# St. Louis River Watershed Stressor Identification Report

## A study of local stressors causing degraded fish and aquatic macroinvertebrate communities in the St. Louis River Watershed

Minnesota Pollution Control Agency

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Clockwise from Top Left: Upper Elbow Creek near Leonidas, Minnesota; Tributary to Kingsbury Creek in Duluth, Minnesota during rainfall event; St. Louis River Main Stem near Floodwood, Minnesota; Manganika Lake Outlet near Virginia, Minnesota

# Table of Contents

# Executive Summary

This report summarizes the principal causes, or “stressors,” contributing to impaired fish and aquatic macroinvertebrate communities in 23 impaired stream reaches within the St. Louis River drainage. Ultimately, the results of this report will be used to guide several processes, including; Total Maximum Daily Load (TMDL) development; defining the need for research and development of new water quality standards; and prioritizing additional monitoring, restoration, and protection strategies. Numerous candidate causes for impairment were evaluated for the impaired streams covered in this document. The stressor ID identified eight primary stressors to the biota in the St. Louis River Watershed:

1. **Elevated Total Suspended Solids (TSS) Concentrations.** Elevated TSS concentrations were identified as a probable stressor in 4 of 24, or 17%, of the impaired streams evaluated. Sources of TSS varied by watershed, but excessive streambank erosion, urban runoff, and algae blooms were recognized as major contributors.
2. **Low Dissolved Oxygen.** Low Dissolved Oxygen (DO) concentrations were identified as a probable stressor in 15 of 24, or 63% of the impaired streams evaluated. The low DO conditions observed in a number of these streams were linked to wetland influences and were not driven by anthropogenic disturbances. In other cases, insufficient DO concentrations were the result of stream eutrophication, stream geomorphology, and altered hydrology.
3. **High DO Flux.** High DO flux was identified as a probable stressor in 3 of 24, or 13%, of the impaired streams evaluated. Hydrologically connected mining features (pits) and/or hypereutrophic lakes/reservoirs were present in all cases where this stressor was identified as a cause of impairment.
4. **Elevated Water Temperatures.** Elevated water temperature was identified as a probable stressor in 4 of 24, or 17%, of the impaired streams evaluated. Sources contributing to water temperature increases included urbanized land-use, beaver impoundments, altered hydrology, lack of riparian shading, and changes in geomorphology.
5. **Poor Physical Habitat Conditions.** Poor physical habitat conditions were identified as a probable stressor in 11 of 24, or 46%, of the impaired streams evaluated. Habitat issues cited include excess fine substrate, lack of riffle and glide features, and ditching.
6. **Altered hydrology.** Altered hydrology was identified as a probable or *potential* stressor in 19 of 24, or 79%, of the streams evaluated. This is a difficult stressor to diagnose with confidence, but many land-use features in the St. Louis River Watershed (urban, mining, industrial, dams) have undoubtedly altered the natural flow regime of many rivers and streams.
7. **Nitrate toxicity.** Nitrate toxicity was identified as a probable cause of impairment in 1 of 24, or 4%, of the streams evaluated. It was also listed as a *potential* cause of impairment in two additional streams. The elevated nitrate concentrations were found to be linked to the discharge of treated wastewater into these streams during low flow conditions.
8. **Elevated Specific Conductivity / Sulfate / Chloride.** These three parameters are interrelated, but were evaluated separately as potential stressors. Elevated conductivity was cited as a *potential* stressor in 8 of 24 (33%) of the streams evaluated. Sulfate (6 of 24; 25%) and chloride (2 of 24; 8%) were also discussed as potential or probable stressors in several streams. Mining, municipal, and

urban land-uses were linked to elevated specific conductivity levels, as well as sulfate/chloride concentrations that are many orders of magnitude higher than natural background conditions. Additional research and state water quality standards for conductivity and sulfate are recommended to improve confidence in stressor diagnosis and TMDL development.

# 1.0 Report Purpose, Process, and Overview

The Minnesota Pollution Control Agency (MPCA), in response to the Clean Water Legacy Act, has developed a strategy for improving water quality of the state's streams, rivers, wetlands, and lakes in Minnesota's 81 Major Watersheds, known as the Watershed Restoration and Protection Strategy (WRAPS). A WRAPS is comprised of several types of assessments. The MPCA conducted the first assessment, known as the Intensive Watershed Monitoring Assessment (IWM), during the summers of 2009 and 2010. The IWM assessed the aquatic biology and water chemistry of the St. Louis River Watershed (SLRW) streams and rivers. The second assessment, known as the Stressor Identification (SID) Assessment, builds on the results of the IWM. The MPCA conducted the SID data collection during follow-up monitoring that spanned the years 2011 – 2014. This document reports on the second step of a multi-part WRAPS for the SLRW, a major drainage to Lake Superior that encompasses a large land area of Northeastern Minnesota.

It is important to recognize that this report is part of a series, and thus not a stand-alone document. Information pertinent to understanding this report can be found in the SLRW Monitoring and Assessment (M&A) Report. That document should be read together with this SID Report and can be found from a link on the MPCA's MRW webpage: http://www.pca.state.mn.us/index.php/view-document.html?gid=19270.

### Organization framework of stressor identification

The SID process is used in this report to weigh evidence for or against various candidate causes of biological impairment (Cormier et al. 2000). The SID process is prompted by biological assessment data indicating that a biological impairment has occurred. Through a review of available data, stressor scenarios are developed that may accurately characterize the impairment, the cause, and the sources/pathways of the various stressors (Figure 1a). Confidence in the results often depends on the quality of data available to the SID process. In some cases, additional data collection may be necessary to accurately identify the stressor(s).

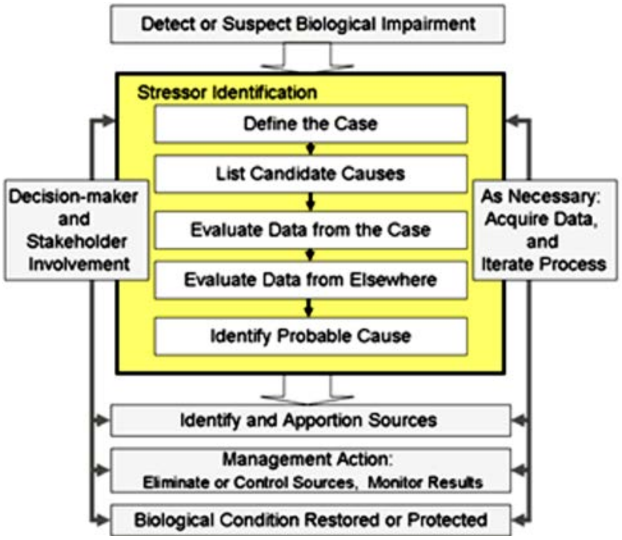


**Figure 1a:** Conceptual diagram of the SID process for identifying the cause(s) of biological Impairment (Cormier et al 2003).

Completion of the SID process does not result in completed TMDL allocations. The product of the SID process is the identification the stressor(s) for which the TMDL load allocation will be developed. For example, the SID process may help investigators identify excess fine sediment as the cause of biological impairment, but a separate effort is then required to determine the TMDL and implementation goals needed to address and correct the impaired condition.

## 1.1 St. Louis River Watershed Zones

The SLRW drains approximately 3,584 square miles of a landscape that has one of the most complex geologic histories of any region in the world. The size and complexity of the SLRW makes it difficult to evaluate potential stressors without further stratifying the drainage area into smaller sections. Although there may be some consistent chemical and physical stressors found throughout the St. Louis River drainage, several stressors are likely acting locally, driven by characteristics and land-uses that are specific to a certain region of the watershed. For the purpose of investigating the causes of biological impairments in this report, the SLRW was stratified into eleven “watershed zones” based on similarities in local geology, land-use, hydrology, and ecological classifications (Figure 2). These watershed zones will serve as an organizational framework for presenting data in this SID Report. Each impaired stream will be discussed and evaluated individually, but the watershed zone groupings will help to place these impaired waters within the overall context of the SLRW.

### Delineation of SLRW Zones

The delineation of SLRW zone boundaries is heavily based on the Minnesota Ecological Classification System (ECS), which was developed through a collaborative effort between DNR and the United States Forest Service (USFS). The primary function of the ECS is to map and describe progressively smaller areas of land with increasingly uniform ecological features. Associations of biotic and environmental factors, such as climate, geology, topography, soils, hydrology, and vegetation are all incorporated into the ECS sections and sub-sections.

Six ECS subsections occur within the SLRW; Glacial Lake Superior Plain, Laurentian Uplands, Mille Lacs Uplands, Nashwauk Uplands, St. Louis River Moraines, Tamarack Lowlands, and Toimi Uplands (Figure 1b). These subsections were used as an initial framework for identifying unique regions of the SLRW that may share similar natural background conditions and tendencies toward specific regional stressors. The subsections were further divided into 11 watershed zones based on known anthropogenic disturbances that are likely to present different stressor scenarios than neighboring watershed zones with similar natural background conditions. Examples of these anthropogenic factors include channelization and ditching of streams, the presence or absence of mining land-uses, urbanization, and industrial or municipal wastewater discharges.

Throughout this report, these 11 watershed zones will be used as a framework for conveying environmental data and conclusions on candidate causes for biological impairment. Additionally, the watershed zone framework serves as an important tool for identifying watershed protection and restoration strategies that can be applied on a much larger scale than an individual impaired stream and its watershed.

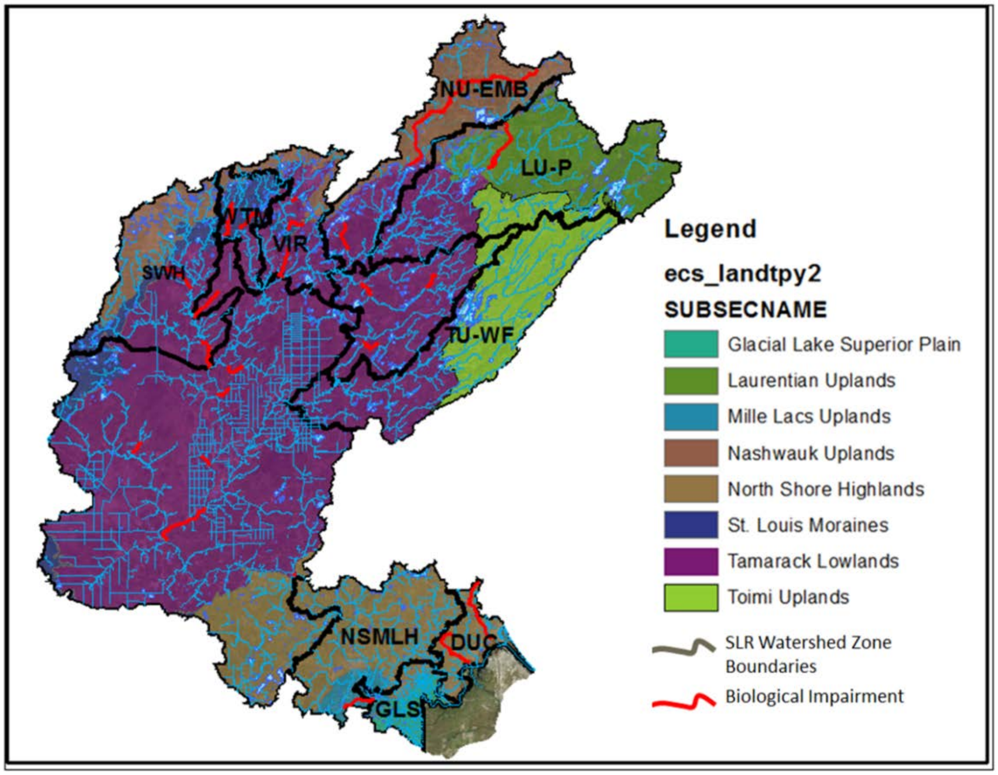


**Figure 1b:** Minnesota DNR's Ecological Classification System (ECS) boundaries (colored) and SLRW zones as delineated by the authors (black outlines).

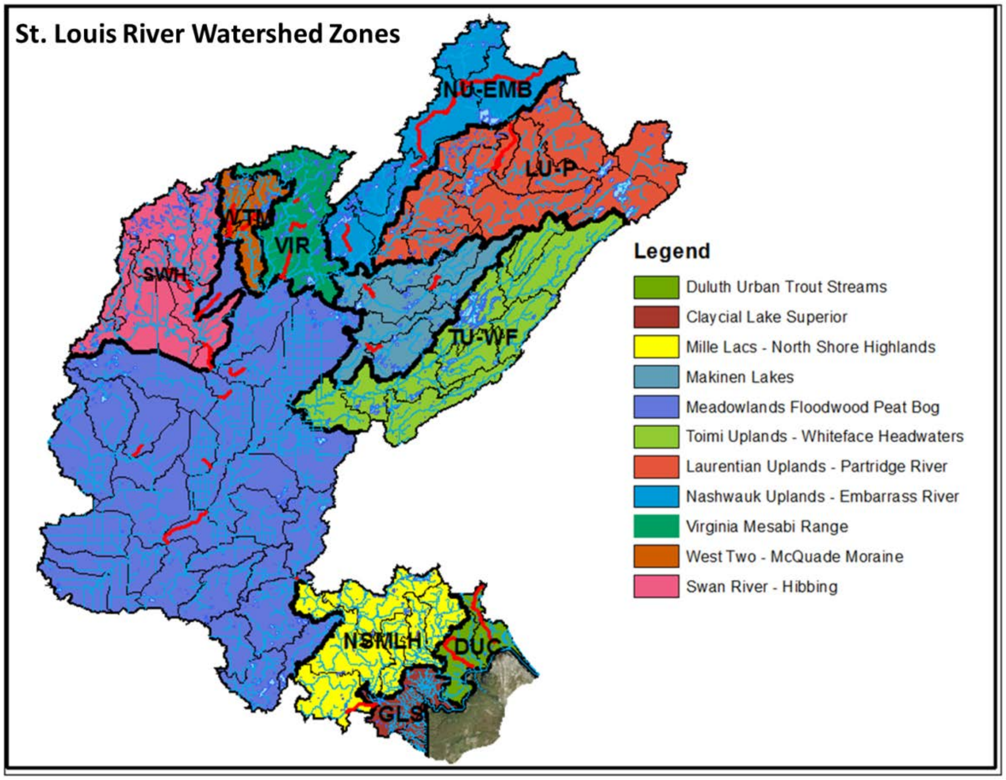


**Figure 2:** The 11 watershed zones of the SLRW. These watershed zones will help to provide context for discussing the impaired streams and organizational structure for this report.

## 1.2 St. Louis River Watershed Characterization

### 1.2.1 Bedrock geologic history of the St. Louis River Basin

The bedrock geology of the St. Louis River basin is ancient and complex. There are four main assemblages representing four very different geologic conditions in Minnesota's past (Figure 3 and 4). The oldest rocks are found in the northernmost regions of the watershed - north of the Mesabi and Cuyuna iron ranges. These rocks date from the Archean period of geologic history and are between 2.5 and 3.0 billion years old. Igneous and metamorphic rock types are dominant and were formed when present-day Minnesota was at the margin of an expanding North American continent.

The second assemblage underlies most of the central portion of the watershed and is from the Animikie Group of Paleoproterozoic rocks – between 1.8 and 2.5 billion years old. Noteworthy among this assemblage are the economically important iron formations, which are metamorphosed oceanic sediments deposited over 2 billion years ago. The conditions for banded iron deposition stopped for unknown reasons about 1.85 billion years ago; thus began the deposition of the second major unit in the Animikie Group. The Virginia formations of shale, siltstone, and greywacke were laid down as oceanic sediments and metamorphosed in a mountain-building event known as the Penokean orogeny. These rocks are mostly covered with recent glacial deposits, but outcrop at the southern base of the Iron Range and more notably along the St. Louis River at Jay Cooke State Park.

The third major geologic assemblage in the St. Louis River basin contains rocks from the Mesoproterozoic and are roughly 1.1 – 0.9 billion years old. These rocks were created in a time during which the North American continent experienced a major rifting event similar to the present-day East-African rift. As the continent began to split apart, volcanic activity increased dramatically and lava poured out in massive flows. The weight of the flows caused the crust to sink and the edges of the rift zone to tilt inwards. The resulting basin collected vast quantities of sediments eroding from the barren landscape – now known as the Hinckley Sandstone and Fold du Lac Formation. The volcanic and metamorphic rocks in this assemblage are erosion-resistant and create the conditions for some of the higher gradient streams in the eastern portions of the SLRW.

The final and youngest geologic assemblage in the basin is the Coleraine Formation of the Cretaceous period (~100 million years old). This group's extent is somewhat minor and only occurs in the western half of the Swan River Watershed. The Coleraine Formation consists mostly of marine sediments deposited when an inland sea invaded Minnesota from the west.

Despite the complex, 3-billion-year geologic history of the basin, almost all of the topography and surficial geology that we see today is the result of only 40,000 years of glacial activity. The ice age and continental glaciers of the Pleistocene Era (10-50 thousand years ago) can be divided into three major periods. The first period of glacial advance came from the northeast and deposited an iron-rich red drift that forms the moraines that extend from Brookston to southwest Lake County and then toward Hibbing. The next period saw the St. Louis Sub Lobe advance from the northwest pushing a lime-rich drift. This lobe formed the moraines that run northeast to southwest, called the Toimi drumlins, and make up much of the Cloquet and Whiteface River Watersheds. In the final period, the Superior Lobe advanced from the northeast out of the Lake Superior basin and deposited a rocky infertile drift along the southern and eastern edge of the SLRW. These deposits essentially dammed the meltwaters of the retreating glaciers and formed an immense, shallow lake called Glacial Lake Upham.

It is the bed of this historic lake that comprises the majority of the central part of the SLRW and is responsible for the extremely low gradients found there. Tributaries of note in this area are the Swan, Whiteface, Floodwood, and Savanna Rivers. Bogs and peatland dominate this region due to the limiting effect of the underlying Virginia Slate on the movement of groundwater. Warm-water conditions are prevalent due to the relative lack of springs and the surface water-fed tributaries (Lindgren and Schuldt 2006).

The southern portion of the watershed - the Mille Lacs – North Shore Highlands and Glacial Lake Superior Watershed zones - contain tributaries fed by springs flowing through the course sediments of the moraines that held back Glacial Lake Upham. The higher gradients of these zones created the conditions for five hydroelectric facilities to be built on the St. Louis River. Major tributaries in this area are Otter Creek, Midway River, and Pine River.

The eastern St. Louis Watershed has a moderate gradient that drains the moraines deposited by all three previously discussed glacial eras. Sediments tend to be very course and productive as a result of lime contained in glacial drift. The Toimi Uplands Watershed zone and eastern portions of the Makinen Lakes and Laurentian Uplands Watershed zones are contained within this area. The northern part of the watershed includes the Laurentian and Nashwauk Uplands, Virginia Mesabi Range, West Two, and Swan River Watershed zones. This area primarily drains infertile red glacial drift. Significant tributaries include East and West Two Rivers, Embarrass and Partridge Rivers.

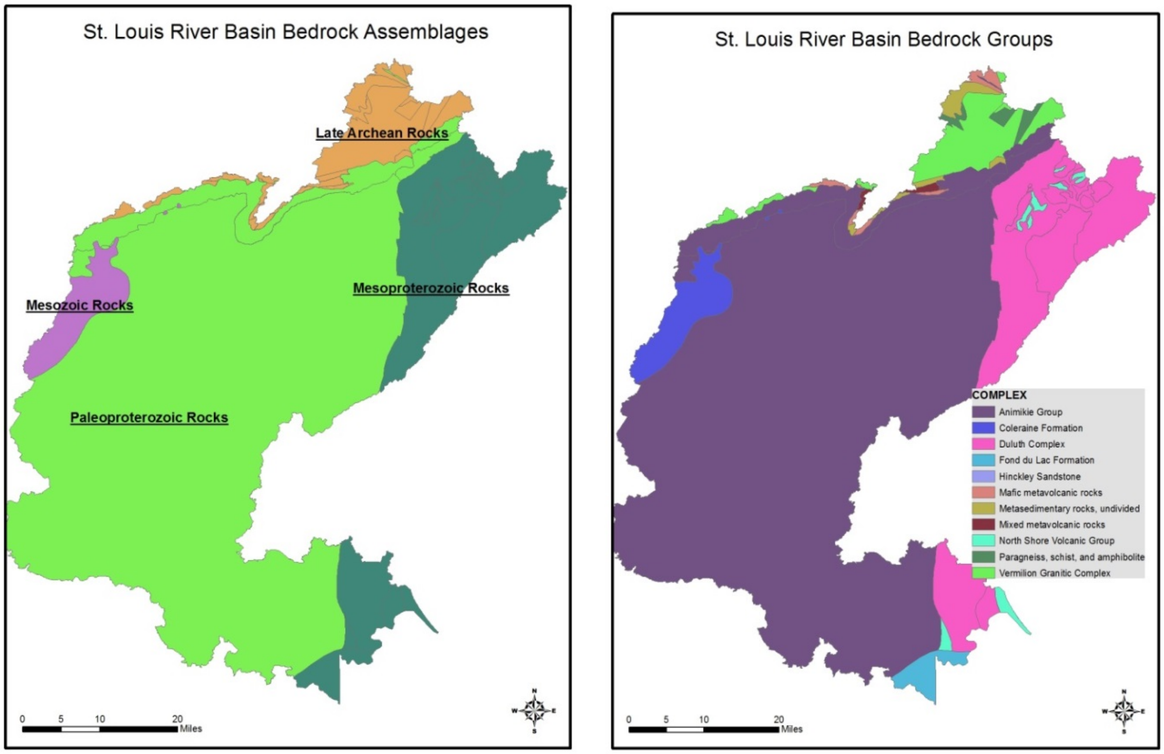


**Figure 3:** Major bedrock assemblages of the SLRW (left) and bedrock groups of the SLRW (right)

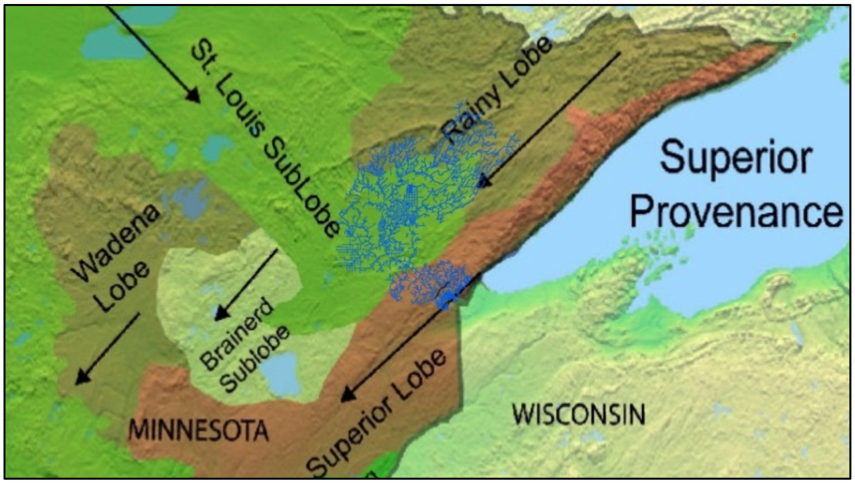


**Figure 4:** Major glacial movements in Northeast Minnesota. www.mngs.umn.edu

### 1.2.2 Geomorphology Overview

The SLRW streams were profiled using LiDAR-derived digital elevation models and the 3D Analyst extension for ArcMap, which shows the change in elevation of a surface along a line. For example, a 3D line drawn on a digital elevation model up the center of a river will show its profile, or a line drawn perpendicular to a valley will show the cross-section for that valley. All impaired streams in the SLRW were profiled, as well as the main stems of major tributaries such as the Whiteface, Floodwood, Savanah, Midway, and Artichoke Rivers (Figure 5). Approximately 980 miles of stream and 516 individual stream reaches were then delineated based on slope and Rosgen channel type and valley type (Rosgen 1994) (Figure 7). For more information on Rosgen stream and valley classifications, see appendix A. Channel types were identified using a combination of aerial and field photos, slope, sinuosity, and stream cross-sections. Valley types were identified using slope, valley cross-sections, and photos.

Figure 6 shows the average slope for each watershed zone. Not surprisingly, the Duluth Urban Trout Stream zone is the steepest with an average slope of almost 1.89% (100 feet per mile of stream). Rivers in the Meadowlands Floodwood Peat Bog are the flattest with an average slope of 0.057% (3 ft/mile). Figures 7 and 8 show the stream and valley type breakdown of each watershed zone. Historic lacustrine valley and alluvial valley types (VIII and X) constitute 60-80% of the SLRW zones. Common within these valley types are low gradient stream types such as Cc and E. These are predominant within the SLRW. However, Miller and Kingsbury Creeks in the Duluth Urban Zone are dominated by altered channels and steeper channels such as Aa+, A and B.

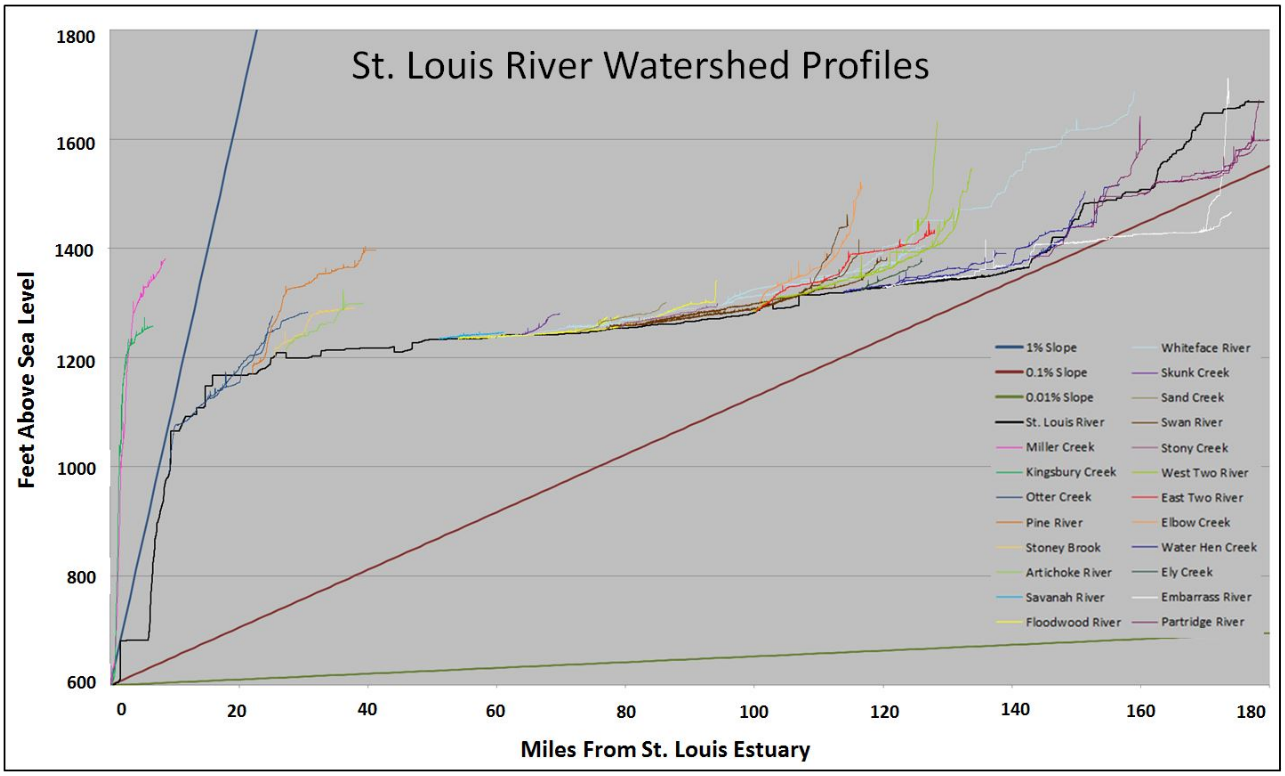


**Figure 5:** Profile of the St. Louis River and all major tributaries, with 1%, 0.1%, and 0.01% slope lines for reference

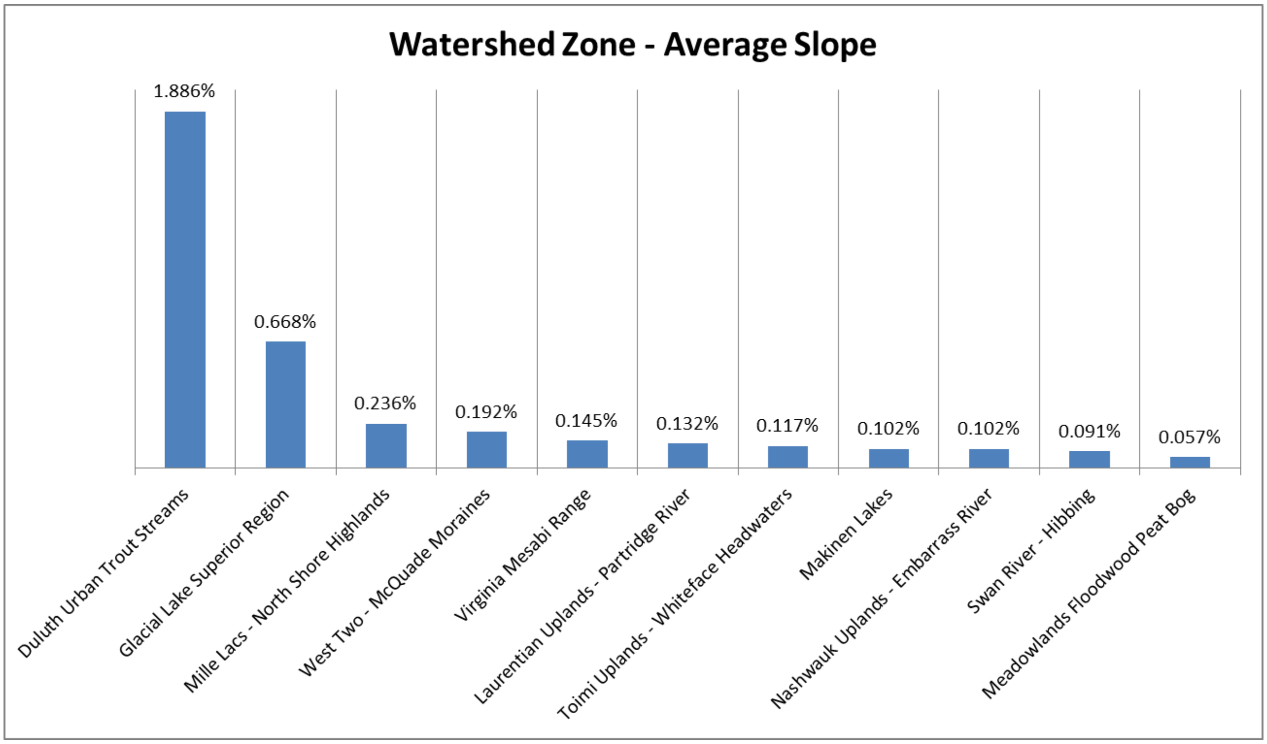
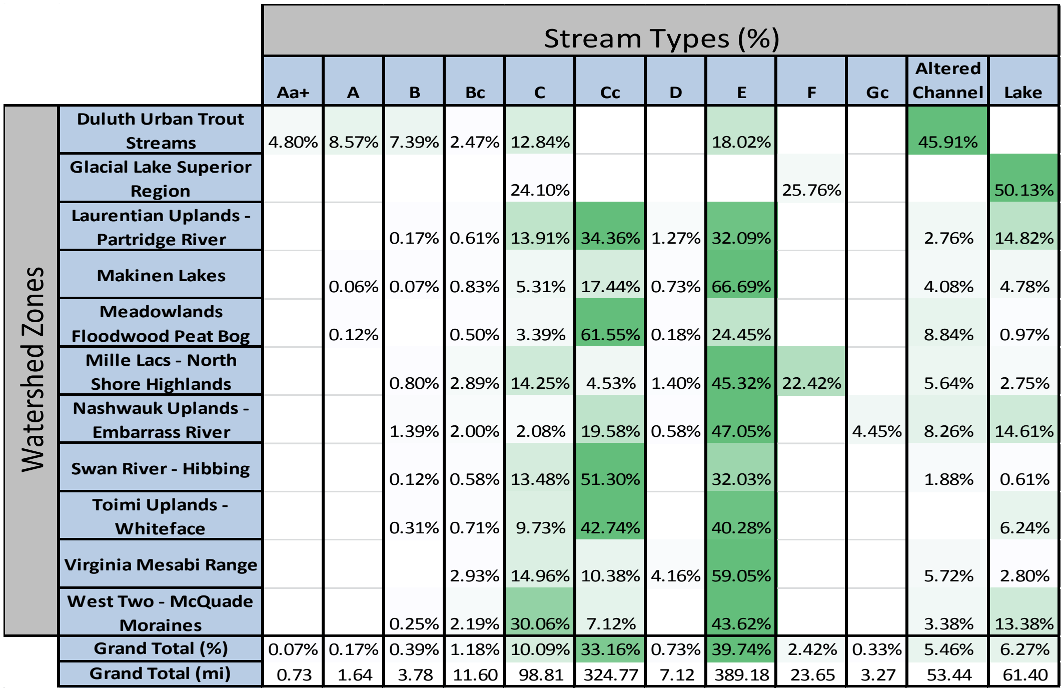
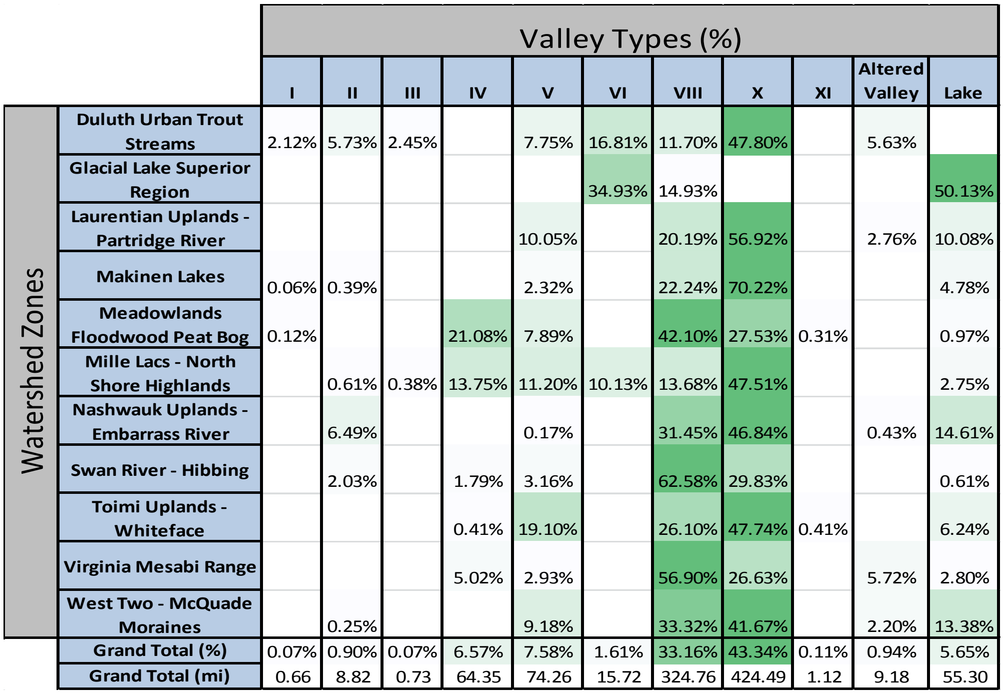


**Figure 6:** Average slope of streams in each SLRW zone



| Watershed Zones | Stream Types (%) | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Aa+ | A | B | Bc | C | Cc | D | E | F | Gc | Altered Channel | Lake |
| Duluth Urban Trout Streams | 4.80% | 8.57% | 7.39% | 2.47% | 12.84% | | | 18.02% | | | 45.91% | |
| Glacial Lake Superior Region | | | | | 24.10% | | | | 25.76% | | | 50.13% |
| Laurentian Uplands - Partridge River | | | 0.17% | 0.61% | 13.91% | 34.36% | 1.27% | 32.09% | | | 2.76% | 14.82% |
| Makinen Lakes | | 0.06% | 0.07% | 0.83% | 5.31% | 17.44% | 0.73% | 66.69% | | | 4.08% | 4.78% |
| Meadowlands Floodwood Peat Bog | | 0.12% | | 0.50% | 3.39% | 61.55% | 0.18% | 24.45% | | | 8.84% | 0.97% |
| Mille Lacs - North Shore Highlands | | | 0.80% | 2.89% | 14.25% | 4.53% | 1.40% | 45.32% | 22.42% | | 5.64% | 2.75% |
| Nashwauk Uplands - Embarrass River | | | 1.39% | 2.00% | 2.08% | 19.58% | 0.58% | 47.05% | | 4.45% | 8.26% | 14.61% |
| Swan River - Hibbing | | | 0.12% | 0.58% | 13.48% | 51.30% | | 32.03% | | | 1.88% | 0.61% |
| Toimi Uplands - Whiteface | | | 0.31% | 0.71% | 9.73% | 42.74% | | 40.28% | | | | 6.24% |
| Virginia Mesabi Range | | | | 2.93% | 14.96% | 10.38% | 4.16% | 59.05% | | | 5.72% | 2.80% |
| West Two - McQuade Moraines | | | 0.25% | 2.19% | 30.06% | 7.12% | | 43.62% | | | 3.38% | 13.38% |
| Grand Total (%) | 0.07% | 0.17% | 0.39% | 1.18% | 10.09% | 33.16% | 0.73% | 39.74% | 2.42% | 0.33% | 5.46% | 6.27% |
| Grand Total (mi) | 0.73 | 1.64 | 3.78 | 11.60 | 98.81 | 324.77 | 7.12 | 389.18 | 23.65 | 3.27 | 53.44 | 61.40 |

**Figure 7:** Percentage of total stream miles by channel type (Rosgen 1994) for each watershed zone of the SLRW



| Watershed Zones | Valley Types (%) | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | I | II | III | IV | V | VI | VIII | X | XI | Altered Valley | Lake |
| Duluth Urban Trout Streams | 2.12% | 5.73% | 2.45% | | 7.75% | 16.81% | 11.70% | 47.80% | | 5.63% | |
| Glacial Lake Superior Region | | | | | | 34.93% | 14.93% | | | | 50.13% |
| Laurentian Uplands - Partridge River | | | | | 10.05% | | 20.19% | 56.92% | | 2.76% | 10.08% |
| Makinen Lakes | 0.06% | 0.39% | | | 2.32% | | 22.24% | 70.22% | | | 4.78% |
| Meadowlands Floodwood Peat Bog | 0.12% | | | 21.08% | 7.89% | | 42.10% | 27.53% | 0.31% | | 0.97% |
| Mille Lacs - North Shore Highlands | | 0.61% | 0.38% | 13.75% | 11.20% | 10.13% | 13.68% | 47.51% | | | 2.75% |
| Nashwauk Uplands - Embarrass River | | 6.49% | | | 0.17% | | 31.45% | 46.84% | | 0.43% | 14.61% |
| Swan River - Hibbing | | 2.03% | | 1.79% | 3.16% | | 62.58% | 29.83% | | | 0.61% |
| Toimi Uplands - Whiteface | | | | 0.41% | 19.10% | | 26.10% | 47.74% | 0.41% | | 6.24% |
| Virginia Mesabi Range | | | | 5.02% | 2.93% | | 56.90% | 26.63% | | 5.72% | 2.80% |
| West Two - McQuade Moraines | | 0.25% | | | 9.18% | | 33.32% | 41.67% | | 2.20% | 13.38% |
| Grand Total (%) | 0.07% | 0.90% | 0.07% | 6.57% | 7.58% | 1.61% | 33.16% | 43.34% | 0.11% | 0.94% | 5.65% |
| Grand Total (mi) | 0.66 | 8.82 | 0.73 | 64.35 | 74.26 | 15.72 | 324.76 | 424.49 | 1.12 | 9.18 | 55.30 |

**Figure 8:** Valley types (Rosgen, 1994) for each watershed zone of the SLRW

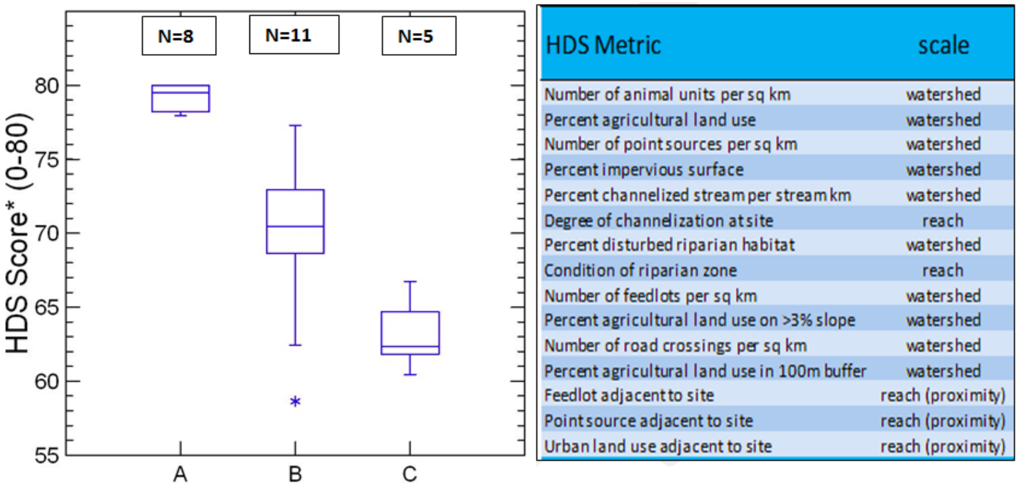
### 1.2.3 Water Quality and Establishment of Reference Conditions

McCollor and Heiskary (1993) compiled long-term water quality data from a set of minimally impacted reference sites to establish water quality goals and targets for Minnesota's seven major ecoregions. A somewhat similar approach was used in this report to characterize water quality conditions of streams and rivers of the SLRW. The primary difference in our methodology was that streams with varying levels of disturbance were selected, as opposed to previous analyses that focused on data from minimally impacted streams (Heiskary 1993; Thingvold et al 1979). Recognizing that good to excellent biological integrity can still be achieved in watersheds that are somewhat impacted, we expanded our investigation to include disturbed sites. Our objective was to identify stream reaches that displayed good to exceptional biological integrity over a general gradient of watershed disturbance, and to summarize water chemistry data for those stations.

Twenty-six monitoring sites (reference sites) with relatively long periods of record were selected from each major zone of the SLRW (Figure 10, Table 1). The vast majority of these reference sites (23 of 26) had accompanying biological data with good to excellent index of biological integrity (IBI) results. Most of the IBI results scored above the upper confidence limit (UCL) for both fish and macroinvertebrate IBI. However, there are several sites that were included based on one biological assessment criterion alone (MIBI or FIBI), and a select few that were chosen based on lack of human disturbance in the watershed as opposed to biological monitoring results.

The reference stations were partitioned into classes (“A”, “B”, and “C”) based on level of watershed disturbance and potential impact from point sources. Initially, a quick review of major disturbances in these watersheds (mining land-use, urban areas, point source discharges) was performed to develop a working list of sites for each grouping. Human disturbance gradient score (HDS), which is an index developed by the MPCA to broadly quantify human disturbances, were then evaluated for each site to determine similarities within each grouping and the level of separation between the three groups (Figure 9). Overall, there appears to be a fair level of agreement between sites within groupings, and a fairly consistent divergence of HDS scores between the three groups. The relatively large span of HDS scores within the “B” grouping was expected, as it was much easier to distinguish relatively pristine sites (A) from impacted sites (C), but harder to quickly quantify impacts at those sites that are moderately disturbed (B).

Summaries of water quality data for each grouping are provided in Table 2. Data are presented for nine water quality variables; specific conductivity, pH, TSS, total ammonia nitrogen, nitrate-nitrite nitrogen, total phosphorous, turbidity, alkalinity, and sulfate. This data set will be used in conjunction with Minnesota water quality standards to evaluate potential stressors to aquatic life.



| HDS Metric | scale |
|---|---|
| Number of animal units per sq km | watershed |
| Percent agricultural land use | watershed |
| Number of point sources per sq km | watershed |
| Percent impervious surface | watershed |
| Percent channelized stream per stream km | watershed |
| Degree of channelization at site | reach |
| Percent disturbed riparian habitat | watershed |
| Condition of riparian zone | reach |
| Number of feedlots per sq km | watershed |
| Percent agricultural land use on >3% slope | watershed |
| Number of road crossings per sq km | watershed |
| Percent agricultural land use in 100m buffer | watershed |
| Feedlot adjacent to site | reach (proximity) |
| Point source adjacent to site | reach (proximity) |
| Urban land use adjacent to site | reach (proximity) |

**Figure 9:** Box plot summary of HDS scores for the three water quality reference station classes (left) and list of metrics used to develop the HDS score (right

**Table 1:** List of stations used to develop water quality summary statistics for streams of the SLRW

| Stream | Group | Watershed Zone | EQUIS ID | Bio Site(s) | FIBI Class | MIBI Class | +/- FIBI UCL* | +/- MIBI UCL * |
|---|---|---|---|---|---|---|---|---|
| **Little Silver Creek** | A | GLS | S000-616 | None | - | - | - | - |
| **Colvin Creek** | A | LU_Partridge | S002-593 | 09LS106 | 7 | 4 | +8 | +9 |
| **SB Partridge R** | A | LU_Partridge | S005-767 | 97LS077 | 6 | 3 | +5 / +6 | +15 / -3 |
| **St. Louis River** | A | LU_Partridge | S000-631 | 97LS080 | 5 | 3 | +17 / +25 | +31 / +7 |
| **Floodwood River** | A | MDWLANDS | S005-761 | 09LS027 | 7 | 4 | - 9 | -5 |
| **Bear Creek** | A | NU_EMB | S006-543 | 09LS098 | 7 | 4 | -7 | +1 |
| **Trib. to West Swan River** | A | Swan-Hibbing | S007-154 | None | - | - | - | - |
| **Bug Creek** | A | Toimi Uplands | S005-766 | 09LS052 | 7 | 4 | +4 | +12 |
| **South Branch Whiteface R** | A | Toimi Uplands | S005-769 | 09LS057 | 7 | 4 | +2 | -10 |
| **South Branch Whiteface R** | A | Toimi Uplands | S005-754 | 97LS019 | 6 | 3 | +20 / +29 | +20 / +21 |
| **Partridge River** | B | LU_Partridge | S004-595 | 09LS105 | 5 | 3 | +28 | +8 |
| **Partridge River** | B | LU_Partridge | S002-596 | None | - | - | - | - |
| **Mud Hen Creek** | B | Makinen Lakes | S005-070 | 09LS090 | 5 | 1 | +19 / +14 | +24 / +15 / +10 |
| **Mud Hen Creek** | B | Makinen Lakes | S007-034 | 09LS091 | 6 | 4 | +21 | +15 |
| **Trib to St. Louis R** | B | MDWLANDS | S005-758 | 09LS011 | 6 | 4 | +20 | -2 |
| **Stoney Brook** | B | MDWLANDS | S004-594 | 09LS016 | 5 | 3 | +22 / +3 | -9 / -3 |
| **Hay Creek** | B | NSMH | S005-942 | 97LS108 | 11 | 8 | +24 | -4 |
| **Trib. to Midway River** | B | NSMH | S005-863 | 97LS112 | 11 | 8 | +5 | 2 |
| **Embarrass River** | B | NU_EMB | S001-680 | 09LS100 | 7 | 4 | -19 * | -5 |
| **West Swan River** | B | Swan-Hibbing | S005-757 | 98NF115 | 5 | 4 | +19 / +16 / + 9 | +3 |
| **Whiteface River** | B | Toimi Uplands | S005-768 | 09LS056 | 5 | 3 | +18 | +17 |
| **Whiteface River** | B | Toimi Uplands | S000-984 | 98LS046 | 5 | 3 | +30 / +20 | +15 / +12 |
| **St. Louis River** | C | MDWLANDS | S000-285 | 97LS027 / 09LS038 | 4 |  | +29 / +31 / +21 | +2 / +35 / +40 /+16 |
| **Embarrass River** | C | NU_EMB | S005-751 | 09LS095 | 5 | 4 | +34 | +23 |
| **West Two River** | C | W Two McQuade | S004-601 | 09LS073 | 5 | 3 | +24 / +18 | -2 |
| **Partridge River** | C | LU_Partridge | S005-752 | 09LS102 / 09LS114 | 5 | 3 | +23 / -15 | +18 |

* Number of points above or below the upper confidence limit for Fish IBI (FIBI) and macroinvertebrate IBI (MIBI)

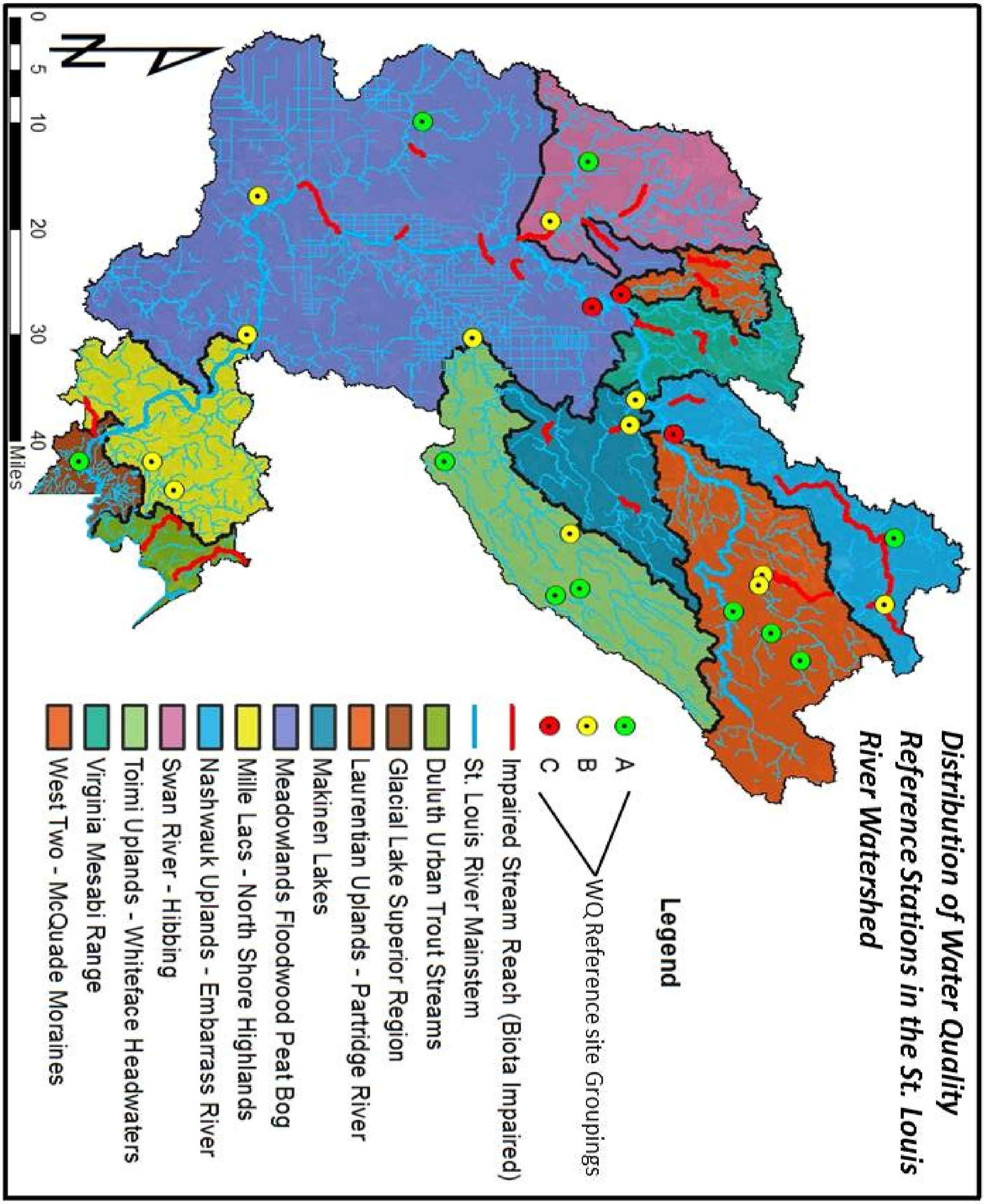


**Figure 10:** Map of reference water quality stations by SLRW zone and disturbance grouping (A, B, C)

**Table 2:** Summary statistics for water quality data associated with reference WQ stations

| **Group A** | | | | MAX | | | MEDIAN | | | MIN |
|---|---|---|---|---|---|---|---|---|---|---|
| Parameter | N | Mean | SD | 100% | 95% | 75% | 50% | 25% | 5% | 0% |
| Conductivity (µS/cm) | 212 | 140 | 83 | 355 | 324 | 171 | 111 | 80 | 54 | 36 |
| pH | 216 | 7.42 | 0.38 | 8.37 | 8.09 | 7.68 | 7.40 | 7.20 | 6.85 | 6.10 |
| TSS (mg/L) | 137 | 4.3 | 2.9 | 20.0 | 8.8 | 5.7 | 3.5 | 2.3 | 1.0 | 0.4 |
| Total Ammonia (mg/L) | 149 | 0.05 | 0.06 | 0.56 | 0.11 | 0.06 | 0.04 | 0.02 | 0.01 | 0.00 |
| NO2 + NO3 (mg/L) | 135 | 0.18 | 0.23 | 0.84 | 0.64 | 0.36 | 0.06 | 0.01 | 0.00 | 0.00 |
| Total Phosphorous (mg/L) | 148 | 0.035 | 0.027 | 0.230 | 0.075 | 0.039 | 0.030 | 0.022 | 0.013 | 0.001 |
| Turbidity (NTU) | 109 | 4.4 | 2.5 | 17.4 | 8.1 | 5.8 | 3.8 | 2.9 | 1.8 | 0.8 |
| Alkalinity (mg/L) | 27 | 44.9 | 25.1 | 145.0 | 69.7 | 48.0 | 40.0 | 29.0 | 16.4 | 13.0 |
| Sulfate (mg/L) | 68 | 3.9 | 3.1 | 15.0 | 11.7 | 5.0 | 3.0 | 2.2 | 1.2 | 1.0 |
| | | | | | | | | | | |
| **Group B** | | | | MAX | | | MEDIAN | | | MIN |
| Parameter | N | Mean | SD | 100% | 95% | 75% | 50% | 25% | 5% | 0% |
| Conductivity (µS/cm) | 124 | 187 | 78 | 380 | 326 | 233 | 180 | 123 | 66 | 25 |
| pH | 126 | 7.53 | 0.53 | 9.00 | 8.28 | 7.90 | 7.54 | 7.21 | 6.50 | 6.23 |
| TSS (mg/L) | 64 | 6.3 | 12.7 | 98.0 | 15.9 | 5.8 | 3.2 | 2.0 | 1.0 | 0.5 |
| Total Ammonia (mg/L) | 30 | 0.062 | 0.093 | 0.390 | 0.281 | 0.050 | 0.024 | 0.016 | 0.010 | 0.009 |
| NO2 + NO3 (mg/L) | 41 | 0.1 | 0.4 | 2.8 | 0.3 | 0.1 | 0.1 | 0.0 | 0.0 | 0.0 |
| Total Phosphorous (mg/L) | 46 | 0.037 | 0.018 | 0.101 | 0.072 | 0.043 | 0.037 | 0.029 | 0.010 | 0.001 |
| Turbidity (NTU) | 73 | 5.8 | 3.8 | 17.7 | 13.2 | 7.0 | 5.0 | 3.0 | 2.0 | 1.3 |
| Alkalinity (mg/L) | 13 | 76.8 | 34.2 | 130.0 | 130.0 | 100.0 | 76.8 | 34.2 | 23.2 | 30.0 |
| Sulfate (mg/L) | 51 | 6.9 | 12.3 | 76.0 | 26.8 | 5.0 | 2.7 | 2.1 | 1.3 | 1.0 |
| **Group C** | | | | MAX | | | MEDIAN | | | MIN |
| Parameter | N | Mean | SD | 100% | 95% | 75% | 50% | 25% | 5% | 0% |
| Conductivity (µS/cm) | 148 | 344 | 223 | 1433 | 556 | 433 | 287 | 222 | 109 | 6 |
| pH | 151 | 7.73 | 0.49 | 9.80 | 8.22 | 8 | 7.75 | 7.55 | 6.9 | 6.28 |
| TSS (mg/L) | 106 | 7.7 | 13.3 | 120 | 17 | 8 | 4 | 2.1 | 0.83 | 0.50 |
| Total Ammonia (mg/L) | 88 | 0.25 | 0.83 | 7.8 | 0.27 | 0.20 | 0.20 | 0.02 | 0.01 | 0.01 |
| NO2 + NO3 (mg/L) | 79 | 0.16 | 0.17 | 0.77 | 0.33 | 0.21 | 0.11 | 0.03 | 0.00 | 0.00 |
| Total Phosphorous (mg/L) | 108 | 0.06 | 0.05 | 0.27 | 0.12 | 0.08 | 0.040 | 0.02 | 0.01 | 0.00 |
| Turbidity (NTU) | 88 | 5.5 | 3.6 | 24.0 | 9.4 | 6.6 | 4.3 | 3.5 | 2.3 | 1.4 |
| Alkalinity (mg/L) | 36 | 78.7 | 33.8 | 180.0 | 115.0 | 92.0 | 82.5 | 53.8 | 28.3 | 25.0 |
| Sulfate (mg/L) | 90 | 72.7 | 77.9 | 434.0 | 190.1 | 83.5 | 41.5 | 27.0 | 15.8 | 2.3 |

### 1.2.4 Overview of Biological Conditions

The SLRW spans one of the most diverse landscapes in Minnesota in terms of geological and hydrological features. Consequently, there are a wide variety of aquatic habitats present within its 3,443 mi$^2$ watershed, which in turn support a diverse population of aquatic organisms. Over 50 species of fish have been documented in the streams and rivers of the SLRW during the MPCA and the Minnesota Department of Natural Resources (DNR) biological monitoring efforts spanning the years 1967 – 2012. The most common gamefish sampled (in streams and rivers) over this period include Smallmouth Bass and Northern Pike from warm to coolwater streams, and Brook and Brown Trout from coldwater streams.

Coldwater trout streams are common in the southern portion of the SLRW. They are particularly prevalent in the steep, rugged drainages that feed St. Louis Bay and in areas of glacial till deposits near Cloquet, Minnesota. Several tributaries to the Swan River, Whiteface River, and Partridge River are also designated trout streams. However, the relative abundance and quality of coldwater streams in these regions is generally much lower. There is a long history of trout stocking in the watershed, dating back to failed attempts to stock pacific salmon into the St. Louis River estuary and cold water tributaries in 1875 (Lindgren and Schuldt 2006). Stocking of Brook Trout, Brown Trout, and Rainbow Trout has occurred throughout the watershed since 1894, but current stocking efforts are limited to a small number of streams, most of which are near the city of Duluth. Today, Brook Trout and Brown Trout are the only salmonids commonly found in streams in the SLRW upstream of barriers to fish migrating from Lake Superior and the St. Louis River estuary.

Minnesota's list of endangered, threatened or special concern species includes several fish species known to have historic ranges within the SLRW. These include Lake Sturgeon, Least Darter, and Pugnose Shiner. There are no recorded observations of these species in the MPCA's biological monitoring records, which include 313 sampling visits to streams within the SLRW. Populations of these fish may have been reduced due to increased presence of the stressors highlighted in Table 3. The Fond du Lac Indian Reservation's Resource Management agency has been stocking and tracking lake sturgeon upstream of the Knife Falls dam for the past six years, and plans to continue this work.

Numerous species of threatened or endangered caddisflies and dragonflies have historic ranges within the SLRW. However, the MPCA macroinvertebrate data does not include species-level identification, making it difficult to know if any of these organisms were collected in the samples.

**Table 3:** Species of special concern or threatened status with historic ranges in the SLRW

| Common Name | Status | Year Listed | Specific Impacts / Stressors |
|---|---|---|---|
| **Lake Sturgeon** | SC | 1984 | Siltation, some agricultural practices, and dam construction reduced habitat availability for the species, resulting in the extirpation or reduction of populations throughout its range (DNR) |
| **Least Darter** | SC | 1996 | Pollution from pesticides, agricultural and urban runoff, eutrophication, and loss of habitat elements such as low velocity waters and aquatic vegetation. Loss of forested habitats around streams, stream reclamation, and the introduction of non-native and predatory fish species (DNR) |
| **Pugnose Shiner** | T | 1996 | Extremely sensitive to increases in turbidity and siltation. Removal of littoral vegetation from lakes and an increase in turbidity in lakes and streams are linked to its demise in other states (DNR) |
| **T = Threatened SC = Special Concern** | | | |

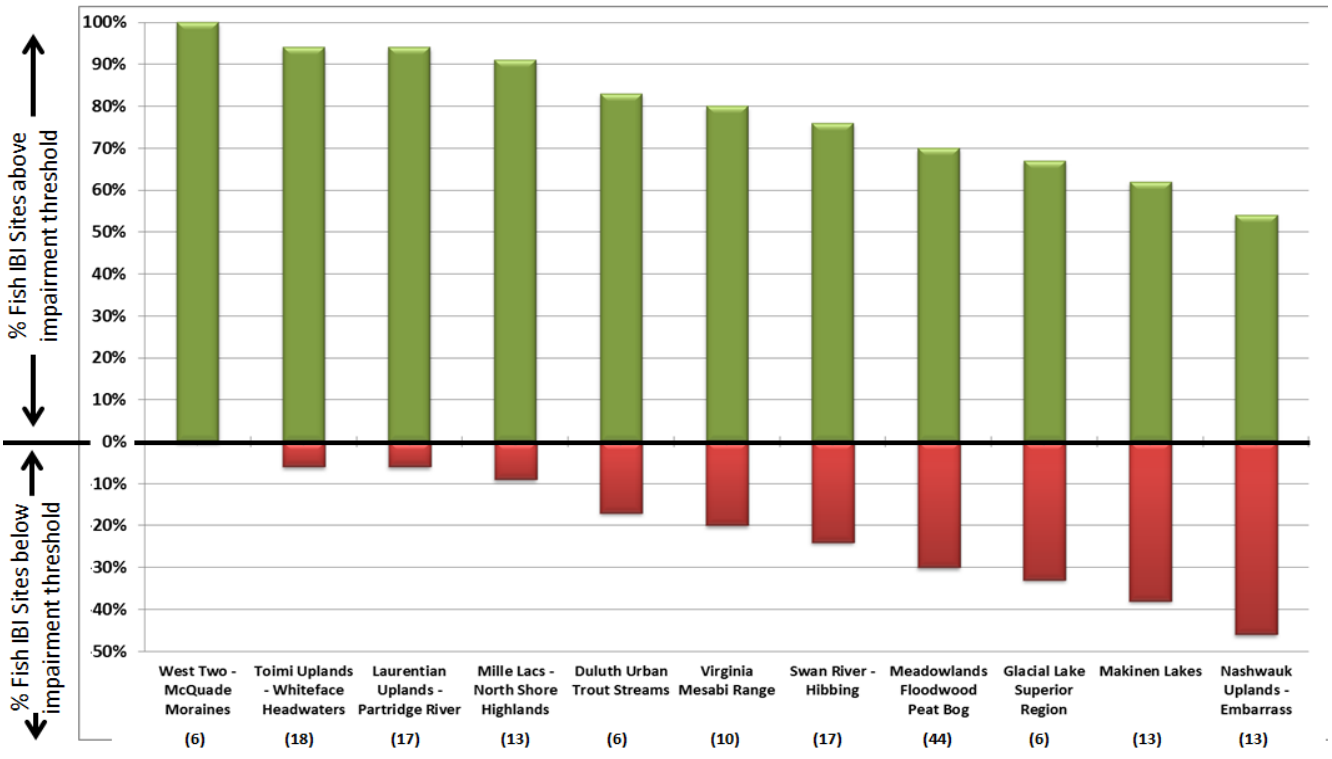
## General Overview of Biological Integrity Results by Watershed Zones

Fish and macroinvertebrate data were analyzed in the context of watershed zones in order to evaluate large-scale spatial trends in biological integrity. Two watershed zones in the SLRW consistently show a high level of biological integrity based on both fish and macroinvertebrate results; the Toimi Uplands-Whiteface Headwaters (TU-WF) and the Laurentian Uplands-Partridge River (LU-P) (Figure 11 and 12). In the TU-WF, 17 of 18 sites (94%) achieved FIBI scores above the impairment threshold. These sites generally surpassed the IBI threshold by a wide margin (average of 27 points above). The MIBI results are equally as impressive in this region of the SLRW. Similar to the FIBI results, 94% of the stations sampled scored above the MIBI impairment threshold, and the average margin above the threshold was 27 points. The IBI scores in the LU-P were slightly lower in comparison to those in the TU-WF, but still exceeded the impairment threshold at a high rate (94% of FIBI scores and 92% of MIBI scores).

The exceptional biological integrity observed within these two watershed zones can be attributed to lower anthropogenic influence, as well as several natural background characteristics that are favorable for supporting healthy streams. Very few of the streams in these two watershed zones have been ditched and straightened, and wetland areas have generally not been altered or drained. Consequently, many of the streams assessed in this region of the SLRW remain in stable physical and hydrological condition and provide exceptional habitat for aquatic life. Relative to other areas of the SLRW, land-cover within the TU-WF and LU-P Watershed zones has changed very little from pre-settlement. Less than 2% of the land-area in both of these watershed zones is categorized as “developed” based on National Land Cover Database (NLCD) data from 2006. Moraine and outwash geological features are common throughout these two watershed zones. The rolling terrain and coarse textured soils found in large portions of these watershed zones facilitates groundwater to surface water exchange, resulting in cooler water temperatures and more stable baseflows for sensitive aquatic life. In addition, the coarser grained material and steeper slopes found in this geologic setting provide a high level of in-stream habitat complexity (riffles, boulders, gravel spawning habitat) that is not found in lower gradient, bog and wetland dominated regions of the SLRW. This region of the watershed should be considered for watershed and stream protection strategies to maintain the high level of biological integrity observed in these streams.

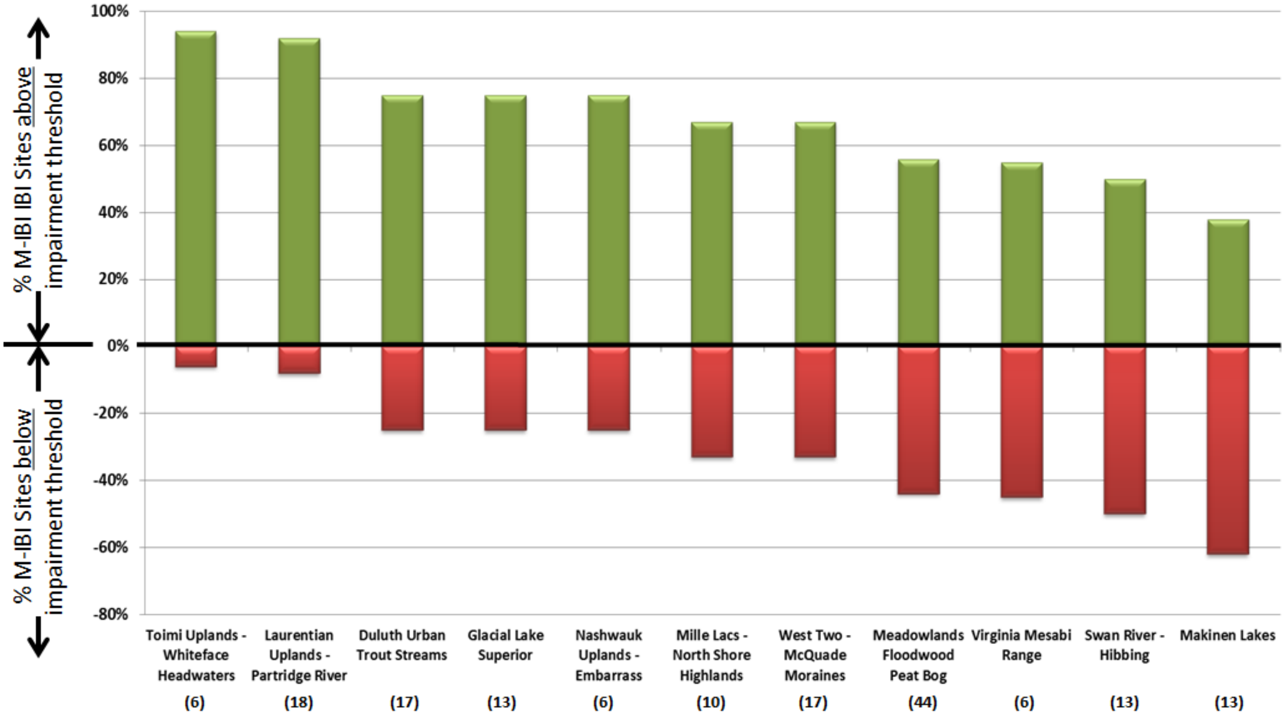
Although biological impairments were observed in nearly every region of the SLRW, several watershed zones were found to produce consistently lower IBI scores for fish and macroinvertebrates (Figure 11 and 12). Within the Makinen Lakes (ML) zone, over 38% (5 of 13) of the fish assessments resulted in IBI scores below the impairment threshold. Although a relatively high percentage of sites scored below the impairment threshold, the fish communities observed within the ML Watershed zone were not severely degraded, with the exception of sites on Paleface River and Paleface Creek. The ML Watershed zone had the highest percentage of MIBI stations score below the impairment threshold (62%). Paleface Creek, Paleface River, and Water Hen Creek were all found to have severely degraded macroinvertebrate assemblages. Two impaired streams in this watershed zone (Paleface Creek and Water Hen Creek) are fed by lakes that are listed as impaired for excess nutrients. The significant amount of wetlands in this watershed zone and low gradient nature of these streams may also be natural background stressors contributing to low IBI scores. These and other stressors in this watershed zone will be evaluated throughout this report.

The highest rate of FIBI scores below the impairment threshold (54% / 7 of 13) was found in the Nashwauk Uplands – Embarrass River (NU-EMB) Watershed zone. Low scoring FIBI sites within this watershed were primarily located on the upper Embarrass River and several of its small tributary streams. Fish results from the upper Embarrass River (the portion upstream of the town of Embarrass) show extremely low fish counts and limited taxa richness. The impaired reach of the Embarrass River flows through expansive wetlands, resulting in extremely tannin stained (tea colored) water that is often low in DO. Two of the impaired streams in this watershed zone, Spring Mine Creek and the Embarrass River, have watersheds that have been heavily altered by resource extraction land-uses such as logging and mining. Both of these streams are discharge points for mine pit dewatering, and water quality sampling results from these streams show elevated specific conductance and sulfate concentrations. These potential stressors will be evaluated in terms of their impact to aquatic life in these watersheds in section 5 of this document.



*Numbers in ( ) indicate the sample size (e.g. fish IBI assessments conducted in each watershed zone)*

**Figure 11:** Percentage of monitoring stations above and below FIBI impairment threshold.



*Numbers in ( ) indicate the sample size (e.g. fish IBI assessments conducted in each watershed zone)*

**Figure 12:** Percentage of monitoring stations above and below MIBI impairment threshold

# 2.0 Biological Impairments in the St. Louis River Watershed

In 2009, the MPCA began an intensive watershed monitoring effort of the SLRW's surface waters. Using the data collected during this effort, aquatic life assessments were completed for 75 stream and river segments (assessment units or "AUIDs") in the spring of 2011. These assessments were carried out in compliance with the federal Clean Water Act, which requires states to monitor and assess waterbodies for various criteria related to aquatic life and recreation. A complete summary of these assessments can be found in the SLRW M&A Report (Anderson 2013). Streams were assessed for a variety of water quality parameters and biological indicators (fish and aquatic macroinvertebrates). This report deals specifically with streams that were identified as impaired using fish and macroinvertebrate data. For background information on the development and implementation of fish and macroinvertebrate IBI standards, refer to the documents listed in Table 4.

Of the 75 AUIDs assessed for aquatic life, 24 (32%) were ultimately listed as "impaired waters" for failing to meet established IBI criteria for fish and/or aquatic macroinvertebrates. The impaired streams are listed in Table 5, and their locations displayed on a map of the watershed in Figure 13. Generally, the biological impairments in the SLRW are located on first and second order headwaters streams, although several impaired segments were identified on larger river systems, including the Swan River, Embarrass River, and a short section of the St. Louis River main stem near Floodwood, Minnesota.

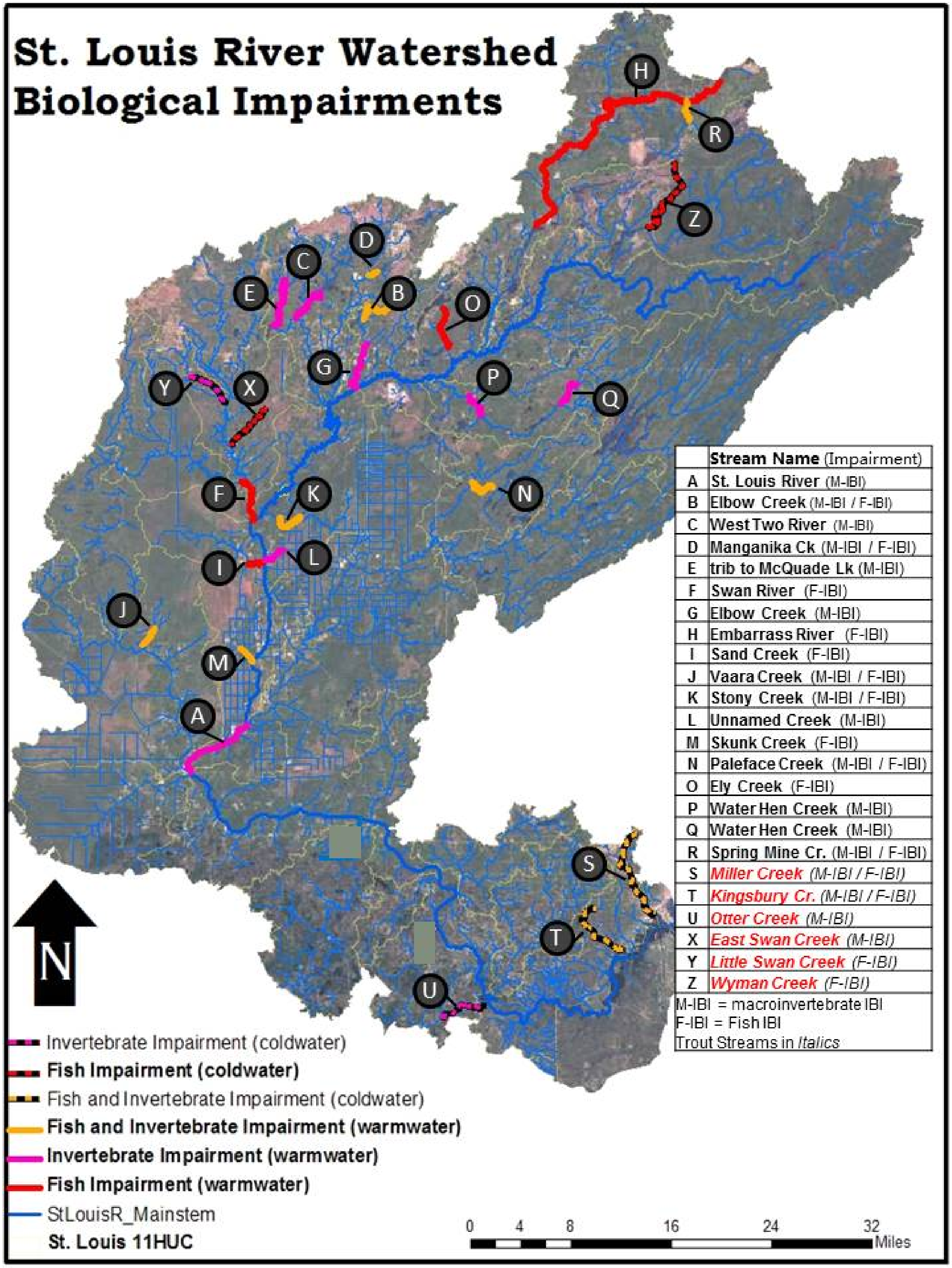
Specific information related to each of these impairments will be presented in Section 4. Fish and MIBI scores, a discussion of biological metric results and symptoms of impairment, and SID data will be presented in detail in that section of this report.

**Table 4:** List of MPCA documents available for background information on IBI development and assessment criteria

| Document Title (Citation) | Internet Link |
|---|---|
| **Development of a fish-based index of biological integrity for Minnesota Rivers and Streams (MPCA 2014)** | http://www.pca.state.mn.us/index.php/view-document.html?gid=21417 |
| **Development of a Macroinvertebrate-based index of biological integrity for Minnesota Rivers and Streams (MPCA 2014)** | http://www.pca.state.mn.us/index.php/view-document.html?gid=21215 |
| **Development of Biological Criteria for Tiered Aquatic Life Uses (MPCA 2014)** | http://www.pca.state.mn.us/index.php/view-document.html?gid=21164 |

| Stream Name | Drainage | AUID | Impairment | Fish Class* | Invert Class* | Watershed Zone |
|---|---|---|---|---|---|---|
| **Kingsbury Creek** | 7.1 | 04010201-626 | F-IBI / M-IBI | 11 | 8 | Duluth Urban Coldwater |
| **Miller Creek** | 8.0 | 04010201-512 | F-IBI / M-IBI | 11 | 8 | Duluth Urban Coldwater |
| **Wyman Creek** | 10.2 | 04010201-942 | F-IBI | 11 | 8 | Laurentian Uplands - Partridge River |
| **Paleface Creek** | 29.5 | 04010201-A24 | F-IBI / M-IBI | 7 | 4 | Makinen Lakes |
| **Water Hen Creek** | 15.9 | 04010201-A35 | M-IBI | 6 | 4 | Makinen Lakes |
| **Water Hen River** | 68.5 | 04010201-A31 | M-IBI | 5 | 4 | Makinen Lakes |
| **Little Swan Creek** | 21.1 | 04010201-891 | F-IBI | 11 | 8 | Meadowlands Floodwood Peat Bog |
| **Sand Creek** | 64.0 | 04010201-607 | F-IBI | 5 | 3, 4 | Meadowlands Floodwood Peat Bog |
| **Skunk Creek** | 15.0 | 04010201-A18 | F-IBI / M-IBI | 6 | 4 | Meadowlands Floodwood Peat Bog |
| **St Louis River** | 1,936.4 | 04010201-508 | M-IBI | 4 | 1 | Meadowlands Floodwood Peat Bog |
| **Stony Creek** | 21.5 | 04010201-963 | F-IBI / M-IBI | 6, 7 | 4 | Meadowlands Floodwood Peat Bog |
| **Vaara Creek** | 26.8 | 04010201-623 | F-IBI / M-IBI | 7 | 4 | Meadowlands Floodwood Peat Bog |
| **Unnamed Trib to St. Louis R.** | 4.8 | 04010201-A17 | M-IBI | 6 | 4 | Meadowlands Floodwood Peat Bog |
| **Otter Creek** | 39.7 | 04010201-629 | M-IBI | 11 | 8 | Mille Lacs - North Shore Highlands |
| **Ely Creek** | 15.5 | 04010201-A26 | F-IBI | 6 | 4 | Nashwauk Uplands - Embarrass River |
| **Embarrass River** | 115.1 | 04010201-579 | F-IBI | 5, 7 | 4 | Nashwauk Uplands - Embarrass River |
| **Spring Mine Creek** | 4.4 | 04010201-A42 | F-IBI / M-IBI | 6 | 4 | Nashwauk Uplands - Embarrass River |
| **East Swan Creek** | 7.1 | 04010201-888 | M-IBI | 11 | 8 | Swan River - Hibbing |
| **Swan River** | 244.3 | 04010201-557 | F-IBI | 5 | 4 | Swan River - Hibbing |
| **Elbow Creek** | 3.2 | 04010201-518 | F-IBI / M-IBI | 6 | 4 | Virginia Mesabi Range |
| **Elbow Creek** | 12.0 | 04010201-570 | M-IBI | 6 | 4 | Virginia Mesabi Range |
| **Manganika Creek** | 5.7 | 04010201-548 | F-IBI / M-IBI | 6 | 4 | Virginia Mesabi Range |
| **Kinney Creek** | 17.5 | 04010201-551 | M-IBI | 6 | 3, 4 | West Two - McQuade Moraine |
| **West Two River** | 33.5 | 04010201-535 | M-IBI | 6 | 4 | West Two - McQuade Moraine |

**Table 5**: List of streams with fish and/or MIBI impairments in the SLRW * see Table 4 for link to class descriptions in IBI Document



| | **Stream Name** (Impairment) |
|---|---|
| A | **St. Louis River** (M-IBI) |
| B | **Elbow Creek** (M-IBI / F-IBI) |
| C | **West Two River** (M-IBI) |
| D | **Manganika Ck** (M-IBI / F-IBI) |
| E | **trib to McQuade Lk** (M-IBI) |
| F | **Swan River** (F-IBI) |
| G | **Elbow Creek** (M-IBI) |
| H | **Embarrass River** (F-IBI) |
| I | **Sand Creek** (F-IBI) |
| J | **Vaara Creek** (M-IBI / F-IBI) |
| K | **Stony Creek** (M-IBI / F-IBI) |
| L | **Unnamed Creek** (M-IBI) |
| M | **Skunk Creek** (F-IBI) |
| N | **Paleface Creek** (M-IBI / F-IBI) |
| O | **Ely Creek** (F-IBI) |
| P | **Water Hen Creek** (M-IBI) |
| Q | **Water Hen Creek** (M-IBI) |
| R | **Spring Mine Cr.** (M-IBI / F-IBI) |
| S | ***Miller Creek*** *(M-IBI / F-IBI)* |
| T | ***Kingsbury Cr.*** *(M-IBI / F-IBI)* |
| U | ***Otter Creek*** *(M-IBI)* |
| X | ***East Swan Creek*** *(M-IBI)* |
| Y | ***Little Swan Creek*** *(F-IBI)* |
| Z | ***Wyman Creek*** *(F-IBI)* |

M-IBI = macroinvertebrate IBI
F-IBI = Fish IBI
Trout Streams in *Italics*

**Figure 13:** Map of biological impairments in the St. Louis River 8-HUC watershed. Coldwater streams in **red.**

# 3.0 Background on Stressors and Applicable Standards

## 3.1 Water Quality Stressors

A broad, high-level review of water quality data is presented in this section with the goal of summarizing current conditions and developing list of candidate stressors related to water quality parameters. Minnesota's water quality standards for water quality parameters are also discussed in this section where applicable. For parameters without associated water quality standards, data from a selection of high quality reference streams were used for comparison.

### 3.1.1 Water Temperature

Fish and macroinvertebrate species are often restricted in their distribution based on the temperature ranges observed within streams, rivers, and lakes. Although adaptations have taken place that allow certain species to live within the colder and warmer extremes of natural waters, very few taxa are able to cope with very high water temperatures. Species that occupy streams with a narrow temperature range are referred to as *stenothermal*, while those that thrive over a wide temperature range are called *eurythermal*. Species common to trout streams in the SLRW, such as Brook Trout and Mottled Sculpin, are considered coldwater stenotherms, because they are unable to survive when water temperatures become elevated.

Water temperature has the most potential to act as a stressor to aquatic life during the cold and warm extremes of the year. The northern latitude of the SLRW renders the biota of the region vulnerable to both of these critical periods. Winter monitoring of water temperatures and below-ice conditions are challenging. Although occasional winter measurements and observations were taken, they were not a major part of this monitoring effort. As a result, most of the focus on water temperature as a stressor will be placed on summer extremes.

**Warmwater & Coolwater Streams of the SLRW**

Seventy-five percent (18 of 24) of the impaired stream reaches in the SLRW are considered warmwater or coolwater streams. These streams have likely never supported Brook Trout or other coldwater species, and are currently managed as non-trout bearing streams. The specific temperature thresholds that separate cold, cool and warmwater stream classes are not defined by rule in Minnesota, and tend to vary by region. Fish and macroinvertebrate species inhabiting these streams are generally able to tolerate wider temperature ranges and higher maximum temperatures. Most warmwater fishes, including esocids (pikes) and cyprinids (minnows) have upper temperature tolerance limits near 30 C.

The highest temperature recorded among all of the study streams was 27.2 C (Stoney Brook, MF-PB Zone), which is still within the suitable range for supporting warmwater fish species. Several impaired streams in the Iron Range district of the watershed show lower maximum temperatures and noticeably narrower ranges between minimum and maximum temperatures. This is likely due to the influence of groundwater and mine pit dewatering to these streams.

Based on the available data for warm and coolwater streams in the SLRW, elevated water temperatures are an unlikely cause of impairment and can be eliminated as a candidate cause.

**Coldwater Streams of the SLRW**

Instantaneous temperature readings from the months of July and August were compiled for the six impaired stream segments on designated trout streams in the SLRW. Continuous temperature loggers were also deployed in these streams, and the data collected during these continuous monitoring periods were also considered in identifying streams with potential impacts related to water temperature. Stream temperatures were found to be in the range of thermal stress for coldwater taxa in all watershed zones of the SLRW. Temperatures considered lethal to Brook Trout were not exceeded in any of the instantaneous measurements, although streams in the DUC Watershed zone (Kingsbury Creek and Miller Creek) had temperatures that approached this threshold.

This screening level assessment of stream temperature data shows that elevated stream temperatures are a candidate cause for impairment in all watershed zones that contain coldwater streams. Therefore, water temperature stressors will be a focus primarily for the impaired coldwater streams featured in this report. These include **Miller Creek** and **Kingsbury Creek** within the city limits of Duluth, **Otter Creek** near the city of Carlton, **East Swan Creek** and **Little Swan Creek** south of Hibbing, and **Wyman Creek** near the city of Hoyt Lakes.

### 3.1.2 Dissolved Oxygen

Dissolved oxygen (DO) refers to the concentration of oxygen gas within the water column. Oxygen diffuses into water from the atmosphere (turbulent flow enhances this diffusion) and from the release of oxygen by aquatic plants during photosynthesis. DO concentrations in streams are driven by several factors. Large-scale factors include climate, topography, and hydrologic pathways. These in turn influence smaller scale factors such as water chemistry and temperature, and biological productivity. As water temperature increases, its capability to hold oxygen is reduced. Low DO can be an issue in streams with slow currents, excessive temperatures, high biological oxygen demand, and/or high groundwater seepage (Hansen, 1975). In most streams and rivers, the critical conditions for stream DO usually occur during the late summer season when water temperatures are at or near the annual high and stream flow volumes and rates are generally lower. DO concentrations change hourly, daily, and seasonally in response to these driving factors.

Human activities can alter many of these driving factors and change the DO concentrations of water resources. Increased nutrient content of surface waters is a common human influence, which results in excess aquatic plant growth. This situation often leads to a decline in daily minimum oxygen concentrations and an increase in the magnitude of daily DO concentration fluctuations due to the decay of the excess organic material, increased usage of oxygen by plants at night, and their greater oxygen production during the daytime. Humans may directly add organic material by municipal or industrial effluents. Other human activities that can change water temperature include vegetation alteration and changes to flow patterns.

Aquatic organisms require oxygen for respiration. Inadequate oxygen levels can alter fish behavior, such as moving to the surface to breathe air, or moving to another location in the stream. These behaviors can put fish at risk of predation, or may hinder their ability to obtain necessary food resources (Kramer 1987). Additionally, low DO levels can significantly affect fish growth rates (Doudoroff and Warren 1965). Fish species differ in their preferred temperature ranges (Dowling and Wiley 1986), so alterations in water temperature (and DO) from the natural condition will alter the composition of fish communities. Low or highly fluctuating concentrations of DO can have detrimental effects on many fish and macroinvertebrate species (Davis 1975; Nebeker et al. 1992). Heiskary et al. (2013) observed several strong negative relationships between fish and macroinvertebrate metrics and higher daily DO fluctuations. Increased water temperature raises the metabolism of organisms, and thus their oxygen needs, while at the same time, the higher-temperature water holds less oxygen. Some aquatic insect species have anatomical features that allow them to access atmospheric air, though many draw their oxygen from the water column. Macroinvertebrate groups (Orders) that are particularly intolerant to low DO levels (with a few exceptions), include Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies).

## Minnesota DO Standards

The class 2B (warmwater) water quality standard for DO in Minnesota is 5 mg/L as a daily minimum, while the class 2A (coldwater) water quality standard for DO in Minnesota is 7 mg/L as a daily minimum. Additional stipulations have been recently added to this standard that require most of the data to be collected during times where sub-optimal DO concentrations typically occur. For more information on this DO standard, refer to the Guidance Manual for Assessing the Quality of Minnesota Surface Waters (MPCA 2009).

## Types of Dissolved Oxygen Data

1. Point Measurements

   Instantaneous (one moment in time) DO data was collected at many locations in the MRW and used as an initial screening for low DO reaches. Because DO concentrations can vary significantly with changes in flow conditions and time of sampling, conclusions using instantaneous measurements need to be made with caution and are not completely representative of the DO regime at a given site.

2. Longitudinal (Synoptic)

   This sampling method involves collecting simultaneous (or nearly so) readings of DO from several locations along a significant length of the stream path. It is best to perform this sampling in the early morning in order to capture the daily minimum DO readings.

3. Diurnal (Continuous)

   Short interval, long time period sampling using deployed YSI™ water quality sondes (a submerged electronic sampling devise) provides a large number of measurements to reveal the magnitude and pattern of diurnal DO flux at a site. This sampling captures the daily minimum DO concentration, and when deployed during the peak summer water temperature period, also allows an assessment of the annual low DO levels in a stream system.

**Candidate Cause Screening: Dissolved Oxygen**

Available DO (point and continuous measurements) data were evaluated for all impaired streams in the SLRW zone to identify areas where low DO may be a candidate cause of biological impairments. This review revealed that low DO concentrations and/or high DO flux are a widespread candidate cause for impairment in the SLRW, with 17 out of 24 (71%) impaired stream reaches showing potentially stressful DO conditions. DO concentrations below state water quality standards were recorded in all watershed zones of the SLRW. The only streams for which DO was eliminated based on monitoring results include **St. Louis River, Sand Creek, Unnamed Trib to St. Louis River, Miller Creek, Otter Creek, and East Swan Creek.**

## 3.1.3 Nutrients and River Eutrophication

Phosphorus (P), an important plant nutrient, is typically in short supply in natural systems, but human presence and activity on the landscape often exports P to waterways, which can impact stream organisms. Nutrient sources can include urban stormwater runoff, agricultural runoff, animal waste, fertilizer, industrial and municipal wastewater facility discharges, and non-compliant septic system effluents. Under anoxic conditions (no DO), P can be released from stream or lake bottom sediments, particularly in the absence of iron and other key elements (Wetzel 2001). P exists in several forms; the soluble form, orthophosphorus, is readily available for plant and algal uptake. While P itself is not toxic to aquatic organisms, it can have detrimental effects via other follow-on phenomena when levels are elevated above natural concentrations. Increased nutrients cause excessive aquatic plant and algal growth, which alters physical habitat, food resources, and oxygen levels in streams. Excess plant growth increases DO during daylight hours and saps oxygen from the water during the nighttime. Additionally, DO is lowered as bacterial decomposition occurs after the abundant plant material dies. Streams dominated with submerged macrophytes experience the largest swings in DO and pH (Wilcox and Nagels 2001). In some cases, oxygen production leads to extremely high levels of oxygen in the water (supersaturation), which can cause gas bubble disease in fish. The wide daily fluctuations in DO caused by excess plant growth are also correlated to degradation of aquatic communities (Heiskary et al. 2013). More information on the effects of P can be found on EPA's CADDIS webpage: http://www.epa.gov/caddis/ssr_nut_int.html

**Minnesota River Nutrient Standards**

Nutrient enrichment (particularly total phosphorous), chlorophyll-a (Chl-a) concentrations, and measures of biological oxygen demand (BOD) are all factors in the DO regime of streams and rivers. The MPCA has developed standards for P designed to protect aquatic life (Heiskary et al. 2013). Total Phosphorus (TP) criteria were developed for three geographic regions (Table 6). The TP standard is a maximum concentration also requiring at least one of three related stressors (Chl-a, DO Flux, $BOD_5$) above its threshold.

Increased sulfate loading may contribute to the production and the availability of P for algal growth. The mechanisms involved are associated with the tendency during decay of organic matter for natural bacteria to convert sulfate to sulfide after oxygen is depleted.

**Table 6:** River eutrophication criteria ranges by River Nutrient Region for Minnesota. The SLRW is placed in the North Region.

| | Nutrient | Stressor | | |
|---|---|---|---|---|
| **Region** | **TP µg/L** | **Chl-a µg/L** | **DO flux mg/L** | **BOD$_5$ mg/L** |
| **North** | **55** | **<10** | **≤4.0** | **≤1.5** |
| **Central** | **100** | **<20** | **≤4.5** | **≤2.0** |
| **South** | **150** | **<40** | **≤5.0** | **<3.5** |

### Types of Phosphorus Data

Water samples were collected from streams and rivers throughout the MRW. The most common data is for TP, though orthophosphorus samples were collected in some cases. Samples are analyzed by a state certified laboratory and the data is stored in a publicly available database: http://cf.pca.state.mn.us/water/watershedweb/wdip/search_more.cfm.

### Candidate Cause Screening: Eutrophication

The TP concentrations exceeding the regional target of 0.055 mg/L are observed in biota-impaired streams within nearly every watershed zone of the SLRW -- with the only exception being Wyman Creek. However, median and 75th percentile TP concentrations for most of these impaired streams are below the regional target. In many cases, the elevated TP results may be limited to high flow events during snowmelt or large summer rain events. Two watershed zones had median TP concentrations above the target for the North nutrient region, Swan River-Hibbing and Virginia Range Streams. Special attention will be given to the role that eutrophication plays in the DO regime of streams in these watershed zones.

## 3.1.4 Altered pH

Acidity is measured on a scale called pH, ranging from 0 to 14, with values of 0 to 6.99 being acidic, 7.0 neutral and above seven being basic. Human effects on pH values can result from agricultural runoff, urbanization, and industrial discharges. Some geology produces naturally high hydrogen ions that can leach into surface water, but it would be rare for this to be the only cause when pH is a stressor. Photosynthesis from unnaturally-abundant plants or algae removes carbon dioxide from the water, causing a rise in pH. Thus, stream eutrophication can be a primary cause of elevated pH in streams.

As pH increases, unionized ammonia (the toxic form of ammonia) increases, and may reach toxic concentrations. Low pH values contribute to elevated specific conductivity of water (more dissolved minerals). High or low pH effects on biology include decreased growth and reproduction, decreased biodiversity, and damage to skin, gills, eyes, and organs. Values of pH outside the range of 6.5 - 9 or highly fluctuating values are stressful to aquatic life. A conceptual model for pH as a stressor can be found on EPA's webpage: http://www.epa.gov/caddis/ssr_ph_int.html#highph .

### Water Quality Standards

The pH standard for Class 2B (warmwater) streams is within the range of 6.5 as a daily minimum and 9 as a daily maximum. The standard for 2A (coldwater) streams is slightly more stringent, requiring waters to be within the range of 6.5 to 8.5. Both of these standards can be found in Minn. Stat. 7050.0222, subp. 4.

**Candidate Cause Screening: pH**

Due to the natural diversity of the watershed and a variety of anthropogenic disturbances, streams of the SLRW display a wide range of pH values. The swamps and peat bogs of the Meadowlands region located in the central portion of the SLRW contribute large amounts of humic acid to streams, resulting in brown or black stained waters and pH values between 4 and 7. Streams near Makinen, Minnesota (**Water Hen Creek** and **Paleface Creek**) and the headwaters of the Embarrass River are also generally acidic due to the presence of wetlands and bogs. The combination of more alkaline geology and soils, along with the industrial and municipal discharges results in pH values between 7 and 9 throughout most of the other watershed zones.

The pH values of several impaired streams in the SLRW are exceeding state water quality standards. Low pH values near the minimum standard of 6.5 have been observed in several impaired streams in the Meadowlands region of the watershed, including **Skunk Creek**, **Vaara Creek**, and **Little Swan Creek**. Due to the fact that these conditions are predominantly due to natural background conditions in these watersheds, pH will not be thoroughly evaluated as a stressor in this report. However, the low productivity and pH levels on these streams are likely limiting fish and macroinvertebrate diversity and abundance.

Elevated pH concentrations, in the range of 8-9, are regularly observed in the Iron Range streams, and occasionally in the Duluth Urban streams that drain to the St. Louis River Estuary. **Manganika Creek** is the only impaired stream that regularly exceeded the 9.0 during the monitoring completed for SID (2011-2014). Historic data from other impaired streams also show elevated pH concentrations above 9.0, but recent data from these streams were all within WQ guidelines, calling into question the validity of the earlier measurements, or the possibility that these streams are in a more suitable pH regime in the present day.

### 3.1.5 Specific conductivity, Sulfate, Chloride, and Total Dissolved Solids

Specific conductance refers to the collective amount of ions in the water. In general, the higher the level of dissolved minerals in water, the more electrical current can be conducted through that water. The presence of dissolved salts and minerals in surface waters does occur naturally, and biota are adapted to a natural range of ionic strengths. However, industry runoff and discharges, road salt, urban stormwater drainage, agricultural drainage, WWTP effluent, and other point sources can increase ions in downstream waters. Aquatic organisms maintain a careful water and ion balance, and can become stressed by an increase in ion concentrations. Ions of many elements, such as calcium, sodium, and magnesium are necessary for aquatic health, but imbalances can be toxic (SETAC 2004). There has not been much research into how specific ions, and at what level, can become toxic to individual species. Associations from research, between species and toxicity levels of ionic strength are limited, and so it may be difficult to confidently conclude that specific conductance is a stressor. The causes and potential sources for high ionic strength are modeled at: http://www.epa.gov/caddis/ssr_ion_int.html .

**Effects on Aquatic Life**

There is debate as to the exact mechanisms responsible for toxicity associated with specific conductivity. Toxicity due to specific conductivity could result from disruption of organisms' osmotic regulation processes, decreases in bioavailability of essential elements, increases in availability of heavy metal ions, increases in particularly harmful ions, changes in ionic composition, absence of chemical constituents that offset impacts of harmful ions, a combination of the above, or other as yet unknown mechanisms. In some instances (perhaps the majority), increased specific conductivity causes shifts in community composition rather than mortality. Thus, specific conductivity, salinity, and TDS levels may be associated with biological impairment and yet be below mortality thresholds.

Biological effects of conductivity are often difficult to quantify. Increased specific conductivity can cause community shifts favoring ion tolerant taxa and an increase in ion tolerant life stages, but it is difficult to separate the role of specific conductivity in this shift from influence of confounding stressors. With increases in specific conductivity, macroinvertebrate taxa richness (particularly Ephemeroptera sp.) has been found to decrease (Piscart et al. 2005). Echols et. al (2009) observed a reduction in EPT abundance as conductivity values increased. A study of Minnesota biological data and stressor linkages found that sites with specific conductivity exceeding 1,000 µS/cm rarely meet the biological integrity impairment thresholds for general use streams (MBI 2012). Laing (personal communication 2014) developed predictive regression models of overall taxa richness and EPT taxa richness and specific conductance values using a paired, statewide and regional data set (Figure 14). Based on these regressions, the probability of a macroinvertebrate community to support EPT richness on par with high quality streams decreases considerably when specific conductivity levels exceed 500 to 1,000 µS/cm (Figure 14).

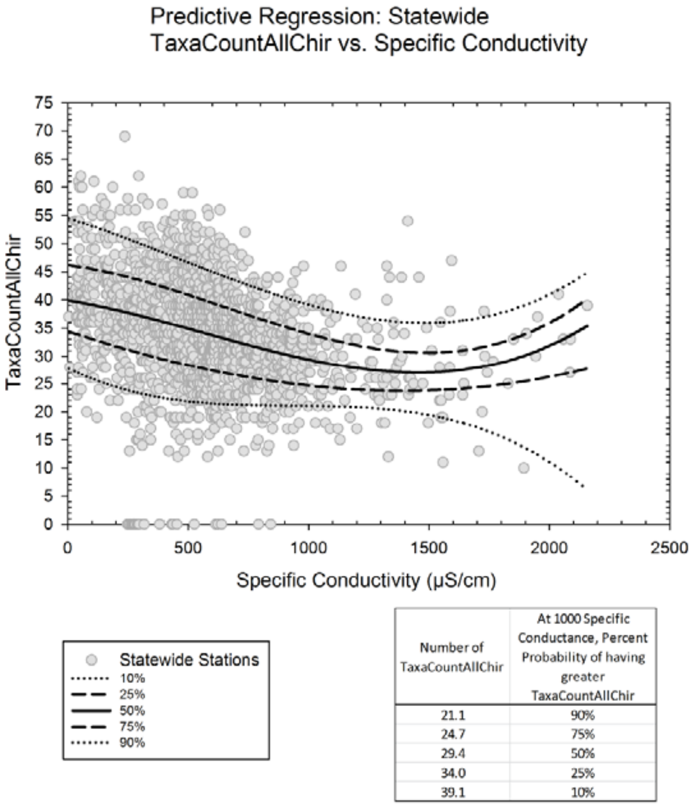
The EPA's technical paper *A Field-Based Benchmark for Conductivity in Central Appalachian Streams* listed a value of 300 µS/cm as a chronic benchmark for protecting sensitive aquatic life (EPA 2011). This benchmark is based on field data from portions of the states of WV and KY, and the authors believe this value is applicable to portions of OH, PA, TN, VA, AL, and MD. Background conductivity values derived from data from this region and used in this report ranged from 66 to 214 µS/cm, which is comparable to the SLRW streams. Although it cannot be used as a water quality standard to determine impairments in the St. Louis River drainage, this benchmark should be considered when evaluating potential impacts of high conductivity in this region and/or developing a conductivity standard for Minnesota streams.

### Water Quality Standards

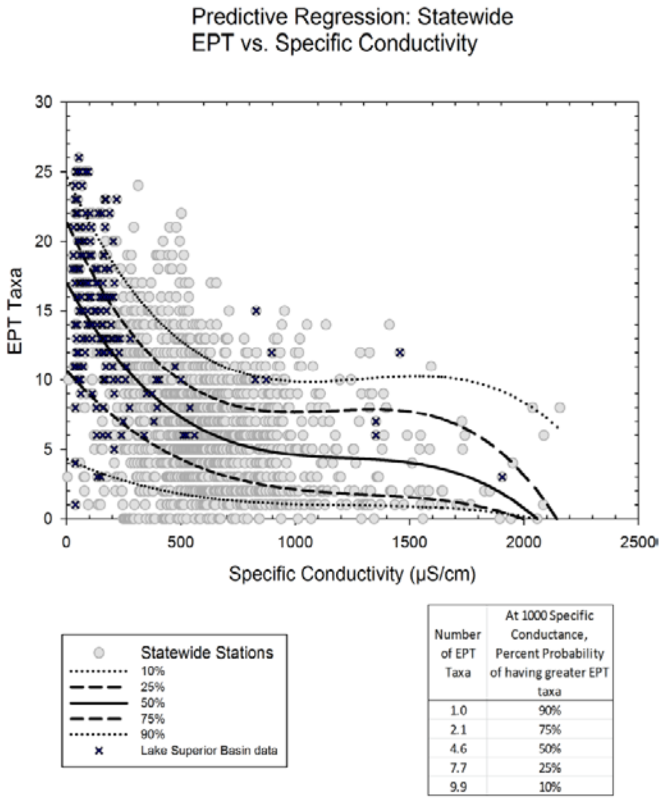
Minnesota does not have an aquatic life standard for Specific Conductance.

### Types of Ionic Strength Data

Specific conductance readings can be collected by deployed devices at defined time intervals, or a single, instantaneous reading taken during a site visit. Specific conductivity data collected in the SLRW involved both of these data collection methods.



| Number of TaxaCountAllChir | At 1000 Specific Conductance, Percent Probability of having greater TaxaCountAllChir |
|---|---|
| 21.1 | 90% |
| 24.7 | 75% |
| 29.4 | 50% |
| 34.0 | 25% |
| 39.1 | 10% |



| Number of EPT Taxa | At 1000 Specific Conductance, Percent Probability of having greater EPT taxa |
|---|---|
| 1.0 | 90% |
| 2.1 | 75% |
| 4.6 | 50% |
| 7.7 | 25% |
| 9.9 | 10% |

**Figure 14:** Predictive regression graphs for specific conductivity and two macroinvertebrate integrity metrics

### Candidate Cause Screening: Specific Conductivity

Specific conductivity values vary widely among the streams of the SLRW due to natural factors (e.g. local geology, wetlands and lakes, groundwater) and anthropogenic land-uses that have altered the natural condition of surface and groundwater (e.g. mining, urbanization, wastewater treatment). In areas of the SLRW that are relatively unaffected by mining, urbanization, or agriculture, stream conductivity values ranged from 36 to 380 µS/cm and were generally below 230 µS/cm (see Section 1.2.3). In general, conductivity values exceeding 500 µS/cm are limited to streams with highly urbanized watersheds and/or effluent discharges from industrial or municipal sources (e.g. mine pits and WWTP). Maximum specific conductivity levels observed at SLRW stations were plotted to examine the spatial distribution of stations that could potentially be impacted by this stressor (Figure 15). Streams of the Iron Range, Duluth Metro area, and the St. Louis River main stem are the three regions of the SLRW where elevated specific conductivity levels are most commonly observed.

Specific conductivity was identified as a candidate stressor in the following impaired streams: **Miller Creek** and **Kingsbury Creek** in Duluth; **Wyman Creek, Spring Mine Creek** and the **Embarrass River** in the extreme headwaters of the SLRW; **East Swan Creek** near Hibbing; **Elbow Creek** and **Manganika Creek** near the towns of Eveleth and Virginia; and the **West Two River** and **Kinney Creek** (a.k.a. Unnamed tributary to McQuade Lake).

The impaired streams located in the central portion of the SLRW exhibit low specific conductivity values in comparison to other streams of the SLRW. Specific conductivity levels rarely exceed 150 µS/cm in some of these streams, and are often well below 100 µS/cm. Stream-dwelling organisms require water of some minimal ionic concentration, and some research indicates that waters low in ionic concentration can limit abundance and diversity of aquatic flora and fauna (Allan 1995). These instances of low conductivity in the watershed are driven by natural background factors, and as a result, will not be analyzed in detail as potential stressors in this report. However, it is important to keep this potential influence in mind when developing plans for improving impaired streams in this region of the watershed.

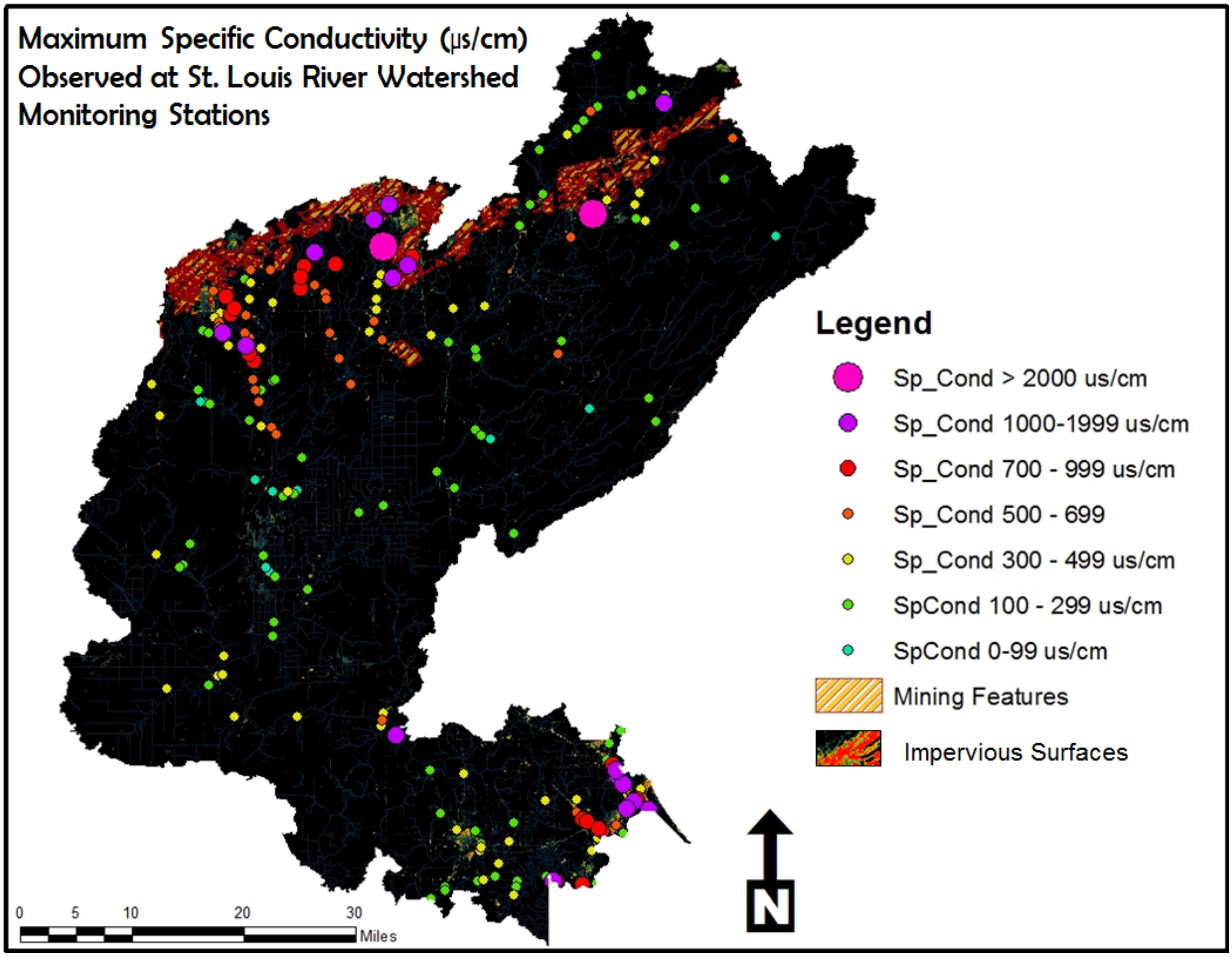


**Figure 15:** Maximum specific conductivity levels observed at stations across the St. Louis River 8 HUC watershed

## 3.1.6 Sulfate Toxicity

Sulfate is a common compound generally found in low concentrations in natural streams. Natural sources of sulfate in surface waters include the decomposition of leaves, atmospheric deposition, or the weathering of certain geologic formations including pyrite (iron disulfide) and gypsum (calcium sulfate) (Pennsylvania DEP). A variety of anthropogenic activities on the landscape can result in elevated $SO_4^-$ concentrations, including wastewaters from mining or industrial processes, and runoff from urban and agricultural areas.

Elevated sulfate concentrations in surface waters of the SLRW have been widely documented. An excerpt from a recent paper by Berndt and Bavin (2012) offers a good summary of sulfate sources and interaction with other elements on the land and in the water column:

**Taken verbatim from Berndt and Bavin (2012):**

> *It has long been known that mining activities on the Iron Range result in release of sulfate (SO4) to the St. Louis River. Most of this SO4 is released from the oxidation of minor sulfide minerals that are exposed to oxygen in waste rock piles and tailings placed on land. Although sulfide oxidation sometimes creates acidic conditions in other ore mining districts, acid produced in this region appears to be fully neutralized by dissolution of carbonate minerals that are abundant in the iron formation. Thus, in addition to elevated* $SO_4$*, these waters tend to have high alkalinity (*$HCO3^-$*) and hardness (mostly* $Mg^{++}$ *and* $Ca^{++}$*) compared to waters from surrounding watersheds without mines (Berndt and Bavin, 2009). SO4 concentrations for major streams in the area rarely exceed 100 mg/L SO4, but waters sampled from pits close to the highest sulfide-bearing waste rock piles can have SO4 concentrations of 1000 mg/L and above.*

Sulfate data for the SLRW is available primarily for streams near the Iron Range mining district and along the main stem of the St. Louis River. However, some data exist for streams which lack mining and other sulfate-loading land uses in their watersheds. The map in Figure 16 on the following page shows the maximum sulfate concentrations recorded at monitoring stations where data are available. Consistent with the statement made in the excerpt from Berndt and Bavin (2012) in the previous paragraph, the spatial distribution of maximum sulfate concentrations clearly show that the highest values are observed in small streams in the immediate vicinity of mining features. Sulfate concentrations are also slightly elevated all along the St. Louis River main stem, but concentrations are diluted significantly due to the volume of water and distance from sulfate sources. Miller Creek, a coldwater stream within the city of Duluth, also shows slightly elevated sulfate levels due to its heavily urbanized watershed.

## Water Quality Standards

The MPCA is working to revise the existing sulfate standard for protection of wild rice bearing waters. The proposed approach takes into account local levels of iron and organic carbon, as these variables can affect the amount of sulfide (more toxic form) that is produced by bacteria in stream/lake bottom sediments.

There is currently no sulfate standard in Minnesota designed to protect fish and aquatic macroinvertebrates.

### Sulfate Reduction, Sulfide Toxicity, and Methylmercury

Sulfate loading also plays a role in mercury methylation and can influence biota in through other pathways under certain environmental conditions.

**(*Text in italics taken verbatim from MPCA 2006*)**

> *Research indicates a correlation between sulfate loading and methylmercury (MeHg) production and P mobilization under certain conditions (MPCA 2006). Many waters of the state are impaired as a result of MeHg in fish tissues and excess nutrients. The mechanisms associated with enhanced MeHg production and P availability are different, but are both associated with the tendency during decay of organic matter for natural*

*bacteria to convert sulfate to sulfide after oxygen is depleted. This group of bacteria is called sulfate-reducing bacteria (SRB).*

The reduction of sulfate to sulfide through this process represents another stressor pathway. Sulfides, especially hydrogen sulfide ($H_2S$), are quite soluble in water and are toxic to both humans and fish, though elevated concentrations are usually restricted to anaerobic conditions. Due to a lack of sulfide data for the streams evaluated for this report, and the lack of any reported fish kills or toxicity symptoms, this specific stressor pathway will not be discussed in great detail in this report. Instead, a greater emphasis will be placed on the toxicity of sulfate. The MPCA and partners are continuing research on the dynamics of sulfate, mercury, and other factors (organic carbon, phosphorous, iron) that interact to create harmful conditions for aquatic biota and human health.

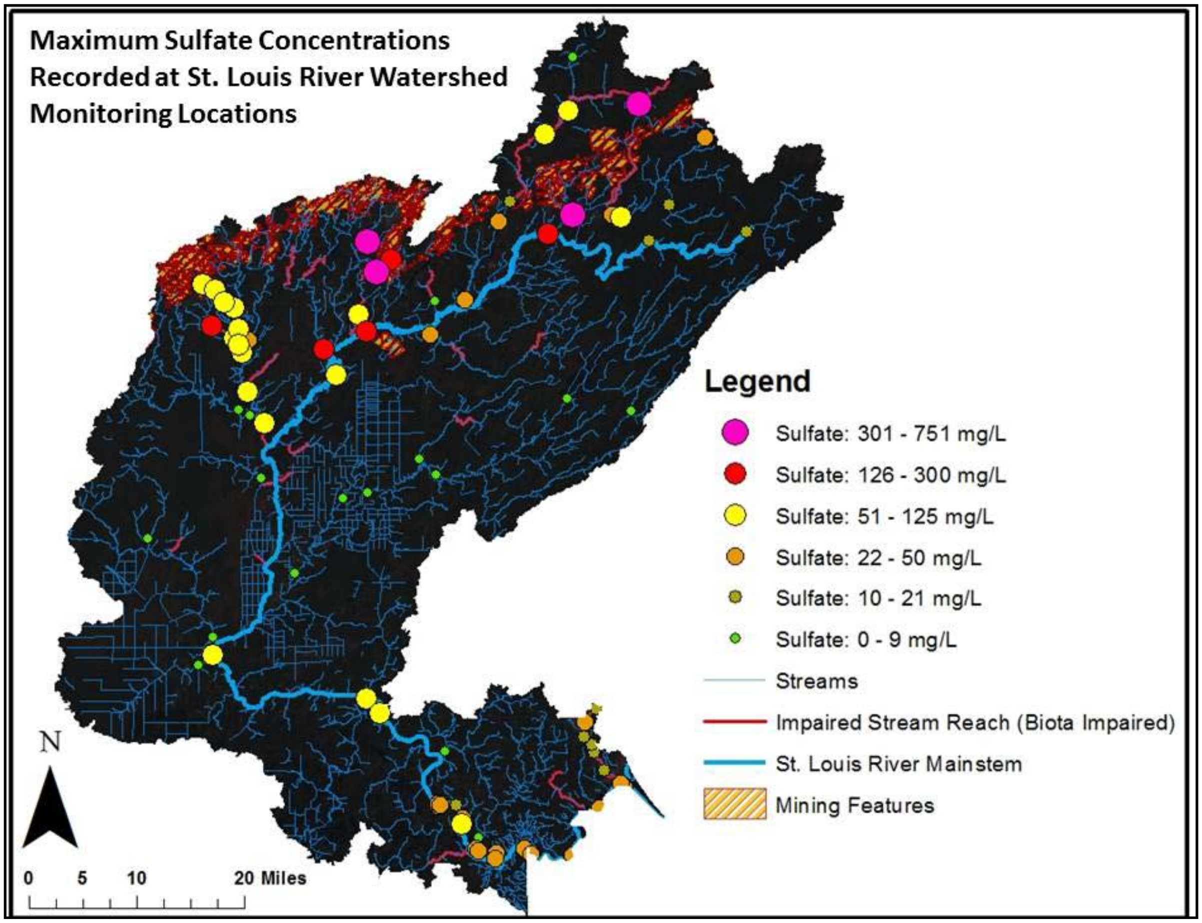


**Figure 16:** Maximum sulfate concentrations observed in streams across the St. Louis River 8 HUC watershed

## Effects of Sulfate on Aquatic Life

The specific effects of sulfate on aquatic life have been investigated for relatively few fish and macroinvertebrate species. As a result, there is a fair amount of uncertainty with respect to sulfate thresholds for protecting sensitive aquatic life. In particular, research related to sulfate toxicity is limited for aquatic macroinvertebrates, which may be more sensitive to sulfate than other commonly used species for toxicity testing.

The limited data available suggests that certain aquatic insects may be vulnerable to impacts from sulfate. Groestch and Palmer (1997) observed that sodium sulfate was “considerably more toxic to *Tricorythus sp.* mayflies than sodium chloride. This study also concluded that the mortality observed could not be linked to conductivity or total dissolved solids (TDS) concentrations, but that the nature of the salt was important for understanding the true cause of the effect. Research from streams in Central Appalachian streams affected by coal mining revealed that the mayfly *Centroptilum traingulifer* and mussel *Lampsilis siliquoidea* were highly sensitive to elevated TDS dominated by sulfate salts, whereas commonly used test species *C. dubia* and *H. azteca* were relatively unaffected.

### Toxicity Testing and Water Quality Standards for Sulfate

Over the past decade, there has been a growing interest in studying the toxicity of sulfate in aquatic ecosystems. Sulfate toxicity has been evaluated in recent years through laboratory testing of various organisms (Elphick et al. 2010; Soucek and Kennedy 2005) and in some cases by various state agencies looking to further understand sulfate related stressors (Rankin 2003, 2004) or develop water quality standards for sulfate (Buchwalter 2013; DEP Pennsylvania; Iowa DNR 2009). Table 8 provides a summary of these investigations and resulting water quality standards or benchmarks for protecting aquatic life from the effects of sulfate.

The research completed by Soucek and Kennedy (2005) has been particularly influential in the development of sulfate standards in the states of Illinois, Iowa, and Pennsylvania. Their research focused on the effects of chloride, hardness, and acclimation on the acute toxicity of sulfate to freshwater macroinvertebrates. The authors concluded that chloride concentrations and water hardness are important variables that control the toxicity of sulfate in surface waters. At low chloride concentrations (between 5 and 25 mg/l) chloride may lessen the toxic effect of sulfate, but at higher concentrations it may add to the toxicity, hence the two equations where chlorides are added in one and subtracted in the other. Chloride concentrations below 5 mg/L had no effect on the toxicity of sulfate. The authors of this study found that hardness ameliorates the toxicity of the sulfate.

### Candidate Cause Screening: Sulfate Toxicity

Given the lack of an aquatic-life based sulfate standard in Minnesota, a combination of the guidelines and standards shown in Table 8 and 9 will be used to evaluate sulfate as a candidate stressor in the SLRW. Several of the standards and guidelines summarized in this section are the focus of ongoing research. Taking this into consideration, some caution will be used in terms of diagnosing sulfate as a stressor without applicable Minnesota water quality standards as an additional piece of supporting evidence.

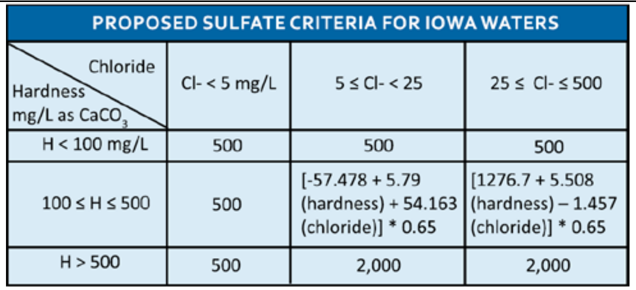
The lowest toxicity value for sulfate included in Table 8 is a chronic criterion of 75 mg/L for soft-water (10-40 mg/L) as reported in Elphick (2010). The biota-impaired streams of the SLRW with hardness within or near the range of 10-40 mg/L for portions of the year generally have low concentrations of sulfate (n = 56, max = 57.7 mg/L; min = > 1 mg/L; median = 6.8 mg/L). This includes all impaired streams in the MDW-PB, NSH-ML, and ML Watershed zones. Based on the monitoring results from these streams, sulfate is eliminated as a candidate cause for impairment in these watershed zones.

The next most-protective sulfate toxicity benchmark cited is a chronic criterion value of 124 mg/L SO4 (Buchwalter 2010). This criterion is not adjusted based on ambient water hardness values or chloride concentrations, and can be considered one of the more “protective” standards. This criterion will be applied to the remaining SLRW biota-impaired streams that are not considered to have “soft” water (hardness 10-40 mg/L) for the purposes of selecting sulfate as a candidate cause for further evaluation. Sulfate concentrations from five streams with IBI impairments exceeded 124 mg/L of sulfate in at least one sample. These include, **Spring Mine Creek**, **Elbow Creek**, **Manganika Creek**, **West Two River**, and **Kinney Creek**. Sulfate toxicity is considered a candidate cause for impairment in these streams and will be further evaluated in Section 5. Sulfate data will also be discussed further in the analysis of stressors for impairments in **Wyman Creek** and the **Embarrass River** as well, due to the presence of mining discharges in close proximity to these impairments.

**Table 9:** Summary of aquatic life standard for sulfate used in several U.S. states

| Location | Author | Test species or biological response variable | Other WQ Factors | Sulfate Target or Biological Response |
|---|---|---|---|---|
| **British Columbia** | Elphick et al.; 2010 | **Invertebrates** *(C. dubia, Brachionus calyciflorous, H. azteca)*<br>**Fish** (Rainbow Trout, coho salmon, Fathead Minnow)<br>**Amphibian** (Pacific tree frog) | Hardness | **Values based on SSD* Data:**<br>Hardness 10-40 mg/L -----> 129 mg/L SO4<br>Hardness 80-100 mg/L -----> 644 mg/L SO4<br>Hardness 160-250 mg/L -----> 725 mg/L SO4<br>**Values based on "safety factor approach":**<br>Hardness 10-40 mg/L -----> 75 mg/L SO4<br>Hardness 80-100 mg/L -----> 625 mg/L SO4<br>Hardness 160-250 mg/L -----> 675 mg/L SO4 |
| **California** | Buchwalter, 2010) | Sulfate toxicity data from EPA's ECOTOX database | None; mentions need to further evaluate impacts of chloride/hardness | Acute Criterion -----> 234 mg/L SO4<br>Chronic Criterion -----> 124 mg/L SO4 |
| **Ohio** | Ohio EPA (Rankin, 2003,2004) | Paired water quality and biological data from wadable streams in Ohio | Chloride | Reduced Invertebrate IBI @ SO4 around 400 mg/L<br>Reduced # Quality EPT Taxa @ SO4 > 500 mg/L<br>*(2004 paper revealed reduced biological response to sulfate toxicity when chloride concentrations are elevated above background levels)* |

**Table 8:** Summary of research focusing on the toxicity of sulfate to fish and macroinvertebrates

| **Location** | **Author** | **Test species or biological response variable** | **Other WQ Factors** | **Sulfate Water Quality Standards** |
|---|---|---|---|---|
| **Illinois** | Soucek and Kennedy (2004) | Toxicity tests of over 30 organisms through a collaboration between State of Illinois and USEPA Duluth Toxicity Laboratory | Hardness / Chloride |  |
| **Pennsylvania** | Pennsylvania Dept. Env. Protection | | | |
| **Iowa** | Iowa DNR (2009) | | | |

PROPOSED SULFATE CRITERIA FOR IOWA WATERS

| Chloride / Hardness mg/L as $CaCO_3$ | Cl- < 5 mg/L | 5 ≤ Cl- < 25 | 25 ≤ Cl- ≤ 500 |
|---|---|---|---|
| H < 100 mg/L | 500 | 500 | 500 |
| 100 ≤ H ≤ 500 | 500 | [-57.478 + 5.79 (hardness) + 54.163 (chloride)] * 0.65 | [1276.7 + 5.508 (hardness) – 1.457 (chloride)] * 0.65 |
| H > 500 | 500 | 2,000 | 2,000 |

### 3.1.7 Chloride

The negative effects of elevated chloride concentrations on aquatic life have been well documented, especially in urban areas. Chloride enters the environment in small amounts through the dissolution of mineral salts, but human uses of chloride salts result in the greatest input to surface waters. Of greatest importance are sources from municipal and industrial discharges containing salt wastes from water softening or process water, and stormwater sources associated with use of chlorides in road de-icing salts, agricultural runoff (livestock waste and fertilizer) and produced water from oil and gas wells. The use of road salt and de-icing products has increased considerably in the United States since 1950, putting more urban streams at risk for this stressor (Kostick 1993). Road-salt runoff to surface water has caused detrimental effects on water quality and aquatic life on local, regional, and national scales (Corsi et al 2010).

#### Water Quality Standards

The recommended national criteria (EPA) and current Minnesota water quality standard for chloride are established at a chronic value of 230 mg chloride/liter, implemented as a four-day average concentration and acute (maximum concentration) of 860 mg chloride / liter, implemented as a one-day average concentration.

#### Specific Conductivity as a Surrogate for Chloride

Researchers at the University of Minnesota-Duluth’s Natural Resources Research Institute (NRRI) in Duluth, Minnesota have developed a regression equation for predicting chloride concentrations based on specific conductivity readings (Figure 17) (duluthstreams.org). The formula was developed from laboratory testing that involved intruding various quantities of city of Duluth road salt to surface water from streams within the city limits. This regression equation will be used to evaluate chloride as a candidate stressor in two coldwater streams located within the city limits of Duluth (**Kingsbury Creek** and **Miller Creek**). This relationship between specific conductivity and chloride is the strongest in urban streams that have no other significant source of dissolved solids other than road salt application. Therefore, the same regression cannot be used in streams that are impacted by point source discharges (e.g. WWTP and industrial effluent).

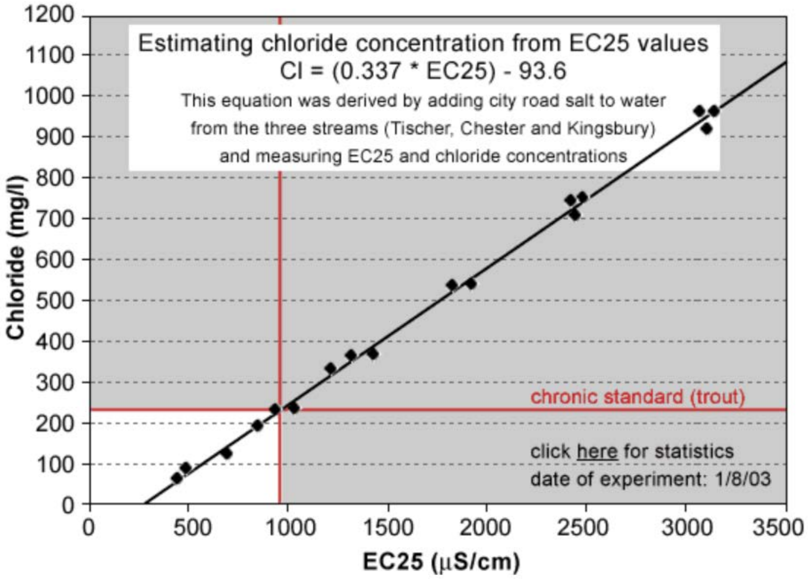


**Figure 17: (from duluthstreams.org, NRRI)** Regression equation for chloride and specific conductivity in Duluth streams

## Effects of Chloride on Aquatic Life

Several studies have found negative relationships between increases in conductivity and biological integrity. The majority of the negative effects have been observed in macroinvertebrate communities, but fish have also been affected in some instances. Roy et al (2003) found that specific conductance was a significant predictor of urbanization and was negatively related to total invertebrate richness, EPT richness, and total invertebrate density. Echols et al (2009) also noted a decrease in EPT taxa richness (when Hydropsychid caddisflies were excluded) downstream of a point source discharge which increased conductivity levels. Other documented impacts of elevated specific conductivity include lower scores in the Shannon (macroinvertebrate) Diversity Index and decreases in intolerant macroinvertebrate taxa (Johnson et al 2012).

**Table 10:** Documented biological responses to elevated chloride concentrations

| Metric | Response | Source |
|---|---|---|
| EPT Richness | Decrease | Roy et al (2003); Echols et al (2009); Johnson et al (2013) |
| Overall Taxa Richness | Decrease | Johnson et al (2013) |
| Ephemeroptera Richness | Decrease | Pond (2004); Hassel et al (2006) |
| Tolerance Indicator Values (Chloride / Sp. Conductivity | Increase | MBDI (Yoder and Rankin, 2012) |

## Candidate Cause Screening: Chloride Toxicity

A review of available SLRW chloride results revealed that concentrations are above natural background conditions in many of the impaired streams. However, very few violations of state WQ standards were observed, with the exception of the two Duluth Urban coldwater streams, Kingsbury Creek and Miller Creek. Therefore, further analysis of chloride toxicity as a candidate cause of impairment will be limited to these two watersheds.

## 3.1.8 Total Suspended Solids and Turbidity

Sediment and turbidity have been shown to be among the leading pollutant issues affecting stream health in the United States (EPA 2003). Recent studies in Minnesota have demonstrated that human activities on the landscape have dramatically increased the sediment entering our streams and rivers since European settlement (Triplett et al. 2009; Engstrom et al. 2009). Sediment can come from land surfaces (e.g., exposed soil), or from unstable stream banks (see geomorphology section for details). The soil may be unprotected for a variety of reasons, such as construction, mining, agriculture, or insufficiently-vegetated pastures. Human actions on the landscape, such as channelization of waterways, riparian land cover alteration, and increased impervious surface area can cause stream bank instability leading to sediment input from bank sloughing. Although sediment delivery and transport are an important natural process for all stream systems, sediment imbalance (either excess sediment or lack of sediment) can be detrimental to aquatic organisms.

### Suspended sediment

As described in a review by Waters (1995), excess suspended sediments cause harm to aquatic life through two major pathways: (1) direct, physical effects on biota (i.e., abrasion of gills, suppression of photosynthesis, avoidance behaviors); and (2) indirect effects (i.e., loss of visibility, increase in sediment oxygen demand). Elevated turbidity levels and TSS concentrations can reduce the penetration of sunlight and can thwart photosynthetic activity and limit primary production (Munawar et al. 1991). Sediment can also cause increases in water temperature as darker (turbid) water will absorb more solar radiation.

### Deposited sediment

Whereas suspended sediment is a stressor operating in the water column, sediment is also deposited onto the stream bottom, and thus can have different effects on organisms oriented to living on or within the streambed substrate (this includes many of the macroinvertebrate taxa). Excess fine sediment deposition on benthic habitat has been proven to adversely impact fish and macroinvertebrate species that depend on clean, coarse stream substrates for feeding, refuge, and/or reproduction (Newcombe et al. 1991). Excessive deposition of fine sediment can degrade macroinvertebrate habitat quality, reducing productivity and altering the community composition (Rabeni et al. 2005 Burdon et al. 2013). Aquatic macroinvertebrates are affected in several ways: (1) loss of certain taxa due to changes in substrate composition (Erman and Ligon 1988); (2) increase in drift (avoidance behavior, using current to seek a new suitable location) due to sediment deposition or substrate instability (Rosenberg and Wiens 1978); and (3) changes in the quality and abundance of food sources such as periphyton and other prey items (Pekarsky 1984).

Fish communities are typically influenced through: (1) a reduction in spawning habitat or egg survival (Chapman 1988); and (2) a reduction in prey items as a result of decreases in primary production and benthic productivity (Bruton 1985; Gray and Ward 1982). Fish species that are simple lithophilic spawners require clean, coarse substrate for reproduction. These fish do not construct nests for depositing eggs, but rather broadcast them over the substrate. Eggs often find their way into interstitial spaces among gravel and other coarse particles in the stream bed. Increased sedimentation can reduce reproductive success for simple lithophilic spawning fish, as eggs become smothered by sediment and become oxygen deprived.

Organic particles (including algae) can contribute to TSS. Testing for Total Suspended Volatile Solids (TSVS) allows for the determination of the particle type, and provides information on the source of the problem. Unusually high concentrations of TSVS can be indicative of excess nutrients (causing algal growth) and an unstable DO regime. Determining the type of suspended material (mineral vs organic) is important for proper conclusions about the stressor and source (erosion vs. nutrient enrichment vs. a wastewater discharge). More information on sediment effects can be found on EPA's CADDIS webpage: http://www.epa.gov/caddis/ssr_sed_int.html.

### Types of Sediment Data

Particles suspended in the water column can be either organic or mineral. Generally, both are present to some degree and measured as TSS. Typically, fine mineral matter is more concerning and comes from soil erosion of land surfaces or stream banks. TSS is determined by collecting a stream water sample and having the sample filtered and weighed to determine the concentration of particulate matter in the sample. To determine the mineral component of the suspended particles, a second test is run using the same procedure except to burn off the organic material in an oven before weighing the remains, which are only mineral material. Quantitative field measurement of deposited sediment (bedload) is very difficult. Deposited sediment is visually estimated by measuring the degree to which fine material surrounds rock or woody substrate within the channel (embeddedness). Deposited sediment is also analyzed by randomly measuring numerous substrate particles (Wolman 1954) and calculating the $D_{50}$ particle size.

### Water Quality Standards

Since the late 1960's, the MPCA has used a turbidity standard measured in nephelometric turbidity units (NTU) as a means of addressing aquatic life use impacts resulting from increased suspended particles (sediment, algae, etc.). Although many rivers remain listed as impaired for turbidity (including several streams in the SLRW), the MPCA recently put into rule a water quality standard based on TSS criteria. Unlike turbidity, TSS is a "concentration-based" parameter, which facilities the development of load allocations during the TMDL process. For additional information on the water quality standard for TSS, refer to the following internet link (http://www.pca.state.mn.us/index.php/view-document.html?gid=14922).

The new TSS criteria are stratified by geographic region and stream class (e.g. coldwater, warmwater) to account for differences in natural background conditions and biological sensitivity. The draft TSS standard for warmwater and coolwater streams of the SLRW is 15 mg/L TSS. Coldwater streams have a slightly lower impairment threshold value of 10 mg/L TSS (Table 11). An impairment listing may occur when these values are exceeded in more than 10% of samples during the months of April through September, within a minimum of three total results that exceed the standard.

**Table11:** Summary of applicable TSS and Secchi tube standard for streams and rivers of the SLRW

| Use class | TSS Region | TSS | Secchi Tube |
|---|---|---|---|
| 2A (Coldwater Trout Streams) | Applies to all | < 10 mg/L | > 55 cm |
| 2B (Warmwater) | Northern RNR | < 15 mg/L | > 40 cm |

For the purposes of SID, TSS results will be relied upon to evaluate the effects of suspended solids and turbidity on fish and macroinvertebrate populations. The available turbidity data for the watershed exists in several different units of measurement, and at times the equipment used to measure turbidity can produce erroneous results if instrumentation is not calibrated adequately. TSS results are available for the watershed from state-certified laboratories and the existing data covers a much larger spatial and temporal scale in the watershed.

## Candidate Cause Screening: Elevated TSS

Reference or background TSS concentrations in the SLRW are relatively low compared to other regions of Minnesota. McCollor and Heiskary (1993) observed that the 75th percentile value for a set of minimally impacted streams of the Northern Lakes and Forests ecoregion was 6.4 mg/L. By comparison, the 75th percentile TSS concentration from the selection of SLRW reference streams discussed in Section 1.2.4 ranged from 5.7 mg/L to 10.5 mg/L depending on the grouping used (Table 12). The median TSS values at SLRW reference sites ranged from 3.2 mg/L to 5.6 mg/L, well below both the warmwater and coldwater TSS standard. These results further support the claim of natural background conditions with low TSS concentrations in the SLRW.

**Table 12:** Summary statistics for TSS calculated using data from SLRW reference stations

| | | | | MAX | | | MEDIAN | | | MIN |
|---|---|---|---|---|---|---|---|---|---|---|
| Parameter | N | Mean | SD | 100% | 95% | 75% | 50% | 25% | 5% | 0% |
| TSS (Reference Group A) | 137 | 4.3 | 2.9 | 20.0 | 8.8 | 5.7 | 3.5 | 2.3 | 1.0 | 0.4 |
| TSS (Reference Group B) | 64 | 6.3 | 12.7 | 98.0 | 15.9 | 5.8 | 3.2 | 2.0 | 1.0 | 0.5 |
| TSS (Reference Group C) | 76 | 9.7 | 15.3 | 120.0 | 31.3 | 10.5 | 5.6 | 2.3 | 0.8 | 0.5 |

Available TSS data for the impaired stream reaches were compiled and evaluated for the impaired streams to identify areas where elevated TSS may be a candidate cause of biological impairments. Impaired streams of the DUC and SR-HIB watershed zones show the highest TSS concentrations among the streams analyzed. These watershed zones include two urban trout streams, **Miller Creek** and **Kingsbury Creek,** as well the **Swan River** on the in the vicinity of Hibbing, Minnesota. TSS concentrations in both of these streams frequently exceed the warmwater and coldwater water quality standard, particularly during spring snowmelt and large rain events. Elevated TSS is considered a candidate cause for impairment in these watershed zones and will be further evaluated. **Wyman Creek**, the lone impaired stream in the LU-P River Watershed zone, exceeded the 10 mg/L coldwater TSS standard. TSS will also be further evaluated as a candidate cause in **Wyman Creek** based on these results.

The TSS concentrations in streams of the MF-PB Watershed zone are slightly elevated and occasionally exceed water quality targets for coldwater and warmwater streams. The majority of the TSS results between 10 and 30 mg/L in this watershed zone were observed in **Little Swan Creek**, a coldwater tributary of the East Swan River. Over 30% (8 of 25 samples) of the TSS results from this stream exceed the 10 mg/L TSS standard for coldwater streams. **Sand Creek** and **Stony Creek** exceeded the 15 mg/L warmwater TSS standard during several spring and summer monitoring events. TSS is considered a candidate cause for impairment and will be further evaluated for linkages to biota impairments in this watershed zone.

**Otter Creek** in the ML-NSH Watershed zone narrowly exceeded the 10 mg/L TSS standard during a 2012 snowmelt sampling event. However, this stream generally exhibits low TSS concentrations, as summer baseflow samples ranged from <1 mg/L to 3 mg/L. TSS is not considered a candidate cause for impairment in Otter Creek and will not be evaluated as a stressor in the ML-NSH Watershed zone.

The TSS will be further evaluated as a candidate cause for impairment in **Manganika Creek**, a short warmwater stream that serves as the outlet of Manganika Lake and a tributary stream to the East Two River. The TSS concentrations in **Manganika Creek** are highest during the summer low flow periods and are at least partially due to algae blooms originating in Manganika Lake upstream from the monitoring station. The other impaired stream in this watershed zone, Elbow Creek, did not show any signs of elevated TSS concentrations.

### 3.1.9 Nitrogen (Nitrate Toxicity)

Nitrate ($NO_3$) and nitrite ($NO_2$) forms of nitrogen are components of the natural nitrogen cycle in aquatic ecosystems. $NO_2$ anions are naturally present in soil and water, and are readily converted to $NO_3$ by microorganisms as part of the denitrification process of the nitrogen cycle. As a result, nitrate is far more abundant than nitrite. Although the water test commonly used measures both nitrate and nitrite, because a very large percent is nitrate, from here on this report will refer to this data as being nitrate.

Elevated nitrate concentrations in surface water have been linked to a variety of sources and pathways. Anthropogenic alterations of the landscape, namely an increase in agricultural land-use, have increased ambient nitrate concentrations in some watersheds to levels that can be toxic to some fish and macroinvertebrates (Lewis and Morris, 1986; Jensen 2003). In addition to agricultural sources, elevated $NO_2$ and $NO_3$ concentrations have also been linked to effluent from facilities producing metals, dyes, and celluloids and sewage. For more information on the sources and effects of nitrate, see the EPA's CADDIS webpages: http://www.epa.gov/caddis/ssr_nut_int.html.

In Minnesota, natural inputs of nitrate to surface waters vary by geographic location. However, when nitrate concentrations in surface water samples from "reference" areas (i.e., areas with relatively little human impact) are compared to samples from areas of greater human impact, the reference areas exhibit much lower nitrate concentrations (Monson and Preimesberger 2010). Nitrate concentrations under "reference" conditions in Minnesota are typically below 1 mg/L (Heiskary and Wilson 2005). A statistical breakdown of nitrate results from 25 reference sites in the SLRW is shown below in Table 13. Aside from a single result of 2.8 mg/L from the Partridge River near Hoyt Lakes, maximum nitrate values were below 1.0 mg/L at all of these locations.

**Table 13:** Summary statistics for nitrate nitrogen data at SLRW reference stations

| | | | | MAX | | | MEDIAN | | | MIN |
|---|---|---|---|---|---|---|---|---|---|---|
| Parameter | N | Mean | SD | 100% | 95% | 75% | 50% | 25% | 5% | 0% |
| Nitrate (Reference Group A) | 135 | 0.18 | 0.23 | 0.84 | 0.64 | 0.36 | 0.06 | 0.01 | 0.00 | 0.00 |
| Nitrate (Reference Group B) | 41 | 0.1 | 0.4 | 2.8 | 0.3 | 0.1 | 0.1 | 0.0 | 0.0 | 0.0 |
| Nitrate (Reference Group C) | 49 | 0.2 | 0.2 | 0.8 | 0.6 | 0.2 | 0.1 | 0.0 | 0.0 | 0.0 |

### Types of Nitrate Data

Nitrate samples have been collected from stream and river locations throughout the MRW. Samples were analyzed by a state certified laboratory and the data is stored in a publicly-available database: http://cf.pca.state.mn.us/water/watershedweb/wdip/search_more.cfm.

### Biological Effects of Elated Nitrate Concentrations

The intake of nitrite and nitrate by aquatic organisms has been shown to convert oxygen-carrying pigments into forms that are unable to carry oxygen, thus inducing a toxic effect on fish and invertebrates (Grabda et a 1974; Kropouva et a 2005). Certain species of caddisflies, amphipods, and salmonid fishes seem to be the most sensitive to nitrate toxicity (Camargo and Alonso 2006). Nitrate toxicity to freshwater aquatic life is dependent on concentration and exposure time, as well as the overall sensitivity of the organism(s) in question. Comargo et al (2005) cited a maximum level of 2 mg/L nitrate-N as appropriate for protecting the most sensitive freshwater species, although the in the same review paper, the authors also offered a recommendation of NO3 concentrations under 10 mg/L as protective of several sensitive fish and aquatic invertebrate taxa.

### Water Quality Standards

Minnesota currently does not have an aquatic life use nitrate standard, though the MPCA is actively developing an aquatic life standard for nitrate toxicity.

### Candidate Cause Screening: Nitrate Toxicity

All available $NO_2$ + $NO_3$ (nitrate) data for biota-impaired stream reaches in the SLRW were compiled and evaluated for potential harmful effects on aquatic life. Based on these data, elevated nitrate concentrations are clearly a candidate cause for impairment in **East Swan Creek**, where results up to 18 mg/L have been observed. Two streams in the Virginia area, **Manganika Creek** and **Elbow Creek**, were also identified as impaired streams with nitrate concentrations significantly exceeding natural background conditions for the SLRW. Nitrate toxicity will be evaluated as a stressor in each of these streams.

## 3.1.10 Ammonia-N Toxicity

Ammonia ($NH_3$) is a common toxicant derived from wastes, fertilizers, and natural processes. Ammonia nitrogen includes both the ionized form (ammonium, $NH_4^+$) and the unionized form (ammonia, $NH_3$). An increase in pH favors formation of the more toxic unionized form ($NH_3$), while a decrease favors the ionized ($NH_4^+$) form. Temperature also affects the toxicity of ammonia to aquatic life. Ammonia is a common cause of fish kills, but the most common problems associated with ammonia relate to elevated concentrations affecting fish growth, gill condition, organ weights, and hematocrit. Exposure duration and frequency strongly influence the severity of effects (Milne et al. 2000) (Text taken from EPA CADDIS).

Ammonia in sediments typically results from bacterial decomposition of natural and anthropogenic organic matter that accumulates in sediment. Sediment microbiota mineralize organic nitrogen or (less commonly) produce ammonia by dissimilatory nitrate reduction. Ammonia is especially prevalent in anoxic sediments because nitrification (the oxidation of ammonia to nitrite [$NO_2^-$] and nitrate [$NO_3^-$]) is inhibited. Ammonia generated in sediment may be toxic to benthic or surface water biota (Lapota et al. 2000). Channel alteration can result in decreased natural conversion of ammonia to nitrate, and alteration or removal of riparian vegetation can reduce the interception of nitrogen compounds in runoff from the surrounding landscape. Channel alteration and water withdrawals can reduce ammonia volatilization by reducing the turbulence of the water. For a more detailed explanation of ammonia sources and causal pathways, see: http://www.epa.gov/caddis/ssr_amm4s.html.

Ammonia also exerts a biochemical oxygen demand on receiving waters (referred to as nitrogenous BOD or NBOD) because DO is consumed as bacteria and other microbes oxidize ammonia into nitrite and nitrate. The resulting DO reductions can decrease species diversity and even cause fish kills. Additionally, ammonia can lead to heavy plant growth (eutrophication) due to its nutrient properties (see the Nutrients module). Conversely, algae and macrophytes take up ammonia, thereby reducing aqueous concentrations. A summary of the commonly observed effects of ammonia on aquatic life are listed in Table 14.

**Table 14:** Common biological effects observed in streams with elevated unionized ammonia concentrations (from EPA CADDIS website)

| Biological Effects of Ammonia Toxicity | Used in Stressor ID Analysis |
|---|---|
| ▪ Reduction or absence of ammonia-sensitive species | Yes |
| ▪ Physiological effects (e.g., decreased nitrogen excretion, decreased oxygen binding to hemoglobin) | No |
| ▪ Behavioral effects (e.g., loss of equilibrium, hyperexcitability, increased breathing) | No |
| ▪ Morphological effects (e.g., proliferation of gill lamellae, lesions in blood vessels, mucus secretion) | Yes |
| ▪ Organismal and population effects (e.g., decreased growth and abundance, mass mortality) | Yes |

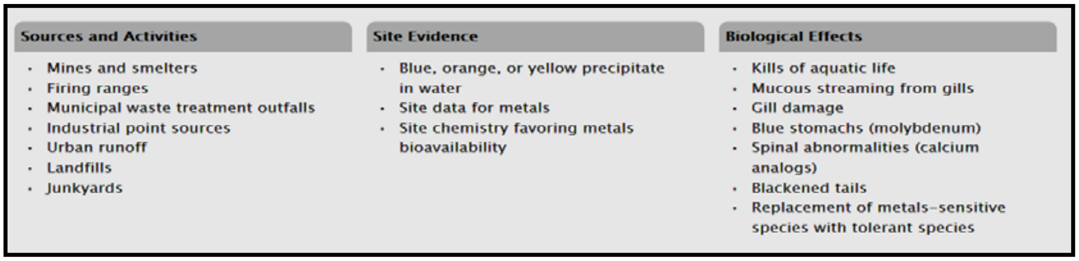
### Candidate Cause Screening: Ammonia Toxicity

All available ammonia data were compiled and evaluated for impaired streams of the SLRW. Only data that were paired with pH and temperature data were considered (required to calculate toxic unionized form). Several of the streams evaluated which receive wastewater effluent from WWTP historically carried extremely high ammonia concentrations, but more contemporary monitoring results show significantly reduced concentrations due to changes in the treatment process. Streams that will be evaluated for ammonia toxicity as a candidate cause of impairment include **Manganika Creek**, **Elbow Creek**, and **East Swan Creek**.

## 3.1.11 Metals Toxicity

While some metals are essential as nutrients, all metals can be toxic at some level, and some metals are toxic in minute amounts. Impairments result when metals are biologically available at toxic concentrations affecting the survival, reproduction, and behavior of aquatic organisms. Metals that are commonly linked to toxic effects include arsenic, cadmium, chromium, copper, lead, inorganic mercury, nickel, selenium, and zinc. A list of anthropogenic sources of metals and common effects on water quality and biota are described in Table 15. There are numerous sources in the SLRW that could contribute to increased concentrations of a variety of metals, including urban runoff, landfills, municipal and industrial point sources, and mining operations.

**Table 15:** Some common sources, indicators, and biological responses to elevated metals concentrations



| Sources and Activities | Site Evidence | Biological Effects |
|---|---|---|
| • Mines and smelters<br>• Firing ranges<br>• Municipal waste treatment outfalls<br>• Industrial point sources<br>• Urban runoff<br>• Landfills<br>• Junkyards | • Blue, orange, or yellow precipitate in water<br>• Site data for metals<br>• Site chemistry favoring metals bioavailability | • Kills of aquatic life<br>• Mucous streaming from gills<br>• Gill damage<br>• Blue stomachs (molybdenum)<br>• Spinal abnormalities (calcium analogs)<br>• Blackened tails<br>• Replacement of metals-sensitive species with tolerant species |

### Water Quality Standards

Trace metals with toxicity-based standards used in Minnesota include aluminum, arsenic, cadmium, chromium, copper, lead, nickel, selenium and zinc. See Table 16 for available WQ standards and internet links to a guidance document for guidance on calculating standards based on dissolved metals and water hardness.

**Table 16:** Summary of Minnesota water quality standards for trace metals

| Metal | CS | MS | FAV | CS | MS | FAV |
|---|---|---|---|---|---|---|
| Aluminum | 87 (Tox) | 748 (Tox) | 1,496 (Tox) | 125 (Tox) | 1,072 (Tox) | 2,145 (Tox) |
| Arsenic | 2.0 (HH) | 360 (HH) | 720 (Tox) | 53 (HH) | 360 (HH) | 720 (Tox) |
| Cadmium | Based on h2O Hardness Values (See http://www.pca.state.mn.us/index.php/view-document.html?gid=21257) | | | | | |
| Chromium | Based on h2O Hardness Values (See http://www.pca.state.mn.us/index.php/view-document.html?gid=21257) | | | | | |
| Copper | Based on h2O Hardness Values (See http://www.pca.state.mn.us/index.php/view-document.html?gid=21257) | | | | | |
| Lead | Based on h2O Hardness Values (See http://www.pca.state.mn.us/index.php/view-document.html?gid=21257) | | | | | |
| Nickel | Based on h2O Hardness Values (See http://www.pca.state.mn.us/index.php/view-document.html?gid=21257) | | | | | |
| Zinc | Based on h2O Hardness Values (See http://www.pca.state.mn.us/index.php/view-document.html?gid=21257) | | | | | |
| **CS** = chronic standard **MS**=Maximum Standard **FAV**=final acute value<br>**Tox** = based on toxicity to aquatic life **HH** = based on human health impacts | | | | | | |

## Aluminum

Aluminum occurs ubiquitously in natural waters as a result of the weathering of aluminum-containing rocks and minerals, but concentrations in surface waters can be increased directly or indirectly by human activity through industrial and municipal discharges, surface run-off, and wet and dry atmospheric deposition (Eisenreich 1980). The use of alum (aluminum sulfate) as a flocculent in water treatment facilities typically leads to high aluminum concentrations in finished waters. Weathering of sulfide ores exposed to the atmosphere in inactive mines and tailings dumps can release large quantities of sulfuric acid and metals such as aluminum (Filipek et al. 1987). The mobilization of aluminum is often episodic in nature, and is regularly associated with pH depressions (acidification) occurring during the spring snowmelt or with erosion from specific storm. At lower pH levels, the aluminum content significantly increases because of increased solubility of aluminum oxide and salts in acidic solutions.

Higher aluminum concentrations have also been observed in waters with elevated humic acid content (Brusewitz 1984). Therefore, streams located in peat-bog dominated regions of the SLRW may have naturally higher aluminum concentrations than streams with other land cover types. Dissolved organic matter (DOM) and sulfates in water may bind with Al and alter its bio-availability. DOM can typically complex 50-70% of the dissolved Al in natural waters at pH 4.5 – 6.5, and the result is a decrease in the bioavailability of Al to aquatic organisms.

Aluminum toxicity has been studied extensively in fish, and to a lesser extent for aquatic macroinvertebrates. In aquatic systems, bioavailability and toxicity of aluminum is closely related to ambient pH. Aluminum is only sparingly soluble in the pH range that is found in most streams of the SLRW (6.0 to 8.5). At moderate pH (5.5-7.0), fish and invertebrates may be stressed due to aluminum adsorption onto gill surfaces and subsequent asphyxiation (Sparlling et al. 1997).

In general, aquatic invertebrates are less sensitive to aluminum toxicity than fish. The addition of 400-500 µg/L Al within a pH range of 4.0 – 4.3 had negligible effects on mortality in clams (*Pisidium* sp.), amphipods (*Hyallela* sp.), snails (*Amnicola* and *Physella* sp.), or insect larvae (*Enallagma* sp., *Lepidostoma* sp., or *Pycnopsyche* sp.). Similarly, additions of neither 350 µg/L nor 1,000 µg/L Al at the same pH range affected survivorship in larval benthic insects (Havas and Likens 1985).

**Water Quality Standard: Aluminum**

The water quality standard for aluminum in is listed in Table 16. Unlike other metals evaluated in this report, the standard for aluminum is not adjusted based on water hardness.

**Candidate Cause Screening: Aluminum Toxicity**

Monitoring results for total aluminum were evaluated for all impaired streams of the SLRW. In general, existing data were minimal for these streams, and in many cases the available results were associated with high flow events. Aluminum concentrations were found to be very high in the **Swan River**, exceeding the maximum standard (MS) and approaching the acute standard (FAV) for warmwater streams. These results were associated with two high flow events in 2012, and the high aluminum levels were undoubtedly associated with extremely high suspended sediment concentrations stemming from erosion and overland runoff. Considering the episodic nature of these results, it is unclear whether or not aluminum concentrations remain elevated for long enough duration to stress fish and macroinvertebrate assemblages. Additional aluminum data is needed for further analysis of this stressor in the **Swan River**.

Many of the impaired streams in the SLRW exceeded the chronic standard (CS) for aluminum. Impaired streams in the Meadowlands region of the watershed, such as **Stony Creek**, **Sand Creek**, and **Skunk Creek** all exceeded the CS, but the sampling events were all associated with snowmelt and rain events and therefore are not representative of conditions that can be considered chronic exposure. A similar scenario is occurring in other impaired streams in the watershed, including **Water Hen Creek** near Makinen, Minnesota and the two impaired trout streams within the city limits of Duluth, **Kingsbury Creek** and **Miller Creek**.

Due to the limited data available and the complexity of aluminum toxicity, no streams will be evaluated for this parameter as a candidate cause for impairment. Additional data should be collected to further investigate this stressor in the impaired streams mentioned above.

## Iron

At certain concentrations, iron can be toxic to aquatic life. Minnesota does not currently implement a water quality standard for iron that is protective of aquatic life. The EPA recommended a criterion of 1,000 µg/L (1 mg/L) for freshwater aquatic life protection, and numerous other US states also use this concentration as an aquatic life standard. Linton et al (2007) produced a set of bio-assessment-based benchmarks for total iron by identifying the iron concentrations associated with major changes in the structure and function of aquatic communities. The benchmarks derived from this work were 210 µg/L (0.21 mg/L, for no or minimal changes in community structure) and 1,740 µg/L (1.74 m/L, for slight to moderate changes in community structure).

### Sources and Pathways of Iron in Surface Water

Wetlands can play a significant role in the amount of iron available, as well as the species of iron delivered to hydrologically connected waterbodies. Many of the wetlands bordering SLRW streams are sedge meadows, with deep peat soils. The water table is very high in these areas, sitting just a few inches below the soil surface. As the wetland soils warm up in summer, the microbes become active in the saturated peat soils, and oxygen levels become anaerobic. Most areas of the SLRW area have high iron concentrations in groundwater, and the so the peat soils have a lot of iron in them due to the groundwater that is passing through them, on its way to the stream. When the peat areas are aerobic (colder periods of the year), the soluble groundwater iron gets stored in the peat as iron oxides. When these peat areas become anaerobic, the iron converts to Fe2+ (soluble) and is carried with the shallow groundwater to the stream.

### Iron Precipitate

Oxygen concentrations play a significant role in dictating the state of iron in surface and groundwater. If ample oxygen is present, iron oxides develop in the form of a rust colored precipitate (Fe3+). In ground water, low DO concentrations are typically observed, causing available iron to remain as Fe2+ (soluble). When it emerges into the stream and is exposed to more oxygen, it precipitates as iron oxide. Iron-fixing bacteria also can play a role in converting convert Fe2+ to Fe3+. At least 18 different types of bacteria are classified as “iron bacteria,” which are long, thread-like organisms that “feed” on iron and secrete slime as a byproduct. Unlike most bacteria, which feed on organic matter, iron bacteria fulfill their energy requirements by oxidizing ferrous iron ($Fe^{2+}$) into ferric iron ($Fe^{3+}$). Ferric iron ($Fe^{3+}$) is insoluble and precipitates out of the water as a rust colored deposit. The effect of these iron forming bacteria on DO concentrations is not known, but it is possible that oxygen is consumed as iron bacteria form in this stream.

The formation of iron precipitates in surface waters is seen in many areas of northern Minnesota, and occurs naturally in areas with very little anthropogenic influence. The additional loading of iron contributed to the stream through anthropogenic sources (mining, urban areas, WWTP) may contribute additional iron to surface waters. It is very difficult to separate out the natural processes from human influence in this case, considering that iron precipitates are a rather commonly observed “impact” observed around the state in watersheds with a wide range of land-uses.

Iron precipitates have the ability to restrict the distribution, abundance, and diversity of fishes (Dahl 1963) in stream. Some of the observed effects of iron precipitates include, (1) accumulation in fish gills, limiting respiration (Dalzell and MacFarlane 1999); (2) constraining food access by macroinvertebrates (Gerhardt 1992); (3) altering the quality and structure of benthic habitats (Letterman and Mitsch 1978). Smith et al. (1991) found observed an increase in $Fe^{2+}$ concentrations following water transport through a beaver dam.

### Candidate Cause Screening: Iron and Iron Precipitate

Geochemistry samples for a set of cations and anions were collected from all of the biota-impaired streams in the SLRW during snowmelt, rain event, and baseflow conditions. The highest iron concentrations during these sampling events were consistently found in watersheds with a significant amount of wetlands and bogs. The impaired streams included in this grouping include the upper **Embarrass River** and **Wyman Creek** in the extreme northern portion of the SLRW, **Water Hen Creek** near Makinen, Minnesota, **Otter Creek** near Carlton, Minnesota, and several streams in the Sax Zim bog region (**Sand Creek**, **Skunk Creek**, **Stony Creek**, **Paleface Creek**). The highest concentrations were observed in **Wyman Creek** (5,540 µg/L during baseflow) and heavy amounts of iron precipitate were observed at the biological monitoring station during several site visits. No significant iron precipitates were observed at any of the other impaired streams.

Due the high iron concentrations and precipitates observed in **Wyman Creek**, iron will be evaluated as a candidate case for impairment in that watershed. Iron may also be contributing as a stressor in several of the streams in and around the Sax Zim bog area, but natural background influences are the suspected source, and precipitates were not observed in these streams.

## Cadmium

Cadmium is a relatively rare element that is a minor nutrient for plants at low concentrations, but is toxic to aquatic life at concentrations only slightly higher. Cadmium can enter the environment from various anthropogenic sources, such as by-products from zinc refining, coal combustion, mine wastes, electroplating processes, iron and steel production, pigments, fertilizers and pesticides.

The primary mechanism of cadmium toxicity, like other metals, is binding to fish gills and disrupting cation transport channels on the membranes of the gills. It is difficult to measure the toxic form of cadmium because it binds to numerous constituents that depend on site-specific water chemistry. Dissolved cadmium is considered the toxic form. Its bioavailability is primarily dependent on the calcium and magnesium concentrations in the water because these cations compete with the cadmium for binding sites (Monson and Monson 2012).

#### Water Quality Standard: Cadmium

The water quality standards for cadmium are based on the hardness of the water being sampled. As hardness decreases, the cadmium thresholds for CS, MS, and FAV also decrease. The lowest hardness value observed among impaired streams in this study was 40 mg/L (Upper Embarrass River). At a hardness of 40 mg/L, the CS for cadmium is 0.55 µg/L.

### Candidate Cause Screening: Cadmium Toxicity

Cadmium concentrations observed in the biota-impaired streams were nearly all below 0.10 µ/L (75 of 76 results; > 98%), with the only exception being a result of 0.91 µg/L on the Upper Embarrass River in July of 1977. Based on the hardness value at the time of this sample, a concentration of 0.91 µg/L exceeded the CS for cadmium (hardness = 52 mg/L, cadmium CS = 0.67 mg/L). However, the other 17 sampling results for cadmium at this monitoring station rest were all below 0.1 µg/L. It is very likely that the 0.91 µg/L result is the result of a short duration event or sampling error. Based on the monitoring results available, cadmium is not considered a candidate cause for impairment in any of the watershed zones.

## Arsenic

Arsenic (As) is a relatively common element that occurs in air, water, soil, and all living tissues. Organisms are exposed to arsenic through numerous pathways, including atmospheric emissions from smelters, coal-fired power plants, herbicide sprays, water contaminated by mine tailings, and natural mineralization processes. Arsenic bioavailability and toxic properties are significantly modified by numerous biological and abiotic factors that include the physical and chemical forms of arsenic tested, the route of administration, the dose, and the species of animal. Arsenic is bio-concentrated by organisms, but not biomagnified in the food chain.

### Water Quality Standard: Arsenic

The water quality standard for aluminum in is listed in Table 16. The CS and MS listed for class 2A and 2B streams is based on human-health, while the FAV listed for both stream classes is based on toxicity data.

### Candidate Cause Screening: Arsenic Toxicity

Arsenic concentrations in biota-impaired streams were generally below the 2.0 µg/L human-health based chronic water quality standard for class 2A (coldwater) streams. Data from all locations were significantly below the class 2B (warmwater) chronic standard. Relative to the other streams, slightly elevated arsenic concentrations were observed in **Manganika Creek** near Virginia, Minnesota, and a tributary to **Wyman Creek** originating from an inactive ore mining pit. Although concentrations were elevated (5 – 11 µg/L), they were significantly below levels that can be considered harmful to aquatic life. Thus, toxicity from arsenic is not considered a candidate cause in any of the watershed zones with biota impairments.

## Copper

Copper is a common natural element that is found in geologic deposits that include cadmium and zinc as well. According to EPA (EPA 2007) naturally occurring copper ranges from 0.20 to 30 µg/L in freshwater. Copper is associated with various anthropogenic activities, including discharges from mining, leather processing, metal fabrication, and electrical equipment production. Copper is found in municipal wastewater because of the corrosion of copper pipes. Copper sulfate is a common algaecide to treat nuisance algal blooms in lakes and ponds but can also toxic to the zooplankton that graze on the algae.

Copper is an essential nutrient at very low levels, but as it increases in concentration it becomes toxic to animal and plant life by binding to key organic molecules (ligands) and interfering with waste removal from blood. Specific biological effects of copper on fish at non-toxic levels make it useful to model the causal pathway between copper and impairments for fish and invertebrates separately.

Copper interferes with olfaction in fish. Fish can detect copper at relatively low levels, changing behavior to avoid low concentrations. Copper is often used to chase fish in to nets due to the strength of avoidance behavior. This change in behavior reduces feeding, inhibits thermoregulation, and ultimately results in lower growth rates. Copper intoxication can also result in etiological shifts that reduce the growth, reproduction, and survival of fish. Fish eggs are particularly sensitive to copper, with little or no survival of eggs at copper levels that are not harmful to adults.

Finally, because different macroinvertebrates exhibit varying copper tolerances, copper can influence macroinvertebrate species composition as well as directly impacting growth, reproduction, survival, and life cycle phenology. In general, benthic invertebrates are most sensitive to copper accumulation in sediments.

#### Water Quality Standard: Copper

Copper toxicity to aquatic life varies with its bio-availability, which is mediated primarily by pH and hardness. Minnesota’s current water quality standard for copper is based on water hardness and is discussed in greater detail in the MPCA document “Guidance Manual for Assessing the Quality of Minnesota’s Surface Waters for Determination of Impairment: 305 (b) Report and 303(d) List” (see internet link in Table 16).

#### Candidate Cause Screening: Copper Toxicity

Dissolved copper concentrations were calculated for all biota-impaired streams for which adequate data were available (Figure 19). Dissolved copper concentrations were elevated in **East Swan Creek**, but a review of the monitoring results indicated that this stream is narrowly meeting the CS for copper toxicity. The only result exceeding the CS within the SLRW was a sample from the upper **Embarrass River** collected in July of 1977 (diss. Copper = 6.53 µg/L, hardness=52 mg/L). Out of a total of 21 results for dissolved copper from the Embarrass River, only one exceeds the chronic standard. One result from **Kingsbury Creek** collected in May of 2012 during a high flow event was narrowly below the CS.

Current data do not provide any evidence in support of copper as a stressor in any of the impaired streams. Additional data collection in the watersheds mentioned above would be beneficial for eliminating this stressor with a higher level of confidence. Analysis of sediment-based copper concentrations would also be valuable in these watersheds.

## Lead

Lead is a non-essential element for plant, animal, and human nutrition, but is ubiquitous in our environment. Aquatic environments receive led through precipitation, fallout of lead dust, street runoff, and both industrial and municipal wastewater discharges. Generally, the solubility of lead in water decreases with increased alkalinity.

Invertebrate species show varying sensitivities to lead. Amphipods (scuds) were reported by Spehar, et al. (1978) to be more sensitive to lead than any other invertebrate thus far tested. Interestingly, this same relationship existed in longer exposures lasting up to 28 days in which the scud was far more sensitive to lead than a snail, cladoceran, chironomid, mayfly, stonefly, and caddisfly (Spehar, et al. 1978; Biesinger and Christensen 1972; Anderson et al. 1980; and Nehring 1976).

Spinal deformities due to lead were noted in a life-cycle test of three generations of Brook Trout (Holcombe et al. 1976). The chronic values obtained by these investigators were 58 to 119 µg/L Pb (total) in water of hardness 44 mg/L as CaCO3.

### Water Quality Standard: Lead

The water quality standards for lead toxicity are based on dissolved concentrations and water hardness. As hardness decreases, the water quality thresholds for CS, MS, and final acute value (FAV) also decrease. The lowest hardness value observed among impaired streams in this study was 40 mg/L (Upper Embarrass River). At a hardness of 40 mg/L, the CS, MS, and FAV for lead are 5.57 µg/L, 7.48 µg/L, and 14.96 µg/L, respectively.

### Candidate Cause Screening: Lead

All available dissolved lead concentrations were plotted against existing water quality standards to screen for potential stressors related to this metal (Figure 18). A total of 29 paired observations of lead and water hardness were available for five biota impaired streams; **Embarrass River**, **Elbow Creek**, **Kingsbury Creek**, **Manganika Creek**, and **East Swan Creek**. Total lead data were available for several other impaired streams, but there were no paired water hardness measurements for calculating dissolved lead and comparing to water quality standards.

Only 2 out of the 29 (7%) available sampling results exceeded either the CS or MS for dissolved lead. A sample collected from the impaired reach of **Kingsbury Creek** in May of 2012 narrowly exceeded the CS (diss. Lead = 3.18 µg/L, hardness = 55 mg/L). Streamflow conditions during this sampling event were elevated due to a large rain event in the Duluth area. Additional samples collected from this site during low flow conditions resulted in lead concentrations below the CS. A single result from the upper **Embarrass River** exceeded the MS concentration for lead in July of 1977 (diss. Lead = 58.5, hardness = 52 mg/L). Elevated levels of dissolved copper were observed during this same sampling event. The other 19 results for dissolved lead from the upper Embarrass River were all below 1 µg/L. The exceedance of the MS in July of 1977 is an extreme outlier, and is likely the result of either a sampling error, data entry error, or a rare discharge or spill in the watershed.

Lead toxicity is considered a candidate cause for impairment in **Kingsbury Creek** and will be further evaluate in Section 5 of this report.

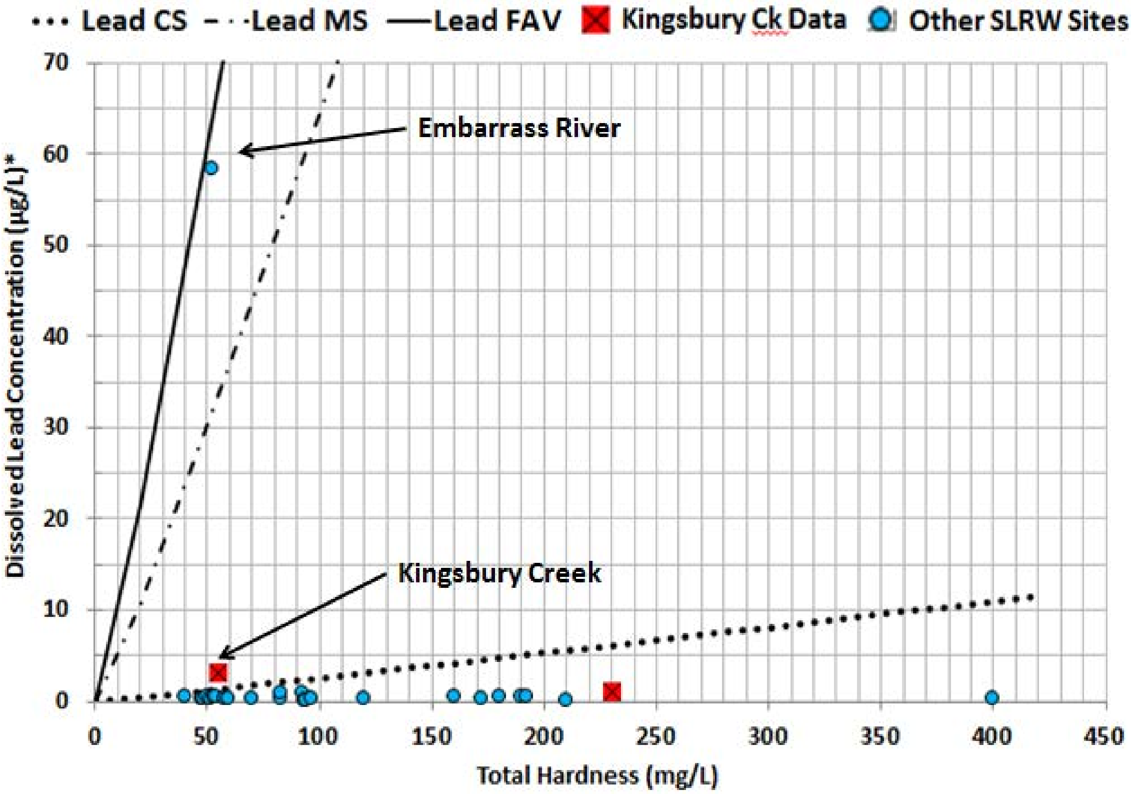


**Figure 18:** Dissolved lead results for SLRW streams with biological impairments

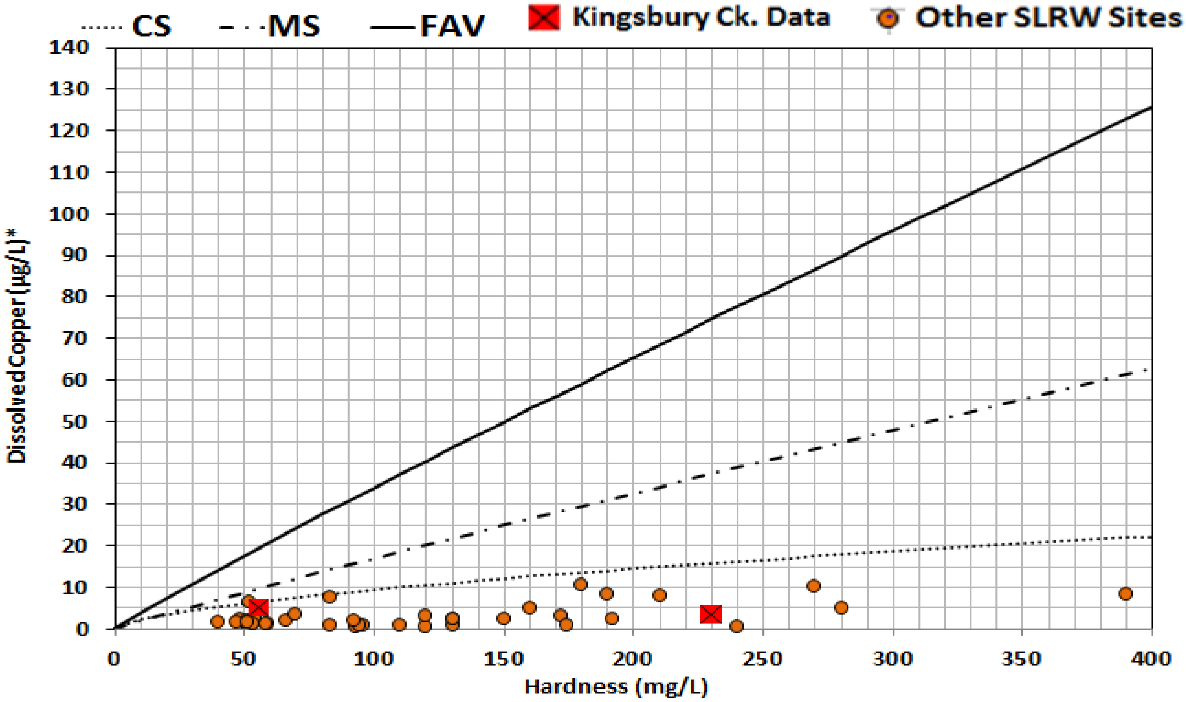


**Figure 19:** Dissolved copper results for SLRW streams with biological impairments

## Zinc

Zinc (Zn) is one of the most commonly occurring heavy metals in natural waters, and is an essential element for most plants and animals. The toxicity of Zn to aquatic life varies widely between species, and is modified by several ambient factors in streams, including water hardness, DO concentration, and temperature. Zinc is acutely toxic to select freshwater organisms at concentrations as low as 90 µg/L (Rabe and Sappington 1970), and the lowest reported chronic effects documented are between 26 and 51 µg/L (Spehar 1976).

### Water Quality Standard: Zinc

The water quality standards for Zn toxicity are based on water hardness. As hardness decreases, the Zn thresholds for CS, MS, and FAV also decrease. The lowest hardness value observed among impaired streams in this study was 40 mg/L (Upper Embarrass River). All other streams evaluated in this study have shown hardness values equal to or greater than 40 mg/L. At a hardness of 40 mg/L, the CS, MS, and FAV for lead are 48.77 µg/L, 53.84 µg/L, and 107.68 µg/L, respectively.

### Candidate Cause Screening: Zinc

Available samplings results for zinc were compiled and evaluated for all impaired SLRW streams. Nearly all of the results were well below the CS at a hardness of 40 mg/L (48.8 µg/L Zn), which represents the softest water sampled among the impaired streams. Zn concentration exceeded 48 µg/L in **East Swan Creek** (near Hibbing, Minnesota) during a single sampling event in August, 1979. However, water hardness is generally much higher in East Swan Creek, which increases the concentration at which Zn becomes toxic. At the time of sampling, hardness was 190 mg/L, which equates to a CS of 182.6 µg/L, nearly four times higher than concentration of this particular sample.

Based on the monitoring results, Zn toxicity is highly unlikely stressor in all of the biota-impaired streams evaluated in this report and can be eliminated as a candidate cause of impairment.

## Nickel, Chromium, and Selenium

Other trace metals that were evaluated as potential candidate stressors include nickel, chromium, and selenium. Concentrations of these trace metals were generally very low in the biota-impaired streams that were focused on as part of this SID study. These metals are not considered a candidate cause for impairment in any of the impaired streams.

# 3.2 Physical Stressors

## 3.2.1 Hydrology

A broad review of watershed hydrology data is presented in this section of the report. The goal is to summarize current conditions and develop a list of candidate stressors related to hydrology or flow alteration. Flow alteration was defined as the change of the stream flow regime caused by anthropogenic sources. Some focus was given to channel geomorphology and physical habitat due to the interconnectedness of flow regime, sediment transport, and channel formation in stream processes.

In general, long-term hydrologic (stream gage) records were scarce in Northern Minnesota compared to the southern half of the state. Continuous water level and discharge records for the biologically impaired reaches within the SLRW were found to be scarce, and short in duration. A Hydrological Simulation Program-Fortran (HSPF) model to simulate flow and load rates was being developed by Tetra Tech during the scope of this study, but had not been completed. A separate Tetra Tech study which combined HSPF model output with groundwater flow model (GFLOW) output to estimate current and native flow conditions in mining area streams was completed in September, 2014. The Upper SLRW Mining Area Hydrology report (Tetra Tech, 2016) was intended as a preliminary assessment of the Minnesota Iron Range hydrology. The groundwater component of the study was a simplified representation of average conditions and was based on limited data. Results for the biologically impaired reaches are highlighted below in the percent mine features section of this report.

Given the lack of available hydrologic data in the SLRW, Geographic information System (GIS) data were used to calculate contributing drainage areas and various flow alteration metrics. The upstream drainage areas were defined for biological impairments, with the respective pour-points defined as the downstream-most point of the impaired reaches (map segments). For drainages that were substantially larger in scale than the length of the impaired segment (St. Louis River, Swan River, Water Hen Creek), both “local” and “cumulative” drainages were defined. The local drainage was defined at the Hydrologic Unit Code (HUC)-12 watershed scale, which had drainage areas between 15 to 62 square-miles. The cumulative drainage was defined as the entire upstream drainage area for that reach. The St. Louis River impairment (AUID 04010201-508), for example, was evaluated at a local scale using the HUC-12 delineation (21 square miles) and cumulative scale that included the entire upstream St. Louis River drainage (1932 square miles). Delineated local and cumulative drainage areas and respective pour points were displayed below in Figures 20 and 21.

A conservative numeric threshold was established for each metric to determine whether the metric was considered a candidate stressor. If results for a given metric were above the threshold or determined to be inconclusive, it was advanced for further analysis. Where stream gage or lake level data was available, it was used to further investigate impairments for a given metric. Where information or data were lacking to make a conclusive candidate cause or pathway determination, a call for additional monitoring or information collection was made. The following metrics were considered:

| Flow Alteration Metrics |
|---|
| **Stream/Road Crossing Density** |
| **Percent Stream Channelization** |
| **Impoundments** |
| **Percent Impervious Surface** |
| **Percent Agriculture (crop, pasture, hay)** |
| **Discharges & Withdrawals** |
| **Percent Mine Features** |

**Table 17:** Landscape/GIS metrics selected for evaluating stressors related to altered hydrology. Percent stream channelization based on MPCA’s Altered Watercourse Layer

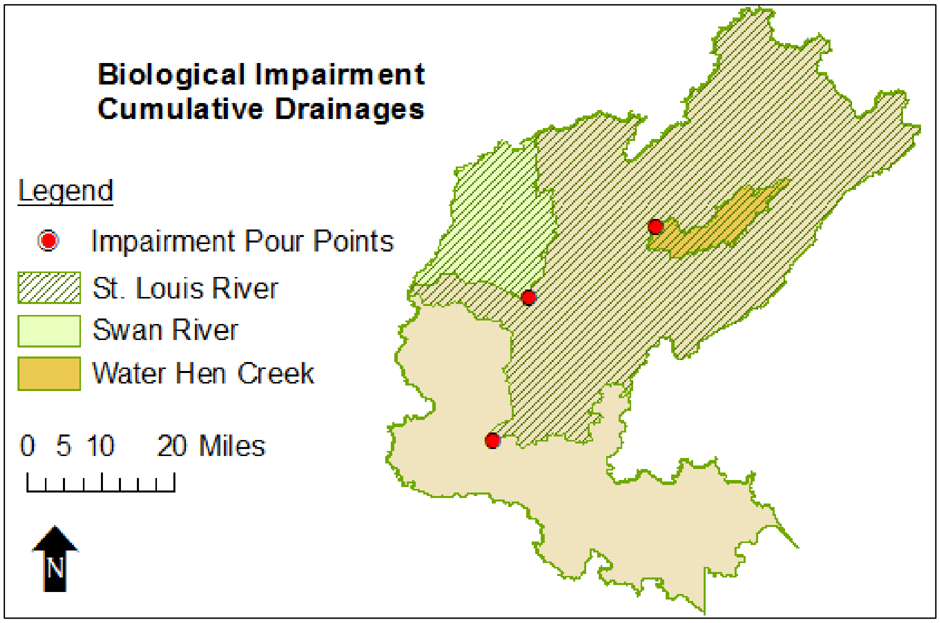


**Figure 20:** Pour point locations for cumulative drainages used in the hydrological stressor analysis

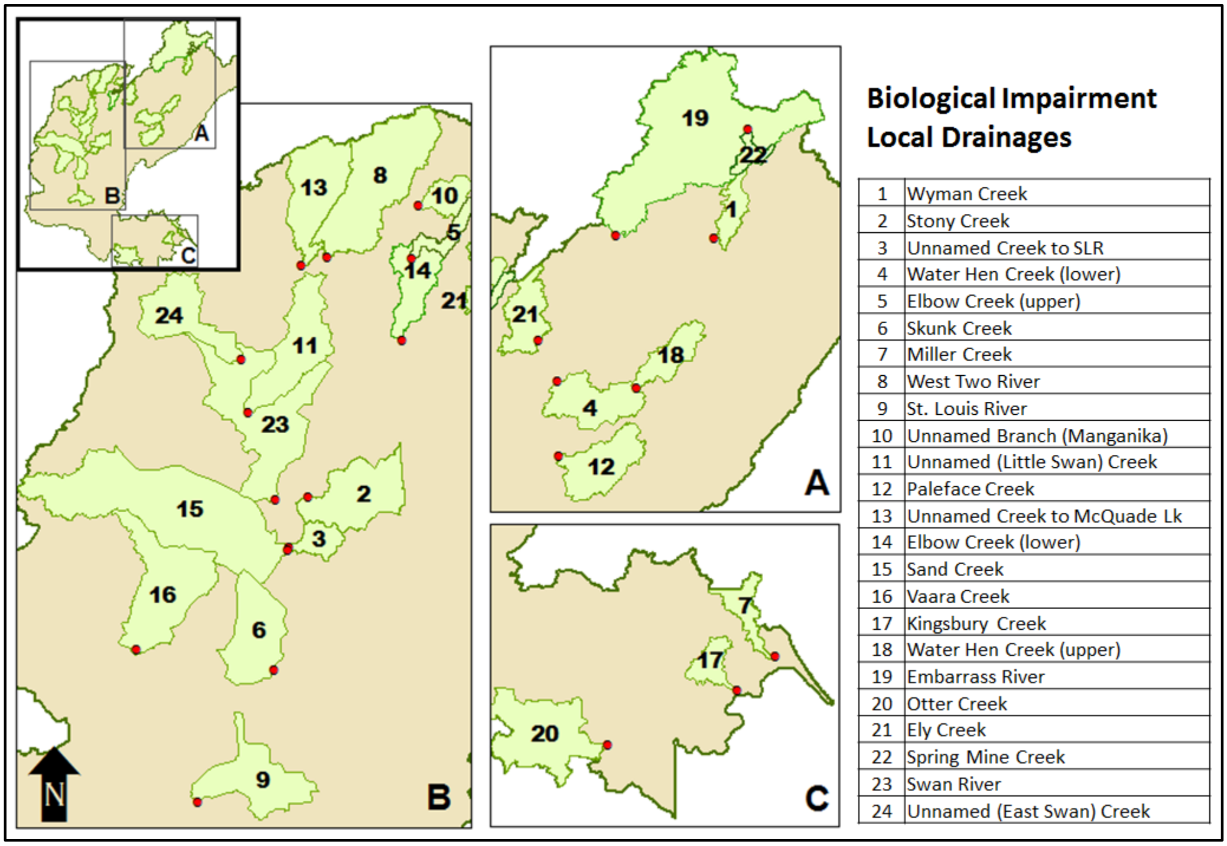


**Figure 21:** Local drainages used for evaluating stressors related to altered hydrology

### Stream/Road Crossing Density

Stream road crossings can change stream flow direction, create fish passage barriers, and alter the overall stream channel geomorphology. Undersized culverts can affect the hydrology of the system by creating temporary impoundments that can alter the hydrology by reducing the peak flows. This reduction can have impacts on the stream system by not allowing for the movement of in stream sediments that would typically be moved at bankfull events or larger floods. Misaligned or improperly sized culverts and bridges can alter the flow regime in a variety of ways: causing scours, over-widened channels, loss of habitat, and localized bank erosion. Culvert crossings more commonly act as barriers to fish movement than bridges. Compared to bridges, culverts typically provide a smaller cross sectional area for channel migration to occur. In addition, culverts that are undersized can cause channel incision on the downstream end, resulting in an outflow that is elevated above the downstream water surface (perched culvert). Perched culverts and increased stream velocities within the culvert can impede fish passage. An excellent review of studies regarding culvert impacts to fish migration, including information specifically from Minnesota, has been conducted by the Minnesota Department of Transportation (MnDOT) (2013).

### Calculating the index

Road Crossing density was defined as the ratio of total number of road-stream crossings to the total length of streams (kilometers) in a watershed. Crossing density was estimated using the intersections of the MnDOT Active Streets GIS layer and the United States Geological Survey's National Hydrography Dataset (NHD) High Resolution stream line work. A threshold of 3.2 crossings per mile (2 crossings per kilometer) of stream length was used based on the findings of Alberti (2007). In the study, a linear relationship between number of road crossings and biological condition in the stream was derived with IBI values approaching poor biological conditions after two crossings per kilometer. A more detailed assessment report, *Connectivity Analysis on SLRW Impaired Streams (CASLRWIS)*, was completed by the South St. Louis Soil and Water Conservation District, found in Appendix C. The assessment accounted for number of bridges and culverts, including the size status of each culvert (undersized, sized correctly, oversized, and undefined) within each biologically impaired AUID.

### Stream/Road Crossing Data Discussion

Based on the watershed scale crossing threshold, an exceedance was observed in the Miller Creek Watershed (3.52 crossings per mile of stream length). Kingsbury Creek Watershed was just below the threshold (2.95 crossings per mile of stream length). The CASLRWIS report found that both Miller and Kingsbury Creeks exceeded the crossing threshold at the more localized AUID scale with respective 4.2 and 4.0 crossings per stream mile. These streams were located in urban landscapes where road densities were high and corresponding stream/road crossings were more abundant.

Miller Creek which was located in an urban environment entirely, from its headwaters to the mouth, had a high crossing density throughout most of the watershed. A 1.5-mile stretch of riparian corridor was established in the vicinity of Lake Superior College, between Anderson Road and Trinity Road, where no road crossings were observed on the main stem and very few were found on the contributing tributaries. The lower 0.6-miles of stream length flowed primarily underground through a culvert system, with a few stream fragments exposed above ground. A total of 36 crossings were identified in the impaired AUID, including 22 culverts. Six of those culverts were undersized, seven were oversized, and 2 undefined resulting in 35% rate in correct culvert sizing. An example of an undersized culvert on Miller Creek is shown below in Figure 22b. The same culvert was also perched during base flow conditions.

Kingsbury Creek, located in a rural/urban environment, also had crossings distributed throughout the entire watershed with a higher density on the main stem than contributing tributaries. The stream segment between Proctor and Duluth had a lower intensity of crossings that the stream miles in the highly developed urban settings. A total of 27 crossings were identified in the impaired AUID, including 13 culverts. One of those culverts was undersized and six were oversized resulting in 46% success rate in correct culvert sizing. An example of a perched culvert on Kingsbury Creek was found at the US Highway 2 crossing. A photograph taken during summer base flow (Figure 22a.) shows little to no flow and inadequate stream depth for fish passage through the culvert during dry periods. This site had a high stream slope which can pose challenges in placement design. Based on bankfull width estimates produced in the CASLRWIS report, the culvert was sized correctly.

Other impairments that were identified as having undersized and/or oversized culverts were Elbow Creek (upper and lower), Skunk Creek, Ely Creek, Little Swan Creek, and Wyman Creek. Detailed crossing locations of crossing type and culvert sizing of all impairments can be found in the CASLRWIS report located in Appendix C.

**Figure 22a. Perched culvert on Kingsbury Creek at US Hwy 2 crossing, acting as a barrier to fish movement.**

**Figure 22b. Culvert on Miller Creek at US Hwy 53 crossing during snowmelt. The culvert becomes perched and acts as barrier to fish passage during the low flow season. The plunge pool downstream is an indication of an undersized culvert.**

**Figure 22c. Miller Creek outflows from an underground culvert that daylights at 26th Avenue West in Duluth, MN. This culvert is not perched, but is representative of the habitat loss present in the urbanized setting.**

**Figure 22d. Miller Creek near Lake Superior College in Duluth, MN with high quality stream corridor. The stream does not intersect roads for 1.5 miles of this segment.**

### Stream Channelization

Ditches can provide important drainage and flood control functions in urban and agricultural landscapes, but ecological services are often lost when previously natural channels become modified for these purposes (Allan, 1995). Schlosser (1982) found that ditched streams experienced a loss of pool habitat, increased organic substrates, and a shift in trophic structure to omnivores and herbivores instead of insectivores and piscivores. In a study conducted in the east-central Indiana Corn Belt region, Lau et al (2006) found that channelized streams had lower quality fish assemblages when compared to natural streams, based on IBI results. In addition, the results of this study showed a reduction in riffle and pool habitats associated with channelization was the most significant factor affecting the fish assemblage. Channel geomorphology, substrate, and in-stream cover were also linked to negative impacts on habitat and associated fish assemblage.

Numerous studies have found conventional trapezoidal ditches to be inferior to natural streams in terms of sediment transport capacity and channel stability over time (Urban and Rhoads, 2003; Landwehr and Rhoads, 2003). Conventional ditches are designed to handle low frequency, high-magnitude flood events. They are generally not designed with a bankfull channel which is optimized for the flow regime of the watershed. This design typically does not support adequate water depth and velocities for transporting sediment and maintaining stream facets (e.g. glide, riffle, run, pool) during more frequent, lower magnitude high flow events channel forming events. The result can be excess sedimentation of the stream bed as particles become immobile and aggrade over time. An opposite affect can occur when stream length is reduced as a result of sinuosity loss during channel straightening. This can lead to increased stream slope gradient and associated increased water velocities, stream power, and erosion hazard.

### Calculating the index

Percent stream channelization was estimated using the Statewide Altered Watercourse (AWC) GIS layer (released in 2013) which consists of a statewide inventory of streams that have altered hydrology (e.g. channelized and impounded). Visual interpretation of multiple years of aerial photography, LiDAR (Light Detection and Ranging) derived “hillshade” imagery, and various other reference data in ArcGIS 10.0 were completed by GIS technicians for the United States Geological Survey’s National Hydrography Dataset (NHD) stream line work in development of the AWC layer. These data were developed by the MPCA and Minnesota Geospatial Information Office (MnGeo).

Using the AWC layer, channelized stream miles, total stream miles, and the ratio of channelized to total stream miles (percent channelized stream miles) were quantified for the AUID length of each impairment and then for the entire stream network within the drainage area. Thresholds of greater than 20% impaired reach length and 40% total watershed stream length were used. Thresholds were based on best professional judgment. A lower threshold was applied to the reach scale to capture affects that may be occurring on a more localized, micro-habitat scale (e.g. stream cover, poor riffle/pool quality, and other habitat loss). If either of these thresholds were triggered, the associated impairment was evaluated in more detail for channelization as a candidate cause.

### Channelization Data Discussion

Channelization threshold exceedances were observed in the six of the nine watershed zones, suggesting that ditching of streams has been a common practice across the SLRW. Nine impaired streams exceeded the watershed-scale threshold of 40% (Table 18). At least four and possibly five impaired streams exceeded the reach-scale threshold of 20%. Kingsbury Creek, Unnamed Creek to SLR, and Skunk Creek exceeded both thresholds. Heavy historical ditching in the Meadowlands Floodwood Peat Bog was apparent, with five impairments in this zone exceeding the watershed-scale threshold. Watershed channelization values for Skunk Creek (95.4%), Unnnamed Creek to SLR (85.1%), and tributaries to St. Louis River-local (62.9%) were exceptionally high. The Kingsbury Creek reach-scale exceedance was also exceptionally high (greater than three times the 20% threshold). There was some uncertainty whether certain sections of the Embarrass River had been altered, resulting in ambiguity of exceedance.

**Table 18: Summary of channelization metrics by impairment and watershed zone. Shaded cell indicate an exceedance of a channelization thresholds at a watershed scale (threshold, >60%) and/or reach scale (threshold, >20%). Uncertainty of channelization on the Embarrass River results in a range of values for percent impaired reach length*.**

| Impairment | Watershed Zone | % Channelization in Drainage Network | % Channelization in AUID (Reach) |
|---|---|---|---|
| Miller Creek | DUC | 28.8 | 24.5 |
| Kingsbury Creek | DUC | 49.2 | 75.4 |
| Unnamed Creek to SLR | MF-PB | 85.1 | 20.5 |
| Skunk Creek | MF-PB | 95.4 | 20.7 |
| St. Louis River - local | MF-PB | 62.9 | 0 |
| Stony Creek | MF-PB | 42.5 | 0 |
| Sand Creek | MF-PB | 44.6 | 0 |
| Embarrass River | NU-EMB | 32.9 | *5.0-38.8% |
| Water Hen Creek (upper) | ML | 45.3 | 0 |
| Manganika Creek | VIR | 55.21 | 0 |
| East Swan Creek | SWH | 56.46 | 6.09 |

A good portion of the ditching in the Meadowlands Floodwood Peat Bog zone was done at the turn of the 20th century as an attempt to drain the land for agricultural use. **Skunk Creek, Unnamed Creek to St. Louis River, Stony Creek, Sand Creek** and the local HUC-12 drainage for the St. Louis River impairment fell into this category. **Skunk Creek** alterations were the most prevalent in a high stream density network of ditching. Field observations described the reach as flashy, exhibiting high velocities and levels during precipitation events and extremely low flow during dry periods. This flow regime was atypical for a wetland environment and was likely the result of the grid of straight channels that most efficiently drain water off of the landscape. This straight channel grid-like pattern was also observed in the other four MF-PB impairments listed above at a lower density. The main stem **St. Louis River** impairment was a natural channel, but received water from multiple ditched, straight channel tributaries. The main stem was large enough in size at the impairment pour-point (1,932 square mile drainage area) that it may have been affected less from the wetland drainage in its HUC-12 delineation than the smaller systems.

**Upper Water Hen Creek** and **East Swan Creek** were also channelized for agricultural use. East Swan Creek channelization occurred in both urban and pasture/hay land use settings, whereas the Water Hen Creek drainage had pasture/hay as the dominant land use. Both watersheds were altered for agricultural drainage, but the channels did not resemble the severe straight-ditch grid pattern that was observed in the MF-PB impairments.

**Miller** and **Kingsbury Creeks** were likely channelized to make room for development of railways, roads, parking lots, buildings, other urban infrastructure, and historically for agriculture. The channel modification on the urban streams was primarily done on the main stem reaches. Kingsbury alterations were located in Midway Township, city of Proctor, and city of Duluth. Miller Creek alterations were located in city of Hermantown and city of Duluth. Unlike Kingsbury Watershed where the majority of the upper and middle main stem Creek was channelized, main stem Miller Creek had intermittent sections throughout the watershed that had not been channelized. Channelization in the lower main stem of both creeks was limited by bedrock geology and steep slopes. As discussed above in the Percent Stream

Channelization section of this report, a segment of Miller Creek, near Lake Superior College, featured good riparian corridor. A 5-mile stretch of stream in this vicinity was identified as natural channel (no channelization).

The **Embarrass River** was rerouted in the vicinity of Embarrass and Wynne lakes in the mid-1900s for iron ore mining. Prior to this, the river flowed from Wynne Lake outlet to the east end of Embarrass Lake. Between the two lakes, the rivers natural path was through a previous natural lake that was in the location of the existing Embarrass Mine Pit (DNR Lake # 69-0429-00). According to the AWC designations, sections of the stream were also straightened upstream of Wynne and Sabin lakes, contributing to the AUID threshold exceedance of 20%. Further analysis of historical aerial photos from Minnesota DNR Landview suggested some level of uncertainty of whether these upper watershed sections were channelized for agricultural drainage at the turn of the Century or whether a lesser sinuosity observed was a natural channel feature. If the questionable sections were a natural channel feature, the Embarrass would not have exceeded the threshold.

Also within the Embarrass R. drainage, there was a diked wetland complex located approximately two miles southwest of the town of Embarrass. The complex was originally developed as a commercial wild rice paddy and is now managed by the DNR as a Wildlife Management Area (Darwin S. Myers WMA). Native plant communities have been impacted in this area and are now dominated by nonnative-invasive Reed Canary grass. It was difficult to speculate the severity of impact on the stream environment, but possible impacts included a change in stream flow rate of rise, peak magnitude, and rainfall release time from channel straightening as well as bank instability and habitat degradation from vegetation change. Future stream flow and geomorphology data collection in this area could help quantify the magnitude. Field observations at the biological monitoring sites outside of this complex suggested that stream sinuosity and bank conditions in most of the upper Embarrass reach were adequate and the associated stream habitat and channel stability were not stressed.

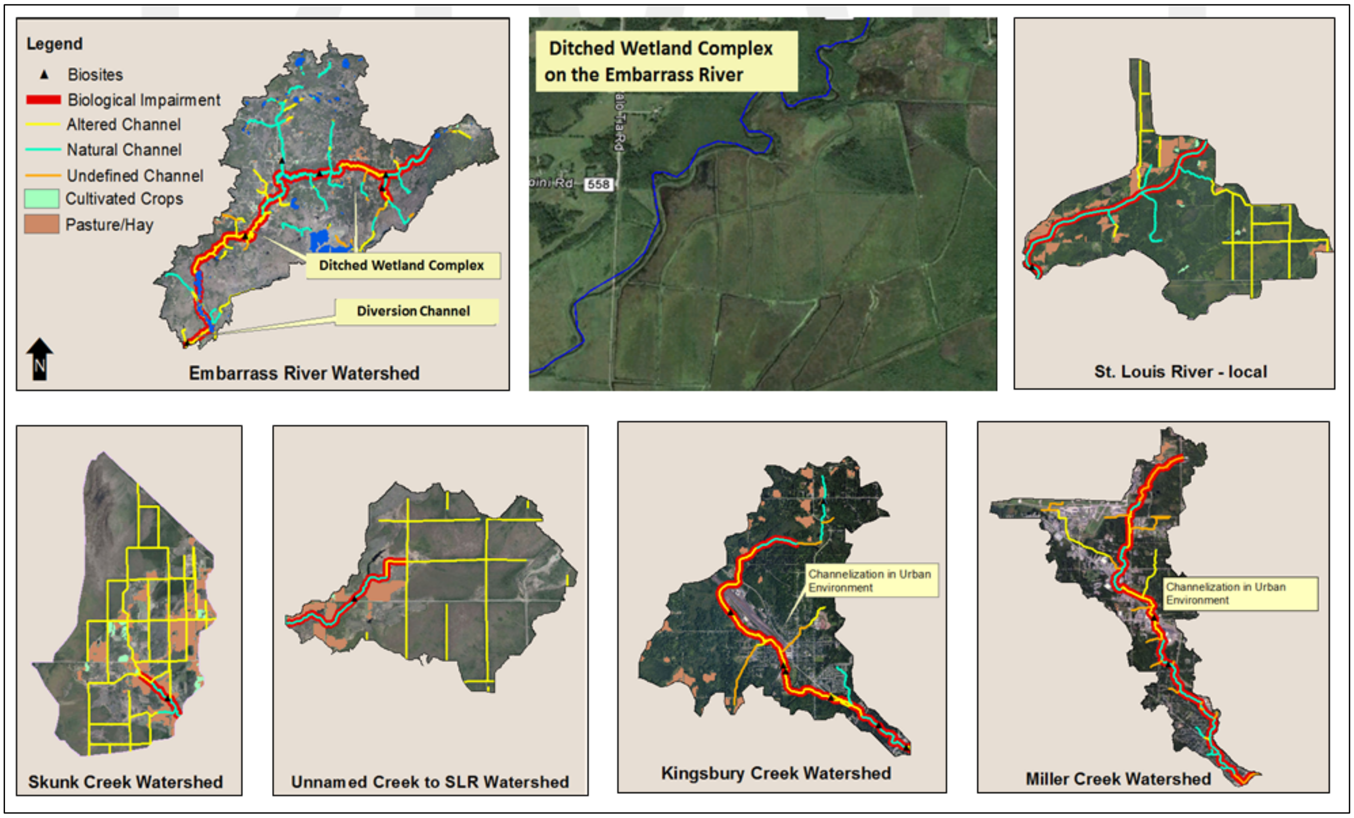


**Figure 23: Watersheds that exceeded channelization thresholds. Images were created at varying scales.**

### Impoundments

In Minnesota, there are more than 800 dams on streams and rivers for a variety of purposes, including flood control, maintenance of lake levels, wildlife habitat, and hydroelectric power generation. Dams change stream habitat by altering streamflow, water temperature, and sediment transport (Cummins 1979; Waters 1995). Dams also directly block fish migration. Both mechanisms can cause changes in fish and macroinvertebrate communities and greatly reduce or even extirpate local populations (Brooker 1981; Tiemann et al. 2004).

The DNR has conducted numerous dam removal projects in recent years which have demonstrated benefits to fish populations. A more detailed presentation of the effects of dams on water quality and biological communities can be found in the MDNR publication “Reconnecting Rivers: Natural Channel Design in Dam Removals and Fish Passage” (Aadland 2010).

### Calculating the index

All impoundments with records in the Minnesota Dams GIS layer and within the impairment drainages were inventoried. The Minnesota Dams layer was created in 2011 by the DNR Dam Safety program and includes dams both under the program’s jurisdiction and not. Impoundments that were located off-stream from any discernable channel were identified, but were not evaluated in further detail. In all cases, the off-stream dams within the impaired watersheds were located on tailings basins that either do not or rarely discharge water directly to adjacent streams. The majority of wastewater from tailings basins is reused in mine processing and lost to evaporation with a lesser and unknown amount lost to surface water and groundwater seepage.

A numeric threshold did not apply to the impoundment variable. Instead the function, structure, and management of individual on-stream (located on discernable channels) impoundments were discussed where the information was available. More detail was given to impoundments with stream or lake water level data. The hydrologic impacts of the flow attenuation of these impoundments can be obtained from future HSPF model results and flow records from the impoundment management authority if it exists.

### Impoundment Data Discussion

The SLRW M&A Report identified 37 dams in the watershed that were recognized by Minnesota DNR Division of Waters. Five hydroelectric dams in the St. Louis Watershed were located on the steep down-gradient section on the lower reaches of the St. Louis River. None of the hydroelectric dams were located within drainages that have identified biological impairments as a result of the MPCA’s assessment process. Thirteen impoundments were located in drainages with identified biological impairments. Four of the thirteen were considered “on-stream” and are identified below in Figure 24. They were the Buhl Detention (**Kinney Creek**), Wynne Lake Dam (**Embarrass River**), West Two Rivers Reservoir (**West Two River**), and Ely Lake Dam (**Ely Creek**).

### Buhl Detention

The Buhl Detention, owned and maintained by DNR Wildlife, was an earthen berm with a concrete drop structure. Total dam height was 11-feet., impounding approximately 3 square miles of wetland drainage and creating a 60-acre normal surface area reservoir. The structure was built in 1988 on an unnamed tributary to **Kinney (McQuade) Creek** to enhance duck habitat. The structure was located approximately a quarter of a mile upstream from the confluence with Kinney Creek which was impaired for invertebrates. In-channel flow and downstream bank stability just downstream of the dam were observed during site reconnaissance in September 2014, the normal dry season. Total impounded drainage area was approximately 14% of the total Kinney Creek impairment upstream drainage area. Based on field observations, the impoundment did not appear to be contributing flow regime related stress to the reach. Potential indirect impacts on downstream water chemistry such as DO were not quantified due to absence of water chemistry data at the impoundment. Future water chemistry monitoring could help quantify the impact, if any exists. For additional information on DO in Kinney Creek, see section 5.21 of this report.

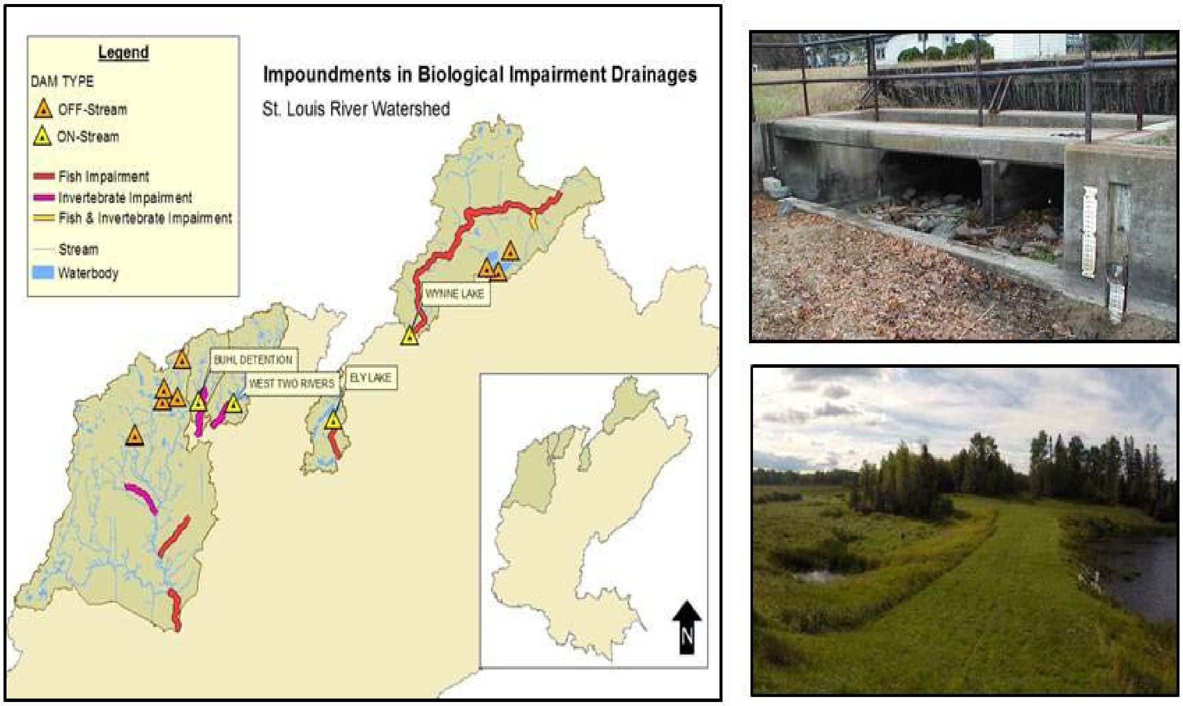


**Figure 24: (Left)** Impoundments within watershed boundaries of biologically impaired reach pour-points. **(Upper Right)** Ely Lake outlet and water level gage during late summer dry period in 2006, with no observed outflow to Ely Creek and the water level gage out of water. **(Bottom Right)** Buhl Detention with reservoir on right and stable downstream channel through wetlands on left

### Embarrass Dam at Wynne Lake

Embarrass Dam at Wynne Lake was a combination stop log and concrete crest weir (set at 5 feet above stream bottom) structure that was built in 1944 to re-route the stream. It was built in the diversion channel located between Wynne and Embarrass Lakes. The structure was State-owned, but was not being managed (stop logs were not in place). Aerial imagery suggested that little to no water was impounded behind the structure. The three biological stations that contributed to the fish impairment on 04010201-579 (**Embarrass River** Headwaters to Embarrass Lake) were located upstream of Wynne Lake. The levels at upstream Wynne Lake were controlled by the lake outlet rather than the downstream Wynne Lake Dam therefore the dam would only have a hydrological influence on the diversion channel below Wynne Lake. The impoundment likely was not causing stress to the system upstream of the lake.

### West Two River Reservoir Dam

West Two River Reservoir Dam was a 40 foot structure with an approximate 20-foot drop from the crest to the river below. It was built in 1966 to supply the Minntac taconite processing facility with water. U.S. Steel was permitted to take up to 6000 gal/min (13cfs) and to maintain a minimum discharge release of 3cfs (through a low flow drain at the base) to the **West Two River**. Extensive pumping took place in the 1970s to 1980s and then slowed to a near halt until 2012 when increased pumping began again, to over 500 MGY. The reservoir drained upstream forest, wetland, and mining landscapes through multiple inlets. It was designated impaired for recreational use due to exceedance of the eutrophication criteria in 2012.

There were no known stream flow records on the West Two River between the reservoir outlet and the confluence with **Kinney Creek**. The closest downstream gage on the West Two River was located 7.5 miles downstream of the reservoir and 3.5 miles downstream of the confluence with Kinney Creek. Median daily flows were compared for time periods before and after the construction of the dam. Datasets were from periods 1954 to 1963 and from 1971 to 1979. Statistical analysis was completed for pre-dam and post-dam median daily flows using the two-sample non-parametric Kolmogorov-Smirnov test for the water year (Jan 1 – Dec 31). Results indicated a significant difference with a p-value = 0.000.

High and low flow portions of the hydrograph were examined in Figure 25 below. The snowmelt or spring flow peak was greater in magnitude and shorter in duration during the post dam period. This change in flow regime was the opposite of the expected outcome of an impoundment placement. More typically, impoundments decrease the flashiness of a system by holding back water and then slowly releasing it over time. This data indicated that the West Two River Reservoir (post-dam) undergoes a quick release of water towards the end of the snowmelt period.

Flows during the low flow period (late summer - early winter) for post-dam (pumping) years were below pre-dam rates 70% of the time. During July through October they were below pre-dam rates 90% of the time. Post-dam flows were higher during the month of December. It is possible that the dam's minimum discharge release rate of 3cfs kept flows lower during the summer and higher in December than pre-dam rates. It was difficult to speculate the degree of impact on the aquatic biology that resulted from flow alteration. Based on the above flow analysis, flashier hydrology during snowmelt and decreased flows during late summer to early winter months could not be eliminated as a candidate causes for the impairment. It was determined that additional flow data or local HSPF modeling was needed at a location below the reservoir and above the confluence with Kinney Creek. The **West Two River** below the confluence with **Kinney Creek** was in full support of fish and MIBI criteria, even with the reduced summer to winter flows. Indirect impacts of the impoundment on downstream water chemistry such as DO can be found in section 5.20 and 5.21 this report.

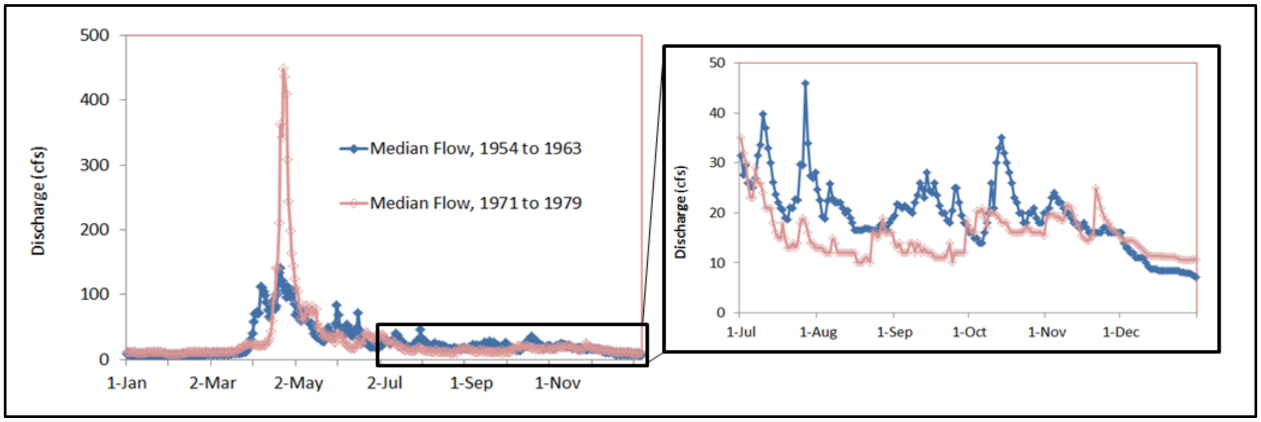
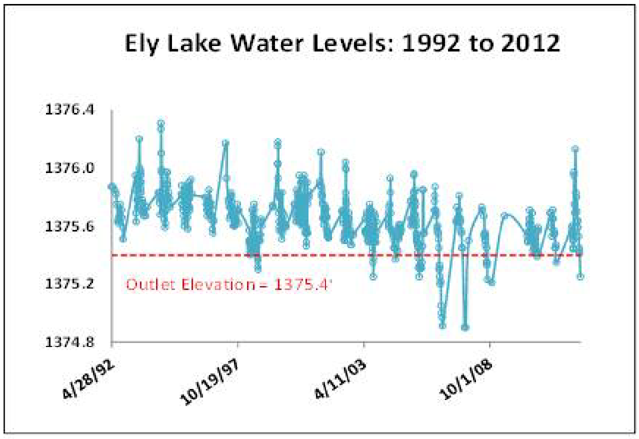


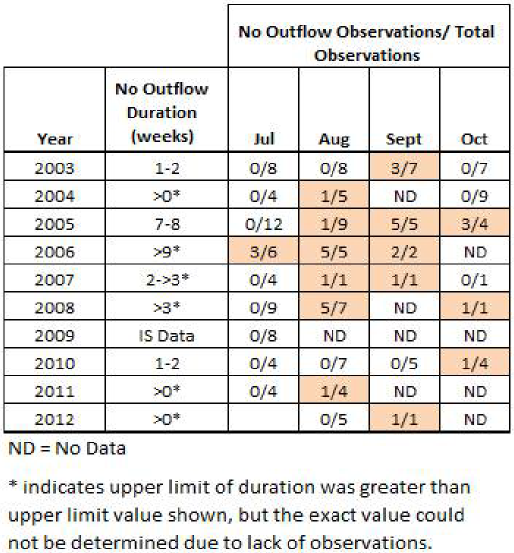
**Figure 25:** Flow comparison analysis of pre-dam (1954-1963) and post-dam/pumping (1971-1979) time periods at USGS stream gage 04019000, West Two River near Iron Junction, MN. The datasets were significantly different (p-value = 0.000). A higher flow magnitude and shorter peak duration were observed during snowmelt. Flows were lower during pumping years than pre-dam years during the majority of the low flow season.

## Ely Lake Dam

Ely Lake Dam was originally a stop log structure built in 1939 with Works Projects Administration funding at the outlet of Ely Lake (DNR Lake #69-0660-00). In 1979, it was reconstructed as a fixed crest concrete sill dam with the control set at 3.0 feet above the sill and a crest elevation equal to 1375.4 feet mean sea level. In addition to the dam, a managed aqueduct on the west end of the lake diverts water to St. Mary's Lake (DNR Lake #: 69-0651-00) according to an Aqueduct Management Plan when levels in St. Mary's Lake are lower than Ely Lake as long as Ely Lake elevations are above a specified elevation. St. Mary's Lake has a permitted withdrawal and is the water supply source to the city of Eveleth. The city of Eveleth is permitted to pump 291 million gallons per year. Private residential wells also scatter the perimeter of Ely Lake. The upstream (Ely Lake – St. Mary's Lake) lakesheds account for approximately 40% of the total **Ely Creek** drainage area.

Ely Lake levels were available from two time periods, 1939 to 1983 and 1992 to date. Level data from the latter were displayed below in Figure 26 and the raw lake level data was summarized in Table 19. The data showed that lake levels began dropping below the outflow elevation annually, beginning in 2003. During periods when levels fell below the dam crest, little to no outflow of lake water was supplied to **Ely Creek**. The approximate duration of weeks for each summer in which levels were too low for outflow was approximated using the data available. The exact duration of weeks could not be determined for every year due to lack of field observations or gaps in lake level readings. Gaps in lake level data occurred most frequently during dry periods, likely because the levels were below the bottom of the water level gage (Figure 24, upper right). Timeframes of a minimum of seven and nine weeks below outflow elevation were recorded in 2005 and 2006, respectively. The maximum depths below outflow elevation (-0.49 to -0.50 feet) were recorded in years 2006 and 2007. The area faced extreme drought conditions in the late summer of 2006 through the end of winter. Year 2007 was more typical of a normal precipitation year, but levels may not have recovered from year 2006.





| | | No Outflow Observations/ Total Observations | | | |
|---|---|---|---|---|---|
| Year | No Outflow Duration (weeks) | Jul | Aug | Sept | Oct |
| 2003 | 1-2 | 0/8 | 0/8 | 3/7 | 0/7 |
| 2004 | >0* | 0/4 | 1/5 | ND | 0/9 |
| 2005 | 7-8 | 0/12 | 1/9 | 5/5 | 3/4 |
| 2006 | >9* | 3/6 | 5/5 | 2/2 | ND |
| 2007 | 2->3* | 0/4 | 1/1 | 1/1 | 0/1 |
| 2008 | >3* | 0/9 | 5/7 | ND | 1/1 |
| 2009 | IS Data | 0/8 | ND | ND | ND |
| 2010 | 1-2 | 0/4 | 0/7 | 0/5 | 1/4 |
| 2011 | >0* | 0/4 | 1/4 | ND | ND |
| 2012 | >0* | | 0/5 | 1/1 | ND |

ND = No Data

* indicates upper limit of duration was greater than upper limit value shown, but the exact value could not be determined due to lack of observations.

**Figure 26 (Top Left): Ely Lake Water Levels were below the lake outflow elevation (no outflow to Ely Creek) annually since 2003 during the late summer. Table 19 (Top Right): Summary of Ely Lake elevations 2003-2012. In 2009 there was no data available for August through October. * signifies uncertainty in exact duration due to gaps in observation data, likely due to levels too low for a staff gage reading. *ND* signifies No Data.**

The city of Eveleth pumping rates were within permit limitations during the scope of this dataset. Pumping rates from 1988 to 2002 were compared with rates after 2003 through 2013, when an annual drop in levels below outlet elevation was observed. An increase in pump rate median values, from 223 million gallons per year to 246 million gallons per year was observed.

Based on the above lake level analysis, intermittent or ceased outflow to **Ely Creek** during late summer to early fall months could not be confirmed or eliminated as a candidate cause for the downstream impairment. Low DO in the late summer months, particularly in low precipitation years may be connected to low flow issues in the stream. Detailed DO analysis for Ely Creek can be found in section 5.16 of this report. Additional flow data or model output is needed to determine the annual and seasonal impacts to Ely Creek due to flow intermittency or cessation at the Ely Lake outlet. The HSPF model that is being developed coincidental to this report may provide the additional context needed to make a determination on low flow as a stressor and, if stressed, help identify the primary pathways.

### Percent Impervious Surface

Impervious surfaces can be artificial man-made structures or compacted soil surfaces in which water cannot infiltrate. Often an increase in impervious surfaces is associated with urbanization. Increased surface water run-off from the landscape to water bodies during precipitation events and lack of infiltration and soil storage of water in the surrounding floodplain to be released during drier periods can lead to an altered flow regime, which can impact the aquatic life within that system. The Minnesota DNR accurately explains the processes in more detail below:

“In many natural systems, most of the rainfall infiltrates into the soil, and then slowly moves through saturated soil to streams, springs, or aquifers. This infiltration serves to buffer the speed of stream flow increases during large rain events, and meter out the subsurface flow during dry periods (Ziemer and Lisle 1998; Poff et al. 2006).

In contrast, impervious surfaces prevent infiltration, lead to increased water flow on the surface of the land, and much more rapid flow into streams. Peak floods become higher and dry-season flows lower, as streams become “flashier” (Schoonover et al. 2005). High flows degrade stream channels by incision or downcutting, increase scour and increase the amount of sediment carried by the stream (White and Greer 2006). High flows also lead to greater bank erosion, as well as increased upland gully erosion, particularly at pipe outlets. Organic and inorganic pollutants are deposited by vehicles and from the atmosphere onto impervious surfaces where surface runoff often carries them directly to streams. These various impacts lead to subsequent changes in the diversity and abundance of stream flora and fauna (Wang et al. 1997; Wang and Kahehl 2003; Lyons 2006; Schiff and Benoit 2007).”

—http://www.dnr.state.mn.us/whaf/about/scores/hydrology/impervious.html

### Calculating the index

Percent impervious cover was estimated using a classification of Landsat satellite images, year 2000. These data were developed by the University of Minnesota Environmental Remote Sensing and Geospatial Analysis Lab. The amount of impervious surface was quantified for each biological impairment drainage area. A threshold of 4% and 10% impervious was used for coldwater and warmwater streams respectively. Studies have shown that once imperviousness reaches 10%, the stream begins to widen through bank erosion and resulting aggradation and valuable habitat is lost (Scheuler 1995). This value for percent impervious cover has been accepted by Nonpoint Education for Municipal Officials (NEMO) as the threshold in which water quality is degraded. At 20-25% imperviousness, the system can be degraded to the point where it is no longer suitable for supporting aquatic life (Center for Watershed Protection 2003; Wang et al 1997). Other studies have shown that sensitive coldwater species are impacted at imperviousness of 4% (EPA 2012. Stranko et al 2008; Wenger et al. 2008).

### Impervious Data Discussion

Percent impervious surface results had a wide range of values in the SLRW. The range for the impaired reach drainages was 0.25% (**Vaara Creek**) to 19.51% (**Miller Creek**). All impaired reach drainages that exceeded the thresholds were located within or downstream of an urban center. The coldwater (trout stream) reaches that exceeded the 4% threshold were **Miller Creek** (19.51%), **Kingsbury Creek** (9.5%) and **East Swan Creek** (7.55%). The watersheds were located fully or partially within the city limits of respective communities: Duluth-Hermantown, Hermantown-Proctor-Duluth, and Hibbing. Upper **Elbow Creek** (11.76%), located at the southern edge of Eveleth, was the one warmwater reach that exceeded the 10% threshold.

**Miller Creek** Watershed had 39% urban land use value (lakesuperiorstreams.org). Between the headwaters and the mouth at Lake Superior, the stream bordered or intersected a shopping district, airport, golf course, college, and residential neighborhoods. Miller Creek had the highest percent impervious surface of all 86 HUC-12 drainages in the SLRW and approached 20% imperviousness, a threshold shown to be associated with non-support of aquatic life (Center for Watershed Protection, 2003). The NRRI calculated a value of 22% using a similar method (Lakesuperiorstreams 2009) which would put it over the higher risk threshold. A 1.6-mile stretch of riparian corridor and resulting lesser percent impervious surface area was established in the vicinity of Lake Superior College between Chambersburg Avenue and Trinity Road.

**Miller Creek** and **Kingsbury Creek** were acknowledged as “flashy” hydrologic systems. Sections of stream with steep gradient, shallow depth-to-bedrock, and high percent impervious surface were likely contributing factors. Flow data from Miller Creek, year 2010, demonstrated a steep rate of rise from 2.5cfs to 177cfs in less than a day and a rapid rainfall release time (flows back to 2.8cfs) of five days (Figure 27). A similar flashy event was also recorded in late October of 2010.

A flow analysis study (Herb & Stefan 2009) on **Miller Creek** indicated flows as low as 1 to 2cfs were determined to be common at weekly time scales in the lower portion of the watershed and flows less than 0.1cfs were possible with a 10- year return period. Due to the common nature of lower flows, it was concluded that a rainfall of moderate magnitude could significantly impact stream flow and temperature. It was also concluded that a large fraction of the flow in **Miller Creek** originated from the upper portion of the watershed, possibly in an undeveloped pocket in the northwest corner of the watershed in which some wetlands remain intact. Although high flow precipitation events and low summer base flows are typical for area streams due to the regional geology, the flashiness of the **Miller Creek** hydrology is compounded by high percent impervious surfaces.

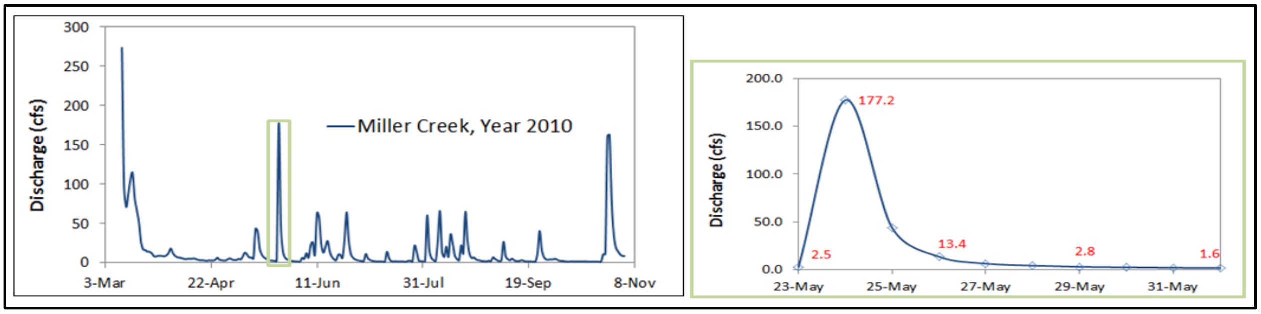


**Figure 27: Hydrograph showing "flashy" hydrology observed in Miller Creek flow data, 2010.**

The metric results for **Kingsbury Creek** (9.5%) were slightly below the 10% threshold for percent impervious and did not meet the 4% threshold for coldwater streams. Development in the watershed and adjacent to stream channel corridors included a rail yard, dirt track motor speedway, city maintenance and stockpile/storage yard, golf course, recreational fields, parking lots, residential neighborhoods, and a city zoo. The majority of the development appeared to be located adjacent to the stream channel or riparian corridor. Areas of lesser development were found in Midway Township, the northeast corner of Proctor city limits, and the upper reaches of the stream found within Hermantown city limits.

**East Swan Creek** exceeded the coldwater threshold with 7.55% impervious area. The majority of impervious development in the watershed was located in the city of Hibbing which is located in the headwaters of the creek and several of its unnamed tributaries. In the eastern watershed, it appeared that much of the stream was fed by stormwater runoff from the city of Hibbing through direct runoff or pooling in seasonally flooded dispersed wetlands. The development in this drainage area consists of residential housing, a shopping/business district, and overburden stockpiles from a nearby mine. Roadways, parking areas, and rooftops account for most of the impervious surface area. In the western part of the watershed, much of the headwaters were in pasture/hay land use which have low impervious surface values. At least one of the tributaries in the western part of the watershed receives drainage from nearby parking lots and shopping centers. Less development occurred south of Hibbing, although there were areas of impervious surfaces in rural residential and farms, subdivisions, and a waste water treatment plant just upstream of the impairment and near Townline Road.

**Upper Elbow Creek** (04010201-518) did not meet the warmwater threshold, with 11.75% of its watershed categorized as impervious area. **Lower Elbow Creek** (04010201-570) drainage which includes the upper drainage did meet the threshold with 5.41%. Elbow Creek had little to no wetlands upstream of 18th Avenue in the city of Eveleth, located just south of EVTAC's south pit. A small wetland area that was contiguous with an industrial park was identified in its upmost headwaters near Mud Lake. The main channel became defined near Highway 53 in Eveleth and then continued in route, passing through a cemetery, park, residential neighborhoods, and into an underground culvert that day-lighted at a wastewater treatment plant (WWTP). Most of the riparian was considered low to medium intensity development which had the potential to cause sheet flow runoff to the stream. The stream continued in route between iron mine pits (no runoff to stream) and mine waste stockpiles. Between 18th Avenue and Elbow Lake, the impervious development was dispersed in the form of a few housing developments.

In this reach, there was a notable increase in wetlands and forested areas. Most of the stormwater runoff and higher impervious values occurred within the city of Eveleth.

### Percent Agricultural Land

Land in agriculture has the potential to increase flow volume during precipitation events, decrease flow volume during dry periods, and alter the stream chemistry through nutrient, sediment, and other anthropogenic chemical inputs (pesticides, herbicides, fertilizers). Surface water runoff from agricultural land can both contribute to high and low flow alterations. Land-use practices that can contribute to the problem include the initial removal of native vegetation, farming steep slopes and long field lengths, removing crop residue, deep tillage, soil compaction, and the overall degradation of soil structure. Rill and gully erosion can be a problem in areas of high surface water runoff and poor soil quality. Lack of stream cover, bank erosion, increased sediment inputs, change in channel geomorphology, and overall loss of habitat are some of the indirect biological and chemical impacts on waterways. In addition to increased stream flow and surface water runoff, bank erosion can be accentuated through uncontrolled cattle access and removal of riparian vegetation.

### Calculating the index

Percent agriculture was estimated using the NLCD, 2006 GIS layer. The NLCD products were created through a partnership of federal agencies, the Multi-Resolution Land Characteristics (MRLC) Consortium. A study on over 100 Wisconsin streams (Wang et al. 1997) compared watershed land use to habitat quality and to biotic integrity. In regards to agricultural land use, the study found that an obvious decline in habitat and biotic integrity was only apparent when the threshold of 50% land in agricultural use was exceeded. Not all sites that exceeded this threshold experienced obvious habitat quality and biotic integrity declines, especially in locations where the streams were not channelized and the bed materials were coarse. Based on this study, a high threshold of 50% land in agricultural use (row crop, pasture, hay) was used. A more moderate threshold for agriculture of 10% drainage area was also analyzed based on 10% thresholds used in both impervious and mining feature analysis.

### Agriculture Data Discussion

Agriculture has been generally non-prevalent in the SLRW due to a short growing season, the topography, and unproductive soils. Pasture, hay, and livestock operations have historically been more prevalent in this region than row crops. No impaired reach drainages exceeded the 50% watershed area threshold. Further analysis of all 86 HUC-12 drainages resulted in no exceedances of the threshold. Lowering the threshold to 10% resulted in two exceedances; Skunk Creek (13.4%) and East Swan Creek (10.57%). Pasture and hay crop were the dominant agricultural practices in both drainages with less than 1.3% area dedicated to row crop. Surface water runoff from agricultural land use (excluding channelization) did not appear to be a major stressor in these watersheds. For more information on channelization, see the Channelization data discussion above.

### Discharges and Withdrawals

Surface water discharges and groundwater/surface water allocations were not analyzed in full detail in this analysis due to the complexity of water management in the upper SLRW. Water was pumped, discharged, piped, and sometimes rerouted across watershed boundaries. Municipal and/or mining water use and management were present in several of the impaired drainages. The Hydrological Simulation Program—Fortran (HSPF) model that was being developed (not yet completed) by Tetra Tech coincidental to the report could better estimate the impact to stream flows by anthropogenic changes to the surface water component.

Regarding the groundwater component, there was insufficient information available to build a model capable of tracking groundwater/surface water interactions in the upper SLRW during the time of this study. Though Tetra Tech was working on a SLRW model using HSPF, this type of model is primarily focused on surface water processes, not groundwater. A project was starting in the spring of 2015 to address this lack of information. Extra borehole data would be collected as part of the Minnesota Geological Survey's County Geologic Atlas effort. Numerous monitoring wells would be constructed to collect water levels and geochemical data, as well as new stream gages to provide flow statistics. The goal of the project was to put together the data necessary to begin to build a regional-scale groundwater model of the upper watershed within four years.

Tetra Tech completed a first draft of the Upper SLRW Mining Area Hydrology report in September 2014, for the MPCA. The results were under review and the model was undergoing adjustments at the same time as the work reported here. Groundwater- surface water interactions were modeled using GFLOW, which is "a simplified representation of the ground water flow system that represents average conditions and is based on limited data" and the results should be viewed as "preliminary insight into the ground water, surface water, and mining interactions in the SLRW" (Tetra Tech 2014).

### Withdrawals

The SLRW Groundwater Review report, written by the MPCA (2014) noted that; in general, surface water withdrawals over the past twenty years had increased with a statistically significant trend and groundwater withdrawals had increased with a small rising trend. Withdrawals in the SLRW were mostly for municipal and industrial use.

Biological impairments that had point source discharges in their "local" drainages included Elbow Creek, Skunk Creek, West Two River, Manganika Creek, Kinney Creek, Embarrass River, Ely Creek, and East Swan Creek. West Two River, Manganika Creek, Kinney Creek, and Elbow Creek had watershed withdrawals ranging from 0.10 to 0.31 Million gallons/day/square-mile (MGD/mi$^2$). Ely Creek had a mid-range value (0.03 MGD/mi$^2$). The potential impact of this withdrawal on the downstream system was discussed above in the Impoundments section. Embarrass River, East Swan Creek, and Skunk Creek had lower discharge per watershed area values, less than or equal to 0.001 MGD/mi$^2$.

### Discharges

Surface water discharges in the watershed likely had a greater effect on low flows than high channel forming flows. In some cases, streamflow to point source discharge ratios were extremely low. In these circumstances, there was likely a larger impact on stream water chemistry than in-channel flow or channel forming processes. An example of this was East Swan Creek where, according to the National Pollutant Discharge Elimination System (NPDES) Permit Program Fact Sheet, 92% of total stream flow during critical low flows was WWTP effluent. More detail on the water chemistry effects due to flow alteration can be found in 5.22 of this report.

Biological impairments that had point source discharges in their "local" drainages included Wyman Creek, Elbow Creek, West Two River, Manganika Creek, Embarrass River, Ely Creek, Spring Mine Creek and East Swan Creek. Manganika Creek Watershed had the greatest point source discharge per watershed area (4.12 MGD/mi$^2$), much higher than the other watersheds. East Swan Creek and Elbow Creek (upper) Watershed had the next highest discharge per watershed values, in the 0.3 to 0.4 MGD/mi$^2$ range. Embarrass River and Ely Creek had lower discharge per watershed area values, less than or equal to 0.001 MGD/mi$^2$.

Based on point discharges and withdrawals reported, Wyman Creek, Elbow Creek, West Two River, Manganika Creek, Spring Mine Creek, East Swan Creek, Kinney Creek, and Ely Creek should be future focus areas in local groundwater/surface water interaction modeling efforts.

### Percent Mine Features

Taconite mining operations have important impacts on the hydrology and water quality of the upper SLRW. Mining operations interact with and affect both surface and subsurface hydrology. The altered hydrologic network is complex and includes water withdrawals, surface water discharges, water reuse and routing, potential subsurface voids, and seepage from open pits and tailings basins to groundwater and/or surface waters.

In addition, mining activity over the past century has resulted in a substantial loss of headwaters streams in some watersheds resulting in a significant reduction in native base flow. The source of water for many of these streams has changed from headwaters wetlands or forested highlands to open pit basins. Surface water runoff is now exposed to waste rock and other mining by-products (altering the water chemistry) as water travels through the open pits and waste rock stockpiles before entering the outlet stream. Watershed boundaries have changed dramatically in some cases, with water being transported across historical boundaries via mining infrastructure or re-shaping of the terrain. Removal of headwater drainage areas significantly alters peak flows (magnitude, frequency, and duration) and base flows which has implications on stream morphology, biological integrity, species richness and diversity, and overall ecological health.

Stream flow in several of these impaired streams is augmented by regular or constant pit de-watering and/or point source discharges. Intercepted groundwater flow accounts for a significant portion of the water retained in the open pits feeding these impaired streams. In some cases, mine pit dewatering flows closely mimic pre-mining mean annual flows. In others, mine pit dewatering greatly exceeds that of pre-mining mean annual flows, elevating base flows and changing the water chemistry.

The mining hydrology index below was calculated to identify drainages with known biological impairments that also have a higher degree of mining activity. It is recommended that additional data be collected in these watersheds to determine whether flow alteration due to the mining industry in these drainages are causing stress to the impaired reaches.

### Calculating the index

Percent mine features was estimated using the Minnesota DNR's 2009 Mine Features GIS layer. Taconite pits, stockpiles (ore, waste rock, and surface overburden), buildings/infrastructure, tailing/settling basins, and roads were included in the dataset. A threshold of 10% mine features was used based on the following:

- A significant portion of mining activity is located in historical natural headwater areas of the impaired streams. Headwaters are a critical component of a healthy stream ecosystem; supplying source water, dissolved and particulate organic matter to downstream reaches; and often providing areas of infiltration, water storage, and canopy cover. A small percentage of watershed area could have a relatively large impact on the overall system due to the critical functions of headwaters.
- Correlates with upper third quartile (10.8%) of percent watershed in mine features for the impaired watershed delineations.
- Correlates with 10% thresholds used in impervious and agricultural land use metrics

**Mine Features Data Discussion**

The mine features threshold was exceeded in the impaired watersheds identified below in Table 20. Mine features percentages for Elbow Creek (upper), Manganika Creek, West Two River, and Kinney Creek were two to four times the 10% threshold. Swan River (cumulative) and Elbow Creek (lower/cumulative) barely exceeded the threshold. The majority of the contributing Elbow Creek (lower/cumulative) mine features resumed in the Elbow (upper) drainage delineation. Mine features located in Barber Creek and to a lesser extent Dempsey Creek, Upper West Swan River, and East Swan River contributed to the cumulative Swan River score. Nearly all of the land area located upstream of Manganika Lake in the Manganika Creek drainage had been converted to mine features.

Wyman Creek and Spring Mine Creek were bracketed within the two extremes. The St. Louis River Mining Hydrology report estimated reductions in base flows associated with current taconite mining operations and reductions in base flows after point source discharges were accounted for. The delineations of several watershed boundaries differed from the delineations presented here. In this report, the delineation was derived for the upstream drainage of the downstream-most point on the impaired AUID segment. The Tetra Tech report had other goals which were represented by DNR Level 08 catchments and sub-catchments within several of the Level 08 catchments.

Estimated values of reduced baseflow for Manganika Creek, West Two River, and Kinney Creek Watersheds before point source discharge consideration were 83.84%, 50.67%, and 52.71% respectively. After point source discharge consideration, the respective reductions were -365.45% for Manganika Creek and 10.90% for West Two River. The calculation for Kinney Creek was still under review. The report did not split out the upper Elbow Creek impairment drainage. Reduction in base flows for Elbow Creek (lower) was 3.39% due to mining operations and was mitigated to -9.55% after point source discharges were considered. No mine pit dewatering to Elbow Creek had occurred since 2002. Wyman Creek and Spring Mine Creek were not considered (or split out of larger drainages) in the Mining Hydrology Report.

Potential for water chemistry change was high in any of the watersheds listed in Table 20 due to the change in soil and water properties and processes and overall land cover changes that occur in mining activities. More information on water chemistry at the impaired AUIDs can be found in section 5 of this report.

**Table 20: Impaired watersheds that exceeded the 10% area threshold for land associated with mining activity features. Cells were shaded where an exceedance of 2 to 4 times the threshold was exceeded.**

| Impairment | Watershed Zone | % Mine Features |
|---|---|---|
| **Wyman Creek** | **LU-P** | **19** |
| **Spring Mine Creek** | **NU-EMB** | **17** |
| **Swan River (cumulative)** | **SWH** | **10** |
| **Elbow Creek (upper)** | **VIR** | **32** |
| **Elbow Creek (lower/cumulative)** | **VIR** | **11** |
| **Manganika Creek** | **VIR** | **46** |
| **West Two River** | **WTM** | **30** |
| **Kinney Creek** | **WTM** | **25** |

### Summary: Is Altered Hydrology a Stressor in Impaired St. Louis River Watersheds?

This section of the report was a generalization of hydrological changes that have occurred in the SLRW impairments. Multiple models were being developed during the time of this report, but were not finalized. Much of the data in this section relied on ArcGIS data interpretation. Potential and probable stressors for the various metrics can be found below in Table 21.

**Table 21:** Summary of the altered hydrology analyses for impaired streams of the SLRW.

| Stream | Crossings | Channelizaion | Impoundments | Impervious Surface | Agricultural Land use | Point Source Discharges & Withdrawals | Mine Features |
|---|---|---|---|---|---|---|---|
| Kingsbury Creek | ● | ● |  | ● |  |  |  |
| Miller Creek | ● | ● |  | ● |  |  |  |
| Otter Creek |  |  |  |  |  |  |  |
| Paleface Creek |  |  |  |  |  |  |  |
| Water Hen Creek - Upper |  | ● |  |  |  |  |  |
| Water Hen Creek - Lower |  |  |  |  |  |  |  |
| Sand Creek |  | ● |  |  |  |  |  |
| Skunk Creek |  | ● |  |  |  |  |  |
| St. Louis River |  | ○ |  |  |  |  |  |
| Stony Creek |  | ● |  |  |  |  |  |
| Unnamed Creek to SLR |  | ● |  |  |  |  |  |
| Vaara Creek |  |  |  |  |  |  |  |
| Wyman Creek |  |  |  |  |  | ○ | ○ |
| Ely Creek |  |  | ○ |  |  |  |  |
| Embarrass River |  | ○ | X |  |  |  |  |
| Spring Mine Creek |  |  |  |  |  | ○ | ○ |
| Elbow Creek - Upper |  |  |  | ● |  | ○ | ● |
| Elbow Creek - Lower |  |  |  |  |  | ○ | ○ |
| Manganika Creek |  | ● |  |  |  | ○ | ● |
| Kinney Creek |  |  | ○ |  |  | ○ | ● |
| West Two River |  |  | ○ |  |  | ○ | ● |
| Swan River |  |  |  |  |  |  | ○ |
| East Swan Creek |  | ● |  | ● |  | ○ |  |
| Little Swan Creek |  |  |  |  |  |  |  |

Key: ● = Probable Candidate Cause ○ = Potential Candidate Cause

### 3.2.2 Poor Physical Habitat Conditions

Habitat is a broad term encompassing all aspects of the physical, chemical, and biological conditions needed to support a biological community. The focus here will be on physical habitat. EPA's CADDIS website lists six broad categories that form a stream's overall physical habitat: 1) stream size and channel dimensions, 2) channel gradient, 3) channel substrate size and type, 4) habitat complexity and cover, 5) vegetation cover and structure in the riparian zone, and 6) channel-riparian interactions. Physical habitat loss is often the result of other stressors (e.g., sediment, flow volumes, DO) and so the reader is directed to other stressor sections for more detail. Degraded physical habitat is a leading cause nationally of impairment in streams on state 303(d) lists.

Specific habitats that are required by a healthy biotic community can be minimized or altered by practices on the landscape by way of resource extraction, agriculture, forestry, urbanization, and industry. Channelizing streams leads to an overall more homogeneous habitat, with loss of important microhabitats needed by particular species (Lau et al., 2006). These landscape alterations can lead to reduced habitat availability, such as decreased riffle habitat, or reduced habitat quality, such as embedded gravel/cobble substrates. In the past, it was common to remove large woody debris (LWD) from stream channels for various reasons. It has now been shown (Gurnell et al. 1995, Cordova et al. 2006, and Magilligan et al. 2008) that LWD is very important in creating habitat (causes scour pools, provides cover for fish and creates pockets of protection from faster currents, and a living surface for macroinvertebrates that cling to hard objects).

### Types of Physical Habitat Data

The MPCA biological monitoring crews conduct a qualitative habitat assessment using the MPCA Stream Habitat Assessment (MSHA) protocol at stream monitoring sites. The MSHA protocol can be found at: http://www.pca.state.mn.us/index.php/view-document.html?gid=6088. The MSHA scores can be used to review habitat conditions at biological sampling locations and compare those conditions against similar-sized streams. The MPCA has explored the relationship between MSHA scores and Index of biological integrity (IBI) scores, developing a probability function of a stream meeting its IBI threshold, given the MSHA score it received. The MPCA and MDNR staffs are collecting stream channel dimension, pattern and profile data at impaired sites and some stream locations having very natural conditions. This data can be used to compare channel form departure from a reference condition (i.e., the norm). Habitat features can be analyzed to determine if a stream has reduced pool depth, incorrect pool spacing, adequate cross sectional area to convey discharge, and various other physical habitat features that are too numerous to list here. The MPCA/DNR use the applied river morphology method developed by Rosgen (1996) to collect and analyze this data.

The Pfankuch Stability Index (PSI) was also used in this study to assess stream channel stability and physical habitat conditions within the impaired streams. The PSI is a rapid, semi-quantitative assessment of conditions in three primary areas of the stream channel; upper streambank, lower streambank, and channel bottom (substrate). The overall score provides an assessment of the condition of the stream channel and its ability to maintain its pattern, profile, and dimension over time.

### Candidate Cause Screening: Poor Physical Habitat Conditions

MSHA and PSI scores were compiled for all impaired streams of the SLRW to determine which streams had marginal to poor physical habitat conditions that could be causing biological impairment. Many streams were found to have habitat conditions that were potentially limiting to aquatic life. Poor habitat conditions are evaluated as a potential stressor for these specific streams in Section 5 of this report.

# 4.0 Overview of Analysis Tools Used to Evaluate Stressors

## 4.1 Tolerance Indicator Values (TIV)

The MPCA biological monitoring staff has developed a set of Tolerance Indicator Values (TIV) as a guidance for how tolerant various fish and macroinvertebrate taxa are to certain stressors. The TIV are calculated using the abundance weighted average of each taxon that is present in conjunction with water quality of physical conditions. For example, Central Mudminnow is a very tolerant fish species that has been observed as the dominant fish species in many streams with low DO conditions in Minnesota. As a result, this species has a TIV value for DO that indicates a very high tolerance to low DO. Each individual species is assigned a TIV value for a given stressor. Community level TIV have also been developed, which is calculated using the abundance weighted average of the tolerance values of each taxon at a station.

This report uses TIV values for the following parameters; TSS, DO, specific conductivity, nitrate, and chloride. The specific TIV values for fish and macroinvertebrate taxa of Minnesota can be provided upon request.

## 4.2 Box-Plot Distribution

Box plot distribution graphs are used throughout this report to compare study streams (impaired waters) against results from comparable reference streams in the SLRW. Box plot graphs are used to compare biological and habitat metric data as well as TIV for various parameters. The objective of these plots is to determine the degree of departure from the reference sites that is observed in the impaired stream.

An example of a box-plot graph from this report is shown in Figure 28. Acronyms are commonly used to describe the x-axis categories. A summary table of these acronyms is provided in Table 21 on the following page.

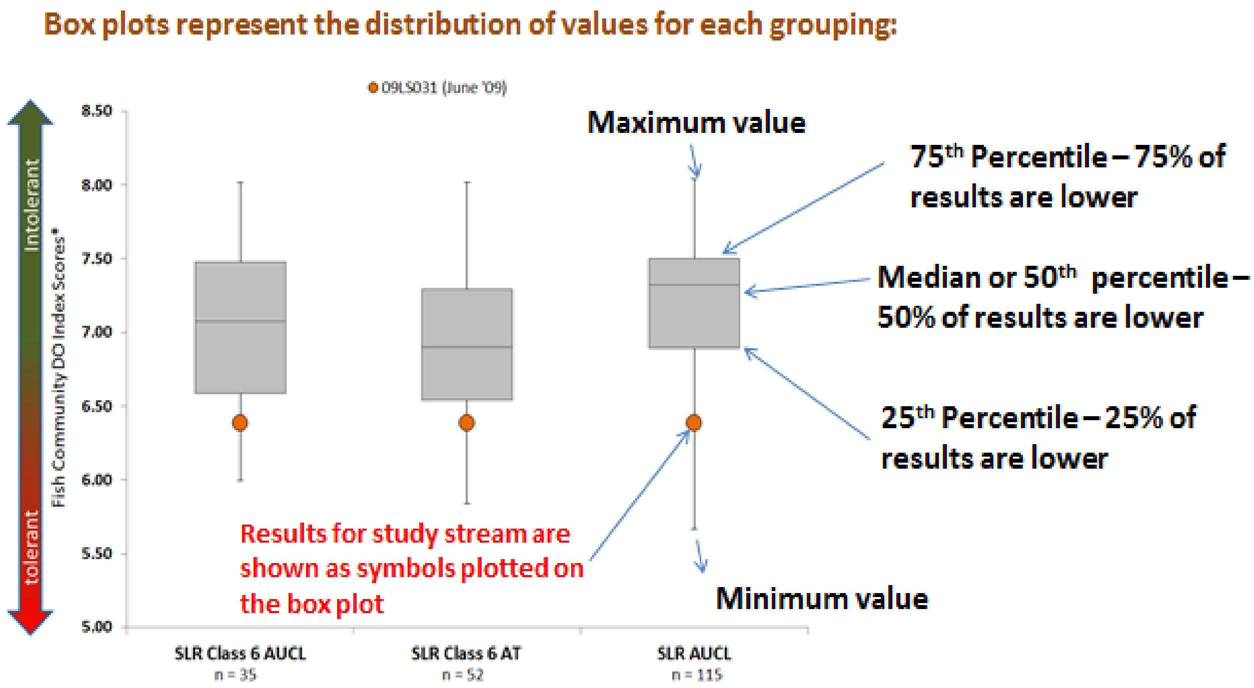


**Figure 28: Example of the box plot graphs used throughout the SLRW stressor ID report**

**Table 21: Descriptions of frequently used acronyms in tables and graphs throughout this report**

| Acronym | Description |
|---|---|
| **Class 6 (or other number)** | **Represents a specific fish or macroinvertebrate IBI station class** |
| **AT** | **Includes all stations scoring above (passed/full support) the IBI impairment threshold. These stations are meet, or are likely to meet general use criteria for aquatic life** |
| **AUCL** | **Includes all stations scoring above the upper confidence limit of the IBI impairment threshold. These are very high quality stations based on IBI results** |
| **Ref class A, B, or C** | **Includes all of the stations that were designated as reference stations based on water quality and/or biological data (see Section 1.2.3)** |

**Table 22: Description of some of the frequently used biological metrics in this report**

| Acronym | Description |
|---|---|
| **TV / TIV / Index Score** | Calculated using the abundance weighted average of each taxon that is present in conjunction with water quality of physical conditions |
| **TaxaCountAllChir** | Total taxa richness of macroinvertebrates |
| **EPTCh / EPT Taxa Richness** | Taxa richness of Ephemeroptera, Plecoptera & Trichoptera (baetid taxa treated as one taxon) |
| **EphemeropteraCh** | Taxa richness of Ephemeroptera (mayflies) |
| **EphemeropteraPct** | Relative abundance (%) of Ephemeroptera (mayfly) individuals in subsample |
| **LeglessChTxPct** | Relative percentage of taxa without legs |
| **Legless Pct** | Relative abundance (%) of legless individuals in subsample |
| **Sprawler Taxa** | Taxa richness of sprawlers (excluding chironomid and baetid sprawler taxa) |
| **Percent Sprawlers** | Relative abundance (%) of sprawler individuals in subsample |
| **Burrower Taxa** | Taxa richness of burrowers (excluding chironomid burrower taxa) |
| **Burrower Pct** | Relative abundance (%) of burrowers in subsample |
| **Long Lived Taxa Pct** | Relative percentage of longlived taxa |

# 5.0 Evaluation of Candidate Causes for Impairment

## 5.1 Duluth Urban Trout Streams

The Duluth Urban Coldwater Watershed zone contains two impaired coldwater (trout) streams, Miller Creek and Kingsbury Creek. Both of these urban trout streams are listed as impaired for failing to meet IBI criteria for fish and macroinvertebrates. Some of the potential stressors in that will be evaluated in this watershed zone include elevated water temperatures, altered hydrology, TSS, chloride toxicity, lack of connectivity, and habitat degradation.

Many of the candidate causes for impairment in this watershed zone are linked to high density urban land-uses and/or natural background limitations due to the bedrock geology of the region. Among the numerous designated trout streams within the city limits of Duluth, Miller Creek and Kingsbury Creek both rank among the top in terms of percent impervious land cover within the watershed. In addition to limitations from human disturbances, many reaches of these have natural features which present challenges for sustaining viable populations of coldwater fish. Waterfalls along both of these streams serve as natural barriers to fish migration, which may prevent trout from accessing critical thermal refugia such as springs and colder tributaries during summer months. Bedrock substrates are also common features of these streams which hinders benthic food production, groundwater-surface water exchange, and spawning success.

**Figure 29:** Representative reaches of impaired coldwater streams in the Duluth Urban Coldwater Watershed zone. (Left) Reach of Kingsbury Creek showing high gradient / bedrock geology nature of these streams. (Center) Impacts of 2012 flood Kingsbury Creek. (Right) Channelized portion of Miller Creek in high-density commercial area.

Common symptoms of macroinvertebrate impairment in this watershed zone include a lack of intolerant taxa, low POET (Plecoptera, Odonata, Ephemeroptera, and Trichoptera) taxa richness, and low scores in the Hilsenhoff Biotic Index (HBI), which is known to respond negatively to many types of disturbance, including organic pollution and thermal stress. The macroinvertebrate assemblages in Kingsbury and Miller Creeks contained a higher relative percentage of non-insect taxa, such as snails, scuds (Amphipoda), crayfish, and aquatic worms. Many of the non-insect macroinvertebrate taxa are more tolerant of stressors like low DO or benthic habitat degradation.

Both of these streams show a reduced number or lack of fish species that are considered “intolerant” or “sensitive” to disturbance in coldwater streams. Examples of sensitive species observed in high-quality coldwater streams within the SLRW include Brook Trout, Longnose Dace, Mottled Sculpin, and longnose sucker. Brook and Brown Trout (Brown Trout were limited to Kingsbury Creek) were present at several monitoring stations on impaired reaches of Kingsbury Creek and Miller Creek during 2009 and 2012 sampling visits, but they accounted for a relatively low percentage of the overall population. Sampling results also indicate that many or all of the adult trout observed in Kingsbury Creek were the result of recent stocking efforts. Pioneer species such as Blacknose Dace, Creek Chub, and White Sucker were typically dominant in these streams. High quality trout streams of smaller stream orders (1-3) typically have low taxa richness, and the species present are highly specialized to thrive in streams with colder water temperatures.

FIBI scores for Miller and Kingsbury Creek were also lower due to an abundance of omnivorous fish taxa. Omnivorous fish species are those that have the physiological ability (usually indicated by the presence of a long coiled gut and dark peritonium) to digest both plants and animals. They are able to utilize any available food resources, and their dominance within a fish community indicates an unstable food base. They are more tolerant of degradation than trophic specialists, because they can survive even if more sensitive food resources (e.g. benthic invertebrates) are reduced or eliminated, by switching to other, less sensitive, food resources. Coldwater obligate species such as trout and sculpin are trophic specialists relying on insect life (aquatic and terrestrial) and the predation of other fish for food.

Miller Creek continues to support a wild (naturally reproducing) population of Brook Trout in select areas despite the significant amount of development in much of its watershed. Portions of the stream corridor remain relatively undeveloped and offer ample shading to reduce water temperatures, as well as inputs organic matter (leaves, twigs) and larger woody cover (fallen trees, root wads). Figure 30 shows a high quality reach of Miller Creek and a wild Brook Trout caught by an angler within this section of the creek. Similar areas exist along Kingsbury Creek, but there not evidence of naturally reproducing Brook Trout

**Figure 30:** Wild Brook Trout caught by anglers on Miller Creek (left). A few reaches of Miller Creek remain in stable condition and provide fair to good habitat conditions for sustaining native Brook Trout populations without stocking (right).

## 5.2 Kingsbury Creek

Beginning at the outlet of Mogie Lake north of Proctor, Kingsbury Creek flows 0.75 miles through a wetland-dominated lacustrine valley as a low gradient, sinuous E channel. Just upstream of the first Ugstad Rd crossing, Kingsbury enters a ditched channel that flows for about 3.0 miles around the rail yard and through the city of Proctor. This reach is mostly straightened, but in spots has started to recreate meanders for itself and is trying to return to a more stable pattern. The valley type in this reach is mostly lacustrine, but in Proctor the valley is very unnatural and constricted, most closely resembling a small glacial trough or colluvial valley. Downstream of Proctor, Kingsbury Creek flows through a transition zone of natural riffle-pool ("C"-type channel) and step-pool ("B"-type) channels before tumbling over a mile down the hillside as a steep "A" channel (see appendix A for Rosgen channel types). The river valley becomes more and more bedrock-controlled throughout these reaches. There are two smaller channelized reaches downstream of Proctor – next to Highway 2 and through the Lake Superior Zoo.

Fish and macroinvertebrate communities were assessed at four monitoring locations on Kingsbury Creek, with sampling years of 1995, 1998, 2009, and 2012. Two stations (98LS003 and 95LS036) are located above the escarpment within the city limits of Proctor, while the other two stations are sighted in the steeper gradient, middle to lower reaches of the creek (Figure 31). Characteristics of the four monitoring stations and a summary of sampling results are provided in Table 23.

**Table 23:** Summary of Kingsbury Creek fish and macroinvertebrate monitoring stations and visit results

**Fish Assessments**

| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | FIBI Class | FIBI Result *(visit year)* | FIBI Result *(visit year)* | Standard | IBI Lower Confidence Limit | IBI Upper Confidence Limit |
|---|---|---|---|---|---|---|---|---|---|
| **98LS003** | 7.08 | 0.48 | 2 | 11 | **20** *(1998)* | **45*** *(2012)* | 35 | 25 | 45 |
| **95LS036** | 7.11 | 0.48 | 2 | 11 | **36** *(2009)* | - | 35 | 25 | 45 |
| **12LS004** | 7.74 | 3.17 | 2 | 11 | **46** *(2012)* | - | 35 | 25 | 45 |
| **12LS005** | 8.49 | 8.93 | 2 | 11 | **36** *(2009)* | - | 35 | 25 | 45 |

**Macroinvertebrate Assessments**

| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | Invert IBI Class | Invert IBI Result *(visit year)* | Invert IBI Result *(visit year)* | Standard | IBI Lower Confidence Limit | IBI Upper Confidence Limit |
|---|---|---|---|---|---|---|---|---|---|
| **98LS003** | 7.08 | 0.48 | 2 | 8 | **19.68** *(1998)* | **14.97** *(2012)* | 32 | 44.40 | 19.60 |
| **95LS036** | 7.11 | 0.48 | 2 | 8 | **5.93** *(2009)* | - | 32 | 44.40 | 19.60 |

Water quality and physical habitat data were used to develop a list of candidate causes for biota impairments in Kingsbury Creek. Based on the results of this review, the following candidate causes of impairment will be evaluated in this section:

1. **Mortality or stress of coldwater fish species due to elevated water temperatures**
2. **Low DO**
3. **Elevated TSS / Turbidity**
4. **Chloride Toxicity and Elevated Specific Conductance**
5. **Poor physical habitat**
6. **Toxicity from heavy metals (Copper & Lead)**

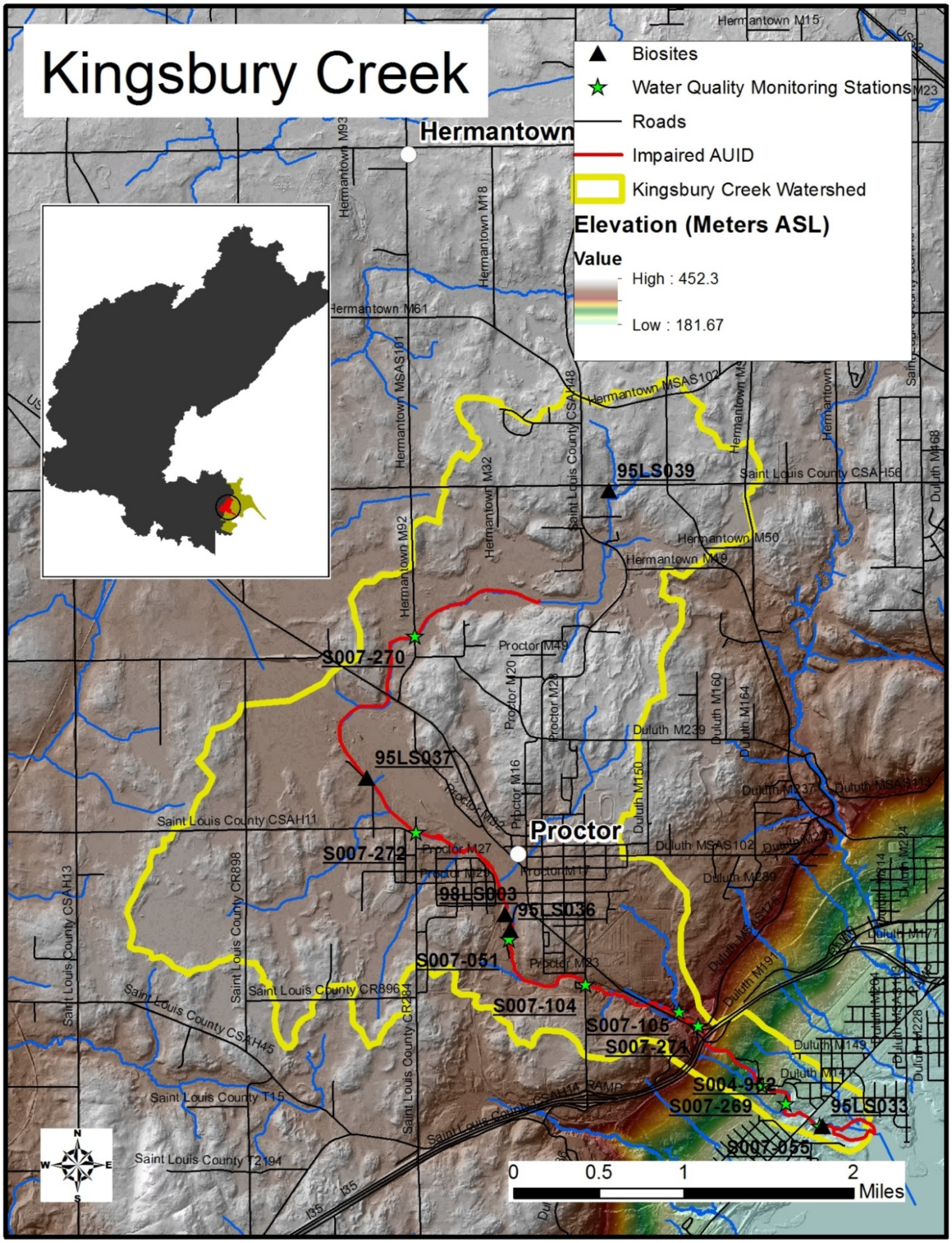


**Figure 31:** Map of Kingsbury Creek Watershed and impaired stream reach

### 5.2.1. Elevated Water Temperatures

Continuous temperature data were collected at three MPCA biological monitoring stations on Kingsbury Creek, (1) 98LS003 upstream of the Proctor High School ball fields, (2) 12LS004 upstream of Highway 2, and (3) 12LS005 upstream of the Lake Superior Zoo (Figure 32). The logger at 12LS005 was swept away during the June 2012 flood and had to be replaced. Therefore, only August 2012 data is available from that site. Additionally, data was used from DNR loggers at five sites between 1998 and 2002. Only data between June 1 and August 31 were analyzed (when stream temperatures are most likely to exceed the stress threshold for coldwater obligate species).

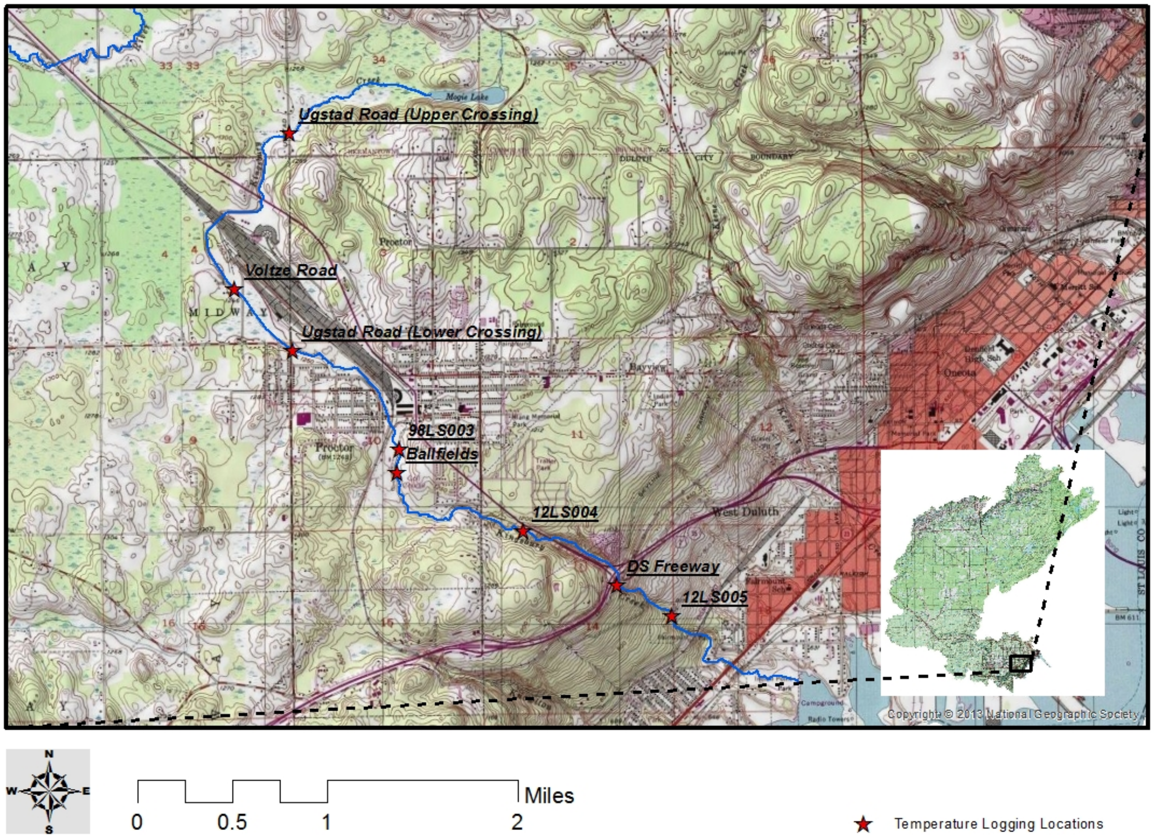


**Figure 32: Temperature monitoring locations on Kingsbury Creek**

Late June to the middle of August seems to be the critical time in Kingsbury Creek for coldwater sensitive species such as Brook Trout. Figure 33 shows the spatial breakdown of the temperature data, which was evaluated using at least 70% of the time in growth temperature (46.0-67.9° F) as the indicator of whether or not BKT should be present based on water temperature alone. At the majority of the Kingsbury Creek sites, the water temperature was adequate for most coldwater species, and was in the Brook Trout growth range at least 70% of the time. Three sites fell below 70% growth – Ugstad Road (Lower Crossing), 12LS004, and the DNR Highway 2 site (which is very close to 12LS004). The Ugstad Road and Highway 2 sites also significantly exceeded the lethal range (>77° F) for trout. At times, water temperatures at these stations were above the lethal threshold for 12 straight hours.

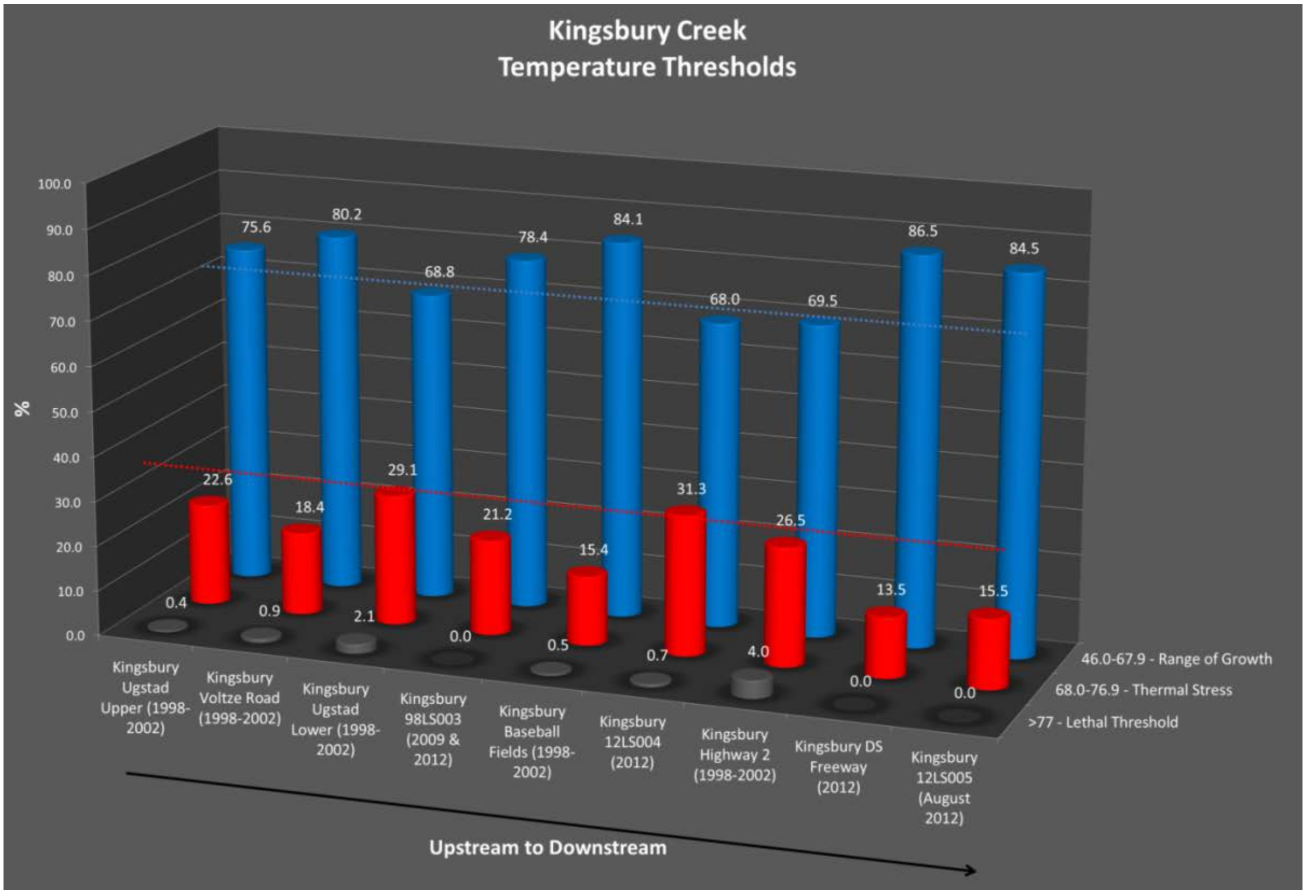


**Figure 33: Percentage of time spent in BKT growth, stress and lethal ranges**

The upper two (Ugstad Rd Upper and Voltze Rd) and lower two (DS Freeway and 12LS005) sites had relatively cold temperatures and could conceivably support coldwater species. Two adult Brook Trout were observed at the DS Freeway site during a stream reconnaissance performed by the authors – anecdotal evidence that the water in that reach is hospitable. The spatial distribution of the three sites with warm temperatures extends through the middle of the Kingsbury Creek Watershed. This part of the watershed contains several possible sources of temperature loading, including runoff from impervious surfaces in the city of Proctor and the Canadian National rail yard, and substantial portions of removed and/or inadequate riparian vegetation. The stream types in this area are mostly B and C channels (see appendix A for channel type descriptions), which have higher width/depth ratios and are more prone to warming from direct solar radiation. In contrast, the riparian corridor in the upper reaches is relatively undisturbed, aside from some channelization of the stream itself. Those reaches are mostly deep, narrow E-type channels with wide, well vegetated riparian corridors. The lower reaches are mostly narrow A-type channels with plentiful shade provided by healthy old growth cedar and pine forests in the riparian corridor.

In addition to daily maximum water temperatures, some research suggests that daily temperature fluctuation is an important variable related to the presence and absence of trout in streams. Temperature data for all Kingsbury Creek sites were plotted (Figure 34) using similar methodology to a trout temperature tolerance study in Wisconsin and Michigan (Wehrly et al. 2007). The X-axis represents the highest average daily temperature recorded at each site for the three-month period between June 1 and August 31. The Y-axis represents the highest temperature range recorded in a single day in the same

June-August time span. In the aforementioned study, fish and temperature data from streams across Michigan and Wisconsin were plotted. A BKT tolerance curve was developed by running a 95% quantile regression on the sites where BKT were found. This same curve is used in the Figure 34.

Almost every site falls outside the tolerance limit for BKT, with the exception of 12LS005, which is only August data. This analysis suggests that temperature fluctuation is also a limiting factor for Brook Trout in Kingsbury Creek in terms of water temperature. Possible causes of high temperature fluctuation are numerous in Kingsbury Creek, including -- (1) relatively small groundwater contributions in low flow conditions, (2) inadequate stream shading leading to direct solar heating, and/or (3) heated runoff from impervious urban and industrial portions of the watershed.

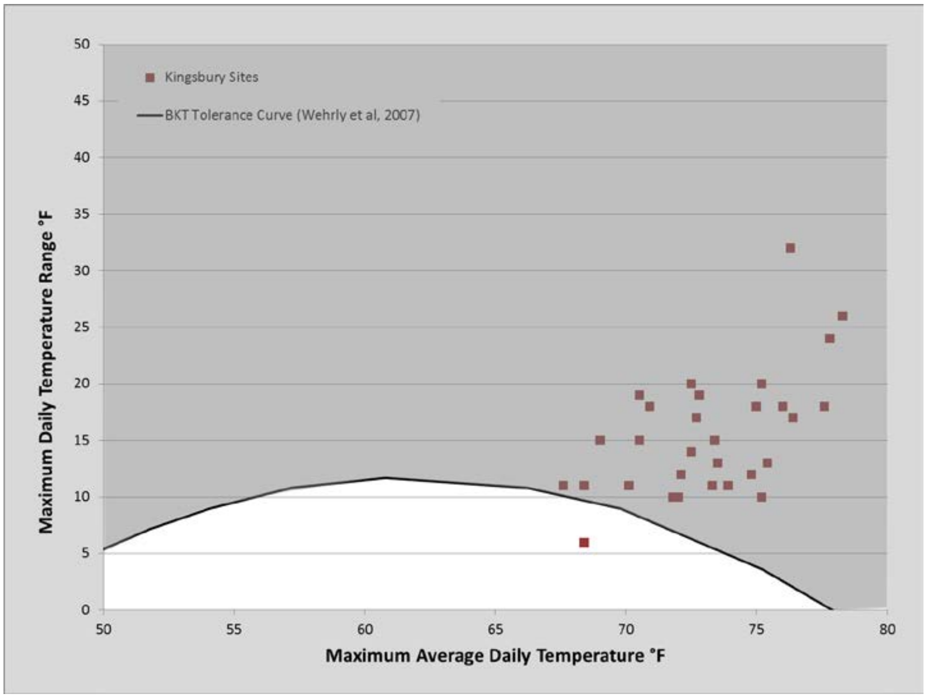


**Figure 34: Maximum daily temperature range and maximum average daily temperature for all Kingsbury temperature loggers**

### Biological Response to Elevated Water Temperatures

The fish community of Kingsbury Creek supports a very few fish species that are considered “intolerant” or “sensitive” to disturbance in coldwater streams. Examples of sensitive species observed in high-quality coldwater streams within the SLRW include Brook Trout, Longnose Dace, Mottled Sculpin, and Longnose Sucker. Brook Trout, as well as non-native Brown Trout, were present at several monitoring stations on impaired reaches of Kingsbury Creek, but accounted for a relatively low percentage of the overall population and were likely present due to DNR stocking efforts. Pioneer species such as Blacknose Dace, Creek Chub, and White Sucker were typically dominant in these streams.

An analysis of coldwater biological metrics reaffirms temperature as a stressor compared with unimpaired coldwater sites in the. Table 24 compares results from several key coldwater FIBI metrics between Kingsbury Creek stations support fewer coldwater fish taxa, a lower relative percentage of coldwater individuals, and a lower relative abundance of sensitive coldwater fish taxa compared to the reference stations. Some of the data from Kingsbury Creek may be altered due to stocked fish, so the results may be even more indicative of an impaired assemblage if no stocking were to take place.

**Table 24:** Coldwater metrics of Kingsbury Creek and unimpaired coldwater streams in the SLRW

| | Visit Number | Stream Name | Field Number | Fish Class | FIBI | Coldwater Taxa Present | Coldwater Intolerant % | Coldwater Sensitive Taxa % |
|---|---|---|---|---|---|---|---|---|
| | 19980006 | Kingsbury Creek | 98LS003 | 11 | 20 | 0 | 0.00 | 25.00 |
| | 20121114 | Kingsbury Creek | 98LS003 | 11 | 45 | 1 | 22.64 | 28.57 |
| | | | | | **Average** | **0.5** | **11.32** | **26.79** |
| Unimpaired Sites | 20091057 | Dutch Slough | 09LS014 | 11 | 49 | 1 | 9.09 | 40.00 |
| | 20091147 | Hay Creek | 97LS108 | 11 | 71 | 2 | 51.49 | 42.86 |
| | 20091156 | Keene Creek | 95LS028 | 11 | 61 | 1 | 27.47 | 25.00 |
| | 20091146 | Little Otter Creek | 09LS116 | 11 | 51 | 2 | 2.33 | 30.00 |
| | 20091012 | Midway River | 09LS117 | 11 | 55 | 2 | 15.43 | 30.77 |
| | 20091013 | Midway River | 09LS118 | 11 | 61 | 3 | 15.71 | 35.71 |
| | 20091068 | Midway River | 09LS117 | 11 | 59 | 2 | 23.44 | 30.77 |
| | 20091037 | Pine River | 09LS013 | 11 | 50 | 2 | 5.42 | 22.73 |
| | 20091065 | Pine River | 09LS013 | 11 | 53 | 2 | 1.78 | 31.25 |
| | 19970052 | Trib. to Midway River | 97LS039 | 11 | 63 | 2 | 2.74 | 57.14 |
| | 20091153 | Trib. to Midway River | 97LS112 | 11 | 52 | 1 | 23.38 | 44.44 |
| | 19970087 | White Pine River | 97LS083 | 11 | 63 | 2 | 15.55 | 38.46 |
| | 20091148 | White Pine River | 09LS115 | 11 | 55 | 1 | 5.88 | 28.57 |
| | | | | | **Average** | **1.8** | **15.36** | **35.21** |

**Summary: Are Elevated Stream Temperatures a Stressor in Kingsbury Creek?**

Based on stream temperature data and FIBI metric results, Kingsbury Creek clearly does not support a high quality coldwater fish assemblage. The biological and water chemistry data are supportive of elevated stream temperature as a cause of impairment.

### 5.2.2 Dissolved Oxygen

The DO was identified as a candidate cause for FIBI and MIBI impairment in Kingsbury Creek. Existing data show DO concentrations below the 7 mg/L water quality standard for coldwater streams. All available instantaneous DO readings from Kingsbury Creek are displayed in Figure 36 by monitoring station and calendar month. The only monitoring station that fails to meet the DO standard on occasion is S007-051, which is co-located with biological monitoring station 95LS036. Poor FIBI and MIBI scores from this particular site were the impetus for the impairment listing, although subsequent monitoring has also revealed poor IBI scores in other reaches of Kingsbury Creek as well.

Limited continuous monitoring of DO concentration also demonstrates sub-optimal conditions within the impaired reach of Kingsbury Creek. Results from continuous monitoring completed at station S007-051 in August of 2012 show DO concentrations dropping below 7 mg/L each day of the 5-day monitoring period, often for extended periods of time (20 – 42 hours) (Figure 37). Stream conditions during the continuous measurement were also symptomatic of a stream that may not support adequate DO concentrations for coldwater fish. Flows were stagnant, water temperatures were relatively warm (> 20° C), and a “scum layer” was observed on the surface of the water, perhaps due to nutrient enrichment (Figure 35). Diurnal DO flux was elevated (> 4 mg/L) during this monitoring period, likely due to the presence of algae and stagnant water. This reach of Kingsbury Creek has been channelized and lacks riffle features, which also limit DO inputs from reaeration.

**Figure 35:** Stagnant water with surface film observed at Kingsbury Creek station 95LS036. This reach has been channelized and lacks channel features (esp. riffles) that provide habitat reaeration of water.

### Sources and Pathways of Low Dissolved Oxygen

Dominant sources and pathways contributing to low DO concentrations in Kingsbury Creek include nutrient enrichment and factors related to geomorphology and stream channelization. Total phosphorous (TP) concentrations in the creek exceeded the 0.055 mg/L water quality standard in 23% (9 of 39) of the samples collected in the impaired reach. Elevated TP concentrations, in conjunction with summer and fall low flow periods and higher water temperatures, have the ability to produce algae blooms in areas of Kingsbury Creek with low gradient and stagnant flows. These conditions result in low DO (< 5 mg/L) concentrations that are unsuitable for trout, and diurnal DO flux greater than 4 mg/L, which can cause stress to sensitive fish and macroinvertebrate taxa (Heiskary 2013).

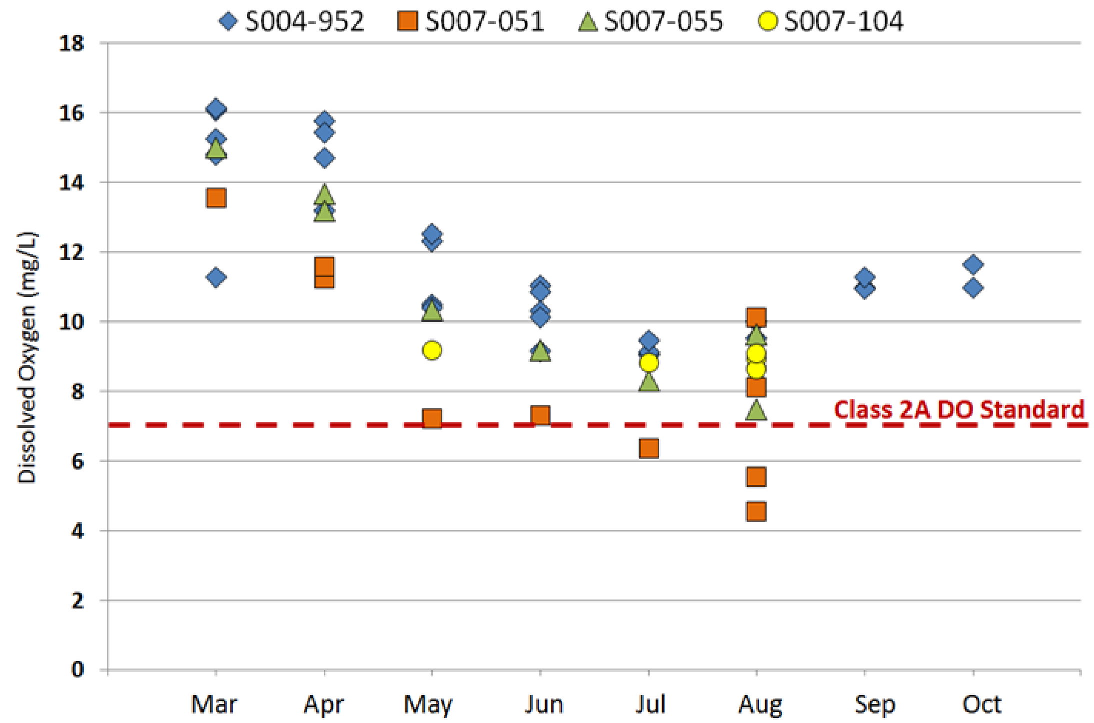
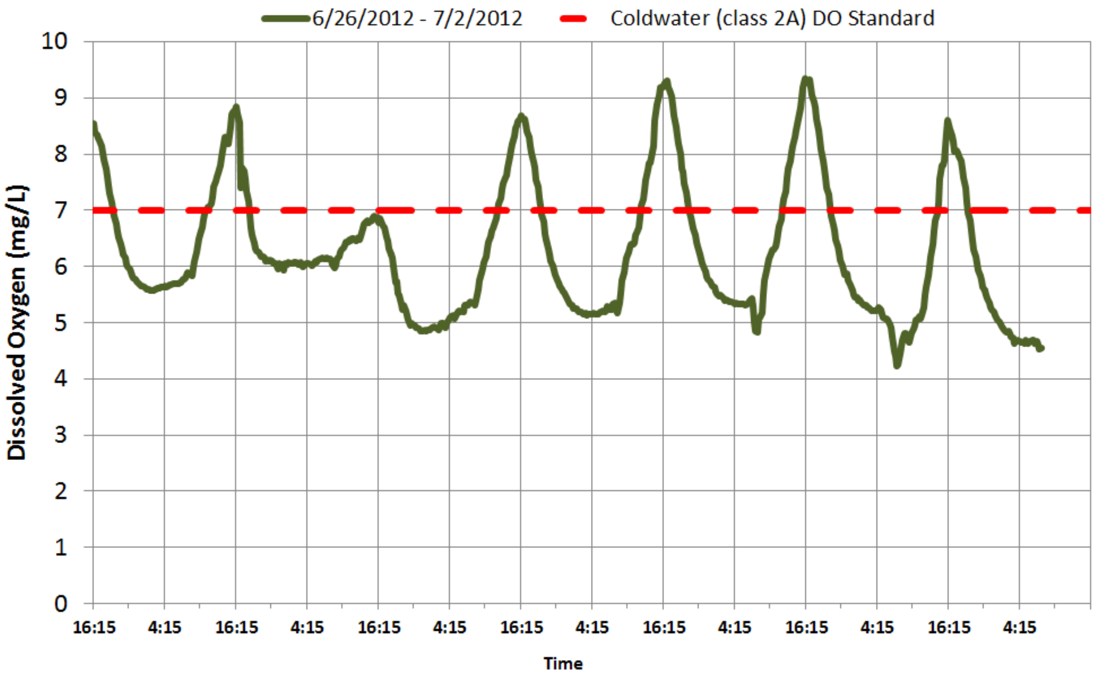


**Figure 36:** Point measurements of DO observed at Kingsbury Creek monitoring stations



| Monitoring Period | # of Readings | DO Min (mg/L) | DO Max (mg/L) | DO Average (mg/L) | DO Max (mg/L) | % Readings below 7 mg/L | Avg. Duration below 7 mg/L | Max Duration below 7 mg/L |
|---|---|---|---|---|---|---|---|---|
| 6/26/2012 - 7/2/2012 | 639 | 4.22 | 9.34 | 3.71 | 4.51 | 75.12% | 20 | 41.75 |

**Figure 37:** Continuous DO data collected at biological monitoring station 95LS036 on Kingsbury Creek

The geomorphology of the Kingsbury Creek Watershed also plays a role in its DO regime. Low DO levels are limited to lower gradient stream reaches in the headwaters area. Much of the upper watershed is channelized as well, which has led to channel widening and a reduction in riffles, which help supply oxygen. The mid to lower reaches of Kingsbury Creek flow through confined valleys (Rosgen types I, II, V) and are mostly Rosgen B and A channel types. There are numerous cascades and step pools within this section of the creek which supply ample oxygen levels through reaeration and colder water temperatures.

### Biological Response

The majority of the fish observed in Kingsbury Creek are species that can be considered neutral in terms of their tolerance to low DO conditions. Creek Chub and White Sucker are examples of fish species that fall within this tolerance class, both of which are abundant in many reaches of Kingsbury Creek. These two species are commonly found in streams with a wide range of DO concentrations, so their abundance in this stream does not offer much evidence for or against DO as a stressor.

Localized areas of the creek show a higher proportion of fish species that can be considered highly tolerant of lower DO concentrations. This is particularly apparent in the corridor along the city of Proctor athletic complex, which lies just downstream of the city of Proctor (stations 98LS003 and 95LS036). Fish species associated with low DO conditions and poor overall habitat conditions, such as Fathead Minnow and Central Mudminnow, were observed in fairly large numbers within this reach. Several minnow species common to wetland and headwater stream environments were also present (Northern Redbelly Dace, Pearl Dace, Finescale Dace). The proportion of these low DO tolerant species to more sensitive fish taxa at this monitoring station varies considerably between the two sampling events (note difference in DO TIV results for station 98LS003 in Figure 38), which may indicate that suitable DO conditions may exist during years when ambient conditions are favorable.

Brook Trout and Brown Trout were the only fish taxa observed that are considered intolerant to low DO conditions. Minnesota DNR routinely stocks catchable size trout in several reaches of Kingsbury Creek, and it is likely that all of the trout observed were introduced to the stream through stocking. The twenty-four Brook Trout individuals collected during the 2012 sampling of station 98LS003 are believed to be stocked fish, as there were no appreciable differences in size between the fish sampled and eroded caudal fins were observed in a large number of the fish. Eroded fins are a common deformity in stocked fish due to time spent in holding tanks. A single Brook Trout was observed at both station 12LS004 and 95LS036. Although there may be some natural reproduction occurring in some areas of Kingsbury Creek, historical and current data from this stream suggest that these fish are also the result of stocking efforts. As a result, the presence of trout at these stations does not factor into the weight of evidence approach for evaluating low DO as a stressor.

The fish assemblage of Kingsbury Creek does not provide overwhelming evidence for or against low DO as a stressor. However, after discounting stocked brook and Brown Trout, low-DO tolerant fish taxa are more abundant in Kingsbury Creek than taxa which are moderately or highly intolerant of low DO. The lack of self-sustaining populations of DO-sensitive fish species like Brook Trout and Longnose Dace provide evidence in support of low DO as a stressor. Fish community TIV values for DO are generally lower (more indicative of DO stress) than values observed at high quality stations of the same IBI class (Figure 39). This is particularly the case at stations where sub-optimal DO concentrations have been observed (stations 98LS003 and 95LS036).

Overall, the macroinvertebrate results from Kingsbury Creek are variable in terms of community level tolerance to low DO concentrations. Over 60% of the macroinvertebrates observed at station 98LS003 during the 1998 sampling event were low-DO tolerant taxa. The sample contained relatively large populations of low-DO tolerant taxa such as the isopod crustacean *Caecidotea*, the non-biting midge *Dicrotendipes*, and the air-breathing snail *Helisoma*. These taxa were not present in the more recent sample from this site, which resulted in a more favorable community level DO TIV value. Station 95LS036, located just upstream and within the same general stream reach as 98LS003, had a TIV result comparable to the median value from high quality stations of the same MIBI class (Northern Coldwater Streams). Aside from the 1998 sample of station 98LS003, the macroinvertebrate community of upper Kingsbury Creek appears to be fairly neutral in terms of tolerance to low DO.

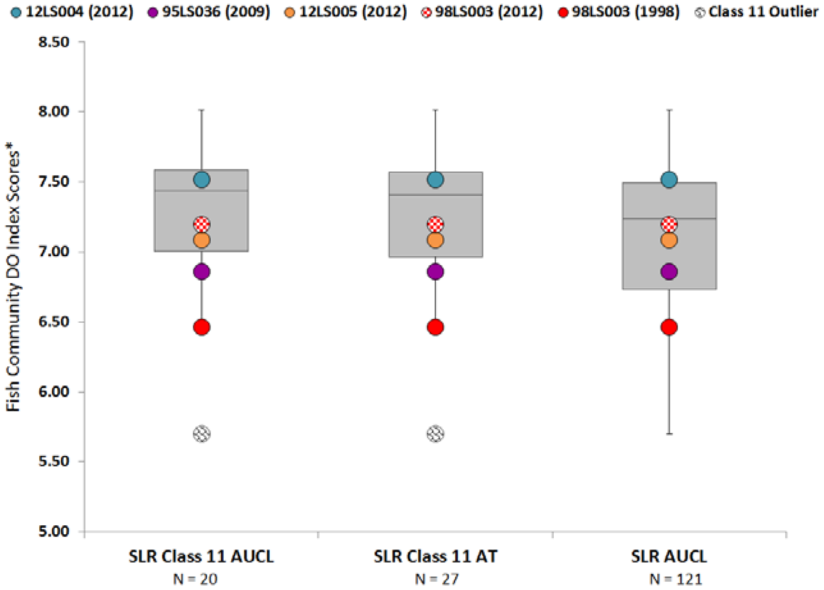


**Figure 38:** Fish community DO TIV results for Kingsbury Creek compared to high quality stations (*see table 21 for acronym descriptions)

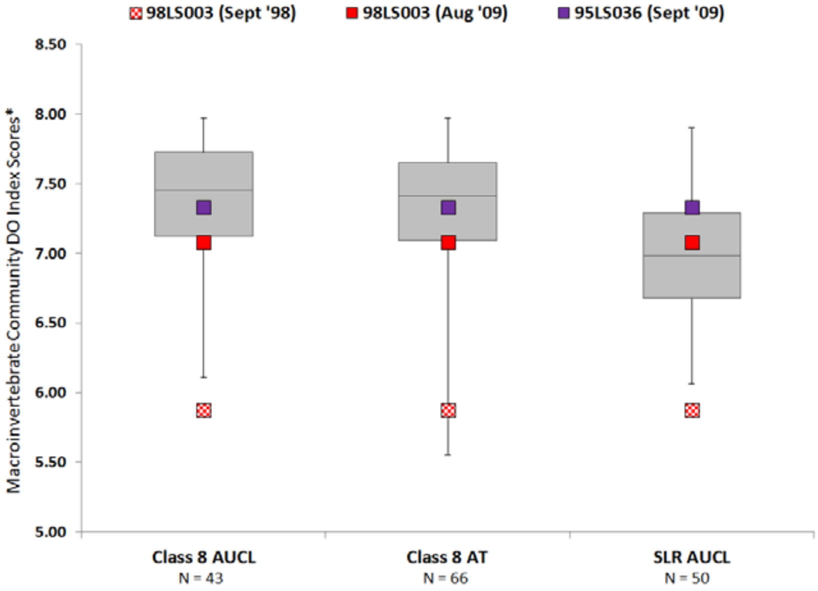


**Figure 39:** Macroinvertebrate community DO TIV results for Kingsbury Creek compared to high quality stations (*see Table 21 for acronym descriptions)

**Summary: Is Low Dissolved Oxygen a Stressor in Kingsbury Creek?**

The DO concentrations in the upper reaches of Kingsbury Creek (above Boundary Avenue) are frequently inadequate for supporting a self-sustaining coldwater fish assemblage. Water chemistry data collected within this reach indicate that concentrations regularly drop below the 7 mg/L DO standard applied to coldwater streams in Minnesota. The current DO regime, which regularly drops to levels below 5 mg/L during late summer and early fall months, supports a variety of warmwater species that can tolerate a wide range of DO concentrations. Sensitive coldwater species are not abundant or naturally reproducing in this reach, and marginal DO concentrations are one of several limiting factors.

Based on availale data, Brook Trout and other coldwater fish species are generally absent from the lower reaches of Kingsbury Creek as well. However, current water quality data suggest that low DO concentrations are not a limiting factor (i.e. stressor) in the lower reaches of Kingsbury Creek (below Boundary Avenue). All of the DO measurements collected in lower Kingsbury Creek to date have been at or above the 7 mg/L standard for coldwater streams. Other stressors, such as limited streamflow, elevated water temperatures, and marginal habitat conditions are more prominent stressors in these lower reaches.

There is some indication that low DO concentrations are contributing to the macroinvertebrate impairment as well, but the evidence is somewhat weaker due to variability between monitoring visits. Macroinvertebrate sampling was limited to montoring stations in the upper reaches of Kingsbury Creek (stations 98LS003 and 95LS036). The taxa present at these stations had a wide range of tolerance levels to low DO, and no firm conclusions can be made based on the macroinvertebrate data. The macroinvertebrate comminity would undoubtedly benefit from any restoration activities designed to improve DO conditions for coldwater fish.

### 5.2.3 Total Suspended Solids & Turbidity

Monitoring data from 2008 – 2012 were used to develop longitudinal summaries of TSS and Secchi tube data for Kingsbury Creek. The data summary by site, average value, number of samples and draft standard exceedance percentage are shown in Table 25. Under the current WQ standards for TSS, a site is considered for an impairment listing if more than 10% of the samples exceed the threshold value. For additional information on the water quality standard for TSS, refer to Section 3.1.8. A map of the Kingsbury Creek stations where at least 10% of samples exceeded the draft standard is shown in Figure 40.

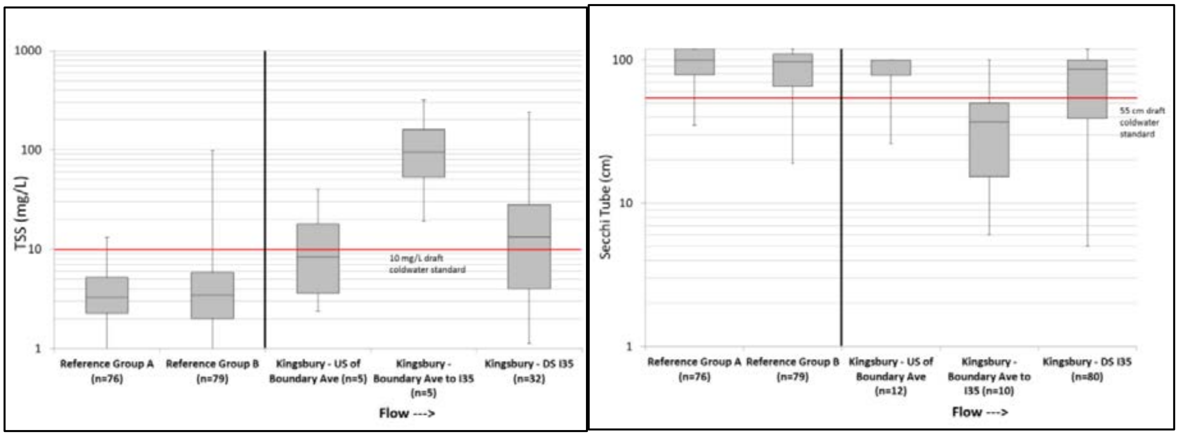
The TSS concentrations in Kingsbury Creek generally increase from upstream to downstream. The two upper sites (S007-270 and S007-272) met the draft standard in 100% of the samples, although it should be noted that there is a small sample size at these two sites. Every site downstream of S007-272 fails to meet the draft standard in over 10% of samples. The site at Point Drive (S007-051) is co-located with 98LS003 (the biological monitoring site that triggered the impairment listing), and failed to meet the draft standard in 3 out of 9 samples (33%). The percentage of standard-exceeding samples fluctuates downstream of S007-051 without any clear trend (Table 25), possibly due to low sample numbers and inconsistent sample timing at some sites (e.g. samples only taken during rain and snowmelt events). Station S004-952 at the Lake Superior Zoo has the most robust TSS dataset, with 98 samples taken between April and September in the last 10 years. The TSS target of 10 mg/L for coldwater streams was exceeded in 33% of the sampled collected at this station, and the average TSS value was three-times higher than the draft standard of 10 mg/L for coldwater streams at 30.7 mg/L.

**Table 25:** Longitudinal TSS and Secchi tube average values and percent draft standard exceedances for Kingsbury Creek

| | Site | Site Description | TSS Average (mg/L) | # of Samples | TSS % Exceeding Standard | Secchi Tube Average (cm) | # of Samples | Secchi Tube % Exceeding Standard | Total # of Samples | Total % Exceeding Standard |
|---|---|---|---|---|---|---|---|---|---|---|
| | S007-270 | KINGSBURY CK AT UGSTAD RD | 3.6 | 1 | 0.0% | 100.0 | 3 | 0.0% | 4 | 0.0% |
| | S007-272 | KINGSBURY CK AT N UGSTAD RD | 8.4 | 1 | 0.0% | 95.3 | 3 | 0.0% | 4 | 0.0% |
| F | S007-051 | KINGSBURY CK AT PIONK DR | 20.1 | 3 | 66.7% | 71.3 | 6 | 50.0% | 9 | 55.6% |
| l | S007-104 | KINGSBURY CK AT S BOUNDARY AVE | 36 | 2 | 100.0% | 64.4 | 5 | 60.0% | 7 | 71.4% |
| o | S007-271 | KINGSBURY CK AT SKYLINE PKWY | 320 | 1 | 100.0% | 25.0 | 2 | 100.0% | 3 | 100.0% |
| w | S007-269 | KINGSBURY CK AT OLD THOMPSON HILL RD | | 0 | | 63.3 | 4 | 50.0% | 4 | 50.0% |
| ↓ | S004-952 | KINGSBURY CK AT LK SUPERIOR ZOO | 30.7 | 29 | 55.2% | 75.3 | 69 | 30.4% | 98 | 37.8% |
| | S007-055 | KINGSBURY CK AT WALKING BR | 109.5 | 3 | 66.7% | 57.0 | 7 | 57.1% | 10 | 60.0% |
| | S007-105 | UNN TRIB TO KINGSBURY CK | 127 | 2 | 100.0% | 16.7 | 3 | 100.0% | 5 | 100.0% |

Box plots of TSS and Secchi tube values for Kingsbury Creek and the "A" and "B" reference streams in the SLRW are shown in Figure 40. Kingsbury data was lumped into three reaches due to the small number of samples at some sites. The three reaches are: 1) the upper watershed upstream of Boundary Aveue, 2) the transitional zone between Boundary Avenue and Interstate 35, and 3) the bedrock-dominated escarpment downstream of Interstate 35. The TSS and Secchi tube datasets for Kingsbury show a clear longitudinal trend, with a consistent violation of the draft standards in the transitional zone between Boundary Ave and Interstate 35. This reach contains many eroding banks and is where the unnamed tributary discussed above enters Kingsbury Creek. TSS and Secchi tube data show improving water quality downstream of the Interstate – most likely due to the bedrock- and boulder-dominated channel and the influence of clear groundwater seepage into the stream in this reach. These trends are reflected in the snowmelt sampling discussed below. The data for the "A" and "B" reference streams overwhelmingly meet the draft standard for both TSS and Secchi tube, indicating that low levels of suspended solids are closely linked with healthy biologic communities in the SLRW.

**Figure 40: Box plots of TSS concentrations (left) and Secchi transparency (right) results for Kingsbury Creek and reference streams**



## Longitudinal snowmelt sampling

During periods of high flow, such as snowmelt events, streams often exhibit higher levels of turbidity. The larger discharges during snowmelt increase stream power and erode stream banks more than periods of low flow. For this reason, conducting longitudinal TSS sampling during these events is vital to locating stream reaches that serve as sources of suspended sediment in a watershed.

The results from a longitudinal sampling of a snowmelt event in April 2014 are shown in Figure 42. The results show a relatively large increase between S007-272 (North Ugstad Road) and S007-051 (Point Drive). The data also reveal significant sediment loading from a small tributary (pictured in Figure 45, upper left) entering Kingsbury Creek just upstream of Interstate 35. A six-fold increase in TSS, 53 mg/l to 320 mg/L, was observed between Boundary Avenue (S007-104) and Skyline Parkway (S007-271). It is no coincidence that the tributary mentioned above drains into Kingsbury Creek between these two sampling points. The tributary itself had a TSS concentration of 160 mg/L, suggesting that a significant amount of sediment is also being introduced from the main stem Kingsbury Creek segment between Boundary Avenue and Skyline Parkway. Figure 41 shows the sample bottles from this sampling effort, and is illustrative of how the water clarity changed from the headwaters to the mouth.

**Figure 41:** TSS sample bottles from the April 2014 snowmelt event

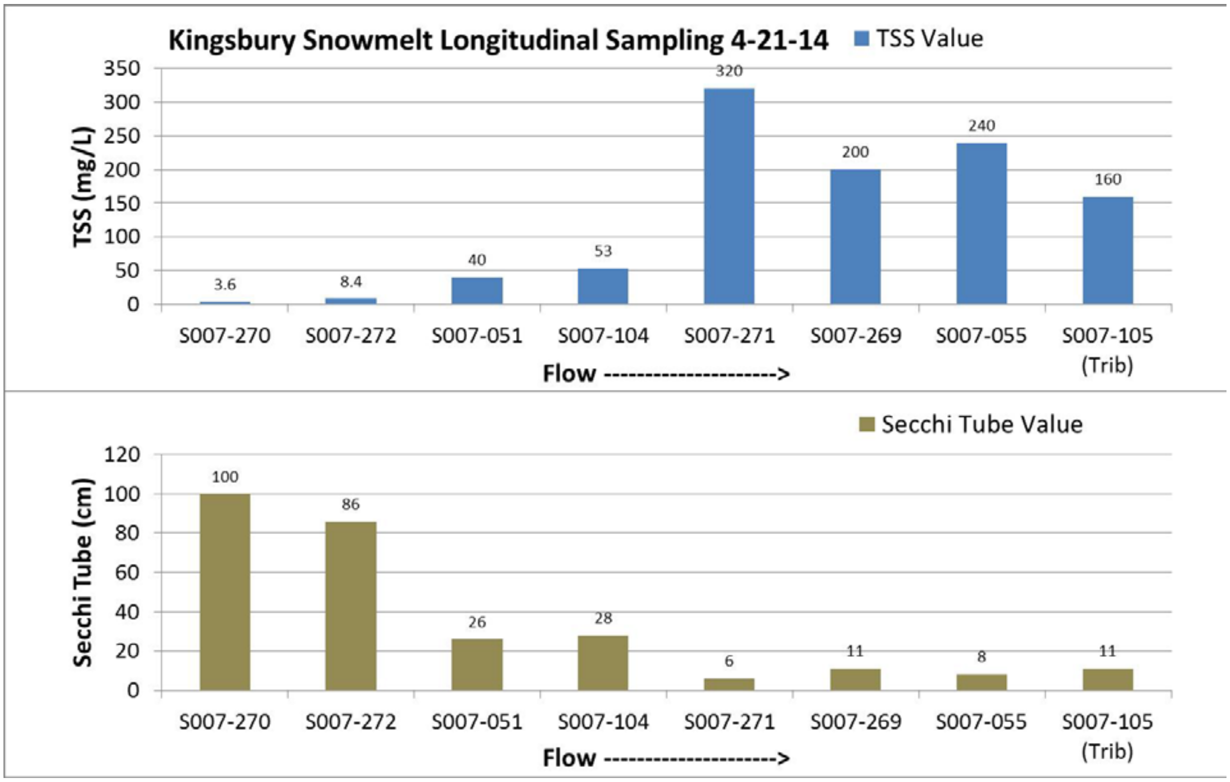


**Figure 42:** TSS and Secchi Tube results from a 4/21/2014 snowmelt sampling event on Kingsbury Creek

### Seasonal variation in total suspended solids

Like most of the streams in the city of Duluth and along the North Shore of Lake Superior, Kingsbury Creek appears turbid during periods of high flow, and clears up significantly during moderate to low streamflow. High TSS numbers are generally limited to short-term events driven by snowmelt and heavy rain events. Higher TSS concentrations tended to take place from mid-April through May, when melting snow and precipitation on already-saturated soil trigger peak stream discharges. This relatively short-lived increase in TSS may have a less deleterious effect on biological communities than more persistently high TSS levels seen in other watersheds in the region (e.g. Nemadji River, Beaver River, Swan River).

## Sources and pathways of sediment in the Kingsbury Creek Watershed

### Ditching/Channelization

Channel straightening and meander bend removal result in a shortening of the channel, which causes the water slope to increase and the velocity of channel-forming discharge to increase. If grade control (culverts, bedrock, etc.) is absent, channel incision will often follow – delivering sediment to the stream from the bed and banks as channel evolution progresses. Approximately 56% of Kingsbury Creek (AUID 04010201-626) has been channelized or straightened (see Figure 43). Most of the channelization has occurred in the upper reaches of the watershed. Based on the current slope and elevation data, the upper reaches of the creek used to be a low gradient (<1% slope) E or C channel in a wide lacustrine or alluvial floodplain. This wide, flat floodplain was targeted by developers in Proctor area, and the channel was straightened and re-aligned to make room for industry, recreation facilities, and urban development (see inset in Figure 43 showing Kingsbury Creek channelized around the rail yard).

Channelized stream reaches are also evident in the vicinity of the US Highway 2 – Skyline Road intersection and near the mouth of Kingsbury Creek within, and downstream of the Lake Superior Zoo (Figure 43). In both cases, the natural pattern and dimension of Kingsbury Creek was altered for the purposes of road construction, stream crossings (culverts), and/or other forms of development.

### Channel Instability/Bank Erosion

Channel instability is common in heavily urbanized watersheds (Booth 1990). Increased impervious surfaces cause augmented peak flows, and without grade control, stream channel incision is often the end result. Figure 44 shows an example of the increase in development and impervious surface in the Kingsbury Creek Watershed over the last half-century. The area displayed in the aerial photos in this Figure drains into an unnamed tributary that delivers high sediment loads to Kingsbury Creek during snowmelt and rain events.

Areas of channel instability and bank erosion were observed mostly in the aforementioned tributary and in Kingsbury Creek between Interstate 35 and Boundary Avenue in Proctor. This is the transitional area between the low gradient headwaters and the bedrock-controlled cascading reaches in the lower part of the watershed. This area alternates between "B" type channels, which are mostly stable, and "C" type channels, which often show signs of lateral instability in the form of bank erosion (see Figure 45). C-type channels are less efficient at transporting sediment than B channels because they are generally flatter and are not as entrenched. Thus, in incised rivers with excess sediment supply, it is predicted that the C channels will be the reaches where aggradation and lateral instability will occur. This has been an observed trend in many North Shore streams, and it is posited that Kingsbury Creek is experiencing a similar scenario with bank erosion (as a consequence of lateral instability) contributing sediment to the stream.

There are also areas that are contributing sediment as a result of vertical instability. A large headcut was observed in the Kingsbury tributary (Figure 45), just downstream of the trailer park. The reach immediately downstream of this headcut is likely contributing a significant amount of suspended sediment as channel evolution progresses.

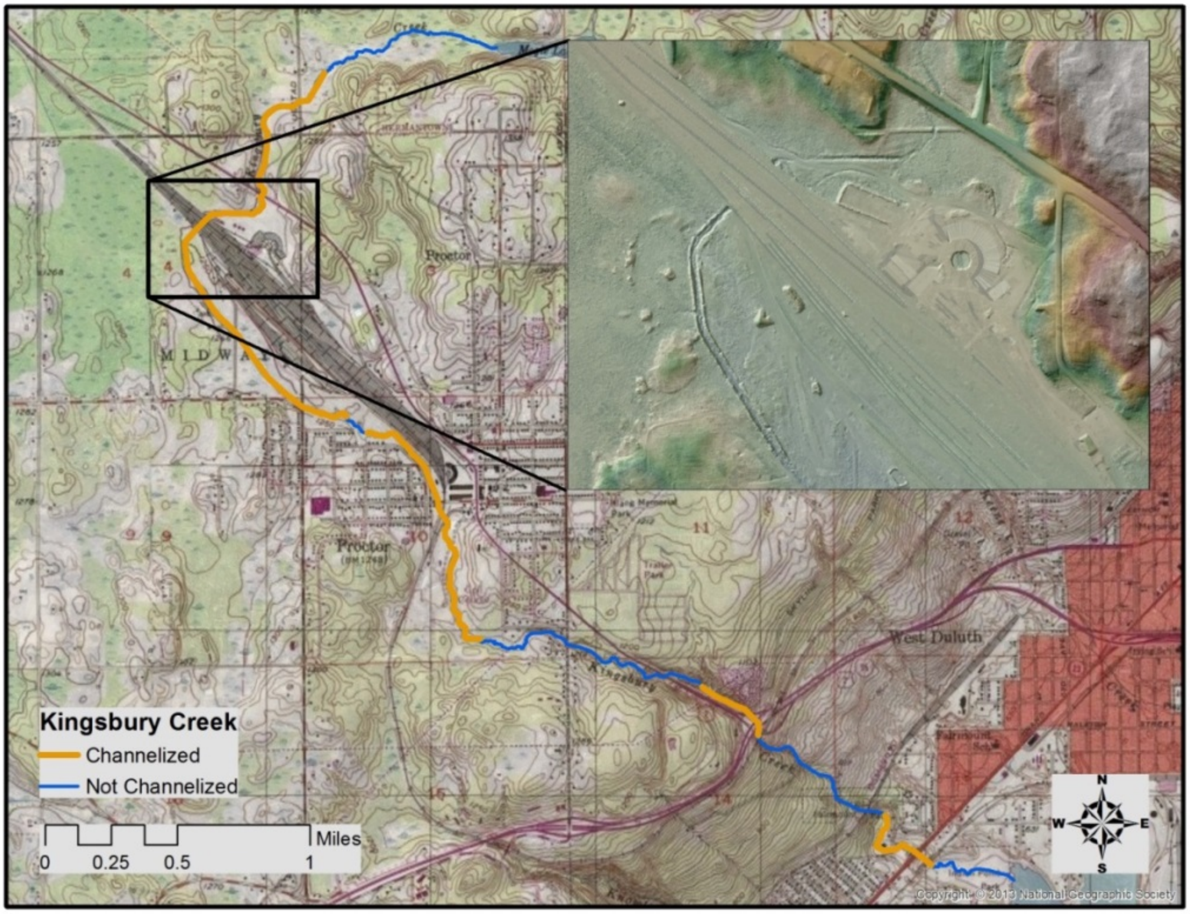


**Figure 43:** Map showing the channelized and straightened reaches of Kingsbury Creek

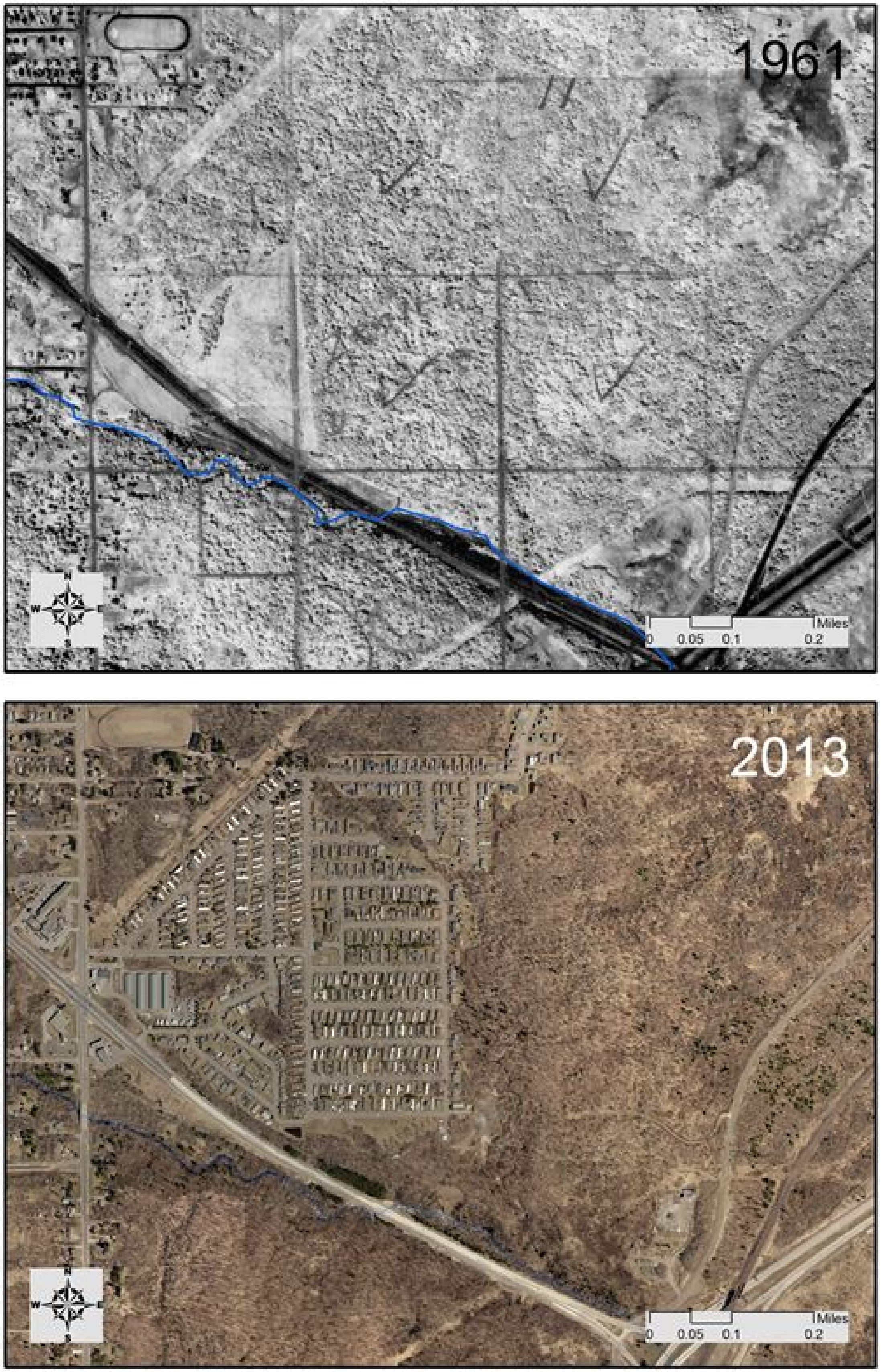


**Figure 44:** A dramatic example of development in the Kingsbury Creek Watershed

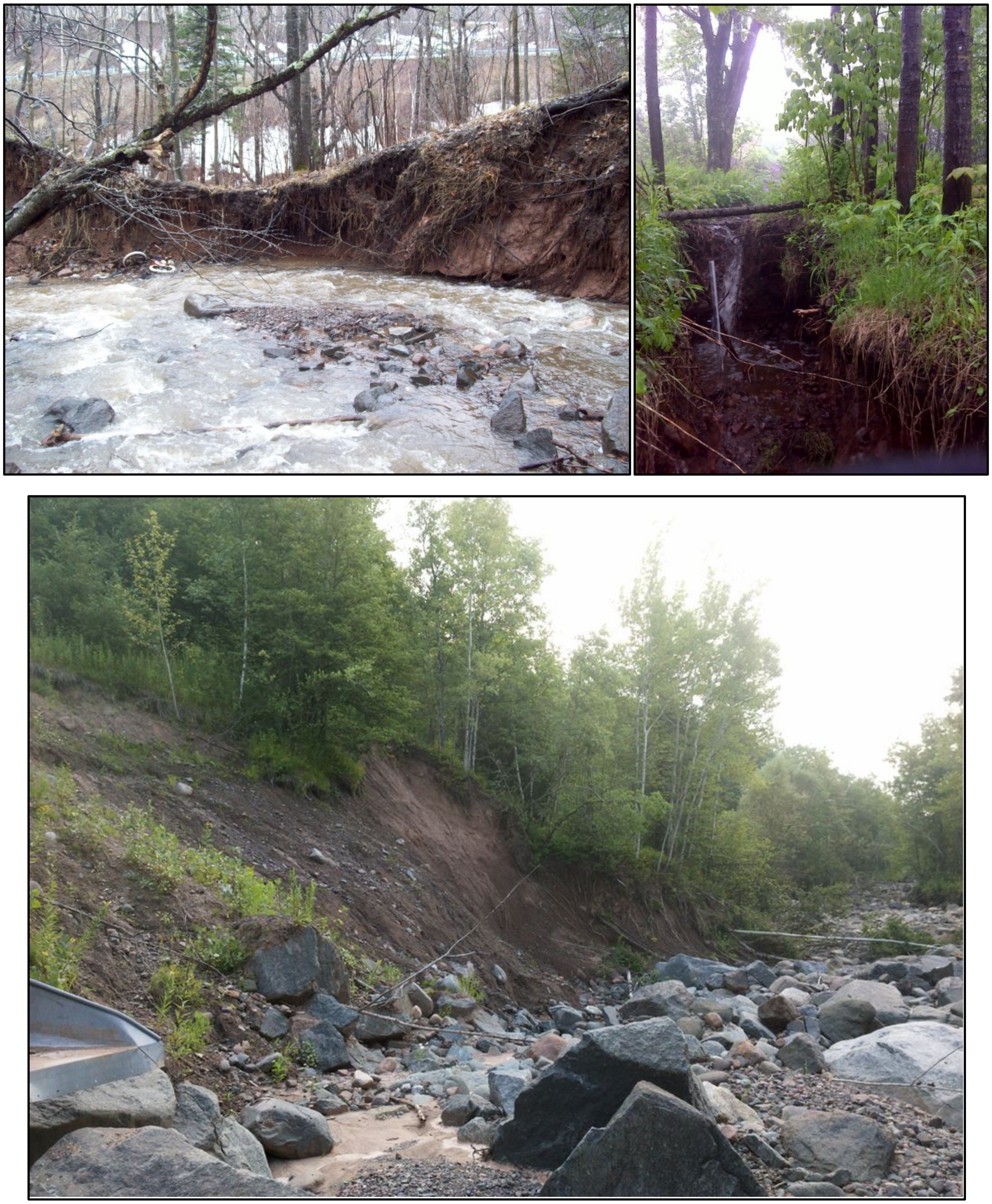


**Figure 45:** Severely eroding bank on the unnamed tributary to Kingsbury Creek (upper left); a ~1.0 meter headcut on the unnamed tributary to Kingsbury Creek (upper right); and major bluff erosion site downstream of HWY 2 (lower)

### Urban Stormwater Runoff

Urban, residential, and industrial development in and around the city of Proctor has increased the amount of impervious surfaces in the Kingsbury Creek Watershed, causing precipitation to flow over the land instead of infiltrating into the ground. This quickly flowing runoff can transport sediment and other particulates into streams. During snowmelt, road sand, litter, and other detritus piled into snowbanks over the course of the winter season (Figure 46) are flushed into stormwater pipes and streams in a relatively short period of time. Soils exposed by construction and industrial activities are susceptible to erosion during rain events. The Kingsbury Creek Watershed has a great deal of developed and barren land (Figure 46), especially in close proximity to the stream channel, thus it is likely that urban runoff is a pathway for suspended solids in Kingsbury Creek.

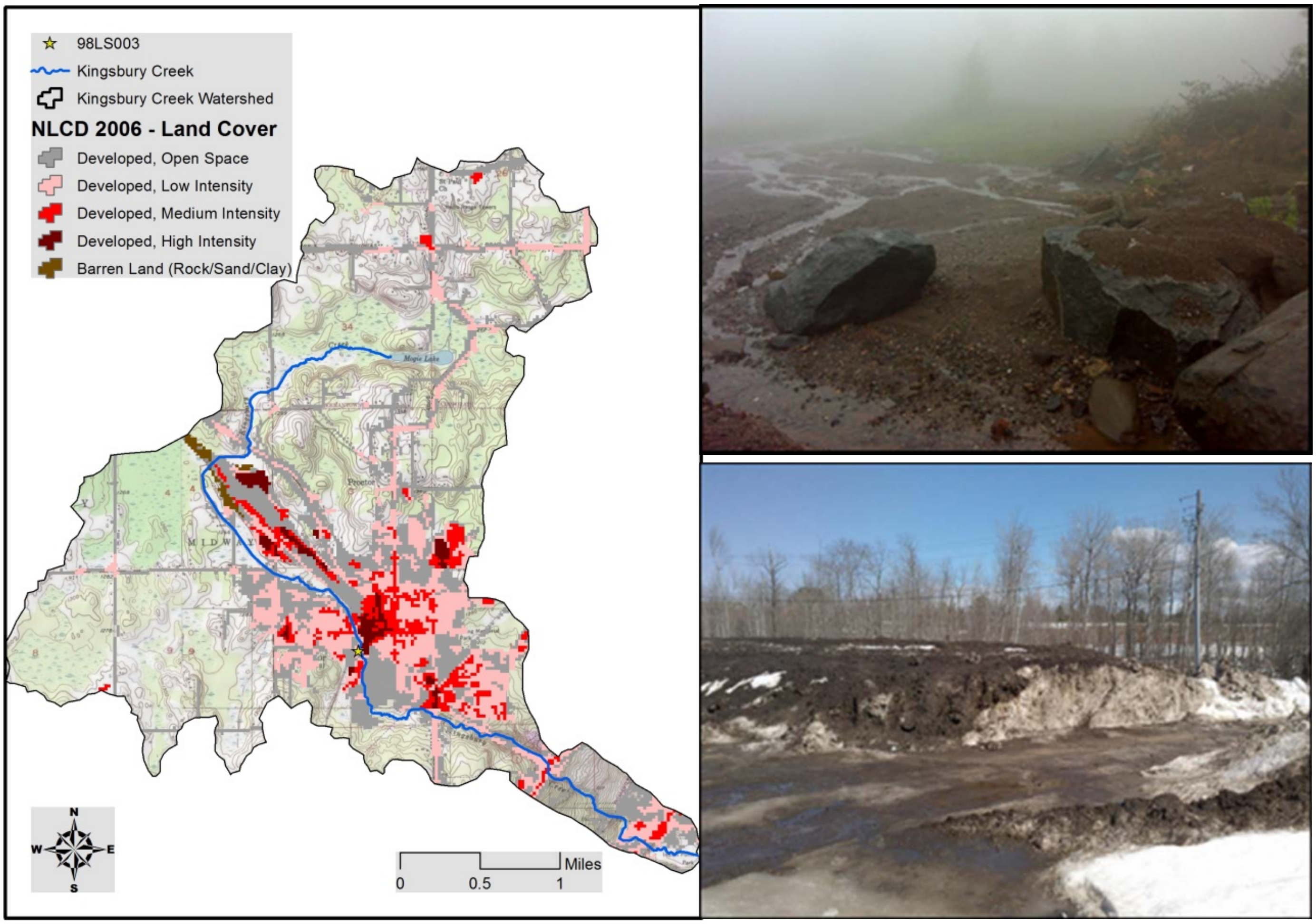


**Figure 46:** Developed and barren land in the Kingsbury Creek Watershed (left); After snowmelt, road sand and other detritus is ready to be washed into nearby Kingsbury Creek (right)

## Impacts of the 2012 Duluth Flood

Over the course of a 24-hour period on June 19-20th, the Duluth area received around 10 inches of rainfall. This precipitation event resulted in damaging floods in many of the watersheds in and around the Duluth metropolitan area, which drastically changed stream channel and floodplain conditions in many of these streams. In the Kingsbury Creek Watershed, many pools were filled with cobbles and boulders that were carried downstream as bedload during this extreme high flow event (Figure 47). The following link includes footage of Kingsbury Creek during this flood event (https://www.youtube.com/watch?v=WZCK0OBcLxA). In this footage, you can hear large boulders knocking as they careen down gradient.

The 2012 flooding caused significant streambank and bluff erosion in the Kingsbury Creek Watershed, and also caused damage to infrastructure along the stream corridor (Figure 48). Many of the impacted areas have yet to recover and are still in a state of vulnerability for further erosion and sediment delivery to the creek. Biological monitoring and other stream assessment data used to list Kingsbury Creek as an impaired water were collected in 2009 and 2010, prior to this flood event, and thus did not factor into the assessment process. However, the widespread impacts of the 2012 flood must be considered in the planning of restoration and implantation activities aimed at reducing sediment loading in this watershed.

**Figure 47:** The photo on the left shows a deep pool where several Brook Trout were sampled on 6/13/12. The photo on right, taken about a month later, shows the extent of pool filling from the 2012 flood. (The red dot is marking the same rock in each photo).

**Figure 48:** Streambank and infrastructure damage along Kingsbury Creek caused by extreme flood event in June 2012.

### Kingsbury Creek BANCS Analysis: An Effort to Pinpoint Bank and Near-Channel Sediment Sources

The BANCS (Bank Assessment for Non-point source Consequences of Sediment) assessments are meant to predict stream bank erosion rates and use two tools for estimating bank erosion: the Bank Erosion Hazard Index (BEHI) and Near-Bank Stress (NBS) (Rosgen 2001). The assessment takes characteristics of individual stream banks and the distribution of energy and shear stress in the water. This combination of BEHI and NBS scores can then be used to estimate an erosion rate in ft/yr using an empirically-derived curve from Yellowstone, Colorado, or elsewhere depending on the local geology. This erosion rate is then multiplied by the length and height of the bank to get a sediment load in cubic feet/year or tons per year. For more information on the actual process of the BANCS assessment see the WARSSS section of the EPA website (http://water.epa.gov/scitech/datait/tools/warsss/pla_box08.cfm).

A detailed BANCS analysis was performed on approximately half of Kingsbury Creek and a small section of the unnamed tributary that empties into Kingsbury Creek upstream of Interstate 35 (see Figure 49). The whole watershed was not assessed due to time constaints.

This assessment predicts that, for the stream reaches we analyzed, 44% of the predicted sediment load from bank erosion is coming from five stream banks (less than 3% of the stream). These banks have erosion rates of at least 0.1 tons/feet/year and are generally located in areas where there is a slope inflection, such as a B-type channel transitioning to a C-type channel. The results of this analysis suggest that a huge reduction in annual bank erosion could be accomplished by working on a small percentage of the channel. However, the TSS data and the longitudinal snowmelt sampling show that a significant amount of sediment is being sourced upstream of Point Drive (S007-051) and in the unnamed tributary discussed earlier. A BANCS assessment of these two areas is recommended in order to further pinpoint the reaches that are contributing sediment to the system.

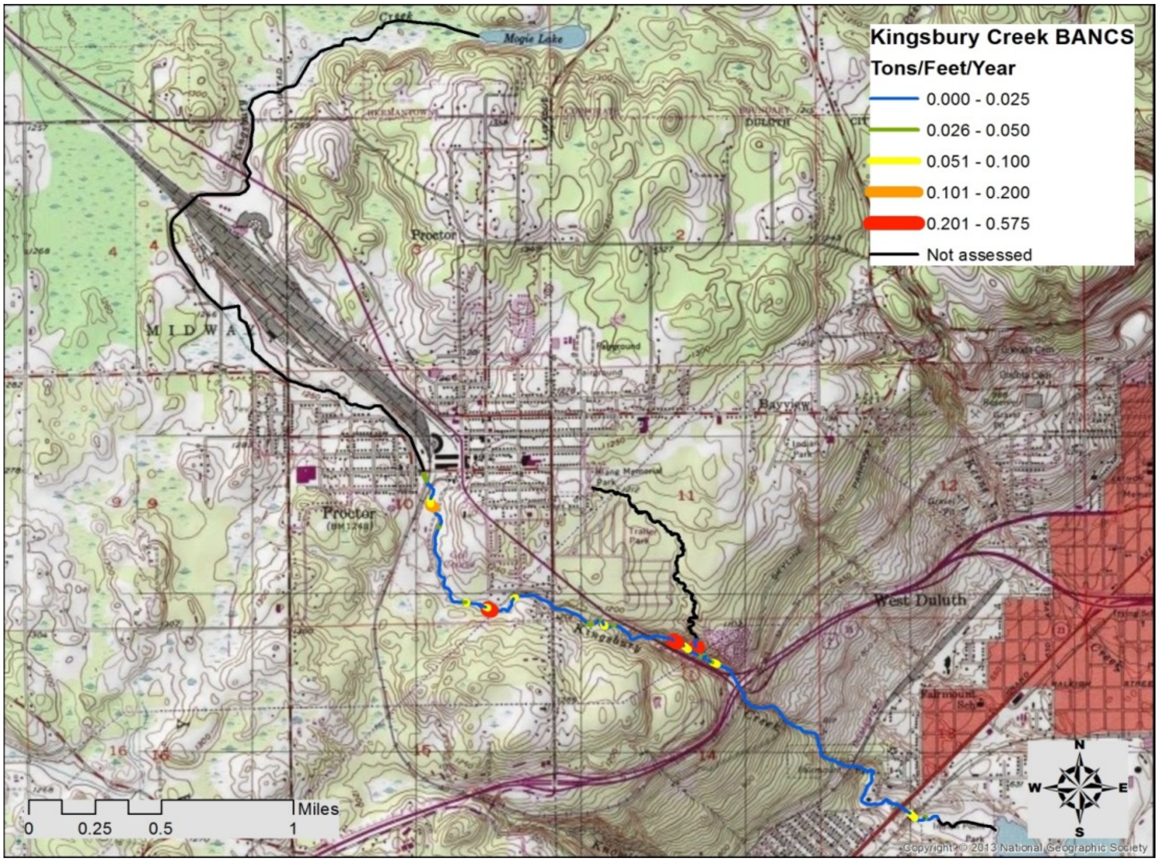


**Figure 49:** Map of Kingsbury Creek BANCS analysis

### Biological Response to Elevated TSS & Turbidity

### Fish Response to TSS

The MPCA biologists have developed TSS tolerance indicator values (TIVs) for all of the fish species found in Minnesota. The TIVs were developed using statewide biological and water chemistry data sets, and are based on the presence, absence, and/or relative abundance of fish taxa found in various water quality conditions. More information on the development and use of TIVs can be found in Section 4.0. For the purpose of evaluating various stressors in this report, TIV values were broken up into percentiles and fish taxa were classified as highly tolerant, moderately tolerant, neutral, moderately intolerant, or highly intolerant. These categories are used here to evaluate the general tolerance of the fish and invertebrate community of Kingsbury Creek to elevated TSS concentrations.

Generally speaking, the FIBI impairment on Kingsbury Creek is due absence or low abundance of fish species that are expected in healthy Northern Coldwater streams (e.g. Brook Trout, Longnose Dace, sculpin sp.) These taxa are considered intolerant or moderately intolerant of elevated TSS concentrations based on TIV values (Table 26). Figure 50 and Table 26 show that the majority of individuals sampled in Kingsbury Creek are “neutral” in terms of their tolerance to elevated levels of TSS. That is, they are neither “tolerant” nor “intolerant.” Neutral species such as Creek Chub, Brook Stickleback, and White Sucker constituted the majority of fish populations in all of the sampling sites. Three stations (09LS003, 95LS036, and 12LS005) had species that are considered “highly intolerant” of elevated TSS, but upon closer inspection of the data these individuals were revealed to be stocked Brook Trout.

Northern Redbelly Dace and Pearl Dace are considered “moderately intolerant” to TSS and were sampled at 98LS003 (1998 and 2012) and 95LS036. Only 8 individuals were collected at the furthermost downstream site in Kingsbury Creek in 2012 (12LS005). Included in this site were two Fathead Minnows (moderately tolerant), four Creek Chubs (neutral), one White Sucker (neutral), and 1 stocked Brook Trout (highly intolerant).

In general, after discounting the stocked Brook Trout, the fish assemblage in Kingsbury Creek shifts slightly toward species that are more “tolerant” as one moves downstream. This may be the result of the longitudinal increases in TSS discussed earlier in this section, but it may also due to differences in thermal regime, habitat type, and habitat related stressors. Considered alone, it is difficult to determine whether the fish community in Kingsbury Creek is generally tolerant or intolerant of TSS because many of the species inhabiting this stream are neither tolerant nor intolerant. However, when considering the water chemistry data and the frequent exceedances of the 10 mg/L TSS standard observed in the lower reaches of Kingsbury Creek, a TSS stressor is a plausible cause of impairment.

In comparison to streams with quality coldwater fish communities, the Kingsbury Creek fish assemblage is much more tolerant of elevated TSS concentrations. Figure 50 compares the TSS TIVs for the five sampling sites with: 1) all the Class 11 (Northern Coldwater) streams in the SLRW that were above the UCL of the IBI threshold 2) the Class 11 streams in the SLRW that scored above the IBI threshold, and 3) streams of all types in the SLRW that were above the UCL of their respective IBI impairment thresholds. As can be seen from the graph, the Kingsbury fish community TIV results show a higher tolerance of TSS than 75% of the Northern Coldwater sites in the SLRW.

**Table 26:** TSS tolerance values of coldwater indicator fish species common to the region, their status in Kingsbury Creek, and a summary of fish surveyed in Kingsbury Creek and their associated TSS tolerance indicator values.

**Coldwater Indicator Species found in Healthy Streams of Northeastern Minnesota**

| Species | TIV | TIV Decile* | TSS Tolerance Category | Status in Kingsbury Ck. |
|---|---|---|---|---|
| Mottled Sculpin | 7.7 | T3 | Moderately Intolerant | absent |
| Longnose Dace | 7.1 | T2 | Highly Intolerant | rare/absent |
| Brook Trout | 7.0 | T2 | Highly Intolerant | rare/absent** |
| Pearl Dace | 9.1 | T3 | Moderately Intolerant | Present (headwaters) |
| Finescale Dace | 11.5 | T4 | Neutral | Present (headwaters) |

**Kingsbury Creek Fish Community**

| Species | TIV | TIV Decile* | TSS Tolerance Category | Number | % of Total |
|---|---|---|---|---|---|
| Creek Chub | 16.0 | T6 | Neutral | 204 | 29.4% |
| White Sucker | 14.2 | T5 | Neutral | 199 | 28.7% |
| Central Mudminnow | 10.9 | T4 | Neutral | 79 | 11.4% |
| Brook Stickleback | 16.6 | T6 | Neutral | 76 | 11.0% |
| Pearl Dace | 9.1 | T3 | Moderately Intolerant | 49 | 7.1% |
| Brook Trout** | 7.0 | T2 | Highly Intolerant | 26** | 3.7%** |
| Northern Redbelly Dace | 9.9 | T3 | Moderately Intolerant | 18 | 2.6% |
| Finescale Dace | 11.5 | T4 | Neutral | 18 | 2.6% |
| Longnose Dace | 7.1 | T2 | Highly Intolerant | 6 | 0.9% |
| Smallmouth Bass | 7.7 | T3 | Moderately Intolerant | 5 | 0.7% |
| Brown Trout | 10.5 | T4 | Neutral | 5 | 0.7% |
| Fathead Minnow | 27.8 | T8 | Moderately Tolerant | 4 | 0.6% |
| Logperch | 6.7 | T2 | Highly Intolerant | 2 | 0.3% |
| Blacknose Dace | 10.2 | T4 | Neutral | 1 | 0.1% |
| Common Shiner | 12.0 | T5 | Neutral | 1 | 0.1% |
| Johnny Darter | 11.6 | T5 | Neutral | 1 | 0.1% |

* Tolerance Indicator Values (TIV) for TSS were arranged in groups by 10th percentiles, or deciles to categorize fish species as highly tolerant (T9-T10), moderately tolerant (T8), neutral (T4-T7), moderately intolerant (T3), or highly intolerant (T1-T2).

** Brook Trout abundance in Kingsbury Creek is influenced by regular stocking by DNR. Very little natural reproduction has been documented in this stream.

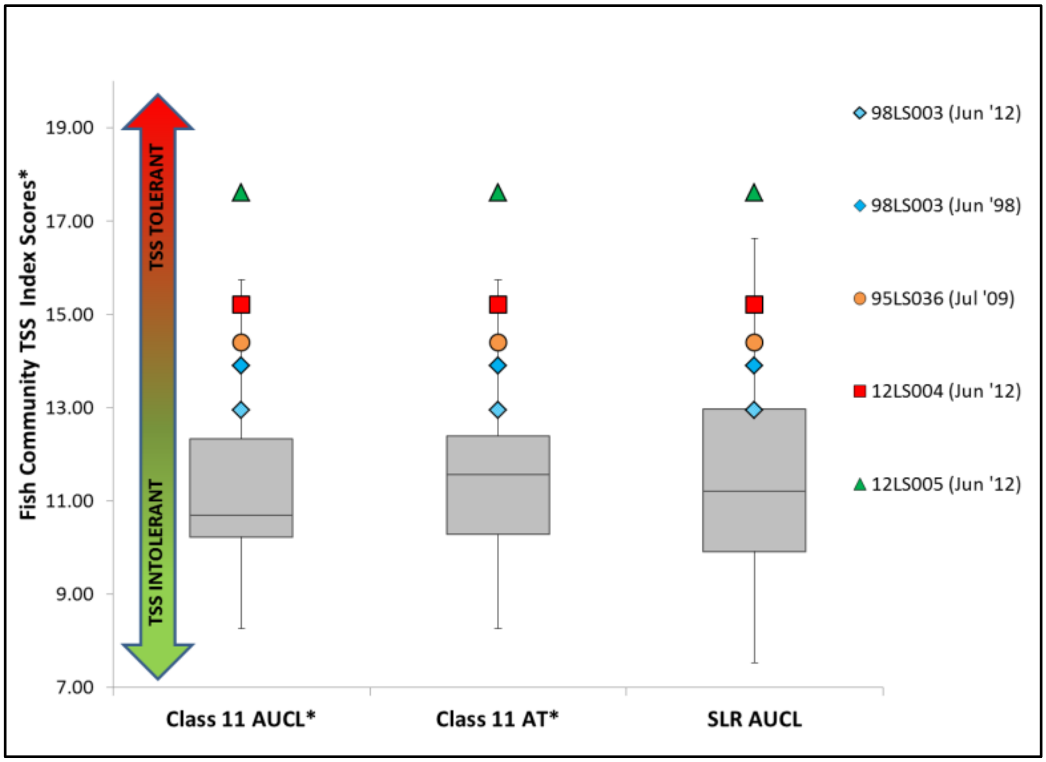


**Figure 50:** Fish community TSS TIVs for Kingsbury Creek, compared to unimpaired streams

### Macroinvertebrate Response to TSS

Similar to the approach taken for evaluating fish response to TSS, the macroinvertebrate taxa in Kingsbury Creek were categorized into four classes based on their tolerance to elevated levels of TSS. These were: 1) highly tolerant, 2) moderately tolerant, 3) moderately intolerant, and 4) highly intolerant. Figure 51 shows that the taxa found in Kingsbury Creek are generally tolerant of elevated levels of TSS. The biological monitoring site 98LS003 was sampled in 1998 and 2009, and the results show that the invertebrate community shifted to more tolerant taxa in the 11 years between sampling. A total of 7 of 16 taxa (44%) in 1998 were either moderately or highly intolerant of TSS. That number dropped to 2 out of 13 taxa (15%) in 2009. Moving downstream, a similar percentage of tolerant taxa were sampled at 95LS036 in 2009. Half as many taxa were sampled at that site, with 1 out of 6 (17%) taxa regarded as "highly intolerant."

TSS index scores, which are a composite value of taxa tolerance and relative abundance measures, are clearly showing that the invertebrate assemblage of Kingsbury Creek is more tolerant of TSS in comparison to high quality stations of the same MIBI class. The box plots in Figure 52 compare data for a series of TSS-related metrics between Kingsbury Creek monitoring stations and 1) Class 8 stations that scored above the UCL of the MIBI threshold, 2) Class 8 stations that scored above the impairment threshold, and all SLRW stations that scored above the UCL of the MIBI threshold. Both of the Kingsbury Creek stations monitored recently exceed 75th percentile values (or greater) for measures of TSS tolerance. The tolerance value for the 1998 sampling score much better, however, and fall at or below the median tolerance value for higher quality biologic communities. These data show a high invertebrate TSS tolerance, with an increasing tolerance over time, and support adding TSS as a stressor to macroinvertebrate communities in Kingsbury Creek.

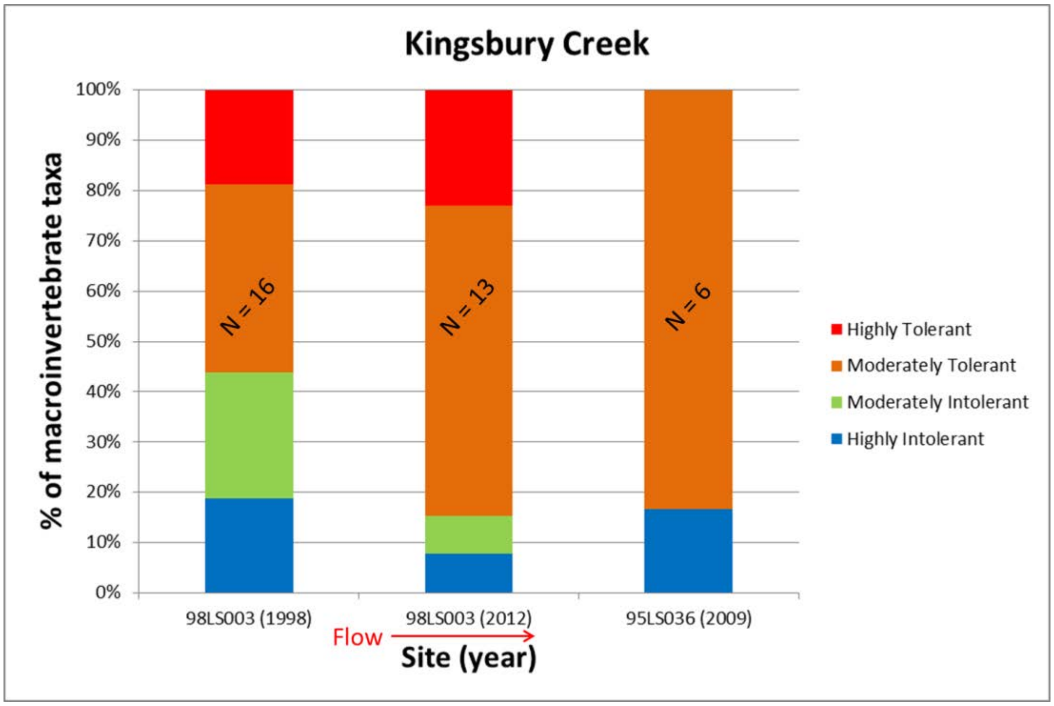


**Figure 51:** TSS tolerance of macroinvertebrate assemblages by site in Kingsbury Creek (n = number of taxa)

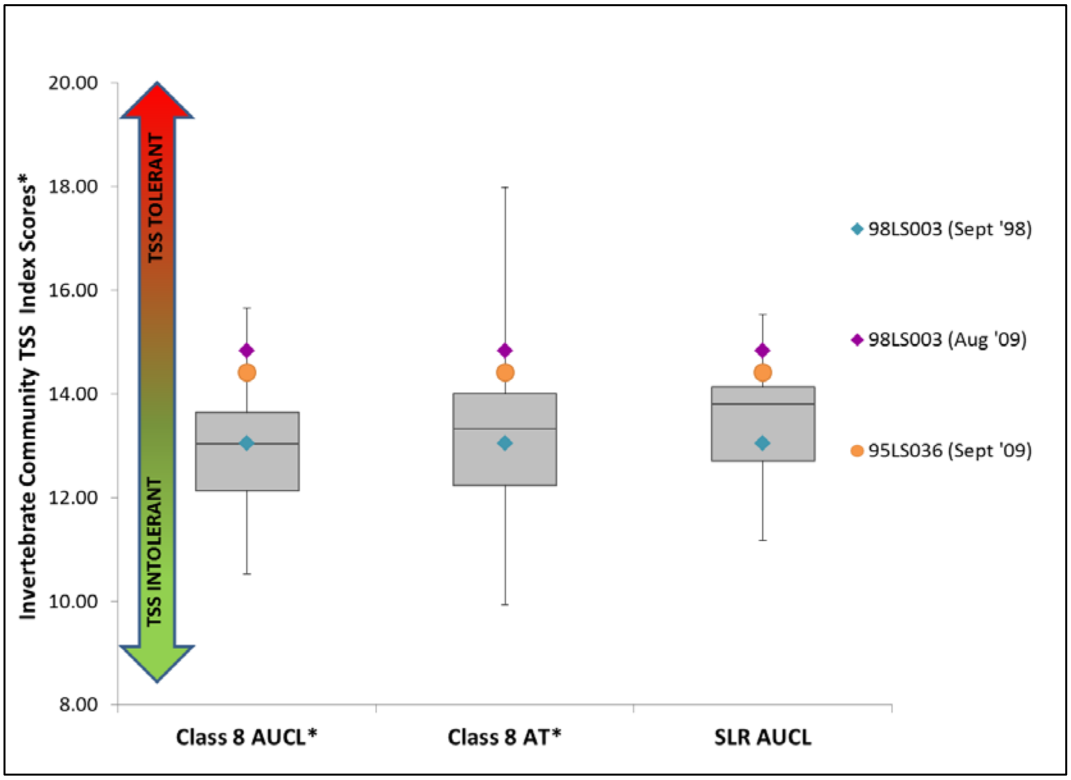


**Figure 52:** Macroinvertebrate community TSS TIVs for Kingsbury Creek compared to unimpaired streams

**Summary: Is TSS a Stressor to Aquatic Life in Kingsbury Creek?**

TSS concentrations in Kingsbury Creek are highly elevated during spring snowmelt, as well as precipitation events during the spring, summer, and fall months. Streambank and bluff erosion, unstable gully and ravine tributaries, and overland runoff from urban areas are all contributing excess sediment to this creek, which is degrading water quality and physical habitat. Although the observed spikes in TSS and turbidity are rather short in duration, the amount of sediment being transported via suspended and bedload is significant enough to limit quality habitat for sensitive fish and macroinvertebrate taxa. Elevated TSS concentrations, and sediment impacts as a whole (including sediment deposited within the stream channel), is therefore considered one of the stressors impairing aquatic life in this system.

### 5.2.4 Chloride Toxicity & Specific conductivity

Increased specific conductivity was identified as a candidate stressor to aquatic life in Kingsbury Creek based on existing data and the urban characteristics of its watershed. The watershed of Kingsbury Creek includes portions of the cities of Proctor and Duluth, as well as two major highway corridors (US 53 and I-35). Overall, impervious surfaces account for approximately 10% of the total watershed area. Elevated chloride and specific conductivity in Kingsbury Creek were previously cited as potential stressors to aquatic life in the MPCA’s snowmelt study of Duluth Streams (Anderson et al. 2000) and the Duluth Streams website (lakesuperiorstreams.org).

#### Chloride: Results and Applicable Water Quality Standards

Minnesota’s water quality standard for chloride is a chronic value of 230 mg chloride/liter, implemented as a four-day average concentration, and an acute (maximum concentration) of 860 mg chloride / liter, implemented as a one-day average concentration. More information on the chloride standard can be found in Section 3.1.7.

Kingsbury Creek is not currently listed as impaired for exceeding the chloride standard. Much of the chloride data collected on Kingsbury Creek was not made available during the most recent assessment process. As a result, it was determined that insufficient data were available to assess Kingsbury Creek for this parameter. However, existing monitoring data shows exceedances of both the chronic and acute thresholds that were established to protect aquatic life from the adverse effects of chloride (Figure 53). Like most urban streams, chloride concentrations in Kingsbury Creek experience a spike during mild to moderate snowmelt events. The existing data shows exceedances of water quality standards in March and November. No data are currently available for the months of December through February, which is often the season where annual maximums are observed in Duluth urban streams due to road salt runoff and low stream flow. Chloride concentrations between 50-100 mg/L are frequently observed in Kingsbury Creek during summer low flow periods, suggesting that road salt applied during the winter months may be infiltrating into the groundwater aquifer that provides baseflow to the creek during drier periods.

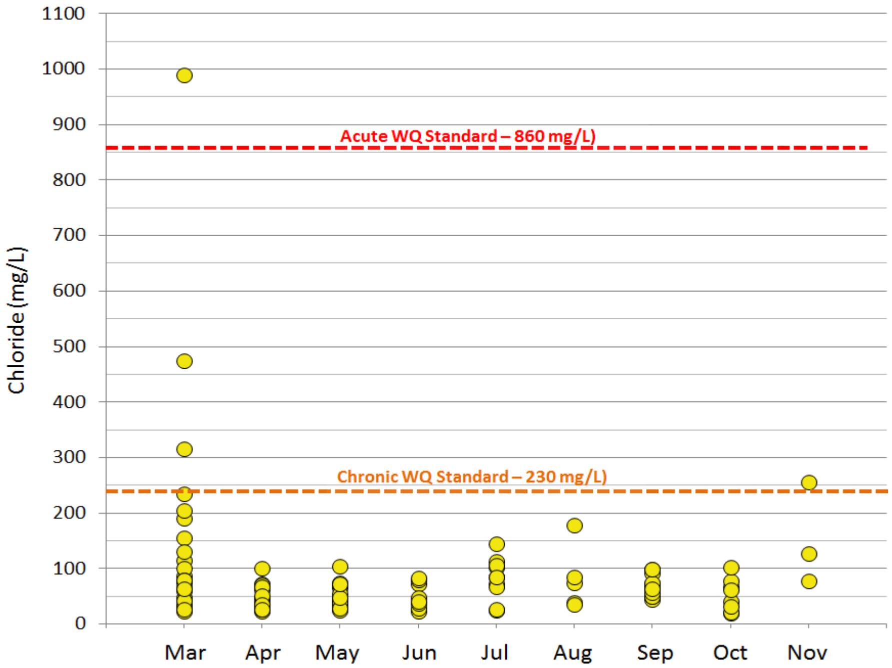


**Figure 53:** Kingsbury Creek chloride grab sampling results arranged by calendar month (all monitoring sites combined).

### Sources and Pathways: Chloride

The sources and pathways of chloride within the Kingsbury Creek Watershed are believed to be very similar to Miller Creek. See Section 5.2.2 to learn more about these sources. Specific "hot spots" for chloride in Kingsbury Creek likely include the city of Proctor, the U.S. Highway 2 and I-35 highway corridors in the middle reaches of the watershed, and the Grand Avenue crossing near the mouth of the creek. Synoptic monitoring at these locations and additional sites within the watershed is recommended to better understand chloride inputs to Kingsbury Creek.

**Figure 54:** Large piles of snow laden with road salt and grit adjacent to Kingsbury Creek (4/21/14). Yellow arrows indicate location of creek.

### Specific Conductivity as a Surrogate Measure of Chloride

Both instantaneous (point) and continuous measurements of specific conductivity are available for several Kingsbury Creek monitoring locations. Point observations are primarily available from early March through November. All available point measurements of conductivity are shown in Figure 55 along with the monthly averages. The average monthly instantaneous data show conductivity peaking in March during snowmelt runoff, and again in July and August during low flow conditions. No instantaneous measurements of specific conductivity are available for the months of December through February.

Specific conductivity levels in Kingsbury Creek have been recorded continuously (15 to 30 minute intervals) above the Lake Superior Zoo by the University of Minnesota-Duluth NRRI since 2002. Due to difficult stream and weather conditions, some of the data for certain years are incomplete. However, these continuous data provide a good record of conductivity levels in the creek for periods where there are no point measurements (mid to late winter). Specific conductivity in Kingsbury Creek follows a fairly consistent pattern in years where continuous data are available. The 2006 data are shown in Figure 57 as an example of the annual specific profile commonly seen in this stream. In 2006, conductivity levels peaked around 3,100 µS/cm in early March. Other peaks can also be seen during the winter months in this data set (2,700 µS/cm in February; 1,750 µS/cm in January). Each of these peaks corresponds with snowfall events when road salt is applied to urban areas around the cities of Duluth and Proctor.

The exceedance probability curve for the 2006 data set shows that the observed spikes in conductivity are extremely short in duration and occur only several times on an annual basis. For example, specific conductivity exceeded 1,000 µS/cm only 12.6% of the time in 2006, while conductivity greater than 1500 µS/cm occurred less than 1% of the time (Figure 58). Based on data from 2006 and other years, aquatic life in Kingsbury Creek typically experience conductivity levels in the range of 400 to 700 µS/cm for much of the year. This range is lower than other urbanized streams in the city of Duluth (e.g. Miller Creek), but still remains elevated above background conditions for the area.

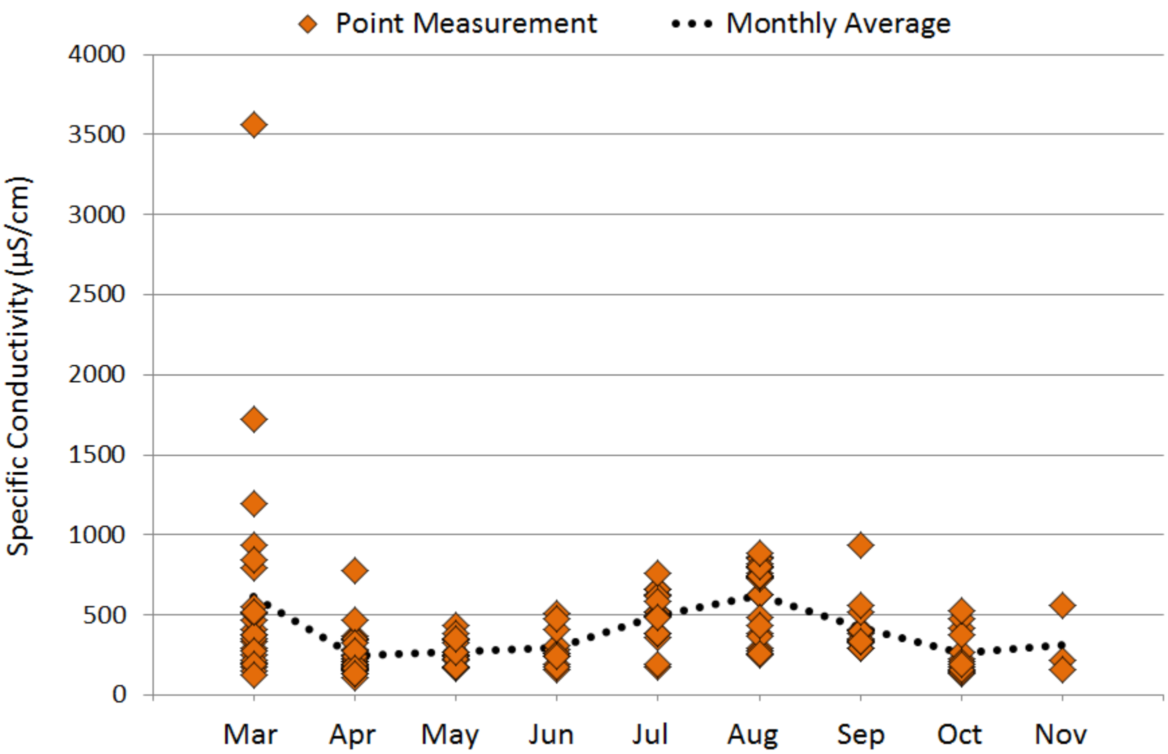


**Figure 55:** Point measurements of specific conductivity in Kingsbury Creek by calendar month

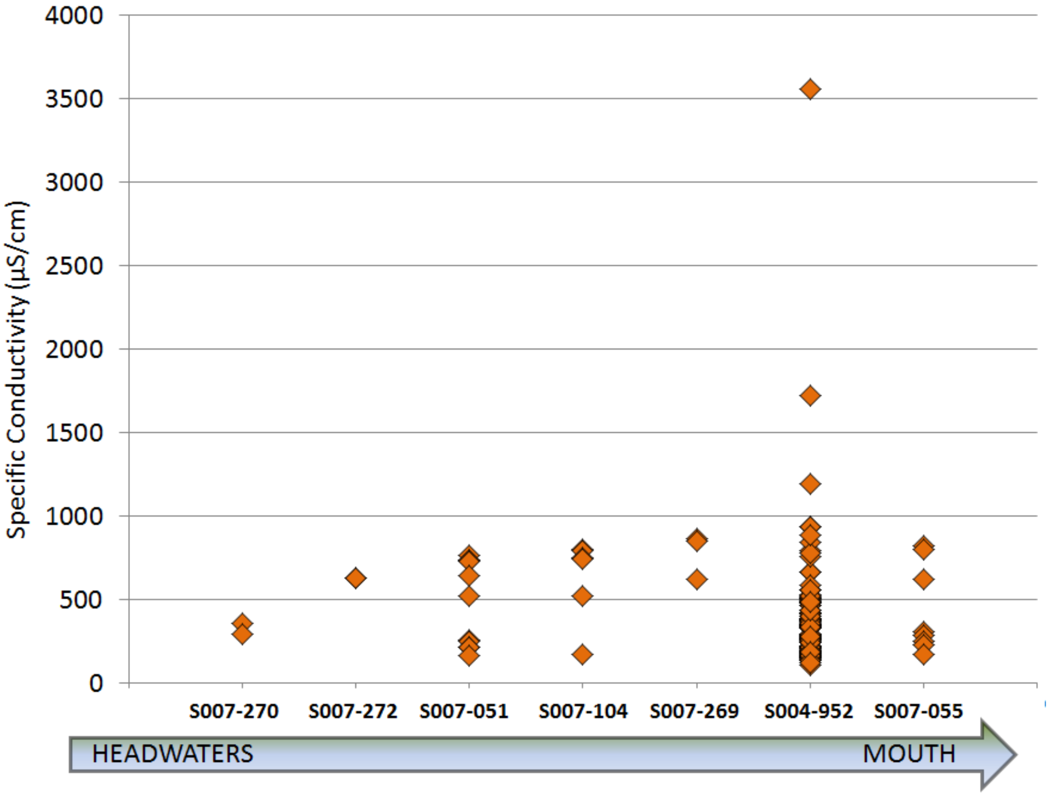


**Figure 56:** Point measurements of specific conductivity in Kingsbury Creek longitudnally by monitoring station

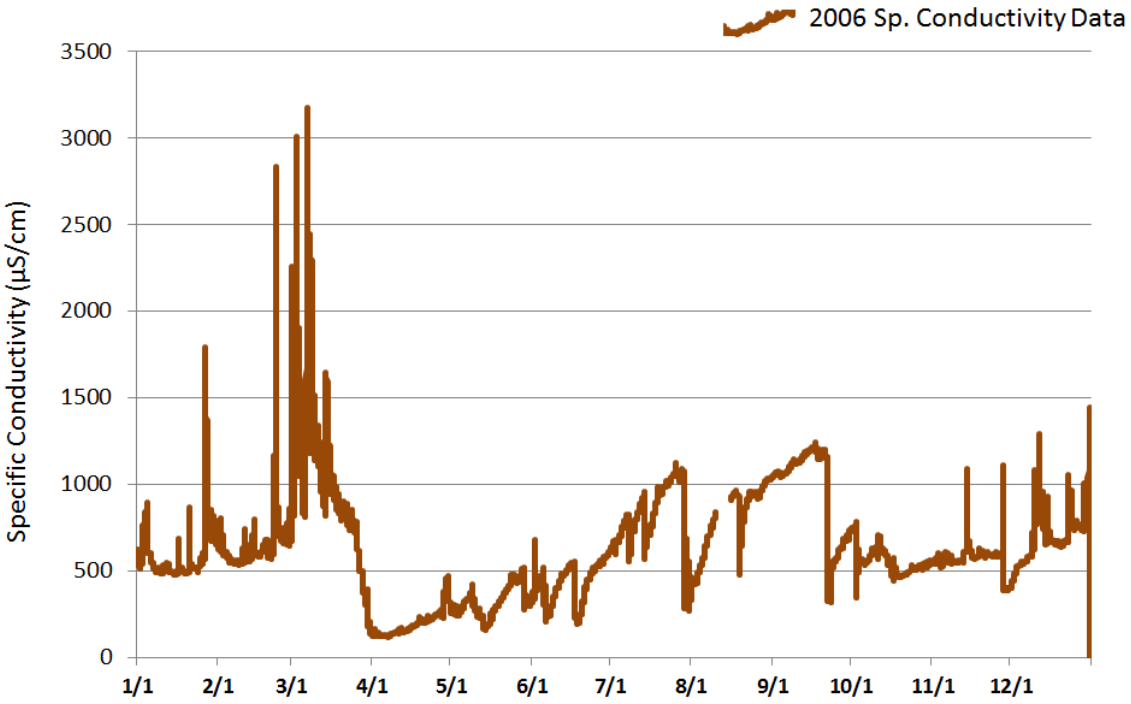


**Figure 57:** Continuous specific conductivity data collected from Kingsbury Creek station S004-952 in 2006

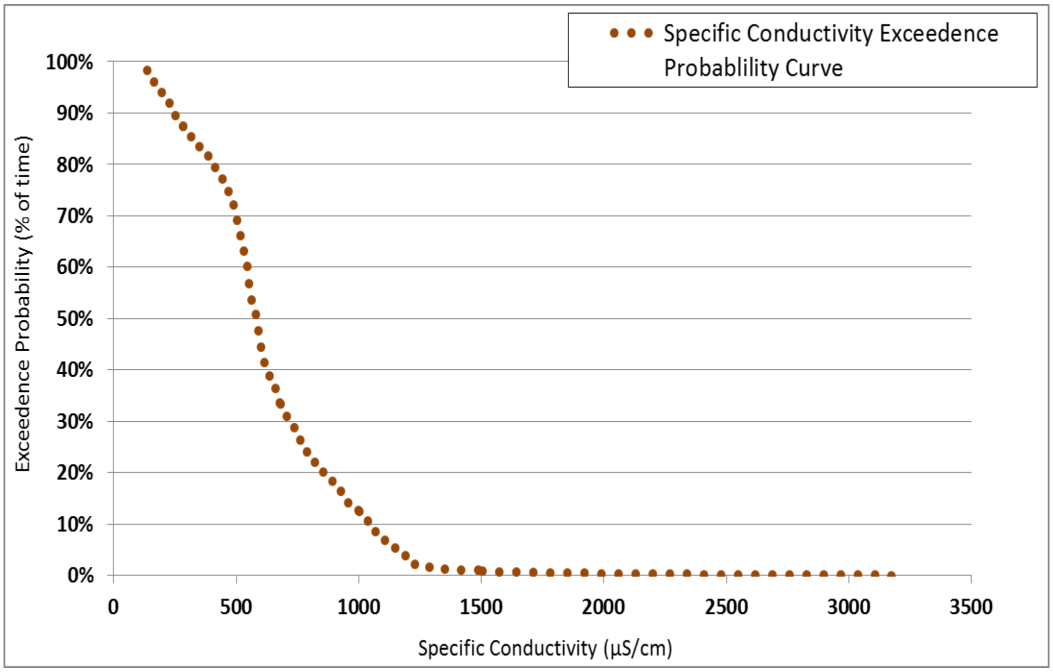


**Figure 58:** Exceedance probability curve for the 2006 specific conductivity data set from station S004-952

## Chloride / Specific conductivity Relationship in Kingsbury Creek

In many instances, a clear relationship can be established between dissolved salts and specific conductance (Allan 1995). Due to differences in local geology, land-use, and other environmental factors, the proportion of the various dissolved ions can vary significantly by watershed. In Kingsbury Creek, a good relationship is observed between chloride concentrations and specific conductance

(Figure 59). A regression of 108 paired chloride and specific conductivity results shows a clear positive relationship, although several early season snowmelt samples fall off of the regression line due to dilution factor from high streamflow. This relationship clearly indicates that chloride concentration is a primary driver of specific conductivity levels in Kingsbury Creek.

The regression equations in Figure 59, developed using data from Kingsbury Creek, can be used to derive estimated chloride concentrations based on specific conductivity readings. In addition, NRRI developed an equation to determine chloride concentrations via conductivity readings by adding city road salt to water from three streams in the Duluth metropolitan area (DuluthStreams.org; Figure 17). The two equations derived from these data sets were applied to specific conductivity readings collected during continuous monitoring efforts by NRRI at the Kingsbury Creek monitoring station just upstream of the Lake Superior Zoo. Elevated specific conductivity readings (>1500 µS/cm) were selected from the continuous data spanning the years 2002-2013 to estimate chloride concentrations. The results, shown in Table 27, clearly show that chloride concentrations (estimated) in Kingsbury Creek exceed the acute water quality standard of 860 mg/L fairly frequently, and in some instances, by a wide margin. If these estimations prove to be accurate after additional sampling, chloride toxicity can be considered a significant threat to sensitive aquatic life in Kingsbury Creek. Winter grab samples during the Duluth Urban WRAPS project (set to begin in 2016) are recommended to verify these estimated chloride concentrations.

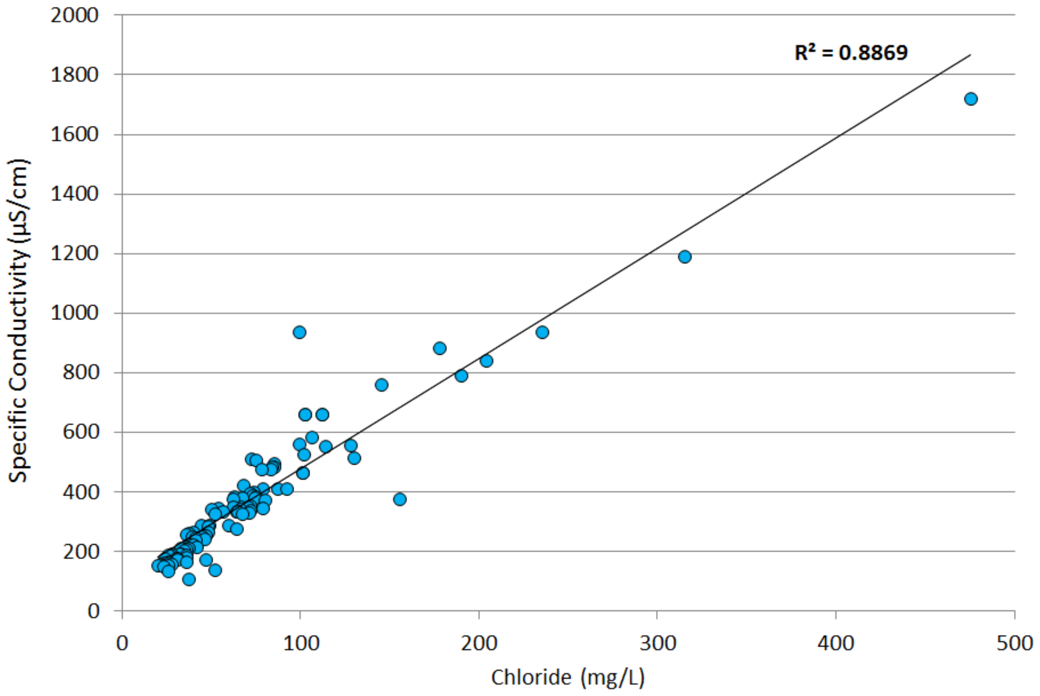


**Figure 59:** Scatter-plot regression of paired chloride and specific conductivity measurements from Kingsbury Creek

**Table 27:** Estimated chloride concentrations based on chloride - sp. conductivity regression equations developed by lab experiments (duluthstreams.org) and paired chloride – sp. conductivity results from Kingsbury Creek.

| Date | Sp. Conductivity (µS/cm) | Chloride Equivalent in mg/L (NRRI Curve) | Chloride Equivalent in mg/L (Kingsbury Data Curve) |
|---|---|---|---|
| **12/24/2005** | 2442 | 729.4 | 630.3 |
| **2/23/2006** | 2835 | **861.8** | 736.2 |
| **3/3/2006** | 2995 | **915.7** | 779.4 |
| **3/7/2006** | 3172 | **975.4** | 827.1 |
| **11/12/2008** | 9885 | **3237.6** | **2636.2** |
| **1/4/2009** | 3324 | **1026.6** | **868.0** |
| **1/11/2009** | 3011 | **921.1** | 783.7 |
| **2/10/2009** | 2793 | 847.6 | 724.9 |
| **2/26/2009** | 3669 | **1142.9** | **961.0** |
| **3/14/2009** | 3031 | **927.8** | 789.1 |
| **1/24/2010** | 2451 | 732.4 | 632.8 |
| **2/18/2010** | 1989 | 576.7 | 508.2 |
| **12/16/2011** | 1899 | 546.4 | 484.0 |
| **12/28/2011** | 1894 | 544.7 | 482.6 |
| **1/8/2012** | 2517 | 754.6 | 650.5 |

## Effects of Elevated Chloride / Specific Conductivity on Aquatic Life

### Chloride

MBI (2012) identified chloride-sensitive macroinvertebrate taxa based on a statewide water chemistry and biological data set. Specific chloride thresholds were not established through this study, however, common macroinvertebrate taxa to Minnesota were classified as sensitive (“S”), Tolerant (“T”), or neither tolerant nor sensitive.

Overall, there is a general lack of chloride sensitive macroinvertebrate taxa in Kingsbury Creek. A very small population of chloride sensitive individuals from the genus *Gerris* (pond-skaters) were observed at station 98LS003, accounting for less than 1% of the total individuals collected at that station. Chloride sensitive macroinvertebrate taxa were absent during all of the remaining four visits to Kingsbury Creek monitoring sites. Chloride tolerant macroinvertebrate taxa were present in relatively large numbers at station 98LS003 during site visits in 1998 and 2011, when 21% and 41% of the invertebrate population were chloride tolerant taxa, respectively. The other three monitoring stations, which are located further downstream from 98LS003, lacked chloride sensitive taxa but had lower percentages of chloride tolerant taxa (8 – 16%).

Although it appears that chloride sensitive invertebrate taxa are more or less absent from Kingsbury Creek, some caution needs to be used in evaluating this stressor with these taxa tolerance values. The influence of confounding stressors may be a factor in that many of the taxa that are considered chloride-tolerant may also be tolerant of other water quality or physical stressors (e.g. low DO or habitat degradation). Comparing data from Kingsbury Creek to other streams in the Duluth area and North Shore of Lake Superior helps provide some context for these tolerance analyses. Several streams with rural and/or relatively undeveloped watersheds included in this analysis (Big Sucker Creek, Mission

Creek, Hay Creek) also supported macroinvertebrate communities with a high percentage of chloride tolerant individuals (see appendix B).

Fish species classified as “tolerant” to chloride by Bouchard (2013) were abundant at many Kingsbury Creek monitoring stations. Creek Chub and Brook Stickleback were the two chloride tolerant species most commonly observed, accounting for between 80-100% of the total fish community at several stations. However, several chloride sensitive species were also observed in small populations in Kingsbury Creek, including Northern Redbelly Dace, Pearl Dace, and Brook Trout. These species were mainly limited to stations in the headwaters, and exposure to chloride may be reduced in magnitude at these locations. Also, many or all of the Brook Trout observed were the result of regular stocking efforts by DNR and should not factor heavily into this species sensitivity analysis. Overall, the fish community of Kingsbury Creek supports fewer fish taxa that may be sensitive of chloride in comparison to streams with less urban development within their respective watersheds.

Appendix B summarizes fish data from nine streams located within the city of Duluth, in the rural outskirts of Duluth, or further up the North Shore of Lake Superior. This selection of streams represents a range of land-use, from relatively pristine, to rural residential/agricultural, to heavily urbanized. Out of 32 total visits to stations on these streams, results from three heavily urbanized streams (Kingsbury Creek, Miller Creek, and Chester Creek) occupy the top 14 positions in terms of the percent of the fish community that are chloride tolerant individuals. The shift towards a fish community dominated by tolerant species such as Creek Chub, Central Mudminnow, and Brook Stickleback in the urbanized streams is very clear. However, it is difficult to link this community shift to chloride toxicity alone given the numerous other stressors present in these urban watersheds. Several of these streams support chloride sensitive minnow species or Brook Trout below areas of high density urban development, which somewhat weakens the case of chloride as the lone stressor, but does not provide evidence to refute it as a stressor given the relatively small population of these sensitive taxa.

## Biological Response to Elevated Specific Conductivity

The effects of elevated conductivity on aquatic life were evaluated using data from Minnesota streams and scientific literature. A summary of this analysis is presented in Section 3.1.5. Based on this work, several biological metrics were selected to evaluate specific conductivity as a stressor in Kingsbury Creek (Table 28).

**Table 28:** Biological metrics selected to evaluate specific conductivity as a stressor in Kingsbury Creek.

| Metric | Response to Increased Specific conductivity / Conductivity | Source |
|---|---|---|
| **EPT Richness** | Decrease | Roy et al (2003); Echols et al (2009); Johnson et al (2013) |
| **Macroinvertebrate Taxa Richness** | Decrease | Johnson et al (2013) |
| **Ephemeroptera Richness** | Decrease | Pond (2004); Hassel et al (2006) |
| **Tolerance Values (Sp. Conductivity)** | Increase | MPCA (2014) |

**Taxa Richness**

With the exception of 1998 data from station 98LS003, macroinvertebrate taxa richness in Kingsbury Creek is relatively low compared to other streams in the SLRW, including streams of the same size and MIBI classification (Figure 61). In 2011, 13 fewer taxa were observed at station 98LS003 compared to the original sampling event in 1998. This decrease suggests that there are short or long term stressors present in this watershed that are creating varying conditions from year to year at this monitoring station. Potential causes include several stressors that are commonly observed in urban watersheds, such as acute stress from specific conductivity or lack of streamflow due to altered watershed land-use. The relatively low taxa richness counts observed in the 2009 and 2011 sampling events provide supporting evidence of specific conductivity as a stressor in Kingsbury Creek, but this stressor cannot be diagnosed based on these results, largely because habitat conditions are also poor at these monitoring stations.

**EPT Richness**

EPT taxa richness in Kingsbury Creek is considerably lower than results observed from high quality streams of the same MIBI class (Figure 61; class 8 - Northern Coldwater Streams). The highest EPT count in Kingsbury Creek (9 taxa) was observed at station 98LS003 in 1998. More recent monitoring resulted in lower EPT taxa counts at 98LS003 (6 taxa) and 95LS036 (5 taxa). Overall, it can be concluded that Kingsbury Creek supports significantly fewer EPT taxa compared to the majority of healthy rivers and streams in the SLRW. Considering that many EPT taxa are sensitive to changes in specific conductivity, the lack of EPT taxa observed in Kingsbury Creek provides supporting evidence in support of specific conductivity as a stressor, yet again; it is difficult to rule out other stressors that can cause the same symptoms (e.g. low DO, poor physical habitat).

**Ephemeroptera Richness**

Kingsbury Creek lacks an abundance and diversity of Ephemeroptera (mayfly) individuals, which provides further evidence in support of specific conductivity as a stressor. Less than five mayfly taxa were observed during all macroinvertebrate sampling events on the creek. These results are below the 25th percentile values observe at high quality reference stations across the SLRW (Figure 61). It is important to note that the difference between mayfly richness in Kingsbury Creek and many high quality streams is only 2-3 taxa. Several high quality coldwater streams located on the outskirts of Duluth (Big Sucker Creek, East Branch Amity Creek) also had relatively low mayfly richness values that were only slightly higher than results from Kingsbury Creek. Miller Creek, which is another impaired coldwater stream within the city limits of Duluth, supports even fewer mayfly taxa than Kingsbury Creek and likely experiences a greater impact from chloride and associated stressors (specific conductivity). Miller Creek is currently listed as impaired for violating the water quality standard for chloride.

**Community Tolerance Values**

Community TIV results for specific conductivity for Kingsbury Creek monitoring sites are plotted with results from reference streams in Figure 61. The majority of the TIV results for both fish and macroinvertebrates plot above the 75th percentile values observed at the high quality SLRW stations. Based on these results, it can be concluded that fish and macroinvertebrate communities of Kingsbury Creek are moderately tolerant of elevated specific conductivity compared to high quality sites in the SLRW.

Overall, a very small percentage of the macroinvertebrate population observed in Kingsbury Creek can be considered “highly tolerant” of elevated specific conductivity (Figure 60). However, several sites had large numbers of Oligochaeta (aquatic worms) and Hirudinea (leeches), which are listed as “no tolerance value” because MPCA has yet to develop a tolerance value for these taxa. Most members of Oligochaeta and Hirudinea are tolerant of many forms of pollution and habitat degradation, so it can be assumed that the macroinvertebrate community of Kingsbury Creek is more tolerant of elevated specific conductivity than depicted in Figure 60. A very low percentage of the macroinvertebrate community of Kingsbury Creek can be considered highly or moderately intolerant of elevated specific conductivity levels.

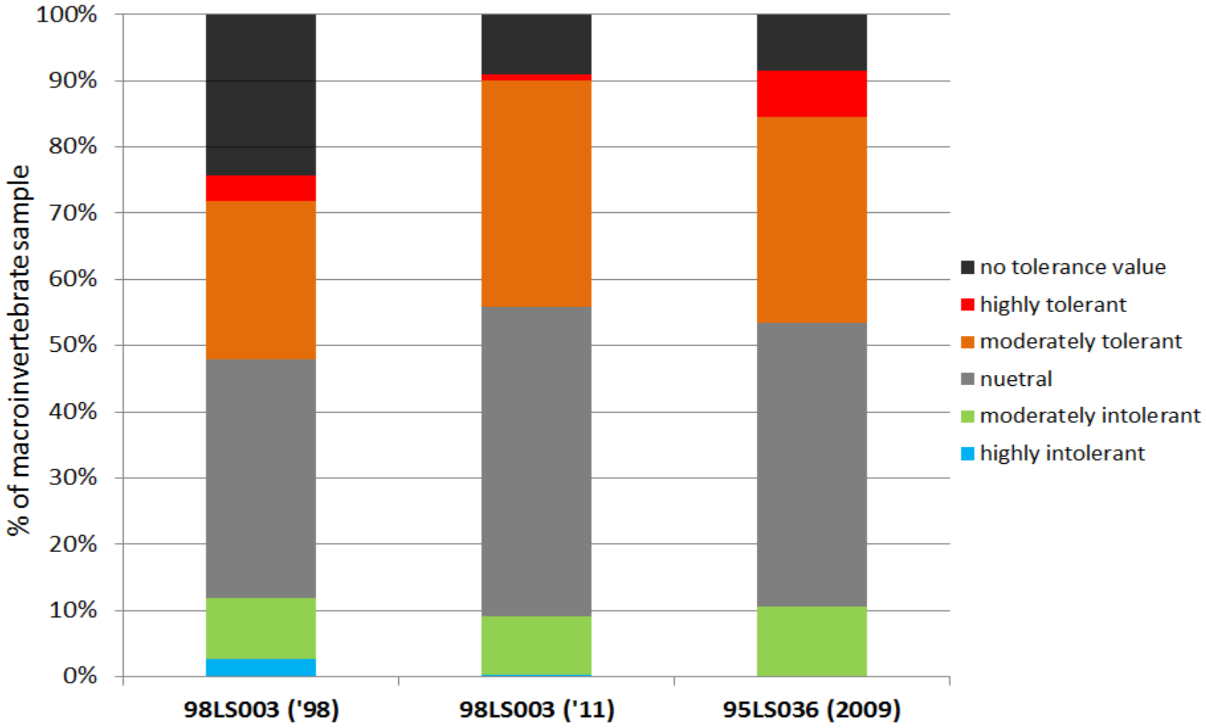


**Figure 60:** Macroinvertebrate community tolerance to specific conductivity in Kingsbury Creek, based on taxon specific TIV values

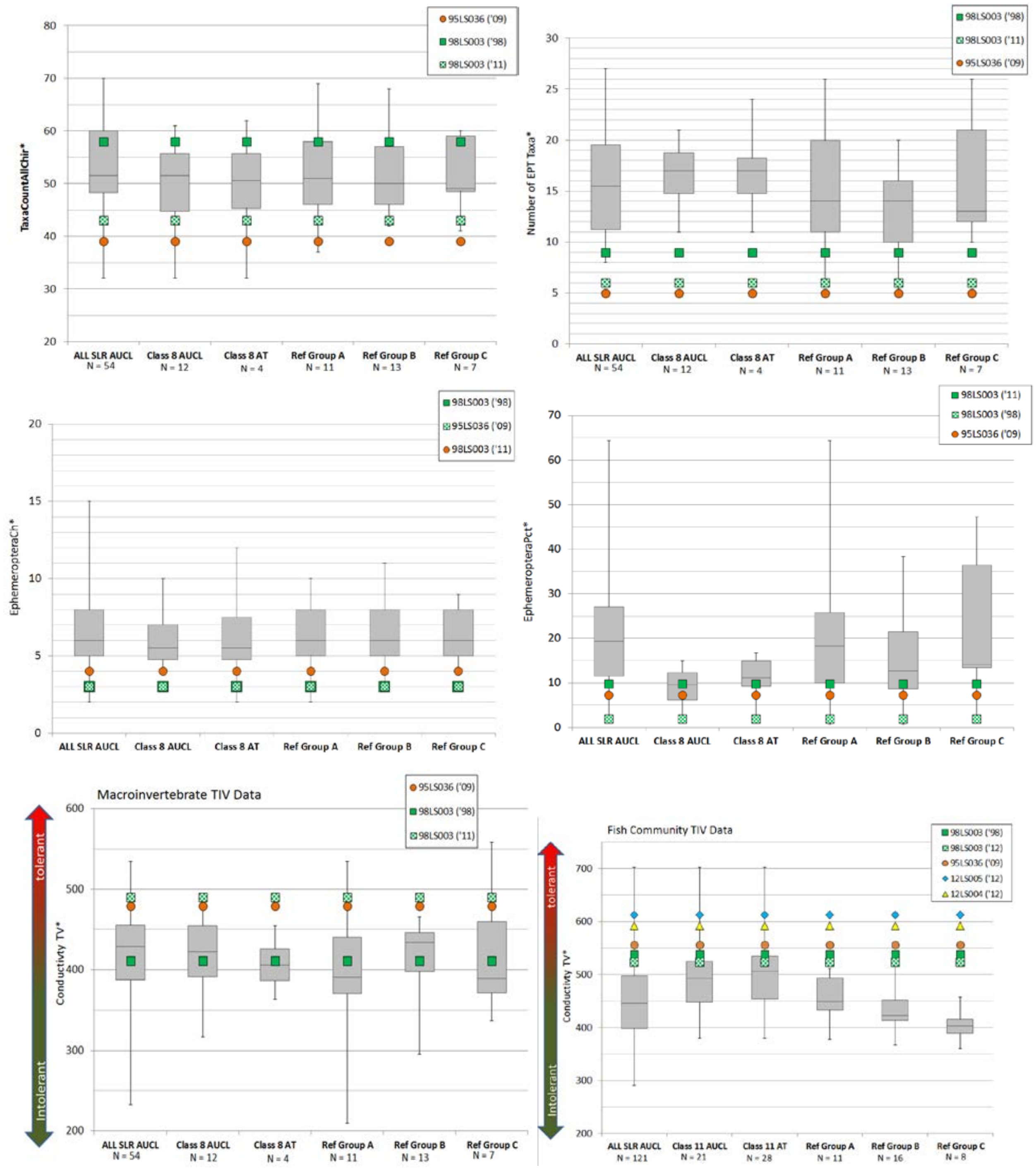


**Figure 61:** Collection of box-plot distribution graphs comparing biological response data from Kingsbury Creek to results from SLRW reference streams. ***** See Section 4 for explanation of TIVs **SLR=**St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above FIBI Threshold

## Summary: Is chloride / specific conductivity a stressor in Kingsbury Creek?

Elevated chloride concentrations and associated short and long tern increases in specific conductivity are clearly evident in the available water chemistry data for Kingsbury Creek. Violations of the chronic and acute chloride water quality standard have been observed in this stream and continuous measurements of specific conductance show that chloride concentrations are often higher than the values observed thus far during grab sampling events. The biological response data evaluated for this stressor shows symptoms that are similar to effects seen in other streams (lack of EPT taxa, low taxa richness, lack of sensitive taxa), but due to the potential for confounding stressors it is difficult to link these responses to elevated specific conductivity alone.

The sources and pathways of these stressors within the watershed are well understood, but additional data would be beneficial. Road salt application during the winter months is clearly the primary driver of this potential stressor, and based on the limited data available, conditions in the stream during this time may be acutely toxic for sensitive forms of aquatic life. Additional monitoring during this critical time (December through February) is recommended, particularly at locations upstream and downstream of areas where intensive road de-icing efforts are occurring. Additional work is also recommended to explore chloride concentrations in the groundwater, stormwater ponds, and riparian corridors connected to Kingsbury Creek. Mid-summer chloride concentrations and specific conductivity levels are also elevated above background conditions and contamination of the groundwater and riparian corridors are a likely source.

Chloride and specific conductivity should be considered a potential stressor in Kingsbury Creek, but additional monitoring focused on the critical periods (mid to late winter) is recommended before a TMDL is initiated. The Duluth urban WRAPS project will have a chloride component, and monitoring of Kingsbury Creek should be included in that effort.

### 5.2.5 Poor Physical Habitat Conditions

**Sources and pathways of degraded habitat in Kingsbury Creek**

Kingsbury Creek biological monitoring stations 98LS003 and 95LS036 both received “fair” ratings based on Minnesota Stream Habitat Assessment (MSHA) scores. Surrounding land use scored poorly due to the urban and recreational development adjacent to the riparian area. Substrate also scored poorly due to the dominance of sand and moderate embeddedness on the channel substrate. The stream at both biological monitoring stations was dominated by “run” features, with very few glide, riffle, and pool habitats present. This general lack of habitat variability is expected in ditched or channelized streams as Kingsbury Creek is at biological monitoring station 98LS003. Among the other attributes of Kingsbury Creek that prevented a good MSHA score were sparse cover and moderate channel stability and development.

Only one PSI rating was computed for both 98LS003 and 95LS036 due to their close proximity. Scores for Kingsbury Creek are typical of a ditched channel that is undergoing channel evolution (see photo in Figure 64), with characteristics such as bank erosion, debris jams, and loose bottom sediments. Kingsbury Creek had a score of 104, which corresponds to a rating of “unstable” for the potential E6 stream type. It is likely that the consequences of channelization, such as a lack of habitat variability, excess fine particle deposition, and pool filling, are causing habitat degradation in Kingsbury Creek.

## Biological effects of degraded habitat

### Fish Response to Habitat

The fish community in Kingsbury Creek is generally tolerant of degraded habitat. The number of non-tolerant fish/meter at 98LS003 in 2012 was higher than the other three biological monitoring stations shown below, and was more than 25% of the unimpaired Class 11 stations in the SLRW. Unfortunately, that number is artificially high due to the presence of 24 stocked Brook Trout. Only six non-tolerant fish were collected other than the Brook Trout (1 Northern Redbelly Dace and 5 Pearl Dace). The other three sampling efforts resulted in much lower numbers of non-tolerant fish, mostly due to the lower numbers of Brook Trout collected. The number of non-tolerant fish/meter at those three sites all fell well below 75% of the Class 11 AT stations in the watershed.

Kingsbury Creek also did not compare well to unimpaired Class 11 streams in the SLRW in terms of the number of habitat-sensitive taxa. All four sampling efforts found the same number of these specific taxa. No benthic insectivore, darter, sculpin, or round-bodied sucker species were found in Kingsbury Creek – well below the SLRW Class 11 AT stations. Stocked Brook Trout represented the one piscivorous and gravel spawning species found at all four sites. Similarly, White Sucker was the only riffle dwelling taxa found at the biostations. One piscivorous species (Northern Pike) was sampled, which is the median for unimpaired Class 11 streams in the watershed. One riffle-dwelling and gravel spawning taxa (White Sucker) was present – also equal to or below 75% of the comparison streams. Brook Trout are stocked in the stream by Minnesota DNR, and White Suckers can thrive in streams with degraded habitat. Therefore the two habitat-sensitive species found in Kingsbury Creek are not good indicators of quality habitat. The taxa makeup of the fish community other than these two species is evidence that habitat is rather degraded in Kingsbury Creek.

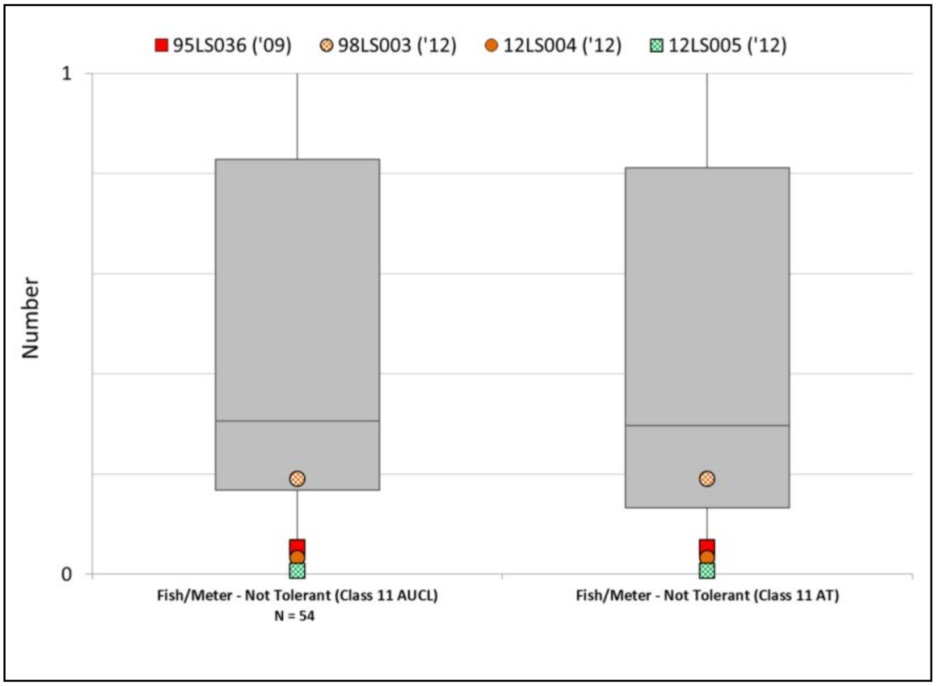


**Figure 62:** Fish/meter (excluding tolerant taxa) in Kingsbury Creek compared to high quality SLRW Class 11 stations. (* see Table 22 for acronym descriptions)

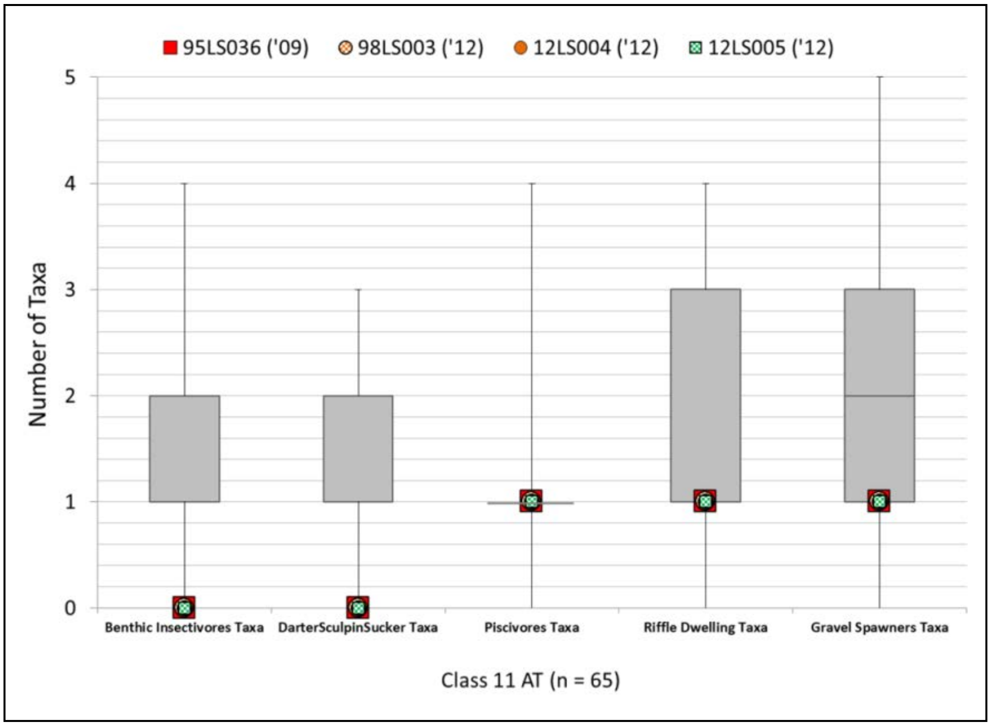
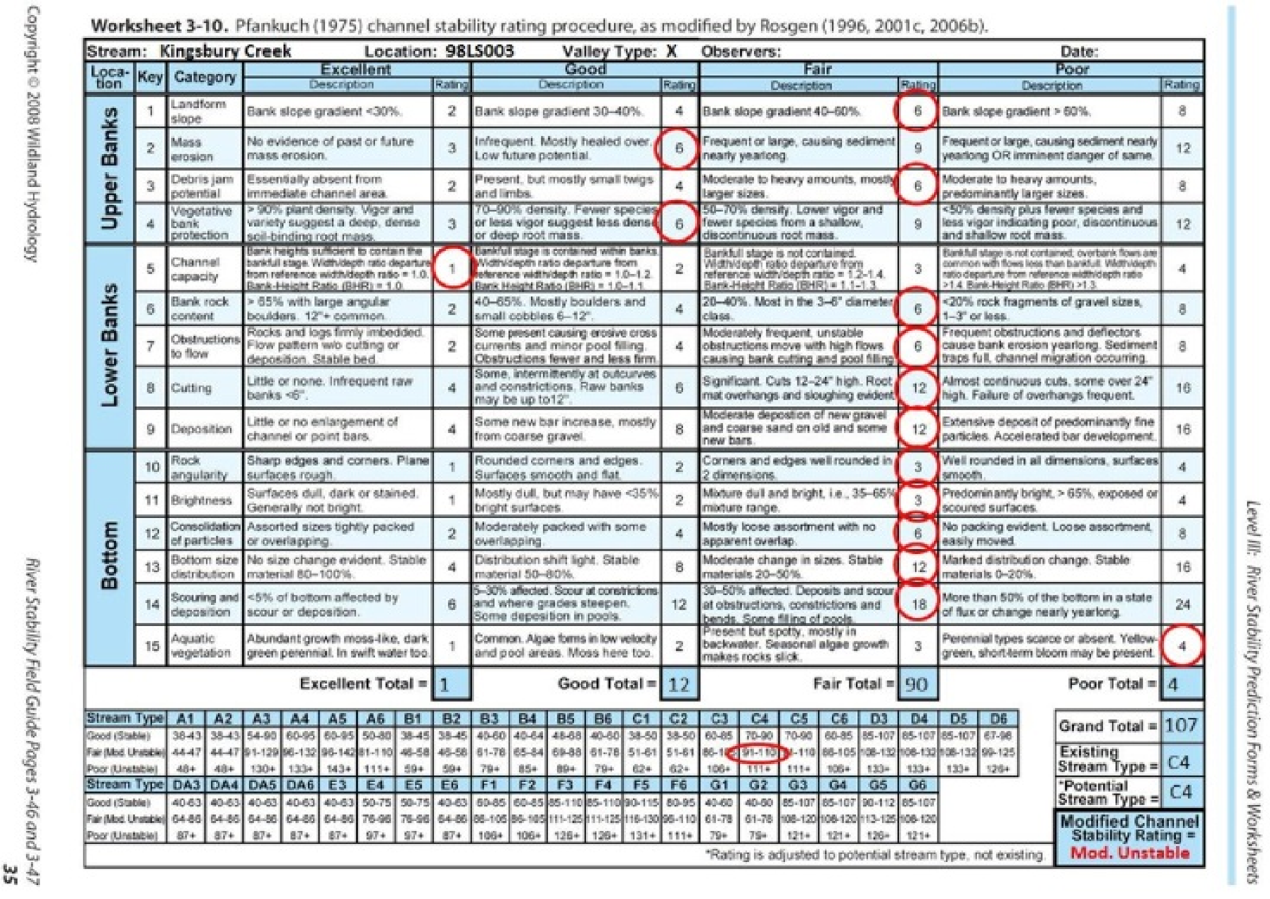


**Figure 63**: Number of fish taxa for various categories in Kingsbury Creek compared to SLRW stations above the impairment threshold. (* see Table 22 for acronym descriptions)



**Worksheet 3-10.** Pfankuch (1975) channel stability rating procedure, as modified by Rosgen (1996, 2001c, 2006b).

Stream: Kingsbury Creek Location: 98LS003 Valley Type: X Observers: Date:

| Location | Key | Category | Excellent Description | Rating | Good Description | Rating | Fair Description | Rating | Poor Description | Rating |
|---|---|---|---|---|---|---|---|---|---|---|
| Upper Banks | 1 | Landform slope | Bank slope gradient <30%. | 2 | Bank slope gradient 30–40%. | 4 | Bank slope gradient 40–60%. | 6 | Bank slope gradient > 60%. | 8 |
| | 2 | Mass erosion | No evidence of past or future mass erosion. | 3 | Infrequent. Mostly healed over. Low future potential. | 6 | Frequent or large, causing sediment nearly yearlong. | 9 | Frequent or large, causing sediment nearly yearlong OR imminent danger of same. | 12 |
| | 3 | Debris jam potential | Essentially absent from immediate channel area. | 2 | Present, but mostly small twigs and limbs. | 4 | Moderate to heavy amounts, mostly larger sizes. | 6 | Moderate to heavy amounts, predominantly larger sizes. | 8 |
| | 4 | Vegetative bank protection | > 90% plant density. Vigor and variety suggest a deep, dense soil-binding root mass. | 3 | 70–90% density. Fewer species or less vigor suggest less dense or deep root mass. | 6 | 50–70% density. Lower vigor and fewer species from a shallow, discontinuous root mass. | 9 | <50% density plus fewer species and less vigor indicating poor, discontinuous and shallow root mass. | 12 |
| Lower Banks | 5 | Channel capacity | Bank heights sufficient to contain the bankfull stage. Width/depth ratio departure from reference width/depth ratio = 1.0. Bank-Height Ratio (BHR) = 1.0. | 1 | Bankfull stage is contained within banks. Width/depth ratio departure from reference width/depth ratio = 1.0–1.2. Bank-Height Ratio (BHR) = 1.0–1.1. | 2 | Bankfull stage is not contained. Width/depth ratio departure from reference width/depth ratio = 1.2–1.4. Bank-Height Ratio (BHR) = 1.1–1.3. | 3 | Bankfull stage is not contained; overbank flows are common with flows less than bankfull. Width/depth ratio departure from reference width/depth ratio >1.4. Bank-Height Ratio (BHR) >1.3. | 4 |
| | 6 | Bank rock content | > 65% with large angular boulders. 12"+ common. | 2 | 40–65%. Mostly boulders and small cobbles 6–12". | 4 | 20–40%. Most in the 3–6" diameter class. | 6 | <20% rock fragments of gravel sizes, 1–3" or less. | 8 |
| | 7 | Obstructions to flow | Rocks and logs firmly imbedded. Flow pattern w/o cutting or deposition. Stable bed. | 2 | Some present causing erosive cross currents and minor pool filling. Obstructions fewer and less firm. | 4 | Moderately frequent, unstable obstructions move with high flows causing bank cutting and pool filling. | 6 | Frequent obstructions and deflectors cause bank erosion yearlong. Sediment traps full, channel migration occurring. | 8 |
| | 8 | Cutting | Little or none. Infrequent raw banks <6". | 4 | Some, intermittently at outcurves and constrictions. Raw banks may be up to12". | 6 | Significant. Cuts 12–24" high. Root mat overhangs and sloughing evident. | 12 | Almost continuous cuts, some over 24" high. Failure of overhangs frequent. | 16 |
| | 9 | Deposition | Little or no enlargement of channel or point bars. | 4 | Some new bar increase, mostly from coarse gravel. | 8 | Moderate deposition of new gravel and coarse sand on old and some new bars. | 12 | Extensive deposit of predominantly fine particles. Accelerated bar development. | 16 |
| Bottom | 10 | Rock angularity | Sharp edges and corners. Plane surfaces rough. | 1 | Rounded corners and edges. Surfaces smooth and flat. | 2 | Corners and edges well rounded in 2 dimensions. | 3 | Well rounded in all dimensions, surfaces smooth. | 4 |
| | 11 | Brightness | Surfaces dull, dark or stained. Generally not bright. | 1 | Mostly dull, but may have <35% bright surfaces. | 2 | Mixture dull and bright, i.e., 35–65% mixture range. | 3 | Predominantly bright, > 65%, exposed or scoured surfaces. | 4 |
| | 12 | Consolidation of particles | Assorted sizes tightly packed or overlapping. | 2 | Moderately packed with some overlapping. | 4 | Mostly loose assortment with no apparent overlap. | 6 | No packing evident. Loose assortment, easily moved. | 8 |
| | 13 | Bottom size distribution | No size change evident. Stable material 80–100%. | 4 | Distribution shift light. Stable material 50–80%. | 8 | Moderate change in sizes. Stable materials 20–50%. | 12 | Marked distribution change. Stable materials 0–20%. | 16 |
| | 14 | Scouring and deposition | <5% of bottom affected by scour or deposition. | 6 | 5–30% affected. Scour at constrictions and where grades steepen. Some deposition in pools. | 12 | 30–50% affected. Deposits and scour at obstructions, constrictions and bends. Some filling of pools. | 18 | More than 50% of the bottom in a state of flux or change nearly yearlong. | 24 |
| | 15 | Aquatic vegetation | Abundant growth moss-like, dark green perennial. In swift water too. | 1 | Common. Algae forms in low velocity and pool areas. Moss here too. | 2 | Present but spotty, mostly in backwater. Seasonal algae growth makes rocks slick. | 3 | Perennial types scarce or absent. Yellow-green, short-term bloom may be present. | 4 |
| | | | Excellent Total = | 1 | Good Total = | 12 | Fair Total = | 90 | Poor Total = | 4 |

| Stream Type | A1 | A2 | A3 | A4 | A5 | A6 | B1 | B2 | B3 | B4 | B5 | B6 | C1 | C2 | C3 | C4 | C5 | C6 | D3 | D4 | D5 | D6 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Good (Stable) | 38-43 | 38-43 | 54-90 | 60-95 | 60-95 | 50-80 | 38-45 | 38-45 | 40-60 | 40-64 | 48-68 | 40-60 | 38-50 | 38-50 | 60-85 | 70-90 | 70-90 | 60-85 | 85-107 | 85-107 | 85-107 | 67-98 |
| Fair (Mod. Unstable) | 44-47 | 44-47 | 91-129 | 96-132 | 96-142 | 81-110 | 46-58 | 46-58 | 61-78 | 65-84 | 69-88 | 61-78 | 51-61 | 51-61 | 86-1 | 91-110 | 1-110 | 86-105 | 108-132 | 108-132 | 108-132 | 99-125 |
| Poor (Unstable) | 48+ | 48+ | 130+ | 133+ | 143+ | 111+ | 59+ | 59+ | 79+ | 85+ | 89+ | 79+ | 62+ | 62+ | 106+ | 111+ | 111+ | 106+ | 133+ | 133+ | 133+ | 126+ |

| Stream Type | DA3 | DA4 | DA5 | DA6 | E3 | E4 | E5 | E6 | F1 | F2 | F3 | F4 | F5 | F6 | G1 | G2 | G3 | G4 | G5 | G6 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Good (Stable) | 40-63 | 40-63 | 40-63 | 40-63 | 40-63 | 50-75 | 50-75 | 40-63 | 60-85 | 60-85 | 85-110 | 85-110 | 90-115 | 80-95 | 40-60 | 40-60 | 85-107 | 85-107 | 90-112 | 85-107 |
| Fair (Mod. Unstable) | 64-86 | 64-86 | 64-86 | 64-86 | 64-86 | 76-96 | 76-96 | 64-86 | 86-105 | 86-105 | 111-125 | 111-125 | 116-130 | 96-110 | 61-78 | 61-78 | 108-120 | 108-120 | 113-125 | 108-120 |
| Poor (Unstable) | 87+ | 87+ | 87+ | 87+ | 87+ | 97+ | 97+ | 87+ | 106+ | 106+ | 126+ | 126+ | 131+ | 111+ | 79+ | 79+ | 121+ | 121+ | 126+ | 121+ |

Grand Total = 107

Existing Stream Type = C4

*Potential Stream Type = C4

Modified Channel Stability Rating = **Mod. Unstable**

*Rating is adjusted to potential stream type, not existing.

River Stability Field Guide Pages 3-46 and 3-47

35

Level III: River Stability Prediction Forms & Worksheets

**Figure 64**: Pfankuch rating for Kingsbury Creek 98LS003 and 95LS036

**Summary: Are poor physical habitat conditions a stressor in Kingsbury Creek?**

Degraded habitat in Kingsbury Creek can partially be attributed to channelization, channel instability and sedimentation. In addition, embeddedness of gravel substrates by road sand flushed into the creek as urban runoff is also likely contributing to habitat degradation in this stream. Visual observations of large sand deposits in riparian snowbanks and at stormwater outfalls support that conclusion. The substrate and channel morphology sections of the MSHA score, as well as the "moderately unstable" Pfankuch Stability rating support the diagnosis of physical habitat as a stressor contributing to fish and macroinvertebrate impairments. The degraded habitat is contributing to a decrease in the biological integrity of the stream, as evidenced by the low number of non-tolerant fish/meter and non-stocked, habitat-sensitive taxa present. It is our conclusion that habitat degradation is a stressor to the fish community in Kingsbury Creek.

## 5.2.5 Metals Toxicity

While some metals are essential as nutrients, all metals can be toxic at some level, and some metals are toxic in minute amounts. Impairments result when metals are biologically available at toxic concentrations affecting the survival, reproduction, and behavior of aquatic organisms. Lead and copper are two metals that have been detected at levels above natural background concentrations in Kingsbury Creek, which is not surprising given the urban and industrial land-uses in this watershed. Based on available metals data, lead and copper toxicity were identified as candidate causes of fish and macroinvertebrate impairments in the impaired reach of Kingsbury Creek. For additional information on lead and copper toxicity, including a discussion of applicable water quality standards, refer to Section 3.1.11.

Water quality standards for metals are separated into several categories based on chronic and acute effects. Table 29 provides a general overview of the three categories used to evaluate potential impairments due to metal toxicity. Due to the limited number of sampling occurrences for metals in the Kingsbury Creek Watershed, some of the specific criteria discussed in Table 29 cannot be evaluated, which provides limitations in the analysis of metals as a stressor in this watershed. For example, available metals data are derived from one-time sampling events and do not address the duration component of exposure, which is a significant factor in determining whether or not chronic impairment thresholds were exceeded.

**Table 29:** Types of water quality standards associated with trace metals

| Standard | Exposure Criteria | Additional Considerations |
|---|---|---|
| **Chronic Standard (CS)** | Value not to be exceeded more than once over a three year period; ***The value is based on the average concentration over a four-day duration.*** | Pollutant concentrations can be quite variable over such periods, depending on factors such as the type and size of the waterbody, weather and flow conditions, and the source and nature of the pollutant. When concentrations are judged to be relatively stable over the 4-day period in question, single samples can be sufficient. When concentrations are more variable, multiple samples or time-weighted composite samples are generally necessary in order to calculate a sufficiently accurate average concentration (text taken verbatim from MPCA, ) |
| **Maximum Standard (MS)** | Maximum standard is the value not to be exceeded at any time; ***the value is based on the average concentration over a one day duration.*** | |
| **Final Acute Value (FAV)** | Used exclusively for effluents, and is the ***concentration in that effluent to never be exceeded.*** | |

### Lead Toxicity

Minnesota's water quality standards for lead toxicity are based on water hardness. As hardness decreases, the concentrations assigned to the CS, MS, and FAV standard also decrease. In other words, the bioavailability, and thus toxicity of lead to aquatic is negatively related to water hardness. Limited water hardness data are available for Kingsbury Creek, but values likely range from near 40-50 mg/L during high flows to near 230 mg/L during low flows. The WQ standard for lead in water with hardness of 40 mg/L is 1.0 mg/L (CS), 25.4 mg/L (MS), and 51.0 mg/L (FAV). In water of 240 mg/L hardness, the applicable standards increase to 9.7 mg/L (CS), 248.9 mg/L (MS), and 499.0 mg/L (FAV). Figure 66 shows the different WQ standards for lead based on water hardness, as well as available paired measurements of lead and hardness for SLRW streams, including Kingsbury Creek.

Total lead concentrations observed at two Kingsbury Creek monitoring stations ranged from 0.9 mg/L to 3.63 mg/L. The maximum concentration of 3.63 mg/L was observed during a high flow event following a significant rainfall (3 inches in 24 hours) that hit the Duluth metropolitan area and pushed Kingsbury Creek to bankfull stage (Figure 65). The WQ standard for lead is based on the bioavailable or dissolved lead concentration, which is calculated using a formula based on water hardness.

Only two of the four lead samples had corresponding water hardness data, and the dissolved lead portions of these results are displayed in Table 29. One of the four samples exceeded the CS for lead toxicity (dissolved lead conc. = 3.18 µg/L; hardness = 55 mg/L) (Table 29). This particular sample was collected at biological monitoring station 95LS036 on 5/24/2012 following the aforementioned 3-inch rain event. Although water hardness data are not available to calculate dissolved lead concentrations for the samples collected on April 18, 2012, the dissolved portion of the lead results from that date are expected to be below the CS. Water hardness would have needed to be in the range of 30 mg/L for these samples to exceed the CS, which is unlikely during a snowmelt event given the high chloride concentrations typically observed in this stream (see Figure 53).

**Figure 65:** Photo of high-flow sampling conditions that produced a dissolved lead result above the CS (left) and an aerial photo of Kingsbury Creek winding through the city of Proctor, Minnesota (right)

The elevated lead concentrations observed during the May 2012 rain event may have been due to the flushing of wetlands and stormwater ponds on and around the DM & IR rail yard site. On the other hand, there is no way to discount the potential contribution of other sources of metals upstream of this site (city of Proctor, gas stations, road runoff). Based on the limited data available, it appears that any violations of the water quality standard for lead are relatively minor (only CS was exceeded) and are driven by high flow events that mobilize contaminants from various sources in the watershed. The baseflow sampling completed during winter low flow resulted in dissolved lead concentration of 1.00 µg/L and a water hardness value of 230 mg/L. This sample was well below the chronic water quality standard for lead.

**Table 29:** Monitoring results for lead and parameters related to the lead toxicity standard, as well as the dissolved lead concentrations for each sample compared to water quality standards

| Station | Station Description | Sample Date | Flow Conditions | Lead (µg/L) | Hardness (mg/L) | Dissolved Lead (µg/L) | Violation of WQ Standard? |
|---|---|---|---|---|---|---|---|
| **S007-051** | KINGSBURY CK AT POINT DR | 5/24/2012 | High (Rain Event) | 3.63 | 55 | **3.18** | **Yes / CS*** |
| **S007-051** | KINGSBURY CK AT POINT DR | 4/18/2012 | High (Snowmelt) | 0.84 | n/a | n/a | No** |
| **S007-051** | KINGSBURY CK AT POINT DR | 2/5/2014 | Low (winter baseflow) | 1.5 | 230 | 1.00 | No |
| **S007-055** | KINGSBURY CK AT GRAND AVE | 4/18/2012 | High (Snowmelt) | 0.9 | n/a | 1 | No** |

Additional sampling should be conducted to determine if lead concentrations in Kingsbury Creek remain below the chronic toxicity standard during normal to low flow conditions. In addition, sampling should be conducted on a daily basis during spring and summer rain events to investigate whether the CS is exceeded for a period of at least four days, as required by the WQ standard of listing a stream as impaired for lead toxicity. At this point, there is insufficient data to diagnose or eliminate lead toxicity as a stressor in Kingsbury Creek. Additional sampling for this parameter will be carried out by the MPCA staff during the Duluth Urban Watershed Assessment and Restoration Project, which is set to begin in 2015.

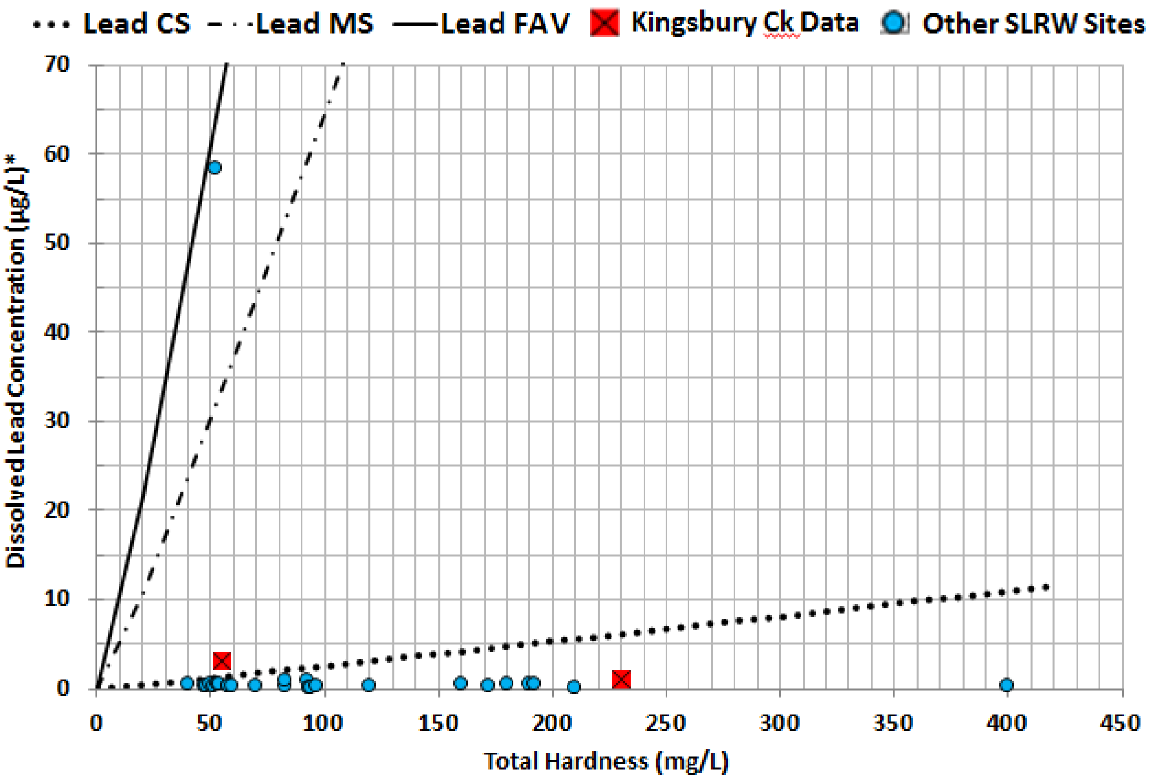


**Figure 66:** Dissolved Lead concentrations (mg/L) observed in Kingsbury Creek and other SLRW streams

### Copper Toxicity

For a summary of common sources of copper and its impacts on aquatic life, refer to Section 3.1.11. Copper toxicity to aquatic life varies with its bio-availability, which is mediated primarily by pH and hardness. As a result, Minnesota's current water quality standard for copper is based on dissolved copper concentration and water hardness. The bioavailability and its potential to cause harm to aquatic organisms decreases with increasing water hardness. Thus, copper toxicity is more commonly observed in streams and lakes with "soft" water hardness (than 60 mg/L). Measurements of copper that are considered for compliance with water quality standards must first be converted to dissolved copper, which is accomplished by applying a conversion factor of 0.960 to total copper values (Minn. R. 7050).

Limited water hardness data are available for Kingsbury Creek, but values likely range from near 40-50 mg/L (i.e. soft) during high flows to near 230 mg/L (i.e. very hard) during low flows (Table 30). The WQ standard for copper in water with hardness of 40 mg/L is 5.4 µg/L (CS), 7.2 µg/L (MS), and 14.4 µg/L (FAV). In water of 240 mg/L hardness, the applicable standards increase to 15.8 µg/L (CS), 37.3 µg/L (MS), and 74.6 µg/L (FAV). Figure 67 shows the different WQ standards for copper based on water hardness, as well as available paired measurements of copper and hardness for SLRW streams, including Kingsbury Creek.

Only two sampling results are available for copper in the Kingsbury Creek Watershed. One sample was collected from biological monitoring station 95LS036 during a high flow event after a 3-inch rainfall when the creek was at bankfull stage. Despite the high water volume, the dissolved copper concentration observed in the sample was elevated (5.45 µg/L). The corresponding water hardness of this sample was 55 mg/L, which sets the CS value for copper at 6.51 µg/L. The other monitoring result is from a baseflow sampling event (February 5, 2014) when the water in Kingsbury Creek was much harder

(230 mg/L). Dissolved copper concentration in the winter sample was 3.52 µg/L, which is well below the CS of 15.81 µg/L (Figure 67). Thus, although copper levels are elevated in Kingsbury Creek compared to many Duluth area streams, the data show no exceedances of the chronic toxicity standard.

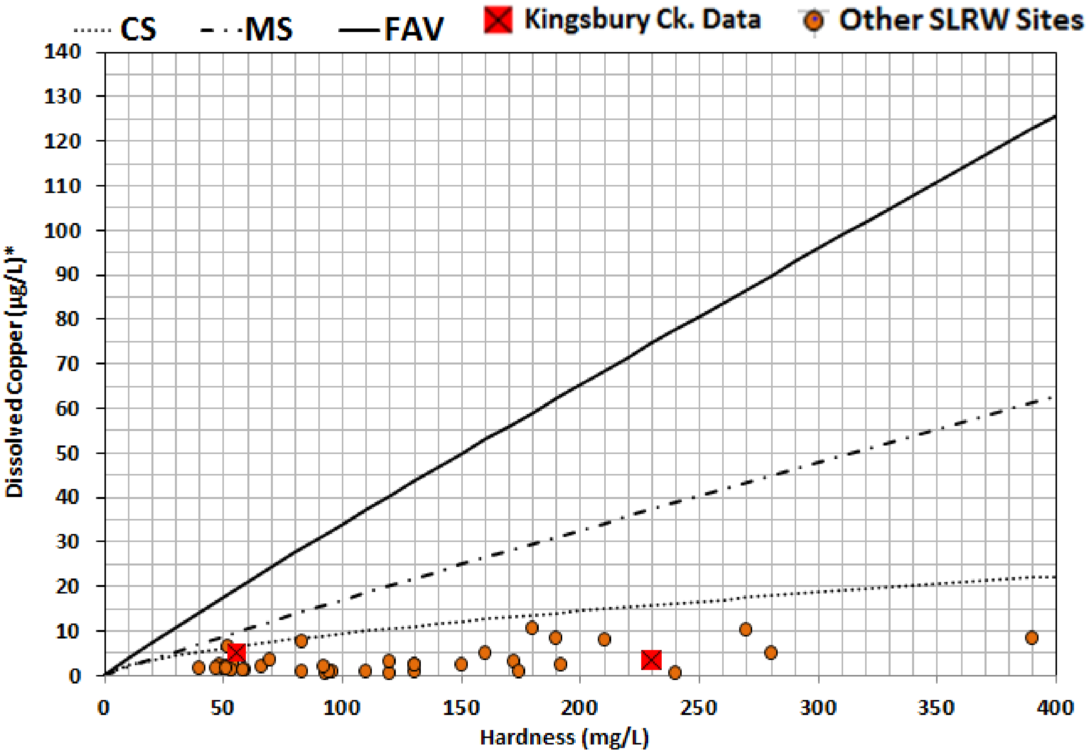


**Figure 67:** Dissolved copper concentrations (mg/L) observed in Kingsbury Creek and other SLRW.

There is not adequate data to diagnose or eliminate copper toxicity as a cause of biological impairment in Kingsbury Creek. Sampling results show no exceedances of water quality standards, but additional sampling over a wider range of flow conditions is required to increase confidence in diagnosing or eliminating this stressor. Additional sampling for copper will be conducted during the Duluth Urban WRAPS, which is slated to begin in 2015.

**Table 30:** Monitoring results for copper and parameters related to the copper standard, as well as the dissolved copper concentrations for each sample compared to water quality standards

| Station | Station Description | Sample Date | Copper (µg/L) | Hardness (mg/L) | Copper CS based on Hardness (µg/L) | Violation of WQ Standard? |
|---|---|---|---|---|---|---|
| **S007-051** | KINGSBURY CK AT POINT DR | 5/24/2012 | 5.23 | 55 | 6.51 | No |
| **S007-051** | KINGSBURY CK AT POINT DR | 2/5/2014 | 3.52 | 230 | 15.81 | No |

## Biological Response to Metals Toxicity and Other Contaminants

Biological response indicators to metals toxicity include individual physical and physiological effects (spinal abnormalities, gill damage, blackened tails) and whole community disturbances (kills of aquatic life, replacement of metals sensitive species with tolerant species). There have not been any reported fish kills in Kingsbury Creek, and the current monitoring approaches used by the MPCA and the DNR staff do not involve investigating physiological stress.

The fish and macroinvertebrate communities of Kingsbury Creek immediately downstream of Proctor are dominated by tolerant species. It is possible that more sensitive species, such as Brook Trout and certain mayfly taxa, are not present in Kingsbury Creek due to elevated metals concentrations. However, there are numerous stressors in this watershed that could be contributing to the lack of sensitive taxa in Kingsbury Creek, such as poor physical habitat, marginal DO concentrations, poor thermal regime for coldwater obligate species.

**Summary and Conclusions: Is Metals Toxicity a Stressor in Kingsbury Creek?**

The urban and industrial land-uses that dominate the upper half of the Kingsbury Creek Watershed are resulting in elevated concentrations (above natural background conditions) of lead, copper, and other contaminants. Dissolved lead has been observed in concentrations that exceed the chronic toxicity standard, although the data is not available to determine whether or not the duration component (four-days) of the CS is violated. The exceedance of the CS occurred following a major rain event, while low flow (baseflow) results were well below the water quality standard. This observation suggests that higher lead concentrations may be linked with precipitation and/or snowmelt events.

Additional monitoring data are needed in order to learn more about metals and other contaminants in Kingsbury Creek. The lack of data makes it difficult to diagnose or refute any of these parameters as stressors with a high level of confidence. Therefore, we suggest that these parameters remain potential causes of the impairment while initial implementation work is directed at restoring physical habitat conditions, lowering water temperatures, improving DO conditions, and addressing the impacts to hydrology caused by impervious surfaces in the watershed**.**

## 5.2.6 Kingsbury Creek: Summary of Stressors to Aquatic Life

**Table 31: Summary of SID results for Kingsbury Creek**

| Candidate Cause | Result |
|---|---|
| **Elevated Water Temperatures** | ● |
| **Low Dissolved Oxygen** | ● |
| **Total Suspended Solids (TSS) / Turbidity** | ● |
| **Chloride Toxicity / Sp. Conductivity** | ○ |
| **Poor Physical Habitat Conditions** | ● |
| **Copper & Lead Toxicity** | ○ |
| **Altered Hydrology** | ○ |
| Key: ● **= confirmed stressor** ○ **= Potential Stressor X = eliminated candidate cause** | |

## 5.3 Miller Creek

Similar to many streams in Duluth, Miller Creek originates as a wetland E channel, flowing through a wide lacustrine valley. However, due to development pressures the stream and valley have both been constricted in many places. The Miller Hill Mall area is a classic example of this. The original stream and valley type is impossible to tell and has been replaced essential by a stormwater conveyance ditch and does not resemble a natural channel until it flows past the Mall. From there, Miller Creek enters a transition zone of alternating C and B channels that continues for over a mile and a half. Near Lake Superior College the stream picks up gradient and begins its steep descent to the St. Louis River. Except for the last reach that is underground, Miller Creek is mostly an A channel with a bedrock controlled valley through this last section. Overall, 38% of Miller Creek is altered, with a smattering of sinuous E and C channels and higher-gradient B and A channels. Miller Creek plummets over 815 feet in 9 miles and has an average slope of 1.8%.

Data from two biological monitoring stations (98LS001 and 09LS003) were considered in the assessment of Miller Creek, which ultimately resulted in an impairment listing for low MIBI scores. Station information and MIBI results are presented in Table 32, and the locations of these monitoring sites are mapped in Figure 68. A previous biological assessment of Miller Creek, completed in 2002, resulted in an impairment listing for a “lack of coldwater fish.” Elevated water temperatures were cited as the main driver of the fish impairment (Source), and a TMDL is currently in development to address this issue. The macroinvertebrate impairment will be the main focus of this SID report.

A total of four macroinvertebrate samples were collected from Miller Creek (two visits to each station). The MIBI results were all below the impairment threshold. The poor MIBI results at these stations were the result of low scores in tolerance-based metrics and a lack of POET taxa (Plecoptera, Odonata, Ephemeroptera, Trichoptera). In all four samples, over half of the macroinvertebrate taxa present at these two sites can be considered “tolerant” of pollution and/or habitat disturbance. Abundance and diversity of EPT taxa were generally low at both Miller Creek stations, although several caddisfly taxa (*Lepidostoma*, *Cheumatopsyche* ) were fairly abundant at station 09LS003. Mayfly taxa were essentially absent from Miller Creek monitoring stations other than very small populations of *Centroptilum* and *Baetis*. The relative abundance of non-insect taxa (e.g. bivalves, aquatic worms, snails) at these stations also contributed significantly to the low overall MIBI scores.

**Table 32:** Summary of Miller Creek fish and macroinvertebrate monitoring stations and visit results

| Macroinvertebrate Assessments | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | Invert IBI Class | Invert IBI Result *(visit year)* | Invert IBI Result *(visit year)* | Standard | IBI Upper Confidence Limit | IBI Lower Confidence Limit |
| **98LS001** | 6.54 | 0.55 | 2 | 8 | **24.62** *(1998)* | **21.54** *(2012)* | 32 | 44.40 | 19.60 |
| **09LS003** | 7.97 | 0.56 | 2 | 8 | **17.12** *(2009)* | **17.92** *(2009)* | 32 | 44.40 | 19.60 |

Candidate causes of impairment that will be evaluated for MIBI impairment in Miller Creek include:

1. **TSS**
2. **Elevated Specific conductivity & Chloride Toxicity**
3. **Flow Alteration**
4. **Elevated Water Temperatures**

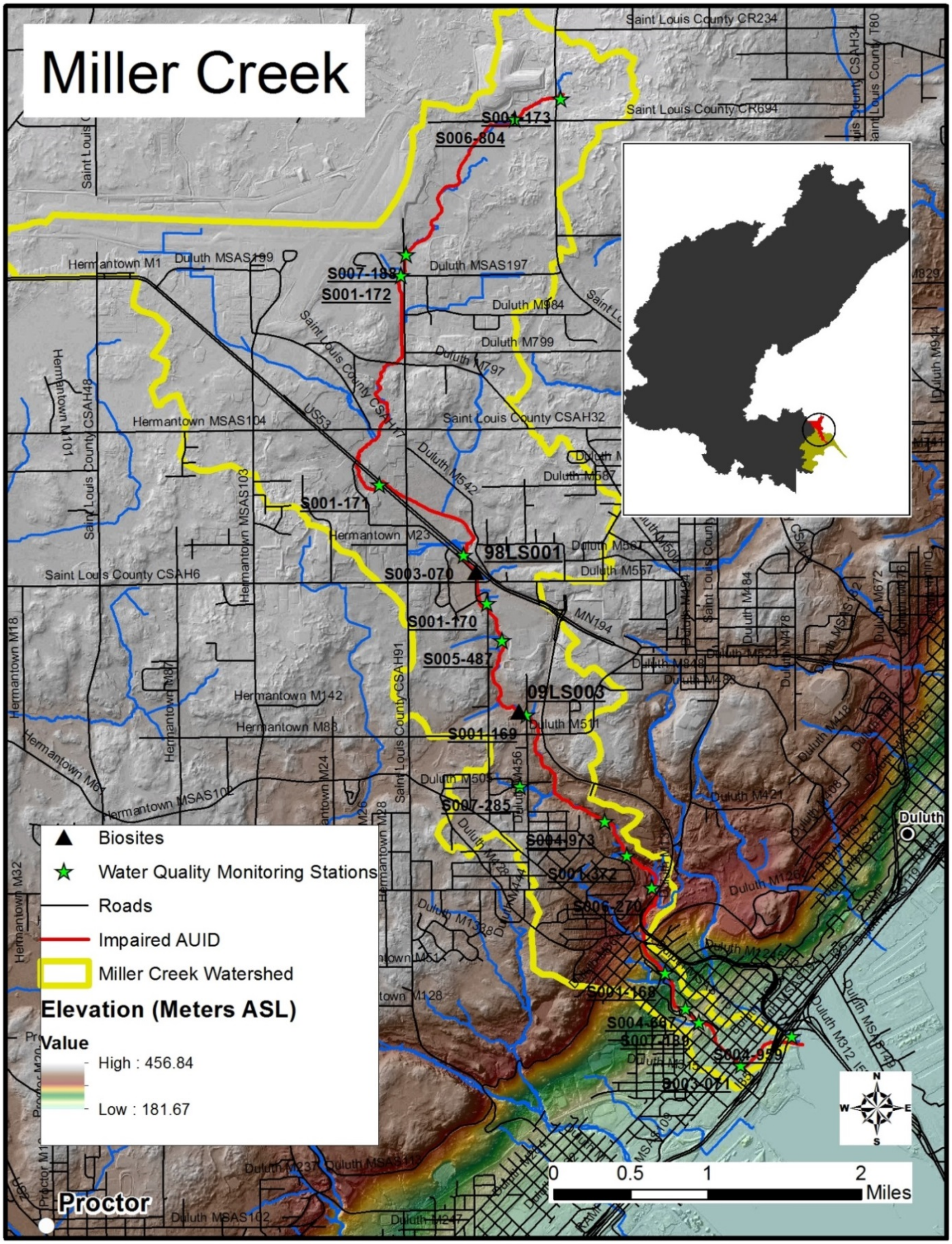


Figure 68: Map of Miller Creek Watershed and impaired stream reach

## 5.3.1 Total Suspended Solids & Turbidity

Monitoring data from 2007 – 2013 were used to develop longitudinal summaries of TSS and transparency tube (transparency) results for Miller Creek. The data summary provided in Table 33 presents these data by monitoring site, average result value, number of samples, and percent of samples exceeding the water quality standard. The TSS concentrations in the headwaters reaches of Miller Creek are generally low, and pose no threat to aquatic life. The two monitoring stations located furthest upstream (S003-070 and S005-487) met the proposed standard in at least 90% of the samples. However, every station downstream of S005-487 exceeded the 10 mg/L TSS standard in 10% or more of samples collected. The site at Chambersburg Road (S001-169) is co-located with biological monitoring station 09LS003 (one site that failed to meet the MIBI threshold), and failed to meet the TSS and Secchi standard in 5 out of 44 samples (11.4%). The average TSS at this station was 6.3 mg/L, which is lower than the proposed standard, but 10 mg/L was exceeded in 4 out of 25 samples (16%). Downstream from this station, TSS and transparency data show mixed results, yet all stations exceed the applicable water quality standards with regularity.

**Table 33:** Longitudinal TSS and Secchi Tube average values and percent standard exceedances for Miller Creek

| | Site | Site Description | TSS Average (mg/L) | TSS % Exceeding Standard | Secchi Tube Average | Secchi Tube % Exceeding Standard | Total # of Samples | Total % Exceeding Standard |
|---|---|---|---|---|---|---|---|---|
| Flow ↓ | S003-070 | UPPER GAGE SITE AT HWY 53 | 4.4 | 10.0% | 107.2 | 3.3% | 60 | 6.7% |
| | S005-487 | SEDIMENT TRAP NR DECKER RD | | | 104.3 | 3.7% | 54 | 3.7% |
| | S001-169 | CHAMBERSBURG RD | 6.3 | 16.0% | 104.2 | 5.3% | 44 | 11.4% |
| | S004-973 | LK SUPERIOR COLLEGE | 19.4 | 77.8% | 102.0 | 10.8% | 46 | 23.9% |
| | S001-372 | UPSREAM OF TRINITY ROAD | | | 56.7 | 10.3% | 29 | 10.3% |
| | S004-667 | E OF LINCOLN PK DR | | | 78.5 | 18.2% | 22 | 18.2% |
| | S003-071 | LOWER SITE AT 26TH AVE W IN DULUTH | 8.3 | 20.6% | 114.7 | 0.0% | 59 | 11.9% |
| | S007-285 | UNNAMED TRIB AT CHAMBERSBURG AVENUE | | | 87.9 | 10.5% | 38 | 10.5% |

Box plots of TSS and transparency values for select Miller Creek sites are compared to results from the A and B class SLRW reference streams (see Section 1.2.3 for reference stations) in Figure 69 . The TSS data for Miller Creek are generally comparable to the resutls from the SLRW reference streams, with the exception the reach adjacent to Lake Superior College (S004-973). The data set for this particular site includes very few measurements (n=9), and several samples collected during spring snowmelt conditions in May of 2010 are highly biasing the overall results. This reach currently supports a wild, self-sustaining Brook Trout population, although the population size is not particularly large and can be considered vulnerable. Although the data suggests that this reach experiences elevated TSS concentrations compared to other areas of Miller Creek, it is likely due to a lack of data from this location.

### Seasonal variation in total suspended solids

The TSS data from station S001-169 (biological monitoring station 09LS003) was plotted by month in order to investigate any possible seasonal trends in TSS (Figure 70). The data points are relatively scattered, although there is a possible trend of higher TSS values in spring and lower values in summer, which is similar to the trend seen in most streams in the Duluth area and along the North Shore of Lake Superior. This seasonal variation may not be as strong in Miller Creek for a variety of reasons. It is possible that the relative channel stability in Miller Creek does not cause higher flows to be as turbid as other streams. Another possibility may be that the main pathway of sediment – urban runoff and road sand – produces mostly *bedload* and is not reflected in the suspended load. Bedload refers to sediment particles transported by a stream by tumbling, sliding, or rolling along the streambed.

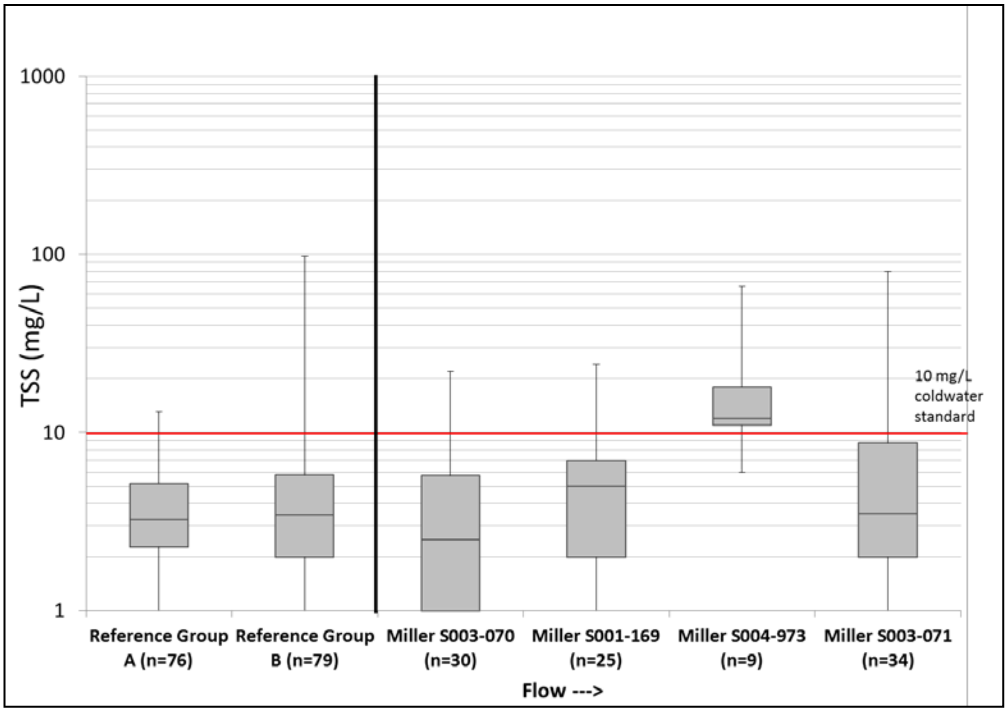


**Figure 69: Box plots of TSS values for Miller Creek sites and reference streams**

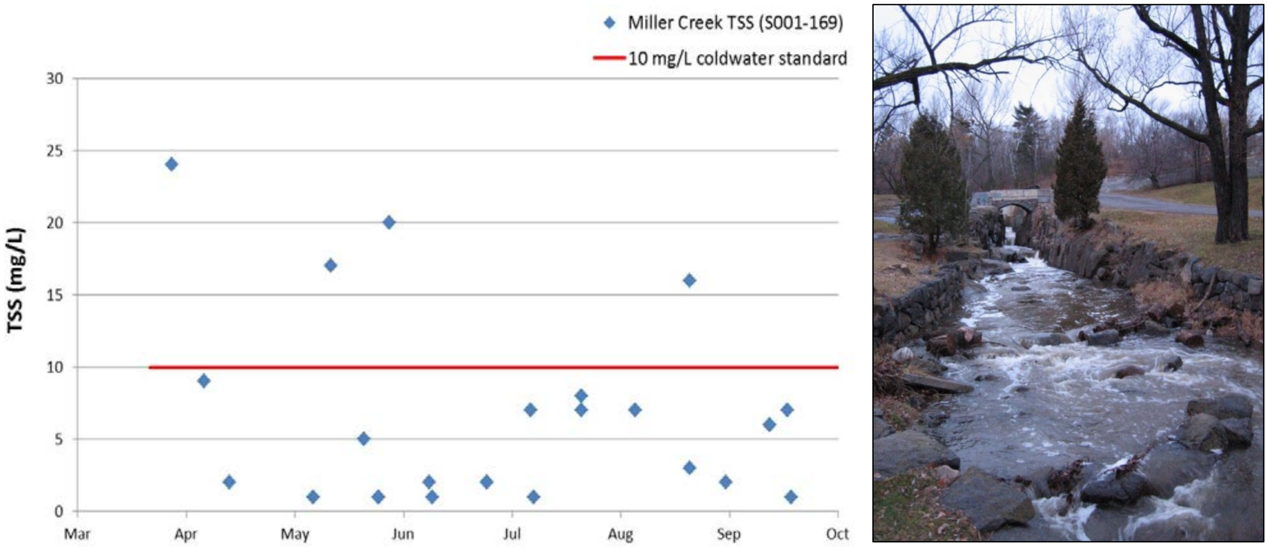


**Figure 70:** TSS data from Miller Creek near 09LS003, plotted by month (left). Slightly elevated TSS after a fall rain event (right)

### Sources and pathways of sediment in the Miller Creek watershed

### Ditching/Channelization

Channel straightening and meander bend removal result in a shortening of the channel, which causes the water slope to increase and the velocity of channel-forming discharge to increase. If grade control (culverts, bedrock, etc.) is absent, channel incision will often follow, delivering sediment to the stream from the bed and banks as channel evolution progresses. 38% of Miller Creek (AUID 04010201-512) has been channelized or straightened (Figure 71). Most of the channelization took place in the upper reaches of the watershed. Based on the current slope and elevation data, Miller Creek used to be a low gradient (<1% slope) E channel in a wide lacustrine or alluvial floodplain. This wide, flat floodplain has been severely constricted by development in the Hermantown and Miller Hill areas, and the channel was straightened and moved to make room for roads, retail stores and urban development (see inset in Figure 71 showing Miller Creek channelized around Highway 53 and shopping outlets in the Miller Hill Mall area).

### Channel Instability/Bank Erosion

Areas of channel instability and bank erosion are minimal in the Miller Creek Watershed. A few unstable gullies and some localized bank sloughing were observed during a reconnaissance hike on the stream (Figure 72); therefore, it cannot be written off as a possible pathway. A much more significant pathway for suspended solids in the Miller Creek Watershed is urban stormwater runoff.

**Figure 71:** Map showing the channelized and straightened reaches of Miller Creek

**Figure 72:** Bank and gully erosion in the Miller Creek Watershed

### Urban Runoff

A significant portion of the land in the Miller Creek Watershed is under urban and residential land-uses (Figure 73). In urbanized watersheds, the combination of buildings, roads, parking lots and other impervious surfaces cause precipitation to flow over the land instead of infiltrating into the ground. This quickly flowing runoff can transport sediment and other particulates into streams. Road sand, litter, and other detritus piled into snowbanks over the course of the winter season are flushed into stormwater pipes and streams in a relatively short period of time during snowmelt. Soils exposed by construction and development are susceptible to erosion during rain events.

The impact of urbanization in the Miller Creek Watershed has been the focus of ongoing work by several agencies and partners. In response to the stream being listed as impaired for elevated water temperatures and poor biological integrity, numerous best management practices (BMPs) have been installed to reduce the impact of urban areas on the creek and the aquatic life it supports. Pervious pavements have been used in mall and restaurant parking lots, and numerous rain gardens have been installed to curb runoff during snowmelt and rain events. An example of pervious pavement in the Miller Creek Watershed is shown in Figure 73.

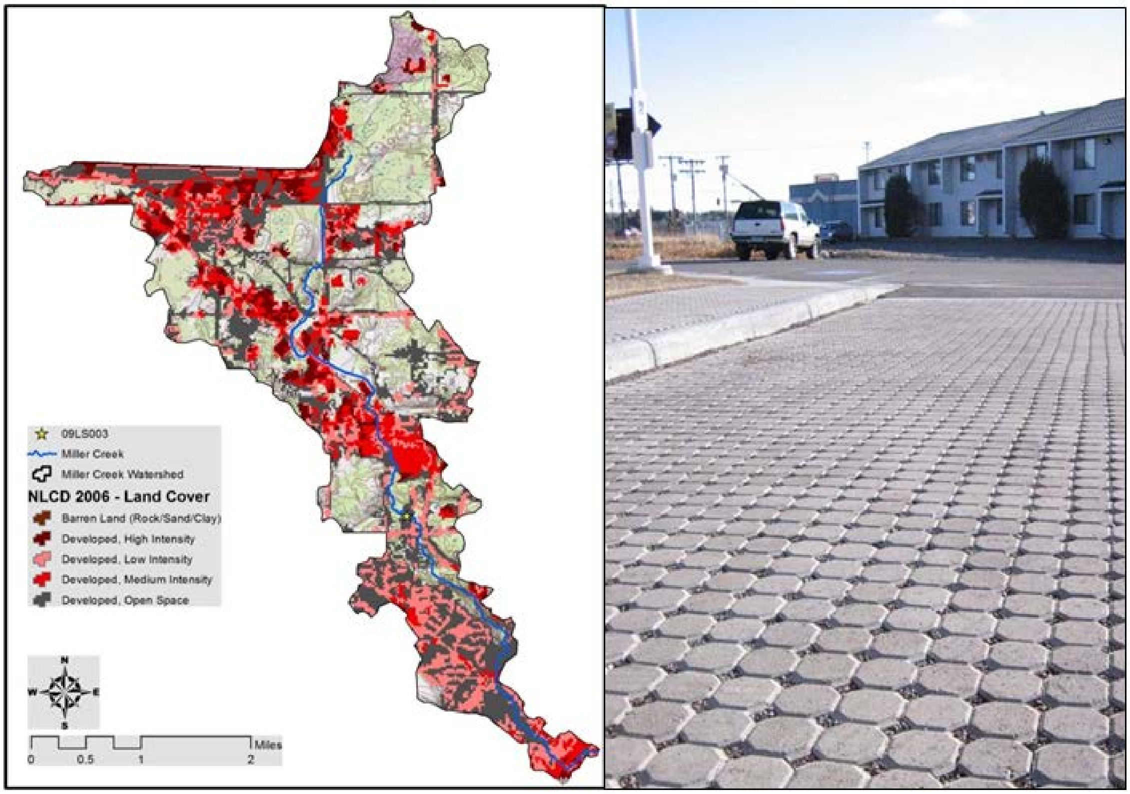


**Figure 73:** Developed and barren land in the Miller Creek Watershed (National Land Cover Database 2006)

### Biological effects of elevated TSS

### Macroinvertebrate Response to TSS

The TIVs for TSS were used to develop four classes (highly tolerant, moderately tolerant, moderately intolerant, and highly intolerant) for evaluating macroinvertebrate response in Miller Creek. Figure 74 shows that the invertebrate taxa found in Miller Creek are fairly evenly distributed among taxa that are intolerant and tolerant of TSS. Highly tolerant macroinvertebrate taxa were only observed at station 98LS001, which is located in the middle of the commercial district around Miller Hill Mall. The TSS results from water quality station S003-070, which is co-located with 98LS001, show relatively low concentrations compared to stations located further downstream. The presence of TSS-tolerant taxa at this station does not appear to be related to water quality conditions, considering that elevated TSS concentrations are generally not observed at this station. Other confounding stressors (chloride, habitat) may be playing a role in the community differences between 98LS001 and 09LS003, which is located downstream of the Miller Hill Mall area.

TSS index scores, which are a composite value of taxa tolerance and relative abundance measures, are clearly showing that the invertebrate assemblage in Miller Creek is similar in TSS tolerance to high quality stations throughout the SLRW. The box plots in Figure 75 compare data for a series of TSS-related metrics between Miller Creek monitoring stations and 1) Class 8 stations that scored above the upper confidence limit (AUCL) of the MIBI threshold, 2) Class 8 stations that scored above the impairment threshold (AT), and all SLRW AUCL stations. The three sites included in this analysis are clustered around the median values for the Class 8 AUCL stations, as well as the Class 8 AT stations. The Miller Creek invertebrate TSS index scored better than the median for all SLRW AUCL stations. These data reflect an invertebrate assemblage that is relatively intolerant of TSS, and do not support adding TSS as a stressor to macroinvertebrate communities in Miller Creek.

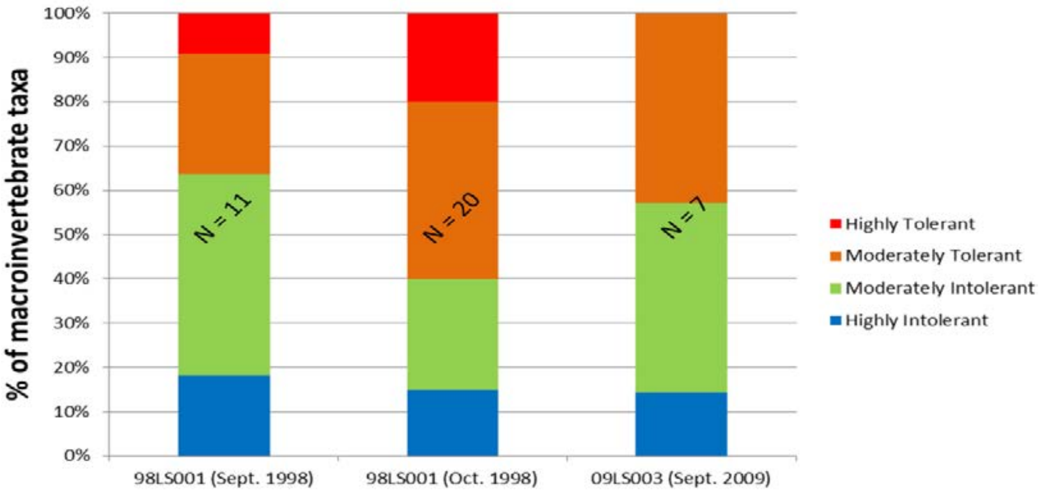


**Figure 74:** TSS tolerance of macroinvertebrate assemblages by site in Miller Creek (n = number of taxa)

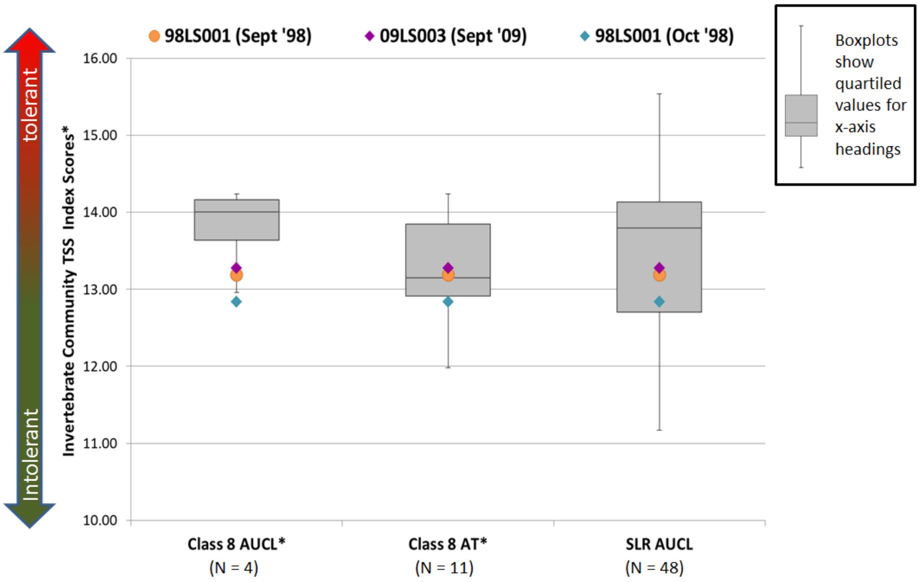


Figure 75: **Macroinvertebrate community TSS TIVs for Miller Creek, compared to unimpaired streams (*** See Section 4 for explanation of TIVs **SLR=**St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above FIBI Threshold

**Summary: Is TSS a stressor to aquatic life in Miller Creek?**

Water quality and biological monitoring results do not provide a strong case for listing TSS as a cause for impairment in Miller Creek. Urban runoff, gully erosion, and bank erosion (to a lesser extent) are resulting in elevated sediment loading to the creek during spring snowmelt and larger precipitation events, but adequate water clarity returns over a short period of time, and exposures to elevated TSS are short-lived. The biological metrics evaluated for symptoms of stress related to elevated TSS were inconclusive.

A more pressing issue related to sediment in the Miller Creek Watershed is the physical habitat degradation caused by the sediment deposition. Significant amounts of the road sand applied during the winter months end up in the creek, smothering important habitat for macroinvertebrates and bottom dwelling fish (e.g. sculpins), while also reducing available spawning habitat for Brook Trout and other gravel spawning fish species.

### 5.3.2 Chloride Toxicity and Specific conductivity

Elevated chloride concentrations and high specific conductivity are candidate stressors that have received a significant amount of attention in the Miller Creek Watershed due to the level of urban and commercial development. A study completed by Duluth MPCA (Anderson et al. 2000) documented some of the effects of urbanization and road salt application on several Duluth area streams, including Miller Creek. This study, along with more contemporary data that is available for these parameters, will be evaluated in this section as potential causes for MIBI impairment. Linkages between chloride concentrations and specific conductivity will be discussed in this section. However, these parameters will be evaluated individually as potential stressors considering that specific conductivity may be affected by other constituents as well.

**Chloride**

{*For background information on this stressor and a list of applicable water quality standards, see Section 3.1.7*}

Miller Creek was listed as impaired during the 2010 assessment cycle for failing to meet the state water quality standard for chloride. Chloride data available from 1988 through 2010 show frequent exceedances of the 230 mg/L chronic standard, and one result that exceeded the acute standard of 860 mg/L (1400 mg/L; February 18, 1999) (Figure 76). Monitoring results indicate that chloride concentrations in Miller Creek peak during the late winter months, and increase again during the late summer and early fall. High streamflow during the spring snowmelt periods of March and April dilute chloride concentrations and stream specific conductivity. Critical periods for chloride impacts to aquatic life appear to be winter months, when both acute and chronic standards are violated, as well as late summer low flow periods when the CS is surpassed. A lack of data exists for the months of November through January. Chloride concentrations during these months are likely elevated due to road salt application and low streamflow.

Chloride data have been collected at nine monitoring stations on Miller Creek, but only three stations have a large data set (> 10 observations). Figure 77 shows a box-plot distribution of chloride results from the four monitoring stations with the most samples collected. These stations are all located in the lower half of the watershed, and several of them bracket areas of high density commercial and/or residential development. Median chloride concentrations at the four monitoring stations range from 46 to 136 mg/L. Maximum values at all stations were above the CS (230 mg/L), and the maximum at station located near the mouth of Miller Creek (S003-071) exceeded the acute standard on one occasion. Given the limited number of monitoring sites with large data sets, it is difficult to pinpoint locations within the watershed where chloride concentrations dramatically increase. An increase appears to occur between stations S003-070 and S001-169, and again near the mouth of Miller Creek. These increases occur downstream of areas that are highly developed. Synoptic chloride monitoring is recommended at existing and additional sites to identify chloride inputs throughout the watershed.

Chloride concentrations can also be derived from continuous specific conductivity data through regression equations using paired conductivity / chloride sampling results (see Figure 79) and the curve developed by University of Minnesota's NRRI in Duluth (duluthstreams.org, Figure 17). Several years of continuous specific conductivity data are available for Miller Creek from station S001-169 (Chambersburg Road) and these results were used to select some conductivity spikes to convert to chloride concentrations. Table 34 summarizes some of the specific conductivity spikes observed during continuous monitoring periods and calculated chloride concentrations. These data suggest that chloride levels are often higher than the results obtained during grab sampling events.

Additional monitoring data from winter months is advised for better understanding the threat of chloride toxicity in the Miller Creek Watershed. There are no data available for the months of November through January. Based on trends from other urban areas, chloride concentrations during these months are often well above state and national water quality targets for protecting aquatic life.

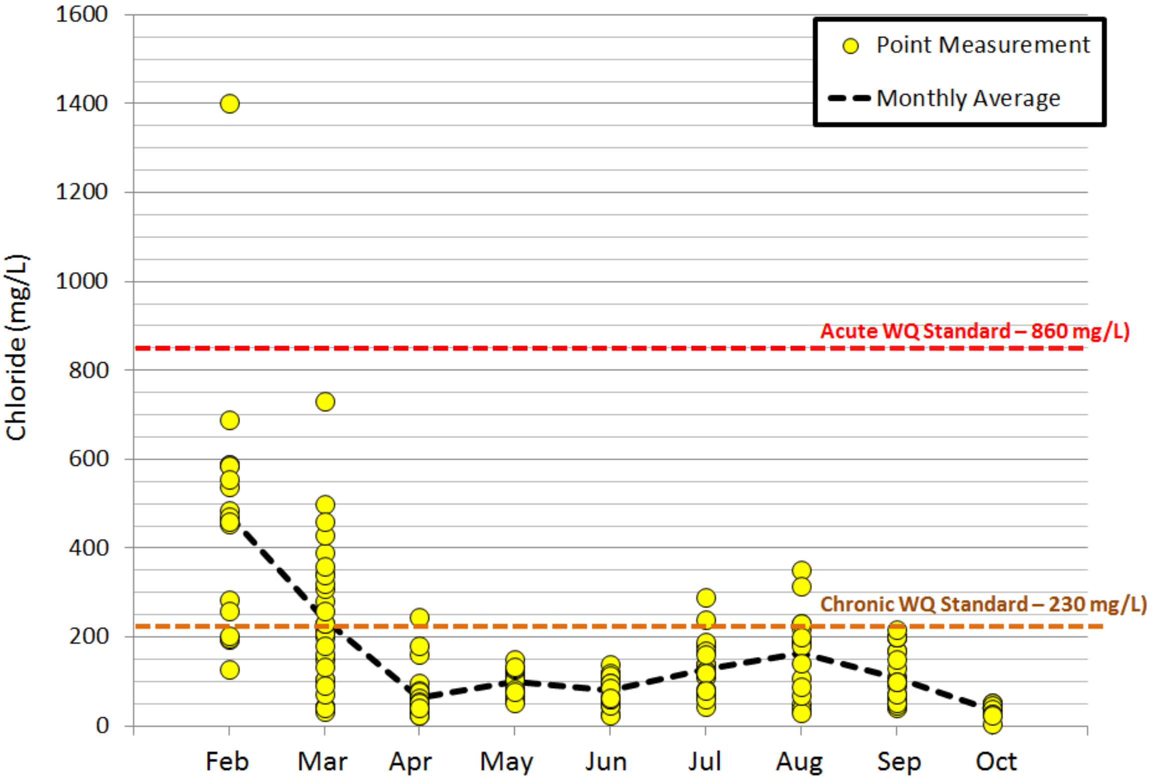


**Figure 76:** Available chloride data for Miller Creek obtained by grab sampling, arranged by calendar month.

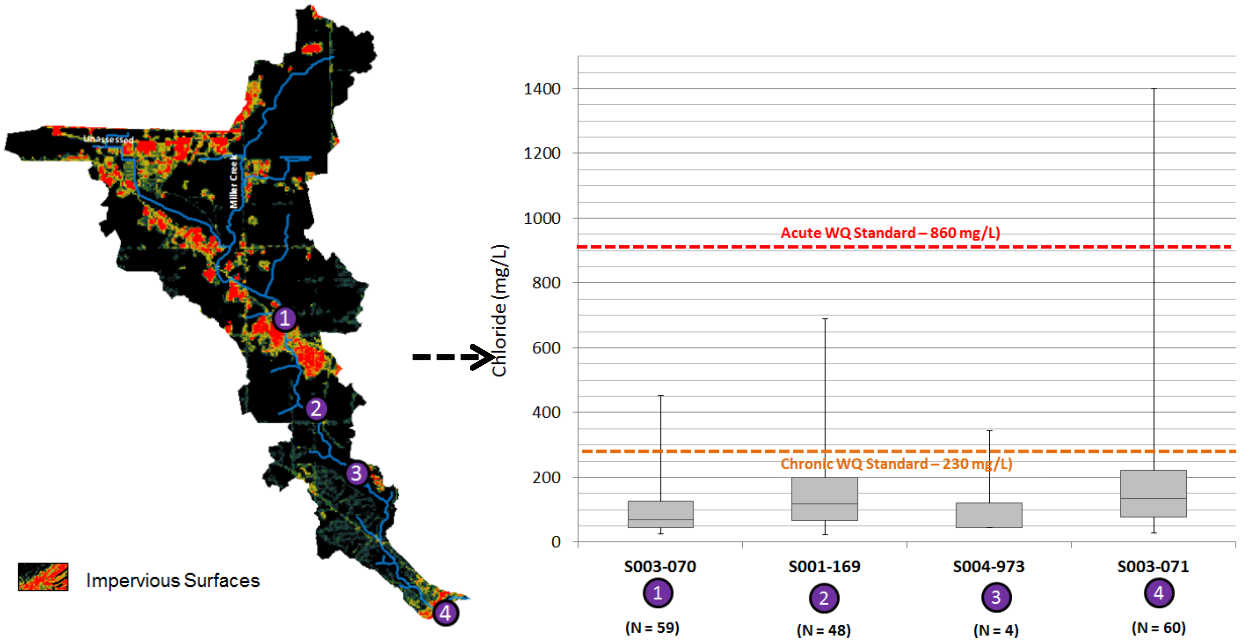


**Figure 77:** Box plots of all chloride sampling results for Miller Creek arranged by monitoring station

**Table 34:** Estimated chloride concentrations based on chloride - sp. conductivity regression equations developed by lab experiments (Axler, source) and paired chloride – sp. conductivity results from Kingsbury Creek.

| Date | Sp. Conductivity (µS/cm) | Chloride Equivalent in mg/L (NRRI Curve) | Chloride Equivalent in mg/L (Miller Creek Data Curve) |
|---|---|---|---|
| 3/6/2009 | 4111 | **1291.8** | **995.7** |
| 2/10/2009 | 3694 | **1151.3** | **892.3** |
| 3/14/2009 | 2793 | 847.6 | 669.1 |
| 2/26/2009 | 2409 | 718.2 | 573.9 |
| 11/26/1997 | 1835 | 524.8 | 431.7 |
| 1/24/1998 | 2540 | 762.4 | 606.4 |
| 2/16/1998 | 3305 | **1020.2** | 795.9 |
| Red = value above Chronic Standard **Bold Red = value above Acute Standard** | | | |

### Sources and Pathways

Road Salt Runoff

The watershed of Miller Creek has been dramatically altered from its original state due to residential and commercial land-uses. In the year 2000, approximately 20% of the Miller Creek Watershed was covered with impervious surfaces (roads, parking lots, sidewalks, rooftops, etc.). Research on the influence of urban land use on aquatic life in streams has identified a level of 7-12% imperviousness where decreases in biotic integrity were observed (Wang et al., 1997, 2003). Due to the cold climate and abundant annual snowfall within the city of Duluth, many of the impervious surfaces within the Miller Creek Watershed are treated throughout the winter months with road deicing chemicals (primarily sodium chloride or NaCl).

Chloride is likely entering Miller Creek through surface water and groundwater pathways. There are no point sources continuously discharging (e.g. WWTP) to Miller Creek that would provide regular chloride inputs. Surface water inputs of chloride to Miller Creek were described in the Duluth Metropolitan Area Streams Snowmelt Runoff Study (Anderson et al. 2000):

> *Urban stormwater impacts to Duluth Metropolitan Area streams can be severe in the snowmelt runoff period for several reasons: (1) Duluth has one of the highest annual snowfall amounts in the state, averaging 79.5 inches per year (in 2013-2014 Duluth received 130.2 inches), and many streams receive their highest annual flows during the snowmelt runoff period. (2) The area's high gradient, thin soils and surficial and bedrock geology reduce the potential for infiltration; (3) Because of the size of the stream's drainage areas [comparatively small], the ability to store, retain, or retard flow for long periods is restricted.*

Groundwater is another delivery pathway for chloride to enter streams and rivers. Contaminated runoff from impervious surfaces can flow from the pavement into unlined ditches or ponds and infiltrate surrounding soil. In addition, road salt applied during snowstorms is generally plowed off the roadway and paved shoulder (Figure 78). When the resulting snowbanks melt, the meltwater, together with the dissolved salt, can migrate through soil and infiltrate into the water table. A study of groundwater influence on stream chemistry in Massachusetts confirmed that chloride originating from highway-deicing application persisted throughout the year as a source of contamination in groundwater, interflow, and surface water even during warmer months (Granato et al. 1995).

**Figure 78**: Large snowbank from a shopping center parking lot adjacent to Miler Creek. These snowbanks can contain significant amounts of road deicing salts and grit (sand/sediment). *Photo Credit*: Tom Estabrooks, MPCA- Duluth.

## Chloride - Specific Conductivity Relationship in Miller Creek

In many instances, an excellent relationship can be established between total dissolved salts and specific conductance (Allan 1995). Due to differences in local geology, land-use, and other environmental factors, the proportion of the various dissolved ions can vary significantly by watershed. In Miller Creek, a very strong relationship can be drawn between chloride concentrations and specific conductance (Figure 79). A regression of 95 paired chloride and specific conductivity results shows a clear positive relationship, although several early season snowmelt samples fall off of the regression line due to dilution factor due to the high streamflow associated with those events. This relationship clearly indicates that chloride in the water column is a primary driver of specific conductivity levels in Miller Creek.

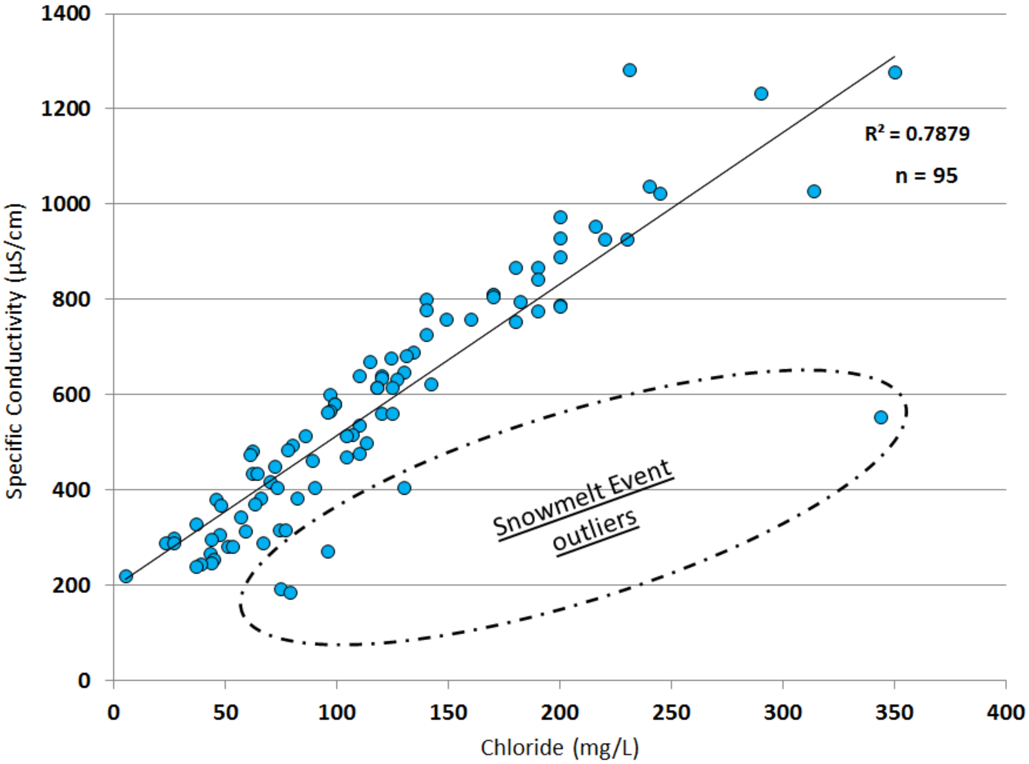


**Figure 79**: Regression of all existing paired chloride and specific conductivity measurements taken in Miller Creek (n=95)

### Specific Conductivity Data

Both instantaneous (point) and continuous measurements of specific conductivity are available for several Miller Creek monitoring locations. Point observations were collected primarily between early April and October, with a just a few observations available from outside of that seasonal window. All available point measurements of conductivity are shown in Figure 80 along with the monthly averages. The average monthly instantaneous data show conductivity peaking in July and August during low flow conditions. Maximum conductivity values during these months range between 1500 and 1900 µS/cm and the average is around 600 – 700 µS/cm. While these observations provide an accurate assessment of conditions in Miller Creek during the open water season, the continuous data provides data for the critical winter months when road salt is being applied to the streets of Duluth and the parking lots of the Miller Hill commercial district.

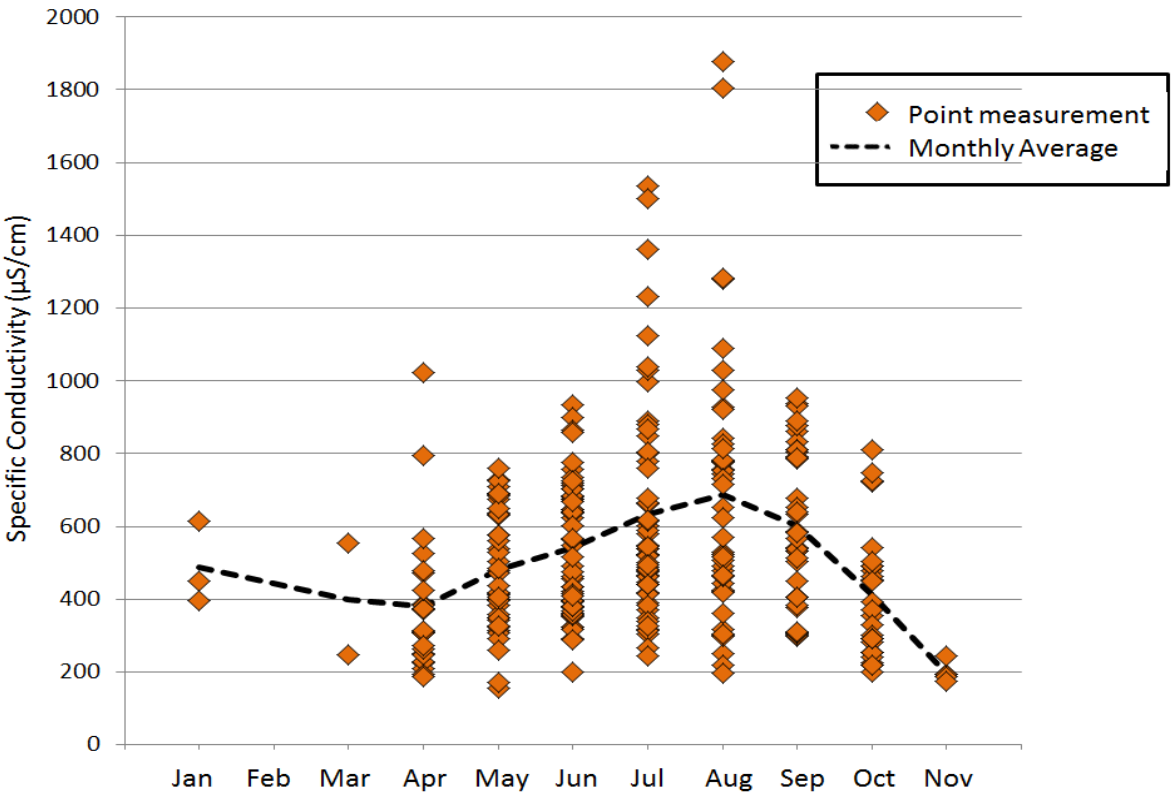


**Figure 80:** Point measurements of specific conductivity in Miller Creek arranged by calendar month

Figures 81 through 83 show continuous conductivity readings from Miller Creek at Chambersburg Road (station S001-169) for the years 1998, 2008, and 2009. Continuous data were collected at 15 or 30 minute intervals nearly year round. High magnitude, short duration pulses of extremely high specific conductivity levels were observed during the months of January through April in all three years of data shown in the Figures. On March 1, 2008, specific conductivity approached 8000 µS/cm, and then topped 5500 µS/cm a few weeks later on March 18, 2008. Similar spikes are observed in the other continuous data sets presented, but maximum conductivity values were slightly lower during the other monitoring years (in the range of 3000 to 4000 µS/cm). Based on the continuous data, it is clear that conductivity levels peak during the winter months when road salt is being actively applied to impervious surfaces and stream flow is fairly low. These graphs also show that the pulses of increased conductivity are extremely rapid. For example, during the March 1, 2008 snow event, specific conductivity rose from 1,747 µS/cm to 7,995 µS/cm in a span of about 16 hours, and then leveled off around 3,000 µS/cm approximately 16 hours later.

Although the major spikes in specific conductance appear to be short-term, the continuous monitoring data does show sustained periods where specific conductivity values were in excess of 1,000 µS/cm. In 2008 and 2009, conductivity at the Chambersburg Road monitoring site remained above 1,000 for several months. These longer term exposures may be a chronic stressor to fish and macroinvertebrate taxa that are sensitive to elevated conductivity levels. Additional analyses on the effects of elevated conductivity in Miller Creek are presented in the next section.

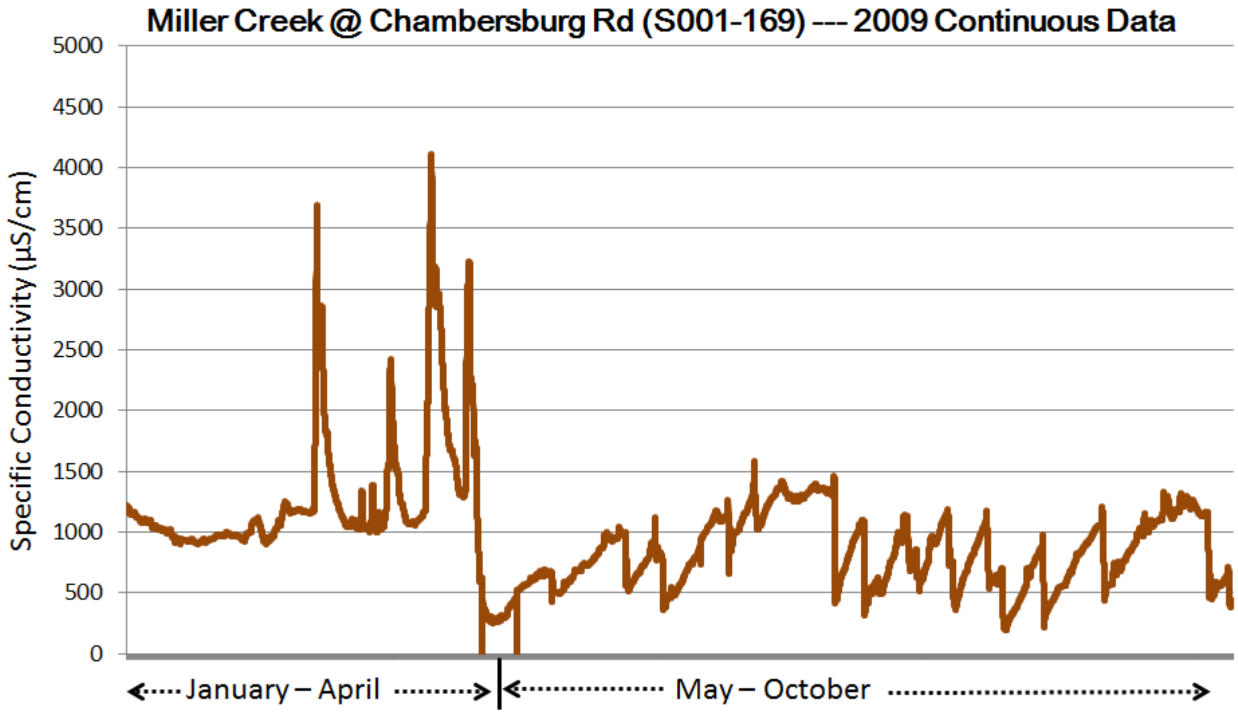


**Figure 81:** Continuous specific conductivity data collected at station S001-169 in 2009

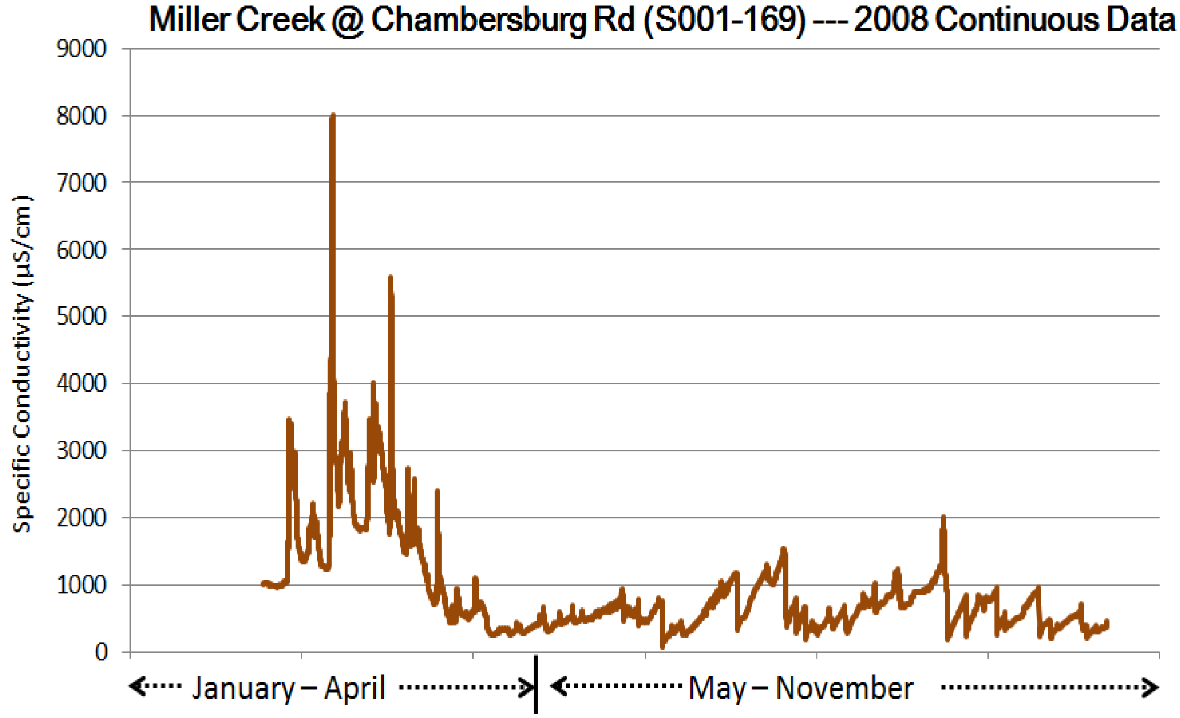


**Figure 82:** Continuous specific conductivity data collected at station S001-169 in 2008

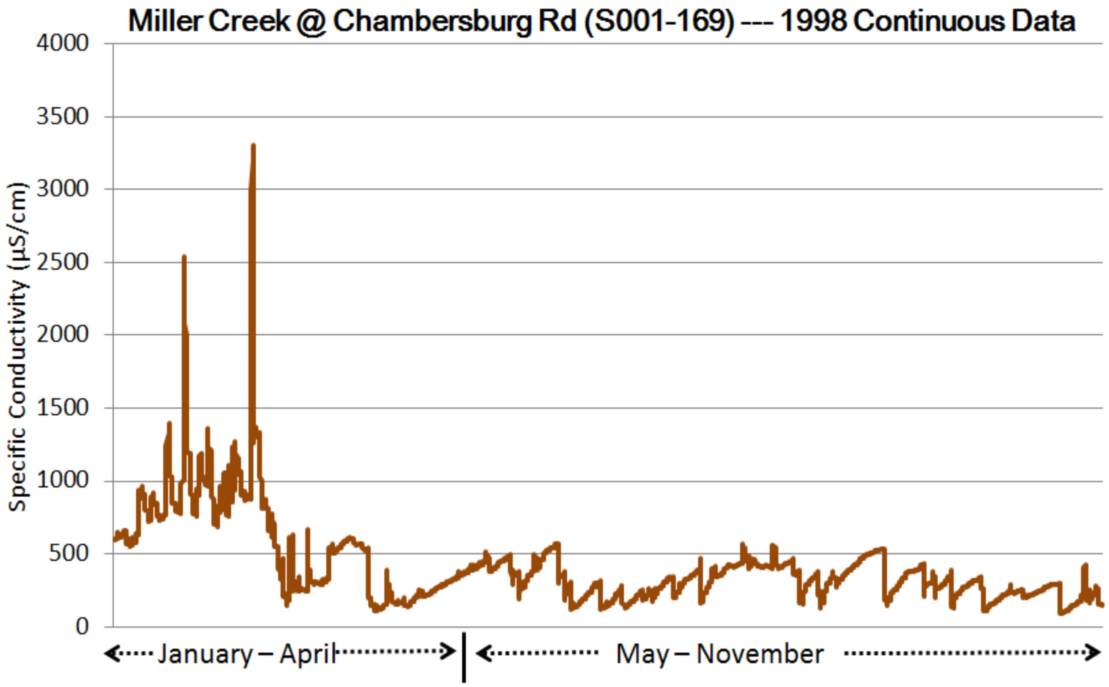


**Figure 83:** Continuous specific conductivity data collected at station S001-169 in 1998

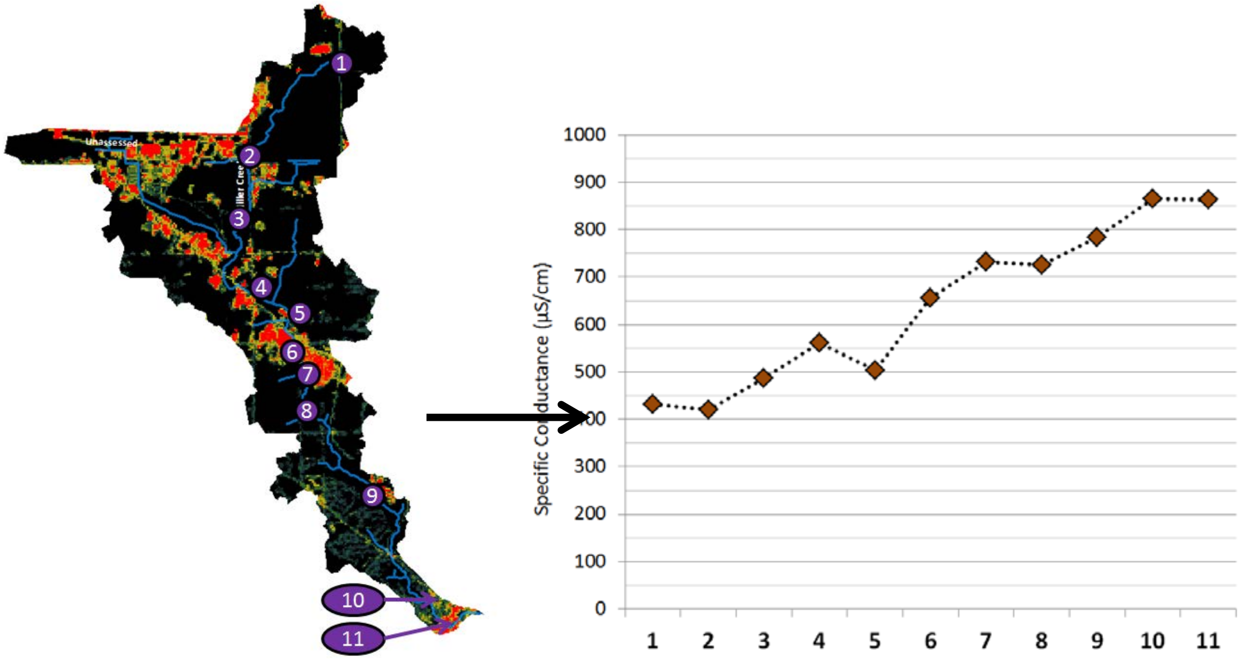


**Figure 84:** Results from longitudinal specific conductivity monitoring conducted September 12, 2014

## Effects of Elevated Chloride / Specific Conductivity on Aquatic Life

### Chloride

The MBI (2012) identified chloride-sensitive macroinvertebrate taxa based on a statewide water chemistry and biological data set. Specific chloride thresholds were not established through this study, however, common macroinvertebrate taxa to Minnesota were classified as sensitive (“S”), Tolerant (“T”), or neither tolerant nor sensitive. Despite its urbanized watershed and elevated chloride concentrations, the macroinvertebrate community of Miller Creek is not generally dominated by taxa that are considered tolerant of chloride concentrations. On average, chloride tolerant individuals accounted for 9.1% of the organisms sampled at station 98LS001, and 25.9% of the organisms at station 09LS003. The increase in chloride tolerant individuals from 98LS001 to 09LS003 may reflect an increase in chloride loading from the Miller Hill Mall and Highway 53 corridor that runs adjacent to the creek between these two stations.

The maximum percentage of chloride tolerant individuals observed in Miller Creek, 34.5%, occurred at station 09LS003 during a 2010 sampling visit. Several stations that are located on streams with far less urbanized land cover (Big Sucker Creek, Hay Creek, East Branch Amity Creek) had higher percentages of chloride tolerant macroinvertebrate during several monitoring visits, which shows that there is not a direction relationship between chloride concentrations and these tolerance values. Big Sucker Creek, which is a rural trout stream located on the outskirts of the city of Duluth, had an average of 35.3% chloride tolerant macroinvertebrate individuals over four separate sampling events at two monitoring stations. The maximum of 46% chloride tolerant organisms was exceeded only by two stations on Mission Creek (located in the city limits of Duluth), which both had over 50% chloride tolerant macroinvertebrates.

In general, the proportion of chloride tolerant macroinvertebrates observed in Miller Creek and other regional streams does not show a consistent relationship with urban land-use or chloride concentrations. Miller Creek station 98LS001, which is located in a highly developed commercial shopping center in the Duluth Metropolitan area, is comparable to Kimball Creek and Kadunce River in terms of overall percentage of chloride tolerant macroinvertebrates. Kimball and Kadunce River are high quality coldwater streams with relatively pristine watersheds and no water quality-based impacts from urban development or road crossings. The most common chloride tolerant organisms found in these two streams were the mayfly *Acentrella* (tiny blue-winged olive mayflies), midge *Polypedilum*, and the caddisfly *Ceratopsyche*.

The only indicator of chloride stress in Miller Creek based on these tolerance data is the lack of chloride sensitive macroinvertebrate taxa. Chloride sensitive macroinvertebrate taxa were absent from all four sampling visits to Miller Creek. It should be noted that several streams with less disturbed watersheds (Kadunce River, East Branch Amity Creek) also did not support any chloride sensitive taxa during at least one sampling event. The majority of the stations evaluated did support at least some chloride sensitive taxa, with the most common being *Eurylophella*, *Capniidae*, and *Ophiogomphus*.

Additional work is needed to better refine the biological indicator tools for chloride toxicity. Clearly, there are false responses taking place in streams like Sucker River, Kimball Creek, and Kadunce River, which show fairly high percentage of chloride-tolerant taxa despite having little to no development in their respective watersheds. Confounding stressors are one possible explanation for these mixed results, as it is impossible to remove the influence of other factors that can control the abundance and diversity of organisms at these stations. Biological indicators for specific conductivity are better understood, and will be used to further evaluate the potential for a chloride-specific conductivity stressor in the Miller Creek Watershed.

### Specific conductivity (Specific Conductivity)

The effects of elevated specific conductivity on aquatic life were evaluated using data from Minnesota streams and scientific literature. A summary of this analysis is presented in Section 3.1.5. Based on this work, several biological metrics were selected to evaluate specific conductivity as a stressor in Miller Creek (see Table 28).

### Macroinvertebrate Taxa Richness

Macroinvertebrate taxa richness in Miller Creek is relatively low compared to other streams in the SLRW, including streams of the same size and MIBI classification (Figure 85). The two Miller Creek stations are either at, or well below the 25th percentile taxa richness values observed at the selection of reference sites included. Each of the Miller Creek monitoring stations show considerable variability in taxa richness from year to year, and even some variability within the same monitoring year. At station 98LS001, there was considerable variability in taxa richness between the two sampling visits to that site in 1998 (35 taxa on 9/14/98; 45 taxa on October 1, 1998). Taxa richness at station 09LS003 dropped from 44 taxa in 2009 to 36 taxa in 2010. This variability is not unusual, but it may represent some instability in the macroinvertebrate community over time, which could be attributed to the rapidly changing conditions that can occur in urbanized watersheds. Overall, the relatively low taxa richness observed in Miller Creek is consistent with observations from other streams with elevated specific conductivity levels.

### Ephemeroptera (mayfly) Taxa Richness and Relative Abundance / EPT Taxa Richness

A reduction in Ephemeroptera (mayfly) taxa richness and/or relative abundance has been observed in streams and rivers with elevated specific conductivity (Pond 2004; Hassel et al. 2006). In Miller Creek, Ephemeroptera richness and relative abundance are both very low compared to many high quality streams within the SLRW (Figure 85). Only one mayfly taxon was observed at 98LS001, *Stenonema*, and was only present during one of two sampling events. *Stenonema* is considered neither sensitive nor tolerant to elevated conductivity. At station 09LS003, below Miller Hill Mall, two Ephemeroptera taxa were observed in 2009 sampling event (*Baetis and Centropilum*) and only one in 2010 (*Baetis)*. Median values for Ephemeroptera richness among the reference sites that were used for comparison ranged from 5.50 – 6.00. Miller Creek clearly supports fewer mayfly taxa than high quality streams in the watershed, and the elevated specific conductivity levels may be limiting more sensitive mayfly taxa from taking hold in this watershed.

Relative abundance of mayfly individuals was also depleted in Miller Creek. Individuals from the order Ephemeroptera accounted for less than 5% of the total macroinvertebrate community during each visit to Miller Creek biological monitoring stations. Ephemeroptera accounted for an average of 0.2% of the total population at 98LS001 over two monitoring visits, and an average of 2.9% at 09LS001 over two visits. Out of the 36 total visits to class 8 (coldwater) MIBI stations in the SLRW, three visits to Miller Creek resulted in the lowest percentage of Ephemeroptera observed among all of the sites sampled (0.00%, 0.3%, 1.29%).

Various Trichopterans (caddisflies) and Plecopterans (stoneflies) are also known to be sensitive to elevated conductivity. Together, along with the order Ephemeroptera, these macroinvertebrate orders are called “EPT” taxa and are often used as an indicator of disturbance when their abundance or overall richness declines. Data from streams around the state of Minnesota show a decline in EPT macroinvertebrate taxa when specific conductivity approaches or exceeds 1,000 µS/cm (see Figure 14). EPT taxa richness in Miller Creek is lower than what was observed at high quality stations of the same MIBI class (Class 8), and also lower than the majority of high quality sites throughout the SLRW (Figure 85).

### Specific Conductivity Tolerance Indicator Values (TIV)

Specific conductivity TIVs for macroinvertebrate taxa were used to determine the percentage of tolerant and intolerant individuals observed at Miller Creek monitoring sites. The distribution is fairly equal between highly tolerant, moderately tolerant, neutral, and moderately intolerant taxa. Taxa which are highly intolerant of elevated specific conductivity were present in relatively low numbers (1%-2% of total community); with the exception of the 2010 visit to station 09LS003, where slightly over 20% of the community were individuals from highly intolerant taxa. A robust population of *Lepidostoma* (Little Brown Sedge) caddisflies accounted for the large increase in intolerant taxa observed during that particular sampling visit. Several of the monitoring stations supported relatively high numbers of individuals from taxa without a TIV value for specific conductivity. Nearly all of the taxa without TIV observed in Miller Creek were members of the subclasses Oligochaeta (worms) and *Acari* (water mites). These taxa tend to be tolerant of many stressors, both physical and chemical.

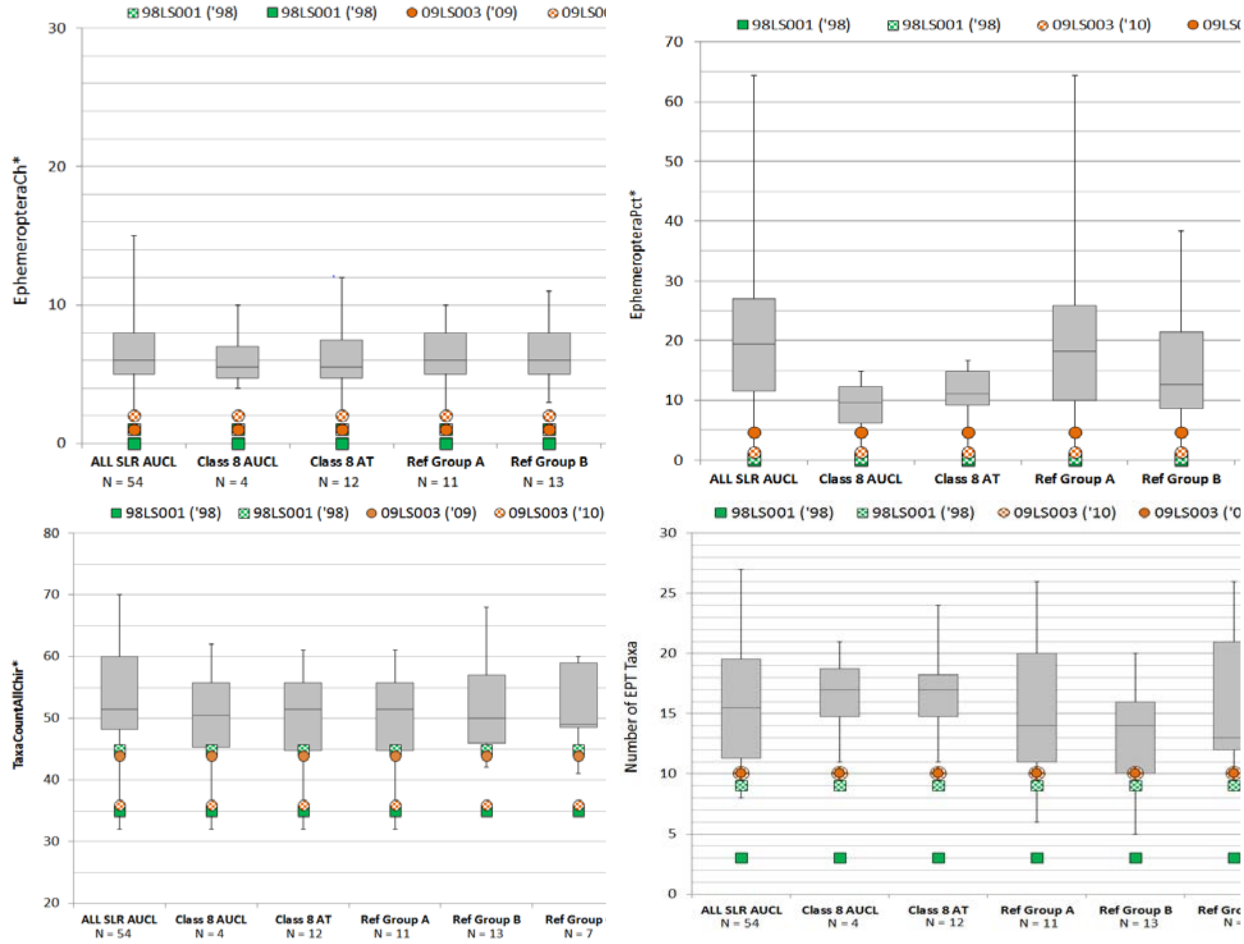


**Figure 85:** Biological response metric data for specific conductivity. Comparison of Mille r Creek data to SLRW reference sites

***** See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

### Summary

Elevated chloride concentrations, and the resulting short and long tern spikes in specific conductivity, are likely contributing to the macroinvertebrate impairment in Miller Creek. The biological response data presented in this report shows a clear suppression and/or absence of macroinvertebrate taxa that are sensitive to exposure to high chloride concentrations and elevated specific conductivity.

The sources and pathways of these stressors within the watershed are well understood, but additional data would be beneficial. Clearly, road salt application during the winter months is the primary driver of this stressor, and based on the limited data available, conditions in the stream during this time may be acutely toxic for sensitive forms of aquatic life. Additional monitoring during this critical time (December through February) is recommended, particularly at locations upstream and downstream of areas where intensive road de-icing efforts are occurring. Additional work is also recommended to explore chloride concentrations in the groundwater, stormwater ponds, and riparian corridors connected to Miller Creek. Mid-summer chloride concentrations and specific conductivity levels are clearly elevated above background conditions, most likely due to contamination of the groundwater and riparian corridors from excessive road salt applications during the winter season.

## **Miller Creek:** Summary of Stressors to Aquatic Life

**Table 31:** Summary of SID results for Miller Creek

| Candidate Cause | Result |
|---|---|
| **Elevated Water Temperatures** | ● |
| **Total Suspended Solids (TSS) / Turbidity** | **X** |
| **Chloride Toxicity / Sp. Conductivity** | ● |
| **Altered Hydrology** | ○ |
| Key: ● **= confirmed stressor** ○ **= Potential Stressor X = eliminated candidate cause** | |

## 5.4 Otter Creek

Otter Creek is the lone impaired stream in this watershed zone, and it appears to share similar symptoms of impairment to coldwater streams in the DUC watershed zone. Otter Creek originates in a series of wetlands, and meanders through a riparian corridor dominated by alder and willow shrubs, interspersed with localized stands of pine and several bedrock outcroppings. Over the past few decades, land in this watershed has been increasingly developed due to the expansion of a large casino, new housing developments, and several gravel mining pits within close proximity of the stream.

The upper reaches of Otter Creek follow a fairly consistent pattern of relatively steep reaches of C channel in short glacial trough valleys connecting E channels winding through longer lacustrine valleys. In areas of development pressure around the Black Bear Casino and Interstate 35, the channel has been re-routed and straightened. These changes were the result of a former gravel mining operation on the site now occupied by the casino. The majority of the impaired reach consists of C and E channel types in a lacustrine valley; although for the final two-thirds of a mile before entering the St. Louis River the stream flows through a bedrock valley as a B channel. The Otter Creek impaired AUID has an average slope of 0.4%, dropping 135 feet in 6 miles.

**Figure 86:** Looking upstream at Otter Creek monitoring station 09LS005 during low flow in September 2012 (left), and downstream during a high flow event in June 2012 (right).

Otter Creek is a designated trout stream for most of its length. Both brook and Brown Trout were observed in moderate abundance during the 2009 sampling of station 09LS005, which is located just upstream of Highway 1 in the city of Carlton, Minnesota. The DNR has stocked Brown Trout in this reach annually since 1993, but Brook Trout are not regularly stocked. All of the trout sampled in 2009 were adults, which may be an indication that natural reproduction is low within this stream. Another possibility is that spawning and rearing occurs in other reaches of Otter Creek or tributary streams that were not sampled. A small population of Brook Trout was also present in Little Otter Creek, which is a major tributary to Otter Creek.

### Background Information on Macroinvertebrate Impairment Listing

Macroinvertebrate data were collected from three stations on Otter Creek. Two of these stations, 09LS005 and 12LS001, are located on the impaired reach. The macroinvertebrate impairment for this stream stems from data collected in 2009 at station 09LS005. Station 12LS001 was not sampled until after the stream was assessed and listed as an impaired water. The third station is located upstream of the impaired reach, and has not been sampled since 1997. Since the data from this site is outside of the MPCA's assessment window of 10 years, data from this site were not considered during the assessment process.

The MIBI scores for stations on the impaired reach are all below the impairment threshold, although several sample visits produced results that are narrowly below the threshold and within the lower confidence limit (Table 32). These results suggest that the magnitude of the impairment is not severe, and restoration potential is high if watershed stressors can be addressed. A repeat visit to station 09LS005 during the fall of 2012 produced the lowest MIBI score observed within the impaired reach. This low score may have been influenced by a major rain event that occurred in the region in June of 2012, which dropped 8-10 inches of rain in a 24-hour period in the area of this watershed. Station 12LS001 was also sampled during the fall of 2012 and scored narrowly above the threshold (0.49 points above). Thus, regardless of the impacts from the flood, station 09LS005 appears to support a poor quality macroinvertebrate assemblage than the upstream station.

Like many of the other macroinvertebrate-impaired streams in the SLRW, Otter Creek supports relatively few sensitive taxa and tolerant taxa are fairly common. Poor scores in the several tolerance based metrics used to assess these sites (# intolerant taxa, % very tolerant taxa) are one of the main reasons this stream failed to meet the MIBI criteria. Data from this stream also indicates large proportion of macroinvertebrate taxa from the functional feeding group (FFG) "collector-gatherers." These organisms feed by collecting fine particulate organic matter (segments of leaves, twigs, and other plant matter) and can be a sign of sedimentation if coarser substrates are buried or embedded under fines.

The more favorable MIBI score at station 12LS001 was due to a higher number of individuals from the orders **P**lecoptera, **O**donata, **E**phemeroptera, and **T**richoptera (POET) observed in the sample. In addition, a lower percentage of non-insect taxa were observed at this station compared to 09LS005, which also factored heavily into the disparity between the MIBI scores at these sites.

**Table 32:** Summary of biological monitoring stations, macroinvertebrate sampling results, and applicable standards

| **Macroinvertebrate Assessments** | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Station** | **Drainage Area (mi2)** | **Gradient (%)** | **Stream Order (Strahler)** | **Invert IBI Class** | **Invert IBI Result *(visit year)*** | **Invert IBI Result *(visit year)*** | **Standard** | **IBI Lower Confidence Limit** | **IBI Upper Confidence Limit** |
| **12LS001** | 36.81 | 0.25 | 3 | 8 | **32.49** *(2009)* | - | 32 | 19.6 | 44.40 |
| **09LS005** | 39.72 | 0.17 | 3 | 8 | **24.86** *(2009)* | **16.54** *(2012)* | 32 | 19.6 | 44.40 |
| **97LS086*** | 10.33 | 0.27 | 2 | 8 | **68.78** *(1997)* | - | 32 | 19.6 | 44.40 |
| **09LS116*** | 16.93 | 0.16 | 2 | 8 | **27.07** *(2009)* | - | 32 | 19.6 | 44.40 |

A review of existing data was performed to develop a list of candidate causes for the Otter Creek M-IBI impairment. Based on the results of this review, the following candidate causes of impairment will be evaluated in this section:

1. **Elevated water temperatures**
2. **Lack of habitat**

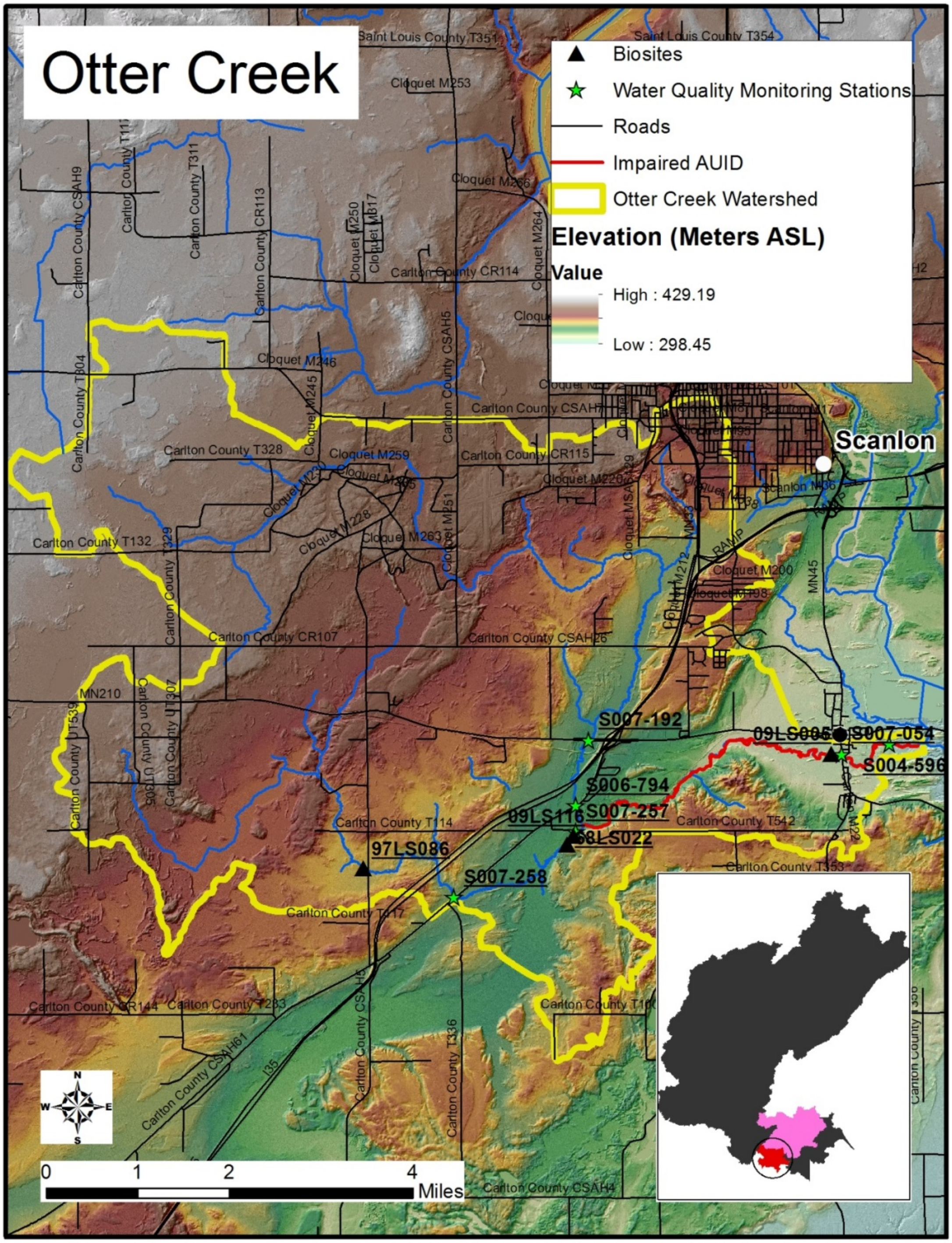


**Figure 87:** Map of Otter Creek Watershed and impaired stream reach

### Elevated Water Temperatures

Water temperature data was collected at three MPCA biological monitoring stations on Otter Creek; station 09LS005, south of the town of Carlton; station 12LS001, near the Munger Trail; and station 68LS022, located upstream of the mouth of Little Otter Creek. The locations of these monitoring sites are shown in Figure 88. Water temperatures were recorded at 15-minute intervals from mid-May through September, but the majority of the data analysis in this section will focus on the period between June and late August, when stream temperatures are most likely to exceed the stress threshold for coldwater-sensitive macroinvertebrate species.

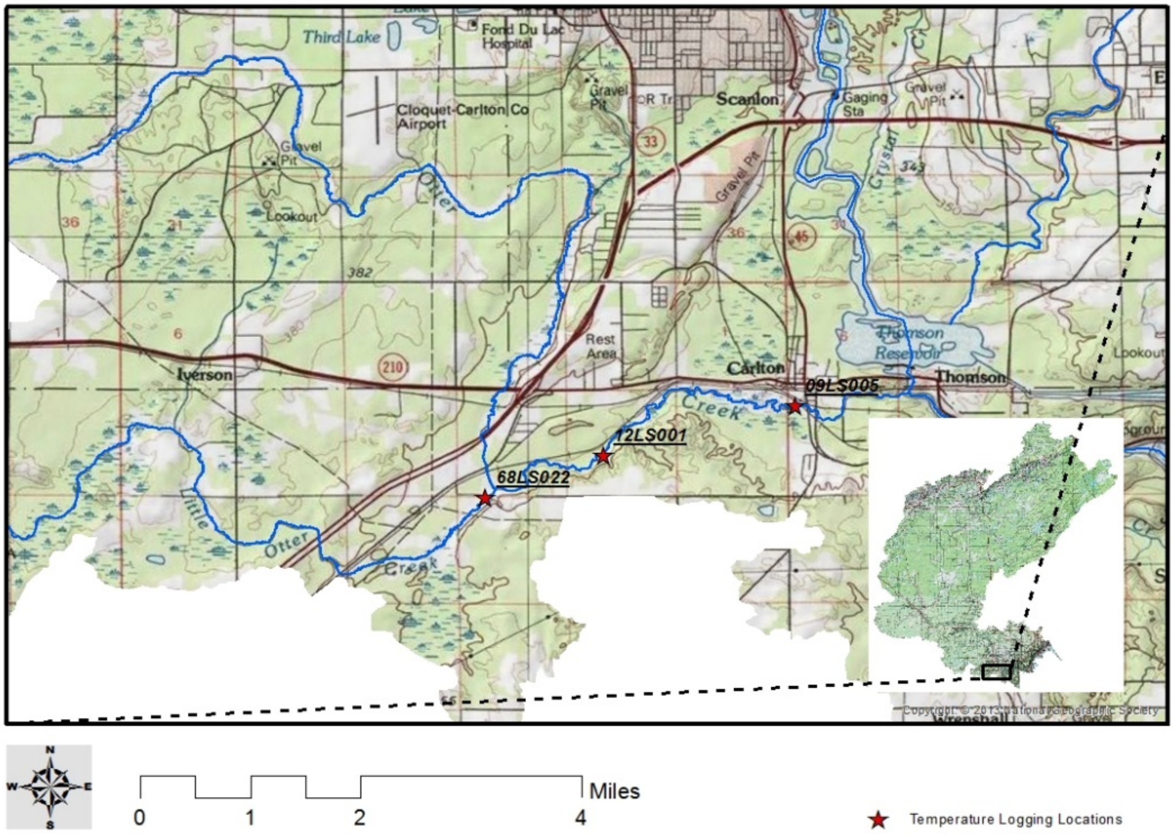


**Figure 88:** Location of biological monitorign stations with continuous temperature data

Stream temperatures in Otter Creek appear to be heavily influenced by ambient air temperature. The temperture logger installed in 2009 (solid red line) recorded average daily temperatures that exceeded the stress threshold for only three periods of short duration (Figure 89). As has been stated before, the 2009 summer was much colder than normal. Conversely, the summer of 2012 (solid black line) saw average daily temperatures in Otter Creek that exceeded the stress threshold of 68° F for essentially the entire month of July. The summer of 2012 was warmer than average, but closer to normal than the summer of 2009.

The average daily temperature in Little Otter Creek (a tributary to Otter Ck) strongly mirrors that of Otter Creek. In fact, during both recorded periods the average water temperature in Little Otter Creek rarely strays by even a degree Fahrenheit from the average in Otter Creek (even in 2009, when the temperature loggers were more than three miles apart). Altlough Little Otter Creek was not ultimately listed as impaired, the MIBI scores were very similar to those from Otter Creek, and these two streams seem to support macroinvertebrate communities with the same thermal preferences (see Figure 90).

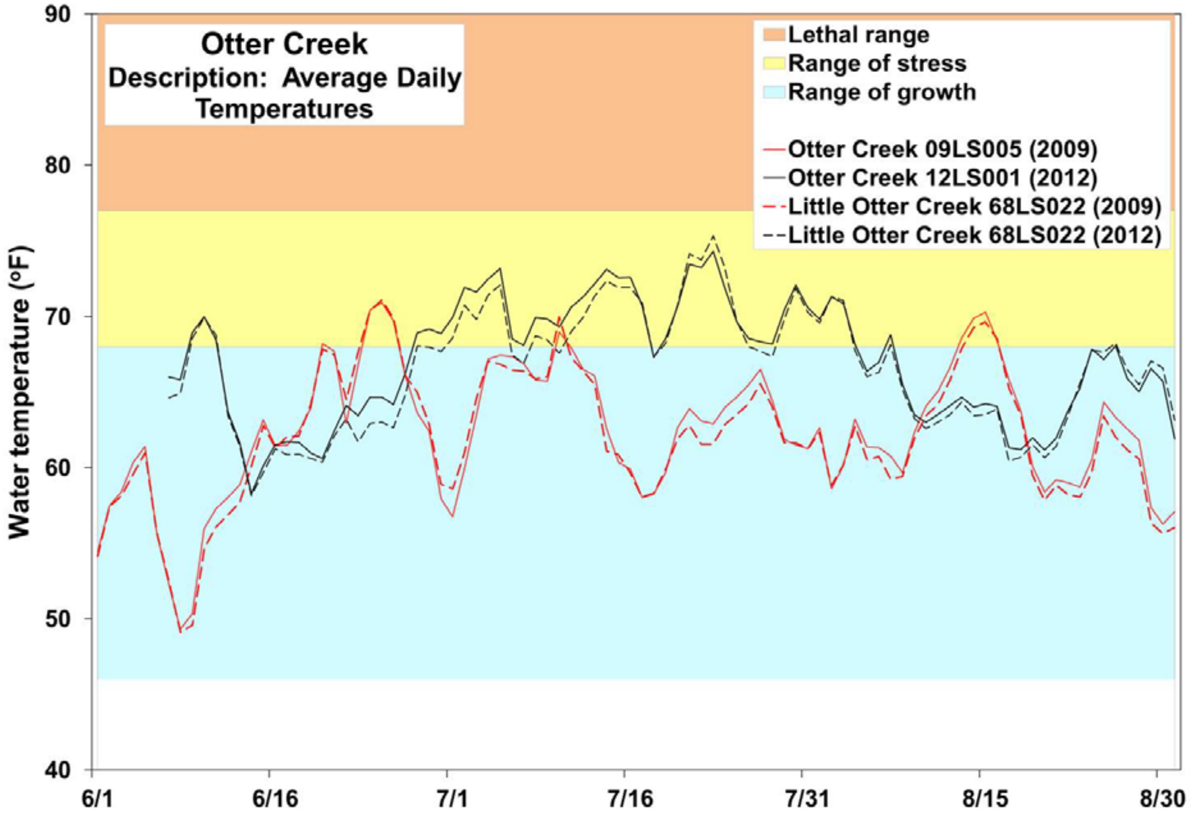


**Figure 89:** Average daily temperatures for Otter Creek at monitoring stations during the years 2009 and 2012

### Biological Response to Elevated Water Temperature

Species specific TIV related to water temperature were used to evaluate the thermal preferences of macroinvertebrates in Otter Creek and several comparable coldwater streams. The TIV values were broken into groupings by 10th percentiles or “deciles” to differentiate between organisms that have been most commonly found in cold, cool, warm, or very warm water in the state of Minnesota. More emphasis should be placed on the cold and warm water groupings, as there is likely a significant amount of overlap in the cool to warm transitional species. Those organisms that are most commonly found in cold water habitats are of most importance in this analysis considering that Otter Creek is an impaired coldwater stream.

Compared to several high quality SLRW coldwater streams that were used for comparison (Keene Creek and Hay Creek), the stations on Otter Creek and Little Otter Creek supported a lower relative percentage of macroinvertebrates that are linked to coldwater conditions (Figure 90). Still, macroinvertebrate taxa associated with cold stream temperatures were still present in Otter Creek, and in some cases, a fairly significant percentage of the organisms sampled are associated with cold or coolwater streams (i.e. 29% at station 09LS005 in 2012, Figure 90). These data suggest that the thermal regime of Otter Creek is somewhat marginal for supporting abundant and diverse communities of coldwater macroinvertebrates compared to high quality coldwater streams of the region.

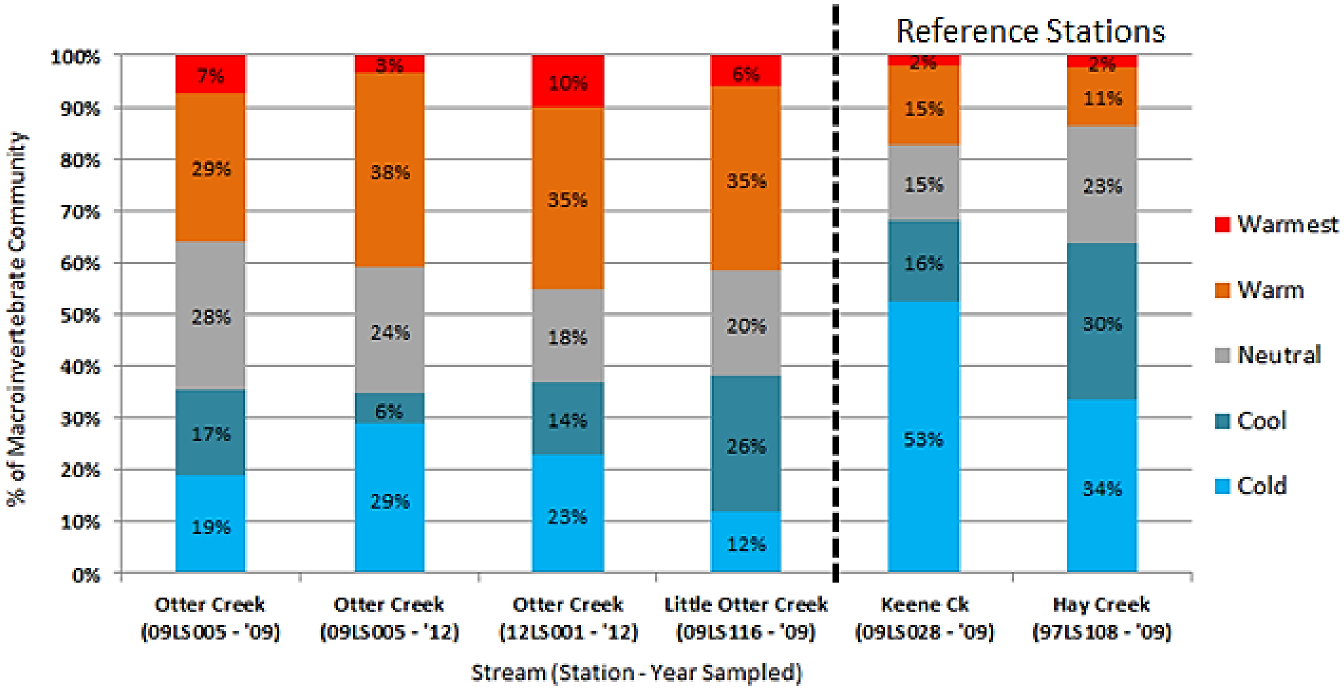


**Figure 90:** Percentage of macroinvertebrate community associated with cold, cool or warm stream temperatures observed in Otter Creek compared to reference stations

### Is Elevated Water Temperature a stressor in Otter Creek?

The temperature regime of Otter Creek is marginal for supporting a high quality coldwater macroinvertebrate assemblage. Stream temperature data collected from several locations in the watershed was highly responsive to ambient air temperatures, which suggests a somewhat limited groundwater influence on water temperatures within the reaches that were sampled. During the summer of 2012 when ambient air temperatures were slightly warmer than average, stream temperatures in Otter Creek were within the "stress level" range for Brook Trout for extended periods of time. However, MN DNR reports that there is a small, but naturally reproducing population of Brook Trout in this stream. This suggests that local temperatures suitable for sustaining coldwater taxa, or that coldwater refuge areas (springs/tributaries) are accessible within this reach of Otter Creek.

The Otter Creek Watershed is not pristine, and there are several potential sources of thermal loading to the creek. These include commercial and residential development, impervious surfaces from roads and interstate highways, stream channelization, and vegetation removal. However, natural background conditions in the watershed may not be entirely favorable for coldwater fish and macroinvertebrates.

Elevated water temperature is considered a possible contributing cause to the low coldwater MIBI scores. However, the confidence level in diagnosing this stressor is somewhat low due to the annual variability in the available temperature data, the presence of some coldwater macroinvertebrate individuals in Otter Creek, and the possible contributions from natural background conditions in the watershed.

### Poor Physical Habitat Conditions (Substrate)

Physical habitat conditions in Otter Creek were evaluated at station 09LS005 using the MSHA methodology. The overall MSHA score of 77.15 out of 100 at this station corresponds to a habitat rating of “good.” However, there are several sub-category scores that suggest that several components of the physical habitat may be somewhat limiting for a healthy macroinvertebrate assemblage. Based on the MSHA results and observations made during several site visits, coarser grained substrates (cobble and gravel) at station 09LS005 are moderately embedded by sand and other fine particles. The MSHA substrate component score for Otter Creek was near the 25th percentile value for coldwater streams in the SLRW (Figure 91), which reflects the moderate level of substrate embeddedness at this station. Excessive deposition of fine sediment can degrade macroinvertebrate habitat quality, reducing productivity and altering the community composition (Rabeni et al. 2005, Burdon et al. 2013).

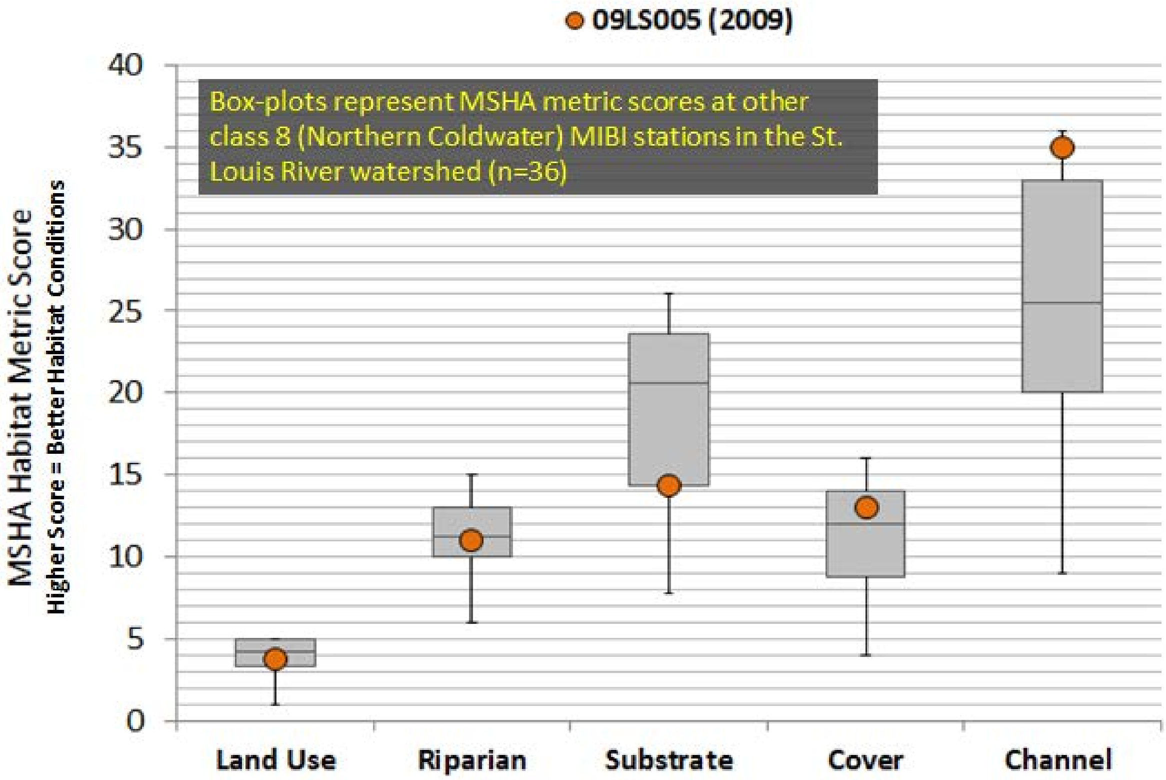


**Figure 91:** MSHA metric scores for Otter Creek station 09LS005 compared to the box plot distribution of results from all other class 8 (Northern Coldwater) stations in the SLRW.

### Lack of riffle/glide Features

The original sampling reach on Otter Creek (09LS005) is low gradient and lacks shallow, fast-water habitats such as riffles and glides. Riffles and glides are often productive microhabitats with diverse and abundant macroinvertebrate assemblages. Compared to other class 8 MIBI stations in the SLRW, 09LS005 is dominated by “run” and “pool” habitats and had one of the lowest percentages of riffle habitat in the entire watershed for streams of this MIBI class (Figure 92). The lack of riffle and glide habitats within this sampling reach are one of several natural background factors that are likely factoring into the low MIBI scores observed.

**Figure 92:** Otter Creek station 09LS005 is low gradient and lacks shallow, fast-water riffle and glide habitats

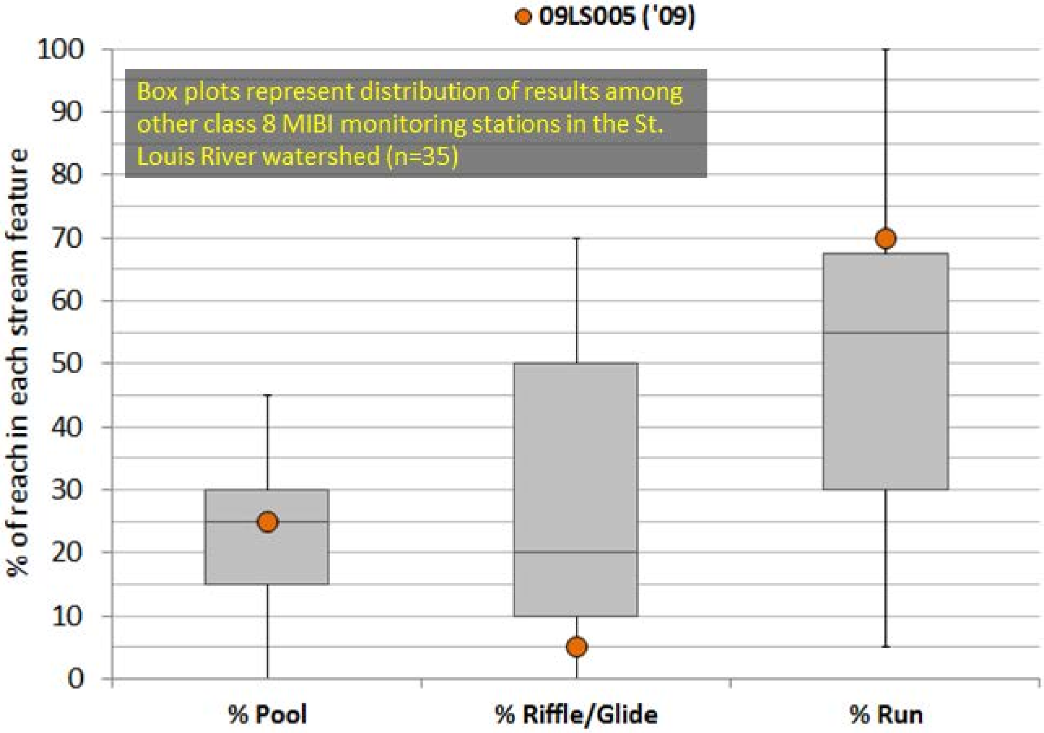


**Figure 92:** Percentage of the sampling reach in pool, riffle/glide, and run habitat at station 09LS005 compared to other class 8 MIBI stations in the SLRW.

## Biological Response to Poor Substrate Conditions

Macroinvertebrate taxa with specific feeding or other life history traits that require clean, coarse substrates are often the first to decrease in richness and abundance in streams with high rates of sedimentation. The following macroinvertebrate metrics cover some of the more sensitive taxa that have shown fairly predictable responses in streams with high rates of embeddedness or those dominated by fine substrate (silt/clay/sand).

### % Clinger Individuals

Clinger macroinvertebrates usually have flattened body forms and attach themselves to firm substrates (mostly rocks, wood) in swift water habitats. The relative percentage of clinger taxa observed during the two sampling visits to Otter Creek station 09LS005 ranged from 16 – 48%. Overall, this station supported a lower percentage of clinger individuals than the majority of the class 8 (Northern Coldwater Streams) monitoring stations in the SLRW that scored above the MIBI impairment threshold (high quality sites) (Figure 93). The 2009 sample (16% clinger individuals) in particular represented a very low relative percentage of clinger individuals compared to high quality stations. Two extreme rain events in 2012 may have scoured away much of the fine substrates in the sampling reach that were embedding the coarser gravels and cobbles. This in turn would've created better habitat conditions for clinger macroinvertebrates, which may explain the fairly significant increase in their relative abundance in the 2012 sample.

Results for this metric generally support substrate embeddedness and poor substrate conditions as a contributing cause to the MIBI impairment in Otter Creek.

### %Sprawler Individuals

Sprawler macroinvertebrates live on the surface of floating aquatic plants or fine sediments, and usually possess adaptations for staying on top of substrate and keeping respiratory surfaces free of silt. Sprawler individuals accounted for 8% and 22% of the total macroinvertebrate community during the two sampling events at station 09LS005. The higher percentage of sprawler individuals (22%) was observed in the 2009 sampling event, which is consistent with other metrics that are being used to evaluate poor substrate conditions as a stressor. Again, the large flood events prior to the 2012 sampling event likely scoured the streambed of fine sediment and significantly changed the macroinvertebrate assemblage. In 2009, Otter Creek 09LS005 supported a higher relative percentage of sprawler individuals than 75% of the class 8 MIBI stations scoring above the impairment threshold (Figure 93). In 2012, the relative percentage of sprawler individuals at this station was *lower* than 75% of those stations. Clearly, the results of this metric and others evaluated here were influenced by the two significant rain events in the early summer of 2012.

### % EPT individuals

The relative abundance of organisms in the order Ephemeroptera (Mayflies), Plecoptera (Stoneflies), and Trichoptera (Caddisflies) (EPT) is widely used indicator of biological health to evaluate many stressors. The relative abundance of EPT individuals has been shown to decrease in streams with degraded habitat conditions (source). Percent EPT individuals ranged from 18% (2009 visit) to 26% (2012 visit) during the two monitoring visits to station 09LS005. Both of these results are poor relative to high quality class 8 MIBI stations, particularly the 2009 result of 18% EPT individuals (Figure 94).

## % Burrower Individuals

Burrower macroinvertebrates inhabit the fine sediments of streams and lakes. A stream dominated by burrower individuals or taxa can be a good indicator that a stream reach is dominated by fines and lacks quality coarse substrate. Only 2% and 7% of the total macroinvertebrate community were “burrowers” in the two sampling events at station 09LS005 (Figure 94). These results are quite low in comparison to high quality stations of the same MIBI class, and were generally lower than the majority of high quality stations in the SLRW. Station 09LS005 was not dominated by silt and sand substrates throughout the reach, and the low percentage of burrower taxa is further evidence of this. It appears that most of the habitat degradation related to sedimentation in Otter Creek, if any, is occurring through the filling of interstitial spaces in riffle and glide areas (embedded coarse substrates).

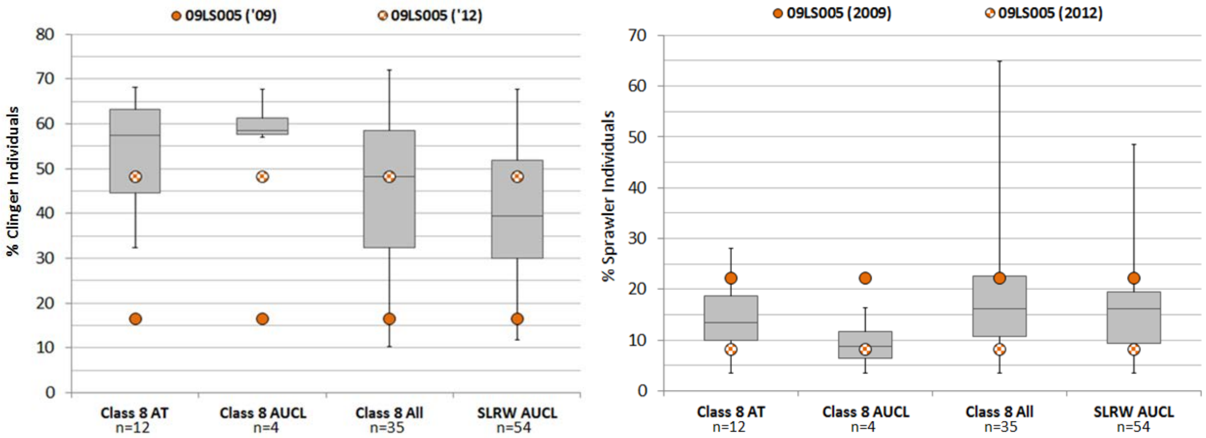


**Figure 93:** % clinger individuals (left) and % sprawler individuals (right) observed at Otter Creek 09LS005 compared results from various groupings of SLRW stations. See Table 21category descriptions.

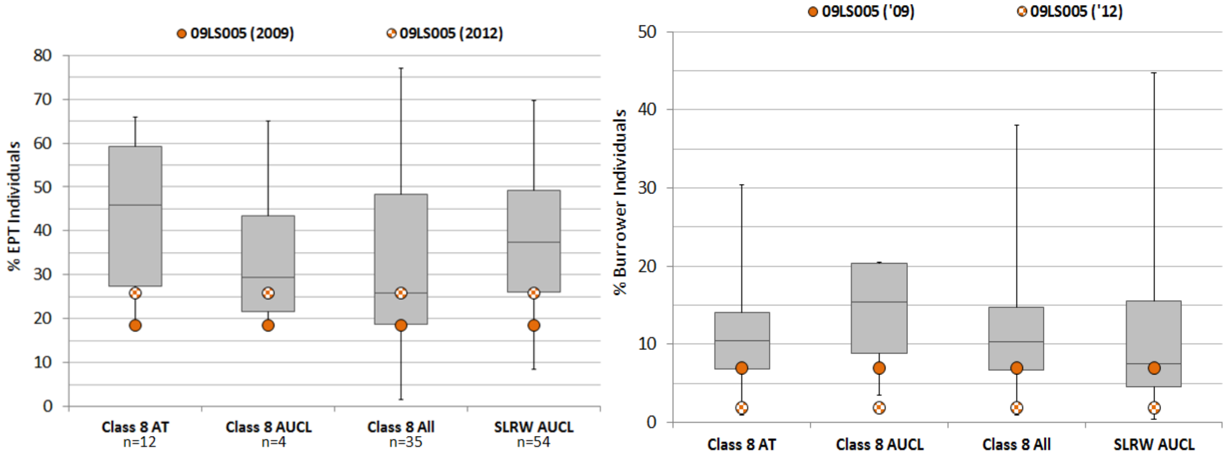


**Figure 94:** % EPT individuals (left) and % burrower individuals (right) observed at Otter Creek 09LS005 compared results from various groupings of SLRW stations. See Table 21category descriptions.

### Are Poor Physical Habitat Conditions a Stressor in Otter Creek?

Overall, habitat conditions in the impaired reach of Otter Creek are in relatively good condition, but several characteristics of the physical habitat may be limiting MIBI scores. The original sampling reach used to assess aquatic life in Otter Creek (09LS005) sits in a lacustrine valley and is bordered by wetlands and bogs. The gradient is very gradual and the stream lacks riffle and glide features. These habitat conditions contrast much of what is observed in the higher quality coldwater streams in Northeaster Minnesota, which tend to feature steeper gradients and abundant riffles. The MSHA results from station 09LS005 also show moderate levels of substrate embeddedness, which further limits the availability and quality of habitat for certain macroinvertebrate taxa.

Marginal physical habitat conditions, specifically the lack of quality substrate and riffle features, are likely contributing to the low MIBI scores observed in Otter Creek. Restoration activities within the impaired reach may not be able to significantly improve the physical habitat for macroinvertebrates. No major bank erosion or other sources of sediment were observed within the vicinity of the biological monitoring site. The pattern, profile, and dimensions of the stream channel within the biological monitoring reach do not indicate a high degree of instability or departure from reference conditions.

## **Otter Creek:** Summary of Stressors to Aquatic Life

**Table 33:** Summary of SID results for Otter Creek

| Candidate Cause | Result |
|---|---|
| **Elevated Water Temperatures** | ○ |
| **Poor Physical Habitat Conditions** | ○ |
| Key: ● **= confirmed stressor** ○ **= Potential Stressor X = eliminated candidate cause** | |

### Meadowlands Floodwood Peat Bog

The Meadowlands Floodwood Peat Bog (MF-PB) Watershed zone exhibits one of the highest impairment rates of any region in the SLRW. The impaired streams in this region many similarities in terms of the symptoms of impairment and potential stressors. For the most part, streams of this watershed zone are very low gradient, and are lacking coarse substrates and riffle-run habitats. With the exception of the impaired reach on the main stem of the St. Louis River, all of the impairments are found on small tributary streams draining a network of ditches within the expansive Meadowlands Sax-Zim peat bog. All of these streams are severely tea-stained in color and low in alkalinity, which is a natural background condition that may be limiting the diversity and abundance of aquatic life.

**Figure 95:** Some representative stream reaches from the Meadowlands Floodwood Peat Bog (MF-PB) Watershed zone. Pictured are Stony Creek (left), Vaara Creek (center), and Sand Creek (right). All of these streams are impaired for biological measures.

The impaired streams of this watershed zone generally support few species of fish, and overall fish abundance is also low in comparison with other streams of similar size in the SLRW. Populations of non-tolerant headwater minnow species, such as Northern Redbelly Dace, Pearl Dace, and Finescale Dace were lacking at the impaired sites in this watershed zone. Instead, the impaired streams were typically dominated by species highly tolerant to low DO conditions (Central Mudminnow, Black Bullhead) or species that are known to migrate into low gradient streams seasonally when conditions are favorable (e.g. Northern Pike).

A lack of insectivorous fish species is another symptom of impairment that was common across most of the impaired stations in the MF-PB Watershed zone. This may be an indication that the food base in degraded streams has been altered by habitat degradation, eutrophication, or other processes. The insect life available as prey may also be lower in these streams due to the natural background conditions found in this watershed zone (low alkalinity, lack of coarse substrates/riffle habitats). Taxa richness of simple lithophils (fish that require non-embedded gravels or cobble for spawning) was also very low at most of the impaired locations. This is another symptom of impairment that is often linked to a lack of coarse substrate in streams.

The macroinvertebrate communities at impaired sites within this watershed zone tend to be “unbalanced,” or in other words, dominated by several taxa. For example, the five most common taxa in Vaara Creek and Skunk Creek accounted for 85% and 73% of the total individuals sampled. An unbalanced macroinvertebrate community can be an indicator of reduced habitat complexity or the presence of a stressor that would affect a broad range of taxa.

Many of the impaired sites in this IBI class scored poorly in a metric that measuring the richness of “clinger” macroinvertebrate taxa. These taxa maintain a relatively fixed position on firm substrates, often in areas where current velocities are higher. Their reliance on coarse substrate and interstitial spaces between substrate particles as habitat renders them vulnerable to benthic habitat degradation, particularly in the form of sedimentation (embeddedness).

## 5.5 St. Louis River

The impaired section of the St. Louis River flows through the bottom end of the historic bed of Glacial Lake Upham, and is thus extremely low-gradient. The impaired reach of the St. Louis River has the flattest gradient of any reach on the entire river, with an average slope of less than 0.01% (six inches/mile). Field level channel cross-section data were not gathered here, but the river is fairly wide and not very sinuous. Thus it was assumed that the river would not type out as an E channel, but instead as a low-gradient C channel (Cc) (refer to Appendix A for channel and valley types). It may be that the St. Louis is an F channel through this reach, but impossible to say with certainty without data showing that the river is entrenched.

This reach of the St. Louis River was listed as impaired based on low MIBI scores. Only one biological monitoring station is located within the impaired reach (97LS090), and it has failed to meet the MIBI impairment threshold in each of the three monitoring visits (Figure 96). MIBI scores, overall taxa richness and intolerant taxa richness were all lower within the impaired reach compared to reaches of the SLRW that are not listed as impaired for MIBI assessments. Abundant macroinvertebrate taxa observed at the impaired biological monitoring station (97LS090) include *Leptophlebiidae* (prong-gilled mayflies), *Baetis* (mayfly genus), *Hyallela* (freshwater amphipods), *Rheotanytarsus* and *Polypedilum* (non-biting midges), and *Macronychus* and *Stenelmis* (riffle beetles).

**Table 34:** Summary of biological monitoring stations, macroinvertebrate sampling results, and applicable standards

| Macroinvertebrate Assessments | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | Invert IBI Class | Invert IBI Result *(visit year)* | Invert IBI Result *(visit year)* | Invert IBI Result *(visit year)* | Standard | IBI Lower Confidence Limit | IBI Upper Confidence Limit |
| **97LS090** | 1,936.3 | 0.01 | 5 | 1 | **33.68**<br>*(2009)* | **45.82**<br>*(2009)* | **35.52** *(2012)* | 49.0 | 38.2 | 59.8 |

A review of available water chemistry, physical habitat, and land-use data was performed to develop a list of candidate causes for the MIBI impairment on this main stem reach of the St. Louis River. The following potential causes were selected for further analysis in this section.

1. **Lack of quality habitat**

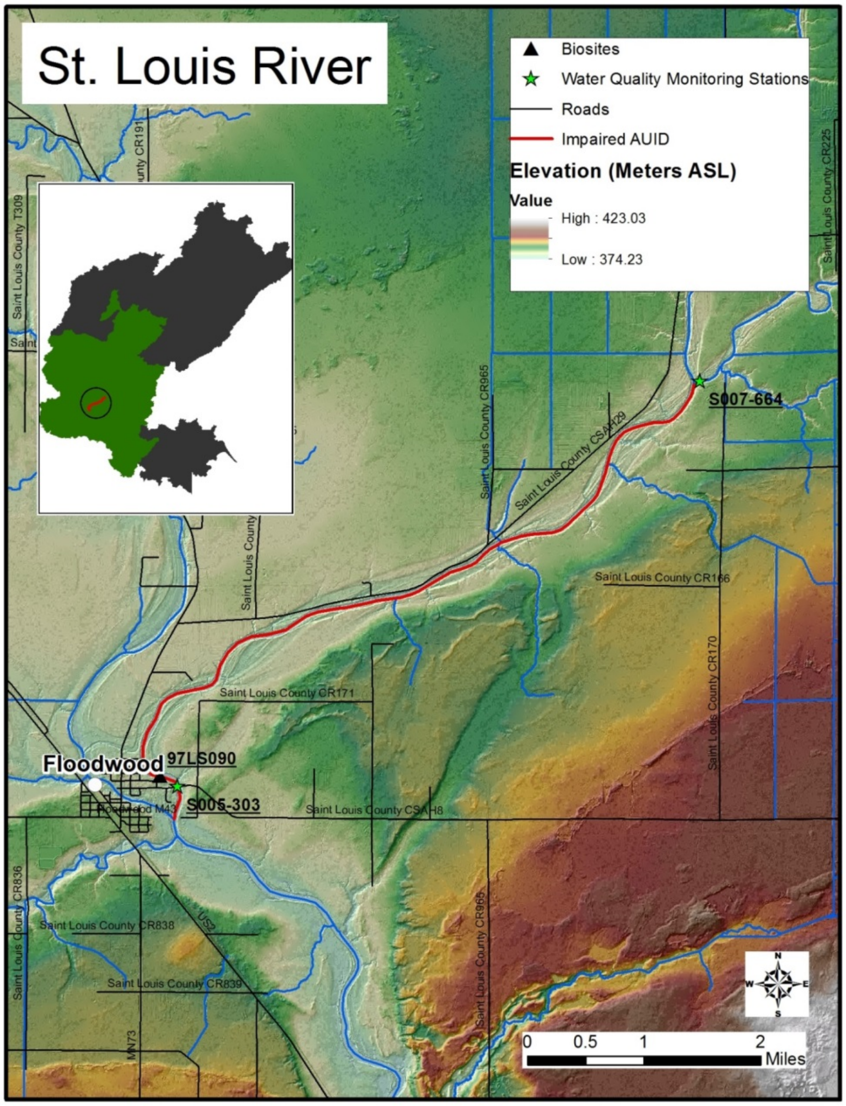


**Figure 96:** Map of impaired reach of the St. Louis River and monitoring locations

### 5.5.1 Poor Physical Habitat Conditions

The impaired reach of the St. Louis River extends from the confluence with the Whiteface River down to the confluence with the Floodwood River. This reach flows through the nearly level bed of Glacial Lake Upham, and as a result it has a much lower gradient than other reaches of the river, and a streambed dominated by fine particles (silt and clay). These conditions, coupled with the infertile geological features in this region, have been cited as potential causes for limited fisheries productivity in this reach of the river (Lindgren et al. 2006). The MIBI scores were below the impairment threshold within this same reach, but elsewhere on the river, scores were generally good to excellent.

Low MIBI scores and similar impairment symptoms in the biota are observed for a 20-mile stretch of the St. Louis River, including areas downstream of the impaired reach. Stream gradient is fairly low throughout this entire reach, and many of these stations with poor MIBI scores share similar habitat limitations. Compared to St. Louis River monitoring stations upstream and downstream, sites in the vicinity of the impaired reach generally have poorer substrate conditions, less in-stream cover, and morphological attributes that are not conducive to quality habitat (Figure 98).

**Figure 97**: Photos from station 97LS090 taken in 2009 (left) and 1997 (right). Note the extremely wide channel with very few defined stream features (riffles, glides) and stagnant flow conditions.

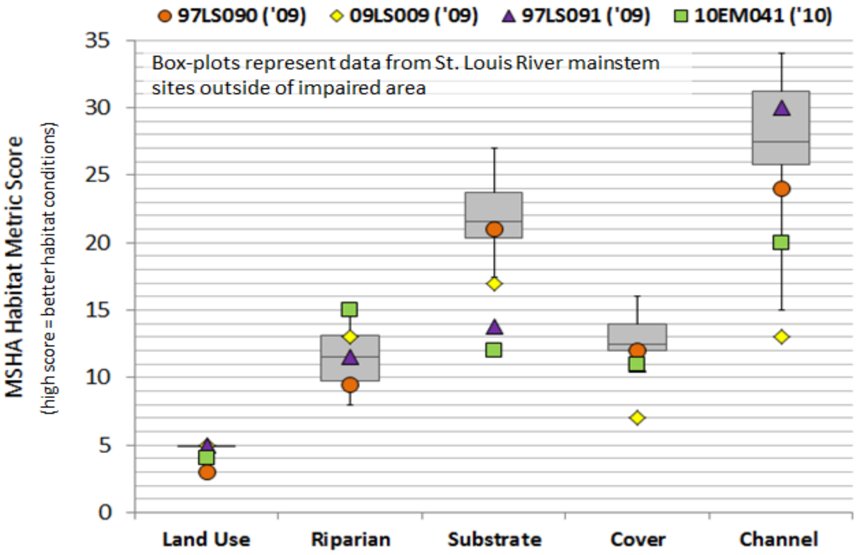


**Figure 98:** MSHA metric scores for St. Louis River sites within and near the impaired reach (points) compared to stations with higher MIBI scores that are located up or downstream

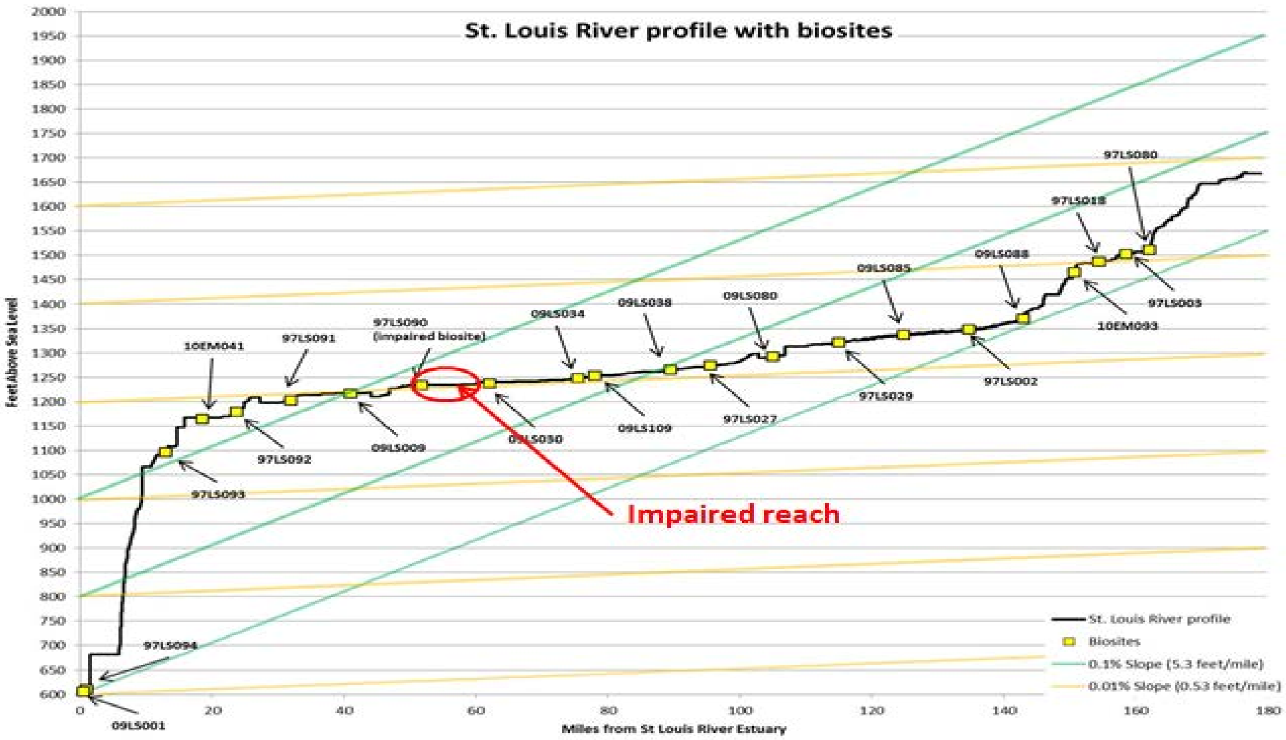


**Figure 99:** Longitudinal elevation profile of St. Louis River. Note the relatively flat gradient of impaired reach in comparison to the upper ½ of the river where biological integrity scores are more favorable.

### Biological Response to Poor Habitat Conditions

Several M-IBI metrics appear to be responding negatively to localized stressors within this low-gradient, habitat-limited reach of the St. Louis River. Similar to observations of the fish community made by Lindgren et al. (2006), overall macroinvertebrate taxa richness begins to decline as the St. Louis River enters the area of former Glacial Lake Upham. Taxa richness counts remain suppressed at several monitoring stations extending nearly 20 miles downstream of the impaired reach (Figure 100). These stations are also lower in gradient, and may be impacted by glacial lake sediments and several tributary streams that are heavily influenced by bogs and wetlands (e.g. Savannah River, Floodwood River). The lack of taxa richness observed at these sites was consistently a factor in the low MIBI scores observed at stations in this reach of the St. Louis River. Low taxa richness can be a symptom of many stressors, but the localized nature of the response and co-location with these lower gradient sites provides good evidence in support of limited habitat as a stressor.

In addition to low overall taxa richness, macroinvertebrate communities within the impaired reach tended to be dominated by several taxa. Observations from station 97LS090, which is the only station officially located on the impaired segment, show a community that was dominated by Leptophlebiidae (Prong-gilled mayflies) and Hyallela (freshwater amphipods). These two genera accounted for 88% and 84% of the macroinvertebrates counted from samples collected during two visits to 97LS090 in the fall of 2009. Drastic changes in the macroinvertebrate assemblage were observed during a repeat visit to this station in 2001, as Leptophlebiidae taxa were not observed, and overall, there was considerably more balance among the taxa that were present.

The percentage of EPT (Ephemeroptera – “mayflies”, Plecoptera – “stoneflies”, Trichoptera – “caddisflies”) observed also drops precipitously downstream of the impaired reach. EPT percentages in the upper reaches of the St. Louis ranged from 40-70% of the total community, but downstream of the impaired reach, only around 2-20% of the community were individuals representing EPT taxa (Figure 102). Station 97LS070, which as the primary site that was assessed to list this stream as impaired, had fairly a fairly high percentage of EPT individuals in several sampling events (Figure 102).

Several of the more common macroinvertebrate metrics used to evaluate degraded habitat conditions did not respond in a manner that supports physical habitat as a stressor. Despite fairly poor MSHA scores for substrate conditions, station 97LS090 supported a fairly large population of macroinvertebrates that cling to hard surfaces (“clingers”), and had a relatively low number of individuals that sprawl out on fine substrates (“sprawlers”). Many of the other habitat impaired streams in the region were dominated by “sprawlers” and supported very low numbers of “clinger” macroinvertebrates.

**Figure 100:** Overall macroinvertebrate taxa richness (left) and % dominant two taxa (right) observed at St. Louis River monitoring stations

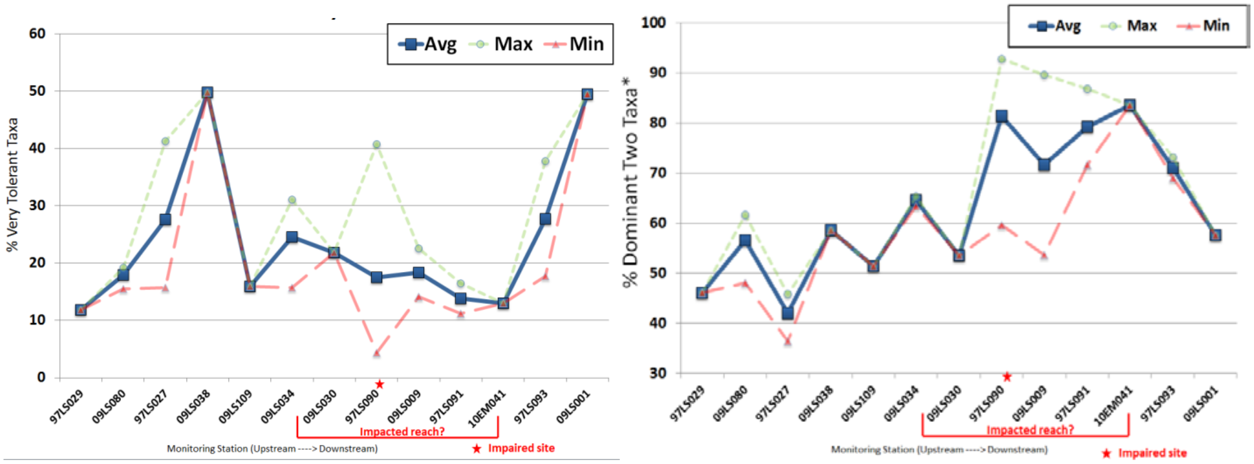


**Figure 101:** A series of macroinvertebrate metric data showing a decline in biological integrity within and around the impaired reach of the St. Louis River

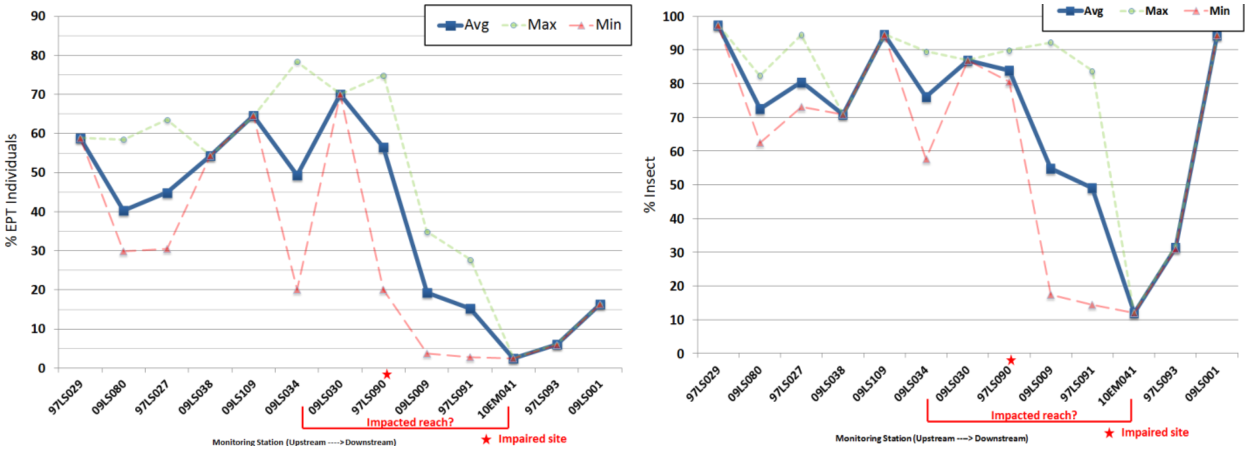


**Figure 102:** A series of macroinvertebrate metric data showing a decline in biological integrity within and around the impaired reach of the St. Louis River

### Summary: Are poor physical habitat conditions a stressor in the St. Louis River?

The impaired reach of the St. Louis River is located in a region of extremely flat topography created by Glacial Lake Upham. Compared to upstream and downstream section of the St. Louis River, the impaired reach is extremely low gradient, with a very high width to depth ratio and a lack of riffle and glide habitats. The lack of habitat heterogeneity within this reach is likely a major factor in the observed declines in macroinvertebrate taxa richness within and around the impaired reach.

Water chemistry variables (temperature, DO, pH, specific conductivity, TSS) within this reach were not substantially different than other non-impaired areas of the St. Louis River. The lack of candidate stressors associated with water chemistry adds strength to the argument for marginal physical habitat conditions as a stressor.

Restoring habitat for macroinvertebrates in this reach may not be feasible or cost effective. FIBI results from this reach have all been good to excellent, and the poor macroinvertebrate scores may be nothing more than a reflection of poor habitat conditions linked to the low gradient, lacustrine valley through which this reach flows

## 5.5.2 St. Louis River: Summary of Stressors to Aquatic Life

**Table 35:** Summary of SID results for the impaired reach of the St. Louis River

| Candidate Cause | Result |
|---|---|
| **Poor Physical Habitat Conditions** | ● |
| Key: ● = **confirmed stressor** ○ = **Potential Stressor X = eliminated candidate cause** | |

## 5.6 Skunk Creek

Skunk Creek begins in a large peat bog complex southwest of Toivola, Minnesota. A significant portion of this stream and all of its tributaries were ditched in an effort to drain the landscape. Starting downstream of CSAH 195, Skunk Creek takes on the characteristics of an E Channel with a sinuous pattern, narrow channel width, and low-gradient slope (0.08%). The impaired AUID of Skunk Creek is 15% ditched, while 85% of it remains in natural channel form.

Due to the small size of its watershed and the high rate of channelization, only one site was established to evaluate the health of the fish and aquatic macroinvertebrate communities of Skunk Creek. Station 09LS031 is located approximately 0.6 miles upstream of the Skunk Creek – St. Louis River confluence (Figure 103). Station information and sampling results are summarized in Table 36. Fish and macroinvertebrate communities were each sampled once in the summer and fall of 2009.

The fish community of Skunk Creek is highly degraded. Station 09LS031 scored a 0 out of a possible 100 on the FIBI, and both taxa richness and abundance were extremely limited. Less than 20 individuals were collected, and the only species represented in the sample are highly tolerant of poor habitat and water quality (Central Mudminnow, Golden Shiner, and White Sucker). The MIBI scores were more favorable than the fish results, but still failed to meet the MIBI criteria for class 4 stations. The MIBI score of 39.64 was below the impairment threshold (51), but within the lower confidence limit, which is evidence that the macroinvertebrate community is less impacted by stressors in this watershed than the fish community.

**Table 36:** Summary of Skunk Creek biological monitoring stations, monitoring results, and applicable standards

**Fish Assessments**

| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | FIBI Class | FIBI Result *(visit year)* | FIBI Result *(visit year)* | Standard | IBI Lower Confidence Limit | IBI Upper Confidence Limit |
|---|---|---|---|---|---|---|---|---|---|
| 09LS031 | 15.01 | 0.11 | 2 | 6 | **0** *(2009)* | - | 42 | 36 | 58 |

**Macroinvertebrate Assessments**

| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | Invert IBI Class | Invert IBI Result *(visit year)* | Invert IBI Result *(visit year)* | Standard | IBI Lower Confidence Limit | IBI Upper Confidence Limit |
|---|---|---|---|---|---|---|---|---|---|
| 09LS031 | 15.01 | 0.11 | 2 | 4 | **39.64** *(2009)* | - | 51 | 37.40 | 64.60 |

Water chemistry and physical habitat data for the Skunk Creek Watershed were used to develop a list of candidate causes for impairment. The candidate causes that will be evaluated in this section include:

1. **Low DO**
2. **Poor physical habitat conditions**

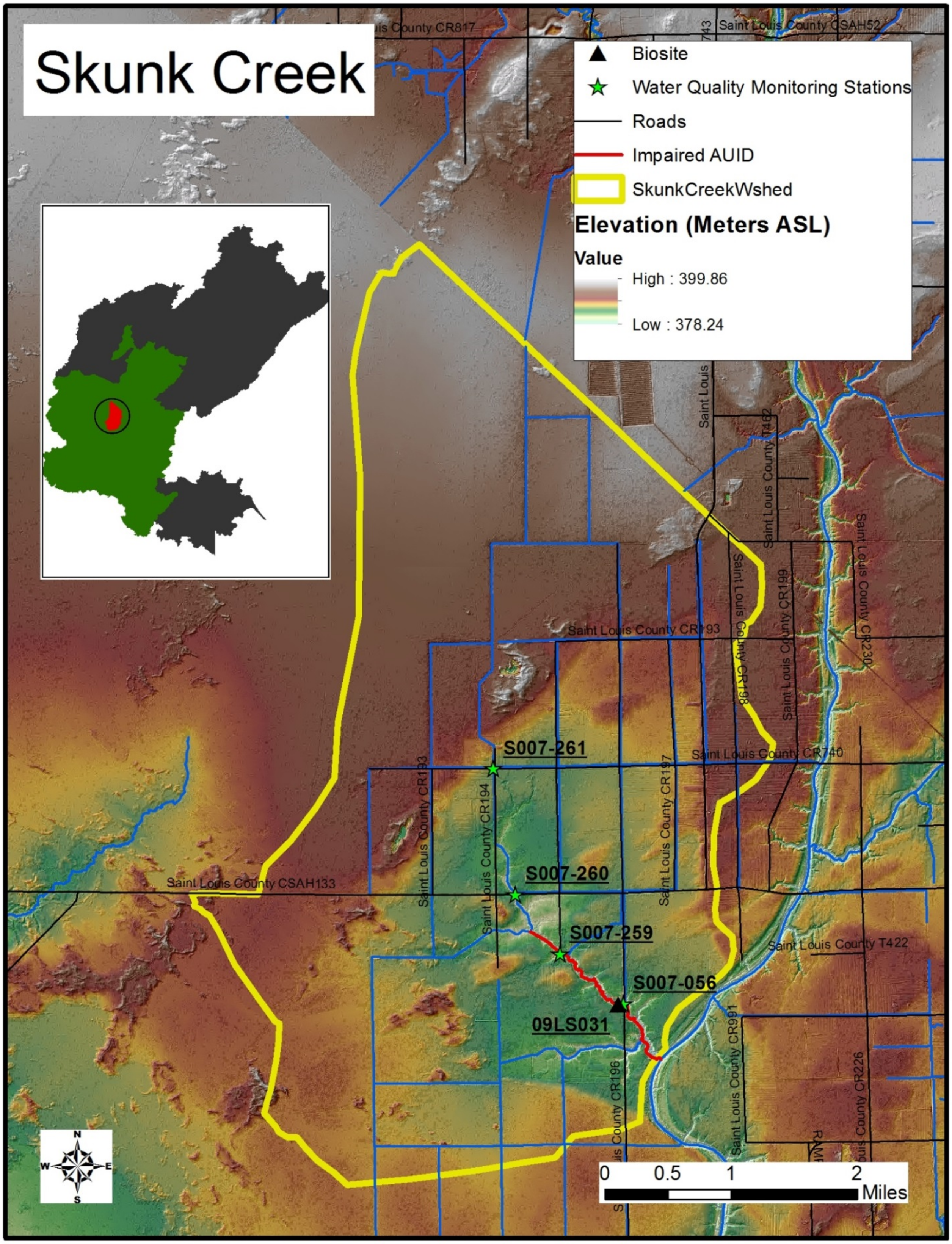


**Figure 103:** Map of Skunk Creek Watershed, impaired stream segments, and monitoring locations.

### 5.6.1 Low Dissolved Oxygen

Low DO concentrations were identified as a candidate cause for fish and MIBI impairments in Skunk Creek. Instantaneous measurements collected between 2009 and 2013 indicate that DO concentrations occasionally fall below the 5 mg/L water quality standard applied to warmwater streams. Approximately 28% of DO readings collected that the lone biological monitoring station (09LS031) were below the DO standard (Figure 105). The lowest concentration observed in Skunk Creek (2.41 mg/L) occurred in September of 2012 during an extended dry period when streamflow was stagnant and impounded by a bridge crossing that was above the water surface elevation (Figure 104). Less than three months earlier, a historic flood (estimated as a 500-year flood in some areas of the SLRW) hit the watershed, causing Skunk Creek to spill out onto its floodplain for several weeks (Figure 104). Due to all of the wetlands and bogs in the watershed that were flushed out during this precipitation event, DO concentrations were fairly low (4.54 mg/L) despite the extremely elevated flow conditions.

**Figure 104:** Skunk Creek at flood stage after major precipitation event in June 2012 (left). At the same station 3 months later, water levels were extremely low, resulting in water surface elevations the below bridge crossing (right).

Short-term continuous measurements of DO were collected at biological monitoring station 09LS031 for approximately one week in July of 2012, and again in late September of 2013. During these continuous monitoring profiles, DO concentrations were sustained just above and below the 5 mg/L DO standard (Figure 106). No sub 5-mg/L DO concentration were observed during the July 2012 profile, but nearly 97% of the measurements collected during the 2013 profile were below 5 mg/L. Both of these profiles show very low diurnal fluctuation in DO concentration (average of less than 1 mg/L) due the lack biological productivity in this stream. Skunk creek contains heavily tannin-stained water and silt/clay substrate, which limits the growth of aquatic vegetation.

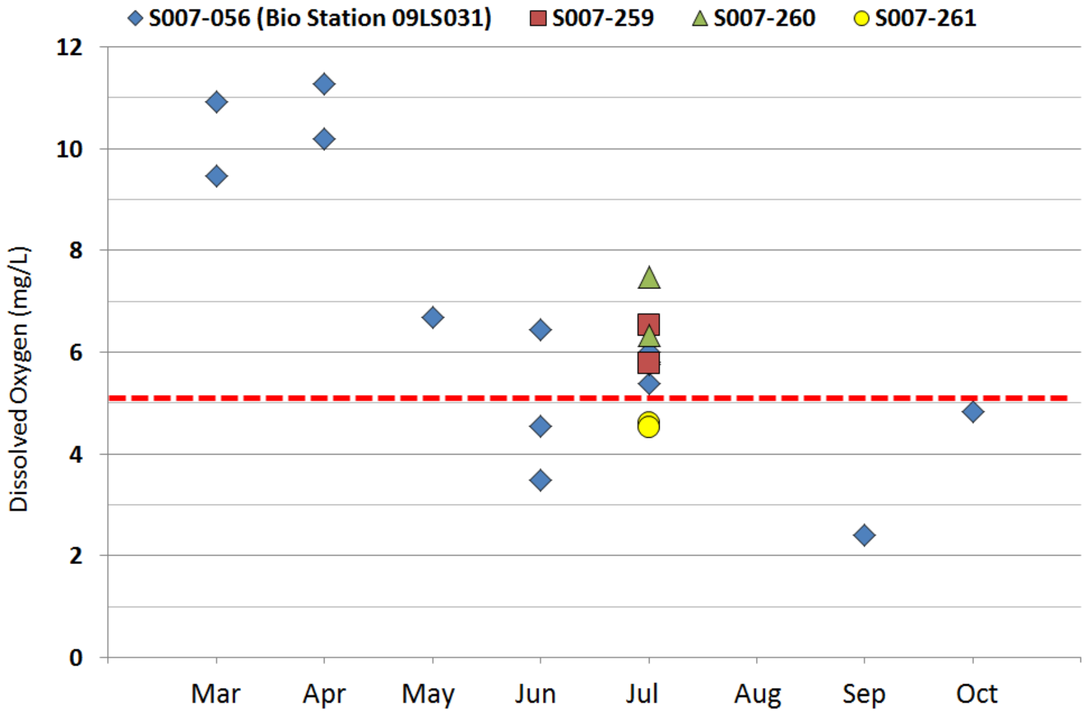
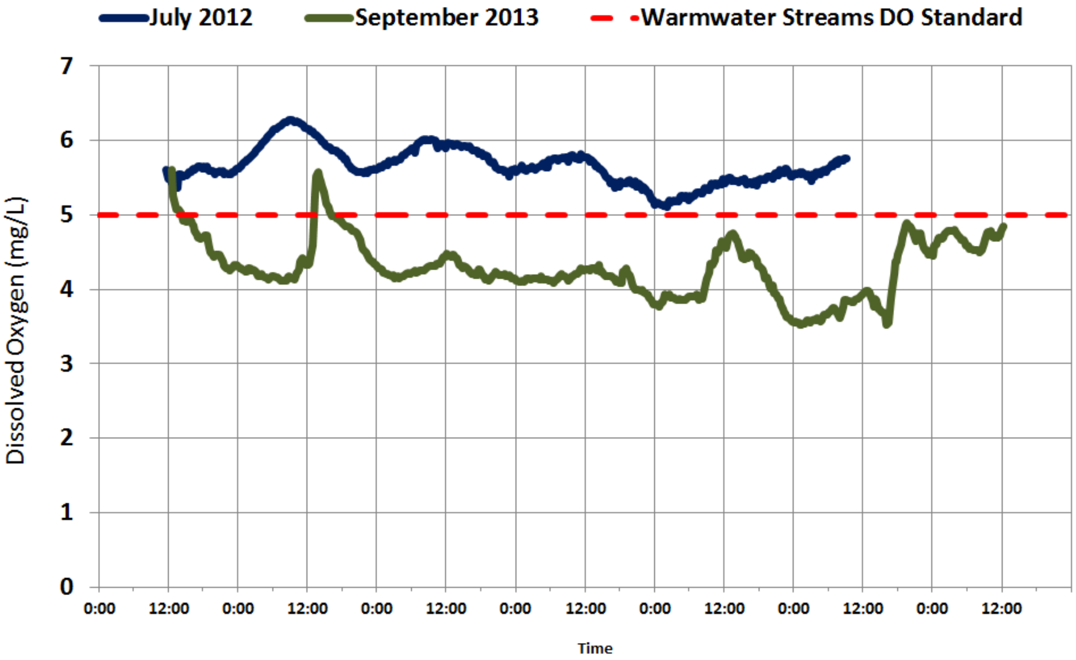


**Figure 105:** Point measurements of DO collected at four locations on Skunk Creek, arranged by calendar month.



| Monitoring Period | # of Readings | Min (mg/L) | Max (mg/L) | Avg. Diurnal Flux (mg/L) | Max Diurnal Flux (mg/L) | % Readings below 5 mg/L | Avg. Duration below 5 mg/L | Max Duration below 5 mg/L |
|---|---|---|---|---|---|---|---|---|
| 7/13/12 – 7/18/12 | 471 | 5.1 | 6.27 | 0.59 | 0.71 | 0% | 0 | 0 |
| 9/27/13 – 10/3/13 | 576 | 3.78 | 5.58 | 0.98 | 1.47 | 97% | 70.5 | 116.25 |

**Figure 106:** Results and summary statistics of two continuous DO monitoring surveys

## Sources and Pathways of Low Dissolved Oxygen

A combination of wetland processes, ditching, and geomorphic features are likely playing a role in the low DO concentrations observed in Skunk Creek. Approximately 67% of the 15 square mile Skunk Creek watershed is classified as woody or emergent herbaceous wetland. Much of the area not classified as wetlands has been converted to livestock pastures or hay fields (12% of the total watershed area). The headwaters of the creek form in a large peat bog complex, the edges of which have been extensively ditched and channelized. Essentially every tributary to Skunk Creek originates from a bog or wetland complex and is channelized for its entire length until meeting the main stem (Figure 103). The intensity of ditching in this watershed is likely resulting in an altered flow regime (see Section 3.2.1), poor physical habitat conditions, and water quality issues such as low DO. The connection between these natural background conditions and anthropomorphic changes to low DO concentrations in Skunk Creek are summarized in Table 37.

**Table 37:** Summary of contributing factors (both natural and anthropogenic) contributing to low DO concentrations in Skunk Creek.

| Source/Pathway | Root Cause(s) | Symptom |
|---|---|---|
| **Flashy Hydrology** | Significant ditching of tributaries and main stem of Skunk Ck. | Very low streamflow conditions and stagnant water are common in summer and fall when low DO conditions are most evident. |
| **Wetlands** | Approximately 67% of watershed area is wetlands. Headwaters of creek is a very large peat bog complex. | Tannin stained water reduces biological productivity. Organics from wetland/bogs in watershed increase BOD, which was fairly high at the biological monitoring station in Skunk Creek (4.5 mg/L) |
| **Geomorphology** | Low gradient stream with no roughness elements (cobble/boulders) | Complete lack of riffles results in no aeration of surface water from the atmosphere |
| **Anthropogenic** | Several livestock pastures within riparian corridor | Possible contributor to elevated BOD concentrations. TP is also elevated (0.056 – 0.123 mg/L) nutrients, but limited primary productivity is occurring in the stream due to tannin stain. |

## Biological Response

Fish abundance and taxa richness measures were very low at station 09LS031. Only three fish species were observed; Central Mudminnow, Golden Shiner, and White Sucker. Each of these species can tolerate a wide range of DO conditions and are commonly observed in streams with low oxygen levels. A total of only 17 individual fish were collected during the electrofishing pass at this station. The low number of fish captured, lack of species richness, and lack of DO sensitive taxa all provide supporting evidence for low DO as a stressor in this stream.

Over 75% of the macroinvertebrate individuals sampled from station 09LS031 belong to taxa that are considered tolerant of low DO concentrations. EPT taxa accounted for only 4.5% of all the taxa observed. The sample was dominated by *Hyallela* (freshwater amphipod), which frequently are found in large populations in streams with low DO concentrations.

Community based TIVs for 09LS031 provide further evidence in support of low DO as a stressor. The DO TIV values for both the fish and macroinvertebrate population were lower than the majority of high quality stations from comparable IBI classes (Figures 107 and 108). These results confirm that Skunk Creek primarily supports fish and aquatic macroinvertebrate taxa that are well adapted to low oxygen levels.

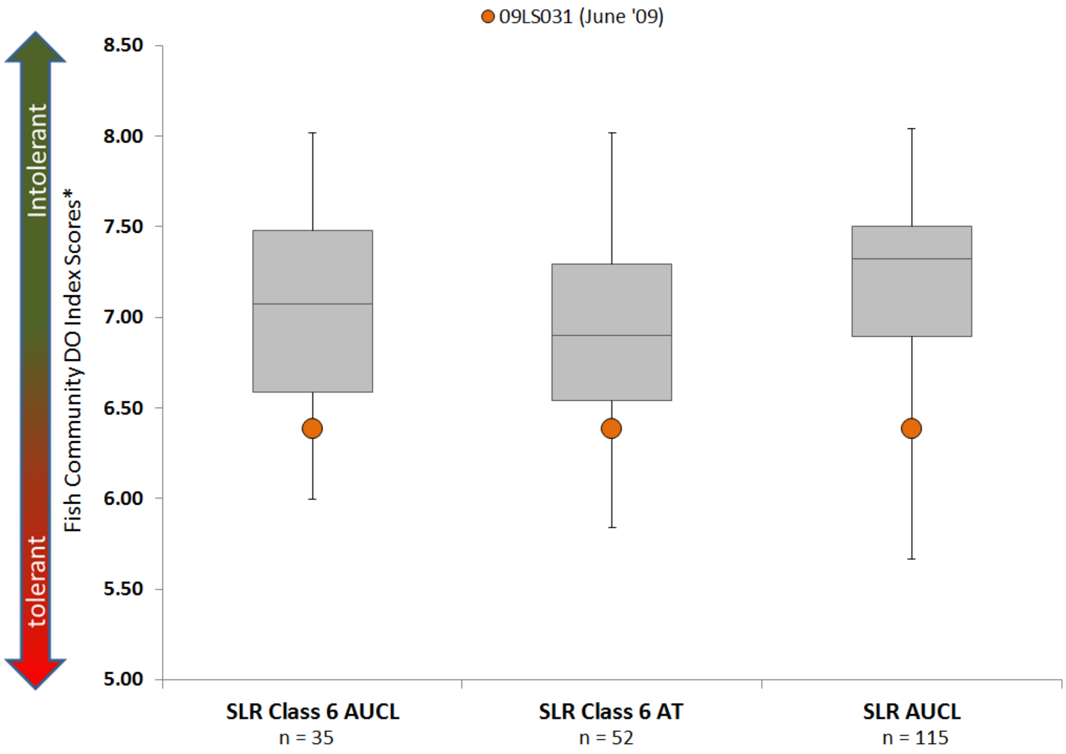


**Figure 107:** Fish community DO TIV results for Skunk Creek station 09LS031 compared to results from high quality stations of the same IBI class. * See Section 4 for explanation of TIVs. **SLR**=St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above FIBI Threshold

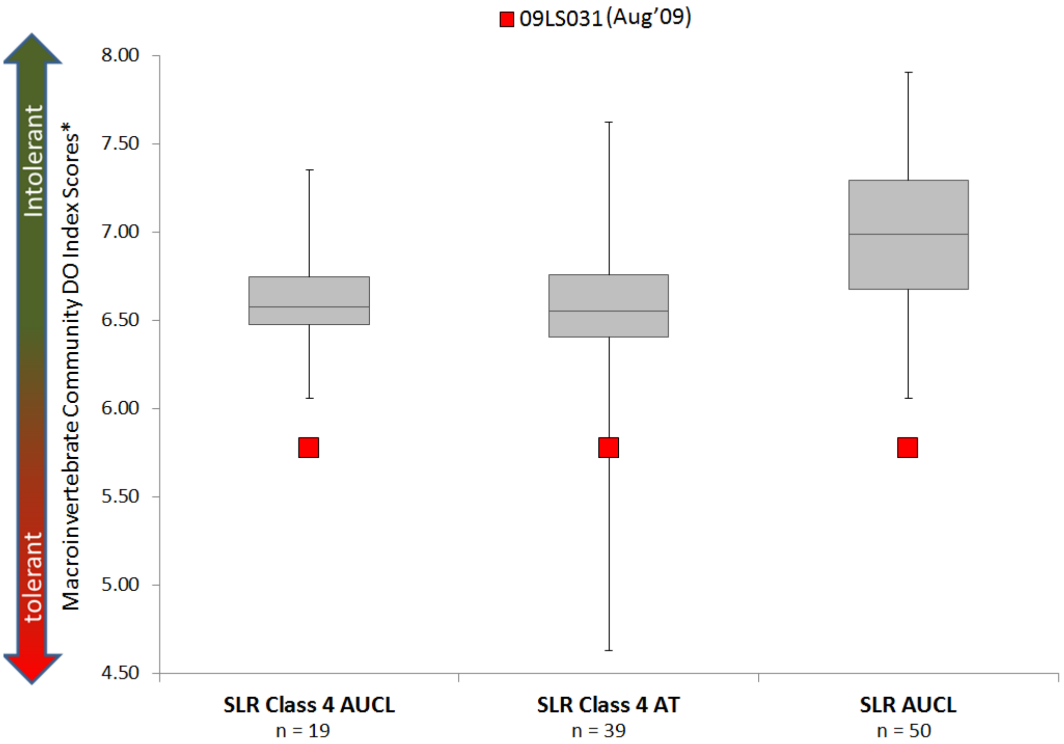


**Figure 108:** Invertebrate community DO TIV results for Skunk Creek station 09LS031 compared to results from high quality stations of the same IBI class. * See Section 4 for explanation of TIVs. **SLR**=St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above FIBI Threshold

**Summary: Is low dissolved oxygen a stressor in Skunk Creek?**

Based on evidence provided by the water quality and biological data, low DO should be considered a cause of biological impairment in Skunk Creek.

## 5.6.2 Poor Physical Habitat Conditions

### Sources and pathways of degraded habitat in Skunk Creek

Skunk Creek station 09LS031 scored 42 out of 100 on the MSHA, which is considered a “poor” habitat score. The land use and cover categories scored fairly well due to the healthy riparian forest adjacent to the stream and the moderate amount of overhead cover in the stream. However, due to the instability of the stream channel and presence of bank erosion in this reach, the riparian, substrate, and channel morphology sections scored very poorly. Substrate scored 7 out of a potential 28 due to the dominance of clay and silt and the complete absence of coarser substrates. Diverse habitat features and channel facets were lacking at this station. Based on MSHA results, the reach was 25% “pool” and 75% “run”, with no riffle or glide sections. As a result, the channel morphology score for station 09LS031 was a dismal 10 out of a possible 35 points.

The PSI scores for Skunk Creek correspond to a rating of “moderately unstable” for the E5 channel type. Characteristics such as intermittent bank erosion, scouring and fine particle deposition are representative of a slightly downcut channel undergoing channel evolution. In fact, there is strong evidence of a change in channel stability above and below the bridge at CR 196. The channel upstream of the bridge is a stable E channel and is narrow, deep, heavily vegetated with little erosion (see Figure 109, bottom left). Just downstream of the bridge the channel is wider and bank erosion is common Figure 109, bottom right). The downstream reach was used to develop the Pfankuch Stability rating of “moderately unstable”.

The condition of Skunk Creek downstream of County Road 196 tells the story of a channel that at some point in the past underwent a channel incision event or series of incision events. From Lane’s stream balance equation (Source), this degradation could have been due to an increase in runoff, increase in water slope, decrease in sediment load, or decrease in sediment size. After incision, the stream no longer has connectivity to its floodplain at bankfull flows and starts to erode its banks to recreate a floodplain at its current elevation. The present condition of Skunk Creek indicates that the channel is in the process of widening to recreate a new floodplain. A possible evolution scenario is represented in Figure 110.

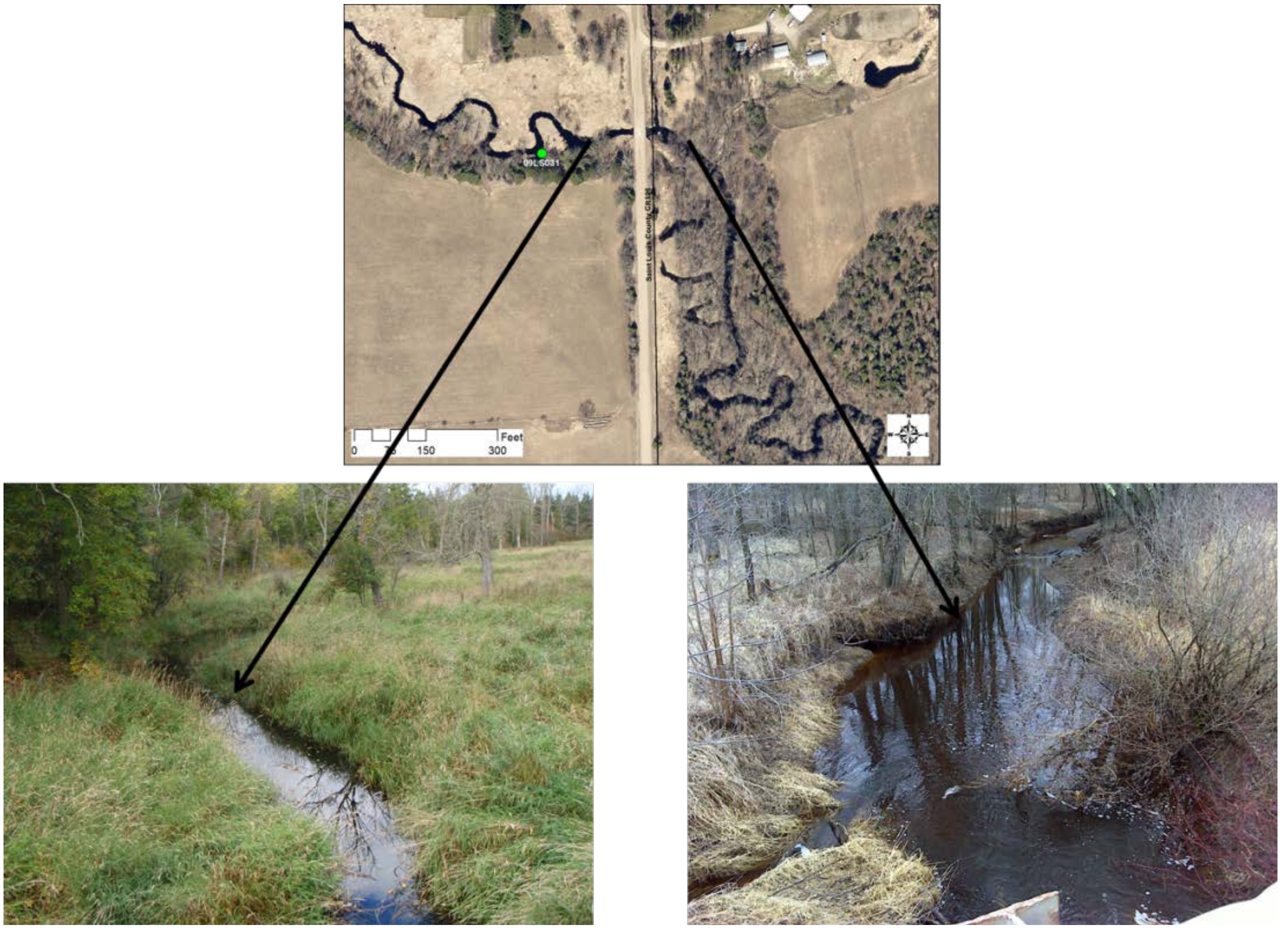


**Figure 109:** Aerial photo of Skunk Creek 09LS031 and CR 196 crossing (top). Upstream of the crossing, the stream is stable (bottom left). Downstream, the stream is widened and eroding (bottom right)

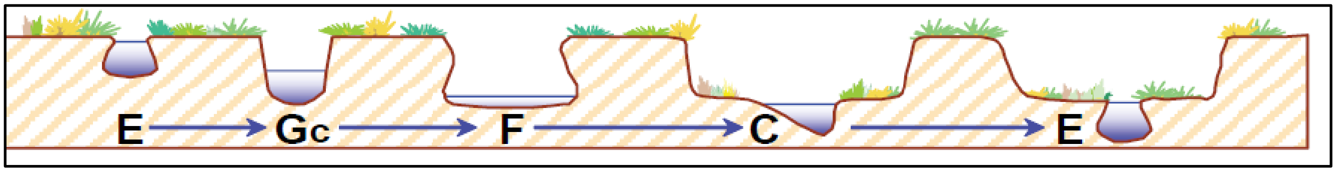


**Figure 110:** Possible Skunk Creek channel evolution scenario (from the River Stability Field Guide, Rosgen 2008)

Further evidence of instability in the downstream reach can be found in the LiDAR data. Figure 111 shows two images of the same reach near County Road 196. The image on the left is a hillshade LiDAR image that shows a high degree of channel incision (downcutting) beginning downstream of the bridge. This is an indication that the channel is degraded, or at least that the elevation difference between the water surface and the stream banks is greater. At right in Figure 111 is an image that combines LiDAR data from 2011 (before the huge June 2012 flood) and fall 2012 (post-flood). Green and yellow colors indicate little to no elevation change, but red indicates degradation from 2011 to 2012.

Worth noting is the fact that the same incision process is not occurring upstream of County Road 196. Usually when a downstream reach cuts down into its bed the channel incision will migrate upstream in the form of a headcut or waterfall. That process has obviously been stopped at CR 196, and one look at the substrate under the bridge provides the answer to why it has been stopped. The coarse cobble pile surrounding the footings of the bridge has provided grade control and prevented the upstream migration of the channel incision. Unfortunately, this means that at base flow the water flows subsurface in the vicinity of the County Road 196 bridge. So, in addition to providing grade control, the cobbles also create a barrier for fish and other aquatic life (see lack of water in Figure 112). Due to the high crest elevation of the cobbles, the water surface profile upstream of the bridge is artificially high and flat during low flows, which is likely eliminating shallow, fast-water habitats (riffles/glides) and create more slack water habitats that dominate this reach (runs/pools).

As a result of channel adjustments there is a lot of sediment deposition and a constant shift in the bed of the stream, which is detrimental to in-stream habitat. Of concern in the Skunk Creek system is the unknown period of time it will take for the stream to adjust and return to a stable state. The low gradient of the stream and relative cohesiveness of the bank material may hinder channel evolution and it may take decades or longer for Skunk Creek to stabilize naturally. In the meantime, like many of the other impaired streams in the area, it is likely that channel instability and excess fine particle deposition are causing habitat degradation in Skunk Creek. Degraded habitat in Skunk Creek will manifest in the fish community if indeed those conditions exist.

The PSI scores for Skunk Creek are representative of slightly downcut channel undergoing channel evolution. Intermittent bank erosion and fine particle deposition led Skunk Creek to a score of 76, which corresponds to a rating of “moderately unstable” for the E5 channel type. Like many of the other impaired streams in the area, it is likely that channel instability and excess fine particle deposition are causing habitat degradation in Skunk Creek. Degraded habitat in Skunk Creek will manifest in the fish community if indeed those conditions exist.

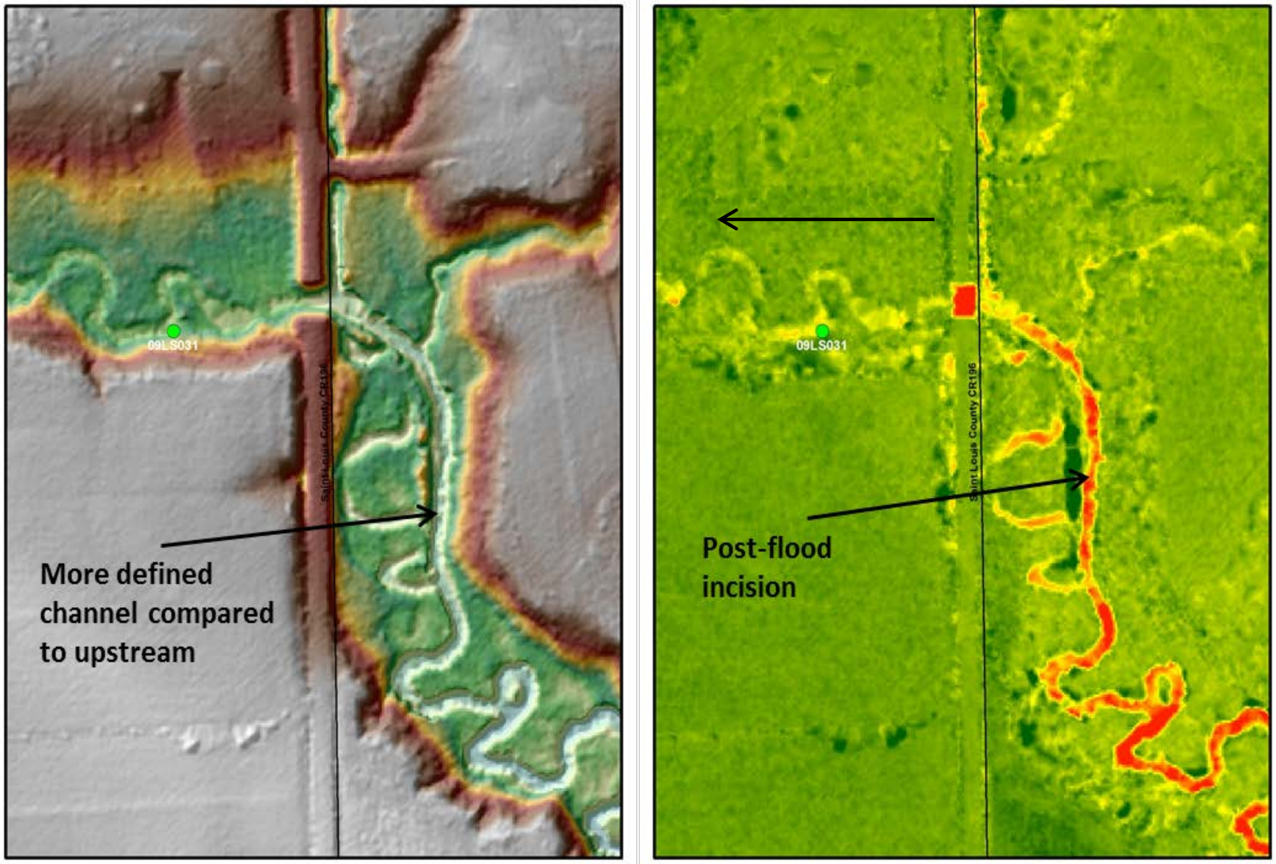


**Figure 111: Two LiDAR images of Skunk Creek 09LS031, one showing an incised channel, the other showing evidence of degradation following the June 2012 flood (scale: green →yellow →red = little change→moderate change→ severe change).**

**Figure 112 (Left): Looking upstream from CR 196 bridge. The cobbles added during bridge construction were set at too high of an elevation. Water flows subsurface through cobbles, and the improper crest elevation of the cobbles is creating a flat water surface upstream. *(Right):* The coarse cobble substrate and lack of water underneath the CR 196 bridge.**

## Biological effects of degraded habitat

The fish community in Skunk Creek is comprised of taxa that are considered tolerant of or neutral to degraded physical habitat conditions. A total of only three fish species were observed during sampling (Central Mudminnow, White Sucker, and Golden Shiner), all of which can be considered habitat generalists with the ability to thrive in streams with marginal and/or degraded physical habitat. The lack of taxa richness and low overall fish count (only 17 total fish were sampled at this site) also support physical habitat as a stressor for this stream reach.

Fish IBI metric results from station 09LS031 provide further evidence of physical habitat limitations in Skunk Creek due to the lack of habitat sensitive fish taxa. Benthic insectivores and darter species were completely absent from Skunk Creek. By comparison, the vast majority of class 6 FIBI stations scoring above the impairment threshold supported one or several fish species which qualify for these metrics. Piscivorous fish species were also absent from Skunk Creek, possibly due to the lack of pools or other forms of cover that these fish depend on. In addition, the number of riffle-dwelling and gravel-spawning fish taxa were also limited in Skunk Creek compared to high quality streams of the same FIBI class. Overall, the fish data from this station provide evidence in support of poor habitat quality as a stressor in Skunk Creek.

The macroinvertebrate assemblage of Skunk Creek is also representative a habitat-limited environment. *Hyalella* (freshwater amphipod crustacean) was far and away the most abundant taxa, accounting for over half of the individuals sampled (58%). Also common at this site were bloodworm midges (*Endochironomus*), mollusks (*Ferrissia*) and aquatic worms (Oligochaeta). These taxon are considered “sprawlers”, which means they sit atop of the substrate (often fine sediment) in habitats with little to no current. Sprawler macroinvertebrates accounted for 60% of the total macroinvertebrate community sampled at station 09LS031. By comparison, the average result observed at other class 4 MIBI stations scoring above the IBI impairment threshold is around 19%. A community dominated by sprawlers as observed in Skunk Creek is often a good indication of poor habitat conditions, and provides evidence in support of poor habitat as a cause of MIBI impairment.

#### Summary: Are poor physical habitat conditions a stressor in Skunk Creek?

There is adequate evidence for diagnosing poor physical habitat conditions a contributing cause of biological impairments in Skunk Creek. Degraded habitat in this stream can be attributed to several factors; (1) channel incision and resulting sedimentation, (2) poor substrate conditions due to local geology and low gradient nature of the stream channel, and (3) poorly installed grade control structure at County Road 196 bridge that is causing slack water effect during periods of lower streamflow.

The substrate and channel morphology sections of the MSHA score, as well as the “moderately unstable” Pfankuch Stability rating support claim that habitat is a limiting factor. This degraded habitat is contributing to a decrease in the numbers of non-tolerant fish/meter and the low numbers of various habitat-sensitive fish and macroinvertebrate taxa.

### 5.6.3 Skunk Creek: Summary of Stressors to Aquatic Life

**Table 38:** Summary of SID results for Skunk Creek

| Candidate Cause | Result |
|---|---|
| **Low Dissolved Oxygen** | ● |
| **Poor Physical Habitat Conditions** | ● |
| **Altered Hydrology** | ○ |
| Key: ● **= confirmed stressor** ○ **= Potential Stressor X = eliminated candidate cause** | |

## 5.7 Vaara Creek

The headwaters of Vaara Creek lie in the bog country between Highway 73 and Toivola, Minnesota. There is almost no variability in the stream and valley types in the Vaara Creek system. A GIS-based analysis showed that the entire creek – from its headwaters to the Floodwood River – is a low-gradient (0.04%), sinuous E channel that flows slowly through a wide wetland-dominated lacustrine valley. This is expected since the watershed lies entirely within the historic Glacial Lake Upham basin, which is very flat and dominated by bogs and other wetland types.

One biological monitoring station was used to evaluate the condition of fish and aquatic macroinvertebrate assemblages in Vaara Creek. Located approximately 0.5 river miles upstream of the Vaara Creek – Floodwood River confluence, station 97LS034 originally established and sampled back in 1997. This same site was re-sampled in 2009 and the data from that sampling event were used in the most recent assessment of this stream, while the older data are now considered “unreportable” given that they are beyond the assessment time window of 10 years. FIBI scores were significantly lower in the 2009 sample (FIBI = 0) than the results from 1997 (FIBI = 41). Several notable species that were present in 1997 were not observed in 2009, including Mottled Sculpin, Burbot, and Northern Pike. Only three species were observed in the 2009 sample (Black Bullhead, Central Mudminnow, Johnny Darter), none of which are particularly sensitive to disturbance or poor water quality.

Unlike the fisheries data, the MIBI results changed very little between the 1997 and 2009 sampling events. Both results are narrowly below the MIBI standard for class 4 streams of 51 (Table 39). Prominent macroinvertebrate taxa observed in the 2009 sample included *Hyallela* (freshwater amphipod), *Coenagrionidae* (narrow-winged damselflies), *Leptophlebiidae* (prong-gilled mayflies), and two taxa of air-breathing freshwater snails *Planorbidae* and *Ferrissia.* Between 40-65% of the taxa observed during the two sampling events are considered “tolerant”, and the percentage of intolerant taxa present during both samples was extremely low (2-5%). The relatively poor overall MIBI scores observed at this station are due to low taxa richness of collector-filterer macroinvertebrates and low non-Hydropsychid caddisfly abundance and taxa richness.

**Table 39:** Summary of biological monitoring stations, biological sampling results, and applicable standards

**Fish Assessments**

| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | FIBI Class | FIBI Result *(visit year)* | FIBI Result *(visit year)* | Standard | IBI Lower Confidence Limit | IBI Upper Confidence Limit |
|---|---|---|---|---|---|---|---|---|---|
| 97LS034 | 27.77 | 0.05 | 2 | 7 | **41** *(1997)* | **0** *(2009)* | 42 | 32 | 52 |

**Macroinvertebrate Assessments**

| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | Invert IBI Class | Invert IBI Result *(visit year)* | Invert IBI Result *(visit year)* | Standard | IBI Lower Confidence Limit | IBI Upper Confidence Limit |
|---|---|---|---|---|---|---|---|---|---|
| 97LS034 | 27.77 | 0.05 | 2 | 4 | **45.45** *(1997)* | **48.06** *(2009)* | 51 | 37.40 | 64.60 |

A review of available water chemistry, biological, physical habitat, and land-cover data were evaluated to develop a list of candidate causes for the FIBI and MIBI impairments in Vaara Creek. The following candidate causes were selected for further analysis in this section:

1. **Low DO**
2. **Poor physical habitat conditions**

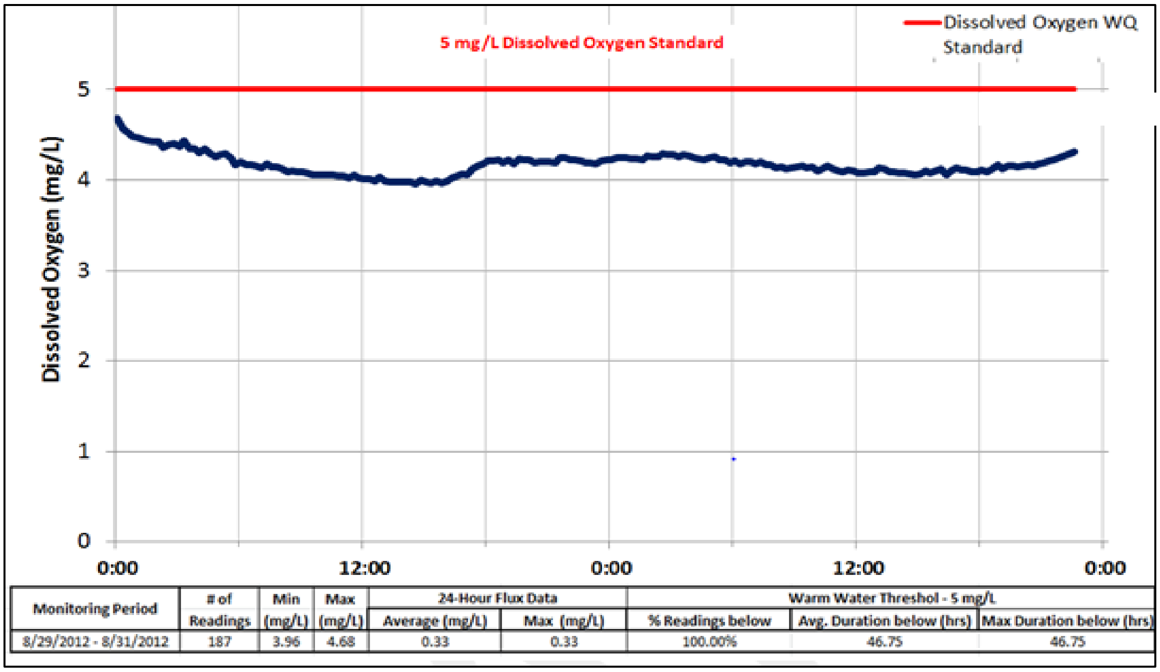
### 5.7.1 Low Dissolved Oxygen

Limited instantaneous DO measurements are available from two stations on Vaara Creek. Station 97LS034 (biological monitoring station) was sampled a total of seven times, with the majority of the DO readings collected in July and August. Summer baseflow DO concentrations at this station hovered near the warmwater DO standard of 5 mg/L, with most of the readings falling just under 5 mg/L. The other monitoring station with DO data, S007-263, is located approximately 0.4 miles downstream of the biological monitoring station near the confluence with the Floodwood River. The area around S007-263 appears to flood frequently and may be influenced by backwater from the Floodwood River. DO data at this site are limited, but this reach appears to have DO concentrations similar to those observed at the biological monitoring station.

A longitudinal DO profile was conducted at three monitoring stations on Vaara Creek in July of 2012. The sample size was limited to three stations due to the lack of road crossings in the watershed. Still, a pattern of lower DO concentrations (below 3 mg/L) in the headwaters increasing to around 4.5 mg/L downstream was evident. The difference between morning and afternoon measurements was negligible, which is another indicator that diurnal DO flux is extremely low in this stream.

A single continuous DO monitoring period on Vaara Creek was initiated in late August of 2012. The 46.75-hour monitoring period was shorter than the typical DO profiles collected throughout the SLRW as part of the SID study. DO concentrations ranged from a minimum of 3.96 mg/L to a maximum of 4.68 mg/L, remaining below the 5 mg/L warmwater standard for the entire monitoring period (Figure 113). DO flux was extremely low (0.33 mg/L), which is similar to other impaired streams in the region with wetland dominated land-cover and heavily bog-stained water. Morning and evening DO measurements were also taken on July 20, 2012 (9:12 am and 3:00 pm) to observe DO flux over that time period. A DO flux of 0.20 mg/L was observed between those two sampling points, and DO concentrations were in the range of 4.35 – 4.55 mg/L.

The available DO data shows that Vaara Creek routinely fails to meet the warmwater DO standard of 5 mg/L. Although data are somewhat limited for this stream, the DO concentrations observed are consistently below 5 mg/L during summer low flow periods



| Monitoring Period | # of Readings | Min (mg/L) | Max (mg/L) | 24-Hour Flux Data | | Warm Water Threshol - 5 mg/L | | |
|---|---|---|---|---|---|---|---|---|
| | | | | Average (mg/L) | Max (mg/L) | % Readings below | Avg. Duration below (hrs) | Max Duration below (hrs) |
| 8/29/2012 - 8/31/2012 | 187 | 3.96 | 4.68 | 0.33 | 0.33 | 100.00% | 46.75 | 46.75 |

**Figure 113:** Results from continuous monitoring at biological monitoring station 97LS034 over a 2-day period in August 2012

## Sources and Pathways Contributing to Low Dissolved Oxygen

### Wetlands

Nearly 80% of the land within the Vaara Creek Watershed is classified as wetland. The main stem of Vaara Creek, and its tributary streams emerge from an expansive region of wetlands that are underlain by mostly peat, all hydric soils (Figure 114). Hydric soils, by definition, are permanently or seasonally saturated by water, resulting in anaerobic conditions. These wetland areas are likely delivering water with depleted oxygen levels to Vaara Creek throughout its length.

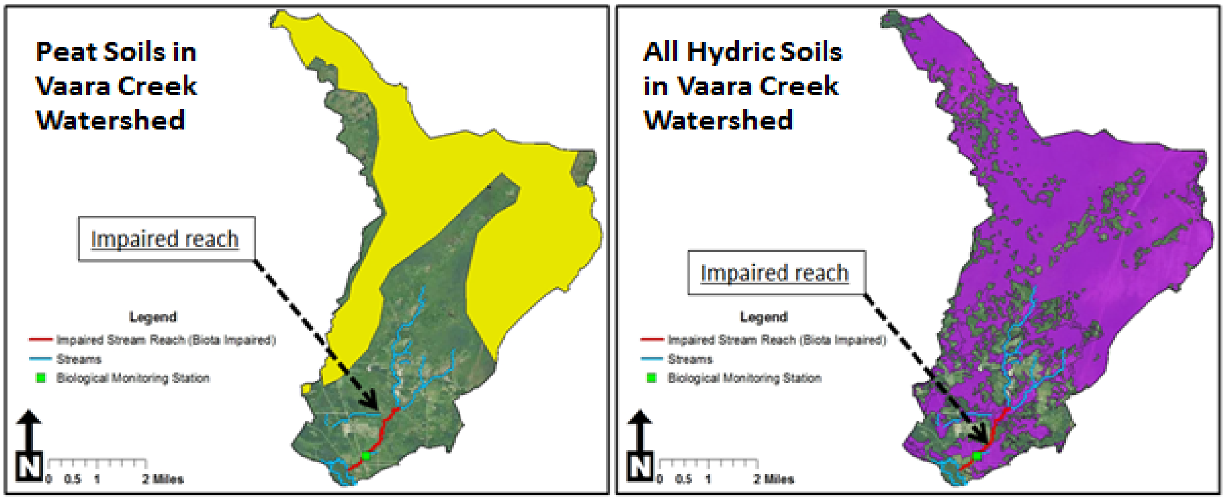


**Figure 114:** Coverage of peat soils (left) and all hydric soils (right) in Vaara Creek Watershed

### Nutrients and Productivity

The TP data is limited for Vaara Creek, but TP concentrations are slightly elevated, and may regularly exceed river nutrient criteria targets for Northern Minnesota. The three results, all from low flow mid-summer conditions, range from 0.032 mg/L to 0.090 mg/L. There are several agricultural fields and small feedlots in the Vaara Creek Watershed that may be contributing to phosphorous loading in the watershed, but natural sources of phosphorous (e.g. internal loading from wetland processes) are likely a significant source as well.

The river nutrient criteria in development for streams and rivers of Minnesota uses TP as the primary nutrient variable, as well as a series of response variables (DO flux, biological oxygen demand, chlorophyll-a) that relate specifically to the stressor caused by elevated nutrient concentrations. The only paired data set for TP and response variables are from samples that were taken during the 48-hr continuous monitoring period in August of 2012 (Table 40). TP concentration was below the nutrient criteria of 0.055 mg/L, and DO flux (0.33 mg/L) was well below 4.0 mg/L.

**Table 40:** Total phosphorous and eutrophication response variable data for Vaara Creek compared to River Nutrient Criteria standards.

| | **Total Phosphorous (TP) (mg/L)** | **24-hr DO Flux (mg/L)** | **BOD(mg/L)** | **Chlorophyll-a (µg/L)** |
|---|---|---|---|---|
| **Vaara Creek @ 97LS034 (08/29/2012)** | 0.032 | 0.33 | 1.7 | No Data |
| **Draft River Nutrient Criteria (Northern MN)** | 0.055 | ≤ 4.0 | ≤ 1.5 | < 10 |

BOD narrowly exceeded the nutrient criteria guideline, but the higher BOD values are likely due to the breaking down of organic compounds produced in the peat bogs upstream, and not primary production in the water column. Chlorophyll-a data was not collected during the continuous monitoring period or at any other time in the SID study. However, Chl-a values are expected to well within river nutrient criteria due to the tannin stained water of Vaara Creek limiting algae growth. Low DO concentrations in Vaara Creek do not appear to be driven by nutrient enrichment and river eutrophication.

### Biological Response to Low Dissolved Oxygen

Fish sampling was conducted at Vaara Creek station 97LS034 in 1997 and 2009. Results from both sampling events show a fish community dominated by fish species that are considered tolerant of low DO conditions. In 1997, 89% of the total fish sampled were species that can be considered tolerant to low DO, and in 2009, this percentage increased to 100%. Overall fish abundance was very low in the 2009 sample, with only 12 individuals caught. This was a significantly lower catch than the 1997 survey in which 97 individuals were sampled. Of the 97 fish sampled in 1997, 73 (75%) were Central Mudminnow, a species that is widely considered one of the most low-DO tolerant species found in Minnesota.

The macroinvertebrate assemblage in Vaara Creek was also dominated by taxa which can be considered tolerant of low DO concentrations. Taking into consideration both monitoring visits, approximately 80-90% of the macroinvertebrates sampled from Vaara Creek station 97LS034 were moderately tolerant or highly tolerant of low DO levels. Mayflies from the genus *Paraleptophlebia* were the most abundant taxa represented in the 1997 sample by a large margin. These mayflies are part of the family Leptophlebiidae (prong-gilled mayflies), and can be considered moderately tolerant of low DO conditions. *Paraleptophlebia* were absent from the sample collected in 2009, and instead, the amphipod *Hyalella* was dominant. *Hyalella* are tolerant of many stressors, including low DO, are abundant in many other impaired streams in the SLRW with low DO concentrations, including nearby Skunk Creek, which is also impaired based on poor MIBI results.

The fish and macroinvertebrate DO index values for Vaara Creek offer further evidence in support of low DO concentrations as a stressor in this watershed. DO index scores in Vaara Creek are lower than the vast majority of scores recorded from reference streams of the same IBI class (Figure 115 and 116). A lower DO index score indicates a community that is more tolerant of low DO conditions. Other biological data that could support low DO as a stressor in this stream include low overall fish abundance, a high percentage of “legless” macroinvertebrate taxa (aquatic worms, fly larvae, midges), and low EPT taxa percent.

### Summary: Is low dissolved oxygen a stressor in Vaara Creek?

Based on the evidence provided by water chemistry and biological data, low DO can be confidently diagnosed as a contributing stressor to biological impairment in Vaara Creek.

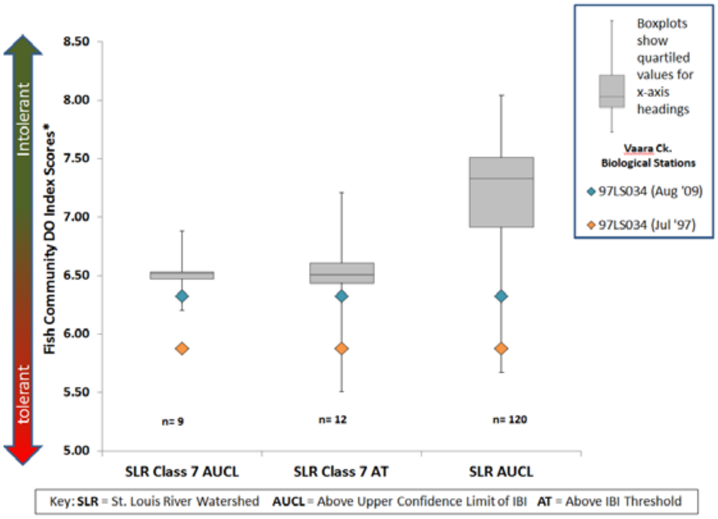


**Figure 115:** Fish community DO TIV results for Vaara Creek compared to high quality reference stations. * See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

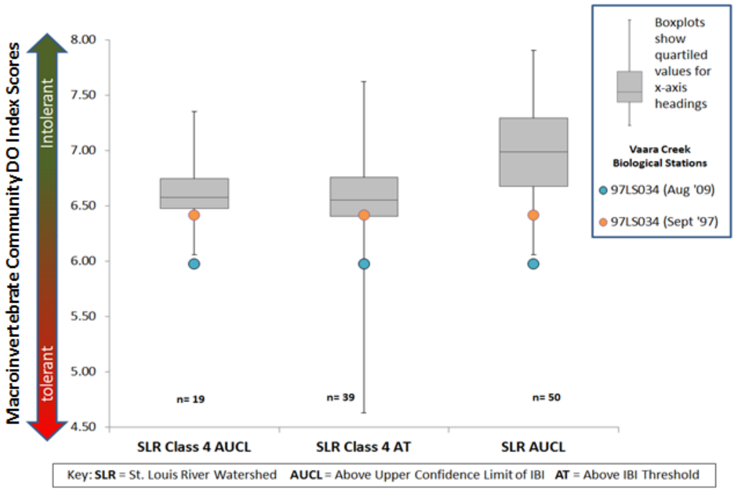


**Figure 116:** Invertebrate community DO TIV results for Vaara Creek compared to high quality reference stations ***** See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

## 5.7.2 Poor Physical Habitat Conditions

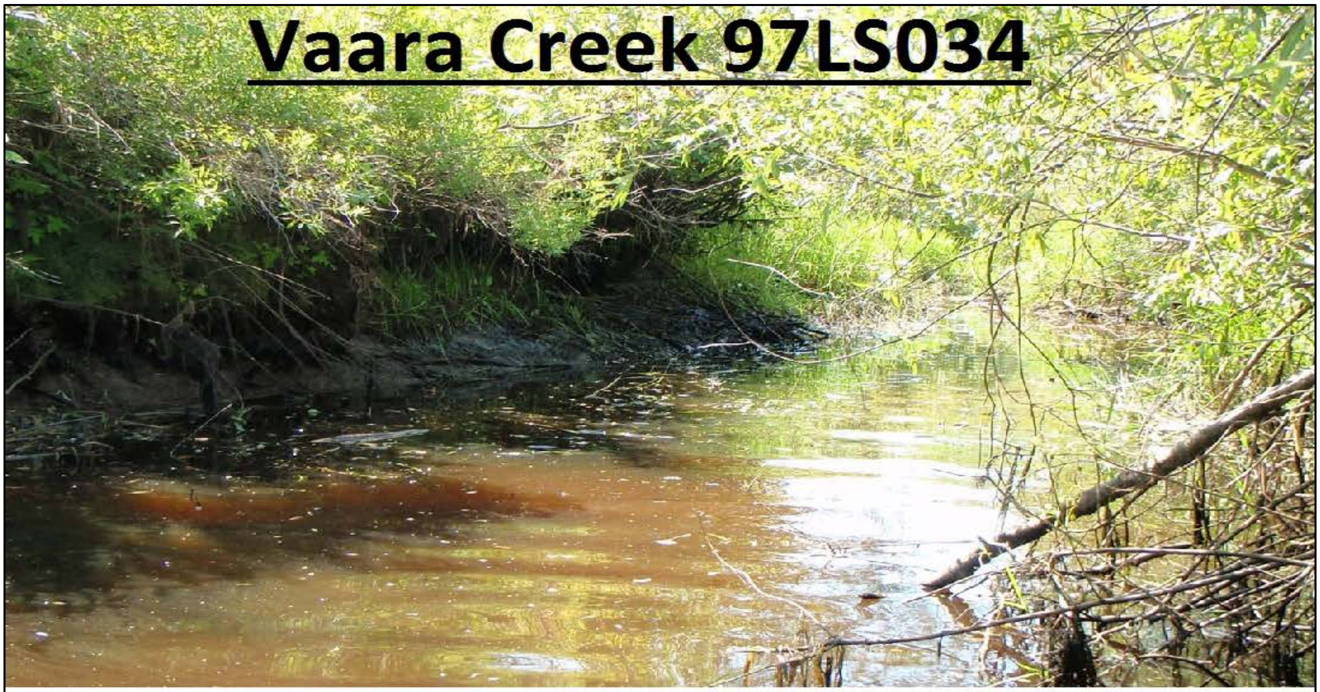
**Sources and pathways of degraded habitat in Vaara Creek**

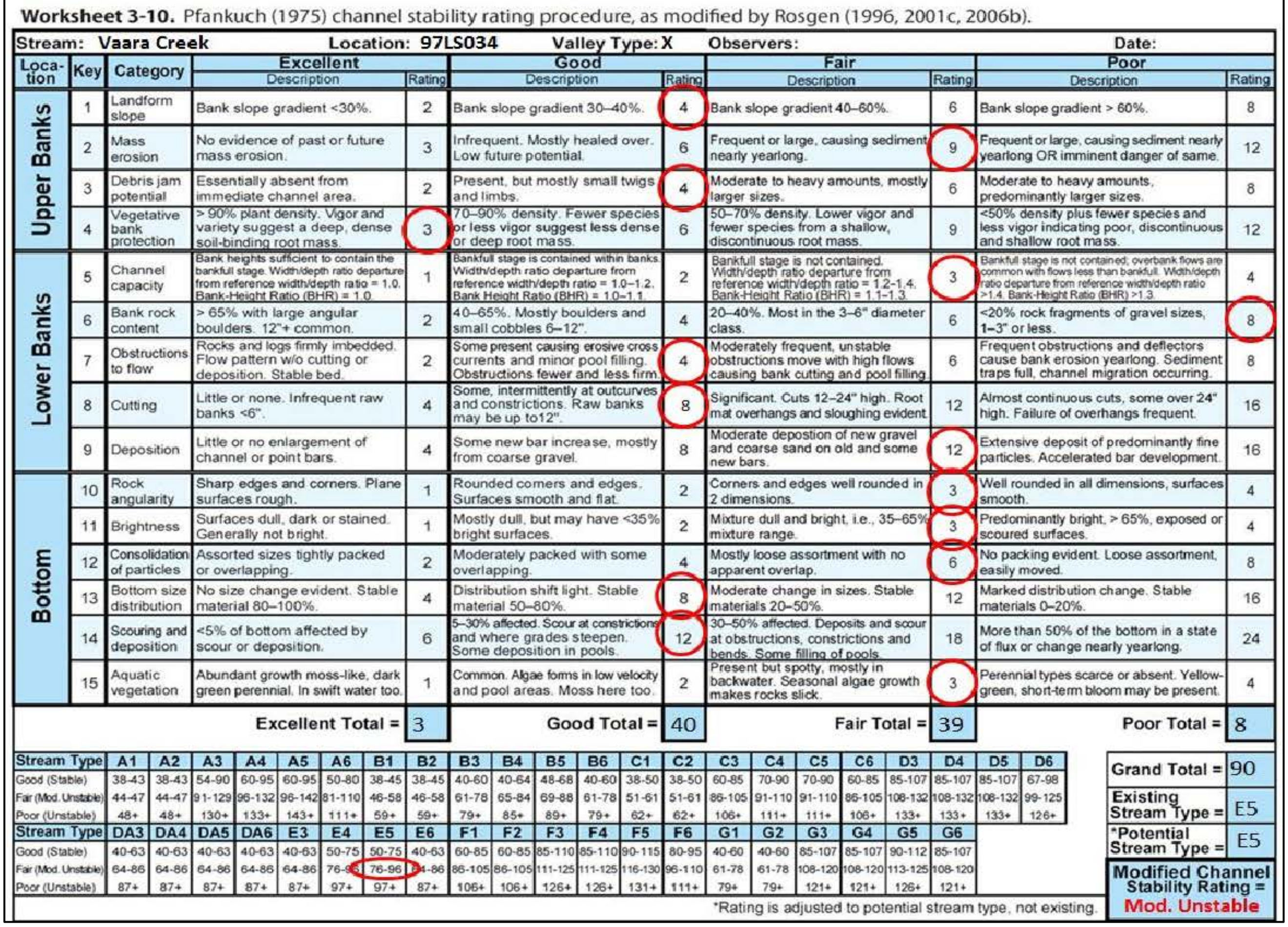
The MSHA result for Vaara Creek was 46 out of 100, which is considered a “fair” score. Excellent scores were achieved in MSHA metrics related to local land use and riparian zone conditions due to the low-impact land use and intact near-stream riparian wetland complex at the biological monitoring station. However, this station scored poorly in the in-stream metrics (substrate and cover) and channel morphology metrics (a total of 28 out of a possible 81). Substrate scored 7 out of a potential 28, due to the dominance of clay and silt and the complete absence of coarser substrates. The stream features were dominated by run (75%) and pool (25%), and lacked shallow/fast water features such as riffles and glides. The lack of suitable cover for fish, particularly piscivorous species, is another habitat limitation in this reach. The stream lacks undercut banks, boulders, rootwads, and overall, over 75% of the reach did not provide adequate cover for fish.

Aquatic vegetation that is suitable for macroinvertebrate habitat and/or fish cover also appears to be severely limited in Vaara Creek. This is likely due to several characteristics of this stream, including; (1) low transparency of the water due to bog staining, (2) peat soil type results in a very compact streambed, and (3) sediment deposition from streambank erosion.

The PSI scores for Vaara Creek are typical of an incised channel, with characteristics such as bank erosion and loose bottom sediments leading to poor habitat conditions. Vaara Creek station 97LS034 scored a 90 on the PSI, which corresponds to a rating of “moderately unstable” for the potential E5 stream type (Figure 118). The channel instability in this reach of Vaara Creek may be due in part to seasonal flow variability. The wide channel and steep, high banks observed in this reach are indicative of stream with highly fluctuating flow conditions. Site visits to this reach have confirmed these conditions, as spring snowmelt conditions fill much of the stream channel and wetted width (width from waters edge to waters edge) during summer baseflow conditions is much narrower than the physical channel width (Figure 117).

**Figure 117:** Low gradient conditions at Vaara Creek biological monitoring station. Note the lack of riffle/glide features




**Worksheet 3-10.** Pfankuch (1975) channel stability rating procedure, as modified by Rosgen (1996, 2001c, 2006b).

**Stream:** Vaara Creek
**Location:** 97LS034
**Valley Type:** X
**Observers:**
**Date:**

| Location | Key | Category | Excellent Description | Excellent Rating | Good Description | Good Rating | Fair Description | Fair Rating | Poor Description | Poor Rating |
|---|---|---|---|---|---|---|---|---|---|---|
| Upper Banks | 1 | Landform slope | Bank slope gradient <30%. | 2 | Bank slope gradient 30–40%. | [✓ 4] | Bank slope gradient 40–60%. | 6 | Bank slope gradient > 60%. | 8 |
| | 2 | Mass erosion | No evidence of past or future mass erosion. | 3 | Infrequent. Mostly healed over. Low future potential. | 6 | Frequent or large, causing sediment nearly yearlong. | [✓ 9] | Frequent or large, causing sediment nearly yearlong OR imminent danger of same. | 12 |
| | 3 | Debris jam potential | Essentially absent from immediate channel area. | 2 | Present, but mostly small twigs and limbs. | [✓ 4] | Moderate to heavy amounts, mostly larger sizes. | 6 | Moderate to heavy amounts, predominantly larger sizes. | 8 |
| | 4 | Vegetative bank protection | > 90% plant density. Vigor and variety suggest a deep, dense soil-binding root mass. | [✓ 3] | 70–90% density. Fewer species or less vigor suggest less dense or deep root mass. | 6 | 50–70% density. Lower vigor and fewer species from a shallow, discontinuous root mass. | 9 | <50% density plus fewer species and less vigor indicating poor, discontinuous and shallow root mass. | 12 |
| Lower Banks | 5 | Channel capacity | Bank heights sufficient to contain the bankfull stage. Width/depth ratio departure from reference width/depth ratio = 1.0. Bank-Height Ratio (BHR) = 1.0. | 1 | Bankfull stage is contained within banks. Width/depth ratio departure from reference width/depth ratio = 1.0–1.2. Bank Height Ratio (BHR) = 1.0–1.1. | 2 | Bankfull stage is not contained. Width/depth ratio departure from reference width/depth ratio = 1.2-1.4. Bank-Height Ratio (BHR) = 1.1–1.3. | [✓ 3] | Bankfull stage is not contained; overbank flows are common with flows less than bankfull. Width/depth ratio departure from reference width/depth ratio >1.4. Bank-Height Ratio (BHR) >1.3. | 4 |
| | 6 | Bank rock content | > 65% with large angular boulders. 12"+ common. | 2 | 40–65%. Mostly boulders and small cobbles 6–12". | 4 | 20–40%. Most in the 3–6" diameter class. | 6 | <20% rock fragments of gravel sizes, 1–3" or less. | [✓ 8] |
| | 7 | Obstructions to flow | Rocks and logs firmly imbedded. Flow pattern w/o cutting or deposition. Stable bed. | 2 | Some present causing erosive cross currents and minor pool filling. Obstructions fewer and less firm. | [✓ 4] | Moderately frequent, unstable obstructions move with high flows causing bank cutting and pool filling. | 6 | Frequent obstructions and deflectors cause bank erosion yearlong. Sediment traps full, channel migration occurring. | 8 |
| | 8 | Cutting | Little or none. Infrequent raw banks <6". | 4 | Some, intermittently at outcurves and constrictions. Raw banks may be up to12". | [✓ 8] | Significant. Cuts 12–24" high. Root mat overhangs and sloughing evident. | 12 | Almost continuous cuts, some over 24" high. Failure of overhangs frequent. | 16 |
| | 9 | Deposition | Little or no enlargement of channel or point bars. | 4 | Some new bar increase, mostly from coarse gravel. | 8 | Moderate depostion of new gravel and coarse sand on old and some new bars. | [✓ 12] | Extensive deposit of predominantly fine particles. Accelerated bar development. | 16 |
| Bottom | 10 | Rock angularity | Sharp edges and corners. Plane surfaces rough. | 1 | Rounded corners and edges. Surfaces smooth and flat. | 2 | Corners and edges well rounded in 2 dimensions. | [✓ 3] | Well rounded in all dimensions, surfaces smooth. | 4 |
| | 11 | Brightness | Surfaces dull, dark or stained. Generally not bright. | 1 | Mostly dull, but may have <35% bright surfaces. | 2 | Mixture dull and bright, i.e., 35–65% mixture range. | [✓ 3] | Predominantly bright, > 65%, exposed or scoured surfaces. | 4 |
| | 12 | Consolidation of particles | Assorted sizes tightly packed or overlapping. | 2 | Moderately packed with some overlapping. | 4 | Mostly loose assortment with no apparent overlap. | [✓ 6] | No packing evident. Loose assortment, easily moved. | 8 |
| | 13 | Bottom size distribution | No size change evident. Stable material 80–100%. | 4 | Distribution shift light. Stable material 50–80%. | [✓ 8] | Moderate change in sizes. Stable materials 20–50%. | 12 | Marked distribution change. Stable materials 0–20%. | 16 |
| | 14 | Scouring and deposition | <5% of bottom affected by scour or deposition. | 6 | 5–30% affected. Scour at constrictions and where grades steepen. Some deposition in pools. | [✓ 12] | 30–50% affected. Deposits and scour at obstructions, constrictions and bends. Some filling of pools. | 18 | More than 50% of the bottom in a state of flux or change nearly yearlong. | 24 |
| | 15 | Aquatic vegetation | Abundant growth moss-like, dark green perennial. In swift water too. | 1 | Common. Algae forms in low velocity and pool areas. Moss here too. | 2 | Present but spotty, mostly in backwater. Seasonal algae growth makes rocks slick. | [✓ 3] | Perennial types scarce or absent. Yellow-green, short-term bloom may be present. | 4 |
| | | | Excellent Total = | 3 | Good Total = | 40 | Fair Total = | 39 | Poor Total = | 8 |

| Stream Type | A1 | A2 | A3 | A4 | A5 | A6 | B1 | B2 | B3 | B4 | B5 | B6 | C1 | C2 | C3 | C4 | C5 | C6 | D3 | D4 | D5 | D6 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Good (Stable) | 38-43 | 38-43 | 54-90 | 60-95 | 60-95 | 50-80 | 38-45 | 38-45 | 40-60 | 40-64 | 48-68 | 40-60 | 38-50 | 38-50 | 60-85 | 70-90 | 70-90 | 60-85 | 85-107 | 85-107 | 85-107 | 67-98 |
| Fair (Mod. Unstable) | 44-47 | 44-47 | 91-129 | 96-132 | 96-142 | 81-110 | 46-58 | 46-58 | 61-78 | 65-84 | 69-88 | 61-78 | 51-61 | 51-61 | 86-105 | 91-110 | 91-110 | 86-105 | 108-132 | 108-132 | 108-132 | 99-125 |
| Poor (Unstable) | 48+ | 48+ | 130+ | 133+ | 143+ | 111+ | 59+ | 59+ | 79+ | 85+ | 89+ | 79+ | 62+ | 62+ | 106+ | 111+ | 111+ | 106+ | 133+ | 133+ | 133+ | 126+ |

| Stream Type | DA3 | DA4 | DA5 | DA6 | E3 | E4 | E5 | E6 | F1 | F2 | F3 | F4 | F5 | F6 | G1 | G2 | G3 | G4 | G5 | G6 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Good (Stable) | 40-63 | 40-63 | 40-63 | 40-63 | 40-63 | 50-75 | 50-75 | 40-63 | 60-85 | 60-85 | 85-110 | 85-110 | 90-115 | 80-95 | 40-60 | 40-60 | 85-107 | 85-107 | 90-112 | 85-107 |
| Fair (Mod. Unstable) | 64-86 | 64-86 | 64-86 | 64-86 | 64-86 | 76-9 | [✓ 76-96] | 4-86 | 86-105 | 86-105 | 111-125 | 111-125 | 116-130 | 96-110 | 61-78 | 61-78 | 108-120 | 108-120 | 113-125 | 108-120 |
| Poor (Unstable) | 87+ | 87+ | 87+ | 87+ | 87+ | 97+ | 97+ | 87+ | 106+ | 106+ | 126+ | 126+ | 131+ | 111+ | 79+ | 79+ | 121+ | 121+ | 126+ | 121+ |

Grand Total = 90
Existing Stream Type = E5
*Potential Stream Type = E5
Modified Channel Stability Rating = Mod. Unstable

*Rating is adjusted to potential stream type, not existing.

**Figure 118:** Pfankuch Stability Index rating for Vaara Creek 97LS034

## Biological effects of degraded habitat

### Fish Response to Habitat

Very low fish counts were observed during the August 2009 sampling visit to station 97LS034. Only 12 individuals, representing three taxa (Black Bullhead, Central Mudminnow, Johnny Darter) were sampled in the 150 meter reach that was sampled via electrofishing methods. The catch-rate per distance sampled was only 0.04 fish per meter (excluding tolerant taxa) during the 2009 visit, which is significantly lower than 75% of the Class 7 streams that were above the IBI UCL and the IBI threshold (Figure 119). The catch rate observed during the 1997 sampling event was higher (0.15 fish/meter, excluding tolerant taxa) but still below 75% of comparable streams that scored above the IBI threshold and UCL. Overall, taxa richness and abundance of non-tolerant fish species is significantly lower in Vaara Creek compared to healthy streams of the same IBI class. This symptom of impairment can be associated with the limited physical habitat available to support abundant and diverse fish populations.

The fish species found in the most recent survey of Vaara Creek can be considered habitat generalists and are generally tolerant of degraded habitat conditions.

The fish community observed at Vaara Creek monitoring station 09LS034 during the 1997 survey contained relatively large numbers of habitat-sensitive taxa, such as Burbot and Mottled Sculpin. However, habitat-sensitive taxa declined across the board in the 12 years between sampling efforts. In 2009 the number of habitat-sensitive taxa in Vaara Creek did not compare favorably to the 13 unimpaired Class 7 streams in the SLRW (Figure 120). One benthic insectivore species – Johnny Darter – was found at this station in 2009. This was equal to or below 75% of the Class 7 streams above the impairment threshold. The same species counts as the one darter/sculpin/sucker taxa found in Vaara Creek, which is the median for the Class 7 SLRW AT stations. No piscivorous, riffle-dwelling or gravel spawning taxa were present –equal to or below 75% of the comparison streams. The general lack of fish taxa with specific habitat requirements supports the diagnosis of poor physical habitat as a stressor in Vaara Creek.

The macroinvertebrate data also show a community shift between 1997 and 2009 samping events. The 1997 sample was dominated by the the mayfly *Paraleptophlebia*, which is a member of the “prong-gilled mayfly” family. Although classified as a “crawler” in terms of its mode of mobility, these mayflies also swim exceptionally well within the water column. Individuals from this mayfly genus accounted for 73% (215 of 296) of the organisms sampled from station 09LS034 in 1997. In contrast, the 2009 sampled was dominated by individuals from the freshwater amphipod genus *Hyalella* (40% of sample) and the Ramshorn Snail, *Planorbidae* (10% of sample). Both of these macroinvertebrate taxa are considered can be considered tolerant of fine substrates and generally poor habitat conditions.

Although the two macroinvertebrate samples were dominated by different taxa, the overall community was fairly similar in terms of the taxa represented in the sample. The MIBI results for the two visits were very similar (45/100 in 1997 and 48/100 in 2009). The biggest differences observed were related to the dominance of certain taxa. In general, the macroinvertebrate community of Vaara Creek lacks EPT taxa richness and abundance, contains a high percentage of “legless” taxa (mostly snails and blackfly larvae), and lacks intolerant taxa. All of these symptoms of impairment can be linked to the poor habitat conditions that exist throughout the impaired reach.

Based on the photos collected during the two sampling visits, it does not appear that physical habitat conditions have changed considerably between the years of 1997 and 2009. Variables other than physical habitat components, such as water temperture and DO, may explain why some of the fish species present in 1999 were not sampled in 2009. Water temperture was higher in 2009 (21.7 C) and DO concentration lower (4.35 mg/L) which may expain the absence of Burbot and Mottled Sculpin in the 2009 sample, as both of these species are sensitive to warmer water tempertures. Still, physical habitat conditions in Vaara Creek are very homogenous and dominated by features that are suitable mainly to generalist fish and macroinvertebrate taxa.

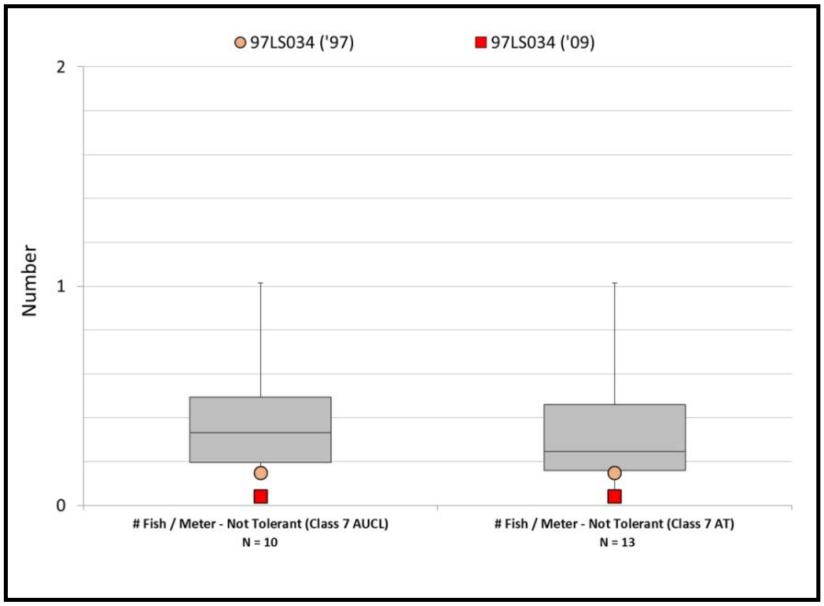


Figure 119: Non-tolerant fish/meter in Vaara Creek compared to high quality Class 7 SLRW stations. * See section 4 for explanation of TIVs AUCL = Above Upper Confidence Limit of FIBI threshold AT = Above Fish IBI Threshold

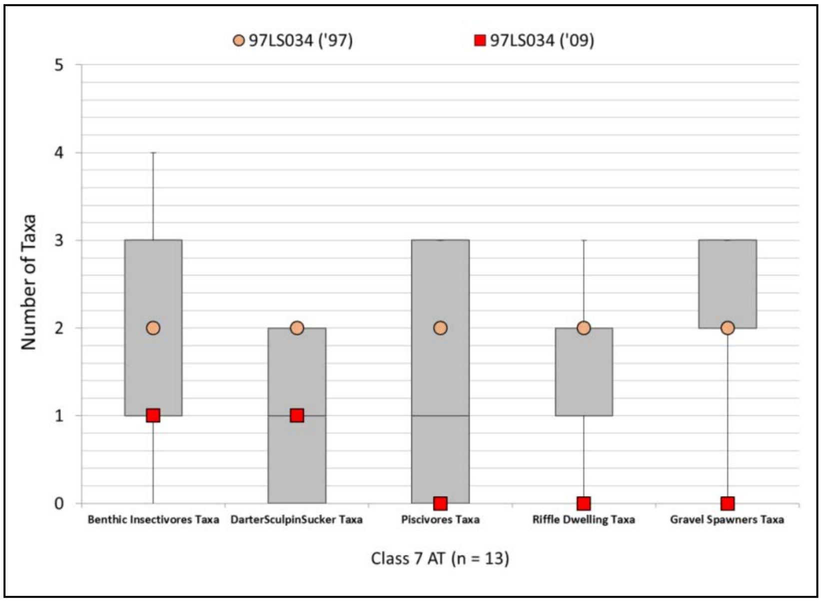


**Figure 120:** Number of fish taxa for various categories in Vaara Creek compared to Class 7 SLRW stations above the impairment threshold. * See section 4 for explanation of TIVs AUCL = Above Upper Confidence Limit of FIBI threshold AT = Above Fish IBI Threshold

**Summary: Is Poor Habitat a Stressor to Aquatic Life in Vaara Creek?**

Degraded habitat in Vaara Creek can partially be attributed to channel instability and sedimentation. The poor substrate and channel morphology sections of the MSHA score and the "moderately unstable" Pfankuch Stability rating support this hypothesis. The degraded habitat is contributing to a decrease in the biological integrity of the stream, as evidenced by the number of non-tolerant fish/meter and the low number of various habitat-sensitive taxa present. It is our conclusion that habitat degradation is a stressor to the fish community in Vaara Creek.

## 5.7.3 Vaara Creek: Summary of Stressors to Aquatic Life

**Table 41:** Summary of SID results for Vaara Creek, along with recommendations for restoration and implementation projects.

| Candidate Cause | Result |
|---|---|
| **Low Dissolved Oxygen** | ● |
| **Poor Physical Habitat Conditions** | ● |
| **Altered Hydrology** | ○ |
| Key: ● **= confirmed stressor** ○ **= Potential Stressor X = eliminated candidate cause** | |

## 5.8 Sand Creek

Sand Creek originates in the vast bog just south of the Swan River Watershed. Much of the beginning reaches of the creek were ditched, presumably in an effort to drain the bog. Eventually the stream flows out of the bog and enters a more shrub-dominated wetland system. This reach is not ditched and still retains its sinuous and narrow E channel characteristics. The upper reaches of Sand Creek flow through very wide and flat lacustrine valleys. The impaired AUID of Sand Creek has little variation in stream type or valley type, and the whole reach is a low-gradient (0.06%) C Channel in an unconfined alluvial (Type 8) valley.

Fish and macroinvertebrate data were collected at two monitoring stations on Sand Creek. An initial monitoring effort occurred in 1998 at station 98LS047, but this site was not resampled during the most recent monitoring work. Instead, a new station was created (09LS033) downstream of 98LS047. The IBI impairment on Sand Creek is based on data collected at station 09LS033 in 2009. The FIBI score for this visit (43) was several points below the impairment threshold, but within the lower confidence limit, which is an indication that the level of impairment at this station is not severe.

**Table 42:** Summary of biological monitoring stations, biological sampling results, and applicable standards

| **Fish Assessments** | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Station** | **Drainage Area (mi2)** | **Gradient (%)** | **Stream Order (Strahler)** | **FIBI Class** | **FIBI Result *(visit year)*** | **FIBI Result *(visit year)*** | **Standard** | **IBI Lower Confidence Limit** | **IBI Upper Confidence Limit** |
| 98LS047 | 59.64 | 0.09 | 2 | 5 | **71** *(1998)* | - | 47 | 38 | 56 |
| 09LS033 | 63.99 | 0.09 | 2 | 5 | **43** *(2009)* | - | 47 | 38 | 56 |

The fish community at 09LS003 consists of 11 taxa with a variety of tolerance levels to degraded habitat and poor water quality. Creek Chub, Johnny Darter, and Bigmouth Shiner were the most abundant fish taxa observed at the impaired site, although the numbers of fish were quite evenly distributed among the 11 taxa represented (Figure 121). Mottled Sculpin were the only “sensitive” taxa observed at this monitoring station, but they were found in much lower numbers compared to station 98LS047 further upstream. Many of the same fish species were observed at the two monitoring stations, although several differences in community structure between the sites are responsible for the disparity in FIBI scores (28 point difference). Several tolerant minnow species were present at the impaired station (09LS033) but absent from 98LS047, such as Fathead Minnow and Bigmouth Shiner. On the contrary, station 98LS047 supported a robust population of Longnose Dace, which is a sensitive minnow species that is intolerant of poor water quality and degraded habitat conditions. Longnose Dace were not sampled at the impaired station.

A review of available water chemistry, biological, physical habitat, and land-use data was completed in order to identify a list of candidate causes for the FIBI impairment observed in Sand Creek. This review resulted in three candidate causes that will be evaluated in this section:

1. **Elevated TSS**
2. **Poor Habitat**

| Station 98LS047 | | Station 09LS033 | |
|---|---|---|---|
| **Shared Species / Count** | | | |
| Burbot | 1 | Burbot | 3 |
| Creek Chub | 1 | Creek Chub | 11 |
| Johnny Darter | 34 | Johnny Darter | 11 |
| Mottled Sculpin | 50 | Mottled Sculpin | 6 |
| Trout-Perch | 4 | Trout-Perch | 7 |
| White Sucker | 4 | White Sucker | 4 |
| **Species unique to Station / Count** | | | |
| Central Mudminnow | 13 | Bigmouth Shiner | 10 |
| Longnose Dace | 63 | Common Shiner | 2 |
| Walleye | 1 | Fathead Minnow | 4 |
| Northern Pike | 1 | Northern Redbelly Dace | 2 |
| | | Golden Shiner | 5 |

98LS047

09LS033

**Figure 121:** Comparison of fish species/abundance observed at the two biological monitoring stations on Sand Creek.

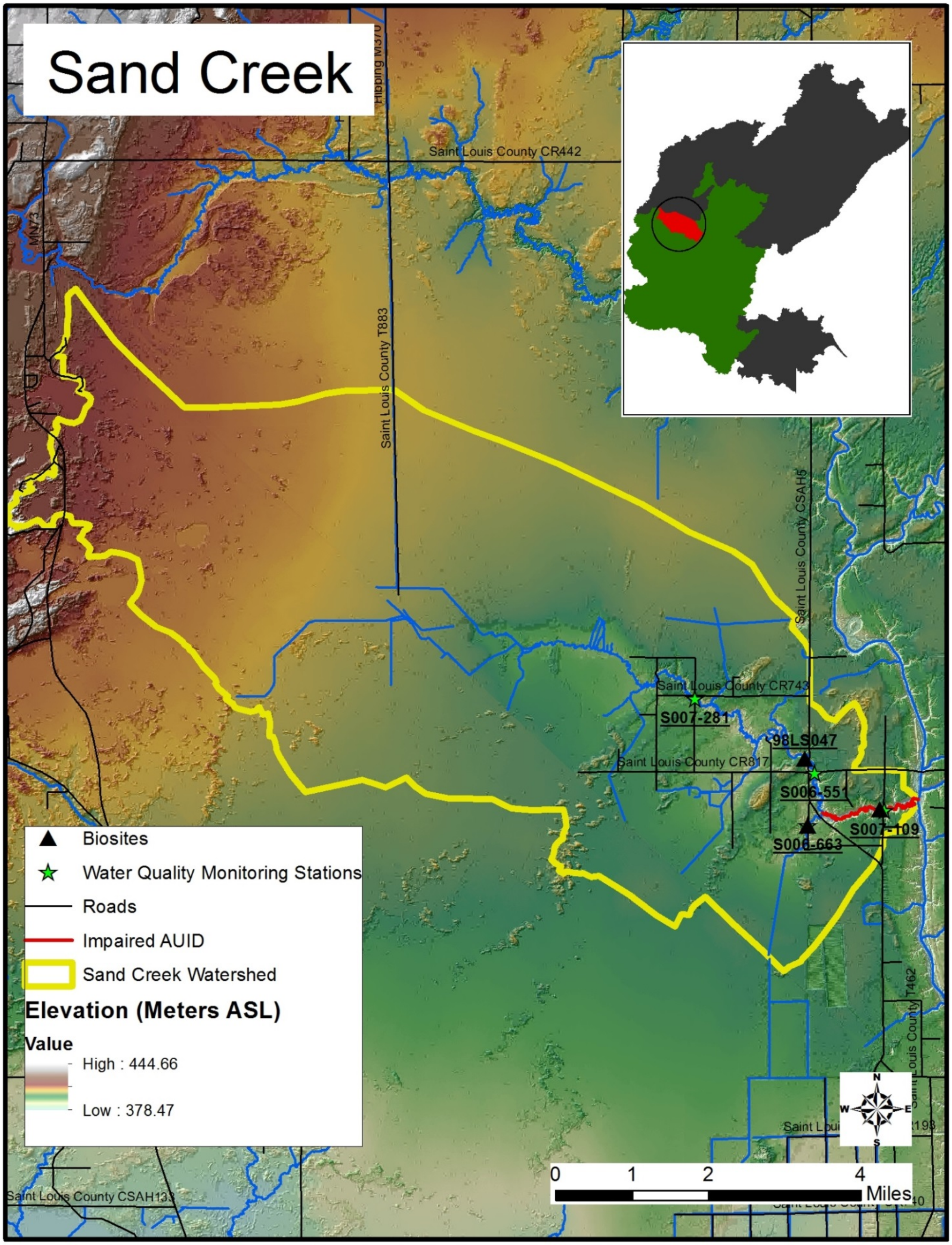


**Figure 122:** The Sand Creek Watershed and biological/water chemistry stations.

### 5.8.1 Total Suspended Solids & Turbidity

A total of 16 samples from three sampling sites over three years (2011-2013) were used to develop a summary of TSS and Secchi tube data. The data summary by average TSS value, number of samples and exceedance percentage are shown in Table 43. The station at CSAH-5 (S007-109) is co-located with biological monitoring site 09LS033, which failed to meet the FIBI threshold during a 2009 sampling event. TSS results from this station failed to meet the 15 mg/L standard in 4 out of 16 samples (25%).

**Table 43**: TSS and Secchi tube average values and percent standard exceedances based on Sand Creek monitoring results

| Sites | Site Description | TSS Average (mg/L) | TSS % Exceeding Draft Standard | Secchi Tube Average | Secchi Tube % Exceeding Draft Standard | Total # of Samples | Total % Exceeding Draft Standard |
|---|---|---|---|---|---|---|---|
| S007-281,<br>S006-551,<br>S007-109 | All Sand Creek sampling locations | 12.5 | 25.0% | 46.7 | 25.0% | 16 | 25.0% |

Box plots of the TSS and Secchi tube values for Sand Creek and the "A" and "B" reference streams in the SLRW are shown in Figure 123 and 124. Results for both of these parameters show that Sand Creek has higher TSS concentrations and lower water transparency than SLRW reference streams. Yet, only the 75th percentile values fell below the draft standards of 15 mg/L for TSS and 40 cm for Secchi tube. The three sites on Sand Creek that were used in this analysis were in violation of the 15 mg/L draft TSS standard frequently enough to technically be considered "impaired" for TSS, but TSS cannot be considered a stressor based on water quality data alone.

#### Seasonal variation in total suspended solids

In contrast to many streams in Northeastern Minnesota, Sand Creek appears to have clearer water during snowmelt than during baseflow (Figure 125). There is no discernible seasonal trend in TSS due to a lack of dated TSS samples (n=2). This may be caused by the relative increase in bog stained surface water during periods of low flow. During rain events and snowmelt, there may be more runoff from surficial sources which may actually increase transparency of the water in Sand Creek. Additional sampling, especially during snowmelt and during rain events, is recommended to make sure that this apparent trend is not merely attributed to a lack of sufficient data.

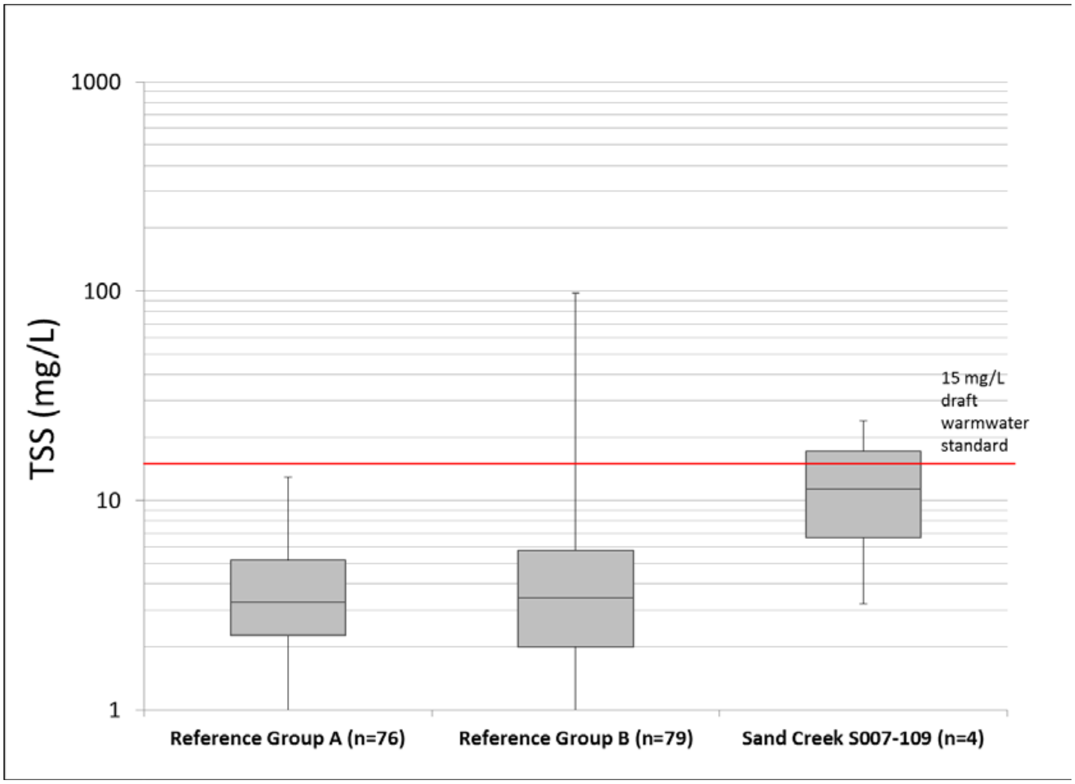


**Figure 123**: Box plots of TSS values for Sand Creek compared to reference streams (see Section 1.2.3 for reference stations).

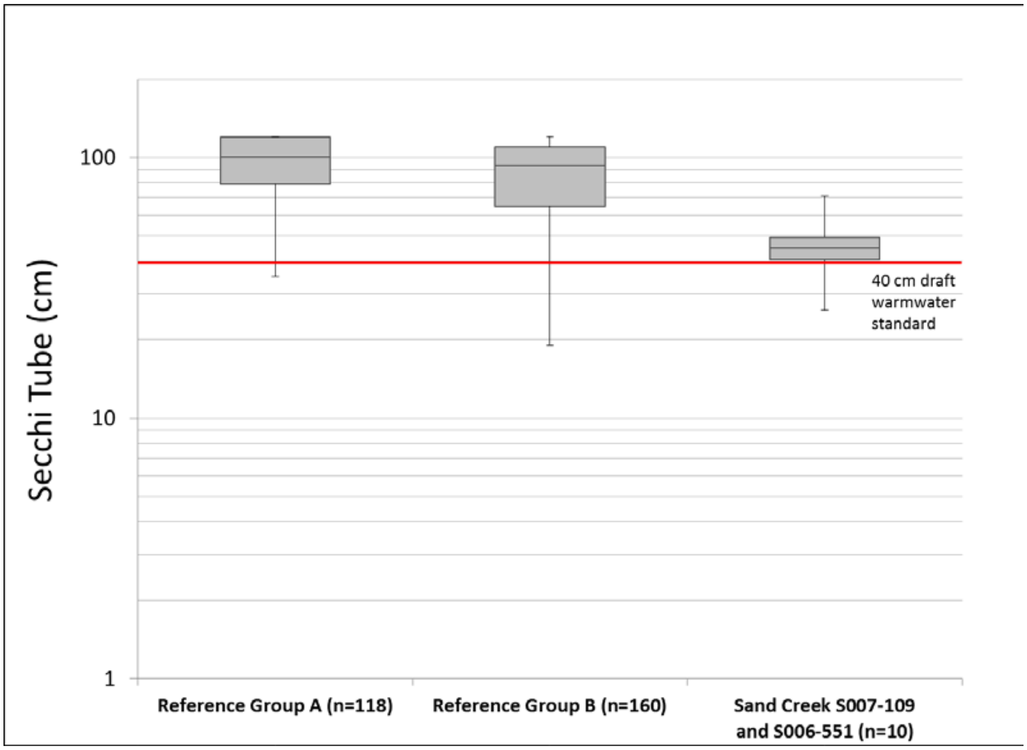


**Figure 124**: Box plots of Secchi tube transparency values for Sand Creek compared to reference streams (see Section 1.2.3 for reference stations

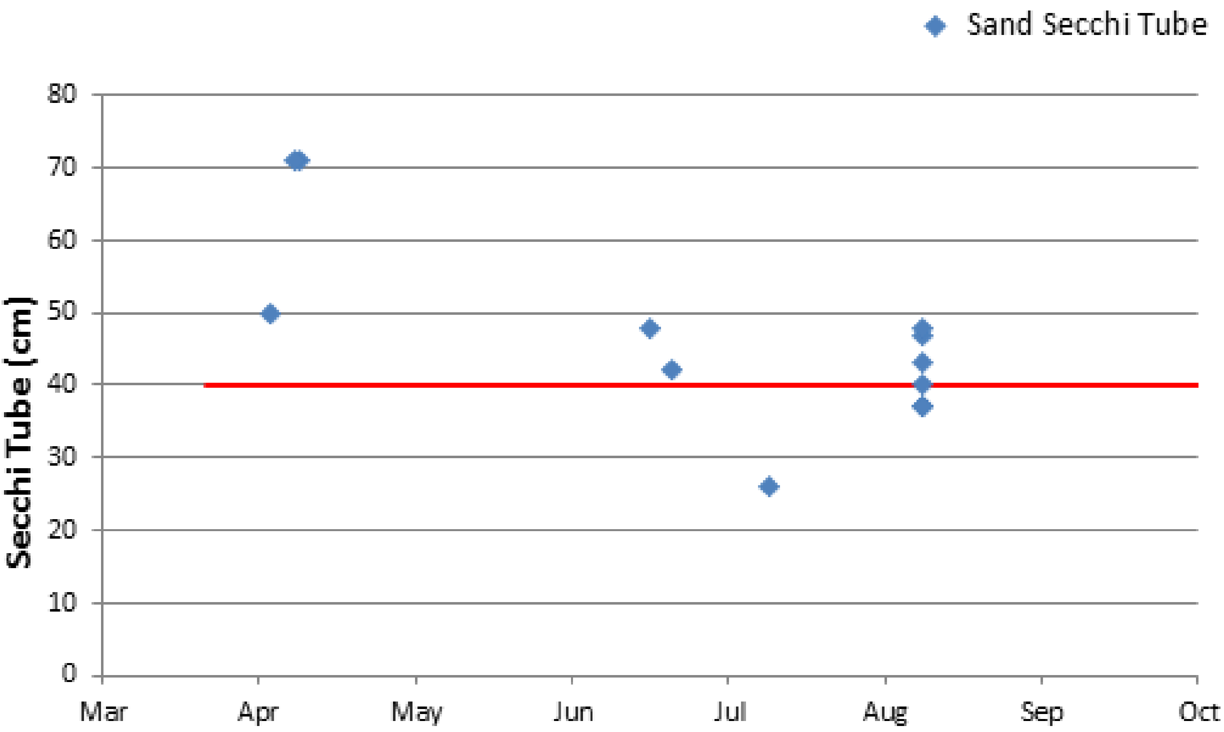


**Figure 125:** Sand Creek Secchi Tube data plotted by month

**Figure 126:** Sand Creek station S006-551 looking upstream during high flow conditions (left) and low flow conditions (right). Note tannin stained water in both photos.

## Sources and pathways of sediment in the Sand Creek Watershed

### Ditching/Channelization

Channel straightening and meander bend removal result in a shortening of the stream channel, which causes the water slope to increase and the velocity of channel-forming discharge to increase. If grade control (e.g. culverts, bedrock, stable riffles) is absent, channel incision will often follow – delivering sediment to the stream from the bed and banks as channel evolution progresses. The upper reaches of Sand Creek and most of its tributaries have been channelized or straightened (see Figure 122). In most instances the stream seems to have been ditched to allow better drainage of the peat bogs that dominate the watershed, which lies entirely within the Glacial Lake Upham basin.

Channel Instability/Bank Erosion

Areas of channel instability and bank erosion were observed at the 09LS033 biological monitoring station (see Figure 127) and other road crossings in the watershed. Channel instability in the Sand Creek Watershed is likely a consequence of increased peak flows as a result of efficient landscape drainage (ditching). Many reaches in Sand Creek show erosion on both sides of the river as well as trees falling into the channel. These are dead giveaways for incision and support the hypothesis that ditching in the watershed increased water slopes and peak flows and led to channel degradation.

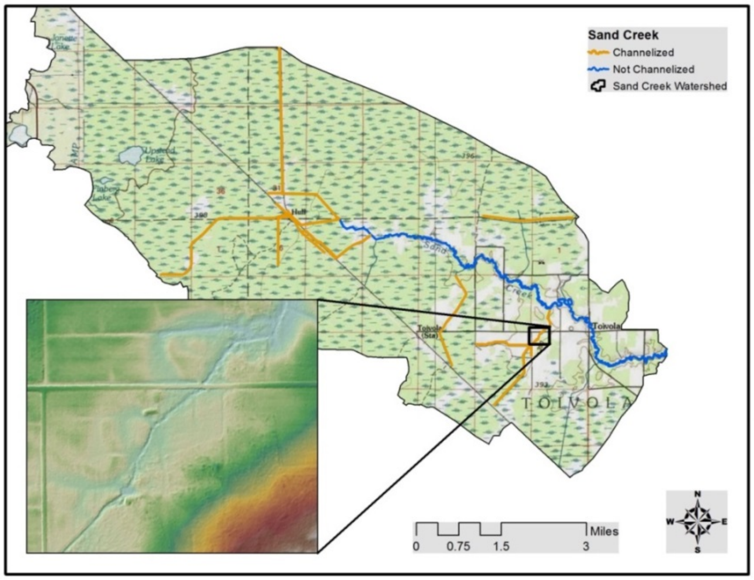


**Figure 127: (Left)** Map showing the channelized and natural channel reaches in the Sand Creek Watershed.

**(Above):** Erosion on both banks and leaning trees are a sign of channel incision in Sand Creek

## Biological effects of elevated TSS

### Fish Response to TSS

The FIBI impairment on Sand Creek is the result of poor metrics related to a lack of sensitive species that are expected in healthy Northern Streams. A total of two biological monitoring stations are located on Sand Creek, one on the impaired reach (09LS033), and one upstream of the impaired reach (98LS047). Spatial comparisons between these stations are somewhat difficult to make due to temporal sampling discrepancies – station 98LS047 was sampled once in 1998, while station 09LS033 was sampled a single time in 2009.

Species-level TIVs for TSS were used to evaluate whether or not the fish community of Sand Creek is tolerant or intolerant of elevated TSS. Data from the two monitoring stations on Sand Creek show discrepant results. Over 65% of the fish community at station 98LS047 consisted of species considered moderately or highly intolerant to elevated levels of TSS. Three intolerant species were present; Mottled Sculpin (n=50), Longnose Dace (n=63) and Burbot (n=3). The rest of the fish community at 98LS947 is considered neutral to TSS and included species such as White Sucker, Trout-Perch, Central Mudminnow and Johnny Darter.

In contrast, the fish community observed at station 09LS033 during the 2009 sampling event exhibited a higher tolerance of elevated TSS. Only 14.3% of the individuals sampled represented taxa that are considered intolerant to elevated TSS. Approximately 80% of the fish sampled at this station can be considered neither tolerant nor intolerant of TSS, and 6.3% were considered moderately tolerant to TSS. There is a fair amount of species overlap between the two sites, although the high numbers of intolerant Longnose Dace present at the 1998 station are absent in the 2009 sample, and the 2009 monitoring effort captured several “neutral” shiner species that were not present in the 1998 station.

Due to the temporal and spatial differences between the two sites, it is difficult to say whether the tolerance contrast between the two assemblages is the result of changes to the watershed over time or different water quality conditions in the two reaches of stream that may still exist today. Because of this, recent biological results from the impaired watershed (09LS033) will be given more weight than the older data from 98LS047. When comparing Sand Creek to the high-quality biological stations in the SLRW, the fish assemblage at 09LS033 is much more tolerant of TSS (Figure 128). The fish community TSS index values from 09LS033 are worse than 100% of comparable stations scoring above the impairment threshold or UCL of the FIBI criteria, and more tolerant than 75% of all of the stations in the SLRW scoring above the UCL of their respective FIBI impairment thresholds.

**Summary: Are elevated TSS concentrations a stressor in Sand Creek?**

The TSS results for Sand Creek show that the TSS and transparency water quality standards are exceeded with some regularity. However, the results also show that most results narrowly exceed the standard, and thus biota in Sand Creek are not exposed to extremely high concentrations of TSS. The naturally-occurring bog stain of surface waters in Sand Creek is also contributing to the low transparency levels, particularly during baseflow when water from the wetlands and bogs the dominant source of water in the creek.

Despite some indication that TSS and transparency conditions are not ideal, we do not recommend listing TSS as a stressor to aquatic life in Sand Creek for several reasons; (1) the existing data set contains relatively few sampling results, (2) the rate and magnitude of TSS and transparency exceedances are not severe, (3) the impaired biological monitoring station (09LS033) is not dominated by fish species that are tolerant of elevated TSS concentrations.

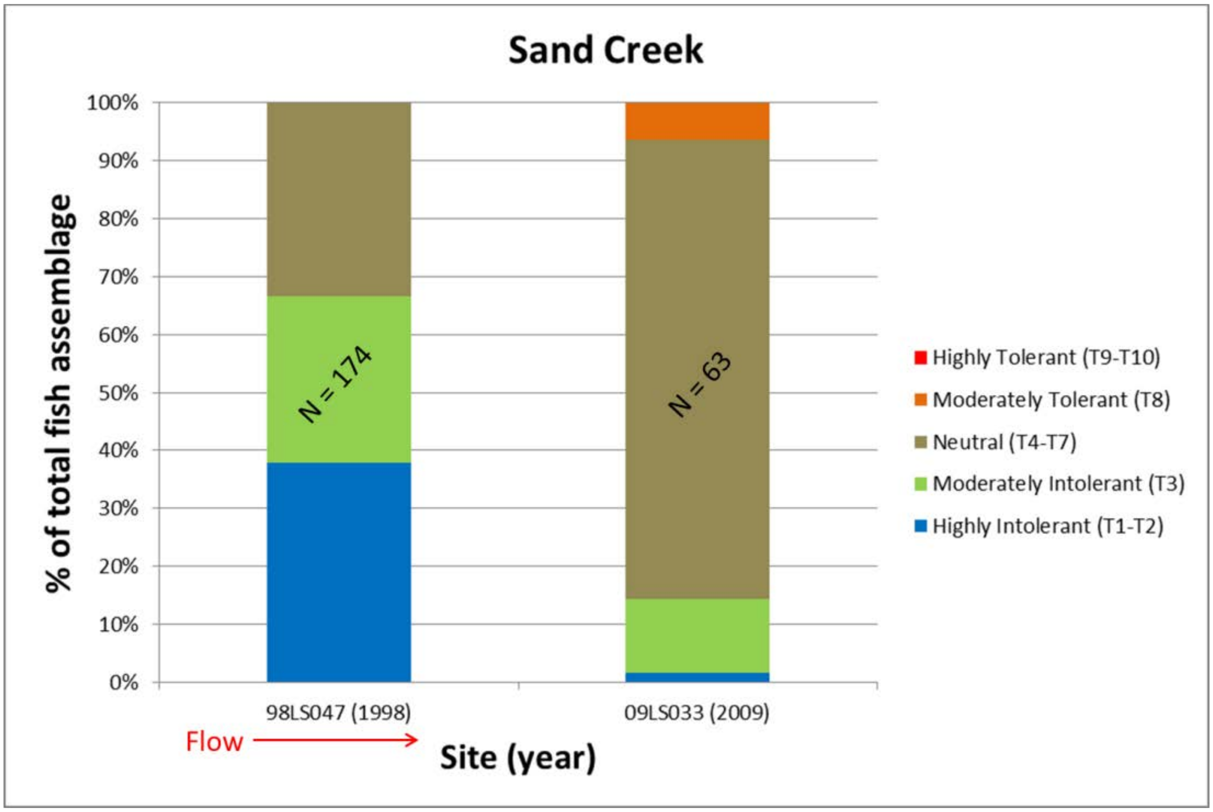


**Figure 128:** TSS tolerance of fish assemblages by site in Sand Creek (n = number of fish)

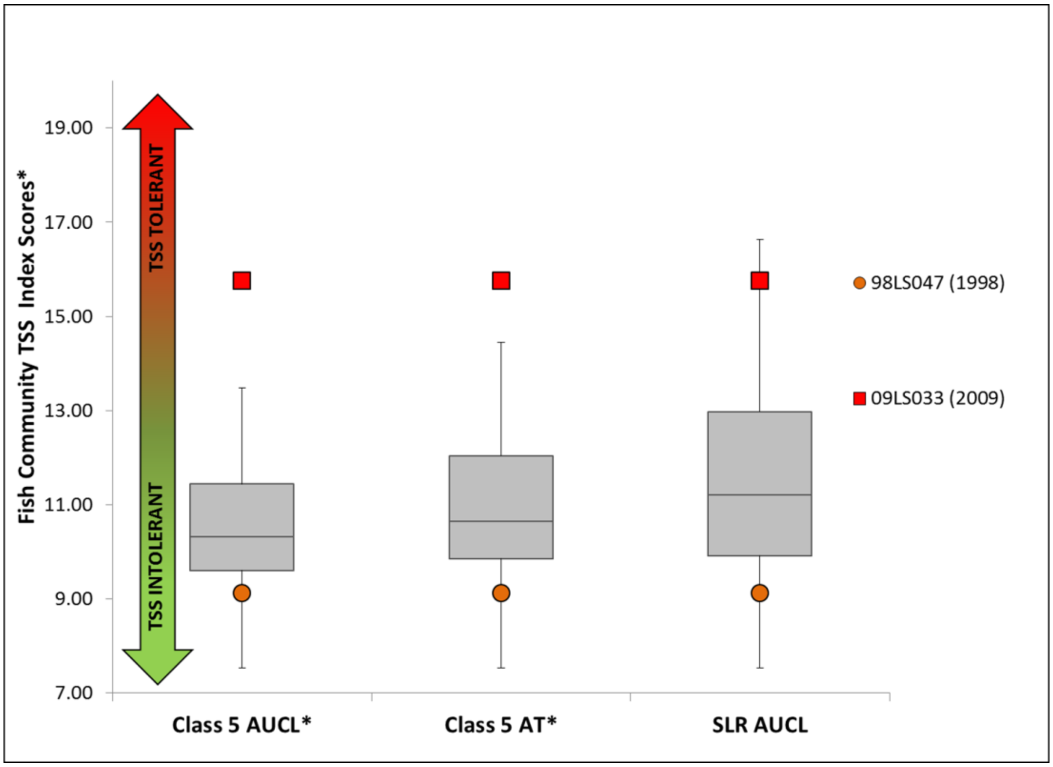


**Figure 129:** Fish community TSS TIVs for Sand Creek, compared to unimpaired streams. * See section 4 for explanation of TIVs SLR= St. Louis River Watershed AUCL = Above Upper Confidence Limit of FIBI threshold AT = Above Fish IBI Threshold

## 5.8.2 Poor Physical Habitat Conditions

The MSHA methodology was used to assess habitat conditions in Sand Creek during the 2009 biological sampling events. Overall MSHA scores were slightly better at station 98LS047, but the results were very comparable (64.1/100 and 61.25/100) and both stations were categorized as having “fair” habitat conditions. Individual habitat metric scores vary considerably between the sites and may be responsible for the discrepant FIBI scores. At the impaired biological monitoring station (09LS033), scores related to substrate conditions were significantly lower, which corresponds to poor substrate conditions for habitat sensitive fish. The poor substrate scores at 09LS033 were largely due to the dominance of clay and silt material, and the complete absence of coarse substrates. Substrate scores were considerably higher at 98LS047 due to the presence of gravel, cobble, and some boulders.

The MSHA results from the two Sand Creek biological monitoring stations also point to a difference in stream feature facets that likely factor into the fish species that are dominant at these stations. The impaired station was dominated by run and pool habitats, and had very few riffle habitats available. The limited riffle habitat available at this station may explain the absence of Longnose Dace from this station (they abundant at the station upstream) and the lower numbers of Mottled Sculpin.

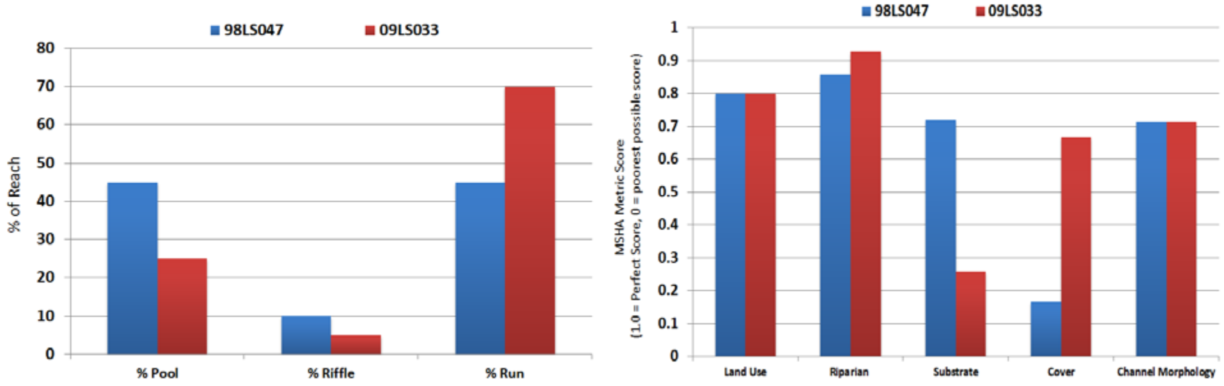
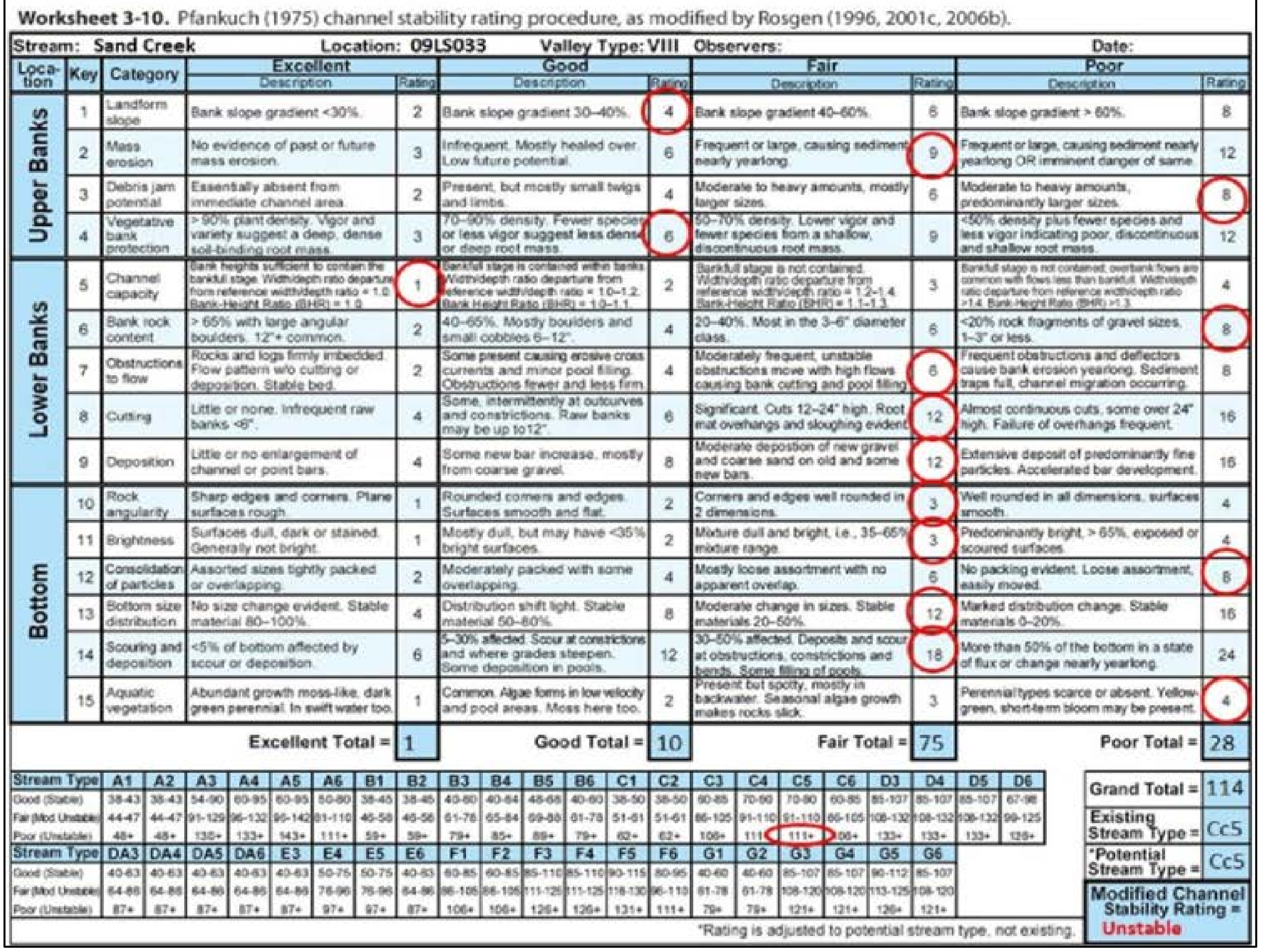


**Figure 130: (Left)** Percentage of Sand Creek monitoring stations in riffle, run, and pool. **(Right)** Summary of key MSHA habitat metric results related to fish habitat

Both monitoring stations received excellent scores for riparian corridor conditions due to the low-impact land use and intact riparian zones. Based on these results and observations collected during several visits to these monitoring sites, riparian land-use can be eliminated as a source of habitat degradation in this watershed. Channel processes, specifically bank erosion, are the main source of habitat loss. The PSI scores for Sand Creek support this claim. Characteristics such as bank erosion, debris jams and loose bottom sediments were common at the impaired site and are typical of an incised, unstable channel (see photo in Figure 131). A PSI score of 114 was recorded at the impaired biological monitoring station, which corresponds to a stability rating of “unstable” for the potential “C5” stream type. It is likely that the consequences of channel incision, such as excess fine particle deposition and pool filling, are causing habitat degradation in Sand Creek.



**Worksheet 3-10.** Pfankuch (1975) channel stability rating procedure, as modified by Rosgen (1996, 2001c, 2006b).

**Stream:** Sand Creek **Location:** 09LS033 **Valley Type:** VIII **Observers:** **Date:**

| Location | Key | Category | Excellent |  | Good |  | Fair |  | Poor |  |
|---|---|---|---|---|---|---|---|---|---|---|
|  |  |  | Description | Rating | Description | Rating | Description | Rating | Description | Rating |
| Upper Banks | 1 | Landform slope | Bank slope gradient <30%. | 2 | Bank slope gradient 30–40%. | [✓ 4] | Bank slope gradient 40–60%. | 6 | Bank slope gradient > 60%. | 8 |
|  | 2 | Mass erosion | No evidence of past or future mass erosion. | 3 | Infrequent. Mostly healed over. Low future potential. | 6 | Frequent or large, causing sediment nearly yearlong. | [✓ 9] | Frequent or large, causing sediment nearly yearlong OR imminent danger of same. | 12 |
|  | 3 | Debris jam potential | Essentially absent from immediate channel area. | 2 | Present, but mostly small twigs and limbs. | 4 | Moderate to heavy amounts, mostly larger sizes. | 6 | Moderate to heavy amounts, predominantly larger sizes. | [✓ 8] |
|  | 4 | Vegetative bank protection | > 90% plant density. Vigor and variety suggest a deep, dense soil-binding root mass. | 3 | 70–90% density. Fewer species or less vigor suggest less dense or deep root mass. | [✓ 6] | 50–70% density. Lower vigor and fewer species from a shallow, discontinuous root mass. | 9 | <50% density plus fewer species and less vigor indicating poor, discontinuous and shallow root mass. | 12 |
| Lower Banks | 5 | Channel capacity | Bank heights sufficient to contain the bankfull stage. Width/depth ratio departure from reference width/depth ratio = 1.0. Bank-Height Ratio (BHR) = 1.0. | [✓ 1] | Bankfull stage is contained within banks. Width/depth ratio departure from reference width/depth ratio = 1.0–1.2. Bank-Height Ratio (BHR) = 1.0–1.1. | 2 | Bankfull stage is not contained. Width/depth ratio departure from reference width/depth ratio = 1.2–1.4. Bank-Height Ratio (BHR) = 1.1–1.3. | 3 | Bankfull stage is not contained; overbank flows are common with flows less than bankfull. Width/depth ratio departure from reference width/depth ratio >1.4. Bank-Height Ratio (BHR) >1.3. | 4 |
|  | 6 | Bank rock content | > 65% with large angular boulders. 12"+ common. | 2 | 40–65%. Mostly boulders and small cobbles 6–12". | 4 | 20–40%. Most in the 3–6" diameter class. | 6 | <20% rock fragments of gravel sizes, 1–3" or less. | [✓ 8] |
|  | 7 | Obstructions to flow | Rocks and logs firmly imbedded. Flow pattern w/o cutting or deposition. Stable bed. | 2 | Some present causing erosive cross currents and minor pool filling. Obstructions fewer and less firm. | 4 | Moderately frequent, unstable obstructions move with high flows causing bank cutting and pool filling. | [✓ 6] | Frequent obstructions and deflectors cause bank erosion yearlong. Sediment traps full, channel migration occurring. | 8 |
|  | 8 | Cutting | Little or none. Infrequent raw banks <6". | 4 | Some, intermittently at outcurves and constrictions. Raw banks may be up to12". | 6 | Significant. Cuts 12–24" high. Root mat overhangs and sloughing evident. | [✓ 12] | Almost continuous cuts, some over 24" high. Failure of overhangs frequent. | 16 |
|  | 9 | Deposition | Little or no enlargement of channel or point bars. | 4 | Some new bar increase, mostly from coarse gravel. | 8 | Moderate deposition of new gravel and coarse sand on old and some new bars. | [✓ 12] | Extensive deposit of predominantly fine particles. Accelerated bar development. | 16 |
| Bottom | 10 | Rock angularity | Sharp edges and corners. Plane surfaces rough. | 1 | Rounded corners and edges. Surfaces smooth and flat. | 2 | Corners and edges well rounded in 2 dimensions. | [✓ 3] | Well rounded in all dimensions, surfaces smooth. | 4 |
|  | 11 | Brightness | Surfaces dull, dark or stained. Generally not bright. | 1 | Mostly dull, but may have <35% bright surfaces. | 2 | Mixture dull and bright, i.e., 35–65% mixture range. | [✓ 3] | Predominantly bright, > 65%, exposed or scoured surfaces. | 4 |
|  | 12 | Consolidation of particles | Assorted sizes tightly packed or overlapping. | 2 | Moderately packed with some overlapping. | 4 | Mostly loose assortment with no apparent overlap. | 6 | No packing evident. Loose assortment, easily moved. | [✓ 8] |
|  | 13 | Bottom size distribution | No size change evident. Stable material 80–100%. | 4 | Distribution shift light. Stable material 50–80%. | 8 | Moderate change in sizes. Stable materials 20–50%. | [✓ 12] | Marked distribution change. Stable materials 0–20%. | 16 |
|  | 14 | Scouring and deposition | <5% of bottom affected by scour or deposition. | 6 | 5–30% affected. Scour at constrictions and where grades steepen. Some deposition in pools. | 12 | 30–50% affected. Deposits and scour at obstructions, constrictions and bends. Some filling of pools. | [✓ 18] | More than 50% of the bottom in a state of flux or change nearly yearlong. | 24 |
|  | 15 | Aquatic vegetation | Abundant growth moss-like, dark green perennial. In swift water too. | 1 | Common. Algae forms in low velocity and pool areas. Moss here too. | 2 | Present but spotty, mostly in backwater. Seasonal algae growth makes rocks slick. | 3 | Perennial types scarce or absent. Yellow-green, short-term bloom may be present. | [✓ 4] |
|  |  |  | Excellent Total = | 1 | Good Total = | 10 | Fair Total = | 75 | Poor Total = | 28 |

| Stream Type | A1 | A2 | A3 | A4 | A5 | A6 | B1 | B2 | B3 | B4 | B5 | B6 | C1 | C2 | C3 | C4 | C5 | C6 | D3 | D4 | D5 | D6 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Good (Stable) | 38-43 | 38-43 | 54-90 | 60-95 | 60-95 | 50-80 | 38-45 | 38-45 | 40-60 | 40-64 | 48-68 | 40-60 | 38-50 | 38-50 | 60-85 | 70-90 | 70-90 | 60-85 | 85-107 | 85-107 | 85-107 | 67-98 |
| Fair (Mod Unstable) | 44-47 | 44-47 | 91-129 | 96-132 | 96-142 | 81-110 | 46-58 | 46-58 | 61-78 | 65-84 | 69-88 | 61-78 | 51-61 | 51-61 | 86-105 | 91-110 | 91-110 | 86-105 | 108-132 | 108-132 | 108-132 | 99-125 |
| Poor (Unstable) | 48+ | 48+ | 130+ | 133+ | 143+ | 111+ | 59+ | 59+ | 79+ | 85+ | 89+ | 79+ | 62+ | 62+ | 106+ | 111 | [✓ 111+] | 106+ | 133+ | 133+ | 133+ | 126+ |

| Stream Type | DA3 | DA4 | DA5 | DA6 | E3 | E4 | E5 | E6 | F1 | F2 | F3 | F4 | F5 | F6 | G1 | G2 | G3 | G4 | G5 | G6 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Good (Stable) | 40-63 | 40-63 | 40-63 | 40-63 | 40-63 | 50-75 | 50-75 | 40-63 | 60-85 | 60-85 | 85-110 | 85-110 | 90-115 | 80-95 | 40-60 | 40-60 | 85-107 | 85-107 | 90-112 | 85-107 |
| Fair (Mod Unstable) | 64-86 | 64-86 | 64-86 | 64-86 | 64-86 | 76-96 | 76-96 | 64-86 | 86-105 | 86-105 | 111-125 | 111-125 | 116-130 | 96-110 | 61-78 | 61-78 | 108-120 | 108-120 | 113-125 | 108-120 |
| Poor (Unstable) | 87+ | 87+ | 87+ | 87+ | 87+ | 97+ | 97+ | 87+ | 106+ | 106+ | 126+ | 126+ | 131+ | 111+ | 79+ | 79+ | 121+ | 121+ | 126+ | 121+ |

|  |  |
|---|---|
| Grand Total = | 114 |
| Existing Stream Type = | Cc5 |
| *Potential Stream Type = | Cc5 |
| Modified Channel Stability Rating = | Unstable |

*Rating is adjusted to potential stream type, not existing.

**Figure 131:** Pfankuch rating for Sand Creek biological monitoring reach 09LS033

**Biological effects of degraded habitat**

Poor physical habitat conditions can limit the abundance and taxa richness of non-tolerant fish species. The number of non-tolerant fish within the impaired reach of Sand Creek was fairly low, at 0.14 per meter. That result is around the 25th percentile value for streams of the same IBI class that are either scored above the IBI threshold or UCL of the IBI threshold. The limited riffle, glide, and pool habitat within the impaired reach is likely a significant factor contributing to the lack of diversity and abundance of non-tolerant fish species. Fish taxa richness and abundance measures were considerably higher at station 98LS047, which provides additional evidence in support of a physical habitat stressor at 09LS033. A rate of 0.57 non-tolerant fish/meter were sampled in the 1998 visit to station 98LS047, compared to 0.14 non-tolerant fish/meter at the impaired site.

In addition, station 98LS047 supported relatively healthy numbers of habitat-sensitive taxa compared to the impaired station (09LS033). Taxa richness of benthic insectivores and darter, sculpin, sucker species were the same at both monitoring stations. However, a higher number of piscivores, riffle dwelling taxa, and gravel spawning taxa were observed at station 98LS047 where habitat conditions are superior. Of the habitat-related metrics evaluated, station 98LS047 had metric values comparable to the median values observed from a set of 47 class 5 FIBI stations that scored above the IBI impairment threshold. On the other hand, station 09LS033 supported fewer riffle dwelling taxa, piscivores, and gravel spawning fish species than the majority of high-quality class 5 FIBI stations.

**Summary: Are poor physical habitat conditions a stressor in Sand Creek?**

The poor MSHA substrate scores and the “unstable” Pfankuch Stability rating suggest that a certain amount of channel instability, erosion, and sedimentation is occurring in Sand Creek, particularly in the lower reach where FIBI scores are poor. The poor habitat in Sand Creek is likely contributing to a decrease in the biological integrity of the stream, as demonstrated by the very low number of non-tolerant fish/meter and the low number of piscivorous, riffle dwelling, and gravel spawning fish species present relative to higher quality sites upstream. There is adequate evidence to suggest that poor physical habitat conditions are a stressor contributing to the fish impairment.

## 5.8.3 Sand Creek: Overall Summary of Stressors to Aquatic Life

**Table 44:** Summary of SID results for Sand Creek, along with recommendations for restoration and implementation projects.

| Candidate Cause | Result |
|---|---|
| **Total Suspended Solids (TSS) / Turbidity** | X |
| **Poor Physical Habitat Conditions** | ● |
| **Altered Hydrology** | ○ |
| Key: ● **= confirmed stressor** ○ **= Potential Stressor X = eliminated candidate cause** | |

## 5.9 Stony Creek

Stony Creek flows primarily through the Sax-Zim bog area. The watershed lies entirely within the Glacial Lake Upham basin and thus is fairly flat in slope. The majority of Stony Creek exhibits the characteristics of a sinuous “E” type channel, but in the last half mile before emptying into the St. Louis River, the stream picks up gradient and loses sinuosity and becomes a type “C” channel. For most of its length, Stony Creek appears to have cut an alluvial valley for itself within the Glacial Lake Upham lake bed. The impaired AUID of Stony Creek is an E channel with a gentle slope of 0.04% (2 feet/mile).

Two biological monitoring stations are located on Stony Creek. Station 67LS020 was originally sampled in 1967 by DNR, and was resampled for fish and macroinvertebrates in 2012. Data collected in 2009 at station 09LS036 were exclusively used for the assessment process, and it is the results from this station which prompted the impairment listing for low fish and MIBI scores. Station information and a summary of fish and MIBI results are listed in Table 45.

The FIBI scores for these two sites are only narrowly below the impairment threshold and do not reflect a highly degraded fish community. The FIBI score for station 09LS036 was 39/100 for both the 2009 and 2012 visits, a score that is only three points below the impairment threshold. Dominant fish species at this station over two sampling visits include Central Mudminnow, White Sucker, and Johnny Darter. A small population of Pearl Dace, and Blacknose Shiner were the only two sensitive minnow species observed at this monitoring station. Low scores in metrics related to fish abundance and taxa richness of minnow species were significant factors in the failure of this station to meet FIBI criteria.

Data from the other station on Stony Creek (67LS020) was not used during the assessment process, as the 1967 data were too old for consideration and the 2012 data were collected after the stream was already listed as an impaired water. The fish community at 67LS020 was dominated by Central Mudminnow, Brook Stickleback, and Fathead Minnow, which are all highly tolerant to poor water quality (esp. low DO) and limited physical habitat. Pearl Dace were also present at this station, accounting for the only sensitive minnow species at this location.

**Table 45:** Summary of biological monitoring stations, sampling visits, and results.

**Fish Assessments**

| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | FIBI Class | FIBI Result *(visit year)* | FIBI Result *(visit year)* | Standard | IBI Lower Confidence Limit | IBI Upper Confidence Limit |
|---|---|---|---|---|---|---|---|---|---|
| 67LS020 | 19.00 | 0.03 | 2 | 7 | **32** *(2012)* | - | 42 | 32 | 52 |
| 09LS036 | 21.54 | 0.06 | 2 | 6 | **39** *(2012)* | **39** *(2009)* | 42 | 26 | 58 |

**Macroinvertebrate Assessments**

| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | Invert IBI Class | Invert IBI Result *(visit year)* | Invert IBI Result *(visit year)* | Standard | IBI Lower Confidence Limit | IBI Upper Confidence Limit |
|---|---|---|---|---|---|---|---|---|---|
| 67LS020 | 19.00 | 0.03 | 2 | 4 | **40.85** *(2012)* | - | 51 | 37.40 | 64.60 |
| 09LS036 | 21.54 | 0.06 | 2 | 4 | **29.06** *(2012)* | **42.48** *(2009)* | 51 | 37.40 | 64.60 |

The MIBI results were comparatively lower than the fish results. The MIBI scores were between 7 and 21 points below the impairment threshold. Similar to the fish data, the 2009 MIBI results from station 09LS036 were the only data considered during the assessment of Stony Creek. Common macroinvertebrate taxa observed in this sampling event included *Stenacron* (flathead mayflies), *Procloeon* (small minnow mayflies), *Pisidiidae* (pill clams), *Caenis* (white-winged Sulphur mayflies), and *Cricotopus* (non-biting midge). The low MIBI result from this station is driven predominantly by a lack of intolerant macroinvertebrate taxa, low overall taxa richness, low numbers of collector-filterer macroinvertebrate individuals, and low numbers of non-Hydropsychid caddisflies.

**Table 46:** Summary of impairment symptoms observed in the biological assemblage of Stony Creek

| Symptoms of Biological Impairment (Fish) | Symptoms of Biological Impairment (Macroinvertebrates) |
|---|---|
| ● Low fish counts<br>● Lack of insectivorous minnow sp.<br>● Lack of headwaters minnow sp.<br>● Lack of sensitive fish taxa | ● Low taxa richness<br>● Low Relative Abundance of Collector-Filterer Taxa<br>● Lack of intolerant taxa<br>● Low relative abundance of Trichopteran (caddisfly) taxa |

## Candidate Causes for Impairment

Water quality and physical habitat data were used identify candidate causes for impairment in Stony Creek. The following candidate causes were selected for further analysis:

1. **Low DO**
2. **TSS / Turbidity**
3. **Poor Habitat Conditions**

### 5.9.1 Low Dissolved Oxygen

Instantaneous DO data are available for two stations on Stony Creek, both of which have co-located biological monitoring data. DO concentrations in Stony Creek regularly fell below the 5 mg/L warmwater standard during the months of July through October (Figure 133). During these periods of lower DO concentrations, stream stage has generally been extremely low and current velocities have been imperceptible. The picture of station 09LS036 in Figure 132 shows an example of the low flow conditions that are commonly observed during the late summer and early fall months on this stream. Although there remains a significant amount of water in the channel, there often is no perceivable flow. At the time this photo was taken (September 12, 2012 @ 11:00), the DO concentration was 1.88 mg/L.

Instantaneous measurements were also collected longitudinally along Stony Creek in order to compare DO concentrations from different stream reaches under similar ambient conditions. The results, shown in Figure 133, show low DO conditions (at or below 3 mg/L) throughout most of Stony Creek, with lower concentrations in the headwaters and tributary streams. Similar to the continuous DO monitoring results, the longitudinal data show very low diurnal change in DO concentrations.

DO and other field parameters (temperature, pH, Sp. Conductivity) were measured continuously over a five day period from July 13, 2012 to July 18, 2012 at station 09LS036. DO concentrations observed during this monitoring period ranged from 3.83 mg/L to 4.40 mg/L, with 100% of the readings falling below the 5 mg/L warmwater DO threshold (Figure 134). DO flux was very low (less than 0.50 mg/L), which suggests that primary productivity in Stony Creek is minimal. Although nutrient concentrations in Stony Creek can be relatively high, water clarity is reduced due to heavy tannin stain and does not provide favorable conditions for supporting the growth of aquatic plants or algae.

**Figure 132:** Low, stagnant flow conditions are common in the impaired reach of Stony Creek. Also note the dark tannin stained appearance of the water, which limits sunlight penetration and macrophytes/algae growth and reduces diurnal DO flux.

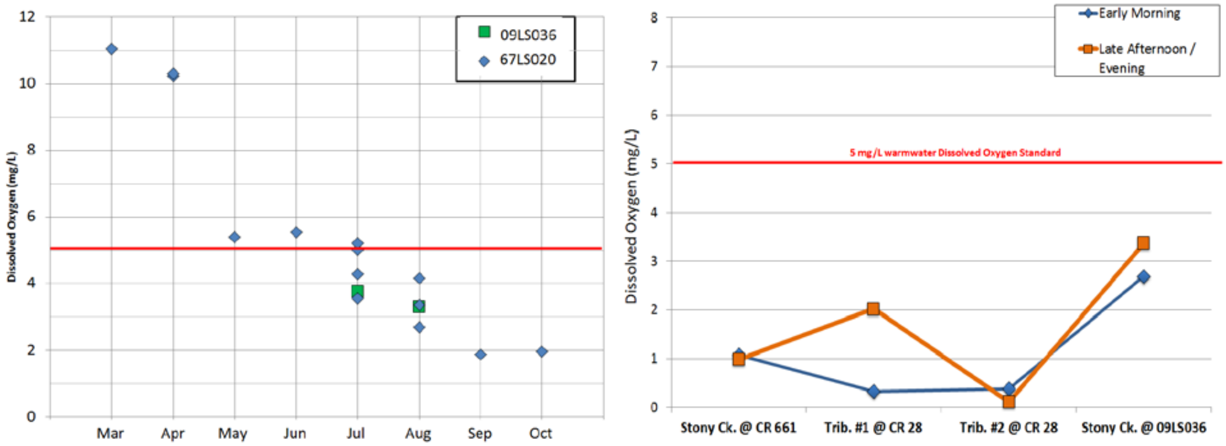
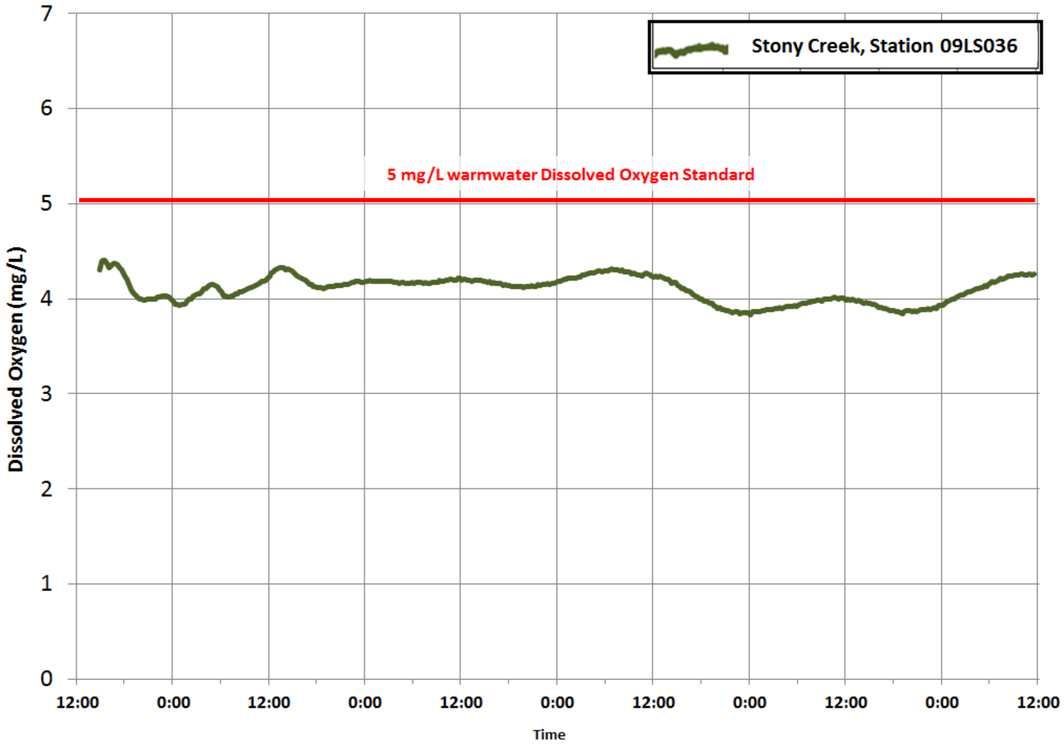


**Figure 133: (Left)** Instantaneous (point) monitoring results for DO at Stony Creek monitoring stations, arranged by calendar month. **(Right)** Results from a longitudinal DO survey conducted August 6, 2012.



| Dates of Monitoring Period | # of Readings | Min (mg/L) | Max (mg/L) | 24-Hour Flux Data Average (mg/L) | Max (mg/L) | % Readings below Warmwater DO Threshold - (5 mg/L) |
|---|---|---|---|---|---|---|
| ***7/13/2012 – 7/18/2012*** | 486 | 3.83 | 4.40 | 0.29 | 0.47 | 100% |

**Figure 134:** Results from continuous DO monitoring at biological monitoring station 09LS036

## Sources and Pathways Contributing to Low Dissolved Oxygen

A combination of wetland processes, ditching, and geomorphic features are likely playing a role in the low DO concentrations observed in Stony Creek. Approximately 63% of the watershed is classified as woody or emergent herbaceous wetland. A very small percentage of the watershed area has been converted to livestock pastures or hay fields (less than 2%), which accounts for the majority of ongoing anthropogenic disturbance in the watershed. The headwaters of the creek form in a large peat bog complex, the edges of which have been extensively ditched and channelized. Several peat mining operations are located on the outer fringes of the Stony Creek Watershed. Essentially every tributary to Stony Creek originates from a bog or wetland complex and is channelized for its entire length until meeting the main stem (Figure 135). The connection between these natural background conditions and anthropomorphic changes to low DO concentrations in Skunk Creek are highlighted in Table 46.

The TP concentrations in Stony Creek are elevated well above river nutrient criteria for northern Minnesota. The TP concentrations during mid-summer to early fall baseflow conditions are generally greater than 0.100 mg/L. A maximum TP concentration of 0.240 mg/L was observed in Stony Creek during a snowmelt runoff event in March 2012. Despite these elevated TP concentrations, productivity in Stony Creek is low, as evidenced by the lack of aquatic macrophytes and flat diurnal DO profile observed in the July 2012 continuous monitoring effort.

**Table 46:** Summary of sources and pathways that potentially limit DO availability in Stony Creek

| Source/Pathway | Root Cause(s) | Symptom |
|---|---|---|
| **Flashy Hydrology** | Significant ditching of tributaries to Stony Creek | Very low streamflow conditions and stagnant water are common in summer and fall when low DO conditions are most evident |
| **Wetlands / Peat Soils** | Approximately 63% of watershed area is wetlands. Much of drainage area is a very large peat bog complex. | Tannin stained water reduces biological productivity. Organics from wetland/bogs in watershed increase BOD, which was fairly high at the biological monitoring station in Skunk Creek (6.3 mg/L; 7/18/2012) |
| **Geomorphology** | Low gradient stream with no roughness elements (cobble/boulders) | Lack of riffles results in no aeration of surface water from the atmosphere |

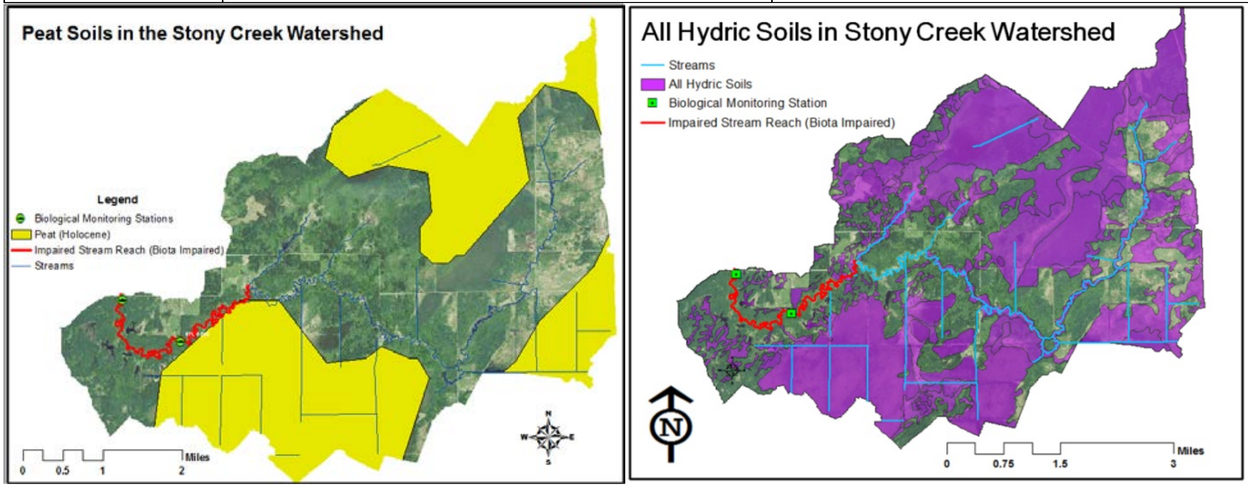


**Figure 135:** Peat and All Hydric soils coverage in the Stony Creek Watershed

## Biological Response to Low Dissolved Oxygen

The fish community of Stony Creek is composed primarily of species that are highly tolerant or moderately tolerant of low DO conditions. This is particularly the case at station 67LS020, where 94% of the total fish community was made up of four species that are very tolerant of low DO; Central Mudminnow (44%), Brook Stickleback (36%), Fathead Minnow (8%), and Golden Shiner (6%). Central Mudminnow was also a dominant or abundant species during the two monitoring visits to 09LS036. Several species that can be considered neutral in terms of their tolerance to low DO were observed at 09LS036 (Johnny Darter, Walleye) which suggests that the DO regime at this site may be more suitable for supporting fish species with more stringent DO requirements.

The fish community DO-index values for Stony Creek monitoring sites are poorer than comparable sites with high biological integrity scores (Figure 136). DO index scores from the 2012 sampling events are particularly low and fall well below the 25th percentile values from the set of reference streams. Clearly, the fish community of Stony Creek is one that is adapted to low DO conditions and lacks the sensitive fish taxa and overall diversity observed in high quality streams of the SLRW with more suitable DO concentrations. Based on these observations and available water chemistry data, low DO is considered a contributing stressor the fish impairment in Stony Creek.

The biological response to low DO concentrations in Stony Creek is more difficult to pin down. Both samples from station 09LS036 were dominated by mayflies from the genus Stenacron. Mayflies of this genus can live in waters that are stagnant and have very low DO concentrations. On the other hand, several macroinvertebrate taxa considered to be intolerant or moderately intolerant of low DO conditions were observed in small numbers in Stony Creek, including individuals from the genera *Neoplasta* (member of dance flies family), *Maccaffertium* (member of flathead mayfly family), and *Cheumatopsyche* (net-spinning caddisfly family). Individuals from intolerant or moderately intolerant taxa accounted for around 10% of the total community at all three monitoring stations, so although they were present, the majority of the organisms observed at these monitoring stations were not sensitive or intolerant to low DO conditions (Figure 139).

The macroinvertebrate DO index scores for Stony Creek do not provide convincing evidence for or against low DO as a stressor (Figure 138). Index scores for station 09LS036 are slightly better than the median score observed at class 4 SLRW biological monitoring stations scoring above the impairment threshold and UCL of the impairment threshold. In other words, the macroinvertebrate community at Stony Creek station 09LS036 is not any more adapted to low DO conditions than many of the higher scoring class 4 MIBI stations in the SLRW. The DO index score at station 67LS020 is slightly lower, but still comparable to many stations with good to excellent MIBI scores.

Summary: Is low dissolved oxygen a stressor in Stony Creek?

Water chemistry and biological monitoring data provide adequate evidence to diagnose low DO as a cause of fish impairment in Stony Creek. The seasonally low DO conditions, coupled with the abundance of DO tolerant fish species suggest that low DO is a limiting factor in this stream. Evidence linking low DO conditions to the macroinvertebrate impairment was not as strong, and other stressors (esp. habitat) may be more limiting to macroinvertebrate community.

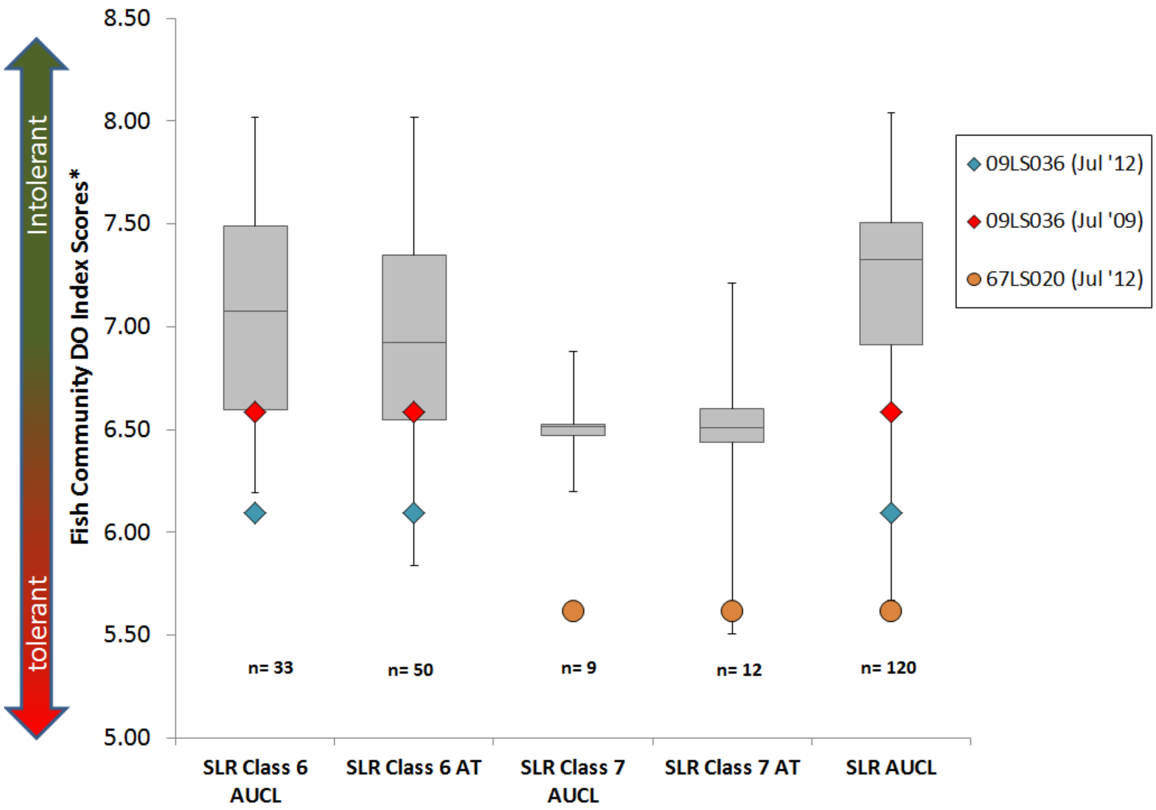


**Figure 136:** Fish community DO TIVs for Stony Creek, compared to unimpaired streams (* see table 21 for acronym descriptions). * See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

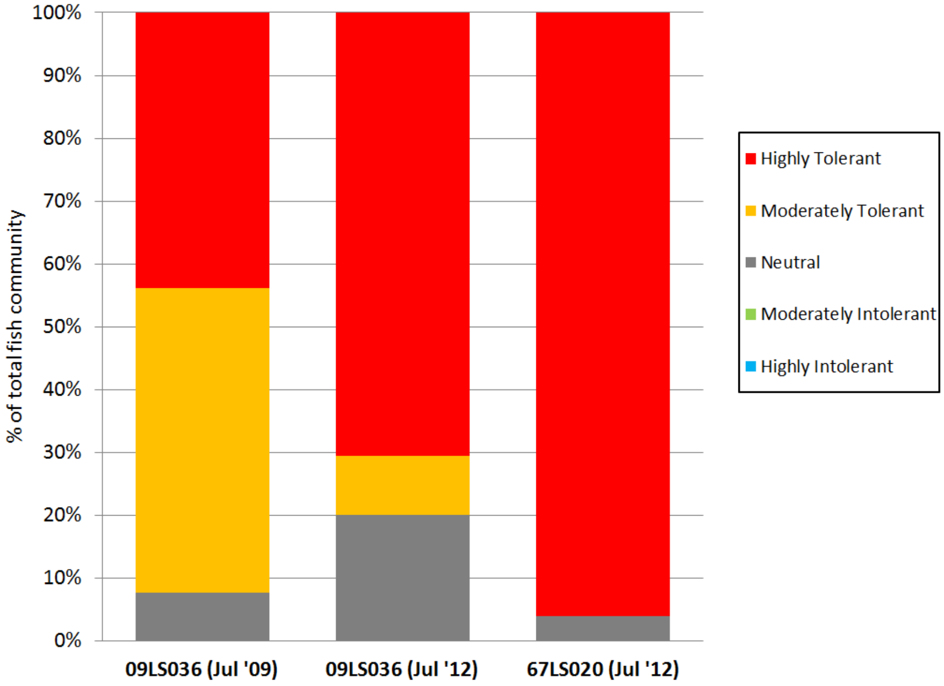


**Figure 137:** Fish community tolerance to low dissolved oxygen in Stony Creek based on taxon specific TIV values

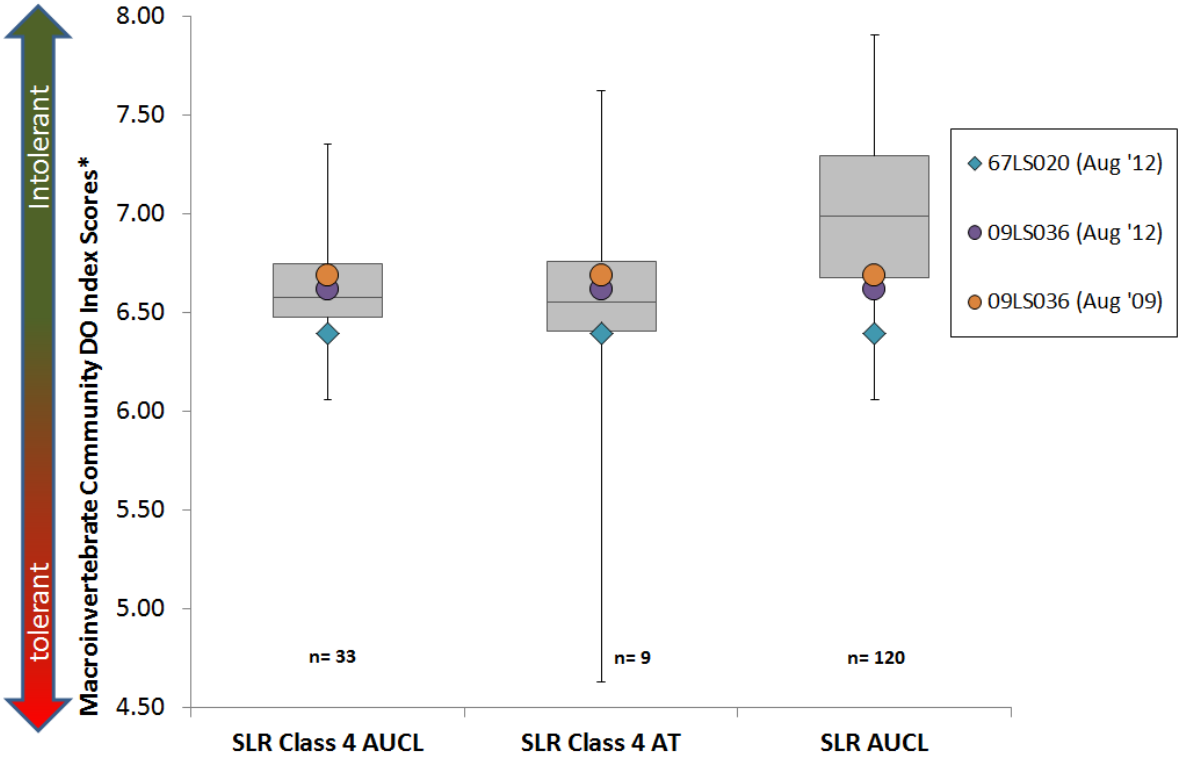


**Figure 138:** Macroinvertebrate community DO TIVs for Stony Creek, compared to unimpaired streams * See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL** = Above Upper Confidence Limit of FIBI threshold **AT** = Above Fish IBI Threshold

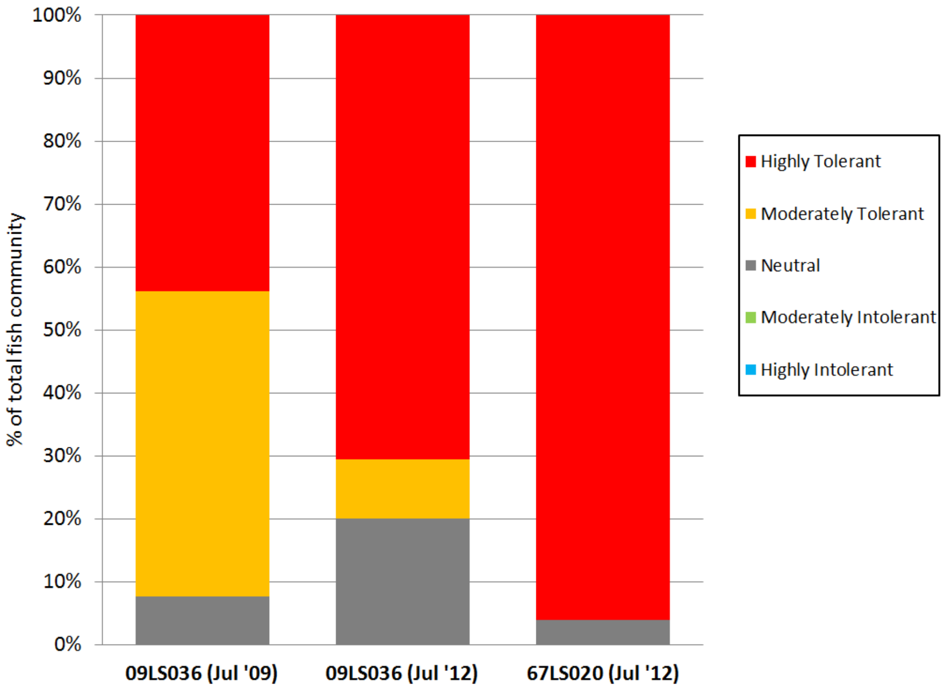


**Figure 137:** Fish community tolerance to low DO in Stony Creek based on taxon specific TIV values

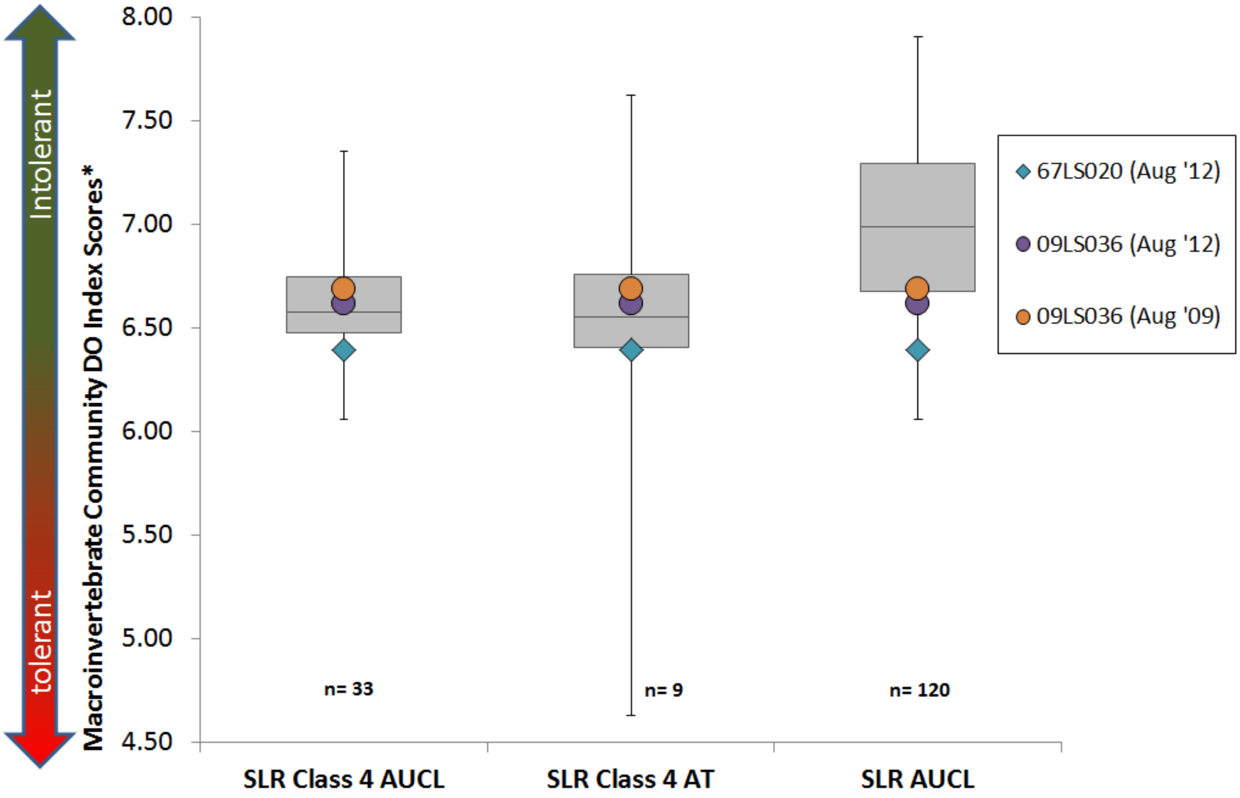


**Figure 138:** Macroinvertebrate community DO TIVs for Stony Creek, compared to unimpaired streams (* see Table 21 for acronym descriptions). ***** See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

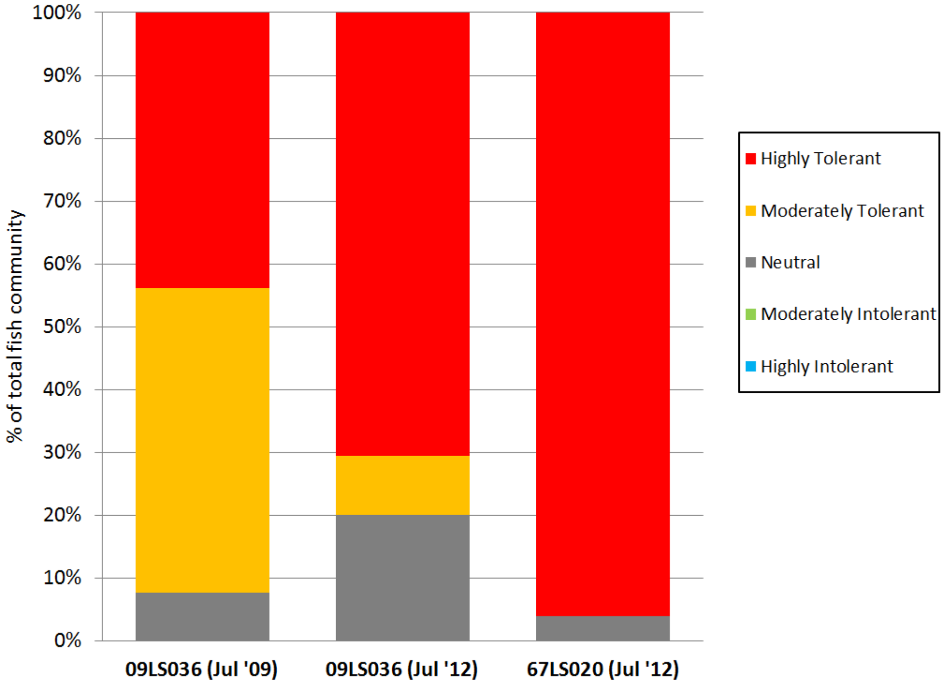


**Figure 137:** Fish community tolerance to low dissolved oxygen in Stony Creek based on taxon specific TIV values

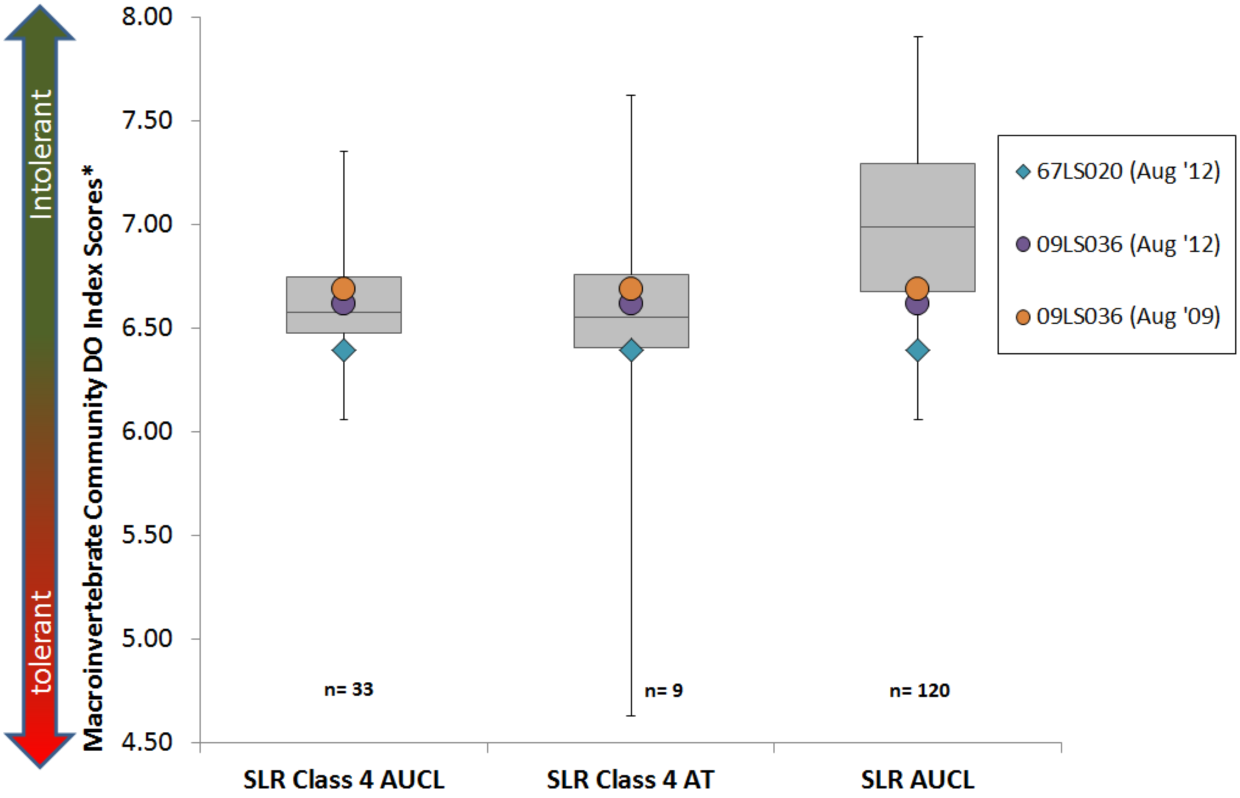


**Figure 138:** Macroinvertebrate community DO TIVs for Stony Creek, compared to unimpaired streams * See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL** = Above Upper Confidence Limit of FIBI threshold **AT** = Above Fish IBI Threshold

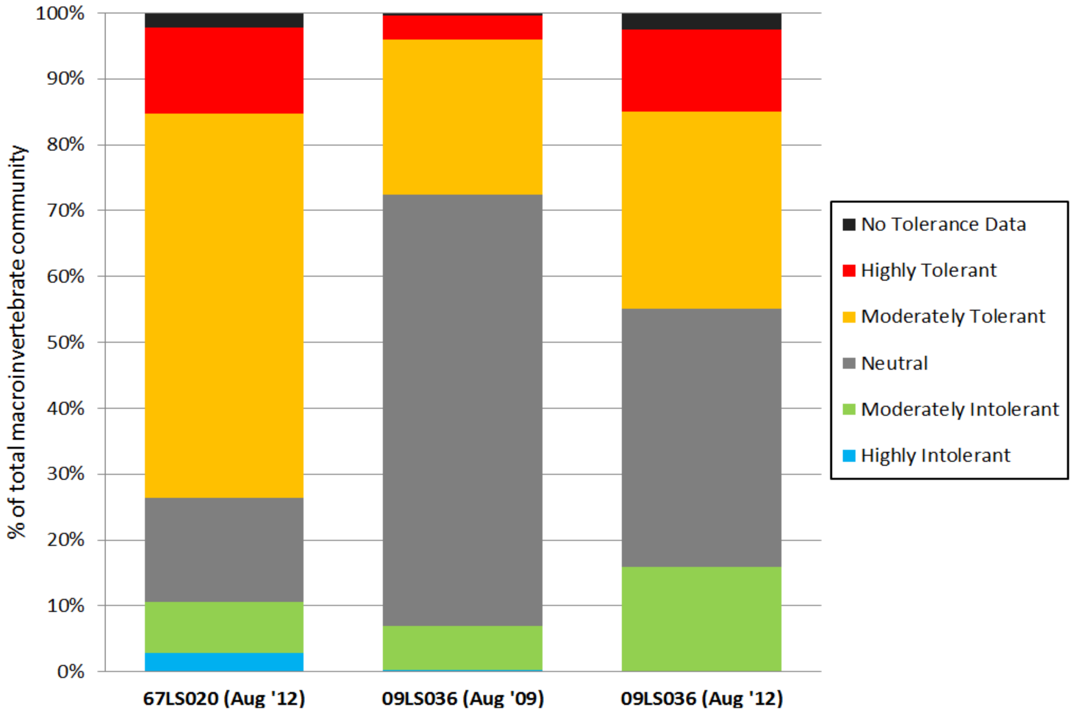


**Figure 140:** Invertebrate community tolerance to low DO in Stony Creek based on taxon specific TIV values

## 5.9.2 Total Suspended Solids & Turbidity

The TSS and Secchi transparency tube (s-tube) data were collected over two years (2012-2013) at station S007-052 on Stony Creek (co-located with biological monitoring station 09LS036). A summary of the data, including the rate at which samples exceeded water quality standards is provided in Table 47. Results from this monitoring station violated the TSS and s-tube standards in 10 out of 14 samples (71.4%). The average TSS was 15.4 mg/L, which is only slightly above the TSS standard of 15 mg/L for Class 2B streams. The average Secchi tube reading of 31.4 cm also violated the draft standard.

**Table 47:** TSS and Secchi tube average values and percent standard exceedances for Stony Creek

| Sites | Site Description | TSS Average (mg/L) | TSS % Exceeding Draft Standard | Secchi Tube Average | Secchi Tube % Exceeding Draft Standard | Total # of Samples | Total % Exceeding Draft Standard |
|---|---|---|---|---|---|---|---|
| S007-052 | STONY CK AT CSAH-83 | 15.4 | 57.1% | 31.4 | 85.7% | 14 | 71.4% |

The TSS and Secchi tube datasets for Stony Creek show a departure from the "A" and "B" reference streams in the SLRW (Figure 142 and 143). Both datasets are poorer than the values for the reference streams, but only the 75th percentile and max TSS values exceed the draft standard of 15 mg/L for TSS. However, even the 25th percentile Secchi tube reading falls below the 40 cm draft standard for Class 2B waters. Evidently water clarity violates the draft standard more frequently than suspended solids, indicating a high level of DOM. The site used in this analysis is technically in violation of the 15 mg/L draft TSS standard frequently enough to be considered "impaired" for TSS, but a closer look at the biological data is needed to verify whether TSS is actually a stressor to fish and macroinvertebrate communities in Stony Creek.

### Seasonal variation in total suspended solids

Stony Creek is opaque in appearance throughout the year. During rains and snowmelt its waters are muddy and during low flows the water is heavily tannin-stained. There is no discernible seasonal trend in TSS due to the low number of dated TSS samples.

**Figure 141:** Stony Creek showing muddiness during snowmelt (left) and its tannin-stained appearance at low flow (right)

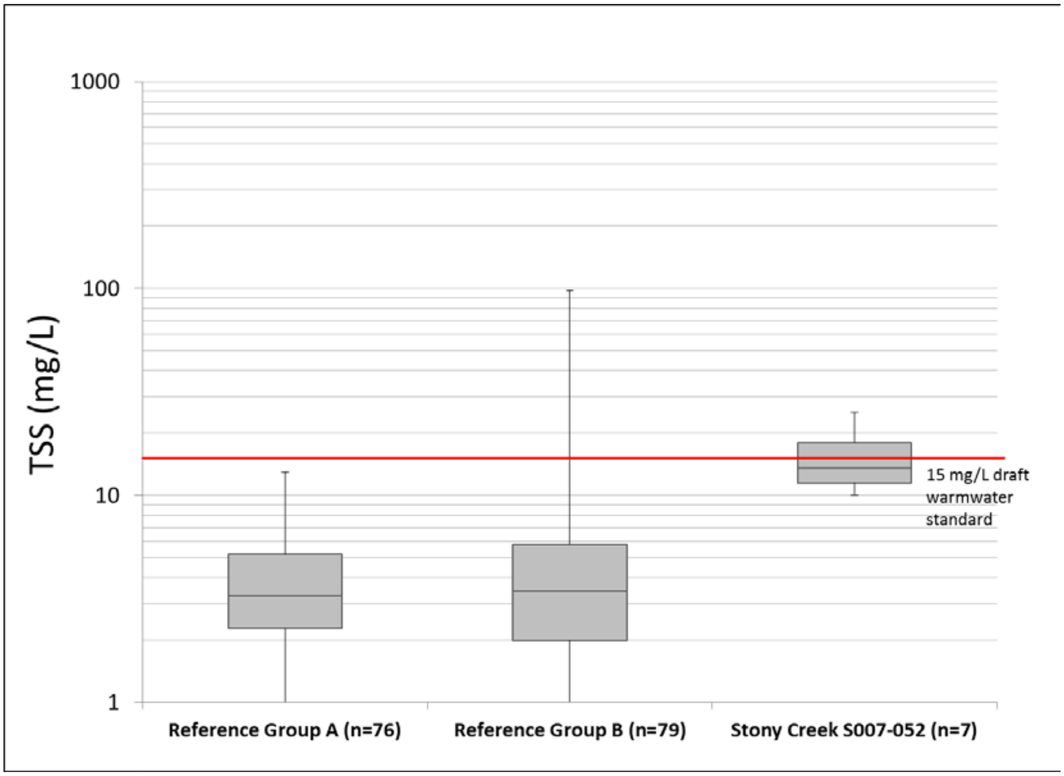


**Figure 142:** Box plots of TSS values for Stony Creek and reference streams. (see section 1.2.3 for reference streams)

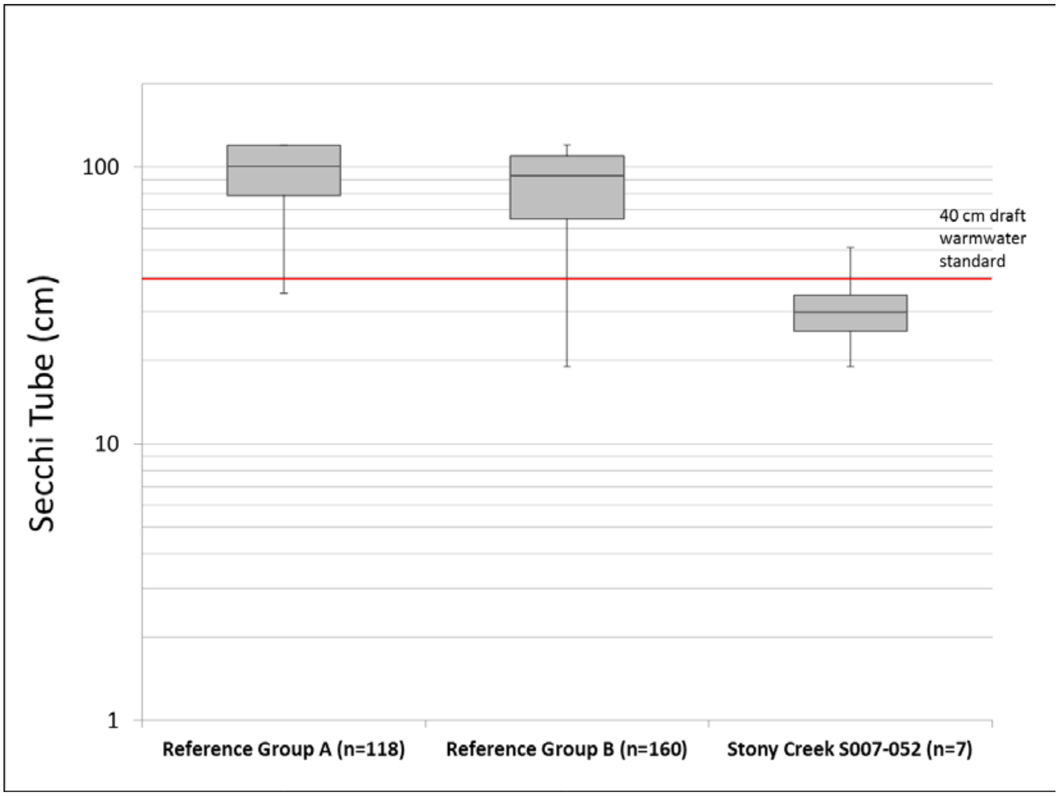


**Figure 143:** Box plots of Secchi tube values for Stony Creek and reference streams. (see section 1.2.3 for reference streams)

## Sources and pathways of suspended solids in the Stony Creek Watershed

### Ditching/Channelization

Channel straightening and meander bend removal result in a shortening of the channel, which causes the water slope to increase and the velocity of channel-forming discharge to increase. If grade control (culverts, bedrock, etc.) is absent, channel incision will often follow – delivering sediment to the stream from the bed and banks as channel evolution progresses. Most of Stony Creek's tributaries have been channelized or straightened (Figure 144). In most instances the streams seem to have been ditched to allow better drainage of the peat bogs that dominate the watershed, which lies entirely within the Glacial Lake Upham basin.

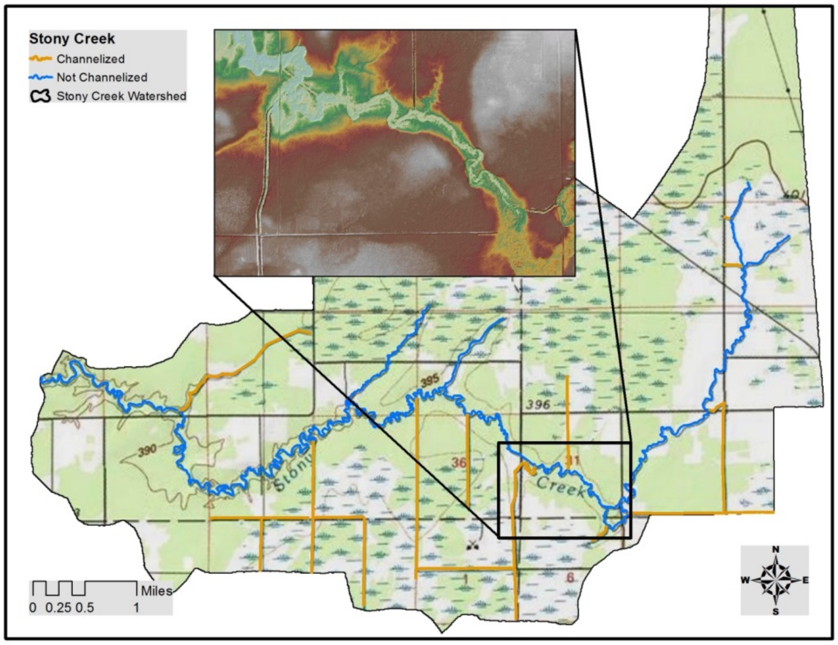


**Figure 144:** Map showing the channelized and straightened reaches in the Stony Creek Watershed

### Channel Instability/Bank Erosion

Areas of debris jams and other indications of channel instability were observed at the 09LS036 biological monitoring station (Figure 145). Channel instability in the Stony Creek Watershed is likely a consequence of increased peak flows as a result of efficient landscape drainage (ditching). Another possible pathway for channel instability may have been a local base level drop in the St. Louis River that caused a headcut to migrate up through the Stony Creek Watershed. More research is needed to determine whether historic landscape alterations (mass logging, ditching, development, etc.) in the SLRW caused peak flows in the St. Louis River to increase. If so, was incision of the St. Louis River a response to these alterations? The lacustrine sediments of the Glacial Lake Upham basin would probably not have been very resistant to downcutting forces. If the main stem St. Louis River incised in the recent past, it would have caused headcuts to migrate up every tributary that fed into the downcut reaches of the river. There is some evidence of this, especially in the part of the St. Louis that flows through the Glacial Lake Upham basin (Figure 146), which shows an incised tributary that feeds into the St. Louis River just upstream of Floodwood, Minnesota). Many of the tributaries in this area show evidence of downcutting, with high degrees of incision, V-notch valleys, and sediment deltas at their mouths.

The stream channel of Stony Creek shows signs of lateral migration, which may be leading to increased sediment loadings (Figure 147). Irregular meanders with oxbows and oxbow cutoffs dominate the lower reaches of Stony Creek where the fish and invertebrate impairment is located. This meander pattern is an indicator of lateral instability (Rosgen 2006).

**Figure 145:** Leaning trees and massive debris jams are a sign of channel instability in Stony Creek

**Figure 146:** Incised tributary to the St. Louis River. Note the V-notch valley and sediment delta at the mouth

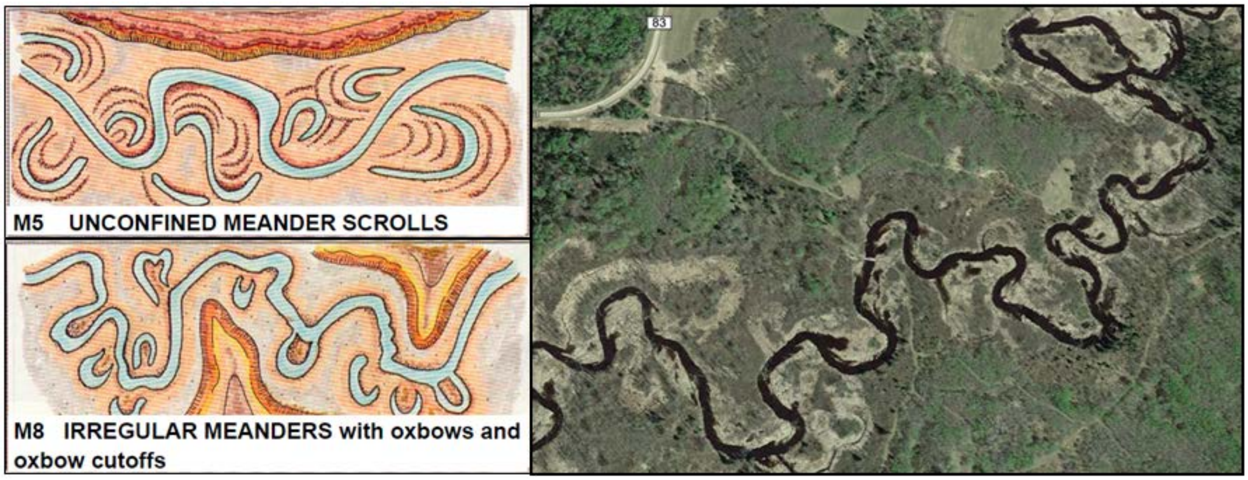


**Figure147: (Left)** Meander patterns that apply to Stony Creek (from Rosgen 2006). **(Right)** Aerial photo of Stony Creek showing irregular meander pattern, oxbows, and oxbow cutoffs (see upper right of photo for large oxbow cutoff)

### Biological effects of elevated TSS

#### Fish Response to TSS

The FIBI impairment on Stony Creek is the result of poor metrics scores related to low fish counts and a lack of species that are expected in healthy headwaters streams (minnow, darter, sculpin sp.). In the 1967 sample of station 67LS020, nearly half of the fish observed were taxa that are considered highly intolerant or moderately intolerant of elevated TSS concentrations. A healthy population of Longnose Dace, and a small number of Mottled Sculpin accounted for this observation. In addition, this sample lacked fish taxa that can be considered tolerant or highly tolerant of TSS. Fish data collected at the same station in 2012 shows a community shift to one that is more tolerant of TSS (Figure 148). Longnose Dace and Mottled Sculpin were not observed in the 2012 sample, and more tolerant species such as Fathead Minnow and Brook Stickleback were observed in their place. Central Mudminnow and Brook Stickleback accounted for nearly 80% of the fish assemblage in 2012. There is no long term record of TSS concentrations or turbidity levels for Stony Creek that can be linked to this fish assemblage change, but clearly, the current fish community is more tolerant of TSS (and other stressors) than the community observed in the late 1960’s.

Fish data from the other biological monitoring station on Stony Creek (09LS036) were only collected in 2009 and 2012. A small population of Pearl Dace, and a single Burbot individual were observed which accounted for the only TSS intolerant species observed at this station. Fish taxa that were prominent in these samples include Central Mudminnow, Johnny Darter, and White Sucker. A small population of fish taxa that are tolerant of TSS (Green Sunfish and Black Bullhead) also showed up in the sample, but the overall fish community at this station can be considered neither tolerant nor intolerant of TSS.

### Invert Response to TSS

Stony Creek was listed as impaired for failing to meet the MIBI criteria at both monitoring stations (67LS020 and 09LS036). Low relative abundance of collector-filterer taxa and a lack of pollution intolerant taxa were two of the primary metrics that contributed to the low MIBI scores at these stations. Both of these metrics have the potential to be negatively influenced by elevated TSS concentrations. Collector-filterer taxa, which obtain food by filtering particles from the water column, have shown to respond negatively to increases in TSS in streams and rivers of northern Minnesota (Markus 2011). Non-Hydropsychid caddisfly taxa were present in relatively low numbers in Stony Creek, which is potentially another measure that has shown a negative response to TSS in streams of northern Minnesota.

The macroinvertebrate community in Stony Creek is more tolerant of TSS in comparison to high quality stations of the same MIBI class. The box plots in Figure 151 compare data for a series of TSS related metrics between Stony Creek monitoring stations and class 4 stations that scored above the UCL of the MIBI threshold (class 4 MIBI AUCL). In both sampling visits to 09LS036, over half of the macroinvertebrate taxa observed are considered tolerant of TSS Figure 150). Both of the Stony Creek monitoring stations exceed 75th percentile values (or greater) for measures of % TSS tolerant and % TSS very tolerant. TSS index scores, which are a composite value of taxa tolerance and relative abundance measures, are clearly showing that the invertebrate assemblage of Stony Creek are more tolerant of TSS than the class 4 MIBI AUCL stations (Figure 151).

**Summary: Is elevated TSS a stressor in Stony Creek?**

Based on the biological and water chemistry data presented in this report, we recommend that TSS be considered a stressor to aquatic life in Stony Creek. Additional water chemisty data would add confidence to this decision, but there is a logical pathway in this watershed connecting source, stressor, and biological response. The primary source of TSS and bedded sediment in this stream appears to be bank erosion caused by channel incision, widening, and bank scour in areas where large debris jams are impeding flow and re-directing currents towards vulnerable banks.

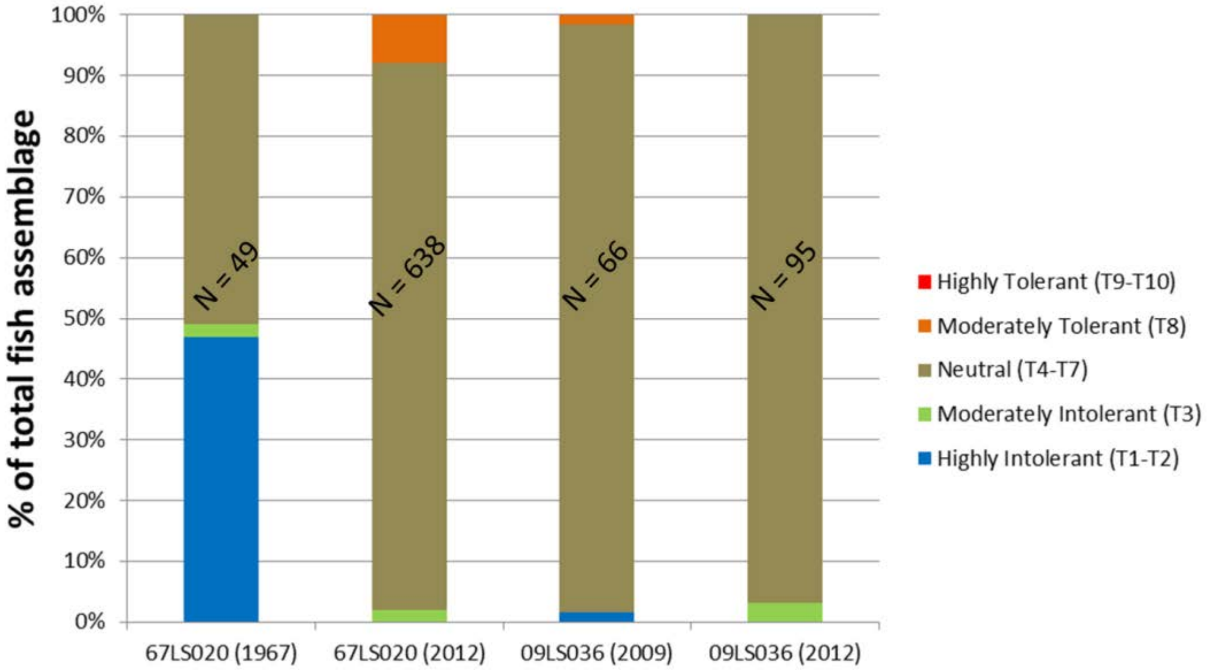


**Figure 148:** TSS tolerance of fish assemblages by site in Stony Creek (n = number of fish)

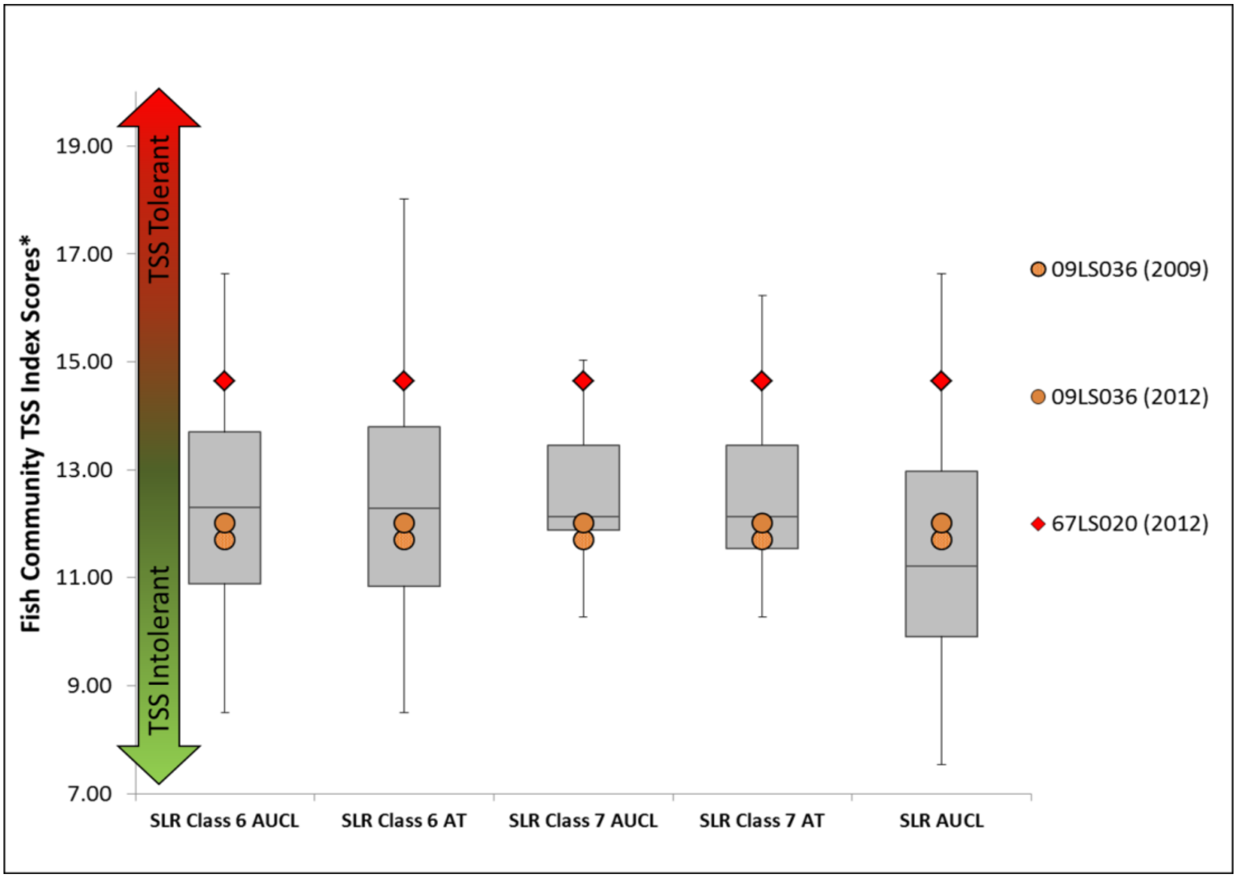


**Figure 149:** Fish community TSS TIVs for Stony Creek, compared to unimpaired streams. * See Section 4 for explanation of TIVs SLR= St. Louis River Watershed AUCL = Above Upper Confidence Limit of FIBI threshold AT = Above Fish IBI Threshold

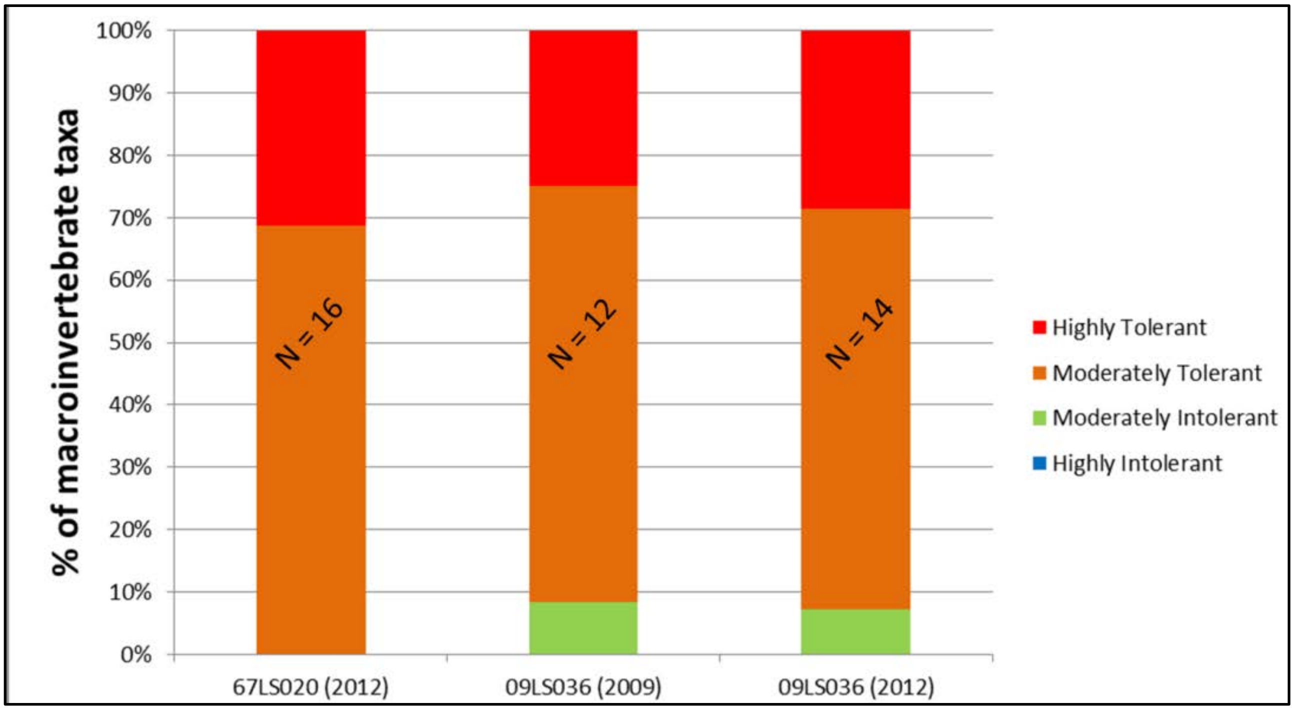


**Figure 150:** TSS tolerance of macroinvertebrate assemblages by site in Stony Creek (n = number of taxa)

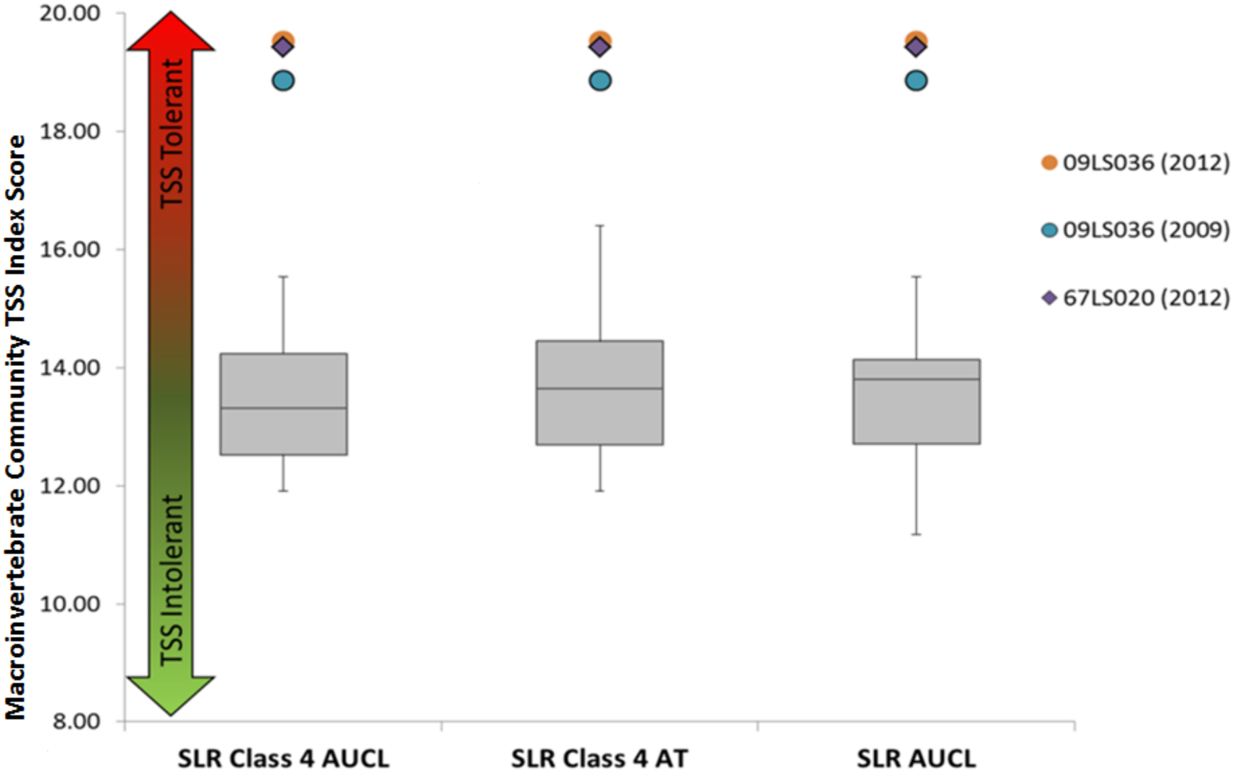


**Figure 151:** Macroinvertebrate community TSS TIVs for Stony Creek, compared to unimpaired streams
* See Section 4 for explanation of TIVs SLR= St. Louis River Watershed AUCL = Above Upper Confidence Limit of FIBI threshold AT = Above Fish IBI Threshold

### 5.9.3 Poor Physical Habitat Conditions

The MSHA methodology was used to assess habitat conditions in Stony Creek at several locations in the summers of 2009 and 2012. Overall MSHA scores were very similar at both monitoring stations (53/54/ 56.5 out of possible 100), and both sites were categorized as having “fair” habitat conditions. Individual habitat metric scores from the two stations show that similar habitat features are lacking at both locations. Specifically, both monitoring stations received poor metric scores related to substrate conditions and channel morphology (Figure 152). The MSHA scores related to substrate, cover, and channel morphology observed at Stony Creek monitoring stations were low compared to high quality streams of the same FIBI class (Figure 153 and 154). Several positive habitat attributes were observed at the two monitoring stations as well, including a healthy riparian corridor and local-land uses that do not result in considerable impacts to the stream.

The low gradient nature of Stony Creek provides minimal shallow, fast-water microhabitats such as riffle and glide features. Both of the monitoring sites on Stony Creek lacked riffle and glide habitats and were dominated by run and pool features, which become predominantly slack-water areas during normal to lower flow conditions. Given that riffle and glide areas are considered areas of high biological productivity, the lack of these features in Stony Creek may be contributing to the low taxa richness and abundance observed in fish and macroinvertebrate community of this stream.

The PSI was also used to evaluate physical habitat and channel stability along Stony Creek. The results observed were typical of many of the incised streams in the Glacial Lake Upham basin, with unfavorable characteristics such as bank erosion, loose bottom sediments, and debris jams. The overall PSI score of 104 at biological monitoring station 09LS036 corresponds to a rating of “unstable” for the potential E6 stream type. The PSI scores from the biological monitoring station were the lowest in the various metrics related to the condition of the lower banks and stream bottom. The upper bank metric scores were quite good due to the undisturbed riparian corridor and the presence of an established forest with many mature trees.

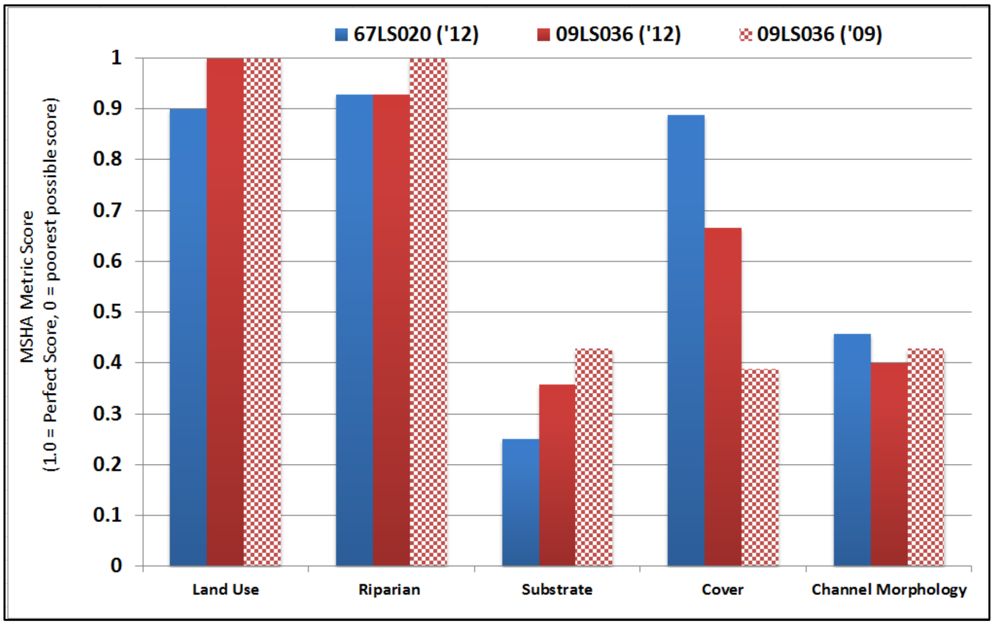


**Figure 152:** Summary of key MSHA habitat metric results related to fish habitat

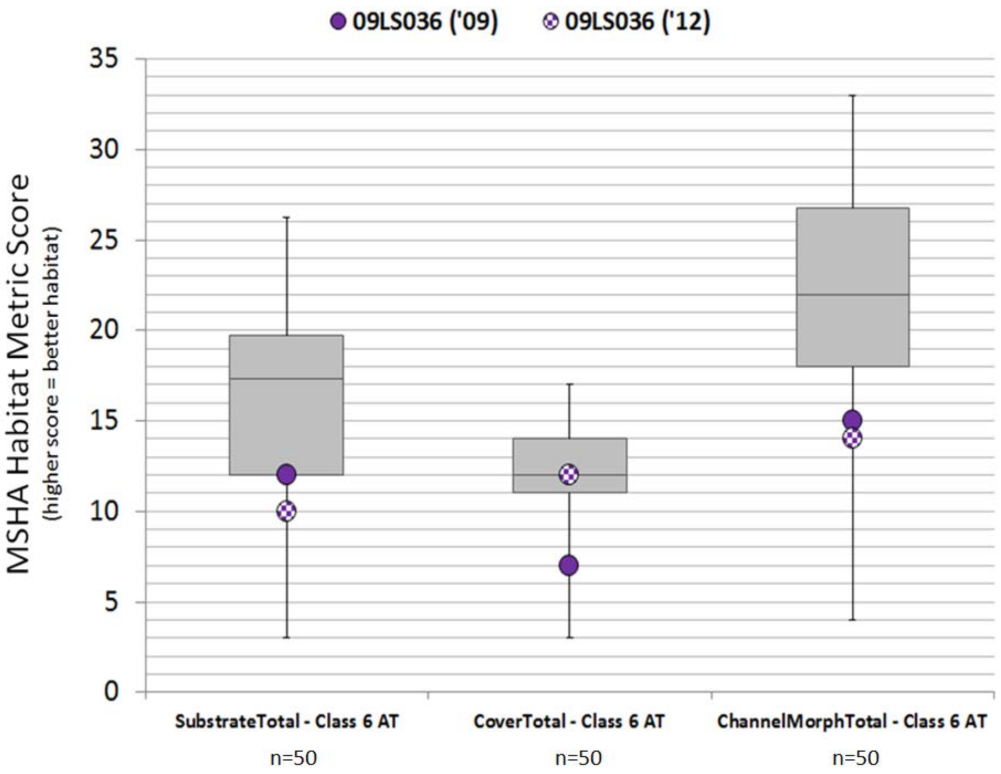


**Figure 153:** MSHA habitat metric scores for Stony Creek station 09LS036 compared to high quality stations of the same IBI class. **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

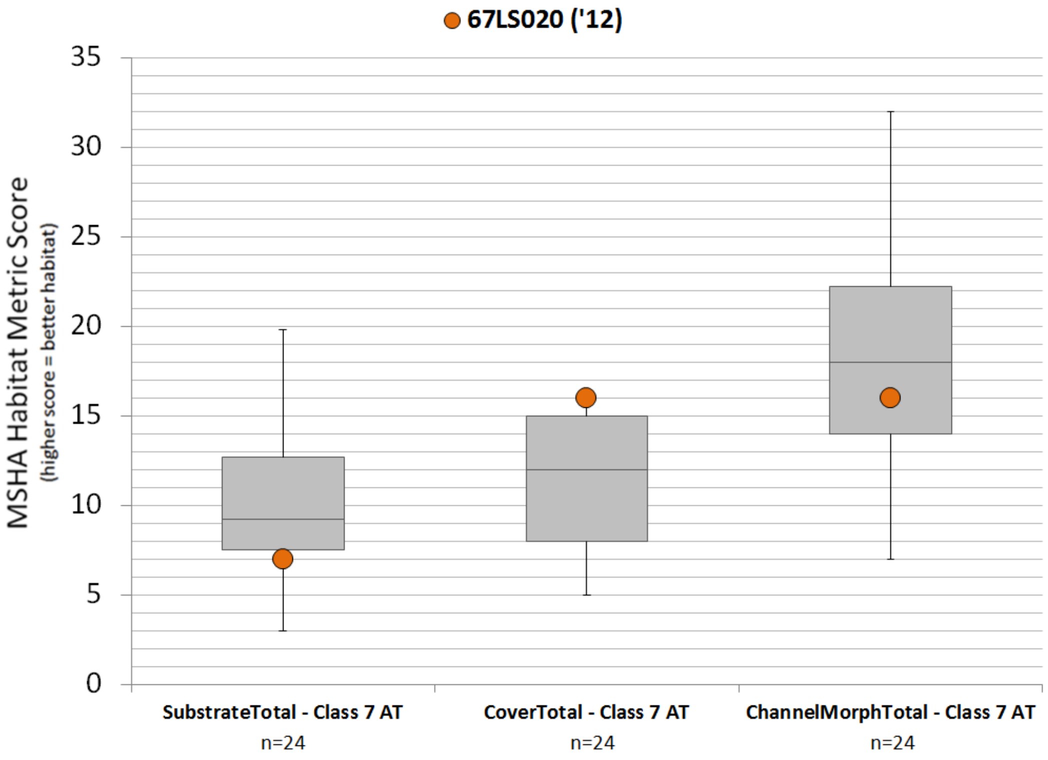
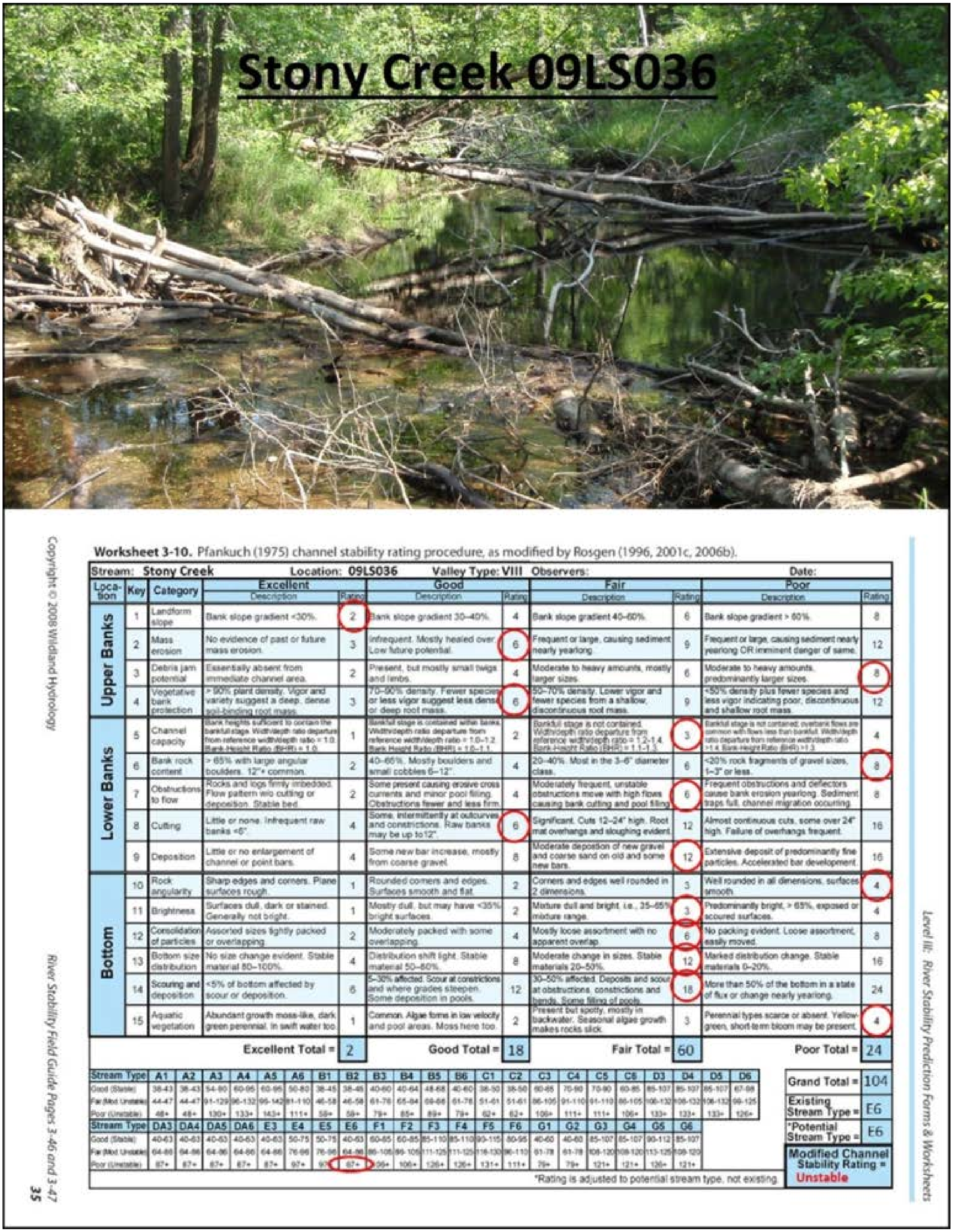


**Figure 154:** MSHA habitat metric scores for Stony Creek station 67LS020 compared to high quality stations of the same IBI class. **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold



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**Worksheet 3-10.** Pfankuch (1975) channel stability rating procedure, as modified by Rosgen (1996, 2001c, 2006b).

**Stream:** Stony Creek **Location:** 09LS036 **Valley Type:** VIII **Observers:** **Date:**

| Location | Key | Category | Excellent |  | Good |  | Fair |  | Poor |  |
|---|---|---|---|---|---|---|---|---|---|---|
|  |  |  | Description | Rating | Description | Rating | Description | Rating | Description | Rating |
| Upper Banks | 1 | Landform slope | Bank slope gradient <30%. | 2 | Bank slope gradient 30–40%. | 4 | Bank slope gradient 40–60%. | 6 | Bank slope gradient > 60%. | 8 |
|  | 2 | Mass erosion | No evidence of past or future mass erosion. | 3 | Infrequent. Mostly healed over. Low future potential. | 6 | Frequent or large, causing sediment nearly yearlong. | 9 | Frequent or large, causing sediment nearly yearlong OR imminent danger of same. | 12 |
|  | 3 | Debris jam potential | Essentially absent from immediate channel area. | 2 | Present, but mostly small twigs and limbs. | 4 | Moderate to heavy amounts, mostly larger sizes. | 6 | Moderate to heavy amounts, predominantly larger sizes. | 8 |
|  | 4 | Vegetative bank protection | > 90% plant density. Vigor and variety suggest a deep, dense soil-binding root mass. | 3 | 70–90% density. Fewer species or less vigor suggest less dense or deep root mass. | 6 | 50–70% density. Lower vigor and fewer species from a shallow, discontinuous root mass. | 9 | <50% density plus fewer species and less vigor indicating poor, discontinuous and shallow root mass. | 12 |
| Lower Banks | 5 | Channel capacity | Bank heights sufficient to contain the bankfull stage. Width/depth ratio departure from reference width/depth ratio = 1.0. Bank-Height Ratio (BHR) = 1.0. | 1 | Bankfull stage is contained within banks. Width/depth ratio departure from reference width/depth ratio = 1.0–1.2. Bank-Height Ratio (BHR) = 1.0–1.1. | 2 | Bankfull stage is not contained. Width/depth ratio departure from reference width/depth ratio = 1.2–1.4. Bank-Height Ratio (BHR) = 1.1–1.3. | 3 | Bankfull stage is not contained; overbank flows are common with flows less than bankfull. Width/depth ratio departure from reference width/depth ratio >1.4. Bank-Height Ratio (BHR) >1.3. | 4 |
|  | 6 | Bank rock content | > 65% with large angular boulders. 12"+ common. | 2 | 40–65%. Mostly boulders and small cobbles 6–12". | 4 | 20–40%. Most in the 3–6" diameter class. | 6 | <20% rock fragments of gravel sizes, 1–3" or less. | 8 |
|  | 7 | Obstructions to flow | Rocks and logs firmly imbedded. Flow pattern w/o cutting or deposition. Stable bed. | 2 | Some present causing erosive cross currents and minor pool filling. Obstructions fewer and less firm. | 4 | Moderately frequent, unstable obstructions move with high flows causing bank cutting and pool filling. | 6 | Frequent obstructions and deflectors cause bank erosion yearlong. Sediment traps full, channel migration occurring. | 8 |
|  | 8 | Cutting | Little or none. Infrequent raw banks <6". | 4 | Some, intermittently at outcurves and constrictions. Raw banks may be up to 12". | 6 | Significant. Cuts 12–24" high. Root mat overhangs and sloughing evident. | 12 | Almost continuous cuts, some over 24" high. Failure of overhangs frequent. | 16 |
|  | 9 | Deposition | Little or no enlargement of channel or point bars. | 4 | Some new bar increase, mostly from coarse gravel. | 8 | Moderate deposition of new gravel and coarse sand on old and some new bars. | 12 | Extensive deposit of predominantly fine particles. Accelerated bar development. | 16 |
| Bottom | 10 | Rock angularity | Sharp edges and corners. Plane surfaces rough. | 1 | Rounded corners and edges. Surfaces smooth and flat. | 2 | Corners and edges well rounded in 2 dimensions. | 3 | Well rounded in all dimensions, surfaces smooth. | 4 |
|  | 11 | Brightness | Surfaces dull, dark or stained. Generally not bright. | 1 | Mostly dull, but may have <35% bright surfaces. | 2 | Mixture dull and bright, i.e., 35–65% mixture range. | 3 | Predominantly bright, > 65%, exposed or scoured surfaces. | 4 |
|  | 12 | Consolidation of particles | Assorted sizes tightly packed or overlapping. | 2 | Moderately packed with some overlapping. | 4 | Mostly loose assortment with no apparent overlap. | 6 | No packing evident. Loose assortment, easily moved. | 8 |
|  | 13 | Bottom size distribution | No size change evident. Stable material 80–100%. | 4 | Distribution shift light. Stable material 50–80%. | 8 | Moderate change in sizes. Stable materials 20–50%. | 12 | Marked distribution change. Stable materials 0–20%. | 16 |
|  | 14 | Scouring and deposition | <5% of bottom affected by scour or deposition. | 6 | 5–30% affected. Scour at constrictions and where grades steepen. Some deposition in pools. | 12 | 30–50% affected. Deposits and scour at obstructions, constrictions and bends. Some filling of pools. | 18 | More than 50% of the bottom in a state of flux or change nearly yearlong. | 24 |
|  | 15 | Aquatic vegetation | Abundant growth moss-like, dark green perennial. In swift water too. | 1 | Common. Algae forms in low velocity and pool areas. Moss here too. | 2 | Present but spotty, mostly in backwater. Seasonal algae growth makes rocks slick. | 3 | Perennial types scarce or absent. Yellow-green, short-term bloom may be present. | 4 |

Excellent Total = 2

Good Total = 18

Fair Total = 60

Poor Total = 24

| Stream Type | A1 | A2 | A3 | A4 | A5 | A6 | B1 | B2 | B3 | B4 | B5 | B6 | C1 | C2 | C3 | C4 | C5 | C6 | D3 | D4 | D5 | D6 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Good (Stable) | 38-43 | 38-43 | 54-90 | 60-95 | 60-95 | 50-80 | 38-45 | 38-45 | 40-60 | 40-64 | 48-68 | 40-60 | 38-50 | 38-50 | 60-85 | 70-90 | 70-90 | 60-85 | 85-107 | 85-107 | 85-107 | 67-98 |
| Fair (Mod. Unstable) | 44-47 | 44-47 | 91-129 | 96-132 | 96-142 | 81-110 | 46-58 | 46-58 | 61-78 | 65-84 | 69-88 | 61-78 | 51-61 | 51-61 | 86-105 | 91-110 | 91-110 | 86-105 | 108-132 | 108-132 | 108-132 | 99-125 |
| Poor (Unstable) | 48+ | 48+ | 130+ | 133+ | 143+ | 111+ | 59+ | 59+ | 79+ | 85+ | 89+ | 79+ | 62+ | 62+ | 106+ | 111+ | 111+ | 106+ | 133+ | 133+ | 133+ | 126+ |

| Stream Type | DA3 | DA4 | DA5 | DA6 | E3 | E4 | E5 | E6 | F1 | F2 | F3 | F4 | F5 | F6 | G1 | G2 | G3 | G4 | G5 | G6 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Good (Stable) | 40-63 | 40-63 | 40-63 | 40-63 | 40-63 | 50-75 | 50-75 | 40-63 | 60-85 | 60-85 | 85-110 | 85-110 | 90-115 | 80-95 | 40-60 | 40-60 | 85-107 | 85-107 | 90-112 | 85-107 |
| Fair (Mod. Unstable) | 64-86 | 64-86 | 64-86 | 64-86 | 64-86 | 76-96 | 76-96 | 64-86 | 86-105 | 86-105 | 111-125 | 111-125 | 116-130 | 96-110 | 61-78 | 61-78 | 108-120 | 108-120 | 113-125 | 108-120 |
| Poor (Unstable) | 87+ | 87+ | 87+ | 87+ | 87+ | 97+ | 97+ | 87+ | 106+ | 106+ | 126+ | 126+ | 131+ | 111+ | 79+ | 79+ | 121+ | 121+ | 126+ | 121+ |

Grand Total = 104

Existing Stream Type = E6

*Potential Stream Type = E6

Modified Channel Stability Rating = Unstable

*Rating is adjusted to potential stream type, not existing.

River Stability Field Guide Pages 3-46 and 3-47

35

Level III: River Stability Prediction Forms & Worksheets

**Figure 155:** Pfankuch rating for Stony Creek 09LS036

### Biological effects of degraded habitat

#### Fish Response

Poor physical habitat conditions can limit the abundance and taxa richness of non-tolerant fish species (source). In general, the fish community of Stony Creek exhibited low taxa richness and low fish counts of non-tolerant species. The number of non-tolerant fish/meter at 09LS036 was well below the rate observed at high quality stations of the same FIBI class (Figure 156). In the 2009 visit, 5 non-tolerant species and 43 individuals were sampled – Blacknose Shiner (1), Burbot (1), Johnny Darter (31), Northern Pike (2), and Yellow Perch (8). The number of non-tolerant fish/meter during the ’09 visit was 0.19 – less than 75% of the Class 6 streams that were above the IBI UCL and above the IBI threshold. The number of non-tolerant fish/meter during the 2012 visit to the station was much less, at 0.08. Five non-tolerant species were present, but accounted for only 22 individual fish. The fish community in 2012 was dominated by tolerant species such as Central Mudminnow and White Sucker. The limited riffle, glide, and pool habitat within the impaired reach is likely a significant factor contributing to the lack of diversity and abundance of non-tolerant fish species

The fish communities observed at the two Stony Creek monitoring sites were fairly similar in terms of the number of habitat-sensitive taxa. The results from both years were at or below 25 percentile values pulled from high quality reference stations in 4 of the 5 metrics (Figure 157). One benthic insectivore and darter species (Johnny Darter) was collected in 2009. None were present in the follow-up 2012 visit. One riffle-dwelling species (White Sucker) and two gravel spawning species (Burbot and White Sucker) were found in both years at this site. Piscivorous taxa were the only group in Stony Creek that rated high in comparison to the class 6 reference streams. Piscivores are somewhat rare in many Class 6 streams (the median is one species), but two piscivorous species were found at 09LS036 in both years. It is possible the significant large woody debris in the channel is providing quality cover for these fish, increasing their numbers and diversity. The poor numbers of taxa in the other four categories are likely a result of the absence of velocity variability and dominance of fine substrate in the stream.

#### Macroinvertebrate Response

In many cases, streams with degraded habitat favor macroinvertebrate taxa that do not require stable benthic habitats with clean, coarse material. Instead, organisms that can suspend themselves on top of fine substrate (“sprawlers”), or burrow into the substrate as part of their life-cycle (“burrowers”) tend to have a competitive advantage in these streams. “Legless” organisms such as aquatic worms, midge larvae, clams, and snails also can dominate systems with degraded habitat and poor substrate conditions. Biological metrics covering these attributes were evaluated to investigate a physical habitat stressor in Stony Creek.

Samples collected from Stony Creek show a high percentage of macroinvertebrate taxa that are considered sprawlers, burrowers, and legless compared to class 4 reference stations (Figure 158-160). However, the overall proportion of individual organisms with these traits seems to be fairly similar to high quality stations in most samples. Macroinvertebrate taxa and individuals with a higher tolerance for marginal substrate conditions were more prominent at most stations during the 2012 sampling event. The 2009 data compare more favorably to the results from high quality class 4 MIBI stations. This could be related to water conditions in 2012, which brought two major flood events in the early summer months, and sustained drought in the late summer and early fall season. Overall, it is hard to distinguish whether or not the metric data show a consistent response to marginal habitat. The abundance of wood debris jams in Stony Creek may have a role in sustaining population of certain macroinvertebrate taxa that require hard substrates for survival and propagation. Other streams in the area with substrate conditions similar to Stony Creek were dominated by sprawler, legless, and burrower taxa to a much greater extent (e.g. Vaara Creek, Sand Creek). These streams generally had fewer debris jams and less "snag" habitats.

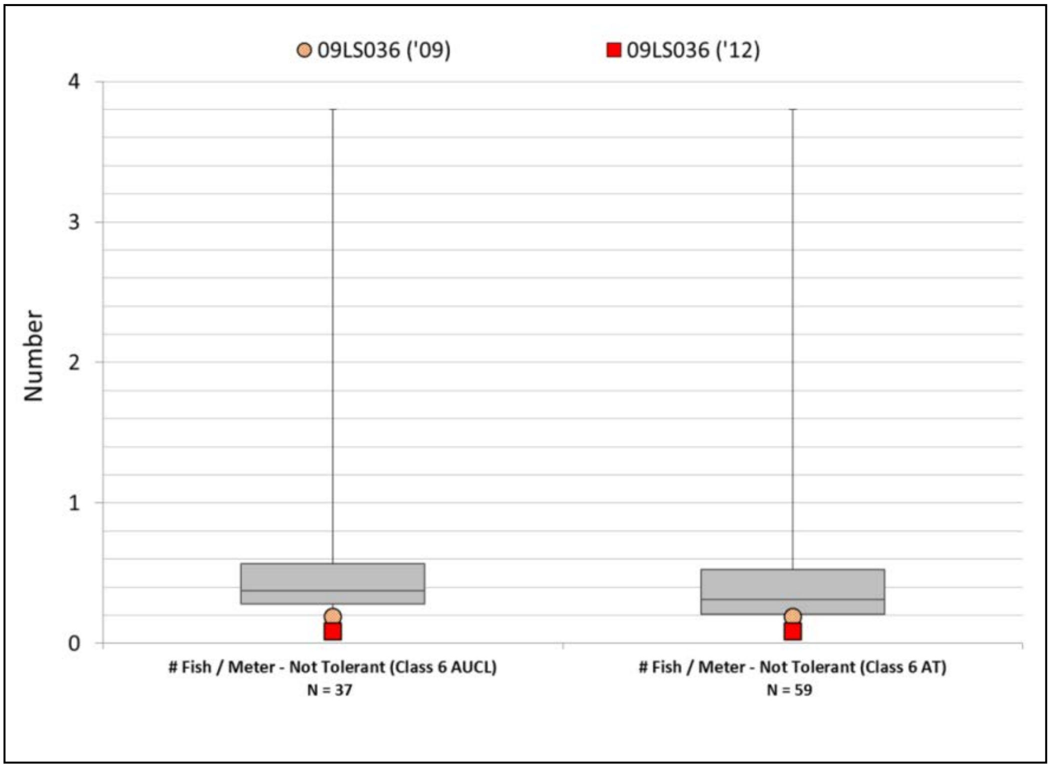


**Figure 156:** Non-tolerant fish/meter in Stony Creek compared to high quality Class 6 SLRW stations AUCL = Above Upper Confidence Limit of FIBI threshold AT = Above Fish IBI Threshold

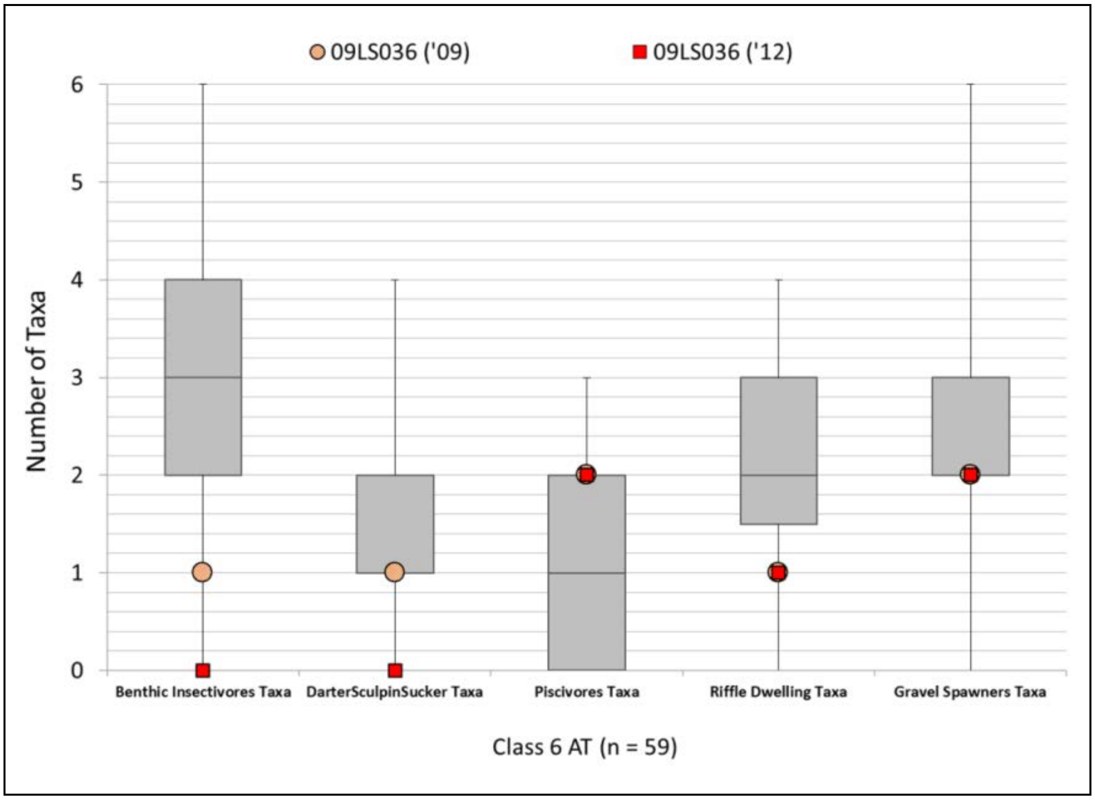


**Figure 157:** Number of fish taxa for various categories in Stony Creek compared to Class 6 SLRW stations above the impairment threshold. AT = Above Fish IBI Threshold

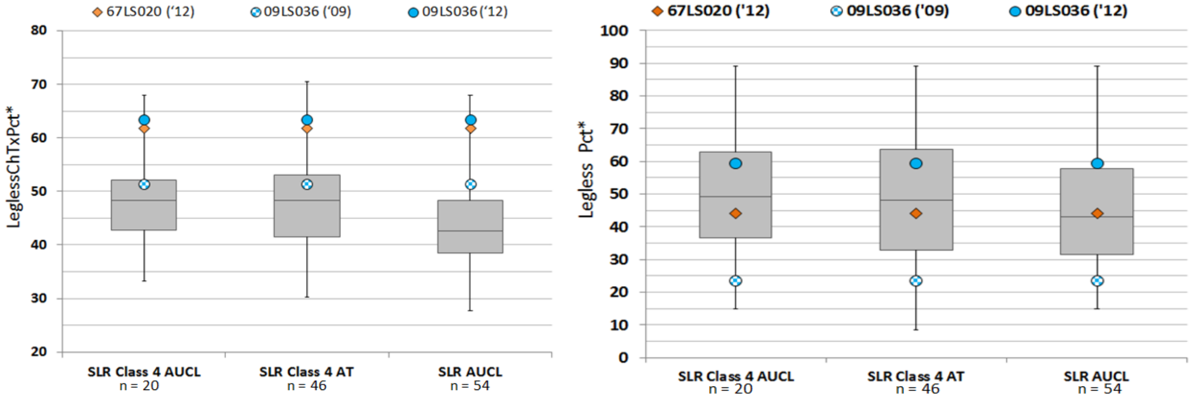


**Figure 158:** Stony Creek metric results compared to high quality reference stations. * See Section 4 for explanation of TIVs SLR= St. Louis River Watershed AUCL = Above Upper Confidence Limit of FIBI threshold AT = Above Fish IBI Threshold

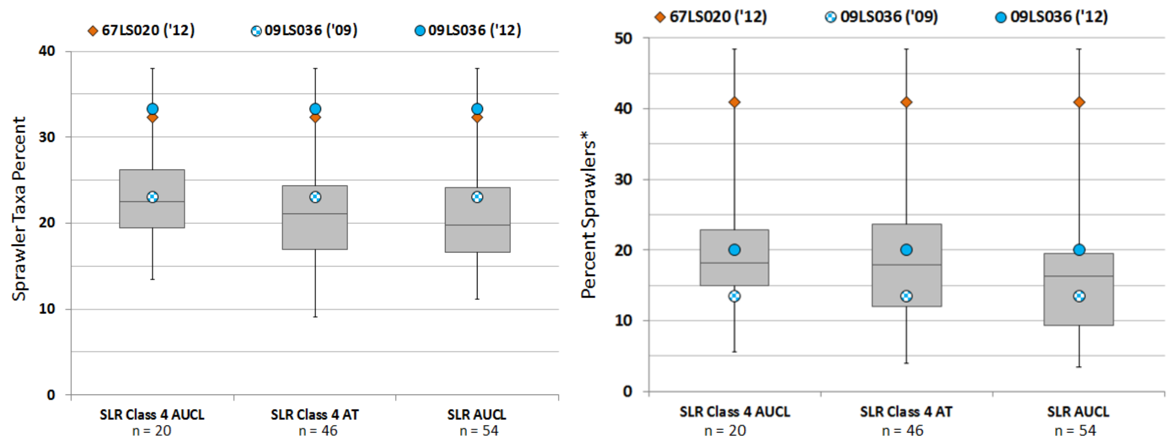


**Figure 159:** Stony Creek metric results compared to high quality reference stations. . * See Section 4 for explanation of TIVs SLR= St. Louis River Watershed AUCL = Above Upper Confidence Limit of FIBI threshold AT = Above Fish IBI Threshold

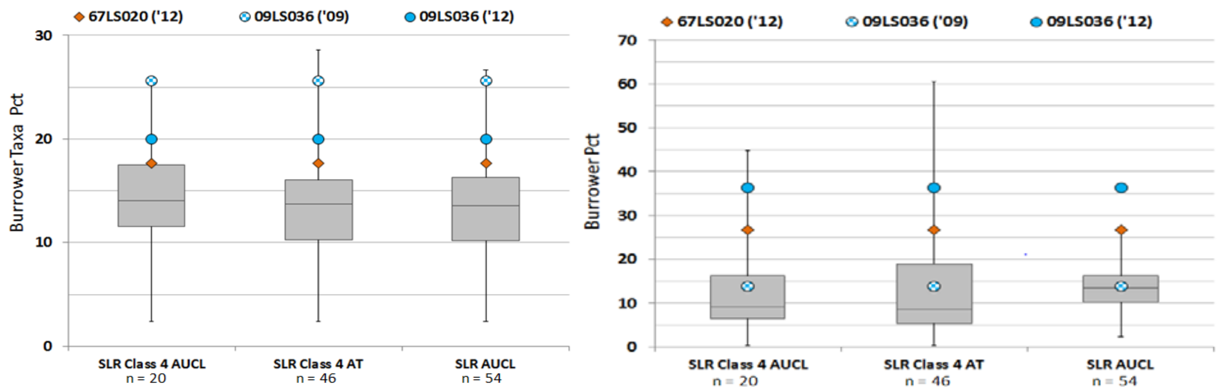


**Figure 160:** Stony Creek metric results compared to high quality reference stations. * See Section 4 for explanation of TIVs SLR= St. Louis River Watershed AUCL = Above Upper Confidence Limit of FIBI threshold AT = Above Fish IBI Threshold

### Summary: Are poor physical habitat conditions a stressor in Stony Creek?

Degraded habitat conditions in Stony Creek can partially be attributed to channel instability and resulting sedimentation. The poor substrate and channel morphology scores observed in the MSHA results and the "unstable" Pfankuch Stability rating support this claim. Marginal habitat conditions in Stony Creek are contributing to the low number of non-tolerant fish and the low number of various habitat-sensitive taxa present. It is our conclusion that habitat degradation is a stressor to the fish community in Stony Creek.

The effects of habitat degradation on the macroinvertebrate community are a little harder to decipher due to some variability in the biological-response data. Macroinvertebrate taxa with attributes favorable for withstanding marginal habitat conditions (e.g. sprawlers, burrowers, etc.) were found to be dominant at several Stony Creek sampling locations. In many cases, however, the number of individuals representing these taxa was fairly low in abundance and did not account for the majority of the overall population. The lack of riffle and glide habitats in Stony Creek are likely limiting overall macroinvertebrate taxa richness, but the amount of woody debris in the channel may be making up for the lack of substrate heterogeneity by providing habitat for organisms that prefer hard surfaces.

Poor habitat conditions should be considered a stressor in Stony Creek. Further monitoring is suggested to identify areas of channel incision, bank erosion, channel widening, and debris jams that are altering the natural course of the stream channel. Due to the local geology and low relief of the landscape, Stony Creek may never provide ideal habitat for aquatic organisms that prefer swift current and coarse substrates. Restoring the proper pattern, profile, and dimensions of the stream channel in specific areas and partial removal some of the major debris jams will improve habitat conditions for aquatic life.

### 5.9.4 Stony Creek: Summary of Stressors to Aquatic Life

**Table 48:** Summary of SID results for Stony Creek, along with recommendations for restoration and implementation projects.

| Candidate Cause | Result |
|---|---|
| **Low Dissolved Oxygen** | ● |
| **TSS / Turbidity** | ● |
| **Poor Physical Habitat Conditions** | ● |
| **Altered Hydrology** | ○ |
| Key: ● **= confirmed stressor** ○ **= Potential Stressor X = eliminated candidate cause** | |

## 5.10 Unnamed Tributary to St. Louis River

Much like other nearby streams, this unnamed tributary to the St. Louis River originates in the extensive bog that covers much of the Meadowlands region. The headwaters were ditched in an attempt to drain the bog, but the lower reaches retain a natural pattern and sinuosity. The impaired AUID is a 0.3% C channel in an alluvial valley. Based on visual observations, the channel below County Road 52 is in poor condition and is somewhat incised with erosion and sedimentation issues. Additional field data would need to be collected to confirm whether the channel is a G (gully) or C channel in that location.

The impaired macroinvertebrate community at station 09LS035 of this unnamed tributary was dominated by “legless” taxa such as pill clams (*Pisidiidae*), black fly larvae (*Simulium*), and air breathing snails (*Physa*). Over 70% of the macroinvertebrate taxa represented at this station, and nearly 90% of the individuals sampled are considered tolerant of pollution or and/or poor physical habitat. The MIBI score of 31.80 is well below the standard and lower confidence limit for class 4 streams (Table 49).

**Table 49:** Summary of biological monitoring stations and data collected from Unnamed Tributary to St. Louis River

| Macroinvertebrate Assessments | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Station** | **Drainage Area (mi2)** | **Gradient (%)** | **Stream Order (Strahler)** | **Invert IBI Class** | **Invert IBI Result *(visit year)*** | **Invert IBI Result *(visit year)*** | **Standard** | **IBI Lower Confidence Limit** | **IBI Upper Confidence Limit** |
| **09LS035** | 4.83 | 0.45 | 1 | 4 | **31.80** *(2009)* | - | 51 | 37.40 | 64.60 |

A comprehensive review of available water chemistry and physical habitat data were review to generate a list of candidate causes for impairment. Due to the relatively small size of this watershed and the limited amount of accessible sampling points, the data set is somewhat limited in terms of sampling visits and parameters assessed. Candidate causes for the MIBI impairment in unnamed tributary to the St. Louis River include:

1. **Lack of streamflow due to altered hydrology**
2. **Poor physical habitat**
3. **TSS / turbidity**

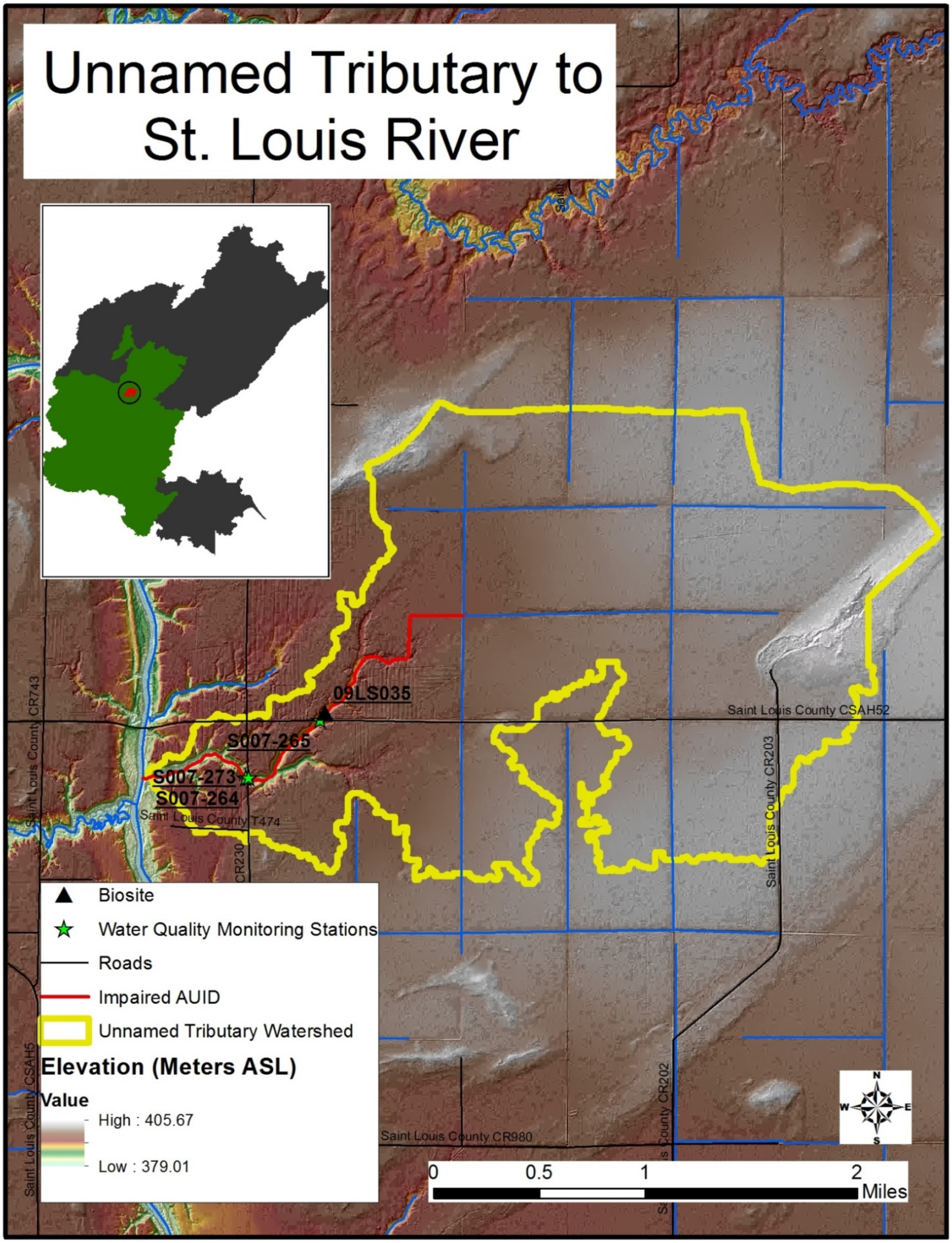


**Figure 161:** Map of Unnamed Tributary to SLRW and monitoring locations.

### 5.10.1 Lack of Flow

Headwaters streams are critical for maintaining hydrologic connectivity and ecosystem integrity at regional scales (Freeman et al. 2007). Yet, the importance of small headwaters streams to watershed health is often underappreciated and ignored. As a result, they are often managed more as water conveyance systems. Due to the small catchment size of these headwaters streams, they are easily influenced by small-scale differences and especially sensitive to disruption. Across the conterminous U.S., Carlisle et al. (2010) found that there is a strong correlation between diminished streamflow and impaired biological communities. Habitat availability can be scarce when flows are interrupted, low for a prolonged duration, or extremely low, leading to a decreased wetted width, cross sectional area, and water volume.

The lone biological monitoring station (09LS035) located on this impaired unnamed tributary to the St. Louis River has a drainage area of only 4.83 square miles. Even under pristine watershed conditions, a stream of this size which lacks a substantial groundwater component may be prone to periods of low flow during seasonal drought conditions. This particular stream is especially vulnerable given that over 85% of the stream miles in its watershed have been channelized. Essentially 100% of the stream miles upstream of the biological monitoring site have been channelized, as most of the stream length that was left in a natural form is downstream of the impaired biological monitoring station. Several site visits were conducted along the impaired reach in 2011, 2012, 2013, and 2014, and at no point has flow been intermittent. During the mid-summer and fall months, water levels do recede to the point where flows become stagnant, and improperly set and/or old road culverts become barriers to fish passage. The crossing located at St. Louis County Road 52 is one of these low flow barriers (Figure 162).

The majority of the channelized streams that feed this impaired tributary of the St. Louis River were not historically streams at all. Instead, they became water conveyance systems as the bogs and wetlands of this watershed were channelized in the early 1900's in an attempt to improve the agricultural capacity of land in this region. There is no long term flow record available to evaluate the impacts that ditching these wetlands areas had on the hydrological regime of this watershed. Hydrologic models (such as HSPF) should be employed to determine whether or not the channelization of wetlands in this watershed can be linked to altered streamflow.

**Figure 162:** Photos of Unnamed tributary to St. Louis River at County Road 52. Note low flow conditions and possible barrier to fish passage (right).

The macroinvertebrate community observed at this station is somewhat symptomatic of a stream that experiences low flow conditions and lacks habitat diversity. Collector-filterer macroinvertebrates were the most common trophic guild represented in the sample, with the majority of these being fingernail clams (*Pisidiidae*) and black fly larvae (*Simulium*). The fingernail clams are a family of the mollusks, and are abundant in wetlands, and uncommon-to-non-existent in flowing waters (limited to backwater areas). Over 50% of the organisms counted from this reach were fingernail clams, which is an indication that conditions at this site include stagnant, wetland-like conditions and favors organisms that thrive in those habitats.

This unnamed tributary is not listed as impaired for fish, as several good indicator species were present (Finescale Dace, Northern Redbelly Dace). Although these species are generally considered indicators of a healthy stream, they are also quite tolerant of wetland conditions where current velocity is non-existent or minimal. The lack of fast-water, riffle-dwelling species at this site may be an indicator of low flow and stagnant current velocities.

**Summary: Is lack of flow a stressor in Unnamed Tributary to St. Louis River?**

The small drainage area, observations of stagnant flow conditions, and biota that favor wetland-like environments are all indictors that low streamflow is a limiting factor in this watershed. What is less clear is how land-use changes such as ditching and wetland removal have altered the flow regime in this watershed. Additional work should be completed via hydrological models to evaluate the degree of departure from natural background conditions, and the overall restoration potential for this stream.

Altered hydrology and low streamflow should remain a candidate cause for impairment for unnamed tributary to the St. Louis River. Hydrologic modeling data (e.g. HSPF output) should be used to further evaluate this stressor and add confidence to decision that confirms or refutes this stressor.

### 5.10.2 Poor Physical Habitat Conditions

Streams with homogenous substrate types often support lower macroinvertebrate taxa richness and abundance, particularly if the dominant substrate type consists of fine sediment (sand, silt, clay). Station 09LS035 on the impaired reach of Unnamed Creek had only two substrate types present --- sand and silt (Figure 163). The lack of substrate types and the dominance of fine particles within this reach resulted in poor scores in the substrate related metrics that are included in the MSHA. The substrate scores for this station were comparable to the 25th percentile scores for class 4 MIBI stations scoring above the impairment threshold (Figure 163). Compared to high-quality MIBI stations in the greater SLRW, the scores from the Unnamed Tributary are quite poor.

#### Lack of Riffle and Glide Features

Riffles and glide habitats are usually shallow in depth with moderate to swift current velocities. These stream features are generally considered to be the most productive habitats for macroinvertebrate life (Buffagni and Comin 2000). A stream reach with a diversity of habitat types (riffle, glide, pool, woody debris jams, aquatic macrophytes) are bound to support a higher diversity of organisms, and likely more taxa which are dependent on specific habitats. Station 09LS035 contains very few riffle habitats (5% of the total reach) and no glide habitats. The lack of these high-productivity habitat types within the reach is likely limiting overall taxa richness and the ability of sensitive taxa to take hold within this reach. Approximately 80% of the biological monitoring reach was classified as “run” habitat at the time of sampling, meaning that it was fairly uniform in depth and surface turbulence and swift water were not observed throughout most of the sampling reach.

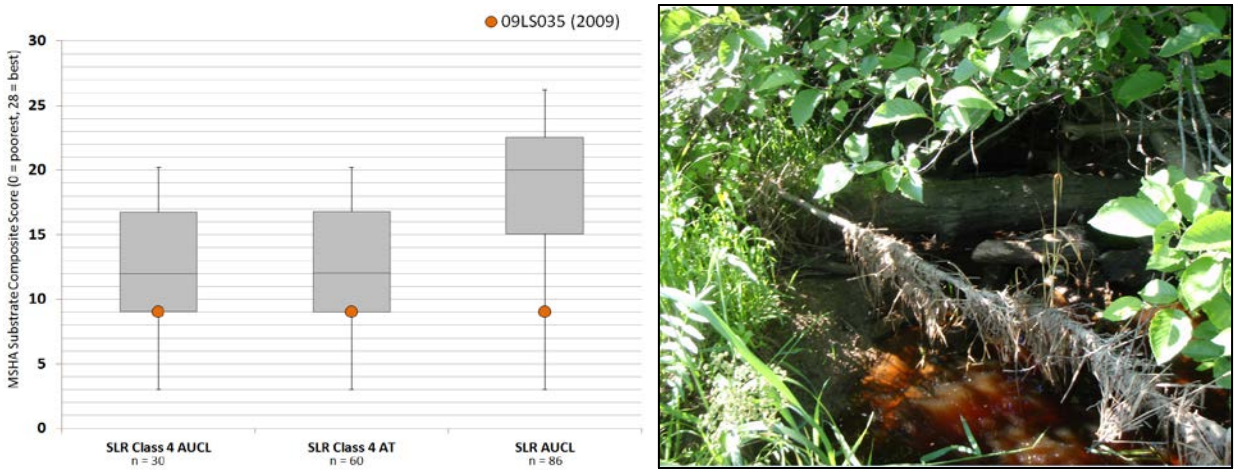


**Figure 163: (Left)** Box plots comparing MSHA substrate scores for Unnamed Creek station 09LS035 to high quality stations in the SLRW ***** See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold. **(Right)** Photo from the biological sampling reach showing sand and silt substrates.

## Biological Response to Poor Habitat Conditions

Numerous macroinvertebrate metrics can be used to evaluate physical habitat limitations. For this particular stream, specific metrics were chosen that focus on the homogenous habitat conditions (runs dominated by fine substrate) and the low flow conditions that are the result of a small watershed with significant ditching of wetlands.

### Overall Macroinvertebrate Taxa Richness

Streams with poor habitat conditions typically support fewer macroinvertebrate taxa, particularly sensitive taxa. Overall macroinvertebrate taxa richness in Unnamed Tributary to the St. Louis River is lower than the majority of class 4 MIBI stations that score above the MIBI impairment threshold and UCL of the MIBI (Figure 164). Clearly, macroinvertebrate taxa richness is lower at station 09LS035 compared to high quality stations in the SLRW with comparable drainage area, slope, and other natural background conditions.

### Sprawler Taxa Richness

Macroinvertebrate taxa that are considered “sprawlers” live on the surface of floating aquatic plants or fine sediments, and usually have physical adaptations for staying on top of substrate and keeping respiratory surfaces free of silt. These organisms are well-suited for living in slow moving streams dominated by fine substrates. Over 30% of the macroinvertebrate taxa observed at station 09LS035 were sprawler taxa. This result was well above the 75th percentile value observed in comparable, high-quality reference streams in the SLRW (Figure 165). The relative abundance of sprawler taxa in the impaired reach is another piece of evidence in support of poor habitat conditions (specifically fine substrate) as a stressor contributing to low MIBI scores.

Long-Lived Taxa Percent

Macroinvertebrate taxa that are considered “long-lived” require more than one year in order to complete their life-cycle. Therefore, these organisms provide an excellent indicator of habitat stability and long-term watershed health. A stream lacking or supporting very few of these taxa relative to the overall population may experience recurring stressors (low flows, frequent flooding, seasonally low DO) and provide conditions that are more suitable for organisms that can rapidly complete their life cycles. Only 5% of the macroinvertebrate taxa at station 09LS035 were long-lived. This percentage was lower than all of the class 4 stations scoring above the upper confidence limit of the MIBI threshold, and lower than nearly all stations scoring above the MIBI threshold (Figure 166). The low percentage of long-lived taxa within the impaired reach provides evidence in support of poor habitat and low flow conditions as contributing stressors.

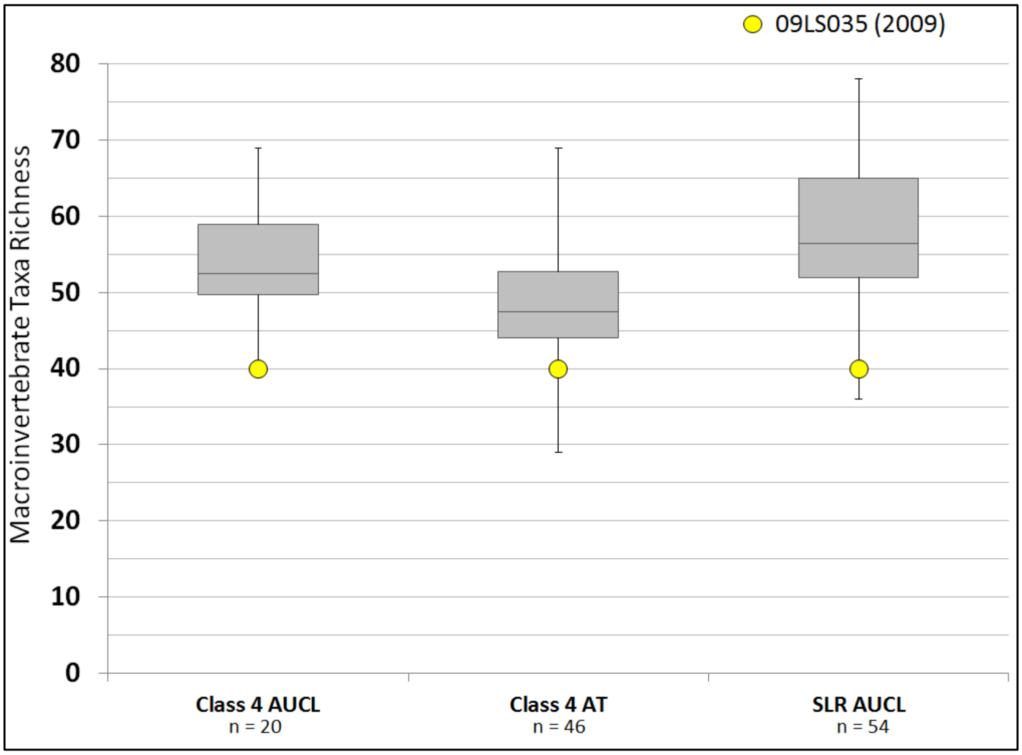


**Figure 164:** Macroinvertebrate taxa richness observed at station 09LS035 compared to high quality reference stations

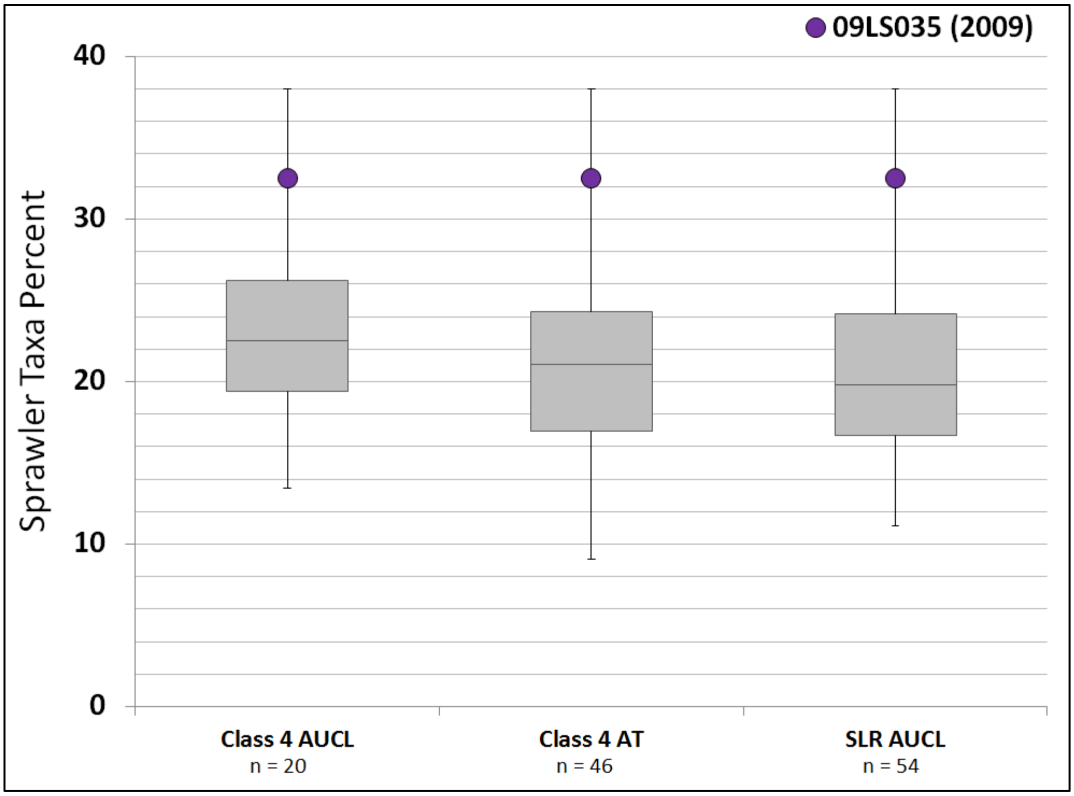


**Figure 165:** Sprawler taxa percent observed at station 09LS035 compared to high quality reference stations. ***** See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

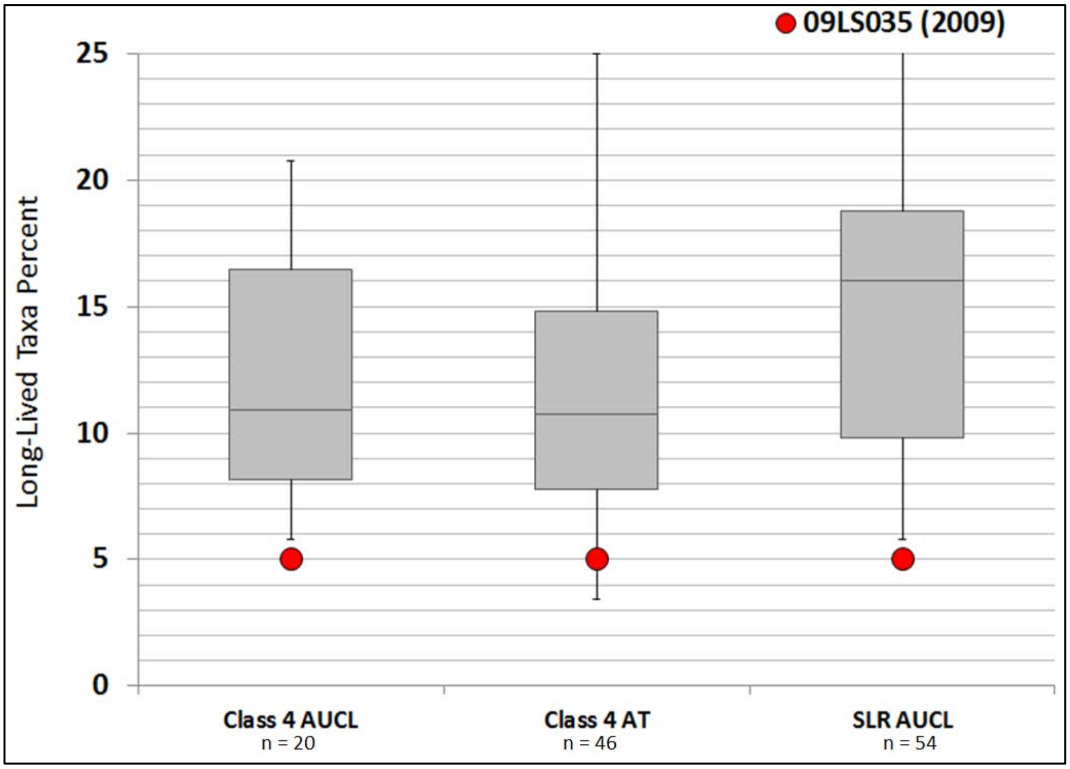


**Figure 166:** Long-lived macroinvertebrate taxa percent observed at station 09LS035 compared to high quality reference stations. ***** See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

**Summary: Are poor habitat conditions a stressor in Unnamed Trib to St. Louis R.?**

This impaired tributary to the St. Louis River provides marginal habitat for macroinvertebrate due to low summer and fall stream flow, lack of riffle and glide habitats, and poor substrate conditions. Natural background conditions may contribute significantly to several or all of these limiting factors. As a result, restoration potential may be minimal in this watershed. Regardless, poor habitat conditions and low streamflow should be considered stressors contributing to the MIBI impairment.

## 5.10.3 Unnamed Trib. to St. Louis River: Summary of Stressors to Aquatic Life

**Table 50:** Summary of SID results for Unnamed Tributary to St. Louis River

| Candidate Cause | Result |
|---|---|
| **Poor Physical Habitat Conditions** | ● |
| **Altered Hydrology** | ○ |
| Key: **• = confirmed stressor ○ = Potential Stressor X = eliminated candidate cause** | |

## 5.11 Little Swan Creek

The source of Little Swan Creek lies within the expansive peat bogs southwest of Cherry, Minnesota. The channel is highly sinuous and narrow and a GIS analysis showed that the entire river types out as an E channel. For the upper half of its length the stream flows through a wide lacustrine valley (Type 10). Close to Highway 5 the river gradually cuts down into the lake bed and has created an alluvial valley for itself. Little Swan Creek has an average slope of 0.1%, falling 44 feet in a little over 7 miles.

One biological monitoring site was sampled to evaluate the fish community of Little Swan Creek. The FIBI impairment listing was based off of the result from 2009, which produced a FIBI score of 34 (out of 100), which is only one point below the impairment threshold for northern coldwater streams IBI class (Table 51). White Sucker was the dominant fish species sampled at Little Swan Creek station 09LS062, accounting for 59% of the total catch. Other species present but not abundant included Trout-Perch, Creek Chub, Central Mudminnow, and Northern Pike. The low FIBI results for Little Swan Creek are related to the lack of coldwater fish taxa in this stream. The presence of Trout-Perch at 09LS062 suggests that the temperature regime of this stream is probably cooler than many streams in the area, but no species that qualify in coldwater metrics of the IBI scoring were observed (Brook Trout, sculpin, Longnose Dace, Pearl Dace).

**Table 51:** Summary of biological monitoring stations, biological sampling results, and applicable standards

| **Fish Assessments** | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Station** | **Drainage Area (mi2)** | **Gradient (%)** | **Stream Order (Strahler)** | **FIBI Class** | **FIBI Result *(visit year)*** | **FIBI Result *(visit year)*** | **Standard** | **IBI Lower Confidence Limit** | **IBI Upper Confidence Limit** |
| 09LS062 | 21.08 | 0.17 | 2 | 11 | **34** *(2009)* | - | 35 | 25 | 45 |

Water quality and physical habitat data were used to develop a list of candidate causes for the FIBI impairment in Little Swan Creek. The following candidate causes were selected for further analysis in this section:

1. **Elevated water temperatures**
2. **Low DO**
3. **TSS / Turbidity**
4. **Poor physical habitat conditions**

### 5.11.1 Elevated Water Temperature

Continuous water temperature data are available from two sites on Little Swan Creek; S007-248 at County Road 5 (2012) and 09LS062 at County Road 444 (2009 and 2012) (Figure 168). Only data between June 1 and August 31 were analyzed, which corresponds to the window of time when stream temperatures are most likely to exceed the stress threshold for Brook Trout and other cold water fish species.

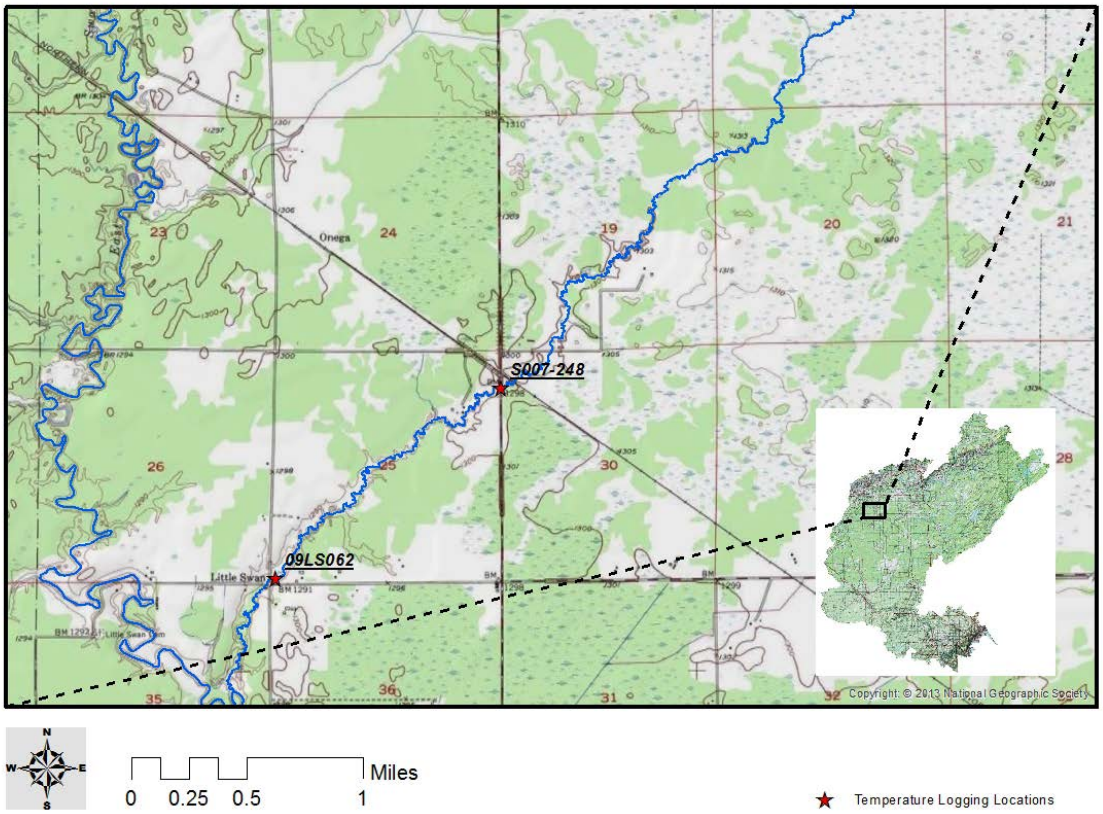


**Figure 168:** Map of Little Swan Creek temperature monitoring stations

Little Swan Creek shows susceptibility to ambient atmospheric conditions. In 2009, a colder-than-normal year, the data from 09LS062 show suitable water temperatures for coldwater fish for nearly the entire summer. Only 10% of the summer was spent in the stress range for Brook Trout. Summer temperatures in nearby Hibbing, Minnesota that year were 4.8°F below average (weather-warehouse.com). In contrast, at that same site during the warmer-than-normal summer of 2012 (about 0.8°F above average), the stream was in the stress range nearly 50% of the time. Stressful water temperatures for coldwater fish were not just observed during peak daylight hours. At station 09LS062 during the 2012 monitoring year, temperatures were in the stress range for Brook Trout for eight straight days. The summer air temperatures in 2012 were much closer to average than 2009, suggesting that 2009 was an anomaly and the stream temperatures in 2012 are much more reflective of an average summer for Little Swan Creek.

Such a strong mirroring of atmospheric conditions suggests that there is poor groundwater input to Little Swan Creek, and/or very little shading for the stream. Both of these scenarios would render the creek vulnerable to heat loading during average summer conditions. The watershed has not been heavily ditched and road densities are low relative to others in the Meadowlands Watershed zone. The riparian corridor is rather healthy as well, with moderate amounts of larger trees and long grasses to provide shade for most of the length of the stream (see Figure 169). However, large-scale logging activities are common in the upper part of the watershed (Figure 169), and the underlying geology of the watershed may be unfavorable to groundwater recharge. Additional study would be required to ascertain the impact of ground water recharge (or lack thereof) on the thermal regime of Little Swan Creek.

Another potential source of temperature loading is beaver activity. Using LiDAR profiles and high-resolution aerial photos, six beaver dams were counted on 7.2 miles of Little Swan Creek (0.8 dams/mile). One especially large, approximately 1 meter-high dam in the upper reaches has created a half-mile long impoundment (Figure 169). It is likely that the beaver activity on Little Swan Creek is having a negative impact on stream temperature and making the stream much more susceptible to the ambient atmospheric conditions.

**Figure 169: (left)** Long grasses and deciduous tree canopies dominate the riparian corridor for Little Swan Creek. **(Right)** Beaver dams and logging in the headwaters of Little Swan Creek

### Biological Response to Elevated Water Temperature

An analysis of coldwater FIBI biological metrics reaffirms temperature as a stressor compared with unimpaired coldwater sites in the SLRW. No coldwater taxa were sampled in the reportable visit to Little Swan biological monitoring station 09LS062. Healthy, non-impaired coldwater streams in the SLRW support one or several coldwater fish taxa, such as Brook Trout, Mottled Sculpin, Pearl Dace, or Longnose Dace. None of these species were observed in Little Swan Creek. Instead, the fish community was dominated by White Sucker and trout perch, which are found in both coldwater and warmwater streams. The other species present at this monitoring station were all warmwater species.

### Summary: Is elevated water temperature a stressor in Little Swan Creek?

Monitoring results show a marginal to poor temperature regime for supporting coldwater fish taxa. In addition, biological monitoring data support elevated water temperatures as a stressor based on the lack of coldwater taxa present observed in the fish community. Elevated water temperature should be considered a stressor in Little Swan Creek.

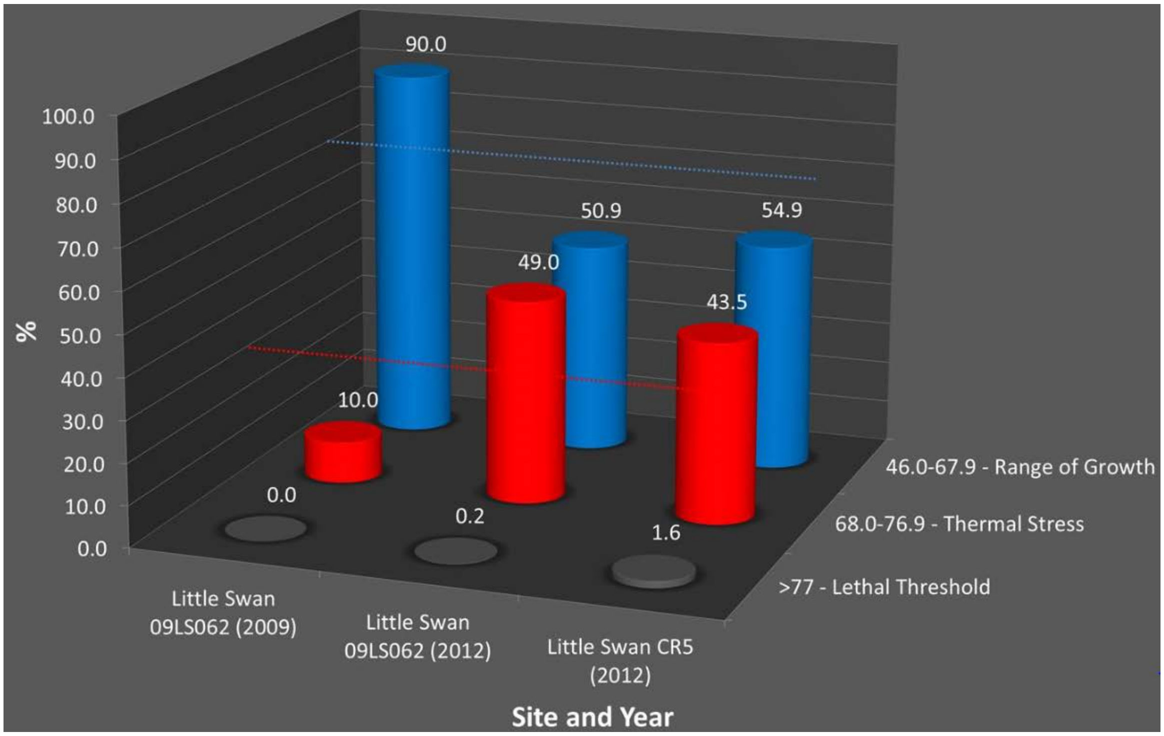


**Figure 170:** Percentage of time spent in BKT growth, stress and lethal ranges

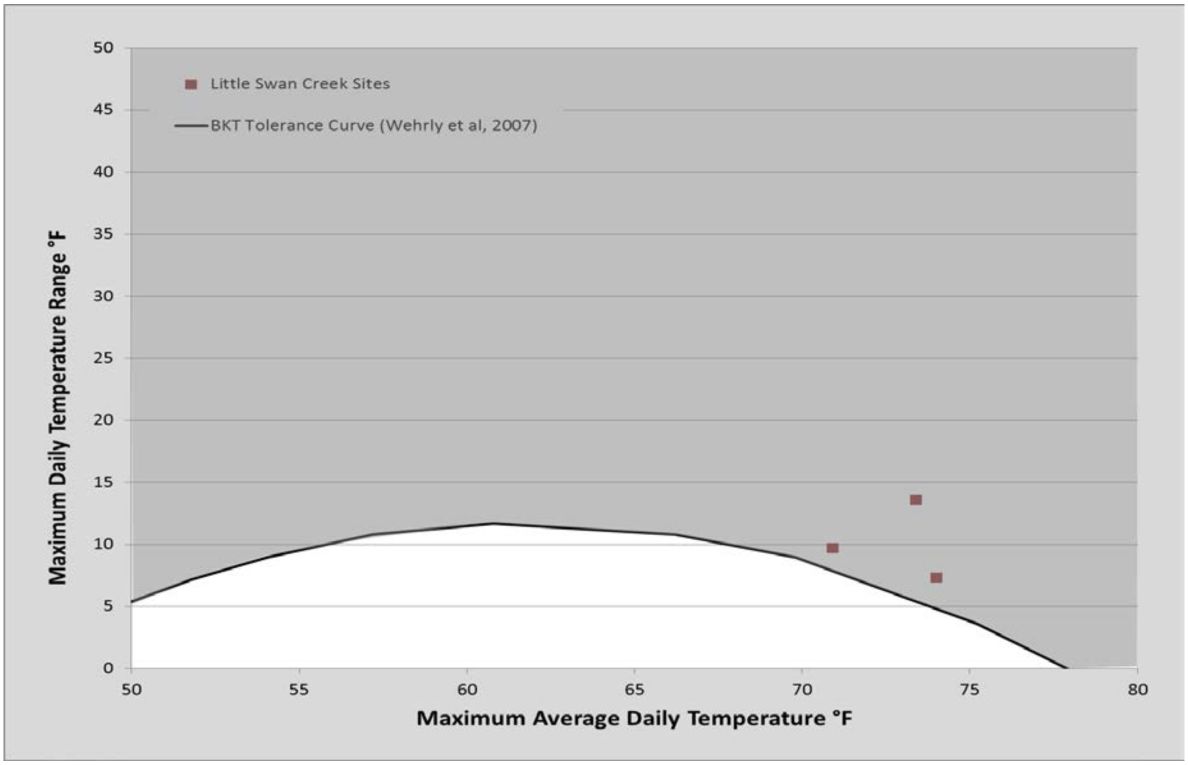


**Figure 171:** Maximum daily temperature and maximum temperature fluctuation recorded on Little Swan Creek

## 5.11.2 Low Dissolved Oxygen

The DO was identified as a candidate cause for the fish impairment in Little Swan Creek. A total of 27 DO measurements were collected from Little Swan Creek between 2009 and 2013. The results, shown in Figure 172, indicate that DO levels frequently fall well below the water quality standard for coldwater streams (7 mg/L). Over half of the DO measurements collected from this stream are below the 7 mg/L standard, including 100% of measurements taken during July and August. Minimum DO concentrations observed at the biological monitoring station are around 3 mg/L.

Continuous monitoring results from Little Swan Creek provide further evidence that DO concentrations are potentially unsuitable for sensitive coldwater fish taxa. In July of 2012, the DO concentrations fell below the 7 mg/L standard for the duration of the five-day monitoring period (Figure 173). The results from August 2013 were more suitable, as DO concentrations remained above 7 mg/L for the entire seven-day monitoring period. The difference in DO concentrations observed in these two continuous profiles can be attributed to differences in water temperature. In 2013, the average water temperature recorded during the continuous monitoring period was 15.9° C, compared to 22.7° C in the 2012 survey. Diurnal fluctuation in DO concentration was low (less than 1 mg/L) during both continuous monitoring events, which is an indicator of low productivity in this stream.

### Sources and Pathways of Low DO

A combination of wetland processes and geomorphic features are likely contributing to the low DO concentrations observed in Little Swan Creek. Over 75% of the land in the watershed is classified as woody or emergent herbaceous wetland. Organics from wetlands and bogs often increase BOD in surface water, resulting in less oxygen available for other forms of aquatic life. The BOD concentration in Little Swan Creek was elevated (7.2 mg/L) on July 18, 2012, but no other data are available for this parameter.

The TP concentrations in Little Swan Creek are elevated in comparison to high quality reference streams in the SLRW. Forty-percent of samples (8 of 20) exceeded the MPCA's River Nutrient TP criterion of 0.055 mg/L applied to Northern Lakes and Forest ecoregion streams. Other than some light agricultural activity (cattle grazing/hay production) and some areas of streambank erosion, most of the sources of phosphorous in the watershed are related to the wetland qualities found within this drainage.

Little Swan Creek is a low gradient stream with a substrate dominated by fine particles such as silt, sand, and clay. Due to these characteristics, the stream lacks riffle features and as a consequence, very little oxygen is supplied to the stream by way of aeration from the atmosphere.

### Biological Response to Low Dissolved Oxygen

Five species of fish were observed at monitoring station 09LS062; White Sucker, Creek Chub, Trout-Perch, Central Mudminnow, and Northern Pike. With the exception of Central Mudminnow, which is considered highly tolerant of low DO, the majority of the species sampled are neither tolerant nor intolerant of low DO conditions. Overall fish abundance was very low, with only 34 individuals sampled in the reach. Nearly 60% of the fish sampled were White Sucker, which are found both in streams with both low and moderate to high DO concentrations. DO TIV results for this station are slightly lower but comparable to the median DO TIV values observed at high quality reference sites (Figure 174).

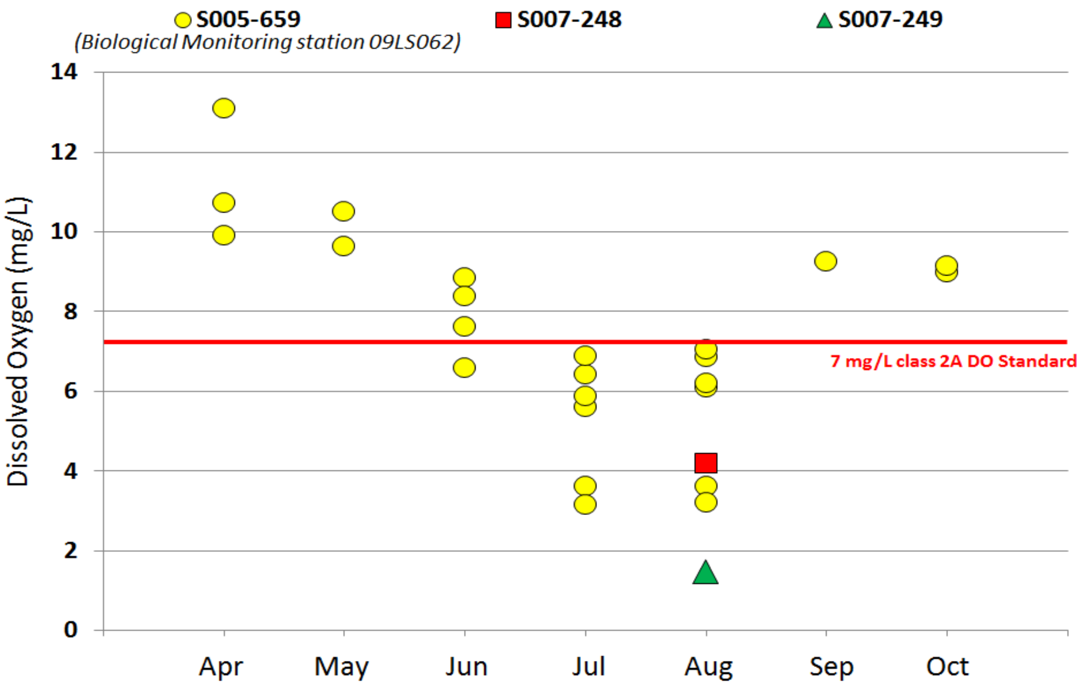


**Figure 172:** Instantaneous measurements of DO by monitoring station and calendar month.

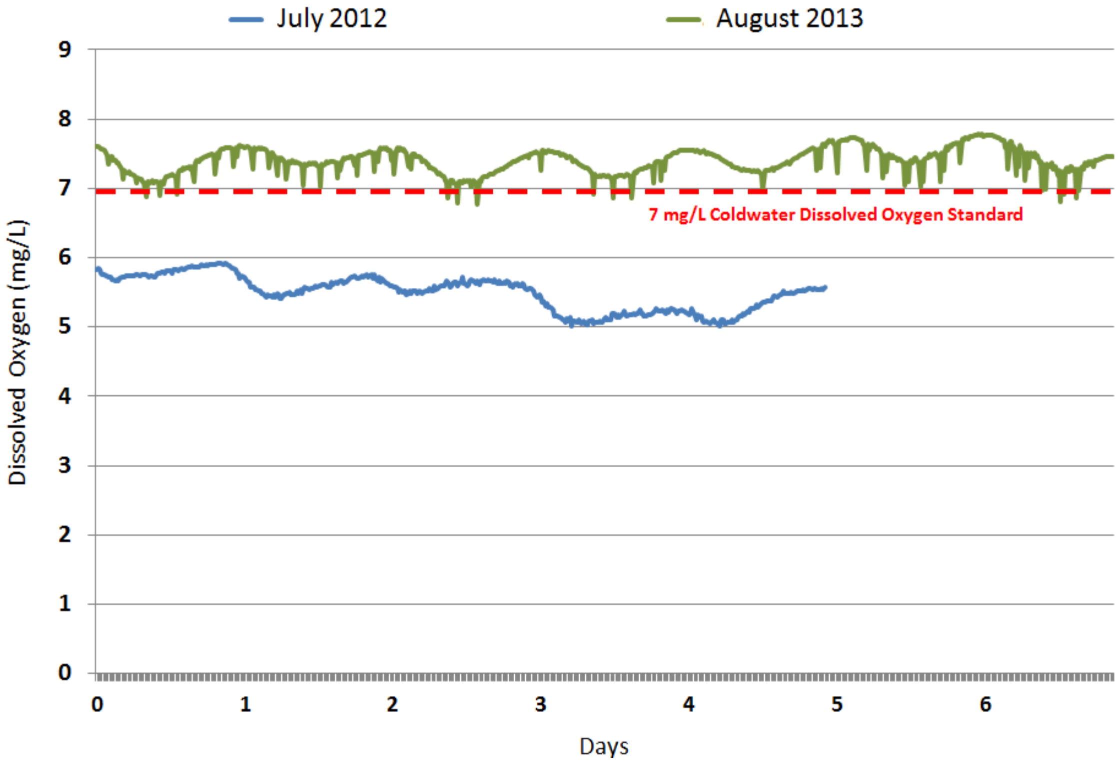


**Figure 173:** Results from two continuous DO surveys at biological monitoring station 09LS062

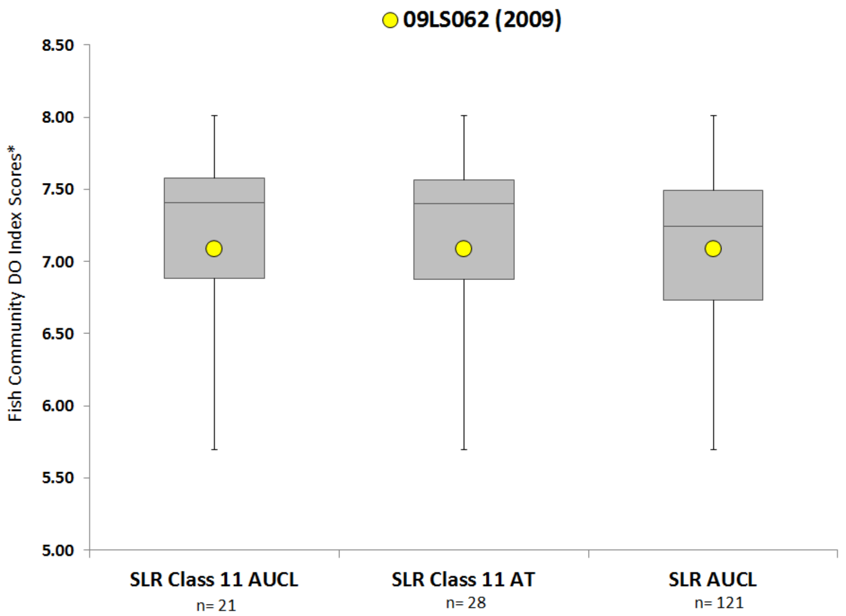


**Figure 174:** Fish community DO Tolerance Indicator Values for station 09LS062 compared to results from high quality reference streams. ***** See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL** = Above Upper Confidence Limit of FIBI threshold **AT** = Above Fish IBI Threshold

**Summary: Is low dissolved oxygen a stressor in Little Swan Creek?**

The fish community observed in this stream is not highly tolerant of low DO conditions. However, DO intolerant species and coldwater taxa were absent and DO values consistently fall well below the 7 mg/L water quality standard for coldwater streams. Based on the water chemistry and biological data available, it appears that DO concentrations in Little Swan Creek are inadequate for supporting coldwater fish taxa with demands for highly oxygenated water. Therefore, low DO should be considered a stressor to aquatic life in this watershed.

## 5.11.3 Total Suspended Solids & Turbidity

A summary of TSS and Secchi transparency tube (s-tube) results was developed (Table 52) based on a total of 47 samples that were collected over a five year period (2009-2013). All samples were collected at station S005-659, which is co-located with the impaired biological monitoring station 09LS062 (Figure 175). Slightly over 40% of the sampling results (20 out of 47) from this station exceed either the TSS or transparency water quality standard. In terms of the TSS results, 5 out of 22 samples (20.8%) collected within the assessment window of April – September exceeded the TSS standard of 10 mg/L for coldwater streams. Streams that exceed the TSS standard in more than 10% of samples (minimum of 20 samples required) can be considered “impaired” for high TSS concentrations.

**Table 52:** TSS and Secchi tube average values and percent standard exceedances for Little Swan Creek.).

| Site | Site Description | TSS Average (mg/L) | # of Samples | TSS % Exceeding Standard | Secchi Tube Average (cm) | # of Samples | Secchi Tube % Exceeding Standard | Total # of Samples | Total % Exceeding Standard |
|---|---|---|---|---|---|---|---|---|---|
| S005-659 | LITTLE SWAN CR AT CR-444 | 8.2 | 24 | 20.8% | 50.6 | 23 | 65.2% | 47 | 42.6% |

Box plots of the TSS and Secchi tube datasets for Little Swan Creek were compared to the datasets from the "A" and "B" reference streams in the SLRW (Figure 176). Both datasets are worse than the values for the reference streams. Only the 75th TSS percentile fails to meet the the draft standard of 10 mg/L. As stated before, water clarity is worse, and the median Secchi reading fails to meet the draft standard of 40 cm. The site used in this analysis is in violation of the draft standards frequently enough to technically be considered "impaired" for turbidity, but the water quality data alone does not make a strong case in determining whether TSS are stressing fish communities.

**Figure 175:** Little Swan Creek during snowmelt (left) and at low flow (right)

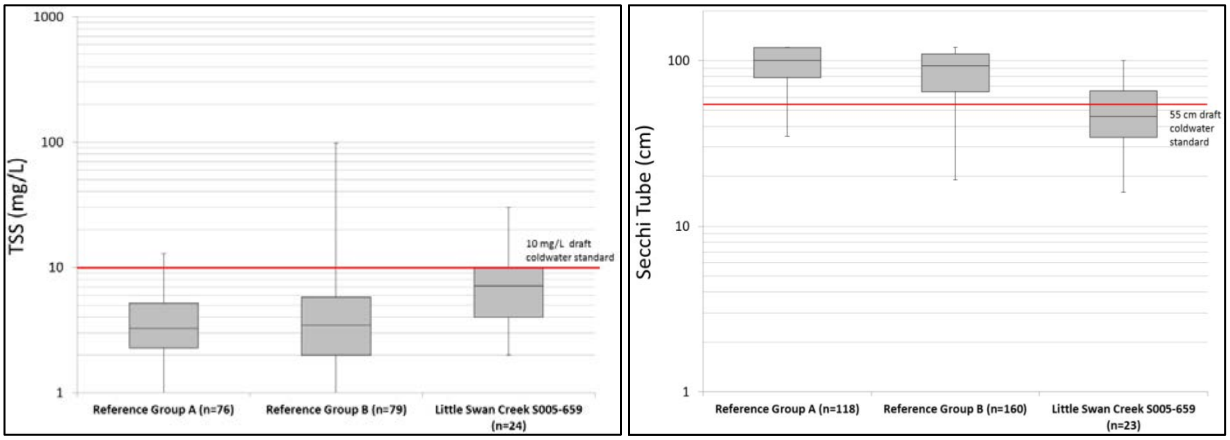


**Figure 176:** Box plots of TSS values for Little Swan Creek and reference streams

### Sources and pathways of Elevated TSS

Channel Instability/Bank Erosion

Areas of channel instability and bank erosion were observed at the 09LS062 biological monitoring station (Figure 178) and other road crossings in the watershed. Channel instability in the Little Swan Creek Watershed is likely a consequence of increased peak flows as a result of road ditches or land cover change (forests converted to agriculture), or the result of a local base level drop in East Swan River which caused a headcut to advance up into the Little Swan Creek Watershed. Many reaches in Little Swan Creek show erosion on both sides of the river and vegetation falling into the channel – both are manifestations of channel incision.

**Figure 178:** Erosion on both banks and leaning trees are a sign of channel incision in Little Swan Creek

### Biological effects of elevated TSS

## Fish Response to TSS

Little Swan Creek is listed as impaired for failing to meet the coldwater FIBI criteria. The lone biological monitoring station on this stream (09LS062) scored poorly in the FIBI due to a lack coldwater species and a high percentage of omnivorous fish taxa, which are typically uncommon in functioning coldwater streams. Overall taxa richness and fish abundance were both relatively low (5 species, 34 total individuals sampled). White Sucker accounted for nearly 60% of the individuals sampled at the lone monitoring station on Little Swan Creek. Also present in the sample were Trout-Perch (6), Creek Chub (4), Central Mudminnow (3), and Northern Pike (1).

None of the fish taxa observed in Little Swan Creek are considered to be strongly associated with elevated TSS concentrations. The entire fish community at 09LS062 can be considered “neutral” in terms of their TIVs) for TSS, meaning that they are neither tolerant nor intolerant of this stressor. White Sucker, the most dominant taxa on Little Swan Creek, is the most common and widespread sucker species in Minnesota (Phillips et al. 1982) and is able to adapt to a variety of environmental conditions. The other fish taxa observed in Little Swan Creek possess a similar ability to adapt to different stream conditions and their presence does not offer a strong biological indicator for TSS-induced stress.

However, in comparison to fish assemblages observed in high quality coldwater streams of the SLRW, the fish community of Little Swan Creek is much more tolerant of TSS. The TSS TIVs for Little Swan Creek station 09LS062 were compared against healthy biological monitoring stations throughout the SLRW. The fish community observed at 09LS062 was more tolerant of TSS than nearly 100% of comparable stations that scored above the UCL of the FIBI. However, the strong presence of generally tolerant species at this station renders it difficult to tell whether this discrepancy is due to elevated TSS and not a combination of other stressors.

Little Swan Creek is not currently listed as impaired for macroinvertebrate IBI. However, the data can be evaluated as another piece of evidence for or against TSS as a stressor. There were eight macroinvertebrate taxa sampled that are considered to be “tolerant” or “very tolerant”, and they accounted for a very small percentage of the overall population (4.0% and 2.6%, respectively). No intolerant species were collected, indicating that the vast majority of macroinvertebrates in Little Swan Creek are considered neutral to elevated TSS. Although fish and invertebrates have different tolerance levels and responses to suspended sediment, the low percentages of TSS tolerant invertebrate taxa is another piece of evidence that weakens the case for TSS as a stressor.

**Summary: Are elevated TSS concentrations a stressor in Little Swan Creek?**

The inconclusiveness of the chemistry and biological data suggests that TSS is not significantly affecting fish communities in Little Swan Creek, and it should not be added as a stressor to aquatic life. Elevated water temperatures, low DO, and physical habitat limitations are believed to be more significant stressors leading to the poor FIBI results.

### 5.11.4 Poor Physical Habitat Conditions

The MSHA methodology was used to assess habitat conditions at biological monitoring station 09LS062 in the summer of 2009. The overall MSHA score for this station was 37 out of a possible 100, which is considered a “poor” rating. In fact, the MSHA result from this site represents the worst score of all the streams analyzed in this region of the SLRW. All of the MSHA metrics evaluated scored poorly at this monitoring station, but of these metrics related to substrate and channel morphology were particularly poor. Scores related to substrate condition were poor due to the dominance of clay and silt and the absence of coarser substrates, while channel morphology metric scores were poor due to the lack of riffle and glide features. The entire sampling reach was composed of “run” and “pool” channel features, which are almost always deep habitats with slow current velocities according the MSHA definitions. Channel stability and channel development scores were both “moderate” and reflect the fairly poor condition of the channel. Among the other attributes of Little Swan Creek 09LS062 that led to its poor MSHA score were the lack of depth and current variability and the absence of macrophytes.

The PSI was used to assess stream channel stability within the biological sampling reach. Results for Little Swan Creek are typical of a slightly incised stream channel, with characteristics such as moderate bank erosion and loose bottom sediments observed as indicators of channel instability. Overall, station 09LS062 received a PSI score of 104, which corresponds to a rating of “unstable” for the potential E6 stream type. Many of the PSI metrics that scored poorly were related to the channel bottom and lower banks, while the metrics pertaining to the condition of the upper banks scored fairly well

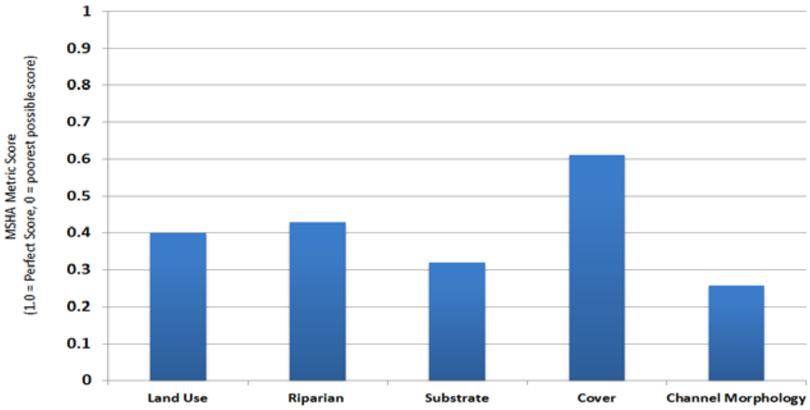
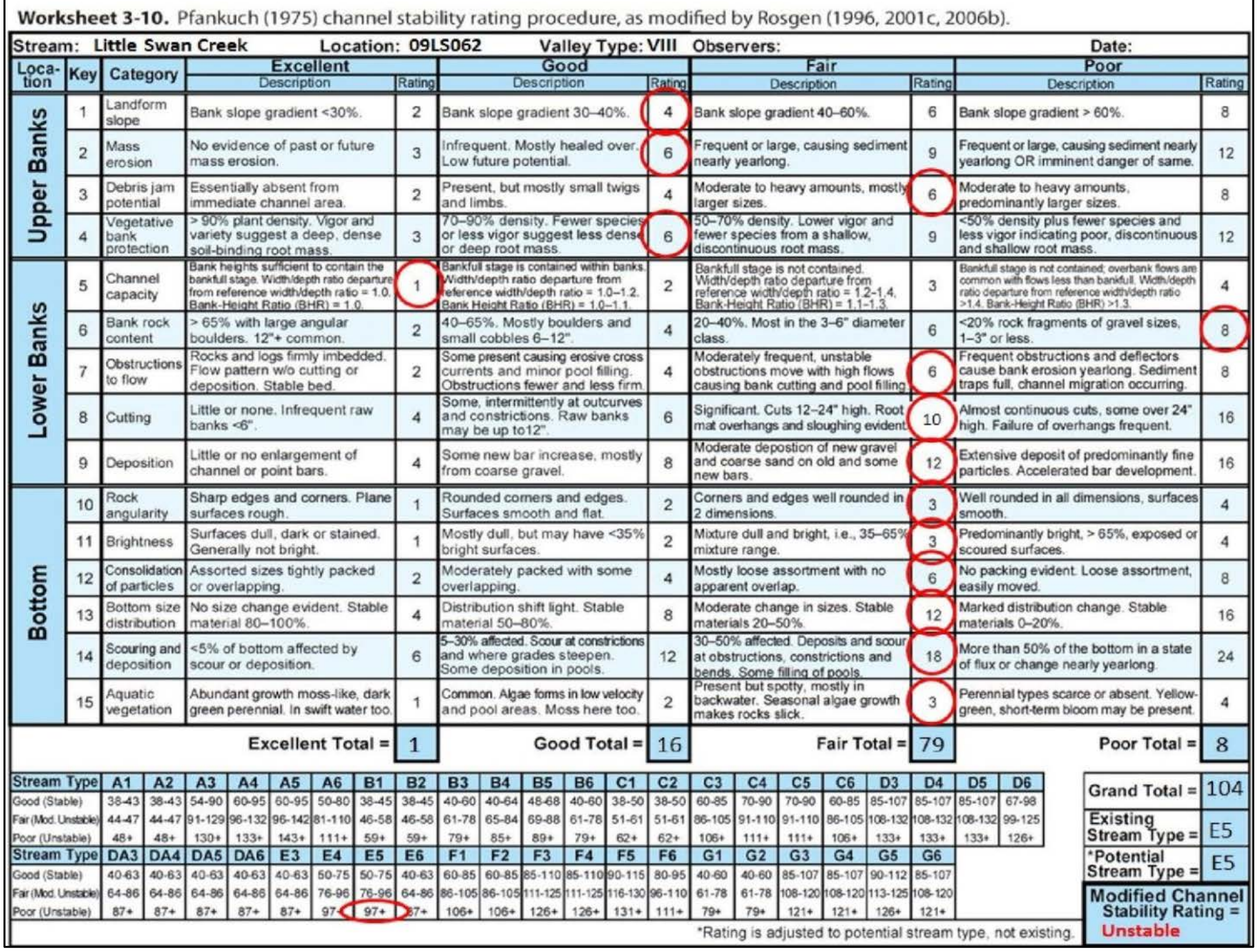


**Figure 179:** MSHA results for various habitat metrics at biological monitoring station 09LS062



**Worksheet 3-10.** Pfankuch (1975) channel stability rating procedure, as modified by Rosgen (1996, 2001c, 2006b).

**Stream:** Little Swan Creek **Location:** 09LS062 **Valley Type:** VIII **Observers:** **Date:**

| Location | Key | Category | Excellent | | Good | | Fair | | Poor | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | Description | Rating | Description | Rating | Description | Rating | Description | Rating |
| Upper Banks | 1 | Landform slope | Bank slope gradient <30%. | 2 | Bank slope gradient 30–40%. | [✓ 4] | Bank slope gradient 40–60%. | 6 | Bank slope gradient > 60%. | 8 |
| | 2 | Mass erosion | No evidence of past or future mass erosion. | 3 | Infrequent. Mostly healed over. Low future potential. | [✓ 6] | Frequent or large, causing sediment nearly yearlong. | 9 | Frequent or large, causing sediment nearly yearlong OR imminent danger of same. | 12 |
| | 3 | Debris jam potential | Essentially absent from immediate channel area. | 2 | Present, but mostly small twigs and limbs. | 4 | Moderate to heavy amounts, mostly larger sizes. | [✓ 6] | Moderate to heavy amounts, predominantly larger sizes. | 8 |
| | 4 | Vegetative bank protection | > 90% plant density. Vigor and variety suggest a deep, dense soil-binding root mass. | 3 | 70–90% density. Fewer species or less vigor suggest less dense or deep root mass. | [✓ 6] | 50–70% density. Lower vigor and fewer species from a shallow, discontinuous root mass. | 9 | <50% density plus fewer species and less vigor indicating poor, discontinuous and shallow root mass. | 12 |
| Lower Banks | 5 | Channel capacity | Bank heights sufficient to contain the bankfull stage. Width/depth ratio departure from reference width/depth ratio = 1.0. Bank-Height Ratio (BHR) = 1.0. | [✓ 1] | Bankfull stage is contained within banks. Width/depth ratio departure from reference width/depth ratio = 1.0–1.2. Bank Height Ratio (BHR) = 1.0–1.1. | 2 | Bankfull stage is not contained. Width/depth ratio departure from reference width/depth ratio = 1.2–1.4. Bank-Height Ratio (BHR) = 1.1–1.3. | 3 | Bankfull stage is not contained; overbank flows are common with flows less than bankfull. Width/depth ratio departure from reference width/depth ratio >1.4. Bank-Height Ratio (BHR) >1.3. | 4 |
| | 6 | Bank rock content | > 65% with large angular boulders. 12"+ common. | 2 | 40–65%. Mostly boulders and small cobbles 6–12". | 4 | 20–40%. Most in the 3–6" diameter class. | 6 | <20% rock fragments of gravel sizes, 1–3" or less. | [✓ 8] |
| | 7 | Obstructions to flow | Rocks and logs firmly imbedded. Flow pattern w/o cutting or deposition. Stable bed. | 2 | Some present causing erosive cross currents and minor pool filling. Obstructions fewer and less firm. | 4 | Moderately frequent, unstable obstructions move with high flows causing bank cutting and pool filling. | [✓ 6] | Frequent obstructions and deflectors cause bank erosion yearlong. Sediment traps full, channel migration occurring. | 8 |
| | 8 | Cutting | Little or none. Infrequent raw banks <6". | 4 | Some, intermittently at outcurves and constrictions. Raw banks may be up to12". | 6 | Significant. Cuts 12–24" high. Root mat overhangs and sloughing evident. | [✓ 10] | Almost continuous cuts, some over 24" high. Failure of overhangs frequent. | 16 |
| | 9 | Deposition | Little or no enlargement of channel or point bars. | 4 | Some new bar increase, mostly from coarse gravel. | 8 | Moderate depostion of new gravel and coarse sand on old and some new bars. | [✓ 12] | Extensive deposit of predominantly fine particles. Accelerated bar development. | 16 |
| Bottom | 10 | Rock angularity | Sharp edges and corners. Plane surfaces rough. | 1 | Rounded corners and edges. Surfaces smooth and flat. | 2 | Corners and edges well rounded in 2 dimensions. | [✓ 3] | Well rounded in all dimensions, surfaces smooth. | 4 |
| | 11 | Brightness | Surfaces dull, dark or stained. Generally not bright. | 1 | Mostly dull, but may have <35% bright surfaces. | 2 | Mixture dull and bright, i.e., 35–65% mixture range. | [✓ 3] | Predominantly bright, > 65%, exposed or scoured surfaces. | 4 |
| | 12 | Consolidation of particles | Assorted sizes tightly packed or overlapping. | 2 | Moderately packed with some overlapping. | 4 | Mostly loose assortment with no apparent overlap. | [✓ 6] | No packing evident. Loose assortment, easily moved. | 8 |
| | 13 | Bottom size distribution | No size change evident. Stable material 80–100%. | 4 | Distribution shift light. Stable material 50–80%. | 8 | Moderate change in sizes. Stable materials 20–50%. | [✓ 12] | Marked distribution change. Stable materials 0–20%. | 16 |
| | 14 | Scouring and deposition | <5% of bottom affected by scour or deposition. | 6 | 5–30% affected. Scour at constrictions and where grades steepen. Some deposition in pools. | 12 | 30–50% affected. Deposits and scour at obstructions, constrictions and bends. Some filling of pools. | [✓ 18] | More than 50% of the bottom in a state of flux or change nearly yearlong. | 24 |
| | 15 | Aquatic vegetation | Abundant growth moss-like, dark green perennial. In swift water too. | 1 | Common. Algae forms in low velocity and pool areas. Moss here too. | 2 | Present but spotty, mostly in backwater. Seasonal algae growth makes rocks slick. | [✓ 3] | Perennial types scarce or absent. Yellow-green, short-term bloom may be present. | 4 |
| | | | Excellent Total = | 1 | Good Total = | 16 | Fair Total = | 79 | Poor Total = | 8 |

| Stream Type | A1 | A2 | A3 | A4 | A5 | A6 | B1 | B2 | B3 | B4 | B5 | B6 | C1 | C2 | C3 | C4 | C5 | C6 | D3 | D4 | D5 | D6 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Good (Stable) | 38-43 | 38-43 | 54-90 | 60-95 | 60-95 | 50-80 | 38-45 | 38-45 | 40-60 | 40-64 | 48-68 | 40-60 | 38-50 | 38-50 | 60-85 | 70-90 | 70-90 | 60-85 | 85-107 | 85-107 | 85-107 | 67-98 |
| Fair (Mod. Unstable) | 44-47 | 44-47 | 91-129 | 96-132 | 96-142 | 81-110 | 46-58 | 46-58 | 61-78 | 65-84 | 69-88 | 61-78 | 51-61 | 51-61 | 86-105 | 91-110 | 91-110 | 86-105 | 108-132 | 108-132 | 108-132 | 99-125 |
| Poor (Unstable) | 48+ | 48+ | 130+ | 133+ | 143+ | 111+ | 59+ | 59+ | 79+ | 85+ | 89+ | 79+ | 62+ | 62+ | 106+ | 111+ | 111+ | 106+ | 133+ | 133+ | 133+ | 126+ |

| Stream Type | DA3 | DA4 | DA5 | DA6 | E3 | E4 | E5 | E6 | F1 | F2 | F3 | F4 | F5 | F6 | G1 | G2 | G3 | G4 | G5 | G6 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Good (Stable) | 40-63 | 40-63 | 40-63 | 40-63 | 40-63 | 50-75 | 50-75 | 40-63 | 60-85 | 60-85 | 85-110 | 85-110 | 90-115 | 80-95 | 40-60 | 40-60 | 85-107 | 85-107 | 90-112 | 85-107 |
| Fair (Mod. Unstable) | 64-86 | 64-86 | 64-86 | 64-86 | 64-86 | 76-96 | 76-96 | 64-86 | 86-105 | 86-105 | 111-125 | 111-125 | 116-130 | 96-110 | 61-78 | 61-78 | 108-120 | 108-120 | 113-125 | 108-120 |
| Poor (Unstable) | 87+ | 87+ | 87+ | 87+ | 87+ | 97+ | [✓ 97+] | 87+ | 106+ | 106+ | 126+ | 126+ | 131+ | 111+ | 79+ | 79+ | 121+ | 121+ | 126+ | 121+ |

| | |
|---|---|
| Grand Total = | 104 |
| Existing Stream Type = | E5 |
| *Potential Stream Type = | E5 |
| Modified Channel Stability Rating = | Unstable |

*Rating is adjusted to potential stream type, not existing.

**Figure 180:** Pfankuch Stability Index scores and rating for Little Swan Creek biological monitoring station 09LS062

### Biological effects of degraded habitat

Poor physical habitat conditions can limit the fish abundance and taxa richness of non-tolerant fish species (source). In general, the fish community of Little Swan Creek exhibited low taxa richness and low fish counts of non-tolerant species. A total of five fish taxa were present at this monitoring station, none of which are particularly sensitive to habitat degradation. In terms of fish abundance, the number of non-tolerant fish/meter at station 09LS062 was well below the rate observed at high quality stations of the same FIBI class (Figure 181).

Little Swan Creek did not compare well to unimpaired Class 11 (Northern Coldwater) streams in the SLRW in terms of the number of habitat-sensitive taxa that were present (Figure 182). One benthic insectivore species (Trout-Perch) was observed at station 09LS062, a value that is equal to or less than 75% of the Class 11 streams above the impairment threshold. No darter, sculpin, or round-bodied sucker taxa were found in Little Swan Creek – well below the SLRW Class 11 AT stations. One piscivorous species (Northern Pike) was sampled, which is the median for unimpaired Class 11 streams in the watershed. One riffle-dwelling and gravel spawning taxa (White Sucker) was present – also equal to or below 75% of the comparison streams. Little Swan Creek generally lacks species that require specific habitat features (e.g. fast-moving water, spawning gravels, interstitial spaces in substrate). The absence of such species is likely due to the very limited habitat variability in the

### Summary: Are poor physical habitat conditions a stressor in Little Swan Creek?

Degraded habitat in Little Swan Creek can partially be attributed to channel instability and sedimentation. The substrate and channel morphology sections of the MSHA score and the “unstable” Pfankuch Stability rating support this hypothesis. This degraded habitat is contributing to a decrease in the biological integrity of the stream, as evidenced by the number of non-tolerant fish/meter and the low number of various habitat-sensitive taxa present. It is our conclusion that habitat degradation is a stressor to the fish community in Little Swan Creek.

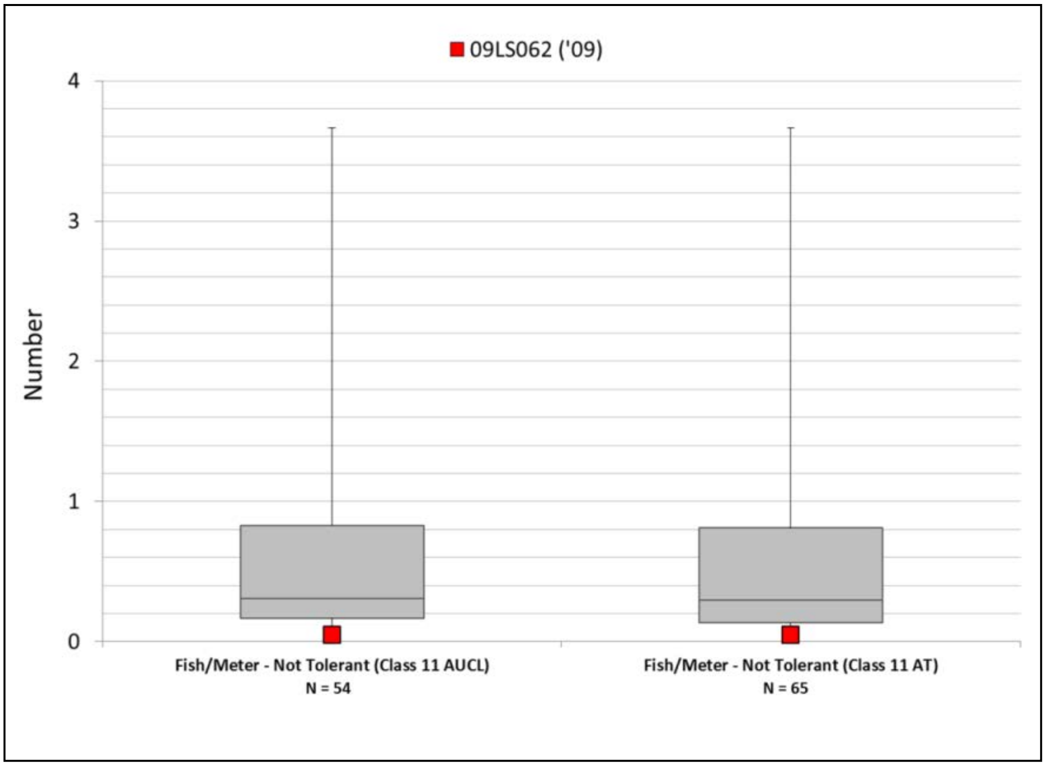


**Figure 181:** Non-tolerant fish/meter in Little Swan Creek compared to high quality SLRW Class 11 stations. AUCL = Above Upper Confidence Limit of FIBI threshold AT = Above Fish IBI Threshold

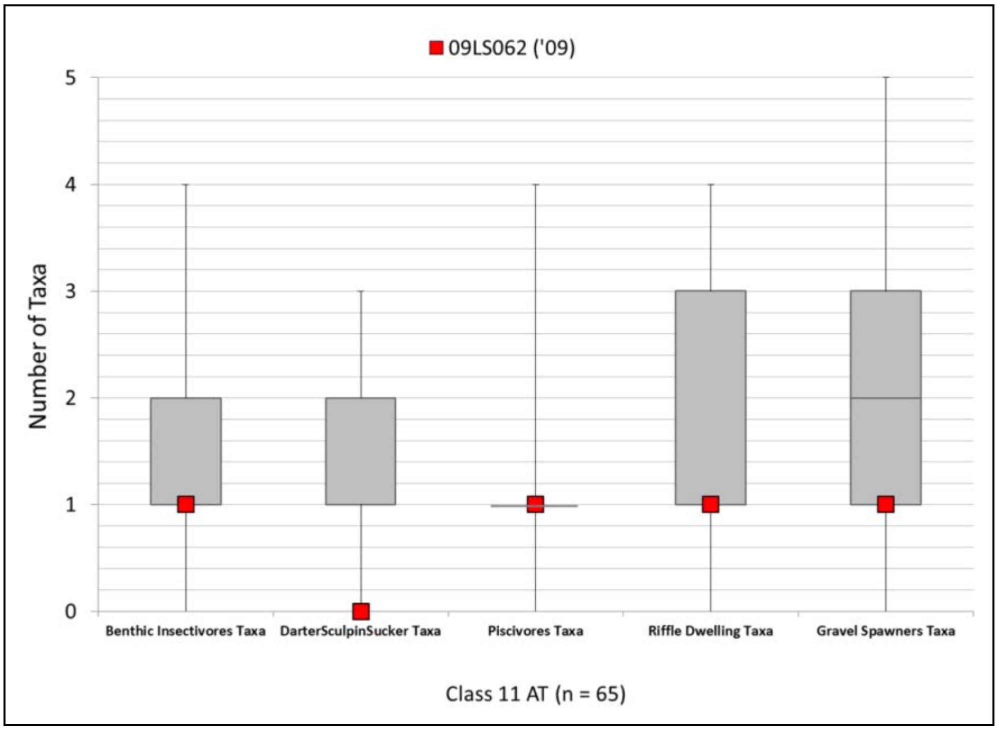


**Figure 182:** Number of fish taxa for various categories in Little Swan Creek compared to SLRW stations above the impairment threshold. AT = Above Fish IBI Threshold

## 5.11.5 Little Swan Creek: Summary of Stressors to Aquatic Life

**Table 53:** Summary of SID results for Little Swan Creek

| Candidate Cause | Result |
|---|---|
| **Elevated Water Temperatures** | ● |
| **Low Dissolved Oxygen** | ● |
| **Total Suspended Solids / Turbidity** | ○ |
| **Poor Physical Habitat Conditions** | ● |
| Key: **• = confirmed stressor ○ = Potential Stressor X = eliminated candidate cause** | |

### Makinen Lakes Watershed Zone

Impaired streams in this watershed zone include Water Hen Creek, Water Hen River, and Paleface Creek. All of these streams are low gradient and feature predominantly glide-pool habitats. This watershed zone contains numerous lakes and wetlands, and most of the streams within it are connected to these features. Generally, the lakes of area are in relatively good condition. However, several of the lakes that are hydrologically connected to the impaired streams are impacted by elevated phosphorous concentrations. The outlet of Dinham Lake, which is impaired for excess phosphorous, enters Paleface Creek just upstream from the impaired reach of that stream and nutrient-impaired Long Lake serves as the headwaters of Water Hen River.

**Figure 183:** Examples of impaired streams in the Makinen Lakes Watershed zone. Paleface Creek (left), Water Hen Creek (middle), and Water Hen River (right).

Symptoms of macroinvertebrate impairment were very similar in all of the impaired streams of this watershed zone. At impaired sites, the macroinvertebrate communities tend to be dominated by non-insect taxa. Freshwater amphipods from the genus Hyallela were very abundant, particularly in the impaired reach of Water Hen River. Members of the genus Hyallela are generally tolerant of disturbance and are important in the breakdown of organic matter in streams and rivers (Bouchard Jr. 2004). Aquatic worms (Oligochaeta), snails (*Physa*, *Hydrobiidae*, and *Planorbidae*), and non-biting midges (*Ablabesmia*, *Tanytarsus, Cricotopus*) were also dominant in samples collected from these streams. Aquatic insect taxa were not abundant or diverse at these locations. The insect taxa present were well-adapted for living in slow-moving or stagnant streams with wetland qualities, and included narrow-winged damselflies (*Coenagrionidae*), small minnow mayflies (*Baetidae*), and prong-gilled mayflies (*Leptophlebiidae*).

Paleface Creek is the only stream impaired for low FIBI in this watershed zone. The fish community in this stream is extremely limited in terms of species diversity and overall fish abundance. Only six species of fish were observed in two sampling visits to this stream; Black Bullhead, Central Mudminnow, Tadpole Madtom, White Sucker, Northern Pike, and Pumpkinseed sunfish. Central Mudminnow individuals accounted for 83% and 66% of the total fish counted during the sampling visits. This fish assemblage is typical of a low gradient stream with significant wetland influence. Extremely low DO values were observed in Paleface Creek throughout most of the open-water season (April – November).

## 5.12 Water Hen Creek / Water Hen River

The headwaters of Water Hen Creek lie in the glacial deposits approximately six miles north of the Whiteface Reservoir. The stream follows a similar stream and valley pattern to other rivers in the area, wherein long narrow lacustrine valleys with E channels are punctuated by shorter, steeper glacial trough valleys with C or B channels. This pattern is a result of the regional geology and glacial activity during the last ice age. Also of interest is a more than 3-mile channelized reach upstream of Highway 16. There are two impaired AUID's on Water Hen Creek. The upstream impaired reach is mostly a flat (0.08%) sinuous E channel in a lacustrine valley. The downstream reach, which is just upstream of the confluence with Mud Hen Creek, is mostly a wider, flat (0.02%) C channel, but has a steeper section in the middle that types out as a 1.9% B channel. The average slope of Water Hen Creek is fairly flat, at 0.09% (4.5 feet/mile).

The MIBI impairment listings for Water Hen Creek and Water Hen River are based on data from two monitoring stations (09LS094 and 09LS092). Station information and MIBI results are summarized in Table 54, and the location of these monitoring stations in the watershed are displayed in Figure 184. The MIBI results obtained from these two stations are all below the MIBI impairment threshold, but the station located on Water Hen River (09LS092) was within the confidence limit of the MIBI criteria and is appears to be less degraded than the station on Water Hen Creek (09LS094). Station 10EM121 was sampled in 2010 and scored narrowly above the impairment threshold. However, these results were not convincing enough to prevent an impairment listing.

The macroinvertebrate community at Water Hen Creek station 09LS094 was dominated by mostly "legless" taxa such as *Hyallela* (freshwater amphipods), *Oligochaeta* (aquatic worms), and *Planorbidae* (air-breathing freshwater snails). Over 80% of the taxa observed at this station are considered "tolerant" or poor water quality and/or degraded habitat conditions. The low overall MIBI score at this station is primarily the result of poor metric scores based on a lack of "clinger" taxa, low numbers of "collector-filterer" individuals, and an unbalanced community in which the five most abundant taxa account for a large percentage of the overall community.

*Hyallela* was also the most abundant taxon observed at the Water Hen Creek monitoring station as well. Also present in high numbers at this monitoring station were *Coenagrionidae* (narrow-winged damselflies), *Hydrobiidae* (mud snails), *Physa* (air-breathing freshwater snails), *Oligochaeta* (aquatic worms), and *Paraleptophlebia* (prong-gilled mayflies).

**Table 54:** Summary of biological monitoring stations, biological sampling results, and applicable standards

| **Macroinvertebrate Assessments** | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Station** | **Drainage Area (mi2)** | **Gradient (%)** | **Stream Order (Strahler)** | **Invert IBI Class** | **Invert IBI Result *(visit year)*** | **Invert IBI Result *(visit year)*** | **Standard** | **IBI Lower Confidence Limit** | **IBI Upper Confidence Limit** |
| **09LS094** | 15.86 | 0.10 | 2 | 4 | **33.79** *(2009)* | - | 51 | 37.40 | 64.60 |
| **10EM121** | 49.36 | 0.04 | 3 | 4 | **52.23** *(2010)* | - | 51 | 37.40 | 64.60 |
| **09LS092** | 68.47 | 0.05 | 3 | 4 | **42.24** *(2009)* | **37.58** *(2009)* | 51 | 37.40 | 64.60 |

A review of available water chemistry, biological, and physical habitat data was performed to develop a list of candidate causes for the MIBI impairments in this watershed. The following candidate causes were selected for further analysis:

### 1. Low DO

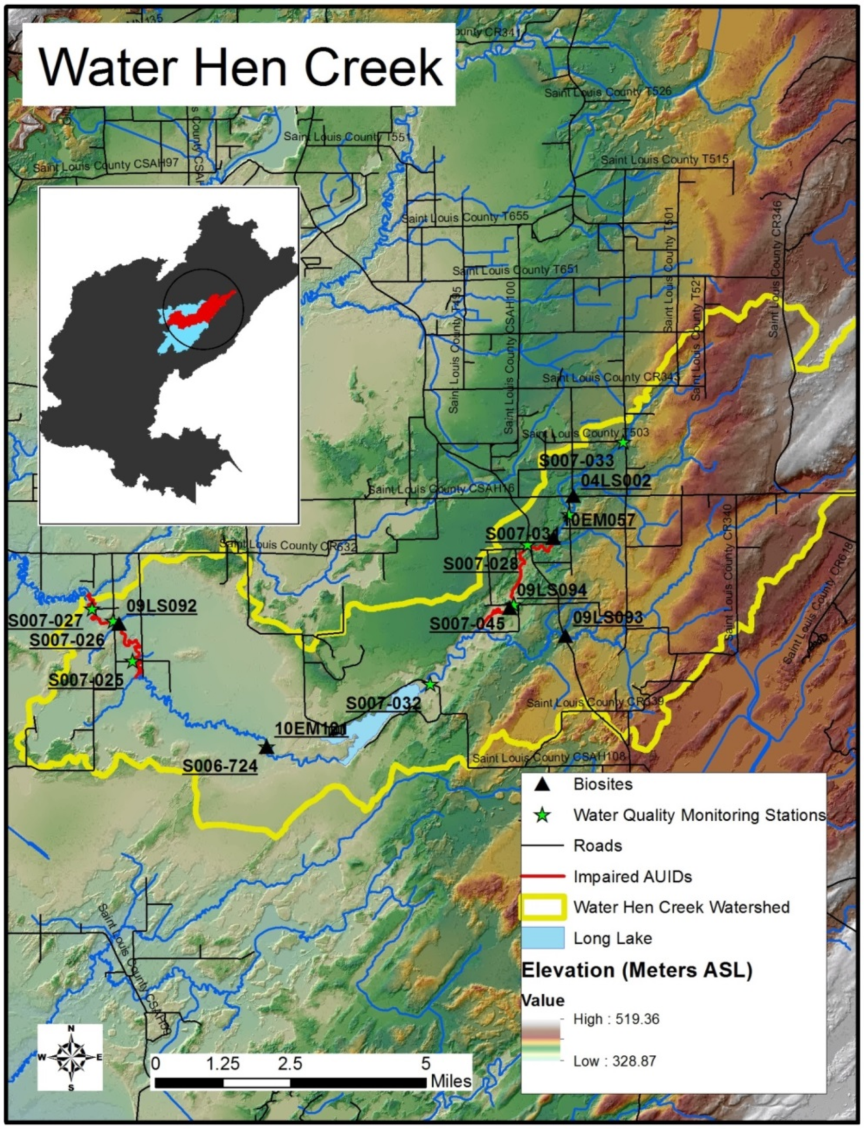


**Figure 184:** Map of Water Hen River Watershed and impaired stream reach

### 5.12.1 Low Dissolved Oxygen

Low DO concentrations were identified as a candidate cause of biological impairment in Water Hen Creek and Water Hen River. Results from instantaneous (point) measurements of DO collected at the two biological monitoring stations between the years of 2009 and 2013 are shown in Figure 185. DO concentrations are lowest during the months of June and July, but are not limited to low flow conditions. Sags in DO have been observed in this stream during and following large June rain events when flushing of the abundant wetlands in the watershed has occurred.

A longitudinal synoptic survey of DO concentration was completed in July of 2011. One sampling run was conducted during the early morning hours before sunlight took full effect, and the other sampling run was carried out in the late afternoon. DO concentrations were low (less than 3 mg/L) in the upper reaches of Water Hen Creek during both the early morning and afternoon sampling runs (Figure 186). This reach includes biological monitoring station 09LS094, where DO concentrations were around 2.5 mg/L. DO concentrations were more suitable at station S007-032, which is located just upstream of Long Lake. The increase in DO observed at this station may be attributed to the addition of the South Branch of Water Hen Creek, which enters just upstream of this monitoring site. Overall, very little change in DO concentration was observed at stations in upper Water Hen Creek between the two sampling events.

DO concentrations dropped significantly downstream of Long Lake during the July 2011 longitudinal sampling run (Figure 186). Due to limited access, the monitoring station downstream of the lake (S007-025) is located approximately five miles downstream of the lake outlet, making it difficult to determine if the observed decrease in DO concentration was driven by lake or stream processes. Several large beaver dams are located in the reach between the lake outlet and monitoring station S007-025, which may be contributing to low DO concentrations. Less than two miles downstream at station S007-026 (biological monitoring station 09LS092), DO concentrations recovered to levels just below the 5 mg/L water quality standard, and increased further just downstream at monitoring station S007-027. Similar to the upper reaches of Water Hen Creek, very little diurnal fluctuation in DO concentration was observed in the lower portion of the creek below Long Lake.

The DO concentrations were monitored continuously for a period of several days at the two biological monitoring stations. These continuous DO profiles were collected in June of 2012 immediately after a major rain event dropped 3-6 inches of rain in the Water Hen Creek Watershed, and again in September of 2012 during low flow conditions. Both of the profiles from the June 2012 high flow event show low DO concentrations and minimal DO flux (Figure 187). The September 2012 profile from monitoring station S007-045 is drastically different, as DO concentrations were much higher, and diurnal DO flux exceeded 9.0 mg/L. Diurnal fluctuation in DO concentrations of more than 4 mg/L can negatively impact aquatic macroinvertebrate communities (Heiskary et al 2013).

**Figure 187:** The June 2012 DO profile was conducted during high flows on the falling limb of the hydrograph after the 2012 flood event (left). An additional DO profile was conducted during low flows during September 2012 (right).

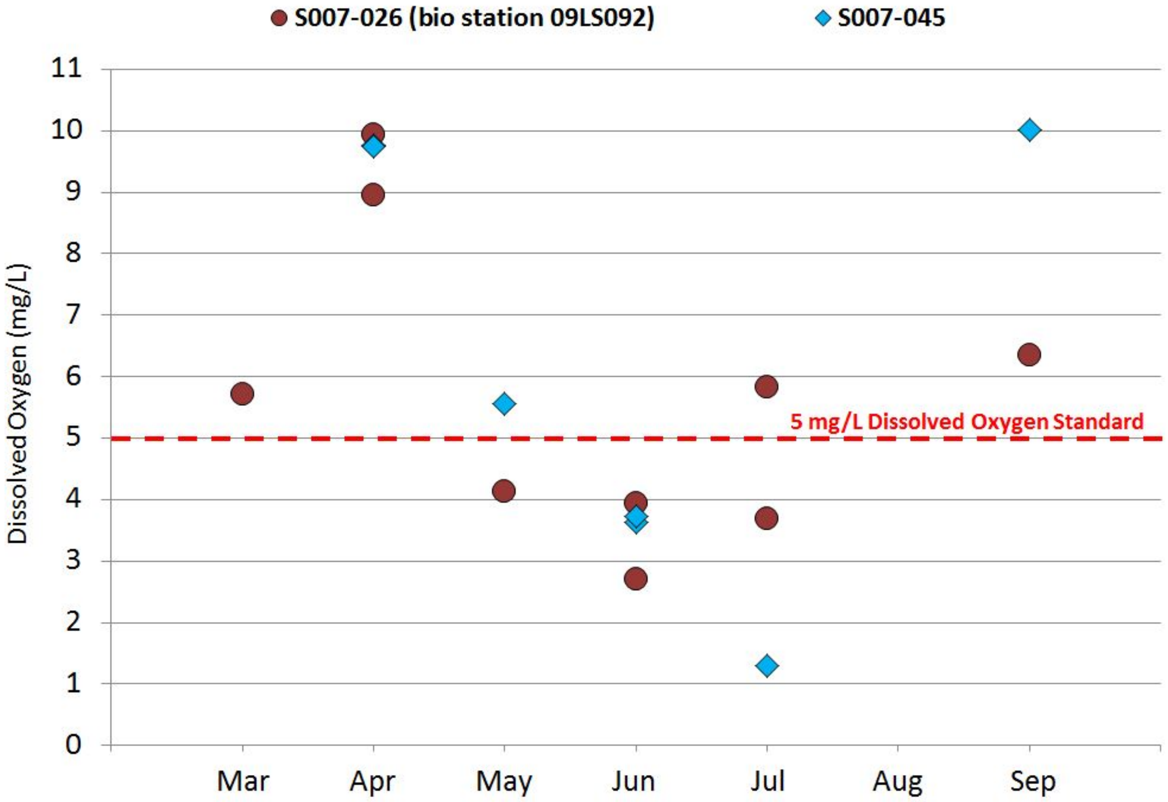


**Figure 185:** Point measurements of DO collected on Water Hen Creek / Water Hen River.

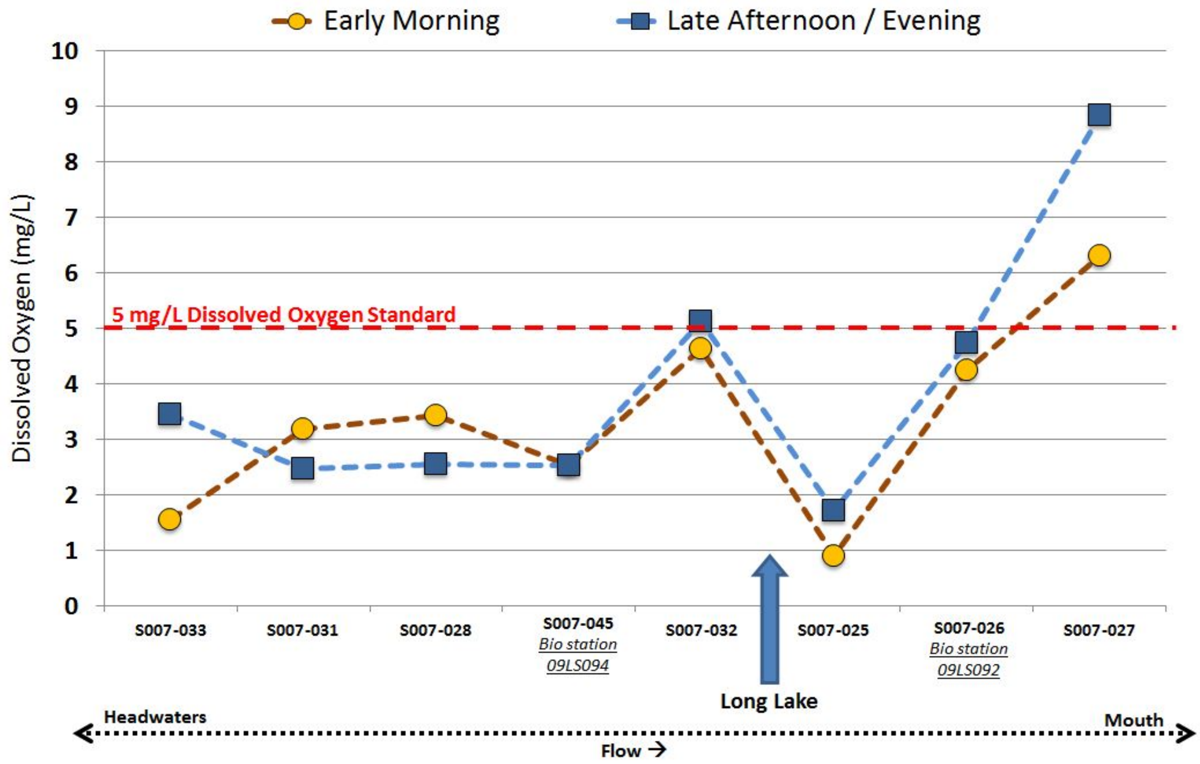


**Figure 186:** Results from the 7/29/2011 longitudinal synoptic DO profile.

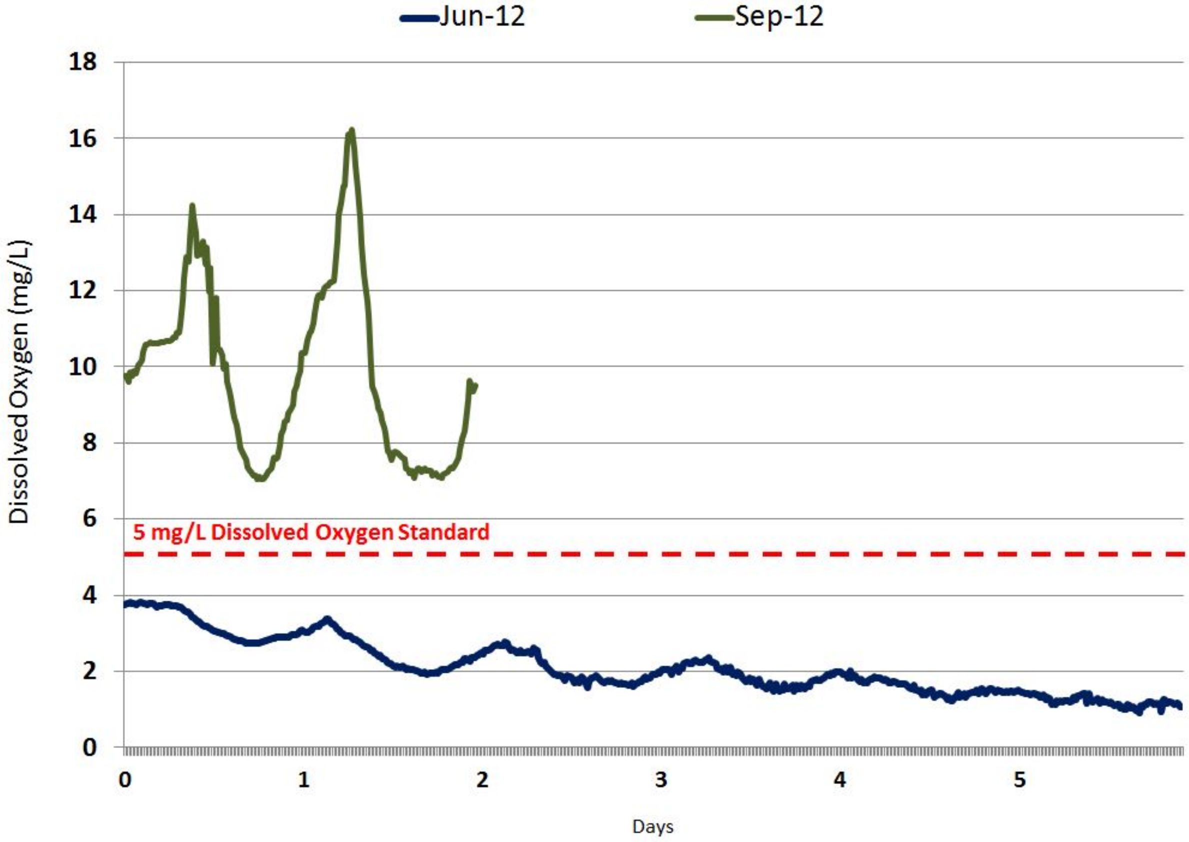


**Figure 187:** Continuous DO profile results from biological monitoring station 09LS094

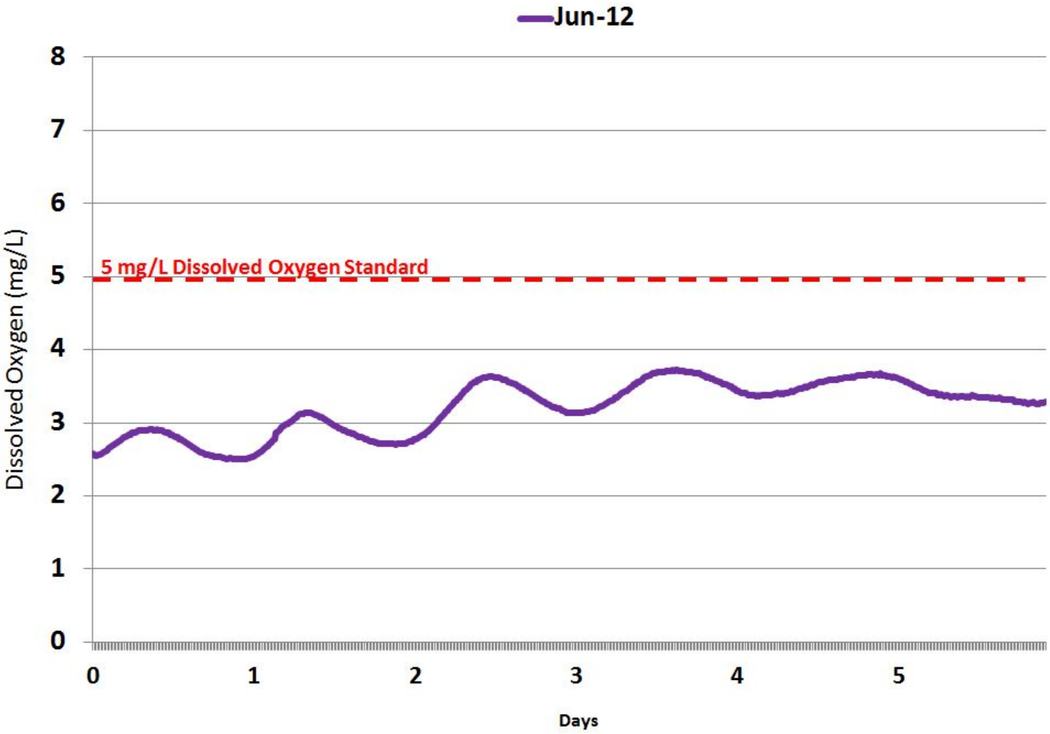


**Figure 188:** Continuous DO profile results from biological monitoring station 09LS092

## Sources and Pathways Contributing to Low Dissolved Oxygen

### Wetlands

Wetlands are a prominent land cover type found in watersheds of Water Hen Creek, Water Hen River, and Mud Hen Creek (Figure 189). Many of the wetland areas in these watersheds are expansive, poorly drained areas with hydric peat soil classifications. Hydric soils are permanently or seasonally saturated by water, often resulting in anaerobic (no oxygen) conditions. The entire impaired reach of Water Hen Creek, from Long Lake to its confluence with Mud Hen Creek, is bordered by a wetland dominated riparian corridor with partially hydric or all hydric soil classifications. Hydric peat soils are particularly abundant and concentrated in the lower portion of the Water Hen Creek Watershed. A large wetland complex with hydric peat soil type is located just upstream and adjacent to the lower impaired reach. It is likely that this wetland complex is delivering anoxic water to this reach of Water Hen Creek.

#### Nutrients and Productivity

The TP concentrations in Water Hen Creek are slightly elevated, and occasionally exceed river nutrient criteria of 0.055 mg/L (Figure 190). Several of the elevated TP results on record have occurred during major runoff events or immediately after these events. These elevated TP results may be related to the flushing of nutrient rich wetlands and/or lakes in the watershed. However, elevated TP concentrations have also occurred during lower flows in Water Hen Creek and tributary streams. Samples collected at biological monitoring stations 09LS092 and 09LS093 during low flows resulted in TP concentrations of 0.075 mg/L and 0.080 mg/L, respectively. TP concentrations in Water Hen Creek have shown to be higher than the adjacent, non-impaired Mud Hen Creek, possibly due to the greater wetland presence in the Water Hen Creek drainage.

Nutrient-impaired lakes are another potential source of TP in this watershed. Water Hen Creek flows through Long Lake (impaired for excess nutrients), which may result in higher nutrient concentrations in the lower reaches of the creek during certain times of the year. The TP data available are insufficient for evaluating the effects of Long Lake on the nutrient dynamics of lower Water Hen Creek.

Despite elevated TP concentrations, DO flux in Water Hen Creek is typically low (0.5 – 2 mg/L), with the exception of the September 2012 DO profile from upper Water Hen Creek (see Figure 187). The tannin stained water of this stream may limit sunlight penetration and reduce primary productivity. Low DO flux in streams with wetland dominated watersheds and tannin stained water is a common observation in the SLRW.

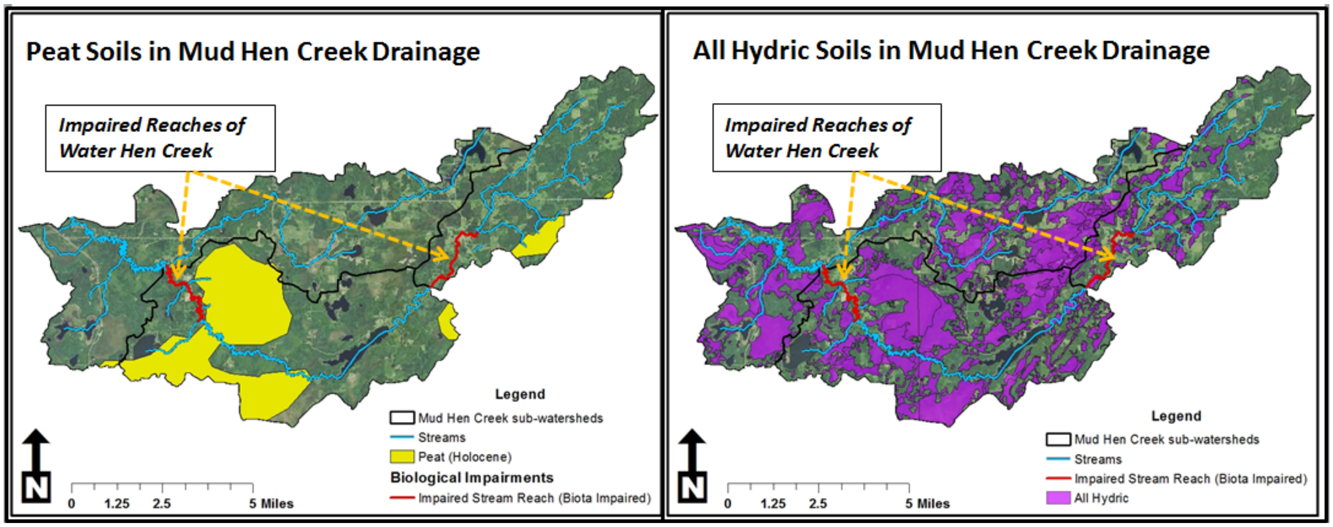
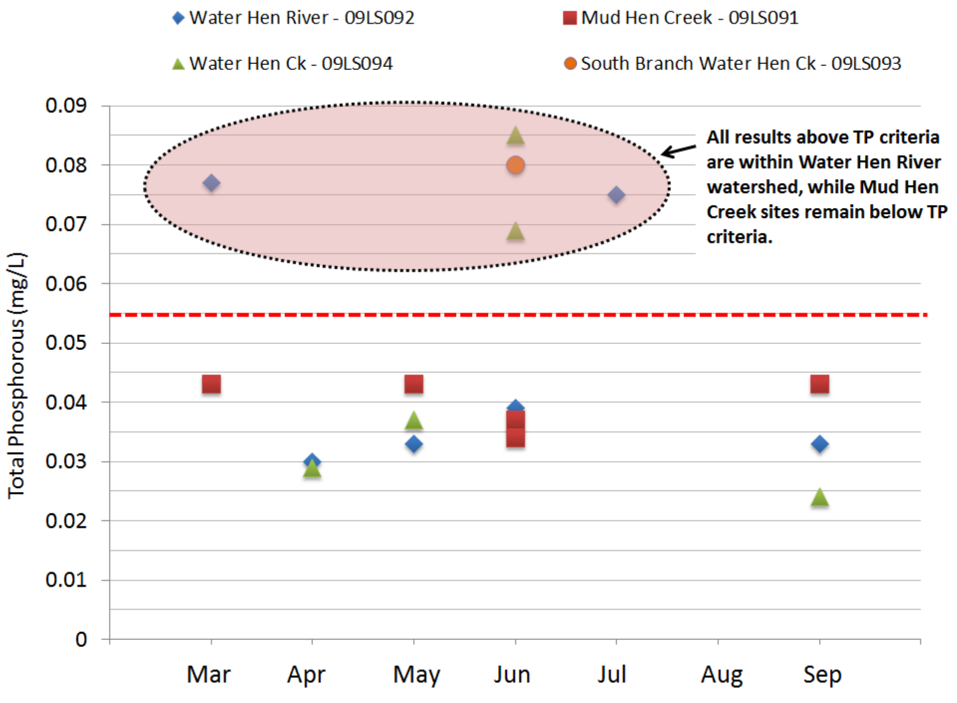


**Figure 189:** Coverage of peat and all hydric soil types in Mud Hen Creek / Water Hen Creek Watershed



**All results above TP criteria are within Water Hen River watershed, while Mud Hen Creek sites remain below TP criteria.**

**Figure 190:** TP results for Water Hen Creek and Mud Hen Creek Watershed

### Biological Response to Low Dissolved Oxygen

The impaired reaches of Water Hen Creek and Water Hen River are both dominated by macroinvertebrate taxa that are tolerant of low DO conditions (Figure 191). Between 66% and 89% of the individuals sampled from these reaches are either moderately or highly tolerant of low DO. The amphipod *Hyallela* was the most abundant taxon at all monitoring stations. These organisms are considered highly tolerant of low DO concentrations among other stressors (nutrient enrichment, poor physical habitat). Freshwater snails from the genera *Ferrissia*, *Planorbidae*, *Physa*, which are also known to tolerate low DO concentrations, were also common within the impaired segments of these two streams.

While the majority of the macroinvertebrate data indicates a community that is tolerant of low DO, only one reach of Water Hen Creek that displayed more DO sensitive macroinvertebrate community. The macroinvertebrate community at station 10EM121 appears to be somewhat less tolerant of low DO. Four macroinvertebrate taxa that are sensitive to low DO levels were observed at this station. Nearly 20% of the individual organisms sampled at this location belong to taxa that are sensitive to low DO. Still, over 50% of the organisms at this station can be considered moderately or highly tolerant of low DO conditions.

Macroinvertebrate community DO Index values in Water Hen Creek are generally lower than comparable reference streams in the SLRW (Figure 192). The DO Index scores at station 09LS92, located near the mouth of Water Hen River, were below all class 4 sites scoring above the UCL and well below the 25th percentile scores for class 4 stations scoring above the impairment threshold. The exception is station 10EM121, where the macroinvertebrate community DO index score is comparable to the majority of high-quality class 4 macroinvertebrate stations. The DO data at this monitoring station are limited to a single measurement taken at the time of sampling in August 2010 (8.33 mg/L).

Another piece of evidence in support of low DO as a stressor in this watershed is the lack of EPT taxa richness and low overall EPT abundance observed in Water Hen Creek. The percent of individuals belonging to EPT taxa ranged from a minimum of 6.3% (09LS092, Sept ’11) to a maximum of 13.5% (10EM121, August 2010), with an average value of 8.3% among the four sampling visits to the three monitoring stations. In comparison, the average EPT % at class 4 stations that scored above the impairment threshold is 27.0% (n=45, 25th percentile =13.5%, min=2.2%, max=69.3%). Low DO concentrations are likely one of several factors limiting EPT richness and abundance in Water Hen Creek.

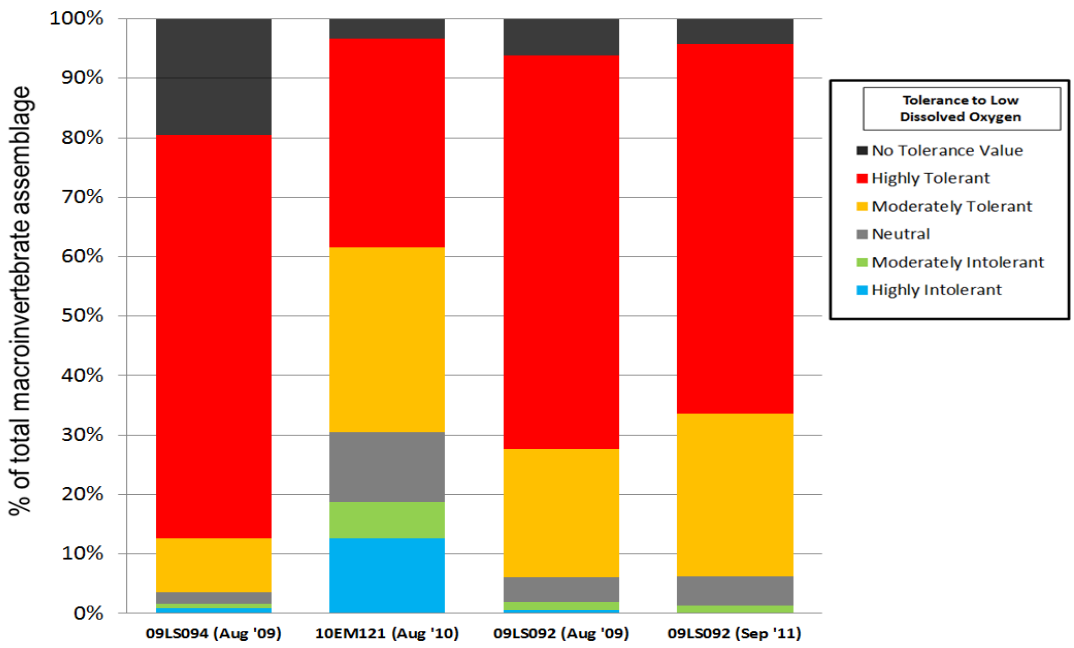


**Figure 191:** Proportion of macroinvertebrates tolerant, neutral, or intolerant of low DO concentrations

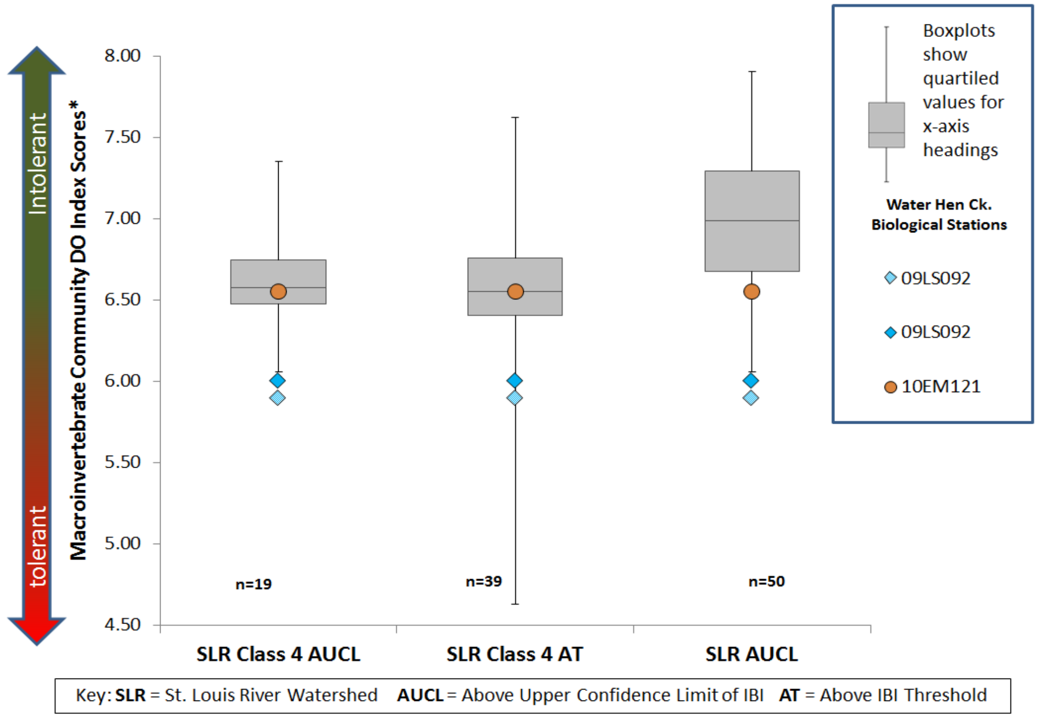


**Figure 192:** Invertebrate community DO TIV results compared to results from high quality stations of the same IBI class. * See Section 4 for explanation of TIVs.

**Summary: Is low dissolved oxygen a stressor in Water Hen Creek?**

Water chemistry and biological data from provide adequate evidence to diagnose low DO as a stressor in both impaired reaches of Water Hen Creek. The DO concentrations regularly fall below the 5 mg/L water quality standard, and extremely high DO flux has been observed in the upper impaired reach (Water Hen Creek). The macroinvertebrate community shows a high percentage of organisms that are highly tolerant of low DO conditions, which is also supported by the community level TIV results.

## 5.12.2 Water Hen Creek: Summary of Stressors to Aquatic Life

**Table 55:** Summary of SID results for Water Hen Creek and Water Hen River

| Candidate Cause | Result |
|---|---|
| **Low Dissolved Oxygen/High DO Flux** | ● |
| Key: ● **= confirmed stressor** ○ **= Potential Stressor X = eliminated candidate cause** | |

# 5.13 Paleface Creek

The origin of Paleface Creek is in the Wabuse and Washusk chain of lakes that lie three miles southwest of the Whiteface reservoir. The stream follows a fairly consistent pattern of alternating stream and valley types. Northeast to southwest trending lacustrine valleys with flat sinuous E channels are interrupted by shorter and steeper glacial trough valleys with “C-type” channels (Rosgen 1994) flowing through them. Paleface Creek has several flow-through lakes, including Wabuse Lake, Washusk Lake Number One, Washusk Lake Number Two, and Morcom Lake. The impaired AUID of Paleface Creek is the outlet of these lake systems, flowing for five miles before it empties into the Paleface River. This reach has no variation in stream or valley type, and is solely an E-type channel in a lacustrine valley. The average slope of the river is 0.06% (three feet per mile) and the slope of the impaired AUID is slightly lower, at 0.04%.

Paleface Creek is impaired for failing to meet FIBI and MIBI standards. Station information, sampling results, and applicable IBI criteria are outlined in table 56. All biological sampling was completed in 2009 at one monitoring station in the lower reaches of Paleface Creek, approximately 0.9 river miles upstream of its confluence with the Paleface River.

The fish community of Paleface Creek at station 09LS049 exhibited low taxa richness and a general lack of individuals. An average of less than 25 fish was collected over the two fish sampling visits in 2009. Central Mudminnow was the dominant taxa observed, accounting for 67% and 83% of the total population during the two visits. Other fish species observed in low numbers at this monitoring station included Northern Pike, White Sucker, Black Bullhead, Pumpkinseed sunfish, and Tadpole Madtom. The vast majority of the fish sampled are common to low gradient streams and environments with low DO concentrations.

The macroinvertebrate community at 09LS049 was dominated by *Oligochaeta* (aquatic worms) and *Hyallela* (freshwater amphipods). These two taxa accounted for approximately 56% of the organisms observed from this station. Over 94% of the total organisms sampled from this station can be considered tolerant of poor water quality and/or degraded habitat conditions.

**Table 56:** Summary of biological monitoring stations, biological sampling results, and applicable standards

**Fish Assessments**

| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | FIBI Class | FIBI Result *(visit year)* | FIBI Result *(visit year)* | Standard | IBI Lower Confidence Limit | IBI Upper Confidence Limit |
|---|---|---|---|---|---|---|---|---|---|
| **09LS049** | 29.45 | 0.03 | 2 | 7 | **31** *(2009)* | **21** *(2009)* | 42 | 32 | 52 |

**Macroinvertebrate Assessments**

| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | Invert IBI Class | Invert IBI Result *(visit year)* | Invert IBI Result *(visit year)* | Standard | IBI Lower Confidence Limit | IBI Upper Confidence Limit |
|---|---|---|---|---|---|---|---|---|---|
| **09LS049** | 29.45 | 0.03 | 2 | 4 | **38.79** *(2009)* | - | 51 | 37.40 | 64.60 |

Water quality and physical habitat data were used to develop a list of candidate causes for the FIBI and MIBI impairments observed in Paleface Creek. The following candidate causes were identified for further analysis in this section:

1. **Low DO**

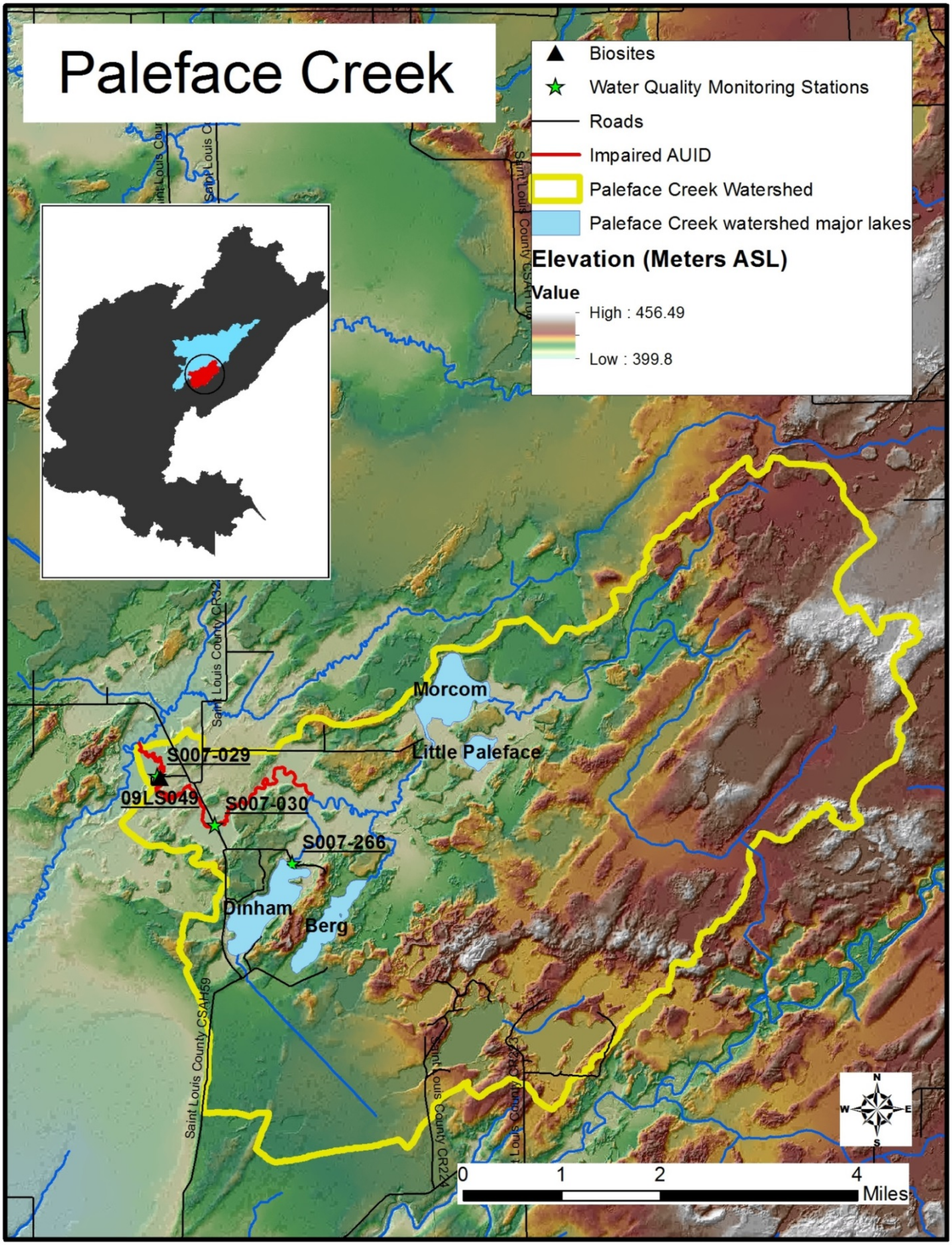


**Figure 193:** Paleface Creek Watershed and impaired stream reach

## 5.13.1 Low Dissolved Oxygen

The DO was identified as a candidate cause of impairment in Paleface Creek after a review of existing data. Severely low DO concentrations (less than 1 mg/L) are common during mid-summer and early fall (Figure 194). Over 90% (12/13) of the DO measurements collected from Paleface Creek were below the 5 mg/L DO standard. Of the readings below 5 mg/L, 60% were less than 2 mg/L. In addition to the one-time measurements, DO concentrations were monitoring continuously for a period of approximately four days in August of 2012. Results were below 1 mg/L for the entire four-day monitoring period, and showed very little diurnal fluctuation due to the overall lack of productivity in this stream, which is heavily influenced by wetland and bog land cover within its watershed.

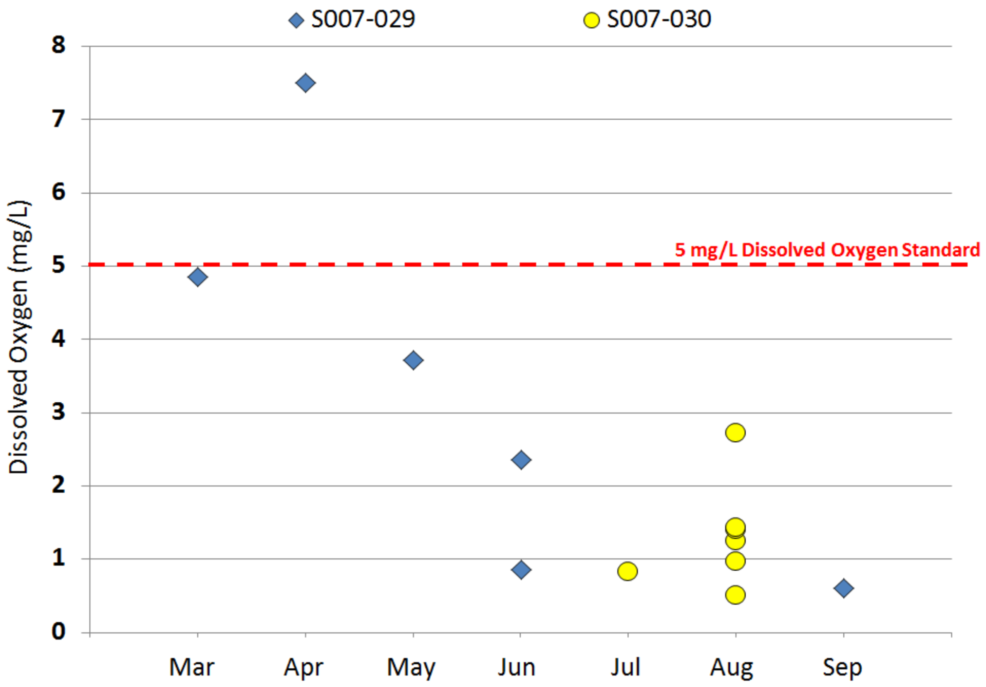


**Figure 194:** Point measurements of DO collected from Paleface Creek.

### Sources and Pathways Contributing to Low Dissolved Oxygen

Paleface Creek drains a watershed with flat topography and an abundance of shallow lakes and wetlands. Over 50% of the watershed area is comprised of wetlands (45%) and open water (7%). The lakes in this watershed are generally shallow and tannin stained. Dinham Lake, which outlets to a tributary stream flowing north into Paleface Creek, is impaired for excess nutrients and failing to meet the MPCA’s lake eutrophication criteria. Less than 2% of the watershed area is developed, with the majority of the developed land located around Dinham and Berg Lake.

The TP concentrations in Paleface Creek are slightly elevated, with several samples exceeding the draft river nutrient TP criteria of 0.055 mg/L. TP data for Paleface Creek are limited, and many of the results are from high flow periods (snowmelt, rain events). A maximum TP concentration of 0.071 mg/L was observed in July of 2009 (n=7, min=0.019 mg/L, mean=0.038 mg/L). Despite TP concentrations above the river nutrient criteria standards, primary productivity in Paleface Creek appears to be limited based on very low diurnal DO flux and sparse to moderate density of aquatic macrophytes in the stream channel. Paleface Creek is extremely tannin stained and the lower water transparency also serves a limiting factor for primary productivity.

The low DO concentrations observed in Paleface Creek are likely driven by interactions between the stream and its adjacent wetlands. The entire length of Paleface Creek is bordered an extensive riparian corridor consisting of woody and emergent herbaceous wetlands (Figure 195). Nearly all of the adjacent wetlands can be categorized as Organic Flat HGM type wetlands. The MPCA biologists who sampled this stream reported 1-2 feet of pure detritus as substrate. Decomposition, not eutrophication, is the primary driver of low DO concentrations observed in this stream.

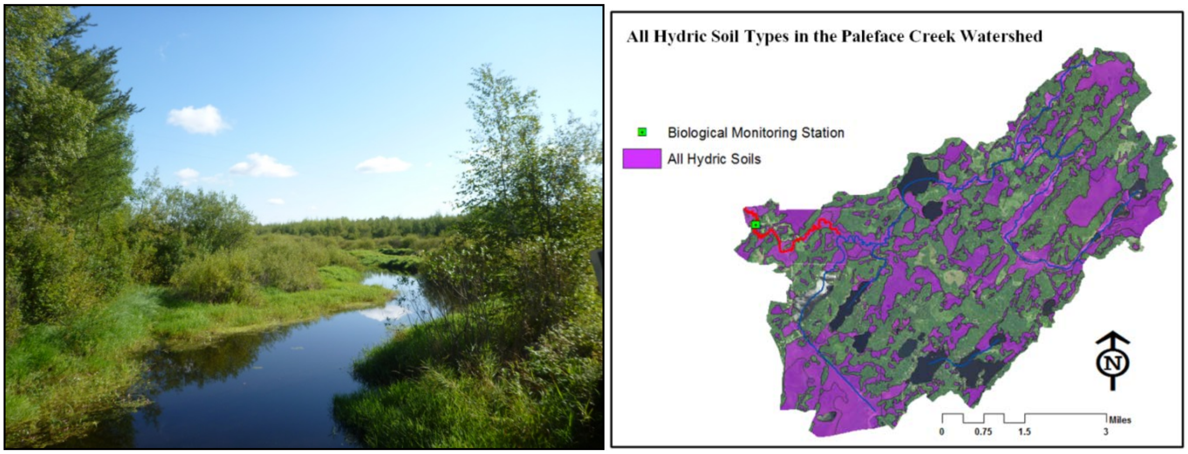


**Figure 195:** Low gradient reach of Paleface Creek bordered by wetlands (left). Map of all-hydric soils in the Paleface Creek Watershed.

## Biological Response to Low Dissolved Oxygen

The fish community observed at Paleface Creek station 09LS049 is dominated by species that are very tolerant of low DO conditions. Central Mudminnow was the most abundant species observed, accounting for 63% and 83% of the fish community during the two sampling visits. Other species that were present in very small numbers include Black Bullhead, White Sucker, Tadpole Madtom, Pumpkinseed sunfish, and Northern Pike. Aside from White Sucker and Tadpole Madtom, every species of fish observed in Paleface Creek can be considered highly tolerant of low DO conditions. Based on data from the two fish sampling visits, approximately 87% to 97% of the fish observed in Paleface Creek are from species known to be highly tolerant of low DO.

Station 09LS049 scored poorly in the fish community DO index, which is another tool used to assess the overall tolerance level of the fish community to low DO. Figure 197 shows a comparison of the DO index score at 09LS049 to scores from reference streams of the same FIBI class in the SLRW. The DO Index scores at 09LS049 (5.8 and 5.6) fall near the bottom of the lower quartile values observed at comparable reference stations. It is clear from this comparison that the Paleface Creek fish community is more tolerant of low DO conditions than high quality streams of the same FIBI class.

In addition, Paleface Creek scored poorly in several other FIBI metrics that are often linked to low DO as a stressor. Station 09LS049 received poor scores for metrics related to overall fish abundance, a lack of headwaters minnow species, and a lack of sensitive fish species. Although these metrics can be responsive to a variety of stressors, these results are further evidence in support of low DO as a stressor in Paleface Creek.

### Macroinvertebrates

Nearly 80% of the macroinvertebrates sampled at station 09LS049 on Paleface Creek belong to taxa that are moderately to highly tolerant of low DO (Figure 196). Other than aquatic worms (Oligochaeta), amphipod crustaceans (Hyallela) were the most abundant macroinvertebrate at this station, accounting for nearly 40% of the total organisms sampled. Members of this genus are considered highly tolerant of low DO conditions and elevated nutrient concentrations. Hyalella individuals feed primarily on decaying organic matter, which appears to be abundant in Paleface Creek due to the low gradient, wetland setting the creek passes through. Other macroinvertebrate taxa that were abundant at this monitoring station included several genera of non-biting midges from the family Chironomidae; *Cricotopus*, *Ablabesmyia*, *Tanytarsus*. These taxa can be considered neutral to moderately tolerant of low DO levels.

The macroinvertebrate community DO Index value at station 09LS049 is lower than the scores observed at comparable reference sites (Figure 198). The low DO index score driven by two factors at this station. First, it supported only one taxa that is considered intolerant of low DO concentrations. Second, 50% of the total macroinvertebrate community was comprised of individuals belonging to taxa that are tolerant of low DO conditions.

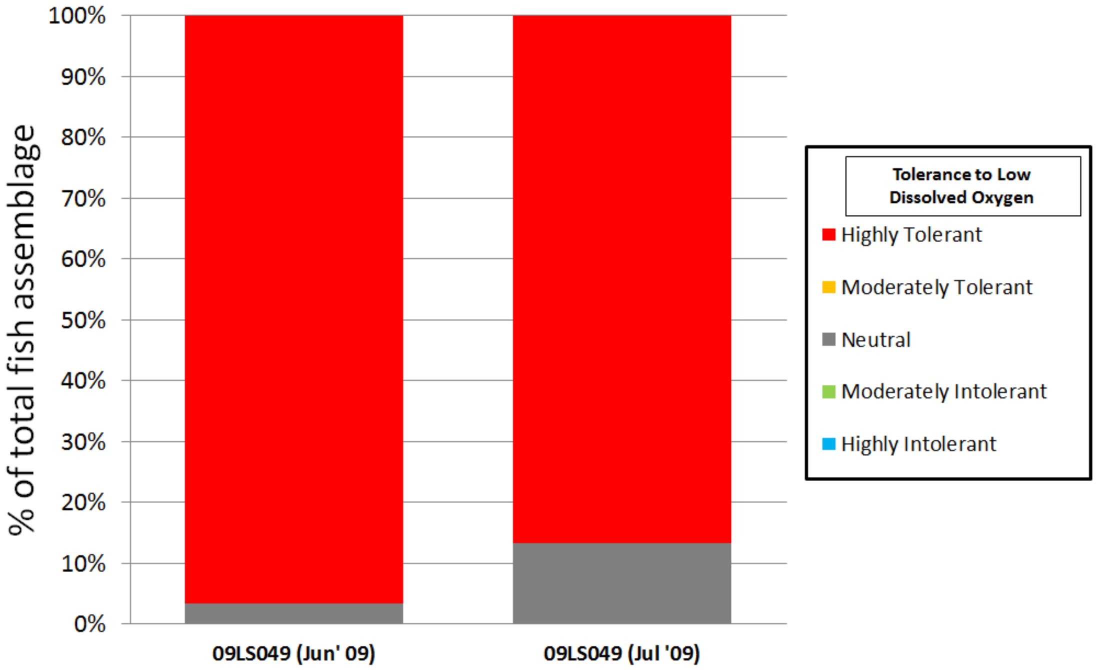
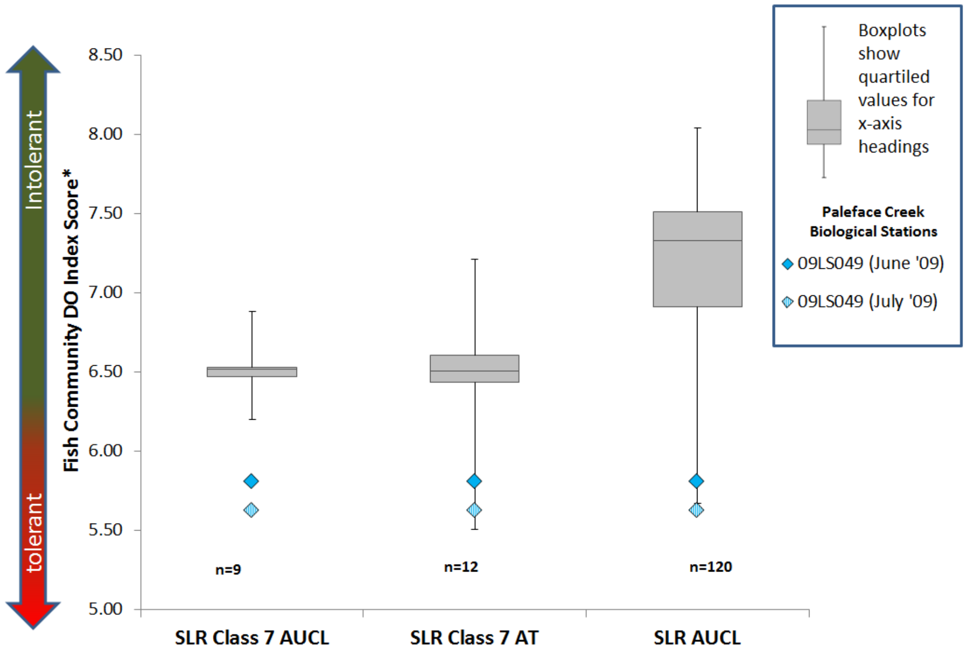
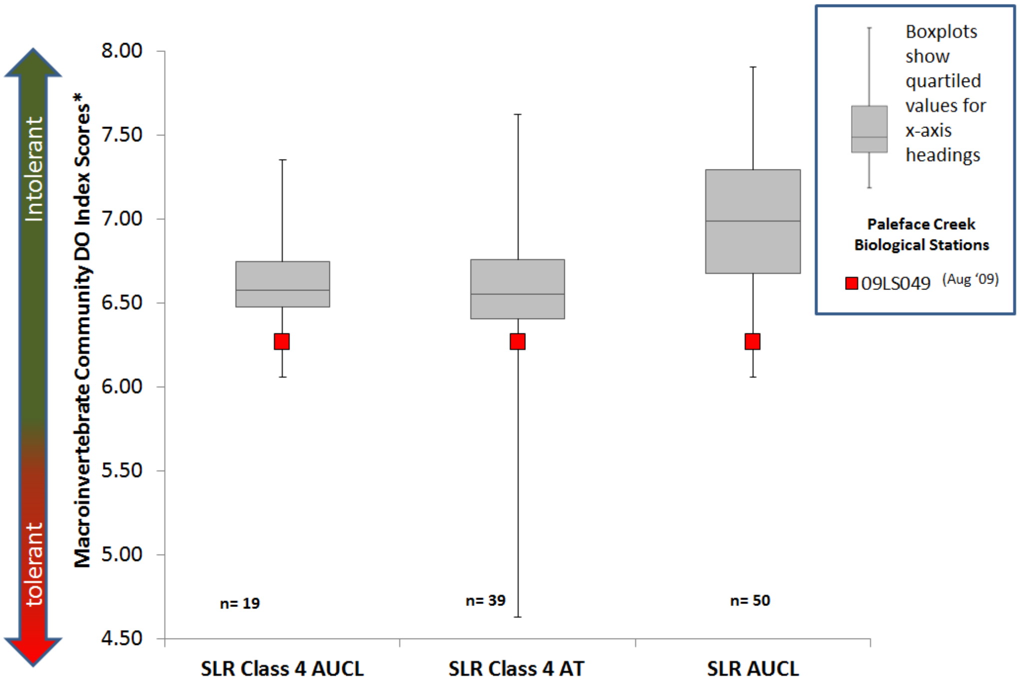


**Figure 196: (Left)** Percentage of fish in each DO tolerance class observed at station 09LS049. **(Right)** Fish community DO tolerance indicator value results compared to high quality reference streams of the same IBI class and greater SLRW.



Key: **SLR** = St. Louis River Watershed **AUCL** = Above Upper Confidence Limit of IBI **AT** = Above IBI Threshold

**Figure 197:** Fish community DO TIV results compared to results from high quality stations of the same IBI class. ***** See Section 4 for explanation of TIVs. TIVs **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above FIBI Threshold



Key: **SLR** = St. Louis River Watershed **AUCL** = Above Upper Confidence Limit of IBI **AT** = Above IBI Threshold

**Figure 198:** Invertebrate community DO TIV results compared to results from high quality stations of the same IBI class. ***** See Section 4 for explanation of TIVs. **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above FIBI Threshold

**Summary: Is low dissolved oxygen a stressor in Paleface Creek?**

Biological data offer firm support of low DO as a stressor in the impaired reach of Paleface Creek. The fish and macroinvertebrate communities are both symptomatic of a DO-limited environment, with low taxa richness and abundance measures, as well as communities that are dominated by taxa that are tolerant of low DO concentrations.

## 5.13.2 Paleface Creek: Summary of Stressors to Aquatic Life

**Table 57**: Summary of SID results for Paleface Creek

| Candidate Cause | Result |
|---|---|
| Low Dissolved Oxygen | ● |
| Key: • = **confirmed stressor** ○ = **Potential Stressor** **X = eliminated candidate cause** | |

## Laurentian Uplands – Partridge River Watershed Zone

# 5.14 Wyman Creek

Wyman Creek is the lone impaired stream in the Laurentian Uplands – Partridge Headwaters (LU-P) Watershed zone. The headwaters of Wyman Creek originate from a series of abandoned mine pits which deliver water to the creek at a fairly constant rate all year round. The influence of these mine pits on the water quality, temperature, and physical habitat of this stream will be discussed later in this report. Historically, small populations of Brook Trout have been sampled in the lower reaches of Wyman Creek, which are steeper in gradient and dominated by cobble and small boulder substrate. The upper reach of Wyman Creek is a sinuous, low gradient channel meandering through bogs and wetlands. Riffle and run features are extremely limited in the upper ¾ of the stream and substrate is dominated by fines (sand/silt) throughout this reach. Beaver dams were observed throughout the length of the creek during a survey completed in August of 2010. One interesting feature of Wyman Creek is a 1.5 mile-long reach where the stream becomes braided and actually divides into two parallel valleys. Based on the historical presence of Brook Trout, Wyman Creek remains a designated trout stream, despite a lack of trout in the more recent monitoring efforts.

**Figure 199:** Photos from Wyman Creek showing beaver impoundments and diversity of channel types.

Fish were sampled from two monitoring stations on Wyman Creek. The station established in 1981 (81LS008) is located in the lower reaches of the stream, approximately 0.5 river miles upstream of its confluence with the Partridge River. This station was re-sampled in 2009, and the data collected during that visit was used for the assessment process. Station 12LS006 was added in 2012, several years after the creek was listed as impaired water, in order to evaluate the fish assemblage further upstream from station 81LS008. Both of these sites were sampled in 2012. Station information and FIBI results are summarized in Table 58.

**Table 58:** Summary of biological monitoring stations, results, and applicable biological assessment criteria.

| Fish Assessments | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | FIBI Class | FIBI Result *(visit year)* | FIBI Result *(visit year)* | Standard | IBI Lower Confidence Limit | IBI Upper Confidence Limit |
| **12LS006** | 6.92 | 0.31 | 2 | 11 | **53** *(2012)* | - | 35 | 25 | 45 |
| **81LS008** | 10.85 | 0.59 | 2 | 11 | **33** *(2009)* | **46** *(2012)* | 35 | 25 | 45 |

Wyman Creek supports several fish species that are commonly found in high-quality trout streams in Northeastern Minnesota, such as Mottled Sculpin, Longnose Dace, Finescale Dace, and Pearl Dace. However, repeat sampling results show robust populations of fish species that often take over marginal or degraded trout streams. Creek Chub, Black Crappie, Yellow Perch, Blacknose Dace, and Common Shiner are examples of undesirable species that were commonly observed in fish surveys. The presence of these species suggests that the stream is a marginal coldwater stream due to natural background conditions, or that is has been degraded due to anthropogenic stressors. The specific stressors impacting Wyman Creek will be discussed in this section.

A review of existing data was performed to develop a list of candidate causes for the FIBI impairment. The following candidate causes were selected for further evaluation in this section:

1. **Elevated Water Temperatures**
2. **Low DO**
3. **TSS**
4. **Habitat Loss due to Iron Precipitate**
5. **Loss of Connectivity and Suitable Habitat due to Beaver Dams**

## 5.14.1 Iron Toxicity / Iron Precipitate

Limited data are available to evaluate iron as a stressor in Wyman Creek. However, the data which are available show concentrations that significantly exceed EPA’s water quality criteria for protection of aquatic life of 1,000 µg/L (Figure 200). Samples were collected at biological monitoring station 81LS008 during three flow conditions (snowmelt, rain event, baseflow) in 2011. Total iron concentrations ranged from a low of 1,210 µg/L (rain event) to a maximum of 5,540 µg/L (baseflow). Iron concentrations in Wyman Creek were the highest among all of the streams sampled during these events, which covered 29 stations from 22 different streams across the SLRW (Figure 201).

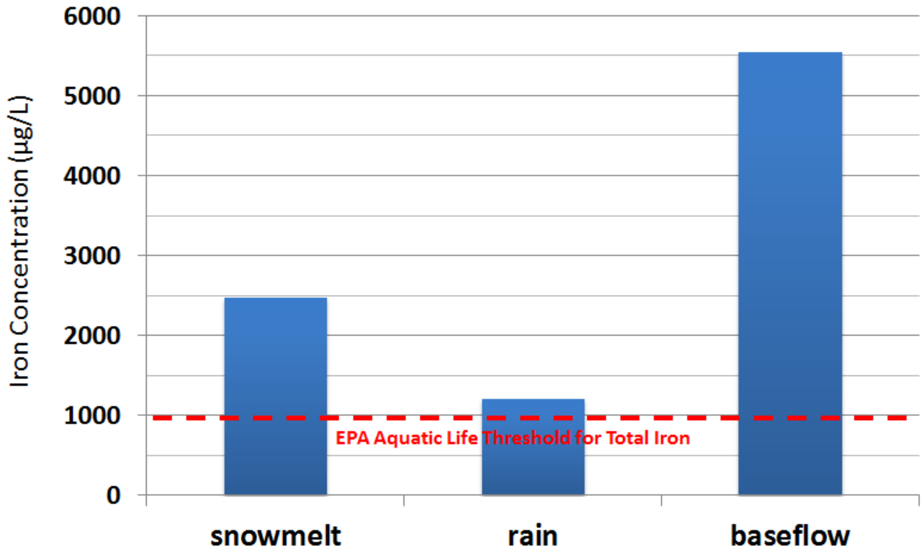


**Figure 200:** Total iron concentrations observed in Wyman Creek over three distinct flow conditions

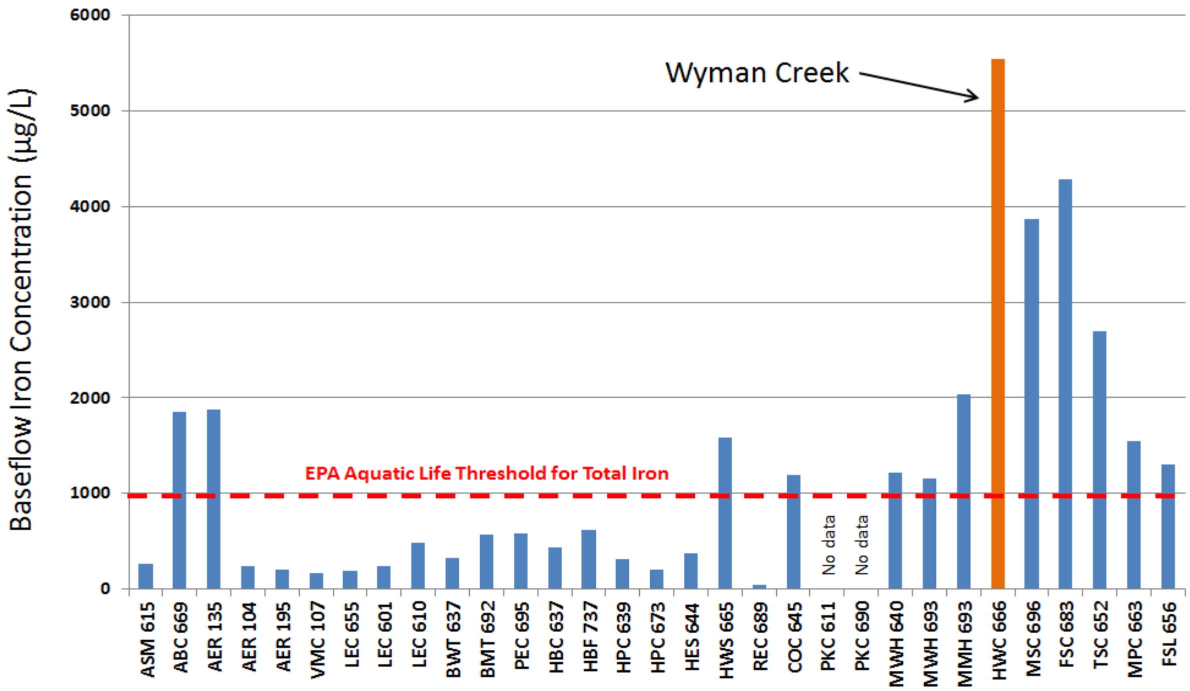


**Figure 201:** Total iron concentrations observed in St Louis River Watershed study streams at baseflow

### Iron Precipitate

The formation of iron precipitates in surface waters is seen in many areas of northern Minnesota, and occurs naturally in areas with very little anthropogenic influence. Wyman Creek receives water from several abandoned iron ore mining pits in its headwaters, but iron precipitates are seen forming in many tributary streams that have no impact from mining. The formation of iron precipitates in Wyman Creek appear to be a natural process resulting from significant groundwater inputs, but the additional loading of iron contributed to the stream through mine pit dewatering may have an additional impact. It is very difficult to separate out the natural processes from human influence in this case, considering that iron precipitates are a rather commonly observed “impact” observed around the state in watersheds with a wide range of land-uses. Iron precipitates have been routinely observed in Wyman Creek during low flow conditions (Figure 202 and 203).

**Figure 202:** Iron precipitate observed at Wyman Creek biological monitoring station 81LS008 on 9/12/2012. The photo on right shows accumulation of precipitate on a barrel that was deployed in the stream for one week.

**Figure 203:** Iron precipitate (Fe3+) rust colored water on upper right) forming in tributary stream to Wyman Creek. This particular stream has no upstream mining impacts, which is evidence that natural conditions in the watershed are favorable for the formation of Fe3+.

## Biological Response to Iron / Iron Precipitate

Published work by Lind et al (2006) suggests that mayflies in the family Leptophlebiidae are one of the most sensitive macroinvertebrate taxa affected by total iron concentrations. These mayflies were fairly abundant in the September 2009 sample collected from station 81LS008, but were not present in the sample collected in August of that year. It is possible that iron concentrations are leading to reduced numbers of these sensitive mayflies and other sensitive taxa, but their relatively robust population offers some evidence against toxic levels of dissolved iron in this system. They also may not have been present during the August sampling time due to life stage (i.e., they might have been in the egg stage, or too small to capture).

Iron precipitates similar to those observed in Wyman Creek have the ability to restrict the distribution, abundance, and diversity of fishes (Dahl 1963; Amelung 1982) in stream. Detailed descriptions of some of the habitat and physiological disturbances that can be caused by iron precipitate are discussed in

Section 3.1.11. One of the more common impacts of iron precipitate is the fouling of benthic substrates, which create adverse conditions for fish and macroinvertebrates that utilize that area of the stream.

Data from several biological metrics related to fish that utilize benthic habitats are shown in Figure 204. These Figures compare results from Wyman Creek to a set of high quality coldwater streams in the SLRW, all of which scored above the IBI threshold and/or UCL of the IBI threshold. The percentage of benthic insectivorous fish observed at station 81LS008 was comparable or higher than the majority of high quality SLRW coldwater streams (Figure 204). A similar pattern can be observed in the relative percentage of benthic minnow and darter species at this site, which is another fish metric that can respond negatively to benthic habitat degradation. A large population of Mottled Sculpin and a smaller population of Longnose Dace accounted for the relatively high percentage of benthic species at station 81LS008 in the 2009 sample. No benthic fish species were observed at 12LS006, but the stressor in question (iron precipitate) has never been observed to be as strong at this station.

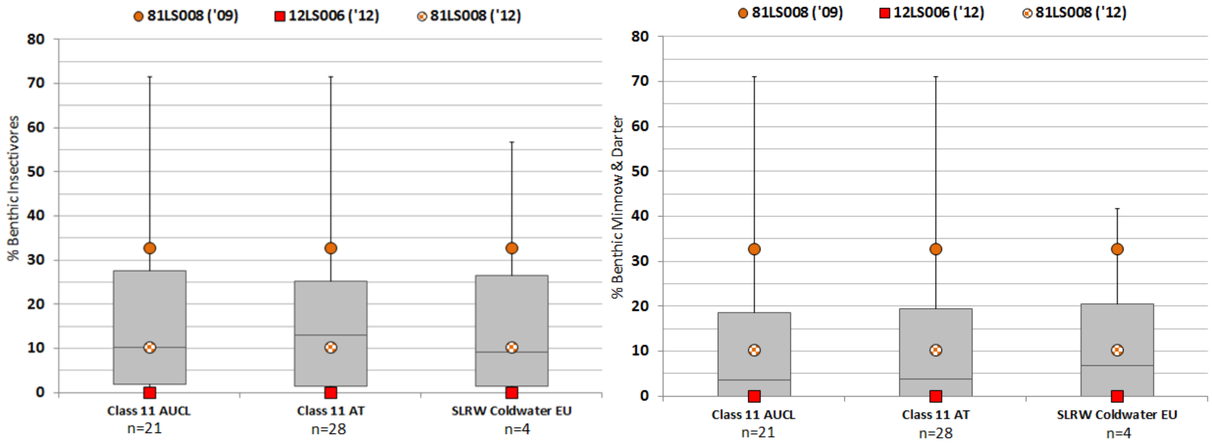


**Figure 204:** Comparison of % benthic insectivorous fish and % benthic minnow and darter individuals observed at Wyman Creek to high quality coldwater stream of the SLRW

Although this reach is not listed as impaired for MIBI, the macroinvertebrate community did not score exceptionally well. Scores of 26 and 40 were recorded at station 81LS008, which equate to 6 points below and 8 points above the impairment threshold (32), respectively. The marginal MIBI scores at this station may be reflective of the seasonally poor substrate conditions that occur when iron precipitates form in the channel.

### Summary: Is iron a stressor in Wyman Creek?

Iron concentrations in Wyman Creek have been recorded at concentrations over five times higher than EPA's aquatic life standard of 1,000 µg/L. Iron concentrations are elevated due to natural background conditions, and possibly due in part to mining land-uses in the headwaters. Iron precipitates have been observed at biological monitoring station 81LS008, and also in many tributary streams draining watersheds that have no mining influence. Biological response data are inconclusive at this time, but iron concentrations and iron precipitates should remain a candidate stressor for further evaluation with future monitoring.

## 5.14.2 Elevated Water Temperature

Water temperature data were collected in at several locations in Wyman Creek during the years 2009, 2012, and 2013. In all, five sites on Wyman Creek were continuously monitored for stream temperature, although all sites were not monitored during every year of data collection. Figure 205 shows the location of temperature logger sites within the Wyman Creek Watershed. Although loggers were typically installed in May and removed in September or October, only data between June 1 and August31 were analyzed for the purposes of this report to evaluate elevated water temperature as a stressor.

Figure 206 shows the site-specific breakdown of the temperature data, which was evaluated using at least 70% of the time in growth temperature (46.0-67.9° F) as the indicator of whether or not BKT should be present based on water temperature alone. It is evident that the water temperature in Wyman Creek is too warm for coldwater species. Every logger except one fell below 70% growth, and the one that exceeded 70% was deployed in 2009 – Minnesota's 7th coldest summer on record (www.noaanews.noaa.gov/stories2009/20090910_summerstats.html).

Temperature data for all Wyman Creek sites was plotted using similar methodology to a trout temperature tolerance study in Wisconsin and Michigan (Wehrly et al. 2007) Field-Based Estimates of Thermal Tolerance Limits for Trout: Incorporating Exposure Time and Temperature Fluctuation, Transactions of the American Fisheries Society, 136:2, 365-374, DOI: 10.1577/T06-163.1). Every site falls outside the tolerance limit for BKT. This analysis suggests that both maximum temperature *and* temperature fluctuation are limiting coldwater species, especially Brook Trout, in Wyman Creek. High temperature fluctuation is most likely due to the high sensitivity of the stream to ambient air temperatures. This situation is exacerbated by the high number of beaver- and man-made impoundments in the watershed.

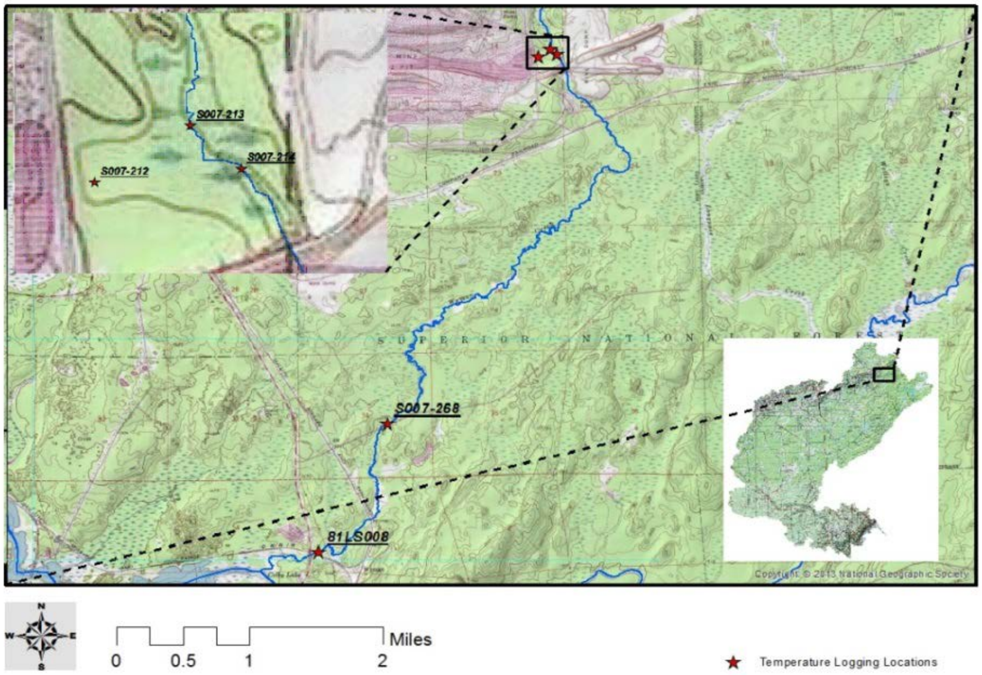


**Figure 205:** Temperature monitoring locations in the Wyman Creek Watershed

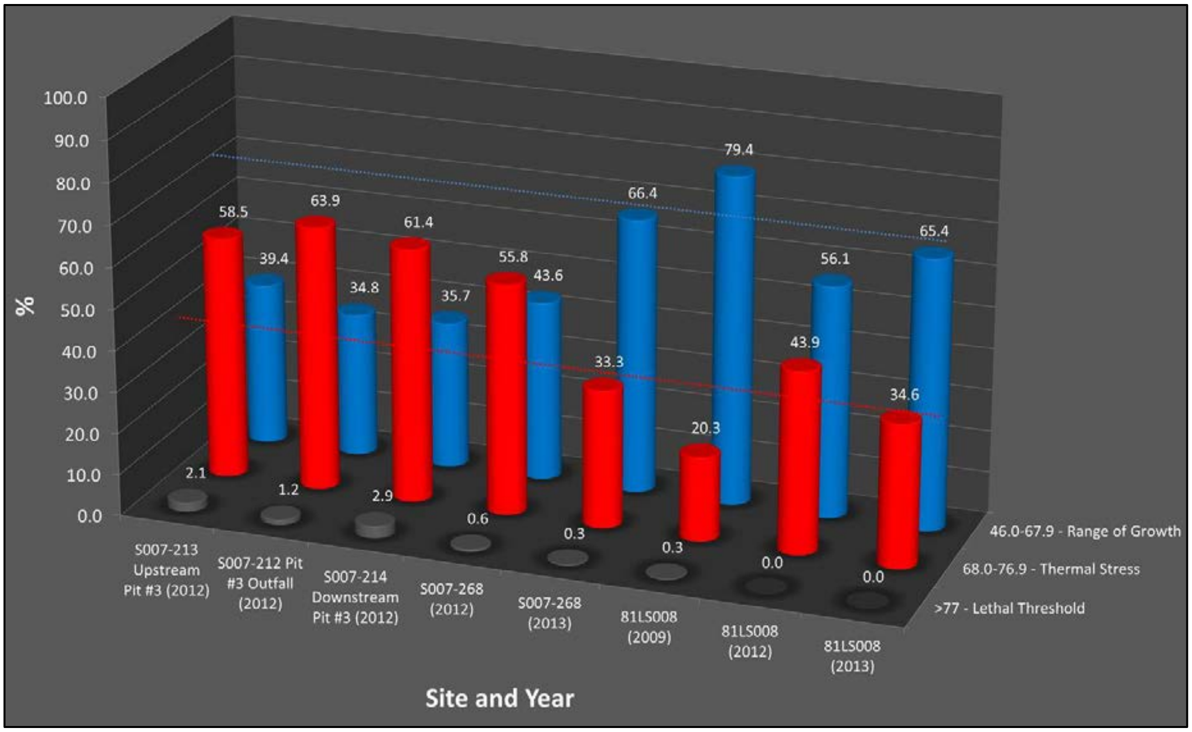


**Figure 206:** Percentage of time spent in BKT growth, stress and lethal ranges

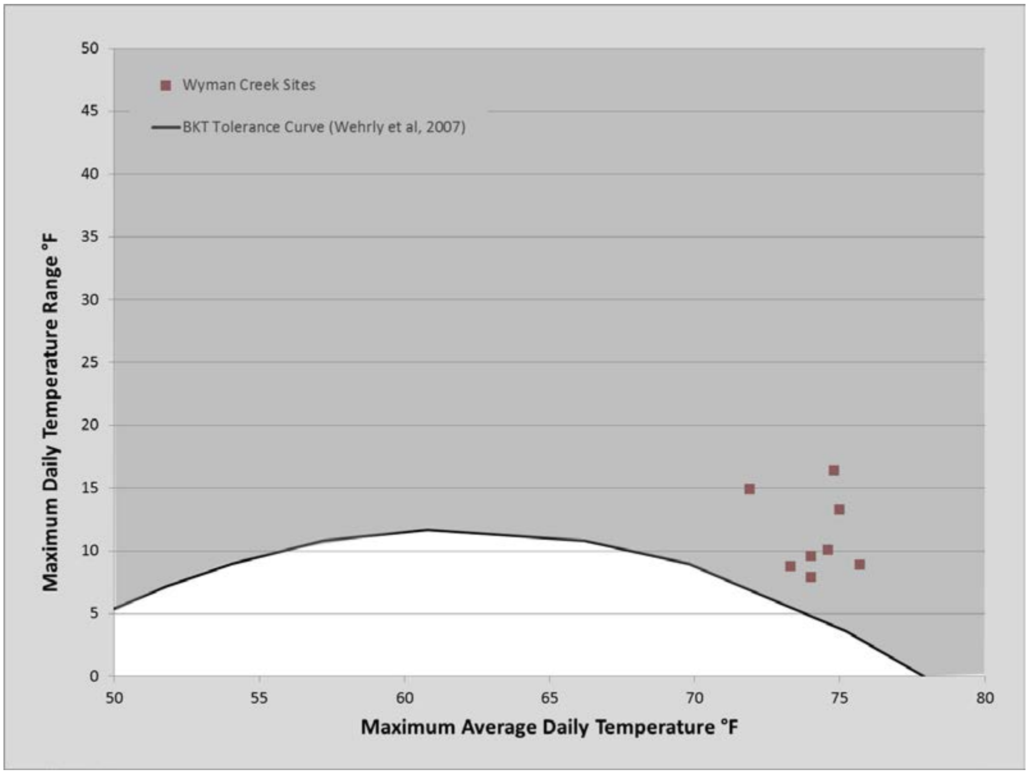


**Figure 207:** Maximum daily temperature and maximum temperature fluctuation recorded in Wyman Creek

## Sources and Pathways Contributing to Elevated Water Temperatures

The Wyman Creek Watershed contains several possible sources of temperature loading, with two prominent potential sources being mine pits and beaver dams. Both impede water flow and increase the surface area available for the stream to absorb solar energy. Beaver dams have the added effect of flooding and killing riparian forests so that even when the beaver dams breach, the shading that used to be provided by the canopy trees is removed. The silt, clay and other organic matter deposited behind beaver dams is also easily re-suspended by wildlife, darkening the water and aiding in heat absorption (Figure 208). The absence of Brook Trout in the most recent fisheries surveys may be in part due to an increase in beaver activity. Historic photos of the watershed indicate a large increase in beaver activity in the last ~25 years. Current beaver dam density is 3.4 dams per mile, while in 1991; the density was 1.6 dams per mile (counted from Google Earth historic imagery).

Additionally, Pit #5 at the headwaters of Wyman Creek was flooded in the 1990's, and now outflows directly into the stream. Temperature data collected from the outfall of this mine pit show temperatures that were in the stress range for Brook Trout approximately 64% of the time in 2012, which is a longer duration than any other site where data were collected that year.

**Figure 208:** An example of a large beaver dam and the turbid water typical throughout Wyman Creek.

**Figure 209:** An example of a connectivity issue on Wyman Creek. This culvert is perched, degraded, and filled with sediment and large wood which reduces or eliminates

An impoundment created by a failing culvert at a railroad crossing may also be contributing to temperature loading (Figure 209). This crossing is located near the headwaters of the creek and is acting as a barrier to fish movement, as well as disrupting sediment transport and likely warming water temperatures, although no data are available to validate these claims.

### Biological Response to Elevated Water Temperatures

Fish community data are available from two stations on Wyman Creek, 81LS008 (sampled twice) and 12LS006. The locations of these monitoring sites are shown in Figure 205. In all, fifteen total fish species were collected from Wyman Creek, covering a wide range of thermal preferences and tolerances. Several taxa were present that are commonly found in streams inhabited by Brook Trout in Northeastern Minnesota, including Mottled Sculpin, Longnose Dace, Pearl Dace, and Finescale Dace. On the other hand, several of the samples were dominated by species that are more typical in warmwater or coolwater streams (Common Shiner, Black Bullhead, Yellow Perch) and often dominant in marginal or poor coldwater streams (Blacknose Dace, Creek Chub). The proximity of monitoring station 81LS008 to Colby Lake (0.5 stream miles) is likely responsible for the presence of warmwater species such as Black Crappie, Black Bullhead, Yellow Perch, and Burbot – which are all species that are more typically found in larger rivers or lake systems.

Several biological metrics were selected to evaluate elevated water temperatures as a stressor in Wyman Creek. These metrics all focus on the relative abundance of individuals or taxa that prefer or require cold water temperatures:

#### % Coldwater Individuals / % Coldwater Taxa

Mottled Sculpin was the only species found in Wyman Creek that are included in the “Cold” fish metric used by the MPCA in their analysis of coldwater streams (metric name was expanded to “Coldwater” in this report). Species included in this metric are limited to salmonids and sculpins. Mottled Sculpin were present and fairly abundant during both sampling events at station 81LS008, but they were not found at the upstream monitoring station, 12LS006. Although this species is included in the coldwater metric, they are regularly found in streams of region that possess marginal thermal regimes for supporting trout and other coldwater species, and they are also observed in some streams with water quality impacts.

Compared to results from SLRW coldwater streams scoring above the IBI threshold and UCL, Wyman Creek supports a lower percentage of coldwater taxa (Figure 210). Many of the higher quality coldwater streams in the SLRW support two or even three coldwater taxa, typically trout and sculpin species. Station 81LS008 compares favorably the high quality coldwater streams of the SLRW in terms of the % of the fish community that were coldwater fish due to the relatively large population of Mottled Sculpin (Figure 210). Based on the moderately tolerant characteristics of this species described earlier, the presence of Mottled Sculpin alone at this monitoring station is not enough evidence to refute elevated water temperature as cause of poor coldwater FIBI results.

#### % Coldwater Intolerant Individuals / % Coldwater Intolerant Taxa

Longnose Dace was the only species found in Wyman Creek that is included in the “Coldwater Intolerant” IBI metric. This metric focuses on fish commonly found in coldwater streams that are sensitive to a variety of stressors, including water temperature. Compared to high quality coldwater streams in the SLRW, Wyman Creek supports very few coldwater intolerant taxa and individuals (Figure

210). Results for this metric were between the 0 – 25th percentile values observed at SLRW coldwater streams scoring above the IBI threshold and/or UCL of that threshold.

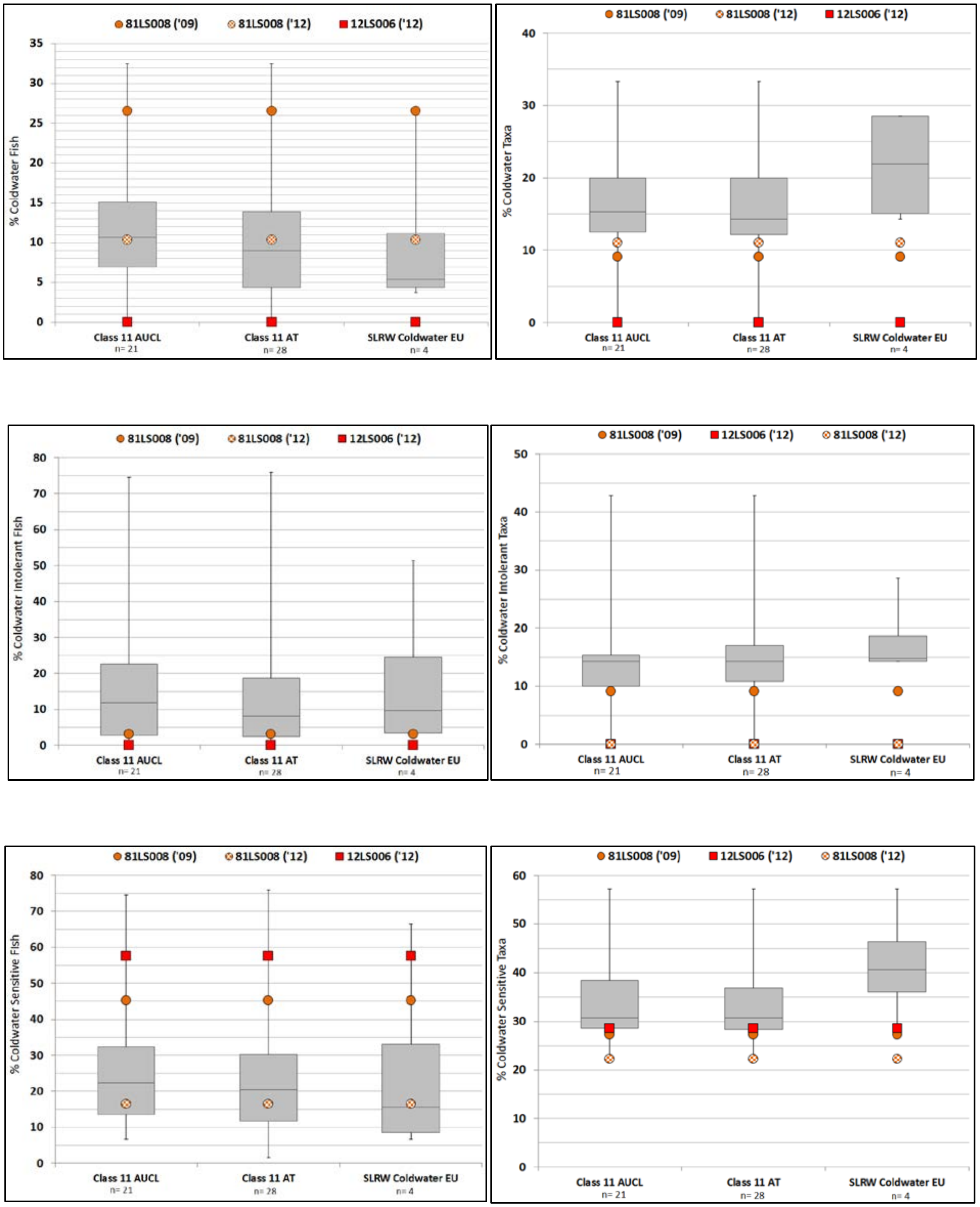


**Figure 210:** Wyman Ck. fish metric results compared to high quality reference streams * See Section 4 for explanation of TIVs. **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above FIBI Threshold

%Coldwater Sensitive Individuals / % Coldwater Sensitive Taxa

The “coldwater sensitive” metric is probably the least temperature-based among the three used here to evaluate data from Wyman Creek. This metric reflects general tolerance to a variety of stressors, within the context of coldwater streams. This metric includes taxa that are coldwater obligate species (e.g. Brook Trout, slimy sculpin) but is also expanded to include species found in coldwater streams that are more tolerant of coolwater conditions that may not be suitable for Brook Trout (e.g. Pearl Dace, Finescale Dace). Pearl Dace were observed in fairly large populations at both Wyman Creek monitoring stations. Finescale Dace were present at the upstream monitoring station (12LS006) in very low numbers. In terms of the relative abundance of coldwater sensitive individuals, Wyman Creek stations are very comparable to or exceed values observed at high quality coldwater streams (Figure 210). The comparison is not so favorable for in terms of coldwater sensitive taxa, as Wyman Creek only supports two taxa that qualify for this metric. The abundance of Pearl Dace in Wyman Creek is not surprising, as this species often has a strong presence in cool, bog-drainage streams with beaver activity (Cunningham 2006).

**Summary: Are elevated water temperatures a stressor in Wyman Creek?**

Data from the early 1980’s confirm that Wyman Creek once supported a small population of naturally reproducing native Brook Trout. Recent surveys conducted by the MPCA did not contain Brook Trout, but healthy populations of several coldwater/coolwater species remain, such as Mottled Sculpin, Longnose Dace, and Pearl Dace. Based on temperature monitoring data from 2009, 2012, and 2013, the thermal regime of Wyman Creek can be considered marginal to poor for supporting Brook Trout and other coldwater obligate fish species. Although no historical temperature data is available for comparison, it is likely that thermal loading generated from the numerous beaver impoundments, and possibly mine pit drainage in the headwaters of the creek, are elevating water temperatures and creating poor conditions for Brook Trout.

We recommend that elevated water temperature be included in the list of stressors contributing to the impaired condition in Wyman Creek. Additional monitoring that focuses on the potential effects of beaver impoundments and mine pit drainage is recommended for developing a restoration plan.

### 5.14.3 Dissolved Oxygen

Instantaneous DO data were collected at four stations along Wyman Creek. Two additional DO readings were collected through the ice in the winter of 2014. Two of these stations (S007-213 & S007-214) are located in the headwaters of the creek in an area of the watershed that is dominated by wetlands and mining land-uses. The other two stations are located in the lower reaches of Wyman Creek and are paired with the biological monitoring stations. Monitoring was conducted during the open water season 2009, 2012, and 2013. The DO concentrations observed at these four stations are displayed in Figure 211.

The DO concentrations in Wyman Creek fell below the class 2A (coldwater trout stream) standard of seven mg/L at all monitoring stations during the months of June through September. Very low DO concentrations (sub 3 mg/L) were observed in the headwaters of Wyman Creek at station S007-213, but approximately 500 ft downstream, DO concentrations increased to around 5 mg/L due to a tributary stream entering the creek draining an abandoned mine pit. This observation suggests that the specific mine-pit tributary mentioned above is not contributing to the sub-optimal DO conditions observed downstream. On the contrary, it may be improving the DO regime of Wyman Creek, at least locally. Other tributaries from mine pits and mining areas enter further upstream, but the effect of these tributaries on the main stem of Wyman Creek is not known.

The DO concentrations at the two biological monitoring stations failed to meet the water quality standard (7 mg/L) for supporting coldwater aquatic life. July and August DO levels at these stations were generally around 4 or 5 mg/L, which is adequate for supporting many warmwater species, but inadequate for supporting sensitive coldwater species such as Brook Trout. This observation is supported by biological data from these sites which shows high taxa richness and fish densities. Early spring and late winter DO concentrations were adequate at these sites, so the period of DO-stress is limited to mid-summer and early fall periods.

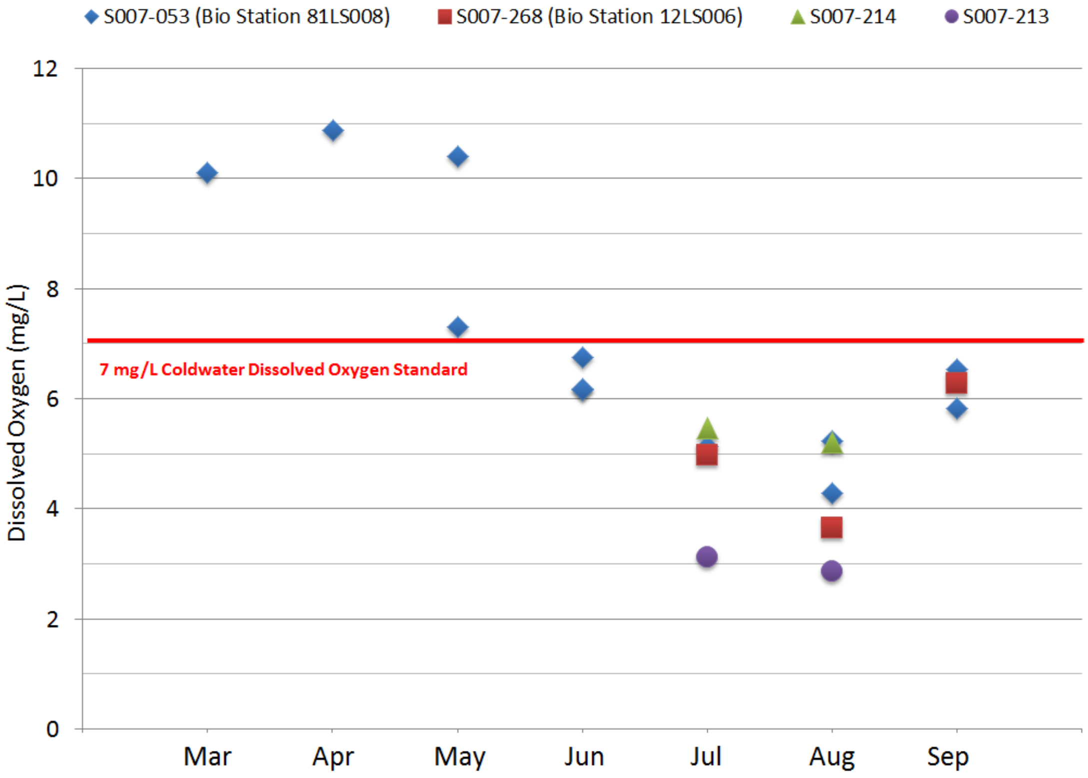
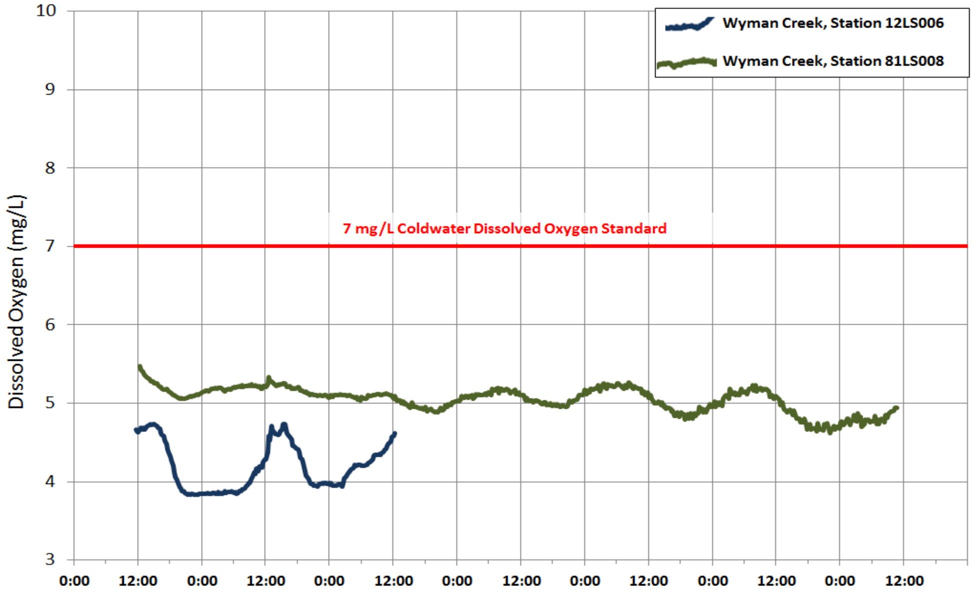


**Figure 211:** Point measurements of DO collected at Wyman Creek monitoring stations

Continuous DO data were collected at two biological monitoring stations on Wyman Creek in the summers of 2012 and 2013 (Figure 212 and 213). Both of these DO profiles were conducted in late August during baseflow conditions. DO concentrations were recorded at 15-minute intervals for a period of 4-6 days. An equipment malfunction limited the 2012 profile at station 12LS006 to a duration of just over 24 hours.

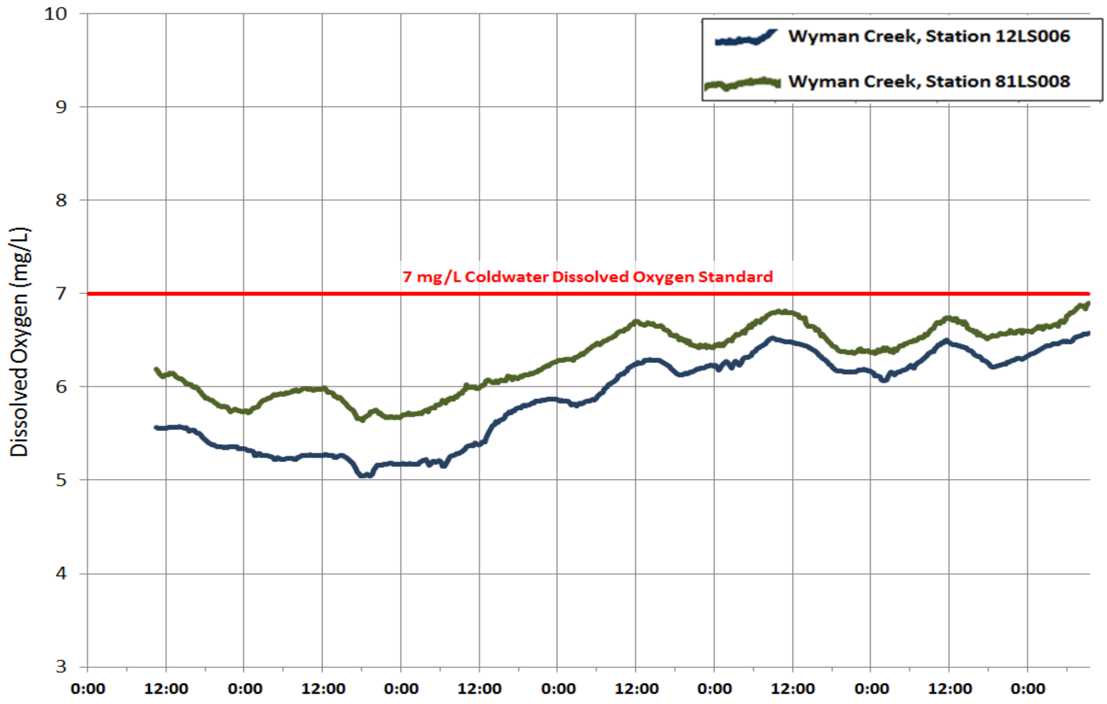
The DO concentrations remained below the 7 mg/L DO standard at both locations for the entire duration of the monitoring periods in both 2012 and 2013. In both profiles, DO concentrations were lower at the upstream monitoring station (12LS006), where DO levels fell below the warmwater standard of 5 mg/L on occasion. Cooler water temperatures near the end of the 2013 survey caused a steady increase in DO concentrations at both monitoring sites. Daily maximum water temperature dropped from 22.8 C down to 18.6 C during this time, which led to an increase in the availability of DO within the water column.

Diurnal DO flux is very minimal in Wyman Creek based on the two continuous monitoring data sets. Maximum DO flux was below 1 mg/L at all monitoring stations during the 2012 and 2013 monitoring efforts. The low diurnal flux in DO concentrations is an indication that primary productivity is low within these two reaches of Wyman Creek, and that the lower DO concentrations observed may be due to factors other than stream eutrophication (i.e. beaver impoundments, lack of stream flow, wetlands)



| Dates of Monitoring Period | # of Readings | DO Min (mg/L) | DO Max (mg/L) | Avg 24-hr Flux (mg/L) | Max 24-hr Flux (mg/) | % Readings below Trout Stream Threshold (7 mg/L) | Avg. Duration below (hours) | Max Duration below (hours) |
|---|---|---|---|---|---|---|---|---|
| ***8/24/2012 - 8/30/12*** | 570 | 4.89 | 5.33 | 0.366 | 0.61 | 100% | 142.5 | 142.5 |
| ***8/24/2012 - 8/26/12*** | 196 | 3.83 | 4.73 | 0.89 | 0.89 | 100% | 49 | 49 |

**Figure 212:** Results of August 2012 continuous DO monitoring at Wyman Creek biological monitoring sites



| Dates of Monitoring Period | # of Readings | Min (mg/L) | Max (mg/L) | 24-Hour Flux Data Average (mg/L) | Max (mg/L) | % Readings below Coldwater DO Threshold - (7 mg/L) |
|---|---|---|---|---|---|---|
| *8/30/2013 - 9/5/13* | 583 | 5.64 | 6.7 | 0.45 | 0.58 | 100% |
| *8/30/2013 - 9/5/13* | 582 | 5.05 | 6.58 | 0.46 | 0.72 | 100% |

**Figure 213:** Results of September 2013 continuous DO monitoring at Wyman Creek biological monitoring sites

### Sources and Pathways Contributing to Low Dissolved Oxygen

The TP concentrations observed in Wyman Creek were all below the stream nutrient criteria northern Minnesota target of 0.055 mg/L (n=11, avg.=0.024 mg/L, max=0.035 mg/L). Very little periphyton algae or aquatic macrophytes were observed in the channel. These observations, along with the low diurnal DO flux observed during continuous monitoring, provide evidence against stream eutrophication as a cause of low DO concentrations.

### Biological Oxygen Demand (BOD)

Samples were collected in August 2012 to evaluate the contribution of BOD to the sub-optimal DO concentrations observed in Wyman Creek. Samples were collected at the two biological monitoring stations final day of the continuous monitoring period (August 30, 2012). The BOD concentrations at the two stations were comparable, 1.6 mg/L at station 12LS006, and 1.9 mg/L at station 81LS008. Both of these results are slightly higher than the BOD target level of 1.5 mg/L cited in the river nutrient criteria that is developed by the MPCA (Heiskary 2013). This suggests that a higher concentration of BOD is observed at these stations in comparison to high quality streams in Northern Minnesota. The other components of stream eutrophication (elevated phosphorous, high DO flux, high Chl-a concentrations) were not observed at these stations. Therefore, the slightly elevated BOD values observed at these sites are not a symptom of eutrophication, but are likely linked to organics from wetlands and/or the presence of iron bacteria (see Section 3.1.11).

### Beaver Dams

Wyman Creek and its tributary streams are frequently impounded by beaver dams. A total of 42 beaver dams were identified along the 10-mile length of Wyman Creek using recent aerial photos, which equates to about 1 impoundment for every 1,200 feet of stream (Figure 214). There is considerable debate on whether or not beaver dams are beneficial or detrimental to stream habitat, but there is no debating that the Wyman Creek stream corridor is heavily influenced by beaver activity. These impoundments may have direct impacts on channel morphology, fish passage, and streamflow throughout the length of the creek. Indirectly, the beaver dams have the potential to increase water temperatures, as well as decreasing DO concentrations due to the warmer water temperatures, lower streamflow velocities, and increases in biological oxygen demand. No data were collected in this study that allow for direct comparisons between reaches influenced by beaver, and those left relatively or completely unaltered. As a result, the connection between beaver dams and low DO in Wyman Creek are unknown, and need further evaluation during the TMDL development and implementation phase of the impaired waters process.

### Iron Precipitate

Iron precipitates have been observed throughout Wyman Creek and in many tributary streams. Water colors darken when this rust-colored orange participate forms (Figure 202). The darker appearance of the water under these conditions is likely to soak up more solar heat, particularly where these precipitates form in stagnant, impounded areas behind beaver dams. In addition, there are at least 18 different types of bacteria are classified as “iron bacteria,” which are long, thread-like organisms that “feed” on iron and secrete slime as a bi-product. Unlike most bacteria, which feed on organic matter, iron bacteria fulfill their energy requirements by oxidizing ferrous iron ($Fe^{2+}$) into ferric iron ($Fe^{3+}$). Ferric iron ($Fe^{3+}$) is insoluble and precipitates out of the water as a rust colored deposit. The effect of these iron forming bacteria on DO concentrations is not known, but it is possible that oxygen is consumed as iron bacteria form in this stream.

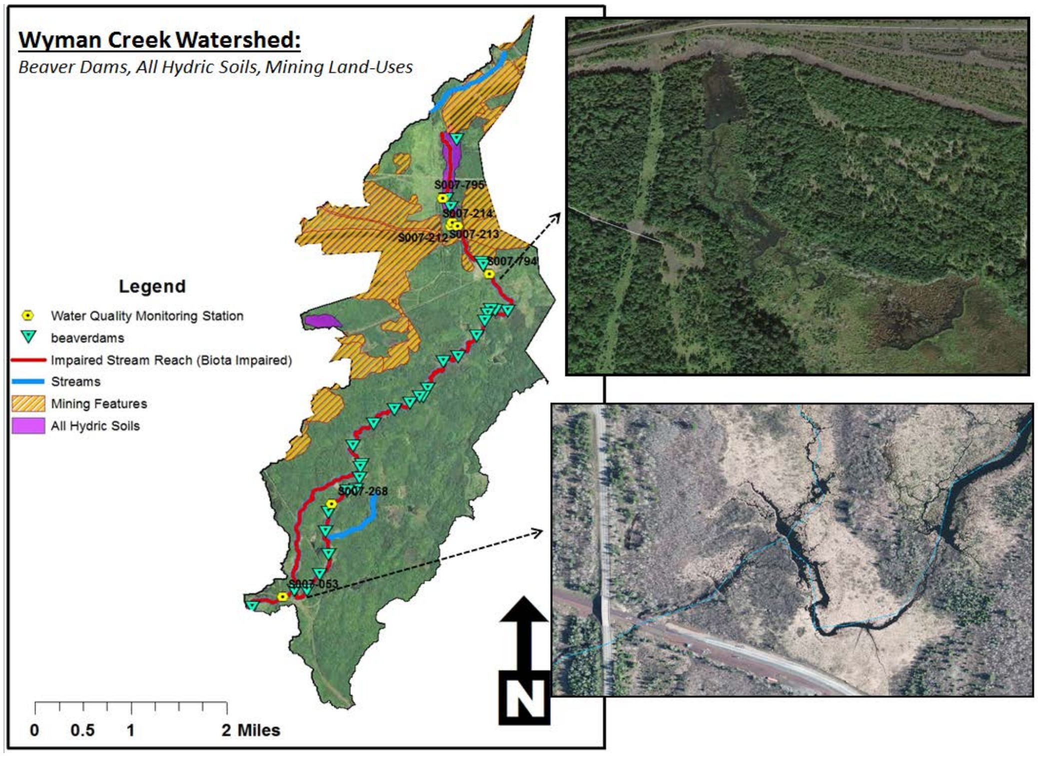


**Figure 214:** Beaver dam locations, mining-related land-use features, and all hydric soil types within the Wyman Creek Watershed.

## Biological Response to Low Dissolved Oxygen

The fish community of Wyman Creek includes a mix of coldwater and warmwater species with a range of tolerance to low DO concentrations. An average of 18% of the fish observed at 81LS008 over the two monitoring visits were coldwater species that are moderately intolerant of low DO. Mottled Sculpin were present at the downstream monitoring station (81LS008) during both sampling events, and were fairly abundant relative to other species present. Longnose Dace, another fish species that is commonly found in higher quality coldwater streams in northeastern Minnesota, were also present at 81LS008 during one sampling visit in limited numbers. The monitoring station located further upstream (12LS006) did not support any coldwater species during the time of sampling. Instead, this reach was Pearl Dace, Blacknose Dace, and Creek Chub. Pearl Dace, a coolwater wetland species with a fairly high tolerance to low DO concentrations, accounted for 56% of the total sample at this station. Overall, species that are very tolerant of low DO (Fathead Minnow, Central Mudminnow, Brook Stickleback) were not a dominant presence in the fish community.

The DO index values for the fish community of Wyman Creek are slightly lower than most of the values recorded at high quality coldwater streams in the SLRW (Figure 215). The DO index for the fish community at 81LS008 falls between the 25th percentile and the median value for comparable reference sites. These results support the earlier claim that this reach is not dominated by species that are tolerant of low DO, but may lack adequate DO concentrations for supporting sensitive coldwater species like Brook Trout. The DO index scores are lower at station 12LS006 due to the large population of Pearl Dace at this station, as well as the presence of northern redbelly and Finescale Dace. These fish species are commonly found in wetland dominated landscapes, which often have more marginal DO conditions for supporting a diverse fish assemblage.

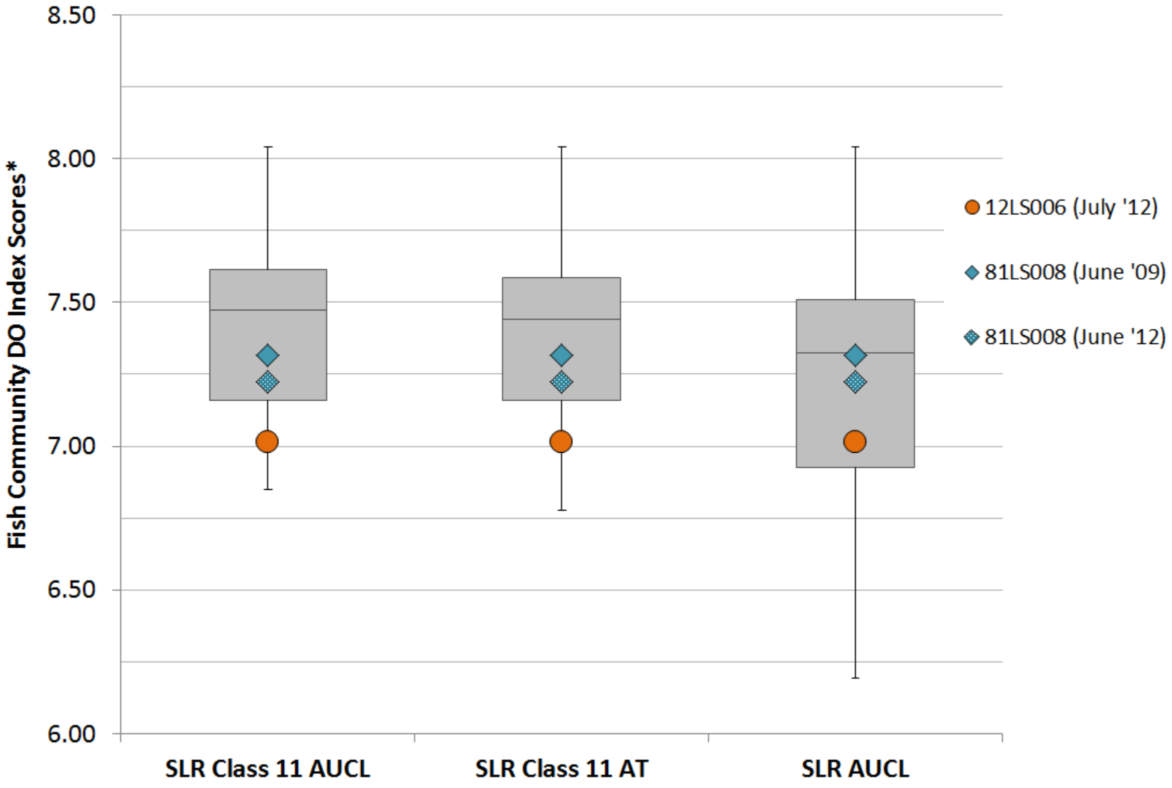


**Figure 215:** Fish community DO TIV results compared to results from high quality stations of the same IBI class. * See Section 4 for explanation of TIVs **AUCL** = Above Upper Confidence Limit of FIBI threshold **AT** = Above FIBI Threshold

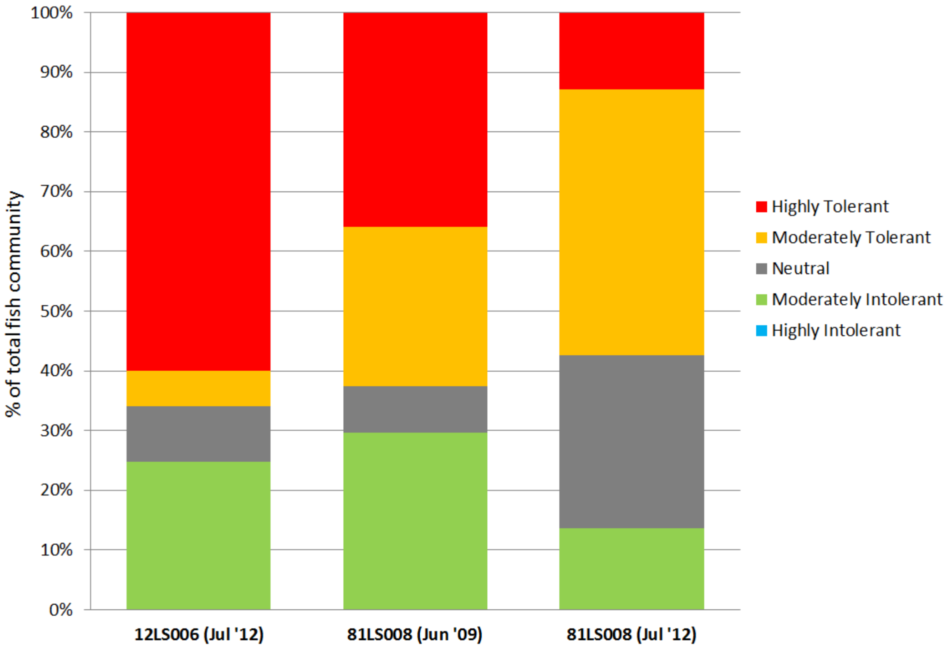


**Figure 216:** Proportion of Wyman Creek fish community considered tolerant or intolerant of low DO conditions

**Summary: Is low dissolved oxygen a stressor in Wyman Creek?**

Based on the fish community and other data, Wyman Creek does not appear to be a stream that is severely limited in terms of DO availability. However, the current DO regime is not favorable for supporting a quality coldwater fish assemblage. DO concentrations are frequently well below the 7 mg/L water quality standard for extended periods of time. Based on the evidence provided through available water chemistry and biological data, we recommend adding low DO as a cause of coldwater fish impairment in Wyman Creek.

### 5.14.4 Total Suspended Solids (TSS) & Turbidity

**Total suspended solids data**

A total of 16 observations from S007-053 over two years (2012 and 2013) were used to develop a summary of TSS and Secchi Tube (s-tube) data. This monitoring station is co-located with biological monitoring station 81LS008, approximately 0.5 miles upstream of the location where Wyman Creek outlets to Colby Lake. TSS concentrations exceeded the 10 mg/L standard for coldwater streams in 2 out of 10 samples (20.0%). The average TSS value over the 10 sampling events was 7.7 mg/L, with a maximum value of 14 mg/L. The exceedances of the WQ standard were relatively low in magnitude and frequency. Similarly, the average s-tube values were well above the WQ standard of 55 cm (87.7 cm) and the only result that failed to meet this standard (50 cm) only did so by a small margin.

Box plots of TSS and s-tube values for Wyman Creek and the “A” and “B” reference streams in the SLRW are shown in Figure 218 and 219. The TSS values for Wyman plot slightly higher than the reference streams but mostly fall below the threshold of 10 mg/L. The Secchi Tube values for Wyman roughly mirror the plotted Secchi data for the reference streams.

**Seasonal variation in total suspended solids**

Visual observations suggest that there is some seasonal variation in the amount of TSS in Wyman Creek. During snowmelt and heavy rain events the stream runs stained yet relatively clear, but during summer and fall low flows the stream appears turbid. This suggests that bank erosion and overland flow are not significant sources of suspended solids in this watershed. During low flows, iron precipitate is frequently observed and the stream appears turbid. This hypothesis is supported by the limited TSS and Secchi Tube data for Wyman Creek. It seems that there is a slight rise in suspended solids during snowmelt, but the maximum amount of suspended solids (and minimum transparency) in Wyman Creek does not occur until August. It should be noted that this is based on a small sample size (n=6) and more sampling is recommended to expand on these observations.

**Figure 217:** Wyman Creek after a May 2012 rain event (left) and during August 2012 at low flow, showing iron precipitate (right)

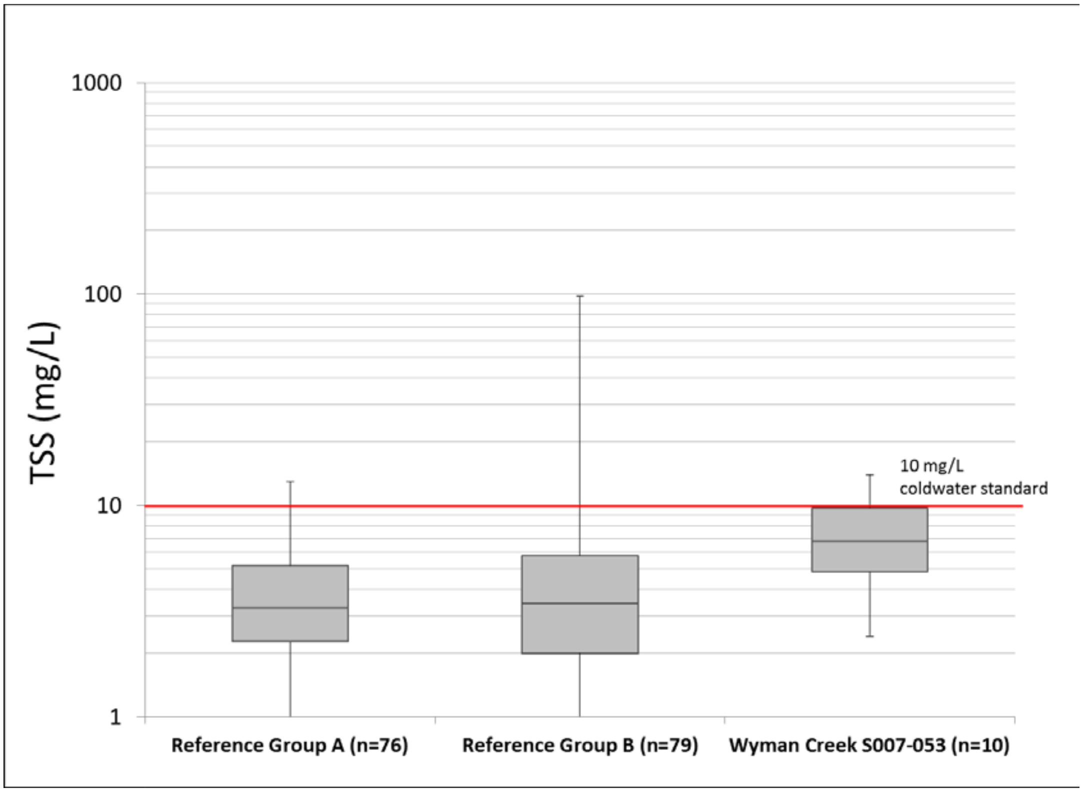


**Figure 218:** Box plots of TSS values for Wyman Creek and reference streams

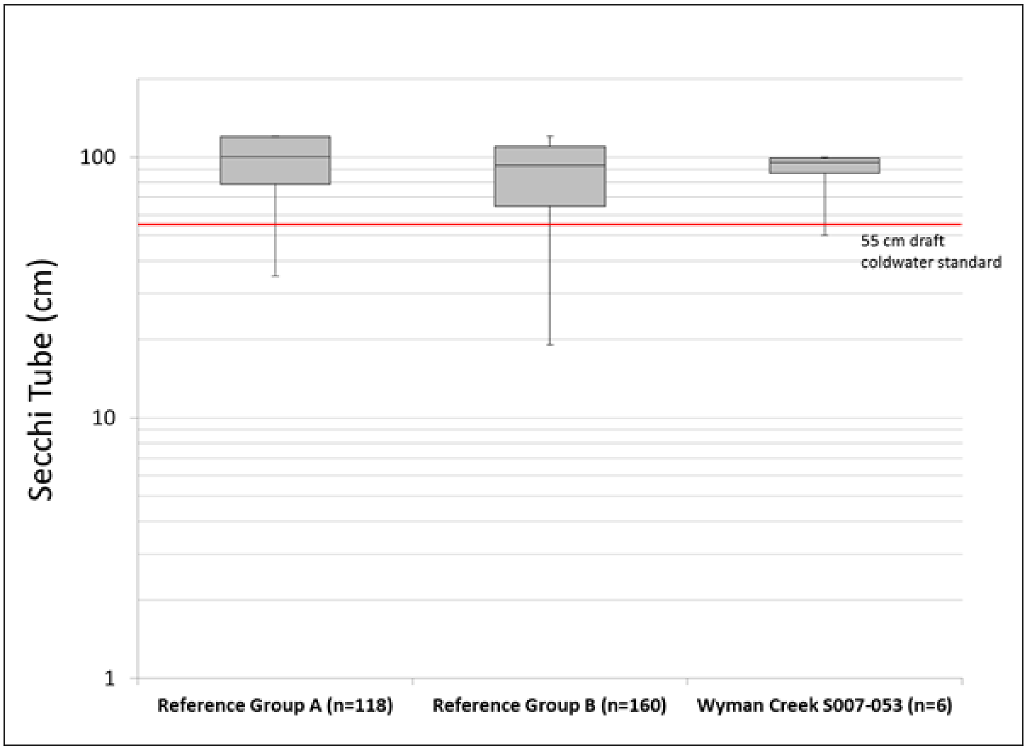


**Figure 219:** Box plots of Secchi Tube values for Wyman Creek and reference streams

### Sources and pathways of suspended solids in the Wyman Creek Watershed

Common sources of TSS and turbidity such as bank erosion and overland runoff do not appear to be issues in the Wyman Creek Watershed. The vast majority of the free flowing stream reaches observed were stable and not producing large sediment loads. The major sources of TSS and turbidity in Wyman Creek are discussed below.

### Iron Precipitate

Iron precipitate has frequently been observed in the lower reaches of Wyman Creek during low flow periods in the summer and fall (see Figure 217). The water chemistry and hydrological processes involved with the formation of iron precipitate are discussed in detail in Section 3.1.11. Elevated TSS concentrations and lower transparency values were observed during several late summer and early fall visits when iron precipitates were present.

### Beaver Impoundments

A significant portion of Wyman Creek and its tributary streams are impounded by beaver dams. These impoundments can have a profound impact on sediment transport, and thus impact both the condition of stream substrates and water clarity. Based on aerial photos and stream reconnaissance photos collected by (consultant), many of the beaver dams along Wyman Creek are associated with higher turbidity and TSS concentrations. The formation of iron precipitates behind these beaver dams accounts for a large portion of the observed increases in turbidity, but the settling and re-suspension of fine particles (silt/clay) is also a factor.

## Biological effects of elevated TSS

### Fish Response to TSS

The FIBI impairment on Wyman Creek is the result of poor metrics related to an abundance of warmwater species (Black Bullhead, Yellow Perch, etc.). Figure 220 shows that the majority of species sampled in two of the three sites are considered "intolerant" to elevated levels of TSS. Common TSS-intolerant species at these two sites include Pearl Dace, Mottled Sculpin, Northern Redbelly Dace, Longnose Dace, and Burbot. The 2012 sampling of 81LS008 only contained 17% of "intolerant" individuals. Creek Chubs and Common Shiners, both "neutral" to TSS, were dominant in 81LS008 that year (176 out of 242 individuals). Some of the other common "neutral" species found in Wyman Creek were Yellow Perch, Black Crappie, Blacknose Dace, and Black Bullhead. No species tolerant to TSS were collected at either monitoring station.

Two of the three monitoring results compare favorably with healthy streams throughout the SLRW. Figure 221 compares the Wyman Creek TSS TIVs with: 1) all the Class 11 AUCL streams in the SLRW, 2) the Class 11 AT streams in the SLRW, and 3) AUCL streams of all types in the SLRW. As can be seen from the graph, the Wyman Creek fish assemblages at station 12LS006 (2012) and station 81LS008 (2009) were less tolerant of TSS than the median of the healthy streams in the SLRW. The 81LS008 (2012) sampling scores much worse, falling near the 75% percentile value of all three groups of quality stations. This result is driven largely by the high numbers of Creek Chubs and Common Shiners.

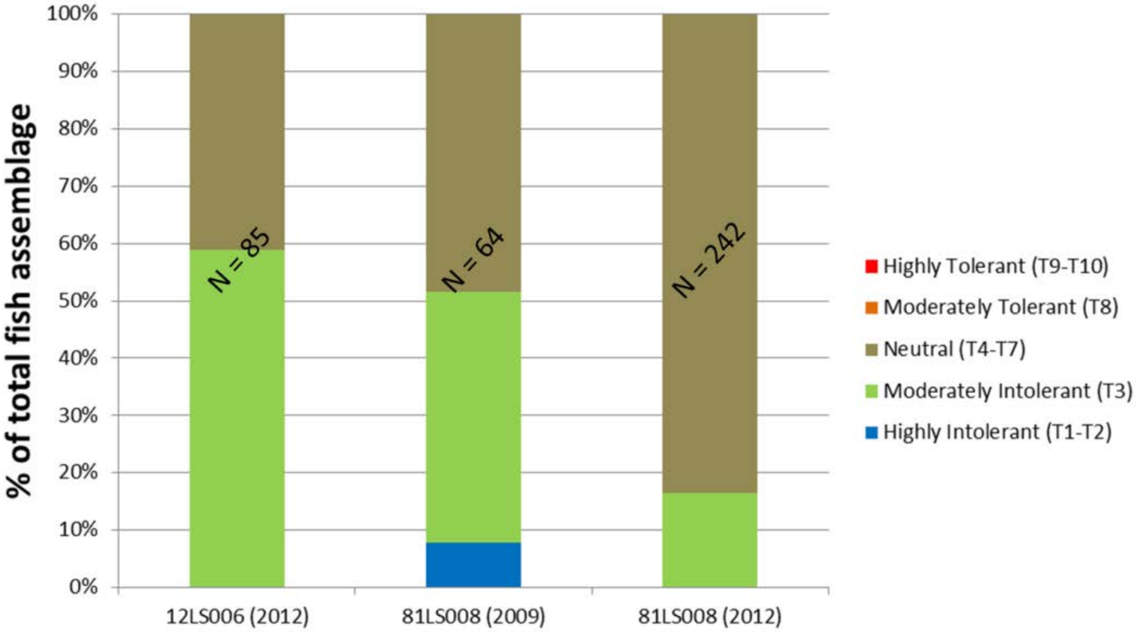


**Figure 220**: Proportion of fish population observed at Wyman Creek monitoring stations that are tolerant, neutral, or intolerant of low DO concentrations

### 5.14.5 Sulfate Toxicity

Sulfate was included as a candidate cause of FIBI impairment in Wyman Creek due to presence of mining land use in its watershed. Further analysis of the available sulfate data revealed concentrations that are higher than natural background conditions in the SLRW. A total of 14 sulfate samples were collected in the Wyman Creek Watershed from five stations (Table 59). Two of these stations were co-located with biological monitoring site, while the other three were located in the headwaters of Wyman Creek bracketing a tributary that originates from an abandoned mine pit.

Sulfate concentrations observed at the two biological monitoring sites located in the mid to lower reaches of Wyman Creek were highly variable. August and September sulfate results from both 81LS008 and 12LS006 were extremely low (around 1 mg/L). Samples collected concurrently from sites in the headwaters of Wyman Creek produced much higher sulfate concentrations, ranging from 15 – 80 mg/L. The highest concentration was observed at station S007-212, which is a tributary to Wyman Creek that originates from an abandoned mine pit. Other samples collected during snowmelt, rain events, and winter baseflow conditions show higher sulfate concentrations in the lower reaches of Wyman Creek at the biological monitoring stations (Table 59). There are some clear seasonal differences in sulfate concentrations even though the source of sulfate is a fairly consistent discharge of mine pit to the creek. Some theories on the variability of sulfate concentrations are discussed later in this section.

Per the requirements of discharge permit MN0042536-SD-12, the mine pit water discharged to Wyman Creek via an unnamed tributary is monitored regularly for a variety of parameters, including sulfate. Based on monthly sampling conducted between the years of 2007 and 2014, the average sulfate concentration of the effluent is between 70 – 80 mg/L and the monthly maximums were around 90 mg/L at their highest (Figure 222). Sulfate concentrations observed at the biological monitoring stations in February of 2014 were very similar those observed in the effluent.

**Table 59:** Summary of sulfate, hardness, and chloride data from Wyman Creek. Sulfate data are compared to WQ standards for aquatic life used in other US states.

| **WQ Station (Bio station)** | **Sample Date** | **Sulfate** (mg/L) | **Magnesium** (mg/L) | **Calcium** (mg/L) | **Hardness** (mg/L) | **Chloride** (mg/L) | **IA, IL, IN, PA Sulfate Standard*** (mg/L) |
|---|---|---|---|---|---|---|---|
| S007-053 (81LS008) | 3/19/2012 | **31.3** | 9.98 | 12.9 | 73.3 | 1.55 | 500 |
| S007-053 (81LS008) | 4/17/2012 | **45.4** | - | - | - | 1.54 | 500 |
| S007-053 (81LS008) | 6/26/2012 | **13** | - | - | - | 1 | 500 |
| S007-053 (81LS008) | 5/25/2012 | **19.3** | 7.02 | 9.74 | 53.2 | 0.5 | 500 |
| S007-053 (81LS008) | 8/1/2012 | **1** | - | - | - | 1 | 500 |
| S007-053 (81LS008) | 9/13/2012 | **1.1** | 17.5 | 23.9 | 131.7 | 0.5 | 500 |
| S007-053 (81LS008) | 8/30/2012 | **1** | - | - | - | 1 | 500 |
| S007-053 (81LS008) | 5/1/2013 | **16.5** | - | - | - | - | 500 |
| S007-214 | 8/1/2012 | **47.9** | - | - | - | 1.22 | 500 |
| S007-213 | 8/1/2012 | **15.4** | - | - | - | 1 | 500 |
| S007-212 | 8/1/2012 | **71.9** | - | - | - | 1.37 | 500 |
| S007-268 (12LS006) | 8/30/2012 | **1** | - | - | - | 1.13 | 500 |
| S007-268 (12LS006) | 2/3/2014 | **97.7** | - | - | - | - | 500 |
| S007-053 (81LS008) | 2/3/2014 | **85.4** | - | - | - | - | 500 |

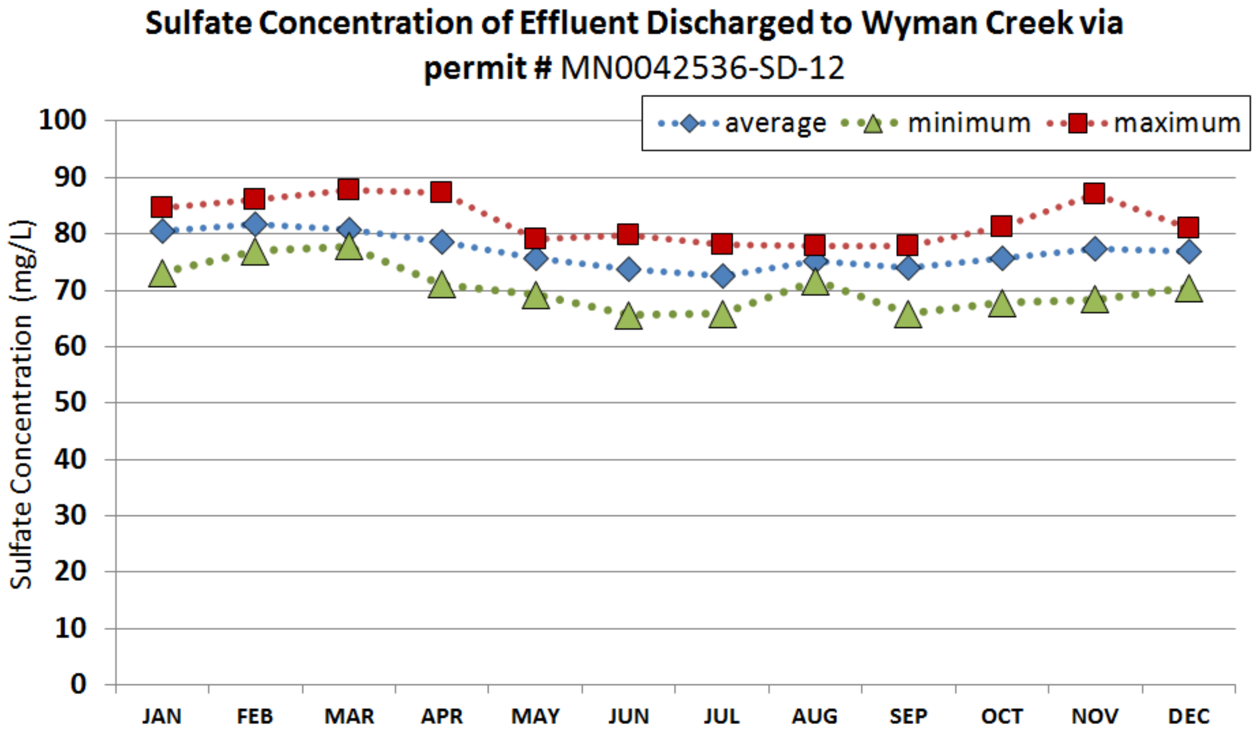


**Figure 222:** Sulfate concentrations of mine pit dewatering effluent discharged to Wyman Creek

Additional monitoring would be beneficial for understanding factors controlling sulfate concentrations in this watershed. Beaver impounded wetland areas along and within Wyman Creek may explain the longitudinal decrease (upstream to downstream direction) in sulfate concentrations observed during the summer and fall months. It is possible that DO concentrations reach very low or anoxic levels within the many beaver impoundments that exist between the headwaters area and the biological monitoring stations. If anoxia is occurring periodically, redox reactions may be occurring within these wetland complexes to convert sulfate to sulfide. These interactions may explain why sulfate concentrations appear to be much higher in the headwaters compared to sites near the mouth during summer months. Unfortunately, no sulfide data were collected during this project to investigate this further.

### Comparison of data to Water Quality Standards & Literature Values

Many of the current sulfate standards in the United States and Canada are based on total sulfate concentration as well as chloride concentrations and water hardness. As chloride concentrations and hardness increase, there is evidence that sulfate becomes less toxic to fish and aquatic macroinvertebrates (Soucek and Kennedy 2004). See Section 3.1.6 for a summary of current sulfate standards being applied in several U.S. states that are based on sulfate concentration, chloride concentration, and water hardness.

Table 59 summarizes total sulfate concentrations observed in Wyman Creek, along with associated chloride and hardness data. Water hardness data were only available for a small number of the sampling events. However, based on the low chloride concentrations in Wyman Creek, a sulfate standard of 500 mg/L would be applied in most, if not all flow conditions in this watershed. Given that the maximum sulfate concentration observed was 97 mg/L, it is highly unlikely that this stream would violate the sulfate standard used in other states (IA, IL, IN, PA) at any time of the year.

Total sulfate concentrations in Wyman Creek are also lower than several values cited in scientific literature and other research. Buckwalter (2013) listed a chronic toxicity value of 124 mg/L for protecting the most sensitive forms of aquatic life, which is not exceeded in any of the sampled collected from Wyman Creek or tributary streams. This chronic toxicity value does not incorporate water hardness or chloride values, and thus differs from the work done to develop water quality standards in several U.S. states (IA, PA, IL). In several research papers based on data from Ohio, (Rankin 2003, 2004) suggests that biological effects from sulfate may be occurring in streams with sulfate concentrations in the range of 300-500 mg/L. Again, sulfate concentrations in Wyman Creek are much lower than these listed values, making it difficult to suggest a sulfate stressor as a cause of fish impairment in this system.

**Summary**

Sulfate concentrations are elevated in Wyman Creek due to the presence of a mine pit dewatering discharge in its headwaters. All sulfate results from this watershed met water quality standards being applied in the states of Illinois, Indiana, Iowa, and Pennsylvania. The lack of a water quality standard in Minnesota presents challenges in building a defensible case for or against sulfate as a stressor to aquatic life. Based on the data and supporting information available at this time, it is unlikely that sulfate is a primary cause of impairment in Wyman Creek. However, the presence of numerous beaver impoundments between the source of sulfate (headwaters mine pits) and biological monitoring stations may be creating conditions that are favorable (e.g. low DO) for reduction of sulfate to sulfide, which is a known toxicant. Therefore, sulfate/sulfide cannot be entirely eliminated as a potential stressor.

## 5.14.6 Wyman Creek: Summary of Stressors to Aquatic Life

**Table 60:** Summary of SID results for Wyman Creek

| Candidate Cause | Result |
|---|---|
| **Elevated Water Temperatures** | ● |
| **Low Dissolved Oxygen** | ● |
| **Loss of Connectivity due to Beaver Dams and Road Crossings** | ● |
| **Total Suspended Solids (TSS) / Turbidity** | X |
| **Habitat Impacts from Iron Precipitate** | ○ |
| **Iron Toxicity** | ○ |
| **Sulfate Toxicity** | ○ |
| Key: ● **= confirmed stressor** ○ **= Potential Stressor X = eliminated candidate cause** | |

### Nashwauk Uplands – Embarrass River Watershed Zone

Impaired streams of this watershed zone include Spring Mine Creek, Ely Creek, and the headwaters reach of the Embarrass River. These impaired segments of these streams share similar natural background qualities in that they are relatively low in gradient, moderately sinuous, and have broad floodplains with wetland qualities. Beaver dams are common features in all three of these streams, and these impoundments appear to have a significant effect on channel pattern, in-stream habitat, and water surface slope. Physical habitat conditions in these streams are somewhat limited due to the lack of riffle-run features and an abundance of fine substrates. Mining activity in this watershed zone introduces the potential for point-source pollution as a stressor, particularly in the case of Spring Mine Creek, which originates from a mine pit high atop Giants Ridge.

**Figure 223:** Typical stream reaches of the three impaired streams in this watershed zone. Spring Mine Creek (left), Embarrass River (middle), Ely Creek (right)

Overall fish counts were low in all three of the impaired streams in this watershed zone, and the species present were generally not sensitive or intolerant of disturbance. Species tolerant of low DO (Central Mudminnow, Brook Stickleback) were present in high numbers relative to other fish species at impaired sites. Aside from a very small population of Pearl Dace observed in Spring Mine Creek, headwaters minnow species and darter species were absent from the impaired reaches.

The habitat conditions available in these low-gradient, wetland dominated watersheds may be naturally limiting in terms of supporting a diverse fish assemblage. Bear Creek, a second-order tributary of the Embarrass River has been used as a reference stream in previous studies involving biological integrity in this region of the SLRW. There is very little development and no mining land-use in the Bear Creek Watershed, but many of the natural limitations (low gradient, wetland riparian corridor, lack of coarse substrate) are shared with the Embarrass River and Spring Mine Creek. Despite its relatively intact watershed, Bear Creek scored only three points higher than the impairment threshold and is comparable to the impaired streams in terms of fish abundance and species distribution. The impaired streams in this watershed are certainly impacted by anthropogenic activity, but further analysis of available reference conditions to base restoration efforts on is recommended. The relative contributions of natural and anthropogenic stressors will be further discussed for these streams in the candidate causes for impairment section.

## 5.15 Spring Mine Creek

Spring Mine Creek originates at the top of the Mesabi Range in Spring Mine Lake, which lies in the middle of an intensely mined area to the southeast of Embarrass, Minnesota. Due to mining activities, the creek is ditched for 0.75 miles immediately after flowing out of Spring Mine Lake. After leaving the mining area the stream plunges down the side of the Range for more than a mile as a 3.0% B channel. Spring Mine Creek then flows into a wide lacustrine valley and takes on the characteristics of a sinuous E channel. The impaired AUID begins in this reach. After this, the stream picks up gradient (1.6%) and makes the final descent into the Embarrass River valley, where the stream type is an E channel and the valley type is lacustrine. Overall, the average slope of Spring Mine Creek is close to 1%, dropping almost 250 feet in its 5-mile journey.

Spring Mine Creek is the only stream in this watershed zone that is listed as impaired for macroinvertebrate bio assessments. The M-IBI results from this stream were narrowly below the impairment criteria and do not suggest severe impairment. However, ancillary information considered in the assessment process (elevated specific conductivity readings; invertebrate samples dominated by *Gammarus* and *Corixidae*) resulted in an impairment listing. Symptoms of impairment observed in Spring Mine Creek include a very low relative percentage of non-Hydropsychid caddisfly taxa (1.6%) and imbalance in the distribution of taxa present. Over 76% of the individuals counted were from the five most abundant taxa in the sample. Bear Creek, which has been discussed as a potential reference stream for this watershed zone, shows more balance among taxa present, supports more intolerant taxa, and better representation from the order Trichoptera (Table 61).

**Table 61:** Summary of biological monitoring stations and sampling results in Spring Mine Creek

| **Fish Assessments** | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Station** | **Drainage Area (mi2)** | **Gradient (%)** | **Stream Order (Strahler)** | **FIBI Class** | **FIBI Result *(visit year)*** | **FIBI Result *(visit year)*** | **Standard** | **IBI Lower Confidence Limit** | **IBI Upper Confidence Limit** |
| 09LS101 | 4.43 | 0.31 | 2 | 6 | **37** *(2009)* | **37** *(2009)* | 42 | 26 | 58 |
| **Macroinvertebrate Assessments** | | | | | | | | | |
| **Station** | **Drainage Area (mi2)** | **Gradient (%)** | **Stream Order (Strahler)** | **Invert IBI Class** | **Invert IBI Result *(visit year)*** | **Invert IBI Result *(visit year)*** | **Standard** | **IBI Lower Confidence Limit** | **IBI Upper Confidence Limit** |
| 09LS101 | 4.43 | 0.31 | 2 | 4 | **46.35** *(1997)* | - | 51 | 37.40 | 64.60 |

**Table 62:** Comparison of several macroinvertebrate metric results between Bear Creek and Spring Mine Creek

| | **Bear Creek** | | **Spring Mine Creek** | |
|---|---|---|---|---|
| ***M-IBI Metric*** | **Metric Value** | **Metric Score** | **Metric Value** | **Metric Score** |
| **Richness of Intolerant Taxa*** | 3 | ***10.0*** | 1 | ***5.0*** |
| **% Trichoptera Taxa** | 18.9% | ***10.0*** | 5.7% | ***3.5*** |
| **% Trichoptera Taxa *(excludes Hydrophyschidae)*** | 5.8% | ***7.8*** | 1.6% | ***3.8*** |
| **POET Taxa Richness**** | 12 | ***7.1*** | 7 | ***3.6*** |
| **% Dominant Five Taxa***** | 65.2% | ***5.4*** | 76.7% | ***3.0*** |

* Taxa richness of macroinvertebrates with tolerance values (TV) less than or equal to 2, using MN TV (Chirhart, source)

** Taxa richness of Plecoptera, Odonata, Ephemeroptera, & Trichoptera (baetid taxa treated as one taxon)

The FIBI scores from the two visits during the summer of 2009 were both 37 (out of a possible 100), which is five points below the impairment threshold and within the lower confidence limit of the IBI standard (Table 61). Common Shiner, White Sucker, Brook Stickleback, and Creek Chub were the dominant taxa present in the June 2009 sampling event. A similar community was observed in the re-sampling of this site in September of the same year. Small populations of several sensitive fish taxa were present in both samples, including Pearl Dace, Burbot, and Blacknose Shiner. Water temperatures in spring mine creek tend to be colder than surrounding stream due to the deep mine pits that feed water to this stream in its headwaters. As a result, fish that prefer cool to cold water temperatures (Burbot, Pearl Dace) seem to be finding suitable conditions in this stream. The low FIBI scores for this station are primarily due to low scores in metrics related to overall fish abundance.

Available water quality, biological, physical habitat, and land-use data were reviewed to develop a list of candidate causes for the fish and macroinvertebrate impairments in Spring Mine Creek. The following candidate causes for impairment will be evaluated in this section:

1. **Low DO / High DO Flux**
2. **High Specific conductivity**
3. **Sulfate Toxicity**

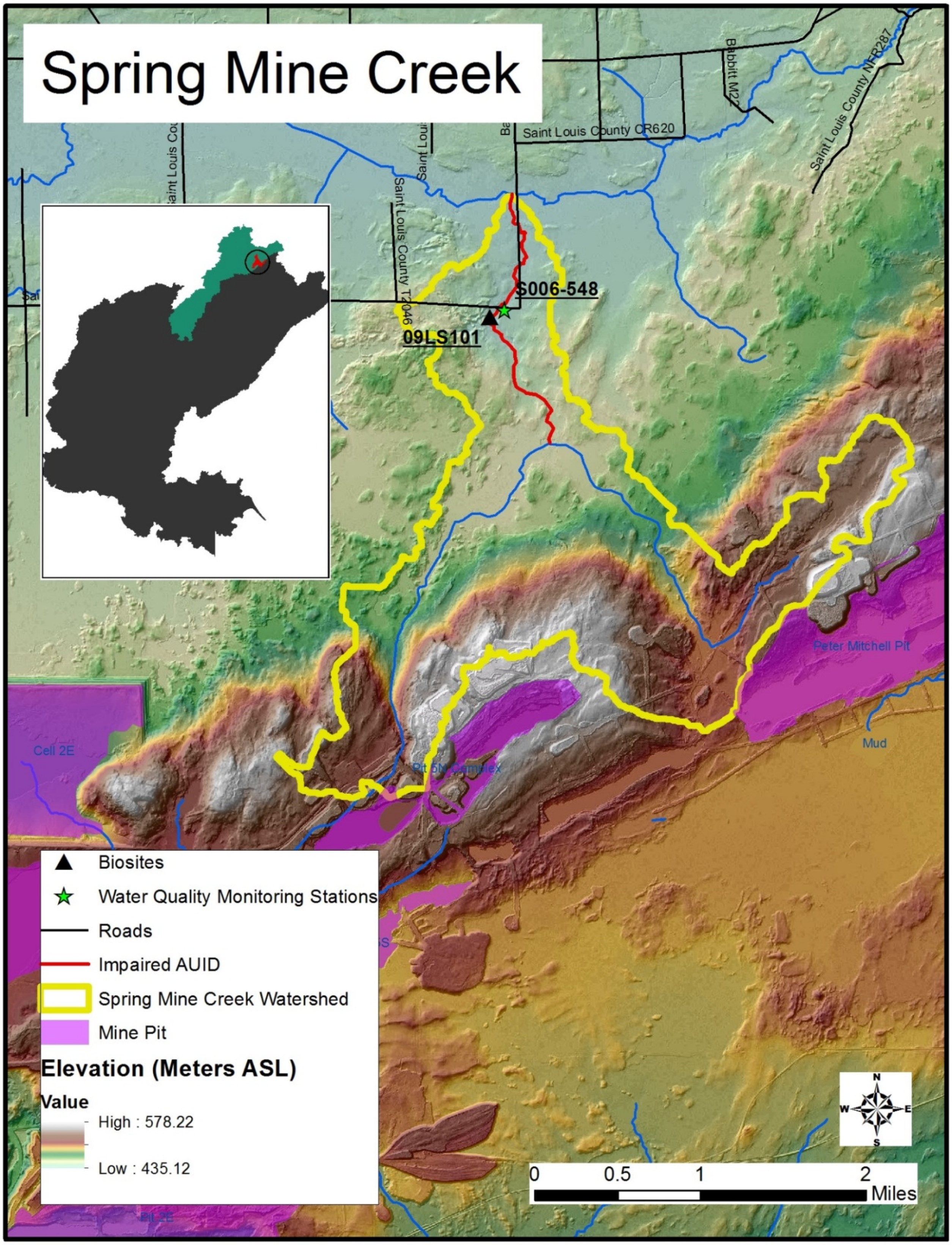


**Figure 224:** Map of Spring Mine Creek Watershed and impaired reach

### 5.15.1 Low Dissolved Oxygen

With only one road crossing and a high percentage of the watershed in private ownership, access to Spring Mine Creek for monitoring purposes was extremely limited. As a result, instantaneous DO data were collected at only one monitoring station. A total of 13 instantaneous DO measurements were collected between the months of April and November, with only one reading falling below the 5 mg/L warmwater DO standard (2.61 mg/L; August 29, 2013) (Figure 225). The majority of mid-summer DO concentrations recorded in Spring Mine Creek are in the range of 5-8 mg/L, which is a suitable range for supporting healthy warmwater fish and macroinvertebrate communities.

Continuous monitoring data for DO and other parameters were collected in 2011, 2012, 2013, and 2015 at biological monitoring station 09LS101. The DO concentrations dropped below the warmwater standard of 5 mg/L periodically for relatively short durations (up to 12 hours) during the months of July and August (Figure 226).

Average DO diurnal flux (DO flux) in Spring Mine Creek ranged from 1.44 mg/L to 5.43 mg/L over the three shorter duration continuous monitoring periods in 2011, 2012, and 2013. The longer duration, 2015 data set shows much higher DO flux of up to 7.49 mg/L and many days with DO flux in the range of 5-6 mg/L. (Figure 227). These values exceed the Northern river nutrient stressor criteria for DO flux, which is set at 4.00 mg/L. Based on these results, DO flux in Spring Mine Creek is highly variable, and is high enough at times to present stressful conditions to sensitive aquatic life.

Minimum DO concentrations in Spring Mine Creek drop below 5 mg/L periodically, but are generally not extremely low (e.g. below 2-3 mg/L) and duration of sub-5 mg/L DO periods tend to be short. DO flux is likely more of a candidate stressor than low minimum DO concentrations. Each of these candidate causes for impairment will be further evaluated in this section.

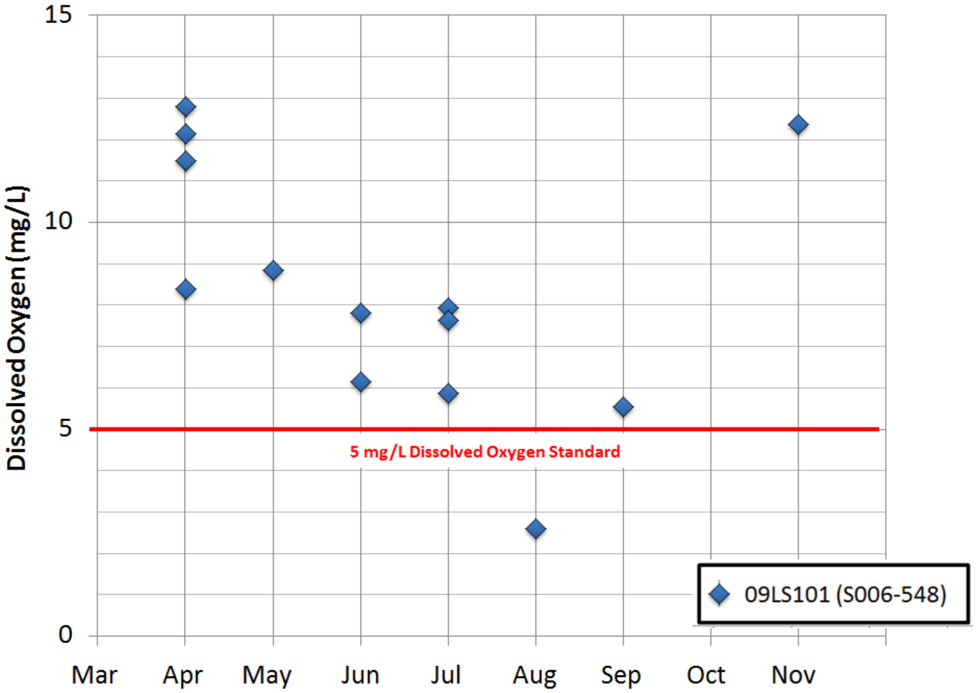
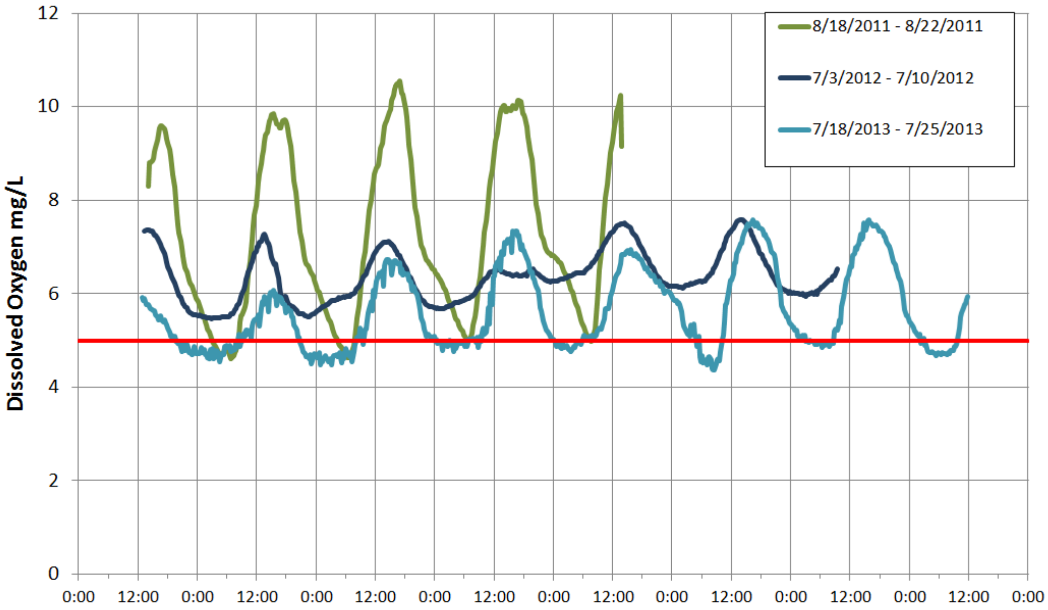


**Figure 225:** Point measurements of DO collected at biological monitoring station 09LS101



| | # of Readings | Min | Max | 24-Hour Flux Data | | Warm Water Threshold - 5 mg/L | | |
|---|---|---|---|---|---|---|---|---|
| | | (mg/L) | (mg/L) | Average (mg/L) | Max (mg/L) | % Readings below | Avg. Duration below (hrs) | Max Duration below (hrs) |
| 8/18/2011 - 8/22/2011 | **384** | **4.62** | **10.55** | **5.43** | **5.92** | **9.4%** | **3** | **5.3** |
| 7/3/2012 - 7/10/2012 | **660** | **5.46** | **7.59** | **1.44** | **1.82** | **0.0%** | **0** | **0** |
| 7/18/2013 - 7/25/2013 | **669** | **4.37** | **7.57** | **2.42** | **3.20** | **32.6%** | **2.7** | **12.3** |

**Figure 226:** Continuous DO monitoring results from station 09LS101

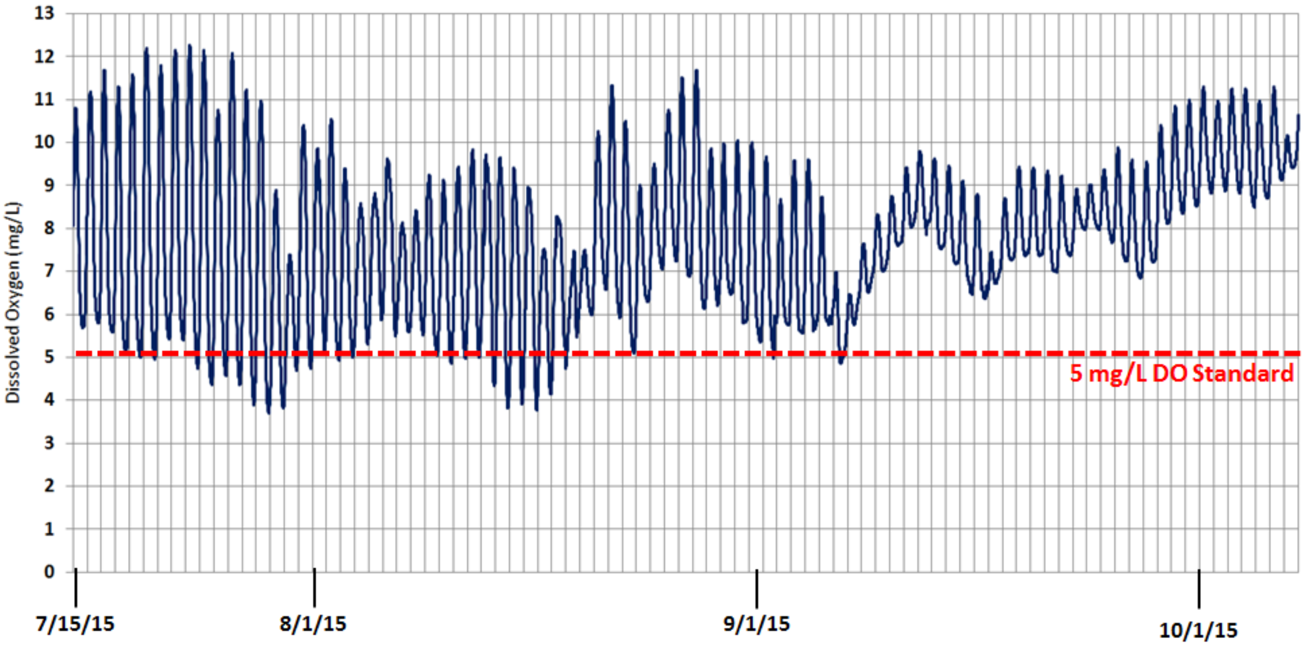


**Figure 227:** Additional continuous DO monitoring results from station 09LS101. Measurements were collected at 15-minute intervals from 7/15/15 to 10/9/15. Each rectangle represents a 24-hour period.

### Sources and Pathways Contributing to Low DO

The TP concentrations in Spring Mine Creek ranged from 0.009 mg/L to 0.074 mg/L (n=10, avg. = 0.033 mg/L). Two of the ten results for TP exceed the 0.055 mg/L nutrient criteria target for northern Minnesota streams and rivers. The maximum of 0.074 mg/L was observed during a snowmelt runoff event, and the other exceedance (0.065 mg/L) was observed during low flow conditions in early fall. Based on the small data set for this parameter, TP concentrations are not elevated throughout the year, but do periodically climb to levels that can lead to excess productivity and cause stress to aquatic life.

The BOD data for this stream are minimal, with two sampling results from summer of 2015. BOD concentrations from these samples were 2.1 mg/L and 1.9 mg/L, which both exceed the 1.5 mg/L benchmark for this parameter in the river nutrient criteria for “northern rivers” stream grouping. Chlorophyll-a data were not collected, but would help to provide a clearer picture of the processes that are resulting in the rare low DO readings and the observations of elevated DO flux.

The DO regime of Spring Mine Creek is likely influenced seasonally by the presence of submergent, emergent, and floating leaf aquatic vegetation. During site visits to station 09LS101, aquatic vegetation was generally present but somewhat sparse in the spring and early summer months. By August and September, a significant portion of the channel was typically covered by these vegetation types (Figure 228). As the amount of aquatic vegetation increases, DO flux generally increases as well due to photosynthesis and respiration process within the plant community. The elevated DO flux observed in Spring Mine Creek in August of 2011 may be linked to the increase in vegetation observed within the stream channel.

**Figure 228:** Photos of station 09LS101 looking upstream. Photo on the left was taken in June of 2009, and the photo on the right was taken in August 2009. Note the increase in floating leaf and emergent vegetation in the August 2009 photo.

Wetlands and hydric soil types are present in the Spring Mine Creek Watershed, particularly in the riparian corridor along the middle to lower reaches of the creek (Figure 229). These features have the potential to release anoxic water, or water with lower DO content to the stream. Given that very few periods of extremely low DO were observed at station 09LS101, the linkage between wetlands and low DO does not appear to be as strong in this impaired sub-watershed. Instead, the wetland influence in Spring Mine Creek appears to be more related to elevated DO flux. Water clarity was generally much higher in Spring Mine Creek compared to other wetland-influenced streams of the SLRW, likely due to mine pits at its headwaters, which discharge very clear water to the creek and accounts for the majority of the flow in Spring Mine Creek during baseflow. The high water clarity and stagnant flows of Spring Mine Creek at bio station 09LS101 promote growth of aquatic macrophytes, and the DO regime is affected during periods of growth and senescence of these plants.

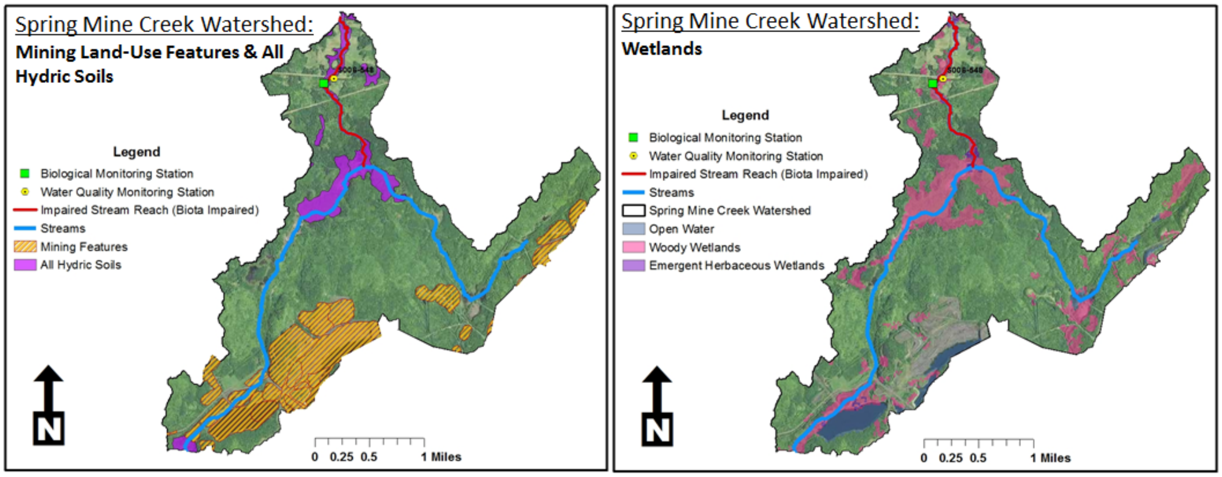


**Figure 229:** Maps of Spring Mine Creek drainage area showing locations of all hydric coils and mine features (left) and wetland features (right).

### Biological Response to Dissolved Oxygen Stressor

The fish community at station 09LS101 of Spring Mine Creek consists primarily of species that are often found in low gradient wetland streams. Generally, the species observed at this station range from neutral to highly tolerant in terms of their tolerance to low DO conditions. Brook Stickleback, Central Mudminnow, Blacknose Shiner, and Pearl Dace are examples of fish species observed at station 09LS101 that tend to occupy or dominate stream reaches with lower DO content. Central Mudminnow and Brook Stickleback are often prevalent in streams with very low DO levels. Combined, these two species accounted for a relatively large percentage (25% and 50%) of the overall fish community over the two sampling visits (Figure 230). No fish species known to be sensitive to low DO concentrations were observed during the two sampling events. However, the presence of “neutral” species such as White Sucker, Creek Chub, and Burbot in this stream suggest that DO concentrations in Spring Mine Creek are not low enough to exclude all but tolerant species.

The fish metric *TolPct* (% of total individuals that are tolerant species) has shown to be responsive to DO flux based on regression analysis using a Minnesota statewide data set (Laing 2014, personal communication). As DO flux increases, the percent of tolerant individuals also tends to increase. Spring Mine Creek supports a higher percentage of tolerant fish than most other SLRW sites of the same FIBI class. Over 72% of the fish sampled in the original fish survey conducted in June of 2009 were tolerant species. The percentage of tolerant fish dropped slightly to 60% in the follow-up survey conducted in September of 2009. The results from Spring Mine Creek for this specific metric are on par with the 75th percentile values observed at SLRW reference streams that scored above the class 6 FIBI impairment threshold.

The macroinvertebrate community did contain several taxa that are found in streams that generally have very good DO conditions. The freshwater amphipod *Gammarus* was the most abundant taxa represented in the sample, accounting for 41% of the total organisms that were identified and counted. Although members of the genus *Gammarus* are often tolerant of other stressors (nutrients, specific conductivity, suspended sediment), they are often associated with cold streams that have ample DO. Their abundance in Spring Mine Creek may be more related to water temperature regime as opposed to DO conditions.

Other than *Gammarus*, no additional low DO intolerant taxa were observed in Spring Mine Creek at station 09LS101. Overall, most of the macroinvertebrate taxa present in Spring Mine Creek are not sensitive to low DO environments. A total of 9 low DO tolerant taxa were observed, and of these, 5 can be considered “very tolerant” of low DO. The total number of taxa sampled at this station was 37, so nearly 25% of the total taxa observed are tolerant of low DO conditions. A fairly small portion of *individuals* sampled were from low DO tolerant taxa (16%). Overall, this location of Spring Mine Creek is not dominated by taxa or individuals that are tolerant of low DO, but there are some tolerant taxa present.

The MPCA’s River Nutrient Criteria (Heiskary et al 2013) observed a negative correlation between DO flux of over 4.0 mg/L and macroinvertebrate taxa richness. Twenty four macroinvertebrate taxa were observed in the sample at station 09LS101, which is considerably lower than the majority of comparable sites that scored above the MIBI standard. By comparison, the median taxa richness observed at high quality class 4 biological monitoring stations in the SLRW ranged from 30 – 37 taxa. The relatively low taxa richness observed at 09LS101 provides evidence in support of DO flux as a stressor. However, this symptom of impairment can be caused by other confounding stressors that may be present in Spring Mine Creek (e.g. elevated conductivity).

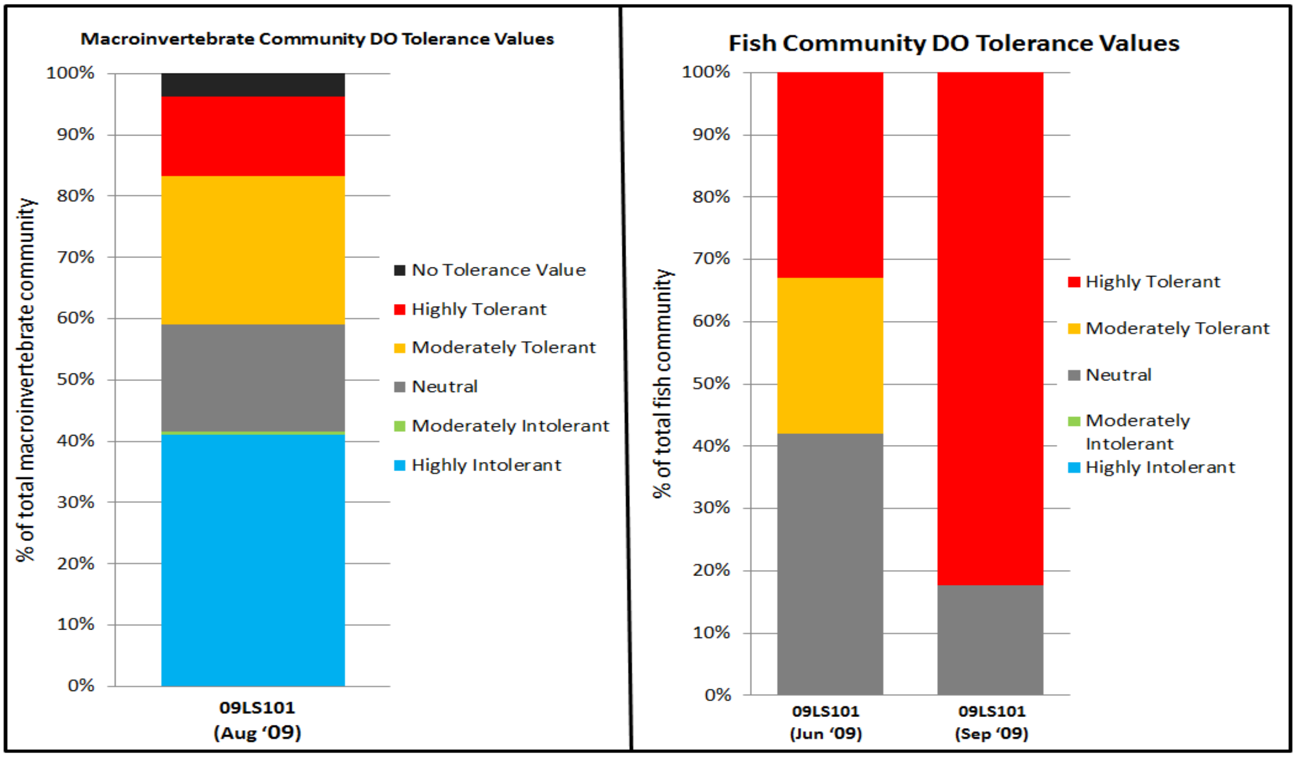


**Figure 230:** Proportion of fish and macroinvertebrate population in Spring Mine Creek that are tolerant, neutral, or intolerant of low DO concentrations

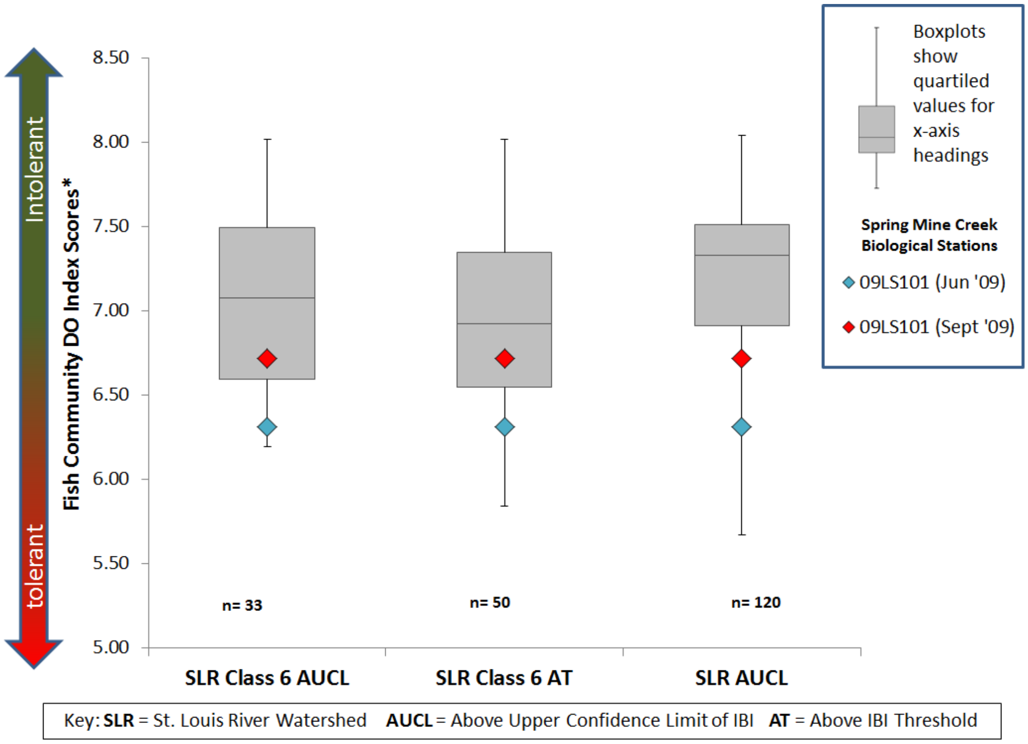


**Figure 231:** Fish community DO TIV results for Spring Mine Creek compared to high quality stations of the same IBI class

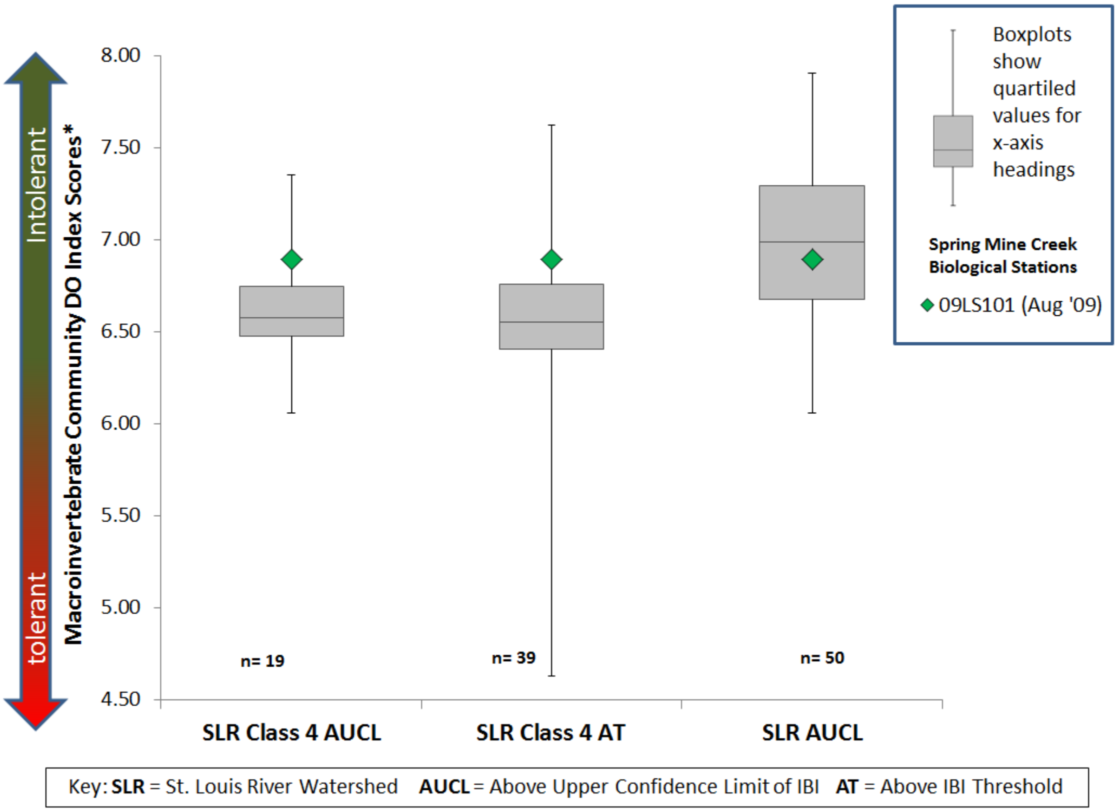


**Figure 232:** Macroinvertebrate community DO TIV results for Spring Mine Ck vs. high quality stations of the same IBI class

**Summary: Is low DO / DO flux a stressor in Spring Mine Creek?**

Very few sub-5mg/L DO measurements were observed in our data set, and the magnitude and durations of exposure to low DO concentrations were not severe. The lowest DO measurement of 2.61 mg/L on August 29, 2013 is cause for some concern, but that data point appears to be somewhat of an outlier when considering all of the data collected from 2011-2015. The relatively high diurnal DO flux (> 7 mg/L) is more problematic than low minimum DO concentrations. The MPCA's River Nutrient Criteria cited a DO flux value of 4.0 mg/L and greater as a stressor symptom of eutrophication. The DO flux at station 09LS100 appears to be greater than 5 to 6 mg/L with regularity.

Fish and macroinvertebrate data from Spring Mine Creek offer some evidence in support of low DO as a stressor. Brook Stickleback and Central Mudminnow (both tolerant of low DO conditions) were found in large populations relative to other species that were present at station 09LS101. The invertebrate sample was dominated by *Gammarus*, which prefer cold streams that typically have good DO levels due to the colder water temperatures. The rather cold water temperatures observed in Spring Mine Creek are probably responsible for the presence of *Gammarus* as opposed to the DO conditions. These organisms are often found in nutrient rich streams, so their presence does not provide evidence against DO flux as a stressor, but may instead be supporting evidence of DO flux as a cause of impairment.

Low minimum DO is an unlikely stressor, but cannot be eliminated entirely due to several measurements that fell below the water 5 mg/L standard. The high DO flux observed in this system should be considered a stressor based on monitoring results and biological response data.

## 5.15.2 Specific Conductivity

Specific conductivity levels observed at biological monitoring station 09LS101 on Spring Mine Creek range from a low of around 100 - 400 µS/cm during spring snowmelt to a high of >1,400 µS/cm during winter baseflow (Figure 233). Continuous specific conductivity data (15 min. intervals) were collected in 2011, 2012, 2013, and 2015. The 2011 – 2013 measurements were relatively short in duration (about 7 days). In 2015, measurements were collected with very minimal gaps from mid-July to early October. Over this monitoring period, specific conductivity at station 09LS101 was predominantly in the range of 800-1,100 µS/cm, with short periods above and below this range in values (Figure 234).

Based on the continuous and point sampling results, aquatic biota in Spring Mine Creek are chronically exposed to specific conductivity levels in the range of 700 - 1,500 µS/cm at biological monitoring station 09LS101. Exposure to higher conductivity levels are likely in reaches of Spring Mine Creek upstream of monitoring station 09LS101, given that the sources of elevated conductivity in this watershed are mine pits in its headwaters. Monitoring results from the permitted mine pit discharge to Spring Mine Creek show conductivity levels leaving the mine pits average 2,000 – 2,400 µS/cm (Figure 235).

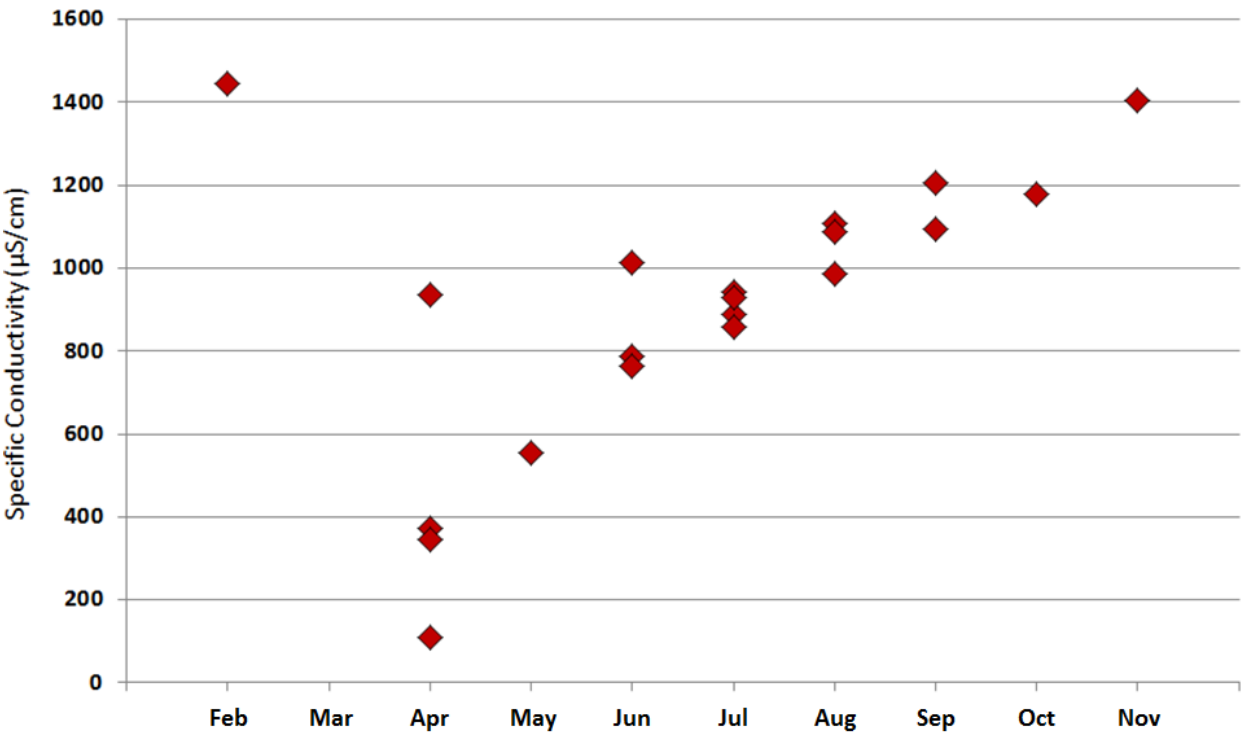


**Figure 233:** Point measurements of specific conductivity collected at biological monitoring station 09LS101 by month. Data was collected between 2011 – 2015.

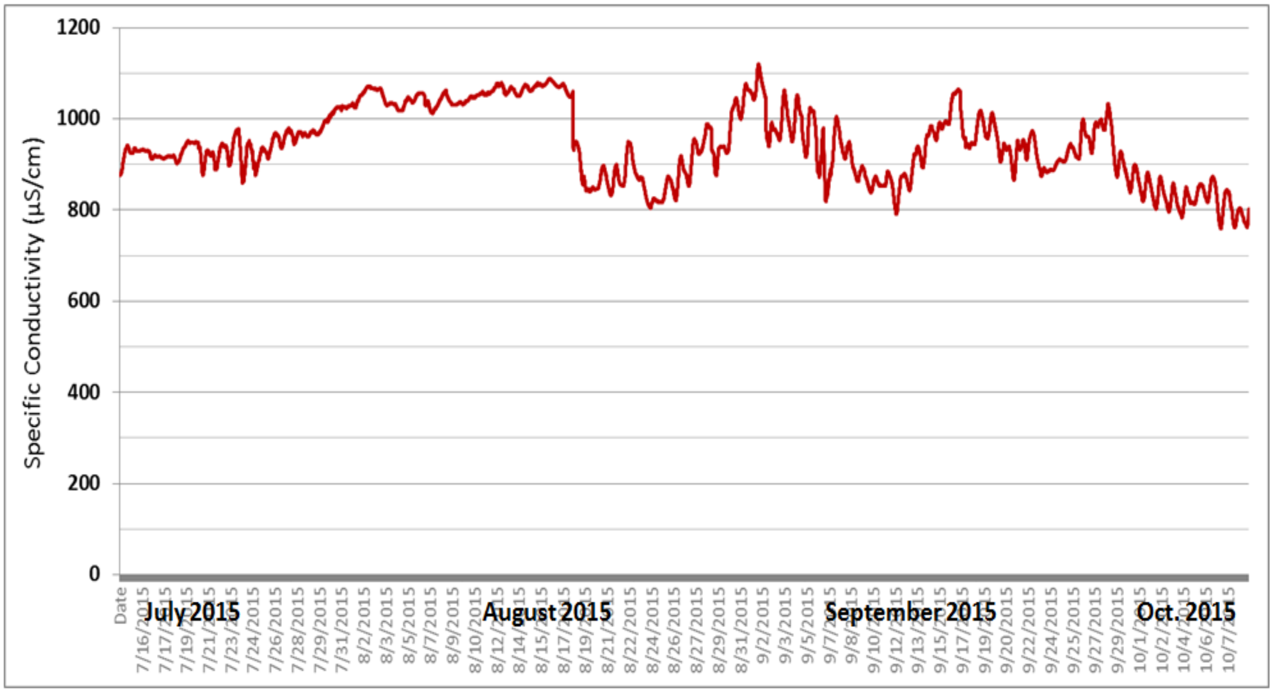


**Figure 234:** Continuous specific conductivity data collected in 15-minute intervals at biological monitoring station 09LS101 during the 2015 season

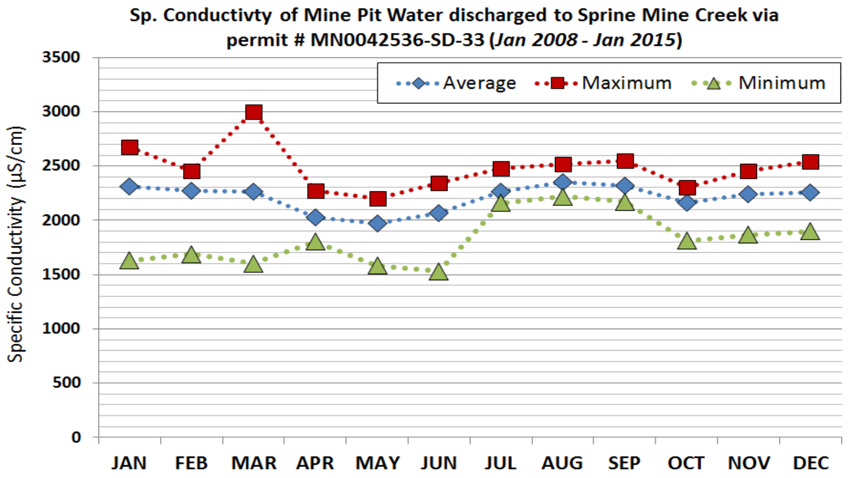


**Figure 235:** Continuous specific conductivity data collected at the outlet of the abandoned mine pit that serves as the headwaters of Spring Mine Creek.

The most common contributors to salinity (a surrogate for elevated specific conductivity) in surface waters are referred to as matrix ions, and include the positively charged cations $Ca^{2+}$, $Mg^{2+}$, $Na^{+}$, $K^{+}$, and negatively charged anions $HCO_3^{-}$, $CO_3^{2-}$, $SO_4^{2-}$, and $Cl^{-}$. Samples were collected from Spring Mine Creek during baseflow, snowmelt, and rain event flow conditions to better understand the geochemical composition of its surface water. Samples were also collected at nearby Bear Creek, which has a relatively undisturbed watershed and may represent natural background conditions for streams in this region of the SLRW.

Figure 236 show the geochemistry results for Spring Mine Creek and Bear Creek. Bear Creek had very low concentrations of most major ions that were evaluated. On the contrary, concentrations of several ions were notably elevated in Spring Mine Creek, particularly sulfate ($SO_4^{2-}$) and magnesium ($Mg^{2+}$). Specific conductivity measurements from these two streams taken during the baseflow sampling event reflect the differences in the concentrations of these ions. At baseflow, specific conductivity in Bear Creek was 120 µS/cm, compared to 1420 µS/cm in Spring Mine Creek

Although several cations and anions were not included in this analysis, it can be concluded that sulfate is a major driver of specific conductivity in Spring Mine Creek. Sulfate toxicity will be evaluated a stressor to aquatic life later in this sections, but it is likely that the effects of sulfate and related increases in specific conductivity are two stressors that are closely linked.

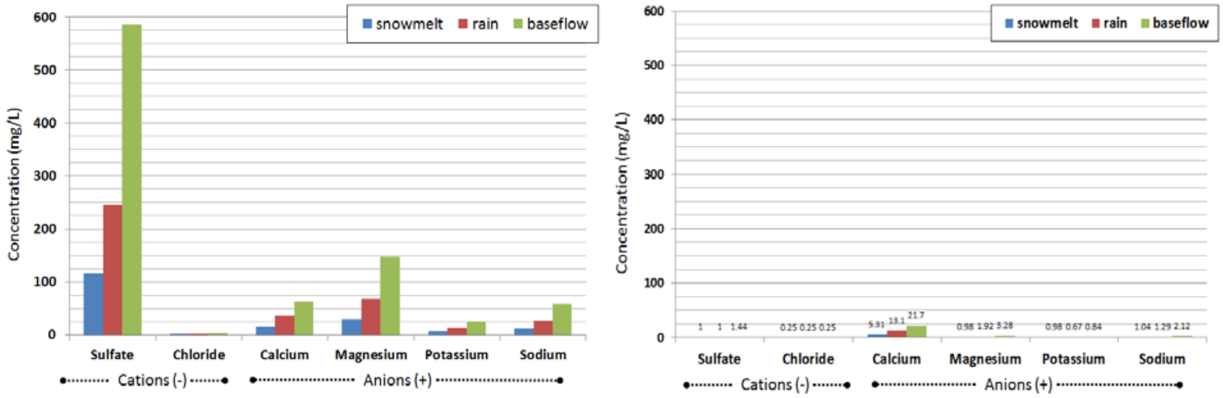


**Figure 236**: Concentrations of major cations and anions observed in Spring Mine Creek (left; impacted) and Bear Creek (right; undisturbed reference).

## Biological Response to Elevated Specific conductivity

The effects of elevated conductivity on aquatic life were evaluated using data from Minnesota streams and scientific literature. A summary of the biological responses that have been observed in the presence of high specific conductivity are presented in Section 3.1.5. Based on the literature that has been compiled, several biological metrics were selected to evaluate specific conductivity as a stressor in Spring Mine Creek (Table 63).

**Table 63:** Summary of biological metrics and literature used to evaluate elevated specific conductivity as a stressor

| Metric | Response to Increased Specific conductivity / Conductivity | Source |
|---|---|---|
| **EPT Richness** | Decrease | Roy et al (2003); Echols et al (2009); Johnson et al (2013) |
| **Overall Taxa Richness** | Decrease | Johnson et al (2013) |
| **Ephemeroptera Richness** | Decrease | Pond (2004); Hassel et al (2006) |
| **Fish and Macroinvertebrate Tolerance Indicator Values (TIV)** | Increase | MBDI (Yoder and Rankin, 2012) |

### EPT Richness

Richness of EPT taxa in Spring Mine Creek is lower than all comparable MIBI stations (Class 4) which score above the impairment threshold. Only 3 EPT taxa were present at 09LS101, while the median EPT taxa richness for high quality biological monitoring stations in the SLRW ranged from 10 – 16 EPT taxa (Figure 237). Although a lack of EPT taxa can be related to other water quality and physical stressors (e.g. low DO and/or lack of habitat), specific conductivity levels in Spring Mine Creek are elevated enough to limit sensitive EPT taxa from becoming established.

### Overall Macroinvertebrate Taxa Richness

Twenty four macroinvertebrate taxa were observed in the sample at station 09LS101, which is considerably lower than the majority of comparable sites that scored above the MIBI standard. By comparison, the median taxa richness observed at high quality class 4 biological monitoring stations in the SLRW ranged from 30 – 37 taxa (Figure 237). The relatively low taxa richness observed at 09LS101 provides evidence in support of conductivity as a stressor. However, this symptom of impairment can be caused by other confounding stressors that may be present in Spring Mine Creek (e.g. low DO).

### Ephemeroptera Taxa Richness

*Eurylophella* was the only Ephemeropteran (mayfly) taxa represented in the sample collected at 09LS101. This mayfly genus tends to be somewhat sensitive to disturbance, so its presence at this monitoring station suggests that Spring Mine Creek does offer some suitable conditions for supporting small populations of sensitive taxa. However, *Eurylophella* were extremely scarce in the sample, representing less than 1% of the organisms counted.

With only one Ephemeroptera taxa represented in the sample, Spring Mine Creek ranks below all of the reference sites in terms of richness for that macroinvertebrate order (Figure 238). Median Ephemeroptera richness at sites with good to exceptional MIBI scores ranged from 4-6 taxa. The lack of Ephemeroptera taxa in Spring Mine Creek provides supporting evidence for elevated conductivity as a stressor.

### Specific Conductivity Tolerance Indicator Values

The MPCA has developed macroinvertebrate TIVs for various parameters based on statewide biological and water chemistry data sets. Community level TIV for specific conductivity are shown for Spring Mine Creek fish and macroinvertebrate communities in Figure 239. The TIV score for the macroinvertebrate community at 09LS101 is very high compared to the vast majority of TIV results from reference sites (Figure 239). This result is an indication that the macroinvertebrate community found at 09LS101 is composed primarily of taxa that are found at locations with elevated conductivity levels.

The results are somewhat different for fish, which tend to be more tolerant of elevated specific conductivity. Fish community TIV scores at 09LS101 were slightly elevated compared to high quality reference sites in the SLRW, but did not show the same level of divergence from the high quality sites as seen in the macroinvertebrate TIV scores.

### Summary: Is elevated specific conductivity a stressor in Spring Mine Creek?

Biological data from Spring Mine Creek provide evidence in support of elevated specific conductivity as a stressor, particularly in the macroinvertebrate community. However, confidence in diagnosing conductivity as a stressor is weakened by the possibility of unrelated confounding stressors that also appear to be present at the site (lack of habitat, DO flux). Additional monitoring sites in the higher gradient reaches upstream of 09LS101, where habitat seems more suitable for aquatic life, could potentially separate the impacts of specific conductivity from other confounding stressors. This additional monitoring is planned for 2015, and the data should add confidence to decisions regarding this potential stressor.

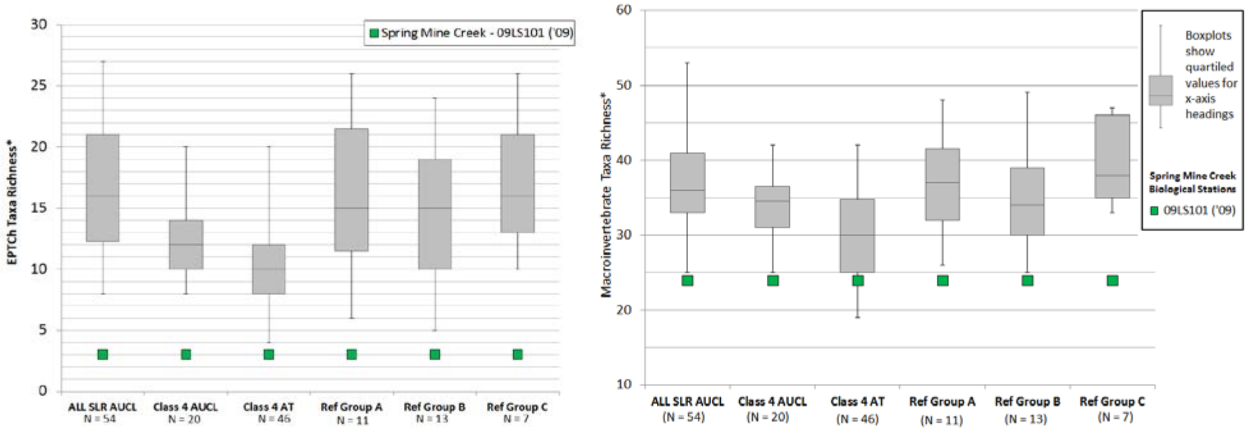


**Figure 237:** EPT taxa richness (left) and overall taxa richness (right) from Spring Mine Creek compared to high quality reference stations. *See section 1.2.3 for list of reference stations. See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

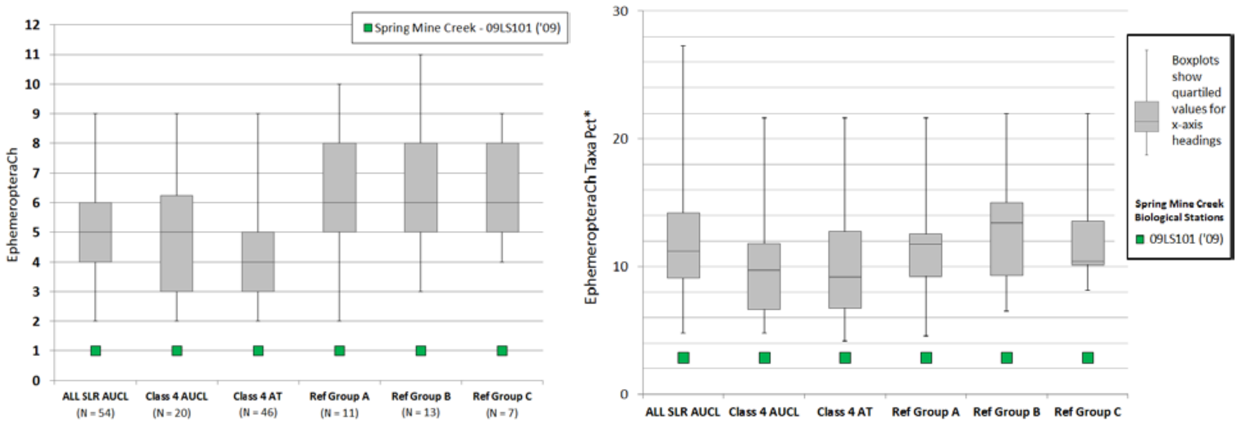


**Figure 238:** Ephemeroptera taxa richness (left) and Ephemeroptera taxa percent (right) from Spring Mine Creek compared to high quality reference stations. *See section 1.2.3 for list of reference stations. See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

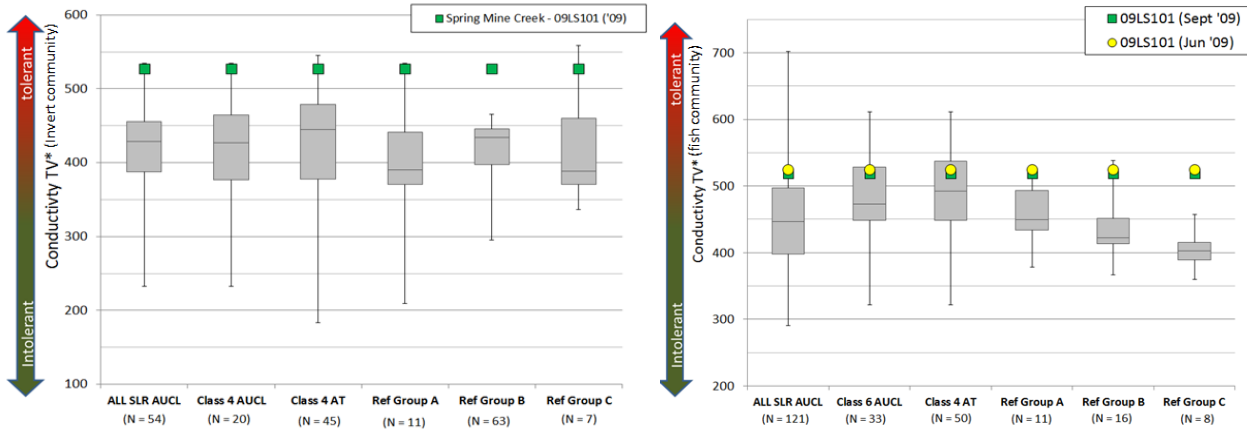


**Figure 239:** Community level tolerance indicator values (TV) for fish (right) and macroinvertebrate (left) populations in Spring Mine Creek compared to high quality reference stations. *See section 1.2.3 for list of reference stations. See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

### 5.15.3 Sulfate Toxicity

Elevated sulfate concentrations were identified as a candidate cause for biological impairment in Spring Mine Creek based on monitoring results from samples collected between the years 2009 and 2015. A total of twelve samples were collected over a range of flow conditions, including winter baseflow samples through ice during the month of February. Sulfate concentrations ranged from a low of 192 mg/L during high flow, snowmelt runoff conditions in April, to a maximum of 751 mg/L during winter baseflow. The sampling results clearly show sulfate concentrations increasing steadily as flows decrease towards annual minimums in late fall and winter (Figure 240).

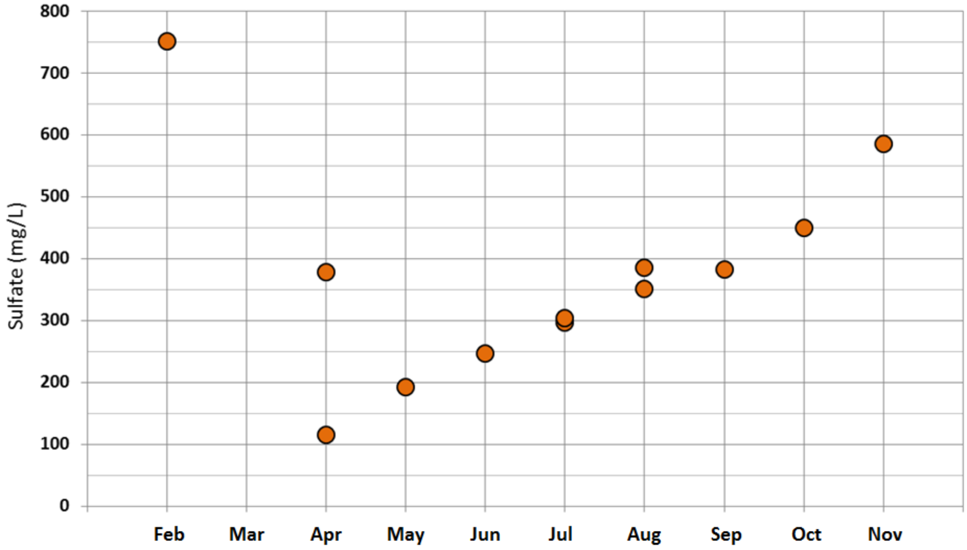


**Figure 240:** Grab sample results for sulfate collected at biological monitoring station 09LS101

#### Sources and Pathways of Sulfate with the Spring Mine Creek Watershed

Sulfate is a common compound generally found in low concentrations in pristine or lightly impacted streams. Sulfate concentrations averaged 3.9 mg/L (min = <1.0 mg/L; max = 15.0 mg/L) in a set of 61 samples from 10 high quality streams in the SLRW (see Table 2). In the northern regions of the SLRW where mining land uses are a prominent feature of the landscape, sulfate concentrations in select streams can be highly elevated. Higher sulfate concentrations are also seen along the length of the St. Louis River, particularly downstream of the Iron Range.

Sulfate is often the dominant contaminant from mine water and can form a wide range of salts (Mining and Environmental Management Magazine 2000). Significant concentrations of sulfate can accumulate in surface water that is not frequently flushed, such as water found in mine pit lakes or behind stream impoundments. This scenario is occurring in the headwaters of Spring Mine Creek, as mine pits contributing flow to the creek are very high in sulfate concentration. The continuous discharge from these pits into Spring Mine Creek is resulting in elevated sulfate concentrations further downstream where biological and water chemistry data were collected.

Monitoring data available per the requirements of NPDES permit MN0042536-SD-33 provide documentation of the quantity and quality of water being discharged to Spring Mine Creek at its headwaters. The location of this monitoring site (shown in Figure 241) is several hundred feet downstream of several mine pits, which discharge continuously to the creek. Bi-monthly flow data collected between 2001 and 2014 show an average monthly maximum discharge ranging between 0.9 and 2.9 cubic feet per second (cfs) (Figure 242). The maximum discharges recorded over this period are in the range of 7 to 8 cfs, and generally occur during spring snowmelt periods, and again in the early fall.

Sulfate concentrations were also monitored at this station as a part of the requirements for the discharge permit. Based on data from 2007 through 2014, the average sulfate concentration of surface water entering Spring Mine Creek from the mine pits is greater than > 1000 mg/L in every month except for May. Snowmelt runoff conditions dilute sulfate concentrations over the months of March through May, and a steady increase in sulfate can be observed as baseflow conditions set in during the summer, fall, and winter months.

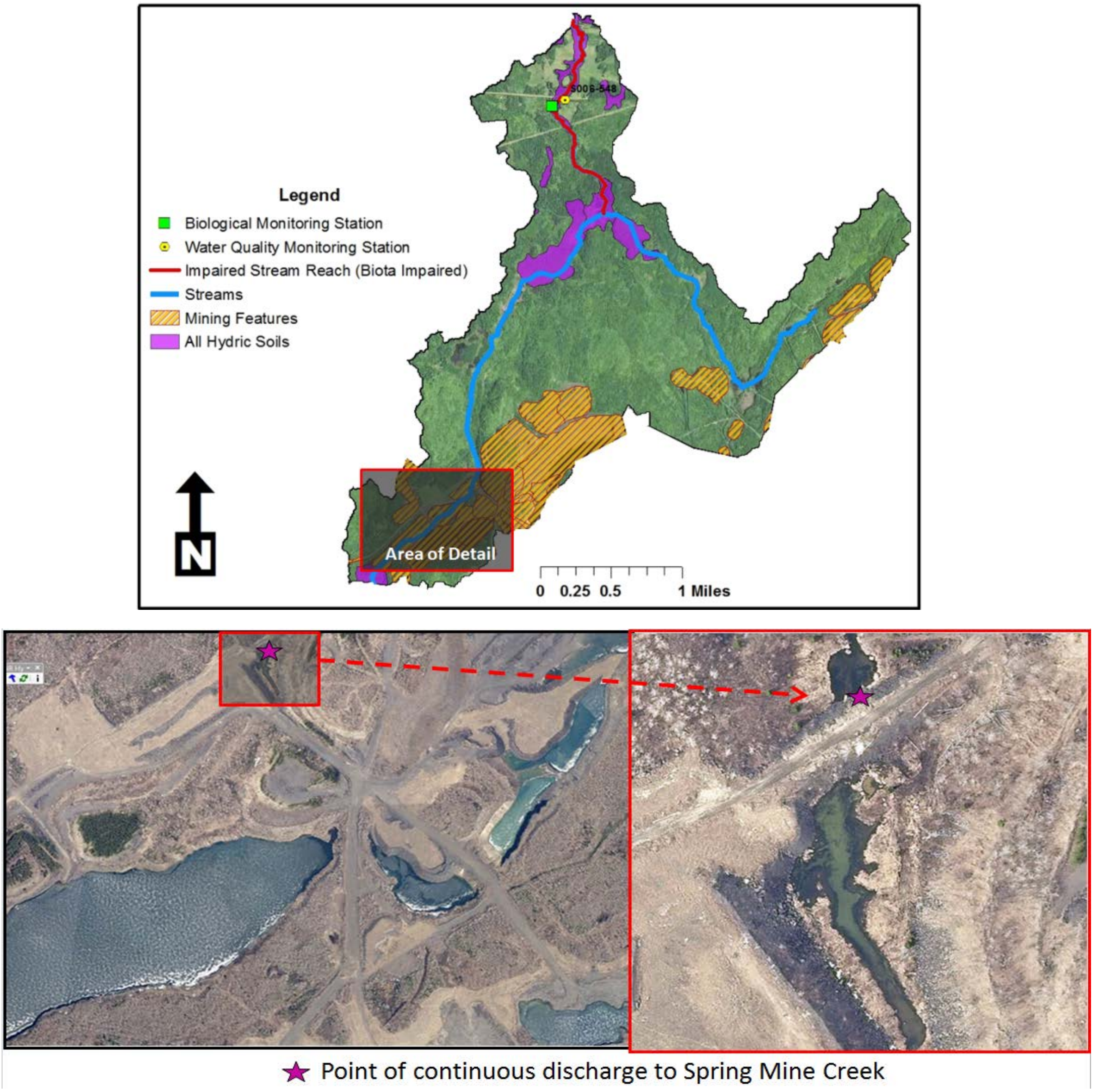


**Figure 241:** The headwaters of Spring Mine Creek are formed by a continuous discharge from a series of abandoned mine pits.

Chloride and hardness data are also important in the evaluation of sulfate as a stressor to aquatic life. Some research has shown that as chloride concentration and hardness are key variables in understanding the potential toxicity of sulfate. As a result, several states have incorporated chloride and hardness based criteria into their sulfate water quality standards (see Section 3.1.6). Chloride and hardness data were collected at the headwaters of Spring Mine Creek on a once per month basis over the years 2007-2014. Chloride concentrations were very low throughout the year, averaging less than 5 mg/L. Hardness values for the water leaving the pit at the headwaters were consistently in the range of 250 – 390 mg/L, with one severe outlier of 1330 mg/L recorded in January of 2012. Based on general guidelines for classifying water hardness, Spring Mine Creek can be classified as having very hard water (>180 mg/L), yet chloride concentrations are quite low compared to most streams receiving mine pit dewatering flow. These chloride and hardness results will be discussed later as they relate to the potential for sulfate acting as a stressor in Spring Mine Creek.

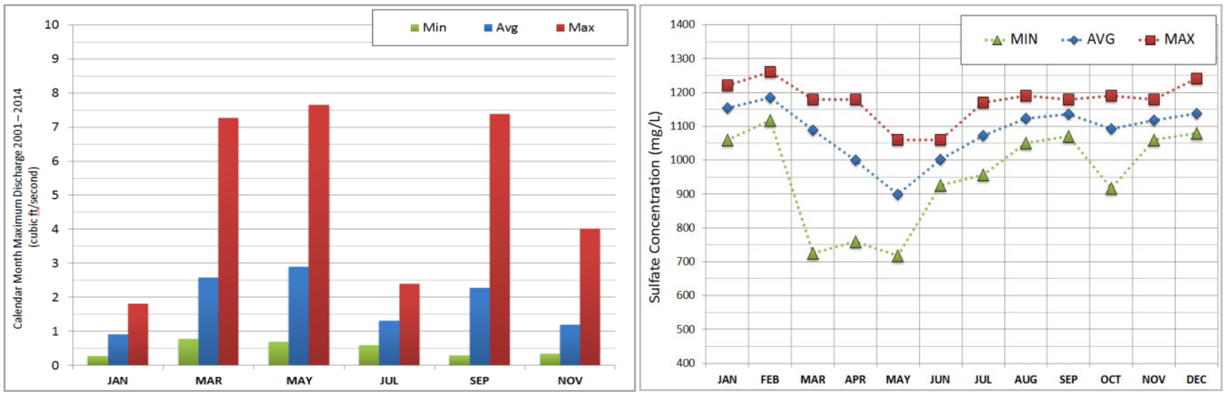


**Figure 242:** Monitoring data for streamflow and sulfate associated with permitted discharge MN0042536-SD-33 to Spring Mine Creek

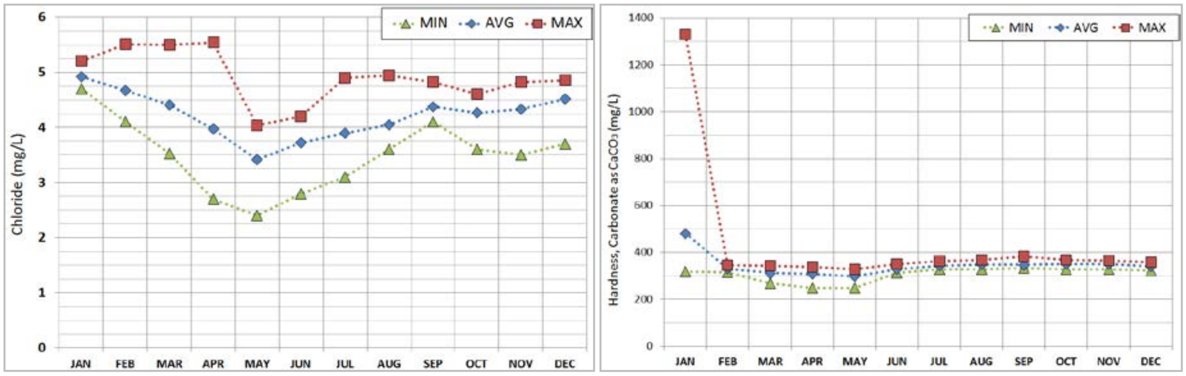


**Figure 243:** Monitoring data for chloride and total hardness associated with permitted discharge MN0042536-SD-33 to Spring Mine Creek

## Water Quality Standards for Sulfate

Minnesota does not currently enforce a sulfate standard for protection of fish and aquatic macroinvertebrates. Several U.S. and Canadian provinces have developed sulfate standards that will be used to evaluate sulfate as a stressor to aquatic life in the SLRW. For more information on these standards, see Section 3.1.6.

## Spring Mine Creek Sulfate Discussion

Sulfate concentrations recorded in the lower reaches of Spring Mine Creek exceed some water quality criteria that are currently being implemented or drafted in other states and Canadian provinces. Below is a brief comparison of Spring Mine Creek sulfate data with some of the sulfate standards currently being implemented in other states and provinces.

### British Columbia, Canada

In a paper by Elphick et al (2010), various sulfate standards are proposed for British Columbia waters based on species sensitivity data (SSD) and a safety factor approach (SFA). Both of these standards are dependent on water hardness, as harder water has been shown to reduce the toxicity effects of sulfate on aquatic life. The sulfate standard proposed in British Columbia for very hard water (>160 mg/L) is 725 mg/L based on the SSD approach, and 675 mg/L based on the SFA approach. Sulfate concentrations exceeded these criteria in 1 of 8 sampling events at the biological monitoring station 09LS101 (751 mg/L; February 3, 2014). These results provide evidence that Spring Mine Creek contains potentially harmful levels of sulfate during winter and summer low flow conditions.

### California

The state of California evaluated sulfate as a stressor to aquatic life in a 2013 study (Buchwalter 2013). This report did not result in the development of sulfate standard for CA, but served as more of a review of existing data and summary of other work involving sulfate. Although many uncertainties involving sulfate toxicity were discussed in this report, the author concluded that there is enough toxicity data by the EPA standards to support an acute toxicity criterion of 234 mg/L $SO_4$ and a chronic criterion of 124 mg/L $SO_4$. These values were not adjusted based on chloride and hardness values like other WQ standards for sulfate, and the author mentions uncertainties in the values stated above based on this detail.

Sulfate levels in Spring Mine Creek exceed the 124 mg/L and 234 mg/L standards mentioned in this report with regularity. The only result falling below the 124 mg/L chronic toxicity value is from a snowmelt sample collected in April of 2011 when concentrations were diluted due to overland runoff.

### Ohio

Several short reports exploring sulfate effects on aquatic life in the state of Ohio were released by the Center for Applied Bioassessment and Biocriteria (Rankin 2003, 2004). These studies linked biological monitoring data with sulfate sampling results across the state of Ohio with the goal of identifying critical thresholds for protecting sensitive forms aquatic life. Although no water quality standards were developed through this work, several conclusions can be drawn from these reports:

1. Many of the most sensitive taxa were not present in streams where sulfate concentrations exceeded 200 mg/L.

2. There is good evidence from Ohio streams that the presence of higher chloride concentrations ameliorates the effects of sulfate

3. Streams with sulfate concentrations above 400 mg/L generally exhibited poor biological integrity scores

4. EPT macroinvertebrate taxa were limited to 10 or less at sites where sulfate concentrations exceeded 500 mg/L

Illinois, Indiana, Pennsylvania, and Iowa

The states of Illinois, Indiana, Pennsylvania, and Iowa have been working towards an aquatic life standard for sulfate and other dissolved solids. Studies by Soucek and Kennedy (2004), Pennsylvania Department of Environmental Protection, and Iowa DNR (Iowa DNR 2009) were compiled to develop the sulfate standard. The specifics of this sulfate standard are provided in Table 9. Unlike some of the sulfate criteria listed above, chloride and water hardness were taken into account in the development of a sulfate standard for these states. Based on the hardness and chloride data available for Spring Mine Creek, the applicable water quality standard for this stream would be 500 mg/L if similar guidelines were used for a sulfate standard in Minnesota.

Figure 244 shows sulfate results for Spring Mine Creek plotted with water hardness values. Although the number of data points is somewhat limited, there is very strong positive correlation between hardness and sulfate in this stream. Of the eight sulfate samples collected from Spring Mine Creek, two exceed the sulfate WQ criteria used by the states of Iowa, Illinois, and Pennsylvania. Given the continuous discharge from headwaters mine pits, and the high sulfate content of the water leaving the pits (>1000 mg/L), additional sampling during low flow periods (November through February) would likely produce more results in exceedance of 500 mg/L sulfate.

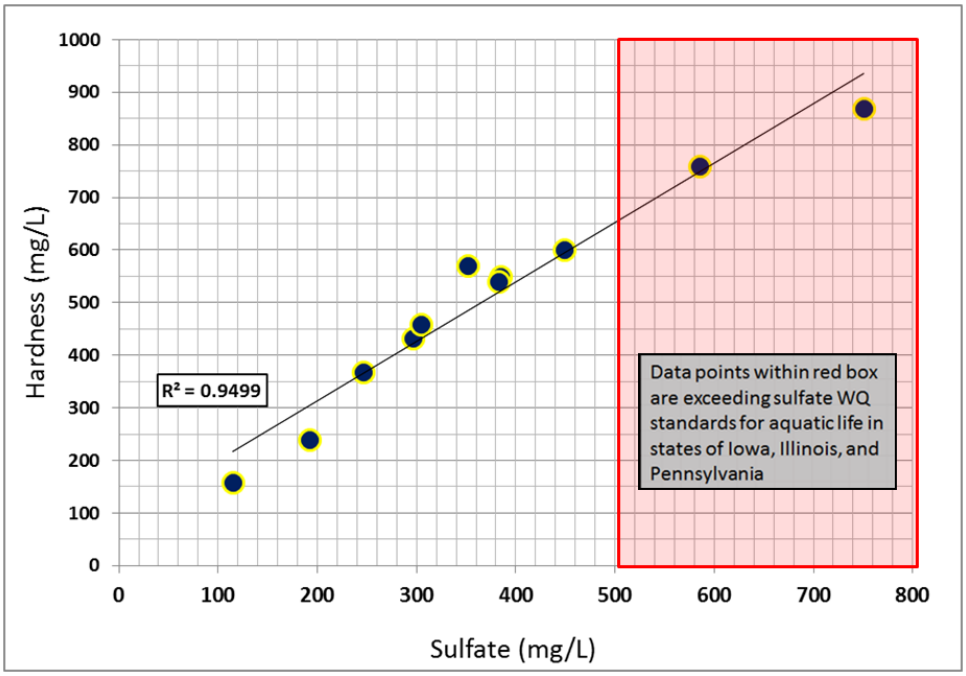


**Figure 244:** Paired sulfate and hardness data for Spring Mine Creek compared to WQ standard enforced in several US States

### Biological Response to Sulfate

The impact of elevated sulfate levels on aquatic life is a subject area that has been receiving more attention in Minnesota, as well as other regions where mining land-uses are common. Sulfate toxicity is a complex issue and a number of factors may interact to determine the responses of various organisms to sulfate-dominated waters. A discussion of available biological response data to elevated sulfate levels is presented in Section 3.1.6 of this report. Based on that summary, the biological metrics listed in Table 64 below will be used to evaluate sulfate as a stressor in Spring Mine Creek. Additional consideration for sulfate as a stressor will be presented in the specific conductivity discussion for this stream, which can be found in Section 5.14.2.

Elevated sulfate concentrations are not considered a strong candidate cause for the fish impairment in Spring Mine Creek. The available toxicity data for fish indicates that they are generally quite tolerant of sulfate, therefore it is not considered to be contributing to the fish impairment.

**Table 64:** Biological metrics selected to evaluate sulfate toxicity as a stressor to aquatic life

| Metric | Description | Relevance |
|---|---|---|
| **EPTCh** | Taxa richness of Ephemeroptera, Plecoptera & Trichoptera (baetid taxa treated as one taxon) | EPT macroinvertebrate taxa were limited to 10 or less at sites where sulfate concentrations exceeded 500 mg/L (Rankin, 2003) |
| **EphemeropteraPct** | Relative abundance (%) of Ephemeroptera individuals in subsample | Sulfate and/or bicarbonate are the likely drivers of reduced macroinvertebrate diversity and abundance (particularly mayflies) in mining impacted streams in West Virginia (Buchwalter, 2013) |

## Biological Response: EPT Taxa Richness (EPT Ch)

Rankin (2003) observed that EPT macroinvertebrate taxa were limited to 10 or less at sites where sulfate concentrations exceeded 500 mg/L. For the purposes of this analysis, the metric EPTCh was selected, which groups all baetid mayfly taxa together as one taxon. In the case of Spring Mine Creek, this detail is unimportant, as the results from 09LS101 are the same regardless of which EPT metric is used. Figure 245 compares EPT richness in Spring Mine Creek to high quality biological monitoring stations from around the SLRW. Only three EPT taxa were observed in Spring Mine Creek, which is fewer than any of the sites scoring above the IBI threshold in the same MIBI class (class 4). Individuals representing EPT taxa at 09LS101 made up less than only 2.4% of the overall subsample; *Eurylophella* (0.9%) *Glyphopsyche* (northern caddisflies - 1.2%), *Leptoceridae* (long-horn caddisflies) (0.3%).

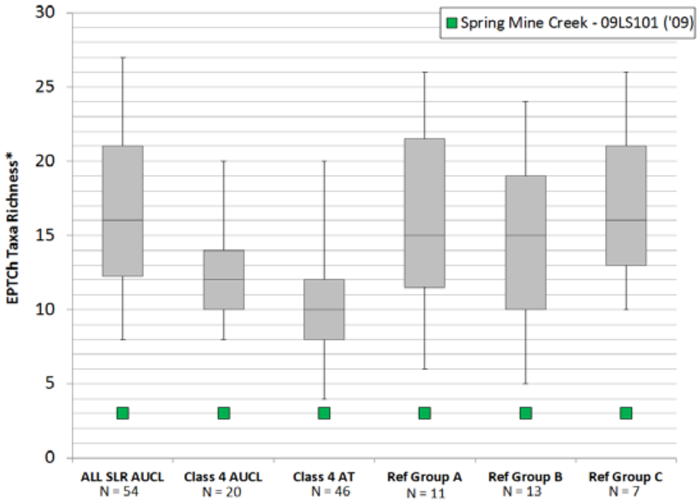


**Figure 245:** EPT taxa richness at Spring Mine Creek 09LS101 compared to results from high quality stations of the same IBI class. * See Section 4 for explanation of TIVs **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above FIBI Threshold

### Ephemeroptera Taxa Richness

*Eurylophella* was the only Ephemeropteran (mayfly) taxa represented in the sample collected at 09LS101. This mayfly genus tends to be somewhat sensitive to disturbance, so its presence at this monitoring station suggests that Spring Mine Creek does offer some suitable conditions for supporting small populations of sensitive taxa. However, *Eurylophella* were extremely scarce in the sample, representing less than 1% of the organisms counted.

With only one Ephemeroptera taxa represented in the sample, Spring Mine Creek ranks below all of the reference sites in terms of richness for that macroinvertebrate order (Figure 238). Median Ephemeroptera richness at sites with good to exceptional MIBI scores ranged from 4-6 taxa. The lack of Ephemeroptera taxa in Spring Mine Creek provides supporting evidence for elevated conductivity as a stressor.

### Summary: Is sulfate toxicity a stressor in Spring Mine Creek?

Sulfate concentrations are elevated in Spring Mine Creek, particularly during late fall and winter low flow periods. Concentrations as high as 751 mg/L have been observed during winter sampling efforts. Although Minnesota does not currently have a water quality standard for sulfate that is protective of fish and macroinvertebrates, the results from this stream show violations of standards used in other U.S. states, and exceed many of the benchmark values cited in scientific literature.

Biological data from Spring Mine Creek show some of the same symptoms seen in other streams with high sulfate concentrations (e.g. lack of EPT richness). However, it is difficult to eliminate the effects of other confounding stressors that can cause similar symptoms of impairment.

We recommend that sulfate toxicity remain a potential cause of impairment in Spring Mine Creek. In order to improve confidence in SID analyses regarding sulfate toxicity, the MPCA and other partners should place a priority on additional data collection that can lead to the development of a standard that is protective of fish and macroinvertebrate assemblages.

## 5.15.4 Spring Mine Creek: Summary of Stressors to Aquatic Life

**Table 65:** Summary of SID results for Spring Mine Creek

| Candidate Cause | Result |
|---|---|
| **Low Dissolved Oxygen** | ○ |
| **High Dissolved Oxygen Flux** | ● |
| **Sulfate Toxicity** | ○ |
| **Elevated Specific conductivity** | ○ |
| Key: **• = confirmed stressor ○ = Potential Stressor X = eliminated candidate cause** | |

## 5.16 Ely Creek

Ely Creek originates in heavily-developed Ely Lake just to the southeast of Eveleth and flows for just over six miles before emptying into the St. Louis River. The water elevation of Ely Lake is regulated by a low-head dam at its outlet to Ely Creek. Thus, flow conditions in Ely Creek are closely tied to lake conditions, and during dry periods, inputs to the creek can be very minimal or non-existent. After leaving Ely Lake, the stream enters a wide lacustrine valley, where it flows as a sinuous Rosgen E-type channel for about 1.5 miles. The stream cuts down into the lacustrine sediments just upstream of the Eveleth airport, creating an alluvial valley for itself. The stream through this reach is incised and types out as a Rosgen G-type channel based on field measurements. The impaired AUID of Ely Creek includes these first two reaches and has an average slope of 8 feet/ mile (0.15%). Just below Bodas Rd Ely Creek connects with the outlet of Pleasant Lake. This unimpaired reach is 1.5 miles long and types out as a Rosgen C-type channel in an alluvial valley.

The FIBI impairment listing for Ely Creek is based on data collected at station 09LS084, which is located in the lower reaches of Ely Creek approximately 1.5 miles upstream of its confluence with the St. Louis River near Makinen, Minnesota. Station information and sampling results are summarized in Table 66. The fish community at this station is characterized by low fish abundance and a lack of sensitive taxa. Only 20 individual fish representing seven different species were sampled from this station. The numbers of fish per taxa were distributed fairly evenly. Species observed at this station, listed in order of most to least abundant include; Common Shiner (n=6), Burbot (n=4), Creek Chub (n=3), White Sucker (n=2), Bluegill (n=2), Central Mudminnow (n=2), and Yellow Perch (n=1).

The low overall FIBI scores for Ely Creek station 09LS084 result from poor metric scores related to low fish abundance, a lack of minnow and headwater minnow species (e.g. shiner sp., dace sp., sculpin sp.), and a low number of sensitive fish taxa.

**Table 66:** Biological monitoring sites and FIBI results from Ely Creek

| **Fish Assessments** | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Station** | **Drainage Area (mi2)** | **Gradient (%)** | **Stream Order (Strahler)** | **FIBI Class** | **FIBI Result *(visit year)*** | **FIBI Result *(visit year)*** | **Standard** | **IBI Lower Confidence Limit** | **IBI Upper Confidence Limit** |
| 09LS084 | 15.45 | 0.12 | 1 | 5 | **34** *(2009)* | - | 42 | 26 | 58 |

Water quality and physical habitat data were used to develop a list of candidate causes for impairment in Ely Creek. The following candidate causes were identified for further analysis in this section;

1. **Poor Habitat Conditions**
2. **Low DO**
3. **Altered Hydrology**

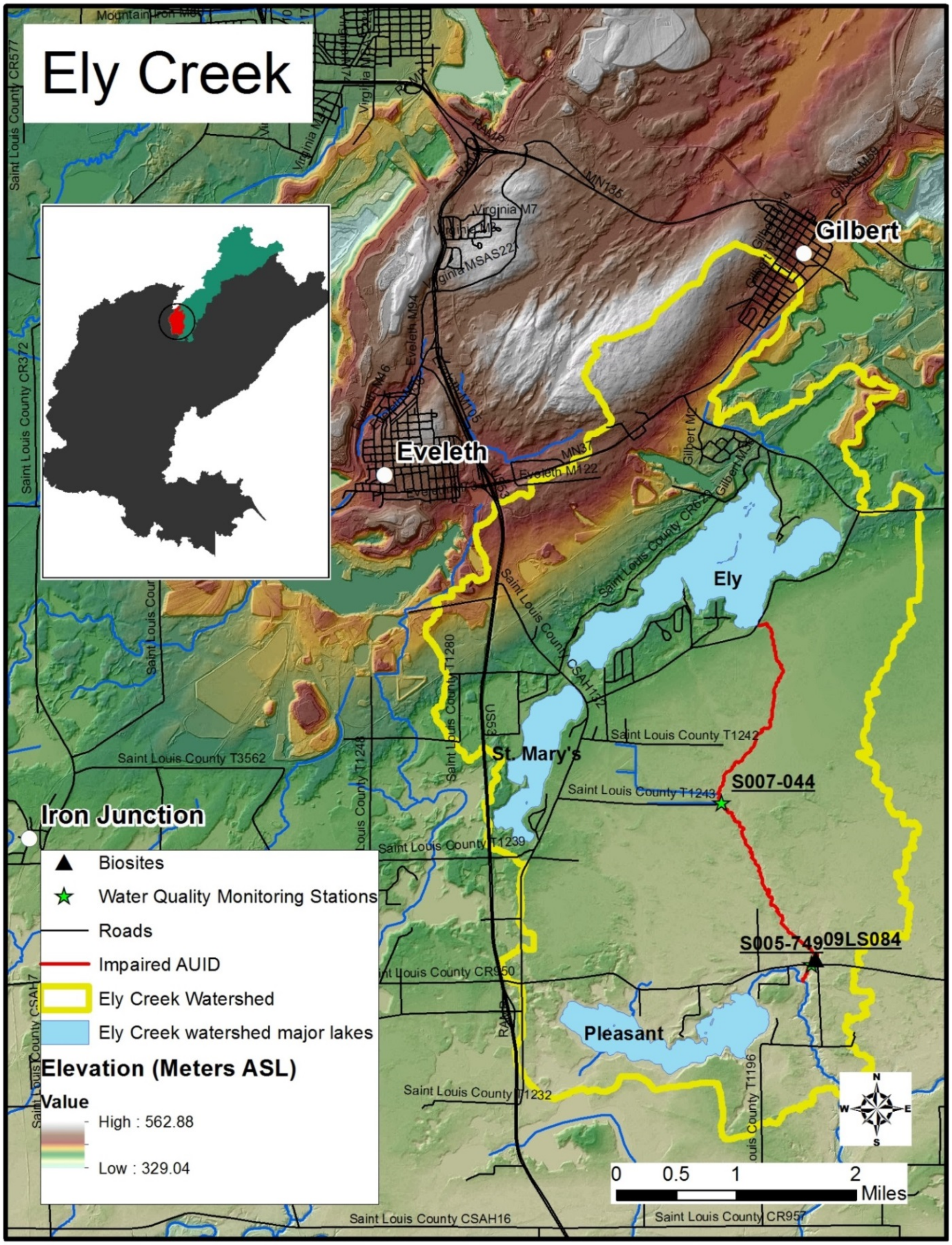


**Figure 239:** Ely Creek Watershed and monitoring locations

### 5.16.1 Poor Physical Habitat Conditions

Overall, physical habitat conditions in Ely Creek are considered “fair” based on MSHA results. Biological monitoring station 09LS084 was the only site evaluated using the MSHA, and it the “fair” rating summarizes the score of 57.5 (out of 100). The MSHA metrics for substrate and channel morphology received particularly poor scores. As shown in Figure 240, the overall scores for substrate at 09LS084 were well below the 25th percentile score observed at non-impaired class 6 fish stations in the SLRW. The poor scores in the substrate metric were due mainly to the dominance of clay and silt and the complete absence of coarser substrates. Results for the other three major MSHA metric categories (riparian, fish cover, channel morphology) were more compared more favorably to results from the non-impaired class 6 FIBI stations (Figure 240).

The lack of riffle and glide features in Ely Creek was another habitat limitation that became evident after reviewing MSHA results. The biological monitoring reach consisted of 25% “pool” features and 75% “run” features, with no riffle or glide sections. The shallow, fast velocity habitats created by riffle and glide features are often found to be productive areas for benthic macroinvertebrates, and many stream fishes make use of these areas for feeding, refuge, and reproduction. On average, streams dominated by one or two habitat types are prone to supporting lower biodiversity and fewer habitat specialists, particularly if “run” features are dominant.

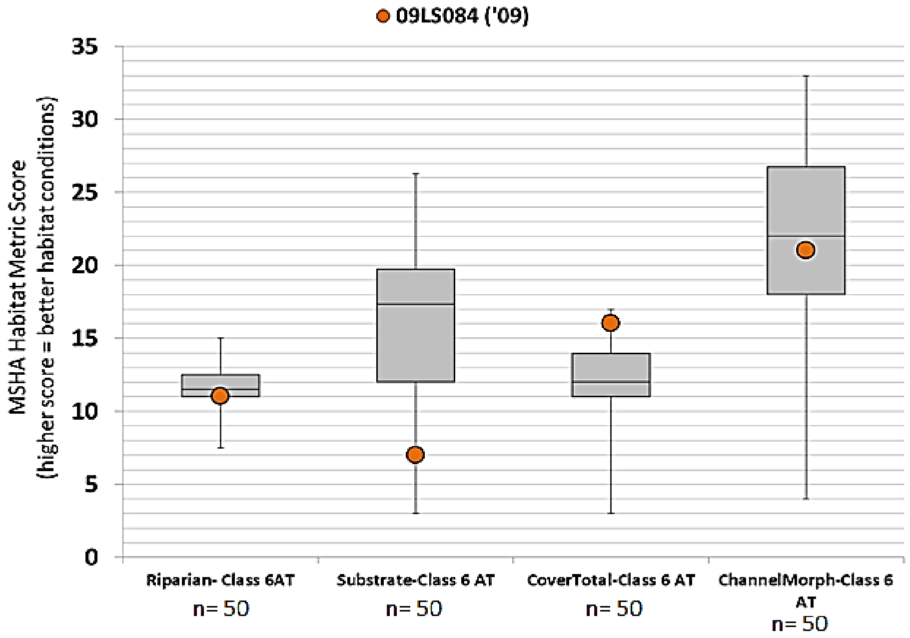
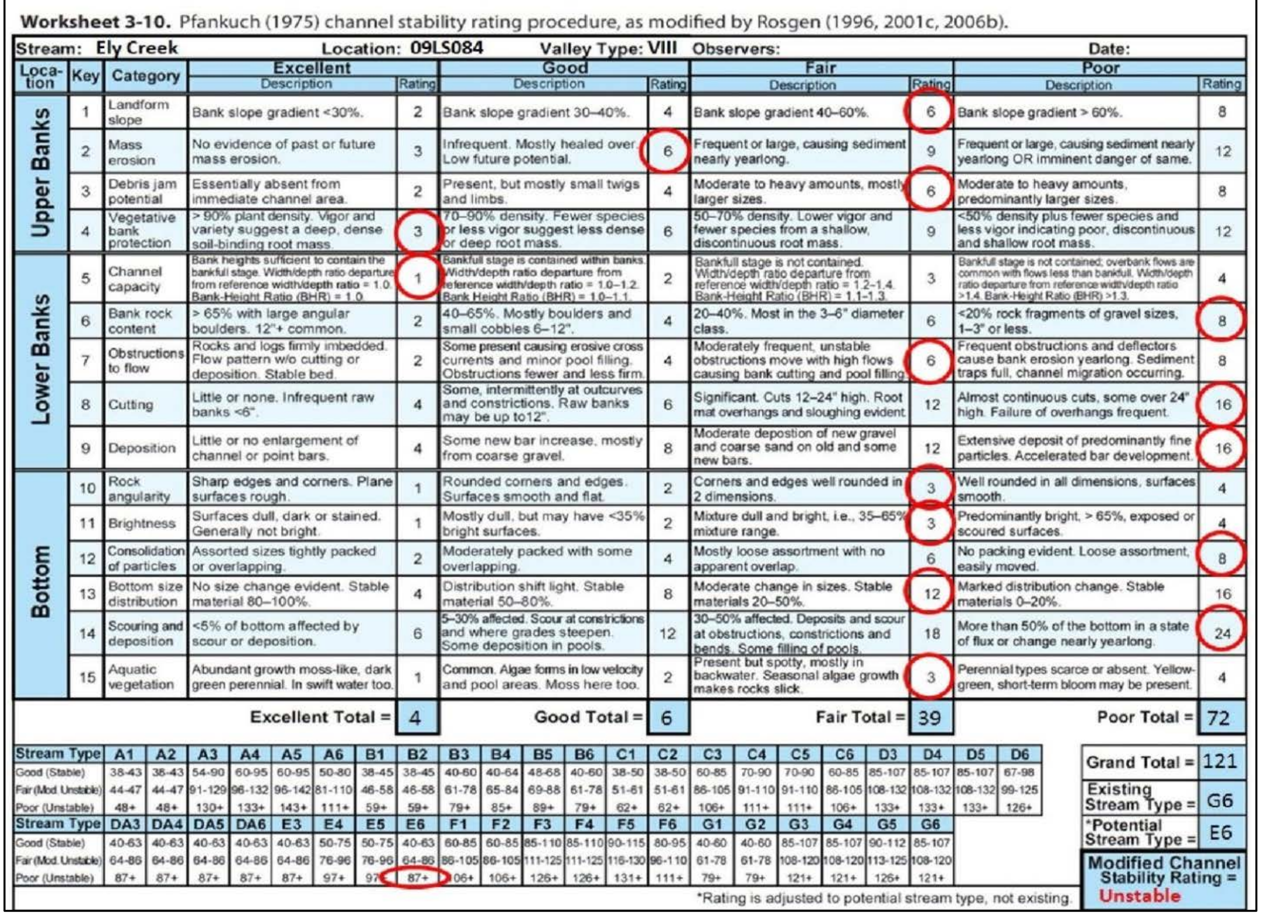


**Figure 240:** Overall scores for the four major habitat categories evaluated in the MSHA, comparing Ely Creek results to results from non-impaired class 6 FIBI stations in the SLRW. * See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

The PSI was also used to evaluate channel stability and habitat quality in Ely Creek. PSI results for station 09LS084 are typical of a severely incised channel. Characteristics such as bank cutting, fine particle deposition, pool filling, and debris jams gave Ely Creek a total PSI score of 121, which corresponds to a rating of “unstable” for the potential stream type (Figure 241). For more background on the unstable geomorphology of Ely Creek at 09LS084, see Appendix X (Ely Creek geomorphological survey). Some of the consequences of channel incision, like undercut banks and large woody debris in the channel, may actually benefit habitat conditions. However, it is likely that the other consequences of incision, such as excess fine particle deposition and pool filling, are causing a net decrease in the amount of quality habitat available to the fish community in Ely Creek.



**Worksheet 3-10.** Pfankuch (1975) channel stability rating procedure, as modified by Rosgen (1996, 2001c, 2006b).

**Stream:** Ely Creek **Location:** 09LS084 **Valley Type:** VIII **Observers:** **Date:**

| Location | Key | Category | Excellent |  | Good |  | Fair |  | Poor |  |
|---|---|---|---|---|---|---|---|---|---|---|
|  |  |  | Description | Rating | Description | Rating | Description | Rating | Description | Rating |
| Upper Banks | 1 | Landform slope | Bank slope gradient <30%. | 2 | Bank slope gradient 30–40%. | 4 | Bank slope gradient 40–60%. | 6 | Bank slope gradient > 60%. | 8 |
|  | 2 | Mass erosion | No evidence of past or future mass erosion. | 3 | Infrequent. Mostly healed over. Low future potential. | 6 | Frequent or large, causing sediment nearly yearlong. | 9 | Frequent or large, causing sediment nearly yearlong OR imminent danger of same. | 12 |
|  | 3 | Debris jam potential | Essentially absent from immediate channel area. | 2 | Present, but mostly small twigs and limbs. | 4 | Moderate to heavy amounts, mostly larger sizes. | 6 | Moderate to heavy amounts, predominantly larger sizes. | 8 |
|  | 4 | Vegetative bank protection | > 90% plant density. Vigor and variety suggest a deep, dense soil-binding root mass. | 3 | 70–90% density. Fewer species or less vigor suggest less dense or deep root mass. | 6 | 50–70% density. Lower vigor and fewer species from a shallow, discontinuous root mass. | 9 | <50% density plus fewer species and less vigor indicating poor, discontinuous and shallow root mass. | 12 |
| Lower Banks | 5 | Channel capacity | Bank heights sufficient to contain the bankfull stage. Width/depth ratio departure from reference width/depth ratio = 1.0. Bank-Height Ratio (BHR) = 1.0. | 1 | Bankfull stage is contained within banks. Width/depth ratio departure from reference width/depth ratio = 1.0–1.2. Bank Height Ratio (BHR) = 1.0–1.1. | 2 | Bankfull stage is not contained. Width/depth ratio departure from reference width/depth ratio = 1.2–1.4. Bank-Height Ratio (BHR) = 1.1–1.3. | 3 | Bankfull stage is not contained; overbank flows are common with flows less than bankfull. Width/depth ratio departure from reference width/depth ratio >1.4. Bank-Height Ratio (BHR) >1.3. | 4 |
|  | 6 | Bank rock content | > 65% with large angular boulders. 12"+ common. | 2 | 40–65%. Mostly boulders and small cobbles 6–12". | 4 | 20–40%. Most in the 3–6" diameter class. | 6 | <20% rock fragments of gravel sizes, 1–3" or less. | 8 |
|  | 7 | Obstructions to flow | Rocks and logs firmly imbedded. Flow pattern w/o cutting or deposition. Stable bed. | 2 | Some present causing erosive cross currents and minor pool filling. Obstructions fewer and less firm. | 4 | Moderately frequent, unstable obstructions move with high flows causing bank cutting and pool filling. | 6 | Frequent obstructions and deflectors cause bank erosion yearlong. Sediment traps full, channel migration occurring. | 8 |
|  | 8 | Cutting | Little or none. Infrequent raw banks <6". | 4 | Some, intermittently at outcurves and constrictions. Raw banks may be up to12". | 6 | Significant. Cuts 12–24" high. Root mat overhangs and sloughing evident. | 12 | Almost continuous cuts, some over 24" high. Failure of overhangs frequent. | 16 |
|  | 9 | Deposition | Little or no enlargement of channel or point bars. | 4 | Some new bar increase, mostly from coarse gravel. | 8 | Moderate depostion of new gravel and coarse sand on old and some new bars. | 12 | Extensive deposit of predominantly fine particles. Accelerated bar development. | 16 |
| Bottom | 10 | Rock angularity | Sharp edges and corners. Plane surfaces rough. | 1 | Rounded corners and edges. Surfaces smooth and flat. | 2 | Corners and edges well rounded in 2 dimensions. | 3 | Well rounded in all dimensions, surfaces smooth. | 4 |
|  | 11 | Brightness | Surfaces dull, dark or stained. Generally not bright. | 1 | Mostly dull, but may have <35% bright surfaces. | 2 | Mixture dull and bright, i.e., 35–65% mixture range. | 3 | Predominantly bright, > 65%, exposed or scoured surfaces. | 4 |
|  | 12 | Consolidation of particles | Assorted sizes tightly packed or overlapping. | 2 | Moderately packed with some overlapping. | 4 | Mostly loose assortment with no apparent overlap. | 6 | No packing evident. Loose assortment, easily moved. | 8 |
|  | 13 | Bottom size distribution | No size change evident. Stable material 80–100%. | 4 | Distribution shift light. Stable material 50–80%. | 8 | Moderate change in sizes. Stable materials 20–50%. | 12 | Marked distribution change. Stable materials 0–20%. | 16 |
|  | 14 | Scouring and deposition | <5% of bottom affected by scour or deposition. | 6 | 5–30% affected. Scour at constrictions and where grades steepen. Some deposition in pools. | 12 | 30–50% affected. Deposits and scour at obstructions, constrictions and bends. Some filling of pools. | 18 | More than 50% of the bottom in a state of flux or change nearly yearlong. | 24 |
|  | 15 | Aquatic vegetation | Abundant growth moss-like, dark green perennial. In swift water too. | 1 | Common. Algae forms in low velocity and pool areas. Moss here too. | 2 | Present but spotty, mostly in backwater. Seasonal algae growth makes rocks slick. | 3 | Perennial types scarce or absent. Yellow-green, short-term bloom may be present. | 4 |
|  |  |  | Excellent Total = | 4 | Good Total = | 6 | Fair Total = | 39 | Poor Total = | 72 |

| Stream Type | A1 | A2 | A3 | A4 | A5 | A6 | B1 | B2 | B3 | B4 | B5 | B6 | C1 | C2 | C3 | C4 | C5 | C6 | D3 | D4 | D5 | D6 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Good (Stable) | 38-43 | 38-43 | 54-90 | 60-95 | 60-95 | 50-80 | 38-45 | 38-45 | 40-60 | 40-64 | 48-68 | 40-60 | 38-50 | 38-50 | 60-85 | 70-90 | 70-90 | 60-85 | 85-107 | 85-107 | 85-107 | 67-98 |
| Fair (Mod. Unstable) | 44-47 | 44-47 | 91-129 | 96-132 | 96-142 | 81-110 | 46-58 | 46-58 | 61-78 | 65-84 | 69-88 | 61-78 | 51-61 | 51-61 | 86-105 | 91-110 | 91-110 | 86-105 | 108-132 | 108-132 | 108-132 | 99-125 |
| Poor (Unstable) | 48+ | 48+ | 130+ | 133+ | 143+ | 111+ | 59+ | 59+ | 79+ | 85+ | 89+ | 79+ | 62+ | 62+ | 106+ | 111+ | 111+ | 106+ | 133+ | 133+ | 133+ | 126+ |

| Stream Type | DA3 | DA4 | DA5 | DA6 | E3 | E4 | E5 | E6 | F1 | F2 | F3 | F4 | F5 | F6 | G1 | G2 | G3 | G4 | G5 | G6 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Good (Stable) | 40-63 | 40-63 | 40-63 | 40-63 | 40-63 | 50-75 | 50-75 | 40-63 | 60-85 | 60-85 | 85-110 | 85-110 | 90-115 | 80-95 | 40-60 | 40-60 | 85-107 | 85-107 | 90-112 | 85-107 |
| Fair (Mod. Unstable) | 64-86 | 64-86 | 64-86 | 64-86 | 64-86 | 76-96 | 76-96 | 64-86 | 86-105 | 86-105 | 111-125 | 111-125 | 116-130 | 96-110 | 61-78 | 61-78 | 108-120 | 108-120 | 113-125 | 108-120 |
| Poor (Unstable) | 87+ | 87+ | 87+ | 87+ | 87+ | 97+ | 97 | 87+ | 06+ | 106+ | 126+ | 126+ | 131+ | 111+ | 79+ | 79+ | 121+ | 121+ | 126+ | 121+ |

| | |
|---|---|
| Grand Total = | 121 |
| Existing Stream Type = | G6 |
| *Potential Stream Type = | E6 |

**Modified Channel Stability Rating =** Unstable

*Rating is adjusted to potential stream type, not existing.

**Figure 241:** Pfankuch Stability Index (PSI) score and rating for Ely Creek biological monitoring station 09LS084

Rosgen Level II Survey Data

Initial observations of station 09LS084 indicated that channel instability and resulting habitat loss due to sedimentation could be a cause of poor FIBI scores. On October 11, 2013, a geomorphological survey was completed (Rosgen Levell II methods) at the 09LS084 biosite in order to quantify the degree of instability and inform any potential future implementation activities that may improve conditions within the reach. Included in this survey were the longitudinal profile, riffle cross-section, reach and riffle pebble counts, and PSI rating.

Survey data confirmed that this reach is mostly incised and the bank height is well above the bankfull elevation (see cross-section in Figure 243), indicating that the channel had undergone a downcutting event sometime in the past. The channel has started to create depositional features a the bankfull elevation in some locations (Figure 244). Due to the flat water surface, distinct riffles and pools were not present, and the riffle cross section was chosen by locating a shallow stretch of channel.

A reach pebble count was also completed at this site and results are summarized in Figure 245. The D50 (median particle size) is 0.09mm, which is in the very fine sand category. 75% of the particles in the stream were classified as silt/clay, very fine sand, or fine sand. There is a complete lack of gravel substrates in this reach. It is posited that the dominance of fines in the substrate is at least partially related to backwater conditions created by the beaver dam downstream of the reach, which induce settling of fine particles.

The poor condition of Ely Creek suggests that it has been destabilized, and is undergoing a channel evolution process to return to a stable state. One channel evolution scenario that may be occurring Ely Creek is presented below (Figure 242). In this scenario, the channel experiences a downcutting event, caused by a change in the hydrology of the watershed, an increase in the water slope, or a decrease in the amount or size of the sediment made available to the stream. After downcutting to a “G”, the channel is severely entrenched and widens to an “F” channel as it tries to recreate a floodplain. The “G” -> “F” stage is erosional and puts a massive amount of sediment into the channel – potentially tons per lineal foot of channel. After widening, the W/D ratio becomes high enough that the river no longer has the competence to transport sediment and deposition starts to occur. This process builds the new floodplain at a lower elevation and creates a wide “C” channel. Over time, riparian vegetation will colonize the banks and eventually cause the channel to narrow to a stable “E” channel.

If this evolution process is indeed happening on Ely Creek, it is probable that the channel is transitioning between the second and third stages (G -> F) (Figure 242). The riparian trees leaning and falling into the channel indicate lateral instability (widening), reinforcing this hypothesis. It is also likely, however, that this channel evolution is occuring at an arrested rate. The presence of grade control downstream in the form of road crossings and beaver dams may have prevented further downcutting and may explain why the channel is not incised enough to classify as a true “G” channel. The relatively low stream power and the cohesiveness of the loamy silt material in the channel banks are also likely contributing to an arrested rate of channel evolution. Without human intervention it may take decades to get back to a stable condition that provides good habitat for aquatic biota.

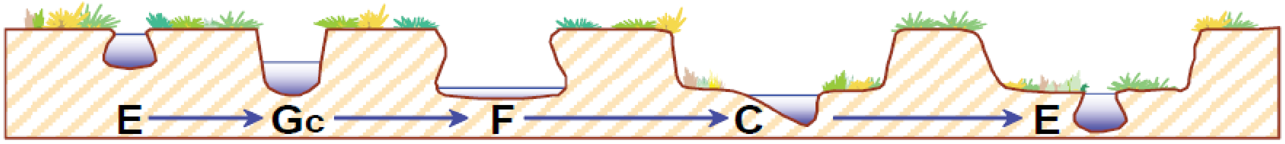


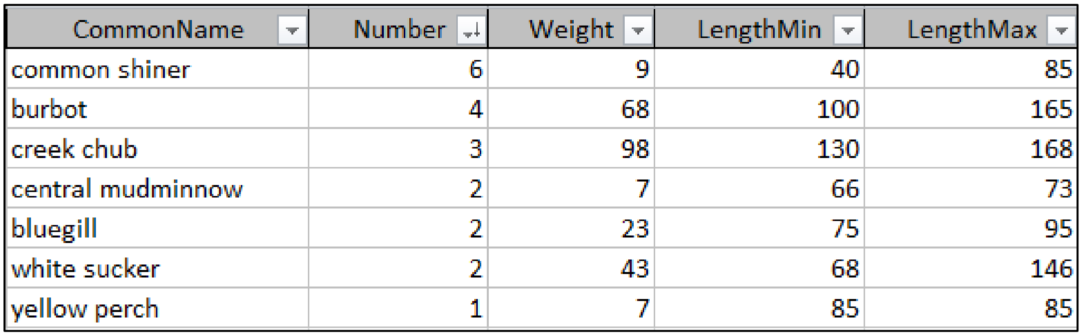
**Figure 242:** Ely Creek channel evolution scenario (from the River Stability Field Guide, Rosgen 2008)

### Biological effects of degraded habitat

#### Fish Response to Habitat

The fish community observed at Ely Creek station 09LS084 is characterized by low taxa richness and low fish abundance. Fish density (fish/meter) results were well below the values observed in the vast majority of non-impaired stations of the same FIBI class. Most of the fish species present in Ely Creek are considered moderately to highly tolerant of poor habitat conditions. The presence of Bluegill, Yellow Perch, and Burbot in the sample is a sign of the close proximity and connectivity of this reach to Pleasant Lake just a short distance downstream. Examples of fish species found in high quality class 6 FIBI stations but absent from Ely Creek include Northern Redbelly Dace, Longnose Dace, Iowa Darter, Blacknose Shiner, and Pearl Dace.

**Table 67:** Summary of the fish species found in Ely Creek, including number caught, maximum and minimum length, and batch weights.



| CommonName | Number | Weight | LengthMin | LengthMax |
|---|---|---|---|---|
| common shiner | 6 | 9 | 40 | 85 |
| burbot | 4 | 68 | 100 | 165 |
| creek chub | 3 | 98 | 130 | 168 |
| central mudminnow | 2 | 7 | 66 | 73 |
| bluegill | 2 | 23 | 75 | 95 |
| white sucker | 2 | 43 | 68 | 146 |
| yellow perch | 1 | 7 | 85 | 85 |

In addition to low taxa richness and abundance, another symptom of impairment observed in the fish community of Ely Creek was the lack of species with specific habitat and trophic requirements. No benthic insectivorous fish were observed at station 09LS084. Fish with this trophic trait prey on aquatic insect life living on stream substrates. The homogenous silt and sand substrates of Ely Creek are not prone to producing large insect populations, and these conditions are not adventageous for fish that feed on benthic macroinvertebrates. Fish species that prefer swift current and riffle habitats were lacking from this reach of Ely Creek. Station 09LS084 received a score of 0 out of 10 for the *DartSclup* (numbe of darter and sculpin species) IBI metric. The lack of species with physical, physiological, and life-history traits that revolve around riffle and swift water habitats is not suprising given the dominance of pool and run features within the biological sampling reach.

The lack of habitat and trophic specialists in Ely Creek provide evidence in support of degraded habitat conditions as a stressor. Several simple lithophilic spawning species (fish that require coarse substrates for spawning) were present at this monitoring station, but given the lack of coarse substrates at station 09LS084, it is likely that they spawn in other accessible stream reaches. In conclusion, Ely Creek seems to lack species that need specific substrate for feeding and refuge, as well as taxa that require variability in current velocities and habitat types.

**Summary: Are poor physical habitat conditions a stressor in Ely Creek?**

Degraded habitat in Ely Creek can likely be attributed to channel incision and resulting sedimentation. The "substrate" and "channel morphology" sections of the MSHA score, as well as the "unstable" Pfankuch Stability provide evidence in support of this stressor. Degraded habitat conditions are limiting the diversity and integrity of biota inhabiting Ely Creek, as evidenced by the number of non-tolerant fish/meter and the low number of various habitat-sensitive taxa present. It is our conclusion that habitat degradation is a stressor to the fish community in Ely Creek.

### 5.16.2 Low Dissolved Oxygen

Low DO was identified as a candidate cause for impairment in Ely Creek due to a series of measurements in 2012 that failed to meet the 5 mg/L water quality standard. Available DO data for Ely Creek include instantaneous (point) measurements and a series of short term (approximately 1 week) continuous monitoring profiles completed in August and September of 2011, 2012, and 2013. Results of point DO measurements collected between the years of 2009 and 2013 are shown in Figure 246 by calendar month. All measurements were collected at station S005-749, which is co-located with the only biological monitoring station on Ely Creek (09LS084). Overall, the point measurements of DO indicate favorable conditions for supporting a healthy fish community, as only 7% of the results (2 of 27) fell below the 5 mg/L water quality standard. Both of the observations of low DO were observed in early September of 2012.

Results from the continuous DO profiles are plotted in Figure 247. The profiles were all collected between the middle of August to early September during baseflow conditions. Results from 2011 and 2013 show adequate DO concentrations for aquatic life. In contrast, the 2012 data show DO concentrations dropping well below 1.0 mg/L during the early morning hours each day of the profile. During the 2012 profile, streamflow at station S005-749 was extremely low due to drought conditions that extended across much of northern Minnesota. The photo in Figure 248 was taken in September of 2012 during this period of very low flow. In addition to lower DO minimums, the level of DO flux over a 24 hour period (diurnal flux) was roughly two times higher during the 2012 profile compared to the other two years of data (Figure 247). Based on all of continuous data profiles, Ely Creek does not exhibit high level of DO flux and can be characterized as a fairly unproductive stream. Typical diurnal flux observed in our data was in the range of 1 – 2.5 mg/L.

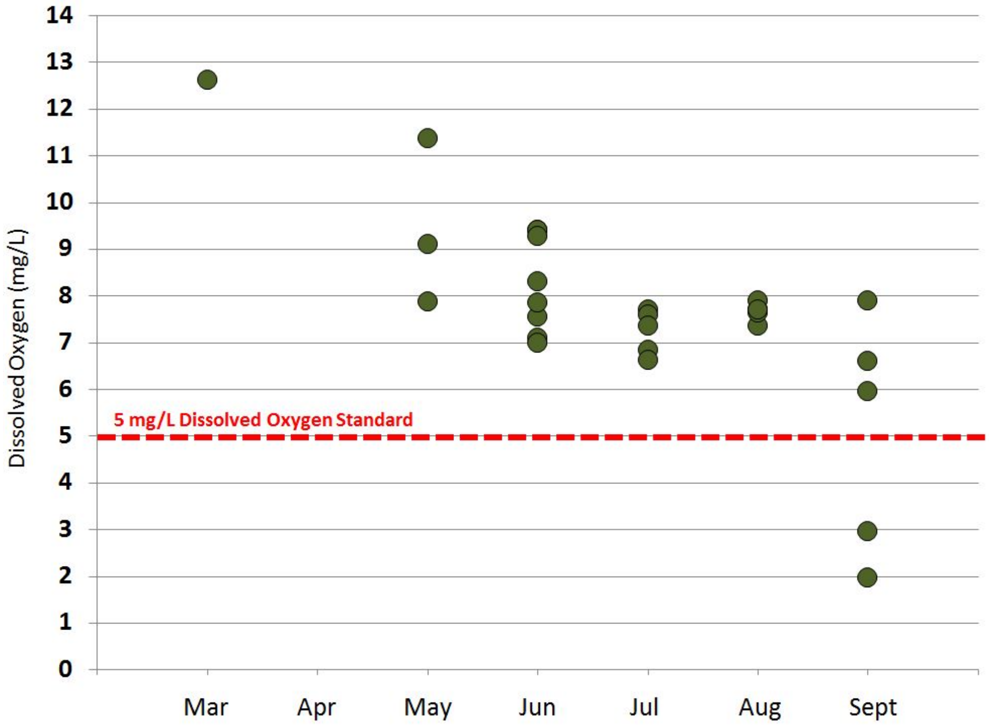


**Figure 246:** Results of point measurements of DO collected at Ely Creek station S005-749 (biological station 09LS084) between 2009 and 2013.

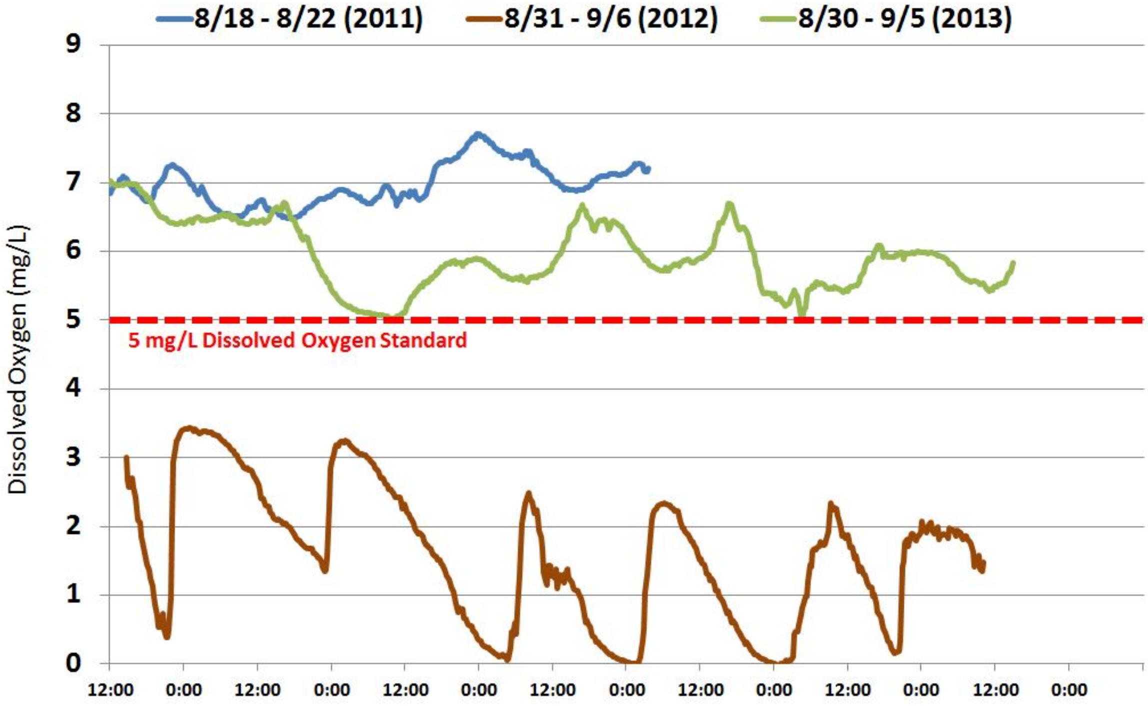


**Figure 247:** Plots of continuous DO measurements collected at station S005-749 (biological monitoring station 09LS084) in 2011, 2012, and 2013. Data are compared to the DO standard for warmwater streams (5 mg/L).

**Figure 248:** Low flow conditions at station S005-749 on September 13, 2012 (looking upstream). Continuous DO measurements revealed significantly lower DO concentration during this dry period.

## Sources and Pathways Contributing to Low Dissolved Oxygen

Over the course of numerous site visits from 2010 to 2013, low or stagnant flow conditions were regularly observed in Ely Creek during the summer and early fall months. Stream conditions were particularly low in 2012, as drought conditions spread across much of northern Minnesota. Total precipitation in the area Ely Creek Watershed between July and September 2012 was seven inches, which ranks as the 2nd lowest total observed between the years 2000-2013 (Table 68). It is likely that the dam at the outlet of Ely Lake exacerbates low flow conditions in the creek during these dry periods by retaining water that historically fed Ely Creek. Lake elevation measurements collected at the location of the dam indicate that lake levels regularly drop below the crest elevation of the dam during summer and fall months (see Section 3.2.1). When this occurs, the surface waters of Ely Lake become disconnected with the creek and downstream wetlands. On September 6, 2012, the crest gauge at the outlet of Ely Lake recorded a lake elevation below the crest of the dam. Therefore, at some point during the 2012 continuous DO profile, Ely Lake was not passing any surface water across the dam and into the creek.

In 2006, only 6.4 inches of precipitation was recorded in the vicinity of the Ely Creek Watershed over the months of July through September. In the 14 years of record spanning the period from 2000-2014, the 2006 total is the only July-September span drier than the 2012 total (Table 68). Water level data from the crest gauge at Ely Lake indicate that lake levels were below the dam crest for a duration of 8-9 weeks during the dry period in 2006. Low precipitation years, in combination with the effects of the dam at Ely Lake, are plausible sources of low streamflow and low DO levels in Ely Creek.

**Table 68:** Monthly precipitation (inches) from May to October over the period of 2000 – 2014

| **Year** | **May** | **Jun** | **Jul** | **Aug** | **Sep** | **Oct** | *Total (May - Oct)* | **Sub Total (July - Sept)** | **July - Sept Rank (Driest to Wettest)** |
|---|---|---|---|---|---|---|---|---|---|
| **2000** | 2.46 | 3.75 | 3.81 | 5.6 | 1.59 | 3.74 | *21.0* | **11.0** | **11** |
| **2001** | 5.6 | 2.51 | 2.62 | 3.77 | 1.54 | 2.97 | *19.0* | **7.9** | **4** |
| **2002** | 1.03 | 3.73 | 4 | 3.69 | 2.64 | 2.54 | *17.6* | **10.3** | **10** |
| **2003** | 2.21 | 3.48 | 6.4 | 1.9 | 5.85 | 0.97 | *20.8* | **14.2** | **13** |
| **2004** | 3.53 | 1.44 | 2.03 | 2.49 | 5.49 | 3.73 | *18.7* | **10.0** | **9** |
| **2005** | 4.38 | 5.08 | 1.66 | 3.34 | 2.66 | 2.79 | *19.9* | **7.7** | **3** |
| **2006** | 4.04 | 2.34 | 3.3 | 1.35 | 1.74 | 1.98 | *14.8* | **6.4** | **1** |
| **2007** | 2.85 | 4.14 | 1.8 | 2.35 | 7.53 | 3.58 | *22.3* | **11.7** | **12** |
| **2008** | 1.9 | 4.07 | 1.49 | 2.18 | 4.76 | 3.15 | *17.6* | **8.4** | **7** |
| **2009** | 2.22 | 3.38 | 3.53 | 4.68 | 1.3 | 4.99 | *20.1* | **9.5** | **8** |
| **2010** | 1.68 | 3.89 | 2.52 | 7.4 | 6.5 | 2.6 | *24.6* | **16.4** | **14** |
| **2011** | 2.08 | 4.86 | 2.7 | 3.81 | 1.62 | 1.51 | *16.6* | **8.1** | **5** |
| **2012** | 6.54 | 7.08 | 4.8 | **1.52** | **0.65** | 2.79 | *23.4* | **7.0** | **2** |
| **2013** | 2.15 | 8.45 | 2.51 | 2.91 | 2.97 | 3.09 | *22.1* | **8.4** | **6** |

Beaver dams have been observed downstream of the biological monitoring station, which also contribute to stagnant flow conditions. A survey completed in the fall of 2013 revealed that water surface slope within the biological monitoring reach was essentially flat at baseflow conditions due to downstream beaver impoundments (0.00005%, or 3 inches per mile). The stagnant surface water resulting from these impoundments, along with the general lack of riffle and glide features within the biological monitoring reach, minimizes or eliminates any oxygen inputs from water surface turbulence.

Nutrient enrichment and productivity are ***not*** considered a source or pathway linked to low DO in Ely Creek. Observations of TP have all been below the 0.055 mg/L river nutrient criteria applied to northern Minnesota streams and rivers (n=11, mean = 0.032 mg/L, max = 0.054 mg/L). Primary productivity in Ely Creek appears to be low based on the low diurnal DO flux and sparse amounts of algae and aquatic vegetation observed within the stream channel.

## Biological Response to Low Dissolved Oxygen

The Ely Creek FIBI impairment is based on data collected from one 2009 monitoring event at station 09LS084. A total of 20 individuals were counted, and 7 fish species were represented in the sample. Species present included Common Shiner (n=6), Burbot (n=4), Creek Chub (n=3), Central Mudminnow (n=2), Bluegill (n=2), White Sucker (n=2), and Yellow Perch (n=1). Only one sensitive fish taxa was observed in the sample (Burbot), while over half of the species present can be considered tolerant or very tolerant of poor water chemistry or degraded habitat conditions. Measures of fish abundance in Ely Creek are significantly lower than high quality streams with comparable drainage area and gradient. Fish density (normalized by reach length surveyed) at station Ely Creek 09LS084 was only 0.13 fish/meter. In comparison, median results from high quality stations of the same FIBI class ranged were in the range of 0.73 to 0.83 fish per meter.

The DO tolerance indicator values (DO TIV) for the fish community of Ely Creek is comparable to results from high quality stations of the same IBI class. These results indicate that the fish community at station 09LS084 is not dominated by taxa which are tolerant of low DO conditions. Several species that can tolerate of low DO conditions were observed in the sample (e.g. Central Mudminnow, White Sucker), but their relative abundance within overall fish community was not high enough to drive down DO TIV results.

Due the variability in DO concentration observed between monitoring years, it's important to consider the temporal relationship between the water chemistry and biological data when evaluating low DO as a stressor. The biological data available was collected in June of 2009 during a summer with average precipitation. Streamflow conditions and DO concentrations were suitable for most warmwater fish at the time of sampling, and physical habitat conditions may have been more of a limiting factor during the 2009 monitoring season.

**Summary: Is low dissolved oxygen a stressor in Ely Creek?**

The DO regime of Ely Creek is closely linked to streamflow and annual precipitation levels in the watershed. In two of the three years in which data was collected, summer precipitation amounts were fairly normal, and DO conditions in Ely Creek were suitable for supporting a quality warmwater fish assemblage. However, during drought conditions in the late summer of 2012, critically low DO concentrations (less than 1 mg/L) were observed within the impaired reach. Under these conditions, sensitive fish would have perished or migrated to another reach or stream with more suitable DO concentrations. Although there are no observations to use as evidence, similar conditions were likely present in Ely Creek in 2006, when even less summer precipitation fell than 2012.

The loss of hydraulic connectivity between Ely Lake and Ely Creek during dry conditions likely exacerbates low flow/low DO conditions within the impaired reach. A large portion of the Ely Creek Watershed drains to Ely Lake, and when lake levels fall below the crest elevation of the dam at the lake outlet, it is possible that the creek loses one of its main sources of surface water. Hydrologic modeling of this watershed using HSPF and other tools would help define the relationship between Ely Lake and Ely Creek, and the impact that the dam has on the relationship between the two. More information on hydrological alteration in the Ely Creek Watershed can be found in Section 3.2.1.

The available biological data do not provide overwhelming evidence for or against low DO as a stressor. DO TIV results show a fish community that is neither tolerant nor intolerant of low DO. Fish abundance, and to a lesser extent fish taxa richness, were suppressed compared to high quality streams of the same FIBI class. The overall lack of abundance and diversity could be an indicator of a stream that experiences intermittent stress related to low streamflow and low DO.

Due to the fact that low DO concentrations occurred only during periods of extremely low flow, altered hydrology should be considered primary a stressor in this watershed, and the resulting low DO a secondary stressor or symptom of altered hydrology. The summer of 2012 was exceptionally dry when compared to the past 14 years of precipitation data, but low flows in Ely Creek are not simply a symptom of climate change or seasonal droughts.

### 5.16.3 Ely Creek: Summary of Stressors to Aquatic Life

**Table 69:** Summary of SID results for Ely Creek

| Candidate Cause | Result |
|---|---|
| **Poor Physical Habitat** | ● |
| **Low DO** | ● |
| **Altered Hydrology** | ● |
| Key: ● **= confirmed stressor** ○ **= Potential Stressor X = eliminated candidate cause** | |

## 5.17 Embarrass River

The Embarrass River has its headwaters in the wetlands to the southwest of Babbitt. Rosgen E-type channels and type lacustrine and unconfined alluvial valleys are the most common in this system, with shorter reaches of other stream types including B anc C type channels. There are several flow-through lakes on the Embarrass River: Sabin, Wynne, Embarrass, Cedar Island, Fourth, and Esquagama Lakes. The river is channelized between Wynne and Embarrass Lakes in the mid-1900's as a result of mining activities. The channel has also been levied upstream of Highway 135 to prevent flooding of the agricultural fields within the riparian corridor. Several mine pits and other mining featured are found within and adjacent to the Embarrass River Watershed. These features have the potential to alter groundwater flows to and from the river, and reduce water quality.

The Embarrass River as a whole is one of the flattest tributaries to the St. Louis River, dropping just 89 feet in a little over 54 miles (0.03%). Over half of that elevation drop happens in just one mile of channel, leaving the remaining 53 miles with a gradient of only 0.015% (<1 foot/mile).

The FIBI impairment on the upper Embarrass River is based on monitoring data from two stations, 09LS100 and 10EM045. Information and monitoring results from these stations and other Embarrass River monitoring stations are summarized in Table 70. The two impaired stations are located within the lower gradient reaches of the river, well above the chain of lakes near the city of Biwabik, Minnesota (Figure 249). Station 97LS005 is also located in the general vicinity of the impaired sites, however, monitoring at this station resulted in much better FIBI scores. Downstream of the chain of lakes, FIBI results Embarrass are exceptional. Station 09LS095 within this reach scored a 93 out of a possible 100 on the FIBI. The character of the river is much different below the chain of lakes, thus, these results cannot be accurately compared to data from the upper portions of the watershed.

A lack of headwaters minnow species and sensitive fish taxa were two key factors in the low FIBI scores at stations 09LS100 and 10EM145 in the upper Embarrass River. These two stations were dominated by tolerant (Central Mudminnow) or highly mobile (Northern Pike, White Sucker) fish taxa. Minnow species that are generally found in healthy headwaters streams of northern Minnesota (e.g. dace sp., shiner sp., darter sp.) were not present at these two monitorign stations. In contrast, station 97LS005 farther downstream supported several of these species (Northern Redbelly Dace, Blacknose Shiner, Johnny

Darter). Poor metric scores in fish abundance metrics were also a major factor in the low FIBI scores observed at the impaired stations.

**Table 70:** Biological monitoring sites and FIBI results from the Embarrass River

| | **Fish Assessments** | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Station** | **Drainage Area (mi2)** | **Gradient (%)** | **Stream Order (Strahler)** | **FIBI Class** | **FIBI Result *(visit year)*** | **FIBI Result *(visit year)*** | **FIBI Result *(visit year)*** | **Standard** | **IBI Lower Confidence Limit** | **IBI Upper Confidence Limit** |
| **09LS100** | 18.89 | 0.02 | 2 | 7 | **31** *(2009)* | - | - | 42 | 32 | 52 |
| **10EM045** | 44.58 | 0.02 | 3 | 7 | **0** *(2009)* | **0** *(2010)* | - | 42 | 32 | 52 |
| **97LS005** | 44.58 | 0.02 | 3 | 5 | **50** *(1997)* | **54** *(1997)* | **52** *(2009)* | 47 | 38 | 56 |
| **09LS095** | 115.07 | 0.03 | 3 | 5 | **93** *(2009)* | - | - | 47 | 38 | 56 |

Water quality and physical habitat data were used to identify a list of candidate causes for the FIBI impairment in the upper Embarrass River. The following candidate causes were selected for further evaluation;

1. **Low DO 2. Sulfate Toxicity**

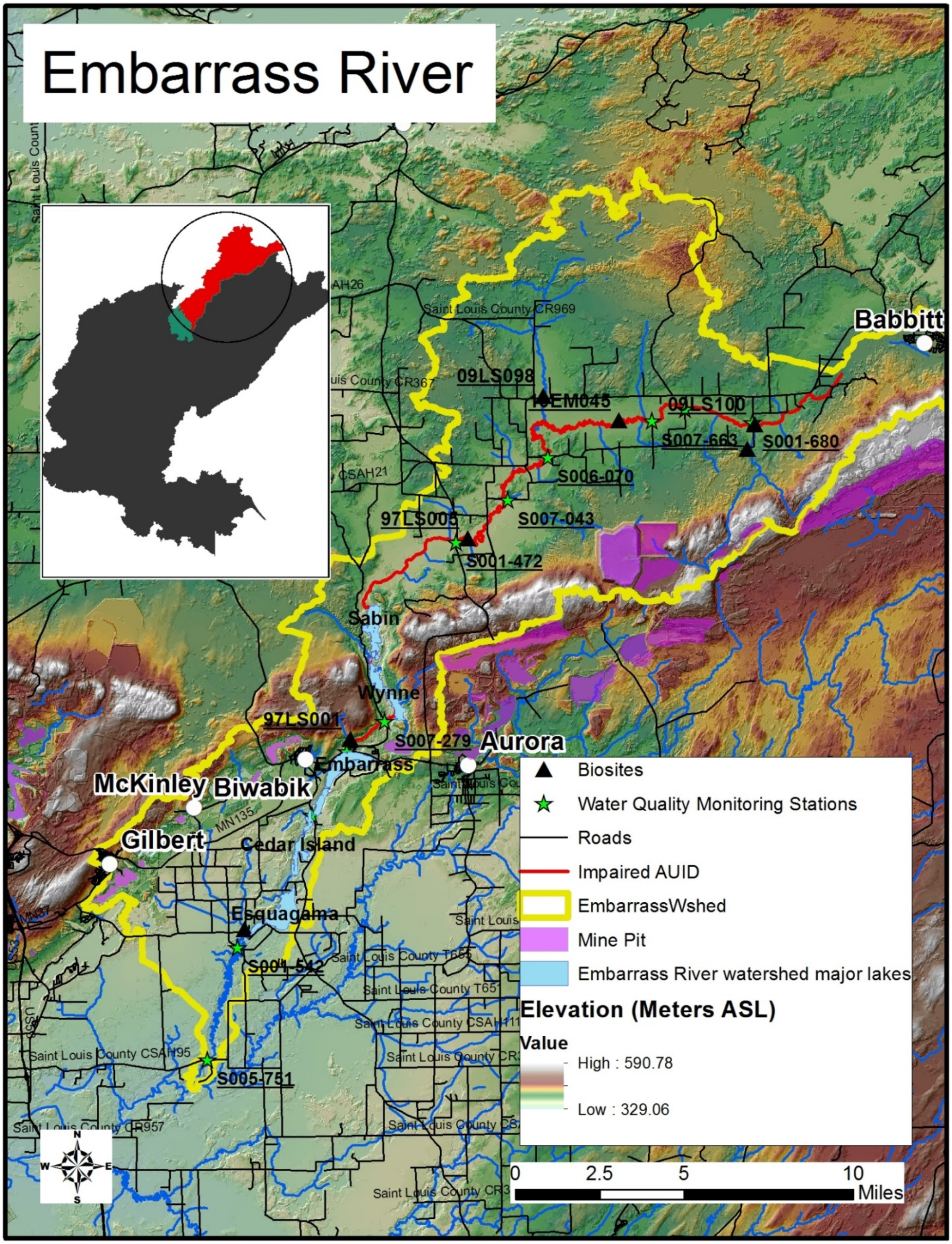


**Figure 249:** Map of Embarrass River Watershed and impaired stream reach

### 5.17.1 Low Dissolved Oxygen

Low DO was identified as a potential cause of low FIBI results in the upper Embarrass River. The impaired reach extends from the headwaters of the Embarrass River to its confluence with Embarrass Lake, just east of Babbitt, Minnesota. Available DO data for the Embarrass River were collected using several methods, including instantaneous (point) measurements, longitudinal synoptic monitoring profiles, and multiple short-term (approximately one week) deployments of continuous monitoring equipment at several locations.

Point measurements of DO with the impaired reach are displayed in Figure 250 by station and calendar month. These data were collected between the years of 1976 and 2013. Results from several stations fall well below the 5 mg/L DO standard between the months of May and September. Several sub-5mg/L observations have also been recorded in the headwaters of the Embarrass River at station S001-680 during winter months. The vast majority of point measurements below 5 mg/L were collected at stations S001-680, S001-472 (a.k.a. bio station 97LS005), and S006-070 during low flow / high air temperature periods in the months of July and August. Overall, just over 44% of the point measurements of DO collected in the impaired reach of the Embarrass River were below the 5 mg/L water quality standard.

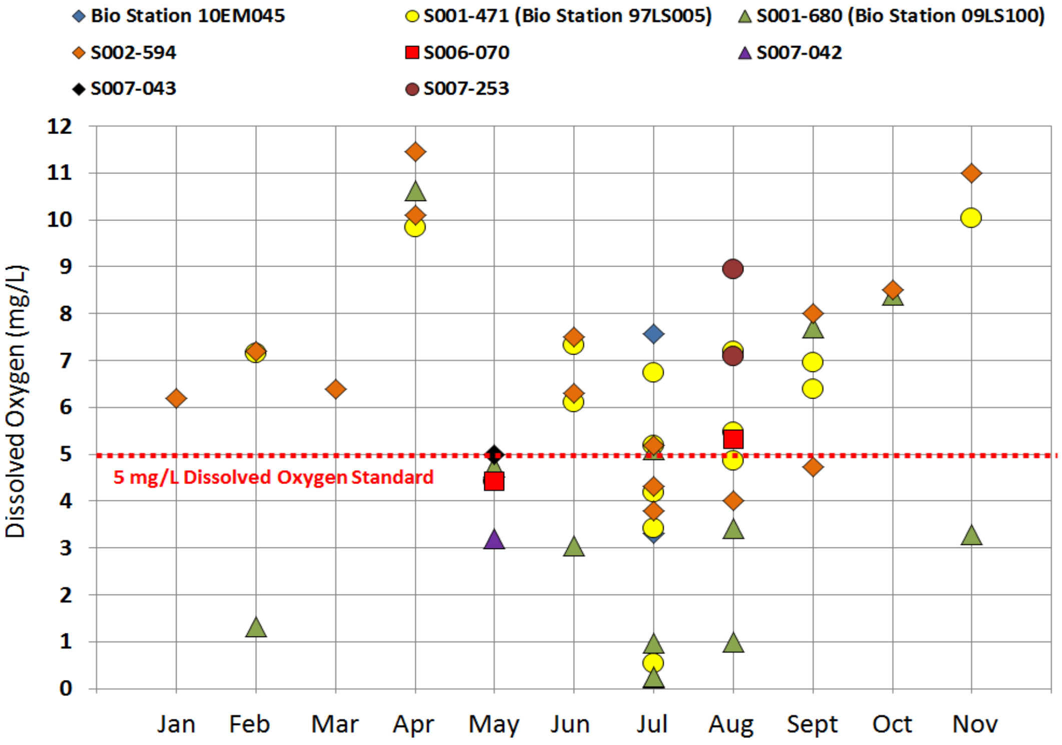


**Figure 250:** Point measurements of DO collected within the impaired reach of the Embarrass River plotted by monitoring station and calendar month.

Longitudinal synoptic measurements of DO were collected on the Upper Embarrass River in August of 2011 and 2012. The goal of synoptic monitoring is to observe conditions at a large number of sites within a short time period, thus providing a snapshot of conditions over a broad area. Typically, morning and afternoon sampling runs are conducted to calculate approximate diurnal DO flux, but for the Embarrass River, only early morning sampling runs were completed due to time constraints. In 2011, DO concentrations were very low at the two stations located nearest to the headwaters of the Embarrass River (Figure 251). DO concentrations at the upper most station, which is co-located with biological monitoring station 09LS100, were around 1 mg/L. DO concentrations increased in a downstream direction, and met the water quality standard at four of the six total stations. Data from the 2012 longitudinal profile did not result is as many sub-5 mg/L observations (Figure 252). The general trend of increasing DO concentrations in a downstream direction was still apparent, but minimum DO concentrations at all stations except S007-042 were above the DO water quality standard.

Continuous DO monitoring profiles were completed in 2011, 2012, and 2013. Data were collected primarily during baseflow or near baseflow conditions. Figure 253 displays the results from a July 2013 continuous monitoring event that includes four monitoring stations, three of which are located in the impaired reach, and one station (S005-571 / 09LS095) which had excellent fish and MIBI scores located on a non-impaired reach downstream of the chain of lakes (see map in Figure 249). Very low DO concentrations (> 2 mg/L) were observed for the entire duration of the profile at the two monitoring stations closest to the headwaters (S001-680 and S007-663). DO conditions improved further downstream at biological monitoring station 97LS005, but remained narrowly below the 5 mg/L water quality standard. Downstream of the chain of lakes at station S007-571 (bio station 09LS095), DO concentrations were well above the 5 mg/L.

Additional results from continuous monitoring of DO are provided in Table 71. Low DO concentrations were observed during DO profiles collected at monitoring station 97LS005 in July and September of 2012, as well as station S007-043 in September of 2012. All continuous DO data from station S005-571 downstream of the chain of lakes are in compliance with state water quality standards.

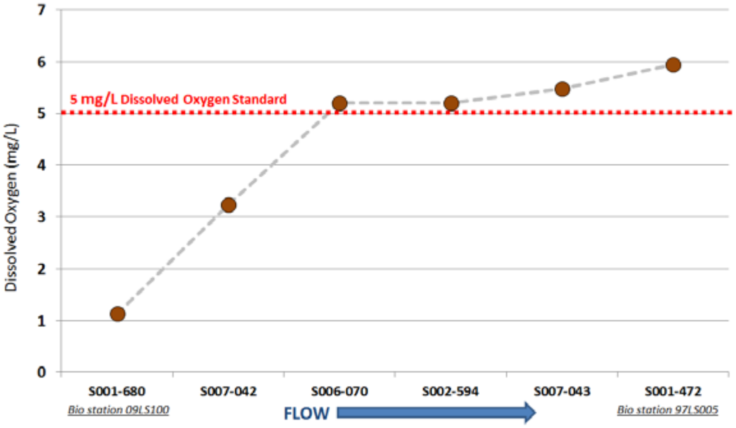


**Figure 251:** Longitudinal DO data from upper Embarrass River monitoring stations (August 2011)

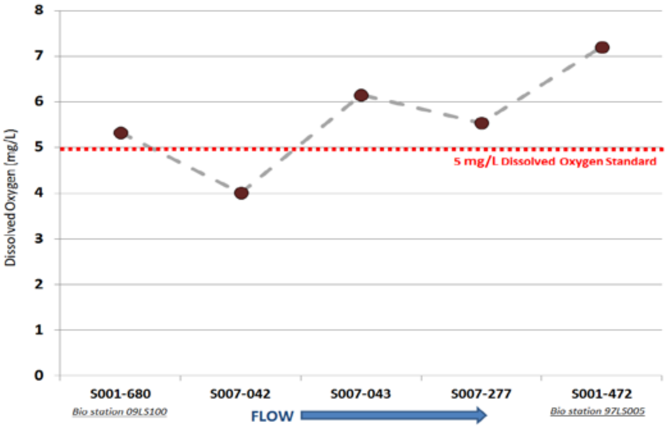
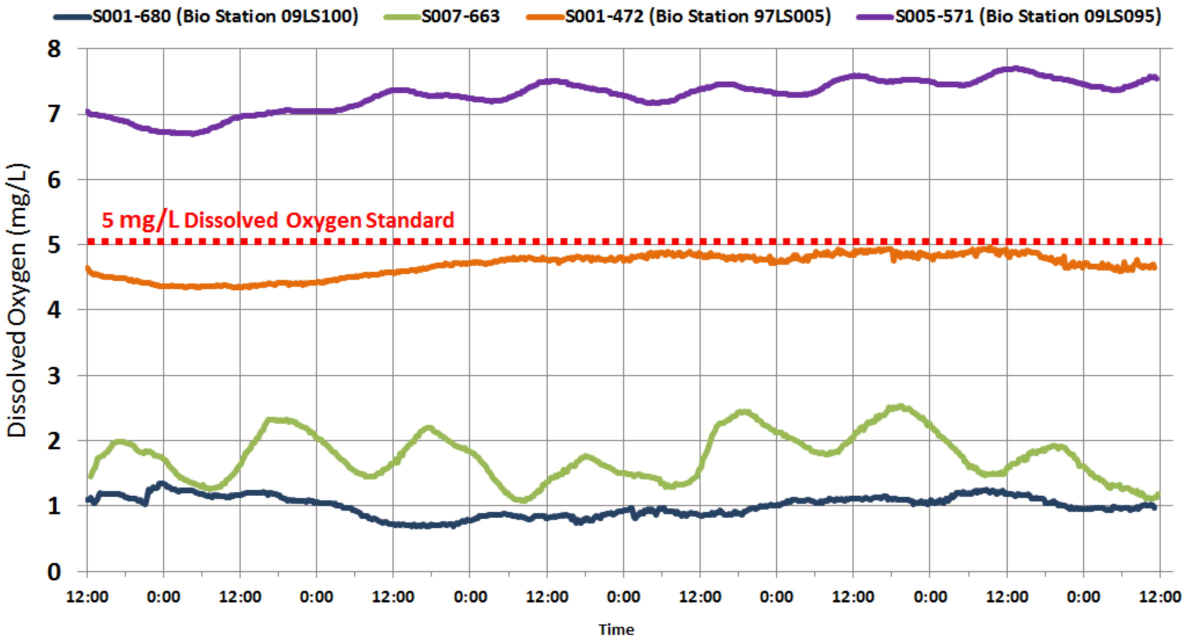


**Figure 252:** Longitudinal DO data from upper Embarrass River monitoring stations (August 2012)



| Station | # of readings | DO Min. (mg/L) | DO max. (mg/L) | Avg. DO Flux (mg/L) | % Readings below 5 mg/L |
|---|---|---|---|---|---|
| S001-680 (Bio Station 09LS100) | 669 | 0.68 | 1.35 | 0.24 | **100%** |
| S007-063 | 669 | 1.07 | 2.54 | 0.89 | **100%** |
| S001-472 (Bio Station 97LS005) | 669 | 4.34 | 4.96 | 0.20 | **100%** |
| S005-751 (Bio Station 09LS095) | 670 | 6.69 | 7.71 | 0.31 | 0% |

**Figure 253:** Results of continuous DO monitoring at four Embarrass River monitoring sites from 7/18/2013 to 7/25/2013. Stations S001-680, S007-663, and S001-472 are located within the impaired reach. Station S005-571 is co-located with a high quality biological monitoring site further downstream.

**Table 71:** Summary statistics for additional continuous DO monitoring in the Embarrass River

| Station | Date | # of Obs. | DO Min. (mg/L) | DO max. (mg/L) | Avg. DO Flux (mg/L) | % Readings below 5 mg/L |
|---|---|---|---|---|---|---|
| **S007-043** | 9/6/12 - 9/12/12 | 667 | **2.31** | 5.25 | 1.52 | **95%** |
| **S001-472 (Bio Station 97LS005)** | 9/6/12 - 9/12/12 | 568 | **4.56** | 6.12 | 0.94 | **27%** |
| **S001-472 (Bio Station 97LS005)** | 7/3/12 - 7/10/12 | 661 | **3.1** | 4.61 | 0.34 | **100%** |
| **S001-472 (Bio Station 97LS005)** | 8/23/13 - 8/29/13 | 565 | 5.01 | 7.14 | 1.16 | 0% |
| **S005-571 (Bio Station 09LS095)** | 8/23/2013 - 8/29/13 | 569 | 6.55 | 8.68 | 1.61 | 0% |

## Sources and Pathways Related to Low Dissolved Oxygen

### Wetlands

The topography of the upper Embarrass River Watershed is generally flat, and organic flat wetlands are a major feature of the landscape. These wetland complexes are the dominant source of flow in the headwaters reaches of the Embarrass River and its tributaries, as precipitation runs off the wetlands via saturation and overland flow. The runoff progresses slowly due to the gentle sloping of the wetlands towards the streams, and can persist for long periods after precipitation events.

Bourdaghs and Gernes (2013) summarized the effects of these wetlands on water quality in the Embarrass River after conducting a series of site visits in July of 2013:

> *The adjacent wetlands at the upper Embarrass Watershed sites were all the Organic Flat HGM type. In terms of the interaction of water flows between the wetlands and the streams—the wetlands are predominantly providing water to the streams here. This occurs via saturation-overland flow. Precipitation slowly runs off the saturated soils bringing with it high DOM, low pH, and low dissolved oxygen soil water that mixes with the precipitation in the top layer of the organic soil/leaf litter before entering the streams. Extensive Organic Flat wetlands are a common feature in this part of the watershed as they appear to be the headwaters (and continue for long stretches) of the Embarrass and tributaries.*

Adjacent (riparian) wetlands to the lower Embarrass River biological monitoring station (09LS095) function differently than the Organic Flat wetlands in the headwaters. An active floodplain has developed here, and the soils of these wetlands are mineral alluvial deposits from regular flooding, mapped as Entisols (Bourdaghs and Gernes 2013). These are classed as "riverine wetlands" where the predominant water source is from overbank flow from the channel and water losses occur via the return of floodwater back to the channel (Bourdaghs and Gernes 2013). Ultimately, the different wetland processes found in the lower watershed are not resulting in the low DO concentrations as observed in the headwaters. Bourdaghs and Gernes (2013) summarized the differences in their report:

> *Interaction of water flows is opposite at the lower Embarrass River site. Here the river is supplying the adjacent wetlands with water and sediment via flooding. Water returns to stream channels when levels recede relatively quickly via lateral/downstream flow and through shallow ground water due to the relatively high hydrologic conductivity of the sandy soils. These processes do not result in prolonged saturation conditions that lead to the buildup of organic material and subsequent delivery of high DOM/low pH/low DO water to the stream channel.*

### Nutrients and Productivity

Elevated phosphorous concentrations are often linked to increases in primary productivity, and can result in low or highly fluctuating DO concentrations (source). A total of 27 observations of TP have been collected in the upper Embarrass River Watershed, with many of the results (24 of 27 samples) being older data from the 1970's. Results are highly variable, ranging from a low of 0.023 mg/L to a maximum of 0.413 mg/L. More contemporary sampling results, collected in conjunction with biological monitoring events, range from 0.024 mg/L to 0.081 mg/L. Based on these observations, it can be concluded that phosphorous levels in the upper Embarrass River are elevated compared to river nutrient criteria value of 0.055 mg/L.

Despite the elevated phosphorous concentrations, primary productivity through algae and other aquatic plants is rather low in this reach, and does not appear to be linked to low DO concentrations. The low diurnal DO flux observed during continuous monitoring supports this claim. Nutrient levels are likely elevated in this reach due to natural background factors related to the organic flat wetlands and their tendency to deliver nutrient rich, low DO water the adjacent water bodies. The only BOD data available for the Embarrass River were collected in response to a fish kill event caused by an industrial wastewater spill and are not representative of normal conditions.

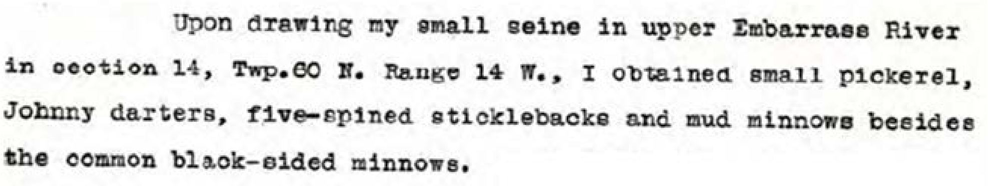
## Biological Response to Low Dissolved Oxygen

The fish community of the upper Embarrass River is characterized by low overall fish abundance and by the dominance of species that can tolerate fairly low levels of DO (Table 72, Figure 254). White Sucker, Yellow Perch, and Central Mudminnow individuals accounted for nearly 60% of the fish sampled in the impaired reach. Central Mudminnow individuals are highly tolerant of low DO concentrations, while Yellow Perch and White Sucker are found in streams with a wide range of DO concentrations. None of the fish species observed in the impaired reach of the upper Embarrass River are considered sensitive to low DO concentrations.

**Table 72:** Combined fish abundance and relative percentages observed at three biological monitoring stations in the upper Embarrass River.

| Species | # observed | % of upper Embarrass Fish Assemblage |
|---|---|---|
| White Sucker | 62 | 29.8% |
| Yellow Perch | 35 | 16.8% |
| Central Mudminnow | 26 | 12.5% |
| Johnny Darter | 24 | 11.5% |
| Rock Bass | 15 | 7.2% |
| Northern Pike | 12 | 5.8% |
| Burbot | 11 | 5.3% |
| Golden Shiner | 10 | 4.8% |
| Common Shiner | 5 | 2.4% |
| Brook Stickleback | 2 | 1.0% |
| Black Bullhead | 1 | 0.5% |
| Blacknose Shiner | 1 | 0.5% |
| Northern Redbelly Dace | 1 | 0.5% |
| Pumpkinseed | 1 | 0.5% |
| Trout-Perch | 1 | 0.5% |
| Walleye | 1 | 0.5% |

Thaddeus Surber conducted an extensive survey of St. Louis River drainage streams and published his findings in 1926. His work included a biological survey of the upper Embarrass River very near the MPCA's station 09LS100. An excerpt from the report describing this sample is shown below:



> Upon drawing my small seine in upper Embarrass River in section 14, Twp.60 N. Range 14 W., I obtained small pickerel, Johnny darters, five-spined sticklebacks and mud minnows besides the common black-sided minnows.

Based on this description, his survey informal survey of the fish population included Northern Pike, ("small pickerel"), Johnny Darter, Brook Stickleback ("five-spined sticklebacks"), Central Mudminnow, and Blacknose Shiner ("common black-sided minnows"). All of these species are still found in the upper Embarrass River today. Based on this historical account, it does not appear that the fish community of the upper Embarrass River has changed significantly since the early 1920's.

Species level DO TIVs were developed by the MPCA biologists based on paired statewide biological and water chemistry data sets. The TIV were developed based on the relative abundance of various fish species observed under different DO conditions (more on TIV development can be found in Section 4. The species level TIV's for DO were sorted into a series of tolerance classes based on their TIV values. The most upstream monitoring station is dominated by high tolerant fish species (Central Mudminnow), and total fish populations were relatively low at the upper stations (09LS100 and 10EM145) compared to stations further downstream. Aside from station 09LS095 which is located downstream of the chain of lakes, all species observed in Embarrass River are neutral to highly tolerant of low dissolved conditions.

Community-level DO tolerance indicator values (DO TIV) were calculated for the fish community of the upper Embarrass River. With the exception of station 10EM045, biological monitoring stations located on the impaired reach (97LS005, 09LS100) had relatively low fish community DO TIV values compared to reference sites from similar IBI classes (Figure 255). Station 09LS100 had a particularly low DO TIV score, as this station was dominated by Central Mudminnow individuals. DO TIV results were significantly improved at station 09LS095, which is located downstream of the impaired reach and has more favorable DO conditions for supporting sensitive taxa.

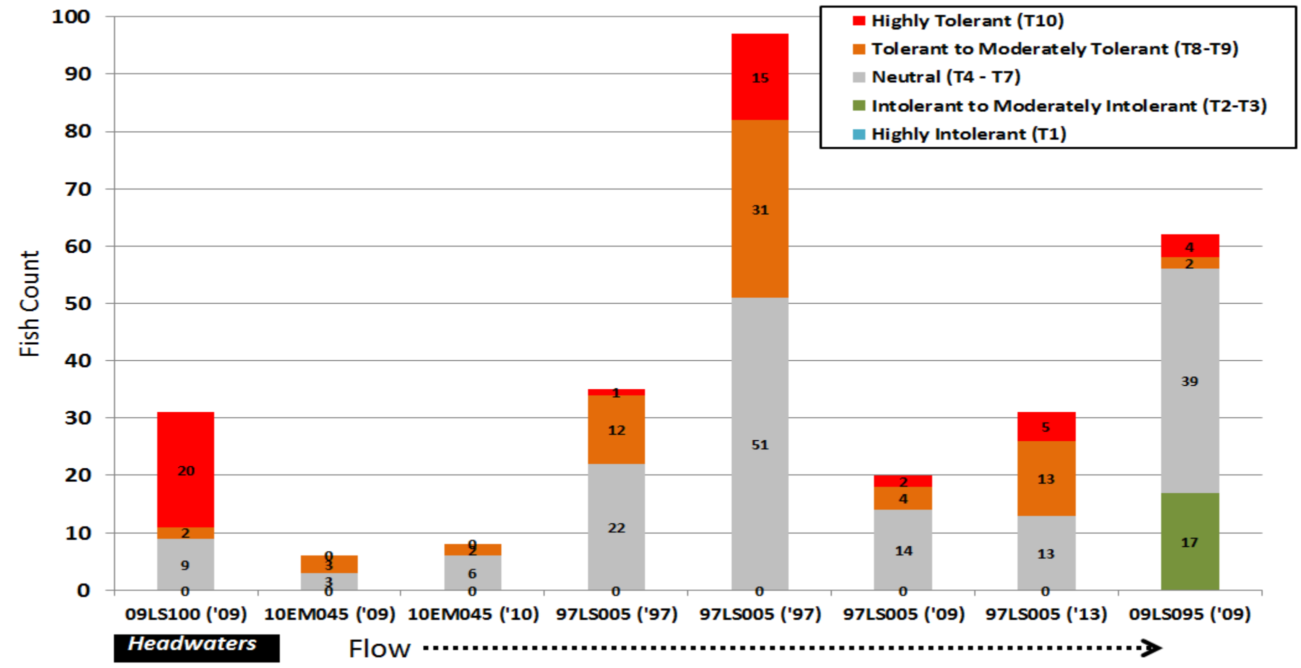


**Figure 254:** Total fish counts and number of individuals within DO tolerance classes observed at biological monitoring stations on the upper Embarrass River.

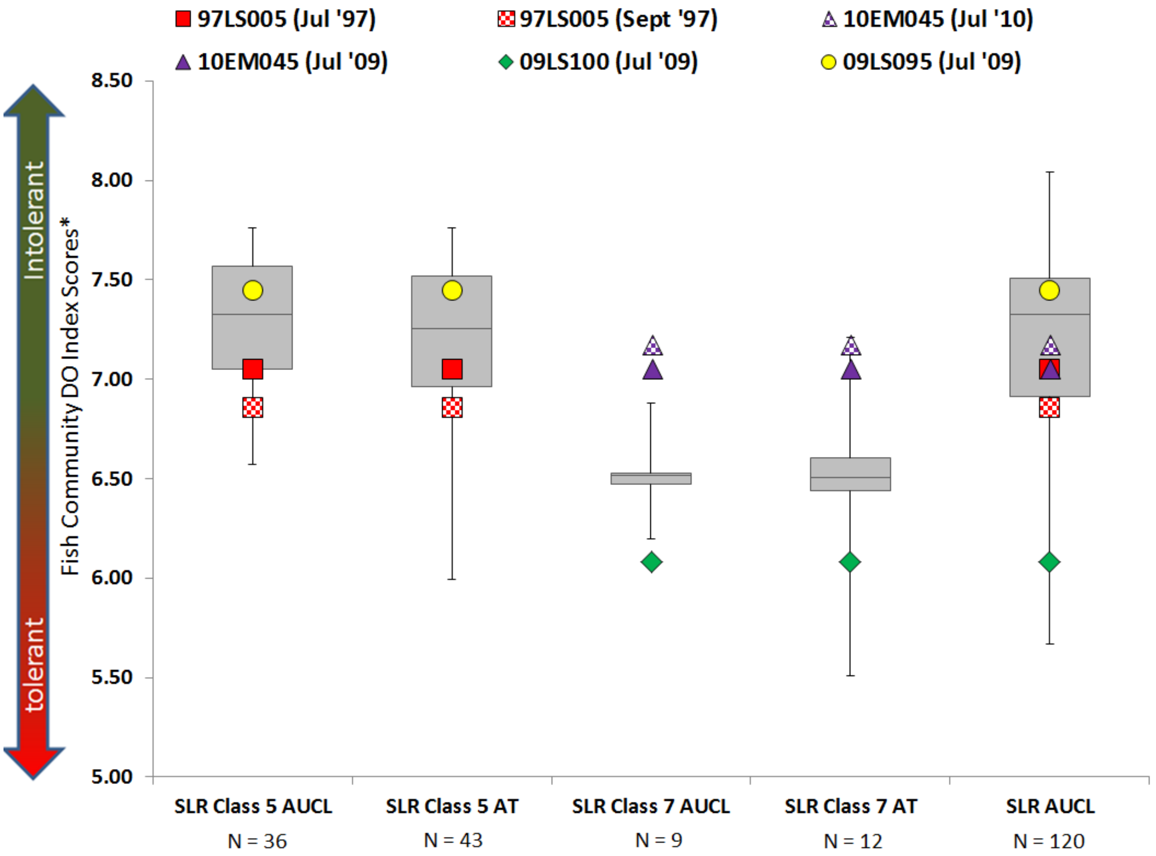


**Figure 255:** Fish counts and number of each individuals within DO tolerance classes at upper Embarrass River stations. * See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

**Summary: Is low dissolved oxygen a stressor in Embarrass River?**

Overall, fisheries data from the upper Embarrass River provide supporting evidence for low DO as a limiting factor. The fish taxa present in this reach are either tolerant of low DO concentrations (e.g. Central Mudminnow, Golden Shiner, Black Bullhead) or have the ability to migrate to areas with more suitable conditions when DO concentrations in the Embarrass River are unsuitable (e.g. Northern Pike, Burbot). The low DO concentrations observed in the water chemistry data are likely linked to the abundant wetlands in the watershed, which deliver nutrient rich, oxygen depleted water to the stream. Additional causes of low DO that cannot be eliminated include flow alteration from mining land-use in the watershed and adjacent drainages and stream channelization.

## 5.17.2 Sulfate Toxicity

Sulfate was initially included as a candidate cause of impairment for the Embarrass River due to presence of mining land use in its watershed. Further analysis of the available sulfate data shows sulfate concentrations that are significantly higher than natural background conditions in the SLRW. A maximum sulfate concentration of 123 mg/L was observed at station S002-594 in February of 2014. However, the sulfate concentrations in the Embarrass River are fairly low overall in comparison to many of the values cited by researchers and other governing agencies as harmful to aquatic life (see Section 3.1.6). Within the impaired reach, the average sulfate concentration is only 30 mg/L (n=43).

Sulfate concentrations in the Embarrass River experience a spike downstream of the confluence with Spring Mine Creek, which carries a very high sulfate concentration (see Section 5.14.2). During a February 2014 sampling event under baseflow conditions, the sulfate concentration observed in the headwaters of the Embarrass River at station S001-680 was 1.29 mg/L. Just downstream of the confluence with Spring Mine Creek at Embarrass River station S002-594, the concentration increased to 123 mg/L. At the time, the sulfate concentration observed in Spring Mine Creek was 751 mg/L. Clearly, Spring Mine Creek is a major source of sulfate in the upper reaches of the Embarrass River.

**Summary: Is sulfate toxicity a stressor in the Embarrass River?**

Based on data collected from the Embarrass River, as well as current research and WQ standard development (see Section 3.1.6), sulfate toxicity is not considered a stressor to fish and macroinvertebrate populations at this time. Sulfate concentrations in the Embarrass River frequently exceed the draft water quality standard for wild rice of 10 mg/l, but do not appear to be high enough to be problematic for fish and macroinvertebrates based on currently knowledge of this stressor.

**Embarrass River: Summary of Stressors to Aquatic Life**

**Table 73:** Summary of SID results for the Embarrass River

| Candidate Cause | Result |
|---|---|
| **Low Dissolved Oxygen** | ● |
| **Sulfate Toxicity** | **X** |
| **Altered Hydrology** | ○ |

Key: ● **= confirmed stressor** ○ **= Potential Stressor X = eliminated candidate cause**

### Virginia Mesabi Range Watershed Zone

Biological impairment listings within the Virginia Mesabi Range (VIR) Watershed zone include two segments of Elbow Creek and the outlet stream of Manganika Lake (Manganika Creek), which is a very short tributary to the East Two River. The watersheds of these two streams have been dramatically altered due to mining land-uses. While Elbow Creek still remains a free-flowing stream has retained most of its original length, much of the stream is channelized and routed around a series of mine pits and waste rock stockpiles. It flows through nutrient-impaired Elbow Lake at its mid-point before joining the St. Louis River near the town of Forbes. Elbow Creek has historically received dewatering flow from abandoned mine pits, and is currently the receiving water for the city of Eveleth's WWTP effluent.

**Figure 256:** Photos from impaired stream segments in this watershed zone. Manganika Creek (left), Upper Elbow Creek (middle), and lower Elbow Creek (right).

## 5.18 Unnamed Tributary to East Two River (Manganika Creek)

Manganika Creek begins just south of the town of Virginia, and receives stormwater runoff from that area. The stream is ditched as it flows past the WWTP on its way to Manganika Lake. The impaired AUID is located downstream of Mangnika Lake to the confluence with the East Two River (Figure 257). Mangnika Creek is a sinuous and narrow E channel within a wide lacustrine valley for the entire length of the impaired AUID. The average slope of the reach is 0.075%, or about 4 feet/mile.

The watershed area of Manganika Creek has been reduced by 48% due to mining and urban development, which equates to reduction in mean annual flow of 2.86 cubic feet per second (cfs). This reduction in flow has been replaced to some degree by current mine pit dewatering permits in the watershed, which discharge 2.45 cfs to the stream. There are currently plans to expand a mine pit in the vicinity of the creek that will alter additional land in the watershed. The city of Virginia WWTP currently discharges effluent to a tributary of Manganika Lake. Manganika Lake is currently listed as impaired for elevated nutrient concentrations.

Fish and macroinvertebrate community data were collected at two stations on Manganika Creek (Table 74). One of these stations is located upstream of Manganika Lake on a channelized portion of the creek (98LS015), and the other is located downstream of the lake outlet on a sinuous, natural channel (09LS078). Due to the channelization of the stream above Manganika Lake, only the lower site (09LS078) was used to assess the biological integrity of this drainage. The fish and macroinvertebrate communities found in Manganika Creek are severely degraded. A FIBI score of 0 out of a possible 100 was recorded for both of the monitoring stations on this stream. Only four fish species were observed; Central Mudminnow, Brook Stickleback, Brassy Minnow, and Yellow Perch. The overall catch in the reach below Manganika Lake was extremely low, as only 13 fish were collected at that station. This fish assemblage represents a dramatic departure from what is typically observed in healthy headwaters streams in northern Minnesota.

The MIBI results from these stations also indicate severe impairment (Table 74). The community was dominated by various chironomid taxa, in particular species from the genera *Glyptotendipes* and Dicrotendipes. Nearly 70% of the organisms identified were from these two genera, which are well known to be very tolerant of many forms of pollution and habitat degradation. *Glyptotendipes* sp. are known to be very tolerant of organic pollution, and Dicrotendipes sp. have been linked to streams with moderate to high water temperatures, organic matter, TSS, pH, phosphates, and sulfates (Al-Shami et al 2010).

**Table 74:** Biological monitoring sites and fish/MIBI results from the Embarrass River

| **Fish Assessments** | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Station** | **Drainage Area (mi2)** | **Gradient (%)** | **Stream Order (Strahler)** | **FIBI Class** | **FIBI Result *(visit year)*** | **FIBI Result *(visit year)*** | **Standard** | **IBI Lower Confidence Limit** | **IBI Upper Confidence Limit** |
| 98LS015 | 1.41 | 0.28 | 1 | 6 | **0** *(1998)* | **0** *(2009)* | 42 | 26 | 58 |
| 09LS078 | 5.70 | 0.10 | 1 | 6 | **0** *(2009)* | - | 42 | 26 | 58 |
| **Macroinvertebrate Assessments** | | | | | | | | | |
| **Station** | **Drainage Area (mi2)** | **Gradient (%)** | **Stream Order (Strahler)** | **Invert IBI Class** | **Invert IBI Result *(visit year)*** | **Invert IBI Result *(visit year)*** | **Standard** | **IBI Lower Confidence Limit** | **IBI Upper Confidence Limit** |
| 98LS015 | 1.41 | 0.28 | 1 | 4 | **4.49** *(1998)* | **0.89** *(2009)* | 51 | 37.40 | 64.60 |
| 09LS078 | 5.70 | 0.10 | 1 | 4 | **14.45** *(2009)* | - | 51 | 37.40 | 64.60 |

Candidate causes for impairment that will be evaluated for Manganika Creek include;

1. **Low Dissolved Oxygen**
2. **Total Suspended Soilds (TSS)**
3. **Elevated pH**
4. **Ammonia Toxicity**
5. **Sulfate Toxicity**
6. **High Specific conductivity**

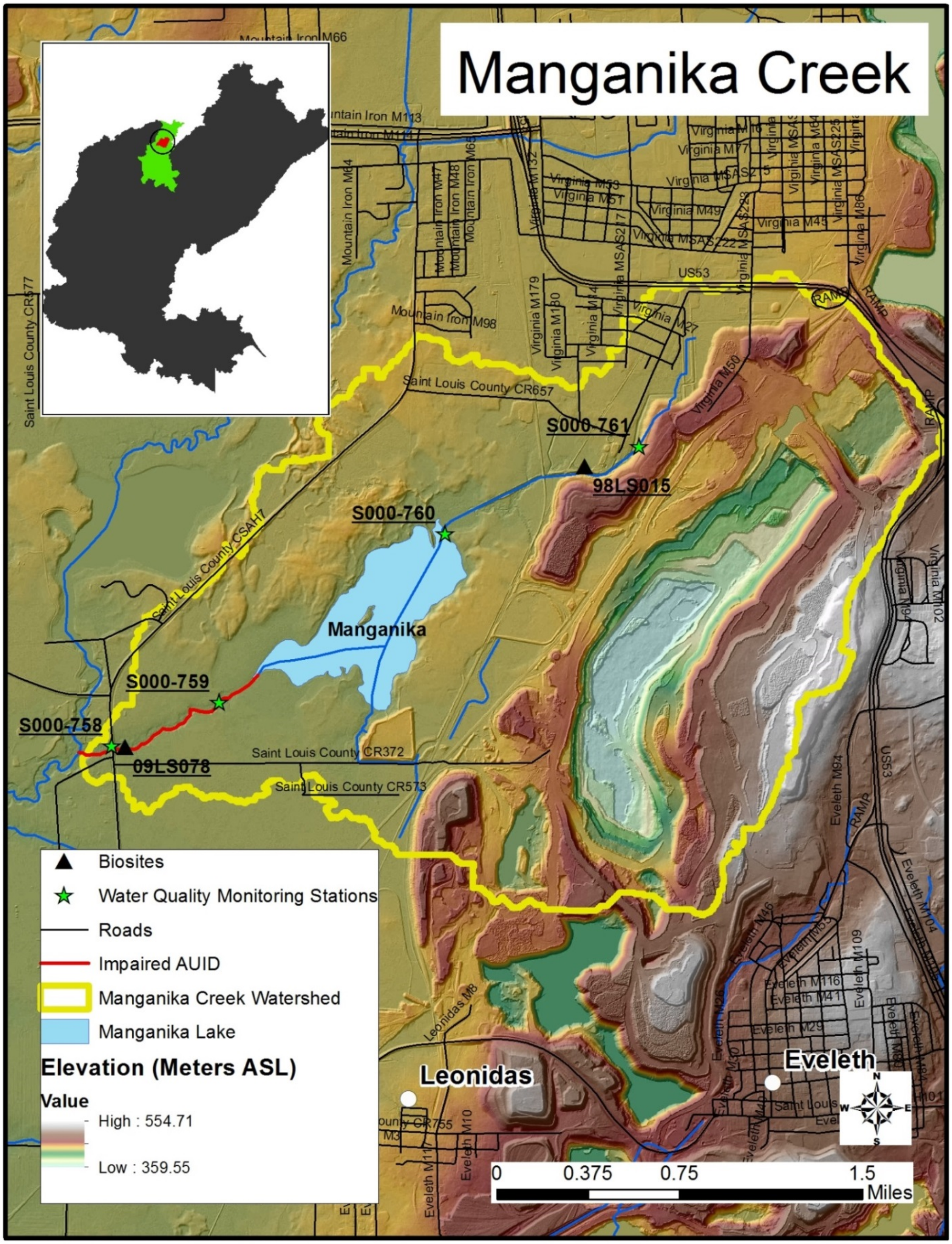


**Figure 257:** Manganika Creek Watershed, impaired stream segments, and monitoring locations

## 5.18.2 Dissolved Oxygen

Low DO was identified as a candidate cause for impairment in Manganika Creek. Available DO data for this impaired stream includes instantaneous (point) measurements and continuous monitoring over short time intervals (approximately 5 days). DO concentrations are generally above the water quality standard, which requires a minimum of 5 mg/L in warmwater streams like Manganika Creek (Figure 260) Several sub 5 mg/L measurements have been observed at the two biological monitoring stations during the months of February and June. Although these data provide an overview of DO conditions in Manganika Creek, very few of these measurements were collected during early morning hours when DO concentrations tend to be at daily minimums.

Continuous monitoring data present a different picture of DO concentrations in Manganika Creek, one that is less favorable for aquatic life. Only one continuous DO profile was collected in Manganika Creek due to difficult in-stream conditions for deploying equipment long-term. Elevated concentrations of algae and fine sediment were found to foul monitoring sensors after only four days of deployment. Results from the August 2011 continuous DO profile revealed much lower DO concentrations than those obtained during spot measurements (Figure 259). Daily minimum concentrations during continuous monitoring ranged from 0.71 mg/L to 2.41 mg/L, and concentration remained below the 5 mg/L water quality standard for nearly the entire four-day monitoring period. A minor rainfall event (approximately 0.2 inch) occurred during the first few days of the monitoring period, which resulted in lower DO concentrations and less diurnal change. Towards the end of the monitoring period, baseflow conditions returned which caused an increase in DO concentrations and diurnal DO flux. DO flux on the final day was around 4.2 mg/L. DO flux greater than 4.0 mg/L is an indicator of excess nutrient enrichment and can be harmful to sensitive aquatic life (Heiskary 2013).

**Figure 258:** Significant algae blooms originating in Manganika Lake cause oxygen depletion and increased suspended solids in Manganika Creek (left photo). Central Mudminnow (*Umbra limi*), which can survive in anoxic conditions in streams and lakes, were the most abundant fish species at station 09LS078 (right photo).

### Sources and Pathways of Low Dissolved Oxygen

The negative effects of low DO (and high DO flux) are seen primarily within the impaired reach of Manganika Creek, which is located immediately downstream of Manganika Lake. The DO concentrations are unsuitable in this reach largely due to processes occurring in the lake, where major algae blooms are commonly observed. Manganika Lake is currently on the impaired waters list for excess nutrients and is one of the most significantly degraded lakes in the Northern Lakes and Forests Ecoregion (Jesse

Anderson, MPCA, personal communication). The photo in Figure 258 provides an example of an algae bloom in this system. The DO regime of Manganika Creek is heavily influenced by the biological and chemical processes taking place in Manganika Lake just upstream.

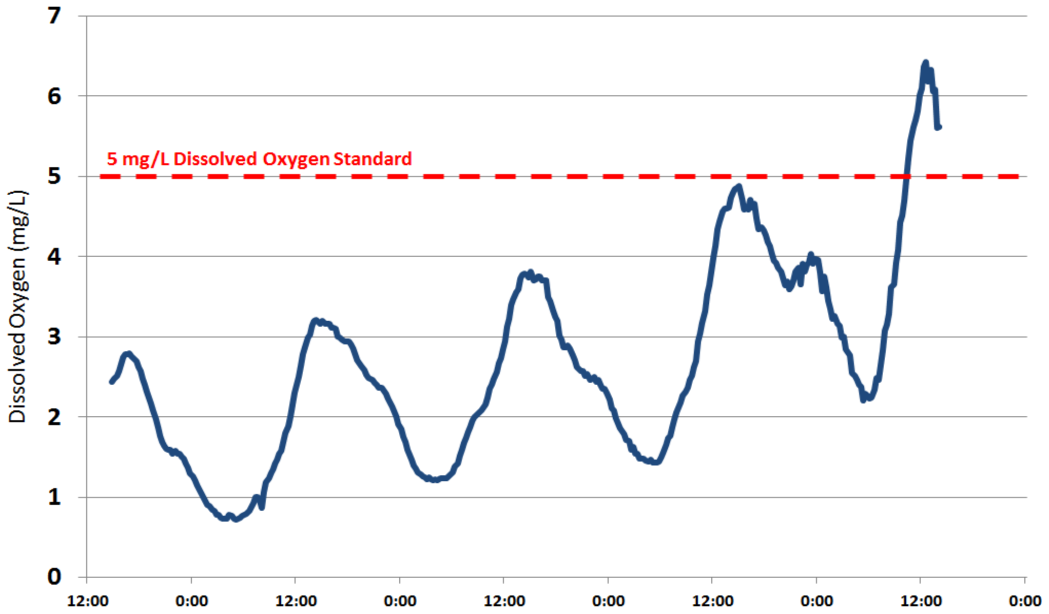


**Figure 259:** Results from continuous DO monitoring at station 09LS078 (August 2011)

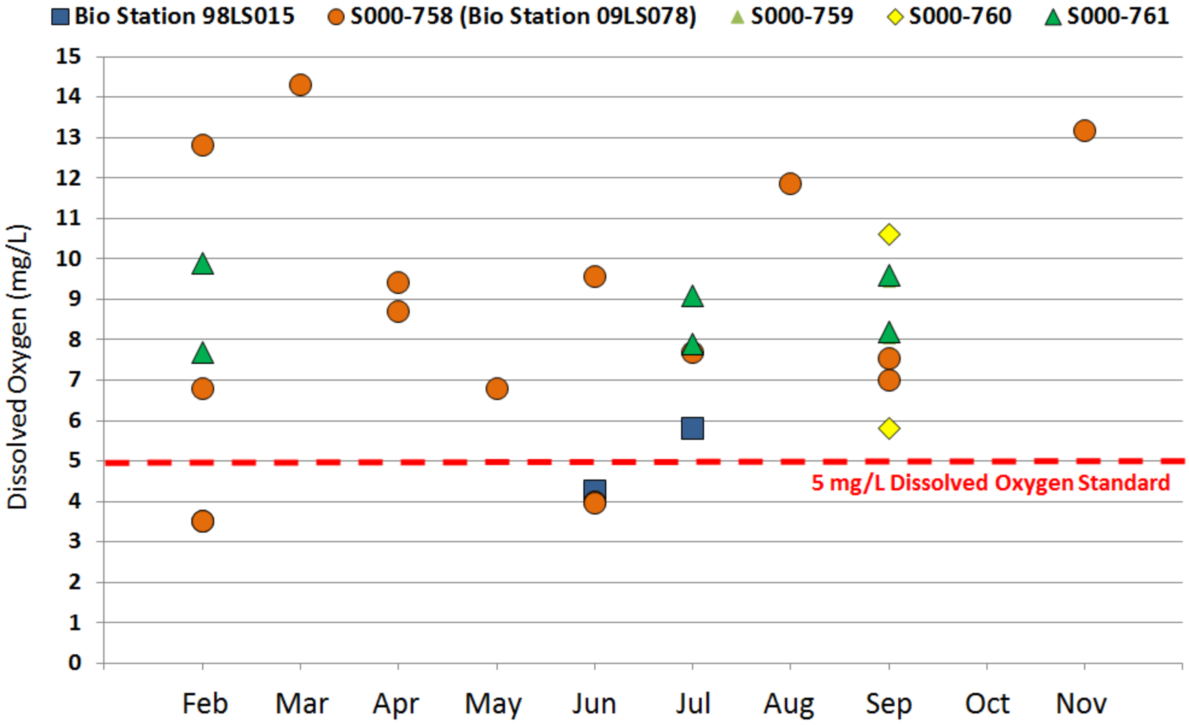


**Figure 260:** Point measurements of DO collected from Manganika Creek

The TP concentrations in Manganika Creek are exceptionally high compared to minimally impacted streams in the SLRW. The average TP concentration in the impaired reach is 0.230 mg/L, compared to mean concentrations of 0.035 to 0.060 mg/L observed in the SLRW reference streams discussed in Section 1.2.3 of this report. Sources of phosphorous in the Manganika Creek Watershed include internal loading driven by high sulfate concentrations, streambank erosion, urban runoff, mining land uses, and effluent from the city of Virginia's WWTP. Aside from the WWTP, which has records of effluent monitoring from 1998 through 2014, TP loading from these watershed sources are difficult to quantify.

The WWTP operators have monitored the TP concentrations of effluent discharged to Manganika Creek since 1998. Calendar month average TP concentrations of the effluent are typically between 0.50 and 0.75 mg/L, but concentrations exceeding 2 mg/L have been recorded on rare occasions (Figure 261). With a few exceptions, this facility shows a good history of staying in compliance with the concentration based TP limit (1.0 mg/L) listed in the current discharge permit. Nonetheless, WWTP effluent remains a major source of nutrient loading to Manganika Lake and its impaired outlet stream.

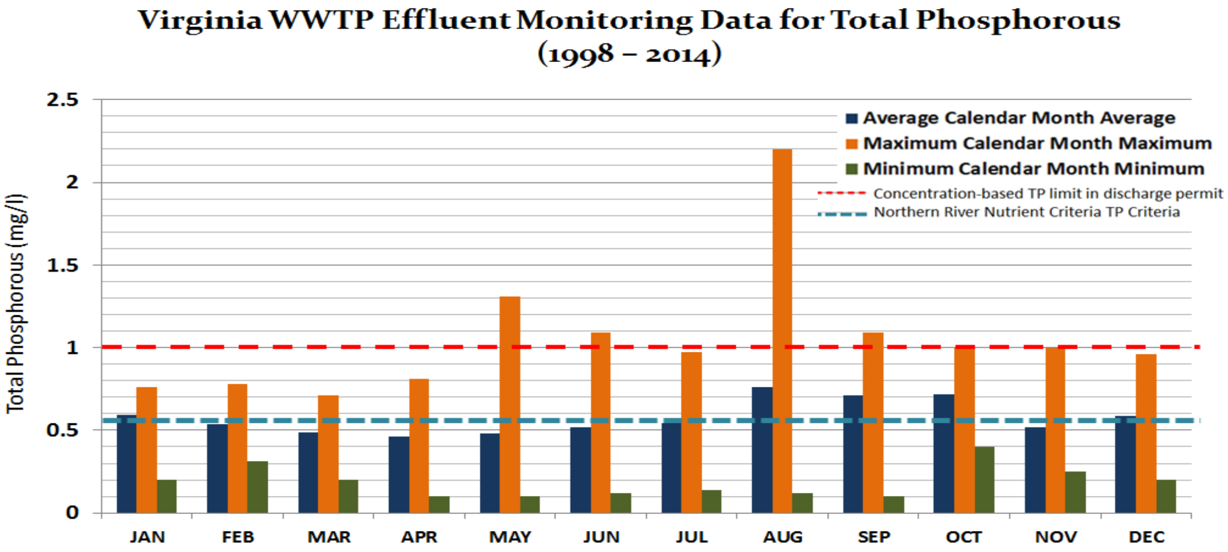


**Figure 261:** Total phosphorous concentrations of Virginia WWTP effluent discharged to Manganika Creek

## Biological Response to Low Dissolved Oxygen

The fish and macroinvertebrate biological integrity scores observed at Manganika Creek monitoring stations were some of the lowest in the entire SLRW. Biodiversity among fish and macroinvertebrate populations are extremely low in this stream, and the organisms present are tolerant of a wide variety of stressors, including low DO.

Fish species present below Manganika Lake (station 09LS078) included Central Mudminnow (n=8), Yellow Perch (n=4), and Brassy Minnow (n=1). Central Mudminnows are particularly tolerant of low DO conditions and have been known to survive in anoxic environments for extended periods of time (Klinger et al. 1982). The DO tolerance indicator value (DO TIV) score for the fish community at this station was well below all of the results observed at non-impaired stations of the same FIBI class in the SLRW (Figure 263), which is an indication that Manganika Creek lacks the diversity and sensitive fish species observed at locations with a more stable DO regime.

Central Mudminnow and Brook Stickleback were the only two fish species observed at the station upstream of Manganika Lake (98LS015). Both of these species are highly tolerant of low DO conditions. Fish community DO TIV results from this station were even lower than those observed at station 09LS078. Overall, the fish community of Manganika Creek both above and below the lake is symptomatic of a system that is stressed due to low DO concentrations.

The macroinvertebrate community of Manganika Creek was dominated by aquatic worms (*Oligochaeta)* and tolerant chironomid (midge) taxa, such as *Dicrotendipes, Glyptotendipes*, and *Chironomus*. Nearly all (95% and 99% over two sampling events) of the organisms sampled from station 98LS015 above Manganika Lake belonged to taxa that are tolerant of low DO conditions. Downstream of the lake at station 09LS078 the percentage of DO tolerant individuals was lower (41%), but this station still lacked taxa that are intolerant of low DO, and supported a much higher percentage of DO tolerant taxa than most comparable sites scoring above the IBI threshold. The DO TIV results further confirm the high level of tolerance exhibited in the macroinvertebrate community of Manganika Creek (Figure 263 and 264).

**Figure 262:** Photo of biological monitoring station 98LS015, located above Manganika Lake. This stream is channelized through this reach, and flows are stagnant and DO conditions are very marginal. All of the fish observed at this monitoring station were highly tolerant of low DO concentrations.

## Summary: Is dissolved oxygen a stressor in Manganika Creek?

Biological and water chemistry data from the Manganika Creek Watershed provide strong evidence to list low DO concentrations as a cause of both fish and macroinvertebrate impairments. Water chemistry data show prolonged periods of DO concentrations well below the warmwater DO standard of 5 mg/L. High DO flux may also be acting as a stressor to aquatic life, as diurnal flux exceeded 4.0 mg/L during continuous measurements collected below Manganika Lake. The fish and macroinvertebrate communities are severely degraded in terms of abundance and species diversity, and the vast majority of the species present are considered tolerant of low DO concentrations.

Unless conditions in Manganika Lake are drastically improved, the DO regime of Manganika Creek will remain unsuitable for sensitive aquatic life. The hypereutrophic conditions in Lake Manganika are likely to continue given the current nutrient loading from wastewater effluent and internal loading from Lake Manganika itself.

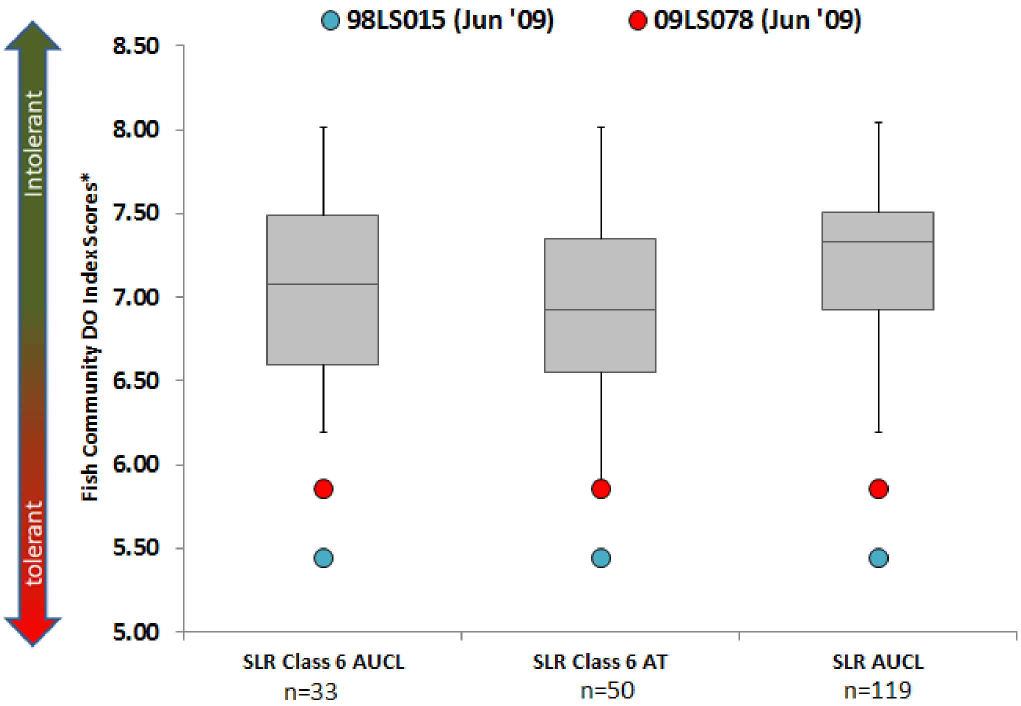


**Figure 263:** Fish community DO tolerance indicator value results for Manganika Creek compared to high quality reference streams of the same IBI class and greater SLRW. * See Section 4 for explanation of TIVs **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above FIBI Threshold

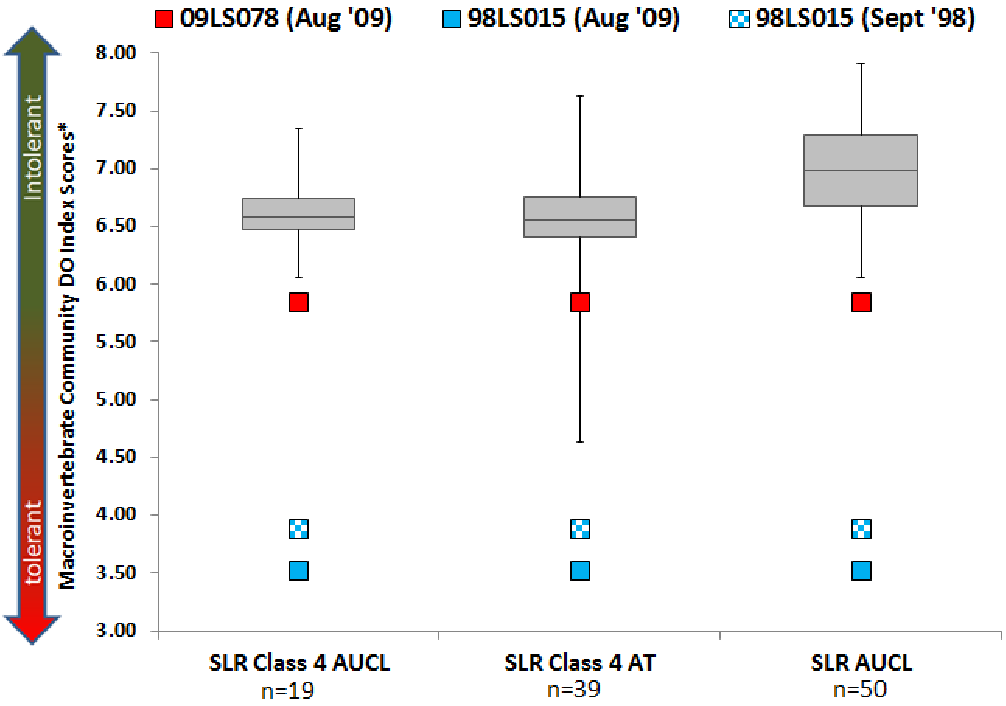


**Figure 264:** Macroinvertebrate community DO tolerance indicator value results for Manganika Creek compared to high quality reference streams of the same IBI class and greater SLRW. * See Section 4 for explanation of TIVs **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above FIBI Threshold

## 5.18.2 Specific conductivity

Elevated specific conductivity was selected as a candidate cause for impairment in Manganika Creek based on available water quality data, which shows conductivity levels frequently exceeding 1,000 µS/cm (Figure 265). Over 80% of the conductivity readings taken on Manganika Creek exceed 1,000 µS/cm. In February of 2014 during extreme low flow conditions, specific conductivity exceeded 2,000 µS/cm at biological monitoring station 09LS078 downstream of Manganika Lake. Data from November and February show that conductivity levels are in the range of 1,500 to 2,000 µS/cm for long durations during late fall and winter baseflow conditions.

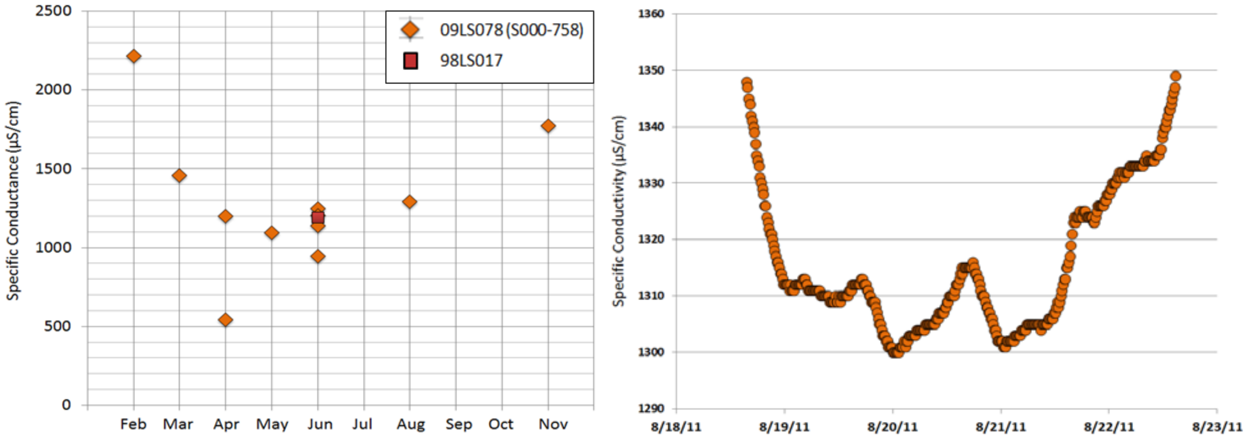


**Figure 265:** Point measurements of specific conductivity (left) and conductivity levels during a 5-day continuous monitoring period at biological monitoring station 09LS078

### Sources and Pathways of Specific conductivity

Elevated specific conductivity is often associated with mining land uses, discharge of sewage and industrial waste, and road salt application (EPA CADDIS 2012). The watershed of Manganika Creek contains very few road crossings, and elevated conductivity readings have been observed upstream of areas where road salt is applied most heavily (Minnesota State Highway 7). It is unlikely that road salt is a source of elevated specific conductivity in this watershed. Discharges from the Virginia WWTP and mining land uses within the watershed are the two most likely sources of elevated specific conductivity in Manganika Creek.

Effluent monitoring for specific conductivity has been performed by the Virginia WWTP since 2010, and is reported in calendar month maximum (CMM) values. The average CMM over this monitoring period is 1,304 µS/cm, and the highest value recorded is 1,700 µS/cm. Effluent monitoring data shows high concentrations of the anions bicarbonate ($HCO_3^-$), chloride ($Cl^-$), and sulfate ($SO_4^-$). Concentrations of the cations sodium ($Na^+$), calcium ($Ca^+$), and magnesium ($Mg^+$) are also high in the effluent being discharged to Manganika Creek. The limited flow data from this facility indicate an effluent discharge of around 1.8 cfs to Manganika Creek. Considering the small drainage area of this watershed, the WWTP plant effluent may account for a considerable portion of streamflow during low flow periods.

Samples were collected at the Manganika Lake outlet station (S000-758) during three distinct flow periods (baseflow, snowmelt, and rain event) and analyzed for major cations and anions. A similar set of samples were collected at stations across the SLRW, including Bear Creek, an undisturbed second-order tributary to the Embarrass River. Although Bear Creek may not be a perfect reference condition site to compare to Manganika Creek, data from this stream provides a general idea of the concentration of these major cations and anions observed at relatively undisturbed sites in this region of the Lake Superior drainage basin. Results from the two stations are compared in Figure 266. Manganika Creek has significantly higher concentrations of all cations and anions, particularly sulfate, chloride, magnesium, and sodium. Elevated concentrations of these compounds are contributing to unnaturally high specific conductivity of surface water in Manganika Creek compared to natural background conditions in the region.

Specific conductivity is elevated in Manganika Creek throughout the year, but particularly in those seasons where baseflow is the dominant flow regime. A lack of data exists for comparing conductivity levels above and below Manganika Lake, although at the time of fish sampling in June of 2009, the two stations had comparable specific conductivity (1,190 µS/cm upstream of the lake, and 1,130 µS/cm at the outlet monitoring site). Additional monitoring upstream and downstream of the lake would provide valuable information in terms of how the lake effects specific conductivity of surface water in this system.

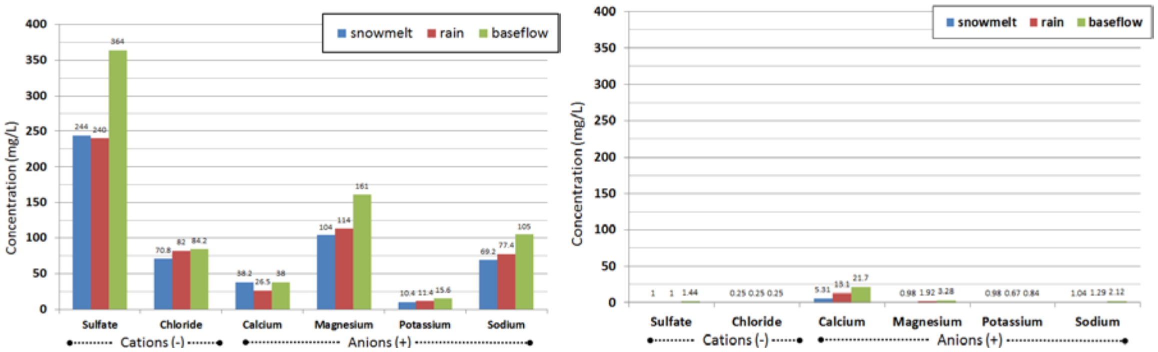


**Figure 266:** Comparison of major cations and anions under different flow conditions in Manganika Creek (left) and Bear Creek (right)

## Biological Response to Specific conductivity

The effects of elevated conductivity on aquatic life were evaluated using data from Minnesota streams and scientific literature. A summary of the biological responses that have been observed in the presence of high specific conductivity are presented in Section 3.1.5. Based on the literature that has been compiled, several biological metrics were selected to evaluate specific conductivity as a stressor in Manganika Creek (Table 75).

**Table 75:** Summary of biological metrics and literature used to evaluate elevated specific conductivity as a stressor

| Metric | Response to Increased Specific conductivity / Conductivity | Source |
|---|---|---|
| **EPT Richness** | Decrease | Roy et al (2003); Echols et al (2009); Johnson et al (2013) |
| **Overall Taxa Richness** | Decrease | Johnson et al (2013) |
| **Ephemeroptera Richness** | Decrease | Pond (2004); Hassel et al (2006) |
| **Fish and Macroinvertebrate Tolerance Indicator Values (TIV)** | Increase | MBDI (Yoder and Rankin, 2012) |

### EPT Richness

Several researches have observed reductions in EPT taxa richness in the presence of high specific conductivity (Roy et al 2003; Echols et al. 2009; Johnson et al 2013). For the purposes of this analysis, the metric EPTCh was selected, which groups all baetid mayfly taxa together as one taxon. In the case of Manganika Creek, this detail is unimportant, as the results from both sites are the same regardless of which EPT metric is used. EPT taxa were completely absent from station 98LS015 during both sampling events (’98 and ’09). A single EPT taxon, *Caenis*, was present in the sample collected at 09LS078 in 2009. *Caenis* is a relatively tolerant genus of mayflies, so essentially sensitive EPT taxa were absent from all monitoring stations on Manganika Creek. EPT richness in Manganika Creek is much lower in comparison to high quality biological monitoring stations from around the SLRW. The median number of EPT taxa observed at class 4 monitoring stations that are meeting the MIBI criteria is 10. No class 4 station scoring above the MIBI criteria had fewer than four EPT taxa present in the sample.

### Overall Taxa Richness

A reduction in overall taxa richness has been observed in response to elevated specific conductivity (Johnson et al. 2013). Macroinvertebrate taxa richness in Manganika Creek is significantly lower than observations from high quality stream reaches within the SLRW. Median taxa richness at other class 4 MIBI sites scoring above the impairment threshold was 44 taxa. In contrast, Manganika Creek monitoring stations supported only 6, 10, and 30 taxa over three sampling visits (Figure 267). Taxa richness is clearly lower in Manganika Creek compared to other SLRW streams, particularly the upper reaches above Manganika Lake. There are not enough conductivity data above and below the lake to determine if differences in specific conductivity can explain the disparity in taxa richness counts. In 2009, conductivity readings were collected above and below the lake during fish sampling efforts. Above the lake, specific conductance was 1,190 µS/cm, while below the lake conductivity was slightly lower, but comparable at 1,130 µS/cm.

Linking the lack of macroinvertebrate taxa richness exclusively to elevated specific conductivity is difficult considering the numerous stressors that can cause this symptom. Still, the biological response of decreasing taxa richness in the presence of high specific conductivity is consistent with observations from Manganika Creek monitoring stations.

### Ephemeroptera (Mayfly) Taxa Richness

Pond (2004) observed Ephemeroptera declined along a gradient of increasing conductivity. This trend can also be observed in statewide macroinvertebrate data from Minnesota monitoring stations (see Figure 267). Mayfly taxa were essentially absent from Manganika Creek, except for a very small population from the genus *Caenis*, which are considered to be tolerant of streams with elevated specific conductivity (Meador and Carlisile 2007). A comparison of Manganika Creek to comparable reference streams further supports the claim that this stream supports fewer mayfly taxa than healthy streams in the SLRW (Figure 267).

### Community Tolerance Indicator Values – Conductivity

Community level tolerance values for specific conductivity were calculated for Manganika Creek and compared to a set of high quality streams in the SLRW. The results, shown in Figure 268, indicate that both the fish and macroinvertebrate communities found in this stream are fairly tolerant of elevated specific conductivity levels. The macroinvertebrate community at station 09LS078 is dominated by taxa which are frequently found in streams with high specific conductance, as the community level TIV calculated for that site was well above the maximum values observed in the sites used in comparison.

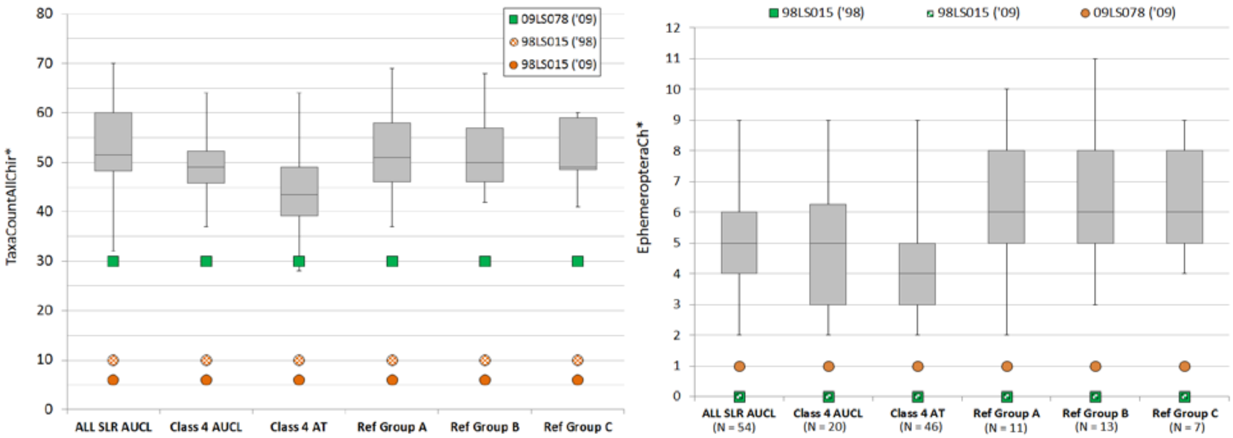


**Figure 267:** Overall taxa richness (left) and mayfly taxa richness (right) in Manganika Creek compared to high quality reference streams. * See Section 4 for explanation of TIVs **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above FIBI Threshold

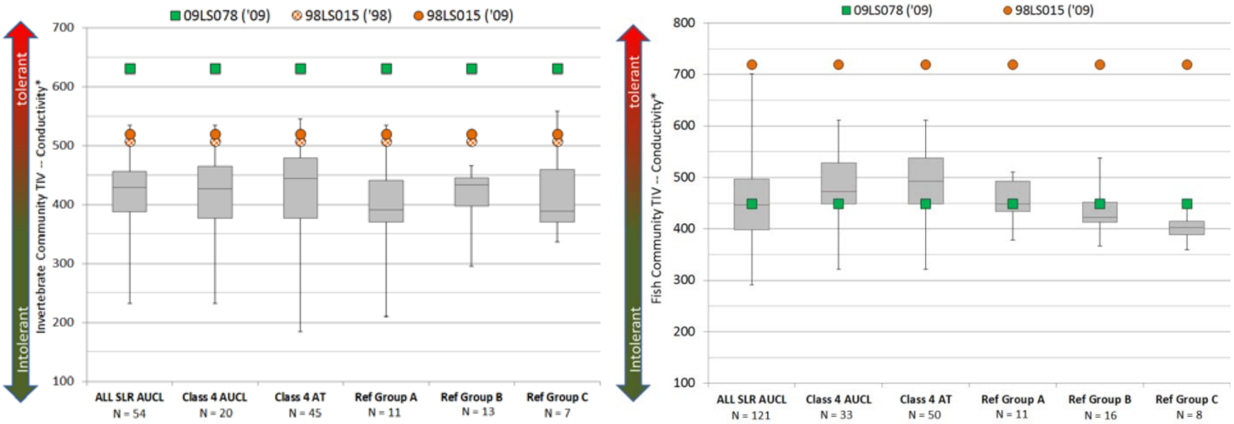


**Figure 268:** Fish and macroinvertebrate community TIV values for specific conductivity in Manganika Creek compared to high quality reference sites. * See Section 4 for explanation of TIVs **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above FIBI Threshold

### Summary: Is specific conductivity a stressor in Manganika Creek?

Specific conductivity levels in Manganika Creek have been shown to exceed 2,000 µS/cm, which is well above the suitable range for many sensitive fish and macroinvertebrate species. Conductivity is elevated during low flows, which suggests that the effects on biota are chronic in nature, and are not episodic. Several ionic compounds are found in high concentrations in Manganika Creek, including sulfate, chloride, magnesium, and chloride. With the exception of sulfate, none of these compounds are found in high enough concentrations to be considered independently toxic. However, it appears that these compounds (and potentially others) are cumulatively resulting in surface waters with an specific conductivity that can be considered a stressor to aquatic life. The potential for a toxic effect from sulfate are discussed further in Section 3.1.6.

The primary sources of this stressor appear to be effluent from the Virginia WWTP and mine pit dewatering from adjacent mine pits. These are all permitted discharges, and the operators are currently discharging within the guidelines established within their respective permits. Currently, specific conductance is not a parameter that is routinely measured in the effluent discharged to Manganika Lake via mine pit dewatering. The addition of this parameter to monitoring plans would provide valuable information in terms of understanding the relative contribution from those sources to this stressor.

### 5.18.3 Sulfate Toxicity

Elevated sulfate concentrations were identified as a candidate cause for impairment in Manganika Creek based on monitoring results that show concentrations well above natural background conditions for the SLRW. Based on seven samples collected between the years of 2011 and 2014, sulfate concentrations in Manganika Creek range from a low of 203 mg/L (6/25/2012) to a high of 597 mg/L (February 3, 2014). The average concentration over seven sampling events is 311 mg/L. Manganika Creek is one of several streams included in this SID study with sulfate concentrations that are significantly higher than streams with less impacted watersheds.

#### Sources and Pathways of Sulfate with the Manganika Creek Watershed

*A general discussion of sulfate sources and pathways in the SLRW can be found in* Section 3.1.6

Sulfate is a common compound generally found in low concentrations in pristine or lightly impacted streams. Sulfate concentrations averaged 3.9 mg/L (min = <1.0 mg/L; max = 15.0 mg/L) in a set of 61 samples from 10 high quality streams in the SLRW (refer to Table 2). In the northern regions of the SLRW where mining land uses are a prominent feature of the landscape, sulfate concentrations are generally higher. Higher sulfate concentrations are also seen along the entire main stem of the St. Louis River, particularly downstream of the Iron Range.

Manganika Creek receives mine pit dewatering inputs from a number of permitted sources, which likely contribute to the elevated sulfate concentrations observed in Manganika Creek. A pit from United Taconite's (UTAC) Thunderbird Mine, which sits adjacent to Manganika Lake to the east, sends pit dewatering flow of approximately 2.5 cfs into the lake. Just to the south, UTAC's Spruce/Nelson Leonidas mine also discharges pit water the lake sporadically, with up to 2.2 cfs allowed under the current permit. Plans are in place for UTAC to mine the area to the north of the current impacted area, and the new pit formed from this future mining effort will likely be hydraulically connected to the three large mine pits to the north. The formation of this "mega-pit" reservoir will likely have surface outflow, but the location is yet to be determined (Mike Crotteau, DNR, personal communication). The current and future surface and groundwater inputs from these mining areas are a very likely source of sulfate in the watershed. However, none of the current discharge permits require monitoring for sulfate, making it difficult to distinguish the significance of these sources from others in the watershed. Piles of waste rock and tailings placed on land adjacent to surface water are another source of sulfate in this watershed.

Effluent from the Virginia WWTP is additional source of sulfate in this watershed. The city of Virginia currently appropriates 4,000 gallons of water per minute (GPM) from the Missabe Mountain Pit Complex, which is north of the Manganika Creek Watershed boundary. However, nearly 1,400 to 3,000 GPM (35 - 75%) of this appropriation exits into the Manganika Creek Watershed via the WWTP. Since 2010, this WWTP has monitored the sulfate concentration of the effluent discharged to the stream. Calendar month maximums over this monitoring period range from 50 mg/L (October 2013) to 102 mg/L (November 2012), with an average of 73 mg/L over 54 total months of monitoring data. Limited flow data are available for this facility, but in August and September of 2014, calendar month average discharge was around 1.8 cubic feet per second.

#### Water Hardness and Chloride Concentrations

Chloride and hardness data are also important in the evaluation of sulfate as a stressor to aquatic life. Some research has shown that as chloride and hardness increase, the concentration at which sulfate can cause stress on aquatic life also increases. As a result, several states have incorporated chloride and hardness based criteria into their sulfate water quality standards.

Chloride and hardness data from Manganika Creek are relatively sparse, but the available results are useful for understanding the properties of surface water in this watershed and the role it plays in the toxicity of sulfate. The water in Manganika Creek can be considered hard to extremely hard, based on existing data. Spring and early summer results for water hardness are in the 450 – 540 mg/L range, and late summer to winter results range from 490 mg/L up to 1300 mg/L. Thus the hardness of the water in Manganika Creek may be buffering aquatic life from the elevated sulfate concentrations observed in this watershed. Chloride concentrations in Manganika Creek show very little seasonal variation. Over eight sampling events, concentrations ranged from 58 mg/L to 84 mg/L and it can be assumed that chloride concentrations consistently fall within or around these values. Independently, chloride and water hardness are not considered stressors, but instead will be important to consider when comparing sulfate values in Manganika Creek to water quality standards in other states that factor hardness and chloride into toxicity thresholds (see Table 9).

### Manganika Creek Sulfate Discussion

Minnesota does not currently enforce a water quality standard for sulfate that is specifically focused on protecting fish and aquatic macroinvertebrates. However, sulfate concentrations recorded in the lower reaches of Manganika Creek exceed several water quality criteria that are currently being implemented or drafted in other states and Canadian provinces. Below is a brief comparison of Manganika Creek sulfate data to some of the sulfate standards available elsewhere.

## British Columbia, Canada:

In a paper by Elphick et al (2010), various sulfate standards are proposed for British Columbia waters based on SSD and a SFA. Both of these standards are dependent on water hardness, as harder water can ameliorate the impact of sulfate on aquatic life. The sulfate standard proposed in British Columbia for very hard water (>160 mg/L) is 725 mg/L based on the SSD approach, and 675 mg/L based on the SFA approach.

To date, the maximum sulfate concentration observed in Manganika Creek is 597 mg/L. Thus, it would not be considered impaired for sulfate based on this water quality standard. The maximum sulfate concentration observed in Manganika Creek is fairly close to the SFA threshold, and additional sampling may show values that exceed this threshold.

### California

The state of California evaluated sulfate as a stressor to aquatic life in a 2013 study (Buchwalter 2013). The author concluded that there is enough toxicity data by EPA standards to support an acute toxicity criterion of 234 mg/L SO4 and a chronic criterion of 124 mg/L SO4. These values were not adjusted based on chloride and hardness values like other WQ standards for sulfate, and the author mentions uncertainties in the values stated above based on this detail. These values have not yet been adopted as water quality standards in California or any other state.

Sulfate levels in Manganika Creek exceed the 124 mg/L and 234 mg/L standards mentioned in this report with regularity. The only result falling below the 234 mg/L acute toxicity value is from a sample collected in June of 2012 following a historic flood event in Northeastern Minnesota.

### Illinois, Indiana, Pennsylvania, and Iowa

The states of Illinois, Indiana, Pennsylvania, and Iowa have been working towards an aquatic life standard for sulfate and other dissolved solids. Studies by Soucek and Kennedy (2004), Pennsylvania Department of Environmental Protection, and Iowa DNR (IDNR 2009) were compiled to develop the sulfate standard. The specifics of this sulfate standard are provided in Table 76. Unlike some of the sulfate criteria listed above, chloride and water hardness were taken into account in the development of a sulfate standard for these states. Based on the hardness and chloride data available for Manganika Creek, the applicable water quality standard for this stream would be range from 2,000 to 2,534 mg/L if similar guidelines were used for a sulfate standard in Minnesota. The highest sulfate concentration observed in Manganika Creek (597 mg/L) is well below the WQ standards being applied in these states.

### Biological Response to Sulfate

Sulfate toxicity is a complex issue and a number of factors may interact to determine the responses of various organisms to sulfate-dominated waters. A discussion of available biological response data to elevated sulfate levels is presented in Section 3.1.6 of this report. Based on that summary, the biological metrics listed in Table 76 below will be used to evaluate sulfate as a stressor in Manganika Creek. Additional consideration for sulfate as a stressor will be presented in the specific conductivity discussion for this stream, which can be found in Section 5.17.2.

Elevated sulfate concentrations are not considered a candidate cause for the fish impairment in Manganika Creek. The available toxicity data for fish indicates that they are generally quite tolerant of sulfate, therefore it is not considered to be contributing to the fish impairment.

**Table 76:** Summary of biological response metrics used to evaluate sulfate toxicity as a stressor

| Metric | Description | Relevance |
|---|---|---|
| **EPTCh** | Taxa richness of Ephemeroptera, Plecoptera & Trichoptera (baetid taxa treated as one taxon) | EPT macroinvertebrate taxa were limited to 10 or less at sites where sulfate concentrations exceeded 500 mg/L (Rankin 2003) |
| **EphemeropteraPct** | Relative abundance (%) of Ephemeroptera individuals in subsample | Sulfate and/or bicarbonate are the likely drivers of reduced macroinvertebrate diversity and abundance (particularly mayflies) in mining impacted streams in West Virginia (Buchwalter 2013) |
| **Intolerant Taxa** | Taxa richness of macroinvertebrates with tolerance values less than or equal to 2, using MN TVs | Rankin (2003) observed that many of the most sensitive taxa were not present in streams where sulfate concentrations exceeded 200 mg/L. |

## **Biological Response:** Low EPT Richness

Rankin (2003) observed that EPT macroinvertebrate taxa were limited to 10 or less at sites where sulfate concentrations exceeded 500 mg/L. For the purposes of this analysis, the metric EPTCh was selected, which groups all baetid mayfly taxa together as one taxon. In the case of Manganika Creek, this detail is unimportant, as the results from both sites are the same regardless of which EPT metric is used. EPT taxa were completely absent from station 98LS015 during both sampling events (’98 and ’09). A single EPT taxon, *Caenis*, was present in the sample collected at 09LS078 in 2009. *Caenis* is a relatively tolerant genus of mayflies, so essentially sensitive EPT taxa were absent from all monitoring stations on Manganika Creek. Figure 269 compares EPT richness in Manganika Creek to high quality biological monitoring stations from around the SLRW. The median number of EPT taxa observed at class 4 monitoring stations that are meeting the MIBI criteria is 10. No class 4 station scoring above the MIBI criteria had fewer than four EPT taxa present in the sample.

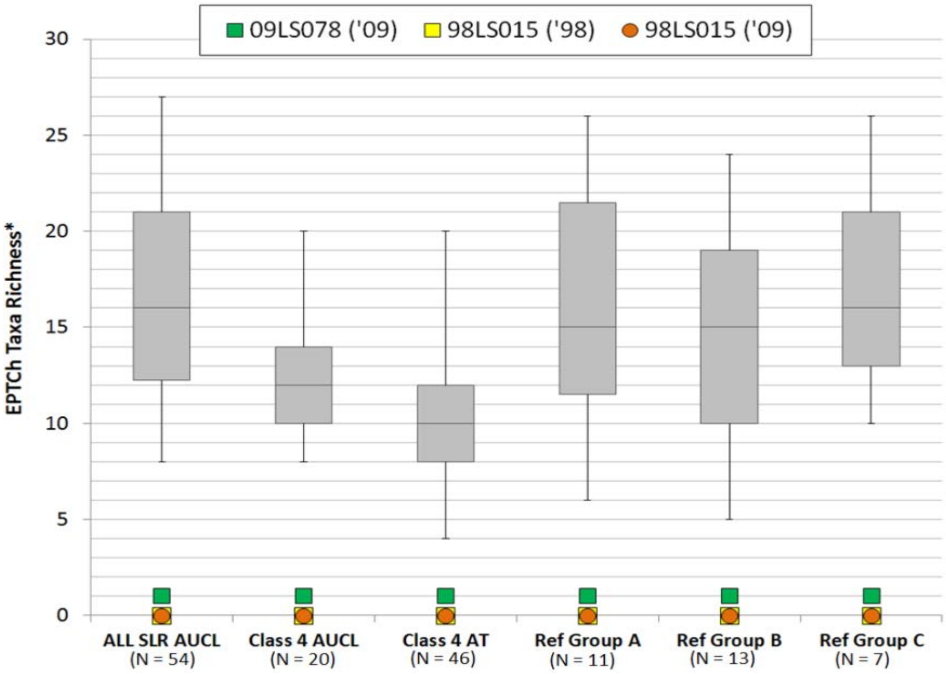


**Figure 269:** EPT taxa richness observed at Manganika Creek monitoring stations compared to high quality reference streams (see section 1.2.3 for list of reference streams). ***** See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

**Biological Response:** Lack of Intolerant Taxa

Rankin (2003) observed a decline or lack of sensitive macroinvertebrate taxa where sulfate concentrations exceeded 200 mg/L. The MPCA's biological metric *Intolerant2Ch* is used to quantify the taxa richness of macroinvertebrates that are considered sensitive or intolerant of disturbance in Minnesota streams. Taxa qualifying for this metric were completely absent from all Manganika Creek monitoring stations. These intolerant organisms respond negatively to a wide variety of stressors, so it is impossible to conclude that their absence from these stations is due to sulfate toxicity alone.

**Biological Response:** Low Ephemeroptera (Mayfly) Taxa Richness

Sulfate and/or bicarbonate are the likely drivers of reduced macroinvertebrate diversity and abundance (particularly mayflies) in mining impacted streams in West Virginia (Buchwalter 2013). Mayfly taxa were essentially absent from Manganika Creek, except for a very small population from the genus *Caenis*, which are considered to be tolerant of streams with elevated specific conductivity (Meador and Carlisile 2007). A comparison of Manganika Creek to comparable reference streams further supports the claim that this stream supports fewer mayfly taxa than healthy streams in the SLRW (Figure 267).

**Summary: Is sulfate toxicity a stressor in Manganika Creek?**

Sulfate concentrations in Manganika Creek are elevated well above natural background conditions for the SLRW, and exceed some of the WQ criteria that are being considered around the United States and Canada. However, current research showing that the combination of hard water and elevated chloride concentrations can reduce or eliminate the toxicity of sulfate must be considered for this watershed. The hardness of the water in Manganika Creek is likely reducing the potential for sulfate to be toxic to aquatic life in this specific watershed. Based on the water quality standards being applied in Iowa, Illinois, and Pennsylvania, sulfate concentrations in Manganika Creek are not high enough to be considered harmful to aquatic life.

Without a sulfate standard that can be applied specifically to Minnesota streams and rivers, it is difficult to eliminate or diagnose sulfate as a cause of impairment based on water quality data alone. The biological data shows consistencies with other sulfate impacted streams in Minnesota and in other states, but it is very difficult to separate the effects of sulfate from other confounding variables.

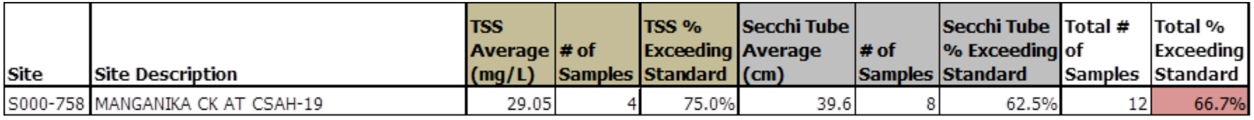
The high sulfate concentrations in Manganika Lake are likely contributing to other diagnosed and potential stressors, including eutrophication/low DO and specific conductivity. Efforts to improve water quality in the lake and outlet stream must focus on sulfate reduction among other constituents.

### 5.18.4 Total Suspended Solids / Turbidity

Current data for TSS and Secchi tube transparency (s-tube) are summarized in Table 77. In all, a total of 12 observations (4 TSS samples, 8 s-tube observations) from one sampling site (biological monitoring station 09LS078) were collected over a three year period from 2011 to 2013. Despite the relatively low number of observations, the monitoring results make a strong case for elevated TSS as a candidate cause of impairment. Results show 75% of TSS samples and over 60% of s-tube measurements in violation of WQ standards for protecting aquatic life.

Box plots of the TSS and Secchi tube datasets for Manganika Creek were compared to the datasets from the "A" and "B" reference streams (see Section 1.2.3) in the SLRW. The TSS data for Manganika regularly exceeded the TSS standard of 15 mg/L, although the sample size is quite small. The data for the “A” and “B” reference streams are well below the draft standards for both TSS and Secchi tube, indicating that low levels of suspended solids are closely linked with healthy biologic communities in the SLRW. However, a closer look at the biological metrics of Manganika Creek is needed to determine whether TSS can be considered a stressor in this system.

**Table 77:** TSS and Secchi tube average values and percent standard exceedances for Manganika Creek



| Site | Site Description | TSS Average (mg/L) | # of Samples | TSS % Exceeding Standard | Secchi Tube Average (cm) | # of Samples | Secchi Tube % Exceeding Standard | Total # of Samples | Total % Exceeding Standard |
|---|---|---|---|---|---|---|---|---|---|
| S000-758 | MANGANIKA CK AT CSAH-19 | 29.05 | 4 | 75.0% | 39.6 | 8 | 62.5% | 12 | 66.7% |

### Sources and pathways of suspended solids in the Manganika Creek Watershed

Algal Blooms in Manganika Lake

The impaired reach originates from Manganika Lake, which is highly eutrophic and listed as impaired for high nutrient concentrations. Significant algal blooms are common in Manganika Lake throughout much of the year (see Figure 270), and the impaired reach of Manganika Creek closely mirrors the condition of its source. The majority of the TSS in Manganika Creek are volatile solids (algae) flowing out of Manganika Lake. Very little suspended sediment has been observed within the impaired reach, and there are no visible signs of streambank instability or other sediment sources downstream of the lake.

### Biological effects of elevated TSS

#### Fish Response to TSS

The FIBI impairment on Manganika Creek is the result of poor metrics related to low fish counts and a lack of species that are expected in healthy streams. Two species were collected at 98LS015 in 2009 – Brook Stickleback (n=50) and Central Mudminnow (n=5). Both species are considered “neutral” in regards to elevated TSS. These two species are commonly found in headwaters streams with wetland influences, which often have lower TSS concentrations. However, they are known to be tolerant of a wide range of water quality and habitat conditions. Only 13 total fish were sampled in a June 2009 visit to the lower biological monitoring station, 09LS078. The majority of individuals sampled (92%) are neutral related to elevated levels of TSS. The dominant taxa, Central Mudminnow and Yellow Perch, are considered neither tolerant nor intolerant based on the MPCA’s TIVs. One “moderately tolerant” Brassy Minnow was also collected at this station.

Considered alone, it is difficult to determine whether the fish community in Manganika Creek is generally tolerant or intolerant of TSS due to the overwhelming numbers of “neutral” individuals. A look at community level TIV results for both Manganika sites reveals a somewhat dichotomous assemblage. Figure X compares the TSS TIVs for the two sampling sites with: 1) all the Class 6 streams in the SLRW that were above the UCL, 2) the Class 6 streams in the SLRW that were above the impairment threshold, and 3) streams of all types in the SLRW that were above the UCL. As can be seen from the graph, the fish assemblage at the impaired biological monitoring station (09LS078) is less tolerant of TSS than the median for healthy Northern Headwater streams, and slightly worse than the median for all of the SLRW AUCL streams. Conversely, the monitoring site upstream of Manganika Lake (98LS015) scored much worse and was well above the 75th percentile tolerance values of the three healthy groupings.

The lack of taxa richness and low fish counts at this station are probably a more reliable symptom of stress related to TSS than the individual tolerances of these species. The significant algae blooms seen in this stream reach reduce visibility and create difficult conditions for supporting a diverse and robust population of fish.

### Invert Response to TSS

Macroinvertebrate taxa in Manganika Creek were categorized into four classes based on their tolerance to elevated levels of TSS. These were: 1) highly tolerant, 2) moderately tolerant, 3) moderately intolerant, and 4) highly intolerant. Only two taxa were present in a 1998 sampling of the 98LS015 biological monitoring station. One is considered “moderately tolerant” and the other “moderately intolerant.” Due to the low number of taxa and the age of this data, more weight should be given to the 2009 sampling of the impaired biological monitoring station at 09LS078. A total of 17 invertebrate taxa were collected at 09LS078. They are all considered tolerant of elevated levels of TSS (10 “moderately tolerant” and 7 “highly tolerant”).

TSS index scores are clearly showing that the invertebrate assemblage of Manganika Creek is more tolerant of TSS in comparison to high quality stations of the same MIBI class. The box plots in Figure 272 and 273 compare data for a series of TSS-related metrics between Manganika Creek monitoring stations and (1) Class 4 stations that scored above the UCL of the MIBI threshold, (2) Class 4 stations that scored above the impairment threshold, and (3) all SLRW stations that scored above the UCL of the MIBI threshold. The 1998 sample shows a lower tolerance to TSS than all of the healthy streams, but that is mainly due to the high percentage of intolerant taxa (50% - 1 out of 2). The invertebrate assemblage at 09LS078 is extremely tolerant to TSS when compared to healthy SLRW streams (Figure 273).

The lack of a precise mechanism for TSS to negatively affect macroinvertebrates weakens the case for this candidate stressor. The elevated TSS concentrations observed in Manganika Creek are driven by volatile solids (algae blooms) as opposed to suspended inorganic particles (sand/silt). The major pathways for TSS to harm to invertebrates are through abrasion, clogging of filtering mechanisms, and habitat degradation from suspended sediments settling out on the stream bottom. As a result, it is difficult to directly link the poor MIBI scores to the elevated TSS concentrations in this stream. There are obvious connections between the algae blooms, elevated TSS concentrations, and poor DO conditions that impact both fish and macroinvertebrate populations.

Elevated TSS concentrations are probably not a direct cause of the macroinvertebrate impairment in this stream. However, TSS is likely a cause of poor FIBI scores, and can be linked to other stressors (esp. low DO) that contribute to poor MIBI results.

**Figure 270:** Algal bloom on Manganika Lake

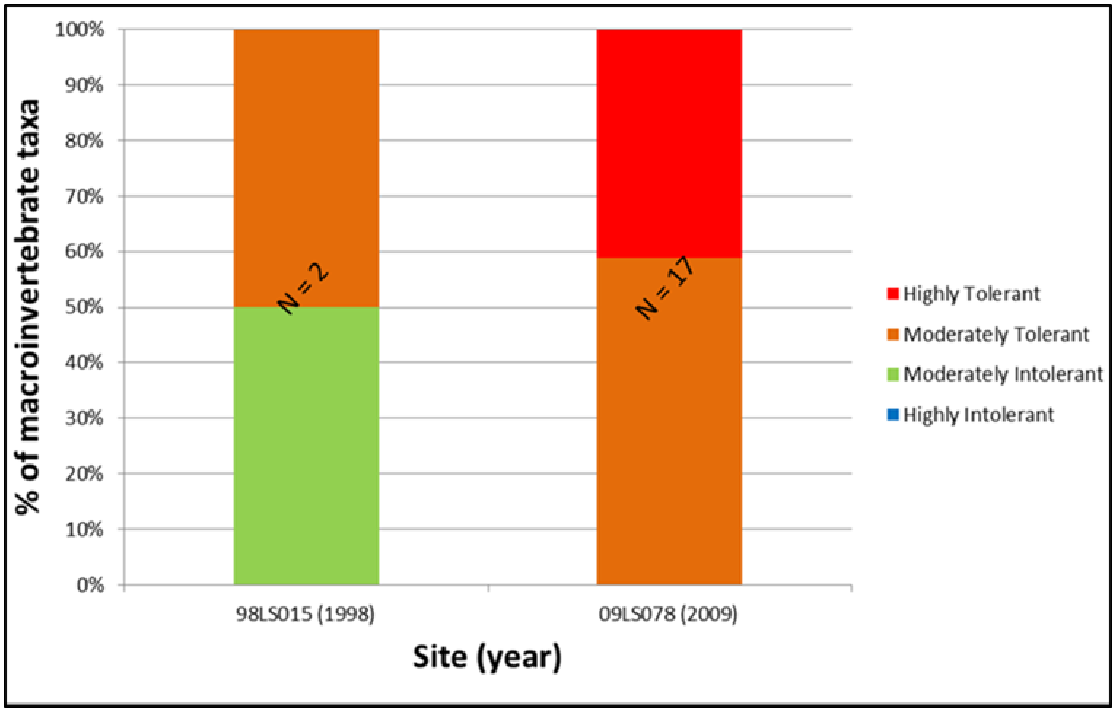


**Figure 271**: Proportion of macroinvertebrate assemblage in Manganika Creek that are considered tolerant, neutral, or intolerant of low DO concentrations based on MPCA tolerance values.

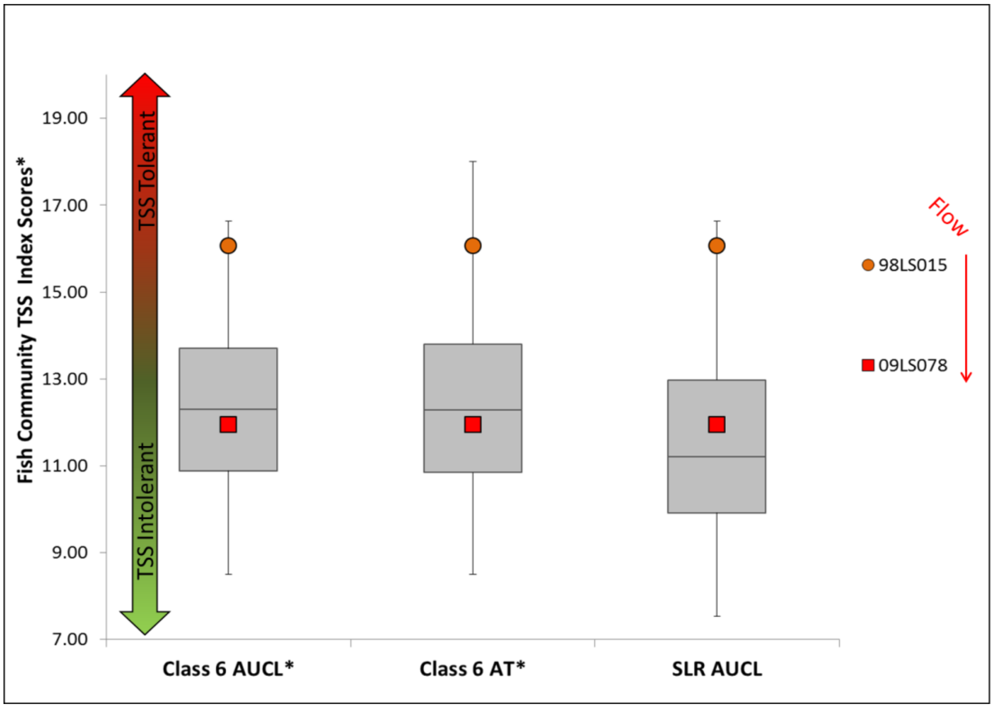


**Figure 272**: Fish community TSS TIV results for Skunk Creek station 09LS031 compared to results from high quality stations of the same IBI class. * See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

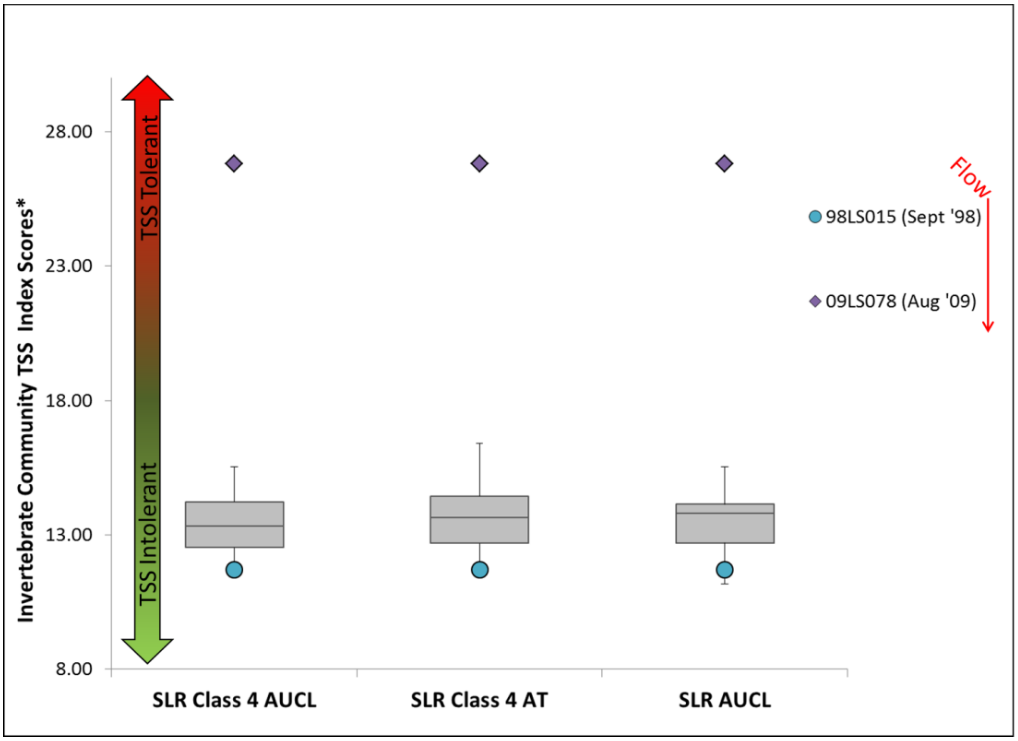


**Figure 273:** Invertebrate community TSS TIV results compared to results from high quality stations of the same IBI class. * See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

**Summary: Are elevated TSS concentrations a stressor in Manganika Creek?**

Water chemistry and biological data provide adequate evidence to diagnose elevated TSS and turbidity as a stressor to aquatic life in Manganika Creek. Efforts to improve the conditions of Manganika Lake will have the added benefit of providing better water quality for the fish and macroinvertebrate populations in this impaired stream.

### 5.18.5 Elevated pH

Elevated pH was identified as a candidate cause for impairment in the Manganika Creek Watershed based on available monitoring data, which show frequent violations of the state water quality standard. Extreme pH values much below 5.0 or above 9.0 are harmful to most organisms. Effects on biota include decreased growth and reproduction, decreased biodiversity, and damage to skin, gills, eyes, and organs. Concentrations of nutrients (especially nitrogen) also play a significant part in pH dynamics, as nitrification and respiration both produce hydrogen ions. For additional background information on this stressors, refer to Section 3.1.4.

Point measurements of pH collected from Manganika Creek are shown in Figure 274. Violations of the pH standard occur most frequently between the months of June and September, when algae blooms on Manganika Lake are likely at their annual peak. All six pH observations from station S000-761 are between 7.0 and 8.0, which is comparable to minimally impacted streams within the SLRW. This station is located upstream of the Virginia WWTP discharge point and Manganika Lake, both of which are potentially linked to elevated pH levels in the lower portions of the watershed. Based on these data, it appears that the upper reaches of Manganika Creek (above the lake) have a fairly comparable pH regime to other streams of the SLRW.

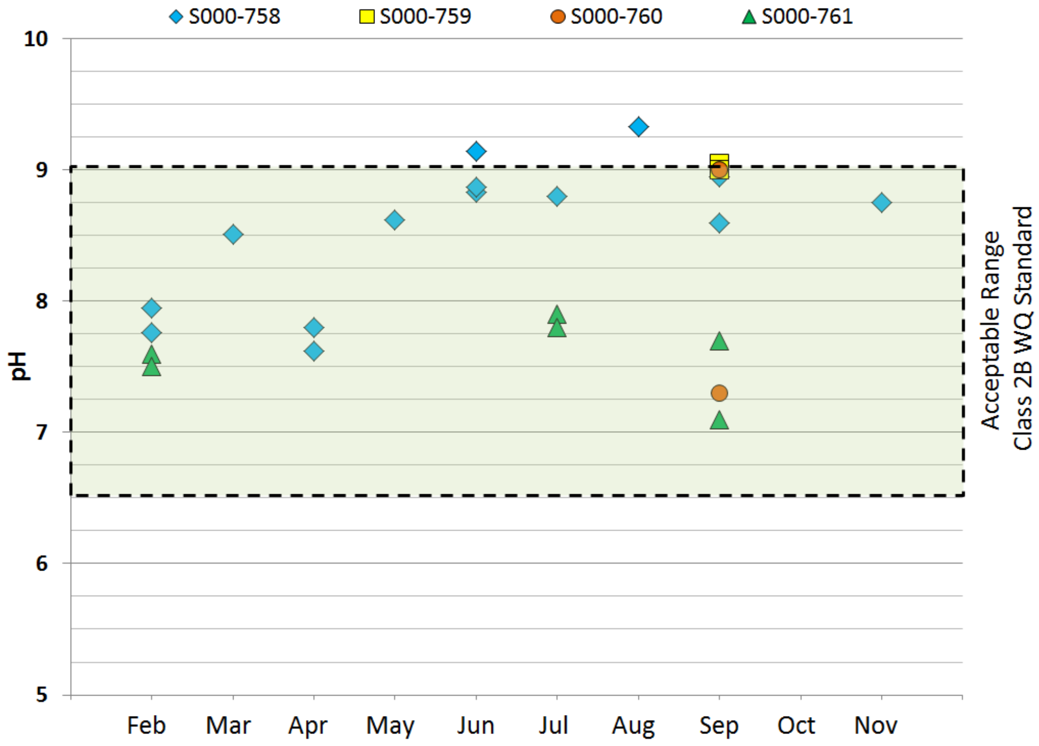


**Figure 274:** pH data from Manganika Creek arranged by calendar month and monitoring stations.

## Sources and Pathways of Elevated pH

The Virginia WWTP and eutrophic conditions of Manganika Lake were identified as potential sources linked to elevated pH regime in the impaired reach of Manganika Creek. The pH values of the effluent leaving the Virginia WWTP have been monitored since 1998. The results show that on average, the effluent discharged from the WWTP has a pH around 7.5, although higher values around 9.0 (December 2010) and up to 9.6 (October 2012) have been observed in rare instances. Based on these monitoring data, it appears that the Virginia WWTP typically meets the pH limit of 9.0 that is required by the NPDES permit issued to the facility. The pH data set for Manganika Creek is insufficient for a thorough comparison of conditions upstream and downstream of the WWTP, but effluent from this facility likely increases the pH of the receiving water when effluent values are greater than 7.6.

Manganika Lake is one of the most degraded lakes in all of northern Minnesota and significant algae blooms occur on this lake throughout the year. High rates of photosynthesis where algae and macrophytes are overly abundant can remove carbon dioxide from the water, which results in a higher pH value. Water chemistry results collected at the time of biological sampling in June of 2009 show the effect that the lake has on pH levels. Upstream of the lake at station 98LS015, a pH of 7.71 was observed, while downstream of the lake, pH increased to 8.83. Elevated phosphorous concentrations (exceeding 0.200 mg/L) are commonly observed in both the lake and stream, which drive the eutrophic conditions in these waters, and ultimately contribute to a pH regime that is outside of the normal range observed in the SLRW.

## Biological Response to pH Regime Alteration

The effects of high pH are usually not specific enough to be considered symptomatic, yet they can be seen in broad level community changes or damage to specific organs in fish (EPA CADDIS 2013). Many of the effects of elevated pH often cited are similar to, or linked with symptoms that occur in the presence of related stressors (e.g. ammonia toxicity), such as decreased growth, mortality, reduced number of species and individuals. These same symptoms were evaluated for ammonia toxicity as a cause of impairment, and in general, the evidence from the biota of Manganika Creek is supportive of either pH or ammonia toxicity as a stressor, or both. Refer to Section 3.1.4. for a summary of these symptoms.

### Summary: Is elevated pH a stressor in Manganika Creek?

The pH levels observed in Manganika Creek are elevated above background conditions due to effluent from the Virginia WWTP and significant algae blooms in Manganika Lake. Exceedances of upper limit of the pH standard (9.0) have occurred during the summer and early fall months, but are observed rather infrequently based on the somewhat limited data available. Additional monitoring during this period would provide useful information in terms of the frequency and duration of pH standard violations. Several symptoms observed in the biota of Manganika Creek are typical of streams with elevated pH -- low fish counts, decreased reproduction, and reduced biodiversity. Based on all of the evidence from this watershed, elevated pH cannot be ruled out as a cause of biological impairment, but additional water chemistry data showing regular observations above 9.0 are recommended to diagnose pH as a stressor.

## 5.18.6 Ammonia Toxicity

Elevated concentrations of unionized ammonia (ammonia) were identified as a candidate cause for impairment in the Manganika Creek Watershed after a review of existing monitoring data and pollution sources within the watershed. Results of samples collected between the years of 1980 and 2014 show frequent violations of the 40 µg/L water quality standard at locations both upstream and downstream of Manganika Lake (Table 78). Concentrations as high as 335.2 µg/L, or eight times over the WQ standard, have been observed upstream of Manganika Lake near biological monitoring station 98LS015. Below the lake at biological monitoring station 09LS078 (co-located with WQ station S000-758), ammonia concentrations are somewhat lower, but have still been observed at over two times the WQ standard.

**Table 78:** Summary of ammonia nitrogen data and other water quality variables that are used to calculate the toxic form (unionized) of ammonia. Values in **red** are in violation of the class 2B (warmwater streams) water quality standard.

| Station | SAMPLE DATE | Water Temp (°C) | pH | Ammonia N (mg/L) | Unionized Ammonia (ug/L) |
|---|---|---|---|---|---|
| S000-759 | 9/18/1980 | 12.00 | 9.00 | 1.88 | 335.2 |
| 98LS015 | 7/8/1998 | 18.0 | 7.60 | 9.54 | 128.1 |
| S000-758 | 8/1/2012 | 26.81 | 9.33 | 0.17 | 98.5 |
| S000-758 | 11/30/2011 | 0.20 | 8.75 | 2.00 | 90.0 |
| 98LS015 | 6/24/2009 | 17.3 | 7.71 | 5.00 | 82.0 |
| S000-758 | 9/18/1980 | 11.00 | 8.60 | 0.78 | 57.7 |
| 09LS078 | 6/25/2009 | 24.6 | 8.83 | 0.20 | 53.0 |
| S000-758 | 5/30/2012 | 13.75 | 8.62 | 0.34 | 31.9 |
| S000-758 | 6/25/2012 | 25.95 | 9.14 | 0.05 | 22.8 |
| S000-758 | 6/23/2011 | 14.32 | 8.83 | 0.11 | 16.4 |
| S000-758 | 4/13/2011 | 6.93 | 7.80 | 1.60 | 14.6 |
| S000-758 | 2/3/2014 | 0.00 | 7.80 | 0.75 | 3.9 |

### Sources and Pathways of Ammonia

Although there are several permitted dischargers of effluent wastewater in its watershed (mine pit dewatering, WWTP), the effluent from the Virginia WWTP has historically been the primary source of ammonia nitrogen inputs to Manganika Creek. Effluent monitoring for ammonia nitrogen was initiated at the Virginia WWTP in 2010 as part of the ammonia limit required by the facility's NPDES permit reissuance. Results of this monitoring effort show elevated concentrations of ammonia being discharged from 2010 until 2012, and drastic reductions in ammonia thereafter once treatment systems were improved (Figure 275)

There are not enough monitoring data from the years 2013 and 2014 to effectively track how the ammonia reductions are changing water quality in Manganika Creek. Considering that the basis of the biological impairment listing are 2009 biological monitoring data and the WWTP upgrades occurred in 2012, the post 2012 data do not factor into the causal analysis for this impairment. Given the treatment changes made in the WWTP facility, follow-up monitoring in Manganika Creek is warranted to determine whether or not the ammonia reductions are improving water quality and biological integrity downstream.

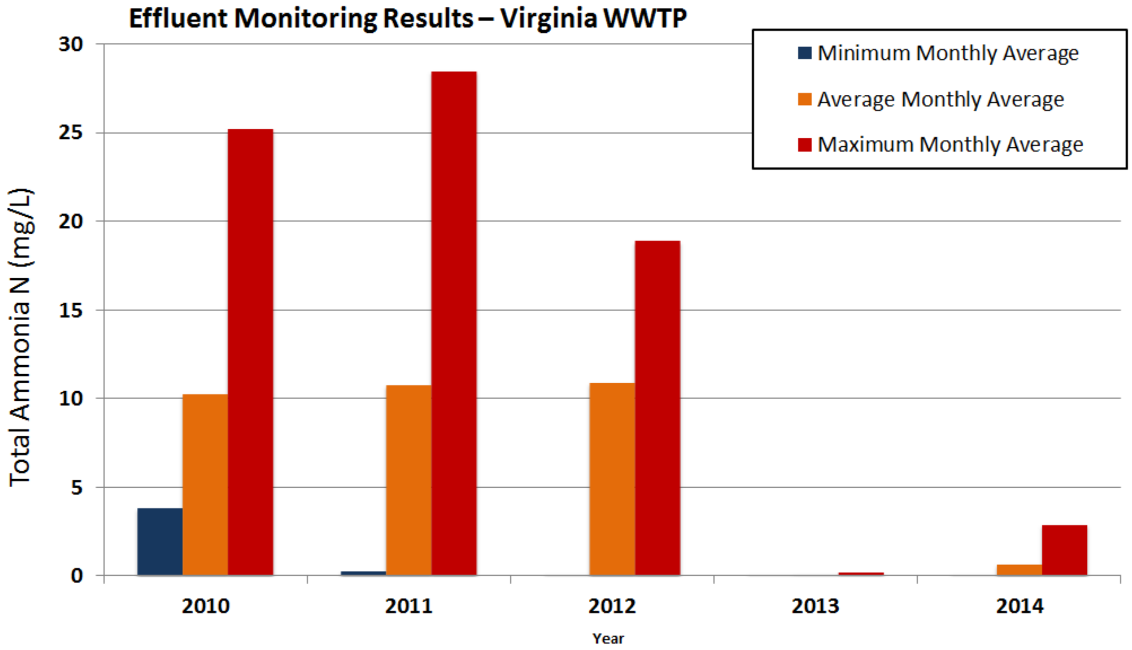


**Figure 275:** Effluent monitoring results for total ammonia discharged from the Virginia WWTP to Manganika Creek

## Influence of Manganika Lake

Manganika Lake is located between the WWTP discharge and the lower reach of Manganika Creek and likely plays a significant role in the fate and transport of nitrogen in the watershed. The 158 acre lake is currently listed as impaired for excess nutrients and it is considered to be one of the most severely impaired lakes in Northeastern Minnesota due to a long history of pollution from wastewater and other sources (e.g. mining, urbanization).

The eutrophic nature of this lake results in algae blooms that persist for most of the year. Algae and macrophytes are known to take up ammonia, and can thereby reduce concentrations in the water column. In late June of 2009, biological and water chemistry sampling were conducted at sites above and below Manganika Lake. Unionized ammonia concentrations were 35% higher at the location above the lake, although both locations exceeded the 40 µg/L WQ standard (Figure 276). FIBI and MIBI scores were very poor at both locations, although the macroinvertebrate community below the lake was slightly improved compared to the location upstream of the lake. The uptake of ammonia in Manganika Lake is a likely scenario in this watershed, but additional data would be required to further evaluate whether or not this is occurring and to what extent.

## Influence of pH and Water Temperature

Ammonia is more toxic to aquatic life at higher temperature and pH values. As pH increases, so does the fraction of unionized ammonia and the toxicity to fish (EPA 1999). The ratio of NH3 to NH4+ increases by 10 times for each one-unit rise in pH, and by approximately 2 times for each 10 degree rise in temperature from 0 - 30° C (EPA 2009). When pH exceeds 9.5, it can become difficult or impossible for fish to excrete $NH_3$ from their systems, which can lead to mortality from autointoxication. Temperature and pH regime of surface water in the Manganika Lake/Creek complex are very suitable for the formation of the toxic unionized form of ammonia. Observations of pH in Manganika Creek are extremely high and frequently exceed the state water quality standard of 9.0. Water temperatures in

Manganika Creek are some of the highest of all the streams in the SLRW, approaching 28 °C (80° F) during the summer months. Elevated water temperatures and pH in Manganika Lake/Creek provide a key pathway in the formation of unionized ammonia in this watershed.

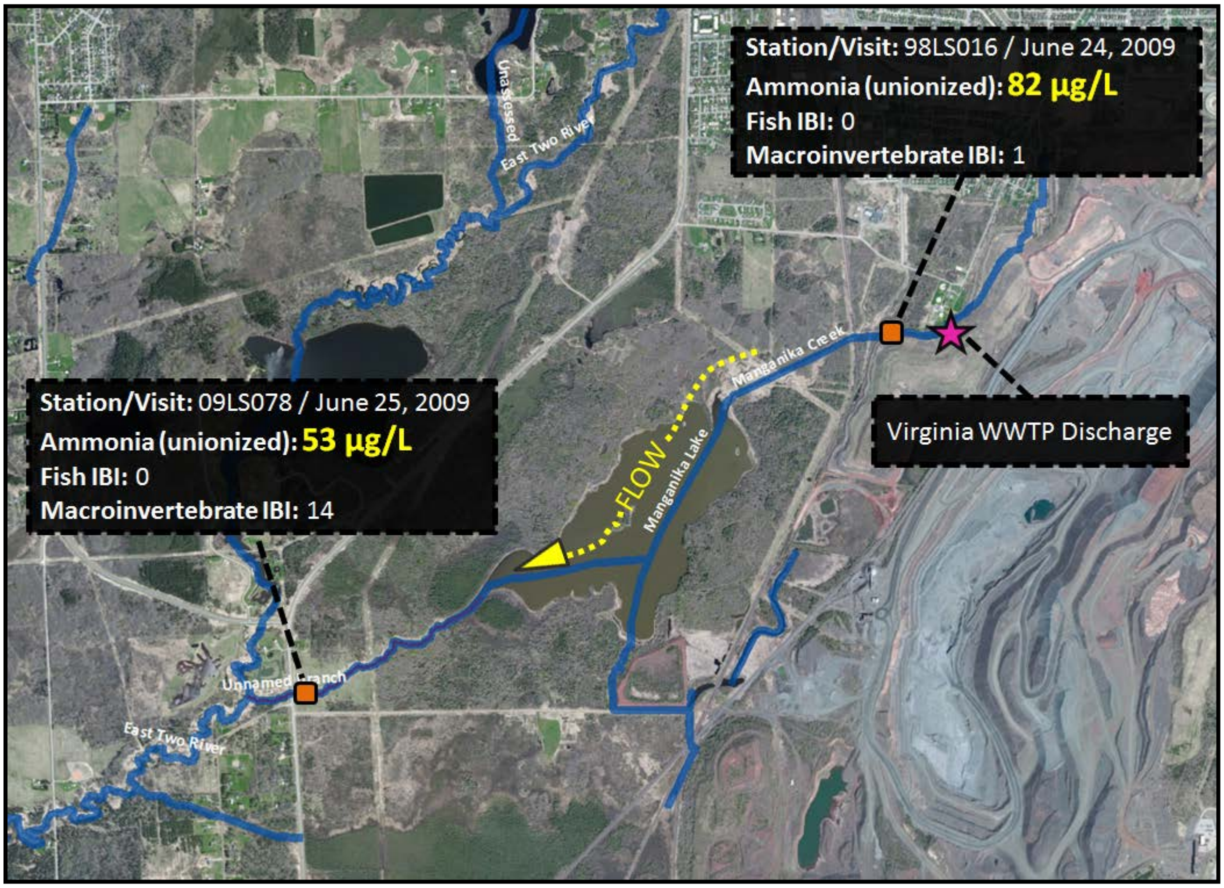


**Figure 276:** Eutrophic conditions in Manganika Lake may be resulting in reduced ammonia concentrations downstream.

### Sediments and Dissolved Oxygen

The accumulation of fine sediments (silt/clay) in Manganika Creek and Manganika Lake are another possible source of ammonia. An abundance of fine sediments have been observed within the biological sampling reaches that were visited. Ammonia in sediments typically results from bacterial decomposition of natural and anthropogenic organic matter that accumulates in sediment. Ammonia is especially prevalent in anoxic sediments because nitrification (the oxidation of ammonia to nitrite [$NO_2^-$] and nitrate [$NO_3^-$]) is inhibited (EPA CADDIS). Oxygen levels in the hypolimnion of Manganika Lake are frequently anoxic, and very low DO concentrations have been observed in the lake outlet. Ammonia generated in sediment may be toxic to benthic or surface water biota (Lapota et al. 2000).

### Effects of Hardness on Ammonia Availability

Ammonia toxicity is also a function of specific conductivity, or the salinity of the water. Several studies have indicated that increasing the hardness of ambient water can in some cases decrease ammonia toxicity (Wicks et al. 2002; Soderberg and Meade 1992). The surface water in Manganika Creek is extremely hard compared to the majority of SLRW streams, exceeding 1,300 mg/L during a February 2014 sampling event. Water hardness is not factored into the current state water quality standard for ammonia. However, it is possible that the negative effects of ammonia are reduced by the extremely hard water found in this drainage.

#### Duration of Exposure

The toxicity of ammonia to fish is impacted by both duration and frequency of exposure. Duration of exposure to ammonia has been demonstrated to be a critical factor affecting fish survival, and should therefore be incorporated into chronic and acute standards. As such, a single or occasional measurement of ammonia concentration is not sufficient to evaluate and regulate ammonia contamination. The MPCA's ammonia standard requires "no more than one exceedance of the chronic water quality standard in any 3-year interval," and the CS is based on a "4-day average" ammonia concentration. The current data set is not sufficient to calculate a running 4-day average in Manganika Creek. Prior to 2013, effluent containing elevated concentrations of ammonia was being discharged continuously from the Virginia WWTP, and most likely resulted in long duration, high frequency exposures of aquatic life to harmful levels of ammonia. Frequency and duration of exposure has likely decreased significantly since 2013, when changes to the treatment system at the WWTP resulted in much lower ammonia concentrations in the effluent stream.

### Biological Effects of Ammonia Toxicity

The most common effects of ammonia on fish include impacts to the central nervous system, increased breathing, cardiac output, oxygen uptake, convulsions, coma, and death (EPA 2013). Lower concentrations of ammonia can cause a reduction in hatching success, reduced growth rate and morphological development, and pathologic changes in the tissues of gills, livers, and kidneys (EPA 2013). For a summary of this stressor and specific impacts to aquatic life, see Section 3.1.10. The MPCA biological sampling protocols include an assessment of external fish condition (discussed in the "DELT" section later), but do not include an evaluation of internal organs and physiological condition. Therefore, it is difficult to determine whether or not the typical symptoms of ammonia stress are occurring within the fish community of Manganika Creek. The biological response metrics will be evaluated using data from Manganika Creek to present evidence for or against ammonia as a stressor.

#### Abundance, Taxa Richness, and Age Structure

Elevated concentration of unionized ammonia have been linked to fish kills in agricultural or urban watersheds, as well as in streams located downstream of a point source discharge. No fish kills have been reported in Manganika Creek. However, fish abundance and taxa richness were low at both monitoring stations, with only three taxa observed at station 09LS078 (Brassy Minnow, Central Mudminnow, and Yellow Perch) and only 13 individuals were collected during sampling. The catch rate at this station was only 2.7 fish per meter of stream sampled, which resulted in a very poor score (0.17 out of 10) in the fish abundance metric (NumPerMeter) used to calculate the overall IBI score. Above Manganika Lake at station 98LS015, the fish community consisted of only two taxa, Central Mudminnow and Brook Stickleback, both of which are very tolerant to a variety of water chemistry stressors.

All of the fish species observed in Manganika Creek reach sexual maturity before the age of one year, with the exception of a single Yellow Perch sampled at 09LS078. A fish community dominated by species with this trait can be an indicator of high annual mortality. It is difficult to directly link this symptom of stress to ammonia toxicity without direct observation of a fish kill and corresponding water chemistry data. However, the low taxa richness and fish abundance measures observed in the creek are symptomatic of a system where fish kills are observed and conditions are unfavorable for supporting fish that reach sexual maturity at a later age (> 1 year).

Macroinvertebrate taxa richness is also very low in Manganika Creek compared to other SLRW streams. An average of only eight taxa were sampled at the biological monitoring station upstream of Manganika Lake (98LS015). Below the lake, taxa richness was much higher at 30 taxa, but remained well below the median taxa richness values observed at high quality monitoring stations in the SLRW. Sensitive and pollution intolerant fish and macroinvertebrate taxa are completely absent from this watershed. Approximately 97% of the macroinvertebrates sampled are considered tolerant to pollution or degraded stream conditions.

Without annual or routine monitoring of these sites, it’s difficult to determine whether or not fish kills occur regularly on Manganika Creek. Based on the chemical and biological data, ammonia concentrations are, at a minimum, spatially correlated with reduced fish and macroinvertebrate abundance, low taxa richness, and a community dominated by short-lived, fast-maturing fish species. All of these symptoms provide evidence in support of ammonia toxicity as a stressor.

### Deformities, Eroded Fins, Lesions, Tumors (DELT)

Elevated concentrations of ammonia have been associated with anomalies in fish appearance and condition (Cormier et al. 2000). The MPCA biological monitoring protocols call for the evaluation of all fish captured for deformities (D), eroded fins (E), lesions (L), and tumors (T), collectively referred to as “DELT” in various biological metrics. No DELT anomalies were observed in Manganika Creek at either of the two monitoring stations. The lack of DELT observations does not completely eliminate the possibility of ammonia causing physical or physiological abnormalities to internal organs and body function.

### Summary: Is ammonia toxicity a stressor in Manganika Creek?

Unionized ammonia concentrations in Manganika Creek frequently exceed the state water quality standard, and several biological metrics show symptoms that can be related to ammonia toxicity. Given the many candidate stressors in this watershed, it is difficult to rule out other stressors that can cause similar effects on aquatic life (elevated conductivity, low DO). One of these other candidate stressors, elevated specific conductivity, may actually be minimizing the toxicity of ammonia in this watershed. In conclusion, ammonia toxicity cannot be ruled out as a stressor in Manganika Creek. Based on the frequent water quality standard violations, additional monitoring is recommended to determine how ammonia concentrations can be reduced to create more favorable conditions for aquatic life.

### 5.18.7 Manganika Creek: Summary of Stressors to Aquatic Life

**Table 79:** Summary of SID results for Manganika Creek

| Candidate Cause | Result |
|---|---|
| **Low Dissolved Oxygen** | ● |
| **Total Suspended Solids (TSS)** | ● |
| **Elevated pH** | ○ |
| **Ammonia Toxicity** | ○ |
| **Sulfate Toxicity** | ○ |
| **Elevated Specific conductivity** | ○ |
| Key: ● **= confirmed stressor** ○ **= Potential Stressor X = eliminated candidate cause** | |

## 5.19 Elbow Creek

Elbow Creek originates in the mining area just south of the towns of Leonidas and Eveleth. Many of the stream miles in the headwaters have been ditched to make room for urban, industrial, municipal, and mining land-uses. The upstream impaired AUID (04010201-518) is almost entirely a sinuous E-type channel (Rosgen 1994) that flows through a combination of wide lacustrine valleys and narrower alluvial valleys. After leaving Elbow Lake, Elbow Creek enters a lacustrine valley and resembles a wide, flat C-type channel (Rosgen 1994). After about a mile the creek narrows into an E channel and flows through a sequence of alluvial and lacustrine valleys. The downstream impaired AUID (04010201-570) is mostly a sinuous and narrow E channel. Overall, Elbow Creek drops a little more than 215 feet in just over 16 miles, with an average slope of 0.25%.

Fish and macroinvertebrate communities were evaluated at four monitoring stations on Elbow Creek. Two of these statons, 98LS016 and 09LS082, are located upstream of Elbow Lake, while stations 11LS073 and 09LS081 are downstrean of the lake outlet (Figure 277). FIBI results from station upstream of Elbow Lake were all below the impairment threshold. In contrast, FIBI scores below the lake all met the IBI critieria and do not indicate an impaired fish community. MIBI results were generally quite low throughout the entire length of Elbow Creek, with the exception of station 11LS073 (Table 80).

**Table 80:** Summary of Elbow Creek biological monitoring stations and FIBI and MIBI results

**Fish Assessments**

| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | FIBI Class | FIBI Result *(visit year)* | FIBI Result *(visit year)* | Standard | IBI Lower Confidence Limit | IBI Upper Confidence Limit |
|---|---|---|---|---|---|---|---|---|---|
| 98LS016 | 1.98 | 1.07 | 1 | 6 | **5** *(1998)* | **31** *(2009)* | 42 | 26 | 58 |
| 09LS082 | 3.13 | 0.42 | 1 | 6 | **26** *(2009)* | - | 42 | 26 | 58 |
| 11LS073 | 12.02 | 0.10 | 2 | 6 | **43** *(2011)* | - | 42 | 26 | 58 |
| 09LS081 | 14.09 | 0.09 | 2 | 6 | **55** *(2009)* | **53** *(2011)* | 42 | 26 | 58 |

**Macroinvertebrate Assessments**

| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | Invert IBI Class | Invert IBI Result *(visit year)* | Invert IBI Result *(visit year)* | Standard | IBI Lower Confidence Limit | IBI Upper Confidence Limit |
|---|---|---|---|---|---|---|---|---|---|
| 98LS016 | 1.98 | 1.07 | 1 | 6 | **18.58** *(1998)* | **15.36** *(2009)* | 51 | 37.40 | 64.60 |
| 09LS082 | 3.13 | 0.42 | 1 | 6 | **29.98** *(2009)* | - | 51 | 37.40 | 64.60 |
| 11LS073 | 12.02 | 0.10 | 2 | 6 | **50.76** *(2011)* | - | 51 | 37.40 | 64.60 |
| 09LS081 | 14.09 | 0.09 | 2 | 6 | **37.52** *(2009)* | **44.43** *(2011)* | 51 | 37.40 | 64.60 |

The fish impairment on Elbow Creek is limited to a reach upstream of Elbow Lake (pictured in Figure 280). Fish survey results from this reach show an assemblage dominated by Brook Stickleback, Northern Redbelly Dace, Central Mudminnow, and Fathead Minnow. These species are commonly found in streams with wetland qualities and are all at least somewhat tolerant of low DO conditions. The Fathead Minnow is considered a “pioneer species,” which means they are highly adaptable to streams that are regularly disturbed by a stressor. Darter species and simple lithophils (gravel spawning fish) were absent from the sampling station on this impaired reach, which also factored significantly into the low FIBI score.

The MIBI impairment on Elbow Creek includes the reach mentioned above, as well as the reach extending downstream of Elbow Lake to the confluence with the St. Louis River. Monitoring stations on the two reaches were assessed using different MIBI criteria due to differences in stream gradient and habitat types. The upper reach, which flows through a low gradient wetland area, had a macroinvertebrate community dominated by non-insect taxa – aquatic worms (Oligochaeta), roundworms (Nematoda), pill clams (*Pisidiidae*), and various midge (*Chironomidae*) taxa. Four dragonfly taxa and one caddisfly taxa (*Ptilostomis*) found at this location, but no stonefly or mayfly taxa were present. Overall, no intolerant or sensitive macroinvertebrate taxa were present at this site. The macroinvertebrate community in the lower reach of Elbow Creek was more evenly distributed among the taxa present, and does not appear to be as degraded as the community in upper Elbow Creek. Compared to high-quality sites of the same M-IBI class, this reach of Elbow Creek supported fewer “clinger” taxa and lacked the stonefly taxa that were present in some high quality streams of the same MIBI class.

## Candidate Causes for Impairment

Water quality and physical habitat data were used to develop a list of candidate causes for the MIBI and FIBI impairments in Elbow Creek. The following candidate causes were selected for further analysis as potential stressors;

1. **Low Dissolved Oxygen**
2. **Elevated Specific conductivity**
3. **Sulfate Toxicity**
4. **Nitrate Toxicity**
5. **Ammonia Toxicity**
6. **Poor Habitat Conditions**

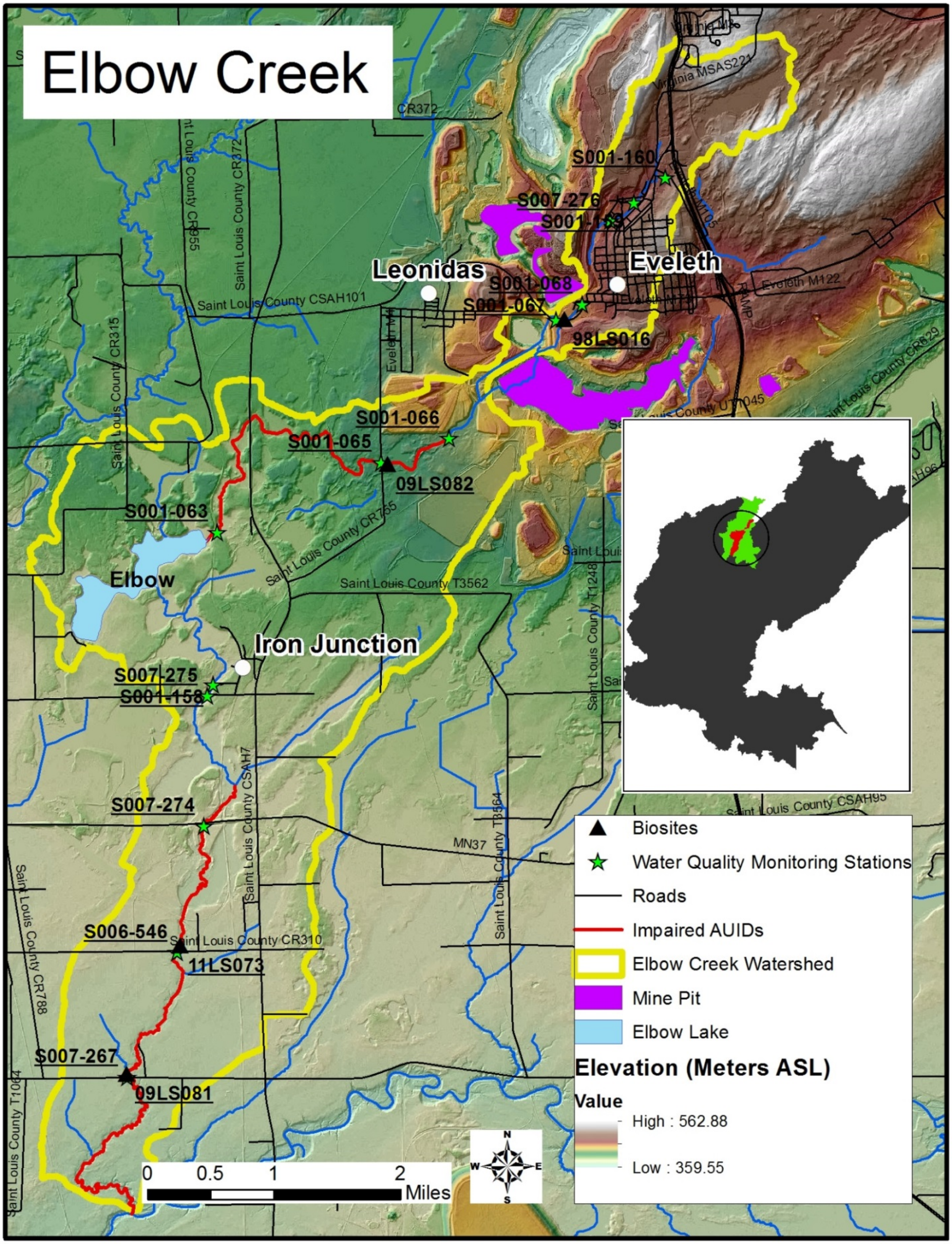


**Figure 277:** Map of Elbow Creek Watershed and impaired stream reach

### 5.19.1 Low Dissolved Oxygen

Low DO concentrations were identified as a potential cause of fish and macroinvertebrate impairments in two impaired reaches of Elbow Creek. Available DO data were collected using several methods, including instantaneous (point) measurements, longitudinal synoptic monitoring profiles, and multiple short-term (approximately one week) deployments of continuous monitoring equipment at several locations. The majority of the DO data were collected between the years of 2009 and 2012, however, some older data from the mid 1980's is available for a few monitoring locations.

Point measurements of DO are displayed in Figure 278 by sampling year and monitoring station and summarized by stream assessment units (AUID) in Table 81. Results below the 5 mg/L DO standard were observed at five of the ten monitoring stations, but only station S001-065 (biological monitoring station co-located 09LS082) had a significant amount of measurements that did not meet the standard. Forty-percent of the measurements (22 of 55) collected at this station were in violation of the DO standard, and over 30% of the readings were less than 3 mg/L. Based on the point measurement data alone, it is clear that the DO regime at station S001-065 is not suitable for supporting sensitive fish and macroinvertebrate taxa.

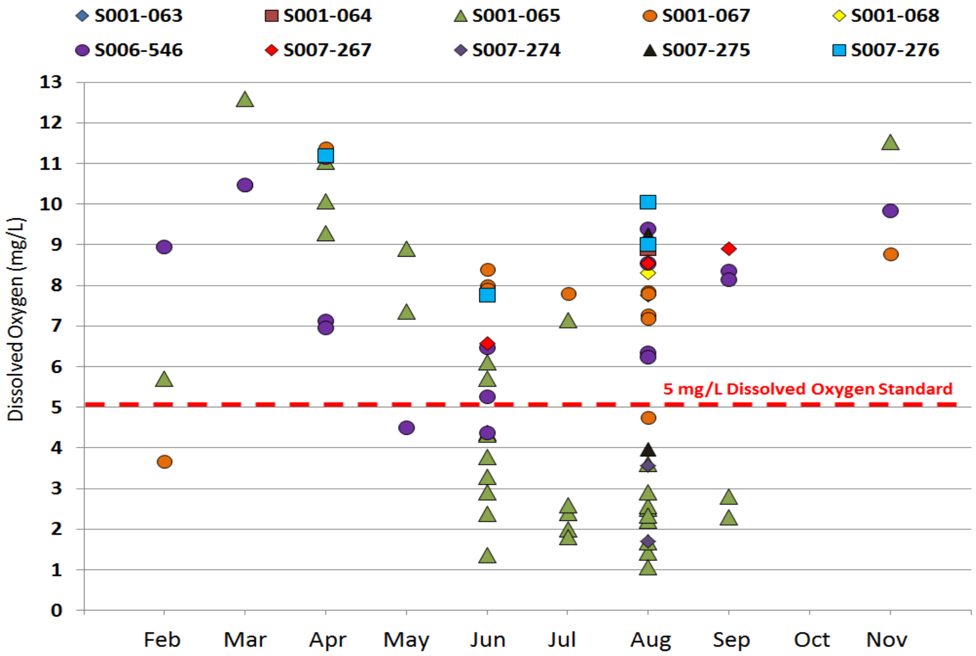


**Figure 278:** Point measurements of DO collected from Elbow Creek arranged by station and month

**Table 81:** Number and percent of DO measurements below the WQ standard by Elbow Creek AUID

| | Stream AUID* | | | | |
|---|---|---|---|---|---|
| | 04010201-979 | 04010201-521 | 04010201-518 | 04010201-572 | 04010201-511 |
| # sub-5 mg/L observation / Total Observations | 0/4 | 3/16 | 22/55 | 1/2 | 2/16 |
| % Observations below 5 mg/L WQ Standard | 0.0% | 18.8% | 40.0% | 50.0% | 12.5% |

The continuous DO data provide additional evidence of sustained periods of low DO at station S001-065. In August of 2012, continuous monitoring devices were deployed for a one week period at three monitoring stations that were co-located with biological data collection. DO concentrations station S001-065 (biological station 09LS082) remained below 2 mg/L for the entire monitoring period, and even dropped below 1 mg/L for several days (Figure 279). Results from the two monitoring stations located downstream of Elbow Lake showed DO concentrations that were more favorable for aquatic life. Minimum DO concentrations at these sites ranged from 5.43 mg/L to 6.12 mg/L. Diurnal DO flux was also considerably higher at the monitoring stations downstream of Elbow Lake, but did not exceed the DO flux criteria of 4 mg/L cited in the MPCA's river nutrient criteria (Heiskary, 2013).

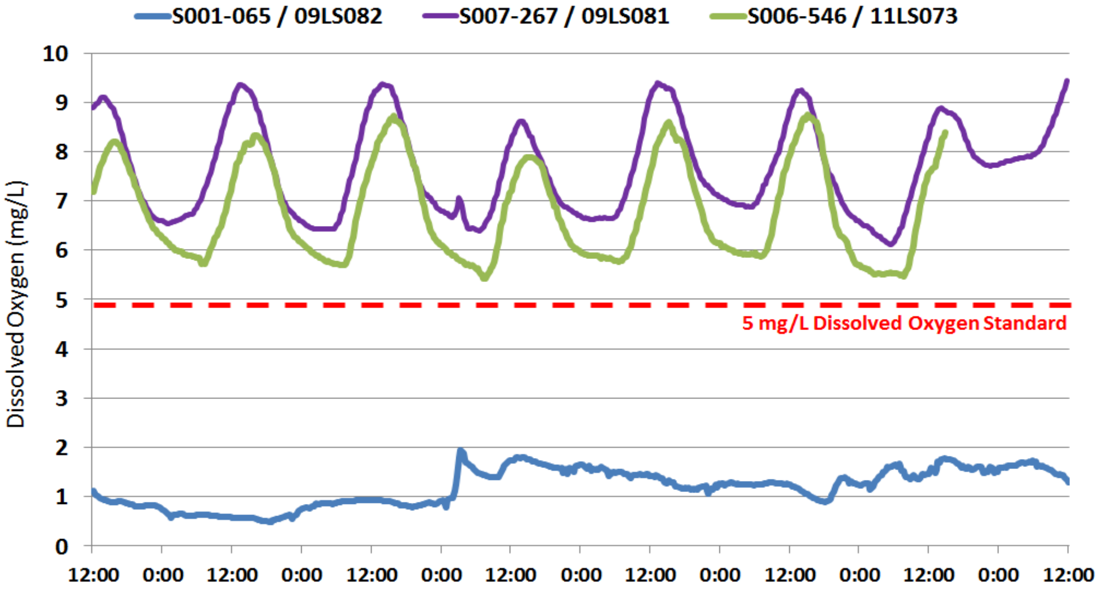


**Figure 279:** Continuous DO data collected at three Elbow Creek biological monitoring stations in August 2012

In the summer of 2013, an additional two continuous DO profiles were collected at station S006-546 to further investigate whether or not DO flux was a stressor to aquatic life at that location. There was some speculation that DO flux may be abnormally high at this monitoring location due to the eutrophic conditions that are frequently observed in Elbow Lake, which is currently listed as impaired for excess nutrient levels. Summary statistics for the two profiles are listed in Table 82. Maximum diurnal DO flux during the 2013 profiles ranged from 1.87 to 2.24 mg/L, which was lower than the 2012 results and well below the river nutrient criteria guidelines. Based on the results and data from other Elbow Creek stations, DO *flux* can be eliminated as a cause of biotic impairment.

**Table 82:** Summary statistics for two continuous DO profiles collected at station 11LS073 in 2012/2013

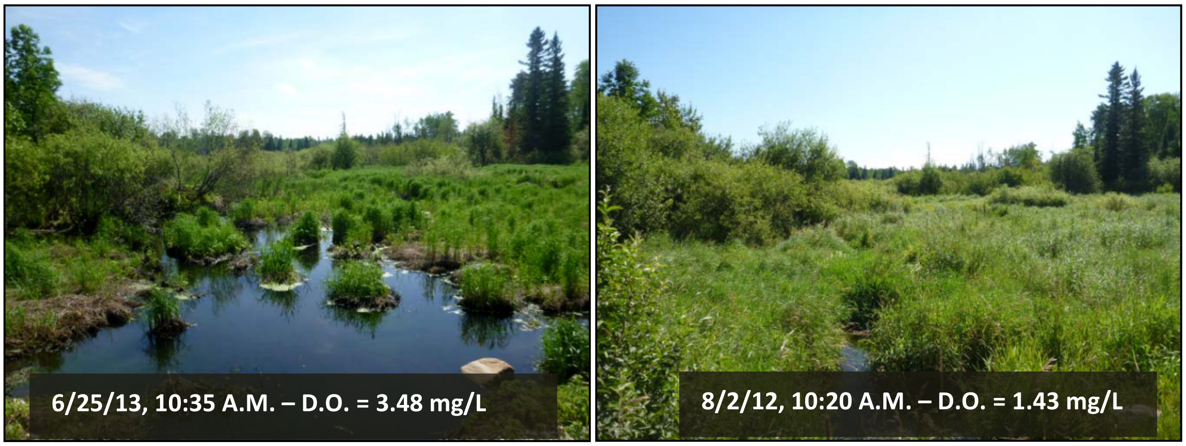
| Station | Date | # of Readings | Min (mg/L) | Max (mg/L) | % Readings below | Average 24 hour DO Flux (mg/L) | Max 24 hour DO Flux (mg/L) |
|---|---|---|---|---|---|---|---|
| S006-546 / 11LS073 | 7/26/13 – 8/1/13 | 579 | 6.9 | 10.47 | 0.0% | 2.02 | 2.24 |
| S006-546 / 11LS073 | 8/30/13 – 9/5/12 | 585 | 5.11 | 8.57 | 0.0% | 1.65 | 1.87 |

## Sources and Pathways Contributing to Low Dissolved Oxygen

Low DO concentrations in Elbow Creek occur most frequently in the stream reaches located just upstream and downstream of Elbow Lake. Potential sources of low DO that will be evaluated in this section include the effects of riparian wetlands, effluent from the Eveleth WWTP, beaver dam impoundments, and nutrient enrichment.

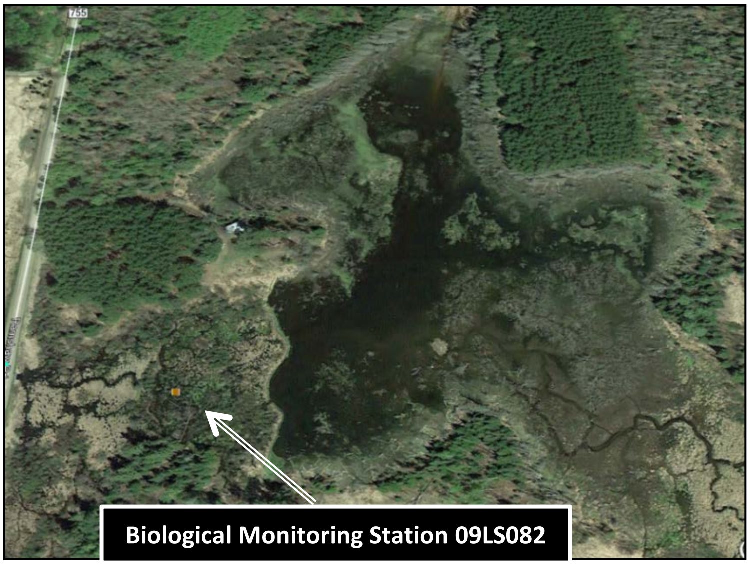
### Wetlands and Beaver Dams

Wetlands are a prominent feature of the riparian corridor or Elbow Creek upstream of impaired biological monitoring station 09LS082. Although riparian wetlands are beneficial for stream health in many ways (maintaining channel stability, filtering sediment and pollutants from runoff, streamflow regulation), they can also deliver oxygen depleted, nutrient rich water to nearby streams (Bourdaghs, 2013). Beaver dams in amongst these wetland dominated areas have the potential to further lower DO concentrations by reducing or eliminating streamflow and reducing capacity to transport organic sediment. Numerous beaver dams were observed upstream of station 09LS082, including a large dam that currently creates an impoundment with an area of approximately 12 acres (Figure 281). The presence of beaver dams and riparian wetlands are likely contributing to the low DO concentrations observed at station 09LS082.


6/25/13, 10:35 A.M. – D.O. = 3.48 mg/L

8/2/12, 10:20 A.M. – D.O. = 1.43 mg/L

**Figure 280:** Photos of biological station 09LS082 showing low flow conditions and wetland dominated riparian corridor



Biological Monitoring Station 09LS082

**Figure 281:** Aerial photo of station 09LS082 showing large beaver dam impoundment just upstream of the monitoring reach

## Nutrient Enrichment

Nutrient enrichment, particularly in the form of excess phosphorous, can be a driver of low or highly fluctuating DO concentrations in streams (Heiskary 2013). TP concentrations in the upper reaches of Elbow Creek (headwaters to Elbow Lake) are elevated well above the MPCA's river nutrient criteria thresholds for the northern nutrient region. Average TP concentrations range between 0.100 mg/L to nearly 0.500 mg/L in the upper impaired reach of Elbow Creek, and have exceeded 0.900 mg/L at station S001-066 (Figure 282). Limited TP data are available downstream of Elbow Lake, but based on available results, TP concentrations decline significantly in that reach. Figure 282 shows declining TP concentrations as the creek approaches its confluence with the lake, and the lowest TP results were found downstream of the lake at station S006-546.

Common sources of nutrient inputs to rivers and streams include effluent and septic systems, animal feedlots or confined animal feeding operations, and runoff from agricultural land and impervious surfaces. Agricultural land-uses are not common in the Elbow Creek Watershed and are unlikely to be a major source of nutrient inputs to Elbow Creek. The urban area of Eveleth, Minnesota (population 3,697) sits at the headwaters of Elbow Creek, and land-uses in and around this community may contribute to nutrient loading in this watershed. Based on available water quality and land-use data, the primary source of nutrient inputs to Elbow Creek is the Eveleth WWTP, which discharges to the creek in its headwaters near the city of Eveleth.

Under the current discharge permit granted to the Eveleth WWTP, the TP concentration of effluent discharged to Elbow Creek must be at or below 1.0 mg/L. Per these permit requirements, Eveleth WWTP has been actively monitoring effluent TP concentrations since the late 1990's. A summary of "calendar month" reporting values is provided in Figure 283. Results show that typical calendar month average TP concentrations between 2002 and 2014 are well above the river nutrient TP criteria of 0.055 mg/L. Monthly maximums during this twelve year monitoring period occasionally exceeded the concentration limit of 1.0 mg/L stated in the discharge permit, and have been as high as 2.5 to 3.1 mg/L.

In addition to the routine WWTP discharge under the current discharge permit, this facility is allowed to bypass untreated waste to Elbow Creek when unable to provide treatment to all wastewater inflow (large rain events, mechanical failures, etc.). Historically, these bypasses occurred fairly frequently and likely resulted in heavy loading of nutrients and other potentially harmful agents commonly observed in untreated wastewater (biological oxygen demand, *E. coli* bacteria). In 1992, a 5.6 million gallon containment pond was constructed on site, allowing operators to contain large amounts of untreated wastewater during rain events or other times when treatment is not possible. The untreated wastewater stored in this pond is eventually treated and discharged to Elbow Creek per permit guidelines. The addition of this containment pond has reduced the frequency of bypasses and reduced nutrient loading to Elbow Creek.

## Biological Oxygen Demand

The BOD is the amount of DO needed by aerobic biological organisms to break down organic matter. Streams with high volumes of organic matter often support large populations of microorganisms that degrade these organic compounds, resulting in high BOD concentrations and less DO available for more advanced forms of aquatic life, such as fish and macroinvertebrates. Effluent from WWTP often contains high concentrations of BOD and can result in lower DO concentrations in receiving waters (Ortiz and

Puig 2007). In addition, natural sources of organic matter and BOD loading can occur in streams with wetland characteristics. Elbow Creek contains both of these common sources of elevated BOD. Thus, a linkage between BOD and low DO in Elbow Creek is highly plausible.

Unfortunately, no BOD data were collected from Elbow Creek during the course of this study. However, BOD results are available from effluent monitoring work performed by staff at the Eveleth WWTP. This information can be used to gain some understanding of BOD inputs to Elbow Creek. Eveleth WWTP staff measure effluent BOD concentrations on a weekly basis (several samples per week) and report them as average and maximum values by calendar month. Based on results from sampling completed between the year of 1999 and 2014, the BOD concentration of the effluent leaving the WWTP is generally around 2.6 to 4.4 mg/L, but can exceed 10 mg/L at times. BOD concentration exceeding 1.5 mg/L were identified as a response variable to excess nutrients in the draft nutrient criteria developed for streams and rivers of northern Minnesota (Heiskary 2013). The relatively high BOD concentrations of the effluent combined with the low flow conditions in Elbow Creek are one possible pathway leading to low DO concentrations in the upper half of the watershed.

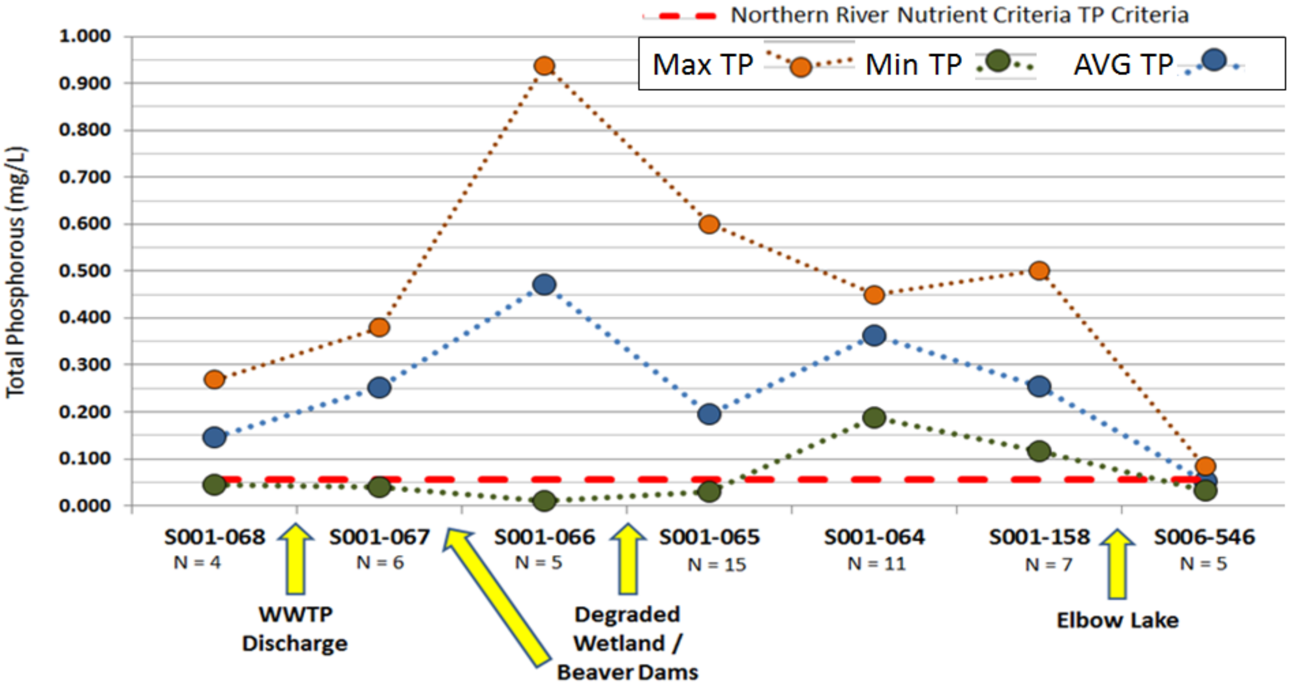


**Figure 282:** Minimum, average, and maximum TP concentrations observed at Elbow Creek monitoring stations.

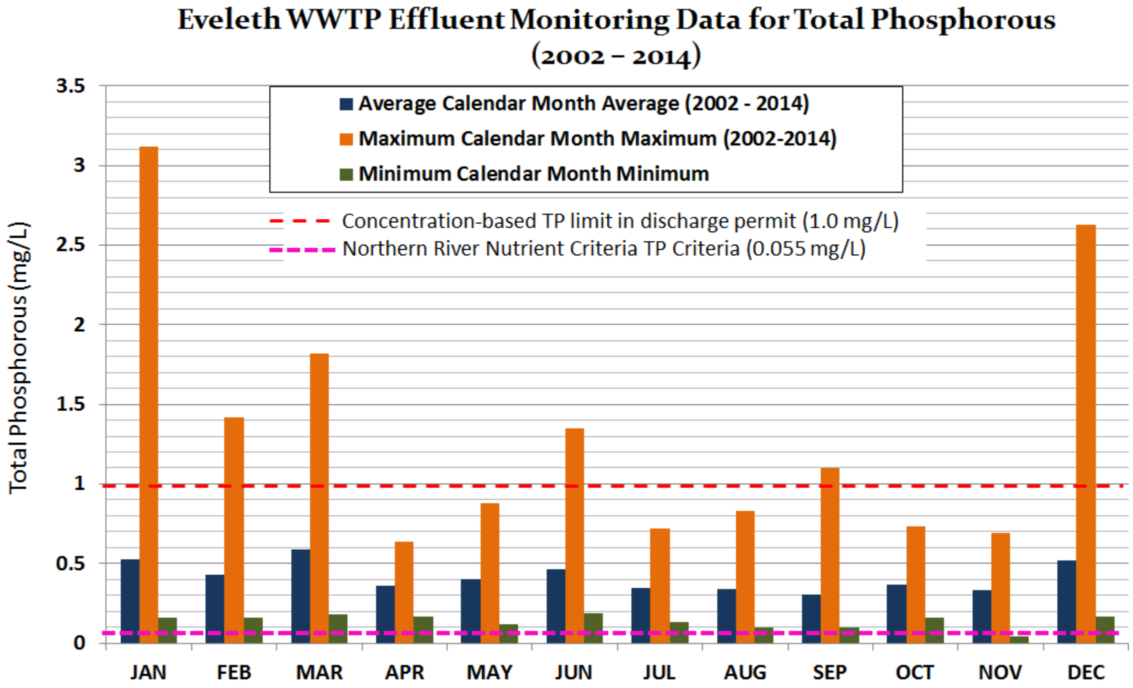


**Figure 283:** Summary of TP concentrations observed in Eveleth WWTP effluent discharged to Elbow Creek

## Biological Response to Low Dissolved Oxygen

The upper impaired reach of Elbow Creek (AUID 04010201-518, Headwaters to Elbow Lake) is impaired for failing to meet aquatic life criteria for measures of fish and macroinvertebrate biological integrity. Two biological monitoring stations are located on this reach (98LS016 and 09LS082) with a total of three monitoring visits each for fish and macroinvertebrate assessments (98LS016 was sampled twice for both fish and inverts). The fish community observed in this reach is dominated by species that are tolerant of low DO concentrations. Combined, Fathead Minnow, Brook Stickleback, and Central Mudminnow accounted for over 76% of the total fish sampled. Each of these species is commonly dominant in low DO environments. Other species that were present in this reach include Northern Redbelly Dace and Finescale Dace. These two species are generally more sensitive to disturbance and are often found in higher quality headwaters stream environments. Yet, they are both fairly adapted to survive headwater streams with wetland influences, which often show low DO conditions.

Below Elbow Lake, the fish community becomes more diverse and includes several species that can be considered sensitive to low DO concentrations and other stressors. As a result, this reach was not listed as impaired for poor FIBI, but was listed as impaired for failing to meet MIBI criteria. Longnose Dace, Mottled Sculpin, and Iowa Darter are examples of sensitive species that were present below Elbow Lake but absent in the FIBI impaired reach upstream of the lake.

Fish community DO TIVs for Elbow Creek monitoring sites are shown in Figure 284. Overall, DO TIV scores were extremely low in Elbow Creek compared to high quality stations of the same FIBI class. The only exception are the results from the 2011 sampling visit to station 09LS081, which are comparable to the median DO TIV scores observed at the reference stations. The relative abundance of DO tolerant individuals upstream of Elbow Lake resulted in extremely low DO TIV scores for stations 98LS016 and 09LS082.

The tolerance level of the macroinvertebrate community to low DO varied considerably between Elbow Creek monitoring sites. The headwaters monitoring station (98LS016) between the towns of Eveleth and Leonidas shows a relatively high percentage of organisms that are moderately intolerant of DO, particularly in the most recent sample collected in 2009. A large population of *Simulium* (black fly) and *Micropsectra* (bloodworm midge) in both 1998 and 2009 sampling events account for most of the intolerant organisms found at this station. Further downstream, Elbow Creek flows through an expansive wetland area that is commonly impounded by beaver activity. Here, at station 09LS082, a dramatic shift in the macroinvertebrate community was observed. The most dominant taxa at this station were aquatic worms (Oligochaeta/Nematoda), midges (*Tanytarsini/Chironomus/Bezzia*), and pill clams (Pisidiidae). These taxa are tolerant of a wide variety of stressors, including low DO concentrations. The dominance of low DO tolerant taxa at this monitoring station resulted in a very low DO TIV score at this station in comparison with the other biological monitoring stations on Elbow Creek (Figure 285).

The reach of Elbow Creek downstream of the lake (AUID 04010201-570) is impaired for low MIBI scores, but the taxa present at these monitoring stations are generally not tolerant of low DO. Several taxa that can be considered intolerant of low DO concentrations were common at station 09LS081, including mayfly from the genera *Baetis* and *Fallceon*, and black fly larvae *(Simulium)*. DO TIV scores for the 2009 visit to station 09LS081 were comparable to the median results from high quality reference sites of the same IBI class in (Figure 285). A slightly lower score resulted from the 2011 visit to the same site but the DO TIV value from this sampling event is not low enough to suggest a dominance of low DO tolerant taxa. Station 11LS073 did not support any taxa that can be considered intolerant of low DO concentrations (Figure 286). Instead, this site predominantly supported taxa that are found in streams with a wide range of DO concentrations. DO TIV results for this station fell near the 25th percentile of scores observed at non-impaired stations of the same MIBI class.

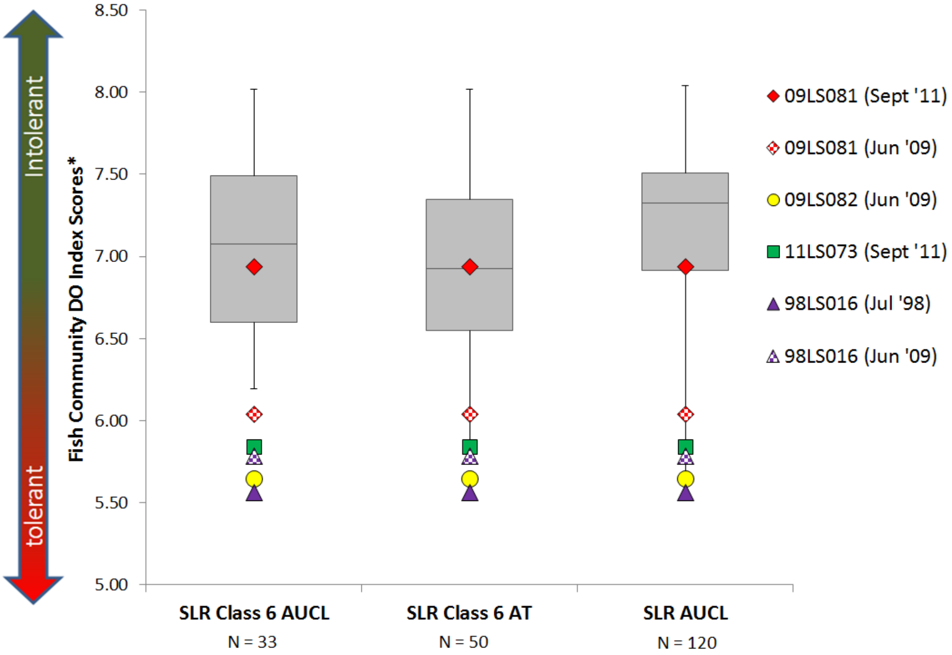


**Figure 284:** Fish community DO TIV results for Elbow Creek monitoring sites compared to high quality reference stations of the same IBI class. * See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL** = Above Upper Confidence Limit of FIBI threshold **AT** = Above Fish IBI Threshold

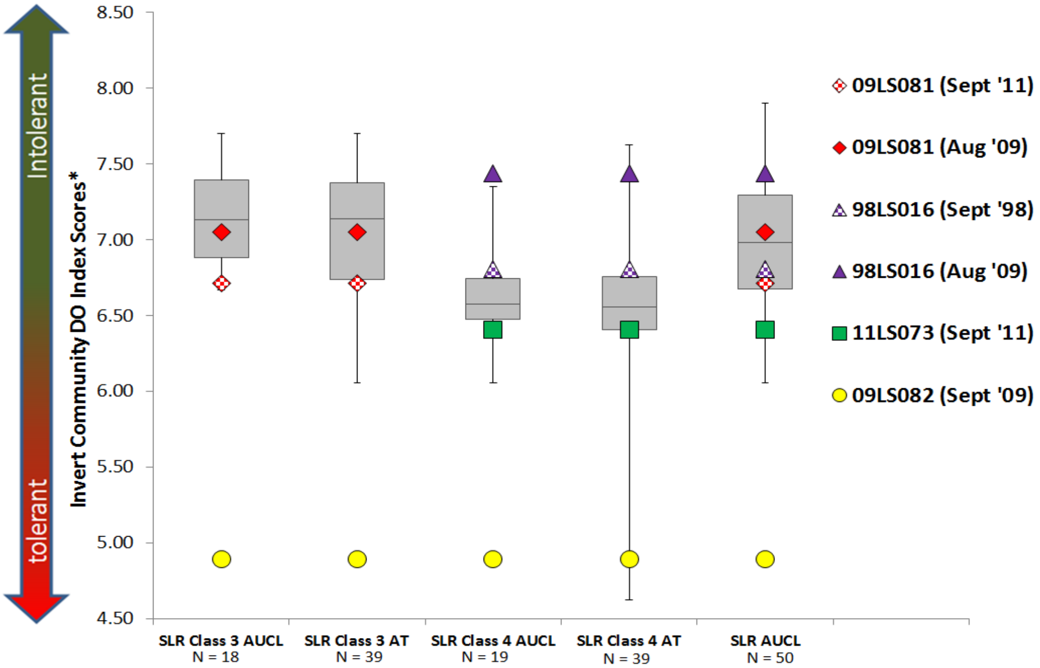


**Figure 285:** Macroinvertebrate DO TIV results for Elbow Creek monitoring sites compared to high quality reference stations of the same IBI class. * See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

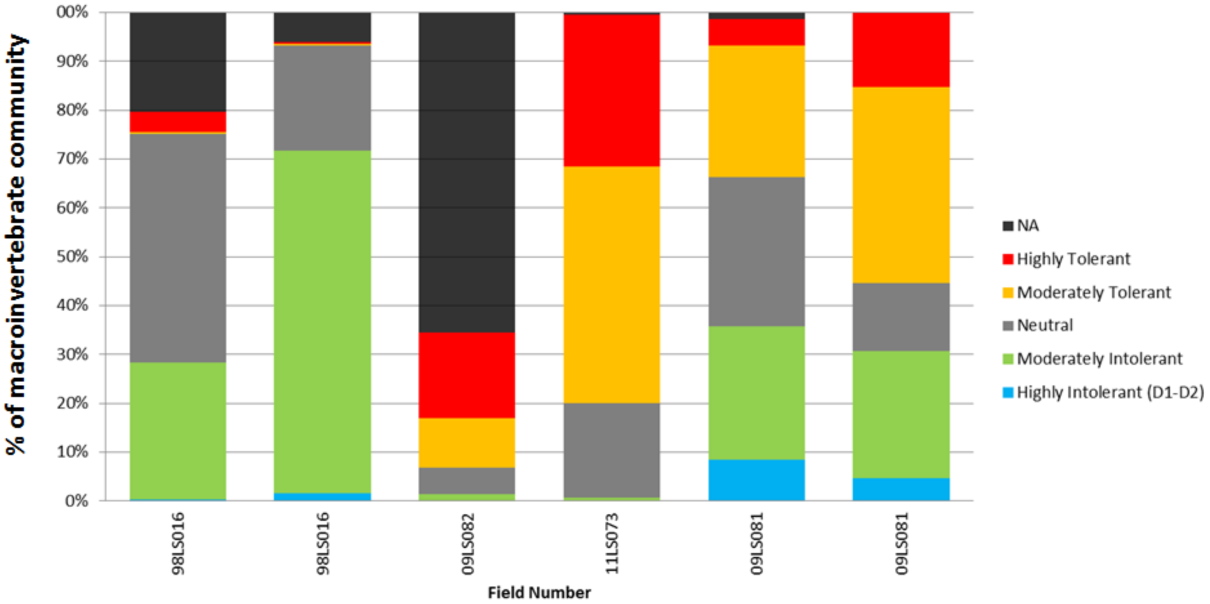


**Figure286:** Proportion of macroinvertebrate assemblage observed at Elbow Creek stations that are considered tolerant, neutral, or intolerant of low DO concentrations based on MPCA tolerance values.

### Summary: Is low dissolved oxygen a stressor in Elbow Creek?

Based on the available biological and water chemistry data, low DO can be confirmed as a stressor to fish and macroinvertebrate populations in the upper impaired reach of Elbow Creek (AUID 04010201-518). DO concentrations regularly dropped below 1 mg/L and remained at that level for extended durations. Highly tolerant fish and macroinvertebrate taxa (Fathead Minnow, Brook Stickleback, aquatic worms) were dominant in reaches where low DO concentrations were observed. The combination of wetlands, beaver dams, low flow conditions, and nutrient/BOD loading is likely driving the low DO conditions observed in this reach.

The impaired reach below Elbow Lake (AUID 04010201-570) does not appear to be impacted by low DO conditions or DO flux. DO concentrations did fall below the 5 mg/L water quality standard on occasion within this reach. However, low DO conditions were not spatially or temporally co-located with biological symptoms of DO stress. Continuous monitoring results from sites above and below Elbow Lake revealed drastic differences in DO concentrations. During the same period of time when DO concentrations were less than 1 mg/L at station 09LS082 upstream of the lake, Considering all of the data at hand, DO can be eliminated as a candidate cause for impairment in this reach.

## 5.19.2 Sulfate Toxicity

Sulfate toxicity was identified as a candidate cause of impairment in Elbow Creek based on the mining land uses within its watershed and available water chemistry data. Available sulfate data for the three monitoring stations are shown in Figure 287. All data were collected during 2011, 2012, and 2014. Sulfate concentrations rarely exceeded 100 mg/L in this watershed (3 of 22 samples, or 14%), but a result of 443 mg/L was observed at station S001-065 during a November 2011 sampling effort. The other two results exceeding 100 mg/L were also observed during this November 2011 sampling event. Overall, the majority of sulfate results for this watershed are within a range that is not considered toxic to aquatic life. However, the elevated results in 2011 demonstrate that sulfate concentrations are high enough at times to be considered a potential stressor.

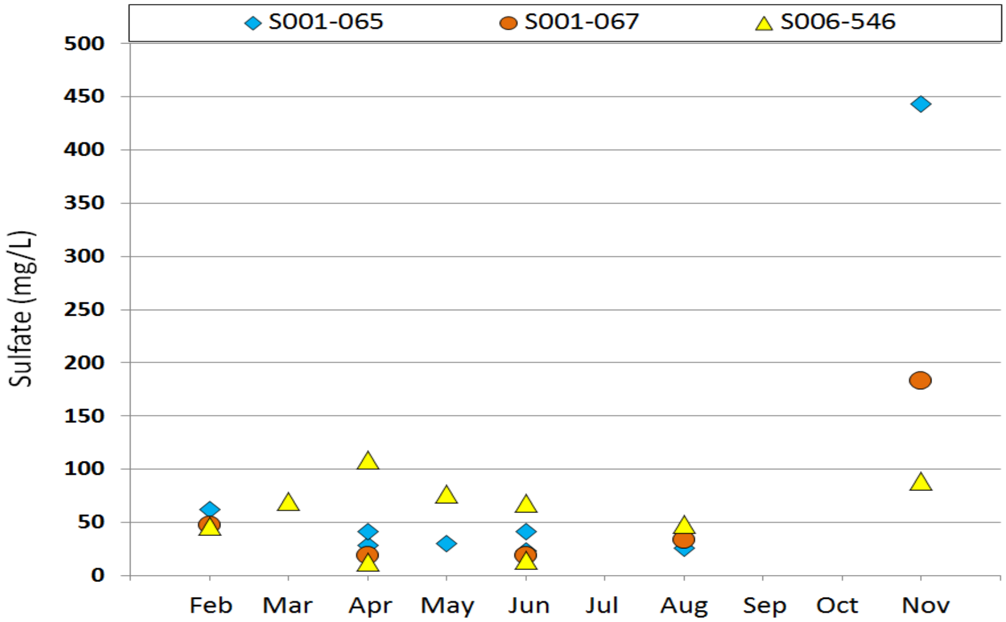


**Figure 287:** All available sulfate date from Elbow Creek. Data collected in 2011, 2012, and 2014.

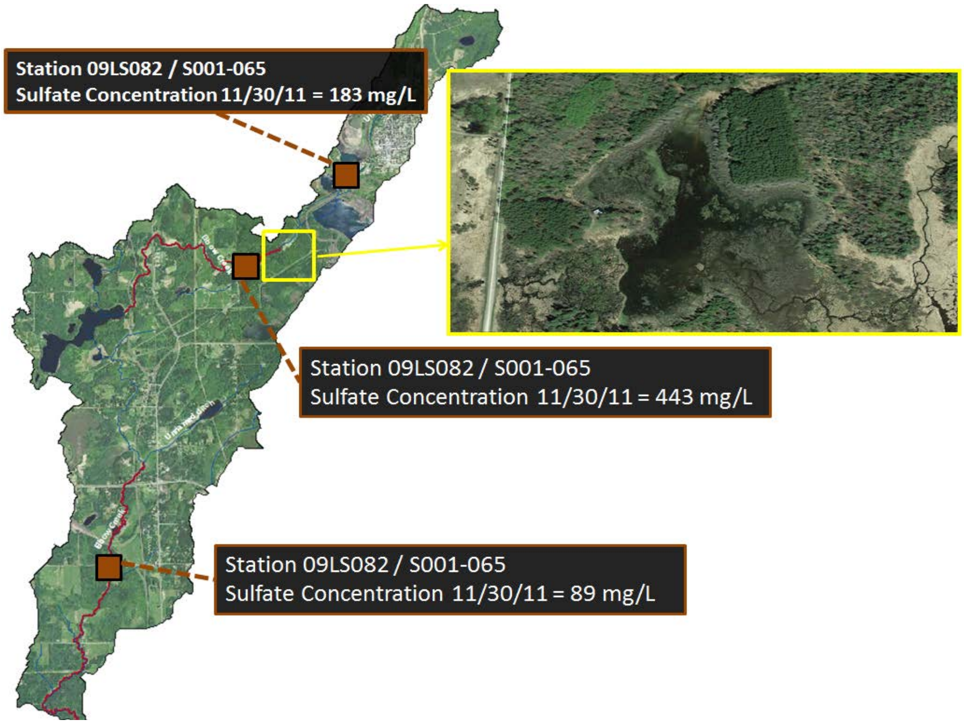
### Sources and Pathways of Sulfate

Major sources of sulfate in the Elbow Creek Watershed include mining land use features and the city of Eveleth WWTP. WWTP effluent enters Elbow Creek in the upper one-third of the watershed, just upstream of biological monitoring station at a rate of around 0.5 to 1 cubic feet per second (cfs). Sulfate was added to the list of sampling parameters required by the facility's NPDES permit in 2013, with the monitoring results reported as "calendar month maximums" (CMM). The average CMM over the course of 2013-2014 is 43 mg/L, with a maximum CMM of 82 mg/L (n=20, min = 26 mg/L).

Historically, several mine pits in the area were dewatered into Elbow Creek. However, these pits have not likely dewatered into the creek since 2002 (Crotteau, personal communication). The lack of continuous pit dewatering may be keeping the sulfate concentrations in Elbow Creek slightly lower than others on the iron range that receive constant regular dewatering inputs (e.g. Spring Mine Creek, Manganika Creek, Kinney Creek).

Waste rock piles and tailings placed on land have been cited as a major source of sulfate in watersheds with mining land uses (Berndt and Bavin 2012). Sulfate is generated from these stockpile and overburden areas due to the oxidation of minor sulfide minerals that are exposed to atmospheric conditions. In the upper half of the Elbow Creek Watershed, there are numerous waste rock stockpiles sitting adjacent to the stream and its tributaries. These stockpiles are potentially a significant source of sulfate delivery to Elbow Creek.

Beaver dams and wetland complexes may be having an effect on the concentrations of sulfate observed in this stream. Upstream of biological monitoring station 09LS082, Elbow Creek is impounded by a large beaver dam, forming an expansive wetland complex on the upstream side (see inset in Figure 288). The DO concentrations as low as 1.06 mg/L have been observed within the biological monitoring reach downstream of this beaver dam impoundment. It is possible that DO concentrations reach anoxic levels within the wetland complex upstream of the dam. If anoxia is occurring periodically, redox reactions may be occurring within this wetland complex to convert sulfate to sulfide. These interactions may explain the variability in sulfate concentrations observed at the monitoring site downstream.



**Station 09LS082 / S001-065**
**Sulfate Concentration 11/30/11 = 183 mg/L**

Station 09LS082 / S001-065
Sulfate Concentration 11/30/11 = 443 mg/L

Station 09LS082 / S001-065
Sulfate Concentration 11/30/11 = 89 mg/L

**Figure 288:** Sulfate monitoring results at three locations on Elbow Creek, 11/30/11. A significant increase in sulfate concentration was observed between the two upper monitoring stations where mining waste rock stockpiles are located.

### Water Quality Standards for Sulfate

Minnesota does not currently enforce a sulfate standard for protection of fish and aquatic macroinvertebrates. Several U.S. and Canadian provinces have developed sulfate standards that will be used to evaluate sulfate as a stressor to aquatic life in the SLRW. For more information on these standards, see Section 3.1.6.

### Sulfate Data Discussion – Elbow Creek

Several U.S. states and Canadian provinces are working towards implementing a sulfate water quality standard for protecting fish and macroinvertebrates. In this section, sulfate data from Elbow Creek will be compared to the criteria being implemented or discussed in other regions. Several sulfate criteria values published in scientific literature will also be cited for comparison.

#### British Columbia, Canada

In a paper by Elphick et al (2010), various sulfate standards are proposed for British Columbia waters based on SSD and a SFA. Both of these standards are dependent on water hardness, as harder water can ameliorate the impact of sulfate on aquatic life. The sulfate standard proposed in British Columbia for very hard water (>160 mg/L) is 725 mg/L based on the SSD approach, and 675 mg/L based on the SFA approach. For water with hardness values between 80-100 mg/L, the sulfate criteria are 625 mg/L (SFA) and 644 mg/L (SSD).

Conditions in Elbow Creek do not exceed any of the proposed sulfate criteria mentioned above.

#### California

The state of California evaluated sulfate as a stressor to aquatic life in a 2013 study (Buchwalter 2013). Although many uncertainties involving sulfate toxicity were discussed in this report, the author concluded that there is enough toxicity data by EPA standards to support an acute toxicity criterion of 234 mg/L SO4 and a chronic criterion of 124 mg/L SO4. These values were not adjusted based on chloride and hardness values like other WQ standards for sulfate, and the author mentions uncertainties in the values stated above based on this detail.

Sulfate levels in Elbow Creek exceeded the 124 mg/L value in 10% of samples collected (2 of 22), and exceeded the 234 mg/L acute value in less than 5% of samples (1 of 22). The sulfate toxicity thresholds cited by Buchwalter (2013) are relatively low compared to the values cited in other WQ standard work and other scientific literature. Therefore, the fact that sulfate concentrations in Elbow Creek rarely exceed these values offers evidence against sulfate as a stressor in this watershed.

#### Ohio

Rankin (2003, 2004) linked biological monitoring data with sulfate sampling results across the state of Ohio with the goal of identifying critical thresholds for protecting sensitive forms aquatic life. Although no water quality standards were developed through this work, several conclusions can be drawn from these reports:

1. ***Many of the most sensitive taxa were not present in streams where sulfate concentrations exceeded 200 mg/L.***

The number of sensitive macroinvertebrate taxa observed in Elbow Creek was highly variable between monitoring stations. The monitoring stations lower in the watershed 09LS081 and 11LS073 supported more intolerant macroinvertebrate taxa (6 and three taxa, respectively). Intolerant taxa were absent from station 98LS016 in both sampling events, and at station 09LS082, only one intolerant taxa was observed.

2. ***There is good evidence from Ohio streams that the presence of higher chloride concentrations ameliorates the effects of sulfate.***

Average chloride concentration in Elbow Creek is around 50 mg/L, with seasonal variation ranging from 25-100 mg/L. Rankin (2004) demonstrated that high biological integrity scores are more frequently achieved in the presence of high sulfate concentrations when chloride concentrations are also elevated (above 20 mg/L). Elevated chloride concentrations in Elbow Creek may be reducing the potential for sulfate to be toxic to aquatic life in this watershed.

3. **Streams with sulfate concentrations above 400 mg/L generally exhibited poor biological integrity scores**

Only one sampling result exceeded 400 mg/L in Elbow Creek. This result was co-located with an impaired biological monitoring station (09LS082).

4. **EPT macroinvertebrate taxa were limited to 10 or less at sites where sulfate concentrations exceeded 500 mg/L**

No sampling results from Elbow Creek exceeded 500 mg/L. The highest concentration recorded was 443 mg/L at biological monitoring station 09LS082. Still, EPT taxa richness was limited to 10 or less at all stations except for 09LS081, which supported 13 EPT taxa in each of the samples collected in 2009 and 2010. EPT taxa richness was extremely low at stations 98LS016 (n=0) and 09LS082 (n=1). The low numbers of EPT taxa observed at these stations may be an indicator of sulfate stress, but this biological metric also responds negatively to other stressors that are present in this watershed (low DO, nitrate toxicity, poor habitat conditions).

### Illinois, Indiana, Pennsylvania, and Iowa

The states of Illinois, Indiana, Pennsylvania, and Iowa have been working towards an aquatic life standard for sulfate and other dissolved solids. The specifics of this sulfate standard are provided in Table 9. Unlike some of the sulfate criteria listed previously in this section, chloride and water hardness were taken into account in the development of a sulfate standard for these states. Based on the hardness and chloride data available for Elbow Creek, the applicable water quality standard for this stream would range from 500 mg/L to 2,671 mg/L if similar guidelines were used for a sulfate standard in Minnesota.

Table 83 provides a summary of paired sulfate, hardness, and chloride results along with the applicable water quality standard for sulfate based on the approach used by the states of IL, PA, IN, and IA. Sulfate concentrations in Elbow Creek remained below the aquatic life standard applied in these states in all 14 instances where sulfate, hardness, and chloride measurement were available.

**Table 83:** Elbow Creek sulfate data paired with hardness and chloride results, and applicable water quality standard based on formula used by several US states.

| Station | Sample Date | Hardness (mg/L) | Chloride (mg/L) | Sulfate (mg/L) | Sulfate Standard (IL, IA, PA) | Exceeds IA, IL, IN, PA Standard? |
|---|---|---|---|---|---|---|
| **S006-546** | 4/13/2011 | 64 | 19 | 13 | 500 | **No** |
| **S006-546** | 5/30/2012 | 170 | 24 | 77 | 1454 | **No** |
| **S006-546** | 6/23/2011 | 108 | 27 | 15 | 1243 | **No** |
| **S006-546** | 11/30/2011 | 240 | 34 | 89 | 1723 | **No** |
| **S006-546** | 2/5/2014 | 210 | 45 | 47 | 1624 | **No** |
| **S001-065** | 5/30/2012 | 120 | 42 | 31 | 1300 | **No** |
| **S001-065** | 11/30/2011 | 860 | 52 | 443 | 500 | **No** |
| **S001-065** | 4/13/2011 | 107 | 53 | 28 | 1264 | **No** |
| **S001-065** | 6/23/2011 | 136 | 68 | 23 | 1383 | **No** |
| **S001-065** | 2/5/2014 | 170 | 103 | 48 | 1536 | **No** |
| **S001-067** | 4/13/2011 | 105 | 59 | 19 | 1263 | **No** |
| **S001-067** | 11/30/2011 | 495 | 71 | 183 | 2671 | **No** |
| **S001-067** | 6/23/2011 | 137 | 89 | 19 | 1406 | **No** |
| **S001-067** | 2/5/2014 | 140 | 100 | 63 | 1426 | **No** |

**Summary: Is sulfate toxicity a stressor in Elbow Creek?**

Sulfate concentrations are elevated in the Elbow Creek Watershed due to mining and municipal wastewater sources. Compared to other streams with similar land uses in the SLRW, Elbow Creek has lower sulfate concentrations, with only one high outlier result of 443 mg/L in November 2011. Although sulfate levels occasionally exceed concentrations that are considered harmful to aquatic life by some researchers, elevated hardness and chloride concentrations are likely buffering the harmful effects of sulfate in this particular stream. All sulfate results from this watershed met water quality standards being applied in the states of Illinois, Indiana, Iowa, and Pennsylvania. Based on the data and supporting information available at this time, it is unlikely that sulfate is a primary cause of impairment in Elbow Creek. However, sulfate cannot be eliminated as a candidate cause of impairment for several reasons; (1) there is scientific literature supporting much lower sulfate toxicity thresholds (e.g. Buchwalter 2013) and; (2) currently there is no sulfate standard for fish and aquatic macroinvertebrates in Minnesota.

### 5.19.3 Specific conductivity

Elevated specific conductivity was identified as a candidate cause for impairment in the Elbow Creek watershed based on a review of current data and land uses within the watershed. A significant number of specific conductivity readings have been collected in this watershed (n=74, nine total monitoring stations). The vast majority of conductivity readings from this stream are between 200 – 700 µS/cm, but in rare instances, conductivity has exceeded 1,600 µS/cm (Figure 289). Both of the readings exceeding 1,600 µS/cm were observed in the upper portion of the watershed on 11/30/2011 during a late season baseflow monitoring event. These high readings co-occurred with a significant spike in sulfate concentrations (see Figure 287 for sulfate data). Downstream of Elbow Lake, all conductivity readings were below 600 µS/cm. It appears that this 165-acre lake provides a dilution or buffer effect, which results in lower conductivity levels downstream.

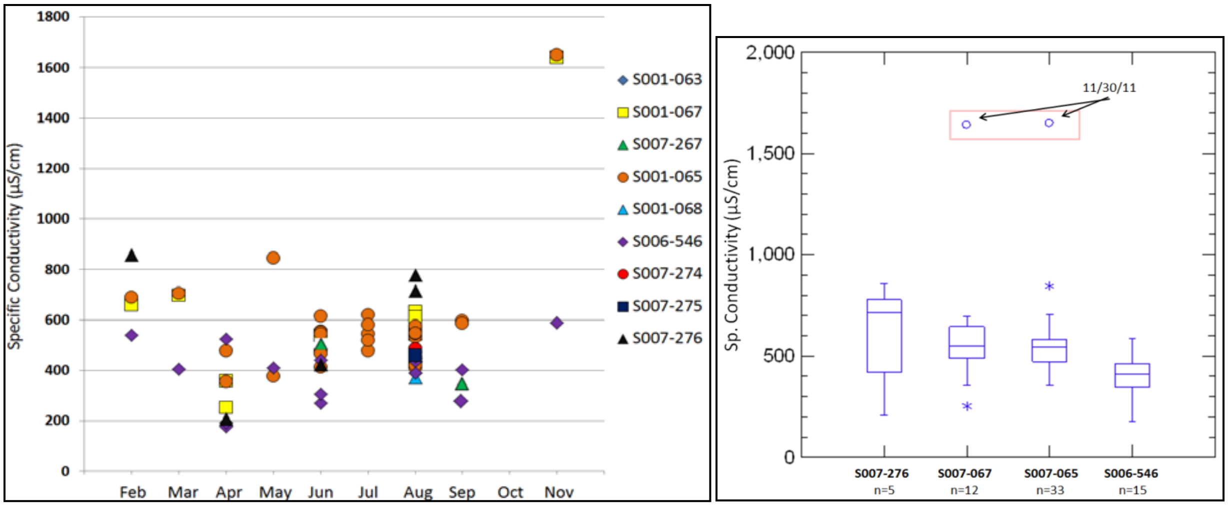


**Figure 289:** Point observations of specific conductivity in Elbow Creek (left), and box plots of results by station (right)

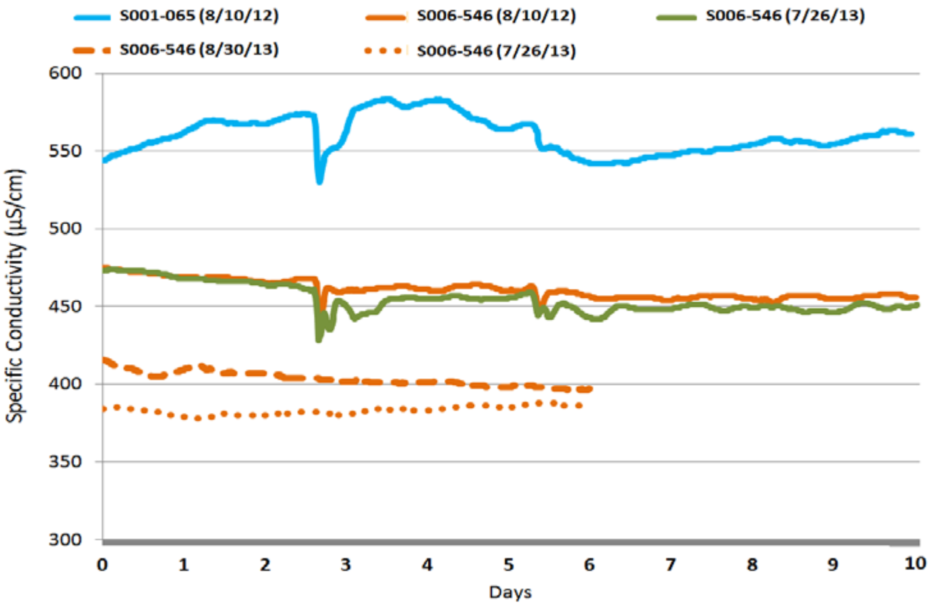


**Figure 290:** Continuous specific conductivity results measured over a 6-10 day period at several Elbow Creek monitoring sites

## Sources and Pathways of Elevated Specific conductivity

Samples were collected from three Elbow Creek monitoring stations during baseflow, rain event flow, and snowmelt flow and analyzed for major cations and anions. The most common salt ions found in surface water that influence specific conductivity levels include positively charged (cations) $Ca^{2+}$, $Mg^{2+}$, $Na^{+}$, $K^{+}$, and negatively charged (anions) $HCO_3^{-}$, $CO_3^{2-}$, $SO_4^{2-}$, and $Cl^{-}$. Figure 291 shows the concentrations of select cations and anions observed during the three geochemistry sampling events. The baseflow sampling results show elevated concentrations of $SO_4$, Cl, Mg, and Ca. These four ionic compounds appear to be the prominent drivers of high conductivity in Elbow Creek. Sulfate, in particular, shows a strong correlation with specific conductivity in these three sampling events. Chloride concentrations did not vary between sampling events as much as the other parameters. Another unique pattern in the chloride data can be observed in the rain event data, which shows an increase in chloride concentration at the two monitoring stations upstream of Elbow Lake (S001-067 and S001-065) during this event. Sodium concentration also increased slightly at these stations during the rain event.

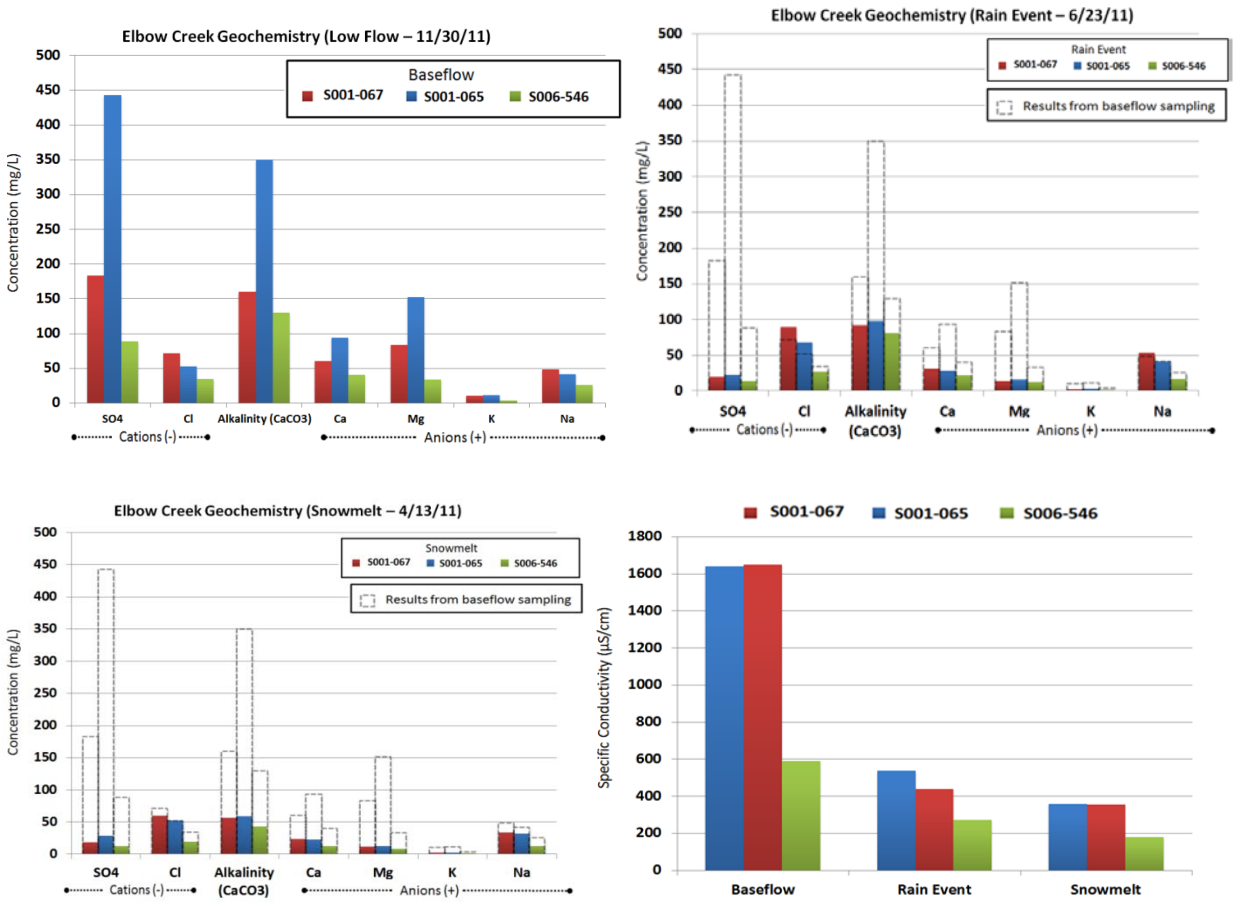


**Figure 291:** Geochemistry sampling results for Elbow Creek during baseflow (upper left), rain event flow (upper right), and snowmelt flow (lower left). Specific conductivity readings for the three monitoring events are shown (lower right)

Effluent monitoring for specific conductivity has been required as part of the Eveleth WWTP's discharge permit since 2013. Treated effluent from this facility is continuously discharged to Elbow Creek at rates of approximately 0.70 to 1.5 cubic feet per second (cfs). Specific conductivity values of the effluent are reported as calendar month maximum values (CMM). In over 20 months of monitoring (February 2013 through September 2014), the average conductivity of effluent discharged to Elbow Creek from this facility is 675 µS/cm (n=20, min=580 µS/cm; max=830 µS/cm). Unless facility upgrades occurred between 2011 and 2013 to improve treatment, it is unlikely that WWTP effluent alone resulted in the elevated conductivity readings (>1,600 µS/cm) observed in November of 2011. The city of Iron Junction WWTP also discharges to Elbow Creek, but the discharge point is downstream of monitoring sites where elevated conductivity levels have been observed.

United Taconite LLC possesses a permit to discharge runoff from a mine shop area to ditch that leads to Elbow Creek. The location of this discharge is in the headwaters of the watershed, approximately one mile upstream of the Eveleth WWTP. This is not a continuous discharge, and monitoring records spanning 1999 through 2014 show very limited discharge from this facility. Over that period of record, the calendar month average discharge ranges from 0 cfs to 0.3 cfs. Specific conductance is not currently included as a parameter in the monitoring plan for this facility. In addition to flow, this permit requires monitoring for TSS and petroleum hydrocarbons. Monitoring records show occasional permit violations for exceeding the limits associated with these parameters. Given the lack of specific conductivity data and the intermittent nature of discharge from this site, it is difficult to determine whether or not this discharge has any connection to elevated conductivity in the upper reaches of Elbow Creek. No discharge was reported from this site in November of 2011 when specific conductivity exceeded 1,600 µS/cm at stations located in the upper reaches of Elbow Creek.

The other permitted dischargers in the Elbow Creek Watershed include several facilities with stormwater permits (Table 84). Several of these facilities are located downstream of the impacted area and would have no linkage to elevated conductivity. The OSI Environmental Inc. (permit MNR05355W) is located in the vicinity of the monitoring stations where high conductivity readings have occurred, but there is no data connecting this facility with those observations.

**Table 84:** List of permitted point source dischargers in the Elbow Creek Watershed.

| Facility Name | Permit ID | Details / Notes |
|---|---|---|
| Eveleth WWTP | MN0023337 | Wastewater Discharge |
| Eveleth WWTP | MN0023337 | WWTP Bypass Discharge point |
| United Taconite, LLC | MN0044946 | Runoff from mine shop area |
| St. Louis County Land Department | MNG490177 | Stormwater Permit |
| Iron Junction WWTP | MNG580049 | Wastewater Discharge |
| OSI Environmental (Petroleum Bulk Stations & Terminals) | MNR05355W | Stormwater Permit |
| Keenan Yard Fueling and Maintenance Facility | MNR0535DX | Stormwater Permit |

Stockpiles of waste rock and overburden from mining operations are a common feature of the landscape in the upper portions of the Elbow Creek Watershed. As previously discussed in Section 3.1.6, exposure of these stockpiles to the atmosphere can introduce sulfate into surface water. Elevated conductivity levels in Elbow Creek are associated with spikes in sulfate concentration. Therefore, runoff from stockpiles is one potential source contributing to this candidate stressor. Sulfate is also discharged from the Eveleth WWTP in fairly high concentrations (up to 82 mg/L based on monitoring data from 2013-2014), but the relatively short monitoring record makes it difficult to determine the source of the elevated sulfate concentration observed on November 30, 2011.

## Biological Response

The effects of elevated conductivity on aquatic life were evaluated using data from Minnesota streams and scientific literature. A summary of this analysis is presented in Section 3.1.6. Based on this work, several biological metrics were selected to evaluate specific conductivity as a stressor in Elbow Creek (Table 85).

**Table 85:** Summary of biological metrics and literature used to evaluate elevated specific conductivity as a stressor

| Metric | Response to Increased Specific conductivity / Conductivity | Source |
|---|---|---|
| **EPT Richness** | Decrease | Roy et al (2003); Echols et al (2009); Johnson et al (2013) |
| **Overall Taxa Richness** | Decrease | Johnson et al (2013) |
| **Ephemeroptera Richness** | Decrease | Pond (2004); Hassel et al (2006) |
| **Fish and Macroinvertebrate Tolerance Indicator Values (TIV)** | Increase | MBDI (Yoder and Rankin, 2012) |

### EPT Richness

EPT taxa richness is extremely low at monitoring stations in the upper reaches of Elbow Creek. No EPT taxa were observed during two sampling visits to 98LS016 (Figure 292). At the next station downstream, 09LS082, only one EPT taxa was observed. The macroinvertebrate communities at these stations were dominated by taxa known to be tolerant of a wide range of stressors, including elevated specific conductivity. Dominant taxa observed include Oligochaeta (worms), a variety of tolerant midges (*Microspectra, Chironomus, Thienemannimia, Tanytarsini*), and *Simulium* (black fly larvae). Between 92-99% of the macroinvertebrates sampled from these two monitoring stations are considered "tolerant" of disturbance.

Taxa richness and relative abundance of EPT increase substantially at monitoring stations downstream of Elbow Lake (Figure 292). The average number of EPT taxa observed at stations 11LS073 and 09LS081 was 8 and 13, respectively. EPT taxa accounted for approximately 23 – 36% of the total taxa observed at these monitoring stations, compared to 0 – 2% at the sites upstream of Elbow Lake.

There is a clear shift in the macroinvertebrate community as Elbow Creek passes through Elbow Lake. As noted earlier, there is a similar shift with regards to a number of stressors that have been evaluated other than specific conductivity. Extreme conductivity spikes (> 800 µS/cm) have only been observed in the upper portions of Elbow Creek, where EPT taxa are essentially absent and the macroinvertebrate community is nearly entirely composed of tolerant taxa. Downstream of the lake, where conductivity levels are still above background conditions, but more moderate (300 – 600 µS/cm), EPT richness and relative abundance is more comparable to less impacted streams in the SLRW (Figure 295 and 296).

### Overall Macroinvertebrate Taxa Richness

Overall, macroinvertebrate taxa richness in Elbow Creek is lower than the majority of healthy streams in the SLRW (Figure 295). Taxa richness in Elbow Creek ranges from a low of 12 taxa (98LS016) to a high of 37 taxa (09LS081). In comparison, stations of the same MIBI class scoring above the impairment threshold have a median taxa richness of around 50 taxa. A longitudinal trend in taxa richness is apparent in Elbow Creek, and shows a similar pattern to EPT taxa richness. Stations with the lowest taxa richness (98LS016 and 09LS082) are located in the upper reaches of Elbow Creek. Station 98LS016, located just downstream of the Eveleth WWTP discharge, is particularly devoid of taxa (Figure 295).

Similar to observations made with EPT taxa, a lack of taxa richness is a clear symptom of impairment linked to the upper reaches of Elbow Creek. Establishing a linkage between this symptom and specific conductivity, as opposed to other candidate stressors, is difficult due to confounding data.

### Ephemeroptera Taxa Richness

Ephemeroptera (mayfly) taxa richness results vary considerably among biological monitoring stations, and shares a similar pattern with other biological indices (taxa richness, EPT taxa richness). Ephemeroptera were completely absent from the two monitoring stations upstream of Elbow Lake (Figure 292 and 296). Downstream of the lake, Ephemeroptera richness ranged from 5 taxa (11LS073) to 7-8 taxa (09LS081). Data for this biological metric provide further evidence for a stressor that is limited to the upper portion of the watershed upstream of Elbow Lake.

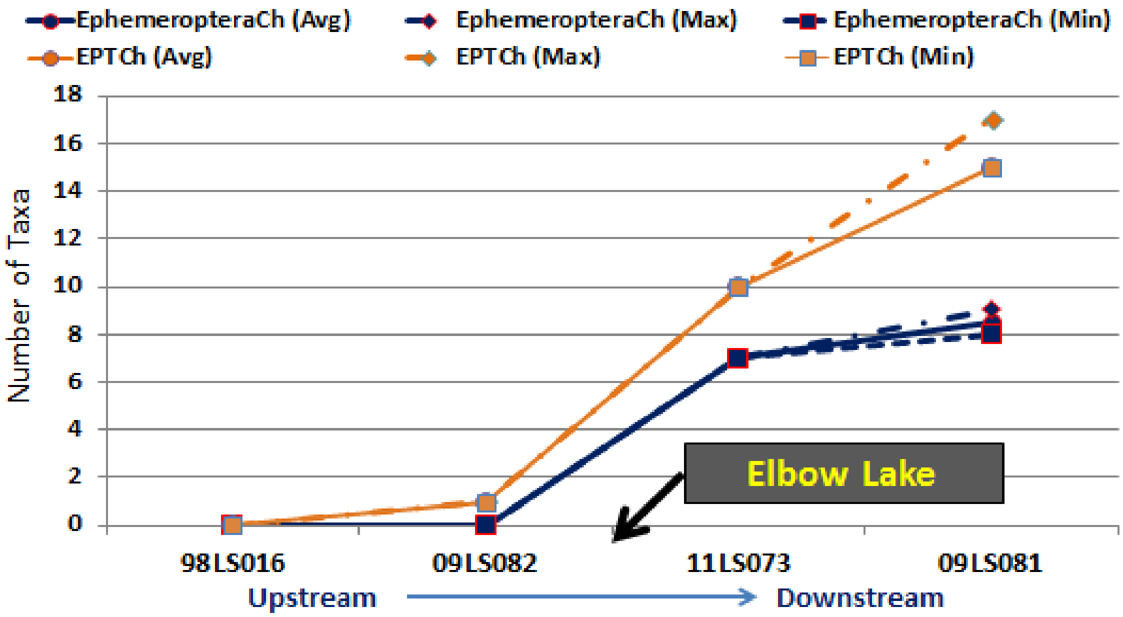


**Figure 292:** Results for EPT taxa richness and mayfly taxa richness shown longitudinally from left (upstream) to right (downstream)

### Specific Conductivity Tolerance Indicator Values

Community TIVs (refer to Section 4 for background on TIV) were calculated for fish and macroinvertebrate communities of Elbow Creek to evaluate community-level tolerance to specific conductivity. The macroinvertebrate community TIV show somewhat of a departure from the other metrics used to evaluate specific conductivity as a stressor, in that a station downstream of Elbow Lake (09LS081) shows the highest TIV value (most tolerant) of specific conductivity. Another inconsistency with this metric can be seen in 09LS082, which had the lowest TIV value of any site by a significant margin. However, the TIV result for 09LS082 is likely inaccurate due to the lack of taxa specific TIV for Oligochaeta, which was the most dominant taxa at this monitoring station. Oligochaetes (aquatic worms) are known to be tolerant of a wide variety of stressors, including specific conductivity. Community level TIV results for many of the monitoring visits were comparable to high quality monitoring stations in the SLRW (Figure 293).

Fish community TIV results for Elbow Creek show a similar pattern to many of the invertebrate metrics, with a clear separation of results between sites upstream and downstream of Elbow Lake. Results from stations downstream of the lake are comparable with high quality stations in the SLRW (Figure 294), which means that the fish community was primarily composed of species that are commonly observed in streams with low to moderate ionic strength. The presence of species such as Mottled Sculpin, Longnose Dace, and Iowa Darter at these monitoring stations factored into the more moderate TIV results. Upstream of the lake, fish community TIV results were higher than all of the class 4 stations scoring above the impairment threshold, and considerably higher than results from SLRW reference stations (Figure 294). Stations above the lake had abundant populations of tolerant species, such as Brook Stickleback, Fathead Minnow, and Central Mudminnow. These species are tolerant of a wide variety of stressors that are unrelated to specific conductivity (low DO, nitrate, habitat degradation), so their relative abundance is not necessarily diagnostic of specific conductivity as a stressor.

### Summary: Is elevated specific conductivity a stressor in Elbow Creek?

Specific conductivity (specific conductivity) is elevated above background levels throughout the Elbow Creek Watershed, particularly in reach upstream of Elbow Lake. Based on paired geochemistry and conductivity data, sulfate concentrations appear to be a major driver of specific conductivity in this watershed. Sulfate concentrations and specific conductivity levels are variable in this watershed, and aside from one set of samples collected in November 2011, both sulfate and conductance are lower in the Elbow Creek Watershed than in other mining affected watersheds (see Spring Mine Creek, Manganika Creek, Kinney Creek). Under 3% of specific conductivity measurements (2 of 80) exceeded 1,000 µS/cm, and these occurred during the same sampling event (November 2011). Extreme conductivity events are uncommon in this watershed, and appear to be short in duration when they do occur.

Biological metrics that were evaluated generally provided evidence in support of specific conductivity as a stressor in the upper watershed (impaired AUID 04010201-518), but due to the possibility of confounding stressors in that reach of Elbow Creek (nitrate and/or ammonia toxicity, low DO), specific conductivity cannot be fully diagnosed as a stressor based on the monitoring results. If specific conductivity levels were more consistently in the range of 800-1,000 µS/cm and above, a stronger case could be made for specific conductivity as a stressor above some of the other candidate causes that are being considered. The biological data are supportive of specific conductivity as a stressor, but there are some uncertainties as to whether or not the chemical component (specific conductivity) is severe enough to cause the effects that are occurring. A more convincing case exists for low DO and nitrate as causes of impairment in this reach.

In summary, increased specific conductivity should remain a candidate stressor for the upper reach of Elbow Creek, from the headwaters to Elbow Lake (AUID 04010201-518). Additional monitoring for specific conductance within this reach may help to further explain the conditions and sources that contribute to occasional conductivity levels that exceed 1,600 µS/cm.

There is suitable evidence to eliminate specific conductivity as a candidate cause for impairment in the lower reach of Elbow Creek, from the outlet of Elbow Lake to its confluence with the St. Louis River (AUID 04010201-570). Conductivity levels within this reach are predominantly in the range of 200- 500 µS/cm, a range that is very unlikely to result in harmful conditions for most fish and macroinvertebrate taxa. Biological data from this reach is also provides solid evidence for eliminating specific conductivity as a candidate cause. Sensitive fish species such as Longnose Dace, Iowa Darter, and Mottled Sculpin were sampled in this reach. A relatively high number of EPT macroinvertebrate taxa were also present in this reach.

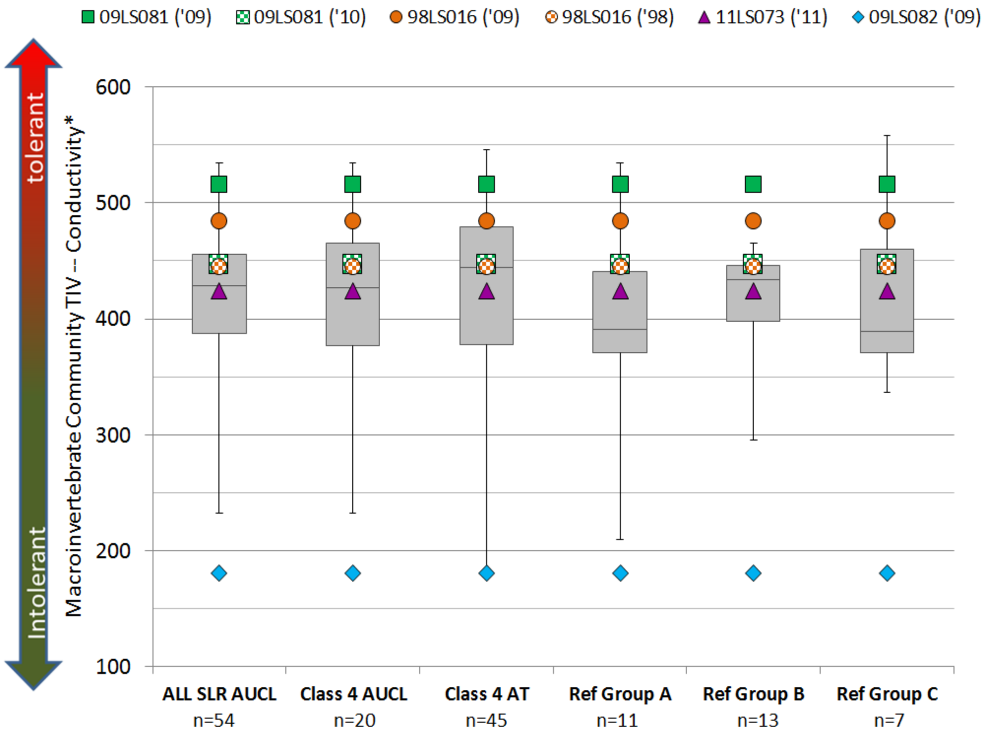


**Figure 293:** Community level TIV values for specific conductivity in Elbow Creek compared to SLRW reference sites. * See section 1.2.3 for list or reference stations. See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL** = Above Upper Confidence Limit of FIBI threshold **AT** = Above Fish IBI Threshold

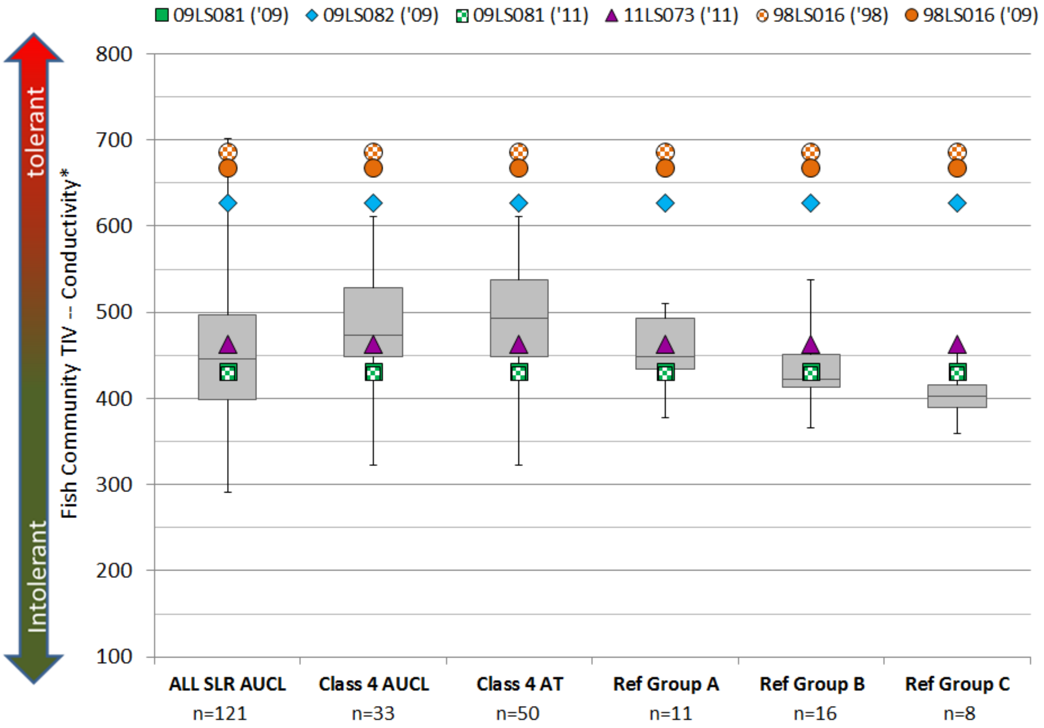


**Figure 294:** Community level TIV values for specific conductivity in Elbow Creek compared to SLRW reference sites. * See section 1.2.3 for list or reference stations. See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL** = Above Upper Confidence Limit of FIBI threshold **AT** = Above Fish IBI Threshold

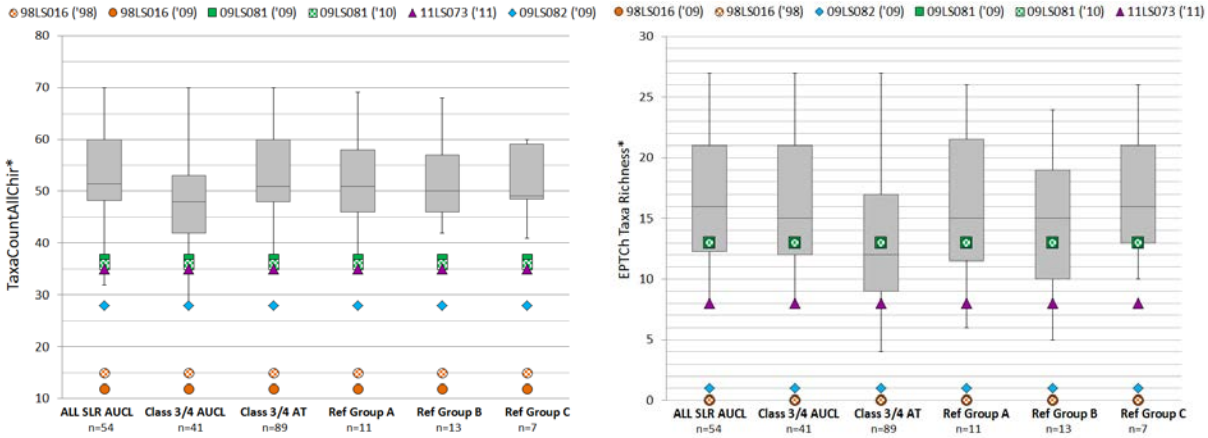


**Figure 295:** Overall taxa richness and EPT taxa richness at Elbow Creek monitoring stations compared to SLRW reference sites. * See section 1.2.3 for list or reference stations. See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

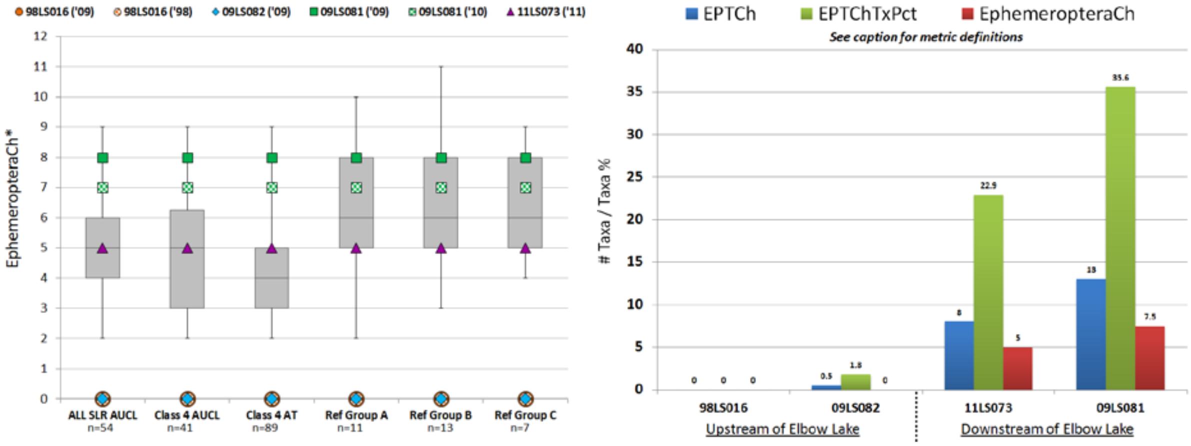


**Figure 296:** Mayfly richness in Elbow Creek compared to high quality reference stations (left). * See section 1.2.3 for list or reference stations. See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

## 5.19.4 Nitrate Toxicity

Nitrate toxicity was identified as a candidate cause for impairment in Elbow Creek based on existing water chemistry data. Elbow Creek has been sampled for nitrate numerous times (n=44) between the years 1986-2012 at seven monitoring stations. The vast majority of the sampling results have yielded low to very low nitrate concentrations. However, elevated nitrate concentrations (4 – 6 mg/L) have been observed at station S001-067 each time this station was sampled. This station is co-located with biological monitoring site 98LS016 which scored below the MIBI impairment threshold. These monitoring sites are approximately 500 feet downstream of the main discharge point for the city of Eveleth’s WWTP, which is likely the primary source of nitrate entering Elbow Creek.

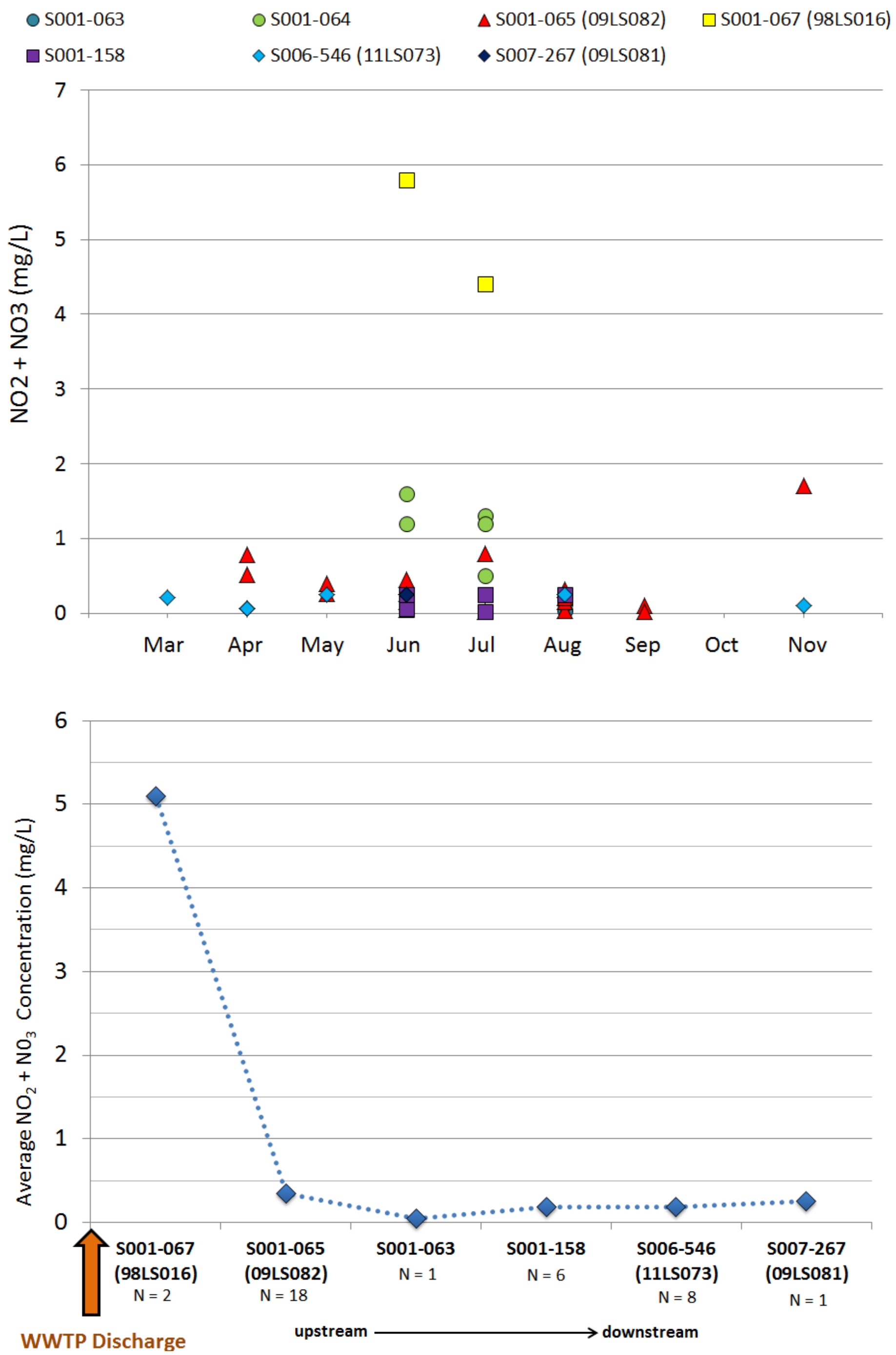


**Figure 297:** Nitrate results from Elbow Creek arranged by monitoring station and month (top). Longitudinal summary of nitrate results showing higher concentrations immediately downstream of Eveleth WWTP (bottom)

### Sources and Pathways of Nitrate

Effluent from the Eveleth WWTP is believed be to the only significant source of nitrate within the Elbow Creek Watershed. Elbow Creek has historically received mine pit dewatering flow from numerous tributaries, but many of these inputs have ceased or decreased in recent years, and are not likely introducing nitrate to the stream in high concentrations. Other common sources of nitrate, such as agricultural or urban land-uses, are not prevalent within the Elbow Creek Watershed.

The Eveleth WWTP currently discharges to Elbow Creek via an NPDES permit which requires regular effluent monitoring for various parameters, including flow and nitrate nitrogen. Data for these parameters are only available for the 2013 and 2014 calendar years. The monitoring results show an average effluent discharge from the WWTP of roughly 0.4 million gallons per day (mgd), which equates to about 0.6 cubic feet per second (cfs) of continuous discharge entering Elbow Creek. Calendar month maximum discharges are roughly 0.9 cfs on average, and have been recorded as high as 1.5 cfs (May 2014). Sampling for nitrate + nitrite (nitrate) is reported as a "calendar month average" in the data report from the WWTP. Data are only available for April and September for both 2013 and 2014. The results for the April sampling range from 4.8 to 5.0 mg/L, while the September results are significantly higher, ranging from 9.3 to 15.0 mg/L.

These results confirm that WWTP effluent is the primary driver of elevated nitrate levels in the upper reaches of Elbow Creek. Although the overall volume of effluent is rather small, nitrate concentrations in the effluent are elevated enough to be a concern for the protection of sensitive aquatic biota. Streamflow data are not available for this reach, and therefore the percentage of baseflow that can be attributed to WWTP effluent cannot be accurately calculated. The drainage area of station 98LS016 in less than 2 square miles and baseflow is estimated to be 1-2 cfs based on visual estimates made during site visits. Based on the elevated nitrate levels and the low flow conditions observed in this reach, it can be concluded that effluent from the WWTP has a significant impact on the quantity and quality of water in the headwaters of Elbow Creek.

**Figure 298:** Elbow Creek just downstream of the Eveleth WWTP outfall, looking upstream from CR 101 (left) and downstream of CR 101 (right).

## Biological Response to Nitrate

Macroinvertebrate data from four biological monitoring stations are available to evaluate the potential impact of elevated nitrate concentrations in Elbow Creek. All four stations are located downstream of the WWTP influence, but several are located several miles downstream and have expansive lakes and/or wetlands buffering them from the effluent discharge. The distance of each monitoring site from the WWTP discharge point, as well as any features that could ameliorate the impacts of the effluent on aquatic life, are listed in Table 86. For more information on these monitoring sites, including a detailed map of site locations, refer to the map in Figure 277.

Nitrate tolerance indicator values (NTIV) have been developed by the MPCA biologists for most fish and macroinvertebrate taxa that are found in the streams and rivers of Minnesota. For more information on NTIV development, refer to Section 4. Individual and community based NTIV values will be used in this section to evaluate the degree to of nitrate tolerance exhibited by fish and macroinvertebrate taxa in Elbow Creek.

**Table 86:** Biological monitoring stations on Elbow Creek and relative location to WWTP outfall

| Station | Distance from WWTP | Comments |
|---|---|---|
| 98LS016 | 500 feet | Station is immediately downstream of WWTP discharge |
| 09LS082 | 2.3 miles | Large wetland complex upstream of monitoring station |
| 11LS073 | 9.3 miles | 3.3 miles downstream of Elbow Lake (165 acres) and several wetland complexes |
| 09LS081 | 10.5 miles | Land-use surrounding site is agricultural (animal pastures and hayfields) |

### Macroinvertebrate Response to Nitrate

The percent of macroinvertebrate individuals tolerant to elevated nitrate levels was calculated for each Elbow Creek monitoring site and compared to set of results from a statewide and regional data set (Figure 299). The results for Elbow Creek were highly variable among the four monitoring stations, and some variability was also observed between multiple visits to the same monitoring stations. The highest percentage of nitrate tolerant organisms was observed during sampling visits to stations 09LS081 (2009 sample), 09LS082 (2009 sample), and 98LS016 (1998 sample), in which just under 60% of the total macroinvertebrate community were tolerant to nitrate. The relative abundance of nitrate tolerant organisms observed during these visits was slightly higher than the statewide median of 53%, and substantially higher than the median values for regional monitoring stations and those within the same MIBI class.

However, repeat visits to two of these stations produced very different results. Percent nitrate tolerant individuals dropped to 25% (98LS016, 2009 sample) and 18% (09LS081, 2010 sample) during the other sampling events. Both of these results are well below the median values for comparison sites (Figure 299). This level of variability calls into question nitrate as a stressor considering that the source of nitrate in this watershed is a continuous discharge from a WWTP. Only one monitoring visit was made to station 11LS073. This station had the lowest percent of nitrate tolerant organisms (8%). Nitrate concentrations at this location have routinely been less than 0.5 mg/L

Macroinvertebrate NTIV results for the six sampling visits to Elbow Creek show a similar pattern (Figure 300). Station 98LS016, which is the monitoring station closest to the WWTP discharge, had the highest NTIV scores. The result from the 1998 visit to this station is one of the highest observed in its MIBI class, the Lake Superior Basin, and the St. Louis River 8 HUC watershed. A more recent visit to this station resulted in a lower NTIV score, although it was still higher than 75% of the results from the northern coldwater MIBI class. The NTIV scores were also somewhat elevated at stations 09LS82 and 09LS81, but these scores were not exceptionally high, and were more comparable to the statewide and regional median values.

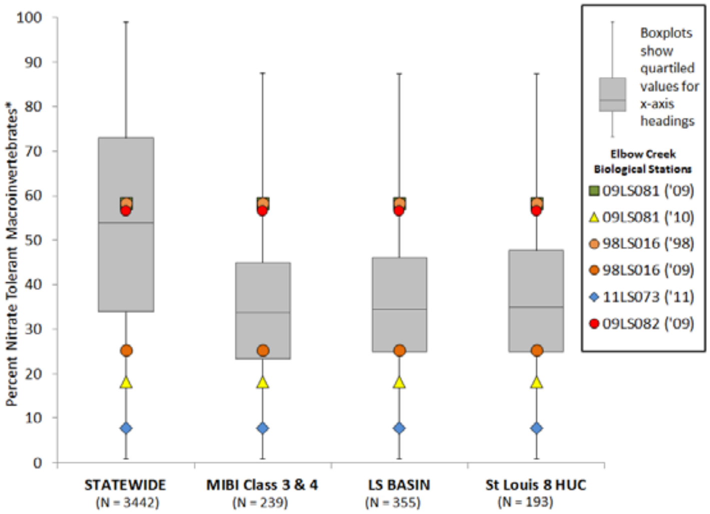


**Figure 299:** Percent nitrate tolerant macroinvertebrates at Elbow Creek monitoring stations compared to statewide and regional monitoring stations. **SLR=** St. Louis River Watershed **LS Basin =** Lake Superior Drainage Basin

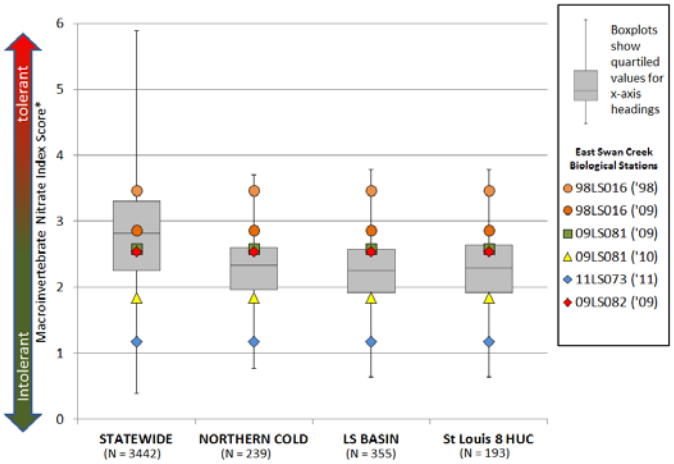


**Figure 300:** Nitrogen tolerance indicator value (NTIV) for Elbow Creek monitoring sites compared to statewide and regional monitoring stations. **SLR=** St. Louis River Watershed **LS Basin =** Lake Superior Drainage Basin

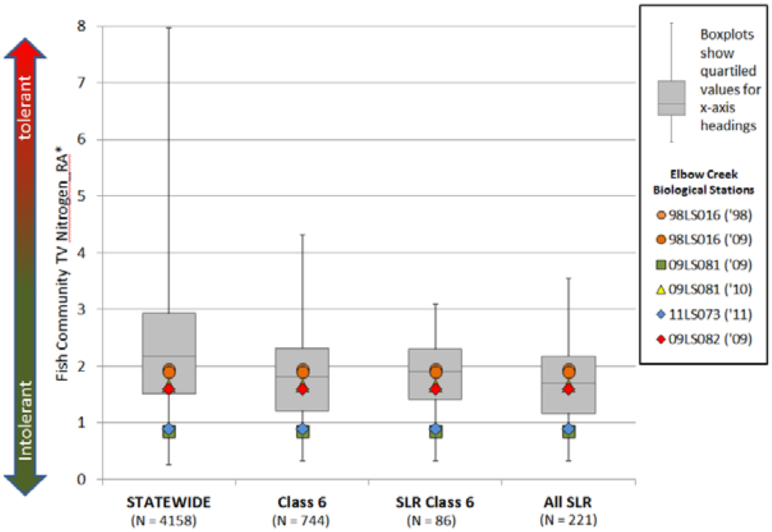


**Figure 301:** Nitrogen tolerance indicator value (NTIV) results for the fish community of Elbow Creek compared to results from statewide and regional monitoring stations. **SLR=** St. Louis River Watershed **LS Basin =** Lake Superior Drainage Basin

**Fish Response to Nitrate**

To evaluate nitrate as a stressor to the fish community of Elbow Creek, community TIV scores based on the relative abundance of nitrogen tolerant fish species were compared to a statewide and regional data set (Figure 301). The community TIV results for Elbow Creek are comparable to or below the median score observed at other class 6 FIBI sites within and outside of the SLRW. In other words, the fish community of Elbow Creek is no more tolerant of elevated nitrate concentrations than other stations in the region where nitrate concentrations are likely a lot lower. Based on these results, it is unlikely that the fish impairment in the upper reaches of Elbow Creek is caused by nitrate toxicity.

**Summary: Is nitrate toxicity a stressor in Elbow Creek?**

Nitrate concentrations are elevated (> 5 mg/L) in the upper reaches of Elbow Creek downstream of the Eveleth WWTP discharge, near biological monitoring station 98LS016. Fairly extensive sampling several miles downstream of the WWTP to the mouth of Elbow Creek shows considerably lower nitrate concentrations. The decrease in nitrate is likely the result of dilution and/or uptake of nitrogen as the stream passes through Elbow Lake or the numerous wetlands complexes along its length. Monitoring data collected per the NPDES permit for the WWTP shows calendar month average nitrate concentrations as high as 15 mg/L, which could impact sensitive aquatic life downstream of the discharge, particularly during low flow periods.

The macroinvertebrate data provide moderate support of nitrate as a stressor. Station 98LS016, located just downstream of the WWTP discharge, received much lower MIBI scores than the rest of the monitoring stations along Elbow Creek, which is an indication that upper Elbow Creek is impacted by a stressor that is not present further downstream. In addition, nitrate TIV scores for 98LS016 were higher than other Elbow Creek locations, as well as the majority of SLRW stations and stations of the same MIBI class. Nitrate toxicity cannot be ruled out as a stressor for the upper impaired reach of Elbow Creek (AUID 04010201-518). Further nitrate monitoring is recommended along this AUID, and this stream should be re-assessed when a nitrate WQ standard is available.

### 5.19.5 Ammonia Toxicity

Ammonia toxicity was identified as a potential cause of biological impairment in Elbow Creek because it is the receiving water for the city of Eveleth's WWTP effluent. Elevated levels of ammonia nitrogen have been observed in the effluent being discharged to Elbow Creek and at several monitoring stations in the upper reaches of the watershed. Contemporary and historical monitoring results for unionized ammonia concentrations in Elbow Creek are shown in Figure 302. The majority of the sampling results are very low (less than 1 µg/L), particularly the results from contemporary monitoring efforts carried out between the years of 2009 – 2012. The single historical (1986) sampling result at station S001-067 was 22 µg/L, which is an indication that unionized ammonia concentrations in the upper reaches of Elbow Creek were much higher in the past. Contemporary sampling results from stations in the upper reaches of Elbow Creek (S001-067 and S001-065) show concentrations that are still somewhat elevated compared to most SLRW streams, but they remain well below the 40 µg/L CS for protecting aquatic life.

Based on these results, it is unlikely that elevated unionized ammonia concentrations are a present day contributor to biological impairments in this stream.

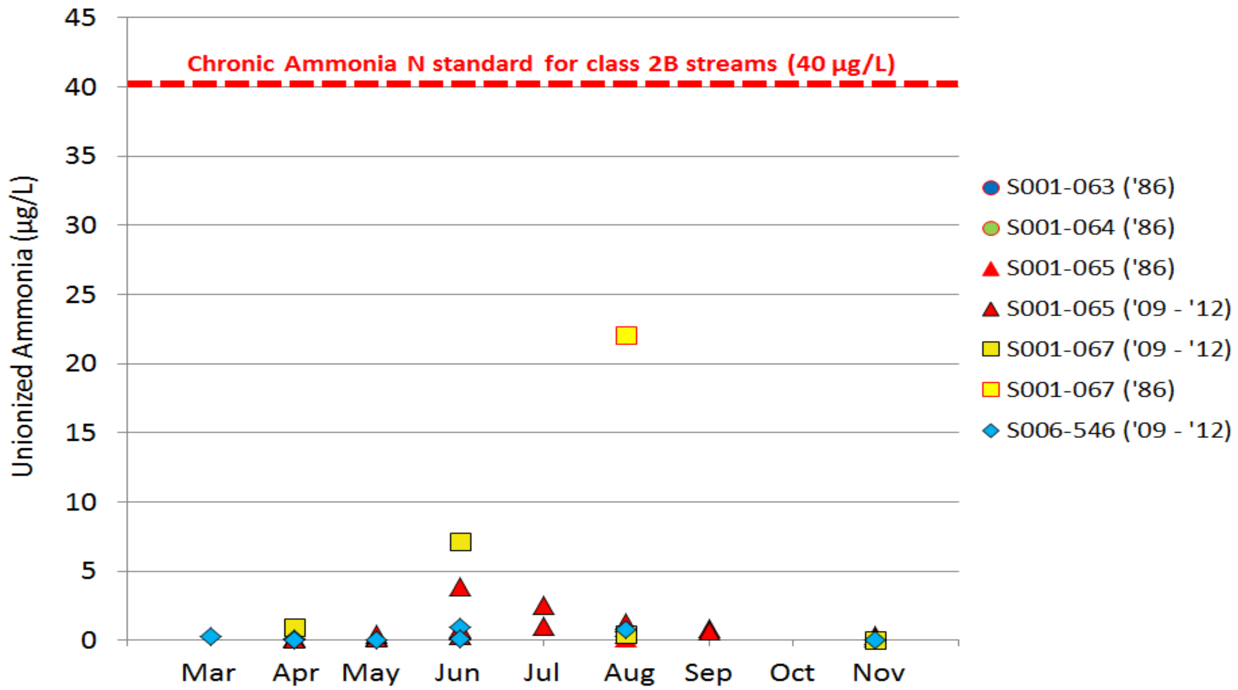


**Figure 302:** Unionized ammonia nitrogen results observed at Elbow Creek monitoring stations, arranged by month

### 5.19.6 Lower Impaired Reach of Elbow Creek (04010201-570)

An analysis of available water quality (WQ) data for this impaired reach did not result in any definitive candidate causes for impairment. Although data are somewhat limited for this segment of Elbow Creek, results for all of the WQ parameters evaluated seemed to be within suitable ranges for supporting healthy fish and macroinvertebrate communities. TP concentrations were fairly high in one of two available samples (0.104 mg/L, June 23, 2009), but DO measurements do not show any minimum concentrations below WQ standards, and DO flux was within a suitable range during a continuous monitoring period in August of 2012. Several parameters which could be impacting the upper impaired reach, such as low DO, elevated specific conductivity, and nitrate toxicity, show no signs of being problematic in the lower impaired reach. Biological data support this claim, as FIBI scores were above IBI impairment threshold, and the EPT macroinvertebrate taxa were abundant at the biological monitoring station.

The individual MIBI metric scores from impaired biological monitoring station (09LS081) on this reach are shown in Figure 303. Low scores in several metrics are driving down the overall MIBI score at this location, particularly the “Plecoptera” (taxa richness of stoneflies) and “Predator” (taxa richness of predators) metrics. Many stonefly taxa are considered predators, and the scores for metric “Odonata” (taxa richness of dragonflies – another common predator) are fairly good. Thus, the low scores in both of these metrics are largely the result of a lack of stonefly taxa observed at this station. Other metrics that scored poorly during the initial 2009 sample include “ClimberCh” (richness of “climber” taxa), “ClingerChTxPct” (percentage of taxa that are “clingers”), and “Trichoptera” (taxa richness of caddisflies).

Metrics involving stonefly and caddisfly richness and “clinger” macroinvertebrate abundance can all be negatively impacted by degraded physical habitat conditions. Given the lack of candidate causes that emerged from the analysis of WQ variables, the physical habitat conditions of station 09LS081 will be evaluated in further detail in this section as a potential candidate cause of MIBI impairment.

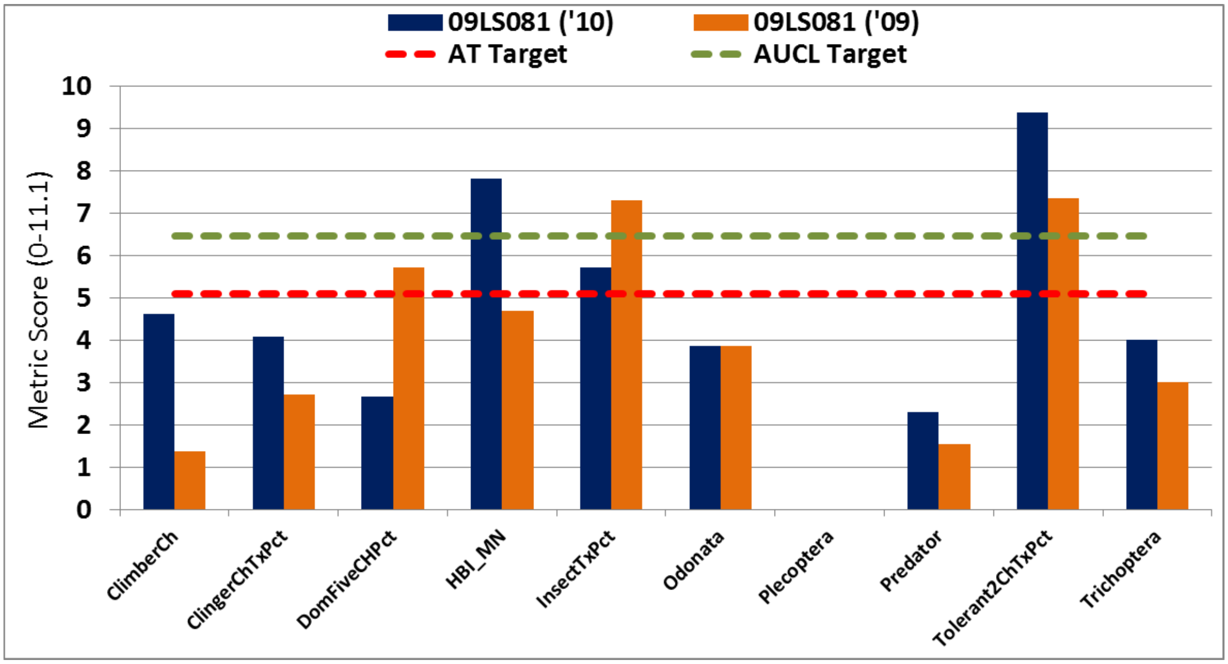


**Figure 303:** Individual MIBI metric scores for station 09LS081. The “AT Target” represents the score needed in each metric to produce a cumulative MIBI score above the impairment threshold (51). The “AUCL Target” represents the score needed in each metric to produce a cumulative MIBI score above the upper confidence limit of the class 3 MIBI impairment threshold (64.6)

## Degraded Physical Habitat Conditions

Physical habitat conditions in the lower reach of Elbow Creek were evaluated at station 09LS081 using the MSHA methodology. Two habitat assessments were performed using these protocols, one during the initial visit in 2009 and the other during a follow up visit in 2011. An overall MSHA score of 77 out of 100 resulted from both visits to this station, which corresponds to a habitat rating of “good.” However, there are several sub-category scores that suggest that several components of the physical habitat may be somewhat limited for supporting a healthy macroinvertebrate assemblage. Based on the MSHA results and observations made during several site visits, coarser grained substrates (cobble and gravel) at station 09LS081 are moderately embedded by sand and other fine particles. The MSHA substrate component score from the 2011 visit to this station was near the 25th percentile value for class 3 MIBI stations in the SLRW (Figure 304), which reflects the moderate level of substrate embeddedness at this station. Results from the 2009 assessment were slightly better. Excessive deposition of fine sediment can degrade macroinvertebrate habitat quality, reducing productivity and altering the community composition (Rabeni et al. 2005, Burdon et al. 2013).

The MSHA scores for the “channel stability” metric also provide some evidence of habitat degradation at station 09LS081. The 2011 assessment categorized the stream channel as moderately unstable due some areas of minor bank erosion. Areas of the stream channel also appear to be incising (downcutting) and widening based on photographs collected within the biological monitoring reach. Figure 305 shows an example of a widened reach during baseflow conditions, and another cross section within the station with stagnant flow and poor local substrate conditions. More investigation is required to determine the cause of instability within this reach, but poor riparian buffers may be one of the factors involved. The impaired station (09LS081) is set within a fairly expansive pasture which extends for approximately 0.4 mile upstream of the station. Although areas of this pasture are well buffered from the stream, there are several areas where a narrow vegetated buffer is present (>10 ft) or a buffer is absent (Figure 306).

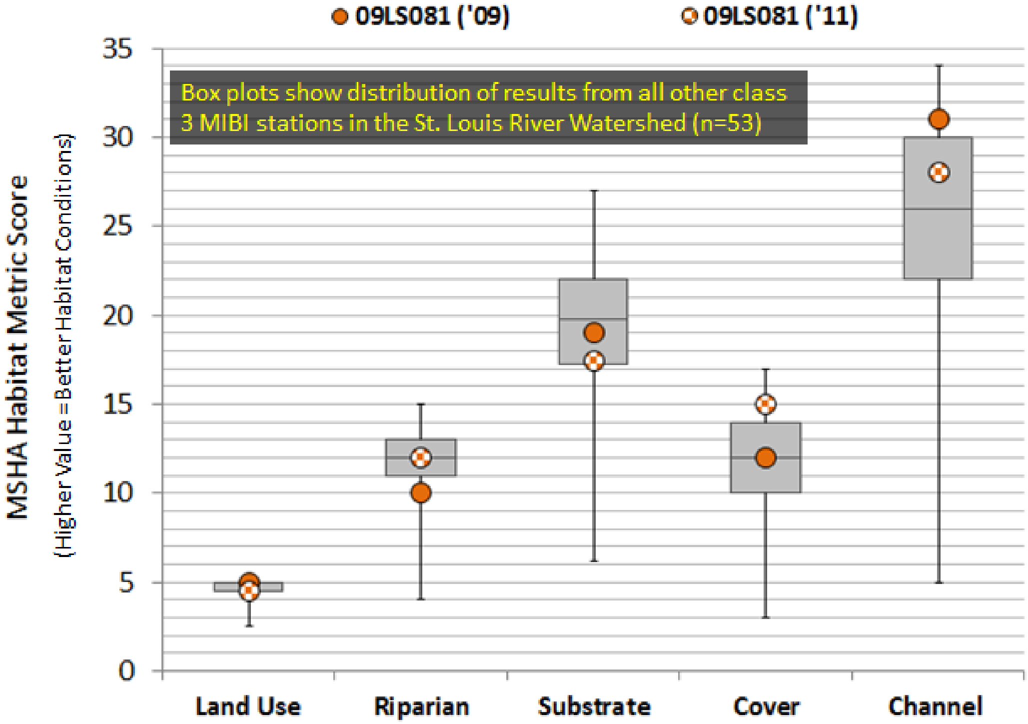


**Figure 304:** MSHA metric scores for Otter Creek station 09LS005 compared to the box plot distribution of results from all other class 8 (Northern Coldwater) stations in the SLRW.

**Figure 305:** Wide, shallow riffle at 09LS081 (left) and low gradient, stagnant conditions near the culvert at HWY 16 within station 09LS081

**Figure 306:** Aerial photo of landscape around station 09LS081. Hay/pasture lands border this monitoring station and along the stream corridor for approximately ½ mile upstream. Riparian buffer conditions are fairly poor along the biological monitoring reach (inset photo on right).

## Biological Response to Degraded Habitat Conditions

Macroinvertebrate taxa with specific feeding or other life history traits that require clean, coarse substrates are often the first to decrease in richness and abundance in streams with high rates of sedimentation. The following macroinvertebrate metrics cover some of the more sensitive taxa that have shown fairly predictable responses in streams with high rates of embeddedness or those dominated by fine substrate (silt/clay/sand).

### % Clinger Individuals and Clinger Taxa %

Clinger macroinvertebrates usually have flattened body forms and attach themselves to firm substrates (mostly rocks, wood) in swift water habitats. The relative percentage of clinger taxa observed during the two sampling visits to station 09LS081 ranged from 15 – 33%. Overall, this station supported a lower percentage of clinger individuals than the majority of the class 3 monitoring stations in the SLRW that scored above the MIBI impairment threshold (high quality sites) (Figure 307). The 2009 sample (15% clinger individuals) in particular represented a very low relative percentage of clinger individuals compared to high quality stations.

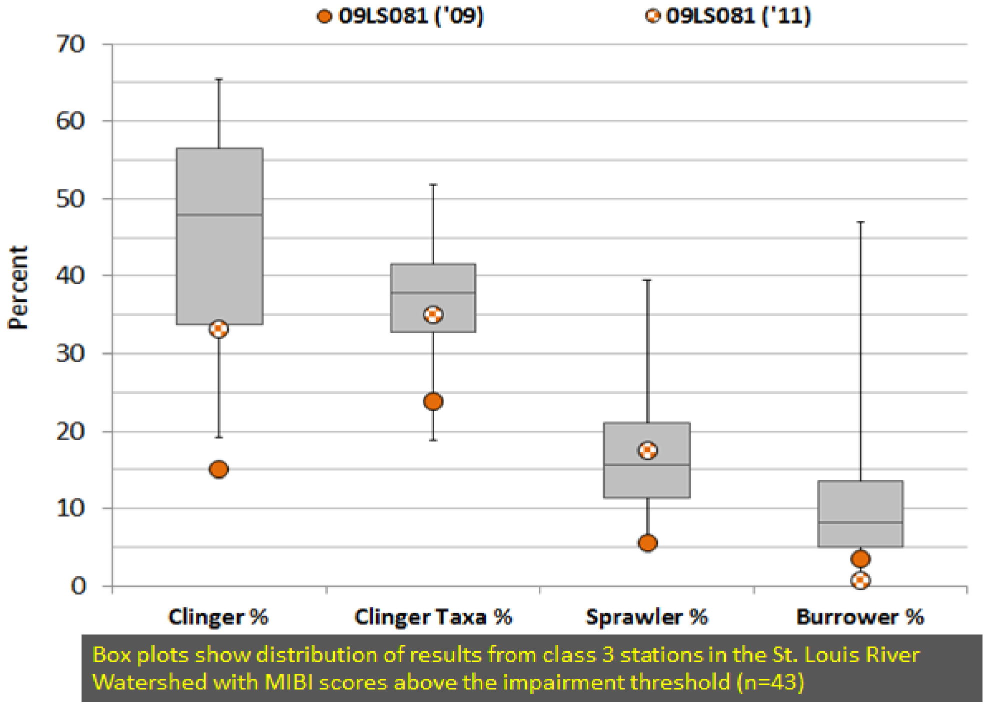
The relative percentage of clinger *taxa* was also lower at this station compared to the majority of class 3 stations in the SLRW (Figure 307). The results for these two metrics support substrate embeddedness and poor substrate conditions as a contributing cause to the MIBI impairment within this reach.

### % Sprawler Individuals

Sprawler macroinvertebrates live on the surface of floating aquatic plants or fine sediments, and usually possess adaptations for staying on top of substrate and keeping respiratory surfaces free of silt. Sprawler individuals accounted for 5% and 18% of the total macroinvertebrate community during the two sampling events at station 09LS081. These values are relatively low in comparison to results from class 3 stations scoring above the MIBI threshold. It can be concluded that the relative percent of sprawler individuals is comparable to high quality stations.

### % Burrower Individuals

Burrower macroinvertebrates inhabit the fine sediments of streams and lakes. A stream dominated by burrower individuals or taxa can be a good indicator that a stream reach is dominated by fines and lacks quality coarse substrate. Only 1% and 4% of the total macroinvertebrate community were “burrowers” in the two sampling events at station 09LS081. These results are quite low in comparison to high quality stations of the same MIBI class, and were generally lower than the majority of high quality stations in the SLRW. Station 09LS081 was not dominated by silt and sand substrates throughout the reach, and the low percentage of burrower taxa is further evidence of this. It appears that most of the habitat degradation related to sedimentation in Elbow Creek, if any, is occurring through the filling of interstitial spaces in riffle and glide areas (embedded coarse substrates).



Box plots show distribution of results from class 3 stations in the St. Louis River Watershed with MIBI scores above the impairment threshold (n=43)

**Figure 307:** Results for select MIBI metrics observed at Elbow Creek station 09LS081 compared to class 3 stations scoring above the MIBI impairment threshold

### Summary: Are poor physical habitat conditions a stressor in lower Elbow Creek?

Overall, physical habitat conditions in the lower impaired reach of Elbow Creek are in decent condition, but several characteristics of the physical habitat may be limiting MIBI scores. The MSHA results from station 09LS081 show moderate levels of substrate embeddedness and some bank erosion, which could be negatively influencing several of the MIBI metrics that scored poorly (Clinger Taxa Richness &

Plecoptera Taxa Richness). Riparian conditions bordering the impaired monitoring station are not ideal. Vegetated buffers between the stream channel and adjacent pasture/hay land are narrow and lacking species diversity and rigor. The culvert crossing at Highway 16 (Town Line Road) at the downstream end of the biological monitoring station may also be impacting physical habitat conditions within the impaired reach. Photographs of the culvert show a large, vegetated bar blocking one of the culverts on the upstream side. This deposited material may be impacting sediment transport through this reach and leading to a flatter stream slope and stagnant flow conditions just upstream of this crossing. Further investigation of this issue is needed to verify these impacts.

Given the lack of WQ issues detected in this impaired reach, restoration activities should focus on improving physical habitat conditions. Poor physical habitat conditions should be considered the leading cause of impairment in this reach based on the lack of other apparent stressors.

### 5.19.7 Elbow Creek: Summary of Stressors to Aquatic Life

**Table 87:** Summary of SID results for Upper Elbow Creek

| Elbow Creek Impaired AUID 04010201-518 | |
|---|---|
| Candidate Cause | Result |
| **Low Dissolved Oxygen** | ● |
| **Nitrate Toxicity** | ○ |
| **Ammonia Toxicity** | **X** |
| **Sulfate Toxicity** | ○ |
| **Elevated Specific conductivity** | ● |
| Key: **● = confirmed stressor ○ = Potential Stressor X = eliminated candidate cause** | |

**Table 87:** Summary of SID results for Upper Elbow Creek

| Elbow Creek Impaired AUID 04010201-570 | |
|---|---|
| Candidate Cause | Result |
| **Low Dissolved Oxygen** | **X** |
| **Nitrate Toxicity** | **X** |
| **Ammonia Toxicity** | **X** |
| **Sulfate Toxicity** | **X** |
| **Elevated Specific conductivity** | **X** |
| **Poor Physical Habitat** | ● |
| Key: **● = confirmed stressor ○ = Potential Stressor X = eliminated candidate cause** | |

### West Two - McQuade Moraines Watershed Zone

This watershed zone includes two impaired streams, the West Two River below West Two Reservoir, and a tributary to McQuade Lake (Kinney Creek). West Two Reservoir was created in 1964 by U.S. Steel Company and is now listed as impaired for elevated nutrient concentrations. The impoundment is currently permitted to release a minimum of 3 cubic feet per second (cfs) of streamflow into the impaired reach of the West Two River. Headwaters streams in both of these watersheds have been removed or reduced due to the presence of mining land-use. The loss of these headwaters streams, as well as current mine-pit dewatering and impounded reservoirs, has altered the hydrological regime of these streams from their natural state.

**Figure 308:** Photos of impaired stream reaches in the West Two McQuade Moraines Watershed zone. West Two River (upper left and upper right) and Unnamed Tributary to McQuade Lake (a.k.a “Kinney Creek”) (below)

## 5.20 West Two River

The West Two River begins in the iron mining region between Mountain Iron and Kinney. Much of the upper reaches are dominated by E-type (Rosgen 1994) channels in wetland-dominated lacustrine valleys. A considerable amount of the stream channel in the upper reaches has also been inundated by the West Two Reservoir. The six-mile long impaired reach is located in the middle of the watershed, and is predominantly a C or E type channel through a lacustrine valley. Approximately one mile of this reach flows through a more confined glacial trough valley. The average slope of the impaired AUID is less than 0.1%. The lower reaches of the West Two River flow across the Glacial Lake Upham basin, although it has cut a new alluvial valley at a lower elevation than the historic lake bed.

Fish and macroinvertebrate data were collected at three monitoring stations (Table 88). The most upstream station, 09LS075, is located immediately downstream of the West Two Reservoir. Data collected from this site in the fall of 2009 led to the macroinvertebrate impairment listing. The other two monitoring stations (12LS002 and 09LS073) are located further downstream and sampling visits to these stations resulted in MIBI scores that were more favorable and did not indicate an impaired condition. As a result, the impaired reach is limited to a 5.5-mile reach downstream of the reservoir outlet (Figure 309).

The macroinvertebrate community at the impaired sampling location (09LS075) was dominated by Hirudinea (leeches), which accounted for over 1/3 of the individuals counted. Also common were individuals from the Chironomid genera *Tanytarsus* and *Dicrotendipes*, which are known to be tolerant of streams with elevated nutrient concentrations, low DO, and predominantly fine substrates. Only six EPT taxa were present at this station, and individuals from the EPT order of insects accounted for a relatively low percentage of the overall community (6%). By comparison, a total of 9-10 EPT taxa were observed at the downstream monitoring locations, and EPT individuals comprised 26-28% of the total population at these sites.

**Table 88:** Overview of West Two River biological monitoring station and MIBI results compared to applicable standards

| Macroinvertebrate Assessments | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | Invert IBI Class | Invert IBI Result *(visit year)* | Invert IBI Result *(visit year)* | Standard | IBI Lower Confidence Limit | IBI Upper Confidence Limit |
| 09LS075 | 33.48 | 0.11 | 3 | 4 | **34.00** *(2009)* | - | 51 | 37.40 | 64.60 |
| 12LS002 | 63.79 | 0.07 | 3 | 4 | **55.41** *(2012)* | - | 51 | 37.40 | 64.60 |
| 09LS073 | 77.81 | 0.06 | 4 | 3 | **61.15** *(2009)* | - | 53 | 40.40 | 65.60 |

### Candidate Causes for Impairment

A review of existing data was conducted in order to identify a set of candidate causes for the MIBI impairment on the West Two River. The followign candidate caueses were selected for detailed analysis in this report;

1. **Low DO / High DO Flux**
2. **Sulfate Toxicity**
3. **Elevated Specific conductivity**

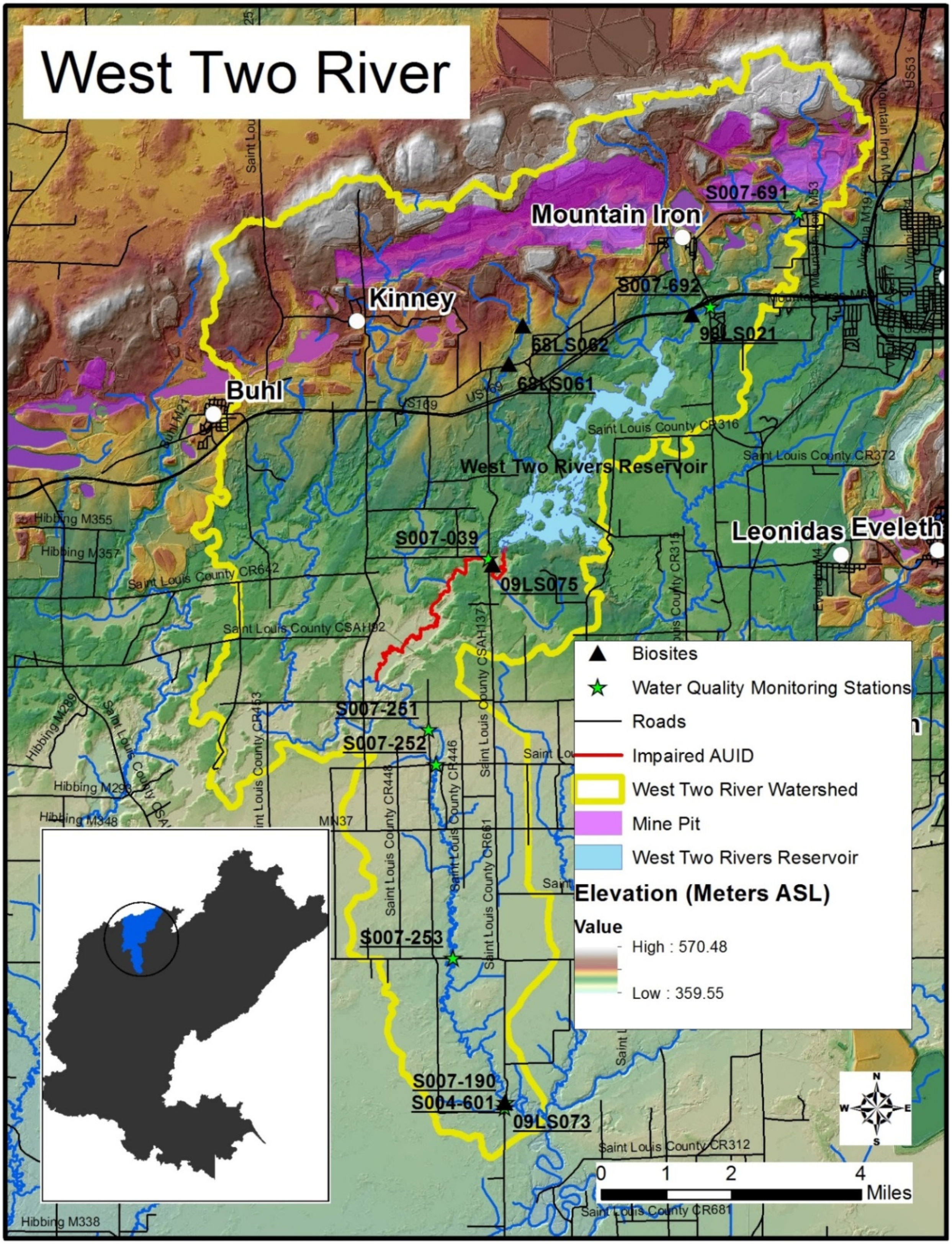


Figure 309: West Two River Watershed, impaired reach, and monitoring stations

### 5.20.1 Low Dissolved Oxygen

The DO was identified as a candidate cause for MIBI impairment on the impaired reach of West Two River. Available DO data include instantaneous (point) measurements, longitudinal synoptic measurements, and continuous monitoring data collected for approximately one week on two separate occasions. Data collected during point measurements are displayed in Figure 310. Only 6% of the DO readings collected from West Two River fell below the 5 mg/L water quality standard, all of which were from the biological monitoring station within the impaired reach (09LS075). DO concentrations as low as 1.74 mg/L were recorded at this monitoring station.

Synoptic longitudinal monitoring of DO was completed at five monitoring stations on August 14, 2012. Two sampling runs were performed, one in the early morning before sunrise, and another in the late afternoon/evening hours. The timing of these sampling runs was designed to estimate diurnal fluctuation (DO flux) in DO concentrations. Minimum DO concentrations increased from in a downstream direction, with the lowest concentration (and only sub 5 mg/L measurement) observed at biological monitoring station 09LS075, which is located just downstream from West Two Reservoir. DO flux was significantly higher at station 09LS075 compared to the other four monitoring stations downstream. A nearly 10 mg/L difference in DO was observed at 09LS075 between the morning and evening measurements (Figure 311). The other monitoring stations exhibited an average change of around 2 mg/L.

Continuous monitoring equipment was deployed at two biological monitoring stations, 09LS075 and 09LS073, during low flow conditions in 2012 and 2013. Based on the resulting DO profiles, it is clear that the DO regime at these two monitoring stations is drastically different in terms of both minimum DO concentrations and DO flux (Figure 312, 313). Just downstream from West Two Reservoir at station 09LS075, maximum DO flux observed during continuous monitoring ranged from 7.5 to 11 mg/L, and minimum DO concentrations dropped below the 5 mg/L water quality standard on a daily basis. Approximately 11 miles downstream at station 09LS073, maximum DO flux observed was less than 2 mg/L, and the DO minimums were all well above the water quality standard. A comparison of DO flux between the two monitoring stations is shown in Figure 311. Proximity to the eutrophic West Two Reservoir, higher water temperatures, and abundance of aquatic macrophytes are all factors that are likely resulting in higher DO flux at station 09LS075.

**Figure 309a:** Photos of biological monitoring stations 09LS075 and 09LS073. Station 09LS075 is located just downstream of the eutrophic West Two Reservoir, and has a riparian dominated by wetlands. Station 09LS073 is located 11 miles downstream and cuts through an alluvial valley. These differences are important factors in the DO regime observed at the two stations.

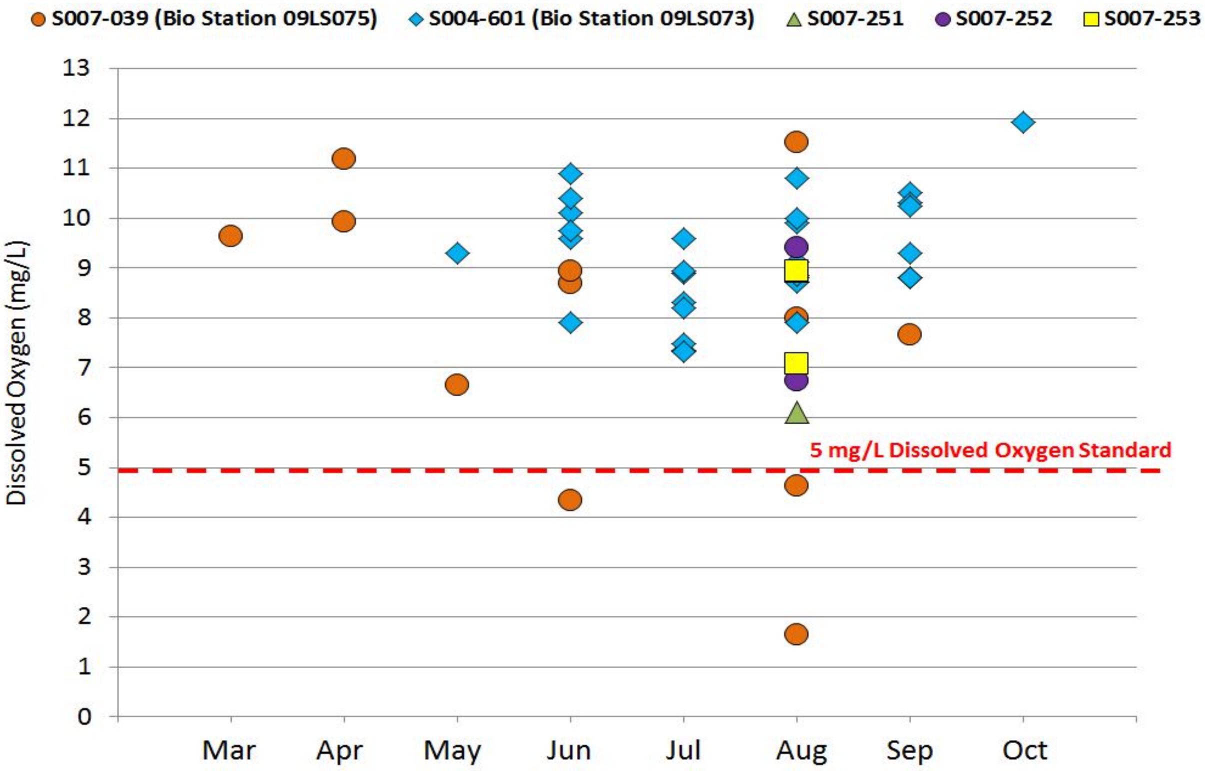


**Figure 310:** Point measurements of DO collected from the West Two River

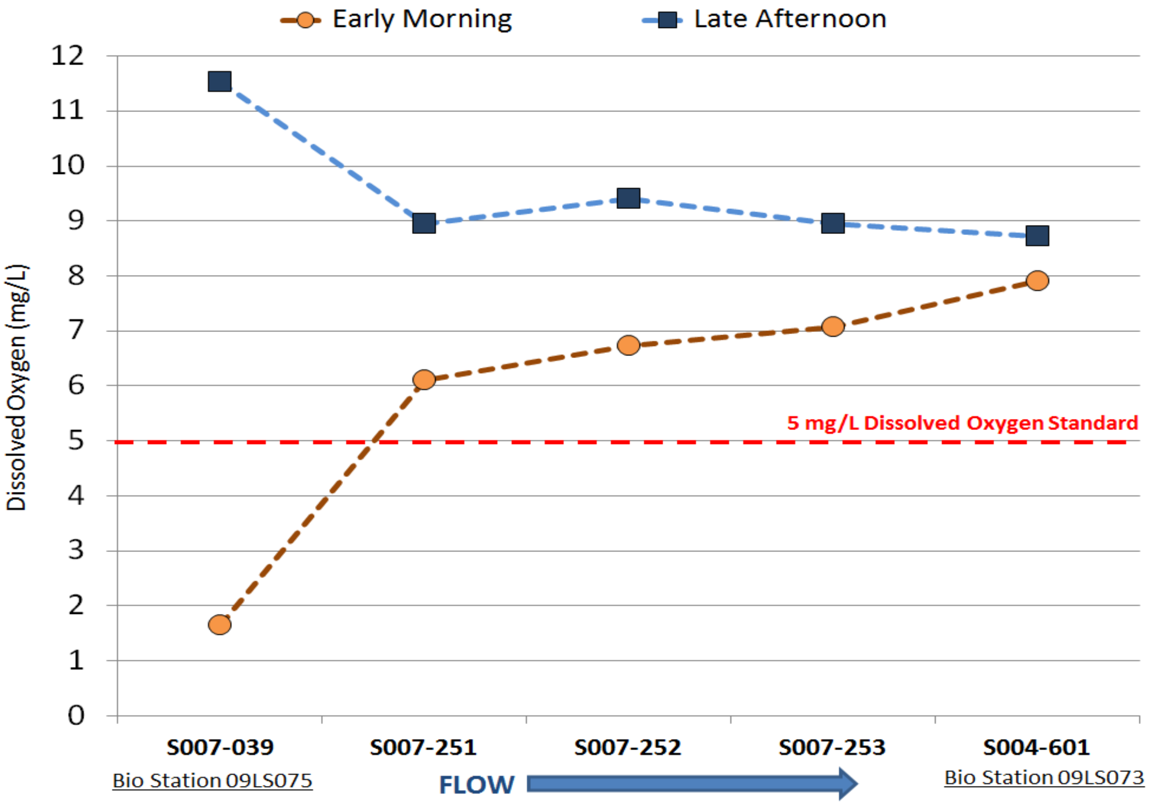


**Figure 311:** Results of longitudinal DO profile collected from the West Two River on August 14, 2012

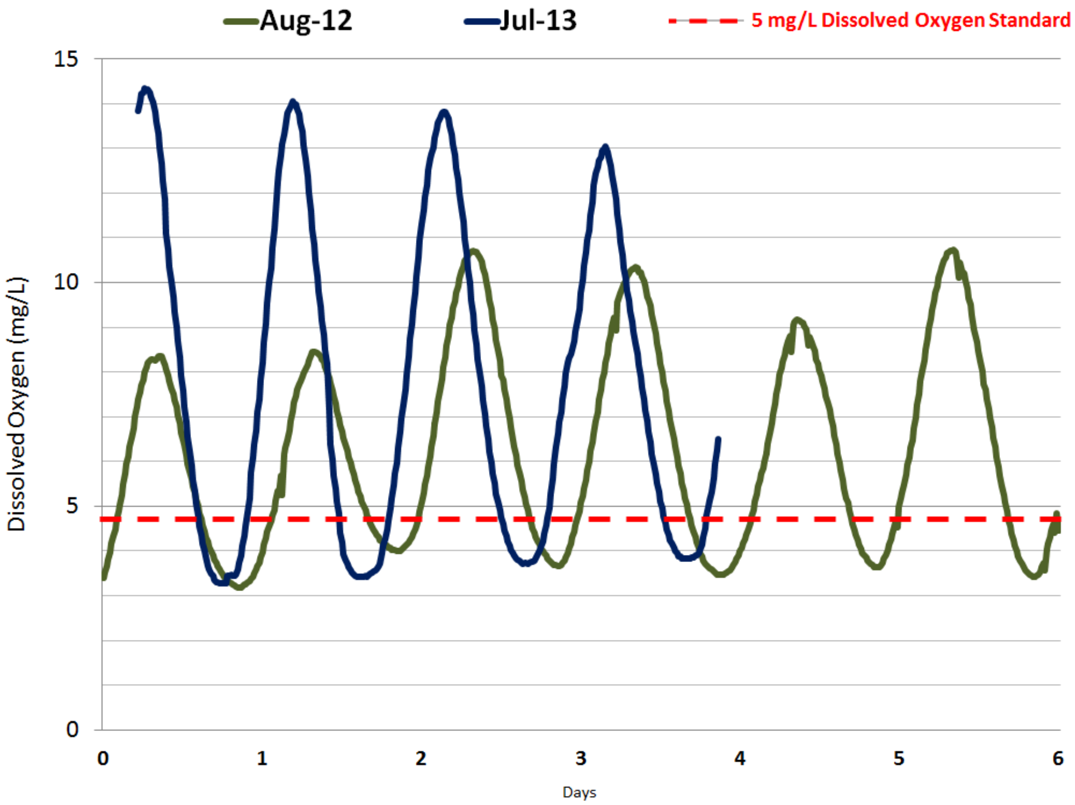


**Figure 312:** Results of continuous DO monitoring at station 09LS075 in August 2012 and July 2013

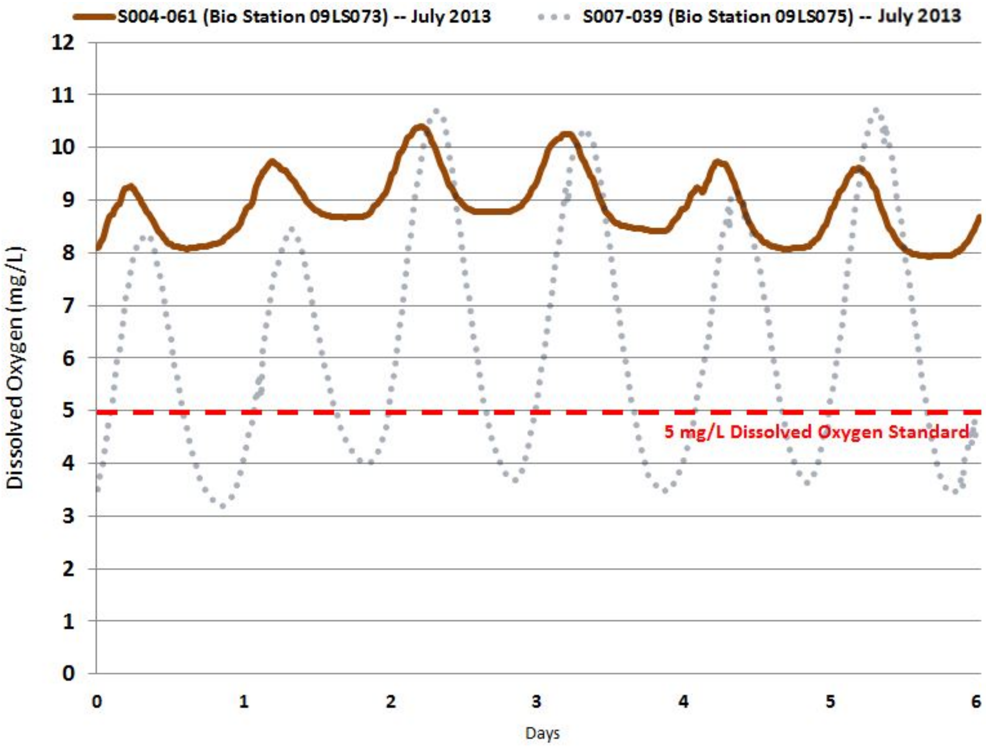


**Figure 313:** Comparison of continuous DO data collected at station 09LS075 (impaired) and 09LS073 (not impaired) in July of 2013

## Sources and Pathways of Low DO / High DO Flux

The negative effects of low DO and DO flux appear to be isolated to the impaired biological monitoring station located immediately downstream from West Two Reservoir (09LS075). The primary source of DO stress within this reach is nutrient enrichment and productivity in West Two Reservoir. This reservoir is currently on the impaired waters list for excess nutrients, and major algae blooms are commonly observed. Figure 314 provides an example of an algae bloom on this reservoir, and shows the close proximity of biological monitoring station 09LS075 to the outlet. The DO regime of the West Two River below the reservoir is heavily influenced by the biological and chemical processes taking place in this impoundment.

Nutrient enrichment and productivity within the lotic (flowing) sections of the West Two River does not appear to be excessive. TP concentrations in the West Two River are only slightly elevated compared to high quality reference streams in the SLRW. Mean TP concentrations at the two biological monitoring stations are both around 0.030 mg/L (n=14), which is below the 0.055 mg/L TP criteria cited for streams in the northern nutrient region of the MPCA's river nutrient water quality standards. Algae blooms are rare in the impaired reach, although moderate amounts of suspended algae have been noted in the river just below the impoundment.

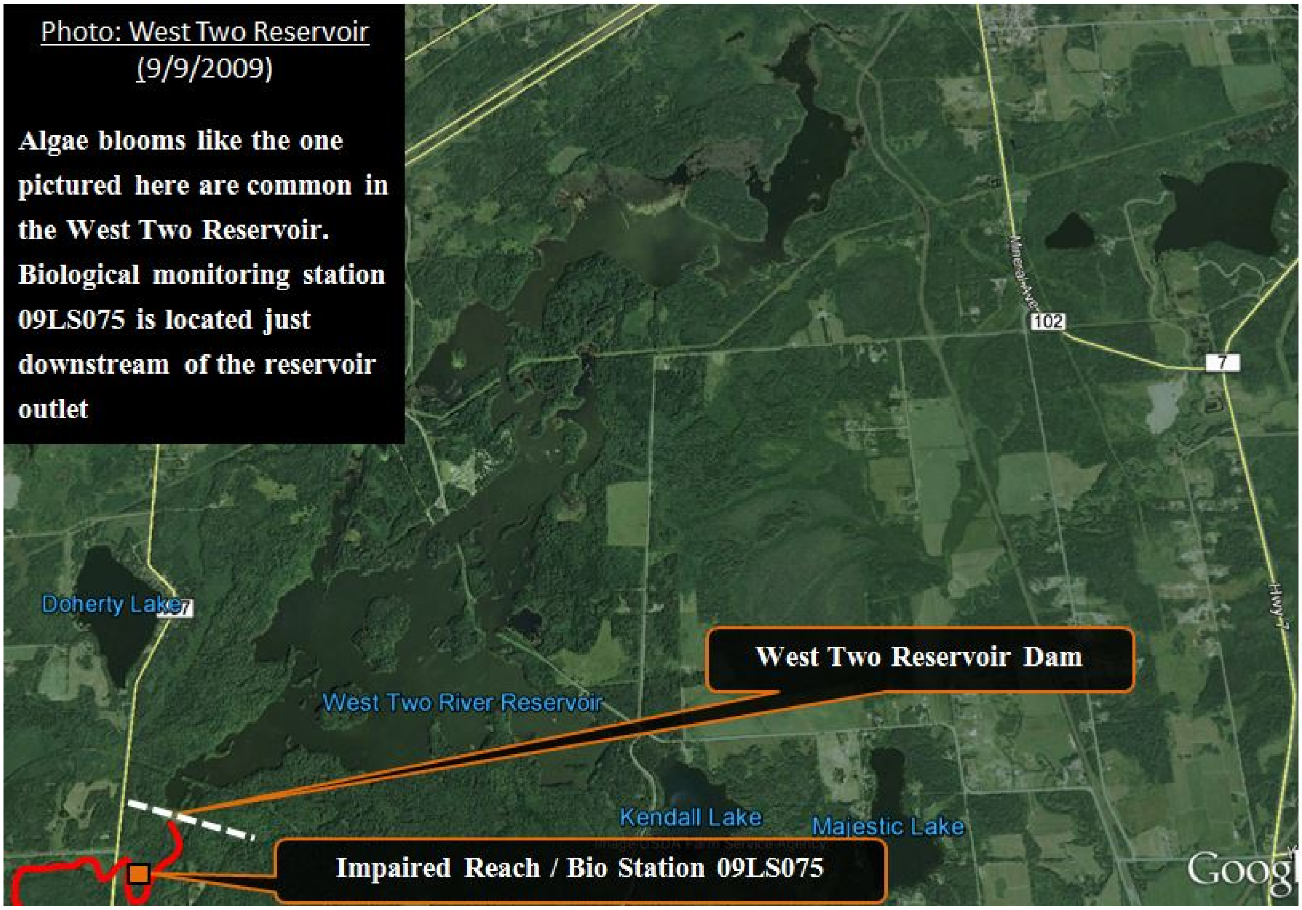


**Figure 314:** Aerial photo showing algae bloom in the West Two Reservoir, and location of biological monitoring station 09LS075 in relation to the reservoir outlet

## Biological Response to Low Dissolved Oxygen / Dissolved Oxygen Flux

### Macroinvertebrate Community DO Index

Macroinvertebrate DO TIV from the three West Two River biological monitoring stations are compared to results from high-quality SLRW stations in Figure 315. Higher DO TIV values indicate a macroinvertebrate community with a larger proportion of taxa that are sensitive to low DO conditions. The DO TIV results from station 12LS002 were higher than 75% of the stations scoring above the MIBI threshold and UCL, which suggest this station supports many organisms that are sensitive to low DO conditions. Results from station 09LS073 were comparable to the median DO TIV results for high quality stations in the SLRW. Therefore, low DO and DO flux is not a likely stressor at these two monitoring locations.

On the contrary, DO TIV results from station 09LS075 were well below the 25 percentile of results from the high quality reference streams. Approximately 1/3 of the individuals sampled from station 09LS075 were members of taxa that are considered tolerant of low DO conditions. In comparison, only 2% and 5% of the individuals sampled were members of low DO tolerant taxa at stations 09LS073 and 12LS002, respectively. There is a clear biological response to low and/or fluctuating DO conditions observed at 09LS075 that is absent from the other two monitoring stations. These biological symptoms are well correlated with water quality data from these monitoring sites, as station 09LS075 was the only station to exhibit DO conditions that can be considered harmful to aquatic life.

### Macroinvertebrate Taxa Richness

Macroinvertebrate taxa richness can decline when DO conditions are below suitable levels and/or fluctuate significantly (> 4 mg/L) over the course of day. Depending on the station, macroinvertebrate taxa richness at West Two River biological monitoring sites is slightly or severely below results from high quality stations of the same MIBI class in the SLRW. Station 09LS073 supported 49 total taxa, which is comparable to many of the high quality sites used in the comparison in Figure 316. Stations 09LS075 and 12LS002 supported 42 and 33 taxa, respectively. These counts are below the 25th percentile taxa richness values for comparable reference sites. Station 12LS002 supported the fewest taxa of the three monitoring stations; however, it had a lower percentage of very tolerant taxa than station 09LS075 where DO conditions are unfavorable. Over 60% of the taxa observed at 09LS075 can be considered “very tolerant” of pollution and/or disturbance, compared to only 27% at station 12LS002 and 20% at station 09LS073.

### Insect Taxa %

Elevated DO flux can lead to an increase in less desirable non-insect aquatic species such as aquatic worms (Oligochaeta), fly larvae (e.g., certain Chironomid (midges) taxa, and snails). Nearly 85% of the taxa observed at stations 09LS073 and 12LS002 were insects, which is slightly lower, but comparable to stations of the same MIBI class that scored above the impairment threshold (i.e. not impaired) (Figure 316). The percentage of insect taxa at station 09LS075 was slightly lower at 76%. Although this is a somewhat small difference based on percentages, very few of the class 3 and 4 MIBI stations in the SLRW with good to excellent scores supported invertebrate communities with less than 80% insect taxa. In other words, station 09LS075 supports a lower relative percentage of insect taxa than the vast majority of high quality streams with comparable drainage area and habitat features (Figure 316).

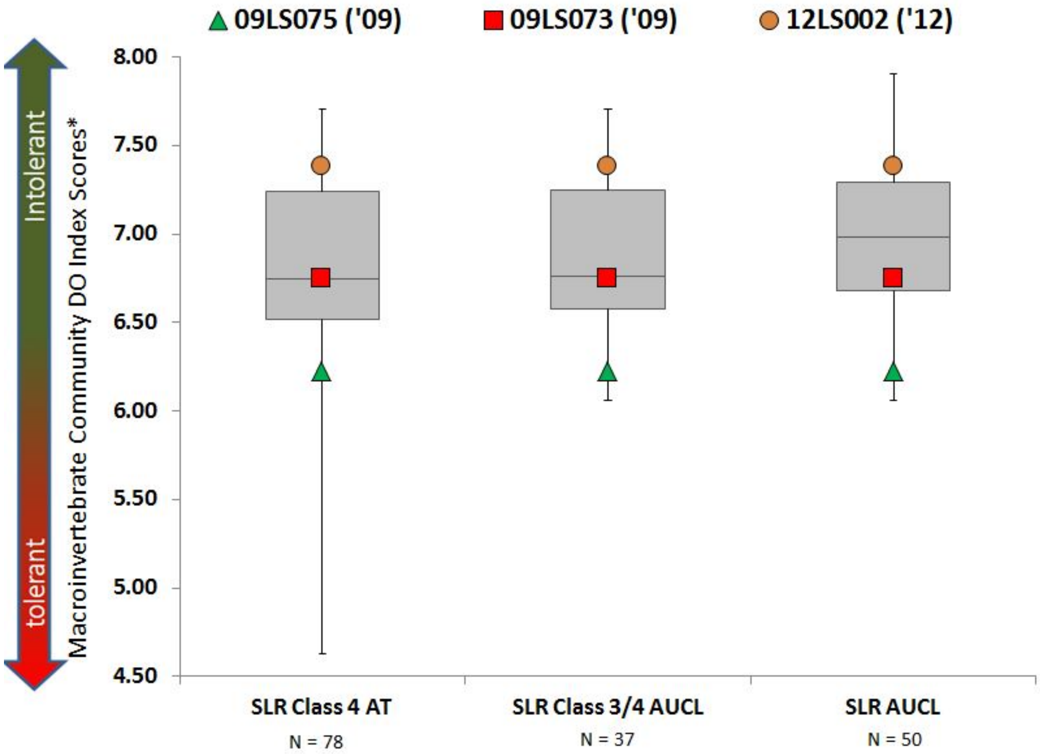


**Figure 315:** Macroinvertebrate community tolerance indicator values (TIV) for DO at West Two River monitoring sites compared to non-impaired streams. See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

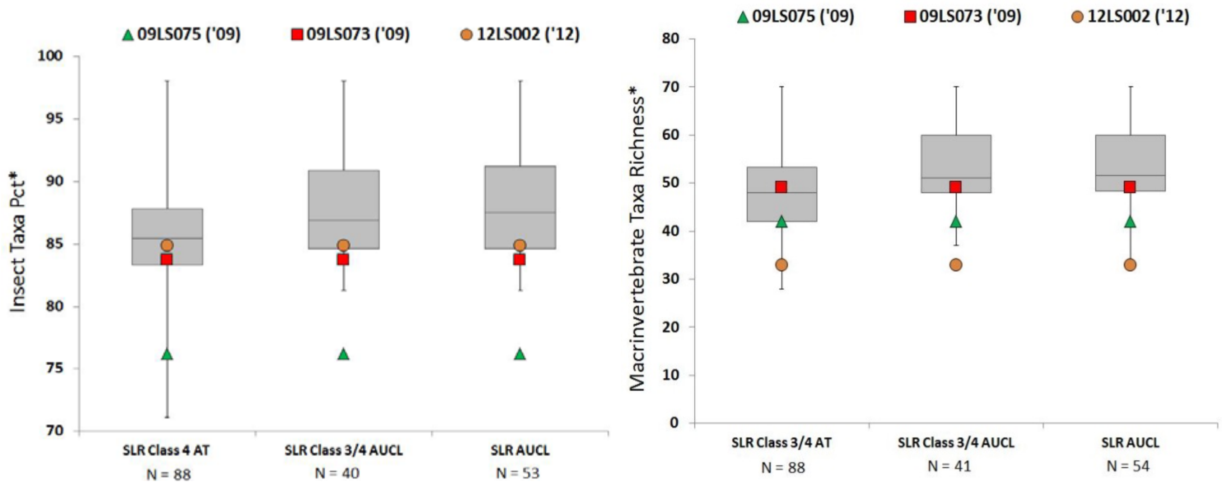


**Figure 316:** Comparison of results for several macroinvertebrate metrics between West Two River stations and non-impaired SLRW reference stations: % Insect Taxa (left) and overall macroinvertebrate taxa richness (right). See section 4 for explanation of TIVs **SLR=** St. Louis River Watershed **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above Fish IBI Threshold

**Summary: Is low DO/DO flux a stressor in the West Two River?**

Water chemistry and biological data provide adequate evidence to diagnose DO as a cause of biological impairment (macroinvertebrate IBI) in West Two River. The presence of low DO and high DO flux, as well as the related effects on aquatic life, are limited to the reach of West Two River immediately downstream of the reservoir. Further downstream, stations 12LS002 and 09LS073 support healthy macroinvertebrate assemblages and adequate DO concentrations. The low DO concentrations and high DO flux observed at station 09LS075 are caused by the eutrophic conditions in West Two Reservoir. Nutrient reduction strategies aimed at improving reservoir conditions will benefit macroinvertebrate populations in the impaired reach of West Two River, although other limiting factors observed at station 09LS075 (esp. habitat) may prevent full support of aquatic life criteria.

## 5.20.2 Specific conductivity and Sulfate Toxicity

Specific conductivity was identified as a candidate cause of MIBI impairment in the West Two River based on existing data and watershed land-uses. Instantaneous (point) measurements of specific conductivity, a surrogate measure of specific conductivity, are plotted by monitoring station and calendar month in Figure 317. Specific conductivity at the two biological monitoring stations (09LS075 and 09LS073) is generally in the range of 400 – 600 µS/cm during the open water season from April through October. Several measurements exceeding 800 µS/cm have been observed at the station immediately downstream of West Two Reservoir (09LS075) during winter and early spring conditions.

Monitoring equipment was deployed in stream to collect short-term continuous specific conductivity data (as well as pH, temperature, DO) at the two biological monitoring stations. Data were logged for an average duration of about five days, and measurements were collected at 15-minute intervals. The results obtained through continuous monitoring were very similar to the data collected via point measurements at these stations. Over the continuous monitoring periods, specific conductivity at these two stations ranged between 500 – 700 µS/cm, with the higher readings observed at the upstream-most station located at the West Two Reservoir outlet.

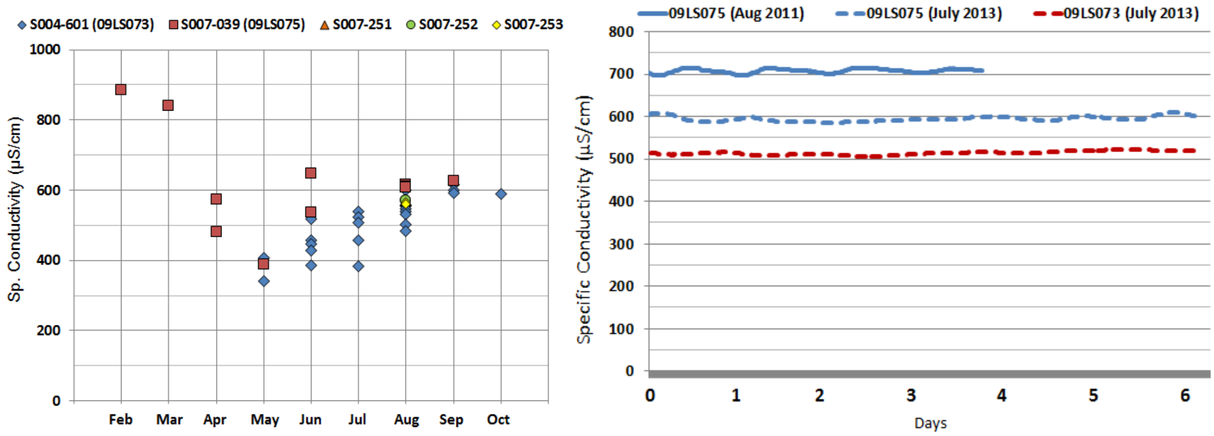


**Figure 317:** Point measurements of specific conductivity collected at West Two River monitoring stations (left), and continuous specific conductivity results from3-5 periods at biological monitoring stations

### Sources and Pathways of Specific conductivity

The West Two River and several of its tributary streams exhibit elevated specific conductivity and water hardness as a result of two primary sources; (1) dewatering of mine pits from US Steel's Minntac Mining Area, and (2) Effluent from the city of Mountain Iron's WWTP. Other common sources of elevated specific conductivity, such as urban development and road de-icing agents may contribute to seasonal increases in specific conductivity, but monitoring results do not suggest that these are severe enough in magnitude to have any impact on aquatic life.

**Table 89:** List of permitted dischargers in the West Two River Watershed

| Facility / Permit # | Discharge Point | Permit Details |
|---|---|---|
| Ulland Brothers – Aggregate MNG490069 | Stormwater permit | Stormwater permit for sand and gravel mining operation |
| US Steel Corporation – Minntac Mining Area / MN0052493-SD-7 | PIPE OUTFALL 070, SUMP #11 | West Minntac Pit dewatered through pipe outfall to Kinross Creek and unnamed wetlands |
| US Steel Corporation – Minntac Mining Area / MN0052493-SD-9 | PIPE OUTFALL 090, WHEELING | Water pumped from Prindle and Wheeling mine pits at combined average and maximum rates of 10 cfs and 24 cfs |
| US Steel Corporation – Minntac Mining Area / MN0052493-SD-4 | PIPE OUTFALL 040, PRINDLE | |
| Mountain Iron WWTP / MN0040835-SD-3 | SURFACE WATER DISCHARGE | Continuous discharge to unnamed tributary to the West Two Reservoir |

## West Two River Geochemistry

Samples were collected at West Two River station S007-039 (biological station 09LS075) during baseflow, rain event flow, and snowmelt flow and analyzed for major cations and anions. The most common salt ions found in surface water that influence specific conductivity levels include positively charged (cations) $Ca^{2+}$, $Mg^{2+}$, $Na^{+}$, $K^{+}$, and negatively charged (anions) $HCO_3^{-}$, $CO_3^{2-}$, $SO_4^{2-}$, and $Cl^{-}$ (CADDIS, 2012). Figure 318 compares concentrations of these major cations and anions observed in the impaired reach of the West Two River and the nearby West Swan River (control), which is not impaired.

Concentrations of all the major cations and anions analyzed were higher in the West Two River compared to the control. Sulfate, in particular, was found to be highly elevated in all three sampling events. Sulfate concentrations in this stream were highest during the snowmelt sampling event (185 mg/L), which differed from other streams studied in mining watersheds (e.g. Spring Mine Creek, Kinney Creek) where sulfate concentrations reached their maximums during baseflow conditions. Snowmelt events in the lower West Two River Watershed are likely regulated to some level by the reservoir above the dam, and it's possible that the stream was still closer to baseflow conditions when it was sampled in March of 2012 as other free-flowing rivers nearby were in snowmelt stage.

Sulfate, magnesium, calcium, and sodium appear to be the major drivers of specific conductivity in the impaired reach of the West Two River. This observation is consistent with results from the other watersheds evaluated in this report that are influenced by mining land-uses and WWTP effluent. The results from the West Swan River, which is not as heavily impacted by mining, provides some context for evaluating departure from reference or natural background conditions. Sulfate concentrations in the West Two River were less than 5 mg/L during all three sampling events. Sulfate levels in West Two River will be evaluated independently as a stressor later in this section.

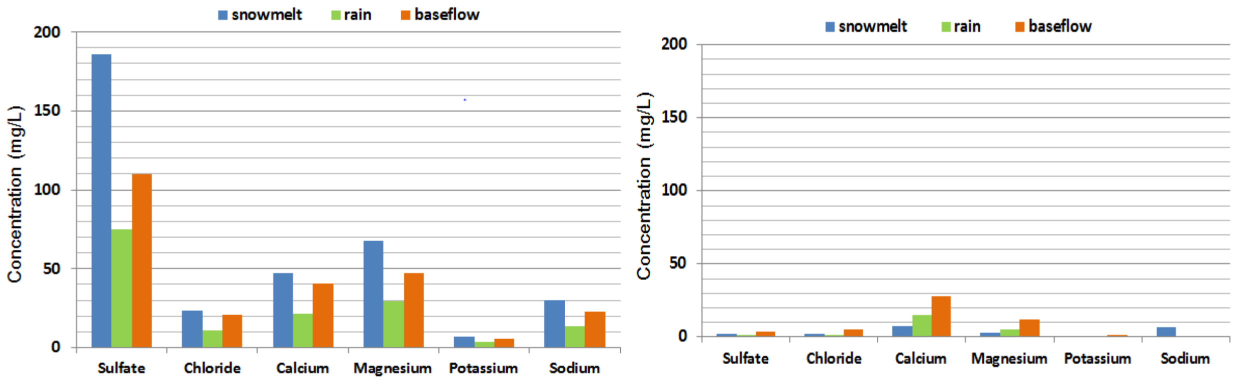


**Figure 318:** Concentrations of major cations and anions observed in West Two River (left) compared to the nearby West Swan River (right)

## Biological Response to Specific conductivity

The effects of elevated conductivity on aquatic life were evaluated using data from Minnesota streams and scientific literature. A summary of this analysis is presented in Section 3.1.6. Based on this work, several biological metrics were selected to evaluate specific conductivity as a cause of MIBI impairment in West Two River.

**Table 90:** Summary of biological metrics and literature used to evaluate elevated specific conductivity as a stressor

| Metric | Response to Increased Specific conductivity / Conductivity | Source |
|---|---|---|
| **EPT Richness** | Decrease | Roy et al (2003); Echols et al (2009); Johnson et al (2013) |
| **Overall Taxa Richness** | Decrease | Johnson et al (2013) |
| **Ephemeroptera Richness** | Decrease | Pond (2004); Hassel et al (2006) |
| **Fish and Macroinvertebrate Tolerance Indicator Values (TIV)** | Increase | MBDI (Yoder and Rankin, 2012) |

### EPT Richness

A total of six EPT taxa were sampled at the impaired biological monitoring site (09LS075). A higher number of EPT taxa (9 and 10) were observed at downstream monitoring stations 12LS002 and 09LS073, which both passed the MIBI criteria and are not considered impaired sites. Overall, results from all three stations were equal to or below the 25th percentile values for EPT taxa richness observed at high quality stations of the same MIBI class in the SLRW (Figure 319). EPT taxa are somewhat limited in the lower West Two River compared to healthy streams in the SLRW, especially at station 09LS075.

### Overall Macroinvertebrate Taxa Richness

Macroinvertebrate taxa richness at Kinney Creek monitoring stations ranged from 33 to 49 taxa. The lowest taxa richness count was observed at station 12LS002, which is not considered impaired. Taxa richness values from stations 09LS075 (located on the impaired reach) and 12LS002 were below the 25th percentile of taxa richness values observed at high quality stations of the same MIBI class. Taxa richness results from station 09LS073 were comparable to many of the high quality sites in the SLRW. Overall, macroinvertebrate taxa richness appears to be moderately to severely limited in the reach just below West Two Reservoir, a symptom not seen in the lower reaches of the river near station 09LS073. Specific conductivity levels are somewhat higher in the reach where taxa richness is more limited, but there is not much of an appreciable difference. Other stressors, particularly low DO and DO flux, show a stronger spatial co-location with the decrease in taxa richness than specific conductivity.

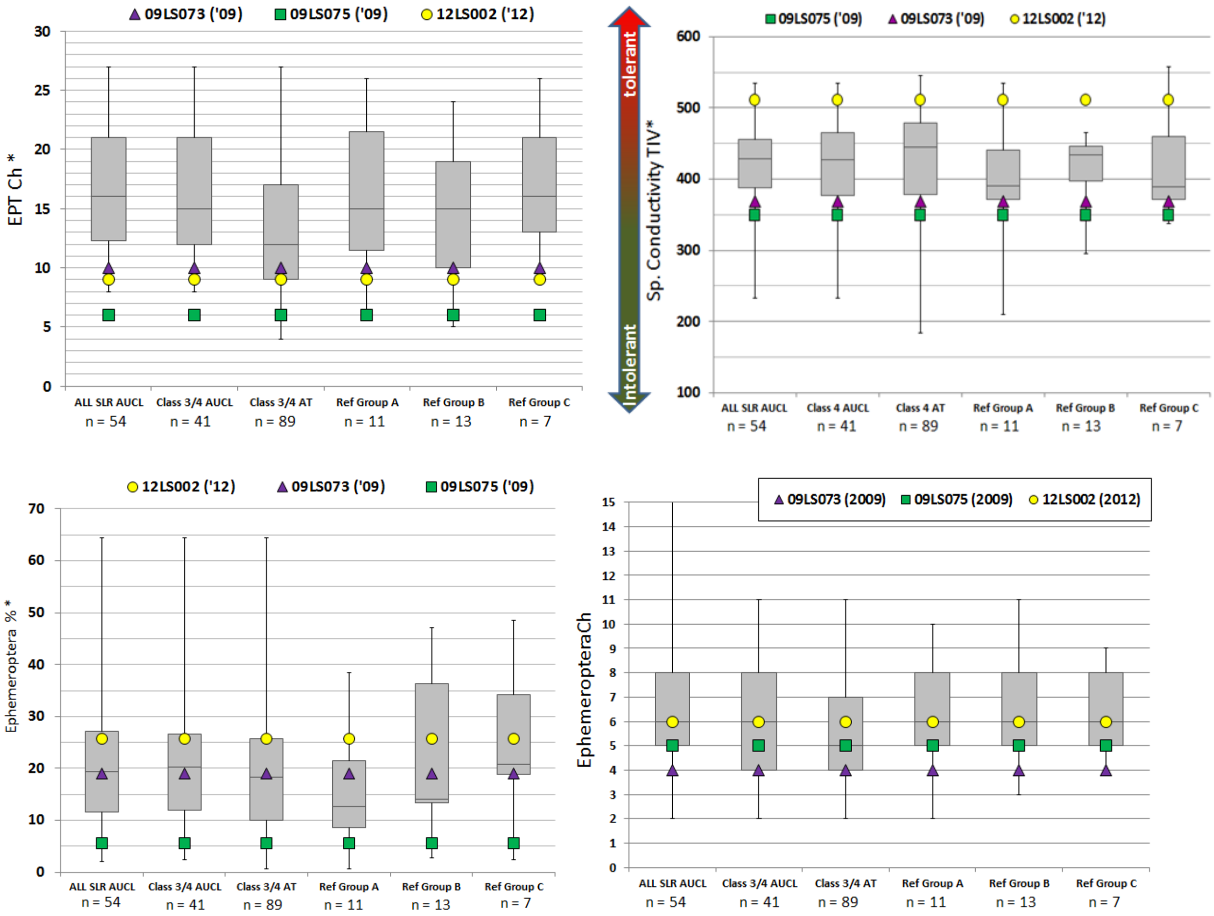


**Figure 319:** Comparison of results for several macroinvertebrate metrics between West Two River stations and non-impaired SLRW reference stations: EPT Taxa Richness (top left); macroinvertebrate community tolerance indicator values (TIV) for specific conductivity (top right); % Ephemeroptera in sample (lower left); Ephemeroptera taxa richness (lower right). ***** See Section 4 for explanation of TIVs. **AUCL =** Above Upper Confidence Limit of FIBI threshold **AT =** Above FIBI Threshold

### Ephemeroptera Taxa Richness and Relative Abundance

Ephemeroptera (mayfly) taxa richness ranged from 4-6 taxa among West Two River macroinvertebrate monitoring stations. The median value for comparable sites in the SLRW scoring above the impairment threshold was six mayfly taxa, so several West Two River monitoring sites have slightly lower mayfly richness than most sites that scored above the MIBI threshold (Figure 319). In other impaired SLRW streams where elevated specific conductivity is a candidate cause for impairment, Ephemeroptera richness was in the range of 0 to 1 taxon (e.g. Manganika Creek and Spring Mine Creek). The presence of 4-6 mayfly taxa in the West Two River may be an indication that the effects of specific conductivity are less severe in this watershed. However, mayfly richness can be impacted by a variety of stressors (e.g. low DO, habitat, turbidity) and this metric cannot be used as a stand-alone diagnostic biological metric of any one stressor.

Relative abundance of mayfly individuals varied considerably among the three West Two River monitoring stations. Abundance at stations 12LS002 and 09LS073 were comparable or higher than the median value observed at non-impaired stations of the same MIBI class (Figure 319). Station 09LS075, located just downstream of the West Two Reservoir, had much lower mayfly abundance values than the vast majority of comparable streams in the SLRW. Similar to other biological response metrics evaluated for these sites, the lack of mayfly individuals at this station may be more closely linked to the poor DO regime associated with this site.

**Specific Conductivity Tolerance Indicator Values**

Specific conductivity TIVs for West Two River macroinvertebrate data are compared to results from high quality SLRW stations in Figure 319. For additional information on the development of TIV, refer to Section 4. TIV results for West Two River stations show a high level of variability between monitoring stations. Results from station 12LS002 indicate a macroinvertebrate community with a rather high tolerance level to specific conductivity, while stations 09LS075 and 09LS73 show TIV values that represent macroinvertebrate community that is more sensitive to high specific conductivity. The spatial pattern of these TIV results is not correlated with water chemistry data for specific conductivity, as there is no appreciable difference in specific conductivity between station 12LS002 and the other monitoring stations. Therefore, it is highly probable that the TIV results are being influenced by other confounding variables and not specific conductivity.

**Summary: Is elevated specific conductivity a stressor in the West Two River?**

Specific conductivity levels in West Two River are elevated compared to natural background conditions observed in less impacted streams of the SLRW (see Table 2). Measurements of up to 900 µS/cm have been observed during baseflow periods, but more typical open water season readings are in the range of 400 – 600 µS/cm. Inputs of sulfate and other dissolved solids from mine pit dewatering and WWTP effluent are the primary sources. Several of the biological metrics evaluated above show symptoms of stress that are commonly observed in streams with high specific conductivity (e.g. low EPT taxa, low taxa richness). However, there are inconsistencies in the biological response data. Specifically, several response metrics (taxa richness, community level TIVs) are more closely related to low DO concentrations and DO flux than specific conductivity.

Biological response metrics show variable and inconsistent responses to specific conductivity. As a result, specific conductivity cannot be diagnosed as a cause of impairment in the West Two River. The monitoring results obtained from this study do not provide enough evidence to eliminate specific conductivity as a stressor with high confidence, but other stressors (esp. DO) appear to be more closely linked to the macroinvertebrate impairment.

### 5.20.3 Sulfate Toxicity

Elevated sulfate concentrations were identified as a candidate cause for MIBI in West Two River based on monitoring results from 2009 through 2014. A total of seventeen samples were collected for sulfate from the reach of the West Two River located downstream of the West Two Reservoir. These samples were collected over a range of flow conditions, including winter baseflow samples through ice during the month of February. An addition two samples were collected from Parkville Creek, which is a tributary stream to the West Two River and a significant source of sulfate in its watershed.

Below West Two reservoir, sulfate concentrations ranged from a low of 51 mg/L in May of 2009, to a high of 212 mg/L in September of 2009 (n=17, average = 108 mg/L) (Table 91). Another elevated concentration of 201 mg/L was observed during the lone winter sample that was collected through the ice in February of 2014. The sampling results clearly show sulfate concentrations increasing steadily as flows decrease towards annual minimums in late summer, fall, and winter months. Monitoring data indicate that chronic exposures to sulfate in the West Two River below the reservoir during normal flows are in the range of 80 – 220 mg/L.

Corresponding chloride and water hardness results are also displayed in Table 91. Chloride concentrations were consistently in the range of 10-30 mg/L and hardness values range from 149 mg/L up to 430 mg/L. Hardness and chloride concentrations were used to calculate the applicable sulfate toxicity standard for aquatic life that is applied in several use states, such as Illinois, Indiana, Iowa, and Pennsylvania (Table 9). Monitoring results show that sulfate levels in West Two River would not be violating water quality criteria in these states due to elevated chloride and hardness values (Table 91). There is currently no sulfate standard in Minnesota designed to protect fish and aquatic macroinvertebrates.

### Sources and Pathways of Sulfate with the West Two River Watershed

Sulfate is often the dominant contaminant from mine water and can form a wide range of salts (Mining and Environmental Management Magazine 2000). Significant concentrations of sulfate can accumulate in surface water that is not frequently flushed, such as water found in mine pit lakes or behind stream impoundments. This scenario is occurring in the headwaters of West Two River, as several mine pits in the upper limits of the watershed are pumped into tributary streams flowing into the West Two Reservoir. Parkville Creek, located between the communities of Kinney and Mountain Iron, is one tributary in particular that delivers high concentrations of sulfate to this drainage. Samples collected from this stream in September of 2013 had sulfate concentrations of 455 mg/L and 457 mg/L.

Another potential source of sulfate in this watershed is the city of Mountain Iron’s WWTP. No sulfate monitoring is required under the current discharge permit, so the contribution from this source cannot be fully accounted for in this report.

### Water Quality Standards for Sulfate

Minnesota does not currently enforce a sulfate standard for protection of fish and aquatic macroinvertebrates. Several U.S. states and Canadian provinces have developed sulfate standards that will be used to evaluate sulfate as a stressor to aquatic life in the SLRW. For more information on these standards, see Section 3.1.6.

**Table 91:** West Two River sulfate data paired with hardness and chloride results, and applicable water quality standard based on formula used by several US states.

| Station | AUID | Sample Date | Sulfate (mg/L) | Chloride (mg/L) | Hardness (mg/L) | Calculated Sulfate Standard (states of IL, IA, IN, PA) (mg/l) ** |
|---|---|---|---|---|---|---|
| S004-601[1] | 04010201-534 | 5/6/2009 | 51.9 | 13.3 | 149 | 992 |
| S004-601[1] | 04010201-534 | 5/21/2009 | 71 | 10.6 | 168 | 968 |
| S004-601[1] | 04010201-534 | 6/4/2009 | 84 | 13.5 | 201 | 1,194 |
| S004-601[1] | 04010201-534 | 6/22/2009 | 65.5 | 12.2 | 191 | 1,111 |
| S004-601[1] | 04010201-534 | 7/8/2009 | 82 | 11.5 | 233 | 1,243 |
| S004-601[1] | 04010201-534 | 7/28/2009 | 84 | 11.6 | 237 | 1,264 |
| S004-601[1] | 04010201-534 | 8/10/2009 | 80 | 11.0 | 236 | 1,238 |
| S004-601[1] | 04010201-534 | 8/24/2009 | 93 | 12.7 | 259 | 1,385 |
| S004-601[1] | 04010201-534 | 9/9/2009 | 212 | 12.6 | 265 | 1,404 |
| S004-601[1] | 04010201-534 | 9/23/2009 | 116 | 12.5 | 283 | 1,468 |
| S007-039[2] | 04010201-535 | 3/29/2012 | 186 | 23.5 | 398 | 2,287 |
| S007-039[2] | 04010201-535 | 4/16/2012 | 91.9 | 14.8 | no data | - |
| S007-039[2] | 04010201-535 | 5/25/2012 | 74.9 | 10.9 | 174 | 1,002 |
| S007-039[2] | 04010201-535 | 6/25/2012 | 130 | 20.7 | no data | - |
| S007-039[2] | 04010201-535 | 8/1/2012 | 110 | 19.9 | no data | - |
| S007-039[2] | 04010201-535 | 9/12/2012 | 110 | 20.6 | 295 | 1,800 |
| S007-039[2] | 04010201-535 | 02/05/2014 | 201 | 27.9 | 430 | 2,343 |
| S007-691[3] | 04010201-537 | 9/5/2013 | 457 | no data | no data | - |
| S007-691[3] | 04010201-537 | 9/5/2013 | 455 | no data | no data | - |
| **[1] = Non-Impaired reach of West Two River** | | | | | | |
| **[2] = Impaired Reach of West Two River** | | | | | | |
| **[3] = Parkville Creek (tributary to West Two River Reservoir)** | | | | | | |

## West Two River Sulfate Discussion

Sulfate concentrations recorded in the impaired reach of the West Two River generally did not exceed water quality criteria that are currently being implemented or drafted in other US states and Canadian provinces. Below is a brief comparison of West Two River sulfate data to some of the sulfate standards available.

### British Columbia, Canada

In a paper by Elphick et al (2010), various sulfate standards are proposed for British Columbia waters based on SSD and a SFA. Both of these standards are dependent on water hardness, as harder water can ameliorate the impact of sulfate on aquatic life. The sulfate standard proposed in British Columbia for very hard water (>160 mg/L) is 725 mg/L based on the SSD approach, and 675 mg/L based on the SFA approach. The maximum sulfate concentration observed in the impaired reach of the West Two River was 212 mg/L, well below the proposed standard for British Columbia.

### California

The state of California evaluated sulfate as a stressor to aquatic life in a 2013 study (Buchwalter 2013). This report did not result in the development of sulfate standard for the state, but served as more of a review of existing data and summary of other work involving sulfate. Although many uncertainties involving sulfate toxicity were discussed in this report, the author concluded that there is enough toxicity data by EPA standards to support an acute toxicity criterion of 234 mg/L SO4 and a chronic criterion of 124 mg/L SO4. These values were not adjusted based on chloride and hardness values like other WQ standards for sulfate, and the author mentions uncertainties in the values stated above based on this detail.

Sulfate levels in West Two River exceeded the 124 mg/L chronic criterion on occasion, and nearly exceeded the 234 mg/L acute threshold during a few sampling events. Given all of the recent research (Soucek and Kennedy 2004; Rankin 2003 and 2004) on the importance of chloride and hardness values in determining the potential toxicity of sulfate, the values proposed by Buchwalter should be applied with caution.

### Ohio

Several short reports exploring sulfate effects on aquatic life in the state of Ohio were released by the Center for Applied Bioassessment and Biocriteria (Rankin 2003, 2004). These studies linked biological monitoring data with sulfate sampling results across the state of Ohio with the goal of identifying critical thresholds for protecting sensitive forms aquatic life. Although no water quality standards were developed through this work, several conclusions can be drawn from these reports:

1. **Many of the most sensitive taxa were not present in streams where sulfate concentrations exceeded 200 mg/L.**

Richness of mayfly, stonefly, and caddisfly taxa (EPT taxa) at West Two River monitoring stations was noticeably lower in comparison to high quality stations in the SLRW. These taxa respond negatively to a variety of chemical and physical stressors, and thus cannot be considered diagnostic of any one stressor. Intolerant or sensitive macroinvertebrate taxa were present in very low numbers in the West Two River. Percent intolerant individuals at stations 09LS075, 12LS002, and 09LS073 were 0.3%, 1.9%, and 2.2%, respectively.

2. **There is good evidence from Ohio streams that the presence of higher chloride concentrations ameliorates the effects of sulfate**

Chloride and water hardness values in West Two River are elevated above natural background conditions, and are likely limiting the toxicity of sulfate to fish and aquatic macroinvertebrates. The various US states that are currently applying a sulfate standard for protection of fish and macroinvertebrates (see next page) are incorporating hardness and chloride values into the WQ standard.

3. **Streams with sulfate concentrations above 400 mg/L generally exhibited poor biological integrity scores**

Sulfate concentrations did not exceed 400 mg/L in the impaired reach of the West Two River. The maximum concentration observed in this reach was 212 mg/L. Sulfate concentrations exceeding 400 mg/L in Parkville Creek, a headwaters tributary of West Two River that is impacted by mining land-uses.

Fish and MIBI scores for Parkville Creek were deemed “poor” based on biological assessments completed in 2013 (Peterson et al 2014). Habitat quality in Parkville Creek was found to be sufficient for supporting quality fish and macroinvertebrate assemblages. With the elimination of habitat as a potential stressor in Parkville Creek, it is more likely that a water quality variable (e.g. sulfate) is responsible for the low fish and MIBI scores.

4. **EPT macroinvertebrate taxa were limited to 10 or less at sites where sulfate concentrations exceeded 500 mg/L**

Sulfate concentrations did not exceed 500 mg/L in the impaired reach of the West Two River or any of its tributary streams for which data are available. Two of the three monitoring stations on Parkville Creek had fewer than 10 EPT taxa (6 and 7 EPT taxa were observed). The station with more than 10 EPT taxa (n=12) was located furthest away from the permitted discharge.

### Illinois, Indiana, Pennsylvania, and Iowa

The states of Illinois, Indiana, Pennsylvania, and Iowa have been working towards an aquatic life standard for sulfate and other dissolved solids. Studies by Soucek and Kennedy (2004), Pennsylvania Department of Environmental Protection (PDEP, no date), and Iowa DNR (IDNR 2009) were compiled to develop the sulfate standard. The specifics of this sulfate standard are provided in Table 9. Unlike some of the sulfate criteria listed above, chloride and water hardness were taken into account in the development of a sulfate standard for these states. Table 91 summarizes paired sulfate, hardness, and chloride data and the resulting sulfate WQ standard as it would be applied in these states.

Water quality sampling results indicate that the impaired reach of the West Two River would not be in violation of the sulfate standard as it is applied in the states of Indiana, Illinois, Iowa, and Pennsylvania.

### Biological Response to Sulfate

Sulfate toxicity is a complex issue and a number of factors may interact to determine the responses of various organisms to sulfate-dominated waters. A discussion of available biological response data to elevated sulfate levels is presented in Section 3.1.6 of this report. Based on that summary, the biological metrics listed in Table 92 will be used to evaluate sulfate as a stressor in West Two River. Additional consideration for sulfate as a stressor will be presented in the specific conductivity discussion for this stream.

**Table 92:** Biological metrics selected to evaluate sulfate toxicity as a stressor to aquatic life

| Metric | Description | Relevance |
|---|---|---|
| **EPTCh** | Taxa richness of Ephemeroptera, Plecoptera & Trichoptera (baetid taxa treated as one taxon) | EPT macroinvertebrate taxa were limited to 10 or less at sites where sulfate concentrations exceeded 500 mg/L (Rankin, 2003) |
| **EphemeropteraPct** | Relative abundance (%) of Ephemeroptera individuals in subsample | Sulfate and/or bicarbonate are the likely drivers of reduced macroinvertebrate diversity and abundance (particularly mayflies) in mining impacted streams in West Virginia (Buchwalter, 2013) |

### EPTCh

Rankin (2003) observed that EPT macroinvertebrate taxa were limited to 10 or less at sites where sulfate concentrations exceeded 500 mg/L. See Figure 319 for a summary of West Two River data related to this biological response metric. The EPT taxa are somewhat limited in the lower West Two River compared to healthy streams in the SLRW, especially at station 09LS075.

### EphemeropteraPct

Buchwalter (2013) concluded that sulfate and/or bicarbonate are the likely drivers of reduced macroinvertebrate diversity and abundance (particularly mayflies) in mining impacted streams in West Virginia. The mayfly abundance metric *EphemeropteraPct* measures the relative abundance of mayflies in the macroinvertebrate sample collected. This metric was selected to evaluate a potential negative response in mayfly abundance in the presence of elevated sulfate concentrations and high specific conductivity. As this metric was also used to evaluate specific conductivity as a stressor, it was previously discussed on page 318.

Relative abundance of mayfly individuals varied considerably among the three West Two River monitoring stations. Abundance at stations 12LS002 and 09LS073 were comparable or higher than the median value observed at non-impaired stations of the same MIBI class. Station 09LS075, located just downstream of the West Two Reservoir, had much lower mayfly abundance values than the vast majority of comparable streams in the SLRW. Similar to other biological response metrics evaluated for these sites, the lack of mayfly individuals at this station may be more closely linked to the poor dissolved oxygen regime associated with this site.

### Lack of Intolerant Taxa

Rankin (2003) observed a decline or lack of sensitive macroinvertebrate taxa where sulfate concentrations exceeded 200 mg/L. The MPCA's biological metric *Intolerant2lessChTxPct* is used to quantify the relative percentage of taxa that are considered sensitive or intolerant of disturbance in Minnesota streams. Macroinvertebrate taxa qualifying for this metric accounted for relatively small percentage of the community at the three West Two River monitoring stations. Intolerant taxa accounted for 10-12% of the total taxa present at these monitoring stations. By comparison, the median values observed at comparable, high-quality reference stations range were between 20 – 26%. Based on these results, the relative abundance of intolerant macroinvertebrate taxa is lower than high quality streams in the SLRW. However, these intolerant organisms respond negatively to a wide variety of stressors, so it is impossible to conclude that their absence from these stations is due to sulfate toxicity alone.

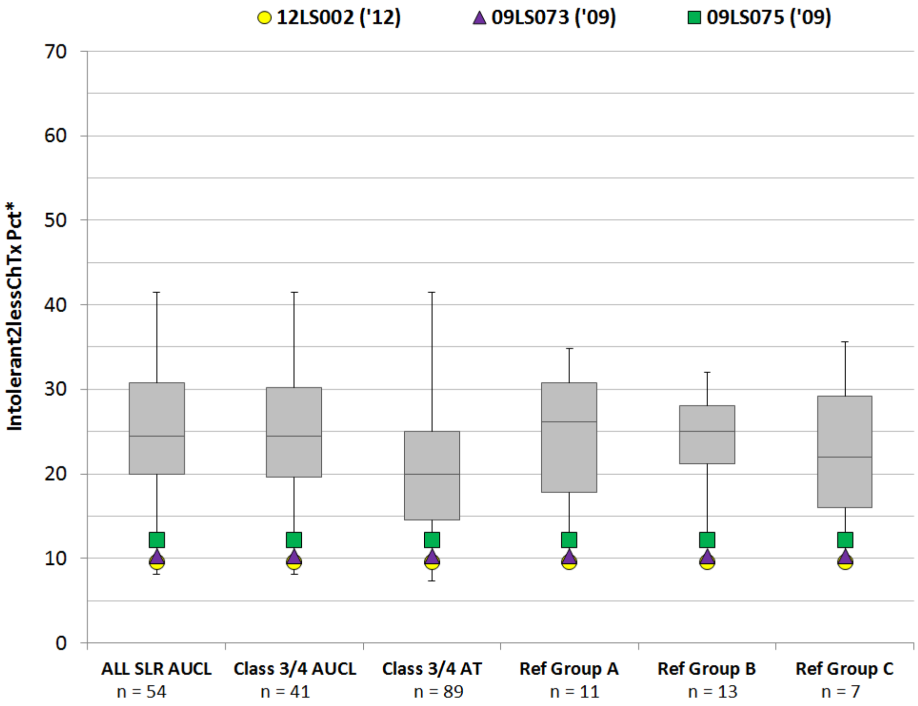


**Figure 320:** The percent of macroinvertebrate taxa present that are considered "intolerant" to disturbance at West Two River monitoring stations compared to results from high quality stations of the same IBI class. * See Section 4 for explanation of TIVs **AUCL** = Above Upper Confidence Limit of FIBI threshold **AT** = Above FIBI Threshold

### Summary: Is sulfate toxicity a stressor in the West Two River?

Sulfate concentrations in the impaired reach of the West Two River are elevated well above natural background conditions for the SLRW, and exceed some of the toxicity thresholds that are included in scientific literature and water quality standard development in the United States and Canada. However, current research suggesting that water hardness and chloride concentrations affect the toxicity of sulfate must be considered for this watershed. The hard water (150 – 450 mg/L $CaCO_3$) and moderately high chloride levels (10 – 25 mg/L) found in the West Two River are likely reducing the potential for sulfate to be toxic to aquatic life. After incorporating water hardness and chloride values in the analysis, the current sulfate concentrations in the West Two River would not be high enough to be exceeding water quality standards currently being implemented in Iowa, Illinois, Indiana, and Pennsylvania.

Several of the biological metrics evaluated are showing symptoms of a sulfate stressor, but the results are inconclusive due to the potential for confounding responses. Several of the symptoms observed -- such as low EPT richness, low mayfly richness, and low numbers of intolerant taxa -- can be caused by a wide range of stressors, and it is impossible to resolve that sulfate is a stressor with a high level of confidence provided that the water quality data is also somewhat inconclusive. Without a sulfate standard that can be applied specifically to Minnesota streams and rivers, it is difficult to eliminate or diagnose sulfate as a cause of impairment based on water quality or biological data alone.

### 5.20.4 West Two River: Summary of Stressors to Aquatic Life

**Table 93:** Summary of SID results for West Two River

| Candidate Cause | Result |
|---|---|
| **Low Dissolved Oxygen / High Dissolved Oxygen Flux** | ● |
| **Sulfate Toxicity** | ○ |
| **Elevated Specific conductivity** | ○ |
| Key: ● **= confirmed stressor** ○ **= Potential Stressor X = eliminated candidate cause** | |

## 5.21 Unnamed Tributary to McQuade Lake (Kinney Creek)

The Unnamed Tributary to McQuade Lake (refered to as “Kinney Creek” from here on) originates in the mining region near the towns of Buhl and Kinney. The upper reaches tend to be steeper, with streches of Rosgen B type channel and higher-gradient C channels. The valley types in this area vary, from wide former lake bottoms to terraced alluvial valleys, to steep colluvial valleys. There is a short stretch of ditched channel just downstream of Highway 25. The impaired AUID follows a consistent alternating pattern of C channels in alluvial valleys and E channels in lacustrine valleys. The impaired reach drops 44 feet in just over 8 miles (0.1% slope).

Fish and macroinvertebrate communities were sampled at two stations on Kinney Creek. The original station, 09LS074, was established in 2009 and was first sampled in the fall of that year. Data from this station were used to form the basis of the MIBI impairment listing for this stream. In 2011, follow-up monitornig was conducted at station 09LS074 again, as well as a new station further upstream (11LS073). Results from the 2011 sampling met the MIBI standard at both locations, but remained within the confidence interval of the impairment threshold (Table 94). Based on the results as a whole, the magnitude of the MIBI impairment in Kinney Creek does not appear to be as severe as other nearby streams (e.g. West Two River / Manganika Creek). Several mayfly taxa (*Acerpenna, Baetis*) and caddisfly taxa (*Cheumatopsyche, Micrasema, Neurclipsis*) were present in relatively high numbers within the impaired reach, which is another indication that this stream is probably not as severely impacted.

The absence of Plecoptera (stonefly) and Odonata taxa (dragonfly) at station 09LS074 is the primary reason that MIBI scores were low enough to list this reach as impaired. Several other characteristics of the invert community at this site also factored heavily into the low MIBI score; including the dominance of several taxa and a high relative percentage of non-insect taxa.

**Table 94:** Biological monitoring stations on Kinney Creek and MIBI results compared to standards

| Macroinvertebrate Assessments | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | Invert IBI Class | Invert IBI Result *(visit year)* | Invert IBI Result *(visit year)* | Standard | IBI Lower Confidence Limit | IBI Upper Confidence Limit |
| 11LS075 | 15.37 | 0.08 | 3 | 3 | **60.00** *(2011)* | | 53 | 40.40 | 65.60 |
| 09LS074 | 17.50 | 0.08 | 3 | 3 | **41.37** *(2009)* | **54.05** *(2011)* | 53 | 40.40 | 65.60 |

## Candidate Causes for Impairment

Water quality and physical habitat data were used to develop a working list of candidate causes for the MIBI impairment in Kinney Creek. Ultimately, the following candidate causes were selected for detailed analysis in this report;

1. **Low DO / High DO Flux**
2. **Elevated Specific conductivity**
3. **Sulfate Toxicity**
4. **Poor Physical Habitat Conditions**

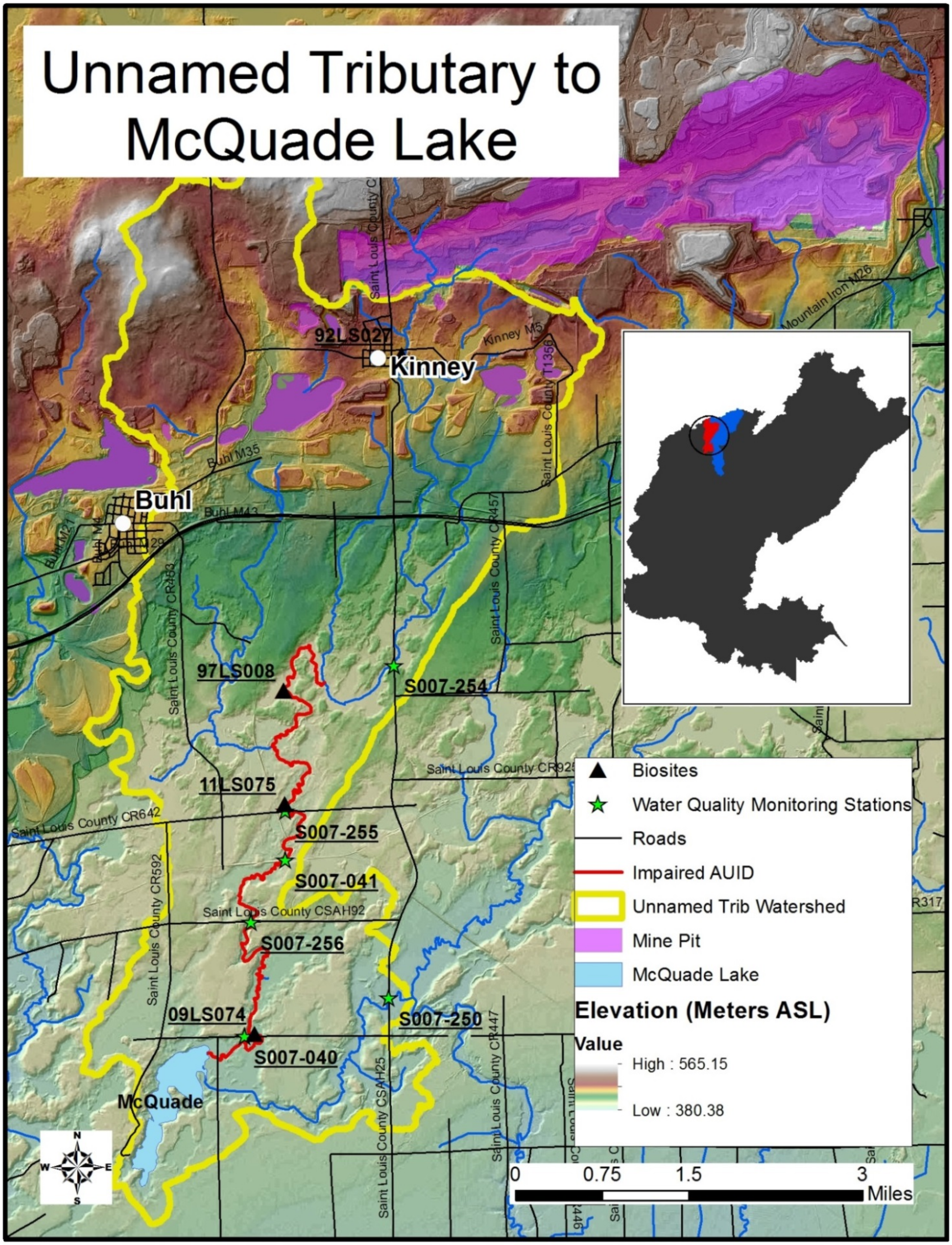


**Figure 321:** Map of the Kinney Creek Watershed, impaired stream reach, and sampling stations

## 5.21.1 Dissolved Oxygen

Instantaneous DO data are available for 5 stations on Kinney Creek, although several stations were only visited a single time. The majority of instantaneous DO data were collected at station S007-040, which is co-located with the biological monitoring station that served as the impetus for the impairment listing (09LS074). The DO concentrations ranged from approximately 5 mg/l to 9 mg/L during the months spanning June through September (Figure 322). DO concentrations in were fairly low at four of the monitoring stations in August of 2013, but none of the results fell below the water quality standard of 5 mg/L.

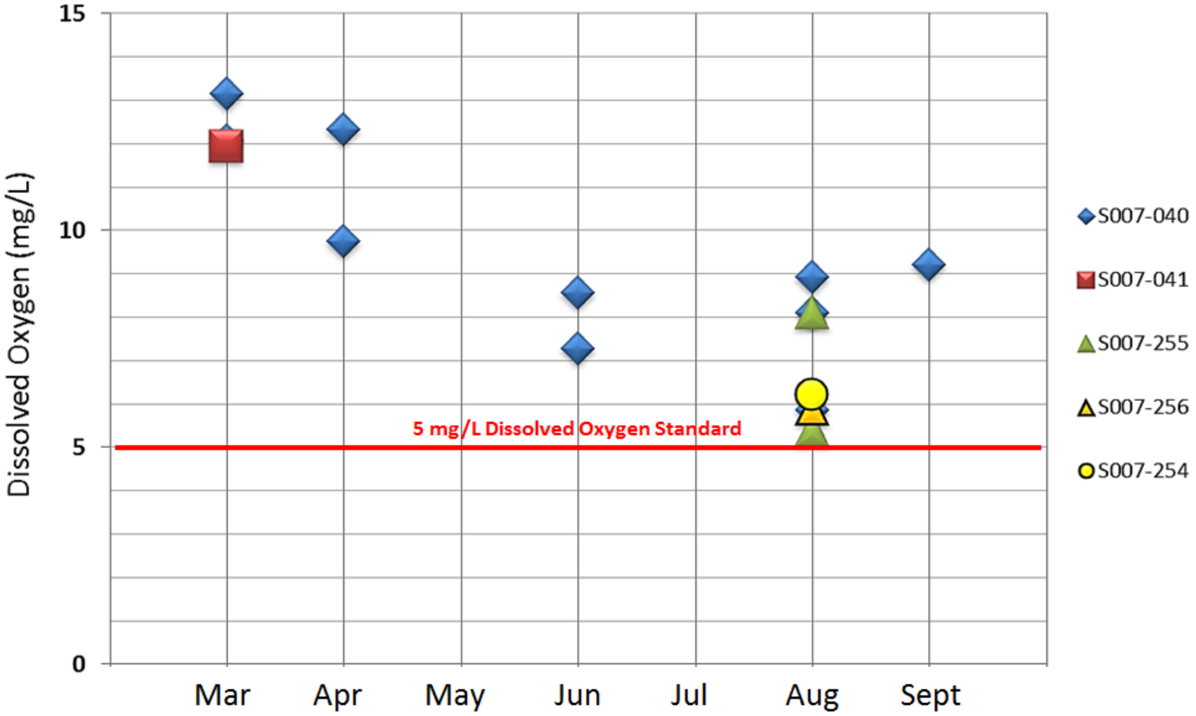


**Figure 322:** Point measurements of DO collected from Kinney Creek monitoring stations

Continuous Dissolved Oxygen Data

Continuous DO monitoring data were collected at two locations on Kinney Creek during the months of July and August in 2011 and 2013. The continuous monitoring sites (S007-040 and S007-255) were co-located with the two biological monitoring stations in order to better evaluate relationships between the biota and DO regime. Multi-parameter water quality monitoring YSI sondes were deployed for a period of 5-7 days at each site and set to record DO concentrations at 15-minute intervals (Figure 323 - 325).

Minimum DO concentrations were found to be suitable for supporting warm and coolwater aquatic life at both locations during the two diurnal monitoring periods. Station S007-255 consistently had higher daily maximum and lower daily minimum concentrations than station S007-040, but both sites maintained DO concentrations above the minimum.

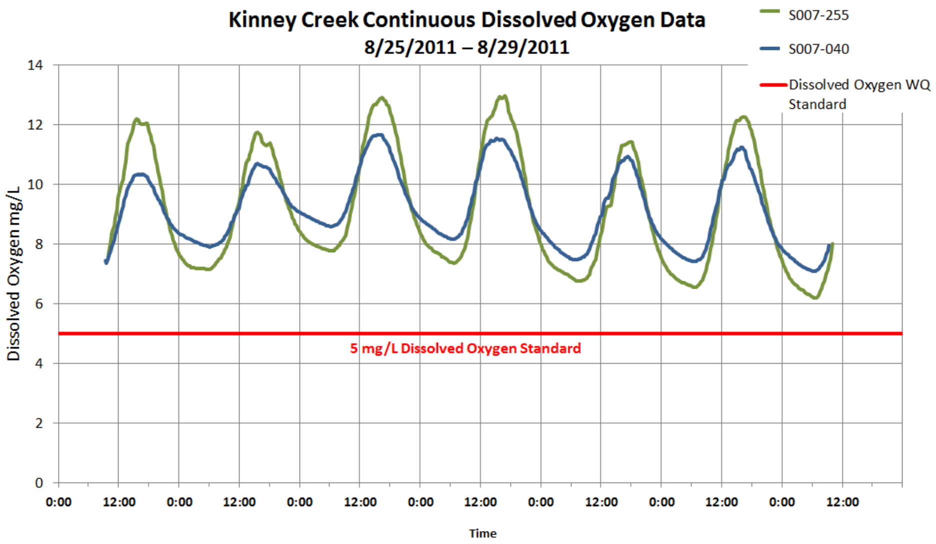


**Figure 323:** Continuous DO data from two Kinney Creek biological monitoring stations in August 2011

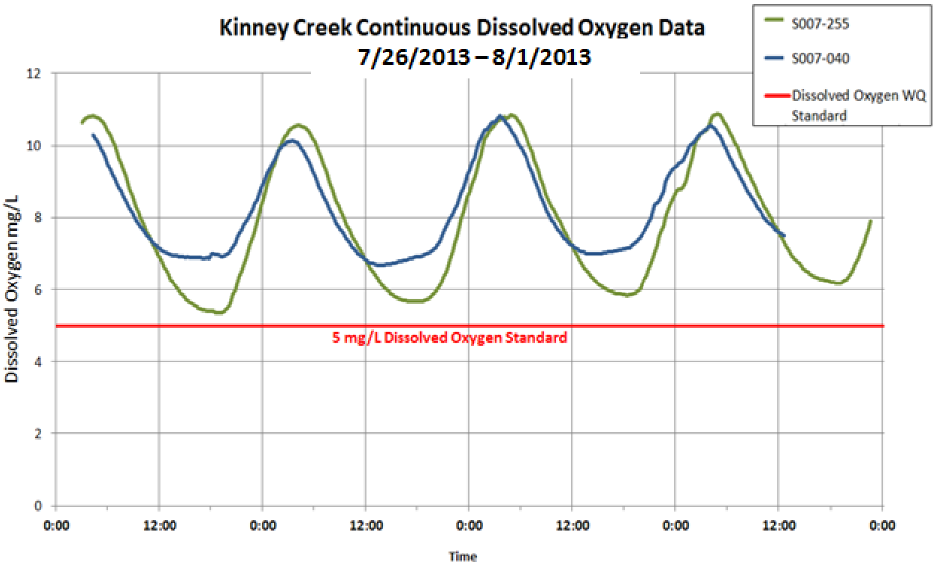


**Figure 324:** Continuous DO data from two Kinney Creek biological monitoring stations in July 2013

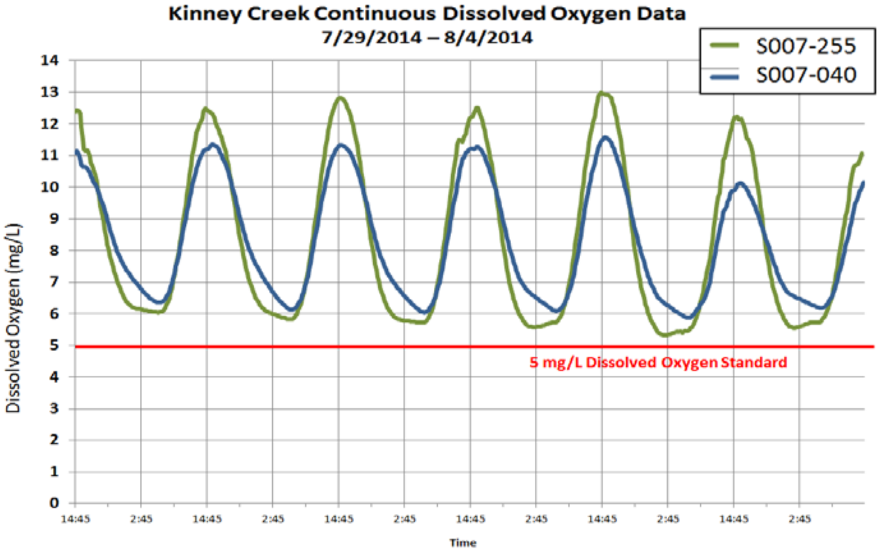


**Figure 325:** Continuous DO data from two Kinney Creek biological monitoring stations in July 2014

Minimum DO concentrations were not found to drop below water quality targets (5 mg/L). However, diurnal DO flux was quite high at both monitoring stations, particularly S007-255. While these short-term deployments of DO loggers may not have measured low minimum values, the large swings in concentration are probably an indicator that there is a high potential for low DO conditions to occur in the system at some point during the year, depending on streamflow and atmospheric conditions. DO levels can become very low during high temperatures, low flow conditions, or during the fall when algae and other plants begin to senesce.

Wide diurnal fluctuation in DO concentrations can also stress aquatic organisms, mostly due to the physiological stress resulting from swings in DO. The MPCA's River Nutrient Criteria (Heiskary 2013) lists a diurnal DO flux greater than 4.0 as a stressor variable for streams and rivers of northern Minnesota. Diurnal DO flux in Kinney Creek exceeded 5 mg/L at station S007-255 during both the August 2011 and July 2013 continuous monitoring periods. In August of 2014, DO flux exceeded 7 mg/L at station S007-255 (Figure 326). DO flux was also greater than 4.0 mg/L at station S007-040 in July of 2013, but only for one, 24-hour period. Based on the continuous monitoring data available, DO flux can be considered a candidate stressor in Kinney Creek, and may be more problematic than low minimum DO concentrations.

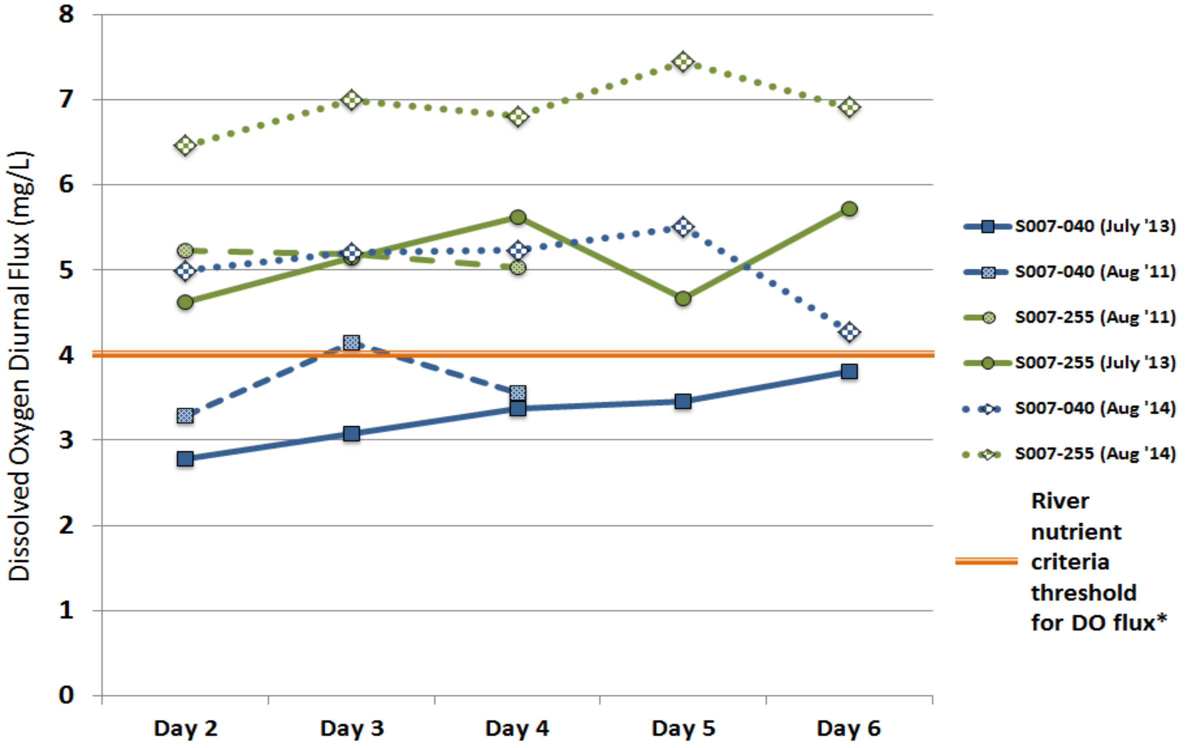


**Figure 326:** 24-hour (diurnal) fluctuation in DO concentration observed at Kinney Creek monitoring stations during continuous monitoring

## Sources and Pathways Contributing to Dissolved Oxygen Stress

Nutrient enrichment, Chl-a concentrations, and measures of BOD are all factors in the DO regime of streams and rivers. The MPCA has developed nutrient criteria for Minnesota rivers with thresholds for TP and several related stressor effects linked to excess nutrients -- high diurnal DO flux, high Chl-a concentrations, and elevated BOD levels. See Section 3.1.3 for more information on the river nutrient criteria. Kinney Creek data, in combination with various benchmarks provided by the river nutrient criteria, are discussed here to investigate potential pathways and sources causing DO stress.

Total Phosphorous

The TP data available for Kinney Creek are fairly limited. Only six results are available from a single monitoring station (S007-040 / biological monitoring station 09LS074), and all of the data are from the 2012 monitoring season. The results show elevated TP concentrations (0.160 mg/L) during the spring snowmelt runoff period and considerably lower concentrations throughout the rest of the summer and early fall (avg=0.03 mg/L, max=0.038, min=0.019). Aside from the snowmelt sample, all TP results are below the draft river nutrient criteria of 0.055 mg/L. Additional monitoring may be required to better understand how TP concentrations vary from year to year, but these data indicate that Kinney Creek is not a highly eutrophic stream.

Biological Oxygen Demand

The BOD data are very limited for this impaired stream. The two biological monitoring stations were each sampled a single time in August of 2014. The BOD concentrations at the upstream station (11LS075) was 8.4 mg/L, which is extremely high compared to the expected value for healthy streams in the Northern River Nutrient region as defined by Heiskary et al (2013). A DO Flux of > 7 mg/L was observed at this station when the BOD value of 8.4 mg/L was observed. Farther downstream at biological monitoring station 09LS074, BOD concentration decreased to 2.3 mg/L. Both of these values are greater than the 1.5 mg/L value listed in Heiskary et al (2013) as a potential indicator of eutrophication.

Chl-a

No data available.

### Biological Response to Dissolved Oxygen Stress

Macroinvertebrate Community Dissolved Oxygen Index

Macroinvertebrate DO Index scores at the three Kinney Creek biological monitoring stations are compared to scores from high-quality stations in Figure 327. DO index scores at Kinney Creek stations are comparable to or more favorable than many of the index scores observed at comparable reference streams in the SLRW. These results indicate that the macroinvertebrate community of Kinney Creek is not highly tolerant of low DO conditions. However, the variability in the DO index score at station 09LS074 between the 2009 and 2011 sampling events show a potential for the community to shift to more DO tolerant organisms at times. Over 75% of class 3 MIBI stations in the SLRW with a passing IBI score had a higher DO index value than the 2011 visit to station 09LS074. A shift in the macroinvertebrate community away from mayflies (Baetis) to more tolerant amphipod crustacean (Hyallela) at station 09LS074 was responsible for the drop in DO index scores observed in the two sampling events.

The DO index score at station 11LS075 was very good -- scoring well above the 75th percentile values of comparable reference streams (Figure 327). This station routinely had the higher DO flux (> 7 mg/L in August of 2014) of the two biological monitoring stations, so it does not appear that DO flux is negatively influencing DO index values in Kinney Creek. The DO index scores are based on an organism's sensitivity to low DO, and DO flux was not considered in the development of the index scores. Therefore, the DO index score may not be a fully appropriate metric to evaluate a DO flux stressor.

EPT Taxa Richness and Overall Taxa Richness

The EPT taxa richness and overall taxa richness are known to decline in streams with low DO concentrations or high DO flux. The number of EPT taxa observed from Kinney Creek monitoring stations ranged from 10 to 15 taxa. These results are comparable to the median EPT taxa richness observed high quality reference sites, which ranged from 12 to 16 taxa (Figure 327). The station with the lowest EPT taxa richness, 11LS075, plots below the 25th percentile result from stations scoring above the upper confidence limit of MIBI threshold (i.e. stations with good to excellent MIBI scores). This provides some evidence that EPT taxa richness may be slightly depressed at this monitoring station compared to less impacted streams in the SLRW.

Nearly 66% of the macroinvertebrate individuals sampled at station 09LS074 (2009 sample) were from EPT families. EPT taxa that were abundant at this station include *Baetis* and *Acerpenna* mayflies, caddisflies from the genus *Cheumatopsyche*, *Neureclipsis*, *Hydropsychidae*, and *Micrasema*. No Plecopteran (stonefly) taxa were observed at this station, or any other station in Kinney Creek. Overall macroinvertebrate taxa richness in Kinney Creek is lower than many of the comparable reference streams in the SLRW (Figure 327). Taxa richness at the two monitoring sites ranged from 24 to 33 taxa. Both monitoring stations on Kinney Creek were at or below the 25th percentile taxa richness values of comparable reference streams.

Insect Taxa %

Increases in DO flux can lead to an increase in less desirable non-insect aquatic species such as aquatic worms (Oligochaeta), midge larvae (Chironomidae), and snails. Between 82-88% of the macroinvertebrate taxa observed in Kinney Creek are considered aquatic insects (Figure 327). The median values for the reference sites compiled in Figure 327 range from 85-90%, with upper quartile values extending into the 92% range. Overall, the relative abundance of insect taxa is slightly lower in Kinney Creek, but comparable to many high quality stations in the SLRW.

**Summary: Is low DO/DO flux a stressor in Kinney Creek?**

Low DO concentrations can be eliminated as a cause of impairment in Kinney Creek. No DO measurements collected within the impaired reach produced results below the 5 mg/L DO standard.

Diurnal DO flux at the two biological monitoring stations frequently exceeds 4 mg/L, which is the value cited as a potential response variable to river eutrophication in the “Northern Rivers” nutrient region of Minnesota (Heiskary, 2013). Diurnal DO flux exceeded 7 mg/L at station S007-255 in August of 2014. Despite the elevated DO flux observed at these stations, macroinvertebrate data were somewhat inconclusive in terms of showing a biological response to this candidate stressor. The number of EPT taxa observed in Kinney Creek was comparable to many high quality stations in the SLRW with comparable drainage area and habitat types. Other metrics, such as the relative abundance of insect taxa and overall taxa richness were more indicative of stress from DO flux. Based on these conflicting results, DO flux cannot be eliminated or diagnosed as a stressor to aquatic life at this time.

Additional investigation of DO flux in this system is recommended. Application of HSPF modeling data to simulate DO conditions and the various potential drivers of DO flux (biological oxygen demand, phosphorous) could be informative in planning restoration activities to restore a more suitable DO regime for aquatic life.

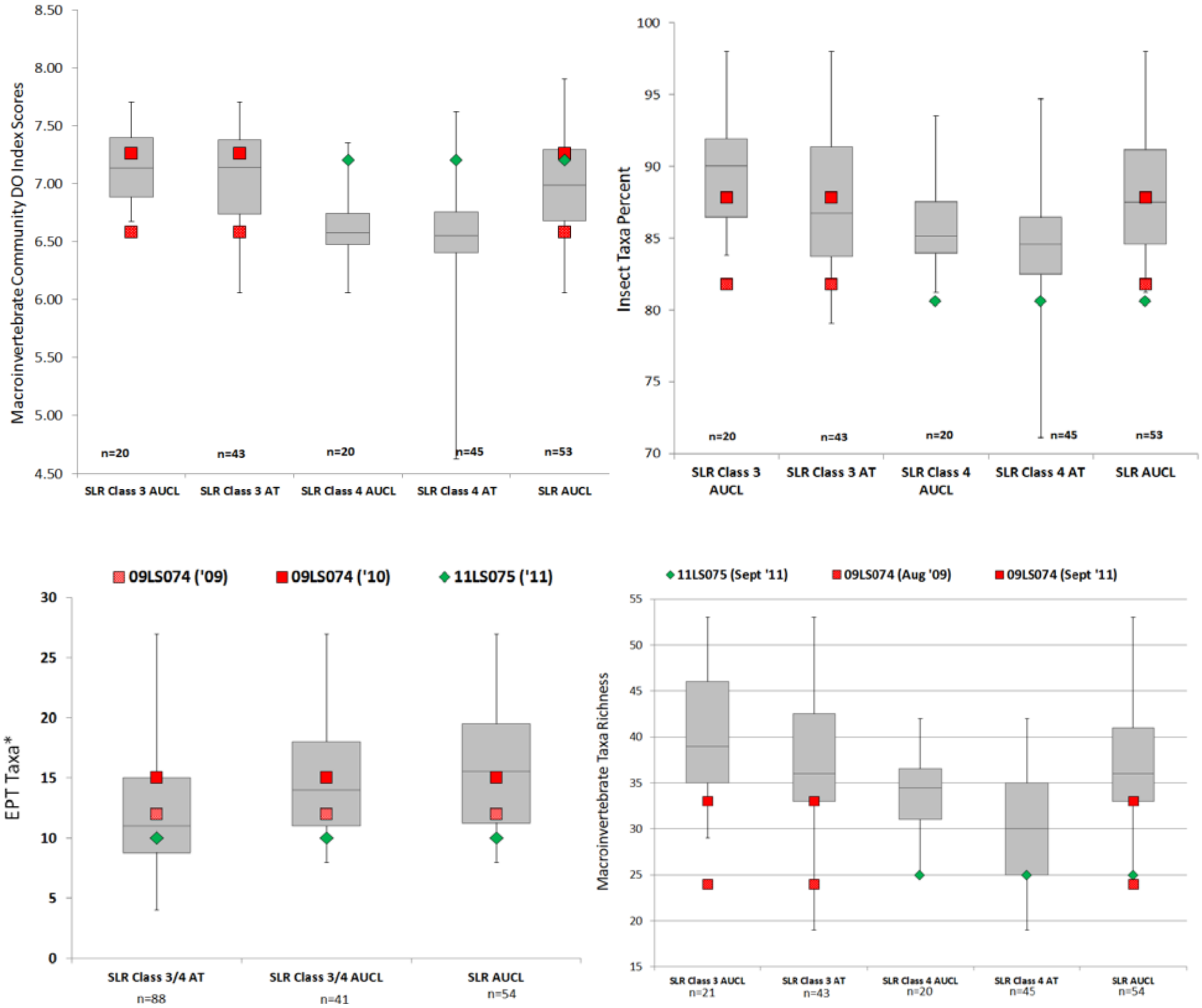


**Figure 327:** Comparison of results for several macroinvertebrate metrics between Kinney Creek stations and non-impaired SLRW reference stations: macroinvertebrate community DO tolerance indicator values (TIV) (top left), % Insect Taxa (top right); EPT Taxa Richness (lower left); overall macroinvertebrate taxa richness (lower right) ***** See Section 4 for explanation of TIVs **SLR**=St. Louis River Watershed **AUCL** = Above Upper Confidence Limit of FIBI threshold **AT** = Above FIBI Threshold

### 5.21.2 Specific conductivity

Specific conductivity (specific conductivity) was identified as a candidate cause for impairment in Kinney Creek based on water quality data and land use factors. A plot of available point measurements of specific conductivity are shown in Figure 328. Specific conductivity levels exceed 1,000 µS/cm in the headwaters region of Kinney Creek Watershed, but are typically somewhat lower (700-900 µS/cm) in the impaired reach further downstream where the biological monitoring stations are located. The highest specific conductivity readings in this stream have been observed in August and September during low flow conditions.

Continuous monitoring for specific conductance was completed at two monitoring stations in July of 2013 and August of 2011. The two sites selected for continuous monitoring were co-located with biological monitoring stations. Continuous monitoring equipment was deployed for a period of 3-6 days, with specific conductivity readings collected at 15 minute intervals. Results show conductivity levels in the range of 750 – 850 µS/cm in the August 2011 sampling period and slightly lower readings in the range of 675 – 775 µS/cm during the July 2013 sampling. During both monitoring periods, specific conductivity was higher at station S007-255 (bio site 11LS075), which is located approximately three river miles upstream of the other station where equipment was simultaneously deployed, S007-040 (bio station 09LS074). This observation is consistent with other monitoring events which show conductivity levels in Kinney Creek decreasing in an upstream to downstream direction (see Figure 329).

A longitudinal monitoring profile of specific conductivity levels in Kinney Creek was completed on September 25, 2014. Six monitoring sites were selected to evaluate changes in conductivity from the headwaters to the mouth. The results show a clear pattern of decreasing conductivity in an upstream to downstream direction (Figure 330). A sharp decrease in conductivity was observed between stations 3 and 4 (difference of 106 µS/cm). Several factors may be contributing to the decrease in conductivity observed between these monitoring locations; (1) the presence of bogs and wetlands, (2) tributary streams entering between these two stations, several of which drain lake and wetland complexes, and/or (3) the distance between these two monitoring sites is longer than the gap between other monitoring sites included in this longitudinal profile. At the two biological monitoring stations within the impaired reach, conductivity levels were around 850-870 µS/cm.

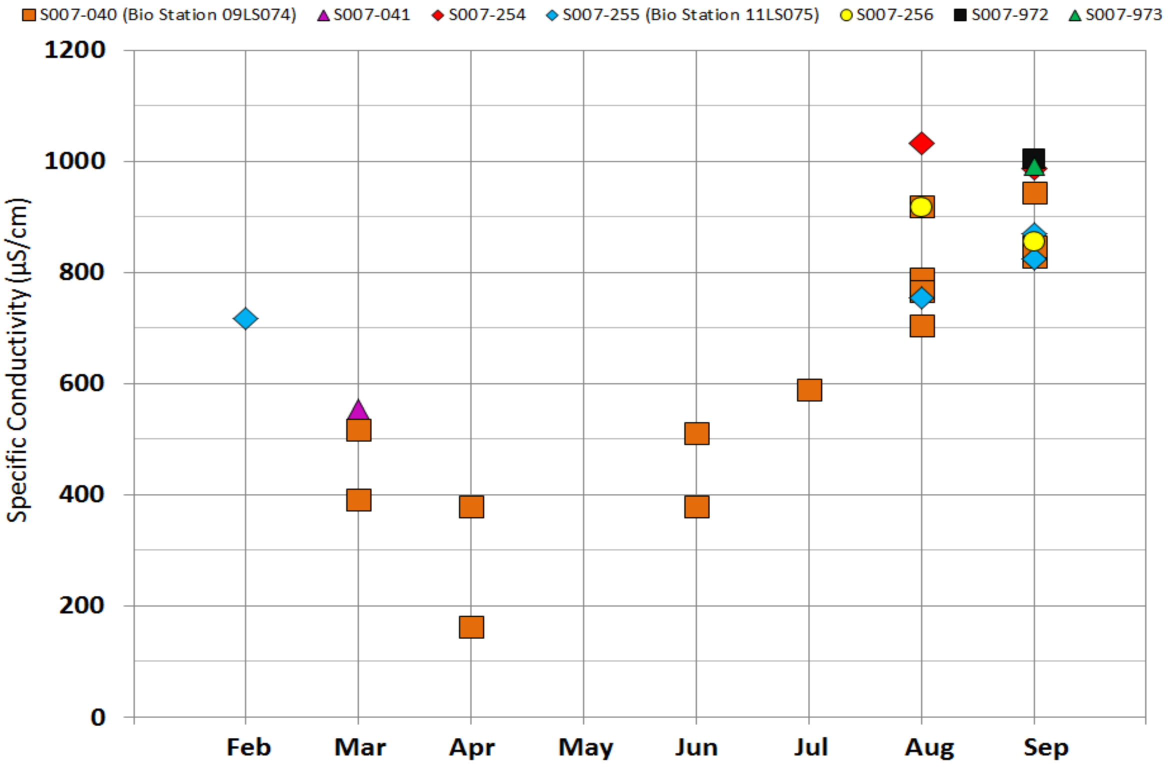


**Figure 328:** Point measurements of specific conductivity collected from Kinney Creek monitoring stations

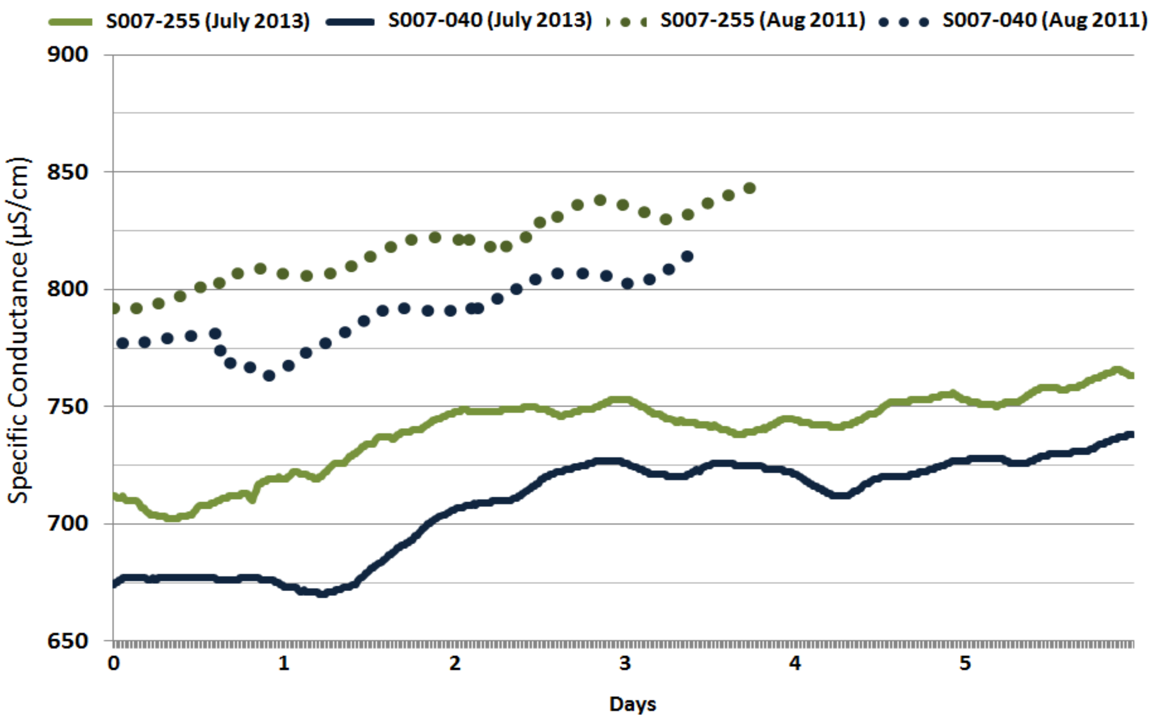


**Figure 329:** Continuous specific conductivity monitoring results from several biological monitoring stations on Kinney Creek

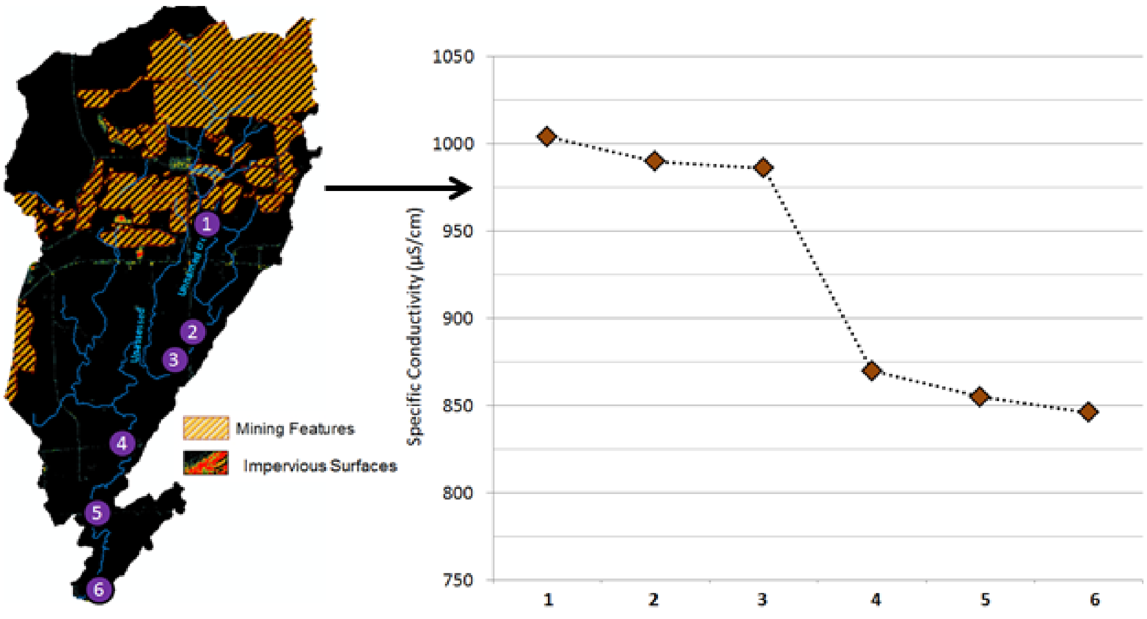


**Figure 330:** Longitudinal specific conductivity results from Kinney Creek; September 2014

**Figure 330:** Longitudinal specific conductivity results from Kinney Creek; September 2014

## Sources and Pathways of Specific conductivity

Inputs from mine pit dewatering are the primary sources of elevated specific conductivity in the surface waters of Kinney Creek. Seasonal application of road salt to county, state, and federal highways (particularly Highway 169) may cause short terms spikes in specific conductivity during the winter months. However, permitted discharges from mining impacted areas of the watershed are the source of sustained periods of high conductivity seen in the monitoring data, and have the highest potential to influence aquatic life in the impaired reach of Kinney Creek. Several other point source discharge permits have been issued in the Kinney Creek Watershed, but none of these are expected to have a significant impact on specific conductivity.

Several permitted discharges to Kinney Creek originate from the US Steel Minntac Mining Area near Kinney, Minnesota. According to the most recent NPDES permit (December 2013), there are two dewatering routes that deliver water to the stream. Specific information on these discharges is provided in Table 95. Based on the permit language and available monitoring data, it can be concluded that discharge point MN0052493-SD-3 is a primary source of surface water in the headwaters of Kinney Creek. An average of 8 cubic feet per second (cfs) of mine pit water is pumped into Kinney Lake via this discharge, located in the extreme headwaters of the watershed. Specific conductivity values of this discharge range from 689 µS/cm to 1389 µS/cm, with an average of 921 µS/cm. This discharge accounts for the vast majority of flow in Kinney Creek and is a primary driver of the elevated specific conductivity readings observed throughout the entire length of the creek.

**Table 95:** List of point source permitted dischargers to Kinney Creek

| Facility / Permit # | Discharge Point | Permit Details | Monitoring Data |
|---|---|---|---|
| **US Steel Corporation – Minntac Mining Area / MN0052493** | MN0052493-SD-3 | Discharge to Kinney Lake, Kinney Creek, and wetlands at average and maximum rates of 13 cfs and 33 cfs. | Monitoring data from 1999 – 2014 shows a typical discharge of around 8 cfs, and a maximum discharge around 31 cfs (September 2001). Specific conductivity of discharge averages 921 µS/cm (min=689 µS/cm; max = 1386 µS/cm) |
| **US Steel Corporation – Minntac Mining Area / MN0052493** | MN0052493-SD-2 | Overflow from pit at average and maximum rates of 13 cfs and 26 cfs. | No monitoring data available from 1999 - 2014. This does not appear to be a regularly used point of discharge. |
| **Mesabi Bituminous Inc./ MNG490021** | 3 Stormwater permits located in Kinney Creek Watershed | 100 mg/L TSS limit. Monitoring required two times per year. | No data available |

Samples were collected at station S007-040 (biological station 09LS074) during baseflow, rain event flow, and snowmelt flow and analyzed for major cations and anions. The most common salt ions found in surface water that influence specific conductivity levels include positively charged (cations) $Ca^{2+}$, $Mg^{2+}$, $Na^{+}$, $K^{+}$, and negatively charged (anions) $HCO_3^{-}$, $CO_3^{2-}$, $SO_4^{2-}$, and $Cl^{-}$ (CADDIS, 2012). Figure 331 shows the concentrations of various cations and anions observed in Kinney Creek compared to two other SLRW streams of similar size that are not impacted by mining (control streams). The baseflow sampling results show elevated concentrations of $SO_4$, Mg, and Ca in Kinney Creek compared to the control streams. These three ionic compounds appear to be the prominent drivers of high conductivity in Kinney Creek. Sulfate, in particular, was highly elevated in the baseflow samples (> 150 mg/L) and shows a strong correlation with specific conductivity in these three sampling events. Chloride concentrations were relatively low (< 25 mg/L) during all three sampling events. Elevated sulfate concentrations are discussed as an independent stressor in Section 5.20.3, but sulfate and specific conductivity show a strong positive relationship in this watershed, and their effects on aquatic life may be confounding.

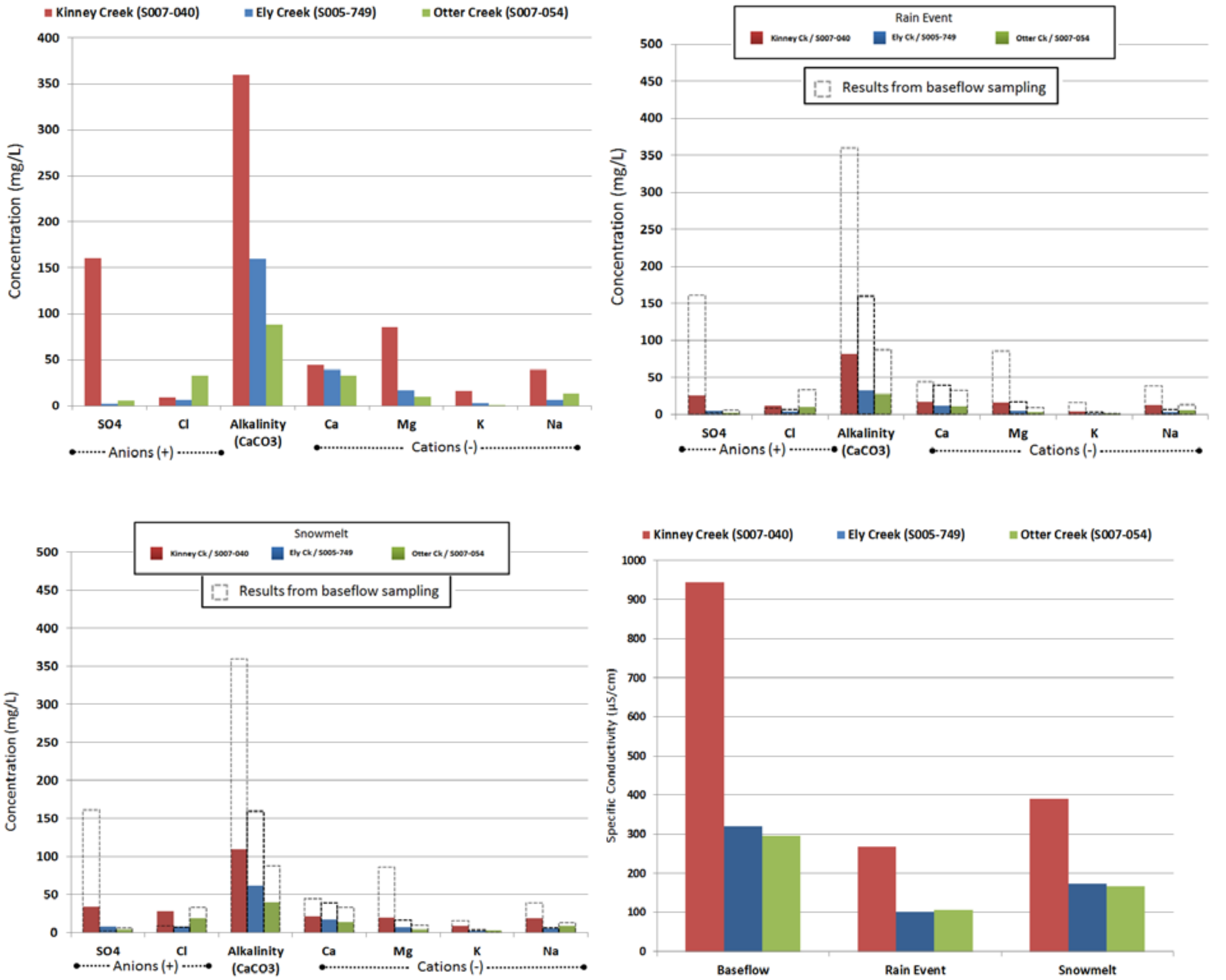


**Figure 331:** Geochemistry sampling results for major cations and anions in Kinney Creek under various flow conditions; baseflow (upper left), rain event (upper right), snowmelt (lower left). Specific conductivity comparisons for Kinney Creek, Ely Creek, and Otter Creek during the same sampling events (lower right).

## Biological Response to Specific conductivity – Kinney Creek

The effects of elevated conductivity on aquatic life were evaluated using data from Minnesota streams and scientific literature. A summary of the background information used to derive these response metrics is presented in section 3.1.6. Based on this work, several biological metrics were selected to evaluate specific conductivity as a cause of MIBI impairment in Kinney Creek.

**Table 96:** Summary of biological metrics and literature used to evaluate elevated specific conductivity as a stressor

| Metric | Response to Increased Specific conductivity / Conductivity | Source |
|---|---|---|
| **EPT Richness** | Decrease | Roy et al (2003); Echols et al (2009); Johnson et al (2013) |
| **Overall Taxa Richness** | Decrease | Johnson et al (2013) |
| **Ephemeroptera Richness** | Decrease | Pond (2004); Hassel et al (2006) |
| **Fish and Macroinvertebrate Tolerance Indicator Values (TIV)** | Increase | MBDI (Yoder and Rankin, 2012) |

### EPT Richness

The EPT taxa richness in Kinney Creek ranges from 11 to 16 taxa, depending on the station and sampling year (Figure 332). These results are comparable to the median EPT taxa richness observed at high quality stations of the same MIBI class in the SLRW. This observation provides evidence against specific conductivity as a stressor in Kinney Creek, as EPT taxa are typically reduced in streams with specific conductivity levels outside of the suitable range for sensitive aquatic life.

### Overall Macroinvertebrate Taxa Richness

Macroinvertebrate taxa richness at Kinney Creek monitoring stations ranged from 31 to 51 taxa. Considerable variability between sampling events occurred at station 09LS074, where a difference of 20 taxa was observed between 2009 and 2011 sampling events. The 2011 sample from this station resulted in an IBI score of 54, which is narrowly above the impairment threshold of 53 (IBI score of 41 was observed in 2009). Overall, taxa richness in Kinney Creek is lower than the majority of comparable reference sites from the same MIBI class in the SLRW (Figure 332). The variable results from 09LS074 demonstrate the ability of this stream to support a diversity of macroinvertebrate taxa that is comparable to non-impaired reference sites under certain conditions.

### Ephemeroptera Taxa Richness

Ephemeroptera (mayfly) taxa richness ranged from 4-6 taxa at Kinney Creek monitoring stations. The median value for comparable sites in the SLRW scoring above the impairment threshold ranged from 5-6 taxa, so there is no appreciable difference in this metric between Kinney Creek monitoring sites and many SLRW sites that scored above the MIBI threshold (Figure 332). In other impaired SLRW streams where elevated specific conductivity is a candidate cause for impairment (e.g. Manganika Creek and Spring Mine Creek), Ephemeroptera richness was in the range of 0 to 1 taxon). The presence of 4-6 mayfly taxa in Kinney Creek suggests that the effects of specific conductivity are less severe in this watershed.

### Specific Conductivity Tolerance Indicator Values (TIV)

Macroinvertebrate community TIVs for specific conductivity are compared to results from high quality SLRW stations in Figure 332. The TIV results for Kinney Creek stations are all well above the 75th percentile values of comparable reference sites, which means that the overall species composition at Kinney Creek sites favors those that are often found in streams with elevated specific conductivity. However, these results are not diagnostic of specific conductivity as the dominant stressor in this watershed, as many of the same taxa that are tolerant of elevated specific conductivity may also be tolerant of other stressors that may be present.

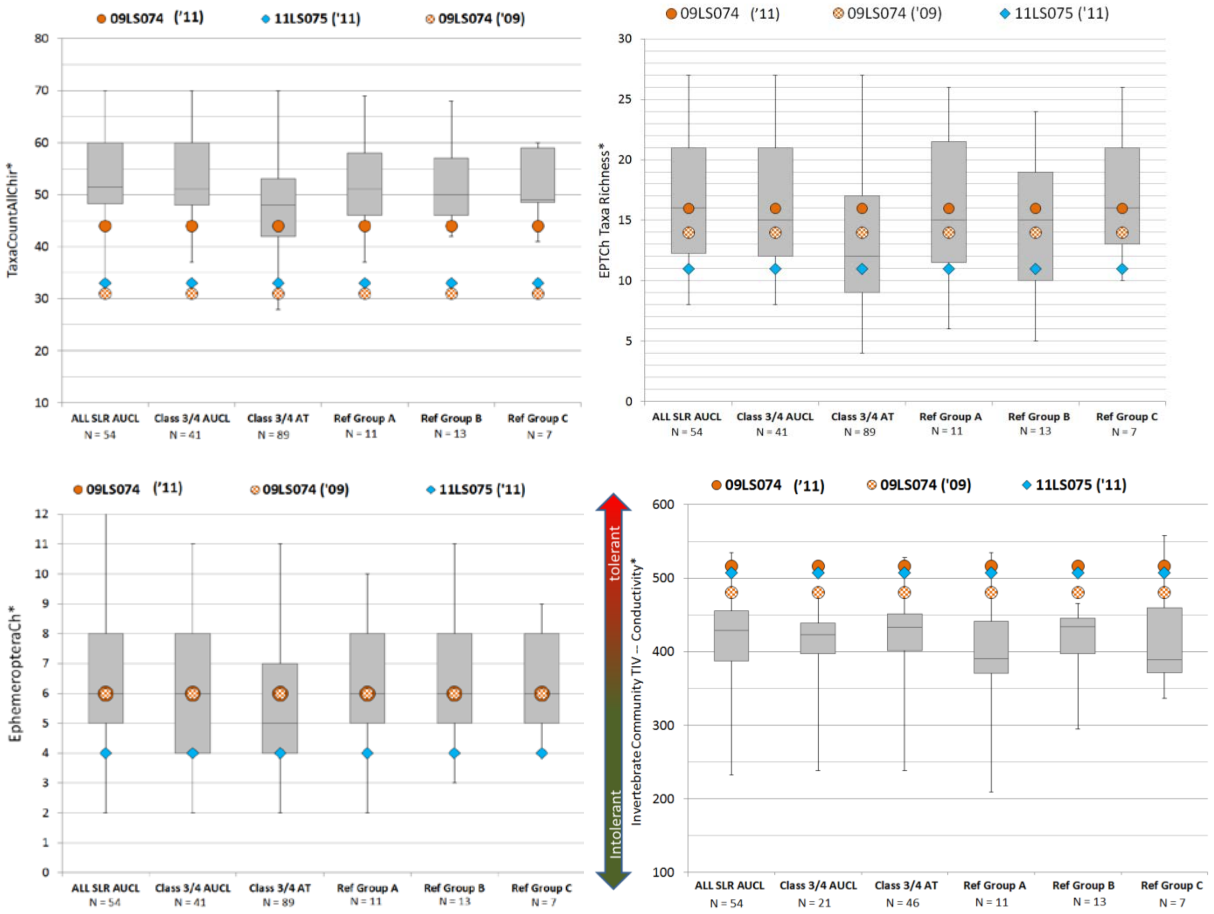


**Figure 332:** Comparison of results for several macroinvertebrate metrics between Kinney Creek stations and non-impaired SLRW reference stations: overall macroinvertebrate taxa richness (upper left); EPT Taxa Richness (upper right); Ephemeroptera taxa richness (lower left); macroinvertebrate community DO tolerance indicator values (TIV) (lower right). ** See section 4 for explanation of TIVs* ***AUCL** = Above Upper Confidence Limit of FIBI threshold* ***AT** = Above FIBI Threshold*

## Summary: Is elevated specific conductivity a stressor in Kinney Creek?

Specific conductivity levels are elevated in Kinney Creek due to inputs from mine pit dewatering in the extreme headwaters of the watershed. Elevated concentrations of sulfate, calcium, magnesium, and sodium are observed during baseflow and low flow conditions, resulting in conductivity levels that occasionally exceed 1,000 µS/cm. More moderate specific conductivity levels (300 – 600 µS/cm) are seen during the months of April through July, so the exposure to elevated specific conductivity is seasonal in this stream and tends to occur in fall and winter conditions when the streamflow is at or near baseflow conditions.

Biological response metrics for this candidate stressor are somewhat inconclusive, and do not show a clear consistent response. The number of EPT and Ephemeroptera taxa observed in Kinney Creek is comparable to many streams that achieved fair, good, or excellent MIBI scores. These taxa are generally sensitive to elevated specific conductivity, and an expected response would be a very low number or complete lack of these taxa. Several metrics did show a stressor response that could be related to specific conductivity. These include a lack of overall macroinvertebrate taxa richness, and community level TIV values that show a fairly high community level tolerance to specific conductivity. However, these metrics can be influenced by a wide variety of stressors and are not considered diagnostic of specific conductivity as a stressor. Specific conductivity should remain a candidate cause of impairment based on the elevated values observed in this stream (>1,000 µS/cm), but the evidence presented here is not definitive enough to diagnose this stressor as a cause of impairment.

### 5.21.3 Sulfate Toxicity

Sulfate was initially included as a candidate cause of MIBI impairment in Kinney Creek due to presence of mining land use in its watershed. Further analysis of the available sulfate data revealed concentrations that are significantly higher than natural background conditions in the SLRW. Seven sulfate samples were collected from station S007-040 (co-located with biological monitoring station 09LS074), most of which were collected during the open water season of 2012. The additional sample was collected through the ice in February of 2014. A maximum sulfate concentration of 161 mg/L was observed during a low flow event on September 12, 2014 (Table 97). In general, the sulfate concentrations in Kinney Creek are relatively low in comparison to many of the values cited by researchers and other governing agencies as harmful to aquatic life (see summary of standards and research in Section 3.1.6). Based on the limited data available, sulfate concentrations in this stream are the highest during late summer and fall low flow periods, with annual maximums ranging somewhere between 100-200 mg/L.

U.S. Steel's Minntac Mining area possesses a permit (MN0052493-SD-3) to discharge mine pit water to Kinney Lake, Kinney Creek, and surrounding wetlands (see details in Table 95). This discharge accounts for the only major source of sulfate in this watershed. Based on the results of effluent monitoring conducted per discharge permit requirements, the sulfate concentration of pit water pumped into Kinney Creek is between 80 – 185 mg/L (Figure 333). These concentrations are slightly higher, yet comparable to sulfate concentrations seen at the monitoring stations located near the mouth of Kinney Creek 8-9 miles downstream.

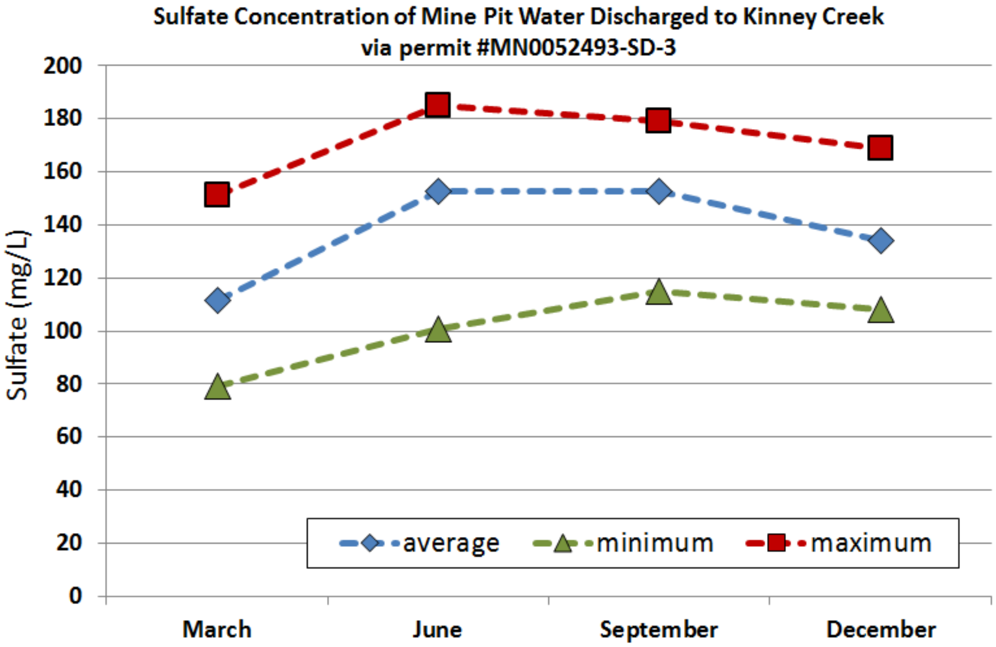


**Figure 333:** Summary of sulfate concentrations observed in the mine pit water pumped into Kinney Creek via permit MN0052493-SD-3. Monitoring data are from samples collected each March, June, September, and December from 2004 through 2014.

All of the current sulfate standards designed to protect fish and aquatic macroinvertebrates used in the United States and Canada are based on total sulfate concentration, as well as chloride concentrations and water hardness. As water hardness increases, there is evidence that sulfate becomes less toxic to fish and aquatic macroinvertebrates (Soucek and Kennedy 2004). Table 97 summarizes total sulfate concentrations observed in Kinney Creek, along with associated chloride and hardness data. Water hardness data were only available for a small number of the sampling events. Therefore, it was impossible to compare all of the results to current sulfate standards being applied in several US states (Table 97). The three sulfate results with adequate supplementary hardness and chloride data were well below the calculated WQ standards applied in the other states.

Total sulfate concentrations in Kinney Creek do exceed one of the more protective sulfate targets listed in current research. Buchwalter (2010) listed a chronic toxicity value of 124 mg/L for protecting the most sensitive forms of aquatic life, which was exceeded by the September 12, 2012 sample (161 mg/L). This chronic toxicity value does not incorporate water hardness or chloride values, and thus differs from the work done to develop water quality standards in several U.S. states (IA, PA, IL).

**Table 97:** Kinney Creek sulfate data paired with hardness and chloride results, and applicable water quality standard based on formula used by several US states.

| **Sample Date** | **Sulfate** (mg/L) | **Magnesium** (mg/L) | **Calcium** (mg/L) | **Hardness** (mg/L) | **Chloride** (mg/L) | **IA, IL, IN, PA Sulfate Standard** (mg/L) |
|---|---|---|---|---|---|---|
| 03/15/2012 | **34.2** | 20.3 | 21.9 | 138.3 | 28.4 | 1298.0 |
| 03/29/2012 | **70.9** | - | - | n/a | 16 | n/a |
| 04/16/2012 | **61.1** | - | - | n/a | 10.2 | n/a |
| 06/25/2012 | **42** | - | - | n/a | 6.35 | n/a |
| 08/01/2012 | **121** | - | - | n/a | 7.9 | n/a |
| 09/12/2012 | **161** | 85.8 | 44.3 | 464.0 | 8.55 | 2009.7 |
| 02/05/2014 | **93.4** | - | - | 340 | 13.2 | 1706.9 |

### Summary: Is sulfate toxicity a stressor in Kinney Creek?

Sulfate concentrations are elevated in Kinney Creek due to the presence of a mine pit dewatering discharge in its headwaters. Although sulfate levels occasionally exceed concentrations that are considered harmful to aquatic life by some researchers, elevated hardness and chloride concentrations are likely buffering the harmful effects of sulfate in this particular stream. All sulfate results from this watershed met water quality standards being applied in the states of Illinois, Indiana, Iowa, and Pennsylvania. The lack of a water quality standard in Minnesota presents challenges in building a defensible case for or against sulfate as a stressor to fish and macroinvertebrate communities. Based on the data and supporting information available at this time, it is unlikely that sulfate is a primary cause of impairment in Kinney Creek. Sulfate should remain a candidate cause until further research or a water quality standard is available that can improve confidence level of this decision.

### 5.21.4 Unnamed tributary to McQuade Lake (Kinney Creek)

**Table 98:** Summary of SID results for Kinney Creek

| Candidate Cause | Result |
|---|---|
| **Low Dissolved Oxygen** | X |
| **High Dissolved Oxygen Flux** | ● |
| **Sulfate Toxicity** | ○ |
| **Elevated Specific conductivity** | ○ |
| Key: ● **= confirmed stressor** ○ **= Potential Stressor X = eliminated candidate cause** | |

**Swan River – Hibbing Watershed Zone**

Impaired streams within this watershed zone include East Swan Creek, a designated trout stream south of the city of Hibbing, and a main stem reach of the Swan River, just above its confluence with the St. Louis River. Portions of this watershed zone have been highly modified due to urban development and mining activities. Many of the streams in this region of the SLRW receive effluent from municipal WWTP and abandoned iron ore mining pits. Urban impacts, such as an increase in impervious surfaces, stream channelization, and pollutants from road runoff also need to be considered as potential stressors in this watershed zone. Elevated turbidity and TSS concentrations are also known problems in several streams in this region. These potential stressors will be covered in detail in the analsysis and discussion of these impaired streams.

**Figure 334:** Photos of impaired stream streams in the Swan River - Hibbing Watershed zone. East Swan Creek (left and middle), Swan River (right)

## 5.22 East Swan Creek

East Swan Creek originates from Bryan Lake and several other small tributary streams within the city limits of Hibbing. Headwaters tributaries of this creek course through many disturbed areas, including high density housing developments, golf courses, junkyards and auto salvage lots, and commercial shopping centers. This area of Hibbing is growing rapidly and becoming increasingly urbanized. Downstream of Hibbing, East Swan Creek receives a continuous, year round discharge of treated wastewater from the Hibbing WWTP. Specific details on the discharge rates and water quality concerns associated with this discharge to the creek will be discussed further in the analysis of candidate causes for impairment.

The impaired reach of East Swan Creek extends from the WWTP outlet down to its confluence with the Swan River. The basis of the impairment listing was poor MIBI scores observed at biological monitoring station 09LS064 (Table 99). Two biological monitoring stations are located on the impaired reach (Table 99, Figure 335), with station 13LS015 being added in 2013 after the stream was already listed as an impaired water. The MIBI results at 13LS105 were considerably better, which provides some evidence that the stressor impacting 09LS064 is not present further downstream.

The most common macroinvertebrates observed at the impaired station (09LS064) include a variety of pollution- tolerant chironomid taxa (*Polypedilum, Cladotanytarsus, Tanytarsus*), aquatic worms (Oligochaeta), and several mayfly and caddisfly taxa that are often present in moderately degraded habitats (*Hydropsyche, Baetis*). Nearly 60% of the taxa observed at the impaired biological monitoring site are considered tolerant of pollution or disturbance (based on tolerance data developed for Minnesota). The impaired site scored poorly in the HBI, which may be an indication that the macroinvertebrate assemblage as a whole is tolerant of organic pollution.

**Table 99:** Summary of biological monitoring stations on East Swan Creek and available MIBI scores compared to impairment threshold criteria

| Macroinvertebrate Assessments | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Station | Drainage Area (mi2) | Gradient (%) | Stream Order (Strahler) | Invert IBI Class | Invert IBI Result *(visit year)* | Invert IBI Result *(visit year)* | Standard | IBI Lower Confidence Limit | IBI Upper Confidence Limit |
| **09LS064** | 7.06 | 0.28 | 2 | 8 | **21.89** *(2009)* | **27.14** *(2013)* | 32 | 19.6 | 44.40 |
| **13LS105** | 16.87 | 0.21 | 2 | 8 | **57.08** *(2013)* | | 32 | 19.6 | 44.40 |

The most common macroinvertebrates observed at the impaired station (09LS064) include a variety of pollution- tolerant chironomid taxa (Polypedilum, Cladotanytarsus, Tanytarsus), aquatic worms (Oligochaeta), and several mayfly and caddisfly taxa that are often present in moderately degraded habitats (Hydropsyche, Baetis). Nearly 60% of the taxa observed at the impaired biological monitoring site are considered tolerant of pollution or disturbance (based on tolerance data developed for Minnesota). The impaired site scored poorly in the HBI, which may be an indication that the macroinvertebrate assemblage as a whole is tolerant of organic pollution.

Water quality and physical habitat data were used to develop a list of candidate causes of impairment in East Swan Creek. The following candidate causes were identified for further evaluation in this Section;

1. **Elevated water temperatures**
2. **Nitrate Toxicity 3. Ammonia Toxicity**

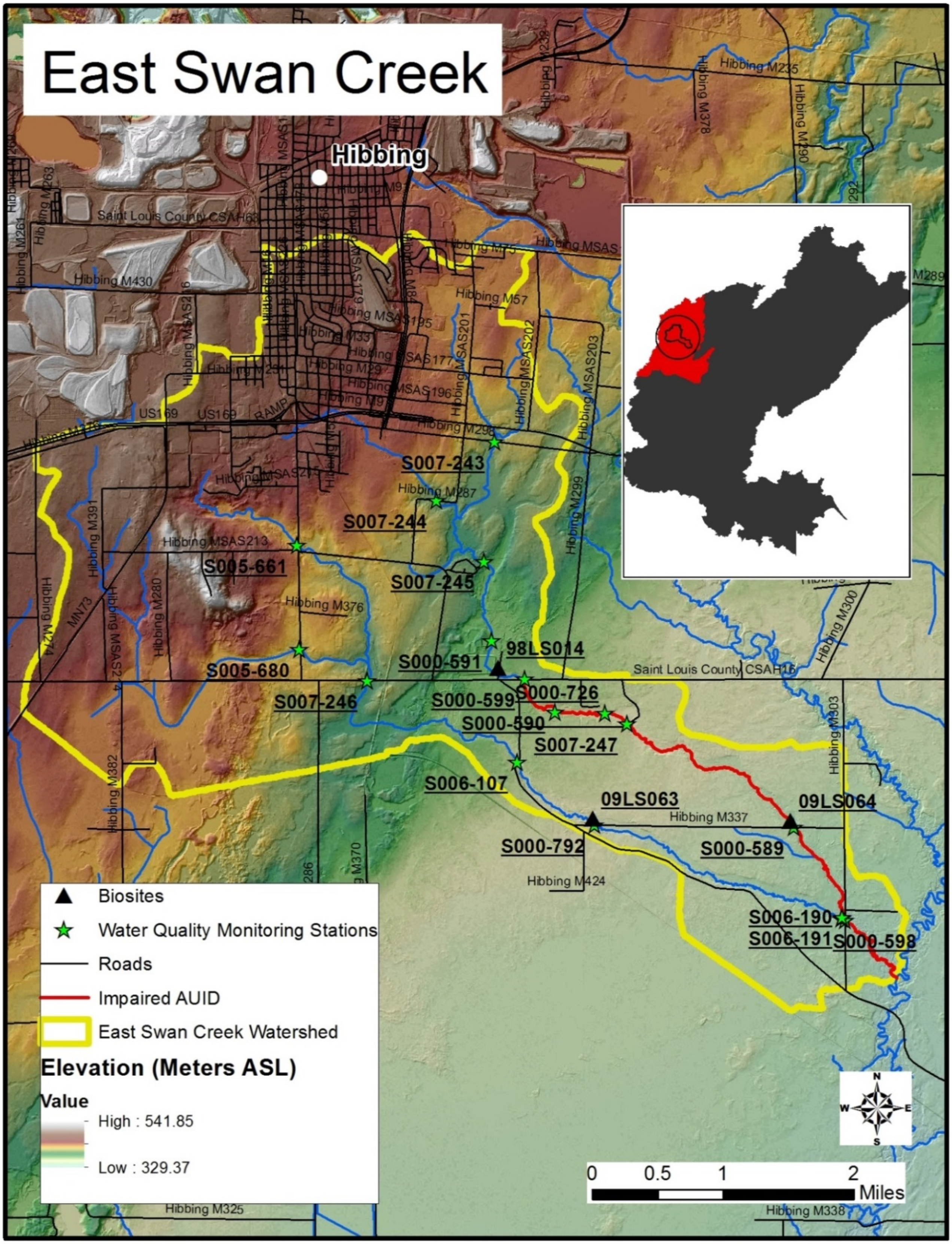


**Figure 335:** Map of East Swan Creek Watershed, monitoring stations, and impaired stream segments

## 5.22.1 Elevated Water Temperature

Continuous temperature data were collected at six stations on East Swan Creek (Figure 336). Although loggers were deployed in May and pulled from the stream in September or October, only data between June 1 and August 31 were analyzed for the purposes of this report, considering that these months are when stream temperatures are most likely to exceed the stress threshold for coldwater-sensitive macroinvertebrate species.

The MIBI impairment on East Swan Creek is based on 2009 sampling results from station 09LS064. Monitoring results show that the average daily temperatures at this station are relatively cold, and remarkably stable throughout the summer months (Figure 337). The stability of the thermal regime is related to the relatively large contribution of wastewater effluent (Hibbing WWTP) to summer baseflow in this stream. Only 2% of the average daily temperatures exceeded the stress threshold during the summers of 2009, 2012, and 2013 (6 days out of 272 recorded). Average stream temperatures were cold even during 2012, which was a much warmer-than-normal summer. By contrast, the average daily temperatures upstream of the wastewater plant were much warmer and showed more fluctuation (Figure 338). This suggests that the discharge of the wastewater plant has a stabilizing and cooling effect on the temperature of the stream.

We also compared water temperature data from East Swan Creek to measurements from a nearby, non-impaired biological monitoring site. Spider Muskrat Creek, located 25 miles to the south, is a coldwater stream with very good to excellent MIBI results. As can be seen from Figure 339, East Swan Creek's temperature regime is actually cooler than that of Spider Muskrat Creek. About 18% of the daily average temperatures in Spider Muskrat Creek exceeded the stress threshold (17 days out of 92).

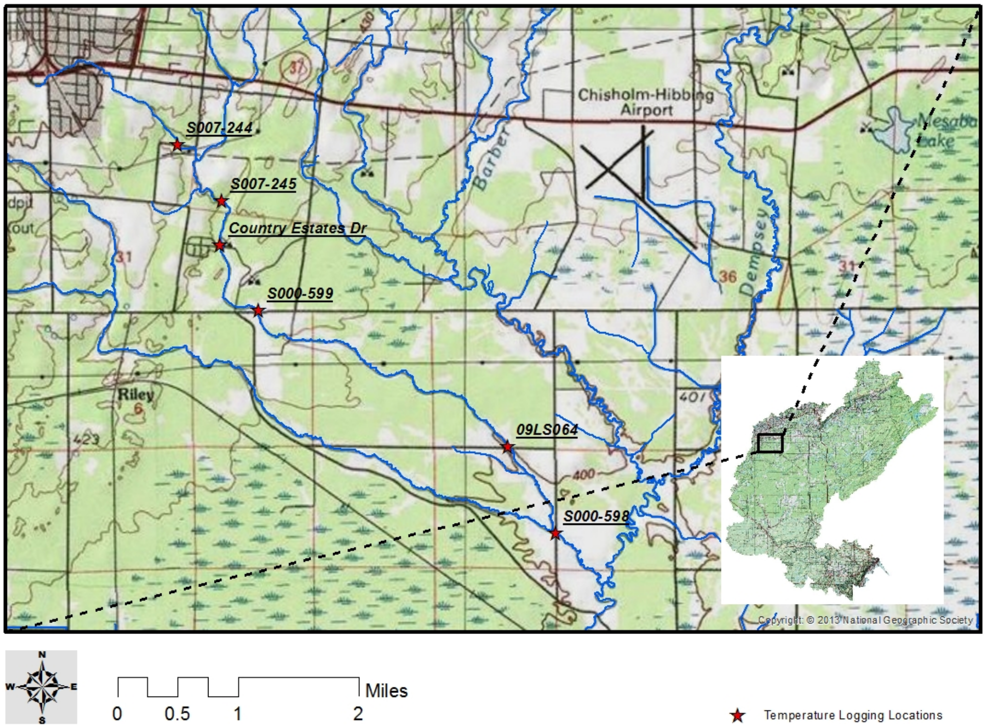


**Figure 336:** HOBO temperature logger locations on East Swan Creek

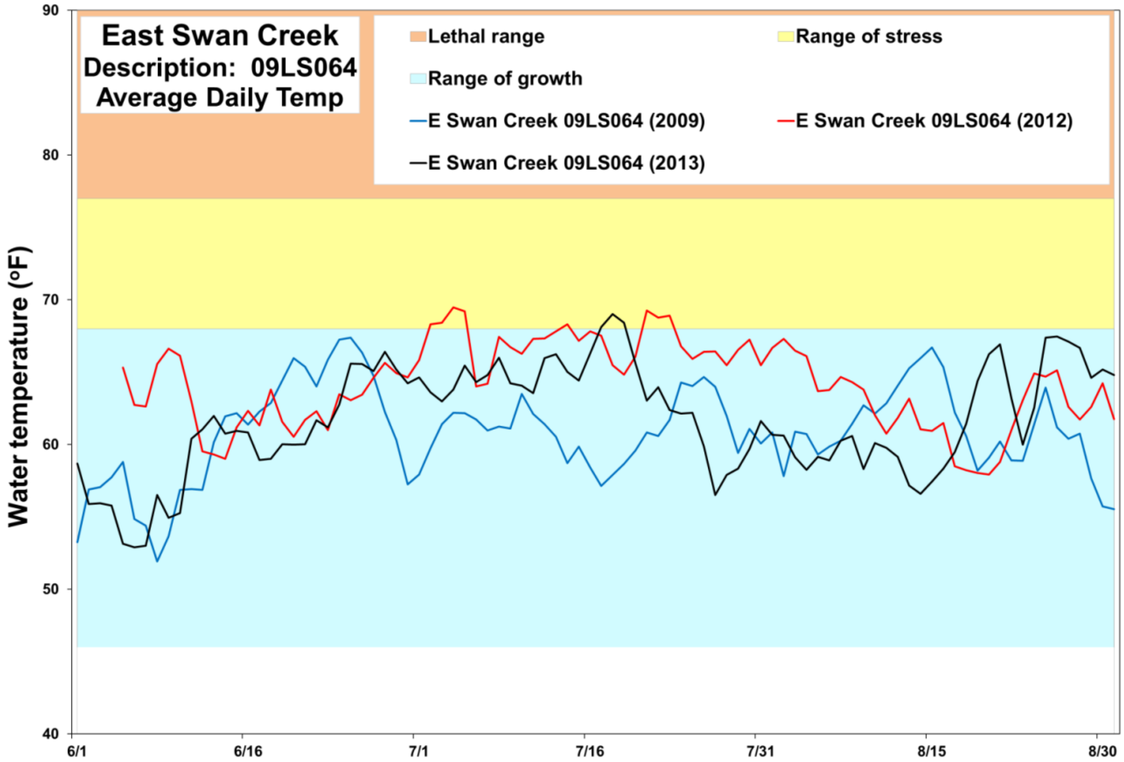


**Figure 337:** Average daily temperatures for 2009, 2012, and 2013 at 09LS064, the impaired biological monitoring station on East Swan Creek

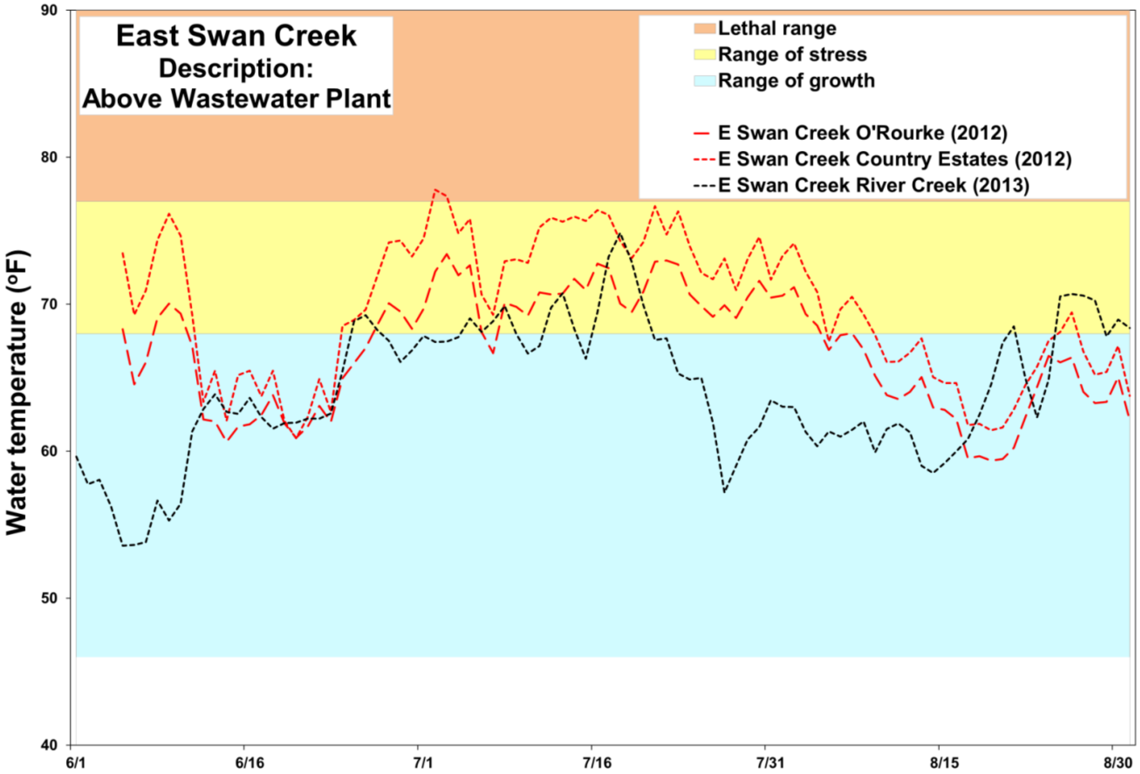


**Figure 338:** Average daily temperatures in East Swan Creek upstream of the WWTP

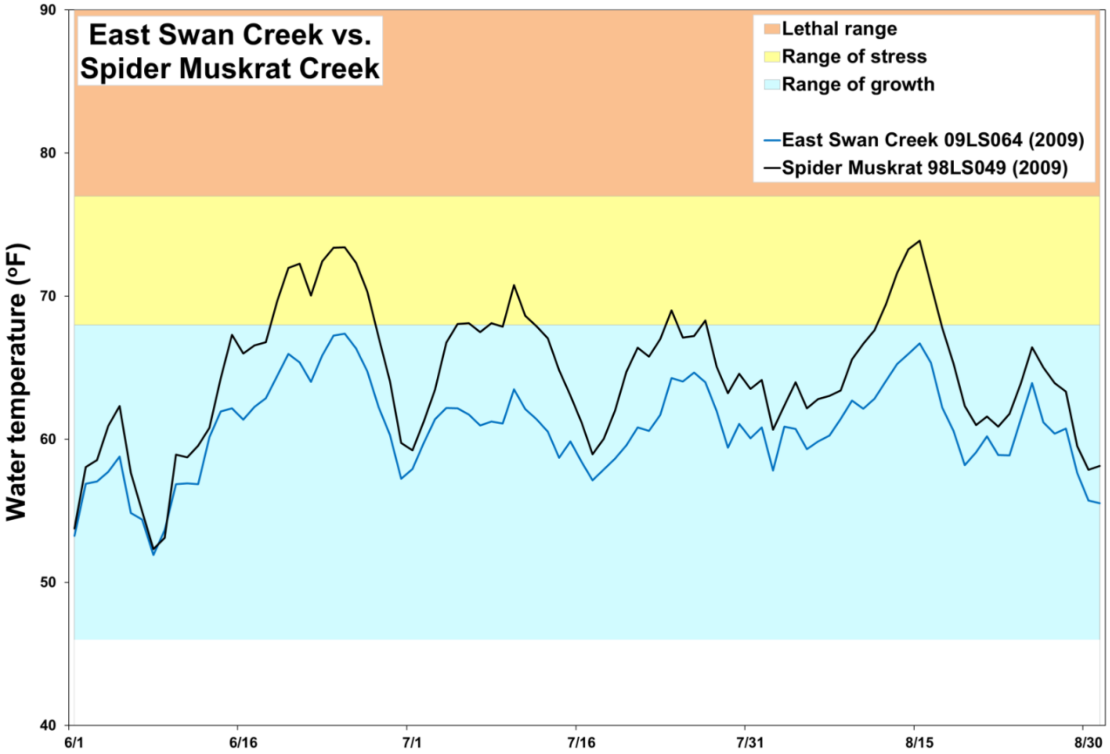


**Figure 339:** East Swan Creek average daily temperatures in 2009, compared to non-impaired Spider Muskrat Creek during the same year

**Summary: Is elevated water temperature a stressor in East Swan Creek?**

The thermal regime of impaired reach of East Swan Creek (downstream of the Hibbing WWTP) is suitable for supporting coldwater fish and macroinvertebrate taxa. Upstream of the WWTP, stream temperatures fluctuate more and are marginal to poor for supporting coldwater taxa., Based on this analysis, it is clear that other factors are limiting the macroinvertebrate community within the impaired reach of East Swan Creek, and stream temperature can be eliminated as a cause of impairment.

### 5.22.2 Specific conductivity

Elevated specific conductivity was identified as a candidate cause for impairment in East Swan Creek based on water quality data and concerns about the impacts of Hibbing WWTP on aquatic communities downstream of the discharge. Available point measurements of specific conductance are displayed in Figure 340 by sampling month and monitoring site. The majority of results are within the range of 400-800 µS/cm, but data from several monitoring locations frequently exceed 800 µS/cm. Three of these monitoring stations have large data sets for conductivity data associated with them, S000-599 (co-located with biological monitoring station 98LS014), S000-589 (co-located with biological monitoring station 09LS064), and S006-191. All three of these monitoring stations are located downstream of the Hibbing WWTP discharge. The two stations closest to the WWTP discharge (S000-589 and S000-599) tend to have the highest specific conductivity levels, as these two stations were the only stations to record readings over 1,000 µS/cm.

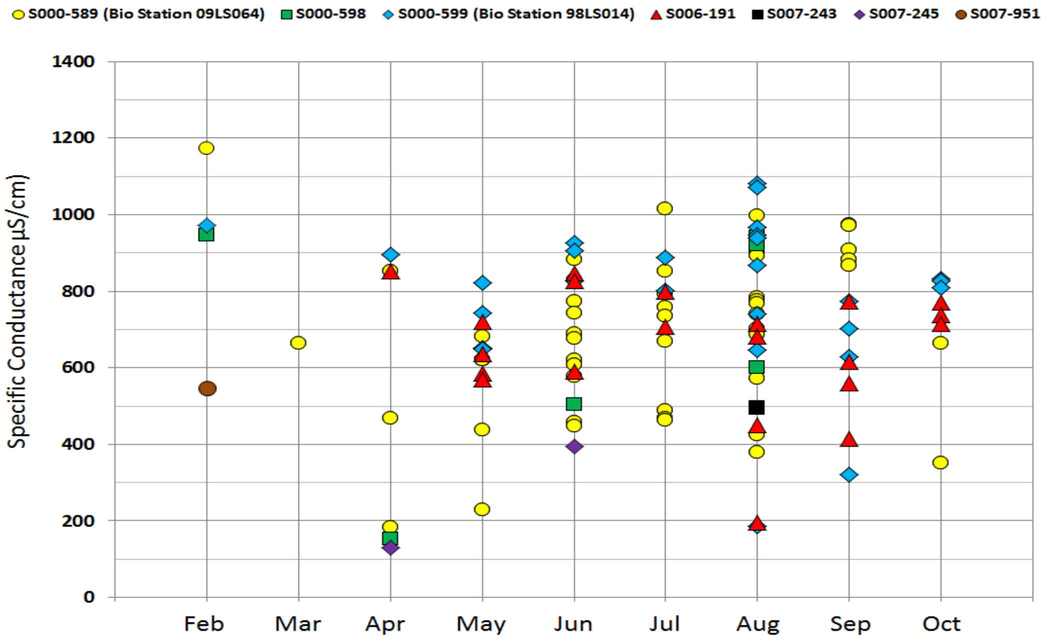


**Figure 340:** Point measurements of specific conductivity collected from East Swan Creek by monitorng station and calendar month

Few measurements of specific conductivity are available for sites upstream of the WWTP, but the limited data available show a spike in conductivity below the discharge point. Results from winter baseflow monitoring show an increase from 548 µS/cm to 973 µS/cm downstream of the discharge (Figure 341). During this monitoring event, specific conductivity continued to increase through the next monitoring station downstream (an additional three miles of stream length) before decreasing at the monitoring station near the outlet. The cause of the increase in specific conductance further downstream is not known.

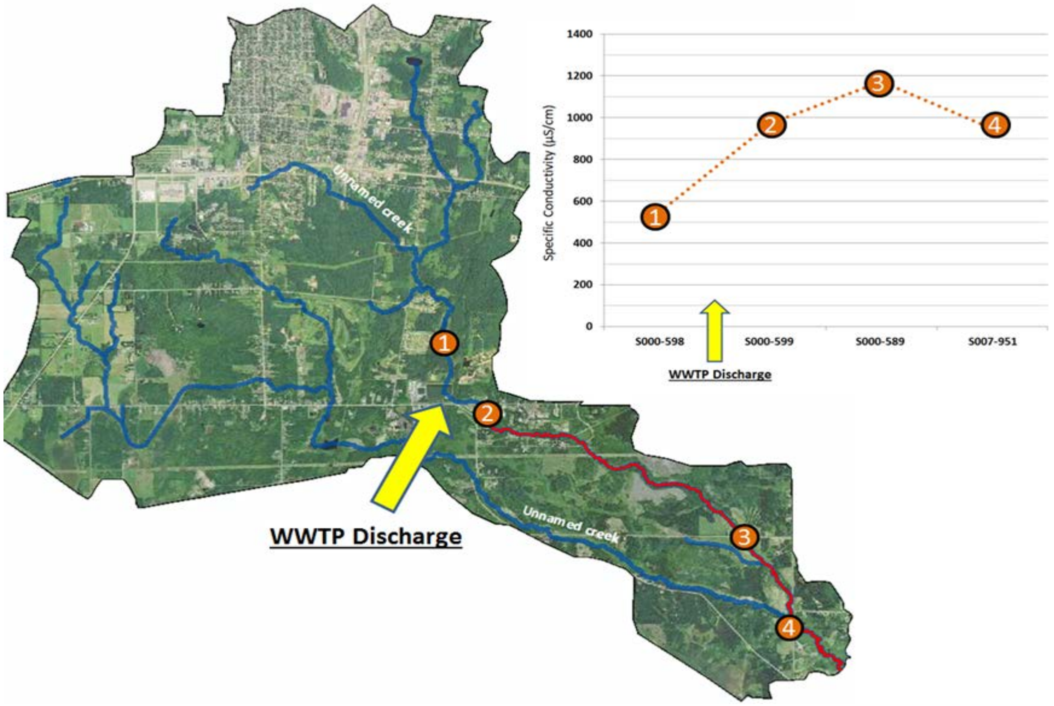


**Figure 341:** Longitudinal measurments of specific conductivity from East Swan Creek collected in February 2014

### Sources and Pathways of Specific conductivity

The most common salt ions found in surface water that influence specific conductivity levels include positively charged (cations) $Ca^{2+}$, $Mg^{2+}$, $Na^{+}$, $K^{+}$, and negatively charged (anions) $HCO_3^{-}$, $CO_3^{2-}$, $SO_4^{2-}$, and $Cl^{-}$ (CADDIS, 2012). Samples were collected from East Swan Creek station S000-589 (co-located with biological monitoring station 09LS064) during baseflow, rain event flow, and snowmelt conditions and analyzed for major cations and anions. Figure 342 shows the concentrations of various cations and anions observed during the three geochemistry sampling events. The baseflow sampling results show elevated concentrations chloride (>100 mg/L), which can be linked back to the WWTP discharge upstream. Stormwater and snowmelt runoff from urban areas in and around the city of Hibbing are another likely source of chloride in this watershed, but not during low flow conditions.

Sodium, sulfate, and calcium were also observed in fairly high concentrations during baseflow in East Swan Creek, and are considered sources of elevated specific conductivity. Sampling results from station S000-589 are closely related to effluent monitoring results for these ionic compounds, which are shown as average monthly maximums in Figure 343. Chloride, sulfate, sodium, and calcium are observed in WWTP effluent in concentrations that are similar to those observed during baseflow sampling at S000-589. These results are not surprising considering that WWTP effluent is the source of most of the water in East Swan Creek during summer and fall low flow periods.

Per NPDES permit requirements, the Hibbing WWTP has been monitored specific conductivity levels of effluent discharged to East Swan Creek since September of 2012. The results, reported in monthly maximum values, indicate that for most of the year, specific conductivity levels of effluent leaving the plant exceed 1,000 µS/cm. The maximum value recorded to date is 1,120 µS/cm in February of 2014. Conductivity levels are elevated during periods of the year where precipitation and runoff are at a minimum, which results in a discharge of higher specific conductivity entering East Swan Creek when the stream is at low flow stage.

Several other NPDES permits have been issued within this watershed, but none of these are likely related to this candidate cause for impairment. The majority of the other point sources in the watershed are stormwater permits designed to prevent runoff from gravel and aggregate production facilities.

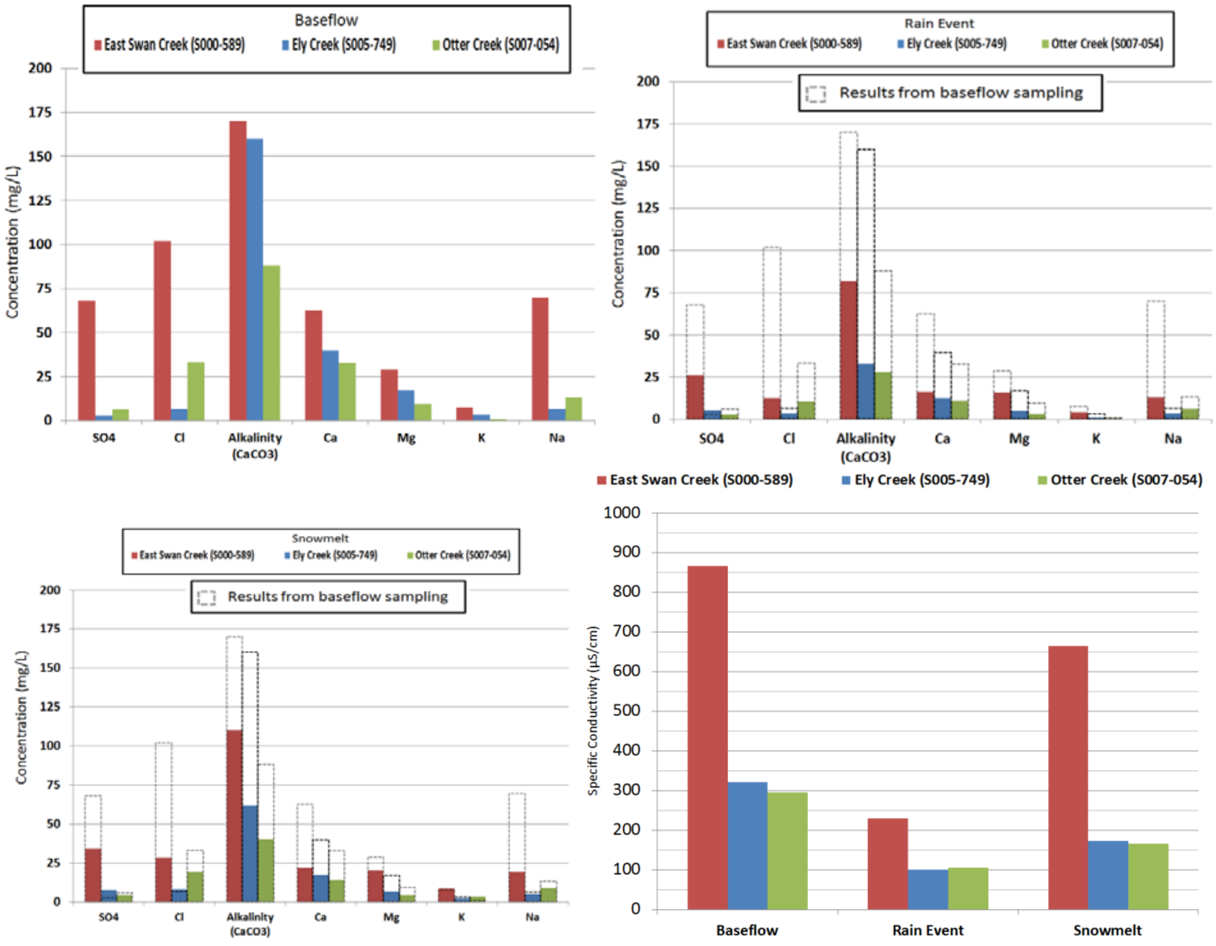


**Figure 342:** Comparison of geochemistry sampling results for major cations and anions in East Swan Creek, Ely Creek, and Otter Creek under various flow conditions; baseflow (upper left), rain event (upper right), snowmelt (lower left). Specific conductivity comparisons for East Swan Creek, Ely Creek, and Otter Creek during the same sampling events (lower right).

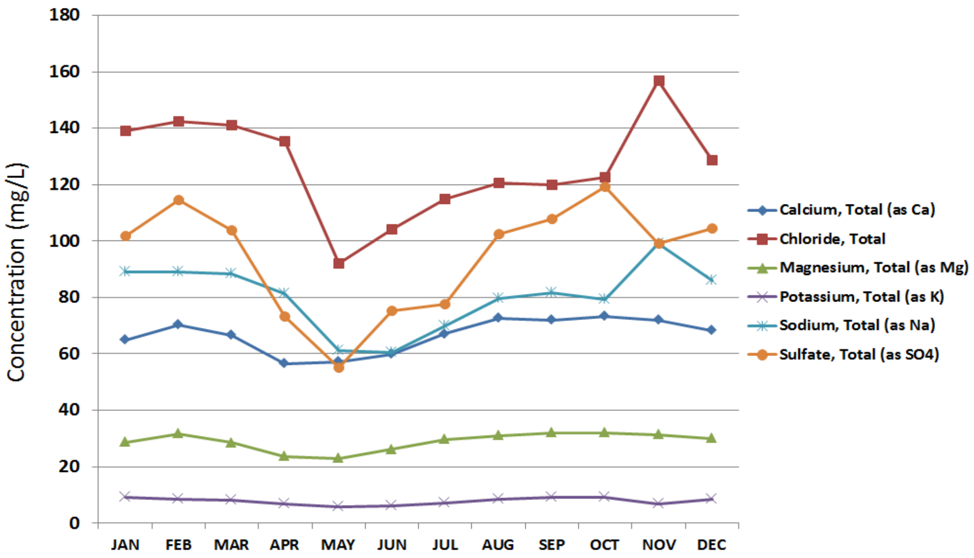


**Figure 343:** Average calendar month maximum concentrations of various parameters measured in the effluent from Hibbing WWTP

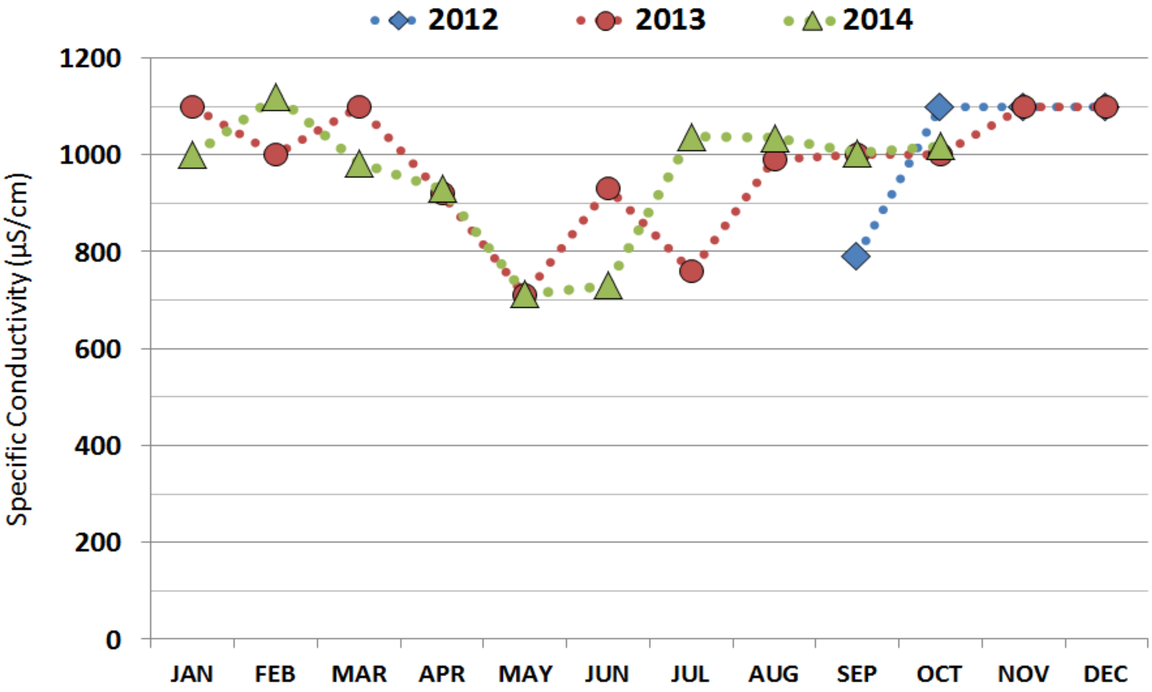


**Figure 344:** Monthly maximum specific conductivity values of effluent discharged from Hibbing WWTP to East Swan Creek

## Biological Response to Specific conductivity

The effects of elevated conductivity on aquatic life were evaluated using data from Minnesota streams and scientific literature. A summary of this analysis is presented in Section 3.1.5. Based on this work, several biological metrics were selected to evaluate specific conductivity as a stressor in Elbow Creek (Table 100).

**Table 100:** Summary of biological metrics and literature used to evaluate elevated specific conductivity as a stressor

| Metric | Response to Increased Specific conductivity / Conductivity | Source |
|---|---|---|
| **EPT Richness** | Decrease | Roy et al (2003); Echols et al (2009); Johnson et al (2013) |
| **Overall Taxa Richness** | Decrease | Johnson et al (2013) |
| **Ephemeroptera Richness** | Decrease | Pond (2004); Hassel et al (2006) |
| **Fish and Macroinvertebrate Tolerance Indicator Values (TIV)** | Increase | MBDI (Yoder and Rankin, 2012) |

### EPT Richness

EPT richness at East Swan Creek monitoring stations ranged from 6-12 taxa (Figure 345). Station 98LS014, located in the reach where WWTP effluent enters the creek, had the lowest EPT taxa count. Several miles downstream of the WWTP outfall at station 09LS064, EPT taxa richness was somewhat variable between two sampling events. In 2009, a total of 9 EPT taxa were observed in the sample from this station, compared to the 12 EPT taxa that were observed in 2013. At station 13LS105, located near the mouth of East Swan Creek, and the site furthest removed from the source of elevated specific conductivity in the watershed, 10 EPT taxa were observed during the only sampling visit in 2013. EPT richness generally increases with distance from the WWTP outfall, although the variability in results from station 09LS064 does not perfectly adhere to this trend. Overall, EPT taxa richness in East Swan Creek is moderately to severely limited compared to high quality stations in the SLRW.

### Overall Macroinvertebrate Taxa Richness

Macroinvertebrate taxa richness in East Swan Creek ranged from a low of 35 taxa (98LS014) to a high of 44 taxa (13LS105). Taxa richness values increase in a downstream direction and with distance from the WWTP outfall. Compared to high quality stations in the SLRW, taxa richness in East Swan Creek is low, particularly at stations 98LS014 and 09LS064. These observations provide evidence in support of specific conductivity as a stressor, but this particularly symptom in the biota may also be responsive to other stressors that may be present (e.g. nitrate).

### Ephemeroptera Taxa Richness

Ephemeroptera (mayfly) richness in East Swan Creek was low, ranging from 2-4 taxa over four monitoring visits to three individual stations. Stations nearest to the WWTP outfall had fewer Ephemeroptera present compared to sites downstream, but results were comparable throughout all of the sites visited.

### Specific Conductivity Tolerance Indicator Values (TIV)

Specific conductivity TIVs for East Swan Creek are at or above the 75th percentile results from unimpaired, high quality streams in the SLRW (Figure 345). These results suggest that the macroinvertebrate community in East Swan Creek is more tolerant of elevated specific conductivity compared to less impacted streams. The highest TIV value (most tolerant) was observed just downstream of the WWTP at station 98LS014. The second highest TIV value was observed at station 09LS063, which is located on a tributary to East Swan Creek. This tributary receives no wastewater, and monitoring results show a conductivity range of 100 – 400 µs/cm in this stream. This observation shows some inconsistency in the TIV results, as other stations on the main stem of East Swan Creek had much higher conductivity readings, yet had TIV values that showed lower tolerance.

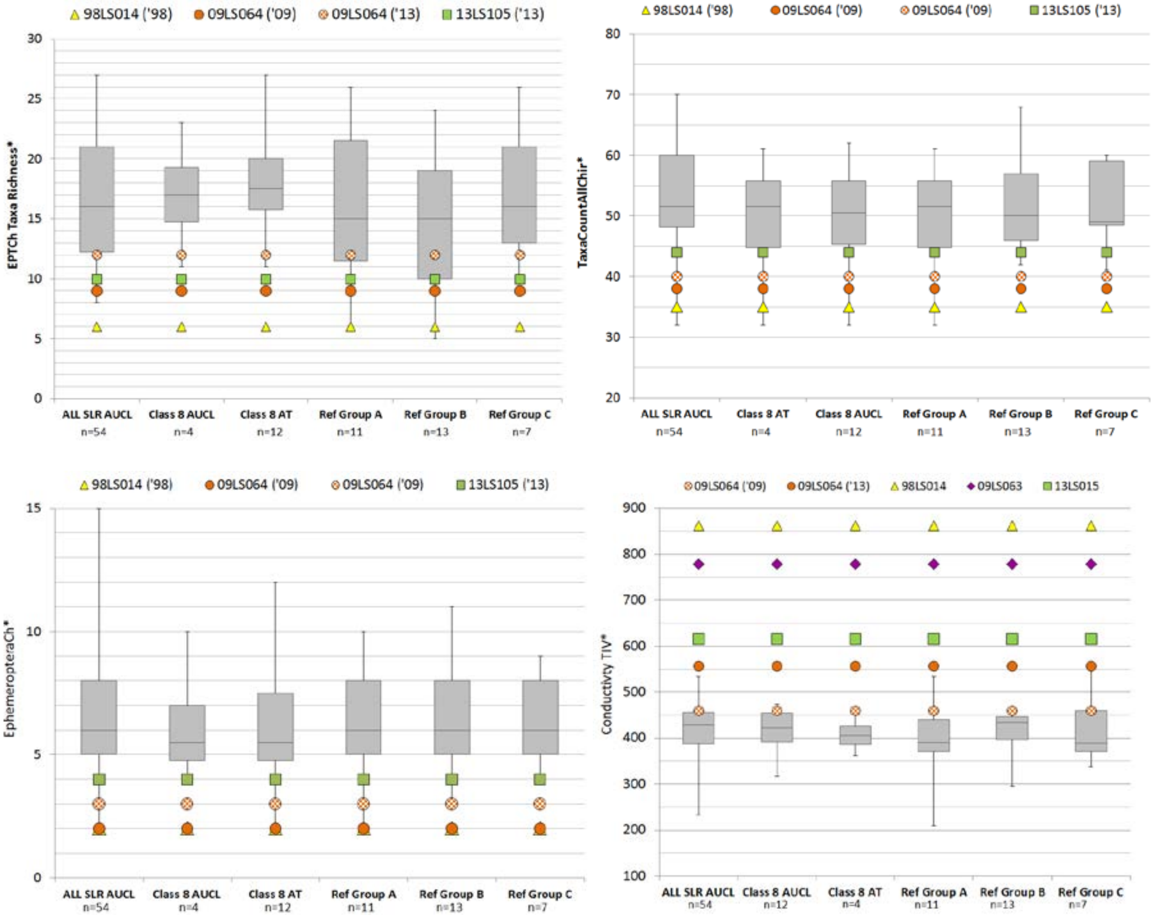


**Figure 345:** Comparison of results for several macroinvertebrate metrics between Kinney Creek stations and non-impaired SLRW reference stations: overall macroinvertebrate taxa richness (upper left); EPT Taxa Richness (upper right); Ephemeroptera taxa richness (lower left); macroinvertebrate community DO tolerance indicator values (TIV) (lower right). ** See Section 4 for explanation of TIVs **AUCL** = Above Upper Confidence Limit of FIBI threshold **AT** = Above FIBI Threshold*

### Summary: Is elevated specific conductivity a stressor in East Swan Creek?

Effluent from the Hibbing WWTP provides much of the baseflow in East Swan Creek, and as a result, specific conductivity levels can be quite high (500 – 1,200 µS/cm) for long durations. The macroinvertebrate data show some common symptoms of stress due to high ionic strength – lack of taxa richness, low EPT taxa richness, lack of mayfly richness, and higher TIV values for specific conductivity.

Specific conductivity should remain a potential stressor to aquatic life in East Swan Creek. It is difficult to eliminate confounding stressors to a point where this stressor can be diagnosed as a cause of impairment with high confidence. Development of state water quality criteria that is based on a larger data set would be a helpful tool for further evaluating this stressor.

## 5.22.3 Nitrate and Ammonia Toxicity

Elevated nitrogen levels have been documented in East Swan Creek as far back as 1979. Between the years 1979-1981, monitoring data show elevated ammonia nitrogen (ammonia) concentrations (up to 13 mg/L) downstream of the Hibbing WWTP. However, nitrate concentrations at this time were relatively low (avg. less than 2 mg/L). More recent data from 2009-2014 shows a reversal of this trend, with higher nitrate concentrations (up to 18 mg/L) and very low ammonia concentrations averaging less than 1 mg/L (Figure 346).

Prior to May 31st, 2008 the city of Hibbing operated two separate WWTPs (North & South). The city of Hibbing wanted to join and modernize the facilities to reduce treatments costs and avoid redundant upkeep costs. A non-degradation review approved by the MPCA allowed the north WWTP to be shut down and the flow rate from the south WWTP expanded to 4.5 million gallons per day (MGD). The expansion of the south WWTP to 4.5 MGD also introduced stringent carbonaceous biological oxygen demand (cBOD5) and ammonia limits in order to be protective of the receiving water of East Swan Creek. The Hibbing WWTP was assumed to be the dominant source of nitrogen to East Swan Creek.

In order to meet the cBOD5 and ammonia limits, a nitrifying WWTP design was created for the expanded south Hibbing WWTP. A nitrifying WWTP uses microbial population dynamics to convert ammonia to nitrate. The decrease in ammonia concentration and the increase in nitrate concentrations in East Swan Creek from 1982 to present can be explained by the Hibbing WWTP fully nitrifying its effluent; effectively converting nitrogen species from ammonia to nitrate.

Historically, ammonia nitrogen may have been a stressor to aquatic life in East Swan Creek. Since 2008, ammonia levels in the creek have decreased substantially and are no longer considered a threat to fish and macroinvertebrate populations. Therefore, only the nitrate form of nitrogen will be evaluated as a candidate cause of the impaired macroinvertebrate community.

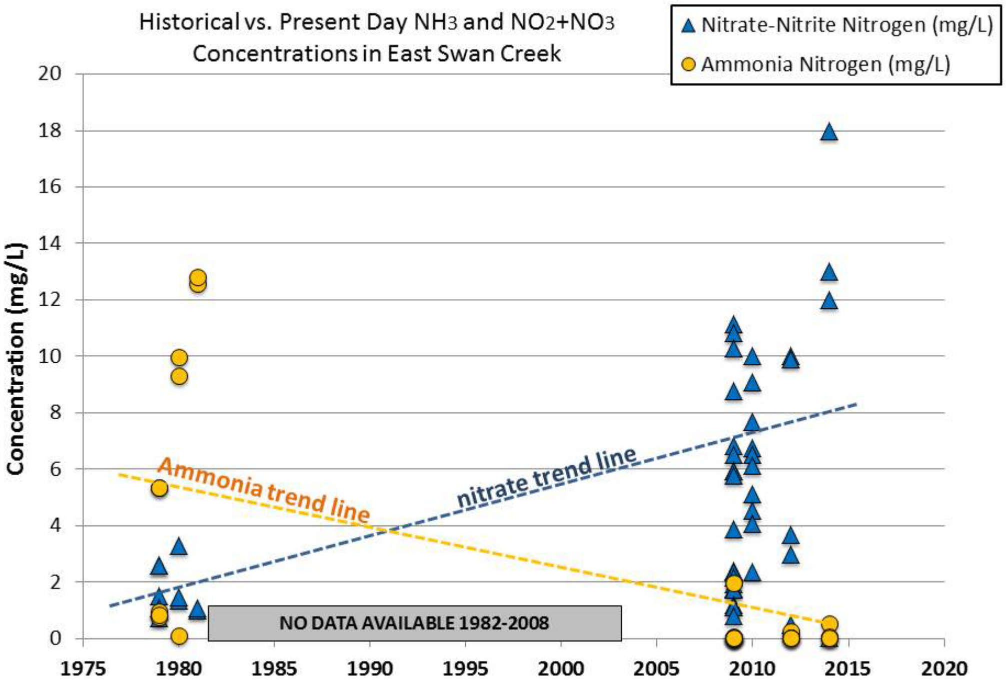


**Figure 346:** Comparison of historic and contemporary ammonia and nitrate nitrogen data collected from East Swan Creek

## Nitrate Toxicity

Nitrate-Nitrite nitrogen (nitrate) sampling results are available for six stations along East Swan Creek. The data were collected primarily between the years of 1979 -1981 and more recently, 2009 – 2014. Nitrate concentrations in East Swan Creek peak during baseflow (low flow) conditions during the late summer, early fall, and mid-winter seasons (Figure 348). Levels as high as 18 mg/L have been recorded (station S000-589, February 2014) and concentrations in the range of 8-12 mg/L are common during summer and fall baseflow periods. Historic data (1979-1981) generally show lower nitrate concentrations, which is most likely due to the different effluent treatment methods that were used at the Hibbing WWTP during this period of time.

In February of 2014, several samples were collected in a longitudinal pattern along East Swan Creek to identify potential sources and pathways of nitrate within the watershed. Four stations were sampled in the lower half of the watershed where elevated nitrate concentrations had been previously observed. Two of these stations, S007-951 and S007-599 were specifically chosen to bracket the location where effluent from the Hibbing WWTP enters East Swan Creek (see map in Figure 349). The results of the longitudinal sampling clearly show that effluent from the WWTP is the primary source of nitrate entering East Swan Creek. Results from the stations above and below the WWTP outfall show an increase from < 0.05 mg/L above the discharge point to 13 mg/L immediately downstream of the discharge. Approximately 2.5 miles downstream from the WWTP discharge at station S000-589, nitrate concentrations were even higher (18 mg/L). Additional sources of nitrate between the WWTP and this station are unknown. The increase in nitrate concentrations between these two stations may be the result of in-stream nitrification processes. At station S000-598, 3.3 miles downstream of the WWTP discharge, nitrate concentrations decreased slightly to 12 mg/L.

**Figure 347:** Signs of untreated wastewater in East Swan Creek observed at CR 16 (Town Line Rd.) in August of 2012

During extreme low flow conditions, effluent discharge from the WWTP accounts for a large portion of the flow in East Swan Creek. Upstream of the WWTP, flows are generally <1 cubic feet per second (cfs) during baseflow periods, which was the case during the February 2014 longitudinal sampling event and other visits to East Swan Creek in late summer or early fall months. Based on permitting reports from the Hibbing WWTP, average discharge rates to East Swan Creek are in the range of 4-5 cfs for most of the year. In wet periods, such as spring snowmelt or large rain events, discharge to the creek can increase to 15-20 cfs. During baseflow conditions, it is likely that wastewater effluent from the Hibbing WWTP accounts for more than 80% of the flow in lower East Swan Creek where the biological monitoring stations are located.

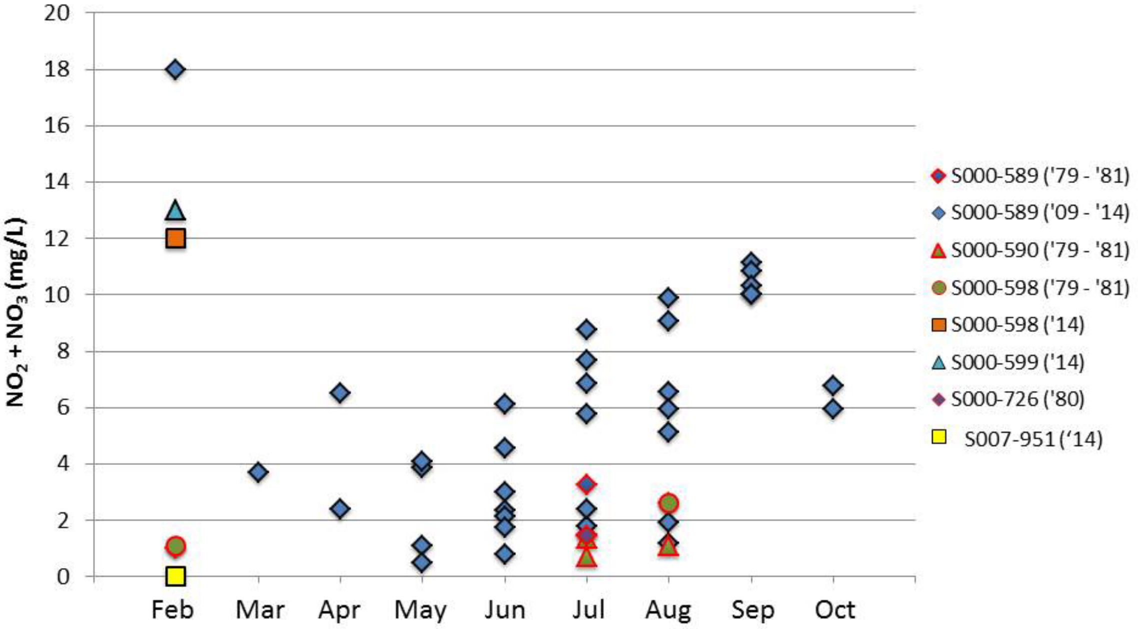


**Figure 348:** Nitrate results from East Swan Creek sorted by monitoring station and calendar month

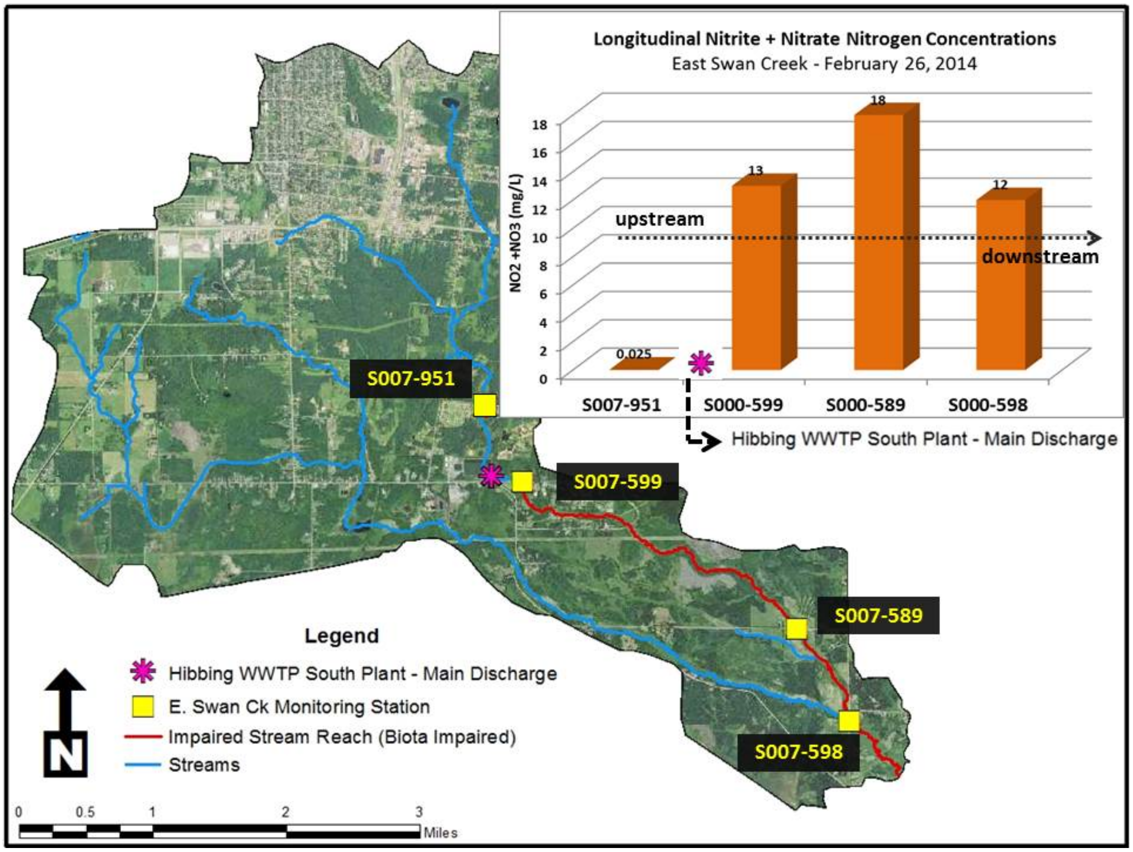


**Figure 349:** Nitrate concentrations observed in East Swan Creek during a February 2014 longitudinal sampling event

### Biological Effects of Elevated Nitrate in East Swan Creek

Macroinvertebrate data from four biological monitoring stations are available to evaluate the potential impact of elevated nitrate concentrations in East Swan Creek. The three stations on the main stem of East Swan Creek are all located downstream of the WWTP influence. In addition to the continuous discharge from the WWTP, there are several wastewater ponds that may be contributing legacy impacts from several overflows that have occurred at this WWTP in the past. Station 09LS063 is located on a tributary to East Swan Creek and is not impacted by WWTP discharge. For more information on these monitoring sites, including a detailed map of site locations, refer to the map in Figure 355.

The NTIV have been developed by the MPCA biological monitoring unit for most macroinvertebrate taxa that have been observed in Minnesota streams. For more information on TIV development, refer to Section 4. Individual and community based NTIV values will be used in this Section to evaluate the degree to of nitrate tolerance exhibited by macroinvertebrate taxa in East Swan Creek.

### NTIV Community Index Scores

Community-based macroinvertebrate NTIV values for East Swan Creek monitoring stations are shown in Figure 350 along with quartile box-plot values from statewide, northern coldwater MIBI class, Lake Superior (LS) basin, and SLRW monitoring stations. In East Swan Creek, the two stations closest to the WWTP produced the highest NTIV scores. A higher NTIV score means that nitrate-tolerant macroinvertebrate taxa were more prevalent at those monitoring locations. Station 09LS064 clearly registered the highest NTIV score of all sites sampled on East Swan Creek, and the score seems to be consistently high based on the close agreement between 2009 and 2013 sampling events. NTI scores at 09LS064 were well above the 75th percentile values for all of the data sets used for comparative purposes. Chemistry results also support 09LS064 as a station of concern for nitrate toxicity, as it recorded the highest nitrate concentration (18 mg/L) during the longitudinal sampling event conducted during the winter of 2014.

The NTIV result from station 98LS014 is also elevated compared to values from stations included in the statewide, Lake Superior basin, and SLRW data sets. Nitrate concentrations were 10 mg/L during the visit when fish data were collected (July 1998). As mentioned earlier, station 98LS014 is located just downstream of the main discharge point for the Hibbing WWTP. However, nitrate concentrations during the 2012 longitudinal sampling event were actually higher at station 09LS064 (2.5 miles downstream). Monitoring results show that the WWTP is the major source of nitrogen in East Swan Creek, but nitrate concentrations and the impact to aquatic life may reach a peak a short distance (1-3 miles) downstream of the discharge point.

Nitrate tolerant macroinvertebrate taxa accounted for a large percentage of the overall population at stations 09LS064, and to a somewhat lesser extent, at 98LS014 as well. In 2009, over 71% of the macroinvertebrate community at 09LS064 consisted of taxa that can be considered tolerant of elevated nitrate concentrations. In 2013, the percentage of nitrate tolerant individuals at this station decreased slightly to 60%. At 98LS014 (sampled in 1998), nitrate tolerant macroinvertebrate individuals accounted for 59% of the total community sampled. Each of the results at both stations is above the 75th percentile value observed at other stations in the northern coldwater MIBI class and other regional reference sites (Figure 351). Based on biological sampling results, it can be concluded that the reach of East Swan Creek from the WWTP downstream 2-3 miles supports a macroinvertebrate community with a relatively high percentage of nitrate tolerant individuals compared to other monitoring sites in Northeastern Minnesota.

Data from the East Swan Creek tributary station (09LS063) and Swinnerton Road station (13LS105) had lower percentages of nitrate tolerant macroinvertebrates. The tributary stream, with no upstream point source discharge and very low nitrate concentrations (n=22, avg = <0.05 mg/L, max = 0.13 mg/L), supported a macroinvertebrate community in which 46% of the individuals sampled can be considered nitrate tolerant. Station 13LS105, located on the main stem of East Swan Creek approximately 3.3 miles downstream of the WWTP, had 49% nitrate tolerant individuals in a 2013 sampling event. While the differences between these two sites and others closer to the WWTP do not seem significant, they do support a gradient of biological effect that decreases with distance from the WWTP outfall.

**Summary: Is nitrate toxicity a stressor in East Swan Creek?**

Available water chemistry and biological data provide adequate evidence to diagnose nitrate toxicity as a cause of impairment in East Swan Creek. Nitrate concentrations as high as 18 mg/L were observed at biological monitoring stations downstream of the Hibbing WWTP. In contrast, stations upstream of the WWTP outfall show nitrate concentrations less than 0.05 mg/L. A clear stressor gradient was observed within the impaired reach showing a higher magnitude of impact closer to the source of nitrate and decreasing impacts in a downstream direction. Biological indicators of nitrate stress observed in the impaired reach include a high percentage of nitrate tolerant organisms and community TIVs that show a high level of tolerance to elevated nitrate levels.

The MPCA does not currently have a water quality standard for nitrate that is based on aquatic life. The development of a TMDL for this parameter will be deferred until there is a water quality standard to base pollutant loading allocations.

### 5.22.4 Unnamed Creek (East Swan Creek):

**Table 101:** Summary of SID results for the impaired reach of East Swan Creek

| Candidate Cause | Result |
|---|---|
| **Elevated Water Temperatures** | X |
| **Elevated Specific conductivity** | ○ |
| **Nitrate Toxicity** | ● |
| **Ammonia Toxicity** | X |
| Key: ● **= confirmed stressor** ○ **= Potential Stressor X = eliminated candidate cause** | |

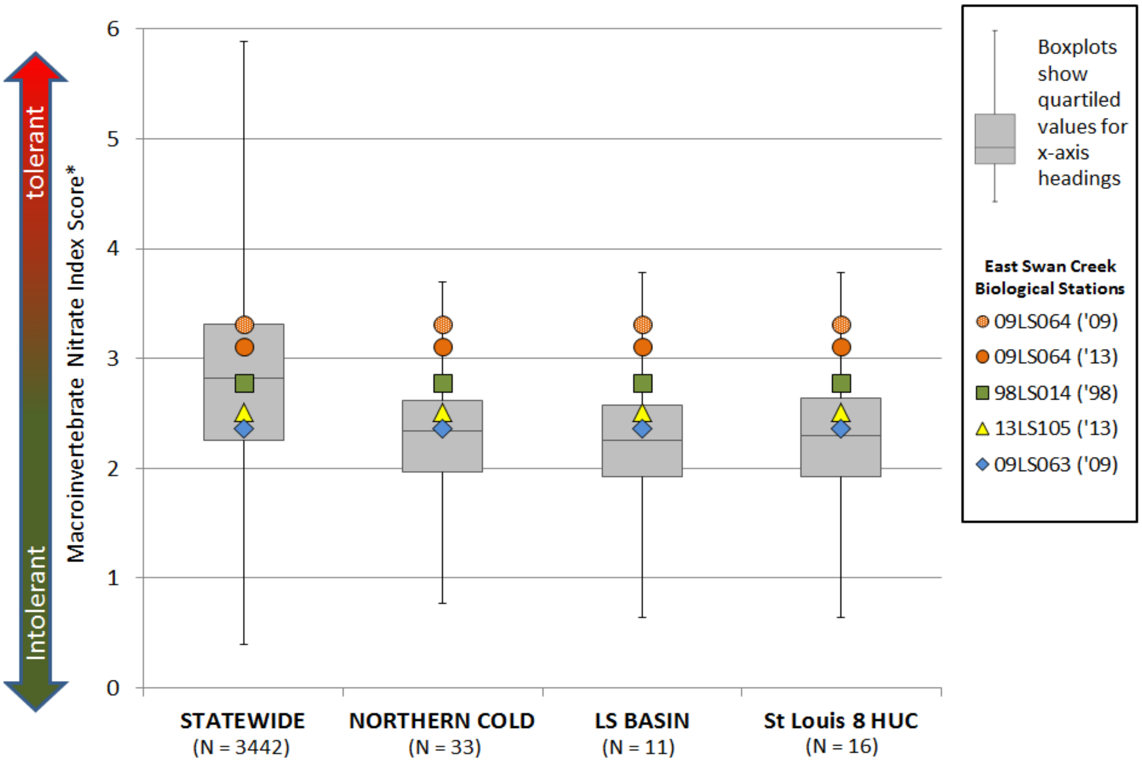


**Figure 350:** Macroinvertebrate community nitrate tolerance indicator values (NTIV) for East Swan Creek monitoring stations compared to results from statewide and regional sites.

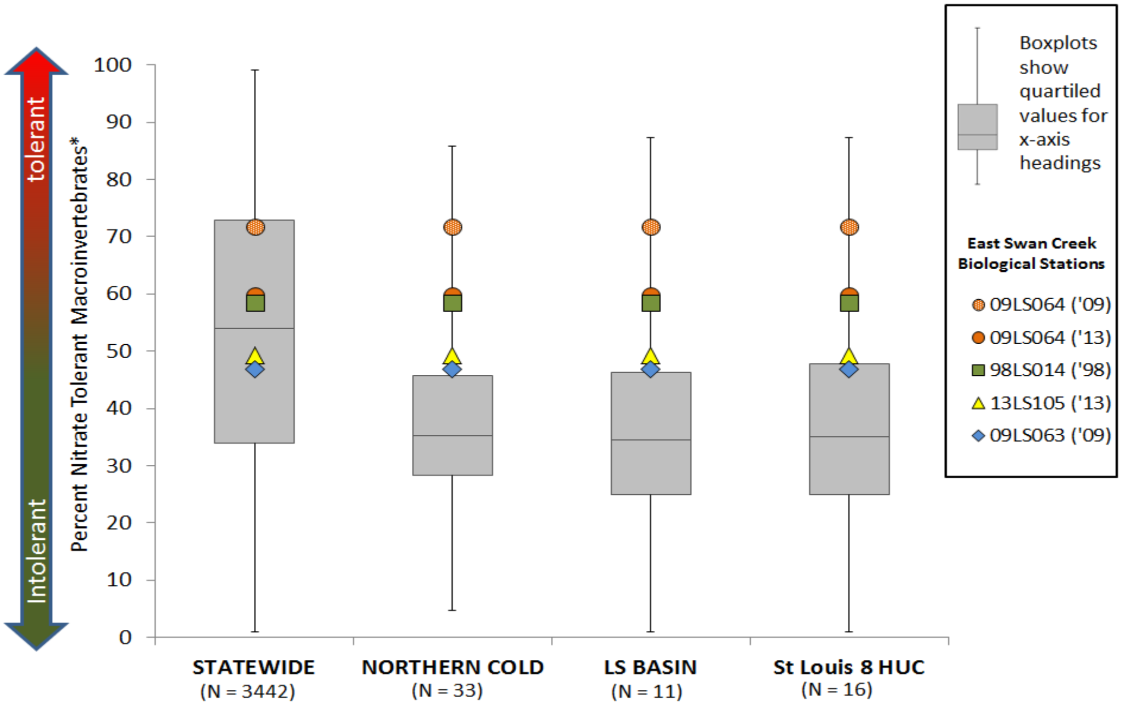


**Figure 351:** Percent nitrate tolerant macroinvertebrates observed at East Swan Creek monitoring stations compared to results from statewide and regional sites.

## 5.23 Swan River

The Swan River is formed at the confluence of two major tributaries; the West Swan River and the East Swan River. The iron mines near Hibbing, Chisholm and Buhl constitute the headwaters of the greater Swan River Watershed. The lower half of the watershed is within the Glacial Lake Upham lake bed. In this region the stream types are mostly low gradient Rosgen C and E type channels. Many stream reaches in this specific area have incised into the lacustrine sediments through which they flow. The impaired reach of the Swan River is only about 5 miles long, and consists mostly of a very low-gradient (<0.01%), somewhat sinuous Rosgen C-type channel. The last half mile of the river steepens (0.3% slope) before entering the St. Louis River.

The impaired reach of the Swan River extends from the confluence of the East Swan River and West Swan River down to the St. Louis River. Currently, this reach is listed as impaired for low FIBI scores and elevated turbidity concentrations. This reach was considered a designated trout stream until recently, when conversations between the DNR and the MPCA resulted in a use-class change to a warmwater designation.

The FIBI results from the two monitoring stations were below the impairment threshold and its lower confidence limit. A total of only seven fish were collected during the sampling of 09LS061, which occurred in the summer of 2009. Smallmouth Bass were the only species represented with more than one individual, and only two individuals were collected in the sample. Other species present, with only one individual sampled, included Creek Chub, Northern Redbelly Dace, Common Shiner, Shorthead Redhorse, Trout-Perch, and White Sucker. The list of species observed at this site includes several that are fairly sensitive to disturbance, but the lack of overall abundance was an indicator that this reach of the Swan River was impacted by one or more stressors. The overall FIBI score was below the impairment threshold due to several characteristics of the fish community; (1) lack of simple lithophils (fish that spawn by broadcasting eggs in clean, coarse substrates, (2) lack of insectivorous fish, (3) lack of taxa richness, and (4) low fish counts.

Station 97LS021, which is located just on the other side of Oja Road from station 09LS061, was sampled only once back in July of 1997. A total of 15 species of fish were observed during this sampling event, over two times the number observed just upstream at station 09LS061 in 2009. The number of fish sampled was also considerably higher, with 140 individuals collected in at 97LS021 compared to 7 at 09LS061. Gamefish species such as Walleye, Rock Bass, Northern Pike and Largemouth Bass were observed at 97LS021, but not 09LS061. Longnose Dace, a sensitive coolwater/coldwater minnow species was also present only at 97LS021. Despite the superior fish assemblage observed at station 97LS021 back in ’07, this station still scored below the FIBI impairment threshold (Table 102).

**Table 102:** Summary of biological monitoring stations on the Swan River, FIBI results, and applicable standards

| | **Fish Assessments** | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Station** | **Drainage Area (mi2)** | **Gradient (%)** | **Stream Order (Strahler)** | **FIBI Class** | **FIBI Result *(visit year)*** | **FIBI Result *(visit year)*** | **FIBI Result *(visit year)*** | **Standard** | **IBI Lower Confidence Limit** | **IBI Upper Confidence Limit** |
| **09LS061** | 244.26 | 0.02 | 4 | 5 | **36** *(2009)* | - | - | 47 | 38 | 56 |
| **97LS021** | 247.20 | 0.03 | 4 | 5 | **43** *(1997)* | - | - | 47 | 38 | 56 |

## Candidate Causes for Impairment

Water quality and physical habitat data were used to develop a working list of candidate causes for the FIBI impairment in the Swan River. The following candidate causes were selected for further evaluation as potential stressors;

1. **TSS / Turbidity**
2. **Poor Habitat Conditions**

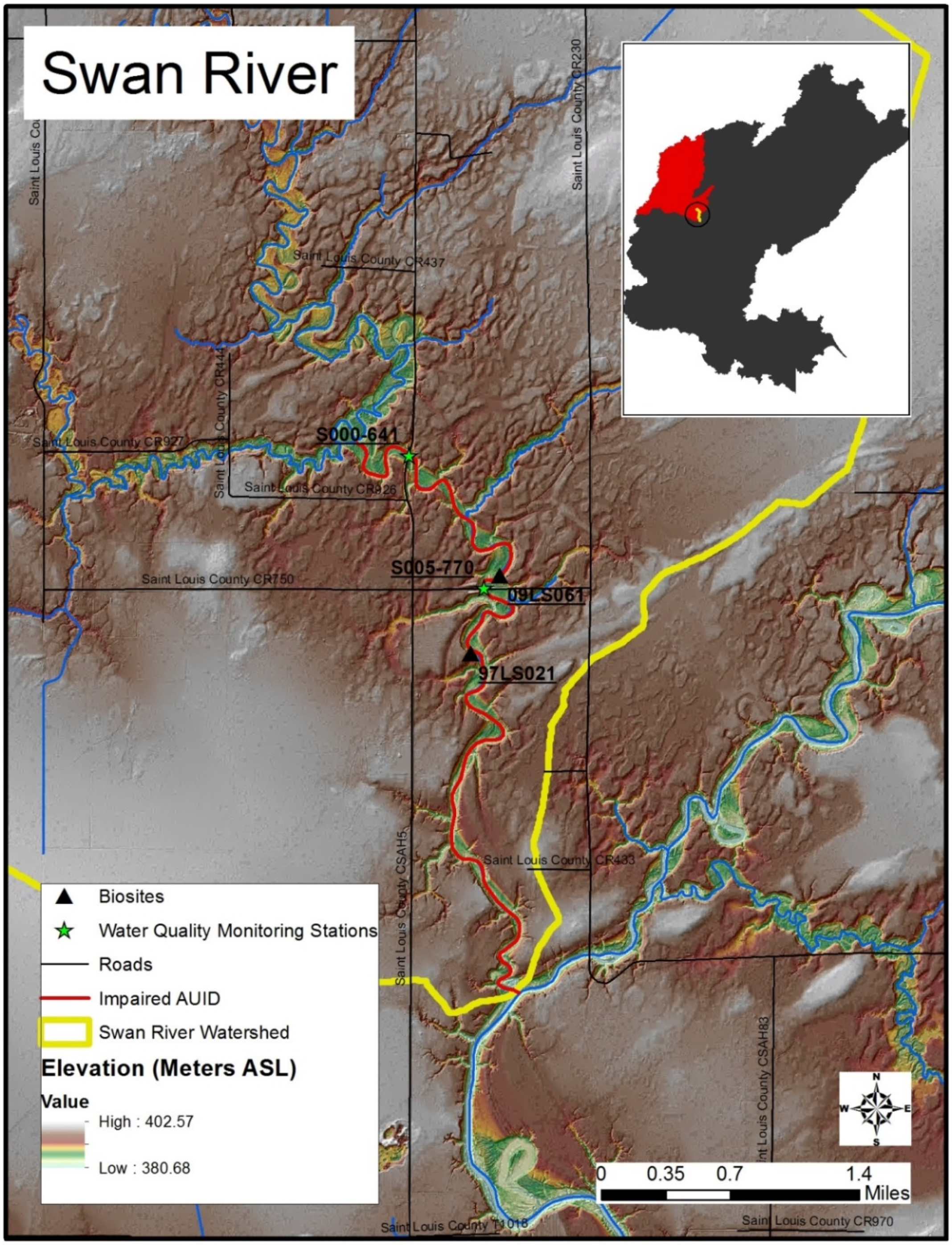


**Figure 352:** Map of Swan River Watershed, impaired stream reach, and monitoring stations

## 5.23.1 Total suspended solids and Turbidity

The TSS and Secchi transparency tube (s-tube) data is available from two stations on the Swan River. A summary of these results are provided in Table 103, along with the rate at which applicable water quality standards were violated. Overall, both of the monitoring stations on the Swan River exceed water quality standards for TSS and s-tube at a high rate (48% – 82% of the samples, depending on site and parameter). This data summary confirms and validates the existing turbidity impairment, and provides good support for listing TSS and low transparency as a candidate cause for the fish impairment. Box plots of the TSS and Secchi Tube values for the Swan River and the “A” and “B” reference streams in the SLRW are shown in Figure 354 and 355. Both datasets show a clear departure from the SLRW reference streams that were presented in Section 1.2.3.

**Table 103**: TSS and Secchi Tube average values and percent standard exceedances for Swan River

| Site | Site Description | TSS Average (mg/L) | # of Samples | TSS % Exceeding Standard | Secchi Tube Average (cm) | # of Samples | Secchi Tube % Exceeding Standard | Total # of Samples | Total % Exceeding Standard |
|---|---|---|---|---|---|---|---|---|---|
| S000-641 | SWAN RIVER 5.5 MI N OF TOIVOLA ON CSAH-5 | 44.1 | 17 | 82.4% | 49.6 | 18 | 61.1% | 35 | 71.4% |
| S005-770 | SWAN R AT CR-750, 4 MI SE OF LITTLE SWAN | 32.1 | 16 | 62.5% | 38.4 | 23 | 47.8% | 39 | 53.8% |

### Seasonal variation in total suspended solids

Elevated TSS concentrations in the Swan River are directly and positively related to discharge. Snowmelt runoff and large rain events result in extremely turbid conditions, but during low flow periods, the Swan River is fairly clear and tannin stained (Figure 353). A plot of all Swan River TSS results by month shows a distinct trend of higher TSS values from April to June, when stream flows tend to be higher as a result of snowmelt events and spring rains that fall on saturated ground (Figure 356 and 357). From July to October TSS in the Swan River maintain consistently low levels and regularly meet the draft 15 mg/L warmwater standard.

**Figure 353:** Turbid conditions (high TSS) in Swan River during snowmelt (left) and its clear appearance at low flow (right)

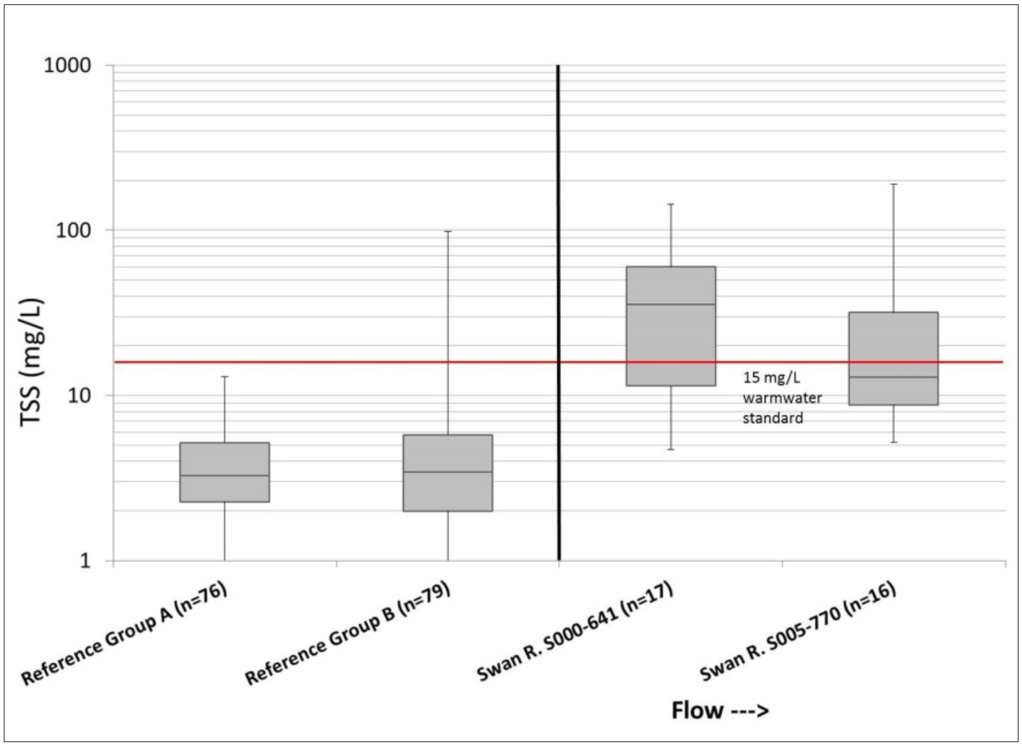


**Figure 354:** Box plots of TSS values for the Swan River and reference streams (see Section 1.2.3 for reference streams)

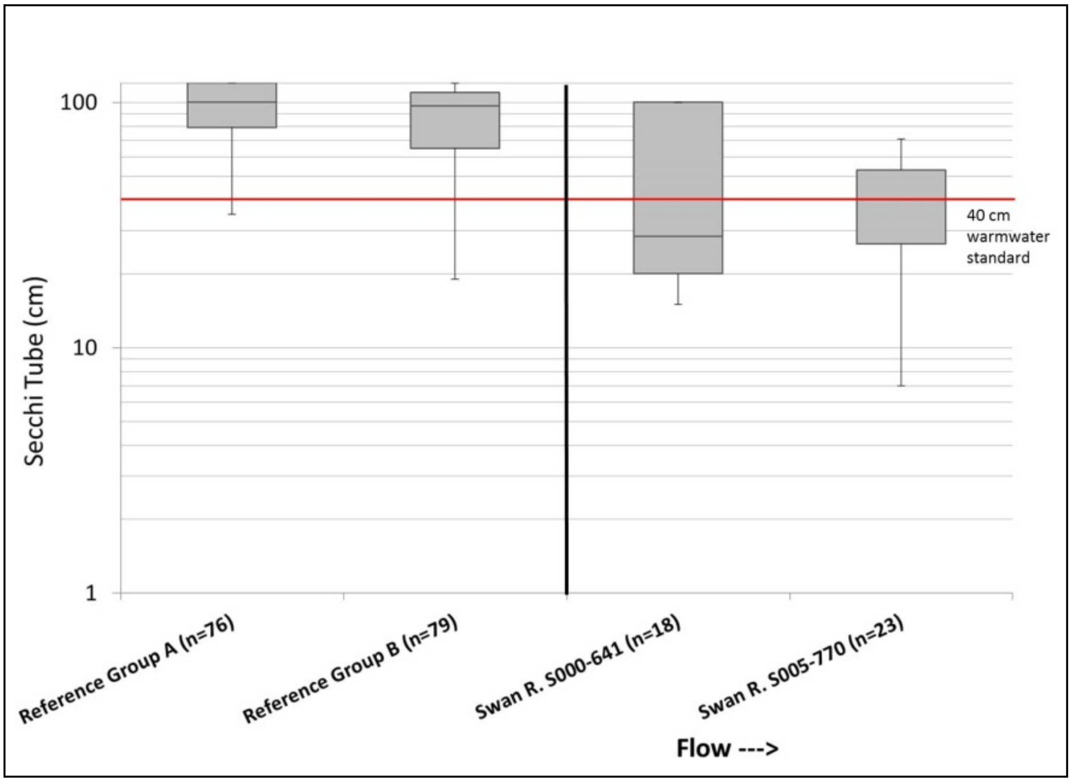


**Figure 355:** Box plots of Secchi Tube values for the Swan River and reference streams (see Section 1.2.3 for reference streams)

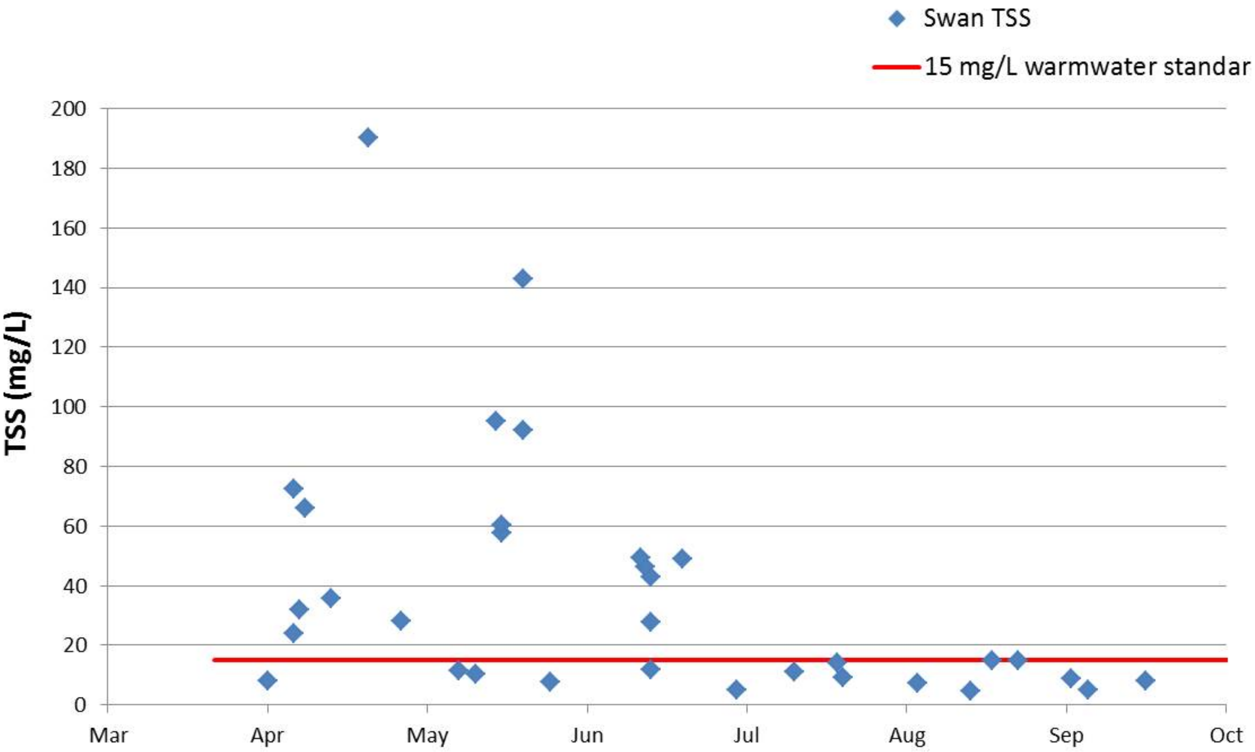


**Figure 356:** Swan River TSS data plotted by month

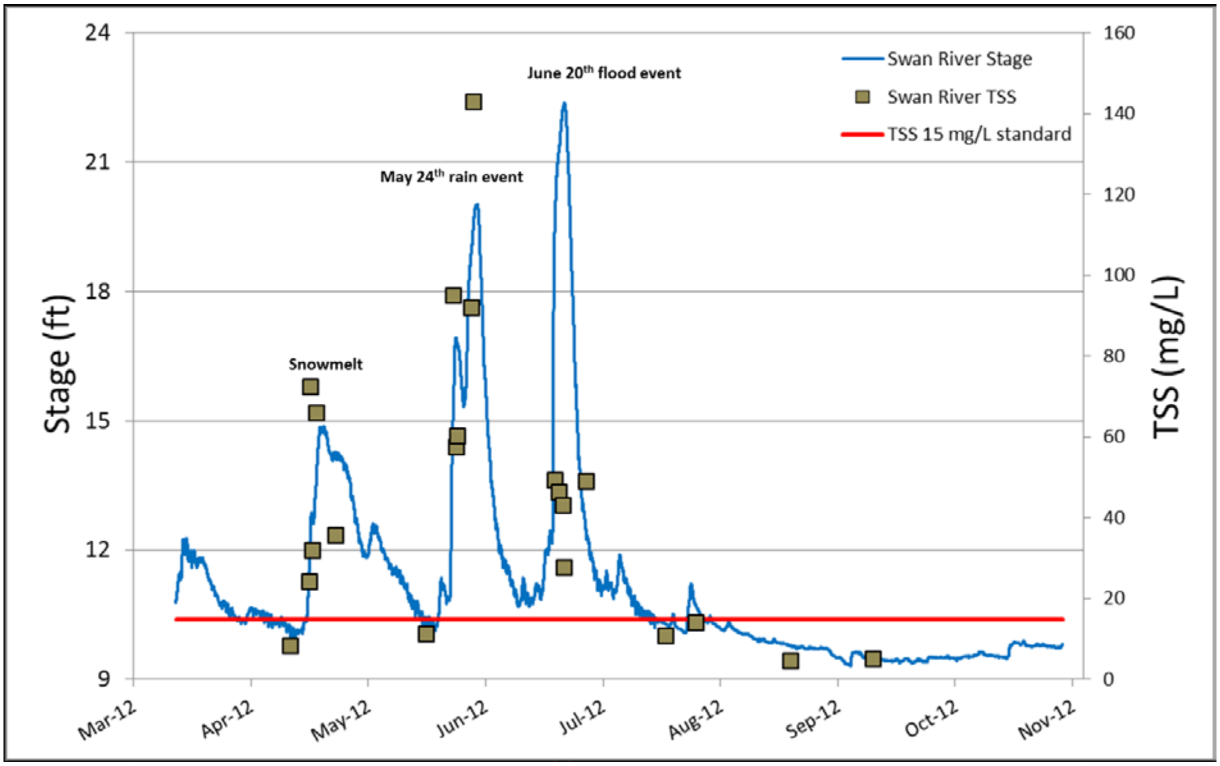


**Figure 357:** Plot of Swan River stage at the Hwy 5 gage versus TSS data from 2012

Paired streamflow and TSS data from collected during the 2012 season displays the strong positive relationship between flow and suspended solids in this system (Figure 357). In the first half of 2012 there were three peaks in flow; a snowmelt event in late April, a May 24th rain event, and the larger June 20th flood event that dropped 9 inches of rain in 24 hours in much of Northeastern Minnesota. These three peaks were accompanied by large increases in TSS. Every sample that was taken during these events violated the Class 2B (warmwater) standard for TSS. Conversely, low flow samples were consistently under the 15 mg/L standard. The Swan River essentially exceeds the draft TSS standard with any significant rainfall event or snowmelt. Following the high flow events, the river tends to stay turbid for long periods of time due the fine silt and clay particles that are suspended. These prolonged periods of exposure have a high potential to stress aquatic biota.

### Sources and pathways of sediment in the Swan River Watershed

Channel Instability/Bank Erosion

Areas of channel instability and bank erosion were observed throughout the Swan River Watershed (Figure 358) and represent a significant source for suspended solids. For more discussion on the causes and repercussions of channel instability in the Swan River Watershed see Swan River Geomorphic Study on MPCA's website.

**Figure 358:** Examples of bank erosion and channel instability in the Swan River watershed

## Urban/Industrial Stormwater Runoff

In urban, industrial and mining areas, roads, parking lots and other impervious surfaces cause precipitation to flow over the land instead of infiltrating into the ground. This rapidly flowing runoff can transport sediment and other particulates into streams, and can also lead to an increase in the rate of streambank erosion in vulnerable areas. During snowmelt, road sand, litter, and other detritus piled into snowbanks over the course of the winter season are flushed into stormwater pipes and streams in a relatively short period of time. Soils exposed by construction and industrial activities are susceptible to erosion during rain events. The Swan River Watershed has a fair amount of developed and barren land in the upper reaches of the watershed (Figure 359), thus there is potential for urban runoff as a pathway for suspended solids in the Swan River.

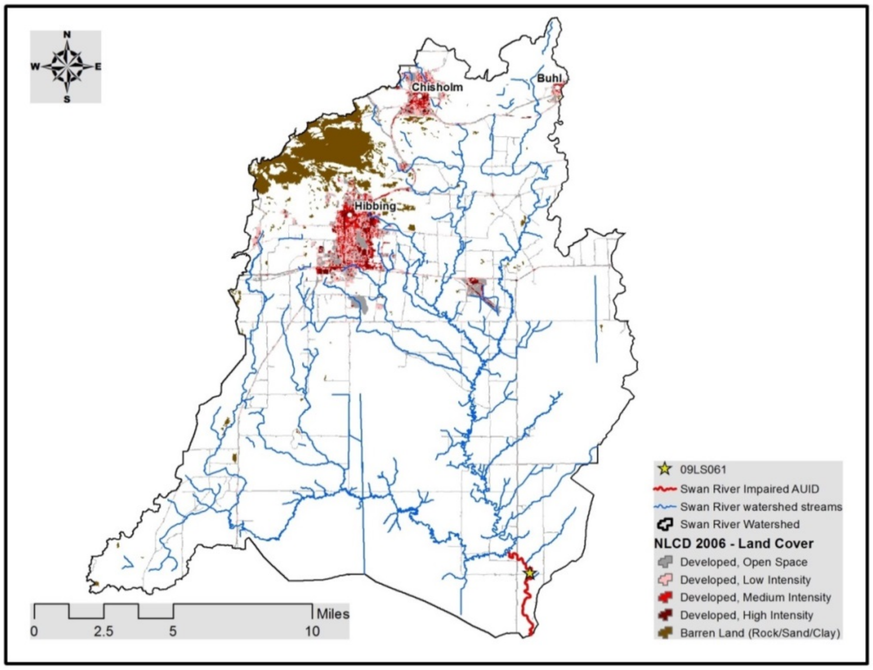


**Figure 359:** Developed and barren land in the Swan River Watershed (National Land Cover Database, 2006)

### Biological effects of elevated TSS

#### Fish Response to TSS

The FIBI impairment in the Swan River is the result of poor metrics related to low fish counts and a lack of species that are expected in healthy Northern Streams. Two stations were sampled on the Swan River, 97LS021 in 1997 and 09LS061 in 2009. Based on species TIVs developed by the MPCA, the majority of fish sampled in the Swan River are “neutral” species in terms of tolerance to elevated levels of TSS. In other words, these are species that are found in a wide variety of water conditions, from semi-turbid to clear water conditions. The most common neutral species observed in the Swan River were Creek Chub, Trout-Perch, Largemouth Bass, and White Sucker. A precipitous decline in the number of species and individuals collected occurred between the two monitoring events. In 1997 at 97LS021, 14 species and 132 individual fish were sampled. In contrast, a depauperate assemblage was sampled at 09LS061 in 2009 that included 7 species and 8 individuals. These two sites are only 0.75 miles from each other and the characteristics of the river are very similar at both locations (same stream type, slope, riparian condition, etc.). This suggests that the substantial decline in fish population may be the result of some recent phenomena and needs further study.

Comparing streams with healthy fish communities to those of the Swan River reveals that the fish assemblage in the Swan is much more tolerant of TSS. Figure 360 compares fish community TSS (TIV) for the two Swan River monitoring sites against; (1) all the Class 5 streams in the SLRW with FIBI scores above the UCL of the impairment threshold (“Class 5 AUCL”), (2) all class 5 streams in the SLRW scoring above the FIBI impairment threshold (“Class 5 AT”), and (3) all stations scoring above the UCL of the FIBI impairment threshold (regardless of FIBI class) in the SLRW (“SLRW AUCL”). Even though 97LS021 had more fish and more species, its TSS index value scores worse than 09LS061 due to the high numbers of more tolerant species at that station, such as Brassy Minnow and Fathead Minnow. The index value of 97LS021 is one of the highest in the SLRW, and is more tolerant than 100% of the stations in the SLRW scoring above the IBI threshold. Stations 97LS061 scores slightly better, but based on TIV results, the fish community at this station remains more tolerant than 100% of the Class 5 AUCL streams and 75% Class 5 AT streams.

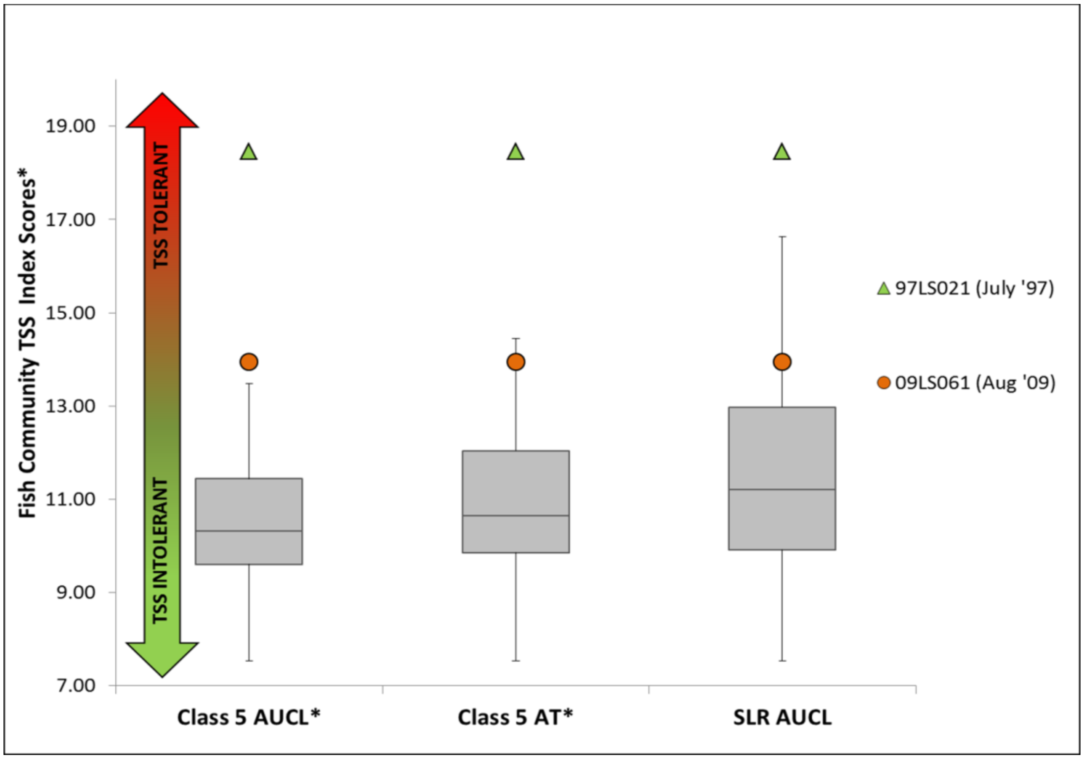


**Figure 360:** Fish community TSS TIV results for Swan River stations compared to results from high quality stations of the same IBI class. * See Section 4 for explanation of TIVs **AUCL** = Above Upper Confidence Limit of FIBI threshold **AT** = Above FIBI Threshold

### Summary: Elevated TSS

Available TSS, turbidity, and s-tube data for the Swan River frequently violates state water quality standards for protecting aquatic life. Although the majority of the extremely high TSS concentrations occur during spring snowmelt and after rain events, turbid conditions persist for long periods of time due to the clay and silt material eroding from streambanks, valley walls, and ravines along the Swan River and tributary streams.

Fish community TIVs for TSS show a community that is highly tolerant of elevated TSS concentrations compared to high quality streams in the greater SLRW. Based on the evidence provided through the water chemistry and biological data, elevated TSS is considered a stressor to the fish community and a contributing cause to the impaired condition.

## 5.23.2 Poor Physical Habitat Conditions

### Sources and pathways of degraded habitat in Swan River

Stream habitat conditions in the Swan River were evaluated using the MSHA semi-quantitative protocols developed by the MPCA's biological monitoring unit (source). MSHA results from a 2009 assessment of the Swan River at station 09LS061 resulted in score of 62.2 out of 100, which corresponds to a "fair" narrative score. The surrounding land use, riparian zone, and substrate of the stream are in fairly good shape and did not significantly affect the overall habitat score, combining for 35.2 out of a possible 47 points. The cover elements in the channel are somewhat limited and produced lower scores than high quality reference streams (Figure 362). Specific cover types that were lacking include undercut banks, boulders, rootwads, large woody cover, and aquatic vegetation. Scores were also relatively low in the channel morphology category, with poor channel stability, channel development, and velocity ratings.

**Figure 361:** Photos from station 09LS061 on the Swan River. Note the lack of cover for fish and the lack of riffle and glide features.

The PSI was also used to evaluate physical habitat and stream channel stability in the Swan River. The PSI scores for station 09LS061 portray a marginally more unstable channel than the MSHA assessment results. The overall PSI score of 92 corresponds to a stability rating of "moderately unstable" for the potential C5 stream type. Characteristics such as bank erosion, pool filling, and loose bottom sediments were common at this site, and are typical of a slightly incised channel. It is possible that the consequences of channel instability are causing habitat degradation in the Swan River. For more information on channel instability in the Swan River Watershed, see Appendix X (Swan River geomorphic study).

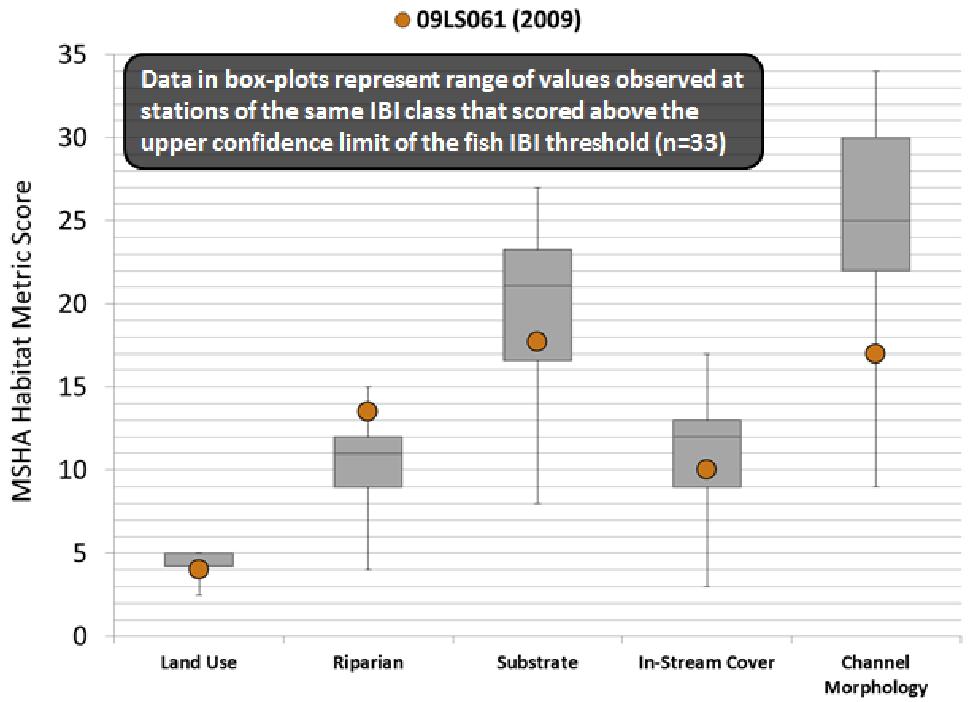


**Figure 362:** Composite scores for 5 MSHA metrics at station 09LS061 (Swan River) compared to results from stations of the same FIBI class (Class 5) that scored above the upper confidence limit of the IBI standard

### Biological effects of degraded habitat

The fish impairment listing for the Swan River is based on data collected from station 09LS061 in the summer of 2009. Seven total species were present in this sample, including several that are fairly sensitive to habitat disturbance (e.g. Shorthead Redhorse, Smallmouth Bass). However, very few fish were captured during this survey, and the overall size of the fish population in this reach does not compare well with healthy streams of the same IBI class in the SLRW. The density of 0.014 non-tolerant fish per meter present at 09LS061 was lower than 75% of the high quality Class 5 stations in the SLRW. The median fish density value for unimpaired Class 5 streams is almost 20 times what was sampled at the Swan River station. In short, the overall number of fish observed in this reach was extremely low considering the drainage area and size of the river at this location. The lack of available cover and fast-shallow water habitat (riffles and glide features) may have a role in the low fish density observed at this station.

**Table 104:** Summary of fish data from station 09LS061, including species observed and total counts for each species

| Station | Stream Name | Common Name | Count |
|---|---|---|---|
| 09LS061 | Swan River | Creek Chub | 1 |
| 09LS061 | Swan River | Northern Redbelly Dace | 1 |
| 09LS061 | Swan River | Common Shiner | 1 |
| 09LS061 | Swan River | Shorthead Redhorse | 1 |
| 09LS061 | Swan River | Trout-Perch | 1 |
| 09LS061 | Swan River | Smallmouth Bass | 2 |
| 09LS061 | Swan River | White Sucker | 1 |

In addition to the low fish abundance measures, station 09LS061 generally supported few habitat-sensitive taxa compared to non-impaired SLRW Class 5 stations. Shorthead Redhorse and Trout-Perch were the only two benthic insectivore species present at 09LS061, while the median number of benthic insectivore species observed at non-impaired class 5 streams was four. Given the dominance of silt and sand substrates in this reach of the Swan River, and the lack of woody debris and aquatic macrophytes, it is of no surprise that benthic feeders are somewhat supressed compared to high quality sites in the SLRW.

Only one piscivorous individual (Smallmouth Bass) was present at 09LS061 in 2009. This value was lower than 75% of the non-impaired Class 5 stations used in the comparison. Walleye and Northern Pike were two species absent from station 09LS061 which were found in a good number of the high quality class 5 stations in the SLRW. Two riffle-dwelling species (Shorthead Redhorse and White Sucker) and three gravel spawning species (Shorthead Redhorse, Common Shiner, and White Sucker) were present, equal to the 25th percentile of the unimpaired stations. The poor numbers of taxa in these categories are likely a result of the relative lack of velocity variability, habitat diversity, and dominance of fine substrate in the stream.

**Summary: Is physical habitat a stressor in the Swan River?**

Poor scores in several MSHA metrics (channel morphology and cover) as well as the “moderately unstable” Pfankuch Stability rating indicate a certain amount of channel instability and subsequent habitat degradation is occurring in the Swan River. These poor habitat conditions are likely contributing to several of the symptoms of impairment observed in the fish community, including low fish density, a lack of habitat-sensitive fish taxa, and few benthic feeding specialists and piscivorous fish. In comparison to many high quality streams of similar size, habitat conditions in the Swan River are far from ideal.

However, there are some streams in the region with good to excellent FIBI scores that possess very similar physical habitat conditions to the lower Swan River. Station 09LS051 on the Whiteface River (shown in Figure 363), which is close in proximity to the Swan River and has a similar morphological setting, produced FIBI results well above the UCL of the standard despite having a slightly poorer overall MSHA result and comparable metric scores related to channel morphology and fish cover. According to the MSHA results, substrate conditions were somewhat more favorable at the Whiteface River station, and water chemistry results show no indication of the TSS and turbidity issues that plague the Swan River system. Overall conditions are likely more favorable in the Whiteface River, but in terms of physical habitat, there are quite a few similarities to the lower Swan River.

We recommend including physical habitat as a stressor in the lower Swan River. Water quality conditions in the Swan River (esp. elevated TSS concentrations) may be a higher priority stressor with more direct effects on the biota, but the physical habitat conditions within the impaired reach are also contributing to low IBI scores.

**Figure 363:** Station 09LS061 on the impaired reach of the Swan River (left) compared to station 09LS051 on the Whiteface River, which had exceptional FIBI scores. Note similar habitat features but poorer WQ conditions in the impaired reach of the Swan River on left.

### 5.23.3 Swan River: Summary of Stressors to Aquatic Life

**Table 105:** Summary of SID results for the Swan River

| Candidate Cause | Result |
|---|---|
| **Total Suspended Solids (TSS)** | ● |
| **Poor Physical Habitat** | ● |
| Key: ● **= confirmed stressor** ○ **= Potential Stressor X = eliminated candidate cause** | |

# 6.0 Stressor ID Conclusions and Future Monitoring Needs

The stressors discussed and evaluated for the 24 biota-impaired streams in this report represent a wide variety of physical and chemical impacts. Several of these stressors, such as low DO and poor physical habitat conditions were observed in all regions of the SLRW. Other stressors were more regionalized and linked to specific local land-uses or municipalities. For example, specific conductivity concerns were only observed in the impaired streams located on the iron range or within the city limits of Duluth. A summary of diagnosed ("probable") stressors and potential stressors for each impaired stream are listed in Table 106.

Additional work is needed to better understand several of the stressors discussed in this document. Below is a list of future monitoring and research needs that would improve confidence level in diagnosing certain stressors:

- Statewide or St. Louis River 8-HUC scale investigation of the effects specific conductivity on aquatic life. Other U.S. states (e.g. West Virginia) have developed, or are in the processes of developing water quality standards for specific conductivity that are based on protection of aquatic life.

- Additional paired monitoring of sulfate and aquatic life (fish and macroinvertebrates) is needed to further our understanding of this potential stressor in the SLRW. An effort is currently underway to develop a sulfate standard for wild rice bearing waters of the state. A similar standard is needed for the protection of fish and macroinvertebrate populations. Other states have developed, or are in the process of developing sulfate standards for fish and macroinvertebrates (e.g. Iowa, Illinois, Pennsylvania). Given the elevated sulfate concentrations observed in streams of the SLRW (and other mining areas of the state), a similar standard is needed to improve our confidence in diagnosing this stressor in impaired waters.

- The urban area of Duluth is unique in that many streams within its city limits support a wild Brook Trout population. Additional work to better understand the sources and negative impacts of urban related stressors (chloride, specific conductivity, flow alteration, elevated water temperatures, metals toxicity) would be highly valuable considering the importance of these resources to the community.

- The Sax-Zim bog area encompasses a large area of the SLRW, and many of the impaired streams discussed in this report are hydrologically connected to this expansive area. Additional monitoring and modeling analysis would be beneficial in this area to better understand the impacts of: (1) the extensive ditch network designed to drain many of these wetland areas, (2) natural background conditions (e.g. peat soils, shallow water table) that are likely contributing to low DO concentrations in many of these streams, (3) the effects of soft water and bog stain on overall biological productivity of these streams and the potential for these streams to support diverse fish and macroinvertebrate assemblages.

- Many streams in the St. Louis River exceed the chronic toxicity standard for Aluminum. The majority of the streams exceeding the standard are located in the Meadowlands/Sax-Zim bog region and the Swan River Watershed. Many of the available monitoring results for this parameter were from high-flow (snowmelt and rain event) conditions, and the elevated aluminum concentrations are likely linked to sediment entering the stream from erosion. Aluminum toxicity was not discussed at a high level in this report, but additional monitoring and analysis is warranted for some of the SLRW sub-watersheds.

**Table 106:** Overall summary of SID results for impaired stream reaches of the St. Louis River 8HUC watershed

| Stream Name | AUID | Impairment | Stressor ID Summary |
|---|---|---|---|
| **Kingsbury Creek** | **04010201-626** | **F-IBI / M-IBI** | **Diagnosed/Probable Stressors:**<br>• Low Dissolved Oxygen<br>• Poor Physical Habitat Conditions<br>• Elevated Water Temperatures<br>• Total Suspended Solids (TSS)<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Chloride / Specific Conductivity<br>• Altered Hydrology and Connectivity<br>• Lead Toxicity |
| **Miller Creek** | **04010201-512** | **F-IBI / M-IBI** | **Diagnosed/Probable Stressors:**<br>• Elevated Water Temperatures<br>• Chloride / Specific Conductivity<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Altered Hydrology and Connectivity |
| **Wyman Creek** | **04010201-942** | **F-IBI** | **Diagnosed/Probable Stressors:**<br>• Elevated Water Temperatures<br>• Low Dissolved Oxygen<br>• Altered Hydrology/Connectivity due to beaver dams<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Habitat Loss due to Iron Precipitate<br>• Iron toxicity<br>• Sulfate Toxicity |
| **Paleface Creek** | **04010201-A24** | **F-IBI / M-IBI** | **Diagnosed/Probable Stressors:**<br>• Low Dissolved Oxygen<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• None |
| **Water Hen Creek** | **04010201-A35** | **M-IBI** | **Diagnosed/Probable Stressors:**<br>• Low Dissolved Oxygen/Dissolved Oxygen Flux<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• None |

**Table 106 (Continued) :** Overall summary of SID results for impaired stream reaches of the St. Louis River 8HUC watershed

| Stream Name | AUID | Impairment | Stressor ID Summary |
|---|---|---|---|
| **Water Hen River** | **04010201-A31** | **M-IBI** | **Diagnosed/Probable Stressors:**<br>• Low Dissolved Oxygen<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• None |
| **Little Swan Creek** | **04010201-891** | **F-IBI** | **Diagnosed/Probable Stressors:**<br>• Low Dissolved Oxygen<br>• Poor Physical Habitat Conditions<br>• Elevated Water Temperatures<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Total Suspended Solids (TSS) |
| **Sand Creek** | **04010201-607** | **F-IBI** | **Diagnosed/Probable Stressors:**<br>• Poor Physical Habitat Conditions<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Altered hydrology |
| **Skunk Creek** | **04010201-A18** | **F-IBI / M-IBI** | **Diagnosed/Probable Stressors:**<br>• Low Dissolved Oxygen<br>• Poor Physical Habitat Conditions<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Altered Hydrology |
| **St Louis River** | **04010201-508** | **M-IBI** | **Diagnosed/Probable Stressors:**<br>• Poor Physical Habitat Conditions<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• None |
| **Stony Creek** | **04010201-963** | **F-IBI / M-IBI** | **Diagnosed/Probable Stressors:**<br>• Low Dissolved Oxygen<br>• Poor Physical Habitat Conditions<br>• Total Suspended Solids (TSS)<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Altered hydrology |

**Table 106 (Continued) :** Overall summary of SID results for impaired stream reaches of the St. Louis River 8HUC watershed

| Stream Name | AUID | Impairment | Stressor ID Summary |
|---|---|---|---|
| **Vaara Creek** | **04010201-623** | **F-IBI / M-IBI** | **Diagnosed/Probable Stressors:**<br>• Low Dissolved Oxygen<br>• Poor Physical Habitat Conditions<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Altered Hydrology |
| **Unnamed Tributary to St. Louis River** | **04010201-A17** | **M-IBI** | **Diagnosed/Probable Stressors:**<br>• Poor Physical Habitat Conditions<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Altered Hydrology and Connectivity |
| **Otter Creek** | **04010201-629** | **M-IBI** | **Diagnosed/Probable Stressors:**<br>• None<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Elevated Water Temperature<br>• Poor Physical Habitat Conditions |
| **Ely Creek** | **04010201-A26** | **F-IBI** | **Diagnosed/Probable Stressors:**<br>• Low Dissolved Oxygen<br>• Poor Physical Habitat Conditions<br>• Altered Hydrology<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• None |
| **Embarrass River** | **04010201-579** | **F-IBI** | **Diagnosed/Probable Stressors:**<br>• Low Dissolved Oxygen<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Altered Hydrology |

| Stream Name | AUID | Impairment | Stressor ID Summary |
|---|---|---|---|
| **Spring Mine Creek** | **04010201-A42** | **F-IBI / M-IBI** | **Diagnosed/Probable Stressors:**<br>• Low Dissolved Oxygen / Dissolved Oxygen Flux<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Sulfate Toxicity<br>• Specific Conductivity<br>• Limited Physical Habitat (low gradient/wetland influence) |
| **East Swan Creek** | **04010201-888** | **M-IBI** | **Diagnosed/Probable Stressors:**<br>• Nitrate Toxicity<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Specific Conductivity |
| **Swan River** | **04010201-557** | **F-IBI** | **Diagnosed/Probable Stressors:**<br>• Total Suspended Solids (TSS)<br>• Poor Physical Habitat Conditions<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Altered Hydrology and Connectivity |
| **Elbow Creek** | **04010201-518** | **F-IBI / M-IBI** | **Diagnosed/Probable Stressors:**<br>• Low Dissolved Oxygen<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Specific Conductivity<br>• Sulfate Toxicity<br>• Nitrate Toxicity |
| **Elbow Creek** | **04010201-570** | **M-IBI** | **Diagnosed/Probable Stressors:**<br>• Poor Physical Habitat Conditions<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• None |

**Table 106 (Continued) :** Overall summary of SID results for impaired stream reaches of the St. Louis River 8HUC watershed

**Table 106 (Continued):** Overall summary of SID results for impaired stream reaches of the St. Louis River 8HUC watershed

| Stream Name | AUID | Impairment | Stressor ID Summary |
|---|---|---|---|
| **Manganika Creek** | **04010201-548** | **F-IBI / M-IBI** | **Diagnosed/Probable Stressors:**<br>• Low Dissolved Oxygen<br>• Total Suspended Solids (TSS)<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Specific Conductivity<br>• Sulfate Toxicity<br>• Ammonia & pH<br>• Nitrate Toxicity |
| **Kinney Creek** | **04010201-551** | **M-IBI** | **Diagnosed/Probable Stressors:**<br>• High DO Flux<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Sulfate Toxicity<br>• Specific Conductivity |
| **West Two River** | **04010201-535** | **M-IBI** | **Diagnosed/Probable Stressors:**<br>• Low Dissolved Oxygen / High Dissolved Oxygen Flux<br>**Potential Stressors & Focus Areas for Additional Monitoring:**<br>• Altered Hydrology<br>• Sulfate Toxicity<br>• Specific Conductivity |

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# APPENDIX A (Submitted by Ann Thompson, South St. Louis Co. SWCD)

## Stream Classification

### Why classify?

Streams are classified in order to have a system that is recognizable and easily communicated. This classification system allows for consistency among researchers and practitioners and allows for organization of data and information. Stream classification allows the ability to reproduce data and to extrapolate data to similarly classified streams.

The Rosgen stream classification is used in this report. This method has been accepted and is used by the Minnesota DNR, the Natural Resources Conservation Service (NRCS), as well as many other agencies. (Rosgen D. L., A Classification of Natural Rivers, 1994)

### How do we classify?

Rosgen stream classification determines stream type based on dimension, pattern, and profile. These three parameters are first determined by investigating the channel shape (cross-section), the channel slope, and the sinuosity. When further analyzing stream type entrenchment and the width-to-depth ratio are also used. When stream type is determined valley type, stream order, watershed area, and channel material are also documented. (Rosgen D. L., A Classification of Natural Rivers, 1994)

Stream types change throughout a river system. Some rivers may have the same stream type for many miles, while other rivers have many reaches of differing stream types within a short distance. Changes in stream type may occur because of changes in geology or changes in hydrology from land cover or an entering tributary (Rosgen D. L., A Classification of Natural Rivers, 1994).

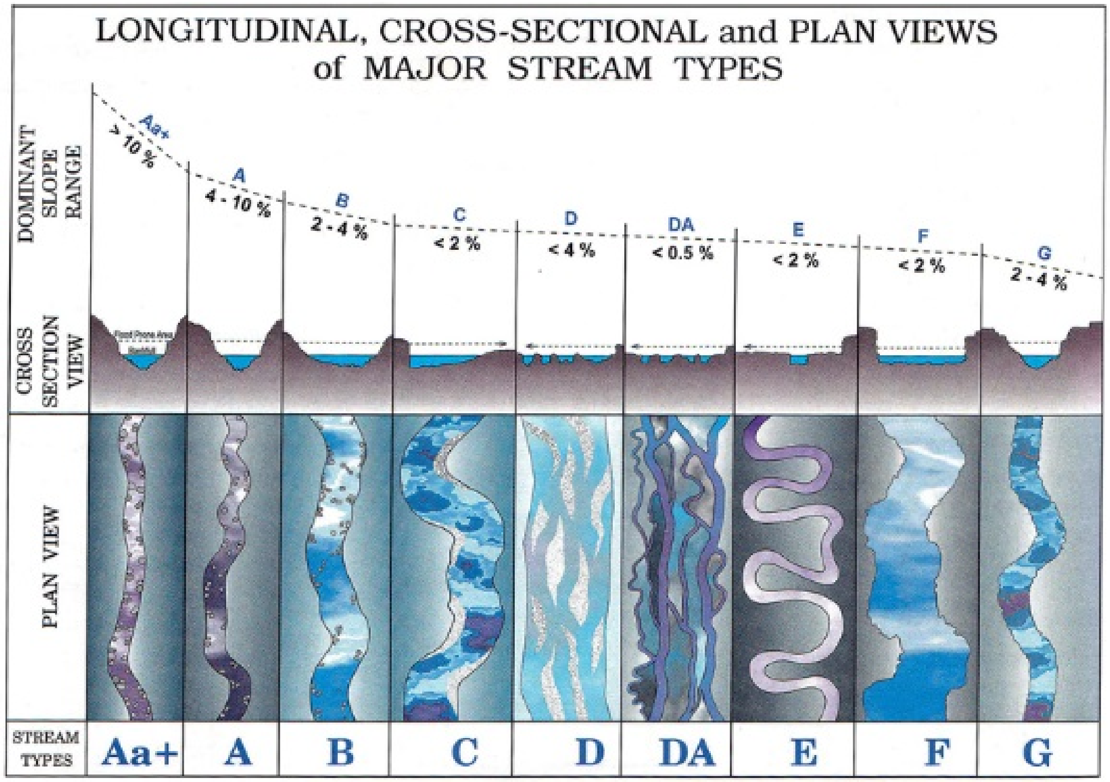


**Figure:** Dave Rosgen's stream type classification. Longitudinal, cross section, and plan views are shown for each stream type. Source: Dave Rosgen, A classification of Natural Rivers, 1994.

Another aspect of stream classification is channel material. Channel material is associated with a number ranging from one (bedrock) to six (silt/clay) (Rosgen D. L., A Classification of Natural Rivers, 1994). This number gives reference to the material that is most dominant in the channel.

It is important to note that stream type often changes when instability occurs. When determining a stream type we must also determine the overall stability of the stream to maintain that classification. Not only do practitioners determine the current stream type they also determine the potential stream type for the reach to become stable.

### Stream types on Stanley Creek and the Main West Branch

Stanley Creek and the MWB have a few main stream types that were documented. Below is a general explanation of each of these stream types.

### B Channel

A B channel is a moderately steep channel with a step-pool sequence. B channels have slopes of 2 - 4% The step pool sequence leads to scour pools forming, but unlike C and E channels, B channels are moderately entrenched and do not have wide floodplains. B channels have a moderate width-to-depth ratio and a sinuosity of greater than 1.2. The steep slopes and entrenchment of B channels make them efficient at moving sediment through their reaches. Deposition of sediment or fine materials is not common in B channels. B channels are often in good condition along the North Shore of Lake Superior and have stable stream banks and beds. B channels are found in confined colluvial and glacial trough valleys. (Rosgen D. L., Applied River Morphology, 1996)

**C Channel**

A C channel is characterized as a meandering stream with a floodplain and riffle-pool sequence. C channels generally have a slope of less than 2%. Moderate sinuosity and a high width-to-depth ratio of greater than 12 are other features of a C channel. Streambanks are made of fine, unconsolidated material while the streambed is composed of larger materials. Streambanks on C channels are vulnerable to high rates of erosion depending on the stability of the stream and the vegetation present. Moderate to high amounts of sediment may be created depending on the bank material. Sediment is often deposited on point bars formed on the inside meaner of a pool, and on other depositional features in C channels. Unlike B channels, C channels are inefficient at moving sediment and sediment is often deposited within the reach. This stream type is often unstable in streams along the North Shore of Lake Superior when downstream of a sediment source. C channels are found in unconfined lacustrine valleys and confined and unconfined alluvial valleys. (Rosgen D. L., Applied River Morphology, 1996)

**E Channel**

A typical E channel is narrow and deep with a low width-to-depth ratio. This narrow and deep channel shape allows E channels to effectively transport sediment and prevents the channel from down cutting. E channels are typically found in wide, lacustrine valleys and have an entrenchment ratio of greater than 2.2. E channels have high sinuosity with many meanders and flat slopes of less than 2%. E channels, like C channels have a riffle-pool sequence. E channels are often stabilized by riparian vegetation. Dense sod mats and/or large amounts of woody vegetation stabilize the banks of the E channel. Finer materials are found in the banks of the channel with larger materials deposited on the streambed. (Rosgen D. L., Applied River Morphology, 1996)

## Stream Stability

A stable stream channel is one that has little to no disturbance. It has minor erosion on the stream banks, contains diverse and suitable habitat, and changes little from year to year. Stable channels are able to withstand large flood events with minor to no effects. Geomorphologist Dave Rosgen defines a stable channel as “the ability of the stream to maintain, over time, its dimension, pattern, and profile in such a manner that it is neither aggrading nor degrading and is able to transport without adverse consequence the flows and detritus of its watershed” (Rosgen D. L., Applied River Morphology, 1996).

# APPENDIX B – Supplemental biological Analyses for Chloride Toxicity

**Table 107:** Chloride sensitivity in the macroinvertebrate populations observed in impaired streams vs. non-urban reference streams

| Visit Year | Station | Stream | #Tot | #T | #S | #U | % T | %S | %U | Abundant Tolerant | Sensitive Taxa |
|---|---|---|---|---|---|---|---|---|---|---|---|
| ***1998*** | ***98LS003*** | ***Kingsbury Creek*** | 316 | 66 | 2 | 248 | **20.9%** | **0.6%** | 78.5% | Polypedilum<br>Chironomus | Gerris |
| ***2009*** | ***95LS036*** | ***Kingsbury Creek*** | 330 | 53 | 0 | 277 | **16.1%** | **0.0%** | 83.9% | Cheumatopsyche<br>Polypedilum | none |
| ***2011*** | ***98LS003*** | ***Kingsbury Creek*** | 330 | 135 | 0 | 195 | **40.9%** | **0.0%** | 59.1% | Phaenospectra<br>Polypedilum | none |
| ***2012*** | ***12LS004*** | ***Kingsbury Creek*** | 304 | 24 | 0 | 280 | **7.9%** | **0.0%** | 92.1% | Cheumatopsyche<br>Polypedilum | none |
| ***2012*** | ***12LS005*** | ***Kingsbury Creek*** | 328 | 44 | 0 | 284 | **13.4%** | **0.0%** | 86.6% | Ceratopsyche<br>Polypedilum | none |
| ***1998*** | ***98LS001*** | ***Miller Creek*** | 357 | 22 | 0 | 335 | **6.2%** | **0.0%** | 93.8% | Polypedilum<br>Chironomus | Neurocordulia present (no count) |
| ***1998*** | ***98LS001*** | ***Miller Creek*** | 298 | 36 | 0 | 262 | **12.1%** | **0.0%** | 87.9% | Hyalella<br>Ceratopsyche | none |
| ***2009*** | ***09LS003*** | ***Miller Creek*** | 300 | 52 | 0 | 248 | **17.3%** | **0.0%** | 82.7% | Ceratopsyche<br>Cheumatopsyche | none |
| ***2010*** | ***09LS003*** | ***Miller Creek*** | 322 | 111 | 0 | 211 | **34.5%** | **0.0%** | 65.5% | Cheumatopsyche<br>Ceratopsyche | none |
| ***1998*** | ***98LS009*** | ***Keene Creek*** | 351 | 37 | 0 | 314 | **10.5%** | **0.0%** | 89.5% | Ceratopsyche<br>Cheumatopsyche | none |
| ***1998*** | ***98LS009*** | ***Keene Creek*** | 350 | 50 | 0 | 300 | **14.3%** | **0.0%** | 85.7% | Heptageniidae<br>Ceratopsyche | none |
| ***2009*** | ***95LS028*** | ***Keene Creek*** | 310 | 59 | 50 | 201 | **19.0%** | **16.1%** | 64.8% | Cheumatopsyche<br>Ceratopsyche | Capniidae<br>Molanna |
| ***2010*** | ***09LS002*** | ***Mission Creek*** | 291 | 157 | 1 | 133 | **54.0%** | **0.3%** | 45.7% | Ceratopsyche<br>Cheumatopsyche | Capniidae |
| ***1997*** | ***97LS047*** | ***Mission Creek*** | 284 | 149 | 1 | 134 | **52.5%** | **0.4%** | 47.2% | Ceratopsyche<br>Heptageniidae | Capniidae |
| ***2009*** | ***97LS108*** | ***Hay Creek*** | 308 | 111 | 7 | 190 | **36.0%** | **2.3%** | 61.7% | Ceratopsyche<br>Hydropsychidae | Paracricotopus<br>Synorthocladius |

| | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| ***2011*** | ***97LS038*** | ***Amity Creek, East Branch*** | 300 | 59 | 0 | 241 | **19.7%** | **0.0%** | 80.3% | Ceratopsyche<br>Thienemanniella | none |
| ***2011*** | ***97LS038*** | ***Amity Creek, East Branch*** | 323 | 60 | 7 | 256 | **18.6%** | **2.2%** | 79.3% | Ceratopsyche<br>Rheotanytarsus<br>Polypedilum | Synorthocladius |
| ***1997*** | ***97LS038*** | ***Amity Creek, East Branch*** | 353 | 119 | 1 | 233 | **33.7%** | **0.3%** | 66.0% | Ceratopsyche<br>Polypedilum | Eurylophella |
| ***1997*** | ***97LS038*** | ***Amity Creek, East Branch*** | 295 | 102 | 4 | 189 | 34.6% | 1.4% | 64.1% | Ceratopsyche<br>Polypedilum | Eurylophella<br>Leptophlebia<br>Glossiphoniidae<br>Ophiogomphus |
| ***2011*** | ***11LS039*** | ***Chester Creek*** | 327 | 40 | 5 | 282 | 12.2% | 1.5% | 86.2% | Ceratopsyche<br>Hydropsychidae | Capniidae |
| ***2010*** | ***97LS089*** | ***Big Sucker Creek*** | 320 | 140 | 3 | 177 | 43.8% | 0.9% | 55.3% | Ceratopsyche, Hydropsychidae | Capniidae, Brachycentridae |
| ***2011*** | ***11LS041*** | ***Big Sucker Creek*** | 313 | 144 | 2 | 167 | 46.0% | 0.6% | 53.4% | Ceratopsyche, Cheumatopsyche | Ophiogomphus |
| ***1997*** | ***97LS089*** | ***Big Sucker Creek*** | 292 | 68 | 1 | 223 | 23.3% | 0.3% | 76.4% | Rheotanytarsus, Polypedilum | Ophiogomphus |
| ***1997*** | ***97LS089*** | ***Big Sucker Creek*** | 287 | 81 | 2 | 204 | 28.2% | 0.7% | 71.1% | Rheotanytarsus, Polypedilum | Ophiogomphus |
| ***2013*** | ***13LS050*** | ***Kadunce Creek*** | 303 | 38 | 0 | 265 | 12.5% | 0.0% | 87.5% | Ceratopsyche, Polypedilum | none |
| ***2013*** | ***13LS011*** | ***Kimball Creek*** | 332 | 59 | 1 | 272 | 17.8% | 0.3% | 81.9% | Ceratopsyche, Acentrella | Capniidae |

**RED = Highly Urbanized ORANGE=Moderately Urbanized GREEN=Little to no urbanized land-use**

**#Tot** = total number of organisms sampled **#T**=number of chloride-tolerant individuals **#S**=number of chloride-sensitive individuals

**#U**=number of individuals sampled with unspecified tolerance to chloride **%T**= percent of sample comprised of chloride-tolerant individuals

**%S**= percent of sample comprised of chloride-sensitive individuals **%U**= percent of sample comprised of individuals with unspecified tolerance to chloride

| Station | Stream | #Tot | #T | #S | #U | % T | %S | %U | most abundant tolerant taxa* | most abundant sensitive taxa* |
|---|---|---|---|---|---|---|---|---|---|---|
| 95LS039 | Kingsbury Creek | 70 | 70 | 0 | 0 | 100.0% | 0.0% | 0.0% | Creek Chub, Central Mudminnow | none |
| 12LS004 | Kingsbury Creek | 41 | 33 | 5 | 3 | 80.5% | 12.2% | 7.3% | Creek Chub | Brook Trout (likely stocked) |
| 95LS037 | Kingsbury Creek | 45 | 25 | 0 | 20 | 55.6% | 0.0% | 44.4% | Creek Chub, Brook Stickleback | none |
| 98LS003 | Kingsbury Creek | 250 | 136 | 48 | 66 | 54.4% | 19.2% | 26.4% | Creek Chub, Brook Stickleback | Pearl Dace, Northern Redbelly Dace |
| 98LS001 | Miller Creek | 484 | 252 | 14 | 218 | 52.1% | 2.9% | 45.0% | Creek Chub, Central Mudminnow | Brook Trout, Pearl Dace, Northern Redbelly Dace |
| 95LS036 | Kingsbury Creek | 54 | 28 | 7 | 19 | 51.9% | 13.0% | 35.2% | Creek Chub, Brook Stickleback | Pearl Dace |
| 12LS005 | Kingsbury Creek | 8 | 4 | 1 | 3 | 50.0% | 12.5% | 37.5% | Creek Chub | Brook Trout |
| 98LS001 | Miller Creek | 493 | 234 | 50 | 209 | 47.5% | 10.1% | 42.4% | Central Mudminnow, Brook Stickleback | Brook Trout, Pearl Dace, Northern Redbelly Dace |
| 09LS002 | Mission Creek | 121 | 53 | 0 | 68 | 43.8% | 0.0% | 56.2% | Creek Chub, Brook Stickleback | none |
| 97LS047 | Mission Creek | 119 | 44 | 0 | 75 | 37.0% | 0.0% | 63.0% | Creek Chub, Brook Stickleback | none |
| 98LS003 | Kingsbury Creek | 106 | 39 | 30 | 37 | 36.8% | 28.3% | 34.9% | Creek Chub, Brook Stickleback | Brook Trout (likely stocked), Pearl Dace |
| 95LS043 | Chester Creek | 48 | 17 | 0 | 31 | 35.4% | 0.0% | 64.6% | Creek Chub, Brook Stickleback | none |
| 95LS036 | Kingsbury Creek | 74 | 24 | 8 | 42 | 32.4% | 10.8% | 56.8% | Creek Chub, Brook Stickleback | Northern Redbelly Dace, Pearl Dace, (one Brook Trout present) |
| 09LS002 | Mission Creek | 311 | 96 | 1 | 214 | 30.9% | 0.3% | 68.8% | Creek Chub, Brook Stickleback | Pearl Dace |
| 68LS014 | Hay Creek | 49 | 11 | 3 | 35 | 22.4% | 6.1% | 71.4% | Brook Stickleback | Brook Trout |
| 89LS010 | Kadunce River | 119 | 23 | 54 | 42 | 19.3% | 45.4% | 35.3% | Creek Chub, Brook Stickleback | Brook Trout, Rainbow Trout |
| 95LS042 | Chester Creek | 133 | 25 | 0 | 108 | 18.8% | 0.0% | 81.2% | Creek Chub, Brook Stickleback | none |
| 95LS026 | Keene Creek | 264 | 43 | 0 | 221 | 16.3% | 0.0% | 83.7% | Creek Chub, Central Mudminnow | none |
| 68LS013 | Hay Creek | 91 | 12 | 4 | 75 | 13.2% | 4.4% | 82.4% | Creek Chub | Brook Trout, Northern Redbelly Dace |
| 68LS015 | Hay Creek | 27 | 3 | 9 | 15 | 11.1% | 33.3% | 55.6% | Brook Stickleback | Brook Trout |
| 09LS003 | Miller Creek | 268 | 29 | 20 | 219 | 10.8% | 7.5% | 81.7% | Creek Chub, Brook Stickleback | Brook Trout, Pearl Dace |
| 95LS028 | Keene Creek | 91 | 5 | 25 | 61 | 5.5% | 27.5% | 67.0% | Creek Chub, Brook Stickleback | Brook Trout |
| 97LS038 | Amity Creek, East Branch | 201 | 9 | 121 | 71 | 4.5% | 60.2% | 35.3% | Creek Chub | Brook Trout |
| 11LS039 | Chester Creek | 503 | 20 | 0 | 483 | 4.0% | 0.0% | 96.0% | Creek Chub, Brook Stickleback | none |
| 97LS038 | Amity Creek, East Branch | 132 | 5 | 75 | 52 | 3.8% | 56.8% | 39.4% | Creek Chub | Brook Trout, Pearl Dace, Northern Redbelly Dace |

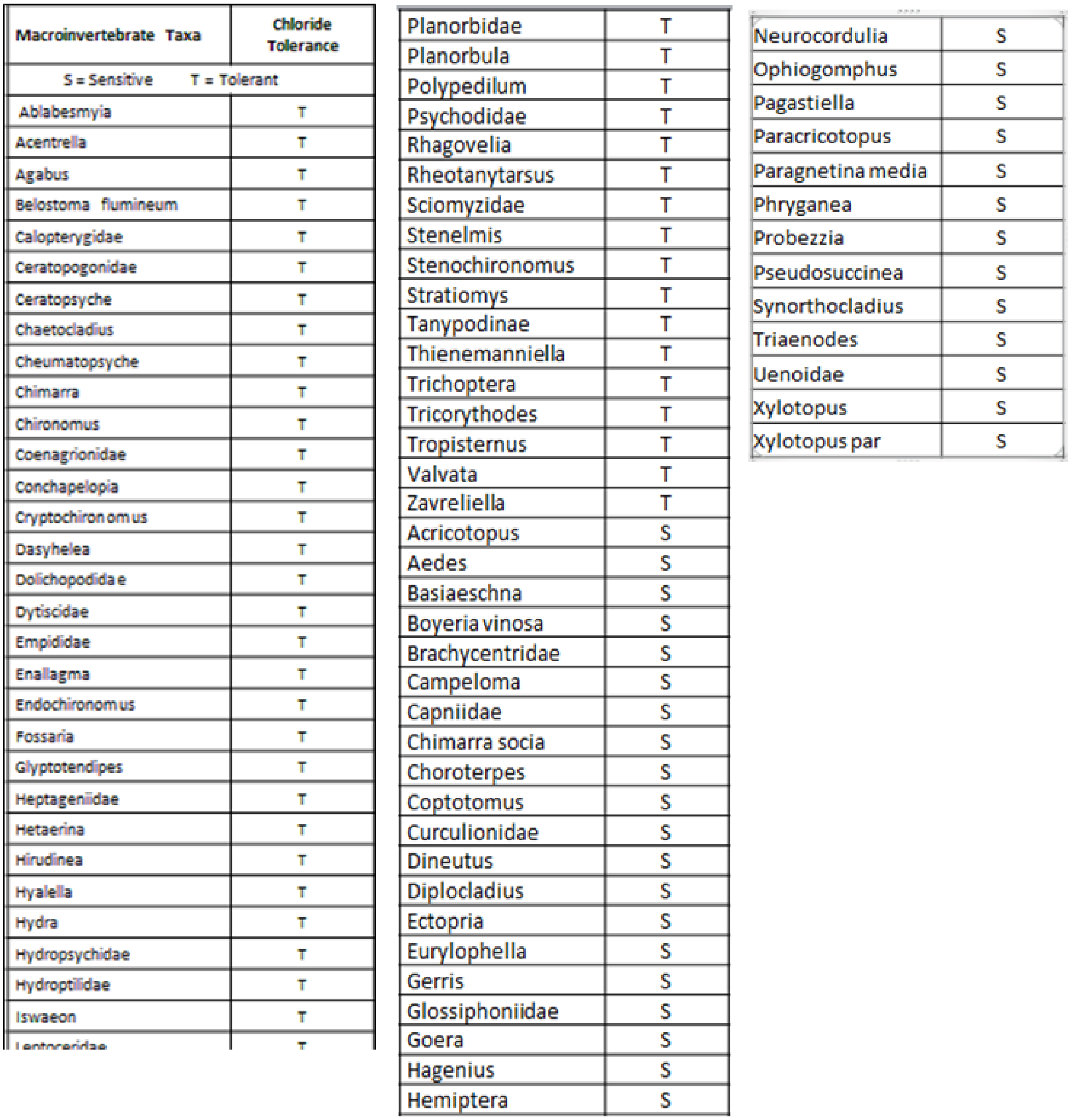
| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| 95LS041 | Chester Creek | 70 | 2 | 0 | 68 | 2.9% | 0.0% | 97.1% | Creek Chub, Brook Stickleback | none |
| 97LS089 | Big Sucker Creek | 699 | 16 | 98 | 585 | 2.3% | 14.0% | 83.7% | Creek Chub, Brook Stickleback | Brook Trout, Pearl Dace, Iowa Darter |
| 97LS038 | Amity Creek, East Branch | 104 | 2 | 58 | 44 | 1.9% | 55.8% | 42.3% | Creek Chub | Brook Trout |
| 97LS089 | Big Sucker Creek | 250 | 4 | 83 | 163 | 1.6% | 33.2% | 65.2% | Creek Chub, Brook Stickleback | Brook Trout, Northern Redbelly Dace |
| 98LS009 | Keene Creek | 287 | 3 | 9 | 275 | 1.0% | 3.1% | 95.8% | Creek Chub | Brook Trout |
| 98LS009 | Keene Creek | 212 | 1 | 16 | 195 | 0.5% | 7.5% | 92.0% | Creek Chub | Brook Trout |
| 89LS008 | Kadunce River | 209 | 0 | 167 | 42 | 0.0% | 79.9% | 20.1% | none | Brook Trout, Rainbow Trout |

**Table 108:** Chloride sensitivity in the fish populations observed in impaired streams vs. non-urban reference streams (see Figure 107 for KEY)

**Table 109:** List of chloride tolerance value assigned to various fish species based on MIBI, 2012.

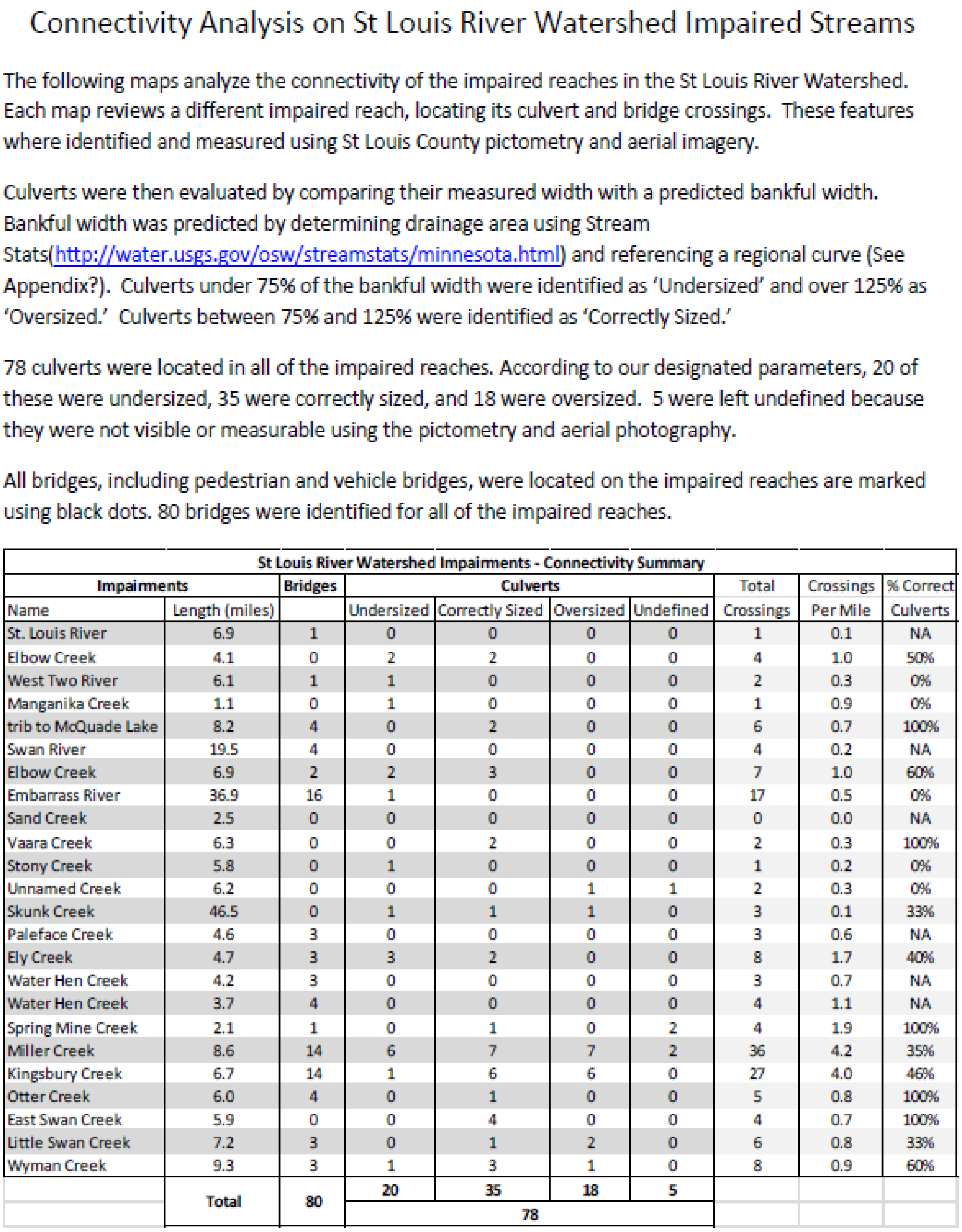
| Species Name | Chloride Sensitivity |
|---|---|
| S = Sensitive T = Tolerant | |
| **Blacknose Shiner** | **S** |
| **Brook Trout** | **S** |
| **Finescale Dace** | **S** |
| **gilt darter** | **S** |
| **Iowa Darter** | **S** |
| **lake sturgeon** | **S** |
| **least darter** | **S** |
| **Mimic Shiner** | **S** |
| **Northern Redbelly Dace** | **S** |
| **Pearl Dace** | **S** |
| **Rainbow Trout** | **S** |
| **river darter** | **S** |
| **river redhorse** | **S** |
| **silver lamprey** | **S** |
| **Smallmouth Bass** | **S** |
| **Trout-Perch** | **S** |
| **banded killifish** | **T** |
| **Bigmouth Shiner** | **T** |
| **Brook Stickleback** | **T** |
| **Central Mudminnow** | **T** |
| **central stoneroller** | **T** |
| **Creek Chub** | **T** |
| **emerald shiner** | **T** |
| **Golden Shiner** | **T** |
| **Green Sunfish** | **T** |
| **hybrid sunfish** | **T** |
| **quillback** | **T** |
| **river carpsucker** | **T** |
| **silver chub** | **T** |
| **stonecat** | **T** |
| **white bass** | **T** |
| **Yellow Bullhead** | **T** |

**Table 110:** List of chloride tolerance value assigned to various macroinvertebrate species based on MIBI, 2012.



| Macroinvertebrate Taxa | Chloride Tolerance |
|---|---|
| S = Sensitive T = Tolerant | |
| Ablabesmyia | T |
| Acentrella | T |
| Agabus | T |
| Belostoma flumineum | T |
| Calopterygidae | T |
| Ceratopogonidae | T |
| Ceratopsyche | T |
| Chaetocladius | T |
| Cheumatopsyche | T |
| Chimarra | T |
| Chironomus | T |
| Coenagrionidae | T |
| Conchapelopia | T |
| Cryptochironomus | T |
| Dasyhelea | T |
| Dolichopodidae | T |
| Dytiscidae | T |
| Empididae | T |
| Enallagma | T |
| Endochironomus | T |
| Fossaria | T |
| Glyptotendipes | T |
| Heptageniidae | T |
| Hetaerina | T |
| Hirudinea | T |
| Hyalella | T |
| Hydra | T |
| Hydropsychidae | T |
| Hydroptilidae | T |
| Iswaeon | T |
| Leptoceridae | T |
| Planorbidae | T |
| Planorbula | T |
| Polypedilum | T |
| Psychodidae | T |
| Rhagovelia | T |
| Rheotanytarsus | T |
| Sciomyzidae | T |
| Stenelmis | T |
| Stenochironomus | T |
| Stratiomys | T |
| Tanypodinae | T |
| Thienemanniella | T |
| Trichoptera | T |
| Tricorythodes | T |
| Tropisternus | T |
| Valvata | T |
| Zavreliella | T |
| Acricotopus | S |
| Aedes | S |
| Basiaeschna | S |
| Boyeria vinosa | S |
| Brachycentridae | S |
| Campeloma | S |
| Capniidae | S |
| Chimarra socia | S |
| Choroterpes | S |
| Coptotomus | S |
| Curculionidae | S |
| Dineutus | S |
| Diplocladius | S |
| Ectopria | S |
| Eurylophella | S |
| Gerris | S |
| Glossiphoniidae | S |
| Goera | S |
| Hagenius | S |
| Hemiptera | S |
| Neurocordulia | S |
| Ophiogomphus | S |
| Pagastiella | S |
| Paracricotopus | S |
| Paragnetina media | S |
| Phryganea | S |
| Probezzia | S |
| Pseudosuccinea | S |
| Synorthocladius | S |
| Triaenodes | S |
| Uenoidae | S |
| Xylotopus | S |
| Xylotopus par | S |

# APPENDIX C – Connectivity Analysis of Impaired Streams in the St. Louis River Watershed



## Connectivity Analysis on St Louis River Watershed Impaired Streams

The following maps analyze the connectivity of the impaired reaches in the St Louis River Watershed. Each map reviews a different impaired reach, locating its culvert and bridge crossings. These features where identified and measured using St Louis County pictometry and aerial imagery.

Culverts were then evaluated by comparing their measured width with a predicted bankful width. Bankful width was predicted by determining drainage area using Stream Stats(http://water.usgs.gov/osw/streamstats/minnesota.html) and referencing a regional curve (See Appendix?). Culverts under 75% of the bankful width were identified as 'Undersized' and over 125% as 'Oversized.' Culverts between 75% and 125% were identified as 'Correctly Sized.'

78 culverts were located in all of the impaired reaches. According to our designated parameters, 20 of these were undersized, 35 were correctly sized, and 18 were oversized. 5 were left undefined because they were not visible or measurable using the pictometry and aerial photography.

All bridges, including pedestrian and vehicle bridges, were located on the impaired reaches are marked using black dots. 80 bridges were identified for all of the impaired reaches.

**St Louis River Watershed Impairments - Connectivity Summary**

| **Impairments** | | **Bridges** | **Culverts** | | | | Total | Crossings | % Correct |
|---|---|---|---|---|---|---|---|---|---|
| Name | Length (miles) | | Undersized | Correctly Sized | Oversized | Undefined | Crossings | Per Mile | Culverts |
| St. Louis River | 6.9 | 1 | 0 | 0 | 0 | 0 | 1 | 0.1 | NA |
| Elbow Creek | 4.1 | 0 | 2 | 2 | 0 | 0 | 4 | 1.0 | 50% |
| West Two River | 6.1 | 1 | 1 | 0 | 0 | 0 | 2 | 0.3 | 0% |
| Manganika Creek | 1.1 | 0 | 1 | 0 | 0 | 0 | 1 | 0.9 | 0% |
| trib to McQuade Lake | 8.2 | 4 | 0 | 2 | 0 | 0 | 6 | 0.7 | 100% |
| Swan River | 19.5 | 4 | 0 | 0 | 0 | 0 | 4 | 0.2 | NA |
| Elbow Creek | 6.9 | 2 | 2 | 3 | 0 | 0 | 7 | 1.0 | 60% |
| Embarrass River | 36.9 | 16 | 1 | 0 | 0 | 0 | 17 | 0.5 | 0% |
| Sand Creek | 2.5 | 0 | 0 | 0 | 0 | 0 | 0 | 0.0 | NA |
| Vaara Creek | 6.3 | 0 | 0 | 2 | 0 | 0 | 2 | 0.3 | 100% |
| Stony Creek | 5.8 | 0 | 1 | 0 | 0 | 0 | 1 | 0.2 | 0% |
| Unnamed Creek | 6.2 | 0 | 0 | 0 | 1 | 1 | 2 | 0.3 | 0% |
| Skunk Creek | 46.5 | 0 | 1 | 1 | 1 | 0 | 3 | 0.1 | 33% |
| Paleface Creek | 4.6 | 3 | 0 | 0 | 0 | 0 | 3 | 0.6 | NA |
| Ely Creek | 4.7 | 3 | 3 | 2 | 0 | 0 | 8 | 1.7 | 40% |
| Water Hen Creek | 4.2 | 3 | 0 | 0 | 0 | 0 | 3 | 0.7 | NA |
| Water Hen Creek | 3.7 | 4 | 0 | 0 | 0 | 0 | 4 | 1.1 | NA |
| Spring Mine Creek | 2.1 | 1 | 0 | 1 | 0 | 2 | 4 | 1.9 | 100% |
| Miller Creek | 8.6 | 14 | 6 | 7 | 7 | 2 | 36 | 4.2 | 35% |
| Kingsbury Creek | 6.7 | 14 | 1 | 6 | 6 | 0 | 27 | 4.0 | 46% |
| Otter Creek | 6.0 | 4 | 0 | 1 | 0 | 0 | 5 | 0.8 | 100% |
| East Swan Creek | 5.9 | 0 | 0 | 4 | 0 | 0 | 4 | 0.7 | 100% |
| Little Swan Creek | 7.2 | 3 | 0 | 1 | 2 | 0 | 6 | 0.8 | 33% |
| Wyman Creek | 9.3 | 3 | 1 | 3 | 1 | 0 | 8 | 0.9 | 60% |
| | **Total** | **80** | **20** | **35** | **18** | **5** | | | |
| | | | **78** | | | | | | |