Ann River Stressor Identification Report

A study of local stressors causing a lack of biotic communities in the Ann River Watershed in Kanabec County

Minnesota Pollution Control Agency

September 2011

Legislative Charge

Minn. Statutes § 116.011 Annual Pollution Report

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Executive Summary

The objective of this report was to evaluate the environmental data available for the Ann River watershed to diagnose the probable causes of biological impairment. Numerous candidate causes for impairment were evaluated using U.S. Environmental Protection Agency's (EPA's) Causal Analysis/Diagnosis Decision Information System (CADDIS), Minnesota Pollution Control Agency's (MPCA's) biological Total Maximum Daily Load (TMDL) protocols, and weight of evidence analysis. The results of the Stressor Identification analysis pointed to five probable causes for the biological impairment in the Ann River. These include:

- loss of habitat due to substrate embeddedness
- low dissolved oxygen (DO) concentrations
- habitat loss resulting from riparian corridor degradation
- loss of connectivity due to impoundment structures
- altered flow regime due to impoundment structures

Loss of habitat due to sedimentation appears to be most problematic in the lower reaches of the river, which are lower in gradient and serve as depositional areas for sediment from upstream sources. Observations made during stream reconnaissance efforts indicate that agricultural land-uses (primarily cattle grazing) are a significant source of sediment delivery in the watershed. In addition, historical logging, and the use of the Ann River as a log driving waterway may also play a role in present day sediment dynamics. Sediment deposition in the lower Ann River reduced pool and riffle habitat quality, which resulted in a lack of game fish and fish species that depend on coarse substrates for feeding and reproduction.

Synoptic longitudinal and continuous (diurnal) measurements for dissolved oxygen were conducted at monitoring stations on the Ann River and Little Ann River during the summers of 2007 and 2008. The data indicates that dissolved oxygen concentrations in the Ann River occasionally drop below the standard of five mg/L during mid to late summer months. Most of the standard violations occurred in the early morning hours before sunrise. This stressor appears to be systemic in nature (watershed-wide) and may be linked to an altered flow regime, increased water temperatures resulting from a lack of stream shading, and climactic events (i.e. drought).

Changes in channel morphology are considered a candidate stressor in the Ann River watershed due to the degradation of riparian buffers and habitat. This disturbance appears to be causing increases in channel width to depth ratios, loss of pool and riffle habitat, decreases in stream shading and woody debris inputs. Channel widening, gully formation, and other erosion processes within the stream corridor appear to be contributing higher than normal sediment loads to the river.

Several impoundment structures located in the Ann River watershed may be altering stream flow and/or impeding fish passage. Extreme low flows were observed in 2007 and 2008 as much of the area experienced drought condition. Although sustaining adequate base flows are an issue that many undisturbed streams in the region face as well, we hypothesize that the impoundments are exacerbating low flow conditions and potentially stressing fish and invertebrate life in the river. Further evaluation of these conditions is needed to determine if they were the result of climactic events or flow alteration due to the impoundment of Ann Lake.

These five stressors and their connections to biological impairments on the Ann River will be evaluated in this report. The EPA's Stressor Identification (SID) and CADDIS will be used to determine key stressors and their sources.

1.0 Background Information and Impairment Description

The Ann River (AUID: 07030004-511) was listed on the 303(d) list of impaired waters in 2002 for failure to meet fish index of biological integrity (IBI) criteria established for [1]class 2B* waters of the St. Croix river basin. It is expected that the same reach will be listed for non-support of the aquatic macroinvertebrate index of biological integrity (IBI) when the 2010 303(d) list is released. The impaired reach for both fish and macroinvertebrates extends from the headwaters of the Ann River (Ann Lake) to the confluence with the Snake River at the outlet of Fish Lake (Figure 1).

The original impairment listing was based on fish community assessments conducted at site seven in 1996 and Site nine in 1998 (Figure 2). Both of these sites scored below the fish IBI criteria, however, Site 7 did score within the confidence interval for the IBI.

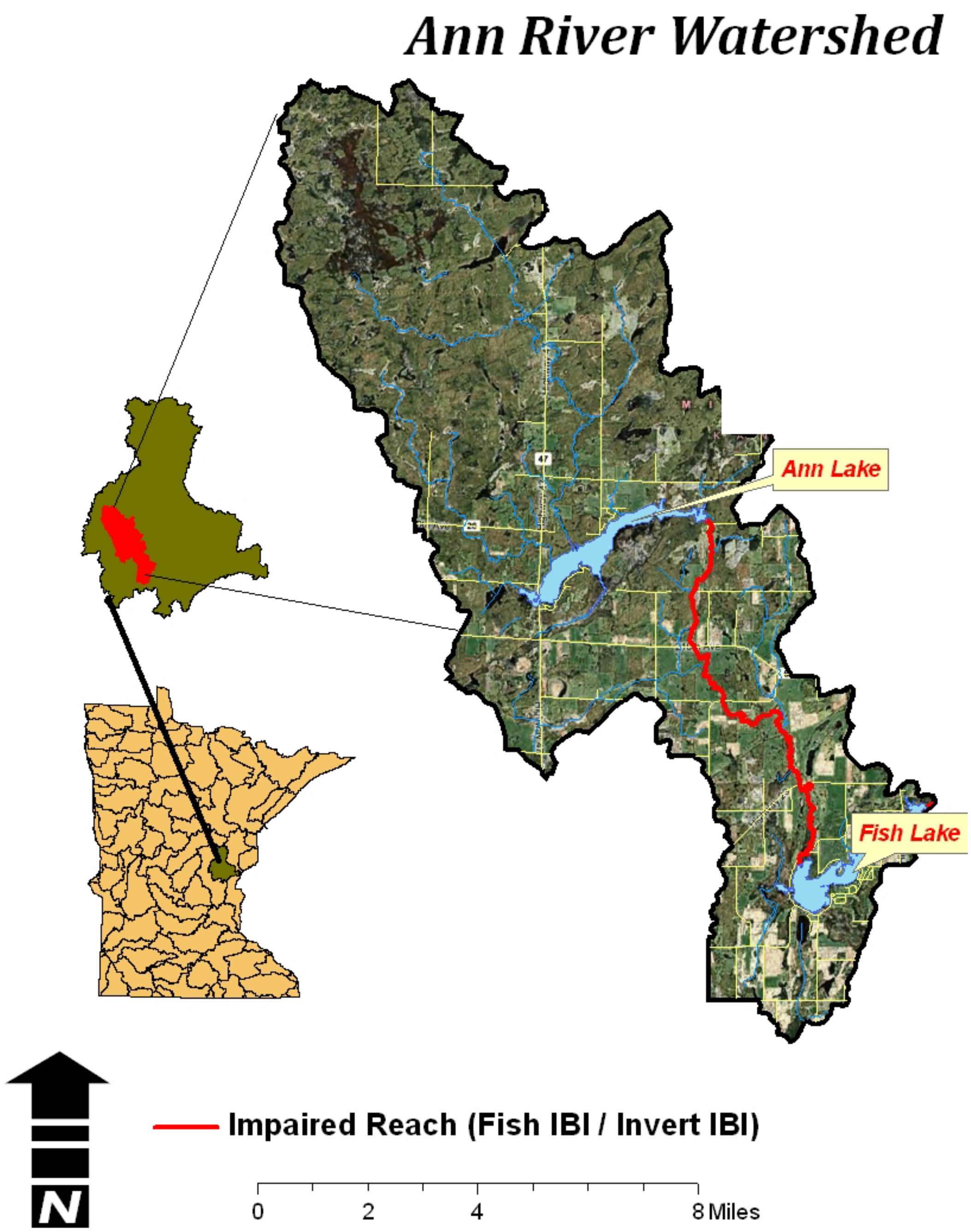
Since the original listing, three additional monitoring stations were established in the watershed. Fish and macroinvertebrate sampling was carried out per MPCA protocols at these new sites, and a selection of existing sites from 2006 - 2008. The data collected over this time period verified fish IBI impairments, and strengthened the case for listing the river as impaired for failure to meet macroinvertebrate IBI criteria.

Although Ann River fish IBI scores are below established standards for the St. Croix basin, the scores do not indicate severe impairment. Several sampling events produced fish IBI scores above the standard or within the established confidence interval. The exception may be Site 9, which has scored well below the fish IBI standard on two out of three surveys.

Drought conditions led to extremely low flows during macroinvertebrate sampling efforts in 2006 and 2007. As a result, the majority of macroinvertebrate data from these sampling events was not used for assessment purposes. A 1996 sampling event at Site 7 produced a macroinvertebrate IBI score below the standard, which is the basis of the invertebrate impairment listing.

[1] *** Class 2B water =** The quality of Class 2B surface waters shall be such as to permit the propagation and maintenance of a healthy community of cool or warm water sport or commercial fish and associated aquatic life, and their habitats. These waters shall be suitable for aquatic recreation of all kinds, including bathing, for which the waters may be usable. This class of surface water is not protected as a source of drinking water.

Figure 1. Map of Ann River Watershed



1.1 Monitoring Approach/Station Descriptions

1.1.1 Biological monitoring

A total of five biological monitoring sites were sampled on the main stem of the Ann River. There are an additional two biological monitoring stations on the Little Ann River, but the results from those stations will not be discussed in this report. The Little Ann River has a significantly smaller drainage area and currently meets state standards for aquatic life. Due to differences in drainage area, sites from the Ann River and Little Ann River cannot be accurately compared in terms of the biological assemblages they can be expected to support. Table 1 provides a list of Ann River watershed monitoring sites and associated identification numbers.

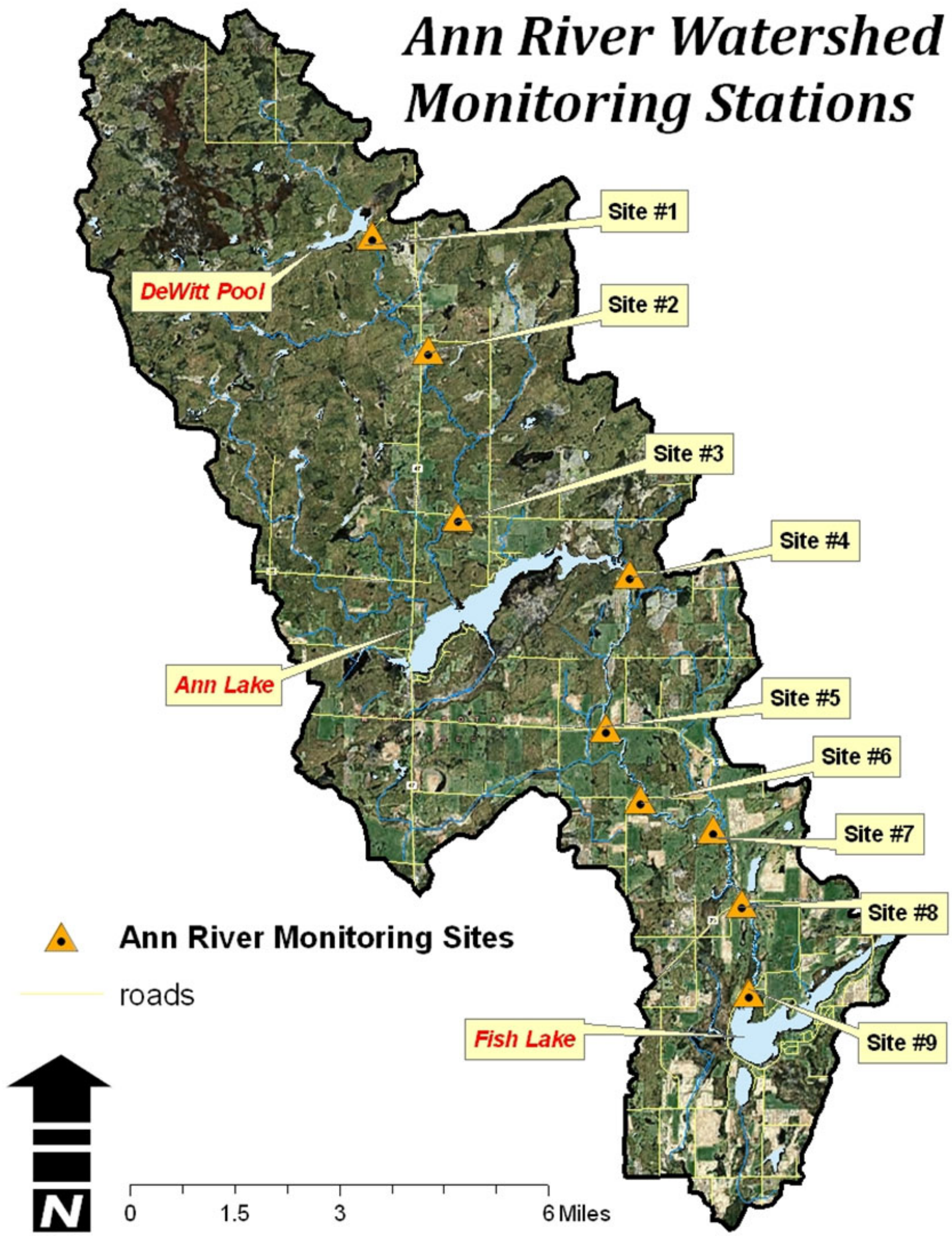
Table 1. MPCA monitoring stations in the Ann River Watershed

Site ID	STORET/biological Site FIELD ID	Location(RM = miles upstream of mouth)	Parameter(s) collected
Site 9	S004-782 / 98SC019	Ann River (RM 0.7)	Biological / Water Chemistry
Site 8	S004-066 / 06SC122	Ann River (RM 1.9)	Biological / Water Chemistry
Site 7	96SC021	Ann River (RM 4.3)	Biological
Site 6	S005-530 / 06SC136	Ann River (RM 5.7)	Biological / Water Chemistry
Site 5	S004-392	Ann River (RM 7.1)	Water Chemistry
Site 4	S004-640 / 07SC006	Ann River (RM 9.5)	Biological / Water Chemistry
Site 3	S004-393 / 06SC138	Little Ann River	Biological / Water Chemistry
Site 2	96SC004	Little Ann River	Biological
Site 1	S004-862 / S004-862	Little Ann River	Water Chemistry

Sampling was conducted to assess fish and aquatic macroinvertebrate populations per MPCA's biological sampling protocols. A total of 10 fish assessments and 8 macroinvertebrate assessments were conducted on the Ann River by MPCA staff from 1998 – 2008. Biological monitoring stations ranged in length from 250 m to 441m, with an average distance of 322.8 m. In additional to fish and macroinvertebrate collection, quantitative habitat data, instantaneous flow, and water chemistry samples were collected at each visit to the sites. Chemistry samples were analyzed for ammonium (NH4), total nitrogen (TN), total phosphorous (TP), and total suspended solids (TSS).

Minnesota Department of Natural Resources (DNR) fisheries staff based out of Hinckley conducted a fisheries survey of the Ann River in September and October of 2006. The survey involved four stations on the main stem of the river at the locations shown in Figure 2. DNR station lengths ranged from 270 m to 512 m. Fish collection involved species identification and counts, minimum and maximum length measurements, and batch weights. For the purposes of the Stressor ID analysis, DNR data will be used as supplemental information to the available MPCA data (Frank, 2006). The IBI metric scores used in this report will be based off of MPCA data.

Figure 2. Ann River Watershed monitoring sites (See Table 1 for associated STORET and field ID numbers)



1.2 Description of Fish Impairments

1.2.1 Fish IBI metric descriptions

All fish stations on the main stem of the Ann River were evaluated using St. Croix basin IBI criteria (Niemela and Feist, 2000) for streams with drainage areas between 55 – 270 sq. miles. The overall IBI score represents the cumulative scores from a set of 10 metrics used to evaluate fish abundance, condition, and species composition (Table 2). A complete description of the metrics used and scoring criteria can be found in Niemela and Feist, 2000.

Table 2. Metrics used to calculate Fish IBI scores for St. Croix River Basin Streams with Drainage Areas between 55 – 270 sq. miles

Metric name	Category	Description	Predicted response to disturbance
# Darter species	Richness	Taxa richness of darter species	Decrease
% piscivore	Trophic	Pct of fish community that is piscivorous	Decrease
# sensitive species	Abundance	Taxa richness of sensitive species	Decrease
% Simple lithophils	Reproduction	Pct of fish community that are simple lithophilic spawners	Decrease
# benthic insectivore species	Trophic	Taxa richness of fish species that feed primarily on aquatic insects on the stream bottom	Decrease
# omnivore species	Trophic	Taxa richness of omnivore species	Increase
Total # species	Abundance	Total species richness	Decrease
% tolerant species		Pct of fish community considered tolerant to pollution and/or disturbance	Increase
# fish per 100 meters	Abundance	Number of fish per 100 meters (excluding tolerant species)	Decrease
% DELT	Condition	Pct of fish with deformities, eroded fins, lesions, and/or tumors	Increase

Fish impairments in the Ann River Watershed appear to be relatively localized and are not severe in nature. The smaller headwaters streams (i.e. Camp Creek/Little Ann River) that feed Ann Lake appear to support a diverse and healthy warmwater fish assemblage. Below Ann Lake, on the main stem of the Ann River, fish IBI results are mixed, with some scores both above and below the established standard (Table 3). Two stations (Site 4 and Site 8) scored narrowly above the fish IBI standard for all sampling events. The remaining monitoring stations (sites 6, 7, and 9) scored below the IBI standard during all sampling events, although it should be noted that several of the scores were within the confidence interval for the St. Croix basin IBI.

Table 3. Station information, Fish IBI scores, and assessment status for Ann River biological monitoring stations

Station	River mile	Drainage area	Fish IBI score(s)	Std deviation	IBI standard	Assessment
Site 4	10.2	56.2	71	n/a	69	Support
Site 6	6.4	64.3	67, 58	6.4	69	Non-support
Site 7	4.7	65.2	65, 65	0	69	Non-support
Site 8	2.3	71.8	71, 72	0.7	69	Support
Site 9	1.1	72.3	44, 68, 58	12.1	69	Non-Support

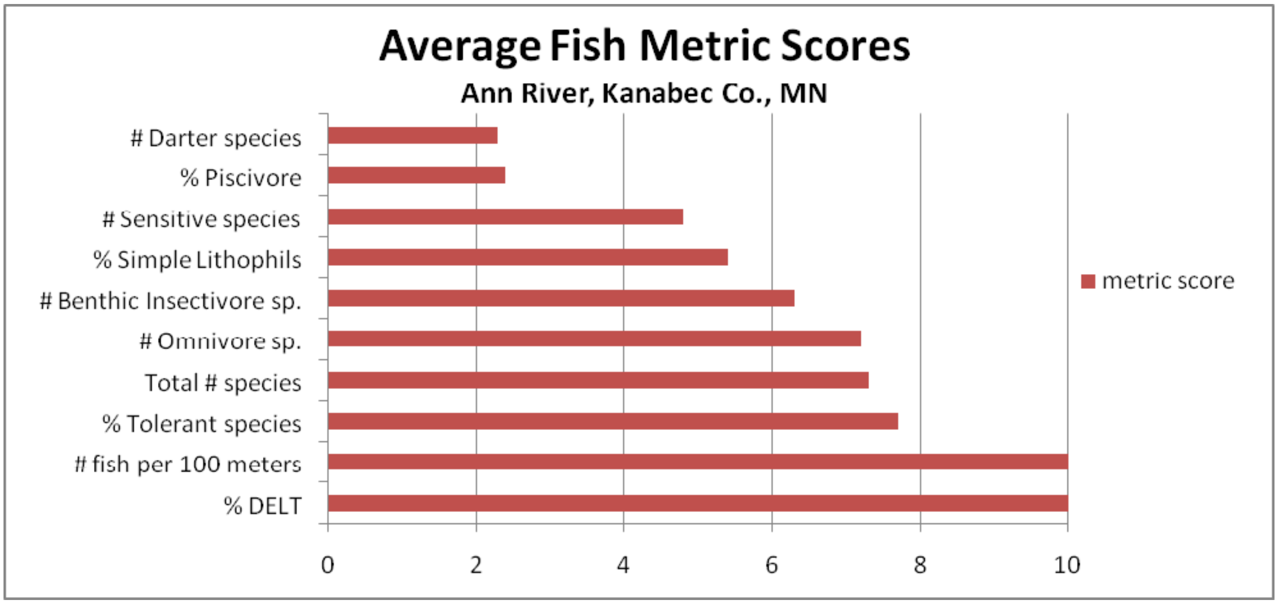
Repeat sampling visits at Sites 7 and 8 showed little variability between the sampling years (std. deviation of 0 and 0.7, table 3). On the contrary, repeat visits to Sites 6 and 9 produced somewhat variable IBI scores, although all of the scores still fell below the impairment threshold. In the case of Site 9, fish IBI scores varied considerably, making it difficult to determine the severity and nature of impairment within this reach. Variability in fish IBI over multiple sampling events can be an indication of stressed communities. Generally, there tends to be less variability in overall IBI score at high quality sites and very poor sites, and greater variability in score at moderately disturbed sites.

The fluctuation in IBI score at Site 9 can be attributed to the inconsistent presence of redhorse (shorthead redhorse, golden redhorse) species and an unusually high abundance of common shiner during one sampling event. In July of 2007, when this reach achieved an IBI score of 68, shorthead and golden redhorse were present, but these species were not sampled in 1998 or in August of 2007. The variability in fish community data at this station is discussed further in the next section.

1.2.2 Systemic indicators of fish impairment

The fish community of the Ann River displays both localized and systemic (watershed-wide) indicators of impairment. Localized indicators occur only at a selection of sites, while systemic indicators appear to be driven by widespread stressors that are present at all sites. Looking at the average of fish metric scores for the Ann River, it appears systemic indicators of fish impairment include low a percentage of piscivore species, low number of darter species, and low numbers of sensitive fish species (Figure 3). Overall taxa richness, fish abundance, and fish abnormalities (i.e. deformities, lesions, tumors, etc.) do not appear to be issues in the Ann River drainage (Figure 3).

Figure 3. Average Fish Metric Scores for all Ann River Biological Monitoring



Low Percent Piscivore

Common piscivorous fish species found in warm water and cool water streams of the St. Croix River basin include smallmouth bass, northern pike, burbot, and rock bass. Fish surveys results from the Ann River show that smallmouth bass, northern pike, burbot, and rock bass are all present, but in very low numbers. The relative scarcity of these species in the Ann River could be attributed to the absence of a variety of chemical, physical, and biological habitat requirements. Most piscivorous fish depend on pool habitat and large structure such as logs, boulders, and undercut banks. The lack of fish species in this trophic group may suggest that these habitat types are limited in the Ann River.

Low Number Darter Species

Darter species sampled from Ann River include logperch (*Percina caprodes*), johnny darter (*Etheostoma nigrum*), and iowa darter (*Etheostoma exile*). Only one Iowa darter individual was collected during sampling efforts (Site 4 in 2006), so it is likely that this species exists in very low numbers. Dater species commonly sampled in other St. Croix region rivers and streams, but not found in the Ann River include, the gilt darter (*Percina evides*), blackside darter (*Percina maculate*), and slenderhead darter (*Percina phoxocephala*).

Low Number of Sensitive Fish Species

Sensitive fish species, as defined by MPCA, are those that are reduced in abundance and/or diversity when habitat conditions become degraded. Typically, these sensitive fish species are the first to be extirpated from river systems where anthropogenic disturbance is prevalent within the watershed. Noteworthy sensitive fish species that are absent or rare in the Ann River include smallmouth bass, gilt darter, and slenderhead darter.

1.2.3 Site specific symptoms of impairment

Evaluating localized symptoms of impairment can provide evidence for or against various candidate stressors. Localized, or site specific biological metric responses were determined by reviewing individual fish metrics for all sampling sites. The variability in fish metric results adds some uncertainty to the process of selecting site-specific impairment trends. The most apparent site-specific indicators of impairment are summarized in Table 4 below. Complete metric scores for fish assessments are shown in Figures 4-8 on the following page.

Table 4. Localized Symptoms of Fish Impairment based on available data

Site #	Site-specific indicator of fish impairment
Site 4	• High % omnivore species • Low # simple lithophilic spawners
Site 6	• Low taxa richness • Low richness of benthic insectivores
Site 7	• Low richness of benthic insectivores
Site 8	• Low # simple lithophilic spawners
Site 9	• Low # simple lithohilic spawners • Low taxa richness • Low richness of benthic insectivores

Figure 4. Individual Fish Metric Scores for Ann River Biological Monitoring Sites

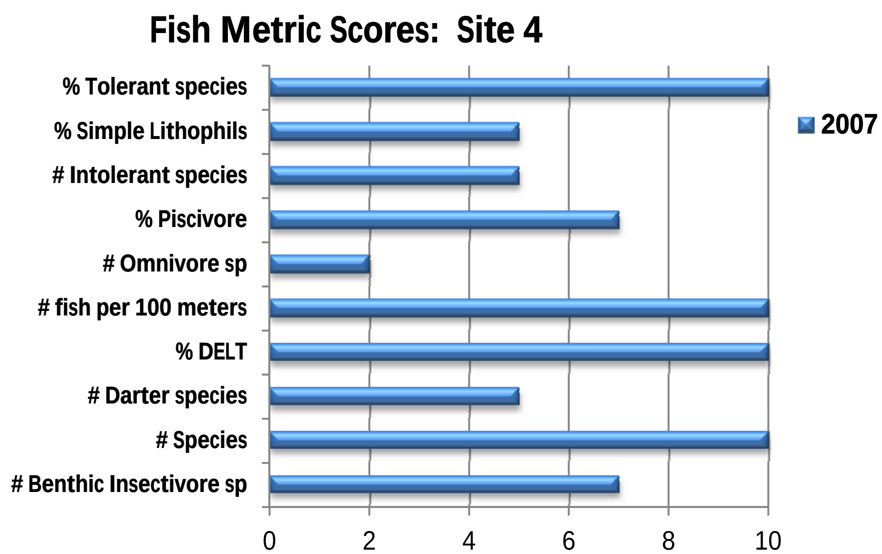


Figure 5. Individual Fish Metric Scores for Ann River Biological Monitoring Sites

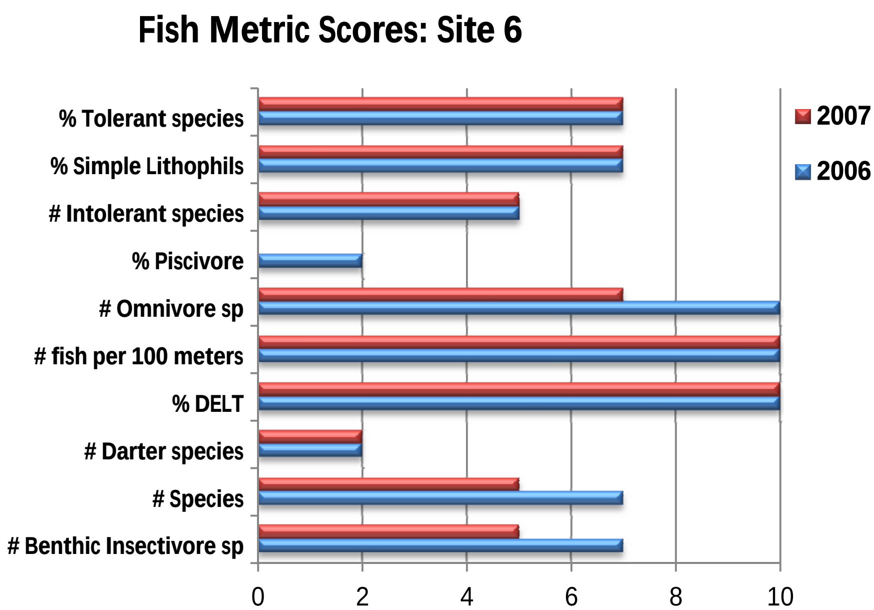


Figure 6. Individual Fish Metric Scores for Ann River Biological Monitoring Sites

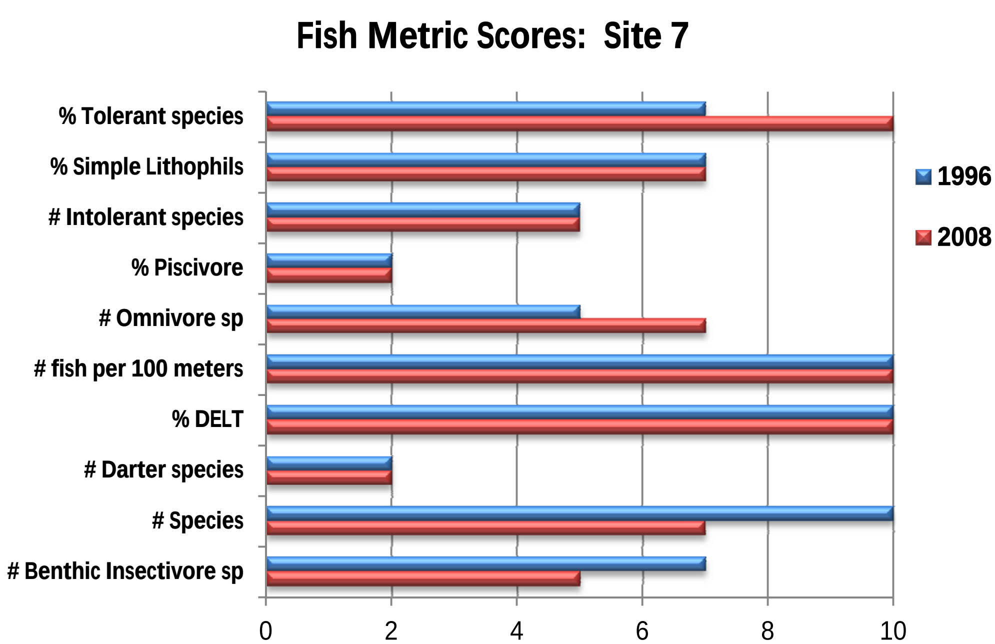


Figure 7. Individual Fish Metric Scores for Ann River Biological Monitoring Sites

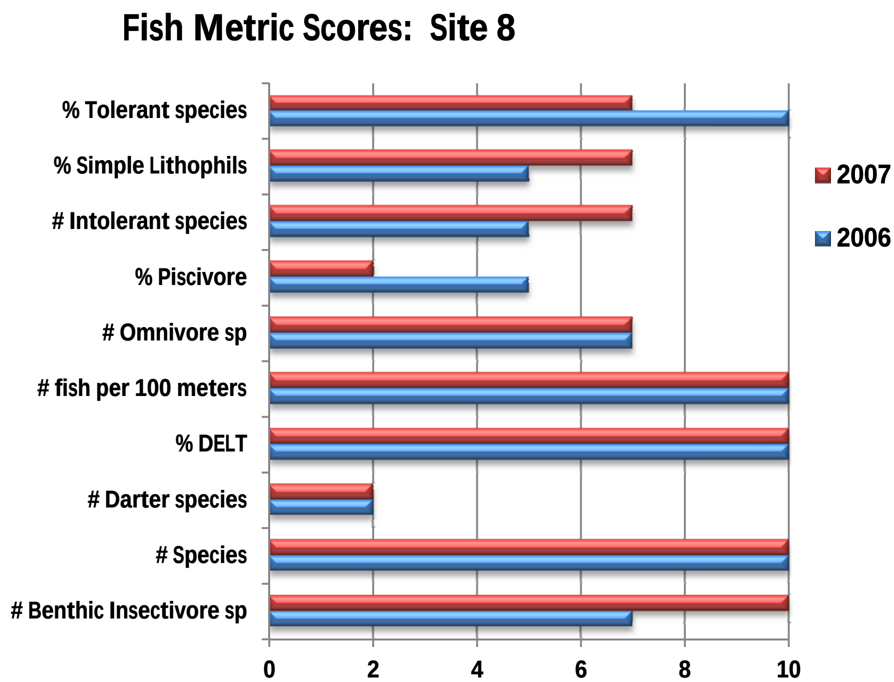
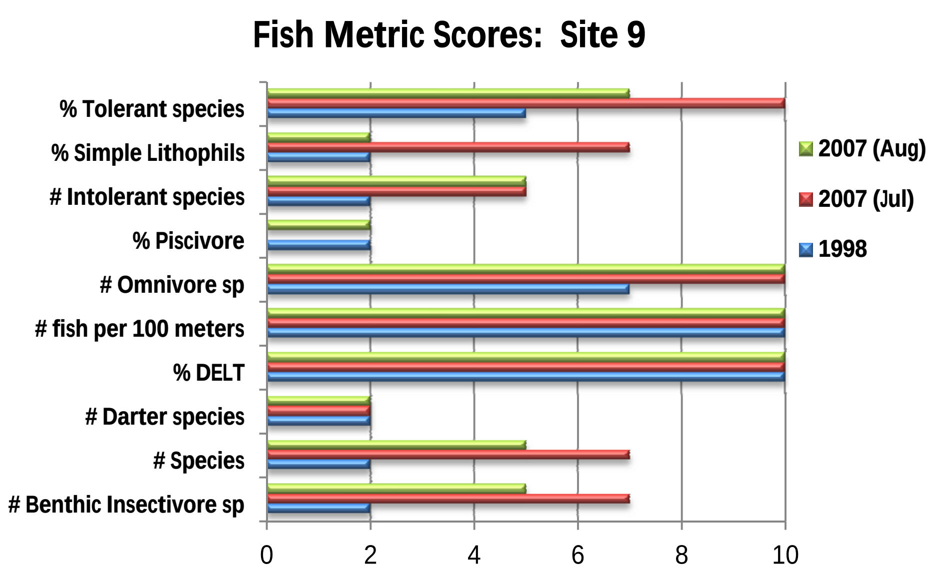


Figure 8. Individual Fish Metric Scores for Ann River Biological Monitoring Sites



1.3 Macroinvertebrate IBI Impairment

1.3.1 Metric descriptions and results

A new statewide macroinvertebrate Index of Biotic Integrity was released by MPCA in the spring of 2010. The IBI metrics used to evaluate macroinvertebrate assemblages in the Ann River can be found in Tables 5 and 6 along with the predicted metric response to an increase in disturbance within the watershed. Due to differences in stream morphology, two distinct sets of macroinvertebrate IBI metrics were applied to Ann River monitoring sites. Sites 4, 6, 7, and 8 were evaluated using invertebrate metrics for *high gradient* wadeable streams with drainage areas less than 500 square miles (Table 5). On the contrary, Site 9, located near the mouth of the Ann River, was evaluated using invertebrate metrics for *low gradient* wadeable streams with drainage area less than 500 square miles (Table 6).

Table 5. Macroinvertebrate Metric Descriptions for Northern Forest Streams Riffle Run (RR)

Metric Code	Description	Response to disturbance
CoenagrionidaePct	Relative percentage of taxa belonging to Coenagrionidae	Increase
EPTCh	Taxa richness of Ephemeroptera, Plecoptera & Trichoptera	Decrease
HBI_MN	A measure of pollution based on tolerance values assigned to each individual taxon developed by Chirhart	Decrease
OTChTxPct	Relative percentage of taxa belonging to Odonata & Trichoptera	Decrease
VeryTolerant2ChTxPct	Relative percentage of taxa with tolerance values equal to or greater than 8 (0-10 scale) using Minnesota tolerance values	Increase

Table 6 Macroinvertebrate Metric Descriptions for Northern Forest Streams Glide Pool (GP)

Metric Code	Description	Response to disturbance
Tricoptera	Taxa richness of Trichoptera (caddisflies)	Decrease
Tolerant2Pct	Relative abundance (%) of macroinvertebrate individuals in subsample with tolerance values equal to or greater than 8 (0-10 scale)	Increase
Intolerant2lessCh	Taxa richness of macroinvertebrates with tolerance values less than or equal to 4 (0-10 scale)	Decrease
HetColChTxPct	Relative percentage of taxa belonging to Heteroptera & Coleoptera	Increase
EPTChTxPct	Relative percentage of taxa belonging to Ephemeroptera, Plecoptera & Trichoptera	Decrease

The spatial extent and severity of macroinvertebrate impairment in the Ann River Watershed is somewhat unknown, as several of the IBI results were likely influenced by drought conditions. Results from the 2006 and 2007 monitoring season indicated severe impairment, but these samples were deemed non-reportable due to low water conditions. The results listed in boldface in Table 7 are those that have passed MPCA data quality standards and were used to assess the Ann River for macroinvertebrate IBI. MPCA's macroinvertebrate sampling is conducted during late summer and early fall, which is when drought conditions were most pronounced in the Ann River watershed during the years of 2006 and 2007. The fish data was collected earlier in the summer during that time period, and were not influenced as severely by the lack of precipitation. As a result, all of the fish assessments provided reportable data.

Based on reportable results only, the results show consistent impairment at Site 7, although the level of impairment is not very severe. The reportable result from Site 6 is above the IBI standard, but within the 90 percent confidence interval, and thus not clearly supporting. Site 9, which was assessed with a unique set of metrics due to the low gradient within the reach, appears to be supporting a healthy macroinvertebrate assemblage. Due to the limited timeframe of this study, many of the stations were not re-sampled for macroinvertebrates, and as a result, spatial coverage of reportable data is relatively poor.

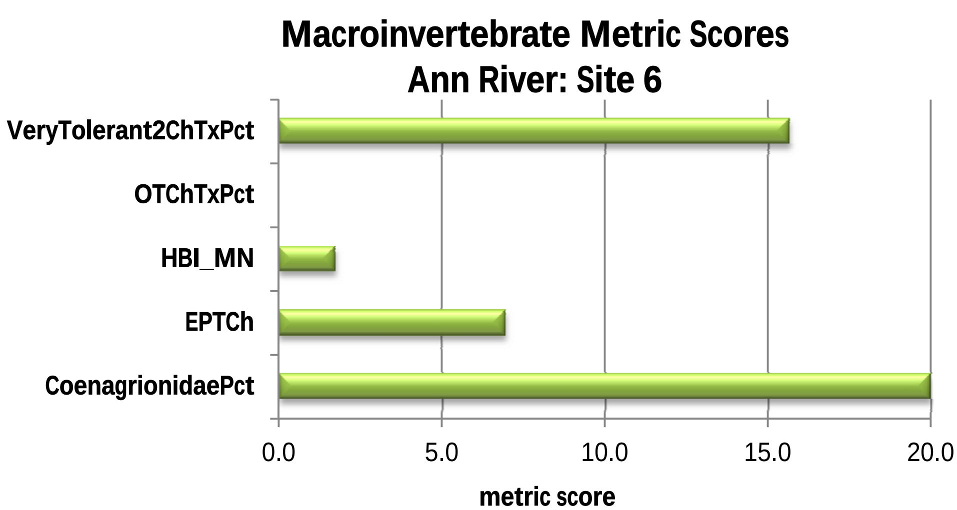
Table 7. Ann River macroinvertebrate monitoring sites and overall IBI scores

Station	Drainage area	IBI Stream classification	M-IBI Scores	MIBI impairment threshold / 90% C.I.
Site 4	56.2	Northern Forest Streams RR	*21*	41.2 (+/- 16.7)
Site 6	66.8	Northern Forest Streams RR	*30, 11,* **44**	41.2 (+/- 16.7)
Site 7	67.5	Northern Forest Streams RR	**31, 31, 39**	41.2 (+/- 16.7)
Site 8	74.0	Northern Forest Streams RR	*18, 25*	41.2 (+/- 16.7)
Site 9	74.8	Northern Forest Streams GP	**68**	39.5 (+/- 17.1)

Bold = reportable data Italics = non-reportable results due to drought conditions

For the purposes of this Stressor Identification study, all invertebrate data (both drought and non-drought years) will be used to some degree. A greater emphasis will be placed on the results from non-drought years at Sites 6, 7, and 9. The data from these sampling events will likely show the effects of stressors that are not related to the drought conditions that were present in the watershed in 2006 and 2007. The macroinvertebrate metric results for these three sites are shown in Figures 9-11.

Figure 9. Macroinvertebrate IBI Metric Scores at Sites 6 and 7*



* These represent the only reportable macroinvertebrate samples from Ann River sites categorized as "high gradient."

Figure 10. Macroinvertebrate IBI Metric Scores at Sites 6 and 7*

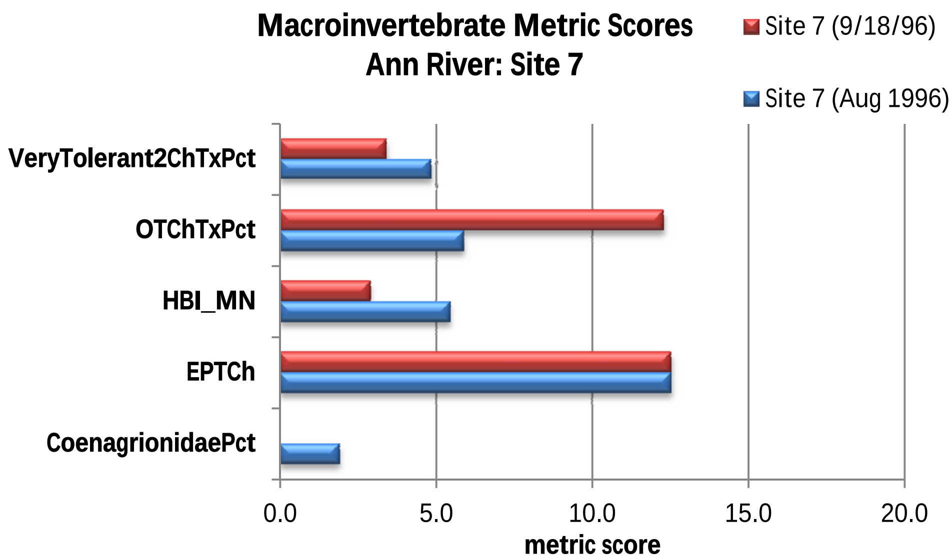
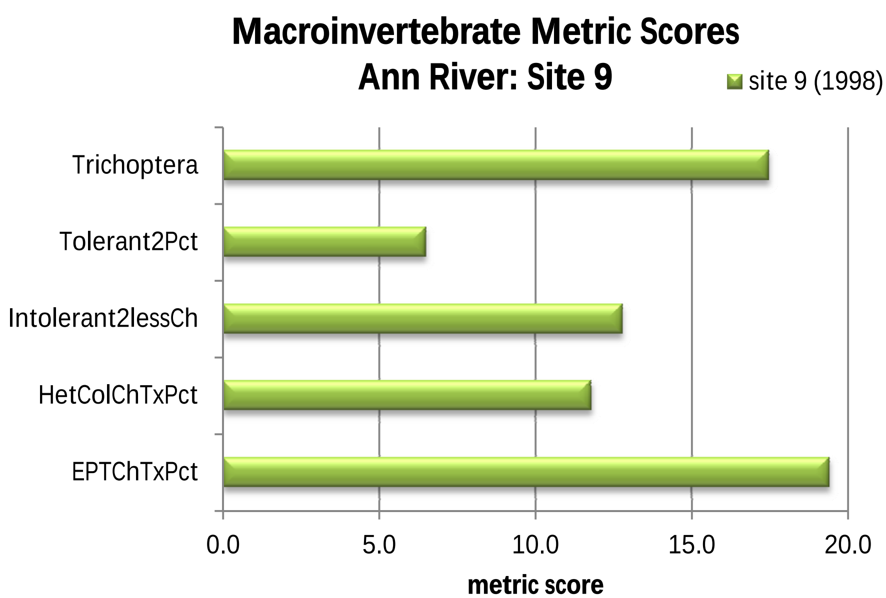


Figure 11. Macroinvertebrate IBI Metric Scores at Site 9, which is classified as a "low gradient" Site for IBI Assessment



1.3.2 Site specific indicators of invertebrate impairment

Macroinvertebrate data from 1996, 1998, and 2008 sampling events at Site 7 form the basis of the M-IBI impairment listing. As mentioned earlier, most of the other data from high gradient sites within the watershed are considered “non-reportable” due to low water conditions at the time of sampling. Site specific indicators of the impaired macroinvertebrate community will be gleaned from station Site 7 and the use of a reference site in the St. Croix basin with similar natural background conditions (stream gradient, drainage area, etc.).

1.3.3 Reference site selection

Criteria for selecting a reference site for the Ann River included several geographic and physical habitat characteristics. The search for a reference site focused on a stream from the same physiographic area with comparable stream gradient, substrate composition, drainage area, and hydrological pathways. The reference site also needed to have an invertebrate IBI score considerably higher than the established impairment threshold.

Site 96SC085 on the Kettle River was chosen as a suitable reference site to compare to the Ann River at Site 7. These two stations have comparable natural background conditions, although in-stream and riparian habitat conditions differ as a result of local land-use. A comparison of key characteristics used to identify the reference site can be found in table 8. Both of these biological monitoring stations are within the “Northern Forest Streams” classification for the macroinvertebrate IBI criteria developed by MPCA.

Table 8. Key Variables used to select a reference site for Ann River site #7

River Name	Station	Drainage area	Strahler stream order	Stream gradient (%)	Dominant substrate
Ann River	Site 7	67.5	3	1.6571	Gravel/Cobble
Kettle River	96SC085	75.6	3	1.6177	Gravel/Cobble

Table 9. Comparison of M-IBI Scores between the Reference Site (96SC085) and Site 7 on the Ann River

River Name	Station	Invertebrate IBI score(s)	Std deviation	Standard	Assessment
Ann River	site 7	31, 31, 40	5.1	41.2	Non-support
Kettle River	96SC085	85	0	41.2	Support

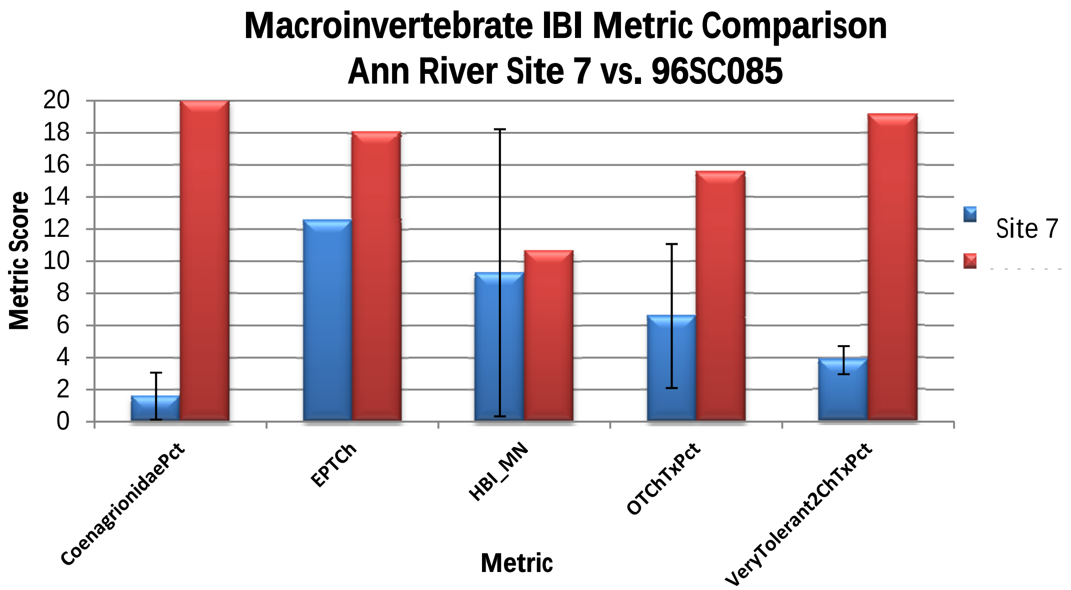
Figure 12. Reference site (98SC085 – Kettle River) on the left. Ann River Site 7, impaired for low macroinvertebrate IBI on the right. Good riparian conditions on the Kettle River at 98SC085 provide shading, bank stability, and woody debris inputs which has resulted in macroinvertebrate IBI scores that meet St. Croix basin standards.

1.3.4 Macroinvertebrate comparisons: reference site vs. Ann River Site 7

The macroinvertebrate communities present at these two sites were compared to learn more about the specific nature of the IBI impairment in the Ann River. The invertebrate community at the reference site on the Kettle River may serve as a target for the Ann River if restoration and/or best management practices (BMP's) are employed in the watershed.

In comparison to the Kettle River reference site, the macroinvertebrate community at Site 7 on the Ann River has a higher percentage of pollution tolerant taxa, fewer taxa from the Ephemeroptera, Tricoptera, and Plecoptera (or EPT) orders, and fewer taxa from the order Odonata (dragonfly/damselfly), and a much higher percentage of Coenagrionidae taxa. The significance of each of these metrics is briefly described in the next paragraph. Most of the metrics used to calculate the overall IBI score respond to disturbance in a predictable manner, though it is somewhat difficult to identify specific stressors (i.e. low DO, turbidity, etc.) from the metric data without looking at species composition and abundance.

Figure 13. Metric Scores for Ann River Site 7 with standard deviation and Kettle River Site 98SC085, which is being considered a potential reference reach



1.3.5 Abundance of very tolerant taxa

Ann River Site 7 appears to contain a greater abundance of very tolerant invertebrate taxa in comparison to the reference site on the Kettle River. The "tolerance value" in this case is defined on a scale from 0 (very sensitive to pollution) to 10 (very tolerant of pollution.) The macroinvertebrate metrics used to calculate IBI scores define "very tolerant" taxa as those with a tolerance value greater than eight. On average, Ann River Site 7 had over 18 very tolerant taxa (mean 18.3, Max 20, Min 17) while the reference site had only 10 very tolerant taxa. The overall percentage of very tolerant taxa at each site did not differ as much, but higher percentage of the macroinvertebrate community at Ann River Site 7 was comprised of very tolerant invertebrate taxa (figure 13).

Table 10. The five most common very Tolerant Taxa found at Ann River Site 7

Genus name	Common name	MN Tolerance Value (0 = least tolerant; 10 = most tolerant)
Stenelmis	Riffle beetles	8.3
Caenis	Mayflies	8.8
Cheumatopsyche	Net-spinning caddisflies	8.0
Polypedilum	midges	8.6
Dicrotendipes	midges	8.2

1.3.6 Percent coenagrionidae

Coenagrionidae is a family of damselflies, also referred to as the narrow-winged or pond damselflies. The majority of Coenagrionidae are wetland dwellers, but there are a few genera that live in stream environments, particularly Enallagma and Ischnura. Enallagma were present in two out of three samples collected at Site 7. Since these species are closely related to the wetland species in the same family, they could be an indicator of poor lotic water quality, especially nutrient enrichment and/or low dissolved oxygen concentrations. Macroinvertebrates from the genera Enallagma are considered climbers, which means that they climb aquatic vegetation or other debris in the stream channel. This quality renders them less dependent on quality benthic habitat and less sensitive to sedimentation of the streambed.

1.3.7 EPT taxa

Taxa richness of the orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and trichoptera (caddisflies), or EPT, is often used as an indicator of overall stream health. Typically, EPT taxa richness declines as watershed disturbance increases and stream habitat conditions worsen. Ann River Site 7 had an EPT taxa richness of 18 all three times that the site was sampled. The reference reach on the Kettle was only sampled once, and had an EPT taxa richness of 22. The main difference in EPT richness between the two sites was due to a greater number of Trichoptera taxa in the Kettle River reference site.

2.0 Candidate Stressors for Biological Impairments

Biological impairments can be caused by a wide-range of chemical, physical, and even biological stressors. The initial step of EPA's stressor identification (SI) process is to list all potential candidate causes for the impairments. Table 11 represents a broad list of stressors that are common causes of biological impairments in river systems. Each of these common candidate causes, and a host of others, were evaluated as a potential cause of biological impairment in the Ann River.

Table 11. Broadest range of candidate causes for fish and macroinvertebrate impairments in the Ann River. (Based on EPA CADDIS)

Low dissolved oxygen	Algaecides
Hydrologic regime alteration	Lampricides
Nutrient regime alteration	Metals
Organic-matter regime alteration	Moluscicides
pH regime alteration	Organic solvents (e.g. benzene, phenol)
Salinity regime alteration	Other hydrocarbons (e.g. dioxins PCBs)
Bed sediment load changes including siltation	Endocrine disrupting chemicals
Suspended solids and/or turbidity alteration	Mixed, cumulative effect
Water temperature regime alteration	Interspecific competition
Habitat destruction	Small population (e.g. inbreeding, stochastic fluctuation, etc.)
Habitat fragmentation	Genetic alteration (e.g. hybridization)
Physical crushing and trampling	Overharvesting or legal, intentional collecting or killing
Toxic substances	Parasitism
Herbicides and fungicides	Predation
Halogens and halides (e.g. chloride, trihalomethanes)	Poaching, vandalism, harassment, or indiscriminate killing
Fish-killing agents (e.g. rotenone)	Unintentional capture or killing
Insecticides	Radiation exposure increase (e.g. increased UV radiation)

Sufficiency of evidence for potential stressors

The Stressor ID/CADDIS framework operates on a strength of evidence platform through which all candidate stressors are evaluated using various types of evidence. The types of evidence that are included in the analysis can be derived from case-specific data or from other outside sources, such as scientific literature, data from similar cases, and anecdotal information from citizen stakeholders. All available evidence is scored using the criteria shown in Table 12. The evidence types each have specific "weights" that can be applied to the overall analysis. These differences are based on the fact that some forms of evidence present a stronger cause-and-effect relationship than others (Table 13). For additional information on the Stressor Identification process, refer to the guidance developed by Cormier et al. (2000).

For the purposes of selecting candidate causes for further analysis, the fish and macroinvertebrate impairments observed in the Ann River watershed will be grouped together. However, analyses of candidate causes for impairment and diagnosis of stressors will be specific to the impaired fish or macroinvertebrate communities, unless a common stressor is responsible for each.

Table 12. Values used to Score evidence in the Stressor Identification process developed by EPA

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

Table 13. Strength of Evidence Scores for various types of evidence used in Stressor ID analysis

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, -

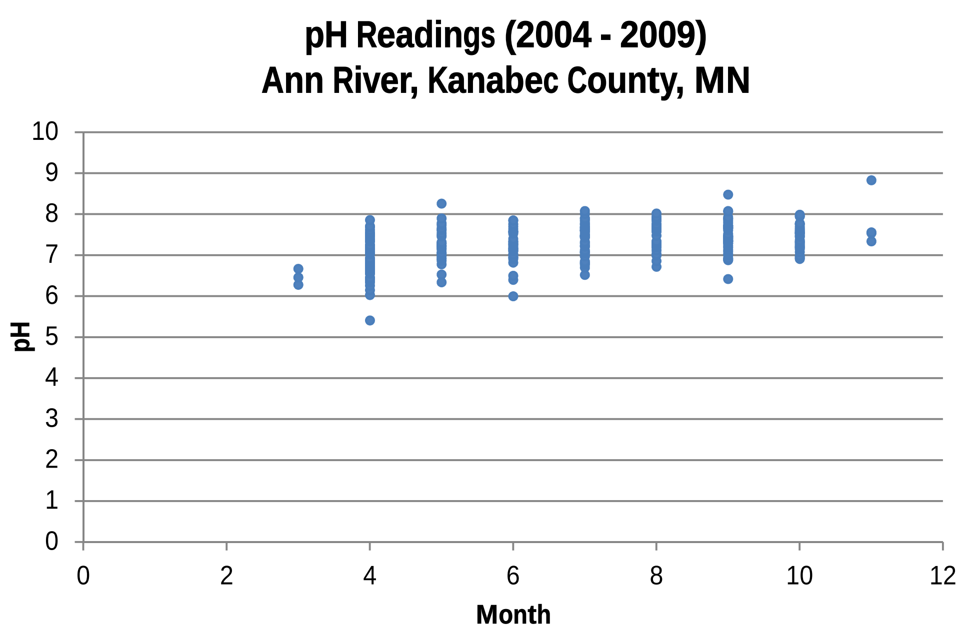
2.1 Elimination of Candidate Causes

The Stressor ID process is essentially a process of elimination. Using data or professional judgment to eliminate candidate causes early in the process simplifies data analysis, and adds strength to the potential causes that remain. Several stressors have been eliminated using existing data from the Ann River watershed, scientific literature, and in some cases, professional judgment stemming from knowledge of the resource. Tables 14 and 15 lists all of the candidate causes that were eliminated as possible causes for biological impairments in the Ann River. Below are some of the more common stressors that were eliminated from consideration, including a brief discussion of the data and/or thought process that was used to eliminate them.

I. Altered pH

Stream pH levels in the Ann River were typically found to be within acceptable ranges for propagation of a coolwater and warmwater fishery. In April of 2008, pH levels at Site 3 on the Little Ann River dropped below the class 2B minimum standard of 6.5 for several weeks. Stream pH readings taken at other Ann River monitoring sites during this period also show lower than normal values, however violations of the class 2B are a rare occurrence on the mainstem of the Ann River.

Figure 14. Ann River pH data by month from 2004-2009



II. Turbidity and total suspended solids

Turbidity levels and total suspended solids (TSS) concentrations in the Ann River are within the desirable range for supporting cool and warmwater fishes and diverse aquatic macroinvertebrate assemblages. Turbidity data from 2006 -2009 did not register one value over the 2B water quality standard of 25 NTU (n = 208, mean = 2.49 NTU, minimum = 0 NTU, maximum = 23.6 NTU). Suspended sediment concentrations measured during the same timeframe fall within the lower percentiles of North Central Hardwood Forest (NCHF) ecoregion reference streams (n = 108; mean = 3.12 mg/L; minimum = 1.00 mg/L; maximum = 22 mg/L). Further analysis of Ann River TSS concentrations and turbidity can be found in section on bedded sediment as a possible stressor.

III. Elevated levels of halogens, halides (i.e. chloride), or salinity

Halogens, halides (i.e. chloride), and salinity were not likely to be problematic in the Ann River watershed given the rural setting of the watershed. Available chloride data indicates that concentrations are well below the state standard of 120 mg/L. Lampricides, piscicides, and moluscides were not specifically sampled for, but the probability of these agents being present in the watershed is low because there are no known uses of these agents for any management purpose. Organic solvents, hydrocarbons, endocrine disruptors, and potentially toxic heavy metals were not evaluated as a part of this sampling effort. It is unlikely that heavy metals, organic solvents, or hydrocarbons (dioxins, polychlorinated biphenyl (PCB)) would be found in high concentrations in the Ann River watershed given the lack of point source pollution sources within the drainage. Fish tissue samples collected from a selection of top predator (smallmouth bass) and roughfish (shorthead redhorse) from the Snake River (into which the Ann River flows) were below detection limits for PCB concentration.

IV. Pesticide and insecticides

Currently, there is no data available on pesticide or insecticide concentrations in the Ann River watershed. However, regional data have been collected by Minnesota Department of Agriculture (MDA) to determine pesticide concentrations in surface water and groundwater. The most recent publication of these results was released for the 2007 monitoring season. Results from streams near the Ann River (Grindstone River, Mission Creek, and Sunrise River) indicate that concentrations of most common pesticides are below detection levels. The few pesticides that were detected in these streams were present in concentrations well below state water quality standards for protection of aquatic life. The MDA groundwater monitoring data from 2007 for Kanabec County does not show any pesticide concentrations above human health risk levels (Minnesota Department of Agriculture, 2010).

V. Interspecific competition

This stressor does not appear to be driver of biological impairment in this system. The fish and macroinvertebrate assemblages present in the Ann River are typical of a coolwater stream with moderate disturbances in the stream and on the landscape. In other words, there are no introduced species or specific conditions that would significantly alter interspecific competition. No unusual levels of interbreeding, genetic hybridization, parasitism, or predation were observed during biological monitoring in the watershed. The Ann River appears to receive moderate fishing pressure at certain locations, but not nearly enough to deplete gamefish numbers.

VI. Heavy metals toxicity

Concentrations of Lead, Nickel, Cadmium, and Chromium were not measured in the Ann River as part of this study. Given the rural setting of this watershed and lack of point source polluters, elevated concentrations of these metals are unlikely causes of biological impairment.

Table 14. Common Stressors to Aquatic Life as presented by EPA CADDIS and Weight of Evidence Scores for the Ann River

Types of evidence	pH regime alteration	Salinity regime alteration	Chloride	Fish killing agents (i.e. Rotenone)	Insecticide/ Pesticide	Lampricide	Moluscicide	Organic Solvents	Other hydrocarbons (dioxins, PCBs)
Evidence using data from Ann River									
Spatial / temporal co-occurrence	---	---	---	NE	NE	NE	NE	NE	NE
Evidence of exposure, biological mechanism	--	--	--	NE	NE	NE	NE	NE	NE
Causal Pathway	-	-	-	NE	NE	NE	NE	NE	NE
Field Evidence of stressor-response	-	-	-	NE		NE	NE	NE	NE
Field experiments / manipulation of exposure	NE	NE	NE	NE	NE	NE	NE	NE	NE
Laboratory analysis of site media	NE	NE	NE	NE	NE	NE	NE	NE	NE
Temporal sequence	NE	NE	NE	NE	NE	NE	NE	NE	NE
Verified of tested predictions	NE	NE	NE	NE	NE	NE	NE	NE	NE
Symptoms	NE	NE	NE	NE	NE	NE	NE	NE	NE
Evidence using data from other sources									
Mechanistically plausible cause	-	-	-	+	+	+	+	+	+
Stressor-response in other field studies	-	-	-	+	-	+	+	+	+
Stressor-response in other lab studies	0	0	0	++	++	++	++	++	++
Stressor-response in ecological models	0	0	0	++	++	++	++	++	++
Manipulation experiments at other field sites	0	0	0	++	++	++	++	++	++

Types of evidence	pH regime alteration	Salinity regime alteration	Chloride	Fish killing agents (i.e. Rotenone)	Insecticide / Pesticide	Lampricide	Moluscicide	Organic Solvents	Other hydrocarbons (dioxins, PCBs)
Analogous stressors	0	0	0	+	+	+	+	+	+
Multiple Lines of Evidence									
Consistency of Evidence	--	--	--	NE	--	NE	NE	NE	NE
Explanatory power of evidence	-	-	-	NE	-	NE	NE	NE	NE

Table 15. Common Stressors to Aquatic Life as presented by EPA CADDIS and Weight of Evidence Scores for the Ann River

Types of Evidence	Endocrine disrupting chemicals	Interspecific competition	Population/ Interbreeding	Genetic Hybridization	Parasitism	Predation	Poaching/ overharvest	Lead	Nickel	Zinc	Cadmium	Chromium
Evidence using data from Ann River												
Spatial / temporal co-occurrence	NE	---	---	---	---	---	NE	NE	NE	NE	NE	NE
Evidence of exposure, biological mechanism	NE	--	--	--	--	--	NE	NE	NE	NE	NE	NE
Causal Pathway	NE	-	-	-	-	-	NE	NE	NE	NE	NE	NE
Field Evidence of stressor-response	NE	-	-	-	-	-	NE	NE	NE	NE	NE	NE
Field experiments / manipulation of exposure	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE
Laboratory analysis of site media	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE
Temporal sequence	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE
Verified of tested predictions	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE
Symptoms	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE	NE
Evidence using data from other sources												
Mechanistically plausible cause	+	+	+	+	+	+	+	NA	NA	NA	NA	NA
Stressor-response in other field studies	+	+	+	+	+	+	+	NA	NA	NA	NA	NA
Stressor-response in other lab studies	++	++	++	++	++	++	NE	NA	NA	NA	NA	NA
Stressor-response in eco ecological models	++	++	++	++	++	++	++	NA	NA	NA	NA	NA

Types of Evidence	Endocrine disrupting chemicals	Interspecific competition	Population/ Interbreeding	Genetic Hybridiz ation	Parasiti sm	Pred ation	Poaching/ overharvest	Lead	Nickel	Zinc	Cadmium	Chromium
Manipulation experiments at other field sites	++	++	++	++	++	++	++	NA	NA	NA	NA	NA
Analogous stressors	+	+	+	+	+	+	+	NA	NA	NA	NA	NA
Multiple Lines of Evidence												
Consistency of Evidence	NE	--	--	--	--	--	NE	NE	NE	NE	NE	NE
Explanatory power of evidence	NE	-	-	-	-	-	NE	NE	NE	NE	NE	NE

2.2 Remaining Candidate Causes and Causal Pathways

Five candidate causes were selected for further analysis to determine the most probable causes of biological impairment in the Ann River.

- loss of habitat resulting from substrate embeddedness
- low dissolved oxygen
- loss of riparian function
- flow alteration due to impoundments
- loss of connectivity due to impoundments

2.3 Candidate Cause #1: Loss of Habitat due to Excess Bedded Sediment

2.3.1 Effects of bedded sediment and applicable water quality standards

Increases in suspended and bedded sediment within aquatic systems are now considered one of the greatest causes of water quality and biological impairment in the United States (EPA, 2003). Although sediment delivery and transport are an important natural process for all stream systems, sediment imbalance (either excess sediment or lack of sediment) can result in the loss of habitat and/or direct harm to aquatic organisms. As described in a review by Waters (1995), excess suspended or bedded sediments cause harm to aquatic life through two major pathways: (1) direct, physical effects on biota (i.e. abrasion of gills, suppression of photosynthesis, avoidance behaviors); and (2) indirect effects (i.e. loss of gravel spawning habitat, increase in sediment oxygen demand).

Excess fine sediment deposition on benthic habitat has been proven to negatively impact fish and macroinvertebrate species that depend on clean, coarse stream substrates for feeding, refugia, and/or reproduction. Aquatic macroinvertebrates are generally affected in several ways: (1) loss of certain taxa due to changes in substrate composition (Erman and Ligon, 1988); (2) increase in drift (avoidance) due to sediment deposition or substrate instability (Rosenberg and Wiens 1978); and (3) changes in the quality and abundance of food sources such as periphyton and other prey items (Peckarsky 1984). Fish communities are typically influenced through: (1) a reduction in spawning habitat or egg survival (Chapman 1988) and (2) a reduction in prey items as a result of decreases in primary production and benthic productivity (Bruton 1985; Gray and Ward 1982).

Based on empirical evidence collected in the Ann River watershed, habitat degradation resulting from bedded sediment was identified as a potential cause of biological impairment. Measurements of substrate embeddedness, particle size distributions, and depositional patterns indicated a degraded benthic habitat and a general lack of stream features (riffles, pools, glides) within impaired areas. These disturbances are more pronounced at selection of river reaches, which is likely a result of localized sediment inputs and differences in transport capacity based on channel slope, form, and overall stability.

2.3.2 Sediment sources, conceptual model, and data analysis

The conceptual model for excess bedded sediment in the Ann River is shown in Figure 19. As depicted in the conceptual model, the primary sources of sediment in the Ann River watershed can be categorized into three main groupings: (1) upland sources, (2) riparian sources, and (3) streambed/banks. "Upland sources" are those that are outside of the immediate stream corridor and riparian area. In the Ann River watershed, these sediment sources include contributions from various agriculture activities (namely crop production and cattle pasturing), mineral extraction (primarily gravel mining), road construction, and timber harvesting. Based on observations during watershed reconnaissance efforts, sediment delivery from the "upland sources" constitutes a small percentage of the overall sediment inputs to the river.

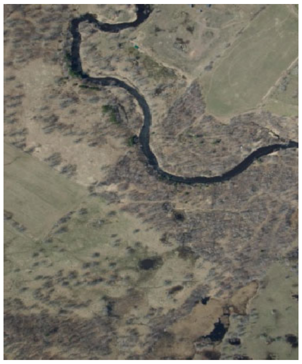
Sources of sediment within the riparian corridor encompass those located with several hundred feet of the river banks. Based on analysis of aerial photos and observations recorded during stream reconnaissance, the majority of sediment sources in the riparian corridor are driven by cattle grazing, timber harvesting, and other land-use activities that have resulted in a reduction of riparian plant diversity and/or rigor. These land-uses appear to be delivering sediment loads to the stream through direct destabilization of stream banks and valley walls (i.e. trampling, vegetation removal) and indirectly by making these landscape features more susceptible to sediment loss during snowmelt and rain events.

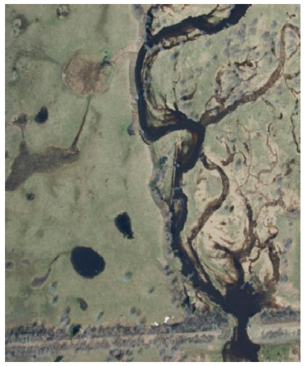
Figure 15. Example of contrasting Riparian Management along the Ann River

The photos in Figure 15 were taken from the same location, looking upstream (left) and downstream (right) at a fence line between forested land and a cattle pasture. The picture on the right shows sediment sources from eroded streambanks and riparian sources due to vegetation removal.

Figures 19 and Figure 20 provide additional evidence of sediment delivery to the river from riparian sources. Analysis of aerial photos taken during the spring of 2008 indicates that several reaches of the Ann River lacking riparian vegetation are beginning to develop braided stream-channels, which tends to increase channel width-to-depth ratio and reduce sediment transport capacity. This channel evolution process can result in higher rates of sediment deposition on the streambed and reduced habitat complexity as pools and riffles become overwhelmed with sediment.

Figure 16. Aerial view of Two Ann River Stream Reaches with contrasting local land-use and riparian quality





The two reaches above are about 0.5 miles apart, and located near Site 7. The photos were taken during spring snowmelt in April 2008. Surface and gully erosion, along with the formation of numerous side channels, is evident within the impacted area on the right. On the contrary, a stable, single-thread channel is maintained within the wooded, non-grazed area on the left.

Figure 17. 1939 Aerial photo (left) compared to 2008 aerial photo (right) of the same reach (Site 7) on the Ann River. Deforestation and agricultural (grazing) land-use appears to have de-stabilized the stream channel and increased sediment loading to the river.

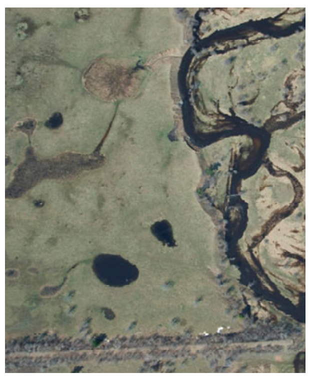


Figure 18. Riparian corridor of the Ann River impacted by deforestation, cattle grazing, and fluvial processes. (This reach is located near Site 7)

The third major sediment source in the Ann River watershed is the sediment load comprised of streambank sediments and those scoured from the streambed. Erosion of streambanks and bed materials is a natural process, occurring to some degree in all streams, even those that are considered stable and undisturbed. Areas of severe bank erosion were observed in several locations during reconnaissance trips down the Ann River. These bank failures were often observed in areas where the riparian and/or bank vegetation was removed or altered (Figure 18). Significant bank erosion was also observed in the lower reaches of the Ann River several miles upstream of Fish Lake. These banks appeared to be eroding as a result of fluvial processes, as most of this reach is a heavily wooded and well-vegetated floodplain forest. Debris jams were prevalent in this reach, which are likely deflecting erosive currents away from the river's thalweg and towards streambanks. This process appeared to be increasing the frequency and magnitude of bank scouring within this lower reach of the Ann River.

Figure 19 Conceptual Model for Sediment as a Stressor to Aquatic Life in the Ann River Watershed

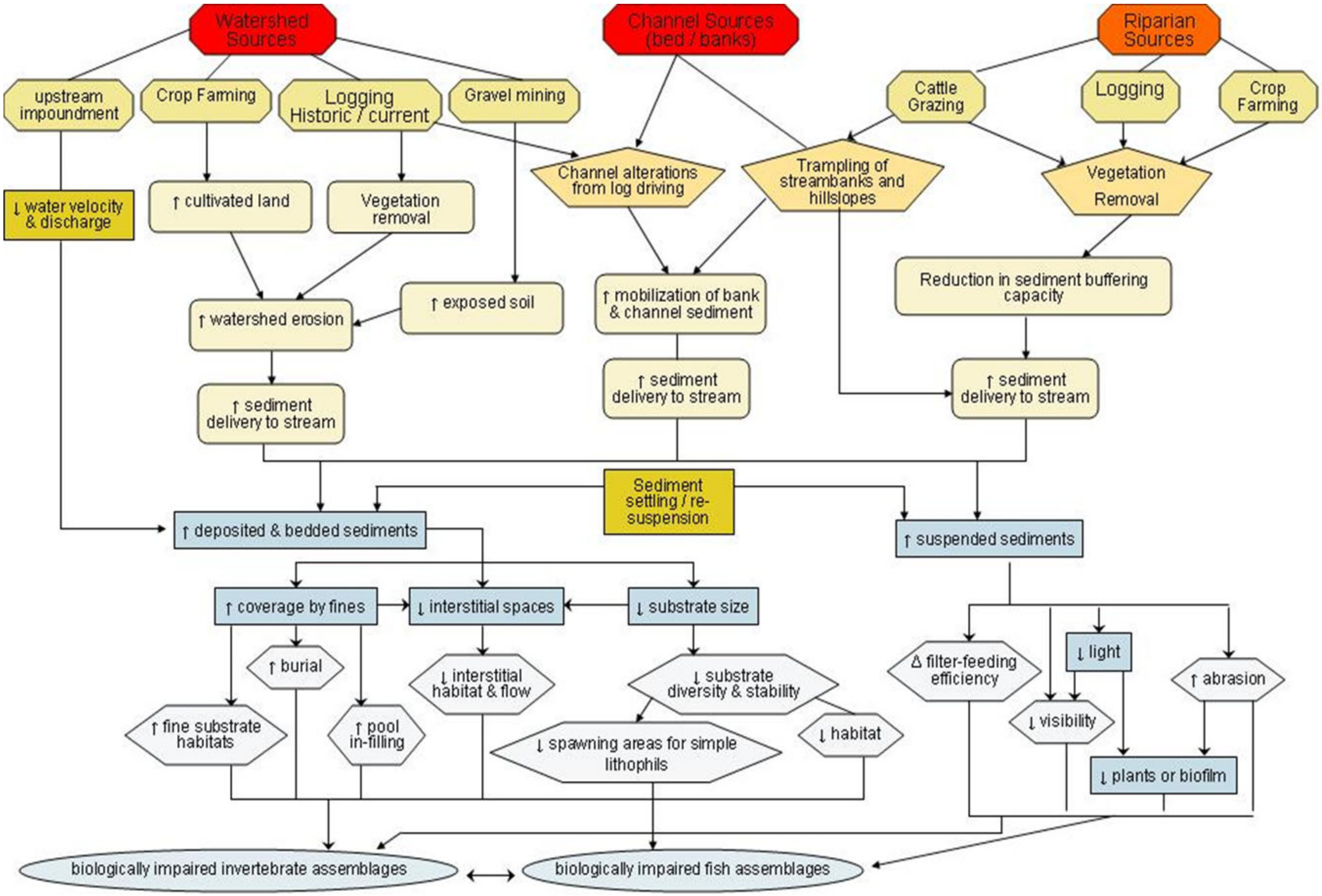


Figure 20. Examples of bank erosion contributing to the sediment load carried by the Ann River.

The photo on right shows fluvial erosion observed near the mouth of the Ann River. The photo on left depicts streambank erosion resulting from cattle access to the river.

Figure 21. Cattle access to the stream channel is common within the Ann River watershed. Bank erosion resulting from cattle traffic near and within the stream was very evident during stream reconnaissance. The above photo below was taken at Site 8.

2.3.3 Suspended sediment versus bedded sediment

Two proximate stressors involving sediment stress were included in the conceptual model in Figure 19. One proximate stressor involves the loss of habitat due to increased bedded sediment, and the other is associated with impacts to biota resulting from an increase in suspended sediment concentrations. Although these two proximate stressors originate from similar sources (streambank erosion, gully erosion, surface runoff), they involve different process within the stream channel and are examined using different data sets. This section explores the connection between suspended and bedded sediment within the Ann River watershed, with the goal of defining which form of sediment stress is more problematic for aquatic life.

The suspended sediment (SS) data available for the Ann River do not suggest that it is a probable stressor to aquatic life. Out of a total of 185 TSS samples collected on the mainstem of the Ann River, 92 percent of the results are below the North Central Hardwood Forest ecoregion median for minimally impacted streams, which is 8.8 mg/L (Figure 22) (McCollor and Heiskary, 1993). This suggests that the suspended sediment loads carried by the Ann River are similar to those of minimally impacted rivers within comparable ecological and geological attributes. Snowmelt events in early spring and summer storm events have been shown to elevate Ann River TSS concentrations to between 12-20 mg/L on occasion, but concentrations are typically below 8.8 mg/L even during spring and summer runoff events.

Figure 22. Total Suspended Sediment (TSS) concentrations by month/day compared to the annual median for reference streams from the North Central Hardwood Forest Ecoregion (McCollor and Heiskary, 1993). Period of record for TSS data is 1999 – 2009.

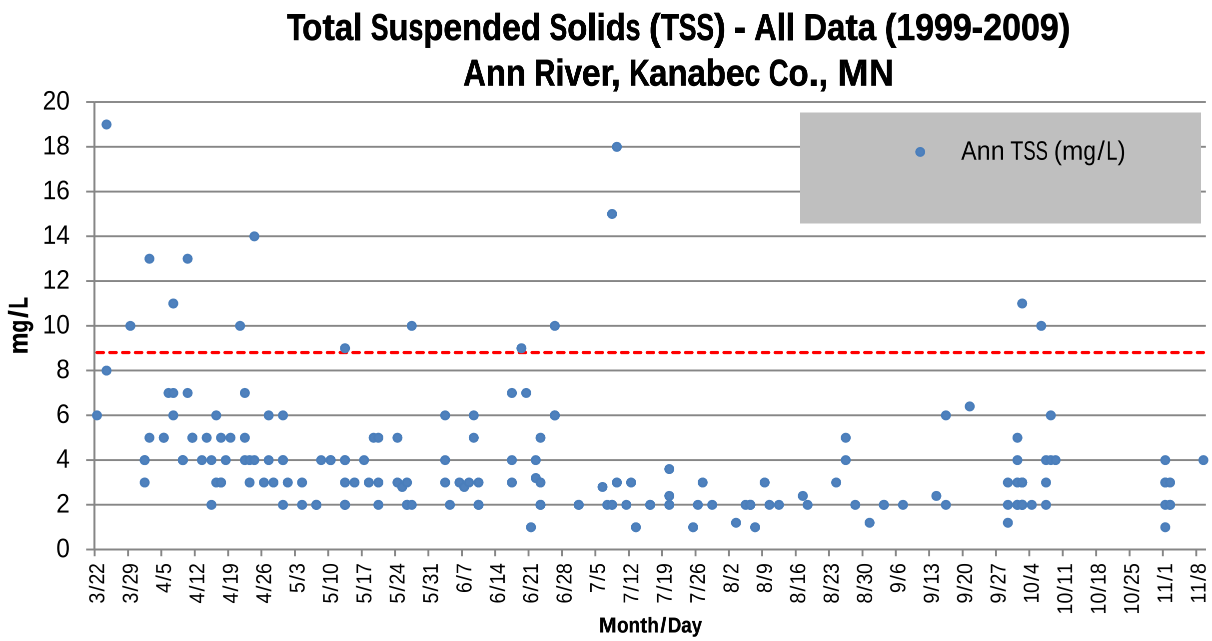
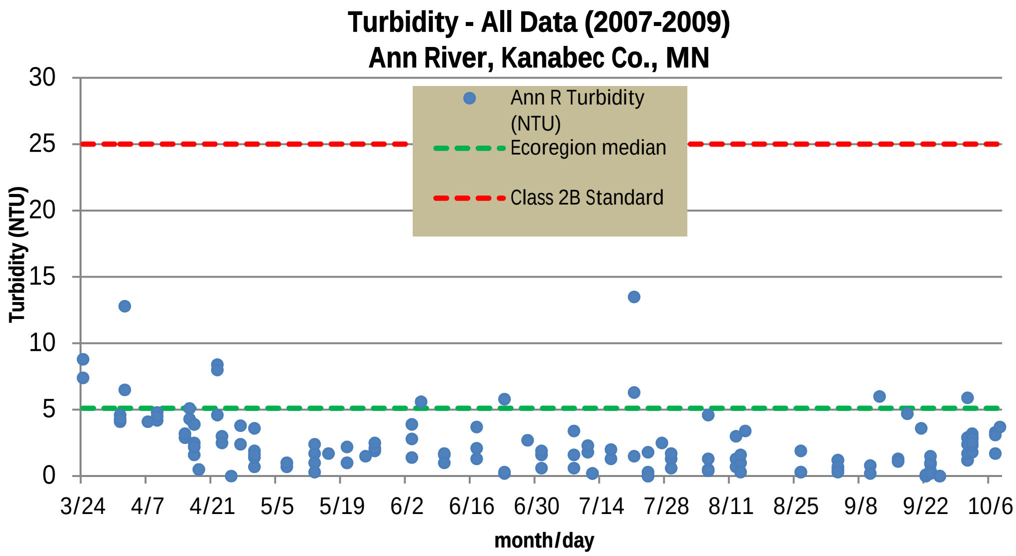


Table 16. All available Ann River TSS data by month (1999-2009)

Month	# samples	TSS (max)	TSS (avg.)	Ecoregion annual median
Mar	4	19	10.75	8.8
Apr	46	14	5.31	8.8
May	31	10	3.45	8.8
Jun	27	10	4.22	8.8
Jul	21	18	3.56	8.8
Aug	16	5	2.30	8.8
Sept	10	6	2.90	8.8
Oct	22	11	3.68	8.8
Nov	9	4	2.67	8.8

A significant amount of work has been done to develop relationships between suspended sediment concentration and turbidity (sources). In Minnesota, there is a strong positive correlation between these two parameters. Within the Ann River watershed, the relatively low TSS concentrations that have been observed are further validated by turbidity readings that are well below class 2B state water quality standard of 25 NTU (Figure 23). Turbidity levels in the Ann River are comparable to those found in minimally impacted streams within the North Central Hardwood Forest ecoregion of Minnesota
(Figure 23). The relatively low turbidity concentrations are likely due to the abundance of coarse particles in the watershed. Most of the sediment transported by the Ann River appears to be coarse sand and small gravel.

Figure 23. Ann River Turbidity Data (2007 – 2009) by month and day.



The combination of low TSS concentrations and low turbidity suggest that bedload is the dominant process for sediment transport in the Ann River. Actual measurements of bedload sediment transport have not been collected in the Ann River watershed due to funding and equipment limitations. Observations made within several unstable reaches of the Ann River revealed large amounts of coarse sand and small gravel particles deposited in the middle of the stream channel (Figure 24). These materials are likely carried as bedload during high flow events, and then deposited out as flows recede. Certain stream reaches appear to be overwhelmed with the amount of coarse-grained material and lack the ability to maintain sediment balance. The result is channel-braiding, a reduction in stream power, and the filling in of interstitial spaces with sand and fine gravel.

Figure 24. Reach of the Ann River showing mid-channel bars of coarse grained sediment within the stream channel. Under high flows, stream power is sufficient to transport these materials along the stream bottom. As flows recede, sand and gravel drop out and reduce benthic habitat quality.

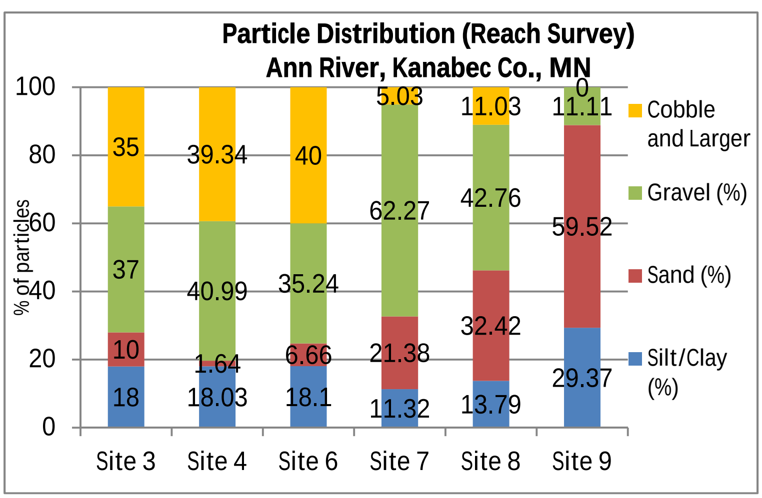
2.3.4 Methodology for evaluating bedded sediment

Field measurements were collected to evaluate substrate embeddedness, percentage of stream substrates that were fines (sand/silt), depth of fine sediment, and particle size distribution. Data collection procedures followed the MPCA's quantitative stream habitat assessment and geomorphological techniques outlined in *Applied River Morphology* (Rosgen, 1996) and Verry (2005). The objectives of the data collection were to characterize the condition of benthic habitats and evaluate the sediment transport capabilities of various Ann River stream reaches. These data were analyzed along with co-located biological data to evaluate potential cause and effect relationships between sedimentation and impaired fish and macroinvertebrate assemblages.

2.3.5 Results: bedded sediment and particle distributions-longitudinal trends

There is a clear longitudinal shift in dominant substrate type within the Ann River watershed. Biological monitoring stations in headwater tributaries, as well as the headwaters and middle reaches of the mainstem Ann River, are dominated by boulder, cobble substrates, with the finer materials being gravel and coarse sand. Lower in the watershed, the dominant substrate shifts to finer materials, such as small gravel, sand, and silt (Figure 25 and 27). This trend is not unusual for river systems, as headwater reaches are often steeper in gradient and can more efficiently transport materials compared to lower gradient, wider stream reaches with lower stream power. In the case of the Ann River, it appears that additional sediment loading from various land-use practices has increased the rate of and extent of sediment deposition in the lower reaches of the river.

Figure 25. Particle Distributions at Biological Monitoring Stations on the Ann River

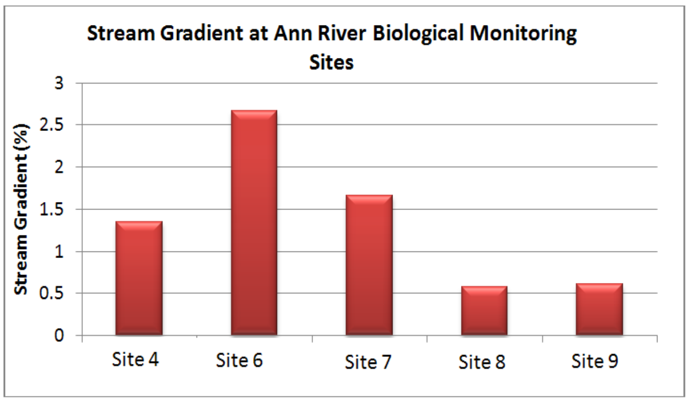


UPSTREAM --> DOWNSTREAM

Based on data collected using longitudinal pebble count methods (Verry, 2005). A large portion of the silt/clay particles sampled were taken from the bank material, not in the active stream bed.

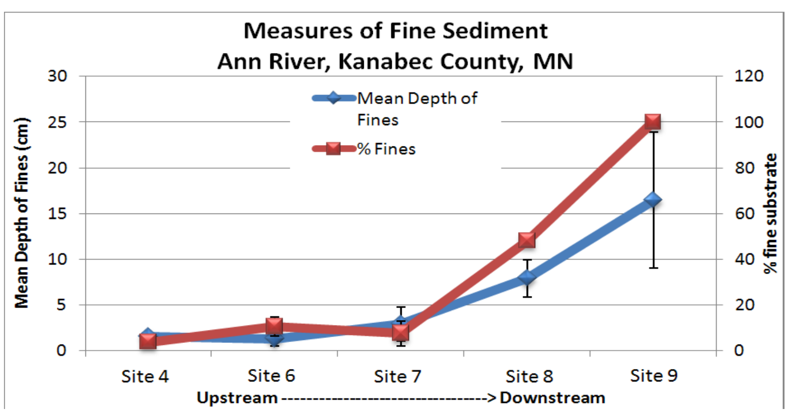
Lane (1955) is frequently referenced when discussing the relationships between channel morphology, streamflow, and sediment transport. Lane concluded that stream gradient (stream channel slope) is one of several key factors in determining the quantity and size of sediment that a stream can transport. Stream slope varies greatly between the biological monitoring stations on the Ann River (Figure 26). Biological monitoring stations in the mid-river reaches are high-gradient sites with enough streampower during elevated flows to move sand and gravel downstream. The more gradual stream gradients found within monitoring stations Site 8 and Site 9 have a reduced ability to transport sediment, which is very likely a contributing factor to the abundance of fine particles (sand/silt/clay) found at these sites.

Figure 26. Comparison of Stream Gradient at Ann River Biological Monitoring Sites



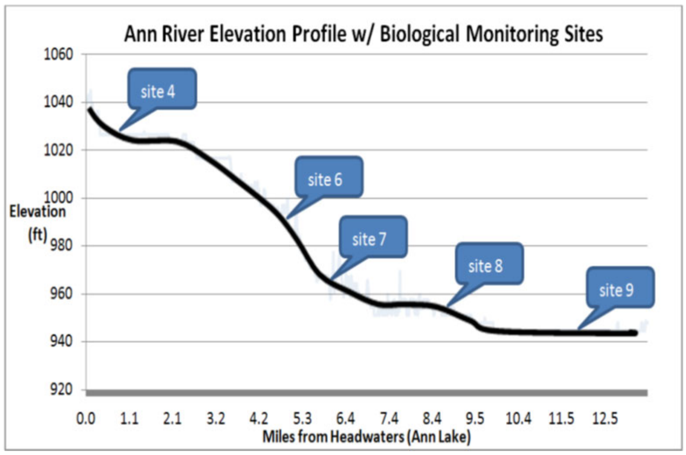
The depth of fine sediment was measured at each biological monitoring site using MPCA's quantitative habitat protocols. Measurements were conducted along cross-sectional transects that were evenly spaced throughout the biological sampling reach. The fine sediment measured as "depth of fines" can be considered the material that is loosely deposited (not compacted) on the stream bottom and could reduce spawning habitat for fish and interstitial spaces used by macroinvertebrates. There is a clear longitudinal trend showing an increase in depth of fine sediment from upstream to downstream in the Ann River (Figure 27).

Figure 27. Average depth of fines and average percentage of fine substrate at Ann River biological monitoring stations (w/ standard deviation).



The relationship between fine sediment deposition on the streambed and stream gradient becomes even more apparent when comparing the elevation profile in Figure 28 against the depth of fines measurements in Figure 27. Anecdotal evidence from residents of Fish Lake suggests that sediment deposition at the river mouth has increased to the point where the lake is filling in some areas. If sediment deposition continues in this area at the current rate, habitat may be compromised for lake dwelling fish as well as those species that migrate between the lake and the river for spawning purposes (northern pike, redhorse, lake sturgeon).

Figure 28. Elevation profile and location of biological monitoring sites. Elevations are based on 30m digital elevation model data



2.3.6 Biotic response to sedimentation

This section will focus on making connections between bedded sediment and impaired biotic communities of the Ann River. The habitat and geomorphological data discussed in the previous section will be evaluated in the context of Ann River biological data and the various metrics that comprise the IBI criteria.

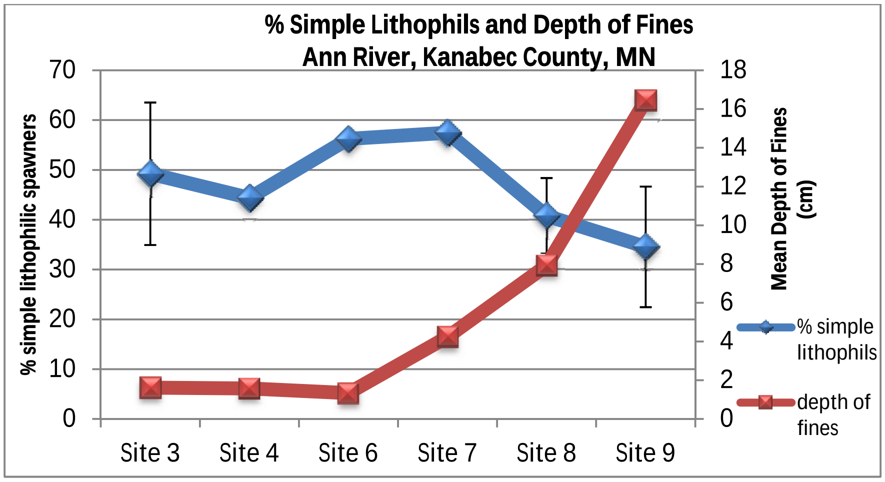
2.3.7 Fish response

2.3.7.1 Reduction in simple lithophilic spawning fish species

Fish species that are simple lithophilic spawners require clean, coarse substrate for reproduction. These fish do not construct nests for depositing eggs, but rather broadcast them over the substrate. Eggs often find their way into interstitial spaces among gravel and other coarse particles in the stream bed. Increased sedimentation can reduce reproductive success for simple lithophilic spawning fish, as eggs become smothered by sediment and become oxygen deprived. Examples of simple lithophils common to coolwater streams of the St. Croix River basin include blacknose dace, common shiner, redhorse species, and logperch.

Within the Ann River watershed, biological monitoring sites with greater abundance of fine sediment generally displayed lower percentages of fish that were simple lithophilic spawners (Figure 29). Although there was a fair amount of variability in simple lithophils percentages between sampling events, there appears to be a noticeable negative relationship between sedimentation and simple lithophils in the Ann River watershed.

Figure 29. Percentage of the Fish community composed of simple lithophils vs. average depth of fine (bedded) sediment at Ann River Biological Monitoring Stations



The standard deviation of percent simple lithophils between sampling events is substantial for Site 8, Site 9, and Site 3. Much of the variability in this metric is the result of fish survey results from August of 2007, which produced lower abundance of simple lithophils, and lower overall IBI scores at Sites 3 and Site 9. Site 3 was sampled twice in 2007, once in July and again in August. The abundance of simple lithophilic spawners and the overall IBI score was considerably lower during the August sample as the drought began to take full effect.

Cumulative data from all three sampling events at Site 9 suggest that numbers of simple lithophils are generally lower at this station when compared to other sites in the watershed. Given the lack of coarse substrate at Site 9, this reach does not likely provide viable year-round habitat for fish that reproduce by this method. Much of the variability in percent simple lithophils at Sites 8 and 9 can be attributed to a greater abundance of common shiners sampled in 2007. Common shiners were generally abundant at both sites across multiple sampling events, but in 2007, they were by far the most abundant species present at both sites. The relative abundance of other simple lithophilic spawning species remains fairly constant at these sites across multiple sampling years.

Figure 30. Species and abundance of Simple Lithophils at Site 8, in 2006 and 2007. The increase in percent simple lithophils observed in 2007 was largely due to a sharp increase in common shiner individuals.

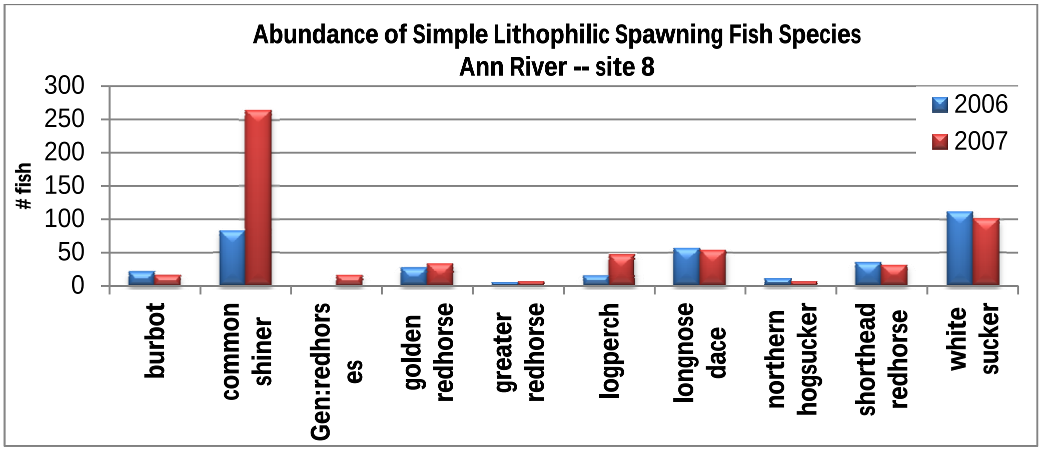
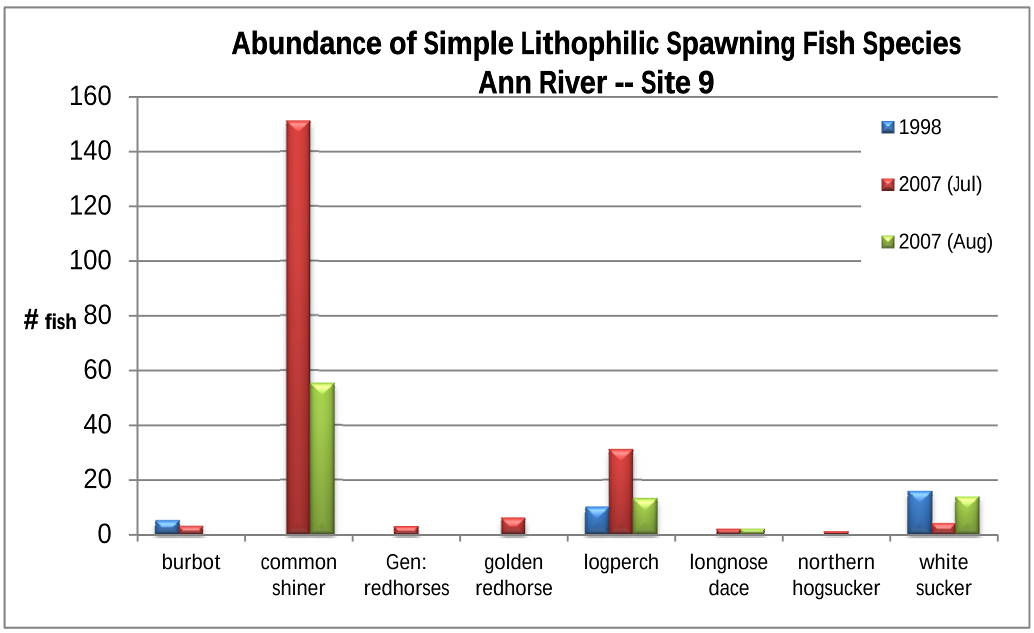


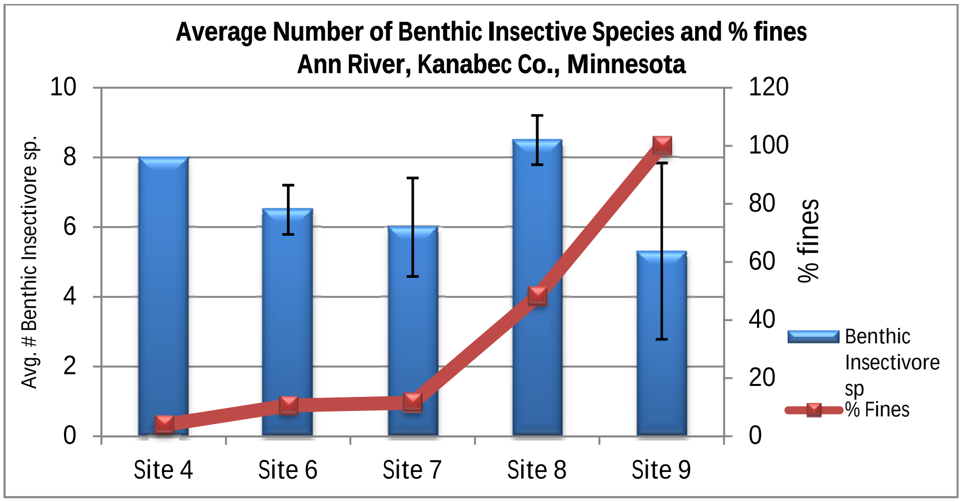
Figure 31. Species and abundance of Simple Lithophils at Site 9 in 1998 and 2007 (July & August). The increase in percent simple lithophils observed in 2007 was largely due to a sharp increase in common shiner individuals.



2.3.7.2 Decrease in fish species that are benthic insectivores

An increase in sediment deposition upon the streambed can reduce habitat for benthic macroinvertebrates causing decreases in taxa richness and overall benthic productivity (Rabeni et al., 2005). This in turn limits food availability for fish species that rely on benthic macroinvertebrates as a primary food source. The diversity and abundance of this trophic group typically decrease with an increase in fine substrates and a reduction in riffle quantity and quality (Berkman and Rabeni, 1987; Rabeni and Smale, 1995; Stauffer et al., 2000).

Figure 32. Benthic insectivore taxa richness vs. percent Fines at Ann River biological monitoring sites



There is some evidence that sedimentation is limiting habitat for benthic insectivores in the Ann River watershed. However, the level of variability in the biological data set, along with some a typical responses in spatial co-occurrence of stressor-response relationships weakens the evidence for this candidate cause. Site 8 repeatedly had one of the highest metric scores for benthic insectivores despite the relative abundance of fine substrate within that reach (Figure 32). Site 8 also has a significant amount of gravel substrate and riffle habitat, which may be sustaining benthic productivity. It will be important to continue monitoring this reach to see if this habitat type remains. There appears to be some streambank instability within this reach and some evidence of a widening stream channel.

Site 9, which is dominated by sand and silt substrates, generally had lower metric scores for benthic insectivores when compared to monitoring stations further upstream. As shown by the error bars in Figure 32, the data from this station was highly variable. Given the abundance of fine substrate at this station, it is unlikely that many benthic insectivorous fish, or other species dependent on coarse substrates, would inhabit this reach for an extended period of time.

Low darter taxa richness

Darters are a general of fishes that have shown to be especially sensitive to changes in habitat. Many species of darter react negatively to impoundment structures, since these fish depend on swift flowing water over relatively clean, coarse stream substrates. Given that coarse substrates are a habitat requirement for many darter species, they are often extirpated or reduced in numbers where excessive siltation of the streambed has taken place (Becker, 1983).

Three species of darter were sampled from the Ann River during MPCA biological sampling efforts; johnny darter (Etheostoma nigrum), logperch (Percina caprodes), and iowa darter (Etheostoma exile). Only one Iowa darter individual was collected (Site 4), so it appears that the population of this species is small and localized within the watershed. Johnny darter, a relatively tolerant darter species, make up a large portion of the darter population in the Ann River (table 17). Karr (1981) found that the presence of Johnny darters in the absence of other darter species can be an indicator of degraded conditions. Logperch are a widely distributed fish species in Minnesota and Wisconsin and can be found in diverse habitats, ranging from swift current to the open water of larger lakes (Becker, 1983).

Table 17. Breakdown of Ann River Darter populations as sampled by MPCA. (Includes all MPCA fish sampling data for the Ann River)

Darter Species (common name)	Total # sampled	% of darter population
Johnny darter	817	73%
Logperch	307	37%
Iowa darter	1	< 1%

Increased siltation of the streambed has reduced darter diversity and abundance in Minnesota and other Midwestern states. In southern Ohio, the slenderhead darter's decline was correlated with increasing levels of siltation within its range in that state (Trautman, 1957). The gilt darter is currently listed as a species of special concern in the state of Minnesota. Although this species can still be found in high numbers in certain St. Croix River basin streams, populations are in decline primarily due to their sensitivity to siltation (Hatch, 1986; Proulx, 2007). Gilt darters favor microhabitats with large amounts of cobble, and to a lesser extent gravel, and generally avoid microhabitats with substantial amounts of depositional substrata (sand, silt or debris) (Skyfield and Grossman, 2007). These microhabitat types do exist in the Ann River watershed, but their abundance and quality are being reduced by sediment inputs from various land-uses.

The absence or scarcity of several darter species in the Ann River is noteworthy, and potentially related to specific stressors caused by land-use and hydrological changes in the watershed. Slenderhead darter, gilt darter, and iowa darter are found in many streams within the St. Croix River drainage basin, but are absent or very scarce in the Ann River. Without historic records of the fishery, it is difficult to know whether or not these species were historically present in greater numbers.

2.3.8 Macroinvertebrate response

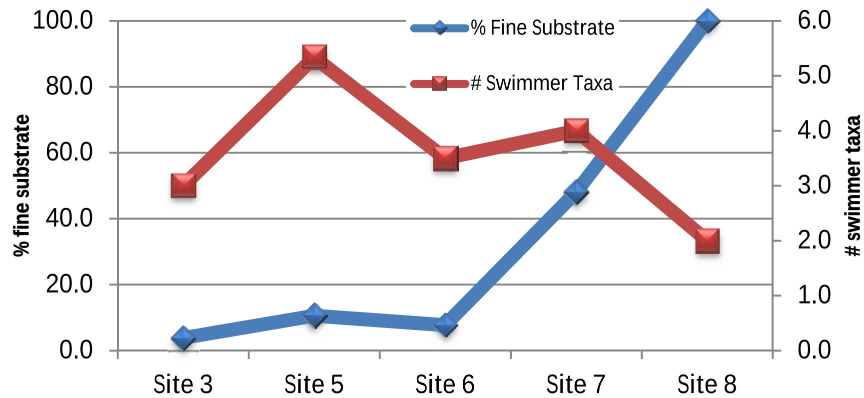
2.3.8.1 Swimmer taxa

Habitat and behavioral designations of aquatic macroinvertebrates relate to the way the organism moves around and searches for food. Various adaptations and morphological features often determine which invertebrate species can occupy a specific habitat type (Merrit et al., 1996). Macroinvertebrate functional and feeding groups from the Ann River were evaluated to evaluate sedimentation as a candidate stressor.

Macroinvertebrate taxa classified as "swimmers" have the ability to control the direction and velocity of their movements, allowing them to occupy diverse habitats throughout the water column. The ability of swimmer taxa to elevate off of the streambed render them less dependent on clean, productive benthic habitats and less vulnerable to sedimentation.

There does not appear to be a positive correlation between percentage of fine substrate and swimmer taxa richness in the Ann River. In fact, swimmer taxa richness appears to be lowest at sites with greater percentage of fines in the benthic zone.

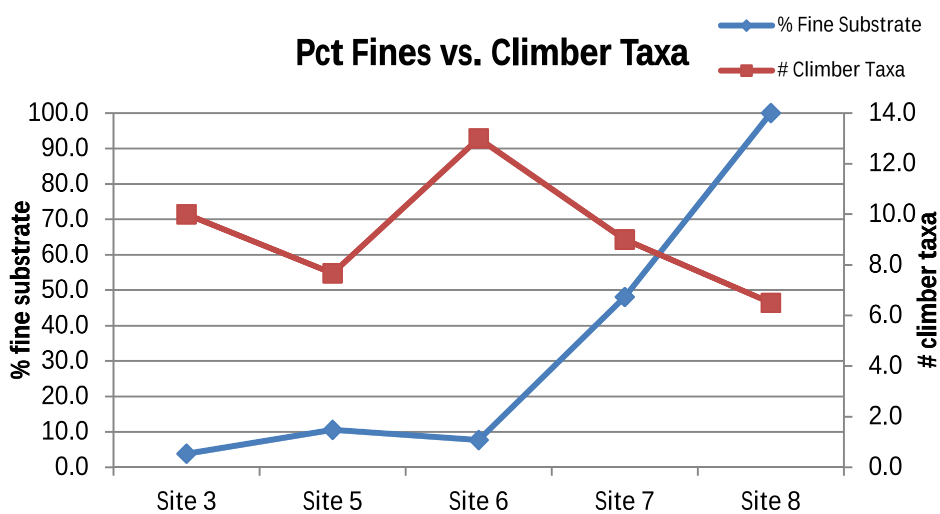
Figure 33. Comparison of Percent Fine Substrate and Average Swimmer taxa Richness at Ann River Biological Monitoring Sites



2.3.8.2 Climber taxa

Climber taxa attach to and feed in submerged vegetation. The density of climber taxa has been shown to significantly increase with increases in deposited sediment (Rebeni et al., 2005). No such relationship was observed when comparing the five biological monitoring sites on the Ann River (Figure 34). It is possible that climber taxa richness is more closely related to the abundance of aquatic macrophytes at these sites.

Figure 34. **Comparison of Percent Fine Substrate and Average Climber taxa Richness at Ann River Biological Monitoring Sites**



2.3.8.3 Clinger Taxa

Aquatic macroinvertebrates classified as "clingers" are often dorsoventrally flattened and have the ability to remain stationary on bottom substrates in fast-flowing water. Given that their preferential habitats are coarse particles on the streambed and the interstitial spaces between them, clinger diversity and abundance can decrease as sedimentation and substrate embeddedness increases (Erhart et al., 2002).

There does appear to be a noticeable trend of decreasing clinger taxa richness as percent fine substrate increases in the Ann River (Figure 35). Although clinger taxa richness varied at each site across multiple sampling events, Site 9 consistently had the lowest taxa richness of this macroinvertebrate habit group. The low gradient conditions and lack of riffle habitat (some of which is probably due to natural features) at Site 9 are likely the main factors behind the lack of clinger taxa. Earlier in this report, Site 7 was compared against a reference reach in the Kettle River. In terms of clinger taxa, the reference site from the Kettle River (see section 1.3 for reference site information) had 25 clinger taxa, while the comparable Ann River Site 7 had an average of 21 (Max = 23, Min = 17).

Figure 35. Percent Fine Substrate compared against Clinger taxa at Ann River Biological Monitoring Sites

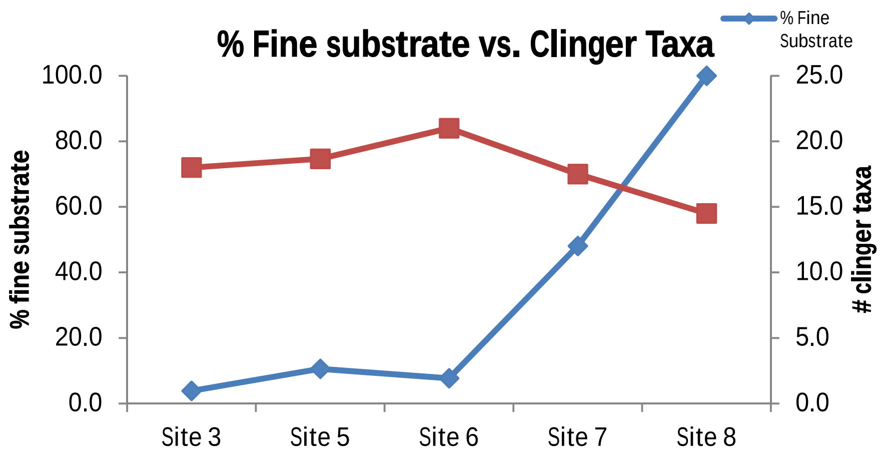


Table 18. **Candidate Cause #1: Substrate Embeddedness (Summary)**

Types of Evidence	Strength of Evidence Score		Comment
Evidence using data from Ann River	Fish	Invert	
Spatial / temporal co-occurrence	+	0	Simple lithophilic spawning fish and darter taxa richness decrease with increasing fine sediment deposition. Invertebrate response to sediment is difficult to determine given lack of reportable data, although high invert scores were observed within reaches with an abundance of fine substrate (i.e. Site 9). Other habitat types present within this reach (i.e. woody debris) likely provide suitable habitat for some invertebrate taxa.
Evidence of exposure, biological mechanism	+	0	The fish that inhabit Site 9 on a regular basis are those that are tolerant of wide variety of habitat types (creek chub, common shiner, white sucker). Missing from Site 9 is a healthy population of longnose dace, which require suitable riffle habitats for reproduction. Longnose dace were sampled in relatively high numbers at several other Ann River monitoring stations that possess stream subtrates that are less embedded with fine sediments.
Causal Pathway	++	++	Land-uses in the watershed appear to be contributing to higher sediment inputs to the river via streambank erosion and riparian disturbance. A higher rate of sediment deposition is being observed in the lower reaches of the river where stream gradient decreases and also within areas of active cattle grazing.
Field Evidence of stressor-response	0	0	Evidence for stressor-response is somewhat inconsistent. While overall fish IBI score, % lithophils, benthic insectivores, and sensitive fish species are the lowest where fine sediment is most abundant (Site 9), the dose-response effect is inconsistent.
Field experiments / manipulation of exposure	NE	NE	No data available.
Laboratory analysis of site media	NE	NE	No data available.
Temporal sequence	0	0	The probable sources of this stressor (agriculture, logging, mineral extraction, development) occurred over a long period of time (100 years or more). The biological record is much shorter, making it difficult to evaluate the temporal sequence of cause and effect. Current land-uses (esp. cattle grazing in riparian corridor) appear to be a threat to stream stability and quality benthic habitat.
Verified of tested predictions	NE	NE	No data available.
Symptoms	+	0	Decrease in simple lithophilic spawners, benthic insectivores, and intolerant taxa at Site 9 are potentially a symptom of increased substrate embeddedness in the lower Ann River.
Evidence using data from other sources			

Types of Evidence	Strength of Evidence Score		Comment
Mechanistically plausible cause	+	+	Stressor is consistent with known principles of biology, chemistry, and physics and with properties of the affected organisms and ecosystem
Stressor-response in other field studies	+	+	Substrate embeddedness resulting from excess fine sediment is a common stressor in watersheds dominated by agricultural land-uses (Nerbonne and Vondracek, 2001; Stewart et. al 2007). The Groundhouse River (adjacent to the Ann River) was determined to have degraded biological communities as a result of this stressor.
Stressor-response in other lab studies	+	+	Sullivan and Watson, 2009;
Stressor-response in ecological models	+	+	Richards and Host, 1994;
Manipulation experiments at other field sites	NE	NE	No data available
Analogous stressors	++	++	Groundhouse River (Minnesota); Hardwood Creek (Minnesota); Little Rock Creek (Minnesota)
Multiple Lines of Evidence			
Consistency of Evidence	+	0	Lack of reportable invertebrate data from several of the monitoring sites introduces a high level of inconsistency in the evidence.
Explanatory power of evidence	++	0	Inverts: The use of a separate IBI for "low gradient" streams at Site 9 may mask some of the negative effects of sedimentation within this reach.

2.4 Candidate Cause #2: Low Dissolved Oxygen

Dissolved oxygen (DO) refers to the concentration of oxygen gas within the water column. Low or unstable concentrations of DO can have detrimental effects on many fish and macroinvertebrate species (Davis, 1975; Nebeker et al., 1991). DO concentrations change seasonally and daily in response to shifts in ambient air and water temperature, along with various chemical, physical, and biological processes within the water column. If dissolved oxygen concentrations become limited or fluctuate dramatically, aerobic aquatic life can experience reduced growth or fatality (Allan, 1995). Many species of fish avoid areas where dissolved oxygen concentrations are below 5 mg/L (Raleigh et al., 1986).

The class 2B* water quality standard for DO in Minnesota is 5 mg/L as a daily minimum. Additional stipulations have been recently added to this standard. The following is from the Guidance Manual for Assessing the Quality of Minnesota Surface Waters (MPCA, 2009):

Under revised assessment criteria beginning with the 2010 assessment cycle, the DO standard must be met at least 90 percent of the time during both the 5-month period of May through September and the seven-month period of October through April. Accordingly, no more than 10 percent of DO measurements can violate the standard in either of the two periods.

Further, measurements taken after 9:00 in the morning during the five-month period of May through September are no longer considered to represent daily minimums, and thus measurements of > 5 DO later in the day are no longer considered to be indications that a stream is meeting the standard.

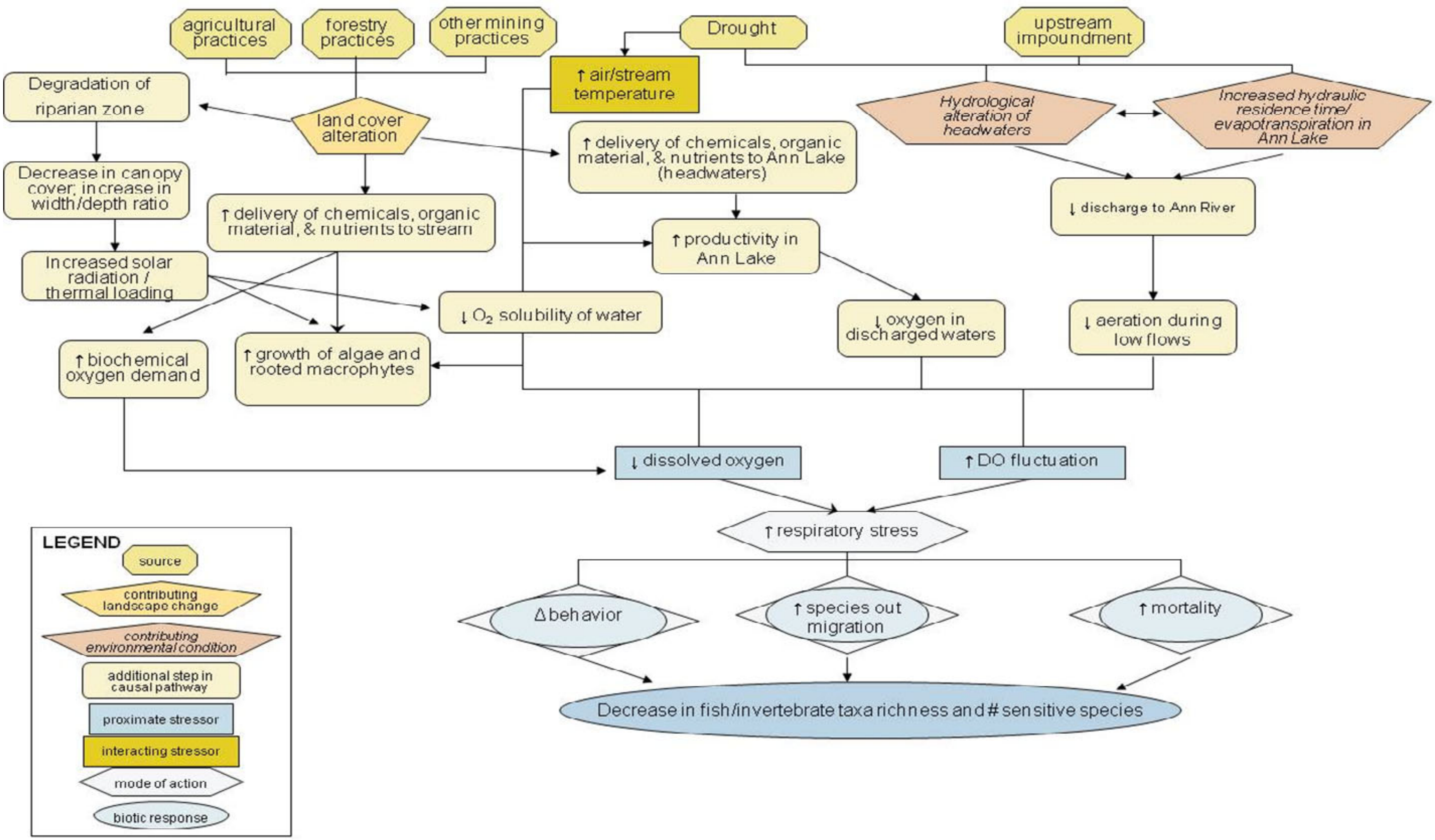
A stream is considered impaired if 1) more than 10 percent of the “suitable” (taken before 9:00 AM) May through September measurements, or more than 10 percent of the total May through September measurements, or more than 10 percent of the October through April measurements violate the standard, and 2) there are at least three total violations.

In most streams and rivers, the critical conditions for stream DO usually occur during the late summer season when water temperatures are high and stream flows are reduced to baseflow. As temperatures increase, the saturation levels of dissolved oxygen decrease. Increased water temperature also raises the dissolved oxygen needs for many species of fish (Raleigh et al., 1986). Low dissolved oxygen can be an issue in streams with slow currents, excessive temperatures, high biological oxygen demand, and/or high groundwater seepage (Hansen, 1975).

2.4.1 Sources, pathways, and biological effects related to low dissolved oxygen

The conceptual model in Figure 36 covers the potential sources, pathways, and biological effects of low dissolved oxygen concentrations in the Ann River watershed. The proposed sources of low dissolved oxygen in the watershed include both anthropogenic and natural factors.

Figure 36. Conceptual Model for Low Dissolved Oxygen Concentrations as a Stressor in the Ann River Watershed



2.4.2 Ann River dissolved oxygen data

Dissolved Oxygen data were collected from the Ann River watershed from 2004-2009. DO measurements include instantaneous readings, diurnal profiles, and synoptic longitudinal profiles. All measurements were collected using calibrated electronic monitoring instruments.

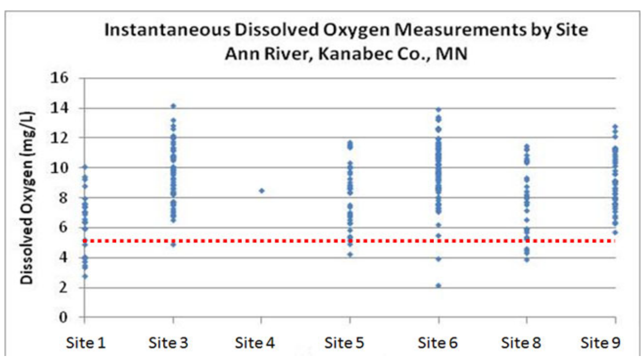
2.4.3 Instantaneous data

Most instantaneous DO measurements from the Ann River watershed were taken during discrete grab sampling events. As a result, the vast majority of readings were taken after 9:00 AM and the results likely have some bias towards daily maximum DO concentration. In addition, the time of sampling was not recorded for a significant amount of the dissolved oxygen readings, which makes these data less valuable for evaluating DO as a potential stressor. Time of day is an important detail for DO sampling because DO concentrations usually reach their daily maximum during the afternoon and early evening hours.

A total of 284 instantaneous measurements of DO were collected, spanning five sites from the headwaters of the Little Ann River to the mouth of the Ann River near Fish Lake. Most instantaneous measurements (94 percent) of dissolved oxygen were above the class 2B standard of 5 mg/L. Based on these data, it appears that dissolved oxygen concentrations are suitable for supporting aquatic life during daylight hours under normal flow conditions. However, several instantaneous DO measurements dropped below the standard of 5 mg/L, most commonly at Sites 5, 7, and 8 on the mainstem of the Ann River.

Site 1 on the Little Ann River also appears to have DO concentrations regularly in violation of the DO standard. This station is located immediately downstream of an impounded reservoir (DeWitt Pool Wildlife Management Area). This reservoir is a shallow impounded wetland area with hydric peat soil/substrate and an abundance of aquatic vegetation. These conditions likely limit the dissolved oxygen concentration of the water that outlets in to the Little Ann River. As shown in Figure 37, DO concentrations on the Little Ann River recover at Site 3, a site that is probably more representative of DO concentrations in that stream.

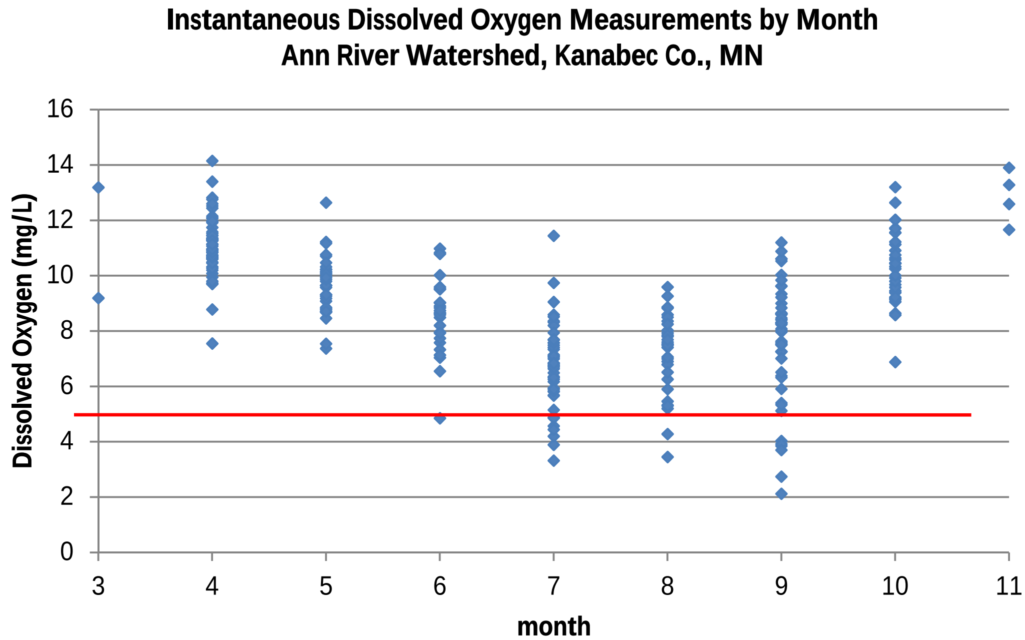
Figure 37. All instantaneous dissolved oxygen measurements by site. (red line represents 5.0 mg/L standard for class 2B)



Sites 1 and 3 are located on the Little Ann River. Sites 4-8 are located on the mainstem of the Ann River. Very few violations of the class 2B DO standard were observed during instantaneous sampling.

The months of July, August, and September appear to be the critical months for low dissolved oxygen concentrations in the watershed. Typically, low DO concentrations during this time of year are correlated with rises in ambient water and air temperature, increases in primary production (photosynthesis and respiration), and decreased streamflow.

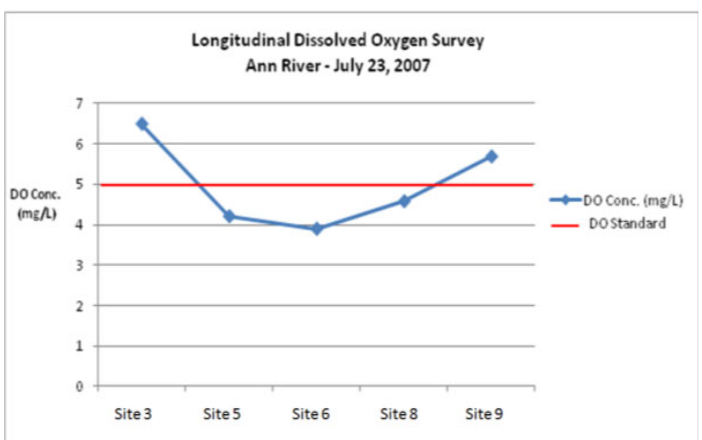
Figure 38. Ann River Instantaneous Dissolved Oxygen Measurements by month



2.4.4 Longitudinal DO data

Longitudinal (synoptic) DO surveys were conducted in July of 2007 and September of 2009. In the July 2007 longitudinal survey, DO concentrations at Sites 5, 6, and 7 fell below the water quality standard of 5 mg/L (Figure 39). The DO concentrations at Site 3 (Little Ann River) and Site 9 (Ann River at CR 14 bridge) were above the DO standard. It is possible that DO levels in Ann Lake play a role in the lower concentrations observed in the upper half of the Ann River. Ann Lake is currently listed as impaired for excess nutrients, and dissolved oxygen profiles taken in the lake indicate that dissolved oxygen concentrations are occasionally below established standards to protect aquatic life.

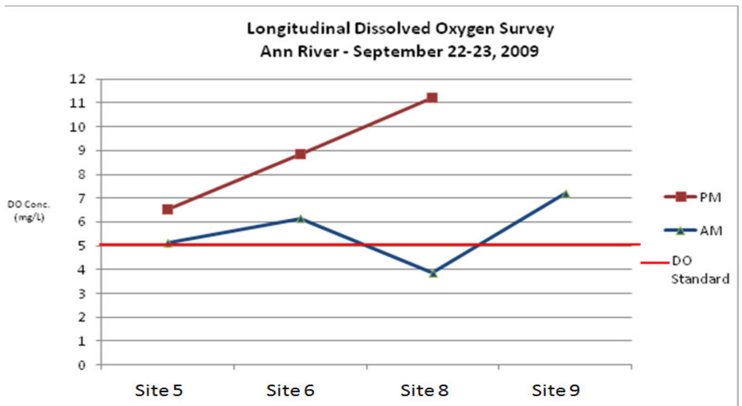
Figure 39. Synoptic Longitudinal DO Profile from the Ann River, July 2007.



Three stations in the watershed were found to be in violation of the class 2B water quality standard for dissolved oxygen.

The synoptic DO survey completed in 2009 included both afternoon and early morning DO measurements. The survey produced only one measurement below the standard-the AM measurement at Site 6 (3.85 mg/L, Figure 39). The evening measurement taken at this station taken the previous evening was 11.20 mg/L. The high level of DO fluctuation within a 12 hour period at this site may be an indicator of significant primary production (photosynthesis and respiration of aquatic plants/algae) due to organic pollution from nutrient enrichment. Ann River nutrient concentrations and their potential effects on dissolved oxygen are discussed later in this section.

Figure 40. Synoptic Longitudinal Dissolved Oxygen data collected in 2009



2.4.5 Diurnal DO

Diurnal dissolved oxygen surveys were conducted at a selection of Ann River sites in 2007, 2008, and 2009. The goal of these surveys was to observe 24-hour (diurnal) fluctuations in dissolved oxygen concentration and observe daily maximum and minimum concentrations. Another benefit of diurnal DO data is the ability to quantify the duration of time that a given site violated the dissolved oxygen water quality standard of 5 mg/L.

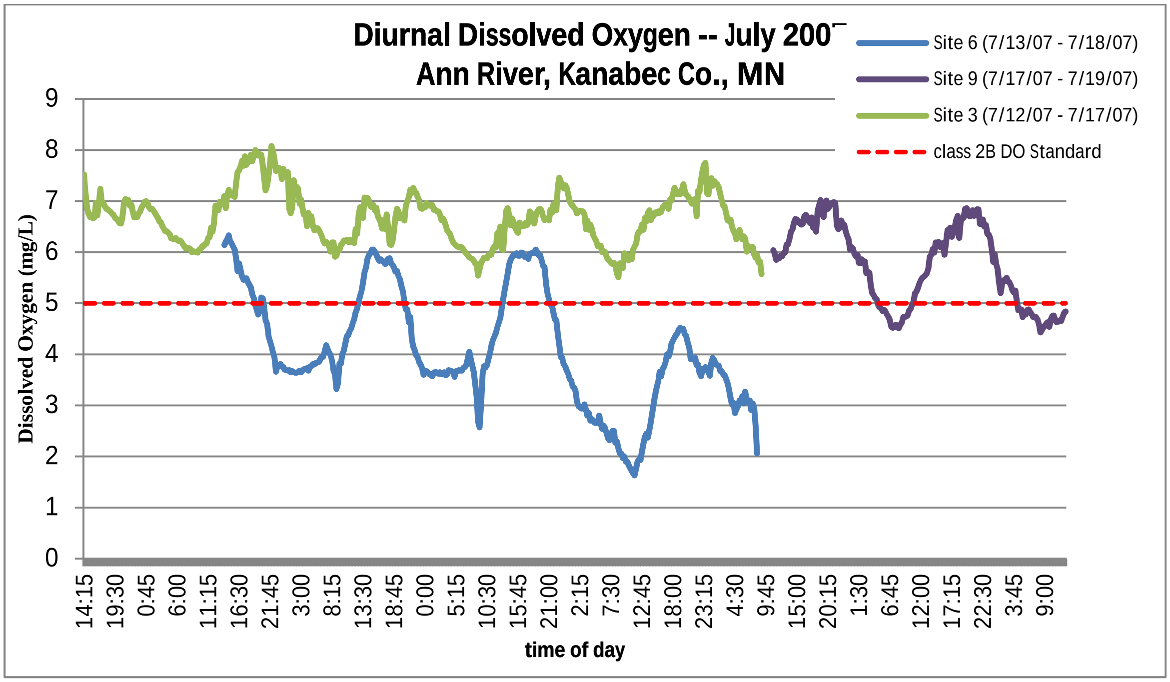
Diurnal dissolved oxygen measurements were collected by deploying YSI 6920 multi-parameter sondes in representative locations within selected river reaches. “Representative” means that the sample locations had adequate flow (no backwater areas) and stream substrate conditions were similar to those found within the rest of the reach. The YSI Sondes were calibrated prior to deployment and set to log stream temperature, specific conductivity, pH, turbidity, and dissolved oxygen at 15-minute intervals. Sondes were deployed for an average duration of 3.9 days (max = 9.1 days; min = 1.8 days).

2.4.5.1 2007 Diurnal data

Diurnal surveys were completed at three stations in the Ann River watershed during July and September of 2007. Both mainstem Ann River Sites (6 and 9) had dissolved oxygen concentrations that fell below water quality standard during the monitoring period (Figure 40). DO concentrations at Site 3 on the Little Ann River met the standard for the entire period of monitoring of 90.75 hours (Figure 40). Violations of the DO standard at Sites 6 and 9 occurred during early morning hours, and the average duration of time below the DO standard was 17.25 hours at Site 6, and 5.75 hours at Site 9.

Site 9 was monitored again during mid-September to evaluate the seasonal differences in DO concentration and flux at this station. No violation of the standard occurred during the 218+ hours of monitoring conducted and the magnitude of diurnal fluctuation in DO concentrations was reduced (Figure 41).

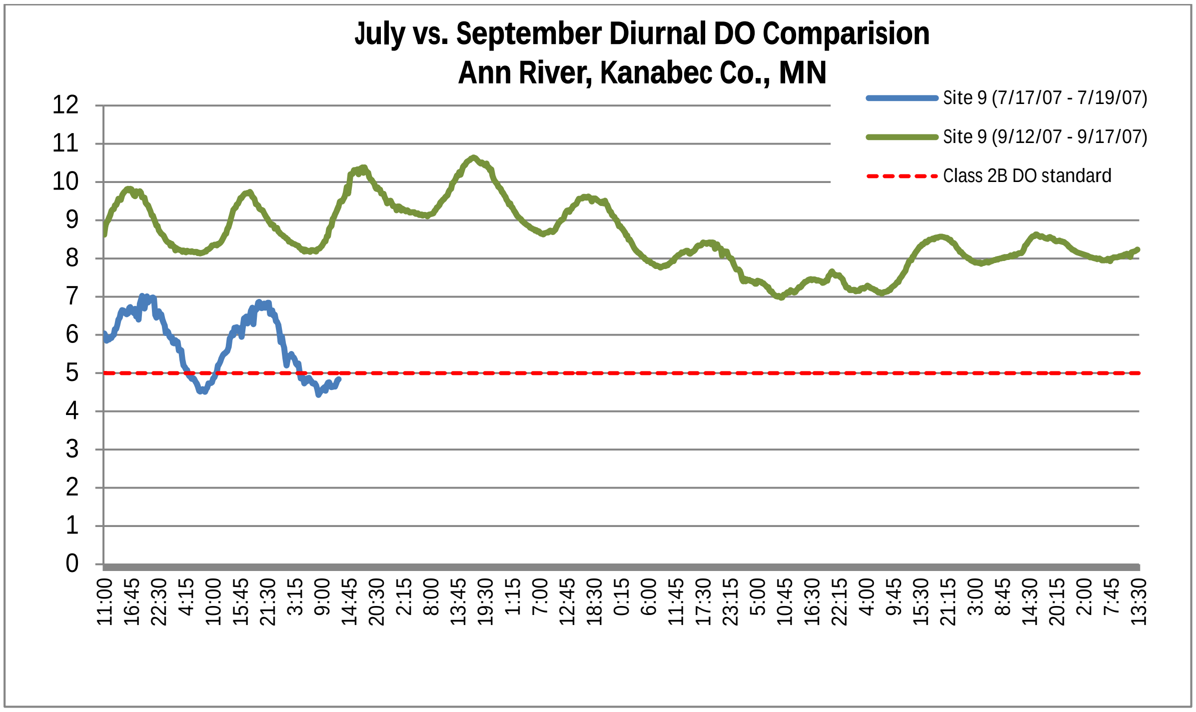
Figure 41. Diurnal Dissolved Oxygen data for Ann River Sites 3, 6, and 9 during the Month of July in 2007.



Site	# Readings	Min (mg/L)	Max (mg/L)	% readings below 5 mg/L	Avg. duration below 5 mg/L (hours)	Max duration below 5 mg/L (hours)	Avg. 24-hr Flux (mg/L)
6	363	1.63	6.33	76%	17.25	35.5*	3.03
9	199	4.43	7.02	29	5.75	8.25*	2.47
3	460	5.51	8.08	0	0	0	1.84

* Monitoring device was pulled when DO concentration was below 5 mg/L, therefore exact duration is unknown

Figure 42. Diurnal Dissolved Oxygen data collected at Site 9 in September 2007*.



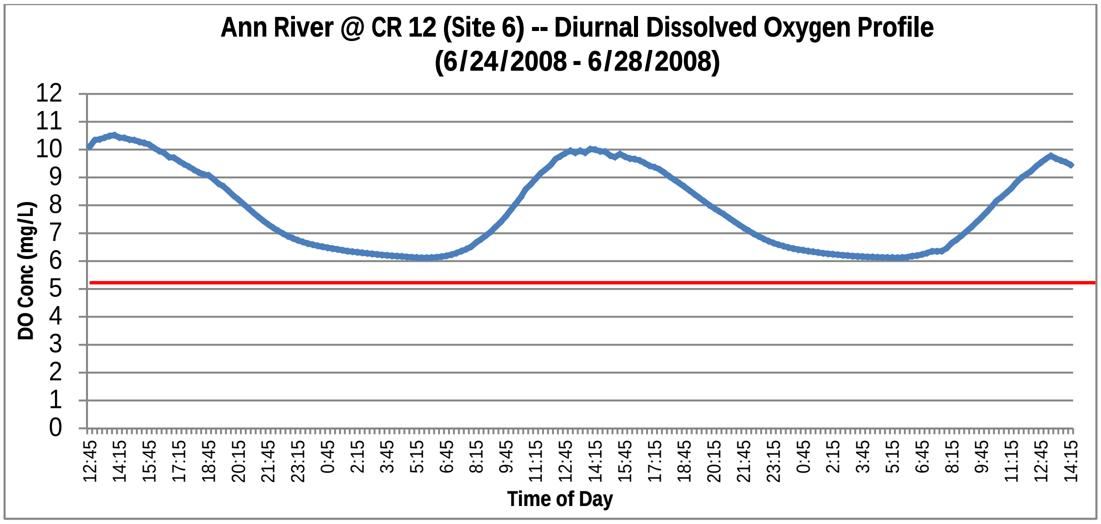
Site	# Readings	Min(mg/L)	Max(mg/L)	% readings below 5 mg/L	Avg. Duration below 5 mg/L (hours)	Max Duration below 5 mg/L (hours)	Avg. 24-hr Flux(mg/L)
9 (July)	199	4.43	7.02	29	5.75	8.25*	2.47
9 (Sept)	874	6.97	10.64	0	0	0	1.45

* July 2007 data is also shown on the graph to illustrate differences between DO concentrations in July and September.

2.4.5.2 2008 Diurnal Data

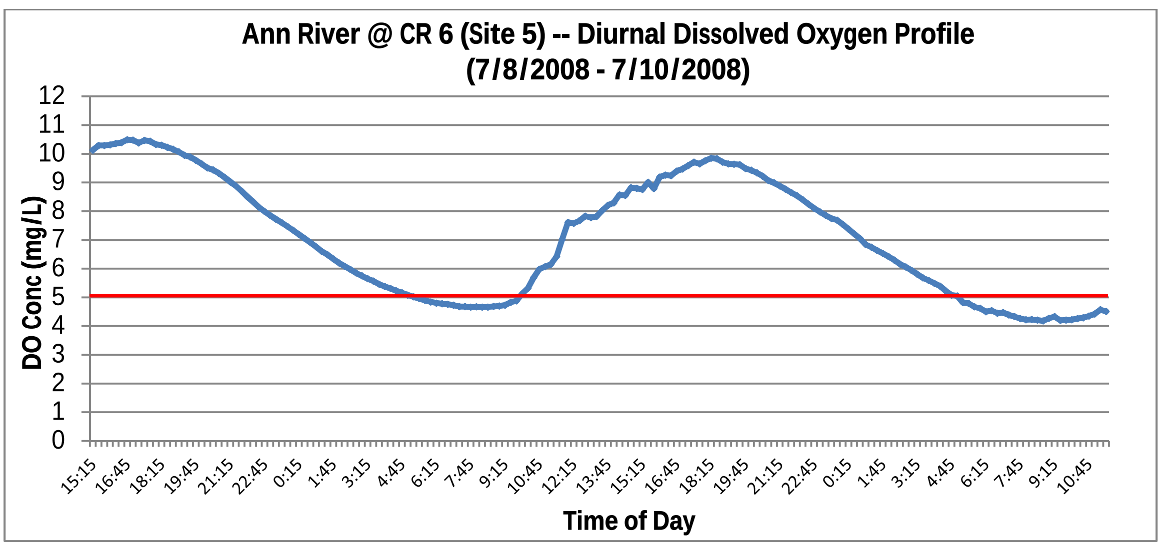
In 2008, diurnal dissolved oxygen measurements were collected in June, July, and August from five monitoring sites on the Ann River. The only measurements below the DO standard occurred in July at Site 5 (Figure 44). The duration of time below the standard was short (4.5 hours) in comparison to violations of the standard observed during the 2007 monitoring year. The minimum DO concentration observed (4.18 mg/L) was also higher than the minimums observed in 2007. Generally, the dissolved oxygen conditions in the Ann River appeared to be much more suitable for aquatic life during the summer and Fall of 2008 than in was in 2007.

Figure 43. Diurnal Dissolved Oxygen data collected at Site 6 in June of 2008



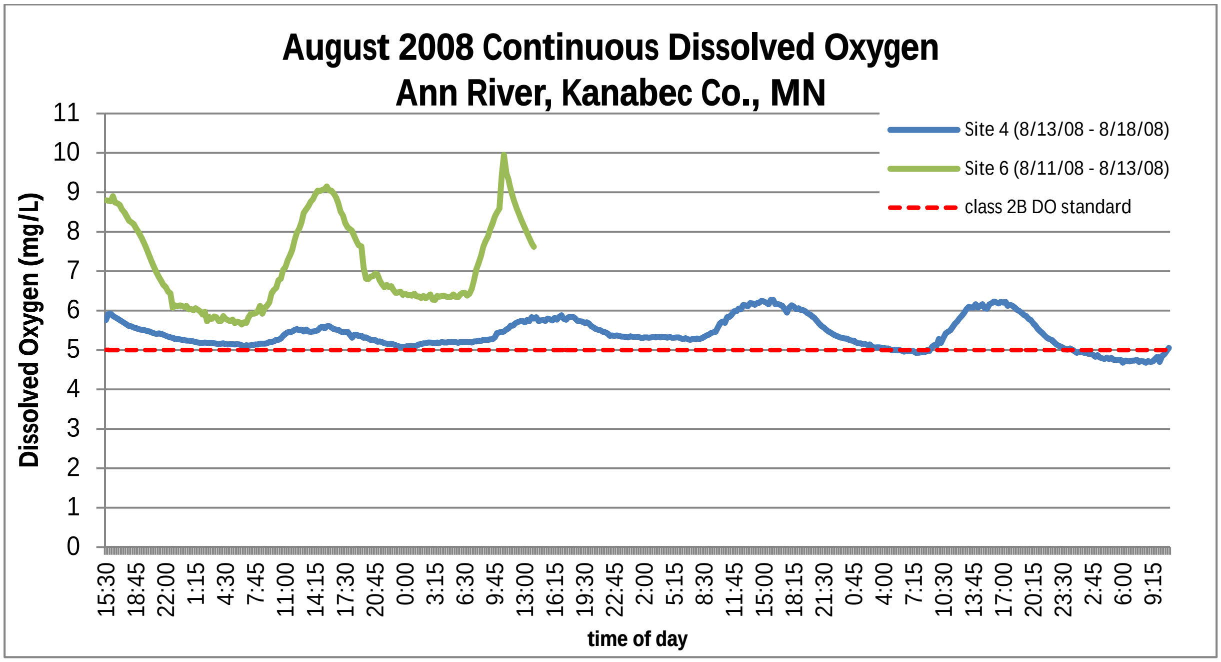
# Readings	Min	Max	% readings below 5 mg/L	Avg. Duration below 5 mg/L (hours)	Max Duration below 5 mg/L (hours)	Avg. 24-hr Flux (mg/L)
199	6.12	10.52	0	0	0	4.14

Figure 44: Diurnal Dissolved Oxygen data collected at Site 5 in July of 2008



# Readings	Min	Max	% readings below 5 mg/L	Avg. Duration below 5 mg/L (hours)	Max Duration below 5 mg/L (hours)	Avg. 24-hr Flux(mg/L)
178	4.18	10.49	25%	4.5	4.5	5.83

Figure 45. Diurnal Dissolved Oxygen data collected at Sites 4 and 6 in August of 2008



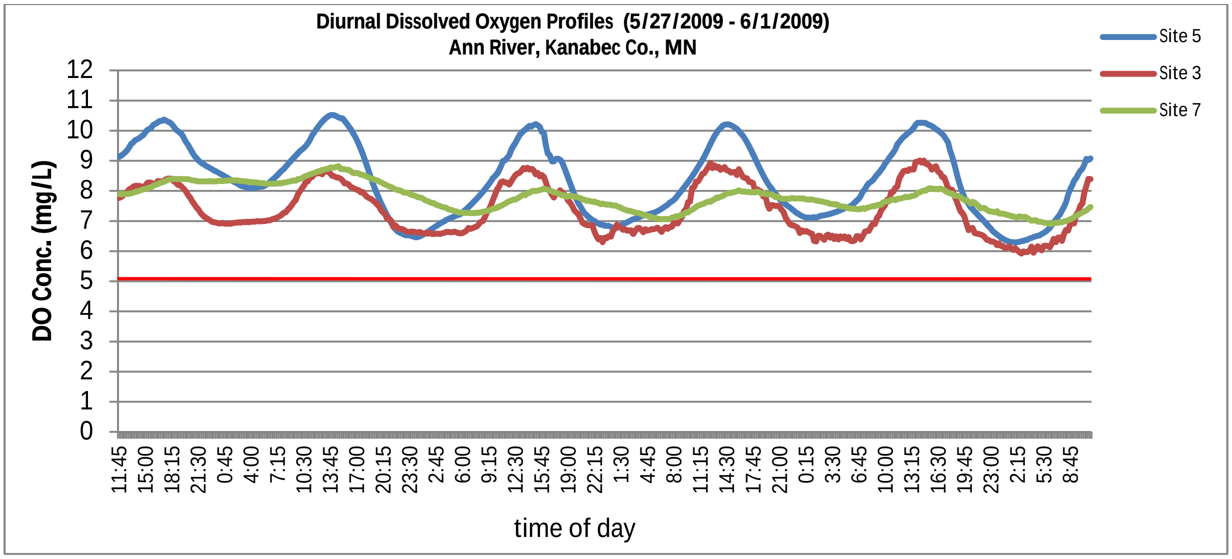
Site	# Reading	Min(mg/L)	Max(mg/L)	% readings below 5 mg/L	Avg. Duration below 5 mg/L (hours)	Max Duration below 5 mg/L (hours)	Avg. 24-hr Flux(mg/L)
4	463	4.68	6.27	12%	2.75	10.25	0.98
6	187	5.65	9.95	0	0	0	3.43

2.4.5.3 2009 Diurnal data

Continuous monitoring data were collected from three Ann River sites during late May to early June. All readings were above the 5 mg/L standard during this monitoring period. Water temperatures during the monitoring period were still several degrees below the typical water temperatures observed in mid-summer months. Therefore, the 2009 data does not likely represent critical periods for low dissolved oxygen concentrations.

The diurnal profiles from 2009 suggest that DO conditions at Site 4 and Site 6 are driven by similar factors, while concentrations at Site 9 appear to be responding to a different set of conditions in the lower reaches of the river. The diurnal curve at Site 9 is flatter than the other two sites. This could be a result of less primary production (photosynthesis and respiration) or the presence of groundwater entering the stream. Site 9 is also located within a reach of the river with a heavily forested riparian buffer, and the shade provided from the forest canopy may be stabilizing stream temperatures and thus flattening the diurnal DO curve.

Figure 46. Diurnal Dissolved Oxygen Profiles from Sites 4, 6, and 9 on the Ann River in May/June 2009



Site	# Readings	Min	Max	% readings below 5 mg/L	Avg. duration below 5 mg/L (hours)	Max duration below 5 mg/L (hours)	Avg. 24-hr flux(mg/L)
4	487	5.92	9.01	0	**0**	0	2.35
6	487	6.29	10.52	0	**0**	0	3.37
9	485	6.90	8.82	0	**0**	0	1.02

2.4.6 Dissolved oxygen and stream temperature

As previously noted, the solubility of oxygen decreases with increasing water temperature. In some cases, the combination of a drop in dissolved oxygen concentrations and increase in stream temperature can induce stress on many species of fish (Allan, 1995). Table 18 provides a summary of the water temperatures recorded during the diurnal DO monitoring periods in 2007 and 2008. There does not appear to be any site specific connections between low dissolved oxygen and stream temperature based on the available data. Average water temperatures were higher during the 2008 continuous monitoring periods, while DO concentrations were above the standard more often during diurnal surveys that year. Further monitoring and/or a modeling effort are recommended in order to better understand the processes driving apparent dissolved oxygen standard violations in the Ann River.

Table 19. Summary of stream Temperature and Dissolved Oxygen Data from Continuous Monitoring conducted in 2007 and 2008

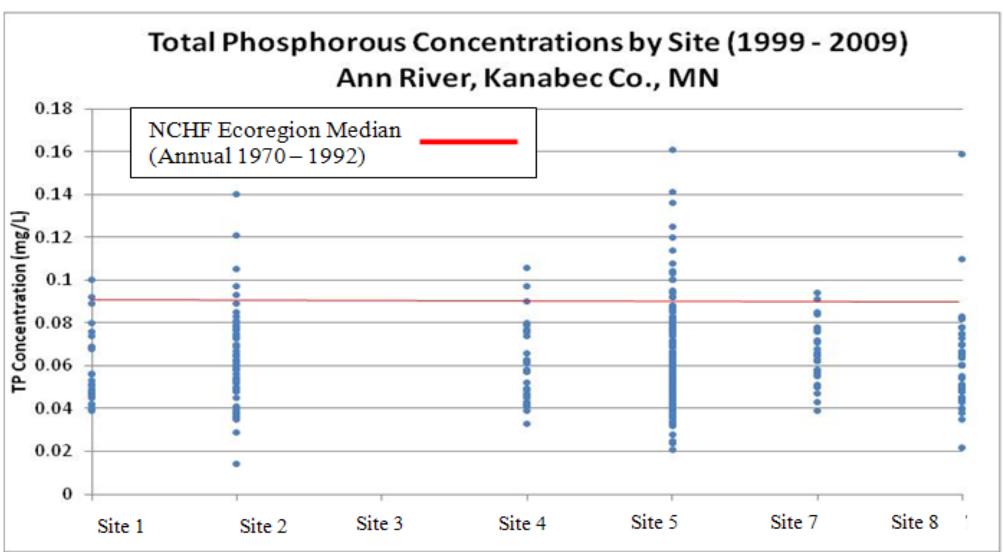
Date (2007)	Site	Max Water Temperature	Min Water Temperature	Average Water Temperature	% DO readings below 5 mg/L
7/19/07 – 7/23/07	6	25.52	17.31	21.08	76%
7/12/07 – 7/17/07	3	23.09	16.64	20.26	0%
7/17/07 – 7/19/07	9	23.91	21.55	22.73	29%
Date (2008)	**Site**	**Max Water Temperature**	**Min Water Temperature**	**Average Water Temperature**	**% DO readings below 5 mg/L**
6/24/08 – 6/26/08	6	27.56	22.26	24.78	0%
7/8/08 – 7/10/08	5	26.52	20.38	23.21	25%
8/13/08 – 8/18/08	4	26.03	18.06	22.15	12%
8/11/08 – 8/13/08	6	23.37	18.12	20.70	0%

2.4.7 Dissolved oxygen and phosphorous

Elevated concentrations of phosphorous and related spikes in primary production have been frequently linked to low dissolved oxygen concentrations in rivers and lakes (Smith et al., 1999; Allan, 1995). Excess nutrients in surface water appear to be an area of concern for the Ann River watershed, as phosphorous concentrations of two lakes connected to the Ann River (Fish Lake and Ann Lake) are routinely above state water quality standards.

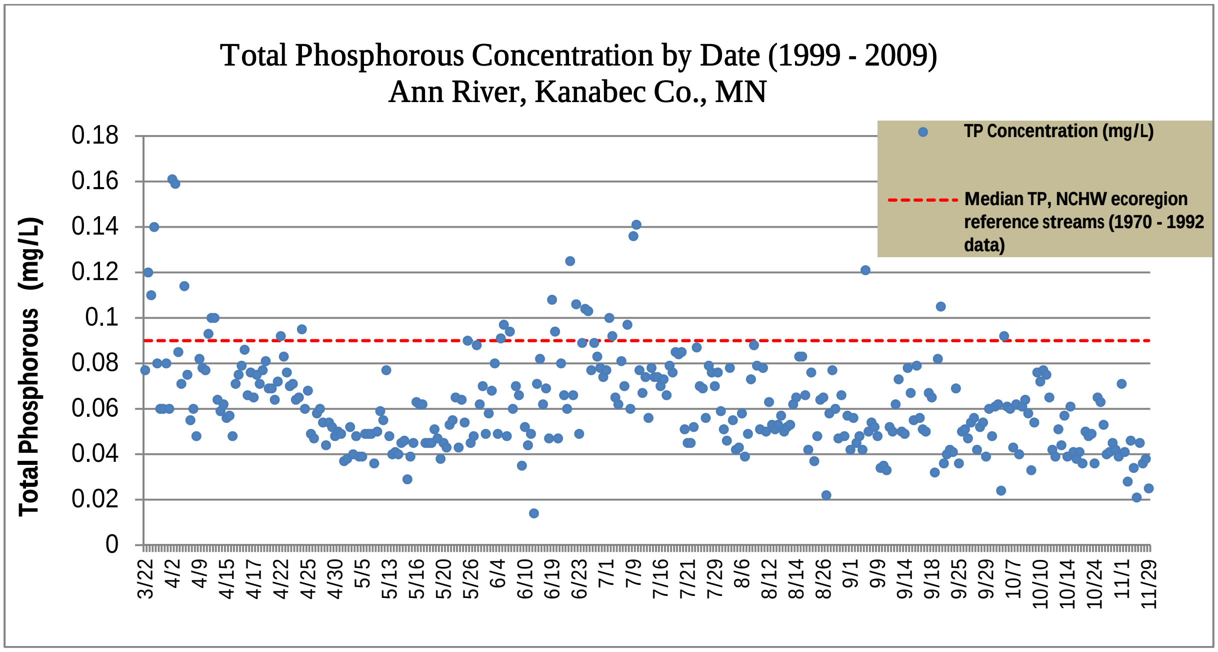
There are currently no nutrient standards for streams and rivers of Minnesota, although comparisons can be made against minimally impacted streams and rivers of Minnesota as presented in McCollor and Heiskary (1993). Total phosphorous concentrations in the Ann River are, on occasion, considerably higher than the median value for minimally impacted streams in the North Central Hardwood Forests ecoregion of Minnesota (Figure 47). Site 6 appears to have some of the higher TP concentrations and was also one of the sites with violations of the dissolved oxygen standard. There is a cattle feedlot and pasture area located upstream of this site and cattle are often in or near the stream, which may contribute to nutrient loading on that reach of the river.

Figure 47. Total Phosphorous Concentration by Monitoring Site



Based on available data, nutrient enrichment may be a factor in the low dissolved oxygen concentrations observed in the Ann River. Diurnal fluctuation in DO concentration during several of the continuous monitoring periods indicate that photosynthetic activity can be relatively high during the daylight hours (producing high DO concentrations) followed by respiration in the evening to early morning (dropping DO concentrations below 5 mg/L). Also, biological data from several sites reveals that a high percentage of the invertebrates collected were from the family amphipoda, which typically thrive in nutrient rich environments.

Figure 48. A Seasonal View of Total Phosphorous Concentrations in the Ann River*



* These data include samples from all Ann River monitoring sites from 1999 – 2009.

2.4.8 Ann River Biological Oxygen Demand (BOD)

Biological Oxygen Demand (BOD) is a chemical procedure for determining the uptake rate of dissolved oxygen by the biological organisms in a body of water. Most pristine rivers will have a five-day carbonaceous BOD below 1 mg/L. Moderately polluted rivers may have a BOD value in the range of 2 to 8 mg/L. Samples were collected for BOD as part of a longitudinal DO survey conducted July 23, 2007. Sampling results indicated that BOD concentrations were below detection levels (0.5 mg/L) at all four sites that were monitored (Table 20). It does not appear that high BOD concentrations are responsible for low dissolved oxygen concentrations in the Ann River. Additional data would be valuable for proving that this is not a plausible causal pathway for low DO conditions in the Ann River.

Table 20. Dissolved Oxygen, stream temperature, and BOD results from a Longitudinal Survey conducted July 24, 2007

Site	Date/Time	Temp (C)	DO (mg/L)	BOD	NCHF Reference**
5	7/24/07 06:40	22.71	4.20	< 0.5 mg/L*	2.2 mg/L
6	7/24/07 06:20	22.31	3.89	< 0.5 mg/L*	2.2 mg/L
8	7/24/07 06:10	21.70	4.57	< 0.5 mg/L*	2.2 mg/L
9	7/24/07 05:45	22.65	5.68	< 0.5 mg/L*	2.2 mg/L

* BOD concentration was below lab detection limit (0.5 mg/L)

**Median BOD concentration of minimally impacted streams and rivers of the North Central Hardwood Forest (NCHF) ecoregion. Includes samples taken during summer months from 1970 – 1992.

2.4.9 Dissolved oxygen and streamflow

Late summer sags in dissolved oxygen concentration are common in Minnesota streams when flow is primarily sustained by groundwater inputs and precipitation events. Streams that do not have strong groundwater inputs, like the Ann River, can be especially susceptible to extremely low baseflow conditions during dry spells or prolonged drought. As streamflow decreases and water within the stream channel becomes stagnant, dissolved oxygen concentrations are often reduced due to lack of re-aeration and an increase in water temperature. Observations from the field during the summer of 2007 suggest that many of the low DO readings taken that summer were the result of low streamflow, which created stagnant pools and reduced re-aeration (figure 48a).

There is no flow record available for the 2007 monitoring year, so it is not possible to quantitatively compare the hydrograph from that year to subsequent years when low flow periods did not seem to be as severe. The short temporal scale of monitoring activities effectively limits the conclusions that can be made regarding the relationships between streamflow and low dissolved oxygen contentrations. However, the evidence collected from the field during the summer of 2007 suggests that reduced streamflow is a plausible source for low DO conditions in the Ann River.

Figure 48a. Photos from the Ann River watershed in August, 2007. Surface water in the stream was reduced to stagnant pools and little to no re-aeration was occurring in riffle areas.

2.4.10 Ann River dissolved oxygen summary and discussion

In summary, the probability of low DO concentrations contributing to biological impairments appears to be the highest within the mid-river reaches at Sites 5, 6, and 8. This is especially apparent when looking at the data from the longitudinal DO surveys that were conducted in 2007 and 2009. Low DO concentrations in the Ann River seem to occur during mid-summer months, when streamflow is at or near baseflow and water temperatures are at their annual peak. Almost all violations of the DO standard were recorded during the months of July, August, and September. In the summer of 2007, the Ann River watershed and surrounding area experienced a severe drought, which appeared to drive DO concentrations below 5 mg/L for extended periods of time at select sites. The magnitude and duration of DO standard violations in 2008 were reduced, possibly due to more stable summer flows than were observed in 2007.

Low dissolved oxygen should be further evaluated as a potential stressor in the Ann River watershed. The existing data is sufficient to identify low DO concentrations as a potential problem. However, there are still many unanswered questions concerning the severity of this stressor, and whether or not climactic events (drought) influenced data collection in 2007 and 2008. It can be concluded that DO concentrations do fall below the state water quality standard of 5 mg/L on occasion, and therefore are potentially stressing some of the more intolerant fish and macroinvertebrate species present in the Ann River drainage.

Table 21. Strength of Evidence chart for Candidate Cause #2 - Low Dissolved Oxygen

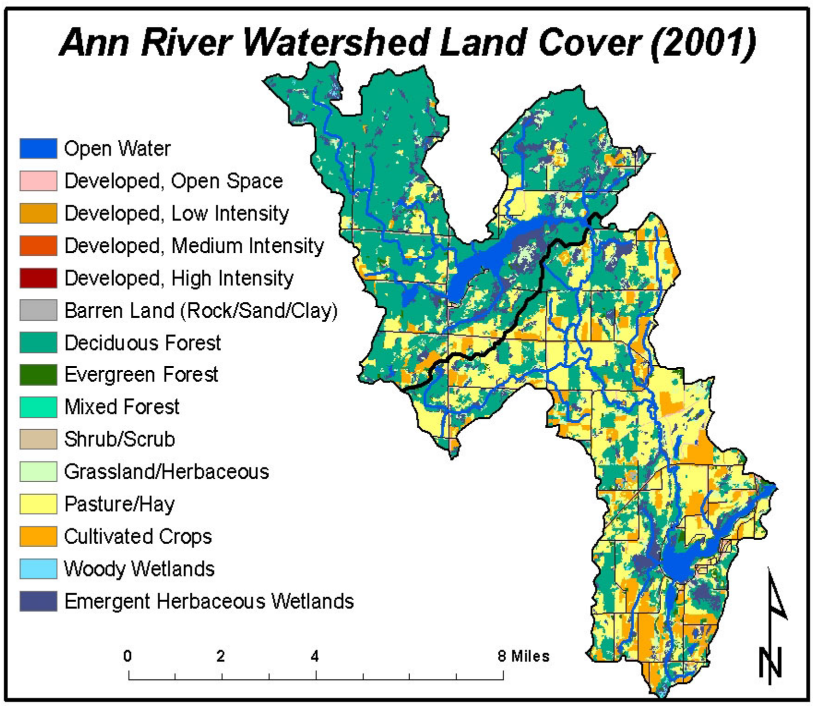
Types of evidence	Strength of evidence score		Comment
Evidence using data from Ann River	Fish	Invert	
Spatial / temporal co-occurrence	+	+	Most Ann River monitoring sites violated the 5 mg/L DO standard at least once during the monitoring period. The magnitude, duration, and frequency of readings below the standard were most significant at Sites 5, 6, and 8 on the Ann River. Monitoring on the Little Ann River, which is not impaired for aquatic life, did not reveal and violations of the DO standard.
Evidence of exposure, biological mechanism	+	+	The fish and macroinvertebrate community of the Ann River lacks sensitive taxa.
Causal Pathway	+	+	Low streamflow (result of impoundments and climactic events), stream channel widening, lack of canopy cover, and nutrient loading are all potential causal pathways that are present in the Ann River watershed.
Field Evidence of stressor-response	0	0	Difficult to interpret because the magnitude, frequency, and duration of low DO concentration changes from one monitoring period to the next. There are inconsistencies in this line of evidence
Field experiments / manipulation of exposure	NE	NE	No data available
Laboratory analysis of site media	NE	NE	No data available
Temporal sequence	NE	NE	No data available
Verified of tested predictions	NE	NE	No data available
Symptoms	0	0	Symptoms of exposure to low dissolved oxygen are ambiguous or occur with many other candidate causes.
Evidence using data from other sources			
Mechanistically plausible cause	+	+	Low DO concentrations can limit diversity and abundance of fish and macroinvertebrates in streams.
Stressor-response in other field studies	+	+	Hardwood Creek (Minnesota) → Jackson River Stressor ID (Virginia) →
Stressor-response in other lab studies	+	NE	(Smale and Rabeni, 1995);
Stressor-response in ecological models	NE	NE	No data available
Manipulation experiments at other field sites	NE	NE	No data available
Analogous stressors	NE	NE	No data available
Multiple Lines of Evidence			
Consistency of Evidence	0	0	Periods of low dissolved oxygen were observed in 2007 and to a lesser extent in 2008. The one diurnal DO survey completed in 2009 did not produce any DO concentrations below 5 mg/L. Low dissolved oxygen appears to be a stressor in the Ann River watershed during mid – to late summer months, when the stream is at baseflow and air temperatures are high. Stagnant flows and elevated phosphorous concentrations may contribute to higher primary production, causing lower DO concentrations as plants respire or decompose.
Explanatory power of evidence	0	0	Additional DO monitoring or modeling would be helpful for determining the explanatory power of this stressor.

2.5 Candidate Cause #3: Loss of Connectivity – Riparian Function

The riparian zone of a stream is generally defined as the transition area between aquatic ecosystems and adjacent upland terrestrial ecosystem (Gregory et al, 1991). High quality, undisturbed riparian corridors provide shading from solar radiation, filtration of overland runoff, mitigation of bank erosion, and inputs of detritus and organic matter that are critical to supporting aquatic life (Cummins and Spengler, 1978; Li and Shen, 1973; Meehan et al, 1977).

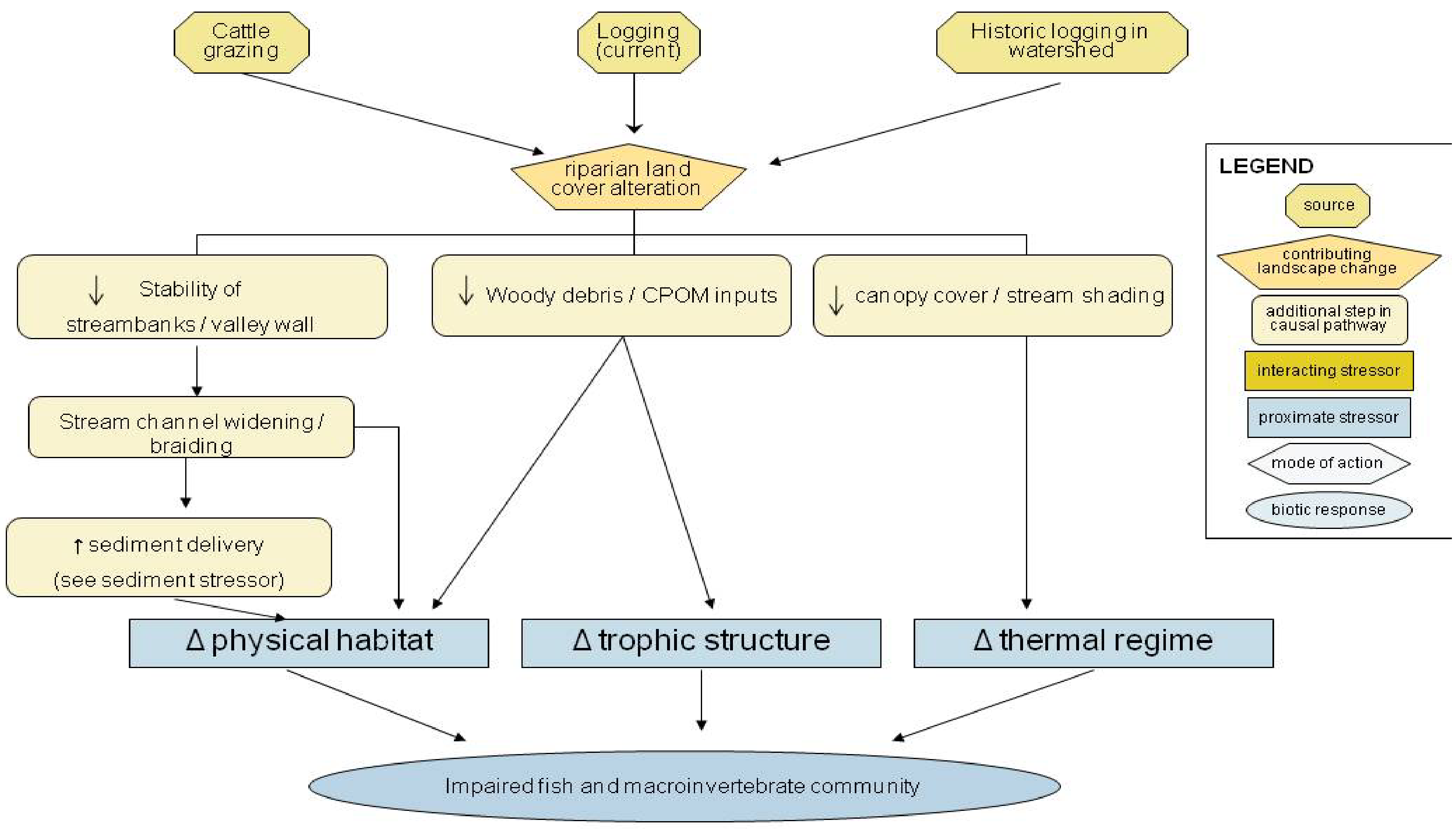
A variety of land-uses and land cover alterations have reduced the quality of the riparian corridor within the Ann River watershed. The pre-settlement vegetation of the area was dominated by mature forest, primarily a mix of coniferous and deciduous species, with some bogs and wooded wetland areas near the two lakes. Post-settlement land cover has been altered by timber harvesting in the late 1800's to early 1900's and more recently, a conversion to agricultural lands, with pasture/hay land being especially prominent along the main stem of the Ann River (Figure 49). The majority of the rangeland and cropland is located in the southern half of the watershed, while the northern half remains a mix of forest, open water, and herbaceous wetland.

Figure 49. Ann River Watershed Land Cover (mn_nlcd 2001)



Forest/Shrub	55%
Rangeland	23%
Wetland	10%
Cropland	6%
Developed	3%
Open Water	3%

Figure 50. Conceptual Model for Riparian Corridor Disturbance

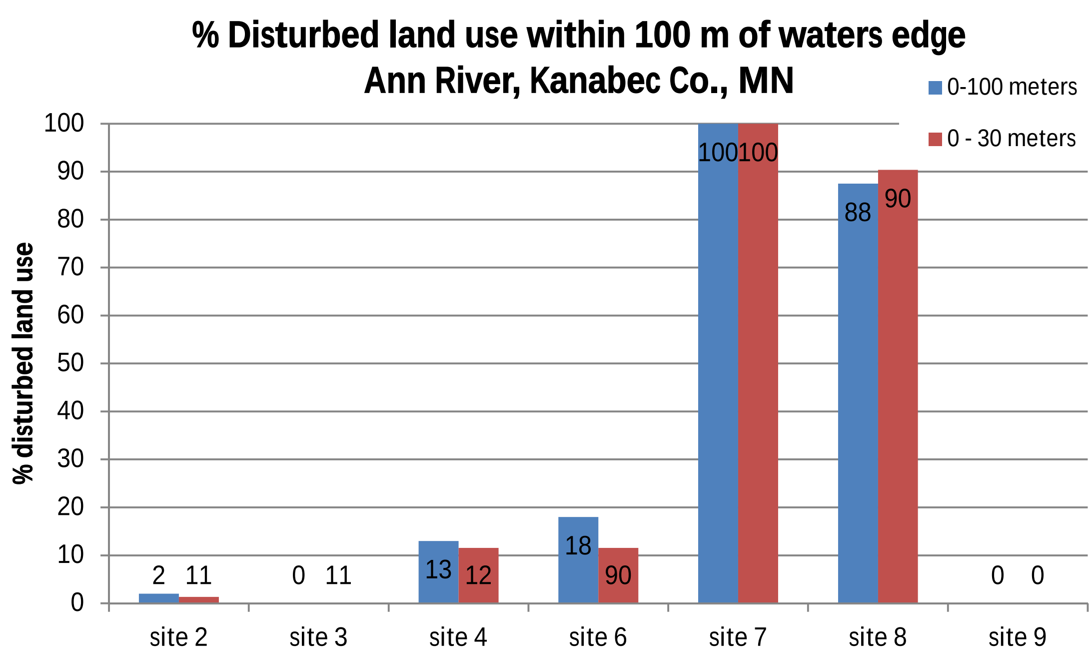


2.5.1 Ann River riparian assessments

Riparian habitat quality was evaluated at Ann River biological monitoring sites using quantitative and qualitative methods developed by MPCA. In all, riparian habitat measurements were collected at seven stations in the Ann River watershed. The qualitative assessment followed Minnesota Stream Habitat Assessment (MSHA) protocols, which evaluate riparian conditions under three main categories – riparian width, bank erosion, and shading. The quantitative assessments of the riparian corridor were completed using the methodologies developed by MPCA's biological monitoring program. These methods include measurements of buffer width, dominant land use, percent "disturbance", overhanging vegetation, and measurements of overhead canopy cover.

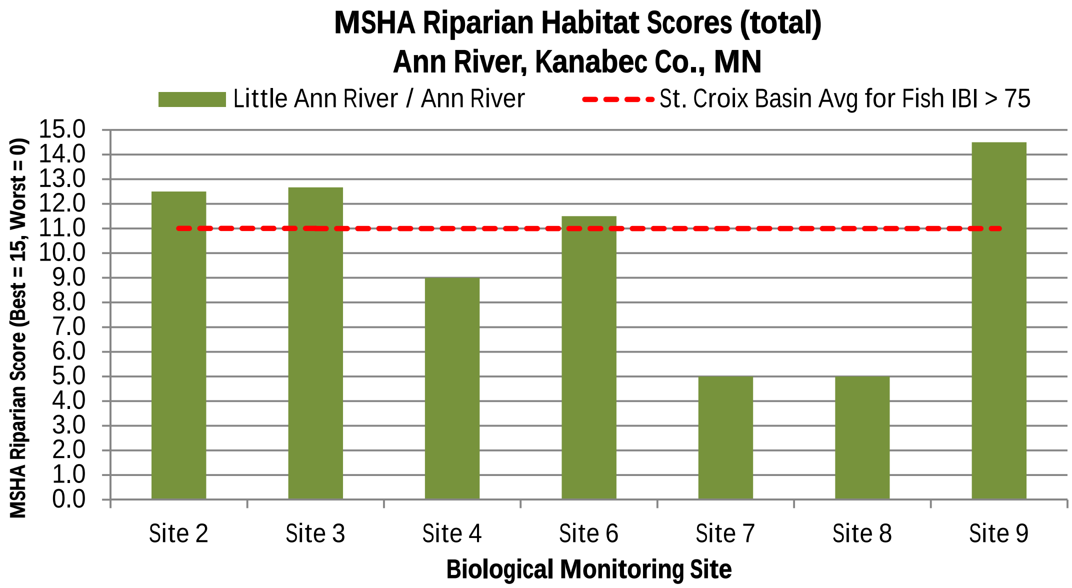
Based on the quantitative and qualitative habitat assessment results, the quality of riparian buffer appears to decline considerably within the middle to lower reaches of the Ann River. At Sites 7 and 8, many beneficial features of the riparian corridor (i.e. shading and erosion control) are likely lost or substantially diminished. The riparian corridor along these two stations has been altered by anthropogenic disturbances to a much greater degree than other areas of the watershed. As shown in Figure 51, almost 100 percent of the riparian area within 100 meters of the river's edge on both banks has been was considered "disturbed" during MPCA's habitat assessments. In the case of these two stations, the riparian disturbances are predominantly connected to cattle grazing operations.

Figure 51. Percent disturbed land-use within 30 and 100 meters of the Ann River



Qualitative habitat scores from MSHA assessments also support poor riparian habitat quality as a potential stressor to aquatic life in the Ann River. Figure 52 shows total riparian habitat scores, which combine individual scores for riparian width, canopy cover, and bank erosion. Again, sites Site 7 and Site 8 score much lower than other sites in the watershed. Riparian disturbance seems to be relatively localized, as other Ann River stations had riparian habitat scores comparable to St. Croix basin streams with healthy fish assemblages (Figure 52).

Figure 52. MSHA scores for Riparian Habitat at Ann River Biological Monitoring Stations



*The red dashed line indicates the basin-wide average for sites that achieve a fish IBI score of at least 75 w/ drainage area between 20 and 76 sq. miles.

2.5.2 Effects of riparian disturbance

2.5.2.1 Channel widening/bank instability

The removal of riparian vegetation can lead to an increase in width-depth ratio, channel braiding, and the loss of undercut banks that can serve as cover for fish (Behnke and Raleigh 1978, Gunderson 1968, Marcuson 1977; Rosgen, 1996). Platts (1990) concluded that riparian vegetation has a major influence on channel shape and contributes to stream bank strength by binding the soil with roots, shielding banks from erosion, and repairing annual damage with sediment deposition.

There is evidence that riparian disturbance has contributed to stream channel widening in the Ann River watershed. Cattle grazing, logging, and herbicide spraying have reduced the rigor and diversity of plant species within the stream corridor. The spraying of herbicide has not been verified in the watershed, although anecdotal evidence suggests that it is (or was) used to control the growth of certain plant species in some locations. These land management tactics have rendered stream banks and riparian terrain more susceptible to sediment loss from fluvial and landscape-driven processes. Based on stream reconnaissance efforts, it does not appear that cattle are fenced off from the stream in any locations where grazing occurs in the riparian corridor. Cattle activity in or near the stream has also been shown to cause widening of the stream channel and higher erosion rates (Behnke & Raleigh, 1978; Ohmart, 1996; Kauffman & Krueger, 1984).

Figure 53. Ann River at Site 7 (left) and Little Ann River at Site 3 (right)

As shown in these photos, the character and quality of the riparian corridor varies widely within the watershed. Note the presence of slumping/mass wasting in the photo of Site 7, highlighted by the yellow square. This is an indicator that the channel is likely widening within this reach.

Rosgen (2006) documented the channel evolution of Wemiuche Creek (Colorado) over a 12-year period after willows were removed by spraying and the land was converted to cattle pasture. Over the course of the study, the pasture became overgrazed and the stream channel of Weminuche Creek underwent a series of adjustments. Ultimately, this transition in land-*use led to higher sediment loading and reduced stable habitat for aquatic and terrestr*ial life (Rosgen, 2006). Table 21 summarizes some of the changes that were observed over the course of the Wemiuche Creek study. Although this case study is from the western United States, there are many similarities between it and observations made in several reaches of the Ann River watershed where cattle grazing is the dominant land-use.

Table 22. Impacts from De-vegetation of Riparian Corridor and Grazing along Weminuche Creek, Colorado (Rosgen, 2006)

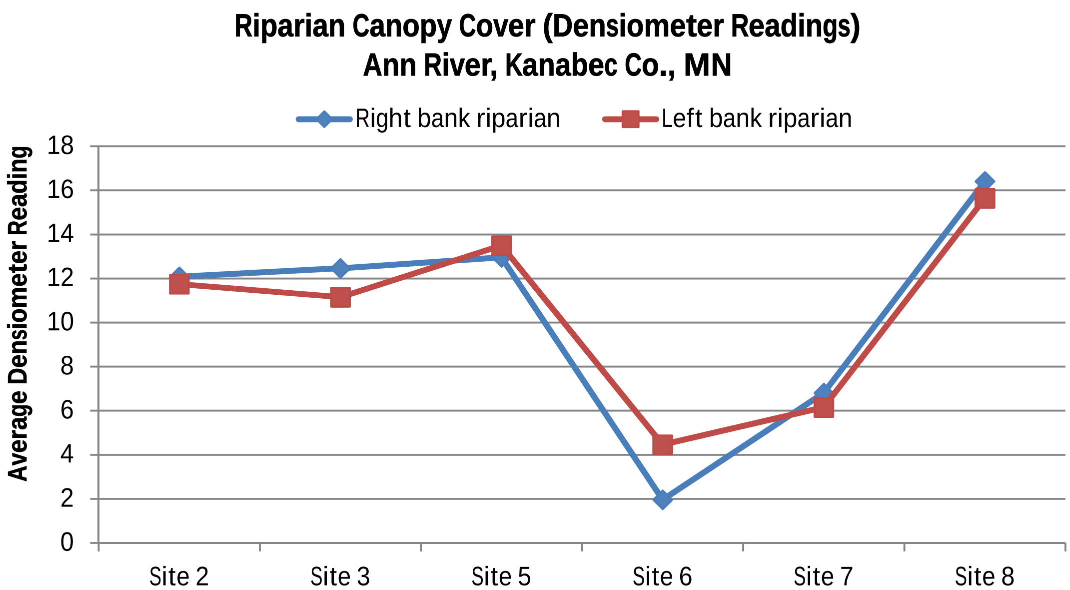
▪ Increases in sediment supply from bank erosion
▪ Decreases in shear stress
▪ Decreases in stream power
▪ Reduced sediment transport capacity
▪ Decreases in sediment competence
▪ Filling in of pools, decreasing fish/invertebrate habitat
▪ Major land loss and decrease in aesthetics

The lack of a long-term data set makes it difficult to evaluate the overall effects of riparian disturbance on the stream channel of the Ann River. Figures 15-18 in this report show stream channels with contrasting riparian conditions in the Ann River Watershed. The aerial photos suggest that stream channels without adequate riparian buffers are widening and beginning to braid. If uncontrolled grazing continues to occur in the riparian corridor of the Ann River, it is likely that many of the processes outlined in Table 21 will unfold within the affected areas of the watershed.

2.5.2.2 Loss of canopy/decrease shading from solar radiation

Altered thermal regimes resulting from the loss of riparian vegetation can result in changes in fish and benthic invertebrate community structure (Newbold et. al 1980; Rabeni et. al, 1997; Sweeny, 1985). The shading provided by riparian vegetation can reduce the rate of longitudinal stream warming and decrease the daily temperature flux, especially in headwater streams (Rutherford et. al., 2004). The Little Ann River and headwater reaches of the Ann River both benefit from a forested riparian corridor, which provides good canopy and plentiful shading. However, there is a significant loss of riparian canopy cover starting at station Site 7 and continuing through Site 8. These sites are located within cattle pasturing areas, which are dominated by forbs and grasses, as opposed to the mature forest found along most of the upper river. The results from densitometer readings taken at biological monitoring sites are shown in Figure 54. The lack of canopy cover within the intensively grazed reaches (Sites 7 and 8) is very evident when compared to non-grazed or forested reaches (Site 6 and 9).

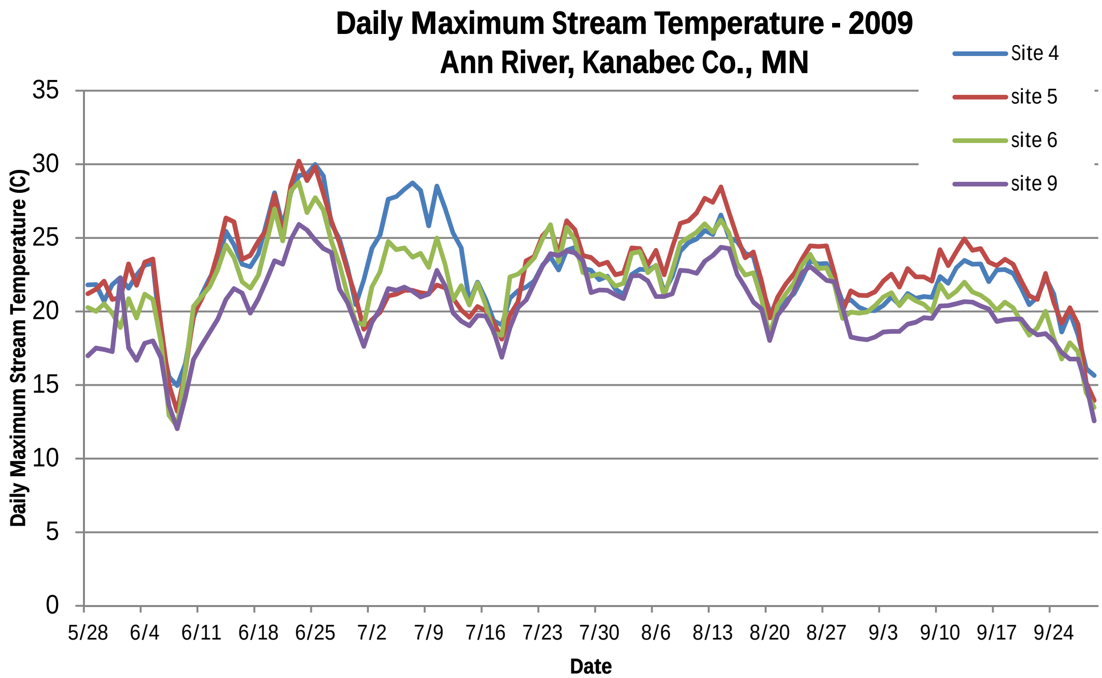
Figure 54. Average Densitometer readings taken from the right and left edge of the stream facing the Riparian Zone



HOBO Pro V2 water temperature loggers were deployed at five locations on the Ann River in 2009. The goal of this monitoring effort was to evaluate stream temperature regime as it relates dissolved oxygen concentrations and the temperature requirements of certain coolwater gamefish (i.e. smallmouth bass). The temperature loggers were deployed at Sites 4, 5, 6, 8, and 9 from early June to late September. Stream temperature was recorded in degrees Celsius at 15-minute intervals. At the end of the monitoring period, the temperature logger at Site 6 was unable to be recovered, so no data is available for that monitoring station.

Relationships between riparian canopy cover and stream temperature in the Ann River watershed are not well defined, although there does appear to be some connection between riparian shading and summer stream temperature. Continuous temperature monitoring data from June-September 2009, indicate that Sites 4, 5, and 8 have similar mid-summer stream temperatures, while Site 9 is consistently several degrees cooler (Figure 55). The different temperature regimes may be attributed to shading provided by the mature deciduous forest in the lower reaches of the river near Site 9.

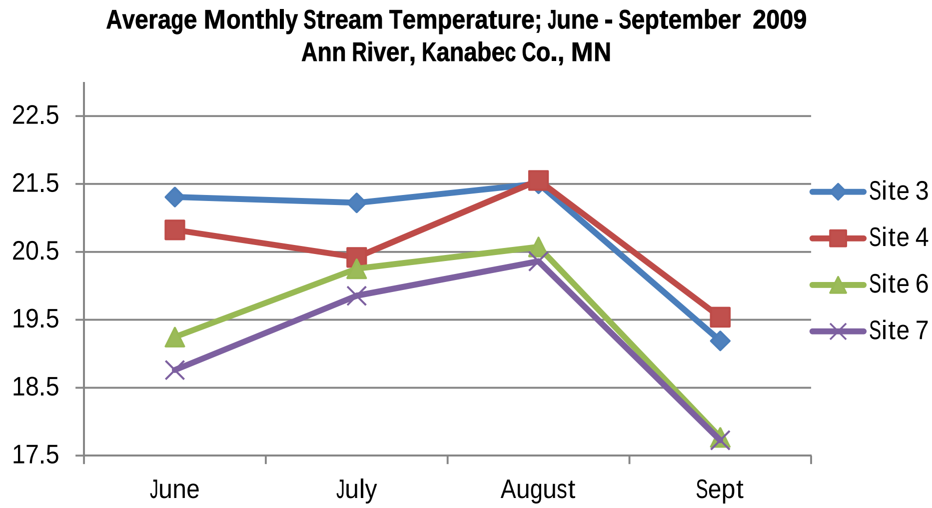
Figure 55. Daily Maximum stream temperature recorded at 4 Ann River Monitoring Sites June – Sept 2009.



Stream temperature tolerances and suitability ranges are less understood for warmwater and coolwater aquatic organisms, as most of the research in this area focuses on coldwater fishes (salmonids). Nonetheless, an increase in maximum stream temperature or daily temperature fluctuations may limit the abundance and growth of some of the coolwater fish species commonly found in the greater St. Croix River watershed. Examples of these fish species include burbot, smallmouth bass, and pearl dace. Growth rates and interspecies competitive advantages in smallmouth bass have been shown to decrease as water temperatures that exceed 22˚ C (Zweifel et al, 1999). This temperature was exceeded regularly at all Ann

The stream temperature data from 2009 clearly shows that the upper and lower Ann River have slightly different temperature regimes, at least for a large portion of the open water season (Figure 56). June and September water temperatures are several degrees cooler at sites located in the lower reaches of the river

Figure 56. Average monthly stream temperature for the Ann River in 2009



2.5.2.3 Loss of woody debris/organic matter inputs

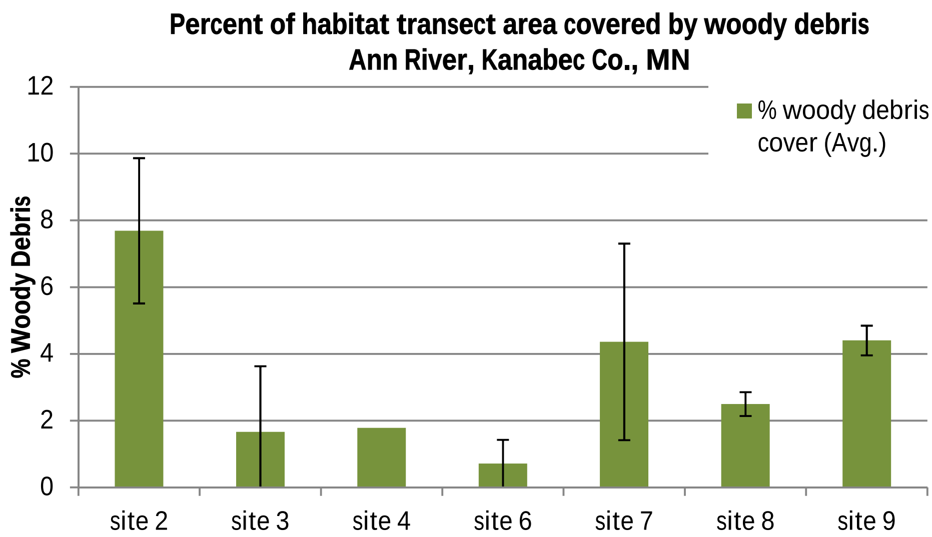
Healthy riparian corridors offer additional benefits to stream ecosystems through contributions of large woody cover and smaller organic material that plays an integral role in food webs and nutrient cycling. Many macroinvertebrate species feed on decaying leaves and other types of organic matter present in the stream, either by filtering the material from the water column or processing detritus and algae from the stream substrate. Riparian vegetation has been found to provide up to 90 percent of organic matter inputs necessary to support headwater stream communities (Cummins and Spengler 1978).

Large woody debris (LWD), such as logs, stumps, and root wads provide cover for fish and can scour additional pool habitat by deflecting current vectors (Abbe and Montgomery, 1996; Rosgen, 1996). LWD has been shown to be a critical habitat component for adult fish in stream environments (Angermeier and Karr, 1984) and a preferred habitat for smallmouth bass (Lobb and Orth, 1991), which is one of the gamefish species present in small numbers within the Ann River watershed. In addition to providing habitat and refuge for aquatic life, the presence of LWD in the stream channel can decrease water velocities and stream power, which alleviates erosive forces acting upon the streambanks and streambed (Macdonald and Keller, 1987).

The LWD in streams is nearly all derived from vegetation within 30 m of the streambanks (Murphy and Koski, 1989). Therefore, management practices that remove woody and herbaceous vegetation from the riparian corridor are likely to reduce LWD abundance in the stream. The amount of LWD ending up in the Ann River is likely lower than historical values as a result of this change in riparian land cover.

Measurements of woody debris abundance and cover were conducted at biological monitoring stations in the watershed using MPCA habitat protocols. The quantitative habitat data from the Ann River does not suggest that riparian vegetation type and in-stream woody debris abundance are positively correlated. Site 9, which is situated in a mature deciduous forest, consistently had the greatest abundance of woody debris in the stream channel (Figure 57). However, Sites 7 and 8 had higher percentages of woody debris in the stream than sites further upstream that are more heavily wooded. Although these results appear somewhat counterintuitive, it is important to understand that woody debris is easily transported downstream under certain streamflow conditions. Physical channel features such as stream gradient, cross-sectional area, and roughness all affect the abundance, location, and orientation of woody debris in stream ecosystems (Abbe and Montgomery, 1996; Cordova et al., 2006). It is possible that forested, high-gradient reaches (such as Site 6) contribute significant amounts of woody debris to the stream that is transported out of the reach and deposited in lower gradient reaches (Site 8 and 9) during higher flow events.

Figure 57. Woody debris abundance at Ann River Biological Monitoring Sites

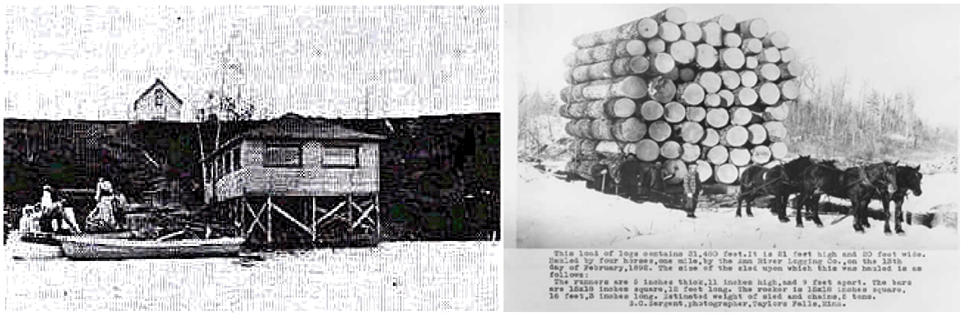


2.5.3 Filtration, infiltration, and fluvial erosion

Vegetated buffer strips located in the riparian corridor have the ability to reduce sediment and nutrient loading to streams (Wilken and Hebel, 1992; Niebling and Alberts, 1979). In addition to trapping sediment and nutrients, effective buffer strips can also remove or reduce pesticides, heavy metals, and other harmful agents from runoff before it reaches a water body. There is a great deal of debate on the relationship between buffer width and overall effectiveness, but wider buffer strips have shown increased ability to filter sediment (Davies and Nelson, 1994).

The conversion of mature forest to grazing rangeland in the Ann River watershed has likely reduced the ability of the landscape to filter runoff and promote infiltration of precipitation. The intensive logging that took place in the watershed around near the turn of the 20th century left many areas devoid of vegetation and very susceptible to the formation of gullies and overland runoff (Figure 58). Many of the areas that were logged off were then converted to pasture for livestock operations. Restoring riparian buffers to woodland or well-managed grassland would likely reduce sediment loading to the Ann River, and mitigate sediment related stressors to aquatic life. Additionally, more forest cover in the watershed would promote infiltration of precipitation and may lead to more stable streamflow during summer months.

Figure 58. Examples of Logging in the Ann River and Snake River Watershed



Logging around the turn of the 20th century significantly reduced the amount of mature forest within the Ann River watershed. The logging efforts opened up the land for cattle pasturing, which is now a dominate land-use in the area.

The lack of re-forestation in some areas of the Ann River watershed has also left much of the terrain more susceptible to erosion from weathering and fluvial processes. The Ann River valley wall is rather steep in some locations, and areas devoid of soil binding vegetation (i.e. tall grasses or trees) show signs of gully and/or landslide formation (Figure 59).

Figure 59 Examples of erosion caused by Weathering, Landslides, and other Fluvial processes in the Ann River Riparian Corridor

The lack of vegetation in certain areas has made the landscape more vulnerable to these forms of erosion.

2.5.4 Summary: loss of riparian function

Historic and current land-uses have degraded the quality of the Ann River riparian corridor, thus reducing the many ecological services these areas provide for aquatic and terrestrial flora and fauna. Apparent effects of the riparian disturbances include stream channel instability, a reduction in the inputs of woody debris and other organic matter, and possible thermal loading as a result of reductions in canopy cover. Although it is difficult to connect riparian disturbance as a whole to a specific biological metric, many of the stressors mentioned in this report (sediment, dissolved oxygen) can be connected to riparian alteration as a source.

Restoration of the Ann River riparian corridor should be a high priority in the overall effort to improve fish and macroinvertebrate IBI scores in the watershed. A restoration plan focused on re-establishing mature forest, plant diversity, and limiting cattle access to the river would address many potential stressors.

Table 23. Strength of Evidence (SOE) scores for riparian corridor disturbance

Types of Evidence	Strength of Evidence Score		Comment
Evidence using data from Ann River	Fish	Invert	
Spatial / temporal co-occurrence	0	+	Spatial co-occurrence is difficult to evaluate for this stressor because the effects of riparian disturbance in one location can impact sites locally or miles downstream. Site 7 is impaired for macroinvertebrates and is located within a highly disturbed riparian corridor. The fish impairment is most severe near the mouth of the river, which has a relatively undisturbed riparian corridor. Sediment delivery from mid-river riparian sources may be impacting fish populations downstream.
Evidence of exposure, biological mechanism	+	+	Invertebrate IBI score is lower within stream reaches that have substantial riparian disturbance. Habitat for both fish and invertebrates has been impacted by the removal of riparian vegetation.
Causal Pathway	+	+	There is a logical causal pathway for this stressor. See conceptual model in Figure 50
Field Evidence of stressor-response	0	+	The lack of a consistent invertebrate data set for the Ann River makes it difficult to observe stressor-response trends. However, invertebrate IBI scores are consistently lower at Site 7 which appears to have the most degraded riparian corridor. Sites with higher quality riparian habitats have shown higher invertebrate IBI scores (Site 6, Site 9, Kettle River reference site -- 96SC085)
Field experiments / manipulation of exposure	NE	NE	No evidence available
Laboratory analysis of site media	NE	NE	No evidence available
Temporal sequence	NE	NE	No evidence available
Verified of tested predictions	NE	NE	No evidence available
Symptoms	0	+	The available biological data does not provide any clear symptoms related to riparian habitat degradation.
Evidence using data from other sources			
Mechanistically plausible cause	+	+	A plausible mechanism for this stressor exists
Stressor-response in other field studies	+	+	Degradation of riparian habitat has been regularly linked to impaired biotic communities
Stressor-response in other lab studies	NE	NE	No data available
Stressor-response in ecological models	+	+	Frimpong et al., 2005
Manipulation experiments at other field sites	+	+	Moerke et al., 2004;
Analogous stressors	+	+	Degradation of the riparian corridor has led to the destruction of aquatic habitat and reduced biological integrity in many regions of the world (Platts, 1991; Kauffman and Krueger, 1984). This is a common stressor in Minnesota in areas where cattle pastures are a prominent feature of the landscape.

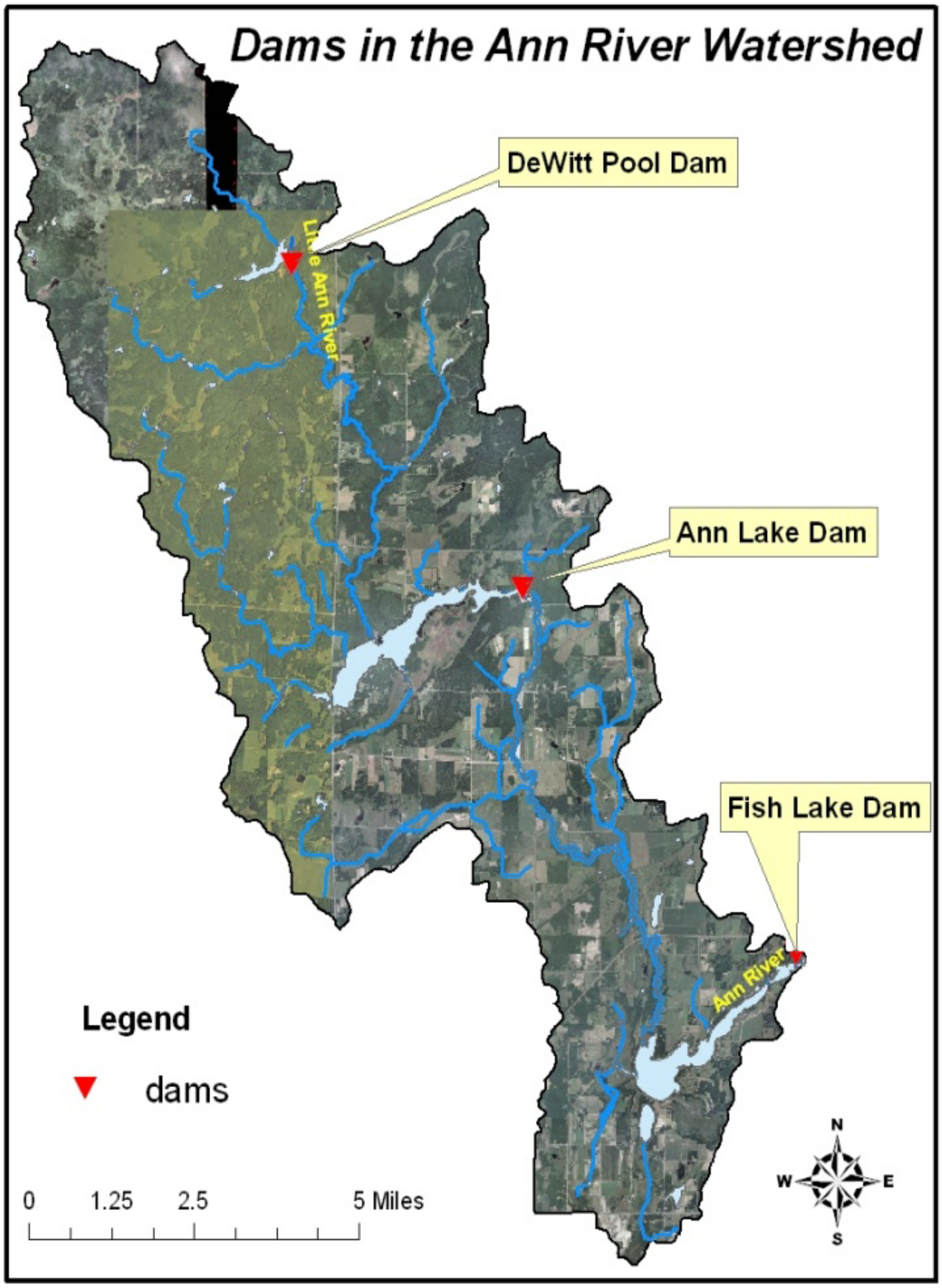
Types of Evidence	Strength of Evidence Score		Comments
Multiple Lines of Evidence			
Consistency of Evidence	0	+	There is a relatively consistent linkage between riparian disturbance and macroinvertebrate impairment within the watershed. However, these results are somewhat difficult to interpret given the lack of quality data.
Explanatory power of evidence	0	++	There are several evidence types that point to riparian degradation as a stressor source (sedimentation) and as a direct stressor (lack of shading and woody debris inputs).

2.6 Candidate Causes #4 and #5: Flow Alteration/Loss of Connectivity due to Impoundments

The presence of impoundment structures on river systems are known to alter steamflow, water temperature regime, and sediment transport processes-each of which can drastically alter fish and macroinvertebrate assemblages (Cummins, 1979; Waters 1995. Dams also regularly limit or impede fish migrations and can greatly reduce or even extirpate local populations (Brooker, 1981; Tiemann et al., 2004). In the state of Minnesota alone, there are over 800 dams on streams and rivers for a variety of purposes, including flood control, wildlife habitat, and hydroelectric power generation.

There are three constructed dams in the Ann River watershed. Both the headwaters of the Little Ann River and Ann River originate from, and are somewhat controlled by impoundment structures. The locations of the dams in the Ann River watershed are shown in Figure 60. Several of these areas have been impounded since the 1800's for the purpose of transporting logs down to the Snake and St. Croix River. The temporary impoundments used for transporting logs were replaced by permanent structures over the last several decades.

Figure 60.. Location of dams in the Ann River watershed



2.6.1 Ann River Watershed Impoundments

2.6.1.1 Dewitt Pool Dam

DeWitt Pool is a shallow reservoir created by a stop-log dam at the headwaters of the Little Ann River in Mille Lacs County. The reservoir is managed for waterfowl and wild rice habitat and water levels are not actively managed. The impoundment structure was reconstructed in 1993. Under the current design, the impoundment has a hydraulic height of about 7.5 feet.

2.6.1.2 Ann Lake Dam

The outlet of Ann Lake has been impounded since the mid to late 1880's. Logging dams were constructed and maintained during the intensive logging that occurred over this time period in order to effectively transport logs down the Ann River. An inspection performed in 1940 indicated that the logging dam at Ann Lake no longer existed, however the earthen dikes remained in place. In the early 1960's, Minnesota DNR obtained land surrounding the outlet of Ann Lake and secured flowage easements in the interest of reconstructing the dam to create waterfowl habitat and hunting opportunities.

In 1965, a water control structure was installed in the Ann River (Figure 61). The structure was designed as a spillway, about 150 feet in length and 12 inches high. This dam created an open-water reservoir and associated adjacent wetlands, enlarging the Ann Lake basin from about 350 acres to 1100 acres and caused a rise in lake elevation of about 4 feet. A significant renovation of the dam took place in the early 1990's to re-stabilize the spillway and replace the stop-logs used to control the outflow.

Figure 61. Ann Lake dam during the summer (left photo) and early fall (right photo)

2.6.1.3 Fish Lake Dam

The Fish Lake dam is a 40 foot long impoundment of the Ann River located near its confluence of the Snake River. The impoundment has a hydraulic height of about 10 feet which creates a 407 acre reservoir known as Fish Lake. During periods of normal to low streamflow, the dam structure appears to limit or completely eliminate connectivity between the Snake River and Ann River (Figure 62).

Figure 62. The Fish Lake dam impounds the Ann River near its confluence with the Snake River (seen in background of photo on right)

2.6.2 Effects on fish passage

The presence of dams in the Ann River watershed likely inhibits migrations of several fish species during low flow periods. Although each of these dams can be passable at high flows after spring snowmelt, for the good portion of the year they are impassable to most fish, especially smaller non-game migratory species. The lack of year round connectivity between the Ann River, Snake River, and the lakes could limit fish diversity, abundance, and tolerance of short-term disturbances in the watershed. Without baseline data or the resources to do tag and recapture studies, it is difficult to determine the true impact that these impoundments are having on fish dispersion and diversity.

Despite the presence of the dams, many migratory fish species are still found in the Ann River drainage. Lake sturgeon (*Acipenser fulvescens*) and numerous redhorse species are among the migratory fish species that are could be impacted or threatened by dams on the Ann River. Shorthead redhorse (*Moxostoma macrolepidotum*), golden redhorse (Moxostoma erythrurum), greater redhorse (Moxostoma valenciennesi), silver redhorse (*Moxostoma anisurum*), and river redhorse (Moxostoma carinatum) have all been recently sampled in the Ann River, and are also present in Fish Lake. Given the presence of these five redhorse species, it does not appear that these impoundments are having a significant negative effect on redhorse diversity and abundance. Yet, it is difficult to make this determination given the lack of fisheries data available from years prior to the construction of the dams.

Lake sturgeon have historically been observed in the Ann River drainage, but did not show up in the most recent surveys of the river or reservoirs. These fish were once abundant in the Snake River and were "taken by wagonloads," with fish weighing in at over one hundred pounds (Waters, 1995). Due to overharvest, impoundments, and other land-use activities, lake sturgeon are now rare in the Snake River and its tributaries. The disappearance of lake sturgeon from the Ann River is likely the result of various stressors that stretch well beyond the outer boundaries of its immediate watershed.

Table 24. Summary of evidence for candidate stressors related to impoundment structures in the watershed

Types of Evidence	Strength of Evidence Score		Comment
Evidence using data from Ann River	Fish	Invert	
Spatial / temporal co-occurrence	0	+	Spatial co-occurrence is difficult to evaluate for this stressor because the effects of riparian disturbance in one location can impact sites locally or miles downstream. Site 7 is impaired for macroinvertebrates and is located within a highly disturbed riparian corridor. The fish impairment is most severe near the mouth of the river, which has a relatively undisturbed riparian corridor. Sediment delivery from mid-river riparian sources may be impacting fish populations downstream.
Evidence of exposure, biological mechanism	+	+	Invertebrate IBI score is lower within stream reaches that have substantial riparian disturbance. Habitat for both fish and invertebrates has been impacted by the removal of riparian vegetation.
Causal Pathway	+	+	There is a logical causal pathway for this stressor. See conceptual model in Figure 50
Field Evidence of stressor-response	0	+	The lack of a consistent invertebrate data set for the Ann River makes it difficult to observe stressor-response trends. However, invertebrate IBI scores are consistently lower at Site 7 which appears to have the most degraded riparian corridor. Sites with higher quality riparian habitats have shown higher invertebrate IBI scores (Site 6, Site 9, Kettle River reference site -- 96SC085)
Field experiments / manipulation of exposure	NE	NE	No evidence available
Laboratory analysis of site media	NE	NE	No evidence available
Temporal sequence	NE	NE	No evidence available
Verified of tested predictions	NE	NE	No evidence available
Symptoms	0	+	The available biological data does not provide any clear symptoms related to riparian habitat degradation.
Evidence using data from other sources			
Mechanistically plausible cause	+	+	A plausible mechanism for this stressor exists
Stressor-response in other field studies	+	+	Degradation of riparian habitat has been regularly linked to impaired biotic communities
Stressor-response in other lab studies	NE	NE	No data available
Stressor-response in ecological models	+	+	Frimpong et al., 2005
Manipulation experiments at other field sites	+	+	Moerke et al., 2004;
Analogous stressors	+	+	Degradation of the riparian corridor has led to the destruction of aquatic habitat and reduced biological integrity in many regions of the world (Platts, 1991; Kauffman and Krueger, 1984). This is a common stressor in Minnesota in areas where cattle pastures are a prominent feature of the landscape.
Multiple Lines of Evidence			
Consistency of Evidence	0	+	There is a relatively consistent linkage between riparian disturbance and macroinvertebrate impairment within the watershed. However, these results are somewhat difficult to interpret given the lack of quality data.
Explanatory power of evidence	0	++	There are several evidence types that point to riparian degradation as a stressor source (sedimentation) and as a direct stressor (lack of shading and woody debris inputs).

3.0 Identification of Probable Causes

Identifying the most probable cause for biological impairment is the final step in the Stressor ID process. Many candidate causes were considered for the Ann River case study, but ultimately, five potential causes were identified as the most likely and were retained for further analysis. These were: *(1) lack of quality benthic habitat due to sedimentation (2)Low Dissolved Oxygen (DO) concentrations (3) Various stressors related to riparian corridor degradation (4) Loss of connectivity due to the presence of dams (5) reduced baseflow due to the presence of dams*. In this final step, the evidence compiled for these five candidate causes will be compared in attempt to rank the stressors in terms of their level of contribution to the impaired condition. Table 23 below is a compilation of the weight of evidence scores for the five candidate stressors.

Table 25. Summary of strength of evidence (SOE) scores for all five candidate stressors

Types of Evidence	Sediment	Low DO	Riparian Degradation	Connectivity	Flow Alteration
Evidence using data from Ann River					
Spatial / temporal co-occurrence	+	+	+	0	NE
Evidence of exposure, biological mechanism	+	+	+	0	NE
Causal Pathway	++	+	++	+	0
Field Evidence of stressor-response	0	0	+		NE
Field experiments / manipulation of exposure	NE	NE	NE	NE	NE
Laboratory analysis of site media	NE	NE	NE	NE	NE
Temporal sequence	0	NE	NE	0	NE
Verified of tested predictions	NE	NE	NE	NE	NE
Symptoms	+	0	0	NE	NE
Evidence using data from other sources					
Mechanistically plausible cause	+	+	+	+	+
Stressor-response in other field studies	+	+	+	+	+
Stressor-response in other lab studies	+	+	NE	NE	NE
Stressor-response in ecological models	+	NE	+	+	+
Manipulation experiments at other field sites	NE	NE	+	+	+
Analogous stressors	++	NE	++	+	+
Multiple Lines of Evidence					
Consistency of Evidence	+	0	+	0	NE
Explanatory power of evidence	++	0	++	0	NE

3.1 Discussion of Evidence for Candidate Stressors

3.1.1 Lack of benthic habitat due to sedimentation

Stream reconnaissance, habitat surveys, and geomorphic analysis all produced solid evidence of sediment stress in the Ann River watershed. There is a definite longitudinal trend of increasing sedimentation of the streambed from upstream to downstream. Stream gradient certainly plays a role in the distribution of fine sediment in the watershed; as the lower gradient river reaches lower in the watershed appear to be acting as depositional areas.

The causal pathway for sediment stress is supported by many forms of evidence. Major sediment sources in the watershed are primarily within the immediate stream corridor. Cattle grazing activity has resulted in a reduction of vegetative cover and plant/diversity near the stream which has increased runoff potential. Cattle activity within and near the immediate stream channel also appears to be causing increases in channel width/depth ratio, bank failure, and channel braiding. Fluvial processes (wind, rain, snowmelt) are creating gullies and washout areas where valley walls are steep and lack vegetation.

The sediment load being carried by the Ann River appears to be dominated by coarse grained materials (sand, small gravel, medium gravel). This step in the causal pathway is supported by the presence of numerous gravel/sand bars within the stream channel and particle size analysis from the streambed and banks. It is our hypothesis that bedload is the main mode of sediment transport in the Ann River as turbidity and TSS concentrations remain low during high flow events.

Although Ann River fish and macroinvertebrate assemblages are not seriously degraded at this point in time, there is some evidence that sedimentation of the streambed is having an effect on species that rely heavily on undisturbed benthic habitats. The diversity of darter fish species is relatively low in the Ann River, which could be indicative of degraded benthic habitat conditions. The abundance of fish that are simple lithophilic spawners also appears to decrease as the substrate shifts from coarse particles to sand and silt. The Ann River appears to be a system in disequilibrium in terms of its sediment budget, and the condition of fish and macroinvertebrate assemblages may continue to decline if sediment inputs continue at their current rate.

3.1.2 Riparian Corridor Degradation

Degradation of the Ann River riparian corridor could very well be considered a “source” of several stressors instead of a stressor itself. In terms of the sediment problems in the watershed, riparian disturbance should be considered a “source.” However, there are potentially a number of direct effects related to riparian disturbance that warrant listing it as an independent stressor. There is strong evidence that a shift in riparian vegetation from a mix of mature coniferous and deciduous tree species to pasture has reduced stream canopy cover and overhanging vegetation along several reaches. The removal of mature trees from the riparian corridor has most likely reduced the input of coarse particular organic matter (CPOM) and large woody debris (LWD), both of which are critical components of stream habitat (Cummins and Spengler 1978; Abbe and Montgomery, 1996; Rosgen, 1996). The loss of overhead canopy has also reduced shading of the stream, and may be resulting in higher water temperatures and lower dissolved oxygen concentrations during critical summer low-flow periods.

3.1.3 Low Dissolved Oxygen

The case for low dissolved oxygen as a primary stressor is weakened by somewhat inconsistent evidence, as DO concentrations varied highly from one monitoring year to the next. There is some evidence of potential low DO conditions in the watershed based on snynoptic and continuous monitoring data, but the presence of drought conditions makes it difficult to determine the specific driver(s) of low DO. Based on available data, the most probable explanation is that low DO conditions do occur in the Ann River, but only when streamflow is critically low and air temperatures are near their annual maximum. Nutrient enrichment from Ann Lake and various sources along the river (i.e. sediment, manure) may be accelerating primary production and leading to DO sags during early morning hours.

3.1.4 Flow Alteation/Loss of Connectivity

The dams located at Ann and Fish Lake may reduce the hydrological and ecological connectivity of the watershed, but the extent of the effects is difficult to evaluate based on the available data. The river supports several fish species that are known to be negatively affected by larger dams (including five species of redhorse), although other species known to be negatively affected by impoundments (e.g. smallmouth bass, gilt darter) were low in numbers or non-existent in recent surveys. The relatively short height of the dams on the Ann River make them passable during higher flows, which likely correspond with spring spawning migrations of several fish species. Several questions remain on the topic of how these dams affect fish passage at normal to low flows, as both appear to be impassable for most of the summer.

The lack of connectivity between the reservoirs and the Ann River may limit spawning habitat for several species that occupy both lotic and lentic habitats. Also, the lack of connectivity between the larger Snake River and the lakes limits available refugia to avoid unfavorable habitat conditions during drought or other temporary disturbances. Lake sturgeon, which have been historically sampled in Fish and Ann Lake, is one fish species that would certainly benefit from greater connectivity within the Ann River watershed and neighboring Snake River. Lake Sturgeon were not documented in the most recent fish surveys completed on Ann Lake (2005) and Fish Lake (2007).

3.1.5 Flow alteration

The lack of a long-term flow record for the Ann River makes it difficult to investigate flow alteration as a stressor. Observations of streamflow during the study period from 2006-2009 suggest that summer baseflow can be limited, especially during years of below average precipitation. Although many streams in the region have limited baseflow during the summer months, the low flow conditions in the Ann River watershed may be exacerbated by the presence of impoundments that regulate water levels at the headwaters.

3.2 Recommendations for Next Steps

The objective of this Stressor Identification study was to inform the Total Maximum Daily Load (TMDL) process by identifying the parameters that will require a load or wasteload allocation. The iterative nature of the Stressor ID process allows some flexibility with these recommendations, as data gaps often arise during the course of the analysis, presenting the need to collect additional data. In addition, stakeholder input is a valuable part of the process and should be considered before any stressors are fully diagnosed.

Based on the evidence presented in this report, it is recommended that TMDL efforts focus on developing target sediment loads for the Ann River that will improve benthic habitat. Target sediment loads should also consider the receiving water of Fish Lake, which is currently impaired for excess nutrients. There is some evidence that sediment is accumulating in the lake near the mouth of the Ann River. The TMDL should attempt to further define the causal linkages between riparian habitat quality and in-stream sediment dynamics. A target setting approach that focuses on the riparian corridor will simultaneously address many of the secondary stressors related to riparian quality (CPOM, woody debris).

Low DO and flow alteration are difficult to eliminate as stressors based on the information available. Without a long-term flow record for the Ann River, it becomes difficult to differentiate between short-term climactic events and an altered flow regime caused by impoundments in the watershed. The extended periods of low DO concentrations appear to coincide with low streamflows, especially during the drought conditions that occurred in 2007. Further evaluation of altered hydrology as a stressor is recommended.

Table 26. List of probable stressors contributing to the biological impairment in the Ann River.

Stressor	Priority*	Comment
Sedimentation	High	TMDL should focus on reducing sediment input from riparian corridor (cattle pastures) and immediate stream channel (stream banks).
Riparian Disturbance	High	TMDL should aim to re-establish quality riparian corridor to increase woody debris, CPOM inputs, and stream shading.
Flow Alteration	Unknown	The impact of impoundments on the flow regime is difficult to determine given the lack of flow data before the impoundments were installed.
Low DO	Medium	Additional data collection summer 2010 to verify low DO conditions. DO should be treated as a secondary stressor.
Connectivity	Medium	Dams are likely fish barriers. Continue to monitor for potential impacts and pursue removal/reconstruction to improve connectivity.

*Also listed is the priority level for inclusion in the TMDL phase of the project and related comments.

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