Elm Creek Watershed Stressor Identification Report

A study of local stressors limiting the biotic communities in the Elm Creek Watershed

Minnesota Pollution Control Agency

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Key Terms & Abbreviations

AUID – Assessment Unit Identification

BMP – Best Management Practice

CADDIS – Causal Analysis/Diagnosis Decision Information System

DELT – Deformities, Eroded fins, Lesions or Tumors

DO – Dissolved Oxygen

DBS – Deposited and Bedded Sediments

DNR – Minnesota Department of Natural Resources

ECS – Ecological Classification System

EPA – United States Environmental Protection Agency

EPT – Ephemeroptera, Plecoptera, Trichoptera

FBI – Family Biotic Index

F-IBI – Fish Index of Biological Integrity

IBI – Index of Biological Integrity

M-IBI – Macroinvertebrate Index of Biological Integrity

MPCA – Minnesota Pollution Control Agency

MSHA – Minnesota Stream Habitat Assessment

MUSA – Metropolitan Urban Service Area

PLS – Public Land Surveys

SID – Stressor Identification

SOE – Strength of Evidence

STORET – STOrage and RETrieval Systems

TAC – Technical Advisory Committee

TALU – Tiered Aquatic Life Uses

TMDL – Total Maximum Daily Load

TP – Total Phosphorus

TSS – Total Suspended Solids

USDA – United States Department of Agriculture

USGS – United States Geological Survey

WRAPS – Watershed Restoration and Protection Strategy

Executive Summary

Over the past few years, the Minnesota Pollution Control Agency (MPCA) has substantially increased the use of biological monitoring and assessment as a means to determine and report the condition of the state's rivers and streams. This basic approach is to examine fish and aquatic macroinvertebrate communities and related habitat conditions at multiple sites throughout a major watershed. From these data, an Index of Biological Integrity (IBI) score can be developed, which provides a measure of overall community health. If biological impairments are found, stressors to the aquatic community must be identified.

Stressor identification (SID) is a formal and rigorous process that identifies stressors causing biological impairment of aquatic ecosystems and provides a structure for organizing the scientific evidence supporting the conclusions (Cormier et al. 2000). In simpler terms, it is the process of identifying the major factors causing harm to aquatic life. The SID is a key component of the major watershed restoration and protection projects being carried out under Minnesota's Clean Water Legacy Act.

Following the 2010 watershed assessment study conducted in by the MPCA in the Elm Creek Watershed, five stream reaches were identified as biologically impaired based on low fish and invertebrate scores using an IBI method. Concurrent to this biological monitoring and assessment work, a wide range of physical and chemical data have been collected throughout the Elm Creek Watershed to inform the processes of watershed management and Total Maximum Daily Load (TMDL) development.

This report summarizes the SID work in the Elm Creek Watershed that has been conducted to identify the likely causative stressors for the existing biological impairments. Stressors have been identified by comparing the structure of the biological assemblages to the relative occurrence of likely stressors known to limit biological integrity in similar stream-types. Throughout the report, potential stressors for fish and invertebrate assemblages are described using the strength of evidence (SOE) approach. This report is organized into five sections section to 1) provide an introduction to the biological assessment and SID processes, 2) provide a summary of the existing data that describe the Elm Creek Watershed and the corresponding biological impairments, 3) describe the potential stressors initially evaluated as potential causes of the biological impairments, 4) describe the likely stressors causing the biological impairments and 5) summarize future management and monitoring recommendations. Throughout this document, stressors are organized by Candidate Cause and discussed relative to the corresponding stream reaches.

After examining many candidate causes for the biological impairments, the following stressors were identified as probable causes of stress to aquatic life:

- Altered Hydrology
- Physical Habitat Alteration
- Excess Sediments
- Excess Phosphorus
- Low Dissolved Oxygen (DO)

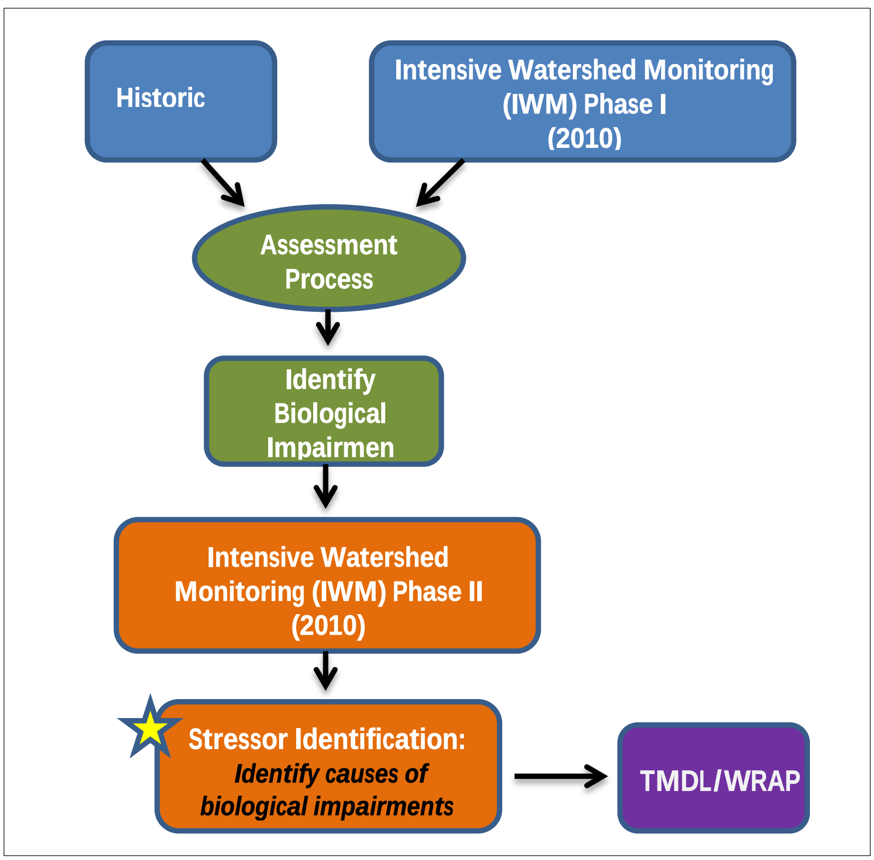
1. Introduction

1.1. Monitoring and Assessment

Water quality and biological monitoring in the Elm Creek Watershed have been ongoing for over 30 years. As part of the MPCA's Intensive Watershed Monitoring (IWM) approach, monitoring activities increased in rigor and intensity during 2010, and focused more on biological monitoring (fish and macroinvertebrates) as a means of assessing stream health. The data collected during this period, as well as historic data obtained prior to 2010, were used to identify stream reaches that were not supporting healthy fish and macroinvertebrate assemblages (Figure 1.1).

Once a biological impairment is discovered, the next step is to identify the source(s) of stress on the biological community. A SID analysis is a step-by-step approach for identifying probable causes of impairment in a particular system. Completion of the SID process does not result in a finished TMDL study. The product of the SID process is the identification of the stressor(s) for which the TMDL may be developed. In other words, the SID process may help investigators nail down 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 restore the impaired condition.

Figure 1.1. Process map of IWM, Assessment, Stressor Identification, and TMDL processes.



1.2. Stressor Identification Process

The MPCA follows the United States Environmental Protection Agency's (EPA) process of identifying stressors that cause biological impairment, which has been used to develop the MPCA's guidance to the SID (Cormier et al. 2000; MPCA 2008). The EPA has also developed an updated, interactive web-based tool, the Causal Analysis/Diagnosis Decision Information System (CADDIS; EPA 2010). This system provides an enormous amount of information designed to guide and assist investigators through the process of the SID. Additional information on the SID process using CADDIS can be found here: http://www.epa.gov/caddis/

A SID is a key component of the major watershed restoration and protection projects being carried out under Minnesota's Clean Water Legacy Act. A SID draws upon a broad variety of disciplines and applications, such as aquatic ecology, geology, geomorphology, chemistry, land-use analysis, and toxicology. A conceptual model showing the steps in the SID process is shown in Figure 1.2. Through a review of available data, stressor scenarios are developed that aim to characterize the biological impairment, the cause, and the sources/pathways of the various stressors.

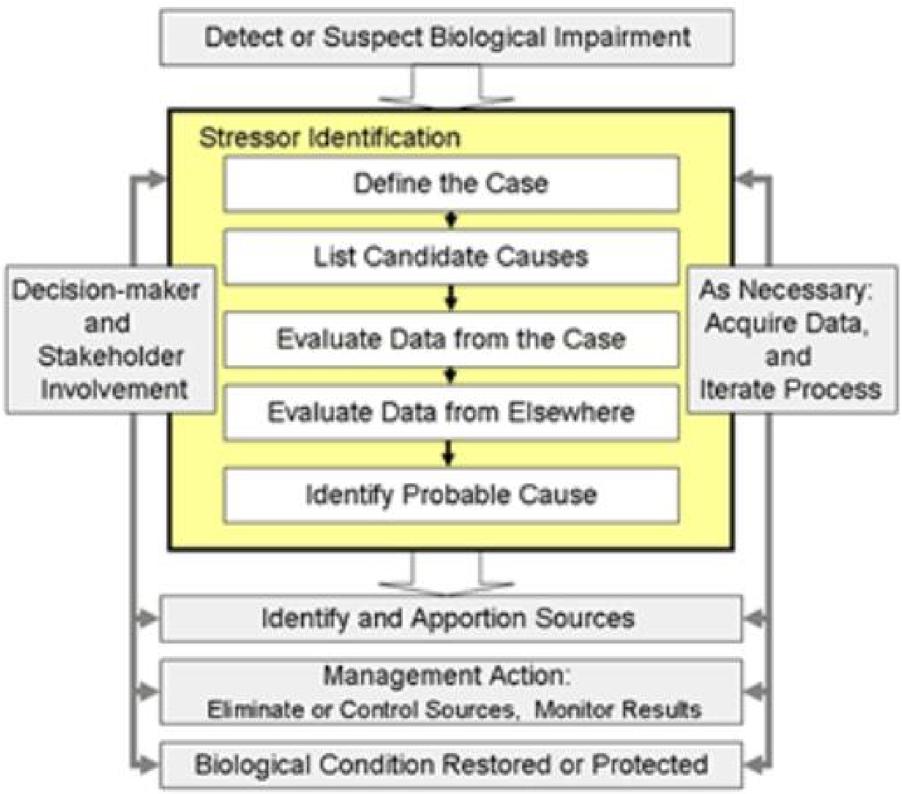


Figure 1.2. Conceptual model of the SID process (Cormier et al. 2000)

The SOE analysis is used to evaluate the data for candidate causes of stress to biological communities. The relationship between stressor and biological response are evaluated by considering the degree to which the available evidence supports or weakens the case for a candidate cause. Typically, much of the information used in the SOE analysis is from the study watershed (i.e., data from the case). However, evidence from other case studies and the scientific literature is also used in the SID process (i.e., data from elsewhere).

Developed by the EPA, a standard scoring system is used to tabulate the results of the SOE analysis for the available evidence (Table A1). A narrative description of how the scores were obtained from the evidence should be discussed as well. The SOE table allows for organization of all of the evidence, provides a checklist to ensure each type has been carefully evaluated and offers transparency to the determination process.

The existence of multiple lines of evidence that support or weaken the case for a candidate cause generally increases confidence in the decision for a candidate cause. The scoring scale for evaluating each type of evidence in support of or against a stressor is shown in Table A2. Additionally, confidence in the results depends on the quantity and quality of data available to the SID process. In some cases, additional data collection may be necessary to accurately identify the stressor(s) causing impairment. Additional detail on the various types of evidence and interpretation of findings can be found here: http://www.epa.gov/caddis/si_step_scores.html

Throughout this report, the causal impacts of different stressors on the structure of biological communities in the Elm Creek Watershed were assessed based on the combined SOE for both stressor exposure and biological response. Stressor exposure was evaluated based on the magnitude and consistency of trends in land use, hydrology, water quality, and stream habitat. Biological impairment was evaluated based observations of divergence of the structure of biological assemblages in the Elm Creek system away from reference conditions (as measured by an IBI). Causal linkage of stressors to the biological response was based on an assessment of the divergence of individual biotic metrics in the Elm Creek system away from sites in similar stream classifications that are meeting the biocriteria (referred to as “unimpaired sites” throughout the document). Individual metrics were considered to have diverged from the unimpaired conditions if the site-specific response of the metric was outside of interquartile range (i.e., between the 25^{th} and 75^{th} percentile) of responses observed across unimpaired sites. Details of the IBI and metric responses are described in Section 2.3.

1.3. Common Stream Stressors

The five major elements of a healthy stream system are stream connections, hydrology, stream channel assessment, water chemistry and stream biology. If one or more of the components are unbalanced, the stream ecosystem may fail to function properly and is listed as an impaired water body. Table 1.1 lists the common stream stressors to biology relative to each of the major stream health categories.

Table 1.1. Common streams stressors to biology (i.e., fish and macroinvertebrates)

Stream Health	Stressor(s)	Link to Biology
Stream Connections	**Loss of Connectivity** · Dams and culverts · Lack of wooded riparian cover · Lack of naturally connected habitats/ causing fragmented habitats	Fish and macroinvertebrates cannot freely move throughout system. Stream temperatures also become elevated due to lack of shade.
Hydrology	**Altered Hydrology** **Loss of habitat due to channelization** **Elevated Levels of TSS** · Channelization · Peak discharge (flashy) · Transport of chemicals	Unstable flow regime within the stream can cause a lack of habitat, unstable stream banks, filling of pools and riffle habitat, and affect the fate and transport of chemicals.
Stream Channel Assessment	**Loss of Habitat due to excess sediment** **Elevated levels of TSS** · Loss of dimension/pattern/profile · Bank erosion from instability · Loss of riffles due to accumulation of fine sediment · Increased turbidity and or TSS	Habitat is degraded due to excess sediment moving through system. There is a loss of clean rock substrate from embeddedness of fine material and a loss of intolerant species.
Water Chemistry	**Low Dissolved Oxygen Concentrations** **Elevated levels of Nutrients** · Increased nutrients from human influence · Widely variable DO levels during the daily cycle · Increased algal and or periphyton growth in stream · Increased nonpoint pollution from urban and agricultural practices · Increased point source pollution from urban treatment facilities	There is a loss of intolerant species and a loss of diversity of species, which tends to favor species that can breathe air or survive under low DO conditions. Biology tends to be dominated by a few tolerant species.
Stream Biology	Fish and macroinvertebrate communities are affected by all of the above listed stressors	If one or more of the above stressors are affecting the fish and macroinvertebrate community, the IBI scores will not meet expectations and the stream will be listed as impaired.

1.4. Report Format

This report is structured to provide continuity between the watershed characteristics, subsequent physical, chemical and biological stressors and the associated changes in assemblage structure of for fish and macroinvertebrates. Section 2 provides an historical overview of the changes in land use, precipitation patterns and jurisdiction throughout the Elm Creek Watershed. Section 3 describes the range of stressors; candidate causes and scope of data that were used identify potential stressors to the biotic assemblages in each assessment unit identifier (AUID). Section 4 describes the specific data sets and trends that were used to evaluated different stressors to the biotic assemblages as well as the evidence for causal relationships between different stressors and biotic responses. Across all sections, data are referred to with respect to AUIDs and stressors responses are grouped according to stressor type (e.g., evidence for potential impacts of altered hydrology to biotic communities are described in the same section, according to individual AUIDs).

2. Overview of Elm Creek Watershed

2.1. Background

The Elm Creek Watershed is located in the northwestern Twin Cities metropolitan area in Hennepin County, Minnesota. The management boundary of the Elm Creek Watershed extends from the Crow River in the northwest, to the Mississippi River in the northeast to the hydrologic border of Elm Creek to the south (Figure 2.1). However, not all of the land area within the watershed management boundary is hydrologically connected. The majority of land (106 square miles) within the management boundary is hydrologically connected and drains to Elm Creek (HUC 07010296). However, two regions within the management boundary are not hydrologically connected and drain directly to the Crow and Mississippi Rivers. For the purposes of this SID study, the term "Elm Creek Watershed" will only be used to describe the hydrologically connected sections of the watershed that drain to the mouth of Elm Creek at the Mississippi River in Champlin, Minnesota.

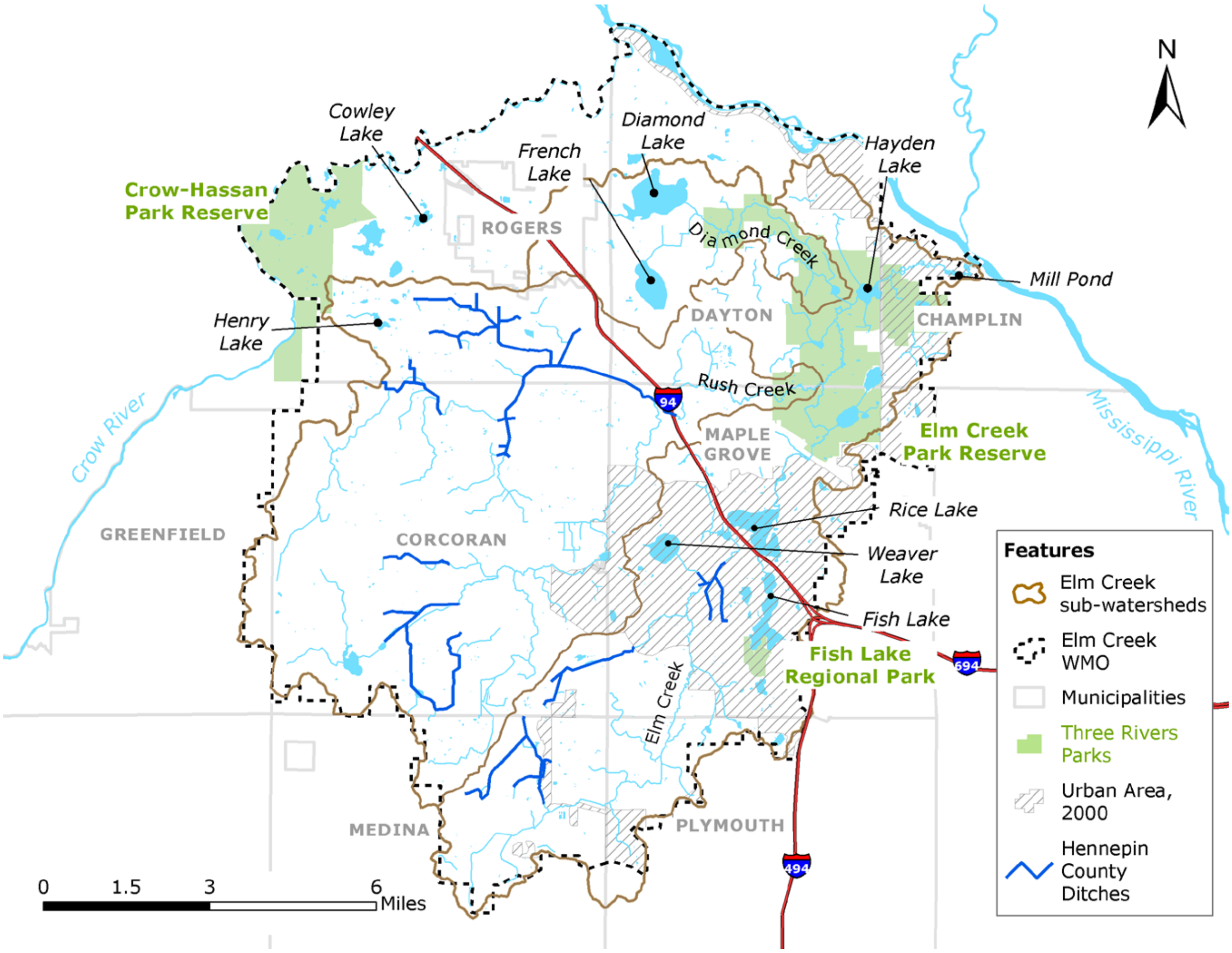


Figure 2.1. Boundaries of the Elm Creek Watershed

Geology and Soils

The current topography in the Elm Creek watershed is a product of the most recent glacial maxima that occurred approximately 10,000 to 12,000 year before present. The Elm Creek Watershed is relatively low gradient, ranging in elevation from a maximum of 332 meters in the headwaters near the city of Medina to a minimum of 252 meters at its outlet to the Mississippi River in the city of Champlin (Figure 1.1 and Figure 2.3). Soils throughout the Elm Creek watershed are dominated by C and D classifications (Figure 2.4); however, A and B soils become more common in downstream areas of the watershed. Classifications of C and D represent soils that are less porous and have lower infiltration rates, while A and B soil types are more porous and have higher infiltration rates (NRCS 2007).

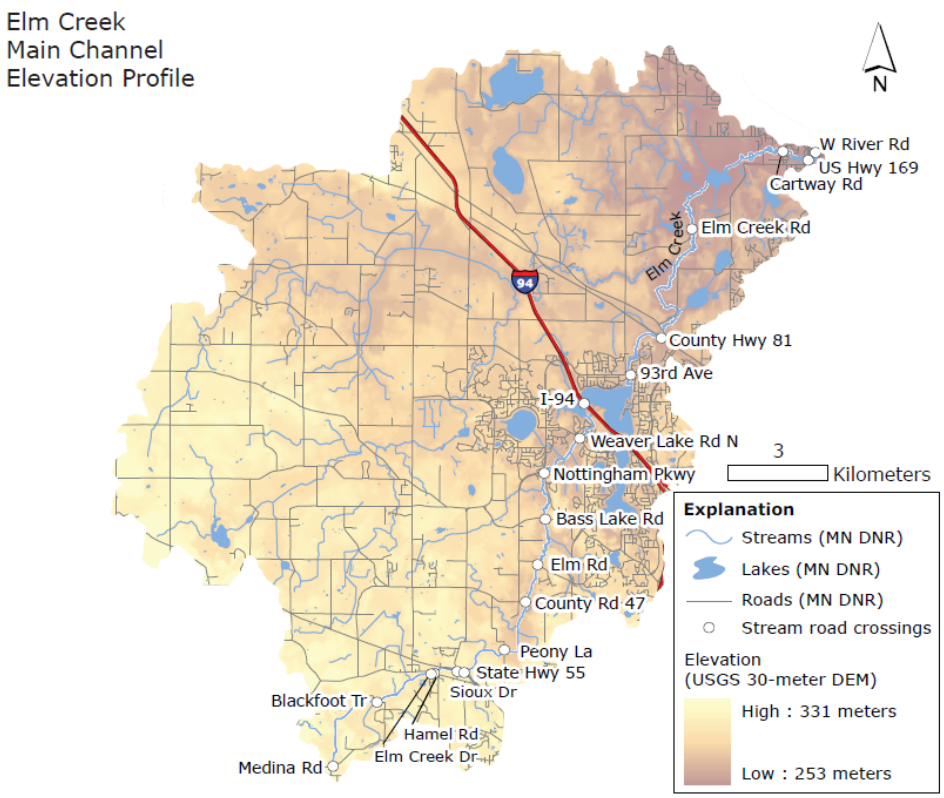


Figure 2.2. Topography and elevation change throughout the Elm Creek Watershed

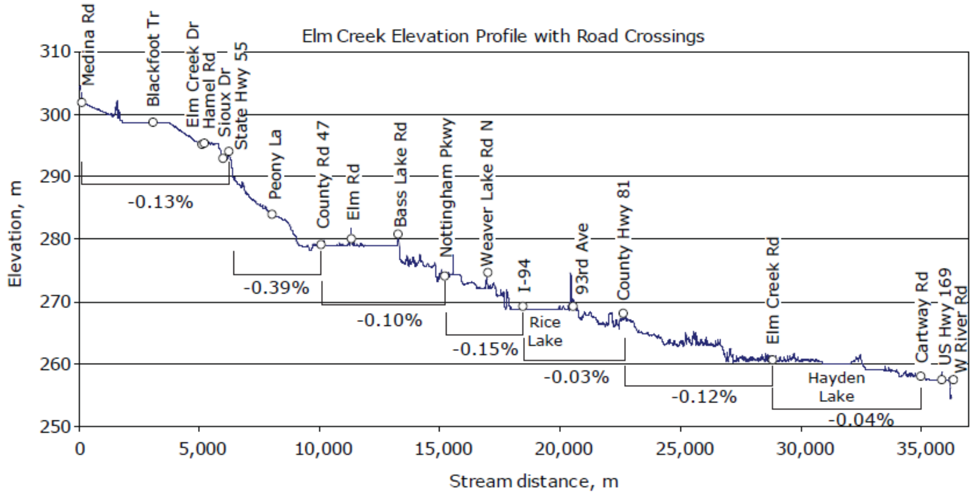


Figure 2.3. Stream channel elevation profile throughout the Elm Creek Watershed

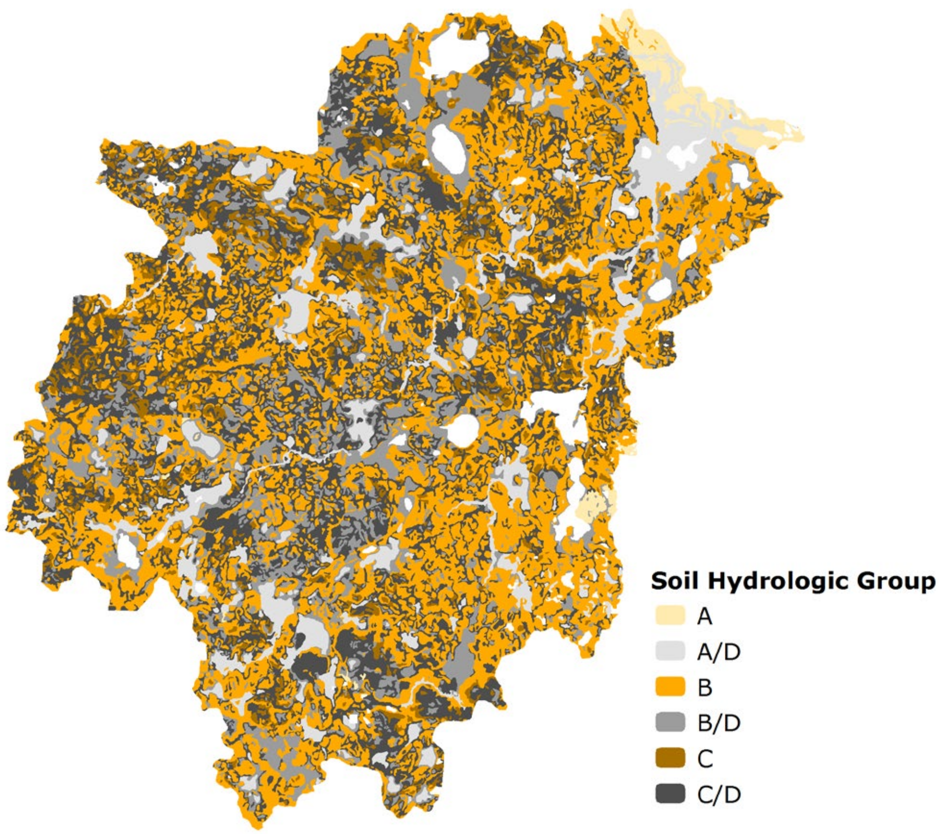


Figure 2.4. Soil type distribution throughout the Elm Creek Watershed based on SURRGO classification

Precipitation and Climate

A series of long-term data sets describing long-term climate and weather conditions exist in and around the Elm Creek Watershed (Figure 2.5). Climate surrounding the Elm Creek Watershed is generally considered temperate, with moderate amounts of snowfall and rainfall and wide seasonal fluctuations in annual temperatures (Table 2.1). Historically, the 100-year 24-hour precipitation event was expected to yield 5.9 inches and most engineering design throughout the basin is based on the TP-40 (Hershfield, 1963). However, precipitation recurrence intervals were recently updated in Atlas 14 (Perica et al. 2013) to account for increased spatial resolution in climatological data and account for any human induced shifts in precipitation patterns. Based on these updates, the 100-year, 24-hr precipitation event in the Elm Creek watershed is now expected to yield 7.9 inches (a 32% increase). However, the Atlas 14 precipitation estimates have only recently become available and have not been incorporated into engineering design and watershed planning work.

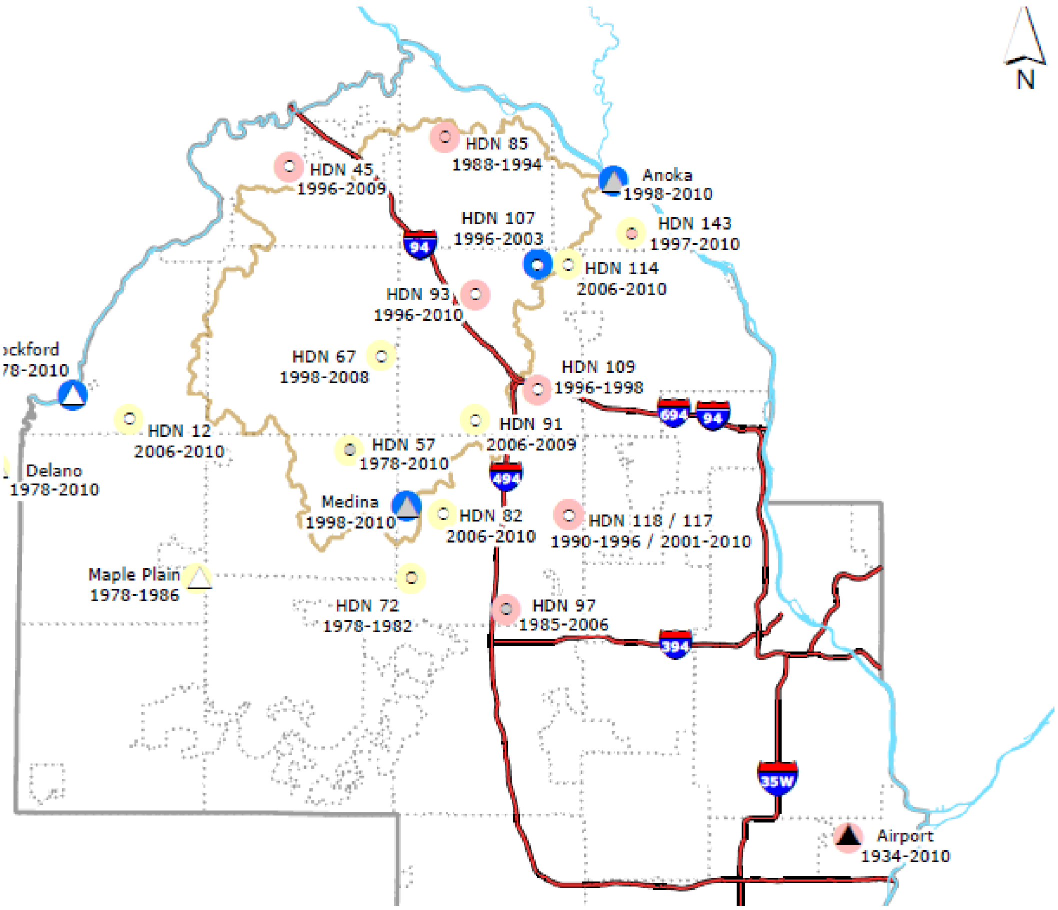


Figure 2.5. Precipitation gage location and period of record throughout the Elm Creek Watershed

Table 2.1. Historical temperature, rainfall and snowfall averages for the Elm Creek watershed. Source: State Climatology Office for the Minneapolis/St. Paul Airport

Month	Average Temp. F°	Precip. inches	Snowfall inches
January	11.8	0.83	9.8
February	17.9	0.85	8.4
March	31	1.60	11.7
April	46.4	2.17	2.8
May	58.5	3.38	0.1
June	68.2	4.17	0
July	73.6	3.55	0
August	70.5	3.40	0
September	60.5	2.89	0
October	48.8	2.01	0.5
November	33.2	1.45	7.9
December	17.9	0.94	9.3
	Annual Average: 44.9	**Total:** 27.24	**Total:** 50.5

Land Use and Historic Land Cover

Current, land use throughout the watershed is comprised primarily of agricultural lands in the headwater areas and is increasingly dominated by low, moderate and high density urban lands in the downstream portions of the watershed (Figure 2.6). Land use in the Elm Creek Watershed has undergone significant change since the 1900s. The Elm Creek Watershed is classified by the EPA as part of the North Center Hardwood forest. Historically, the Elm Creek Watershed was comprised of Oak Savanah with pockets of deciduous forest stands and a high density of wetlands. Starting in the late 1800s and early 1900s, the Elm Creek Watershed was broadly developed for agricultural production, such that by 1980, approximately 52% of the watershed was in agricultural land use. Since the 1980s, the area of urban development has grown by 55% (Figure 2.7).

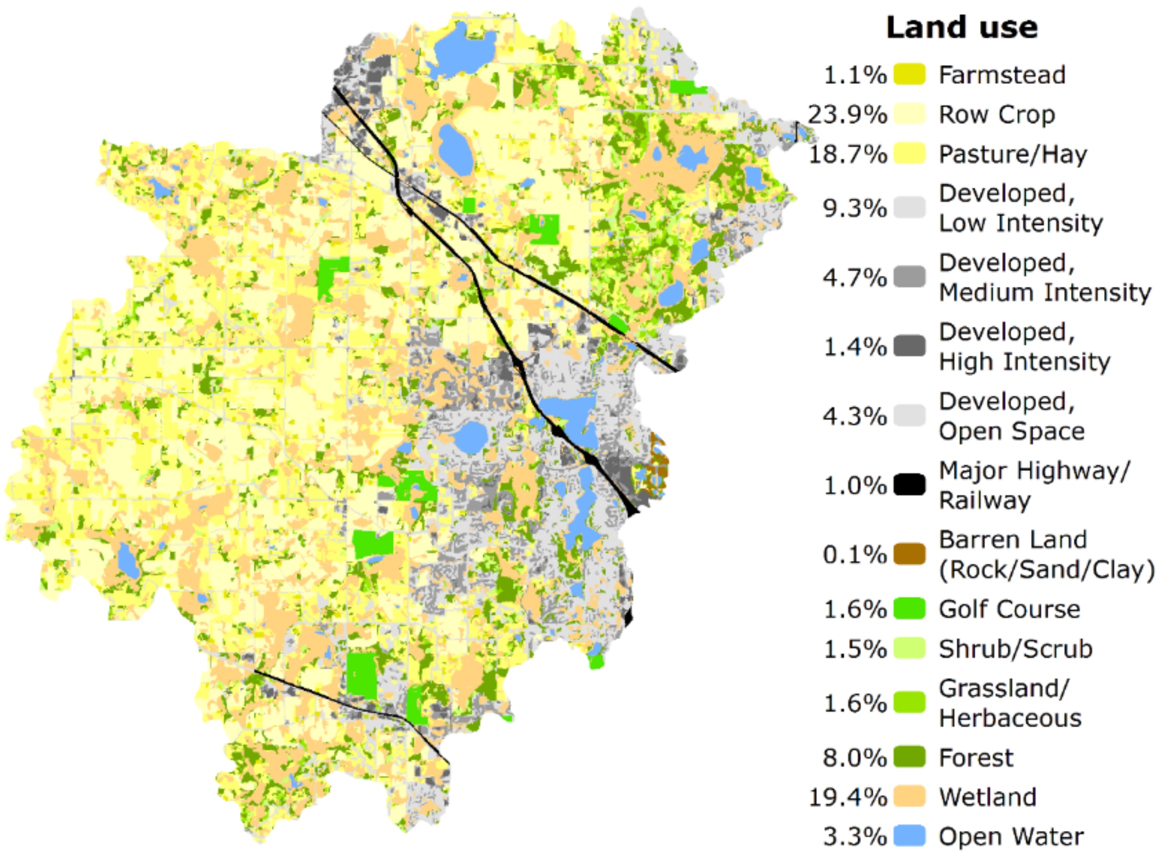


Figure 2.6. Land use distribution throughout the Elm Creek Watershed based a combination of the NLCD (2006), Metropolitan Council Generalized land use (2005) and Minnesota Land Cover Classification System wetlands

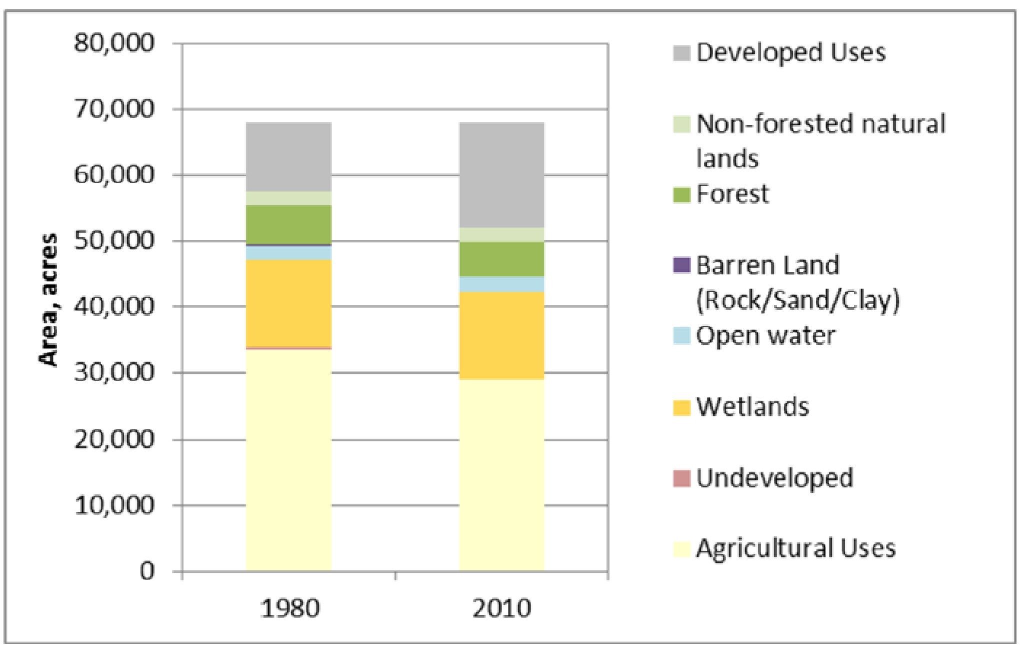


Figure 2.7. Proportional land use change in the Elm Creek Watershed from 1980 to 2010

Urban lands throughout the Elm Creek Watershed represent eight municipalities (Figure 2.8). Most urban areas throughout the Elm Creek Watershed are served by municipal storm sewer systems (Figure 2.8). Similarly, urban development is served by sanitary sewer, except for areas outside of the Metropolitan Urban Service Area (MUSA) boundary. Outside of the MUSA boundary, most stormwater is treated on site (if at all) and wastewater is processed by on-site septic systems (Figure 2.9).

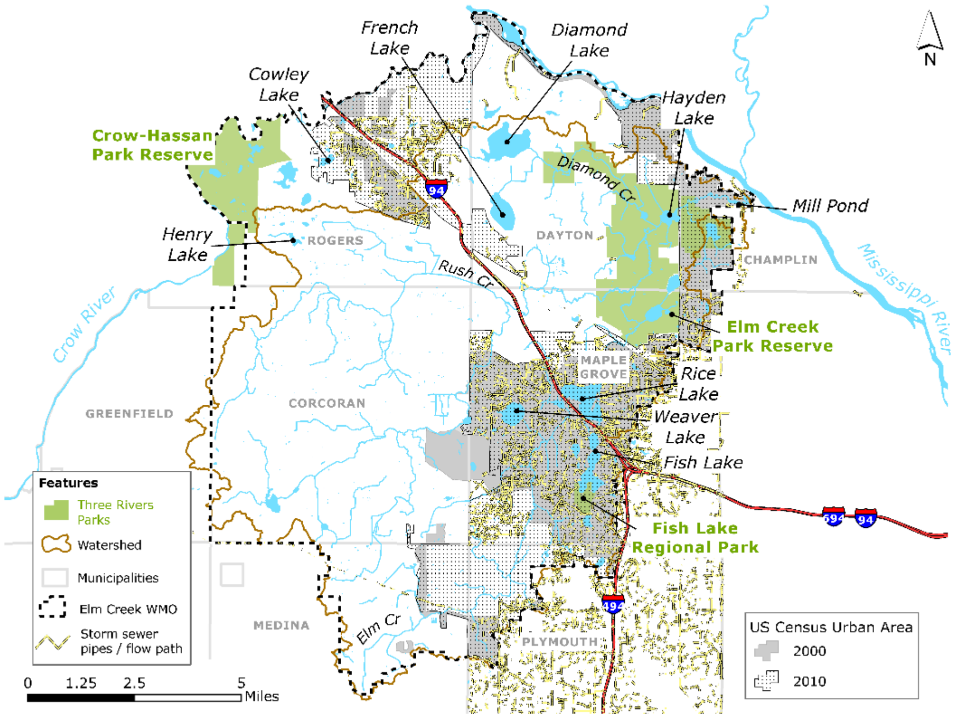


Figure 2.8. Storm sewer system distribution throughout the Elm Creek Watershed

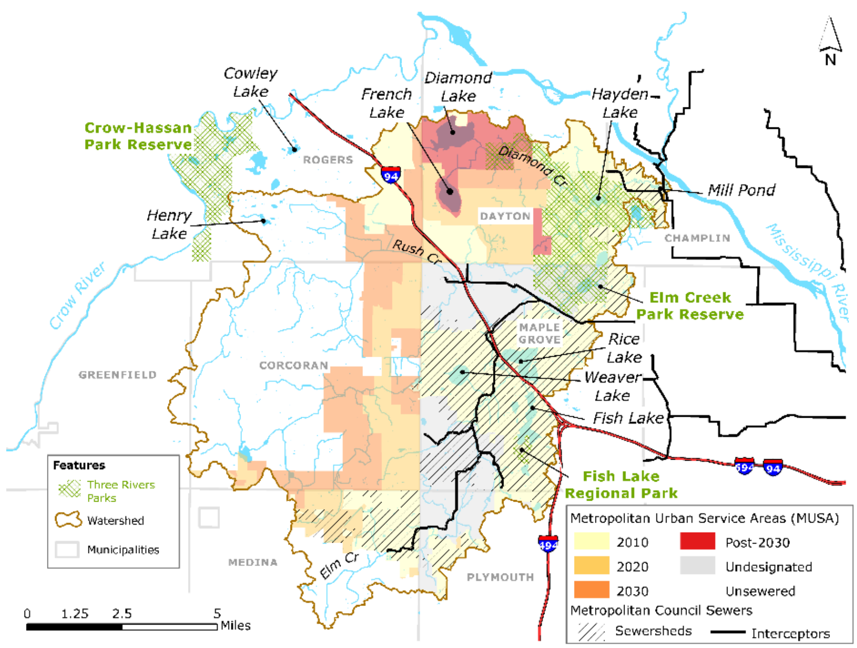


Figure 2.9. Sanitary sewer system coverage throughout the Elm Creek Watershed

Agricultural lands are dominated by row crops (corn and soybeans), but wheat, alfalfa and hay are also common on the landscape (Figure 2.10). Given the extent of C and D soils throughout watershed many agricultural lands have been ditched and tile drained to facilitate increased drainage rates. An inventory of formal regulated ditching has been completed (Figure 2.11). The full extent of ditching and tile drainage is not known. However, based on observations throughout the region, it is likely that areas in which row crop agriculture exists on C and/or D soils, some level of ditching and tile draining also likely exists. In addition to crop agriculture, livestock are also common throughout the watershed, particularly in upstream reaches (Figure 2.12). Livestock are primarily a part of small scale production and hobby farms, but several larger livestock producers are present in the watershed as well.

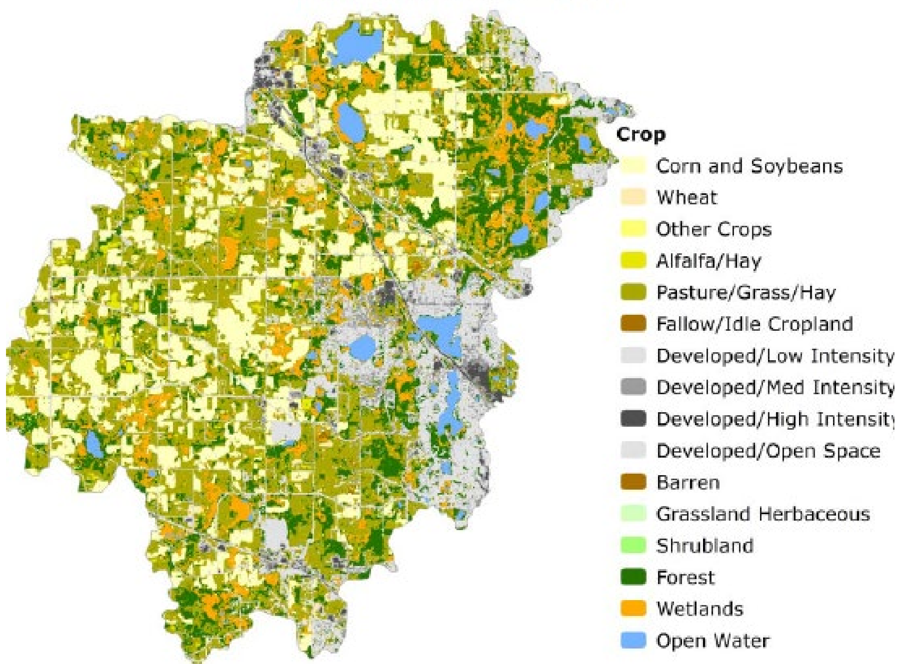


Figure 2.10. Cropland cover classification throughout the Elm Creek Watershed (2006-2011)

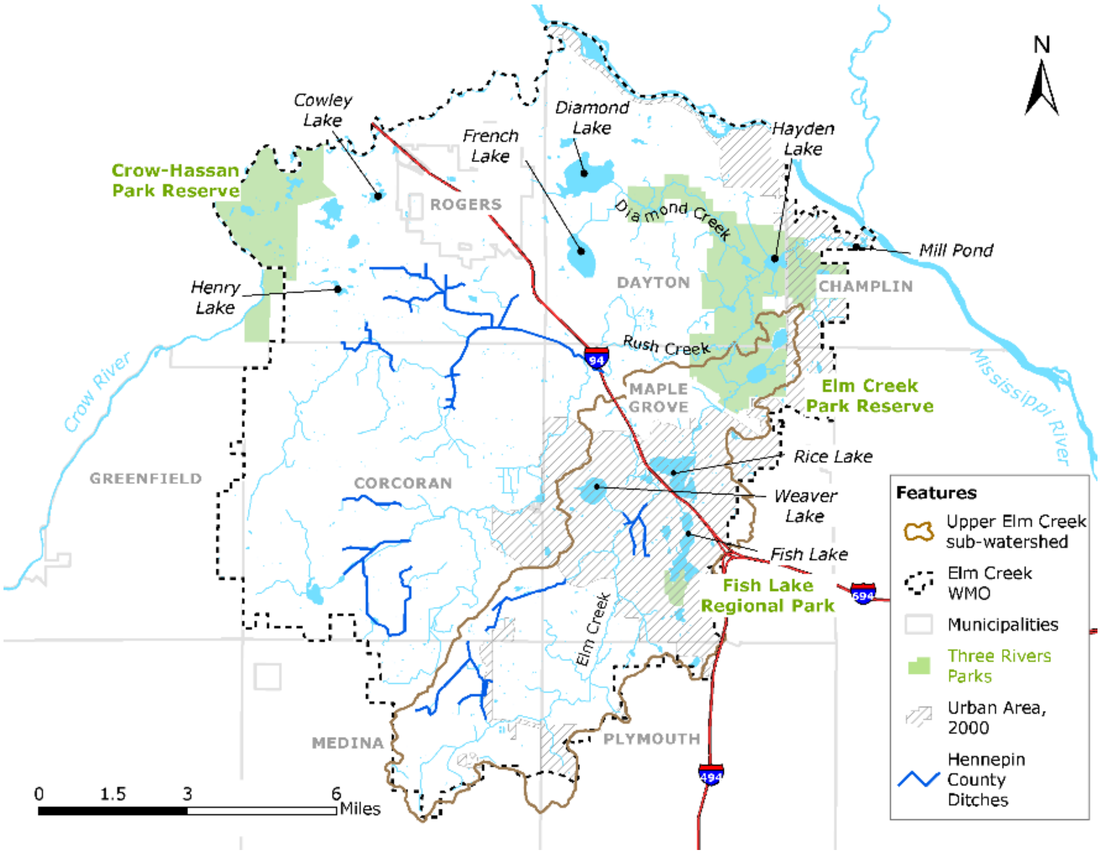


Figure 2.11. Extent and location of agricultural ditching throughout the Elm Creek Watershed

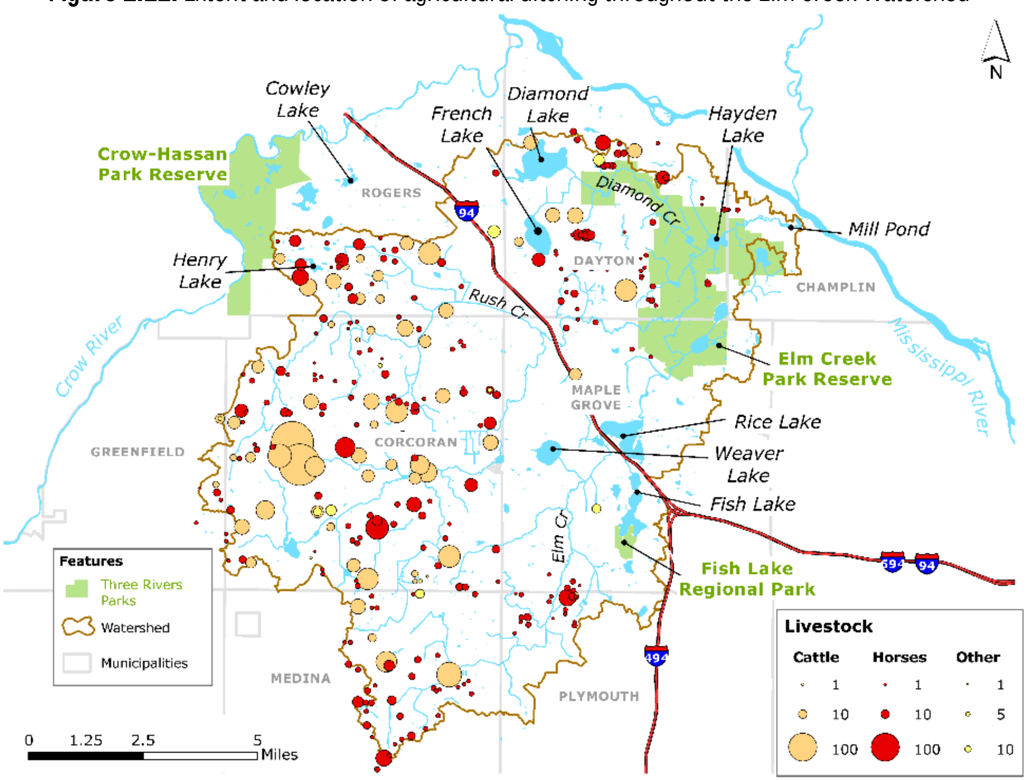


Figure 2.12. Livestock animal unit densities throughout the Elm Creek Watershed

Groundwater Resources

Groundwater within the Elm Creek Watershed is contained within three major aquifers: Prairie Du Chien-Jordan, Franconian-Ironton-Galesville and Mt. Simon-Hinckley. Since the 1990s, ground water extraction has steadily increased, such that approximately 325 million gallons of groundwater are extracted each year from groundwater resources within the Elm Creek Watershed (Figure 2.13). The Prairie du Chien-Jordan aquifer is the primary source of groundwater within Hennepin County, MN.

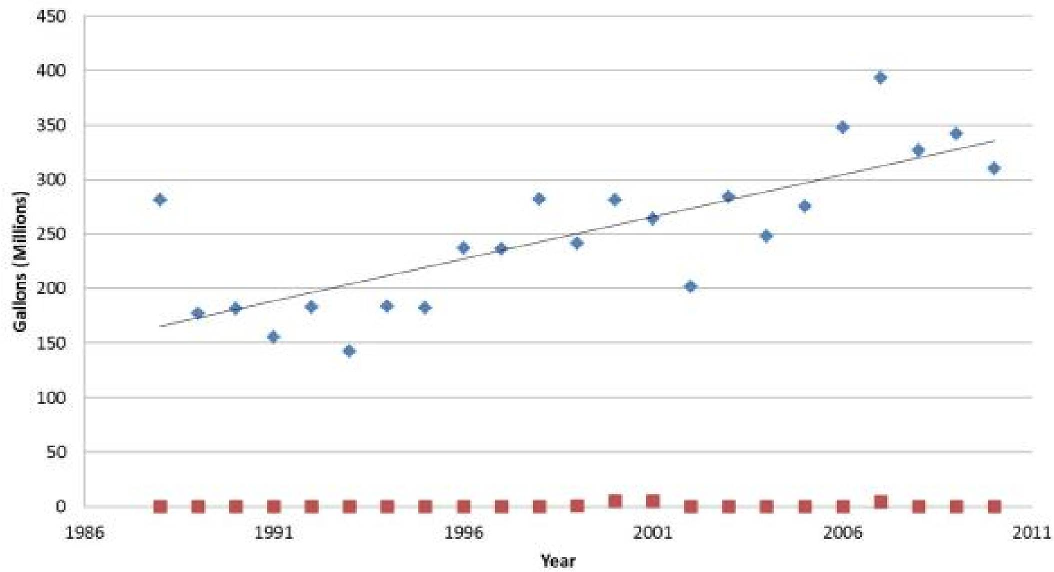


Figure 2.13. Total groundwater (blue diamonds) and surface water (red squares) withdrawals in the Elm Creek Watershed between 1988 and 2010 (from MPCA 2013)

2.1.1 Subwatersheds

The Elm Creek Watershed is functionally comprised of four major sub-basins, Upper Elm Creek, Lower Elm Creek, Rush Creek and Diamond Creek, each of which has unique land use and stream habitat characteristics (Figure 2.14). For regulatory and management purposes, the Elm Creek Watershed is divided into 15 AUID reaches (Figure 2.15). All three digit AUIDs listed within this report are preceded by 07010206- .

Elm Creek is 21 miles long and described as two AUIDs (508 and 577). Functionally, Elm Creek is separated into lower and upper segments which are delineated by the outflow control structure on Rice Lake. Although the headwater reaches are comprised primarily of agricultural lands, the upper Elm Creek Watershed is the most highly urbanized sub-basin throughout the watershed. Most of lower Elm Creek is contained within the Elm Creek Park Reserve (managed by Three Rivers Park District) and much of this sub-basin is dominated by flow through large wetland complexes, and with a smaller area of dense urban land use near the confluence with the Mississippi River. Although hydrologically connected to the Mississippi River, Elm Creek is biologically disconnected by the outlet control structure that forms Mill Pond, just upstream of the confluence.

Diamond Creek is 6 miles long and is described as three AUID (525, 599 and 598). Diamond Creek is fed by both French and Diamond Lakes and comprises the smallest sub-basin throughout the Elm Creek Watershed. Land use in headwater areas of Diamond Creek is dominated by agriculture, while downstream reaches are part of the Elm Creek Park Reserve. Diamond Creek empties into lower Elm Creek inside the three Rivers Park District boundary at the Hayden Lake wetland complex in the city of Dayton.

Rush Creek is comprised of two primary sub-basins, the south fork (5 miles) and main stem (17 miles) and is described by 10 AUIDs (528, 732, 760, 761, 762, 763, 776, 778, 779 and 780). Land use in both the main stem and south fork of Rush Creek is almost entirely occupied by agricultural lands and large wetland complexes, although areas of medium density urban development are found in the downstream reaches of the south fork of Rush Creek. Ditching is common throughout both sub-basins, but the upper main stem has the highest concentration of regulated ditch systems. Rush Creek empties into lower Elm Creek inside the three Rivers Park District boundary in the city of Dayton.

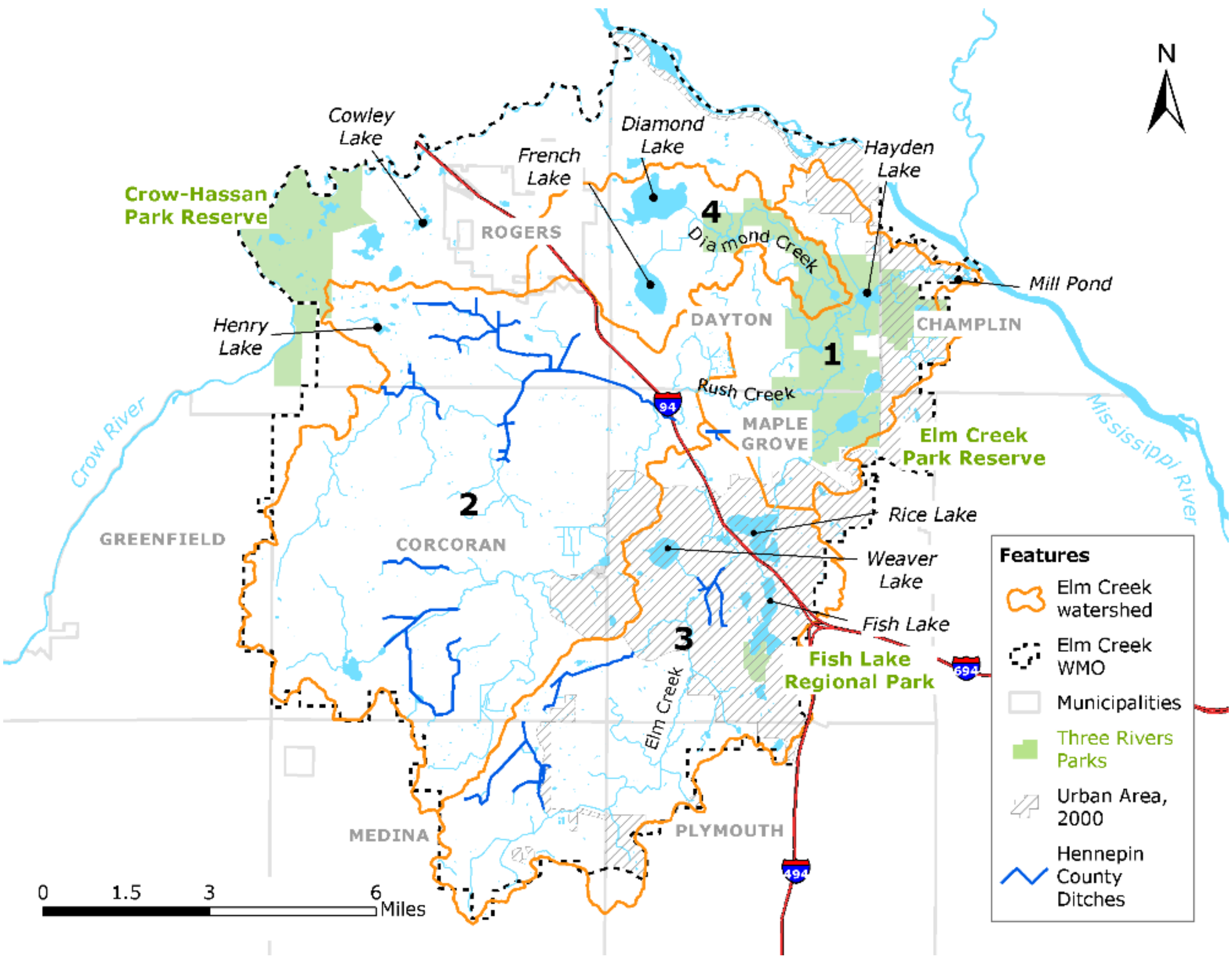


Figure 2.14. Hydrologic sub-basins in the Elm Creek Watershed: 1=Lower Elm Creek; 2=Rush Creek; 3=Upper Elm Creek; 4= Diamond Creek

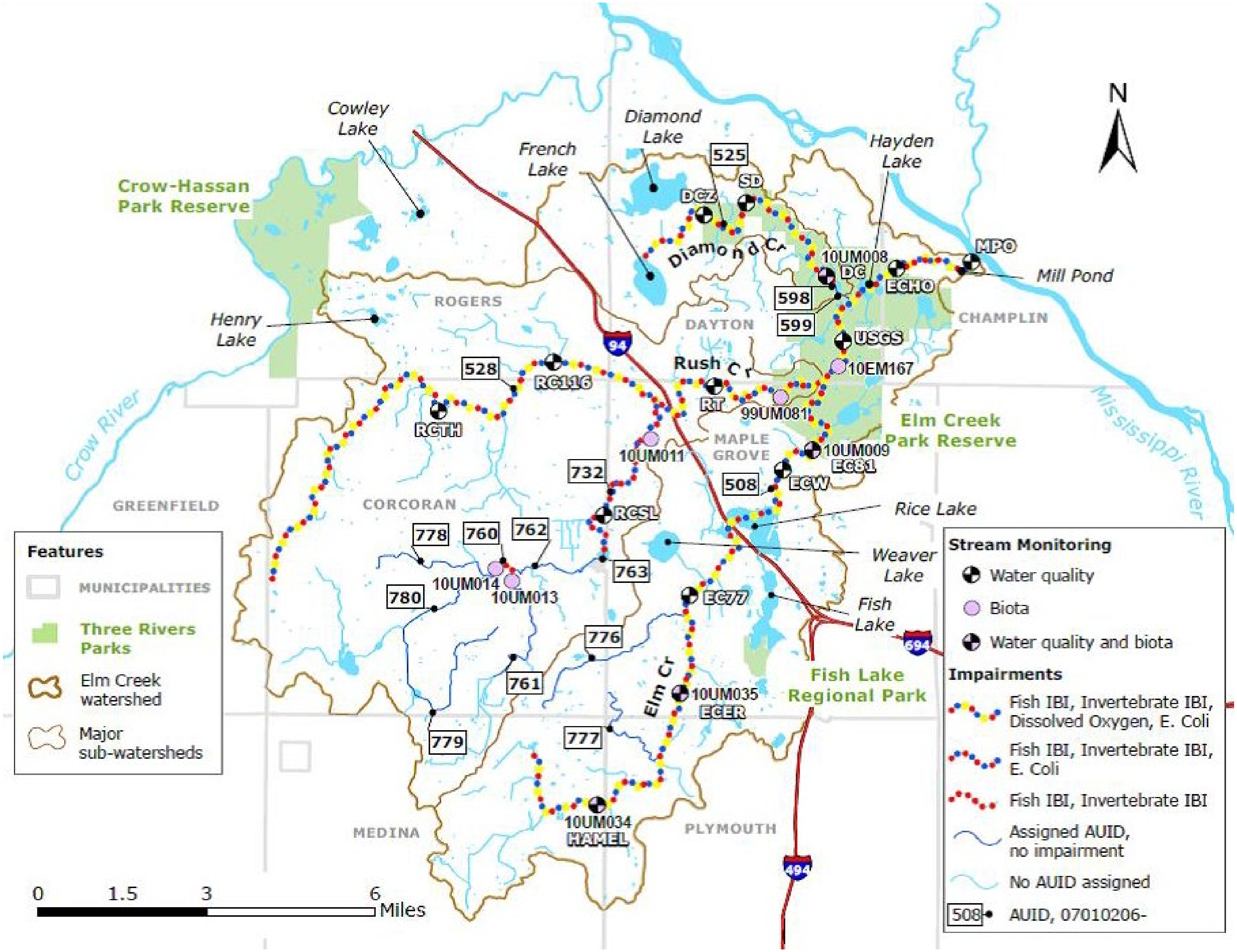


Figure 2.15. Location biotical and chemical monitoring stations in relationship to of AUID currently impaired for fish and invertebrate IBI scores

2.2. Monitoring Overview

A range of physical, chemical and biological monitoring have been conducted to suport the Elm Creek SID (Figure 2.16). Since 1978, different groups have supported the ongoing operation and maintenance of the United States Geological Survey (USGS) guage station on lower Elm Creek. Starting in the 1990s, Hennepin Parks (now Three River Park District), in collaboration with the Elm Creek Watershed Management Commission and regional municipalities have expand this monitoring throughout all sub-basins within the watershed. This collaborative effort has resulted in a water quality and hydrology data collection at different sites for over 25 years. The collective body of this work is summarized in the Elm Creek Surface Water Monitoring Grant report (Ashling et al. 2008) and Elm Creek TMDL – Data Summary Report (Elm Creek Watershed Management Commission, et. al. 2011) supplemented with updated data collected through 2012.

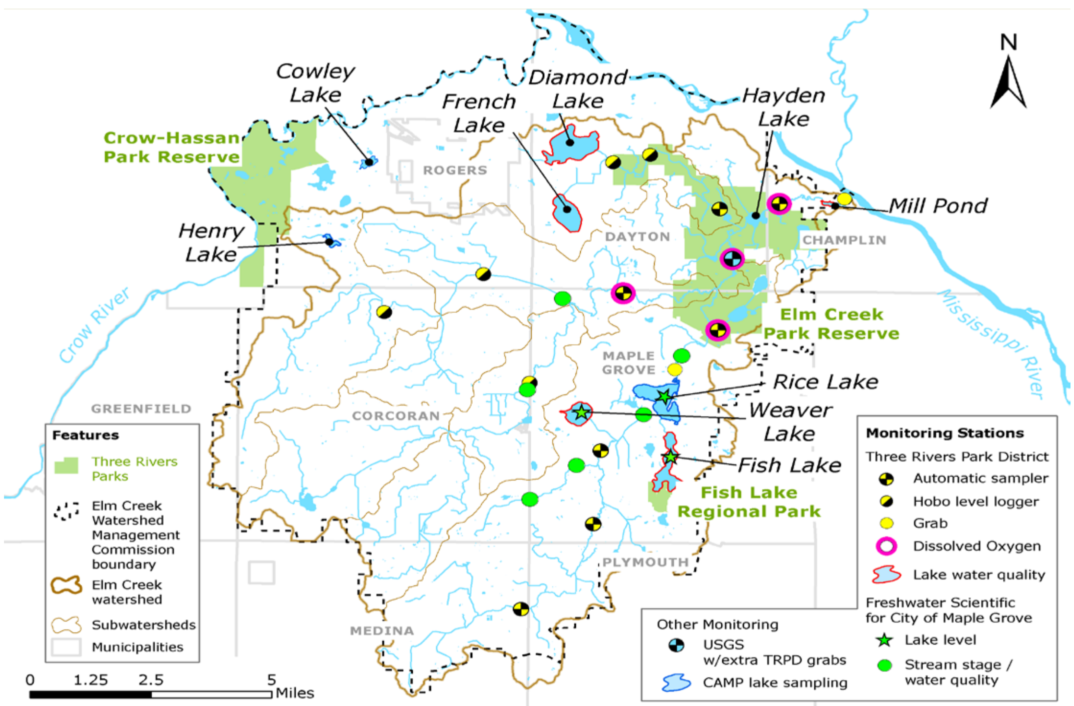


Figure 2.16. Water sampling locations and procedures and cooperating agencies contributing data used in the SID

In addition to this ongoing water quality management work, a range of investigators have assessed the biological communities and physical habitat throughout the Elm Creek watershed. Since 1995, the Minnesota Department of Natural Resources (Schmidt, 1994; Schmidt and Talmage 2001) has surveyed fish communities throughout the watershed. Most recently the MPCA updated the biological monitoring work (and conducted an associated stream habitat assessement) in 2010 as part of the comprehensive watershed monitoring work (MPCA 2013). Over this time period, fish and macroinvertebrate assemblages have been monitored at ten different stations (Figure 2.15). In parrallel, physical habitat throughout the Elm Creek wtaershed was comprehensively inventoried by Hennepin County Conservation District (Dindorf and Miesbauer 2002) hydrological processes were modeled to better understand flooding and erosional processes (Bonestroo 2007). Taken together, these efforts to understand the physical, chemical and biological elements of the Elm Creek Watershed have generated a wealth of data to inform this SID and the associated TMDL analyses

2.3. Summary of Biological Impairments

The approach used to identify biological impairments is primarily based on an assessment of fish and aquatic macroinvertebrates communities and related habitat conditions at sites throughout a watershed. Resident biological communities are subject fluctuations in environmental stressors over space and time in stream ecosystems. As a result, the structure of biological assemblages is commonly utilized to describe a spatially and temporally integrated representation of aquatic ecosystem condition. Using this concept, determinations of impairments to aquatic life can be assessed using an IBI approach. Using this approach, different attributes (i.e., metric) of the sampled assemblages are compared to assemblages from reference sites of similar physical/chemical structure. The IBIs calculated for stream reaches in the Elm Creek watershed were based 10-12 different metrics.

Fish and macroinvertebrates within each AUID were compared to a regionally developed threshold and confidence interval using a weight of evidence approach (Table 2.2 and Table 2.3). Minnesota water quality standards call for the maintenance of a healthy community of aquatic life. The IBI scores provide a measurement tool to assess the health of the aquatic communities. The IBI scores higher than the impairment threshold indicate that the stream reach supports aquatic life. Conversely, scores below the impairment threshold indicate that the stream reach does not support aquatic life. Confidence limits around the impairment threshold help to ascertain where additional information may be considered to help inform the impairment decision. When IBI scores fall within the confidence interval, interpretation and assessment of the waterbody condition involves consideration of potential stressors, and draws upon additional information regarding water chemistry, physical habitat, and land use, etc.

In the Elm Creek Watershed, five AUIDs have been identified as impaired for a lack of biological assemblage (Table 2.4). The purpose of the SID is to interpret the data collected during the biological monitoring and assessment process. Trends in the IBI scores can help to identify causal factors for biological impairments. Across all AUIDs, one macroinvertebrate IBI score was above the regional threshold value and the remaining four IBI scores were between the threshold and lower control limit scores (Figure 2.17). For fish assemblages, all IBI scores were below the regional threshold and four were below the lower confidence limit (Figure 2.18). Details of the general IBI metric response are described below with respect to the individual subwatersheds. Additional details of the IBI response will be discussed in Section 4.

Table 2.2. Fish classes with respective IBI thresholds and upper/lower confidence limits (CL) found in the Elm Creek Watershed

Class	Class Name	IBI Thresholds	Upper CL	Lower CL
5	Northern Streams	50	59	41
6	Northern Headwaters	40	50	30

Table 2.3. Macroinvertebrate classes with respective IBI thresholds and upper/ lower confidence limits (CL) found in the Elm Creek Watershed

Class	Class Name	IBI Thresholds	Upper CL	Lower CL
6	Southern Forest GP	46.8	60.4	33.2

Table 2.4. Biologically impaired AUIDs in the Elm Creek Watershed

			Impairments	
Stream Name	**AUID #**	**Reach Description**	**Biological**	**Water Quality**
Elm Creek	**07010206-508**	Headwaters (Lk Medina 27-0146-00) to Mississippi R	F-IBI: 26 and 24 Invert IBI: 65.13 and 45.13	Dissolved Oxygen; Bacteria
Diamond Creek	**07010206-525**	Headwaters (French Lk 27-0127-00) to Unnamed lk	F-IBI: 19 Invert IBI: 46.78	Dissolved Oxygen; Bacteria
Rush Creek, Main	**07010206-528**	Headwaters to Elm Cr	F-IBI: 30 Invert IBI: 42.63	Dissolved Oxygen; Bacteria
Rush Creek, South Fork	**07010206-732**	Unnamed lk (27-0439-00) to Rush Cr	F-IBI: 20 Invert IBI: 31.31	Dissolved Oxygen; Bacteria
Rush Creek, South Fork	**07010206-760**	Unnamed ditch to County Ditch 16	F-IBI: 1 Invert IBI: 37.90	Bacteria

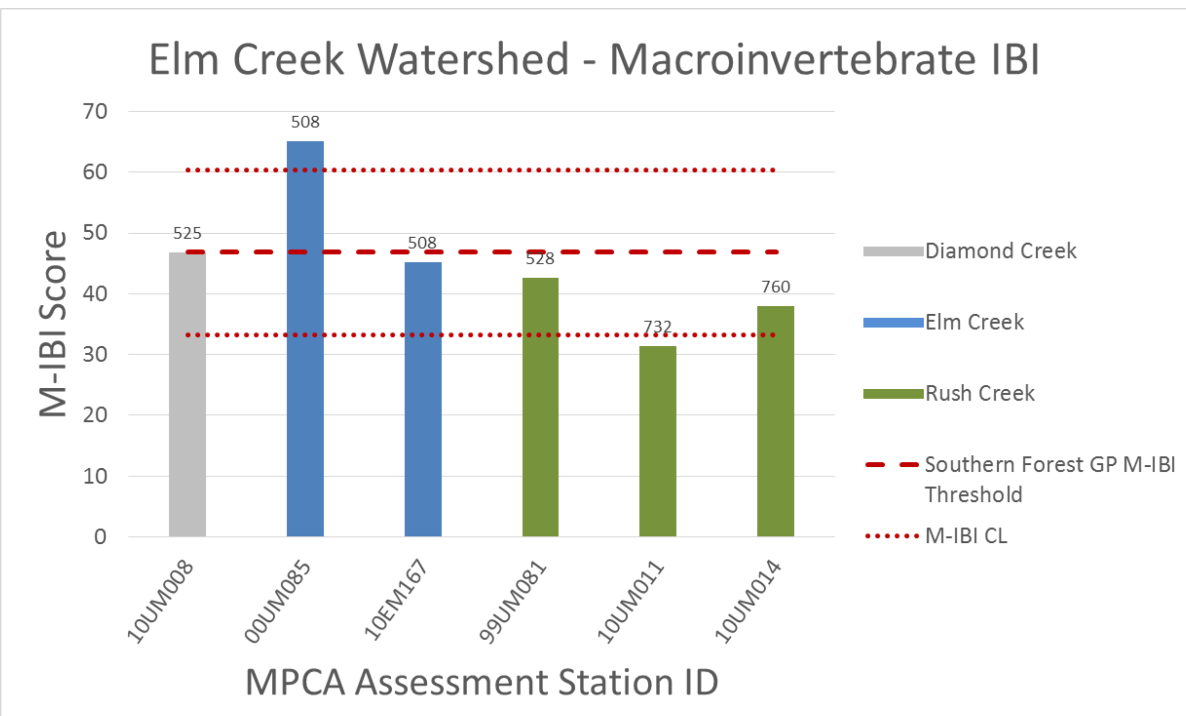


Figure 2.17. The IBI scores for macroinvertebrate assemblages sampled throughout the Elm Creek Watershed

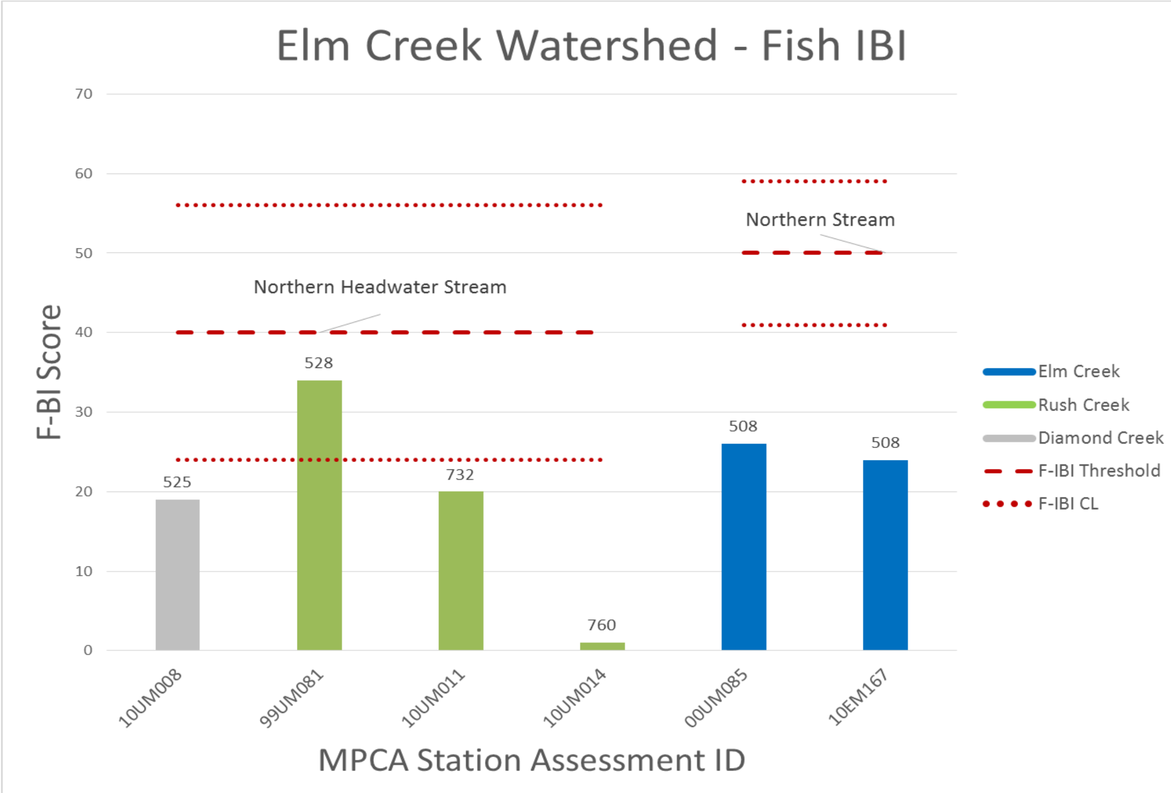


Figure 2.18. The IBI scores for fish assemblages sampled throughout the Elm Creek Watershed

Elm Creek

Two sites, comprising one AUID (508) were assessed to describe the biological assemblages in upper and lower Elm Creek. Biotic assessments for fish and macroinvertebrate assemblages are based on two classifications—Northern Stream (Fish) and Southern Forest Stream (Macroinvertebrates) in Elm Creek. Within Elm Creek, 10 sites have been sampled. Of the 10 sites initially sampled by the MPCA, two were considered "Assessable" based on stream channel characteristics and the timing of the most recent survey (MPCA 2013). Both sites (Station IDs 10EM167 and 00UM085) are located in lower Elm Creek (i.e., downstream of the Rice Lake impoundment). Of these two sites, neither met biocriteria for fish assemblages and one met biocriteria for macroinvertebrate assemblages. Most metrics used to calculate the Fish Index of Biological Integrity (F-IBI) in Elm Creek were below the IBI metric average and lower quartile (Figure 2.19), suggesting that multiple factors are likely impacting fish communities in Elm Creek. Of the metrics used to calculate the Macroinvertebrate Index of Biological Integrity (M-IBI), the lowest scores were related to the relative abundance of sensitive and predator taxa (Figure 2.21), suggesting that a proportionally smaller number of stressors are likely affecting macroinvertebrate communities throughout Elm Creek. The level of divergence from average taxa values was lowest in Elm Creek, suggests a lower level of biological impairment in this sub-basin.

Rush Creek

The assessment of Rush Creek is based on biological assemblages sampled from three AUIDs that cover the south fork (732 and 760) and main stem (528). Biotic assessments for fish and macroinvertebrate assemblages in Rush Creek are based on two classifications—Northern Headwater Stream (Fish) and Southern Forest Stream (Macroinvertebrates). Within Rush Creek six sites have been sampled. Of the six sites initially sampled by the MPCA, three were considered “Assessable” based on stream channel characteristics and the timing of the most recent survey (MPCA 2013). Two sites are located in the south fork of Rush Creek (Station IDs 10UM014 and 10UM011) and one is located on the main stem (Station ID 99UM081). Of these three sites, none met biocriteria for fish or macroinvertebrate assemblages. Most metrics used to calculate the F-IBI in Rush Creek were below the IBI metrics average and lower quartile (Figure 2.20), suggesting that multiple factors are likely impacting fish communities in Rush Creek. Of the metrics used to calculate the M-IBI (Figure 2.21), the lowest scores were related to the relative abundance of sensitive and predator taxa, suggesting that a lower number of stressors are likely affecting macroinvertebrate communities throughout Rush Creek. The level of divergence from average taxa values was greater in Rush Creek than in both Elm and Diamond Creeks; therefore, suggesting a more significant level of biological impairment in this sub-basin.

Diamond Creek

The assessment of Diamond creek is based on biological assemblages sampled from one AUID (525) that extends from the outlet of Diamond Lake to a small reservoir just upstream of the confluence with Elm Creek. Biotic assessments for fish and macroinvertebrate assemblages in Diamond Creek are based on two classifications—Northern Headwater Stream (Fish) and Southern Forest Stream (Macroinvertebrates). Within Rush Creek three sites have been sampled. Of the three sites initially sampled by the MPCA, one was considered “Assessable” based on stream channel characteristics and the timing of the most recent survey (MPCA 2013). The assessed sample site (Station ID 10UM008) located at the downstream end of Diamond Creek, just upstream from the confluence with Elm Creek. Neither fish nor macroinvertebrate assemblages met biocriteria in Diamond Creek (although the M-IBI score was just below the biocritera threshold). Most metrics used to calculate the F-IBI in Diamond Creek were below the IBI metric average and lower quartile (Figure 2.20), suggesting that multiple factors are likely impacting fish communities in Diamond Creek. Of the metrics used to calculate the M-IBI (Figure 2.21), the lowest scores were related to the relative abundance of a small number of sensitive, suggesting that a relatively small number of stressors are likely affecting macroinvertebrate communities throughout Diamond Creek. The level of divergence from average taxa values was at a moderate level Diamond Creek, suggesting a moderate level of biological impairment in this sub-basin.

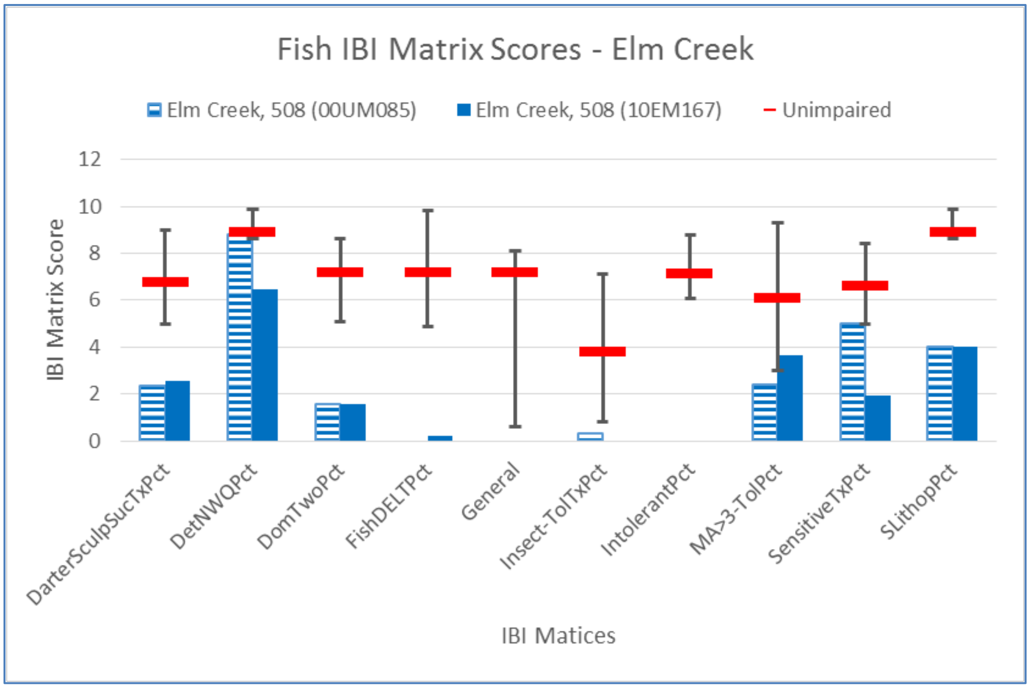


Figure 2.19. Relative metric scores used to calculate the F-IBI in Elm Creek as compared to unimpaired sites within the Northern stream classification. Horizontal bars represent average metric responses and "whiskers" represent upper and lower quartiles at unimpaired sites.

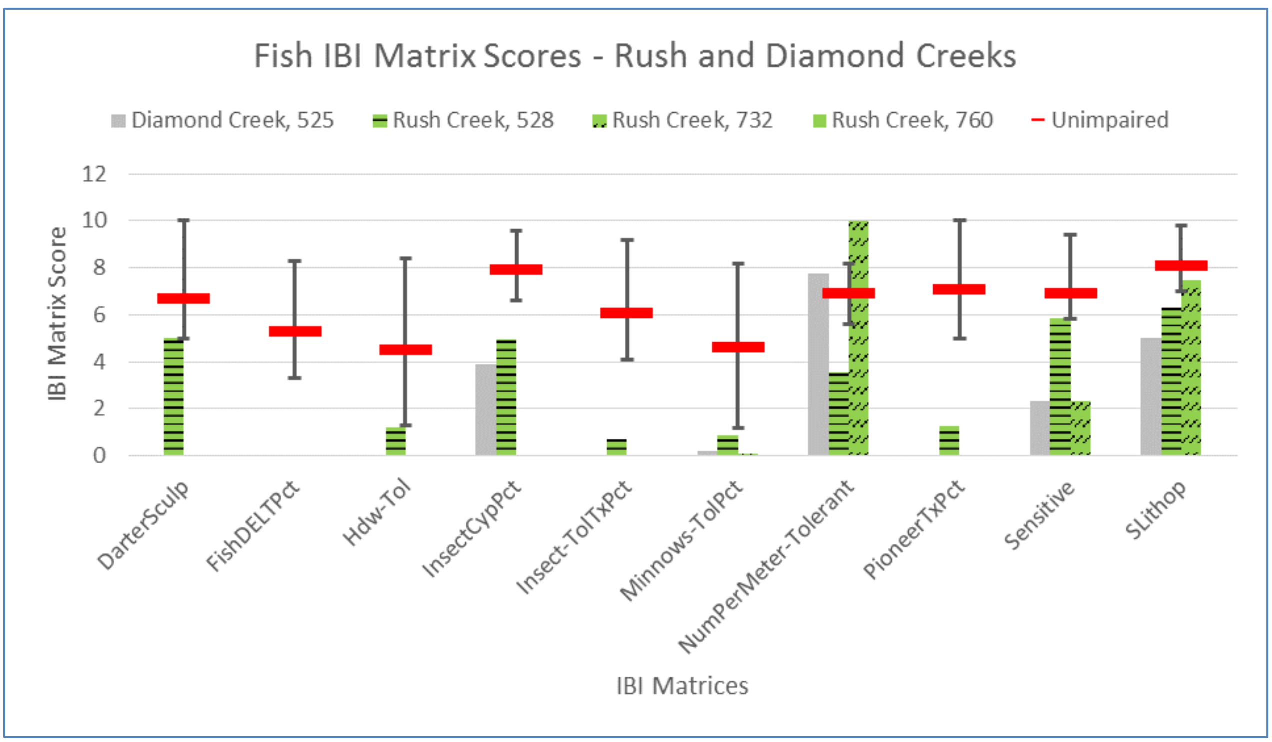


Figure 2.20. Relative metrics scores used to calculate the F-IBI in Rush and Diamond Creeks as compared to unimpaired sites within the Northern Headwater Stream classification. Horizontal bars represent average metric responses and "whiskers" represent upper and lower quartiles at unimpaired sites.

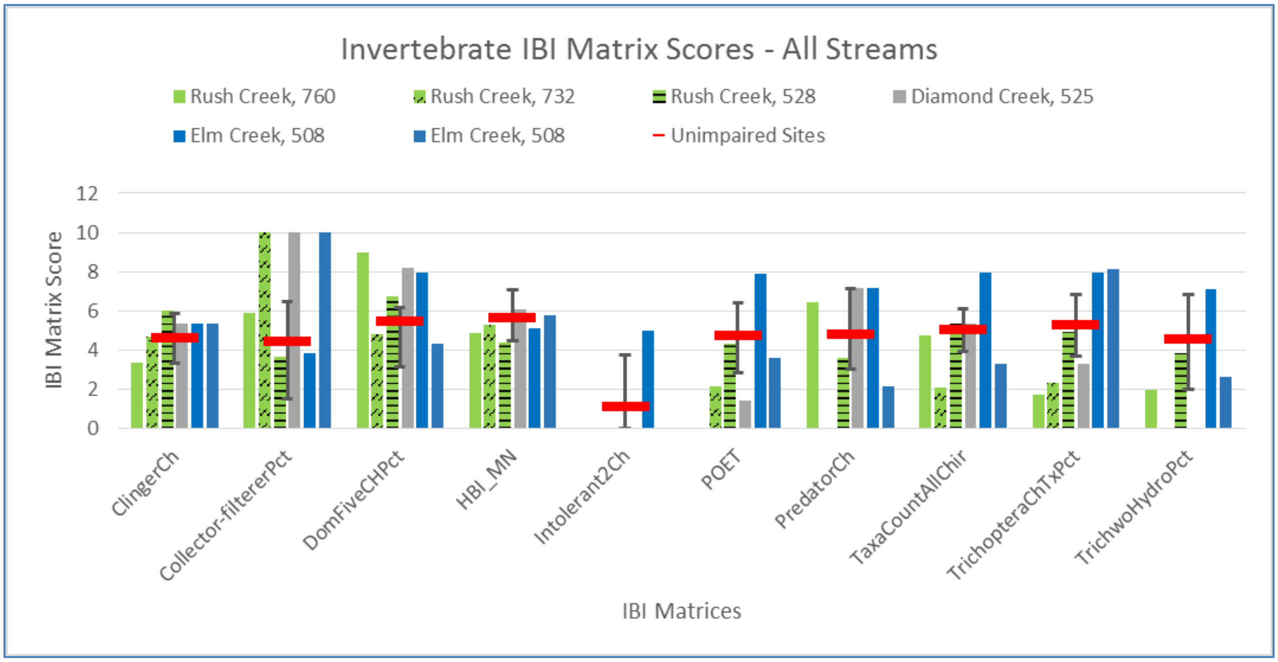


Figure 2.21. Relative metric scores used to calculate the M-IBI throughout the Elm Creek watershed as compared to unimpaired sites within the Southern Forest stream classification. Horizontal bars represent average metric responses and "whiskers" represent upper and lower quartiles at unimpaired sites.

3. Possible Stressors to Biological Communities

A comprehensive list of potential stressors to aquatic biological communities compiled by the EPA can be found here (http://www.epa.gov/caddis/si_step2_stressorlist_popup.html). This comprehensive list serves two purposes. First, it can serve as a checklist for investigators to consider all possible options for impairment in the watershed of interest. Second, it can be used to identify potential stressors that can be eliminated from further evaluation. In some cases, the data may be inconclusive and limit the ability to confidently determine if a stressor is causing impairment to aquatic life. It is imperative to document if a candidate cause was suspected, but there was not enough information to make a scientific determination of whether or not it is causing harm to aquatic life. In this case, management decisions can include modification of sampling plans and future evaluation of the inconclusive case. Alternatively, there may be enough information to conclude that a candidate cause is not causing biological impairment and therefore can be eliminated. The inconclusive or eliminated causes will be discussed in more detail in the following section.

3.1. Eliminated Causes

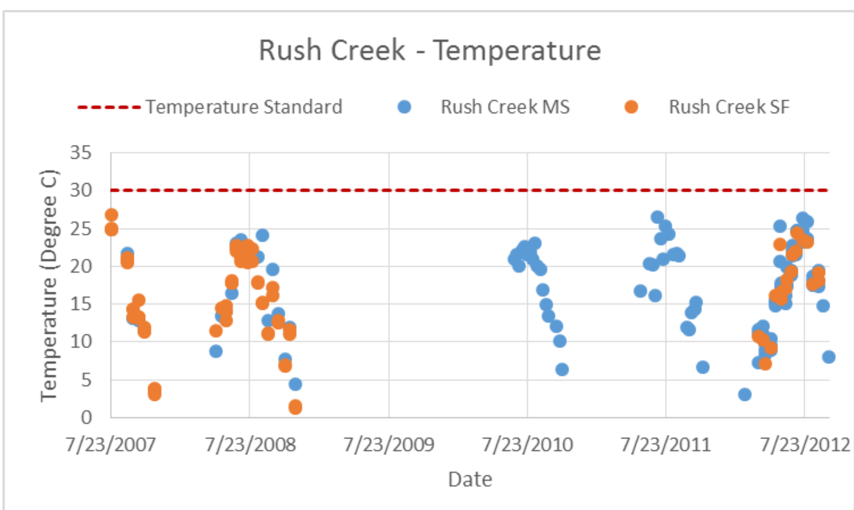
Of the initial stressors considered, four were eliminated as candidate causes: temperature, pH, un-ionized ammonia and nitrate. Details of the rationale for eliminating these potential stressors as causes of the biological impairments in Elm Creek are described below.

3.1.1. Eliminated Cause: Temperature

Temperature is a critical element of all aquatic ecosystems. Most aquatic organisms are ectothermic and thus, even small changes in water temperature can significantly affect biochemical process in exposed organisms. Daily and seasonal fluctuations in water temperature are a natural response to changes in the intensity and duration of solar radiation and most aquatic organisms can tolerate a range of temperatures (i.e., thermal range).

Temperature in aquatic ecosystem can be modified through a variety of processes, but loss of riparian canopy cover, reduced groundwater inputs and elevated temperatures in point source discharges are the most common sources of temperature alteration in streams. Most human alterations to streams result in increases in temperature. Shifts in temperature outside of (particularly above) optimal thermal ranges for different organisms can directly impact rates of growth, reproduction and ultimately survival. Indirectly, changes in temperature can impact levels of dissolved oxygen (DO) and contaminant toxicity (indirect effects of temperature on DO and contaminant toxicity are described further in Sections 4.5 and 3.1.3). To protect aquatic life from changes in stream temperature, Minnesota has established a water quality standard for temperature of "5°F [-15 ^{o}C] above natural in streams and 3°F [-16.1 ^{o}C] above natural in lakes, based on monthly average of the maximum daily temperatures, except in no case shall it exceed the daily average temperature of 86°F [30^{o}C]."

From 2006 to 2012, 837 discrete temperature samples were collected from 15 sites in Elm, Rush and Diamond Creeks. Additionally, in 2010, continuous (15 minute intervals) temperature data were collected at three sites (one in each Elm, Rush and Diamond Creeks). Throughout these water quality data sets, no temperature measurements exceeded the 30 ^{o}C standard and similarly no significant deviations from seasonal averages were present (Figure 3.1and Figure 3.2). Taken together these results suggest that elevated temperatures are not likely impacting biota in the Elm Creek Watershed.



a)

b)

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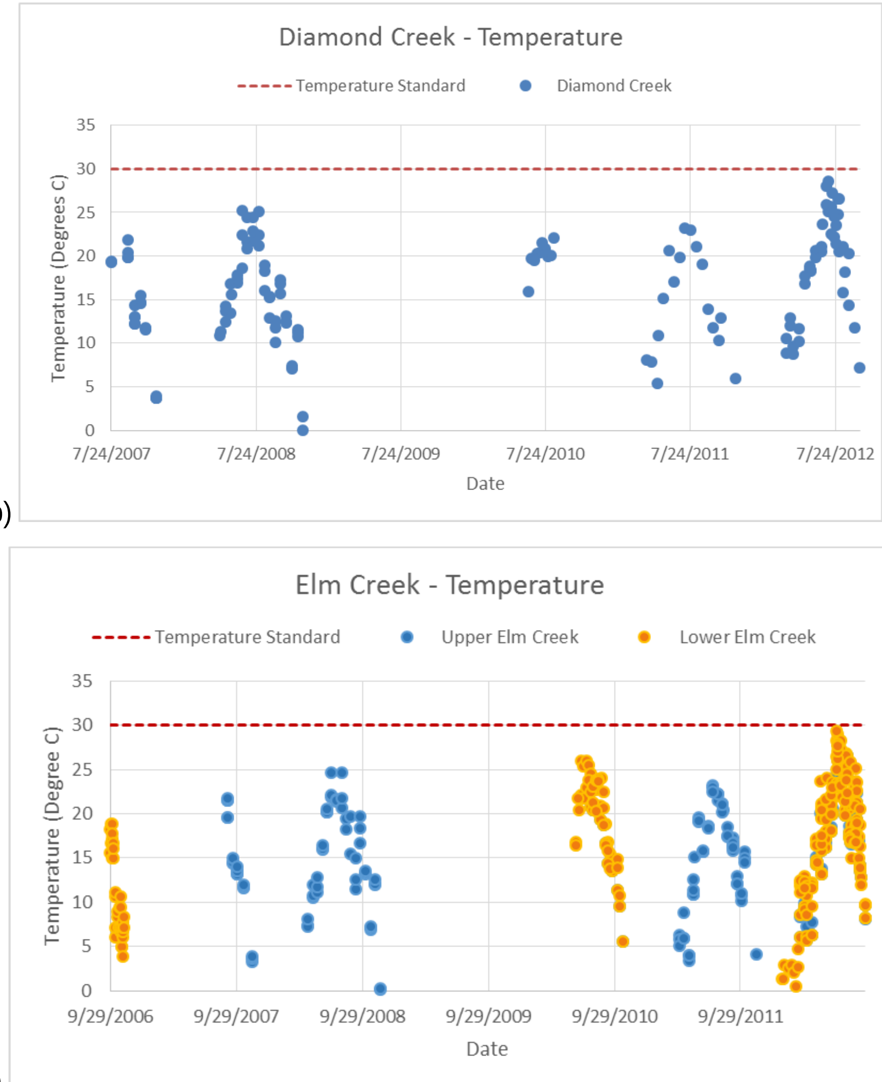
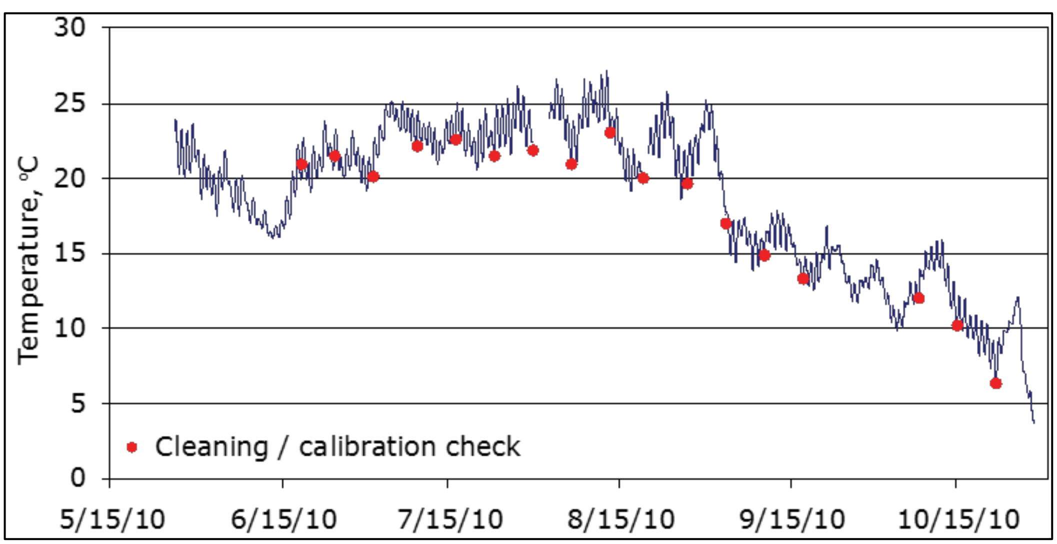
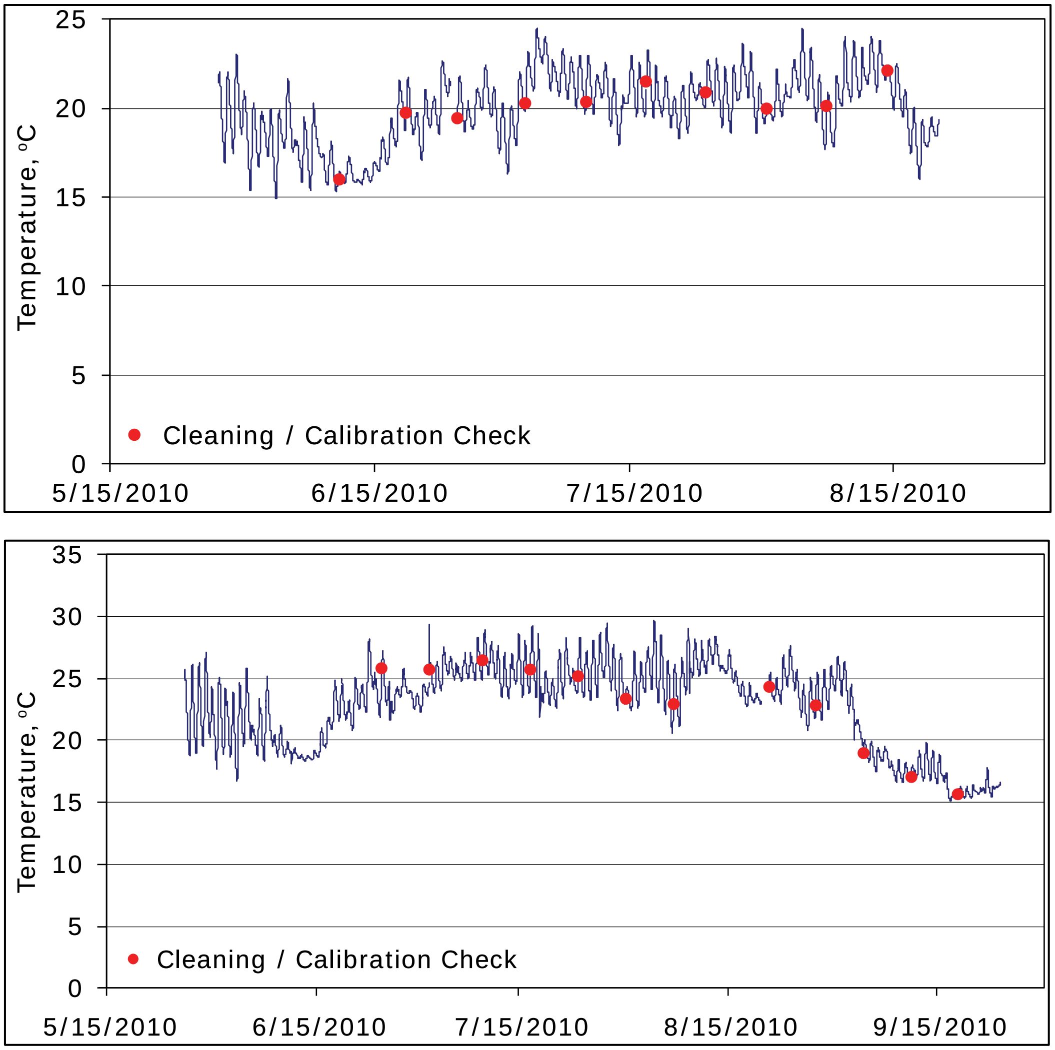


Figure 3.1. Discrete temperature measurements in a) Rush, b) Diamond and c) Elm Creek from 2006-20012



a)



b)

c)

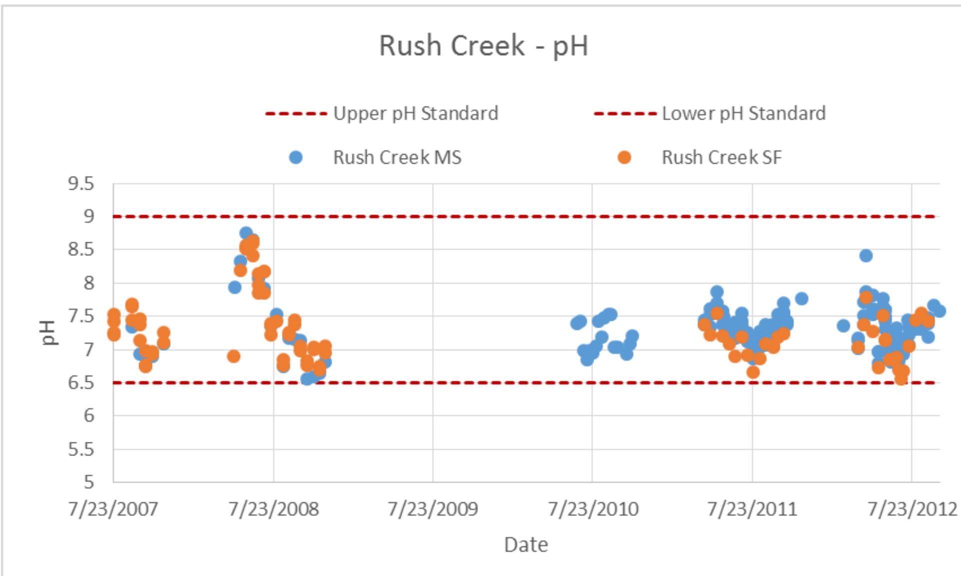
Figure 3.2. Continuous temperature measurements in a) Rush (RT), b) Diamond (DC) and c) Elm Creek (EC81) in 2010

3.1.2. Eliminated Cause: pH

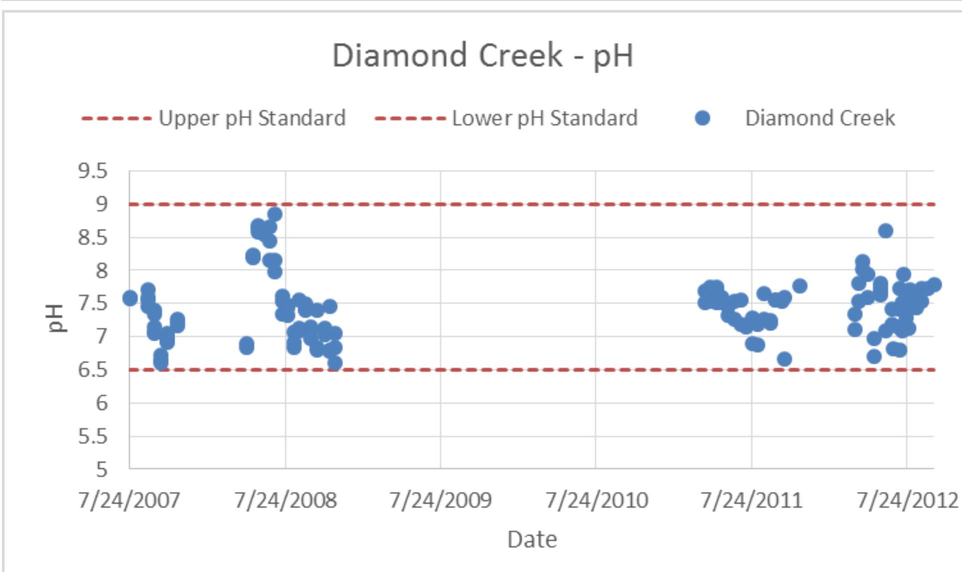
The pH is a measure of free hydrogen ions in water. The pH naturally fluctuates in stream ecosystems in response to geology, hydrology and biotic respiration. The primary source of hydrogen ions in water is from carbonic acid, which is produced in response to CO_2 from atmospheric diffusion and respiration by stream biota. Different geologic features (and in some case biota) can also impact stream pH by increasing the number of free ion sources from different minerals and altering the buffering capacity of the water.

Stream water pH can be altered through a variety of processes. However, the primary sources of altered pH in streams are atmospheric deposition, industrial discharges, excess ammonification and alkali runoff from agricultural lands and mineral extraction operations. Response to altered pH varies depending on the direction of the pH change. Low pH is most commonly associated with negative impacts to aquatic organisms. As pH is reduced, free hydrogen ion increase oxidative stress in cells, ultimately resulting in tissue damage and potentially whole organism toxicity. Lowered pH also can indirectly affect organisms by altering the toxicity of different contaminants, particularly heavy metals. Elevated pH primarily affects aquatic organisms by impacting gas transfer, epithelial and dermal cells. High pH also has the potential to indirectly increase the toxicity of ammonia, by increasing the relative occurrence of the more toxic form of un-ionized ammonia (see Section 3.1.3). To protect aquatic life from effects of altered pH, Minnesota has established a water quality standard for pH of between 6.5 and 9 in Class 2B streams.

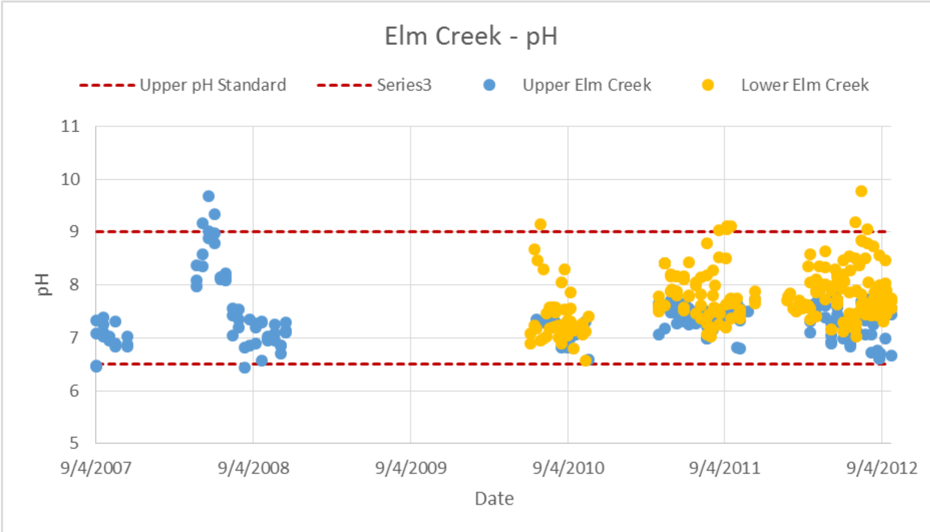
From 2007 to 2012, 816 discrete pH samples were collected from 15 sites in Elm, Rush and Diamond Creeks. Additionally, in 2010, continuous (15 minute intervals) pH data were collected in Elm Creek. Throughout these water quality data sets, no data points were observed outside of the upper and lower standard limits in all AUIDs except 508 (Figure 3.3 and Figure 3.4). In AUID 508, 12 samples were observed in excess of the upper standard. Taken together, these results suggest that pH is unlikely directly affecting most biotic communities in the Elm Creek watershed, but should not be ruled out as a potential stressor in AUID 508 (see Section 3.2.1 below). The potential for indirect affects through un-ionized ammonia and heavy metal toxicity are described below (see Sections 3.1.3 and 3.2.2).



a)



b)



c)

Figure 3.3. Discrete pH measurements in a) Rush, b) Diamond and c) Elm Creek from 2007-2008

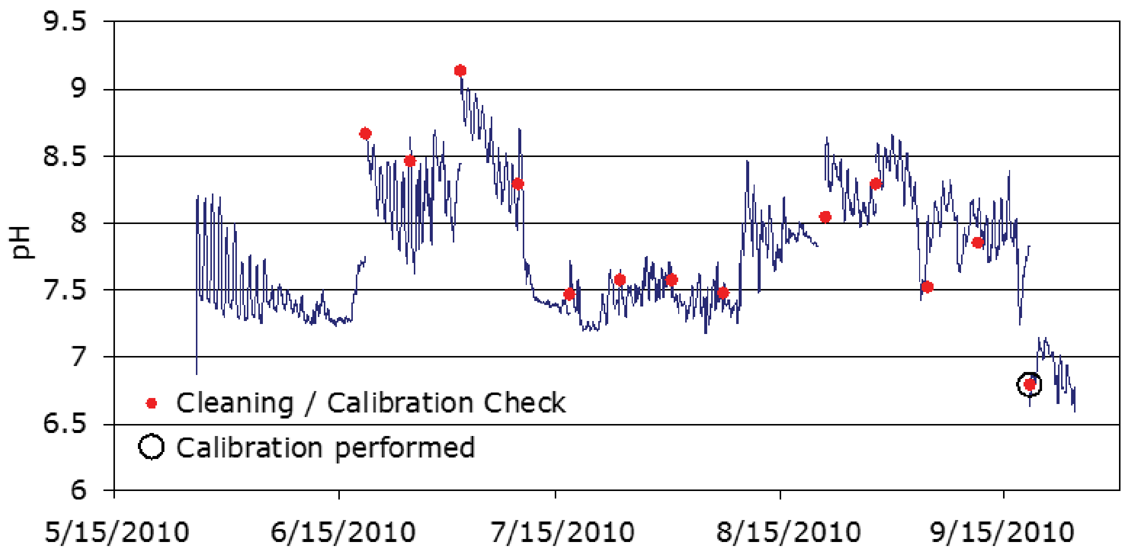


Figure 3.4. Continuous pH measurements in a) Elm (EC81) in 2010

3.1.3. Eliminated Cause: Un-ionized Ammonia

Ammonia is a commonly occurring chemical constituent in freshwater streams. Ammonia naturally enters into streams as a product of the nitrogen cycle. The majority of the global nitrogen supply exists in the atmosphere as N_2. Atmospheric nitrogen is converted to ammonium (NH_4^+) through the process of nitrogen fixation, which is primarily driven by symbiotic bacteria. As ammonia enters into terrestrial and aquatic systems, it is converted to nitrite (NO_2^-) and nitrate (NO_3^-) through the process of nitrification. Depending on local environmental conditions and biotic communities, NH_4^+ and NO_3^- are taken up by different plant species, and nitrogen is either passed up the food chain or reenters the ecosystem as (NH_4^+) when plants decompose and/or animals excrete urine/feces. When in aquatic ecosystems as a free chemical species, ammonia can be in either the ionized (NH_4^+) or the un-ionized (NH_3) form, depending on corresponding pH and temperature of the system. Unionized ammonia is significantly more toxic to aquatic life than NH_4^+.

Humans can significantly alter the concentrations of un-ionized ammonia in streams through a variety of processes. Most commonly, ammonia in streams is increased from fertilizer application and/or waste streams from humans and animals. However, increases in organic matter loads to stream and modification of temperature and pH can also increase un-ionized ammonia concentrations. To protect aquatic life from ammonia toxicity, Minnesota has established a numeric standard for un-ionized ammonia in streams of 0.04 mg/L.

Throughout the Elm Creek Watershed, a range of data have been collected to assess the potential for acute and chronic toxicity from un-ionized ammonia. Un-ionized ammonia concentrations were not directly measured throughout the Elm Creek Watershed. However, un-ionized ammonia concentrations can be calculated based on total ammonia measurements using concurrent temperature and pH (Emerson et al. 1975). Concurrent temperature and pH measurements were not collected along with ammonia measurements, but a theoretical maximum un-ionized ammonia concentration can be calculated based on the maximum temperature (28^{o}C) and pH (9.5) measurements observed throughout the Elm Creek system (see Sections 3.1.1 and 3.1.2 for a discussion of the temperature and pH ranges observed throughout Elm Creek).

From 1998 to 2010, a total of 743 ammonia samples were collected in upper and lower Elm Creek. Based the conversion of these data using maximum temperature and pH values, the theoretical maximum concentration of un-ionized ammonia was below the 0.04 mg/L standard in all but six samples (Figure 3.5). These data suggest that un-ionized ammonia is unlikely to be affecting biota in AUID 508 and can be ruled out as a potential stressor. Given that the samples sites that have the most consistent un-ionized ammonia records are located in the downstream portions of the watershed, it suggests that un-ionized ammonia is likely a relatively minor stressor to biota throughout the Elm Creek system. However, given the absence of data outside of AUID 508, un-ionized ammonia cannot be completely eliminated as a potential stressor in the remaining AUIDs.

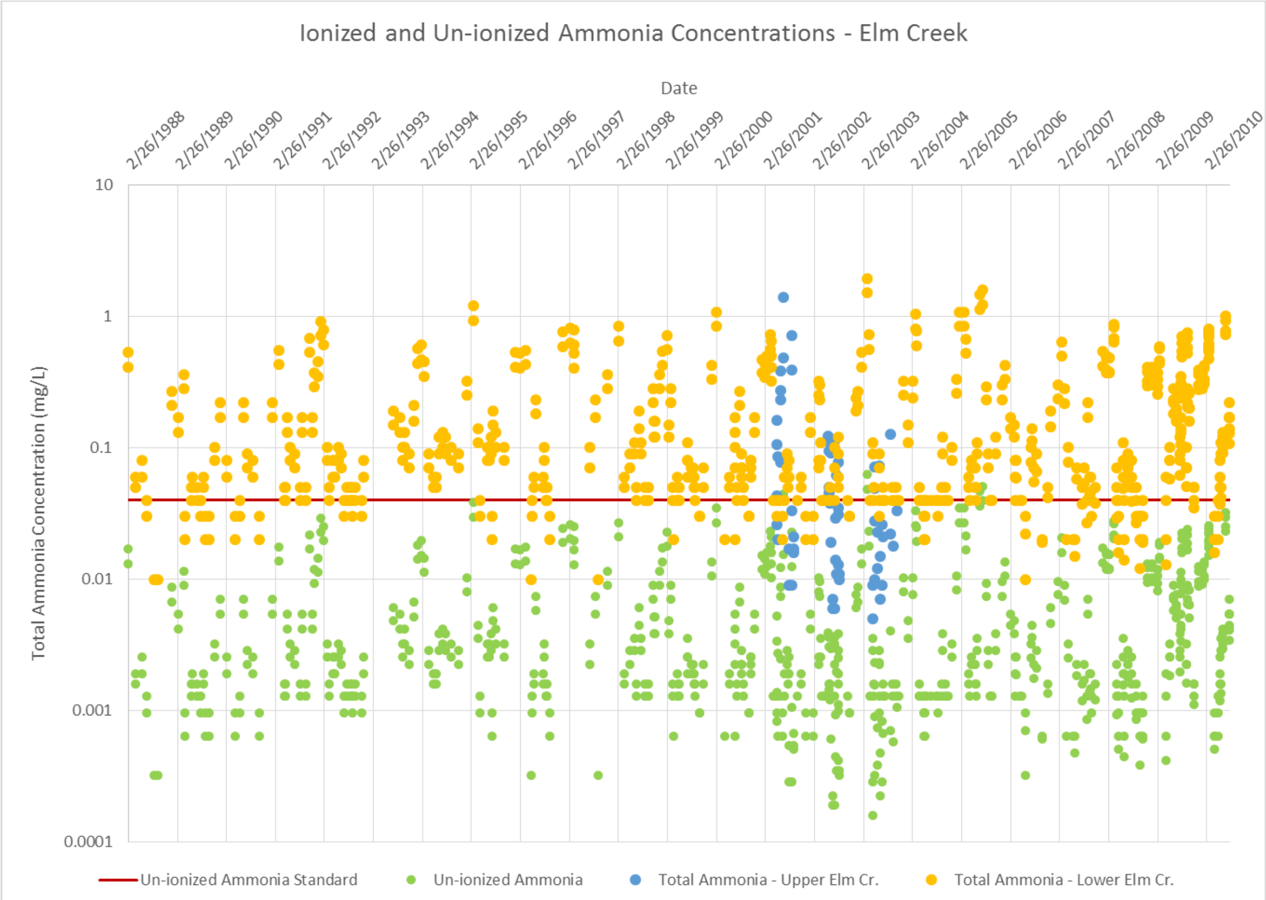


Figure 3.5. Total ammonia and un-ionized ammonia theoretical maximum concentrations measured from 1998-2010 in Elm Creek

3.1.4. Eliminated Cause: Excess Nitrate

Nitrate is a commonly occurring chemical constituent in freshwater ecosystems. Nitrate naturally enters into streams as a product of the nitrogen cycle. Nitrate concentrations are generally below 0.1 mg/L in stream, but can be increased through a variety of mechanisms (see Section 3.1.3 for further discussion on the nitrogen cycle). Elevated nitrate levels in surface and groundwater can elicit a variety of toxic responses in fish, wildlife and humans. To protect aquatic life from nitrate toxicity, Minnesota has proposed a draft standard of 41 mg/L (1-day Maximum Concentration) and 4.9 mg/L (4-day Chronic Maximum), which is currently under review for Class 2B streams.

A total of 958 samples were collected in upper and lower Elm Creek to assess the potential for nitrate contribution to the biological impairments. Throughout the data sets, 14 (1.5 %) samples did not meet the chronic standard and no measurements exceeded the acute standards (Figure 3.6). These data suggest that nitrate is unlikely to be affecting biota in AUID 508. Given that the samples sites that have the most consistent nitrate records are located in the downstream potions of the watershed, it suggests that nitrate is likely a relatively minor stressor to biota in Elm Creek. However, given the absence of data outside of AUID 508, nitrate cannot be completely eliminated as a potential stressor in the remaining AUIDs.

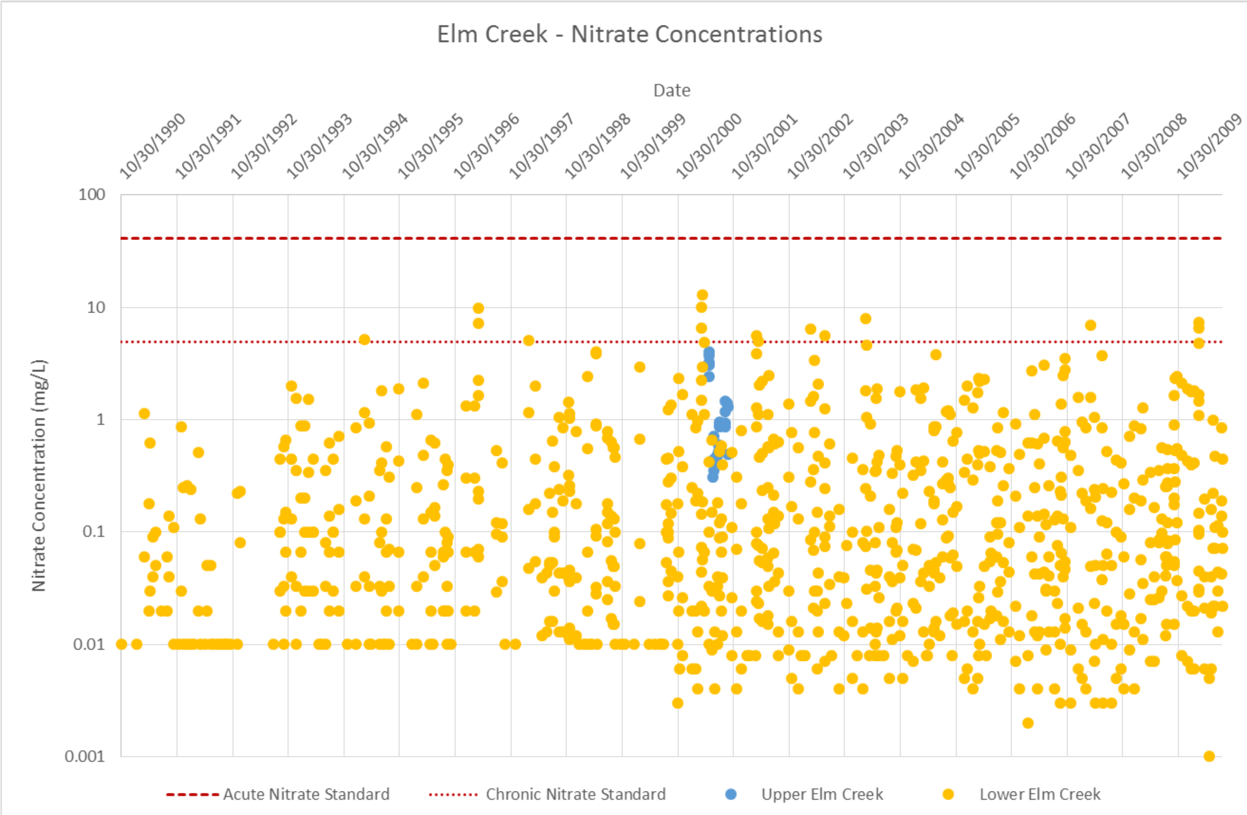


Figure 3.6. Nitrate concentrations measured in Elm Creek from in 2001

3.2. Inconclusive Causes

Two suites of pollutants (organic and inorganic contaminants) have been identified as inconclusive causes of the biological impairments throughout the Elm Creek Watershed and three potential stressors have been identified as inconclusive causes in select AUIDs (while having been ruled out in other AUIDs). In addition to the data described above for pH, ammonia and nitrate, a series of data sets have been collected to describe the potential impact of organic and inorganic pollutants on the biotic communities of the Elm Creek Watershed. These data sets (described below) suggest that organic and inorganic contaminants are not likely having an impact on biota throughout the Elm Creek Watershed. However, given the limited scope of these data sets, and the probable exposure to these chemical resulting from both high intensity agricultural and urban land uses, these pollutant groups have not been eliminated as potential candidate causes and are being listed as inconclusive causes.

3.2.1. Inconclusive Cause: pH (AUID 508)

As described in section 3.1.2 above, 12 samples were observed to be above the upper water quality standard for pH in Elm Creek. These data suggest that alkali stress is likely relatively uncommon in Elm Creek, but given the recurring observation of this potential stressor, it cannot be ruled out as a potential contributor to the biological impairments observed in AUID 508. The source of these pH spikes is currently unclear.

3.2.2. Inconclusive Cause: Un-ionized Ammonia (AUIDs 525, 528, 732 and 760)

As described in section 3.1.3 above, 734 samples were collected from AUID 508, which integrates runoff from all impaired AUIDs (i.e., is furthest downstream). These results suggest that un-ionized ammonia is unlikely to be a significant contributor to the biological impairments observed throughout the watershed. However, because un-ionized ammonia was not consistently measured in closer proximity to the remaining, biologically-impaired stream reaches, it cannot be completely ruled out as a stressor in AUIDs 525, 528, 732 and 760.

3.2.3. Inconclusive Cause: Nitrate (AUIDs 525, 528, 732 and 760)

As described in section 3.1.4 above, 958 samples were collected from AUID 508, which integrates runoff from all impaired AUIDs. These results suggest that nitrate is unlikely to be a significant contributor to the biological impairments observed throughout the watershed. However, because nitrate was not consistently measured in closer proximity to the remaining, biologically-impaired stream reaches, it cannot be completely ruled out as a stressor in AUIDs 525, 528, 732 and 760.

3.2.4. Inconclusive Cause: Organic Contaminants

A range of organic pesticides and herbicides have been shown to negatively affect biotic integrity in aquatic ecosystems. Pesticides and herbicides are commonly applied to urban and agricultural lands and have been shown to runoff in concentrations that can negatively affect the integrity of biotic communities in streams. Given the high density of agricultural and urban lands throughout the Elm Creek Watershed, runoff of these chemicals into Elm Creek is likely particularly following large rain events. To protect aquatic life against organic chemical exposures, Minnesota has adopted a number of water quality criteria for different pesticides and herbicides (see Table 3.1 for a listing of relevant standards).

In 1997, the USGS sampled 48 potential organic pollutants at the long-term monitoring site in lower Elm Creek. Within these samples, all but seven analytes were below detection limits. All analytes detected were below existing regulatory standards (Table 3.1). These results suggest that organic contaminant toxicity is unlikely to be a significant contributor to the biotic impairments in the Elm Creek Watershed. However, these parameters are being listed as inconclusive, given the significant time period between chemical and biological sampling and the limited geographic distribution of the sampling (samples were only collected at the USGS site in lower Elm Creek).

Table 3.1. Mean and maximum concentrations for potential organic contaminants in the Elm Creek Watershed

Stream	Chemical	Observed Concentration	Regulatory Standard	
			Chronic	Maximum
Elm Creek	2-Chloro-4isopropylamin-6-amino-s-traizine	0.011 ug/L	NS	
Elm Creek	Atrazine	0.027 ug/L	10 ug/L	323 ug/L
Elm Creek	Cyanazine	0.019 ug/L	NS	
Elm Creek	Fluoride	0.17 mg/L	NS	
Elm Creek	Metolachlor	0.004 ug/L	23 ug/L	271 ug/L
Elm Creek	Prometon	0.02 ug/L	NS	
Elm Creek	Simazine	0.005 ug/L	NS	
NS = No Minnesota Water Quality Standard				

3.2.5. Inconclusive Cause: Inorganic Contaminants

A range of inorganic contaminants (particularly heavy metals) have been shown to negatively affect biotic integrity in aquatic ecosystems. Heavy metals are common by products of mineral extraction/disposal, road traffic and industrial effluents and have been shown to runoff in concentrations that can negatively affect biotic communities in streams. Relatively few potential sources of heavy metals exist in Elm Creek, but given the high density of roads throughout the watershed, the potential exposure to heavy metals cannot be completely ruled out. To protect aquatic life against organic chemical exposures, Minnesota has adopted a number of water quality standards for different inorganic contaminants (see Table 3.2 for a listing of relevant standards).

From 1995-1998, the USGS conducted a series of analyses to determine heavy metal concentrations in lower Elm Creek (Table 3.2). Over this time approximately 22 samples were collected and analyzed for 22 different analytes. Minnesota has developed water quality standards for seven of the 22 parameters. Mean and maximum concentrations for all seven parameters are below the corresponding chronic and maximum regulatory standards (based on a corresponding average hardness value of 200 mg/L). These results suggest that heavy metal toxicity is unlikely to be a significant contributor to biotic impairment in the Elm Creek Watershed. However, these parameters are being listed as inconclusive, given the significant time period between chemical and biological sampling and the limited geographic distribution of the sampling (samples were only collected at the USGS site in lower Elm Creek).

Table 3.2. Mean and maximum concentrations for potential inorganic contaminants in the Elm Creek watershed.

Stream	Chemical	Sample Number (n)	Mean Concentration (ug/L)	Maximum Concentration (ug/L)	Regulatory Standard (ug/L)		Mean Hardness (mg/L)
					*Chronic	*Maximum	
Elm Creek	Cadmium	22	1	2	2	73	197
Elm Creek	Chromium	22	BDL	6	365	3064	
Elm Creek	Cobalt	22	BDL	6	5	436	
Elm Creek	Copper	22	BDL	10	15	34	
Elm Creek	Lead	22	BDL	20	7.7	197	
Elm Creek	Silver	22	BDL	1	1	13	
Elm Creek	Zinc	22	2.5	15	191	221	

*Calcualted Based on a Mean Hardness of 200 mg/L

3.3. Summary of Candidate Causes in the Elm Creek Watershed

Six candidate causes were selected as possible drivers of biological impairments in the Elm Creek Watershed. A summary of the following candidate causes is described below: altered hydrology, physical habitat alteration, excess sediments, excess phosphorus, low DO, and elevated chloride ions.

3.3.1. Candidate Cause: Altered Hydrology

Stream hydrology is a product of climate, weather and watershed structure and is the primary contributor to, and determinant of, most stressors in stream ecosystems. Stream hydrology is initially determined by the amount, timing and rate of precipitation—all of which influence the erosive potential of the stream (e.g., Poff et al. 1997). Stream channels form as water erodes a path of least resistance as governed by gravitation pull and the surrounding geology (e.g., Woman and Miller 1960; Montgomery and Dietrich 1988). In actively forming streams, the rate of erosion is high, as the stream down-cuts to a non-erodible geological feature and/or establishes a floodplain where erosive power is dissipated. Over time, stream channels stabilize and establish a dynamic equilibrium with the floodplain that facilitates vegetative establishment along the stream banks. As the upland watershed becomes more densely vegetated, an increasing percentage of discharge is contributed to the stream by groundwater inputs. In the Midwest region, on average, less than 10% of annual rainfall would be expected to directly drain into the stream. As a result, the hydrograph in an undisturbed stream is often very broad—as precipitation slowly drains through the soil to the stream. As riparian areas become more heavily vegetated, instream biotic process and habitat become more heavily influenced by streambank vegetation (Bilby and Likens 1980).

3.3.1.1. Water Quality Standards

No formal standards have been developed to assess hydrologic stability and/or alteration. Instead, hydrologic disturbance must be evaluated on using a weight-of-evidence approach that links known sources of hydrologic disturbance with concurrent and/or sequential changes in hydrologic regime.

3.3.1.2. Sources and Causal Pathways Model for Altered Hydrology

Stream hydrology can be altered by a variety of processes, but changes in precipitation patterns, channel structure, watershed land use and surface/groundwater extraction are four of the most common stressors (e.g., Booth and Jackson 1997). Historical changes in precipitation patterns, channel structure (including drainage networks), watershed land use and surface/groundwater extraction throughout the Elm Creek watershed are describe in detail in Section 2.1. In general, in first and second order streams, most changes in precipitation, channel structure and/or watershed land use increase the volume and rate of runoff to the channel. As a result, stream hydrology often becomes more volatile (i.e., "flashier"), where both high and low flow events become more common and intense. Surface and groundwater withdrawals can further exacerbate the magnitude and duration of low-flow conditions.

In response to increased water yield and erosive potential, streams begin to reestablish a channel that corresponds to the "new" hydrologic regime. Initially, increased water yield generally results in channel down-cutting (e.g., Booth and Jackson 1997). As the channel down-cuts, the confined, or "entrenched", stream dissipates energy against the newly exposed, vertical streambanks. In response, lateral, stream bank erosion increases and the channel widens. These changes in hydrologic and erosional processes can have direct impacts on aquatic organisms, but more commonly the secondary effects of altered hydrology (e.g., habitat alteration, erosion and sedimentation, nutrient loading) are the direct sources of impacts to stream biota. The biotic impacts of habitat alteration, sedimentation and nutrient loading are describe in detail in subsequent sections (see Section 4.2, 4.3 and 4.4).

Stream hydrology can also be modified such that the hydrologic regime becomes less flashy. In particular, as water is retained behind dams, or discharge rates are spatially constricted by undersized road crossings, peak flows and floodplain interaction can be reduced (e.g., Poff et al 1997). This reduction in peak flow can also be accompanied by loss of course sediments, as large sediment particle settle out in low velocity waters upstream of retention structures. Changes in hydrology toward a less flashy regime are most common in larger, actively regulated reservoirs systems and less likely in Elm Creek, despite the occurrence of several small retention structure (for further discussion, see Section 2.1).

3.3.1.3. Overview of Altered Hydrology in the Elm Creek Watershed

As described in Section 2.1, the long-term changes in precipitation patterns, altered land use, channel modification and agricultural drainage systems that of often result in altered stream hydrology are commonly occurring throughout all sub-basins within the Elm Creek Watershed. Consequently, the occurrence of altered hydrology is highly probable throughout the Elm Creek watershed. For a more complete review of hydrologic data in the Elm Creek Watershed see Section 4.1.

3.3.2. Candidate Cause: Altered Physical Habitat

Habitat complexity is a critical element of biotic integrity in stream ecosystem. Stream habitat forms in response to hydrological processes and establishes around the dynamic hydrological regime that develops throughout the process of stream channel formation (see Section 3.3.1 for further discussion). As a stream channel forms, it develops a sinuous pattern in response to soil erodibility and stream power. Consequently, because stream water flows linearly along a gravitational gradient, a range of current velocities and flow conditions exist throughout the channel cross section. Areas of high velocity have a coarse particle structure and area of low velocity has a finer particle structure. This structural diversity facilitates the development of different velocity refugia and food sources. As riparian areas become more densely vegetation, large woody debris is often recruited to the stream, further influencing habitat diversity and channel complexity.

In addition to the localized complexity created by channel structure and velocity, habitat diversity is exists along longitudinal gradients (e.g., Vannote et al. 1980). As streams increase in size, interaction with the floodplain, riparian vegetation and the source of primary productivity shifts, although this varies depending on the region of the country. In many Midwestern streams, narrow first order streams were historically free of significant canopy cover, as a result of fire-induced tree suppression (e.g., Wiley et al. 1990). As these streams widened, the water formed a natural fire break which allowed an enclosed canopy to form. As these streams continued to widen and flow through wetland and lake complexes the influence of the canopy decreased and the importance of autochthonous processes become more important.

3.3.2.1. Water Quality Standards

Quantitative habitat criteria and/or standards have not been established for stream ecosystems in Minnesota. In the absence of statewide habitat standards, habitat alteration must be evaluated on a site-specific basis. Several qualitative and semi-quantitative methodologies have been utilized for assessing stream habitat alteration and its potential contribution to biotic impairments. Although each of these methods differs in field application, most focus on the characterization of geomorphic change induced by anthropogenic sources—often measured by a divergence of current habitat quality from potential habitat quality using a stream-type model.

3.3.2.2. Sources and Causal Pathways Model for Altered Physical Habitat

Alteration of physical habitat in stream ecosystems can result from a variety of processes. However, given the importance of hydrological regime in the creation and maintenance of stream habitat, hydrologic alteration is also one of the key drivers of habitat degradation (e.g., Poff et al. 1997). As stream becomes flashier, and erosional potential increases, the stream channel widens and fine sediments are deposited more uniformly across the stream bottom (e.g., Hammer 1972)—reducing particle size diversity and the availability of velocity refugia in intestinal sediment spaces (see Section 3.3.3 for further discussion). If stream power and erosional processes increase significantly, riparian vegetation is often eroded into the stream channel reducing canopy cover and increasing the occurrence of blockages and stagnant water.

Physical stream channel alteration is also a significant source of habitat modification. Many streams throughout the Midwest (particularly in the Elm Creek Watershed) have been significantly channelized to facilitate drainage of agricultural and urban lands and promote navigation (see Section 2.1 for further discussion). Channelization has a similar effect on habitat homogenization as does altered hydrology (Lau et al. 2006). As a channel is straightened, flow volume is concentrated and the erosional potential of the stream (i.e., stream competence) increases (e.g., Prestegaard et al. 1994). In response, channelized reaches often either down-cut as a result of erosional processes or are anthropogenically widened or deepened (i.e., dredged) to increase channel volume and minimize flooding. In general, these processes result in more uniform cross-sectional channel habitat that is entrenched and poorly connected to riparian habitat (e.g., Prestegaard 1988). Additionally, many channelized reaches are actively managed and often less shaded by riparian vegetation, minimizing the availability of riparian-influence instream habitat.

3.3.2.3. Overview of Altered Physical Habitat in the Elm Creek Watershed

As described in Section 2.1, the long-term changes in land use, hydrology, and channel modification that often result in altered physical habitat in streams are commonly occurring throughout all sub-basins within the Elm Creek Watershed. Consequently, the occurrence of altered hydrology is highly probable throughout the Elm Creek Watershed. For a more complete review of habitat data in the Elm Creek Watershed see Section 4.2.

3.3.3. Candidate Cause: Excess Sediments

Fine sediments are a naturally occurring component of all stream ecosystems. As described in Section 3.3.1, elevated concentrations of fine sediments in actively forming streams is relatively common. In a geologically stable stream system, erosional and deposition processes within a given reach are relatively balanced (although the stream as a whole is a net sediment exporting system). In these well-established streams systems, fine sediments are primarily retained by watershed vegetation or rapidly washed downstream. As a result, stream bed sediments are often comprised of large diameter particles with significant intestinal spacing. However, even in very stable stream systems, fine sediments continue to serve an important habitat role for different species—and significant loss of fine sediment out of stream systems can negatively impact biological communities.

3.3.3.1. Water Quality Standards

Minnesota has one existing turbidity and one total suspended solids (TSS) water quality standard that can be used to characterize sediment as a stressor. The turbidity standard for Class 2B streams is 25 Nepheolometic Turbidity Units (NTU). A stream is in violation of the turbidity standard and listed as impaired if:

More than 10% water quality samples collected in a 10-year period exceed the standard

The TSS standard for Class 2B streams is 30 mg/L. A stream will be considered in violation of this standard and listed as impaired if:

More than 10% water quality samples collected in a 10-year period exceed the standard

3.3.3.2. Sources and Causal Pathways Model for Excess Sediments

The primary sources of sediment to streams are altered hydrology and land use change (e.g., Nelson and Booth 2002). As describe above (see Section 3.3.1), when stream hydrology is altered, it most commonly increases the erosive potential of a stream and sediment delivery increases until the stream reestablishes a new hydrologic regime, which can take decades to centuries, depending on the magnitude of the initial disturbance. Similarly, when land use is altered, most commonly this results in reduced vegetative cover and more exposed sediment, which in turn leads to the delivery of more sediment per unit area of watershed to the stream. Once sediments of an erodible particles size enter the stream, they remain in the system until they are deposited in a floodplain, wetland or downstream receiving waterbody. Often sediments delivered from large erosional events, are transported downstream episodically each spring when Midwestern streams generally have the highest erosive potential.

3.3.3.3. Overview of Excess Sediments in the Elm Creek Watershed

As described in Section 2.1, the altered land use, hydrologic and habitat conditions that are common sources of sediment impacts to stream biota are commonly occurring throughout all sub-basins within the Elm Creek Watershed. Consequently, elevated sediment loads are highly probable throughout the Elm Creek Watershed. For a more complete review of sediment data in the Elm Creek Watershed see Section 4.3.

3.3.4. Candidate Cause: Excess Phosphorus

Phosphorus is commonly the primary limiting nutrient in stream ecosystems—and, paradoxically, one of the most common pollutants in freshwater ecosystems. Most of the global phosphorus supply is sequestered within geologic features and biota. As a stream ages (geologically), phosphorus is washed into the water column from the surrounding landscape—often bound to sediment particles. In newly forming streams, erosion and sediment release are high, as a stream establishes its channel—consequently; phosphorus concentrations can be somewhat elevated. As a stream channel and upland/riparian vegetation become more clearly established, phosphorus enters into a dynamic equilibrium, where inputs and outputs are relatively balanced in any given reach (particularly in higher gradient streams). As stream channels decrease in gradient and become more sinuous, streams become more highly influenced by floodplains, connected lakes and wetland habitat—which seasonally can function as both sources and sinks of phosphorus (often depending on land use history).

3.3.4.1. Water Quality Standards

Minnesota has established a numeric phosphorus standard of 100 ug/L for Class 2B streams (Central Region). Using this standard, a stream is considered impaired if:

> Average phosphorus concentrations over a two year period exceed 100 ug/L and standards for one (or more of three) causative variable are concurrently exceeded: 18 ug/L Chlorophyll-a; 3.5 mg/L daily DO flux; 2 mg/L BOD

3.3.4.2. Sources and Causal Pathways Model for Excess Phosphorus

Instream phosphorus concentration can be significantly increased through a variety of mechanisms. In general, phosphorus concentrations increase as rates of erosion in stream channels or the associated upland watersheds increase, or as phosphorus is imported into a watershed—often in the form of inorganic fertilizers, human waste and livestock feed. Phosphorus is generally bound to sediment particles under aerobic conditions. As sediment loads to streams increase, so does phosphorus concentrations. Similarly, many fertilizers, and all human waste and livestock manure contain significant levels of phosphorus. As such, runoff from agricultural and urban lands, as well as discharges from wastewater facilities and septic systems has the potential to increase phosphorus concentrations in stream systems. Rates of phosphorus delivery to streams from undisturbed lands are often less than 0.2 lbs/acre/year, whereas phosphorus delivery from agricultural and urban lands can be 1-2 lbs/acre/year, or greater.

Over time, phosphorus accumulates in watershed soils and/or wetland and lake sediments as a result of natural and anthropogenic processes. As sediment and soil phosphorus concentrations increase over time, it can become significant phosphorus sources to stream ecosystems. For example, phosphorus saturation in historical agricultural fields can continue to deliver phosphorus to aquatic systems decades after agricultural usage has ceased. Similarly, as particulate phosphorus accumulates in wetland and lake sediments, it can be re-released in dissolved form during times of sediment anoxia or hypoxia. This anoxic release can be so pronounced that lakes and wetland that may have historically served as phosphorus sinks in a watershed may become significant, long-term sources. Additionally, as a streams become disconnected from its associated riparian and hyporheic ecosystems, phosphorus loss from the stream can be reduced.

3.3.4.3. Overview of Phosphorus in the Elm Creek Watershed

As described in Section 2.1, the altered land use, hydrologic and habitat conditions that are common sources of phosphorus to stream biota are commonly occurring throughout all sub-basins within the Elm Creek Watershed. Consequently, elevated phosphorus loads are highly probable throughout the Elm Creek Watershed. For a more complete review of phosphorus data in the Elm Creek watershed, see Section 4.4.

3.3.5. Candidate Cause: Low Dissolved Oxygen

The DO is critical to most aquatic life. Temperature is the primary factor governing the concentration of oxygen in the water column—oxygen dissolves to a higher concentration in cold water than in warm water. Although the saturation concentration of oxygen in water is governed by temperature, oxygen levels are also affected by the production, consumption and diffusion of oxygen in stream systems. Oxygen enters stream water primarily through atmospheric diffusion and photosynthetic production. As water becomes more turbulent, oxygen is more easily able to diffuse into and out of the water column toward its thermal equilibrium point. Similarly, as primary production increases in the water column, oxygen concentrations often increase in response to photosynthesis. However, excess primary productivity has the potential to lower DO concentrations as a result of decomposition and heterotrophic respiration processes (see Section 4.4 for further discussion).

3.3.5.1. Water Quality Standards

Minnesota has established a water quality standard for DO of 5 mg/L (daily minimum) for Class 2B streams. Using this standard, a stream would be to consider impaired by low DO if:

> "more than 10% of "suitable" (taken before 9:00 AM) May through September measurements, or more than 10% of the total May through September measurements, or more than 10% of the October through April measures me violate the [5 mg/L] standard, and 2) there are at least three violations"

3.3.5.2. Sources and Causal Pathways Model for Low Dissolved Oxygen

Given the complexity of the oxygen cycle, a wide range processes and stressors can result in low DO conditions. In fact, oxygen fluctuation in response to diurnal cycles of photosynthesis and respiration are common in stream ecosystems, particularly those with low gradient wetland complexes. Diurnal DO fluctuation of 3 mg/L is often observed in undistributed, reference systems. Stressors that amplify diurnal DO fluctuation and/or completely suppress DO levels across a natural diurnal cycle have the potential to significantly impact stream biota.

The most widely studied source of oxygen demand (i.e., oxygen loss) in stream ecosystems is phosphorus-induced, cultural eutrophication. As described in Section 3.3.4, elevated phosphorus levels have the potential to increase rates of primary productivity. As autotrophs enter into respiration (i.e., autotrophic respiration), or die and are decomposed (i.e., heterotrophic respiration) oxygen in removed from the water column, often leading to short-term and/or long-term suppression of oxygen levels (i.e., an oxygen "sag"). In addition to the phosphorus-induced DO suppression described above, oxygen demand can be created by denitrification processes as well as sediment respiration and heterotrophic response to organic matter loading.

A variety of physical sources of oxygen stress also exist in streams. As streams become wider, shallower and less shaded (often in response to altered hydrologic and riparian habitat conditions) stream temperatures can increase dramatically, significantly limiting the oxygen saturation level of the water. Similarly, as stream habitat becomes simplified (particularly in response to channelization) water turbulence generally decreases, reducing the potential diffusion of atmospheric oxygen.

3.3.5.3. Overview of DO in the Elm Creek Watershed

As described in Section 2.1, the altered land use, hydrologic and habitat conditions and elevated nutrient levels that are common sources of low DO in streams are commonly occurring throughout all sub-basins within the Elm Creek Watershed. Consequently, low DO concentrations are highly probable throughout the Elm Creek Watershed. For a more complete review of DO data in the Elm Creek Watershed, see Section 4.5.

3.3.6. Candidate Cause: Excess Chloride

High concentrations of chloride ions are relatively uncommon in most stream systems. In certain, coastal ecosystem and watersheds comprised of chloride rich geological features, chloride ions can be somewhat elevated in stream ecosystems. However, in most stream systems, the presence of chloride ions is most commonly associated with anthropogenic disturbances.

3.3.6.1. Water Quality Standards

Minnesota has established two water quality standards for chloride concentration of 230 mg/L (chronic standard) and 860 mg/L (maximum standard). Using these standards, a stream would be considered impaired by elevated chloride concentrations if:

> Chloride concentrations exceed 860 mg/L for more than one hour and/or an average of 230 mg/L over a four day period

3.3.6.2. Sources and Causal Pathways Model for Excess Chloride

A range of anthropogenic sources have the potential to contribute chloride ions to stream ecosystems. However, the primarily source of chloride ions delivered to stream ecosystems in the Midwest is associated with chloride containing deicer salt application for ice control on hard surfaces. Different salt formulations have the ability to suppress the freezing point of water and are commonly utilized to control ice buildup on automotive and pedestrian travel corridors. Following salt application, as ice and snow cover begin to melt, chloride ions are often directly conveyed to different receiving waters. As a result, elevated chloride concentrations in streams are most common during snow melt and spring runoff events. In some cases (often near bridge crossings), repeated salt application has resulted elevated chloride groundwater concentrations. In the presence of elevated chloride groundwater concentrations, instream chloride concentrations often coincide for periods of baseflow when ground water inputs dominate streamflow (often mid to late summer in the upper Midwest).

3.3.6.3. Overview of Chloride in the Elm Creek Watershed

As described in Section 2.1, the altered land use and high density of transportation corridors that are common sources of chloride runoff to streams are commonly occurring throughout all sub-basins within the Elm Creek Watershed. Consequently, elevated chloride concentrations are highly probable throughout the Elm Creek Watershed. For a more complete review of chloride data in the Elm Creek Watershed, see Section 4.6.

3.4. Summary of Water Quality and Hydrology Data

Water quality and hydrology data have been collected from 26 stream and stormwater sites throughout the Elm Creek Watersheds over the past 25-years (Figure 3.7). Existing data have been collected for a variety of purposes, but the majority of the sampling sites and parameters were selected specifically for the purposes of SID and TMDL development. Most sites have been monitored by paring continuous water level and discharge data with recurring water quality sampling. Five sites have been monitored with continuous water quality sensors that log temperature, pH, conductivity and DO at hourly intervals. Across all sites, between 11 and 16 water quality parameters have been collected. For a detailed summary of all water quality and hydrology data collected throughout the Elm Creek Watershed, see the Water Quality Data report (Appendix A) and the Mississippi River Twin Cities—Monitoring and Assessment Report (Anderson et al. 2013).

In general, water quality and hydrology data are indicative of watershed conditions that have been significantly impacted by agricultural and urban development. Stream flows are consistently flashy across all sites. Nutrient and sediment concentrations are consistently elevated and generally above established or proposed water quality standards. The DO concentrations are generally low across most of the watershed and biochemical oxygen demand is often correspondingly high. Chloride concentrations are generally low, but frequently above established acute and chronic standards.

Throughout all AUIDs multiple long-term stations have been monitored except in AUID 760. Given the close proximity of AUID 760 to AUID 732, water quality data from the long-term monitoring site on the south fork of Rush Creek (RCSL) was used as a surrogate AUID 760. Long-term water quality data records from the USGS site in lower Elm Creek were used as a surrogate for the watershed to evaluate the potential for biological impacts from un-ionized ammonia and nitrate. Detailed, parameter-specific responses in individual AUIDs are described throughout Section 4.

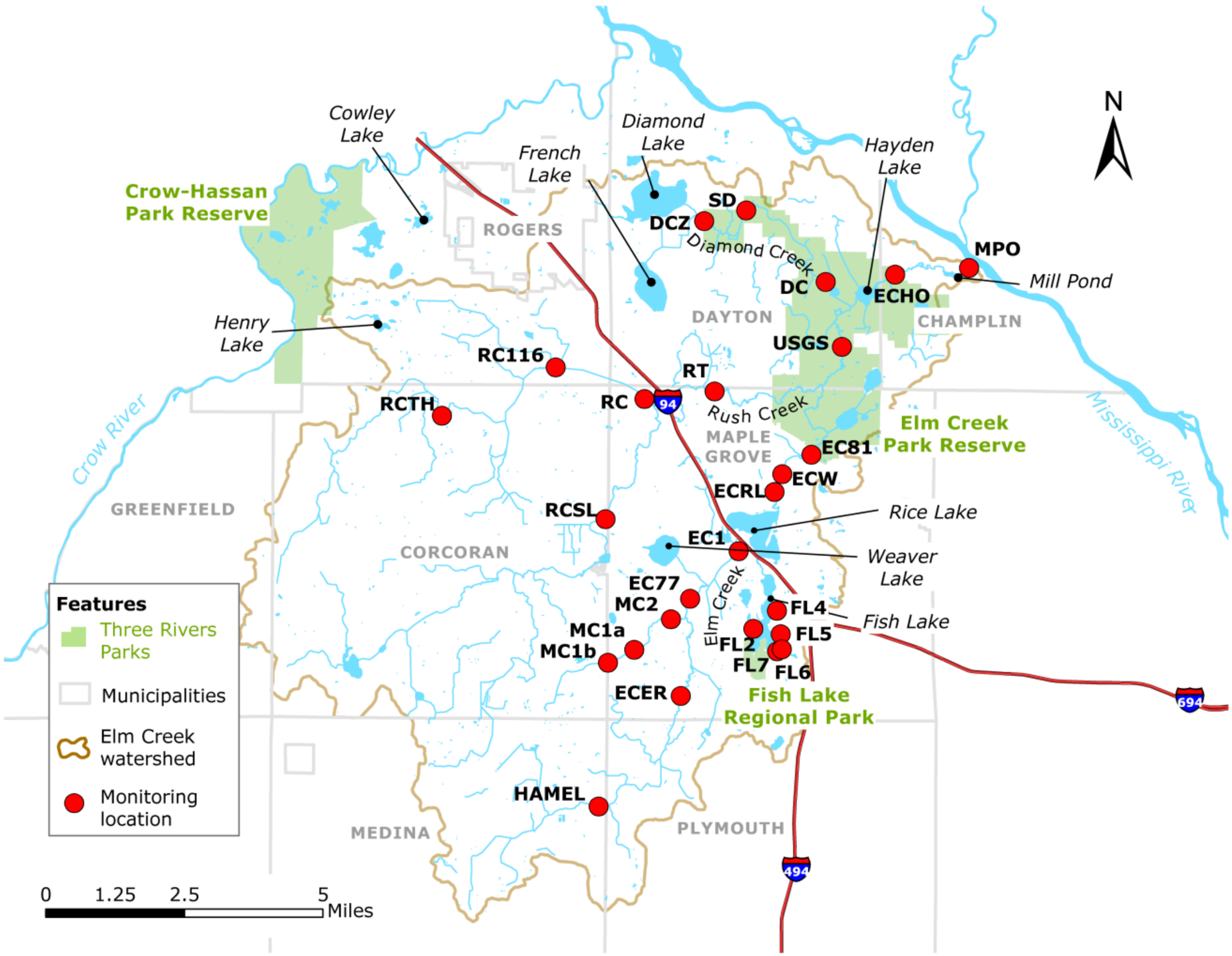


Figure 3.7. Location of stream and stormwater sampling sites throughout the Elm Creek Watershed

Table 3.3. Summary of the parameters (long-term averages) and historical timeline of data collected for stream and stormwater sites within the Elm Creek Watershed. Blank spaces indicate an absence of data. All site locations are depicted in Figure 3.7 above. *Adopted from the Elm Creek Watershed Management Commission, et. al. (2011)* .

			Laboratory Parameters																						Field Parameters									
		Data Years	TN		NH4		NO3		TP		SRP		TS		TSS		VSS		Chl		BOD		E. Coli		DO		Temp		Cond		pH		t-Tube	
Site Name	Impairment		mg/L	n	mg/L	n	mg/L	n	mg/L	n	mg/L	n	mg/L	n	mg/L	n	mg/L	n	mg/L	n	mg/L	n	100ml	n	mg/L	n	Deg C	n	us/cm	n		n	cm	n
Lower Elm Creek																																		
ECRL		1	3.86	11					0.456	11	0.247	11			21.45	11			118.7	22	6.5	11	138	29										
ECW		1	1.77	16					0.211	16	0.095	16			13.72	16	8.4	15	86.7	16	5.0	12	73	30										
EC81		3	1.99	76					0.282	79	0.139	69			18.90	76	6.7	33	110.0	43	3.8	28	202	59										
USGS	DO; E. Coli;	23	1.74	267	0.20	681	1.56	205	0.170	831	0.230	73	341.00	65	20.00	419			52.2	204					8.4	243	10.3	265	568.0	326	7.7	322		
USGS (TRPD)		2	1.06	41					0.165	41	0.107	41			3.43	41	< 4	19	51.6	41	1.6	27	121	59			10.2	20					60.0	20
ECHO		2	1.71	60					0.345	60	0.210	56			6.48	60	< 4	28	75.2	42	3.0	29	97	59			10.6	20					42.7	20
MPO		2																					67	50										
Upper Elm Creek																																		
Hamel		7	1.75	182	0.10	32	0.70	14	0.243	182	0.091	129	205.00	4	36.73	158	3.4	40	182.9	76	2.1	29	399	97	6.7	24	14.0	23	1421.3	22	7.3	24	85.5	21
ECER		7	1.62	159	0.07	31	1.21	13	0.306	160	0.130	117	505.33	3	39.25	137	7.8	33	69.3	69	2.7	28	369	91	7.5	20	13.7	20	565.4	20	7.4	20	93.2	18
EC77	DO; E. Coli; Chloride (?)	4	1.43	129					0.222	131	0.106	85			41.39	132	7.1	35	89.2	71	2.5	28	297	99	8.6	22	14.7	22	572.9	21	7.3	21	96.1	24
MC1A and MC1B *		2	2.73	17					0.383	20	0.183	12			16.42	16																		
MC2		2	1.51	29					0.167	32	0.068	23			26.99	29																		
EC1		3	1.23	45					0.209	48	0.143	40			12.33	46																		
Rush Creek																																		
RT		4	1.79	97					0.391	99	0.243	62			24.92	99	< 4	27	69.7	70	2.7	29	137	99	6.8	25	14.8	25	465.8	22	7.4	24	77.0	36
RCSL / RC101*		4	2.09	76					0.571	76	0.400	49			83.66	75	< 4	17	148.8	60	2.3	24	449	97	4.3	23	14.4	22	1105.1	21	7.2	22	70.0	21
RC116	DO; E. coli; Biota	4	3.15	61					0.490	61	0.274	35			33.76	60	< 4	17	60.2	62	4.7	24	398	98	5.7	23	16.0	22	572.7	23	7.3	21	67.1	19
RCTH		4	2.39	46					0.518	46	0.213	32			21.09	43	< 4	17	75.2	47	2.5	22	427	70	5.7	11	14.5	9	649.0	8	7.4	9	77.3	12
RC		2	1.69	12					0.362	12	0.301	10			3.64	15																		
Diamond Creek																																		
DC		4	1.35	96					0.209	100	0.126	64			15.54	96	< 4	25	39.6	71	2.3	26	340	99	9.2	24	14.3	24	512.1	23	7.4	24	88.7	36
SD	E. coli	3	3.07	38					0.310	38	0.144	14			9.60	38			59.4	40	2.6	12	504	66	5.8	22	14.3	22	472.8	20	7.4	20	80.1	27
DCZ		4	4.24	60					0.440	60	0.115	35			57.06	60	6.7	19	75.9	62	3.2	23	604	94	5.9	23	13.9	22	461.6	21	7.4	22	23.5	29
Fish Lake																																		
FL2		1							0.232	16	0.122	16																						
FL4		1							0.589	11	0.233	11																						
FL5	Nutrients	1							0.502	8	0.283	8																						
FL6*		2	1.32	7					0.158	27	0.395	21			6.30	4																		
FL7		1							0.422	16	0.175	16																						

**indicates the presence of multiple locations for a single sampling site (data have been averaged across site)*

4. Evaluation of Candidate Causes

4.1. Candidate Cause #1 – Altered Hydrology

4.1.1. Data Evaluation

Assessment of hydrologic conditions throughout the Elm Creek Watershed is based on a variety of data sources (see Section 2.1 for additional data). The most commonly referenced work related to the hydrology of the Elm Creek system in the "Elm Creek Channel Study" (Bonestroo 2007). As part of this study, the investigators predicted flooding frequency and magnitude under a range of precipitation conditions at 45 channel cross-sections throughout the watershed using a HydroCAD model (based on an original TR-20 model). Based on this work, the investigators conclude that hydrologic modification is widespread throughout the Elm Creek Watershed and that the occurrences of peak flows that will exceed bankfull conditions are, and will likely remain, common.

To further assess the magnitude of hydrologic alteration throughout the Elm Creek Watershed, precipitation, discharge and hydrograph patterns were compared between the time periods of 1978-1988 and 2003-2013. All analyses were performed using daily average precipitation and discharge data, which were collected from 1978-1988 and 2003-2013 from the Minneapolis-St. Paul Airport (MSP) record and USGS stream gage in lower Elm Creek. *Note: Given that the discharge and precipitation data are in daily averages and that the MSP weather station is approximately 20 miles to the south of the Elm Creek watershed, precise, quantitative comparisons of temporal changes in hydrologic response is not possible. However, the qualitative patterns in the precipitation-hydrograph response provide insight to support or refute the assessment of hydrologic alteration in the Elm Creek watershed over time.*

Temporal Changes in Precipitation and Discharge Patterns

To compare the relationship between precipitation and stream discharge patterns, annual averages were compared for the relative occurrence of precipitation and discharge conditions of different magnitudes. Precipitation patterns were summarized in terms of the relative occurrence of daily precipitation accumulation values ranging from zero to three inches. Discharge patterns were summarized in terms of the relative occurrence of daily average discharge values ranging from 0.5 to 300 cubic feet per second.

Annual precipitation and discharge patterns varied significantly between the time periods of 1978 to 1988 and 2003 to 2013 (Table 4.1). In the 1978 and 1988 period, annual average precipitation was greater than from the time period between 2003 and 2013, while the relative occurrence of different sized precipitation events was relatively consistent between time periods. Conversely, the annual, daily-average discharge from the 1978 to 1988 period was less than from 2003 to 2013 period, while the occurrence of days with high and low flows was greater from the 2003 to 2013 period. These observations are consistent with altered hydrologic conditions, where the occurrence of both high and low flow increases in response to altered land use. The observations of increase low flow conditions are also consistent with the increased groundwater extraction described in Section 2.1.

Table 4.1. Change in annual precipitation and discharge patterns in the Elm Creek Watershed between the time periods 1978-1988 and 2003-2013

Precipitation			
Metric	**78 to 88**	**03 to 13**	**RPD**
Annual Ave.	30.5	27.7	-10%
Days > 0	1160	1083	-7%
Days > 0.1	585	516	-13%
Days > 0.5	178	173	-3%
Days > 1	56	58	3%
Days > 2	12	10	-20%
Days > 3	1	1	0%

Discharge			
Metric	**78 to 88**	**03 to 13**	**RPD**
Ave. Daily CFS	31	40	23%
CFS < 0.5	11	9	-22%
CFS < 1	50	238	79%
CFS < 2	184	851	78%
CFS < 3	518	1229	58%
CFS > 100	339	424	20%
CFS > 200	109	134	19%
CFS > 300	49	50	2%

Temporal Changes in Hydrograph Pattern

Temporal change in hydrograph patterns was analyzed by comparing the storm hydrographs associated with similarly size precipitation events that occurred between the two time periods 1978 to 1988 and 2003 to 2013 (Figure 4.1). Within each time period, similarly sized precipitation events (representing small, moderate and large precipitation events) were identified and plotted along with the corresponding discharge records. When possible, precipitation events that were preceded by base flow conditions were selected.

The discharge record from the USGS gage in lower Elm Creek suggests that hydrologic processes in the Elm Creek Watershed have been significantly altered over the last 30 years. Across all three precipitation regimes, the discharge pattern is consistent, transitioning from a relatively stable hydrograph during the 1978-1988 period to a relatively flashy hydrograph during the 2003-2013 period. As described above, as a stream becomes flashier, water enters the channel more quickly. As such, the "leading edge" of the hydrograph becomes steeper, as discharge increases more rapidly—this pattern can be seen in all three hydrographs from the 2003-2013 period. Similarly, as groundwater recharge and evaporative/interception-based losses are reduced, a larger percentage of precipitation enters directly into the stream channel, resulting in higher peak flows. Peak flows in hydrographs from the 2003-2013 period are substantially increased over 1978-1988 period, such that even proportionally larger precipitation events in the 1978-1988 period resulted in smaller peak flows than were observed in the 2003-2013 period.

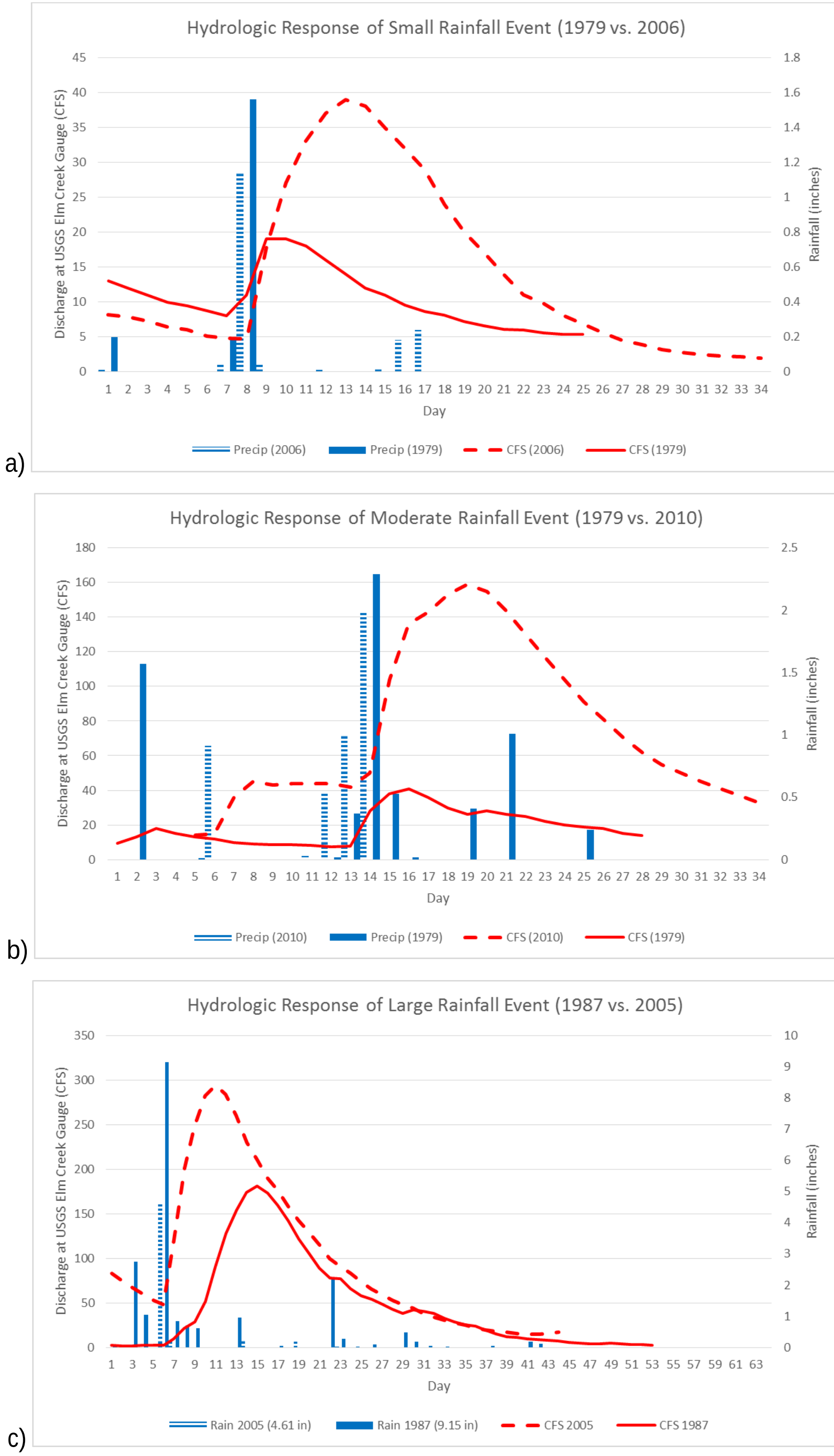


Figure 4.1. Hydrographic responses of Elm Creek to similar sized precipitation in the 1978 - 1988 period versus the 2003 - 2013 period

4.1.2. Stressor Pathway

Altered hydrologic processes in streams have a variety of direct and indirect impacts to biota (e.g., Roy et al. 2005). Ironically, although hydrologic alteration is a primary source of a range of biotic stressors, the direct impacts of altered hydrology are relatively limited. As a result, precise endpoints that selectively diagnose altered hydrology as a stressor to biological communities have not been identified. However, there are a range of endpoints and metrics in both macroinvertebrate and fish assemblage data that would be expected if hydrology had been altered within a stream ecosystem (http://www.epa.gov/caddis/ssr_flow4s.html).

Biotic response to altered hydrology can be characterized with respect to flow as an agent of disturbance and the resulting hydrologic regime (Poff and Allan 1995). As a stream becomes flashier, flood-based disturbance increases in magnitude, and often frequency. As the stream begins to reform and stabilize around a new hydrologic regime, the resulting flow regime is generally characterized by short periods in which high velocity flow exists across most reaches of the stream and longer periods in which low velocity flow dominates most reaches.

Response to this altered flow is dependent on organismal life-history (e.g., Poff and Allan 1995). For relatively sedentary organisms (e.g., macroinvertebrates), that cannot easily migrate back to a particular stream reach following a flood event, flashy conditions favor taxa that can retain position within stream habitat in high flow scour events (e.g., clingers and sprawlers). For organisms, like fish, that are more mobile and reestablish habitat occupancy following flood-based disturbance, conditions favor taxa that can survive across a range of habitat conditions (e.g., generalists) and prefer low-flow conditions. Across both macroinvertebrates and fish assemblages, flashy streams create conditions that favor r-selected species that are short-lived, prolific spawners and compete well in disturbed habitat (e.g., pioneer species).

4.1.3. Biological Communities

Biological assemblages observed in the Elm Creek Watershed are generally consistent with those expected in streams with altered flow regimes (Figure 4.2 to Figure 4.4). However, the magnitude and scope of these responses varies across AUIDs.

In Elm Creek (AUID 508), the structure of both fish and macroinvertebrate assemblages are consistent with altered hydrology. Fish assemblages at both sites in Elm Creek are disproportionately dominated by younger individuals ("MA>1Pct"; "MA>2Pct") from generalist ("GeneralPct") and pioneer ("PioneerPct") taxa and individuals representing long lived ("LLvdPct"; "MA>3Pct"; "MA>4Pct"), meager ("MSpnPct"), or sequential spawning taxa ("SSpnPct") were less common, as compared to corresponding unimpaired sites (Figure 4.3; although this response was somewhat variable between sites). Similarly, individuals from taxa representing clingers ("ClingerPct") and sprawlers ("SprawlerPct"; to a lesser degree) represent a disproportionately high percentage of the sampled assemblages, while individuals from climber taxa ("ClimberPct") were relatively uncommon, as compared to unimpaired sites (Figure 4.2). There was significant variability in metric responses between sites, suggesting that the impacts of altered hydrology may vary throughout AUID 508.

Fish assemblages in Elm Creek (AUID 508) are also generally consistent with those likely to be observed in flashy streams, dominated by prolonged periods of low flow (Figure 4.4). At both sites, relatively few individuals were sampled from taxa that prefer moderate ("MorFnotSH20Pct") and fast-flowing ("FnotSH20Pct") waters, while individuals that represent taxa specific to non-lentic waters ("Non-LacustrinePct") were common. However, although trends in metric response were consistent in both sites in Elm Creek, observed values were within the interquartile range of responses observed at unimpaired sites, suggesting that the cumulative impact of altered hydrology may be relatively moderate.

Responses of fish and macroinvertebrate communities in Rush Creek were similar to Elm Creek. However, the magnitude and consistency of the response of hydrologically sensitive endpoints was generally reduced and more varied among AUIDs, particularly in macroinvertebrate assemblages. Similar to Elm Creek, fish assemblages in AUIDs 760 and 528, were generally dominated by younger, shorter lived species, but assemblages in AUID 732 were more consistently dominated by older, longer-lived species (Figure 4.2). Similarly, relatively few, if any, individuals from taxa that prefer fast-flowing water were observed (Figure 4.4), although the range of responses observed at unimpaired sites for these metrics is highly variable. The magnitude of the macroinvertebrate assemblage response in Rush Creek was similar to Elm Creek, but varied across sites. Invertebrate assemblages in all AUIDs in Rush Creek were dominated by clingers and sprawlers, but this pattern was least pronounced AUID 760 (Figure 4.3).

The response of hydrologically sensitive endpoints in Diamond Creek (AUID 525) was mixed between fish and macroinvertebrate assemblages. Fish assemblages were relatively balanced with respect the occurrence of individuals from different age groups and life-histories (Figure 4.2). However, no fish that are meager or sequential spawners or prefer moderate and fast moving water were sampled. The general response of macroinvertebrate assemblages was similar to Elm Creek, as it was dominated by individuals from clinger and sprawler groups, while individuals from climber taxa were relatively uncommon (Figure 4.3). However, most metric scores observed in Diamond Creek invertebrate assemblages were within the interquartile distributions observed at unimpaired sites.

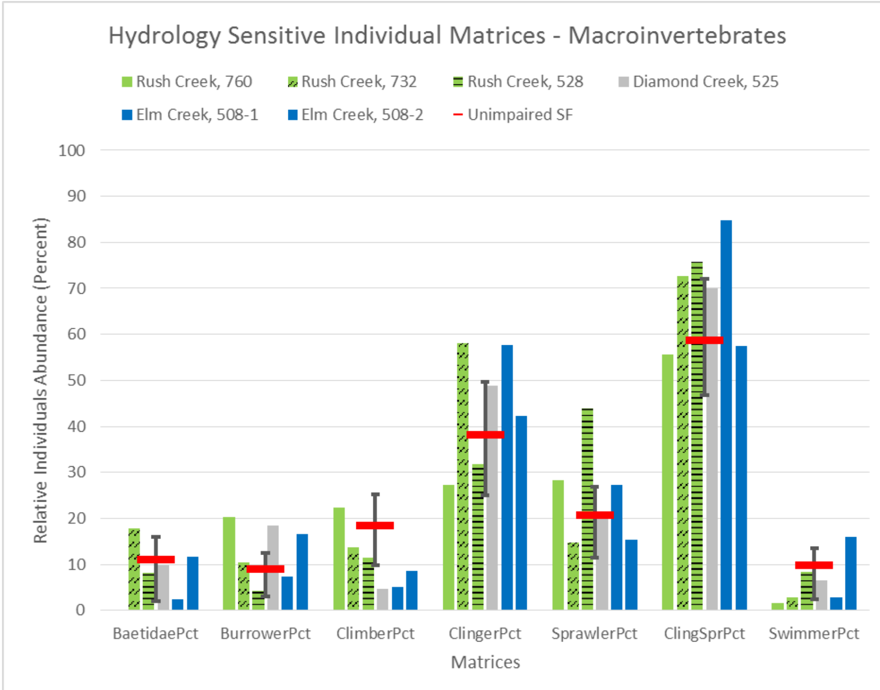


Figure 4.2. Response of macroinvertebrate individual metrics likely impacted by hydrologic disturbance and regime. Horizontal bars represent average metric responses and "whiskers" represent upper and lower quartiles at unimpaired sites.

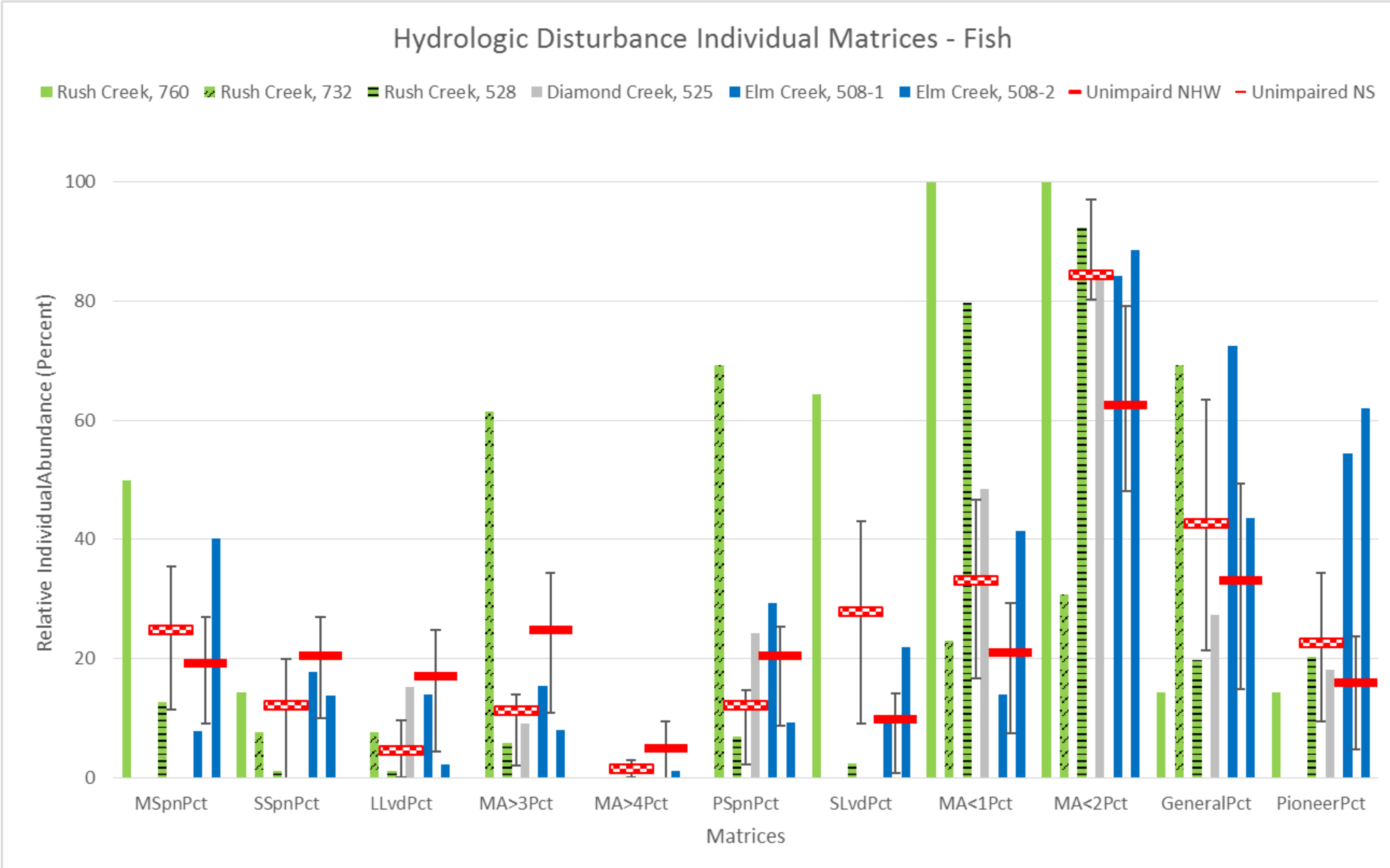


Figure 4.3. Response of fish assemblage metrics likely impacted by hydrologic disturbance in each AUID as compared to unimpaired reaches throughout the Northern Headwaters stream classification ("Unimpaired NHW") for Rush and Diamond Creek and the Northern Streams classification ('Unimpaired NS") for Elm Creek. Horizontal bars represent average metric responses and "whiskers" represent upper and lower quartiles at unimpaired sites.

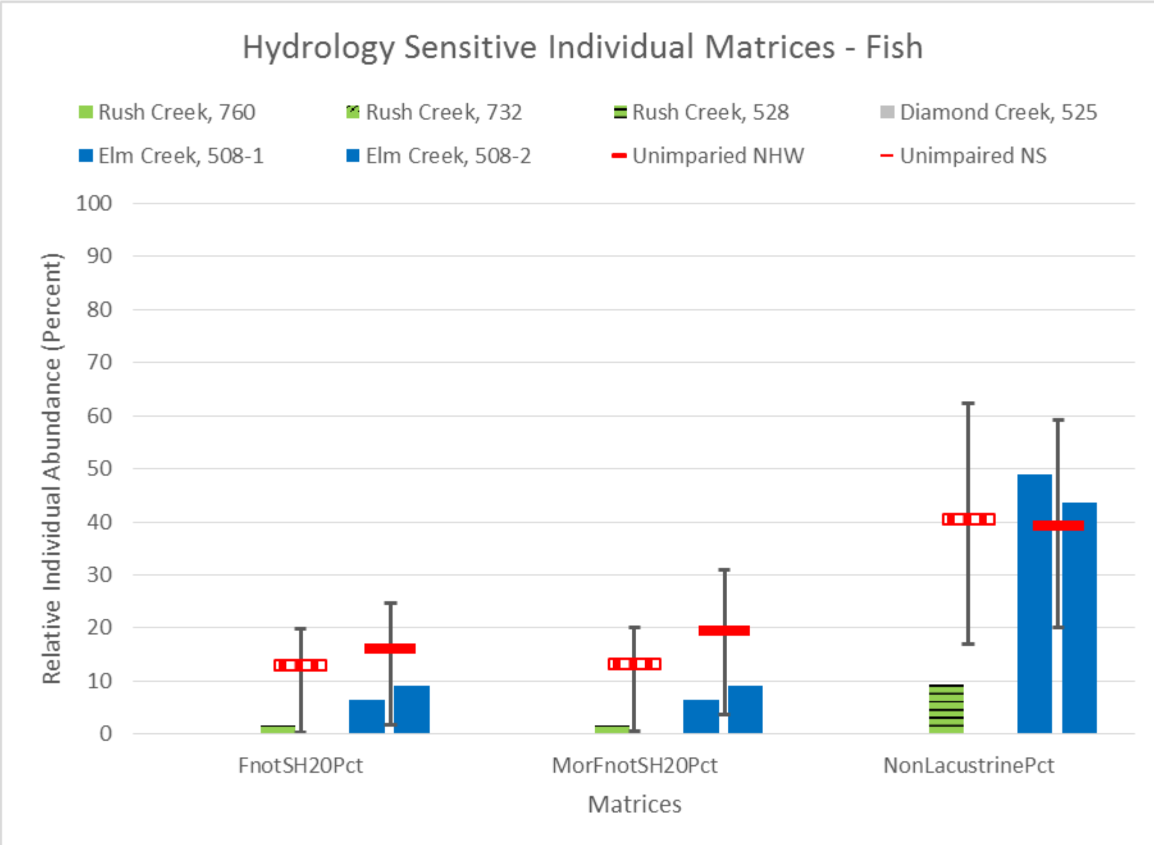


Figure 4.4. Response of fish assemblage metrics likely impacted by hydrologic regime in each AUID as compared to unimpaired reaches throughout the Northern Headwaters stream classification ("Unimpaired NHW") for Rush and Diamond Creek and the Northern Streams classification ('Unimpaired NS") for Elm Creek. Horizontal bars represent average metric responses and "whiskers" represent upper and lower quartiles at unimpaired sites.

4.1.4. Summary of the Strength of Evidence

Evidence for biotic impacts from altered hydrology is relatively common across all AUIDs (Table 4.2). Land use and precipitation patterns necessary to alter stream hydrology have clearly shifted over the last 30 years (see Section 2.1). Concurrently, discharge patterns in Elm Creek have shifted to a flashier hydrologic regime (Table 4.1 and Figure 4.1) and the physical and chemical stressors (see Sections 4.2, 4.3 and 4.4) that commonly result from altered hydrology have increased throughout the watershed. Given the lack of historical biological data, biotic assemblages within Elm Creek cannot be analyzed for temporal response. However, the current fish and macroinvertebrate assemblages are generally consistent with those commonly observed in flashy hydrologic systems, in which high-flow events episodically scour stream habitat, but the majority of the hydrograph is dominated by low-flow conditions (Figure 4.2 and Figure 4.4). Hydrologically-induced shifts in fish assemblage composition are most pronounced and consistent across AUIDs, while the response of macroinvertebrate metrics is less pronounced and consistent across AUIDs, suggesting that flow-regime structure may be a more significant driver of community structure than peak erosive potential and that habitat conditions may vary within Elm Creek watershed (see Section 4.2 for further discussion). Among AUIDs, the response to altered hydrology is most pronounced in 732 and 528 and least pronounced in 508, 760 and 525. Although biotic responses are consistent with observed effects from hydrologic alteration and divergent from corollary biotic assemblages in unimpaired systems, all of these biological endpoints are potentially affected by multiple stressors. Despite the lack of a selective, diagnostic endpoint for impacts to biota from altered hydrology, the SOE suggests that altered hydrology is a primary stressor for AUIDs 525 (fish), 528 (fish and macroinvertebrates), 732 (fish and macroinvertebrates) and 760 (fish and macroinvertebrates) and a secondary stressor in AUIDs 508 (fish and macroinvertebrates), 525 (macroinvertebrates) and 760 (macroinvertebrates).

Table 4.2. Summarizes the weight of evidence supporting altered hydrology as a causative stressor for the biological impairments observed throughout the Elm Creek Watershed -- See Appendix A for definitions

Strength of Evidence Table -- Altered Hydrology					
	Scores for Impaired Reaches				
	Elm Creek	Rush Creek			Diamond Creek
Types of Evidence	508	528	732	760	525
Spatial/Temporal concurrence	+	+	+	+	+
Temporal sequence	0	0	0	0	0
Evidence of exposure, biological mechanism	+	+	+	+	+
Causal pathway	++	++	++	++	++
Field evidence of stress response	+	+++	+++	++	+
Field experiments/manipulations of exposure	NE	NE	NE	NE	NE
Laboratory analysis of site media	NE	NE	NE	NE	NE
Verified or tested predictions	+	+	+	+	+
Symptoms	+	+	+	+	+
Mechanically plausible cause	+	+	+	+	+
Stressor-response in other field studies	+	+	+	+	+
Stressor-response in other lab studies	NE	NE	NE	NE	NE
Stressor-response in ecological models	NE	NE	NE	NE	NE
Manipulation experiments at other sites	NE	NE	NE	NE	NE
Analogous stressors	NE	NE	NE	NE	NE
Consistency of evidence	+	+	+	+	+
Explanatory power of evidence	+	+	+	+	+

4.2. Candidate Cause #2 – Altered Physical Habitat

4.2.1. Data Evaluation

Habitat alteration in the Elm Creek Watershed is described most comprehensively by Dindorf and Miesbauer (2002); locally referred to as the "Elm Creek Habitat Study". This project assessed stream habitat conditions throughout 60 stream-miles, across 45 reaches within Elm, Rush and Diamond Creeks. Habitat assessment sites overlapped with all AUIDs currently listed for biological impairment in the Elm Creek Watershed. Across all sites, instream habitat and channel stability were assessed using a Rosgen methodology. Habitat data from Dindorf and Miesbauer (2002) were summarized for each stream system with respect to channel stability, aquatic habitat condition and sediment type based on the relative occurrence of different classifications categories (Table 4.3).

Stream habitat was also assessed by the MPCA at each assessment stream reach at the time samples were collected as part of the biotic assessment. The MCPA assessment process implements the Minnesota Stream Habitat Assessment (MSHA) and Channel Condition and Stability Assessment (CCSA) to describe all stream reaches that correspond to biotic assessment sites. Results from the Elm Creek assessment report (MCPA 2013) are described below (Table 4.4 and Table 4.5).

Table 4.3. Habitat conditions and sediment types as measured by Dindorf and Miesbauer (2002)

Stream	Number of Reaches Surveyed	Channel Stability					Aquatic Habitat			Sediment Type				
		Poor	Poor-Fair	Fair	Fair-Good	Good	Poor	Fair	Good	Clay	Silt	Sand	Gravel	Cobble
Elm Creek	24	7	3	10		3	3	18	3		5	16	3	
Rush Creek (Main)	12	3	1	3	1	4		9	3		8	4		
Rush Creek (South)	11	5		4		2	1	10			2	9		
Diamond Creek	9	2		3	1	2	3	5			4	5		
Total	**56**	**17**	**4**	**20**	**2**	**11**	**7**	**42**	**6**	**0**	**19**	**34**	**3**	**0**

Table 4.4. The MSHA for Elm Creek Watershed

Biological Station ID	Reach Name	Land Use (0-5)	Riparian (0-15)	Substrate (0-27)	Fish Cover (0-17)	Channel Morph. (0-36)	MSHA Score (0-100)	MSHA Rating
10UM008	Diamond Creek	5	13	10.8	15	20	63.8	Fair
10UM014	Rush Creek, South Fork	0	10	7	13	8	38	Poor
10UM011	Rush Creek, South Fork	0	10	10.8	16	26	62.8	Fair
99UM081	Rush Creek	3.5	11.5	13.4	13	24	65.4	Fair
10EM167	Elm Creek	4.3	12	16.5	12	17	61.8	Fair

Note: Biological Station ID # 00UM085 was not sampled at the time of biological survey.

Table 4.5. Channel Condition and Stability Assessment (CCSA) for Elm Creek Watershed

Biological Station ID	Stream Name	Upper Banks (4-43)	Lower Banks (5-46)	Substrate (3-37)	Channel Evolution (1-11)	CCSI Score (13-137)	CCSI Rating
10UM008	Diamond Creek	6	18	26	3	43	Fairly stable
10UM014	Rush Creek, South Fork	8	13	13	5	39	Fairly stable
10UM011	Rush Creek, South Fork	18	32	27	3	80	Moderately unstable
99UM081	Rush Creek	24	5	8	3	40	Fairly stable
10EM167	Elm Creek	13	18	8	5	44	Fairly stable

Note: Biological Station ID # 00UM085 was not sampled at the time of biological survey.

Results from both the MPCA habitat assessments and Dindorf and Miesbauer (2002), suggest that habitat throughout the Elm Creek Watershed has been significantly altered and is in “Fair” to “Poor” condition with relatively limited substrate diversity. Across both studies, stream channel stability was classified as “Fairly” to “Moderately” stable. In general, Elm Creek and the main stem of Rush Creek had the highest levels of habitat quality and stability and Diamond Creek and the south fork of Rush Creek had proportionally lower habitat quality and stability. However, the lack of diversity in habitat quality and stability scores prevents a gradient analysis of habitat impacts on biota.

4.2.2. Stressor Pathway

Physical habitat alteration can impact biotic communities through a variety of process (http://www.epa.gov/caddis/ssr_phab4s.html). As habitat becomes more homogenous, so does the associated biological communities (e.g., Wang 1997). Stream assemblages that occupy reaches with degraded habitat are generally less species rich and diverse than those from reaches with unimpaired habitat (e.g., Lau et al. 2006). In general, biological communities become dominated by generalist species that utilize fine particle substrate types, emergent vegetation and are adapted to low-flow conditions. Although, species richness and diversity in these degraded stream reaches generally decreases, total biomass can often increase in response to concurrent increases in nutrient loads and reductions in interspecific competition. Given the overlap between biotic responses to physical habitat alteration and hydrologic alteration, discrete attribution of changes in biotic assemblage structure to either of these stressors is difficult. Instead of assigning specific attribution of assemblage changes to habitat alteration, assemblage structure will be described with respect to its consistency with altered habitat conditions.

4.2.3. Causal Analysis of Biological Response

Biological assemblages observed in the Elm Creek watershed are inconsistent with those expected in streams with altered physical habitat (Figure 4.5 - Figure 4.10). However, this response is highly variable among AUIDs.

Evidence for physical habitat impacts varies between fish and macroinvertebrate assemblages in Elm Creek (AUID 508). At both sampling sites, the richness of fish taxa (Figure 4.5) is somewhat reduced relative to corresponding unimpaired reaches (although within the interquartile distribution), while the richness and diversity of macroinvertebrate assemblages varied between sites, relative to unimpaired reaches (Figure 4.6). The relative dominance of different taxa in macroinvertebrate (“Dom___ChPct”) and fish assemblages (“Dom___Pct”) are generally consistent with unimpaired sites (Figure 4.8 and Figure 4.9). However, the occurrence of riffle dependent fish taxa (“RifflePct”) and clinger invertebrate taxa are divergent from the corresponding unimpaired sites. Taken together, these observations suggest that habitat alteration may have a larger impact on fish assemblages than macroinvertebrates in Elm Creek, but these divergences from unimpaired sites are relatively small and potentially explain by co-occurring stressors (e.g., altered hydrology, see Section 4.1).

Biological evidence for altered habitat is more pronounced in Rush Creek, but variable across assemblages and AUIDs. In general, the diversity and richness of fish and macroinvertebrate assemblages are reduced (Figure 4.5-Figure 4.7) and the proportional dominance of individual fish taxa is increased, relative to corresponding unimpaired sites (except AUID 528). In AUID 528, the relative diversity and richness of fish and macroinvertebrate taxa are consistent with unimpaired reaches, while the relative dominance of individual fish taxa is consistently elevated. Similarly, the relative occurrence of different macroinvertebrate habitat groups in AUID 528 is relatively consistent with unimpaired reaches, while in AUIDs 760 and 732 habitat groups are more divergent from unimpaired reaches; however, this may be a result of co-occurring stressors (e.g., altered hydrology and sediment). Interestingly, the relative dominance of individual taxa in AUID 760 is suppressed relative to unimpaired reaches, suggesting the presence of a more robust macroinvertebrate assemblage.

The biological evidence for altered physical habitat in Diamond Creek is similar to that observed in AUIDs 760 and 732, but less pronounced (most responses are within the interquartile distribution observed at corresponding reference sites)—suggesting altered habitat conditions likely have a reduced impact on the biota in this system.

Figure 4.5. Tax richness of fish assemblages throughout the Elm Creek watershed in each AUID as compared to unimpaired reaches throughout the Northern Headwaters stream classification ("Unimpaired NHW") for Rush and Diamond Creek and the Northern Streams classification ('Unimpaired NS") for Elm Creek. Horizontal bars represent average metric responses and "whiskers" represent upper and lower quartiles at unimpaired sites.

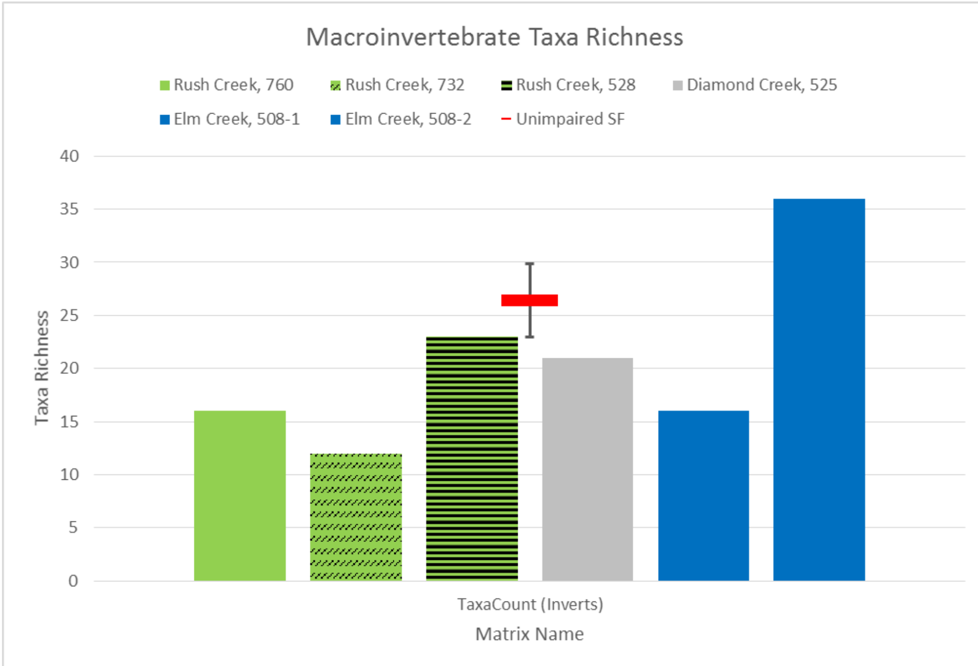


Figure 4.6. Tax richness of macroinvertebrate assemblages throughout the Elm Creek watershed in each AUID as compared to unimpaired reaches throughout the Southern Forest GP stream classification ("Unimpaired SF"). Horizontal bars represent average metric responses and "whiskers" represent upper and lower quartiles at unimpaired sites.

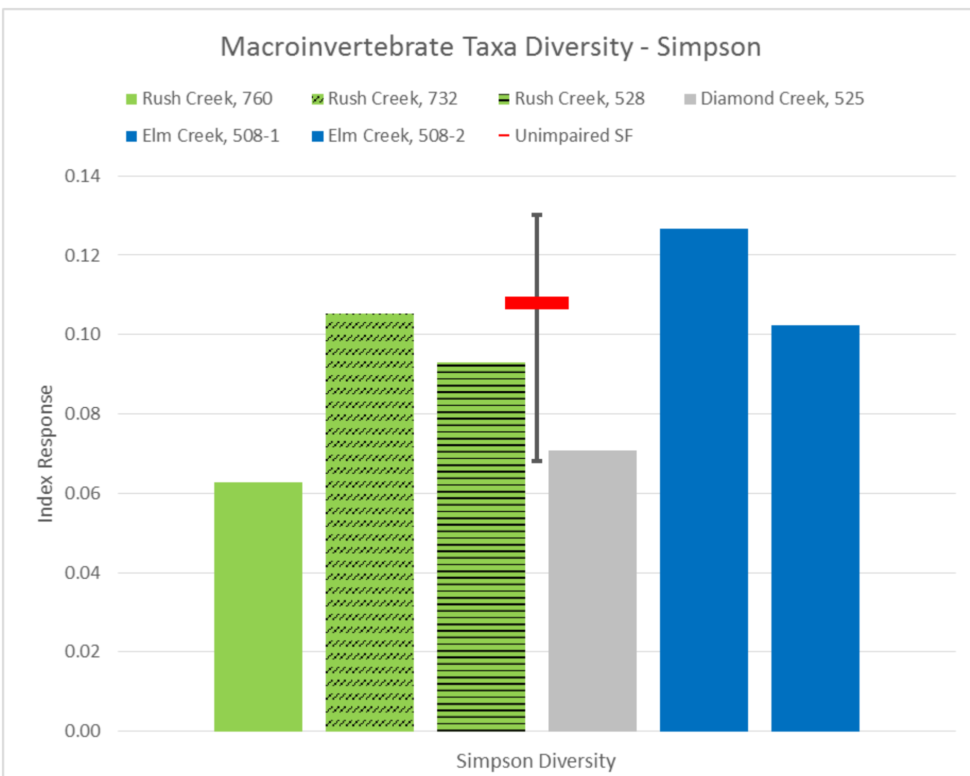


Figure 4.7. Diversity of macroinvertebrate assemblages throughout the Elm Creek watershed as measured by the Simpson Diversity index in each AUID as compared to unimpaired reaches throughout the Southern Forest GP stream classification ("Unimpaired SF") Horizontal bars represent average metric responses and "whiskers" represent upper and lower quartiles at unimpaired sites.

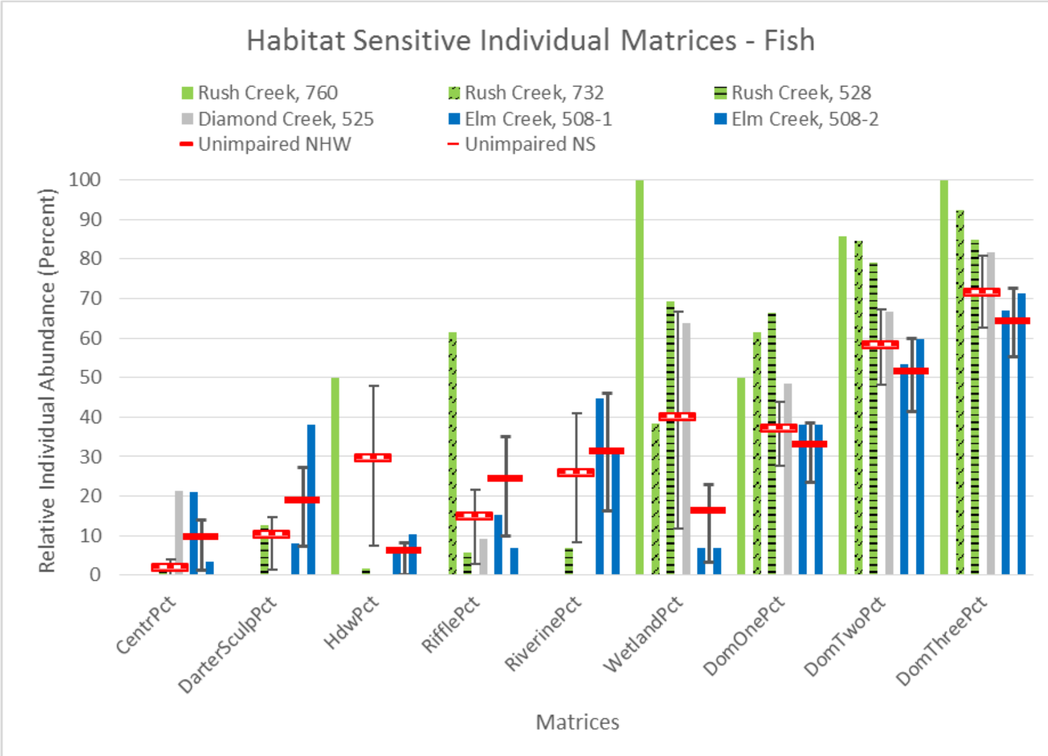


Figure 4.8. Response of fish assemblage metrics likely impacted by physical habitat alteration in each AUID as compared to unimpaired reaches throughout the Northern Headwaters stream classification ("Unimpaired NHW") for Rush and Diamond Creek and the Northern Streams classification ('Unimpaired NS") for Elm Creek. Horizontal bars represent average metric responses and "whiskers" represent upper and lower quartiles at unimpaired sites.

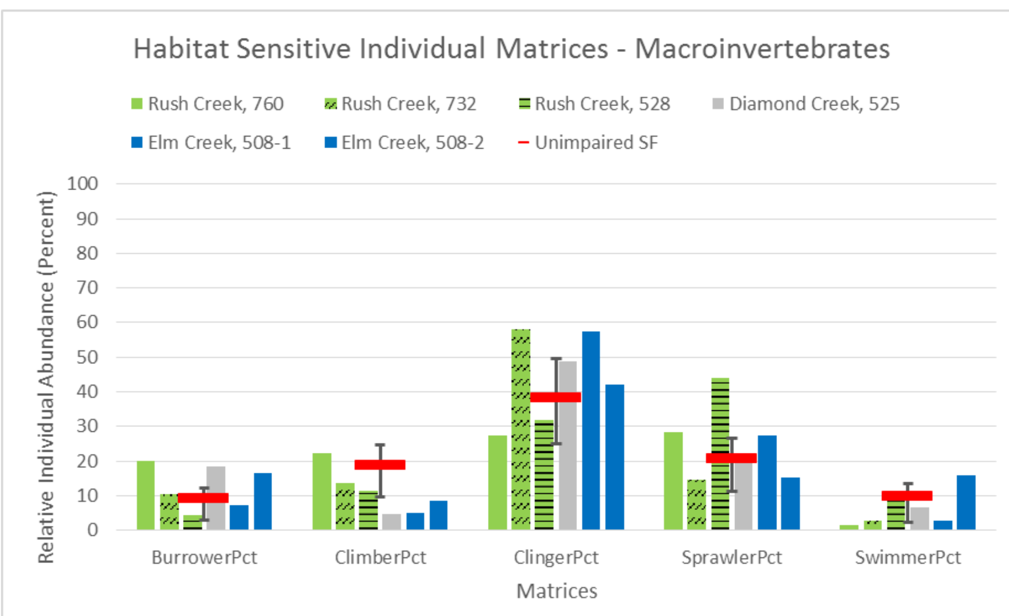


Figure 4.9. Response of macroinvertebrate assemblage metrics likely impacted by habitat alteration in each AUID as compared to unimpaired reaches throughout the Southern Forest GP stream classification ("Unimpaired SF") Horizontal bars represent average metric responses and "whiskers" represent upper and lower quartiles at unimpaired sites.

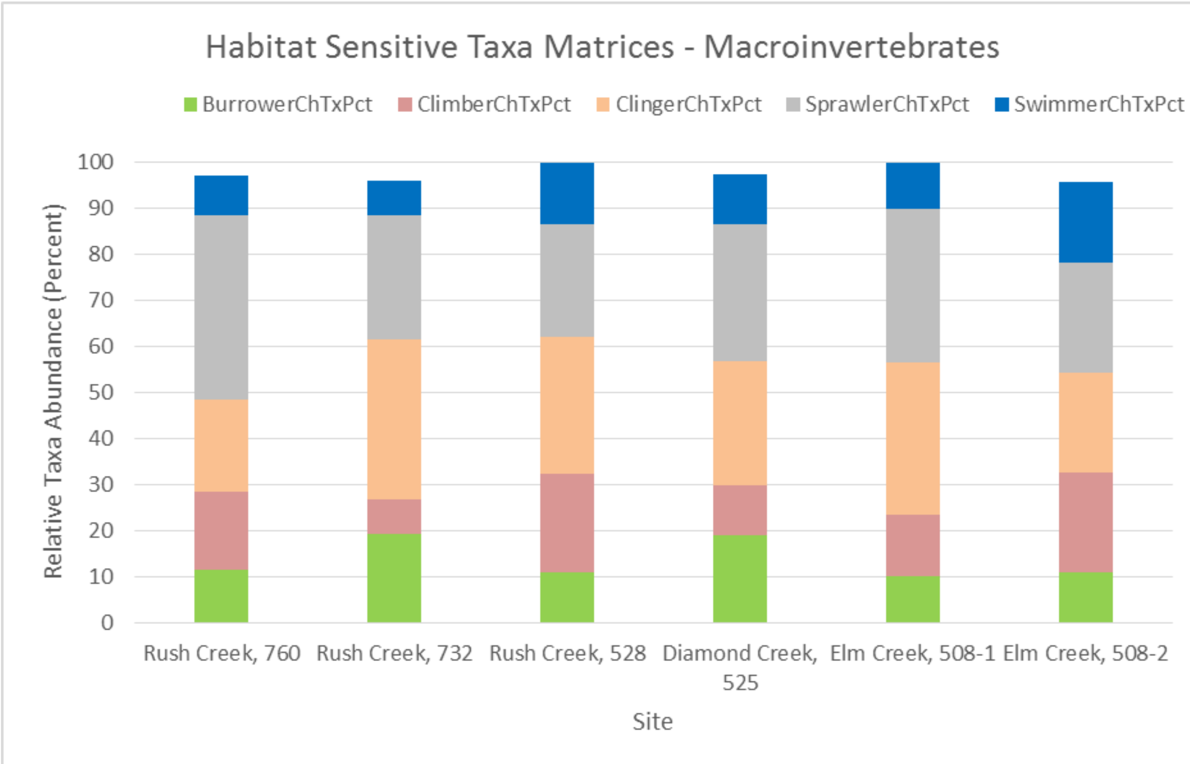


Figure 4.10. Response of macroinvertebrate behavioral group metrics likely impacted by habitat alteration in each AUID as compared to unimpaired reaches throughout the Southern Forest GP stream classification ("Unimpaired SF")

4.2.4. Summary of Strength of Evidence

Evidence for biotic impacts from physical habitat alteration is mixed across AUIDs and assemblages (Table 4.6). Altered hydrologic and land use patterns that commonly result in altered physical habitat in streams have clearly shifted throughout the watershed over the last 30 years (see Section 2.1). Concurrently, physical habitat in streams throughout the Elm Creek watershed has shifted to a more homogenous, lower quality condition (Table 4.3-Table 4.5; although areas of moderate to good habitat continue to persist in lower Elm and Rush Creeks). Given the lack of historical biological data, assemblages within Elm Creek cannot be analyzed for temporal response to physical habitat alteration. Current fish assemblages in all AUIDs (except 508 and 528) are consistent with homogenous, lower quality stream habitat conditions—richness of fish assemblages is generally reduced over unimpaired sites and dominated by relatively few taxa that utilize a limited number of habitat types (or are generalists across a range of habitat types: Figure 4.2 and Figure 4.8). However, richness, diversity and proportional dominance of taxa for macroinvertebrate assemblages in all AUIDs are relatively consistent with unimpaired sites (Figure 4.6 and Figure 4.7). Although biotic responses are consistent with observed effects from physical habitat alteration and divergent from biotic assemblages in unimpaired streams, all of these biological endpoints are potentially affected by multiple stressors. As such, no specific diagnostic endpoints exist in which altered physical habitat is selectively identified as a stressor. Despite the lack of a selective, diagnostic endpoint for habitat impacts to biota, the SOE suggests that habitat impairment is a primary stressor for AUIDs 525 (fish), 732 (fish and macroinvertebrates) and 760 (fish and macroinvertebrates) and a secondary stressor in AUIDs 508 (fish and macroinvertebrates), 525 (macroinvertebrates) and 528 (fish and macroinvertebrates).

Table 4.6. Summarizes the weight of evidence supporting habitat alteration as a causative stressor for the biological impairments observed throughout the Elm Creek watershed See Appendix A for definitions

Strength of Evidence Table -- Habiat Alteration					
	Scores for Impaired Reaches				
	Elm Creek	Rush Creek			Diamond Creek
Types of Evidence	508	528	732	760	525
Spatial/Temporal concurrence	+	+	+	+	+
Temporal sequence	0	0	0	0	0
Evidence of exposure, biological mechanism	+	+	++	++	++
Causal pathway	+	+	++	++	++
Field evidence of stress response	+	+	+++	+++	++
Field experiments/manipulations of exposure	NE	NE	NE	NE	NE
Laboratory analysis of site media	NE	NE	NE	NE	NE
Verified or tested predictions	0	0	0	0	0
Symptoms	+	+	+	+	+
Mechanically plausible cause	+	+	+	+	+
Stressor-response in other field studies	+	+	+	+	+
Stressor-response in other lab studies	NE	NE	NE	NE	NE
Stressor-response in ecological models	NE	NE	NE	NE	NE
Manipulation experiments at other sites	NE	NE	NE	NE	NE
Analogous stressors	NE	NE	NE	NE	NE
Consistency of evidence	+	+	++	++	++
Explanatory power of evidence	+	+	+	+	+

4.3. Candidate Cause #3 – Excess Sediments

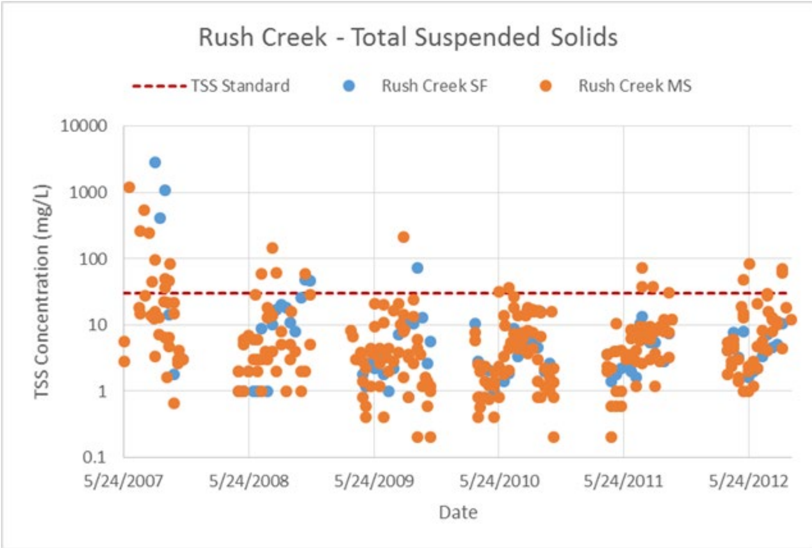
4.3.1. Data Evaluation

A total of 1438 samples have been collected to assess sediments as a potential contributor to the biological impairments throughout the Elm Creek watershed (Figure 4.11). Across all streams, the TSS standard of 30 mg/L was exceeded in 7% or more of samples collected (Table 4.7). Taken together these data suggest that, sediments consistently exceed the established water quality standards (see Section 3.3.1) and are a likely contributor to the observed biological impairments in Elm and Diamond Creeks, but potentially less important in Rush Creek.

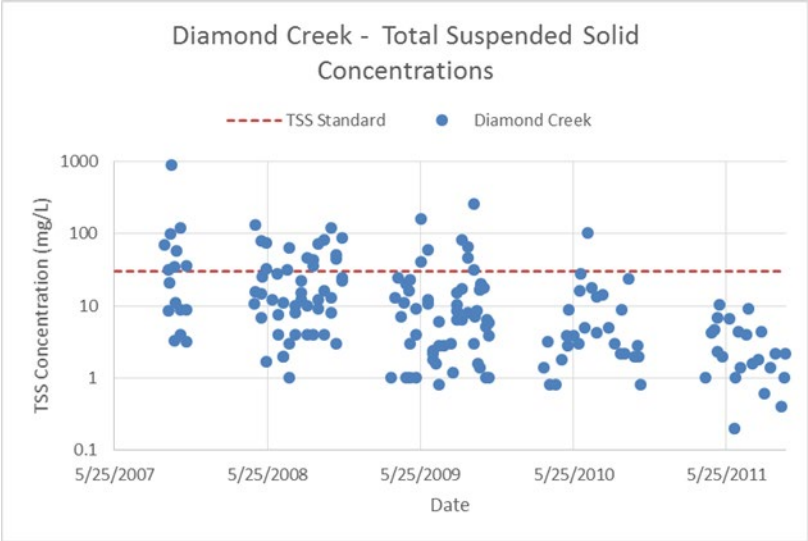
Table 4.7. A comparison of TSS measurements throughout the Elm Creek Watershed the 30 mg/L standard

Stream	AUID	TSS Standard	Total Samples	Number of Exceedences	Percent Exceedence
Elm Creek	508	30 mg/L	983	183	19%
Rush Creek (Main)	528	30 mg/L	83	6	7%
Rush Creek (South)	732; 760*	30 mg/L	290	25	9%
Diamond Creek	525	30 mg/L	82	24	29%
Total			**1438**	**238**	**16%**

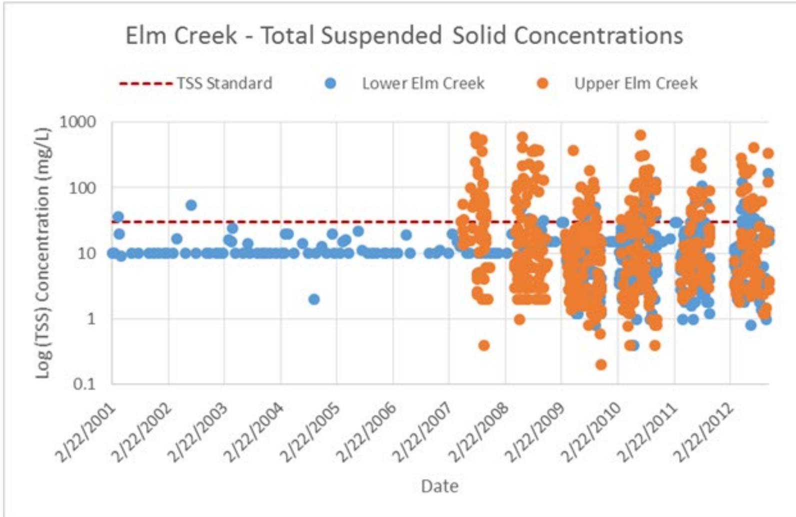
*Based on spatial proximity, water chemistry from AUID 732 is being used as a surrogate for AUID 760



a)



b)



c)

Figure 4.11. The TSS concentrations measured in a) Rush, b) Diamond and c) Elm Creeks from 2007-2009

4.3.2. Stressor Pathway

Increased sediments can affect stream biota through a variety of mechanisms (e.g., Sullivan and Watzin 2010; see http://www.epa.gov/caddis/ssr_sed4s.html). In the most extreme cases, high suspended sediment concentrations can directly affect stream organisms though gill abrasion and inhibition of sight feeding (e.g., Abrahams and Kattenfeld 1997). More commonly, sediments impact biota as they settle and modify stream bottom habitat (e.g., Rabeni and Smale 1995). Biological communities in streams have evolved to exploit the range of habitat types that exist within the diversity of sediments and substrate types—most of which are created by lateral and longitudinal variations in discharge velocities (see Section 4.2 further discussion). As stream bottoms become embedded by fine sediments, the historically heterogeneous habitat becomes homogenous with respect to particle size and habitat diversity. In response to excess sediments, stream communities often become less diverse (Lammert and Allan 1999) and dominated by species that thrive in habitats comprised of smaller particles (e.g., sand and silt). Species that require larger, hard substrate ("lithophiles") are among the first species displaced by increased sedimentation, which further reduces competition for fine particle specialists (e.g., Waters 1995).

Functional feeding group composition also commonly shifts in response to increased sedimentation. As sedimentation increases, the availability of substrate for periphyton decreases and as a result, the primary food source becomes coarse particulate organic matter (CPOM). In response, the proportion of shredder and collector-gatherer species generally increases. Interestingly, although the proportional availability of CPOM increases, the relative abundance of collector-filterers often decreases—generally because of the direct impact of sediments on filtering or the loss of substrate for filterer attachment. Common collector-filterers indicator taxa that often decrease in response to sedimentation are net spinning caddisflies and bivalves.

4.3.3. Causal Analysis of Biological Response

Biological assemblages observed in the Elm Creek watershed are consistent with high sediment conditions (Figure 4.12 - Figure 4.13). However, the response of sediment sensitive endpoints is variable between fish and macroinvertebrate assemblages.

In general, evidence for suspended sediment impacts to biota is limited and/or confounded by mixed stressor impacts. Although, sediment impacts have been observed to co-occur with changes in the relative abundance of benthic feeders, carnivores, herbivores, tolerant centrarchids, tolerant percifiromes, intolerant species, long-lived species and sensitive species, these endpoints are heavily influenced by multiple stressors (particularly elevated nutrients and altered hydrologic disturbance). Given the limited utility of these endpoints for the diagnosis of sediment-specific impacts and the relatively low suspended sediment loads observed throughout this study, this causal analysis focused on the use of endpoints that characterize sediment habitat usage by different reproductive and functional feeding groups.

Evidence for sediment impacts to biota is variable across AUIDs. In Elm Creek (AUID 508), fish assemblages at both site are relatively consistent with unimpaired sites, with non-hard substrate spawning individuals ("NestNoLithPct") being slightly over represented, as compared corresponding unimpaired sites. In all Rush Creek AUIDs (except AUID 732), fish assemblages are dominated by individuals from species that do not use hard substrate for spawning or actively modify/maintain ("CompLithPct") nests sites, while individuals from taxa that directly utilize riffle habitat ("RifflePct") require large particle sizes for spawning ("SLithopPct") are proportionally less common and divergent from unimpaired streams. In AUID 732, white sucker, a lithophilic, riffle species, comprised 60% of the sampled population (8 individuals). Fish assemblages in Diamond Creek (AUID 525) are similar to those observed in AUID 528, suggesting moderate levels of sediment impact.

The structure of macroinvertebrate assemblages is inconsistent across metrics and AUIDs with respect to sediment sensitive endpoints. In AUID 508, the increased relative occurrence of individuals from clinger, burrower ("BurrowerPct") and collector-gatherer taxa ("Collector-gathererPct") and the relative decrease in individuals from collector-filterer taxa ("Collector-filtererPct") is consistent with sediment impacts. However, this response is not consistent across sample sites within AUID 508, and is contradicted by the increased relative occurrence of net-spinning caddisflies ("HydropsychidaePct") and bivalves ("BilbalviaPct"), relative to unimpaired reaches. These results suggest that sediment impacts may be variable across reaches and generally a less important driver of biological conditions in Elm Creek.

In Rush Creek, the structure of macroinvertebrate assemblages in AUID 528 is generally consistent with sediment impacts—proportionally higher relative occurrences of collector-gatherer and sprawler taxa and proportionally lower occurrences of collector-filterer, clinger, hydropsychid and bivalve taxa. This response is similar, but less pronounced in AUID 760 and generally reversed in AUID 732, although the magnitude of the response in both 760 and 732 is generally within the interquartile distribution observed in corresponding unimpaired sites.

The structure of macroinvertebrate communities in Diamond Creek is the mirror image of what would be expected under high sediment conditions, as this site is dominated by collector-filterers, clingers, hydropsychids and bivalves, and collector-filterers are proportionally less common. Interestingly, individuals from burrower taxa are more common in Diamond Creek, which suggest higher density fine sediments.

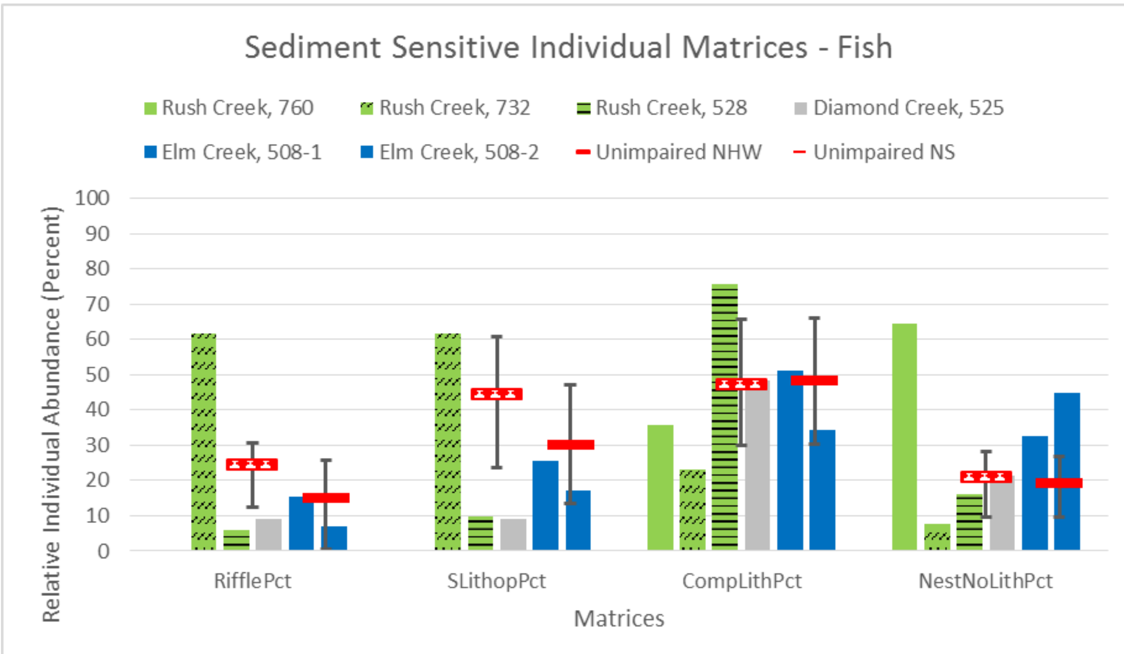


Figure 4.12. Response of fish assemblage metrics likely impacted by sediment in each AUID as compared to unimpaired reaches throughout the Northern Headwaters stream classification ("Unimpaired NHW") for Rush and Diamond Creek and the Northern Streams classification ('Unimpaired NS") for Elm Creek. Horizontal bars represent average metric responses and "whiskers" represent upper and lower quartiles at unimpaired sites.

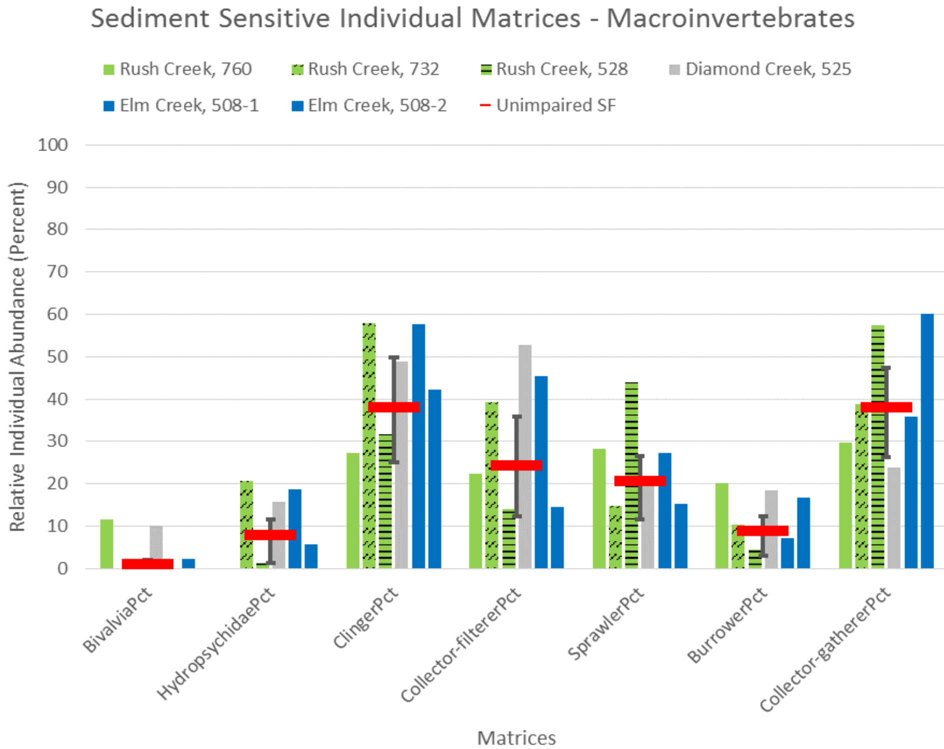


Figure 4.13. Response of macroinvertebrate assemblage metrics likely impacted by sediment in each AUID as compared to unimpaired reaches throughout the Southern Forest GP stream classification ("Unimpaired SF") Horizontal bars represent average metric responses and "whiskers" represent upper and lower quartiles at unimpaired sites.

4.3.4. Summary of Strength of Evidence

Evidence for biotic impacts from excess sediment is mixed across AUIDs and biotic assemblages (Table 4.8). Altered hydrologic and land use patterns that commonly result in excess stream sediment have clearly shifted over the last 30 years (see Section 2.1). Concurrently, the occurrence of elevated suspended sediment levels and fine grain bed sediments is often above established standards (although this response is most pronounced in upper Elm Creek and has been decreasing throughout Rush and Diamond Creeks over the period of record). Given the lack of historical biological data, assemblages within the Elm Creek Watershed cannot be analyzed for temporal response to sedimentation. Current fish assemblages in all AUIDs (except 508 and 732) are consistent with those commonly observed in streams with high suspended sediment loads and highly embedded stream sediments—lithophylic and riffle species are relatively uncommon. The reduced occurrence of elevated TSS concentrations and response of sediment sensitive biotic endpoints in AUID 732 and 508 (specifically, lower Elm Creek) suggest that sediment impacts to biota are less pronounced in these subwatersheds. However, sediments are likely impacting biota in AUID 528 and 760. Conflicting evidence between chemical and biological data for impairment from sediments in AUID 528 may be explained by residual impacts from historical sediment deposition (i.e., high TSS concentrations previously observed, may have created an embedded stream bottom that continues to affect local biota—which is consistent with substrate data). Evidence for sediment impacts to macroinvertebrate assemblages in Diamond Creek (AUID 525) is limited, suggesting excess sediments are potentially impacting this system to a lesser degree. Although the many of biotic responses are consistent with observed effects from excess sediment and divergent from biotic assemblages in unimpaired systems, all of these biological endpoints are potentially affected by multiple stressors. Despite the lack of a selective, diagnostic endpoint for impacts to biota from excess sediment, the SOE suggests that excess sediment is a primary stressor for AUIDs 525 (fish), 528 (fish and macroinvertebrates), and 760 (fish) and a secondary stressor in AUIDs 508 (fish and macroinvertebrates), 525 (macroinvertebrates), 732 (fish and macroinvertebrates) and 760 (macroinvertebrates).

Table 4.8. Summarizes the weight of evidence supporting excess sediments as a causative stressor for the biological impairments observed throughout the Elm Creek Watershed See Appendix A for definitions.

Strength of Evidence Table -- Excess Sediments					
	Scores for Impaired Reaches				
	Elm Creek	Rush Creek			Diamond Creek
Types of Evidence	508	528	732	760	525
Spatial/Temporal concurrence	+	+	+	+	+
Temporal sequence	0	0	0	0	0
Evidence of exposure, biological mechanism	+	0	0	0	0
Causal pathway	+	+	+	+	+
Field evidence of stress response	0	++	0	++	+
Field experiments/manipulations of exposure	NE	NE	NE	NE	NE
Laboratory analysis of site media	NE	NE	NE	NE	NE
Verified or tested predictions	0	0	0	0	0
Symptoms	+	+	+	+	+
Mechanically plausible cause	+	+	+	+	+
Stressor-response in other field studies	+	+	+	+	+
Stressor-response in other lab studies	NE	NE	NE	NE	NE
Stressor-response in ecological models	NE	NE	NE	NE	NE
Manipulation experiments at other sites	NE	NE	NE	NE	NE
Analogous stressors	NE	NE	NE	NE	NE
Consistency of evidence	0	0	+	0	0
Explanatory power of evidence	0	+	+	+	+

4.4. Candidate Cause #4 – Excess Phosphorus

4.4.1. Data Evaluation

A total of 1668 samples were used to assess phosphorus as a potential contributor to the biological impairments throughout the Elm Creek Watershed (Figure 4.4). Across all streams, the total phosphorus (TP) standard of 0.1 mg/L was exceeded in 64% or more of samples collected (Table 4.9). Taken together, these data suggest that TP exceeds the established water quality standards (see Section 3.4) and is a likely contributor to the observed biological impairments in all stream reaches.

Table 4.9. A comparison of TP concentrations measured throughout the Elm Creek Watershed to the 0.1 mg/L water quality standard

Stream	AUID	TP Standard	Total Samples	Number of Exceedences	Percent Exceedence
Elm Creek	508	0.1 mg/L	1180	993	84%
Rush Creek (Main)	528	0.1 mg/L	88	84	95%
Rush Creek (South)	732; 760*	0.1 mg/L	294	287	98%
Diamond Creek	525	0.1 mg/L	306	195	64%
Total			**1868**	**1559**	**85%**

*Based on spatial proximity, water chemistry from AUID 732 is being used as a surrogate for AUID 760

4.4.2. Stressor Pathway

Phosphorus can affect stream biota through a variety of mechanisms (http://www.epa.gov/caddis/ssr_nut4s.html). In extreme cases, phosphorus itself can be acutely or chronically toxic to aquatic organisms, but these cases are rare. More commonly, phosphorus impacts stream biota through secondary mechanisms associated with eutrophication (e.g., Miltner and Rankin 1998). Since phosphorus levels often limit primary productivity in streams, even small increases in phosphorus concentrations can stimulate growth of plankton, periphyton and rooted macrophytes (e.g., Carpenter et al. 1998). When phosphorus concentrations exceed ecological thresholds, increased rates of plant growth/mortality results in increased rates of heterotrophic decomposition, which creates an oxygen demand surrounding areas of high productivity stream reaches. This process often occurs naturally in wetlands, but can be greatly exacerbated and induced in non-wetland habitat. These low oxygen conditions, and in some cases, biofouling of habitat are often primarily responsible for impacts to stream biota.

Biological responses to excess phosphorus are often characterized by shifts in community structure toward low DO specialists and away from an allochthonously driven food web structure. Most stream organisms require relatively high levels of oxygen to survive, although some species specialize in exploiting low DO environments (response to low DO will be describe in greater detail under the DO section below). Similarly, most stream communities in first and second orders streams (similar to those found through the Elm Creek watershed) are comprised of a range of functional feeding groups that exploit different energy resources in the food web. For example, food web structure in headwater streams is often driven primarily by allochthonous food sources, which are dominated by shredder and collector-gatherer macroinvertebrate feeding guilds. As phosphorus concentrations increase (and potentially as sunlight availability increases with reduced canopy cover from habitat alteration; see Section 4.2 for further discussion), autochthonous food sources become more abundant in the stream. In response, functional feeding group composition often shifts, away from collector-gatherer guilds and toward herbivorous (i.e., scrapers), plantivorous and detritivorous guilds.

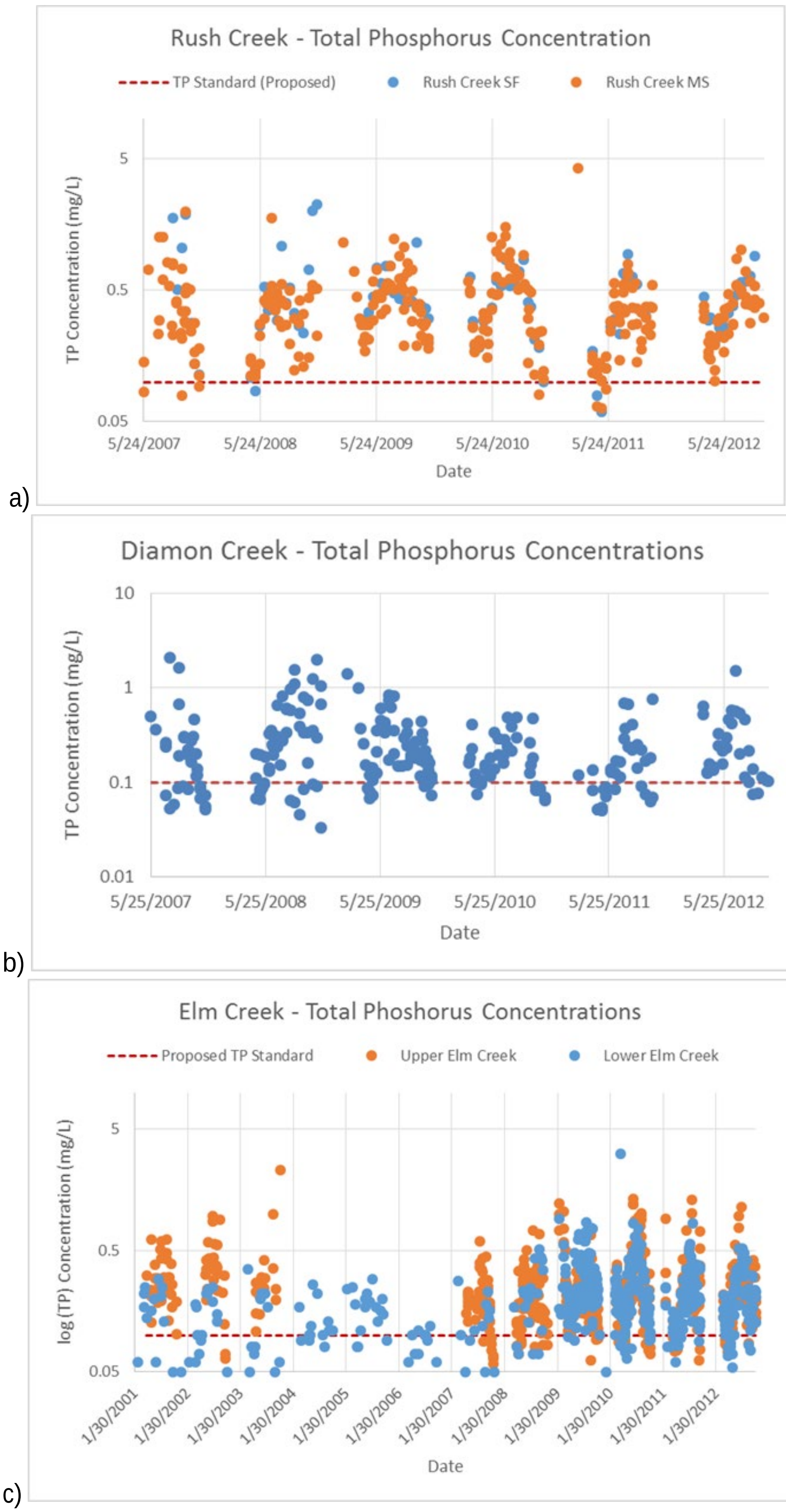


Figure 4.14. The TP concentrations measured in a) Rush, b) Diamond and c) Elm Creeks from 2001-2009

4.4.3. Causal Analysis of Biological Response

Structures of the biotic assemblages throughout the Elm Creek Watershed are consistent with impacts from excess phosphorus (Figure 4.15-Figure 4.17). However, these responses are mixed across AUIDs and assemblages.

Elm Creek has the most direct evidence of elevated phosphorus impacts on biota. Primary productivity in Elm Creek is significantly elevated—in 2007 periphyton biomass was measured at 2289.1 g/m^2 in depositional habitat adjacent to the long-term USGS gage in lower Elm Creek (Elm Creek Watershed Management Commission, et. al. 2011). Given the similarity in nutrient concentrations among Rush, Diamond and Elm Creeks, similar levels of production would be expected (although no direct measurements of production have been made in these reaches). Concurrent to this change in productivity, macroinvertebrate assemblages are disproportionately overrepresented (relative to unimpaired sites) by individuals from collector taxa, and individuals from shredder taxa are relatively uncommon (Figure 4.15). Interestingly, the relative occurrence of collector-gatherer taxa and collector-filter taxa is reversed across the two Elm Creek sites, suggesting that these sites may be responding to different stressor levels and/or combination.

In general, the relative occurrence of individuals from scraper taxa is lower than would be expected based on the high levels of instream productivity. However, this is potentially as a result of the limited availability of hard substrate and low DO concentrations (see Section 4.5 below) often necessary to support many grazer taxa. Fish assemblages in Elm Creek are relatively consistent with those from unimpaired reaches, and the relative occurrence of individuals from detritivorous and planktivorous taxa is lower than expected based on the high levels of periphyton productivity. However, the relative occurrence of individuals from carnivorous taxa (“CarnPct”) is increased, which can indicate increased productivity.

The structure of phosphorus sensitive endpoints within biotic assemblages in Rush Creek is variable. In AUIDs 528 and 732, macroinvertebrate assemblages are dominated by collector taxa, and the relative occurrence of shredder taxa is reduced in AUID 528 (but not 732), relative to unimpaired reaches. Fish assemblages in AUID 528 are relatively consistent with unimpaired reaches, but are dominated by detritivorous (DetNWQPct”) species in AUID 732. In AUID 760, this pattern is reversed, where macroinvertebrate assemblages are inconsistent with elevated phosphorus conditions (a reduced occurrence of collector species), but fish assemblages are dominated by plantivorous species (“PlnkNWQPct”), which suggests a shift in food web structure toward increased primary productivity.

The structure of biotic assemblages in Diamond Creek (AUID 525) is relatively similar to Elm Creek, as macroinvertebrate assemblages are dominated by individuals from collector taxa and individuals from shredder taxa are underrepresented, relative to unimpaired reaches. Fish assemblages in Diamond Creek are similar to those from unimpaired reaches, except that carnivorous species are overrepresented—potentially suggesting an increase in primary productivity.

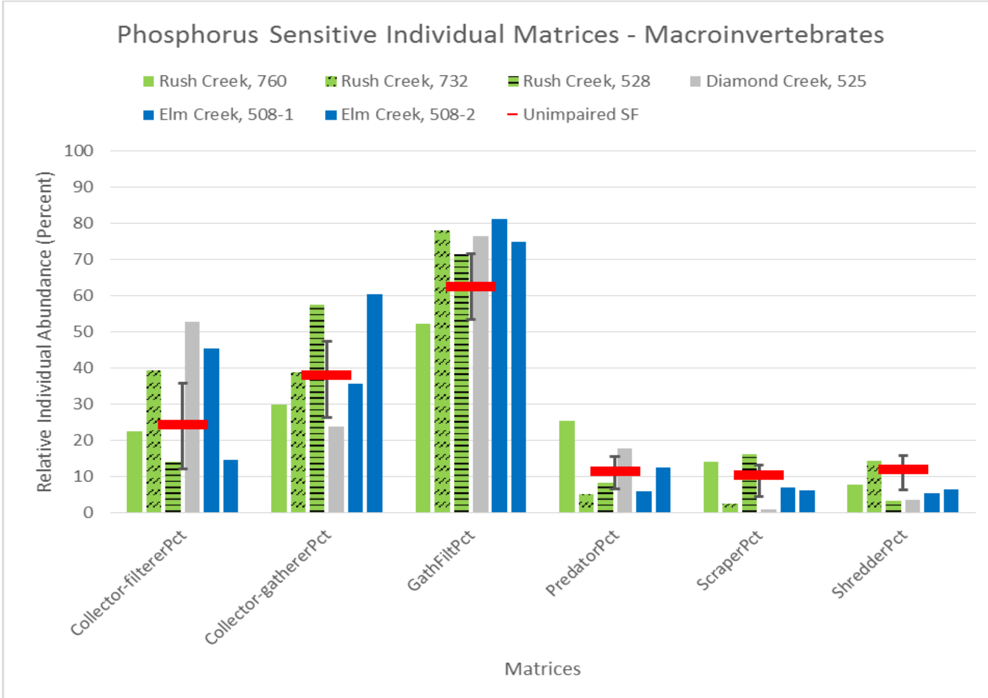


Figure 4.15. Response of macroinvertebrate feeding guild metrics likely impacted by elevated phosphorus levels in each AUID as compared to unimpaired reaches throughout the Southern Forest GP stream classification ("Unimpaired SF"). Horizontal bars represent average metric responses and "whiskers" represent upper and lower quartiles at unimpaired sites.

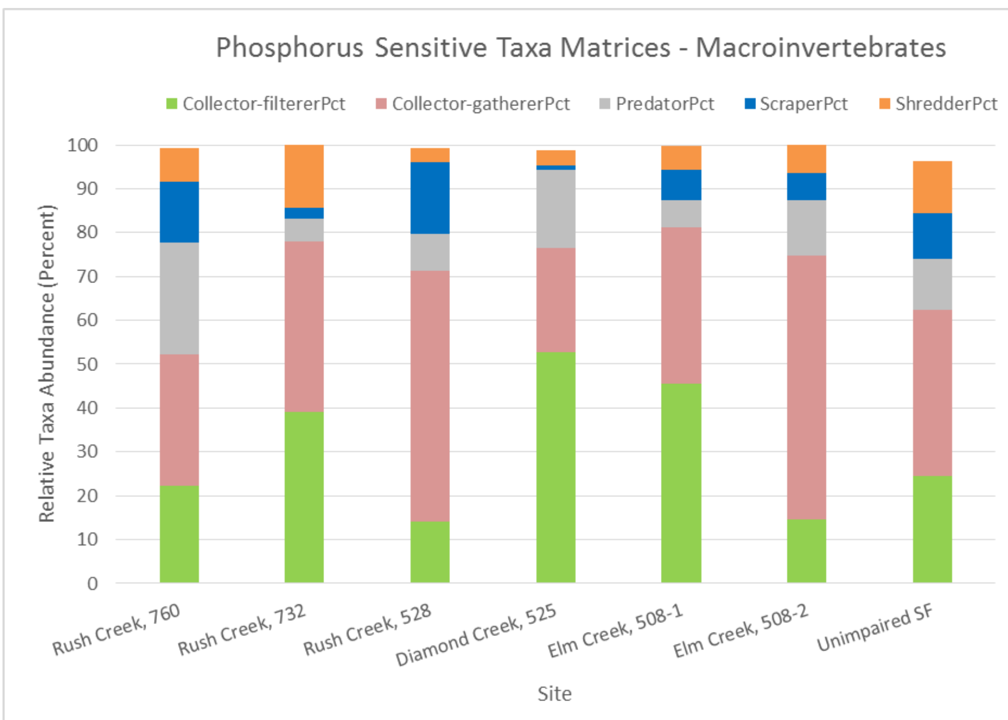


Figure 4.16. Response of macroinvertebrate feeding guild metrics likely impacted by elevated phosphorus levels in each AUID as compared to unimpaired reaches throughout the Southern Forest GP stream classification (Unimpaired SF).

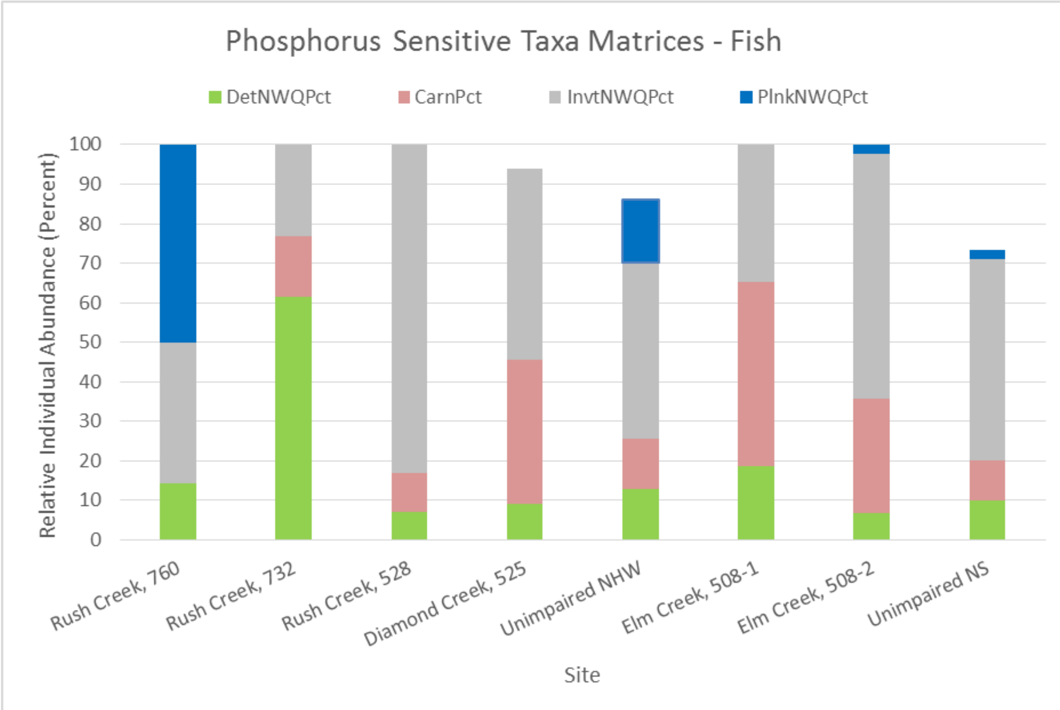


Figure 4.17. Response of fish feeding guild metrics likely impacted by elevated phosphorus levels in each AUID as compared to unimpaired reaches throughout the Northern Headwaters stream classification ("Unimpaired NHW") for Rush and Diamond Creek and the Northern Streams classification ('Unimpaired NS") for Elm Creek.

4.4.4. Summary of Strength of Evidence

Evidence for biotic impacts from excess phosphorus is relatively common across all stream reaches, but the magnitude of the biological response varies across assemblages (Table 4.10). The altered hydrologic and land use patterns that commonly result in excess phosphorus have clearly shifted over the last 30 years (see Section 2.1). Concurrently, the occurrence of elevated phosphorus concentrations and periphyton growth has increased (Figure 4.11). Given the lack of historical biological data, biotic assemblages within the Elm Creek Watershed cannot be analyzed for temporal response to phosphorus. Current macroinvertebrate assemblages are consistent with those commonly observed in streams with high phosphorus—individuals from collector taxa dominate all stream reaches (except AUID 760), while the occurrence of individuals from shredder taxa is reduced (lack of scraper species is unexpected given the high phosphorus concentration, but may be as a result of limited substrate availability and frequent disturbance, see Sections 4.1 and 4.2). Fish assemblage composition in Rush Creek (specifically AUID 732 and 760) is dominated by detritivorous and plantivorous species and consistent with impacts from elevate phosphorus. Fish assemblages in Diamond and Elm Creeks are dominated by insectivorous and carnivorous species and inconclusively impacted by elevated phosphorus levels. Although the biotic responses are consistent with observed effects from excess phosphorus and divergent from biotic assemblages in corollary unimpaired streams, all of these biological endpoints are potentially affected by multiple stressors. Despite the lack of a selective, diagnostic endpoint for impacts to biota from excess phosphorus, the SOE suggests that excess phosphorus is a primary stressor for AUIDs 508 (macroinvertebrates), 525 (macroinvertebrates), 732 (fish and macroinvertebrates) and 760 (fish) and a secondary stressor in AUIDs 508 (fish), 525 (fish), 528 (fish) and 760 (macroinvertebrates).

Although the evidence for direct effects of phosphorus on biota is variable throughout the Elm Creek Watershed, this historical loading of phosphorus (and nitrogen) into wetlands is likely a key driver of the DO fluctuation at observed at some sites (see Figure 4.19 and Section 4.5 for additional detail).

Table 4.10. Summarizes the weight of evidence supporting excess phosphorus as a causative stressor for the biological impairments observed throughout the Elm Creek Watershed See Appendix A for definitions.

Strength of Evidence Table -- Excess Phosphorus					
	Scores for Impaired Reaches				
	Elm Creek	Rush Creek			Diamond Creek
Types of Evidence	508	528	732	760	525
Spatial/Temporal concurrence	+	+	+	+	+
Temporal sequence	0	0	0	0	0
Evidence of exposure, biological mechanism	++	++	++	++	++
Causal pathway	+	+	+	+	+
Field evidence of stress response	+	+	++	+	+
Field experiments/manipulations of exposure	NE	NE	NE	NE	NE
Laboratory analysis of site media	NE	NE	NE	NE	NE
Verified or tested predictions	NE	NE	NE	NE	NE
Symptoms	+	+	+	+	+
Mechanically plausible cause	+	+	+	+	+
Stressor-response in other field studies	+	+	+	+	+
Stressor-response in other lab studies	NE	NE	NE	NE	NE
Stressor-response in ecological models	NE	NE	NE	NE	NE
Manipulation experiments at other sites	NE	NE	NE	NE	NE
Analogous stressors	NE	NE	NE	NE	NE
Consistency of evidence	+	+	+++	++	+
Explanatory power of evidence	+	+	+	+	+

4.5. Candidate Cause #5 – Low Dissolved Oxygen

4.5.1. Data Evaluation

A total of 675 discrete samples were collected across Diamond, Rush, and Upper and Lower Elm Creeks to assess the potential for low DO contribution to the biological impairments (Figure 4.18). Across all streams, the DO daily minimum standard of 5 mg/L was exceeded in 20% or more of samples collected (Table 4.11). These point sample data are further corroborated by data collected from five sites using continuous deployment sondes, in which daily average DO levels did not meet the 5 mg/L standard at five sites (Figure 4.19). Taken together, these data suggest that DO exceeds the established water quality standards (see Section 3.3.5) and is a likely contributor to the observed biological impairments in all AUIDs.

The patterns of seasonal and daily DO fluctuation measured at the continuous monitoring sites are of particular interest. The patterns of DO fluctuation are highly variable across sites, and this variation is potentially as a result of different drivers of the oxygen demand. At sites associated with significant wetland complexes, the magnitude of daily DO fluctuation is significantly increased (e.g., EC81; up to 10 mg/L over a 24 hr period) relative to sites in which wetlands are less associated with stream channel habitat (e.g., USGS, 2-5 mg/L over a 24 hr period). These results suggest that respiration processes in wetland complexes may be a significant driver of low DO throughout the Elm Creek Watershed and that biotic impacts of low DO may be variable within individual AUIDs (particularly 508). Additionally, given that low DO levels are present in Rush Creek (RT), but the pattern of DO fluctuation (and site location) is inconsistent with the influence of wetland complexes, an additional source of oxygen demand may be affecting DO concentrations throughout this sub-basin. Given the proportionally higher concentration of total nitrogen throughout this sub-basin, it is possible that increased rates of nitrification (and the corresponding nitrogen oxygen demand) are suppressing DO concentrations.

Table 4.11. A comparison of discrete DO measurements throughout the Elm Creek Watershed the daily minimum standard of 5 mg/L

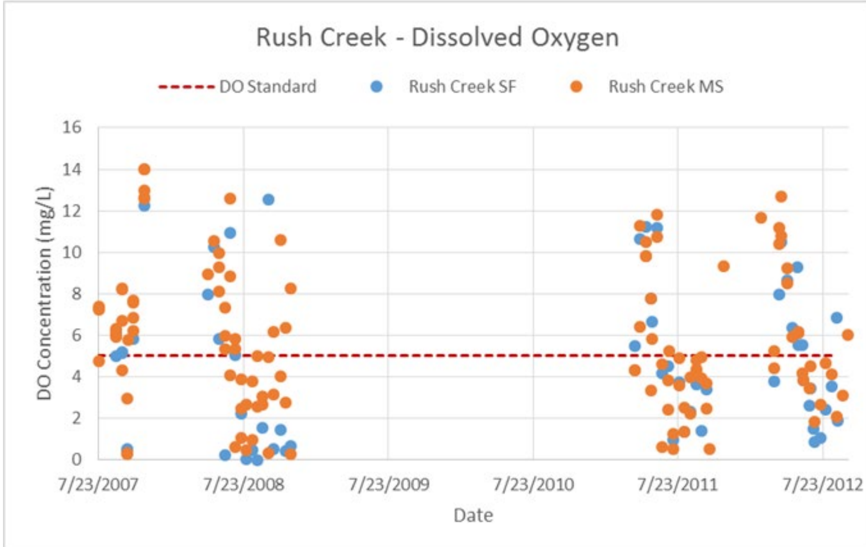
Stream	AUID	DO Standard	Total Samples	Number of Exceedences	Percent Exceedence
Elm Creek	508	5 mg/L	306	62	20%
Rush Creek (Main)	528	5 mg/L	58	32	55%
Rush Creek (South)	732; 760*	5 mg/L	164	87	53%
Diamond Creek	525	5 mg/L	147	50	34%
Total			**675**	**231**	**41%**

*Based on spatial proximity, water chemistry from AUID 732 is being used as a surrogate for AUID 760

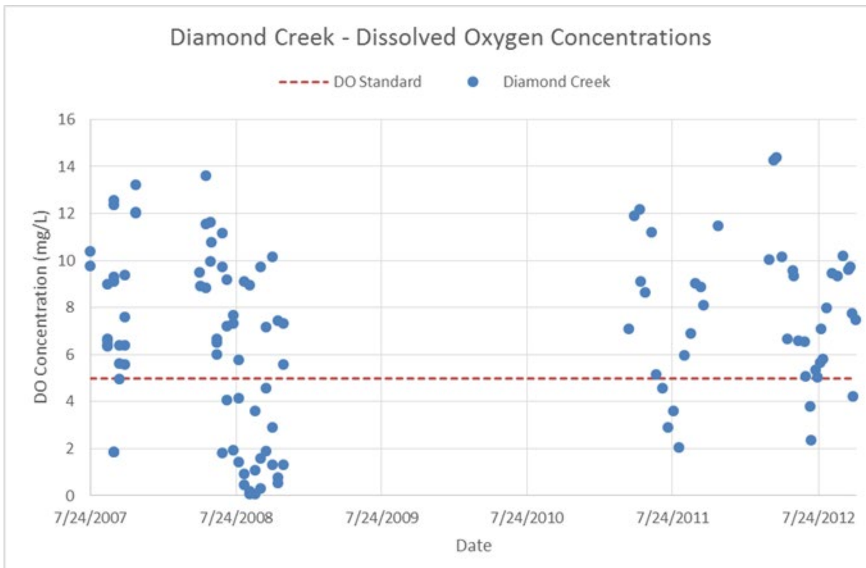
Biological and chemical evidence suggests that the severity of DO impairment decreases from upstream to downstream and that downstream DO fluctuation, may be primarily related to wetland respiration (described above). This trend is strongly correlated with TP-BOD relationships (Figure 4.20). Rush Creek (AUID 528) has the strongest evidence for DO impairment and similarly, the second highest average BOD concentration—and BOD concentrations at this site are positively correlated to TP concentrations. This relationship between DO impairment, BOD, and TP concentration is consistent across sites in south Rush Creek, Diamond Creek and Upper Elm Creek, where sites with the strongest levels of DO impairment also have the highest BOD concentrations and/or strongest TP-BOD correlations.

The DO-BOD-TP relationship in lower Elm Creek is of particular interest. Lower Elm Creek has consistent chemical data to support DO impairment, but the corresponding biological communities are less consistent with low DO conditions. Given the diurnal variation in DO concentrations associated with wetland influenced sites in lower Elm Creek, low DO conditions are likely driven by BOD that originates from wetland plant decomposition (that likely stems from historical accumulation of TP). The BOD-TP relationship from lower Elm Creek corroborates the chemical and biological data. Relatively high BOD concentrations were observed in lower Elm Creek, but the correlation to TP was particularly weak. Taken together, these data suggest that localized areas of low DO likely exist in lower Elm Creek, but that this is primarily driven by wetland processes (and historical phosphorus accumulation) and not current water quality conditions. Given this relationship, TP can be seen as a surrogate for DO that is mediated through BOD.

a)



b)



c)

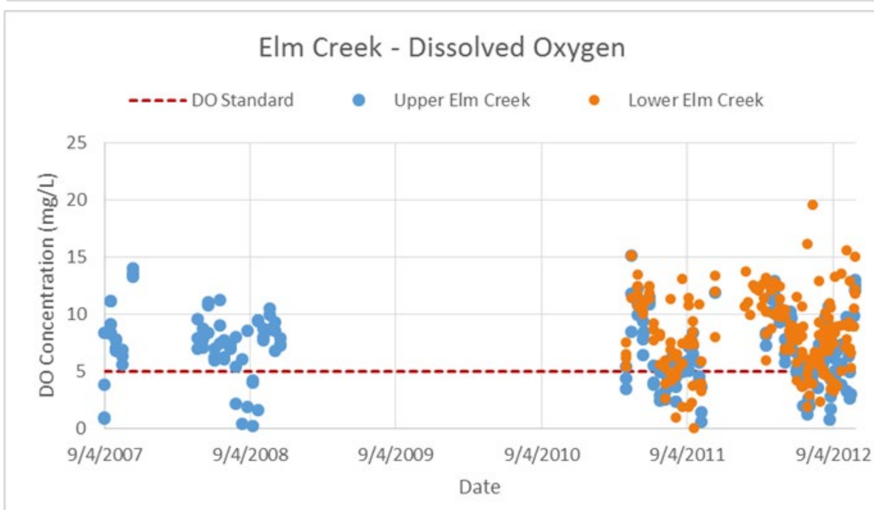


Figure 4.18. The DO concentrations measured in a) Rush, b) Diamond and c) Elm Creeks from 2007-20012

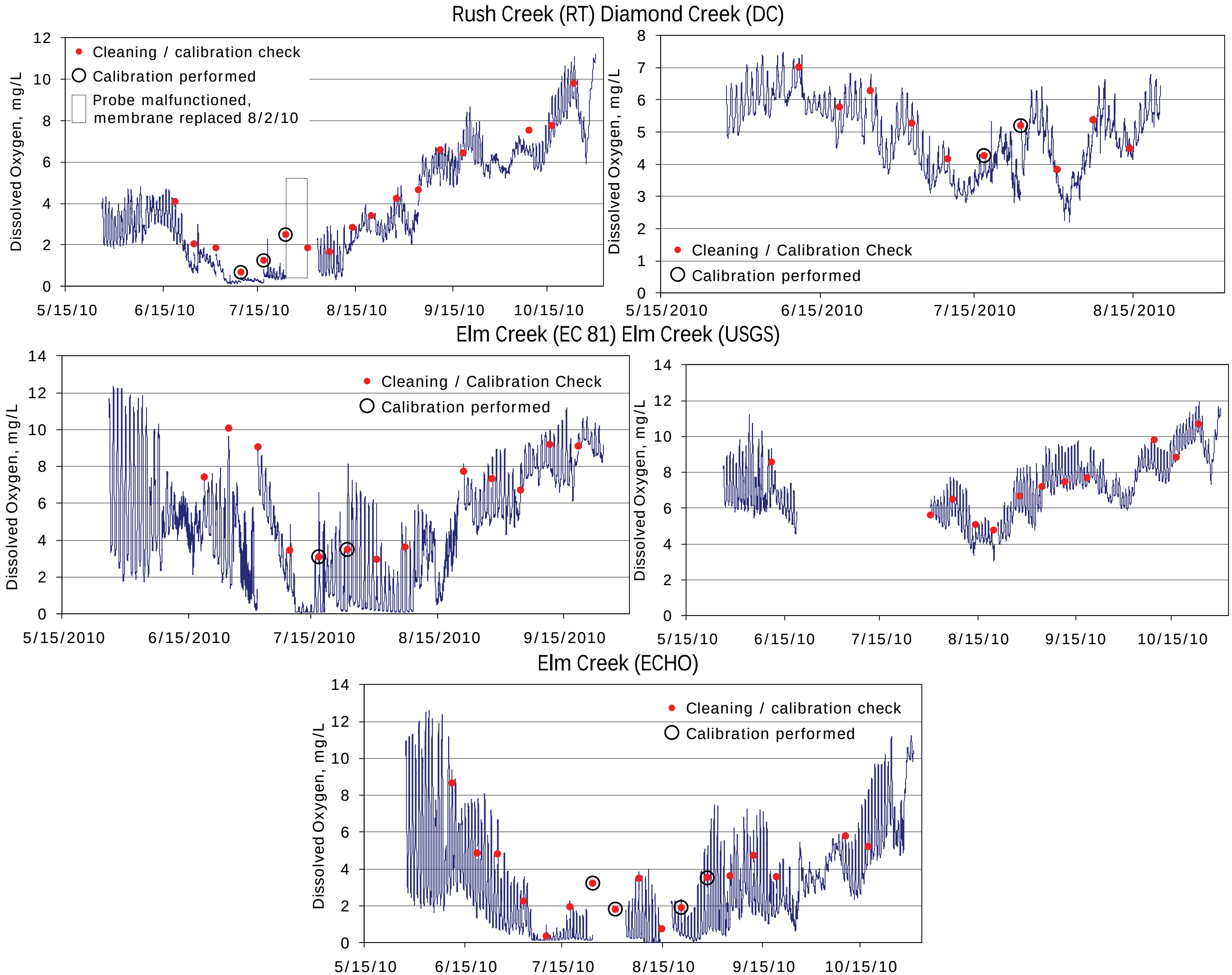
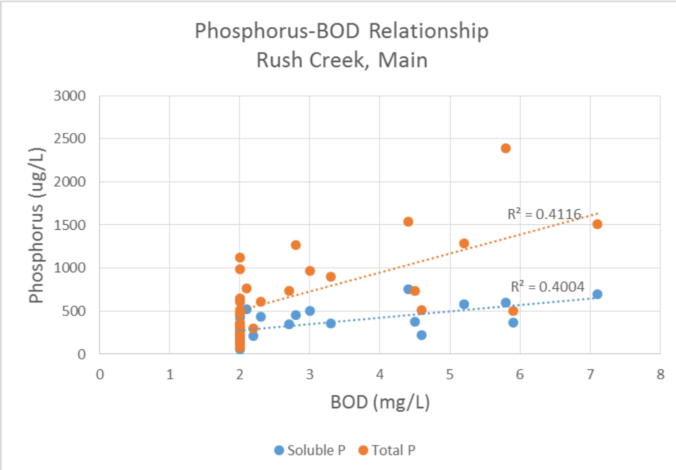
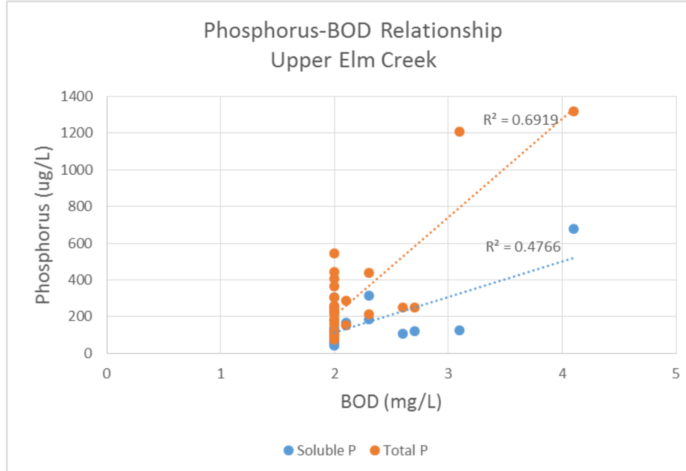


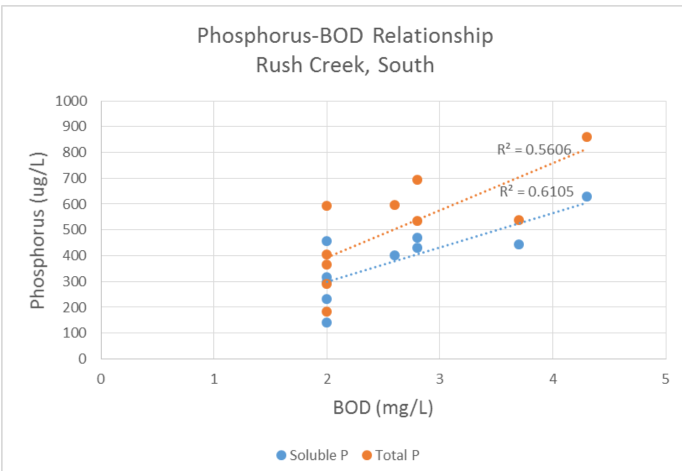
Figure 4.19. Continuous DO concentrations measured in Rush (RT), Diamond (DC) and Elm (EC81, USGS, ECHO) Creeks, 2010

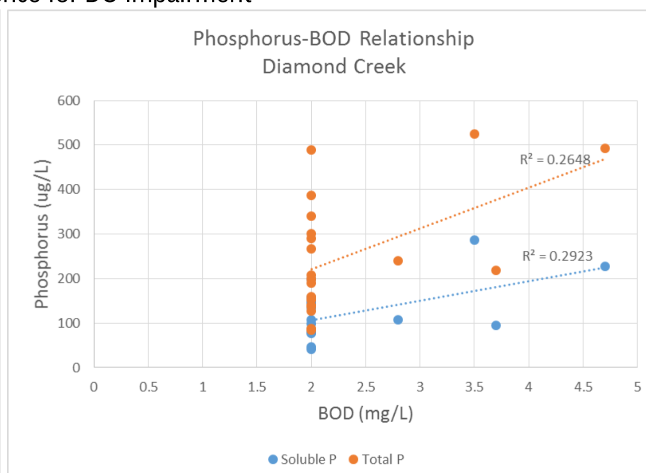
Sites with Strong Evidence for DO Impairment





Sites with Moderate Evidence for DO Impairment





Sites with Limited Evidence for DO Impairment

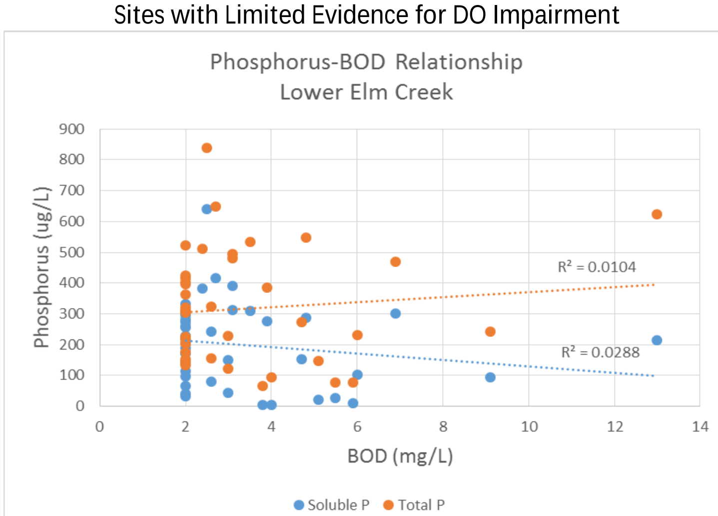


Figure 4.20. Phosphorus-BOD relationship across a gradient of sites with different levels of DO impairment

4.5.2. Stressor Pathway

Low DO conditions can impact stream biota through a variety of mechanisms (http://www.epa.gov/caddis/ssr_do4s.html). As described above (see Section 4.4), low DO conditions most commonly results in a shift in biological community structure away from cool water species toward low DO specialists. As DO levels decrease, macroinvertebrate assemblages often become dominated by individuals representing tolerant and chironomid taxa, while oxygen sensitive taxa (e.g., Ephemeroptera, Plecoptera and Trichoptera; EPT) become relatively less common. Although species tolerance and intolerance metrics are often sensitive to low DO conditions, additional stressors (e.g., habitat alteration) can also affect the relative abundance of tolerant and sensitive taxa. As such, these endpoints should not be viewed as precisely diagnostic, but instead as corroborating evidence of low DO conditions.

Two macroinvertebrate taxa that are particularly affected by DO concentrations are Tanytarsus (which often decreases in response to low oxygen conditions) and Tanypodinae (which generally increases in response to low DO). Similarly, macroinvertebrate responses to organic pollution and low DO conditions have been summarized in a series of stressor-specific indices. One index that is commonly used to characterize organic pollution and low DO conditions is the Hillsenhoff Biotic Index (HBI) that ranks macroinvertebrate communities on a scale of 1-10 (Hillsenhoff 1988; as index values approach 10, low DO conditions are probable).

4.5.3. Causal Analysis of Biological Response

Biotic indicators of low DO stress are common among all AUIDs in the Elm Creek watershed, but mixed with respect to biotic assemblages (Figure 4.20 through Figure 4.22). In AUID 508, relatively few, if any individuals from fish and macroinvertebrate taxa representing sensitive ("SensitivePct"), intolerant ("IntolerantPct"; "Intolerant2LessPct") species were sampled. Interestingly, fish assemblages are overrepresented (relative to unimpaired sites) by individuals from taxa representing tolerant ("Toerlant2Pct"; "TolPct") and very tolerant (VeryTolerant2Pct"; "VtolPct") species, while invertebrate assemblages are consistent with those observed at unimpaired sites.

The structure of biotic assemblages in Rush Creek is similar to that of Elm Creek. However, the response to low DO conditions is more pronounced in Rush Creek. Fish and macroinvertebrate assemblages from AUID 760 exhibit the most pronounced response to low DO conditions. Fish assemblages in AUID 760 are dominated by tolerant, wetland species ("WetlandPct"; "Wetland-TolPct"), which are adapted to low and fluctuating DO conditions. Macroinvertebrate assemblages in AUID 760 are dominated by legless ("LeglessPct") and chironomid ("ChironomidaeChPct"; "ChironomoniniPct") taxa and sensitive taxa ("EPTPct"; "POETPct"; "TanytarsiniPct") are absent or relatively limited in abundance. Fish assemblages from AUID 528 and 732 were also dominated by tolerant wetland species, but the magnitude of this response was significantly reduced, as compared to AUID 760. The structure of biotic assemblages in Diamond Creek is similar to that of Rush Creek, but less pronounced, suggesting that low DO conditions are likely having a reduced, but measureable impact on biotic communities within this system.

Within macroinvertebrate assemblages, the response of oxygen sensitive indices was relatively consistent across AUIDs. HBI scores were between five and seven for all sites, suggesting a moderate potential for low DO impacts to biota (Figure 4.22)—although the Minnesota-specific HBI ("HBI_MN") suggest that DO response at all sites is consistent with those observed in unimpaired systems. Interestingly, the relative occurrence Tanytarsinid taxa was significantly increased over Tanypodinid taxa ("TanypodinaePct") at all sites except AUID 760, suggesting that low DO conditions may be having less of an impact on macroinvertebrate communities outside of AUID 760.

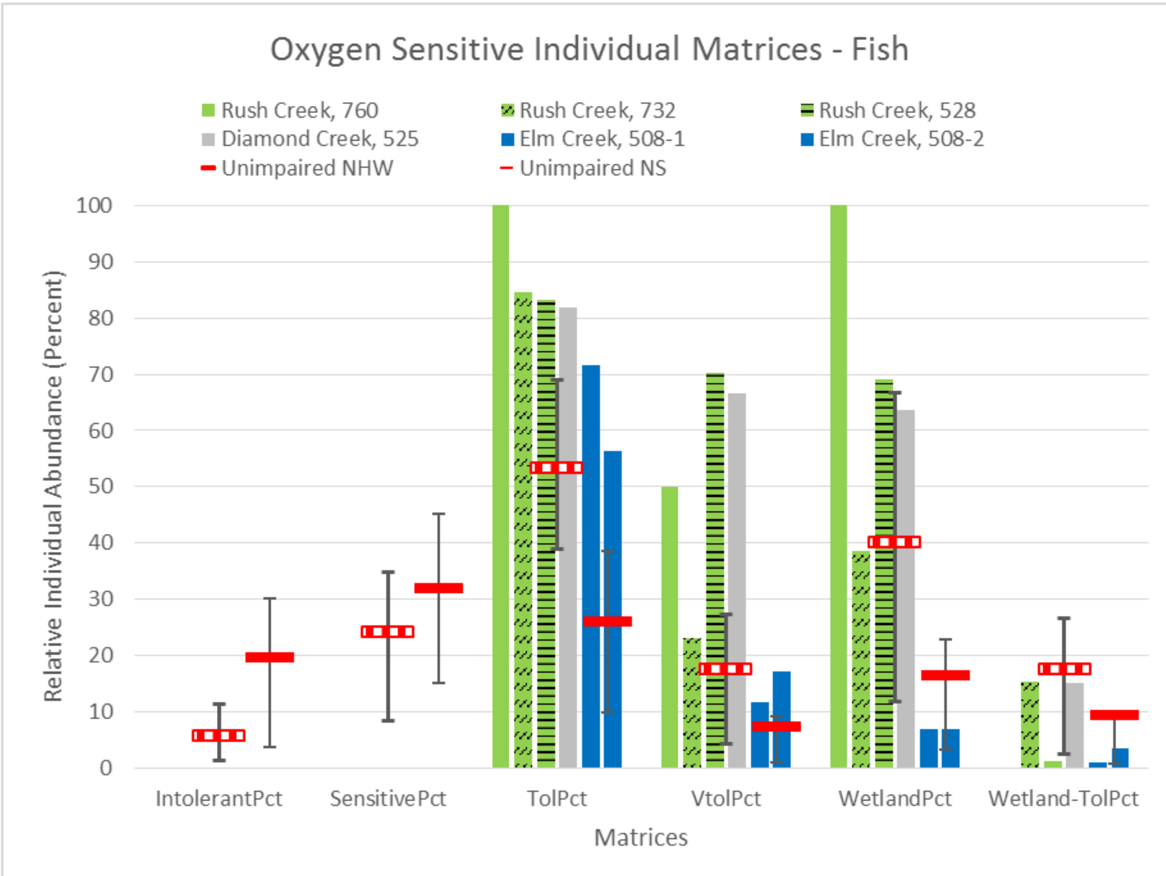


Figure 4.21. Response of fish assemblage metrics likely impacted by low DO levels in each AUID as compared to unimpaired reaches throughout the Northern Headwaters stream classification ("Unimpaired NHW") for Rush and Diamond Creek and the Northern Streams classification ('Unimpaired NS") for Elm Creek. Horizontal bars represent average metric responses and "whiskers" represent upper and lower quartiles at unimpaired sites.

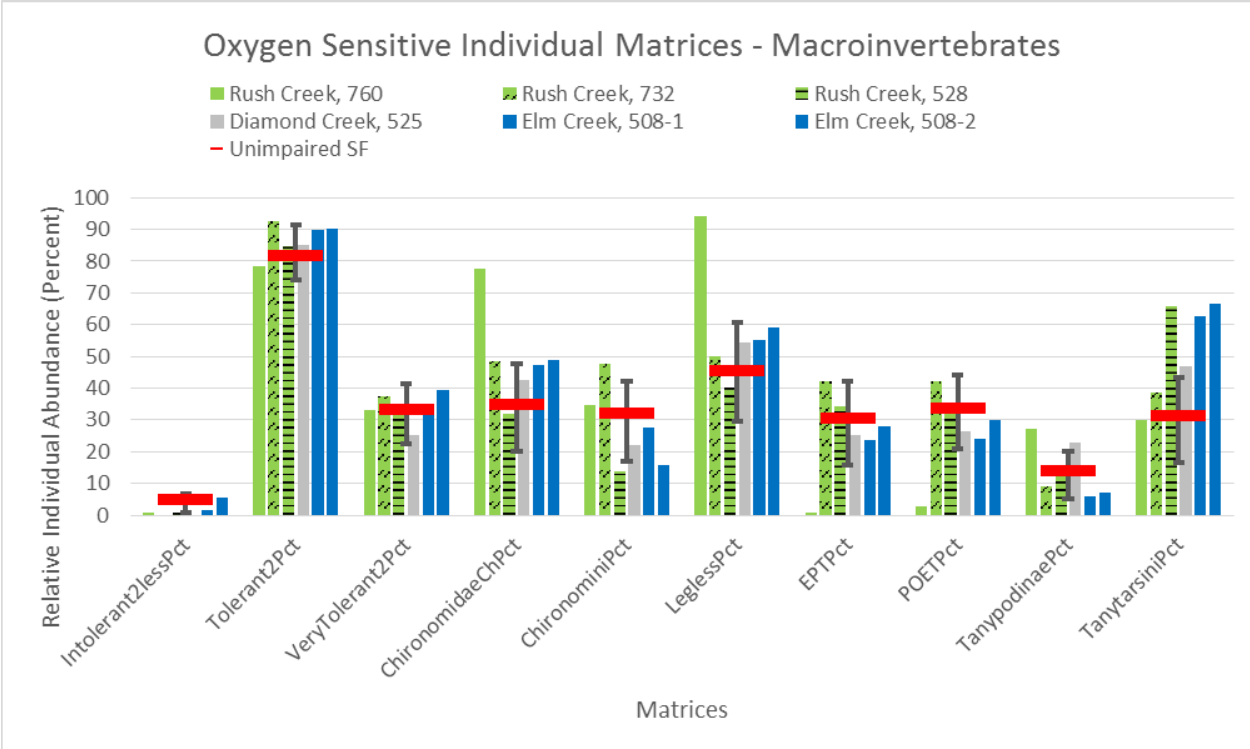


Figure 4.22. Response of macroinvertebrate assemblage metrics likely impacted by low DO levels in each AUID as compared to unimpaired reaches throughout the Southern Forest GP stream classification ("Unimpaired SF") Horizontal bars represent average metric responses and "whiskers" represent upper and lower quartiles at unimpaired sites.

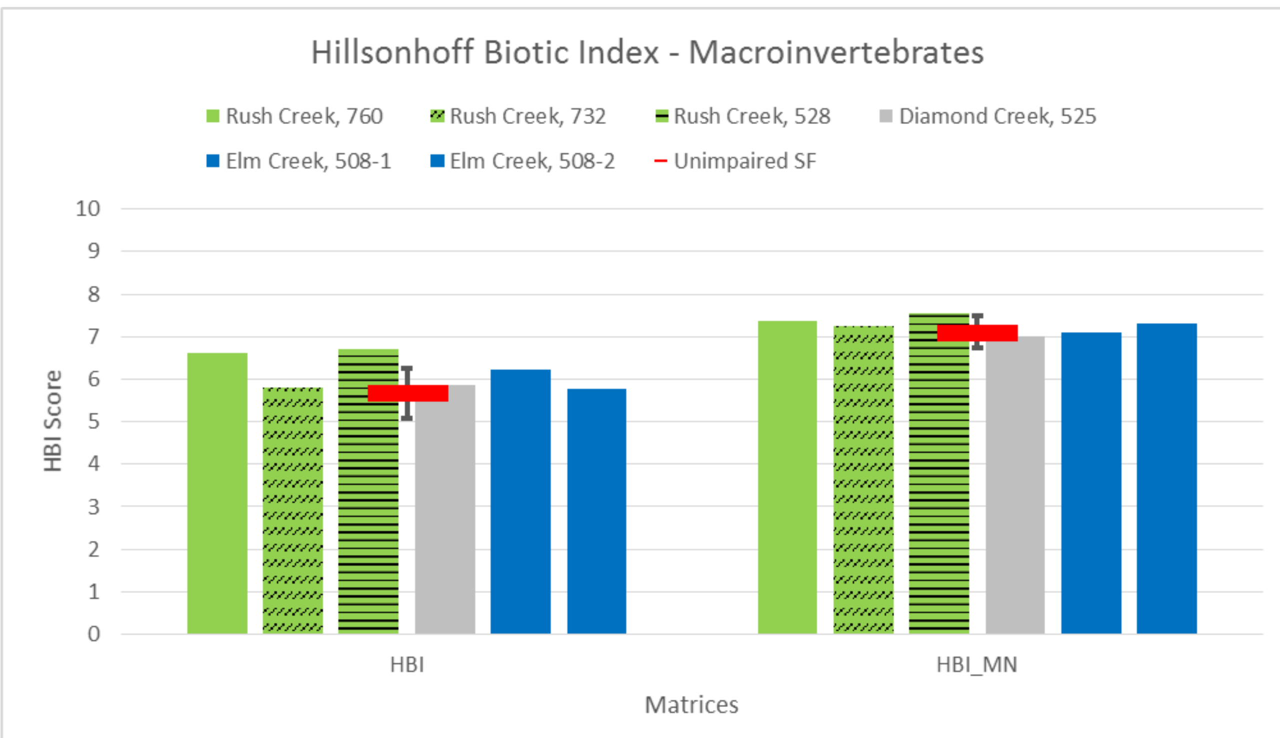


Figure 4.23. Response of macroinvertebrate assemblage metrics likely impacted by low DO levels in each AUID as compared to unimpaired reaches throughout the Southern Forest GP stream classification ("Unimpaired SF") Horizontal bars represent average metric responses and "whiskers" represent upper and lower quartiles at unimpaired sites.

4.5.4. Strength of Evidence

Evidence for biotic impacts from low DO are relatively common across all stream reaches, but most pronounced in Rush Creek and Diamond Creeks (Table 4.12). Altered hydrologic, land use patterns and nutrient concentrations that commonly result in low DOs have clearly shifted over the last 30 years (see Section 2.1). Concurrently, the occurrence of low DO conditions has been observed throughout the Elm Creek Watershed, although low DO conditions have been observed less consistently in Elm Creek (Figure 4.18 and Figure 4.19). Given the lack of historical biological data, assemblages within the Elm Creek Watershed cannot be analyzed for temporal response to low DO. Current macroinvertebrate assemblages are consistent with those commonly observed in streams with low DO—the relative occurrence of DO sensitive species have declined while the occurrence of DO tolerant species has increased (Figure 4.21). Fish assemblage composition in Elm Creek (AUID 508) is less consistent with low DO conditions than in Rush (AUIDs 528, 732 and 760) and Diamond Creeks (AUID 525), suggesting that low DO is potentially impacting biotic communities more significantly in these sub-basins (Figure 4.20). Although the biotic responses are consistent with observed effects from low DO and divergent from biotic assemblages in unaltered systems, all of these biological endpoints are potentially affected by multiple stressors. Despite the lack of a selective, diagnostic endpoint for impacts to biota from low DO, the SOE suggests that low DO is a primary stressor for fish in all AUIDs and a secondary stressor for invertebrates in all sites (except 760, where it is likely a primary stressor). Interestingly, although most biotic data suggest some level of low DO impairment in all AUIDs, the relative response of two oxygen sensitive and tolerant macroinvertebrate taxa are inconsistent with low DO conditions at all sites except in AUIDs 760 (Figure 4.21).

Table 4.12. Summarizes the weight of evidence supporting DO as a causative stressor for the biological impairments observed throughout the Elm Creek Watershed See Appendix A for definitions

Strength of Evidence Table -- Dissolved Oxygen					
	Scores for Impaired Reaches				
	Elm Creek	Rush Creek			Diamond Creek
Types of Evidence	508	528	732	760	525
Spatial/Temporal concurrence	+	+	+	+	+
Temporal sequence	0	0	0	0	0
Evidence of exposure, biological mechanism	+	+	+	++	++
Causal pathway	+	+	+	++	++
Field evidence of stress response	+	+	+	++	+
Field experiments/manipulations of exposure	NE	NE	NE	NE	NE
Laboratory analysis of site media	NE	NE	NE	NE	NE
Verified or tested predictions	0	0	0	0	0
Symptoms	+	+	+	++	+
Mechanically plausible cause	+	+	+	+	+
Stressor-response in other field studies	++	++	++	++	++
Stressor-response in other lab studies	++	++	++	++	++
Stressor-response in ecological models	NE	NE	NE	NE	NE
Manipulation experiments at other sites	+	+	+	+	+
Analogous stressors	NE	NE	NE	NE	NE
Consistency of evidence	+	+	+	++	+
Explanatory power of evidence	+	+	+	+	+

4.6. Candidate Cause #6 – Excess Chloride

4.6.1. Data Evaluation

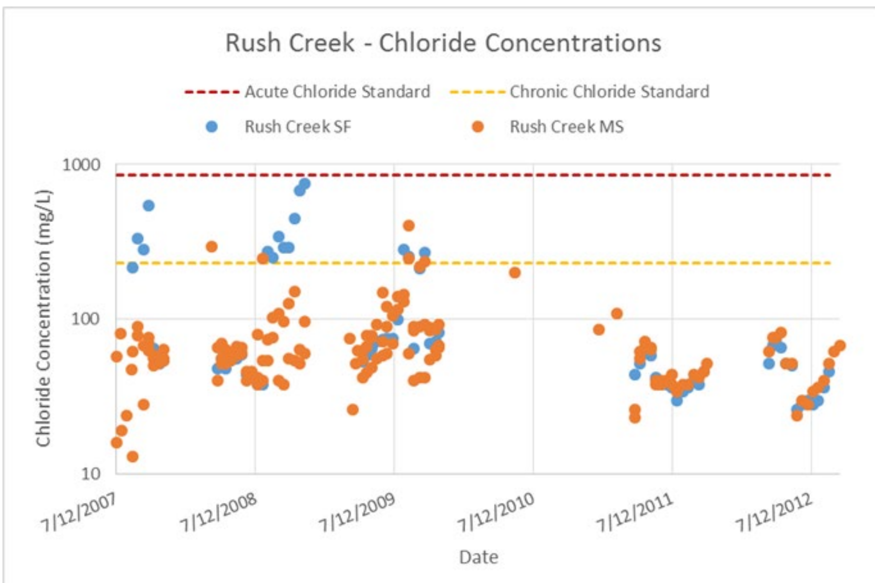
A total of 505 samples were used to assess chloride as a potential contributor to the biological impairments throughout the Elm Creek Watershed (Figure 4.23). Across all streams, the acute chloride standard of 860 mg/L and chronic standard of 230 mg/L were exceeded in between 1% and 20% of the collected samples (Table 4.12). In both Rush and Diamond Creeks, no samples exceeded the acute standard, but in Elm Creek the acute standard was exceeded in 2% of samples. Taken together, these data suggest that chloride exceeds the established acute water quality standards in Elm Creek and may exceed the chronic standard in the main stem of Rush Creek (although trends suggest recent decreases in concentrations). Given that the timing of sampling was predominantly focused on summer months and peak chloride concentrations are commonly observed in winter months, it is possible the current data sets do not fully capture the extent of chloride exposure in the Elm Creek system. However, within the scope of the existing data, the frequency of chloride exceedances of water quality standards are relatively reduced as compared to previous stressors described above. But, given the limited seasonal scope of the available datasets, chloride cannot be ruled out a contributor to the biological impairments in the Elm Creek Watershed.

Table 4.13. A comparison of discrete chloride measurements throughout the Elm Creek Watershed to the chronic and acute standards

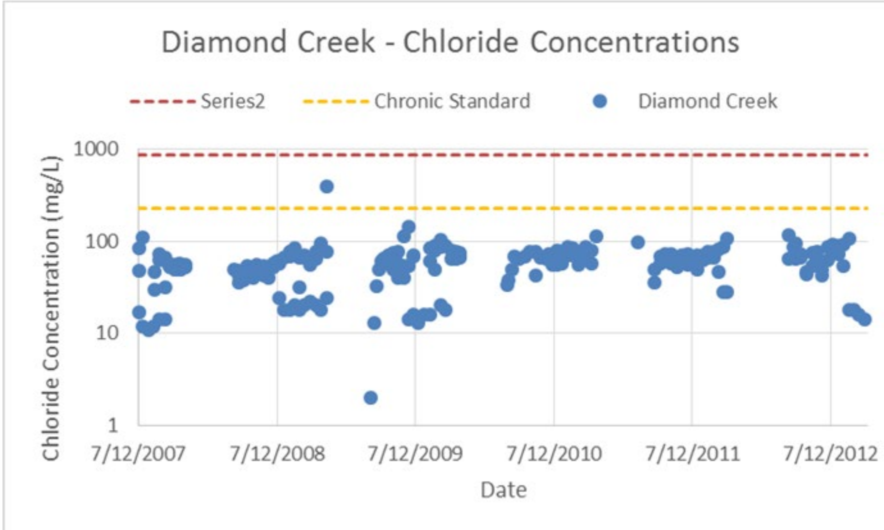
Stream	AUID	Acute Chloride Standard	Total Samples	Number of Exceedences	Percent Exceedence
Elm Creek	508	860 mg/L	607	14	2%
Rush Creek (Main)	528	860 mg/L	69	0	0%
Rush Creek (South)	732; 760*	860 mg/L	204	0	0%
Diamond Creek	525	860 mg/L	221	0	0%
Total			**1101**	**14**	**1%**

Stream	AUID	Chronic Chloride Standard	Total Samples	Number of Exceedences	Percent Exceedence
Elm Creek	508	230 mg/L	607	19	3%
Rush Creek (Main)	528	230 mg/L	69	14	20%
Rush Creek (South)	732; 760*	230 mg/L	204	5	2%
Diamond Creek	525	230 mg/L	221	1	0%
Total			**1101**	**39**	**7%**

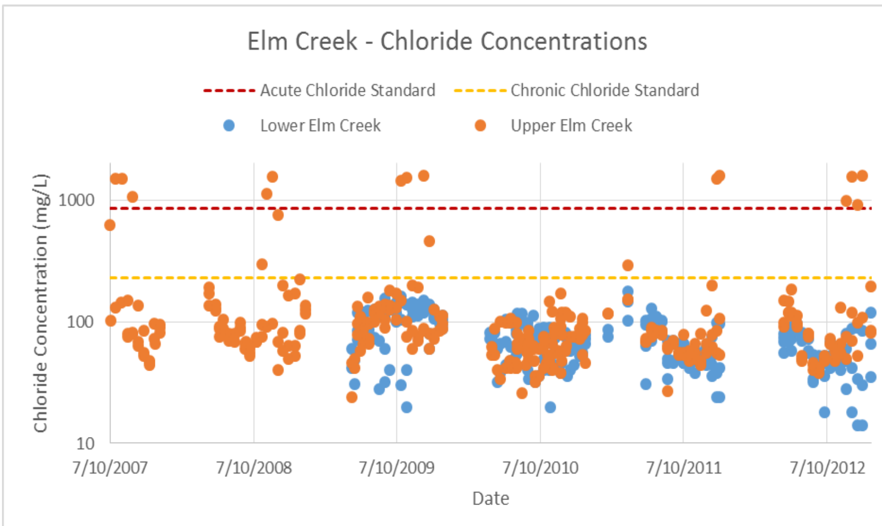
*Based on spatial proximity, water chemistry from AUID 732 is being used as a surrogate for AUID 760



a)



b)



c)

Figure 4.24. Chloride concentrations measured in Elm, Rush and Diamond Creeks from 2007-20012

4.6.2. Stressor Pathway

Specific endpoints to precisely diagnose impacts from chloride pollution on biota are not available in the Elm Creek Watershed. The specific mechanism by which elevated chloride concentrations affect stream biota is not well understood, but may be related to osmotic and ionic regulation (http://www.epa.gov/caddis/ssr_ion4s.html). In areas of know chloride pollution, reductions in organism density, taxa richness and diversity have been observed (e.g., Blasius and Merritt 2002). However, given the diversity of stressors potentially affecting biota in the Elm Creek Watershed (most of which have the potential to impact the density, richness and diversity of different taxa), any changes in these endpoints in not likely to be specifically indicative of chloride exposure.

4.6.3. Causal Analysis of Biological Response

Laboratory studies suggest that different species have developed different toxicological thresholds for chloride exposure. Based on these studies, the most sensitive organisms to chloride pollution are mayflies, which have 96 hour, LC50 values of 415 mg/L (e.g., Wicard, 1975) to over 3800 mg/L (e.g., Blasius and Merritt 2002). Of the organisms sampled throughout Elm Creek Watershed, mayfly taxa are among the most sensitive to chloride pollution (Figure 4.24). However, the relative abundance of mayflies is consistent with assemblages from unimpaired sites (except in AUID 760). Given the lack of mayflies in AUID 760 and the historically elevated chloride concentrations observed in the south fork of Rush Creek, it is possible that chloride exposure has contributed to the biological impairment in this reach. However, given the recent trends in reduced chloride concentration and the diversity of stressors affecting this reach it is unlikely that chloride exposure is the dominant driver of biotic impairment at this site. Given that the other observation of elevated chloride levels have been isolated to upper Elm Creek, it is unlikely that chloride is contributing to the biological impairments observed in AUID 508, which are based on data from assemblages in lower Elm Creek. However, given the timing of sampling (most samples in the summer) and potential for acute responses to exposure, chloride cannot be completely eliminated as a potential cause of biological impairment throughout the Elm Creek watershed.

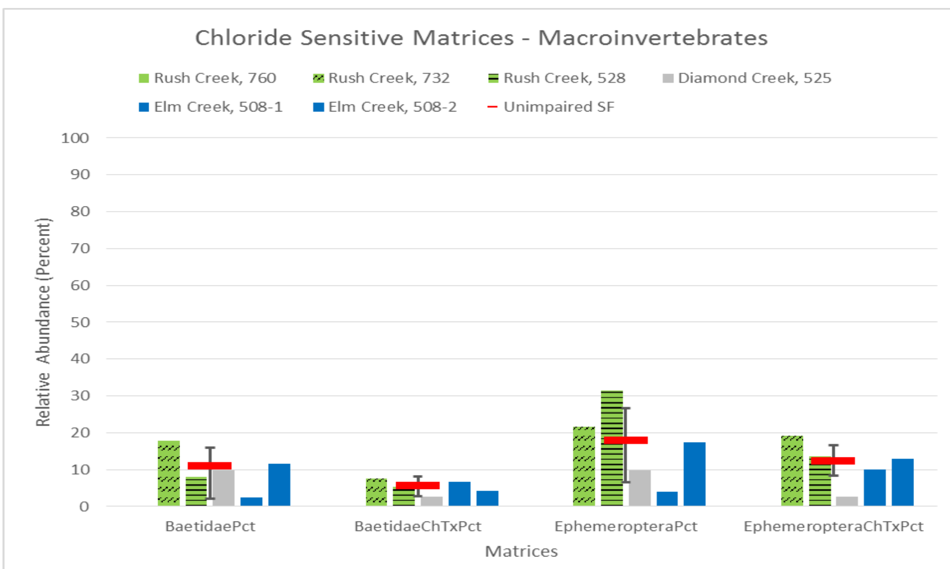


Figure 4.25. Relative abundance of potentially chloride sensitive macroinvertebrate taxa in each AUID as compared to unimpaired reaches throughout the Southern Forest GP stream classification (Unimpaired SF) Horizontal bars represent average metric responses and “whiskers” represent upper and lower quartiles at unimpaired sites

4.6.4. Summary of Strength of Evidence

Evidence for biotic impacts from chloride is inconclusive across all stream reaches, but most probable in Rush and Elm Creeks (Table 4.14). Altered land use patterns (specifically urban development) that commonly result in elevated chloride concentrations have clearly shifted over the last 30 years (see Section 2.1). Concurrently, elevated concentrations of chloride have been observed throughout the Elm Creek Watershed, although elevated chloride levels have been relatively less common in Diamond

Creek. Given the lack of historical biological data and chloride-specific responses in biota, assemblages within Elm Creek cannot be effectively analyzed for response to chloride. The most sensitive macroinvertebrates are relatively low in abundance throughout the watershed, but are generally within the range expected at unimpaired sites. Despite the lack of a selective, diagnostic endpoint for impacts to biota from chloride, the SOE suggests that chloride cannot be ruled out as a stressor in AUIDs 508 and 760, but is unlikely a significant driver of the biotic communities in AUIDs 528, 732 and 525.

Table 4.14. Summarizes the weight of evidence supporting chloride ions as a causative stressor for the biological impairments observed throughout the Elm Creek watershed See Appendix A for definitions

Strength of Evidence Table -- Chloride					
	Scores for Impaired Reaches				
	Elm Creek	Rush Creek			Diamond Creek
Types of Evidence	508	528	732	760	525
Spatial/Temporal concurrence	+	0	+	+	--
Temporal sequence	0	0	0	0	0
Evidence of exposure, biological mechanism	+	0	+	+	--
Causal pathway	0	0	0	0	0
Field evidence of stress response	-	-	-	-	-
Field experiments/manipulations of exposure	NE	NE	NE	NE	NE
Laboratory analysis of site media	NE	NE	NE	NE	NE
Verified or tested predictions	NE	NE	NE	NE	NE
Symptoms	0	0	0	0	0
Mechanically plausible cause	+	+	+	+	+
Stressor-response in other field studies	0	0	0	0	0
Stressor-response in other lab studies	-	-	-	-	-
Stressor-response in ecological models	NE	NE	NE	NE	NE
Manipulation experiments at other sites	NE	NE	NE	NE	NE
Analogous stressors	NE	NE	NE	NE	NE
Consistency of evidence	-	-	-	-	-
Explanatory power of evidence	0	0	0	0	0

5. Conclusions and Recommendations

5.1. Summary of Probable Stressors

A range of stressors are likely impacting fish and macroinvertebrate communities throughout the Elm Creek Watershed. Six primary stressors have been identified throughout all AUIDs within the Elm Creek Watershed. However, the relative impact of these different stressors varies based on AUID. The relative impact of these different stressors is described below according to stressor type (Table 5.1).

Altered hydrology is the most common stressor throughout the watershed and likely influences biotic communities in all AUIDs. The relative impact of altered hydrology is most significant for fish assemblages and should be considered a primary stressor in all AUIDs (except 508). The impact of altered hydrology on macroinvertebrate assemblages is common throughout all sites, but most pronounced in AUIDs 525 and 528, where it should be considered a primary stressor. In all other AUIDs, altered hydrology should be considered a secondary stressor for macroinvertebrate assemblages.

Altered physical habitat is a common stressor throughout the watershed and likely influences biotic communities in all AUIDs. The relative impact of altered physical habitat is most significant for fish assemblages and should be considered a primary stressor in all AUIDs except 528 and 508 (where altered hydrology should be considered a secondary stressor). The impact of altered physical habitat on macroinvertebrate assemblages is common throughout all sites, but most pronounced in AUIDs 732 and 760, where it should be considered a primary stressor. In all other AUIDs, altered physical habitat should be considered a secondary stressor for macroinvertebrate assemblages.

Excess sediment is likely impacting biotic communities in all AUIDs. However, the relative importance of excess sediment as a stressor varies throughout the watershed. Sediments are likely significantly impacting fish in all AUIDs except 508 and 732, and should be considered a primary stressor. In AUIDs 508 and 732, sediments should be considered a secondary stressor for fish. Sediment impacts to macroinvertebrate communities should be considered a secondary stressor in all AUIDs except 528, where sediments should be considered a primary stressor.

Excess phosphorus is likely impacting biotic communities in all AUIDs. However, the relative importance of phosphorus as a stressor varies throughout the watershed. Phosphorus is likely significantly impacting both fish and macroinvertebrate communities in AUIDs 528 and 732 and should be considered a primary stressor—excess phosphorus in these AUIDs is also likely to be contributing to low DO as a stressor (described below). Excess phosphorus is likely impacting both fish and macroinvertebrate communities to a lesser degree in AUIDs 508 and 525, and should be considered a secondary stressor. In AUID 760, excess phosphorus should be considered a primary stressor for fish communities and a secondary stressor for macroinvertebrate communities.

Low DO is likely impacting biotic communities in all AUIDs. However, the relative importance of low DO as a stressor varies throughout the watershed. Low DO is likely significantly impacting fish communities in all AUIDs, and should be considered a primary stressor. Low DO is likely impacting macroinvertebrate communities to a lesser degree, and should be considered a secondary stressor in all AUIDs except 760.

However, the impacts of low DO in the Elm Creek system are potentially reach-specific and driven by diurnal/seasonal respiration in the associated wetland complexes.

Evidence for impacts of excess chloride on biotic communities is limited throughout the Elm Creek watershed. Chloride concentrations have infrequently exceeded water quality standards in all AUIDs and occasionally exceeded toxicity thresholds for sensitive fish and macroinvertebrate species. At certain times of the year (following snow melt and during periods of low flow), elevated chloride levels are present in stream reaches adjacent to major transportation corridors, but the relative contribution of excess chloride to the existing biotic impairments is unclear, particularly given the relative impact of different stressors describe above.

Table 5.1. Summary of stressors to biotic assemblages in the Elm Creek Watershed

HUC-8 Subwatershed	AUID (Last 3)	Stream	Reach Description	Biological Impairment	Primary Stressor					
					Altered Hydrology	Altered Physical Habitat	Excess Sediment	Excess Phosphorus	Low Dissolved Oxygen	Excess Chlorides
7010206 Mississippi River- Twin Cities	508	Elm Creek	Headwaters (Lk Medina 27-0146-00) to Mississippi River	Fish	**o**	**o**	**o**	**o**	•	/
				Macroinvertebrates	**o**	**o**	**o**	•	**o**	/
	525	Diamond Creek	Headwaters (French Lk 27-0127-00) to Unamed Lake	Fish	•	•	•	**o**	•	/
				Macroinvertebrates	**o**	**o**	**o**	•	**o**	/
	528	Rush Creek, Main Stem	Headwaters to Elm Creek	Fish	•	**o**	•	**o**	•	/
				Macroinvertebrates	•	**o**	•	•	**o**	/
	732	Rush Creek, South Fork	Unnamed lk (27-0439-00) to Rush Creek	Fish	•	•	**o**	•	•	/
				Macroinvertebrates	•	•	**o**	•	**o**	/
	760	Rush Creek, South Fork	Unnamed ditch to County Ditch 16	Fish	•	•	•	•	•	/
				Macroinvertebrates	**o**	•	**o**	**o**	•	/

• = Primary Stressor
o = Secondary Stressor
/ = Inconclusive Stressor

5.2. Recommendations

5.1.1. Management Recommendations

Based on the existing data, a range of potential actions are necessary to address the impaired biotic communities throughout the Elm Creek Watershed. However, given the variable impacts of different stressors throughout the different AUIDs in the Elm Creek Watershed and the similarity of best management practices (BMP) used to address these stressors, TMDLs, and implementation work should be prioritized to maximize improvements in biotic integrity.

The highest priority stressor to be addressed in the Elm Creek Watershed is altered hydrology. A wide range of physical, chemical and biological data suggest that altered hydrology is likely contributing to the impairment of biotic communities throughout the Elm Creek Watershed. Additionally, hydrologic alteration is also likely exacerbating the impacts of additional stressors (e.g., altered physical habitat, excess sediments, excess phosphorus and low DO) on the biotic communities throughout the Elm Creek Watershed. Implementation of the BMPs should focus on reductions in the rate and volume of runoff, increases in the annual percentage of groundwater infiltration and increased connection of the stream channel with the historically associated floodplain and wetland systems. Given the linkage between altered hydrology and secondary stressors, the BMPs that simultaneously address additional stressors should be prioritized over the BMPs in which the efficacy is more specifically focused on hydrologic modification. Implementation of the BMPs to address altered hydrology should be initially focused in upstream reaches of the watershed to maximize the extent of hydrologic restoration.

The second highest priority stressors are altered physical habitat and excess sediment. Both of these stressors will be addressed to a certain extent by mitigation of the hydrologic modification (described above). However, the BMPs that actively target eroding stream banks should also be considered. Stream bank restoration efforts should prioritize bioengineering techniques that simultaneously enhance habitat and reduce sediment erosion. Given the extent of channel modification, the instream BMPs should also be considered. Instream BMPs should focus on creating a diversity of flow regimes and be focused in areas where channel incision/widening is most significant. As with the hydrologic BMPs (described above), implementation in upstream reaches should be prioritized to maximize watershed-wide benefits.

Implementation of the BMPs to mitigate altered hydrology, altered physical habitat, and excess sediments will secondarily mitigate the impacts of both excess phosphorus and low DO. The aforementioned BMPs will primarily reduce particulate phosphorus concentrations throughout the Elm Creek system, but management efforts may also need to clarify/focus on the potential contributions of soluble phosphorus release from wetland sediment complexes. In the short-term (as the sources of low DO are addressed), projects that promote re-aeration (e.g., constructed riffles) and access to low DO refugia (e.g., associated lake habitat like Mill Pond, and Rice Lake) may play a significant role as remnant assemblages of DO intolerant species expand throughout the watershed.

5.1.2. Science and Monitoring Recommendations

Although this SID is grounded by a robust series of data sets, there are a range of data sets and technical studies that would enhance the management and restoration of biological communities throughout the Elm Creek Watershed. Recommendations are described below to enhance the technical understanding of the sources of different stressors, relative importance of different stressors and response of biotic communities to management activities over time.

Given the importance of phosphorus and DO as stressors in the Elm Creek Watershed, it is important to better understand the relative contribution of soluble reactive phosphorus (and potentially nitrogen) to low DO conditions, particularly with respect to connected lakes and wetlands. Historically accumulated phosphorus in lake and wetland sediments is released in soluble form under anoxic conditions. Because this phosphorus is in soluble form, it is easily taken up by periphyton, which can increase oxygen demand through autotrophic and (secondarily) heterotrophic respiration. Because these respiration process are likely key drivers of the low DO conditions in and downstream of large wetland complexes, it is important to understand the relative contribution of this historically accumulated phosphorus, relative to newly deposited phosphorus. Additionally, given the variation in oxygen profiles throughout different AUIDs, it will be important to further examine the spatial distribution low DO conditions, as it is likely that some reaches are more influence by low DO conditions than others within the same AUID (particularly 508). Similarly, additional monitoring/modeling work should be conducted to better understand the contribution of nitrification (i.e., nitrogen oxygen demand) to low DO conditions, particularly in Rush Creek.

To effectively manage biological communities and prioritize restoration work into the future, it will be important to clarify the relationship between assessment (i.e., AUIDs) and management (watershed/hydrologic and jurisdictional) boundaries. The structure of assessment units throughout the watershed is highly variable. For example, AUID 508 extends from the headwaters to the mouth of the main stem of Elm Creek, while AUID 760 represents a stream reach of less than one mile in Rush Creek. Currently, each of these AUIDs have equivalent influence on management decisions throughout the watersheds—and this may results in an (over)underrepresentation the overall health of the biotic communities throughout Elm Creek. Given the variability of stressors and biotic communities observed across AUIDs (in 508 in particular) it is important to clarify benchmarks and targets for success; for example, would a reduction in the occurrence of stressors over a large area (e.g., the upper Elm Creek Watershed) be viewed more, or less, favorably than an improvement in the biotic community in a relatively isolated and small stream reach (e.g., AUID 760)? Similarly, if different assessment sites (or biological communities) within the same AUID respond differently to management/restoration work—how should these results be viewed in the context of restoration progress?

Additional monitoring of stressors that are currently identified as inconclusive will also clarify the need for directed management/restoration work. In particularly, the available nitrate, ammonia, pH and chloride data all suggest that these potential stressors are not having a significant impact on the structure of biotic communities throughout the Elm Creek Watershed. However, all of the data sets for these parameters are limited by either their spatial or temporal coverage. Because of these “data gaps”, it is possible that these stressors are having an impact that is masked by the limitations in the available data sets. To address these data gaps, additional sampling should clarify the 1) extent of ammonia and nitrate pollution throughout the watershed (beyond lower Elm Creek), 2) relative biological importance and source of episodic high pH events observed in AUID 508 and 3) scope and extent of chloride pollution, potentially by expanding winter (and low flow) monitoring efforts—particularly adjacent to high use transportation corridors.

Given the potential conflict between the TSS concentrations and the response of sediment sensitive fish and macroinvertebrate metrics, future monitoring and assessment work should clarify the relative contribution of sediment impairment from ongoing runoff vs. historical sedimentation and stream bottom embedding. Monitoring data suggests that the TSS values have been declining in Rush and Diamond Creeks, such that chemical measurements alone would indicate compliance with the 30 mg/L TSS water quality standard in these systems. However, the structure of the fish and macroinvertebrate assemblages in these AUIDs continue to suggest potential sediment impacts. Clarification of the driver of the biotic response will enhance the efficacy of the BMP implementation.

Table 5.2. Recommended prioritization of TMDLs relative to the stressors contributing to the biological impairment in the Elm Creek Watershed

Stressor	Priority	Comment
Altered Hydrology	High	TMDL should focus on reestablishing historical hydrologic patterns.
Altered Physical Habitat	High	TMDL should focus on increasing the diversity of sediment/substrate types and functionality of large woody debris.
Excess Sediments	High	TMDL should be conducted concurrent to altered habitat to focus on increasing the diversity of bed sediment size.
Excess Phosphorus	Medium	TMDL should focus on addressing the current loads and historical accumulation of phosphorus in wetlands (should potentially be expanded to include nitrogen).
Low DO	Medium	Additional monitoring should be conducted to describe the relative contribution of low DO from wetland complexes and the alignment of biological monitoring stations with different habitat types. TMDL should focus on historical accumulation of phosphorus in wetlands and nitrification (particularly in Rush Creek).

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7. Appendix A – Supplemental Tables

Table A1. Values used to score evidence in the SID Process.

Rank	Meaning	Caveat
+++	*Convincingly supports*	*but other possible factors*
++	*Strongly supports*	*but potential confounding factors*
+	*Some support*	*but association is not necessarily causal*
0	*Neither supports nor weakens*	*(ambiguous evidence)*
-	*Somewhat weakens support*	*but association does not necessarily reject as a cause*
--	*Strongly weakens*	*but exposure or mechanism possible missed*
---	*Convincingly weakens*	*but other possible factors*
R	*Refutes*	*findings refute the case unequivocally*
NE	*No evidence available*	
NA	*Evidence not applicable*	
D	*Evidence is diagnostic of cause*	

Table A2. The SOE Scores for various types of evidence

Types of Evidence	Possible values, high to low
Evidence using data from case	
Spatial / temporal co-occurrence	+, 0, ---, R
Evidence of exposure, biological mechanism	++, +, 0, --, R
Causal pathway	++, +, 0, -, ---
Field evidence of stressor-response	++, +, 0, -, --
Field experiments / manipulation of exposure	+++, 0, ---, R
Laboratory analysis of site media	++, +, 0, -
Temporal sequence	+, 0, ---, R
Verified or tested predictions	+++, +, 0, -, ---, R
Symptoms	D, +, 0, ---, R
Evidence using data from other systems	
Mechanistically plausible cause	+, 0, --
Stressor-response relationships in other field studies	++, +, 0, -, --
Stressor-response relationships in other lab studies	++, +, 0, -, --
Stressor-response relationships in ecological models	+, 0, -
Manipulation of exposure experiments at other sites	+++, +, 0, --
Analogous stressors	++, +, -, --
Multiple lines of evidence	
Consistency of evidence	+++, +, 0, -, --
Explanatory power of evidence	++, 0, -