LEGACY EFFECTS OF HISTORICAL GOLD MINING ON CHIRONOMID ASSEMBLAGES FROM GEGOGAN LAKE, NOVA SCOTIA

BY

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Abstract

Historical gold mining in Nova Scotia (NS), Canada, began in the 1860s, and released contaminants into the environment through tailings that were transported by wind and surface waters. I investigated the remains of benthic invertebrates and geochemical proxies archived in the sediment of a shallow lake downstream from the province's most productive historical gold mine. Sedimentary concentrations of metal(loid)s were used to determine mining inputs of tailings as well as establish the pre-mining and post-mining time periods within the lake sediment record. Chironomid assemblages were assessed as an ecological response to long-term environmental changes. Total mercury levels peaked at six times above the sediment quality guidelines, and arsenic concentrations peaked at 165 times. Sedimentary contaminants have declined but remain above probable effect levels. There was a significant difference ($p = 0.001$) in assemblage compositions between the pre-mining and post-mining time periods. There were seven indicator taxa in the pre-mining and three indicator taxa in the post-mining period. In the pre-mining time period, there was an average of 1448 ± 186 chironomid head capsules per gram of sediment which declined to an average of 355 ± 257 per gram in the post-mining time period. The relative abundances of several metal(loid) tolerant chironomid taxa have increased in the post-mining period. For example, during the pre-mining period *Constempellina* was present at an average of 5% abundance and increased to 12% abundance post-mining. Findings from this study show that pollution from historical gold mining continues to make its way into and negatively affect the benthos of Gegogan Lake.

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Introduction

History of Gold Mines in Nova Scotia

In the mid-1800s gold mines in Nova Scotia (NS) opened across the province. From 1862 to 1938, a total of 64 gold mining districts were established (Bates, 1987). The gold mining industry supported the local economy of many regions in the province. An estimated 1.2 million troy ounces (~37,000kg) of gold were produced in NS (Bates, 1987). The most productive historical mining district was Goldenville, located near the village of Sherbrooke, NS. Goldenville produced approximately 6,500kg of gold during its operation from 1862 to 1941 (Bates, 1987). Although the historical mines of Goldenville have been closed for over 80 years, the environmental impacts associated with past mining activities has likely continued to negatively impact the nearby ecosystem, such as persistent and elevated levels of mining-associated contaminants, like mercury and arsenic (Tindale et al., 2016; Wong et al., 1999; LeBlanc et al. 2020; Clark et al. 2021).

Historical gold mining practices and past waste management strategies differ greatly from those used and required by law today. To extract gold from the crushed ore, mercury amalgamation techniques were used at Goldenville (Wong et al., 1999). In this process, ore is crushed into fine particles and the gold is "dissolved" in mercury added to the crushed ore. In Nova Scotia, gold is typically hosted with arsenopyrite (Meunier et al., 2010). The solution is then heated to evaporate the mercury amalgam and isolate the gold (Bates, 1987). For maximum yield, about one ounce of mercury was used to extract one ounce of gold (Parsons et al., 2012). The residual sand and silt, also known as tailings, accumulated near the historical gold mines in large piles and were often deposited directly in nearby surface waters (LeBlanc et al., 2020; Tindale et al., 2016). A field of tailings currently occupies an area of 0.25km^2 nearby the abandoned mine processing area (Tindale et al., 2016).

Early mining methods were wasteful and environmental practices were not as good as they are today (Bates, 1987; LeBlanc et al., 2020). The accumulation of tailings has resulted in many toxic contaminants left to harm the nearby ecosystem. There is approximately 6800kg of mercury in the nearby tailings field at Goldenville (Wong et al., 1999). There are also high concentrations of arsenic in the tailings, reaching concentrations of up to 47,000mg of arsenic

per kilogram of tailings (Parsons et al., 2012). After being crushed, the tailings have high concentration of metal(loids) which poses a threat to the biota (Wong et al., 1999).

Tailings in the Environment

Mercury and arsenic are heavy metal(loid)s, which means they are more dense than water, and are harmful to living organisms at high concentrations (Graeme & Pollack, 1998). To limit adverse effects on organisms, the concentration of contaminants in the environment should be below the probable effect level (PEL), for total mercury in sediments it is 486µg/kg (CCME, 1999b). For arsenic, the PEL in sediments is 17mg/kg (CCME, 1999a). These two contaminants are often associated with historical gold mining pollution in Nova Scotia. High concentrations of both mercury and arsenic have been reported in the surrounding sediments and plant life near historical gold mines in Nova Scotia (Meunier et al., 2010; Parsons et al., 2012; LeBlanc et al. 2020). These contaminants are found in the tailing fields surrounding the historical Goldenville processing area and can enter the surface waters through weathering and erosion (Wong et al., 1999). Thus, pollution from historical mining can be transported many kilometers from where it was produced (Wong et al. 1999; Leblanc et al. 2020; Clark et al. 2021).

Mercury from mine tailings is able to make its way into the aquatic environment and subsequently impact biota exposed to high concentrations of this pollutant (Wong et al., 1999; Zillioux et al., 1993). When in water, mercury can either be dissolved or in a particulate state (Wang et al., 2004). Mercury can bioaccumulate in aquatic organisms and biomagnify in the food web. There is a correlation between mercury and organic matter, when there are high concentrations of organic matter, there tends to be more mercury (Sigel & Sigel, 1997). In the water column, microbes and phytoplankton transform inorganic mercury into a more toxic and bioaccumulative form through methylation recognized as methylmercury (Gascón Díez et al., 2016). Mercury also blocks the use of enzymes and organelles within the cell, which inhibits the growth of the algae (Le Faucheur et al., 2014). This is also a concern to aquatic biota as the assimilation efficiency of methylmercury is three times more than that of inorganic mercury (Mason et al., 1996). There are severe ecological consequences when mercury builds up in aquatic ecosystems, especially to consumers at the top of the food web (Drevnick &

Sandheinrich, 2003; Finley et al., 2016). Once mercury settles into the sediment, less of it is methylated, and its ability to enter the food web is reduced (Gascón Díez et al., 2016).

Arsenic is also detected in surface waters and tailings near historical gold mines like Goldenville (Parsons et al., 2012; Wong et al., 1999). The crushing of gold-bearing arsenopyrite ores to extract gold produced tailings rich in arsenic. The tailing piles weathered and eroded and moved into nearby aquatic ecosystems over time (Meunier et al., 2010; Wong et al., 1999). Like that of mercury, arsenic can bioaccumulate in aquatic systems, although not as easily as mercury. Phytoplankton uptake arsenic, and transform it through processes like methylation (Hellweger & Lall, 2004). They transform arsenate (As^{5+}) to the more toxic arsenite (As^{3+}), as well as other methylated species (Hellweger & Lall, 2004). However, there are some prokaryotes that can reverse the transformation and produce arsenate from arsenite (Ellis et al., 2001; Tsai et al., 2009). Arsenic bioaccumulates and has ecological effects when it builds up in invertebrates and fish (Cott et al., 2016; Mason et al., 2000). Regardless of the oxidation state or chemical species of arsenic, there are negative implications for the food web.

Understanding Paleolimnology

As there is little to no monitoring data available for ecosystems near historical gold mines during their operation, another method of quantifying a response to historical contaminants is needed. Lake sediments are natural archives that contain vast amounts of indicators such as subfossils and geochemical markers, they are often studied in combination to make inferences about past conditions (Smol, 2010; Korosi et al., 2017). Paleolimnology is the science of utilizing indicators in lake sediments. Paleolimnological techniques allow for an ordered timeline to be reconstructed through the process of sedimentation (Smol, 2010). Newer sediment sinks and accumulates on top of older sediments. Autochthonous materials originate from inside the lake, and allochthonous materials originate from outside the lake. Both materials accumulate along with sediments and may be well preserved with the other materials that were inputted. The age of sediments can be determined with some estimate of uncertainty and the preserved materials can be studied to infer geochemical and ecological conditions (Korosi et al., 2017; Smol, 2010). Studying lake sediments, and what is preserved in them, is now a common approach to determine long-term environmental changes.

There are many physical, chemical, and biological indicators that are preserved in the sediment. Physical indicators that can tell you something about the past include sediment size, elemental composition, and colour, as well as presence or absence of debris transported into the lake (Almas et al., 2022; Foster et al., 2011; Dunnington et al. 2020). Chemical indicators are usually quantified analytically. Common chemical indicators are those that were never high in concentration until introduced anthropogenically, like trace metals, organic pollutants, and other legacy contaminants (LeBlanc et al., 2020; Roberts et al., 2019). They are also important for the dating process that includes radiometric analysis of certain isotopes (e.g., ^{210}Pb, ^{14}C) to estimate the age of sediments (Burge et al., 2018). However, with the help of biological indicators or bioindicators, more can be learned about the past ecology of a lake from an ecological perspective (Walker et al., 1991). Common bioindicators are fossilized remains of organisms that preserve well in sediments, like chitin, silica, or pollen and spores. Examples of common bioindicators are diatoms, Cladocera, chironomids, and pollen (Korosi et al., 2017; Wojewódka & Hruševar, 2020). The ecologies of many of the organisms are known and aid in the reconstruction of past environmental changes experienced by the lake.

The larval head capsules of chironomids (order Diptera) are well preserved and abundant in lake sediments. Chironomid eggs are laid in the water, and before re-emerging as an adult insect, will go through a series of four larval growth stages called instars (Walker, 2001). In their larval stage, they generally occupy benthic habitats in lakes, meaning they dwell on or in the sediment of lakes (Walker, 2001; Broderson & Quinlan 2006). Between these instars, they shed their chitinized head capsule, which can often be identified to the species or genus level (Walker, 2001). Earlier instars are usually not identified due to their small size and incomplete morphology. Chironomids are useful bioindicators of the benthic habitat for multiple reasons. They are ubiquitous and a diverse group of benthic invertebrates that can be used to track changes in benthic conditions (Brodersen & Quinlan, 2006). Within a lake system, some taxa, like many in the subfamily Orthocladiinae, prefer to occupy the littoral zone, while others prefer the profundal zone (Armitage et al., 2012).

Using chironomids as bioindicators to assess how an ecosystem has changed is now common given that most chironomids live in close contact with the sediment-water interface and thus are exposed to sedimentary pollution to a greater extent than bioindicators living exclusively in the water column. In addition to past climatic change and dissolved oxygen conditions,

chironomid assemblages can be used to assess biotic responses to pollutants associated with sediment deposition (Korosi et al., 2017; Smol, 2010). Chironomids have been studied to understand the biotic responses to many forms of contamination due to human activities, including legacy pollutants and metal(loid) contamination from mining (arsenic, nickel, copper, lead, etc.) (Ilyashuk et al., 2003; Pelletier et al., 2016; Thienpont et al., 2016). The richness and diversity of chironomids has been found to correlate with the input of pollution associated with mining (Ilyashuk et al., 2003). Arsenic, which entered a lake via aerial deposition from roaster stack emissions, also shifted the chironomid assemblage following the opening of one of Canada's largest gold mines located in Yellowknife (Thienpont et al., 2016). Chironomid taxa respond to contamination in different ways due to their varying ecological preferences. Shifts in the assemblage composition and overall production can be recognized following the historical inputs of mining pollution.

Research Objectives

I used paleolimnological techniques to assess the long-term impacts of historical gold mining from the Goldenville Mining District near Gegogan Lake, a shallow polymictic lake 6km downstream from the tailings field at Goldenville, NS. I investigated the long-term effects of historical gold mine pollution on the benthic environment of Gegogan Lake. Geochemical measures were used as proxies for historical gold mine pollution and chironomid head capsules were used as bioindicators to this pollution. I addressed the following questions:

1) Has the assemblage composition and production of chironomids changed because of historical mining pollution?
2) Has geochemical and biological recovery occurred after mine closure in ~1940s, and if not, what environmental factors may explain the lack of recovery from past mining activities?

I predicted that when sedimentary concentrations of total mercury and arsenic increased in Gegogan Lake, chironomid assemblages responded with a compositional shift and reduced production. Taxa tolerant of heavy metal pollution and generalist taxa were expected to increase

in the sedimentary record after historical mining began in the 1860s. Following the closure of mining in the 1940s, I predicted that there will be biological recovery if there is a decrease in geochemical pollution to concentrations below probable effect levels that are known to cause biological harm. This study will provide insight into how historical gold mines have affected Nova Scotian shallow lake ecosystems that received inputs of pollutants. Findings will contribute to the ongoing research assessing the impacts of historical gold mines in Nova Scotia waterways.

Methods

Study Site

Gegogan Lake (45.0791; -62.0047) is a remote lake located in eastern NS in Guysborough County. The polymictic lake has a surface area of 24ha, maximum depth of ~2.0m, mean depth of 1.1m, and a pH of 6.2 (Table 1). Its watershed is part of the Acadian Forest (Figure 1). The Acadian Forest is a transitional zone composed of both boreal forest and temperate tree species, such as Balsam Fir, White Birch, Sugar Maple, and Yellow Birch (Noseworthy & Beckley, 2020). The average air temperature of Sherbrooke, NS from 1993 to 2002 was 6.8°C, and it is below 0°C in December to February and above 20°C in July-September (Environment and Climate Change Canada, 2023). Total precipitation per month ranges from 22.6mm to 145mm (Environment and Climate Change Canada, 2001). Other than the historical gold mines 6km upstream of this lake, Highway 7 is the only major development of urbanization in the vicinity. The small village of Sherbrooke, NS, is 7km north of the lake. There is one seasonal cottage on the lake.

The Goldenville Mining District was directly upstream of Gegogan Lake (Wong et al., 1999). Goldenville was the most productive gold district in NS (Parsons et al., 2012). Tailings occupy an area of 0.25km^2 nearby the abandoned mine processing area (Tindale et al., 2016). Tailings are known to have moved downstream via Gegogan Brook, into Gegogan Lake, and into the Atlantic Ocean via Gegogan Harbour (Wong et al., 1999).

On April 25th, 2022, a 23.5cm sediment core was retrieved from near the centre of Gegogan Lake where Gegogan Brook flows in. This was obtained using a Glew gravity corer at depth of 2.0m (Glew et al., 2001). Following retrieval, the sediment core was extruded and sectioned into 0.5cm intervals using a vertical extruder (Glew et al., 2001). Each interval was stored in a labeled Whirl-Pak® bag and stored on ice in a cooler. Upon returning to the Environmental Change and Aquatic Biomonitoring Laboratory at Mount Allison University, sediment was stored in a refrigerator.

Geochemistry and Sediment Dating

A total of 26 sediment intervals from throughout the core were freeze-dried for further analysis of geochemistry and dating. To estimate sediment ages, radiometric ^{210}Pb dating was performed. Approximately 0.3g of sediment from 17 of the freeze-dried samples were placed into labelled gamma tubes and sealed with epoxy. These tubes were then sent to Queen's University for dating analysis using gamma spectroscopy with a germanium crystal detector.

To measure the concentration of select mining-associated contaminants, ultratrace inductively coupled plasma-mass spectrometry (ICP-MS) was performed at the Analytical Services Unit of Queen's University in Kingston, Ontario following USEPA Method 200.7. The Analytical Services Unit is a Canadian Association for Laboratory Accreditation certified facility. Approximately 0.5g of sediment from every freeze-dried sample were diluted for ICP-MS analysis. Total mercury and arsenic were the sedimentary contaminants focused on in this thesis given their association to historical mining activities (Parsons et al., 2012; Wong et al., 1999; Clark et al., 2021). Mercury samples were analysed using a direct mercury analyser (DMA-80) following USEPA Method 7473.

Chironomid Preparation and Identification

All 26 freeze-dried sediment intervals were sampled for chironomid head capsules and *Chaoborus* mandibles. On average, 0.25g of sediment was needed to reach the target count of 75 whole mentum equivalents per sediment interval (Quinlan & Smol, 2001). Preparation followed Walker (2001). Sediments were deflocculated in 80mL of warm potassium hydroxide for 20 mins, then sieved using a 100μm mesh. Materials retained on the sieve were then rinsed into a beaker with about 20mL of deionized water. The solution was then poured into Bogorov trays and sorted through to collect all identifiable midge remains. Using a dissecting scope at 20-40x magnification, chironomid head capsules that were more than half intact, or chaoborid mandibles, were picked out and placed onto coverslips using forceps. To ensure all remains were isolated from the Bogorov tray, two passes of the surface and bottom of the tray were completed with a stir of the tray contents between passes. Head capsules were spread evenly on each

coverslip to prevent them from obscuring one another. Coverslips were permanently mounted onto slides using Entellan® mounting medium.

Head capsules were identified using bright field microscopy at 200x or 400x magnification. Each remain was identified to the lowest taxonomic rank possible, usually genus or species level, using Brooks et al. (2007) and Wiederholm (1983). Identifications were based on the ventromental plates, mentum, antennal pedestals, and mandible when possible. Tanytarsini often did not have antennal pedestals, which are important identifying traits, when these were not present, they were counted as indeterminable (IND).

Data Analysis

The concentration of chironomid head capsules per gram of sediment was calculated for each sediment interval. Relative abundance was also calculated for each taxon using the total number of individuals found in the sediment interval. Taxa that were considered rare were removed from further statistical analysis. Rare taxa were categorized as being observed in two samples or fewer at less than 2% relative abundance. Pre-mining and post-mining time periods were established based on an increase of sedimentary total mercury concentrations (~8.25cm sediment interval). This worked out to be 14 samples in the pre-mining period, and 12 samples in the post-mining period. Data were plotted in a stratigraphy using C2 graphing software. All statistical analysis were performed using R (version 4.2.2) with an R Studio interface (RStudio Team, 2023).

To determine if there was a compositional difference of assemblages between pre-mining and post-mining periods, a permutational multivariate analysis of variances (PERMANOVA) was performed. A PERMANOVA is similar to an analysis of variance (ANOVA) but allows for more broad applications and meaningful analysis of a high dimensional data set than an ANOVA (Anderson, 2017). It is an effective analysis even when the data set may have variables that are not normally distributed, as well as can accommodate for heterogeneity of within group dispersions (Anderson, 2017). As there were only two time periods to compare, further pairwise testing of differences was not necessary. To determine if there was a difference in the concentration of chironomid head capsules per gram of sediment, a Welch two sample t-test was performed.

Indicator taxa for each time period were determined using an Indicator Value (IndVal) analysis in the package labdsv. IndVal considers the relative abundances of a taxa across pre-defined groups (Dufrêne & Legendre, 1997). The IndVal analysis outputs which group a taxa belongs to, the indicator value, and a p-value. The best indicators are those with an indicator value close to one.

Results

Sediment Chronologies

A reliable chronology could not be developed for the sediment core retrieved from Gegogan Lake. The ^{210}Pb activities in the sediments were very low and thus reliable ages could not be determined for much of the sediment record. Although low ^{210}Pb activities in sediments are rarely reported from temperate regions, Dunnington et al. (2020) also reported low levels of ^{210}Pb activities in the sediments of a shallow lake from northeastern Nova Scotia. In the absence of a ^{210}Pb-based chronology, other geochemical proxies can be used to identify major periods in the sediment record. In this study, the increase of total mercury concentrations around 8.25cm was used to identify the 1860s when mining began in Goldenville. Sedimentary mercury concentration increased from 843µg/kg to 1400µg/kg between the intervals 9.25cm and 8.25cm, which is a ~66% increase from background levels of mercury. Assuming that this increase corresponded with the beginning of the historical mining, milling, and amalgamation operations in the 1860s, sediments below 8.25cm are thus recognized from the pre-mining time period. Although the Goldenville Mining District closed in 1941, it is difficult to identify this time period without the help of more sophisticated dating techniques.

Trends in Geochemistry

In the pre-mining period, mercury ranged between 244µg/kg and 843µg/kg, with a mean of 521µg/kg. Following the increase at 8.25cm from background levels, the highest recorded concentration was 2880µg/kg and sediments continue to reflect concentrations above 2300µg/kg. The concentration of mercury in Gegogan Lake sediments remained at elevated levels in the post-mining period (Figure 1). On average, there was a 4.7 times increase in mercury from the pre-mining to the post-mining time period.

In the pre-mining period, arsenic ranged between 200mg/kg and 1600mg/kg, with an average of 686mg/kg. The trend of arsenic in the post-mining period differed from that of mercury. Arsenic concentration peaked at 2800mg/kg in the 8.25cm sediment interval, and later decreased to 410mg/kg at the 3.25cm interval. The trend continued to stay below 500mg/kg.

Although there was an increase in arsenic concentration after mining began, the amounts in the upper sediments have returned to similar levels from the pre-mining time period.

Chironomid Assemblages

A total of 63 chironomid taxa were identified in Gegogan Lake (Appendix 1). Additionally, *Chaoboridae* mandibles were also identified in intervals, but were excluded from further analysis due to their low numbers. After excluding rare taxa, and grouping similar taxa, 46 chironomid taxa were determined to be common throughout the sediment core. The PERMANOVA indicated that the assemblage is significantly different between the pre-mining and post-mining time periods (Table 2). The IndVal results indicated that there were seven indicator taxa in the pre-mining period, and three indicator taxa post-mining (Table 3). All indicator taxa in the post-mining time period were on average higher in abundance than they were in the pre-mining time period. The relative abundance of *Constempellina* sp. increased the most, which was by ~2.5 times. *Parakiefferiella triquetra* was identified in more post-mining intervals than pre-mining and was not identified in the oldest parts of the sediment core.

All but one indicator taxon in the pre-mining time period were on average higher in abundance than they were in the post-mining time period. The one taxon, *Neozavrelia*, was identified in only one interval in the post-mining period, compared to being identified in 8 intervals pre-mining. *Zalutschia* sp. and *Unniella* also experienced a similar trend, where they were identified in more intervals in their indicator period but also at higher relative abundances. The relative abundance of *Microtendipes pedellus*, *Endochironomus*, and *Cricotopus/Orthocladius* decreased by ~41%, ~40%, and ~28% respectively. The results of a t-test indicated that there was a significant difference in the number of chironomid head capsules per gram of sediment between the pre-mining (M = 1448, SD = 186) and post-mining periods (M = 355, SD = 257); $t(19.7) = -12.23$, $p = <0.001$ (Figure 3).

Discussion

Environmental Changes in Response to Historical Gold Mining in Gegogan Lake

The sedimentary record of Gegogan Lake suggest that it was impacted by historical gold mining from the upstream Goldenville Mining District (Figure 2). The increase in sedimentary arsenic and mercury in lakes near to mining operations has been suggested to be linked to past mining activities (Fashola et al., 2016; Thienpont et al., 2016; Clark et al. 2021). At Gegogan Lake, following a large increase in arsenic and mercury concentrations there is a decrease in chironomid production (Figure 2 and Figure 3). The relative abundance of certain taxa has also changed and did not recover to abundances observed during the pre-mining period, further emphasizing a shift in the chironomid assemblage following historical gold mining pollution.

In the pre-mining period, arsenic had an average concentration of 686mg/kg, which is higher compared to PEL from the Canadian sediment quality guidelines (CCME, 1999a). The elevated levels of arsenic in the pre-mining sediments may be due to local bedrock geology (LeBlanc et al., 2020). Relatively high background levels of sedimentary arsenic were also observed in dated sediment cores from two lakes near Dartmouth, NS (Clark et al., 2021). In the post-mining time period, the arsenic concentration peaked at 2800mg/kg, which is well above the Canadian PEL of 17mg/kg (CCME, 1999a). Wong et al. (1999) recorded a peak of ~900mg/kg of arsenic. This increase is likely due to the transport of tailings via surface waters into Gegogan Lake (Wong et al., 1999). Within recent sediments, arsenic concentrations recover to about what they were in pre-mining conditions, reaching as low as 110mg/kg (Figure 2).

The average concentration of sedimentary mercury in the pre-mining time period was 521µg/kg, which is also above the PEL of the Canadian sediment quality guidelines (CCME, 1999b). In the post-mining period, the concentration of mercury persists above 2000µg/kg, with a peak of 2880µg/kg and an average of 2438µg/kg. This is well above the PEL of 486µg/kg (CCME, 1999b), meaning that adverse biological effects can be expected from benthic organisms exposed to this pollution. Wong et al. (1999) recorded a peak of ~1650µg/kg at the surface sediments in Gegogan Lake. The persisting levels of mercury in the sediment is likely due to continuous weathering and transport of tailings that are located upstream (Eckley et al., 2021; Wong et al., 1999). Gegogan Lake is considered dystrophic, it is acidic, dark watercolor,

and has high dissolved organic material (Table 1). There is a link between the amount of organic matter and mercury concentration in aquatic systems (Sigel & Sigel, 1997). With higher amounts of organics, there will be more dissolved mercury. As Gegogan is dystrophic, it likely has high dissolved organic mercury and may continue to have high concentrations of mercury in the water column (Wong et al., 1999). As long as the tailings field continues to exist upstream from Gegogan Lake, mercury will likely continue to persist in the water column and accumulate in the sediments.

It is likely that the legacy pollutants from historical gold mining are driving ecosystem changes at Gegogan Lake. It is a remote lake where the nearest settlement is the small village of Sherbrooke, which is 7km away. There is also a minor highway directly next to the lake, but these anthropogenic factors are not likely major stressors to the surrounding environment.

Drivers of Chironomid Assemblage Composition Changes

The assemblage composition of chironomids at Gegogan Lake differed between the pre- and post-mining time period (Table 2). This change occurred following the increase of legacy gold mining pollutants. In the post-mining time period, several taxa decreased in abundance and/or were not identified in any sediment intervals. However, there were some taxa that experienced an increase in relative abundance. This is not unusual as some chironomids are known to be tolerant to metal(loid)s, including arsenic (Canfield et al., 1994). Regardless, chironomid assemblage composition has been known to shift in response to gold mine pollution (Stewart et al., 2018; Thienpont et al., 2016). A ~75% decrease in production of chironomids also occurred between mining periods (Figure 3). The production decrease at Gegogan Lake emphasizes that the numbers of chironomids have not recovered since the mine closure over 80 years ago. Total production may have decreased by ~4 times but how individual taxa respond to contaminants varies. As legacy pollutants like mercury persists, this is likely the driver of ecological change within the benthos of Gegogan Lake.

There were three taxa identified to be indicators of the post-mining time period, *Constempellina* sp., *Tanytarsini* (IND), and *Parakiefferiella triquetra*. The average relative abundances of *Constempellina* sp. and *Tanytarsini* (IND) increased by 7% and 6% respectively, they were identified in every sediment interval. As for *Parakiefferiella triquetra*, the average

relative abundance increased from 1.2% to 1.7%, a minimal amount. It was identified in eight post-mining intervals, which is double compared to the pre-mining time period. The increase of taxa in the Tanytarsus sub-family, *Constempellina* sp. and *Tanytarsini* (IND), is similar to that of Stewart et al. (2018). In Stewart et al., *Tanytarsus* were found to increase following mining and sewage inputs. In contrast, Thienpont et al. (2016) found that this genus decreased after the Giant Mine opened in Yellowknife, NT. The different responses of this taxa following gold mining related contaminants should be further studied as there may be other stressors affecting the ecology of *Tanytarsus* species. The genus *Parakiefferiella* are known to be a tolerant taxa that inhabit cooler waters (Medeiros et al., 2012; Odume et al., 2016). Although a slight increase in relative abundance, the signal from *Parakiefferiella triquetra* may be due to the fact that is becoming more common in the upper sediments. Following contamination events there is generally a lag period in individual taxa responses. The decrease in other taxa may have also triggered the change for this taxa. The response of *Parakiefferiella* to metal(loid) contamination is also rarely reported in literature. Further understanding of how taxa, or groups of taxa, respond to mining-associated pollutants is needed to build on what we already know of their ecological preferences.

There were seven taxa identified to be indicators of the pre-mining time period, four of them had indicator values greater than 0.5 and relative abundances greater than one. The four were *Unniella*, *Endochironomus*, *Microtendipes pedellus*, and *Cricotopus/Orthocladius*. In the post-mining period, they all experienced a decrease in relative abundances and how many intervals they were identified. *Unniella* is rarely reported in the paleolimnological literature (Engels & Cwynar, 2011). It has been reported that *Cricotopus/Orthocladius* in lake systems eventually decrease in relative abundance following exposure to mine waste (Little et al., 2020). However, a study in New Brunswick found that *Cricotopus/Orthocladius* abundance significantly increases when exposed to mine effluent (Swansburg et al., 2002). Though it is important to note that these data were collected in river systems and not lakes. It is clear that the *Cricotopus/Orthocladius* in Gegogan Lake experienced an initial decrease. The current trend may indicate that they have returned to pre-mining abundances (Figure 2). This is not the case for the other three indicator species for the pre-mining time period.

Biological recovery for many taxa is not yet seen in Gegogan Lake despite the mining being closed since the 1940s. The relative abundances of *Unniella*, *Endochironomus*, and

Microtendipes pedellus remain at levels below their pre-mining baseline. They were also rarely identified in the top sediment intervals. This may represent the disappearance of these taxa due to intolerable conditions of the benthic habitat. The taxa with the highest indicator value, *Unniella*, was identified in 86% of the pre-mining intervals and 33% of the post-mining intervals, relative abundance decreased by 44%. It is usually found in lakes with a low pH and high oxygen levels (Wazbinski, 2011). The pH of Gegogan Lake has been reported to be slightly acidic (Table 1), if *Unniella* are disappearing, this further emphasizes that there is a change in the local environment. The loss of *Endochironomus* represents a gradual change as the relative abundance only decreases later in the sediment core. *Endochironomus* is known to be tolerant to arsenic pollution (Little et al., 2020), therefore a lagged decrease is not unexpected. Another factor(s), like the persisting mercury levels, may be the reason for the loss of *Endochironomus*. As for *Microtendipes pedellus*, multiple studies show this taxon's intolerance to mining pollution (Little et al., 2020; Swansburg et al., 2002). Little et al. (2020) found that *Microtendipes pedellus* occurred at 6% relative abundance in a lake with 0.9µg/L of arsenic and did not occur in lakes with 89.6µg/L of arsenic. Although higher arsenic concentrations, *Microtendipes pedellus*, prefers a more shallow and sandy habitat (Engels & Cwynar, 2011). The arsenic may be trapped in deeper sections of this lake and further down in the sediment where it is not available to biota. Given arsenics mobility, preventing anoxic events will help to keep the concentration at the water-sediment interface as low as possible (Smedley & Kinniburgh, 2002). There has been little to no biological recovery at Gegogan Lake based on the chironomid assemblage. Following the influx of legacy contaminants, the overall production of chironomids continues to stay much lower than that of the pre-mining time period. As long as legacy contaminants are persisting at elevated levels and entering the lake system, it is unlikely that we will see any biological recovery due to the varying tolerance level of taxa.

Implications

Historical gold mining practices have contaminated the landscape and surrounding watersheds in Nova Scotia for decades (Clark et al., 2021; LeBlanc et al., 2020; Parsons et al., 2012). Mining activities in the Goldenville District, Nova Scotia's most productive historical gold mine, has impacted the benthic bioindicators that are presented in this study. It is known

that the large tailings field near Goldenville have released contaminants into Gegogan Lake for over a century (Wong et al., 1999). It is likely that this will continue to occur as long as the tailings exist in the surface environment upstream of the lake. Further, these metal(loids) may be transported into the Atlantic Ocean and impact a larger community and ecosystem (Wong et al., 1999).

The high levels of mercury and arsenic in Gegogan Lake have changed the chironomid assemblage, specifically assemblage composition and production of chironomids. This has implications for the aquatic food web and the processing of detritus by benthic organisms like chironomids. Assuming that these organisms are taking up mercury and arsenic, the ability to move across the food web to a higher trophic position is possible. They are both known to readily bioaccumulate in biota (Hellweger & Lall, 2004; Sigel & Sigel, 1997). Mercury more so than arsenic, but both may result in negative ecosystem implications. The mobility of arsenic in sediments also poses a problem. It often takes more than a few decades to see biological recovery in post-mining time periods, if there will be any at all (Little et al., 2020; Thienpont et al., 2016). If the physical environment is polluted, then returning to pre-mining baselines is unlikely. Additionally, climate change may trump historical mining pollution as a dominant stressor on the aquatic ecosystem.

The current trend for *Cricotopus/Orthocladius* may be the first sign of biological recovery at Gegogan Lake. However, it is just one of many indicator organisms present in this study. The majority of the indicator taxa in the pre-mining time period have not been identified in recent sediments and may have disappeared from this lake. As for post-mining, the rise in *Parakiefferiella triquetra* is concerning as it was not always present in the sediment record. *Constempellina* sp. may have always been present, but now has elevated abundances, this taxon is dominating and may take up resources from other taxa preventing their success. Chironomid assemblage recovery at Gegogan Lake in regard to the production and abundances will take a very long time.

Conclusions

This study used paleolimnological techniques to provide an assessment on the impacts of the legacy contaminants from the Goldenville Mining District on the benthic environment of Gegogan Lake. A shift was detected in the chironomid assemblage, which coincided with a large increase of sedimentary concentrations of mercury and arsenic, common wastes from historical gold mining. Taxa known to be intolerant to contamination were not identified in the recent sediments, and those that are tolerant to ecological change increased in abundance. Additionally, there was a significant and sustained decline in chironomid production as observed in the number of head capsules per gram of sediment. Arsenic has since decreased in concentration from its peak but is still above PEL. Mercury persists at elevated levels above PEL in the sediments, these contaminants are still likely transported into the lake. Gegogan is a remote lake and is unlikely affected by anthropogenic factors other than the tailings field directly upstream from it. Gegogan Lake was and continues to be affected by historical gold mining contaminants. It is important to work towards understanding how these contaminants move and effect organisms so that aquatic habitats may return to safe conditions.

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Tables

Table 1. Select physical, chemical, and biological characteristics of Gegogan Lake and its surface waters. *Based on data collected by Alexander et al. (1986) and [+]data collected by Wong et al. (1999).

Characteristic	Value
Coordinates	45.079394N, 62.003802W
Trophic Status	Dystrophic
Maximum depth (m)	2.0
Surface Area (ha)	24*
pH	6.2*[+]
Dissolved Oxygen (mg/L)	8[+]
Total Carbon	21.5[+]
Suspended solids (mg/L)	<5[+]

Table 2. Permutational multivariate analysis of variance (PERMANOVA) results comparing chironomid relative abundances in the pre-mining (n = 14) and post-mining (n = 12) time periods.

Lake		Df	SS	MS	F.Model	R2	P-value
Gegogan	Period	1	0.22	0.22	3.94	0.14	0.001
	Residual	24	1.33	0.06		0.86	
	Total	25	1.55			1	

Table 3. Chironomid taxa identified as significant indicators of time periods and their corresponding IndVal coefficients, *p*-values, average abundances per depth interval, and number of occurrences.

Time Period	Taxon	IndVal	*p*-value	Average Abundance	Number of Occurrences
Post-Mining	*Constempellina* sp.	0.71	0.001	12.5	187
	Tanytarsini (IND)	0.56	0.03	26.9	369
	Parakiefferiella triquetra	0.51	0.02	1.1	18
Pre-Mining	*Unniella*	0.73	0.001	3.2	44
	Endochironomus	0.69	0.009	3.5	48
	Microtendipes pedellus	0.65	0.02	3.0	43
	Cricotopus/Orthocladius	0.60	0.05	5.1	70
	Neozavrelia	0.47	0.02	0.9	10
	Tanytarsus sp.	0.43	0.02	2.5	32
	Zalutschia sp.	0.40	0.05	0.8	10

Figures

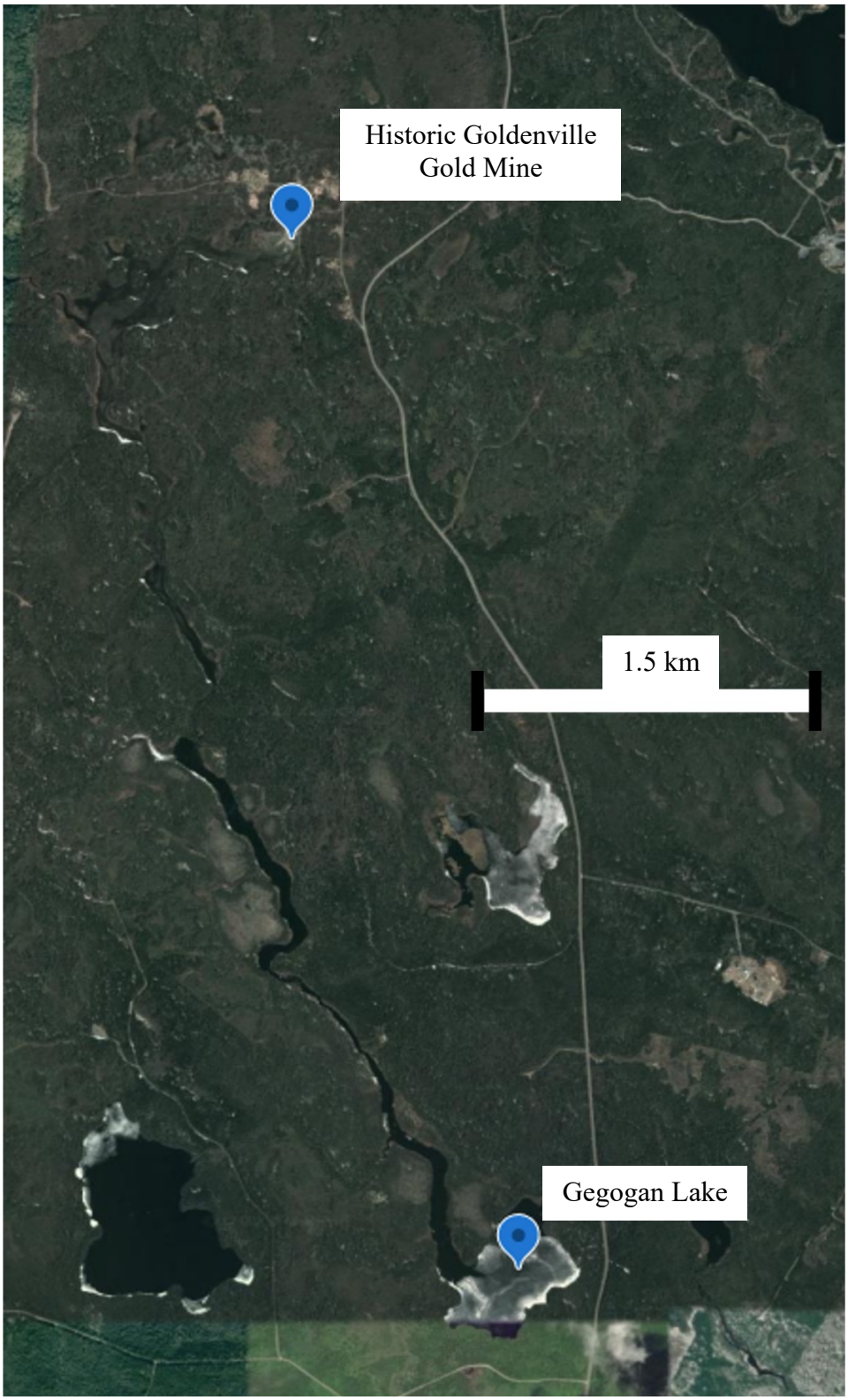

Figure 1. Google Earth image of the Goldenville Mining District and surrounding freshwater and terrestrial ecosystems.

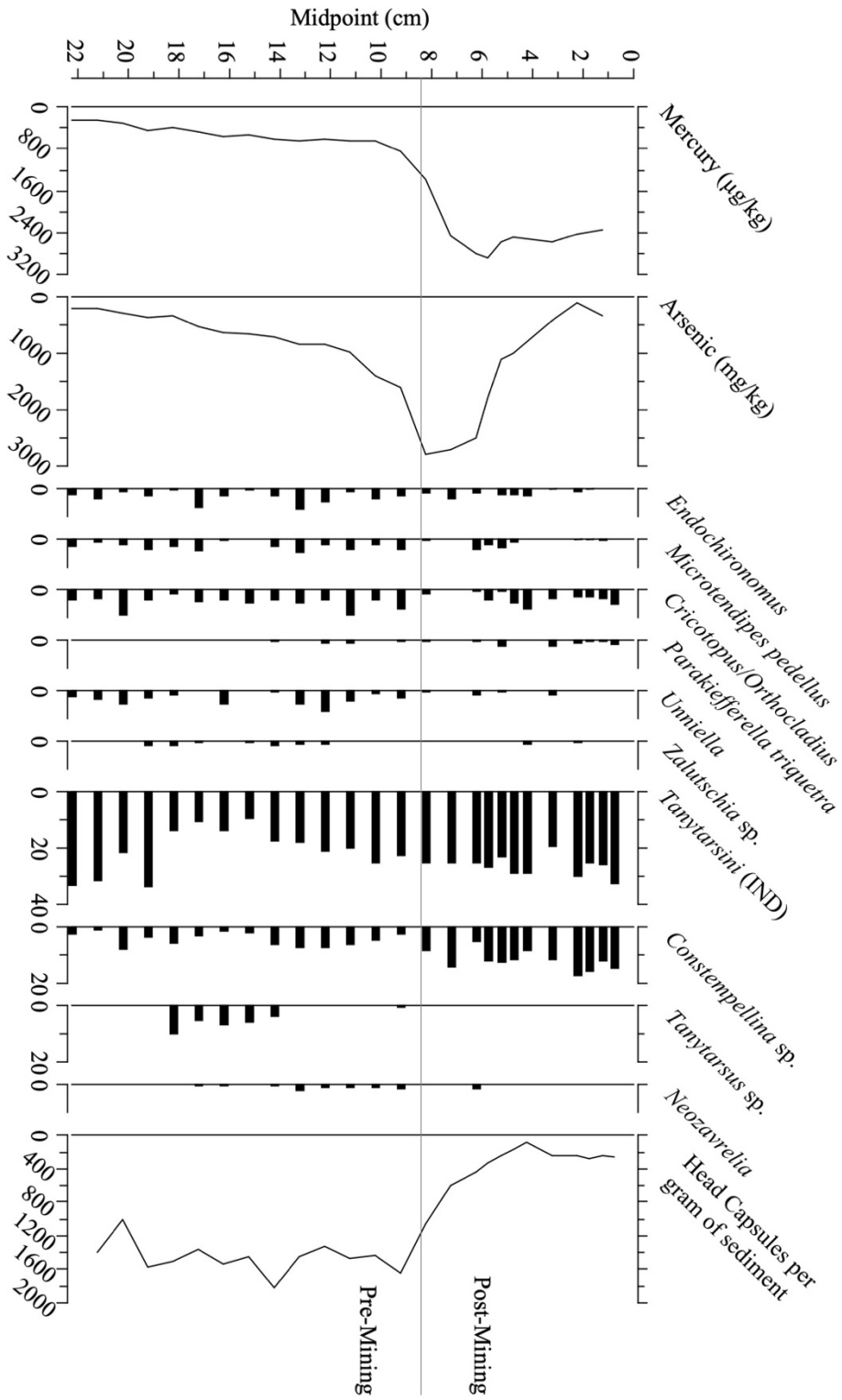

Figure 2. Stratigraphy of Gegogan Lake sedimentary measures showing total mercury and total arsenic concentrations, chironomid relative abundances of indicator taxa, and the number of head capsules per gram of sediment. Mining periods are indicated.

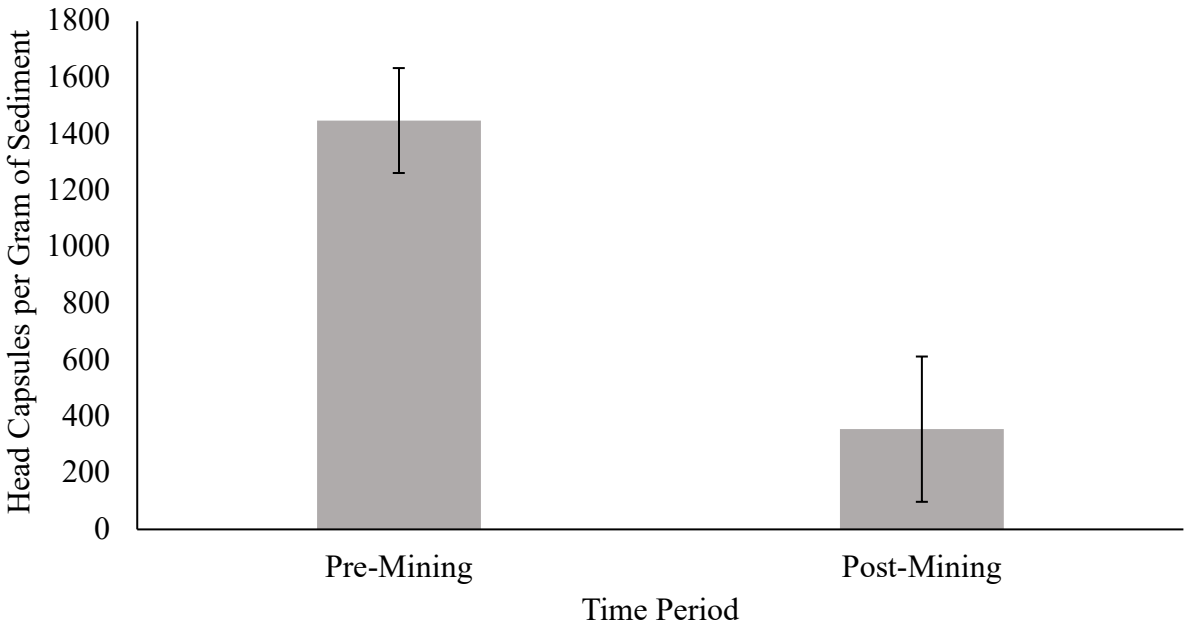

Figure 3. Average chironomid head capsules per gram of sediment at Gegogan Lake during the pre-mining (n = 14) and post-mining (n = 12) time periods.

Appendix

Raw counts of chironomid head capsules at Gegogan Lake

Depth (cm)	*Tanypodinae* (IND)	*Procladius* sp.	*Pentaneurini-Ablabesmyia*	*Labrundinia* sp.	*Chironomini.* (IND)	*Chironomus anthracinus*	*C. plumosus*	*Cladopelma. lateralis*	*Cryptochironomus*	*Dicrotendipes nervosus*	*Endochironomus*	*Glyptotendipes pallens*	*G. barbipes*	*Paracladoplema*	*Microtendipes pedellus*	*M. rydalensis*	*Nilothauma*	*Pagastiella*
0.5-1	0	3	1	0	0	0	0	1	1	3	0	0	0	1	0	0	0	0
1-1.5	0	8	4	1	0	0	0	0	0	1	0	1	0	0	1	0	0	0
1.5-2	0	7	1	0	0	0	0	7	0	6	1	2	0	5	1	0	0	2
2-2.5	0	6	3	0	0	0	0	3	1	1	4	3	0	3	1	0	0	3
3-3.5	0	9	2	0	0	0	0	2	2	6	1	0	1	2	0	0	0	0
4-4.5	0	1	4	0	0	0	0	3	2	2	2	2	0	0	0	0	0	2
4.5-5	0	7	2	0	0	1	0	0	0	3	2	1	0	0	1	0	0	1
5-5.5	0	7	2	1	0	0	0	3	1	0	2	0	0	0	3	0	0	0
5.5-6	0	8	1	0	0	0	0	4	0	2	0	2	0	0	2	0	0	1
6-6.5	0	8	6	0	0	0	0	3	0	3	2	0	0	0	4	0	0	3
7-7.5	0	3	4	0	0	0	0	0	0	1	3	0	0	0	0	1	0	1
8-8.5	0	7	3	0	0	0	0	5	0	0	2	0	0	1	1	0	1	1
9-9.5	0	5	5	1	0	0	0	0	1	5	3	0	0	0	4	0	0	0
10-10.5	0	0	2	1	1	0	0	4	0	2	3	2	0	0	2	0	0	0
11-11.5	0	1	0	1	0	0	0	1	0	1	1	0	1	0	3	0	0	0
12-12.5	0	3	2	0	0	0	0	0	0	1	4	0	0	0	2	0	0	1
13-13.5	0	2	6	0	0	0	0	1	0	2	6	0	0	0	4	0	0	0
14-14.5	0	4	3	0	0	1	0	3	1	5	3	1	0	0	3	0	0	1
15-15.5	0	5	0	0	0	2	0	0	1	4	1	2	1	0	0	0	0	0
16-16.5	0	10	0	0	0	0	0	4	0	0	3	2	0	2	1	0	0	2
17-17.5	0	9	3	0	0	0	0	0	0	2	6	3	0	0	4	0	0	0
18-18.5	3	7	2	0	1	0	1	2	0	2	1	0	0	0	3	0	2	1
19-19.5	0	2	3	0	0	0	0	2	0	1	3	0	0	0	4	0	0	1
20-20.5	0	7	0	1	0	2	0	2	1	3	1	1	1	0	2	1	0	0
21-21.5	0	15	7	1	0	2	0	1	0	3	6	0	0	0	2	0	0	0
22-22.5	0	25	10	2	0	2	0	3	3	10	7	4	0	2	9	0	0	3

Depth (cm)	*Parachironomus*	*Paratendipes*	*Phaenopsectra*	*Polypedilum nubifer*	*Sergentia coracina*	*Stenochironomus*	*Tribelos*	*Zavreliella*	*Orthocladiinae* (IND)	*Corynoneura arctica*	*Corynoneura/ Thienemanniella*	*Cricotopus/ Orthocladius*	*Heterotanytarsus*	*Heterotrissocladius*	*H. marcidus*	*H. grimshawi*	*Limnophyes/ Paralimnophyes*	*Metriocnemus fuscipes*
0.5-1	0	0	0	3	0	0	0	0	1	1	0	5	3	0	1	0	0	0
1-1.5	0	0	0	6	0	0	0	0	2	0	0	3	2	0	2	0	0	0
1.5-2	1	0	0	4	0	1	0	2	2	3	0	8	11	0	7	0	0	0
2-2.5	0	0	0	5	0	0	0	2	4	5	0	7	15	0	7	0	0	0
3-3.5	1	0	0	10	0	0	0	0	7	2	2	7	11	0	6	0	2	0
4-4.5	0	0	0	1	0	0	0	0	2	1	0	5	0	0	2	0	0	0
4.5-5	0	0	0	4	0	0	0	0	0	2	0	4	3	0	3	0	0	0
5-5.5	0	0	0	5	0	0	0	1	1	1	3	1	3	0	2	0	2	0
5.5-6	0	0	0	1	0	0	0	0	2	1	1	3	2	0	1	0	1	0
6-6.5	0	1	0	5	0	0	0	1	2	0	1	1	2	0	4	0	1	0
7-7.5	0	0	0	1	0	0	0	0	1	0	2	0	2	0	0	0	2	0
8-8.5	0	0	0	2	1	0	0	1	0	0	2	2	4	0	4	0	1	0
9-9.5	0	0	0	1	0	0	0	1	2	0	2	7	4	0	2	0	0	0
10-10.5	0	0	1	8	0	0	0	2	2	0	0	3	2	0	3	0	0	0
11-11.5	0	0	0	2	3	0	0	1	2	0	1	7	4	0	4	0	1	0
12-12.5	0	0	0	5	0	0	0	1	0	0	0	3	3	0	3	0	0	0
13-13.5	1	0	2	1	1	0	0	1	3	1	1	4	2	0	1	0	0	0
14-14.5	0	0	1	1	0	0	1	0	1	0	0	4	3	1	5	0	0	0
15-15.5	1	0	0	4	0	0	0	0	1	0	1	4	2	1	4	0	0	0
16-16.5	0	0	0	0	0	0	0	1	2	0	2	4	2	0	3	0	0	0
17-17.5	0	0	0	1	0	0	0	0	2	2	0	4	1	0	3	0	0	0
18-18.5	0	0	0	3	1	1	1	1	3	1	1	2	2	0	1	0	1	0
19-19.5	0	0	0	4	0	0	0	2	5	1	1	4	3	0	3	1	0	0
20-20.5	0	0	1	0	0	0	0	0	1	0	0	7	1	0	1	0	0	1
21-21.5	0	0	0	7	0	0	0	1	6	0	5	5	2	0	6	0	0	0
22-22.5	0	0	0	13	0	0	0	2	4	0	5	12	7	0	5	0	1	0

Depth (cm)	*Nanocladius*	*Eukiefferiella claripennis*	*Parakiefferiella*	*P. triquetra*	*P. type A*	*Parachaetocladius*	*Paracladius*	*Paraphaenocladius/ Paramet.*	*Psectrocladius sordidellus*	*P. calcaratus*	*P. barbatipes*	*P. psilopterus*	*Pseudosmittia*	*Pseudorthocladius*	*Rheocricotopus*	*Smittia*	Species 416	*Synorthocladius*
0.5-1	0	0	0	2	3	0	0	0	3	0	2	0	0	0	0	0	0	0
1-1.5	0	0	0	1	1	0	0	0	1	0	1	0	0	0	0	0	0	0
1.5-2	0	0	0	3	5	0	1	0	3	0	6	0	1	0	0	0	0	0
2-2.5	0	0	0	3	0	0	0	0	7	0	2	0	0	0	0	0	0	0
3-3.5	4	0	0	5	0	2	0	0	4	0	1	0	0	0	0	0	1	0
4-4.5	1	0	0	0	0	0	0	0	2	0	0	0	0	1	1	0	0	0
4.5-5	0	0	0	0	1	0	0	0	3	0	1	0	0	0	0	0	0	1
5-5.5	1	0	0	2	1	0	0	0	4	0	0	0	1	0	0	0	1	0
5.5-6	0	0	0	0	0	0	0	1	2	0	0	0	0	0	0	0	0	0
6-6.5	0	0	0	1	1	0	0	0	1	0	0	0	3	0	0	0	0	1
7-7.5	0	0	0	0	1	1	0	0	3	0	0	1	0	0	0	0	0	1
8-8.5	1	0	0	1	0	0	0	0	3	0	0	0	0	0	0	0	0	0
9-9.5	1	0	0	1	0	1	0	0	4	0	0	0	1	0	1	1	0	0
10-10.5	1	1	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	1
11-11.5	0	0	0	1	0	0	0	0	1	1	0	1	0	0	0	0	0	1
12-12.5	2	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0
13-13.5	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1
14-14.5	2	0	0	1	0	0	0	0	2	2	0	0	2	0	0	0	0	0
15-15.5	1	0	0	0	0	2	0	0	3	0	0	0	0	0	0	0	0	2
16-16.5	1	0	0	0	0	1	0	0	3	0	0	1	0	0	0	0	0	0
17-17.5	1	0	0	0	0	0	0	0	3	0	0	0	0	0	1	0	0	0
18-18.5	1	0	1	0	1	0	0	1	3	0	0	0	1	0	0	0	0	1
19-19.5	4	0	0	0	2	0	0	0	3	1	2	0	0	0	1	0	1	1
20-20.5	0	0	0	0	0	0	0	0	0	0	2	0	0	0	2	0	1	2
21-21.5	0	0	0	0	0	1	0	0	2	0	5	0	0	0	1	0	0	0
22-22.5	1	1	0	0	4	3	0	0	1	0	5	0	0	0	0	0	0	2

Depth (cm)	*Unniella*	*Zalutschia* sp.	*Z. zalutschiacola*	*Tanytarsini* (IND)	*Pseudochironomus* sp.	*Cladotanytarsus* sp.	*Micropsectra* sp.	*Neozavrelia*	*Constempellina* sp.	*Stempellina*	*Stempellinella/ Zavrelia*	*Tanytarsus* sp.	*T. chinyensis*	*T. lactescens*	*Monodiamesa*	*Chaoboridae*
0.5-1	0	0	3	28	0	0	0	0	13	1	0	0	7	0	0	0
1-1.5	0	0	3	21	0	0	2	0	10	4	0	0	7	0	0	0
1.5-2	0	0	12	69	0	0	12	0	43	3	5	0	33	0	0	1
2-2.5	0	2	4	67	0	2	3	0	39	0	4	0	19	0	0	0
3-3.5	4	0	4	38	0	0	7	0	23	0	4	0	24	4	0	0
4-4.5	0	1	2	20	0	1	1	0	6	1	0	0	3	1	0	0
4.5-5	0	0	1	22	0	0	0	0	9	0	0	0	2	1	0	0
5-5.5	1	0	2	20	0	0	1	0	11	0	1	0	1	0	0	0
5.5-6	0	0	5	19	0	0	1	0	9	0	2	0	1	0	0	0
6-6.5	2	0	1	23	0	0	0	2	5	0	2	0	3	0	0	1
7-7.5	0	0	5	19	0	0	6	0	11	0	1	0	3	2	0	0
8-8.5	1	0	3	23	0	1	3	0	8	0	4	0	3	0	0	0
9-9.5	3	0	5	22	0	0	2	2	3	0	2	1	3	1	0	0
10-10.5	1	0	1	19	0	2	1	1	4	1	2	0	2	0	0	0
11-11.5	3	0	6	15	0	1	0	1	5	0	0	0	5	1	0	0
12-12.5	6	1	5	16	0	0	1	1	6	0	5	0	1	0	0	0
13-13.5	4	1	1	14	0	0	1	2	6	0	2	0	5	1	0	1
14-14.5	1	2	4	18	1	2	4	1	7	0	1	4	4	2	0	1
15-15.5	0	1	4	8	0	1	6	0	2	0	2	5	8	2	0	0
16-16.5	5	0	3	13	1	1	12	1	2	0	1	7	4	0	0	2
17-17.5	0	1	4	9	0	0	13	1	3	0	1	5	3	0	0	1
18-18.5	2	2	5	14	0	0	2	0	6	0	1	10	9	0	1	0
19-19.5	3	2	4	34	0	0	0	0	4	1	0	0	1	0	0	0
20-20.5	4	0	4	16	0	0	1	0	6	1	0	0	1	0	0	1
21-21.5	5	0	3	43	0	0	2	0	2	1	3	0	4	0	0	1
22-22.5	7	0	18	94	0	1	2	0	9	0	5	0	3	0	0	1