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ASSESSING BENTHIC RECOVERY IN THE TORCH LAKE AOC

By

Anya Clare Leach

A THESIS

Submitted in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

In Environmental Engineering

MICHIGAN TECHNOLOGICAL UNIVERSITY

2019

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This thesis has been approved in partial fulfillment of the requirements for the Degree of
MASTER OF SCIENCE in Environmental Engineering.

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Acknowledgements

Many thanks to my advisor, Dr. McDonald, and my committee members, Dr. Urban and Dr. Kerfoot, for their valuable time and input to this thesis. I have been incredibly fortunate to work on this project with Dr. McDonald and I am sincerely grateful for the opportunity and mentorship.

The PI for the Torch Lake AOC Benthos Monitoring 2018 study was Cory McDonald.

Funding for this study was provided by the Torch Lake Public Action Council (TLPAC) and the Michigan Office of the Great Lakes Areas of Concern Program.

Boat access was courtesy of Great Lakes Research Center and Michigan Tech's Biology Department.

Pilots for field sampling were Stephen Roblee and Dayton Pax.

Laboratory assistants were Brett Cianek and Kenny Larsen.

Geospatial Research Facility workspace was courtesy of Don Lafreniere and Ryan Williams.

The bathymetry maps were originally digitized by Foad Yusef.

Abstract

Torch Lake, located in Michigan's Upper Peninsula, was a site of copper ore processing from the 1860s through the 1960s. During this time, various mining by-products were deposited into the lake and along its shoreline, including mine tailings and associated heavy metals contaminants (e.g., copper). It was anticipated that natural sediment deposition would cover and attenuate toxic sediments over time, but a comprehensive assessment of the benthos has not been performed since a baseline survey conducted in 1999 and 2000. This study was conducted in 2018 to evaluate the current benthic community status as well as sediment composition and metal concentrations, in order to evaluate natural recovery and spatial drivers of community density.

In comparison with the baseline, an overall increase in macroinvertebrate densities was observed and recovery rates are spatially variable. Macroinvertebrate densities were depth-dependent and exhibited a northeast gradient across the nearshore of the lake. Greatest densities occurred along the eastern shoreline and near the main tributary, while the lowest densities may be influenced by proximity to historical mining activity along the western shoreline. The northern basins, near the main tributary input, were significantly greater in macroinvertebrate density from nearshore sites in the lake's main basin. Habitat quality in the northern basins sites was highly heterogeneous and drivers of macroinvertebrate density in this region include currents and increased sediment deposition zones.

The relative effects of mine tailings and metal toxicity on macroinvertebrate density are modulated by spatially dependent physical and chemical properties in the nearshore, as well as the depth dependence of sediment composition between the offshore and nearshore. Considering habitat variability in future sampling efforts would facilitate a more complete assessment of environmental drivers of benthic conditions and recovery. In addition, an appropriate reference site is needed to more effectively gauge the degree of benthic degradation and to provide future restoration targets for the AOC and potential BUI removal.

1 Introduction

1.1 The Legacy of Copper Mining

Torch Lake, located in Michigan's Keweenaw Peninsula, was a hub of industrial copper ore processing between the 1860s and late 1960s. More than half of the 5 million tons of native (elemental) copper extracted from the Keweenaw Peninsula during this time was processed along Torch Lake's western shoreline (US EPA, 2001).

Figure 1.1 Map of the Torch Lake site and surrounding area

Copper extraction involved crushing (stamping) mined bedrock and employing gravity separation in a fluid to sort denser copper ore from the lighter crushed bedrock. Separated copper ore went to smelters, while the mine tailings and contaminated wastewaters were deposited into Torch Lake and other sites throughout the Keweenaw Peninsula (Urban et al., 2018). Approximately 200 million tons of mine tailings (stamp

sands and slime clays, common names for the coarse and fine mine tailings, respectively) were deposited into Torch Lake and along its western shoreline, forming large piles that protruded from the water surface (MNDR, 1987, Kerfoot et al., 1994). Tailing deposition is estimated to have filled 51 percent of the lake's original volume (Urban et al., 2018), and significantly altered the lake's bathymetry as illustrated in Figure 1.2. Net positive bathymetry changes are focused around the stamp sand piles along the western shoreline and into the northern basins.

Figure 1.2 Change in Torch Lake's bathymetry from pre- to post-mining eras (1865 to 2004)

In the 1910s, copper reclamation technology improved, prompting the dredging and reprocessing of submerged tailings in Torch Lake. By the 1930s, milling activity was predominately focused on dredging and copper ore reclamation (Urban et al., 2018). Dredging increased turbidity (Figure 1.3) and redistributed new and deposited byproducts throughout the lake, further altering bathymetry. Stamp sand piles were highly erodible

via wind or water, also contributing to tailing deposition to the lake's sediments (Baker, 2007).

Figure 1.3 Aerial imagery of increased turbidity from dredged sediment and mining byproducts in Torch Lake (1951). Photo courtesy of Michigan Technological University Archives.

Coinciding with the Clean Air Act (1970) and Clean Water Act (1972), environmental investigations were conducted to evaluate effect of mining pollution on fish, sediment, and water quality in Torch Lake beginning in the early 1970s (Urban et al., 2018). Surveys found Torch Lake sediments contained high concentrations of heavy metals (copper, mercury, etc.). In addition, the faunal benthos (bottom-dwelling invertebrate community) was determined to be low in density, diversity, and biomass based on several benthic surveys and sediment toxicity studies (Massey, 1970; Evans, 1973; Kraft & Sypniewski, 1981; Malueg et al., 1984; MDNR, 1987). Ecological disturbances in the benthos have widespread impacts that propagate through the food web, and slow organic matter processing and nutrient cycling (Urban et al., 2018; Covich et al., 1999).

In 1985, the US EPA and the International Joint Commission's (IJC) Water Quality Board designated Torch Lake as a Great Lakes Area of Concern (AOC) under the International Water Quality Agreement. Three Beneficial Use Impairments (BUIs) were

assigned to the Torch Lake AOC: fish tumors or other deformities¹, restrictions on fish and wildlife consumption, and degradation of benthos. In 1986, Torch Lake and surrounding areas were also included on the National Priorities List as the Torch Lake Superfund Site, composed of three operable units (OU). Sediments and submerged tailings within the Torch Lake, the northern portion of Portage Lake, the Portage Lake Canal, Keweenaw Waterway, the north entry to Lake Superior, Boston Pond, Calumet Lake, and other waterbodies within the Keweenaw drainage basin were included in OU II of the Superfund Site.

Remediation actions were often joint efforts under AOC and Superfund jurisdictions, focusing primarily on ecological disturbances (Urban et al., 2018). In 1987, the Remedial Action Plan (RAP) for Torch Lake AOC did not propose a remedial action for the degraded benthos due to the immense areal coverage and volume of tailing deposits in the lake. Natural attenuation was considered the “only feasible course of action” for the degraded benthos (MDNR, 1987). In 1994, the Superfund Record of Decision (ROD) for lake sediment in OU II also selected no remedial action for lake sediments in the various waterbodies included in the Site (US EPA, 1994). Natural sediment transportation and deposition was anticipated to cover and detoxify metal-rich sediments over time, as well as provide a habitable zone for the faunal and floral benthic communities. Both remedy decisions relied on remedial soil and vegetation cover of the shoreline tailings, selected in the 1992 ROD for OU I and III, to limit erosion and further deposition of tailings into the lake (US EPA, 1992).

In addition, the 1994 ROD established long-term monitoring to evaluate the progress of natural recovery in Torch Lake, as it was anticipated to be more degraded than other OU II waterbodies. It was assumed that any recovery observed in Torch Lake could be extrapolated to indicate equal or better conditions in other waterbodies within the Superfund Site (US EPA, 2001).

¹ The BUI for fish tumors and other deformities was removed in 2007.

The US EPA conducted the first monitoring survey, establishing a baseline condition for Torch Lake in 1999 and 2000 (US EPA, 2001). Ten offshore sampling stations were established in 1999, followed by 20 more nearshore sites in 2000. The baseline survey reported physical, chemical, and biological conditions within Torch Lake as well as chemical conditions of groundwater in the surrounding area. The baseline survey also outlined surveying methods for long-term monitoring to facilitate effective condition comparisons as well as to assess temporal changes in the lake. A few highlighted objectives from the baseline survey included measuring the distribution and concentration of metals in surface water and sediments, analyzing sediments for particle size, and evaluating the condition of the faunal benthic (macroinvertebrate) community. In 2004, the MDEQ completed another monitoring assessment of natural recovery in Torch Lake (MACTEC, 2008), including surveys of the benthic macroinvertebrate community, sedimentation rate, and sediment quality. Benthic community resampling occurred at 8 of the 30 baseline sites to assess changes in the density and diversity of the benthic community. However, large discrepancies in water depth limited direct comparison of benthic conditions to the baseline survey (MACTEC, 2008).

1.2 Benthic Macroinvertebrates

Benthic macroinvertebrates are a key component of lacustrine ecosystems, facilitating nutrient cycling and energy flow through food webs (Covich et al., 1991; Jonasson, 2004). Different macroinvertebrate species perform different ecological functions, such as accelerating detrital decomposition, releasing bound nutrients, and facilitating nutrient replenishment into the water column via feeding, excretion, and burrowing. Therefore, the diversity within a community is often more critical than quantity of organisms (Covich et al., 1991). Changes in the distribution or abundance of species can cause imbalances in the community structure and limit ecological functions.

Ecoregion, watershed characteristics, and lake morphometry affect benthic community structure, but the local habitat quality has greater influence on taxonomic and functional composition (Johnson & Goedkoop, 2002; Free et al., 2009). Habitat

characteristics include the physical and chemical properties of sediment and overlying water column, such as temperature, pH, alkalinity, dissolved oxygen, conductivity, sediment chemistry, and substrate composition (Jonasson, 2004; Free et al., 2009). Macroinvertebrate abundance is highly sensitive to sediment chemistry (Free et al., 2009), whereas in ecologically disturbed areas, physical habitat quality (sediment particle size or availability of food) can be a primary limitation on benthic communities (Heatherly et al., 2005; De Haas et al., 2006). Other limnological properties that influence benthic communities include wind-induced currents and seasonal turnover, that are responsible for circulating food and oxygen that in turn influence community structure in the various zones (Jonasson, 2004).

Community structure is highly dependent on water depth within a lake. Aquatic habitats in stratified lakes are often divided into littoral, sublittoral, and profundal zones, summarized roughly by depth relative to the water surface and thermocline (Mandaville, 2002; Jonasson, 2004). Generally, the macroinvertebrate community is more dense and diverse in the littoral zone. Sediment heterogeneity and macrophyte colonization create microhabitat niches and alter food availability, which contribute to increased diversity. In the littoral zone, many biotic and abiotic characteristics in benthic habitats are influenced by water depth, such as wave action (hydraulic stress), habitat stability, sediment particle size, light attenuation, algal productivity, and temperature (Baumgärtner et al., 2008). In contrast, the profundal zone is more homogeneous in temperature, sediment composition, and macrophyte diversity, thus limiting microhabitat variety (Mandaville, 2002). In addition, lower temperature, oxygen, and food availability restricts the types of species and inhibits colonization in the profundal zone (Jonasson, 2004).

Seasonality of abiotic and biotic factors, primarily plant growth, yield seasonal variation in benthic community. In the littoral zone, summer months of high plant growth coincide with dominant herbivore populations, which are succeeded by carnivores later in the year (Jansson, 2004). The profundal zone is dominated by detritivores that rely on organic material produced in the littoral zone. Sedimentation rates in the profundal zone peak during spring/fall turnover (James & Barko, 1993).

Benthos Monitoring and Biotic Metrics

Macroinvertebrate communities are commonly used as indicators of either ecological health or water quality, or to monitor disturbances in ecosystems through static measures or temporal trends in the density, diversity (composition), and abundance of specified taxonomic groups (New, 1995; Bailey et al., 1995). Macroinvertebrates are relatively sedentary and can provide localized ecological assessment (Goodyear & McNeill, 1999; Beck & Hatch, 2009). Early benthos monitoring focused on the effects of nutrient pollution in rivers and lakes. With the concerns rising over more severe chemical contaminants in recent years, monitoring studies have transitioned to the use of biotic metrics to assess changes in community diversity (Cairns & Pratt, 1993).

Biotic metrics relying on taxa richness and tolerance (organism sensitivity) are often more robust than measures of density (Johnson, 1998; Cairns & Pratt, 1993). In addition, interpretation of multiple metrics is preferable to reliance on a single metric. The most effective diversity metrics for assessing ecological disturbances include taxa richness, community indices (including tolerance), and functional groups (Resh & Jackson, 1993; Cairns & Pratt, 1993).

Taxa richness refers to the number of distinct taxa, a taxonomic group of any kind. Species richness is assessed most commonly, but higher taxa or a specific indicator taxon can also be used. Greater richness corresponds to greater diversity. Disturbances to a system will tend to decrease the taxa richness (Resh & Jackson, 1993; Mandaville, 2002).

Community indices incorporate taxa richness and abundance (counts of individuals within a specified taxon) for a statistical summary of community diversity as ranked by numerical “scores”, and often weighted by abundance (Resh & Jackson, 1993; Mandaville, 2002). Some simply rate the diversity of a community (e.g., Shannon Index, H'), while others incorporate unique characteristics of taxa, such as organism tolerance/sensitivity to nutrient pollution (e.g., Hilsenhoff Biotic Index, HBI).

Functional groups (also referred to as trophic groups or feeding guilds) indicate the feeding mechanisms and food preference of macroinvertebrates (Goodyear & McNeil, 1999; Cummins, 1973). The general functional groups are filterers, collectors, scrapers, shredders, predators, and parasites. Functional group dominance reflects habitat quality, food availability, and the functional capability of the faunal benthos.

Most biotic metrics were designed for water quality evaluations in rivers and streams but are often altered for application to other systems (Beck & Hatch, 2009). Similarly, those metrics can be altered to reflect the characteristics of the community or the degree of ecological disturbance (Back & Hatch, 2008), but a metric's sensitivity and robustness to ecological disturbances should be validated before comparing communities in different systems (Resh & Jackson, 1993).

1.3 Sediment Toxicity

In the late 1960s and early 1970s, recorded surveys on Torch Lake's faunal benthos reported severely degraded communities with sparse organism counts and minimal taxa variety (Yanko, 1969; Massey, 1970; Wright et al., 1973). Lower water column toxicity and reduced turbidity in recent years have contributed to an increase of pelagic species, yet benthic invertebrate densities are still low (Kerfoot et al., 2007).

Numerous studies have linked sediment toxicity to degraded benthic communities in Torch Lake and connecting waterbodies in the Keweenaw Waterway (Evans, 1973; Sypniewski, 1977; Kraft & Sypniewski, 1981; Malueg et al., 1984; West et al., 1993; Cusack & Mihelcic, 1999; US EPA, 2001). Consequently, Torch Lake's degraded benthos is attributed to the toxicity of metal-rich sediments, particularly high copper concentrations associated with mine tailings (Urban et al., 2018).

Toxicity can occur by direct exposure to *in situ* sediments and pore water or transfer from the water column, and through consumption of contaminated organisms (MacDonald & Ingersoll, 2002). Direct exposure of contaminants is commonly accepted

as the main route of contaminant exposure for benthic macroinvertebrates (Adams et al., 1992; Goodyear & McNeill, 1999).

Sediment toxicity is influenced by organism-specific factors modifying toxicity exposure, such as different life histories, feeding mechanisms, and varying sensitivity (West et al., 1993). Macroinvertebrate sensitivity to metal contaminants is influenced by food availability, morphology, functional group, and competition, as well as physical and chemical processes modulating the bioavailability of metals (Buikema & Benfield, 1979; Jeong et al., 1999). Thus, metal concentrations alone are not an effective proxy for sediment toxicity.

1.4 Limitations of Natural Recovery

Natural attenuation is facilitated by the burial of toxic sediments, where the rate of sediment deposition exceeds the rate of sediment scouring or resuspension (Magar, 2001). Natural sedimentation consists of particulate and organic material inputs via watershed reaches and internal production in littoral zones (Baker, 2007). Suspended material is deposited, often in an alluvial fan, as tributary water velocity slows upon entering a lake (Coduto et al., 2010). In offshore regions of the lake, sediment focusing contributes more to sediment accumulation than seasonal mixing in dimictic lakes (James & Barko, 1993).

Sediment transport mechanics, such as erosion, basin scouring, and focusing, may continue to contribute to spatial variability of tailings and associated solid-phase metal concentrations (Baker, 2007; McDonald et al., 2010; Mandelia et al., 2016; Urban et al., 2018). The National Resources Conservation Service (NRCS) estimated that the rate of exposed stamp sands piles erosion into Torch Lake was approximately 0.1 inch per year (Baker, 2007). However, previous sediment trap studies concluded erosion from stamp sand piles are not the primary source of copper in sediments (McDonald et al., 2010, Kerfoot et al., 2016).

Analysis of sediment cores, comprised of post-mining sediment accumulation underlain with mine tailings (Figure 1.4), have shown upward copper concentration gradients between layers (McDonald et al., 2010). Elevated copper concentrations in post-mining sediment have been observed in various Torch Lake sediment core surveys (Cusack & Mihelcic, 1999; Long et al., 2002; Kerfoot et al., 2007; McDonald et al., 2010). This is attributed to continued mobilization of copper from tailings to the water column via pore water transfers (Jeong et al., 1999; McDonald et al., 2010).

Figure 1.4 A sediment core with the distinct boundary between the post-mining sediment and underlying mining era sediment (Original image taken from McDonald, 2018).

The copper budget for Torch Lake indicates dissolved copper concentrations in surface water must be derived from legacy contamination, rather than external contribution such as runoff or precipitation (Warburton, 1987). Dissolved copper concentrations in surface water are controlled by chemical processes such as precipitation, dissolution, adsorption, complexation/methylation, diffusion of pore water from sediments and stamp sands piles (Jeong et al., 1999; Urban et al., 2018).

Natural attenuation relies on abiotic and biotic processes facilitating immobilization of solid-phase metals in sediments (Mulligan & Yong, 2003; Jeong et al., 1999), but processes facilitating immobilization of metals are often reversible, resulting in bioaccumulation and legacy water quality issues (Mulligan & Yong, 2003; Medine & McCutcheon, 1989).

Model forecasts estimate over 100 years of sediment accumulation will not significantly reduce copper concentration in sediments through burial mechanisms (McDonald & Urban, 2004). Despite the evidence against natural attenuation's effectiveness in reducing metal concentrations in sediment and the relationship between metal concentration and sediment toxicity, the status of the benthic community and spatial drivers of benthic recovery were, prior to this study, uncertain.

1.5 Objectives

- 1) To conduct an updated survey of the benthic community to assess any evidence of natural recovery in the lake through a statistical comparison of density and diversity with the baseline survey.
- 2) To survey the northern basins to assess benthic communities in an isolated region of intensified sedimentation and to compare with macroinvertebrate densities in the rest of the lake.
- 3) To evaluate spatial drivers of benthic community density across all sample sites to provide insight to necessary conditions for supporting benthos recovery.

Based on historic densities (USEPA, 2001; MACTEC, 2008), community density was anticipated to remain low with limited diversity, potentially insufficient for an effective statistical evaluation. Flow from the northern basins is hydrologically impeded by a large stamp sand pile, which restricts the depositional area of nearby tributaries resulting in increased sedimentation. Natural recovery was anticipated to be driven by natural sediment accumulation; thus conditions for recovery in benthic communities may be related to the thickness of the post-mining, organic-rich sediment layer (Cusack & Mihelcic, 1999; Kerfoot et al., 2007).

2 Methods

2.1 Site Description

Torch Lake is an oligotrophic, dimictic lake with a surface area of 11 km² and a maximum depth of 38 m. The primary tributary is the Trap Rock River, which discharges into the northern portion of the lake (Figure 2.1). The Torch Lake watershed (approximately 94 km²) includes numerous smaller tributaries, such as McCallum Creek, Sawmill Creek, and Dover Creek. The lake drains via a channel into Portage Lake. The retention time of Torch Lake is approximately one year (Warburton, 1987).

Figure 2.1 Hydrology of Torch Lake

The Torch Lake drainage basin is comprised of the Torch Lake and Trap Rock River subwatersheds, under the USDA hydrologic unit coding system (NRCS, 2007). The Trap Rock River subwatershed has an area of 111 km², comprising over half of the Torch Lake

drainage basin (Warburton, 1987). Both subwatersheds are components of the Portage Lake watershed which ultimately drains into Lake Superior (Figure 1.1).

Land cover in the Torch Lake drainage basin is predominately forested (53%), with smaller areal coverage by wetlands (18%) and developed areas (10%) (Table A.4.1). The western shoreline is mostly developed, lined with the small communities of Lake Linden, Hubbell/Tamarack City, and Mason (Figure 2.2). Beneficial uses of Torch Lake include fishing, boating, and swimming as well as treated municipal waste assimilation (Urban et al., 2018). Wastewater treatment lagoons are located on the Lake Linden and Hubbell/Tamarack City stamp sand piles (Figure 2.2).

Figure 2.2 Land cover by category surrounding the Torch Lake site (NLCD, 2016)

The stamp sand piles surrounding Torch Lake are covered with vegetation varying from scrub to wetlands. Vegetative growth has increased in recent years since the initial remedial coverage in 1999 – 2001 but is not indicated in recently released National Land

Cover Database (NLCD) 2016 data (<https://www.mrlc.gov/data/nlcd-2016-land-cover-conus>). Instead, the stamp sand piles are designated as barren land, as illustrated in Figure 2.2. This discrepancy prevents evaluation of the shoreline cover in relation to sediment quality or the benthos using NLCD.

The 2004 bathymetry of Torch Lake is shown in Figure 2.3. The bathymetry of the lake does vary in depth and slope, particularly along the western shoreline. A more gradual descent is observed along the eastern shoreline towards the lake's outlet. The northeast basin is approximately 6m deeper than the northwest basin, which is not evident from Figure 2.3.

Figure 2.3 Bathymetry (left) and basin slope (right) of Torch Lake (2004)

2.2 Sampling

Sampling and analytical procedures from the baseline survey (US EPA 2001) were maintained to limit the influence of methodological differences on comparisons. Ten new sites located in the northern basins were added to the 30 original baseline sites. Sampling referenced coordinates, water depth, and site-specific identification codes. Site coding reflects the year of initial sampling at each location: 99**, 00**, 18**, respectively (Figure 2.4)². The 99**, 00**, and 18** sites will be referenced as offshore, nearshore, and northern sites, respectively. Results regarding sampling locations and benthic communities from the baseline survey (US EPA, 2001) are summarized in Table A.2.1 through Table A.2.6 in the appendix.

Figure 2.4 Sampling sites with site categorization

² Site codes are abbreviated versions of the station location code. The numerical association between site code and station location does vary for the 99** sites. Consequently, the site code is preferentially used in this study.

Due to the expected relationship between water depth and benthic community structure (Baumgärtner et al., 2008; Jonasson, 2004), sampling at similar water depths was prioritized over coordinate location at baseline sites. If water depth at a site location differed from the sample plan, location was adjusted perpendicularly to the shoreline until water depth agreed with baseline data.

Initial sampling for all 40 sites occurred during August 6 – August 15, 2018. A calibration error in the sonar equipment was subsequently detected, yielding a 2.5m overestimation in water depth. The benthic community was resampled in the nearshore and northern sites on September 3, 2018. As the calibration error was less significant at greater depths, the offshore samples were not resampled. In addition, sediment characteristics were assumed to be similar within a general area per site location, based information from both the sediment characterization survey and sediment reports in the baseline survey, as well as the relative distance of sampling location perturbation compared to distances between sites. Therefore, sediments were not resampled for metals, particle size, and organic content.

Following the baseline methodology, offshore grabs used a full Ponar dredge (0.052 m² sampling area), while a petite Ponar dredge (0.023 m² sampling area) was used for nearshore and northern grabs. Sampling was conducted from MTU's R/V Aggasiz, R/V Polar, and a jon boat, depending on water depth and accessibility of site.

Post-mining sediment was identified as brown silt, based on previous sediment studies (Kerfoot et al., 2007; McDonald et al., 2010). A clear boundary between post-mining sediment and slime clay layers was observed at certain sites, depicted in Figure 1.4, similar to previous sediment core observations (Cusack & Mihelcic, 1999; Kerfoot et al., 2007; Alexander, 2008; McDonald et al., 2010). In such cases, only the top layer of post-mining sediment was collected. Metal subsampling avoided slime clay contamination to the extent possible, so that only post-mining sediment (ie. the natural sediment cap) was analyzed for metal content for evaluation of natural remediation. Prior to disturbing the grab with subsampling, the thickness of the post-mining sediment layer

(Figure 1.4) was measured and the presence of visible stamp sands, slime clays, or macrophytes in grab samples was recorded.

2.3 Benthic Macroinvertebrates

Following baseline methodology, benthic macroinvertebrate samples from the offshore sites were processed using a hinged elutriation table in the field, while the nearshore and northern samples were elutriated on shore. Elutriation technique followed EPA on-station ponar sampler procedures (US EPA, 2002). Ponar grabs were gently rinsed and elutriated into a 500 μm sieve until the decanted water was clear. Elutriated and coarse material were stored separately; both were preserved in 70 to 90 percent ethanol. All retained, preserved material was stored at 4°C prior to sorting. Retained material was picked in entirety due to low organism density, following EPA benthic macroinvertebrate laboratory methods (US EPA, 2012), using an Intertek magnifier lamp (1.75x magnification). Picked organisms were sorted to the lowest taxonomic identification possible (preferably genus) using taxonomic keys, Merritt & Cummins (1996) and Thorp & Covich (2010), through an Olympus dissecting microscope. Sorted organisms were stored in pure ethanol in labeled vials at room temperature.

Macroinvertebrates were reported as individuals per taxa under the major phyla of Annelida, Arthropoda, Nematoda, and Mollusca, following the taxonomical conventions established in the baseline reports. Vertical migratory taxa, such as Chaoborus, and pelagic taxa, such as certain Cladocerans, were reported, following baseline conventions. Individual totals were converted into organism (macroinvertebrate) density, reported as individuals m^{-2} , to account for the difference in ponar cross-sectional area between offshore (0.052 m^2) and nearshore/northern sites (0.023 m^2).

2.3.1 Diversity Metrics

The Shannon Index (H') is a statistical measure of diversity, with greater H' values corresponding to greater diversity (Burton, 1999). It assumes that all taxa are represented in a randomly collected sample, thus the distribution of taxa in a given sample should resemble the population. H' was modified to assess diversity at the family level.

$$H' = \sum_{i=1}^s \frac{n_i}{N} \ln \left(\frac{n_i}{N} \right) \quad (1)$$

where

n_i ... individuals of one particular taxon

N ... total number of individuals recovered (all taxa)

s ... total number of taxa

Functional feeding group identifiers and tolerance values (0 to 10 as most sensitive to least sensitive taxon) were assigned by genus using tabulated references from Barbour et al. (1999) and Bode et al. (1996). Tolerance values listed for the Midwest are derived from water quality surveys by the Wisconsin Department of Natural Resources (Hilsenhoff, 1987), although these have been generalized for most ecological disturbances by later studies (Barbour et al., 1999; Mandaville, 2002). The presence of more sensitive (less tolerant) species is interpreted as an indicator of higher water quality in regard to nutrient pollution (Hilsenhoff, 1982)

The Hilsenhoff Biotic Index (HBI) computes tolerance within a community, weighted by relative abundance of each taxon. Typically, HBI is used to assess a community's response to nutrient pollution but can also be used to measure community response to increasing perturbations in water quality (Table 2.1). HBI was modified for family-level evaluations (Hilsenhoff, 1988) using family-level tolerance values (Table A.1.1).

$$HBI_f = \frac{\sum n_i a_i}{N} \quad (2)$$

where

n_i ... individuals of one particular taxon (family)

a_i ... tolerance value of taxon (family)

N ... total number of individuals recovered (all taxa)

**Table 2.1 Hilsenhoff family biotic index and associated water quality rating
(Modified from Table 2, Hilsenhoff, 1988)**

<i>Family Biotic Index</i>	<i>Water Quality</i>
0.00-3.75	Excellent
3.76-4.25	Very good
4.26-5.00	Good
5.01-5.75	Fair
5.76-6.50	Fairly poor
6.51-7.25	Poor
7.26-10.00	Very poor

2.4 Sediment Properties

2.4.1 Heavy Metals

Sediment subsamples were analyzed for solid-phase concentrations (mg/kg) of Arsenic, Cadmium, Chromium, Copper, Lead, Mercury, Nickel, and Zinc. Heavy metal concentrations were measured by White Water Associates, Inc. (Amasa, MI); procedures are outlined in the Torch Lake AOC Benthos Monitoring QAPP (McDonald, 2018). Metals were analyzed by Inductively Coupled Plasma-Optical Emission Spectroscopy, with the exception of mercury which was analyzed by atomic absorption spectroscopy. Metal concentrations below the method detection limits (MDL) were reported as non-detects (Table 2.2).

Table 2.2 Method detection limits (MDL) and method quantification limits (MQL)

	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Zinc
MDL (mg/kg)	4.2	0.25	0.10	0.13	2.1	0.0079	0.40	0.69
MQL (mg/kg)	10	1	0.30	1	10	0.019	3.0	3.0

2.4.2 Organic Content

The proportion of organic matter in the sediment samples was assessed by Loss on Ignition analysis (LOI) to determine percent volatile solids, or the proportion of combustible benthic material in the solid fraction. During sample preparation for volatile solids (VS) analysis, total solids (TS) and water content (WC) were determined (reported

as percent by mass). Procedures for total solids and water content analyses were taken from US EPA Standard Method 160.4 Residue, Volatile (gravimetric, ignition at 550°C) by Muffle Furnace as well as ASTM standard designation D2216-10, respectively.

Sediments were homogenized and subsampled in triplicate, and a single scoop of mass ranging from 25-50g was collected for TS and LOI analysis. Samples were dried at 105°C for 24 hours, weighed, and dried again at 105°C at one-hour intervals until dry mass remained consistent within four percent. Dried ground samples were ignited at 550°C in a pre-heated muffle furnace for one hour, weighed, and ignited again at 550°C in 30-minute intervals until ignited mass remained consistent within four percent. Wet mass, final dry mass, and final ignited mass were recorded to a tenth of a gram at room temperature.

2.4.3 Particle Size Distribution

Sediment subsamples were analyzed in ‘as-received’ moisture conditions and homogenized to include any pore water which may have separated in storage. Gradation assessment of benthic sediment material was conducted to generate particle size distributions from combined sieve and hydrometer analysis. Despite the heterogeneity in sediment composition, sieve and hydrometer analysis procedures were performed for all sites following recommended procedures for the maximum particle size across all sites.

Sieve analysis determines the mass contribution of various particle diameter ranges to the composite sediment sample. Particle size ranges are defined using square-grid sieves of defined mesh size corresponding to retained particle diameter limit. Hydrometer analysis (sedimentation) is specified for particle diameter less than 75 µm.

Procedures followed were ASTM standard designation D6913/D6913M-17 (sieve analysis), Method B for a maximum particle size equal to or less than 4.75 mm (No. 4 sieve), and D7928-17 (hydrometer analysis) for a 152H hydrometer. ASTM standards do not specify a particle size distinction between clay and silt, therefore USDA particle size

classification limits (Burt, 2011) were used to determine particle size proportions from particle size distribution curves.

Moist subsample preparation, without air or oven drying, was chosen as a consistent preparation method due to high variability of sediment composition between sites. A 250-mL volume ‘scoop’ was homogenized and subsampled according to the minimum dry mass requirements from Table A.1.2, ASTM D6913/D6913M-17 (Method B), and

Table A.1.3, ASTM D7928-17, respectively. Dry mass (M_d) was converted to wet mass (M_m) using water content (w_c) calculated for each sample:

$$M_d = \frac{M_m}{1 + \left(\frac{w_c}{100}\right)} \quad (3)$$

where

M_d ... Dry mass (g)

M_m ... Wet (moist) mass (g)

w_c ... Water content (%)

Sieve Analysis

Composite sieving, which uses multiple sieves in order of decreasing particle size, was chosen as a consistent sieving method as the maximum particle size at some sites fell within the diameter range of 4.75 mm to 1.9 mm. Sediment subsamples were washed through the composite sieves to effectively separate various particle sizes. Sieves included No. 5, 12, 40, 60, and 120 mesh sizes corresponding to retained particle diameters of 1, 2, 0.5, 0.25, and 0.125 mm, respectively (Table 2.3).

Table 2.3 Square-grid sieves by number, mesh size, and particle size limit

Sieve Number	No. 5	No. 12	No. 40	No. 60	No. 120
Mesh Size	5 mesh	10 Mesh	35 Mesh	60 Mesh	115 Mesh
Particle Size Limit	4.00 mm	2.00 mm	0.500 mm	0.250 mm	0.125 mm

Material retained in each sieve was transferred to an individual container of known mass and dried for 24 hours, weighed, and dried again at 105°C in one-hour intervals until dry mass remained consistent within four percent. Visible organic debris was removed from dried, retained material prior to obtaining final dried mass. Dried retained mass was weighed at room temperature and recorded to a hundredth of a gram.

Sediment material was also qualitatively assessed prior to drying retained samples or disposing of the fines-water slurries. These observations are subjective, but consistently applied at all sites to identify presence of mining byproducts (stamp sands, slime clays).

Dry specimen mass was calculated from the wet specimen mass of the sieve subsample and its associated water content. The ratio of the dried retained mass (M_r) to dry specimen mass (M_d) represents the percent retained material (P_r):

$$P_r = 100 \left(\frac{M_r}{M_d} \right) \quad (4)$$

where

P_r ... Percent retained material (%)

M_r ... Retained dry mass (g)

M_d ... Dry mass (g)

Cumulative percent retained outputs, in order of decreasing sieve size, were used to compute percent finer values for the coarse particle component of the particle size distribution curves.

Hydrometer Analysis

All samples were tested via hydrometer analysis without prior quantification of percent fines due to the limited sample mass available for testing. Consequently, hydrometer results from analyzed sites with less than 5% fine-grain material were disregarded upon conclusion of hydrometer tests, adhering to minimal requirements of fines (ASTM D7928-17).

The hydrometer specimen was placed in a dispersion cup with 2.5 g sodium hexametaphosphate (dispersant) and mixed vigorously to generate a soil-water slurry. The slurry was diluted to one liter in a sedimentation cylinder and particles agitated into suspensions for initiation of the hydrometer test. Hydrometer (152H) and concurrent temperature measurements were recorded for 1, 2, 4, 15, 30, 60, 240, and 1440 minutes of elapsed time. Hydrometer measurements were recorded to the nearest half-division mark and temperature was recorded to tenth of a degree Celsius. Temperature and

hydrometer measurements were recorded for the control sedimentation cylinder at the sample start time and if the temperature of the sample slurry changed more than 0.5°C between measurements, for density-temperature calibration measurements concurrent with sample analysis.

At the conclusion of the 1440-minute measurement, the sample slurry was washed through a No. 200 sieve and rinsed with excess distilled water to remove the dispersant. Retained material was transferred to an individual container of known mass and dried for 24 hours, weighed, and dried again at 105°C in one-hour intervals until dry mass remained consistent within four percent. Dried, retained mass was recorded at room temperature and to a hundredth of a gram.

For generation of the fines component of the particle size distribution curves, specific gravity of the sediment subsamples was estimated from literature values as 2.70 for both sand-silt-clay mixtures and dark-colored sand specimens³ (Coduto et al., 2010). Similar to sieve analysis, dry specimen mass was calculated from the wet specimen mass of the hydrometer subsample and its associated water content.

The hydrometer readings were corrected for temperature-density variations using the calculated hydrometer offset. Corrected hydrometer readings were used to compute mass percent finer (N_m) for particles in suspension for each hydrometer reading:

$$N_m = 0.6226 \left(\frac{G_s}{G_s - 1} \right) \left(\frac{V_{sp}}{M_d} \right) (r_m - r_{d,m}) \left(\frac{100}{1000} \right) \quad (5)$$

where

N_m ... Mass percent finer (%)

G_s ... Specific gravity of soil (dimensionless)

V_{sp} ... Volume of Suspension (cm³)

M_d ... Dry specimen mass (g)

r_m ... Hydrometer reading (cm)

$r_{d,m}$... Hydrometer offset (g/L)

³ Specific gravity estimation did not account for stamp sand inclusion in sediment samples.

Next, effective depth (H_m) was calculated using the mass percent finer for each hydrometer reading:

$$H_m = H_{r2} + \left(\frac{H_{r1} - H_{r2}}{r_2 - r_1} \right) (r_2 - r_m + C_m) - \left(\frac{V_{h,b}}{2A_c} \right) \quad (6)$$

where

H_m ... Effective depth (cm)

H_{r1} ... Hydrometer maximum, 18cm

H_{r2} ... Hydrometer minimum, 7 cm

r_1 ... Maximum hydrometer reading (cm)

r_2 ... Minimum hydrometer reading (cm)

r_m ... Hydrometer reading (cm)

C_m ... Meniscus correction, 0.25 g/L

$V_{h,b}$... Hydrometer bulb volume, 58 cm³

A_c ... Cross-sectional area of the sedimentation cylinder, 7.5 cm²

Finally, the maximum particle diameter (D_m) were calculated for a given elapsed time were calculated using the determined effective depths:

$$D_m = 10 \sqrt{\left(\frac{18\mu}{\rho_w g (G_s - 1)} \right) \left(\frac{H_m}{t_m} \right)} \quad (7)$$

where,

D_m ... Particle diameter (mm)

μ ... Viscosity of water, 0.01 g/cm s at 20°C

ρ_w ... Mass density of water, 0.99821 g/cm³

g ... Acceleration due to gravity, 980.7 cm/s²

G_s ... Specific gravity of soil (dimensionless)

H_m ... Effective depth (cm)

t_m ... Elapsed time (s)

Hydrometer Calibration

Calibration of the hydrometer required a temperature-density correction computed as the hydrometer offset value associated with a given slurry temperature; equations are associated with the 152H hydrometer. The hydrometer readings for each elapsed-time measurement were corrected using a temperature-dependent hydrometer offset ($r_{d,m}$):

$$r_{d,m} = B_{avg} - 0.01248 T_c - 0.00795 T_c^2 \quad (8)$$

where,

$r_{d,m}$... Hydrometer offset (g/L)

B_{avg} ... Average mass shift (g/L)

T_c ... Control temperature (C)

The hydrometer offset was computed using an average of the mass reading shift (B):

$$B = m + 0.01248 T_c + 0.00795 T_c^2 \quad (9)$$

where,

B ... Mass reading shift (g/L)

m ... Dispersant mass (g)

T_c ... Control temperature (C)

The particle size distribution curves for both fine and coarse-grain components were constructed from the mass percent finer values associated with particle diameters computed (hydrometer) or acquired from sieve size (Figure A.5.1 through Figure A.5.40). USDA particle size classification limits (Burt, 2011) were used to determine particle size classifications from the generated particle size distribution curves.

3 Results

3.1 Benthic Macroinvertebrate Community

3.1.1 Macroinvertebrate Density

Macroinvertebrate density⁴ per site varied greatly across Torch Lake (Figure 3.1), ranging from 77 to 9,783 individuals m⁻², with a median of 1,478 individuals m⁻². A spatial gradient within macroinvertebrate density was observed (Figure 3.2), with general trends of increasing density from west to east as well as north to south across the lake.

Nearshore sites were regionally categorized by shoreline (Table 3.1), based on similarities in macroinvertebrate densities, proximity to historical mining activity, and sediment composition, described later in this section. Similarly, the northern sites were separated by basin morphometry and sediment composition (Figure 3.1) into northeast and northwest basin and the channel sites (Table 3.1).

A maximum macroinvertebrate density of 9,783 individuals m⁻² was observed at site 0004 on the eastern shoreline (Figure 3.1). Some of the greatest density observations stretch from the channel portion of the northern basin (sites 1806 and 1807) down the eastern shoreline to sites 0005, 0006, 0008, and 0009. The northern basins also contain large densities of organisms, with the northwest basin sites (1808 to 1810) slightly greater than the northeast basin sites (1801 to 1805). Some of the lowest macroinvertebrate densities were observed at offshore sites, with a minimum density of 77 individuals m⁻². In addition, very low densities peppered the western shoreline, occurring at sites 0007, 0017, and 0018. Lastly, the remaining sites 0010 to 0014 along the southern shoreline contained moderate macroinvertebrate densities, in the mid-range of the entire lake's distribution.

⁴ The benthic community results with assigned tolerances and functional groups, as well as corresponding water depths, are provided in Table A.2.7 through Table A.2.10 in the appendix.

Table 3.1 Macroinvertebrate densities (individuals m⁻²)

	Offshore sites	Nearshore sites			Northern sites		
		Western Shoreline	Eastern Shoreline	Southern Shoreline	Channel	Northeast Basin	Northwest Basin
Median	250	609	3,130	1,696			
Mean	262±122	750±590	4,400±3,090	1,939±751	4,543±1,875	2,139±791	3,420±176
Site Codes	99**	0002, 0003, 0007, 0015-0019	0004-0009	0010-0014	1806-1807	1801-1805	1808-1810

Figure 3.1 Macroinvertebrate density

Figure 3.2 Interpolated map of macroinvertebrate density

3.1.2 Diversity Metrics

Eighty-one and 24% of the recovered individuals were not specifically identified (NSI) to the genus and family level, respectively. Consequentially, diversity metrics were computed for the family taxa in lieu of genus or species. In addition, identification of soft-bodied taxa, annelids and nematodes, was limited to higher taxonomical levels, similar to the sampling resolution in the baseline survey.

Taxa Richness

Twenty-one distinct family taxa were identified across all sites. The dominant family was Chironomidae (29%), followed closely by Valvatidae and Chaoboridae (24% and 18%, respectively) across all sites (Figure 3.3). The family taxa richness for offshore, nearshore, and northern sites were 9, 17, and 14 distinct taxa, respectively. Chironomidae was dominant in offshore, nearshore, and northern sites, whereas Valvatidae was not observed in offshore sites.

Figure 3.3 Family taxa richness of identified individuals

Relative Abundance

The ratio of phylum taxa for the entire lake was 24% annelids, 1% nematodes, 42% arthropods, and 33% mollusks (Figure 3.4). Individuals recovered from offshore sites were mostly arthropods (92%). The western shoreline, eastern shoreline, and channel sites had similar ratios of annelids, arthropods, and mollusks. The southern shoreline was predominately mollusks (90%). The northeast basin had more annelids, while the northwest basin was mostly arthropods.

Figure 3.4 Macroinvertebrate density by phylum

For the entire lake, the computed Shannon Index values⁵ ranged from 0.23 to 1.55, with a median of 0.81. Median H' values for offshore, nearshore, and northern sites were

⁵ All calculated H' values per site are provided in Table A.4.1 in the appendix.

0.76, 0.83, and 0.81, respectively. The lowest diversity was observed in the western shoreline and northeast basin sites, with average H' values of 0.6 and 0.64, respectively. The greatest diversity occurred in the channel sites and along the eastern and southern shorelines, with average H' values of 1.32, 0.98, and 0.91, respectively. Shannon Index calculations did not account for family-level diversity within the Annelida and Nematoda phyla, which may alter diversity interpretations at sites exhibiting greater counts of soft-bodied organisms (e.g., northwest basin).

Functional Feeding Groups

The dominant functional feeding group for all sites were gatherer-collectors (35%), followed by scrapers, predators, and filterer-collectors (28%, 15%, and 13%, respectively) (Figure 3.5). The dominant functional feeding group for offshore sites were gatherer-collectors (70%), while nearshore, and northern sites had more variability in feeding group composition. Southern sites were predominately scrapers (87%), while northeast and northwest basin had more predators (27 and 54%, respectively). Approximately 7% of individuals were not assigned a functional feeding group (Table A.2.7 through Table A.2.10).

Figure 3.5 Functional feeding group proportions

Tolerance

Genus-level tolerance values ranged from 4 to 9 for all sites, indicating few sensitive taxa. Approximately 40% of recovered individuals have tolerance values of 8 and 9, whereas only 2% had a tolerance value of 4 (Figure 3.6). The most common tolerance value among recovered individuals was 6 for offshore sites (46%) and 8 for nearshore and northern (41% and 43%, respectively). More sensitive individuals were observed along the eastern shoreline and in northeast basin and channel sites. Approximately 10% of individuals were not assigned a tolerance value (Table A.2.7 through Table A.2.10).

Figure 3.6 Genus-level tolerance value distribution for recovered individuals

For the entire lake, the Hilsenhoff family biotic indices⁶ ranged from 0.4 to 8.0, with a median of 5.49. The mean HBI_f value for Torch Lake was 4.9 ± 2.3 with a good water quality rating. Median HBI_f values for offshore, nearshore, and northern sites were 6.02 (22%NA), 5.81 (23%NA), and 1.98 (58%NA), respectively. HBI_f water quality ratings indicate offshore and nearshore had fairly poor water quality. Northern sites had excellent water quality, particularly in the northwest and northeast basin sites. In contrast, the

⁶ Calculated HBI_f values per site are summarized in Table A.4.2.

southern shoreline had very poor water quality on average. Both the western and eastern shoreline had an average water quality rating of good.

Given that low density sites (0007, 0018) and high-density sites (0004, 1806-07) had similar water quality ratings, the population weight of the HBI_f calculation may be skewing tolerance ratings for small populations. In addition, approximately 37% of individuals were not assigned a family-level tolerance value, which may affect the validity of computed Hilsenhoff family biotic index values.

Figure 3.7 Hilsenhoff family biotic indices and associated water quality rating

3.2 Sediment Properties

3.2.1 Heavy Metals

Solid-phase metals concentrations (mg/kg)⁷ are summarized and compared with consensus-based Threshold Effect Concentrations (TEC) and Probable Effect Concentrations (PEC) (MacDonald et al., 2000) in Table 3.2 and mapped in Figure 3.8 through Figure 3.15.

Metal concentrations at most sites were below TEC levels, except for cadmium, copper, and nickel (Table 3.2). Arsenic, lead, and mercury concentrations were below detection limit (ND) at some sites. In contrast, most sites exceeded PEC levels for cadmium and copper. Generally, greater metal concentrations occurred along the western shoreline and in the offshore and northwest basin sites, whereas the eastern shoreline possessed the lowest metal concentrations. Arsenic and mercury concentrations were low across the lake, with no spatial pattern.

Table 3.2 Number of sites exceeding heavy metal TEC and PEC

	ND		Less than TEC		Between TEC & PEC		Greater than PEC	
Arsenic	36	90%	3	8%	1	3%	0	0%
Cadmium	0	0%	4	10%	9	23%	27	68%
Chromium	0	0%	25	63%	14	35%	1	3%
Copper	0	0%	2	5%	3	8%	35	88%
Lead	2	5%	23	58%	12	30%	3	8%
Mercury	10	25%	29	73%	1	3%	0	0%
Nickel	0	0%	11	28%	21	53%	8	20%
Zinc	0	0%	24	60%	15	38%	1	3%

⁷ All solid-phase trace metals concentrations (mg/kg) are provided in Table A.2.11 and Table A.2.12 with TEC and PEC in the appendix.

Figure 3.8 Arsenic concentrations in sediment samples

Figure 3.9 Cadmium concentrations in sediment samples

Figure 3.10 Chromium concentrations in sediment samples

Figure 3.11 Copper concentrations in sediment samples

Figure 3.12 Mercury concentrations in sediment samples

Figure 3.13 Lead concentrations in sediment samples

Figure 3.14 Nickel concentrations in sediment samples

Figure 3.15 Zinc concentrations in sediment samples

3.2.2 Organic Content

Volatile solid (VS, proxy for organic content)⁸ across Torch Lake ranged from 0.1% to 15.1%, with a median of 5.8%. The median VS content for offshore and northern sites was 10.4% and 1.7% for nearshore sites. Organic content was lowest along the eastern shoreline, at the channel sites, and at sites positioned near the stamp sand piles (Figure 3.16).

Figure 3.16 Percent volatile solids in sediment samples

⁸ The proportions of volatile solids, total solids, and water content in the sediment samples are summarized in Table A.2.13 and Table A.2.14 in the appendix.

3.2.3 Particle Size Distributions

Torch Lake sediments are highly variable in composition⁹, comprised of silts and sands, either natural or stamp sands (Figure 3.17). Offshore sediments were predominantly composed of fines (44 to 70%) with a visible distinction between silty post-mining sediment and slime clays in each Ponar grab (Figure 1.4). In contrast, nearshore and northern sites were highly varied in composition, ranging from silts to fine gravel with varying contributions of particle sizes (Figure 3.17). The western shoreline appears to contribute most to this variability, with sediments majorities ranging from fine gravel to silty fines. The eastern shoreline, channel, and northwest basin sites possess similar sediment particles sizes in the range of fine silt to medium sand. Variation in sediment composition is visually apparent (Figure 3.18) with samples ranging from dark brown silt (site 0001) to light brown natural sand (eg. site 0005) to coarser stamp sand (eg. site 0007) to purplish-grey of slime clay impacted sediment (eg. site 0014).

⁹ All particle size classifications with majority particle size are summarized in Table A.2.15 through Table A.2.16 in the appendix.

Figure 3.17 Particle size distribution in sediment samples

Figure 3.18 Nearshore sediments with site codes

3.2.4 Sediment Observations

Post-mining sediment thickness (PMST)¹⁰ ranged from greater than the depth of the petite Ponar (approximately 11cm) to not accruing in a measurable vertical quantity. In such cases, post-mining sediment was minimal to not present, especially in sand-dominant sediments. Post-mining sediment was identified as brown silt but did not consider natural sand deposition from the watershed. PMST is greatest in the northern basins (Figure 3.19) and minimal from the channel sites down the eastern shoreline.

Figure 3.19 Post-mining sediment thickness

¹⁰ PMST, sediment color, and presence of mining byproducts (stamps sands, slime clays), and presence of macrophytes are summarized in Table A.2.17 and Table A.2.18 in the appendix.

Macrophytes were present in some of the nearshore sites, mostly in the southern end of the lake (Figure 3.20). Stamp sands were observed at sites near the stamp sand piles and along the eastern shoreline (Figure 3.21). Slime clays were observed primary in offshore sites and in the southern end of the lake but were not observed along the eastern shoreline and in the northern sites (Figure 3.22).

Figure 3.20 Macrophyte presence

Figure 3.21 Stamp sand presence

Figure 3.22 Slime clays presence

4 Discussion

4.1 Comparison to Baseline

To ensure effective comparison to the baseline benthic conditions, sampling and processing of the benthic community samples followed baseline methodologies. Taxonomical reporting conventions established in the baseline survey were maintained but sorting efficiency may differ between surveys. Comparisons were only made between the 30 original sites (offshore and nearshore sites, excluding northern sites). Given differences in site coordinates and water depths between surveys, individual site comparisons between 2018 and the baseline were not considered.

4.1.1 Macroinvertebrate Density

Across the lake, 26 of the 30 original sites had greater macroinvertebrate densities, suggesting an overall increasing trend (Figure 4.1)¹¹. The range remained similar to the 1999/2000 data, but the median density increased from 152 to 622 individuals m⁻². However, the mean densities for each sample population are not significantly different (paired t-test, $p = 0.23$). When considering log-transformed distributions, the mean for this study was significantly greater than the baseline mean (paired t-test, $p < 0.05$), implying large density outliers are positively skewing both distributions. If outliers (greater than $1.5 \times Q3$) are removed from each distribution, the mean for the modified updated survey (1,017 individuals m⁻²) is significantly greater (paired t-test, $p < 0.05$) than the modified baseline (262 individuals m⁻²), supporting an overall increase in macroinvertebrate densities with large variance within each sample distribution.

¹¹ All benthic macroinvertebrate results from the baseline survey (USEPA, 2001) are summarized in Table A.7.1 through Table A.7.5 in the appendix.

Offshore densities were significantly greater than the baseline (paired t-test, $p < 0.05$). On the other hand, changes in nearshore densities were more varied. It was unexpected that the greatest increases in density occurred in the offshore (Figure 4.1), but increases in individual counts at low densities may appear more dramatic than similar increases at greater densities.

Figure 4.1 Pairwise comparison of macroinvertebrate density per site surveys

Offshore densities remained lower compared with densities in the rest of the lake (Figure 4.2). Nearshore densities were statistically distinct between the eastern, southern, and western shorelines in 2018 (ANOVA single-factor and t-test assuming unequal variances, $p < 0.05$) but were not in the baseline (ANOVA single-factor, $p = 0.65$). In addition, eastern shoreline densities were significantly greater than those reported in the baseline (paired t-test, $p < 0.05$) as well as rest of the lake in 2018 (t-test assuming unequal variance, $p < 0.05$). This implies spatial dependence for macroinvertebrate density and recovery.

Figure 4.2 Macroinvertebrate densities for the baseline (left) and 2018 (right) surveys

4.1.2 Macroinvertebrate Diversity

Taxa Richness

The number of family taxa observed in Torch Lake was greater in 2018 compared to the baseline, suggesting increased diversity within 30 original sites. Taxa richness for offshore and nearshore sites in the updated survey was also greater. Chironomidae (30%) and Hydrobiidae (20%) were the dominate taxa, similar to the baseline, implying the overall community structure remained the same.

Relative Abundance

The relative abundance per phylum has increased since the baseline (Figure 4.3), particularly with the number of site observations of mollusks (scrapers), suggesting the possibility of increased algae/biofilm productivity in some area of the lake (Covich et al., 1999). However, there were no significant differences (paired t-test, $p > 0.05$) in any macroinvertebrate density distributions within each phylum between baseline and 2018, similar to the lake-wide comparison of density, implying spatial dependency for increases in macroinvertebrate diversity.

Figure 4.3 Relative abundance of phyla observed per site

The range of site-specific Shannon Indices (0.23 – 1.26) was similar to the baseline (0.35 – 1.23), with a slight increase in the median H' value, from 0.69 to 0.81 (Figure 4.4). However, means for each H' distribution were not significantly different (paired t-test, $p = 0.48$), implying similar diversity within the benthos. Despite the greater taxa richness, the overall community structure and diversity remained the same.

Figure 4.4 Shannon Indices (H') from 2018 and baseline

Functional Feeding Groups

The dominant functional feeding group from the baseline findings was also gatherer-collectors (43%), with 11% of recovered individuals for not assigned a functional feeding group. The distribution of functional feeding groups was different between 2018 and baseline conditions (Figure 4.5); there are fewer gatherer-collectors and more scrapers. Fewer gatherers-collectors implies less detrital processing within the benthos (Cummins, 1973). Shredders and scrapers feed on vascular plant tissue (macrophytes) and attached algae or biofilms, respectively (Covich et al., 1999). Greater populations of scrapers may imply greater primary productivity in the lake as well as potential changes in habitat for supporting plant growth (Hodkinson & Jackson, 2005). In addition, the number of individuals in the omnivore and predator classes are slightly increased in 2018 conditions. Increased numbers of omnivores and predators imply a more complex benthic

food web, able to support higher trophic levels. Finally, decreased numbers of filterer-collectors in 2018 may imply a decrease in suspended particles within the water column (Goodyear & McNeill, 1999).

Figure 4.5 Functional group distributions from 2018 and baseline

Tolerance

The general distributions of genus-level tolerance values were similar between 2018 and baseline conditions, with a maximum of 10 versus 9 and a minimum of 4 (Figure 4.6). A slightly greater portion of individuals fell within tolerance of 7 – 8 in 2018. Tolerance values represent organism sensitivity to ecological disturbances (Bode et al., 1996; Barbour et al., 1999), therefore similar community sensitivity between 2018 and the baseline may imply similar habitat quality. In addition, more sensitive taxa, such as Mysis, Gammarus, and Ephemeroptera (mayflies) were observed in 2018, which has not been previously reported.

Figure 4.6 Tolerance value distributions from 2018 and baseline

The range of HBI_f values remained similar between baseline and 2018 conditions (0.8 – 8.0 and 0.6 – 8.0, respectively). The baseline median was 6.0 (mean of 5.4 ± 2.1), corresponding to a water quality rating of fair to fairly poor. The HBI_f values calculated for each survey distribution were not significantly different (paired t-test, $p = 0.71$) (Figure 4.7). Water quality rating also did not change for offshore and nearshore sites (fairly poor). The lack of a statistically significant difference in HBI_f values indicates water quality remained similar from baseline to 2018.

Figure 4.7 Comparison of calculated Hilsenhoff family biotic indices (HBI_f)

4.1.3 Is there evidence of recovery?

A slight but statistically significant increase in macroinvertebrate density was observed since the baseline survey in 1999/2000, implying modest evidence of recovery. Interpreting the temporal trend implied by these surveys is complicated by the high variance within density distributions. Abundance of each phylum per site and taxa richness has increased, but the overall dominance in the community structure was unchanged. Generally, the tolerance level of the benthos has not changed, implying continued suppression by ecological disturbances or maintained water quality. However,

observations of more sensitive taxa were site specific, indicating benthic recovery varies spatially.

The eastern shoreline exhibited greater macroinvertebrate densities than the rest of the lake as well as a statistically significant difference in baseline and 2018 conditions, therefore conditions along the eastern shoreline may be facilitating recovery. In contrast, site 0007, along the western shoreline, has very low macroinvertebrate densities and high metal concentrations due to its close proximity to the Hubbell smelter and slag pile (Mandelia, 2016; Warburton, 1987), and exhibited minimal changes in macroinvertebrate density. Consequently, this may be a region where natural processes are not facilitating recovery.

Variable benthic community conditions and rates of recovery may be related to habitat heterogeneity in the lake. Functional group composition indicates an increase in trophic levels in the lake as well as a shift in benthic functions from detrital processing to primary and secondary consumption. This implies a more complex food web as well as changes in benthic habitat and food availability (Goodyear & McNeil, 1999; Cummins, 1973).

Assessing only the baseline and 2018 survey does not clearly describe temporal trends in the benthic community. Broadening the scope to consider data from earlier studies of Torch Lake, however, provides additional evidence that macroinvertebrate densities have increased. In 1970, Massey (1970) observed a maximum of 172 individual m^{-2} near the mouth of the Trap Rock River. Also, ranges of 22 – 1,774 individual m^{-2} in 1970, 0 – 1,539 individual m^{-2} in 1972, and 0 – 301 individual m^{-2} in 1979 were reported for Torch Lake (Evans, 1973; MDNR, 1987). The 2004 monitoring survey reported macroinvertebrate densities ranging from 0 – 250 individual m^{-2} at select baseline sites in Torch Lake (MACTEC, 2008). While these studies may not be directly comparable in terms of sampling methodologies and locations, a general trend of recovering benthic community densities can be inferred. More importantly, this also shows the need for an appropriate reference site to gauge degree of benthic degradation (Grapentine, 2009;

Duffy et al., 2017), which is not incorporated in many of the benthos and toxicity studies for Torch Lake (West et al., 1973; MDNR, 1987; Cusack & Mihelcic, 1999; US EPA, 2001; MACTEC, 2008).

There are limited studies on benthic macroinvertebrate communities in undisturbed, oligotrophic lakes. Macroinvertebrate surveys are predominately focused on river and stream systems (Beck & Hatch, 2009). Most of the lakes and reservoirs surveyed are ecologically disturbed by nutrient pollution, such as Spirit Lake or Lake Winnebago, WI (Breneman, 2000; Heling et al., 2018). These lakes are typically located in highly developed areas and possess very shallow bathymetry, contributing to problems with eutrophication. Large, deep lakes, such as the Laurentian Great Lakes or Lake Tahoe, CO, are often oligotrophic, but are also very deep which alters the distribution of dissolved oxygen, temperature, and suspended solids (Nalepa et al., 1998; Lozano et al., 2003; Nalepa et al., 2007; Frantz & Cordone, 1996; Ayuushsuren & Shcherbina, 2015). These lake characteristics determine the condition and structure of the benthic community, and therefore restrict inter-lake comparisons of macroinvertebrate communities (Beck & Hatch, 2009).

Considering abundances generally reported in benthic surveys, total organism densities range from 1,000 to 10,000 individuals m^{-2} . Most surveys do not report less than 1,000 individuals m^{-2} . In Torch Lake, macroinvertebrate densities observed at the northern sites and along the eastern and southern shorelines do fall within this range (Figure 3.1). In contrast, the offshore sites and sites along the western shoreline are much lower in density. It can be inferred that macroinvertebrate densities on the eastern side of Torch Lake are potentially approaching a recovered state, whereas other areas in the lake are still heavily impaired.

Changes in ecological systems, either recovery or degradation, can be difficult to detect with a degree of confidence if they are relatively small in comparison to natural variability, seasonally or annually (Johnson, 1998). Seasonality is an important consideration for monitoring accuracy (Resh & Jackson, 1993; Johnson, 1998). The 2018

sampling (August 6 - 7 & September 3, 2018) occurred later in the summer than the baseline sampling (July 29, 1999 & July 17 – 18, 2000), coinciding with the late summer emergence period of most insects (Jansson, 2004). Consequentially, arthropod distributions may be different if sampled earlier in the summer.

Another variable in long-term benthic monitoring is the efficiency of sieving and sampling methods (Jonasson, 2004; Johnson, 1998). Large variation can often occur between replicate samples and extensive efforts are required to obtain enough data to detect statistically significant changes in fauna (Baumgärtner et al., 2008).

Sampling protocols were established in the baseline survey and maintained in the updated survey for comparison purposes, such as the inclusion of vertically migratory species such as *Chaoborus*, which may conflict with other established benthic sampling regimes. In addition, family-level metrics computed for both surveys did not include soft-bodied taxa such as annelids and nematodes.

4.2 Northern Sites

The mean macroinvertebrate density for the northern sites was significantly different than nearshore sites in the main portion of the lake (t-test assuming unequal variance, $p = 0.12$). The northern and the eastern shoreline sites had the greatest observed macroinvertebrate densities across the lake (Figure 3.1). Among the northern sites, channel macroinvertebrate densities were greatest. The northern sites are quite heterogeneous in basin morphology and sediment composition (Figure 3.17), and consequentially categorized as the northeast basin, northwest basin, and channel sites (Table 3.1).

Sediments in the channel sites was composed of mostly stamp sands with low organic content and PMST, similar to the course-grained sediments along the eastern shoreline. On the other hand, the sediments in northeast and northwest basins had greater silt fractions, organic content, and PMST.

Channel sites are potentially influenced by currents that provide a continuous supply of oxygen and particulate food sources, similar to a river or stream. The northeast basin is connected directly to the mouth of the Trap Rock River and contains predominately silty sediments with some stamp sands (e.g. site 1804) due to the proximity to the stamp sand pile. In contrast, the northwest basin is hydrologically isolated and has a single connection to the northeast basin. Sediments in the northwest basin (site 1808) were black with a notable sulfide smell, indicating anoxic conditions potentially due to limited hydrological exchange in the basin.

There is a strong link between the Trap Rock River and sediment deposition in the northeast basin sites, whereas the northwest basin may rely on in-lake productivity to drive new sediment accumulation. In the main portion of the basin, offshore and nearshore sites may depend on in-lake currents and sediment focusing to provide new sediment rather than direct tributary input.

Metal concentrations were greater in the northwest basin and lower in the channel sites, which may be linked to distribution of dissolved oxygen and sediment composition at these sites. However, macroinvertebrate densities in the northwest basin are similar to channel sites and the eastern shoreline. This implies macroinvertebrate densities may be influenced by currents and depositional zones resulting from proximity to the Trap Rock River.

4.3 Spatial Drivers of Density

Considering all sites, macroinvertebrate density was negatively correlated with water depth (Spearman's $\rho = -0.66$, $p < 0.05$)¹², as anticipated prior to sampling. Water depth was significantly different between the offshore and nearshore/northern sites (t-test assuming unequal variance, $p < 0.05$), therefore these site categories were evaluated separately. In contrast, basin slope was not statistically different between offshore, nearshore, and northern sites (t-test assuming unequal variance, $p < 0.05$), therefore it was not considered as a distinguishing factor between sites.

Water depth was consistent among offshore sites and not correlated to offshore macroinvertebrate densities. Without considering offshore sites, macroinvertebrate density was still negatively correlated with water depth (Spearman's $\rho = -0.46$), indicating depth-dependency within the shallower sites. As water depth was not statistically distinct between nearshore and northern sites (t-test assuming unequal variance, $p = 0.65$), these sites were combined for correlation analysis and referenced collectively as the nearshore.

In addition, macroinvertebrate density across all sites was positively correlated with latitude (Spearman's $\rho = 0.44$), indicating an increasing spatial trend from south to north in the lake (Figure 3.2). Without considering offshore sites, macroinvertebrate density was positively correlated with latitude (Spearman's $\rho = 0.45$) and negatively correlated with longitude (Spearman's $\rho = -0.56$), indicating a northeastern trend of increasing macroinvertebrate density within the nearshore. It can be inferred that spatial dependency in macroinvertebrate density results from heterogeneity in the lake.

Table 4.1 Spearman correlation coefficient matrix of site properties and macroinvertebrate density (all sites, $p < 0.05$)

	Density	LAT	LON	Depth	Slope	TS%	VS%	Clay	Silt	Sand	Gravel	As	Cd	Cr	Cu	Pb	Hg	Ni	Zn	PMST
Density	0.44																			
LAT	0.44	0.44																		
LON	-0.66	-0.57	0.44																	
Depth	-0.43	0.49	0.52	0.44																
Slope	0.44	-0.79	0.49	-0.43	0.44															
TS%	0.66	0.66	0.52	0.49	0.44	0.44														
VS%	0.44	0.49	0.52	0.49	0.44	0.44	0.44													
Clay	0.66	0.66	0.52	0.49	0.44	0.44	0.44	0.44												
Silt	0.50	0.50	0.52	0.49	0.44	0.44	0.44	0.44	0.44											
Sand	0.64	0.64	0.52	0.49	0.44	0.44	0.44	0.44	0.44	0.44										
Gravel	0.54	0.45	0.45	0.65	0.71	0.71	0.59	0.81	0.85	0.85	0.71	0.71	0.71	0.71	0.71	0.71	0.71	0.71	0.71	0.71
As	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43	0.43
Cd	-0.64	-0.64	-0.64	-0.64	-0.64	-0.64	-0.64	-0.64	-0.64	-0.64	-0.64	-0.64	-0.64	-0.64	-0.64	-0.64	-0.64	-0.64	-0.64	-0.64
Cr	-0.67	0.71	0.71	0.71	0.71	0.71	0.71	0.71	0.71	0.71	0.71	0.71	0.71	0.71	0.71	0.71	0.71	0.71	0.71	0.71
Cu	-0.55	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54
Pb	-0.51	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73
Hg	-0.44	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54
Ni	-0.60	0.65	0.65	0.65	0.65	0.65	0.65	0.65	0.65	0.65	0.65	0.65	0.65	0.65	0.65	0.65	0.65	0.65	0.65	0.65
Zn	-0.47	0.66	0.66	0.66	0.66	0.66	0.66	0.66	0.66	0.66	0.66	0.66	0.66	0.66	0.66	0.66	0.66	0.66	0.66	0.66
PMST		0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55

Correlation significance: VERY HIGH ... $|p| > 0.9$ HIGH ... $0.9 > |p| > 0.7$ MODERATE ... $0.7 > |p| > 0.4$

Table 4.2 Spearman correlation coefficient matrix of site properties and macroinvertebrate density (offshore sites, $p < 0.05$)

	Density	LAT	LON	Depth	Slope	TS%	VS%	Clay	Silt	Sand	Gravel	As	Cd	Cr	Cu	Pb	Hg	Ni	Zn	PMST
Density																				
LAT																				
LON	0.79																			
Depth	0.70	0.79																		
Slope	0.79	0.70	0.79																	
TS%	-0.66	0.73																		
VS%	0.73	0.79	0.73																	
Clay	-0.66	0.73	0.79	0.73																
Silt	-0.66	0.73	0.79	0.73	-0.66															
Sand	-0.66	0.73	0.79	0.73	-0.66	-0.79														
Gravel	-0.66	0.73	0.79	0.73	-0.66	-0.63	-0.66													
As	-0.66	0.73	0.79	0.73	-0.66	-0.63	-0.66	0.70												
Cd	-0.66	0.73	0.79	0.73	-0.66	-0.63	-0.66	0.70	-0.72											
Cr	-0.66	0.73	0.79	0.73	-0.66	-0.63	-0.66	0.70	-0.72	0.72										
Cu	-0.66	0.73	0.79	0.73	-0.66	-0.63	-0.66	0.70	-0.72	0.72	0.71	0.80								
Pb	-0.66	0.73	0.79	0.73	-0.66	-0.63	-0.66	0.70	-0.72	0.72	0.71	0.80	0.79							
Hg	-0.66	0.73	0.79	0.73	-0.66	-0.63	-0.66	0.70	-0.72	0.72	0.71	0.80	0.79	0.79						
Ni	-0.66	0.73	0.79	0.73	-0.66	-0.63	-0.66	0.70	-0.72	0.72	0.71	0.80	0.79	0.79	0.75					
Zn	-0.66	0.73	0.79	0.73	-0.66	-0.63	-0.66	0.70	-0.72	0.72	0.71	0.80	0.79	0.79	0.75	0.87				
PMST	-0.66	0.73	0.79	0.73	-0.66	-0.63	-0.66	0.70	-0.72	0.72	0.71	0.80	0.79	0.79	0.75	0.87	0.80			

Correlation significance: **VERY HIGH** ... $|p| > 0.9$ **HIGH** ... $0.9 > |p| > 0.7$ **MODERATE** ... $0.7 > |p| > 0.4$

Table 4.3 Spearman correlation coefficient matrix of site properties and macroinvertebrate density (nearshore and northern sites, $p < 0.05$)

	Density	LAT	LON	Depth	Slope	TS%	VS%	Clay	Silt	Sand	Gravel	As	Cd	Cr	Cu	Pb	Hg	Ni	Zn	PMST
Density																				
LAT	0.45		-0.56	-0.46																
LON	-0.56	-0.82																		
Depth	-0.46																			
Slope																				
TS%				-0.59																
VS%																				
Clay																				
Silt																				
Sand																				
Gravel																				
As																				
Cd	-0.49	0.51																		
Cr	-0.51			0.53																
Cu		0.42																		
Pb				0.57																
Hg																				
Ni																				
Zn																				
PMST																				

Correlation significance: VERY HIGH ... $|p| > 0.9$ HIGH ... $0.9 > |p| > 0.7$ MODERATE ... $0.7 > |p| > 0.4$

4.3.1 Tailings, Metals, and Macroinvertebrate Density

Across all sites, macroinvertebrate density was also negatively correlated with all metal concentrations (other than arsenic due to the amount of non-detects, Table 4.1). Metal concentrations tend to exhibit a longitudinal gradient with increasing concentrations from east to west, as well as depth-dependency with greater concentrations in the offshore and along the western shoreline. Previous spatial analysis of metals in Torch Lake describes greater metal concentrations along the western shoreline, except for copper which was consistently elevated across the lake (Mandelia, 2016)¹³. In addition, chromium, mercury, and zinc concentrations tended to increase towards the center of the lake. Spatial variability in metal concentrations is attributed to dredging, varying metal content in ores and waste stream composition during the mining era, and sediment transport mechanics, such as erosion, basin scouring, and focusing (Mandelia, 2016; Urban et al., 2018, Baker, 2007; McDonald et al., 2010).

While macroinvertebrate densities in nearshore were negatively correlated with cadmium and chromium concentrations, they are not correlated with copper (Table 4.3). However, copper is considered the primary driver of sediment toxicity in Torch Lake (Mandelia, 2016; Urban et al., 2018). Copper concentrations were consistently elevated across Torch Lake, with only 2 of the 40 sampled sites falling below TEC levels. Consistently elevated copper concentrations and spatially varying macroinvertebrate densities implies that spatially-dependent factors modulate the effect of copper on the benthos.

Nearshore/northern macroinvertebrate densities were linked with the presence of slime clays (t-test assuming unequal variances, $p < 0.05$), with as the mean densities of $1,513 \pm 745$ and $2,700 \pm 2,112$ individuals m^{-2} were observed with and without the presence of slime clays, respectively. In contrast, the presence of stamps sands in

¹³ Metal concentrations were determined from inclusive sediment samples, incorporating post-mining and mining-era sediment, as well as potential byproducts.

nearshore/northern sediments did not statistically affect macroinvertebrate density (t-test assuming unequal variances, $p = 0.48$). Slime clays have higher copper concentrations than stamp sands due to the larger surface area for adsorption (Kerfoot et al., 2007). Observed presence of tailings did not account for quantity, species, or particle size, other than distinguishing slime clays from stamp sands. Therefore, particle size and habitat preference may drive differences in macroinvertebrate abundance.

4.3.2 Sand versus Silt

Nearshore/northern metal concentrations were negatively correlated with the fraction of sand in sediments (Table 4.3). The eastern shoreline and channel sites exhibited some of the greatest macroinvertebrate densities in the lake, and were primarily composed of natural sand and some stamp sand. On the other hand, sites on the western shoreline (0007, 0018), which were also composed of mostly coarse-grained stamp sand, exhibited some of the lowest macroinvertebrate densities. In addition, sites along the western shoreline on stamp sands piles were much lower in macroinvertebrate density compared to sites near the northern stamp sand pile, which exhibited some of the greatest macroinvertebrate densities in the lake. These differences may be attributed to the proximity of historical mining activity on the western shoreline as well as the proximity to the Trap Rock River on the eastern/northern shoreline.

Offshore macroinvertebrate densities exhibited a high positive correlation with volatile solids (Spearman's $\rho = 0.73$). PMST is positively correlated with water depth and silt content, that are also correlated with volatile solids (Table 4.2). It can be inferred that an organic cap is forming, which may promote detoxification of solid-phase copper through natural organic complexation (Jeong et al., 1999; Kerfoot et al., 2007; Cusack & Mihelcic, 1999).

In contrast, metal concentrations in nearshore were positively correlated with silt content (Table 4.3). Previous core studies have shown the high porosity (70-95%) in deposited, silty sediments facilitates upward diffusion of dissolved metal concentrations

from underlying toxic sediments and tailings (Kerfoot et al., 2007; McDonald et al., 2010).

The difference in PMST and macroinvertebrate densities in the offshore and nearshore may be attributed to lake zones. The littoral zone possesses herbivore populations reliant on plant growth, whereas the profundal zone is dominated by detritivores that rely on organic material produced in the littoral zone (Jansson, 2004). These functional groups interact with sediments differently, resulting in different toxicity sensitivity and routes of exposure (West et al., 1993; Buikema & Benfield, 1979). In addition, physical and chemical characteristics (temperature, oxygen, light, etc.) are different in these two zones, which also modulate the extent to which solid-phase metals are mobilized (Jeong et al., 1999).

Mine tailings, metal concentrations, and macroinvertebrate densities are not directly related. Instead, the effects of tailings and metal concentrations on macroinvertebrate abundance are modulated by spatially dependent physical and chemical properties in the nearshore, as well as the depth dependence of sediment properties between the offshore and nearshore.

4.3.3 Sediment Transport and In-Lake Productivity

In addition to the Trap Rock River, there are other smaller perennial tributaries listed in the National Hydrology Dataset (NHD, <https://viewer.nationalmap.gov/basic/>) for the Torch Lake watershed. However, only a few sampling sites are located near a perennial tributary inlet in the main portion of the lake: 0018, 0009, and 0016. This is in contrast with the proximity of the Trap Rock River in the northern basins, which is effectively extended beyond its denoted inlet by the channelization between the stamp sand pile and the eastern shoreline.

Watershed characteristics and in-lake productivity affect sediment composition, which in turn affects the benthic community (Johnson & Goedkoop). Shoreline wetlands and littoral productivity may increase production of organic material to facilitate natural

organic complexation and detoxification of copper (Kerfoot et. al, 2007; Cusack & Mihelcic, 1999). However, macrophytes are sparse in Torch Lake (USEPA, 1992), and the presence of macrophytes in littoral habitats did not have a statistically significant effect on macroinvertebrate density (t-test assuming unequal variance, $p < 0.05$) in this study. Wetland coverage appears to have increased in recent years along the vegetated stamp sand piles and is relatively high surrounding the northern basins (Figure 2.2). The shift in benthic functional groups may imply increases in primary productivity in the nearshore.

4.4 Future Monitoring Recommendations

Habitat heterogeneity is an important consideration for improving monitoring accuracy and reducing sample variance (Johnson, 1998; Resh & Jackson, 1993). Historic dredging, byproduct deposition, and the inherent variability of large lake systems contribute to Torch Lake's complexity and render this sampling scheme too coarse to effectively determine the effects of localized characteristics on the benthic community.

Spatial patterns in the lake are driven by spatial variation within the nearshore/northern sites. Therefore, refining a nearshore/northern sampling regime would facilitate a more complete assessment of environmental drivers of benthic conditions and recovery. Habitat differences also influence benthic community structure. Stratifying sampling by habitat differences (water depth, lake zone, substrate type, etc.) or expanding the lake-wide sampling for replication of habitat types would improve benthic monitoring, while also assessing shoreline productivity. Similarly, updating temperature and oxygen profiles for Torch Lake would facilitate defining lake zones (littoral, sublittoral, profundal) and also improve benthic monitoring (Jansson, 2004).

The relationships between sediment composition and macroinvertebrate density highlight the importance of physical and chemical characteristics of habitat and water quality in modulating the condition of the benthic community. Few studies on water quality and sediment transport in Torch Lake have been conducted; the focus of historical surveys has been on metal concentrations. Metal concentrations do not appear to be

drastically changing in recent years (cf. Mandelia, 2016), but here we have presented evidence of ecological improvements.

Most importantly, future monitoring efforts require an appropriate reference site to gauge the degree of benthic degradation (Grapentine, 2009; Duffy et al., 2017), which has not yet been incorporated in benthic monitoring surveys (US EPA, 2001; MACTEC, 2008). Lake morphometry, algal productivity, and water chemistry are viewed as good parameters for comparisons of benthic communities between lakes (Bailey et al., 1995; Jackson & Harvey, 1993), further highlighting the need for a better understanding of water quality in Torch Lake to facilitate establishing future reference sites.

In addition, biotic metrics or integrity indices are often better alternatives to intensive taxonomical identification (Beck & Hatch, 2009; Hodgkinson & Jackson, 2005). An effective reference site may facilitate establishing appropriate restoration targets for Torch Lake as well as developing robust biotic metrics for target conditions to simplify benthic monitoring and potentially allow for more frequent surveys to offset annual variability (Resh & Jackson, 1993; Barbour et al., 1999).

5 Conclusions

In comparison with the baseline (1999 and 2000), an overall increase in macroinvertebrate densities was observed. However, there was a large degree of variance within each density distribution.

Macroinvertebrate densities exhibited a northeast gradient across the lake, with the greatest densities observed in the channel portion of the northern basin and along the eastern shoreline to outlet of the lake. These areas also exhibited high diversity, based on functional group, tolerance, and phylum composition and abundance. In contrast, offshore and western shoreline sites had very low macroinvertebrate densities and diversity.

Water depth was a primary driver of benthic community density and composition. Offshore sites generally were low in macroinvertebrate diversity. However, although nearshore and northern sites were statistically similar in water depth distributions, macroinvertebrate densities, tolerance distributions, community structure, and functional group composition varied greatly in different areas in the lake. The northern and the eastern shoreline sites exhibited the greatest observed macroinvertebrate densities across the lake, which may result from proximity to the Trap Rock River, creating currents and depositional zones.

Most metal concentrations displayed a longitudinal gradient, increasing east to west, as well as a depth-dependency, with greater concentrations in deeper water. Most sites exceeded PEC levels for cadmium and copper, but below consensus-based TEC levels, except for cadmium, copper, and nickel. Arsenic and mercury concentrations were low across the lake and copper concentrations were consistently elevated, without spatial trends. Metal concentrations were correlated to high silt content and lower concentrations were observed in high sand sediments.

Mine tailings, metal concentrations, and macroinvertebrate density are not directly related. The presence of stamp sands did not significantly affect macroinvertebrate

density, but organism abundances were lowered in the presence of slime clays. The relative effects of tailings and metal toxicity on macroinvertebrate abundance are modulated by a spatially dependent physical and chemical properties in the nearshore, as well as the depth dependence of sediment properties between the offshore and nearshore.

Variability in density and diversity are attributed to habitat heterogeneity across the lake, specifically within the nearshore/northern sites. Stratifying sampling by habitat differences (water depth, lake zone, substrate type, etc.) to control for habitat variability in the nearshore/northern region would facilitate a more complete assessment of environmental drivers of benthic conditions and recovery. An appropriate reference site is also needed to gauge degree of benthic degradation and to provide future restoration targets for the AOC.

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A Appendix

A.1 Methods Reference Tables

Table A.1.1 Tolerance values for modified Hilsenhoff Family Biotic Index

Order	Family	Tolerance Value	Order	Family	Tolerance Value	Order	Family	Tolerance Value	Order	Family	Tolerance Value
Diptera	Athericidae	2	Amphipoda	Gammaridae	4	Odonata	Aeshnidae	3	Trichoptera	Oligochaeta	8
	Blephariceridae	0		Hyalellidae	8		Calopterygidae	5		Brachycentridae	1
	Blood-red Chironomidae (Chironomini)	8		Talitridae	8		Coenagrionidae	9		Calamoceratidae	3
	Other	6		Baetidae	4		Cordulegastridae	3		Glossosomatidae	0
	Chironomidae	6		Baetiscidae	3		Cordulidae	5		Helicopsychidae	3
	Ceratopogonidae	6		Caenidae	7		Gomphidae	1		Hydropsychidae	4
	Dolichopodidae	4		Ephemerellidae	1		Lestidae	9		Hydroptilidae	4
	Dryopidae	5		Ephemeridae	4		Libellulidae	9		Lepidostomatidae	1
	Elmidae	4		Heptageniidae	4		Macromiidae	3		Leptoceridae	4
	Empididae	6		Leptophlebiidae	2		Capnidae	1		Molannidae	6
Ephemeroptera	Ephydriidae	6	Ephemeroptera	Metretopodidae	2		Chloroperlidae	1		Odontoceridae	0
	Muscidae	6		Oligoneuridae	2		Leuctridae	0		Philopotamidae	3
	Psephenidae	4		Polymitarcyidae	2		Limnephilidae	4		Phryganeidae	4
	Psychodidae	10		Potomanthidae	4	Plecoptera	Nemouridae	2		Polycentropodidae	6
	Simuliidae	6		Siphonuridae	7		Perlidae	1		Psychomyiidae	2
	Syrphidae	10		Tricorythidae	4		Perlodidae	2		Rhyacophilidae	0
	Tabanidae	6		Pyrilidae	5		Pteronarcyidae	0		Sericostomatidae	3
	Tipulidae	3		Sabelidae	6		Taeniopterygidae	2		Uenoidae	3

References: Bode et al., 1996; Hauer and Lamberti, 1988; Hilsenhoff, 1988; Plafkin et al., 1989; Table modified from Mandaville, 2002.

Table A.1.2 Minimum Mass Requirements for Sieve Analysis Subsampling

Site Code	Wet Mass (g)	Site Code	Wet Mass (g)	Site Code	Wet Mass (g)	Site Code	Wet Mass (g)
9901	500	1	300	11	200	1801	300
9902	400	2	600	12	200	1802	300
9903	500	3	400	13	200	1803	300
9904	600	4	100	14	200	1804	100
9905	400	5	100	15	200	1805	200
9906	600	6	100	16	200	1806	100
9907	300	7	100	17	200	1807	100
9908	300	8	200	18	100	1808	400
9909	500	9	100	19	300	1809	300
9910	600	10	200	20	100	1810	500

Reference: ASTM D6913/D6913M-17 (sieve analysis), Method B

Table A.1.3 Minimum Mass Requirements for Hydrometer Analysis Subsampling

Site Code	Wet Mass (g)	Site Code	Wet Mass (g)	Site Code	Wet Mass (g)	Site Code	Wet Mass (g)
9901	300	1	100	11	100	1801	100
9902	200	2	300	12	100	1802	200
9903	200	3	200	13	100	1803	200
9904	300	4	100	14	100	1804	100
9905	200	5	100	15	100	1805	100
9906	300	6	100	16	100	1806	100
9907	100	7	100	17	100	1807	100
9908	200	8	100	18	100	1808	200
9909	300	9	100	19	100	1809	100
9910	300	10	100	20	100	1810	300

Reference: ASTM D7928-17 (hydrometer analysis) for a 152H hydrometer

A.2

Survey Datasets (2018)

Table A.2.1 Benthic community sample information (nearshore sites)

Site Code	Station Location	Sample ID	Collection Date	Sample Time	Process Date	Boat	Sampling Equipment	Latitude	Longitude	Water Depth (m)	Water Depth (ft)
0001	TL0001	TL18-0001-RE	09/03/18	1:56 PM	09/04/18	Jon Boat	Petite Ponar	47.18231	88.40385	3.3	10.8
0002	TL0002	TL18-0002-RE	09/03/18	12:01 PM	09/05/18	Jon Boat	Petite Ponar	47.17873	88.4155	13.9	45.6
0003	TL0003	TL18-0003-RE	09/03/18	10:00 AM	09/07/18	Jon Boat	Petite Ponar	47.15244	88.45279	6.6	21.7
0004	TL0004	TL18-0004-RE	09/03/18	1:47 PM	09/04/18	Jon Boat	Petite Ponar	47.18074	88.39364	1.2	3.9
0005	TL0005	TL18-0005-RE	09/03/18	1:27 PM	09/04/18	Jon Boat	Petite Ponar	47.17404	88.39577	1.3	4.3
0006	TL0006	TL18-0006-RE	09/03/18	1:14 PM	09/05/18	Jon Boat	Petite Ponar	47.16718	88.40016	2.4	7.9
0007	TL0007	TL18-0007-RE	09/03/18	12:10 PM	09/05/18	Jon Boat	Petite Ponar	47.17322	88.42524	5.2	17.1
0008	TL0008	TL18-0008-RE	09/03/18	1:05 PM	09/05/18	Jon Boat	Petite Ponar	47.16072	88.40477	2.2	7.2
0009	TL0009	TL18-0009-RE	09/03/18	12:48 PM	09/05/18	Jon Boat	Petite Ponar	47.15732	88.40902	1.8	5.9
0010	TL0010	TL18-0010-RE	09/03/18	11:00 AM	09/06/18	Jon Boat	Petite Ponar	47.15571	88.41985	2.5	8.2
0011	TL0011	TL18-0011-RE	09/03/18	11:09 AM	09/06/18	Jon Boat	Petite Ponar	47.15672	88.42573	2.5	8.2
0012	TL0012	TL18-0012-RE	09/03/18	9:45 AM	09/06/18	Jon Boat	Petite Ponar	47.15010	88.44279	3	9.8
0013	TL0013	TL18-0013-RE	09/03/18	9:36 AM	09/06/18	Jon Boat	Petite Ponar	47.14314	88.44776	1.6	5.2
0014	TL0014	TL18-0014-RE	09/03/18	8:34 AM	09/06/18	Jon Boat	Petite Ponar	47.13393	88.45474	1.7	5.6
0015	TL0015	TL18-0015-RE	09/03/18	9:01 AM	09/07/18	Jon Boat	Petite Ponar	47.13851	88.46136	1.5	4.9
0016	TL0016	TL18-0016-RE	09/03/18	10:16 AM	09/07/18	Jon Boat	Petite Ponar	47.15621	88.44414	3.3	10.8
0017	TL0017	TL18-0017-RE	09/03/18	10:27 AM	09/07/18	Jon Boat	Petite Ponar	47.15984	88.4349	7.2	23.6
0018	TL0018	TL18-0018-RE	09/03/18	9:15 AM	09/07/18	Jon Boat	Petite Ponar	47.14540	88.45744	1.3	4.3
0019	TL0019	TL18-0019-RE	09/03/18	12:23 PM	09/05/18	Jon Boat	Petite Ponar	47.17065	88.42991	7.8	25.6
0020	TL0020	TL18-0020-RE	09/03/18	12:35 PM	09/06/18	Jon Boat	Petite Ponar	47.16386	88.42541	5.2	17.1

Nearshore sites

Table A.2.2 Sediment sample information (nearshore sites)

Site Code	Station Location	Sample ID	Collection Date	Sample Time	Boat	Sampling Equipment	Latitude	Longitude	Water Depth (m)	Water Depth (ft)
0001	TL0001	TL18-0001-SED	08/07/18	10:00 AM	R/V Agassiz	Petite Ponar	47.18228	-88.40373	0.5	1.6
0002	TL0002	TL18-0002-SED	08/06/18	4:40 PM	R/V Agassiz	Petite Ponar	47.1788	-88.41544	11.9	39.0
0003	TL0003	TL18-0003-SED	08/06/18	10:45 AM	R/V Agassiz	Petite Ponar	47.15251	-88.45283	3.8	12.5
0004	TL0004	TL18-0004-SED	08/09/18	12:40 PM	Polar	Petite Ponar	47.18071	-88.39349	1.2	3.9
0005	TL0005	TL18-0005-SED	08/09/18	1:10 PM	Polar	Petite Ponar	47.17404	-88.39586	1.4	4.6
0006	TL0006	TL18-0006-SED	08/09/18	1:40 PM	Polar	Petite Ponar	47.16709	-88.40047	2.4	7.9
0007	TL0007	TL18-0007-SED	08/06/18	3:40 PM	R/V Agassiz	Petite Ponar	47.17321	-88.42493	2.8	9.2
0008	TL0008	TL18-0008-SED	08/09/18	2:00 PM	Polar	Petite Ponar	47.16069	-88.40188	2.2	7.2
0009	TL0009	TL18-0009-SED	08/09/18	2:30 PM	Polar	Petite Ponar	47.15733	-88.40919	1.8	5.9
0010	TL0010	TL18-0010-SED	08/09/18	8:20 AM	Polar	Petite Ponar	47.1557	-88.41953	2.5	8.2
0011	TL0011	TL18-0011-SED	08/09/18	8:50 AM	Polar	Petite Ponar	47.15546	-88.42016	2.5	8.2
0012	TL0012	TL18-0012-SED	08/06/18	11:10 AM	R/V Agassiz	Petite Ponar	47.14999	-88.4429	1.3	4.3
0013	TL0013	TL18-0013-SED	08/09/18	9:20 AM	Polar	Petite Ponar	47.1455	-88.45129	1.6	5.2
0014	TL0014	TL18-0014-SED	08/09/18	10:00 AM	Polar	Petite Ponar	47.1339056	-88.45469	1.5	4.9
0015	TL0015	TL18-0015-SED	08/09/18	10:20 AM	Polar	Petite Ponar	47.1351	-88.46132	1.5	4.9
0016	TL0016	TL18-0016-SED	08/09/18	11:15 AM	Polar	Petite Ponar	47.15616	-88.44419	3.3	10.8
0017	TL0017	TL18-0017-SED	08/06/18	12:05 PM	R/V Agassiz	Petite Ponar	47.15986	-88.43487	4.7	15.4
0018	TL0018	TL18-0018-SED	08/09/18	10:45 PM	Polar	Petite Ponar	47.14532	-88.4573	1.3	4.3
0019	TL0019	TL18-0019-SED	08/06/18	1:45 PM	R/V Agassiz	Petite Ponar	47.17066	-88.42997	5.5	18.0
0020	TL0020	TL18-0020-SED	08/06/18	12:30 PM	R/V Agassiz	Petite Ponar	47.16395	-88.42545	2.8	9.2

Table A.2.3 Benthic community sample information (offshore sites)

Site Code	Station Location	Sample ID	Collection Date	Sample Time	Process Date	Boat	Sampling Equipment	Latitude	Longitude	Water Depth (m)	Water Depth (ft)
9901	TL9901	TL18-9901-BEN	08/07/18	10:50 AM	08/07/18	R/V Agassiz	Full Ponar	47.18297	-88.41146	21.3	69.9
9902	TL9905	TL18-9902-BEN	08/06/18	4:00 PM	08/06/18	R/V Agassiz	Full Ponar	47.17428	-88.41332	26.5	86.9
9903	TL9906	TL18-9903-BEN	08/06/18	3:15 PM	08/06/18	R/V Agassiz	Full Ponar	47.17162	-88.42209	27.5	90.2
9904	TL9903	TL18-9904-BEN	08/07/18	9:25 AM	08/07/18	R/V Agassiz	Full Ponar	47.17589	-88.39853	18.9	62.0
9905	TL9909	TL18-9905-BEN	08/07/18	9:00 AM	08/07/18	R/V Agassiz	Full Ponar	47.1665	-88.41062	27	88.6
9906	TL9907	TL18-9906-BEN	08/06/18	1:19 PM	08/06/18	R/V Agassiz	Full Ponar	47.16968	-88.42789	18.4	60.4
9907	TL9910	TL18-9907-BEN	08/07/18	8:30 AM	08/07/18	R/V Agassiz	Full Ponar	47.1596	-88.41413	11.7	38.4
9908	TL9912	TL18-9908-BEN	08/06/18	9:45 AM	08/06/18	R/V Agassiz	Full Ponar	47.15209	-88.44955	17.7	58.1
9909	TL9914	TL18-9909-BEN	08/06/18	9:21 AM	08/06/18	R/V Agassiz	Full Ponar	47.14433	-88.45462	14.3	46.9
9910	TL9915	TL18-9910-BEN	08/06/18	8:40 AM	08/06/18	R/V Agassiz	Full Ponar	47.13783	-88.45555	7.9	25.9

Offshore sites

Table A.2.4 Sediment sample information (offshore sites)

Site Code	Station Location	Sample ID	Collection Date	Sample Time	Boat	Sampling Equipment	Latitude	Longitude	Water Depth (m)	Water Depth (ft)
0001	TL0001	TL18-0001-SED	08/07/18	10:00 AM	R/V Agassiz	Petite Ponar	47.18228	-88.40373	0.5	1.6
0002	TL0002	TL18-0002-SED	08/06/18	4:40 PM	R/V Agassiz	Petite Ponar	47.1788	-88.41544	11.9	39.0
0003	TL0003	TL18-0003-SED	08/06/18	10:45 AM	R/V Agassiz	Petite Ponar	47.15251	-88.45283	3.8	12.5
0004	TL0004	TL18-0004-SED	08/09/18	12:40 PM	Polar	Petite Ponar	47.18071	-88.39349	1.2	3.9
0005	TL0005	TL18-0005-SED	08/09/18	1:10 PM	Polar	Petite Ponar	47.17404	-88.39586	1.4	4.6
0006	TL0006	TL18-0006-SED	08/09/18	1:40 PM	Polar	Petite Ponar	47.16709	-88.40047	2.4	7.9
0007	TL0007	TL18-0007-SED	08/06/18	3:40 PM	R/V Agassiz	Petite Ponar	47.17321	-88.42493	2.8	9.2
0008	TL0008	TL18-0008-SED	08/09/18	2:00 PM	Polar	Petite Ponar	47.16069	-88.40188	2.2	7.2
0009	TL0009	TL18-0009-SED	08/09/18	2:30 PM	Polar	Petite Ponar	47.15733	-88.40919	1.8	5.9
0010	TL0010	TL18-0010-SED	08/09/18	8:20 AM	Polar	Petite Ponar	47.1557	-88.41953	2.5	8.2

Offshore sites

Table A.2.5 Benthic community sample information (northern sites)

Site Code	Station Location	Sample ID	Collection Date	Sample Time	Process Date	Boat	Sampling Equipment	Latitude	Longitude	Water Depth (m)	Water Depth (ft)
1801	TL1801	TL18-1801-RE	09/03/18	3:50 PM	09/12/18	Jon Boat	Petite Ponar	47.19185	88.39528	9.1	29.9
1802	TL1802	TL18-1802-RE	09/03/18	4:12 PM	09/12/18	Jon Boat	Petite Ponar	47.19135	88.39474	8.8	28.9
1803	TL1803	TL18-1803-RE	09/03/18	4:31 PM	09/04/18	Jon Boat	Petite Ponar	47.18995	88.39513	5.2	17.1
1804	TL1804	TL18-1804-RE	09/03/18	4:21 PM	09/11/18	Jon Boat	Petite Ponar	47.19103	88.39619	4.4	14.4
1805	TL1805	TL18-1805-RE	09/03/18	4:00 PM	09/12/18	Jon Boat	Petite Ponar	47.19152	88.39347	3	9.8
1806	TL1806	TL18-1806-RE	09/03/18	4:43 PM	09/04/18	Jon Boat	Petite Ponar	47.1876	88.39697	2	6.6
1807	TL1807	TL18-1807-RE	09/03/18	4:53 PM	09/04/18	Jon Boat	Petite Ponar	47.18454	88.39609	1.7	5.6
1808	TL1808	TL18-1808-RE	09/03/18	3:10 PM	09/11/18	Jon Boat	Petite Ponar	47.19262	88.40576	3.2	10.5
1809	TL1809	TL18-1809-RE	09/03/18	3:22 PM	09/11/18	Jon Boat	Petite Ponar	47.19528	88.40339	2.8	9.2
1810	TL1810	TL18-1810-RE	09/03/18	3:34 PM	09/11/18	Jon Boat	Petite Ponar	47.19444	88.4005	2.6	8.5

Northern sites

Table A.2.6 Sediment sample information (northern sites)

Site Code	Station Location	Sample ID	Collection Date	Sample Time	Boat	Sampling Equipment	Latitude	Longitude	Water Depth (m)	Water Depth (ft)
0001	TL0001	TL18-0001-SED	08/07/18	10:00 AM	R/V Agassiz	Petite Ponar	47.18228	-88.40373	0.5	1.6
0002	TL0002	TL18-0002-SED	08/06/18	4:40 PM	R/V Agassiz	Petite Ponar	47.1788	-88.41544	11.9	39.0
0003	TL0003	TL18-0003-SED	08/06/18	10:45 AM	R/V Agassiz	Petite Ponar	47.15251	-88.45283	3.8	12.5
0004	TL0004	TL18-0004-SED	08/09/18	12:40 PM	Polar	Petite Ponar	47.18071	-88.39349	1.2	3.9
0005	TL0005	TL18-0005-SED	08/09/18	1:10 PM	Polar	Petite Ponar	47.17404	-88.39586	1.4	4.6
0006	TL0006	TL18-0006-SED	08/09/18	1:40 PM	Polar	Petite Ponar	47.16709	-88.40047	2.4	7.9
0007	TL0007	TL18-0007-SED	08/06/18	3:40 PM	R/V Agassiz	Petite Ponar	47.17321	-88.42493	2.8	9.2
0008	TL0008	TL18-0008-SED	08/09/18	2:00 PM	Polar	Petite Ponar	47.16069	-88.40188	2.2	7.2
0009	TL0009	TL18-0009-SED	08/09/18	2:30 PM	Polar	Petite Ponar	47.15733	-88.40919	1.8	5.9
0010	TL0010	TL18-0010-SED	08/09/18	8:20 AM	Polar	Petite Ponar	47.1557	-88.41953	2.5	8.2

Northern sites

Table A.2.7 Benthic community results for nearshore sites (0001-0010)

Phylum	Class	Subclass	Order	Suborder	Family	Subfamily	Site										Tolerance ⁽⁵⁾	Functional Groups ⁽⁶⁾															
							Total No. Individuals Depth of Sample (m)																										
Annelida	NSI	Oligochaeta	NSI	NSI	NSI	NSI	NSI	0001	0002	0003	0004	0005	0006	0007	0008	0009	0010																
								42	10	18	225	57	72	5	56	96	39																
	Nematoda	Arachnida	Brachiopoda	Diplostira	NSI	NSI	NSI	NSI	5	1	15	29	6	1	40	20	1																
									8	1	1	4	6	10	1	34	8																
									5								5																
									8								8																
									8								8																
									8								8																
									8								8																
									8								8																
Arthropoda	Crustacea	Malacostraca	Copepoda	Amphipoda	Cyclopoida	Cyclopoidae	Gammarus	NSI	Mysis	Mysidae	NSI	NSI	NSI	NSI	NSI	NSI	NSI																
																		4	OM	4	GC												
																		8															
																		4															
																		8															
																		4															
																		8															
																		4															
																		8															
																		4															
Mollusca	Bivalvia	Gastropoda	Sphaeniidae	Placidum	Hydrobiidae	Physidae	Planorbidae	Valatidae	NSI	NSI	NSI	NSI	NSI	NSI	NSI	NSI	NSI																
																		22	2	45	2	7	2	19	31								
																		8	7	6	15	2	1	8	6								
																		8	7	6	15	2	1	8	6								
																		8	7	6	15	2	1	8	6								
																		8	7	6	15	2	1	8	6								
																		8	7	6	15	2	1	8	6								
																		8	7	6	15	2	1	8	6								
																		8	7	6	15	2	1	8	6								
																		8	7	6	15	2	1	8	6								
Mollusca	Bivalvia	Gastropoda	Sphaeniidae	Placidum	Hydrobiidae	Physidae	Planorbidae	Valatidae	NSI	NSI	NSI	NSI	NSI	NSI	NSI	NSI	NSI																
																		22	2	45	2	7	2	19	31								
																		8	7	6	15	2	1	8	6								
																		8	7	6	15	2	1	8	6								
																		8	7	6	15	2	1	8	6								
																		8	7	6	15	2	1	8	6								
																		8	7	6	15	2	1	8	6								
																		8	7	6	15	2	1	8	6								
																		8	7	6	15	2	1	8	6								
																		8	7	6	15	2	1	8	6								

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NSI – Not Specifically Identified;⁽¹⁾ References: Barbour et al., 1999; Boite et al., 1996; ⁽²⁾FC – Filterer-collector, GC – Gather-collector, PR – Predator, SC – Scraper, SH – Shredder, PA – Parasite, OM – Omnivore, PI – Placer

Nearshore

Table A.2.11 Solid-phase heavy metals in sediment samples (nearshore sites)

Site Code	Station Location	Sample ID	Collection Date	Units	Arsenic		Cadmium		Chromium		Copper		Lead		Mercury		Nickel		Zinc	
					R	Q	R	Q	R	Q	R	Q	R	Q	R	Q	R	Q	R	Q
0001	TL0001	TL18-0001-SED	8/7/18	mg/kg	ND		6.1		45		1000		34		0.052		47		130	
0002	TL0002	TL18-0002-SED	8/6/18	mg/kg	5.1	J	11		61		2000		130		0.085		58		210	
0003	TL0003	TL18-0003-SED	8/6/18	mg/kg	ND		8.9		46		1400		30		0.083		45		130	
0004	TL0004	TL18-0004-SED	8/9/18	mg/kg	ND		1.5		11		120		3	J	ND		12		25	
0005	TL0005	TL18-0005-SED	8/9/18	mg/kg	ND		0.5	J	2.7		21		ND		ND		2	J	8.4	
0006	TL0006	TL18-0006-SED	8/9/18	mg/kg	ND		0.5	J	3.2		29		2.2	J	ND		2.6	J	8	
0007	TL0007	TL18-0007-SED	8/6/18	mg/kg	ND	M	14		700		7200		130	M	ND		14		600	
0008	TL0008	TL18-0008-SED	8/9/18	mg/kg	ND		0.7	J	4.9		89		3.5	J	0.009	J	5.8		16	
0009	TL0009	TL18-0009-SED	8/9/18	mg/kg	ND		0.4	J	2.4		35		ND		ND		3.2		8.3	
0010	TL0010	TL18-0010-SED	8/9/18	mg/kg	ND		2.1		14		310		5.9	J	0.055		12		35	
0011	TL0011	TL18-0011-SED	8/9/18	mg/kg	ND		6.8		48		740		22		ND		46		110	
0012	TL0012	TL18-0012-SED	8/6/18	mg/kg	ND		7.7		44		1000		19		0.13		48		110	
0013	TL0013	TL18-0013-SED	8/9/18	mg/kg	ND		1.4		7.8		210		6.6	J	ND		7.4		19	
0014	TL0014	TL18-0014-SED	8/9/18	mg/kg	ND		6.4		30		700		16		0.12		30		77	
0015	TL0015	TL18-0015-SED	8/9/18	mg/kg	ND	M	10		41		1500		21		0.27		41		100	
0016	TL0016	TL18-0016-SED	8/9/18	mg/kg	ND		4.8		31		570		13		0.049		29		77	
0017	TL0017	TL18-0017-SED	8/6/18	mg/kg	ND		5.8		37		930		12		0.052		32		80	
0018	TL0018	TL18-0018-SED	8/9/18	mg/kg	ND		7.2		30		2900		9.4	J	0.041		37		81	
0019	TL0019	TL18-0019-SED	8/6/18	mg/kg	ND		5.8		34		1700		50		0.073		32		120	
0020	TL0020	TL18-0020-SED	8/6/18	mg/kg	ND		5.5		38		360		8.5	J	0.009	J	32		56	
			PEC	mg/kg	33		5		111		149		128		1.06		49		459	
			TEC	mg/kg	9.8		1		43		31.6		36		0.18		23		121	

R – Results; Q – Laboratory Qualifier; ND – Not Detected; J – The quantitation is an estimated value because the results is less than the sample quantitation limit but greater than the detection limit; M – A matrix effect was present

Table A.2.12 Solid-phase heavy metals in sediment samples (offshore and northern sites)

Site Code	Station Location	Sample ID	Collection Date	Units	Arsenic			Cadmium			Chromium			Copper			Lead			Mercury			Nickel			Zinc		
					R	Q		R	Q		R	Q		R	Q		R	Q		R	Q		R	Q		R	Q	
9901	TL9901	TL18-9901-SED	8/7/18	mg/kg	ND			6.7			40			1300			70			0.091			42			140		
9902	TL9905	TL18-9902-SED	8/6/18	mg/kg	5.4	J		8.8			63			1900			89			0.099			61			230		
9903	TL9906	TL18-9903-SED	8/6/18	mg/kg	10			9.6			69			2200			130			0.12			66			240		
9904	TL9903	TL18-9904-SED	8/7/18	mg/kg	ND			6.3			45			1200			44			0.092			43			150		
9905	TL9909	TL18-9905-SED	8/7/18	mg/kg	ND			10			62			1700			71			0.086			60			210		
9906	TL9907	TL18-9906-SED	8/6/18	mg/kg	ND			6			44			1300			55			0.087			44			120		
9907	TL9910	TL18-9907-SED	8/7/18	mg/kg	5.3	J		9.3			64			1400			51			0.088			68			180		
9908	TL9912	TL18-9908-SED	8/6/18	mg/kg	ND			9.6			60			1800			47			0.12			57			190		
9909	TL9914	TL18-9909-SED	8/6/18	mg/kg	ND			9.5			53			1600			51			0.16			54			160		
9910	TL9915	TL18-9910-SED	8/6/18	mg/kg	ND	M		11			53			1400			34			0.15			51			140		
1801	TL1801	TL18-1801-SED	8/15/18	mg/kg	ND			3.7			22			440			12			0.072			19			60		
1802	TL1802	TL18-1802-SED	8/15/18	mg/kg	ND			5.2			32			600			18			0.059			25			82		
1803	TL1803	TL18-1803-SED	8/15/18	mg/kg	ND			5.5			35			650			20			0.077			28			91		
1804	TL1804	TL18-1804-SED	8/15/18	mg/kg	ND			3.2			26	M		800			7.3	J		ND			21			65		
1805	TL1805	TL18-1805-SED	8/15/18	mg/kg	ND			2.6			17			280			10			0.074			15			50		
1806	TL1806	TL18-1806-SED	8/14/18	mg/kg	ND			3.4			27			590			7.1	J		ND			31			73		
1807	TL1807	TL18-1807-SED	8/14/18	mg/kg	ND			3			24			490			5	J		ND			29			54		
1808	TL1808	TL18-1808-SED	8/14/18	mg/kg	ND			7			43			1700			78			0.086			41			230		
1809	TL1809	TL18-1809-SED	8/14/18	mg/kg	ND			5.4			30			1300			54			0.11			27			210		
1810	TL1810	TL18-1810-SED	8/14/18	mg/kg	ND			7.4			37			1700			47			0.061			32			200		
			PEC	mg/kg	33			5			111			149			128			1.06			49			459		
			TEC	mg/kg	9.8			1			43			31.6			36			0.18			23			121		

Offshore sites

Northern sites

R – Results; Q – Laboratory Qualifier; ND – Not Detected; J – The quantitation is an estimated value because the results is less than the sample quantitation limit but greater than the detection limit; M – A matrix effect was present

Table A.2.13 Total solids, water content, and volatile solids content in sediment samples (nearshore sites)

Site Code	Station Location	Sample ID	Collection Date	Units	Average Total Solids (TS)	SDV (±)	Average Water Content (WC)	SDV (±)	Average Volatile Solids (%)	SDV (%)
0001	TL0001	TL18-0001-SED	8/7/18	%	33.5	0.2	189.4	9.2	6.25	0.08
0002	TL0002	TL18-0002-SED	8/6/18	%	16.6	0.5	456.0	58.4	12.67	0.80
0003	TL0003	TL18-0003-SED	8/6/18	%	24.8	0.8	274.9	36.0	7.17	1.05
0004	TL0004	TL18-0004-SED	8/9/18	%	71.2	0.2	39.8	0.8	0.35	0.07
0005	TL0005	TL18-0005-SED	8/9/18	%	73.3	0.1	35.1	0.0	0.35	0.04
0006	TL0006	TL18-0006-SED	8/9/18	%	71.4	1.0	39.3	1.3	0.56	0.17
0007	TL0007	TL18-0007-SED	8/6/18	%	85.0	1.0	17.5	1.6	0.10	0.01
0008	TL0008	TL18-0008-SED	8/9/18	%	65.7	1.9	50.9	3.1	0.75	0.01
0009	TL0009	TL18-0009-SED	8/9/18	%	74.0	0.8	34.7	1.1	0.39	0.04
0010	TL0010	TL18-0010-SED	8/9/18	%	57.9	0.6	70.5	0.9	1.57	0.18
0011	TL0011	TL18-0011-SED	8/9/18	%	51.8	1.6	89.5	8.8	2.20	0.20
0012	TL0012	TL18-0012-SED	8/6/18	%	47.1	1.0	105.5	3.5	4.39	0.36
0013	TL0013	TL18-0013-SED	8/9/18	%	62.0	1.7	59.6	5.8	2.36	0.22
0014	TL0014	TL18-0014-SED	8/9/18	%	57.7	0.4	70.6	3.3	2.27	0.28
0015	TL0015	TL18-0015-SED	8/9/18	%	58.4	0.5	69.4	2.9	1.86	0.02
0016	TL0016	TL18-0016-SED	8/9/18	%	62.8	0.2	57.3	1.2	1.23	0.06
0017	TL0017	TL18-0017-SED	8/6/18	%	50.3	0.1	96.1	2.7	2.23	0.04
0018	TL0018	TL18-0018-SED	8/9/18	%	77.6	1.2	28.4	1.6	0.69	0.30
0019	TL0019	TL18-0019-SED	8/6/18	%	35.0	0.1	176.2	9.3	7.84	0.02
0020	TL0020	TL18-0020-SED	8/6/18	%	70.1	0.2	41.7	0.4	0.45	0.09

Nearshore sites

Table A.2.14 Total solids, water content, and volatile solids content in sediment samples (offshore and northern sites)

Site Code	Station Location	Sample ID	Collection Date	Units	Average Total Solids (TS)	SDV (±)	Average Water Content (WC)	SDV (±)	Average Volatile Solids (%)	SDV (%)
9901	TL9901	TL18-9901-SED	8/7/18	%	28.6	5.5	242.2	48.3	9.36	0.24
9902	TL9905	TL18-9902-SED	8/6/18	%	15.9	2.4	502.1	142.3	12.90	0.08
9903	TL9906	TL18-9903-SED	8/6/18	%	17.5	2.0	450.4	102.3	11.27	0.22
9904	TL9903	TL18-9904-SED	8/7/18	%	16.3	1.4	438.4	29.6	11.54	0.58
9905	TL9909	TL18-9905-SED	8/7/18	%	15.7	0.3	494.1	47.7	12.52	0.63
9906	TL9907	TL18-9906-SED	8/6/18	%	19.7	0.4	384.0	44.7	10.33	0.09
9907	TL9910	TL18-9907-SED	8/7/18	%	18.0	2.5	428.2	120.3	10.46	0.36
9908	TL9912	TL18-9908-SED	8/6/18	%	17.2	0.9	448.5	79.0	8.99	0.72
9909	TL9914	TL18-9909-SED	8/6/18	%	22.4	0.8	330.4	38.9	7.68	0.28
9910	TL9915	TL18-9910-SED	8/6/18	%	21.2	6.6	364.8	117.7	5.26	0.15
1801	TL1801	TL18-1801-SED	8/15/18	%	37.7	0.7	152.1	4.3	10.40	0.08
1802	TL1802	TL18-1802-SED	8/15/18	%	32.4	1.3	198.0	2.4	13.48	0.17
1803	TL1803	TL18-1803-SED	8/15/18	%	30.3	1.6	223.1	28.7	13.51	0.23
1804	TL1804	TL18-1804-SED	8/15/18	%	69.9	1.6	41.9	4.3	0.71	0.01
1805	TL1805	TL18-1805-SED	8/15/18	%	39.1	3.2	152.3	26.5	10.41	0.11
1806	TL1806	TL18-1806-SED	8/14/18	%	74.8	0.5	33.0	0.6	0.49	0.00
1807	TL1807	TL18-1807-SED	8/14/18	%	78.2	0.5	27.9	0.8	0.32	0.05
1808	TL1808	TL18-1808-SED	8/14/18	%	23.6	0.2	292.3	30.8	13.33	0.17
1809	TL1809	TL18-1809-SED	8/14/18	%	35.2	0.1	172.5	11.2	7.71	0.21
1810	TL1810	TL18-1810-SED	8/14/18	%	18.4	0.8	371.2	9.7	15.06	0.28

Offshore sites

Northern sites

Table A.2.15 Particle size distribution in sediment samples (nearshore sites)

USDA Particle Size Limit Classification (mm)														
					<0.002	>0.002-0.05	>0.05-0.1	>0.1-0.25	>0.25-0.5	>0.5-1	>1-2	>2-10		
Site Code	Station Location	Sample ID	Collection Date	Units	Clay Fine	Silt Fine	Very Fine Sand	Fine Sand	Medium Sand	Course Sand	Very Course Sand	Fine Gravel	Majority (%)	Majority Grain Size
0001	TL0001	TL18-0001-SED	8/7/18	%	2.0	32.5	42.7	21.3	1.1	0.2	0.1	0.1	42.7	Very Fine Sand
0002	TL0002	TL18-0002-SED	8/6/18	%	1.7	34.3	49.5	13.6	0.4	0.2	0.1	0.2	49.5	Very Fine Sand
0003	TL0003	TL18-0003-SED	8/6/18	%	0	39.0	46.5	13.8	0.2	0.2	0.1	0.2	46.5	Very Fine Sand
0004	TL0004	TL18-0004-SED	8/9/18	%	2.2	8.3	1.5	70.9	16.5	0.3	0.2	0.1	70.9	Fine Sand
0005	TL0005	TL18-0005-SED	8/9/18	%	0.5	1.7	4.3	66.9	23.7	2.4	0.1	0.2	66.9	Fine Sand
0006	TL0006	TL18-0006-SED	8/9/18	%	0.5	3.7	6.5	76.2	10.1	2.1	0.7	0.2	76.2	Fine Sand
0007	TL0007	TL18-0007-SED	8/6/18	%	1.0	1.2	0.5	1.1	0.5	7.3	12.9	51.8	51.8	Fine Gravel
0008	TL0008	TL18-0008-SED	8/9/18	%	1.2	1.5	3.8	84.5	7.3	1.3	0.3	0.2	84.5	Fine Sand
0009	TL0009	TL18-0009-SED	8/9/18	%	1.2	0.3	0.5	66.9	27.4	3.3	0.2	0.3	66.9	Fine Sand
0010	TL0010	TL18-0010-SED	8/9/18	%	16.0	31.7	27.0	16.0	4.3	2.2	2.0	0.6	31.7	Silt Fine
0011	TL0011	TL18-0011-SED	8/9/18	%	13.5	44.7	30.3	9.2	1.1	0.2	0.4	0.5	44.7	Silt Fine
0012	TL0012	TL18-0012-SED	8/6/18	%	11.7	30.3	36.5	15.2	4.5	1.0	0.6	0.2	36.5	Very Fine Sand
0013	TL0013	TL18-0013-SED	8/9/18	%	5.2	12.5	16.0	51.4	11.0	2.1	0.9	0.4	51.4	Fine Sand
0014	TL0014	TL18-0014-SED	8/9/18	%	24.0	34.0	31.5	9.7	0.4	0.2	0.1	0	34.0	Silt Fine
0015	TL0015	TL18-0015-SED	8/9/18	%	4.7	47.3	36.5	11.1	0.2	0	0.1	0.1	47.3	Silt Fine
0016	TL0016	TL18-0016-SED	8/9/18	%	11.5	27.0	24.2	36.6	0.5	0.2	0	0	36.6	Fine Sand
0017	TL0017	TL18-0017-SED	8/6/18	%	2.0	33.0	38.0	26.2	0.4	0.1	0.1	0.2	38.0	Very Fine Sand
0018	TL0018	TL18-0018-SED	8/9/18	%	0.5	0.5	0.2	12.7	19.7	20.6	21.2	22.3	22.3	Fine Gravel
0019	TL0019	TL18-0019-SED	8/6/18	%	3.0	23.7	31.0	28.8	3.0	2.7	3.2	4.2	31.0	Very Fine Sand
0020	TL0020	TL18-0020-SED	8/6/18	%	1.0	2.0	4.5	65.2	24.6	2.6	0	0.1	65.2	Fine Sand

ND - Not Detected

Table A.2.16 Particle size distribution in sediment samples (offshore and nearshore sites)

					USDA Particle Size Limit Classification (mm)								
					<0.002	>0.002-0.05	>0.05-0.1	>0.1-0.25	>0.25-0.5	>0.5-1	>1-2	>2-10	
Site Code	Station Location	Sample ID	Collection Date	Units	Clay Fine	Silt Fine	Very Fine Sand	Fine Sand	Medium Sand	Course Sand	Very Course Sand	Fine Gravel	Majority Grain Size
9901	TL9901	TL18-9901-SED	8/7/18	%	9.5	58.5	20.5	10.9	0.3	0.2	0	0.1	Silt Fine
9902	TL9905	TL18-9902-SED	8/6/18	%	29.0	49.5	9.0	11.8	0.4	0.1	0	0.2	Silt Fine
9903	TL9906	TL18-9903-SED	8/6/18	%	31.0	49.0	10.0	9.1	0.3	0.2	0.2	0.2	Silt Fine
9904	TL9903	TL18-9904-SED	8/7/18	%	15.0	58.5	8.2	16.9	0.8	0.4	0.1	0.1	Silt Fine
9905	TL9909	TL18-9905-SED	8/7/18	%	28.7	48.5	10.8	11.2	0.2	0.2	0.3	0.1	Silt Fine
9906	TL9907	TL18-9906-SED	8/6/18	%	12.2	57.5	19.3	10.1	0.4	0.2	0.1	0.2	Silt Fine
9907	TL9910	TL18-9907-SED	8/7/18	%	22.0	44.0	13.5	17.0	2.7	0.4	0.2	0.2	Silt Fine
9908	TL9912	TL18-9908-SED	8/6/18	%	28.7	53.5	5.8	11.2	0.6	0.1	0	0.1	Silt Fine
9909	TL9914	TL18-9909-SED	8/6/18	%	20.2	64.0	6.8	8.7	0.1	0.1	0	0.1	Silt Fine
9910	TL9915	TL18-9910-SED	8/6/18	%	20.5	69.5	2.0	7.3	0.3	0.3	0	0.1	Silt Fine
1801	TL1801	TL18-1801-SED	8/15/18	%	6.0	32.5	22.2	37.7	0.7	0.4	0.3	0.2	Fine Sand
1802	TL1802	TL18-1802-SED	8/15/18	%	6.0	49.7	29.0	15.0	0.2	0.1	0	0.1	Silt Fine
1803	TL1803	TL18-1803-SED	8/15/18	%	7.0	50.5	31.2	10.3	0.6	0.3	0	0.1	Silt Fine
1804	TL1804	TL18-1804-SED	8/15/18	%	0	5.0	5.7	60.3	20.3	5.4	2.8	0.5	Medium Sand
1805	TL1805	TL18-1805-SED	8/15/18	%	6.0	40.0	32.7	20.9	0.3	0.1	0	0	Silt Fine
1806	TL1806	TL18-1806-SED	8/14/18	%	0.7	0.3	0	14.3	47.8	21.1	14.9	0.9	Medium Sand
1807	TL1807	TL18-1807-SED	8/14/18	%	ND	ND	ND	11.2	29.6	30.2	27.3	1.2	Medium Sand
1808	TL1808	TL18-1808-SED	8/14/18	%	14.0	64.7	15.5	5.3	0.2	0.1	0	0.2	Silt Fine
1809	TL1809	TL18-1809-SED	8/14/18	%	10.7	46.3	31.0	11.6	0.1	0	0.1	0.1	Silt Fine
1810	TL1810	TL18-1810-SED	8/14/18	%	7.2	30.8	36.0	24.9	0.5	0.3	0.2	0.1	Silt Fine
ND - Not Detected													

ND - Not Detected

Offshore sites

Northern sites

Table A.2.17 Sediment samples observations (nearshore sites)

Site Code	Station Location	Sample ID	Collection Date	Post-Mining Sediment Thickness (cm)	Observed Slime Clays (Y/N)	Observed Slime Clays (Y/N)	Observed Macrophytes (Y/N)	Color
0001	TL0001	TL18-0001-SED	8/7/18	3-4	Y	N	N	Brown
0002	TL0002	TL18-0002-SED	8/6/18	<1	N	Y	N	Brown
0003	TL0003	TL18-0003-SED	8/6/18	2	Y	Y	Y	Light Brown
0004	TL0004	TL18-0004-SED	8/9/18	NR	Y	N	N	Light Brown
0005	TL0005	TL18-0005-SED	8/9/18	NR	Y	N	Y	Light Brown
0006	TL0006	TL18-0006-SED	8/9/18	NR	N	N	N	Light Brown
0007	TL0007	TL18-0007-SED	8/6/18	NR	Y	N	Y	Dark Brown, Red
0008	TL0008	TL18-0008-SED	8/9/18	NR	N	N	N	Light Brown
0009	TL0009	TL18-0009-SED	8/9/18	NR	N	N	Y	Light Brown
0010	TL0010	TL18-0010-SED	8/9/18	<1	N	Y	Y	Light Brown and Grey
0011	TL0011	TL18-0011-SED	8/9/18	<1	Y	Y	N	Light Brown and Grey
0012	TL0012	TL18-0012-SED	8/6/18	<1	N	Y	Y	Light Brown
0013	TL0013	TL18-0013-SED	8/9/18	3	Y	Y	Y	Grey
0014	TL0014	TL18-0014-SED	8/9/18	2	Y	Y	Y	Grey
0015	TL0015	TL18-0015-SED	8/9/18	2	Y	Y	N	Grey
0016	TL0016	TL18-0016-SED	8/9/18	<1	Y	Y	N	Grey
0017	TL0017	TL18-0017-SED	8/6/18	>10	Y	N	N	Brown
0018	TL0018	TL18-0018-SED	8/9/18	NR	Y	N	Y	Dark Brown
0019	TL0019	TL18-0019-SED	8/6/18	6	N	Y	N	Dark Brown
0020	TL0020	TL18-0020-SED	8/6/18	<1	Y	N	N	Brown

Table A.2.18 Sediment samples observations (offshore and northern sites)

Site Code	Station Location	Sample ID	Collection Date	Post-Mining Sediment Thickness (cm)	Observed Slime Clays (Y/N)	Observed Stamp Sands (Y/N)	Observed Macrophytes (Y/N)	Color
9901	TL9901	TL18-9901-SED	8/7/18	1-3	Y	Y	N	Brown
9902	TL9905	TL18-9902-SED	8/6/18	5	Y	N	N	Brown
9903	TL9906	TL18-9903-SED	8/6/18	5-6	Y	Y	N	Brown
9904	TL9903	TL18-9904-SED	8/7/18	6-7	Y	N	N	Brown
9905	TL9909	TL18-9905-SED	8/7/18	3-5	Y	N	N	Brown
9906	TL9907	TL18-9906-SED	8/6/18	5-6	Y	N	N	Brown
9907	TL9910	TL18-9907-SED	8/7/18	4-6	Y	N	N	Brown
9908	TL9912	TL18-9908-SED	8/6/18	4-6	Y	N	N	Brown
9909	TL9914	TL18-9909-SED	8/6/18	5	Y	N	N	Brown
9910	TL9915	TL18-9910-SED	8/6/18	5	Y	N	Y	Brown
1801	TL1801	TL18-1801-SED	8/15/18	>10	N	N	N	Brown
1802	TL1802	TL18-1802-SED	8/15/18	>10	N	N	N	Brown
1803	TL1803	TL18-1803-SED	8/15/18	>10	N	N	N	Brown
1804	TL1804	TL18-1804-SED	8/15/18	NR	N	Y	N	Red-Brown
1805	TL1805	TL18-1805-SED	8/15/18	>10	N	N	N	Brown
1806	TL1806	TL18-1806-SED	8/14/18	NR	N	Y	N	Brown and Red
1807	TL1807	TL18-1807-SED	8/14/18	NR	N	Y	N	Brown and Red
1808	TL1808	TL18-1808-SED	8/14/18	8-10	N	N	N	Brown
1809	TL1809	TL18-1809-SED	8/14/18	6	N	N	N	Brown
1810	TL1810	TL18-1810-SED	8/14/18	7-9	N	N	N	Brown

Offshore sites

Northern sites

A.3 Diversity Metrics

Table A.3.1 Computed Shannon Indices (H')

Site Code	H'	Site Code	H'	Site Code	H'	Site Code	H'
9901	0.90	0001	1.06	0011	1.08	1801	0.45
9902	0.23	0002	0.41	0012	0.90	1802	0.35
9903	1.00	0003	0.24	0013	0.87	1803	0.91
9904	0.44	0004	1.26	0014	1.13	1804	0.57
9905	0.70	0005	0.96	0015	0.61	1805	0.93
9906	0.41	0006	1.20	0016	0.63	1806	1.55
9907	0.83	0007	0.73	0017	1.04	1807	1.10
9908	1.16	0008	0.71	0018	0.69	1808	0.42
9909	0.85	0009	0.78	0019	0.46	1809	0.71
9910	0.69	0010	0.56	0020	1.15	1810	1.11

Table A.3.2 Computed Hilsenhoff Family Biotic Indices (HBI_f)

Site Code	HBI_f	Site Code	HBI_f	Site Code	HBI_f	Site Code	HBI_f
9901	3.75	0001	7.00	0011	8.00	1801	0.53
9902	7.00	0002	0.60	0012	7.06	1802	0.35
9903	2.93	0003	7.89	0013	7.94	1803	4.32
9904	7.18	0004	5.38	0014	7.00	1804	1.18
9905	5.88	0005	5.61	0015	3.63	1805	2.12
9906	6.17	0006	4.64	0016	8.00	1806	5.80
9907	6.40	0007	5.60	0017	4.00	1807	4.26
9908	4.82	0008	2.14	0018	4.00	1808	0.62
9909	6.89	0009	5.38	0019	6.19	1809	1.83
9910	3.50	0010	8.00	0020	6.00	1810	6.56

A.4 Geospatial Datasets

Table A.4.1 Land cover in the Torch Lake drainage basin

Classification	Category	Area		
		(km ²)	(ac)	(%)
Open Water	Waterbody	10.6	2,614	5%
Woody Wetlands	Wetlands	34.2	8,453	18%
Emergent Herbaceous Wetlands		3.4	849	
Deciduous Forest	Forest	40.3	9,970	53%
Evergreen Forest		6.7	1,643	
Mixed Forest		62.1	15,348	
Shrub/Scrub	Shrubland/Herbaceous	2.7	673	3%
Herbaceous		4.4	1,079	
Hay/Pasture	Cultivated	16.8	4,155	8%
Cultivated Crops		0.2	50	
Developed, Open Space	Developed	8	1,979	10%
Developed, Low Intensity		6.4	1,584	
Developed, Medium Intensity		3.8	941	
Developed, High Intensity		1.4	348	
Barren Land	Barren	4.3	1,053	2%

(NLCD, 2016)

Table A.4.2 Calculated basin slope (percent rise, %)

Site Code	Slope (%)	Site Code	Slope (%)	Site Code	Slope (%)	Site Code	Slope (%)
9901	22.9	0001	34.6	0011	10	1801	12.1
9902	28.7	0002	11	0012	20.9	1802	12.4
9903	29.7	0003	21.2	0013	NA	1803	4.2
9904	31.2	0004	11.2	0014	5.6	1804	10.5
9905	5.6	0005	NA	0015	5.7	1805	3.7
9906	14.4	0006	20.5	0016	32.5	1806	NA
9907	14.3	0007	NA	0017	6	1807	NA
9908	10.7	0008	5.4	0018	14.2	1808	0.2
9909	9	0009	1.5	0019	6.5	1809	0.2
9910	6.9	0010	7.1	0020	27.9	1810	0.9

Reference: bathymetry data layer (2004); NA – Not Available

A.5 Additional Figures

Figure A.5.1 Particle size distribution curves for site 0001

Figure A.5.2 Particle size distribution curves for site 0002

Figure A.5.3 Particle size distribution curves for site 0003

Figure A.5.4 Particle size distribution curves for site 0004

Figure A.5.5 Particle size distribution curves for site 0005

Figure A.5.6 Particle size distribution curves for site 0006

Figure A.5.7 Particle size distribution curves for site 0007

Figure A.5.8 Particle size distribution curves for site 0008

Figure A.5.9 Particle size distribution curves for site 0009

Figure A.5.10 Particle size distribution curves for site 0010

Figure A.5.11 Particle size distribution curves for site 0011

Figure A.5.12 Particle size distribution curves for site 0012

Figure A.5.13 Particle size distribution curves for site 0013

Figure A.5.14 Particle size distribution curves for site 0014

Figure A.5.15 Particle size distribution curves for site 0015

Figure A.5.16 Particle size distribution curves for site 0016

Figure A.5.17 Particle size distribution curves for site 0017

Figure A.5.18 Particle size distribution curves for site 0018

Figure A.5.19 Particle size distribution curves for site 0019

Figure A.5.20 Particle size distribution curves for site 0020

Figure A.5.21 Particle size distribution curves for site 9901

Figure A.5.22 Particle size distribution curves for site 9902

Figure A.5.23 Particle size distribution curves for site 9903

Figure A.5.24 Particle size distribution curves for site 9904

Figure A.5.25 Particle size distribution curves for site 9905

Figure A.5.26 Particle size distribution curves for site 9906

Figure A.5.27 Particle size distribution curves for site 9907

Figure A.5.28 Particle size distribution curves for site 9908

Figure A.5.29 Particle size distribution curves for site 9909

Figure A.5.30 Particle size distribution curves for site 9910

Figure A.5.31 Particle size distribution curves for site 1801

Figure A.5.32 Particle size distribution curves for site 1802

Figure A.5.33 Particle size distribution curves for site 1803

Figure A.5.34 Particle size distribution curves for site 1804

Figure A.5.35 Particle size distribution curves for site 1805

Figure A.5.36 Particle size distribution curves for site 1806

Figure A.5.37 Particle size distribution curves for site 1807

Figure A.5.38 Particle size distribution curves for site 1808

Figure A.5.39 Particle size distribution curves for site 1809

Figure A.5.40 Particle size distribution curves for site 1810

A.6 Baseline Survey Datasets (1999/2000)

Table A.6.1 Sediment and benthic community sample location coordinates Phase I, Baseline Study (USEPA, 2001)

Site Code	Station Location	Sample ID	Collection Date	Sample Method	Latitude	Longitude	Water Depth (m)	Water Depth (ft)
9901	TL9901	TL99PON01-B,C	07/29/99	grab	47.182215	-88.409653	23.8	78
9902	TL9905	TL99PON02-B,C	07/29/99	grab	47.174100	-88.413567	29.3	96
9903	TL9906	TL99PON03-B,C	07/29/99	grab	47.171683	-88.420119	31.4	103
9904	TL9903	TL99PON04-B,C	07/29/99	grab	47.175600	-88.399085	22.6	74
9905	TL9909	TL99PON05-B,C	07/29/99	grab	47.168300	-88.410450	31.4	103
9906	TL9907	TL99PON06-B,C	07/29/99	grab	47.169517	-88.427483	21.0	69
9907	TL9910	TL99PON07-B,C	07/29/99	grab	47.159950	-88.414717	13.7	45
9908	TL9912	TL99PON08-B,C	07/29/99	grab	47.151767	-88.448450	19.2	63
9909	TL9914	TL99PON09-B,C	07/29/99	grab	47.144900	-88.453617	17.1	56
9910	TL9915	TL99PON10-B,C	07/29/99	grab	47.137833	-88.455550	10.4	34

Reference Table A.1, Baseline Study Report (USEPA, 2001)

Table A.6.2 Sediment and benthic community sample location coordinates Phase II, Baseline Study (USEPA, 2001)

Site Code	Station Location	Sample ID	Collection Date	Sample Method	Latitude	Longitude	Water Depth (m)	Water Depth (ft)
0001	TL0001	TL18PON00-01	07/17/00	grab	47.184041	-88.404813	2.9	9.6
0002	TL0002	TL18PON00-02	07/17/00	grab	47.182151	-88.403600	15.2	50.0
0003	TL0003	TL18PON00-03	07/18/00	grab	47.178327	-88.415531	7.2	23.5
0004	TL0004	TL18PON00-04	07/18/00	grab	47.152582	-88.452617	1.1	3.6
0005	TL0005	TL18PON00-05	07/18/00	grab	47.180740	-88.393426	0.9	3.0
0006	TL0006	TL18PON00-06	07/18/00	grab	47.174138	-88.396182	2.3	7.6
0007	TL0007	TL18PON00-07	07/19/00	grab	47.167072	-88.400560	4.8	15.6
0008	TL0008	TL18PON00-08	07/18/00	grab	47.173256	-88.424985	2.2	7.2
0009	TL0009	TL18PON00-09	07/18/00	grab	47.160772	-88.404948	1.8	6.0
0010	TL0010	TL18PON00-10	07/18/00	grab	47.157402	-88.409180	2.3	7.6
0011	TL0011	TL18PON00-11	07/17/00	grab	47.155796	-88.419919	2.4	7.8
0012	TL0012	TL18PON00-12	07/18/00	grab	47.156835	-88.425605	2.7	8.8
0013	TL0013	TL18PON00-13	07/19/00	grab	47.149887	-88.442835	1.5	5.0
0014	TL0014	TL18PON00-14	07/19/00	grab	47.143249	-88.447856	1.8	6.0
0015	TL0015	TL18PON00-15	07/19/00	grab	47.133906	-88.454599	1.5	4.9
0016	TL0016	TL18PON00-16	07/17/00	grab	47.138588	-88.460933	3.3	10.8
0017	TL0017	TL18PON00-17	07/17/00	grab	47.156353	-88.444071	6.4	20.8
0018	TL0018	TL18PON00-18	07/18/00	grab	47.158995	-88.434601	1.2	4.0
0019	TL0019	TL18PON00-19	07/17/00	grab	47.145157	-88.457138	7.6	24.9
0020	TL0020	TL18PON00-20	07/18/00	grab	47.170484	-88.429796	4.8	15.6

Reference Table A.1, Baseline Study Report (USEPA, 2001)

Table A.6.4 Surface/Grab Sediment Sample Benthic Community Results, Baseline Study (nearshore sites, 0011-0020)

Phylum	Class	Order	Family	Subfamily	Genus	0011	0012	0013	0014	0015	0016	0017	0018	0019	0020															
Annelida	Oligochaeta	NSA	Polychaeta		NSA	98			5		1		1	12	1															
		NSA	NSA		NSA																									
Nematoda	Crustacea	NSA	NSA		NSA																									
Arthropoda		Amphipoda	Daphniade	NSA	Gammarus	NSA																								
						NSA	1					5			1															
						NSA																								
						Polyphemus	NSA																							
						Harpacticoida	NSA																							
						NSA	1																							
						NSA																								
						NSA	3																							
						Ostrococha	NSA																							
						Collembola	Seira																							
Insecta	Diptera	Ceratopogonidae	Bezzia	Probezzia	1																									
					NSA																									
					Chaoboridae																									
					Chironomidae																									
					Chironomina	Chironomid larval exuvias																								
						Chironomus								1																
						Cryptochironomus																								
						Dicrotendipes													1											
						Endochironomus																								
						Harniaschia																								
Pagastrella																														
Parachironomus											1				1															
Paralauterborniella																														
Polypedium													1																	
Damesinae	Orthocladiinae	Protanypus	Eukiefferella	Hydrobaenus	Psectrocladius	Procladius	Thienemannimyia gr.	Cladotanytarsus	Cloeor.	Phycocentropus	Oxyethira	NSA	NSA	NSA	NSA															
																1														
Ephemeroptera	Trichoptera	Baetidae	Dipseudopsidae	Hydroptilidae	NSA	Hydrobiidae	Physidae	Valatiidae	NSA	NSA	NSA	NSA	NSA	NSA	NSA															
																4														
Arachnida	Gastropoda	Hydracarina	NSA	Hydrobiidae	Physidae	Valatiidae	NSA	NSA	NSA	NSA	NSA	NSA	NSA	NSA	NSA															
																33														
Mollusca	Gastropoda	Hydracarina	NSA	Hydrobiidae	Physidae	Valatiidae	NSA	NSA	NSA	NSA	NSA	NSA	NSA	NSA	NSA															
																48														
Mollusca	Gastropoda	Hydracarina	NSA	Hydrobiidae	Physidae	Valatiidae	NSA	NSA	NSA	NSA	NSA	NSA	NSA	NSA	NSA															
																48														
Mollusca	Gastropoda	Hydracarina	NSA	Hydrobiidae	Physidae	Valatiidae	NSA	NSA	NSA	NSA	NSA	NSA	NSA	NSA	NSA															
																48														