THE WHITESAND RIVER: AN ASSESSMENT OF BROOK TROUT HABITAT SUITABILITY FOLLOWING MINE SITE CLOSURE

by

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ABSTRACT

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The Winston Lake Mine operated from 1988-1999 within the Whitesand River watershed located near Schreiber, ON. Operations resulted in releases of elevated cations and anions from tailings pond discharges above Cleaver Lake. The lake entered a period of meromictic stagnation but has since been reported to be experiencing seasonal turnover as a product of natural remediation. The overall goal of environmental studies in the watershed is to determine the likelihood of a successful reintroduction of brook trout. Suitable habitat within Cleaver Lake and the outflow into the Whitesand River is required for this to occur. This thesis involved the Whitesand River. Water quality data provided by First Quantum Minerals was used to assess the potential effect of heavy metals, Total Dissolved Solids (TDS), Dissolved Oxygen (DO), pH, temperature and water hardness on survivability of brook trout. Comparison of pre-operational data and recent water quality records indicated the return of examined water quality and benthic invertebrate populations to conditions statistically similar to the historical habitat. Rising seasonal temperatures were identified as an area of concern having potential to impact future health of cold-water fish communities in the river.

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1. INTRODUCTION, LITERATURE REVIEW AND THESIS OBJECTIVE

Effects of Mining Activities

For many countries, mining is a major source of economic growth. Canada's mineral rich sites host deposits that feature valued minerals including gold, silver, copper, iron, zinc, coal, uranium, limestone and nickel (Sandlos and Keeling 2015). Despite market growth provided by the mining industry, environmental damage caused by mine activity is prevalent through all stages of operation, lowering many of the intrinsic and extrinsic values of the affected area.

The most recognized source of environmental damage created by mines is the damage operations can have on water resources. Water use can be tracked through every stage of operation including exploration, excavation and processing. The presence of such activities within a watershed tends to disrupt the natural hydrologic balances, in addition, many operations also produce water from excavation sites (Younger and Wolkersdorfer 2004). Drawdown from excess pumping activity can result in the local water table lowering, depriving nutrients to shallow-rooted plants within the area and posing a threat to drought intolerant species. Drier conditions may also have a variety of deleterious effects on the local ecosystem, including increased susceptibility to forest fires. Changes to the groundwater systems often also impact surface water levels that balance with groundwater levels. Mine-influenced water generally represents the composition of the host strata, usually with high hardness and elevated sulphate levels. Mine site effluent usually contains elevated levels of copper, arsenic, cadmium, zinc and/or mercury - all of which can be harmful to fish and wildlife and have deleterious effects on the surrounding environment. The toxicity of heavy metal pollutants to aquatic life is not only dependent on concentrations, but can be influenced by factors including pH, hardness and the occurrences of other enabling agents (Tiwary 2000). Depending on the supporting water quality parameters of the host water, metal concentrations of the same level can have little to no effect on aquatic life or acute to chronic toxic effects on local biota.

Mine influenced waters that have a generally low pH, high metal concentrations and metal precipitates are commonly referred to as Acid Mine Drainage or AMD (Udayabhanu and Prasad 2010). The general categorization of AMD accommodates the fact that geochemistry and physical characteristics can complicate chemical processes and vary between sites. At coal and hard rock sites, AMD occurs when the mineral pyrite (FeS) and/or other sulfide-containing minerals are introduced to oxygen and water. Many studies also report high conductivity and low pH values resulting from the presence of AMD in waterbodies, and this condition can allow for further dissolution of present metals and other contaminants, further amplifying the effects of the mine activity on local systems. According to Tripole et al. (2006), bodies of water located within substratum that has a high level of silicate and or granite within the soil can be more susceptible to the impacts of acid stress.

Regardless of the specific categorization of mine site effluent, it is common for elevated levels of contaminants to remain around pumping sites and slowly diminish further downstream. Biodiversity around impacted sites often inversely reflects this pattern. Species richness and diversity is lower at and around the source of contamination and higher downstream. As diversity decreases, community composition changes around the contamination site, substituting sensitive species for more tolerant ones (Tripole et al. 2006). Legacy metals and other contaminants from mine drainage can persist in aquatic systems for decades following the cessation of operations at a mine site (Udayabhanu and Prasad 2010). Often these legacy effects need to be quantified before they can be remediated.

Before and After Analysis

One common approach is to use a Before and After (BA) analysis of the effects of mining on the environment (Aberg and Weber, 2018). Both changes to the chemical and biological habitat are normally quantified. Normally mines in Canada have detailed environmental assessments conducted before, during and after mining has ceased. Detailed water and sediment characteristics are collected at the same sampling stations often over several decades. This is true for the example in this thesis and the water quality results collected prior to the opening of the mine provided reference conditions for the Before data used in the analysis.

The biotic community also will change during a mine's operation. One indicator used to determine the biological health of a waterbody is an assessment of the benthic invertebrate community. Benthic invertebrates spend most of their lives in or around the same general area. They cannot escape localized water pollutants because their means of travel is limited in comparison to migrating fish. Generally, waterbodies with low pollutants will have variation in benthic community structure and a sample would include some taxa that are generally intolerant of pollution. The Regional Aquatic Monitoring Program (RAMP 2018) for the province of Alberta and various other reports consider the orders Ephemoptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies), as taxa that are highly sensitive to pollutants. The percentage of the Ephemoptera, Plecoptera and Trichoptera in a sample make up the EPT index value, commonly used to estimate water quality. Tripole et, al. (2006) found that the benthic communities of Ephemoptera and Trichoptera decreased significantly in affected streams compared to the reference area. The EPT index values should be considered for quantifying the natural rehabilitation progress of the stream. A rise in the presence of EPT taxa can indicate prey abundance and availability. Analysis of pollution tolerant taxa in the stream will also aid assessment of the likelihood of heavy metal biomagnification in fish tissues.

Often rehabilitation of mine influenced waters is targeted at specific organisms. In this particular case, it is the reintroduction of brook trout and a first step is to examine their specific requirements for survival and fecundity.

Brook Trout Ecology

General Characteristics

Brook trout (*Salvelinus fontinalis* Mitchill) are recognized for being highly adaptable compared to other salmonids. They are considered a cold-water species and range from 6-16 inches in length at maturity (Raleigh 1982). In the case of Northwestern Ontario, smaller brook trout size compared to the mean for its native range is generally reflective of habitat limitations. Brook trout are known to become anadromous if the current habitat does not meet the required coverage and water quality standards for year-round life.

Seasonal behaviour

According to the Whitewater to Bluewater Partnership (2018), brook trout movements between inshore and offshore habitats are related mostly to spawning behavior and water temperature. If temperatures rise above 22 degrees Celsius, brook trout will migrate into cooler areas such as deep pools and lakes. Although thermal conditions and lack of spawning habitat can drive brook trout to travel long distance, if current habitats do satisfy thermal requirements and contain adequate spawning areas nearby, they will not travel as much. Spawning occurs during the fall season, between September to October dependant on the temperature and oxygen content (Government of Newfoundland and Labrador 2018).

Life Cycle

Eggs and young brook trout require slightly warmer temperatures than mature fish. Therefore, brook trout will migrate to shallow waters to lay their eggs. The ideal bottom substrate should have enough gravel that the eggs, which are free floating and have no adhesive ability, are buried from sight; the gravel is manipulated into a depression by the female trout, which serves as a protective medium for the eggs that successfully settle within its walls (Whitewater to Bluewater Partnership 2018). Ideally, sediment size should range from 3-50 mm. However, some studies have narrowed the allowable range in spawning substrate to 6–25 mm (Adams et al. 2008). Eggs hatch in roughly 100 days, but the young brook trout remain within the safety of the gravel substrate until the yolk sac is fully absorbed (Government of Newfoundland and Labrador. 2018). After the yolk sac has diminished, fry (juvenile brook trout) prefer shallow stream edges and brooks with adequate coverage. Fry are territorial and live in solitary conditions, because their small size makes them extremely vulnerable to predators. This seclusive way of life continues until they grow large enough to venture out into more open areas (Wild Trout Trust 2019). Maturity is reached generally around two years of age, but in smaller streams that accommodate brook trout with less room to grow, it is possible for them to reach reproductive maturity within their first year of life (Whitewater to Bluewater Partnership 2018).

Habitat Requirements

<u>Water Quality Parameters</u>

Habitat and/or water quality are limiting factors to brook trout stocking success (Kerr 2000). The water quality parameters that may influence stocking success are temperature and dissolved oxygen. Brook trout in any habitat generally require cooler streams. Water that reaches and remains 25 °C and above is lethal to brook trout (Smith 1962). Brook trout growth is slowed above 16 °C and stops around 23.4 °C (Chadwick and McCormick 2017). The optimal temperature range for brook trout has been identified by the Fisheries Board of

Canada (2019) as 10-20°C. Brook trout can survive well below the 10°C base of the range, but growth also slows under such conditions.

Sediment in the water column and apparent colour also may aid in determining the likelihood of brook trout survival. Moderate to excessive turbidity can decrease the reactive distance between predator and prey (Raleigh 1982). Brook trout are a drift feeding species therefore, decreased reactive distance due to high turbidity can be expected to lower prey consumption rates (Sweka and Hartman 2001). Salmonids also require gravelly substrate for spawning; excessively turbid water could affect breeding and development, decreasing the abundance and overall health of new fry (Bash, et al. 2001). Increased organic matter and smaller substrate material can lower available oxygen for eggs and create a suffocating effect, lowering seasonal birth rates and prevent the emergence of new fry (Bash, et al. 2001).

The young stages of a brook trout's life have habitat requirements that are slightly more demanding than adult stages, as younger trout are much more vulnerable during their developmental stages of life. Eggs, and newly hatched brook trout require high dissolved oxygen (Teears 2016). As identified by the Canadian Environmental Quality Guidelines for Aquatic Life, the minimum DO that cold-water systems should have in order to support early life stages of aquatic life is 9.5 mg/L and 6.0 mg/L applies to all other life stages. An ideal stream habitat will also provide an abundant supply of the required mineral ions Calcium (Ca^{2+}) and Magnesium (Mg^{2+}) (Smith 1962). Vital for development, their presence is usually the product of the local geology and plays role in controlling the Acid Neutralizing Capabilities (ANC) of the water as well as influencing water hardness levels. Findings from Teears (2016), suggest that high ANC of water results in increased growth rates and survival trends in young brook trout.

Brook trout are hardy compared to many other salmonids. Moderately low pH can be tolerated by brook trout but, increased H+ concentrations associated with low pH can interfere with gill function and provoke excess sodium loss (Packer and Dunson 1972). To avoid dehydration and salt-depletion, freshwater fish must maintain a hyperosmotic state compared to their external environment. Excess sodium loss can stress the balance between water intake (via gills) and the simultaneous release of hypotonic solution (urine) into the water. When exposed to low pH (2.00), brook trout suffered extreme sodium losses prior to death (Packer and Dunson 1972). Other studies have reported that <5.0 pH has resulted in fewer viable eggs produced in test hatcheries (Menendez 1972). Similarly, values below 6.5 pH often result in decreased hatching success and post hatch survival (Teears 2016). The optimal pH range identified by the U.S. Fish and Wildlife Service for brook trout is 6.5-8.0 (Schowalter-Hay 2019). Values below 6.5 pH can allow heavy metals in the water to increase in toxicity.

The general habitat requirements for brook trout as defined by Kerr (2000), in the OMNRF brook trout stocking manual are as follows: The pH should be >6.0, dissolved oxygen should be greater than 5.0 mg/L, water clarity is optimal at 0-30 JTU, and temperatures below 20°C sufficiently meet the temperature requirements of brook trout. A review of literature on the effects of Total Dissolved Solids (TDS) by Scannell and Jacobs (2001) found that various fish species, including brook trout, were most sensitive to TDS exposure during fertilization and egg hardening life stages. TDS concentrations around 750 mg/L are identified as having significant impacts on the fertilization rates of various fish species. After egg hardening, however, the concentrations of TDS tolerable by fish increases. The range identified for significant adverse effects on benthic invertebrates sits between 1692 to 2430 mg/L.

Heavy Metals

The presence of heavy metals in the water can also affect the health of freshwater fish. Most often, they occur as a product of the natural bedrock. However, mining activity is known to cause increased levels of metal contaminants as ore, tailings and mine water are brought to the surface. The survival of brook trout in an environment influenced by mining depends largely on the concentration of metals within the system as well as some associative water quality parameters (Pascoe et al. 1986).

Cadmium

Cadmium is not biologically essential and can be extremely toxic to brook trout at low concentrations (Pascoe et al.1986). Cadmium toxicity has been identified as an inverse reflection of water hardness; many publications state that (Ca^{+2}) and (Mg^{+2}) associated with increased water hardness has antagonistic values that helps to decrease Cadmium toxicity (Besser and Leib 2007). A toxicity test using Rainbow trout found that the 96-hour Lethal Concentration at 50% (LC50) threshold in soft water was 1.3 mg/L Cd, which is doubled in harder water (2.6 mg/L Cd; Pascoe et al. 1986). The Cadmium limits and corresponding hardness levels defined by the Canadian Water Quality Guidelines (CWQG) for aquatic life and are featured below (Table 1).

Table 1. CWQD Guidelines for Freshwater Aquatic Life: Cadmium Concentration vs Hardness (CCME 1999).

Hardness (mg/L as $CaCO_3$)	Cadmium (μg/L)
0-60	0.2
60-120	0.8
120-180	1.3
>180	1.8

Copper

The Lowest Observed Effect Concentration (LOEC) for copper on brook trout survival in the Animas watershed in Colorado and New Mexico (a location historically affected by mining activity) is 25 μg/L and the LOEC for growth and biomass is 6.3 μg/L (Besser and Leib 2007). However, the CWQG threshold for copper is 2 μg/L at unknown hardness levels and varies between 0-82 mg/L for water with low hardness levels (CCME 1999). The copper limits and corresponding hardness levels defined by the CWQG for aquatic life and are featured below (Table 2).

Table 2: Canadian Water Quality Guidelines for Freshwater Aquatic Life: Copper Concentration vs Hardness (CCME 1999).

Hardness (mg/L as CaCO3)	Copper (µg/L)
0 – 60	2
60-120	2
120-180	3
>180	4

Zinc

The LOEC identified by Besser and Leib (2007) is >2000 µg/L zinc for brook trout survival and biomass, and 1000 µg/L for brook trout growth. These values were produced from water with 113 mg/L hardness as $CaCo_3$. The toxicity of zinc to salmonid species is identified Chapman (1978) as 93-97 µg/L in soft waters.

Nickel

The allowable nickel concentrations for freshwater aquatic life as identified by the CDWG are below (Table 3).

Lead

The limits on lead concentrations and corresponding hardness levels defined by the CWQG for aquatic life and are featured below (Table 4).

Table 3. Canadian Water Quality Guidelines for Freshwater Aquatic Life: nickel concentration vs hardness (CCME 1999).

Hardness (mg/L as $CaCO_3$)	Nickel (µg/L)
0-60	25
60-120	65
120-180	110
>180	150

Table 4. Canadian Water Quality Guidelines for freshwater Aquatic Life: lead concentration vs hardness (CCME 1999).

Hardness (mg/L as CaCO3)	Lead (µg/L)
0-60	1
60-120	2
120-180	4
>180	7

Aluminum

Aluminum can have adverse effects on aquatic life in streams with lower pH levels as acidity has a direct correlation with the element's solubility and toxicity. Higher pH values also lower the toxicity of present aluminum (Besser and Leib 2007). The aluminum limits and corresponding pH levels defined in the CWQG for aquatic life and are featured below (Table 5).

Table 5. Canadian Water Quality Guidelines for Freshwater Aquatic Life: aluminum concentration vs pH (CCME 1999).

pH	Aluminum (mg/L)	Additional Parameters
<6.5	0.005	<4 mg/L Ca +2; <2.0 mg/L DO
>6.5	0.1	>4 mg/L Ca +2; >2.0 mg/L DO

It is noteworthy that the CWQG are intended to encompass many species less tolerant than brook trout. They are a reference for determining the general health of an aquatic environment. However, the survival capabilities of brook trout can be considered beyond these guidelines. Besser and Leib (2007) define the threshold for acute effects to brook trout as 1.8 mg/L aluminum at >6.5 pH.

<u>Iron</u>

The rationale for the CWQG reports a 50% reduction in hatchling success from brook trout eggs at 12 mg/L iron concentrations. The guidelines state that the concentration of total iron in aquatic system should not exceed 0.3 mg/L based on varying tolerances of aquatic life (CCME 1999).

Physical Habitat Requirements

The amount of in stream coverage available for refuge can also limit the success of brook trout survival. Sufficient coverage including boulders, large woody debris, pools and undercut banks are essential for brook survival as they provide shade, further cooling stream water and providing adequate protection

(Raleigh 1982). The Community Stream Steward program (2018) describes ideal trout habitat as having a meandering stream channel, well-defined pools and riffles, healthy vegetation, assorted stream bed materials, and large woody debris.

Food Preferences

Brook trout are opportunistic feeders and will select their diet based on food abundance and availability. Insects (both aquatic and terrestrial) are not limited to but noted by many as the most common choice of prey by brook trout. Other prey commonly selected for includes snails, crayfish and leeches (Kerr 2000).

Thesis Objective

The overall objective of this thesis was to determine if data collected over a period of nearly 40 years could be used to estimate whether that brook trout could be re-introduced into a watershed that previously had a self-sustaining population of this species prior to mining activities. A closed copper/zinc mine was used as the example.

2. ESTIMATING THE FEASIBILITY OF BROOK TROUT REESTABLISHMENT AT THE WINSTON LAKE MINE

The Winston Lake Mine operated from 1988-1999 within the Whitesand River watershed near Schreiber, Ontario. Operations at the mine site resulted in discharges that greatly impacted water quality. Mine water released into the system eventually caused Cleaver Lake (the receiving water body) to enter a period of meromictic stagnation. The loss of turnover in the lake resulted in a highly anoxic hypolimnion layer with elevated concentrations of cations and anions.

Cleaver Lake and segments of the watershed receiving outflow have since ceased to support the same degree of aquatic biota that it supported historically. Thus far, remedial efforts on the lake itself have followed a passive approach, without any anthropogenic influence in the remediation process. Remaining infrastructure at the mine site includes a tailings pond and water treatment system that uses lime amendment and carbon dioxide to precipitate metals and alter the waters pH levels (Ecometrix 2019). Water is discharged from the treatment system on site and directed into the Whitesand River above Cleaver lake (Figure 1). During October 1983, levels of zinc and copper were theoretically high enough to cause acute mortality in any fish species present directly below the tailings discharge point at the time (Beak 1984). Concentrations of cations and anions are now reduced, and, as a result, natural fall and winter turnovers occur in Cleaver Lake (Pun 2020). Fish community assessments have shown that white sucker, lake chub and pearl dace are now present in the system (Stantec 2018).

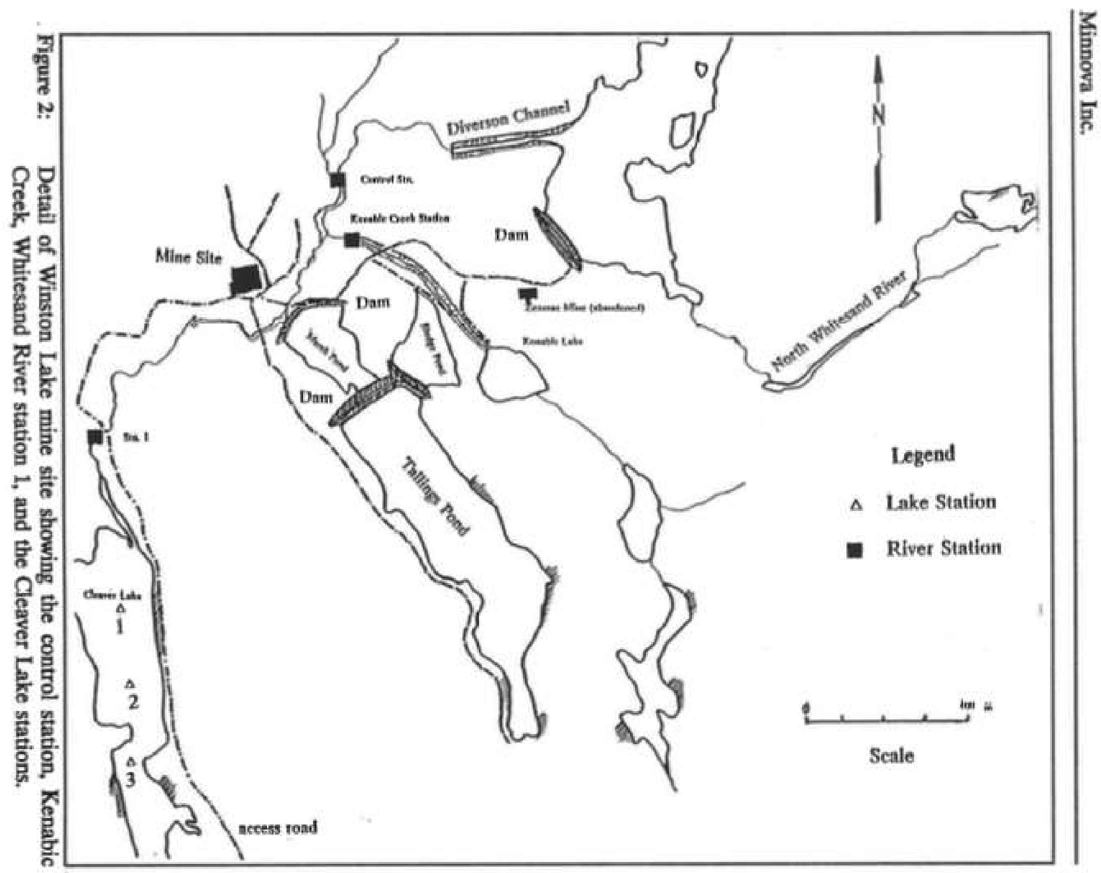

Figure 1. Winston Lake mine site tailings pond location relative to the Whitesand River (Beak 1984).

The rehabilitation goal is to repopulate the upper Whitesand River with brook trout stock, when conditions are favorable in the watershed for their survival. There have been reports of brook trout in segments of the river from Hornblende Lake towards Whitesand Lake (Appendix II; XIV), and below the discharge point, there are a few steep gradients and waterfalls along the river that would require a large flow event to support upward fish migration (Appendix III). It is unconfirmed whether the lack of brook trout in the river above these points is due to the absence of flow events allowing migration or the inability of the affected river segments to meet the habitat requirements for brook trout.

Recently completed research suggested passive remediation of Cleaver Lake has resulted in suitable habitat for brook trout (Pun 2020). However, it is also necessary for the downstream habitat in the Whitesand River to be acceptable, requiring knowledge on whether any intervention is needed to permit normal migration and spawning for brook trout.

Hypothesis

It is hypothesized that that the Whitesand River has recovered enough through passive restoration to support a population of brook trout without the need for human intervention.

Methods

The data for the analysis were provided by First Quantum Minerals. It included aquatic assessments from 2003-2018 of the Whitesand River by Stantec Consulting for recent data on the 1984 Winston Lake project, environmental studies by IEC Beak for historic data prior to mine operation, and reports compiled by Beak International Incorporated for the years 1983, 1991, 1998 and 1999.

For each of the aquatic assessments, field samples were collected at pre-determined sampling stations in early fall (Figure 2). Data from each of the documents was transferred into Excel tables for each parameter by year and sampling location. Changes to water quality parameters were converted into graphic displays. Station numbers 3-8 were used for the analysis. The locations of each station are described below (Table 6).

Table 6. Sampling station locations and reference numbers on the Whitesand River, Ontario.

Location	Reference number
Cleaver Lake inlet	Station 3
Cleaver Lake outlet	Station 4
Gumboot Lake outlet	Station 5
Hornblende Lake outlet	Station 6
Lyne Lake inlet	Station 7
Ross creek (reference)	Station 8

Data collection on the Whitesand River occurred periodically over decades, between various reporting entities. Therefore, some adjustments were necessary for data analysis. In circumstances where multiple readings existed for the same location and time, the average of those values was used for statistical analysis. In circumstances where a given value was reported as “less than” a value by the reporting body, one unit below the given value or another value below the threshold was applied where required. For example, where a value was reported as being <1 mg/L, the value was assumed to be 0.9 mg/L for the purpose of statistical analysis. The only water hardness reading for the 2018 data was reported for the Cleaver lake inlet. In the absence of hardness values for the other stations, an equation developed from a regression analysis for magnesium concentrations and water hardness from the 1983 report was used to develop a set of estimates of hardness at each of the sampling stations in

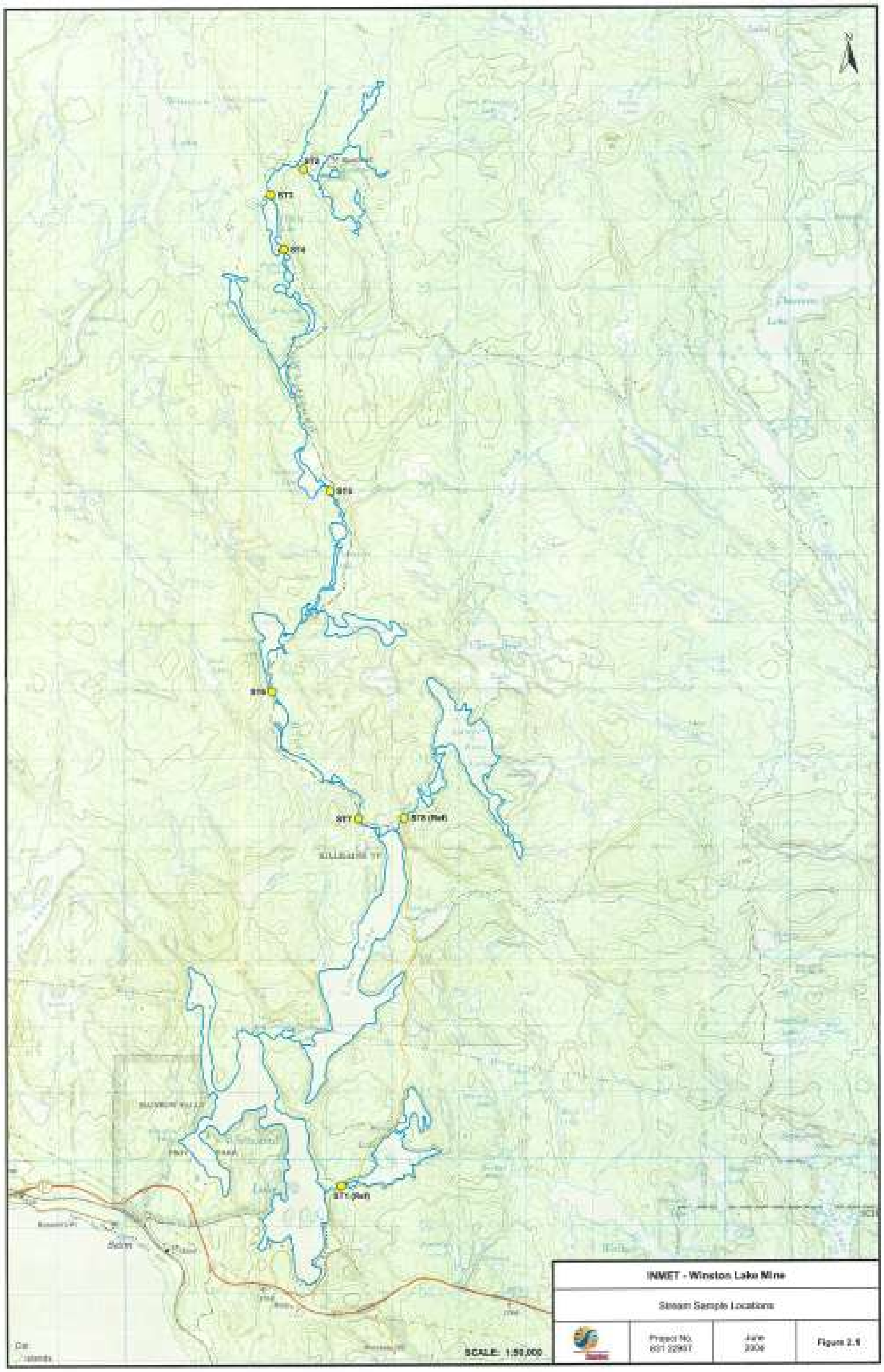

Figure 2. Map of Whitesand River sampling stations (Stantec 2003).

2018. A second regression analysis, between alkalinity and hardness values from the 1983 data was used to predict hardness values at each sampling station for 2018. The two sets of estimations were then averaged to generate a final set of predictions for hardness values in 2018. For the analysis of each of the metal concentrations and supporting water quality data, a paired t-test was calculated to compare data from 1983 and 2018 at the Cleaver Lake outlet, the Gumboot Lake outlet and Hornblende Lake outlet stations. The data analysis tool pack in Microsoft Excel was used for statistical analysis.

RESULTS

Water Quality

Zinc concentrations in the 1983 report ranged from 0.10 – 0.29 mg/L. In the 2018 report they ranged from 0.023-0.110 mg/L zinc. However, the Ross Creek reference sample from 1997 was 1.78 mg/L zinc. This value was noted by the reporting agency as a potential error (Figure 3).

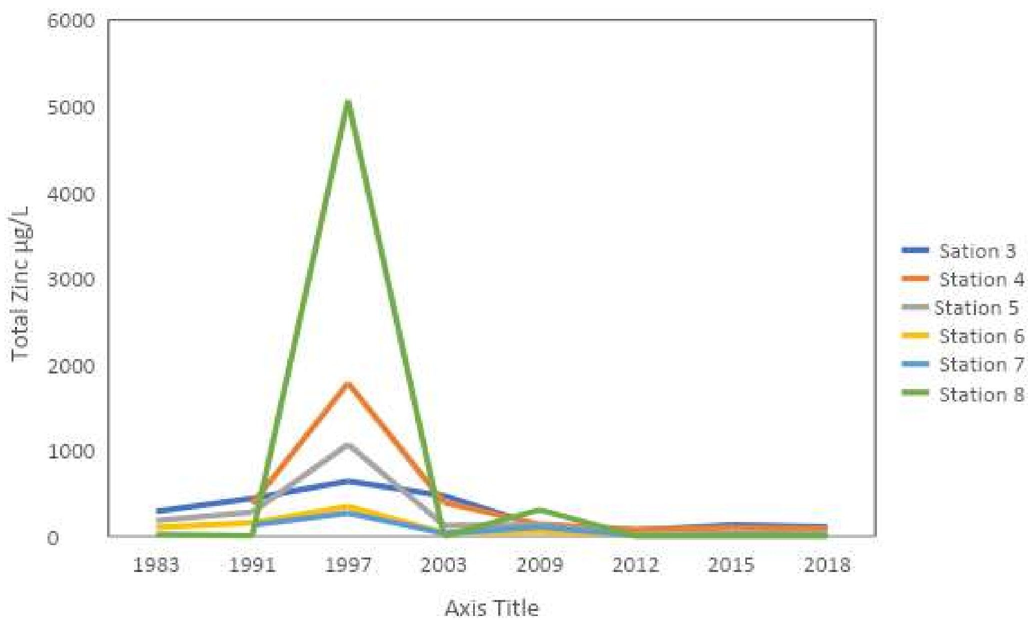

Figure 3. Total zinc concentrations (µg/L Zn) from stream sampling stations 1983-2018.

The same data converted to mg/L zinc with the suspected error removed is illustrated below (Figure 4). In 1998, prior to the opening of the Winston Lake Mine, the zinc concentration at the Cleaver Lake Inlet was 0.29 mg/L. In 2018 the same location had 0.11 mg/L Zn at the time of testing. A paired t-test of the zinc concentrations from 1983 and 2018 showed these initial and final zinc concentrations to be significantly different (P = 0.04).

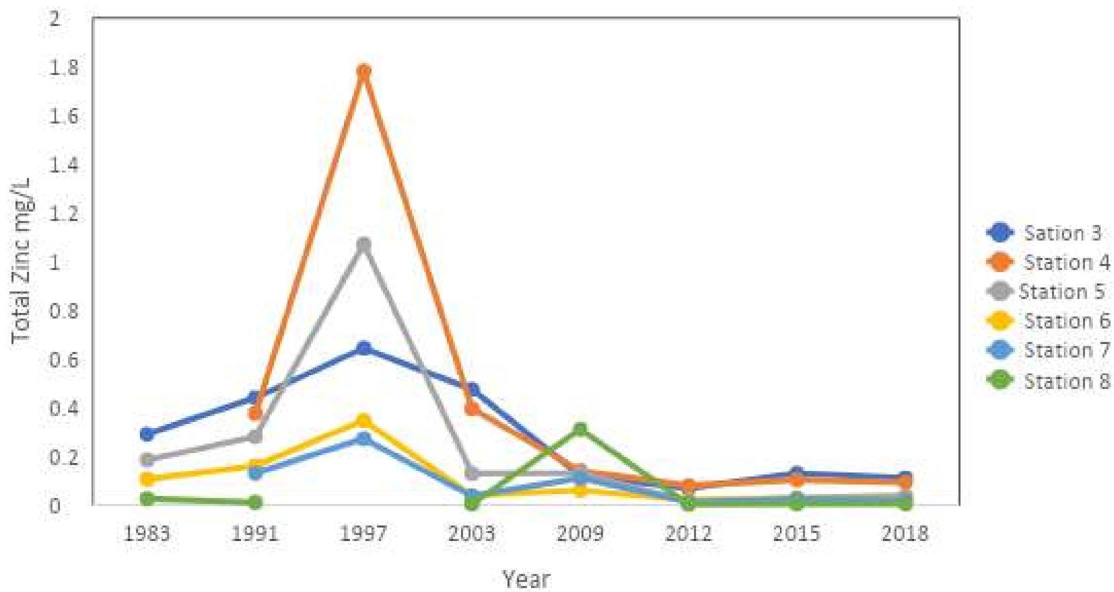

Figure 4. Total Zinc concentrations (mg/L Zn) from stream sampling stations 1983-2018. Suspected error removed.

The Cleaver Lake inlet (sampling station no. 3) samples contained 8.5 µg/L total copper in 1983 and 3.3 µg/L total copper in September 2018. The general trend in decline of total copper concentrations at all sampling stations is illustrated below (Figure 5). The paired t-test of copper concentrations from the 1983 and 2018 reports showed these initial and final copper concentrations to be significantly different (P = 0.005).

The highest recorded concentration of iron in the Whitesand River, 5.14 mg/L, was at the Cleaver lake inlet in 1997 (Figure 6). Iron dropped to 0.35 mg/L the following year. For an improved visual of the overall trends in iron concentrations, the 1997 spike in iron was removed in a second graph of iron

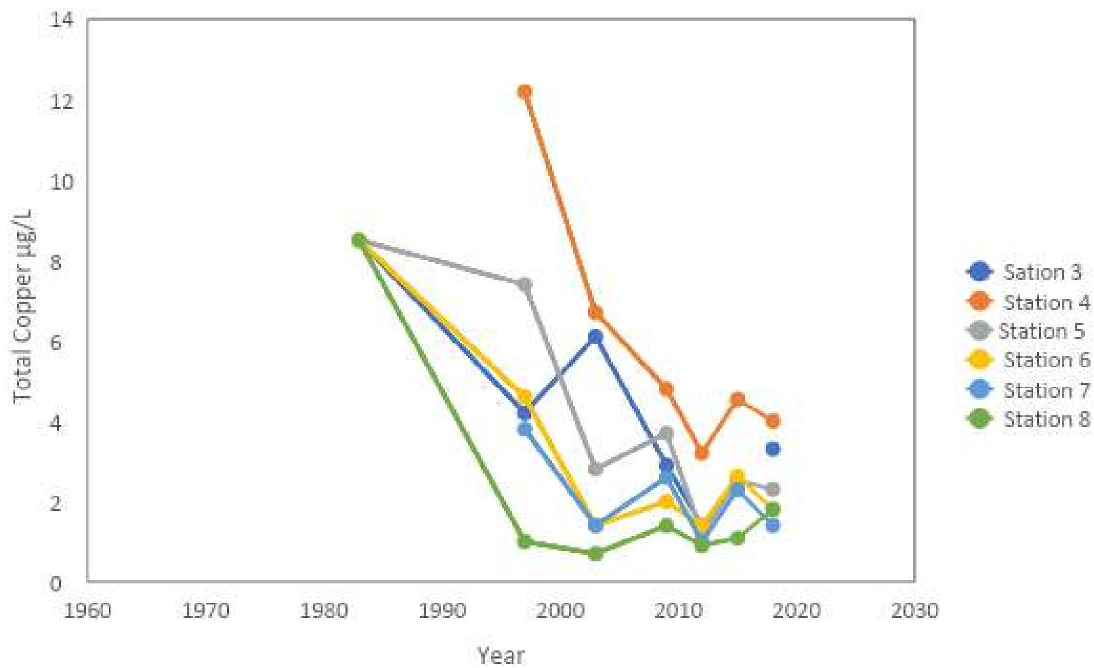

Figure 5. Total copper concentrations (µg/L Cu), from stream sampling stations 1983-2018.

concentrations (Figure 7) and concentrations were also converted to mg/L. Iron concentrations at the Cleaver Lake inlet station was reported as 0.28 mg/L in 1983 and 0.51 mg/L in 2018. All other sampling stations including the Ross Creek reference station experienced a return in iron concentrations that averaged ± 0.035 mg/L from the pre-operational or earliest known records. A paired t-test on the data for iron concentrations from 1983 and 2018 showed no significant difference between these years (P = 0.60).

The data for cadmium concentrations were often expressed as 'less than' particular values in most of the aquatic assessments (EcoMetrix 2009; 2012; 2015; 2018) and as 5 µg/L at each station in the 1983 environmental studies. In creating the graphic display for cadmium concentrations at each station (Figure 8), a value of 7 µg/L was given to each of the samples reported as <10 µg/L. In

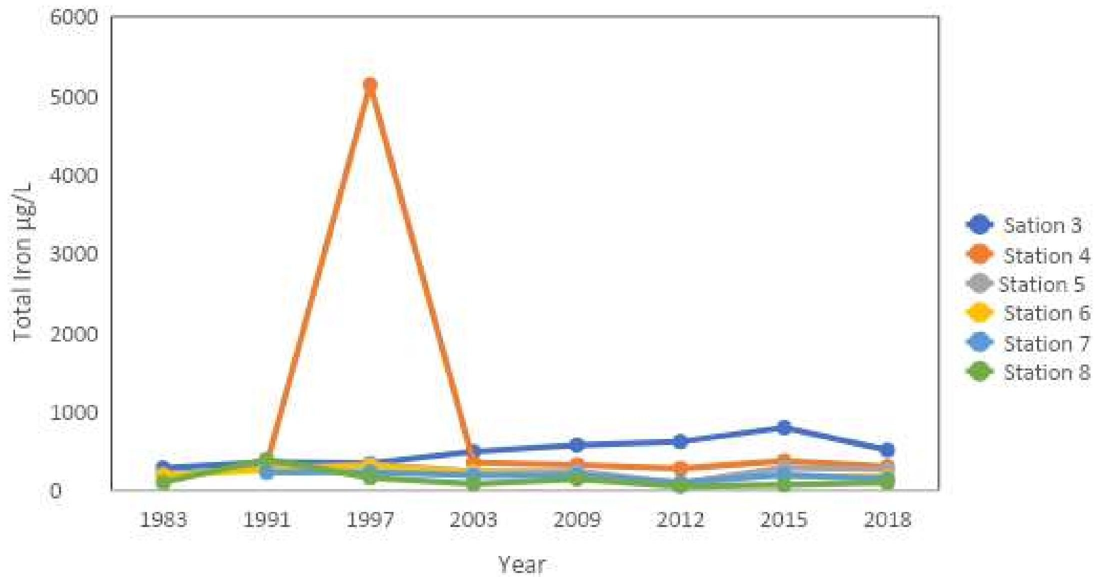

Figure 6. Total iron concentrations (µg/L Fe), from stream sampling stations 1983-2018.

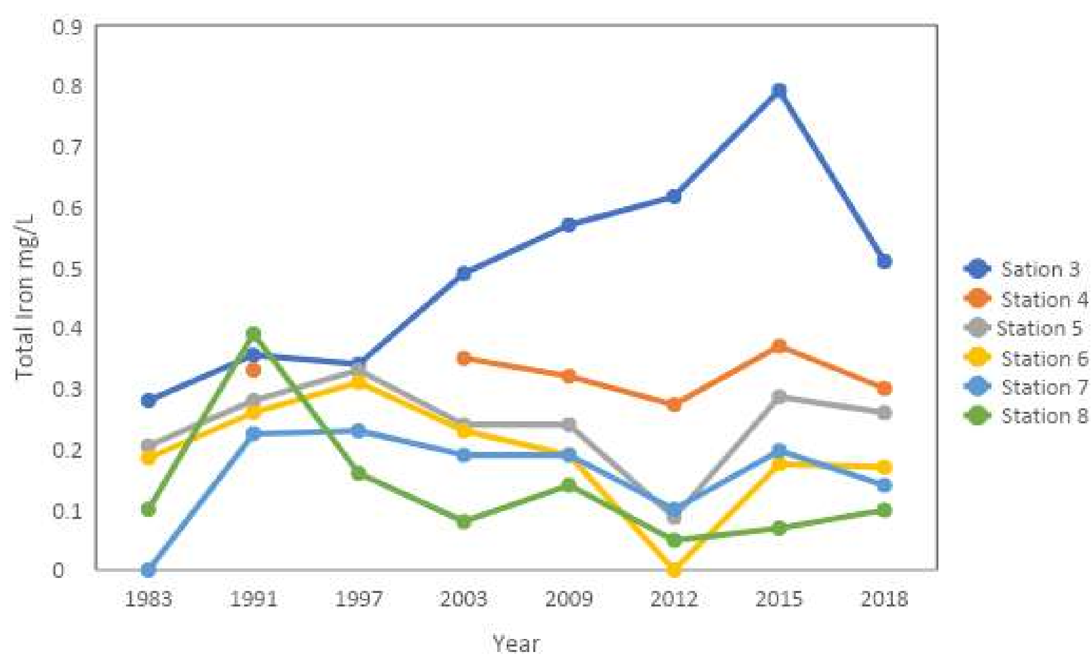

Figure 7. Total iron concentrations (mg/L Fe), from stream sampling stations (data spike removed).

1991, the samples for cadmium were individually reported and values ranged from between 3.00 and 0.09 μg/L total cadmium at each sampling station.

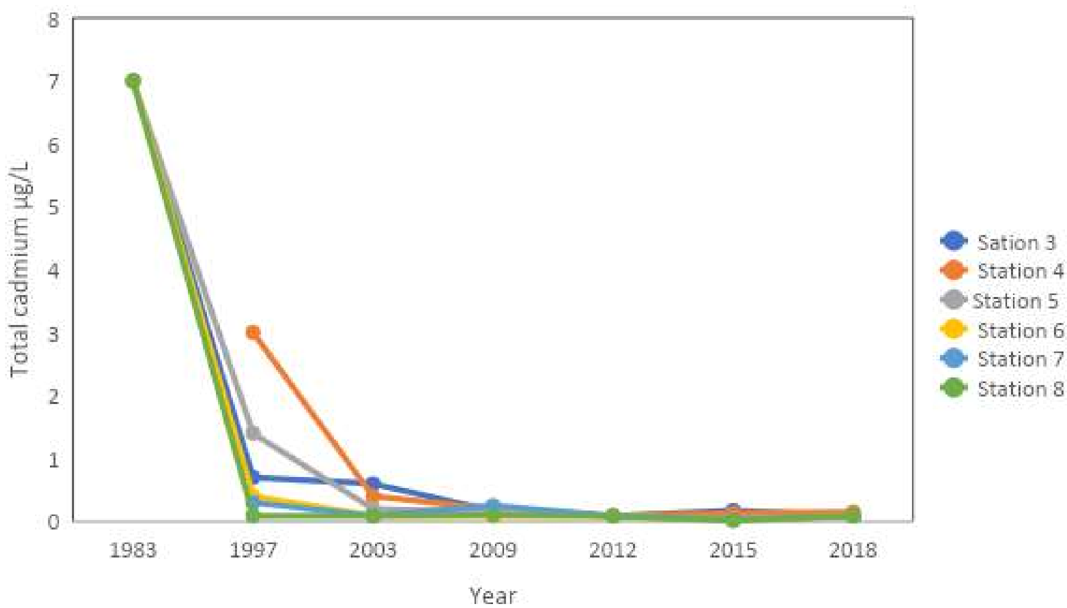

Figure 8. Total Cadmium concentrations (μg/L Cd), from stream sampling stations 1983-2018.

In the 1983 environmental studies, samples averaged 0.785 mg/L magnesium. In 2018, the average magnesium concentration for the Whitesand River was 1.008 mg/L Mg. The changes in the total magnesium concentrations are illustrated below (Figure 9). A paired t-test of the 1983 and 2018 magnesium concentrations showed no significant difference between these years ($P = 0.19$).

The average aluminum concentration in 1983 was 0.252 mg/L and in 2018 was 0.168 mg/L. Concentrations for total aluminum at each sampling station in 2018 were lower than the earliest known concentrations for each segment of the river (Figure 10), but by paired t-test the differences between 1983 and 2018 were not significant ($P = 0.09$).

The recorded values for pH from 1983–2018 are displayed in Figure 11. In 2018, the highest pH value of 7.37 was recorded at the Cleaver Lake Inlet. The pH values for all samples that year ranged from between 6.63-7.37. A paired t-test comparing 1983 and 2018 showed that these years were not significantly different in pH ($P = 0.09$). Water hardness values, alkalinity, and metal concentrations for each station are displayed in Figure 12.

A regression analysis of the 1983 data points between the magnesium concentrations and water hardness at each sampling station returned the equation $y = 12.7x + 0.063$ with $R^2 = 0.99$ (Figure 13). The estimated water

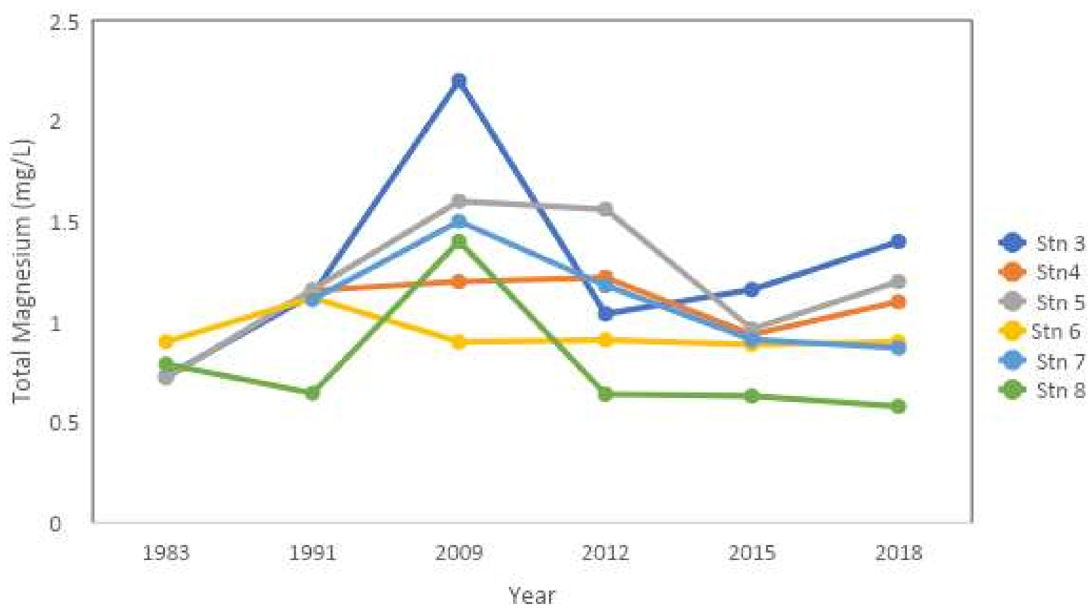

Figure 9. Total magnesium concentrations (mg/L Mg), from stream sampling stations 1983-2018.

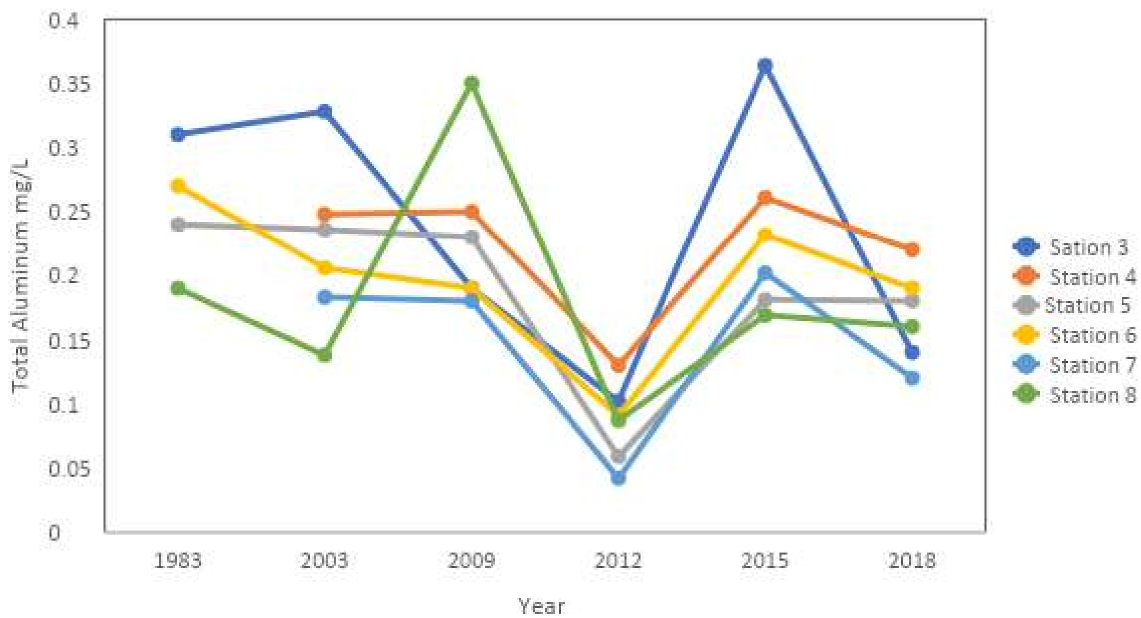

Figure 10. Total aluminum concentrations (mg/L Al), from stream sampling stations 1983-2018.

Hardness values calculated from the magnesium and hardness regression equation for 2018 are 18.5 mg/L $CaCO_3$ at Station 3; 14.7 mg/L $CaCO_3$ at station 4; 15.9 mg/L $CaCO_3$ at station 5; 12.1 mg/L $CaCO_3$ at station 6; 11.7 mg/L $CaCO_3$ at station 7. A second regression, using alkalinity and water hardness from 1983 produced the following equation, y= 12.7x + 0.63 (Figure 14). The estimated water hardness values calculated from the alkalinity and hardness regression equation are 239.98 mg/L $CaCO_3$ at station 3; 133.39 mg/L $CaCO_3$ at station 4; 123.70 mg/L $CaCO_3$ at station 5; 91.72 mg/L $CaCO_3$ at station 6; 103.35 mg/L $CaCO_3$ at station 7. The final estimations (from the averages of the regression estimations) are as follows: 129.2 mg/L $CaCO_3$ at station 3; 74.0 mg/L $CaCO_3$ at station 4; 69.8 mg/L $CaCO_3$ at station 5; 51.9 mg/L $CaCO_3$ at station 6; and 57.5 mg/L $CaCO_3$ at station 7. The final estimations of water

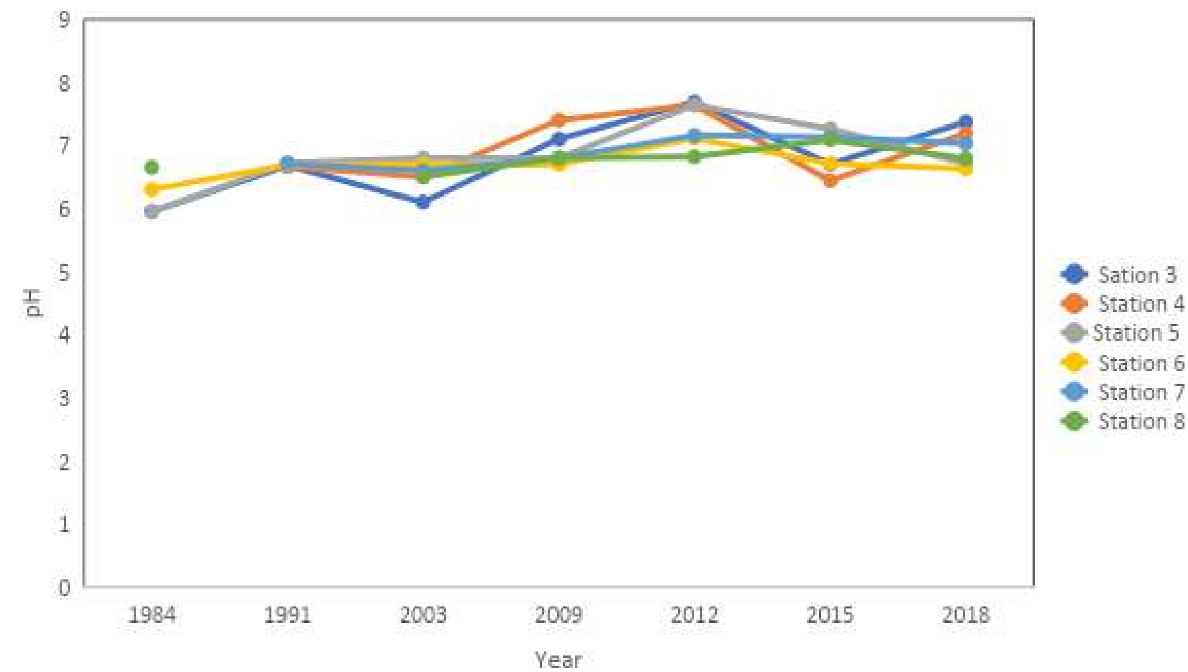

Figure 11. pH from stream sampling stations 1983-2018.

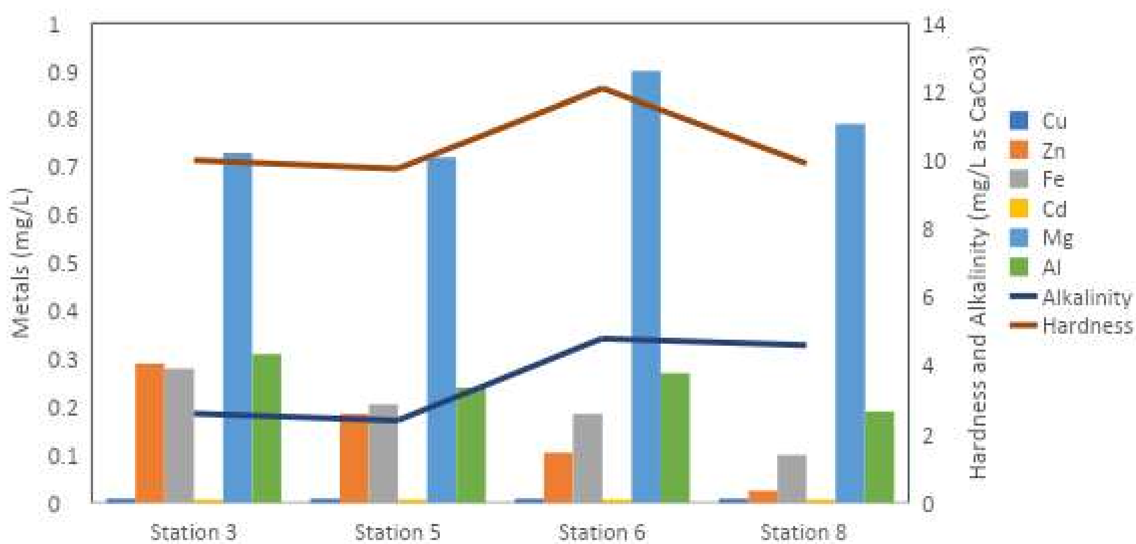

Figure 12. 1983 Total metal concentrations mg/L and alkalinity and hardness as $CaCO_3$.

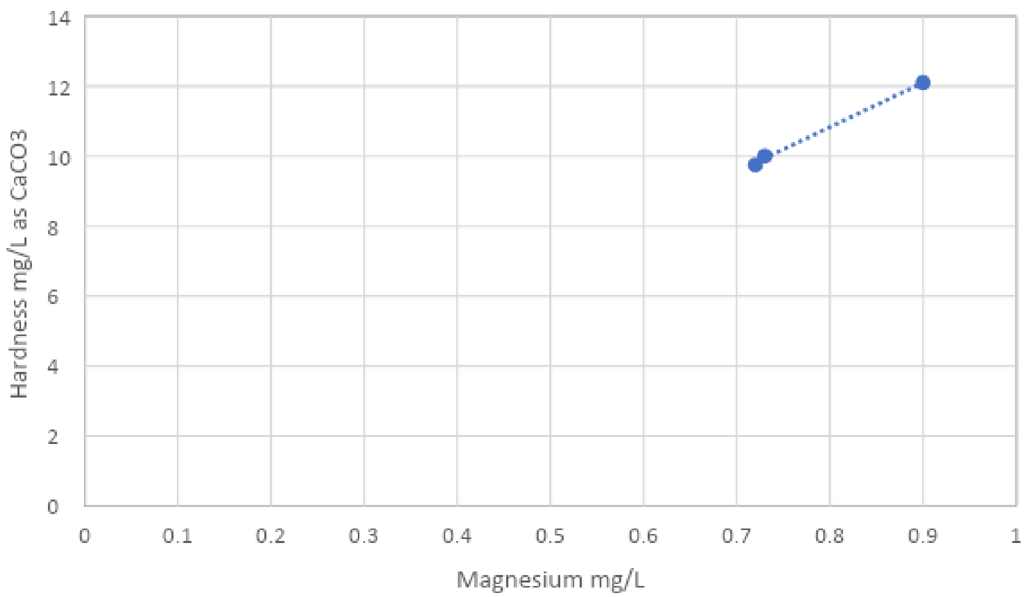

Figure 13. Total magnesium and water hardness regression from 1983 data.

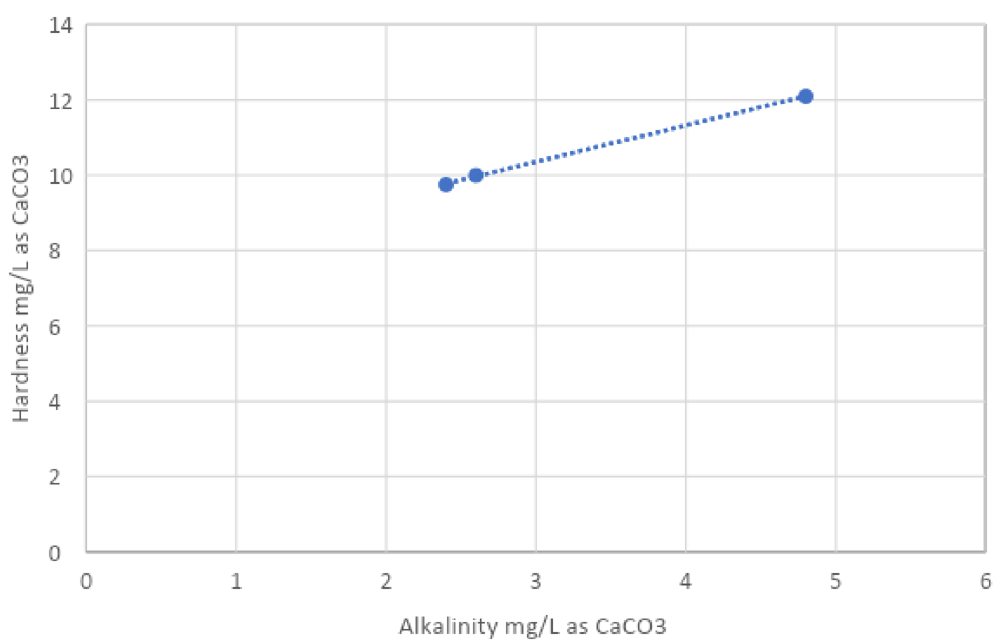

Figure 14. Alkalinity and water hardness regression from 1983 data.

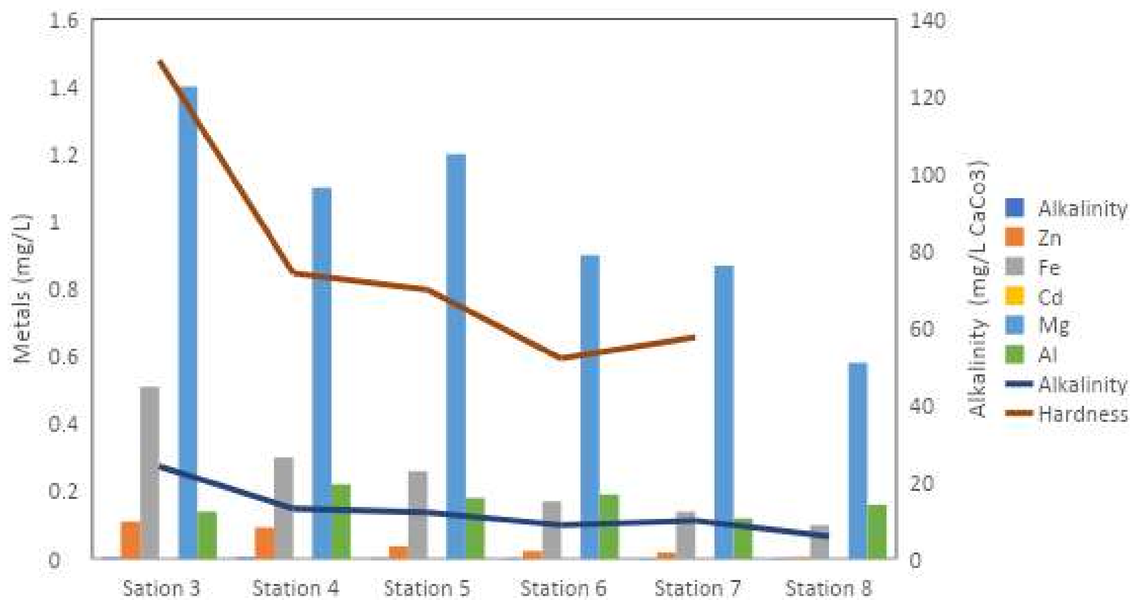

Figure 15. 2018 total metal concentrations, alkalinity mg/L as $CaCO_3$ mg/L and estimated water hardness as $CaCO_3$ mg/L.

hardness were plotted with alkalinity and metal concentrations below (Figure 15).

The changes in TDS concentrations over time, as well as known concentrations of sulphate at each station are displayed in Figure 16. Paired t-tests between 1983 and 2018 for the differences in TDS showed they were not significant (P = 0.97) nor for sulphate (P = 0.23). The dissolved oxygen (DO) content reported in 2018 ranged between 82-99% (Figure 17).

Temperatures recorded in September of 2018 are as follows: station 3, 17.2 °C; station 4, 18.4 °C; station 5, 20.4 °C; station 6, 18.1 °C; station 7, 18.8 °C; and 18.5 °C at station 8 (Figure 18). The dissolved oxygen values for 2018 are displayed with corresponding temperature data below (Figure 19).

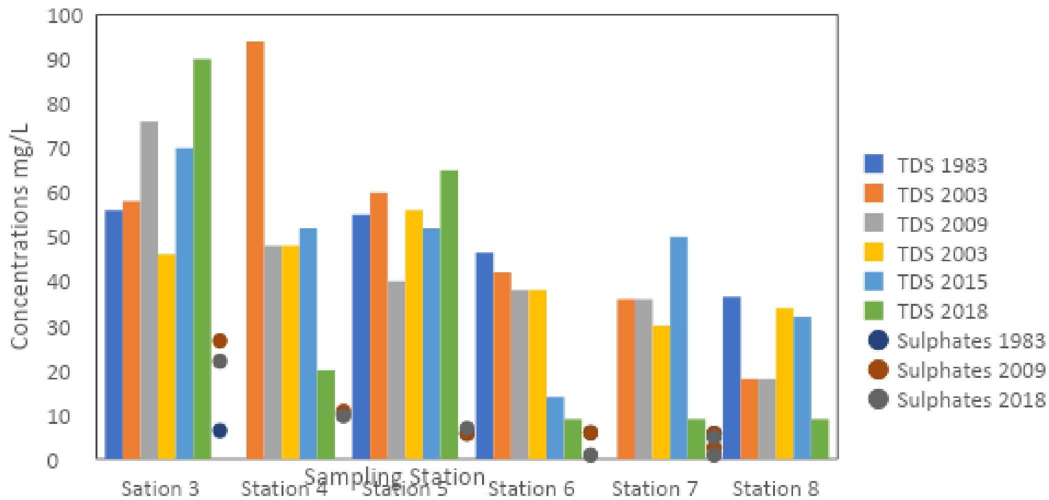

Figure 16. Total dissolved solids and sulphate concentrations for stream sampling stations 1983-2018.

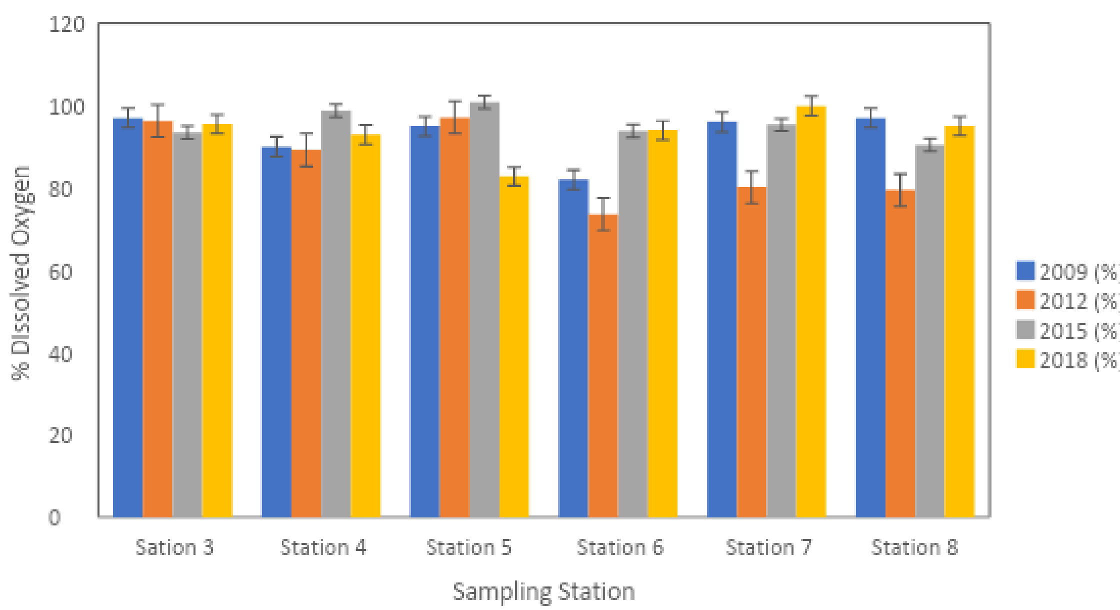

Figure 17. Percent dissolved oxygen at stream sampling stations 2009-2018.

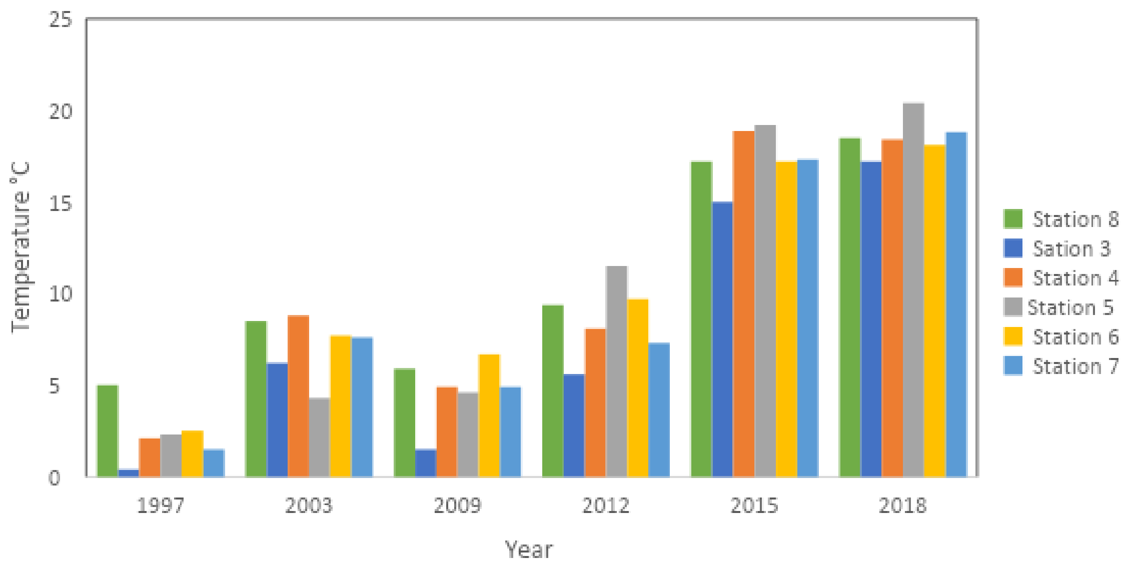

Figure 18. Fall temperatures from sampling stations 1997 to 2018.

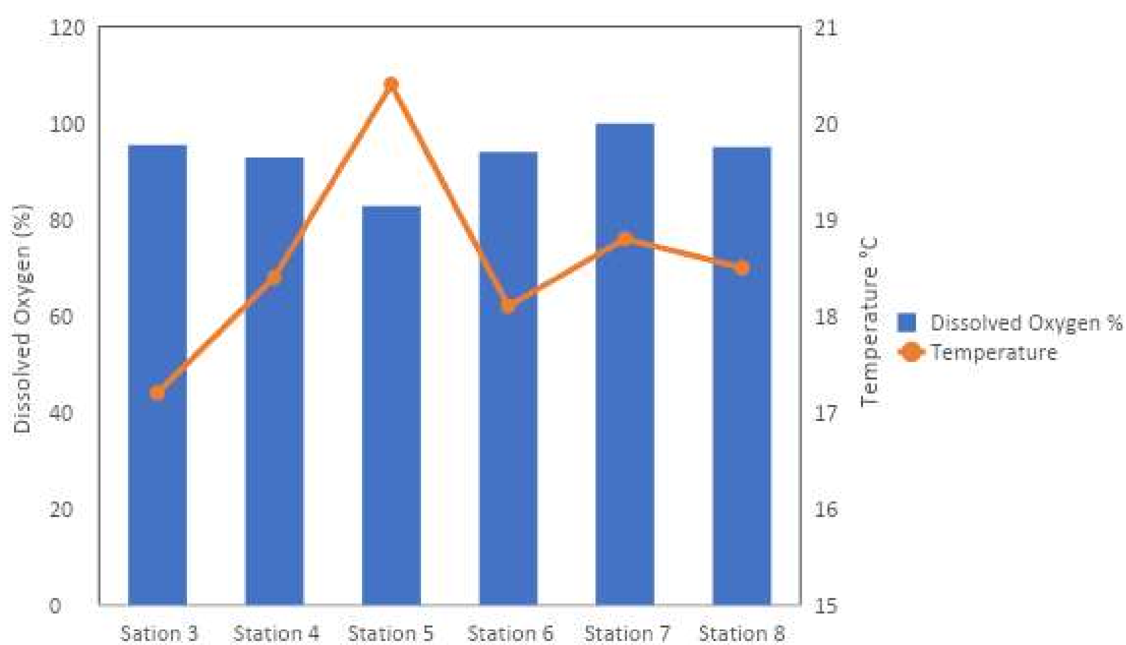

Figure 19. Percent dissolved oxygen and temperature 2018.

<u>Benthic Invertebrates</u>

Early environmental studies In the area indicated that a sample of invertebrates taken from above Cleaver Lake (relative to station 3) contained only a few caddisflies in 1984. Since site closure, EPT taxa have been present at all stations (Table 7).

Table 7. Number of EPT Index taxa identified for 1998, 2006, 2012 and 2018.

Year	Station 3	Station 4	Station 5	Station 6	Station 7	Station 8
1998	12	7	17	20	20	17
2006	16	12	18	23	24	14
2012	19	16	19	20	15	17
2018	19	19	17	22	22	16

The ANOVA for the number of EPT taxa comparing 1988 (the baseline study), 2006, 2012 and 2015 illustrated that there were no significant differences between years (P = 0.44).

DISCUSSION

Before and After Analysis

Overall, the B/A analysis showed that conditions in 1983 were similar to those in 2018. The mean pH along the river was 6.21 in 1983, which represents conditions of the river prior to operations at the Winston Lake mine. During operations, and following the decommissioning of the site, the highest average pH reported was 7.3 and the lowest reported pH was 6.1. Cadmium, copper and zinc concentrations declined after production ceased at the mine site. The levels of zinc in the river increased during mine site production. Samples taken in 1997 had the highest average zinc concentrations for all water stations.

Assessment of Metal Toxicity

Metal toxicity is dependent on hardness of water. Hardness was not always listed in the historical water quality reports, but regressions were able to correct for the missing information. The estimates of water hardness from the regression for alkalinity generated high values. The level of confidence indicated by the correlation coefficient is only applicable to estimates generated with alkalinity values that fall within the same range as in 1983. However, the known alkalinity measurements can also provide a ‘minimum’ value for water hardness where greater certainty is required surrounding the current conditions.

Cadmium concentrations and corresponding hardness values fall within the limits defined by the Canadian Water Quality Guidelines (CWQG) for aquatic life. The values for opper at Stations 4 and 5 are beyond the allowable concentrations defined in the CWQG for unknown or ‘low’ hardness values between 0 and 82 mg/L $CaCO_3$. Copper concentrations at the stations located downstream of Hornblende Lake are under maximum allowable concentrations of 0.2 μg/L within hardness values of unknown limits, ranging to 60 mg/L $CaCO_3$. The estimated value for water hardness at the Cleaver Lake inlet was the highest, and copper concentrations still fell within the allowable guidelines given the correction was appropriate. However, copper concentrations are higher than the threshold identified in the CWQG although none of the stations exceeded the LOECs defined in the review of literature for effects on growth and biomass of brook trout (6.3 μg/L copper). In addition, exceeding the CWQG for copper will not likely impact brook trout survival.

Aluminum concentrations were above the threshold defined in the CWQG for pH concentrations greater than 6.5. All stations with the exception of station 7 exceed the threshold of 0.12 mg/L aluminum at >6.5 pH identified for chronic effects on brook trout. However, no samples from 2018 exceeded the threshold of 1.8 mg/L at >6.5 pH for acute effects to brook trout. Concentrations rangde from 0.12-0.22 mg/L aluminum and could potentially affect growth and biomass of brook trout. It is not expected that aluminum concentrations will impact brook trout survival if the pH remains above 6.5.

In 2018, zinc concentration at station 4 was 1 μg/L and below the threshold identified by Chapman (1978) for having acutely lethal effects on salmonid species in soft waters. Zinc concentrations were lower at stations further downstream of the mine site and it is unlikely that elevated zinc at the Cleaver Lake outlet will cause adverse effects to brook trout. Iron concentrations in the river exceeded the limits defined in the CWQG, especially at stations close to the mine site. Values exceeding the CWQG are still not expected to affect brook trout success based on the LOEC effects identified in the literature review.

Supporting Water Quality Data

The dissolved oxygen at each sampling station is high and has not experienced any significant changes between 2009-2018. The oxygen content along the river is within the requirements defined in the literature to support brook trout populations.

Water temperatures in the river have increased over the past decade. Continuation of thermal increases in the stream, or temperatures remaining above 20°C, would adversely impact the suitability of the Whitesand River as brook trout habitat. Factors including time of day, season, weather, and various other influences may contribute to fluctuations in temperature. Elevated temperatures in early fall hint of seasonal restrictions in the amount of preferable or optimal brook trout habitat available in the river. However, no thermal records on the Whitesand River are above the threshold for brook trout survival.

Sulphate concentrations that were initially a high percentage of TDS in years during mining operation are now low and should not impact brook trout survival. Even if anoxic conditions develop in the hypolimnion, it is probable that iron in the water column will precipitate sulphates as iron sulphide.

Benthic Invertebrate Analysis

The return of pollution intolerant taxa and their persistence in the Whitesand River suggests that the benthic community is capable of supporting a brook trout population similar the community prior to mine operations.

CONCLUSION

Between 1983 and 2018, pH and specific conductivity have remained statistically the same in the White Sand River. Cadmium, copper and zinc values have all decreased from the impacted values and should not adversely affect brook trout survival. Zinc concentrations in the river are lower than the LOECs for chronic toxicity from the review of literature. Therefore, it is unlikely that zinc concentrations will impact the brook trout survival. Aluminum concentrations from 2018 may impact brook trout growth and biomass. However, impacts on survival are unlikely given the river remains near pH 7. Exceeding the CWQG for aquatic life occurs for iron concentrations at the Cleaver Lake inlet and outlet; however, the dissipation of concentrations further downstream suggests, from the literature I have reviewed, that concentrations should not impact brook trout survival in the river.

The values for total dissolved solids from 1983 and 2018 are statistically similar. The literature I reviewed also suggests that TDS concentrations in the Whitesand River should not adversely impact brook trout survival. Dissolved oxygen requirements of brook trout are sufficiently met by the river and the current benthic community structure suggests sufficient food availability.

Temperature increases over the past few decades in the river suggest a decrease in the amount of optimal brook trout habitat, but they do not exceed the threshold for brook trout survival. However, if warming patterns persist in the region, a loss of brook trout habitat is probable. It is unclear with the available data (limited to early autumn) whether temperatures are remaining near to above the threshold for brook trout longevity during other seasons.

Trends in the data and the comparisons between 1983 and 2018 indicate many improvements in the water quality of the Whitesand River. Given the results of my analysis, the hypothesis that the river has recovered sufficiently to support a population of brook trout is supported. However, given the observed increases in water temperatures, it is not clear how far into the future the river will meet the thermal requirements of brook trout.

RECOMMENDATIONS

As 2018 was the first year where fall temperatures reached 20 °C in the Whitesand River, it would be valuable to monitor stream temperatures into the future to ensure that exceeding the thermal tolerances of brook trout does not occur or persist for prolonged periods of time. It may also be valuable to assess regional climatic trends for insights into the likelihood of the river remaining within the thermal requirements for brook trout survival and for the persistence of similar coldwater species in the future.

LITERATURE CITED

Adams, P., C. James, and C. Speas. 2008. Brook trout (*Salvelinus fontinalis*) Species and Conservation Assessment. Grand Mesa, Uncompahgre, and Gunnison National Forests. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/ stelprdb5199816.pdf. Accessed Feb. 19 2019.

B.A.R Environmental. 1992. Biological Monitoring of the Whitesand River. Unpublished report, October 1991.

Bash, J., C. Berman, and S. Bolton. 2001. Effects of Turbidity and Suspended Solids on Salmonids. Retrieved from https://www.wsdot.wa.gov/research/reports /fullreports/526.1.pdf.

Beak International Incorporated. 1999. Aquatic Environmental Conditions, Whitesand River System 1997. Unpublished report, BEAK Ref: 21000.1.

Beak International Incorporated. 1984. Winston Lake Project: Environmental Studies. Prepared for Corporation Falconbridge Copper. Project No. 2214.1.

Besser, J. M., and K.J. Leib. 2007. Environmental Effects of Historical Mining in the Animas ... Retrieved from https://pubs.usgs.gov/fs/2007/3051/pdf/FS07-3051_508.pdf. Accessed Mar. 16, 2019.

Canadian Council of Ministers of the Environment. 1999. Canadian water quality guidelines for the protection of aquatic life. In: Canadian Environmental Quality Guidelines. Unpublished report, Canadian Council of Ministers of the Environment, Winnipeg.

Chadwick, J. G., and S. D. McCormick. 2017. Upper thermal limits of growth in brook trout and their relationship to stress physiology. *Journal of Experimental Biology*, *220* (21), 3976-3987. doi:10.1242/jeb.161224.

Chapman, G. A. 1978. Toxicities of cadmium, copper, and zinc to four juvenile stages of chinook salmon and steelhead. *Transactions of the American Fisheries Society*, *107*(6), 841-847.

Curry, A. R., and W. Scott Macneill. 2004. Population level responses to sediment during early life in brook trout. *Journal of the North American Biological Society*, *23*(1), 140-150.

Department of Fisheries Board Canada. 2019. Brook Trout. Retrieved from https://www.ontario.ca/page/brook-trout Feb. 5, 2019

Government of Newfoundland and Labrador. 2018. Brook Trout. Retrieved from https://www.flr.gov.nl.ca/wildlife/all_species/animals/inland_fish/brook_trout.html. Accessed Feb. 6, 2019.

Kerr, S. J. 2000. Brook Trout Stocking: An annotated bibliography and literature review with an emphasis on Ontario waters. Fish and Wildlife Branch, Ontario Ministry of Natural Resources, Peterborough Ontario. Retrieved from https://mffp.gouv.qc.ca/ faune/peche/ensemencement/Pdf/omble-fontaine.pdf. Accessed Mar. 27, 2019.

Menendez, R. 1976. Chronic Effects of Reduced pH on brook trout *(Salvelinus fontinalis*). *Journal of the Fisheries Research Board of Canada*, *33*(1), 118-123.

OFAH. 2018. Community Stream Steward Program: Trout Habitat Enhancement. Retrieved from https://www.ofah.org/streamsteward/files/Resources/Trout%20Habitat%20Enhancement.pdf. Accessed Mar. 28, 2019.

Packer, R. K., and W. A. Dunson. 1972. Anoxia and sodium loss associated with the death of brook trout at low pH. *Comparative Biochemistry and Physiology Part A: Physiology*, *41*(1), 17-26.

Pascoe, D., S. A. Evans, and J. Woodworth. 1986. Heavy metal toxicity to fish and the influence of water hardness. *Archives of Environmental Contamination and Toxicology*, *15*(5), 481-487.

Pun, A. 2020. Utilization of hydrodynamic modeling in predicting water quality and brook trout habitat: Reclamation of a mine impacted Boreal Shield Lake. M.Sc. thesis, Lakehead University.

RAMP. (2018). Using Benthic Invertebrates as Bioindicators. Retrieved from http://www.ramp-alberta.org/river/ecology/life/invertebrates/using benthic invertebrates as bioindicators.aspx

Sandlos, J., and A. Keeling. 2015. Mining. The Canadian Encyclopedia, 04 March 2015, Historica Canada. https://www.thecanadianencyclopedia.ca/en/article/mining. Accessed Feb. 19, 2019.

Scannell, P. W., and L. L. Jacobs. 2001. Effects of total dissolved solids on aquatic organisms. Technical Report No. 01-06. Alaska Department of Fish and Game. https://www.adfg.alaska.gov/static/home/library/pdfs/habitat/01_06.pdf. Accessed Feb. 19, 2019.

Schowalter-Hay, E. 2019. The optimum pH level of water that trout can live in. Retrieved from https://www.trails.com/facts_41600_water-that-trout-can-live.html.

Smith, M. W. 1962. Establishing and Managing Artificial Trout Ponds in Maritime Provinces. Fisheries Research Board of Canada. Manuscript Report Series. 727. http://www.dfo-mpo.gc.ca/Library/59288.pdf. Accessed Feb. 15 2019.

Stantec Consulting Ltd. 2004. Whitesand River Aquatic Assessment 2003. Ref: 63122957.

Stantec Consulting Ltd. 2010. Whitesand River Aquatic Assessment 2009. Ref: 09-1664

Stantec Consulting Ltd. 2013. Whitesand River Aquatic Assessment 2012. Ref: 12-1964

Stantec Consulting Ltd. 2016. Whitesand River Aquatic Assessment 2015. Ref: 15-2185

Sweka, J. A., and Hartman, K. J. 2001. Effects of turbidity on prey consumption and growth in brook trout and implications for bioenergetics modeling. *Canadian Journal of Fisheries and Aquatic Sciences, 58*(2), 386-393.

Teears, T. 2016. The Effect of Water Quality on the Survival and Fitness of Brook Trout (Salvelinus fontinalis) Eggs, Alevins and Fry in Aquaculture and Deep Springs along the South River in Waynesboro, VA. MSc thesis, North Carolina State University.

Tiwary, R. K. 2000. Environmental impact of coal mining on water regime and its management. *Water, Air, and Soil Pollution*, *132*, 185-199.

Tripole, S., P. Gonzalez., A. Vallania., M. Garbagnati, and M. Mallea. 2006. Evaluation of the impact of acid mine drainage on the chemistry and the macrobenthos in the Carolina stream (San Luis – Argentina). *Environmental Monitoring and Assessment*, *114*, 377-389.

Udayabhanu, S. G., and B. Prasad. 2010. Studies on environmental impact of acid mine drainage generation and its treatment: An appraisal. *Indian Journal of Environmental Protection*, 30(11), 953-967.

Younger, P. L., and C. Wolkersdorfer. 2004. Mining impacts on the fresh water environment: Technical and managerial guidelines for catchment scale management. *Mine, Water, and the Environment*, *23*(S1).

APPENDICES

Map of the Whitesand River and associated lakes. (Beak 1984).

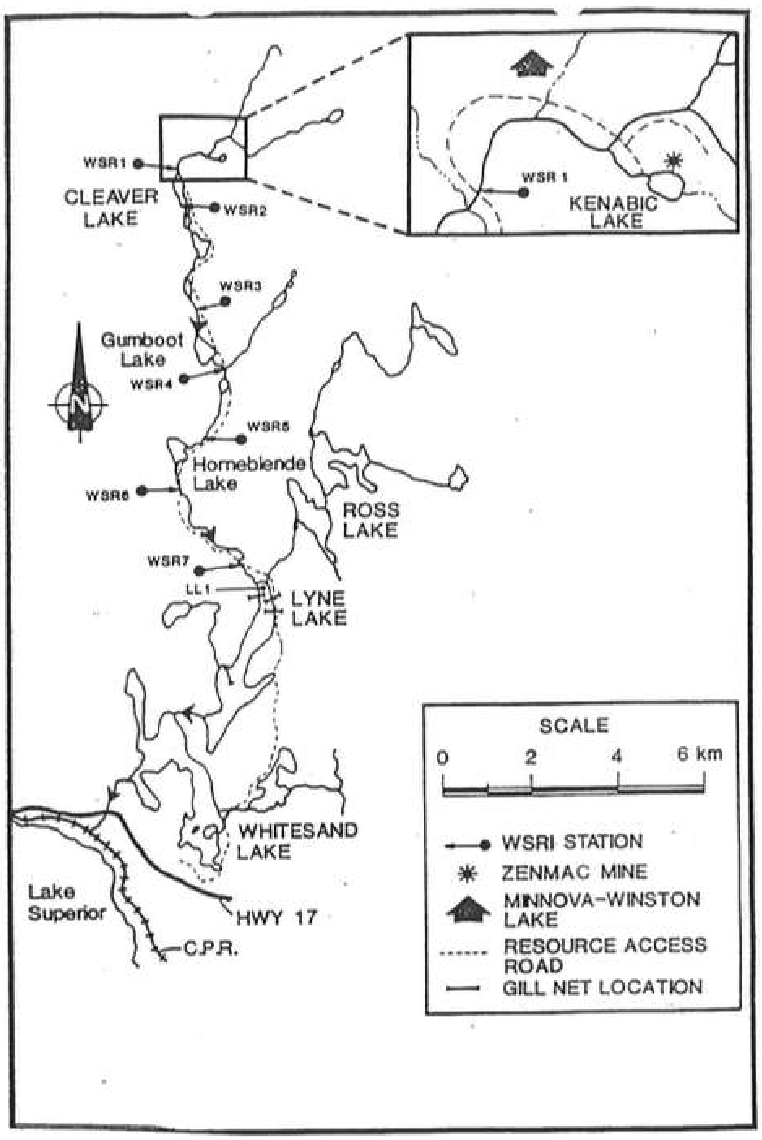

FIGURE 3.5.1 WHITESAND RIVER DETAIL AND SAMPLING LOCATIONS

EcoMetrix. 2015. Whitesand River aquatic assessment. Fish Community Assessment. Waterfalls. Possible barriers to upwards fish migration along the Whitesand River.

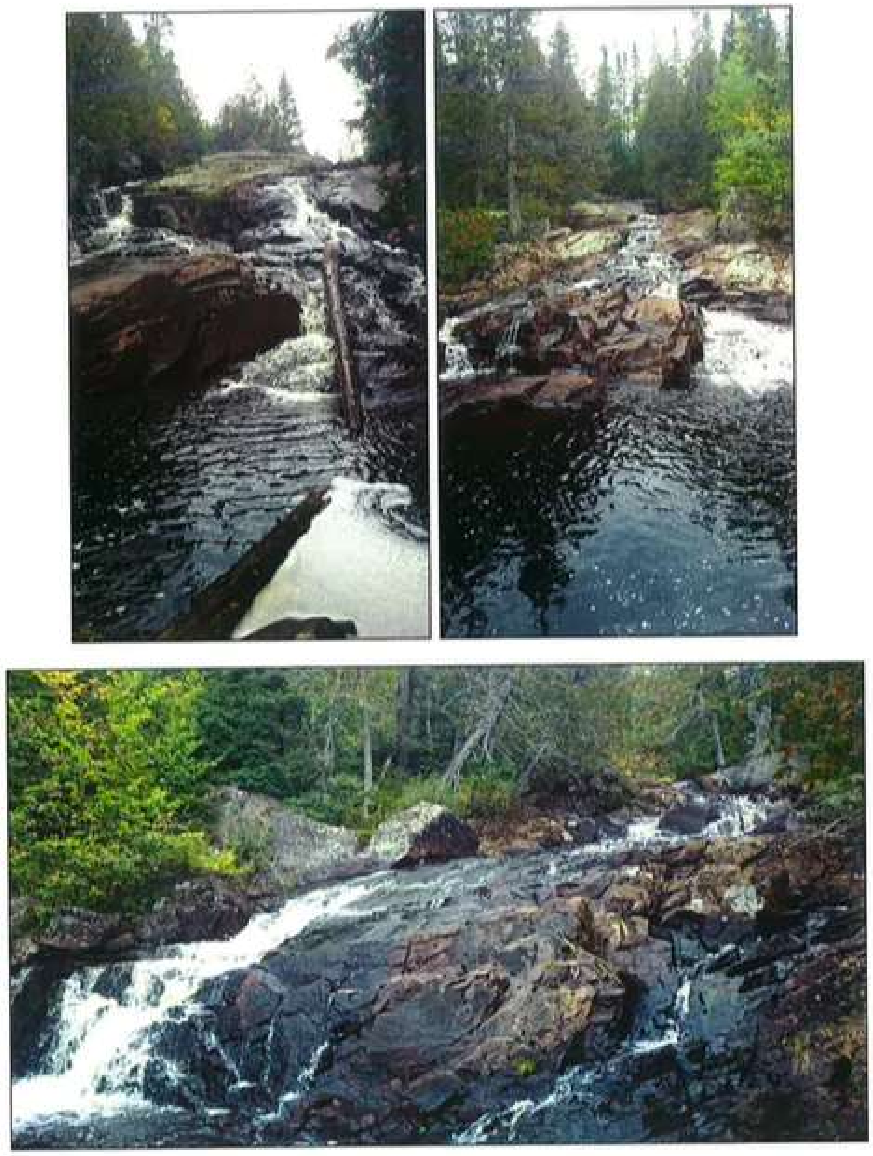

Photo 6.1: Top Left – Waterfall Upstream of Hornblende Lake; Top Right – Waterfall Upstream of Demijohn Lake; Bottom – Waterfall Downstream of Demijohn Lake

Statistical outputs for heavy metal comparisons in the Whitesand River.
(Total zinc) t-test: paired.

	1983	*2018*
Mean	0.19333333	0.05666667
Variance	0.00860833	0.00218233
Observations	3	3
Pearson Correlation	0.95670721	
Hypothesized Mean Difference	0	
df	2	
t Stat	4.73679917	
P(T<=t) one-tail	0.02089725	
t Critical one-tail	2.91998558	
P(T<=t) two-tail	0.0417945	
t Critical two-tail	4.30265273	

(Total copper) t-test: paired

	1983	*2018*
Mean	0.0085	0.00246667
Variance	0	5.8333E-07
Observations	3	3
Pearson Correlation	#DIV/0!	
Hypothesized Mean Difference	0	
df	2	
t Stat	13.6823139	
P(T<=t) one-tail	0.00264965	
t Critical one-tail	2.91998558	
P(T<=t) two-tail	0.00529929	
t Critical two-tail	4.30265273	

(Total Iron) t-test: paired

	1983	*2018*
Mean	0.245	0.276
Variance	0.0035125	0.02133
Observations	5	5
Pearson Correlation	0.57620804	
Hypothesized Mean Difference	0	
df	4	
t Stat	-0.5684964	
P(T<=t) one-tail	0.30004714	
t Critical one-tail	2.13184679	
P(T<=t) two-tail	0.60009428	
t Critical two-tail	2.77644511	

(Total magnesium) t-test: paired

	1983	*2018*
Mean	0.78333333	1.16666667
Variance	0.01023333	0.06333333
Observations	3	3
Pearson Correlation	-0.8969012	
Hypothesized Mean Difference	0	
df	2	
t Stat	-1.9228188	
P(T<=t) one-tail	0.09721211	
t Critical one-tail	2.91998558	
P(T<=t) two-tail	0.19442421	
t Critical two-tail	4.30265273	

(Total aluminum) t-test: paired

	1983	*2018*
Mean	0.27333333	0.17
Variance	0.00123333	0.0007
Observations	3	3
Pearson Correlation	-0.807183	
Hypothesized Mean Difference	0	
df	2	
t Stat	3.05452076	
P(T<=t) one-tail	0.04627128	
t Critical one-tail	2.91998558	
P(T<=t) two-tail	0.09254256	
t Critical two-tail	4.30265273	

Single factor anova of EPT taxa from 1998, 2006, 2012, 2018

Anova: Single Factor

SUMMARY

Groups	Count	Sum	Average	Variance
1998	6	93	15.5	25.9
2006	6	107	17.8333333	23.3666667
2012	6	106	17.6666667	3.86666667
2018	6	115	19.1666667	6.16666667

ANOVA

Source of Variation	SS	df	MS	F	P-value	F crit
Between Groups	41.458	3	13.81	0.932	0.44	3.09
Within Groups	296.5	20	14.8			
Total	337.958	23				

.

Statistical output for supporting water quality data

Total dissolved solids t-test: paired

	1983	*2018*
Mean	52.5	54.6666667
Variance	27.25	1720.33333
Observations	3	3
Pearson Correlation	0.97798876	
Hypothesized Mean Difference	0	
df	2	
t Stat	-0.1031324	
P(T<=t) one-tail	0.46363377	
t Critical one-tail	2.91998558	
P(T<=t) two-tail	0.92726753	
t Critical two-tail	4.30265273	

Alkalinity t-test: paired

	1983	*2018*
Mean	3.266666667	14.9
Variance	1.773333333	64.83
Observations	3	3
Pearson Correlation	-0.609017794	
Hypothesized Mean Difference	0	
df	2	
t Stat	-2.257540563	
P(T<=t) one-tail	0.076275293	
t Critical one-tail	2.91998558	
P(T<=t) two-tail	0.152550587	
t Critical two-tail	4.30265273	

Conductivity t-test: paired

	1983	*2018*
Mean	25.2	59
Variance	8.3125	1029
Observations	3	3
Pearson Correlation	-0.5960396	
Hypothesized Mean Difference	0	
df	2	
t Stat	-1.7281807	
P(T<=t) one-tail	0.11304851	
t Critical one-tail	2.91998558	
P(T<=t) two-tail	0.22609703	
t Critical two-tail	4.30265273	

Excel tables adapted from Stantec and BEAK (Raw data used for figures 3-19)

Copper µg/L

Year	Station 3	Station 4	Station 5	Station 6	Station 7	Station 8
1983	8.5		8.5	8.5		8.5
1997	4.2	12.2	7.4	4.6	3.8	1
2003	6.1	6.7	2.8	1.4	1.4	0.7
2009	2.9	4.8	3.7	2	2.6	1.4
2012	1.4	3.2	1.2	1.4	1	0.9
2015		4.54	2.53	2.63	2.29	1.08
2018	3.3	4	2.3	1.8	1.4	1.8

Zinc µg/L

Year	Station 3	Station 4	Station 5	Station 6	Station 7	Station 8
1983	290		185	105		25
1991	440	375	280	160	130	10
1997	643	1780	1070	347	272	5070
2003	474	395	129	39	37	4
2009	120	140	130	60	110	310
2012	68.6	79.1	14.8	19.8	11.8	2
2015	130	103	27.4	28.6	24.1	4
2018	110	92	37	23	19	4

Cadmium µg/L

Year	Station 3	Station 4	Station 5	Station 6	Station 7	Station 8
1983	7		7	7		7
1997	0.7	3	1.4	0.4	0.3	0.09
2003	0.6	0.4	0.2	0.09	0.09	0.09
2009	0.17	0.2	0.15	0.09	0.25	0.1
2012	0.09	0.09	0.09	0.09	0.09	0.09
2015	0.173	0.135	0.022	0.036	0.036	0.016
2018	0.09	0.15	0.09	0.09	0.09	0.09

Magnesium mg/L

Year	Station 3	Station 4	Station 5	Station 6	Station 7	Station 8
1983	0.73		0.72	0.9		0.79
1991	1.135	1.155	1.16	1.12	1.11	0.645
2009	2.2	1.2	1.6	0.9	1.5	1.4
2012	1.04	1.22	1.56	0.91	1.18	0.64
2015	1.16	0.937	0.964	0.888	0.911	0.631
2018	1.4	1.1	1.2	0.9	0.87	0.58

Aluminum mg/L

Year	Station 3	Station 4	Station 5	Station 6	Station 7	Station 8
1983	0.31		0.24	0.27		0.19
2003	0.328	0.248	0.236	0.206	0.183	0.138
2009	0.19	0.25	0.23	0.19	0.18	0.35
2012	0.102	0.13	0.059	0.091	0.042	0.087
2015	0.364	0.261	0.181	0.232	0.202	0.169
2018	0.14	0.22	0.18	0.19	0.12	0.16

Iron µg/L

	Station 3	Station 4	Station 5	Station 6	Station 7	Station 8
1983	280		205	185		100
1991	355	330	280	260	225	390
1997	340	5140	330	310	230	160
2003	490	350	240	230	190	80
2009	570	320	240	190	190	140
2012	617	273	87		100	49
2015	792	370	286	175	197	69
2018	510	300	260	170	140	99

Dissolved oxygen and temperature °C (2018)

	Station 3	Station 4	Station 5	Station 6	Station 7	Station 8
Dissolved Oxygen %	95.5	92.9	82.8	94	99.9	95
Temperature °C	17.2	18.4	20.4	18.1	18.8	18.5

Total dissolved solids mg/L

Year	Station 3	Station 4	Station 5	Station 6	Station 7	Station 8
1983	56		55	46.5		36.5
2003	58	94	60	42	36	18
2009	76	48	40	38	36	18
2012	46	48	56	38	30	34
2015	70	52	52	14	50	32
2018	90	20	65	9	9	9

Temperature °C

Year	Station 3	Station 4	Station 5	Station 6	Station 7	Station 8
1997	0.4	2.1	2.3	2.5	1.5	5
2003	6.2	8.8	4.3	7.7	7.6	8.5
2009	1.5	4.9	4.6	6.7	4.9	5.9
2012	5.6	8.1	11.5	9.7	7.3	9.4
2015	15	18.9	19.2	17.2	17.3	17.2
2018	17.2	18.4	20.4	18.1	18.8	18.5

Conductivity µmhos/cm

Year	Station 3	Station 4	Station 5	Station 6	Station 7	Station 8
1983	24.2		22.95	28.45		27
1991	104.5	100	100	89.5	86	
1997	1073	386	222	148	143	9
2003	40.1	83.1	51.1	39.3	37.3	16.5
2009	107	51	45	38	42	22
2012	43	53	57	33	39	21
2015	42	35	37	32	32	21
2018	94	58	52	31	30	21

pH

Year	Station 3	Station 4	Station 5	Station 6	Station 7	Station 8
1984	5.95		5.95	6.3		6.65
1991	6.68	6.66	6.72	6.69	6.71	
2003	6.1	6.5	6.8	6.7	6.6	6.5
2009	7.1	7.4	6.8	6.7	6.8	6.8
2012	7.69	7.64	7.64	7.11	7.16	6.82
2015	6.7	6.44	7.26	6.71	7.13	7.09
2018	7.37	7.19	6.71	6.63	7.03	6.79

% Dissolved oxygen

Year	Station 3	Station 4	Station 5	Station 6	Station 7	Station 8
2009	97	90	95	82	96	97
2012	96.3	89.2	97.1	73.6	80.2	79.5
2015	93.4	98.8	100.8	93.8	95.3	90.4
2018	95.5	92.9	82.8	94	99.9	95

Total Dissolved Solids & Sulphates

	Station 3	Station 4	Station 5	Station 6	Station 7	Station 8
TDS 1983	56		55	46.5		36.5
TDS 2003	58	94	60	42	36	18
TDS 2009	76	48	40	38	36	18
TDS 2003	46	48	56	38	30	34
TDS 2015	70	52	52	14	50	32
TDS 2018	90	20	65	9	9	9
Sulphates 1983	6.4		5.75	5.93		5.8
Sulphates 2009	26.6	10.7	5.7	5.9	5.7	2.6
Sulphates 2018	22	9.7	6.9	0.9	0.9	4.9

pH to H+ conversions for 1983 and 2018.

1983		
station	pH	H+
Cleaver inlet	5.95	1.12202E-06
Cleaver outlet		
Gumboot Outlet	5.95	1.12202E-06
Hornblende Outlet	6.3	5.01187E-07
Lyne Lake Inlet		
2018		
station	pH	H+
Cleaver inlet	7.37	4.2658E-08
Cleaver outlet	7.19	
Gumboot Outlet	6.71	1.94984E-07
Hornblende Outlet	6.63	2.34423E-07
Lyne Lake Inlet	7.03	

2018 Hardness mg/L as CaCo3 estimations for sampling stations

2018	Hardness (alkalinity regression)	Hardness (magnesium regression)	Average (Final estimation)
Cleaver inlet	239.9753	18.4752	129.22525
Cleaver outlet	133.3853	14.652	74.01865
Gumboot Outlet	123.6953	15.9264	69.81085
Hornblende Outlet	91.7183	12.1032	51.91075
Lyne Lake Inlet	103.3463	11.72088	57.53359

Water Quality Before/After table for specific conductivity, TDS and total metals

	Conductivity µmhos/cm	TDS mg/L	Alkalinity mg/L as CaCo3	Cu mg/L	Zn mg/L	Fe mg/L	Cd mg/L	Mg mg/L	Al mg/L
1983	**1983**	**1983**	**1983**	**1983**	**1983**	**1983**	**1983**	**Mg (1983)**	**1983**
Station 3	24.2	56	2.6	0.0085	0.29	0.28	0.007	0.73	0.31
Station 4						0.33			
Station 5	22.95	55	2.4	0.0085	0.185	0.205	0.007	0.72	0.24
Station 6	28.45	46.5	4.8	0.0085	0.105	0.185	0.007	0.9	0.27
Station 7						0.225			
2018	**2018**	**2018**	**2018**	**2018**	**2018**	**2018**	**2018**	**2018**	**2018**
Station 3	94	90	24	0.0033	0.11	0.51	0.0009	1.4	0.14
Station 4	58	20	13	0.004	0.092	0.3	0.00015	1.1	0.22
Station 5	52	65	12	0.0023	0.037	0.26	0.00009	1.2	0.18
Station 6	31	9	8.7	0.0018	0.023	0.17	0.00009	0.9	0.19
Station 7	30	9	9.9	0.0014	0.019	0.14	0.00009	0.87	0.12

Besser and Leib (2007). Toxicity thresholds of copper and zinc to brook trout and other species

Table 4. Toxicity thresholds for copper and zinc from laboratory tests.

[Hardness of test water=113 µg/L. LOEC, lowest observed effect concentration; IC25, 25 percent inhibition concentration; EC50, median (50 percent) effect concentration. See text and Besser, Allert, and others (2001) for derivation of toxicity thresholds]

Species	Endpoint	Copper (µg/L)			Zinc (µg/L)		
		LOEC	IC25	EC50	LOEC	IC25	EC50
Amphipod (*H. azteca*)	Survival	100	64	79	250	183	220
Fathead minnow (*P. promelas*)	Survival	12.5	17	35	500	458	704
	Growth	100	9.2	55	>1,000	>1,000	>1,000
	Biomass	50	8.7	11.4	500	>1,000	>1,000
Brook trout (*S. fontinalis*)	Survival	25	24	29	>2,000	>2,000	>2,000
	Growth	6.3	8.1	17	1,000	>2,000	>2,000
	Biomass	6.3	9.8	17	>2,000	>2,000	>2,000