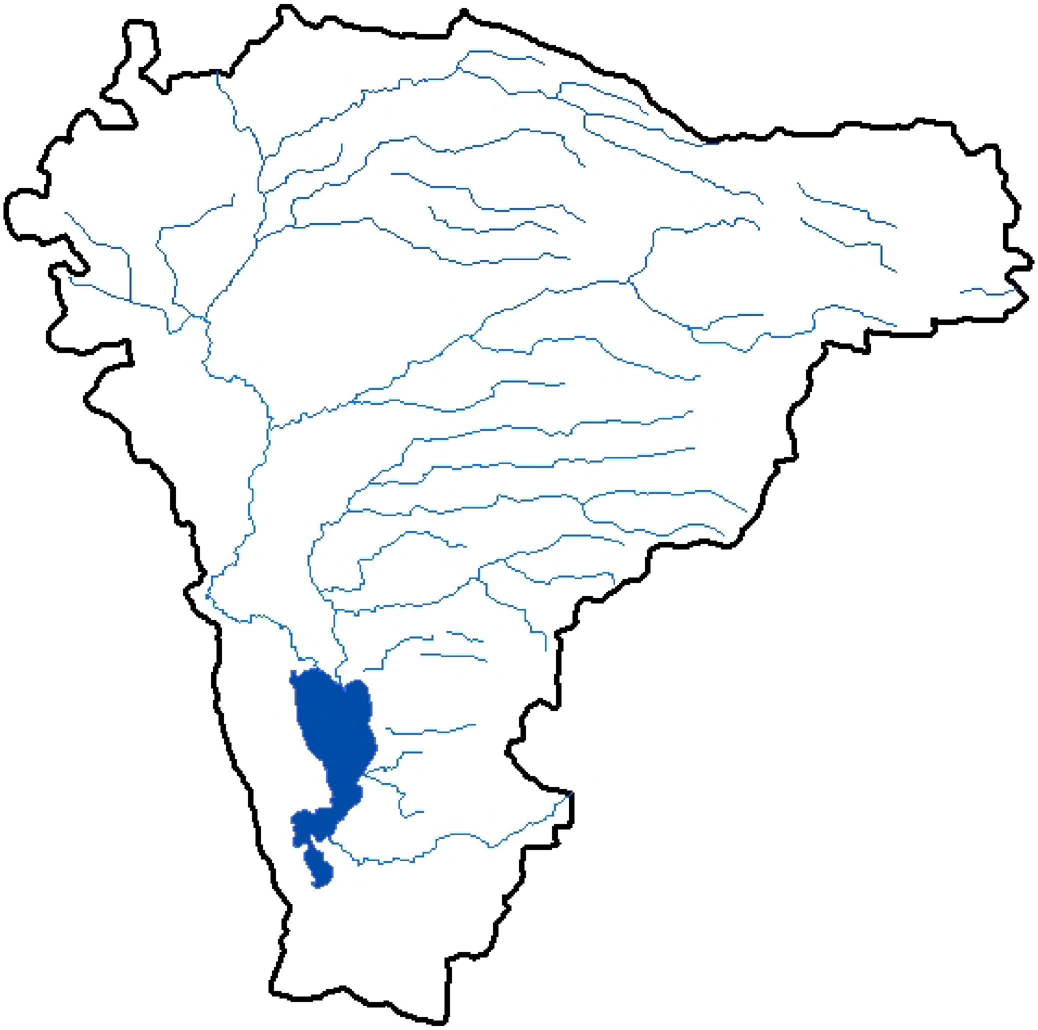
Little Rock Creek

STRESSOR IDENTIFICATION REPORT



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ABBREVIATIONS

AUID	Assessment Unit Identification
CADDIS	Causal Analysis/Diagnosis Decision Information System
DNR	Department of Natural Resources
DO	Dissolved oxygen
EPA	Environmental Protection Agency
GIS	Geographic Information Systems
IBI	Index of Biological Integrity
LRCW	Little Rock Creek Watershed
MPCA	Minnesota Pollution Control Agency
NCHF	North Central Hardwood Forest
NRRI	Natural Resources Research Institute
OBWELL	Observation Well
SI	Stressor Identification
SOE	Strength of Evidence
STORET	STOrage and RETrieval system
SWCD	Soil and Water Conservation District
TC	Technical Committee
TMDL	Total Maximum Daily Load
TP	Total phosphorus
USEPA	United States Environmental Protection Agency
YOY	Young of year

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ABSTRACT

Little Rock Creek is a DNR designated trout stream (Class 1A Brook and Brown Trout) located in central Minnesota near the City of Rice. The Little Rock Creek Watershed is located in Benton and Morrison Counties. Little Rock Creek was listed on the State's 303(d) List of Impaired Waters in 2002 for lack of coldwater fish assemblage. Few trout were captured in the 1999 MPCA fish survey; there was also an absence of sculpin and burbot.

Before Total Maximum Daily Loads (TMDLs) can be developed for this water body, a stressor identification analysis is necessary to determine the specific physical and/or chemical factors that are causing the biological impairment. This analysis identifies stressors in the Little Rock Creek Watershed using a series of logical steps based on the United States Environmental Protection Agency (USEPA) Stressor Identification Guidance Document and the Minnesota Pollution Control Agency Biota TMDL Protocols and Submittal Requirements document. Impairments were evaluated, candidate causes of impairment were described, relationships between causes, stressors, and biotic conditions were assessed, and probable stressors were identified based on strength of evidence from all available data.

Upon completion of the Little Rock Creek stressor identification analysis, it was determined that TMDLs will be developed for temperature, bedded sediment, nitrates, and dissolved oxygen, by calculating the total pollutant load with reference to flow as source of impairment.

INTRODUCTION

This report describes a Stressor Identification study on Little Rock Creek, a biologically impaired trout stream located in central Minnesota (Figure 1). The Little Rock Creek Watershed is 67,650 acres and is nearly evenly split between Benton (36,030 acres) and Morrison (31,620 acres) counties.

The watershed receives approximately 28 inches of precipitation per year (National Weather Service). The average maximum air temperature for the months of May through September is approx 75 degrees F, average min temp is 55 degrees (National Climatic Data Center).

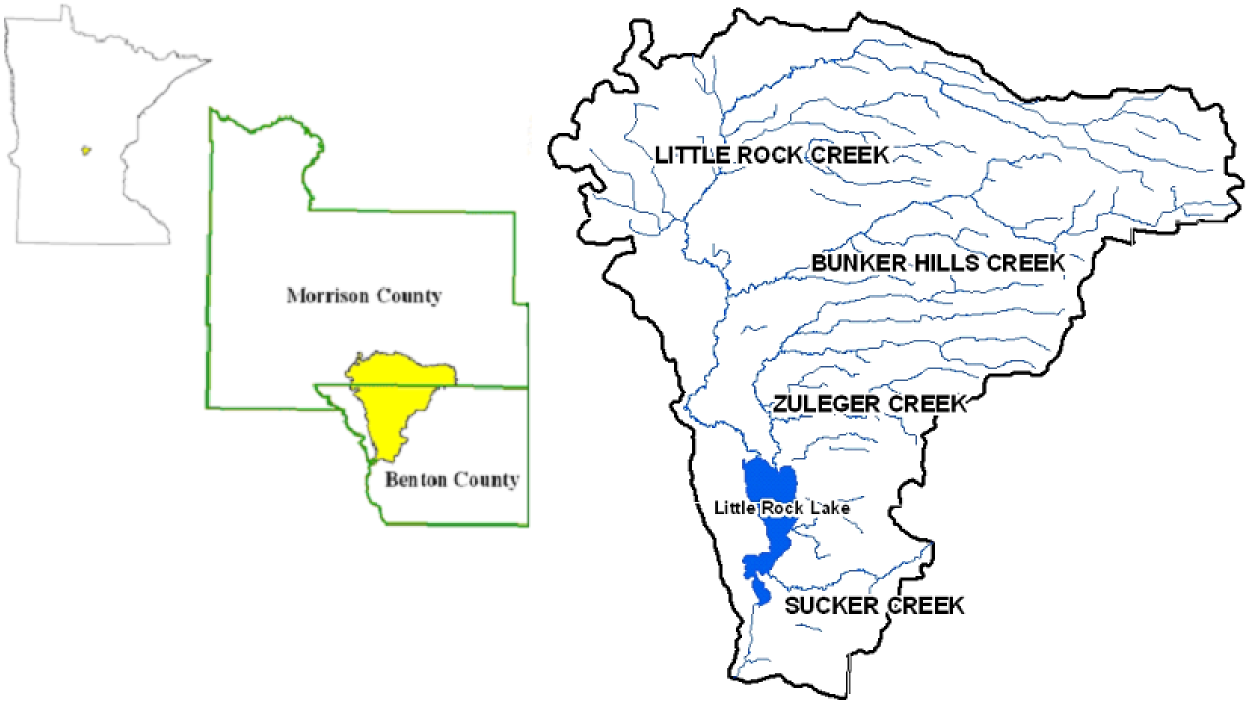


Figure 1. Little Rock Creek Watershed location

Little Rock Creek is made up of two main stream types, intermittent and perennial. Intermittent streams flow at different times of the year, or seasonally, when there is enough water from rainfall, springs, or other surface sources such as melting snow. Perennial streams flow year-round (USEPA). The main stream of Little Rock Creek is perennial, whereas, a majority of the tributaries are intermittent or have been converted to drainage ditches (Figure 2).

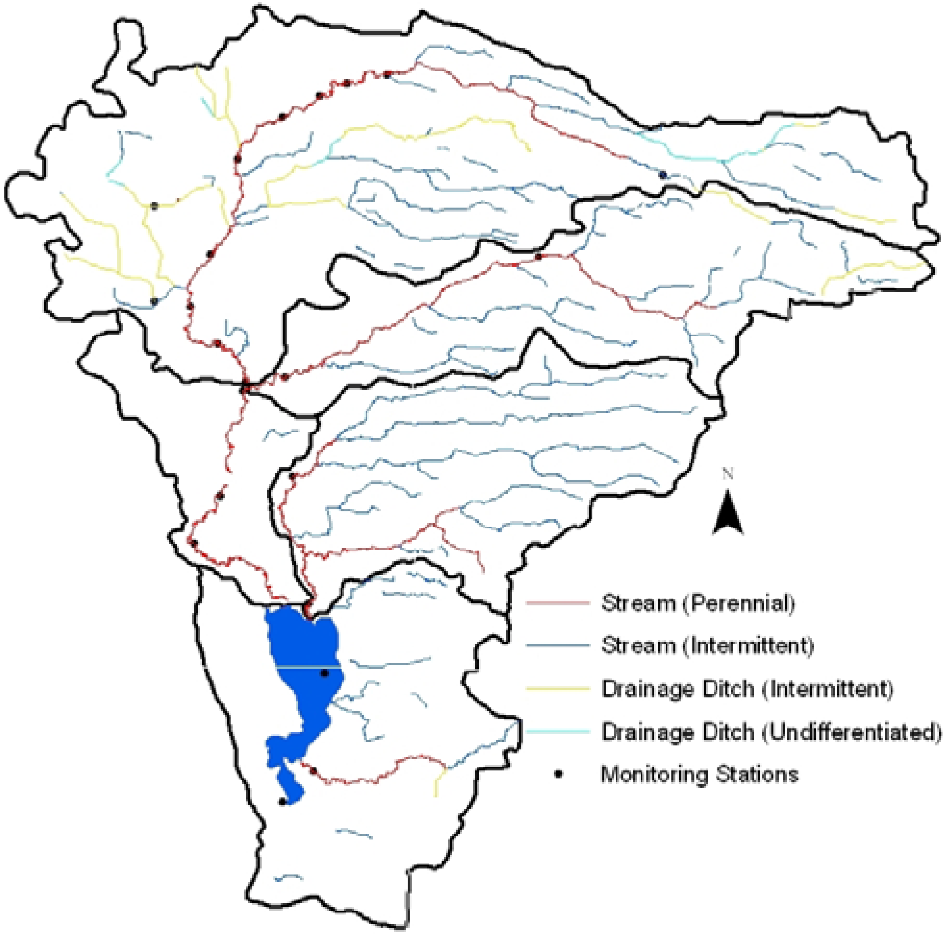


Figure 2. Little Rock Creek stream types

The drainage area for Little Rock Creek has been defined by a groundwater model developed for this study. Note that the watershed boundary relates to the basin where surface water drains whereas the groundwater boundary delineates where ground water discharges. The drainage area for the Little Rock Creek Watershed is 215,701acres or approximately 337 square miles (Figure 3). The drainage area is the project boundary, which is outlined in red Figure 3. The elevation in the watershed ranges from 1,296 feet in the northeast, to 1,017 feet at the outlet of Little Rock Lake into the Mississippi River. Groundwater flows west through the Little Rock Creek Watershed, discharging both to the Creek and to the Mississippi River.

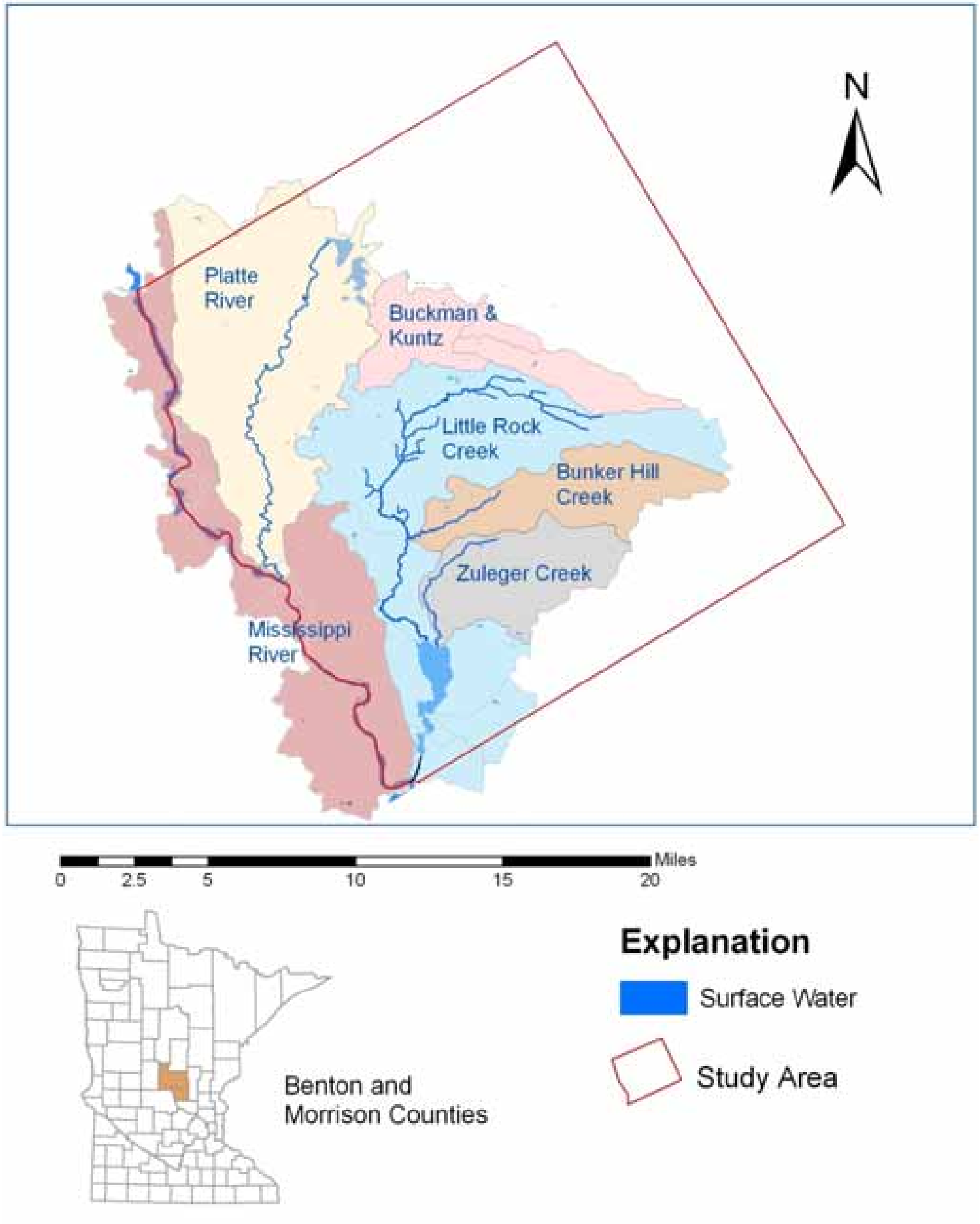
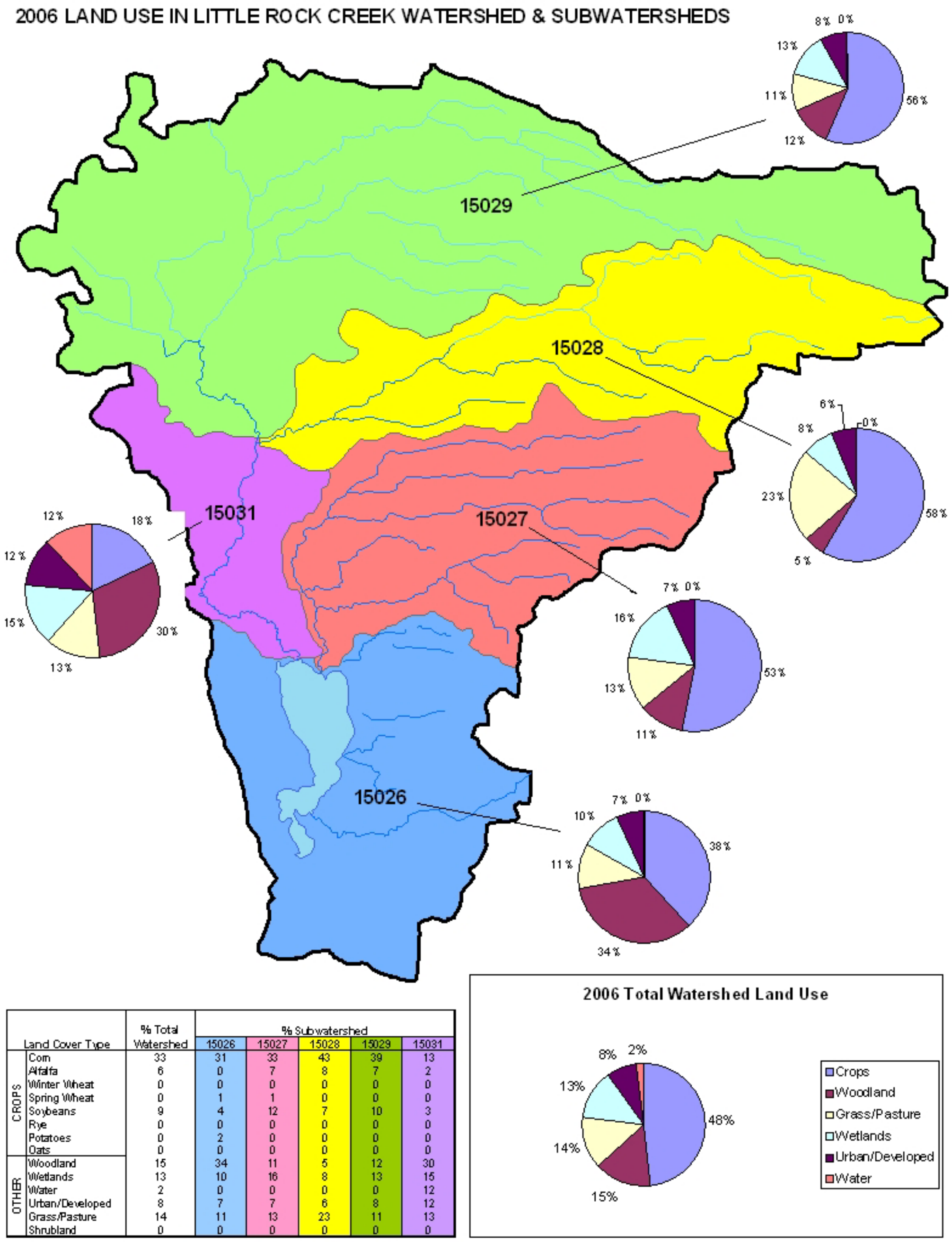


Figure 3. Little Rock Creek drainage area boundary

Land Use

According to the National Agricultural Statistical Service, in 2006 the land use in the watershed consisted of 48% crops, 15% woodland, 15% grass/pasture, 13% wetlands, 8% urban development, and 2% water (Figure 4). Due to the predominance of sandy soils in the watershed, many croplands are irrigated. Channelization is not prevalent on the mainstem of Little Rock Creek although many tributaries in the upper watershed have been ditched and straightened.



	Land Cover Type	% Total Watershed	% Subwatershed 15026	15027	15028	15029	15031
CROPS	Corn	33	31	33	43	39	13
	Alfalfa	6	0	7	8	7	2
	Winter Wheat	0	0	0	0	0	0
	Spring Wheat	0	1	1	0	0	0
	Soybeans	9	4	12	7	10	3
	Rye	0	0	0	0	0	0
	Potatoes	0	2	0	0	0	0
	Oats	0	0	0	0	0	0
OTHER	Woodland	15	34	11	5	12	30
	Wetlands	13	10	16	8	13	15
	Water	2	0	0	0	0	12
	Urban/Developed	8	7	7	6	8	12
	Grass/Pasture	14	11	13	23	11	13
	Shrubland	0	0	0	0	0	0

Figure 4. 2006 Land use

Soils

The watershed has alluvial soils made up predominantly of fine sands. The topography is flat to gently rolling. Most of the watershed is in the Agram Sand Plain (37,799 acres) and the Pierz Drumlin Plain (31,322 acres) (Figure 5). A very small part of the watershed near the Little Rock Lake outlet is in the Mississippi Sand Plain (less than 4 acres).

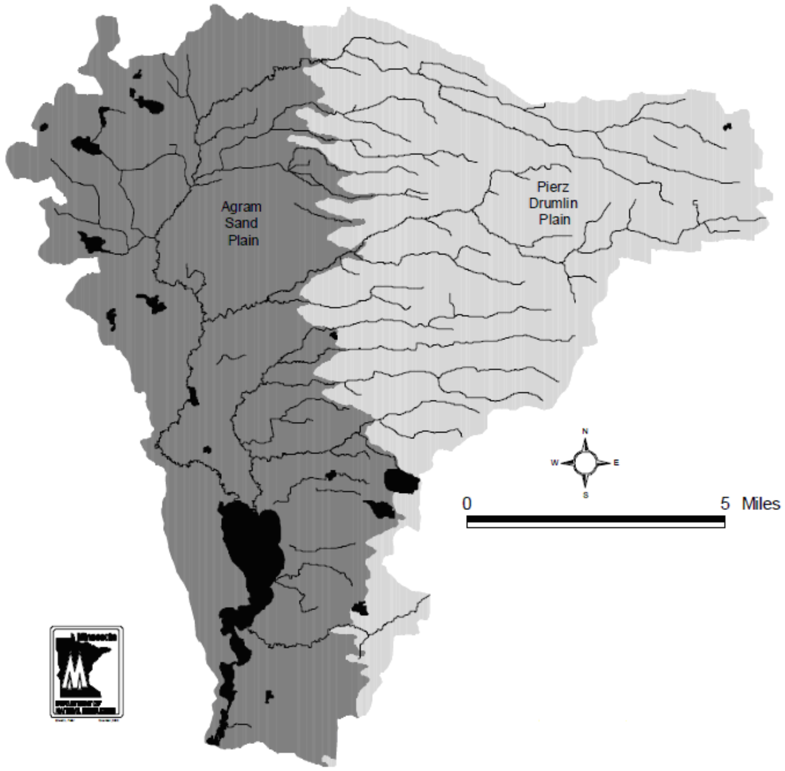


Figure 5. Watershed soils

Stream habitat in Little Rock Creek changes throughout its course. The upstream portion is slow, marshy and warmer than downstream sections. From 230th Avenue in Morrison County and downstream, the creek picks up ground water from springs and the temperature drops (Figure 7). The section defined as DNR trout water begins at 230th Avenue and extends downstream to the confluence with Bunker Hill Creek. Substrates in this section change from sand and silt in the vicinity of 230th Avenue to boulder, rock, gravel and sand near Co. Road 26 (Nature Road). Coarse substrates persist from Co. 26 downstream approximately 1.5 river miles to where the stream slows and becomes more meandering. Sand and silt substrates dominate throughout the remainder of the stream. Riffle, pool and undercut bank type habitat components do not appear to be limiting factors on brown trout abundance (MN DNR).

Ecoregion

The EPA defines ecoregions for Minnesota based on areas of relative homogeneity for land use, soils, landform, and potential natural vegetation (MPCA). The Little Rock Creek watershed is located within the North Central Hardwood Forest Ecoregion (US EPA). This ecoregion is an area of transition between the forested areas to the north and east and the agricultural areas to the south and west. The terrain varies from rolling hills to smaller plains. Upland areas are forested by hardwoods and conifers. Plains include livestock pastures, hay fields and row crops such as potatoes, beans, peas and corn.

DISCUSSION OF THE FISH COMMUNITY

Little Rock Creek is lacking a diverse coldwater fish community. The most popular coldwater species in the creek is the brown trout.

Trout fishery

The following is an excerpt from the 1999, 2000, and 2001 Little Falls DNR Fisheries Stream Population Assessments.

> Little Rock Creek has supported a wild brown trout population since they were introduced into Little Rock Lake in 1908. Brook trout were stocked annually from 1962 through 1971 and provided angling opportunities for large size fish although no natural reproduction was documented. Brown trout were present in routine stream assessments through the late 1980's. The population assessment done in 1992, however, failed to document the presence of brown trout suggesting that population may have become critically low during the drought years of the late 1980's and early 1990's. The stream contains a diverse fish community with 28 species sampled in a prior assessment (1992). White sucker, blacknose dace, Johnny darter and creek chub were the most common species sampled. In an effort to reestablish a self-sustaining brown trout population in Little Rock Creek, 455, 452 and 450 wild brown trout were stocked in the springs of 1995, 1996 and 1997 respectively. In the spring of 1998 excess brood stock brown trout were obtained from the Lanesboro Hatchery and stocked in the stream. This stocking consisted of 1,978 yearling trout (adipose fin clip) that were raised from eggs gathered from wild trout from two streams. Beaver Creek and Trout Run in southeastern Minnesota were the sources for all stocked trout. To aid in the reestablishment process a permanent rule change was pursued and obtained to close the stream to brown trout harvest for 5 years beginning in 1996. In addition, angling for all species was limited to the use of artificial lures. The upstream boundary for these regulation changes was established at 230th Avenue in Morrison County and the downstream limit was established at the upstream side of the dam at the Sartell Wildlife Management Area in Benton County (Figure 7).
>
> Stream temperature recorders deployed during summers from 1995 through 1999 have consistently indicated temperatures within the preferred range for brown trout. In addition, temperature recorders were also deployed in suspected spawning areas during the winter of 1997-1998 to monitor winter temperatures. Winter oxygen tests done during 1989, 1990 and 1992 at several sites detected saturated oxygen levels.
>
> Trout management problems
>
> - Flow patterns and water temperatures should be monitored as irrigation increases in the watershed. The effect of irrigation on groundwater inputs is a concern. A renewed interest in cleaning out ditches is also a concern.
> - Competition with other species could be a limiting factor on brown trout abundance. High numbers of white suckers, young of year (YOY) northern pike, creek chubs and shiner species may compete for food while pike predation may also limit trout abundance. Northern pike abundance fluctuates probably in association with flow patterns and is predominantly made up of YOY individuals. Adult northern pike may prey upon brown trout during the spring spawning period. The presence of the Sartell Wildlife Management Area impoundment and the lack of a suitable site for a barrier make northern pike control difficult. Stop logs were removed and the impounded area drawn down during the winters of 1996 through 1998 to cause winterkill and/or

movement of northern pike and carp to downstream reaches. Stop logs were replaced prior to the spring pike spawning run to prevent northerns from returning to Little Rock Creek. The effectiveness of these drawdowns is unknown.

- Studies of stream productivity are necessary to determine brown trout carrying capacity. A study of benthic invertebrates was recently completed by a graduate student from the St. Cloud State University Biology Department. Results of this study showed the invertebrate community to be dominated by Chironomid Species. Several intolerant species were present in the stream although their numbers were low. Using the insect community as a barometer of stream health, it was suggested that habitat quality increased in a downstream direction. Agricultural impacts were blamed for poor habitat quality in upstream areas. This study employed the Hilsenhoff Biotic Index, which uses insects as a measure of ecological integrity. This index suggested fair to good water quality in Little Rock Creek.
- Rock covering a pipeline crossing near Co. 26 (Nature Road) and a rock bridge approximately 1.5 river miles downstream of Co. 26 are potential barriers to movement during low flow periods (Figure 7). Due to these barriers, trout may be excluded from some spawning sites during drought conditions.
- Natural recruitment of brown trout in Little Rock Creek has been low to nonexistent since the 1980's. Ten naturally produced fingerlings were captured in 1999, one in 1998, and none were seen in fall assessments from 1995 through 1997. Sexually mature trout have been present since 1996. The quality and quantity of spawning habitat is not known and few redd sites have been identified. Embeddedness may also be a factor in redd failures resulting in low recruitment. Erosion has covered some spawning areas used in the past. Harsh winters in 1995-1996 and 1996-1997 may have caused high winter egg mortality. High spring flows in 1996 and 1997 may have also caused year class failures. Mild winters in 1997-1998 and 1998-1999 may have made conditions favorable to allow some natural reproduction to occur.
- Adult brown trout numbers were extremely low in 1999 despite annual stocking of yearling and adult fish. Active redds were observed by local Trout Unlimited members in the fall of 1998 but not in 1999. T. U. members failed to observe brown trout during a spring 1999 stream cleanup outing. A rapid, catastrophic brown trout population decline appears to have occurred in response to some unknown environmental factor. High winter mortality or a high incidence of emigration may account for low numbers of fish seen during this assessment. Reports of brown trout being caught in the Mississippi River and one caught during a spring ice out trapnetting on lower Little Rock Lake suggests that some emigration did occur. Survival of brown trout in the Mississippi River for prolonged periods is doubtful unless cool microclimates are available to carry trout through warm water periods. Winter mortality factors and their degrees of severity are unknown. Fish species sampled during this assessment were mainly pollution tolerant species such as central mudminnows, white suckers and creek chubs suggesting a potential water quality problem. Specific reasons for low brown trout inhabitation of Little Rock Creek need to be identified before future time and monetary resources are spent.

Other cool/cold water species

Trout aren't the only cool/cold water species of concern in the creek. Other sensitive cool/cold water species such as burbot and hornyhead chub have not been seen since the early 1990's. Logperch and longnose dace are currently present in low numbers. Sculpins have not been recorded in contemporary surveys but are present in other streams in the region.

The impairment is also supported by the fact that recent surveys reveal that the fish community is dominated by tolerant warm water species, such as blacknose dace, creek chub, white sucker, central mudminnow, and bigmouth shiner. Very few obligate cold or cool water species were captured, including brook trout, brown trout, longnose dace, and brook stickleback. The presence and increased abundance of sensitive cool/cold water species would be good indicators of improved water quality.

DESCRIPTION OF IMPAIRMENT

During the late 1990's the MPCA's Biological Monitoring Program developed biological criteria for rivers and streams in Minnesota. Indices of Biological Integrity (IBI) were developed for the Upper Mississippi Basin, the St. Croix Basin and Minnesota Basins, as well as the Lake Agassiz Plain Ecoregion. The MPCA monitoring strategy had two parts. One part involved sampling target sites within each basin to obtain information regarding reference conditions. The other was to establish a condition monitoring program, where sites were established and sampled using a stratified sampling design. Fifty sites were randomly selected in the Upper Mississippi River Basin. One of those sites, station 2 (Figure 7), was located on Little Rock Creek. The MPCA collected fishery IBI data in 1999 at the site, which resulted in Little Rock Creek being listed for lack of coldwater assemblage on Minnesota's 303(d) List of Impaired Waters in 2002 due to a low fish IBI score (Table 1). The entire reach is impaired from the headwaters to the Mississippi River. Little Rock Creek is classified under Minnesota Rule 6264.0050 Subpart 4 as a designated trout stream.

Table 1. Little Rock Creek 303(d) listing information

Reach	Description	Year Listed	River ID#	Affected use	Aquatic Life	TMDL Target start	TMDL Target completion	CALM Category*
Little Rock Creek	T39 R30W S27, south line to T38 R31W S28, east line (trout stream)	2002	07010201-548	Aquatic life	Lack of a coldwater assemblage	2004	2012	5C

*CALM (Consolidation Assessment and Listing Methodology)
5C: Impaired by one pollutant and no TMDL study plan is approved by EPA

Water quality standards

The federal Clean Water Act (CWA) requires states to adopt water-quality standards to protect waters from pollution. These standards define how much of a pollutant can be in the water and still allow it to meet designated use(s). "Impaired waters" are those waters that do not meet water-quality standards for one or more pollutants, thus they are "impaired" for their designated use(s) (MPCA).

“Designated uses” are the uses that water resources and their associated aquatic communities provide. Seven designated uses, designated Class 1 through 7, are defined in Minn. R. 7050.0140. Little Rock Creek is classified in Minnesota Rule 7050.0470 Subpart 4 for three designated uses:

Class 1B: Domestic Consumption – Treated water will meet both the primary and secondary drinking water standards with approved disinfection.

Class 2A: Aquatic Life and Recreation – Waters will be such as to permit the propagation and maintenance of a healthy community of cold water sport or commercial fish and associated aquatic life.

Class 3B: Industrial Consumption – Waters will be such as to permit its use for general industrial purposes, except for food processing, with only a moderate degree of treatment.

History of impairment classification

The Little Rock Creek biological impairment is for "lack of a coldwater fish assemblage". This is a correction to the 2002 Impaired Waters listing. The MPCA gives the following information as the justification and history for relisting Little Rock Creek.

2002

Little Rock Creek was initially placed on the 2002 TMDL list due to a biological impairment as indicated by a poor warmwater fish IBI score at station 2. The fish community at Little Rock Creek station 2 was characterized by a prevalence of highly tolerant warmwater species, no intolerant species, and complete absence of species indicative of coldwater habitats. These fish community attributes along with others suggest that the fish community at station 2 deviates substantially from what one would expect in an unimpaired coldwater system (S.Niemela, pers comm.).

A disappearance of trout was noted through surveys by the DNR area fisheries office, Mid Minnesota Trout Unlimited and local anglers in 1999. This information further supports the listing of this stream for biological impairment.

2004

In 2002, it was understood by MPCA that the warmwater use designation (2B) was applicable to all of Little Rock Creek. During the 2004 assessment cycle the reach was split into three sections (-547, -548, and -549) based on the recognition that the designated use for part of the headwaters was not Class 2B (warmwater), but was Class 2A (coldwater) as listed in Minn.R. 7050.0470 and also a designated trout stream as listed in Minn.R 6264.0050 subp.4. The impairment was associated with the AUID 07010201-548 (T39 R30W S27 south line to T38 R31W S28 east line) based on the location of the biological monitoring site.

After consultation with biologists at the Minnesota Department of Natural Resources office in Little Falls, it was determined that the upstream reaches (-547 and part of -548) of Little Rock Creek (i.e. upstream of the eastern boundary of 230th Avenue) lacked the potential to support a coldwater fishery. Thus, for management purposes the MN DNR considered these reaches to be a warmwater stream. The application of a warmwater IBI to determine the impairment status was in their opinion, a reasonable approach.

2006
During the 2006 assessment cycle Little Rock Creek was removed from the Impaired Waters list as a correction when a routine examination of designated uses of assessed waterbodies revealed that the Creek was a designated Class 2A coldwater stream and at that point MPCA did not have a tool developed for assessing the biology of coldwater streams.

2008
During the 2008 assessment process the MPCA assessment team recognized that the unique history of the assessment, listing and designated uses on this AUID necessitated a reevaluation of its removal from the Impaired Waters list. While the data are preliminary and do not meet the typical criteria for listing, they do indicate that impaired conditions exist not only for biota, but also dissolved oxygen, turbidity, fecal coliform, and excessive nutrients in the lake. Therefore, it was determined that Little Rock Creek will be placed back on the Impaired Waters list and assessed as impaired using a narrative interpretation of MN Rule Chapter 7050.0150 as a Class 2A waterbody.

A follow up meeting on biological impairment was held on 4/16/07 and attended by Scott Niemela, Jim Hodgson, Howard Markus, Carol Sinden, and Doug Hansen. The result was to assess Aquatic Life use as non-supporting and develop terminology for the new impairment. This is a 2A segment, and the 2B Fish IBI does not apply. The narrative standards and local resource staff agree there is an impairment to the biology.

PROJECT OVERVIEW

The Little Rock Creek TMDL project was scheduled to be completed in three phases. Phase 1 consisted of collecting and organizing existing data into a GIS map project and entered into STORET. The data was reviewed by technical and stakeholder committees. The committees explored watershed information and landscape activities and developed a list of potential stressors. The technical committee constructed a project flow chart and also identified additional data needs for Phase 2 (Figure 6). Phase 1 was completed in 2003.

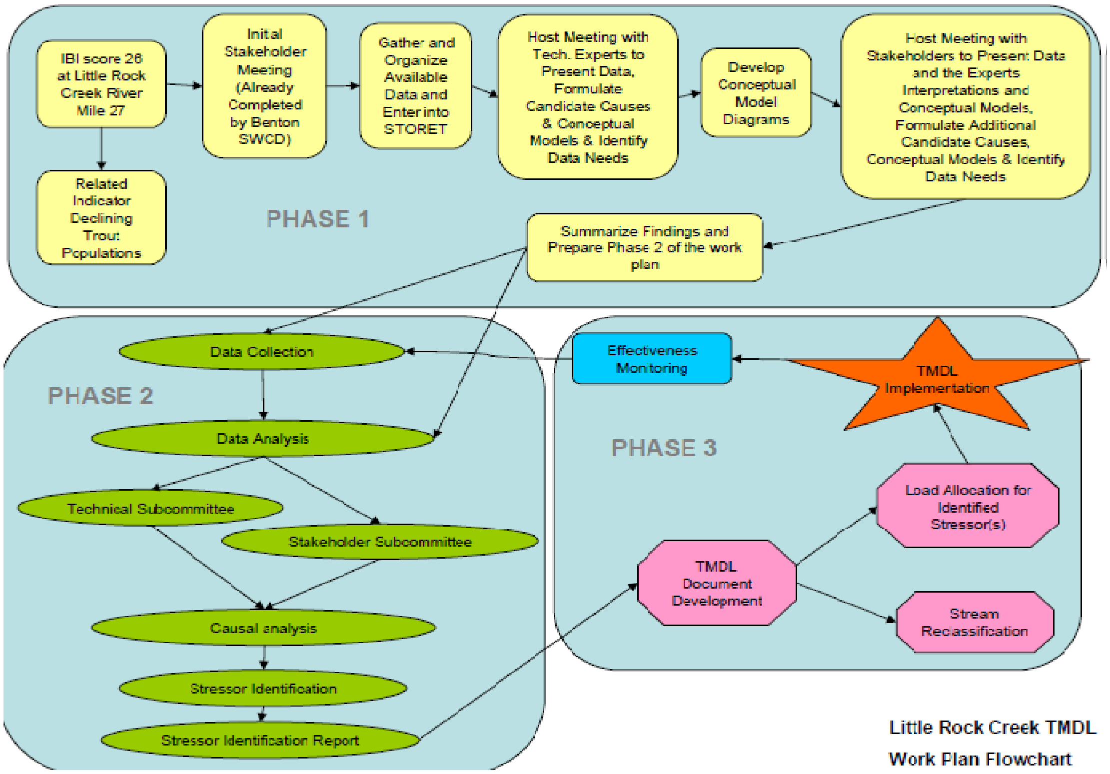


Figure 6. Little Rock Creek TMDL Phase 1-3 Flow Chart

Phase 2 consists of three main tasks: 1) collect additional physical, chemical, and biological data, 2) general project administration, coordinating, and bookkeeping, 3) Stressor Identification Report compilation. The product of Phase 2 will be a Stressor Identification Report that includes watershed GIS data, stakeholder meeting comments, technical group meetings and coordination, causal analysis and stressor identification documentation.

Phase 2 began in 2006 and is scheduled to be completed by September 30, 2009. This document contains the complete stressor identification for lack of cold water fish assemblage.

As required, TMDL development is to be undertaken for applicable pollutants identified through the 303(d) listing process. However, TMDLs cannot be computed on 303(d) listings due to biological impairment because the actual stressors causing the impairment have not been determined. The purpose of this study is to identify stressors contributing to lack of cold water fish assemblage in Little Rock Creek. This report describes impairments, identifies candidate causes of impairment, evaluates relationships of these with the biological community, and identifies the most likely stressors using elimination and strength of evidence analyses. Based on the stressor identification analysis, TMDLs will be developed for applicable pollutants.

During Phase 3, TMDLs will be developed for temperature, bedded sediment, nitrates, and dissolved oxygen, by calculating the total pollutant load with reference to flow as the source of impairment. An implementation plan will also be developed to address these stressors and their sources.

EXISTING DATA

During Phase 1 of the project, all available water quality and other environmental data and information for Little Rock Creek were reviewed. These data were compiled from various sources including the Benton SWCD, DNR Fisheries, DNR Waters, and the MPCA STORET database. A complete listing of compiled data can be found in the Appendix.

NEW DATA COLLECTION

Twenty-two water quality monitoring stations were established during Phase 2 in order to collect additional data (Figure 7). Most of the stations were located on the main stem of Little Rock Creek.

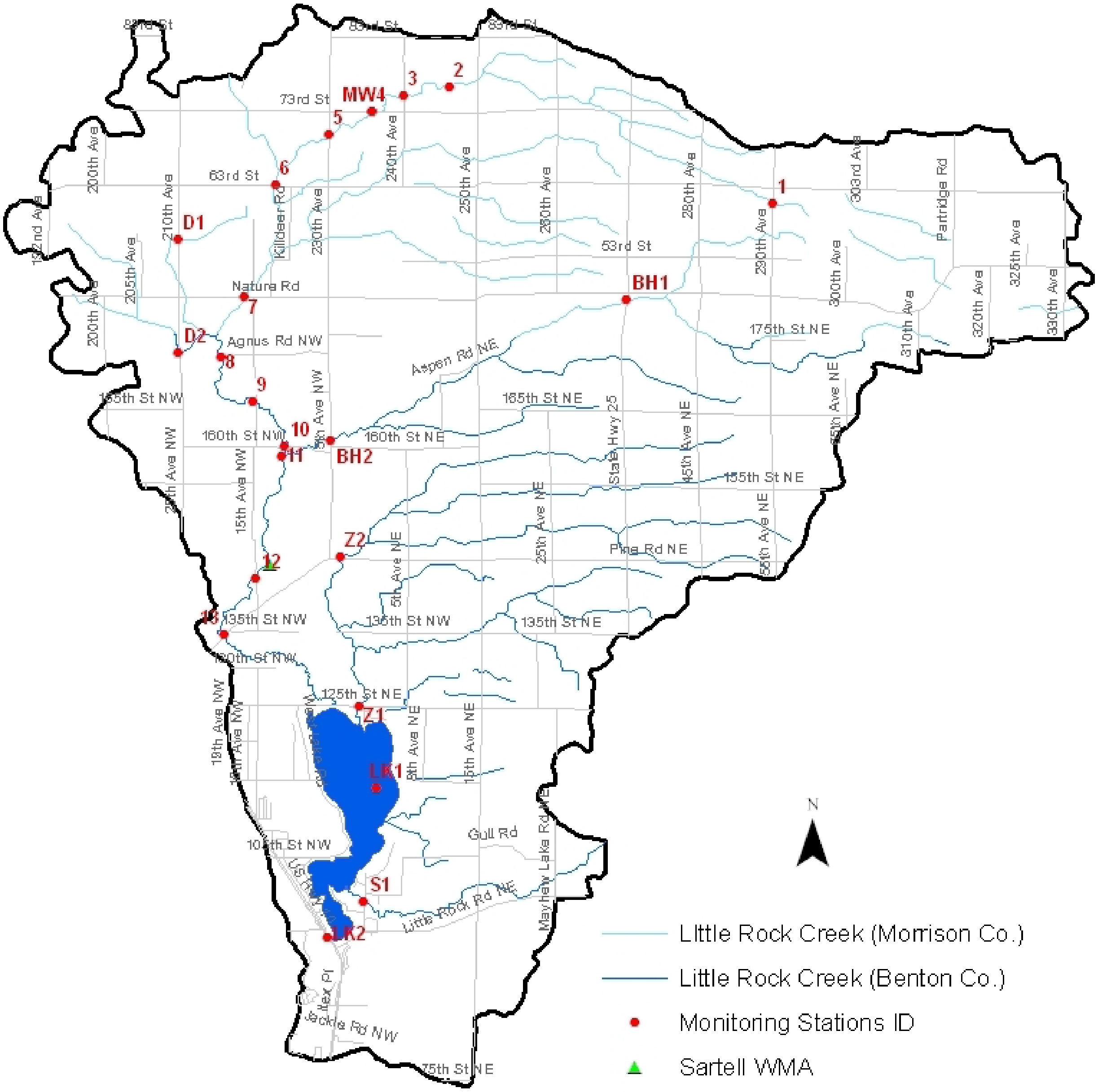


Figure 7. Phase II Little Rock Creek monitoring station locations

The data were collected by a variety of organizations and agencies and each used their own site identification numbers. In order to understand and compare data, a universal monitoring station code was developed for the project. A comprehensive list of the monitoring station identification numbers, responsible agency, data collected and the STORET number associated with the station is located in the Appendix.

The Sartell WMA - DNR Impoundment was identified in the monitoring station map in Figure 7. Little Rock Creek was impounded by a sheet pile structure in the late 1970's. The DNR installed and operates the impoundment in the Sartell Wildlife Management Area, creating Wetland 50P (Figures 7 and 8). The structure consists of an inline sheet pile weir with two stop log bays for pool manipulation.

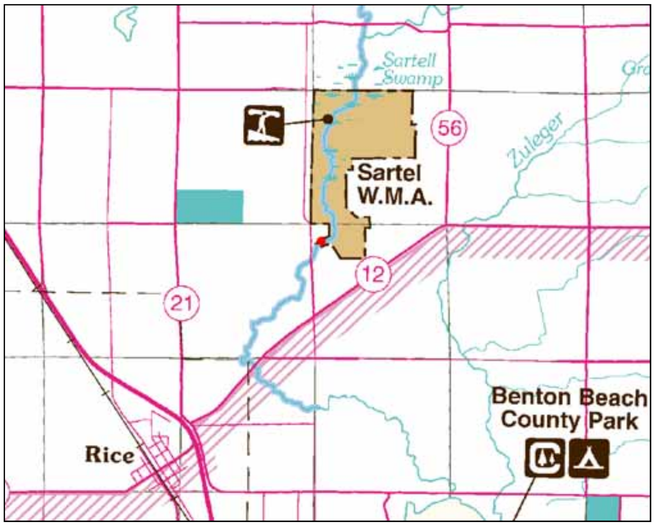


Figure 8. Sartell Wetland Management Area location

Benton SWCD

The SWCD collected water quality data over a period of three consecutive years beginning in 2006 on stations 10, 13, BH2, Z1, S1, and LK1. A total of 46 sample collection events took place, ten collection events in 2006, twenty-one in 2007, and fifteen in 2008. Field data including temperature, pH, transparency, turbidity, specific conductivity, and dissolved oxygen were collected using a YSI Sonde 6920. Water samples were analyzed for total suspended solids, biochemical oxygen demand, ammonia, nitrite + nitrate nitrogen, Kjeldahl nitrogen, orthophoshorus, total phosphorus, fecal coliform, and chloride.

Stream data collection in 2006 began in May and continued through October on a bi-weekly basis. Severe drought in 2006 caused Benton County to be declared a Drought Disaster County. Streams exhibited low base flows, and S1 dried up completely in June. Little Rock Lake data was collected once a month May through October.

In 2007 data collection took place March through November. Samples were collected weekly from March through May, bi-weekly from June through July, and monthly from August through November. Drought persisted through most of the summer months. S1 and BH2 stations dried up in July. Little Rock Lake data was collected once a month from May through October. A severe algae bloom occurred late in the summer on the lake and algal toxin levels exceeded 80,000 ppb.

In 2008, stream water quality data was collected as part of the Little Rock Lake TMDL project. The data was also used for the Little Rock Creek project. Samples were collected on a bi-weekly basis from May 30 through October 29. The same field data was collected at each site as in the two previous years. Water samples were analyzed for alkalinity, ammonia, chloride, color, Kjeldahl nitrogen, nitrite + nitrate nitrogen, orthophoshorus, total phosphorus, turbidity, and total suspended solids.

Little Falls DNR Fisheries

DNR Fisheries conducted fish surveys in mid-July 2006 at stations 6, 7, 10, 13, BH2, Z1, and Z2. The sites were selected in cooperation with NRRI to ensure comparable data. The sites were selected based on access, location of Benton SWCD routine water quality samples sites, or proximate to MPCA defined ‘pour points,’ which allow contributing watersheds to be isolated at a sub-basin level Sampling locations were also considered if an established SWCD and MPCA gauging stations were in operation. DNR Fisheries and NRRI (discussed below) had planned to sample three additional sites within the watershed, but sample collection at those sites was prohibited by drought conditions.

DNR completed fish IBI surveys using MPCA procedures. Fish survey station lengths ranged from 150 to 200 meters. Fish collection gear consisted of either a single backpack electrofishing unit or a two-probe generator electrofishing system mounted in a pull behind barge (“stream shocker”). Gear selection depended primarily on stream size and depth. All fish captured during the survey were identified to the species level and counted. Minimum and maximum individual lengths were recorded for each species.

Natural Resources Research Institute

The Natural Resources Research Institute (NRRI) conducted an invertebrate and habitat survey on stations 6, 7, 9, 13, and Z1 in August 2006 (Figure 7). Low waters conditions in both 2006 and 2007 prevented proposed sampling on Bunker Hill Creek, Sucker Creek, and a second Zuleger Creek location. A combination of sampling gear at available habitats resulted in 8-12 invertebrate samples being processed per site, as well as periphyton rock scrubs and a variety of habitat characterization observations.

Habitat Characteristics

Habitat data for the Little Rock Creek sites were collected both from transects established across the channel perpendicular to flow and from whole-reach observations. A minimum of 10 transects were placed at 10 meter intervals (100 meter minimum reach length) to evaluate substrate characteristics, stream features, bank conditions, and available habitats. A schematic stream reach and diagram noting habitat characteristics, and a cross-section diagram at each transect were completed for all sites.

Transect points - Point estimates were used to evaluate stream features, discharge rates, substrate type and proportional coverage, substrate embeddedness by fine particles, in-stream habitat cover, bank and riparian condition, and riparian corridor extent. Five points evenly spaced along each transect were used to quantify substrate size categories and composition (percent coverage).

Substrate - Within each grid (25 cm^2), the extent (in percent surface area covered) and types of substrate particles were estimated. Classification schemes adhered to standardized particle size categories (e.g., Brusven and Prather 1974, Friedman and Sanders 1978, Gee and Bauder 1986). The extent of embeddedness of large substrate particles by fine particles (sand, silt, and clay) was also estimated (as percent embedded) at one point within each grid. An additional sediment depth measurement along each transect was recorded to determine the maximum depth of fine particle deposition using a sediment rod (Brady et al. 2004). Depositional areas within the stream were targeted, and fine particle accumulation was measured by inserting a sediment rod to a depth of maximum resistance. This measurement was repeated to obtain a maximum sediment depth per transect. Finally, fine sediments were collected using a 7.62 cm diameter core from three locations along the stream reach and returned to the laboratory for particle size analysis.

Flow - Stream discharge was estimated from flow recordings at 15-20 points on a single transect that provided the most laminar flow. Water depth was recorded at each transect point and flow rates were recorded from a point equivalent to 60% of the total water depth. Instructions for flow-weighted averaging (FWA) are provided in the 1990 Marsch-McBirney Flow-mate operators' manual.

In-stream cover - Where transect lines intersected in-stream habitat cover, the type, size, and stability were described. Schematic diagrams of the size, shape, and dimensions of habitat cover, such as large boulders, islands, etc., were also recorded. Large wood pieces (greater than 1 m in length and 10 cm diameter), debris dams, roots wads, etc., that intersected each transect were recorded in detail, noting length or surface area, stability, and position along each transect. Total amount of large wood per reach was also estimated by counting the number of intact units that occurred within transects, and then summing together transect counts. A reach survey qualitative habitat evaluation index to rank overall stream condition was also completed for each site following the sampling event. QHEI categories include substrate, cover, channel type, riparian zone, width/depth ratio, and riffle/run quality; the gradient metric was not calculated or included in the final score due to the lack of GIS elevation data and watershed area estimate for a particular stream reach.

Bank structure - Bank or shoreline structure and condition (stable or unstable) were evaluated on all transects by noting bank substrate type and the presence or absence of undercut banks. Bankfull width was recorded, as well as high water marks or indicators of flood extent.

Riparian corridor – Densiometer readings at a mid-stream point on each transect were used to estimate stream shading. Riparian width and vegetation type within 10 meters were described using predetermined land cover categories. Additional notes were also recorded for adjacent riparian and landuse characteristics from 10-30 meters.

Physicochemical parameters - Water chemistry parameters at each location were recorded with a

YSI 556 multi-probe meter to establish baseline information on water temperature, dissolved oxygen, conductivity, and pH during the sampling effort. Water clarity observations were completed in triplicate using a 120 cm transparency tube.

Minnesota Pollution Control Agency

The MPCA Northern Biological Monitoring Unit conducted biological surveys during July and August, 2007, at stations 2, 5, 7, 9, and 12 (Figure 7). A follow up survey was conducted in 2008 on Little Rock Creek at stations 5, 6, 7, 8, 9, 11, 12, and 13. Stations were typically located near road crossings, within reaches included in the current TMDL study, and selected to correspond with locations of previous fish surveys by MPCA and the MN DNR.

A continuously recording HOBO temperature logger was deployed in each sampling reach in early June to monitor thermal temporal variability; loggers were retrieved and data downloaded in late September. At the time of each fish survey, MPCA crews recorded stream discharge, water temperature, pH, conductivity, and dissolved oxygen concentration, collected samples for laboratory water chemistry analyses, and conducted a quantitative habitat evaluation.

MPCA Fish Survey Methods - Fish survey station lengths ranged from 165 to 262 meters. Fish collection gear consisted of either a single backpack electrofishing unit or a two-probe generator electrofishing system mounted in a canoe (“stream shocker”). Gear selection depended primarily on stream size at a given reach. All fish captured during the survey were identified to the species level and counted; voucher specimens for each species were retained and preserved for taxonomic confirmation. Minimum and maximum individual lengths were recorded for each species, as was the batch weight of all individuals of each species. Species of interest (e.g. trout) were measured and weighed individually. Complete Standard Operating Procedures for MPCA fish sampling can be found on the web (http://www.pca.state.mn.us/water/biomonitoring/bio-streams-fish.html).

The MPCA Watershed Unit collected dissolved oxygen data in August 2008. A diurnal dissolved oxygen survey was conducted on stations 7, 11 and 13. A longitudinal dissolved oxygen survey was also conducted on stations 3 – 7, 9, 10, 12 and 13. Data was collected from the stations starting at the most upstream station in late afternoon/early evening and the sample collection was repeated at sunrise the next day. A longitudinal look at the dissolved oxygen levels was observed on August 20th and 21st. The goal of this monitoring was to gather the dissolved oxygen level at approximately the peaks of the diurnal sine wave, both the highest and the lowest levels, throughout the stream. In the evening of the 20th, dissolved oxygen was collected beginning at site 13 at 4:30pm and finishing at site 4 at 5:35pm. On August 21st, monitoring occurred from 5:10 to 6:30am, with sunrise at 6:24am.

Geochemical samples were collected in 2007 and 2008 for snowmelt, storm-flow and base-flow conditions, along with routine water chemistry sampling. Two rain event samples were captured on June 28th, 2007 and October 8, 2008. Snowmelt samples were taken on April 23rd, 2008 and base flow samples on August 21st, 2008. The geochemistry stations (1, 4, 6, 10, 13, BH1, D1, D2, LK2, and MW) were selected to learn about the hydrologic pathways into Little Rock Creek. Stream, ditch and lake samples were collected with a swing sampler. The mini-well sample was taken approximately 10 feet from the stream using a potential-probe.

The MPCA Watershed Unit conducted a Level 1 and 2 Rosgen geomorphic field survey on Stations 2, 5 – 9, and 11 – 13 in 2008 (Figure 7). Channel bed assessments were obtained at biological stations (6, 7, 8, 11, and 13) to assess fluvial processes and habitat conditions. These measurements were made by inserting a copper rod, with one hand, into the bed surface until resistance was met.

MPCA collected synoptic stream-flow measurements eight times at six locations to evaluate discharge before, during, and after the growing season. DNR Waters collected continuous stage data at station 9 in 2006 and 2008. The data logger failed in 2007 and no data was collected.

The MPCA Watershed Unit from the Brainerd Regional division collected continuous stage data from stations 13, BH2, Z2, S1, and LK2 from 2006 through 2008. SR 50 Ultrasonic stage recorders were installed at all stations except LK2, where an Acoustic Doppler recorder was installed at Station LK2. Paired stage-discharge measurements were collected to develop rating curves and calculate stream flow rates. All data is stored and calculated in Hydstra and can be accessed from the DNR/PCA stream cooperative web page: (http://www.dnr.state.mn.us/waters/csg/index.html).

Data informing about the influence of groundwater on Little Rock Creek was also collected during the investigation. Monthly irrigation pumping information was gathered from DNR Waters. Ground water observation well data in and near Little Rock Creek was obtained from the Minnesota State Climatology Office.

All data collected in Phase 2 was reviewed by the Little Rock Creek TMDL Technical Committee. The Technical Committee led the stressor evaluation process.

PUBLIC PARTICIPATION

Public participation was incorporated into the stressor analysis and identification process in order to receive input from stakeholders and to apprise them of the progress made. Public participation for this process was combined with efforts associated with the development of the Little Rock Lake TMDL.

On September 3rd, 2003 a local stakeholder meeting was organized by the Benton SWCD to inform participants of the stream listing, TMDL process and solicit ideas and information. Subsequent meetings and communications with technical committee members comprised of SWCD, Local Water Managers, MPCA, DNR Division of Waters and Fisheries and MDA staff have revealed a need to gather and organize existing data in a format that can be more effectively interpreted.

The second public meeting was held on September 23, 2007 at the Sauk Rapids Middle School in Sauk Rapids Minnesota to inform the stakeholders about the TMDL process and the stressor identification phase. Project information and related materials were available for public distribution at the meeting. Approximately 30 people attended the meeting.

The third public meeting was held September 16, 2009 at the Sauk Rapids Middle School in Sauk Rapids. The findings of the Stressor Identification Report were presented to the stakeholders. Public comment was received. Approximately 36 people attended the meeting.

CANDIDATE STRESSORS

The Causal Analysis / Diagnosis Decision Information System (CADDIS) was used to systematically review and evaluate all data. CADDIS is an online EPA application that guides the user through the stressor identification process, which is a method for identifying causes of impairments in impaired water bodies (http://cfpub.epa.gov/caddis/). CADDIS was used to evaluate, identify, and rank the stressors causing the biological impairments in Little Rock Creek.

Original list of candidate stressors

The first step in CADDIS is to list and evaluate the candidate causes of impairment. A list of potential stressors was developed by the original Technical committee in 2003. In 2008 the new Technical Committee separately compiled a list of stressors, guided by the CADDIS process. The lists created by the 2003 & 2008 Technical Committees were compiled (Table 2). The candidate causes are grouped by categories of stressors: physical, chemical, biological, and other.

Table 2. Comprehensive list of candidate stressors

Physical	Chemical	Biological	Other
Air temperature	Ammonia	Exotics	Activity on lake
Connectivity/dam	BOD	Fecal coliform	Changing human priorities
Dissolved oxygen	Chloride	Fish IBI's	Human
Elevation	Nitrogen/Nitrates	Food supplies	
Flow regime	Pesticide	growth	
Habitat Variety	pH	Harvest	
In-stream habitat	Phosphorus (fertilizer)	Mortality/Survival	
Lakebed sediment		Predation	
Land use		Reproduction	
Precipitation		Vegetation	
Riparian zone			
Sediment load			
Sinuosity			
Suspended Solids			
Temperature			
Warmwater vs. coldwater environments			
Wetlands/drainage			

Elimination of potential stressors

Preliminary elimination of very unlikely stressors is recommended in the EPA guidance to keep the SI process from being too unwieldy. Upon completion of the Phase 2 data review, the Technical Committee began to eliminate stressors from the original comprehensive list (Table 3). Many potential stressors can be eliminated simply because they or their effects are not present in Little Rock Creek.

Table 3. Stressors eliminated from comprehensive list

Physical	Chemical	Biological	Other
Air temperature Elevation Habitat Variety In stream habitat Lakebed sediment Land use Precipitation Riparian zone Suspended Solids Warm-water vs. cold-water environments Wetlands/drainage	Chloride Pesticide pH Phosphorus (fertilizer)	Exotics Fecal coliform Fish IBI's Food supplies growth Harvest Mortality/Survival Reproduction Vegetation	Activity on lake Changing human priorities Human

Physical Stressors

Elevation, habitat variety, in stream habitat, land use, riparian zone, warm-water vs. cold-water environments, lakebed sediment, and wetlands/drainage were eliminated because they were not deemed to be primary causes of impairment based on the group's professional judgment. The remaining stressors in the physical category had sufficient evidence to be removed. Air temperature is apt to be a relatively weak determinant of stream temperature (Johnson, Sherri L. 2004). Reduced water inputs due to insufficient rainfall were eliminated because precipitation amounts have increased in Minnesota in the past 80 years (discussed in Altered Flow section). Total suspended solids data from 2006-2008 are below North Central Hardwood Forest (NCHF) ecoregion 75% percentile value of 16 mg/L (McCollor & Heiskary) (Figure 9).

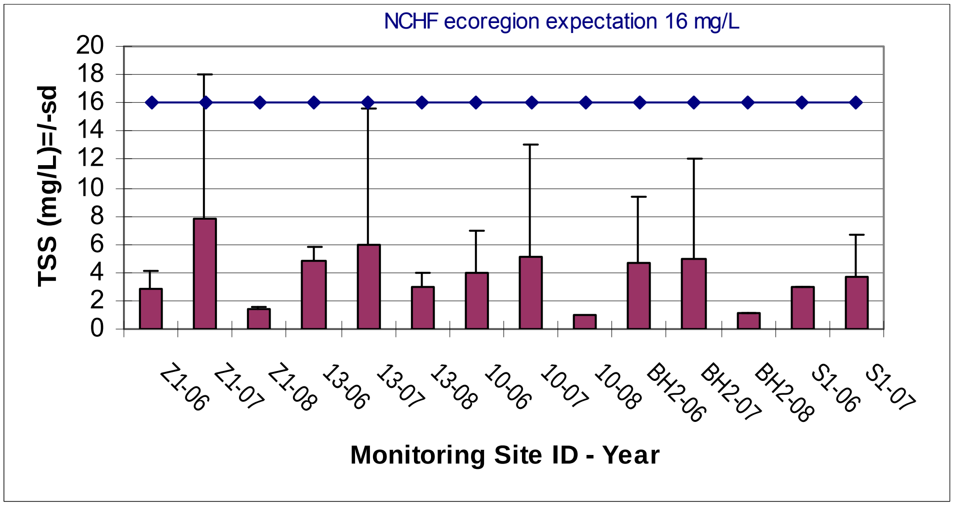


Figure 9. Mean total suspended solids in LRC 2006-2008

Chemical Stressors

Chloride levels were well below the 230 mg/L water quality standard for Class 2A streams (Figure 10). The pH levels were within the 25% and 75% percentile values for reference streams in the NCHF ecoregion (6.5 – 8.5 pH) (Figure 11).

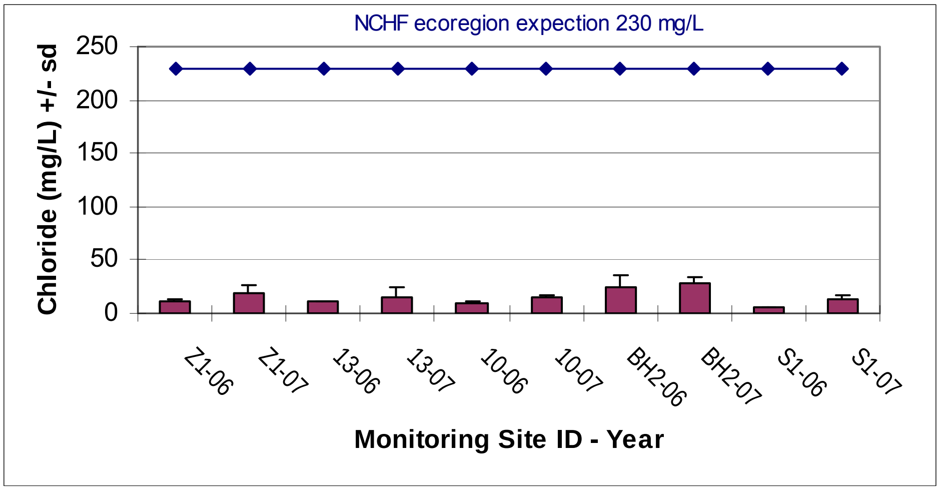


Figure 10. Mean chloride in LRC 2006-2007

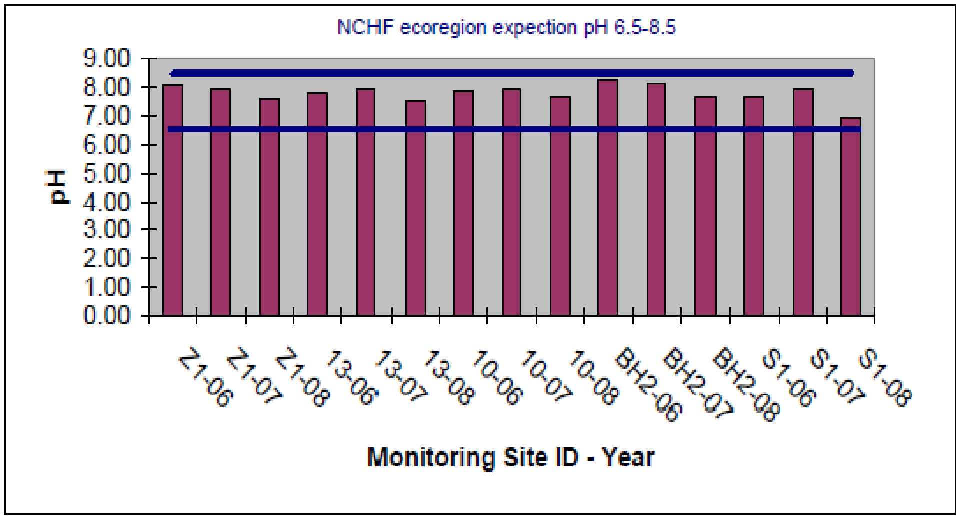


Figure 11. Mean pH in LRC 2006-2008

At stations 1, 2, 4, 10, 13, MW, D1, BH1, BH2, S1, and Z1, phosphorus levels exceeded the 75th percentile value (0.15 mg/L) for reference streams in the NCHF ecoregion (Figure 12). The highest level of phosphorus collected in Little Rock Creek was at site 13 with 1.42 mg/L. The highest level in the watershed was at BH2 with 2.09 mg/L of phosphorus.

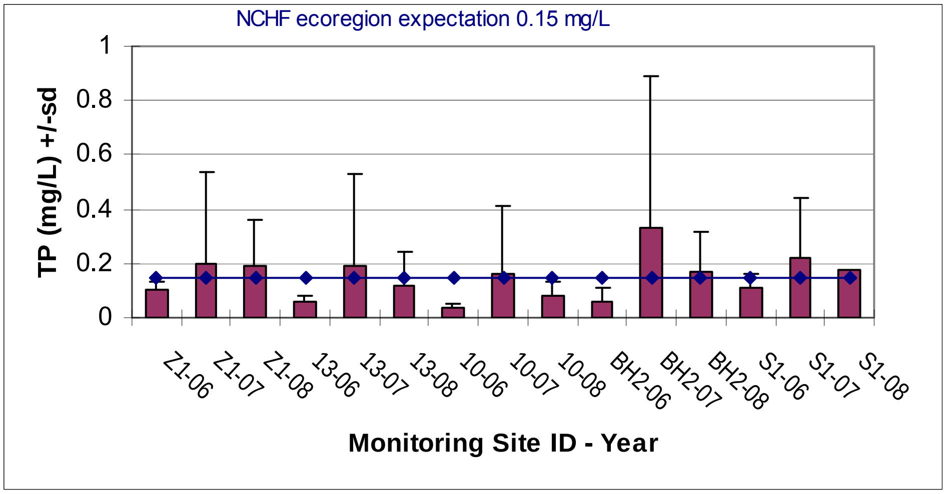


Figure 12. Mean total phosphorus in LRC 2006-2008

In a stream system, the phosphorus cycle tends to move phosphorus downstream as the current carries decomposing plant and animal tissue and dissolved phosphorus. It becomes stationary only when it is taken up by plants or is bound to particles that settle to the bottom of pools.

Excessive phosphorus concentrations in surface water can cause nuisance algae and plant growth, which in turn can adversely affect aquatic ecosystems and water-related recreational activities. Although the high total phosphorous levels in Little Rock Creek potentially indicate excess nutrient inputs to the stream, field surveys found no evidence of excessive algae growth. If some large algae blooms were observed along with dramatic swings in the diel dissolved oxygen from morning to evening, then total phosphorus may be a higher priority stressor. The total phosphorus concentrations will be addressed within the dissolved oxygen candidate stressor section.

Pesticides

The Minnesota Department of Agriculture (MDA) began sampling Little Rock Creek in 2004 as part of their Surface Water Monitoring Program. In 2006, the MDA began monitoring surface waters in Minnesota utilizing a tiered structure. Within the tiered structure, there are three different levels of monitoring intensity. Tier 1 site locations are sampled four times during an eight week period from mid-May through mid-July. The objective is to provide a general assessment of water quality during peak pesticide detection periods from watersheds throughout the state. At Tier 2 and 3 site locations, the frequency of sampling increases to provide better information for a duration assessment or the length of time pesticide concentrations remain at a particular level. In an expanded Tier 3 evaluation the monitoring focus shifts from concentration and duration to a greater emphasis on the concentration-flow relationship to provide for better estimates of contaminant loading and pesticide transport dynamics.

One Little Rock Creek site was selected for this study because levels of atrazine exceeded standards for more than 50% of the samples in 2005. Little Rock Creek is considered a Tier 2 site based upon pesticide results for the previous three years (2003, 2004, 2005). Tier 2 sites were selected based upon pesticide sampling within the last five years that indicated the potential for concentrations at levels of concern (i.e. above 50% of the appropriate reference values). None of the samples collected exhibited concentration in excess of reference values during the time period of 2006-2007, so no time-weighting was necessary for this evaluation.

Biological stressors

Exotic species were eliminated from the list because they are not considered to be a likely candidate stressor for aquatic life impairments in small streams (CADDIS). Field investigations in Little Rock Creek documented no problematic exotic species. Carp are present in the impoundment and Little Rock Lake, but have not been found in the biological monitoring sites.

Fecal coliform values did exceed ecoregion standards in all stations with the exception of S1, but these bacteria are not considered toxic to fish (Figure 13). The elevated levels are however a human health concern, which will be addressed as a separate aquatic recreation and human consumption use impairment.

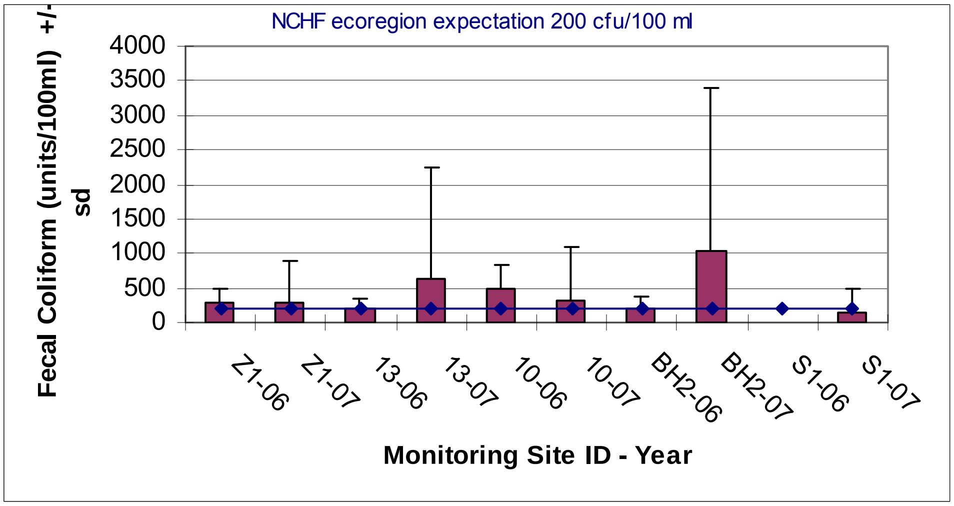


Figure 13. Fecal coliform in LRC 2006-2007

Growth, mortality/survival, reproduction are not direct stressors to the biological community, instead they are results of stressors. There is no evidence that the remaining stressors: food supply (lack of), harvest (over harvest), and vegetation (lack of riparian habitat) are significant enough to cause the impairment in Little Rock Creek.

Other

It is unlikely that activity on the lake is causing a direct stressor on the biota. The migration of warm-water piscivores from the lake up into the creek is discussed later in this report. Humans and changing human priorities may be causing one of the stressors in Little Rock Creek.

Summary: Identification of stressors

Seven main stressors were identified for Little Rock Creek, all of which were confirmed by the sufficiency of evidence analysis. They are listed below in order of priority:

- **Flow alteration**
- **Temperature**
- **Sediment – deposited and bedded**
- **Nitrates (toxin)/Ammonia**
- **Dissolved oxygen/BOD**
- **Connectivity**
- **Predation of trout by pike and other warmwater piscivores**

Stressor evaluations are treated individually in the causal pathway sections to follow, where supporting data and background research are discussed in detail.

CAUSAL PATHWAY MODELING USING CADDIS

The six identified stressors are addressed individually in the following sections using the CADDIS stressor identification process, including detailed analysis of evidence and models of causal pathways. The strength of evidence is assessed for each causal pathway for each stressor. Conceptual models represent relevant ecosystem processes, causal pathways, and candidate sources. All inferences are documented with references to specific data analysis and background literature.

Types of Case-Specific Evidence

The CADDIS Stressor Identification (SI) process is designed to work with many types of evidence. The more types of evidence that support a candidate cause for impairment, the stronger the case for it being the true cause of impairment. The types of evidence presented below are some of the more commonly used forms in ecological and water resource management studies. The evidence generated by analyzing associations among data from the case will likely fall into one of the types listed in below.

Spatial/Temporal Co-occurrence

The biological effect is observed where and when the causal agent is observed and is not observed in the absence of the agent.

Evidence of Exposure or Biological Mechanism

Measurements of the biota show that relevant exposure has occurred or that other biological processes linking the causal agent with the effect have occurred.

Causal Pathway

Precursors of a causal agent (components of the causal pathway) provide supplementary or surrogate evidence that the biological effect and causal agent are likely to have co-occurred.

Stressor-Response Relationships from the Field
The intensity or frequency of biological effects at the site increases with increasing levels of exposure to the causal agent or decrease with decreasing levels.

Manipulation of Exposure
Field experiments or management actions that decrease or increase exposure to a causal agent decrease or increase the biological effect.

Laboratory Tests of Site Media
Laboratory tests of site media can provide evidence of toxicity, and Toxicity Identification Evaluation (TIE) methods can provide evidence of specific toxic chemicals, chemical classes, or non-chemical agents.

Temporal Sequence
The cause must precede the biological effect.

Verified Prediction
Knowledge of the causal agent’s mode of action permits prediction of unobserved effects that can be subsequently confirmed.

Symptoms
Biological measurements (often at lower levels of biological organization than the effect) can be characteristic of one or a few specific causal agents. A set of symptoms may be diagnostic of a particular cause if they are unique to that cause.

Types of Evidence that Use Data from Elsewhere

The types of evidence that use data from elsewhere are listed below. Each piece of evidence should be associated with only one type of evidence, to avoid double counting.

Stressor-Response Relationships from Other Field Studies
The causal agent in the case is at levels that are associated with similar biological effects in other field studies.

Stressor-Response Relationships from Laboratory Studies
The causal agent in the case is at levels that are associated with related effects in laboratory studies. The laboratory studies may test chemicals, materials, or contaminated media from sites contaminated by the same chemical, mixture or other agent as the case. If the effects or conditions in the laboratory and field are dissimilar, extrapolation models may improve the correspondence.

Stressor-Response Relationships from Ecological Simulation Models
The causal agent in the case is at levels that are associated with similar effects in mathematical models that simulate ecological processes.

Mechanistically Plausible Cause
The relationship between the causal agent and biological effect is consistent with known principles of biology, chemistry and physics and with properties of the affected organisms and the receiving environment.

Manipulation of Exposure at Other Sites
At similarly affected sites, field experiments or management actions that alter exposure to a causal agent also alter the biological effects.

Analogous Stressors
Evidence that agents that are similar to the candidate causal agent in the case cause effects similar to the effect observed in the case is supportive of that candidate causal agent as the cause.

Multiple lines of evidence
When evidence supports more than one candidate cause, there are potentially multiple causes. Although this issue should have been addressed when defining the case and listing the candidate causes, it should be reconsidered here if the results are unclear. New evidence or new understanding may reveal relationships among agents that were not apparent in the beginning.

Strength of evidence (SOE) analysis
The strength of analysis step organizes all the information relevant to each candidate cause so that it can be easily compared and analyzed in a logical manner. This analysis provides a consistent and systematic approach for evaluated whether the evidence supports, or doesn't support each candidate cause. Furthermore, for multiple stressors, it identifies the stressor(s) that are best supported by the evidence with a level of confidence assigned to each stressor by the investigators according to the lines of logic. The strength of evidence analysis for Little Rock Creek was completed for the six candidate causes and twelve lines of logic.

After the data is organized and evaluated for the various types of evidence, the strength and consistency of the evidence for candidate cause should be scored for comparative purposes. The scores for the various types of evidence will become particularly valuable in the next step of the CADDIS process, when the most probable cause for impairment will be identified. The scoring system for all the evidence types used in the stressor identification is listed in Table 4. The sign of the score is based on whether the type of evidence supports the candidate cause (+), weakens the candidate cause (-) or has no impact (0).

The number of pluses and minuses increases with the degree to which the evidence either supports or weakens the case for a candidate cause. Evidence can score up to three pluses (+++) or three minuses (---) (Table 5). However, the maximum number recommended for a particular type of evidence depends on the likelihood that an associate might be observed because of chance rather than because of the true cause. Therefore, the highest scores are given to the types of evidence:

- That use data from the case
- That are based on more than one association
- That closely link the proximate cause and the effect

If the available data cannot be analyzed in a way that can be used to evaluate a type of evidence, it is scored as “no evidence” (NE). If other candidate causes in the analysis do have this type of evidence, CADDIS recommended including the NE to help compare the relative strength of evidence across candidate causes.

Table 4. Possible values in the CADDIS stressor identification system

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, -

Table 5. Key to the values assigned evidence in the CADDIS stressor identification system

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*	

The following is the Strength of Evidence analysis for the seven stressors.

FLOW ALTERATION

Movement of water through stream and river channels influences all processes and biota within. Flow characteristics vary throughout a watershed, longitudinally along a stream channel, laterally from channel to floodplain, and longitudinally within groundwater, as a function of landscape features. The variability of flow, both regionally and temporally, results from variance in rainfall patterns, vegetation, development, geology and other watershed characteristics. Biological characteristics at a given site relate to volume, velocity, and variance of flow (EPA CADDIS).

Flow alteration refers to modification of flow characteristics, relative to reference or natural conditions. For the purposes of this case study, altered flow regime refers to a decrease in base discharge (or baseflow). Decreases in baseflow may result in decreased aquatic habitat availability in terms of wetted channel width or depth and decreased habitat quality in terms of flow heterogeneity.

Flow Standard

The minimum stream flow, according to Minnesota State Statute 7050.0210 Subpart 7, for point and nonpoint sources of water pollution shall be controlled so that the water quality standards will be maintained at all stream flows that are equal to or greater than the 7Q10 (the lowest or highest streamflow for 7 consecutive days that occurs on average once every 10 years) for the critical month or months, unless another flow condition is specifically stated as applicable in this chapter.

Effects of altered flow in streams

Zorn et al (2008) conducted a regional scale habitat suitability model to assess the effects of flow reduction on fish assemblages in Michigan streams. Michigan streams are low gradient, sand and gravel, hydrologically-stable environments with temperature governed principally by groundwater inputs that can be quantified as base flow yields. Habitat conditions during summer base flows limit fish distributions as water temperatures peak during this time period and have a dominant effect on fish physiology, growth, and survival. Reductions in base flow may also significantly alter other habitat variables (e.g., dissolved oxygen or flow velocity) and critical ecosystem functions (e.g., sediment transport or channel maintenance). Base flow yield not only provides a measure of the amount of groundwater in a river’s drought flow, it also indexes many key aspects of river habitat, such as summer and winter water temperatures, hydrologic flashiness, current velocity, substrate conditions (i.e., presence of fines), dissolved oxygen conditions, depth, and permanence of instream habitats.

The Michigan models found that base flow yield and temperature both had powerful effects on species’ distributions and abundances, but base flow yield had the dominant influence on declines in species scores. The models indicated that 20 to 50% of the summer base flow could be removed without serious adverse impact to fish assemblages in all river types except “Cold Transitional.”. Fish communities in cold-transitional streams and rivers had the steepest responses to base flow reduction, indicating the need for caution in setting limits on withdrawal.

Zorn's model predictions of species response to base flow reduction were consistent with previous studies and simulations in Michigan and elsewhere of fish species responses to flow reduction and temperature increase. The Michigan model predicted changes in the density of salmonids and other coldwater fishes with base flow reduction in cold small streams and rivers. These changes were similar to modeling results in other studies of various salmonid species and life stages (e.g., Jager et al. 1997; Van Winkle et al. 1997). Declines in these species are attributable to changes in both flow and temperature. For some species, the reduction in density with base flow is more likely a response to temperature changes, especially as temperature nears species' tolerance limits, while for others it may be more a response to flow changes. The Nuhfer & Baker (2004) study on Hunt Creek in Michigan found that water withdrawals increased the warming rate in the study reach up to 28 times or more (9 times on average), which would likely have negatively impacted growth or survival of salmonids in downstream reaches. The Zorn et al. empirical model predicted that base flow reductions would cause declines in abundances of cool and warm water species including smallmouth bass, rainbow darter, blackside darter, burbot, common shiner, logperch, northern hogsucker, northern pike, and pumpkinseed in Michigan streams.

Trout habitat has a strong relationship with the annual flow regime and is highly dependent on the baseflow period (Raleigh, 1982). Binns and Eiserman (1979) identified late summer stream flow, annual stream flow variation, and maximum summer stream temperatures as primary limiting factors to trout density. Brook trout spawning occurs in areas of groundwater upwelling (Curry and Noakes, 1995). If there is less groundwater upwelling in Little Rock Creek, it may be one reason for the biotic impairment. Brook trout may also be affected by decreased water velocity, as juvenile and adult salmonids require certain velocities for optimal foraging and growth (Baker and Coon, 1997). Additionally, lack of flow and increased sediment aggradation also add to the depth of the impairment. Stream discharge does have a significant influence on peak water temperatures during low flow periods in the summer months; high water temperatures may be reduced with an increased in-stream flow (Sinokrot and Gulliver, 2000).

Data evaluation of altered flow in Little Rock Creek

One hypothesis for impaired biota in Little Rock Creek is that increased pumping of groundwater near the creek has intercepted water that would otherwise have discharged into the creek. Less water in the creek raises temperatures and reduces habitat. A first step toward testing this and related hypotheses is to analyze all relevant hydrologic datasets for patterns or trends that would explain the change in the condition of the creek. Where applicable, the trends are submitted to statistical tests of significance. The datasets are used to build a groundwater model, which is a simulation of groundwater flow. The patterns and trends discovered in the supporting datasets are used to make the model more responsive to the conditions found in the field. The groundwater model can then be used to answer questions about the interaction between groundwater and surface water, including the impact of pumping at particular locations, and how this pumping may affect flow volumes in the creek.

Context

The study area (Figure 3) encompasses six subwatersheds, lying between topographic highlands to the east and the Mississippi River to the west. The aquifer is a mixture of sand and clay, and behaves as a single aquifer system. The groundwater flow direction is from the east to the Mississippi River.

In order to evaluate the relationship between the hydrologic datasets it is useful to think of the study area as a simplified system that balances water inputs against outputs. The first input, which adds water to the system, is groundwater flow from beyond the northeast boundaries of the model. Precipitation falling as rain and snow across the study area is another input. Outputs, which take water from the local system, include evapotranspiration, groundwater discharge to the Mississippi River, and groundwater pumped by large volume users to irrigate crops.

The four datasets reviewed here are instrumental to the building of a groundwater conceptual model which describes the connection between creek flow and the many other facets of the hydrogeologic system. They are precipitation, groundwater pumping, observation well measurements and stream flow volumes.

Statistical tests

The stream flow data was analyzed with the t-test, while the others were analyzed for evidence of trends using a combination of the Mann-Kendall and the Sen's Method statistical tests. While all the data are useful, datasets that show statistically significant trends are of particular value in exploring current and historical relationships between different elements of the hydrologic cycle. Trends with statistical significance are identified with their respective 'p' values. A 'p' value of <0.05 indicates that there is a better than 95% chance that the positive trend result is not due to random chance, and this threshold is commonly accepted as a minimum level of significance. A 'p' value smaller than 0.05 is an indication of stronger support for a claim of statistical significance.

Precipitation

Three National Weather Service (NWS) stations are located within 20 miles of the study area: St. Johns University in Collegeville, Pierz and Royalton. All three gauging stations yield positively trending regression lines, though none of the trends are statistically significant. (Figure 14).

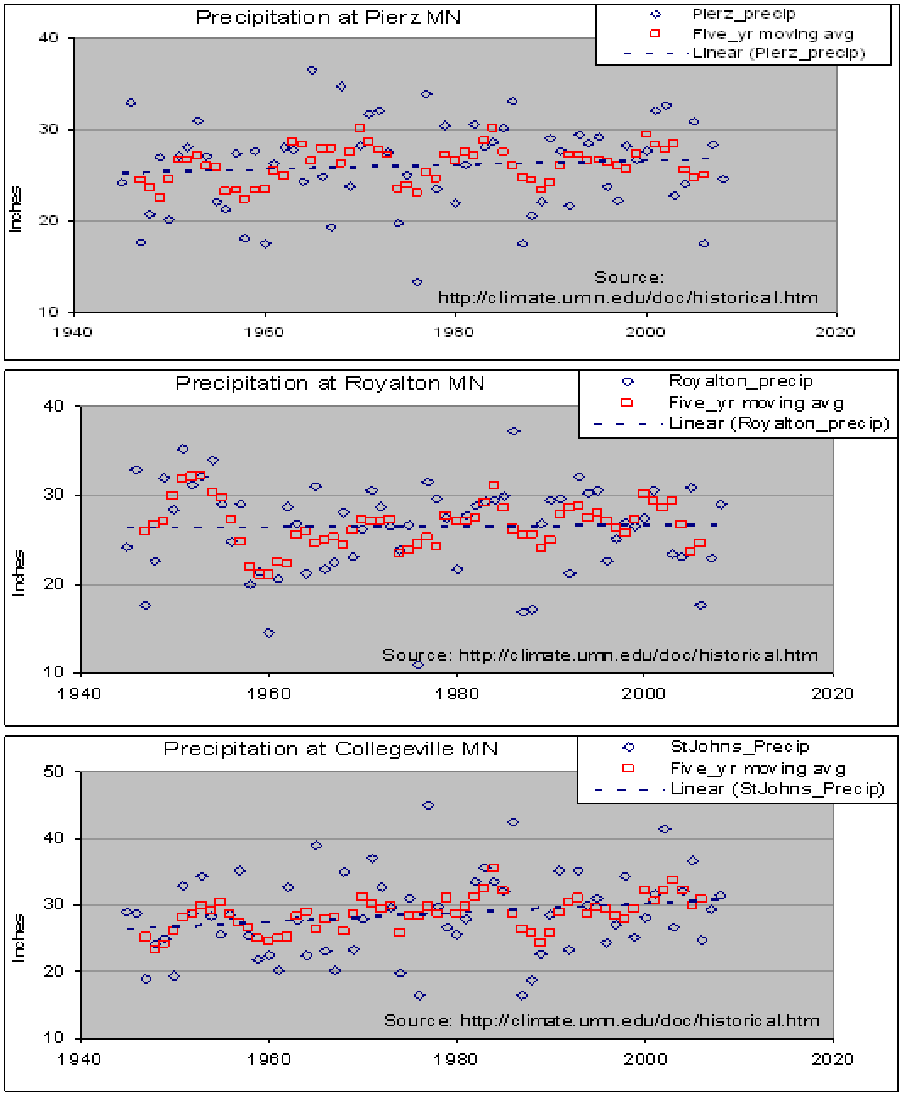


Figure 14. Pierz, Royalton, Collegeville precipitation summary

Because precipitation generally varies greatly over short distances a more appropriate way to analyze for patterns than individual stations is to use state-wide areal averages, combining the readings from hundreds of monitoring stations to produce a spatially robust result. The average areal precipitation trend for Minnesota (Figure 15) is strongly positive for the last 80 years ($p < 0.001$). This positive precipitation trend has been identified in all neighboring states and in northwest Ontario (source- Mark Seeley of the University of Minnesota State Climatology office).

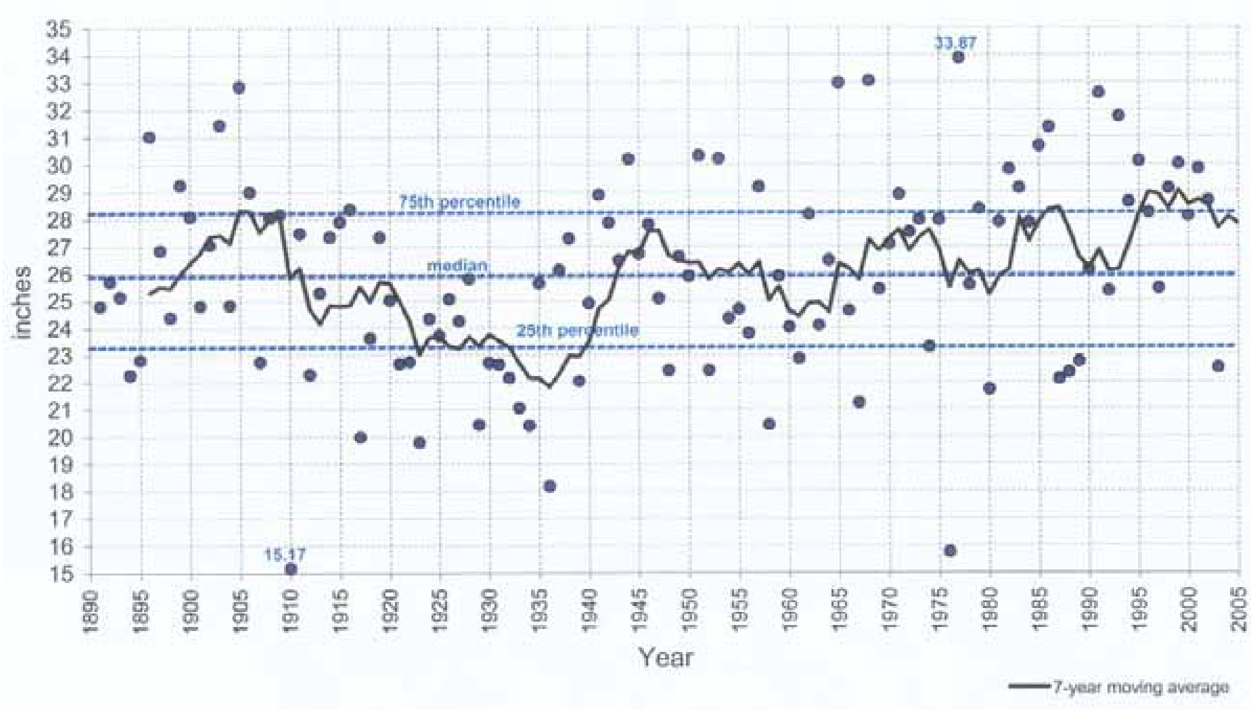


Figure 15. Minnesota areal average annual precipitation summary (MN State Climatology office)

Another important point to consider is that rainfall does not immediately affect ground water levels. Most rain that falls is lost to runoff (especially during heavy downpours) or to evaporation. Even in Benton County's sandy soil, water can take years to move through the unsaturated zone to reach the water table. Ground water levels therefore respond slowly to individual year's contributions, though they can eventually reveal broader precipitation trends if other variables hold steady, which as this report will demonstrate is not the case here.

Groundwater Pumping

Historically, agricultural producers in Minnesota have used groundwater for irrigation since the late 1800's. But it wasn't until 1976 that the Department of Natural Resources began appropriating large volume groundwater pumping through permits. The DNR regulates the use of groundwater in amounts over 10,000 gallons a day, or 1 million gallons a year (Minn. Stat. § 103G.271). Minnesota doesn't differentiate "high-capacity wells" from other types, and there is a complex system of fees per use (Minn. Stat. §§ 103G.271, subd. 6; 103G.301, subds. 2-3).

Most of the DNR pumping permits issued in the Little Rock Creek watershed are for crop irrigation. There were 52 active permits in the Little Rock Creek watershed in 2006, compared to 38 active permits in 2000 and 25 active permits in 1990 (Figure 16). In addition to the increase in the number of permits issued, the quantity of water pumped has increased.

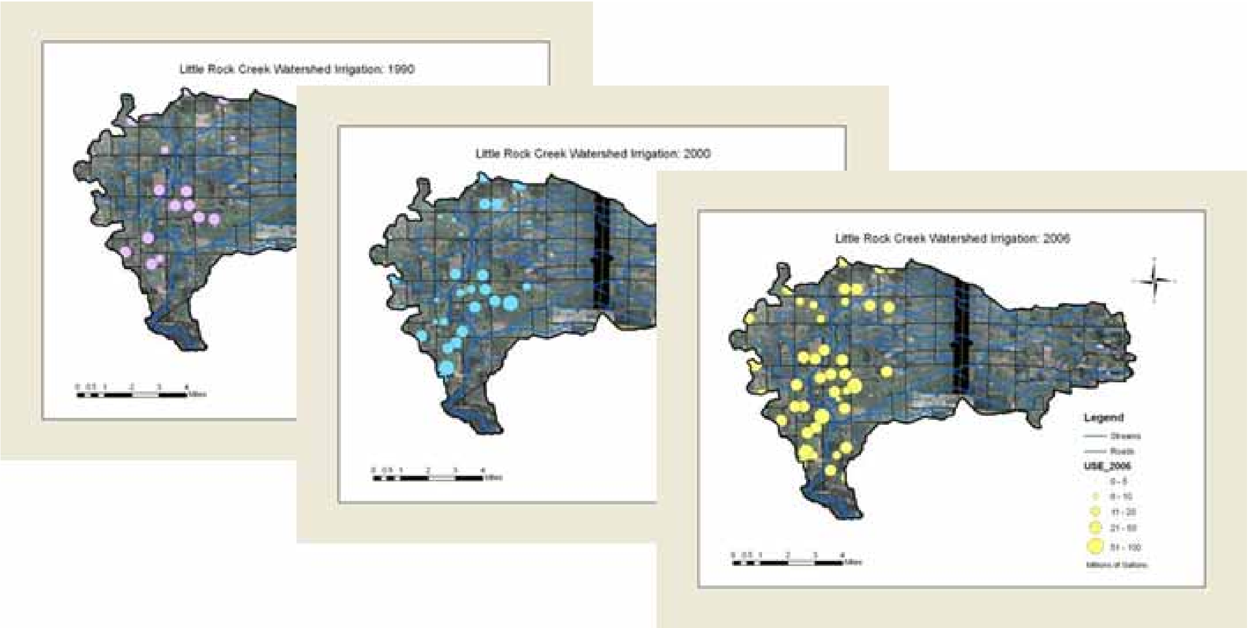


Figure 16. Active pumping permit locations 1990, 2000, 2006

According to data compiled by the MN DNR, permitted pumping increased slightly by 7.3% between 1988 and 1999 in the Little Rock Creek watershed. Pumping rates from 2000 to 2007 increased by 162.31% (Figure 17).

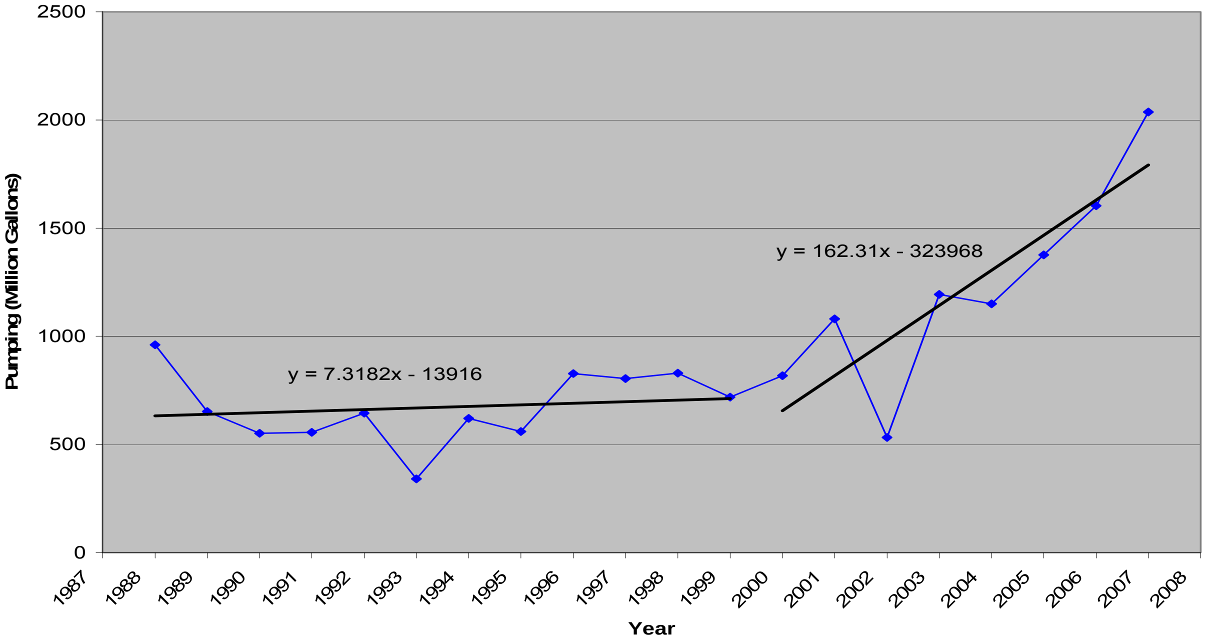


Figure 17. DNR Permitted Pumping Records Summary

Because surface water and groundwater are connected, it is important to recognize the cumulative effects of groundwater pumping. Streams either gain water from inflow of ground water or lose water by outflow to ground water. A pumping well can change the quantity and direction of flow between an aquifer and stream in response to different rates of pumping. The adjustments to pumping of a hydrologic system may take place over many years, depending upon the physical characteristics of the aquifer, degree of hydraulic connection between the stream and aquifer, and locations and pumping history of wells. Reductions of streamflow as a result of ground-water pumping are likely to be of greatest concern during periods of low flow, particularly when the reliability of surface-water supplies is threatened during droughts.

Analyses

The first analysis on the pumping data was done by selecting all DNR permitted (high capacity) wells located within one kilometer of Little Rock Creek (Figures 18 and 19). Total pumping from 1985 through 2008 shows a positive trend with a statistical significance of p<0.001. The second analysis used those wells found within 1 kilometer of the creek that were also exclusively upgradient of the regional groundwater flow (Figures 18 and 19). The trend is also positive, at p<0.001. The results show statistically significant increasing trends in the amount of water pumped within the immediate vicinity of the creek.

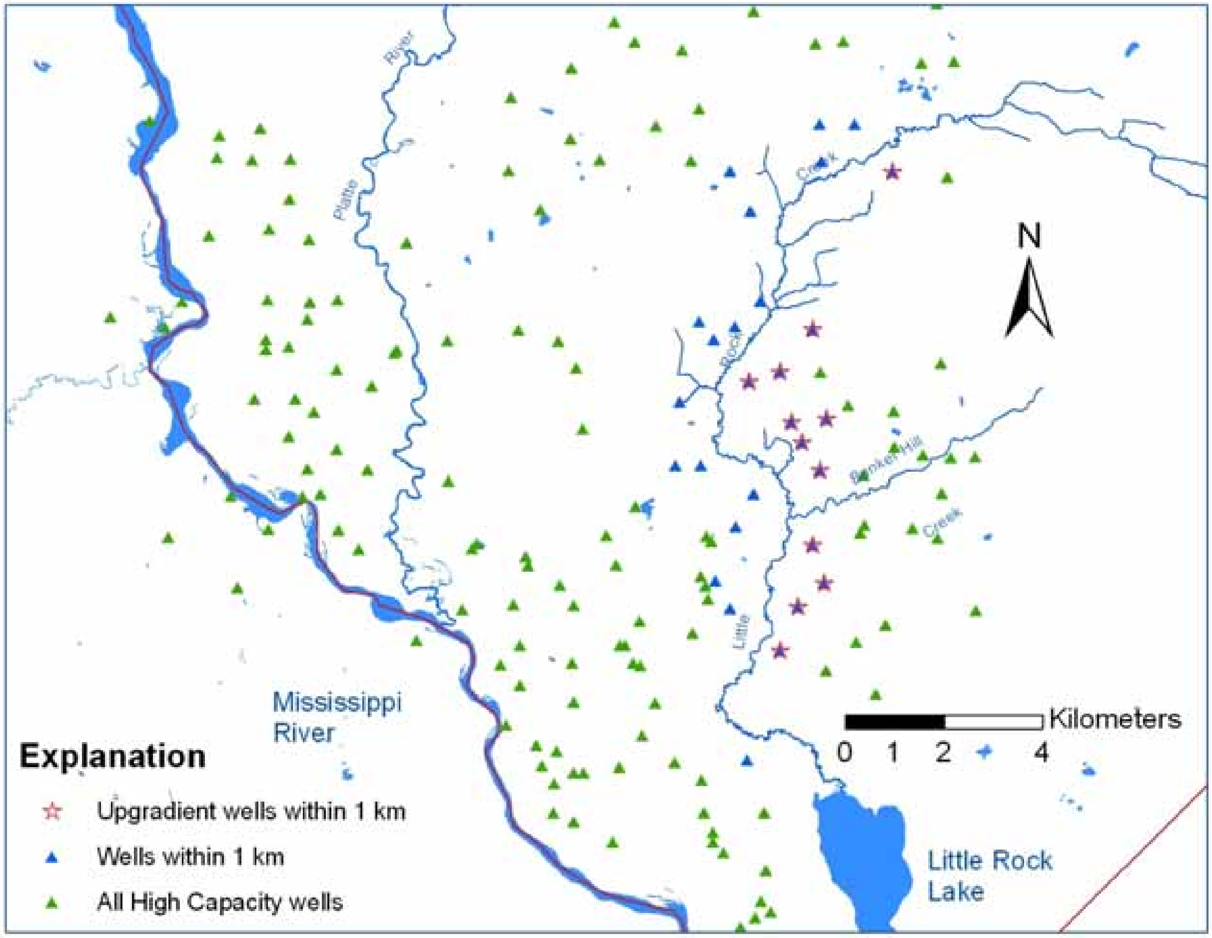


Figure 18. Location of irrigation wells by dataset

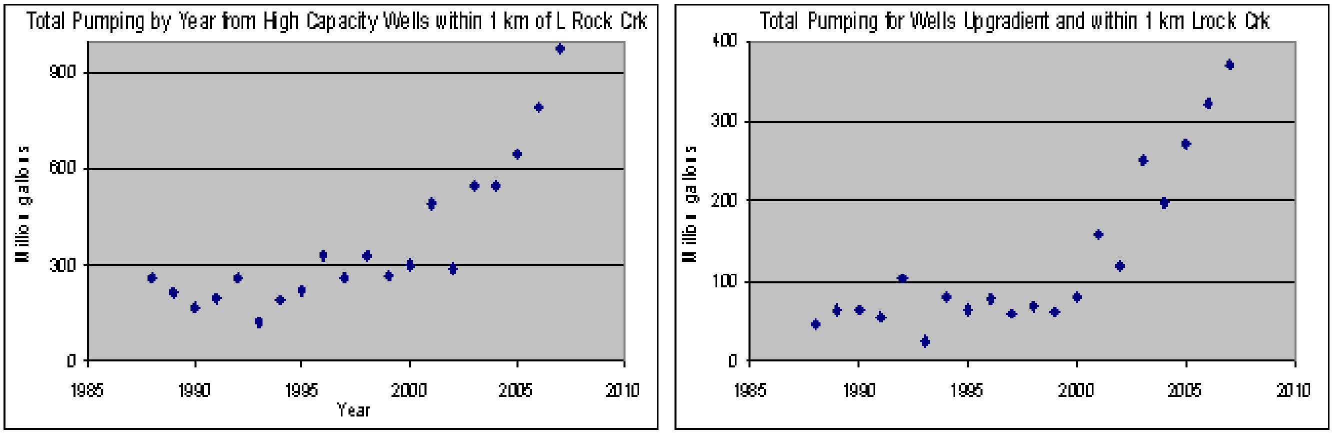


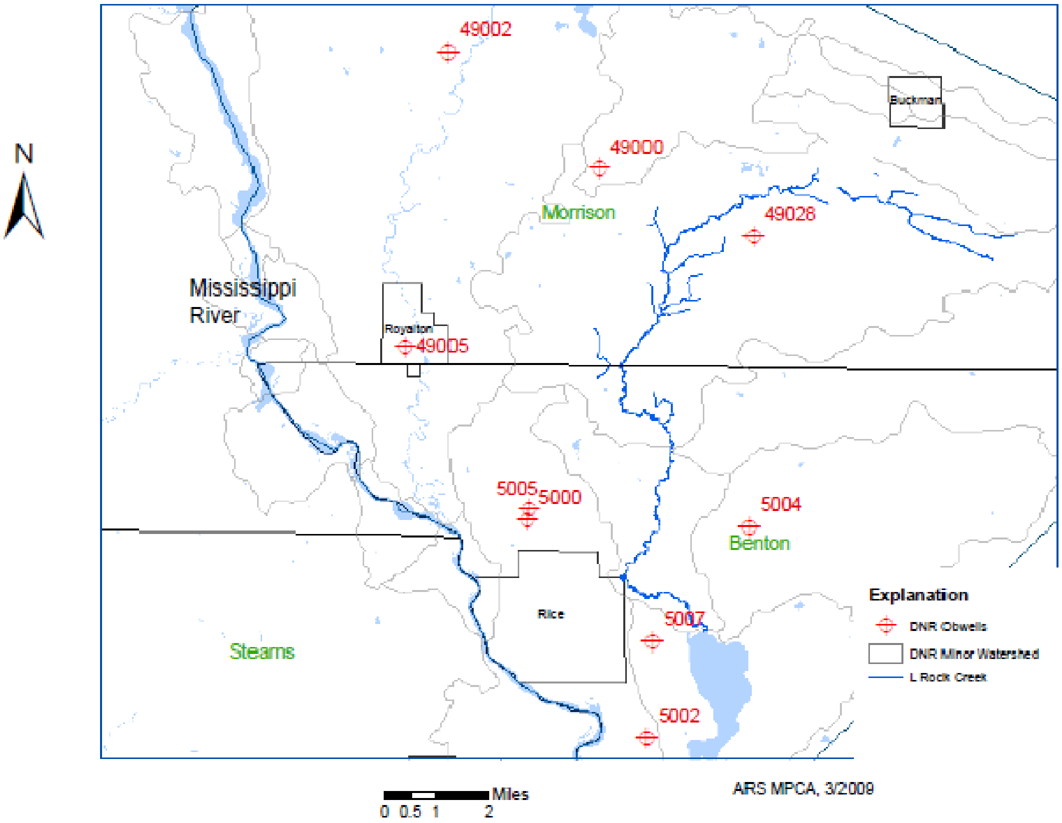
Figure 19. Changes in groundwater pumping

<u>Observation Wells</u>

Since 1944, DNR Waters has managed a statewide network of water level observation wells (obwells). Data from these wells are used to assess ground water resources, determine long term trends, interpret impacts of pumping and climate, plan for water conservation, evaluate water conflicts, and otherwise manage the water resource. Soil and Water Conservation Districts under contract with DNR Waters measure the wells monthly and report the readings to DNR Waters. Readings are also obtained from volunteers at several locations (MN DNR).

DNR obwell hydrographs display groundwater surface elevation changes with time. We have already established that at the same time that the supply of water to the system from precipitation has been increasing, the volume of ground water removed from the system through irrigation pumping has also increased. Measurement of ground water levels can provide information that can be used to understand how these two important parameters interact.

There are nine active obwells in the project area (Figure 20). All of the wells were installed in a water table aquifer, except well 49000 and 5007, which were installed in buried artesian aquifers. The groundwater model is based on the assumption that there is only one aquifer, and that it is unconfined. Three well locations were not used for evidence in this case. Well 49015 only has a five year record (1968-1973). Well 5007 and 5003 are located very close to Little Rock Lake, and because the lake is hydraulically connected with the Mississippi river, the well water elevations are ultimately controlled by the levels on the Mississippi.



ARS MPCA, 3/2009

Figure 20. Observation Well Locations

Analyses

The first analysis involved looking for changes in observation well static water levels over the entire well period of record, typically about 30 years. For this interval all the obwells located north of the Little Rock Lake had negative, or declining, groundwater surface elevations. Two wells had statistically significant declines: Obwell 49000 with significance of $p<0.1$, and Obwell 49002 with a significant trend of $p<0.001$.

The second analysis looked at the change in static water levels for the time period of 2002-2008, when most obwells show a dramatic decline in groundwater elevations. This corresponds well with the previously discussed increase in pumping, with the addition of what appears to be a two year time lag in the groundwater surface decline. This delayed decline supports the thesis that increased pumping is responsible for the decline in observed heads. All but one obwell display a statistically significant decreasing trend. Obwells 49000, 49005, 49028, 5000, 5005 had trends of $p<0.05$. Obwell 5004 had a trend of $p<0.1$. Obwell 49002 had no significant trend.

For the full period of record, only two Morrison County observed wells showed a statistically significant decreasing trend. For the period covering the last eight years, all but one of the wells in both counties displayed a statistically significant decreasing trend. And the wells located in the vicinity of the Mississippi River-dominated Little Rock Lake showed little variation in groundwater levels throughout their records.

Stream Discharge

Stream flow hydrographs quantify stream volume discharge changes. Stream discharge was measured at sites 6, 7, 9, 11 and 13 on Little Rock Creek in 2008 (Figure 21). The change in discharge was examined between pumping and non-pumping seasons. Stream discharge for each sampling period was compared against all other sampling periods with the t-test (or the Mann-Whitney Rank Sum where the datasets are non-normal) to identify statistical differences in datasets (Table 6).

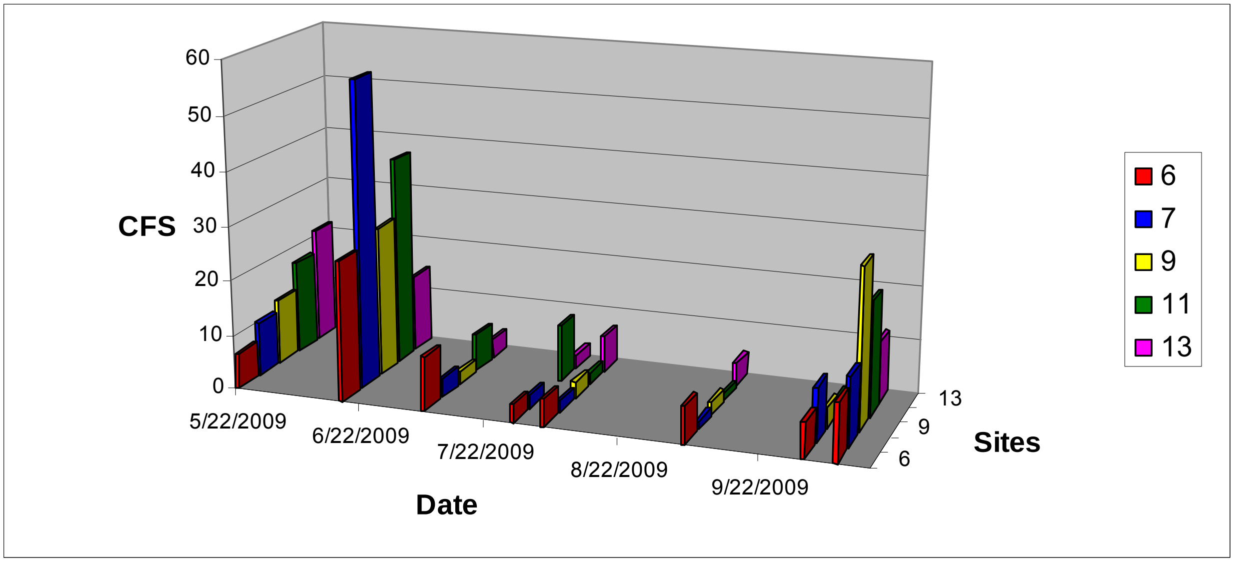


Figure 21. 2009 Stream discharge sites 6, 7, 9, 11, 13

Table 6. Calculated p values for t-tests between datasets

	May 22	June 17	July 7	July 28	Aug 4	Sept 5	Oct 1	Oct 8
May 22		**<0.05**	**<0.05**	**<0.05**	**<0.01**	**<0.01**	**<0.05**	>0.1
June 17			**<0.01**	**<0.05**	**<0.01**	**<0.01**	**<0.01**	**<0.1**
July 7				>0.1	>0.1	>0.1	>0.1	**<0.05**
July 28					>0.1	>0.1	>0.1	**<0.05**
Aug 4						>0.1	>0.1	**<0.01**
Sept 5							>0.1	**<0.01**
Oct 1								**<0.05**
Oct 8								

The stream discharge data collected on May 22, 2008 at five locations on the stream was compared against the discharge data for the remaining seven monitoring events. At the ά=0.05 level, May's discharge was statistically different and higher from the data taken from all datasets that follow with the exception of October 8, 2008 (Table 6). All the datasets collected from July through Oct 1 are statistically similar (ά=0.10). All the data from the first and last collection period are statistically similar to each other, and statistically different from the datasets collected during the summer periods.

The statistical tests show that the creek discharges measured in the summer pumping season are different from non-pumping season values. The cause is most likely a combination of increased evapotranspiration and the diversion of groundwater from the creek to irrigation during the growing season. A more detailed accounting of the loss of water to the creek will be performed by the completed groundwater model.

Models of sources and causal pathway for altered flow in Little Rock Creek
The sources and pathways involved with this stressor are shown in Figure 22. All of the sources in the model stem from human activities, which can significantly alter flow, and may lead to biological impairment. Each pathway was evaluated using CADDIS, shown in Table 7.

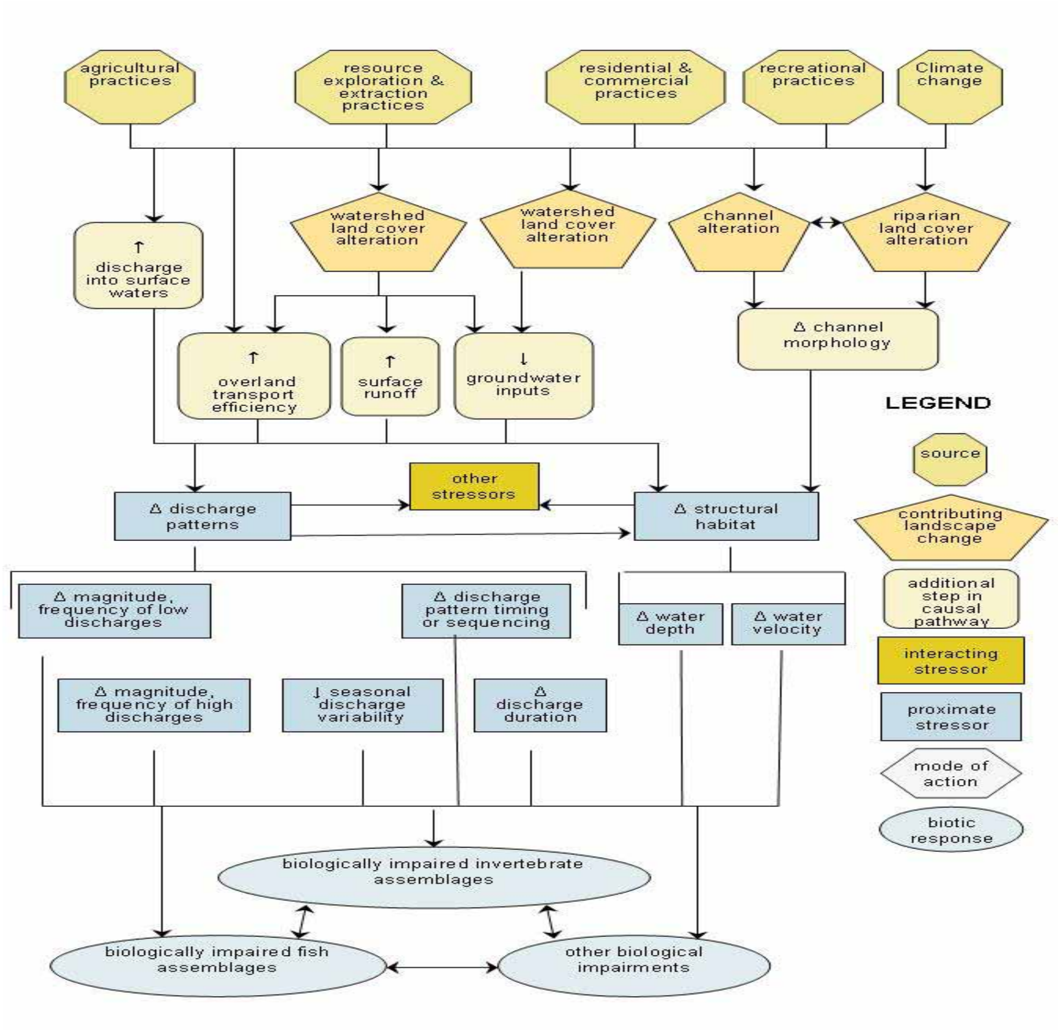


Figure 22. Model of sources and causal pathways for altered flow in Little Rock Creek

Climate change has an effect on precipitation. Streamflow has been found to be sensitive to precipitation in the Midwest of the United States, including most of Southern Minnesota. Stream flows in Minnesota reflect observed changes in precipitation, including increased mean annual precipitation, a larger number of intense rainfall events, more days with precipitation, and earlier and more frequent snowmelt events. Floods due to rainfall events are more likely. Higher summer and winter base flows are expected. Water quality and aquatic ecosystems should benefit from increases in low flows in both the summer and winter since water quality stresses are usually largest during low flow periods. (Novotny and Stefan, 2007).

According to a 2008 report by Galatowitsch, Frelich, and Phillips, climate change projections suggest that by 2069, average annual temperatures will increase approximately 5.8 degrees F; annual precipitation will increase 6-8%, but summer precipitation will decline. Stream flows in Minnesota reflect observed changes in precipitation with increases in mean annual precipitation, a larger number of intense rainfall events, more days with precipitation and earlier and more frequent snowmelt events (Novotny and Stefan). This means an increase in peak flows due to rainfall events in summer, the number of days with higher flows (flow durations), and in summer & winter base flows.

Table 7. Sufficiency of evidence for sources of altered flow in Little Rock Creek

Types of Evidence	Agricultural practices	Resource exploration & extraction practices	Residential & commercial practices	Recreational practices	Climate change
Evidence using data from Little Rock Creek					
Spatial / temporal co-occurrence	+	NE	NE	NE	-
Evidence of exposure, biological mechanism	NE	NE	NE	NE	NE
Causal pathway	+	NE	NE	NE	-
Field evidence of stressor-response	NE	NE	NE	NE	NE
Field experiments / manipulation of exposure	NE	NE	NE	NE	NE
Laboratory analysis of site media	NE	NE	NE	NE	NE
Temporal sequence	+	NE	NE	NE	+
Verified or tested predictions	NE	NE	NE	NE	NE
Symptoms	NE	NE	NE	NE	
Evidence using data from other systems					
Mechanistically plausible cause	+	NE	NE	NE	-
Stressor-response in other field studies	+	NE	NE	NE	NE
Stressor-response in other lab studies	NE	NE	NE	NE	NE
Stressor-response in ecological models	+	NE	NE	NE	-
Manipulation experiments at other sites	NE	NE	NE	NE	NE
Analogous stressors	NE	NE	NE	NE	NE
Multiple lines of evidence					
Consistency of evidence	+	0	0	0	-
Explanatory power of evidence	++	0	0	0	-

Conclusion

While the Minnesota areal 80-year average precipitation trend is positive, groundwater pumping rates in the Little Rock Creek watershed have steadily increased in the last decade, currently representing a 162% increase since the year 2000. There has also been a statistically significant increase in the amount of water pumped within the immediate vicinity of the creek. Reference data indicate that reduced flow in a stream can be a stressor to the fish community. The observed increases in water withdrawals from the Little Rock Creek watershed may be partially responsible for the recent findings that the brown trout community in Little Rock Creek no longer is self-sustaining, and efforts to reestablish a naturally reproducing community have been unsuccessful. The strength of evidence illustrates that agricultural practices are the most likely source of reduced flow in Little Rock Creek (Table 7). A key to the assigned values can be found in Table 5.

TEMPERATURE

Temperature Standard

Fish can be negatively impacted due to increases in temperature (Raleigh et al., 1986). Little Rock Creek fish surveys, in 2007 and 2008, have documented communities more representative of a warmwater stream than a coldwater stream, representing a shift from historical (1900's) conditions. The state standard for temperature in Class 2A streams is "no material increase" (7050.0222 Specific Water Quality Standards for Class 2 Waters of the State; Aquatic Life and Recreation). The temperature at which physiological stress, reduced growth and egg mortality occur for brown trout is called threat temperature at 18.3°C and critical temperature for life occurs at 23.9°C (Wherly et al., 2007).

Effects of increased temperature in streams

When temperatures rise close to 21°C, other fish have been shown to have a competitive advantage over trout for the food supply (Behnke, 1992). Fish continue to feed and gain weight at what is considered their functional feeding temperatures. Brown trout growth is maximized between 4 – 19.5 degrees Celsius (Elliot and Elliot, 1995); however for brown trout egg development temperatures between 0 and 15 degrees Celsius are required (Elliot, 1981). Functional feeding temperatures for brook trout are between 12.7°C and 18.3° (Raleigh, 1982). Temperatures near 22.2°C can be briefly tolerated, but temperatures of 23.8°C for a few hours are typically lethal (Flick, 1991). Density of juvenile brook trout has been shown to be negatively correlated with July mean water temperatures (Hinz and Wiley, 1997). Juvenile brook trout are highly dependent, for growth and distribution, on temperature (McCormick et al. 1972). For more than 80 years (1907 – 1990) stream temperatures in Little Rock creek supported a wild coldwater fishery (brown trout), but current conditions suggest a warming trend in the creek and an associated transition to a cool- or warmwater fish assemblage.

Data evaluation for temperature in Little Rock Creek

Temperature loggers deployed in Little Rock Creek from June to September 2008, collected 15 minute interval measurements. The temperatures ranged from 7.895 to 28.196 degrees Celsius. The temperature logger at site 12 malfunctioned and the logger at site 11 may have been out of the water for a period of time from August 13th to 19th.

This study utilizes temperature data collected in 2008 because fish surveys were conducted concurrently. It is important to note that other stream temperature data is available, including 2002 Trout Unlimited data from 7 sites and MN DNR data from 2005 and 2006. Both Trout Unlimited and MN DNR used HOBO temperature loggers. The Trout Unlimited data was not compared to 2005, 2006, or 2008 temperature data due to project time constraints. The 2005 and 2006 DNR temperature data was reviewed. In 2005, site 13 exceeded the threat temperature most of the time between the months of June and August. In 2006, sites 7 and 9 exceeded the threat temperature between the months of May and August. Sites 7, 9 and 13 did have some readings that exceeded the critical temperature of 23.9 degrees Celsius.

The 2008 daily maximum, minimum and mean water temperatures were calculated for Little Rock Creek (Figures 23 through 25). The temperature fluctuation was also calculated for each of the sites (Figure 26). The number of measurements that were greater than 18.3 °C varied each month, with station 13 having over 2500 measurements greater than the threat temperature in July, which is over 90% of the measurements (Figure 27 & 28). The mean monthly temperatures varied among sites, with site 13 having the warmest July monthly average at 20.70 °C (Figure 28 & Table 8). Station 9 was the coldest location, in July, with a mean temperature of 16.02 °C. Figures 29 through 35 depict each 15 minute interval that is above the brown trout threat temperature. Stations 11 and 13 were the only sites to have measurements above the brown trout critical temperature of 23.9 °C (Figure 36). Figures 37 and 38 show the measurements greater than the critical temperature, for stations 11 and 13.

Trout are present in Little Rock Creek, but it is unknown as to how much of that is due to stocking and how much is natural reproduction. In 2008 sampling by the MPCA, less than 1% of the fish surveyed at station 9 were coldwater fish and less than 2% at station 7 (Figure 39). Stations 5 and 7 had presence of coolwater fish, but less than 3% and 1% respectively. The other stations, 6, 8, 11, 12 and 13, did not have cold or coolwater fish at the time of surveying.

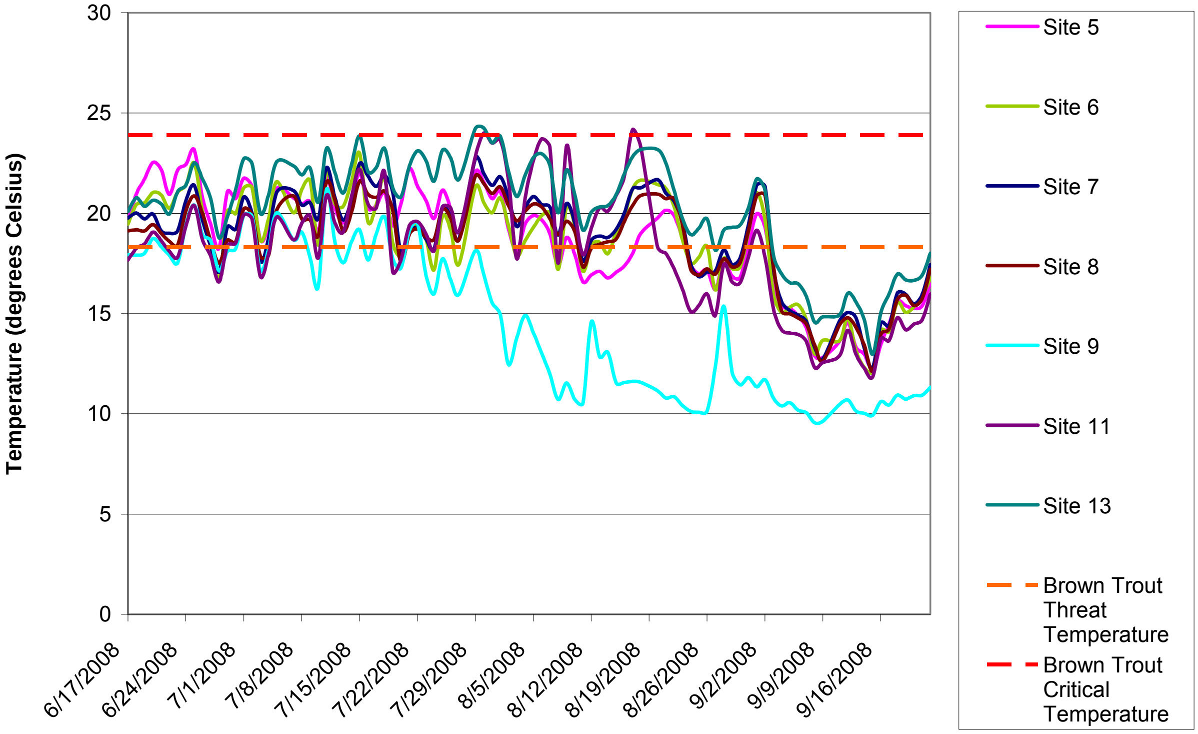


Figure 23. Daily maximum water temperature in Little Rock Creek 2008

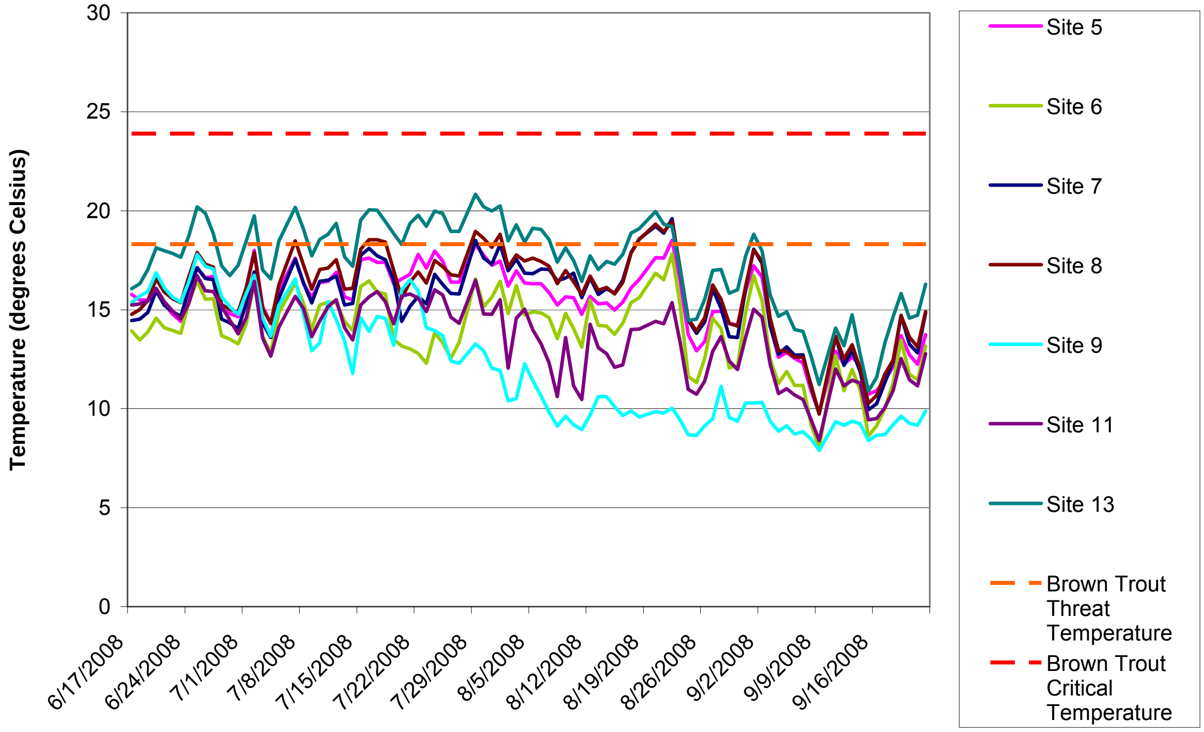


Figure 24. Daily minimum water temperature in Little Rock Creek 2008

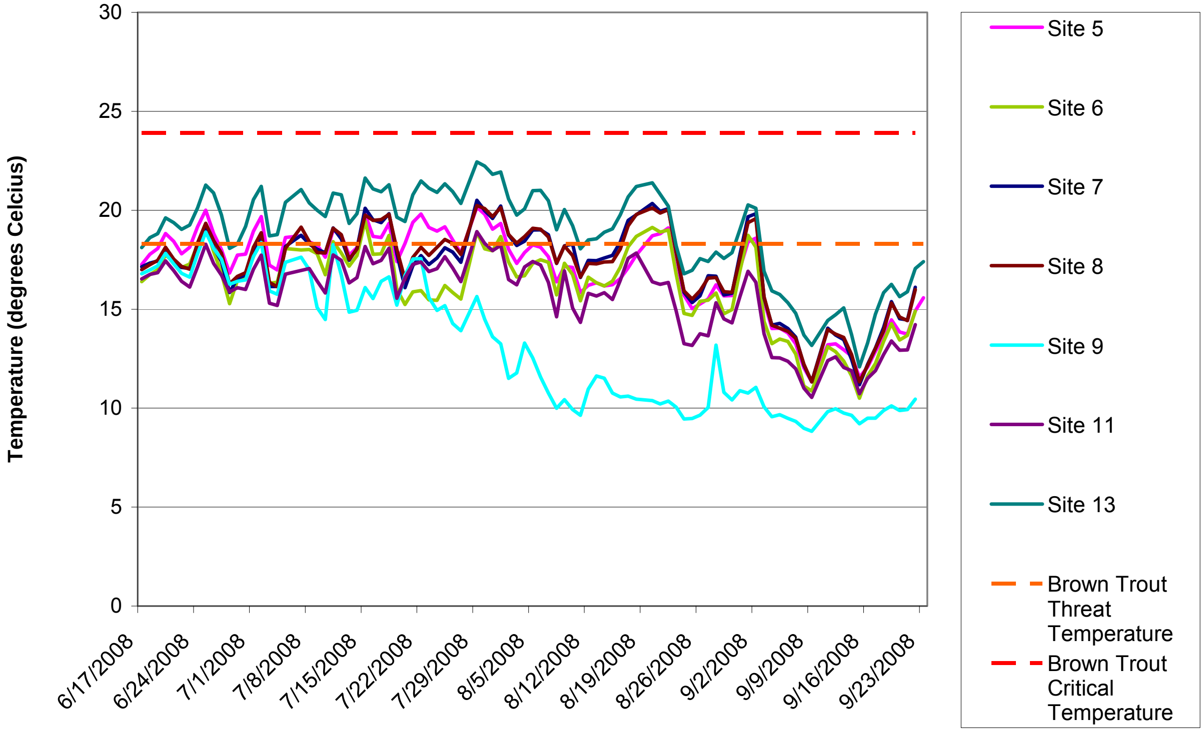


Figure 25. Mean daily water temperature in Little Rock Creek 2008

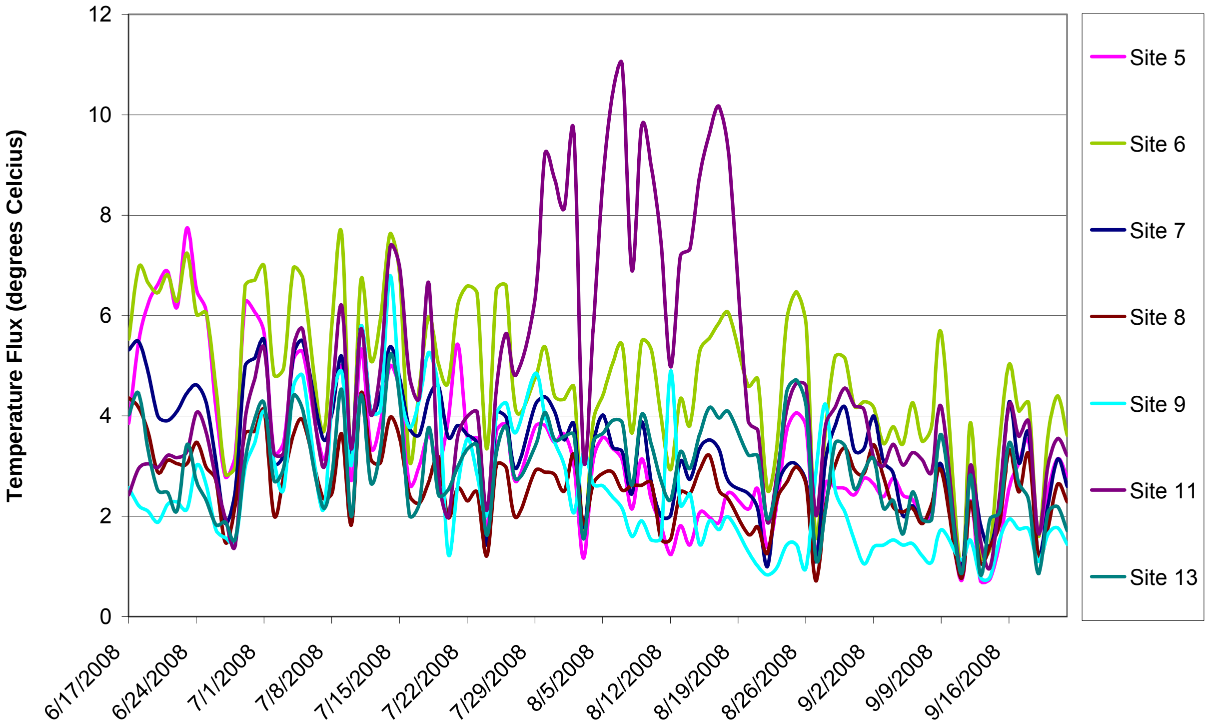


Figure 26. Daily water temperature flux in Little Rock Creek 2008

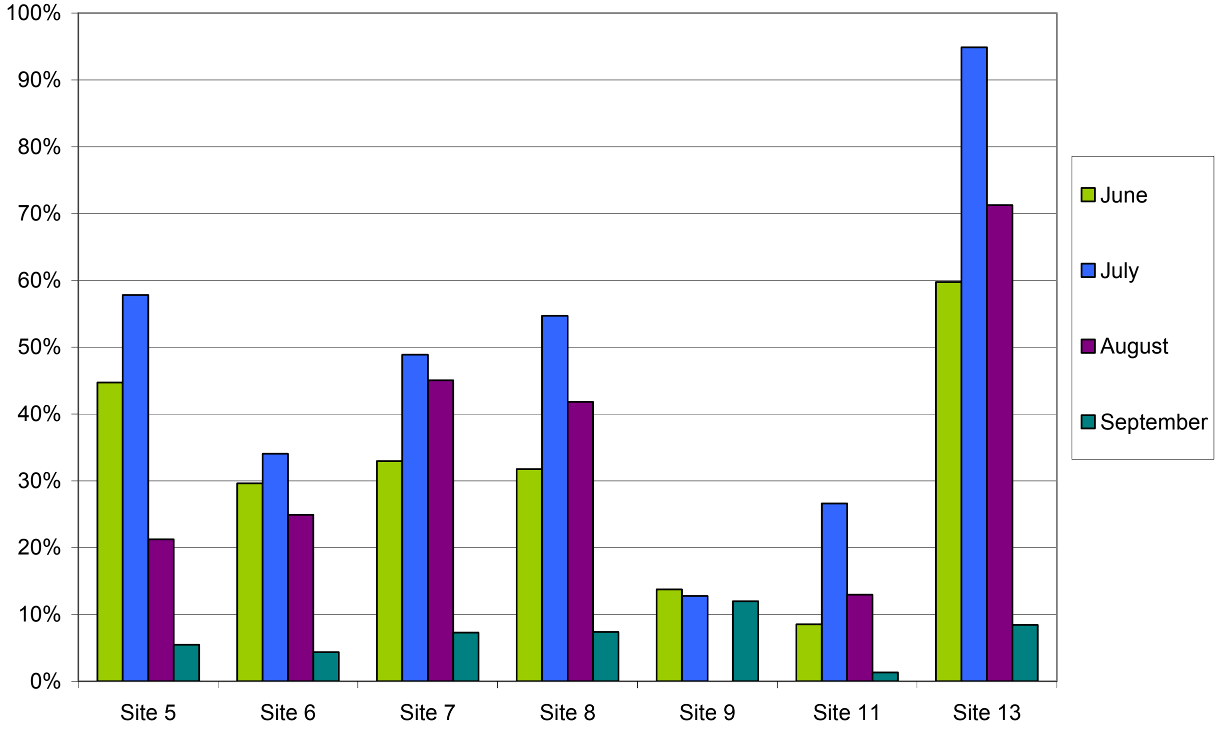


Figure 27. Percentage of measurements greater than 18.3 degrees Celsius in Little Rock Creek 2008

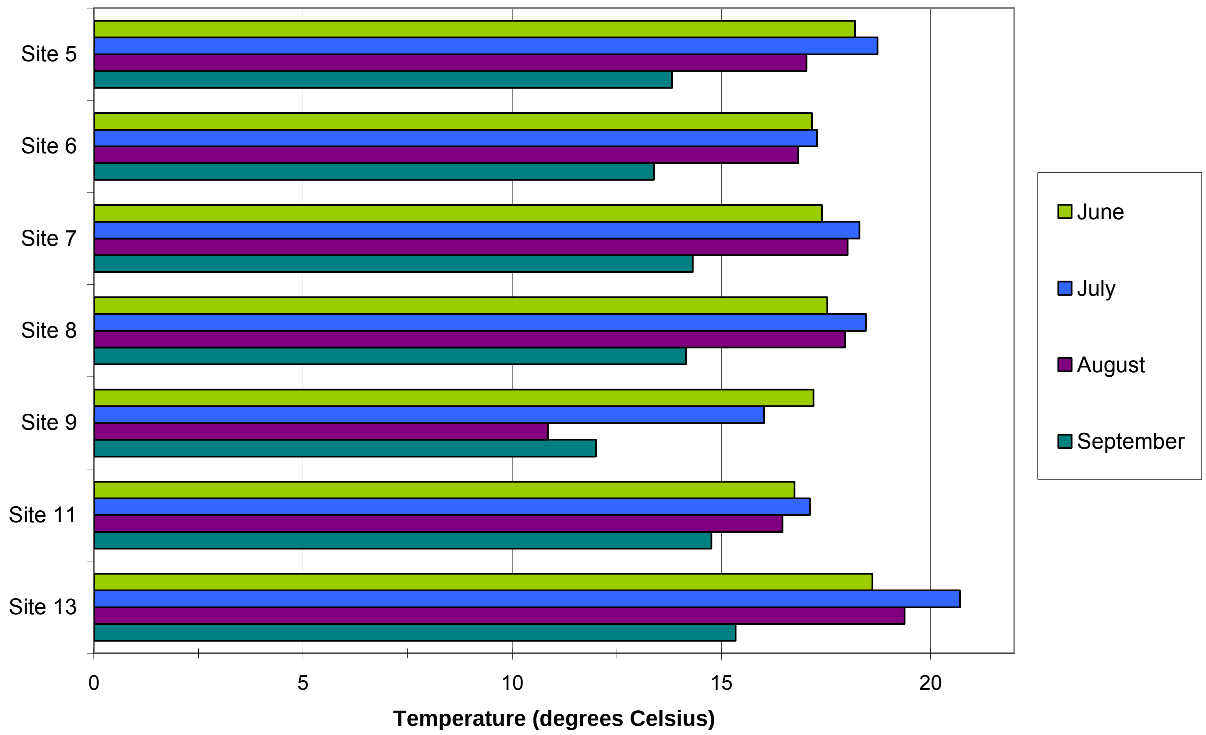


Figure 28. Mean monthly temperatures in Little Rock Creek 2008

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Table 8. July mean temperatures for 2008

Sites	5	6	7	8	9	11	13
July Mean Temperature (°C)	18.73	17.28	18.30	18.45	16.02	17.11	20.70

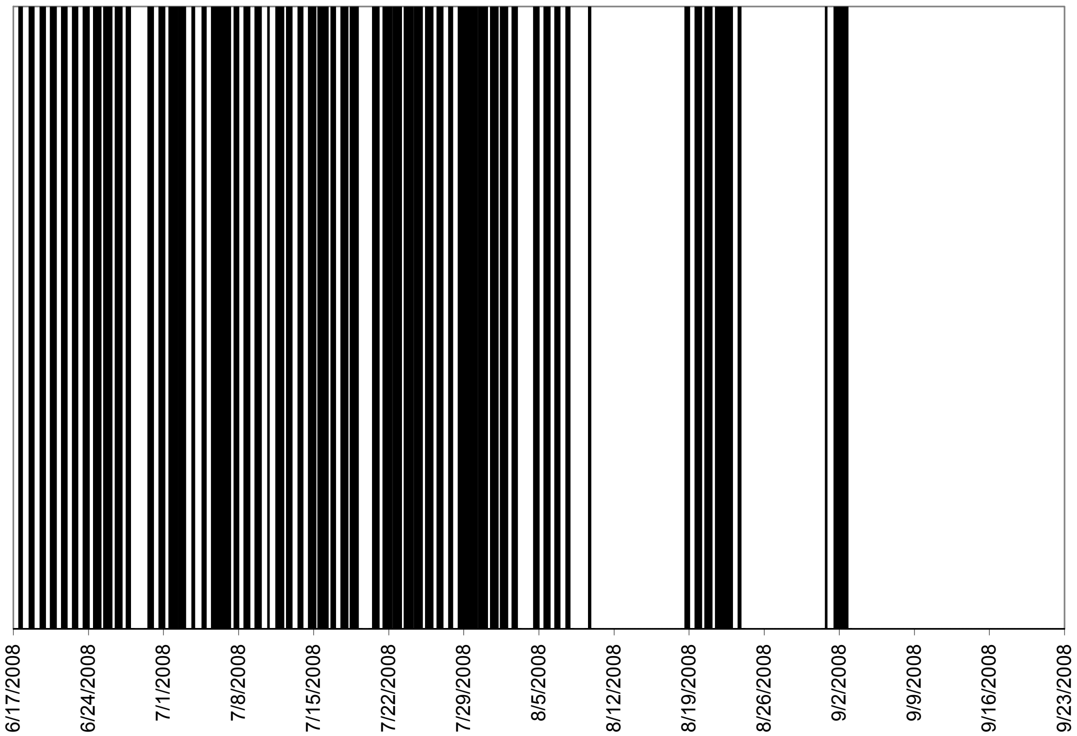

Figure 29. Frequency of 15 minutes periods above brown trout threat temperature (18.3 degrees Celsius) at site 5

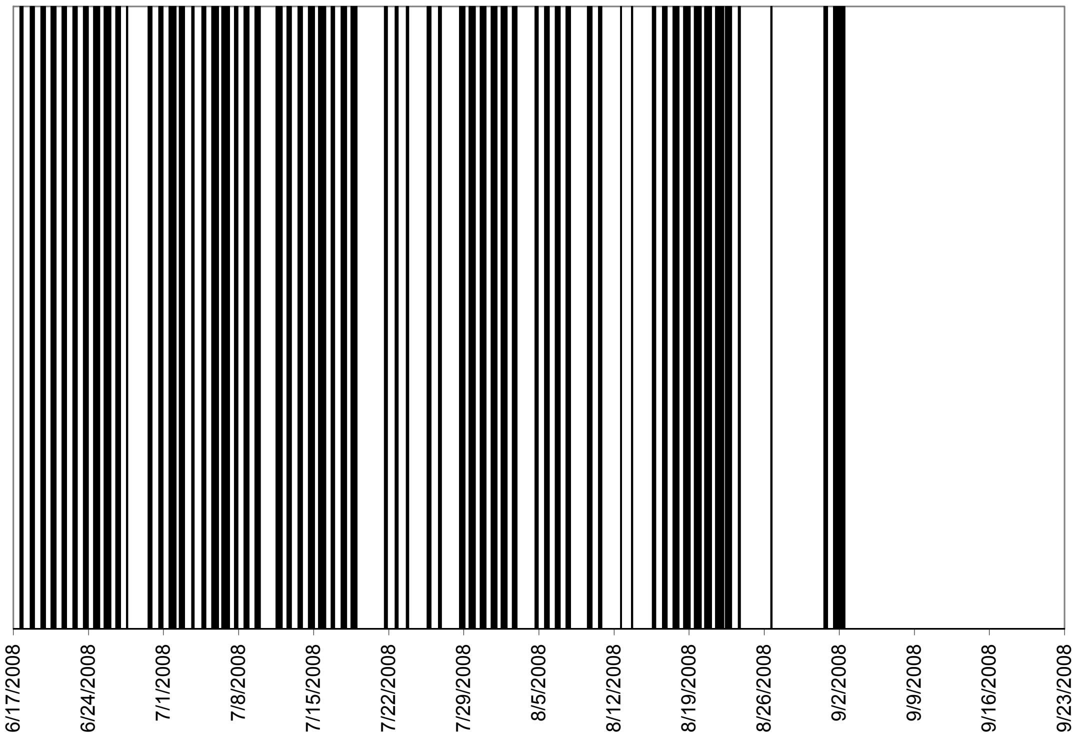

Figure 30. Frequency of 15 minute periods above brown trout threat temperature (18.3 degrees Celsius) at site 6

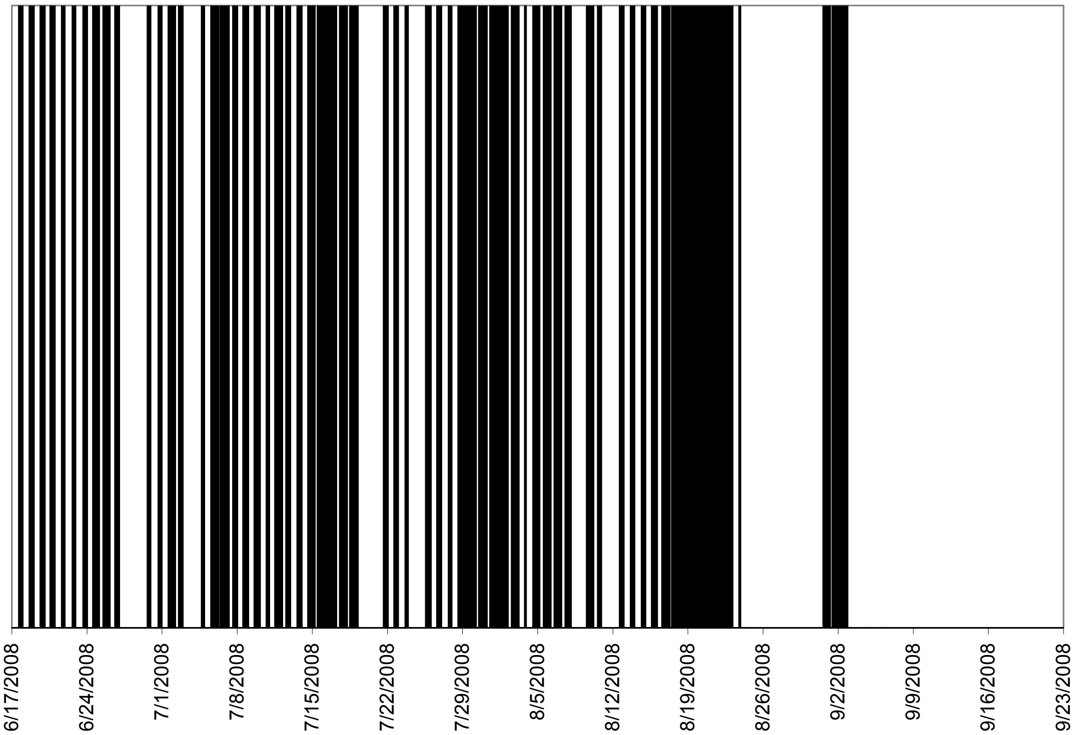

Figure 31. Frequency of 15 minutes periods above brown trout threat temperature (18.3 degrees Celsius) at site 7

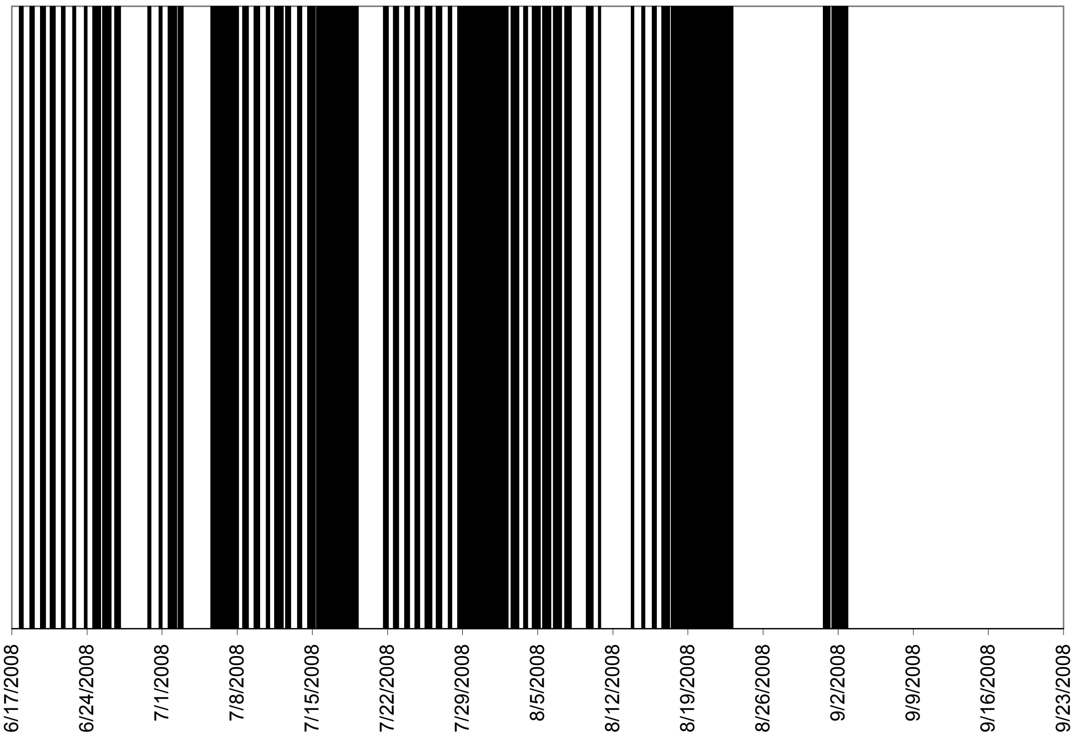

Figure 32. Frequency of 15 minute periods above brown trout temperature (18.3 degrees Celsius) at site 8

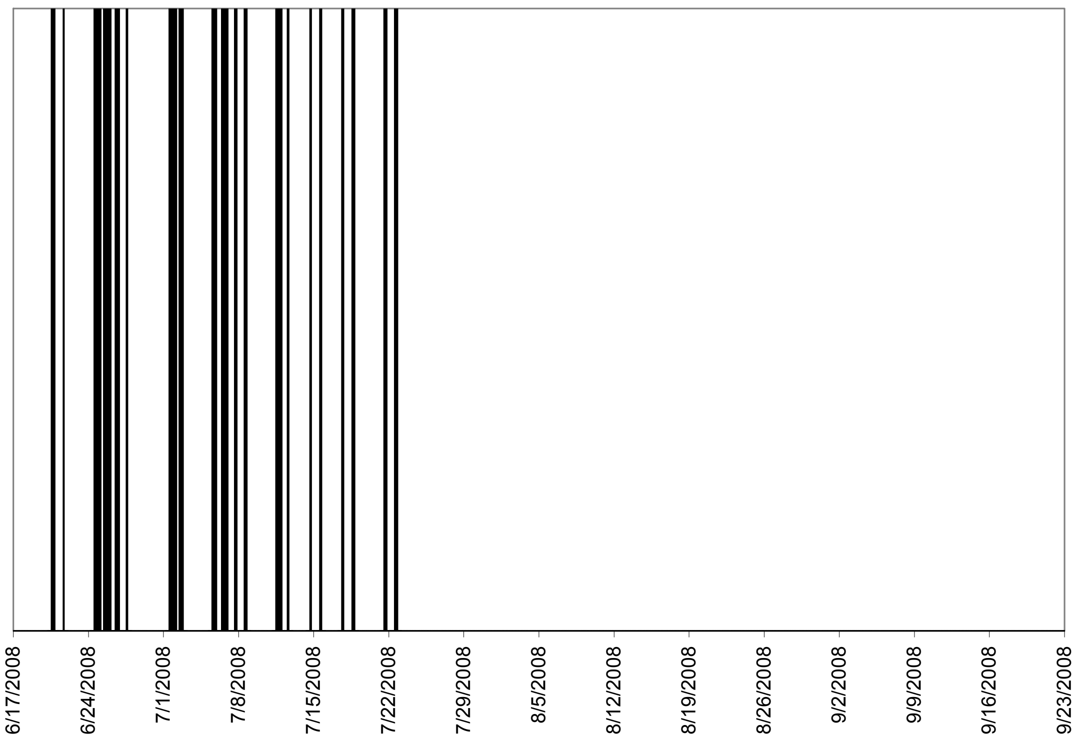

Figure 33. Frequency of 15 minute periods above brown trout threat temperature (18.3 degrees Celsius) at site 9

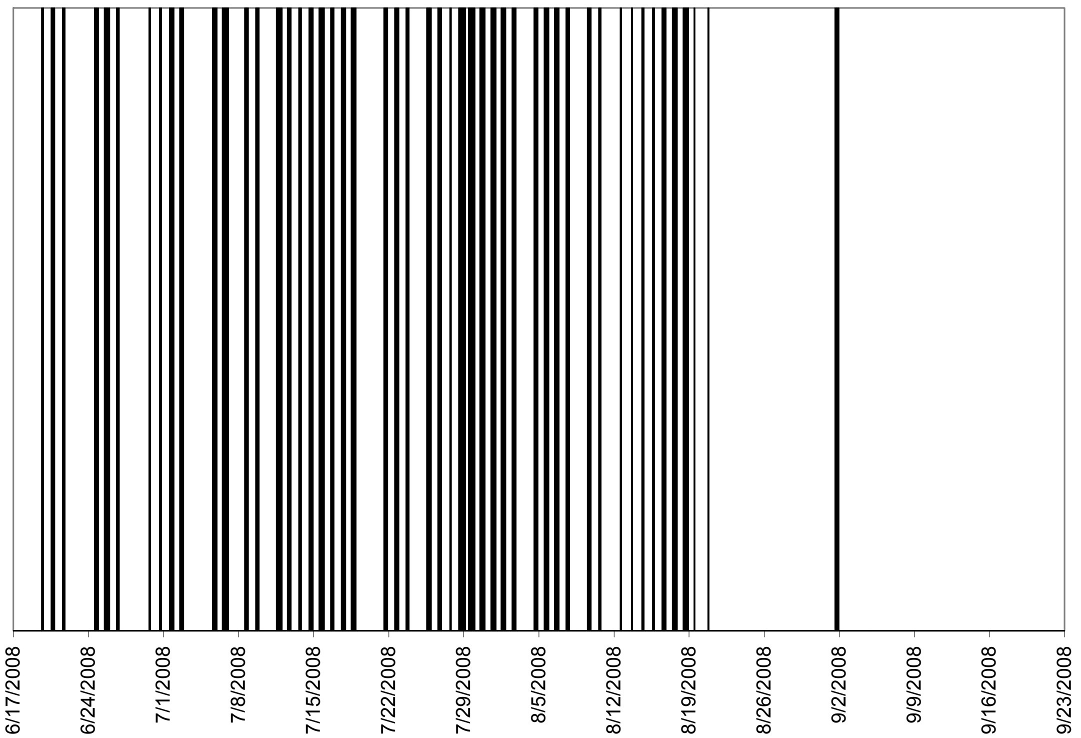

Figure 34. Frequency of 15 minute periods above brown trout threat temperature (18.3 degrees Celsius) at site 11

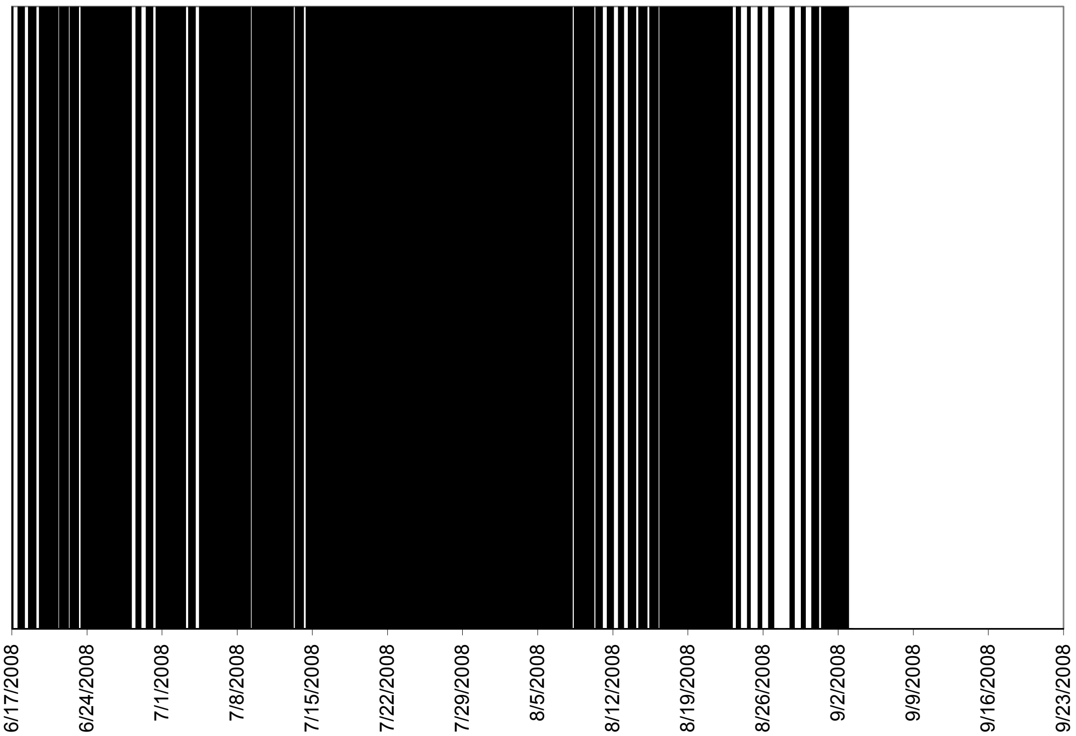

Figure 35. Frequency of 15 minutes periods above brown trout threat temperature (18.3 degrees Celsius) at site 13

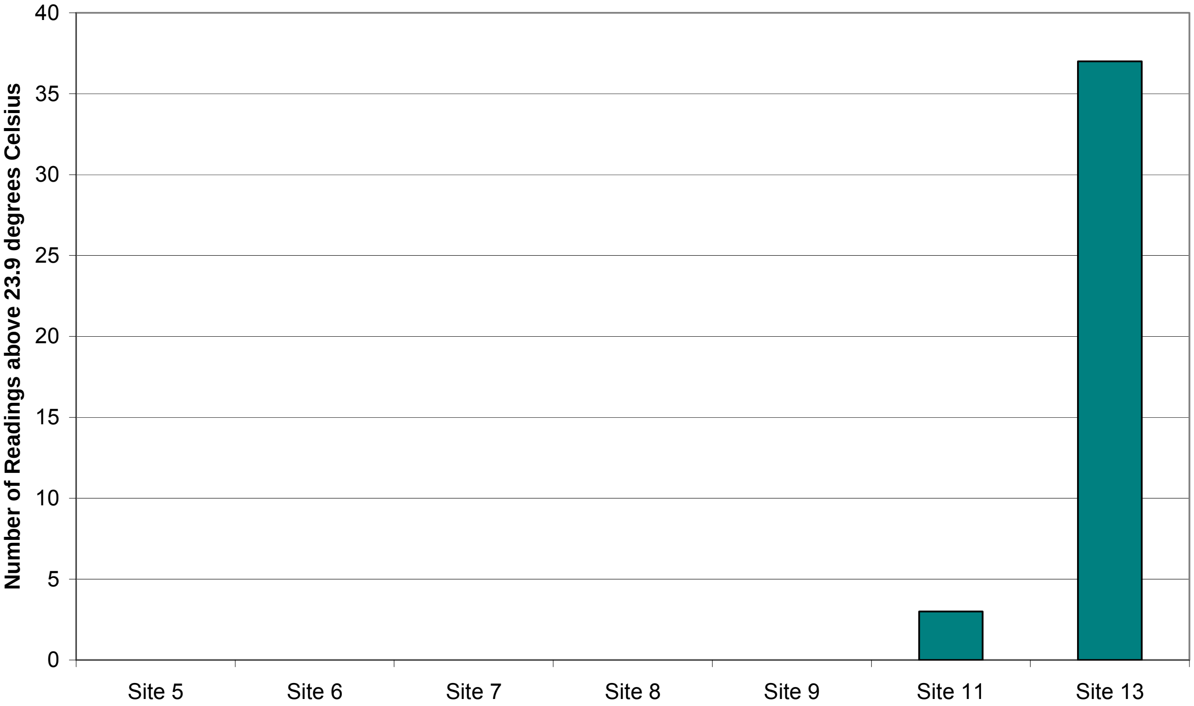

Figure 36. Number of measurements above 23.9 degrees Celsius in Little Rock Creek 2008

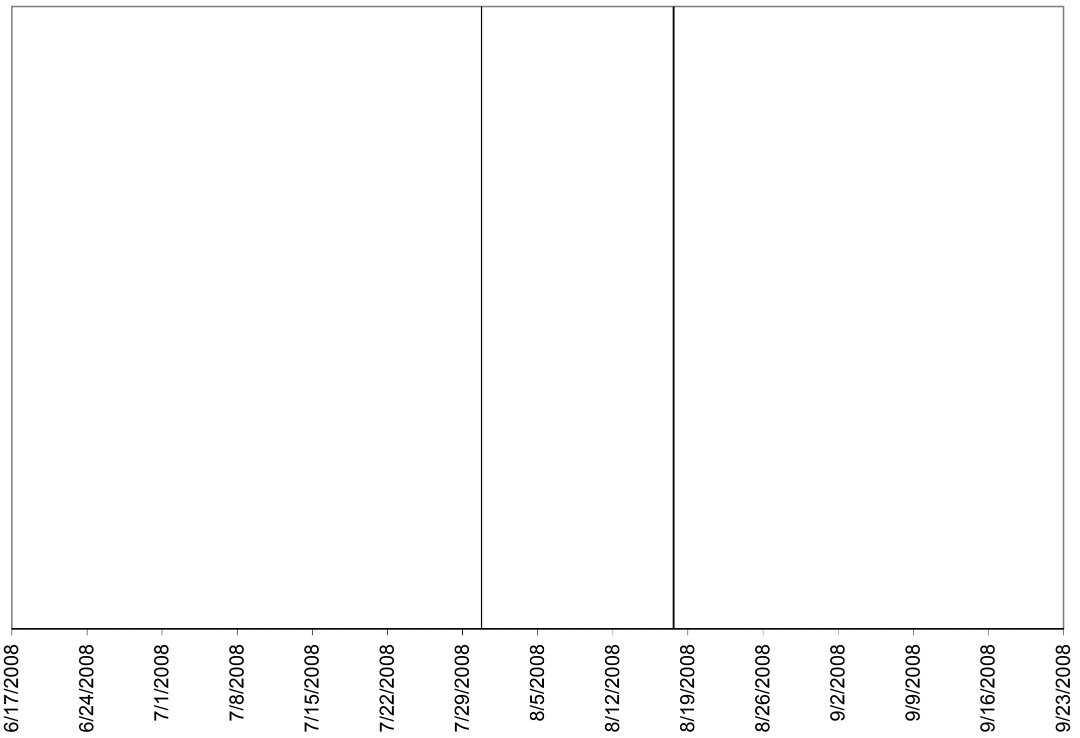

Figure 37. Frequency of 15 minute periods above brown trout critical temperature (23.9 degrees Celsius) at site 11

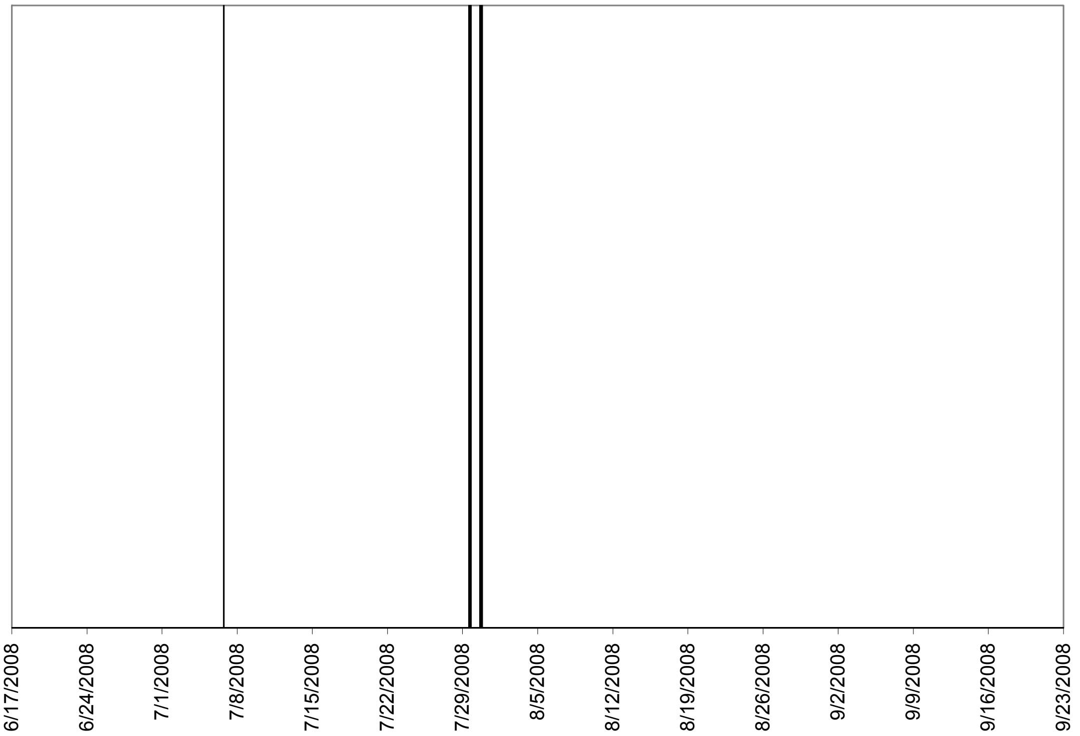

Figure 38. Frequency of 15 minute periods above brown trout critical temperature (23.9 degrees Celsius) at site 13

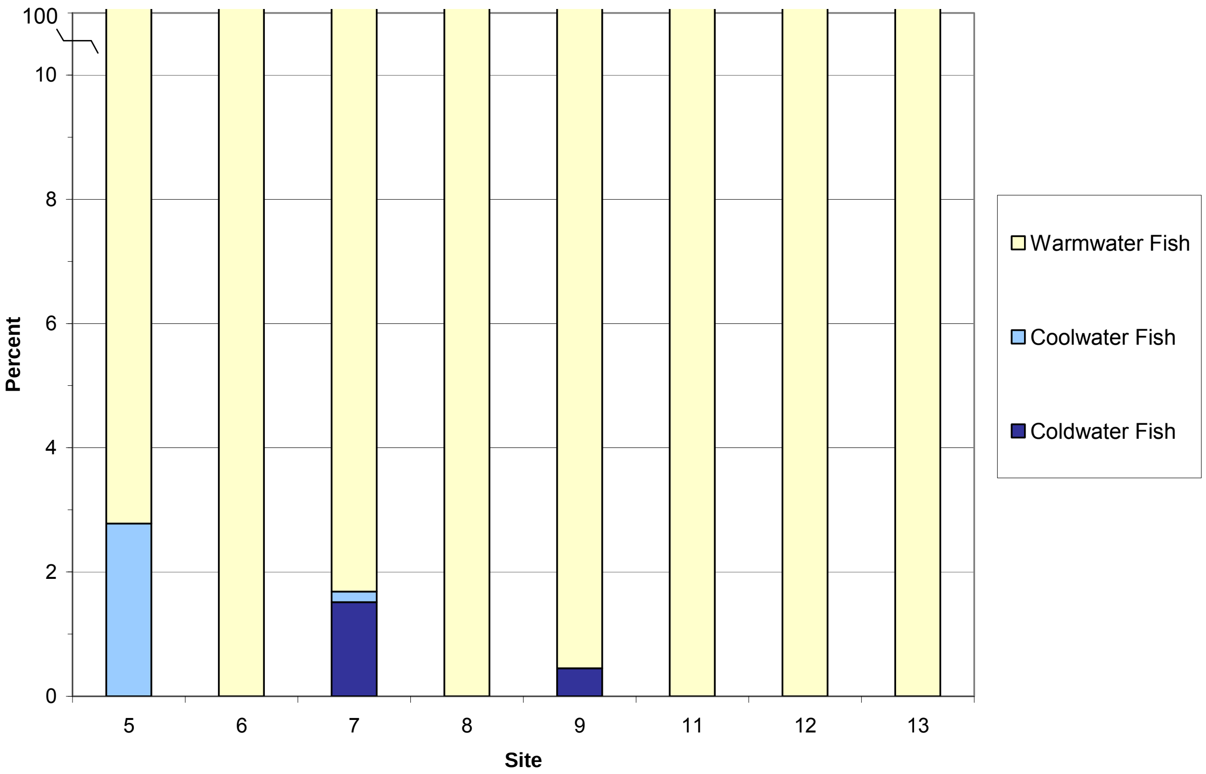

Figure 39. Percentage of warmwater, coolwater and coldwater fish in Little Rock Creek 2008

Models of sources and causal pathways for temperature increase in Little Rock Creek

The sources likely to impact stream temperature in Little Rock creek are shown in Figure 41. Each pathway was evaluated using CADDIS, shown in Table 9. It is likely that with groundwater withdrawal, there is a decrease in coldwater inputs to the creek leading to the increased temperatures.

The increased surface area in the impounded pool above the dam is likely the source of the increased mean July temperature at Site 13. Stream structure determines if the stream will resist warming or cooling (Poole and Berman, 2001). With increased channel widths there is a greater surface area for heat to be exchanged between the water and air. Additionally, stream flow decreases and stream bed aggradation can lead to increased stream temperatures. In most streams, the width-depth ratio typically increases as you move downstream (Gordon et al., 2004). However, the width-depth ratio of surveyed reaches in Little Rock Creek, ratio does not follow this general pattern, higher than expected width-depth ratios were observed at site 6 and to a lesser extent at stations 8 and 9 (Figure 40). Streams that are aggrading are typically wider and shallower when compared to stable or degrading streams. It is likely Little Rock Creek is aggrading since it has a high width-depth ratio and undercut banks with some slumping and eroding banks, further discussion can be found in the sediment section.

Land cover alteration in the watershed and in the riparian corridor may be connected to the observed increase in Little Rock Creek's stream temperature. In-stream habitat assessments have noted some areas of decreased canopy cover due to changes in land use. Mid-stream locations at stations 7 and 9 were found to have more canopy cover than station 6 and 13. In 2006, the watershed comprised of 48% row crops. The clearing of forested land and conversion to row crop agricultural land may be leading to increases in solar heating and warm surface runoff.

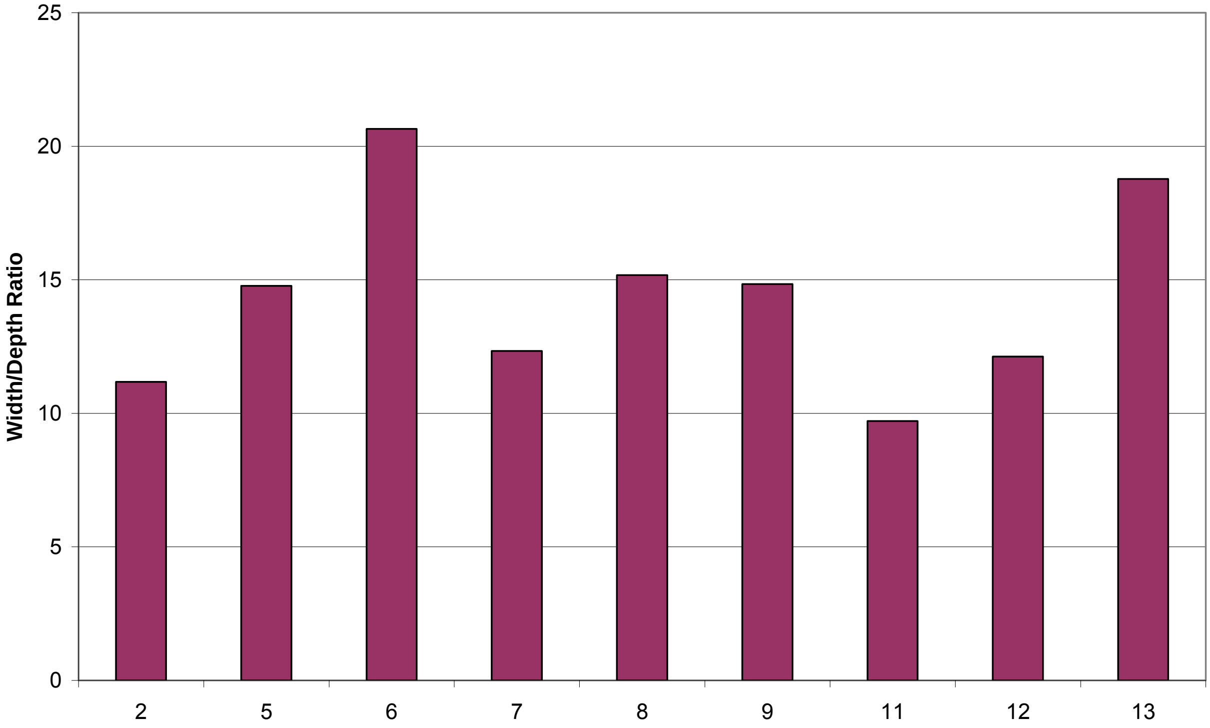

Figure 40. Width-depth ratio for the sites in Little Rock Creek

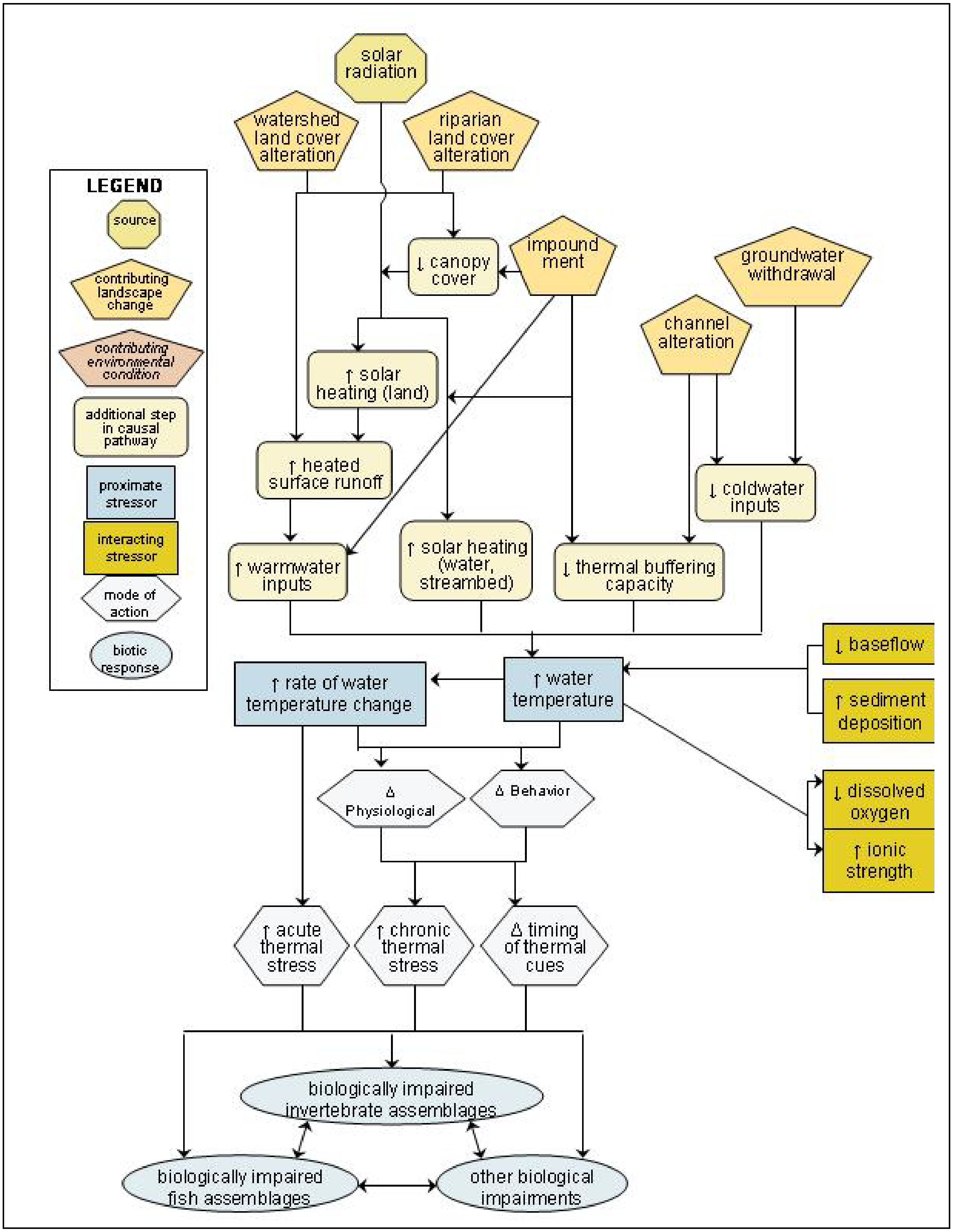

Figure 41. Model of sources and causal pathways for temperature increases in Little Rock Creek

Table 9. Sufficiency of evidence for sources of temperature increase in Little Rock Creek

Types of evidence	Watershed land cover alteration	Riparian land cover alteration	Impound-ment	Channel alteration	Groundwater Withdrawal
Evidence using data from Little Rock Creek					
Spatial/temporal co-occurrence	0	0	+	0	+
Evidence of exposure, biological mechanism	+	+	+	+	+
Causal pathway	+	+	+	+	+
Field evidence of stressor-response	0	0	0	++	0
Field experiments / manipulation of exposure	NE	NE	NE	NE	NE
Laboratory analysis of site media	NE	NE	NE	NE	NE
Temporal sequence	+	+	+	+	+
Verified or tested predictions	NE	NE	NE	NE	NE
Symptoms	0	0	0	0	0
Evidence using data from other systems					
Mechanistically 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	NE	NE	NE	NE	NE
Analogous stressors	-	-	-	-	-
Multiple lines of evidence					
Consistency of evidence	+	+	+	+	+
Explanatory power of evidence	++	++	++	++	++

Conclusion

Temperature in Little Rock Creek exceeds threat and at times critical temperatures for brown trout. The data strongly suggest that the increased temperatures in the stream are a stressor to the biological community. The strength of evidence shows that the impoundment and groundwater withdrawal are likely causing increased stream temperatures.

SEDIMENT

Sediment is a natural part of aquatic habitats, and the dynamics of sediment movement in a stream affect its biota Equal inputs and exports of sediment within the stream creates a stable substrate condition where the stream is neither aggrading nor degrading, and provides a habitat conducive to stable and healthy biological communities. However, insufficient sediment or excess sediment can negatively impact biota. When a stream is sediment starved (insufficient), the channel responds by obtaining sediment from either the channel bed or banks, which can result in bank erosion and/or channel incision. Typically the mechanism for this is an increase in shear stress and sediment competency due to velocity increases. The channel then can either over-widen or incise, both of which have connectivity issues if severe enough and are also indicators of an unstable system.

There are two types of sediment that may affect stream biota: suspended sediment and deposited/bedded sediment. Sediment transport can be separated into two primary categories based on the mechanism by which sediment moves through the system; suspended or bedload.

Sediment Standard

Suspended sediment is primarily fine inorganic particles of clay and silt but also includes sand and particulate organic matter suspended in the water column. Suspended sediment was not determined to be a stressor in Little Rock Creek, as mentioned previously, because the sample values are all below the 75% percentile ecoregion standard of 16 mg/L.

Deposited and bedded sediments are mineral and organic particles that settle out of the water column and collect on the bed of a water body, or that travel primarily by rolling along a stream bed rather than moving in the water column. It includes surficial deposits, as well as, deeper, coarser deposits and bedded layers within the depths used by organisms. There is no bedded sediment standard for surface waters in Minnesota. There is also no ecoregion standard for bedded sediments in Minnesota.

Effects of bedded sediment in streams

In their comprehensive literature review, Wood and Armitage (1997) discuss how the natural variability of river flow, from the extremes of flood to low flows, results in variations in the concentration of suspended solids and their deposition. A reduction in flow velocity, particularly during low flow conditions during the summer months, can lead to large volumes of fines and decaying organic matter being deposited onto the riverbed (Giles et al 1991). This problem is particularly acute in groundwater-fed streams, which rely on precipitation for aquifer recharge (Wright and Berrie 1987). At the same time, the particle size of deposited stream bed sediments affects the flow resistance in the channel, the stability of the bed, and the amount of available aquatic habitat types. These interacting factors determine the sediment regime of a stream, and exert a significant influence on macroinvertebrate and fish communities (Mebane, 1999).

In general, biological communities can withstand short-term increases in suspended and benthic sediments, and recover rapidly from short-term additions of fine particulate material due to human disturbance. However, continuous high levels of sediment input, often associated with disturbances such as agriculture and surface mining activity, may completely change the natural faunal assemblage. Specific responses of the stream biota to changes in a stream's sediment regime vary based on the characteristics (e.g. behavioral, life history, trophic) of the organisms involved, as well as characteristics of the disturbance (e.g. type and duration).

Invertebrates

Wood and Armitage (1997) define four ways that fine sediment suspension and deposition affects benthic invertebrates: 1) by altering substrate composition and changing its suitability for some taxa; 2) by increasing drift due to sediment deposition or substrate instability; 3) by affecting respiration due to the deposition of silt on respiration structures or low oxygen concentrations associated with silt deposits; and 4) by impeding filter feeding due to an increase in suspended sediment concentrations, reducing the food value of and reducing the density of prey items. For example, Mebane (1999) observed a strong negative correlation of the number of invertebrate taxa belonging to the orders Ephemeroptera, Plecoptera, and Trichoptera with increasing fine surface sediment, and that the abundance of the Scraper functional feeding group decreased dramatically when fine sediment occupied 40% or more of the stream channel.

Alterations to the substrate composition by the deposition of sand and other fine particles are a particular problem that is highlighted in many studies. The deposition of sand indirectly affects benthic fauna by impeding their upstream migration, even at low current velocities. Sand is an inherently unstable substrate with most benthic taxa being found in the uppermost layers of the substrate and some small taxa reach very high densities. It has also been recognized that the timing of sand deposition, peaking during base flow conditions, coincides with the period of dispersion and colonization by young benthic macroinvertebrates (Wood and Armitage). Sediment deposition effects invertebrate biomass, diversity, density and abundance, feeding and respiration, and drift.

Fish

Wood and Armitage define at least five ways in which high concentrations of fine sediment adversely affect lotic fisheries: 1) by adversely acting on the fish swimming in the water and either reducing their rate of growth, reducing their tolerance to disease or killing them; lethal concentrations primarily kill by clogging gill rakers and gill filaments; 2) by reducing the suitability of spawning habitat and hindering the development of fish eggs, larvae and juveniles; all of these stages appear to be more susceptible to suspended solids than adult fish; 3) by modifying the natural migration patterns of fish; 4) by reducing the abundance of food available to fish due to a reduction in light penetration and as a result photosynthesis, primary production, and a reduction of habitat available for insectivore prey items; and 5) by affecting the efficiency of hunting, particularly in the case of visual feeders.

Sediment deposition effects fish biomass, diversity and abundance, feeding and respiration, and spawning. Fish community level responses to varying siltation were most consistently described by changes in functional characteristics of the communities. In studies, herbivores, benthic insectivores and simple lithophilic spawners were most sensitive to siltation while other guilds were not (Rabeni and Smale, 1995).

Data evaluation of deposited and bedded sediment in Little Rock Creek

MPCA Study Summary

The MPCA Watershed Unit conducted a Level 2 Rosgen field survey on Stations 2, 5 – 9, and 11 – 13 in 2008 (Figure 7). Rosgen's stream classification involves a four level classification hierarchy. The first level begins with a geomorphic characterization; this is followed by a morphological description which is the second level. The third level involves an assessment of stream's condition. The fourth level is monitoring and validation. Each level rests on the information derived from the previous level. The first two levels rely heavily on the conditions during bankfull discharge (Epstein).

The level 2 data collected for the nine stations provides a geomorphic characterization and morphological description of the stream at those stations. Level 2 data cannot determine channel stability and/or sediment sources. However it can offer limited insights on stream conditions, but additional data collection specifically Level 3 and 4 need to be completed for a quantitative assessment of stability.

Two out of the nine stations were classified as E5 and the remaining stations were C classifications with minor differences based on sediment composition and slope. More than half of the stations had ratio's that suggest a channel shift is underway. Channel type shifts can either indicate a move to a more stable or unstable condition, but is indicative of the stream responding to a change within the watershed, be it hydrology, riparian cover, landuse, connectivity or some other alteration.

Estimated bankfull discharges were also calculated and show an unexpected pattern. Sites six (upstream) through nine (downstream) show bankfull discharge fluctuating, which could be a result of increased irrigation, disconnection to groundwater flows or surface flow going subterranean. Discharge increases at site 11 due to inputs by Bunker Hill Creek and sites 12 and 13 are impacted by the Sartell Wildlife Management impoundment.

Table 10. Summarized geomorphic data

Site	**Classification**	**Width/depth ratio**	**Entrenchment ratio**	**Sinuosity**	**Slope (ft/ft)**	**Bankfull Discharge (cfs)**	**Bankfull Velocity (fps)**
2	E5	11.2	2.3	1.6	.0006	28	1.7
5	C5	14.8	5.3	1.8	.0015	52	2.1
6	C5	20.7	6.3	1.5	.0003	128	2.6
7	C4	12.3	5.0	1.3	.0043	106	3.2
8	C5	21.9	6.6	1.5	.0035	264	2.8
9	C5	14.8	3.9	1.8	.0007	170	3.9
11	E5	8.9	37.2	1.7	.0012	432	5.4
12	C5c-	12.1	2.9	1.4	.0004	303	3.3
13	C5	18.8	4.4	1.9	.0007	372	2.8

As part of the Level 2 surveys, pebble counts were completed at each station at the riffle cross section and reach wide. The reach pebble count is used to classify the reach, while the riffle counts are used to calculate discharge. In 2007 and 2008, MPCA completed pebble counts at the stations in a cross section of a riffle, as well, as in the longitudinal profile. Figures 42 and 43 compare the material found at each of the stations in the riffle cross sections and longitudinal surveys. Station 7 also had the steepest slope, which also helps explain the relative reduction of sand. The MPCA field survey team recorded finding cobble buried under an arm's length of sand.

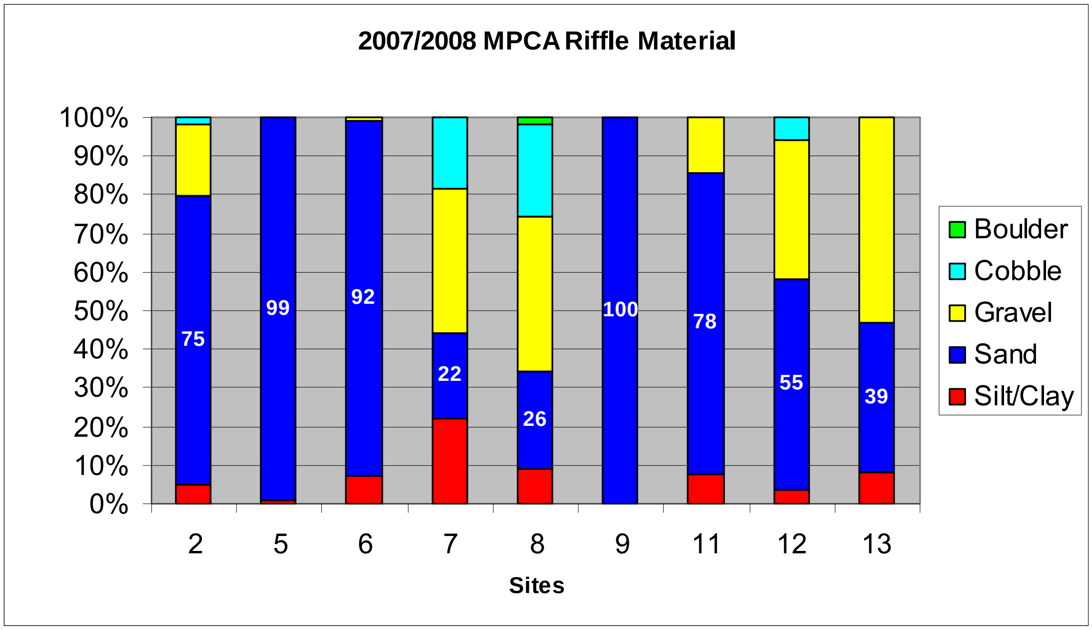

Figure 42. 2007/2008 MPCA riffle material

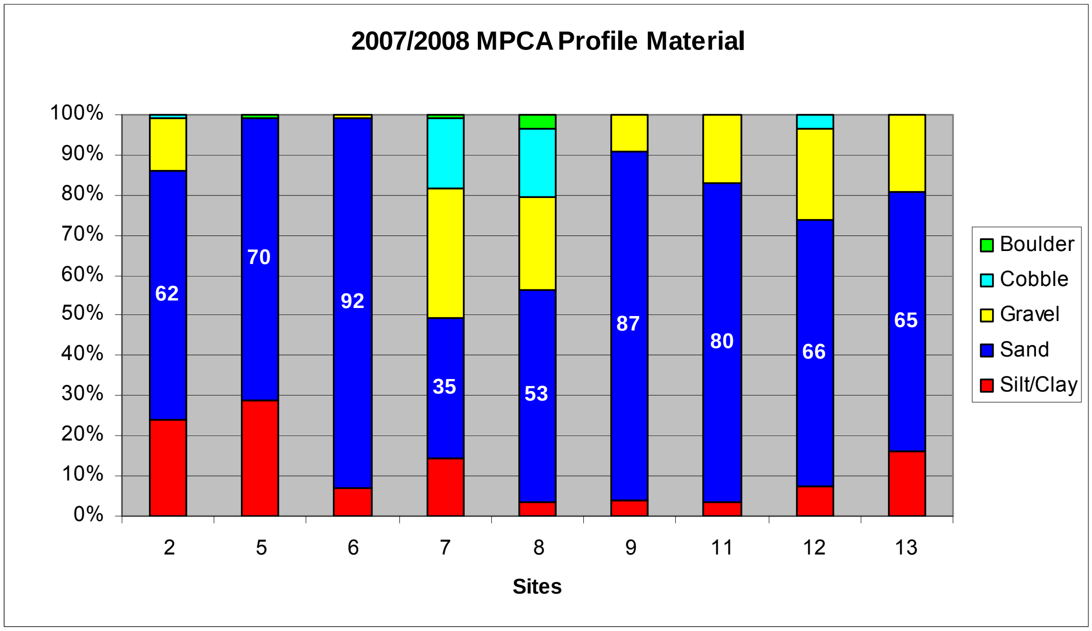

Figure 43. 2007/2008 longitudinal profile material

One method used by MPCA to evaluate sedimentation in streams is the measurement of "Depth of Fines". Using a graduated rod, the water depth is recorded both from the top of the stream bottom, and after pushing the rod into the sediment as far as possible (without leaning on the rod or otherwise using body weight). The difference between the two measured values is Depth of Fines; large values for this metric may indicate excess sedimentation, which in turn may have negative effects on stream biota. Statewide biomonitoring data collected by the MPCA indicates that several positive biological metrics (indicative of healthy biological communities) demonstrate negative relationships with Depth of Fines, including Percent Benthic Insectivore Fish Individuals, Percent Sensitive Fish Individuals, and Percent Simple Lithophil Fish Individuals (Figure 44).

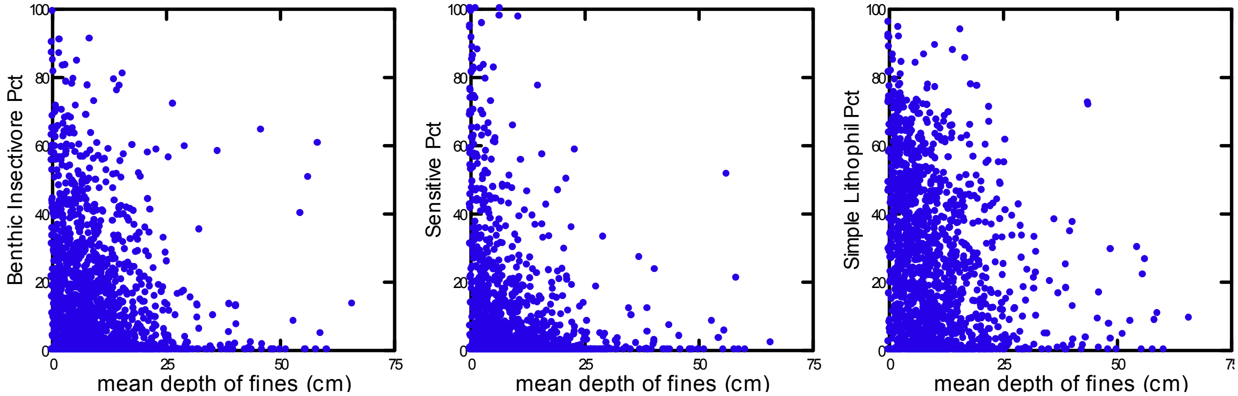

Figure 44. Three positive biological metrics (Benthic Insectivore %, Sensitive %, and Simple Lithophil %) plotted against Mean Depth of Fines. Data from MPCA statewide biological monitoring sites.

The wedge-shaped response distribution of these biological metrics likely reflects limitation by other, unmeasured factors at certain locations and at certain times (Cade and Noon 2003). However, the maximum biological response is clearly depressed as mean depth of fines increases, and causative mechanisms can be readily identified. For example, Benthic Insectivores, which search for prey living on the stream bottom, may experience decreased foraging efficiency or an overall decline in prey availability due to accumulation of fine sediment. In a similar manner, fine sediment may smother clean, coarse substrate, which Simple Lithophils require to carry out their spawning activities.

In Little Rock Creek, relationships between these biological metrics and Mean Depth of Fines largely correspond with those of the statewide dataset (Figure 45). However, a small sample size and the influence of multiple stressors may be partially confounding the response. For example, Sensitive individuals are extremely rare in Little Rock Creek, even at sites where Mean Depth of Fines is quite low.

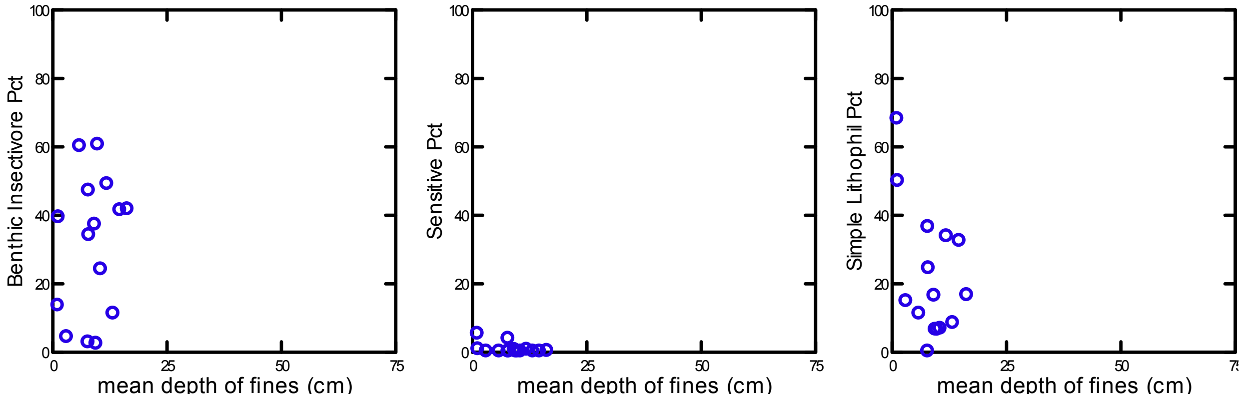

Figure 45. Three positive biological metrics (Benthic Insectivore %, Sensitive %, and Simple Lithophil %) plotted against Mean Depth of Fines. Data from MPCA biological monitoring sites in Little Rock Creek.

Mean Depth of Fines in Little Rock Creek ranged from 1.2 to 16.5 cm, averaging 8.9 cm across the 15 site visits (Table 11). The site with the lowest amount of fine substrate (Station 7) also had the highest percentage of Simple Lithophils and Sensitive individuals, suggesting that improving sediment conditions in Little Rock Creek may have a positive influence on the biological community. While it is unclear how much of the accumulated sediment in Little Rock Creek may be attributed to anthropogenic disturbance, the observation of buried coarse substrates at most sites, coupled with anecdotal accounts from local residents regarding the stream's former state, only reinforce the inference that biological condition in Little Rock Creek is likely negatively affected by the accumulation of fine sediments.

Table 11. Mean Depth of Fines and relevant biological metrics from MPCA biomonitoring sites in Little Rock Creek.

Site	Year	Mean Depth of Fines	Benthic Insectivore Pct	Sensitive Pct	Simple Lithophil Pct
2	1999	9.7	2.3	0.0	6.4
2	2007	3.2	4.2	0.0	14.7
2	2007	8.0	2.7	0.0	0.0
5	2007	10.7	24.0	0.0	6.7
5	2008	13.4	11.1	0.0	8.3
6	2008	6.1	60.0	0.0	11.1
7	2007	1.3	13.5	5.2	68.0
7	2008	1.4	39.2	0.7	49.8
8	2008	8.0	47.0	3.8	36.4
9	2007	16.5	41.6	0.2	16.5
9	2008	9.4	37.1	0.5	16.3
11	2008	12.1	48.9	0.5	33.7
12	2007	8.1	34.0	0.0	24.3
12	2008	10.0	60.4	0.0	6.3
13	2008	14.9	41.3	0.0	32.3

NRRI – Study site summary

The Natural Resources Research Institute collected habitat data in 2006 at stations 6, 7, 9, and 13. Stream habitat was characterized at base flow conditions, and channel structure at all sites was incised, although bank structures were shallow (~1 meter in depth, unpublished data).

Minimal differences between wetted-width and bank-full width measurements at most transects (Table 12) resulted in step-bank structure (59% of bank observations along transects approaching a 90 degree slope). Bank structure typically showed signs of eroding or slumping. Undercut banks exposing root wad structures were recorded at approximately 33% of the transect bank descriptions and were common at all sites except station 9. At base flow conditions, the Little Rock Creek TMDL sites followed a typical up-stream to down-stream trend, with mean depth and flow increasing further downstream (Table 12), which is counter to the bankfull estimated discharges in Table 10. The riparian zone at all TMDL sample sites was mixed vegetation including grass, shrubs, and mixed forest. Land cover adjacent to the riparian zone (beyond ~30 meters) was mainly mixed vegetation (grass, shrub, and deciduous and coniferous trees), with the exception of station 7, where the land cover beyond the riparian zone was predominantly agricultural row crop (Table 13).

Table 12. Physical characteristics of LRC, MPCAA, and NRRI regional comparison streams

Data are presented as mean values. Stream name and site abbreviation (for Little Rock Creek sites) correspond to Figure 1b. Sites from the current study (LRC-TMDL) are in blue. "Shade (%)"is expressed as a mean percentage of canopy cover at the center of the stream channel. Woody debris is the proportion of a sample transect covered by wood material suitable as fish habitat (see MPCA protocols). Little Rock Creek TMDL woody debris is expressed as a mean proportion of transect surface area covered with large wood.

Stream-site	Wetted width (m)	Depth (m)	Flow (m/s)	Temp [C]	Shade (%)	Woody debris (%)
TMDL Sites						
LRC Site 13	7.04	0.16	0.151	19.5	29.6	3.7
LRC Site 9	3.60	0.13	0.148	19.5	71.7	1.5
LRC Site 7	5.17	0.11	0.054	18.8	79.6	1.5
LRC Site 6	2.85	0.12	0.051	22.1	50.4	0.6
LRC Site Z1	3.13	0.15	0.056	18.7	60.8	4.5
MPCA regional comparison sites						
Bogus Brook	6.3	0.449	0.15	21.5	20.36	6.9
Briggs Creek	6.05	0.376	0.12	17.4	3.05	4.6
Browns Creek	3.12	0.204	0.05	17.5	78.85	8.9
County Ditch # Twelve	3.19	0.274	0.06	22.6	51.24	0
Groundhouse River	4.70	0.305	0	18.0	12.10	4.3
Hay Creek	4.40	0.219	0.23	16.8	54.20	3.5
Knife River (Dry Run)	8.14	0.402	0	21.8	0	0
Little Rock Creek-99	3.07	0.465	0.07	20.9	81.11	23.5
Mayhew Creek	4.41	0.424	0.10	20.0	4.19	0
Mike Drew Brook	2.82	0.564	0.02	27.0	1.47	3.1
Moose Horn River	4.20	0.477	0.01	19.8	3.05	8.6
Skunk River	3.36	0.541	0.16	21.4	0.68	3.6
South Fork Watab River	4.31	0.483	0.23	22.4	22.85	2.7
trib. to Sauk River	2.81	0.406	0.04	18.5	31.00	1.5
West Fork Crooked Creek	5.70	0.476	0.12	22.4	1.26	1.1
West Fork Groundhouse River	7.24	0.397	0	30.08	2.26	7.1
NRRI regional comparison sites						
Crane Creek A	3.80	0.28	0.01	10.00	0	0.00
Crane Creek B	3.95	0.36	0.03	11.11	0	0.00
Fall Creek	3.60	0.45	0.05	10.00	91.6	3.47
Headwaters North	4.61	0.43	0.13	7.78	34.2	0.13

Headwaters South	6.28	0.25	0.10	10.00	0	0.00
Lesuer River A	4.97	0.27	0.05	10.00	0	0.12
Lesuer River B	4.08	0.26	0.00	10.00	58.8	1.32
Little Lesuer River	3.44	0.39	0.07	10.00	10.8	0.17
Maple Creek	7.52	0.50	0.01	7.78	0	0.45
McKenzie Creek	5.17	0.22	0.07	10.00	89.4	0.19
Medford Creek	3.24	0.37	0.10	6.67	73.6	0.59
Mud Creek	4.33	0.40	0.03	12.22	0	0.95
North Zumbro A	4.43	0.22	0.15	10.00	4.2	0.00
North Zumbro B	4.49	0.26	0.05	10.00	5.8	0.00
Rush Creek B	9.98	0.46	0.06	10.00	18.2	0.39
Straight River East	4.92	0.21	0.15	7.78	0	0.00
Straight River West	4.50	0.35	0.12	8.89	0	0.00
Turtle Creek	5.64	0.39	0.05		0	0.00

Table 13. LRC study site riparian habitats for the TMDL sampling locations

Riparian observations include a buffer from 0-30 m from the stream bank. Entries divided by a "/" indicate dominant conditions separately for each bank; otherwise the banks are similar. Riparian zones defined as 'MixFor' include a deciduous/conifer mixture. Bank substrate and substrate percent are the percent of the dominant substrate as total bank surface area along transects. Adjacent landuse is that adjacent to the riparian zone. QHEI score was calculated without including the gradient component, worth 10 pts. Undercut bank is the percent occurrence of undercut banks on transects. Amount of organic matter in sediments is expressed as grams dry weight after ashing (Organic). Large woody debris (LWD) are expressed as meter length counts per reach.

Site	Bank substr	Substr %	Ripn zone	Ripn width (m)	Adj landuse	QHEI score*	Under bank (%)	Organic (g)	LWD
13	Sand	74	Grass/Mix For	>50	Forest	46	45	1.9	261
9	Silt	49	Grass/Mix For	>50	Forest	42	0	2.0	53
7	Sand	45	Grass/Mix For	25/40	Agr	53	65	3.9	74
6	Sand	55	Grass/Mix For	>50	Forest	47	10	8.0	17
Z1	Sand	78	Grass/Mix For	>50	Forest	45	55	1.2	144

Large wood (LWD) is an important invertebrate habitat and fish refuge (Wallace et al. 1993), and can serve as the only source of stable substrate in some settings (c.f., Johnson et al. 2003). Occurrence of large wood (as a proportion of transect surface area) from the four Little Rock Creek sample sites followed an upstream to downstream trend, with LWD increasing downstream. Z1 and Station 13 contained the largest accumulation of large wood, both across transects and within a reach (Tables 12 and 13). Canopy cover values measured with a densitometer typically decrease in the downstream direction as systems generally become wider further downstream. However, stream shading is often site-specific and heavily influenced by local landuse characteristics and stream width. Sample sites in this study were more shaded at Stations 7 and 9, with the upper site and the furthest downstream site containing the least amount of stream shade (Table 12). Z1 was intermediate in extent of shade with respect to LRC sites.

In-stream stable refugia in the form of hard substrates larger than gravel was not readily available throughout the sample locations visited. Cobble and gravel riffle/run areas were periodically exposed, and those reaches received a concentrated sampling effort. The substrate at Little Rock Creek sites was heavily dominated by fine substrates (sands, silts, and clay), with the exception of station 7, which contained a substantial deposit of cobble-size particles (Table 14).

Most sample locations contained cobble and gravel-size particles that were buried by a substantial layer of sand and silt. These particles were detected while probing along transects for fine sediment depth. Where larger substrates were uncovered, substrate embeddedness was 20 to 25% at upstream sites and 50% at station 13, the most downstream site in Little Rock Creek; substrate embeddedness averaged 30% on Zuleger Creek (Table 14).

Table 14. Substrate characteristics of TMDL and MPCA regional comparison stream sites

Measurements from the TMDL study are shown in blue. Substrates were characterized as bedrock (bed), boulder (bldr), cobble (cbl), pebble (pbl), sand, and silt and clay (st/cl) and are expressed as percents. Total fines (Tfines) is the sum of percents of sand, silt, and clay. Depth of fines is the depth of fine sediments in slow current areas. Embeddedness is the extent to which large substrates (boulders to pebbles or wood) are surrounded by fine substrates.

Site	Bed (%)	Bldr (%)	Cbl (%)	Pbl (%)	Sand (%)	St/cl (%)	Tfines (%)	Depth fines (m)	Embed (%)
TMDL sites									
13	0	0	3	0	72	25	97	0.36	50
9	0	3	7	0	50	40	90	0.2	25
7	0	0	**55**	**8**	**30**	**7**	**37**	**0.07**	**19.7**
6	0	0	0	0	44	56	100	0.23	26.8
Z1	0	0	0	0	78	22	100	0.35	32.1
MPCA regional comparison sites									
Bogus Brook	0	2	17	6	6	62	68	0.11	53.8
Briggs Creek	0	0	0	0	27	52	79	0.32	-
Browns Creek	0	0	0	31	48	21	69	0.15	24
County Ditch # Twelve	0	10	4	23	31	33	64	0.09	51.3
Groundhouse River	0	4	14	4	59	18	77	0.09	25
Hay Creek	0	0	0	16	78	5	83	0.12	94.4
Knife River (Dry Run)	0	0	0	0	0	100	100	0.16	-
Little Rock Creek-99	0	0	13	2	62	12	74	0.1	20
Mayhew Creek	0	2	2	2	79	15	94	0.22	75
Mike Drew Brook	0	0	0	0	38	47	85	0.17	-
Moose Horn River	0	0	0	0	37	23	60	0.29	-
Skunk River	0	0	0	0	41	10	51	0.26	87.5
South Fork Watab River	0	0	0	13	35	33	68	0.22	35.7
trib. to Sauk River	0	0	10	10	33	46	79	0.14	75
West Fork Crooked Creek	0	0	0	0	27	73	100	1.45	-
West Fork Groundhouse River	0	2	0	2	40	44	84	0.2	62.5
NRRI regional comparison sites									
Crane Creek A	0	20	4	50	26	76	0		0.0
Crane Creek B	4	41	15	39	2	41	4		6.7
Fall Creek	0	34	26	6	34	40	0		44.6
Headwaters North	0	62	10	2	26	28	0		0.0
Headwaters South	6	84	0	2	8	10	6		0.0
Lesuer River A	0	54	4	14	28	42	0		24.0
Lesuer River B	4	92	0	2	2	4	4		42.0
Little Lesuer River	0	0	0	0	100	100	0		0.0
Maple Creek	0	8	0	8	84	92	0		0.0
McKenzie Creek	20	64	0	16	0	16	20		10.0
Medford Creek	0	0	0	92	8	100	0		16.0
Mud Creek	6	70	0	8	16	24	6		46.0

North Zumbro A	0	78	10	8	4	12	0	72.0
North Zumbro B	0	0	74	4	22	26	0	30.3
Rush Creek B	0	42	4	0	54	54	0	38.0
Straight River East	2	64	0	12	22	34	2	2.0
Straight River West	0	0	0	92	8	100	0	42.0
Turtle Creek	4	40	19	23	6	29	4	0.0

Station 7 contained significantly greater amounts of >4 mm, 4-2 mm, and 2-1 mm particle size fractions than were found at all other TMDL sample locations (Table 15). This was the only site with substrate not dominated by sand. Within the 1-0.5 mm and smaller fractions, all sites except station 7 contained significantly more fine sediments than other size classes. Station 7 had only one quarter the amount of fine sand (<0.25 mm size fraction) as the other sites.

Table 15. LRC fine substrate particle distribution and organic content at TMDL sample sites

Values are means followed by 1 standard error. Sediment organic content is measured as AFDW = ash free dry weight in grams.

Site	Substrate AFDW (g)	> 4 mm	4-2 mm	2-1 mm	1-0.5 mm	0.5-0.25 mm	0.25-0.063 mm	< 0.063 mm
13	1.9 (0.12)	101.9 (6.93)	7.3 (0.12)	3.4 (0.18)	14.3 (0.17)	197.8 (5.41)	113.4 (3.52)	0.9 (0.13)
9	2.0 (0.01)	146.2 (34.2)	35.5 (2.31)	20.5 (0.69)	31.7 (2.67)	116.0 (11.40)	116.6 (11.40)	1.9 (0.26)
7	3.9 (1.38)	264.7 (15.1)	56.4 (6.11)	35.1 (2.56)	23.8 (0.92)	63.6 (1.86)	29.6 (2.18)	0.7 (0.01)
6	8.0 (1.25)	0.4 (0.06)	1.17 (0.15)	3.5 (0.40)	22.5 (3.74)	149.0 (21.80)	111.7 (8.59)	5.1 (0.81)
Z1	1.2 (0.06)	3.03 (0.06)	21.5 (1.21)	39.9 (0.56)	82.9 (1.54)	182.0 (1.28)	102.7 (1.22)	2.4 (0.23)

Macroinvertebrates - Total number of invertebrate taxa per site for all habitats was highest (109 taxa) at the furthest downstream Little Rock Creek station 13. The fewest taxa per site (61 taxa) were recorded at the most upstream sample location (site 6), significantly fewer taxa than collected at station 13 ($p = 0.02$) (Figure 46a). Sample locations 9 and 7 included 93 and 99 total taxa, respectively. Mean number of macroinvertebrate taxa per gear-type followed a similar pattern, with station 13 having the highest average taxa richness. Again, the lowest average number of taxa occurred at station, 6, and was similar to that associated with the Zuleger Creek site. Zuleger Creek (Z1), a third order stream located near the confluence with Little Rock Lake, is in closer proximity to station 13, but similar to the site 4 in terms of watershed size (Table 16).

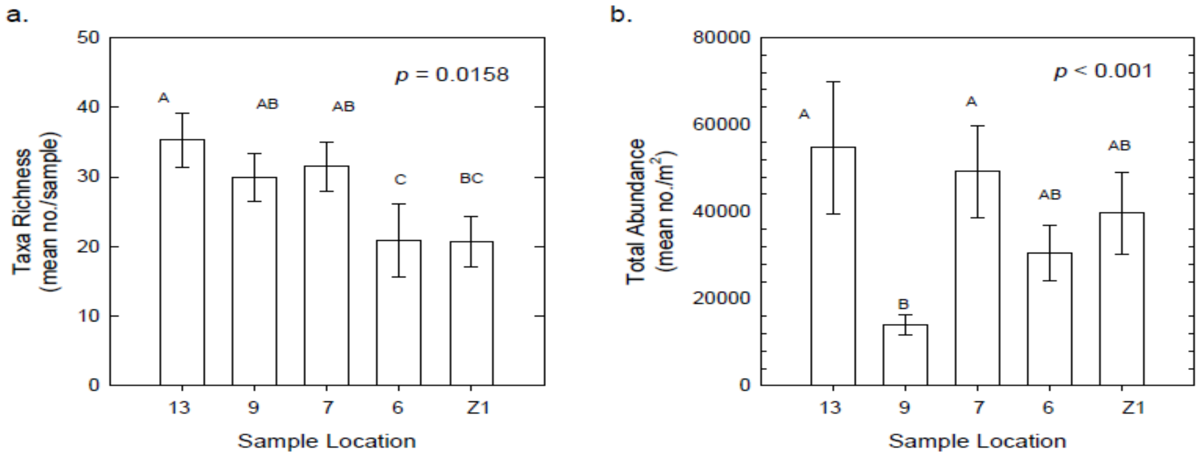

Figure 46. a) Macroinvertebrate taxa richness (combining all gear types), b) mean (#/m2) total abundance of individuals (qualitative core and Hess devices)[1]

Table 16. Land use/land cover characteristics of watersheds, 50m, and 100m buffers along LRC TMDL study stream and MPCA comparison stream sites

Site	Wshed area (ha)	Wshed Agr	Wshed Forest	Wshed Dev	Wshed Wetland	50 m Agr	50 m Forest	50 m Dev	50 m Wetland	100 m Agr	100 m Forest	100 m Dev	100 m Wetland
Bogus	2535	0.071	0.341	0.080	0.268	0.032	0.402	0.046	0.377	0.030	0.397	0.055	0.362
Briggs	2839	0.316	0.316	0.080	0.194	0.127	0.377	0.053	0.379	0.140	0.410	0.053	0.325
Browns	7126	0.100	0.277	0.103	0.174	0.049	0.324	0.082	0.237	0.050	0.322	0.088	0.218
Ditch12	4016	0.187	0.390	0.055	0.210	0.059	0.471	0.023	0.380	0.069	0.472	0.027	0.350
Ground	7542	0.006	0.745	0.019	0.182	0.009	0.557	0.011	0.369	0.006	0.585	0.014	0.338
HayCrk	4552	0.147	0.235	0.097	0.161	0.077	0.182	0.140	0.171	0.095	0.181	0.110	0.179
Knife	3266	0.043	0.412	0.071	0.291	0.016	0.338	0.058	0.514	0.014	0.346	0.059	0.489
LRC Site 13	18231	0.526	0.126	0.060	0.180	0.259	0.237	0.043	0.341	0.322	0.204	0.050	0.303
LRC Site 9	11664	0.508	0.131	0.061	0.195	0.240	0.234	0.046	0.370	0.306	0.201	0.053	0.328
LRC Site 7	9559	0.485	0.135	0.060	0.202	0.253	0.218	0.047	0.363	0.317	0.188	0.054	0.321
LRC Site 6	5356	0.405	0.151	0.058	0.228	0.191	0.204	0.043	0.393	0.253	0.179	0.048	0.351
LRC Site 2	3559	0.381	0.135	0.059	0.218	0.176	0.166	0.046	0.399	0.241	0.147	0.053	0.346
Mayhew	11591	0.435	0.119	0.074	0.246	0.176	0.182	0.051	0.468	0.228	0.175	0.057	0.416
MikeDrew	2482	0.020	0.571	0.038	0.191	0.004	0.607	0.022	0.295	0.007	0.616	0.023	0.274
MooseHorn	5665	0.005	0.772	0.040	0.093	0.000	0.728	0.031	0.124	0.000	0.740	0.036	0.109
SaukTrib	1575	0.616	0.053	0.059	0.158	0.552	0.058	0.022	0.268	0.591	0.061	0.028	0.221
SFWatab	12672	0.169	0.363	0.099	0.175	0.098	0.322	0.079	0.306	0.117	0.316	0.087	0.279
Skunk	941	0.009	0.536	0.033	0.314	0.000	0.324	0.034	0.545	0.000	0.391	0.027	0.473
WFCrooked	3400	0.000	0.891	0.006	0.068	0.000	0.760	0.016	0.175	0.000	0.775	0.020	0.158
WFGround	3272	0.003	0.744	0.015	0.192	0.006	0.532	0.010	0.417	0.005	0.574	0.012	0.373
LRC Site Z1	4833	0.537	0.154	0.066	0.163	0.350	0.256	0.049	0.277	0.402	0.215	0.056	0.253

Individual taxa counts from quantitative gear types (Hess and sediment cores) were standardized by surface area to compare abundance values by habitat and sites. Total abundance (in mean number per m2) was highest at stations 13 and 7, and significantly greater than station 9 (Figure 46b). Station 9 contained the lowest abundance. Mean abundances associated with stations 6 and Z1 were not significantly different from other sample locations.

Trait descriptions are a composite measure of taxa behaviors or characteristics that help smooth out the variability of taxa interactions, and help interpret the broader community's response to the environment. Dominant behaviors and life history characteristics can be used to infer mechanistic relationships between stressors and responses in stream ecosystems (Richards et al.

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[1] Value p is from the overall ANOVA. Sample locations with the same letter are not significantly different.

1997). For example, one species of macroinvertebrate that scrapes substrate surfaces to obtain food (typically periphyton) may not provide a robust picture of stream condition, but in combination with other grazers, a shift from one dominant feeding process to another can be identified.

A strong association exists between the Little Rock Creek invertebrate community and the relative abundance of cobble or gravel substrate at sites. Grazer-scraper numbers were highest at site 7, the only site with substantial rocky substrate, and were very low at the remaining sites (Figure 47). A close association between stream substrate and the macroinvertebrate community can be seen in both mayfly and riffle beetle numbers. Mayflies (Ephemeroptera) responded to the hard substrate associated with site 7 with significantly greater numbers than were found at sites 9 and 6 (Figure 48a). Mean abundances of mayflies at the remaining stations 13 and Z1, were not significantly different than stations 9 and 6. Although larval riffle beetles (Elmidae: Coleoptera) function efficiently in a variety of habitats including fine sediments, adult beetles have been shown to select for particular substrates (c.f., Minshall 1984). Both adult and larval numbers were significantly greater at station 7 than at any other sample site (Figure 48b). These data suggest that riffle beetle numbers are also responding to the dominate cobble substrate at station 7. There was little difference in riffle beetle abundances among the remaining sites, although the abundance of beetles at station 9 was significantly lower than at stations 13 and 6. Beetle abundance in Z1 was not significantly different than any station except 7.

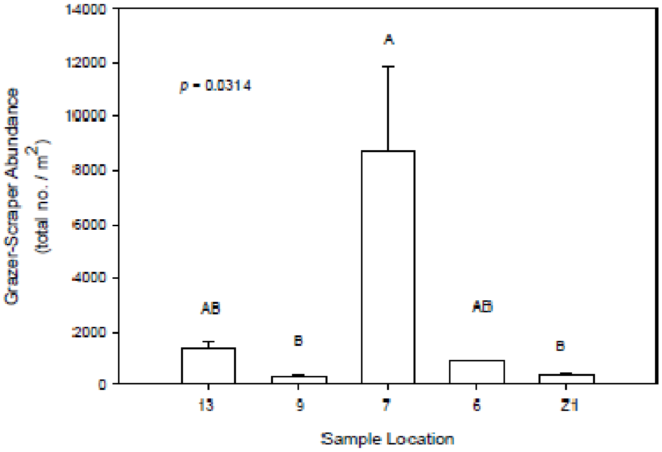

Figure 47. Total abundance (#/m2) of macroinvertebrates classified as grazer-scraper feeders associated with LRC TMDL samples locations.[2]

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[2] Value p is from the overall ANOVA. Sample locations with the same letter are not significantly different.

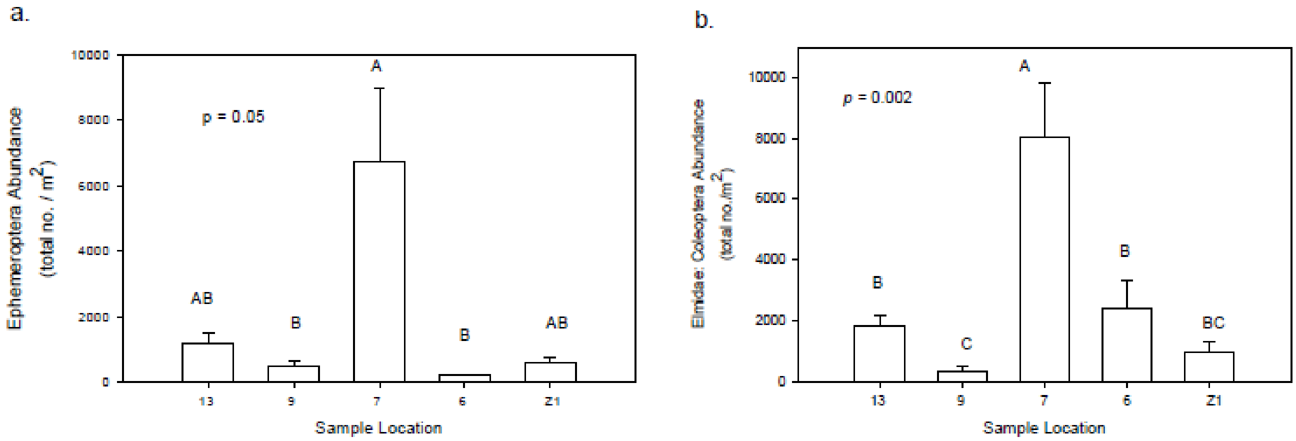

Figure 48. a) Total abundance (#/m2) of Ephemeroptera, and b) Elmidae (Coleoptera) associated with LRC TMDL sample locations (qualitative core and Hess devices)

Burrowing invertebrates are often associated with habitats dominated by fine, organic and inorganic sediments, and thus their abundance and taxonomic diversity are indicative of invertebrate community responses to substrate type at these stations (Table 14). Modifications to body structures include spade-shaped legs or tusks to better move sediment or cover plates over fragile gill appendages to reduce abrasion; these structures enable this group of organisms to more effectively utilize soft sand and silt. Burrower abundance was lowest at station 9, the only station dominated by rocky substrate, significantly increased at station 7, and then significantly increased again at station 6 (Figure 49). Burrower abundances observed at station 13 were not significantly different than at station 6 or Z1. Climbing invertebrates were also lowest in number at station 9, and significantly lower than stations 13 and 6. All Little Rock Creek sites had climber abundances significantly lower than the Z1 sample location (data not shown).

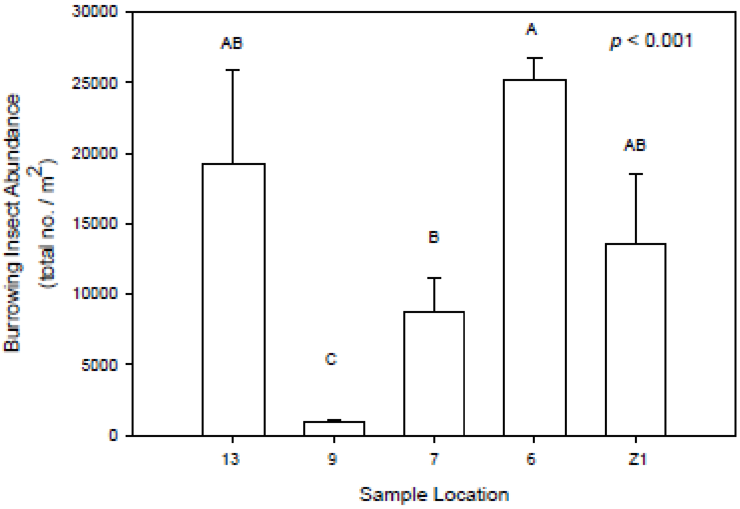

Figure 49. Total abundance (#/m2) of macroinvertebrates classified as burrowers associated with LRC TMDL sample locations (qualitative core and Hess devices)[3]

Summary of TMDL site conditions and biota;

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[3] Value p is from the overall ANOVA. Sample locations with the same letter are not significantly different.

- All sample locations are within watersheds with approximately 50% agricultural row crops; riparian corridors were relatively intact at all sample sites.
- The riparian corridor at station 7 was the narrowest of all the sites, but the stream reach associated with station 7 was heavily shaded, with scattered amounts of large woody debris.
- Station 7 contained more cobble substrate (55%), less fine sediment, and was less embedded (19.7%) than any other TMDL site sampled.
- The macroinvertebrate assemblage responded positively to local habitat characteristics, particularly the rocky substrate, at station 7, resulting in a more heterogeneous community there than was present at other sample locations.
- HBI scores ranked station 7 as 'Fair' and the other TMDL stations as 'Poor' condition.

TMDL Sites Compared to MPCA and NRRI Regional Comparison Datasets

Stream data from sixteen watersheds with relatively similar geologies and in close proximity to Little Rock Creek were provided by the MPCA Surface Water Monitoring Division (Figure 50). Sampling protocols varied between efforts, but the comparisons provided below were generated based on combining macroinvertebrate samples from all habitat types so that TMDL data could be directly compared to the MPCA data. In most cases, data were collected during different years (1999-2006), but were relatively consistent in terms of seasonality (late August through early October). The NRRI regional comparison samples (n=18 sites) were collected in 1998 from a heavily agricultural area in southeastern Minnesota (Figure 51). Sampling methods were very similar to those used for the LRC TMDL samples, thus reducing methodological issues with the data comparison. These samples were collected from streams covering a wide variety of substrate types that were located in a predominantly agricultural setting, making them a good comparison dataset for the TMDL samples. All data were transformed prior to analysis or standardized as percent of total numbers, if necessary, when making direct comparisons.

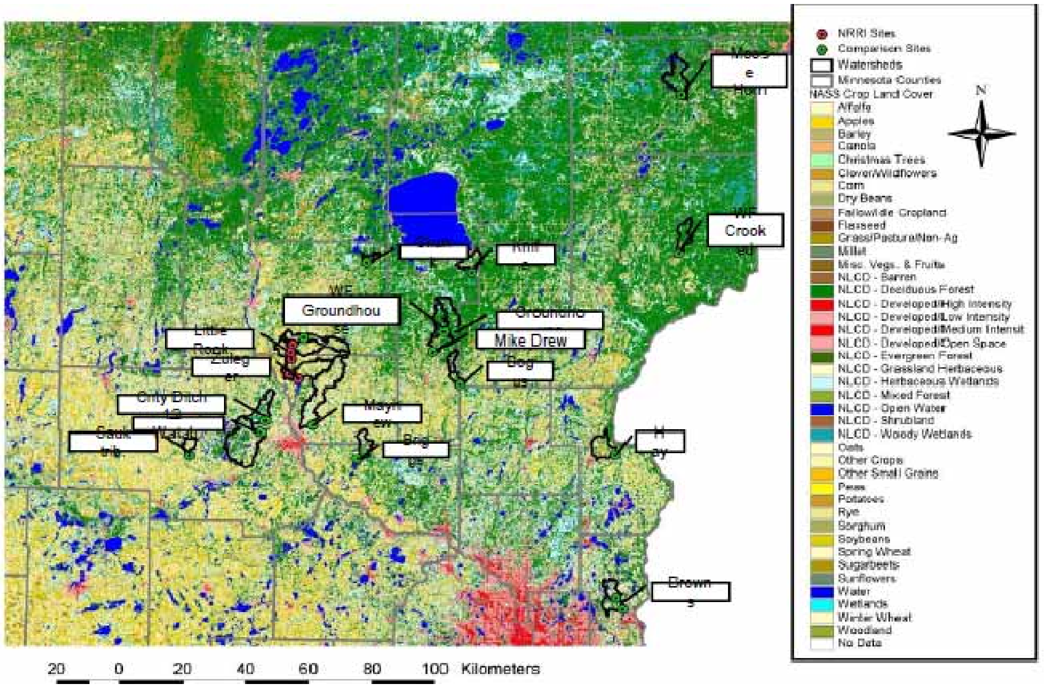

Figure 50. MPCA regional comparison stream sites & TMDL study sites plotted on a land cover map

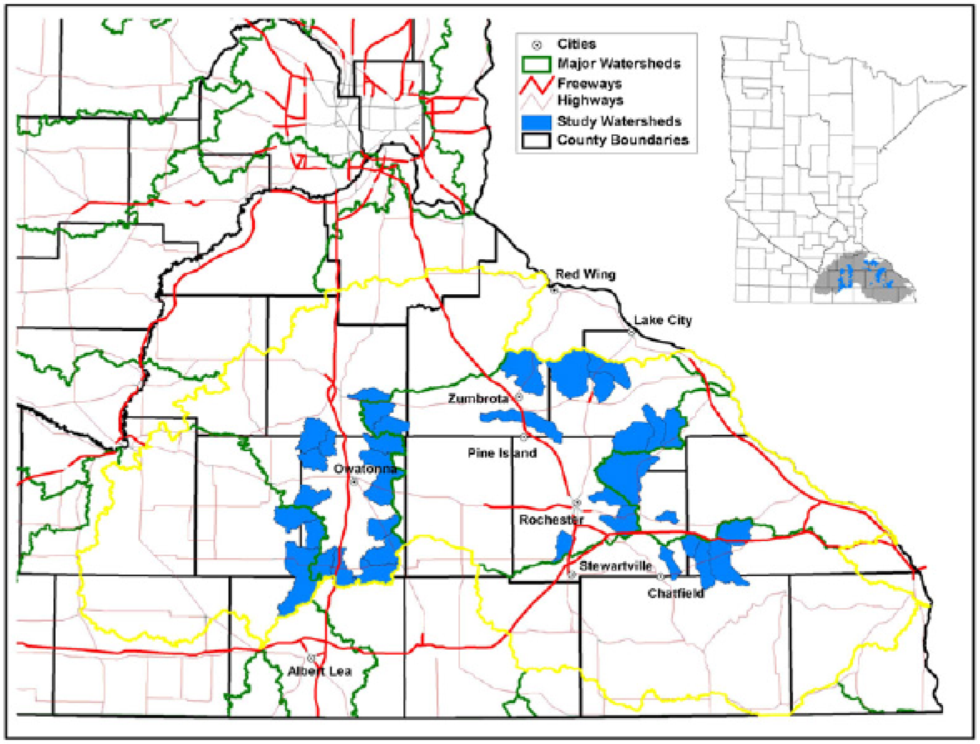

Figure 51. NRRI regional comparison sites in a heavily agricultural region of SE MN.

Watershed landuse/land cover - Landuse among the MPCA regional comparison site watersheds was less dominated by agriculture compared to the TMDL sites for all but the tributaries of the Sauk River and Mayhew; agriculture was 62% and 43%, respectively in those systems, and ranged from 0 to 31% in the other regional comparison sites (Table 16). The Little Rock Creek site sampled by MPCA in 1999 (station 2) was located in a watershed with 38% agricultural land use. Across the MPCA regional comparison watersheds, urban development ranged from 0.5 to 10%; forest land cover ranged from 6 to 89%. Land use within a 100 meter buffer surrounding the stream was also summarized. In general, these data paralleled the results for each watershed with the exception that herbaceous wetlands were a prominent feature of the 100 meter buffers, ranging from 11 to 49% (Table 16). The TMDL sites had watersheds comprised of at least 50% agricultural land use (Table 16). NRRI regional comparison sites, in contrast, had even higher amounts of agriculture in their watersheds, making the TMDL sites among the least agricultural of the watersheds sampled. NRRI sites had watersheds with >80% agricultural land use.

Local habitat conditions - Low water levels over the course of the study likely affected width-depth ratios for Little Rock Creek, because reaches sampled in this study are clearly different from both regional comparison datasets sampled in earlier years (Figure 52 a, b). Wetted widths were not a major contributor to the intense difference, but water depths at the TMDL sample locations were typically only 20 cm deep, while both regional comparison dataset streams had low flow depths of 20-60 cm. With a few exceptions, low flow rate and water temperatures were fairly similar among all sites, as were estimated proportions of woody debris per reach (Table 12). Stream shade (percent canopy cover) is often site-specific and was highly variable (range 0 to >80%) among sites, with no strong relationships apparent.

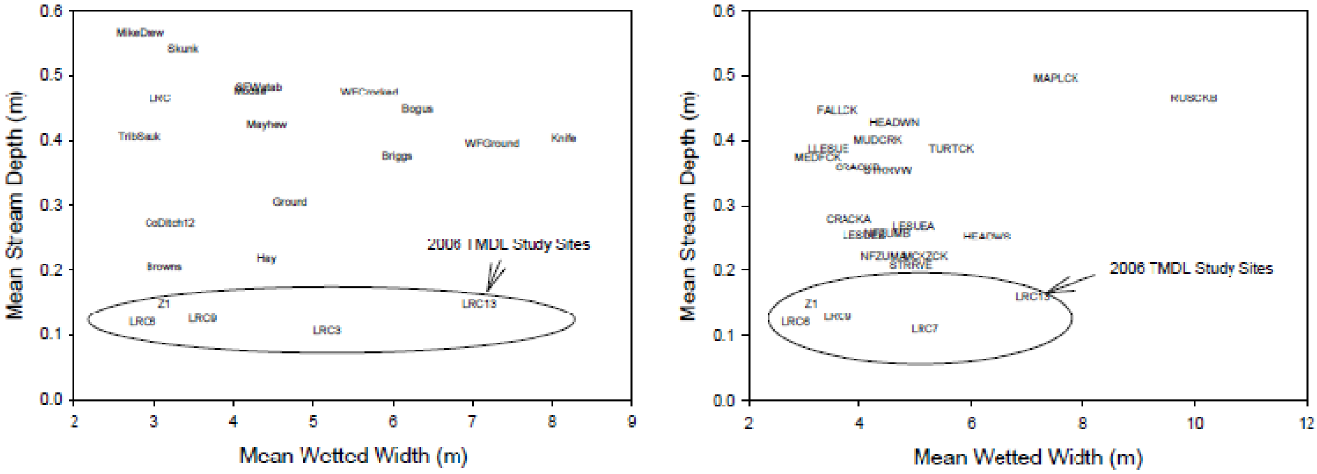

Figure 52. Ratio of stream width to water depth for LRC TMDL sample locations compared to (a) MPCA regional comparison stream sites and (b) NRRI regional comparison sites.

About half of the sites from the regional comparison datasets contained discernable amounts of large rock substrate (e.g., > 50% of boulder, cobble, and gravel (Table 15). However, with the exception of hard substrate associated with station 7, TMDL sites had among the highest amounts of total fine sediment accumulation (summed percent of sand, silt, and clay. TMDL sites had on average nearly 20% greater amounts of fine sediments than did sites associated with the 16 MPCA regional comparison streams (Table 14). With station 7 included, TMDL study sites still contained 10% more total fines than were found in the MPCA comparison streams. Five of the NRRI regional comparison sites contained comparable amounts of fine sediment to the TMDL sites. Mean total depth of fine sediments was similar for all sites, but the extent hard substrates was embedded by fine particle was variable. This was in part due to the embeddedness observation, which requires that hard substrate particles be visible for an embeddedness estimate to be made.

Embeddedness in stream reaches naturally dominated by coarse sand may not place the same adaptive pressures on a biological community as sites with intense pulses of fine sediment filling in hard substrate and quickly altering condition. For example, station 6 and Z1 were 26% and 32% embedded, respectively (Table 14), but no substrate particles larger than sand were recorded. Thus, the embedded variable in this case represents sediment deposits around large woody debris. Wood deposits were sometimes the only hard substrate in a reach, so comparisons to sites with traditional hard substrate would be very difficult. The dominant substrate particles at station 7 (cobbles) were estimated to be 19% embedded, on average, with all five TMDL sites containing substrates that were, on average, 30% embedded. Extent of embedded substrate at comparison sites ranged from 20 to 94%.

Fish assemblage - In addition to the station 6, 7, 9, 13 sample locations, fish assemblages were also collected by MN DNR during the TMDL study (Table 17) at Bunker Hill (BH2) and Zuleger Creeks (Z1 and Z2). Total fish taxa varied among TMDL sites (6-14 taxa at station 13 and Z1, respectively), with a wider range of taxa found at the regional comparison sites (e.g., 6-30 taxa). Average number of fish taxa among the TMDL sample locations was somewhat lower than the regional comparison sites, with means of 10.1 and 14.4, respectively. Taxa richness at comparison sites with a similar proportion of agricultural landuse in the watershed (previously described as Briggs, LRC-99, Mayhew, and Sauk River Tributary) were more similar, with a mean of 12.5 taxa at those sites.

Table 17. Fish community traits from LRC TMDL sample locations and MPCA regional comparison streams

Bunker Hill (BH2) and Zuleger Creek (Z1 and Z2) were additional Minnesota DNR fish sampling locations for the TMDL study, and data for regional comparison sites were provided by the MPCA. Tolerant taxa are based on an EPA tolerance score and ranked as ‘tolerant,’ ‘intolerant,’ or ‘intermediate.’

Site	Total taxa	Native taxa (%)	Tolerant taxa (%)	Total abundance	Omnivore abundance (%)
TMDL Sites					
BH2	10	60	30	205	72
13	6	66	33	60	33
9	9	77	33	253	41
7	13	84	30	341	52
6	10	90	30	466	81
Z2	9	66	44	378	44
Z1	14	71	28	565	68
MPCA regional comparison sites					
Bogus Brook	14	78	35	195	56
Briggs Creek	9	55	33	167	64
Browns Creek	22	81	50	1054	99
County Ditch # Twelve	16	75	25	427	43
Groundhouse River	18	61	22	1116	34
Hay Creek	30	76	26	317	49
Knife River (Dry Run)	14	50	28	1866	86
Little Rock Creek-99	16	81	43	752	88
Mayhew Creek	14	57	21	277	46
Mike Drew Brook	6	50	33	313	53
Moose Horn River	9	55	22	252	37
Skunk River	21	66	42	38	42
South Fork Watab River	9	77	11	206	47
trib. to Sauk River	11	81	45	131	81
West Fork Crooked Creek	10	60	20	122	54
West Fork Groundhouse River	12	58	33	371	43

EPA tolerance rankings of ‘tolerant taxa’ (Table 17) as a percent of total taxa were, on average, 30.9 at the MPCA sites and 32.9 at the TMDL sample locations. Fish species classified as omnivores are also an indicator of tolerance, and the MPCA sites had a mean percent abundance of 58% compared to a mean of 46% at the TMDL sample locations.

A multivariate procedure exploring the relationships between fish assemblage traits and local habitat variables indicated a significant relationship ($p = 0.04$). The first ordination axis was most highly correlated with shade (percent) and cobble (percent), and was negatively associated with water depth (Figure 53). The second axis was positively correlated with wetted width, and negatively correlated with low flow. The ordination clustered TMDL stations 13 and 9 with respect to channel characteristics associated with increasing shade (percent) and cobble substrate, while the other TMDL sites were most associated with decreasing flow. The regional comparison sites with the highest proportion of agricultural landuse in the watershed (Briggs, site 2, Mayhew, Sauk River Tributary) were associated with deeper, more turbid sites, with more wood, and decreasing flow (Figure 53).

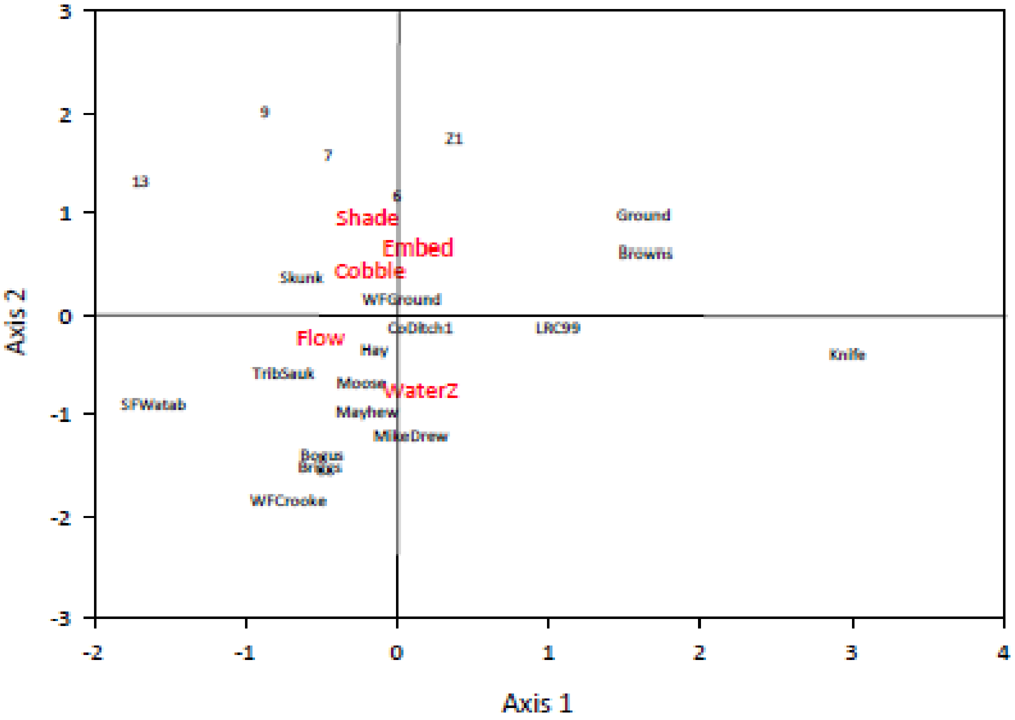

Figure 53. Multivariate bi-plot of fish community metrics with vectors from local habitat variables for LRC TMDL samples locations and regional sites.[4]

Ordination of fish species traits and summary metrics suggests that abundance of intolerant and exotic taxa was associated with cobble substrate, higher percent shade, and embedded substrates (data not shown). Sites with these characteristics were TMDL stations 9 and 13. In contrast, tolerant taxa abundance was associated with deeper, more turbid sites with large wood (i.e., regional comparison sites with relatively high levels of agricultural land use). The presence of exotic fish species in conjunction with intolerant taxa in an environment with embedded cobble substrates is suggestive of conditions that are either beginning to degrade or are undergoing recovery. The fact that these sites are segregated from the regional comparison sites with similar land use patterns suggests that habitat conditions are fundamentally different between the two sets of sites, either due to a local disturbance or landform differences that we have not accounted for in our analyses. Potential factors include differences in landform and/or quaternary sediments. Such factors have been found to play a large role in controlling hydrologic regime and water chemistry (Richards et al. 1996, 1997, Johnson et al. 1997, Hutchens et al. 2009).

Macroinvertebrates - Macroinvertebrate data from TMDL sites incorporated all habitats sampled to better match the MPCA regional comparison dataset. Despite this, we found large differences in taxa richness between the two studies, suggesting that sample processing procedures created systematic differences in the data. For example, MPCA protocols include a multi-habitat

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[4] Relationships are significant (redundancy analysis using a Monte Carlo permutation p=0.0400 in CANOCO), with the relative influence of local habitat characteristics explained on the first two ordination axes (Axis 1 = 58.3% var.).

composite D-net sampling effort per stream reach, followed by a 300-individual count. TMDL survey sample methods involved collecting macroinvertebrates from all dominant stream habitats using several gear (D-nets, Hess samplers, and core tubes), and much larger quantities of the sample were processed (typically at least 25% of each sample). Metrics including mean number of taxa and percent EPT taxa were much lower for MPCA sampled sites than for the TMDL study sites (Table 18). An example of the result of this difference in processing can be seen by comparing the taxa richness and taxa metrics for MPCA Little Rock Creek station 2 and TMDL stations 13 through 6 (Table 18). Richness was much lower at station 2, and the metrics typically would be interpreted as the site having poorer condition. Large differences in taxa richness seem likely to be caused by the differences in sample processing, and not field collection. Both field methods target multiple habitats and the MPCA field method should result in a greater number of samples collected prior to the composite step. In addition, taxonomic resolution appears comparable between datasets (for example, Chironomidae: Diptera are identified to genus in both datasets).

Table 18. Macroinvertebrate metrics calculated from all invert sample types collected from TMDL and NRRI regional comparison sites

MPCA regional comparison streams consist of D-net data gathered from all-habitat types. See Methods for complete description. TMDL study sites are in blue. "# Taxa" indicates total richness count, as does insect and EPT taxa categories. Relative abundance of EPT taxonomic orders is expressed as "%EPT". "Tol score" is site tolerance score. "#Sensit" is number of sensitive taxa. "% Tol" is percentage of tolerant insects in samples (EPA tolerance values ≥7). "%Hydropsych" is proportion of Trichoptera from the family Hydropsychidae. Hilsenhoff biotic index (1982), Hilsenhoff improved biotic index (1987), and Minnesota index of biotic integrity (MIBI) site scores are calculated according to published criteria (Hilsenhoff 1982, Hilsenhoff 1987, Genet and Chirhart 2004).

Stream-site	#Taxa	#Insect taxa	#EPT taxa	% EPT	Tol score	# sensit	% tol	% Hydro-psych	HBI'82	HIBI'87	MIBI
TMDL Sites											
13	84	70	19	5	6	20	57	0.52	4.07	4.95	22
9	78	63	16	37	5	15	31	0.5	4.43	2.79	22
7	90	75	19	17	5	22	38	0.1	3.65	4.41	26
6	85	72	13	2	6	25	44	0.7	5.01	2.97	19
Z1	72	57	14	2	6	18	53	0.17	4.88	4.44	26
MPCA regional comparison sites											
Bogus Brook	49	38	15	52	5	16	25	0.89	3.39	3.6	24
Briggs Creek	52	44	18	15	6	18	15	0.11	2.42	4.73	22
Browns Creek	66	48	16	35	5	33	31	0.32	2.91	3.43	24
County Ditch # Twelve	52	42	13	26	6	18	36	0.08	3.47	1.98	24
Groundhouse River	33	21	3	5	7	15	10	0	1.43	1.15	16
Hay Creek	60	50	15	22	5	27	39	0.81	4.37	3.93	22
Knife River (dry run)	32	24	5	17	7	17	69	0	4.23	6.01	14
Little Rock Creek-99	41	34	11	50	6	12	22	0.81	1.86	4.19	16
Mayhew Creek	19	14	5	4	6	8	6	0	2.91	5.77	12
Mike Drew Brook	41	30	5	9	6	19	27	0.5	3.53	2.29	18
Moose Horn River	51	41	10	7	6	18	49	0	2.89	5.01	24
Skunk River	39	31	5	6	6	15	79	0	4.85	1.6	20
South Fork Watab River	55	46	17	32	5	17	30	0.43	4.35	2.85	26
trib. to Sauk River	33	28	7	28	6	4	32	1	3.23	3.41	20
W. Fork Crooked Creek	45	34	12	12	5	21	48	0.25	3.06	3.41	28

W. Fork Groundhouse R.	51	42	4	27	7	18	47	0.75	1.68	5.12	22
NRRI regional comparison sites											
Crane Creek A	57	51	2	1	3	23	31	0	2.43	6.34	36

Stream-site (cont.)	#Taxa	#Insect taxa	#EPT taxa	% EPT	Tol score	# sensit	% tol	% Hydro-psych	HBI'82	HIBI'87	MIBI
Fall Creek	72	68	13	15	5	21	31	6.4	2.14	4.99	38
Headwaters North	71	67	13	13	5	19	25	0	4.24	5.58	32
Headwaters South	52	47	5	1	5	14	42	0	3.22	6.1	20
Lesuer River A	49	47	10	23	3	15	32	0	2.72	6.85	20
Lesuer River B	44	42	7	1	5	11	73	0	4.45	5.01	30
Little Lesuer River	56	54	13	5	5	12	42	1.13	3.99	5.67	30
Maple Creek	67	60	6	0	5	24	21	0	3.08	5	24
McKenzie Creek	48	45	8	17	4	14	27	2.82	2.41	4.18	24
Medford Creek	61	57	12	13	6	14	37	4.66	3.38	5.48	30
Mud Creek	49	43	6	1	3	12	79	0.1	1.48	6.64	24
North Zumbro A	48	44	7	12	5	11	57	4.71	3.94	5.78	22
North Zumbro B	50	44	10	17	5	15	58	12.08	1.99	4.81	28
Rush Creek B	68	62	16	21	5	20	50	14.71	1.58	4.8	32
Straight River East	57	54	10	7	5	19	65	0	2.55	5.95	26
Straight River West	53	49	8	2	5	19	34	0.76	3.98	5.64	22
Turtle Creek	55	49	12	13	5	18	67	1.01	3.37	6	28

Such methodological differences should not be a concern with the NRRI regional comparison dataset because sampling methods were very similar to those used at LRC TMDL sites. However, to keep the analyses similar to those done with the MPCA regional comparison dataset, we combined all sample (habitat) types for invertebrate analyses because rocky habitats were not common among the sites. In comparison with the NRRI sites, several of the TMDL sites had the highest percent composition of EPT invertebrates. The other two TMDL sites were intermediate among NRRI sites in this respect. EPT abundance and taxa richness are often indicative of higher amounts of habitat space among rockier substrates in streams, but invertebrates may also cling to woody substrate, and a few are adapted to sandy conditions. As is typical for many streams with large amounts of fine substrate or higher amounts of organic material, Diptera (particularly the Chironomidae) dominated sample abundances for almost all NRRI and TMDL sites.

Data conversion to summary indices appears to have reduced the differences created by the two protocols. For example, both Hilsenhoff index scores were similar between datasets, as were other metrics, including the tolerance score and overall MIBI (Table 18). Although a correction factor for early fall sampling was applied (Hilsenhoff 1981), HBI scores ranked station 7 as "Fair" and the other TMDL stations as in 'Poor' conditions. MPCA comparison streams had HBI rankings ranging from 'Excellent' at the Groundhouse River (HBI 1.43) to 'Very Poor' at the Skunk River (HBI 4.85). MPCA regional comparison sites that were similar to the TMDL sites in terms of percent agricultural land use (e.g., Briggs, site 2, Mayhew, and trib to Sauk River) had similar average HIBI'87 values. Averages of HIBI'87 scores for these four regional comparison sites and the five TMDL sites were 4.52 and 4.04, respectively. In general, MIBI scores were similar among sites and, with the exception of Brown's Creek, other metrics, including sensitive taxa and percent of tolerant taxa, were also similar between TMDL and MPCA regional comparison sites (Table 18).

Comparing MIBI scores and metrics between LRC TMDL sites and NRRI regional comparison sites both reduced the variability due to methodological differences and highlighted the impact of high amounts of agricultural land use in the watershed (Table 19). In general, the LRC TMDL sites appear to be in better condition according to these metrics than most of the NRRI regional comparison sites. MIBI scores calculated using all gear types and habitat types are higher, sometimes much higher, than MIBI scores calculated using only D-net samples (Table 20). This is partly because D-nets were used in this methodology only to sample habitats that could not be sampled with one of the other gear types (e.g., undercut banks, overhanging vegetation, wood, leaf packs, etc.). This example helps illustrate why it is important to understand exactly how macroinvertebrate samples were collected and processed when attempting to compare sites that were sampled by different crews and/or methods. Overall, the LRC TMDL stations 13, 9, and 7 generated higher MIBI scores than station 6 and Z1 or any of the NRRI sites.

Table 19. MIBI scores for LRC TMDL sits (blue) and NRRI regional comparison sites, calculated from D-net samples only and from all gear types to illustrate the difference in the calculations

	MIBI Score	
SITE	**D-net Only**	**All gear-types**
Crane Creek A	24	20
Crane Creek B	24	20
Fall Creek	32	30
Headwaters North	16	30
Headwaters South	20	22
Lesuer River A	10	24
Lesuer River B	na	26
Little Lesuer River	22	30
Maple Creek	22	24
McKenzie Creek	18	24
Medford Creek	28	30
Mud Creek	30	24
North Zumbro A	18	22
North Zumbro B	32	28
Rush Creek B	28	32
Straight River East	22	26
Straight River West	18	22
Turtle Creek	20	28
13	22	36
9	22	38
7	26	38
6	19	32
Z1	26	28

Based on Ephemeroptera-Plecoptera-Trichoptera percent abundance (EPT; a metric associated with “healthy” streams), TMDL sites were interspersed among MPCA regional comparison sites, with sites 6 and Z1 on the ‘Poor” condition end and station 9 on the ‘Good’ condition end. The MPCA Little Rock Creek study site (station 2) was upstream of the TMDL station 6 and had similar EPT abundance to station 9, although the distribution of taxa within the EPT metric differed. Station 2 was dominated by Ephemeroptera, while station 9 contained mostly Trichoptera . However, in the context of the more heavily agricultural-area NRRI streams, LRC stations 13, 9, and 7 are among the best with respect to EPT abundance, with station Z1 the only site described on the "Impaired" end of this relative comparison.

TMDL station 9 was among very few sites in either dataset to contain Plecoptera, most likely due to the lack of rocky substrate at the other sites. The abundance of EPT at station 7 was substantially lower than at stations 2 and 9. Study stations 13, 6, and Z1 contained even lower EPT numbers. As EPT abundance declined, midge larvae (Chironomidae: Diptera) tended to increase, along with other taxa. Several of the MPCA regional comparison sites had a large proportion of their community represented by "other" taxa. For example, amphipods represented a large percent of MPCA Briggs and Mayhew Creeks (41 and 56%, respectively). Non-insects also comprised over 30%, 40%, and 80% of the macroinvertebrate population at West Fork Crooked Creek, Mike Drew, and Groundhouse River, respectively.

When comparing metrics generated using tolerance rankings for individual taxa, a site tolerance score using US EPA criteria places the TMDL sites generally in the middle of MPCA regional comparison sites. A similar value quantifying the percent abundance of individuals with an elevated tolerance score (> 7 on a scale of 0 to 10) spreads TMDL sites evenly among MPCA regional comparison sites. Abundance of tolerant insects identified at both downstream locations (stations 13 and Z1) indicates a community more tolerant of disturbance. This observation may be an artifact of the downstream locations both being dominated by fine substrates (which tends to favor burrowing invertebrates), and being influenced by Little Rock Lake. Both of these conditions promote transitional communities that are adapted to deeper water and depositional sediments. Generalist taxa are better adapted to these habitats, and because they are highly adaptable, generalists are often more tolerant of disturbance.

Generalist taxa are typically collector-gatherers in their manner of obtaining food (functional feeding groups or FFGs) because this feeding habit allows adaptation to many different conditions, stream ecologists often consider communities dominated by gatherers, as opposed to more highly specialized predators or more delicate filterers (for example), as more tolerant of adverse conditions. TMDL sites had low to moderate percentages of collector-gatherers compared to most NRRI sites. But the TMDL sites also had few grazers and shredders.
In this case, this occurrence is probably due to the lack of stable grazing surfaces and interstitial spaces that capture and hold leaves, and minimal accumulation of larger organic detritus favored by shredders. Filter-feeding taxa are relatively abundant, however, which is a bit surprising due to the amount of shifting sediment and constant scour activity occurring in these streams. It seems probable that filterers were utilizing woody debris as their stable point of attachment.

Taxa behavioral traits can also indicate the types of conditions to which invertebrates of any particular system are adapted. In the case of streams with high amounts of fine sediments, burrowers may become more dominant, but only if the sediments are relatively stable. Large amounts of shifting sand tend to favor invertebrates that can swim well, sprawl on top of the shifting substrate, or climb on wood or overhanging vegetation. TMDL sites contain low to moderate amounts of burrowers (greatest at stations 13 and 6), with moderate amounts of sprawlers. Climber and swimmer relative abundances are quite low. Clinger invertebrates, which cling to rocks and sometimes wood, make up a large proportion of the invertebrate assemblages at stations 9 and 7, indicating a fair number of surfaces for attachment.

Summary of LRC-TMDL sites in the context of regional comparison sites:

- MPCA regional comparison site habitat characteristics encompass a range of conditions. Little Rock Creek stations tended to fall in the center of that range, with one important exception being the much shallower water depths at the time of sampling for this study.
- Although multi-metric invertebrate indices used to identify biotic health tended to dampen assemblage differences between LRC-TMDL and MPCA regional comparison sites, certain variables used to generate those metrics were substantially different between data sets. Sample processing procedures appear to be the primary cause of these differences.
- LRC-TMDL stations were generally in the middle of the MPCA regional comparison sites in terms of biotic condition according to the multi-metric and summary indices for macroinvertebrates.
- Fish assemblages appear to be responding to local conditions more so than landscape conditions.
- A greater percent of intolerant taxa were found at station 13 corresponding to areas with extensive riparian forest and wetland.
- Although TMDL sites are at the high end of amount of fine substrate relative to the NRRI comparison sites, the TMDL invertebrate indicators and metrics suggest that the invertebrate communities are in better condition than most of the NRRI sites. This is probably due to the amount of agriculture in the NRRI stream watersheds and the relatively intact buffer along many of the TMDL sites.

Landuse Influence on Biological Community and Habitat

Macroinvertebrates - As was seen with the macroinvertebrate metrics, the differences in sample processing between the NRRI-collected samples and those collected by MPCA left a methods signature on the macroinvertebrate multivariate ordinations. Depending on the transformation used, methods differences appear to account for at least 43% of the variability in the entire combined macroinvertebrate dataset of TMDL and MPCA regional comparison sites (e.g., Figure 54). Most of the environmental and habitat variables and macroinvertebrate metrics are correlated with this primary axis separating the two datasets, and seem to indicate major differences among the datasets. It is also possible; however, that the TMDL LRC sites are themselves quite different and unique relative to the MPCA’s sampled sites.

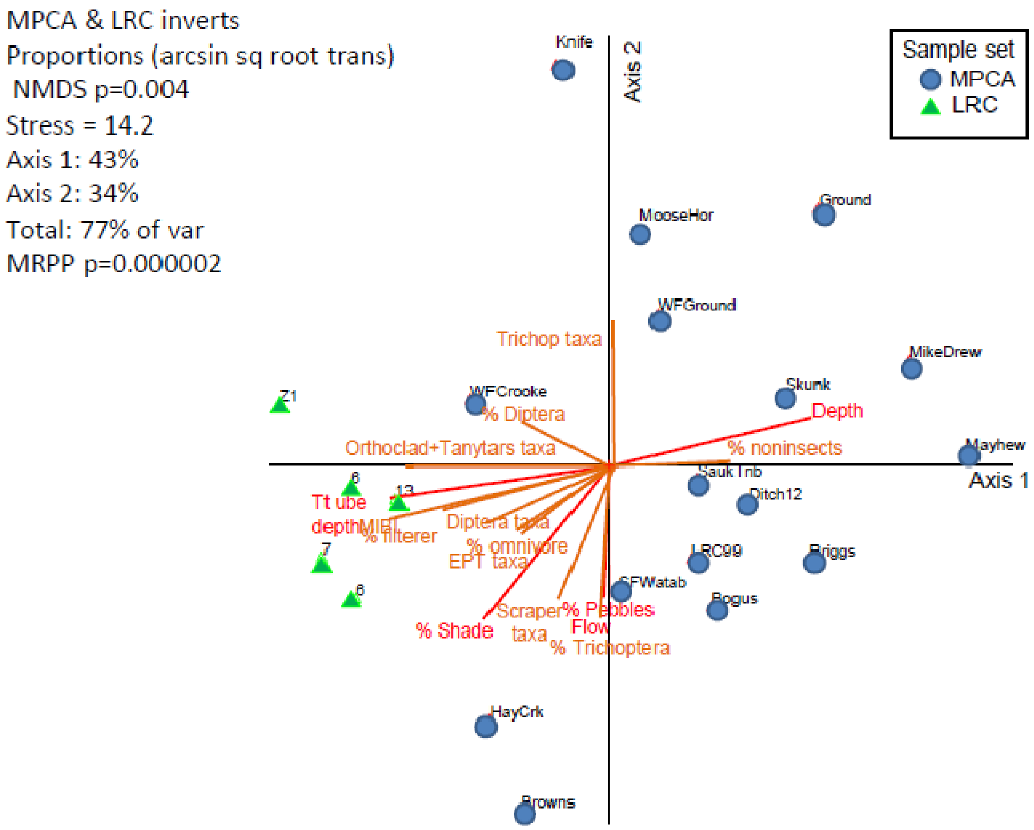

Figure 54. Nonmetric multidimensional scaling ordination of LRC TMDL & MPCA regional comparison sites based on macroinvertebrate relation abundances[5]

To better differentiate between sampling method differences or environmental condition at these sites, we used a data transformation (relative by the maximum) to standardizes each taxa from 0-1 across all sites, greatly down-weighting the significance of highly abundant and highly rare taxa. By using this transformation, a significant non-metric multidimensional scaling (NMS) ordination was obtained. This technique allowed for analysis of watershed landuse/landcover and habitat influences on macroinvertebrate community assemblages from the TMDL and MPCA comparison sites (Fig. 55). The ordination was highly significant (p = 0.004), and captured 75% of the dataset variability in the first 3 axes (46% on axis 3, 16% on axis 2, and 13% on axis 1). Axis 3 primarily separated TMDL sites from all other sites so a clear difference caused by sampling methods or an environmental influence between the sets of sites could not be determined. However, axes 2 and 1 separated sites based on other macroinvertebrate community characteristics (Figure 55).

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[5] All gear types combined, arcsin square root transformation of macroinvertebrate proportions. The ordination was significant for 2 axes (p=0.004, based on Monte Carlo runs), with moderate stress = 14.2. Axis 1 captured 43% of the variation and axis 2 captured 34%, for 77% of the total variability captured. Overlays of landscape and habitat characteristics (red) and macroinvertebrate metrics (brown) have correlations of r = 0.5 or greater. Multiresponse permutation procedure (MRPP) was highly significant (p=0.000002) for differences between the two datasets, LRC TMDL and MPCA.

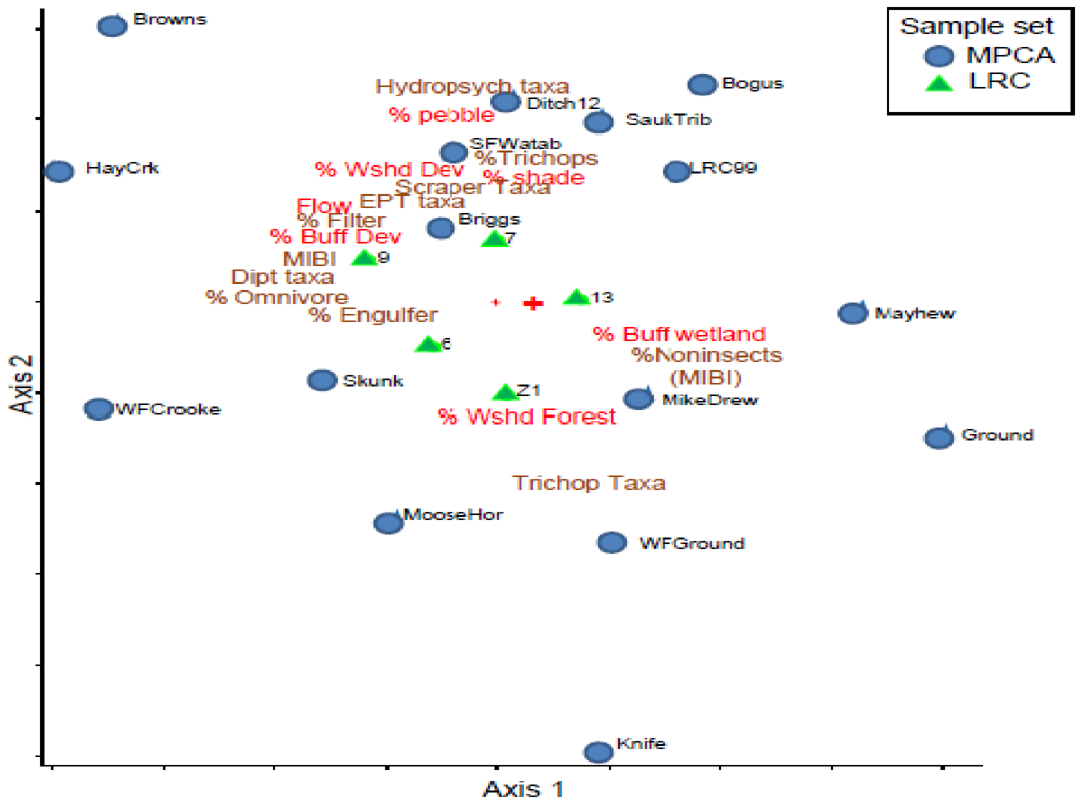

Figure 55. Nonmetric multidimensional scaling ordination of TMDL & MPCA regional comparison sites based on macroinvertebrate relative abundances[6]

Axis 2 separates sites with higher amounts of watershed development (positive on axis 2) from those that have higher amounts of watershed forest (negative on axis 2) (Figure 55, red metrics). The sites in developed watersheds had some habitat characteristics that may moderate the effects of development. The positive direction on axis 2 is also correlated with higher stream flow, higher amounts of pebble (gravel) substrate, and greater stream shading (all $r \geq 0.45$) (see also Table 19). Axis 1 had no habitat characteristics correlating with it at $r \geq 0.45$, and only one land use characteristic, proportion herbaceous wetland within the 100 meter buffer (TMDL sites had quite high amounts of wetland in their stream buffers). Significantly, watershed area did not correlate strongly with any axis, indicating that the size of the watershed did not create significant macroinvertebrate community differences.

Overlaying macroinvertebrate metrics (Figure 55, brown metrics) on the base ordination shows which traits are correlated with the communities that are driving the solution. Only the number of Trichoptera taxa is correlated with the sites that have higher amounts of forest in the watershed. Perhaps because of the buffering effects of in-stream rocky substrate and riparian zone habitat,

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[6] All gear types combined; 'relativize by the maximum' transformation. The ordination was significant ($p=0.004$, based on Monte Carlo runs). Axis 2 captured 17% of the variability in the dataset, and axis 1 captured 13%. (An additional 45% of the variability was captured on axis 3 (not shown) which separated TMDL sites from MPCA comparison sites). Overlays of landscape and habitat characteristics (red) and macroinvertebrate metrics (brown) have correlations of $r=0.45$ or greater. The centroid is indicated with a red plus symbol.

sites with more watershed urbanization also have greater EPT taxa richness, scraper-grazer richness, and proportions of Trichoptera. Filterers, particularly Hydropsychidae: Trichoptera, are also abundant, which indicates a steady supply of water column particulates for them to filter and consume (as organic inputs to streams increase, filterers may increase to take advantage of the additional nutrients). On Axis 1, the separation is between sites with higher MIBI scores, more omnivores, Diptera taxa, and predators that engulf their prey vs. sites with more non-insect invertebrates.

In order to eliminate methods signature as a source of high variability in the dataset, we found that the TMDL LRC sites also had quite different communities from the NRRI regional comparison sites (Figs. 56, 57). Ordinating macroinvertebrate community proportion data again produced a highly significant ordination between the two datasets (MRPP p=0.00005). However, only about 15% of the variability in the dataset seemed to be primarily related to separating the two datasets. NRRI sites were all located in watersheds with more agriculture than all of the TMDL LRC sites. This indicates that there may be a true environmental difference between the LRC sites and both of the regional comparison datasets since neither of the datasets have watersheds with a truly comparable landscape setting.

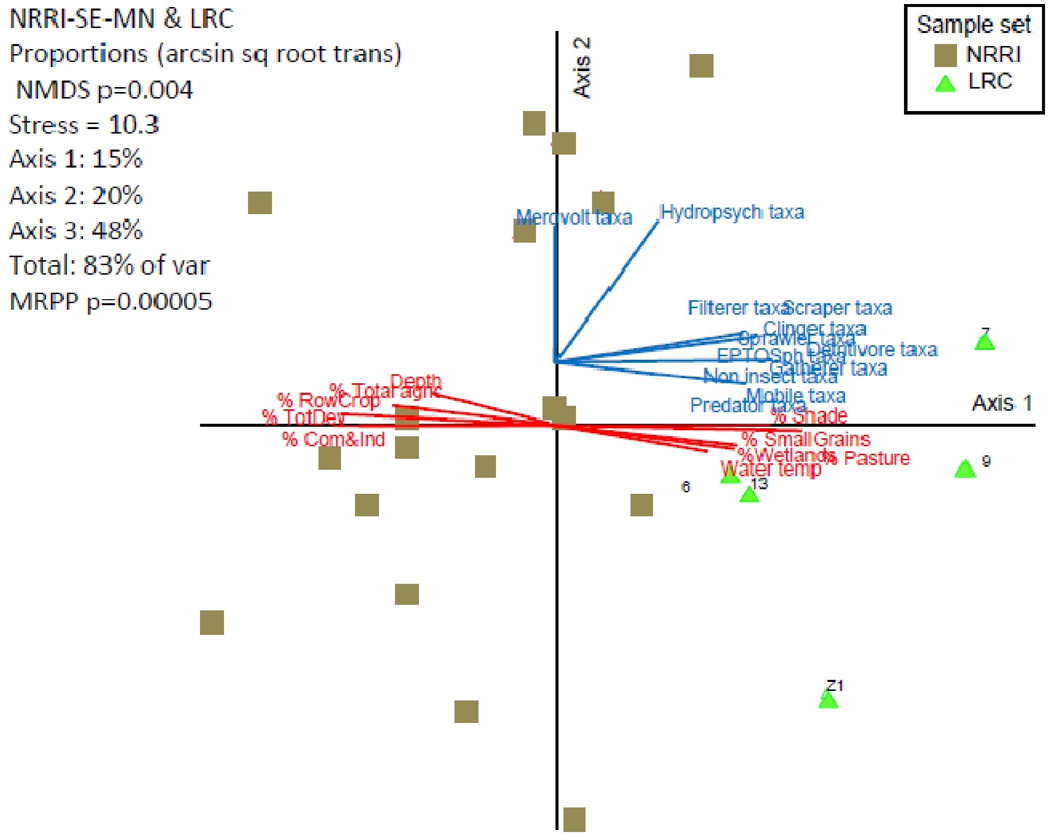

Figure 56. Nonmetric multidimensional scaling ordination of TMDL & NRRI regional comparison sites based on macroinvertebrate relative abundances[7]

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[7] All gear types combined, arcsin square root transformation of proportions. The ordination was significant (p=0.004, based on Monte Carlo runs), with moderately low stress = 10.3. Axes 1 and 2 are shown. Axis 1 captured 15% of the variation, axis 2 captured 20%, and axis 3 captured

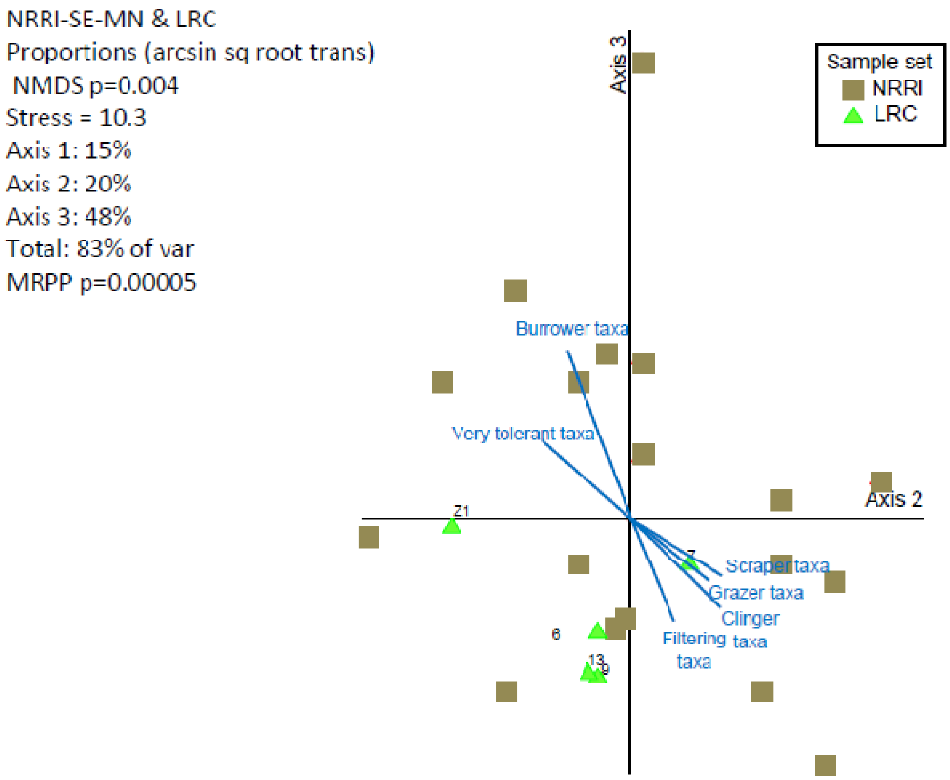

Figure 57. Nonmetric multidimensional scaling of TMDL & NRRI regional comparison sites based on macroinvertebrate relative abundances[8]

LRC TMDL sites tend to be warmer, but contain more shade, and are in watersheds with less agriculture and more wetlands near the stream compared to the NRRI sites (Figure 56). LRC sites also have less residential, commercial, and industrial development in their watersheds. Almost all the macroinvertebrate metrics that correlated strongly with the first 2 axes of the ordination are pointing toward the LRC sites, indicating higher values for the LRC sites than for the NRRI sites. These include nearly all taxonomic count metrics (e.g., number of EPT taxa, number of mobile taxa, filterer taxa, scraper taxa, etc; Figure 56).

Ordination axes 2 and 3 capture the majority of the variability (Figure 57). Unfortunately, none of the environmental or habitat variables that were measured or calculated correlate well with

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48%, for 83% of the total variability captured. Overlays of landscape and habitat characteristics (red) and macroinvertebrate metrics (blue) have correlations of r=0.5 or greater. Macroinvertebrate metrics are offset from origin for ease of viewing. Multiresponse permutation procedure (MRPP) was highly significant (p=00006) for differences between the two datasets, LRC TMDL and NRRI

[8] All gear types combined, arcsin square root transformation of proportions. The same ordination as Figure 53, this time showing axes 2 and 3. The ordination was significant (p=004, based on Monte Carlo runs), with moderately low stress = 10.3. No landscape or habitat variable correlated with either axes at r> 0.5. Axis 1 captured 15% of the variation, axis 2 captured 20%, and axis 3 captured 48%, for 83% of the total variability captured. Macroinvertebrate metrics are shown in blue (correlation r>0.5). Multiresponse permutation procedure (MRPP) was highly significant (p=00006) for differences between the two datasets.

either of these axes, making their interpretation difficult. There are a few macroinvertebrate metrics that correlate with the axes, however. In general, sites with more burrowers and tolerant taxa are in the upper left quadrant, while sites with more scrapers, grazers, clingers, and filterers are in the lower right quadrant (Figure 57).

A final way to investigate sites using ordinations is to ordinate sites based on community metrics, rather than on the communities themselves. This greatly simplifies the comparison by forcing all the community data into a few similar metrics. An ordination on the macroinvertebrate metrics and traits based on counts of taxa still shows most LRC TMDL sites as quite different from the NRRI sites, although stations 6 and Z1 are least different (Figure 58). Agricultural land use again shows up as a dominant correlate on this axis, but the macroinvertebrate metrics (shown as overlays in Figure 58) highlight the diversity of these communities. None of the macroinvertebrate metrics points in the direction of the LRC sites, and instead the metrics are spread relatively evenly around the graph's origin. The ordination done on macroinvertebrate traits (Figure 59) is somewhat easier to interpret. Again the ordination captures a very high amount of the variability in the datasets (92%), and the two datasets still test as significantly different (MRPP p=0.0002). Clingers and filterers are two traits that make up a greater proportion of the community at LRC sites, while sprawlers and burrowers are more dominant at some of the NRRI sites.

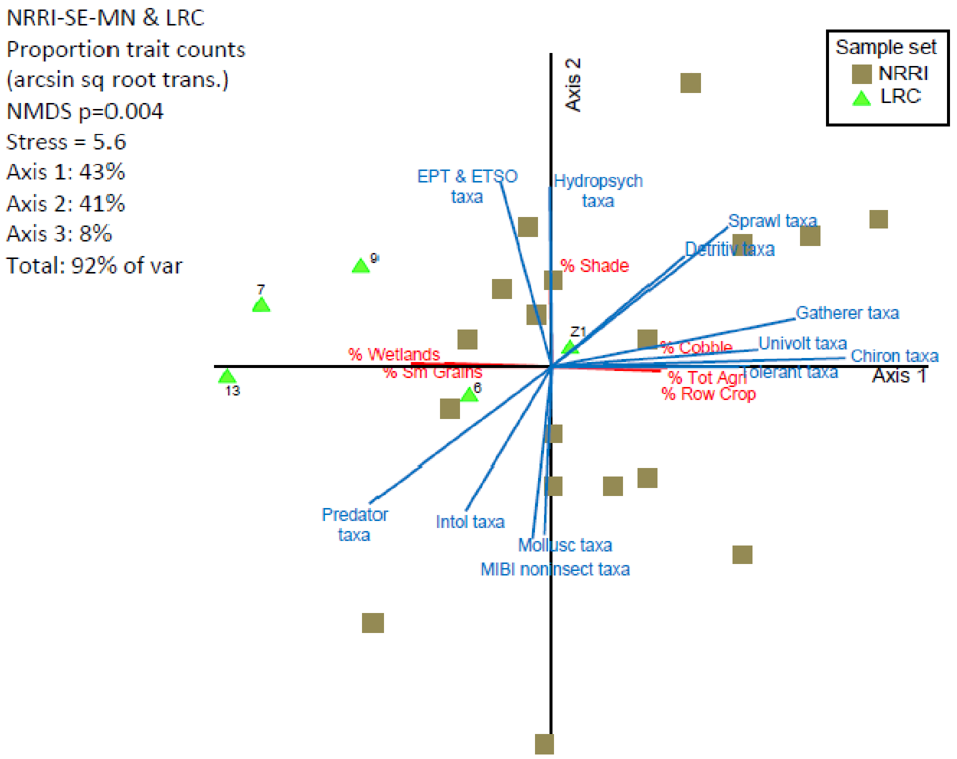

Figure 58. Nonmetric multidimensional scaling ordination of TMDL & NRRI regional comparison sites based on trait taxonomic counts calculated from all gear types combined (no transformation).[9]

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[9] This 2-dimensional ordination was significant (p=0.004, based on Monte Carlo runs), with moderately low stress = 9.4. Axis 1 captured 0.2% of the variation and axis 2 captured 98%, for 98% of the total variability captured. Overlays of landscape and habitat characteristics (red) and macroinvertebrate metrics (blue) have correlations of r=0.5 or greater. Macroinvertebrate metrics are offset from origin for ease of viewing.

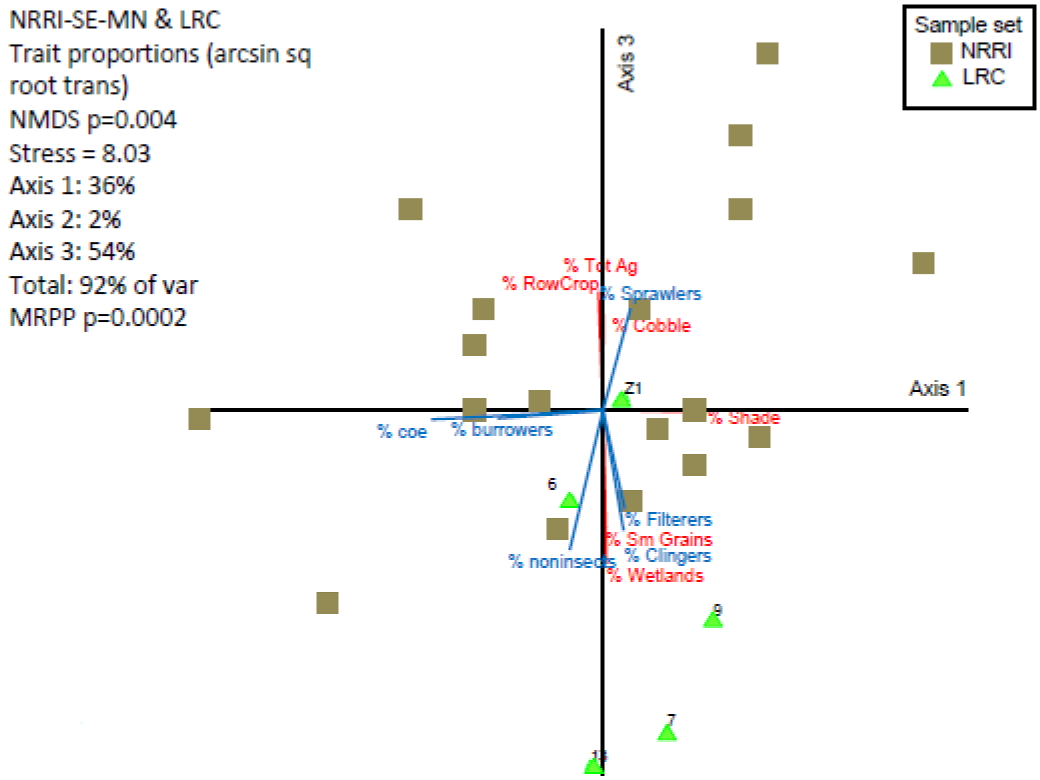

Figure 59. Nonmetric multidimensional scaling ordination of TMDL and NRRI regional comparison sites based on trait proportional metrics calculated from all gear types combined (arcsin square root transformation of proportions).[10]

Few significant correlations were observed among the assemblage data and land use characteristics using multiple regressions. Among the many taxa and metrics, the mobility trait and percent Ephemeroptera were found to be predicted by percent agriculture at the watershed scale at the 0.05 level. Despite the small range of developed land use within these watersheds, a number of significant correlations were observed, including percent Trichoptera ($p = 0.027$), percent Mollusca ($p = 0.0007$), percent MIBI non-insect taxa ($p = 0.008$), and percent burrowing taxa ($p = 0.055$). A Spearman rank correlation performed with land use, percent intolerant taxa, MIBI, and HIBI87 found that watershed percent agriculture was negatively correlated with percent intolerant taxa ($r = -0.47$); and percent forest and percent developed land use in the 100 meter buffer were both correlated with HIBI ($r = 0.45$, $r = 0.48$, respectively; $p < 0.05$). Percent intolerant and tolerant taxa were significantly different between the TMDL and MPCA regional comparison sites ($p = 0.04$, $p = 0.03$, respectively).

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[10] The ordination was significant (p=0.004, based on Monte Carlo runs), with low stress = 8.03. MRPP p=0.0002. Only axes 1 and 3 are shown. Axis 1 captured 36% of the variation, axis 2 captured 2%, and axis 3 captured 54%, for 92% of the total variability captured. Overlays of landscape and habitat characteristics (red) and macroinvertebrate metrics (blue) have correlations of r=0.5 or greater. Multiresponse permutation procedure (MRPP) was very significant (p=0.0002) for differences between the two datasets, LRC TMDL and NRRI.

Local habitat characteristics - We performed a one-way ANOVA to assess whether the MPCA regional comparison sites were inherently different from the TMDL sites in terms of their local habitat conditions. Significant differences were observed with respect to embeddedness ($p = 0.02$), turbidity ($p = 0.04$), depth ($p = 0.0014$), and percent shade ($p = 0.0011$). These findings suggest that regional comparison sites were inherently different, although differences in substrate were not significant. Thus, surficial geology may be similar among sites, but flow regime and channel characteristics potentially from Little Rock Creek.

Only modest correlations were observed between land use and in-stream characteristics. Significant Spearman rank correlations were observed between conductivity and percent agriculture, percent forest (negatively), and percent developed land use at the whole watershed scale ($r = 0.59$, -0.55, and 0.49, respectively), and for percent agriculture within the buffer ($r = 0.57$). Depth was also negatively correlated with percent agriculture ($r = -0.47$) and positively correlated with percent forest ($r = 0.51$) at the watershed and 100 m buffer scale ($r = -0.59$). Percent shade at the sample site was negatively correlated with percent forest in the watershed and the buffer ($r = -0.64$ and -0.62, respectively). While the negative correlation with percent forest seems counterintuitive, inspection of aerial photographs revealed that many sample sites were located within herbaceous wetlands, and thus had open canopy cover at the location where densitometer measurements were taken. Percent shade measures are largely dependent upon local scale conditions including buffer land cover and stream width (wider streams have less canopy cover), despite being correlated with land cover measures at larger spatial scales.

Land use gradient - A preliminary measure of landscape-based stress was quantified by accumulating the proportion of agricultural land use within the 247 subcatchments of the Little Rock Creek watershed. Because agricultural land use impacts are moderated by riparian vegetation, we subtracted out the proportion of forested land within the buffer. Figure 60 depicts the distribution of subcatchments, ranked from “least” to “most” agriculture. The general trend is that headwater catchments show relatively low amounts of agriculture in the eastern part of the basin, while those in the western portion of the basin are more intensively cropped. Furthermore, there is an increase in the amount of forested land cover in the floodplain and vicinity of the main stem of the river beginning upstream of station 9. Thus, the riparian buffer is likely moderating the impacts of the agricultural land use occurring upstream. This is apparent at stations 13 and Z1, in particular. Stations 6 & 7 have the greatest potential to experience the localized effects of agricultural activities, since each of those sample locations are in the vicinity of intensively cropped subcatchments. Local habitat observations confirm the proximity of cropped land at station 7, but not at station 6.

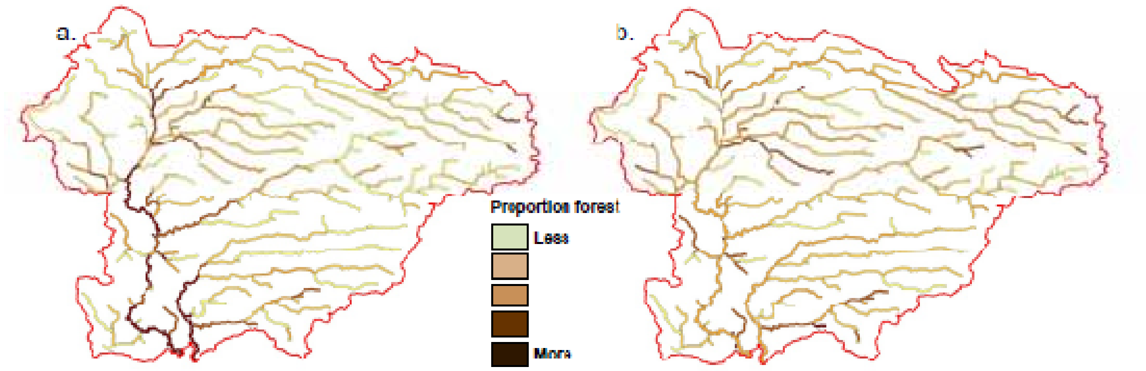

Figure 60. a) Proportion of forest cover associated with a 50 m stream buffer within the Little Rock Creek watershed, and b) the accumulated forest cover influence using the 50 m forest cover-type along the stream network

NRRI Summary and Conclusions

At most Little Rock Creek TMDL sites, the macroinvertebrate community is characterized by taxa that can utilize sandy habitat or substrate such as wood, and may cause a reduction, substantial at some sites, in the number of invertebrates considered sensitive to stress, such as the EPT taxa. Little Rock Creek communities differ substantially from those characteristic of rocky habitats. These differences are evident in the macroinvertebrate community ordination, with sites having more flow and more exposed rocky substrate, especially pebbles, showing an association in the ordination with macroinvertebrate metrics associated with rockier substrates (e.g., Trichoptera, scrapers, EPT taxa).

TMDL sites were sampled in 2006, a drought year, in which we found very low flows and areas with low dissolved oxygen that also undoubtedly affected the macroinvertebrate community. The low flows may have exacerbated the amount of fine sediments in the stream due to the lower stream power to move sediment through the system. At several sites we found indications of rocky substrate buried beneath the fine substrates. This suggests that more rocky substrate was exposed at these sites in the past. Historic land use change, which is associated with altered hydrology, may have both contributed to the amount of sediment in the stream while also reducing stream power to move the sediment through the channel (e.g., Fitzpatrick et al. 2006). Sand-bottomed stream reaches on Wisconsin’s Bayfield peninsula have been shown by the Wisconsin DNR to be able to move enough sand through the reach to re-expose buried rocky substrate when stream power is increased by removing sand-trapping woody debris (Dennis Pratt, WDNR, pers. comm.).

Fundamental differences among TMDL and MPCA sites were observed with respect to the local habitat characteristics such as the amount of shade, channel depth, and amount of fine sediments. Biological communities appear to be responding to these features. We believe land use at the watershed scale is ameliorated by extensive riparian wetlands and forest cover in the flood plain. This becomes particularly clear when the TMDL sites are compared to the NRRI sites. In those comparisons, LRC sites are in better condition than the NRRI sites which are more heavily agricultural, and contain less stream buffering by wetlands and forests. Had stream flows not been so extremely low during the sampling period for the TMDL sites, the macroinvertebrate communities might have been in even better condition than they appear in this analysis. These results indicate either the TMDL sites are truly degraded compared to sites containing hard substrate, or that the EPT metric is not a robust measure of condition in streams that are naturally sandy. However, the small distance separating TMDL sample sites suggests differences in local condition, over changes in geologic condition, favor the former as the prevailing explanation.

Model

The causes and potential sources of bedded sediment in Little Rock Creek are modeled in Figure 61. Each pathway was evaluated using CADDIS, shown in Table 20.

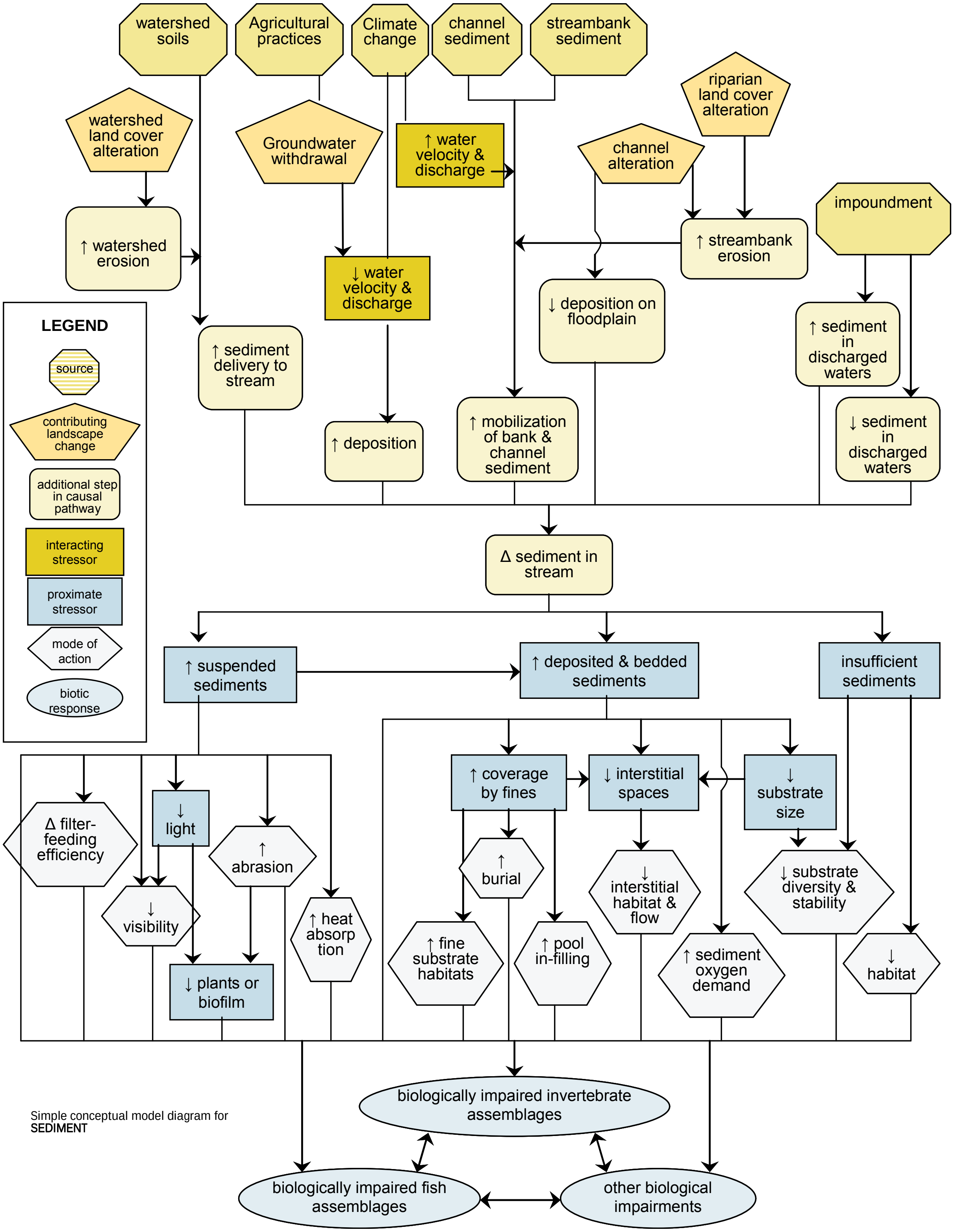

Figure 61. Model of sources and causal pathways for deposited and bedded sediment in Little Rock Creek

Table 20. Sufficiency of evidence for increased bedded sediment in Little Rock Creek

Types of Evidence	Watershed Soils	Agricultural practices	Climate change	Channel sediment	Streambank sediment	Impound-ment
Evidence using data from Little Rock Creek						
Spatial / temporal co-occurrence	+	+	0	+	+	0
Evidence of exposure, biological mechanism	+	NE	NE	+	+	NE
Causal pathway	NE	++	0	+	+	+
Field evidence of stressor-response	+	NE	NE	+	+	NE
Field experiments / manipulation of exposure	NE	NE	NE	NE	NE	NE
Laboratory analysis of site media	NE	NE	NE	NE	NE	NE
Temporal sequence	NE	0	NE	NE	NE	NE
Verified or tested predictions	NE	NE	NE	NE	NE	NE
Symptoms	+	NE	NE	+	+	+
Evidence using data from other systems						
Mechanistically plausible cause	+	+	NA	+	+	+
Stressor-response in other field studies	+	+	0	+	+	+
Stressor-response in other lab studies	NE	NE	NE	NE	NE	NE
Stressor-response in ecological models	+	+	NE	+	+	NE
Manipulation experiments at other sites	NE	NE	NE	NE	NE	NE
Analogous stressors	NE	NE	NE	NE	NE	NE
Multiple lines of evidence						
Consistency of evidence	+	+	0	+	+	0
Explanatory power of evidence	+	+	0	+	+	0

Conclusion

The bedded sediment levels in Little Rock Creek is a plausible stressor causing the lack of coldwater assemblage. The highest percentage of simple lithophils and sensitive individuals in Little Rock Creek were found at station 7, which had the lowest amount of fine substrates. Greater numbers of mayflies and larval riffle beetles were found at station 7, which had greater amounts of coarse substrate. Burrowing invertebrate abundances were lowest at stations 9 and 7, which had hard substrate. The macroinvertebrate community, as measured by several metrics (Tricoptera, scapers, EPT taxa), responds positively to sites having more flow and more exposed rocky substrate, especially pebbles. The source(s) of the bedded sediment is uncertain, but altered flow may be the reason for this problem.

NITRATE

Nitrate Standard

According to Minnesota Rule 7050.0470 Subpart 4 Little Rock Creek is classified as a 1B stream. This classification designates that treated water will meet both the primary and secondary drinking water standards with approved disinfection. The nitrate levels in Little Rock Creek were considerably higher than the standard 10mg/L.

Effects of high nitrates in streams

Nitrates have a toxic effect on fish and invertebrates by the conversion of oxygen-carrying pigments to forms that are not capable of carrying oxygen (Grabda et al, 1974). Camargo et al. (2005) reviewed of literature regarding nitrate toxicity to aquatic animals and concluded that toxicity to aquatic organisms increases with increased exposure and nitrate concentrations. The mimic shiner and hornyhead chub, nitrate sensitive species, which have been reported in Little Rock Creek historically were not found in the most recent surveys. In a study of species' tolerance along physiochemical gradients, both species had mean tolerance indicator values (TIV) for nitrate + nitrite of less than 1 mg/L (Meador and Carlisle, 2007), suggesting that increased nitrate + nitrite may be to some extent responsible for their disappearance from Little Rock Creek.

Long exposures to elevated nitrate concentrations can cause detrimental effects to some invertebrates, fish and amphibians. Some salmonid fish populations can suffer from impaired reproductive success from nitrate levels in the range of 10 mg/L (Kincheloe et al., 1979). Nitrate levels in Little Rock Creek have shown to be above 10 mg/L at times during the summer months. These high levels of nitrates may reduce reproductive capabilities of coldwater fish, as well as lead to low dissolved oxygen through nitrification processes.

Data evaluation for nitrate in Little Rock Creek

Sites 4, 7, 9, 10, MW and BH2 surpassed the standard at least one time in the months of June, July or August (Figure 62). The northern part of the watershed is where the highest nitrate level (27 mg/L) was observed in this study, at the mini-well location. The highest level of nitrate in Little Rock Creek was at Site 4. The highest level of nitate in Bunker Hill Creek was 12.4 mg/L. Lower levels of nitrate (below 1 mg/L) from April to September were also recorded in the Little Rock Creek watershed, indicating background levels less than 1 mg/L.

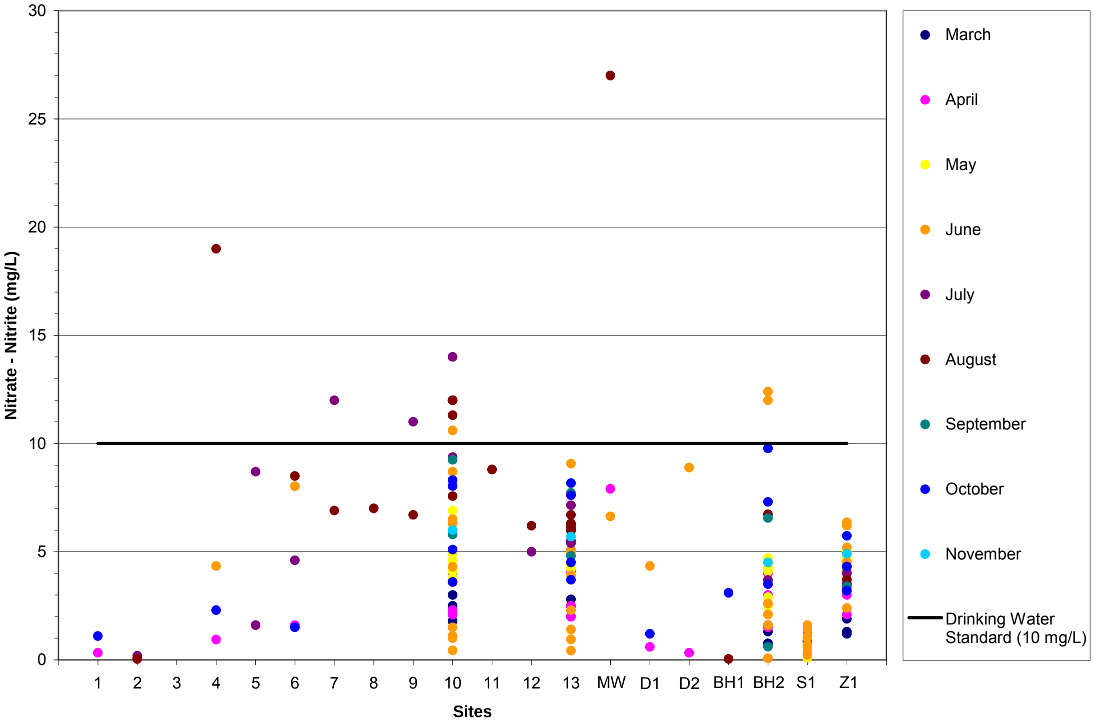

Figure 62. Nitrate-nitrite levels by site in Little Rock Creek watershed

The number of sensitive fish individuals, in the Upper Mississippi River Basin, decreased with increasing nitrate levels, as well as the percentage of sensitive species (Figures 63 and 64). A few outliers do exist in the dataset between 2 and 7 mg/L in the Upper Mississippi River Basin.

The MN DNR historically found two sensitive fish species, the mimic shiner and hornyhead chub, which have not been observed in 1999, 2007, or 2008 surveys by the MPCA. The mimic shiner was last found in a survey in 1987 and similarly, the hornyhead chub was found in 1987 and the most recent observation was in 1992. Brook trout and longnose dace are also sensitive species and have been maintaining a limited level of population in Little Rock Creek.

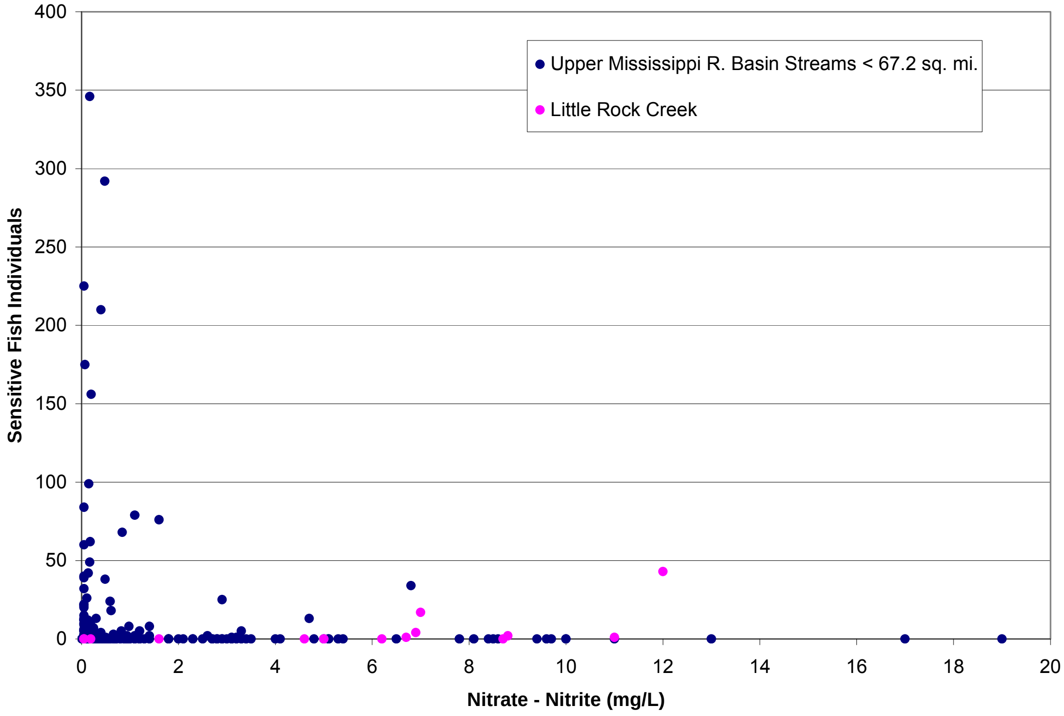

Figure 63. Sensitive fish individuals and nitrate-nitrite levels for the Upper Mississippi R. Basin (with drainage areas less than 67.2) and Little Rock Creek

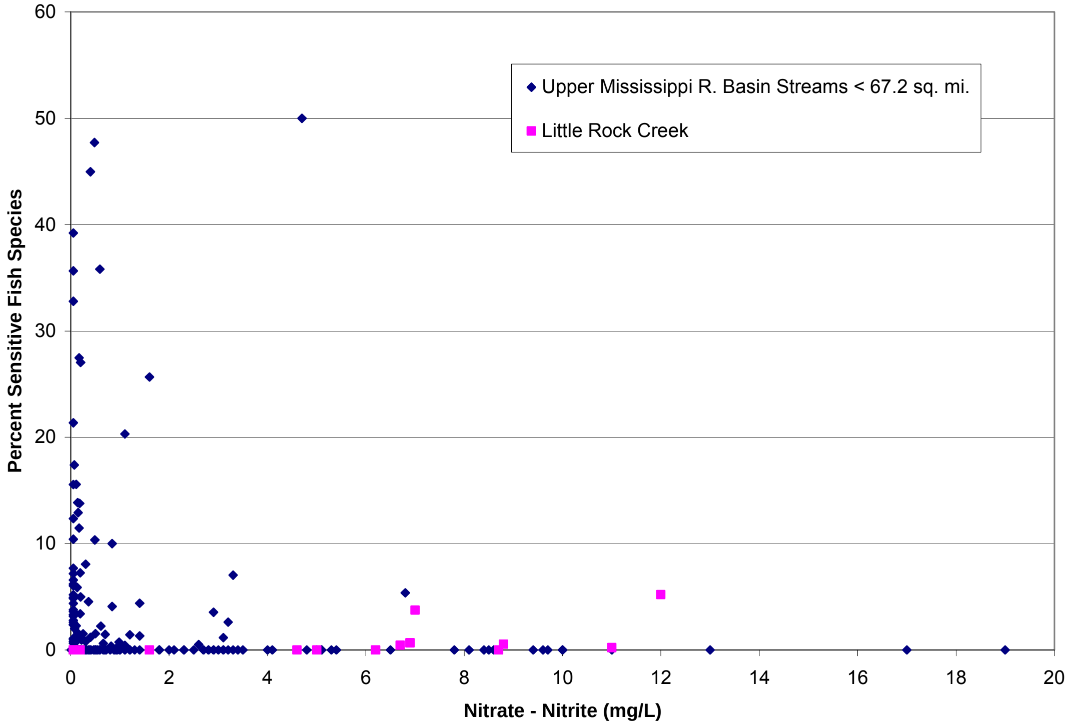

Figure 64. Percent sensitive fish species and nitrate-nitrite levels for the Upper Mississippi R. Basin (with drainage areas less than 67.2) and Little Rock Creek

In comparison to the North Central Hardwood Forests Ecoregion 75th percentile for ammonia, of 0.21 mg/L, levels were elevated in Little Rock Creek. Total Ammonia was considered high at sites 1, 4, 10, 13, D1, BH2, S1, and Z1. With the highest level of total ammonia found at site Z1 with 4.07 mg/L. Site 10 had the highest recorded total ammonia in Little Rock Creek with 3.16 mg/L. Un-ionized ammonia was below the Minnesota state standard of 16 μg/L.

Models of sources and causal pathways for nitrate-nitrite in Little Rock Creek
The causes and potential sources for nitrate-nitrite in Little Rock Creek are modeled in Figure 65. Each pathway was evaluated using CADDIS, shown in Table 21.

Generally the pathway of nitrate loading into streams is by shallow groundwater (Hayashi and Rosenberrry, 2002). Helsel (1995) reported below agricultural or urban areas nitrate concentrations were the highest. Nitrate levels were also highest in areas where geologic characteristics encourage quick movement of water to the aquifer, such as unconsolidated sands. Nitrates present in sandy soils have the potential to leach quickly beyond the level for plant utilization, because sandy soils do not hold as much water as finer grained soils, such as clay (Bundy et al., 1994). Jordan et al. (1997) examined Maryland watersheds with variable amounts of agricultural land with regards to their nitrate levels and baseflow. Nitrate levels were higher in watersheds with a greater percentage of cropland and in streams with high baseflow to total flow ratios. The high levels of nitrate during baseflow indicated that the nitrate is entering the stream through groundwater flow. Soil mineralization, commercial fertilizers, livestock wastes and residue from plants that are able to fix nitrogen from the air and convert it to organic forms are the main sources of nitrogen in agricultural settings (Bundy et al., 1994). Nitrogen found in groundwater can additionally come from septic systems or livestock holding areas.

Magner et al. (1990) studied residential wells in an area west of Little Rock Creek with elevated nitrate levels. Ground water samples were analyzed for nitrate between May 1987 and September 1988 with 52% of the wells nitrate levels exceeding 10 mg/L. In 11% of the wells sampled, nitrate levels were found to be less than 1 mg/L, indicating a natural background condition less than 1 mg/L nitrate.

Nitrogen is commonly applied as a crop fertilizer. Due to the fact that 48% of the Little Rock Creek watershed is planted to agricultural crops, it is assumed that nitrogen and forms of nitrogen, such as ammonia, are being applied to the cropland throughout the watershed.
Soils and geology in the study area are conducive for nitrogen fertilizer to leach to the groundwater. Magner et al. study concluded that the nitrate contamination was from the irrigated cropland, with the higher levels of nitrate found in the wells down from the irrigated field.

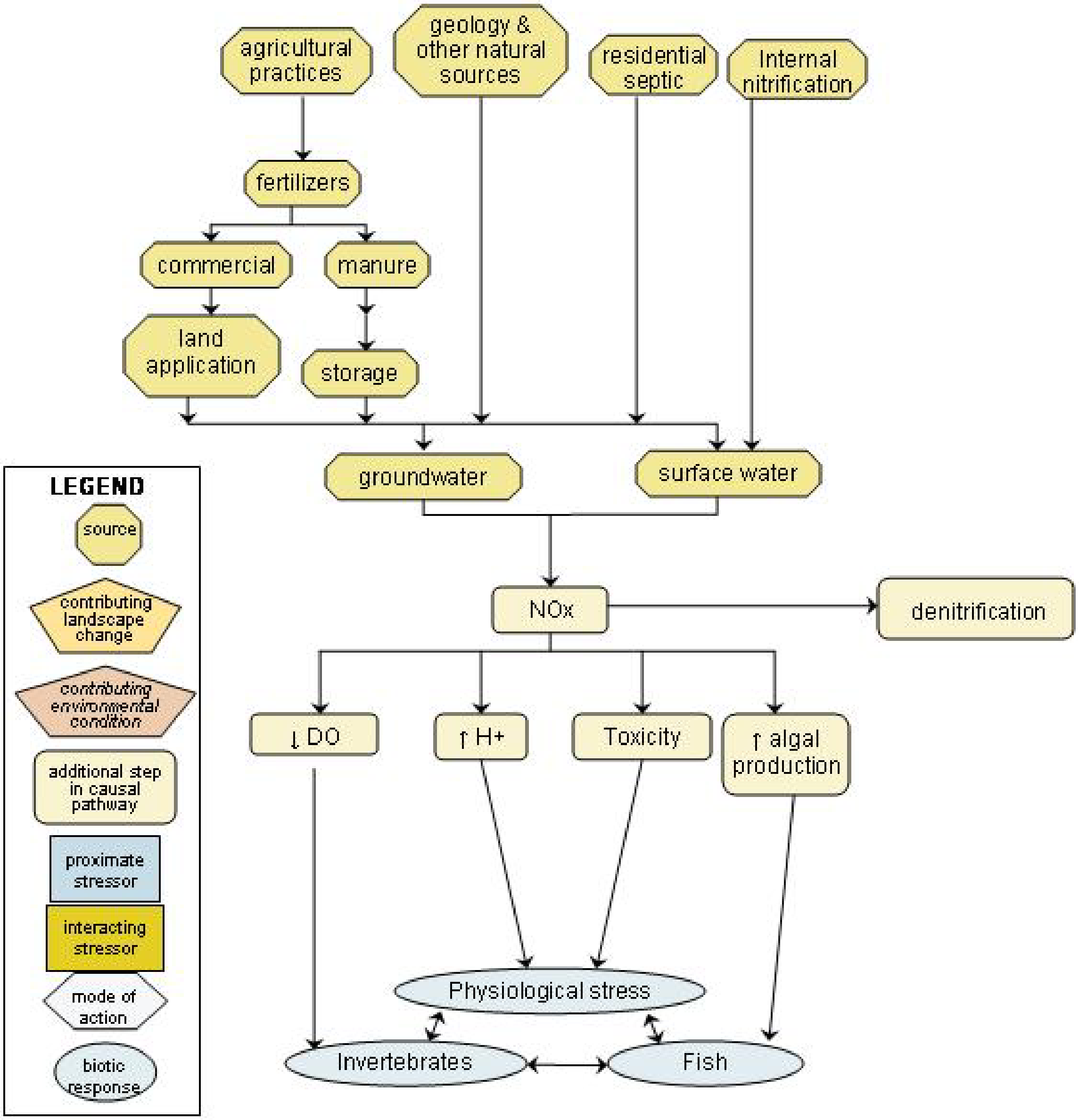

Figure 65. Model of sources and causal pathways for nitrate-nitrite in Little Rock Creek

Table 21. Sufficiency of evidence table for sources of nitrate-nitrite in Little Rock Creek

Types of evidence	Agricultural Practices	Geology & other Natural Sources	Residential Septic	Internal nitrification
Evidence using data from Little Rock Creek				
Spatial/temporal co-occurrence	+	0	0	0
Evidence of exposure, biological mechanism	+	0	0	0
Causal pathway	++	0	0	0
Field evidence of stressor-response	+	0	0	0
Field experiments / manipulation of exposure	NE	NE	NE	NE
Laboratory analysis of site media	NE	NE	NE	NE
Temporal sequence	0	0	0	0
Verified or tested predictions	0	0	0	0
Symptoms	+	0	0	0
Evidence using data from other systems				
Mechanistically plausible cause	+	+	+	+
Stressor-response in other field studies	+	+	+	+
Stressor-response in other lab studies	+	+	+	+
Stressor-response in ecological models	NE	NE	NE	NE
Manipulation experiments at other sites	+	+	+	+
Analogous stressors	0	0	0	0
Multiple lines of evidence				
Consistency of evidence	+	0	0	0
Explanatory power of evidence	++	0	0	0

Conclusion

The nitrate-nitrite levels are a plausible stressor causing the lack of coldwater assemblage in Little Rock Creek, considering that levels exceed the state standard for drinking water and the other evidence presented. Nitrate levels in Little Rock Creek ranged from below 1 mg/L to 18 mg/L. These levels have proven to be detrimental to some invertebrates, fish and amphibians. It is unlikely that the elevated nitrate levels in LRC are due to the geology or other natural sources in the watershed, because the background condition of nitrates in the watershed is below 1mg/L (shown by samples that were less than 1mg/L). It is unknown that the nitrate is due to residential septic. It is unknown if the increase in nitrates is due to internal nitrification. The nitrate-nitrite levels are likely due to agricultural practices in the watershed.

DISSOLVED OXYGEN

Dissolved oxygen standard

In class 2A streams, the Minnesota standard for dissolved oxygen is 7.0 mg/L as a daily minimum. It also requires compliance with the standard 50 percent of the days at which the flow is equal to the $7Q_{10}$.

Effects of low dissolved oxygen in streams

Biological organisms depend on dissolved oxygen for life. Low dissolved oxygen can have detrimental effects on fish communities. Fish, such as brown trout, have been known to avoid areas of water with dissolved oxygen less than 5 mg/L (Raleigh et al., 1986), and the daily minimum dissolved oxygen levels largely impact fish growth rates (Doudoroff and Warren, 1965).

Low dissolved oxygen is possibly derived from the high nitrate concentrations and lack of aeration from minimal instream habitat due to aggradation and the lack of aeration by riffles. The higher temperatures can also lead to decreased dissolved oxygen levels. As temperatures increase the saturation levels of dissolved oxygen decrease, while the increased temperature increases the dissolved oxygen needs for fish (Raleigh et al., 1986). Low dissolved oxygen can be an issue in streams with slow currents, excessive temperatures, high biological oxygen demand and high groundwater seepage (Hansen, 1975)

Data evaluation for dissolved oxygen in Little Rock Creek

Dissolved oxygen has been below the state standard for class 2A streams (7 mg/L) numerous times throughout the Little Rock Creek watershed (Figures 66 and 67). During the initial assessment, site 2 had 5.1 mg/L DO at 11:15am, on July 15th, 1999. Sites 5, 9, and 12 were below the standard during biological sampling in 2007 with 0.42, 5.4, and 6.76 mg/L DO respectively. In 2007, Site 7 was above the standard with 12.26 mg/L DO. Of the field measurements made during biological sampling in 2008, the only site below the standard with 5.74 mg/L DO was site 5. The other biological sites were above the DO standard at the time of fish sampling. In 2007 and 2008 the biological crews indicated water levels were below normal water levels. At the time of the sampling in 1999, biological crews indicated normal water levels.

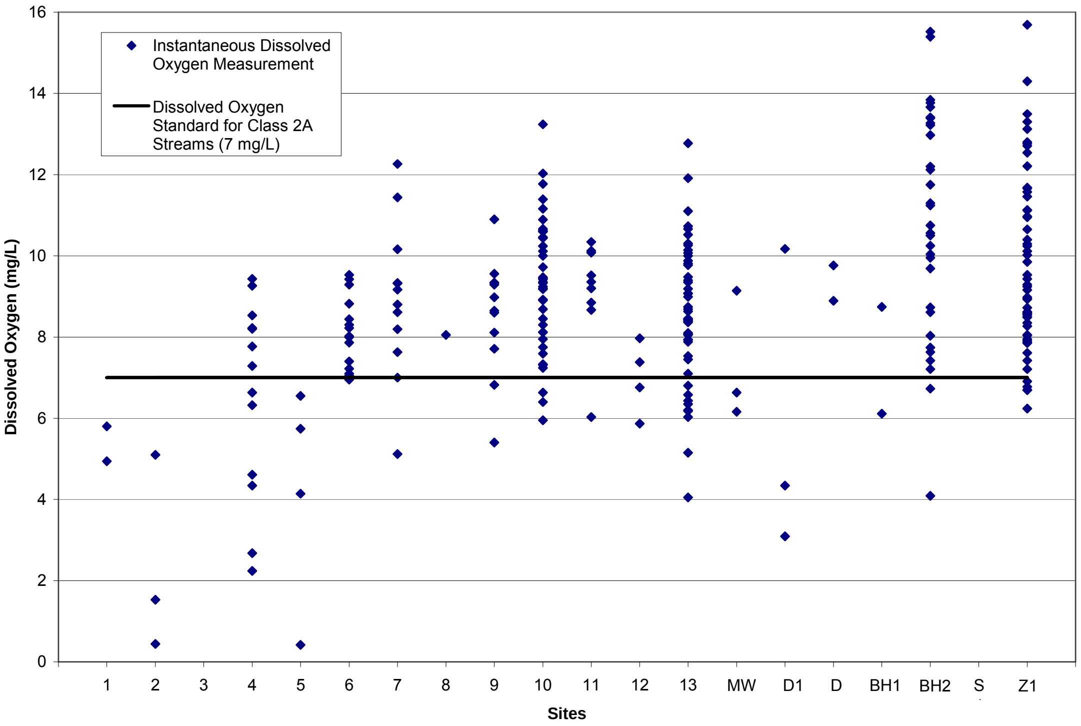

Figure 66. Instantaneous dissolved oxygen measurements by site

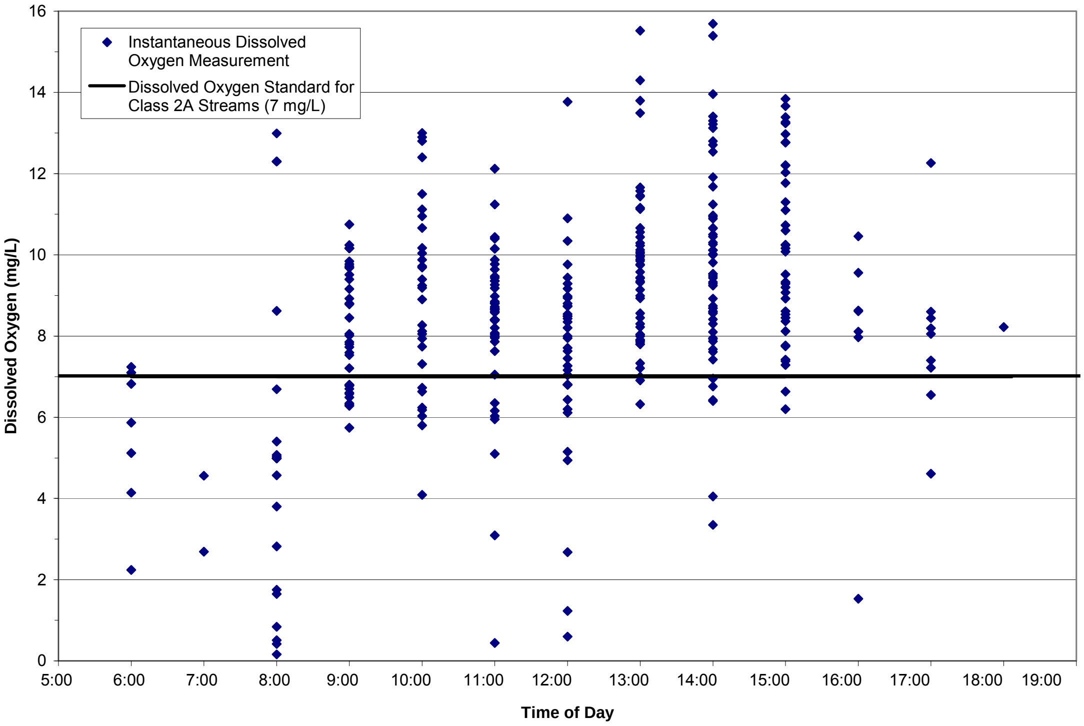

Figure 67. Instantaneous dissolved oxygen measurements by time of day

A longitudinal dissolved oxygen survey along Little Rock Creek indicated levels of DO below the standard at sites 4, 5, 7, 9, and 12 (Figure 68). In both the evening and the morning surveys, sites 4 and 5 were less than the 7 mg/L standard. Dissolved oxygen was measured at other times in the field collection, but typically in the middle of the day when dissolved oxygen levels would be likely to be near the highest levels due to productivity, yet there were still dissolved oxygen levels below the standard in the middle of the day. Diurnal measurements at sites 7, 11, and 13 were recorded from August 19^{th} – 21^{st}, 2008. Dissolved oxygen levels were below the standard at sites 7 and 11 (Figures 69 through 71).

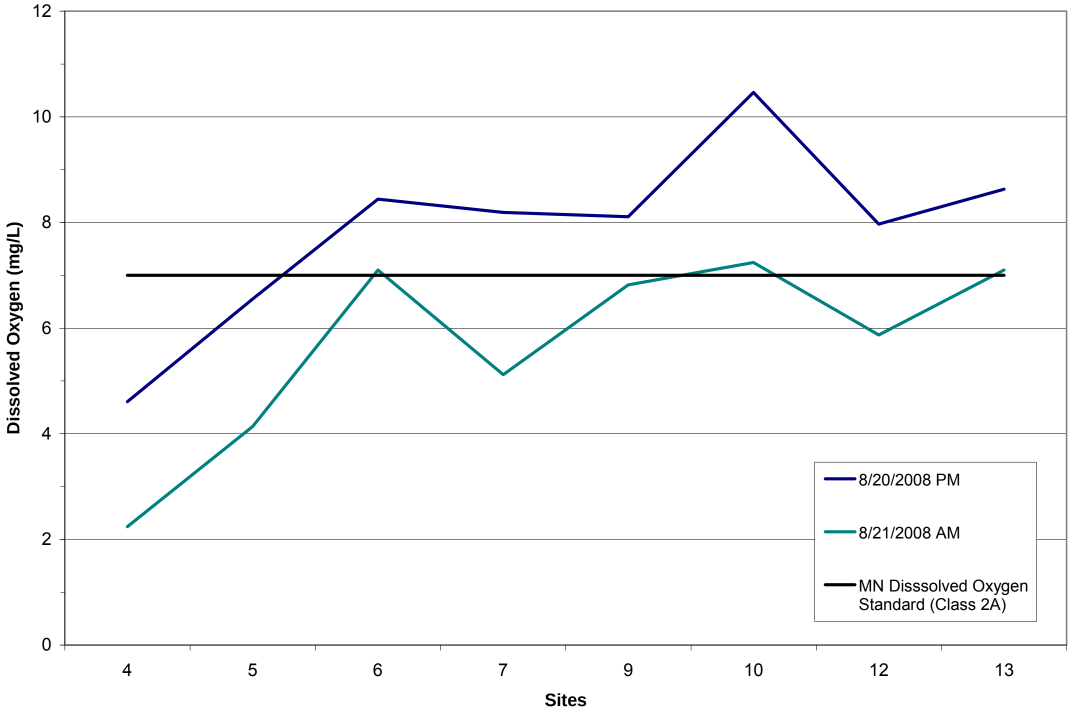

Figure 68. Longitudinal dissolved oxygen profile, August 20th and 21st, 2008

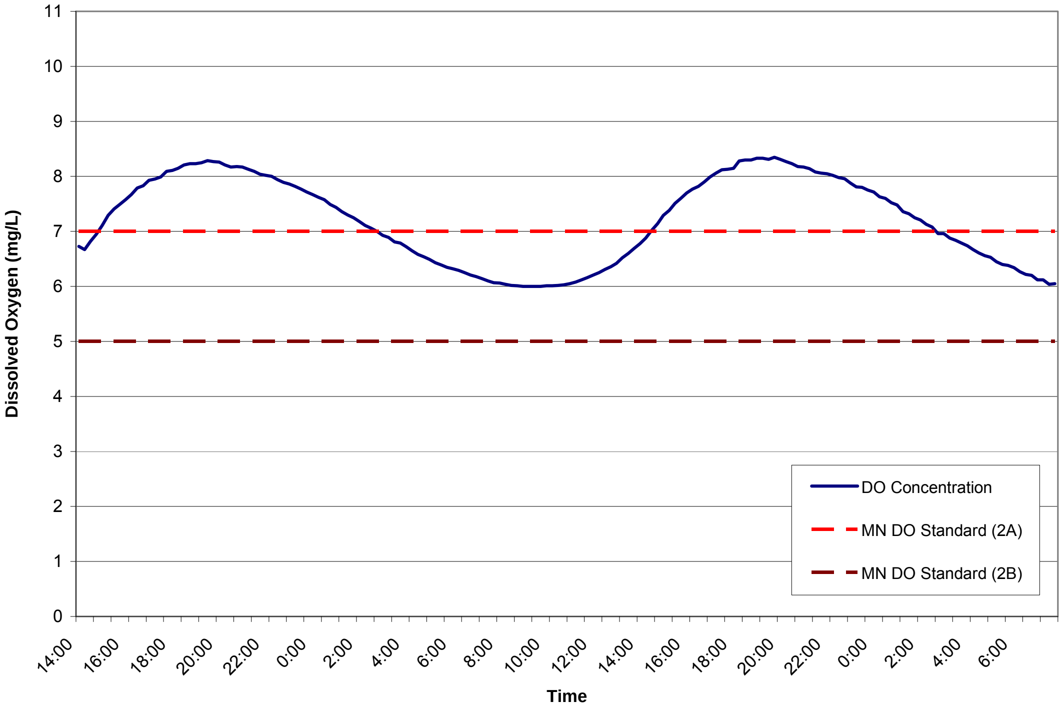

Figure 69. Site 7 dissolved oxygen diurnal measurements

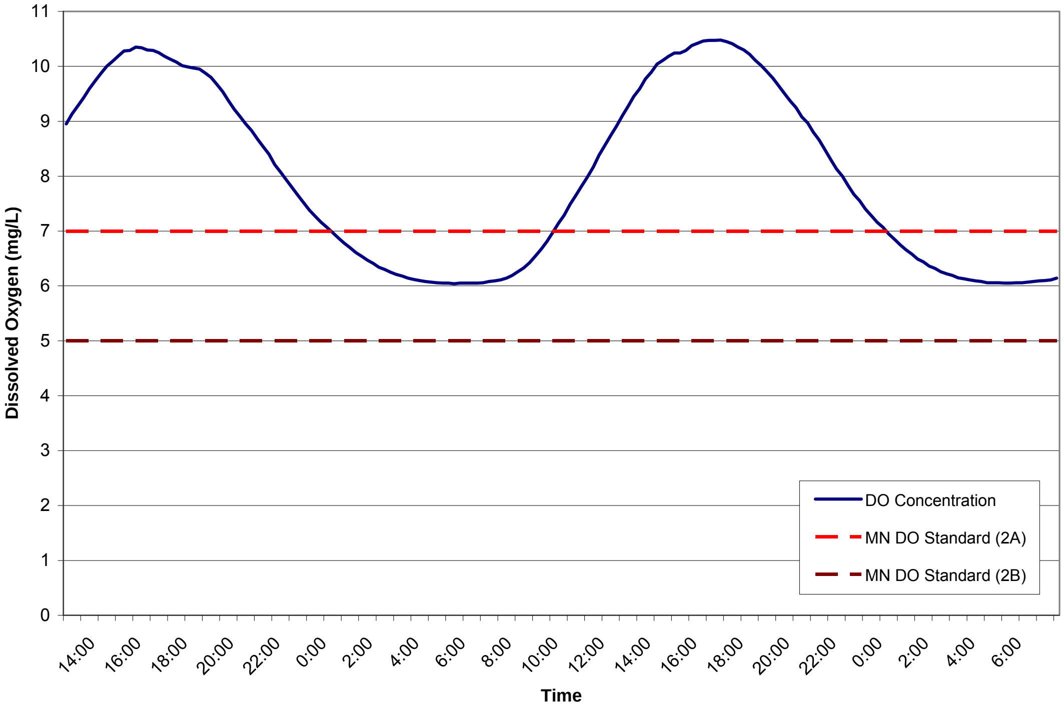

Figure 70. Site 11 dissolved oxygen diurnal measurements

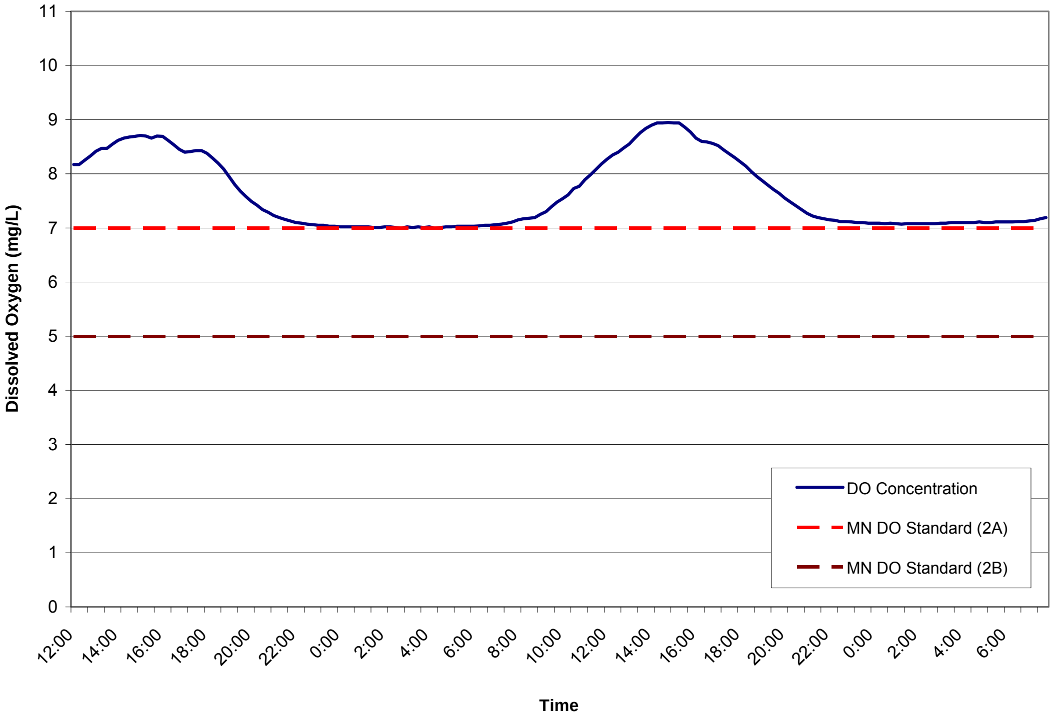

Figure 71. Site 13 dissolved oxygen diurnal measurements

Using Meador and Carlisle (2007) tolerance indicator values for dissolved oxygen, the fish in Little Rock Creek where quartiled were the 1st quartile contains fish that are less sensitive to low dissolved oxygen and the 4th quartile is sensitive to low dissolved oxygen (Table 22).

Table 22. Little Rock Creek fish quartiled based on tolerance indicator value (TIV) for dissolved oxygen (Meador and Carlisle, 2007)

1st Quartile		2nd Quartile		3rd Quartile		4th Quartile	
Common Name	TIV	Common Name	TIV	Common Name	TIV	Common Name	TIV
white crappie	5.7	logperch	7.7	creek chub	8.4	longnose dace	8.8
bluegill	7	mimic shiner	7.8	common carp	8.5	fathead minnow	8.8
black crappie	7.3	walleye	8	central stoneroller	8.6	brook trout	8.9
johnny darter	7.4	northern pike	8	common shiner	8.6	brown trout	8.9
largemouth bass	7.4	black bullhead	8	white sucker	8.6	bluntnose minnow	8.9
yellow perch	7.6	hornyhead chub	8.3	spotfin shiner	8.7	blacknose dace	9.1
pumpkinseed	7.6					sand shiner	9.2
						bigmouth shiner	9.9

Less sensitive to DO → **Sensitive to DO**

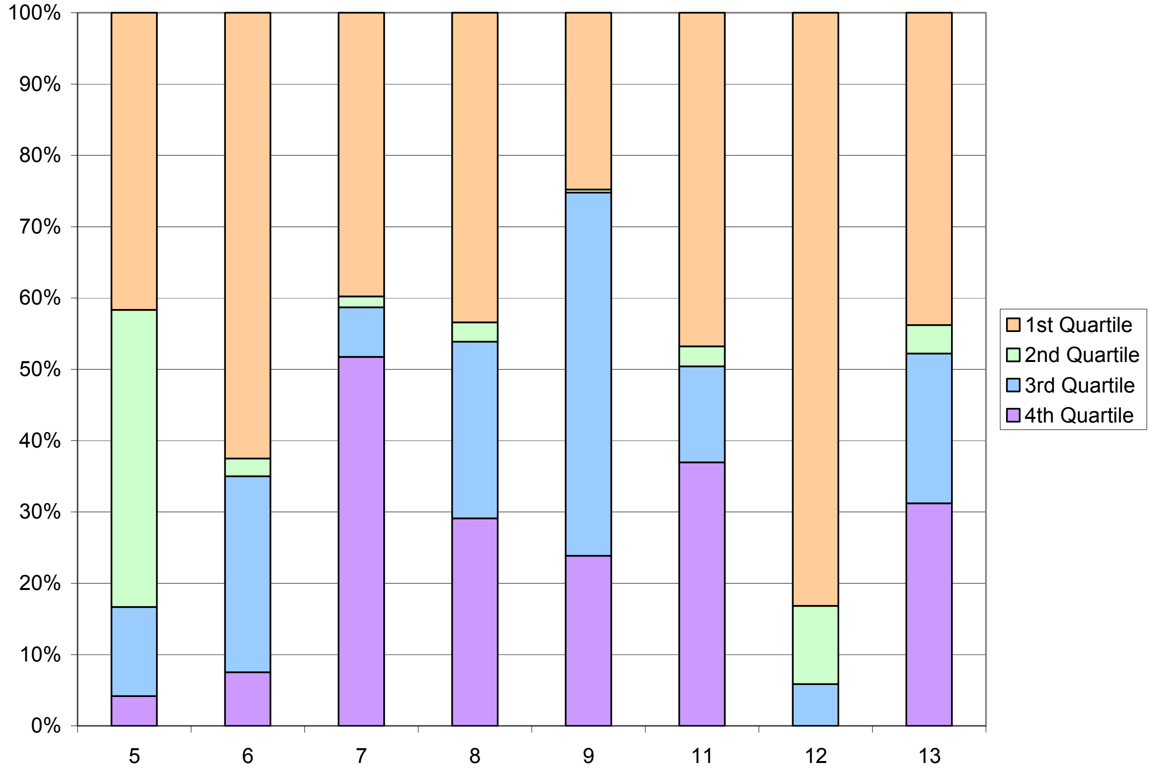

Figure 72. 2008 Fish data for Little Rock Creek quartiled by dissolved oxygen tolerance indicator values

The sites with low dissolved oxygen during the longitudinal dissolved oxygen survey, in August 2008, coincide with the lower percentage of the 4th quartile fish species that are more sensitive to periods of low dissolved oxygen (Figure 72). Similarly, it appears as though there is a weak pattern showing that a low percentage of EPT occur at the same sites as the low dissolved oxygen was measured (Table 23).

Table 23. Percent Ephemeroptera, Plecoptera, and Tricoptera by year and site for Little Rock Creek

Year	Stream-site	% EPT
1999	2	49.8
2006	6	2.25
2006	7	17.19
2006	9	37.47
2006	13	4.96
2006	Z1	1.69
2007	9	15.9
2007	12	22.3

Models of sources and causal pathways for low dissolved oxygen in Little Rock Creek

The sources likely to impact dissolved oxygen in Little Rock creek are shown in Figure 75. Each pathway was evaluated using CADDIS, shown in Table 26.

It is likely that agricultural practices and some land cover alterations in the watershed are contributing to the increase in organic material and nutrient to the stream. These increases are evident in the increases in biological oxygen demand, phosphorus and nitrates (see nitrate section).

Biochemical oxygen demand or BOD is a chemical procedure for determining the rate of uptake of dissolved oxygen by the rate biological organisms in a body of water use up oxygen. At times, the biological oxygen demand was found to be higher than 3.4 mg/L, the 75th percentile values for reference streams in the North Central Hardwood Forests Ecoregion (McCollor and Heiskary, 1993) (Table 24 and Figure 73). All of the sites with BOD5 measurements collected had maximum BOD5 levels well above the ecoregion 75th percentile, with the highest BOD5 level of 51.0 mg/L at site 13. The mean BOD5 values were nearer to the 75th percentile of least impacted streams, with the highest mean BOD5 at site 13 with 3.77 mg/L.

Table 24. BOD5 in Little Rock Creek

	Site 10	Site 13	Site BH2	Site S1	Site Z1
Minimum	0.50	0.50	0.80	0.90	0.50
Maximum	39.00	51.00	22.00	21.00	44.00
Mean	2.81	3.77	2.43	3.37	3.08
N	28	28	21	16	28

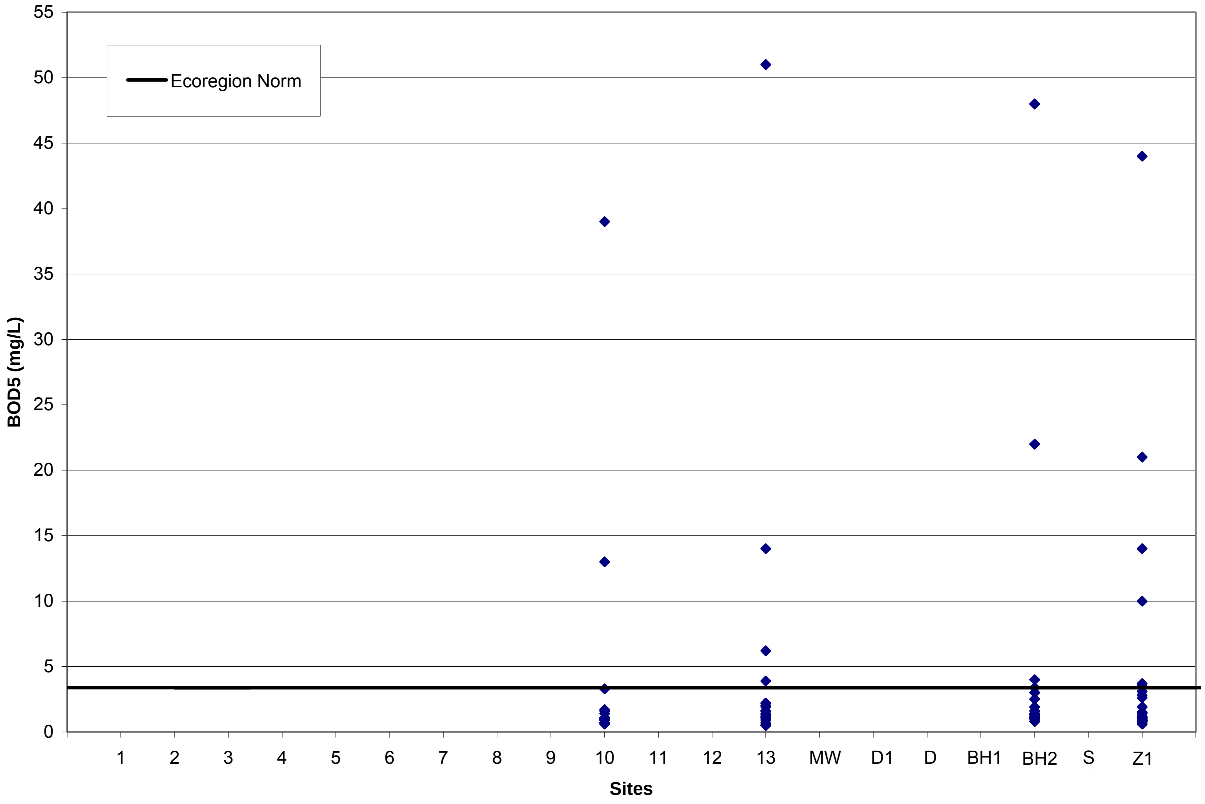

Figure 73. BOD5 measurements by site in Little Rock Creek

BOD demand increases as phosphorus levels increase. Phosphorus levels exceeded the 75th percentile values for least impacted streams in the North Central Hardwood Forests Ecoregion at sites 1, 2, 4, 10, 13, MW, D1, BH1, BH2, S1, and Z1 (Table 25 and Figure 74). The phosphorus ecoregion norm is 0.27 mg/L. Site 13 had the highest level of phosphorus collected in Little Rock Creek with 1.42 mg/L. The highest level in the Little Rock Creek watershed was in Bunker Hill Creek at BH2 with 2.09 mg/L of phosphorus.

Table 25. Phosphorus (mg/L) in Little Rock Creek

	Site 1	Site 2	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9	Site 10	Site 11
Minimum	0.08	0.23	0.06	0.10	0.06	0.06	0.07	0.04	0.03	0.04
Maximum	1.24	1.41	0.47	0.25	0.20	0.17	0.07	0.05	1.01	0.04
Mean	0.66	0.92	0.22	0.17	0.11	0.11	0.07	0.04	0.12	0.04
N	2	3	3	2	4	2	1	2	39	1

	Site 12	Site 13	Site MW	Site D1	Site D2	Site BH1	Site BH2	Site S1	Site Z1
Minimum	0.06	0.03	0.19	0.13	0.10	0.14	0.02	0.06	0.05
Maximum	0.06	1.42	0.43	0.42	0.10	0.43	2.09	0.82	2.05
Mean	0.06	0.14	0.31	0.28	0.10	0.29	0.23	0.19	0.23
N	2	40	2	2	1	2	32	24	36

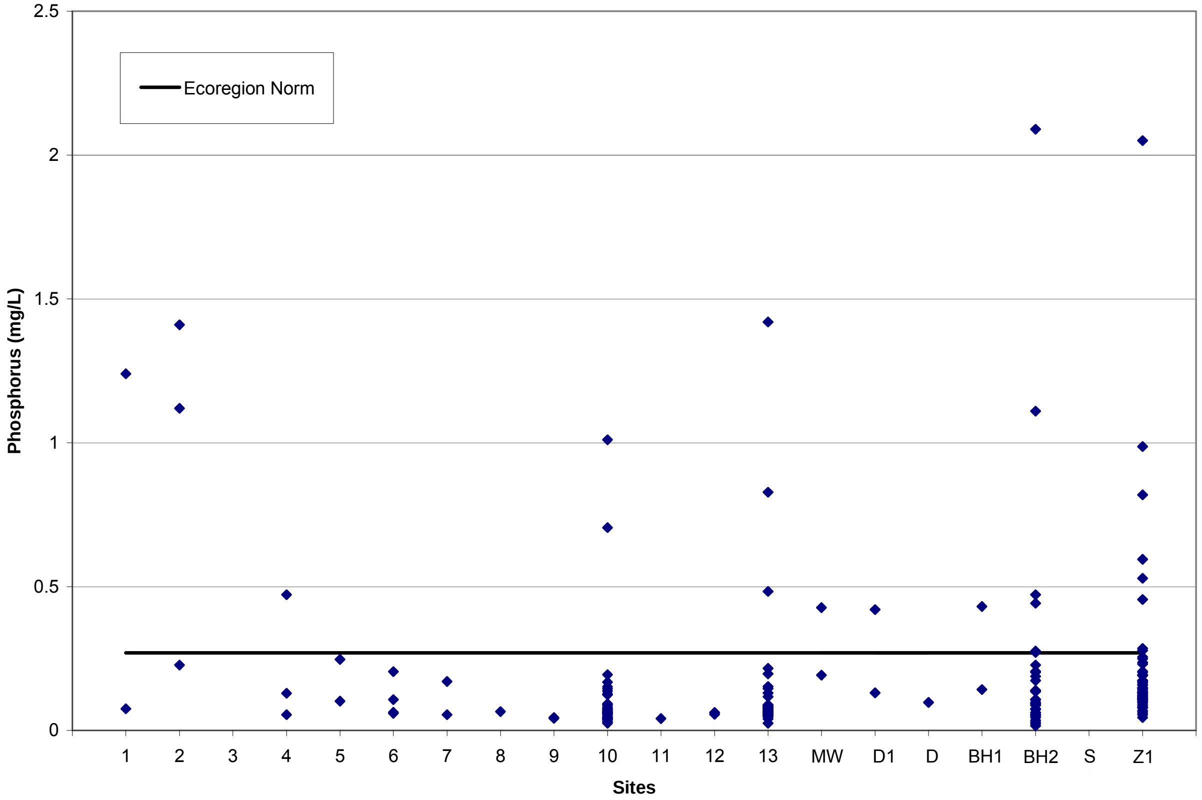

Figure 74. Phosphorus levels in Little Rock Creek

It is unknown if residential practices are contributing to the low dissolved oxygen levels. The impoundment may be contributing to low dissolved oxygen below the dam, as the levels at site 13 are above 7mg/L DO during the diurnal measurements, but below that value during some of the instantaneous measurements.

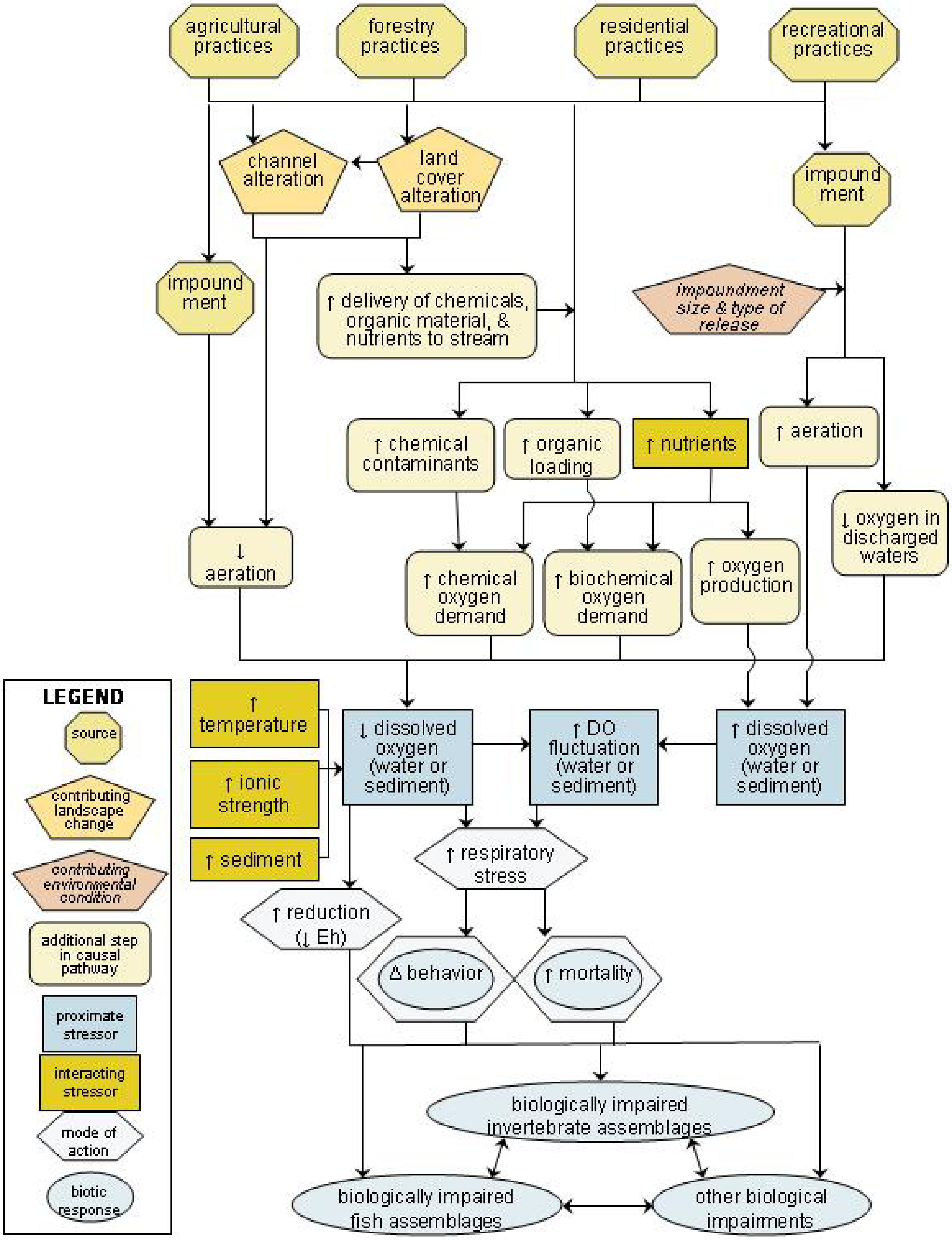

Figure 75. Model of sources and causal pathways for low dissolved oxygen in Little Rock Creek

Table 26. Sufficiency of evidence table for sources of low dissolved oxygen in Little Rock Creek

Types of evidence	Agricultural Practices	Forestry Practices	Residential Practices	Recreational Practices	Impound-ment
Evidence using data from Little Rock Creek					
Spatial/temporal co-occurrence	+	0	0	0	+
Evidence of exposure, biological mechanism	+	0	0	0	+
Causal pathway	++	0	0	0	+
Field evidence of stressor-response	0	0	0	0	0
Field experiments / manipulation of exposure	NE	NE	NE	NE	NE
Laboratory analysis of site media	NE	NE	NE	NE	NE
Temporal sequence	0	0	0	0	0
Verified or tested predictions	+	+	+	+	+
Symptoms	0	0	0	0	0
Evidence using data from other systems					
Mechanistically plausible cause	+	0	0	0	+
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	NE	NE	NE	NE	NE
Analogous stressors	NE	NE	NE	NE	NE
Multiple lines of evidence					
Consistency of evidence	+++	0	0	0	0
Explanatory power of evidence	++	0	0	0	0

Conclusion

The dissolved oxygen levels in Little Rock Creek were below the state standard for Class 2A streams. These levels show that low dissolved oxygen is a stressor to the biological community. Low levels of dissolved oxygen were measured throughout the creek and are likely due to a combination of impacts from agricultural practices.

CONNECTIVITY

Connectivity of a river system refers to the flow, exchange, and pathways that move organisms, energy, and matter through these systems (Annear et al. 2004). Connectivity is more complex than simply the upstream downstream longitudinal interruptions within a river system, such as dams however this section will focus on those stressors which are better understood on the Little Rock Creek. For more information regarding lateral, vertical and time connectivity in river systems, see Annear et al. 2004 and Ward 1989.

Three primary longitudinal connectivity stressors for the Little Rock Creek include: Little Rock Lake, Sartell Wildlife Management Area's wetland impoundment, rock-dams, and numerous culvert crossings. Each one of these structures can change the hydrology of the system, as well as larger watershed changes, which can create unintended impacts for both the physical and biological entities of the creek.

Connectivity Standard

Currently, there is no applicable water quality or biological standard for connectivity impacts. Further data collection would allow natural resource managers to compare the impact of this stressor versus others that have been discussed in this report.

Effects of connectivity within streams

Many riverine organisms rely on migrations up and downstream in order to complete various aspects of their life cycle. When this migration is impeded or even stopped, it can have significant impacts for the species involved. There are many examples of fish species that have been extirpated or suffer significant population declines due to the construction of dams. In Minnesota, the skipjack herring is one example of a once abundant fish species within the Mississippi, St. Croix and Minnesota Rivers and now is rarely ever sampled. Another effect of dams on riverine systems is the interruption of sediment, where the suspended and bedload sediments are dropped out in the impoundment because of the sudden decrease in velocity and competency of the stream to carry those particles. These sediments build upstream of the dam, while the downstream side can be sediment starved, which can induce channel incision. The impounded sediments can create logistical problems if the dam is ever removed. Finally, dams create impoundments and with the increased water surface area, water temperature can also increase. This relationship is especially important in Little Rock Creek, a cold water stream that had a self-sustaining population of brown trout throughout the century up until the late 1980's. The temperature in and of itself can be a stressor to cold water species, however with warming water temperatures, additional cool and even warm water species can become abundant and compete for available habitat and food.

Culverts can become barriers as well when placed incorrectly or a system wide shift in hydrology occurs that delivers more or less flow than the culvers were designed for. In Little Rock Creek, there are many culvert crossings that exhibit sediment aggradation upstream; while downstream a scour pool has developed. Over time, if these scour pools become deep and large enough, they can become fish barriers at low flows and increase water temperatures at all flows.

Finally, Little Rock Creek is also isolated from the Mississippi River and the rest of its downstream network by Little Rock Lake. The warm water, nutrient impaired lake posses a significant migratory barrier to sensitive, coldwater, and/or coolwater taxa (e.g. mottled sculpin, burbot, hornyhead chub, mimic shiner). These taxa are present in the Mississippi River and if they were to successfully colonize the creek, it would improve measures of Biological Integrity. Connectivity challenges may still exist even if Little Rock Lake's water quality were to improve; there will be a warm water thermal regime present which would continue to be a barrier to coldwater species, such as burbot & mottled sculpin.

Data evaluation for connectivity in Little Rock Creek

There are few data available to directly evaluate connectivity as a stressor in the Little Rock Creek watershed. However a system wide alteration of hydrology has been documented in the altered flow section of this report, which would likely result in connectivity being an issue. The magnitude of this issue is unknown. Due to time restrictions, only an outline of possible data collection/data analysis needs was reported (see list below). Additional data collection and analysis will be completed this fall by DNR staff prior to Phase 3.

Future data collection/analysis needs:

- Document any changes in temperature above and below water control structures and culverts.
- Determine at what flows, if any, do culverts become fish or other biotic community migration barriers and/or sediment barriers.
- Determine if the biotic sampling results indicate barrier issues.
- Document any relationship with the increased irrigation and groundwater connectivity.
- Overall, determine how mobile the bedload is and where is aggradation/degradation occurring within the watershed.

Models of Sources and Pathways for Longitudinal Connectivity Stressor

The sources and pathways involved with this stressor are shown in Figure 76. The model was developed by the Little Rock Creek TMDL technical team as a tool to evaluate the potential sources, pathways, and biological impact of this stressor in the lack of coldwater assemblage impairment. The stressor was evaluated using CADDIS, shown in Table 27.

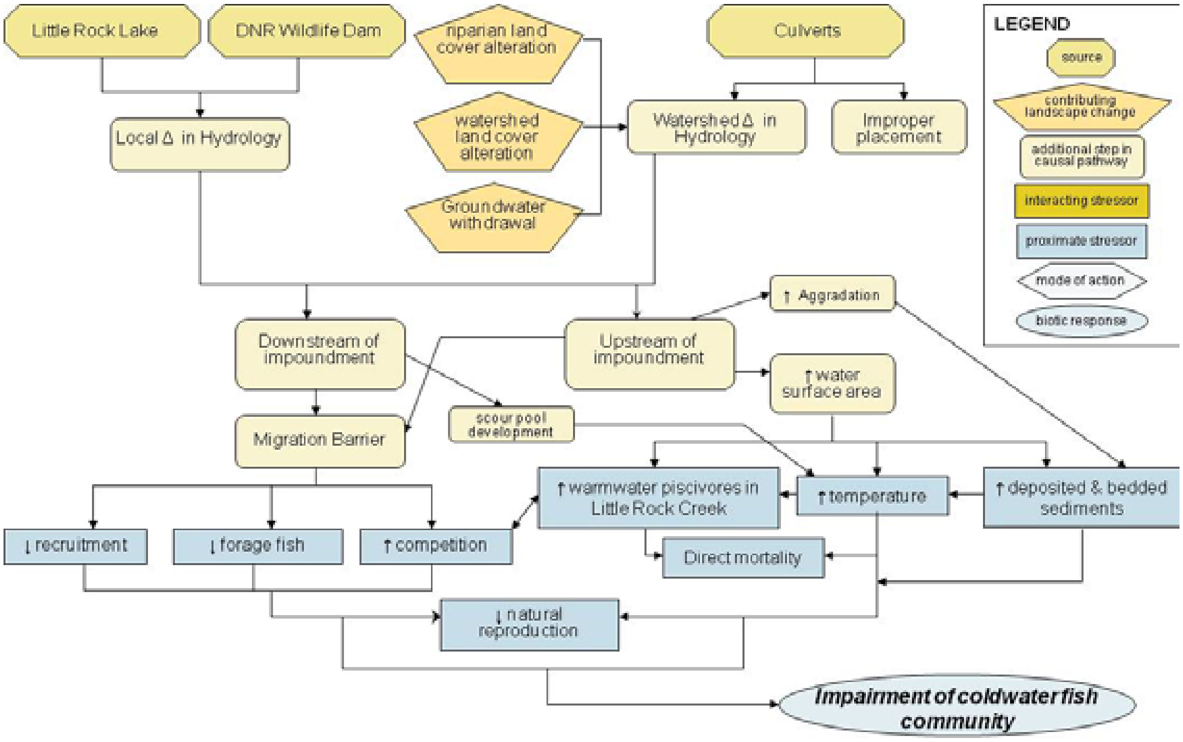

Figure 76. Model of sources and causal pathways for longitudinal connectivity stressors for Little Rock Creek

Table 27. Sufficiency of evidence table for sources of longitudinal connectivity stressors in Little Rock Creek

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

Conclusion

Quantifying the negative effects of connectivity in Little Rock Creek is difficult given the lack of data available. Connectivity is interrelated to hydrology in which flow is one of the primary biological stressors for this system. The presence of dams and numerous culverts at a minimum suggest migration barriers, which can have significant biological impacts as demonstrated by scores derived by using an index of biotic integrity or IBI.

PREDATION OF TROUT BY NORTHERN PIKE AND OTHER WARMWATER PISCIVORES

Undisturbed coldwater streams generally support few fish species which are adapted to thrive in temperature regimes ranging from 14-20 degrees Celsius. As a result of habitat degradation and illegal stockings, numerous coldwater streams around the world have been negatively affected by the presence of invasive predators typically associated with warmwater aquatic habitats. One well-known example of this stressor to coldwater fish comes from Montana’s Flathead River, where northern pike predation is responsible for the loss of over 16,000 native cutthroat and bull trout each year (CBB news bulletin, 2008). Little Rock Creek contains many of the same warmwater fish species (i.e. northern pike and walleye) that are responsible for the loss of coldwater fisheries in other areas.

Predation Standard

Currently, there is no applicable water quality or biological standard for predation impacts. Further data collection would allow fisheries managers to compare the impact of this stressor versus others that have been discussed in this report.

Effects of predation within streams

The primary pathway of this stressor is direct predation of adult or juvenile trout by warmwater species. Yet, indirect effects such as increased competition for food, refugia, and predator avoidance behaviors have also been observed in systems with introduced predators (Fraser and Cerri, 1992; Fraser and Gilliam, 1992). For example, Vehanen and Hamari (2004) found that the presence of northern pike in coldwater habitats reduced activity levels in brown trout and caused them to occupy different microhabitats typically not preferred by trout. These behavioral changes can lead to reductions in feeding efficiency, inability to avoid predation, and ultimately, reductions in survival or growth rates.

Data evaluation for predation in Little Rock Creek

Fisheries surveys on Little Rock Creek from 1975 through 2008 indicate the presence of five warmwater piscivore species that may be preying on and competing with coldwater species. These include northern pike (Esox lucius), walleye (Sander vitreus), largemouth bass (Micropterus salmoides), and burbot (Lota lota). Burbot have not been sampled in Little Rock Creek since 1992. Northern pike account for the majority of warmwater piscivores sampled in the creek (Figure 77), and are generally the largest in size when compared to other warmwater fish species found in LRC.

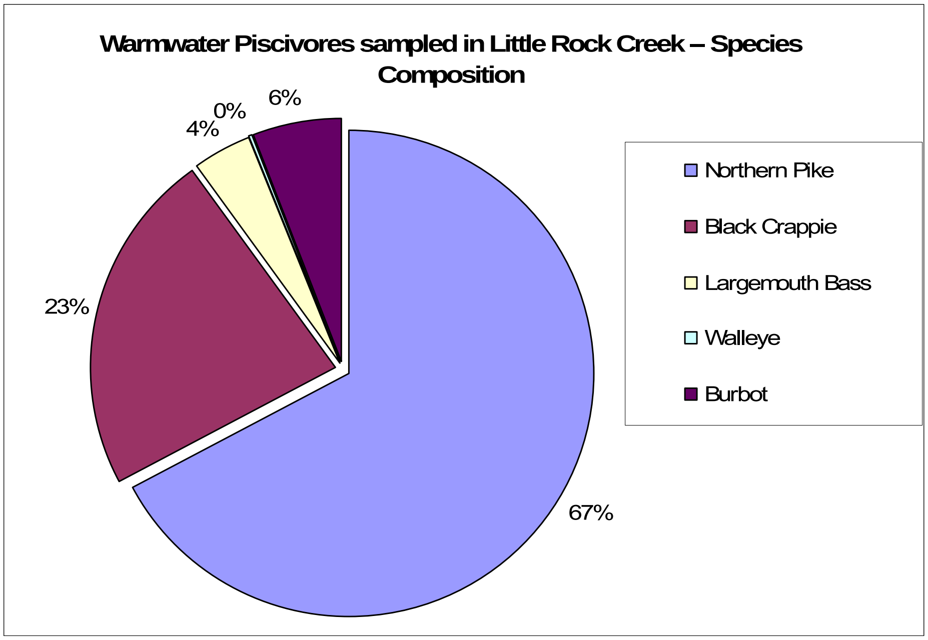

Figure 77. Species composition chart for warmwater piscivore species sampled in Little Rock Creek. Data includes all samples taken between 1975 and 2008.

Fish community sampling protocols of the Minnesota DNR (MN DNR) and Minnesota Pollution Control Agency (MPCA) call for sampling during summer months (June – August). The lack of fish community data from spring months may be significant for this specific stressor, as northern pike spawning activity has been documented within Little Rock Creek during that time of year. Summertime sampling may underestimate abundance and size of northern pike that can be considered predators of juvenile and adult coldwater fish in Little Rock Creek. Existing data suggest that most of the northern pike individuals occupying the creek during summer months are smaller adult fish (up to 17 inches) and young-of-year (YOY) individuals (Figure 78).

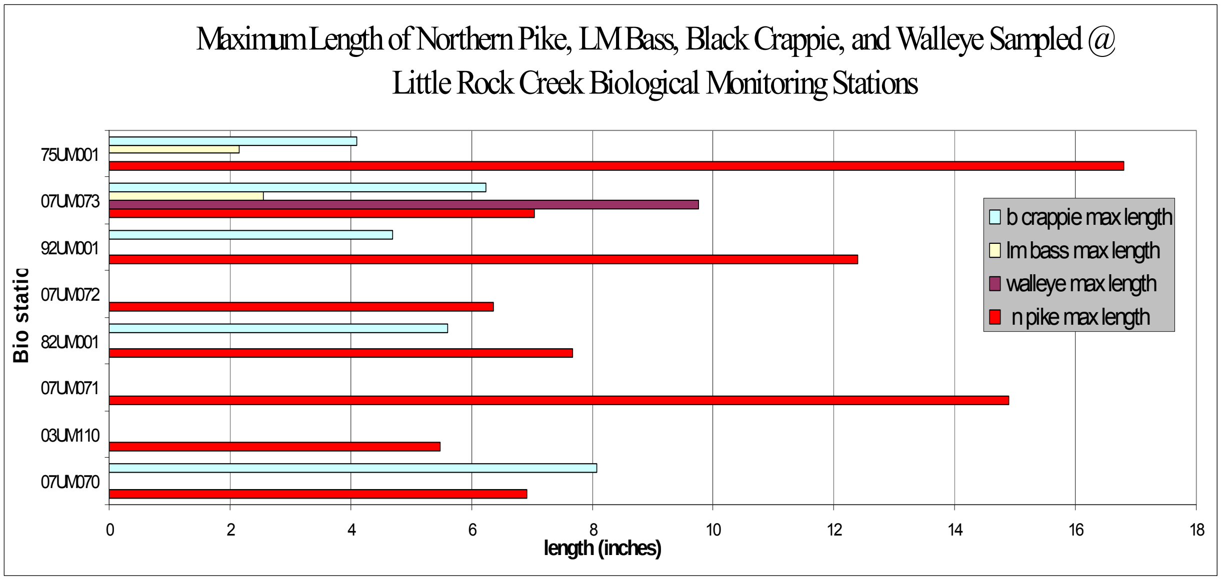

Figure 78. Maximum length of northern pike, largemouth bass, black crappie, and walleye individuals sampled at Little Rock Creek monitoring stations. Data spans 1975-2008.

Models of Sources and Pathways for Predation by Northern Pike

The sources and pathways involved with this stressor are shown in Figure 79. The model was developed by the Little Rock Creek TMDL technical team as a tool to evaluate the potential sources, pathways, and biological impact of this stressor in the lack of coldwater assemblage impairment. The stressor was evaluated using CADDIS, shown in Table 29.

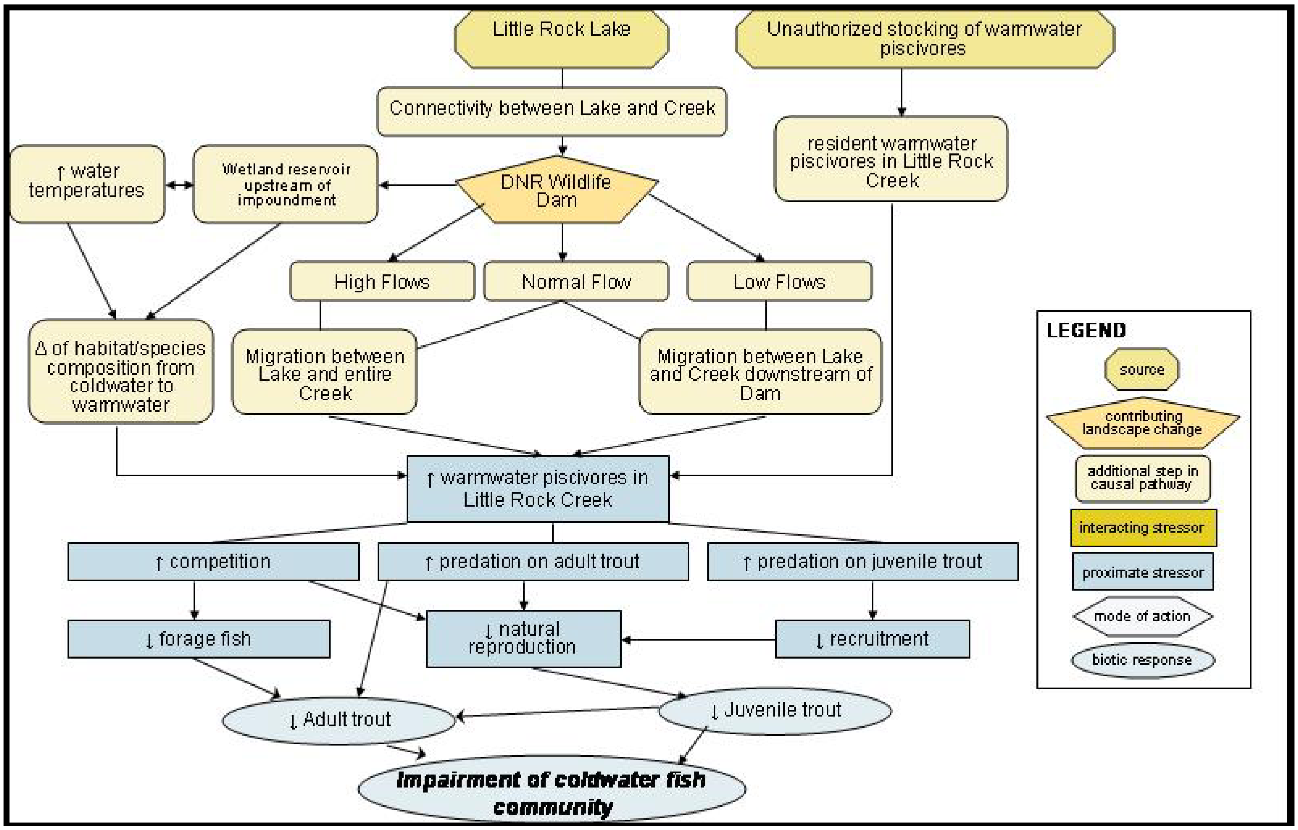

Figure 79. Model of sources and causal pathways for predation on trout by northern pike and other warmwater piscivores in Little Rock Creek

There have been no documented reports of illegal plantings of warmwater piscivores into LRC. It is likely that the main source of northern pike and other warmwater piscivores in Little Rock Creek is Little Rock Lake. Due to the colder temperature regime and general lack of biological productivity in LRC, these fish are likely occupying the creek seasonally or during specific life stages. The exception to this may be the downstream sides of several road crossings, where high flows have scoured out deep pools that remain throughout the summer.

As shown in the conceptual model in Figure 79, there are some uncertainties about the connectivity of Little Rock Lake and the upper reaches of the creek under normal to low streamflows. The DNR manages a wildlife dam approximately 1 mile upstream of Little Rock Lake, and the structure appears to be a barrier to fish passage under summer baseflow conditions. Not surprisingly, this seems to result in a higher density and percentage of warmwater piscivores present in the fish community downstream of the dam (Figure 80). Although connectivity may be limited under low flow conditions, frequent spring flooding in the lower reaches of the creek likely allow northern pike and other warmwater species access to stream reaches upstream of the dam. Warmwater piscivore species have been documented at every monitoring station except site 2, which is the upstream-most biological monitoring site and appears to be an intermittent reach.

The presence of northern pike and other warmwater fish above and below the dam indicate that predation on trout is feasible throughout LRC, and possibly other coldwater tributaries (i.e. Bunker Hill Creek). The available fisheries data suggest that coldwater fish populations downstream and immediately upstream of the dam are most vulnerable to predation by northern pike and other warmwater fish.

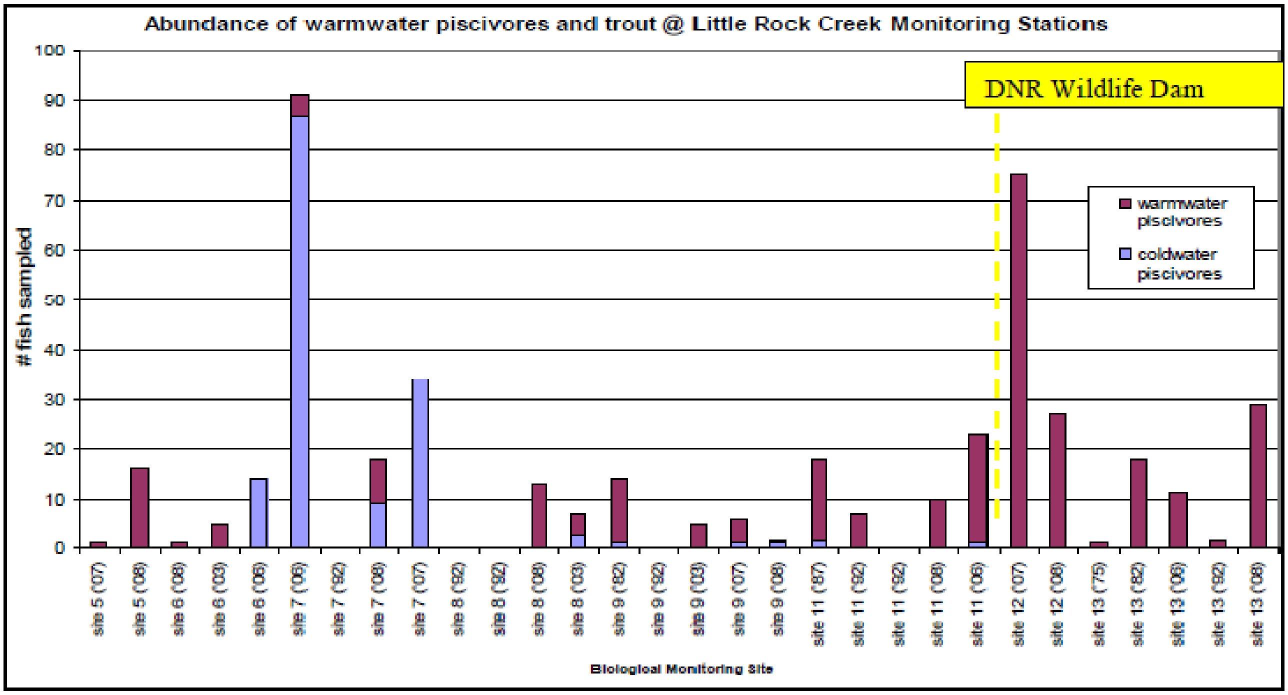

Figure 80. Total number of warmwater piscivore fish surveyed by year at LRC monitoring stations. Sartell Wildlife Management Area dam is located between site 11 and 12

Northern Pike Trends in Little Rock Lake

Historical fisheries data from Little Rock Lake suggests that northern pike abundance and size may be increasing. Data from 1960, 1974, 1980, 1990, and 1996 show an increase in the amount of fish and total fish weight during gillnet surveys (Figure 81). With a greater abundance of northern pike in the lake, especially larger individuals, it is likely that more fish would be using Little Rock Creek for spawning purposes in the spring.

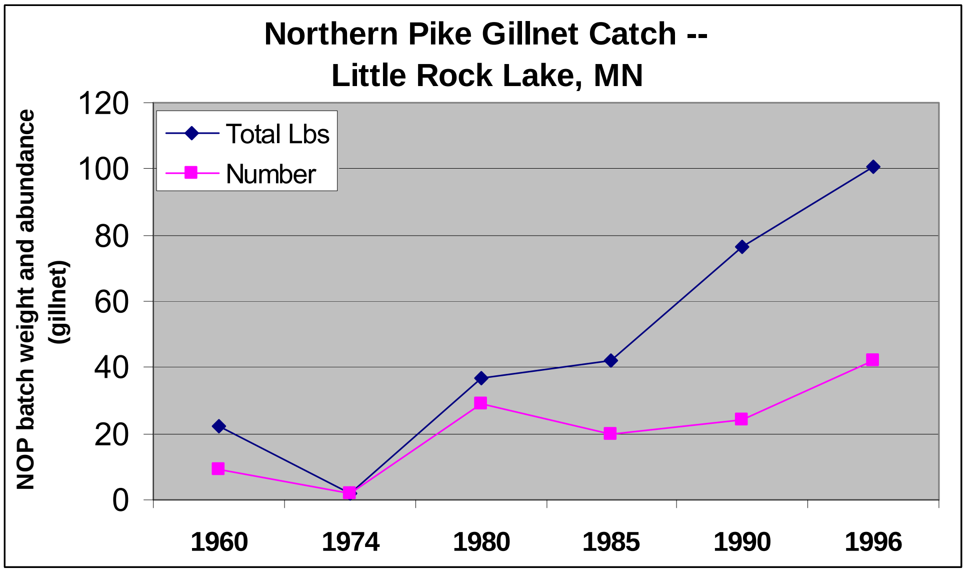

Figure 81. Abundance and total weight of northern pike caught during Minnesota DNR gillnet survey

Minnesota DNR has attempted to decrease northern pike spawning activity in LRC. The presence of the Sartell Wildlife Management Area impoundment and the lack of a suitable site for a barrier makes northern pike control difficult. Stop logs were removed and the impounded area drawn down during the winters of 1996 through 1998 to cause winterkill and/or movement of northern pike and carp to downstream reaches. Stop logs were replaced prior to the spring pike spawning run to prevent northern pike from returning to Little Rock Creek. The effectiveness of this drawdown is unknown. Recent data shows pike are regularly found at almost every sampling site on the creek, so it appears that northern pike activity in the stream is still an issue of concern.

Conceptual Model – Proximate Stressors

The conceptual model in Figure 79 presents the proximate stressors and specific biological responses associated with them. The proximate stressor, or causal agent, is the stressor that directly induces the biological response. Proximate stressors used in the conceptual model for predation of trout by northern pike and other warmwater piscivores include:

(1) Predation on adult and juvenile trout
(2) Competition for available habitat and food resources

The following section will present data from the case to evaluate these proximate stressors and their association with the impairment.

Evaluation of Evidence - Predation on Adult and Juvenile trout

Northern pike are top predators with the ability to deplete or completely eliminate other fish species in a waterbody (He and Kitchell, 1990; Vashro, 1990). To date, there have been no documented cases of pike predation on adult trout within Little Rock Creek. This is likely due to a lack of direct monitoring focused on this predator-prey relationship. Gut content analysis of larger northern pike, especially the mature fish that use the creek for spawning purposes in the spring, would provide a valuable data set for further evaluating this stressor.

The size of prey northern pike and other warmwater piscivores can ingest is limited by gape size. Gape size refers to the width of the mouth opening, which in northern pike, is calculated as a linear function of body length (Gape = 0.098TL – 0.339, where TL is total length of northern pike) (Nilsson and Bromark, 2000). Nilsson and Bromark (2000) calculated gape sizes for northern pike of various lengths and then determined the maximum prey body depth of ingestible prey using fishes from the cyprinid family, common bream (*Abramis brama L.*) and roach (*Rutilus rutilus L.*). Relationships of pike body length, gape size, and prey body depth are detailed in Figure 82, taken from Nilsson and Bronmark (2000).

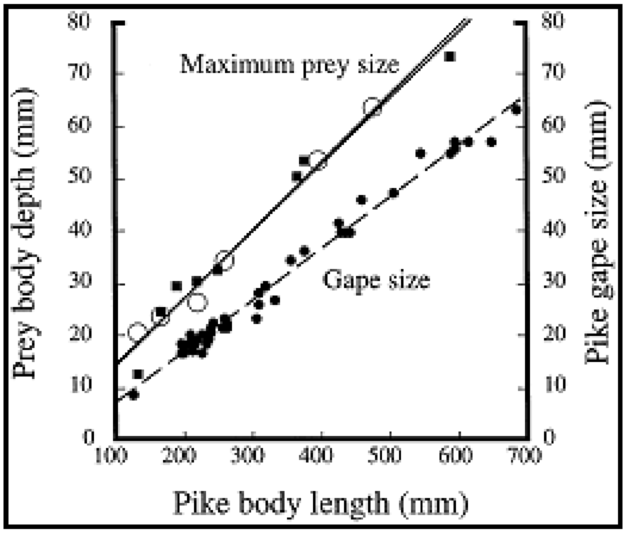

Figure 82. Pike gape size measurements (Nilsson and Bromark, 2000)[11]

Understanding the gape limitations of predators in a waterbody provides useful information on predator – prey interactions. With adequate data, these interactions can be explored with scientific models. Hambright et al. (1991) developed a model to calculate the vulnerabilities of planktivore fish to predation by piscivores based on predator gape size and prey body depth. Hyvarinen and Vehanen (2004) applied this model to a coldwater fishery in Finland to predict and validate the relative vulnerability of age-3 and age-4 stocked brown trout to northern pike predation. They found that age-3 brown trout were much more vulnerable to predation than age-4 brown trout, and that trout with body depths that exceeded the largest pike gape size enjoyed complete refuge from the pike predation.

Unfortunately, MPCA and MN DNR fish sampling protocols do not provide the data necessary to employ the Hambright model to predict vulnerability of trout to pike predation in LRC. MN DNR did not measure lengths of warmwater fish in LRC until 2006, while MPCA measures only the minimum and maximum length for each species captured. However, it is possible to investigate the vulnerability of brown and brook trout to predation in LRC using available pike gape size and trout body depth measurements.

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[11] Pike gape size measurements (broken line, filled circles) and maximum ingestible prey size as a function of pike body length. Pike gape size was measured with a cone, whereas the maximum ingestible prey size (solid lines) was obtained as the body length of the largest prey fish (common bream, filled squares, and roach, open circles) ingested in laboratory experiments.

Brown trout body depth can be calculated using the equation developed by Hyvarinen and Vehanen (2004), body depth = (0.222)(TL) – 2.403 (TL = total brown trout length). No body depth calculations were available for brook trout. For purposes of the stressor identification, the equation for brown trout body depth can applied to brook trout to provide some estimate of brook trout vulnerability to predation. Table 28 shows the maximum length of ingestible brown trout based on the maximum length of northern pike sampled at each LRC monitoring station. For example, the longest northern pike sampled at site 6 was 137 mm; the approximate gape size of that fish was 13.1 mm; prey body depth limit using Nilsson and Bronmark (2000) for that fish was 18.3 mm, which would be about a 93.5 mm (3.68 in) brown trout in length.

Table 28. Northern pike maximum length, gape size, approximate limit for prey body depth, and estimated maximum length of brown trout that can be considered prey

LRC TMDL Site	# NP	Max NP Length	Max NP gape size (mm)	Approx body depth limit of prey for Max length NP (mm)***	Length of Max Ingestible BNT (in)
5	10	126	12.0	16.9	3.4
5	1	173	16.6	23.1	4.5
6	1	137	13.1	18.3	3.7
7	4	373	36.2	49.3	9.2
7	9	216	20.8	28.7	5.5
8	12	192	18.5	25.6	5.0
9	1	150	14.4	20.1	4.0
9	5	159	15.2	21.2	4.2
11	21	310	30.0	41.0	7.7
11	10	202	19.5	26.9	5.2
12	26	176	16.9	23.5	4.6
12	19	149	14.3	19.9	4.0
13	23	186	17.9	24.8	4.8
13	7	422.5	41.1	55.7	10.3

***** based on Nilsson and Bronmark data using Roach as prey species**

Based on the size structure of northern pike sampled in Little Rock Creek, it appears that brown trout lengths of 3 to 5 inches would be the most vulnerable to predation. Some of the larger northern pike sampled in the creek would have the ability to consumer larger adult fish around 9-10 inches in length. Without length-frequency data for northern pike and other warmwater piscivores, it is difficult to accurately model this predator-prey relationship. The available data suggests that adult trout greater than 7 inches in length may have considerable size refuge from northern pike predation, at least based on the data collected during summer months.

Catch per Unit Effort (CPUE) trends for Northern Pike and Trout

The frequent stocking of trout in LRC makes it difficult to use trout abundance as an accurate measure of stream health. Hatchery raised fish are not fin-clipped in LRC as they are in some other coldwater streams, so it is very difficult to distinguish a wild fish from a hatchery fish. In a natural system without stocking activity, it would be possible to evaluate the interactions of northern pike and trout using spatial co-occurrence and temporal trends.

However, catch per unit effort (CPUE) figures for young of the year (YOY) trout can be used as measure of stream health. Fluctuations in number of YOY trout in LRC are more closely connected to the chemical, physical, and biological attributes of the stream, and less so to management activities. CPUE numbers are calculated using the number of individuals captured per unit of time. For example, Little Rock Creek CPUE data was calculated as # fish captured per hour of electrofishing.

Figure 83 shows CPUE figures for YOY trout and northern pike. No trout YOY were found during the 1995, 1996, and 1997 fisheries surveys. Northern pike YOY were abundant during the 1995 and 1997 surveys, but were not present in large numbers in 1996. The CPUE figures for trout and northern pike YOY seem to be negatively correlated for the years 1998 through 2006. This could be a result of predation or interspecific competition, but it may also be indicative of habitat conditions that were more favorable to warm or coldwater fish species (i.e. water temperatures, dissolved oxygen).

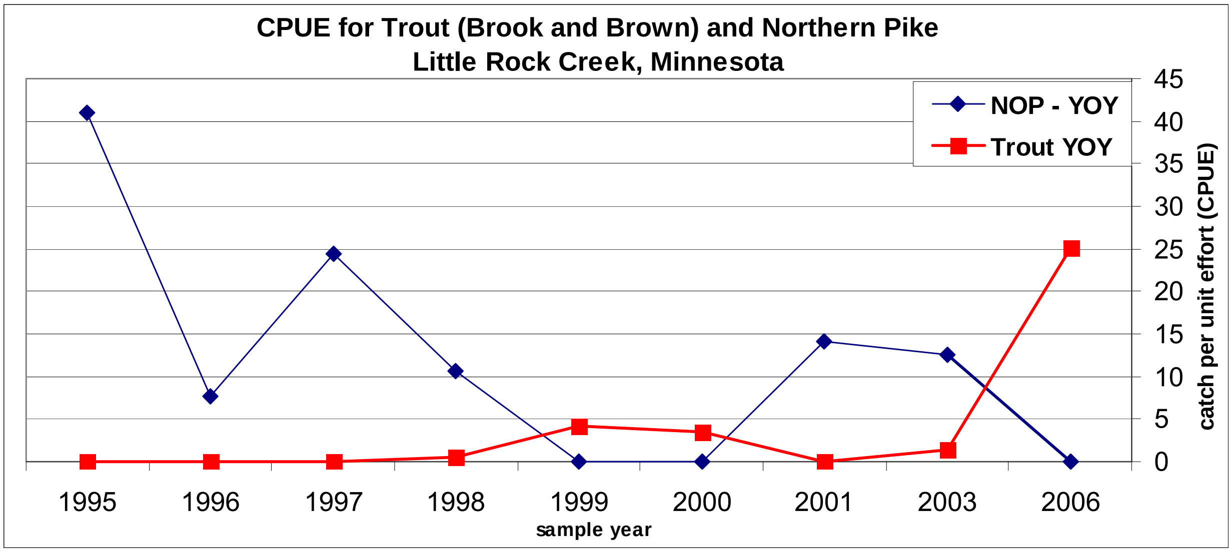

Figure 83. Catch per unit effort (CPUE) of trout and northern pike young of the year YOY). Data collected by MN DNR fisheries Little Falls office

Figure 84 again shows CPUE for trout YOY, but the northern pike CPUE is shown with YOY and adult fish combined. Many of the same patterns emerge when looking at the data this way. However, in 2006, CPUE for trout YOY was at a nine-year high with higher than average CPUE for northern pike. This may be evidence that natural reproduction of LRC trout, as represented by YOY data, is driven by factors other than northern pike abundance.

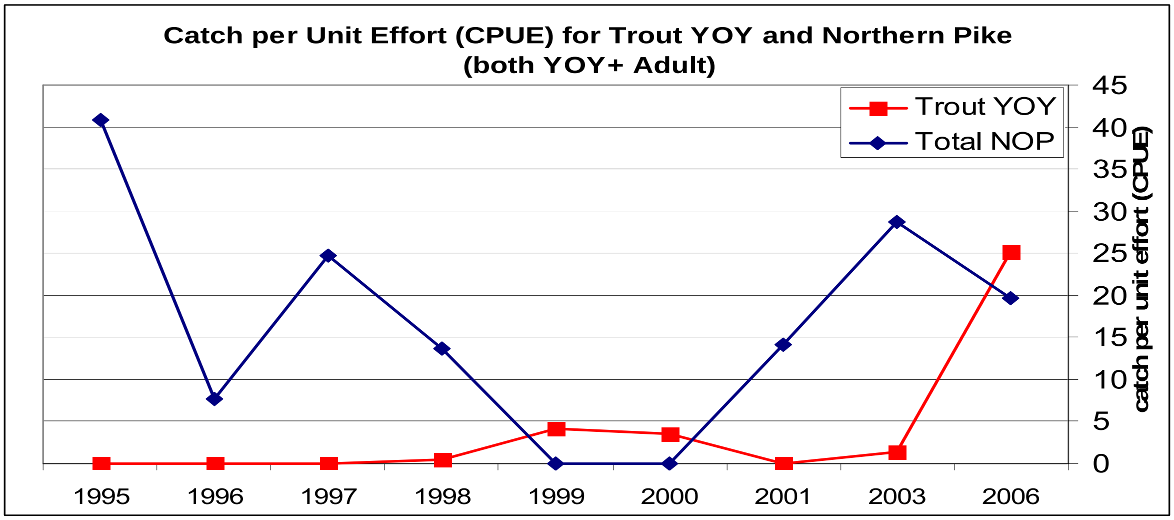

Figure 84. Catch per unit effort (CPUE) of trout YOY and northern pike (YOY and adult fish combined)

LRC Trout Stocking and Vulnerability to Predation

Coldwater fish populations in Little Rock Creek have been heavily supported by stocking for many decades. Brown trout fry were originally stocked in LRC at the mouth near Little Rock Lake back in 1908. A naturally reproducing brown trout population was sustained from the original stocking up until the late 1980's. Stocking efforts were started again in the early 1960's with plantings of brook trout, and have continued on and off to the present day. Recent management strategies have focused primarily on the stocking of fingerling, yearling, and adult brook trout Natural reproduction of both brook and brown trout has been mostly unsuccessful in recent years. A complete stocking history for Little Rock Creek is located in the Appendix.

The age/size and of stocked trout, and the timing of the stocking may play a significant role in the susceptibility of trout to predation by northern pike and other warmwater piscivores in LRC. Based on available fisheries data, the stocking of adult fish greater than 6" could provide the introduced fish with size refuge from the majority of northern pike that occupy LRC during summer months. Still, the ability of these fish to holdover for multiple seasons will depend on their ability to survive during subsequent spring pike migrations where larger pike are found in the stream. In addition, abiotic stressors, such as high water temperatures and elevated nitrate concentrations (discussed in other sections) ultimately play an important role in whether or not stocked trout survive in LRC.

Understanding the size classes, timing of spawning runs, and the use of LRC and its tributaries by warmwater piscivores during summer months would provide valuable information that could be used for evaluating stocking practices and other management goals for the coldwater fishery. The current data set does not provide the detailed information to model predator-prey scenarios in LRC. Modeling, along with field studies such as gut-content analysis of northern pike would provide a better base of evidence to determine where this stressor ranks in importance in comparison to other stressors covered in this study (low DO, sedimentation, etc.).

Increased Competition from Northern Pike

The presence of northern pike and other warmwater piscivores in LRC likely alters the behaviors of coldwater fish co-inhabiting the stream. An increase in predators commonly results in restricted habitat use by species lower on the food chain, as potential prey shift their location to reduce encounters with predators. Studies have shown that when prey species sacrifice foraging opportunities to minimize predation, growth rates decrease as a result of a reduction in feeding efficiency (Cooper, 1984).

Competition between trout and warmwater species may be intensified in Little Rock Creek due to the limited amount of quality habitat available for fish and other aquatic life. Physical habitat and geomorphic assessments of Little Rock Creek conducted by MPCA and DNR have produced concerns about excess fine sediment in the stream. The apparent inability of the stream to transport sediment has resulted in a lack of deep pools, riffle habitat, and general holding areas for adult and juvenile fish (see Sediment section). These physical habitat limitations likely create greater competition for food and refugia between coldwater and warmwater fish in LRC.

Table 29. Sufficiency of evidence table for sources of predation on trout by northern pike and other warmwater piscivores in Little Rock Creek

Types of Evidence	predation and competition from warmwater piscivores
Evidence using data from Little Rock Creek	
Spatial / temporal co-occurrence	0
Evidence of exposure, biological mechanism	NE
Causal pathway	++
Field evidence of stressor-response	0
Field experiments / manipulation of exposure	NE
Laboratory analysis of site media	NE
Temporal sequence	0
Verified or tested predictions	NE
Symptoms	NE
Evidence using data from other systems	
Mechanistically plausible cause	+
Stressor-response in other field studies	++
Stressor-response in other lab studies	NE
Stressor-response in ecological models	+
Manipulation experiments at other sites	+
Analogous stressors	+
Multiple lines of evidence	
Consistency of evidence	0
Explanatory power of evidence	0

Conclusion

Quantifying the negative effects of warmwater piscivores in Little Rock Creek is difficult given the data that is available. The presence of northern pike at nearly every biological monitoring station during summer months suggests that trout are subject to direct predation, as well as competition for available habitat and food. The brook and brown trout fingerlings currently used to stock the stream may be especially vulnerable to predation given their smaller size.

FINAL CONCLUSIONS

Strength of evidence analyses revealed that several candidate stressors may be contributing to the biological impairment of Little Rock Creek. Based on the evidence available, it is probable that altered flow, temperature, sediment, dissolved oxygen, and nitrates may be causing a biological impairment in Little Rock Creek. Furthermore, altered flow is a dominant stressor because it serves as a step in the causal pathways of other stressors including bedded sediment and temperature.

Although altered flow is a dominant stressor in Little Rock Creek, EPA does not believe that flow, or lack of flow, is a pollutant as defined by CWA Section 502(6). This is because EPA interprets section 303(d)(1)(C) to require that TMDLs be established for “pollutants” and does not believe “low flow” is a pollutant (Federal Register 2000).

Instead, low flow is a condition of a waterbody (i.e., a reduced volume of water) that when man-made or man-induced would be categorized under the CWA as pollution, provided it altered the physical, biological, and radiological integrity of the water. Many forms of human activity, including the introduction of pollutants can cause water pollution. Not all pollution causing activities, however, must be analyzed and allocated in a TMDL. Section 303(d) is a mechanism that required an accounting and allocation of pollutants introduced into impaired waters (whether from point or nonpoint sources). If low flow in a river, even if man-induced, exacerbates or amplifies the impairing effect of a pollutant in that river by increasing its concentration, that factor is to be accounted for and dealt with in the TMDL by calculating and allocating the total pollutant load in light of, among other things, seasonal variations in flow (Federal Register 2000).

Subsequently, in Little Rock Creek, TMDLs will be developed for temperature, bedded sediment, nitrates, and dissolved oxygen, by calculating the total pollutant load with reference to flow as the source of impairment.

Evidence regarding predation on trout by pike and other warmwater piscivores and connectivity was inconclusive due to insufficient evidence, but suggests that each has the potential to contribute to the biological impairment. Further monitoring could shed light on the potential effects of predation and connectivity as stressors to the biological community of Little Rock Creek.

A fecal coliform impairment was identified in this study. Additional data collection may be necessary to determine the spatial and temporal trends of the impairment. It will be addressed as a separate aquatic recreation and human consumption use impairment in Phase 3.

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APPENDIX