comprised two sites (Site codes = 08-004, 08-005) located in the northwest region of the watershed with the remaining three assemblages composed of between 22 and 30 sites (Figure 8).

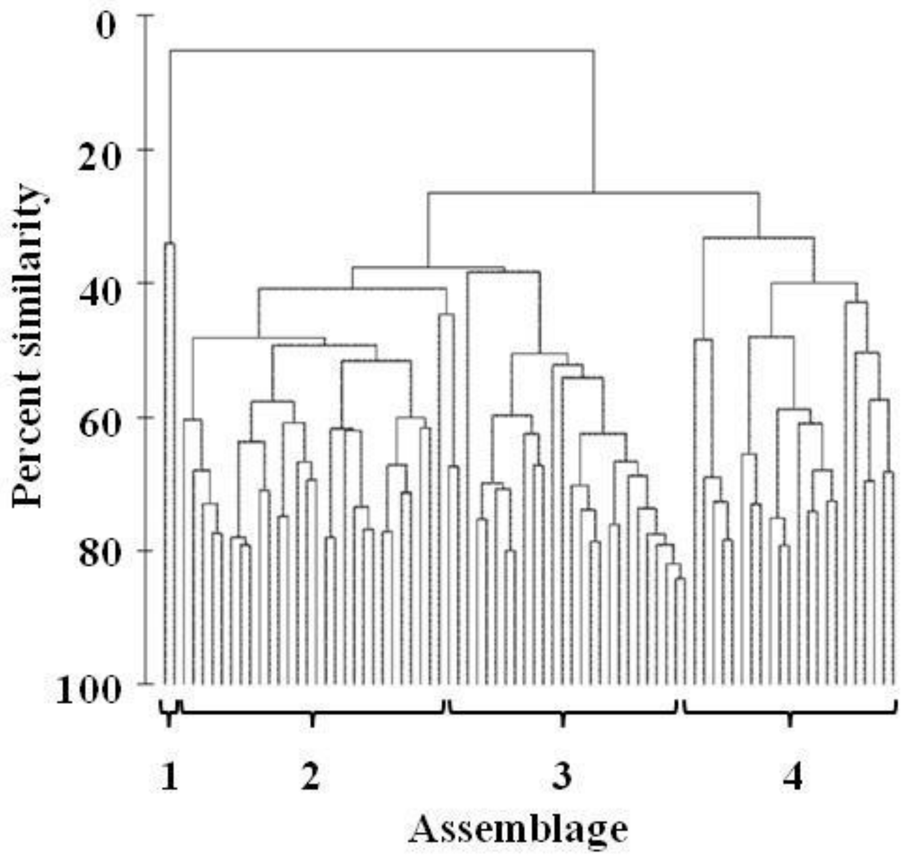

Figure 8. Cluster analysis of 78 reference sites in the South Nahnni Watershed, Northwest Territories sampled in 2008 and 2009. Assemblage 1 is a group of two sites whereas the three numerically dominant assemblages (assemblages 2 to 4) consisted of 22 to 30 sites.

Ordination showed that benthic macroinvertebrate assemblages varied along two gradients reflecting differences in total abundance of benthic macroinvertebrates and abundances of mayfly larvae of Heptanegeniidae and Baetidae (Figure 9). Assemblage 1 comprising only two sites and supported the lowest abundances of macroinvertebrates collected. Sites within the remaining three assemblages varied with total macroinvertebrate abundance (Axis 1) and numbers of Heptageniid and Baetid mayflies (Axis 2) (Figure 9). We evaluated differences among the four assemblages using the SIMPROF test, which is a permutation test developed by Clarke et al. (2008). These analyses showed statistically significant ($P < 0.01$) differences among all four assemblages (Table 7).

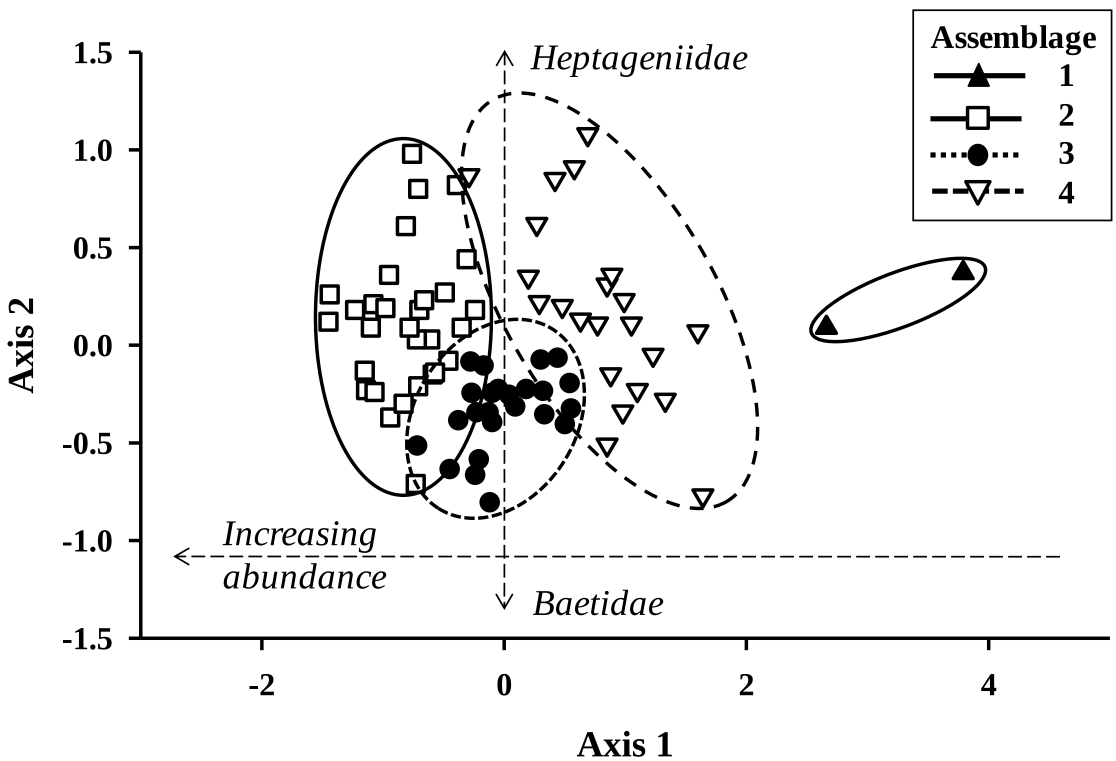

Figure 9. Multi-dimensional scaling ordination of benthic macroinvertebrate communities collected from 78 reference sites from the South Nahanni Watershed, Northwest Territories in 2008 and 2009. Ordination and the discriminate function analyses revealed four assemblage types associated most strongly associated with variance in Heptageniidae and Baetidae mayflies and total numbers of macroinvertebrates collected. Ordination stress = 0.13. Confidence ellipses enclose all sites within each assemblage (group) type. Assemblages 2 and 3 represent reference groups for test sites in the Flat River and Prairie Creek, respectively.

Table 7. Summary of analysis of similarity (ANOSIM) scores for the cluster analysis showing statistically significant (P < 0.01) differences among all four assemblages. Global r for the analysis = 0.69.

	Assemblage		
Assemblage	1	2	3
2	1.000		
3	1.000	0.593	
4	0.958	0.857	0.575

4.3.2 Discriminant Function Analyses

The forward selection discriminant function analysis showed that the six variables of latitude, stream order, percent forest cover, percent ice cover, maximum June air temperature, and average water velocity successfully discriminated among the four macroinvertebrate assemblages (Table 8). The discriminant function models had an overall classification success of 69% (i.e., 54 of the 78 sites). Classification success for assemblages 3 (79.2% [19 of 24 sites]) and 4 (72.7% [16 of 22 sites]) exceeded that for assemblage 2 (63.3% [19 of 30 sites]). The discriminant model was unable to correctly classify either of the two sites in assemblage 1.

Table 8. Summary of environmental (A) and faunal (B) charactreritics of the four benthic macroinvertebrate assemblages from the South Nahnni Watershed, Northwest Territories. Assemblages 2 and 3 represent reference communities for test sites in Prairie Creek and the Flat River, respectively.

A) Environmental variable		Assemblage			
		1	2	3	4
Number of sites		2	30	24	22
Latitude (UTMN)		62.64 ± 0.01	62.02 ± 0.08	61.77 ± 0.08	61.82 ± 0.08
Stream order		4.0 ± 1.0	4.1 ± 0.1	4.1 ± 0.1	5.0 ± 0.2
% Forest		24.5 ± 5.2	23.8 ± 3.3	30.0 ± 3.8	29.9 ± 2.7
% Ice		6.5 ± 4.4	25.9 ± 3.4	4.4 ± 1.6	6.2 ± 2.1
Maximum June air temperature (ºC)		18.0 ± 0.0	16.6 ± 0.5	16.2 ± 0.4	15.0 ± 0.6
Mean water velocity (m/s)		0.43 ± 0.06	0.63 ± 0.03	0.55 ± 0.01	0.59 ± 0.02
B) Invertebrate communities					
Familial richness		9.0±3.0	16.0±0.4	16.5±0.6	12.9±0.9
Total abundance		20.5±11.5	2173.5±179.4	752.1±68.5	327.1±50.7
Class, Order					
Ephemeroptera	Ameletidae	0±0	11.7±2.8	11.9±3.9	6.1±1.4
	Baetidae	2.5±2.5	225.1±33.5	30.8±4.6	110.4±30.9
	Ephemerellidae	0.5±0.5	297.3±58.7	29.5±5.7	18.2±7.5
	Heptageniidae	2.0±2.0	527.3±70.6	290.1±27.0	81.7±10.4
Plecoptera	Capniidae	0±0	183.8±39.8	110.0±17.6	7.0±2.2
	Chloroperlidae	1.5±1.5	124.4±17.6	45.8±8.3	16.5±3.2
	Nemouridae	2.5±2.5	165.3±31.3	66.5±17.5	7.2±2.1
	Perlodidae	0±0	40.3±9.5	7.1±1.2	6.5±1.2
	Taeniopterygidae	0±0	6.0±2.2	11.2±3.0	8.4±2.9
Trichoptera	Hydropsychidae	0.5±0.5	9.1±2.7	8.1±1.8	2.7±1.0
	Rhyacophilidae	0±0	11.6±3.7	3.2±1.0	0.6±0.2
Diptera	Chironomidae	6.0±2.0	429.4±60.5	60.8±14.2	38.7±16.5
	Simuliidae	0.5±0.5	13.4±6.2	26.5±23.0	4.3±2.5
	Tipulidae	0±0	5.6±1.2	0.9±0.5	0.2±0.1
Oligochaeta	Lumbricilidae	0.5±0.5	8.6±3.5	20.2±9.6	6.0±3.3
	Tubificidae	0±0	3.1±1.8	0.1±0.1	0.1±0.1
	Planariidae	0.5±0.5	12.1±7.4	5.0±1.7	0.2±0.2
	Naididae	0±0	18.6±17.5	0±0	0±0
Unionidae	Sphaeriidae	0±0	5.6±5.0	0±0	0±0
Hydarchnidia	Sperconidae	0±0	15.9±3.3	11.7±2.7	5.9±2.7

4.3.3 Structure of the four macroinvertebrate assemblages

Group 1 was characterized by very low abundance and richness with dipterans (Chironomids and Tipulids) and stoneflies (nemourids and chloroperlids) as the characteristic families (Table 8). Group 2 is high abundance and richness, and is a mayfly dominated (50%) community, primarily the Heptageniidae, but with abundant Baetidae and Ephemerellidae and Chironomidae. Group 3 is of intermediate abundance and high richness comprised of Heptageniidae (percent composition = 38%) and Capniidae, Nemouridae and Chloroperlidae stoneflies that comprised (30%) of the entire community. Chironomids are the least common in this community.
Finally, Group 4 is low abundance and intermediate richness with mayflies (Baetidae and Heptageniidae) as the dominant taxa (>60%) and Chironomidae as the second most frequent organisms. Stoneflies are least abundant in this community.

4.4 Application of the reference model to assess site impairment

We tested for potential environmental impairment by assessing the ecological condition of eight sites located from 0.1km to 18.0 km downstream of the Prairie Creek Mine on Prairie Creek, and 13 sites located adjacent to, and between 1.4 to 159 km downstream of the main tailings pond at the Cantung Mine on the Flat River (Tables 9 and 10). We completed site assessments using the online Analytical tools module developed by the Canadian Aquatic Biomonitoring Network (CABIN) (**http://cabin.cciw.ca/BEAST/Tools/Project.aspx**). These tools apply standard reference condition approach analyses by comparing the structure of benthic macroinvertebrate community at a site of unknown condition (i.e., a test site) to an appropriate reference group by firstly applying the discriminant function model to identify the most appropriate reference group and secondly, using ordination (a semi-strong hybrid algorithm), to determine the similarity of the test site to the distribution of reference sites. To minimize distortion of the distribution of reference sites, test site were evaluated one at a time.

As stated previously, we assessed site condition by determining if the test site was equivalent or not to reference probability ellipses were constructed around the reference sites using the three ellipses (90, 99, and 99.9%) to establish four assessment bands (Band 1 - ≤ 90% (equivalent to reference sites [unimpaired]), Band 2 - 90 to 99% (possibly different from reference sites) and Band 3 - 99% to 99.9% [impaired], Band 4 > 99.9% highly impaired).

4.5 Biological assessments of site condition in Prairie Creek

4.5.1 Present study

Our ordination analyses revealed no evidence of biological impairment in 2008, but evidence of possible biological impairment in 2009 (Table 9, Figure 10). In 2008, all eight test sites located 0.3 to 9.2 km downstream of the confluence of Harrison Creek and Prairie Creek fell within Band 1 in ordination space indicating that all sites were unimpaired (i.e., in reference condition). In 2009, our analyses showed that sites 43, 46 and Site 45 located 0.31, 0.54, and 1.69 km downstream of the metal rich discharge to Prairie Creek were possibly impaired. Interestingly our assessment also showed that Site 44 located 1.1 km downstream of the confluence between Harrison Creek and Prairie Creek was unimpaired, but was bounded by a site upstream (Site 46) and downstream (Site 45) that were possibly impaired (Figure 10). Our finding of a spatial mosaic of adjacent sites differing in ecological condition (i.e., impairment) could arise if local inputs of ground water dilute metal-enriched river water. Alternatively, the spatial mosaic of possibly impaired sites with sites in reference condition could reflect the fact that site conditions only varied by one ordination band (i.e., reference versus potential impairment). In these situations, sites in adjacent bands in ordination space share a common suite of benthic macroinvertebrates taxa and that our model is not overly sensitive to discriminating between these assessment states.

Table 9. Summary of environmental assessments of test sites located downstream of the Canadian Zinc mine in the Prairie Creek Watershed, Northwest Territories. Site assessments are derived from ordinations of test sites within the appropriate reference group. Distances are kilometres downstream of the metal rich discharge from the mine site to Prairie Creek.

Site	Distance (km) downstream of the mine discharge	Site assessment	
		2008	2009
43	0.1 km	Unimpaired	Possibly impaired
46	0.54 km	Unimpaired	Possibly impaired
44	1.08 km	Unimpaired	Unimpaired
45	1.69 km	Unimpaired	Possibly impaired
70	3.17 km	Unimpaired	Unimpaired
69	6.00 km	Unimpaired	Unimpaired
68	9.22 km	Unimpaired	Unimpaired
67	18.0 km	Unimpaired	Unimpaired

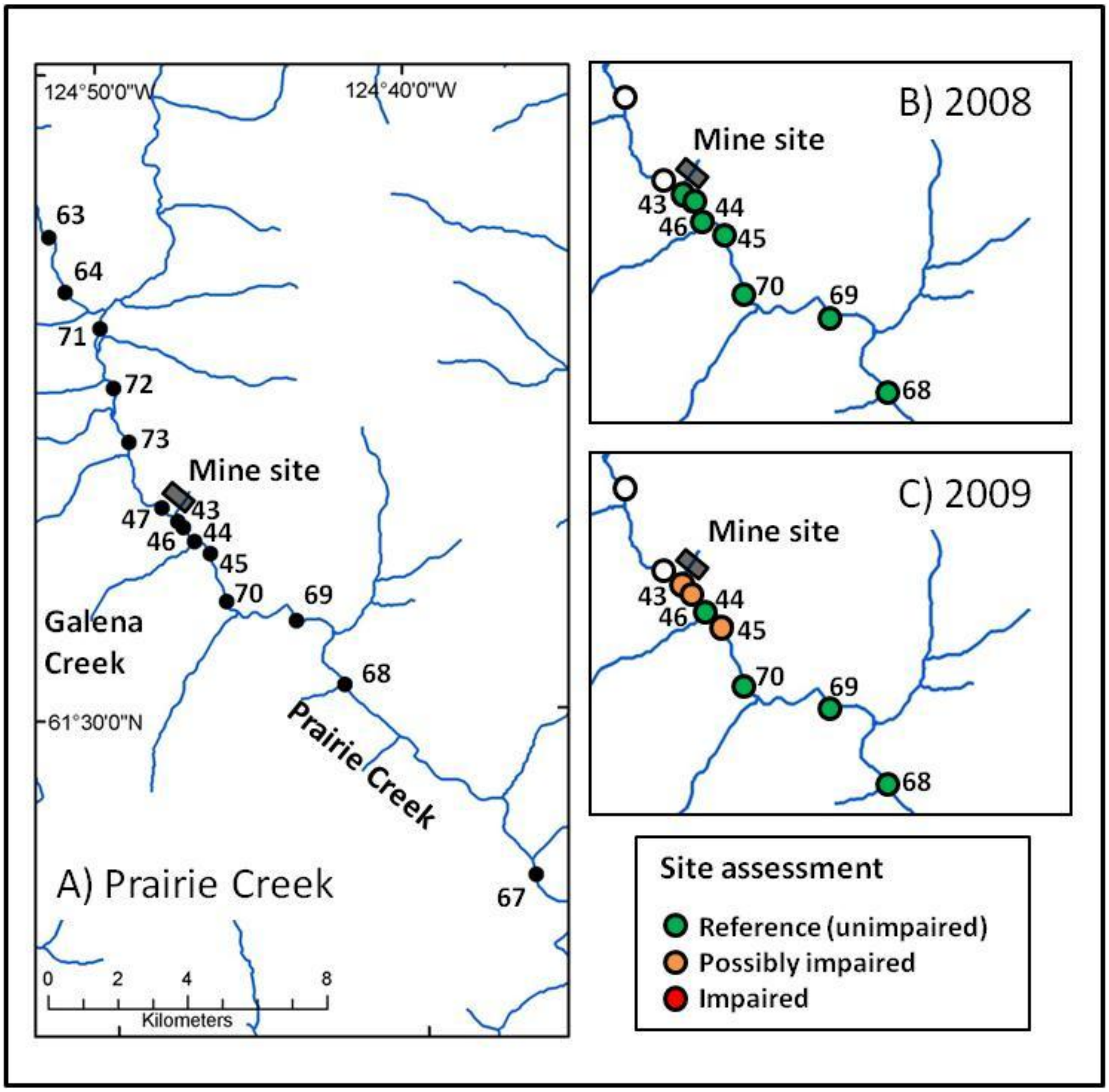

Figure 10. Location and ecological conditions of test sites located downstream of the Canadian Zinc mine in the Prairie Creek Watershed, Northwest Territories.

We used the results of site assessments to estimate the zone of ecological impairment, expressed in kilometres. Because all sites in 2008 were unimpaired, the zone of impairment in 2008 is zero kilometres. In 2009, site assessments showed that Site 45 (located 1.69 km downstream of the Harrison Creek-Prairie Creek confluence) was potentially impaired whereas Site 70, located 3.17 km downstream, was unimpaired. Thus, the zone of potential impairment could range from 1.69 to 3.17 km (Table 9, Figure 10).

4.5.1 Comparisons with published studies

Comparisons of ecological communities at sites upstream relative to those downstream of the Prairie Creek Mine on Prairie Creek have also completed by Spencer et al. (2008) and Bowman et al. (2010) and results from these studies provide insights on the magnitude and spatial extent of impairment in Prairie Creek. Spencer et al. (2008) used a multi-trophic approach to assess impairment including comparisons of algal and benthic macroinvertebrate community structure at an upstream reference site, and near- and far-field field sites located about 0.1 and 2.1 km downstream of the confluence of Harrison Creek and Prairie Creek. Bowman et al. (2010) also used multiple biological attributes to assess ecological impairment of the same near-and far field sites downstream of the mine site. However, in contrast to Spencer et al. (2008) who used a single site to define reference conditions, Bowman et al. (2010) derived a regional reference condition model using data from 20 sites located in the South Nahanni Watershed.

Spencer et al. (2008) assessed impairment by quantifying differences in multiple descriptors of biological communities, including those describing the structure of algal and benthic macroinvertebrate communities, at the upstream reference site relative to the downstream sites potentially exposed to nutrient and metal rich mine discharge. Our assessments of their data suggest that the zone of biological impairment in Prairie Creek varies depending on the biological descriptor used to assess impairment. For example the lack of statistical differences in benthic Chl *a* suggest that sites downstream of the mine are equivalent to upstream site and thus the zone of impairment is less than 0.1 km downstream of the mine discharge or does not exist. When assessments are based comparisons of the percent composition of the diatom *Acnanthes minutissima* the zone of impairment extends between 0.1 to 2.1 km downstream of the mine discharge, whereas the zone of impairment extends between at least to 2.1 km downstream of the mine if assessments are based on comparisons of algal diversity and algal family richness (See Spencer et al. 2008, Figure 3).

Assessments of the biological impairment of near-and far-field sites downstream of the mine discharge to Prairie Creek reported by Bowman et al. (2010) varied depending on whether the assessments were completed using metrics describing algal or benthic macroinvertebrate community structure. Assessments using benthic algal metrics and ordinations suggested that sites downstream of the mine were in reference condition (i.e., unimpaired) and that the zone of influence is located within 0.1 km of the mine discharge or that it does not exist (i.e., the zone of impact = 0 km) (Bowman et al. 2010). By contrast, assessments of site condition using metrics describing benthic macroinvertebrate community structure showed impairment of the near-field, but not the far-field site. Consequently, the zone of influence identified using benthic macroinvertebrates in Prairie Creek extends between 0.1 to 2.1 km downstream of the confluence of Harrison Creek and Prairie Creek.

4.6 Biological assessments of site condition in the Flat River

4.6.1 Present study

Our ordination analyses revealed evidence of possible impairment in both 2008 and 2009 (Table 10, Figure 11). In 2008, our analyses showed that sites 38 and 2 located 0.01 and 2.45 km downstream of the tailings pond were possibly impaired, whereas in 2009 sites 38 and 39 located 0.01 and 1.42 km downstream of the tailings pond were possibly impaired (Table 10, Figure 11). All other sites located immediately downstream of the tailings pond were unimpaired (i.e., in reference condition).

Zones of potential impairment varied between years. In 2008, the zone of potential impairment extends between Site 2 and Site 32, that is, between 0.01 and 2.45 km downstream of the tailings pond. In 2009, the zone of potential impairment extends between Site 39 and Site 2, that is, between 1.42 to 2.45 km downstream of the tailings pond (Table 10, Figure 11).

4.6.2 Comparisons with published studies

Our assessments of the data presented by Spencer et al. (2008) suggest that the zone of biological impairment in the Flat River varies depending on the biological descriptor used to assess impairment. For example, the statistically significant difference in mean benthic Chl *a* between the upstream site and the far-field site suggests that the zone of impairment extends to at least 2.4 km downstream of the tailings pond. Reduced levels of algal richness at the far-field site relative to the upstream site also suggest a zone of influence of at least 2.4 km. By contrast, zone of impairment based on comparisons of: i) algal diversity and ii) percent composition of the diatom *Acnanthes minutissima* suggests a shorter zone of impairment that falls between 0.1 to 2.4 km downstream of the tailings pond (See Spencer et al. 2008, Figure 3).

Table 10. Summary of environmental assessments of test sites located adjacent and downstream of Cantung Mine in the Flat River Watershed, Northwest Territories. Site assessments are derived from ordinations of test sites within the appropriate reference group. Site 1 is located upstream of Tailings Pond 3 but is considered to be exposed since it is located in an area of the floodplain that received mine tailings. The distance from Site 38 to Tailings Pond 3 is 0.01 km since it is located immediately adjacent to Tailings pond 3. Site assessments are derived from ordinations of appropriate reference groups and test sites using the reference condition model. Distances are kilometres upstream or downstream of Tailings Pond 3.

Site	Distance (km) upstream or downstream of Tailings Pond 3	Site assessment	
		2008	2009
1	1.1 km upstream	Unimpaired	Unimpaired
38	0.01 km downstream	Possibly impaired	Possibly impaired
39	1.42 km downstream	Unimpaired	Possibly impaired
2	2.45 km downstream	Possibly impaired	Unimpaired
32	3.10 km downstream	Unimpaired	Unimpaired
31	3.83 km downstream	Unimpaired	Unimpaired
19	6.12 km downstream	Unimpaired	Unimpaired
30	8.64 km downstream	Unimpaired	Unimpaired
29	10.98 km downstream	Unimpaired	Unimpaired
28	15.32 km downstream	Unimpaired	Unimpaired
27	19.51 km downstream	Unimpaired	Unimpaired
26	23.67 km downstream	Unimpaired	Unimpaired
33	158.9 km downstream	Unimpaired	Unimpaired

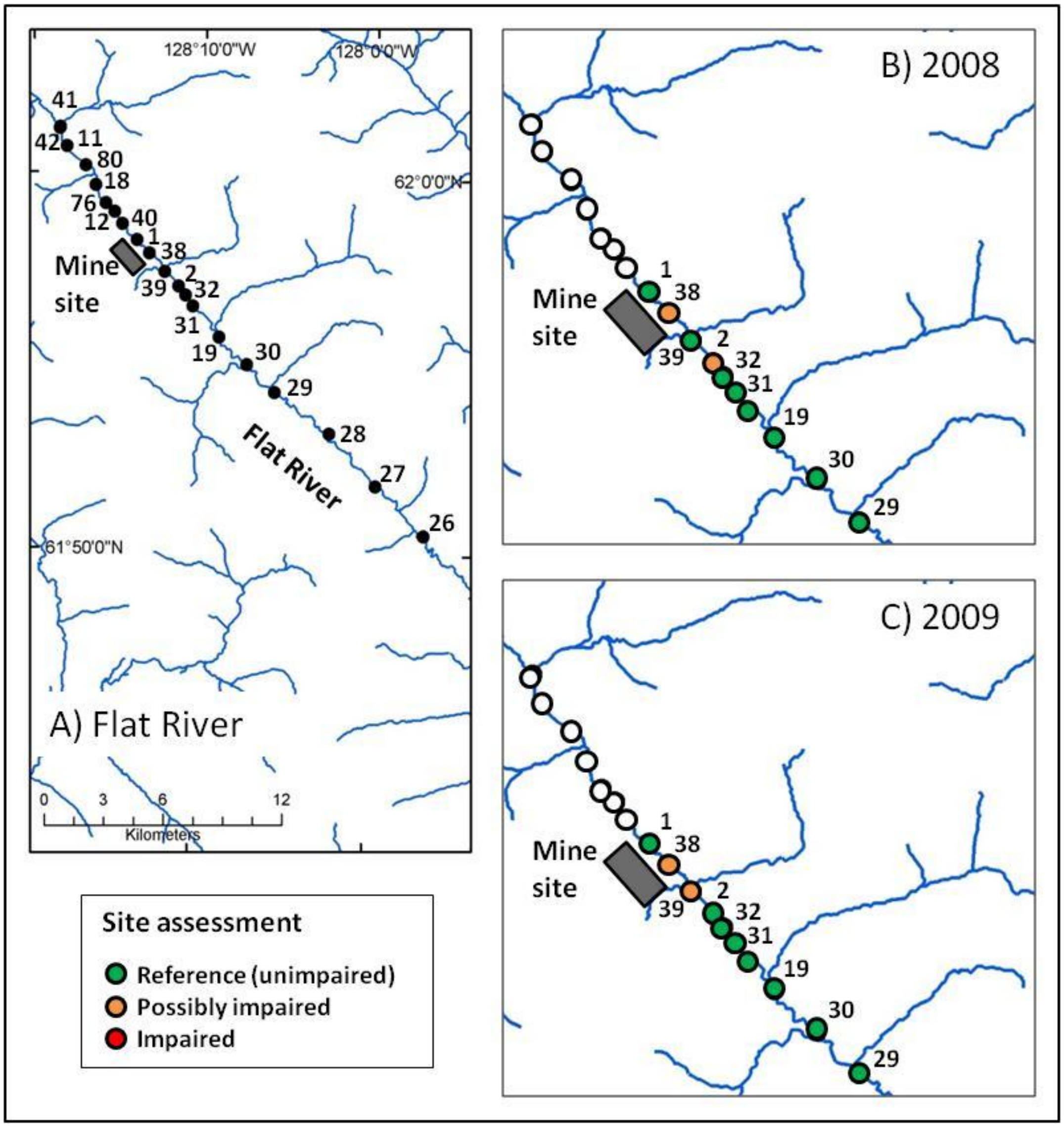

Figure 11. Location and ecological conditions of test sites located downstream of the Cantung Mine in the Flat River Watershed, Northwest Territories.

Bowman et al. (2010) used multiple biological attributes to assess ecological impairment of the same near-and far field sites located 0.4 and 2.4 km downstream of the tailings pond used by Spencer et al. (2008). Bowman et al. (2010) reported that benthic Chlorophyll *a* at the near-field, but not the far-field site, was outside of reference condition indicating a zone of influence of between 0.4 to 2.4 km downstream of the tailings pond. Site assessments using benthic macroinvertebrate metrics showed that both the near- and far field sites were unimpaired (i.e., within the range of reference conditions). Ordinations of community structure also showed that the near-and far-field sites were located were located within 95% confidence ellipse of the range of reference communities. Taken together, assessments using benthic macroinvertebrates suggest that the zone of influence falls within < 0.4 km downstream of the tailings pond or that a zone of does not exist (i.e., zone of influence = 0 km).

4.7 Assessments of duplicate samples and samples taken with a fine mesh net

4.7.1 Assessments of duplicate samples

We evaluated error rates, i.e., differences in site assessments, by collecting duplicate samples from each of eight sites in 2008 and 2009. We controlled for potential differences in sample composition by ensuring that a single CABIN-certified field crew member collected both the original sample and the duplicate sample. We assessed the condition of both samples using online the analytical tools in CABIN as described previously. We defined sample comparability (i.e., concordance in site assessments) by comparing the percentage of times that the first and second fell in the same ordination space (i.e., band).

Our analyses showed that six of the eight duplicate samples (i.e., 75%) were located in the same assessment band (Table 11). While these analyses suggest moderately high levels of sample comparability, they are based on small sample sizes. We suggest that additional efforts to quantify sample comparability using larger data sets including the those from different reference condition models is required to fully understand error rates.

4.7.1 Assessments of samples taken with fine mesh

We assessed comparability of benthic macroinvertebrate samples taken with a CABIN kick net fitted a fine mesh (250 µm) versus that fitted with a coarse and more commonly used 400 µm net. Our analyses showed that the structure of benthic macroinvertebrates samples collected using the fine 250 µm mesh net were not highly comparable to that collected using the standard

400 μm mesh net. In fact, ordinations showed that only four of the 18 fine mesh samples (22%) fell in the same ordination band as the routine sample collected with net fitted with 400 μm mesh (Table 12).

Table 11. Comparisons of site assessments of duplicate samples with regular samples taken from eight sites in the Flat River, the South Nahanni Watershed, Northwest Territories in 2008 and 2009.

Site and year code	Site location	Site condition		Concordance in site assessment
		First regular sample	Duplicate sample	
01-2009	1.08 km upstream of the tailings pond	Unimpaired	Unimpaired	Yes
02-2009	2.45 km downstream of the tailings pond	Unimpaired	Unimpaired	Yes
12-2009	3.11 km upstream of the tailings pond	Unimpaired	Unimpaired	Yes
18-2009	4.89 km upstream of the tailings pond	Unimpaired	Possibly impaired	No
31-2009	3.83 km downstream of the tailings pond	Unimpaired	Stressed	No
38-2009	0.01 km downstream of the tailings pond	Possibly impaired	Possibly impaired	Yes
39-2009	1.42 km downstream of the tailings pond	Possibly impaired	Possibly stressed	Yes
76-2008	3.9 km upstream of the tailings pond	Unimpaired	Unimpaired	Yes

Table 12. Comparison of site assessments using benthic macroinvertebrate samples collected in 2009 with a fine mesh net (250 µm) versus that collect using the standard 400 µm mesh net.

Sample number	Site location	Site condition		Concordance
		Regular sample taken with 400 µm mesh net	Fine mesh sample taken 250 µm mesh net	
2009-01	Adjacent to mine site, Flat River Watershed	Unimpaired	Unimpaired	Yes
2009-02	2.45 km downstream of tailings pond, Flat River Watershed	Unimpaired	Possibly impaired	No
2009-12	2.3 km upstream of tailings pond, Flat River Watershed	Unimpaired	Possibly impaired	No
2009-15	Little Nahanni River, Little Nahanni Watershed	Unimpaired	Possibly impaired	No
2009-16	Little Nahanni River, Little Nahanni Watershed	Unimpaired	Possibly impaired	No
2009-17	Little Nahanni River, Little Nahanni Watershed	Unimpaired	Possibly impaired	No
2009-18	4.5 km upstream of tailings pond, Flat River Watershed	Unimpaired	Highly impaired	No
2009-31	3.83 km downstream of tailings pond, Flat River Watershed	Unimpaired	Impaired	No
2009-38	0.01 km downstream of tailings pond, Flat River Watershed	Possibly impaired	Possibly impaired	Yes

Table 12 – continued

Sample number	Site location	Site condition in 2009		Concordance
		Regular sample taken with 400 µm mesh net	Fine mesh sample taken 250 µm mesh net	
2009-39	1.42 km downstream of tailings pond, Flat River Watershed	Possibly impaired	Highly impaired	No
2009-43	0.1 km downstream of mine site, Prairie Creek Watershed	Possibly impaired	Possibly impaired	Yes
2009-44	1.08 km downstream of mine site, Prairie Creek Watershed	Unimpaired	Possibly impaired	No
2009-45	1.69 km downstream of mine site, Prairie Creek Watershed	Possibly impaired	Unimpaired	No
2009-46	0.54 km downstream of mine site, Prairie Creek Watershed	Possibly impaired	Possibly impaired	Yes
2009-47	0.58 km upstream of mine site, Prairie Creek Watershed	Unimpaired	Possibly impaired	No
2009-70	3.17 km downstream of mine site, Prairie Creek Watershed	Unimpaired	Possibly impaired	No
2009-76	3.9 km upstream of tailings pond, Flat River Watershed	Unimpaired	Possibly impaired	No
2009-77	Tributary of the Little Nahanni River, Little Nahanni Watershed	Unimpaired	Impaired	No

4.8 Monitoring conclusions, and operational considerations

4.8.1 General conclusion on the application of the reference condition model for the South Nahanni Watershed

Our data suggest that the reference condition approach is a viable alternative to traditional biological assessment approaches and can be used to: i) assess the potential effects of mining on the health of the Flat River and Prairie Creek, and ii) when combined with ongoing monitoring can be used to describe possible changes in stream health through time, including evaluations of potential effects of climate changes and long-range deposition of airborne contaminants. The existing reference condition approach model is well suited to supporting both: i) broad-scale assessment and reporting needs such as that related to "State of the Park" reporting, and ii) fine-scale assessments, such as that associated with determining impairment immediately downstream of industrial activities.

4.8.2 Future applications and cost estimates

Identifying the number of streams that need to be sampled and the frequency of sampling required to satisfy broad-scale "State of the Park" reporting and finer-scale assessments depends on the specific questions being asked, and the labour and cash resources available to complete the work. Satisfying "State of the Park" reporting would at a minimum require, one large scale sampling event every 4 to 5 years and could comprise sampling of 30 to 40 river sites, including rivers whose watersheds include a high percentage of glaciers. Assuming that field crews used mine sites (i.e., Cantung Mine, Prairie Creek) as operating bases, a trained and well organized two-person field crew should be able to collect samples from 30 to 40 sites in 4 to 5 days. Assuming that the return flight to transport the field crew from Fort Simpson to the mine site is $3,000 and costs to access the majority of sites with helicopter is $40,000 (i.e., 4 days @$10,000 per day), the total travel costs would be $43,000. Costs to identify benthic macroinvertebrates (35 samples @350.00 per sample = $12,250) and analyze water samples for a suite of metals and some fractions of nutrients, including total phosphorus and total nitrogen (35 sample @170 =

$5,950) would total $18,200. When these two cost categories are combined total project cost, excluding staff time, would be $61,200. When expressed over a 5-year amortization period (i.e., the need to complete a "State of the Park" report every 5 years), annual costs to satisfy "State of the Park" by reporting on stream health would be about $12,240. Because information gained through site assessments using the reference condition model could also support each of the mines' environmental monitoring program (e.g., monitoring to fulfil environmental effects monition under the Metal Mining Effluent Regulations) substantial cost-sharing opportunities exist. Financial costs to Parks Canada Agency could also be reduced if funds could be secured through external sources such Aboriginal Affairs and Northern Development Canada's Cumulative Impact Monitoring Program.

4.8.3 Model Verification and Recalibration

The current reference condition model for the South Nahanni Watershed was constructed using biological data collected in 2008 and 2009. An important consideration in the use of reference condition models is to verify that the structure of biological communities at reference sites is not changing dramatically and systematically through time. Appreciable changes in the structure of biological communities at reference sites would result in appreciable Type I and Type II errors and would compromise assessments of site condition. Confirming that biological communities at reference sites have not shifted appreciably could be conducted in 3 to 5 years (2015 to 2018) as one component efforts to complete a State of the Park report for Nahanni National Park Reserve. This would require re-sampling and assessments of about 10 to 15 reference sites that had been previously sampled in 2008 and 2009. The RCA model would be verified if the majority of these samples were located within the 90% reference ellipse, i.e., are unimpaired, and would indicate that the existing model is still valid and does not require re-calibration.

The reference condition approach model for the South Nahanni Watershed would however require re-calibration under at least three scenarios. First, if: i) biological communities from watersheds that included appreciable (>40% glaciers) amounts were not well strongly predicted to one of the existing biological community types and ii) Park Managers wanted to report on the biological condition of these glacier-dominated streams. Second, if the majority of streams sampled in 2015 to 2018 were located outside of the 90% reference ellipse, i.e., were possibly impaired, impaired, or highly impaired. Third, if site assessments using the existing model derived from data collected in 2008 and 2009 differed from that using a hybrid model comprised of data collected in 2008-2009 and reference sites in 2015 to 2018. Re-calibration of the RCA model could be completed by Parks Canada staff or by a qualified consultant. A consultant would likely charge $6,000 to $12,000 to re-calibrate the RCA model for the South Nahanni Watershed.

Figure 12. A glacial- fed river in the South Nahanni Watershed.

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