Minnesota Pollution Control Agency

Groundhouse River Total Maximum Daily Loads for Fecal Coliform and Biota (Sediment) Impairments

Final Report

March 2009

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Groundhouse River TMDL Project for Fecal Coliform and Biota (Sediment) Impairments

Final TMDL Report

March 2009

Submitted to:
Minnesota Pollution Control Agency

Submitted by:
Tetra Tech

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TMDL Summary Table

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<tr><th>EPA/MPCA Required Elements</th><th colspan="3">Summary</th><th>TMDL Page #</th></tr>
<tr><td>Location</td><td colspan="3">Drainage Basin, Part of State, County, etc.</td><td>1-3</td></tr>
<tr><td>303(d) Listing Information</td><td colspan="3">Describe the waterbody as it is identified on the State/Tribe's 303(d) list:
• Waterbody name, description and ID# for each river segment, lake or wetland
• Impaired Beneficial Use(s) - List use(s) with source citation(s)
• Impairment/TMDL Pollutant(s) of Concern (e.g., nutrients: phosphorus; biota: sediment)
• Priority ranking of the waterbody (i.e. schedule)
• Original listing year</td><td>1</td></tr>
<tr><td>Applicable Water Quality Standards/ Numeric Targets</td><td colspan="3">List all applicable WQS/Targets with source citations. If the TMDL is based on a target other than a numeric water quality criterion, a description of the process used to derive the target must be included in the submittal.</td><td>4-20</td></tr>
<tr><td>Loading Capacity (expressed as daily load)</td><td colspan="3">Identify the waterbody's loading capacity for the applicable pollutant. Identify the critical condition.
For each pollutant: LC = X/day; and Critical Condition Summary</td><td>52-63
54; 61-63</td></tr>
<tr><td rowspan="6">Wasteload Allocation</td><td colspan="3">Portion of the loading capacity allocated to existing and future point sources [40 CFR §130.2(h)].
Total WLA = X/day, for each pollutant</td><td></td></tr>
<tr><td>Source</td><td>Permit #</td><td>Individual WLA</td><td></td></tr>
<tr><td>Permitted Stormwater (i.e. MS4, constr.)</td><td>Various</td><td>Various</td><td>54-55</td></tr>
<tr><td>Straight Pipe Septic</td><td>NA</td><td>0</td><td>54-55, 61-63</td></tr>
<tr><td>WWTP (Ogilvie)</td><td>MN0021997</td><td>Sediment - 0.04 US tons/d

Fecal Coliform – 1,741 Million Org/day</td><td>54

61-62</td></tr>
<tr><td>Reserve Capacity? (and related discussion in report)</td><td>NA</td><td>Sediment – 0
Fecal Coliform – 0</td><td>54
61</td></tr>
</table>

Load Allocation	Identify the portion of the loading capacity allocated to existing and future nonpoint sources and to natural background if possible [40 CFR §130.2(g)]. ***Total LA = X/day, for each pollutant***		
	Source	**LA**	
	NP Source Groundhouse - Sediment	11.45 US ton/day	54
	NP Source South Branch Groundhouse - Sediment	11.05 US ton/day	55
	NP Source Groundhouse – Fecal Coliform	Various	60-63
	Natural Background?	Included in the LA	54-55 60-63
Margin of Safety	Include a MOS to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality [CWA §303(d)(1)(C), 40 CFR §130.7(c)(1)]. ***Identify <u>and explain</u> the implicit or explicit MOS for each pollutant***		53, 60
Seasonal Variation	Statute and regulations require that a TMDL be established with consideration of seasonal variation. The method chosen for including seasonal variation in the TMDL should be described [CWA §303(d)(1)(C), 40 CFR §130.7(c)(1)] ***Seasonal Variation Summary for each pollutant***		53, 60
Reasonable Assurance	***Summarize Reasonable Assurance*** *Note: In a water impaired by both point and nonpoint sources, where a point source is given a less stringent WLA based on an assumption that NPS load reductions will occur, reasonable assurance that the NPS reductions will happen must be explained.* *In a water impaired solely by NPS, reasonable assurances that load reductions will be achieved are not required (by EPA) in order for a TMDL to be approved.*		76-78
Monitoring	***Monitoring Plan included?*** *Note: EPA does not approve effectiveness monitoring plans but providing a general plan is helpful to meet reasonable assurance requirements for nonpoint source reductions. A monitoring plan should describe the additional data to be collected to determine if the load reductions provided for in the TMDL are occurring and leading to attainment of water quality standards.*		76

Implementation	***1. Implementation Strategy included?*** The MPCA requires a general implementation strategy/framework in the TMDL. *Note: Projects are required to submit a separate, more detailed implementation plan to MPCA within one year of the TMDL's approval by EPA.* ***2. Cost estimate included?*** The Clean Water Legacy Act requires that a TMDL include an overall approximation ("…a range of estimates") of the cost to implement a TMDL [MN Statutes 2007, section 114D.25]. *Note: EPA is not required to and does not approve TMDL implementation plans.*	64 Appendix F
Public Participation	• Public Comment period (dates) • Comments received? • Summary of other key elements of public participation process *Note: EPA regulations require public review [40 CFR §130.7(c)(1)(ii), 40 CFR §25] consistent with State or Tribe's own continuing planning process and public participation requirements.*	80

Executive Summary

The Groundhouse River is located in east-central Minnesota in the Snake River watershed. The majority of the watershed is located in Kanabec and Mille Lacs counties with a small area in Isanti County. The watershed has a drainage area of approximately 139 square miles. The Groundhouse River and the South Fork Groundhouse River are listed on Minnesota's final 2006 and draft 2008 303(d) lists as being impaired due to not supporting their designated aquatic life and aquatic recreation uses. As required by the Clean Water Act, Total Maximum Daily Loads have been developed to address these impairments.

An evaluation of potential stressors indicates that excessive fine sediment (primarily the sand component) is the most likely cause of the impaired aquatic life in the Groundhouse River watershed. Natural features, such as low gradient streams and soils prone to erosion, may also be playing a role in certain reaches. Secondary stressors, such as low dissolved oxygen and elevated nutrient concentrations, were also identified but are not thought to be as significant as the excessive fine sediment and no TMDLs were developed for these secondary stressors.

Fecal coliform counts were found to exceed both the monthly geometric mean and the daily not-to-exceed components of Minnesota's water quality standards. Two sites, one on the Groundhouse River upstream of Ogilvie and one on the South Fork Groundhouse River, exhibited the highest counts of fecal coliform and impairments were observed at these two sites during high flow and low flow conditions. Values that exceeded water quality standards at other locations in the watershed were generally only found during high flows.

Various techniques were used to estimate the most significant sources of sediment and fecal coliform, including the application of a watershed model. The most significant sources of sediment were found to be erosion from cropland and streambank erosion, and the most significant sources of fecal coliform were found to be animal operations and failing onsite wastewater treatment systems. Considerably more uncertainty is associated with the estimates of the fecal coliform sources since the sources and fate and transport of bacteria in the environment are not yet well understood.

Allowable loads of sediment were estimated based on reducing sediment loads from all of the significant anthropogenic sources in the watershed such that future loads will better approximate "natural" conditions. Allowable loads of fecal coliform were calculated based on the use of the available flow data, the numeric water quality standard, and appropriate conversion factors. Approximately a 30 percent reduction in sediment loads was determined to be necessary, with the load reductions needed for fecal coliform varying by location and by flow condition.

Various Best Management Practices (BMPs) to address the various sources of sediment and fecal coliform are identified and described in the TMDL report and include riparian buffers, filter strips, fencing, manure management, conservation tillage, and the fixing of failing onsite wastewater systems. Stream restoration projects may also be required to improve the channel substrate and habitat required to support aquatic organisms at certain sites in the watershed, especially a site near Ogilvie with naturally low gradient. The expected costs and effectiveness of the various BMPs are identified and considerations are provided for the prioritization of the BMPs.

Following approval of the TMDLs, a formal implementation plan should be developed by the local stakeholders to identify the most practical and cost-effective BMPs for this watershed. Stakeholder input will be crucial to the success of the plan. Educational programs that focus on the importance of BMPs for protecting water quality and biological health as well as the cost-share programs available in the area should be advertised to raise awareness and increase levels of voluntary participation. Following implementation, continued monitoring will be needed to determine if fish and macroinvertebrate IBI scores are improving and fecal coliform loads are declining. Implementation of BMPs should continue until all monitoring sites in the watershed achieve water quality standards.

1 Introduction

The Groundhouse River is located in east-central Minnesota in the Snake River watershed (Cataloguing Unit 07030004) (Figure 1). The majority of the Groundhouse River watershed is located in Kanabec and Mille Lacs counties with a small area in Isanti County. The watershed has a drainage area of approximately 139 square miles.

The Clean Water Act and U.S. Environmental Protection Agency (USEPA) regulations require that states develop TMDLs for waters identified as impaired on the Section 303(d) lists. The Groundhouse River and the South Fork Groundhouse River are listed on Minnesota's final 2008 303(d) list as described in Table 1. Impaired waters listings based on the 2008 303(d) list are shown in Figure 2. The listings for the impairment of recreational use are due to high levels of fecal coliform bacteria exceeding both the monthly five-sample geometric mean standard (200 orgs/100 mL) and the standard based on individual samples (10 percent equal to or greater than 2,000 orgs/100 mL). The listings for the impairment of aquatic life were based on the results of biological monitoring which showed that Indices of Biological Integrity (IBI) ranked below acceptable levels for both fish and invertebrate communities found in similar streams in the St. Croix River Basin. Based on the final 2008 list, the target date for the completion of TMDLs for these listings is 2008.

Table 1. 2008 Final 303(d) List Information for the Groundhouse River Watershed

River ID	Name	Description	Designated Uses	Basis of Impairment	Year Listed
07030004-512	Groundhouse River	From South Fork Groundhouse River to Snake River	Aquatic Recreation	Fecal Coliform	2008
07030004-513	Groundhouse River	Headwaters to South Fork Groundhouse River	Aquatic Life Aquatic Recreation	Fish and Invertebrate IBIs Fecal Coliform	2002, 2004 2002
07030004-573	South Fork Groundhouse River	Headwaters to Groundhouse River	Aquatic Life Aquatic Recreation	Fish and Invertebrate IBIs Fecal Coliform	2004, 2008 2008

Source: MPCA (2008)

The Minnesota Pollution Control Agency (MPCA) contracted with Tetra Tech to provide technical support for the fecal coliform and biota (sediment) TMDLs for the Groundhouse River watershed. The following reports have been submitted by Tetra Tech in support of this project and are attached as appendices to this TMDL report:

- Water Quality and Fecal Coliform Evaluation, Revised Draft November 14, 2006 (Appendix A)
- Watershed Characterization and Water Quality Modeling Report, Initial Model Setup - Revised Draft January 12, 2007 (Appendix B)
- Summary of Field Reconnaissance of the Groundhouse River Watershed (November 17, 2006; (Appendix C))
- Biological Data Assessment Report, Revised Draft January 14, 2008 (Appendix D)

- Load Duration Curve Results (Appendix E)
- Implementation Strategies (Appendix F)

This draft TMDL report will summarize the information presented in the supporting documents, describe the major sources of pollutant loading in the Groundhouse watershed, determine the required allocations for each pollutant, and outline an implementation plan that MPCA may build on to reach the water quality goals for this watershed.

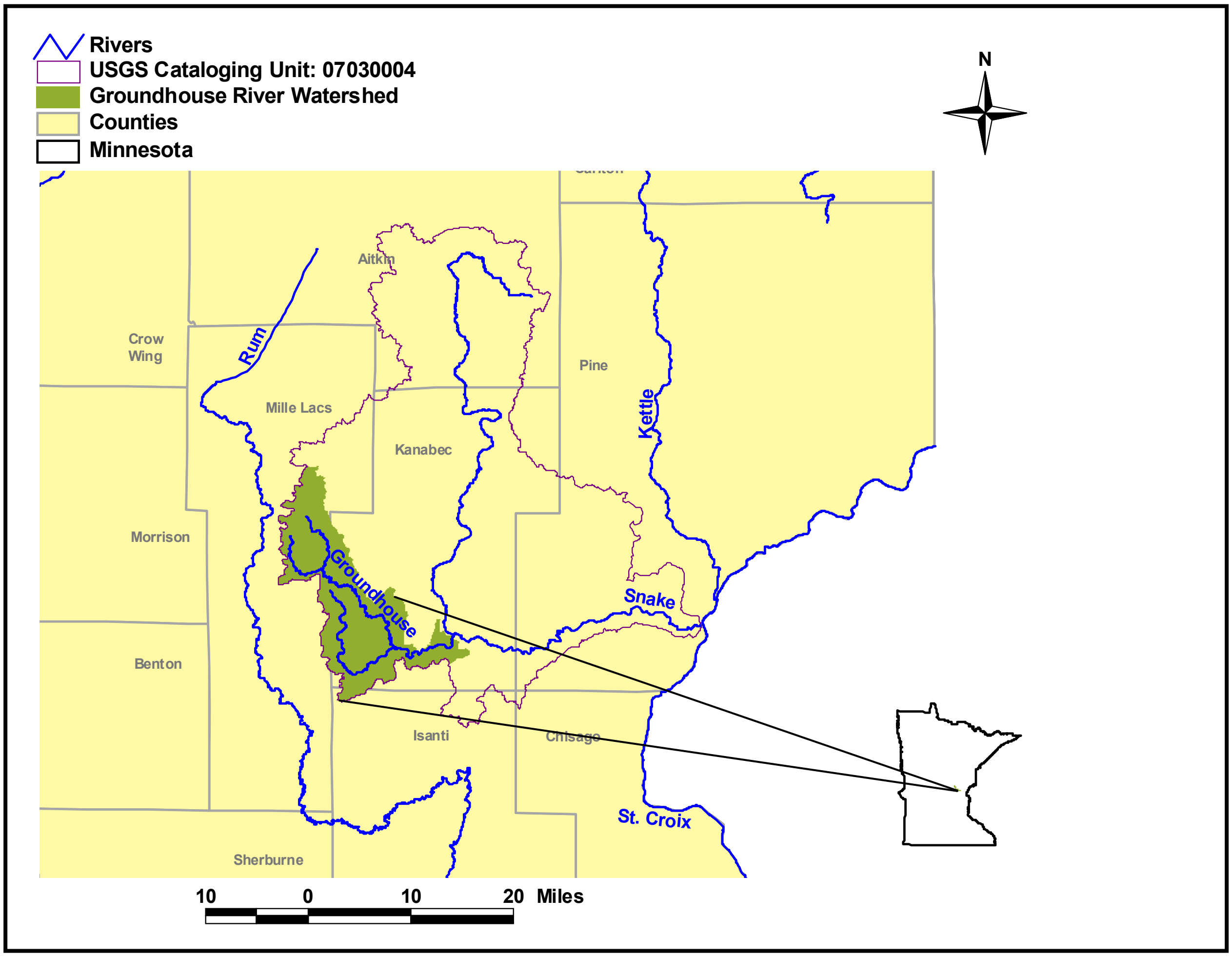

Figure 1. Location of the Groundhouse River Watershed

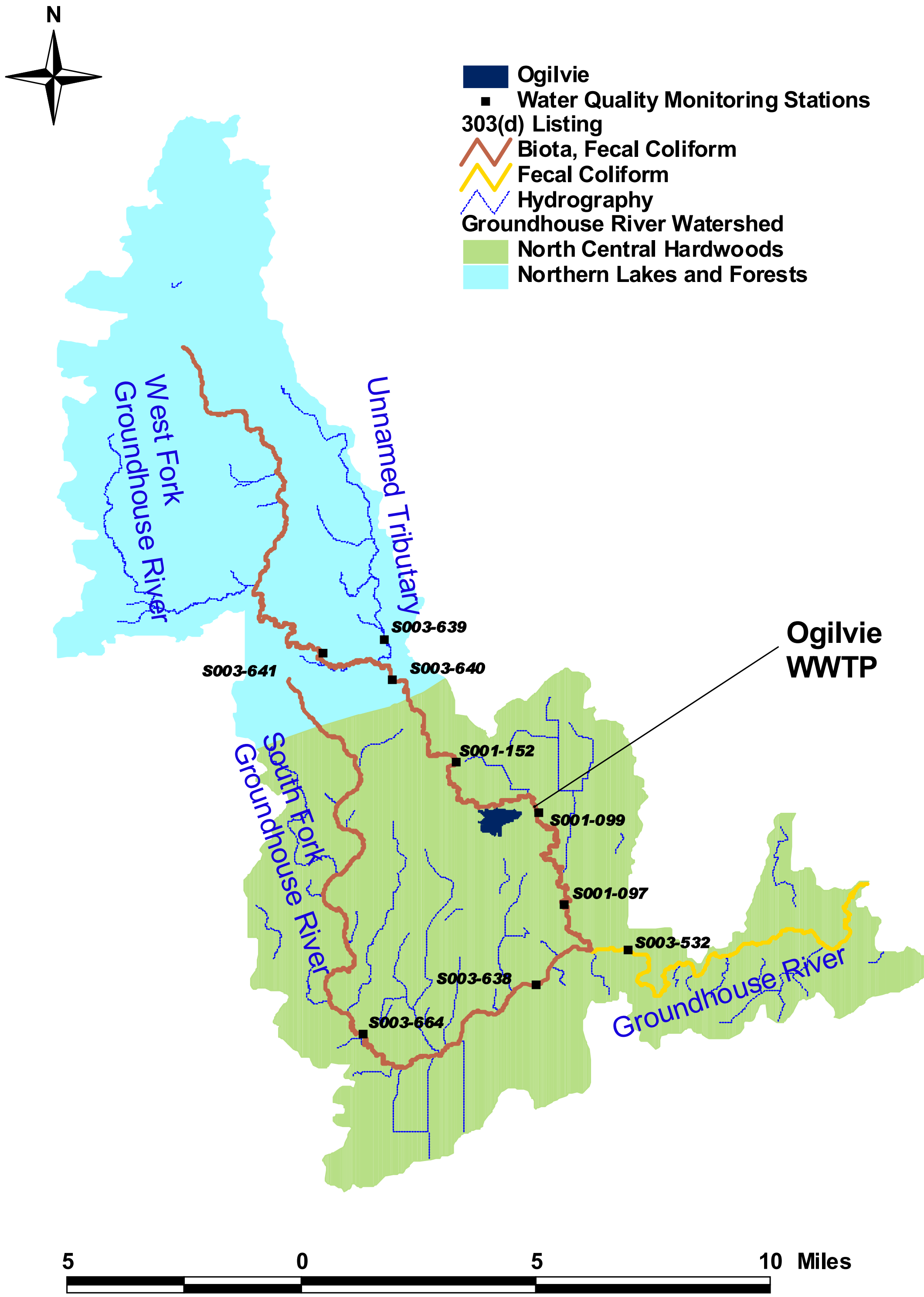

Figure 2. Location of 2008 Section 303(d) Impaired Segments, Monitoring Stations, and Level-Three Ecoregions in the Groundhouse River Watershed

2 Water Quality Standards and Review of Available Data

The purpose of a TMDL is to identify the maximum amount of a pollutant that a waterbody can receive and still meet water quality standards. As such, it is very important to understand the water quality standards that apply to the impaired waterbody. This section of the report provides information on the water quality standards that are relevant to the Groundhouse River watershed TMDLs.

2.1 WATER QUALITY STANDARDS

Minnesota adopted its first statewide water quality standards in 1967. These standards have been updated by adding new standards and regulations periodically since then. The comprehensive Clean Water Act amendments of 1972 require states to adopt water quality standards that meet the minimum requirements of the federal Clean Water Act. Minnesota's water quality standards meet or exceed the federal requirements.

Under the Clean Water Act, every state must adopt water quality standards to protect, maintain, and improve the quality of the nation's surface waters. These standards represent a level of water quality that will support the Act's goal of "fishable and swimmable" waters. Water quality standards consist of three components: beneficial uses, numeric or narrative standards, and a nondegradation policy. Minnesota's water quality standards are summarized in Table 2 and explained in greater detail below.

Table 2. Minnesota Water Quality Standards

Component	Description
Beneficial Use	Beneficial uses are the uses that states decide to make of their water resources. The process of determining beneficial uses is spelled out in the federal rules implementing the Clean Water Act.
Numeric Standards	Numeric water quality standards represent safe concentrations in water that protect a specific beneficial use. If the standard is not exceeded, the use should be protected.
Narrative Standards	A narrative water quality standard is a statement that prohibits unacceptable conditions in or on the water, such as floating solids, scum, visible oil film, or nuisance algae blooms. Narrative standards also prohibit serious impairment of the normal fisheries and lower aquatic biota upon which they are dependent and the use thereof, material alteration of the species composition, material degradation of stream beds, and the prevention or hindrance of the propagation and migration of fish and other biota normally present. The commissioner will consider all readily available and reliable data and information for the following factors of use impairment: an index of biological integrity calculated from measurements of attributes of the resident fish community, the resident aquatic invertebrate community, and the resident aquatic plant community by assessments of refuge for fish and invertebrates and excessive sedimentation.
Nondegradation	(Equivalent to the federal term "antidegradation.") The fundamental concept of nondegradation is that lakes, rivers, and streams whose water quality is better than the applicable standards should be maintained at that high level of quality and not allowed to degrade to the level of applicable standards.

Water quality standards and related provisions can be found in several Minnesota rules, but the primary rule for statewide water quality standards is Minnesota Rules Chapter 7050. Included in this rule are the following:

- A classification system of beneficial uses for both surface and groundwaters
- Numeric and narrative water quality standards
- Nondegradation provisions
- Provisions for the protection of wetlands
- Treatment requirements and effluent limits for wastewater discharges
- Other provisions related to protecting Minnesota's water resources from pollution

MPCA provides guidance for assessing impairment status in its Guidance Manual for Assessing the Quality of Minnesota Surface Waters for Determination of Impairment (MPCA, 2007).

The Groundhouse River and South Fork of the Groundhouse River are classified as Class 2B; the West Fork of the Groundhouse River is Class 2C. 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 waters is also protected as a source of drinking water." Minnesota rules specify that the quality of Class 2C surface waters "shall be such as to permit the propagation and maintenance of a healthy community of indigenous fish and associated aquatic life, and their habitats. These waters shall be suitable for boating and other forms of aquatic recreation for which the waters may be usable." The narrative and numeric criteria that apply to these beneficial uses are described in the sections below.

2.2 BIOLOGICAL DATA

Measures of biological health such as the Index of Biological Integrity (IBI) can be used as an assessment of the overall health of the fish and macroinvertebrate communities and are an excellent way to determine whether designated aquatic life uses are being supported. They also provide a quantitative method by which to interpret the narrative aquatic life criterion. Fish and macroinvertebrate IBI data for the Groundhouse River watershed are presented in this section of the report.

Metrics that are components of the IBI respond to known anthropogenic disturbance; some metrics can be used as a general indicator of disturbance, while others can be an indicator of a specific stressor (Niemela and Feist, 2000; Chirhart, 2003). Because fish and macroinvertebrates respond to different sets of disturbance and stressors, it is helpful to use multiple assemblages in assessment of the biological health of an aquatic system.

Examination of each biological assemblage contributes to a broader assessment of ecosystem conditions on both a spatial and temporal scale. For instance, fish are mobile and integrate larger-scale disturbance at the reach scale and have life stages that can be used to determine sensitivity to certain pollutants or physical disturbances. Benthic macroinvertebrate assemblages are integrators of chemical and physical conditions at a localized scale. They are not as mobile as fish and will experience physicochemical effects at specific points in a stream. Each assemblage (i.e., fish and benthic macroinvertebrates) is sensitive to groups of stressors (e.g., metals, pesticides, physical habitat, etc.) and on different time scales (e.g. slug of pollution or sustained exposure). Using multiple assemblages in assessments has an advantage in identifying potential for severe pollution problems by early detection and opportunity for abatement.

IBI scores for fish assemblages are numeric thresholds used to identify biotic impairment in Minnesota rivers and streams. Scores can fall within any of the five condition categories (e.g., very poor, poor, fair, good, and excellent) that are used to determine level of impairment, if any, at a river site. Table 3 shows the threshold values for both fish and macroinvertebrate biotic impairment. Fish assemblage condition thresholds vary by drainage area and differ among the three categories. Each of the categories is defined by scoring thresholds reported in Table 3. Additional scoring categories below the reference threshold for each drainage size (Table 3) were derived from stream condition observations in the bottom 5% percentile of reference streams sampled for a region. Descriptions for individual biometrics that comprise the IBI, scoring ranges, and regional expectations were described in Lyons (1992). The lower scores from reference sites within each region and for each drainage size unique responses to regional human influence on rivers and streams and so will have non-uniform scoring ranges when comparisons are made between stream reaches within differing drainage sizes.

Table 3. Impairment Thresholds for Fish and Macroinvertebrate IBI Scores in the St. Croix River Basin

Drainage Area (mi^2)	Fish Threshold	Macroinvertebrate Threshold
0 to 20	46	50
20 to 54	68	50
55 to 200	69	50

The cause of biological impairment can be difficult to determine as many factors, including pollutants and habitat, can stress a biological community. An initial effort for identifying existing pollutants in the Groundhouse River used the Stressor Identification (SI) process (USEPA, 2004a). This evaluation indicated that a primary cause for biological impairment was "loss of suitable habitat from unstable or unsuitable substrates caused by excess fines less than 2 mm in diameter." While fine sediment was indicated as the likely cause of the biological impairment, an intensive monitoring program to evaluate other potential stressors was performed in 2005. These data were reviewed during development of the TMDL and confirm that fine sediment is the most likely primary stressor in the watershed. Several secondary stressors, for which TMDLs have not been developed, were also identified and are more fully discussed in Appendix D.

Fish and macroinvertebrate community composition data have been collected at 26 sites in the Groundhouse River watershed by the MPCA, Minnesota Department of Natural Resources (MNDNR), and the University of Minnesota. Although data were collected by multiple management agencies, standard protocols developed by the MPCA were followed (Anon., 2002). Figure 3 shows the locations of the sampling stations along the mainstem of the Groundhouse River and the South Fork.

The biological data were collected between 1996 and 2006, typically between the months of June and September. Many of the sites were visited only once; however, several sites were visited two to three times. Sites were initially selected either at random, following U.S. EPA EMAP protocols, or were specifically targeted reference sites (Niemela and Fiest, 2000). Additional non-random sites were added to investigate possible impairment after one site did not attain MPCA standards.

Data from the mainstem of the Groundhouse River and the South Fork Groundhouse River were evaluated to determine which water quality or physical habitat factors might explain responses in each of the biological communities. Biological impairments were identified through an analysis of benthic invertebrate communities and fish communities collected from select sites along with supporting chemical and physical data.

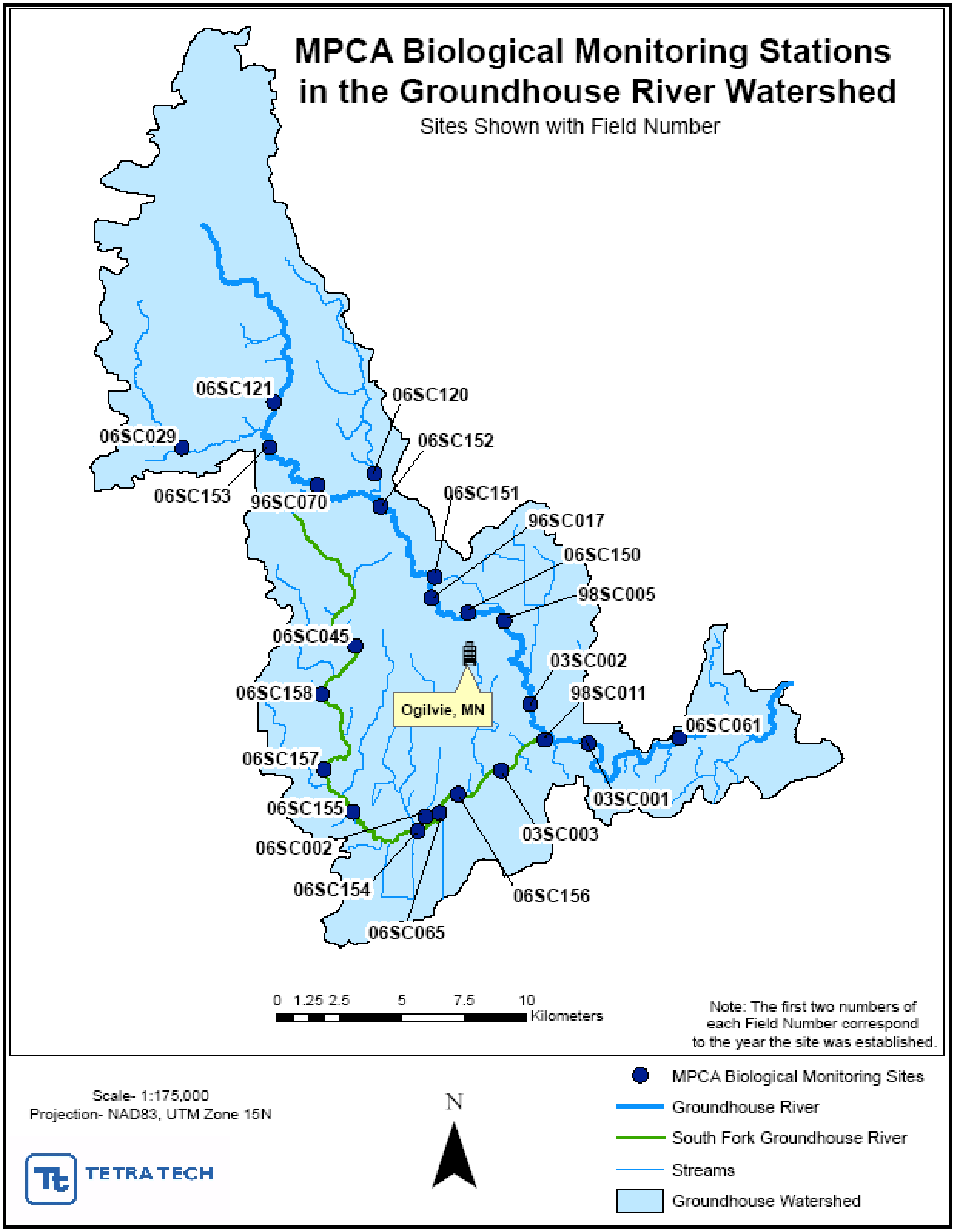

Figure 3. Location of Biological Monitoring Sites in the Groundhouse Watershed

2.2.1 Fish IBI Scores and Impairments

Fish IBI scores have been calculated at the 21 sites in the watershed that were sampled in 2006 (Figure 4). Based on the fish IBI thresholds, four sites are impaired and the impairment is limited to two distinct geographic areas. The first area includes three sites at and near the town of Ogilvie on the main fork of the Groundhouse River. The second area includes only one site located near the headwaters of the South Fork Groundhouse River. Since these two areas differ dramatically in size (drainage area), surrounding land use, and channel morphology, data from the two areas were analyzed separately. The three sites at or near the town of Ogilvie are referred to as Impaired Area 1, (sites 98SC005, 06SC150, and 96SC017), and the site at the headwaters of the South Fork of the Groundhouse River is referred to as Impaired Area 2 (site 06SC045).

The most likely stressors identified in the Biological Assessment Report (Appendix D) for Impaired Area 1 were fine sediments, particularly fine sands (0.06-2 mm) and combinations of silt/clay/muck. Presence of increased fine sediment (%Fines) habitat corresponded with impaired biological condition. Embeddedness was measured as the percentage of fines that buried larger substrate particles; %Embeddedness was high in areas where high %Fines measurements were made. At sites where biological impairments were observed, %Fines and Total Suspended Solids (TSS) concentrations were high. However, this was not always the case and identification of either %Fines or TSS were considered surrogates for loss of habitable substrate. Anthropogenic sources of fine sediment exist in this area. Most notably, livestock operations located adjacent to and upstream of the impaired sampling locations allow cattle direct access to the stream, exacerbating stream bank instability and the potential for erosion due to flowpath alteration.

Natural features of the landscape in Impaired Area 1 may also contribute to fine sediment deposition and retention. First, Impaired Area 1 has a lower stream slope than most other areas of the watershed. Stream reaches with lower slope may have a lower capacity to transport fine sediment because of reduced stream power. This means that even without a supply of excess sediment coming from upstream, Impaired Area 1 may be a natural depositional zone for fine sediment. Second, soil types in Impaired Area 1 are more easily eroded than other soil types in the watershed. (Appendix B discusses soil erodibility factors in the watershed.) Higher erodibility values in Impaired Area 1 indicate that this area is at a higher risk for erosion.

It is important to note that other non-impaired sites also fall within the area of higher erodibility. However, the combinations of anthropogenic stressors that exist in Impaired Area 1, including livestock operations, do not exist in other areas with high erodibility. In addition, stream slope values at other sites are higher, enabling the stream to transport excess sediment downstream. It is likely that although the natural conditions might favor excess sediment, nearby anthropogenic disturbances have triggered a more significant sediment problem.

Impaired Area 2 in the headwaters of the South Fork Groundhouse River is most likely impaired by low dissolved oxygen levels. Two dissolved oxygen measurements are available for Impaired Area 2 with concentrations of 4 mg/L and 3.28 mg/L. Both of these measurements are below the MPCA standard for warm water fisheries of “not less than 5 mg/L as a daily minimum” and are well below the average concentration for the Groundhouse River watershed. In fact, Impaired Area 2 has the lowest recorded dissolved oxygen concentrations in the watershed.

Flow measurements are also the lowest recorded in the watershed in Impaired Area 2, at 0.071 cubic feet per second (ft^3/sec) and 0.014 ft^3/sec. This low flow may be a significant contributor to the observed dissolved oxygen levels. Impaired Area 2 may be functionally behaving more like a wetland than a stream ecosystem. Wetland ecosystems generally have lower levels of dissolved oxygen than other aquatic systems (McCormick and Liang, 2003). To confirm these observations, MPCA plans to monitor DO at this location during the summer of 2008.

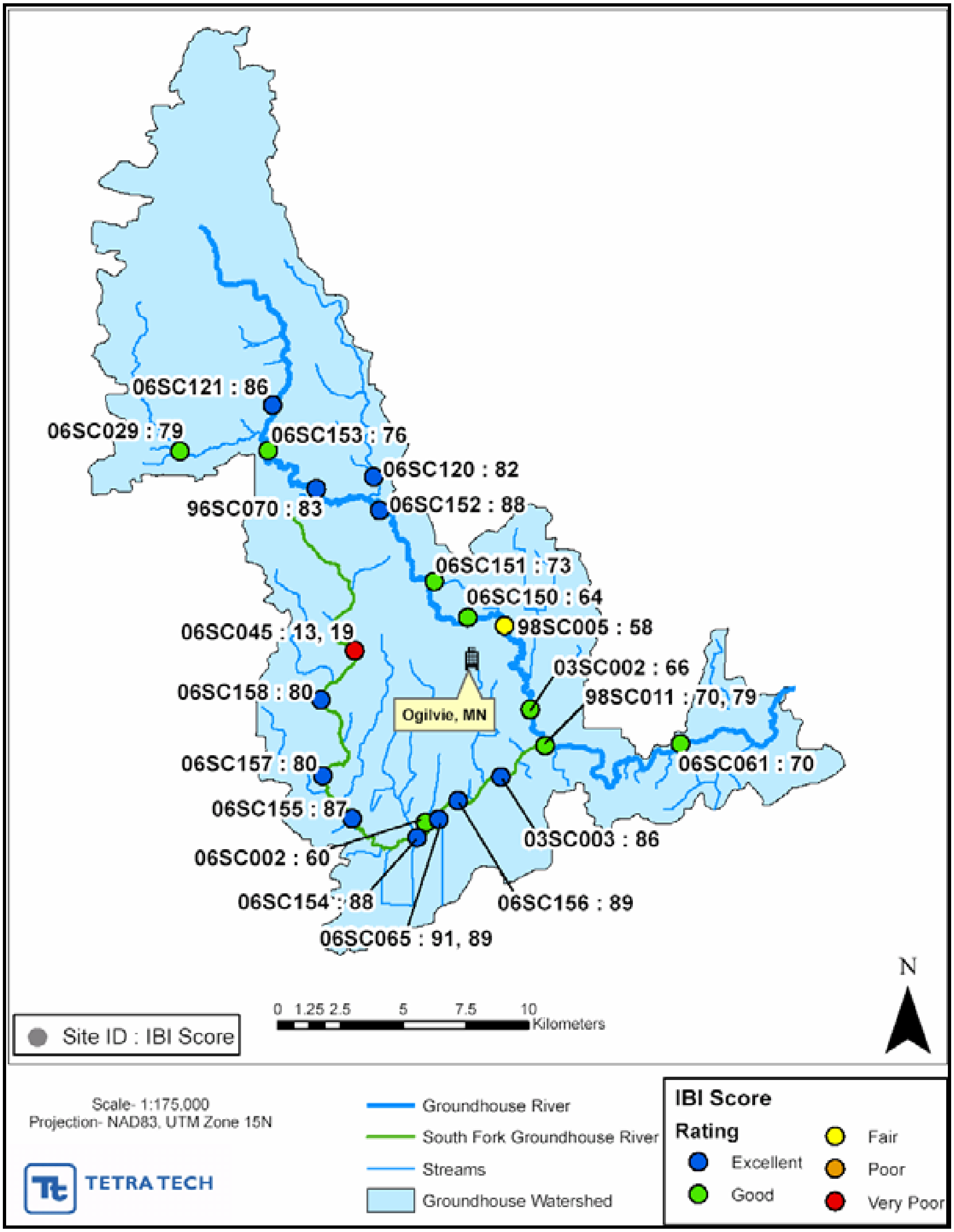

Figure 4. Fish IBI Scores Collected in the Groundhouse Watershed in 2006 (missing site 96SC017)

2.2.2 Macroinvertebrate IBI Scores and Impairments

Macroinvertebrates were found to be impaired throughout the Groundhouse River watershed with only 40 percent of sites in the watershed scoring better than the impairment threshold for MIBI scores (50). Only four sites (16 percent of samples) had MIBI scores greater than 60. Figure 5 shows the MIBI scores calculated at the 24 sites in the watershed.

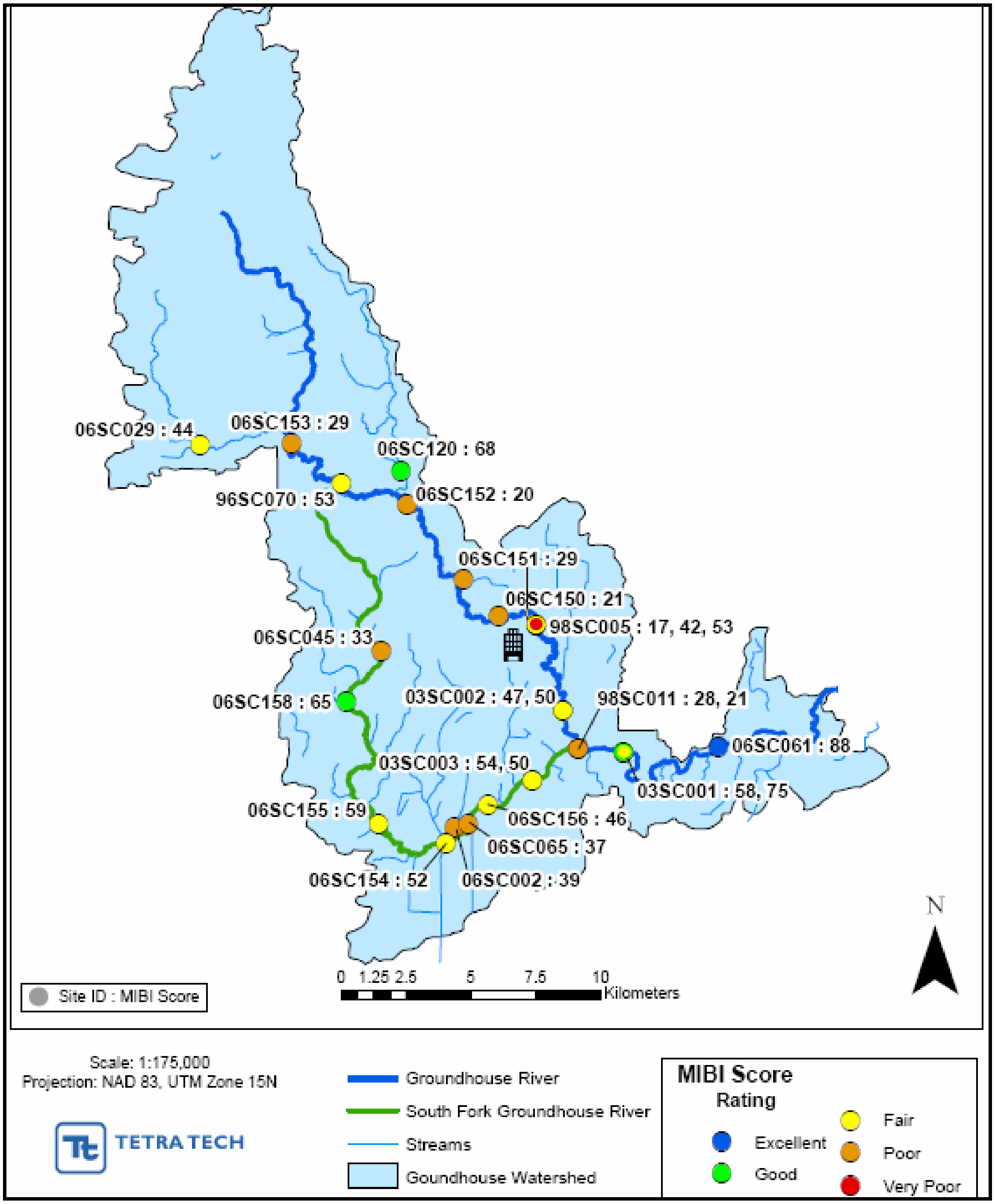

Figure 5. Macroinvertebrate IBI Scores Collected in the Groundhouse Watershed in 2006

Based on the widespread impairment of the benthic invertebrate communities, it appears that the invertebrate community is being impacted to a greater extent than the fish community. The data indicate that this response is due to increases in fine sediment accompanied by decreases in channel gradient. The decrease in channel gradient reduces the energy of the river system in these areas, limiting the transport of fine sediments and facilitating sediment deposition. The data also indicate a response to ammonia concentration despite the fact that ammonia concentrations throughout the watershed are generally low (below MPCA water quality standards) and close to those concentrations measured in comparable Minimally Impacted Streams (Appendix A). The benthic invertebrate communities may be responding to excessive fine sediment, nutrient concentrations, or a coupling of the two.

The most consistent impairment among all South Fork Groundhouse River sites was an increase or presence of high levels of fine sediments. The presence of fine sediments also corresponded with a low-gradient channel. It is likely, then, that low gradient corresponds to low energy to move bedded sediment throughout the South Fork Groundhouse River watershed. The dominant presence of "burrower" taxa and consumers of plant organic material (e.g., *Oligochaeta*, *Polypedilum*, and *Dicrotendipes*) further suggest the presence of excessive fine sediments.

The uppermost site (06SC045) and the lowermost site (98SC011) in the South Fork Groundhouse River appear to have the same type of related causes for biological impairments. Site 06SC045 had very low dissolved oxygen concentrations to which increased ammonium concentrations can likely be attributed. Ammonium concentration is considered an indicator for low dissolved oxygen concentrations; increased ammonium concentrations indicate the presence of increasing anoxia. Although ammonium concentrations did not reach levels known to affect salmonids, identification of this parameter as a stressor that impaired benthic macroinvertebrate communities is a surrogate for presence of increasing eutrophic or anoxic conditions in portions of the drainage. Low gradient at this site and slow water movement explains the dominance of burrowing benthic macroinvertebrates, free-swimming taxa, and species with high tolerance to warm water and low dissolved oxygen concentrations. Many of the same taxa were found to be dominant at Site 98SC011 and indicate that some of the same stressors are responsible for the observed impaired biological conditions.

The mainstem Groundhouse River sites with severely impaired benthic invertebrate communities were separated into an upper river group and a lower river group. The upper river region can be defined as those impaired sites above the Ogilvie Wastewater Treatment Plant (06SC150, 06SC151, 06SC152, and 06SC153). These sites were more frequently identified as having low gradient stream channels with some increased percent fines in habitable substrates. Water chemical characterization showed elevated ammonium (NH_4) concentrations in these same reaches.

The lower group of sites on the mainstem Groundhouse River (98SC005 and 03SC002) is located in or downstream of the sandy area near Ogilvie (i.e., Impaired Area 1 from the fish analysis). Site 03SC002 is located downstream of the Waste Water Treatment Plant (WWTP) and in a higher gradient reach than further upstream. Potential stressors at these sites included differing concentrations of total suspended solids (TSS), ammonium (NH_4), and total phosphorus (TP). TSS concentration was lower at the low gradient site (98SC005) and higher where the site gradient was higher (03SC002). Factors such as increased flow from treated effluent along with re-suspension of organics below this outfall could affect results for water quality characteristics. One example includes increased total phosphorus near the Ogilvie WWTP with limited concentrations at the downstream site (03SC002). The lower river sites were consistently dominated by filter-feeding benthic invertebrates (e.g., Hydropsychid caddisflies, etc.) further confirming the volume of available suspended materials in the water column. Higher concentrations of total nitrogen and ammonium suggest an accumulation of organics that serve as a reservoir for some of the elevated nutrient concentrations. Nitrogen dissociates more easily in surface water and showed higher concentrations at the mainstem site (03SC002) below the Ogilvie WWTP.

The lowermost site on the mainstem Groundhouse River (06SC061) had one of the highest biological condition scores despite trends at other sites showing biological impairment with high nutrient levels and percent fines. Site 06SC061 is a larger stream channel with a complex riparian structure and instream substrate structure that provides a variety of physical refugia. The cumulative impacts from pollutants on the biological community in large rivers do not appear to be as severe as individual effects from each pollutant on biological communities in smaller streams. Biota appear to be able to escape pollution impacts at the lowermost portion of the Groundhouse River (site 06SC061) despite the influence of these stressors in other portions of this drainage.

Evaluation of multiple biological assemblages (e.g., fish and benthic macroinvertebrates) showed differing extent of impairments at sites throughout the Groundhouse River drainage. These discrepancies in site assessments indicate that fish and benthic macroinvertebrate assemblages are responding to unique pollutant stressors (i.e., physical habitat or water quality). Currently, the benthic macroinvertebrates indicate more extensive impairments than do fish and this means stressors are present throughout the watershed. The benthic macroinvertebrates serve as an early warning for the presence of pollutants and provide guidance for developing a staged, management effort that prioritizes restoration projects in order to improve water quality and physical habitat.

2.3 FECAL COLIFORM DATA

Fecal coliform bacteria are an indicator organism, meaning that not all the species of bacteria of this category are harmful, but they are usually associated with harmful organisms transmitted by fecal contamination. They are found in the intestines of warm-blooded animals (including humans). The presence of fecal coliform bacteria in water suggests recent contamination from fecal matter and the possible presence of harmful bacteria (e.g., some strains of *E. coli*), viruses and protozoa (e.g., *Giardia* and *Cryptosporidium*) that are pathogenic to humans when ingested (USEPA, 2001). Minnesota, like many other states and jurisdictions, has used fecal coliform bacteria for its standard rather than actual pathogenic organisms.

The Minnesota rules state that fecal coliform concentrations in Class 2C and 2B waters shall "not exceed 200 organisms per 100 milliliters as a geometric mean[1] of not less than five samples in any calendar month, nor shall more than ten percent of all samples taken during any calendar month individually exceed 2,000 organisms per 100 milliliters. The standard applies only between April 1 and October 31."

The sections below compare the available data to these water quality standards and also evaluate potential spatial and temporal trends and relations to flow and precipitation.

2.3.1 Spatial Analysis

Review of the fecal coliform data in the Groundhouse River watershed show a wide range of reported values which is consistent with the behavior of bacteria in natural systems (Table 4). Median and geometric mean values are relatively similar at all stations with a few exceptions. The highest overall

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[1] Geometric means are used to represent average fecal coliform concentrations. A geometric mean is appropriate for summarizing the central tendency of environmental data that are not normally distributed (Helsel and Hirsch, 1991). Unlike an arithmetic mean, a geometric mean tends to dampen the effect of very high or very low values. It is calculated by taking the n^{th} root of the product of n numbers (or by taking the antilog of the arithmetic mean of log-transformed numbers).

geometric mean values of fecal coliform have been observed at S003-664 on the South Fork Groundhouse River. However, data at this location were only collected during July through October which are the months with the highest concentrations across the watershed. The highest median and 95th percentile and second highest geometric mean values are seen at station S001-152. The lowest overall values are seen at S003-639 located on an unnamed tributary near the confluence with the mainstem.

To highlight the spatial variability of fecal coliform throughout the watershed, the median and geometric mean fecal coliform concentrations were plotted as shown in Figure 6 and Figure 7. No clear spatial pattern (e.g., counts increasing in a downstream direction) is apparent in these figures.

Table 4. Fecal Coliform (organisms/100 mL) Summary for 2005 (April to October)

Station ID	Number of Samples	5th Percentile	Median	Geomean	90th Percentile	95th Percentile
Groundhouse River Above Confluence with South Fork						
S003-641	31	20	110	119.9	890	2,350
S003-639 (unnamed tributary)	30	5	55	45.2	133	316
S003-640	30	5	61	57.5	183	465
S001-152	31	16	230	219.5*	2,700*	8,050
S001-099	31	18	150	142.6	600	1,050
S001-097	31	22	91	101.1	360	785
South Fork Groundhouse River						
S003-664 (July – October)	19	90	170	279.3*	2,180*	4,410
S003-638	30	13	135	158.9	1,930	2,475
Groundhouse River Below Confluence with South Fork						
S003-532	26	26	120	160.1	785	1,975

* Indicates exceedance of water quality standards.

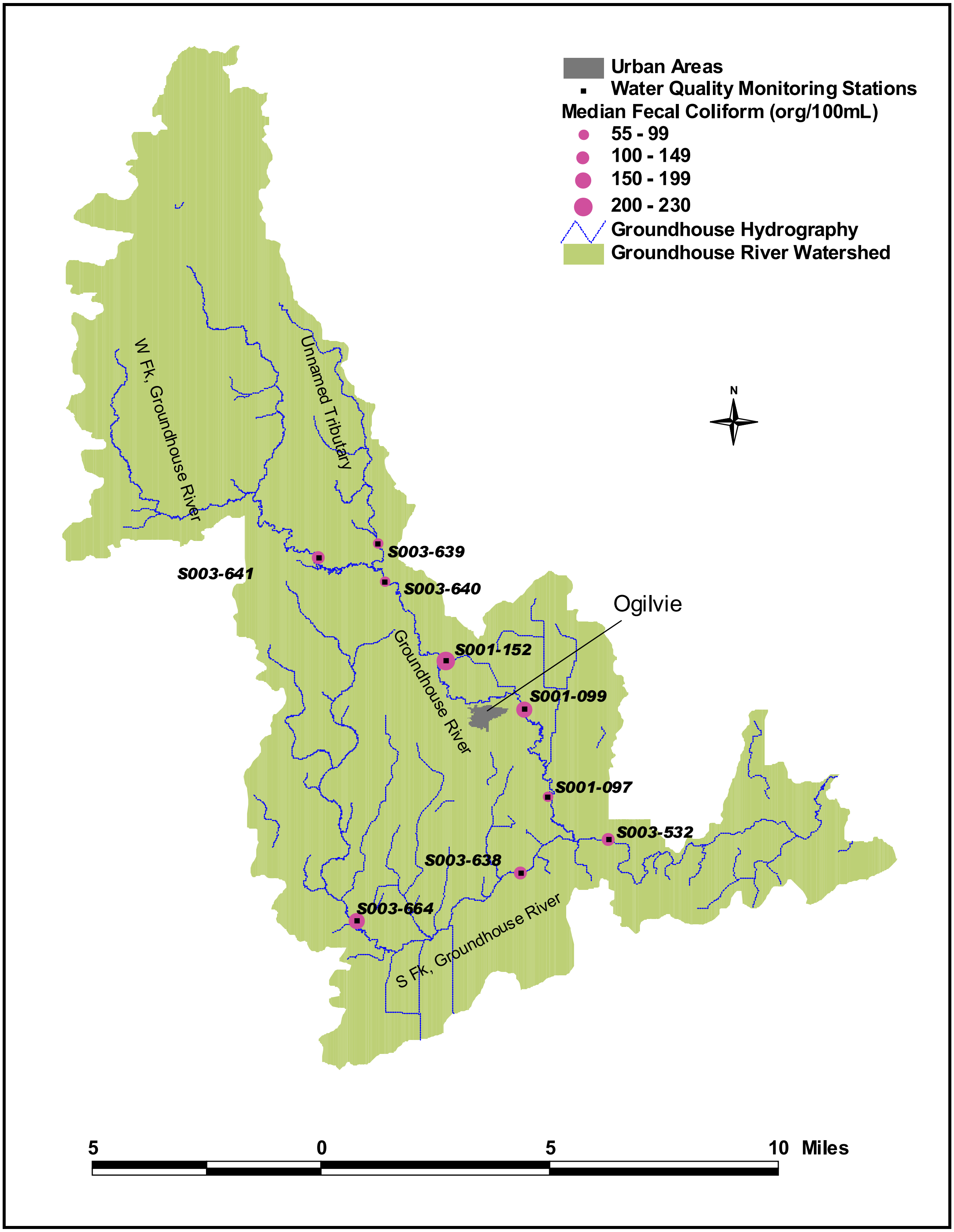


Figure 6. Spatial Distribution of Median Fecal Coliform in the Groundhouse River Watershed

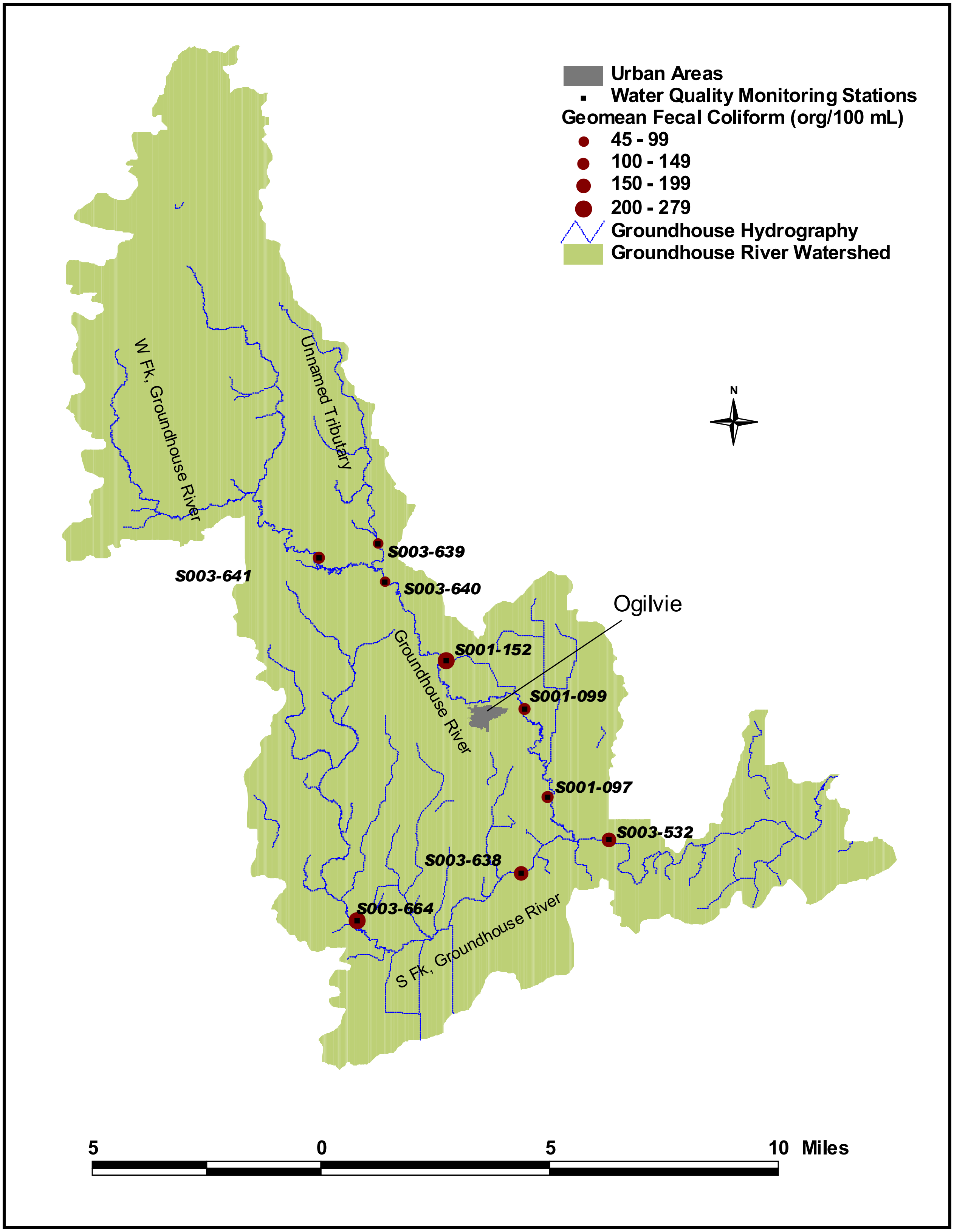


Figure 7. Spatial Distribution of Fecal Coliform Geomean in the Groundhouse River Watershed

2.3.2 Temporal Analyses

A long-term evaluation of fecal coliform counts in the Groundhouse River watershed is only possible at station S001-152 as data have been collected in the late 1980s, the late 1990s, and 2005. Review of the median and geometric mean fecal coliform concentrations aggregated by the decade of sampling shown in Figure 8 indicate a potential long term increase in fecal coliform concentrations which could be a result of some type of shift in land use or management.

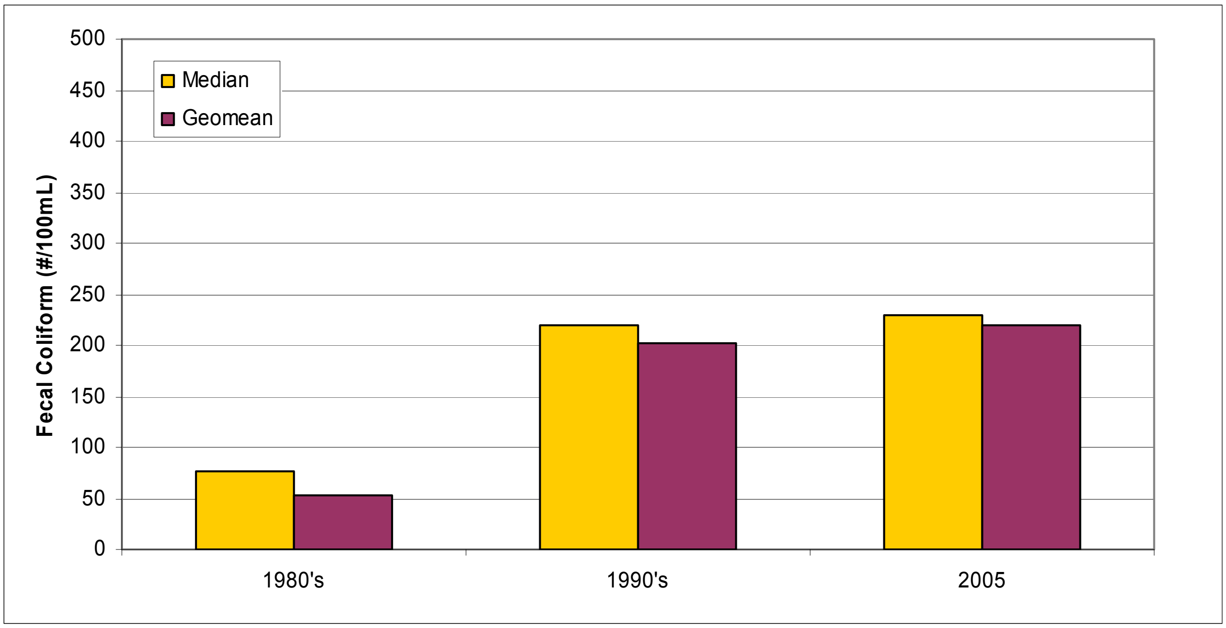
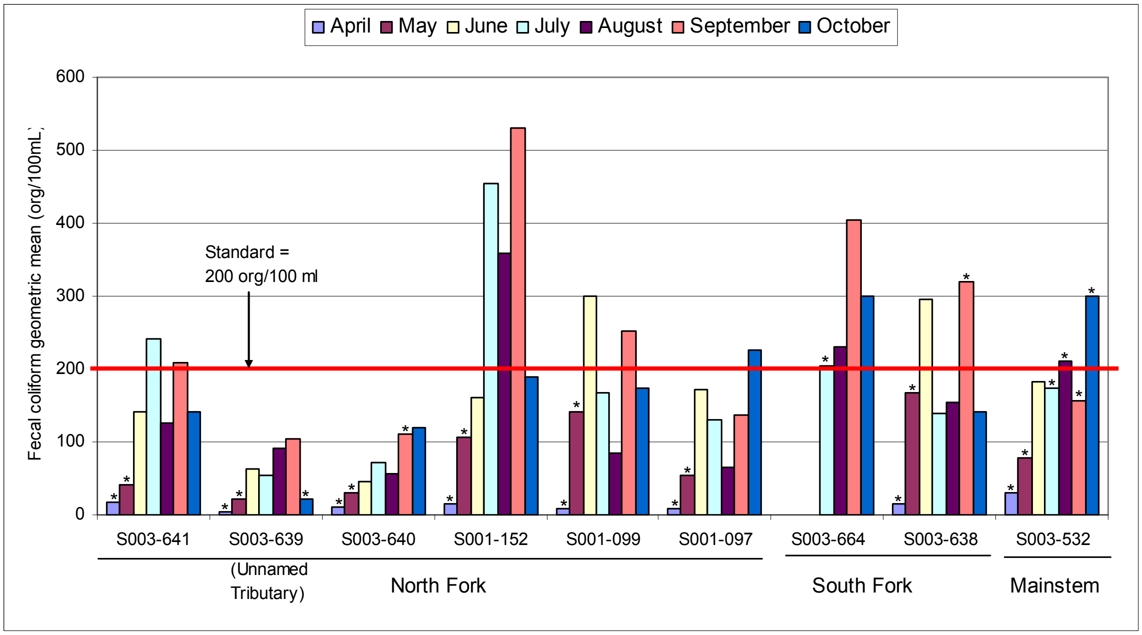


Figure 8. Median and Geometric Mean Annual Fecal Coliform Concentrations at S001-152

The 2005 data throughout the watershed were also evaluated with a monthly geometric mean component to evaluate potential seasonal trends (Figure 9). It should be noted that the April and May values are based on sample sizes less than 5, which is less than required for evaluation of the water quality standard. The monthly geometric mean values are highly variable by month, potentially due to the occurrence of rain events during any given month.



*Fewer than 5 samples obtained during this month.

Figure 9. Geometric Mean Fecal Coliform Concentrations by Month

Generally, storm events are the primary cause of nonpoint source loading to streams. To evaluate the importance of stormwater runoff on instream concentrations, the 2005 data set was evaluated based on antecedent rainfall. Monitoring events which occurred within 24 hours of at least a 0.5-inch rainfall event or 48 hours of at least a 1-inch rainfall event were considered to be wet (w) sampling. The remaining sampling was considered to have been done under dry (d) conditions. A box and whisker plot was developed to illustrate the differences between the wet and dry monitoring (Figure 10). A systematic increase in fecal concentrations is seen under wet conditions at all locations.

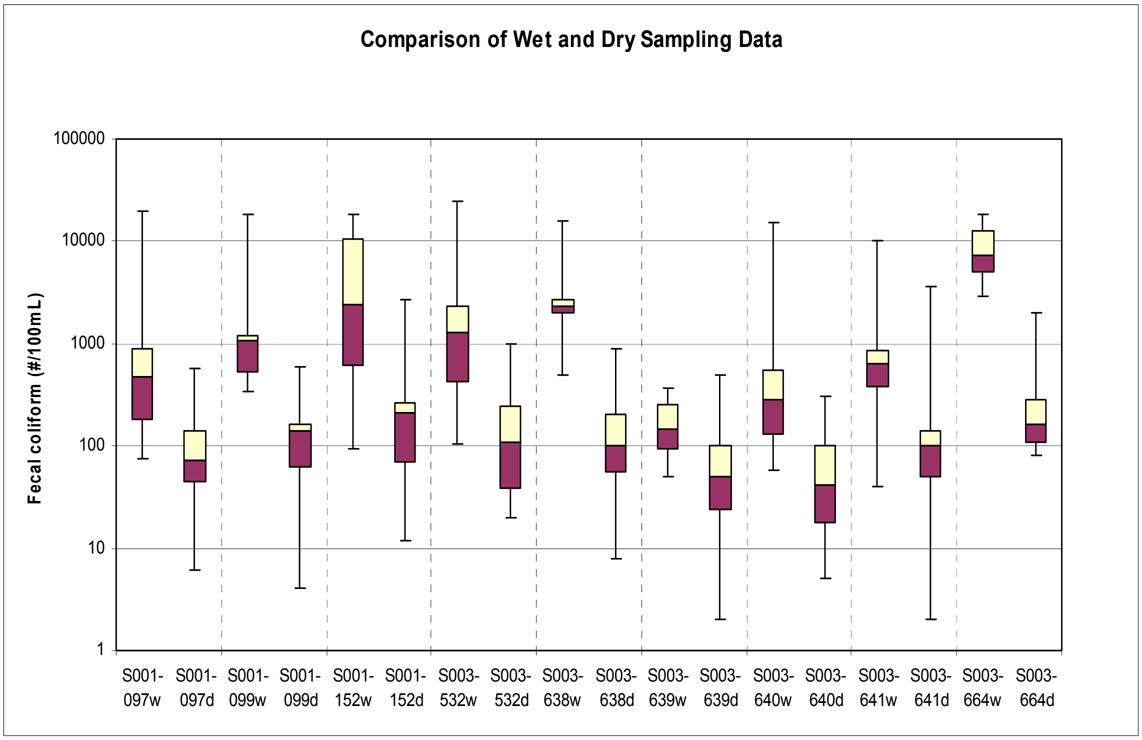


Figure 10. Comparison of Wet and Dry Weather Fecal Coliform Sampling[2]

2.3.3 Flow Duration Analyses

It is often useful to evaluate water quality observations in relation to flow because many water quality impairments manifest themselves differently during different flow conditions. One useful tool for such evaluations is a flow duration curve, which is developed by generating a flow frequency table (i.e., high flows to low flows) and plotting the data points to form a curve. The data reflect a range of natural occurrences from extremely high flows to extremely low flows. By plotting water quality concentrations (or loads) on the flow curve, various patterns can become apparent.

Flow gaging data have been collected in the Groundhouse River from 1999 through 2005 during the recreation season (April through October). Since the fecal coliform standard is evaluated during the recreational season, the seasonal flows collected in the Groundhouse are applicable and were used to develop flow duration curves. The dataset for 2004 was incomplete and was not used in the calculation of the flow duration curves. Isolated periods of missing data were also filled in based on scaled flows measured at USGS 05338500 on the Snake River near Pine City, MN.

Monthly fecal coliform geomean values were plotted against the flow duration intervals and the results for stations S001-152 and S003-664 are shown in Figure 11 and Figure 12 (results for all stations are included in Appendix A). They indicate that impairments at station S001-152 are primarily associated with low flow conditions, whereas the geomeans at station S003-664 exceed the standard during all available flow conditions. High fecal coliform counts at low flows can be associated with septic systems,

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[2] The box and whisker plot shows the range from the 5th to the 95th percentile of the observations at each station during wet and dry conditions. The first quartile (25th percentile), median, and third quartile (75th percentile) define the size of each box.

direct discharges, access of animals to the stream, or possibly illicit connections. Exceedances during high flows are typically associated with nonpoint source runoff and could also represent fecal coliform being re-suspended from the stream bottom.

Flow duration curves were also generated for the individual fecal coliform observations (see Figure 13 and Figure 14 for results at stations S001-152 and S003-664, respectively; results for all stations are shown in Appendix A). While the single sample water quality standard of 2000 organisms/100 mL is infrequently exceeded, the results at stations S001-152 and S003-664 indicate impairments during both high and low flow conditions. Results at other stations indicate that the majority of excursions of the instantaneous fecal coliform standard occur during high flow conditions.

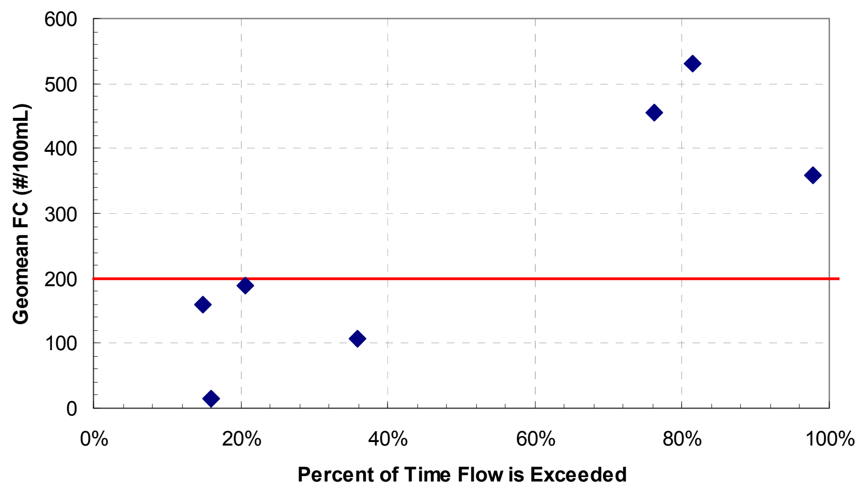

Figure 11. Fecal Coliform Monthly Geometric Mean Flow Duration Curve for S001-152

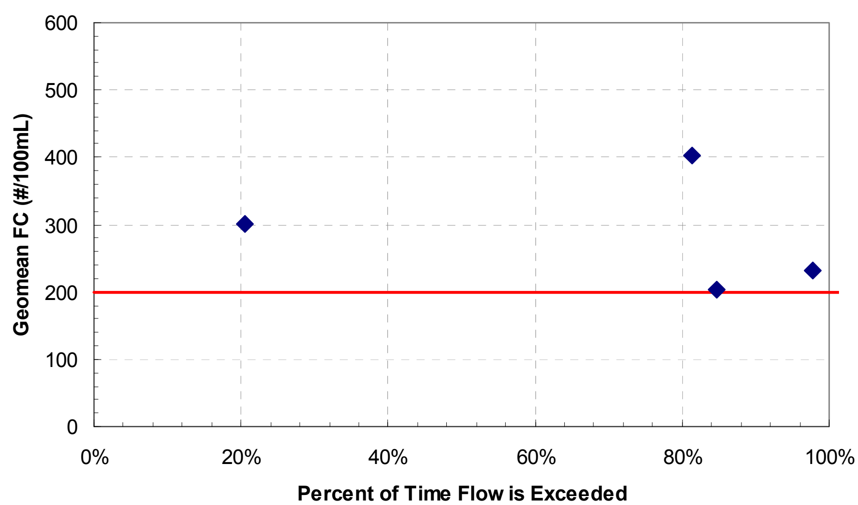

Figure 12. Fecal Coliform Monthly Geometric Mean Flow Duration Curve for S003-664

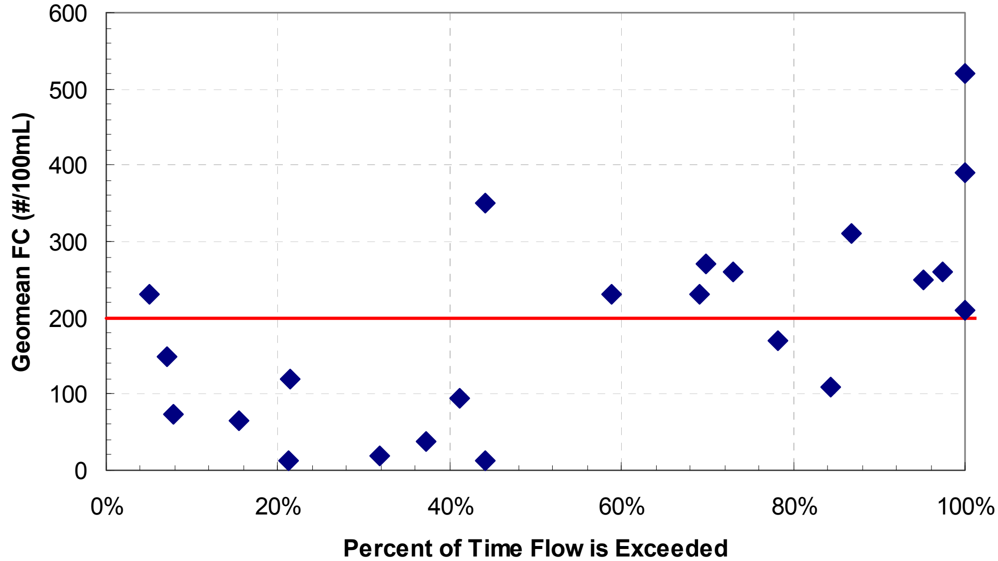

Figure 13. Fecal Coliform Flow Duration Curve for S001-152

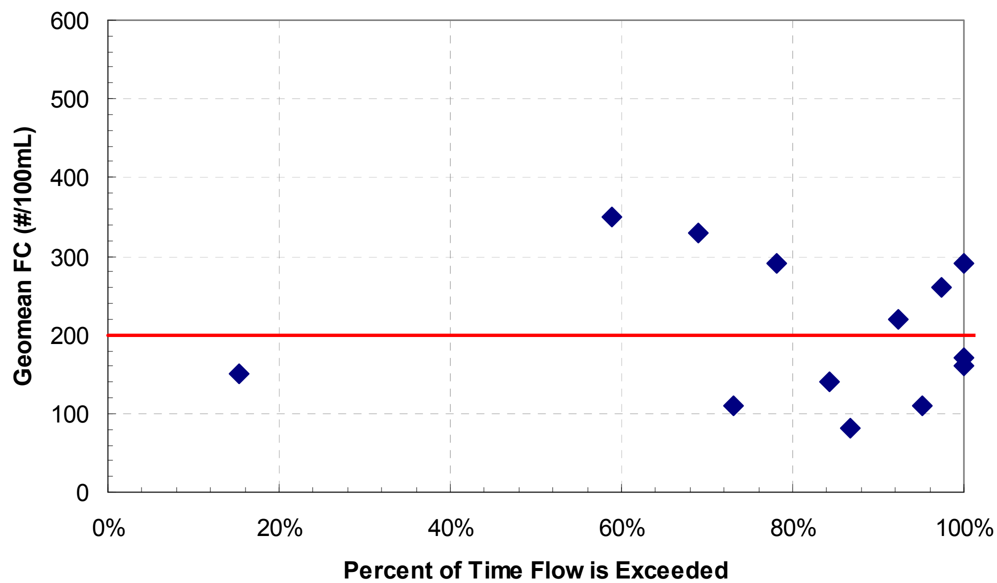

Figure 14. Fecal Coliform Flow Duration Curve for S003-664

3 Source Assessment

This section identifies the potential point and nonpoint pollutant sources that may contribute to the biota and/or fecal coliform impairments in the Groundhouse River watershed. Loads from these sources were quantified, where possible, using the watershed model and other tools and are presented in Section 4.

3.1 Point Sources

The following regulated point sources exist in the Groundhouse River watershed that are potential sources of sediment and/or fecal coliform.

3.1.1 Wastewater Treatment Plant

The Groundhouse River watershed contains one regulated wastewater treatment plant (WWTP), maintained by the city of Ogilvie on the mainstem of the Groundhouse River. The plant discharges directly to the Groundhouse River at Minnesota State Highway 23, and represents the largest point source of treated discharge in the watershed. Wastewater is treated using a trickling filter/chlorine disinfection system, and is permitted by the MPCA to discharge up to 230,000 gallons per day (USEPA, 2004a). MPCA records indicate that the plant has been cited for operational violations in the past. Some of these violations have been associated with the inability of the plant to effectively limit carbonaceous biochemical oxygen demand (CBOD) and total suspended solids (TSS) in discharged water. One violation of the monthly fecal coliform geometric mean standard also occurred in April 2005 (308 organisms/100 mL).

The WWTP is located downstream of the site near Ogilvie where the greatest impairment of the fish community is observed, so it is unlikely that the discharge is impairing the fish community. The macroinvertebrates in the watershed appear to be impaired by fine sediments, and the watershed loading model (Section 4) indicates that solids released from the facility comprise less than 0.3 percent of the sediment load. It is therefore unlikely that the WWTP is contributing to the macroinvertebrate impairment either. The stressor identification report (USEPA, 2004a) also states that the WWTP is not a likely cause of biological impairment.

The mainstem of the Groundhouse River is also listed as impaired for fecal coliform. The Ogilvie WWTP does not appear to be a major contributor to the fecal coliform load in this system because 1) the site with highest level of impairment is upstream of the WWTP, and 2) all excursions of the instantaneous standard downstream of the facility occur during high flow events when the contribution from WWTPs is typically proportionally low. In addition, excursions of similar magnitude also occur upstream of the facility during high flow events. Though the effluent from the WWTP may affect the water quality conditions observed downstream in the Groundhouse River, it was not found to be a significant source of either the fecal coliform or biota impairment during the sampling period.

3.1.2 Stormwater

There are no Municipal Separate Storm Sewer Systems (MS4s) within the Groundhouse River watershed that are required to have NPDES permits; therefore, the only potential sources of stormwater containing sediment and subject to permitting are construction sites. The MPCA issues construction permits for any construction activities disturbing one acre or more of soil, activities which are part of a larger

development or activities which disturb less than one acre of soil but are determined to pose a risk to water resources.
Construction storm water activities are considered in compliance with provisions of the TMDL if they obtain a Construction General Permit under the NPDES program and properly select, install and maintain all BMPs required under the permit, including any applicable additional BMPs required in Appendix A for discharges to impaired waters, or meet local construction stormwater requirements if they are more restrictive than requirements of the State General Permit.

3.1.3 Gravel Pits

The number of gravel pits currently located in the Groundhouse River watershed is not fully known. According to permitting data from the MPCA, 16 gravel pit facilities are active permit holders in the watershed. The number of sites with permits, however, may not accurately reflect the number of active facilities. It is illegal to actively pump wastewater from gravel pits into local waterbodies. This activity may occur, however, and may lead to increases in the fine sediment load of the Groundhouse River. Without data regarding the size and location of active facilities or the wastewater procedures followed, it is not possible to accurately assess the impact of gravel pits on the sediment load in the Groundhouse River watershed. However, conservatively assuming that all runoff water to the pits is discharged to a surface waterbody at a TSS concentration of 25 mg/L yields a percent contribution relative to the total sediment load of 0.01 percent in the Groundhouse watershed and 0.03 percent in the South Fork watershed (see Section 4 for details). Though illegal pumping may cause localized impacts on water quality at the time of discharge, gravel pits are not considered the primary source of high sediment loads in the watershed.

3.2 NONPOINT SOURCES

A large portion of the Groundhouse River watershed is used for agriculture. According to data available from the University of Minnesota, approximately 32 percent of the land use/land cover in the watershed is either row crop cultivation or pasture for livestock. Based on the results of the watershed loading model, agricultural practices, along with streambank erosion, comprise the majority of sediment loading to both listed segments. Fecal coliform loads are mostly attributed to beef and dairy operations.

3.2.1 Feedlots and Pasture Lands

Runoff associated with animal feeding operations (AFOs) and pasture used for grazing cattle may also have an impact on the chemical and physical loads for both the Groundhouse and South Fork Groundhouse rivers. According to data compiled by the Kanabec County Soil and Water Conservation District during drive-by surveys in combination with permit information available from the MPCA, 34 animal operations are located in the Groundhouse River watershed. These include 1,312 dairy cattle animal units; 2,876 beef cattle animal units; 576 swine animal units; 37 buffalo animal units; 30 horse animal units; and 1 chicken animal unit (see Section 4.2.5 for more details). Based on available data, these operations potentially contribute a significant proportion of the fecal coliform load to the South Fork Groundhouse and the mainstem Groundhouse. In addition, pasture lands contribute to the upland sediment load.

Cattle with access to the stream channel can also have a significant impact on stream bank erosion. Stream bank instability can significantly contribute to fine sediment deposition. Stream instability can also directly impact the geomorphology of the stream and hydrologic flow paths. An increase in fine sediment fills interstitial spaces in the stream bed important for fish as well as macroinvertebrates. The cattle will also contribute to the nutrient load of the system, potentially creating toxic environments for other organisms, or further driving eutrophication.

Although this information is very useful for developing general estimates of the impact of feedlots, the windshield surveys and watershed modeling do not reveal the full range of activity and potential for impacts to water quality from feedlots and pasture lands. For example, the timing of the surveys is critical and additional work would be needed to provide more insight into how hydrologic conditions affect pollutant transfer, how combinations of animal density and environmental conditions result in potential for stream impairments, and how other activities involving animal operations could contribute to degradation of aquatic resources (e.g., physical habitat and water quality).

The impact of animal operations and management practices on river ecosystems is multi-fold. It is possible that effluent runoff from improperly stored manure or runoff generated during precipitation events could elevate sediment, nutrient, and fecal coliform levels in the water column. Removal of riparian vegetation during the conversion of an area to pasture can decrease the ability of the landscape to retain nutrients, can increase the speed and volume of overland flow reaching the stream channel, and may increase water temperature by reducing the available cover. An increased nutrient load can drive a decrease in dissolved oxygen by increasing respiration, particularly of photosynthetic organisms. This effect may be exacerbated by the increase in temperature associated with reduced vegetative cover.

It is likely that animal feedlots and pasture lands used for grazing contribute to both the fecal coliform and biota impairments of the Groundhouse and South Fork Groundhouse watersheds.

3.2.2 Row Crop Agriculture

Approximately 38 percent of the land in the Groundhouse River watershed is used for production of corn and soybeans. Conventional tillage practices disturb the vegetation and upper layers of soil and lead to increased rates of erosion. Sediment transfer from cultivated lands to nearby waters is often much higher than other disturbed land uses, such as managed pasture or stabilized urban development.

In addition to increased rates of sediment loading, crop production may also be a source of pesticides, herbicides, and fertilizers to surface water systems. Based on the available biological and water quality data, impairments to the macroinvertebrate community have been linked to increased concentrations of total phosphorus and ammonia.

The watershed loading model indicates that crop production contributes approximately 30 percent of the total sediment load to the Groundhouse River and 47 percent of the sediment load to the South Fork. It is likely that crop production is a significant source of impairment to biological communities of the Groundhouse and South Fork Groundhouse rivers.

3.2.3 Streambank Erosion

Stream channels naturally change shape over time due to erosion of bank material and redeposition of sediments to the channel floor. In an undisturbed watershed, the process is in a state of dynamic equilibrium where the losses and gains are balanced. In watersheds where developed land uses such as agriculture or urban areas are present, the water balance shifts more towards runoff than infiltration to shallow groundwater zones. As runoff increases, so does the velocity and volume of water in the stream channel following precipitation events. The channel begins to erode in an attempt to reach equilibrium with the new flow regime. If development and storm water are not controlled, this can lead to excessive amounts of streambank erosion. In lower gradient segments of the channel, such as the section near Ogilvie, much of the upstream eroded material may redeposit in large quantities and impair aquatic communities.

Stream channels that lack riparian vegetation are more sensitive to changes in runoff flow patterns than well buffered segments. Rates of streambank erosion are typically higher where vegetation has been removed; habitat is also generally poor. In addition to managing runoff volumes from developed areas, replanting riparian zones should be a priority in the Groundhouse watershed. A well buffered stream also limits cattle access, which causes severe trampling and sloughing of bank material into the channel.

The analysis presented in Section 4 estimates that streambank erosion contributed 54 percent of the sediment load to the mainstem of the Groundhouse River and 39 percent to South Fork Groundhouse. This erosion, whether due to runoff events or cattle access, likely impairs the macroinvertebrate and fish communities in both watersheds. Impairments to the fish community near Ogilvie may be a direct result of sediment load originating from channel and landscape erosion further upstream. The deposition of sediment is a result of the low gradient characteristics of this reach in comparison to the higher gradient characteristics immediately upstream.

3.2.4 Onsite Wastewater Treatment Systems

Onsite wastewater treatment systems are not typically a significant source of pollutant loading if they are operating as designed. However, if the failure rate of systems in this watershed is high, then the loading from this source may be significant. At this time, the database of onsite wastewater treatment systems is incomplete for the Groundhouse River watershed, so it is difficult to estimate levels of performance.

In a properly functioning septic system, wastewater effluent leaves the septic tank and percolates through the system drainfield. Fecal coliform concentrations are typically reduced by 99.99 percent (Siegrist et al., 2000). Failing systems which short circuit the soil adsorption field result in ponding on the ground surface, or backup into homes that will have concentrations typical of raw (untreated) sewage. Direct discharge systems that intentionally bypass the drainfield by connecting the septic tank directly to a waterbody or other transport line (such as an agricultural tile drain) will also have concentrations similar to raw sewage.

According to the estimates of delivered fecal coliform loads, onsite sewage treatment systems contribute 4 to 15 percent of the load to the listed segments in the Groundhouse watershed. The impact on fine sediment loading is likely insignificant.

3.2.5 Wildlife and Domestic Pets

Domestic pets, such as cats and dogs, and wildlife, such as deer, geese, and ducks, can be significant sources of pollutant loading in watersheds that have high densities of urban populations or rural communities with relatively undisturbed land use patterns. In the Groundhouse River watershed, where the majority of land is used for agricultural uses, these sources are likely not significant relative to the loading from animal operations and failing onsite wastewater treatment systems. The total delivered load from wildlife and pets is estimated to range from 0.4 to 1.1 percent in this watershed.

3.2.6 Power Right-of-Way

USEPA (2004a) listed the power line right-of-way located along the mainstem of the Groundhouse River as a potential source of impairment. According to the report, the vegetation along this right-of-way may be treated with herbicides and pruned in order to prevent interactions and complications between the plants and the power lines. These herbicides may then enter the Groundhouse River, adversely affecting the biota. Additionally, the lack of woody vegetation along the right-of-way may contribute to bank destabilization increasing sloughing and erosion. Recent pesticide data collected on the mainstem and South Fork of the Groundhouse River indicate that the power line right-of-way is not contributing significant levels of herbicides. The lack of woody vegetation, however, is most likely leading to increased bank sloughing, thereby increasing the sediment load to the river and potentially contributing to the biota impairment.

4 Estimation of Source Loads

A watershed model and other tools were used to quantify the potential loads of sediment and fecal coliform from the various sources identified in Section 3 of this report. There are a variety of approaches for developing watershed-based pollutant loading models, ranging from simple export coefficient models to complex hydrodynamic models. For the Groundhouse River TMDLs, the use of a watershed model that falls between the simple and complex level was considered appropriate and the Generalized Watershed Loading Function (GWLF) model (Haith et al., 1992) was selected. The complexity of GWLF falls between that of detailed simulation models, which attempt a mechanistic, time-dependent representation of pollutant load generation and transport, and simple export coefficient models, which do not represent temporal variability. GWLF provides a basis to estimate pollutant load allocations by addressing overland runoff and groundwater discharge into streams. Separate estimates of streambank erosion were made using literature values of bank erosion rates and regional curves estimating channel cross sectional area. A post-processing spreadsheet model was used to incorporate loading due to point sources and onsite wastewater treatment.

It should be noted that the output from the GWLF model and the estimates of streambank erosion are for total sediment loads, whereas it is only the fine sediment component of the total load that is believed to be primarily impairing aquatic life. Simulating the sand component of the total sediment load would have required a more advanced watershed model that would likely not have justified the additional expense as total sediment is believed to be an acceptable surrogate for fine sediment in this watershed. However, this distinction is important and should be considered during future implementation and monitoring efforts.

4.1 The GWLF Model

GWLF simulates runoff and streamflow by a water-balance method, based on measurements of daily precipitation and average temperature. Precipitation is partitioned into direct runoff and infiltration using a form of the Natural Resources Conservation Service's (NRCS) Curve Number method. The curve number determines the amount of precipitation that runs off directly, adjusted for antecedent soil moisture based on total precipitation in the preceding five days. A separate curve number is specified for each land use by hydrologic soil grouping. Infiltrated water is first assigned to unsaturated zone storage, where it may be lost through evapotranspiration. When storage in the unsaturated zone exceeds soil water capacity, the excess percolates to the shallow saturated zone. This zone is treated as a linear reservoir that discharges to the stream or loses moisture to deep seepage at a rate described by the product of the zone's moisture storage and a constant rate coefficient.

Flow in rural streams may derive from surface runoff during precipitation events or from groundwater pathways. The amount of water available to the shallow groundwater zone is strongly affected by evapotranspiration, which GWLF estimates from available moisture in the unsaturated zone, potential evapotranspiration, and a cover coefficient. Potential evapotranspiration is estimated from a relationship to mean daily temperature and the number of daylight hours.

Monthly sediment delivery from each land use is computed from erosion and the transport capacity of runoff, whereas total erosion is based on the universal soil loss equation (Wischmeier and Smith, 1978), with a modified rainfall erosivity coefficient that accounts for the precipitation energy available to detach soil particles (Haith and Merrill, 1987). Thus, erosion can occur when there is precipitation, but no surface runoff to the stream; delivery of sediment, however, depends on surface runoff volume. The basic processes addressed in the GWLF simulation are shown schematically in Figure 15. Actual implementation of the model made use of the Windows-based version known as BasinSim (Dai and Wetzel, 1999).

The GWLF application requires information on land use distribution, meteorology, and parameters that govern runoff, erosion, and nutrient load generation. In addition to the land use database, four primary data input classes are used to develop the model parameters for the watershed simulations:

1) soil and hydrologic properties
2) pollutant concentration, buildup, and runoff assumptions
3) onsite wastewater disposal information
4) meteorological data.

The land use, watershed delineations, population, septic numbers, and meteorology data were collected and processed to generate a 10-year time series (April 1996 – March 2006 meteorology), which was used to derive annual loading rates by land use.

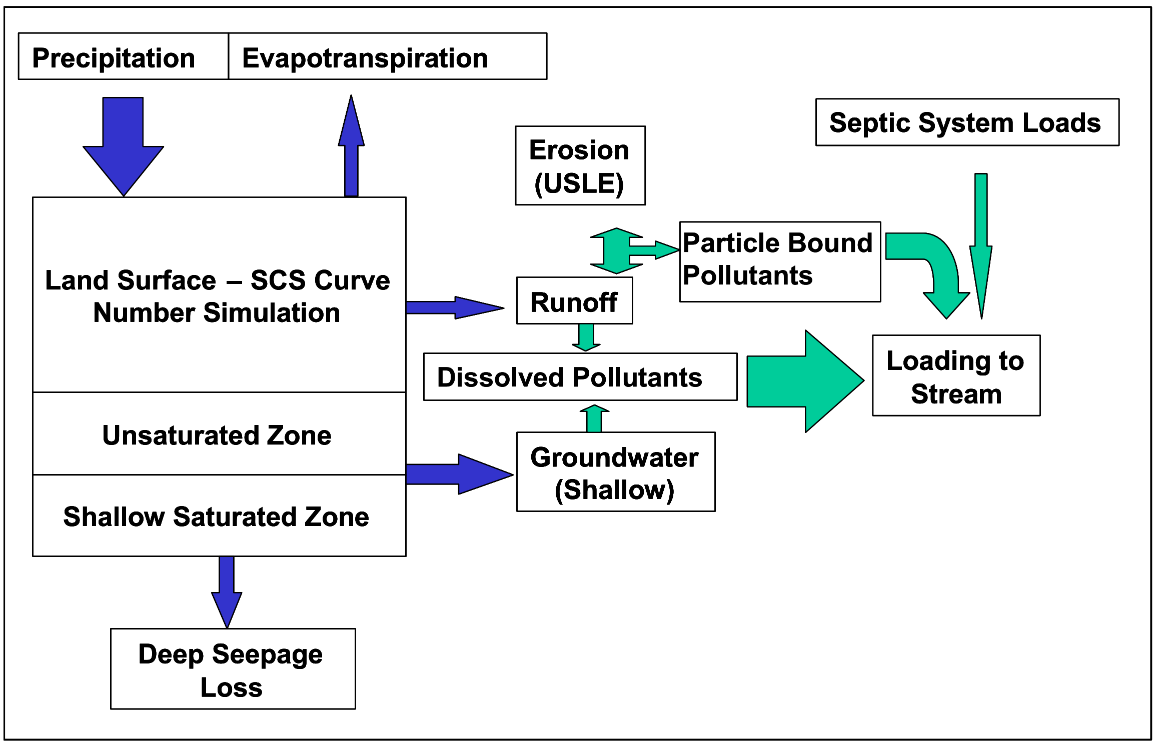

Figure 15. Schematic Representation of the GWLF Model

4.1.1 Watershed Boundaries

The Groundhouse River watershed has a drainage area of approximately 139 mi^2 where it joins the Snake River. The section of the Groundhouse River listed for aquatic life and recreation impairments is above the confluence with the South Fork Groundhouse River and has a drainage area of approximately 72 mi^2. The South Fork Groundhouse River is listed for impairment of aquatic life and has a drainage area of approximately 51 mi^2. Subwatershed boundaries for the Groundhouse River watershed were obtained from DNR. The watershed was divided into 10 subwatersheds with an average area of 14 mi^2. Table 5 lists each subwatershed by GWLF code, name, and drainage area, and Figure 16 shows the modeling subwatersheds.

Table 5. Description of the Groundhouse River Modeling Subwatersheds

Code	Name	Drainage Area (mi^2)
	Mainstem Groundhouse Watershed	
GRN1	Groundhouse River Below the Confluence with South Fork	15.4
GRN2	Groundhouse River Above the Confluence with South Fork	10.5
GRN3	Groundhouse River Above the Confluence with Unnamed Tributary 2	3.8
GRN4	Groundhouse River Above the Confluence with West Fork (Headwaters)	23.5
UTB1	Unnamed Tributary to the Groundhouse River	6.9
UTB2	Unnamed Tributary to the Groundhouse River	11.6
WFK1	West Fork Groundhouse River	15.9
Total Mainstem	**Headwaters of the Groundhouse River to Below the Confluence with the South Fork (excluding the South Fork drainage area)**	**87.6**
	South Fork Groundhouse Watershed	
SFK1	South Fork Groundhouse River Above the Confluence with Mainstem	8.0
SFK2	Unnamed Tributary to the South Fork Groundhouse River	4.7
SFK3	Headwaters of the South Fork Groundhouse River	38.5
Total South Fork	**Headwaters of the South Fork Groundhouse River to the Confluence with the Mainstem**	**51.2**

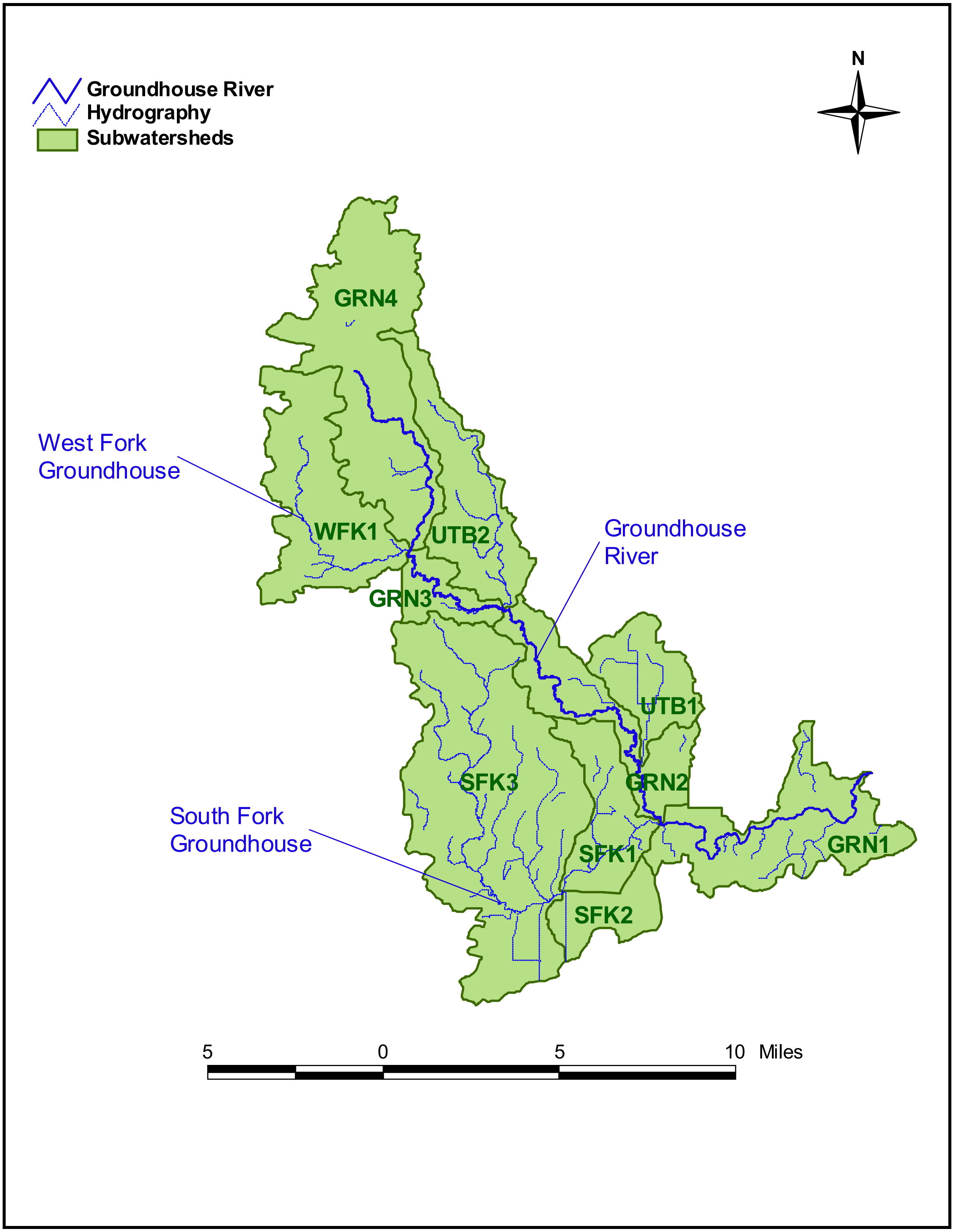

Figure 16. Groundhouse River Modeling Subwatersheds

4.1.2 Weather Data

Hydrologic simulation in GWLF is driven by daily precipitation totals and maximum and minimum daily temperatures. Potential evapotranspiration is calculated from temperature. The meteorological data required by GWLF was collected and processed for the meteorological stations at Isle, MN (Station 214103), Mora, MN (Station 215615), and Milaca, MN (Station 215392) to represent the range of conditions across the watershed (Figure 17). The raw data were obtained from the National Climatic Data Center for 1996 through 2006. Meteorological stations were assigned to subwatersheds as presented in Table 6.

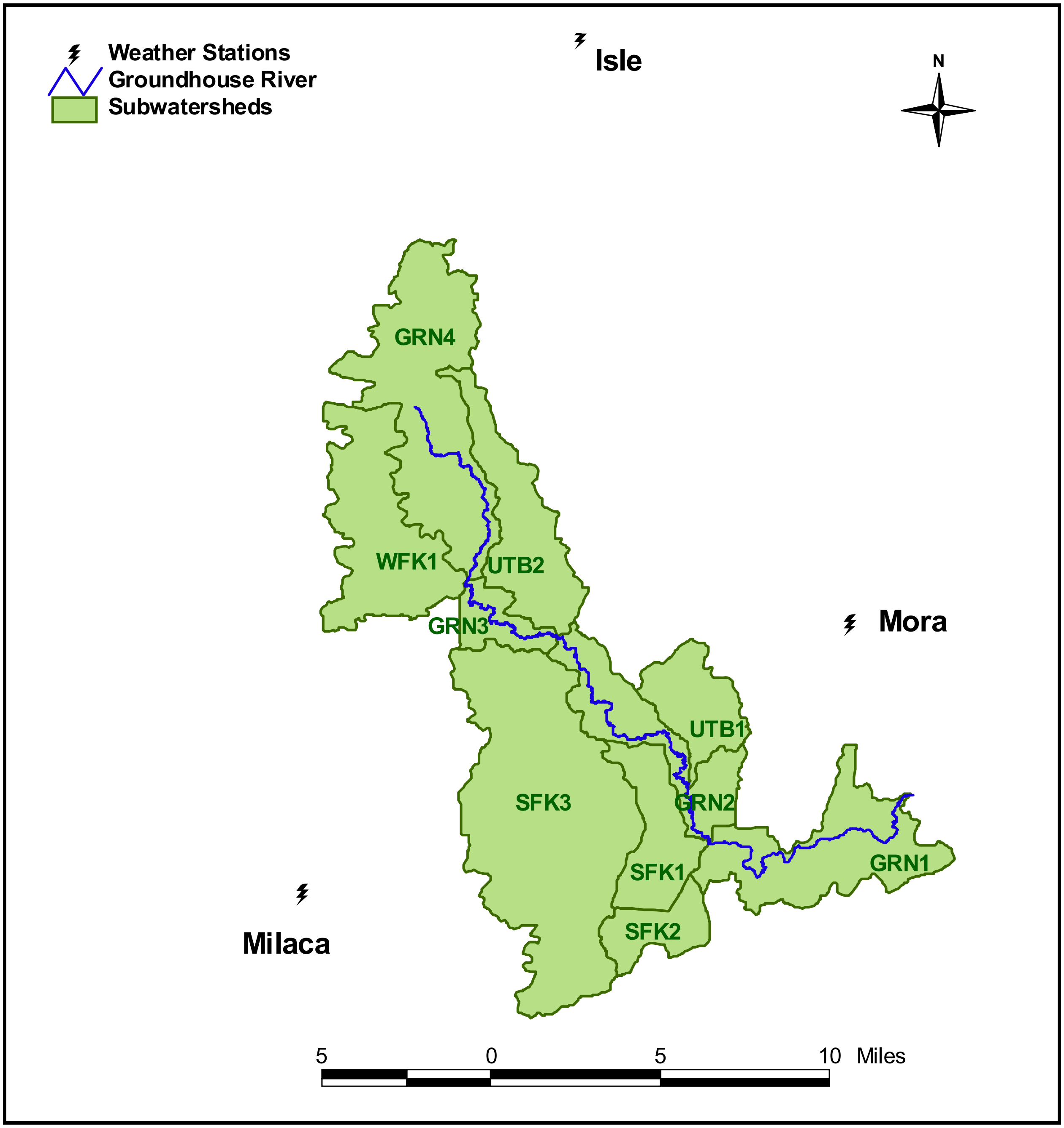

Figure 17. Location of Weather Stations in the Vicinity of the Groundhouse Watershed

Table 6. Assignment of Modeling Subwatersheds to Weather Stations

Subwatershed Code	Weather Station
GRN1	Mora
GRN2	Mora
GRN3	Isle
GRN4	Isle
SFK1	Milaca
SFK2	Milaca
SFK3	Milaca
UTB1	Mora
UTB2	Isle
WFK1	Isle

4.1.3 Overlay of Land Use and Soils Data

The first requirement for a watershed model is an accurate description of land use, land cover, and flow paths within the watershed. MPCA provided GIS coverages of University of Minnesota land use cover and NRCS State Soil Geographic (STATSGO) soils data. Supplemental datasets were obtained from the National Oceanic & Atmospheric Administration (NOAA), U.S. Geological Survey (USGS), Minnesota Department of Transportation (MNDOT), and the Kanabec and Mille Lacs County Soil and Water Conservation districts (SWCD).

The Minnesota 2000 land use data and STATSGO soils data were intersected in a GIS environment to define the land use soil combinations prevalent in the watershed. Combinations comprising less than 5 percent of any subwatershed area were aggregated into another combination with the same land use and similar soil properties. The aggregation resulted in 16 land use/soil combinations (Figure 18). Urban areas include paved and unpaved roads, gravel pits, and bare ground. These areas were post-processed outside of GIS and are not shown distinctly in the figure.

Supplemental information was acquired from the University of Minnesota 2000 impervious cover dataset, the Minnesota DOT road databases, and the 1992 National Land Cover Database (NLCD) from the Multi-Resolution Land Characterization (MRLC) Consortium (USGS, 2000) to better simulate roadways and urban and agricultural land uses. These datasets are further described in Appendix B, as are the soil and hydrologic parameters used during the setup of the model and the model calibration results.

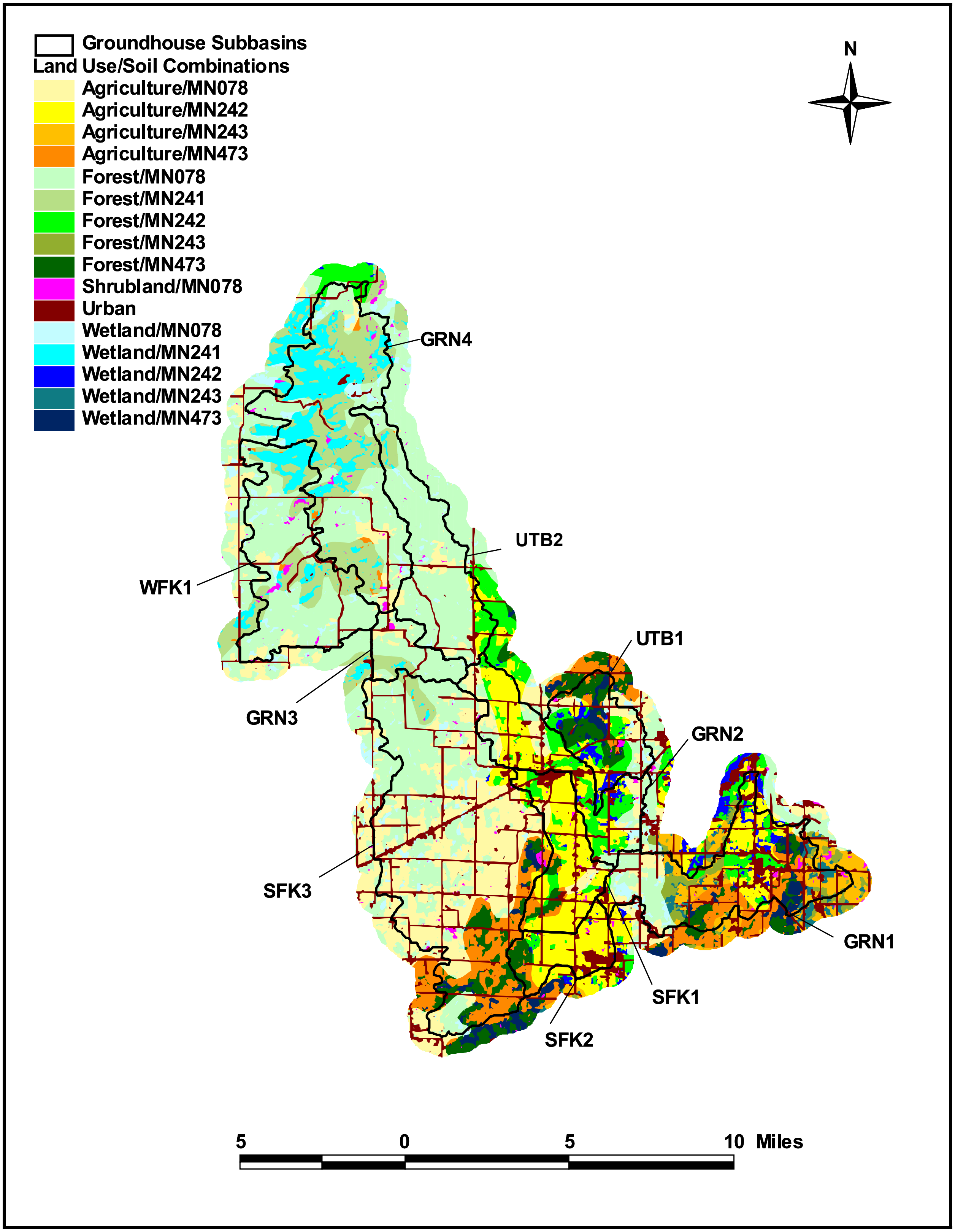

Figure 18. Aggregation of Land Use / Soil Combinations in the Groundhouse Watershed

4.1.4 Watershed Loading Results

Sediment loads were simulated from each subwatershed for the years 1996 through 2005. Table 7 presents the cumulative loads to each listed segment for each simulation year (cumulative loads to the Groundhouse mainstem do not include loads generated in the South Fork watershed since a separate TMDL is being developed for that portion of the watershed). The GWLF User's Manual (Haith et al., 1992) suggests that the first simulation year be discounted from the modeling results to provide a "spin up" period; the average shown in the table is for years two through ten only.

Table 7. Annual Upland Sediment Loads Simulated for the Listed Segments in the Groundhouse River Watershed

Simulation Year	Cumulative Sediment Load to the South Fork (US tons/yr)	Cumulative Sediment Load to the Mainstem (US tons/yr)
1996	1,602.7	1,003.8
1997	1,937.2	1,250.9
1998	2,322.7	1,844.7
1999	4,877.7	3,006.8
2000	2,398.9	1,982.6
2001	6,549.2	2,474.2
2002	6,433.9	4,530.7
2003	3,076.3	3,398.6
2004	3,708.7	2,883.3
2005	5,105.7	3,854.2
Average (1997 – 2005)	**4,045.6**	**2,802.9**

4.2 EXTERNALLY PROCESSED LOADS

Several additional sources of pollutant loading were analyzed outside of the GWLF model, including the Ogilvie WWTP, gravel pit operations, onsite wastewater disposal systems, streambank erosion, animal operations, and domestic and wildlife animals.

4.2.1 Ogilvie Wastewater Treatment Plant

The Ogilvie WWTP is the only wastewater treatment plant in the Groundhouse River watershed. The facility outfall is located above the State Highway 23 bridge. This facility is regulated under permit number MN0021997 and has a permitted discharge rate of 0.23 million gallons per day (MGD), a monthly fecal coliform geometric mean limit of 200 organisms per 100 mL, an average monthly TSS limit of 45 mg/L, and a weekly maximum TSS limit of 68 mg/L.

For the purposes of TMDL development, the permit limits are used for the waste load allocation. Permitted loads of TSS and fecal coliform are 15.8 US tons/year and 1,741 million organisms per day, respectively. The average monthly TSS permit limit of 45 mg/L was used to calculate the permitted load for TSS.

4.2.2 Gravel Pit Operations

The NLCD land use coverage was used along with maps and information provided by Minnesota DOT, Kanabec County Snake River Watershed Management Board, and the Mille Lacs County SWCD to locate active gravel pits in the Groundhouse watershed. The area of disturbed land at each site was traced on aerial photographs to estimate the area of gravel pit disturbance in each subwatershed (Table 8). The locations of the active gravel pits in the watershed are shown in Figure 19.

Table 8. Area Simulated as Gravel Pit

Subwatershed Code	Area (ac)	Percent of Subwatershed Area (percent)
Mainstem Groundhouse Watershed		
GRN1	17.0	0.17
GRN2	3.2	0.05
GRN3	0.2	0.01
GRN4	0.0	0.00
UTB1	0.0	0.00
UTB2	17.6	0.24
WFK1	0.0	0.00
Total Mainstem	38.0	0.07
South Fork Groundhouse Watershed		
SFK1	0.8	0.02
SFK2	116.4	3.86
SFK3	6.2	0.03
Total South Fork	**123.4**	**0.38**

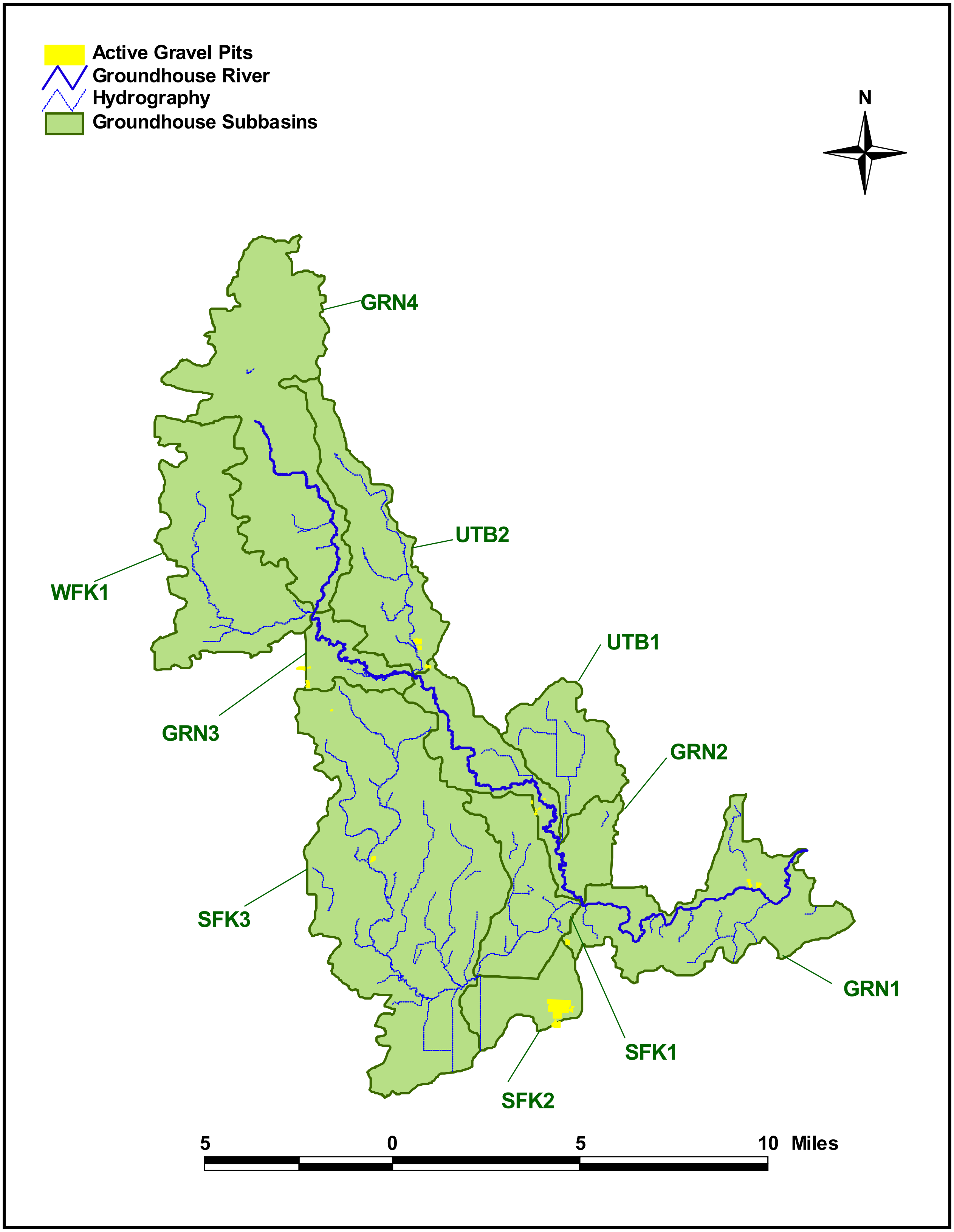

Figure 19. Location of Gravel Pits in the Groundhouse Watershed

Gravel pits in the Groundhouse River watershed are typically below grade operations that collect stormwater runoff in onsite ponds. These sites are not permitted to discharge stormwater from the collection ponds, but illegal pumping has occurred from some facilities in the past (Affeldt, 2006).

In 2004, EPA issued technical guidelines for gravel pit dewatering (USEPA, 2004b). The 30-day average TSS concentrations obtained by best practicable control technologies are listed at 25 mg/L; daily maximum concentrations are listed as 45 mg/L. These values were used in combination with GWLF estimates of runoff from gravel pits to estimate an average annual load for comparison with other sources. Gravel pits in the mainstem Groundhouse watershed are estimated to contribute 0.6 US tons per year of total suspended solids; contributions in the South Fork watershed are estimated as 1.9 US tons per year of total suspended solids.

For the TMDL, the wasteload allocation from these sources will be set to zero. Education and, when necessary, enforcement actions from MPCA will be relied upon to stop any illegal pumping.

4.2.3 Onsite Wastewater Disposal Systems

GWLF is capable of modeling conventional septic systems for the assessment of pollutant loading from onsite wastewater treatment. However, a database of permitted onsite systems along the Groundhouse River shows that 29 percent of permitted systems are mound systems, which provide a higher level of treatment. To simulate both types of onsite treatment systems, a spreadsheet analysis was used to estimate TSS and fecal coliform loading.

An estimate of the population served by onsite wastewater treatment systems was obtained by intersecting the subwatershed boundaries with the coverage of townships in Minnesota (obtained from the Minnesota DOT website). US Census data for the year 2000 was used to calculate the population density of each township. Township areas and population densities were then multiplied to estimate the population of each subwatershed. All residents within the town of Ogilvie were assumed served by the Ogilvie WWTP. The population served by onsite systems is summarized in Table 9.

Table 9. Population Served by Onsite Wastewater Treatment Systems in the Groundhouse Subwatersheds

Subwatershed	Population Served by Onsite Wastewater Treatment Systems
Mainstem Groundhouse Watershed	
GRN1	556
GRN2	291
GRN3	72
GRN4	156
UTB1	262
UTB2	165
WFK1	105
Total Mainstem	**1,607**
South Fork Groundhouse Watershed	
SFK1	138
SFK2	89
SFK3	787
Total South Fork	**1,014**

TSS and fecal coliform loading rates for the conventional and mound systems were estimated for both normal and failing conditions. A normal system is operating as designed; a failing system releases much higher rates of pollutants to the environment due to system malfunctions. An onsite failure rate of 20 percent was assumed for the Groundhouse River watershed based on information presented in the North Branch Sunrise Fecal Coliform TMDL (MPCA, 2006).

A database of onsite systems located in the shore zone of the Groundhouse River was provided by the Kanabec County SWCD. Of the 123 systems in the database, 45 systems were permitted (36 percent). For the permitted systems, the database included information concerning the age and type of system. No information about the non-permitted systems was available. Of the permitted systems, 29 percent were mound systems. Based on this information, it is assumed that 36 percent of the systems in the Groundhouse watershed are permitted and that 29 percent of these systems are mound type. All non-permitted systems are assumed conventional because the determination that an alternative system is required usually occurs during the permit process.

Table 10 summarizes the loading assumptions used to model the four classes of onsite wastewater treatment systems. The failing systems were assumed to have pre-absorption field concentrations. Normally functioning systems were then assumed to percolate through the soil absorption field. The concentrations are based on data presented at the 2000 Decentralized Wastewater Management Research Needs Conference (Siegrist et al., 2000). The resulting TSS loads to the mainstem and South Fork Groundhouse Rivers are 5.5 US tons/year and 3.4 US tons/year, respectively.

Table 10. Assumptions Used to Estimate TSS Loading Rates from Onsite Wastewater Treatment Systems

Parameter	Normal Conventional	Failing Conventional	Normal Mound	Failing Mound
Volumetric Loading Rate (gal/capita/day)	125	125	125	125
Hydraulic Losses (percent)	25	0	25	0
TSS (mg/L)	7.5	75	1.5	15
Fecal Coliform (#/100 mL)	1,000	10,000,000	0.01	100

Fecal coliform loads that enter the environment do not necessarily reach a stream channel. Natural die off will occur on the land surface or groundwater zone. MPCA (2006) uses a delivery factor approach to estimate the fraction of loading from each fecal coliform source that is expected to reach a stream. Failing septic systems are assumed to have a high delivery factor (0.04) while systems operating as designed are assumed to have a delivery factor of 0. Table 11 summarizes the estimated loads from onsite wastewater treatment systems from each subwatershed.

Table 11. Delivered Fecal Coliform Loads from Failing Onsite Wastewater Treatment Systems

Subwatershed	Fecal Coliform (million organisms/day)
Mainstem Groundhouse River	
GRN1	141,422
GRN2	73,929
GRN3	18,263
GRN4	39,664
UTB1	66,592
UTB2	41,991
WFK1	26,776
Total Mainstem	**408,637**
South Fork Groundhouse River	
SFK1	35,058
SFK2	22,659
SFK3	200,026
Total South Fork	**257,743**

4.2.4 Sediment Loads from Streambank Erosion

A simplified approach was used to estimate the contribution of sediment loading from eroding streambanks based on channel surface area, channel length, and literature values for bank erosion rates. A key input for this analysis is the estimation of channel width. Aerial photographs were used along with the GIS measurement tool to estimate the average width of channel at the midpoint of each subwatershed (Table 12). Only main channels were included in this analysis; it was assumed that the smaller channels would contribute proportionally less sediment at the subwatershed scale.

Table 12. Average Channel Width at the Midpoint of Each Subwatershed

Subwatershed Code	Average Width (ft)
GRN1	22
GRN2	22
GRN3	20
GRN4	8
SFK1	14
SFK2	6
SFK3	10
UTB1	8
UTB2	5
WFK1	10

The erosion rates utilized in this procedure were obtained from two studies concerning the influence of land use practices on streambank erosion in central and northeast Iowa (Zaimes et al., 2005; Zaimes et al., 2006). Both studies used erosion pins and measured bulk densities to determine erosion rates. Table 13 shows the erosion rates determined in each study. For the Groundhouse River watershed, bank erosion was calculated for both the minimum and maximum reported values in the Zaimes et al. (2006) study for each land use. The two land uses not included in the literature, wetland and urban, were assigned erosion rate ranges of 4 to 142 mm/yr (assumed equal to forest) and 60 to 464 mm/yr (assumed 20 percent higher than row crop), respectively.

Table 13. Published Streambank Erosion Rates (mm/yr) by Land Use

Reference	Land Use	Erosion Rate
Zaimes et al., 2005 (northeast Iowa)	riparian forest	4 to142
	row crop	50 to 387
	pasture	29 to 295
Zaimes et al., 2006 (central Iowa)	continuous pasture	17.1 (mean)
	rotational pasture	17.0 (mean)
	cattle excluded from	2.2 (mean)
	riparian forest	1.5 (mean)

In a 2005 study by Magner and Brooks, a strong relationship was developed between stream cross-sectional area (A_{xs}) and drainage area (DA) for east-central Minnesota. The developed regression equation takes the following form:

$$y = 4.9765x^{0.6721} \qquad (R^2 = 0.954)$$

where x equals drainage area (sq mi) and y equals stream cross-sectional area (sq ft).

Once the cross-sectional area was calculated from the cumulative drainage area at each subwatershed reach midpoint and the stream width was measured from GIS aerial photos, the bank height and subsequently the erodible bank area were calculated for each subwatershed (Table 14).

Table 14. Calculated or Measured Stream Geometries and Erodible Bank Areas

Subwatershed ID	Stream Length (m)	Total DA (mi^2)	A_{XS} (ft^2)	W (ft)	H (ft)	Bank Area (m^2)
GRN1	17,015	129.62	130.9	22	5.9	61,687
GRN2	23,000	60.08	78.1	22	3.5	49,730
GRN3	11,100	41.67	61.0	20	3.1	20,647
GRN4	12,885	17.64	34.3	8	4.3	33,627
SFK1	6,430	48.43	67.5	14	4.8	18,902
SFK2	3,280	2.36	8.9	6	1.5	2,953
SFK3	24,670	19.26	36.3	10	3.6	54,626
UTB1	7,260	3.45	11.4	8	1.4	6,330
UTB2	13,615	5.23	15.1	5	3.0	25,119
WFK1	13,330	9.52	22.6	10	2.3	18,382

Using STATSGO soils data provided by the NRCS, the average bulk densities were first weighted by proportion of soil classes within each MUID polygon. For each subwatershed, the bulk densities were further weighted by the proportion of the MUID's along the length of channel. The weighted bulk densities, in g/cm^3, are included in Table 15. Note the relatively low bulk density for subwatershed SFK2. The soil along this segment is primarily muck-peat.

Table 15. Weighted Soil Bulk Densities

Subwatershed	Weighted Bulk Density (g/cm^3)
GRN1	1.18
GRN2	1.38
GRN3	1.37
GRN4	1.21
SFK1	0.84
SFK2	0.30
SFK3	1.37
UTB1	1.05
UTB2	1.37
WFK1	1.37

Initially, the bank erosion losses were calculated using this range for both the minimum and maximum values from all the reported erosion rates (4 and 387 mm/yr), regardless of land use composition in each subwatershed. Due to the high variability in published bank erosion rates, an extremely high range of calculated bank erosion losses was estimated. In order to improve the accuracy of the bank erosion estimates, the erosion rates were weighted by land use adjacent to each stream segment. The results of each approach are shown in Table 16.

Table 16. Ranges of Estimated Bank Erosion Losses

Subwatershed ID	Minimum Rate of Bank Erosion (US ton/yr)	Maximum Rate of Bank Erosion (US ton/yr)	Lower Rates Weighted by Land Use (US ton/yr)	Upper Rates Weighted by Land Use (US ton/yr)
Mainstem Groundhouse Watershed				
GRN1	320.8	31,043.0	568.8	12,767.9
GRN2	302.0	29,175.7	1,193.8	15,618.5
GRN3	124.6	12,079.0	282.2	5,303.2
GRN4	179.7	17,407.5	598.5	8,689.4
UTB1	29.8	2,845.0	112.4	1,492.5
UTB2	152.1	14,694.8	313.1	6,288.6
WFK1	111.3	10,754.0	180.8	4,322.1
Total Mainstem	**1,220.3**	**117,999.0**	**3,249.6**	**54,482.3**
South Fork Watershed				
SFK1	69.4	6,750.5	508.2	4,856.7
SFK2	4.4	381.4	20.9	234.8
SFK3	330.7	31,956.8	2,081.1	21,193.9
Total South Fork	**404.5**	**39,088.7**	**2,610.2**	**26,285.4**

Several studies have reported percentages of bank erosion to total sediment load for the north-central US. Odgaard (1984) states that streambank erosion contributes 45 percent to 50 percent of the total sediment load in Iowa streams and 30 percent to 40 percent of the sediment load for two Iowa rivers (Odgaard, 1987). Wilkin and Hebel (1982) reported streambank erosion contributions up to 50 percent for two Illinois streams. Streambank slumping alone accounts for between 31 percent and 44 percent of the TSS load at the mouth of the Blue Earth River in Minnesota and has been reported to range between 17 percent and 92 percent based on a review of past studies (Sekely et al., 2002).

Based on the available literature, the actual bank erosion losses within the Groundhouse River are likely more accurately represented by the lower range of land use weighted erosion rates. In the mainstem and South Fork Groundhouse River watersheds, these estimates are approximately 54 and 39 percent, respectively, of the total sediment load (watershed loads plus streambank loads). In addition, using the lower end of the range (based on adjacent land use) is likely appropriate since Zaimes et al. (2005 and 2006) calculated bank erosion rates for incised streams in Iowa, and most of the stream channels within east-central Minnesota are well connected to their floodplain and riparian areas (Magner and Brooks, 2005).

4.2.5 Fecal Coliform Loading from Animal Operations

Agricultural animal operations typically have high animal populations and are a potentially large source of fecal coliform loading if adequate best management practices (BMPs) are not in place to protect surface waters. GIS coverages of state-registered (large scale) and non-registered (small scale) feedlots in the Groundhouse watershed were supplied by the county SWCDs. Figure 20 shows the spatial distribution of feedlots in the watershed, and Table 17 summarizes the animal units in each modeling subwatershed. Table 18 lists the number of animal units per head of animal as defined by Minnesota Rule 7020.0300 subpart 5.

Table 17. Summary of Animal Units for Each Modeling Subwatershed

Subwatershed Code	Dairy Cow (AU)	Beef Cow (AU)	Swine (AU)	Buffalo (AU)	Horse (AU)	Chicken (Layer) (AU)
GRN1	31	69	0	0	0	0
GRN2	41	168	256	0	2	0
GRN3	0	0	0	0	0	0
GRN4	0	339	0	0	0	0
SFK1	118	960	298	0	7	1
SFK2	62	138	0	0	0	0
SFK3	771	1,068	21	37	11	0
UTB1	11	24	0	0	0	0
UTB2	0	0	0	0	0	0
WFK1	311	178	1	0	10	0
Total	**1,312**	**2,876**	**576**	**37**	**30**	**1**

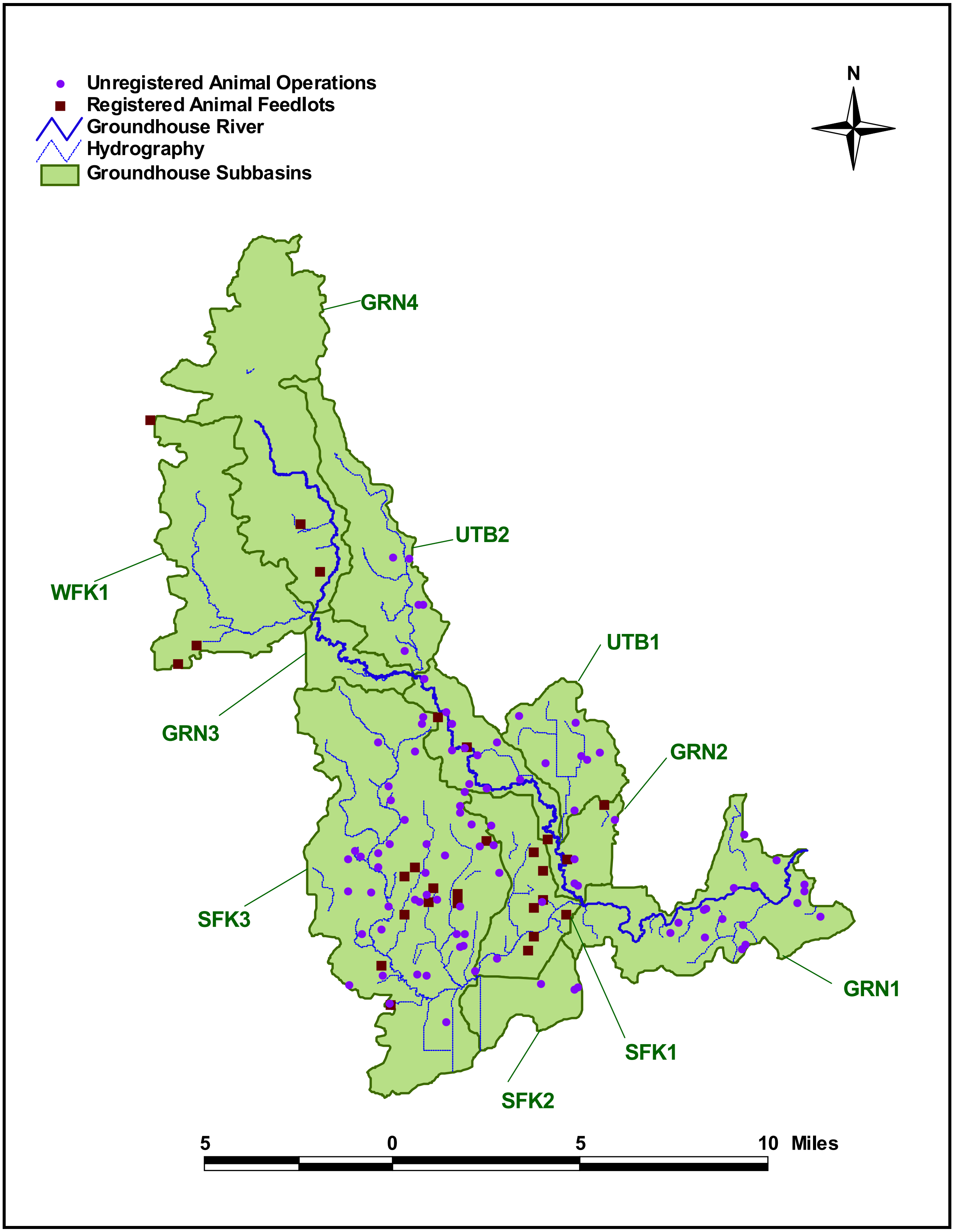

Figure 20. Location of Animal Feedlots in the Groundhouse Watershed

Table 18. Number of Animal Units Per Head by Animal Type

Animal Type	Number of Animal Units per Head
Dairy Cattle	
Mature cow (milked or dry) over 1,000 pounds	1.4
Mature cow (milked or dry) under 1,000 pounds	1
Heifer	0.7
Calf	0.2
Beef Cattle	
Slaughter steer/heifer, stock cow, or bull	1
Feeder cattle (stocker or backgrounding) or heifer	0.7
Cow and calf pair	1.2
Calf	0.2
Swine	
Over 300 pounds	0.4
Between 55 and 300 pounds	0.3
Under 55 pounds	0.05
Horse	1
Sheep or lamb	0.1
Veal	0.2
Chicken: Layer Hens or Broilers	0.033

Fecal matter deposited by animals in and around the stream is assumed a continuous source of fecal loading. Potential fecal coliform loading from animal operations in the Groundhouse watershed is estimated using a spreadsheet analysis that inputs the animal counts by subwatershed with daily loading rates reported by MPCA (2002) and ASAE (1998) (Table 19).

Table 19. Fecal Coliform Loading Rates by Animal Unit

Animal	Total Manure Production (lb/day per AU)	Fecal Coliform Loading (organisms/day per AU)	Loading Rate Source
Dairy cow	86	5.82E+10	(MPCA, 2002)
Beef cow/buffalo	58	8.91E+10	(MPCA, 2002)
Hog	84	3.27E+10	(MPCA, 2002)
Sheep	40	2.00E+11	(ASAE, 1998)
Horse	51	4.20E+08	(ASAE, 1998)
Chicken (Layer)	64	3.40E+10	(ASAE, 1998)

Similar to methods described in Mulla et al. (2001), this TMDL analysis utilizes fecal coliform delivery rates that describe water quality risk associated with manure production by livestock type and surface water proximity to each facility. To facilitate the calculation of numeric load allocations and to account for the fact that not all of the fecal material deposited by each animal will reach a stream channel, a percentage scale (4% = high to 0.1% = low) was used to describe fecal coliform delivery rates for various animal operations. Facilities with surface water on site have a high delivery factor (0.04), facilities with no surface water have a low delivery factor (0.001), and all unregulated operations for which surface water proximity is not known have a delivery factor of 0.02. Resulting fecal coliform loads from registered and unregistered cattle, swine, and horse operations are shown by subwatershed in Table 20. In both watersheds, beef cattle are expected to contribute the majority of loading from animal operations with dairy cattle ranking second.

Table 20. Delivered Fecal Coliform Loads from Agricultural Animals

Subwatershed	Dairy Cattle (million org/day)	Beef Cattle (million org/day)	Swine (million org/day)	Horses (million org/day)	Total (million org/day)
Mainstem Groundhouse Watershed					
GRN1	-	178,200	-	260	178,460
GRN2	40,740	352,551	23,936	294	417,521
GRN3	no known animal operations in this subwatershed				
GRN4	-	916,304	-	-	916,304
UTB1	-	62,370	-	101	62,471
UTB2	-	-	-	218	218
WFK1	722,844	74,844	13,080	-	810,768
Total Mainstem	**763,584**	**1,584,269**	**37,016**	**874**	**2,385,743**
South Fork Groundhouse Watershed					
SFK1	208,707	2,453,636	315,817	77	2,978,236
SFK2	-	356,400	-	34	356,434
SFK3	786,159	2,412,721	12,164	1,100	3,212,145
Total South Fork	**994,866**	**5,222,757**	**327,981**	**1,211**	**6,546,815**

4.2.6 Fecal Coliform Loading from Domestic and Wildlife Animals

Domestic animals such as dogs and cats may also contribute fecal coliform loads to surface waters. For comparison to other fecal coliform sources, the TMDL assumed one pet per household in the watershed based on best professional judgment and the lack of site-specific data. The number of households in each subwatershed is based on US Census data from the year 2000. Table 21 summarizes the number of pets assumed in each subwatershed. Fecal coliform loads for household pets are defined per animal, not animal unit as with agricultural animals.

Table 21. Pet Population in the Groundhouse Watershed

Subwatershed Code	Pet Population
GRN1	200
GRN2	164
GRN3	26
GRN4	60
SFK1	170
SFK2	32
SFK3	279
UTB1	93
UTB2	61
WFK1	40
Total	**1,125**

Wildlife such as deer, geese, and ducks are also potential sources of fecal coliform. The densities of wildlife populations in the Groundhouse watershed were provided by MN DNR (Pauly, 2006) and are summarized in Table 22. Table 23 summarizes the animal populations by subwatershed. Animal units are not defined for wildlife species.

Table 22. Wildlife Population Densities Assumed for the Groundhouse Watershed

Animal	Density (number/mi^2)
Deer	25
Geese	2
Ducks	5

Table 23. Wildlife Animal Population in the Groundhouse Watershed

Subwatershed Code	Deer	Geese	Ducks
GRN1	385	30	77
GRN2	263	20	53
GRN3	95	7	19
GRN4	588	46	118
SFK1	291	16	40
SFK2	200	9	24
SFK3	118	75	193
UTB1	963	13	35
UTB2	173	23	58
WFK1	397	31	79
Total	**3,473**	**271**	**694**

Fecal coliform loading rates from domestic animals and wildlife are based on data presented in the Lower Mississippi River Basin Fecal Coliform TMDL (MPCA, 2002). Loading rates by animal type are summarized in Table 24.

Table 24. Fecal Coliform Loading Rates for Domestic and Wildlife Animals

Animal	Loading Rate (organisms/animal/d)
Dogs	4.50E+09
Deer	5.00E+08
Geese	4.00E+08
Ducks	4.00E+08

Delivery factors for pets and animals were used to estimate the portion of fecal coliform loading that reaches a stream channel. Based on values presented by MPCA (2006), the delivery factor for animals is 0.01, for pets living outside the city is 0.001, and for pets living inside the city is 0.04. Table 25 summarizes the fecal coliform load delivered from non-agricultural animals in the Groundhouse watershed.

Table 25. Delivered Fecal Coliform Loads from Wildlife and Pets

Subwatershed	Deer (million org/day)	Geese (million org/day)	Ducks (million org/day)	Pets (million org/day)	Total (million org/day)
Mainstem Groundhouse Watershed					
GRN1	1,925	120	308	900	3,253
GRN2	1,315	80	212	11,462	13,069
GRN3	475	28	76	116	695
GRN4	2,940	184	472	272	3,868
UTB1	4,815	52	140	417	5,424
UTB2	865	92	232	276	1,465
WFK1	1,985	124	316	178	2,603
Total Mainstem	**14,320**	**680**	**1,756**	**13,620**	**30,376**
South Fork Groundhouse Watershed					
SFK1	1,455	64	160	21,980	23,659
SFK2	1,000	36	96	143	1,275
SFK3	590	300	772	1,258	2,920
Total South Fork	**3,045**	**400**	**1,028**	**23,380**	**27,853**

4.3 EXISTING SEDIMENT LOADS

The calibrated watershed model was used to estimate total sediment loads from land uses in the watershed. In addition, estimates of sediment loading from streambank erosion and total suspended solids (TSS) loading from onsite wastewater treatment systems, gravel pits, and the Ogilvie WWTP were accounted for externally. It is acknowledged that there is a great deal of uncertainty in these estimates, especially for the loads from streambank erosion.

Two segments in the Groundhouse watershed are listed for biota impairments that are mostly attributed to fine sediments. Though the GWLF model outputs the total sediment load from each potential source in the watershed, the percent contributions can be extrapolated to fine sediments because the coarser sediments likely redeposit on the land surface during transport, as simulated by the sediment delivery ratio (Appendix B).

The average annual total sediment load estimated to originate in the South Fork watershed is 6,661.1 US tons/per year. Figure 21 shows the estimated percent contribution of each source in the watershed. Only sources contributing more than 0.2 percent display in the pie chart. The 7,000 acres of row crop production contribute over 47 percent of the load, and streambank erosion contributes over 39 percent of the load. Lands classified as pasture make up most of the remaining load (over 9 percent).

The annual sediment load in the mainstem Groundhouse watershed is 6,074.4 US tons/yr. Figure 22 shows the percent contribution from the sources in the watershed that contribute more than 0.2 percent of the total load. Again, the majority of the sediment load originates from either streambank erosion (over 53 percent) or row crop production (approximately 30 percent) with nearly 10 percent from pasture lands.

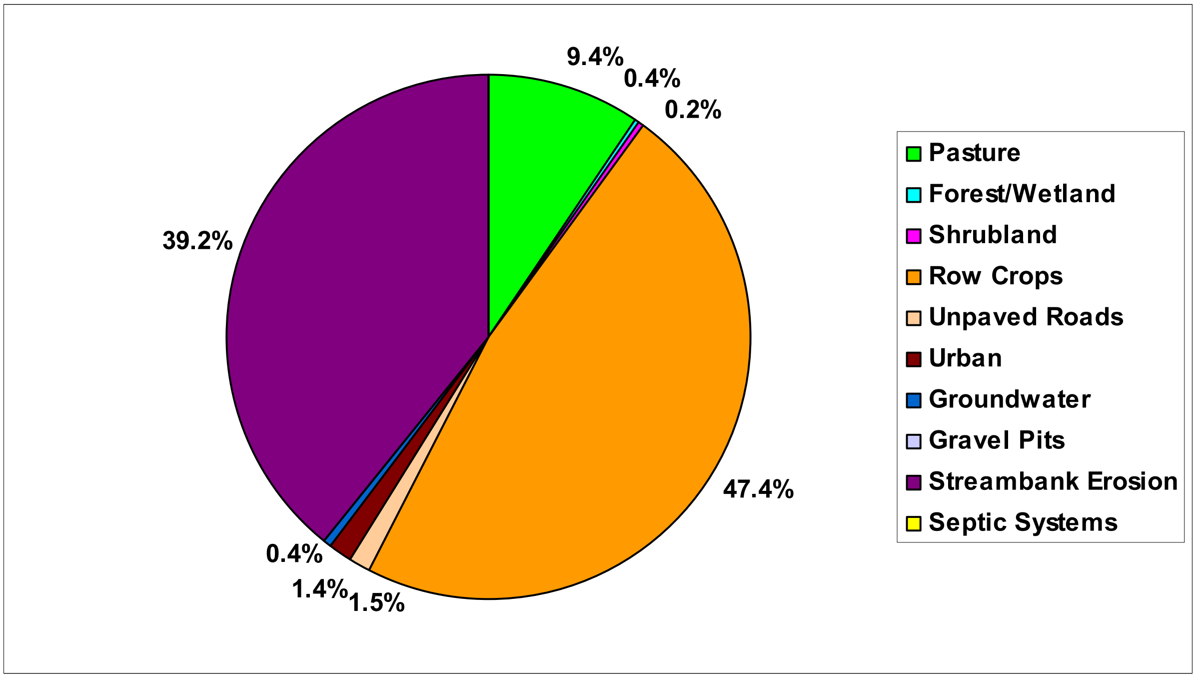

Figure 21. Percent Contribution of Sediment Sources in the South Fork Groundhouse Watershed

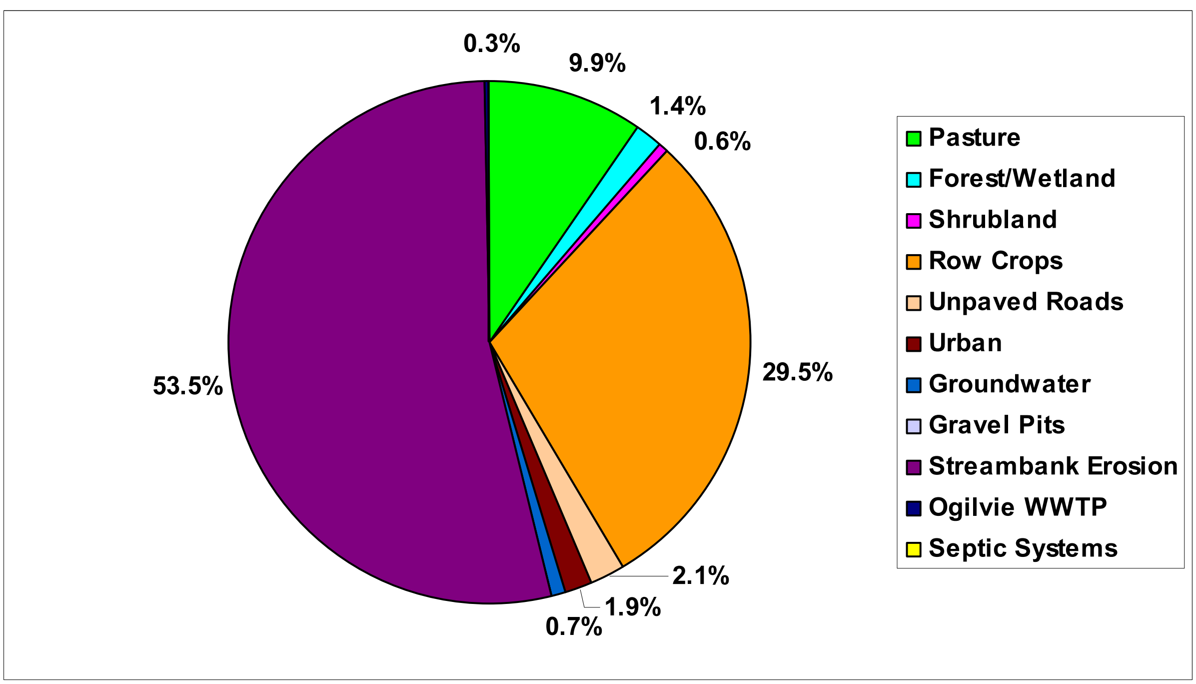

Figure 22. Percent Contribution of Sediment Sources in the Mainstem Groundhouse Watershed

4.4 Existing Fecal Coliform Loads

Fecal coliform loads from each major source were estimated using watershed data, literature values of fecal coliform loading rates, and a delivery factor approach developed by MPCA (2006). The estimated daily fecal coliform load delivered to the South Fork Groundhouse River is 6,832,411 million organisms per day, and the load delivered to the mainstem is estimated to be 2,826,497 million organisms per day. It is acknowledged that there is a great deal of uncertainty in the loading estimates for all sources of fecal coliform.

Figure 23 shows the estimated percent contribution of the fecal coliform sources in the South Fork watershed. Almost 96 percent of the delivered load likely originates from animal operations; onsite wastewater treatment systems make up most of the remaining load at just under 4 percent. The load from wildlife and pets is not significant.

Figure 24 shows the percent contribution from the sources in the mainstem watershed that contribute more than 0.2 percent of the total load. Again, the majority of the delivered fecal coliform load comes from animal operations (over 84 percent) with most of the remaining load from onsite wastewater treatment systems (over 14 percent). Wildlife and pets contribute approximately 1 percent of the delivered load. The load from the Ogilvie WWTP is only 0.06 percent of the total load and does not display on the pie chart.

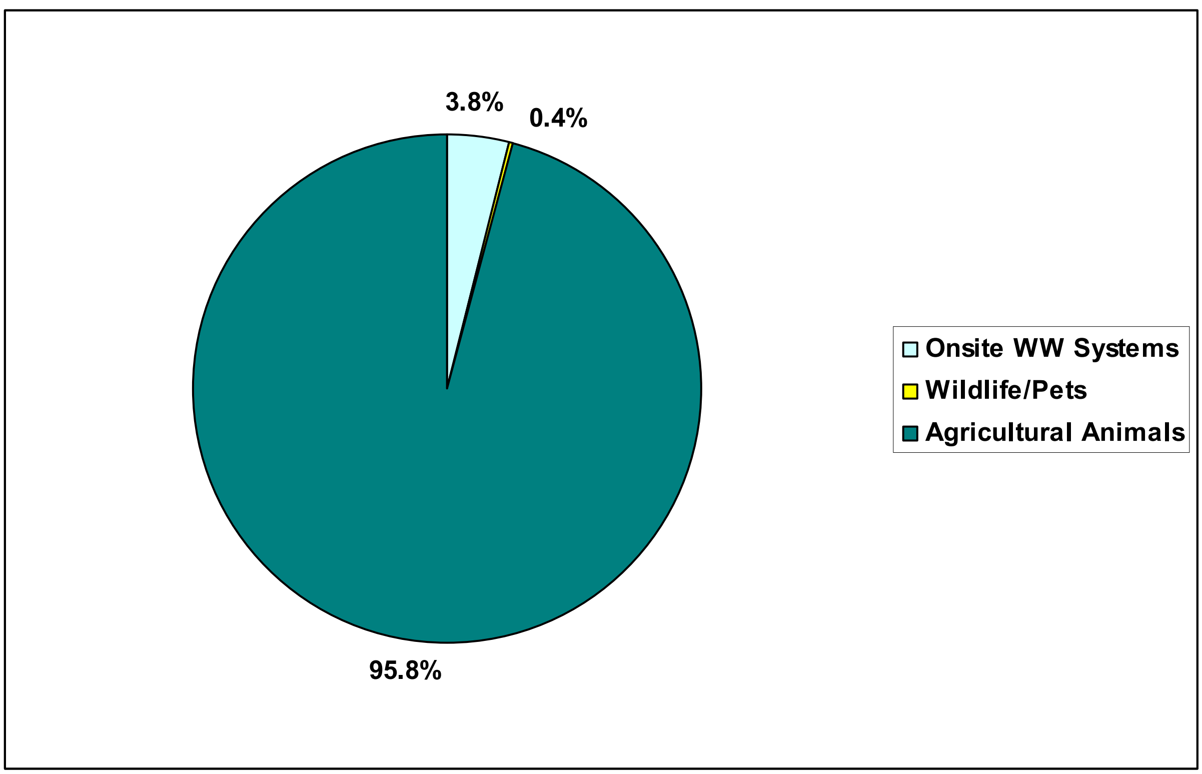

Figure 23. Percent Contribution of Fecal Coliform Sources in the South Fork Groundhouse Watershed

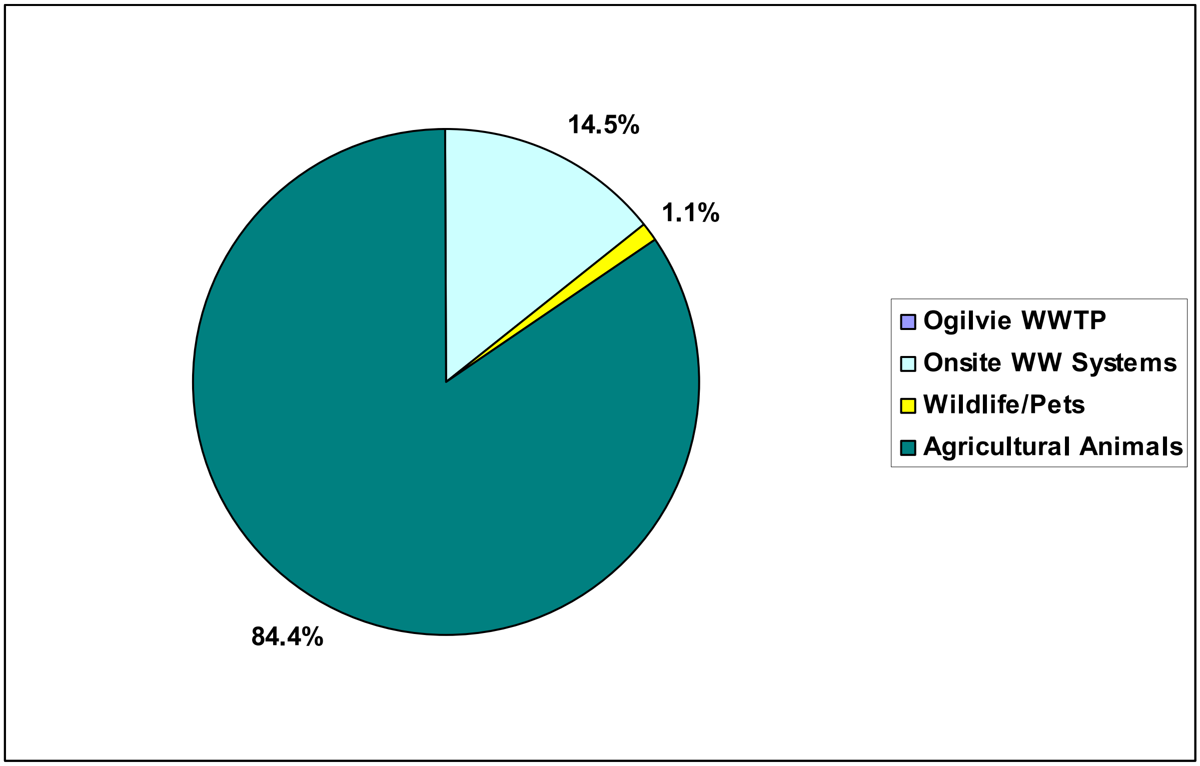

Figure 24. Percent Contribution of Fecal Coliform Sources in the Mainstem Groundhouse Watershed

5 TMDL Development and Determination of Allocations

A TMDL is the total amount of a pollutant that can be assimilated by the receiving water while still achieving water quality standards. TMDLs can be expressed in terms of mass per time or by other appropriate measures of loading rate. TMDLs are composed of the sum of individual wasteload allocations (WLAs) for point sources and load allocations (LAs) for nonpoint sources and natural background levels. In addition, the TMDL must include a margin of safety (MOS), either implicitly or explicitly, that accounts for the uncertainty in the relationship between pollutant loads and the quality of the receiving waterbody. Conceptually, this is defined by the equation:

$$\text{TMDL} = \text{WLA} + \text{LA} + \text{MOS}$$

This section of the report presents the TMDLs for the biota/sediment and fecal coliform impairments for the Groundhouse River watershed.

5.1 SEDIMENT

Excessive fine sediment deposited in low gradient stream segments has been identified as the primary stressor for macroinvertebrate and fish communities throughout the Groundhouse watershed. The most pronounced impairment of the fish community occurs upstream from Ogilvie and is also due to a combination of low gradient and excessive fines.

5.1.1 Technical Approach

Similar to most states, Minnesota does not have numeric water quality standards for TSS which could be directly used to quantify the allowable load of sediment in the Groundhouse River watershed. Furthermore, the data suggest that excessive fine sediment, rather than TSS, is the real cause of the fish and macroinvertebrate impairments. The TMDL therefore is based upon estimating the loads that correspond to natural background levels of fine sediments within the watershed. This approach is supported by Minnesota's regulations which state, in part, that *"…background levels may be used as the standards for controlling the addition of … pollutants from point or nonpoint source discharges in place of the standards."* [Minnesota Rules 7050.0170]

Few data exist to determine the natural background levels of fine sediment in the Groundhouse River watershed. However, it appears that a value of less than 25 percent is frequently associated with fish and macroinvertebrate IBI scores that are above the impairment thresholds (see Appendix D for details; note that there are several exceptions). Because GWLF (and most other models) cannot directly relate watershed loads to measures of in-stream fine sediment, best professional judgment was used to determine that all significant anthropogenic sources in the watershed need to be reduced to achieve the fine sediment target. Specifically, the following load reductions are recommended:

- Sediment loads from gravel pits and animal operations should be reduced to zero.
- Erosion from row crops should be decreased by 50 percent through the increased use of BMPs such as conservation tillage, cover crops, grassed waterways, and filter strips.

- Streambank erosion should be reduced to rates reported for natural watersheds in this geographic area, as reported in Zaimes et al. (2005 and 2006).

Reducing loads to these levels should result in achieving the fine sediment target, which then in turn should allow the streams to fully support their designated aquatic life uses.

5.1.2 Wasteload Allocations

WLAs are established for facilities with individual National Pollutant Discharge Elimination System (NPDES) permits. The only facility in this watershed permitted to discharge TSS is the Ogilvie WWTP. Based on the design flow and permit limits for TSS, the WLA for this facility is 15.8 US tons/year.

A WLA was also established for discharges from construction sites required to obtain an NPDES stormwater permit. A query of the MPCA Delta database in June 2008 indicates that there were 18 active NPDES stormwater permits for construction sites in Kanabec County. The total acreage of these sites covered approximately 0.135% of the land area of the county. Therefore, the WLA for construction stormwater will be set at 0.135% of the TMDL or 5.5 US tons/year.

Though other operations subject to permitting exist in the watershed, none are permitted to discharge solids or sediment to any waterbodies. For this reason, all WLAs for gravel pits, animal operations, and straight pipe dischargers have been set to 0.

5.1.3 Load Allocations

Load allocations for the Groundhouse River watershed represent the load identified after reducing anthropogenic sources to rates comparable to "natural" conditions minus the WLA established for the Ogilvie WWTP and permitted construction sites; the Load Allocations are shown in Table 26. The specific sources associated with the Load Allocations are described in Sections 3 and 4.

5.1.4 Margin of Safety

The margin of safety is accounted for implicitly for the sediment TMDLs. Existing loads are likely over-estimated because 1) existing BMPs are not currently accounted for and 2) the volume of water simulated by GWLF is over 7 percent higher than that measured at the Groundhouse gage. Because GWLF predicts sediment loads based on flow events, the estimated existing loads are likely conservative. In addition, the GWLF model predicts total sediment loads, where as the biota impairment is due primarily to the fine fractions of the sediment loads.

5.1.5 Critical Conditions and Seasonality

Critical conditions and seasonality for sediment loading to the Groundhouse River and South Fork Groundhouse River are primarily precipitation events that cause upland and streambank erosion. The surface water quality analysis (*Appendix A*) based on the duration curve approach used in this project related water quality to flow variability. TSS levels were consistently low at all stations and no violations of the water quality standard were observed. Elevated levels of TSS were associated with storm events. Because the GWLF model is a precipitation-based model, the existing and allowable sediment loading estimates account for critical conditions and seasonal variation.

5.1.6 Reserve Capacity

A reserve capacity to accommodate future increases in sediment loading from permitted sources is not included in this TMDL. The current TSS effluent limit for the Ogilvie wastewater treatment facility is 45 mg/L as a monthly average because the facility uses a trickling filter as the principal unit for biological treatment. This concentration is roughly equivalent to the 25 NTU turbidity water quality standard, and the organic solids that are discharged are subject to assimilation in the river. Finally, the most significant biological impairment occurred upstream from the wastewater outfall. Therefore, solids from the Ogilvie facility are not believed to be a significant contributor to the deposition of fine sediment in the river, the causative factor in the biological impairment found in the Groundhouse River system. If the future permitted design flow from this facility were to increase, TSS discharged would not be expected to contribute to impairment. Furthermore, the trickling filter at this facility is old and has at times been difficult to maintain. Thus, it is possible that a future upgrade of this facility to increase its design flow would see this unit replaced, resulting in the lowering of the permitted monthly limit to 30 mg/L TSS.

Reserve capacity for stormwater runoff from construction sites subject to permitting is not required because the level of construction activity in the watershed is not expected to significantly increase over time.

5.1.7 Sediment TMDLs

Table 26 and Table 27 summarize the TMDL components for the sediment loading to the Groundhouse and South Fork Groundhouse listed segments on an annual and daily basis.

Table 26. Sediment TMDL for the Groundhouse River

Component	Load (US ton/yr)	Load (US ton/d)
Current Load	6,074.4	16.64
TMDL= LA+WLA+MOS	4,203.5	11.51
LA	4,182.0	11.45
WLA: Facilities	15.8	0.04
WLA: Construction Sites	5.7	0.02
MOS	Implicit	Implicit
TMDL Reduction (percent)	30.8	30.8

Table 27. Sediment TMDL for the South Fork Groundhouse River

Component	Load (US ton/yr)	Load (US ton/d)
Current Load	6,661.1	18.25
TMDL= LA+WLA+MOS	4,036.6	11.06
LA	4,031.2	11.05
WLA: Facilities	0	0
WLA: Construction Sites	5.4	0.01
MOS	Implicit	Implicit
TMDL Reduction (percent)	39.4	39.4

5.2 FECAL COLIFORM

The Groundhouse River is classified as a Class 2B waterbody. The fecal coliform water quality standards for 2B waters state that fecal coliform concentrations shall not exceed 200 organisms/100 mL as a geometric mean of not less than 5 samples collected in one month, nor shall more than 10 percent of all samples taken during any month individually exceed 2,000 organisms/100 mL. These standards only apply to the recreation season which begins April 1 and ends October 31. The pending revision of Minnesota water quality standards will shift from the use of fecal coliform to *E. coli* in determining the acceptability of waters for recreational use.

This TMDL study focuses on the monthly geometric mean component of the fecal coliform standard (200 organisms/100 mL) as opposed to the “acute” standard (2,000 organisms/100 mL) based on individual samples. It is believed that achieving the necessary reductions to meet the geometric mean component of the standard will reduce the exceedances of the acute standard, therefore complying with both parts of the water quality criteria. For comparison, Appendix E provides load duration curve results for both standards.

5.2.1 Technical Approach

This section of the report presents the technical approach used to estimate allowable loading of fecal coliform to the Groundhouse and South Fork Groundhouse rivers. As discussed below, a load duration approach was used to make these estimates. Appendix E contains the load duration curve results for each water quality station.

5.2.1 Load Duration Curves

Load reductions were determined through the use of load duration curves. This approach involves calculating the allowable loadings over the range of flow conditions expected to occur in the impaired stream by taking the following steps:

1. A flow duration curve for the stream is developed by generating a flow frequency table and plotting the data points to form a curve. The data reflect a range of natural occurrences from extremely high flows to extremely low flows.

2. The flow curve is translated into a load duration (or TMDL) curve by multiplying each flow value by the water quality standard/target for a particular contaminant, then multiplying by a conversion factor. The resulting points are plotted to create a load duration curve (LDC).

3. Each water quality sample is converted to a load by multiplying the water quality sample concentration by the average daily flow on the day the sample was collected. Then, the individual loads are plotted as points on the TMDL graph and can be compared to the water quality standard, or LDC.

4. Points plotting above the curve represent deviations from the water quality standard/target and the daily allowable load. Those plotting below the curve represent compliance with standards and the daily allowable load. Further, it can be determined which locations contribute loads above or below the water quality standard/target.

5. The area beneath the TMDL curve is interpreted as the loading capacity of the stream. The difference between this area and the area representing the current loading conditions is the load that must be reduced to meet water quality standards.

6. The final step is to determine where reductions need to occur. Those exceedances at the right side of the graph occur during low flow conditions, and significant sources might include septic systems, illicit sewer connections, or animals depositing waste directly to the stream; exceedances on the left side of the graph occur during higher flow events, and potential sources include a variety of activities related to runoff. Using the LDC approach allows the MPCA and local planners to determine which implementation practices are most effective for reducing loads based on flow regime. If loads are significant during wet weather events, implementation efforts can target those BMPs that will most effectively reduce storm water runoff.

An example load duration curve is presented in Figure 25 and illustrates that observed fecal coliform loads exceed allowable loads across all of the flow regimes. The figure indicates that excessive loads occur during high flow events and also when subsurface flows exceed surface flows. The proportion of surface versus subsurface flows was determined using the sliding-interval method for streamflow hydrograph separation contained in the USGS HYSEP program (Sloto and Crouse, 1996). Algorithms from HYSEP were incorporated into the load duration analysis to determine the proportion of daily mean discharge that was overland runoff (surface) or groundwater discharge (subsurface) components. A surface flow threshold value of 50 percent was used to identify water quality samples that were collected during primarily surface runoff events.

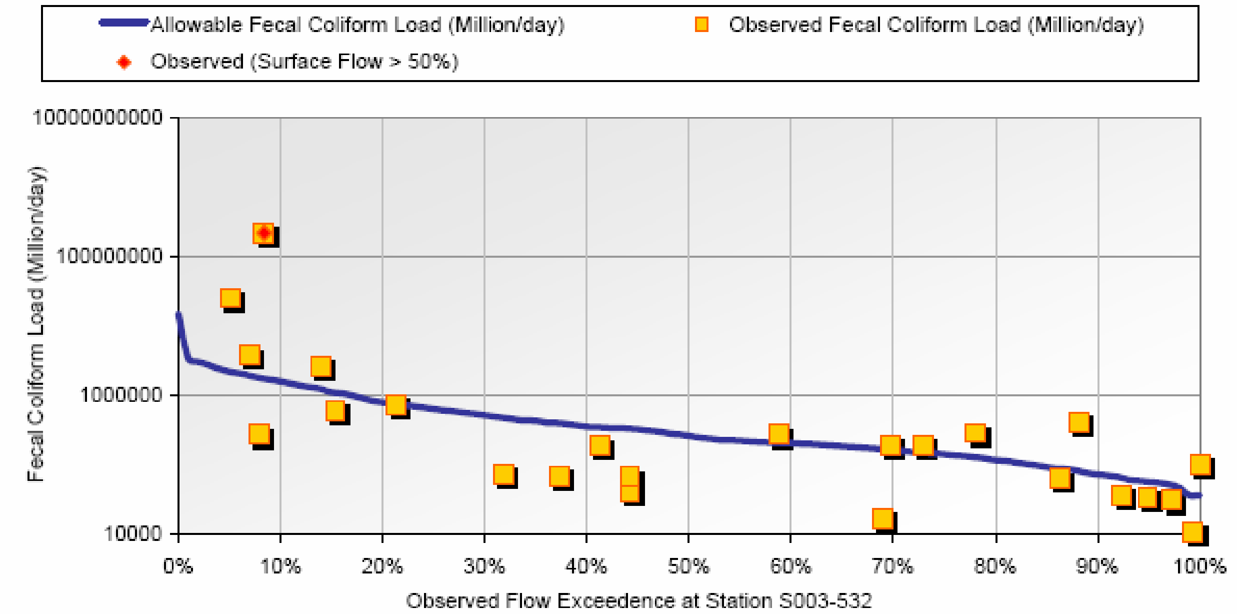

Figure 25. Fecal Coliform Load Duration Curve Example for the S003-532 Sampling Station Located on the Groundhouse River

The stream flows displayed on a load duration curve may be grouped into various flow regimes to aid with interpretation of the load duration curves. The flow regimes are typically divided into 10 groups which can be further categorized into the following five "hydrologic zones" (USEPA, 2007):

- High flow zone: stream flows that plot in the 0 to 10 percentile range, related to flood flows.
- Moist zone: flows in the 10 to 40 percentile range, related to wet weather conditions.
- Mid-range zone: flows in the 40 to 60 percentile range, median stream flow conditions.
- Dry zone: flows in the 60 to 90 percentile range, related to dry weather flows.
- Low flow zone: flows in the 90 to 100 percentile range, related to drought conditions.

The load reduction approach also considers critical conditions and seasonal variation in the TMDL development as required by the Clean Water Act and EPA's implementing regulations. Because the approach establishes loads based on a representative flow regime, it inherently considers seasonal variations and critical conditions attributed to flow conditions.

5.2.1 Stream Flow Estimates

Daily stream flows for each monitoring site of interest are needed to apply the load duration curve. However, there is only one stream flow gage on the Groundhouse River located at sampling station S003-532. This gage drains approximately 125 square miles and seasonal flows were collected April through October from 1999 to 2005 (with the exception of 2004). This flow gage was selected as a surrogate gage for extrapolating flows to other locations that do not have observed flows. Stream flows were extrapolated using the following equation:

$$Q_{ungaged} = \frac{A_{ungaged}}{A_{gaged}} \times Q_{gaged}$$

Where,

$Q_{ungaged}$: Flow at the ungaged location
Q_{gaged}: Flow at gage S003-532
$A_{ungaged}$: Drainage area of the ungaged location
A_{gaged}: Drainage area at gage S003-532

It is acknowledged that a longer period of flow data would be desirable for calculating the TMDLs in the Groundhouse River watershed. A longer period of record would ensure that the flows used to calculate the loading capacities include the full range of conditions likely to be observed in the watershed. Long-term data are not available, however, and approaches that would have generated such data (e.g., simulation modeling or use of a nearby surrogate gage with historical flows) have their own disadvantages. For example, modeled flows would only be estimates with potential errors, as would flow estimates made using a gage from a different watershed and a regression or drainage area weighting approach. Furthermore, precipitation data suggest that the period from 1999 to 2005 includes a representative range of annual rainfall conditions with 2002 being a fairly wet year and 2000 being a fairly dry year (Figure 26).

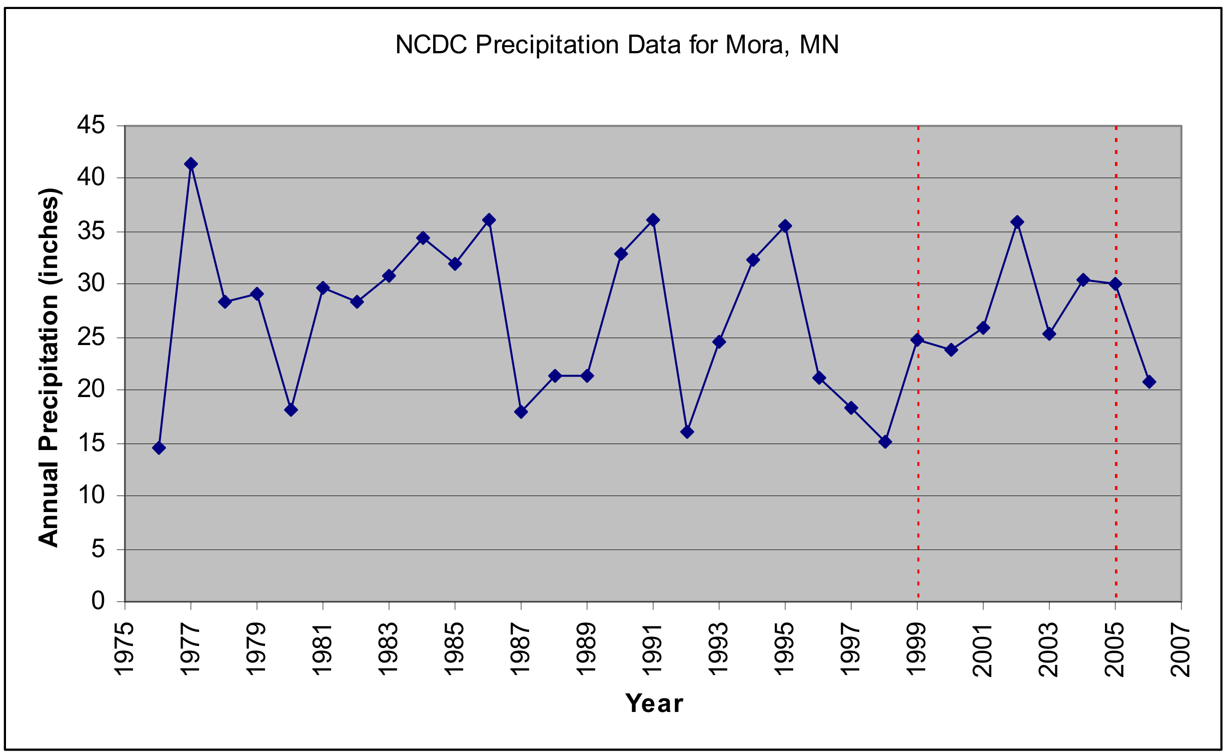

Figure 26. Annual precipitation data for the National Climatic Data Center station at Mora, MN.

To evaluate the potential problems caused by only using six years worth of flow data, the flow duration intervals for the Groundhouse River data were compared to those from the nearby Snake River. USGS gage 05338500 is located on the Snake River near Pine City, MN, or approximately 15 miles from the gage on the Groundhouse River near Mora, MN. The range of flow conditions at this gage should therefore approximate the range of flow conditions that occur in the Groundhouse River (e.g., drought conditions in the Snake River are likely to occur about the same time as drought conditions in the Groundhouse River).

Flow data are available for the Snake River gage for the period from 1958 to 2008. A comparison of the flow duration intervals from this long-term period of record for the Snake River and the flow duration intervals based on the six years of flow data in the Groundhouse River is shown in Table 28. The table indicates that the relative difference between the two sets of flow duration intervals is fairly consistent between flow duration intervals 0.01 and 0.95 (i.e., the Groundhouse flow duration intervals are usually about 15 to 20 percent less than the Snake River flow duration intervals). The differences are larger at extremely high flows and extremely low flows, which could be due to the shorter period of record for the Groundhouse gage but could also be due to differences in the size of the drainage areas or characteristics of the watersheds. (The Snake River gage drains 924 square miles and the Groundhouse gage drains 125 square miles).

If the differences between the flow duration intervals of the Snake and Groundhouse Rivers are due to the shorter period of record for the Groundhouse gage, the potential implications are that the fecal coliform loading capacities for extremely high and low flows could be biased. However, the vast majority of the loading capacities should be accurate, including those reported in the TMDL summary tables in Section 5.2.7 (flow duration intervals 0.05, .25, .50, 0.75, and 0.95).

Table 28. Comparison of Snake River and Groundhouse River recreational season flow duration intervals.

Flow Duration Interval	Snake River: 1958 to 2008 (cfs)	Groundhouse River: 1999 to 2003 and 2005 (cfs)	Percent Difference
0.000	14,200	3,099	21.82%
0.001	10,334	2,561	24.78%
0.003	9,550	2,151	22.52%
0.010	6,580	743	11.29%
0.050	3,119	480	15.39%
0.100	1,990	338	16.98%
0.150	1,530	241	15.75%
0.200	1,160	169	14.57%
0.250	887	137	15.45%
0.300	711	112	15.75%
0.350	576	91	15.80%
0.400	469	77	16.42%
0.450	388	69	17.78%
0.500	315	58	18.41%
0.550	263	47	17.87%
0.600	225	44	19.56%
0.650	196	40	20.41%
0.700	169	36	21.30%
0.750	146	31	21.23%
0.800	124	27	21.77%
0.850	104	22	21.15%
0.900	86	17	19.77%
0.950	63	13	20.63%
0.990	40	10	25.00%
0.999	31	8	25.81%
1.000	26	8	30.77%

5.2.2 Wasteload Allocations

The only facility in this watershed permitted to discharge fecal coliform is the Ogilvie WWTP. Based on the design flow (0.23 MGD) and permit limits for fecal coliform (200 organisms/100 mL), the WLA for this facility is 1,741 million organisms per day for all flow zones.

5.2.3 Load Allocations

The load allocation is the allocated load that originates from nonpoint sources and natural background. Therefore, the remaining capacity (after subtracting the WLA and MOS) is allocated to the LA. The use of the load duration approach results in flow-varying Load Allocations, which are summarized for each sampling station in the tables below. The specific sources associated with the Load Allocations are described in Sections 3 and 4.

5.2.4 Margin of Safety

The margin of safety (MOS) required in calculating a TMDL accounts for uncertainties in both characterizing current conditions and in the relationship between the load and wasteload allocations and in-stream water quality. The purpose of the MOS is to account for the uncertainty that the allocations will result in attainment of water quality standards.

An explicit MOS has been applied as part of all of the fecal coliform TMDLs by reserving five percent of the allowable load (see allocation tables). Five percent was considered an appropriate MOS based on the following considerations:

- The use of the load duration curve approach minimizes a great deal of uncertainty associated with the development of TMDLs because the calculation of the loading capacity is simply a function of flow multiplied by the target value. Most of the uncertainty is therefore associated with the estimated flows in each assessed segment which were based on extrapolating flows from the one existing flow gage.
- The fecal coliform TMDLs include an implicit MOS in that they were based on the geometric mean component of the standard rather than the not-to-exceed standard based on individual samples. Using the not-to-exceed standard would have resulted in larger loading capacities.

5.2.5 Critical Conditions and Seasonality

The Clean Water Act requires that TMDLs take into account critical conditions for stream flow, loading, and water quality parameters as part of the analysis of loading capacity. The analysis of water quality data (Sections 2.3.2 and 2.3.3) indicates that critical conditions, defined as those periods with the highest fecal coliform counts, primarily occur during high flow periods. Through the load duration curve approach, the allowable loads for every flow condition, including the critical conditions, are determined.

The allocation of point source loads (i.e., the WLA) also takes into account critical conditions by assuming the facilities will always discharge at their maximum design flows and permitted concentration limits. In reality, facilities typically discharge below design flows and display effluent quality that is better than their assigned effluent limits.

The Clean Water Act also requires that TMDLs be established with consideration of seasonal variations which are addressed in this TMDL by only assessing conditions during the season when the water quality standard applies (April through October). The load duration approach also accounts for seasonality by evaluating allowable loads on a daily basis over the entire range of estimated flows and presenting daily allowable loads that vary by flow. It is worth noting that fecal coliform typically do not exceed water quality standards in April or May (Figure 9).

5.2.6 Reserve Capacity

Reserve capacity refers to the load allocated for future growth. A reserve capacity to accommodate future increases in fecal coliform loading from permitted sources is not included in this TMDL. The major source of fecal coliform in the watershed is livestock, accounting for approximately 84 percent for the Groundhouse River and 96 percent for the South Fork Groundhouse River. While no trend information is available on livestock numbers, fecal coliform loads can reasonably be expected to decrease as residential development occurs in the watershed. Flows at the Ogilvie wastewater treatment facility may show small increases over time, leading to an eventual increase in the permitted design flow for the city's discharge; however, fecal coliform effluent limits are not set above the water quality standards, and as long as these limits are not exceeded, this source will not cause a water quality standards violation. With respect to individual sewage treatment systems, new systems will be constructed to serve new construction, and some systems at existing homes will be upgraded tending to tend to reduce loads overall. As population increases, the number of pets may also increase; however, the increase in fecal coliform load from pets is likely to be more than offset by the decrease expected from fewer livestock.

5.2.7 Fecal Coliform TMDLs

Table 29 through Table 37 summarizes the TMDL components for the fecal coliform loading to all of the sample stations with available data on the Groundhouse and South Fork Groundhouse rivers. Median flow values presented in the TMDL tables below were calculated using all available flow values for each flow zone. However, the loading capacities were calculated using flows that match the fecal coliform samples found to exceed water quality standards at each site.

Table 29. Fecal Coliform TMDL for the S003-532 Sample Station

TMDL Component (Million Org/Day)	Flow Zone				
	High (0-10) 456.6 cfs	**Moist (10-40) 131.9 cfs**	**Mid-Range (40-60) 54.1 cfs**	**Dry (60-90) 29.5 cfs**	**Low (90-100) 11.8 cfs**
TMDL= LA+WLA+MOS	1,939,815	1,256,368	212,394	141,452	36,732
LA	1,841,083	1,191,809	200,033	132,638	33,154
WLA: Ogilvie WWTP	1,741	1,741	1,741	1,741	1,741
MOS	96,991	62,818	10,620	7,073	1,837

Table 30. Fecal Coliform TMDL for the S001-097 Sample Station

TMDL Component (Million Org/Day)	Flow Zone				
	High (0-10) 258.9 cfs	Moist (10-40) 74.8 cfs	Mid-Range (40-60) 30.7 cfs	Dry (60-90) 16.7 cfs	Low (90-100) 6.7 cfs
TMDL= LA+WLA+MOS	1,099,953	437,885	120,436	90,511	25,185
LA	1,043,214	414,250	112,673	84,244	22,185
WLA: Ogilvie WWTP	1,741	1,741	1,741	1,741	1,741
MOS	54,998	21,894	6,022	4,526	1,259

Table 31. Fecal Coliform TMDL for the S001-099 Sample Station

TMDL Component (Million Org/Day)	Flow Zone				
	High (0-10) 222.9 cfs	Moist (10-40) 64.4 cfs	Mid-Range (40-60) 26.4 cfs	Dry (60-90) 14.4 cfs	Low (90-100) 5.8 cfs
TMDL= LA+WLA+MOS	947,095	613,409	168,642	80,498	21,685
LA	897,999	580,998	158,469	74,732	18,860
WLA: Ogilvie WWTP	1,741	1,741	1,741	1,741	1,741
MOS	47,355	30,670	8,432	4,025	1,084

Table 32. Fecal Coliform TMDL for the S001-152 Sample Station

TMDL Component (Million Org/Day)	Flow Zone				
	High (0-10) 212.6 cfs	Moist (10-40) 61.4 cfs	Mid-Range (40-60) 25.2 cfs	Dry (60-90) 13.7 cfs	Low (90-100) 5.5 cfs
TMDL= LA+WLA+MOS	929,150	585,066	157,000	71,582	20,683
LA	882,693	555,813	149,150	68,003	19,649
WLA: No Upst. Facilities	0	0	0	0	0
MOS	46,457	29,253	7,850	3,579	1,034

Table 33. Fecal Coliform TMDL for the S003-640 Sample Station

TMDL Component (Million Org/Day)	Flow Zone				
	High (0-10) 201.7 cfs	Moist (10-40) 58.3 cfs	Mid-Range (40-60) 23.9 cfs	Dry (60-90) 13.0 cfs	Low (90-100) 5.2 cfs
TMDL= LA+WLA+MOS	779,194	341,140	148,936	52,504	19,621
LA	740,234	324,083	141,489	49,879	18,640
WLA: No Upst. Facilities	0	0	0	0	0
MOS	38,960	17,057	7,447	2,625	981

Table 34. Fecal Coliform TMDL for the S003-639 Sample Station

TMDL Component (Million Org/Day)	Flow Zone				
	High (0-10) 40.4 cfs	Moist (10-40) 11.7 cfs	Mid-Range (40-60) 4.8 cfs	Dry (60-90) 2.6 cfs	Low (90-100) 1.1 cfs
TMDL= LA+WLA+MOS	171,635	111,163	29,830	10,516	3,930
LA	163,053	105,605	28,338	9,990	3,734
WLA: No Upst. Facilities	0	0	0	0	0
MOS	8,582	5,558	1,492	526	196

Table 35. Fecal Coliform TMDL for the S003-641 Sample Station

TMDL Component (Million Org/Day)	Flow Zone				
	High (0-10) 153.4 cfs	Moist (10-40) 44.3 cfs	Mid-Range (40-60) 18.2 cfs	Dry (60-90) 9.9 cfs	Low (90-100) 4.0 cfs
TMDL= LA+WLA+MOS	670,245	422,039	113,253	43,616	12,339
LA	636,733	400,937	107,590	41,435	11,722
WLA: No Upst. Facilities	0	0	0	0	0
MOS	33,512	21,102	5,663	2,181	617

Table 36. Fecal Coliform TMDL for the S003-638 Sample Station

TMDL Component (Million Org/Day)	Flow Zone				
	High (0-10) 176.3 cfs	Moist (10-40) 50.9 cfs	Mid-Range (40-60) 20.9 cfs	Dry (60-90) 11.4 cfs	Low (90-100) 4.6 cfs
TMDL= LA+WLA+MOS	748,924	485,059	133,355	48,589	18,257
LA	711,478	460,806	126,687	46,160	17,344
WLA: No Upst. Facilities	0	0	0	0	0
MOS	37,446	24,253	6,668	2,429	913

Table 37. Fecal Coliform TMDL for the S003-664 Sample Station

TMDL Component (Million Org/Day)	Flow Zone				
	High (0-10) 82.5 cfs	Moist (10-40) 23.8 cfs	Mid-Range (40-60) 9.8 cfs	Dry (60-90) 5.3 cfs	Low (90-100) 2.1 cfs
TMDL= LA+WLA+MOS	318,620	189,083	38,367	25,669	9,411
LA	302,689	179,629	36,449	24,386	8,940
WLA: No Upst. Facilities	0	0	0	0	0
MOS	15931	9454	1918	1283	471

Appendix E displays the fecal coliform load duration curve reports for the stations presented above. Two reports are provided for each station- the first set uses the 200 organisms/100 mL geometric mean standard (used in TMDL calculation), and the second set uses the standard for individual samples of 2,000 organisms/100 mL for comparison. The fecal coliform reports were created using an acute TMDL evaluation. This type of evaluation solely focuses on the fecal coliform samples that exceed water quality standards (e.g. for the reports using the 200 organisms/100 mL standard, TMDLs are based on all samples with fecal coliform concentrations exceeding 200 organisms/100 mL).

The "median observed flows" provided for each flow zone (high, moist, mid-range, dry, and low) in Appendix E are the median of all flows matching the exceeding fecal coliform concentrations. Because two different standards were used for the analyses, there are different numbers of exceeding samples for each report presented. Therefore, the two analyses (200 organisms/100 mL vs. 2,000 organisms/100 mL) at each station will also display different "median observed flow" values based on the different number of matching flows.

6 Implementation Planning

This section of the report provides an overview of BMPs that could be used to address the identified fecal coliform and sediment load reductions. Following approval of the TMDLs, a more detailed implementation plan should be developed by the local stakeholders with assistance from MPCA and using the results of the TMDL study.

6.1 Best Management Practices (BMP)

Controlling pollutant loading to the impaired reaches of the Groundhouse watershed will require implementation of various BMPs. This section lists the BMPs which may be used to reduce loading of sediment, TSS, or fecal coliform from point source dischargers, onsite wastewater treatment systems, agricultural operations, and streambank erosion.

For further information on these BMPs, their effectiveness, and the costs associated with them; information can be found in Attachment F of this document.

- Proper Maintenance of Onsite Wastewater Treatment Systems
- Conservation Tillage
- Cover Crops
- Filter Strips
- Grassed Waterways
- Riparian Buffers
- Controlled Drainage
- Wetland Restoration
- Constructed Wetlands
- Sedimentation Basins
- Proper Manure Handling, Collection, and Disposal
- Composting
- Alternative Watering Systems
- Cattle Exclusion from Streams
- Grazing Land Management
- Stream Bank Erosion BMPs
- Stream Habitat Restoration

6.2 Prioritization of BMPs

This section of the report summarizes the BMP information by source category and offers some considerations with regard to prioritization of BMP implementation. Figure 27 and Figure 28 show aerial photographs for the upper and lower half of the Groundhouse River watershed to illustrate the location of crop land, animal operations, and lack of riparian buffers.

6.2.1 Failing Onsite Wastewater Treatment Systems

Pollutant loads associated with failing onsite wastewater treatment systems likely contribute 4 to 15 percent of the fecal coliform load to the listed segments in the Groundhouse watershed. Reducing the number of failing systems will require ongoing education of system owners, periodic inspections, regular maintenance, and replacing systems when needed. These measures were discussed together in Appendix F, Section 1.1.1.

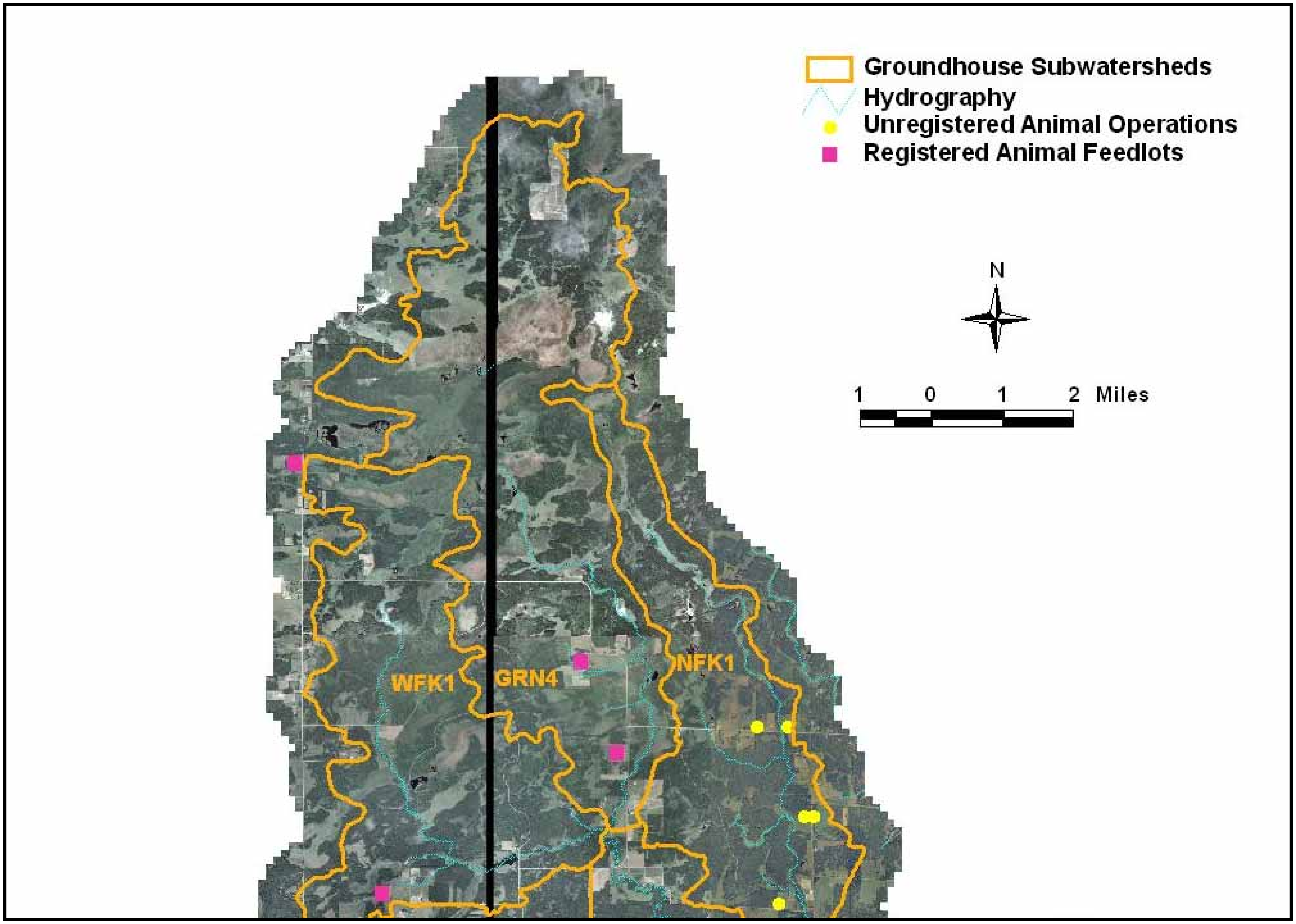

Figure 27. Upper Half of the Groundhouse River Watershed

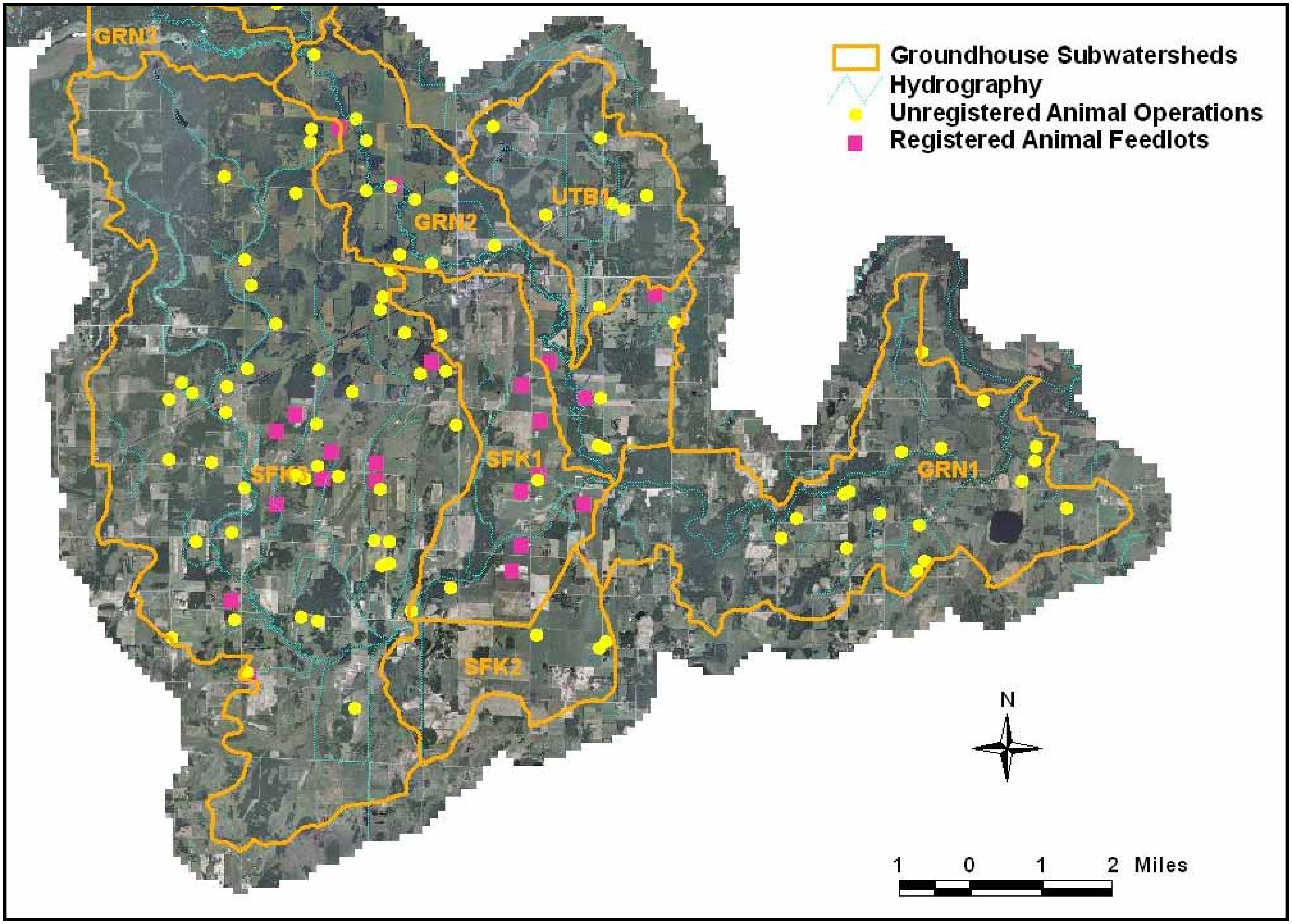

Figure 28. Lower Half of the Groundhouse River Watershed

6.2.2 Animal Operations

The BMPs that are applicable to agricultural operations in the Groundhouse River watershed are summarized in Table 39 and include the percent reductions for five parameters as well as additional information concerning streambank protection and impacts on dissolved oxygen. Managing pollutant loading from animal operations will likely be necessary to meet the TMDL requirements for the listed segments. The effectiveness of BMPs applicable to animal operations is summarized in Table 40.

Fecal coliform and sediment impairments occur throughout the Groundhouse watershed. Management strategies at animal operations should therefore focus on BMPs that address both issues. Excluding cattle from streams either by fencing or the creation of riparian buffers should be a top priority to reduce streambank erosion and fecal coliform loading. Use of constructed wetlands, filter strips, and grassed waterways are also effective. Manure composting is highly effective at reducing fecal coliform concentrations, but may be more expensive than the other options.

The only subwatershed that contains a fecal coliform sampling station that did not observe excursions of the fecal coliform standard was subwatershed UTB2. Each of the other monitored subwatersheds (GRN1, GRN2, GRN3, SFK1, and SFK3) had excursions of either the geometric mean standard or the instantaneous standard. Based on the water quality, biota, and GIS data, prioritization of animal BMPs is as follows:

- Subwatersheds GRN2 and SFK3 have the highest geometric mean and instantaneous fecal coliform concentrations in the watershed. They also have the lowest fish and macroinvertebrate scores in the watershed (with the exception of the low fish scores in Impaired Area 2). These subwatersheds should have the highest priority for implementation of BMPs at animal operations. Those facilities located adjacent to a stream or river should exclude cattle from the stream channels and institute manure management practices. Facilities with no surface water onsite, or those that already exclude cattle, should focus on properly handling, storing, and disposing of manure.

- Subwatershed SKF1 has the highest density of animal units per square mile in the watershed. Fish scores are good to excellent in this watershed, but macroinvertebrate scores are typically poor to fair. Though excursions of the geometric mean fecal coliform standard have been observed, the values are not as high as those measured in GRN2 and SFK3. Further implementation of BMPs at animal operations will likely be required to eliminate excursions of the fecal coliform standard and to improve macroinvertebrate scores. This watershed should be considered a medium priority for implementation, ranking below GRN2 and SFK3.

- Subwatershed GRN1 has seen excursions of the geometric mean standard, though technically none of the excursions are based on five or more samples within the same month. The monitoring station in this subwatershed is located upstream of the 17 unregistered feedlots in this drainage. Examination of the flow duration curves indicates that fecal coliform loading at this site originates from sources in upstream subwatersheds. Fish and macroinvertebrate scores in this watershed are above thresholds. Determining if the feedlots in this subwatershed require implementation of BMPs will require collection of fecal coliform data near the outlet of the drainage or inspection of each facility. Based on the current information, this subwatershed is a lower priority for the implementation of BMPs at animal operations.

- Excursions of the geometric mean fecal coliform standard have been observed in subwatershed GRN3, and macroinvertebrate scores are poor to fair; fish scores rank good to excellent. No animal operations are known to exist in this subwatershed, though four registered feedlots are located in the two upstream subwatersheds (WFK1 and GRN4). Fecal coliform data has not been collected in these subwatersheds, so prioritizing BMP implementation for animal operations is difficult. According to the registered animal feedlot database, only one of the four feedlots in this drainage has either a manure stockpile area or a manure storage basin. Inspection of these animal operations is necessary to determine where additional BMPs may be needed.

- No excursions of the fecal coliform standards were observed at the water quality station in the UTB2 subwatershed, and fish and macroinvertebrate scores are above thresholds. There are currently five unregistered feedlots in this subwatershed. Implementing additional BMPs at these facilities is a lower priority.

- Subwatershed UTB1 has seven unregistered feedlots and the streams and drainage ditches are poorly buffered. No fecal coliform or biota data has been collected in this subwatershed. Facilities in this drainage should be inspected to determine if manure management BMPs and cattle exclusion are likely necessary to protect downstream water quality.

6.2.3 Crop Production

Lands used for crop production contribute sediment to the impaired waters and result in reduced macroinvertebrate scores. Figure 29 shows an example of gully erosion from an agricultural field in the watershed. In addition, application of fertilizers and pesticides may contribute excess nutrients and toxic chemicals to nearby waterbodies. Table 41 summarizes the crop production BMPs that will most efficiently reduce sediment loads from this source.

(Photo courtesy of KCSWCD)

Figure 29. Gully Erosion from Agricultural Field

As sediment is the primary pollutant of concern from land used for crop production, cost effective BMPs for sediment reduction should be prioritized for implementation. Conservation tillage practices offer the best reductions in terms of costs. Given that the impairments related to crop production occur throughout the watershed, encouraging conservation tillage practices should be a top priority. Other effective measures include grassed waterways, cover crops, and filter strips. Riparian buffers are highly effective and will also benefit aquatic organisms by providing shading, habitat, and food sources. The impact of storing water behind outlet control devices on the erosive forces exerted in receiving stream channels has not been quantified. A higher priority for implementing these BMPs should be given to those fields draining directly to an unbuffered stream or drainage ditch.

Improving fish and macroinvertebrate scores in the watershed will require implementation of crop production BMPs. Subwatersheds may be ranked for priority based on biota scores and land uses adjacent to surface waterbodies:

- The highest priority subwatershed for implementing crop BMPs is SKF1. Crop production makes up 39 percent of the land in this subwatershed and macroinvertebrate scores range from poor to fair. Subwatershed SFK2 is also a high priority subwatershed because 45 percent of its land is used for crop production. Though no biota sampling stations are located in SFK2, this subwatershed drains to SFK1 and may contribute sediment loading that impacts downstream biota.

- Subwatershed GRN2 ranks as a medium-high priority subwatershed. Biota scores are relatively poor, but crop lands only comprise 16 percent of the land use. Fields located adjacent to the

Groundhouse River or with ditches draining directly to the river should implement crop production BMPs to decrease sediment loading. Subwatershed UTB1 has approximately 6 percent of land used for crop production. No biota scores have been measured in this subwatershed, but it drains to Subwatershed GRN2. In addition, most of the channels and ditches in this subwatershed are straight with little riparian buffer. This subwatershed also ranks medium-high for implementation of crop BMPs.

- Subwatershed SFK3 has a medium priority ranking for implementation of crop BMPs. Macroinvertebrate scores range from poor to good, and 15 percent of the land is used for crop production. GRN3 also ranks medium. Even though a small percentage of crop land is present in this drainage (3 percent), aerial photographs indicate that fields are located close to unbuffered segments of the river. In addition, macroinvertebrate scores range from poor to fair.

- Subwatershed GRN1 has 16 percent of its land used for crop production, but biota scores are fair to excellent. GRN1 therefore ranks as a lower priority for implementation of crop BMPs. Subwatersheds GRN4, UTB2, and WFK1 also rank low. Each of these subwatersheds has less than 3 percent of its area used for crop production and macroinvertebrate scores are fair to good. Of these three subwatersheds, WFK1 had the lowest macroinvertebrate scores and were below the threshold. Crop land located adjacent to streams in this subwatershed may need additional BMPs.

6.2.4 Streambank Erosion

Erosion of streambanks is the primary contributor of sediment to the mainstem Groundhouse River and the second largest source to the South Fork Groundhouse River. Stabilizing the eroding banks in these watersheds will be necessary to improve biota scores.

The most successful BMPs for reducing these loads are the restoration of riparian areas and exclusion of cattle from stream channels. Based on assessment of riparian cover with the use of aerial photographs of the watershed and geographical locations of both registered and unregistered feedlots, exclusion of cattle from streams and creation of riparian buffers should be a priority in the following locations:

- Subwatersheds GRN2 and SFK1 have the lowest macroinvertebrate scores in the watershed. In addition, GRN2 has relatively low fish scores and the second highest fecal coliform concentrations. These two subwatersheds should rank high for stream bank erosion BMPs and riparian buffer restoration.
- Subwatershed SFK3 has the highest fecal coliform concentrations, and macroinvertebrate scores that are below the threshold, but higher than those measured in GRN2 or SFK1. This subwatershed ranks medium-high for stream bank erosion BMPs and riparian buffer restoration.
- Subwatersheds SFK2 and UTB1 are drained by straightened ditches with little riparian cover. These segments should be inspected for signs of erosion. Riparian buffers or vegetated buffer strips should be placed along both sides of these ditches to intercept sediment and other pollutants from the adjacent crop fields and to reduce the volume and velocity of runoff. Both of these subwatersheds drain to either SFK1 or GRN2 and should have a medium-high ranking for streambank erosion BMPs.
- The majority of the mainstem Groundhouse is well buffered in subwatershed GRN3, with the exception of 0.5 miles in the northern part of the watershed which appears to be sparsely buffered, or only buffered on one side of the channel. Macroinvertebrate scores are poor in this

watershed. This section of river should have a high priority for stream bank erosion BMPs and riparian buffer restoration.

- Subwatersheds WFK1 and GRN4 have a medium priority for stream bank erosion BMPs and restoration of riparian areas. The majority of the streams in these subwatersheds are already well buffered. However, each subwatershed contains agricultural land adjacent to a stream or tributary with no riparian buffer to protect the channel. In addition, macroinvertebrate scores within or downstream of these drainages indicate biota impairment.
- Subwatersheds UTB2 and GRN1 have a lower priority for stream bank erosion BMPs and restoration of riparian zones. Macroinvertebrate and fish scores collected in these drainages are above thresholds, and riparian zones are nearly continuous along the river. There are tributaries in GRN1 that drain agricultural land with no riparian buffer, and some of these have been straightened. Macroinvertebrate scores have not been collected on the Groundhouse downstream of these drainages, so it is not clear if restoration measures are needed.

Table 38 summarizes the prioritization of BMPs for each subwatershed.

Table 38. Summary of Prioritization Ranking for Subwatersheds in the Groundhouse River Watershed

Subwatershed Code	Animal Operations	Crop Production	Streambank Erosion
Mainstem Groundhouse Watershed			
GRN1	L	L	L
GRN2	H	M-H	H
GRN3	No known animal operations	M	H
GRN4	Insufficient data to determine prioritization	L	M
UTB1	Insufficient data to determine prioritization	M-H	M-H
UTB2	L	L	L
WFK1	Insufficient data to determine prioritization	L	M
South Fork Groundhouse Watershed			
SFK1	M	H	H
SFK2	Insufficient data to determine prioritization	H	M-H
SFK3	H	M	M-H

A summary of BMPs that may be considered for these subwatersheds is presented in Table 39. Table 40 and Table 41 present the information separately for each source and include cost information as well.

Table 39. Summary of BMPs Reducing Impairments Due to Agricultural Operations

<table>
<tr><th>BMP</th><th>Sediment Reduction (percent)</th><th>Fecal Coliform Reduction (percent)</th><th>Phosphorus Reduction (percent)</th><th>BOD_5 Reduction (percent)</th><th>Atrazine Reduction (percent)</th><th>Additional Benefits for Stream Health</th></tr>
<tr><td>Conservation Tillage</td><td>50 to 90</td><td>na</td><td>68 to 76</td><td>na</td><td>67 to 90</td><td>Reduces runoff losses by 69 percent, which may reduce rates of streambank erosion.</td></tr>
<tr><td>Cover Crops</td><td>90</td><td>na</td><td>70 to 85</td><td>na</td><td>unknown</td><td>Reduces runoff losses by 50 percent, which may reduce rates of streambank erosion.</td></tr>
<tr><td>Filter Strips</td><td>65</td><td>55 to 87</td><td>65</td><td>unknown</td><td>11 to 100</td><td>Slows rates of runoff and may reduce volume via infiltration. May reduce rates of streambank erosion.</td></tr>
<tr><td>Grassed Waterways</td><td>68</td><td>5</td><td>30</td><td>unknown</td><td>25 to 35</td><td>Slows rates of runoff and may reduce volume via infiltration. May reduce rates of streambank erosion.</td></tr>
<tr><td>Riparian Buffers (30 ft wide)</td><td>70 to 90</td><td>34 to 74</td><td>25 to 30</td><td>unknown</td><td rowspan="3">80 to 90

(width not specified in study)</td><td>Slows runoff and may reduce quantity via infiltration. Protects stream channel from erosion and canopy disturbance.</td></tr>
<tr><td>Riparian Buffers (60 to 90 ft wide)</td><td>At least 70 to 90</td><td>At least 34 to 74</td><td>70 to 80</td><td>unknown</td><td>Slows runoff and may reduce quantity via infiltration. Protects stream channel from erosion and canopy disturbance.</td></tr>
<tr><td>Riparian Buffers (200 ft wide)</td><td>At least 70 to 90</td><td>87</td><td>At least 70 to 80</td><td>62</td><td>Slows runoff and may reduce quantity via infiltration. Protects stream channel from erosion and canopy disturbance.</td></tr>
<tr><td>Constructed Wetlands</td><td>53 to 81</td><td>92</td><td>42</td><td>59 to 80</td><td>50</td><td>Slows runoff and may reduce quantity via infiltration, evaporation, and transpiration.</td></tr>
<tr><td>Controlled Drainage (new tile system)</td><td>na</td><td>na</td><td>65</td><td>na</td><td>na</td><td>Reduces peak flow volumes and velocities by storing water; may allow for volume reduction via transpiration.</td></tr>
<tr><td>Controlled Drainage (retrofit tile system)</td><td>na</td><td>na</td><td>35</td><td>na</td><td>na</td><td>Reduces peak flow volumes and velocities by storing water; may allow for volume reduction via transpiration.</td></tr>
</table>

Table 39. Summary of BMPs Reducing Impairments Due to Agricultural Operations (continued)

BMP	Sediment Reduction (percent)	Fecal Coliform Reduction (percent)	Phosphorus Reduction (percent)	BOD_5 Reduction (percent)	Pesticide Reduction (percent)	Additional Benefits for Stream Health
Sedimentation Basin	47 to 80	70 to 78	19 to 51	unknown	unknown	Reduces the volume and velocity of runoff during storm events. Reduces pollutant loading to stream channels.
Proper Manure Handling, Collection, and Disposal	na	90 to 97	unknown	unknown	na	Reduces loads of nutrients and biodegradable organic material entering waterways which may improve dissolved oxygen concentrations.
Manure Composting	na	99	na	unknown	na	Stabilized manure that reaches waterbodies will degrade more slowly and not consume oxygen as quickly as conventional manure.
Application of Composted Manure	68	na	na	na	unknown	Application of composted manure improves soil infiltration and may reduce runoff volumes by 56 percent, potentially reducing rates of streambank erosion.
Alternative Watering Systems with Cattle Exclusion from Streams	unknown	29 to 46	15 to 49	unknown	na	Prevents streambank trampling and therefore decreases loads of manganese to the stream. Reduces direct deposition of manure into stream channel, reduce loads of BOD_5 in addition to nutrients and fecal coliform.
Grazing Land Management	unknown	40 to 90	49 to 60	unknown	na	Increased vegetative ground cover will reduce soil erosion and associated manganese. Improvements in dissolved oxygen concentrations should occur as a result of lower concentrations of BOD_5 in runoff (reduced proportionally by the change in number of cattle per acre.)

Table 40. BMPs for Animal Operations

BMP	Sediment Reduction (percent)	Fecal Coliform Reduction (percent)	Annualized Costs
Proper Manure Handling, Collection, and Disposal	na	90 to 97	Varies by operation and waste handling system (see Appendix F, Section 1.1.11, Table 12)
Manure Composting	na	99	$1.25 to $11.25 per head of swine $16.50 to $151.25 per head of dairy cattle $10.75 to $99.75 per head of beef or other cattle
Application of Composted Manure	68	na	Not quantified
Alternative Watering Systems with Cattle Exclusion from Streams	87	29 to 46	$5 to $8.25 per head of beef or other pastured cattle
Grazing Land Management	Not quantified	40 to 90	Variable – costs may be covered by fencing and alternative watering locations
Filter Strips	65	55 to 87	$4 to $6 per head of cattle
Grassed Waterways	68	5	$0.05 to $0.12 per head of cattle
Riparian Buffers (30 ft)	70 to 90	34 to 74	$0.03 per ft of channel
Riparian Buffers (60 to 90 ft)	At least 70 to 90	At least 34 to 74	$0.05 to $0.07 per ft of channel
Riparian Buffers (200 ft)	At least 70 to 90	87	$0.17 per ft of channel
Constructed Wetlands	53 to 81	92	$2.50 per head of dairy cattle $4.50 per head of swine

Table 41. BMPs for Crop Production

<table>
<tr><th>BMP</th><th>Phosphorus Reduction (percent)</th><th>Sediment Reduction (percent)</th><th>Pesticide Reduction (percent)</th><th>Annualized Costs per Acre Treated</th></tr>
<tr><td>Conservation Tillage</td><td>68 to 76</td><td>50 to 90</td><td>67 to 90</td><td>$1.25 to $2.25</td></tr>
<tr><td>Cover Crops</td><td>70 to 85</td><td>90</td><td>Unknown</td><td>$20.50</td></tr>
<tr><td>Controlled Drainage (new)</td><td>65</td><td>unknown</td><td>na</td><td>$2.50</td></tr>
<tr><td>Controlled Drainage (retrofit)</td><td>35</td><td>unknown</td><td>na</td><td>$0.75 to $1.50</td></tr>
<tr><td>Filter Strips</td><td>65</td><td>65</td><td>11 to 100</td><td>$24.75</td></tr>
<tr><td>Grassed Waterways</td><td>30</td><td>68</td><td>25 to 35</td><td>$2.25 to $6.50</td></tr>
<tr><td>Riparian Buffers (30 ft)</td><td>25 to 30</td><td>70 to 90</td><td rowspan="3">80 to 90 (width not specified in study)</td><td>$6</td></tr>
<tr><td>Riparian Buffers (60 to 90 ft)</td><td>70 to 80</td><td>At least 70 to 90</td><td>$12 to $24</td></tr>
<tr><td>Riparian Buffers (200 ft)</td><td>At least 70 to 80</td><td>At least 70 to 90</td><td>$53</td></tr>
<tr><td>Sedimentation Basin</td><td>19 to 51</td><td>47 to 80</td><td>unknown</td><td>$33.50</td></tr>
</table>

6.3 Monitoring Plan

Managing impairments in the Groundhouse watershed will likely involve multiple BMPs. Continuing to monitor water quality and biota scores in the listed segments will determine whether or not stream habitat restoration measures are required to bring the watershed into compliance. At a minimum, fish and macroinvertebrate sampling should be conducted by the MPCA every six to ten years during the summer season at each established location until compliance is observed for at least two consecutive summers, and fecal coliform monitoring should occur at least five times per month from April through October at each water quality station. The Snake River Watershed Management Board, a four-county joint powers board, is expected to begin a 2008-2010 surface water monitoring program which includes the Groundhouse and South Fork Groundhouse rivers.

Tracking the implementation of BMPs while continuing to monitor water quality and biological conditions in the watershed will assist the stakeholders and public agencies in determining the effectiveness of the implementation plan. If concentrations remain above the water quality standards or biota scores remain below, further encouragement of the use of BMPs across the watershed through education and incentives will be a priority. It may also be necessary to begin funding efforts for localized BMPs such as riparian buffer and stream restoration.

6.4 Reasonable Assurance

USEPA requires that a TMDL provide reasonable assurance that the required load reductions will be achieved and water quality will be restored. For this watershed, BMPs to control loading from crop production, animal operations, and streambank erosion are the primary management strategies to reach these goals. Participation of farmers and landowners is essential to improving water quality, but lack of information and upfront cost may deter participation. Educational efforts and cost share programs will likely increase participation to levels needed to protect water quality.

Two of the incentive programs discussed below were administered under the 2002 Farm Bill, which expired September 30, 2007. The Conservation Reserve Program will continue to pay out existing contracts, but new enrollments will not be allowed until the bill is reinstated; no official date of reinstatement has been announced. Though the Environmental Quality Incentives Program was also part of the 2002 Farm Bill, it was extended beyond fiscal year 2007 by the Deficit Reduction Act of 2005 (Congressional Research Reports for the People, 2007).

This section briefly describes the programs available in the watershed. Incentive amounts for each BMP will be summarized in the full implementation plan.

6.4.1 Environmental Quality Incentives Program (EQIP)

Several cost share programs are available to farmers and landowners who voluntarily implement resource conservation practices in the Groundhouse watershed. The most comprehensive is the NRCS Environmental Quality Incentives Program (EQIP) which offers cost sharing and incentives to farmers statewide who utilize approved conservation practices to reduce pollutant loading from agricultural lands. In order to participate in the EQIP cost share program, all BMPs must be constructed according to the specifications listed for each conservation practice.

The specifications and program information can be found online at:
http://www.mn.nrcs.usda.gov/programs/eqip/.

6.4.2 Conservation Reserve Program (CRP)

The Farm Service Agency of the USDA supports the Conservation Reserve Program (CRP) which rents land converted from crop production to grass or forestland for the purposes of reducing erosion and protecting sensitive waters. This program is available to farmers who establish vegetated filter strips or grassed waterways.

More information about this program is available online at:
http://www.nrcs.usda.gov/programs/crp/.

The Conservation Reserve Program also sponsors the Farmable Wetlands Pilot Program. The goal of this program is to restore 500,000 acres of wetland and buffer areas to a more natural hydrologic and vegetative condition.

More information about this program is available online at:
http://www.fsa.usda.gov/FSA/webapp?area=home&subject=copr&topic=fwp.

The CRP also sponsors the Conservation Reserve Enhancement Program, which provides incentives to land owners who retire environmentally sensitive agricultural lands.

More information about this program is available online at:
http://www.bwsr.state.mn.us/easements/crep/factsheet.html.

6.4.3 Wetlands Reserve Program

The USDA NRCS sponsors the federal Wetlands Reserve Program which encourages voluntary participation of farmers and land owners to enhance, restore, and protect wetland environments. The program provides support through technical assistance and cost share programs.

More information about this program is available online at:
http://www.nrcs.usda.gov/programs/wrp/.

6.4.4 Wildlife Habitat Incentives Program

The USDA NRCS also sponsors the Wildlife Habitat Incentives Program (WHIP). This program offers technical assistance and cost sharing to farmers and land owners who want to improve fish and wildlife habitat. Eligible lands include grassland, woodland, pastureland, wetlands, streams, and riparian areas. Only land not eligible for other federal or state conservation programs, such as the Wetlands Reserve Program or the Conservation Reserve Program, may be considered for WHIP assistance.

More information about this program is available online at:
http://www.mn.nrcs.usda.gov/programs/whip/fact.pdf.

6.4.5 AgBMP Loan Program

The AgBMP Loan Program offered through the Minnesota Department of Agriculture provides low-interest loans to assist farmers or land owners who implement conservation practices aimed at reducing water pollution caused by agricultural activities or failing onsite wastewater treatment systems. Examples of covered practices include feedlot improvements, manure storage basins, manure handling equipment, conservation tillage equipment, repair of onsite wastewater treatment systems, grassed waterways, streambank protection, sedimentation basins, wind breaks, and other erosion control practices.

More information about this program is available online at:
http://www.mda.state.mn.us/grants/loans/agbmploanmore.htm.

6.4.6 Sustainable Agriculture Grant Program (SARE)

The Sustainable Agricultural Grant Program funds research, education, and outreach efforts for sustainable agricultural practices. Private landowners, organizations, educational, and governmental institutions are all eligible for participation in this program.

More information concerning the Sustainable Agricultural Grant Program can be found online at:
http://www.mda.state.mn.us/about/divisions/esap.htm.

6.4.7 Local Soil and Water Conservation Districts (SWCDs)

The local Soil and Water Conservation Districts (SWCDs) issue State cost-share funds administered by the Minnesota Board of Water and Soil Resources.

The Kanabec County SWCD can be contact via email:
kelly.osterdyk@mn.nacdnet.net.

The Mille Lacs County SWCD maintains the following website:
http://www.millelacsswcd.org/.

6.4.8 Snake River Watershed Management Board

The Snake River Watershed Management Board offers cost share incentives through a continuation of the Minnesota Clean Water Partnership Grant Program.

More information concerning the Snake River Watershed Management Board can be found online at:
http://kanabeccounty.govoffice2.com/index.asp?Type=B_BASIC&SEC={9210D5BC-C702-4D79-870F-CC66C94C1A84}.

6.5 Implementation Time Line

This implementation strategy for the Groundhouse watershed defines a phased approach for achieving the water quality and biota standards. Ideally, implementing fecal coliform and sediment control measures on nonpoint sources of loading will be based on voluntary participation which will depend on 1) the effectiveness of the educational programs for farmers, landowners, and owners of onsite wastewater treatment systems, and 2) the level of participation in the programs. This section outlines a schedule for implementing the control measures and determining whether or not they are sufficient to meet the standards.

Phase I of this implementation plan should focus on education of farm owners and rural land owners concerning the benefits of agricultural BMPs on crop yield, soil quality, and water quality as well as cost share programs available in the watershed. It is expected that initial education through public meetings, mass mailings, TV and radio announcements, and newspaper articles could be achieved in less than 6 months. Assistance with educational programs is available through the following agencies: Minnesota Department of Agriculture, the Minnesota Environmental Protection Agency, and the local Soil and Water Conservation Districts.

Phase II of the implementation schedule will involve voluntary participation of farmers and rural land owners using BMPs such as cattle exclusion from streams, proper management of manure, use of filter strips, composting, constructed wetlands, conservation tillage, and grassed waterways, and creation of riparian buffers. The local Natural Resources Conservation Service office, Soil and Water Conservation Districts, and the Snake River Water Management Board will be able to provide technical assistance and cost share information for these BMPs. In addition, initial inspections of all onsite wastewater treatment systems and necessary repairs may begin. Continued monitoring of water quality and biological integrity in the watershed should continue throughout this phase, which will likely take one to three years.

If fecal coliform concentrations measured during Phase II monitoring remain above the water quality standards or biota scores remain below, Phase III of the implementation plan will be necessary. The load reductions achieved during Phase II should first be estimated by 1) summarizing the areas where BMPs are in use, 2) calculating the reductions in loading from BMPs, and 3) determining the impacts on pollutant loads measured before and after Phase II implementation. If BMPs are resulting in decreased loads of fecal coliform and sediment, and additional areas could be incorporated, further efforts to include more stakeholders in the voluntary program will be needed. If the Phase II BMPs are not having the desired impacts on pollutant concentrations, or additional areas of incorporation are not available, additional measures, such as habitat restoration, will be needed. If required, this phase may last five to ten years.

7 Public Participation Record

Public Participation for the Groundhouse TMDL study consisted of a two-day meeting and workshop with local project partners held during the Stressor Identification Process development and four stakeholder meetings targeting landowners during the development of the TMDL document. One meeting of a newly formed implementation committee, including several local residents, has also been held to help develop and carry out an Implementation Plan. Meetings averaged from numbers of 15 to 30 people; which included not only local organizations but concerned local citizens and the agricultural community.

Local Partners provided updates and meeting notices to local stakeholders by sending out letters to individuals, news releases, and provided annual updates to the county board. The local Soil and Water Conservation Districts also provided quarterly updates to their boards and presented the project along with installed Best Management Practices at the local fairs.

Stakeholder meetings were held on:

- June 29, 2006
- October 24, 2006
- February 6, 2007
- July 31, 2008
- September 2, 2008 – Implementation Plan Discussion

Attendee organizations at one of more of these meetings included the following:

City of Ogilvie
Ann Lake Township
Arthur Township
Brunswick Township
Hayland Township
Kanabec Township
Mudgett Township
Southfork Township
Kanabec County
Kanabec Soil and Water Conservation District
Kanabec County Environmental Services
Mille Lacs County
Mille Lacs Soil and Water Conservation District
Snake River Watershed Management Board
IMPACK 6
Board of Water and Soil Resources
St. Croix Basin Planning Team
Minnesota Department of Natural Resources
Natural Resources Conservation Service
Tetra Tech

This TMDL will be put out for public comment in early 2009.

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1 Introduction

The Groundhouse River located in east-central Minnesota has been included on the Minnesota 303(d) list of impaired waters for fecal coliform and biological impairment (Figure A-1 and Table A-1). Currently available information suggests the biological impairment is associated with excessive fine sediments in the channel (USEPA, 2004). In accordance with the Clean Water Act, a Total Maximum Daily Load (TMDL) is to be calculated to determine the allowable loads of fecal coliform and sediment and reductions needed to meet water quality standards. A TMDL is defined as "the sum of the individual wasteload allocations for point sources and load allocations for nonpoint sources and natural background" such that the capacity of the waterbody to assimilate pollutant loadings is not exceeded. A TMDL is also required to be developed with seasonal variations and must include a margin of safety that addresses the uncertainty in the analysis.

Table A-1. Section 303(d) listed segments in the Groundhouse River watershed.

Stream Name	Description	Year Listed	Reach ID	Designated Use	Impairment
Groundhouse River	Headwaters to S Fork Groundhouse River	2002	07030004-513	Aquatic recreation	Fecal coliform
Groundhouse River	Headwaters to S Fork Groundhouse River	2002	07030004-513	Aquatic life	Fish IBI
Groundhouse River	Headwaters to S Fork Groundhouse River	2004	07030004-513	Aquatic life	Invertebrate IBI
Groundhouse River, South Fork	Headwaters to Groundhouse River	2004	07030004-539	Aquatic life	Invertebrate IBI

Minnesota has established a detailed procedure for determining the designated use and condition of each waterbody. Inclusion on the Minnesota 303(d) List can be a result of low scores on measures of biological health such as the Index of Biological Indicators (IBI) or observed exceedances of the applicable water quality standard.

The cause of biological impairment can be fairly difficult as many factors, including pollutants and habitat, can stress a biological community. Application of the Stressor Identification (SI) process to the Groundhouse River (EPA 2004) indicated that the most probable cause of impairment was "loss of suitable habitat from unstable or unsuitable substrates caused by excess fines less than 2 mm in diameter." While fine sediment was indicated as the likely cause of the biological impairment, an intensive monitoring program for a number of water quality parameters was performed in 2005. These data are reviewed in this document to determine whether stressors in addition to fine sediment might be present. Potentially relevant water quality standards for the Groundhouse River, classified as a Class 2B warm-water fishery, are presented in Table A-2. Fecal coliform impairment is determined through comparison to observed maximum and a geometric mean standard.

Table A-2. Class 2B Water Quality Standards for Relevant Parameters

Parameter	Standard	Comment
Fecal coliform	200 #/100 mL geometric mean or Not to exceed 2,000 #/100 mL in more than 10% of samples collected per month	Applies April 1 – October 31 Geometric mean to be calculated on not less than 5 samples per month
Turbidity	25 nephelometric turbidity units (NTU)	
Dissolved Oxygen	Not less than 5 mg/L as a daily minimum	
Ammonia (unionized)	0.04 mg/L un-ionized ammonia	Total ammonia, pH, and temperature must be collected to determine the determine the un-ionized ammonia fraction

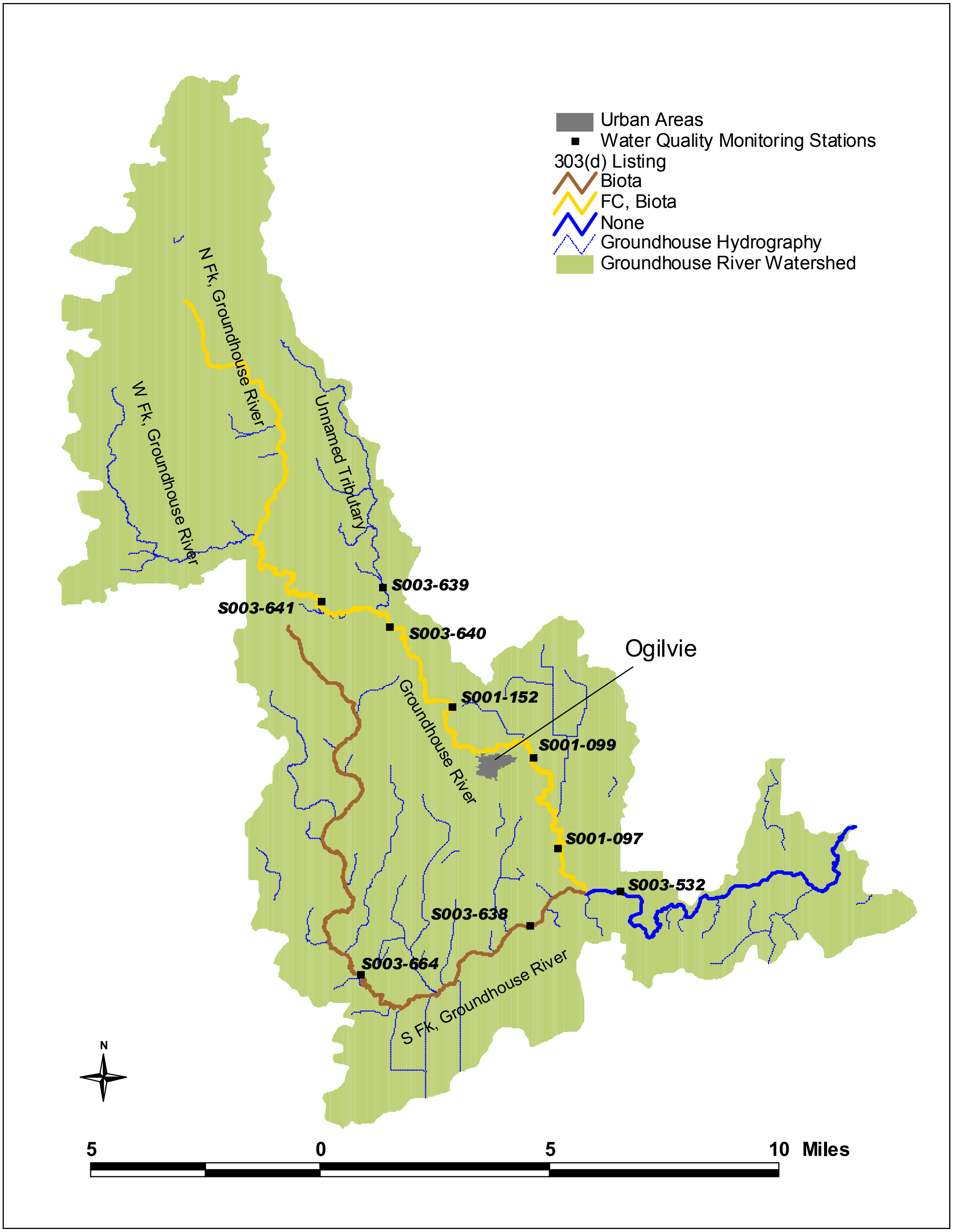

Figure A-1. Listed Impairments and Location of Monitoring Stations.

2 Review of Available Data

Water quality in the Groundhouse River watershed was monitored by the Minnesota Pollution Control Agency (MPCA) at a number of locations over the period from 1987 through 2005 (Figure A-1). The majority of the historic water quality sampling was performed at S001-152 (Groundhouse River on County Road 55, one mile northwest of Ogilvie). However, a significant expansion of the monitoring program in the Groundhouse River was initiated in 2005 in support of the proposed TMDL (Table A-3). The 2005 data were collected at nine stations throughout the watershed and will provide the basis of the majority of the analyses presented in this document.

Table A-3. Station Summary of Observations for all Water Quality Parameters.

STATION_ID	1987	1988	1989	1997	1998	2005	Total
S001-097	20					290	310
S001-099	21					212	233
S001-152	12	61	60	11	30	212	386
S003-532						176	172
S003-638						288	288
S003-639						211	211
S003-640						207	207
S003-641						212	212
S003-664						131	131
Total	53	61	60	11	30	1,939	2,154

The available monitoring data were reviewed and analyzed using a number of techniques. A comparison of the data statistics to water quality characteristics of minimally impacted watersheds was performed along with a review of spatial patterns that might be seen throughout the watershed. Trend analyses for fecal coliform and TSS were performed for station S001-152 as it has been periodically monitored since 1987. Finally, flow duration curves were developed to determine the flow conditions under which the fecal coliform standards are exceeded and when TSS concentrations are elevated.

To perform this assessment, data from the 2005 monitoring events were summarized. Data were collected for the following water quality parameters:

- fecal coliform (FC)
- total suspended solids (TSS)
- suspended volatile solids (VS)
- specific conductance
- water temperature
- turbidity
- dissolved oxygen (DO)
- total ammonia nitrogen (NH_3)
- pH
- Total Kjeldahl nitrogen (TKN)
- nitrate/nitrite (NO_2+NO_3)
- total phosphorus (TP)
- total ortho-phosphorus (TOP)

The sampling for NO_2, TP, and PO4 was conducted only at two locations (S001-097 and S003-638). Where possible, the 2005 monitoring data statistics are compared to data collected for minimally impacted streams (MIS) in the seven Minnesota ecoregions. The Groundhouse watershed straddles the Northern Lakes and Forests (NLF) and the North Central Hardwood Forest (NCHF) ecoregions. The headwater stations (S003-641, S003-640, and S003-639) fall within the NLF ecoregion with the remainder transitioning to the NCHF ecoregion.

2.1 Fecal coliform

Fecal coliform is often used as an indicator of bacteriological contamination from animal and human waste. The Minnesota water quality standard (WQS) for fecal coliform in Class 2B waters is a monthly geometric mean of 200 organisms/100mL and a not-to-exceed standard of 2,000 organisms/100mL for more than 10 percent of samples from April through October. Review of the fecal coliform data show a wide range of reported values which is consistent with the behavior of bacteria in natural systems. Two samples were reported as being less than 10 organisms/100mL and were set to one half of this value (5) for the statistics analysis. Median and geometric mean values are relatively similar at all stations with a few exceptions. The highest overall geometric mean values of fecal coliform are seen at S003-664 on the South Fork of the Groundhouse River. It should be noted however that data was collected at this station only during July through October which are the months with the highest concentrations across the watershed. The highest median and 95th percentile and second highest geometric mean values are seen at station S001-152. The lowest overall values are seen at S003-639 located on the North Fork of the Groundhouse River near the confluence with the mainstem. In general, the head water data greatly exceed the NLF MIS statistics. The 5th percentile values for the station in the NCHF ecoregion are higher than the MIS values but fall very close to the NCHF median and 95th percentile values with the exception of S001-152.

To highlight the spatial variability of fecal coliform throughout the watershed, the median and geometric mean fecal coliform concentrations have been associated with the station locations. Station S001-152 appears to be the primary station with recurring high fecal coliform values. Further analyses of the fecal coliform data are provided in Section 2.1 and the following sections.

2.2 TSS and VSS

The biotic impairment in the Groundhouse River was attributed to an excess of fine sediments (EPA, 2004), specifically at study site 3 near Ogilvie (which is located close to MPCA monitoring site S001-099). The most relevant parameter for this cause of impairment, percent fines <2mm, is not specifically collected as part of the water quality monitoring. However, total suspended solids can provide some insight into the levels of sediment being transported through the system at the time of sampling (although it is recognized that TSS measurements do not capture the movement of bedload sediments). Table A-4 summarizes the available TSS data for the Groundhouse monitoring locations and relevant ecoregion data. Four samples were reported as being less than detection limits (MDL) and were set to one half of MDL for the statistics analysis. Sediment levels in the headwater segments are very close to or below the NLF statistics. The remaining stations generally show significantly lower TSS concentrations than those seen in the NCHF data. Only S003-532 has a 95th percentile value which is slightly greater than those observed in minimally impacted areas. These results suggest that TSS is likely not a problem in the Groundhouse River as most locations exhibit concentrations near or better than reference conditions. A spatial map of the average TSS values is shown in Figure A-2.

Table A-4. Total Suspended Solids (mg/L) Summary for 2005

STATION_ID	Number of Samples	5th Percentile	Median	95th Percentile
North Fork Groundhouse River				
S003-641	18	0.93	2.50	9.00
S003-639 (unnamed tributary)	18	1.00	2.50	4.75
S003-640	18	1.00	2.00	4.60
S001-152	18	1.00	2.50	12.50
S001-099	18	1.00	2.00	6.75
S001-097	18	1.00	2.00	12.90
South Fork Groundhouse River				
S003-664	12	2.55	4.00	9.15
S003-638	18	1.85	3.00	9.10
Groundhouse River (Mainstem)				
S003-532	13	2.00	3.00	20.80
Minnesota Minimally Impacted Streams (1986-1992)				
Northern Lakes and Forests	-	0.8	7.8	8.2
North Central Hardwood Forests	-	1.4	7.7	20.0

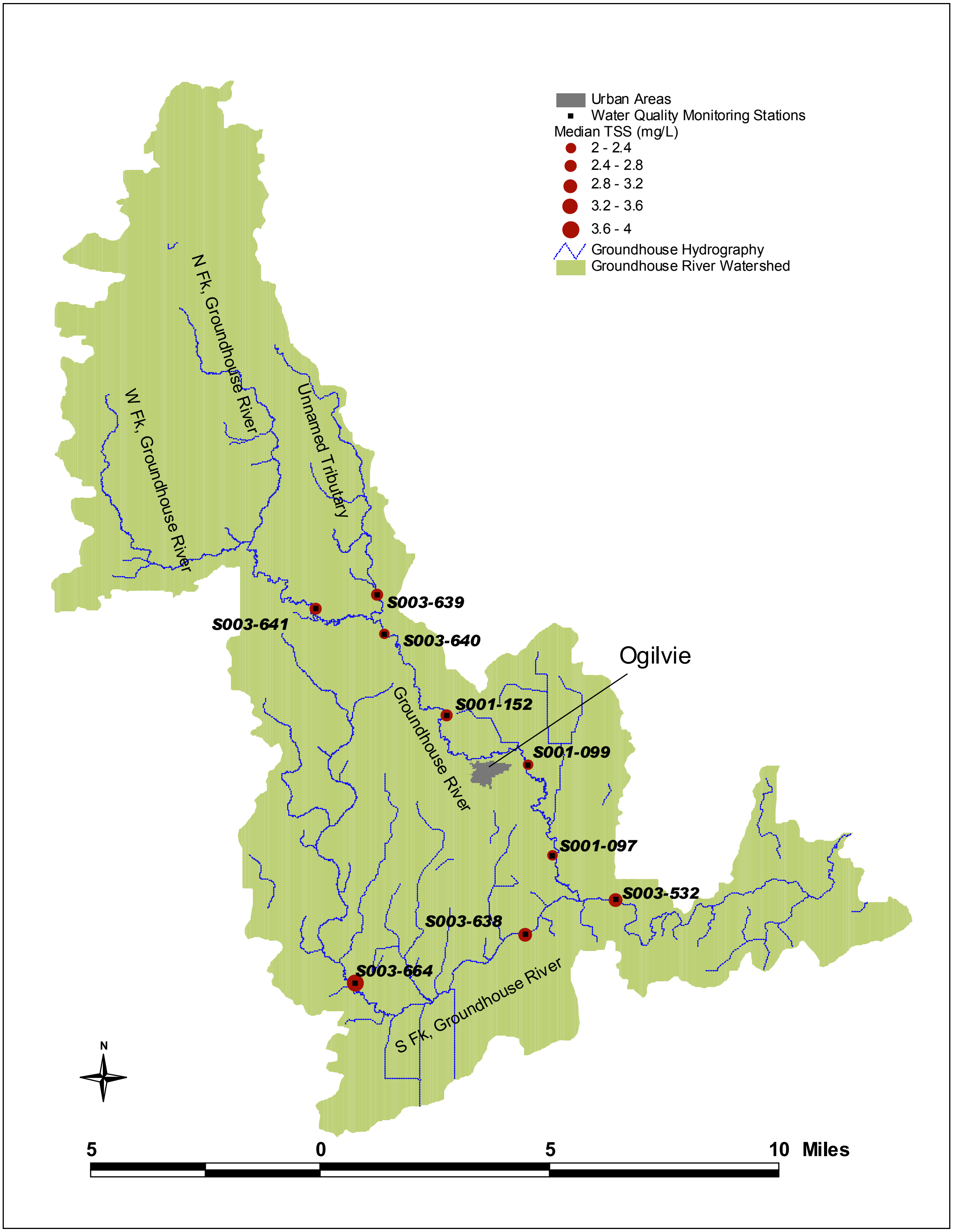

Figure A-2. Spatial Distribution of Median TSS in the Groundhouse River Watershed.

Volatile suspended solids were collected at the nine water quality monitoring sites. VSS is a measure of the organic content of a water sample and does not have a direct impact on aquatic life and does not have a Minnesota water quality standard. Volatile solids are not directly related to fine sediment loads but may give an indication of changes in activities or sources at a given location. Review of the monitoring data shows comparable results at all sites with a slight increase in the 95th percentile at S003-641 and S001-152. In this case, volatile solids are fairly low and constant throughout the watershed (Table A-5). No MIS data are available for comparison. Fifty-seven samples were reported as being less than detection limits (MDL) and were set to one half of MDL for the statistics analysis.

Table A-5. Volatile Suspended Solids (mg/L) Summary for 2005

STATION_ID	Number of Samples	5th Percentile	Median	95th Percentile
North Fork Groundhouse River				
S003-641	18	0.50	1.00	5.15
S003-639 (unnamed tributary)	18	0.50	1.00	2.15
S003-640	18	0.50	1.00	2.15
S001-152	18	0.50	1.50	4.60
S001-099	18	0.50	1.00	2.15
S001-097	18	0.50	1.00	3.60
South Fork Groundhouse River				
S003-664	12	0.50	2.00	3.00
S003-638	18	0.50	1.00	2.45
Groundhouse River (Mainstem)				
S003-532	13	0.50	1.00	4.40

2.3 Specific Conductance

Specific conductance provides an indication of the amounts of dissolved solids found at a specific location. While this is not directly related to the presence or absence of fine sediments, it can indicate whether a significant chemical or physical change occurs at any specific location. The 5th percentile and median conductance observations are very consistent throughout the watershed with the exception of sites S003-664 and S003-638. The conductance in the South Fork appears to be elevated in relation to the remainder of the watershed (See Table A-6 and Figure A-3) raising some question about the activities in this area. The headwater segments show slightly higher 5th percentile values as compared with the MIS data but show significantly lower median and 95th percentile values. The remaining stations have conductance statistics which are near or below the values determined for the NCHF ecoregion.

Table A-6. Specific Conductance (µS) Summary for 2005.

STATION_ID	Number of Samples	5th Percentile	Median	95th Percentile
North Fork Groundhouse River				
S003-641	29	55.0	85.0	186.4
S003-639 (unnamed tributary)	29	56.0	95.0	165.4
S003-640	27	55.9	82.0	238.6
S001-152	29	59.4	95.0	176.0
S001-099	29	68.2	113.0	269.6
S001-097	29	68.0	118.0	245.6
South Fork Groundhouse River				
S003-664	17	83.0	307.0	376.4
S003-638	29	150.4	251.0	356.6
Groundhouse River (Mainstem)				
S003-532	25	109.2	165.0	280.0
Minnesota Minimally Impacted Streams (1986-1992)				
Northern Lakes and Forests	-	43	180	320
North Central Hardwood Forests	-	210	295	360

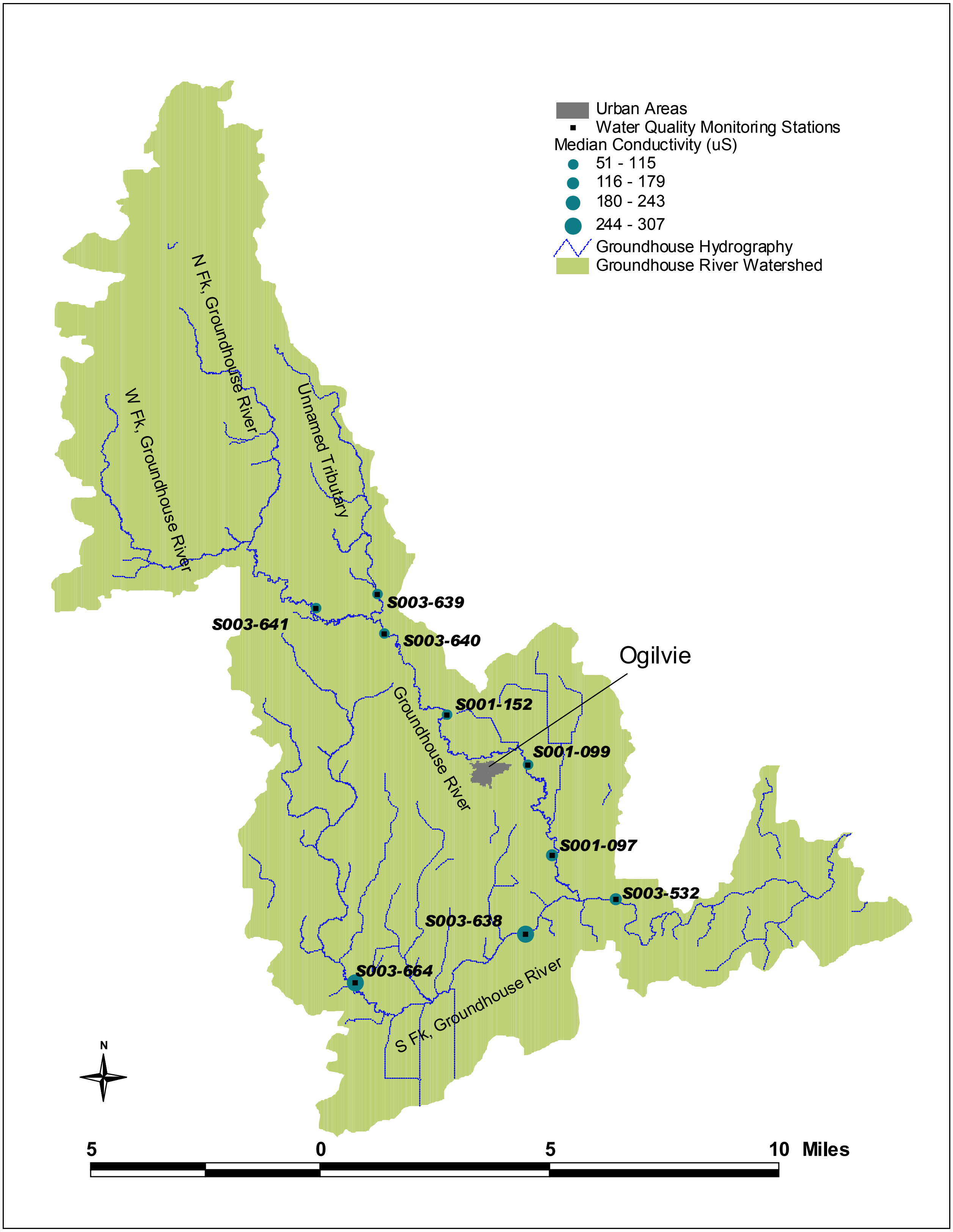

Figure A-3. Spatial Distribution of Median Conductivity in the Groundhouse River Watershed

Turbidity can be an excellent indicator of the transport of fine sediments in a river system. It measures the scattering of light in the water column and is affected by the presence of organic and inorganic matter. While organic matter such as algae and detritus can be a significant cause for high turbidity numbers, this does not seem to be the case in the Groundhouse River watershed as indicated by the low volatile solids observations. While the majority of turbidity in the Groundhouse River may be attributed to inorganic particles, the overall average turbidity is much lower than the water quality standard of 25 NTU at all locations. The highest levels of turbidity occur near the mouth of the Groundhouse River (S003-532) but in general turbidity is consistent and low throughout the watershed (Table A-7 and Figure A-4).

Table A-7. Turbidity (NTU) Summary for 2005

STATION_ID	Number of Samples	5th Percentile	Median	95th Percentile
North Fork Groundhouse River				
S003-641	15	1.98	4.00	11.30
S003-639 (unnamed tributary)	15	2.18	3.40	13.40
S003-640	15	1.81	3.60	9.55
S001-152	15	2.27	3.20	8.06
S001-099	15	2.01	2.60	8.50
S001-097	15	2.12	3.10	11.55
South Fork Groundhouse River				
S003-664	12	3.16	4.40	11.29
S003-638	15	3.57	4.90	12.47
Groundhouse River (Mainstem)				
S003-532	11	1.75	3.70	19.20

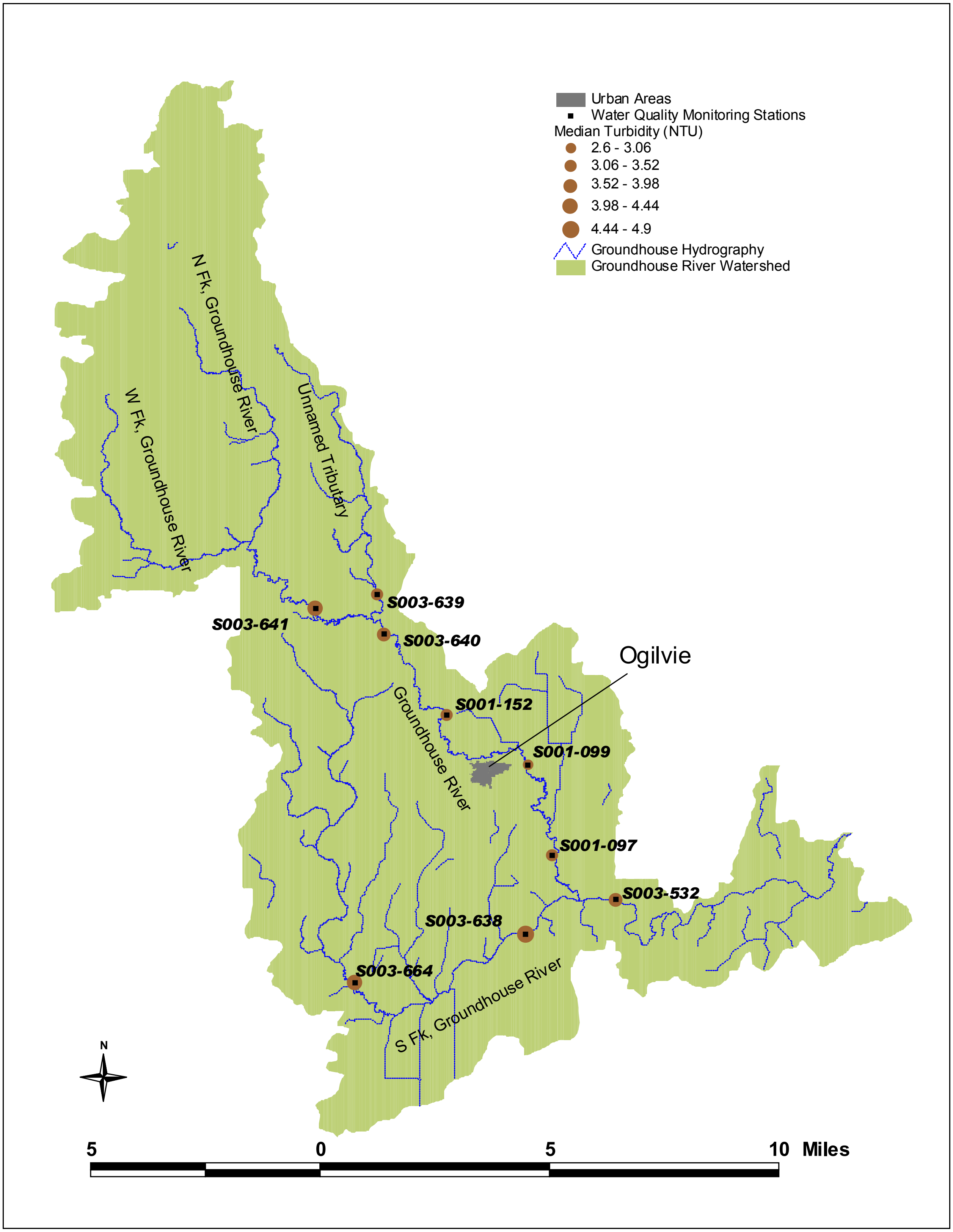

Figure A-4. Spatial Distribution of Median Turbidity in the Groundhouse River Watershed.

2.4 Dissolved Oxygen

Dissolved oxygen (DO) is a critical water chemistry property and required in sufficient amounts to support aquatic life. Low DO is one common cause of biotic impairment. The aquatic life standard for warm water fisheries (Class 2B waters) is “not less than 5.0 mg/L as a daily minimum”. Table A-8 shows the statistics for the monitoring done at each of the nine water quality monitoring locations. No ecoregion data are available for the MIS report for comparison. However, the 5th percentile of DO concentrations at all locations are in compliance with the water quality standard. While this data may not capture short term fluctuations in DO, it does indicate the DO levels are generally very good. Station S001-640 is the only location where the data seem to be consistently lower than the other stations. DO levels appear to return to the higher levels seen across the watershed by the next most downstream monitoring location (S001-152).

Table A-8. Dissolved oxygen (mg/L).

STATION_ID	Number of Samples	5th Percentile	Median	95th Percentile
North Fork Groundhouse River				
S003-641	29	6.13	8.52	12.17
S003-639 (unnamed tributary)	29	6.08	8.29	12.10
S003-640	28	5.88	7.36	10.56
S001-152	29	6.24	8.57	11.48
S001-099	29	6.44	8.32	11.24
S001-097	29	6.63	8.80	11.60
South Fork Groundhouse River				
S003-664	17	6.53	7.69	11.45
S003-638	29	5.81	8.21	11.41
Groundhouse River (Mainstem)				
S003-532	25	6.61	8.71	11.78

2.5 Ammonia

Un-ionized ammonia can be toxic to aquatic life at elevated concentrations. The water quality standard for Class 2B waters is 0.04 mg/L. The monitoring data collected for the Groundhouse and the MIS studies, however, measures total ammonia (both the ionized and unionized forms). The fraction of ammonia that is unionized depends on the pH at the time of the sample. Review of the Groundhouse River data indicates that observed values are at or below those compiled for the MIS study (See Table A-9 and Figure A-5). Ammonia observations are consistently low throughout the watershed with very low values seen at station S003-664. Two hundred and ninety-four samples were reported as being less than detection limits (MDL) and were set to one half of MDL for the statistics analysis.

Table A-9. Ammonia (mg/L) Summary

STATION_ID	Number of Samples	5th Percentile	Median	95th Percentile
North Fork Groundhouse River				
S003-641	14	0.010	0.010	0.098
S003-639 (unnamed tributary)	14	0.010	0.020	0.100
S003-640	14	0.010	0.015	0.084
S001-152	14	0.010	0.015	0.081
S001-099	14	0.010	0.035	0.122
S001-097	14	0.010	0.030	0.104
South Fork Groundhouse River				
S003-664	8	0.010	0.010	0.017
S003-638	14	0.020	0.045	0.100
Groundhouse River (Mainstem)				
S003-532	13	0.010	0.030	0.072
Minnesota Minimally Impacted Streams (1986-1992)				
Northern Lakes and Forests	-	0.02	0.05	0.11
North Central Hardwood Forests	-	0.02	0.08	0.2

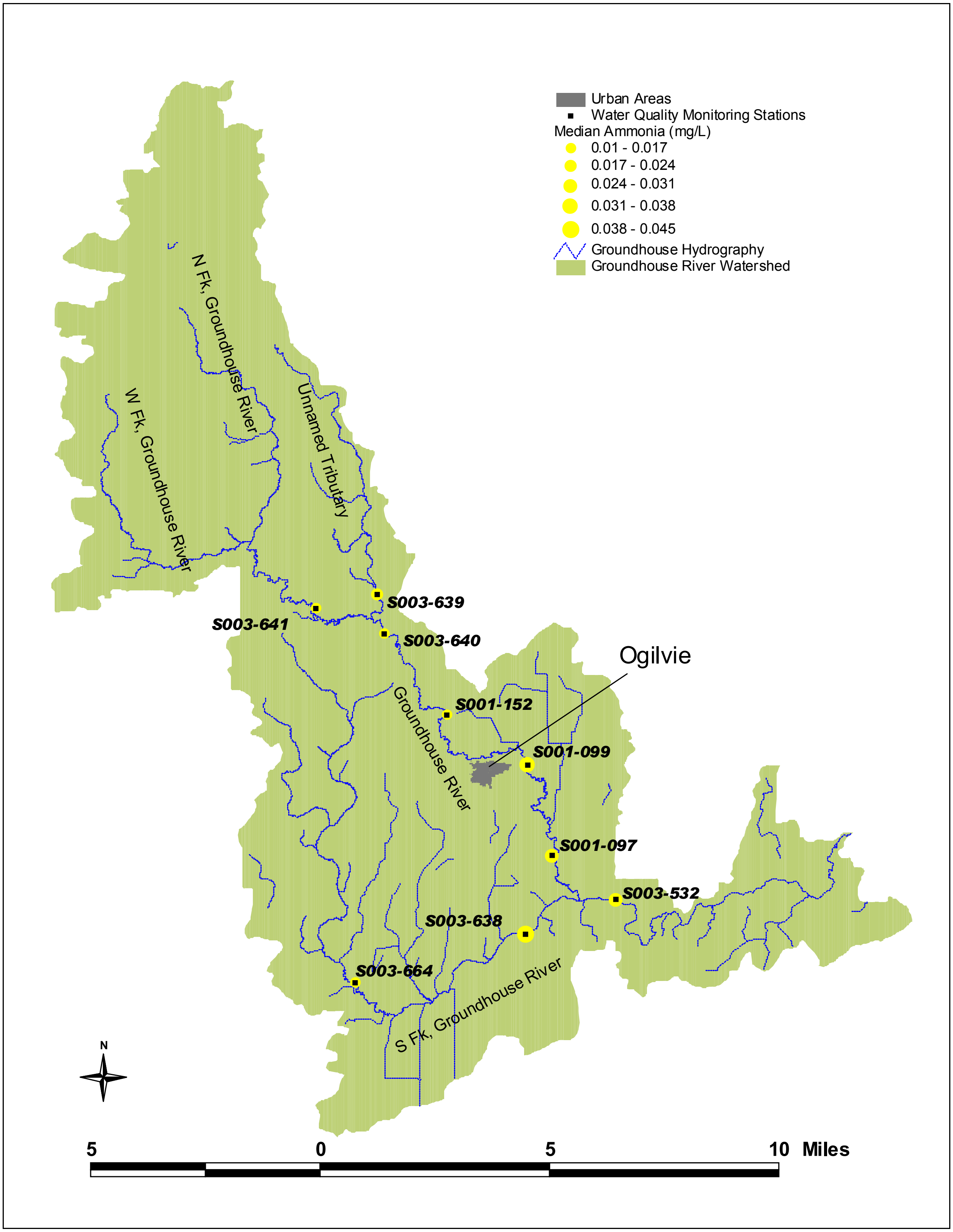

Figure A-5. Distribution of Median Ammonia in the Groundhouse River Watershed

2.6 pH

Extreme high or low pH values can indicate some type of water quality issue and can be harmful to aquatic life. The water quality standard for Class 2B waters is a minimum of 6.5 and a maximum of 8.5 Review of the monitoring data show that the Groundhouse falls well within this range but does show values that are consistently below those seen in the MIS studies. While the values do not indicate a water quality problem, some type of influence can be seen (Table A-10 and Figure A-6).

Table A-10.Summary of pH Data.

STATION_ID	Number of Samples	5th Percentile	Median	95th Percentile
North Fork Groundhouse River				
S003-641	29	6.82	7.36	7.79
S003-639 (unnamed tributary)	29	6.67	7.36	7.62
S003-640	28	6.62	7.14	7.76
S001-152	29	6.77	7.44	8.04
S001-099	29	6.82	7.31	7.79
S001-097	30	6.84	7.36	7.99
South Fork Groundhouse River				
S003-664	17	7.14	7.78	8.04
S003-638	29	6.91	7.53	7.96
Groundhouse River (Mainstem)				
S003-532	25	7.01	7.51	8.05
Minnesota Minimally Impacted Streams (1986-1992)				
Northern Lakes and Forests	-	7.3	7.8	8.2
North Central Hardwood Forests	-	7.6	8.1	8.7

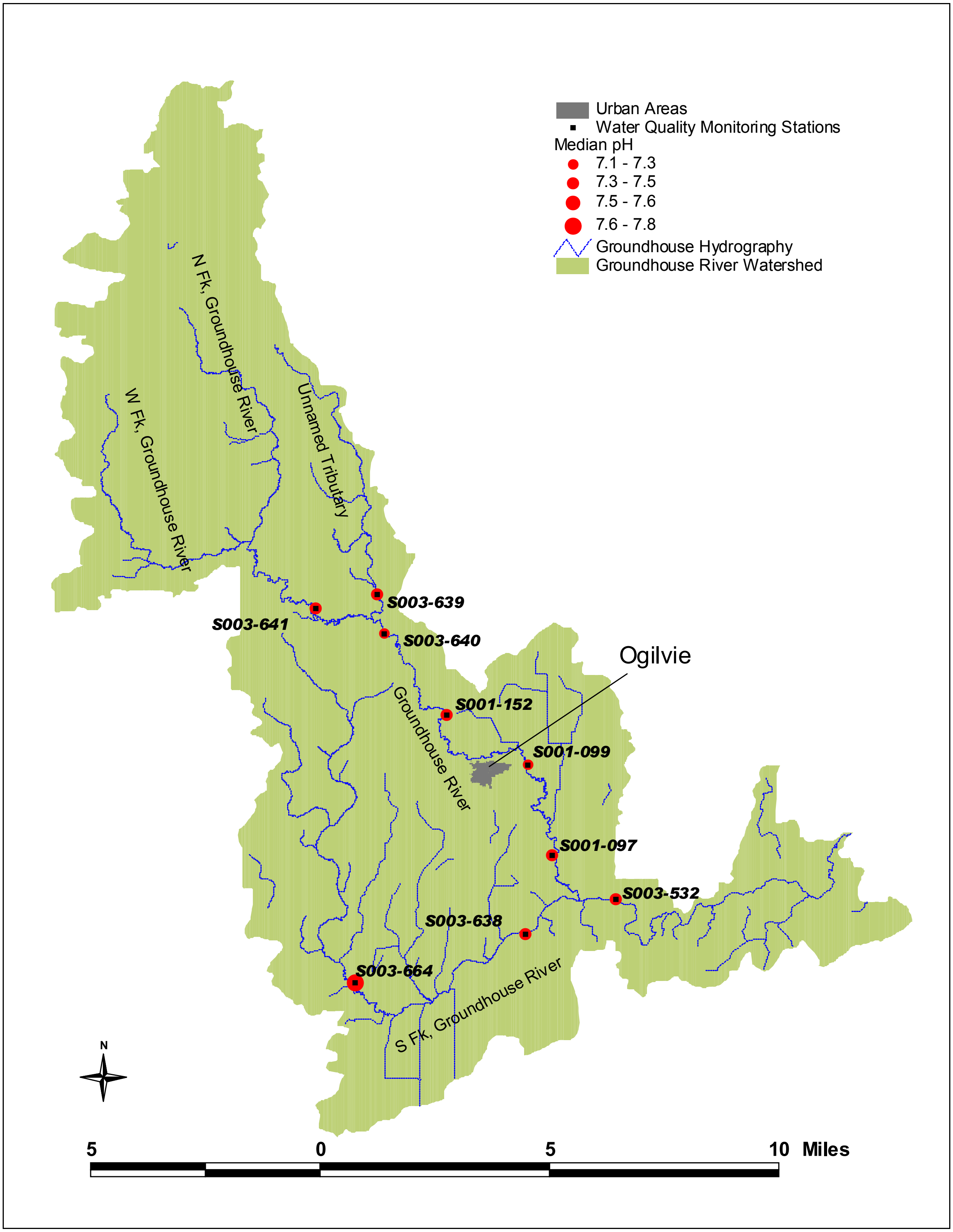

Figure A-6. Distribution of Median pH in the Groundhouse River Watershed

2.7 Nitrogen

Additional nutrient monitoring data were collected during 2005 at a limited number of stations. These data do not have associated water quality standards and only have MIS data for comparison in two cases. Review of the data can determine if a significant difference is observed between water quality in the North Fork (Station S001-097) compared to the South Fork (S003-638).

Total Kjeldahl Nitrogen is a measure of the organic and ammonia component of nitrogen found in the water column. Review of the data in Table A-11 does not show any significant difference between the North and South Forks.

Table A-11.Total Kjeldahl Nitrogen (mg/L) Summary

STATION_ID	Number of Samples	5th Percentile	Median	95th Percentile
North Fork Groundhouse River				
S001-097	19	0.29	0.70	1.13
South Fork Groundhouse River				
S003-638	19	0.48	0.90	1.31

Nitrate and Nitrite measure the inorganic component of nitrogen in the water column. Comparison of the data for the two sampling stations with MIS data show that the 95th percentile values found in the Groundhouse are much higher than those found in minimally impacted areas (See Table A-12) In addition, the median value for the North Fork is higher than those measured for the NCHF ecoregion. These elevated values may be related to agricultural inputs throughout the watershed. Five samples were reported as being less than detection limits (MDL) and were set to one half of MDL for the statistics analysis.

Table A-12.Nitrate and Nitrite (mg/L) Summary

STATION_ID	Number of Samples	5th Percentile	Median	95th Percentile
North Fork Groundhouse River				
S001-097	19	0.005	0.150	2.030
South Fork Groundhouse River				
S003-638	19	0.005	0.080	2.700
Minnesota Minimally Impacted Streams (1986-1992)				
North Central Hardwood Forests	-	0.01	0.08	0.48

2.8 Phosphorus

Phosphorus is most commonly the controlling nutrient required for plant growth. Elevated phosphorus levels can provide the opportunity for excessive algal growth. The phosphorus values found in the Groundhouse are consistent between the North and South Fork but are generally higher than those found in minimally impacted areas (See Table A-13). These elevated values may also be from agricultural inputs throughout the watershed.

Table A-13.Phosphorus (mg/L) Summary

STATION_ID	Number of Samples	5th Percentile	Median	95th Percentile
North Fork Groundhouse River				
S001-097	23	0.042	0.064	0.176
South Fork Groundhouse River				
S003-638	23	0.055	0.087	0.173
Minnesota Minimally Impacted Streams (1986-1992)				
North Central Hardwood Forests	-	0.01	0.07	0.13

Orthophosphate is a measure of the organic component of phosphorus found in the water column. Review of the data in Table A-14 does not show any significant difference between the North and South Forks.

Table A-14.Orthophosphate (mg/L) Summary

STATION_ID	Number of Samples	5th Percentile	Median	95th Percentile
North Fork Groundhouse River				
S001-097	16	0.016	0.027	0.094
South Fork Groundhouse River				
S003-638	16	0.017	0.035	0.115

3 Temporal Analysis

The loading of fecal coliform and sediment to the Groundhouse can vary significantly as a result of environmental conditions. A long term evaluation of fecal coliform and TSS is only possible at S001-152 as data were collected frequently since 1987. Figure A-7 displays the observed fecal coliform values observed at S001-152 since 1987.

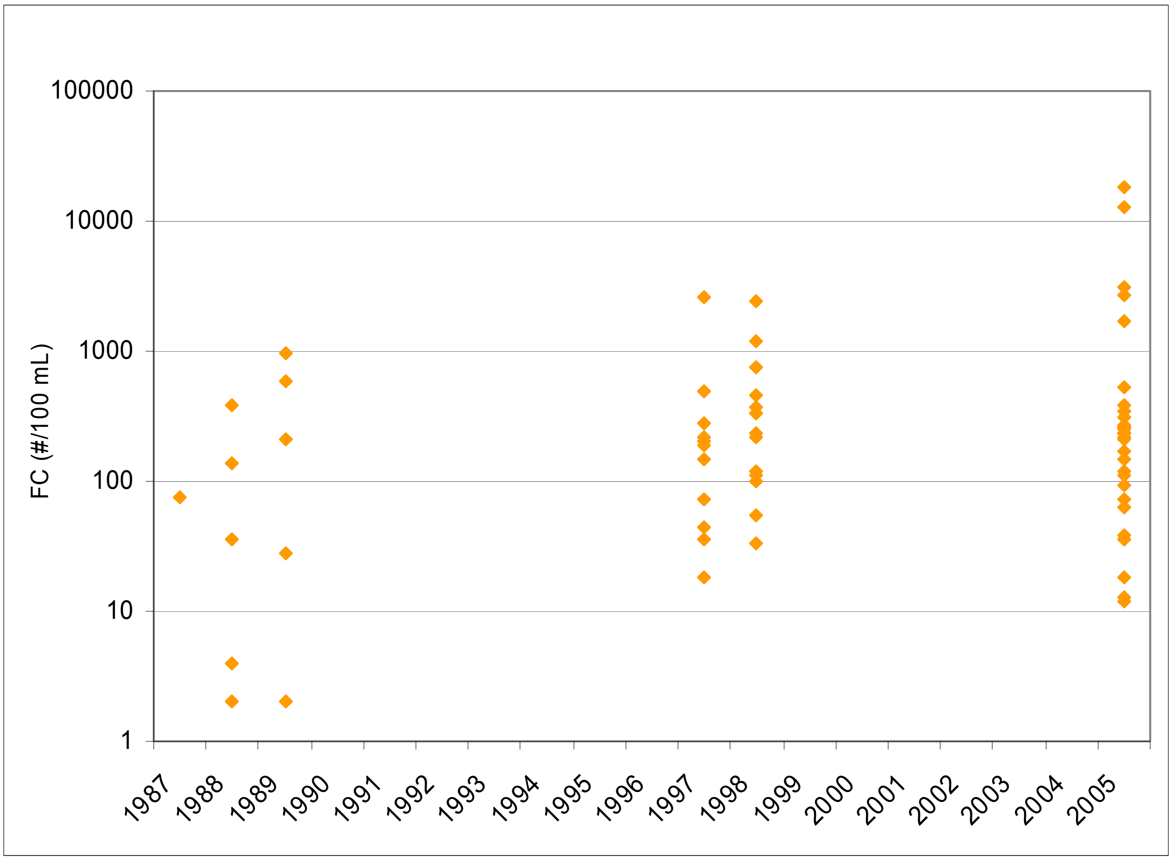

Figure A-7. Fecal Coliform Monitoring at S001-152 from 1987 through 2005

A potential trend can be seen from 1987 through 2005. While individual years may vary depending on natural conditions, the overall trend should not increase. Review of the median and geometric mean fecal coliform concentrations aggregated by the decade of sampling shown in Table A-15 do seem to support a long term increase in fecal coliform concentrations indicating some type of shift in land use or management.

Table A-15. Average Annual Fecal Coliform Concentrations

Year	Median FC (#/100mL)	FC Geometric Mean (#/100mL)
1980's	76	52.2
1990's	220	202.3
2005	230	219.5

While this trend is not conclusive, it does lend credence to the idea that changes in the watershed such as land use patterns and agricultural animal counts can affect water quality. More detailed fecal coliform analyses relating source type to pollutant concentrations are presented in Section 4.

A focus on the intensive sampling performed in 2005 show elevated fecal coliform concentrations which occasionally exceeded the 2,000 #/100mL standard.

Total suspended sediment was also reviewed for long term trends at station S001-152. The results are shown in Figure A-8 and do not suggest any long-term increases in concentrations. The TSS concentrations remain consistently below 20 mg/L.

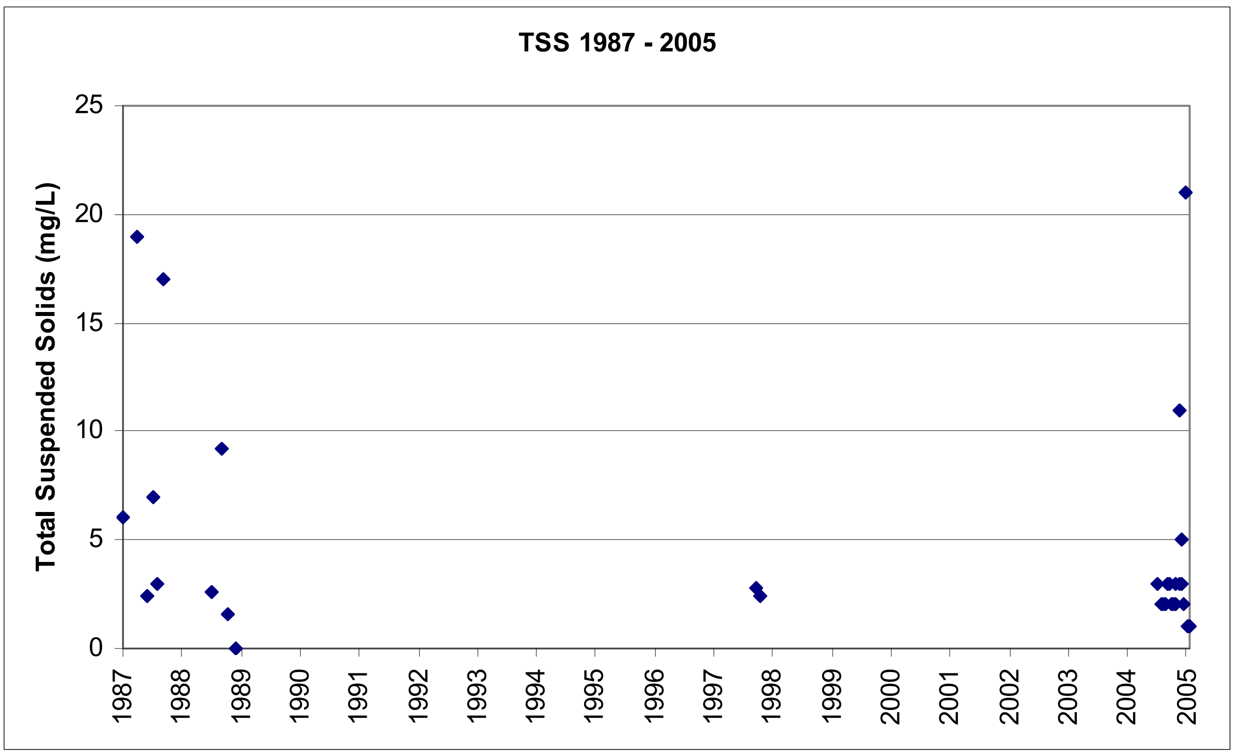

Figure A-8. TSS Monitoring at S001-152 from 1987 through 2005.

The 2005 data were also evaluated with a monthly geometric mean component to evaluate the seasonal trends which may occur at each station. It should be noted that the April and May values are based on samples sizes of less than 5 which is less than is required for evaluation of the water quality standard. The monthly geometric mean values are highly variable by month, potentially due to the occurrence of rain events during any given month.

4 Flow Duration Analyses

It is often useful to evaluate water quality observation in relation to flow regimes. This "flow duration analysis" plots water quality observations against the flow percentile as ranked from 0% to 100%. Flow gaging data were collected in the Groundhouse River from 1999 through 2005 during the recreation season (April through October). Since the fecal coliform standard is also evaluated during the recreational season, the limited flows collected in the Groundhouse are applicable and were used to develop the following fecal coliform flow duration curves. The dataset for 2004 was incomplete and was not used in the calculation of the flow exceedances. Isolated periods of missing data were filled based on the scaled flows calculated based on data collected at USGS 05338500 on the Snake River near Pine City, MN.

The flow duration analysis can help identify the flow regimes under which high values most frequently occur. For example, in Figure A-9, the geometric mean of 209 has a corresponding flow of 26 cfs which is exceeded 65% of the time. This would indicate a mix of point and nonpoint. Exceedances during low flow tend to indicate some type of direct discharge such as point sources or cattle in the stream whereas exceedances during high flows are associated with nonpoint source runoff. Flow duration curves using the geometric mean of the fecal coliform observations and record flows for the nine stations are presented in Figure A-9 through Figure A-15.

Review of the flow duration curves indicate the geometric mean standard is exceeded at least once at seven of the nine stations. Station S003-664 had the highest number of exceedances (4) with the second highest number (3) occurring at S001-152. These high coliform levels at low flows can be associated with septic systems, direct discharges, access of cattle to the stream, or possibly illicit connections. Many of the high geomean numbers are seen during moderate flows and are likely to have a mix of point and nonpoint source contributions. The exceedances seen at S001-152 seem to be very localized with only a few exceedances seen upstream at S003-641 and none upstream at S003-639 and diluted levels downstream at S001-097 or S001-099. This same pattern seemed to be followed with high levels on the South Fork at S003-664 and lower values downstream at S003-638 and S003-532.

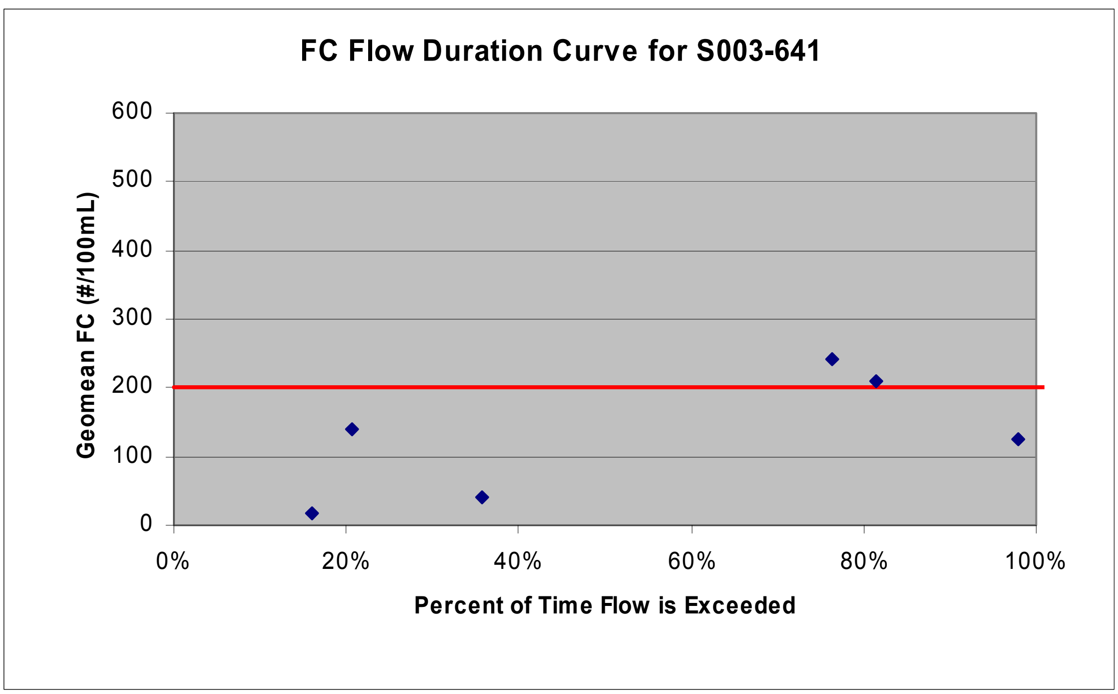

Figure A-9. Fecal Coliform Geometric Mean Flow Duration Curve for S003-641

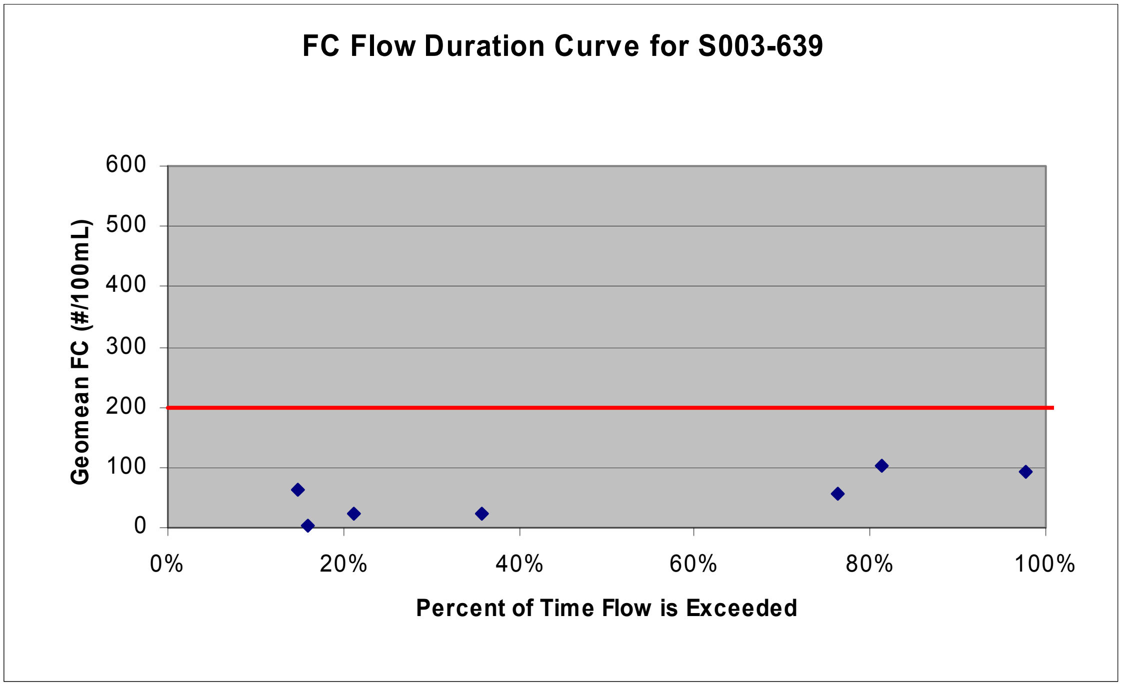

Figure A-10. Fecal Coliform Geometric Mean Flow Duration Curve for S003-639

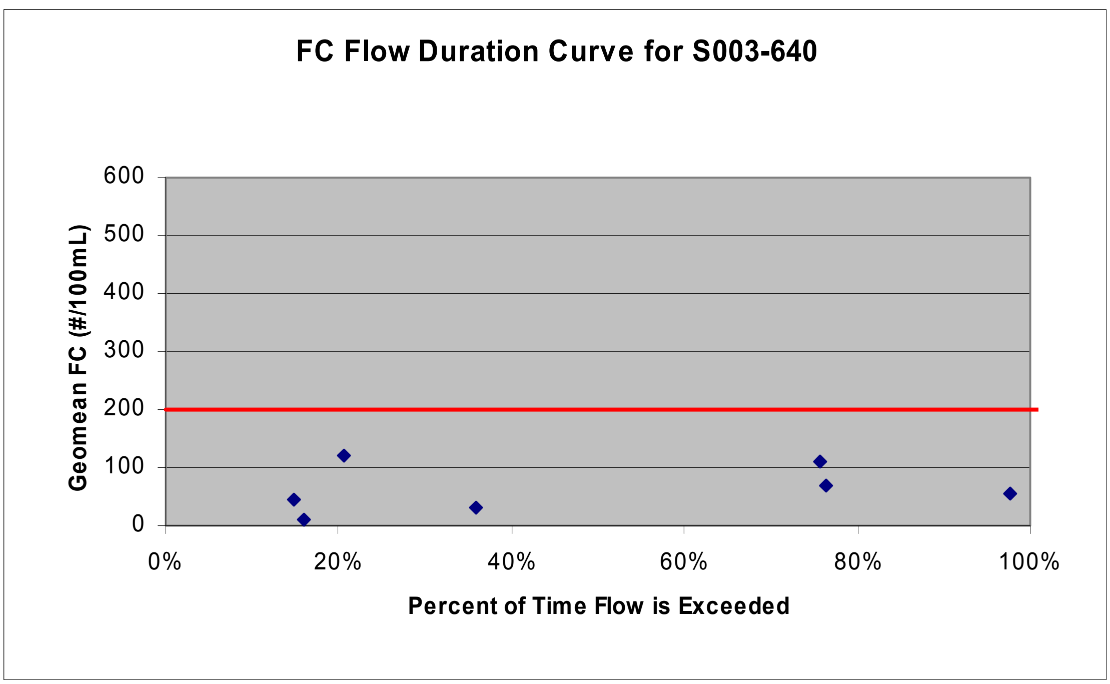

Figure A-11. Fecal Coliform Geometric Mean Flow Duration Curve for S003-640

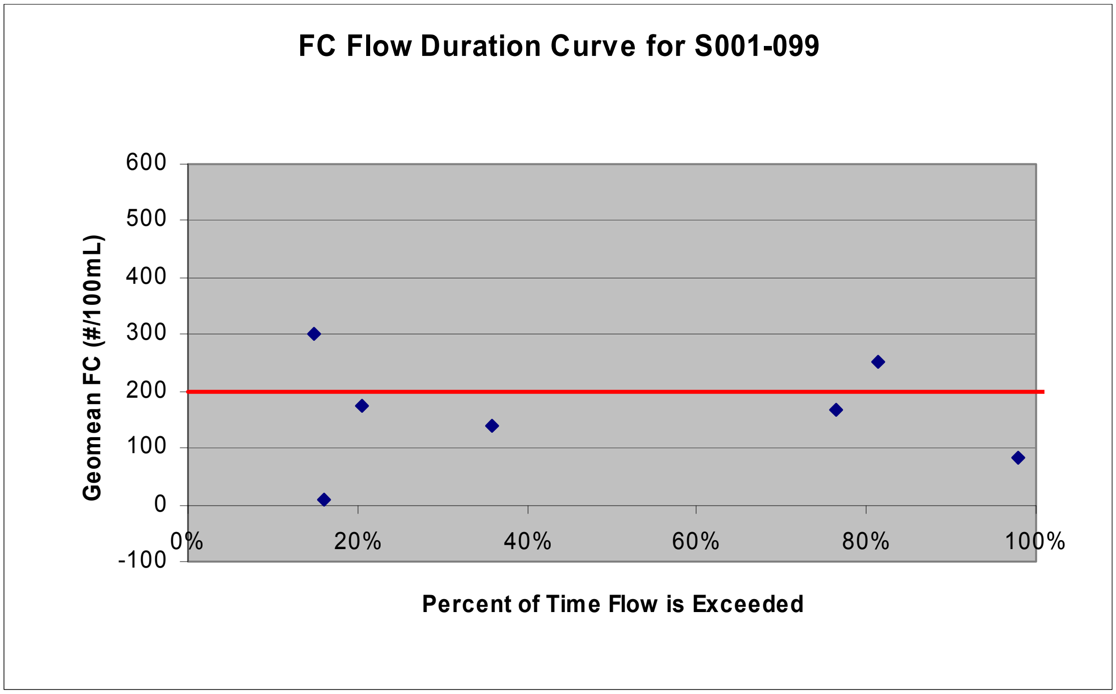

Figure A-12. Fecal Coliform Geometric Mean Flow Duration Curve for S001-099

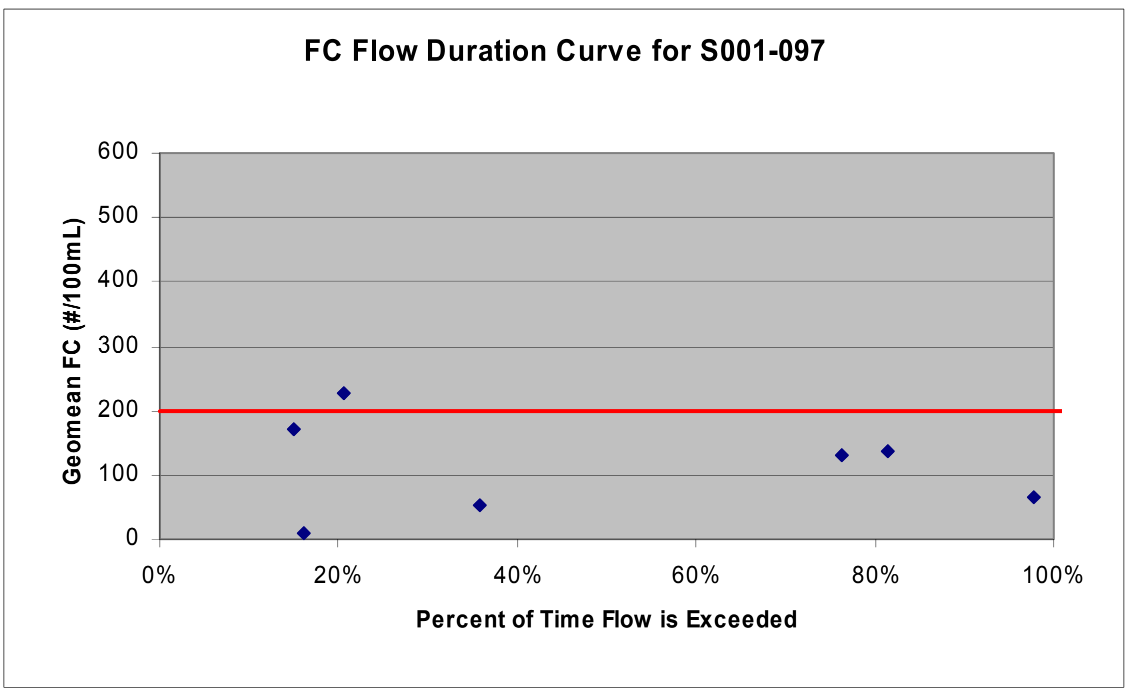

Figure A-13. Fecal Coliform Geometric Mean Flow Duration Curve for S001-097

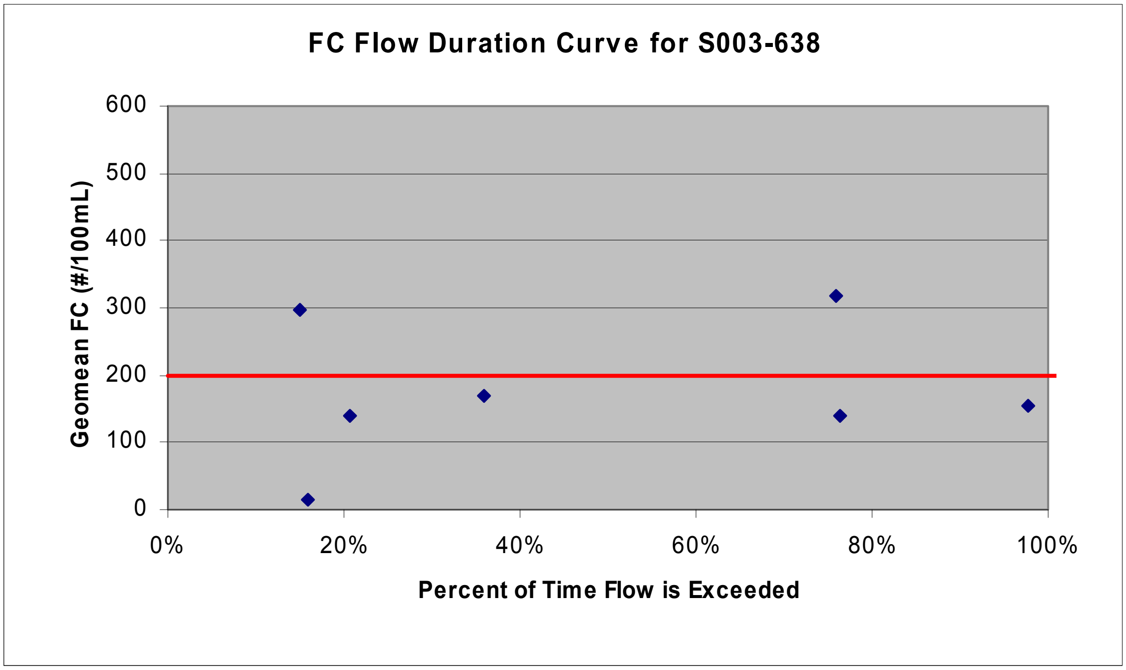

Figure A-14. Fecal Coliform Geometric Mean Flow Duration Curve for S003-638

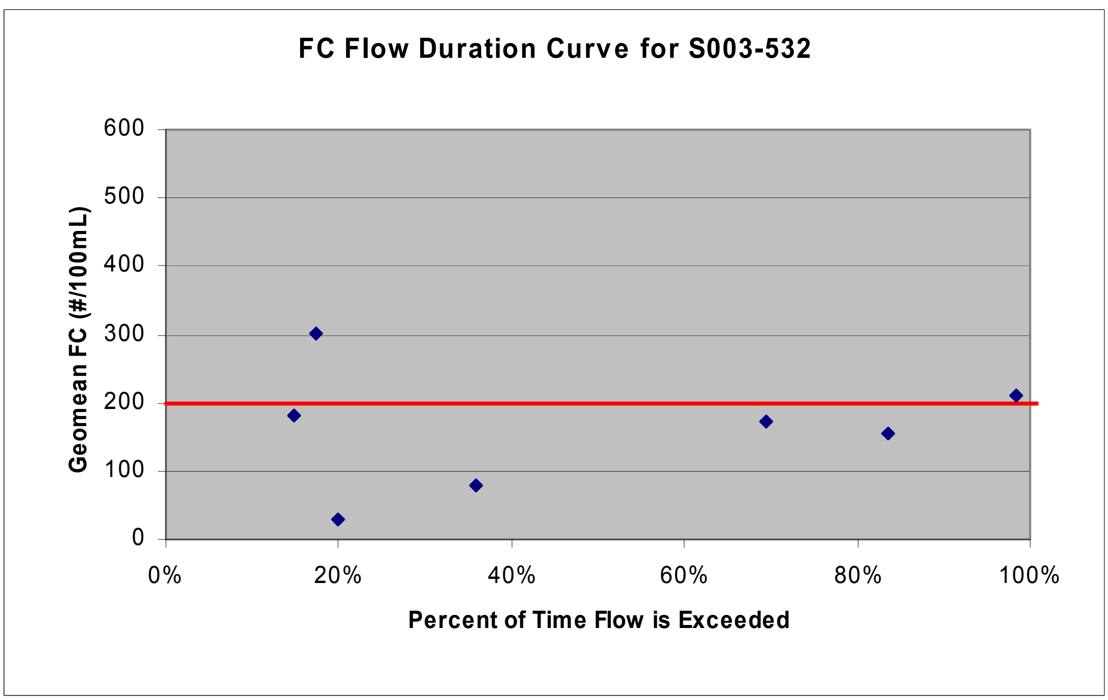

Figure A-15. Fecal Coliform Geometric Mean Flow Duration Curve for S003-532

Flow duration curves were also generated for the individual fecal coliform observations (See Figure A-16 through Figure A-22). While the single sample water quality standard of 2000 #/100 mL is infrequently exceeded, it is informative to identify the flows during which extreme values are recorded. For example, at Station S003-641 an elevated value of 10,000 is recorded during a high flow event. Subsequent observations during the same month are low and an exceedance of the geometric mean standard does not occur. A similar pattern of exceedances during both high and moderate flows is seen in both the geometric mean and individual point flow duration curves.

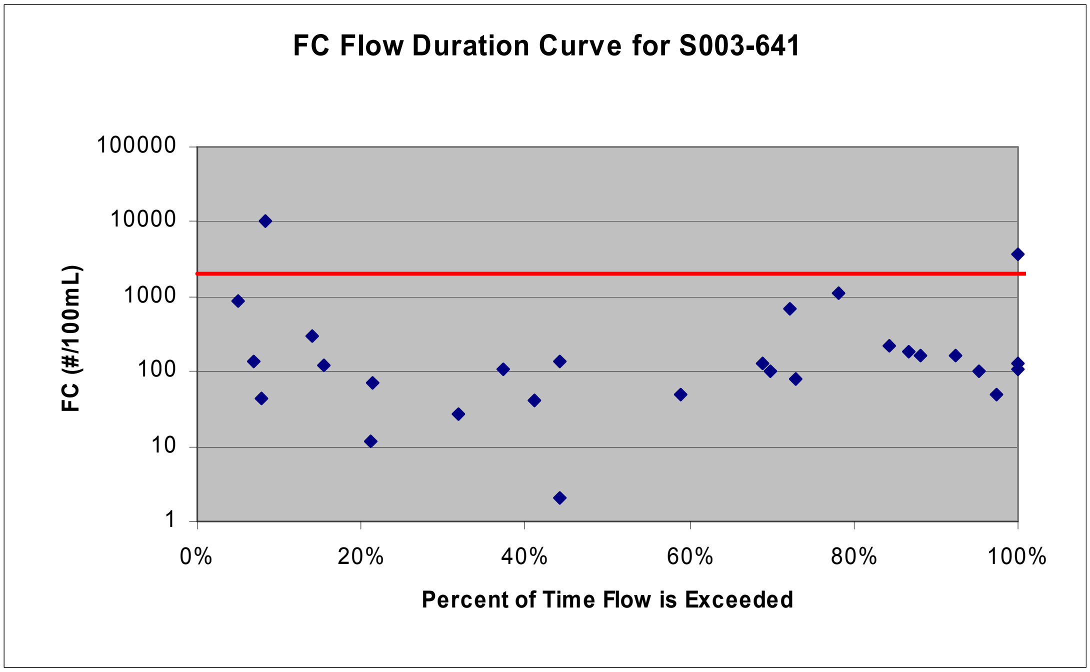

Figure A-16. Fecal Coliform Flow Duration Curve for S003-641

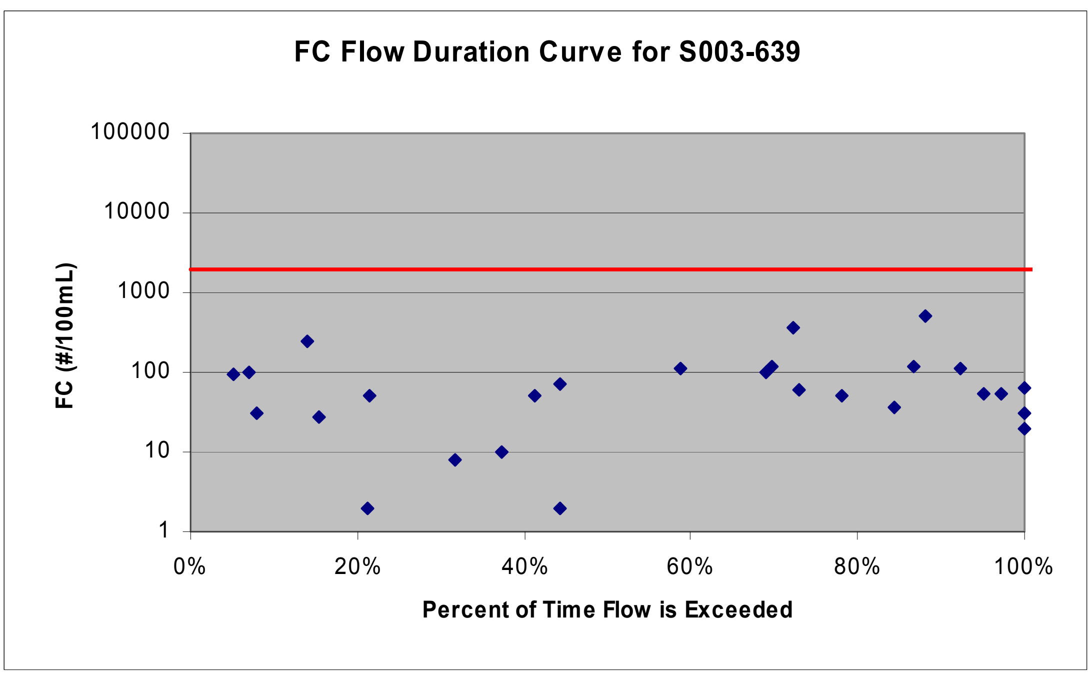

Figure A-17. Fecal Coliform Flow Duration Curve for S003-639

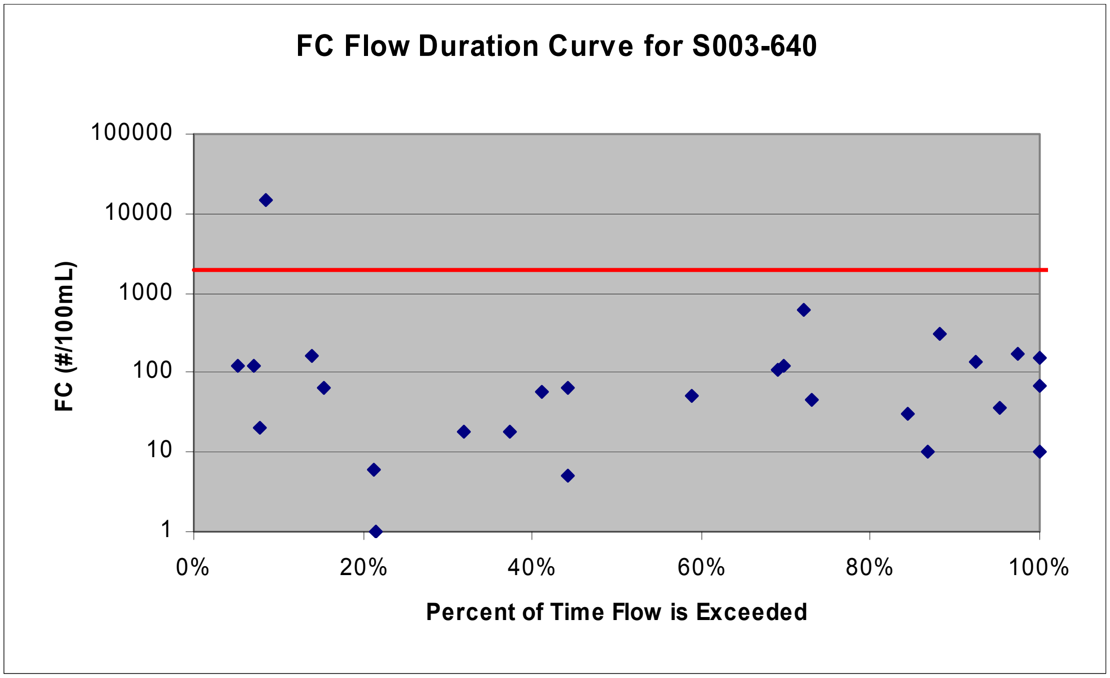

Figure A-18. Fecal Coliform Flow Duration Curve for S003-640

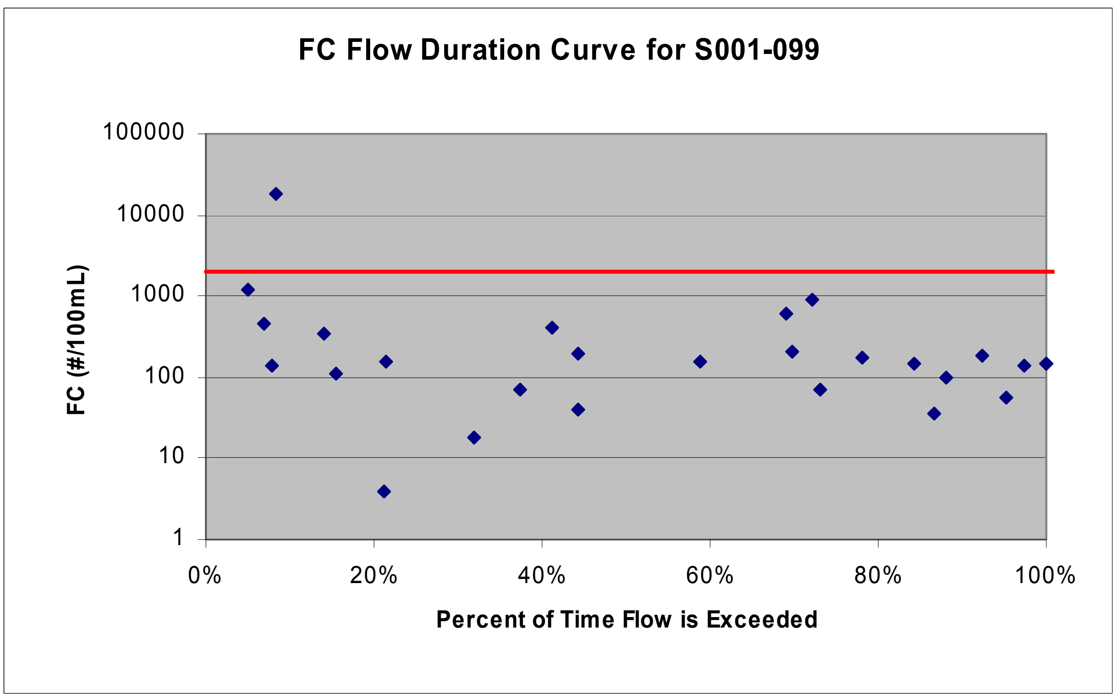

Figure A-19. Fecal Coliform Flow Duration Curve for S001-099

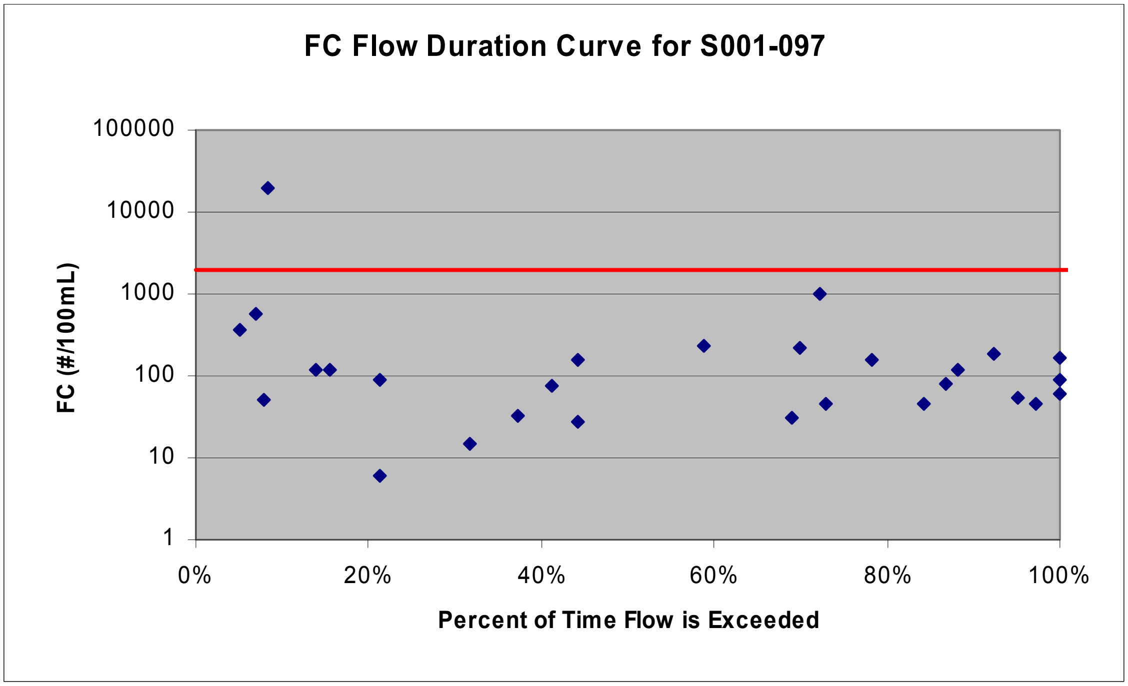

Figure A-20. Fecal Coliform Flow Duration Curve for S001-097

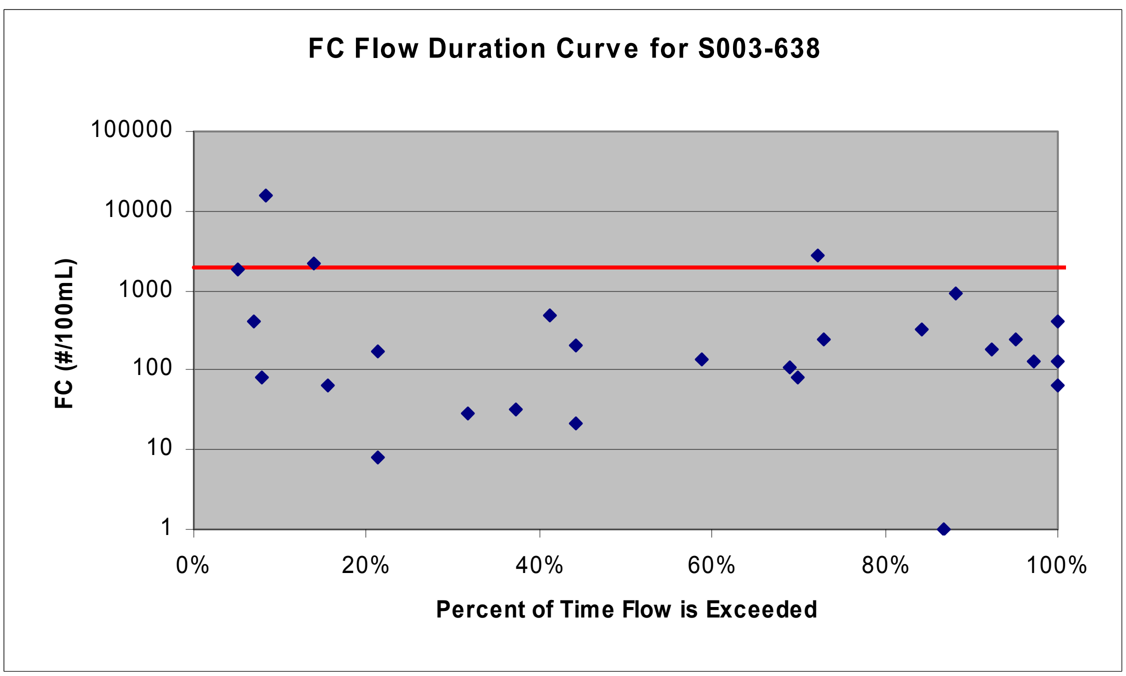

Figure A-21. Fecal Coliform Flow Duration Curve for S003-638

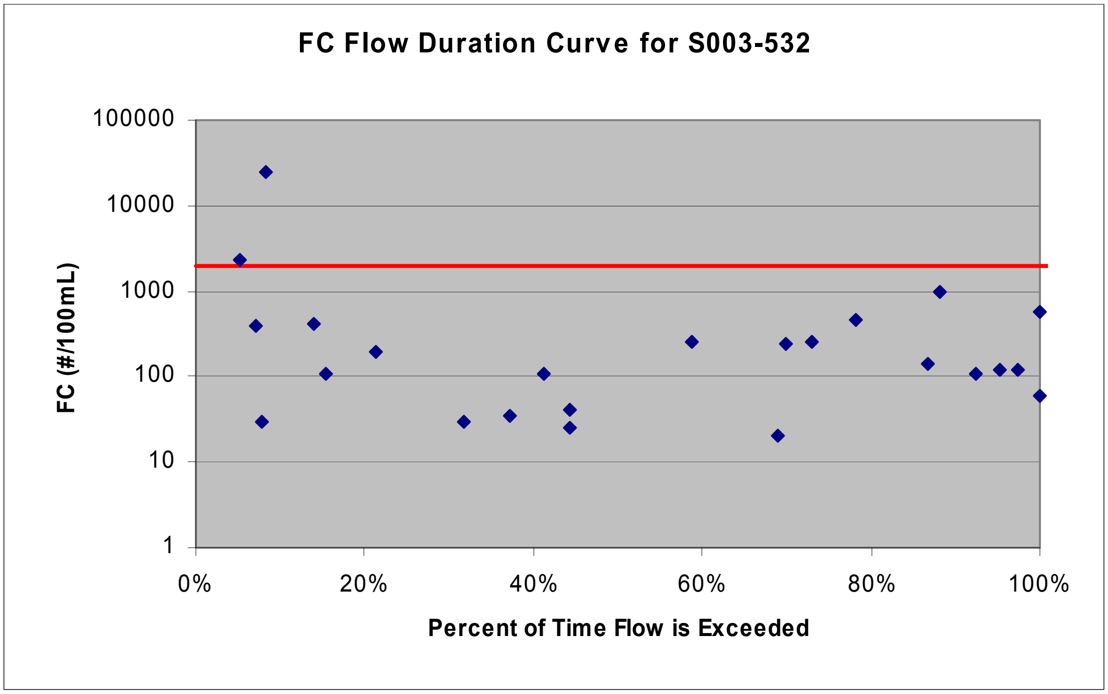

Figure A-22. Fecal Coliform Flow Duration Curve for S003-532

TSS and Nutrient Flow Duration Plots

TSS values were also plotted on a flow duration basis to determine under what conditions sediment was being loaded and transported to the stream. For TSS, TN, and TP, the full range of flows seen during a year are of interest. Flow gaging data was not collected in the Groundhouse River for full years. For this reason, an area-weighted estimate was calculated based on data collected at USGS 05338500 on the Snake River near Pine City, MN.

As seen in Figure A-23 through Figure A-31, TSS levels are consistently low at all stations with almost all the higher readings observed during high flows occurring during a storm event on October 5, 2005. This is not surprising since sediment is most often transported during storm events. However it does suggest that other nonpoint source contributions associated with low flows (such as cattle access to streams) is not a significant problem for erosion except near station S001-152. A number of elevated observations of TSS occur at S001-152 during a variety of flow conditions, supporting the findings of the fecal coliform flow duration analysis that some type of local discharge or disturbance, such as animals in the stream, may occur in this area.

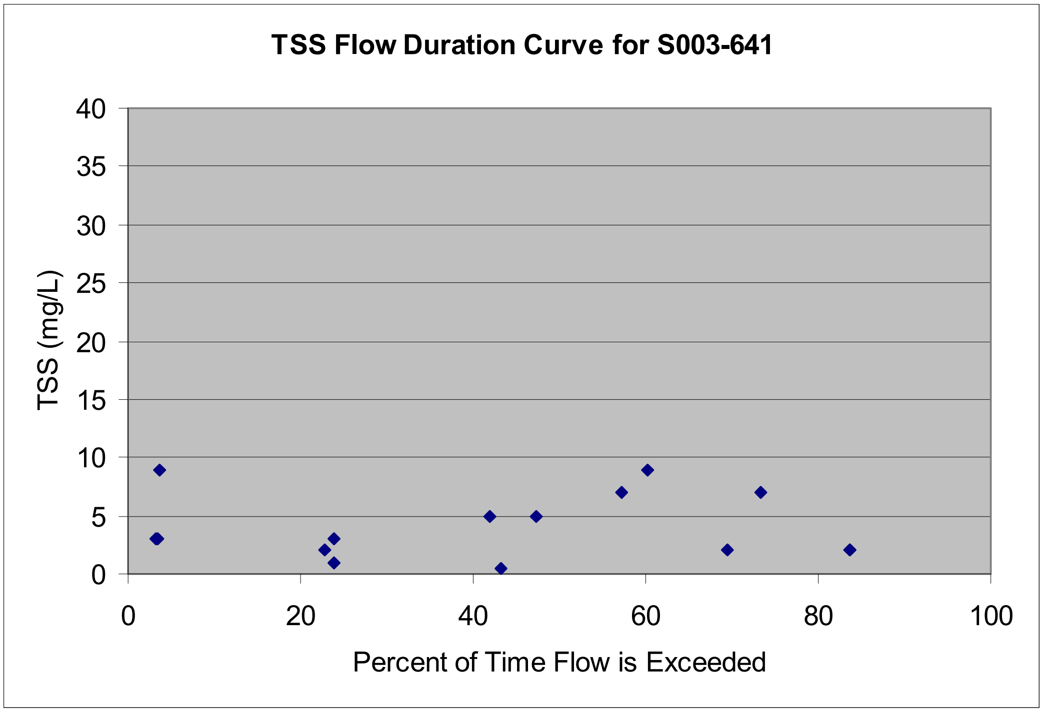

Figure A-23. TSS Flow Duration Curve for S003-641

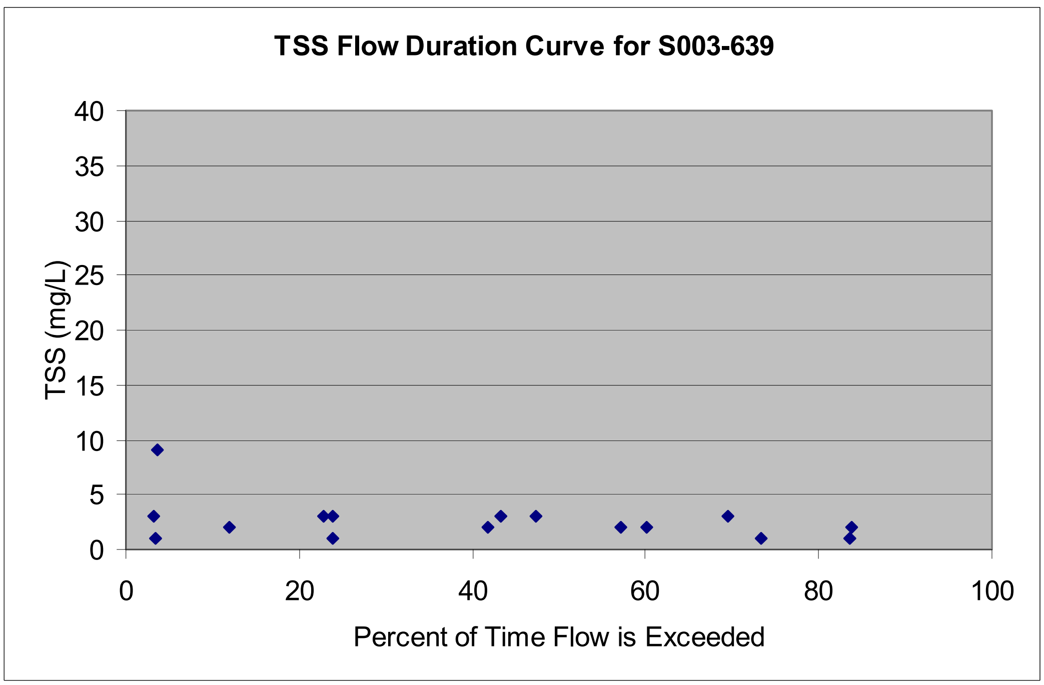

Figure A-24. TSS Flow Duration Curve for S003-639

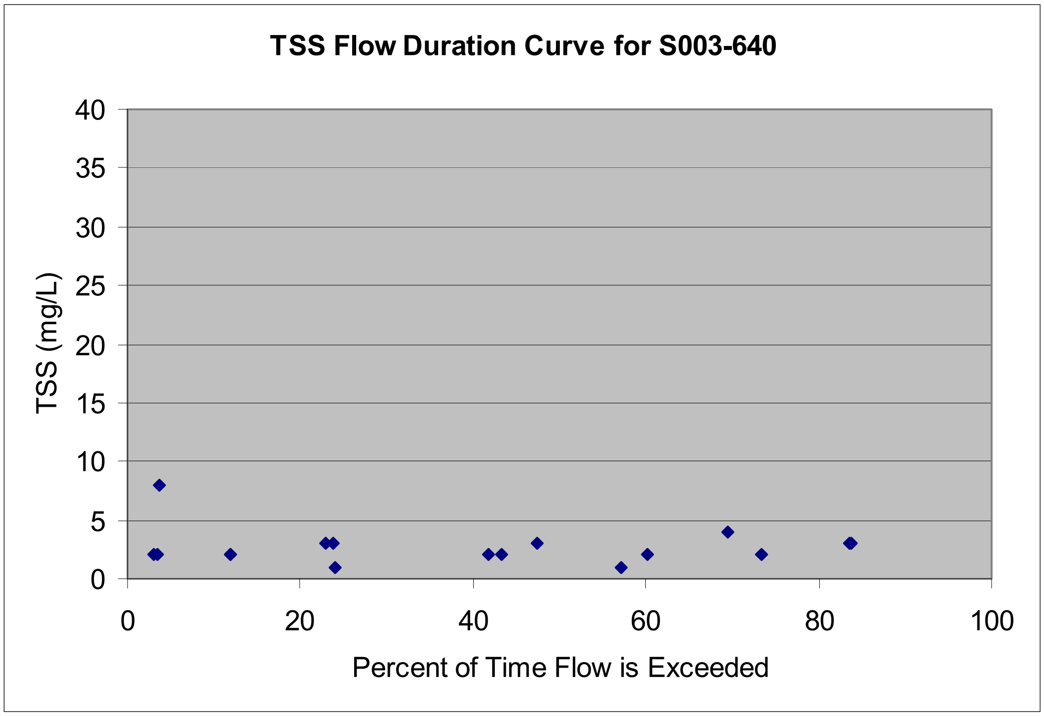

Figure A-25. TSS Flow Duration Curve for S003-640

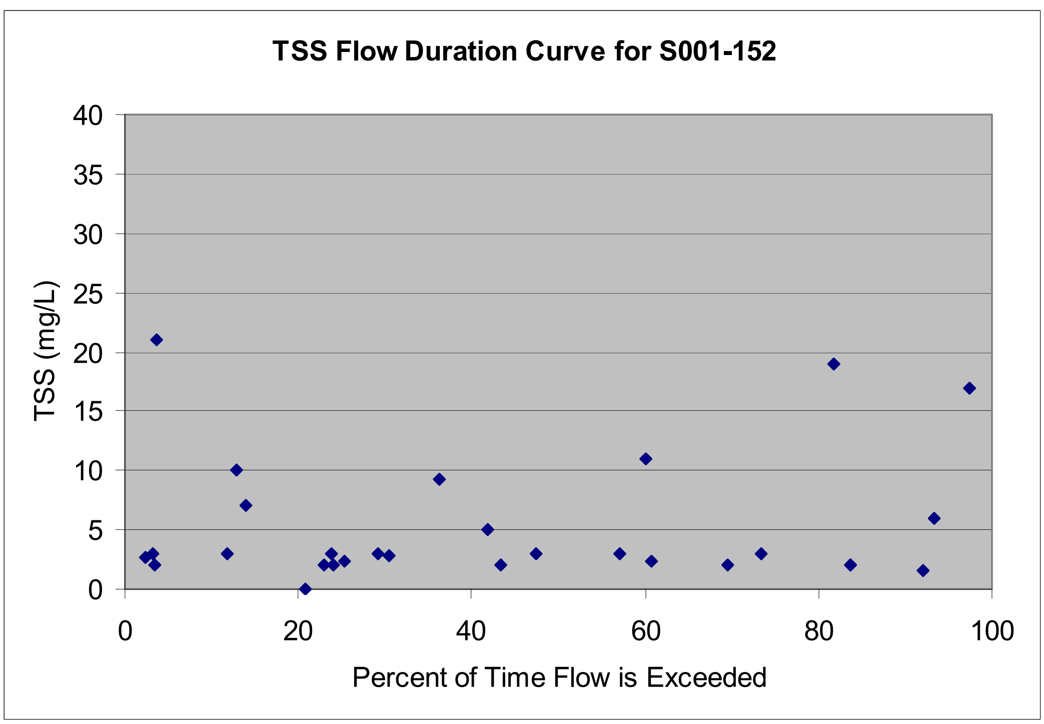

Figure A-26. TSS Flow Duration Curve for S001-152

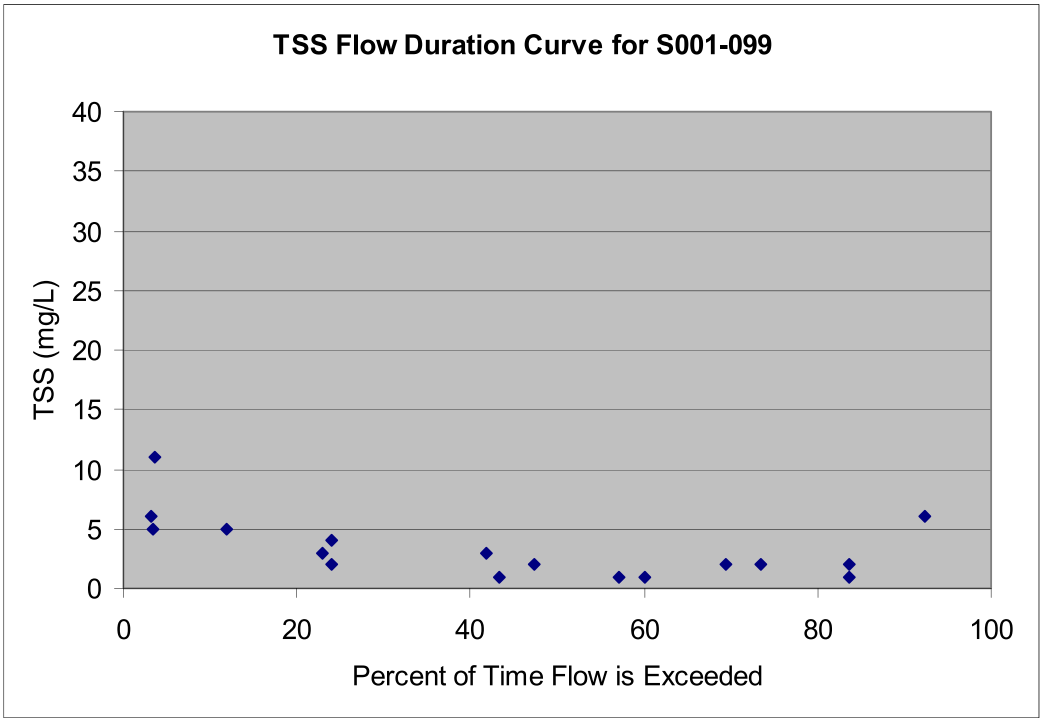

Figure A-27. TSS Flow Duration Curve for S001-099

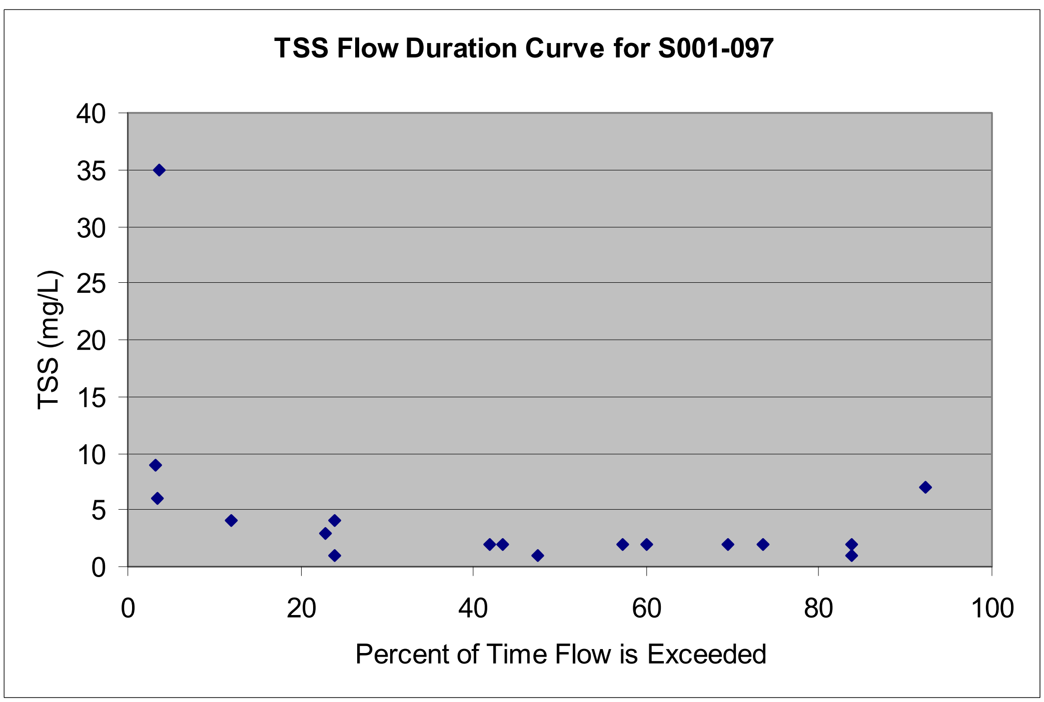

Figure A-28. TSS Flow Duration Curve for S001-097

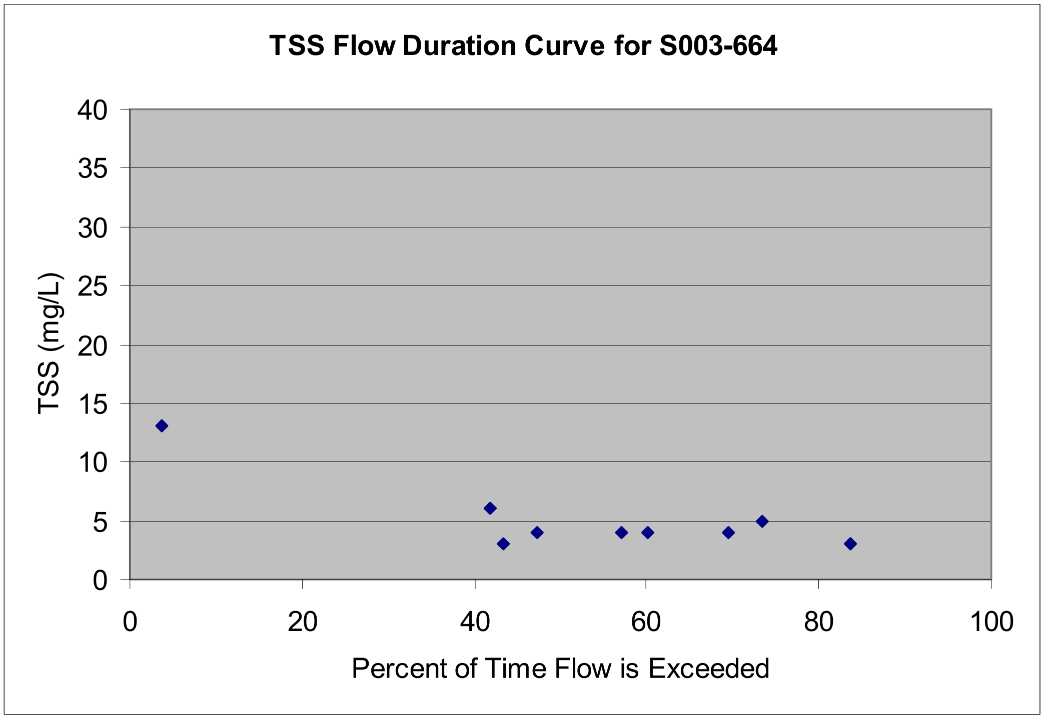

Figure A-29. TSS Flow Duration Curve for S003-664

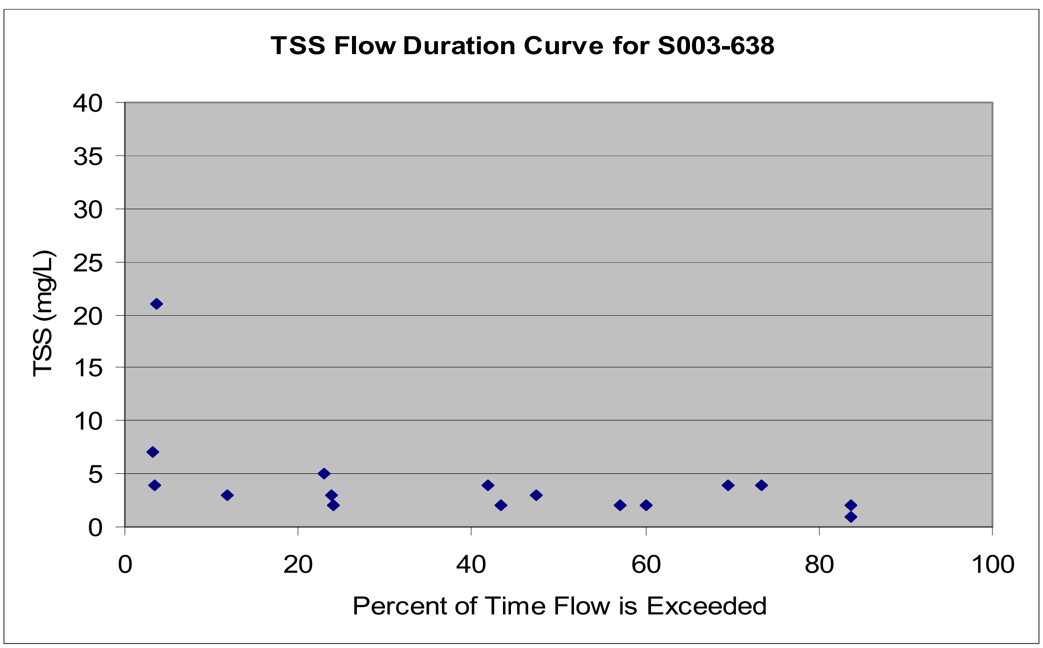

Figure A-30. TSS Flow Duration Curve for S003-638

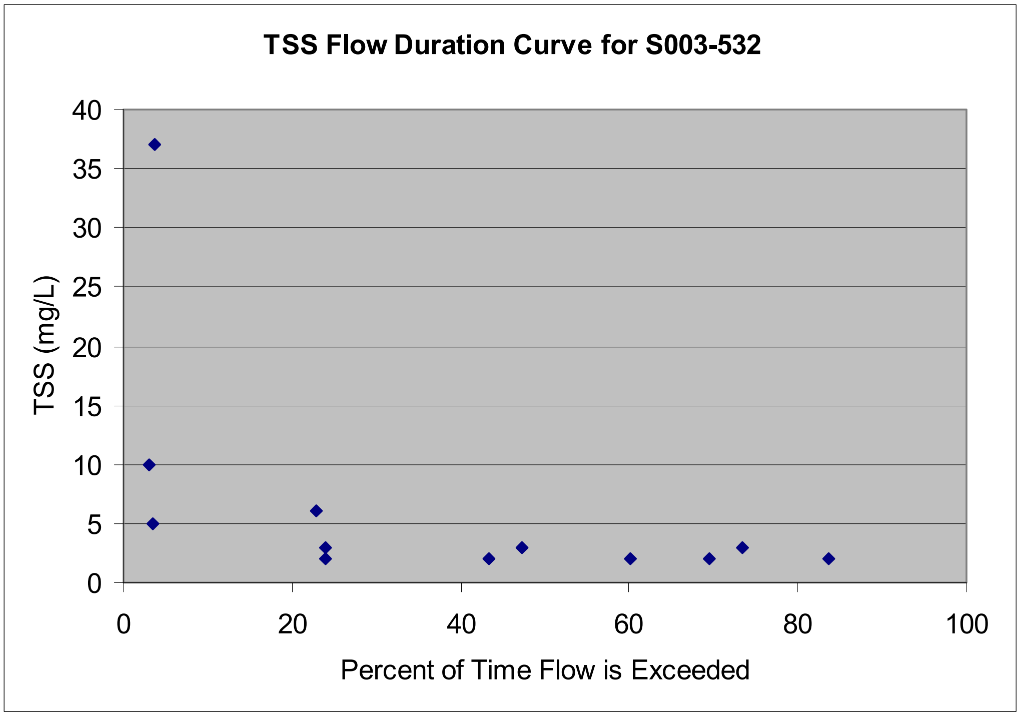

Figure A-31. TSS Flow Duration Curve for S003-532

Nutrient observation data were also evaluated relative to the flow conditions under which they were collected. Total phosphorus and orthophosphate flow duration curves were developed and are shown in Figure A-32 through Figure A-35. As expected, elevated phosphorus levels are associated with higher flow events as a result of stormwater runoff of surface phosphorus loads as well as phosphorus associated with sediment that is mobilized during the higher surface flows.

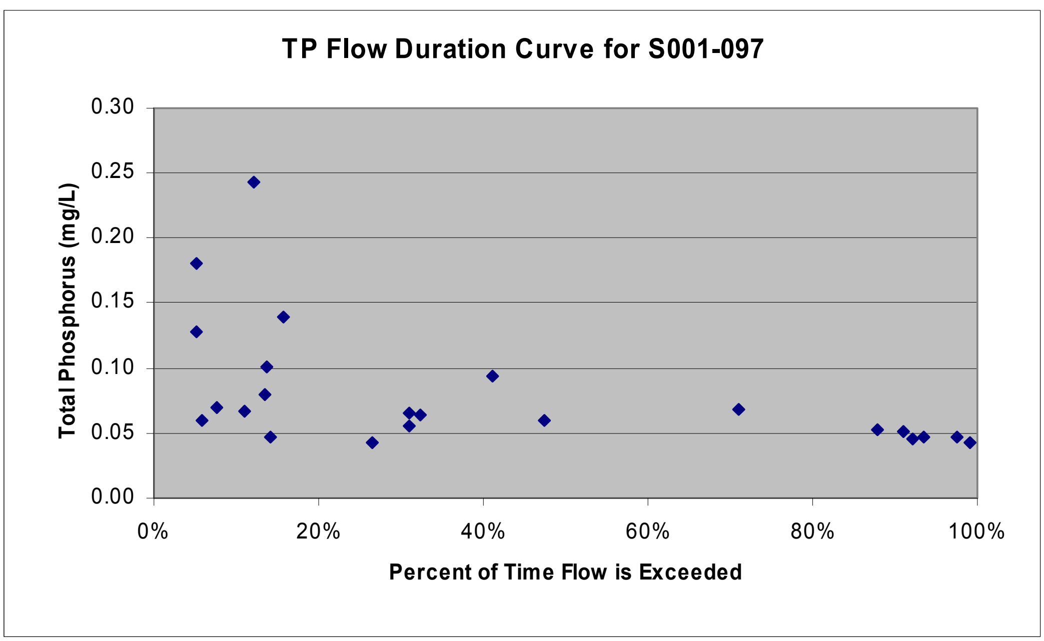

Figure A-32. TP Flow Duration Curve for S001-097

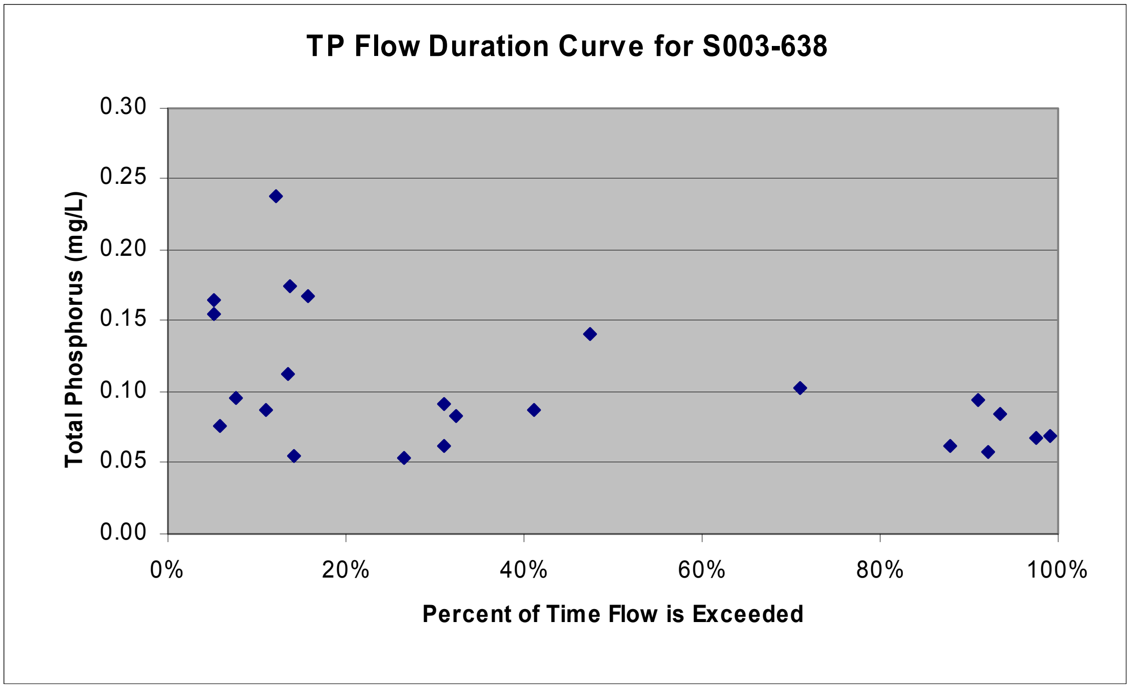

Figure A-33. TP Flow Duration Curve for S003-638

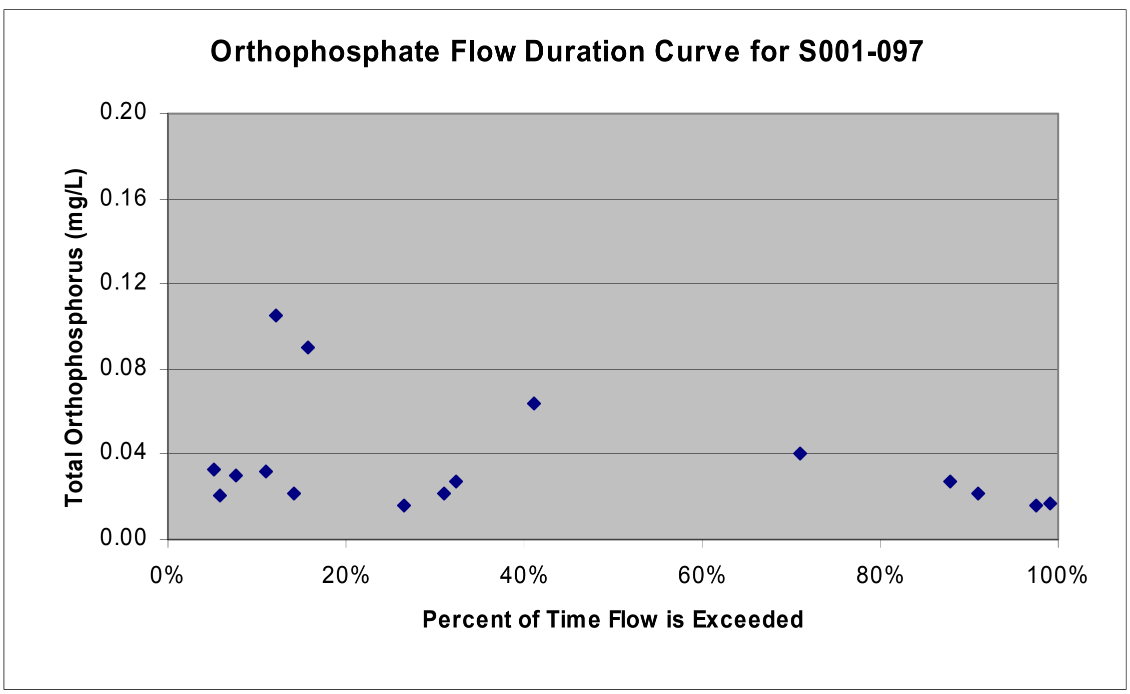

Figure A-34. TOP Flow Duration Curve for S001-097

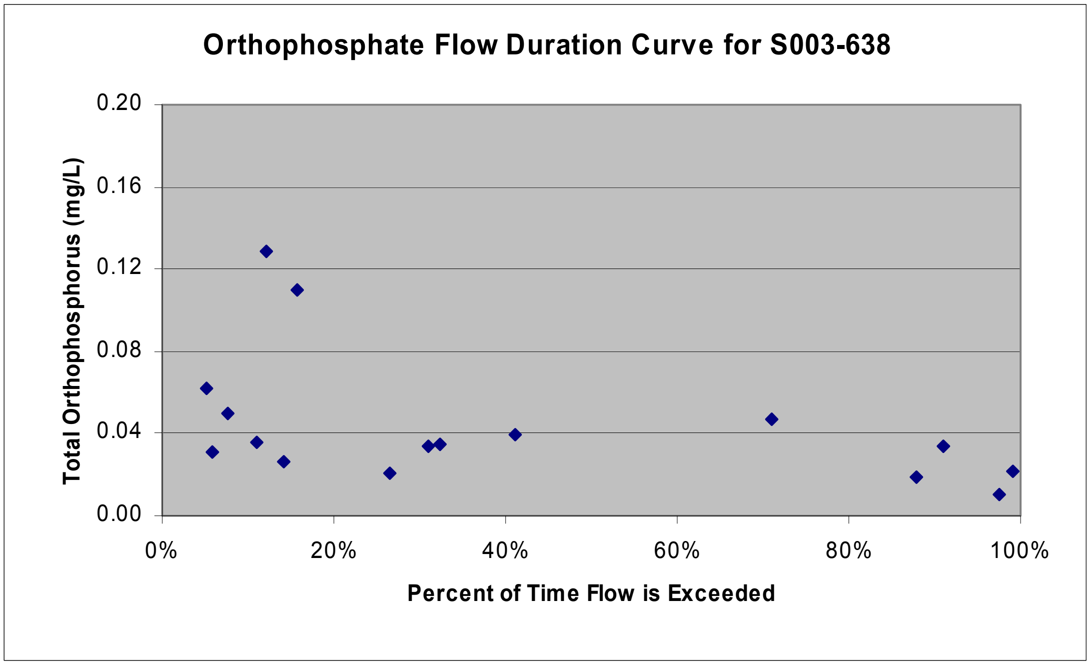

Figure A-35. TOP Flow Duration Curve for S003-638

Flow duration curves for the nitrogen species were developed and are shown in Figure A-36 through Figure A-48. In many cases the nitrogen data were only collected at S001-097 and S003-638. The curves generally show fairly low values during all flow regimes with a slight elevation during storm events. The exceptions to this rule are the NOx curves which show a strong bias towards elevated values during low flow events. This is possibly due to the dominance of groundwater during these periods. Organic fertilizer can be transformed into nitrate and nitrite and transported via groundwater pathways. Another potential source during dry flows is failing septic systems.

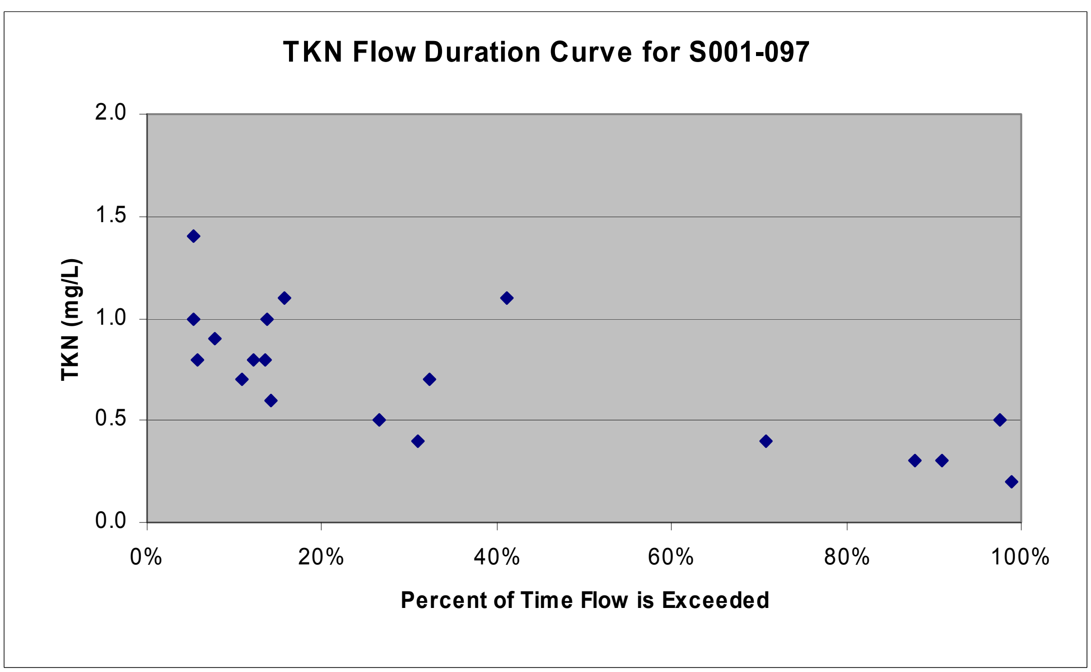

Figure A-36. TKN Flow Duration Curve for S001-097

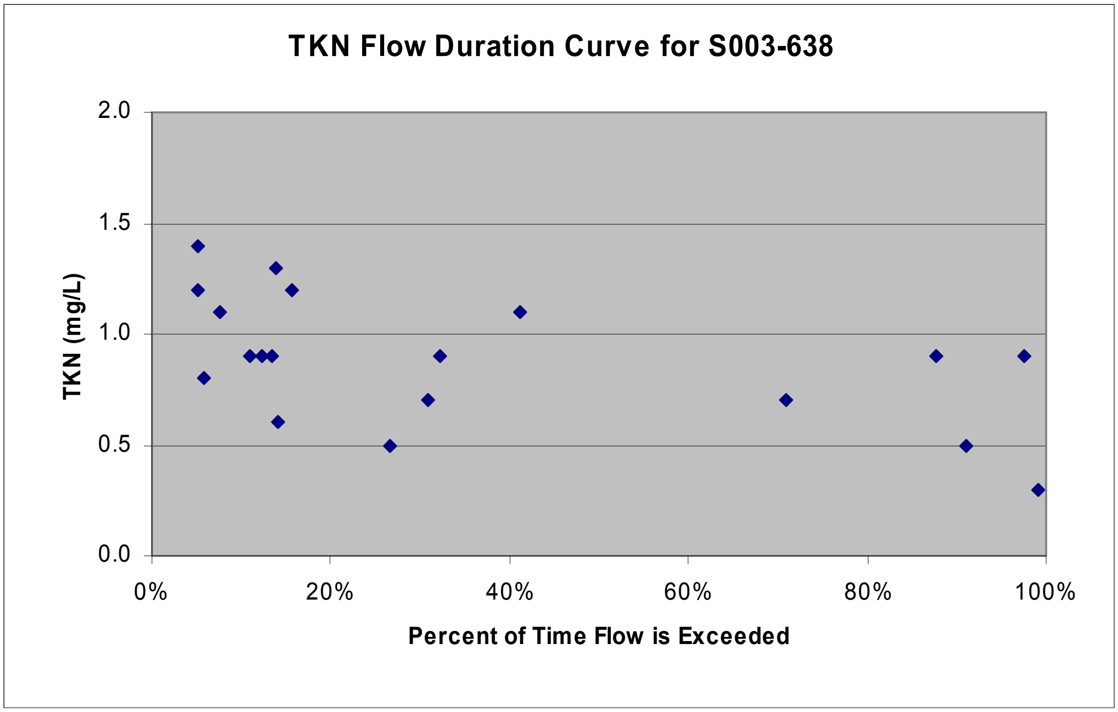

Figure A-37. TKN Flow Duration Curve for S003-638

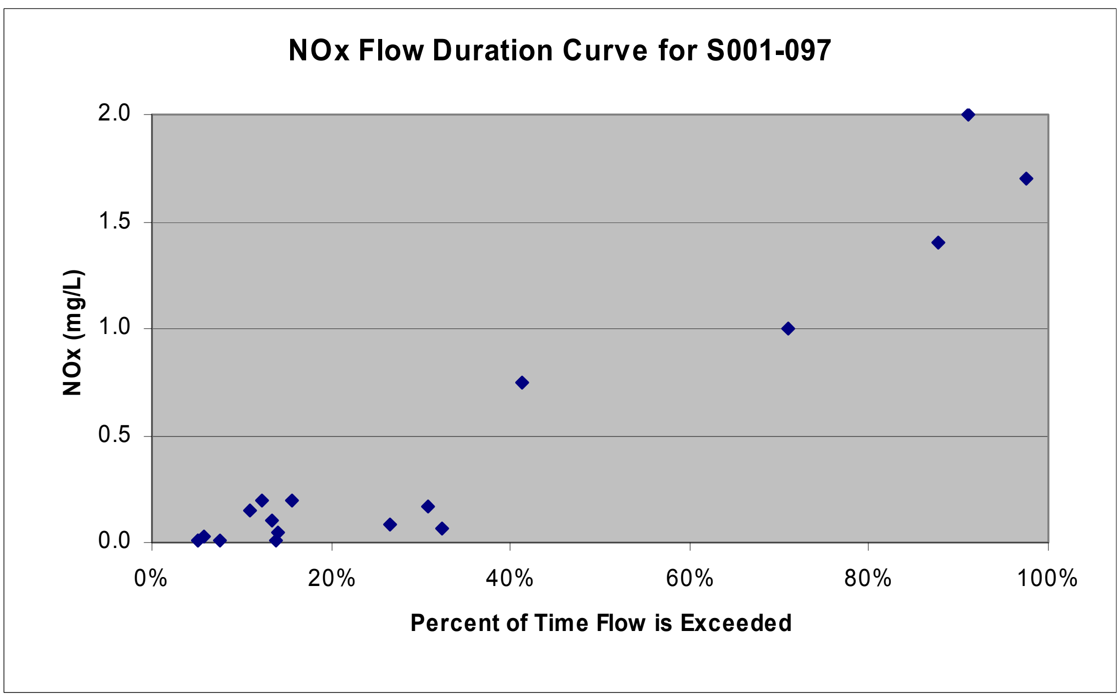

Figure A-38. NOx Flow Duration Curve for S001-097

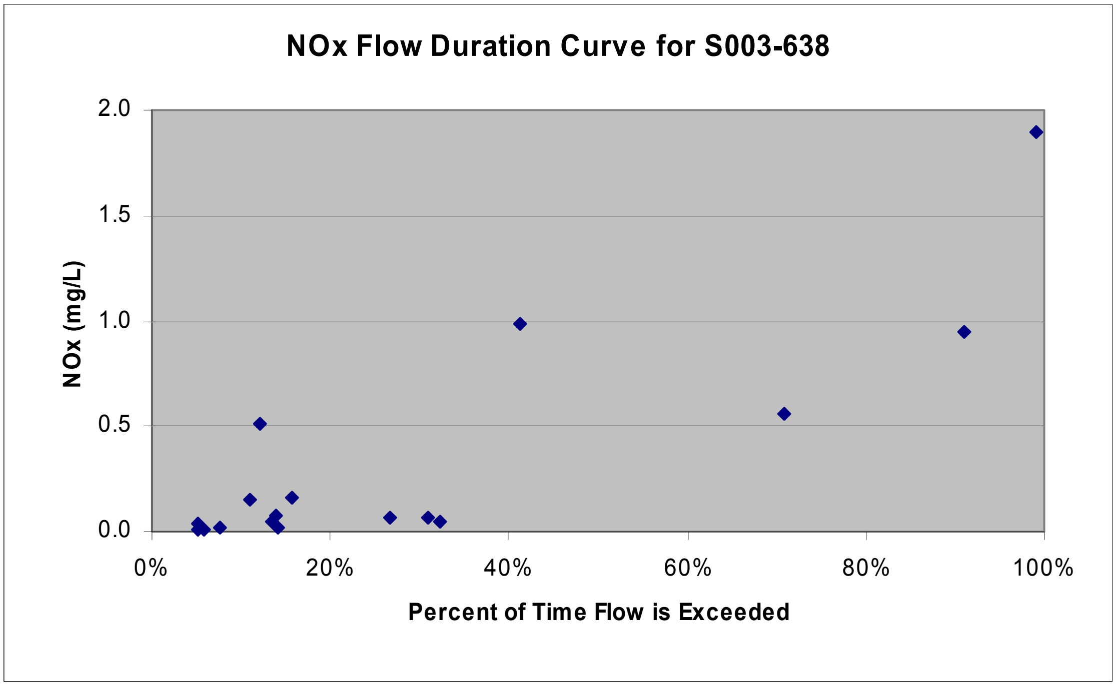

Figure A-39. NOx Flow Duration Curve for S003-638

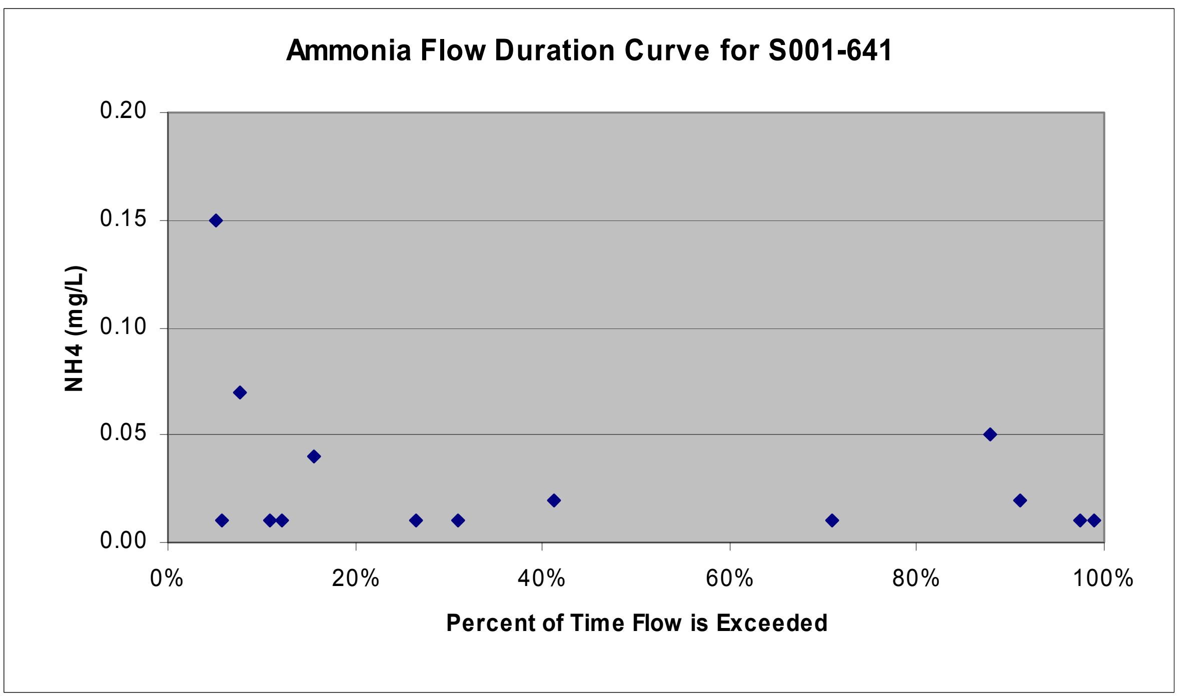

Figure A-40. Ammonia Flow Duration Curve for S003-641

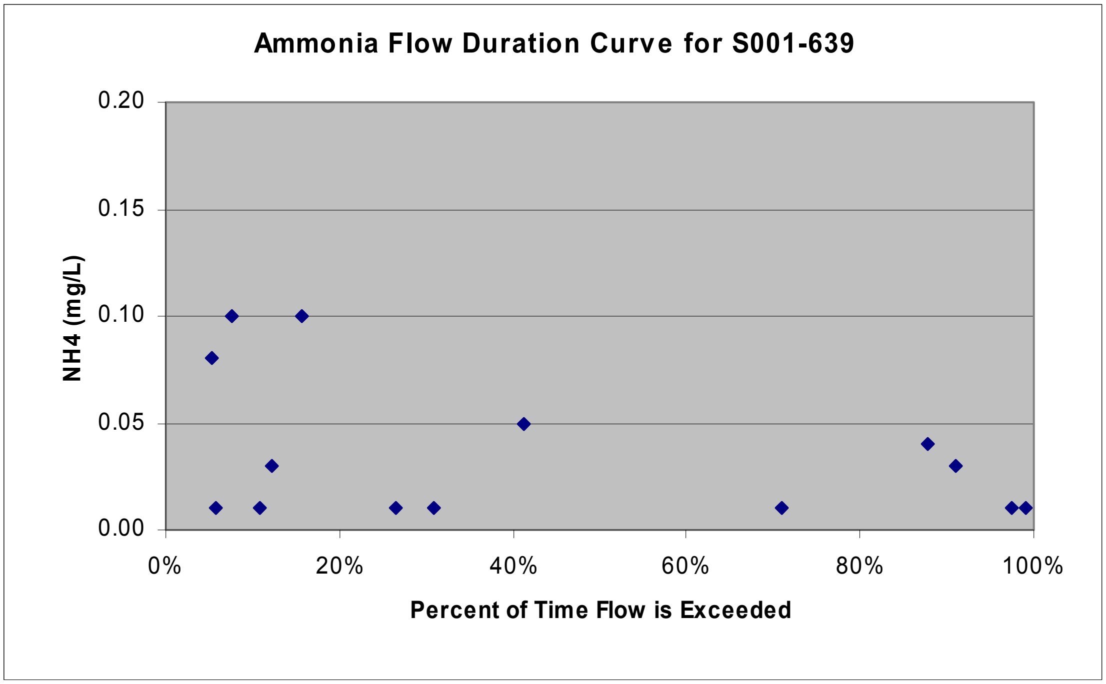

Figure A-41. Ammonia Flow Duration Curve for S003-639

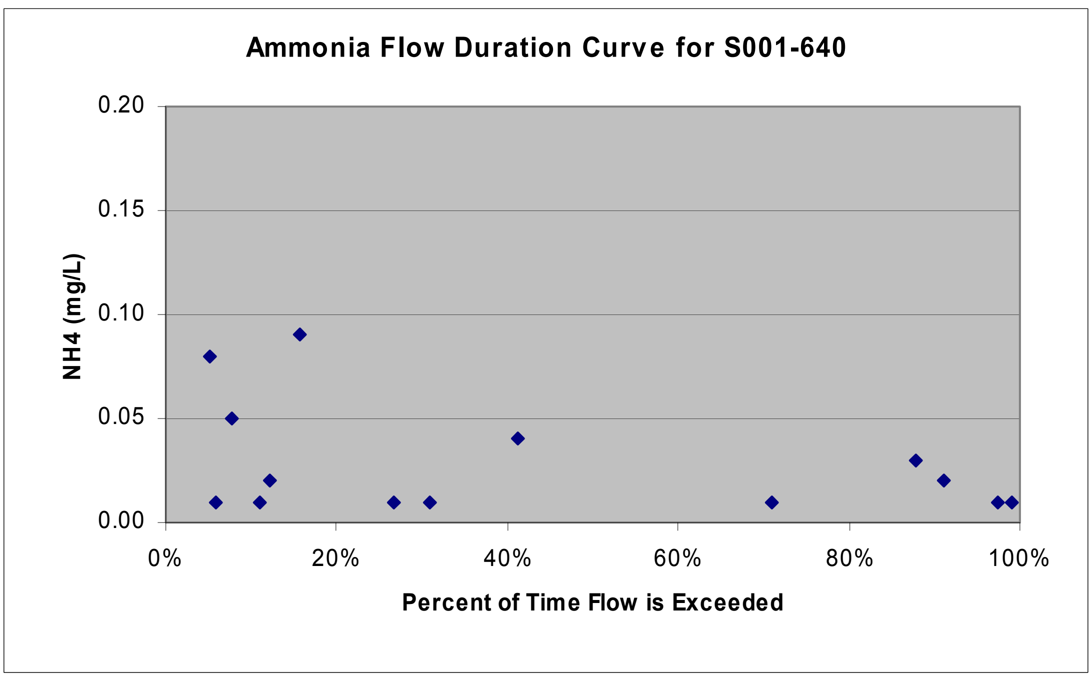

Figure A-42. Ammonia Flow Duration Curve for S003-640

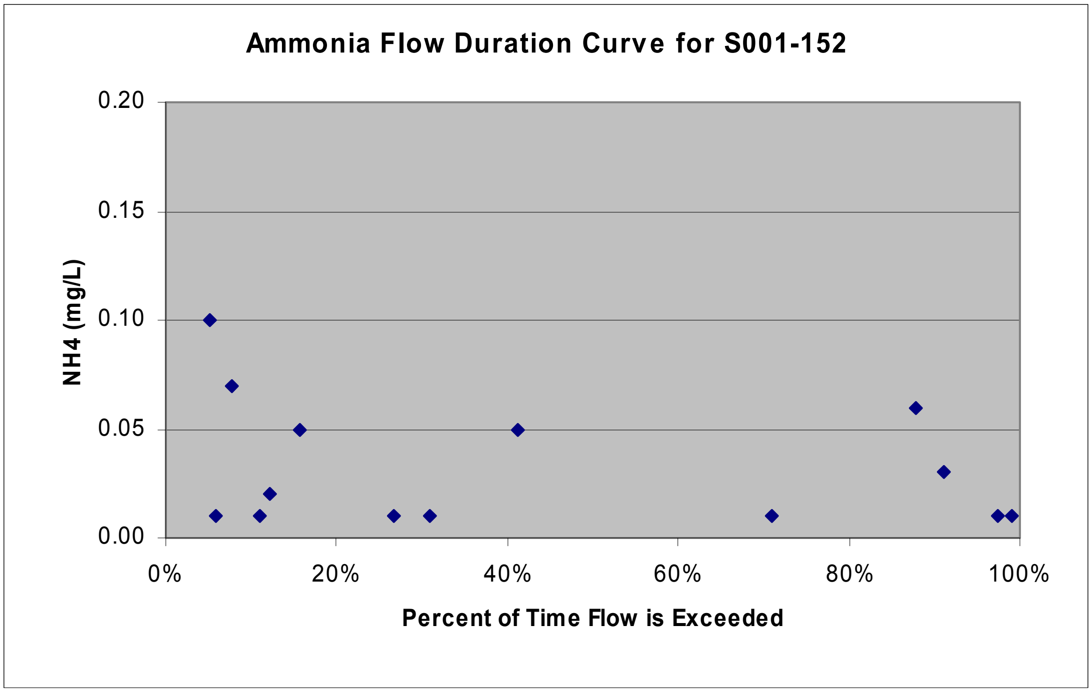

Figure A-43. Ammonia Flow Duration Curve for S003-152

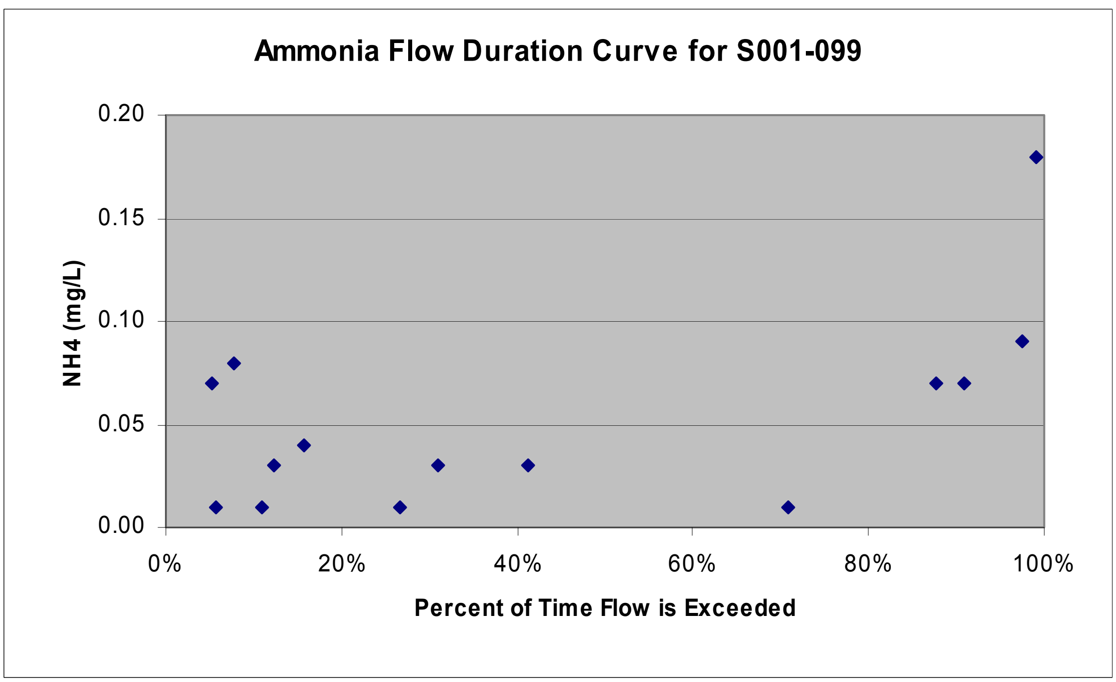

Figure A-44. Ammonia Flow Duration Curve for S001-099

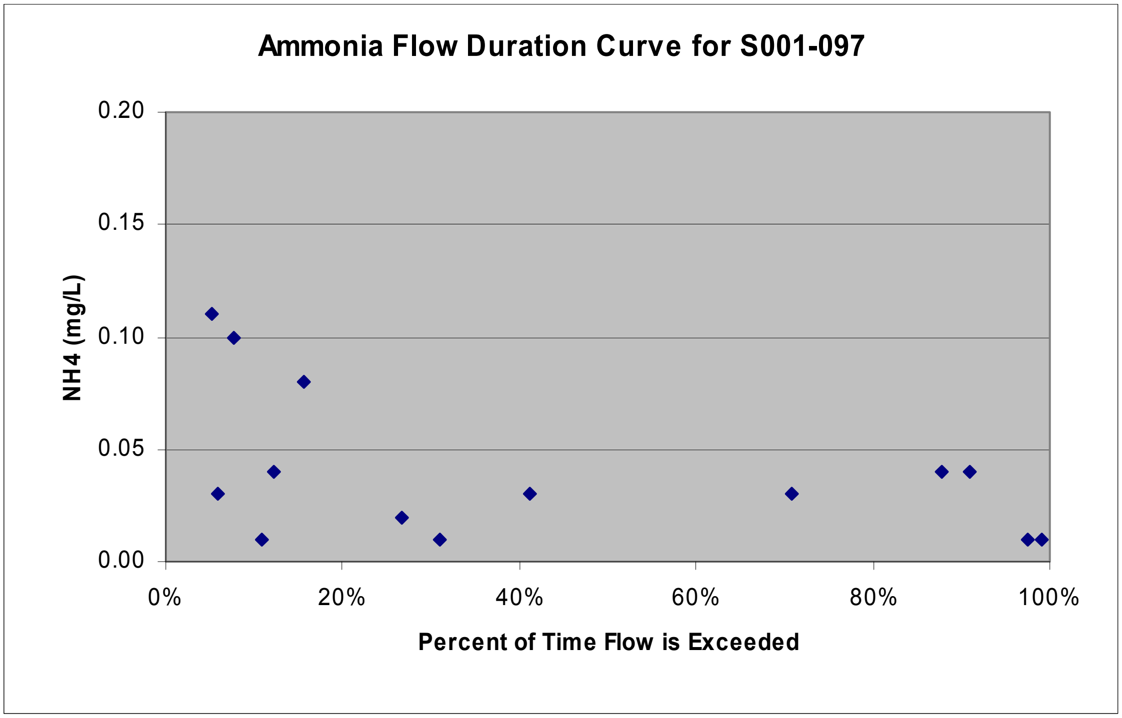

Figure A-45. Ammonia Flow Duration Curve for S001-097

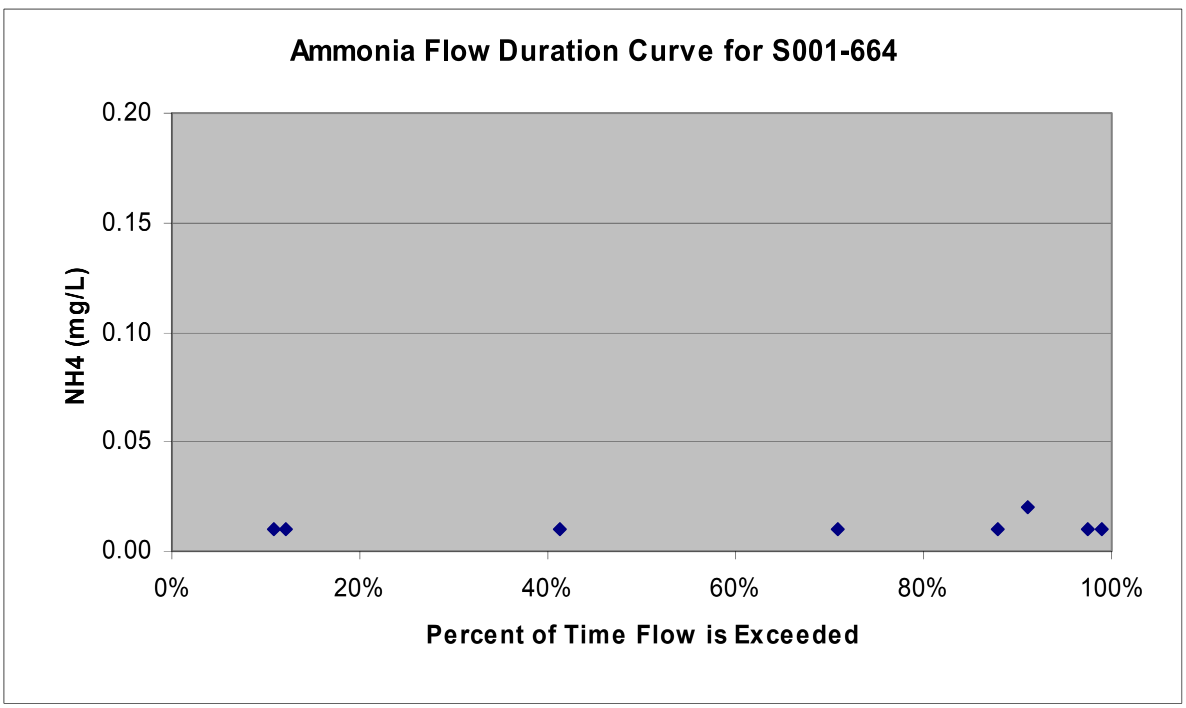

Figure A-46. Ammonia Flow Duration Curve for S003-664

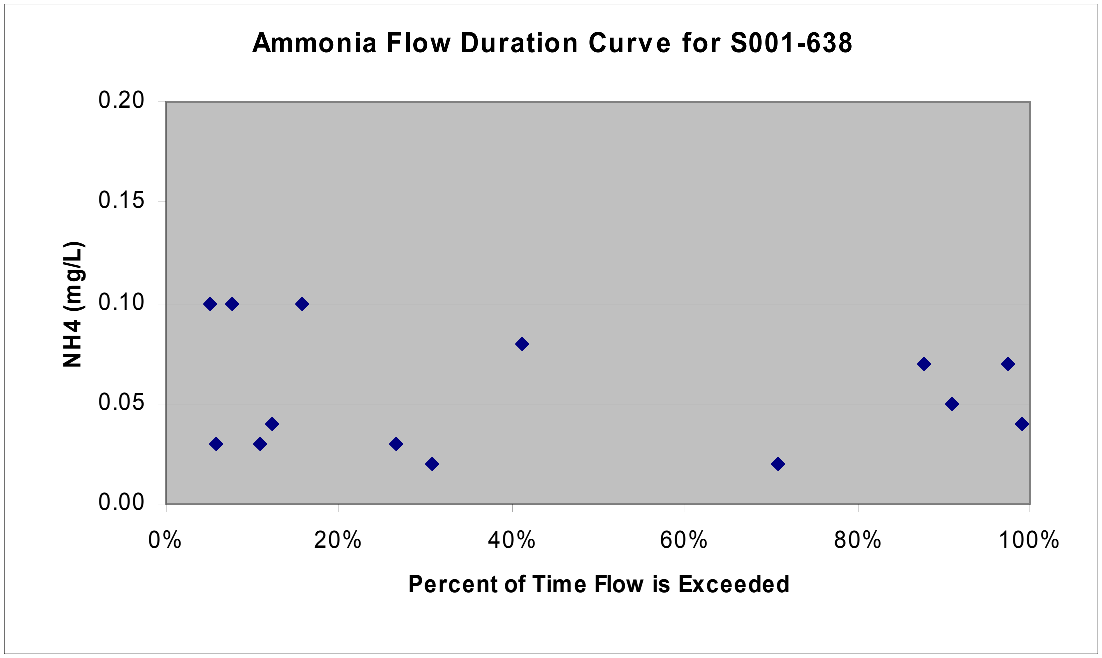

Figure A-47. Ammonia Flow Duration Curve for S003-638

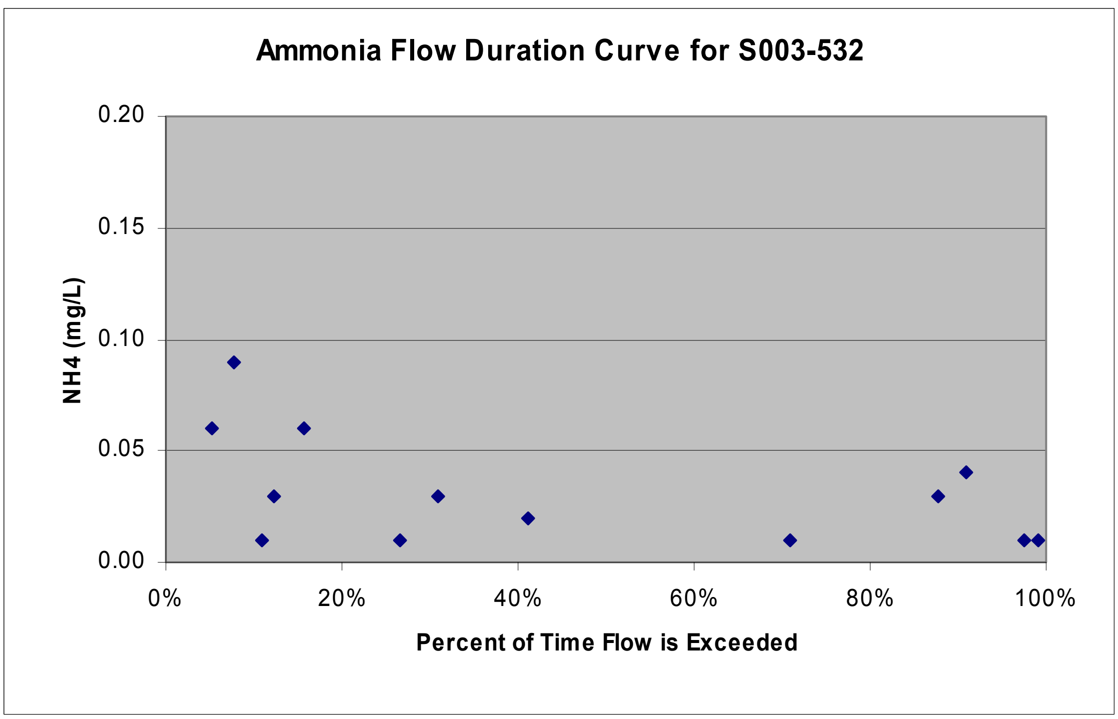

Figure A-48. Ammonia Flow Duration Curve for S003-532

5 Conclusions

Review of the summary data and maps indicate that water quality based on physical and chemical parameters is very good in the Groundhouse River. No potentially significant sediment or nutrient related pollutant levels were seen. Only station S001-152 shows a regular difference from the other monitoring stations. In most cases, water quality met or exceeded the statistics compiled for the minimally impacted streams study. Soil surveys will provide key information to the modeling in identifying areas of high potential for sediment erosion. Since much of the sediment loading can be very site specific, field investigations will also be used to support the modeling efforts by providing a direct assessment of sources such as stream bank erosion. It seems possible that and biotic impairment may be due to localized, possibly natural, habitat issues.

For the biotic impairment, the observations do not in general confirm the SI causal analysis. It is likely that the conditions evaluated by the analysis are localized and a result of natural conditions or the channel bed was impacted by historical sediment impacts that no longer occur. In either case, natural recovery or a reassessment of the natural habitat in this segment may be more relevant than a TMDL.

The geometric mean fecal coliform standard was exceeded occasionally across the watershed but seem to be driven by sources near S001-152 AND S003-664. The FC geometric mean was exceeded in all flow frequency ranges. This implies that the source is not exclusively a wet weather washoff or a relatively constant point source which would dominate at low flows. It could be caused by sporadic direct animal input or possibly septic systems that are highly diluted with high flows and do not reach the stream during low flows.

Subsequent tasks in the TMDL process include the development of a watershed model and field investigations. The watershed model will use landuse, septic system, agricultural census, and feedlot data to identify potential pollutant sources of these fecal coliform exceedances.

6 References

US EPA. 2004. *Screening Level Causal Analysis and Assessment of an Impaired Reach of the Groundhouse River, Minnesota*. U.S. Environmental Protection Agency and the Minnesota Pollution Control Agency.

Appendix B. Watershed Characterization and Water Quality Modeling

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1 Watershed Characteristics

The first requirement for a watershed model is an accurate description of land use, land cover, and flow paths within the watershed. The Minnesota Pollution Control Agency (MPCA) provided Tetra Tech with GIS coverages of MN Department of Natural Resources (DNR) subwatershed boundaries, University of Minnesota land use cover, and Natural Resources Conservation Service (NRCS) State Soil Geographic (STATSGO) soils data. Supplemental datasets were obtained from the National Oceanic & Atmospheric Administration (NOAA), U.S. Geological Survey (USGS), Minnesota Department of Transportation (MNDOT), and the Kanabec and Mille Lacs County Soil and Water Conservation districts (SWCD).

1.1 UNIVERSITY OF MINNESOTA LAND USE AND IMPERVIOUS COVER DATA

The University of Minnesota has developed a statewide land use coverage from 2000 Landsat Thematic Mapper and Enhanced Thematic Mapper satellite images. Table B-1 summarizes the area and percent land cover for the entire watershed. The statewide data classifies land use/ land cover with seven cover classes: urban, agriculture, grassland, forest, water, wetland, and shrubland.

Table B-1. Summary of Minnesota 2000 Land Use Cover for the Groundhouse Watershed

Land Use	Area (ac)	Percent (%)
Forest	42,255	47.5
Agriculture	28,253	31.8
Wetland	11,550	13.0
Urban	5,223	5.9
Shrubland	965	1.1
Grassland	452	0.5
Water	182	0.2
Total	**88,880**	**100**

For modeling purposes, an important characteristic of a land use is the extent of impervious coverage. Impervious areas represent the amount of the land surface that rainfall does not penetrate and include roads, parking lots, and sidewalks. Imperviousness increases with the amount and density of development and affects the quantity and velocity of runoff and the quantity of contaminant wash off. To subdivide the urban land use category into modeling classes, the University of Minnesota 2000 impervious cover dataset was used (Figure B-1). Most of the watershed has very little impervious cover.

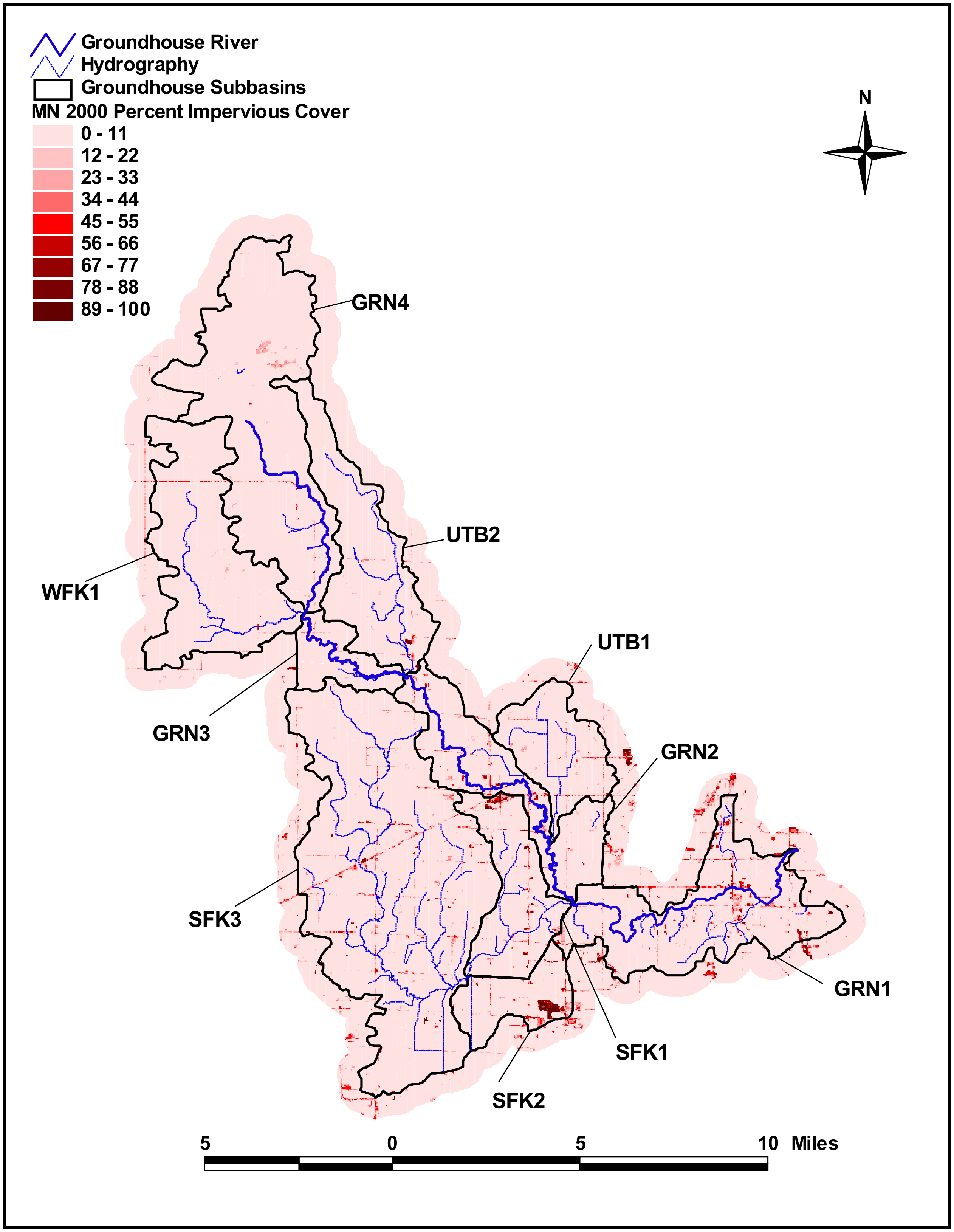

Figure B-1. University of Minnesota 2000 Percent Impervious Cover

The Minnesota 2000 percent impervious cover dataset lists the percent imperviousness in one percent increments for all land uses classified as urban in the Minnesota 2000 land cover dataset. For modeling purposes, Tetra Tech aggregated the urban areas into four modeling classes based on percent impervious area. Table B-2 lists the watershed area in each urban modeling class, the percent of the urban area that each class comprises, and the percent of the total watershed area that each class comprises.

Table B-2. Urban Modeling Classes Aggregated by Percent Impervious Cover in the Groundhouse Watershed

Range of Percent Impervious Cover	Area (ac)	Percent of Urban Land Use Area (%)	Percent of Total Land Use Area (%)
1% to 20%	1,889	36.2	2.1
20% to 50%	2,486	47.6	2.8
50% to 70%	529	10.1	0.6
70% to 100%	319	6.1	0.4
Total Urban	**5,223**	**100**	**5.9**

1.2 NLCD LAND USE DATA

The GLWF model requires a distinction of the separate agricultural land uses for each subwatershed, i.e. pasture, row crops, small grains. However, the University of Minnesota land use data only classifies land use as "Agricultural". To determine the relative proportion of agricultural lands in each subwatershed that are pasture, row crops, or small grains, the 1992 National Land Cover Database (NLCD) from the Multi-Resolution Land Characterization (MRLC) Consortium (USGS, 2000) was used. The NLCD is based on interpretation of 30-meter Landsat satellite thematic mapper imagery. The images were recorded between 1988 and 1994 for Minnesota. Though the NLCD data may be dated in terms of precise land use area, the relative percentages within the agricultural class should be accurate. Small grains were never more than two percent of the area of any subwatershed, so this class was omitted from the modeling analysis. Table B-3 lists the percentages of the modeled pasture versus row crop land use types by subwatershed that resuled from the analysis of the NLCD data. The percent pasture and row crop in each subwatershed were then used to weight the various modeling parameters such as curve numbers, cover coefficients, etc.

Table B-3. Percent of Agriculture in Pasture or Row Crop for Each Modeling Subwatershed Based on the 1992 NLCD

Subwatershed Code	Percent of Agriculture in Pasture	Percent of Agriculture in Row Crop
GRN1	60.1	39.9
GRN2	67.0	33.0
GRN3	77.2	22.8
GRN4	67.3	32.7
SFK1	41.9	58.1
SFK2	33.5	66.5
SFK3	68.6	31.4
UTB1	73.8	26.2
UTB2	81.4	18.6
WFK1	87.3	12.7
Total	**62.3**	**37.7**

1.3 STATSGO SOILS DATA

GWLF simulates rural soil erosion using the universal soil loss equation (USLE). The NRCS STATSGO coverage of the Groundhouse watershed was used to estimate the soil specific parameters such as the soil erodibility factor (K), average slope, percent fines, and hydrologic group (Section 2.2.1). There are six predominant soil types in the watershed, as shown in Figure B-2. Though SSURGO data have recently been made available for Kanabec and Mille Lacs Counties, the coarser resolution of the STATSGO data is more appropriate for this scale of watershed modeling.

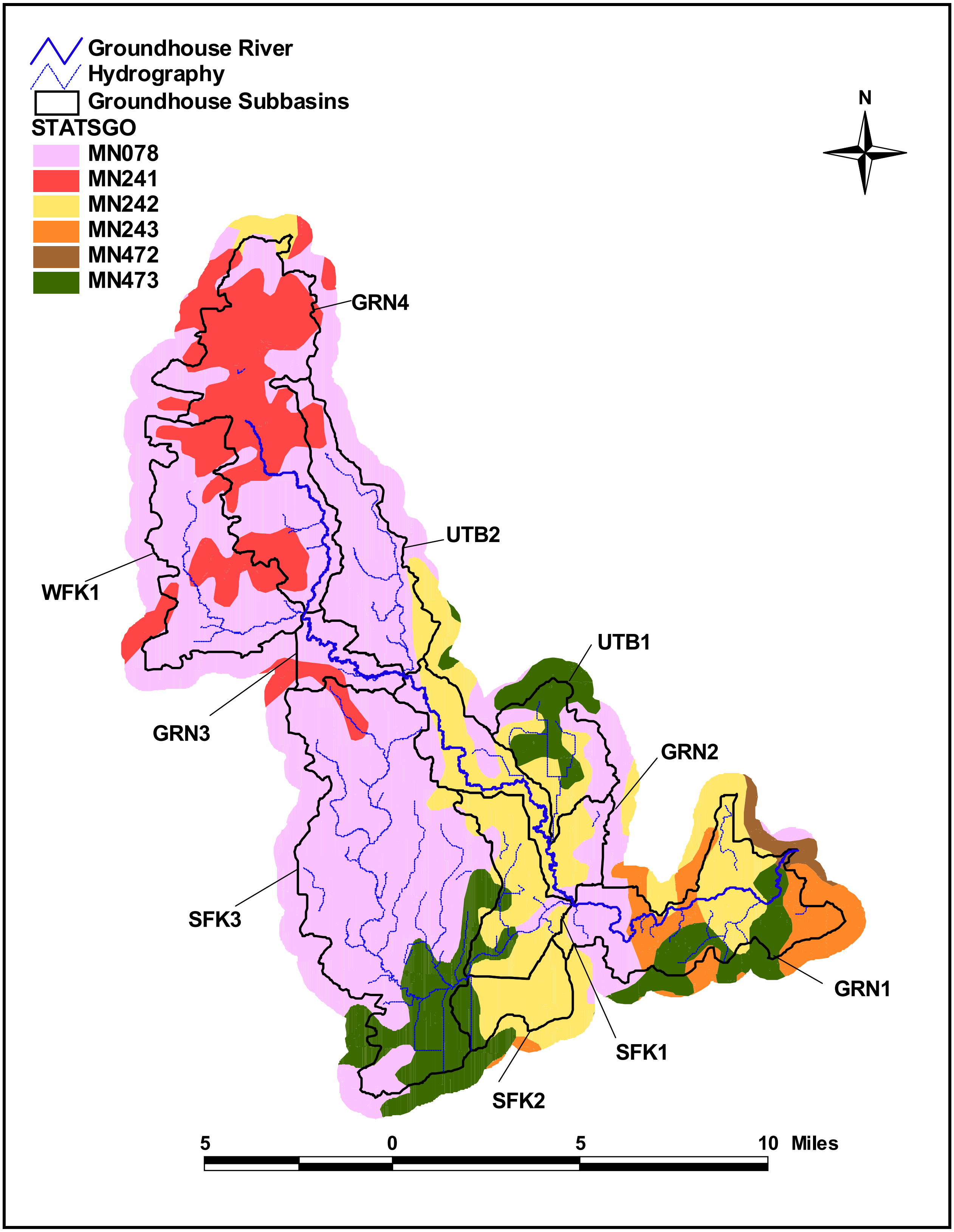

Figure B-2. STATSGO Soil Coverage for the Groundhouse Watershed

1.4 Overlay of Land Use and Soils Data

The Minnesota 2000 land use data and STATSGO soils data were intersected in a GIS environment to define the land use soil combinations prevalent in the watershed. Combinations comprising less than 5 percent of any subwatershed area were aggregated into another combination with the same land use and similar soil properties. The aggregation resulted in 16 land use/soil combinations (urban and gravel pit areas were post-processed outside of GIS (Section 1.1)).

1.5 USDA Aerial Photography

Aerial ortho-photographs of Kanabec and Mille Lacs counties were downloaded from the Minnesota Land Management Information Center website. Images were created by the U.S. Department of Agriculture, Farm Services Agency, Aerial Photography Field Office based on photographs taken in 2003. The photos were used to estimate the area of gravel pits in each subwatershed and to determine the average width of unpaved roads (Section 1.6).

1.6 Unpaved Road Data

The Minnesota 2000 land use data classifies roads in the urban category. This assumption is appropriate for paved roads, but unpaved roads will have much higher sediment loading rates than impervious urban surfaces. GIS road coverages were downloaded from the Minnesota DOT website. To date, this database does not classify the roads by surface type. Plat maps showing paved and unpaved roads in Mille Lacs and Kanabec counties were used to assign a road surface feature to the DOT GIS road coverage as shown in Figure B-3.

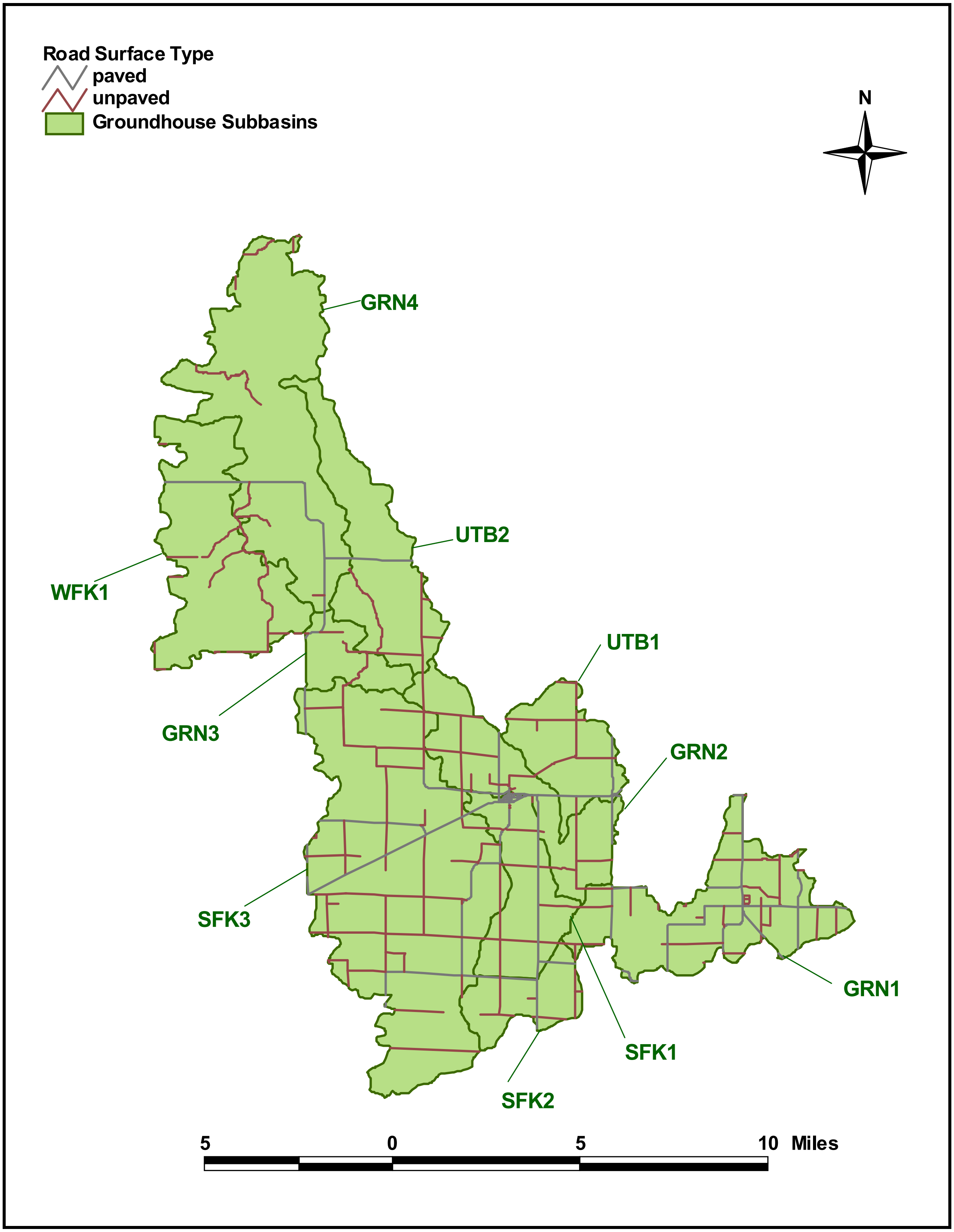

Figure B-3. Road Surface Types in the Groundhouse Watershed

Aerial photographs created by the U.S. Department of Agriculture, Farm Services Agency, were used to measure the widths of unpaved roads in the Groundhouse Watershed. The average width of unpaved road was determined to be 25 feet. The polyline road segments of the MN DOT GIS coverage were buffered to create unpaved road areas. Table B-4 summarizes the area of unpaved roads in each subwatershed.

Table B-4. Area of Unpaved Roads in the Groundhouse Subwatersheds

Subwatershed Code	Area (ac)	Percent of Subwatershed Area (%)
GRN1	51.4	0.52
GRN2	38.8	0.58
GRN3	7.7	0.32
GRN4	25.2	0.17
SFK1	23.2	0.45
SFK2	14.6	0.48
SFK3	131.1	0.53
UTB1	22.8	0.52
UTB2	23.0	0.31
WFK1	30.7	0.30
Total	**368.5**	**0.41**

1.7 POINT SOURCE DISCHARGE DATA

The USEPA Water Discharge Permits (PCS) query form was used to identify permitted dischargers of TSS and fecal coliform in the Groundhouse watershed. The only facility identified with discharge limits for these parameters is the Ogilvie WWTP. The facility outfall is located above the State Highway 23 Bridge. This facility is regulated under permit number MN0021997 and has a permitted discharge rate of 0.23 MGD, a monthly fecal coliform geometric mean limit of 200 per 100 mL, an average monthly TSS limit of 45 mg/L and a weekly maximum TSS limit of 68 mg/L. For the purposes of TMDL development, permit limits will be used for the waste load allocation (Section 5).

1.8 INFORMATION CONCERNING CLOSED TOWN DUMPS

Based on information provided by MPCA and stakeholder input, there are two closed dumps in the Groundhouse Watershed. The Ogilvie town dump located near the intersection of Highway 23 and Highway 52 southwest of Ogilvie was closed and covered in 1975. In May of 1980 the site was inspected and determined appropriately closed in accordance with Minnesota Rule SW-12. The 1980 inspection report states that the site “has been very well covered and revegetation is excellent.” The site was entirely fenced though the gate was broken and a small amount of recent dumping had occurred. An aerial view of the Oglivie town dump taken in 2003 by USDA is shown in Figure B-4.

Based on the 1980 inspection report it is unlikely that the Ogilvie town dump is contributing significant sediment or fecal coliform loading to the waterbodies of the Groundhouse watershed.

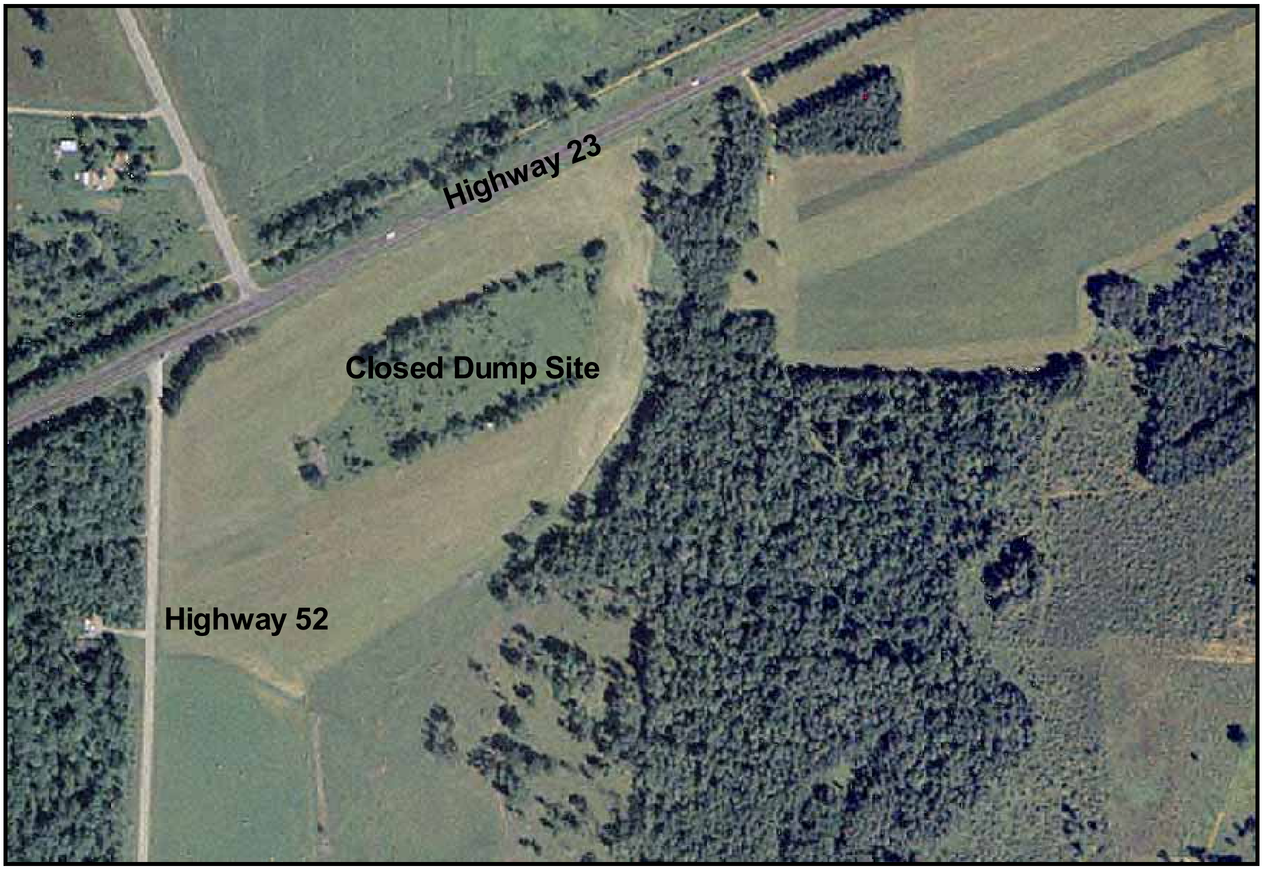

Figure B-4. 2003 Aerial Photograph of the Closed Ogilvie Town Dump

Bernie Klejeski is a citizen in the watershed who owns land north of Ogilvie near an old creamery. Based on personal communication (Klejeski, 2006), Mr. Klejeski reports that an old, closed dump site exists at the end of George Street, and that during heavy rains and spring flooding, the site erodes sediment and trash into the river. The site is shown in Figure B-5, though not much can be seen from the aerial photograph.

Figure B-5. 2003 Aerial Photograph of the Reported Closed Dump Off George Street

The communication with Mr. Klejeski did not occur until after the field reconnaissance. Though Tetra Tech associates did drive down George Street, they encountered an apparent recycling facility with gates, fences, and no trespassing signs, and therefore were not able to exam this section of the river. The field team has since been in contact with Joel Chirhart at MPCA who reportedly has pictures and information concerning this section of the river, which is upstream of the biota impaired site (USEPA, 2004a). Any additional information will be included in future reports.

2 Watershed Model Development

The Groundhouse and South Fork Groundhouse rivers are currently listed on the Minnesota 303(d) list for biota impairment likely due to the presence of sediment in the streambed. In addition, the Groundhouse River is listed for fecal coliform contamination.

There are a variety of approaches for developing watershed-based pollutant loading models, ranging from simple export coefficient models to complex hydrodynamic models. Tetra Tech proposes the use of a watershed model that falls between the simple and complex level and is capable of predicting both annual and seasonal loads for varying hydrologic years. The nonpoint component of the analysis will be based on the Generalized Watershed Loading Function (GWLF) model (Haith et al., 1992

GWLF provides a basis to estimate pollutant load allocations by addressing overland runoff and groundwater discharge into streams. Separate estimates of streambank erosion will be made using the results of a week-long geomorphologic assessment conducted in October 2006 (Section 5). A post-processing spreadsheet model will incorporate loading due to point sources, animals, and onsite wastewater treatment.

The sources of fecal coliform in the watershed are human and animal-based. The fecal loads are simulated separately in three spreadsheet loading models that represent direct discharge from the centralized wastewater treatment plant in Ogilvie, loading from onsite wastewater treatment systems (septic tanks and mound systems), and loading from domestic, agricultural, and wildlife animal populations.

2.1 The GWLF Model

GWLF simulates runoff and streamflow by a water-balance method, based on measurements of daily precipitation and average temperature. Precipitation is partitioned into direct runoff and infiltration using a form of the Natural Resources Conservation Service's (NRCS) Curve Number method. The curve number determines the amount of precipitation that runs off directly, adjusted for antecedent soil moisture based on total precipitation in the preceding five days. A separate curve number is specified for each land use by hydrologic soil grouping. Infiltrated water is first assigned to unsaturated zone storage, where it may be lost through evapotranspiration. When storage in the unsaturated zone exceeds soil water capacity, the excess percolates to the shallow saturated zone. This zone is treated as a linear reservoir that discharges to the stream or loses moisture to deep seepage, at a rate described by the product of the zone's moisture storage and a constant rate coefficient.

2.2 Weather Data

Hydrologic simulation in GWLF is driven by daily precipitation totals and maximum and minimum daily temperatures. Potential evapotranspiration is calculated from temperature. The meteorological data required by GWLF was collected and processed for the meteorological stations at Isle, MN (Station 214103), Mora, MN (Station 215615), and Milica, MN (Station 215392) to represent the range of conditions across the watershed. The raw data were obtained from the National Climatic Data Center for 1996 through 2006.

2.2.1 Soil and Hydrologic Properties

GWLF simulates rural soil erosion using the universal soil loss equation (USLE). This method has been applied extensively, so parameter values are well established. The Minnesota 2000 land use data and STATSGO data were overlain to create unique land use/soil combinations, e.g., forest land use on soil MN078. This section describes the parameters used to simulate the various combinations.

Runoff Curve Numbers: The direct runoff fraction of precipitation in GWLF is calculated using the curve number method from the NRCS TR55 method (SCS, 1986), based on imperviousness and soil hydrologic group. The hydrologic soil group of each soil type was determined using the STATSGO database. Curve numbers for each modeled land use are defined in Table B-5.

Table B-5. Curve Numbers for Antecedent Soil Moisture Condition II by Land Use and Soil Hydrologic Group

Land Use	Hydrologic Group A	Hydrologic Group B	Hydrologic Group C	Hydrologic Group D
Forest	30	55	70	77
Agriculture - Pasture	49	69	79	84
Agriculture - Row Crop	67	78	85	89
Wetland	45	66	77	83
Shrubland	30	48	65	73
Unpaved Roads	76	85	89	91
Urban 1% to 20% Impervious Surface	45	64	76	82
Urban 20% to 50% Impervious Surface	60	74	82	86
Urban 50% to 70% Impervious Surface	74	83	88	91
Urban 70% to 100% Impervious Surface	89	92	94	95

Evapotranspiration Cover Coefficients: The portion of rainfall returned to the atmosphere is determined by temperature and the amount of vegetative cover, which differs for each land use and varies by season (growing and dormant). For rural land uses, evapotranspiration (ET) rates were based on seasonal values reported in the GWLF manual (row crop data are specified by month); for urban land uses, ET was calculated as 1 minus the impervious fraction. ET growing and ET dormant are the same for urban land uses whose pervious area is mostly lawn and landscaped plants. Barren land is assumed to have no significant plant cover, but water is still lost through evaporation. Table B-6 summarizes the average growing season and dormant season ET values for each subwatershed.

Table B-6. Evapotranspiration Coefficients for the Groundhouse Subwatersheds

Subwatershed	Growing Season ET	Dormant Season ET
GRN1	0.93	0.53
GRN2	0.95	0.59
GRN3	0.99	0.41
GRN4	0.99	0.36
SFK1	0.91	0.60
SFK2	0.88	0.56
SFK3	0.96	0.58
UTB1	0.97	0.46
UTB2	0.99	0.39
WFK1	0.99	0.36

Soil Water Capacity: Water stored in soil may evaporate, be transpired by plants, or percolate to groundwater below the rooting zone. The amount of water that can be stored in soil and is available to evapotranspiration—the soil water capacity—varies by soil type and rooting depth. Average soil water capacity was estimated from STATSGO information on available water capacity in the soil column, assuming a rooted depth of 100 cm. Table B-7 lists the average soil water capacity for each subwatershed.

Table B-7. Soil Water Capacity for the Groundhouse Subwatersheds

Subwatershed	Soil Water Capacity (cm)
GRN1	26.5
GRN2	19.6
GRN3	22.6
GRN4	30.5
SFK1	22.9
SFK2	25.0
SFK3	25.1
UTB1	29.5
UTB2	20.8
WFK1	25.3

Recession Coefficients: The rate of groundwater discharge to streams is governed by the recession coefficient. In theory, this coefficient can be determined by examining the flow hydrograph when gaging data are available. An initial value of 0.01 was assumed for the Groundhouse watershed. This coefficient was used as a calibration factor and was set to 0.05 in the calibrated model.

Deep Seepage Coefficient: The GWLF model has three subsurface zones: a shallow unsaturated zone, a shallow saturated zone (aquifer), and a deep aquifer zone. The deep seepage coefficient is the portion of the moisture content in the shallow saturated zone that seeps to the deep aquifer zone and does not reappear as surface flow, effectively removing it from the watershed system. To model this process, the saturated zone is treated as a linear reservoir in which the moisture lost equals the moisture content multiplied by the saturation coefficient. For the Groundhouse watershed, deep seepage is expected to be a small fraction, so that precipitation on the land surface eventually either returns to the atmosphere or flows to the river. As an initial estimate, the deep seepage coefficient was set to zero and did not require adjustment during hydrologic calibration.

Soil Erodibility (K Factor): Erosion in the GWLF model is simulated with the Universal Soil Loss Equation (USLE), for which four input factors are required (K, LS, C, and P). The first of these is the soil erodibility factor, K, which indicates the propensity of a given soil type to erode. Soil erodibility factors from the STATSGO database were analyzed for each MUID. Composition-weighted values by MUID vary from 0.04 (muck/peat) to 0.40 (silt) and are shown in Figure B-6.

Length-Slope (LS) Factor: Erosion potential varies by slope as well as soil type. The LS factor is the length (L) that runoff travels from the highest point in the watershed to the point of concentrated flow, multiplied by the slope (S) which represents the effect of slope steepness on erodibility for each soil type. LS factors for the Groundhouse watershed were calculated using the methodology described in the GWLF User's Manual (Haith et al., 1992). LS factors for the soil types range from 0.12 to 0.22.

Cover and Management (C) and Practice (P) Factors: The mechanism by which soil is eroded from a land area and the amount of soil eroded depends on soil treatment resulting from a combination of land uses (e.g., forestry versus row-cropped agriculture) and the specific manner in which land uses are carried out (e.g., no-till agriculture versus non-contoured row cropping). Land use and management variations are represented by cover and management factors in the universal soil loss equation and in the erosion model of GWLF. Cover and management factors were selected by land use from values reported in the GWLF User's Manual (Haith et al., 1992) and are summarized in Table B-8. A C factor of 1 indicates no vegetative or surface cover. Practice factors are set to 1 for this simulation.

Table B-8. Cover and Management (C) and Practice (P) Factors for Rural Land Uses in the Groundhouse Watershed

Rural Land Use	C	P
Forest	0.003	1
Agriculture – Pasture	0.09	1
Agriculture – Row Crop	0.51	1
Wetland	0.003	1
Shrubland	0.04	1
Unpaved Roads	0.75	1
Urban Pervious	0.01	1

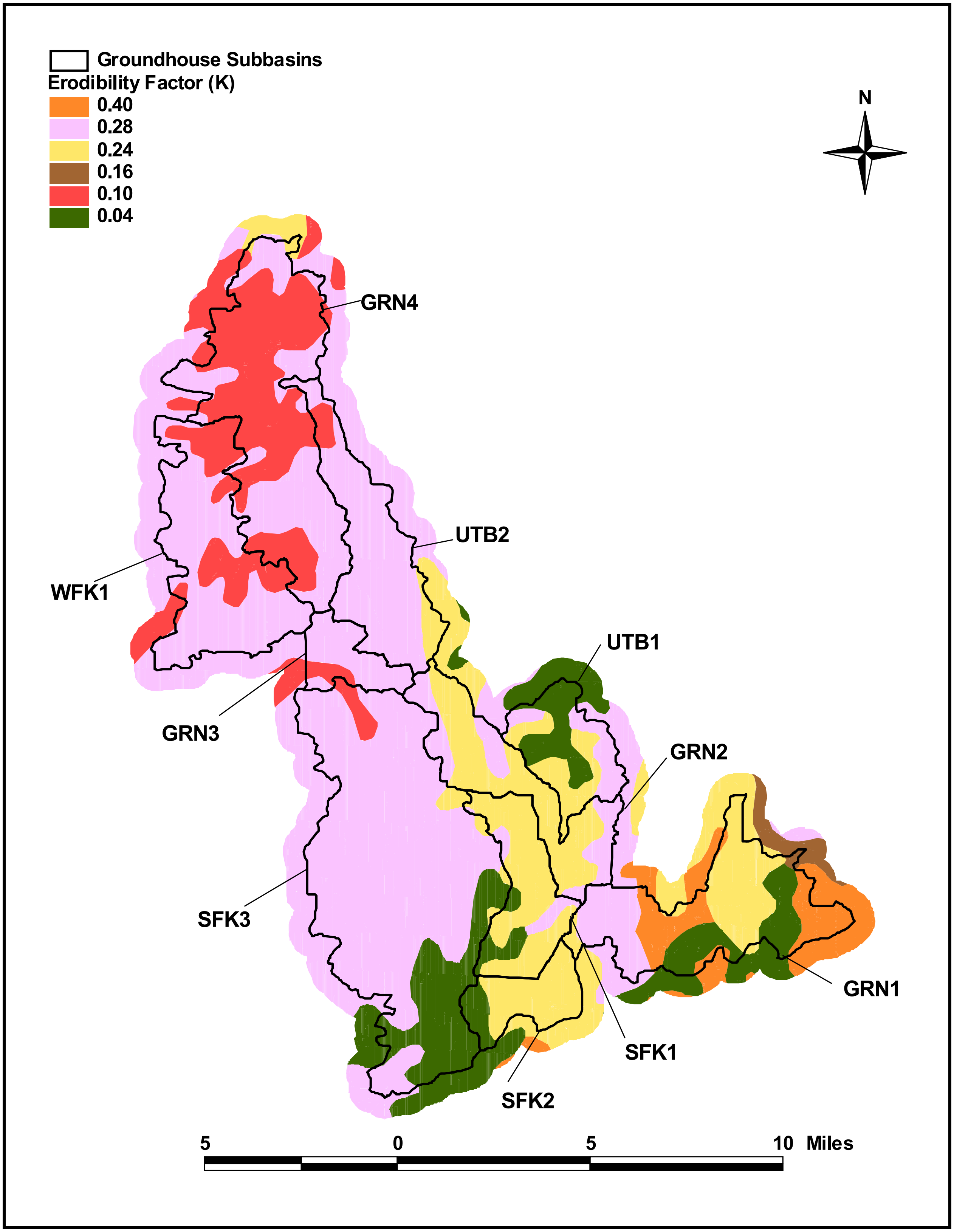

Figure B-6. Soil Erodibility Factor (K) for Soils in the Groundhouse Watershed

Build up-Washoff Rates: GWLF simulates pollutant loading from urban areas using a buildup-washoff formulation. Build up rates of total suspended solids (TSS) for the four urban classes are summarized in Table B-9 based on default values for urban pervious and urban impervious surfaces presented in the GWLF User's manual for three densities of residential development and a central business district (Haith et al., 1992).

Table B-9. Buildup-Washoff Rates for Urban Land Uses in the Groundhouse Watershed

Urban Land Use	TSS Build up-Washoff Rate (lb/ac/d)
1% to 20%	1.3
20% to 50%	2.6
50% to 70%	4.1
70% to 100%	2.2

Sediment Delivery: GWLF uses the USLE equation to estimate erosion from land surfaces and then applies a sediment delivery ratio (SDR) to account for trapping during overland flow. The sediment delivery ratio is calculated for each subwatershed based on drainage area using the tool included with the Windows version of the GWLF model (Dai and Wetzel, 1999). The sediment delivery ratios applied to the Groundhouse subwatersheds are listed in Table B-10.

Table B-10. Sediment Delivery Ratios for the Groundhouse Subwatersheds

Subwatershed	Sediment Delivery Ratio
GRN1	0.1617
GRN2	0.1771
GRN3	0.2217
GRN4	0.1457
SFK1	0.1885
SFK2	0.2117
SFK3	0.1281
UTB1	0.1948
UTB2	0.1729
WFK1	0.1606

2.2.2 Hydrologic Calibration of the GWLF Model

Initial calibration of the GWLF model for the Groundhouse watershed used flow data collected on the Groundhouse River at County Road 12. Flows were monitored in April through October from 1999 through 2005. The GWLF model produces reliable hydrologic output at a monthly time step. For comparison to the observed flows in the watershed, the simulated monthly flows were compared to corresponding observed flows. Permitted flows from the Ogilvie WWTP were accounted for as well (approximately 7 million gallons (MG) per month). The flows observed at County Road 12 were scaled up by a ratio of 139 to 125 to account for the additional area downstream of the gage.

During the calibration, adjustments to the recession coefficient of the river were tested. Though increasing the coefficient from 0.01 to 0.05 improved the model fit for most months, several months were unaffected by the change. Alterations to variables such as monthly evapotranspiration coefficients and curve numbers were tested, but did not improve the fit for those months.

For this reason, the model output was also compared to area-scaled flows measured by the USGS at Snake River near Pine City, MN (Gage 05338500). Flows from this gage were scaled down by a ratio of 139 over 974 to reflect the difference in drainage area of the two watersheds. For those months that the simulated model did not fit well with flows observed on the Groundhouse, it did perform well compared to the flows measured nearby by the USGS. It is likely that the three weather stations surrounding the Groundhouse watershed did not accurately reflect the local weather patterns occurring in the watershed for certain periods during the 10 year simulation.

Figure B-7 shows the simulated monthly flows plotted with scaled flows measured on the Groundhouse and Snake Rivers. The model tends to under predict flows during dry months relative to flows measured on the Groundhouse. However, most of the sediment load is delivered during runoff events, so matching base flows was not a high priority. During wetter months, the model follows the patterns of the measured flows fairly well, but does over predict flows during most months. The total flow volume simulated by the model is approximately 7.5 percent higher than that measured on the Groundhouse and 2.8 percent higher than that measured on the Snake River. The over prediction of flow results in a more conservative estimate of sediment loading since flow events drive erosive processes in the watershed. The sediment loading results, by simulation year, are shown in Table B-11.

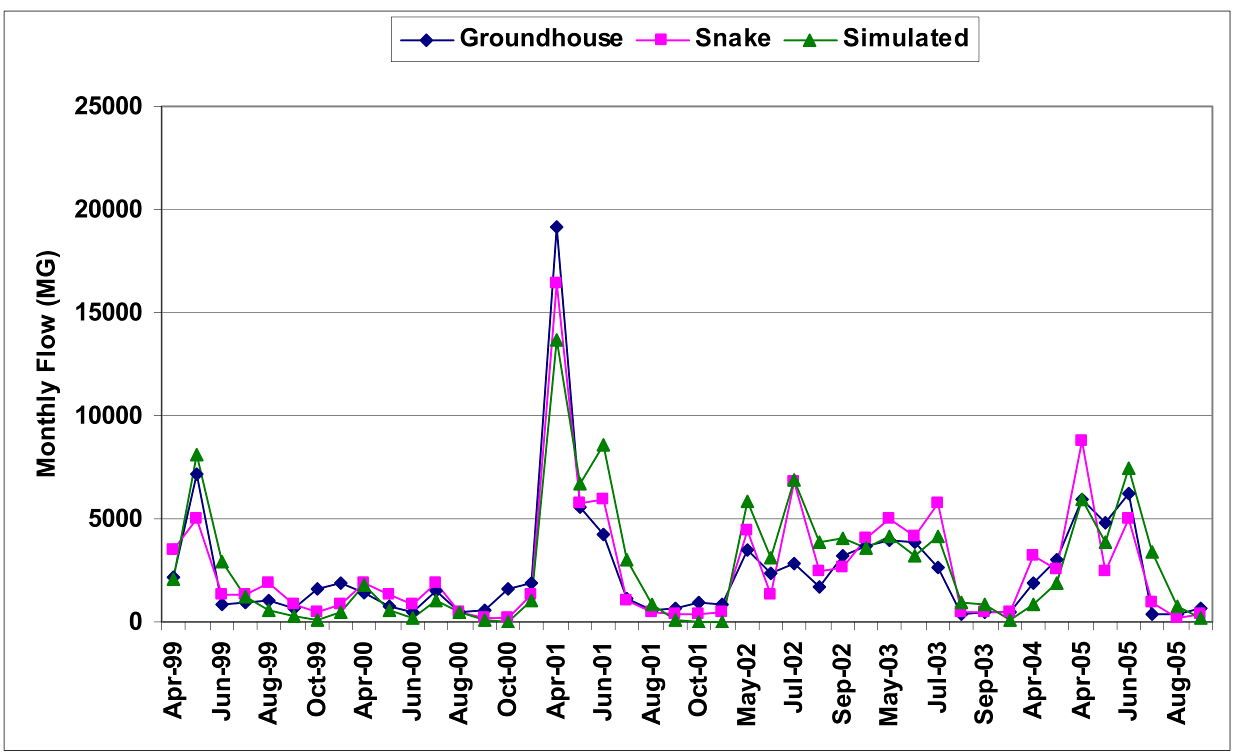

Figure B-7. Comparison of Simulated Monthly Flows to Measured Flows on the Groundhouse and Snake Rivers

Table B-11. Average Sediment Loads for Subwatersheds in the Groundhouse and South Fork Groundhouse Watersheds

Subwatershed	Upland Sediment Load (US tons/yr)
Mainstem Groundhouse Watershed	
GRN1	1,104.8
GRN2	743.8
GRN3	104.6
GRN4	254.7
UTB1	246.4
UTB2	209.0
WFK1	139.6
Total Mainstem	**2,802.9**
South Fork Groundhouse Watershed	
SFK1	1,180.0
SFK2	830.6
SFK3	2,034.9
Total South Fork	**4,045.6**

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Appendix C. Summary of Field Reconnaissance of the Groundhouse River Watershed

1 Watershed Assessment

The assessment was conducted September 26 – 28, 2006 by two Tetra Tech staff - an aquatic biologist and a fluvial geomorphologist. According to input from individuals in the watershed, the summer 2006 precipitation was characterized as a drought; only a few weeks prior to the assessment had any appreciable rain fallen. The drought conditions can influence observed conditions because high flow events are most likely to cause erosion and instability. In the absence of these events, vegetation can colonize exposed surfaces making it appear more stable.

The assessment focused primarily on instream conditions in the Groundhouse River and the South Fork of the Groundhouse River. In particular, reaches were walked at locations noted in previous studies as well as locations where MPCA conducts biological monitoring. While traveling between these reaches, upland conditions that could influence instream biota and fecal coliform bacteria concentrations were noted.

1.1 GROUNDHOUSE RIVER BIOTA

The assessment began by walking the degraded reach of the Groundhouse River from the Highway 23 crossing near Ogilvie for a distance of approximately 2,750 feet (840 meters). Additionally, another 425 feet (130 meters) were walked immediately downstream of the Highway 23 crossing. A power line corridor runs along one edge of the meander belt width. In this corridor, no woody vegetation is present and herbaceous vegetation appears managed. Where the river meanders through the corridor, streambanks appear most unstable; where the stream meanders back into areas with natural canopy species, the streambanks are lower and contribute considerably less sediment to the river. From the downstream limit of the walked reach, substrate noticeably coarsened in the upstream direction. At the downstream end, nearly the entire streambed was formed in soft sand; at the upstream limit of the observed reach, cobble riffles were present.

This reach is located in a glacial outwash region (the Superior Lobe). Glacial outwash material is typically very well sorted and stratified, and the materials are not well consolidated. Additionally, as the materials are fluvially transported, clay size fractions may not be represented as much as the sand and silt size fractions. According to the NRCS soil survey for Kanabec County (2006), the top 6 – 7 feet of soil along the listed reach of the Groundhouse River is silt loam. The top six inches is 0 – 40 percent sand, 50 – 80 percent silt, and 10-25 percent clay; between depths of six to 80 inches, 0 – 84 percent of the soil matrix is sand, 15 – 75 percent is silt, and 1 – 25 percent is clay.

As additional reaches were evaluated in the upstream direction, streambed substrate coarsened to cobbles and boulders, and stream bank erosion was much less prevalent. The one exception to this trend was a reach of approximately 2,000 feet upstream of the Highway 47 crossing north of Ogilvie. Access to this reach was limited due to a private pasture, but excessive streambank erosion was noted along a cattle pasture. The streambed substrate appeared to consist primarily of gravel and sand. The common visible characteristic with the listed reach was the lack of any woody vegetation along the streambanks. No riparian fencing was installed to keep livestock out of the stream, and only grasses were observed along this reach. Between this reach and the impaired reach, the stream returned to a coarser substrate streambed and a stable channel morphology that did not appear to be contributing excessive sediment to the river.

Two reaches were evaluated downstream of the impaired reach on the Groundhouse River. At both of these sites, streambed substrate consisted primarily of cobbles. Further, the streambanks did not appear to be actively contributing sediment to the river.

Based on the comparison of morphologic conditions upstream and downstream of the listed reach on the Groundhouse River to morphologic conditions in the listed reach, two reasonable interpretations can be drawn. The first is that the lack of woody vegetation on the streambanks due to clearing (for either a power line corridor or for a pasture) eliminates woody root systems that would otherwise reinforce the erosion prone glacial outwash material in the streambanks. In this case, other sites without woody vegetation should exhibit actively eroding streambanks, and other sites with woody vegetation should not exhibit sandy substrates and streambank erosion. However, this was not observed to be the case. For example, the MPCA site visited off of Dill Road (approximately 8,000 feet upstream of the eroding reach through the pasture) is located in an open meadow without woody vegetation, but the streambanks were not observed to be actively eroding. Additionally, MPCA staff visited a reach immediately downstream of the Highway 47 crossing that was well wooded and exhibited streambank erosion as bad as or worse than the listed reach (personal communication, Joel Chirhart, MPCA to David Pizzi, Tetra Tech, October 25, 2006). The second interpretation is that the underlying geology plays a key role in the morphology of a river reach. In particular, the extensive sorting and stratification of the glacial outwash in the Superior Lobe provides for some reaches to be formed through deposits of sand and silt. The location of these deposits could explain the intermittent nature of the stream reaches observed to be actively eroding. On its own, this inference is likely more valid than the inference regarding woody vegetation. However, more detailed geologic mapping or soil testing is recommended to confirm this interpretation.

1.2 GROUNDHOUSE RIVER AND SOUTH FORK GROUNDHOUSE RIVER FECAL COLIFORM BACTERIA

After the instream conditions in the Groundhouse River, the West Fork, and the South Fork were assessed, the focus shifted toward an evaluation of potential sources of fecal coliform bacteria. The Groundhouse River from its headwaters to the confluence with the South Fork as well as the South Fork from its headwaters to the confluence with the Groundhouse River has fecal coliform bacteria concentrations that exceed state water quality standards. Potential sources that were investigated include animal operations, septic systems, wastewater treatment outfalls, and straight pipes (i.e., sewer outfalls where no treatment is provided).

Some minor livestock operations were observed in the watershed; however, no large-scale operations were noted. At most, a few dozen cattle were observed in any one pasture during the visit. This is not to insinuate the larger operation don't operate at other times of the year or in pastures not observed during the visit. Cattle exclusion from the stream channel varied widely. Riparian fencing was observed in some pastures to limit livestock access to the water; in other pastures cattle were observed in the water. Some pastures had wide (greater than 100 feet) vegetated buffers whereas others had bare streambanks. The livestock could be a source of elevated fecal coliform bacteria concentration at operations that do not employ best management practices.

The reach of the South Fork located near 140th Avenue (approximately 3,500 feet upstream of the confluence with the Groundhouse River) was noted to have a very strong manure odor. A cattle pasture was observed a short distance upstream, but the condition of a buffer, if any, could not be seen. The odor was strong enough that it was obvious just standing on the culvert crossing at the creek.

Due to the low population density in the watershed, municipal water and sewer systems are not common. Except for structures within Ogilvie that are connected to a sewer system that treats waste at the Publicly Owned Treatment Works (POTW), septic systems are used. These systems are commonly mound systems. While the density of these systems is low, some of the mounds were observed within a few hundred feet of the stream. Due to the potential for excessively high permeability through some deposits of glacial outwash, the possibility exists that septic system leachate enters the groundwater and the rivers without receiving adequate treatment.

The wastewater treatment outfall from the City of Ogilvie POTW was observed. Some green filamentous algae was observed around the outfall, as well as on the streambed downstream of the outfall (noting that the streambed was primarily sand along this reach). Historic operational records should be reviewed to evaluate this outfall as a point source of fecal coliform bacteria.

Field Photograph Inventory

Photographs taken September 26, 2006.

Upstream view of Groundhouse River immediately upstream of the Hwy 23 crossing. This is the downstream limit of the reach listed for impacted aquatic communities. The streambed substrate was firm sand and gravel.

Upstream view from same vantage as previous photo. Note small point bar along right bank. Due to the drought conditions, the vegetation does not necessarily indicate a stable bar.

Downstream view at same location as previous photos. Toe scour was observed at the base of the tree, but the size (e.g., age) of the tree indicates the erosion is a long-term process. Debris in the tree was approximately 6-feet above the elevation of the thalweg.

Upstream view slightly farther upstream from previous photos. This streambank is nearly vertical, approximately 6-feet high, supports little woody vegetation (the reed canary grass has only shallow roots), and shows block failures at the toe. While a pool is still present, the substrate here is less firm.

Upstream view of point bar opposite the unstable streambank shown in the previous photo. The depositional material primarily consisted of coarse sand.

Upstream view beneath though the power line corridor. The taller streambanks (5-6' high) and eroding bank material are likely attributed to the lack of woody vegetation. The corridor runs along one side of the meander belt such that approximately half the bends are located in this cleared corridor.

Downstream view of streambank in the power line corridor. Note the lower streambank height as the stream flows back into naturally wooded area. Streambank material is sandy loam/silt loam.

Downstream view of streambank shown in the previous photo. Scour at the toe of the bank leads to undercutting and slumping of the upper bank. This was the primary mechanism for streambank sediment input observed along this reach of the Groundhouse River.

Upstream view of large depositional (point) bar in the wooded area just upstream of the previous two photos. The material is mostly coarse sand – representative of streambank material. Note the wider channel and lower streambank heights where the woody vegetation has not been cleared.

Close-up of the overbank area shown in the previous photo. Note the staining on the tree trunks likely indicating inundation during certain times. If flood waters draw down relatively rapidly, pore water pressure in the streambanks could contribute to mass wasting.

Fluvially transported woody debris along the right bank. This debris indicates that hydraulic stresses, particularly on the toe of the streambank are sufficient to erode and transport streambank material.

Downstream view toward another crossing of the power line corridor. A debris jam has created a backwater area with excessive green algal growth. The algae is likely due to the open canopy, slow moving water, and nutrient load provided by upstream sources.

First cobble riffle encountered upstream of the Hwy 23 crossing. Noted stoneflies and mayflies on substrate. Only minor embeddedness of surface substrates was observed.

Right bank along cobble riffle shown in previous photo. Note the lower streambank height and the influence of woody roots. The tree at the left of the photo appears to have been recently undercut. Channel response is likely widening due to the more erosion resistant substrate in this reach.

Depositional bar substrate just upstream of the cobble riffle shown in the previous photos. The substrate shape is sub-rounded.

Downstream view of right bank under power line crossing upstream of the cobble riffle shown in the previous photos. The depositional bar on the left bank appears to be storing some of the sediment generated by the erosion of the right bank.

Close up view of large block of streambank material detached from the streambank. The location along the power line corridor indicates the lack of woody vegetation contributes to the susceptibility of the streambanks to destabilizing forces.

Separation of sediment block from the streambank as seen from the top of the bank.

View of the block failure shown in previous photos as seen from the top of the bank.

Close-up view of the streambank material where the failure block is separating as shown in the previous two photos.

Growth of green algae on a depositional bar. The presence of soft, fine depositional material indicates this is a slower flowing region of the channel.

Upstream view of the upper limit of the reach walked from the Hwy 23 crossing. The grassy depositional bars are located where the power line crossing is located. More large woody debris was observed in the channel upstream of this point.

Depositional material at the downstream end of one of the depositional bars shown in the previous photo.

Downstream view downstream of the Hwy 23 crossing. This was the downstream limit of the reach walked from Hwy 23. Bottom substrate was considerably less consolidated than the upper section.

Growth of green algae on bottom substrate in the reach shown in the previous photo.

Outfall from the Ogilvie publicly owned wastewater treatment works (POTW) located on the right streambank immediately upstream of the Hwy 23 crossing. No changes in water quality were observed downstream of the outfall.

Upstream view of the left bank of the Groundhouse River at a site off of Dill Road (located upstream of the previous site). Note the cobble substrate and the more stable morphology of the bank – particularly along the outside of a meander bend.

Mid-channel depositional bar formed of gravel and cobble. Both submerged and emergent vegetation were observed in this reach. The reach is located in a meadow with little woody vegetation or canopy cover, the streambanks were generally low and not sources of excess sediment to the reach.

Downstream view of right bank at the head of a bend. The boulders to the left of the photo were likely placed by the adjacent landowner to minimize streambank erosion. Note the contrast with the upstream conditions where no boulders were installed.

Upstream view of a cobble riffle at the upstream end of the reach walked off of Dill Road. Approximately 0.5'-feet of fall occurs over this riffle – the third observed in this reach.

Example of mound-type septic systems commonly used in the watershed. This mound was located approximately 250-feet from the Groundhouse River (other mounds were observed closer).

Transition from asphalt paved road to graveled road near the intersection of Hwy 10 and 170th Avenue. In general the gravel roads were very well maintained and no evidence of sediment input at stream crossings was observed. Roadside ditches were not common.

Buffalo operation along Hwy 10 in the Groundhouse watershed.

Another view of the buffalo operation.

An example of cattle in a pasture in the Groundhouse watershed.

An example of a mound septic system (note the white pipes in the foreground of the house) within 100 feet of the Groundhouse. This site is just upstream of the Hwy 10 crossing.

Third site visited where Apple Road crosses the Groundhouse River in the Rum River State Forest. This location is immediately upstream of the crossing (which is reported to have been washed out multiple times over the previous 5 years). Note the confinement by the valley wall along the right bank. The woody vegetation and cobble/boulder substrate limit the sediment generated in this reach.

Sandy deposition on a point bar along the left bank. More sand was observed under the surface substrate layer in this reach than in downstream reaches.

Upstream view of a cobble riffle. The cobbles were between 50 – 75 percent embedded with sand. The source of the sandy material was not evident.

Upstream view of left bank upstream of riffle shown in previous photograph. Minor undercutting of the bank is shown – but this does not appear to be significant enough to be the sources of the sediment deposited in this reach.

Another sandy depositional bar observed in this reach. In places, sand was the predominant material on the streambed.

Downstream view at the upstream end of this reach. Note the boulders along the left bank. Due to the presence of the large substrate clasts, the sandy material observed in this reach is likely transported from upstream sources and not indicative of underlying geology.

Flashboard riser structure to control flows from a large lake/wetland to a small tributary to the Groundhouse that enters just downstream of the site visited off of Apple Road.

Another view of the outlet structure.

Downstream end of the culvert from the outlet structure shown in the previous photographs. The 36-inch CMP discharges to placed riprap before entering the small tributary.

Upstream view of “wetland flowage” near, but downstream, of the confluence of the West Fork of the Groundhouse and the Groundhouse. This reach exhibited a very low gradient, and no erosion was observed.

Downstream view from of the “wetland flowage” shown in the previous photo. A culvert crossing is located near the vehicle on the left bank.

Upstream view of impounded wetland area in the headwaters of the West Fork watershed.

Control structure used to maintain the water surface elevation of the impounded area shown in the previous photograph. The culvert barrel is a CMP 8-feet in diameter.

Upstream view of the Groundhouse River upstream of the Hwy 26 crossing. This site is upstream of the confluence with the West Fork. The channel is approximately 6-feet wide and 2-3-feet deep. Little active erosion was noted, but the streambed was soft sand and gravel. The reach is located in a broad wetland.

Downstream view from previous photo vantage point. The dual 60-inch diameter RCP culverts clearly show a high water mark approximately 3-feet above the thalweg elevation. Downstream of this culvert, lots of organic material and much was located on the streambed. Ongoing decomposition was observed by the methane bubbles released by disturbing the sediment. Likely low dissolved oxygen.

Surface gravel mining operation observed along Hwy 10. This operation was not noted on the DOT maps.

Stockpiles of aggregate in the same operation as shown in the previous photo. The operation is approximately 6-acres in size. The western edge abuts the riparian area along the North Fork of the Groundhouse. The confluence with the Groundhouse River is downstream of the site in the Rum River State Forest – thus, this is not the source of the sandy material noted in that reach.

Highway 10 crossing of the Groundhouse. Despite the gravel road surface, even across the bridge, no evidence of excessive input to the river was observed.

A point where road runoff can directly enter and transport sediment to the Groundhouse River.

A second view of the mound septic system at the Hwy 10 crossing showing the proximity to the River.

View facing south along Hwy 10 at the Groundhouse crossing. Note the well maintained condition of the road surface. The grade of the road is sloping away from the River.

Upstream view of the Groundhouse upstream of the Highway 57 crossing. The substrate in this reach consisted of cobbles and boulders, as indicated by the riffle in the center of the photo. An abundance of algae was observed on the substrate.


Upstream view of a second riffle observed upstream of the previous photo.

Upstream view at the upper end of the reach upstream of the Hwy 57 crossing. Note the wide channel with low streambanks. The shape is likely the result of the larger, more erosion resistant streambed substrate.

Downstream view from vantage point of previous photo. A cobble/boulder riffle spans the entire width of the channel. Sand was noted in interstitial spaces indicating deposition of finer material over the coarse material – not the presence of an armor layer over finer material.

Sign at the entrance to the solid waste facility along Hwy 47. GPS coordinates later revealed this facility is located outside of the Groundhouse River watershed.

Solid waste in the landfill left uncovered after the facility closed for the day.

Surface mining operation adjacent to the solid waste facility – this site also turned out to be outside of the watershed boundary.

Photographs taken September 27, 2006.

Downstream view of the Groundhouse downstream of the 150th Avenue crossing (first site visited south of Ogilvie). Despite the eroding streambanks observed at the next upstream site (at the Hwy 23 crossing), this wide reach (25 – 50 feet) contained cobble/boulder substrates with only minor sandy deposition.

Upstream view of a depositional bar along the inside of a bend. The vegetated bar consisted primarily of gravels.

Downstream toe of a mid-channel bar showing 18 – 24 inches of deposition. The finest material is deposited at the toe of the bar – in this case clean, coarse sand with little if any silt or clay. The finer size fractions are likely transported through this reach.

Downstream view of a run typical of this reach. The average depth was approximately 18 inches. A thin coating of fines was observed over the coarse gravel/cobble substrate, but this is likely indicative of the drought conditions throughout the summer; not of excessive deposition and infilling.

Upstream view of a pool along the toe of the left bank. A submerged depositional bar along the toe of the right bank was observed, but it is likely that the material is generated from upstream sources and not changes in the morphology of this reach. The pool depth was greater than 3 feet.

Downstream view of a large mid-channel depositional bar. The reach primarily consisted of long, low gradient runs interspaced with short, deep riffles. Finer depositional material was observed at the tail of each run, typically leading vegetated mid channel bar at the head of each riffle.

Upstream view the right bank showing the influence of understory vegetation (alder) on bank stability along the outside of a meander bend. This type of vegetation could be an appropriate means for reducing sediment generation along the cleared sections of the power line corridor at the next upstream site.

Upstream view of most downstream reach visited on the Groundhouse River (located off Hickory Street near intersection of Hwy 2 and 145th Avenue). The reach was approximately 60-feet wide with deep riffles and cobble substrate. The cobbles were partially embedded with sand-sized depositional material.

Downstream view showing long, low gradient runs. Broad floodplains were observed along both sides of the reach.

Example of a riffle in this reach. The riffles were relatively short and flow depth was deep. The water surface did not exhibit the turbulence typical of riffles. Boulder sized material was observed in the riffles.

Upstream view at the upstream end of the reach – just upstream of the riffle shown in the previous photo. Another long run is shown. Active deposition of sandy material was noted along the top of both streambanks (both of which were approximately 5 feet above the thalweg elevation).

Downstream view of most downstream site (off of 140th Avenue) on the South Fork Groundhouse River, just upstream of the confluence with the mainstem. The geometry was prismatic (approx. 50-ft wide, 2-ft deep, 7-8-ft bank heights), likely indicating historic channelization.

View of nearly vertical right bank. Deposition was evident at the toe of the bank. The valley wall confined the right overbank area, but a broad, accessible floodplain was observed on the left bank.

Upstream view of channelized reach. Spoil piles line both banks, although trees are established on the right bank spoil pile. The substrate was primarily sand. No cobbles or boulders were observed, nor were any riffles or pools (other than small scour pools).

Downstream view of the spoil pile along the right bank. Based on the size of the largest trees growing on the spoil pile, it was estimated that the channelization occurred no sooner than 50 years ago.

Upstream view of the South Fork at the 140th Street crossing. Note the herd of cattle on the hill in the background of the photo – the river smelled strongly of cow manure.

Rain gage and acoustic Doppler velocity (ADV) meter on the left of three culverts under the 140th Street crossing of the Groundhouse. No information was posted as to the owners of the equipment.

Upstream view of South Fork Groundhouse River from the Hwy 47 crossing. The historic channelization noted downstream is still evident at this reach – although the size of the channel is considerably smaller.

Animal operation along the left bank of the South Fork. The cattle were fed approximately 250-300 feet from the channel; however, a vegetated buffer was located between the two.

Row crop agriculture along the right bank. Again, note the presence of a vegetated buffer strip between the row crops and the channel.

View of roadside drainage ditch entering the South Fork from CR-49. The depositional fan at the toe of the riprap ditch indicates the graveled road surface does contribute some fine material to the system.

Upstream view of CR-49 crossing of the South Fork. The prismatic shape still indicates historic channelization of this reach.

Southeast corner of the intersection of CR-49 and CR-50 illustrating the influence of corn on ground cover.

Southwest corner of the intersection of CR-49 and CR-50 illustrating the influence of soy beans on ground cover.

Another example of graveled road cover, approximately 25-feet wide, crowned, and bladed in the previous few days – possibly move picture.

Downstream view of the South Fork at the CR-49 crossing. Note the influence of cattle on the streambanks where no riparian exclusion measures are in place. No cattle were observed in this reach.

Downstream view of CR-4 crossing the South Fork. Note the influence of cattle on the channel morphology as well as the green algae in the pool at the right of the photo.

Downstream view of Delta Street crossing of the South Fork Groundhouse. At this point, the channel did not exhibit any indicators of historic channelization.

Downstream view of CR-4 crossing. Fencing is installed to prevent cattle access to the channel; however, abundant filamentous, green algae was observed in the pool.

Upstream view of the South Fork at the CR-4 crossing. Boulders or possibly rip rap placed at the toe of the streambank along with the willows effectively control erosion along the toe of the bank. Row crop (corn) is visible in the background of the photo.

Downstream view of the South Fork at the CR-9 crossing. The channel curves toward the top-right corner of the photo (as indicated by the vegetation line).

Upstream view of turbid slow moving water in the South Fork at the CR-9 crossing between 130th and 140th Avenues. It is likely that this reach exhibits higher water temperatures and lower dissolved oxygen.

Downstream view of the South Fork at the same crossing as the previous photo. Note the turbid water and undisturbed water surface.

Upstream view of the South Fork at the CR-9 crossing (north of Hwy 23). The South Fork flows through a broad wetland at this point.

Downstream view of the South Fork from the same vantage point as the previous photo. About one dozen cattle were observed along this reach. The cattle have open access to the cannel, as indicated by the trampled streambanks.

Gravel pit (#33062) off of Hwy 23 where a reported drainage ditch is connected to the South Fork of Groundhouse River. The gravel pit no longer appears active, and the water in the pond was crystal clear.

View of the same pond as shown in the previous photo.

View of the gravel pit showing the contributing drainage area to the pond. The vegetation limits the potential for erosion of surface soils.


Another view across the pond toward the ditch to the South Fork.


View of the water clarity in the pond – note the submerged aquatic vegetation thriving in the clear water.

View of drainage ditch from pond toward the Groundhouse. No evidence of sediment input from this ditch was observed.

A view of gravel pit #33038 just east of CR-52. While a small drainage ditch runs through this property, the gravel pit does not appear to be a source of sediment to the South Fork of the Groundhouse River.

Another view of gravel pit #33038. Gravel pit #33039 was not found.

Close up view of gravel pit #33038. It appears that a paving company is now using the pit as a disposal site for asphalt and concrete waste – not as a source of gravel aggregate.

Example of a cattle pasture. A few dozen cattle were observed in this pasture.

Close up view of ground cover in the pasture shown in the previous photo.

Another example of ground cover observed in the cattle pasture shown two photos previous.

Gravel pit #33055 and or #33056 at the intersection of Hwy 47 and 110th Avenue. While marked as an active pit on the 2003 MNDOT map, this pit does not exhibit any signs of recent activity. As there is no outflow, it is unlikely that it is, or was, a source of sediment to the South Fork.

Gravel pit #33054 on the northeast side of the intersection shown in the previous photo. While the 2003 MNDOT map indicates this is an active pit, the vegetation indicates otherwise.

A view toward the North from 110th Avenue at commercial aggregate site #33087. The portion of the pit in the foreground does not appear active; the stockpiles in the background indicate where excavation activities were ongoing during the visit.

Inactive portion of aggregate pit #33087.

Close-up of groundcover in inactive portion of pit #33087.

Aggregate stockpiles at the eastern side of commercial aggregate pit #33093. As with pit #33087, the commercial operations are large-scale pits that disturb tens of acres. However, as precipitation is likely trapped in the pit, it is not expected that these operations are sources of sediment to the South Fork.

Gravel pit #33041 indicated as active on the 2003 MNDOT map no longer appeared active.

Close-up view of groundcover where row crops have been harvested.

Gravel pit #33057 along Hwy 47 just south of Ogilvie. While clearly no longer an active pit, it appears this area may now function as a wetland.

Another close-up of groundcover where row crops (corn) had recently been harvested. Note the patch of corn still standing in the top right corner of the photo.

Photographs taken September 28, 2006.

Close-up example of soybean cover.

Close-up example of groundcover between rows of corn.

Downstream view of the Groundhouse River at CR-53 crossing (north of CR-55). The channel appeared to be under a backwater, possibly from the influence of a beaver impoundment.

Upstream view of the Groundhouse River from the same vantage as the previous photo. The water was too dark to assess the size of the streambed substrate.

Upstream view of the Groundhouse River at the CR-55 crossing (east of CR-53). At this point (downstream of the previous photos), the channel no longer appears to be impounded. The two cobble riffles shown in this reach indicate a free flowing system.

Downstream view of the Groundhouse from the same vantage point as the previous photo. Another cobble riffle was observed. The morphology of the channel did not appear to be an active source of sediment to downstream reaches.

Hay or clover (?) field in the Groundhouse watershed after a recent harvest.

Close-up of groundcover (hay or clover?) shown in the previous photo.

Inactive gravel pit #33048 on the eastern side of Hwy 47 just north of Ogilvie.

Extensive streambank erosion along the Groundhouse River off of 172nd Avenue to the west of Hwy 47 (just north of Ogilvie). Note the mid-channel bar in the foreground. While cattle were not observed, it is obvious that animal have access to the river throughout this pasture.

Downstream view of the Groundhouse River from the same vantage point as previous photo. Streambank erosion continues through this reach.

Close-up of streambank slumping. Based on the knickpoint at the top of the streambank and the grass on the streambank, it is clear the toe of the streambank is being eroded and the upper bank slumps into the river.

Another close-up view of the streambank erosion. This reach of the river is located in glacial outwash sands that afford little resistance to hydraulic stresses – particularly when no woody vegetation is present and livestock can trample the banks.

Upstream view of the Groundhouse from the Hwy 47 crossing – approximately 2,000 feet downstream from the previous photos. Note the gravel cobble substrate on the streambed and the stable streambanks.

Downstream view of the Groundhouse from the Hwy 47 crossing. Where the channel is overly wide due to the road crossing, a gravel mid-channel bar has formed. Note the ATV access trail and tire tracks on the bar.

View of dead fish in the Groundhouse at the upstream side of the CR-60 (Hill Ave) crossing in Ogilvie. It is likely the fish were dumped into the river. Also not the large cobbles and boulders in the channel.

Downstream view of the Groundhouse from the CR-60 crossing. The streambanks appear stable with accessible floodplain on both streambanks. Coarse substrate is again prevalent in the streambed.

Downstream view of the South Fork at the CR-50 crossing. This reach appeared to have experienced historic channelization. Due to the open canopy and high nutrient load, filamentous green algae was abundant.

Upstream view of drainage ditch along CR-50 (facing south). Due to the well maintained road surface and the vegetation in the ditch, most eroded sediment is likely trapped in the ditch before being transported to the river.

View to the east of a lateral ditch into an agricultural field. The lateral ditch connects to the drainage ditch that runs along CR-50.

Downstream view of drainage ditch in the South Fork watershed as seen from 110th Avenue. The ditch drains agricultural areas. An animal, resembling a beaver, was observed swimming in this ditch. Possibly a beaver impoundment is backing up flow and enhancing algal growth.

Upstream view of drainage ditch as seen from previous vantage point. Again, note the abundance of green algae.

Downstream view of the Groundhouse River at the Hwy 65 crossing (downstream and out of the watershed).

Upstream view of the Groundhouse River from the Hwy 65 crossing.

Appendix D. Biological Data Assessment

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1 Introduction

The Stressor Identification process is a formal method by which the causes of biological impairment may be identified through a step-by-step procedure (Cormier et al., 2000). In this process, existing biological, chemical, physical, and land-use data are analyzed to determine probable causes of impairment for aquatic organisms. This procedure lists candidate causes for impairment, examines available data for each candidate, and characterizes the probable cause(s) (Figure D-1).

A report by Lane and Cormier in 2004 enumerated possible causes of impairment to the Groundhouse River using the Stressor Identification process. Based on the limited data available for analysis, this initial report did not establish definite causes for the previously identified biological impairments. However, the report was able to eliminate possible causes from consideration, while directing future data collections. According to Lane and Cormier, the hope was that future collections would provide the basis for a definitive analysis that would directly implicate one or more causes for impairment.

This report is intended to build upon the report by Lane and Cormier (2004) by utilizing updated field data, spatial datasets, and techniques for linking possible impairment sources to the observed impairments. Findings and recommendations from Lane and Cormier (2004) are used in cases when new or updated data are not available.

Data are limited for some parameters assessed in this report. For example, dissolved oxygen and temperature data fluctuate throughout the day and continuous measurements would allow for a more thorough analysis of the relationship to biological response. As a result, the project team had to rely on best professional judgment to determine the dominant stressors leading to impaired biota within the Groundhouse River system.

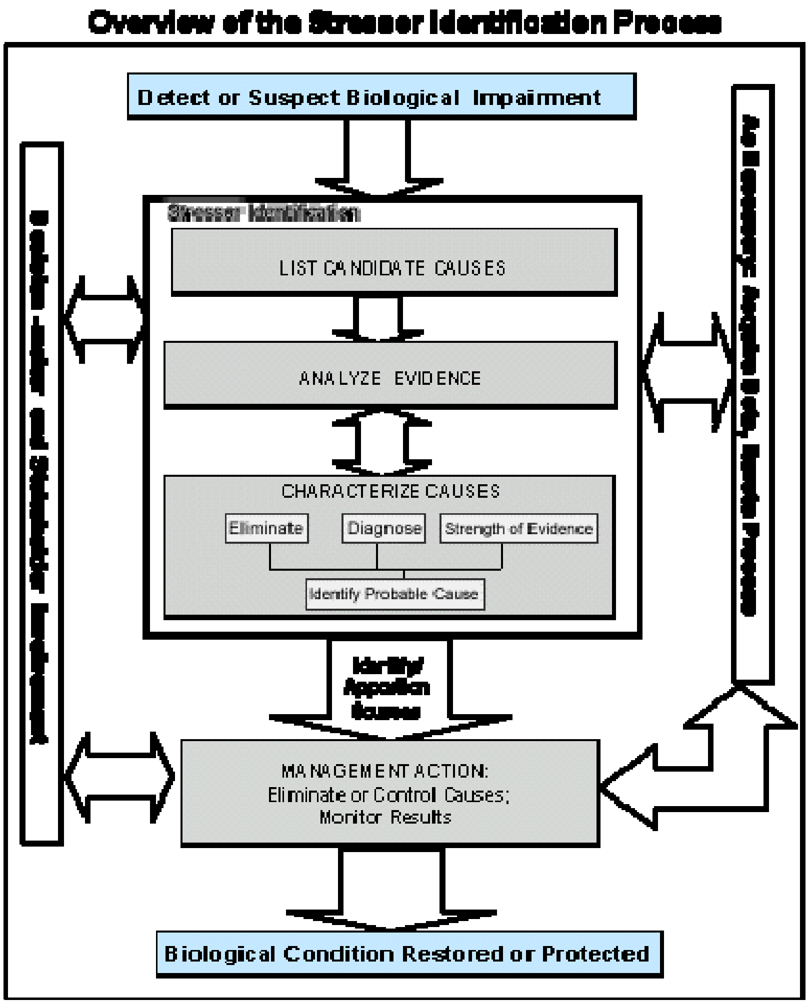


Figure D-1. The Step-by-Step Procedure of the Stressor Identification Process taken from (Cormier et al, 2000).

2 Background

The primary land use/land cover types in the Watershed are deciduous forest (47.5%), pasture/hay (20%), cultivated crops (11.8%), and emergent herbaceous wetlands (13%) (Figure D-2). Like many other rivers in the St. Croix Basin, the upper reaches of the Groundhouse flow through areas dominated by wetlands and forests. The Groundhouse River receives flow from several tributaries, most notably the North and South Fork Groundhouse Rivers. Urban development is generally light; the largest city in the Watershed, Ogilvie, has a population of approximately 500 people.

Ecoregions (Omernik, 1995) can be used to broadly classify the landscape of the watershed. Ecoregions are defined by biotic and abiotic variables including: "geology, physiography, vegetation, climate, soils, land use, wildlife, and hydrology" (EPA 2002). The Groundhouse River Watershed is split between two Level III ecoregions. The northern half of the watershed lies within the Northern Lakes and Forests Ecoregion, while the southern half lies within the North Central Hardwood Forests Ecoregion. According to descriptions from the USEPA, the Northern Lakes and Forests Ecoregion is characterized by nutrient poor or sandy soils associated with glaciations and outwash plains and also characterized by an abundance of lakes, many less productive than southern lentic systems. The dominant ecosystems are coniferous and northern hardwood forests. The North Central Hardwood Forests Ecoregion is a transitional area between the coniferous and hardwood forests to the north and the agriculture based ecoregions found to the south. The area is characterized by lake and wetland areas, hardwood forests, cropland, pasture, and other livestock operations (EPA 2002).

The watershed also spans two agro-ecoregions. Agro-ecoregions are similar to ecoregions, but expand regional differences to include land management practices or strategies as well as natural resource concerns. The Groundhouse River Watershed contains two agro-ecoregions. The western section of the watershed lies within the Drumlins agro-ecoregion, while the eastern half lies within the Central Till agro-ecoregion.

Other information on the characteristics of the Groundhouse River watershed (e.g., soils, population, potential pollutant sources) is available in the *Draft Watershed Characterization and Water Quality Modeling Report* (Tetra Tech, 2006).

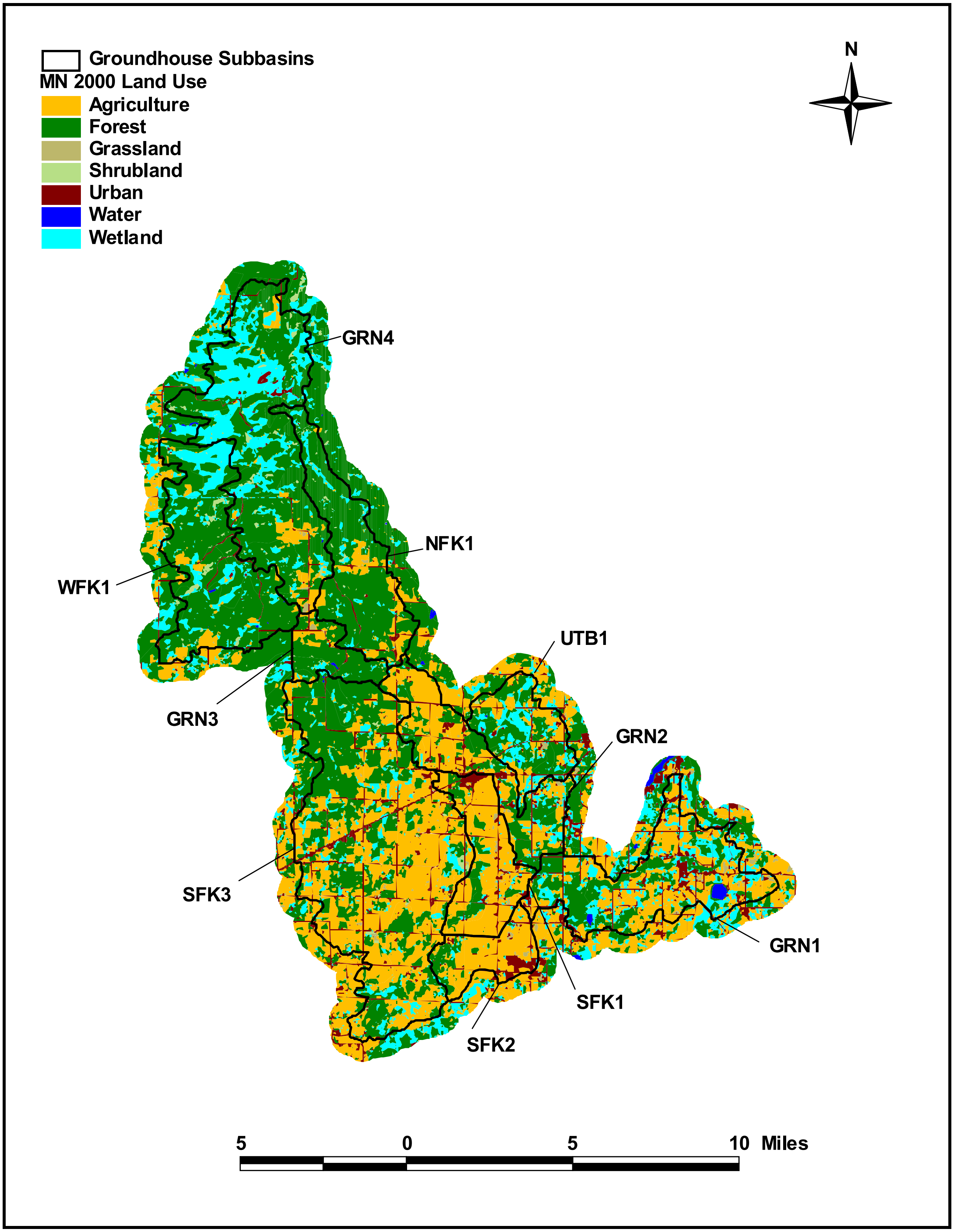


Figure D-2. University of Minnesota 2000 Landsat Land Cover Classification

3 Data Sources

This section of the report presents information on the data used to conduct the biological assessment.

3.1 Biological

Data from the main stem of the Groundhouse River and the South Fork Groundhouse River watersheds were evaluated to determine which water quality or physical habitat factors are related to biological impairments. Biological impairments were identified by analysis of benthic invertebrate communities and fish communities collected from select sites along with supporting chemical and physical data.

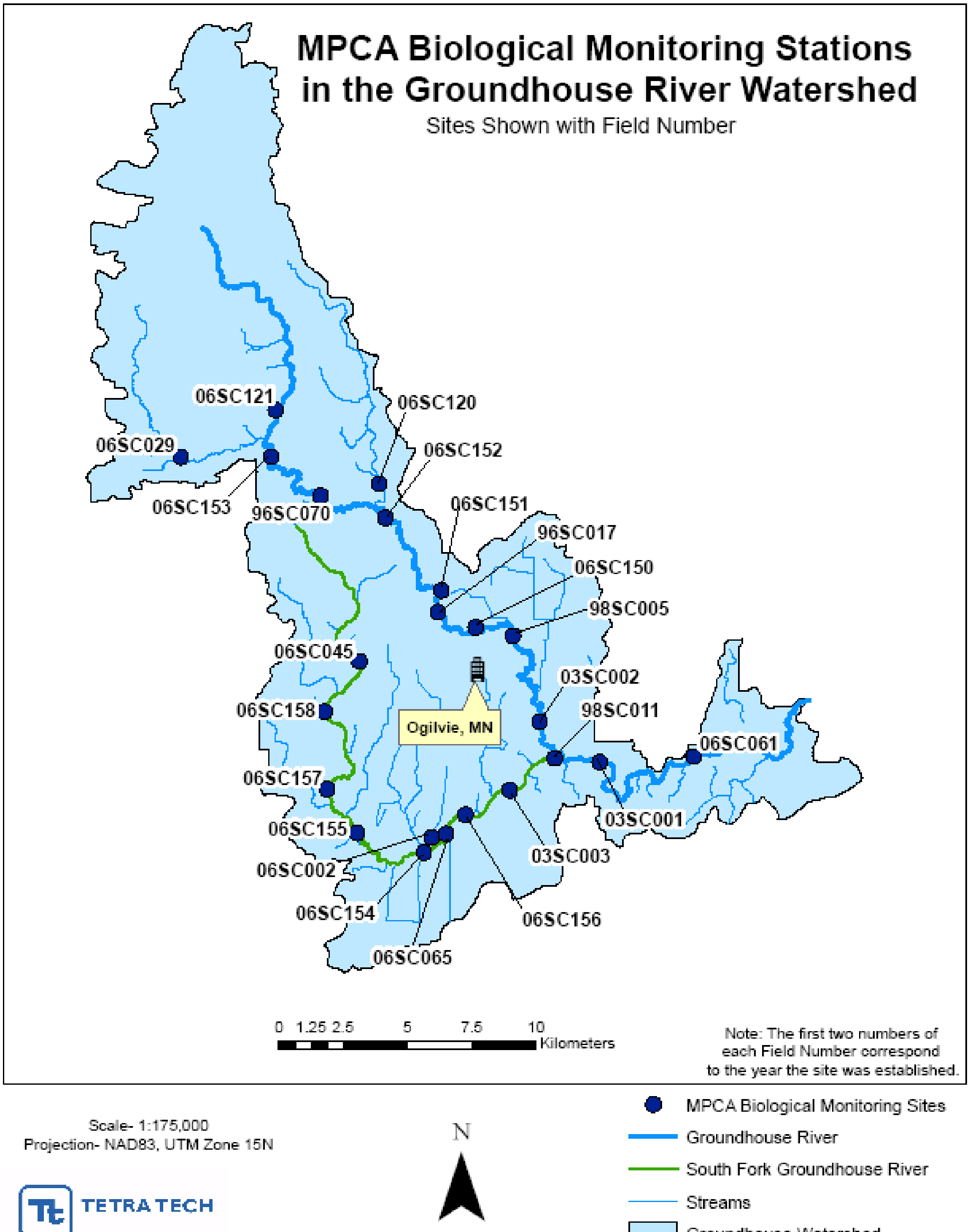

MPCA Biological Monitoring Stations in the Groundhouse River Watershed
Sites Shown with Field Number
06SC121
06SC120
06SC029
06SC152
06SC153
96SC070
06SC151
96SC017
06SC150
98SC005
06SC045
03SC002
Ogilvie, MN
06SC158
98SC011
06SC061
06SC157
03SC001
06SC155
03SC003
06SC002
06SC154
06SC156
06SC065
0 1.25 2.5 5 7.5 10
Kilometers
Note: The first two numbers of each Field Number correspond to the year the site was established.
Scale- 1:175,000
Projection- NAD83, UTM Zone 15N
N
MPCA Biological Monitoring Sites
Groundhouse River
South Fork Groundhouse River
Streams
Groundhouse Watershed
TETRA TECH

Table D-1. Data available for analysis of South Fork Groundhouse River.

Site	Year	Biological	Physical Habitat	Water Quality
98SC011	1998	F, I	X	X
98SC011	2006	F		
98SC011	2006	F, I	X	
03SC003	2003	F, I	X	
03SC003	2005	F, I	X	X
03SC003	2006	F		
06SC156	2006	F		
06SC065	2006	F		
06SC065	2006	F, I	X	X
06SC154	2006	F, I	X	X
06SC155	2006	F, I	X	X
06SC156	2006	F, I	X	X
06SC158	2006	F, I	X	X
06SC045	2006	F		
06SC045	2006	F, I	X	X

Note: F = Fish community characterization
I = Benthic invertebrate community characterization

Table D-2. Data available for analysis of Mainstem Groundhouse River.

Site	Year	Biological	Physical Habitat	Water Quality
06SC061	2006	F, I	X	X
03SC001	2003	F, I	X	X
03SC002	2003	F, I	X	X
03SC002	2005	F		
03SC002	2006	F, I	X	X
98SC005	1998	F, I	X	X
98SC005	2003	F, I	X	X
06SC150	2006	F, I	X	X
96SC017	1996	F		
06SC151	2006	F, I	X	X
06SC152	2006	F, I	X	X
96SC070	1996	F		
96SC070	2005	F		
96SC070	2006	F, I	X	X
06SC153	2006	F, I	X	X
06SC121	2006	F		

Note: F = Fish community characterization
I = Benthic invertebrate community characterization

Fish and macroinvertebrate community composition data were used to generate indices of biotic integrity (IBIs) (Niemela and Fiest, 2000; Chirhart, 2003). Numeric IBI scores are computed by applying multi-metric criteria to raw fish and macroinvertebrate community composition data. Metrics included in the IBI are chosen based on their response to human alterations on the landscape – only metrics that respond to anthropogenic affects are used. Chosen metrics are typically summed to produce a final IBI score. The metrics are calibrated to reflect a negative response to increasing stress in the aquatic environment. Metrics of the fish IBI and macroinvertebrate IBI (hereafter referred to as 'MIBI') metrics fall into three and four categories, respectively (Table D-3).

Table D-3. Metric categories of the IBI and MIBI (Niemela and Fiest, 2000; Chirhart, 2003).

Fish	Macroinvertebrates
Species Richness/Composition	Species Richness
Trophic Composition and Reproductive Function	Species Tolerance
Abundance and Condition	Species Composition
	Trophic Structure

Since metrics respond to anthropogenic disturbance differently based on a site's location within a watershed or the stream morphology type, MPCA uses different metrics based on the drainage area (IBI) and stream type (MIBI) (Niemela and Fiest, 2000; Chirhart, 2003) (Table D-4and Table D-5).

Table D-4. Metrics Used in the St. Croix River Basin Macroinvertebrate IBI, by Reach Type (Chirhart, 2003)

Metric Name	Glide Pool	Sm. Riffle-Run	Lg. Riffle-Run
# Ephemeroptera Taxa		X	
# Plecoptera Taxa		X	
# Trichoptera Taxa		X	X
# Chironomidae Taxa	X	X	
# POET Taxa	X		
# Intolerant Taxa	X	X	X
% Tolerant Taxa	X		X
# Clinger Taxa	X	X	X
# Tanytarsini Taxa	X	X	
# Gatherer Taxa	X	X	
# Filterer Taxa			X
% Amphipoda Taxa	X	X	X
% Dominant 2 Taxa	X	X	

Table D-5. Metrics Used in the St. Croix River Basin Fish IBI, by Drainage Area (Niemela and Feist, 2000)

Metric Name	<20 mi.^{2}	20 - 54 mi.^{2}	55 - 270 mi.^{2}
Total # Species	X	X	X
# Headwater Species	X		
# Minnow Species	X	X	
# Intolerant Species		X	X
# Darter Species			X
% Tolerant Species	X	X	X
% Dominant Two Species	X	X	
# Benthic Insectivore Species	X	X	X
# Omnivores Species			X
% Piscivore Species			X
% Simple Lithophils	X	X	X
# Fish per 100 m *	X	X	X
% DELT Anomalies	X	X	X
# Insectivore Species	X		

Final IBI scores can be used as an indicator of the overall health of the fish and macroinvertebrate communities. Metrics that make up the IBI should respond to anthropogenic disturbance; some metrics can be used as a general indicator of disturbance, while others can be an indicator of a specific stressor (Niemela and Feist, 2000; Chirhart, 2003). Because fish and macroinvertebrates are able to respond differently to disturbance and stress, it is helpful to look at both communities in the assessment of the biological health of an aquatic system. For instance, fish are mobile and may be better able to respond to impairment in isolated reaches of stream ecosystems than macroinvertebrate communities which depend on bed substrate and local water quality.

Further use of genus/species identities provides additional evidence for identification of biological effects from a specific pollutant(s). Several benthic macroinvertebrate taxa are strong indicators for presence of certain types of environmental conditions. Examination of species lists from a longitudinal gradient of sites reveals the dominant environmental gradients for biological response. These are important to identify as the gradients may be a result of human disturbance or may be natural features that can modify the effect of a pollutant on biological communities.

Data collected by the MNDNR in the summer of 2005 were used to complete a standard MNDNR Stream Survey report (Frank, 2005), and to calculate IBI scores consistent with MPCA protocols. Some of these data do not appear to be calculated correctly, however, and in the absence of verification from MNDNR were not included in this report's analysis.

3.2 WATER QUALITY

Water chemistry grab samples were collected at each of the biological sampling sites. Parameters measured include: water temperature (^{o}C), conductivity (µmhos/cm), dissolved oxygen (mg/L), pH, turbidity (Nephelometric Turbidity Units or NTU), total suspended solids (mg/L), total phosphorus (mg/L), ammonia (mg/L), and nitrate + nitrite nitrogen (mg/L). Typically, these parameters were measured only once during a site visit.

Additional water chemistry data were collected at 11 sites specifically for the Groundhouse River TMDL and are stored in the EPA STORET database (Figure D-3). These data were collected during the summer of 2005 over a range of dates. Not every site was sampled for the same parameters, but most sites included water temperature, dissolved oxygen, pH, and total suspended solids.

Although most of the STORET water quality sites include numerous grab samples over an individual summer, continuous data are not available for any parameter.

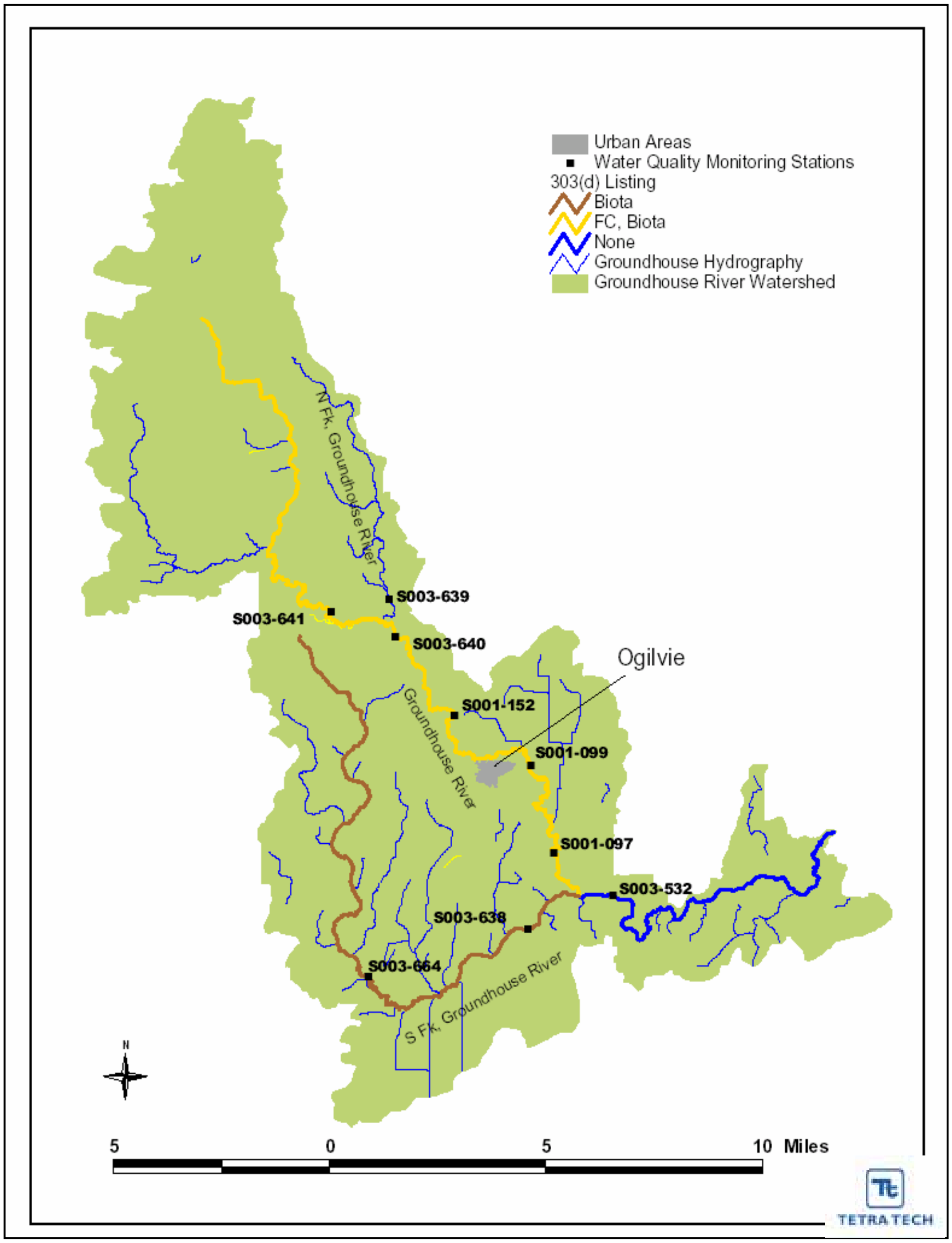


Figure D-3. STORET water quality stations in the Groundhouse River Watershed

3.3 HABITAT / GEOMORPHIC

Detailed habitat data were collected at several of the biological sampling sites; detailed habitat data were not collected as part of the MNDNR Stream Survey in 2005. Data were collected following standard procedures developed by the MPCA (Anon., 2002b). Habitat parameters included: stream bed composition (multiple variables), sinuosity, stream morphology type (i.e. pool, riffle, run), gradient, erosion, and fish cover. A select set of these measurements used in further analysis of biological response were gradient (%) and percent fine sediment. Qualitative habitat and geomorphic data were collected by Tetra Tech in October 2006 and were also available from a University of Minnesota study of the watershed conducted in 2005 (Hansen, 2005).

4 Pollutant Sources and Candidate Causes of Biological Impairment

This section begins by looking at possible point and nonpoint pollutant sources in the Groundhouse River watershed, followed by an overview of the various potential causes of impairment.

4.1 Potential Point Sources

According to land use/land cover data available from the University of Minnesota, the land cover in the Groundhouse River Watershed is dominated by forest (47.5%) and agriculture (31.8%). Only 5.9% of the watershed is urban. There are relatively few facilities or industrial activities at a large enough scale to represent point sources for the Groundhouse River. These few, however, may be significant and are described below.

4.2 Potential Nonpoint Sources

A large portion of the Groundhouse River Watershed is used for agriculture. According to data available from the University of Minnesota, approximately 32 percent of the land use/land cover in the watershed is used for either row crop cultivation or as pasture for livestock. The amount of land used as pasture versus the amount planted as row crop varies per sub basin; however, the majority of the land draining to the main stem of the Groundhouse River is classified as pasture.

4.2.1 Feedlots

Table D-6. Number of Animal Units per Head by Animal Type

Animal Type	Number of Animal Units per Head
Dairy Cattle	
Mature cow (milked or dry) over 1,000 pounds	1.4
Mature cow (milked or dry) under 1,000 pounds	1
Heifer	0.7
Calf	0.2
Beef Cattle	
Slaughter steer/heifer, stock cow, or bull	1
Feeder cattle (stocker or backgrounding) or heifer	0.7
Cow and calf pair	1.2
Calf	0.2
Swine	
Over 300 pounds	0.4
Between 55 and 300 pounds	0.3
Under 55 pounds	0.05
Horse	1
Sheep or lamb	0.1
Veal	0.2
Chicken: Layer Hens or Broilers	0.033

The impact of AFOs and AFO management practices on river ecosystems is multi-fold. It is possible that effluent run-off from improperly stored manure or run-off generated during precipitation events could elevate sediment, nutrient, and fecal coliform levels in the water column. Removal of riparian vegetation during the conversion of an area to pasture can decrease the ability of the landscape to retain nutrients, can increase the speed and volume of overland flow reaching the stream channel, and may increase water temperature by reducing the available cover. An increased nutrient load can drive a decrease in dissolved oxygen by increasing respiration, particularly of photosynthetic organisms. This effect may be exacerbated by the increase in temperature associated with reduced vegetative cover. Land converted to row crop agriculture that is accompanied by removal of riparian vegetation can have a similar impact.

4.2.2 Soil Erosion from Cultivated Farmland

Cultivated farmland in the Groundhouse River watershed is also a potential source of nutrients and sediment. Accumulation of nitrate + nitrite and total phosphorus on cropland occurs from decomposition of residual crop material, fertilization with chemical (e.g., anhydrous ammonia) and manure fertilizers, atmospheric deposition, wildlife excreta, irrigation water, and application of waste products from municipal and industrial wastewater treatment facilities. Surface erosion from bare fields is the primary source of TSS from agricultural lands, although streambank erosion can also be worsened by agricultural activities that remove riparian stream corridor vegetation and de-stabilize streambanks.

4.3 IDENTIFIED CANDIDATE CAUSES

The previous report by Lane and Cormier (2004) considered four candidate causes for biological impairment:

1) loss of habitat from unstable or unsuitable substrates
2) decreased dissolved oxygen availability associated with excessive nutrient loading
3) altered food source caused by excessive nutrient loading
4) chronic or acute toxicity from chemical compounds

Based on data compiled for the TMDL (Tetra Tech, 2006), it is unlikely that toxic chemical compounds are impacting the biological communities of the Groundhouse River. For example, a component (metric) of the fish IBI can be used as an indicator of acute toxicity. Decreases in the DELT (Deformities, Eroded Fins, Legions, and Tumors) component of the IBI are often associated with environmental degradation due to industrial pollutants (Sanders et al, 1999; Ohio EPA, 1988). In the Groundhouse River watershed, every one of the 26 sites received a DELT score of 10 out of 10, indicating little possibility of the presence of toxic chemicals. Monitoring data for toxic chemicals is not available in the Groundhouse River watershed, so a review of this candidate cause is not possible. When toxic chemical data are not available, the presence of common chemical sources must be considered (CADDIS, 2005). However, neither urban development nor high-intensity agricultural operations are present in the watershed, making acute toxicity from chemicals a highly unlikely cause for biotic impairment.

Based on the potential impacts of the point and nonpoint sources identified in the watershed, for the purposes of this report, three candidate causes are considered: (1) an altered temperature regime, (2) low dissolved oxygen levels, and (3) habitat degradation due to fine sediment deposition. The candidate cause “altered food source caused by excessive nutrient loading” was not directly addressed but is indirectly evaluated through the low dissolved oxygen analysis because nutrient availability promotes some of the food sources available to benthic macroinvertebrates and were used as a surrogate measure. There were no available data that directly relate to available food sources other than surrogate indicators and visual analysis for autochthonous input from riparian areas.

4.3.1 Candidate Cause 1: Altered Stream Temperature Regime

Water temperature can shape the overall structure of an aquatic ecosystem; average and flux of water temperature can affect every aspect of a stream system (Table D-7).

Table D-7. Attributes of Aquatic Systems Affected by Temperature (CADDIS, 2005)

Category	Example attributes
Physical:	Water density, thermal stratification, solubility of oxygen and other chemicals
Chemical:	Rates of nutrient cycling, contaminant transformation rates
Biological:	Organism survival, growth, reproduction, development, behavior, habitat preference, competition

Since different aquatic taxa require different thermal habitats, observing changes in taxa may indicate a shift in the temperature regime. Many anthropogenic activities can disrupt stream thermal dynamics resulting in increased water temperatures (Allan, 2005; Poole and Berman, 2001), thus potentially altering fish and macroinvertebrate assemblages.

Changing temperature regimes can affect fish and macroinvertebrate communities at multiple levels. In closed ecosystems (i.e. lakes), changing temperature could result in organism mortality. It could also have non-lethal affects, such as decreased growth or decreased fecundancy. In open systems (i.e. streams), organisms may have the opportunity to move to locations with more suitable thermal regimes.

Anthropogenic activities that can result in increasing stream temperatures include the discharge of heated water from an industrial facility, removal of riparian vegetation (Moore at al, 2005), and straightening of channels (Paul and Meyer, 2001).

4.3.2 Candidate Cause 2: Decreased Dissolved Oxygen Levels

Dissolved oxygen (DO) is required by all aerobic stream organisms. Fish and macroinvertebrates move water across gill structures (passively and actively) to obtain the oxygen required for respiration.

In an open system, fish leave areas with low DO levels. For macroinvertebrates, however, or in a closed system, the impact of decreased oxygen availability can be much more severe limiting the number of organisms able to survive in particular environments, or even causing large scale deaths. “Fish kill” events where most or all of a fish population are found dead is a symptom of anoxic and eutrophic aquatic environments.

DO flux, or the diurnal change in dissolved oxygen, can also have a significant impact on biota. A stream with low DO during even a limited time each day or seasonally can be intolerable to fish or macroinvertebrate species.

Low dissolved oxygen may also decrease nitrification rates. This will decrease available nitrate while increasing available ammonia. This effect will increase with increasing nutrient loading. High concentrations of ammonia can adversely impact biotic communities, particularly benthic macroinvertebrates. Biological communities can often act as indicators of habitat quality trends. It is possible that biological communities will respond to changing conditions before the conditions themselves are defined as problems. For instance, biological communities may respond to increased nutrient loads prior to those loads exceeding local nutrient guidances.

4.3.3 Candidate Cause 3: Excess Fine Sediment Leading to Habitat Loss

According to the EPA, excess fine sediment is one of the leading causes of water quality impairment in the U.S. (CADDIS, 2005). Excess sediment can have detrimental effects on stream biota, including reduced feeding ability, depressed growth and reproduction rates, decreased oxygen levels, increased temperature, and loss of habitat (Henley et al, 2000; Waters, 1995).

Fine sediment in streams can be both suspended in the water column, and deposited in the stream bed. Suspended sediment is often measured by the turbidity or clarity of the water. It can affect stream biota directly by clogging gill rakers and limiting growth, and indirectly by reducing feeding ability. Bedded sediment is often measured as the percent of the stream bed that is covered in fine sediment, or the depth of fine sediment covering the bed. Bedded sediment can affect stream biota by filling interstitial spaces in course substrate used by fish and invertebrates for feeding and habitat (Henley, 2000).

Excess fine sediment can be introduced to a stream system in several ways. Loss of vegetation in the upland watershed may facilitate the deposition of erodible material during precipitation events and subsequent runoff. Channel alteration can disrupt stream banks and beds, allowing more sediment to be entrained in the water column. Livestock grazing near a stream can lead to loss of vegetation in the riparian zone and to trampled stream banks, allowing more fine sediment to reach the stream (CADDIS, 2005; Zimmerman et al, 2003). It is also possible that a stream with decreased velocity and energy due to natural or altered geomorphology may have a decreased ability to move fine sediment. In this way, even in the absence of an excess sediment load, bedded sediment may increase more than is tolerable to resident biota.

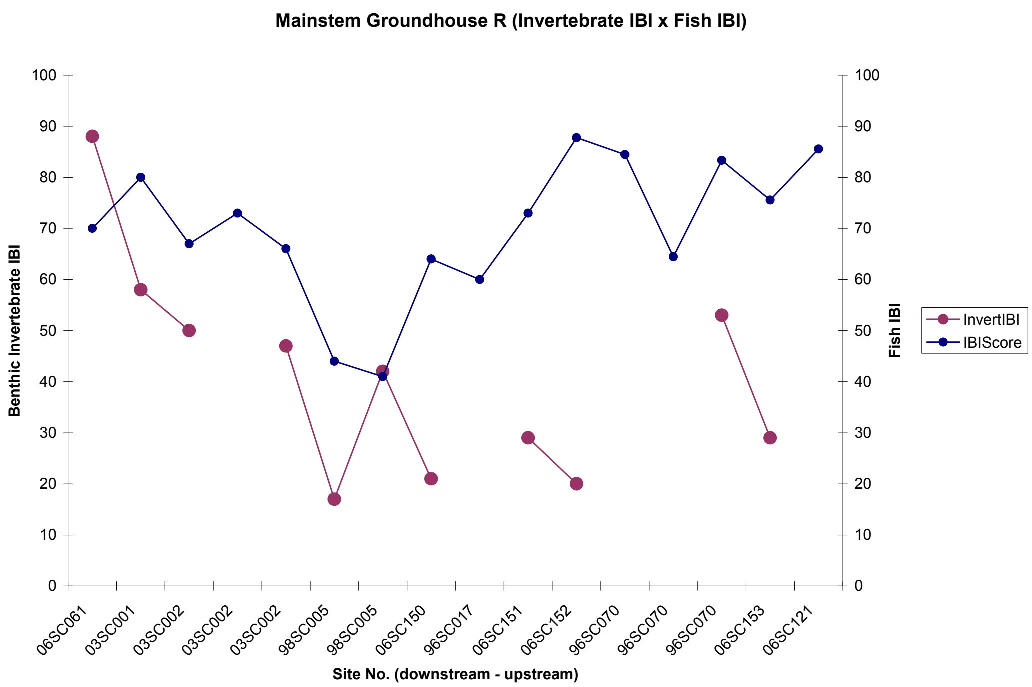
5 Data Review

This section of the report summarizes the available fish and macroinvertebrate data.

5.1 Multiple Assemblages and Selective Stressor Response

Using multiple biological assemblages (e.g., fish and benthic invertebrates) for detecting impairments is advantageous in situations where specific stressors selectively affect an assemblage. Both fish and benthic invertebrate data were collected at most sites on the main stem and South Fork of the Groundhouse River drainage. A comparison of the two assemblages is included in the following figures.

Analysis of this data identifies sites where response of each assemblage is divergent (responding in opposite directions). This can be interpreted as selective sensitivity to specific stressors measured at the stream site. Determining assemblage sensitivity to specific stressors can help in the design of an assessment program that is efficient and whose indicators will detect the presence of sources for impairments before they permanently harm biological communities.



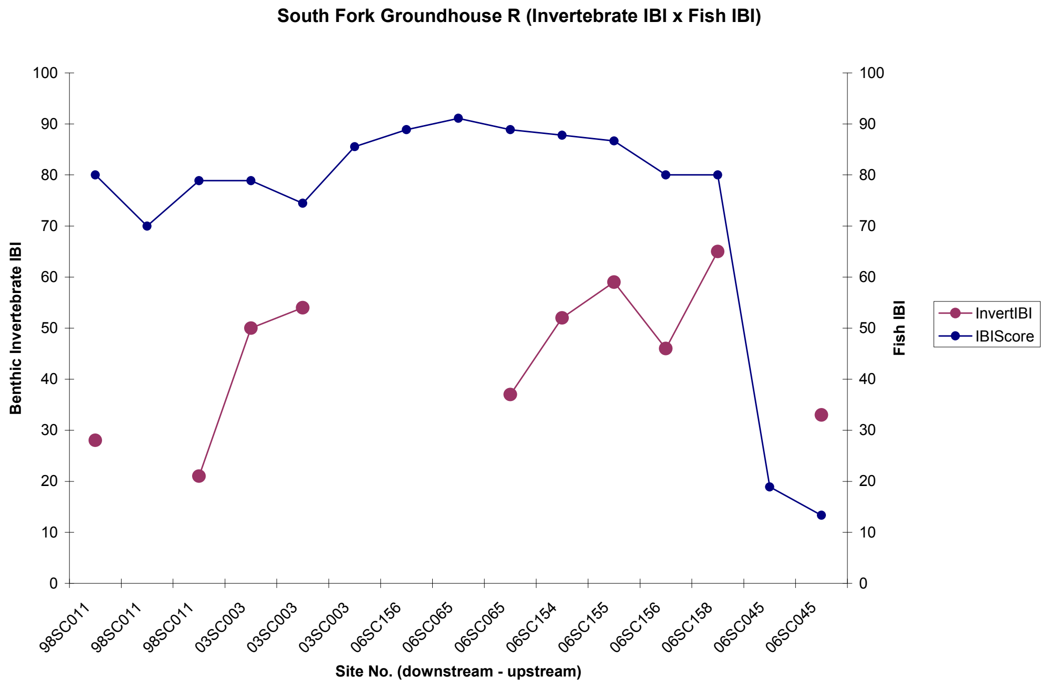


Figure D-4. Comparison of benthic invertebrate and fish community conditions for sites in the Mainstem Groundhouse River and South Fork Groundhouse River.

Fish

This section of the report discusses the available fish data in more detail and identifies the most likely priority stressors.

5.2 Impaired Area 1 – Ogilvie, MN

Fish community data have been collected at 13 sites along the main stem of the Groundhouse River (Table D-2). Only the 3 sites in Impaired Area 1 fall below the impairment threshold; IBI scores at the remaining 9 sites indicate relatively healthy fish communities, with a mean IBI score of 76. The availability of fish community data at so many sites along the main stem provides a unique opportunity for analyzing data along a geographic gradient. Many of the non-impaired sites are similar to the impaired sites in drainage area, adjacent land use, and channel morphology. Examining differences in the non-impaired and impaired sites may therefore provide evidence for or against the candidate causes of impairment. Six sites along the main stem were not used in the geographic gradient analysis because they had either dramatically different substrate types or contributing drainage areas. It is important to note that, due to changing drainage areas, the geographic gradient analysis includes IBI scores calculated with some technical changes. Sites above 06SC151 are calculated using a "small stream" IBI, while sites below are calculated using a "moderate stream" IBI (Niemela and Feist, 2000). The main differences between the two IBIs are the metrics used to calculate the scores. Metrics that change between the two stream sizes were not compared as part of the geographic gradient analysis.

Figure D-5 shows fish IBI scores along the geographic gradient of the Groundhouse main stem. Mean scores were calculated when sites had been visited more than once. Confidence intervals and standard error were not calculated due to the small sample size at each site; bars in Figure D-5 indicate the measured IBI range. Note that one sites (03SC002) provided a single IBI score that fell at or near the impairment threshold. This analysis will focus on those sites with scores that have consistently fallen below the impairment threshold or were sampled only once and found to be below the impairment threshold, namely sites 96SC017, 06SC150, and 98SC005 (Impaired Area 1). The Lane and Cormier (2004) report focuses almost exclusively on the site adjacent the town of Ogilvie, MN; that site is highlighted in red in each geographic gradient analysis figure.

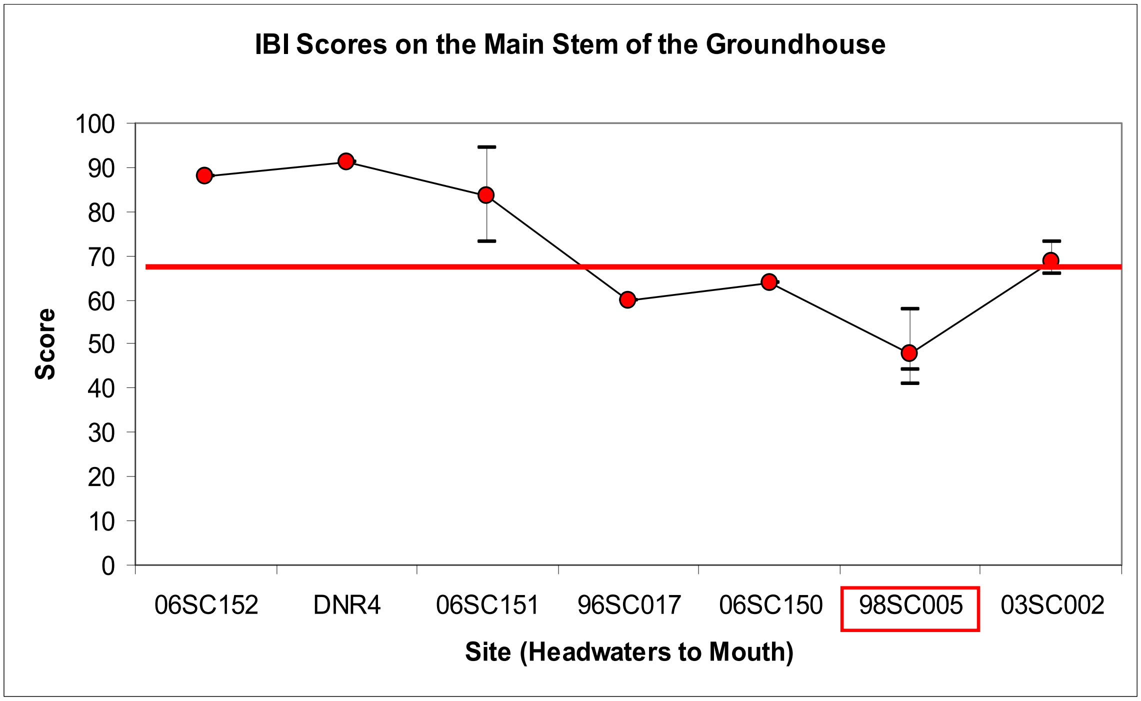


Figure D-5. IBI scores along a geographic gradient on the Mainstem of the Groundhouse River, MN. Red line indicates the approximate impairment threshold. Red dots indicate mean scores and bars indicate measured score range. Red box highlights site adjacent to the town of Ogilvie, MN. Note that the threshold for sites upstream of site 06SC151 is 46.

5.2.1 Candidate Cause 1: Altered Temperature Regime

Although IBI scores at Impaired Area 1 indicate disturbance, neither the IBI score itself or any of the individual metrics respond solely to changes in thermal regimes. Therefore, water temperature data were directly examined to assess temperature change as a candidate cause of impairment.

Several sources that could potentially increase water temperature or change the temperature regime can be identified near Impaired Area 1. First, riparian vegetation has been removed inside of Impaired Area 1 for a power line right-of-way. Vegetation has also been removed for animal feeding operations with cover and bank stability further decreased by cattle access. In these areas, reduced canopy cover has let to an increase in solar radiation reaching the stream surface.

Figure D-6 shows water temperature data collected at STORET water quality sites along the main stem of the Groundhouse River. The majority of temperature data was collected in 2005 and those data are displayed here. Station S001-099 (red line) is located within Impaired Area 1, close to the most impaired site in the area. The data indicate that stream temperatures near Impaired Area 1 follow the same pattern observed at other stations along the main stem. In fact, it appears that Station S001-099 has a lower temperature in July than any of the other sites along the Mainstem of the Groundhouse. This trend continues: temperatures at Impaired Area 1 were often lower than temperatures observed at other locations.

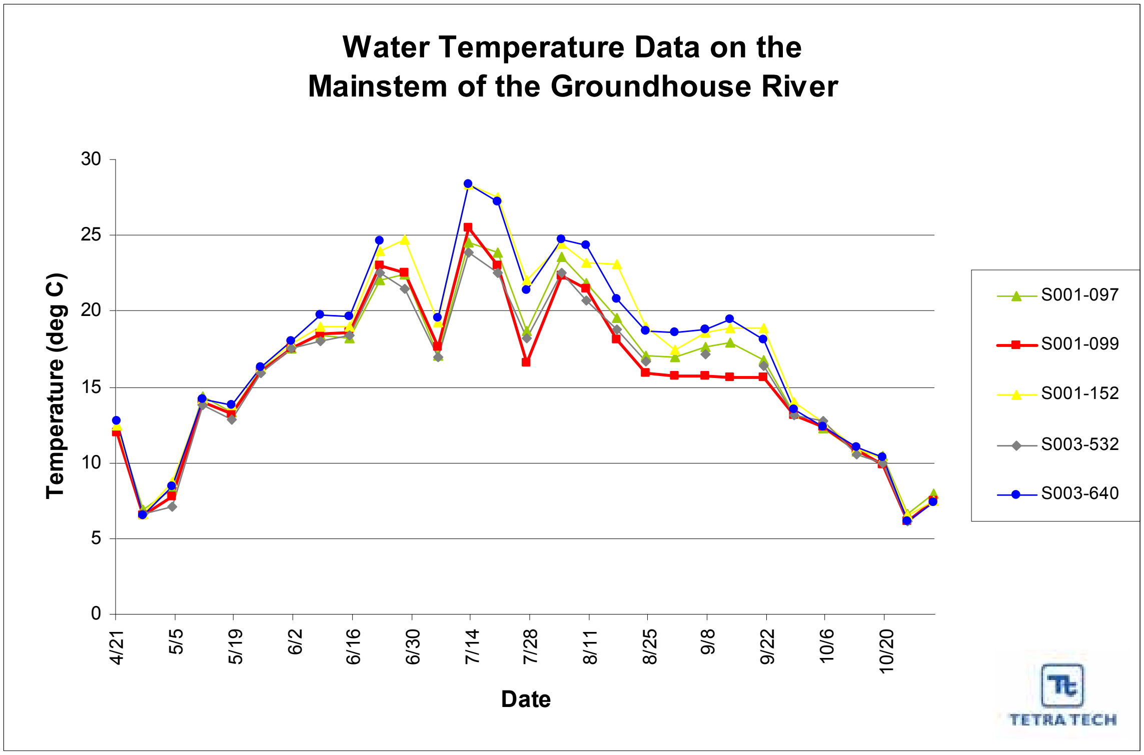


Figure D-6. Water Temperature Values along the Mainstem of the Groundhouse River. Site S001-099 (in red) is Closest to Impairment Area 1.

Based on the data shown in Figure D-6, changed temperature regime is an unlike stressor on the fish community at Impaired Area 1. If temperature were a stressor, we would expect to see a difference between temperature data at Impaired Area 1 and other non-impaired areas of the Mainstem. Despite potential sources driving temperature increase in Impaired Area 1, the data do not support the hypothesis that an altered temperature regime is responsible for the impairment of the fish community.

5.2.2 Candidate Cause 2: Decreased Dissolved Oxygen Levels

Although IBI scores at Impaired Area 1 indicate disturbance, neither the IBI score itself or any of the individual metrics respond solely to changes in dissolved oxygen. Therefore, dissolved oxygen data were directly examined to assess dissolved oxygen concentration as a candidate cause of impairment.

Figure D-7 shows dissolved oxygen concentrations at STORET stations along the Mainstem of the Groundhouse River from the 2005 sampling effort. Station S001-099 (red line) is located closest to 98SC005, the site focused on by Lane and Cormier (2004) and located within Impaired Area 1. Dissolved oxygen data at Impaired Area 1 follow the same trend as dissolved oxygen data at other locations in the watershed. However, the minimum dissolved oxygen concentrations at station S001-099 in July are lower than those of most other stations, which may be contributing to stress on the fish community.

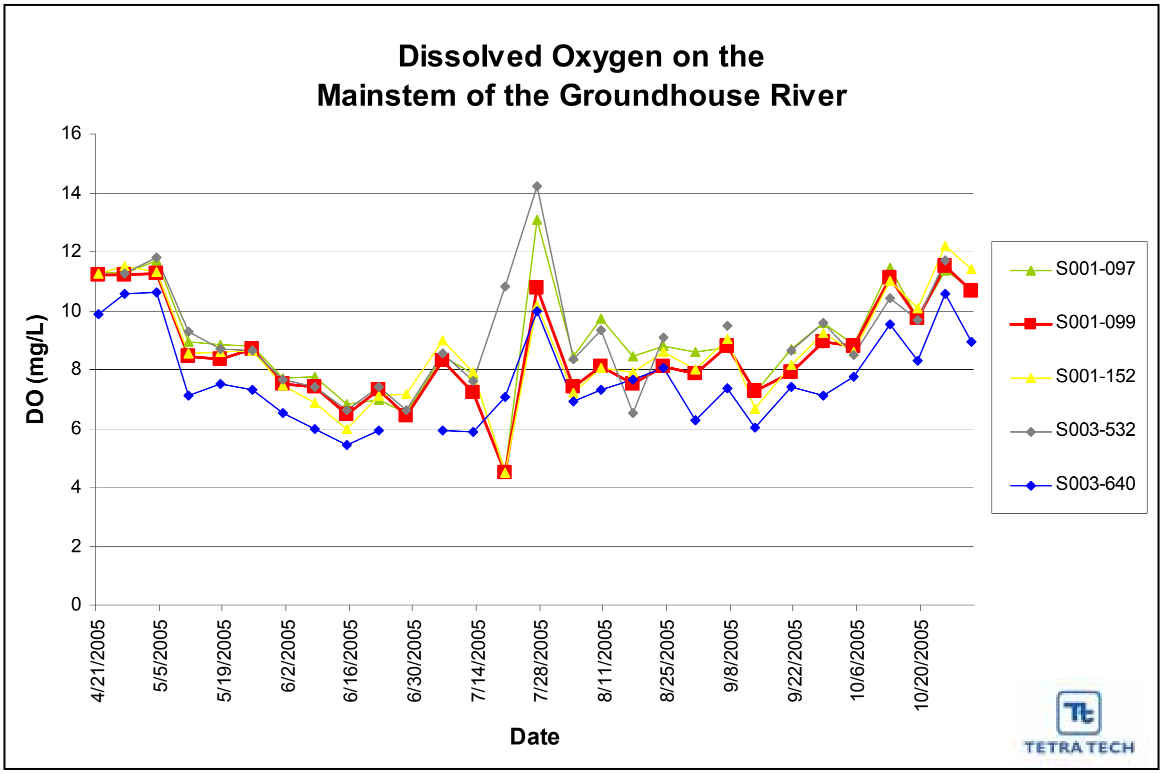


Figure D-7. Dissolved Oxygen Values Along the Mainstem of the Groundhouse River. Station S001-099 (in red) is closest to Impaired Area 1.

Figure D-8 shows STORET dissolved oxygen values at Ogilvie, MN inside Impaired Area 1 over the 2005 season. Measured values fell below the MPCA dissolved oxygen standard of 5 mg/l only once.

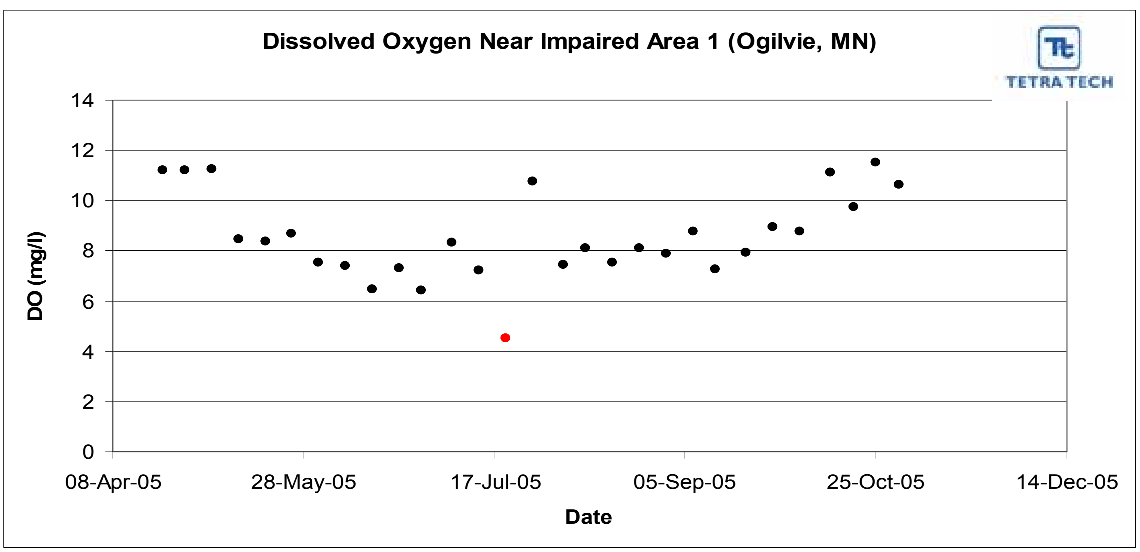


Figure D-8. Dissolved Oxygen Data for STORET station S001-099 through 2005.

Figure D-9 shows median and 95% confidence intervals of STORET dissolved oxygen data on the main stem of the Groundhouse River. An analysis of variance (ANOVA) indicated that there is no significant difference among dissolved oxygen values from all sites (df = 4, =0.95, p=0.08). Although the p-value is very close to indicating a significant difference, the p-value rises when data from STORET site S003-640 is removed. IBI scores collected around STORET site S003-640 do not indicate biological impairment.

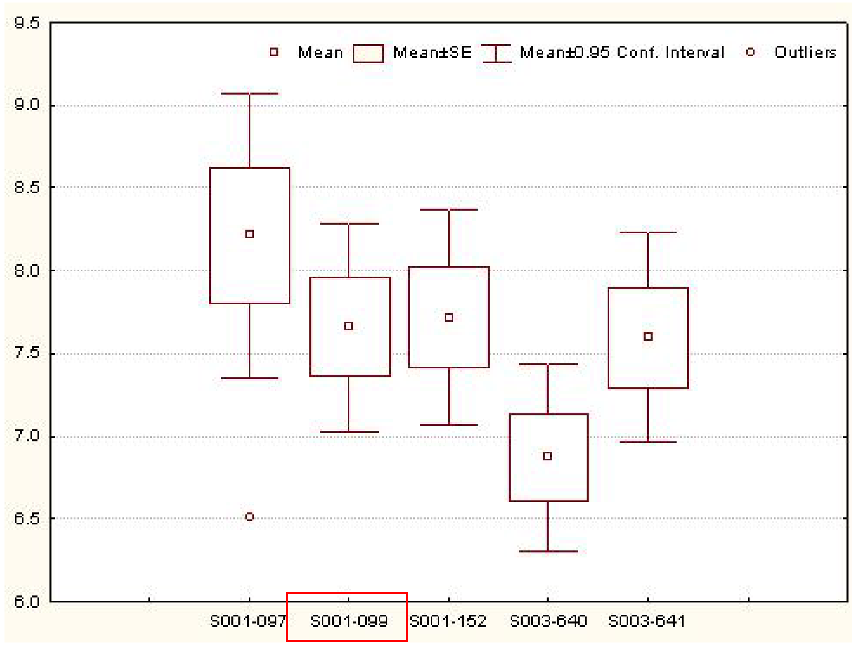


Figure D-9. Median values and 95% confidence intervals for dissolved oxygen concentrations at STORET sampling locations over 2005.

Dissolved oxygen flux is also important to consider. In areas with higher nutrient concentrations and associated increases in respiration and decomposition, dissolved oxygen values can change significantly over the course of a day (Allan, 1995). Late afternoon hours may see higher dissolved oxygen values associated with primary production fueled by sunlight, while non-daylight hours can have lower dissolved oxygen values as primary producers and animals respire. In addition, if the wastewater treatment plant in Ogilvie is discharging nutrient-rich water leading to increases in primary production, the decomposition of the primary producers could trigger periods of low dissolved oxygen. The plant may also be a potential source of biochemical oxygen demand, if it were not performing properly.

The only way to empirically measure diurnal changes in dissolved oxygen is with continuous data loggers; such data are not available in the Groundhouse watershed. Because of this, oxygen flux cannot be eliminated as a potential stressor on the stream biota. However, several pieces of information suggest that DO flux is not a major stressor of the fish community at Impaired Area 1. First, although a wastewater treatment plant does discharge water into Impaired Area 1, the worst impairment is observed *upstream* of the discharge point. And, although one might expect to see depressed dissolved oxygen concentrations leading to depressed IBI scores downstream of the wastewater discharge point, IBI scores rebound quickly after the site at Ogilvie. Second, most of the field measurements were collected in mid- to late morning when we would expect to see dissolved oxygen levels near the daily minimum.

Dissolved oxygen values inside Impaired Area 1 closely follow the pattern observed at other locations along the Groundhouse River main stem where biological impairment is not observed. Although there is a period during the summer where values fall away from the trend observed at other sites, only one concentration was observed below the dissolved oxygen impairment threshold. Additionally, an ANOVA analysis indicates there is no significant difference in dissolved oxygen concentrations between any of the sites on the Groundhouse River main stem. One of the recognized possible triggers for low dissolved oxygen, a wastewater treatment plant, discharges below the point with the most severe impairment. Based on the above analysis, it is unlikely that low dissolved oxygen is a significant stressor on the fish community in Impaired Area 1.

5.2.3 Candidate Cause 3: Excess Fine Sediment Leading to Habitat Loss

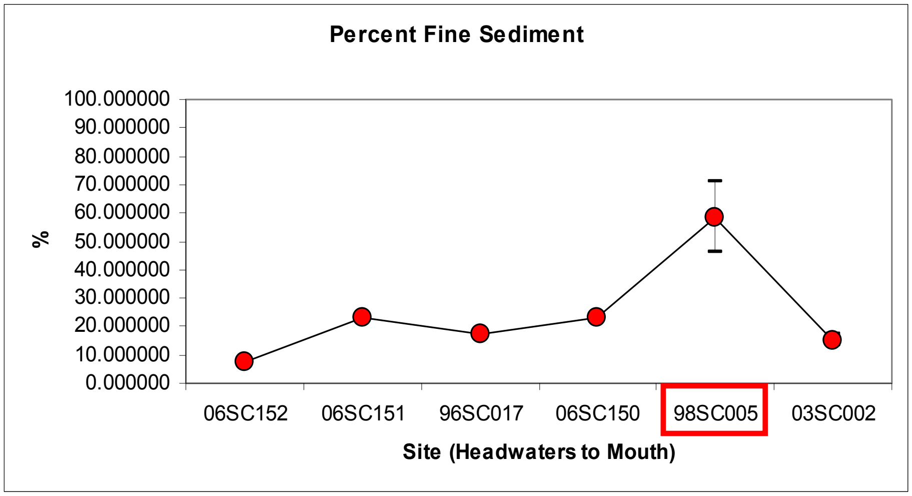
As part of the biological assessment, the MPCA collected habitat data at each site. Several of the parameters are direct measures of fine sediment (Anon, 2002b). The first parameter, *percent fine sediment*, calculates the percent of the transect points covered in fine sediment. This parameter provides an overall estimate of the extent of fine sediment. Second, *percent embeddedness* indicates how much of the coarse substrate is embedded in fine substrate. This parameter is important when examining the response of stream biota to habitat loss from the filling of interstitial spaces in coarse substrate with fine material. Third, *maximum depth of fine sediment* estimates the depth of the fine sediment, averaged from all of the transect points. Finally, both *turbidity* and *total suspended solids* (TSS) are collected to measure the amount of fine sediment plus organic material suspended in the water column.

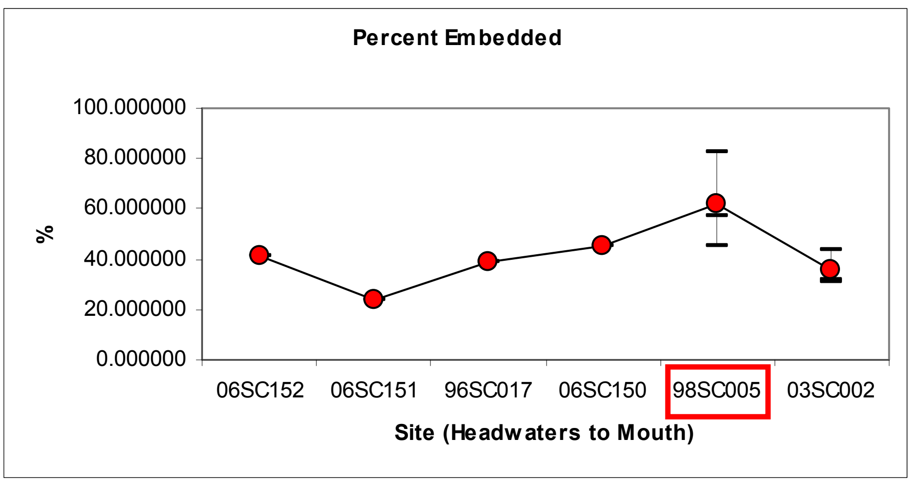
The most relevant parameter for this cause of impairment, percent fines <2mm, is not specifically collected as part of the water quality monitoring. However, TSS can provide some insight into the levels of sediment being transported through the system at the time of sampling (although it is recognized that TSS measurements do not capture the movement of bed load sediments). Table D-8summarizes the available TSS data for the Groundhouse monitoring locations and relevant ecoregion data. Sediment levels in the headwater segments are very close to or below the Northern Lakes and Forests statistics. The remaining stations generally show significantly lower TSS concentrations than those seen in the North Central Hardwood Forests data. Only S003-532 has a 95th percentile value which is slightly greater than those observed in minimally impacted areas. These results suggest that TSS is likely not a problem in the Groundhouse River as most locations exhibit concentrations near or better than reference conditions.

Table D-8. Total Suspended Solids (mg/L) Summary for 2005.

STATION_ID	Number of Samples	5th Percentile	Median	95th Percentile
North Fork Groundhouse River				
S003-641	18	0.93	2.50	9.00
S003-639 (unnamed tributary)	18	1.00	2.50	4.75
S003-640	18	1.00	2.00	4.60
S001-152	18	1.00	2.50	12.50
S001-099	18	1.00	2.00	6.75
S001-097	18	1.00	2.00	12.90
South Fork Groundhouse River				
S003-664	12	2.55	4.00	9.15
S003-638	18	1.85	3.00	9.10
Groundhouse River (Mainstem)				
S003-532	13	2.00	3.00	20.80
Minnesota Minimally Impacted Streams (1986-1992)				
Northern Lakes and Forests	-	0.8	7.8	8.2
North Central Hardwood Forests	-	1.4	7.7	20.0

Figure D-10 shows each of the measured fine sediment parameters over the geographic gradient on the main stem of the Groundhouse River. The site adjacent to the town of Ogilvie, MN shows higher values in each of the categories. These data indicate that, compared to other sites on the Groundhouse River, the site near Ogilvie has a higher percent of fine sediment covering the streambed, coarse substrate that is more embedded with fine sediment, and deeper depths of fine sediment.





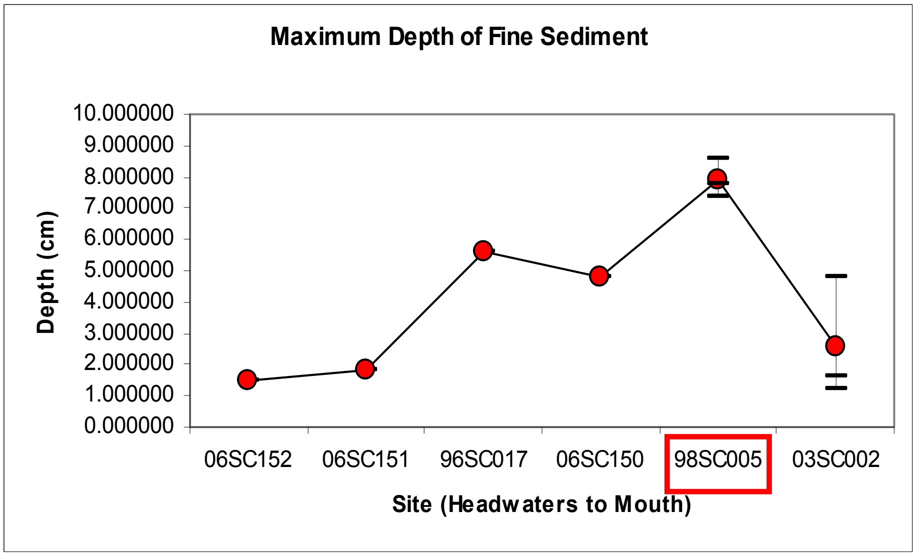


Figure D-10. Fine Sediment Parameter Values along a geographic gradient of the Mainstem of the Groundhouse River, MN. Red dots indicate average values and bars indicate the measured range. Site 98SC005 (outlined in red) is adjacent to the town of Ogilvie, MN.

Two metric scores, percent gravel spawners and number of invertivores, used by the MPCA to compute IBI scores can also be used evaluate the biological response to increased fine sediment. This is done by evaluating changes in the types of functional feeding and reproductive groups. Figure D-11 shows metric scores for *percent gravel spawners* along the main stem of the Groundhouse River. Gravel spawners require clean, coarse gravel to spawn. This metric should be inversely related to excess bedded fine sediment (Niemela and Feist, 2000; Berkman and Rabeni, 1987). Metric scores for the site adjacent Ogilvie (inside Impaired Area 1) are the lowest scores found along the main stem. This indicates that the "gravel spawners" as a reproductive group are responding to a stressor in the area near Ogilive, most likely an increase in fine bend sediment.

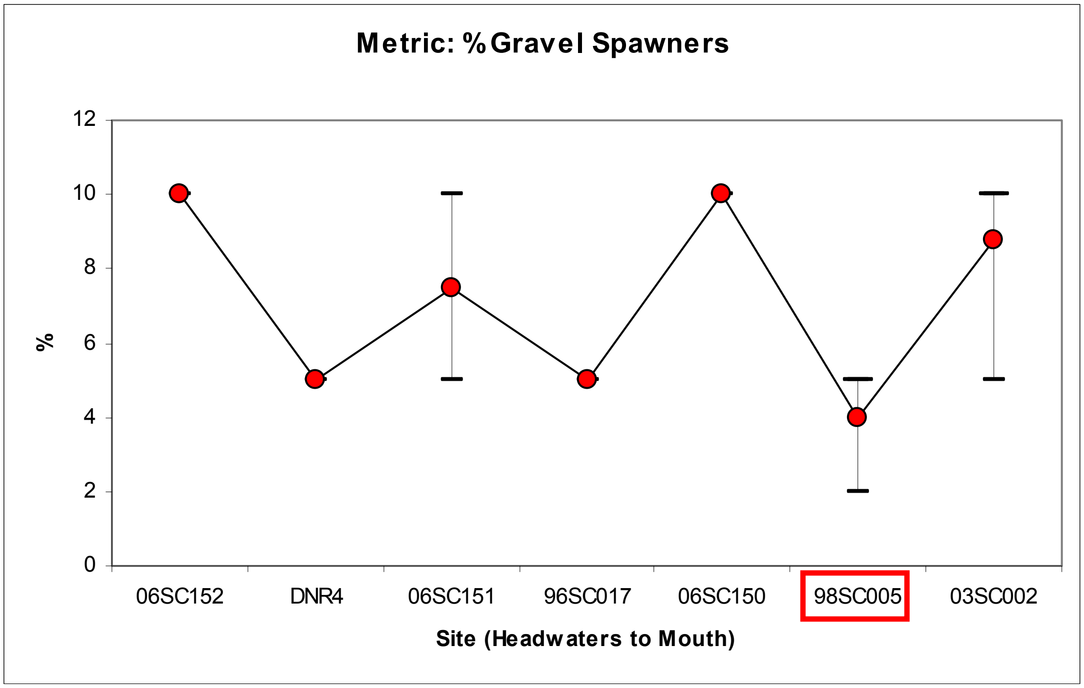


Figure D-11. Metric scores for percent gravel spawners on the Mainstem of the Groundhouse River, MN. Red dots indicate mean scores and bars indicate the measured range. Site 95SC005 (in red) is adjacent to Ogilvie, MN.

Figure D-12 shows metric scores for "*number of benthic insectivores*" along the main stem of the Groundhouse River. Benthic insectivores require undisturbed stream bed habitats to feed and reproduce. Many of the species in this metric require diverse benthic habitats including coarse substrate and woody debris (Neimela and Feist, 2000). Again, metric scores near Ogilvie are consistently lower than scores found at other sites along the main stem, which suggests that benthic habitats near Ogilvie have been disturbed.

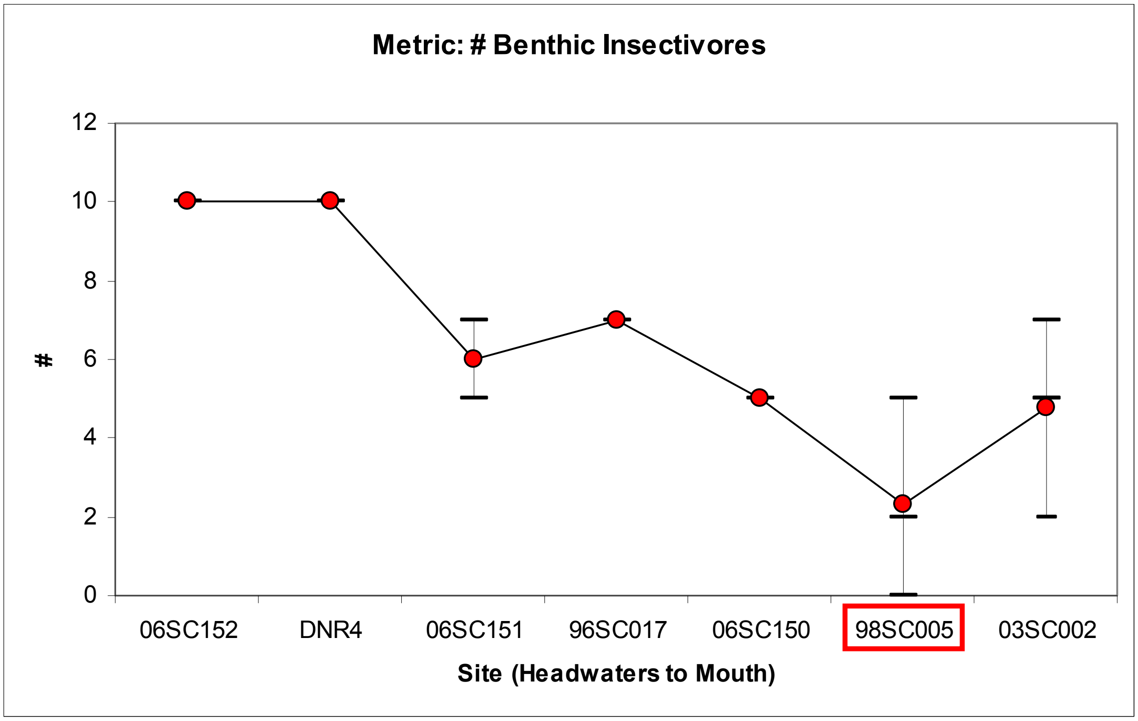


Figure D-12. Metric scores for the number of benthic insectivores on the Mainstem of the Groundhouse River, MN. Red dots indicate mean scores and bars indicate the measured range. Site 95SC005 (in red) is adjacent to Ogilvie, MN.

Anthropogenic sources of fine sediment exist in Impairment Area 1. Most notably, livestock operations located adjacent to and upstream of the impaired sampling locations allow cattle direct access to the stream, exacerbating stream bank instability and the potential for erosion due to flowpath alteration. It is also important to consider the natural features of the landscape in Impaired Area 1 that may contribute to fine sediment deposition and retention.

First, according to MPCA data, Impaired Area 1 has a lower stream slope than most other areas of the watershed. Stream reaches with lower slope may have a lower capacity to transport fine sediment because of reduced stream power. This means that even without a supply of excess sediment coming from upstream, Impaired Area 1 may be natural depositional zone for fine sediment. Figure D-13 shows a decrease in stream slope in the main stem of the Groundhouse near Ogilvie that is accompanied by an increase in the percent of bedded fine sediment. This pattern is not seen upstream of Impaired Area 1 (the sampling locations to the right of the Ogilvie site in Figure D-13) but does continue downstream.

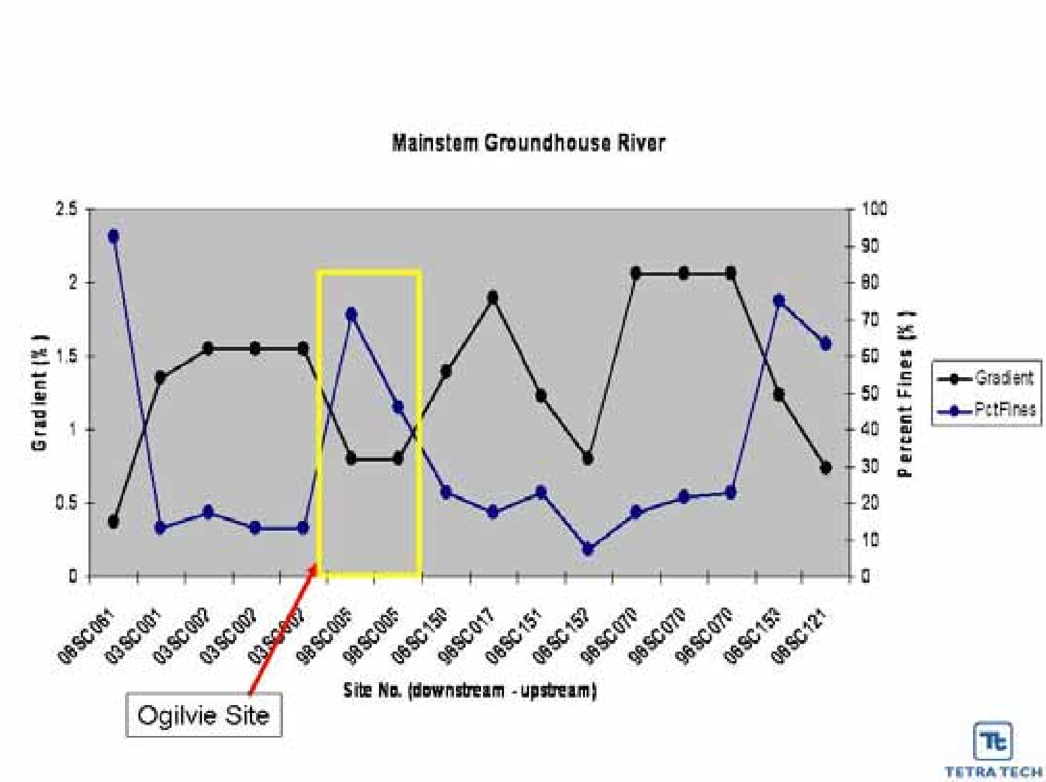


Figure D-13. Stream slope and percent fine sediment in the Mainstem of the Groundhouse River. Some sampling locations are included twice to include two sets of measurements.

Second, soil types in Impaired Area 1 are more easily eroded than other soil types in the watershed. Figure D-14 shows the soil erodibility factors in the Groundhouse River Watershed. Higher erodibility values in Impaired Area 1 indicate that this area is at a higher risk for erosion.

It is important to note that other non-impaired sites also fall within the area with the higher erodibility factor. However, the combinations of anthropogenic stressors that exist in Impaired Area 1, including livestock operations, do not exist in other areas with the high erodibility factor. In addition, stream slope values at other sites are higher, enabling the stream to transport any excess sediment downstream. It is likely that although the natural conditions might favor excess sediment, nearby anthropogenic disturbances have triggered a more significant sediment problem.

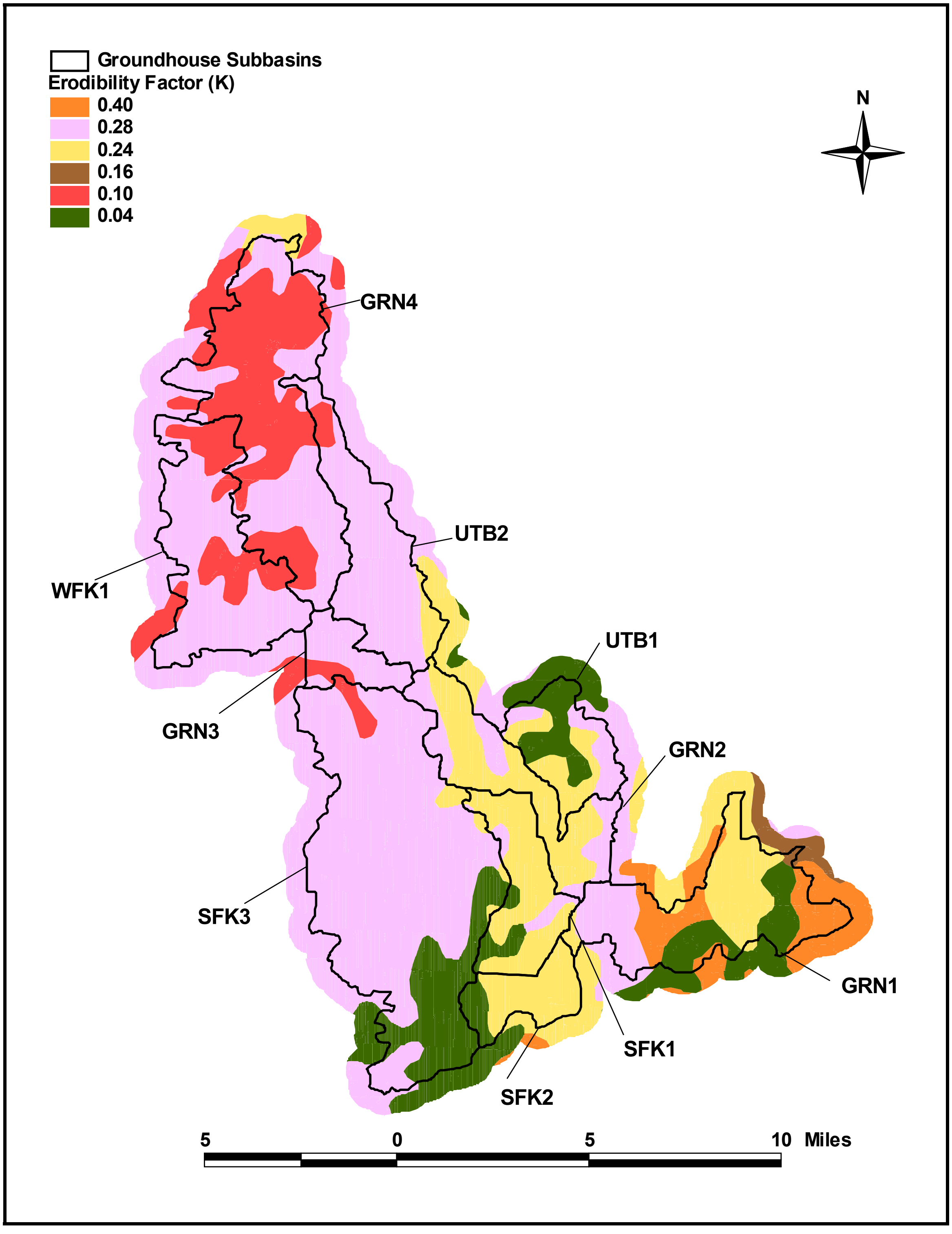


Figure D-14. Soil Erodibility Factor (K) for Soils in the Groundhouse Watershed

5.3 Impaired Area 2 – Headwaters of the South Fork of the Groundhouse River

Fish community data have been collected at nine sites along the South Fork of the Groundhouse River (Table D-1). Only one of the sites, 06SC045 or Impaired Area 2, falls below the impairment threshold (Figure D-15); IBI scores at the remaining eight sites indicate relatively healthy fish communities, with a mean IBI score of 83.

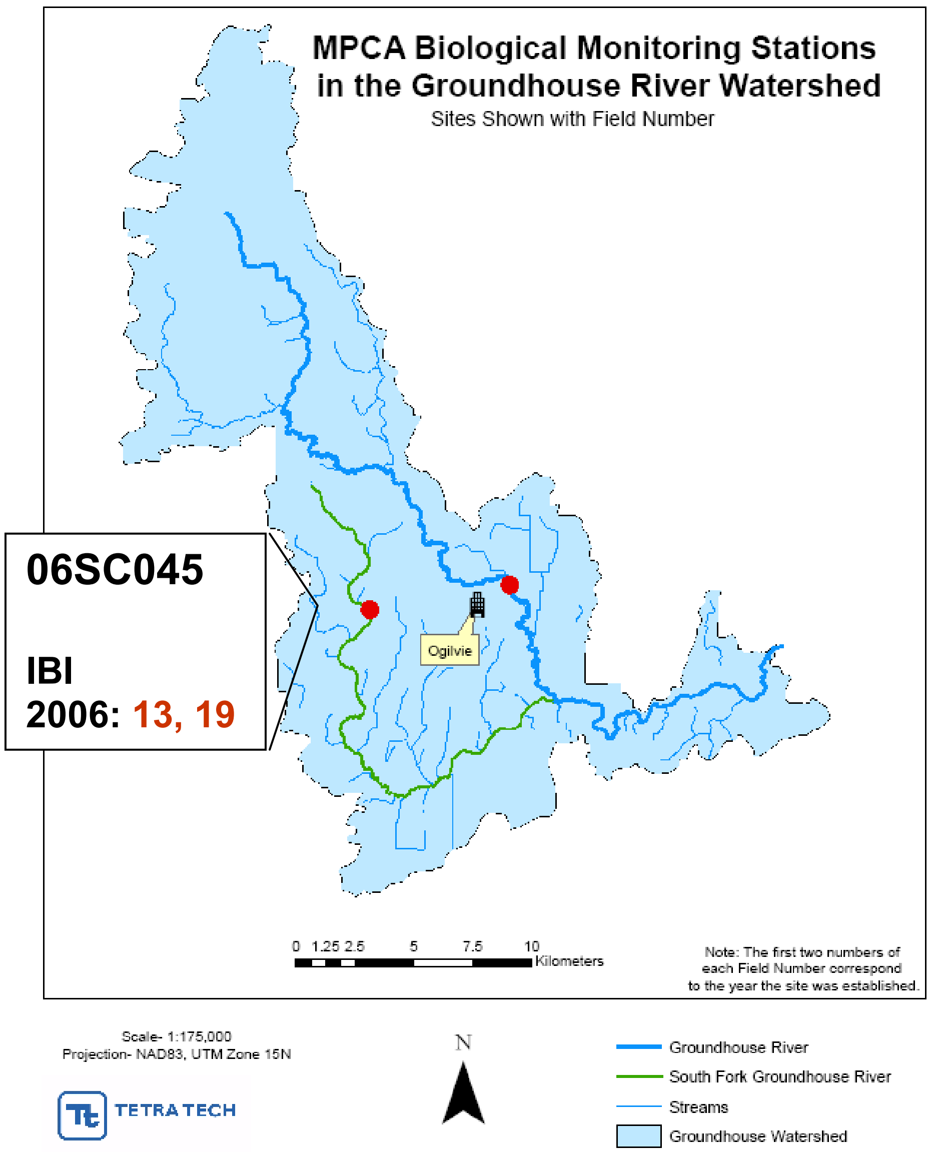


Figure D-15. Site 06SC045 is the location identified as impaired on the South Fork of the Groundhouse River. The site was sampled twice in 2006 with IBI scores of 13 and 19.

Impaired Area 2 is the most upstream site assessed on the South Fork of the Groundhouse. Prior to 2006, the closest site with STORET water quality data available is located more than five miles downstream.

Impaired Area 2 was sampled only in 2006. Because of data limitations, it is not possible to examine water quality or IBI scores over time as was possible with Impaired Area 1. The site has the second smallest drainage area (5.8 square miles) of locations sampled in the watershed.

Impaired Area 2 is also a narrow and shallow reach of the South Fork. This is probably attributable to its proximity to the headwaters. Figure D-16 shows the mean stream width and mean stream depth along the South Fork of the Groundhouse.

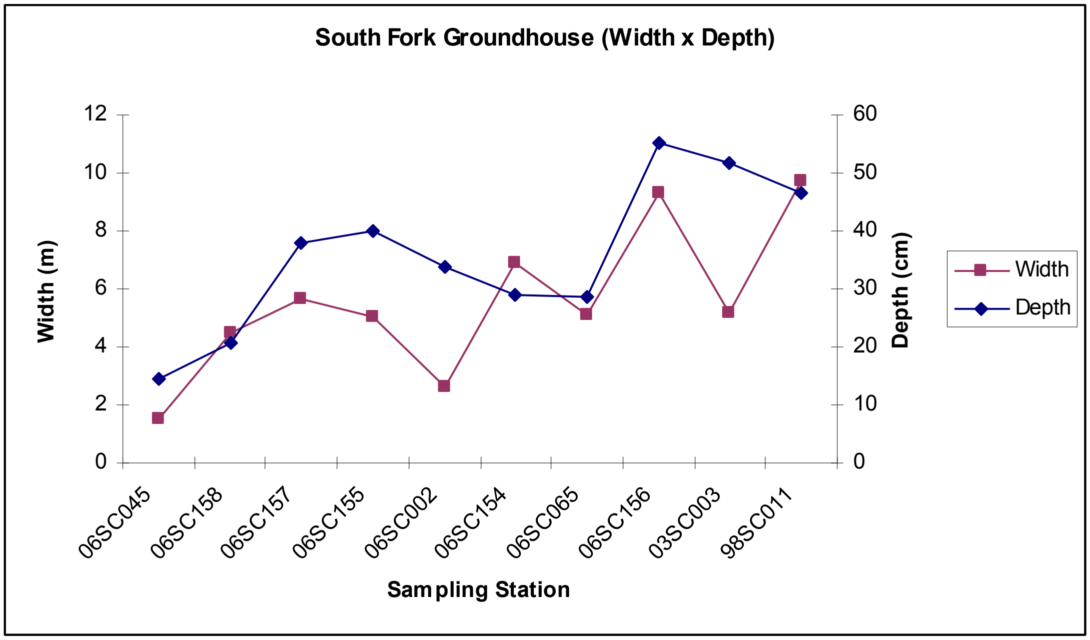


Figure D-16. Mean stream width (m) and mean stream depth (cm) from upstream to downstream along the South Fork of the Groundhouse River. Impaired Area 2 is the most upstream sampling location (06SC045).

According to the MPCA, the variability in IBI scores in minimally disturbed sites throughout the St. Croix River Basin suggests that not all stream ecosystems can be characterized by the same set of metrics (Niemela and Fiest, 2000). The MPCA provides the West Branch of the Kettle River as an example. "This site was essentially a low gradient glide/pool system that flowed through a large wetland. The dissolved oxygen concentration in the middle of the afternoon was <3mg/l," which is most likely attributable to naturally occurring site conditions. The Kettle River Watershed was described as having less than 15 percent disturbance. Disturbance is higher in the South Fork Groundhouse River watershed. However, Impaired Area 2 is a very small headwater stream draining a small area with similar levels of dissolved oxygen (3.64 mg/L) and similar IBI scores. It is possible that the IBI metrics do not well represent the health of the fish community at this site.

5.3.1 Candidate Cause 1: Altered Temperature Regime

Although IBI scores at Impaired Area 2 indicate disturbance, neither the IBI score itself or any of the individual metrics respond solely to changes in temperature. Therefore, temperature data are directly examined to assess changes in the temperature regime as a candidate cause of impairment.

The temperature data available for Impaired Area 2 are limited with only two sampling events available for analysis. Figure D-17 shows the temperature taken at the stations assessed for fish in 2006. Although there are insufficient data to establish long term trends, the water temperature at Impaired Area 2 is relatively consistent with the water temperature at other sampling stations in the South Fork. The temperature is slightly elevated in contrast to the other sampling stations in mid-July. Without a larger data set, however, it is not possible to attribute this to site or weather conditions.

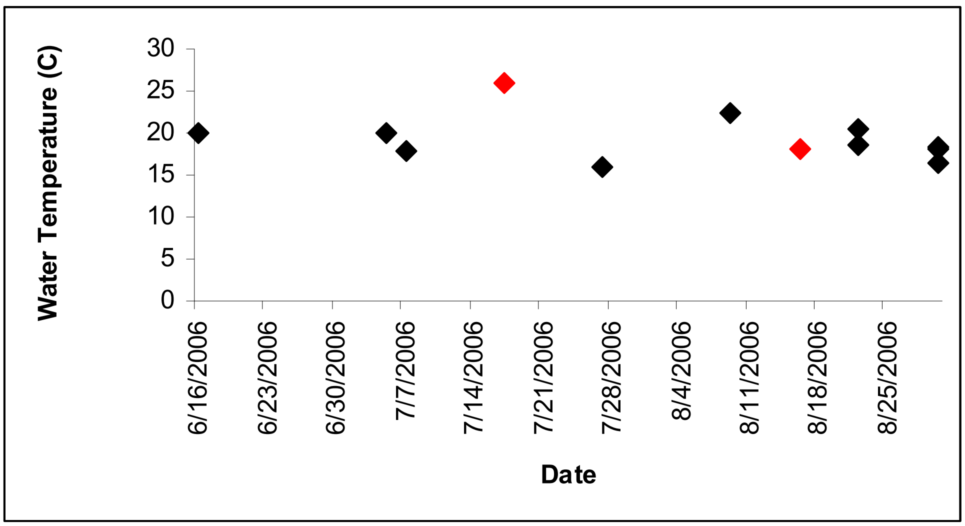


Figure D-17. Water Temperature from June through August 2006 along the South Fork of the Groundhouse River. Red dots represent the temperature at Impaired Area 2 (06SC045).

Although the temperature data are limited for Impaired Area 2 and other sites along the South Fork of the Groundhouse, available data do not suggest significantly high temperatures at Impaired Area 2. Without a more detailed analysis of temperature trends over time it is not possible to discount changes in temperature regime as a biological stressor. However, the available data suggest that changes in the local temperature regime are likely not responsible for the low fish IBI scores in Impaired Area 2.

5.3.2 Candidate Cause 2: Low Dissolved Oxygen

Although IBI scores at Impaired Area 2 indicate disturbance, neither the IBI score itself nor any of the individual metrics respond solely to changes in dissolved oxygen. Therefore, the dissolved oxygen data are directly examined to assess low dissolved oxygen levels as a candidate cause of impairment.

Throughout the Groundhouse River watershed, dissolved oxygen levels are relatively high (Table D-9). The mean of all data including all available STORET measurements (n=244) is 7.29 mg/L. The two

STORET sampling locations on the South Fork of the Groundhouse River are located more than five miles downstream of Impaired Area 2.

Table D-9. Median dissolved oxygen levels throughout the Groundhouse Watershed in mg/L.

STATION_ID	Number of Samples	5th Percentile	Median	95th Percentile
North Fork Groundhouse River				
S003-641	29	6.13	8.52	12.17
S003-639 (unnamed tributary)	29	6.08	8.29	12.10
S003-640	28	5.88	7.36	10.56
S001-152	29	6.24	8.57	11.48
S001-099	29	6.44	8.32	11.24
S001-097	29	6.63	8.80	11.60
South Fork Groundhouse River				
S003-664	17	6.53	7.69	11.45
S003-638	29	5.81	8.21	11.41
Groundhouse River (Mainstem)				
S003-532	25	6.61	8.71	11.78

Research by McCormick and Liang (2003) showed dissolved oxygen reduction in response to phosphorus additions in subtropical wetlands. Although the Groundhouse River watershed is not subtropical, dissolved oxygen levels in Impaired Area 2 may be responding similarly to anthropogenic nutrient loading. Generally, phosphorus levels in the watershed are similar to reference values established for Minnesota Minimally Impacted Streams (Table D-10).

Table D-10. Phosphorus concentration summary in mg/L.

STATION_ID	Number of Samples	5th Percentile	Median	95th Percentile
North Fork Groundhouse River				
S001-097	23	0.042	0.064	0.176
South Fork Groundhouse River				
S003-638	23	0.055	0.087	0.173
Minnesota Minimally Impacted Streams (1986-1992)				
North Central Hardwood Forests	-	0.010	0.070	0.130

The mean phosphorus concentration was 0.121 mg/L. The phosphorus concentrations observed in Impaired Area 2 were 0.273 mg/L and 0.212 mg/L, approximately double the watershed average. These values represent the second and third highest values observed. Figure D-18 shows both phosphorus concentration and IBI score along the South Fork of the Groundhouse. There is not a consistent trend between phosphorus level and IBI score. Total phosphorus concentration is high at Impaired Area 2, but it is also high at sampling station 06SC065, which exhibited the highest IBI score in the South Fork watershed.

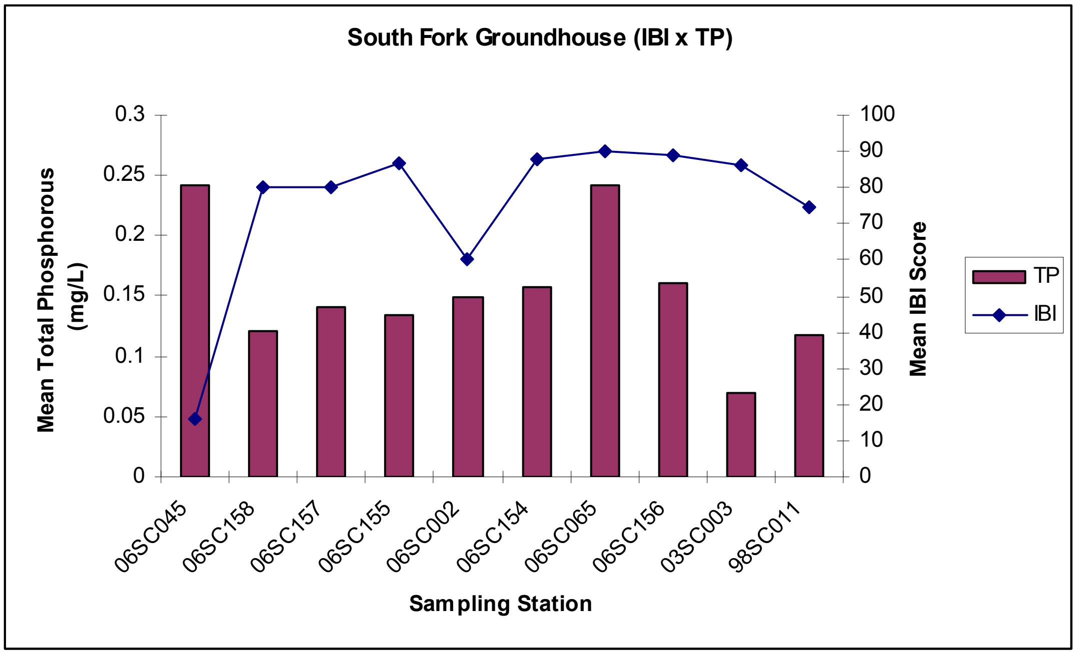


Figure D-18. Summary of total phosphorus and IBI score from upstream to downstream along the South Fork of the Groundhouse River. Impaired Area 2 is the most upstream sampling location (06SC045).

It is possible that eutrophication driven by excess phosphorus is reducing available dissolved oxygen. It is also possible that low dissolved oxygen is a stressor on the fish community in Impaired Area 2. However, based on the inconsistency between nutrient and dissolved oxygen levels at this site and others along the South Fork of the Groundhouse it may be more likely that the dissolved oxygen levels and depressed IBI scores are attributable to the wetland characteristics of Impaired Area 2.

5.3.3 Candidate Cause 3: Excess Fine Sediment Leading to Habitat Loss

Many of the sites on the South Fork of the Groundhouse River have relatively high levels of fine sediments. Impaired Area 2 does not have a level of fine sediment inconsistent with the rest of the South Fork Watershed (Figure D-19).

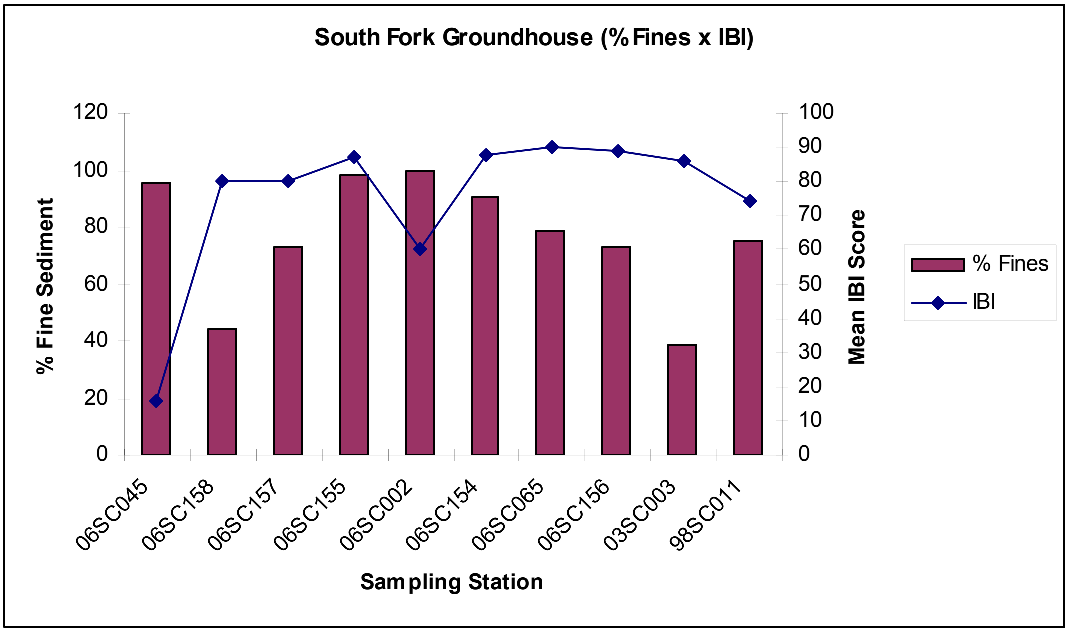


Figure D-19. Percent fine sediment at sampling locations along the South Fork of the Groundhouse River. The sites progress from upstream to downstream.

The mean percent of fine sediment in the South Fork Watershed is 70 percent. The mean percent fine sediment in Impaired Area 2 is 85 percent. Other sampling locations along the South Fork of the Groundhouse have levels of fine sediment equivalent to or higher than the percent of fine sediment observed at Impaired Area 2. However, IBI scores are depressed at the two sampling stations with the highest percent of fine sediment. Because of the number of sites with considerably higher IBI scores and similar levels of fine sediments, it is not likely that the presence of fine sediments is the only cause for impairment on the in the South Fork Watershed, but it does appear to be partially contributing to the impairment.

6 Macroinvertebrates

This section of the report discusses the available macroinvertebrate data in more detail and identifies the most likely priority stressors.

6.1 Extent of Impairment

Macroinvertebrate scores are depressed throughout the Groundhouse River watershed. Unlike the fish community, it is not possible to isolate particular areas of the watershed for analysis or restoration efforts and therefore this analysis was conducted for the entire watershed.

6.1.1 Stressor and Response Variables

Biological community conditions respond to stressors in the aquatic environment. Several physical habitat and water quality variables were qualitatively compared with biological response at each sampling location and are summarized below. Most of the data were from the stream assessments conducted in the summer 2006. Scoring for each stressor variable (e.g., physical habitat and water quality) was based on the stressor/biological response assumptions listed in Table D-11. Evaluation of individual biometrics used in the MIBI as well as sensitive orders of benthic macroinvertebrates (e.g., Ephemeroptera, Plecoptera, and Trichopera) were used to determine biological condition at a site. In addition, where multiple years of data were available it was possible to determine if any short-term patterns in biological condition (e.g., decline or improvement) occurred. These interpretations of biological data served as the basis for conclusions in Table D-12and Table D-13.

Table D-11. Expected relationships between stressors and biotic (benthic macroinvertebrate) communities.

Stressor	Stressor Condition	Biological Condition	Relationship Type
Gradient (%)	Decreases	Declines	Direct
% Fines	Increases	Declines	Inverse
Ammonia	Increases	Declines	Inverse
TSS	Increases	Declines	Inverse
TP	Increases	Declines	Inverse
TN	Increases	Declines	Inverse

Scoring interpretations in the following tables are as follows:

+ denotes that biological conditions were not impaired and responded to decreasing stress
- denotes that biological conditions were impaired and responded to increasing stress
+/- denotes that biological response opposite to that expected with increasing/decreasing stress

Table D-12. Sampling sites on the South Fork Groundhouse River and how biological communities (benthic macroinvertebrates) responded to physical habitat and water quality stressors.

Site	Year	Gradient	% Fines	NH_4	TSS	TP	TN
98SC011	1998	-	-	-	+/-	-	-
98SC011	2006						
98SC011	2006	-	-				
03SC003	2003	-	+				
03SC003	2005	+/-	+	+/-	+	+	+
03SC003	2006						
06SC156	2006						
06SC065	2006						
06SC065	2006	-	-	+/-	-	-	+/-
06SC154	2006	-	+/-	+	+	+	+
06SC155	2006	+/-	+/-	+	+	+	+
06SC156	2006	-	-	-	+/-	-	+/-
06SC158	2006	+/-	+	+	+/-	+	+
06SC045	2006						
06SC045	2006	+/-	-	-	+/-	-	+/-

Table D-13. Sampling sites on the Mainstem Groundhouse River and how biological communities (benthic macroinvertebrates) responded to physical habitat and water quality stressors.

Site	Year	Gradient	% Fines	NH_4	TSS	TP	TN
06SC061	2006	+/-	+/-	+/-	+	+/-	+
03SC001	2003	+	+	+/-	+	+/-	+/-
03SC002	2003	+	+	-	-	-	-
03SC002	2005						
03SC002	2006	+/-	+/-	-	-	+/-	-
98SC005	1998	-	-	+/-	-	-	+/-
98SC005	2003	-	-	-	-	-	+/-
06SC150	2006	+/-	+/-	-	+/-	+/-	+/-
96SC017	1996						
06SC151	2006	-	+/-	-	+/-	+/-	+/-
06SC152	2006	-	+/-	-	-	+/-	+/-
96SC070	1996						
96SC070	2005						
96SC070	2006	+	+	+/-	+	+	+/-
06SC153	2006	-	-	-	-	-	+/-
06SC121	2006						

6.1.2 Assemblage-specific Responses

It is possible that the response to a variety of stressors will be different by fish and macroinvertebrates. Sites along each of the sub-drainages (e.g., South Fork Groundhouse River and main stem Groundhouse River) where benthic invertebrate community assessments differed from fish community assessments are identified below in Table D-14and Table D-15. These differences in response to site conditions may be related to specific stressors.

Table D-14. Identification of divergent response between benthic invertebrate and fish assessments on the South Fork Groundhouse River including identification of stressors impairing biological communities.

Site No.	Benthic Invertebrate Response	Stressor
06SC045	Negative (fish and invertebrates)	% Fines, NH_4, TP
06SC156	Negative	% Fines, NH_4, TP
06SC065	Negative	% Fines, TSS, TP
03SC003 (2003)	Negative	Gradient
98SC011 (2006)	Negative	% Fines
98SC011 (1998)	Negative	% Fines, NH_4, TP, TN

Note: Low Gradient was determined to be a factor that corresponded with increased % Fines at each site.

Table D-15. Identification of divergent response between benthic invertebrate and fish assessments on the Mainstem Groundhouse River including identification of stressors impairing biological communities.

Site No.	Benthic Invertebrate Response	Stressor
06SC153	Negative	Gradient, % Fines, NH_4, TP, TN
06SC152	Negative	Gradient, NH_4, TSS
06SC151	Negative	Gradient, NH_4
06SC150	Negative	NH_4
98SC005 (2003)	Negative (fish and invertebrates)	Gradient, % Fines, NH4, TSS, TP
98SC005 (1998)	Negative (fish and invertebrates)	Gradient, % Fines, TSS, TP
03SC002 (2006)	Negative	NH_4, TSS, TN
03SC002 (2003)	Negative	NH_4, TSS, TP, TN

Based on the widespread impairment of the benthic invertebrate communities, it appears that the invertebrate community is being impacted by habitat conditions to a greater extent than the fish community. The data indicate that this response is due to increases in fine sediment accompanied by decreases in channel gradient. The decrease in channel gradient reduces the energy of the river system in these areas, limiting the transport of fine sediments and facilitating sediment deposition. The data also indicate a response to ammonia concentration despite the fact that ammonia concentrations throughout the watershed are generally low, below MPCA standards, and close to those concentrations measured in comparable Minimally Impacted Streams (see *Water Quality and Fecal Coliform Evaluation*). The benthic invertebrate communities may be responding to nutrient concentrations, the amount of fine sediment, or a coupling of the two.

6.1.3 Indicator Taxa Response (Genus/species)

Taxa lists are commonly reviewed for occurrence of tolerant species and for dominance patterns among the species present at a site. Species that have higher than usual dominance compared to others on the list are sometimes referred to as "bloom taxa." This means that some stimulus is promoting opportunistic feeding behavior or promoting optimal living conditions for a particular species. Such stimulus could include water quality elements and/or the habitat available in the water column (e.g., predatory avoidance, water column feeding, etc.). Lists of taxa from sites identified in Table D-16(South Fork Groundhouse River) and in Table D-17(Mainstem Groundhouse River) were constructed in order to identify the habitat tolerances of species present in the Groundhouse River Watershed and to evaluate whether these taxa indicate certain types of impairment based on life histories and habitat requirements (e.g., tolerance to low dissolved oxygen and high surface water temperatures).

Tolerance levels for individual taxa were determined using the designations reported by MPCA scientists. Tolerance to stressors was based on designations reported in Hilsenhoff (1987). These tolerance values are assigned to taxa on a scale of 1 to 10 (ten being the most tolerant to stressors). Originally, taxa tolerance designations reported by Hilsenhoff were used as indicators of the severity of organic pollution in Midwestern streams. This tolerance scale for individual taxa has since been adapted to reflect a variety of stressors throughout the United States including inorganic pollutants (e.g., sediment), pesticides, and metals. No intolerant species were observed in the South Fork Watershed during any year in which data were collected. All of the species observed in Impaired Area 2 were tolerant species. Intolerant and facultative species are found throughout the main stem Groundhouse River Watershed.

Mainstem Groundhouse River Taxa

A greater diversity of species was identified on the main stem Groundhouse River taxa list than that for the South Fork Groundhouse River. Species richness for benthic macroinvertebrate communities is related to complexity of habitats and to characteristics of larger channels. The number of tolerant chironomid taxa was higher at sites where water quality changes were previously identified (e.g., *Dicrotendipes*, *Ablabesmyia*, *Paratendipes*, *Cricotopus bicinctus*). Filter feeding was the dominant feeding strategy of the species in the main stem, as indicated by *Ceratopsyche* and *Hydropsyche* caddisflies. Filter feeders dominate hard-bottomed sediments with increased volume of suspended organic matter. The addition of these dominant taxa in the main stem indicates the presence of suspended sediments and, therefore, suggests that there must be sources for introduction or re-suspension of organic material. Some of the upper drainage sites on the Mainstem Groundhouse River (*e.g.*, 06SC150, 06SC151, 06SC152, 06SC153) supported taxa found in stillwater environments like the mayfly *Caenis* and *Eurylophella*.

The lowermost site on the Mainstem Groundhouse River (06SC061) had one of the highest biological condition scores despite trends at other sites showing biological impairment with high nutrient levels and percent fines. Site 06SC061 is a larger stream channel with a complex riparian structure and instream substrate structure that provides a variety of physical refugia (Figure D-20). The cumulative impacts from pollutants on the biological community in large rivers do not appear to be as severe as individual effects from each pollutant on biological communities in smaller streams. Biota appear to be able to escape pollution impacts at the lowermost portion of the Groundhouse River (site 06SC061) despite the influence of these stressors in other portions of this drainage.

Figure D-20. Lower Mainstem of the Groundhouse River near site 06SC061 showing riparian condition and instream structure.

South Fork Groundhouse River Taxa

The number of taxa identified in the South Fork Groundhouse River was less than in the main stem. Many of the taxa collected from the upper reach of the South Fork Groundhouse River are tolerant to elevated surface water temperatures and represent top predators in the food chain (e.g., *Enallagma*; dragonfly nymph and Corixidae). Mollusks (e.g., *Ferrissia* and *Amnicola*) were identified at many of the sites. The presence of mollusks is often indicative of substantial water chemistry changes. The mollusks and the dominant caddisfly (e.g., *Helicopsyche*) feed on attached algae (periphyton) found on larger and stable substrates. Remaining dominant taxa were "burrowers" and consumers of plant organic material (e.g., Oligochaeta, *Polypedilum*, and *Dicrotendipes*) indicating presence of fine sediments that served as primary habitat. Dominant taxa identified in this sub-drainage were generally more tolerant to environmental stressors than those identified in the main stem.

Table D-16. Dominant taxa from sites in the South Fork Groundhouse and level of tolerance to environmental stressors.

Site No.	Taxa List	Tolerance Level
06SC045	*Enallagma* *Hyalella azteca* *Amnicola* Oligochaeta	Tolerant Tolerant Tolerant Tolerant
06SC156	*Procloeon* *Polypedilum* *Hyalella azteca* *Ferrissia* Oligochaeta	Facultative Facultative Tolerant Facultative Tolerant
06SC065	*Polypedilum* *Dubiraphia* *Enallagma* *Caenis* *Hyalella azteca* *Amnicola*	Facultative Facultative Tolerant Tolerant Tolerant Tolerant
03SC003 (2006)	*Polypedilum* *Enallagma* *Hyalella azteca* Oligochaeta	Facultative Tolerant Tolerant Tolerant
98SC011 (2006)	Oligochaeta *Hyalella azteca* Corixidae *Sigara* (Corixidae) *Polypedilum*	Tolerant Tolerant Tolerant Tolerant Facultative
98SC011 (1998)	Oligochaeta *Amnicola* *Hyalella azteca* *Dubiraphia* *Helicopsyche* *Dicrotendipes*	Tolerant Tolerant Tolerant Facultative Facultative Tolerant

Note: Tolerant = able to survive in polluted conditions
Facultative = able to survive in moderately polluted conditions
Intolerant = requires unpolluted conditions in order to survive

Table D-17. Dominant taxa from sites in the Mainstem Groundhouse River and level of tolerance to environmental stressors.

Site No.	Taxa List	Tolerance Level
06SC153	*Caenis*	Tolerant
	Gyraulus	Tolerant
	Amnicola	Tolerant
	Oligochaeta	Tolerant
06SC152	*Tanytarsus*	Facultative
	Dicrotendipes	Tolerant
	Ablabesmyia	Tolerant
	Triaenodes	Facultative
	Caenis	Tolerant
	Pisiidae	Facultative
	Physa	Tolerant
	Ferrissia	Facultative
	Amnicola	Tolerant
06SC151	*Tanytarsus*	Facultative
	Helicopsyche	Facultative
	Stenelmis	Facultative
	Caenis	Tolerant
	Stenacron	Tolerant
	Physa	Tolerant
	Amnicola	Tolerant
06SC150	*Tanytarsus*	Facultative
	Paratendipes	Tolerant
	Dicrotendipes	Tolerant
	Helicopsyche	Facultative
	Dubiraphia	Facultative
	Coenagrionidae	Tolerant
	Caenis	Tolerant
	Eurylophella	Facultative
	Acari (water mites)	Tolerant
	Physa	Tolerant
98SC005 (2003)	*Physa*	Tolerant
	Pisiidae	Facultative
	Macronychus	Facultative
	Hydropsychidae	Facultative
	Hydropsyche	Facultative
	Ceratopsyche	Intolerant
	Cricotopus bicinctus	Tolerant
	Rheotanytarsus	Facultative
98SC005 (1998)	*Amnicola*	Tolerant
	Paraleptophlebia	Intolerant
	Dubiraphia	Facultative
	Helicopsyche	Facultative
	Dicrotendipes	Tolerant
	Microtendipes	Facultative
	Polypedilum	Facultative
03SC002 (2006)	*Atherix*	Intolerant
	Rheotanytarsus	Facultative
	Helicopsyche	Facultative
	Micrasema	Intolerant
	Pisiidae	Facultative
03SC002 (2003)	Oligochaeta	Tolerant
	Baetis	Facultative
	Hydropsychidae	Facultative
	Ceratopsyche	Intolerant
	Simulium	Facultative
	Cricotopus bicinctus	Tolerant
	Rheocricotopus	Facultative
	Polypedilum	Facultative
	Rheotanytarsus	Facultative

6.2 Candidate Causes of Biological Impairment

In this section, the same candidate causes of biological impairment will be addressed as were addressed for the impairments associated with the fish community. However, because there is a correlation between the benthic macroinvertebrate community and nutrient levels including ammonia and phosphorus, nutrient loading will be considered as a candidate cause for impairment in conjunction with low dissolved oxygen levels.

6.2.1 Candidate Cause 1: Altered Temperature Regime

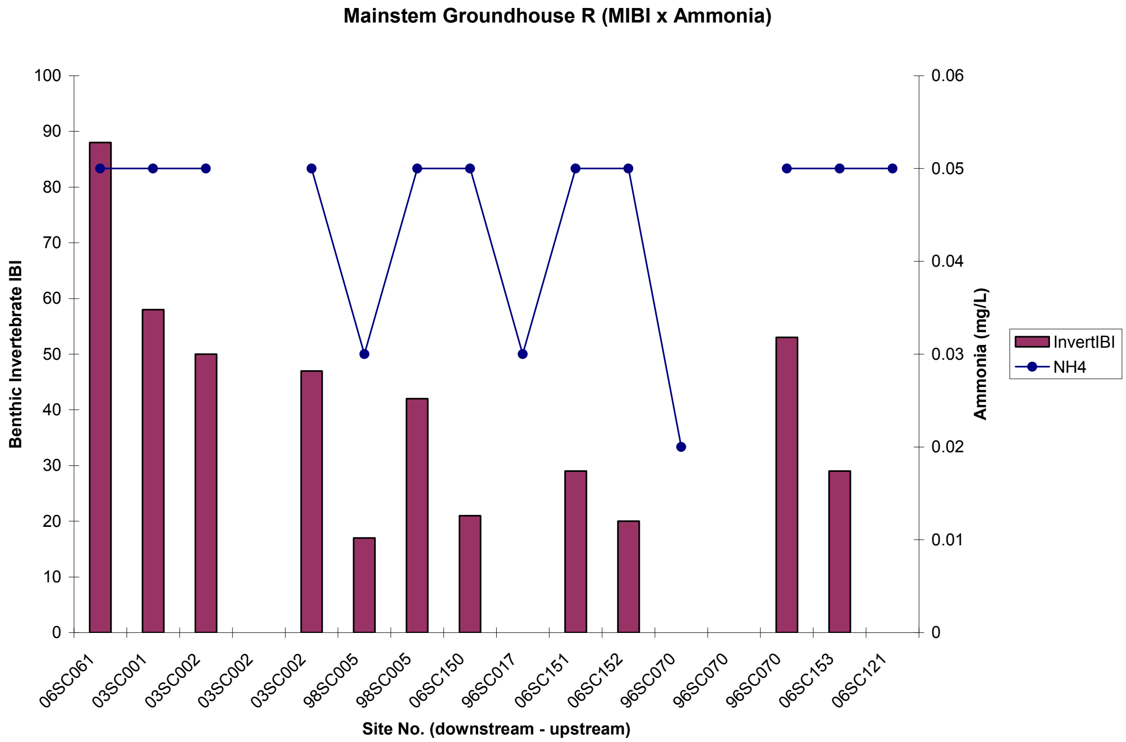
Although MIBI scores indicate disturbance throughout the watershed, neither the IBI score itself or any of the individual metrics respond solely to changes in thermal regimes. Therefore, water temperature data were directly examined to assess temperature change as a candidate cause of impairment.

Figure D-6 shows water temperature data collected at STORET water quality sites along the main stem of the Groundhouse River. The majority of temperature data in the watershed was collected in 2005 as part of the Groundhouse River TMDL; those data are displayed. It does not appear that temperature serves as a stressor in any part of the Groundhouse River watershed.

6.2.2 Candidate Cause 2: Low Dissolved Oxygen and Excess Nutrient Loading

Organic deposition or introduction into this sub-drainage is prevalent throughout and may indicate the occurrence of low dissolved oxygen stress and increasing toxicity of related analytes like ammonia. Even though the concentrations of ammonia were low they are early warning signs that select sites have a propensity to be affected by these poor water quality conditions. Figure D-21 shows the ammonia concentrations in the mainstem and South Fork Groundhouse Rivers as compared to invertebrate IBI scores. Figure D-22 and Figure D-23 show invertebrate IBI scores compared to total phosphorus concentration and total nitrogen concentration.

The uppermost site (06SC045) and the lowermost site (98SC011) in the South Fork Groundhouse River appear to have the same type of related causes for biological impairments. The uppermost site (06SC045) had very low dissolved oxygen concentrations to which increased ammonia concentrations can likely be attributed. Low gradient at this site and slow water movement explains the dominance of burrowing benthic macroinvertebrates, free-swimming taxa, and the species with high tolerance to warm water and low dissolved oxygen concentrations. Many of the same taxa were found to be dominant at the lowermost site (98SC011) and indicate that some of the same stressors are responsible for impaired biological conditions.



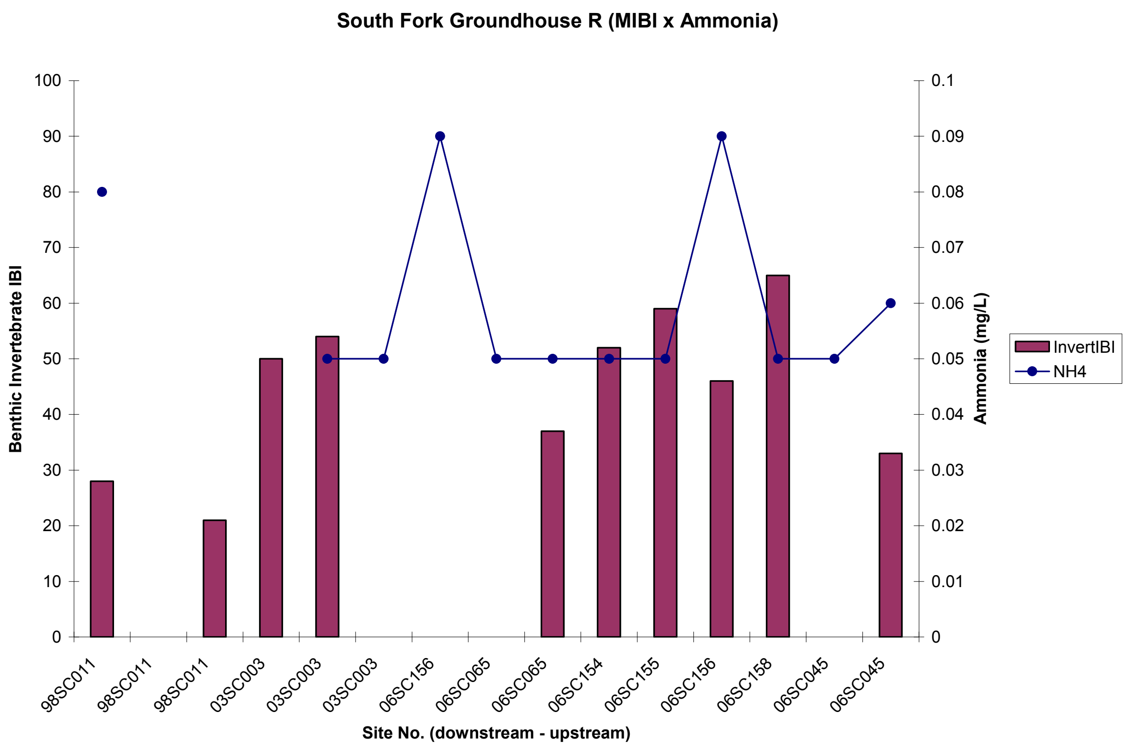
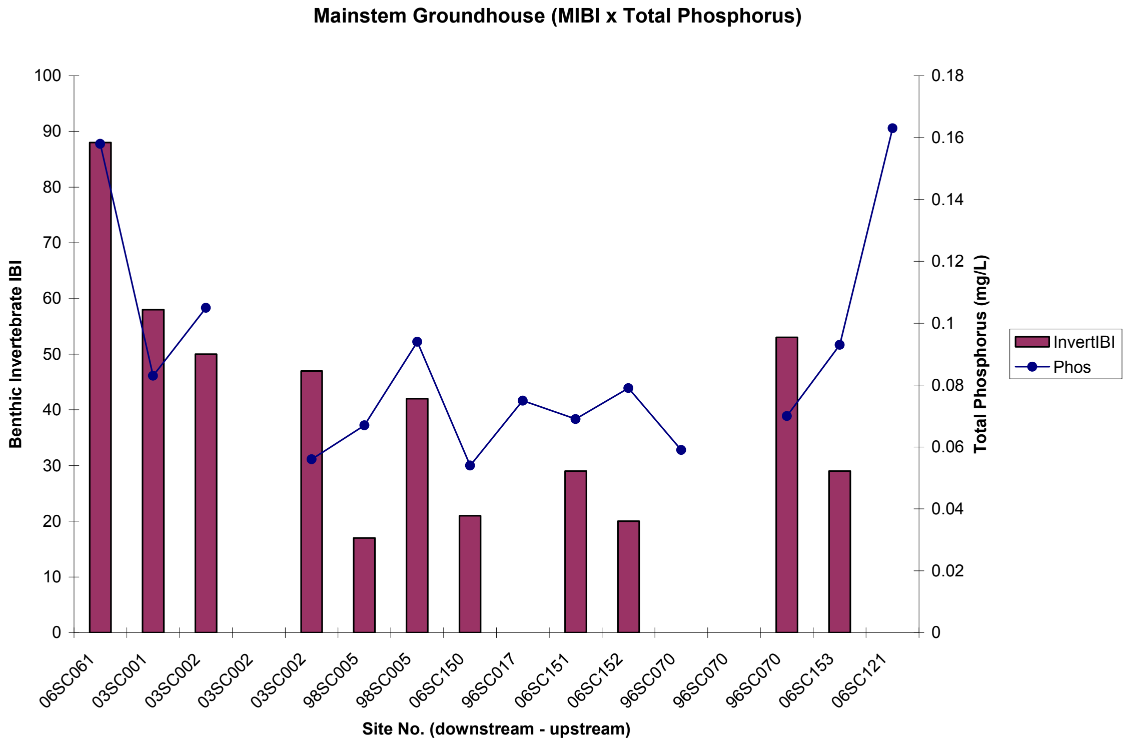


Figure D-21. Ammonia concentrations and MIBI scores along the Groundhouse River.

Mainstem Groundhouse (MIBI x Total Phosphorus)



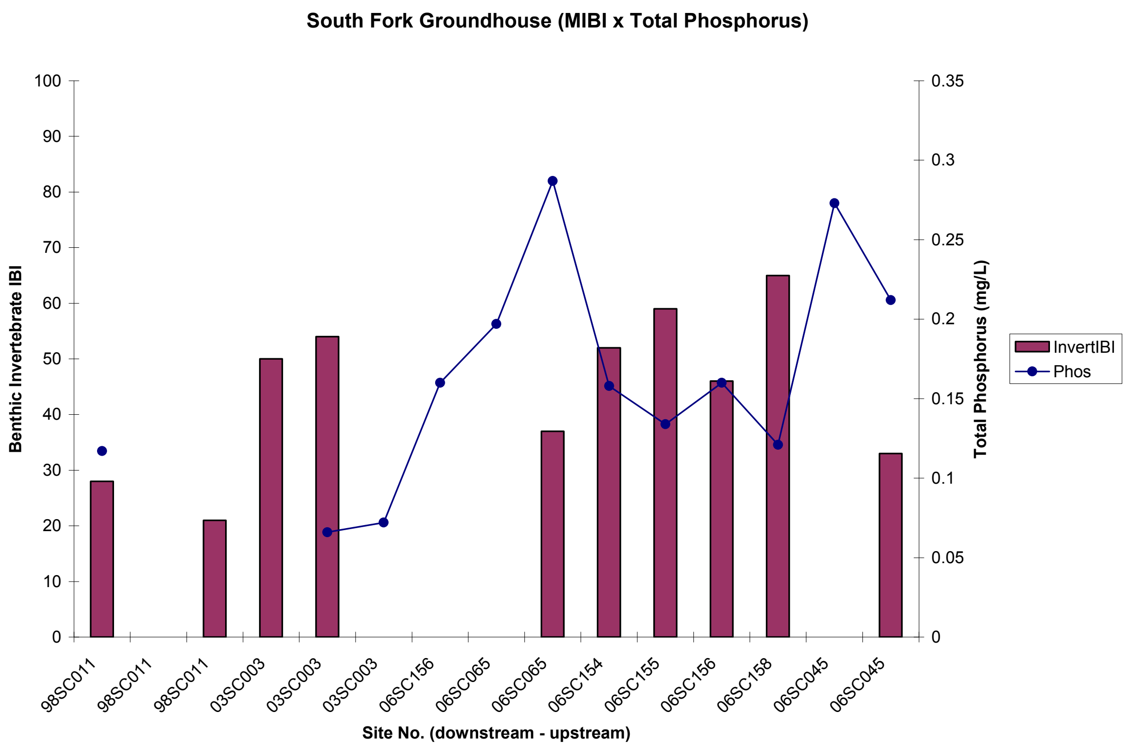
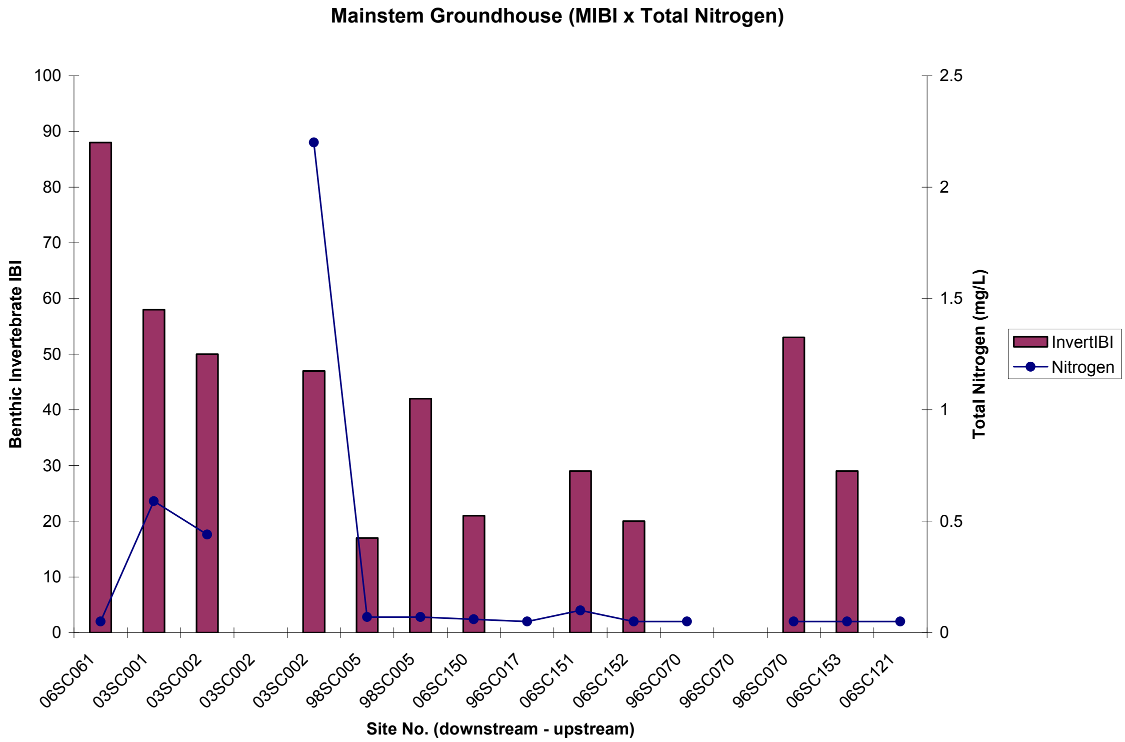


Figure D-22. Total phosphorus concentrations and MIBI scores along the Groundhouse River.



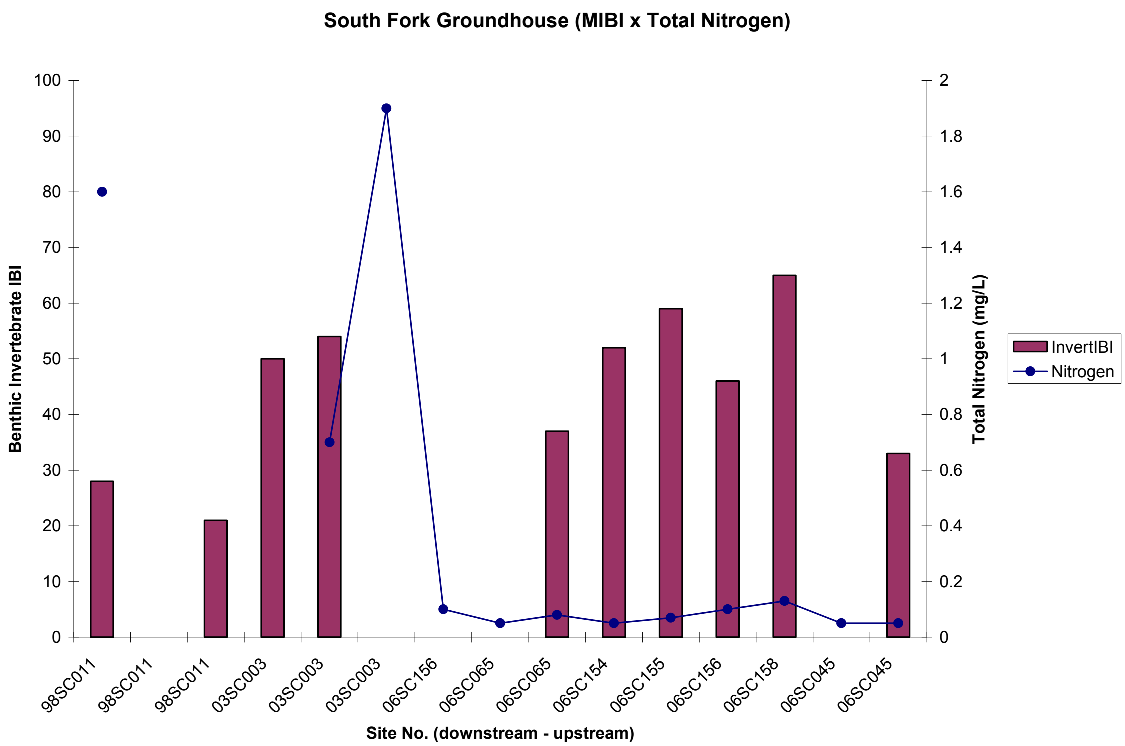


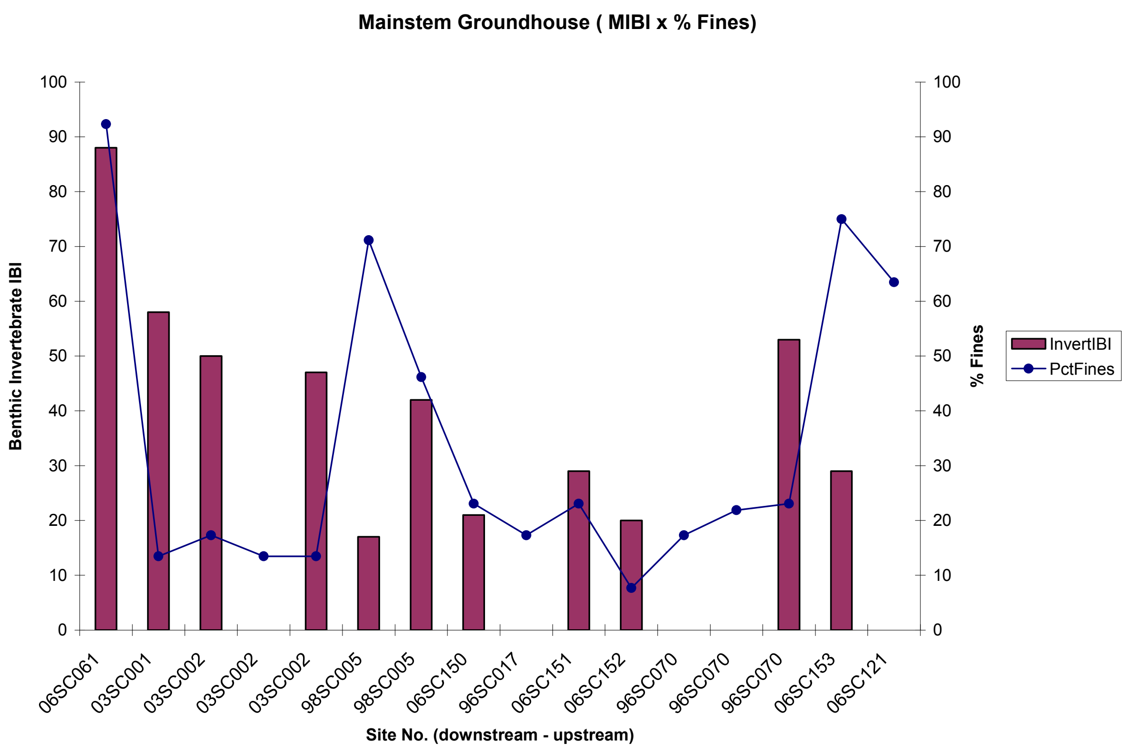
Figure D-23. Total nitrogen concentrations and MIBI scores along the Groundhouse River.

6.2.3 Candidate Cause 3: Excess Fine Sediment Leading to Habitat Loss

Throughout the Groundhouse River watershed, impaired macroinvertebrate communities are associated with increases in bedded sediment (e.g., % Fines; Figure D-24).

Sites along the main stem of the Groundhouse River where benthic invertebrate community condition was depressed corresponded with both increased ammonia levels and high total suspended solids concentrations (Figure D-25). Suspended sediment mobilization in this river channel is likely the result of a convergence between increased gradient and the introduction of eroded materials from stream bank and/or tributary. Benthic invertebrate response to increased sedimentation, including suspended solids, is increased drift behavior in order to escape being buried. The behavioral drift changes community composition which is then reflected in an impaired community condition.

The most consistent impairment among all South Fork Groundhouse River sites was an increase or presence of high levels of fine sediments. The presence of fine sediments also corresponded with a low gradient channel (Figure D-26). It is likely, then, that low gradient corresponds to low energy to move bedded sediment throughout the South Fork Groundhouse River Watershed. The dominant presence of "burrower" taxa and consumers of plant organic material (e.g., Oligochaeta, *Polypedilum*, and *Dicrotendipes*) further suggest the presence of fine sediments serving as primary habitat.



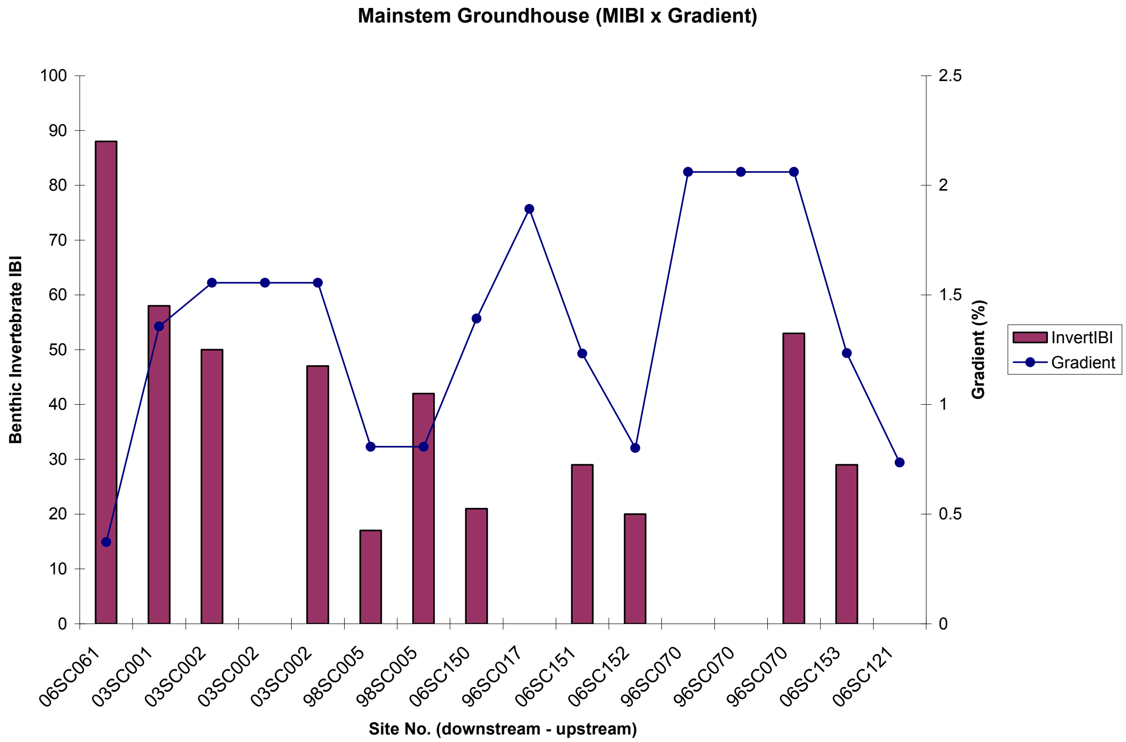
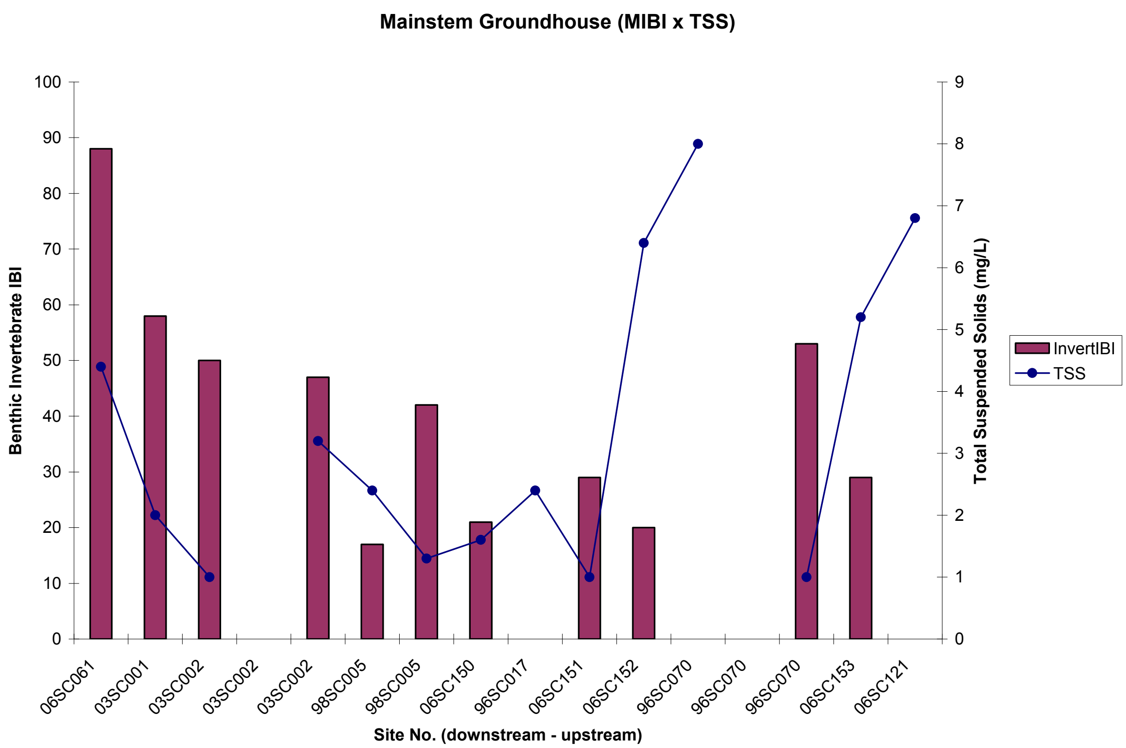


Figure D-24. Benthic invertebrate IBI compared to percent fine sediment and gradient in the Mainstem Groundhouse River.



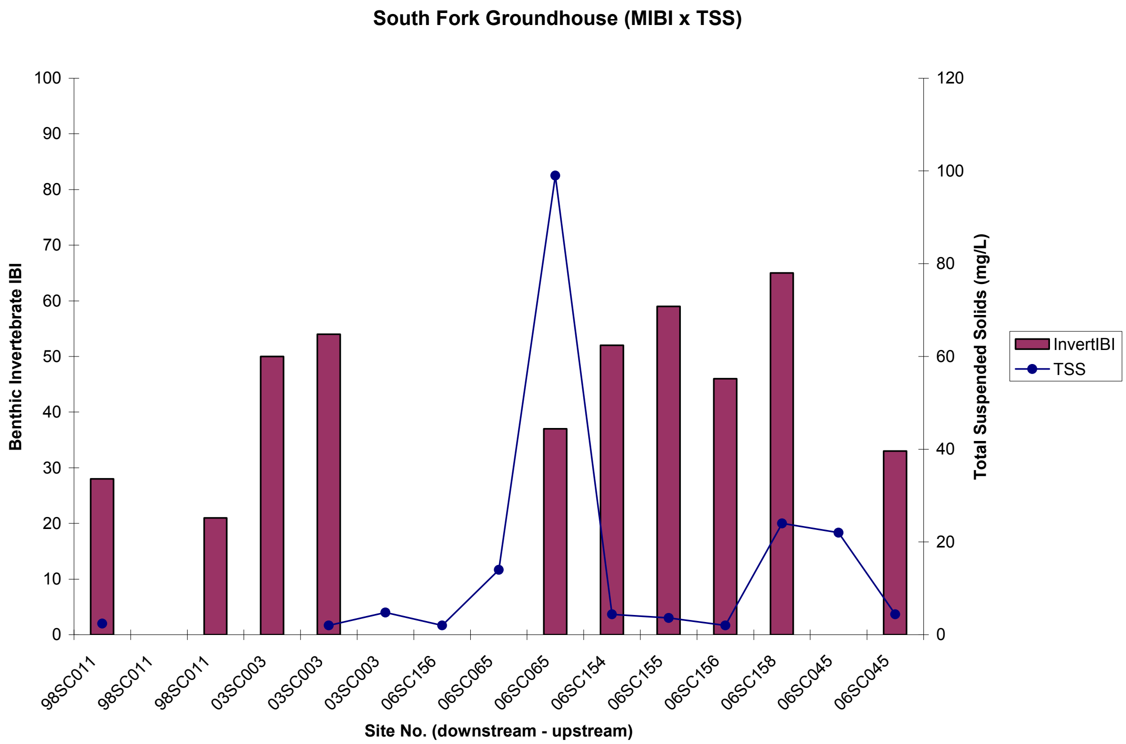
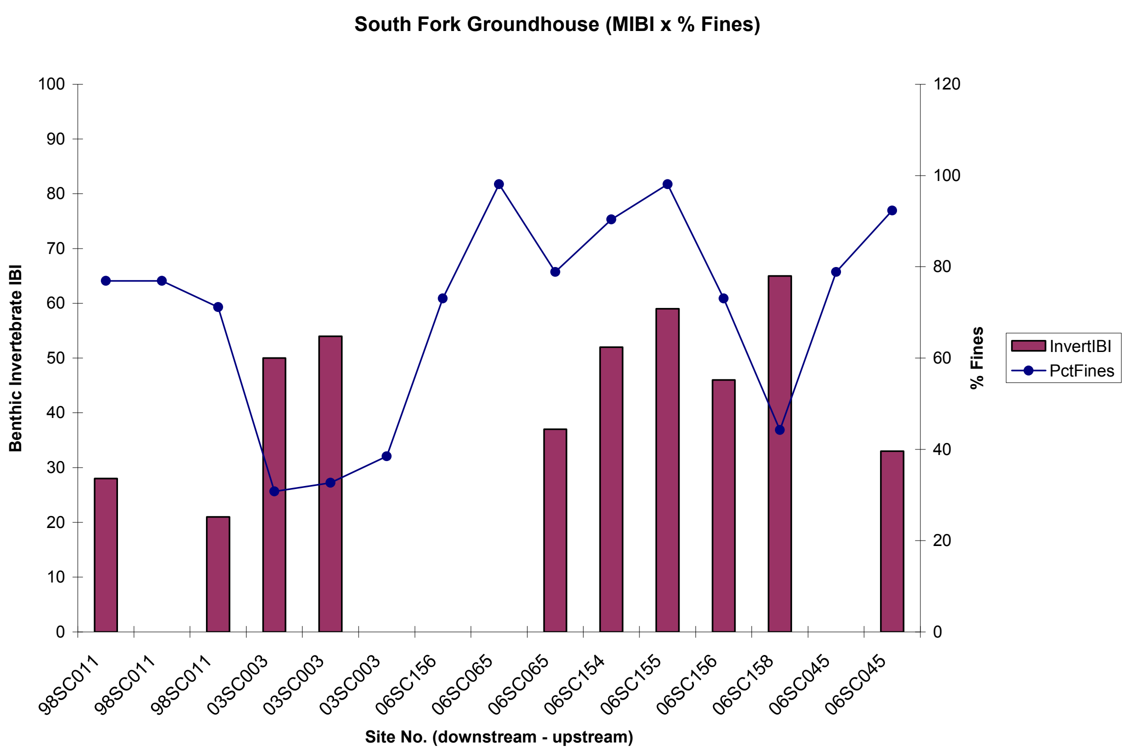


Figure D-25. Total suspended solids concentrations and MIBI scores along the Groundhouse River.



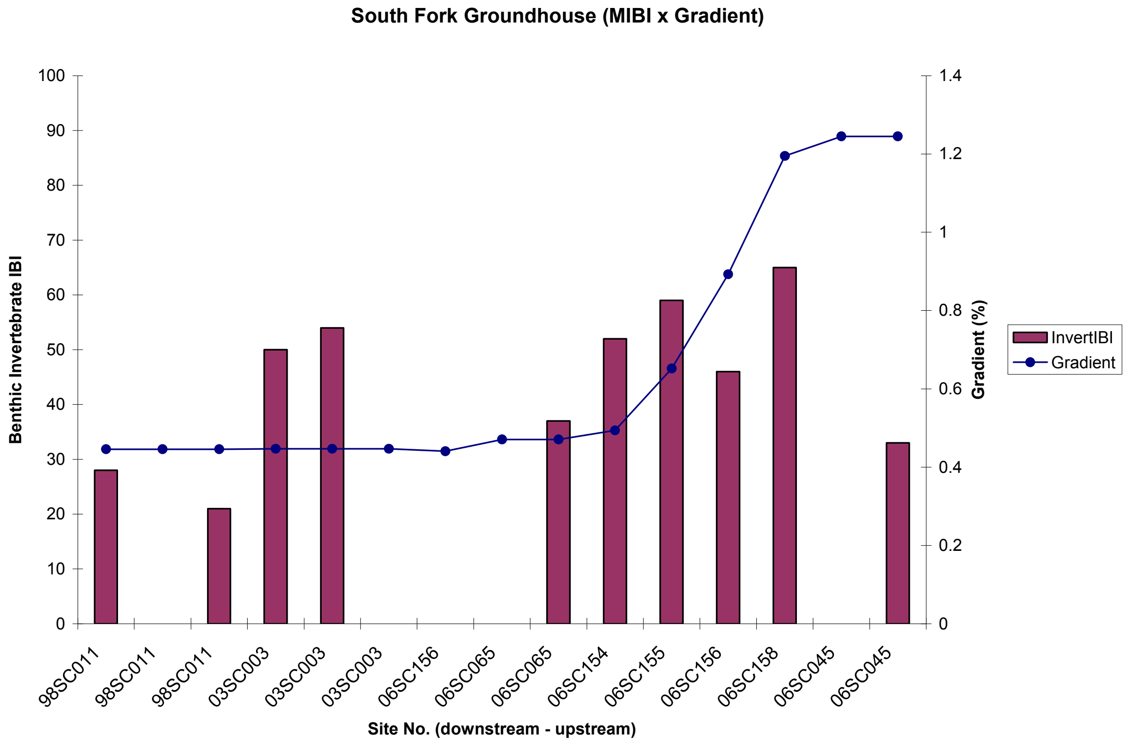


Figure D-26. Benthic invertebrate IBI compared to percent fine sediment and gradient in the South Fork Groundhouse River.

6.3 MACROINVERTEBRATE SUMMARY

A common theme for biological community impairments in the Groundhouse River watershed is that sediment impairments (e.g., % Fines) and select nutrient analytes (e.g., NH_4 and TP) with high concentrations were associated in each case.

Mainstem Groundhouse River sites with severely impaired benthic invertebrate communities could be identified by two unique groups of stressors and were associated with an upper river group and a lower river group. The upper river region can be defined as those impaired sites (e.g., 06SC150, 06SC151, 06SC152, and 06SC153) above the Ogilvie Wastewater Treatment Plant. These sites were more frequently identified as having low gradient stream channels with some increased % fines in habitable substrates. Water chemical characterization showed elevated ammonia (NH_4) concentrations in these same site reaches.

The lower group of sites (e.g., 98SC005 and 03SC002) on the Mainstem Groundhousse River is in or is following the sandy area more susceptible to erosion defined as Impaired Area 1 in the fish analysis. Site 03SC002 is also located downstream of the Waste Water Treatment Plant. These sites were characterized with elevated concentrations of total suspended solids (TSS), increased ammonia (NH_4), and total phosphorus (TP) concentrations. Factors such as increased flow from treated effluent discharge along with annual hydrograph pattern and re-suspension of organics below this outfall could affect results for water quality characteristics. The lower river sites were consistently dominated by filter-feeding benthic invertebrates (e.g., Hydropsychid caddisflies, etc.) further confirming the volume of available suspended materials in the water column.

Sites along the Mainstem Groundhouse River where benthic invertebrate community condition was depressed corresponded with increased ammonia levels and with high total suspended sediment concentrations.

Select sites along the South Fork Groundhouse River were impaired by either physical habitat alterations or by water quality conditions. The most consistent impairment among all South Fork Groundhouse River sites was an increase or presence of high levels of fine sediments. The presence of fine sediments also corresponded with a low gradient channel condition. The next most prevalent stressors at these sites were the potential for dissolved nutrients like ammonia and phosphorus. The presence of ammonia in concentrations high enough to influence the benthic invertebrate community may be coupled with low dissolved oxygen concentrations.

7 Recommendations

Primary stressors identified in reaches of each sub-drainage (e.g., South Fork and Mainstem of the Groundhouse River) are related to contributions from surrounding land uses and impairments exacerbated by the natural channel characteristics. Some reaches are more susceptible to water quality and biological impairment based on specific features of the channel.

Restoration and/or measures for protection of water quality and biological communities need to consider the following factors: seasonality (for water quality issues and biological communities) and sensitive habitat critical for completion of life cycles for indicator species (i.e., benthic invertebrates and fish). Signs of impairment in both water quality characteristics and in biological communities are identifiable. In some cases, there are signs of severe impairments to biological communities. Preserving existing habitat and implementing best management practices at key locations will provide for expanding important refugia. In at least one location complex refugia appears to offset potential impairments caused by nutrient and other physical habitat stressors. Biological communities that can continue to serve as a pool for further re-colonization in reclaimed portions of the drainage will promote recovery. The TMDL and associated implementation plan should identify important habitat for specific indicator taxa and implementation steps for preventing pollutant entry into the river channel.

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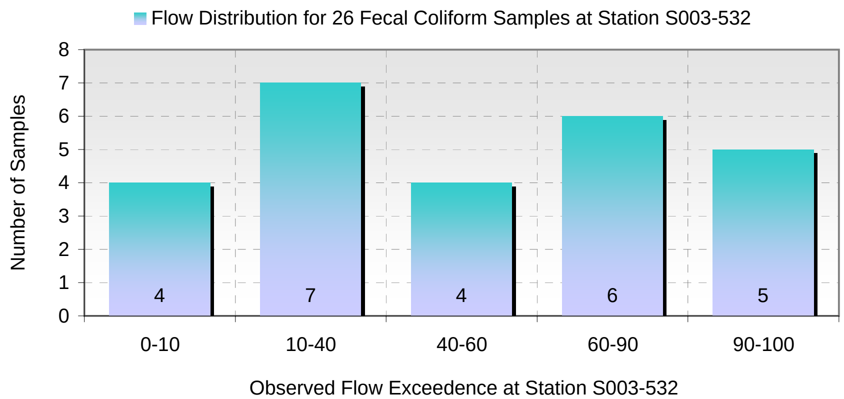
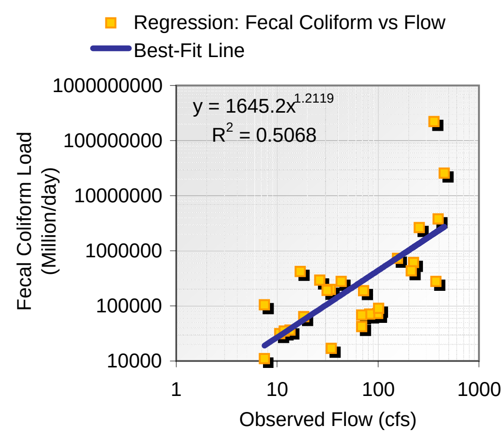
Appendix E. Load Duration Curve Results

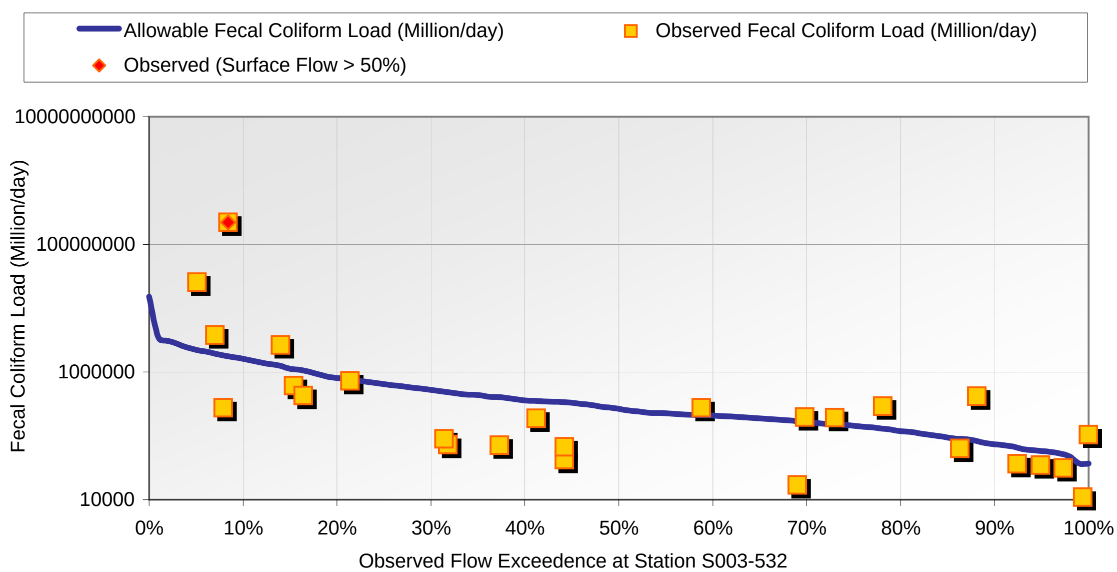
[200 org/100ml Standard](#)
[[1: Fecal Coliform at Station S003-532 (MPN/100mL)] -vs- [1: Flow at Station S003-532 (cfs)]](#)

1. Data Assessment and Trend Confirmation





2. Load Exceedence Analysis

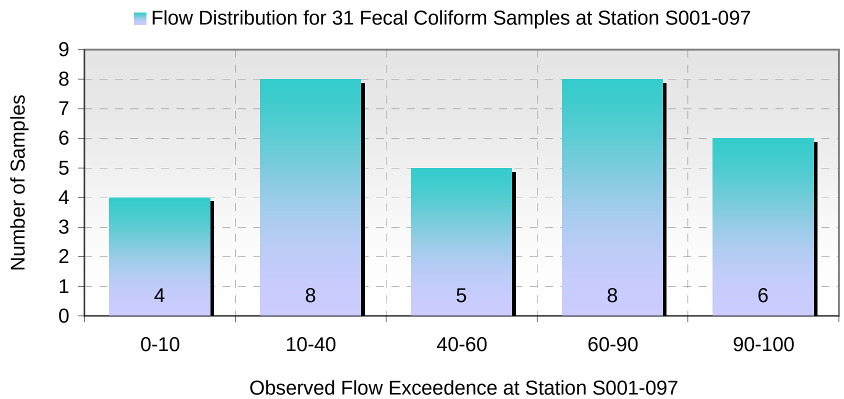


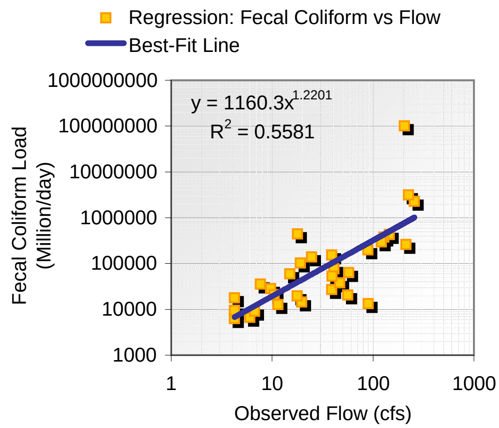
3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	26-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	4	396.43	1,939,815	25,606,657	92.4%
10-40	7	256.76	1,256,368	2,638,373	52.4%
40-60	4	43.41	212,394	276,112	23.1%
60-90	6	28.91	141,452	244,995	42.3%
90-100	5	7.51	36,732	104,687	64.9%

[200 org/100ml Standard](#)
[[5: Fecal Coliform at Station S001-097 (MPN/100mL)] -vs- [5: Flow at Station S001-097 (cfs)]](#)

1. Data Assessment and Trend Confirmation





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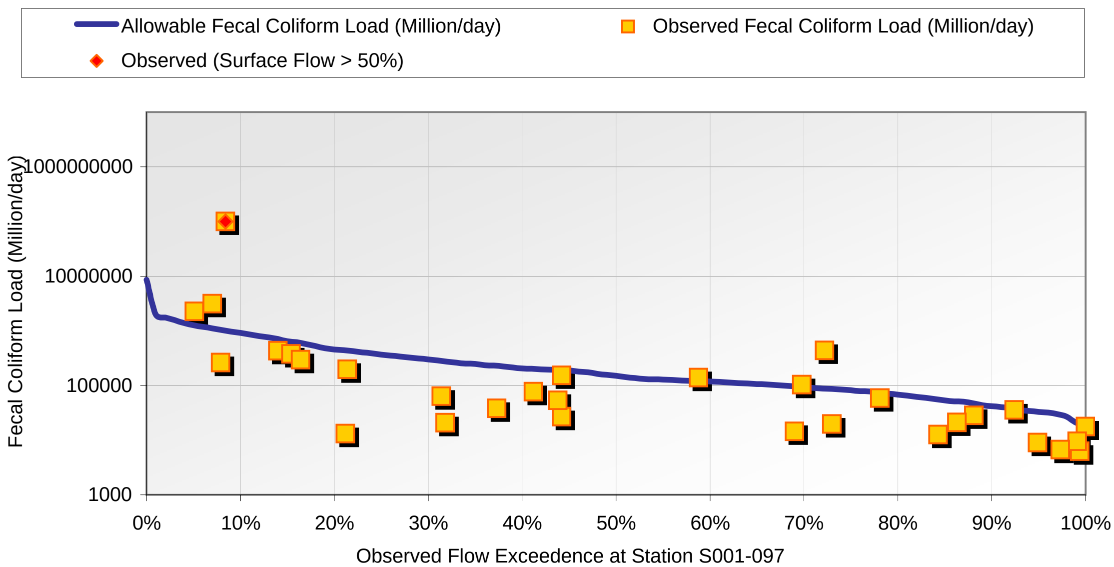
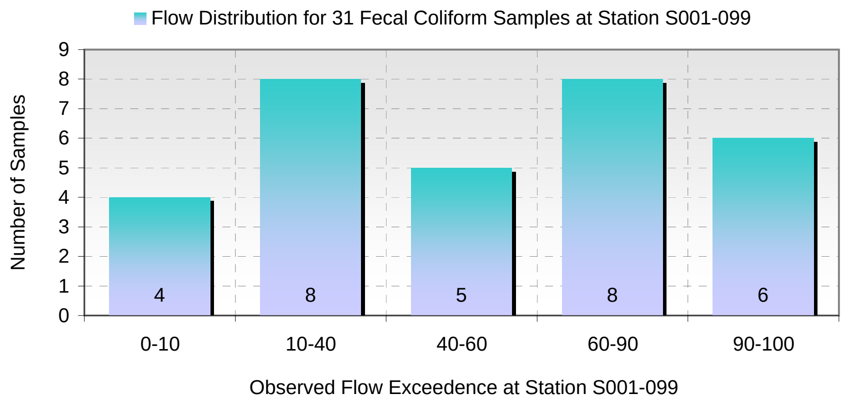


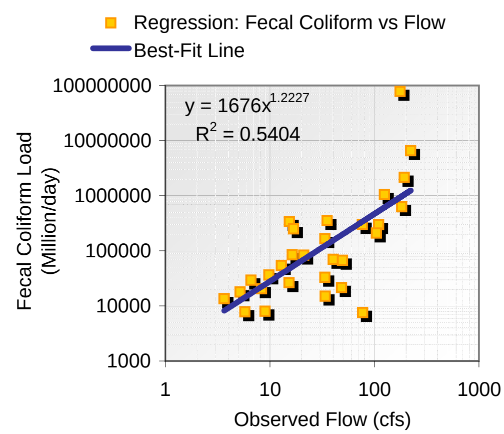
3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	31-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	4	224.79	1,099,953	3,134,865	64.9%
10-40	8	89.49	437,885	130,793	0.0%
40-60	5	24.61	120,436	138,501	13.0%
60-90	8	18.50	90,511	270,248	66.5%
90-100	6	5.15	25,185	9,248	0.0%

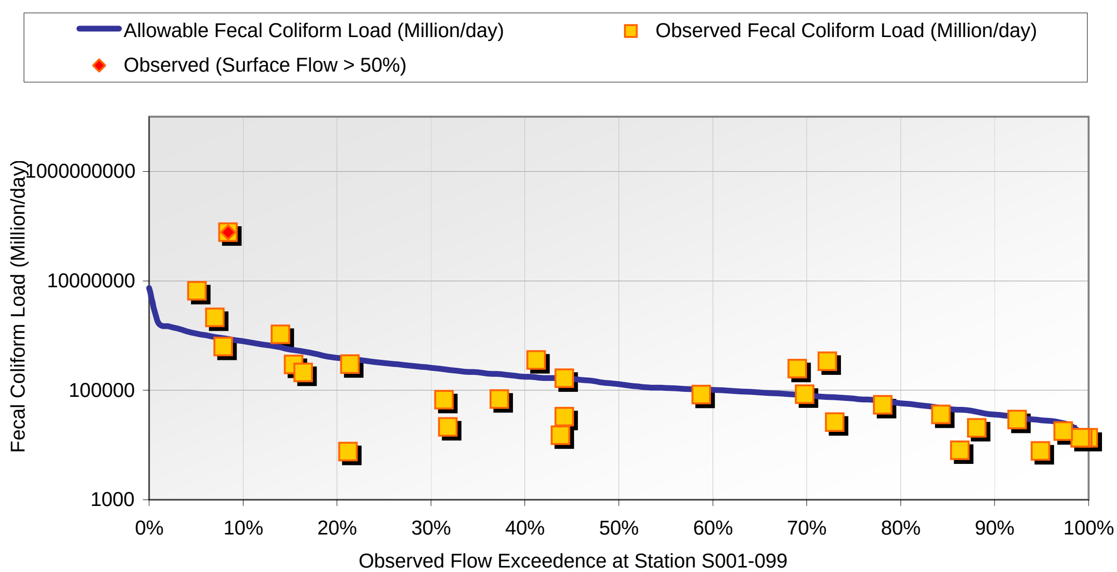
[200 org/100ml Standard](#)
[[6: Fecal Coliform at Station S001-099 (MPN/100mL)] -vs- [6: Flow at Station S001-099 (cfs)]](#)

1. Data Assessment and Trend Confirmation





2. Load Exceedence Analysis

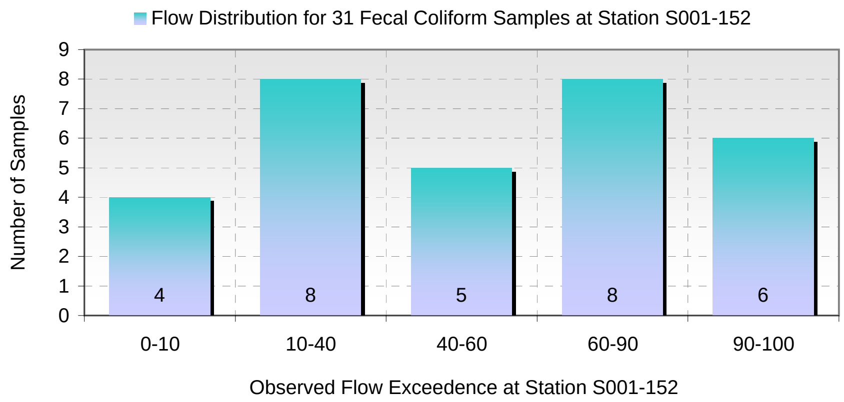


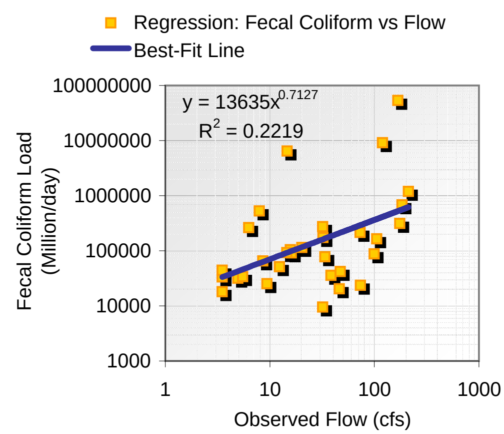
3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	31-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	4	193.56	947,095	6,522,884	85.5%
10-40	8	125.36	613,409	1,042,796	41.2%
40-60	5	34.46	168,642	259,298	35.0%
60-90	8	16.45	80,498	248,134	67.6%
90-100	6	4.43	21,685	13,451	0.0%

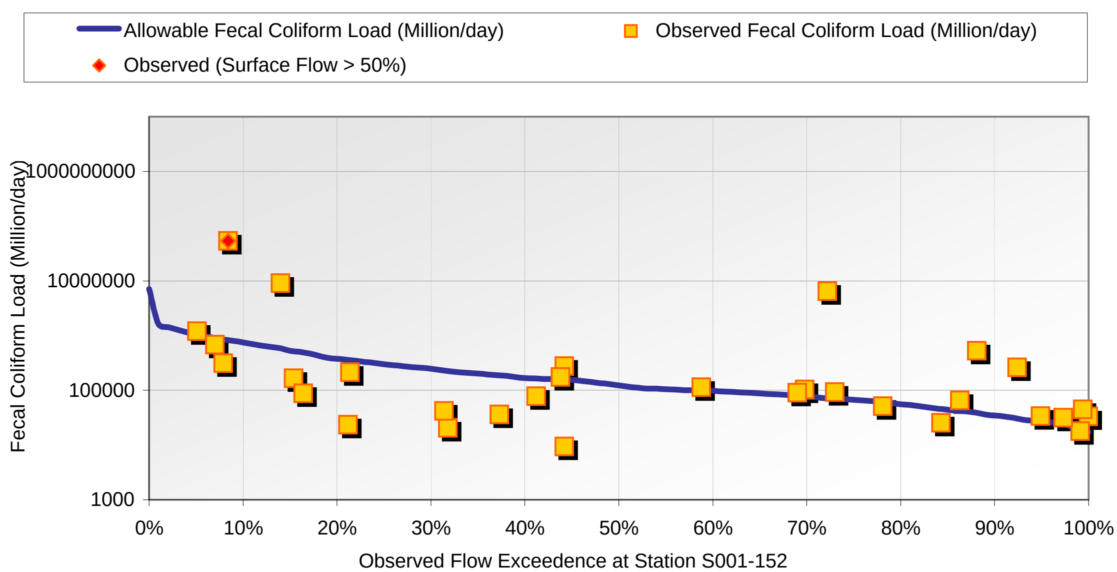
200 org/100ml Standard
[3: Fecal Coliform at Station S001-152 (MPN/100mL)] -vs- [3: Flow at Station S001-152 (cfs)]

1. Data Assessment and Trend Confirmation





2. Load Exceedence Analysis

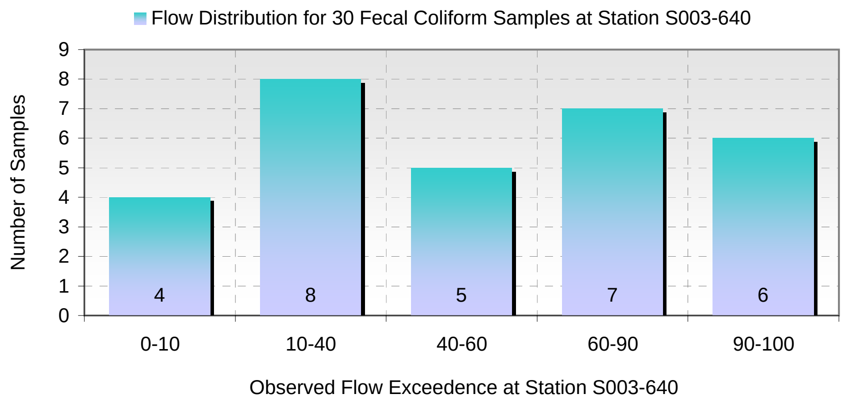


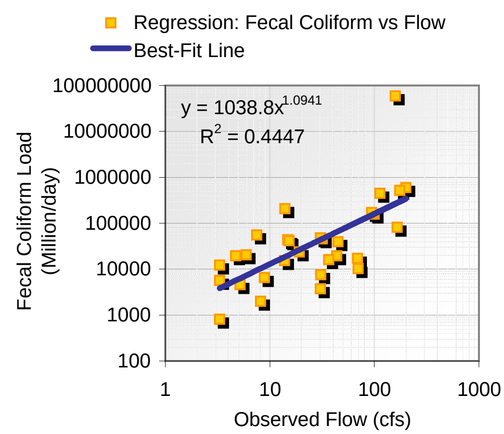
3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	31-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	4	189.89	929,150	27,291,248	96.6%
10-40	8	119.57	585,066	9,068,516	93.5%
40-60	5	32.09	157,000	174,096	9.8%
60-90	8	14.63	71,582	98,157	27.1%
90-100	6	4.23	20,683	33,513	38.3%

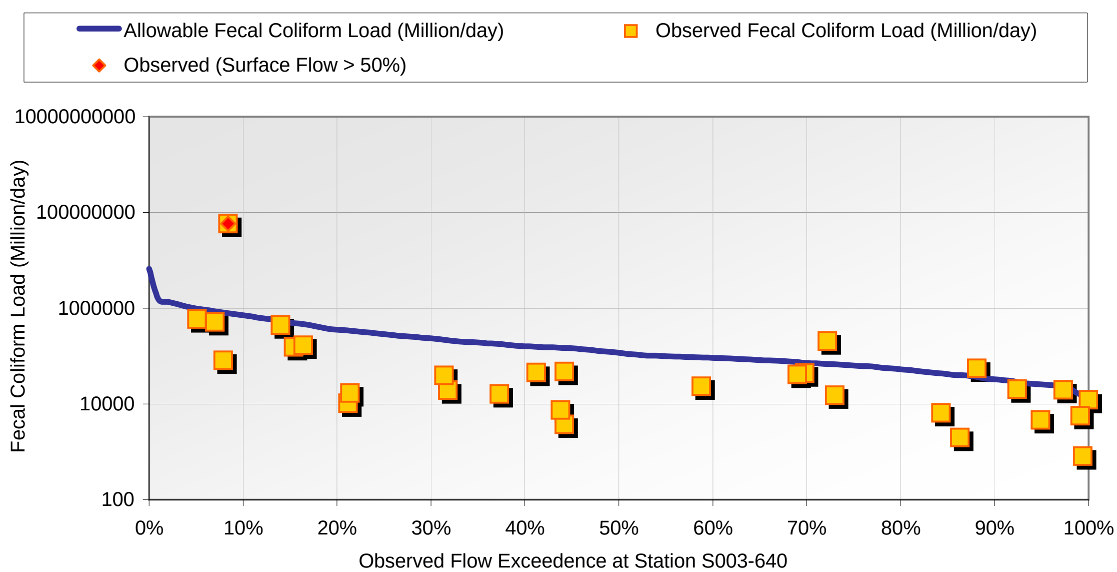
[200 org/100ml Standard](#)
[[7: Fecal Coliform at Station S003-640 (MPN/100mL)] -vs- [7: Flow at Station S003-640 (cfs)]](#)

1. Data Assessment and Trend Confirmation





2. Load Exceedence Analysis

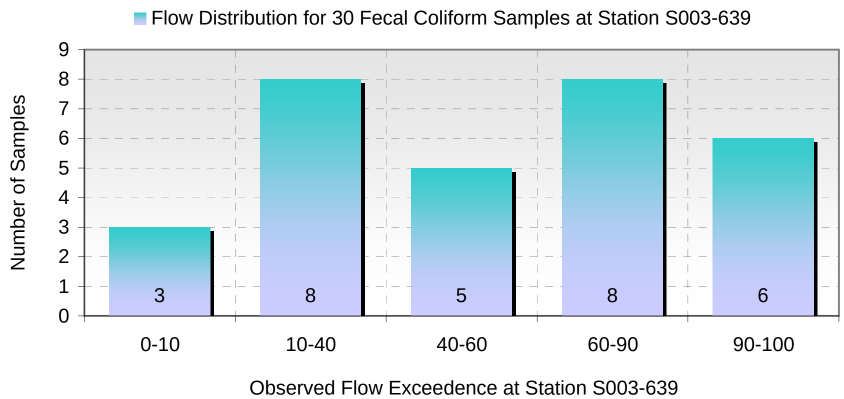


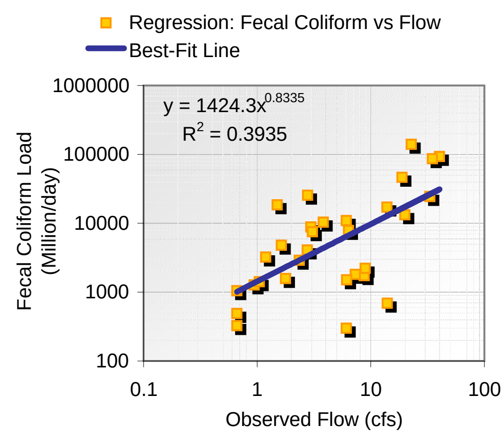
3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	30-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	4	159.24	779,194	58,439,572	98.7%
10-40	8	69.72	341,140	29,411	0.0%
40-60	5	30.44	148,936	23,457	0.0%
60-90	7	10.73	52,504	129,901	59.6%
90-100	6	4.01	19,621	8,925	0.0%

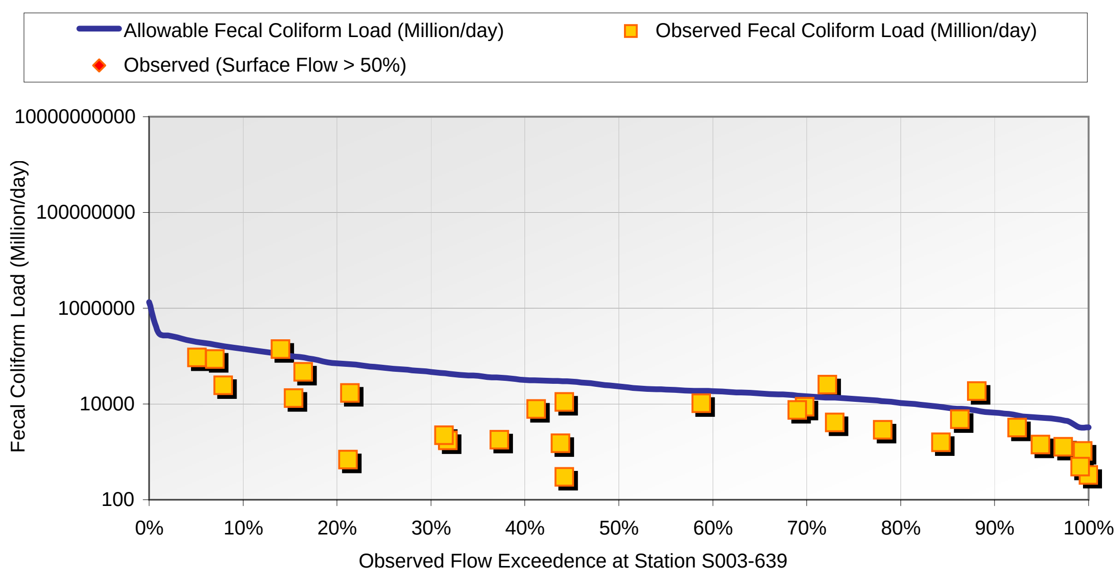
200 org/100ml Standard
[9: Fecal Coliform at Station S003-639 (MPN/100mL)] -vs- [9: Flow at Station S003-639 (cfs)]

1. Data Assessment and Trend Confirmation





2. Load Exceedence Analysis

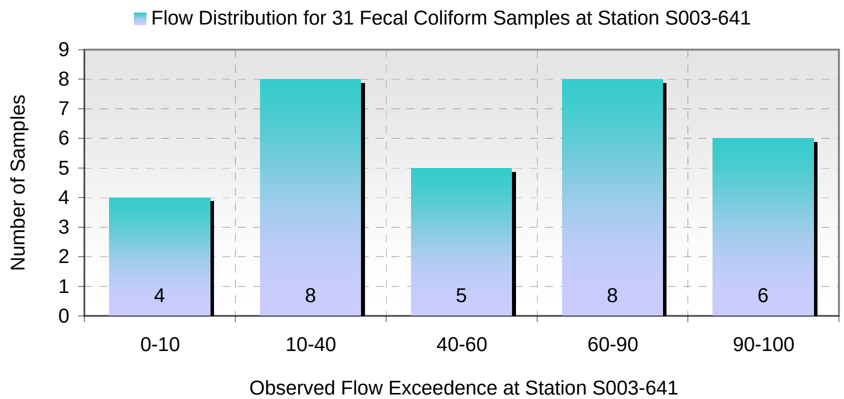


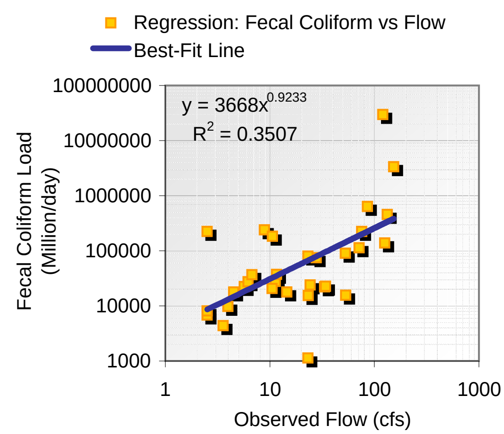
3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	30-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	3	35.08	171,635	85,817	0.0%
10-40	8	22.72	111,163	138,954	20.0%
40-60	5	6.10	29,830	7,823	0.0%
60-90	8	2.15	10,516	21,851	51.9%
90-100	6	0.80	3,930	1,154	0.0%

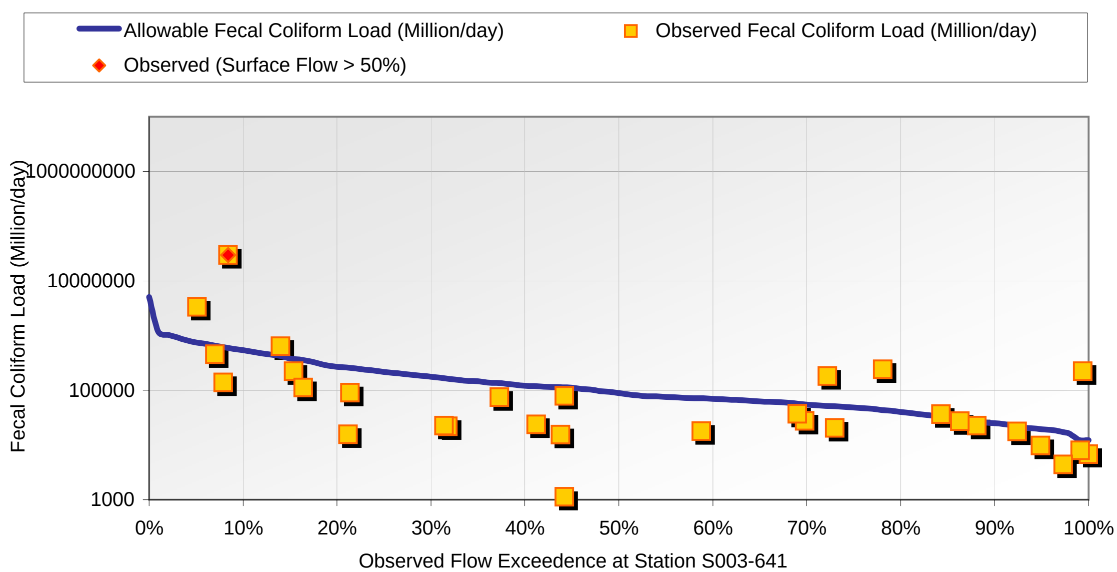
[200 org/100ml Standard](#)
[[8: Fecal Coliform at Station S003-641 (MPN/100mL)] -vs- [8: Flow at Station S003-641 (cfs)]](#)

1. Data Assessment and Trend Confirmation





2. Load Exceedence Analysis

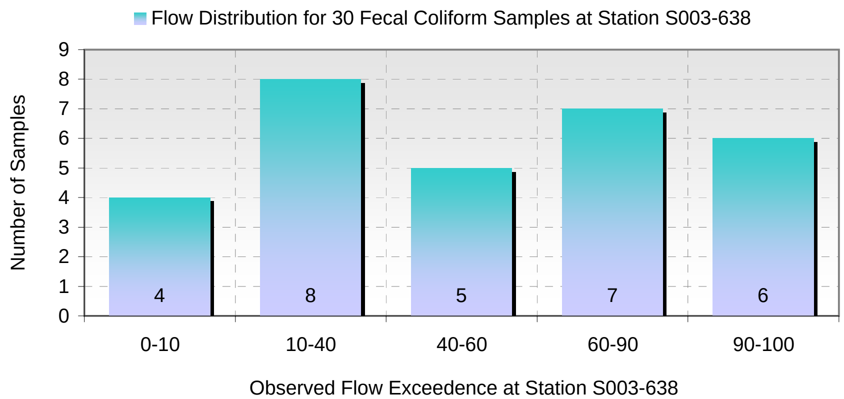


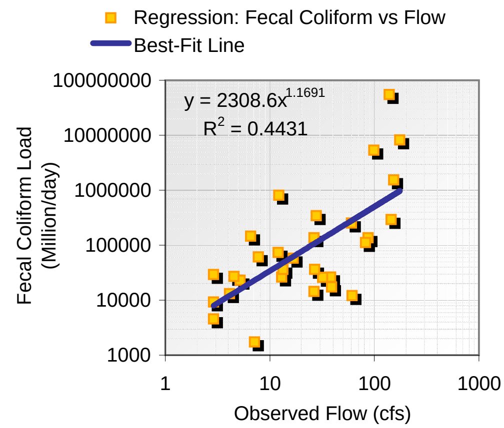
3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	31-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	4	136.98	670,245	16,476,993	95.9%
10-40	8	86.25	422,039	633,059	33.3%
40-60	5	23.15	113,253	17,837	0.0%
60-90	8	8.91	43,616	181,490	76.0%
90-100	6	2.52	12,339	222,105	94.4%

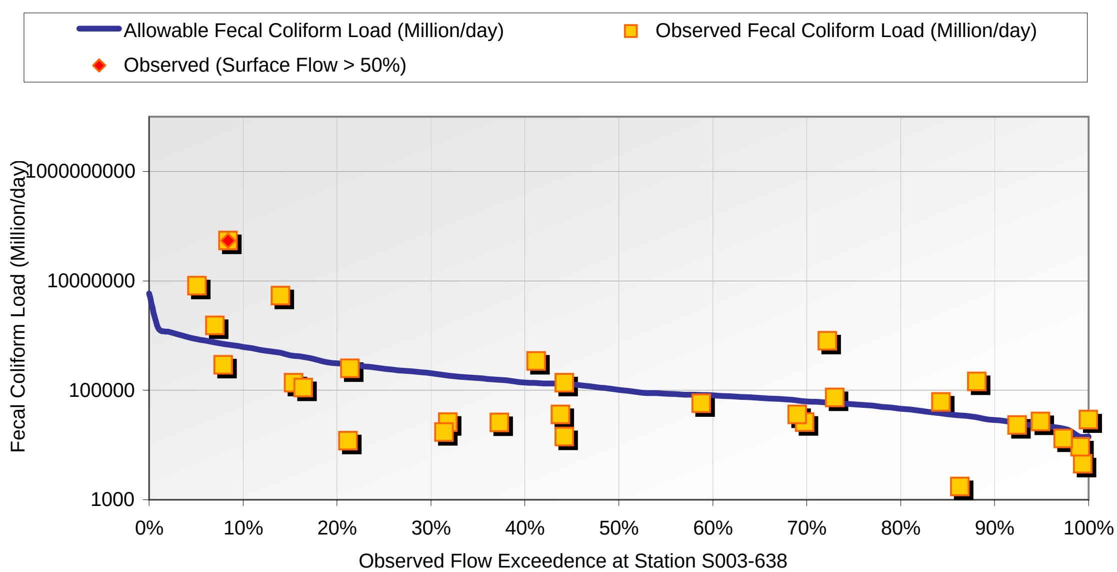
200 org/100ml Standard
[4: Fecal Coliform at Station S003-638 (MPN/100mL)] -vs- [4: Flow at Station S003-638 (cfs)]

1. Data Assessment and Trend Confirmation





2. Load Exceedence Analysis

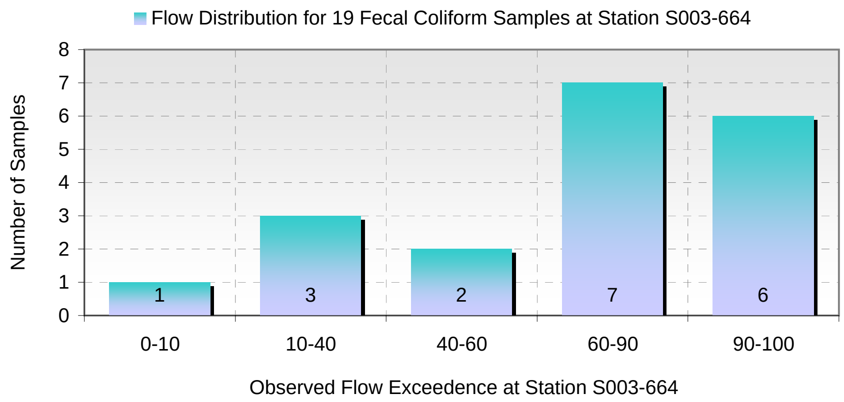


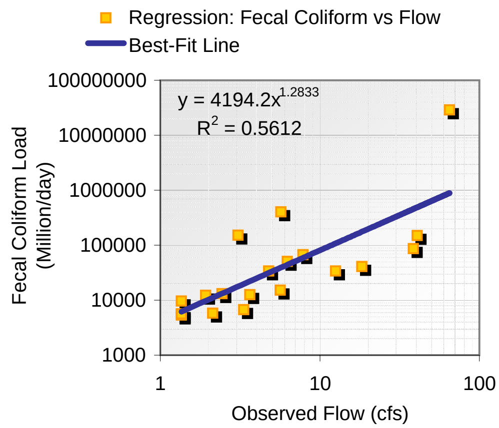
3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	30-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	4	153.06	748,924	8,166,875	90.8%
10-40	8	99.13	485,059	5,335,645	90.9%
40-60	5	27.25	133,355	239,019	44.2%
60-90	7	9.93	48,589	109,330	55.6%
90-100	6	3.73	18,257	27,935	34.6%

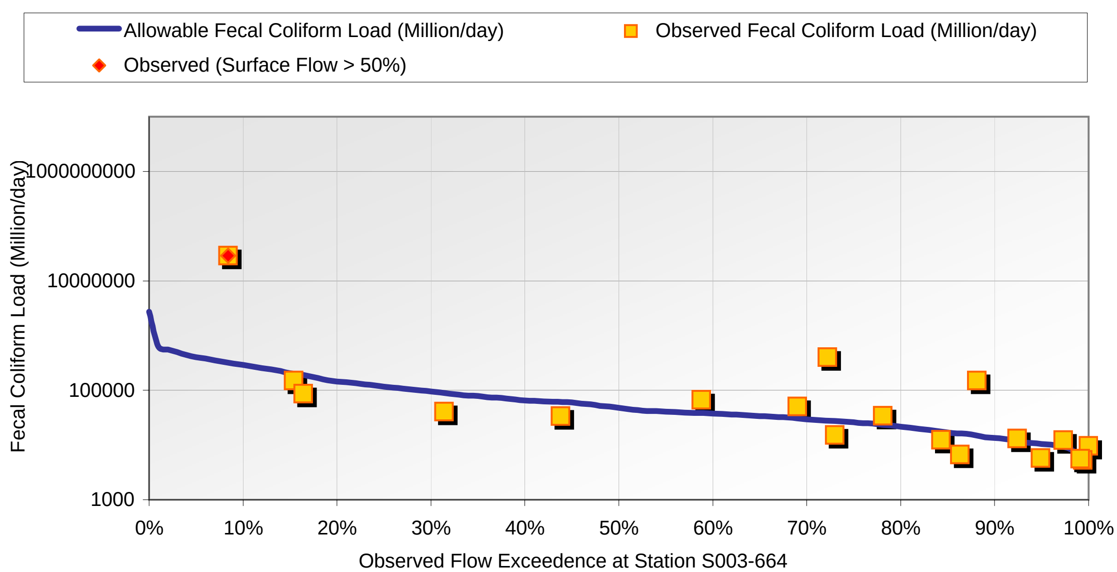
[200 org/100ml Standard](#)
[[2: Fecal Coliform at Station S003-664 (MPN/100mL)] -vs- [2: Flow at Station S003-664 (cfs)]](#)

1. Data Assessment and Trend Confirmation





2. Load Exceedence Analysis

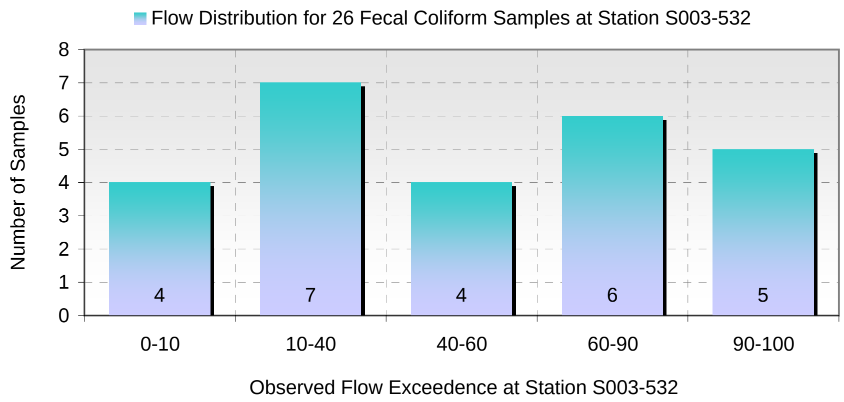


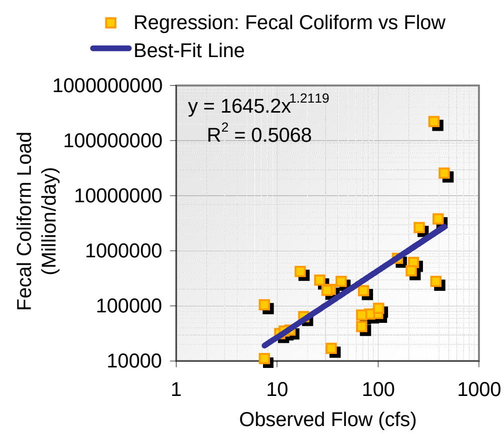
3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	19-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	1	65.12	318,620	28,675,816	98.9%
10-40	3	38.64	189,083	86,033	0.0%
40-60	2	7.84	38,367	67,142	42.9%
60-90	7	5.25	25,669	100,519	74.5%
90-100	6	1.92	9,411	12,234	23.1%

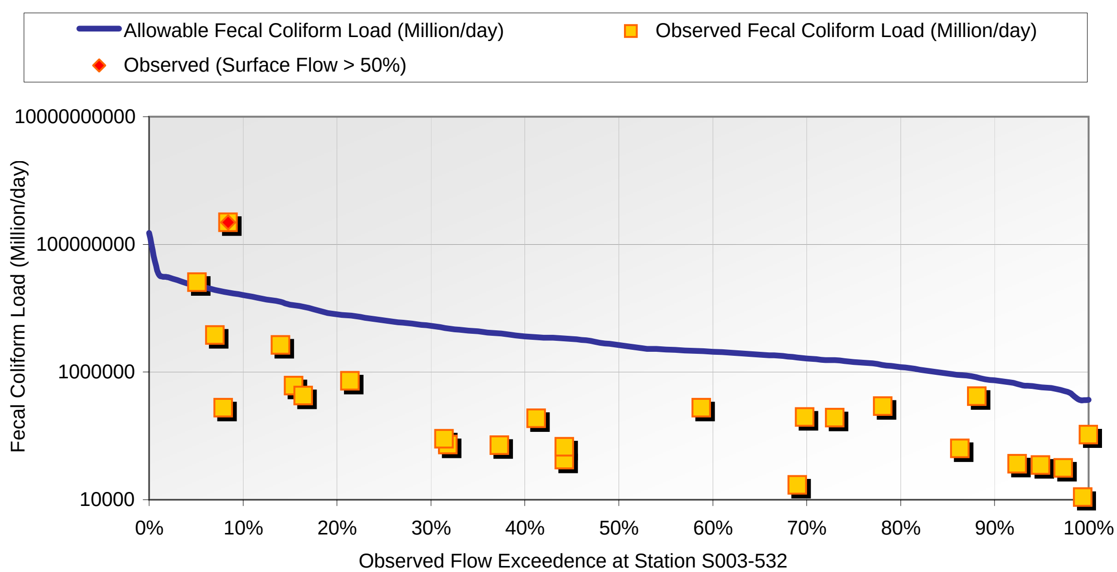
2,000 org/100ml Standard
[1: Fecal Coliform at Station S003-532 (MPN/100mL)] -vs- [1: Flow at Station S003-532 (cfs)]

1. Data Assessment and Trend Confirmation





2. Load Exceedence Analysis

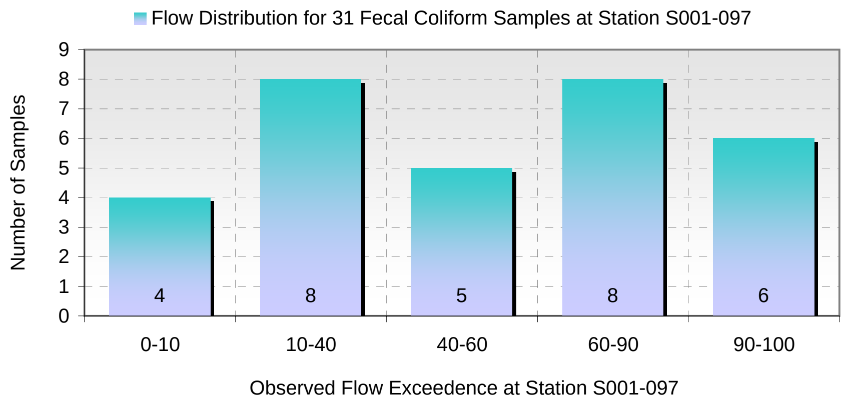


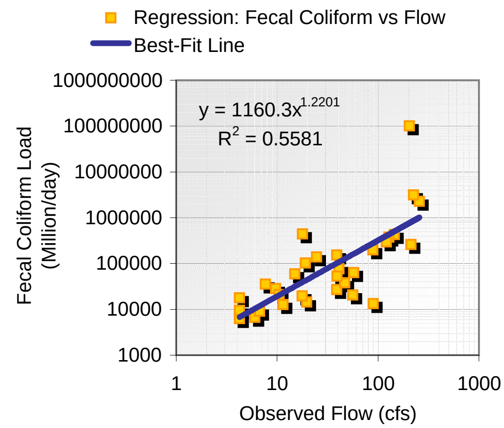
3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	26-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	4	407.76	19,952,533	123,043,375	83.8%
10-40	7	156.99	7,681,793	429,163	0.0%
40-60	4	68.90	3,371,214	127,438	0.0%
60-90	6	28.91	1,414,523	194,589	0.0%
90-100	5	10.65	520,972	34,705	0.0%

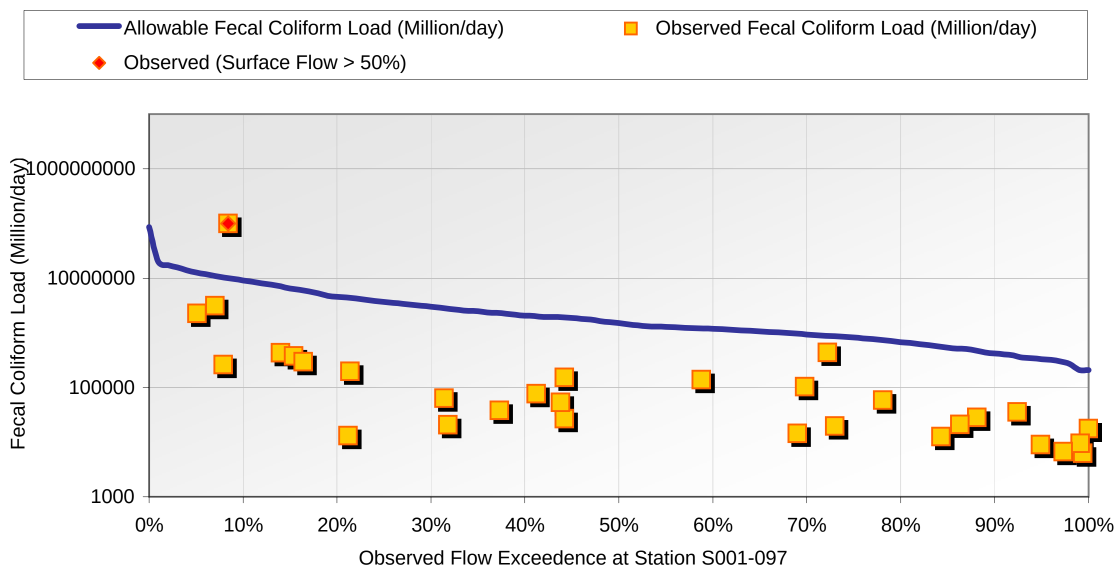
[2,000 org/100ml Standard](#)
[[5: Fecal Coliform at Station S001-097 (MPN/100mL)] -vs- [5: Flow at Station S001-097 (cfs)]](#)

1. Data Assessment and Trend Confirmation





2. Load Exceedence Analysis

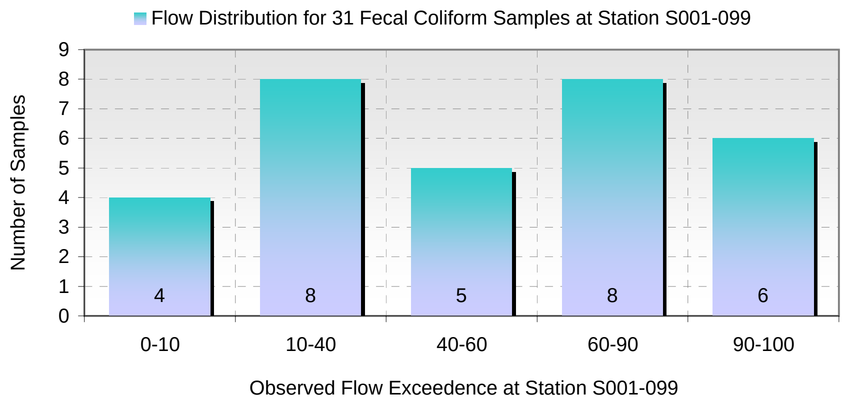


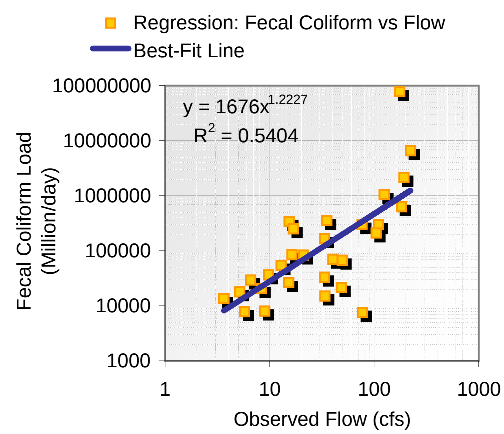
3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	31-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	4	204.40	10,001,683	100,016,827	90.0%
10-40	8	89.49	4,378,853	130,793	0.0%
40-60	5	39.07	1,911,729	76,208	0.0%
60-90	8	16.39	802,091	24,671	0.0%
90-100	6	5.15	251,850	9,248	0.0%

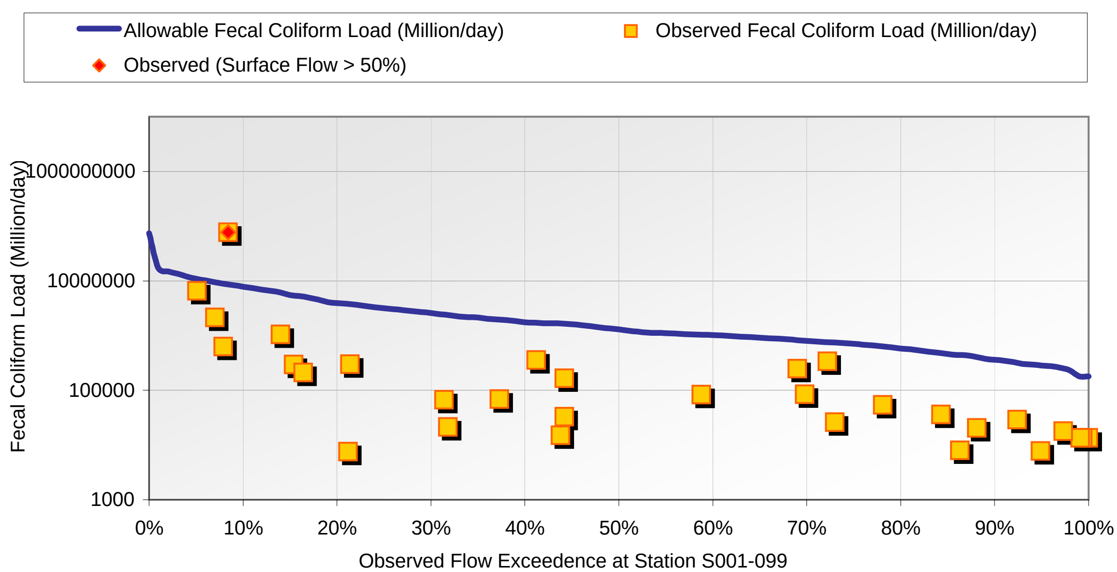
[2,000 org/100ml Standard](#)
[[6: Fecal Coliform at Station S001-099 (MPN/100mL)] -vs- [6: Flow at Station S001-099 (cfs)]](#)

1. Data Assessment and Trend Confirmation





2. Load Exceedence Analysis

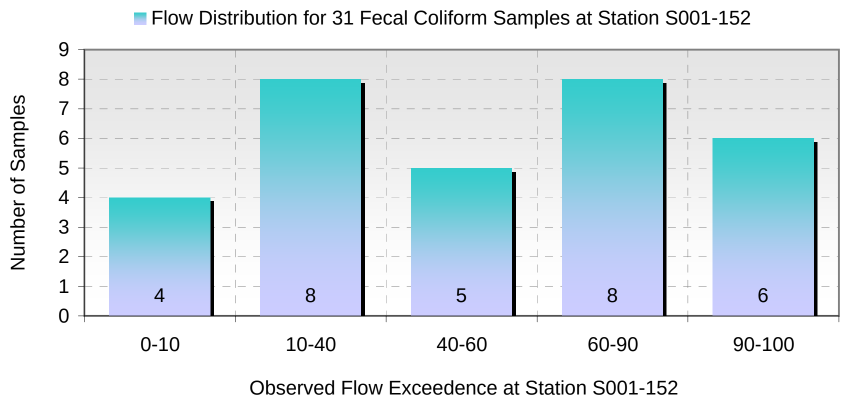


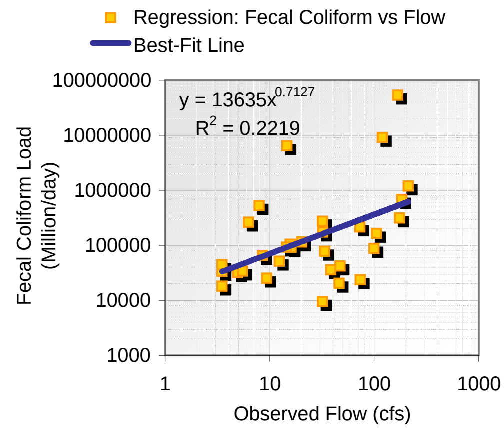
3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	31-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	4	176.00	8,611,776	77,505,981	88.9%
10-40	8	77.05	3,770,335	139,404	0.0%
40-60	5	33.64	1,646,061	82,959	0.0%
60-90	8	14.11	690,626	45,002	0.0%
90-100	6	4.43	216,851	13,451	0.0%

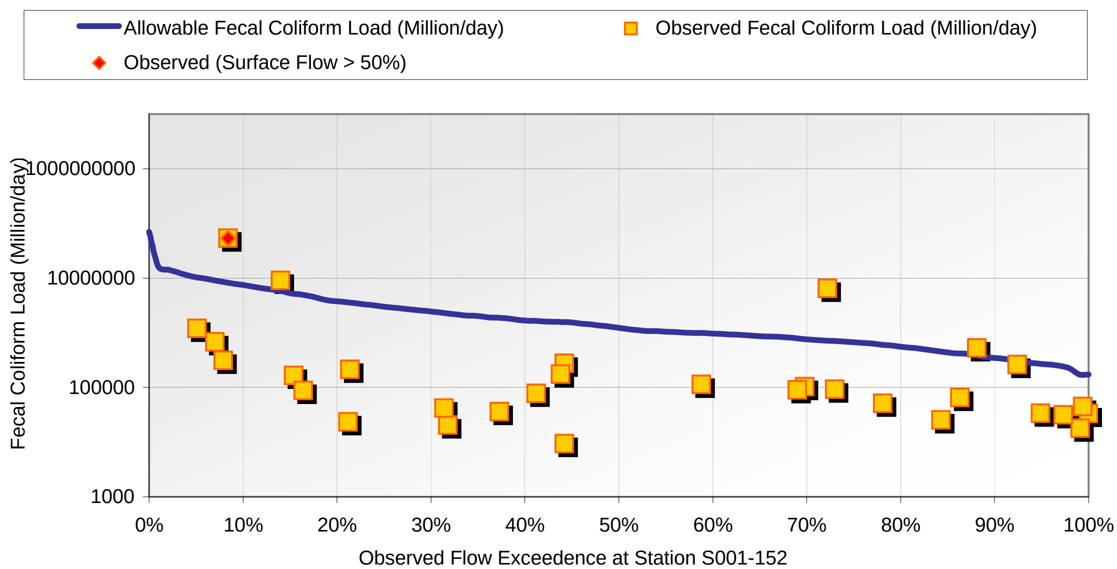
[2,000 org/100ml Standard](#)
[[3: Fecal Coliform at Station S001-152 (MPN/100mL)] -vs- [3: Flow at Station S001-152 (cfs)]](#)

1. Data Assessment and Trend Confirmation





2. Load Exceedence Analysis

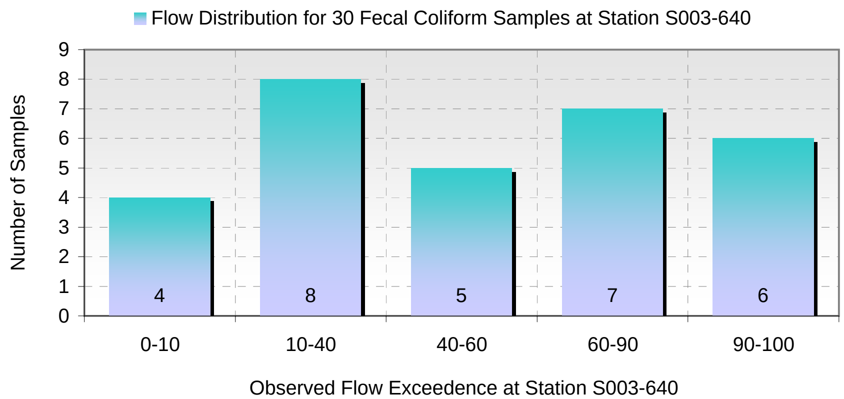


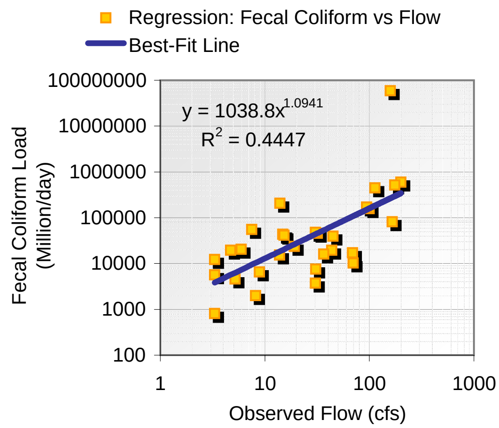
3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	31-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	4	167.86	8,213,853	53,390,046	84.6%
10-40	8	119.57	5,850,655	9,068,516	35.5%
40-60	5	32.09	1,570,002	113,744	0.0%
60-90	8	11.31	553,474	3,496,783	84.2%
90-100	6	4.23	206,831	33,513	0.0%

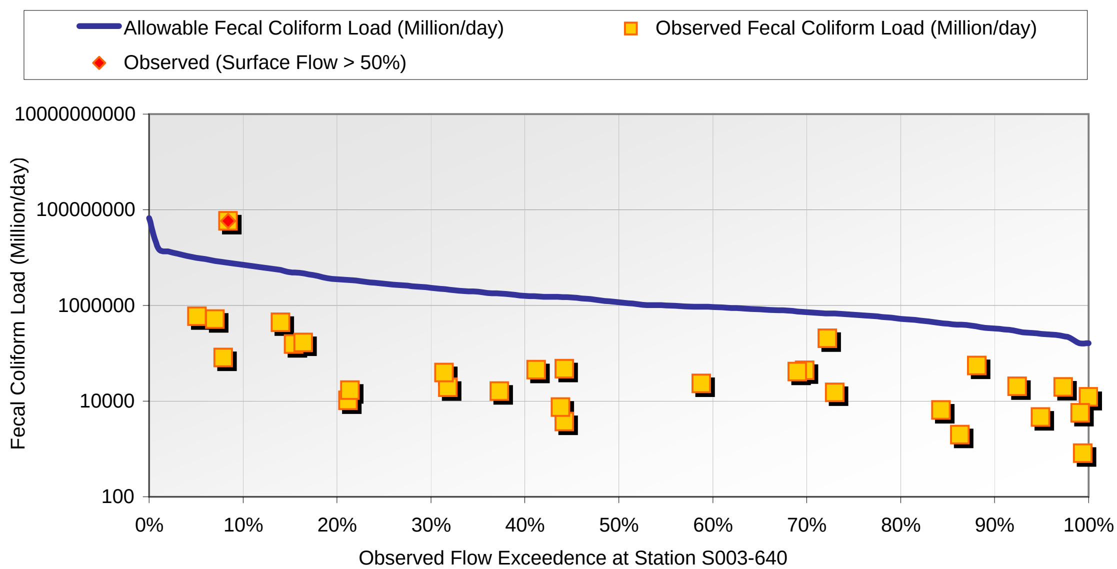
2,000 org/100ml Standard
[7: Fecal Coliform at Station S003-640 (MPN/100mL)] -vs- [7: Flow at Station S003-640 (cfs)]

1. Data Assessment and Trend Confirmation





2. Load Exceedence Analysis

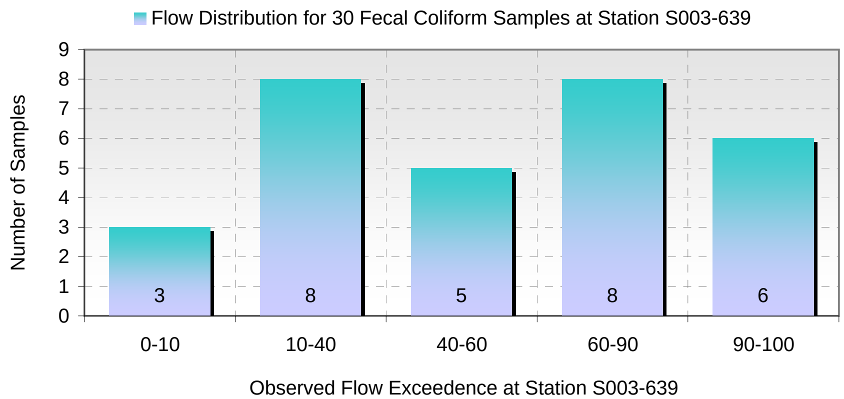


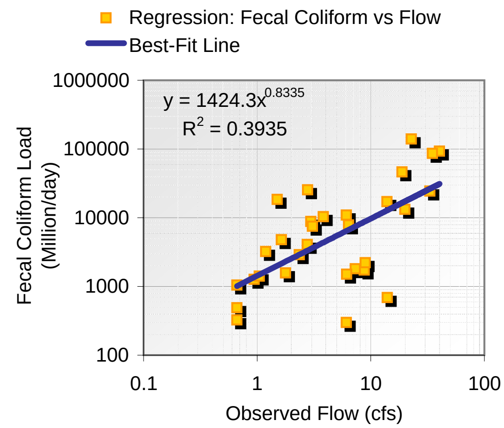
3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	30-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	4	159.24	7,791,943	58,439,572	86.7%
10-40	8	69.72	3,411,403	29,411	0.0%
40-60	5	30.44	1,489,358	23,457	0.0%
60-90	7	13.82	676,175	41,160	0.0%
90-100	6	4.01	196,207	8,925	0.0%

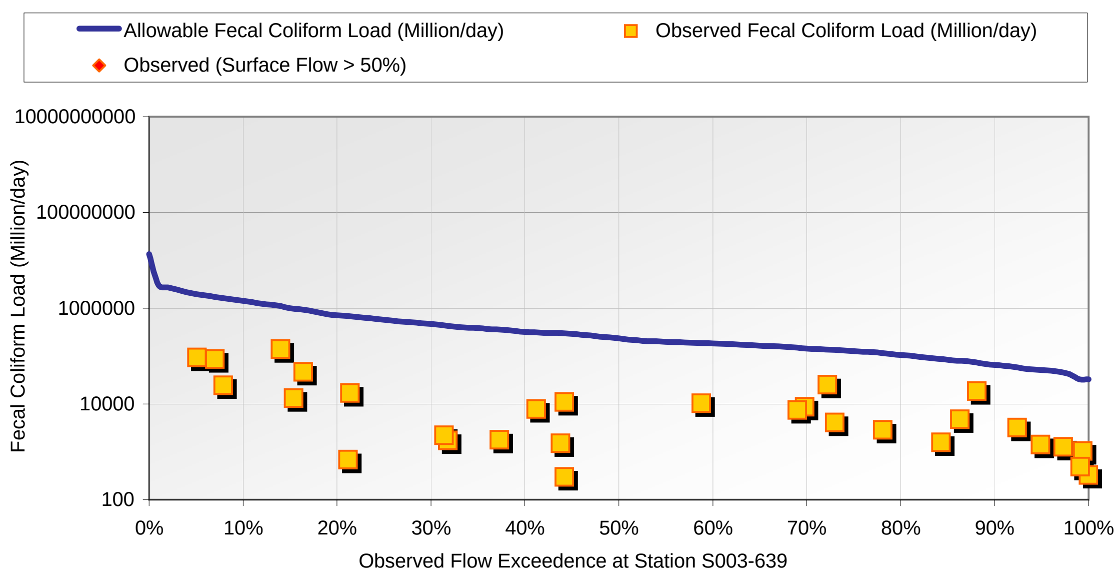
[2,000 org/100ml Standard](#)
[[9: Fecal Coliform at Station S003-639 (MPN/100mL)] -vs- [9: Flow at Station S003-639 (cfs)]](#)

1. Data Assessment and Trend Confirmation





2. Load Exceedence Analysis

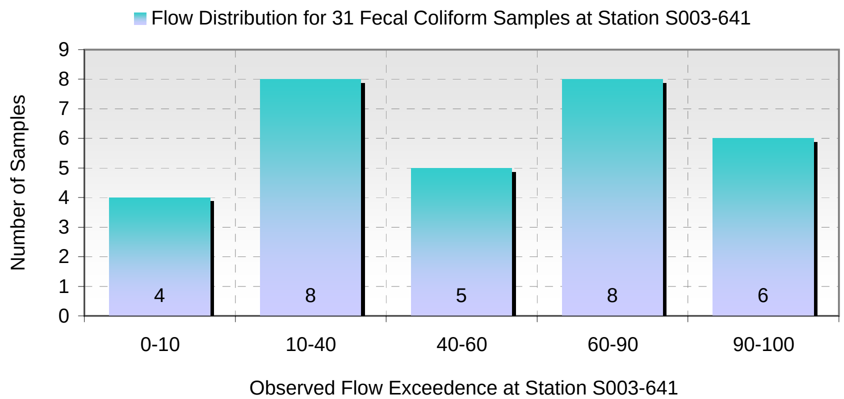


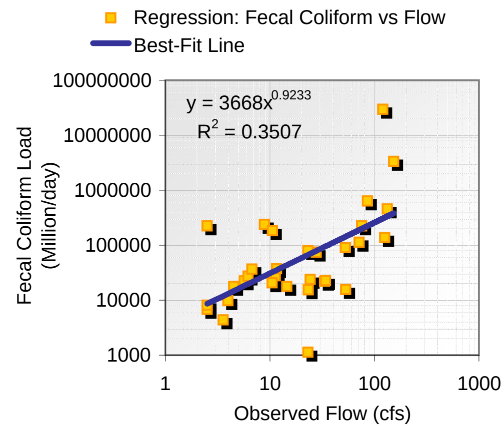
3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	30-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	3	35.08	1,716,348	85,817	0.0%
10-40	8	13.96	683,269	7,702	0.0%
40-60	5	6.10	298,303	7,823	0.0%
60-90	8	2.56	125,157	6,144	0.0%
90-100	6	0.80	39,298	1,154	0.0%

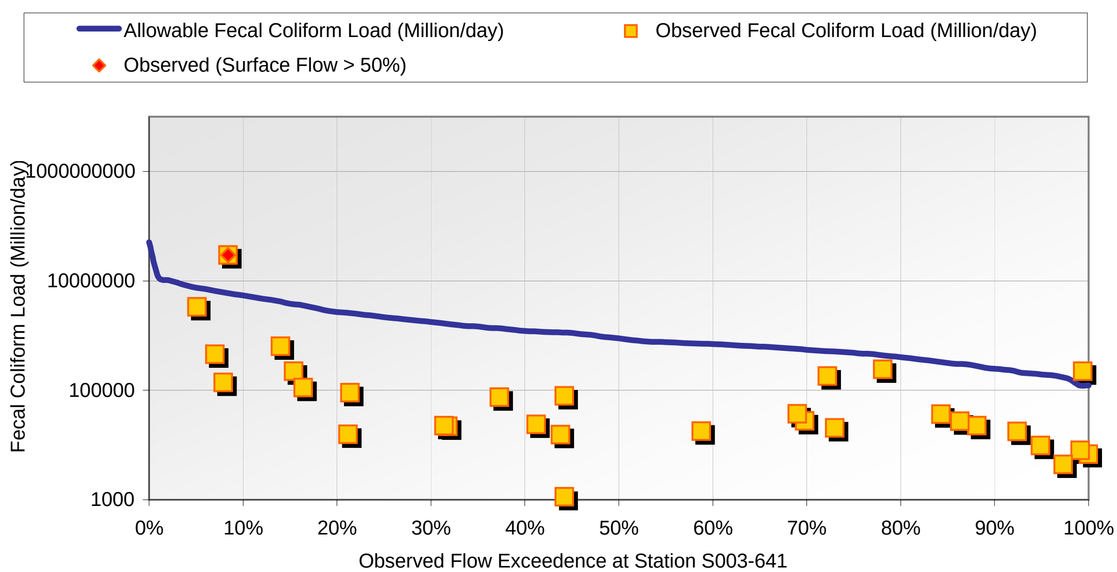
[2,000 org/100ml Standard](#)
[[8: Fecal Coliform at Station S003-641 (MPN/100mL)] -vs- [8: Flow at Station S003-641 (cfs)]](#)

1. Data Assessment and Trend Confirmation





2. Load Exceedence Analysis

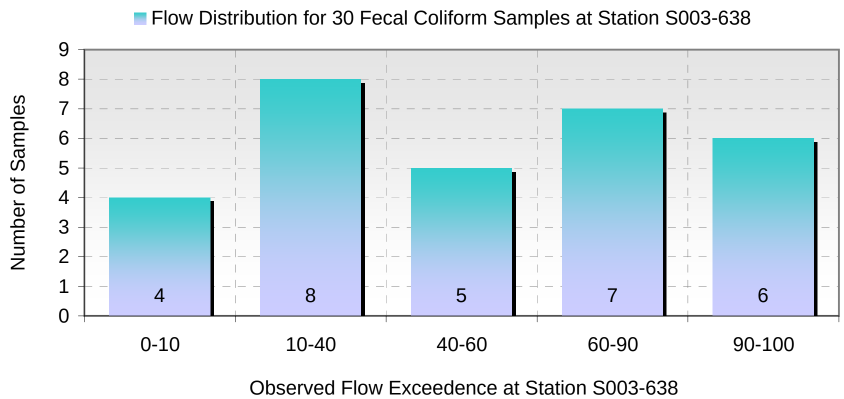


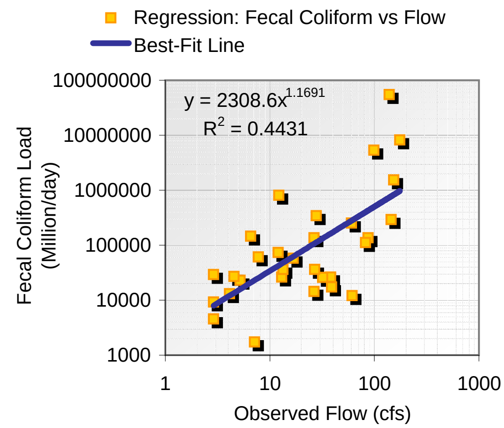
3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	31-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	4	121.09	5,925,094	29,625,469	80.0%
10-40	8	53.01	2,594,075	82,607	0.0%
40-60	5	23.15	1,132,527	17,837	0.0%
60-90	8	9.71	475,166	32,071	0.0%
90-100	6	2.52	123,392	222,105	44.4%

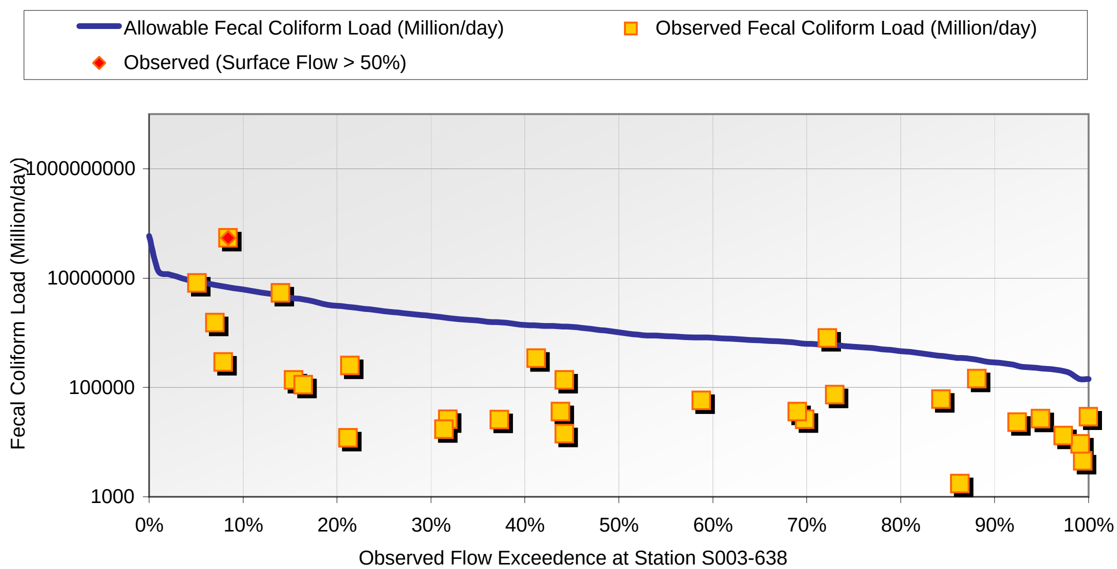
[2,000 org/100ml Standard](#)
[[4: Fecal Coliform at Station S003-638 (MPN/100mL)] -vs- [4: Flow at Station S003-638 (cfs)]](#)

1. Data Assessment and Trend Confirmation





2. Load Exceedence Analysis

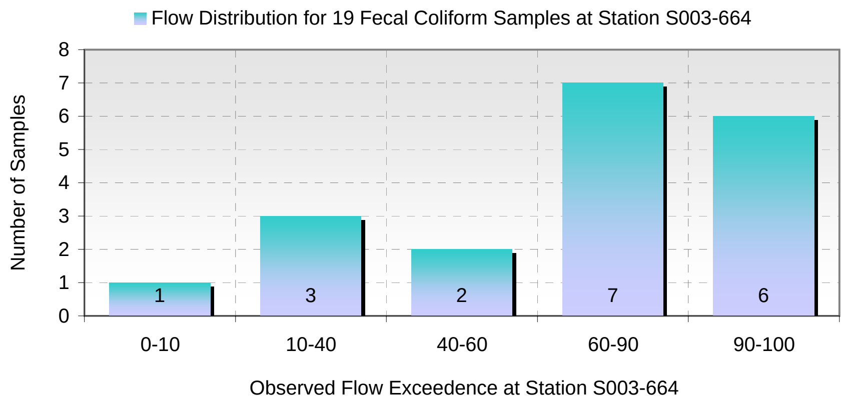


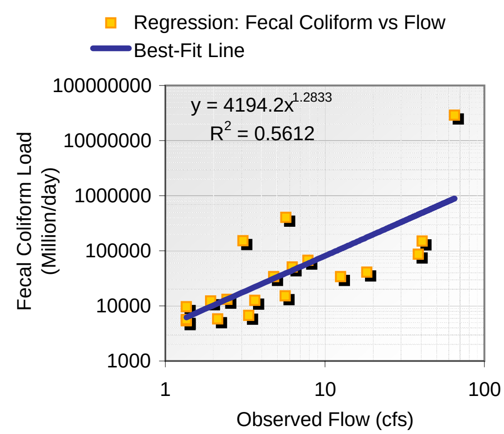
3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	30-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	4	139.17	6,809,836	54,478,691	87.5%
10-40	8	99.13	4,850,586	5,335,645	9.1%
40-60	5	26.60	1,301,637	57,401	0.0%
60-90	7	12.18	595,973	804,564	25.9%
90-100	6	3.50	171,477	18,015	0.0%

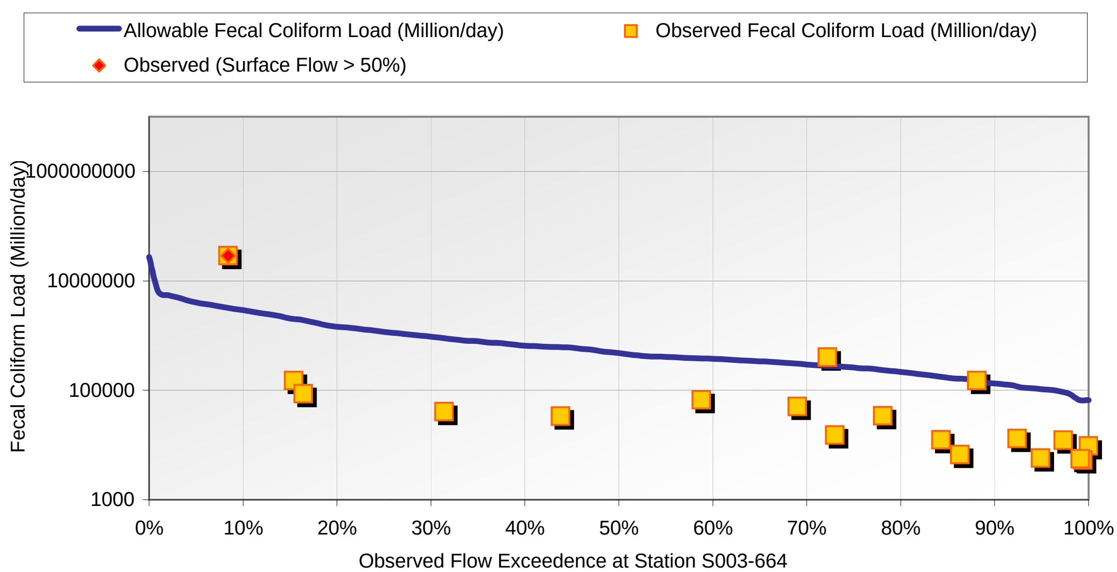
[2,000 org/100ml Standard](#)
[[2: Fecal Coliform at Station S003-664 (MPN/100mL)] -vs- [2: Flow at Station S003-664 (cfs)]](#)

1. Data Assessment and Trend Confirmation





2. Load Exceedence Analysis



3. Estimated TMDL Loads by Flow Exceedence Range

Flow Exceedence Ranges	19-Sample Distribution	Median Observed Flow (cfs)	Allowable Load (Million/day)	Median Load (Million/day)	Estimated Reduction (%)
0-10	1	65.12	3,186,202	28,675,816	88.9%
10-40	3	38.64	1,890,829	86,033	0.0%
40-60	2	10.19	498,801	50,454	0.0%
60-90	7	4.39	214,696	277,436	22.6%
90-100	6	1.64	80,231	7,684	0.0%

Appendix F. Best Management Practices, Costs, and Effectiveness

1 Best Management Practices, Costs, and Effectiveness

This section of the report provides an overview of BMPs that could be used to address the identified fecal coliform and sediment load reductions. Following approval of the TMDLs, a more detailed implementation plan should be developed by the local stakeholders with assistance from MPCA and using the results of the TMDL study.

1.1 Description of Best Management Practices (BMPs)

Controlling pollutant loading to the impaired reaches of the Groundhouse watershed will require implementation of various BMPs. This section describes BMPs that may be used to reduce loading of sediment, TSS, or fecal coliform from point source dischargers, onsite wastewater treatment systems, agricultural operations, and streambank erosion.

Although the TMDL focused on reducing loads of fecal coliform and sediment, reported reductions in nutrient, pesticide, and organic loading are presented as well. At some stations, poor biota scores were correlated to increases in nutrient concentrations. Reducing organic loads will improve dissolved oxygen concentrations which may be impacting ammonia levels. Organic loads are expressed as BOD_5, or the 5-day biochemical oxygen demand. Reducing pesticide loading will also benefit aquatic species.

The net costs associated with the BMPs described in this plan depend on the cost of construction (for structural BMPs), maintenance costs (seeding, grading, etc.), and operating costs (electricity, fuel, labor, etc). In addition, some practices require that land be taken out of farm production and converted to treatment areas, which results in a loss of income from the cash crop. Alternatively, taking land out of production does save money on future seed, fertilizer, labor, etc., and this must be accounted for as well. This section describes how the various costs apply to each BMP, and presents an estimate of the annualized cost spread out over the service life. Incentive plans and cost share programs are discussed separately.

Market prices can fluctuate significantly from year to year based on supply and demand factors, so applying straight rates of inflation to convert crop incomes from one year to the next is not appropriate. The costs presented in this section are discussed in year 2006 dollars because this is the latest year for which gross income estimates for corn and soybean production are available. Gross 2006 income estimates for corn and soybean in Minnesota are $484/ac and $279/ac, respectively. These values are based on gross incomes per bushel of $2.90 for corn and $6.10 for soybeans. Accounting for direct and overhead expenses results in net incomes from corn and soybean farms of $87/ac and $33/ac (University of Minnesota, 2007). The average net annual income of $60/ac was therefore used to estimate the annual loss from BMPs that take a portion of land out of farm production. The average value is considered appropriate since most farms operate on a 2-year crop rotation.

The cost to construct, maintain, and operate the BMPs is assumed to follow a yearly inflation rate of 3 percent since these components are not as dependent on such factors as weather and consumer demand. Therefore, all prices for BMP costs have been converted to year 2006 dollars to develop a net cost for each BMP. Inflated prices are rounded to the nearest quarter of a dollar since most of the costs were reported in whole dollars per acre, not dollars and cents.

1.1.1 Proper Maintenance of Onsite Wastewater Treatment Systems

The most effective BMP for managing loads from septic systems is regular maintenance. Unfortunately, many people do not think about their wastewater systems until a major malfunction occurs (e.g., sewage backs up into the house or onto the lawn). When not maintained properly, septic systems can cause the release of pathogens and excess nutrients into surface water. Good housekeeping measures relating to septic systems are listed below (CWP, 2004; University of Minnesota, 2006):

- Inspect system annually and pump the septic tank at least every three years.
- Refrain from trampling the ground or using heavy equipment above a septic system (to prevent collapse of pipes).
- Prevent septic system overflow by conserving water, not diverting storm drains or basement pumps into septic systems, and not disposing of trash through drains or toilets.

Education is a crucial component of reducing pollution from septic systems. Many owners are not familiar with USEPA recommendations concerning maintenance schedules. Education can occur through public meetings, mass mailings, and radio and television advertisements.

The USEPA recommends that septic tanks be pumped every 3 to 5 years depending on the tank size and number of residents in the household (2002b). Annual inspections, in addition to regular maintenance, ensure that systems are functioning properly. An inspection program would help identify those systems that are currently connected to tile drain systems. All tanks discharging to tile drainage systems should be disconnected immediately.

Some communities choose to formally regulate septic systems by creating a database of all the systems in the area. This database usually contains information on the size, age, and type of system. All inspections and maintenance records are maintained in the database through cooperation with licensed maintenance and repair companies. These databases allow the communities to detect problem areas and ensure proper maintenance. At this time, approximately 36 percent of systems in the watershed are permitted and registered in the Kanabec County database. It is not known what percent of systems are registered in Mille Lacs County.

1.1.1 Effectiveness

The reductions in pollutant loading resulting from improved operation and maintenance of all systems in the watershed depend on the wastewater characteristics and the level of failure present in the watershed. Reducing the level of failure to 0 percent may result in the following load reductions (Siegrist et al., 2000):

- TSS loads may be reduced by 90 percent
- Fecal coliform loads may be reduced by 99.99 percent
- Total phosphorus loads may be reduced by up to 100 percent
- BOD_5 loads may be reduced by 90 percent

1.1.1 Costs

Septic tanks are designed to accumulate sludge in the bottom portion of the tank while allowing water to pass into the drain field. If the tank is not pumped out regularly, the sludge can accumulate and eventually become deep enough to enter the drain field. Pumping the tank every three to five years prolongs the life of the system by protecting the drain field from solid material that may cause clogs and system back ups.

The cost to pump a septic tank in the Kanabec County area is typically $150 to $200 depending on how many gallons are pumped out and the disposal fee for the area. If a system is pumped once every three years, this expense averages out to less than $100 per year. Failure to properly maintain septic tanks may eventually damage drainfields leading to repairs or replacement costing from $2,000 to $10,000.

The cost of developing and maintaining a watershed-wide database of the onsite wastewater treatment systems in the watershed depends on the number of systems that need to be inspected. Based on Census data collected in 2000, there are approximately 1,125 households in the watershed. After the initial inspection of each system and creation of the database, only systems with no subsequent maintenance records would need to be inspected. A recent inspection program in South Carolina found that inspections cost approximately $170 per system (Hajjar, 2000).

Education of home and business owners that use onsite wastewater treatment systems should occur periodically. Public meetings; mass mailings; and radio, newspaper, and TV announcements can all be used to remind and inform owners of their responsibility to maintain their systems (Table 1). The costs associated with education and inspection programs will vary depending on the level of effort required to communicate the importance of proper maintenance and the number of systems in the area.

Table 1. Costs Associated with Maintaining and Replacing an Onsite Wastewater Treatment System

Action	Cost per System	Frequency	Annual Cost per System
Pumping	$150 to $200	Once every 3 years	$50 to $70
Inspection	$170	Initially all systems should be inspected, followed by 5 year inspections for systems not on record as being maintained	Up to $35, assuming all systems have to be inspected once every five years, which is not likely
Replacement	$2,000 to $10,000	With proper maintenance, system life should be 30 years	$70 to $350
Education	$1	Public reminders should occur once per year	$1

1.1.2 Conservation Tillage

Conservation tillage practices and residue management are commonly used to control erosion and surface transport of pollutants from fields used for crop production. The residuals from harvested crops not only provide erosion control, but also provide a nutrient source to growing plants, and continued use of conservation tillage results in a more productive soil with higher organic and nutrient content. Increasing the organic content of soil has the added benefit of reducing the amount of carbon in the atmosphere by storing it in the soil. Researchers estimate that croplands and pasturelands could be managed to trap 5 to 17 percent of the greenhouse gases produced in the United States (Lewandrowski et al., 2004).

Several practices are commonly used to maintain the suggested 30 percent residual surface cover:

- No-till systems disturb only a small row of soil during planting, and typically use a drill or knife to plant seeds below the soil surface.
- Strip till operations leave the areas between rows undisturbed, but remove residual cover above the seed to allow for proper moisture and temperature conditions for seed germination.
- Ridge till systems leave the soil undisturbed between harvest and planting: cultivation during the growing season is used to form ridges around growing plants. During or prior to the next planting, the top half to two inches of soil, residuals, and weed seeds are removed, leaving a relatively moist seed bed.
- Mulch till systems are any practice that results in at least 30 percent residual surface cover, excluding no-till and ridge till systems.

The NRCS provides additional information on these conservation tillage practices:

no-till and strip till:	http://efotg.nrcs.usda.gov/references/public/MN/329mn.pdf
ridge till:	http://efotg.nrcs.usda.gov/references/public/MN/346mn.pdf
mulch till:	http://efotg.nrcs.usda.gov/references/public/MN/345mn.pdf

Corn residues are more durable and capable of sustaining the required 30 percent cover required for conservation tillage. Soybeans generate less residue, the residue degrades more quickly, and supplemental measures or special care may be necessary to meet the 30 percent cover requirement (UME, 1996). Figure 1 shows a comparison of ground cover under conventional and conservation tillage practices.

Figure 1. Comparison of Conventional (left) and Conservation (right) Tillage Practices

Though no-till systems are more effective in reducing sediment loading from crop fields, they tend to concentrate phosphorus in the upper two inches of the soil profile due to surface application of fertilizer and decomposition of plant material (IAH, 2002; UME, 1996). This pool of phosphorus readily mixes with precipitation and can lead to increased concentrations of dissolved phosphorus in surface runoff. Chisel plowing may be required once every several years to reduce stratification of phosphorus in the soil profile.

1.1.2 Effectiveness

Czapar et al. (2006) summarize past and present tillage practices and their impacts on erosion control and nutrient delivery. Historically, the mold board plow was used to prepare the field for planting. This practice disturbed 100 percent of the soil surface and resulted in basically no residual material. Today, conventional tillage typically employs the chisel plow, which is not as disruptive to the soil surface and tends to leave a small amount of residue on the field (0 to 15 percent). Mulch till systems were classified as leaving 30 percent residue; percent cover was not quantified for the no-till systems in this study. The researchers used WEPP modeling to simulate changes in sediment and nutrient loading for these tillage practices. Relative to mold board plowing, chisel plowing reduced phosphorus loads leaving the field by 38 percent, strip tilling reduced loads by 80 percent, and no-till reduced loads by 85 percent. If chisel plowing is now considered conventional, then the strip till and no-till practices are capable of reducing phosphorus loads by 68 percent and 76 percent, respectively (Czapar et al., 2006).

USEPA (2003) reports the findings of several studies regarding the impacts of tillage practices on pesticide loading. Ridge till practices reduced pesticide loads by 90 percent and no-till reduced loads by an average of 67 percent. In addition, no-till reduced runoff losses by 69 percent, which will protect streambanks from erosion and loss of canopy cover (USEPA, 2003).

The reductions achieved by conservation tillage reported in these studies are summarized below:

- 68 to 76 percent reduction in total phosphorus
- 50 percent reduction in sediment for practices leaving 20 to 30 percent residual cover
- 90 percent reduction in sediment for practices leaving 70 percent residual cover
- 90 percent reduction in pesticide loading for ridge till practices
- 67 percent reduction in pesticide loading for no-till practices
- 69 percent reduction in runoff losses for no-till practices

1.1.2 Costs

Conservation tillage practices generally require fewer trips to the field, saving on labor, fuel, and equipment repair costs, although increased weed production may result in higher pesticide costs relative to conventional till (USDA, 1999). In general, conservation tillage results in increased profits relative to conventional tillage (Olson and Senjem, 2002; Buman et al., 2004; Czapar, 2006). The HRWCI (2005) lists the cost for conservation tillage at $0/ac.

Hydrologic inputs are often the limiting factor for crop yields and farm profits. Conservation practices reduce evaporative losses by covering the soil surface. USDA (1999) reports a 30 percent reduction in evaporative losses when 30 percent ground cover is maintained. Harman et al. (2003) and the Southwest Farm Press (2001) report substantial yield increases during dry years on farms managed with conservation or no-till systems compared to conventional till systems.

Depending on the type of equipment currently used, replacing conventional till equipment with no-till equipment can either result in a net savings or slight cost to the producer. Al-Kaisi et al. (2000) estimate that converting conventional equipment to no-till equipment costs approximately $1.25 to $2.25/ac/yr, but that for new equipment, purchasing no-till equipment is less expensive than conventional equipment.

Other researchers report a net gain when conventional equipment is sold to purchase no-till equipment (Harman et al., 2003).

Table 2 summarizes the available information for determining average annual cost for this BMP.

Table 2. Costs Calculations for Conservation Tillage

Item	Costs and Frequency	Annualized Costs (Savings)
Conversion of Conventional Equipment to Conservation Equipment	Costs presented in literature were already averaged out to yearly per acre costs: \$1.25/ac/yr to \$2.25/ac/yr	\$1.25/ac/yr to \$2.25/ac/yr
Operating Costs of Conservation Tillage Relative to Conventional Costs	\$0/ac/yr	\$0/ac/yr
Average Annual Costs		**\$1.25/ac/yr to \$2.25/ac/yr**

1.1.3 Cover Crops

Grasses and legumes may be used as winter cover crops to reduce soil erosion and improve soil quality (IAH, 2002). These crops also contribute nitrogen to the following crop, reducing fertilizer requirements. Grasses tend to have low seed costs and establish relatively quickly, but can impede cash crop development by drying out the soil surface or releasing chemicals during decomposition that may inhibit the growth of a following cash crop. Legumes take longer to establish, but are capable of fixing nitrogen from the atmosphere, thus reducing nitrogen fertilization required for the next cash crop. Legumes, however, are more susceptible to harsh winter environments and may not have adequate survival to offer sufficient erosion protection. Planting the cash crop in wet soil that is covered by heavy surface residue from the cover crop may impede emergence by prolonging wet, cool soil conditions. Cover crops should be killed off two or three weeks prior to planting the cash crop either by application of herbicide or mowing and incorporation, depending on the tillage practices used. Use of cover crops is illustrated in Figure 2.

(Photo Courtesy of NRCS)

Figure 2. Use of Cover Crops

The NRCS provides additional information on cover crops at:
http://efotg.nrcs.usda.gov/references/public/MN/340mn.pdf

1.1.3 Effectiveness

The effectiveness of cover crops in reducing pollutant loading has been reported by several agencies. In addition to these benefits, the reduction in runoff losses will reduce erosion from streambanks and allow for the establishment of vegetation and canopy cover. The reported reductions are listed below:

- 50 percent reduction in soil and runoff losses with cover crops alone. When combined with no-till systems, may reduce soil loss by more than 90 percent (IAH, 2002)
- 70 to 85 percent reduction in phosphorus loading on naturally drained fields (HRWCI, 2005)
- Reduction in fertilizer and pesticide requirements (OSUE, 1999)
- Useful in conservation tillage systems following low-residue crops such as soybeans (USDA, 1999)

1.1.3 Costs

The National Sustainable Agriculture Information Service recommends planting ryegrass after corn harvest and hairy vetch after soybeans (Sullivan, 2003). Both seeds can be planted at a depth of ¼ to ½ inch at a rate of 20 lb/ac or broadcast at a rate of 25 to 30 lb/ac (Ebelhar and Plumer, 2007; OSUE, 1990).

Researchers at Purdue University estimate the seed cost of ryegrass and hairy vetch at $13 and $32/ac, respectively. Savings in nitrogen fertilizer (assuming nitrogen fertilizer cost of $0.30/lb (Sample, 2007)) are $4/ac for ryegrass and $30/ac for hairy vetch. Herbicide application is estimated to cost $15/ac. Yield increases in the following crop, particularly during droughts, are reported at 10 percent and are expected to offset the cost of this practice (Mannering et al., 1998).

Accounting for the seed cost, herbicide cost, and fertilizer offset results in an average net cost of approximately $20.50/ac assuming that cover crop planting recommendations for a typical 2-year corn/soybean rotation are followed (Mannering et al., 1998). These costs do not account for yield increases which may offset these costs completely. Table 3 summarizes the costs and savings associated with ryegrass and hairy vetch.

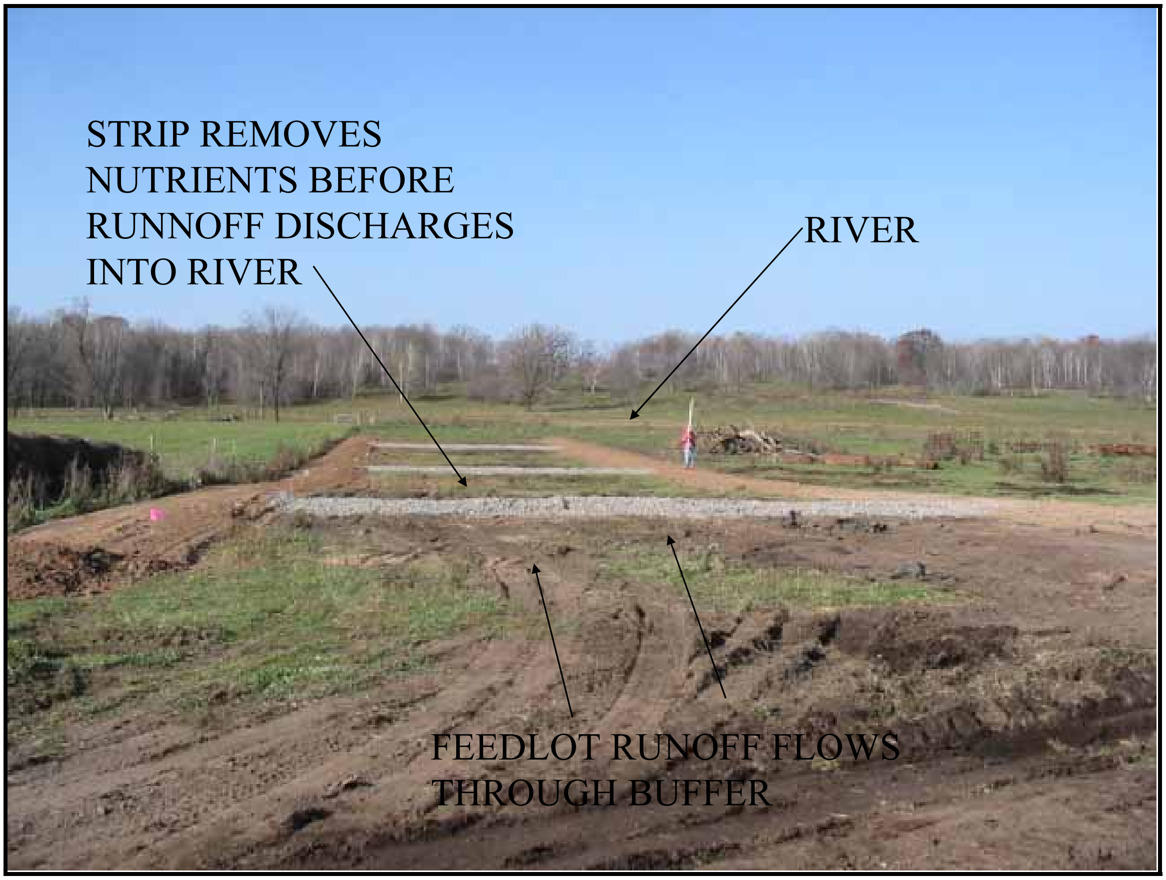
Table 3. Costs Calculations for Cover Crops

Item	Ryegrass	Hairy Vetch
Seed Costs	$13/ac	$32/ac
Nitrogen Fertilizer Savings	($4/ac)	($30/ac)
Herbicide Costs	$15/ac	$15/ac
Annual Costs	$24/ac	$17/ac
Average Annual Cost Assuming Ryegrass Follows Corn and Hairy Vetch Follows Soybeans: $20.50/ac		

1.1.4 Filter Strips

Filter strips are used in agricultural and urban areas to intercept and treat runoff before it leaves the site. If topography allows, filter strips may also be used to treat effluent from tile drain outlets. For small dairy operations, filter strips may be used to treat milk house washings and runoff from the open lot (NRCS, 2003).

Filter strips will require maintenance, including grading and seeding, to ensure distributed flow across the filter and protection from erosion. Periodic removal of vegetation will encourage plant growth and uptake and remove nutrients stored in the plant material. Filter strips are most effective on sites with mild slopes of generally less than 5 percent, and to prevent concentrated flow, the upstream edge of a filter strip should follow one elevation contour (NCDENR, 2005). A filter strip at a feedlot adjacent to the Groundhouse River in Kanabec County is shown in Figure 3.



(Photo Courtesy of KCSWCD)

Figure 3. Grass Filter Strip Protecting Stream from Adjacent Feedlot

The NRCS provides additional information on filter strips at:
http://efotg.nrcs.usda.gov/references/public/MN/393mn.pdf

Filter strips also serve to reduce the quantity and velocity of runoff, which should reduce erosive forces to receiving stream channels. Filter strip sizing is dependent on site specific features such as climate and topography, but at a minimum, the area of a filter strip should be no less than 2 percent of the drainage area for agricultural land (OSUE, 1994). The minimum filter strip width suggested by NRCS (2002) is 30 ft. The strips are assumed to function properly with annual maintenance for 30 years before requiring replacement of soil and vegetation.

1.1.4 Effectiveness

Filter strips have been found to effectively remove pollutants from agricultural runoff. The following reductions are reported in the literature (USEPA, 2003; Kalita, 2000; Woerner et al., 2006):

- 65 percent reduction in sediment
- 55 to 87 percent reduction in fecal coliform
- 11 to 100 percent reductions for atrazine
- 65 percent reductions for total phosphorus
- Slows runoff velocities and may reduce runoff volumes via infiltration

1.1.4 Costs

Filter strips cost approximately $0.34 per sq ft to construct, and the system life is typically assumed 20 years (Weiss et al., 2007). Assuming that the required filter strip area is 2 percent of the area drained (OSUE, 1994) translates to 870 square feet of filter strip required for each acre of agricultural land treated. The construction cost to treat one acre of land is therefore $296/ac. The annualized construction costs are $14.75/ac/yr. Annual maintenance of filter strips is estimated to cost $0.01 per sq ft (USEPA, 2002c) for an additional cost of $8.75/ac/yr of agricultural land treated (rounded to the nearest quarter). In addition, the area converted from agricultural production to filter strip will result in a net annual income loss of $1.25. Table 4 summarizes the costs assumptions used to estimate the annualized cost to treat one acre of agricultural drainage with a filter strip.

Table 4. Costs Calculations for Filter Strips Used in Crop Production

Item	Costs Required to Treat One Acre of Agricultural Land with Filter Strip
Construction Costs	$296
System Life (years)	20
Annualized Construction Costs	$14.75
Annual Maintenance Costs	$8.75
Annual Income Loss	$1.25
Average Annual Costs	**$24.75/ac treated**

Filter strips used in animal operations typically treat contaminated runoff from pastures or feedlot areas or washings from the milk houses of small dairy operations (NRCS, 2003). The NRCS (2003) costs for small dairy operations (75 milk cows) assume a filter strip area of 12,000 sq ft is required. Similarly, for the pasture operations, it is assumed that a filter strip area of 12,000 sq ft (30 ft wide and 400 ft long) would be required to treat runoff from a herd of 50 cattle (NRCS, 2003). The document does not explain why more animals can be treated by the same area of filter strip at the dairy operation compared to the pasture operation.

For animal operations, it is not likely that land used for growing crops would be taken out of production for conversion to a filter strip. Table 5 summarizes the capital, maintenance, and annualized costs for filter strips per head of animal.

Table 5. Costs Calculations for Filter Strips Used at Animal Operations

Operation	Capital Costs per Head	Annual Operation and Maintenance Costs per Head	Total Annualized Costs per Head
Small dairy (75 milking cows)	$51 per head of cattle	$1.50 per head of cattle	$4 per head of cattle
Beef or other (50 cattle)	$76 per head of cattle	$2.50 per head of cattle	$6 per head of cattle

1.1.5 Grassed Waterways

Grassed waterways are stormwater conveyances lined with grass that prevent erosion of the transport channel. They are often used to divert clean up-grade runoff around contaminated feedlots and manure storage areas (NRCS, 2003). In addition, the grassed channel reduces runoff velocities, allows for some infiltration, and filters out some particulate pollutants. A grassed waterway providing surface drainage for a corn field is shown in Figure 4.

(Photo Courtesy of NRCS)

Figure 4. Grassed Waterway

The NRCS provides additional information on grassed waterways at:
http://efotg.nrcs.usda.gov/references/public/MN/412mn.pdf

1.1.5 Effectiveness

The effectiveness of grass swales for treating agricultural runoff has not been quantified. The Center for Watershed Protection reports the following reductions in urban settings (Winer, 2000):

- 5 percent reduction in fecal coliform
- 68 percent reduction of total suspended solids
- 30 percent reduction in total phosphorus
- May reduce runoff volumes via infiltration

In addition, grassed waterways that allow for water infiltration may reduce atrazine loads by 25 to 35 percent (Kansas State University, 2007).

1.1.5 Costs

Grassed waterways cost approximately $0.56 per sq ft to construct (USEPA, 2002c). These stormwater conveyances are best constructed where existing bare ditches transport stormwater, so no income loss from land conversion is expected with this practice. It is assumed that the average area required for a grassed waterway is approximately 0.1 to 0.3 percent of the drainage area, or between 44 and 131 sq ft per acre. The range is based on examples in the Illinois Drainage Guide, information from the NRCS Engineering Field Handbook, and a range of waterway lengths (100 to 300 feet). Waterways are assumed to remove pollutants effectively for 20 years before soil, vegetation, and drainage material need to be replaced (Weiss et al., 2007). The construction costs spread out over the life of the waterway range from $1.25 to $3.75/year for each acre of agriculture draining to a grassed waterway. Annual maintenance of grassed waterways is estimated at $0.02 per sq ft (Rouge River, 2001) for an additional cost ranging from $1 to $2.75/year. Table 6 summarizes the annual costs assumptions for grassed waterways.

Table 6. Costs Calculations for Grassed Waterways Draining Cropland

Item	Costs Required to Treat One Acre of Agricultural Land
Construction Costs	$24.75 to $73.25
System Life (years)	20
Annualized Construction Costs	$1.25 to $3.75
Annual Maintenance Costs	$1 to $2.75
Annual Income Loss	$0
Average Annual Costs	**$2.25 to 6.50/ac treated**

Grassed waterways are primarily used in animal operations to divert clean water away from pastures, feedlots, and manure storage areas. Table 7 summarizes the capital, maintenance, and annualized costs of this practice per head of cattle as summarized by NRCS (2003).

Table 7. Costs Calculations for Grassed Waterways Used in Cattle Operations

Capital Costs per Head	Annual Operation and Maintenance Costs per Head	Total Annualized Costs per Head
$0.50 to $1.50	$0.02 to $0.04	$0.05 to $0.12

1.1.6 Riparian Buffers

Riparian corridors, including both the stream channel and adjacent land areas, are important components of watershed ecology. The streamside forest slowly releases nutrients as twigs and leaves decompose. These nutrients are valuable to the fungi, bacteria, and invertebrates that form the basis of a stream's food chain. Tree canopies of riparian forests also cool the water in streams which can affect the composition of the fish species in the stream, the rate of biological reactions, and the amount of dissolved oxygen the water can hold. Channelization or widening of streams moves the canopy farther apart, decreasing the amount of shaded water surface, increasing water temperatures, and decreasing dissolved oxygen concentrations.

Preserving natural vegetation along stream corridors can effectively reduce water quality degradation associated with human disturbances. The root structure of the vegetation in a buffer enhances infiltration of runoff and subsequent trapping of nonpoint source pollutants. However, the buffers are only effective in this manner when the runoff enters the buffer as a slow moving, shallow "sheet"; concentrated flow in a ditch or gully will quickly pass through the buffer offering minimal opportunity for retention and uptake of pollutants.

Even more important than the filtering capacity of the buffers is the protection they provide to streambanks. The rooting systems of the vegetation serve as reinforcements in streambank soils, which help to hold streambank material in place and minimize erosion. Riparian buffers also prevent cattle access to streams, reducing streambank trampling and defecation in the stream. Due to the increase in stormwater runoff volume and peak rates of runoff associated with agriculture and development, stream channels are subject to greater erosive forces during stormflow events. Thus, preserving natural vegetation along stream channels minimizes the potential for water quality and habitat degradation due to streambank erosion and enhances the pollutant removal of sheet flow runoff from developed areas that pass through the buffer. A riparian buffer protecting the stream corridor from adjacent agricultural areas is shown in Figure 5.

(Photo Courtesy of NRCS)

Figure 5. Riparian Buffer Between Stream Channel and Agricultural Areas

The NRCS provides additional information on riparian buffers at:
http://efotg.nrcs.usda.gov/references/public/MN/390mn.pdf *and*
http://efotg.nrcs.usda.gov/references/public/MN/391mn.pdf

1.1.6 Effectiveness

Riparian buffers should consist of native species and may include grasses, grass-like plants, forbs, shrubs, and trees. Minimum buffer widths of 25 feet or more are required for water quality benefits. Higher removal rates are provided with greater buffer widths. Riparian corridors typically treat a maximum of 300 ft of adjacent land before runoff forms small channels that short circuit treatment. Buffer widths based on slope measurements and recommended plant species should conform to NRCS Field Office Technical Guidelines. The following reductions are reported in the literature:

- 70 to 90 percent reduction of sediment (NCSU, 2002)
- 34 to 74 percent reduction of fecal coliform for 30 ft wide buffers (Wenger, 1999)
- 87 percent reduction of fecal coliform for 200 ft wide buffers (Wenger, 1999)
- 25 to 30 percent reduction of total phosphorus for 30 ft wide buffers (NCSU, 2002)
- 70 to 80 percent reduction of total phosphorus for 60 to 90 ft wide buffers (NCSU, 2002)
- 62 percent reduction in BOD_5 for 200 ft wide buffers (Wenger, 1999)
- 80 to 90 percent reduction of atrazine (USEPA, 2003)
- Increased canopy cover provides shading which may reduce water temperatures and improve dissolved oxygen concentrations (NCSU, 2002). Wenger (1999) suggests buffer width of at least 30 ft to maintain stream temperatures.
- Increased channel stability will reduce streambank erosion

1.1.6 Costs

Restoration of riparian areas costs approximately $106/ac to construct and $500/ac to maintain over the life of the buffer (Wossink and Osmond, 2001; NCEEP, 2004). Maintenance of a riparian buffer should be minimal, but may include items such as periodic inspection of the buffer, minor grading to prevent short circuiting, and replanting/reseeding dead vegetation following premature death or heavy storms. Assuming a buffer width of 90 ft on either side of the stream channel and an adjacent treated width of 300 ft of agricultural land, one acre of buffer will treat approximately 3.3 acres of adjacent agricultural land. The cost per treated area is thus $32/ac to construct and $151/ac to maintain over the life of the buffer. Assuming a system life of 30 years and an annual income loss of $18 for the area converted to a buffer results in an annualized cost of $24/yr for each acre of agricultural land treated (Table 8). The majority of this cost would represent income loss in the event that land was taken out of production to maintain and install a buffer.

Table 8. Costs Calculations for Riparian Buffers

Item	Costs Required to Treat One Acre of Agricultural Land
Costs per Acre of Riparian Buffer	
Construction Costs	$106
Maintenance Costs Over System Life	$500
Costs to Treat One Acre of Agricultural Land (assuming 0.3 ac of buffer)	
Construction Costs	$32
Maintenance Costs Over System Life	$151
System Life (years)	30
Annualized Construction Costs	$1
Annualized Maintenance Costs	$5
Annual Income Loss	$18
Average Annual Costs	**$24/ac treated**

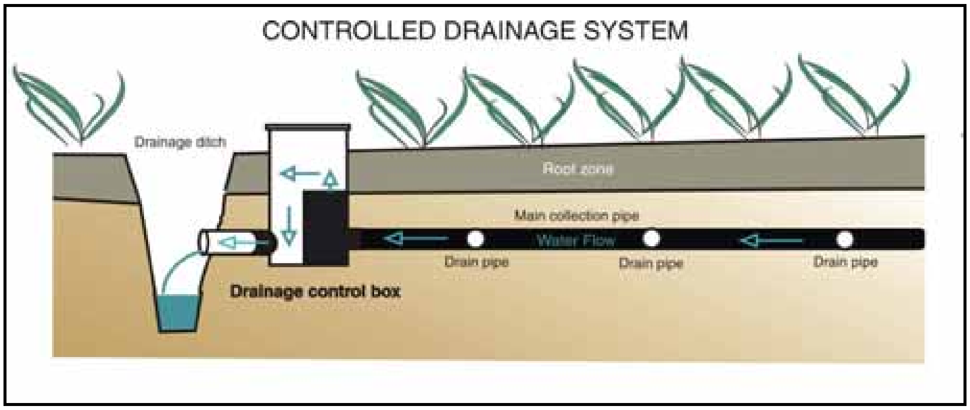
Restoration of riparian areas will protect the stream corridor from cattle trampling and reduce the amount of fecal material entering the channel. The cost of this BMP depends on the length of channel to be protected, not the number of animals having channel access. Fecal coliform reductions have been reported for buffers at least 30 ft wide (Wenger, 1999). Larger reductions are reported for 200 ft wide buffers. The costs per length of channel for 30 ft and 200 ft wide buffers restored on both sides of a stream channel are listed in Table 9 based on the per acre construction and maintenance costs suggested by Wossink and Osmond (2001) and NCEEP (2004). A system life of 30 years is assumed.

Table 9. Costs Calculations for Riparian Buffers Per Foot of Channel

Width	Capital Costs per ft	Annual Operation and Maintenance Costs per ft	Total Annualized Costs per ft
30 ft on both sides of channel	$0.15	$0.02	$0.03
60 ft on both sides of channel	$0.30	$0.04	$0.05
90 ft on both sides of channel	$0.45	$0.06	$0.08
200 ft on both sides of channel	$1.00	$0.14	$0.17

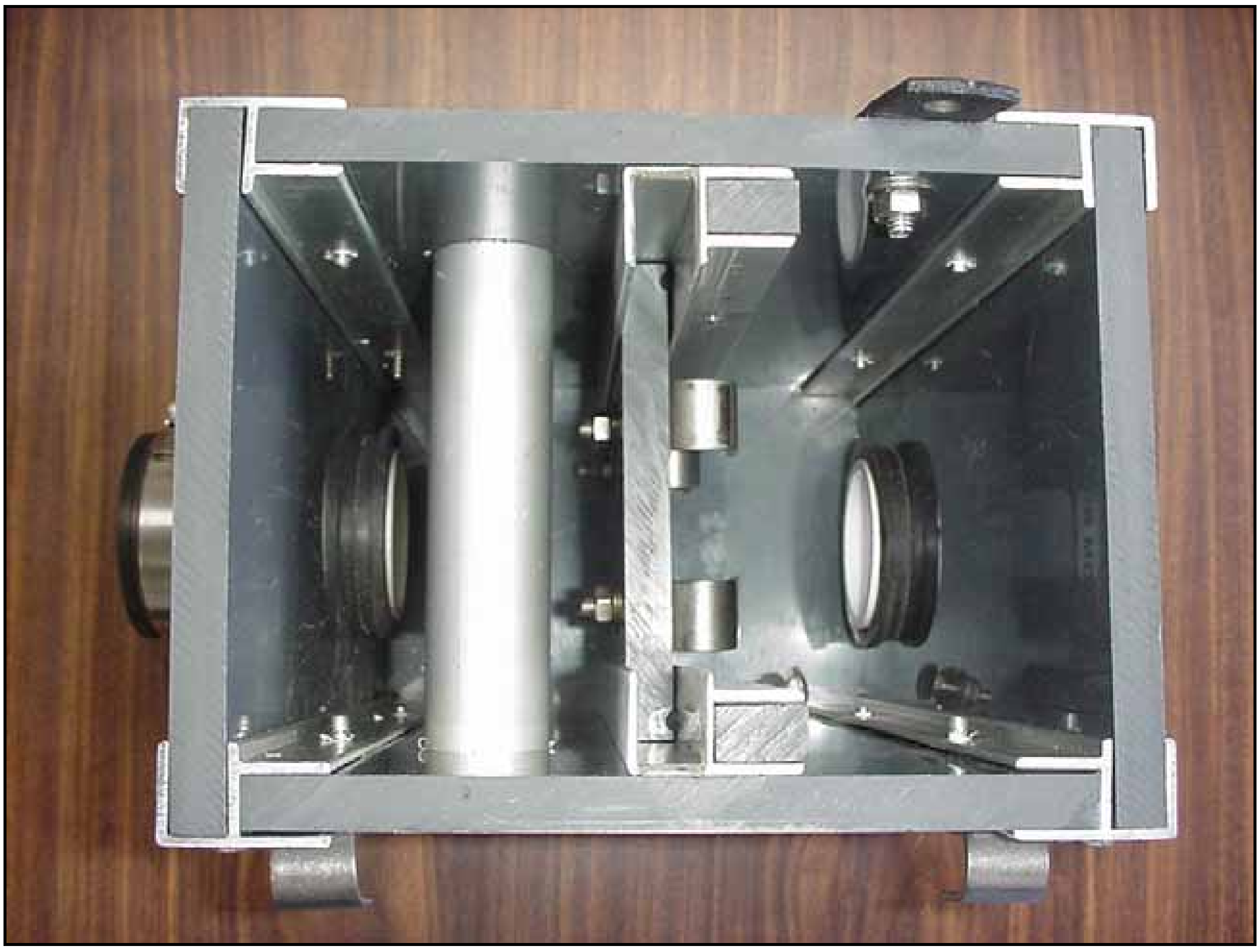
1.1.7 Controlled Drainage

A conventional tile drain system collects infiltrated water below the root zone and transports the water quickly to a down-gradient surface outlet. Placement of a water-level control structure at the outlet (Figure 6 and Figure 7) allows for storage of the collected water to a predefined elevation. The stored water becomes a source of moisture for plants during dry conditions and undergoes biological, chemical, and physical processes that result in lower nutrient concentrations in the final effluent. Installation of outlet control structures can also be used to plug old farm ditches and restore wetland areas.



(Illustration Courtesy of the Agricultural Research Service Information Division)

Figure 6. Controlled Drainage Structure for a Tile Drain System



(Photo Courtesy of CCSWCD)

Figure 7. Interior View of a Drainage Control Structure with Adjustable Baffle Height

The NRCS provides additional information on water level controls at:
http://efotg.nrcs.usda.gov/references/public/MN/587mn.pdf.

1.1.7 Effectiveness

Use of control structures on conventional tile drain systems in the coastal plains has resulted in the reduction of total phosphorus loading by 35 percent (Gilliam et al., 1997). Researchers at the University of Illinois also report reductions in phosphorus loading with tile drainage control structures. Concentrations of phosphate were reduced by 82 percent, although total phosphorus reductions were not quantified in this study (Cooke, 2005). Going from a surface draining system to a tile drain system with outlet control reduces phosphorus loading by 65 percent (Gilliam et al., 1997).

Storage of tile drain water for later use via subsurface irrigation has shown decreases in dissolved phosphorus loading of approximately 50 percent (Tan et al., 2003). However, accumulated salts in reuse water may eventually exceed plant tolerance and result in reduced crop yields. Mixing stored drain water with fresh water or alternating irrigation with natural precipitation events will reduce the negative impacts of reuse. Salinity thresholds for each crop should be considered and compared to irrigation water concentrations.

1.1.7 Costs

Tile mapping services using color infrared photography are available to assist farmers in identifying the exact location of their tile drain lines for approximately $2.25/ac. Cooke (2005) estimates that the cost of retrofitting tile drain systems with outlet control structures ranges from $21 to $42 per acre. Construction of new tile drain systems with outlet control is approximately $80/ac. The yield increases associated with installation of tile drain systems are expected to offset the cost of installation (Cooke, 2005). It is assumed that outlet control structures have a system life of 30 years. Cost assumptions for retrofitting existing systems and installing new tile drain systems with outlet control devices are summarized in Table 10.

Table 10. Costs Calculations for Outlet Control Devices on Tile Drain Systems

Item	Costs to Retrofit Existing Systems	Costs to Install a New System
Mapping Costs per Acre	$2.25	$0
Construction Costs	$21 to $42/ac	$80/ac
System Life (years)	30	30
Average Annual Costs	**$0.75 to $1.50/ac treated**	**$2.50/ac treated**

1.1.8 Wetland Restoration

Wetland restoration is appropriate for areas that were historically functioning as a wetland environment, but were altered to accommodate other land uses, such as agriculture. Because wetlands are typically located between upland areas and receiving streams or rivers, they serve as natural filters for pollutants such as sediment and nutrients. They also provide habitat for wildlife and reduce peak flows in stream channels by storing flood waters. Natural or restored wetland areas should not be used to treat point or nonpoint pollution (USEPA, 2003); constructed wetlands may be created for this purpose, as discussed in Section 1.1.7 of the Groundhouse River TMDL Report. Figure 8 shows a restored wetland supporting wildlife in Iowa.

Wetland restoration must include the rehabilitation of the soils, hydrology, and vegetation to a natural condition. Practices to consider include constructing embankments or dikes, plugging drainage ditches, removing tile drain lines, or installing outlet control devices on existing tile systems (Section 1.1.7).

(Photo Courtesy of NRCS)

Figure 8. Restored Wetland Providing Wildlife Habitat

The NRCS provides additional information on wetland restoration at:
http://efotg.nrcs.usda.gov/references/public/MN/mn651.pdf.

Design recommendations for Minnesota are discussed here:
http://www.bwsr.state.mn.us/wetlands/publications/nativewetveg.pdf.

1.1.8 Effectiveness

Pollutant removal efficiencies of restored wetlands have not been quantified. This is likely because natural and restored wetland systems are not intended to treat high-pollutant wastewater. In general, restored wetlands will offer some reductions of nutrients and sediment and will also protect downstream stream channels from the erosive forces of high peak flows.

1.1.8 Costs

The Kanabec County SWCD reports that project costs for restored wetlands typically range from $5,000 to $8,000 per project.

1.1.9 Constructed Wetlands

Constructed wetlands used to treat animal wastes are typically surface flowing systems comprised of cattails, bulrush, and reed plants. Prior to treating animal waste in a constructed wetland, storage in a lagoon or pond is required to protect the wetland from high pollutant loads that may kill the vegetation or clog pore spaces. After treatment in the wetland, the effluent is typically held in another storage lagoon and then land applied (USEPA, 2002a). Alternatively, the stored effluent can be used to supplement flows to the wetland during dry periods. Constructed wetlands that ultimately discharge to a surface waterbody will require a permit, and the receiving stream must be capable of assimilating the effluent during low flow conditions (NRCS, 2002). Figure 9 shows an example of a lagoon-wetland system.

(Photo courtesy of USDA NRCS.)

Figure 9. Constructed Wetland System for Animal Waste Treatment

The NRCS provides additional information on constructed wetlands at
http://efotg.nrcs.usda.gov/references/public/MN/658mn.pdf
and
ftp://ftp.wcc.nrcs.usda.gov/downloads/wastemgmt/NEH637Ch3ConstructedWetlands.pdf

1.1.9 Effectiveness

Wetland environments treat wastewater through sedimentation, filtration, plant uptake, biochemical transformations, and volatilization. Reported pollutant reductions found in the literature are listed below:

- 92 percent reduction in fecal coliform (USEPA, 2002a)

- 53 to 81 percent reduction in total suspended solids (USEPA, 2002a)
- 42 percent reduction in total phosphorus (USEPA, 2003)
- 59 to 80 percent reduction in BOD_5 (USEPA, 2002a)
- 50 percent reduction in atrazine in wetlands with a retention time of 35 days (Moore, 1999)

1.1.9 Costs

Researchers of the use of constructed wetlands for animal waste management generally agree that these systems are a lower cost alternative compared to conventional treatment and land application technologies. Few studies, however, actually report the costs of constructing and maintaining these systems. A Canadian study (CPAAC, 1999) evaluated the use of a constructed wetland system for treating milk house washings as well as contaminated runoff from the feedlot area and manure storage pile of a dairy operation containing 135 head of cattle. The treatment system was comprised of a pond/wetland/pond/wetland/filter strip treatment train that cost $523 per head to construct. Annual operating and maintenance costs of $7.25 per head include electricity to run pumps, maintenance of pumps and berms, and dredging the wetland cells once every 10 years. Reductions in final disposal costs due to reduced phosphorus content of the final effluent were $22 per head.

Another study evaluated the use of constructed wetlands for treatment of a 3,520-head swine operation in North Carolina. Waste removal from the swine facility occurs via slatted floors to an underlying pit that is flushed once per week. This new treatment system incorporated a settling basin, constructed wetland, and storage pond treatment system prior to land application or return to the pit for flushing.

Capital and maintenance costs reported in the literature for dairy and swine operations are summarized per head in Table 11 assuming a system life of 30 years. No example studies including costs were available for beef cattle operations, which should generate less liquid waste than the other two operations. It would therefore be expected that constructing a wetland for beef cattle operation would cost less than for a dairy or swine operation.

Table 11. Costs Calculations for Constructed Wetlands

Example	Capital Costs per Head	Annual Operation and Maintenance Costs per Head	Total Annualized Costs per Head
Dairy farm	$522	-$14.75	$2.50
Swine operation	$110	$1.00	$4.50

1.1.10 Sedimentation Basins

Sedimentation basins are used to settle out sediment and any attached pollutants before runoff leaves the field. They can also be used to protect wetlands from high levels of pollutant loading that may disturb the flow and water quality functions of the system. Basins need to be dredged periodically when the sediment storage capacity is full. A sedimentation basin constructed in Kanabec County is shown in Figure 10.

(Photo Courtesy of KCSWCD)

Figure 10. Sedimentation Basin

The NRCS provides additional information on sedimentation basins at:
http://efotg.nrcs.usda.gov/references/public/MN/350mn.pdf.

1.1.10 Effectiveness

Sediment control structures offer the following pollutant reduction benefits (Winer, 2000):

- Fecal coliform reductions of 70 to 78 percent
- Sediment reductions of 47 to 80 percent
- Total phosphorus reductions of 19 to 51 percent

1.1.10 Costs

Not many studies report the costs of constructing and maintaining sedimentation basins draining agricultural areas. Data from a study in Indiana (Richardson, 2003) indicate that the annual costs of constructing and maintaining sedimentation basins is $33.50 per acre of agriculture per year. The local project shown in Figure 10 was constructed for $10,850 and treats 23 acres of crop land. Assuming a system life of 20 years, the annualized construction costs are approximately $24/ac/yr (this does not include maintenance costs).

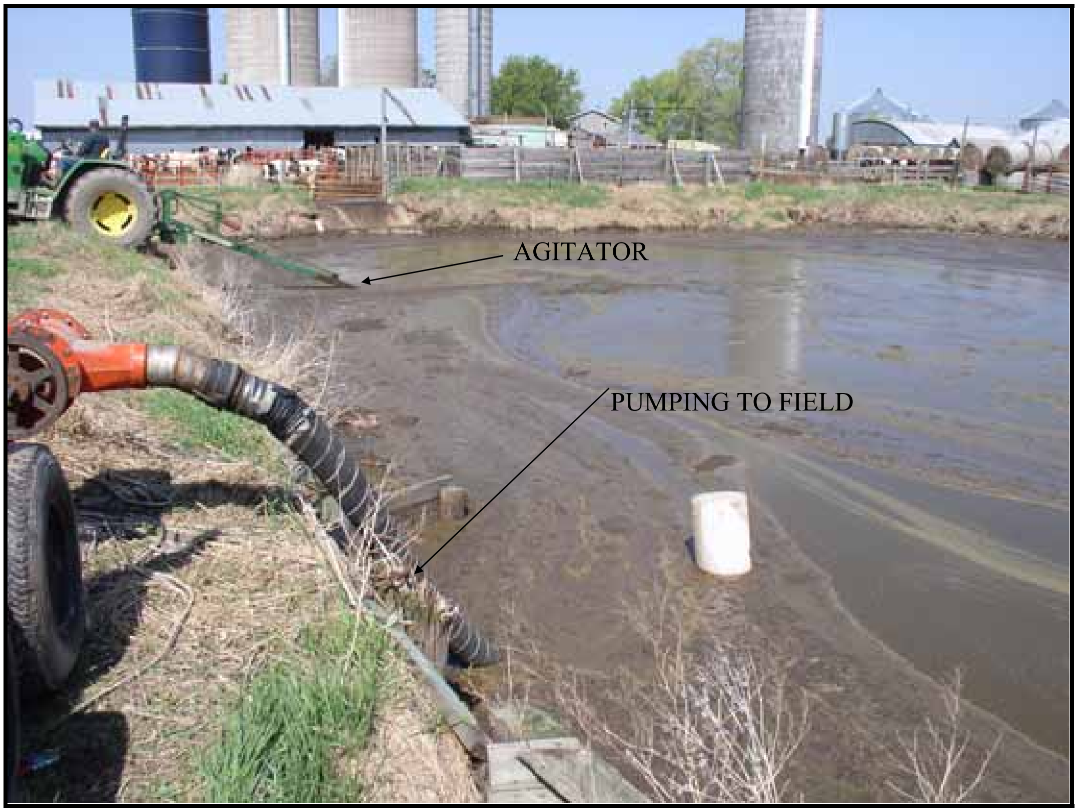
1.1.11 Proper Manure Handling, Collection, and Disposal

Animal operations are typically either pasture-based or confined, or sometimes a combination of the two. The operation type dictates the practices needed to manage manure from the facility. A pasture or open lot system with a relatively low density of animals (1 to 2 head of cattle per acre (USEPA, 2002a)) may not produce manure in quantities that require management for the protection of water quality. If excess manure is produced, then the manure will typically be scraped with a tractor to a storage bin constructed on a concrete surface. Stored manure can then be land applied when the ground is not frozen and precipitation forecasts are low. Rainfall runoff should be diverted around the storage facility with berms, grassed waterways, or tile-drain inlets. Runoff from the feedlot area is considered contaminated and is typically treated in a lagoon or filter strip.

Confined facilities (typically dairy cattle, swine, and poultry operations) often collect manure in storage pits located under slatted floors. Wash water used to clean the floors and remove manure buildup combines with the solid manure to form a liquid or slurry in the pit. The mixture is usually land applied or transported offsite.

Final disposal of waste usually involves land application on the farm or transportation to another site. Manure is typically applied to the land once or twice per year. To maximize the amount of nutrients and organic material retained in the soil, application should not occur on frozen ground or when precipitation is forecast during the next several days.

A photo of an earthen manure storage pit taken by the Kanabec County Soil and Water Conservation District is shown in Figure 11.



(Photo courtesy of KCSWCD)

Figure 11. Earthen Manure Storage Pit

The NRCS provides additional information on waste storage facilities and cover at
http://efotg.nrcs.usda.gov/references/public/MN/313mn.pdf
http://efotg.nrcs.usda.gov/references/public/MN/367mn.pdf

and on anaerobic digestors at
http://efotg.nrcs.usda.gov/references/public/MN/366mn.pdf

1.1.11 Effectiveness

Though little change in total phosphorus or organic content have been reported, reductions in fecal coliform as a result of manure storage have been documented in two studies:

- 97 percent reduction in fecal coliform concentrations in runoff when manure is stored for at least 30 days prior to land application (Meals and Braun, 2006)
- 90 percent reduction in fecal coliform loading with the use of waste storage structures, ponds, and lagoons (USEPA, 2003)

1.1.11 Costs

Depending on whether or not the production facility is pasture-based or confined, manure is typically deposited in feedlots, around watering facilities, and within confined spaces such as housing units and milking parlors. Except for feedlots serving a low density of animals, each location will require the collection and transport of manure to a storage structure, holding pond, storage pit, or lagoon prior to final disposal.

Manure collected from open lots and watering areas is often collected by a tractor equipped with a scraper. This manure is in solid form and is typically stored on a concrete pad surrounded by three walls that allow for stacking of contents. Depending on the climate, a roof may be required to protect the manure from frequent rainfall. Clean water from rooftops or up-grade areas should be diverted around waste stockpiles and heavy use areas with berms, grassed channels, or other means of conveyance (USEPA, 2003). Waste storage lagoons, pits, and above ground tanks are good options for large facilities. Methane gas recovered from anaerobic treatment processes can be used to generate electricity. The NRCS (2003) has developed cost estimates for the various tasks and facilities typically used to transport, store, and dispose of manure. Table 12 summarizes the information contained in the NRCS report and lists the capital and operating/maintenance costs reported per head of animal. Annual maintenance costs were assumed 3 percent of capital costs except for gutter downspouts (assumed 10 percent to account for animals trampling the downspouts) and collection and transfer (assumed 15 percent to account for costs associated with additional fuel and labor). The costs presented as a range were given for various sizes of operations. The lower values reflect the costs per head for the larger operations which are able to spread out costs over more animals.

The full NRCS document can be viewed at
http://www.nrcs.usda.gov/Technical/land/pubs/cnmp1.html.

The useful life for practices requiring construction is assumed 20 years. The total annualized costs were calculated by dividing the capital costs by 20 and adding the annual operation and maintenance costs. Prices are converted to year 2006 dollars.

Table 12. Costs Calculations for Manure Handling, Storage, and Treatment Per Head

Item	Application	Capital Costs per Head	Annual Operation and Maintenance Costs per Head	Total Annualized Costs per Head
Collection and Transfer of Solid Manure, Liquid/Slurry Manure, and Contaminated Runoff				
Collection and transfer of manure solids (assuming a tractor must be purchased)	All operations with outside access and solid collection systems for layer houses	$138.50 - dairy cattle $98.25 - beef cattle $0 - layer[1] $39.25 - swine	$20.75 - dairy cattle $14.50 - beef cattle $0.04 - layer $5.75 - swine	$27.50 - dairy cattle $19.25 - beef cattle $0.04 - layer $7.75 - swine
Collection and transfer of liquid/slurry manure	Dairy, swine, and layer operations using a flush system	$170 to $212 - dairy cattle $.50 - layer $6.00 to $4.75 - swine	$13.00 - dairy cattle $0.03 - layer $0.25 - swine	$21.50 to 23.50 - dairy cattle $0.05 - layer $0.50 - swine
Collection and transfer of contaminated runoff using a berm with pipe outlet	Fattened cattle and confined heifers	$4.25 to $9.50 - cattle	$0.13 to 0.25 - cattle	$0.25 to $0.75 - cattle
Feedlot Upgrades for Cattle Operations Using Concentrated Feeding Areas				
Grading and installation of a concrete pad	Cattle on feed (fattened cattle and confined heifers)	$37 - cattle	$1 - cattle	$3.00 - cattle

Clean Water Diversions				
Roof runoff management: gutters and downspouts	Dairy and swine operations that allow outside access	$17 - dairy cattle $2.50 - swine	$1.70 - dairy cattle $0.25 - swine	$2.75 - dairy cattle $0.50 - swine
Earthen berm with underground pipe outlet	Fattened cattle and dairy operations	$26.75 to $36.60 - cattle	$0.75 to $1.00 - cattle	$2 to $3.00 - cattle
Earthen berm with surface outlet	Swine operations that allow outside access	$1 - swine	$0.03 - swine	$0.08 - swine
Grassed waterway	Fattened cattle and confined heifer operations: scrape and stack system	$0.50 to $1.50 - cattle	$0.02 to $0.04 - cattle	$0.05 to $0.12 - cattle

[1] Costs presented by NRCS (2003) as operating and maintenance only.

Table 12. Costs Calculations for Manure Handling, Storage, and Treatment Per Head (continued)

Item	Application	Capital Costs per Head	Annual Operation and Maintenance Costs per Head	Total Annualized Costs per Head
Storage				
Liquid storage (contaminated runoff and wastewater)	Swine, dairy, and layer operations using flush systems (costs assume manure primarily managed as liquid)	$260 to $283.25 - dairy cattle $2 - layer $83.25 to $84.75 - swine	$7.75 - dairy cattle $0.06 - layer $2.75 - swine	$20.75 to $21.75 - dairy cattle $0.16 - layer $7 - swine
Slurry storage	Swine and dairy operations storing manure in pits beneath slatted floors (costs assume manure primarily managed as slurry)	$110 to $135- dairy cattle $16.50 to $20.75 - swine	$3.50 to $4 - dairy cattle $0.50 - swine	$8.75 to $10.75 - dairy cattle $1.25 to $1.50 - swine
Runoff storage ponds (contaminated runoff)	All operations with outside access	$133 - dairy cattle $148.50 - beef cattle $24.50 - swine	$4 - dairy cattle $4.50 - beef cattle $0.75 - swine	$10.50 - dairy cattle $12 - beef cattle $2 - swine
Solid storage	All animal operations managing solid wastes (costs assume 100% of manure handled as solid)	$208 - dairy cattle $136.75 - beef cattle $1 - layer $15 - swine	$6 - dairy cattle $4 - beef cattle $0.03 - layer $0.50 - swine	$16.50 - dairy cattle $10.75 - beef cattle $0.25 - layer $1.25 - swine

Table 12. Costs Calculations for Manure Handling, Storage, and Treatment Per Head (continued)

Item	Application	Capital Costs per Head	Annual Operation and Maintenance Costs per Head	Total Annualized Costs per Head
Final Disposal				
Pumping and land application of liquid/slurry	Operations handling manure primarily as liquid or slurry.	Land application costs are listed as capital plus operating for final disposal and are listed as dollars per acre for the application system. The required number of acres per head was calculated for each animal type based on the phosphorus content of manure at the time of application. Pumping costs were added to the land application costs as described in the document.		\$20.75 - dairy cattle \$0.25 - layer \$3.00 - swine
Pumping and land application of contaminated runoff	Operations with outside feedlots and manure handled primarily as solid	Pumping costs and land application costs based on information in NRCS (2003). Assuming a typical phosphorus concentration in contaminated runoff of 80 mg/L to determine acres of land required for agronomic application (Kizil and Lindley, 2000). Costs for beef cattle listed as range representing variations in number of animals and manure handling systems (NRCS, 2003). Only one type and size of dairy and swine operation were included in the NRCS document.		\$4.25 - dairy cattle \$4.00 - beef cattle \$4.75 - swine
Land application of solid manure	Operations handling manure primarily as solid	Land application costs are listed as capital plus operating for final disposal and are given as dollars per acre for the application system. The required number of acres per head was calculated for each animal type based on the phosphorus content of manure at the time of application. No pumping costs are required for solid manure.		\$11.50 - dairy cattle \$0.25 - layer \$1.50 - swine \$10.75 - fattened cattle

1.1.12 Composting

Composting is the biological decomposition and stabilization of organic material. The process produces heat that, in turn, produces a final product that is stable, free of pathogens and viable plant seeds, and can be beneficially applied to the land. Like manure storage areas, composting facilities should be located on dry, flat, elevated land at least 100 feet from streams. The landowner should coordinate with local NRCS staff to determine the appropriate design for a composting facility based on the amount of manure generated. Extension agents can also help landowners achieve the ideal nutrient ratios, oxygen levels, and moisture conditions for composting on their site.

Composting can be accomplished by simply constructing a heap of the material, forming composting windrows, or by constructing one or more bins to hold the material. Heaps should be 3 feet wide and 5 feet high with the length depending on the amount of manure being composted. Compost does not have to be turned, but turning will facilitate the composting process (University of Missouri, 1993; PSU, 2005). Machinery required for composting includes a tractor, manure spreader, and front-end loader (Davis and Swinker, 2004). Figure 12 shows a poultry litter composting facility.

(Photo courtesy of USDA NRCS.)

Figure 12. Poultry Litter Composting Facility

The NRCS provides additional information on composting facilities at
http://efotg.nrcs.usda.gov/references/public/MN/317mn.pdf
and
ftp://ftp.wcc.nrcs.usda.gov/downloads/wastemgmt/neh637c2.pdf

1.1.12 Effectiveness

Composting stabilizes the organic content of manure and reduces the volume that needs to be disposed of. In addition, the following reductions in loading are reported:

- 99 percent reduction of fecal coliform concentrations as a result of the heat produced during the composting process (Larney et al., 2003)
- 56 percent reduction in runoff volumes and 68 percent reduction in sediment as a result of improved soil infiltration following application of composted manure (HRWCI, 2005)

1.1.12 Costs

The costs for developing a composting system include site development costs (storage sheds, concrete pads, runoff diversions, etc.), purchasing windrow turners if that system is chosen, and labor and fuel required to form and turn the piles. Cost estimates for composting systems have not been well documented and show a wide variation even for the same type of system. The NRCS is in the process of developing cost estimates for composting and other alternative manure applications in Part II of Costs Associated with Development and Implementation of Comprehensive Nutrient Management Plans (NRCS, 2003). Once published, these estimates should provide a good comparison with the costs summarized for the Midwest region in Table 12. For now, costs are presented in Table 13 based on studies conducted in Wisconsin, Canada, and Indiana.

Researchers in Wisconsin estimated the costs of a windrow composting system using four combinations of machinery and labor (CIAS, 1996). These costs include collection and transfer of excreted material, formation of the windrow pile, turning the pile, and reloading the compost for final disposal. The Wisconsin study was based on a small dairy operation (60 head). Costs for beef cattle, swine, and layer hens were calculated based on animal units and handling weights of solid manure (NRCS, 2003). Equipment life is assumed 20 years. The costs presented in the Wisconsin study are much higher than those presented in Table 13 for collection, transfer, and storage of solid manure. However, the Wisconsin study presented a cost comparison of the windrow system to stacking on a remote concrete slab, and these estimates were approximately four and one-half times higher than the values summarized by NRCS. It is likely that the single data set used for the Wisconsin study is not representative of typical costs.

The University of Alberta summarized the per ton costs of windrow composting with a front end loader compared to a windrow turner (University of Alberta, 2000).

The Alberta Government presented a per ton estimate for a windrow system with turner: this estimate is quite different than the University of Alberta study. These per ton costs were converted to costs per head of dairy cattle, beef cattle, swine, and layer hens based on the manure generation and handling weights presented by NRCS (2003).

In 2001, the USEPA released a draft report titled "Alternative Technologies/Uses for Manure." This report summarizes results from a Purdue University research farm operating a 400-cow dairy operation. This farm also utilizes a windrow system with turner.

Table 13 summarizes the cost estimates presented in each of the studies for the various composting systems. None of these estimates include the final costs of land application, which should be similar to those listed for disposal of solid manure in Table 12, as no phosphorus losses occur during the composting process.

Table 13. Costs Calculations for Manure Composting

Equipment Used	Capital Costs per Head	Annual Operation and Maintenance Costs per Head	Total Annualized Costs per Head
2006 Costs Estimated from CIAS, 1996 – Wisconsin Study			
Windrow composting with front-end loader	\$344 - dairy cattle \$226.50 - beef cattle \$1.75 - layer \$25.25 - swine	\$190.75 - dairy cattle \$125.75 - beef cattle \$1 - layer \$14.00 - swine	\$208 - dairy cattle \$137 - beef cattle \$1 - layer \$15 - swine
Windrow composting with bulldozer	\$282.25 - dairy cattle \$186 - beef cattle \$1.50 - layer \$20.75 - swine	\$190.75 - dairy cattle \$125.75 - beef cattle \$1 - layer \$14 - swine	\$205 - dairy cattle \$135 - beef cattle \$1 - layer \$15.25 - swine
Windrow composting with custom-hire compost turner	\$282.25 - dairy cattle \$186 - beef cattle \$1.50 - layer \$20.75 - swine	\$228.25 - dairy cattle \$150.50 - beef cattle \$1.25 - layer \$16.75 - swine	\$242.75 - dairy cattle \$159.50 - beef cattle \$1.25 - layer \$17.75 - swine
Windrow composting with purchased compost turner	\$654.50 - dairy cattle \$431 - beef cattle \$3.50 - layer \$48 - swine	\$248.50 - dairy cattle \$163.75 - beef cattle \$1.25 - layer \$18.25 - swine	\$281.50 - dairy cattle \$185.50 - beef cattle \$1.50 - layer \$20.75 - swine
2006 Costs Estimated from University of Alberta, 2000			
Windrow composting with front-end loader	Study presented annualized costs per ton of manure composted.		\$25.25 to \$50.50 - dairy cattle \$16.75 to \$33.25 - beef cattle \$0.13 to \$0.25 - layer \$1.75 to \$3.75 - swine
Windrow composting with compost turner	Study presented annualized costs per ton of manure composted.		\$75.50 to \$151.25 - dairy cattle \$49.75 to \$99.75 - beef cattle \$0.50 to \$0.75 - layer \$5.50 to \$11.25 - swine
2006 Costs Estimated from Alberta Government, 2004			
Windrow composting with compost turner	Study presented annualized costs per ton of manure composted.		\$33.50 - dairy cattle \$22 - beef cattle \$0.25 - layer \$2.50 - swine
2006 Costs Estimated from USEPA, 2002a Draft			
Windrow composting with compost turner	Study presented annualized costs per dairy cow.		\$16.50 - dairy cattle \$10.75 - beef cattle \$0.09 - layer \$1.25 - swine

1.1.13 Alternative Watering Systems

A primary management tool for pasture-based systems is supplying cattle with watering systems away from streams and riparian areas. Livestock producers who currently rely on streams to provide water for their animals must develop alternative watering systems, or controlled access systems, before they can exclude cattle from streams and riparian areas. One method of providing an alternative water source is the development of off-stream watering using wells with tank or trough systems. These systems are often highly successful, as cattle often prefer spring or well water to surface water sources. Figure 13 shows a centralized watering tank allowing access from rotated grazing plots and a barn area.

Landowners should work with an agricultural extension agent to properly design and locate watering facilities. One option is to collect rainwater from building roofs (with gutters feeding into cisterns) and use this water for the animal watering system to reduce runoff and conserve water use (Tetra Tech, 2006). Whether or not animals are allowed access to streams, the landowner should provide an alternative shady location and water source so that animals are encouraged to stay away from riparian areas.

(Photo courtesy of USDA NRCS.)

Figure 13. Centralized Watering Tank

The NRCS provides additional information on these alternative watering components:

Spring development:
http://efotg.nrcs.usda.gov/references/public/MN/574mn.pdf,
Well development:
http://efotg.nrcs.usda.gov/references/public/MN/642mn.pdf,

Pipeline:
http://efotg.nrcs.usda.gov/references/public/MN/mn516.pdf,
Watering facilities (trough, barrel, etc.):
http://efotg.nrcs.usda.gov/references/public/MN/614mn.pdf

1.1.13 Effectiveness

The USEPA (2003) reports the following pollutant load reductions achieved by supplying cattle with alternative watering locations and excluding cattle from the stream channel by structural or vegetative barriers:

- 29 to 46 percent reductions in fecal coliform loading
- 15 to 49 percent reductions in total phosphorus loading

Some researchers have studied the impacts of providing alternative watering sites without structural exclusions and found that cattle spend 90 percent less time in the stream when alternative drinking water is furnished (USEPA, 2003). Prohibiting access to the stream channels will also prevent streambank trampling, decrease bank erosion, protect bank vegetation, and reduce the loading of organic material to the streams.

1.1.13 Costs

Alternative drinking water can be supplied by installing a well in the pasture area, pumping water from a nearby stream to a storage tank, developing springs away from the stream corridor, or piping water from an existing water supply. For pasture areas without access to an existing water supply, the most reliable alternative is installation of a well, which ensures continuous flow and water quality for the cattle (NRCS, 2003). Assuming a well depth of 100 ft and a cost of installation of $24 per ft, the cost to install a well is approximately $2,400 per well. The well pump would be sized to deliver adequate water supply for the existing herd size. For a herd of 150 cattle, the price per head for installation was estimated at $16.

After installation of the well or extension of the existing water supply, a water storage device is required to provide the cattle access to the water. Storage devices include troughs or tanks. NRCS (2003) lists the costs of storage devices at $24.50 per head.

Annual operating costs to run the well pump range from $9.50 to $23.25 per year for electricity (USEPA, 2003; Marsh, 2001), or up to $0.16 per head. Table 14 lists the capital, maintenance, and annualized costs for a well, pump, and storage system assuming a system life of 20 years.

Table 14. Costs Calculations for Alternative Watering Facilities

Item	Capital Costs per Head	Annual Operation and Maintenance Costs per Head	Total Annualized Costs per Head
Installation of well	$16	$0	$2
Storage container	$24.50	$0	$1.25
Electricity for well pump	$0	$0.16	$0.16
Total system costs	**$40.50**	**$0.16**	**$2.25**

1.1.14 Cattle Exclusion from Streams

Cattle manure is a substantial source of nutrient and fecal coliform loading to streams, particularly where direct access is not restricted and/or where cattle feeding structures are located adjacent to riparian areas. Direct deposition of feces into streams may be a primary mechanism of pollutant loading during baseflow periods. During storm events, overbank and overland flow may entrain manure accumulated in riparian areas resulting in pulsed loads of nutrients, total organic carbon (TOC), biological oxygen demand (BOD), and fecal coliform bacteria into streams. In addition, cattle with unrestrained stream access typically cause severe streambank erosion. The impacts of cattle on stream ecosystems are shown in Figure 14 and Figure 15.

Figure 14. Typical Stream Bank Erosion in Pastures with Cattle Access to Stream

Figure 15. Cattle-Induced Streambank Mass Wasting and Deposition of Manure into Stream

An example of proper exclusion and the positive impacts on the stream channel are shown in Figure 16.

(Photo courtesy of USDA NRCS.)

Figure 16. Stream Protected from Cattle by Fencing

The NRCS provides additional information on fencing at:
http://efotg.nrcs.usda.gov/references/public/MN/472mn.pdf

Allowing limited or no animal access to streams will provide the greatest water quality protection. On properties where cattle need to cross streams to have access to pasture, stream crossings should be built so that cattle can travel across streams without degrading streambanks and contaminating streams with manure. Figure 17 shows an example of a reinforced cattle access point to minimize time spent in the stream and mass wasting of streambanks.

(Photo courtesy of USDA NRCS.)

Figure 17. Restricted Cattle Access Point

The NRCS provides additional information on use exclusion and controlled access at:
http://efotg.nrcs.usda.gov/references/public/MN/472mn.pdf

1.1.14 Effectiveness

Fencing cattle from streams and riparian areas using vegetation or fencing materials will reduce streambank trampling and direct deposition of fecal material in the streams. As a result, eroded sediment and BOD_5 loads will decrease. The USEPA (2003) reports the following reductions in phosphorus and fecal coliform loading as a result of cattle exclusion practices:

- 29 to 46 percent reductions in fecal coliform loading
- 15 to 49 percent reductions in total phosphorus loading

1.1.14 Costs

The costs of excluding cattle from streams depends more on the length of channel that needs to be protected than the number of animals on site. Fencing may also be used in a grazing land protection operation to control cattle access to individual plots. The system life of wire fences is reported as 20 years; the high tensile fence materials have a reported system life of 25 years (Iowa State University, 2005). Fencing materials vary by installation cost, useful life, and annual maintenance cost as presented in Table 15.

Table 15. Installation and Maintenance Costs of Fencing Material per Foot

Material	Construction Costs (per ft)	Annual Maintenance Costs (per ft)	Total Annualized Costs (per ft)
Woven Wire	$1.55	$0.27	$0.34
Barbed Wire	$1.26	$0.21	$0.28
High tensile (non-electric) 8-strand	$1.16	$0.15	$0.19
High tensile (electric) 5-strand	$0.72	$0.10	$0.12

NRCS reports that the average operation needs approximately 35 ft of additional fencing per head to protect grazing lands and streams. Table 16 presents the capital, maintenance, and annualized costs per head of cattle for four fencing materials based on the NRCS assumptions.

Table 16. Installation and Maintenance Costs of Fencing Material per Head

Material	Capital Costs per Head	Annual Operation and Maintenance Costs per Head	Total Annualized Costs per Head
Woven Wire	$46.25	$3.75	$6.00
Barbed Wire	$35.50	$3.00	$4.75
High Tensile (non-electric) 8-strand	$32.50	$1.75	$3.00
High Tensile (electric) 5-strand	$24.50	$1.50	$2.50

1.1.15 Grazing Land Management

While erosion rates from pasture areas are generally lower than those from row-crop areas, a poorly managed pasture can approach or exceed a well-managed row-crop area in terms of erosion rates. Grazing land protection is intended to maximize ground cover on pasture, reduce soil compaction resulting from overuse, reduce runoff concentrations of nutrients and fecal coliform, and protect streambanks and riparian areas from erosion and fecal deposition. Figure 18 shows an example of a pasture managed for land protection. Cows graze the left lot while the right lot is allowed a resting period to revegetate.

(Photo courtesy of USDA NRCS.)

Figure 18. Example of a Well Managed Grazing System

The NRCS provides additional information on prescribed grazing at:
http://efotg.nrcs.usda.gov/references/public/MN/528mn.pdf

And on grazing practices in general at:
http://www.glti.nrcs.usda.gov/technical/publications/nrph.html

1.1.15 Effectiveness

Maintaining sufficient ground cover on pasture lands requires a proper density of grazing animals and/or a rotational feeding pattern among grazing plots. Increased ground cover will also reduce transport of sediment. Dissolved oxygen concentrations in streams will likely improve as the concentrations of BOD_5 in runoff are reduced proportionally with the change in number of cattle per acre.

The following reductions in loading are reported in the literature (USEPA, 2003; Government of Alberta, 2007):

- 40 percent reduction in fecal coliform loading as a result of grazing land protection measures
- 90 percent reduction in fecal coliform loading with rotational grazing
- 49 to 60 percent reduction in total phosphorus loading

1.1.15 Costs

The costs associated with grazing land protection include acquiring additional land if current animal densities are too high (or reducing the number of animals maintained), fencing and seeding costs, and developing alternative water sources. Establishment of vegetation for pasture areas costs from $41/ac to $73/ac based on data presented in the EPA nonpoint source guidance for agriculture (USEPA, 2003). Annual costs for maintaining vegetative cover will likely range from $6/ac to $11.50/ac (USEPA, 2003). If cattle are not allowed to graze plots to the point of requiring re-vegetation, the cost of grazing land protection may be covered by the fencing and alternative watering strategies discussed above.

1.1.16 Streambank Erosion BMPs

Reducing erosion of streambanks in the watershed will decrease sediment loading to the listed segments and improve temperature and dissolved oxygen conditions by allowing vegetation to establish. Filter strips, riparian area BMPs, and the agricultural BMPs that reduce the quantity and volume of runoff or prevent cattle access will all provide some level of streambank erosion protection.

In addition, the streambanks in the watershed should be inspected for signs of erosion. Banks showing moderate to high erosion rates (indicated by poorly vegetated reaches, exposed tree roots, steep banks, etc.) can be stabilized by engineering controls, vegetative stabilization, and restoration of riparian areas. The effectiveness and costs of stream restorations are site specific and highly variable. Watershed planners and water resource engineers should be utilized to determine the reaches where restoration will result in the most benefit for the watershed as a whole.

The NRCS provides additional information on streambank erosion controls at:
http://efotg.nrcs.usda.gov/references/public/MN/584mn.pdf

1.1.17 Stream Habitat Restoration

Stream restoration activities may be required in certain locations on the mainstem and South Fork of the Groundhouse River to increase biota scores to acceptable levels. A stream restoration plan has already been developed for the most impaired reach around Ogilvie (Magner, 2006). Following implementation of source control BMPs, it may be necessary to restore habitat where conditions have not been mitigated by pollutant reduction alone.

The restoration plan developed by MPCA (Magner, 2006) for the reaches near Ogilvie may be briefly summarized by the following concepts:

- Use fallen logs to create cross vanes and root wads that provide habitat and food sources for macroinvertebrates
- Slightly modify the channel to restore hydraulic capacity

A copy of the restoration plan may be obtained from MPCA.

The effectiveness and costs of stream restorations are site specific and highly variable. Watershed planners and water resource engineers should be utilized to determine the reaches where habitat restoration will result in the most benefit for the watershed as a whole.

The NRCS provides additional information on stream habitat improvement at:
http://efotg.nrcs.usda.gov/references/public/MN/395mn.pdf

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