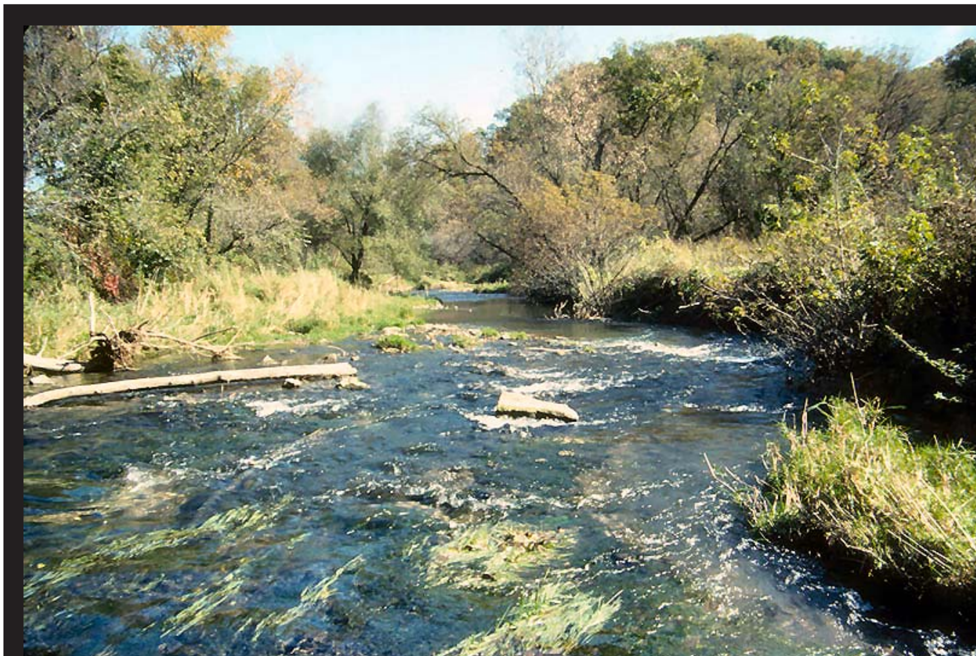
SNY MAGILL NONPOINT SOURCE POLLUTION MONITORING PROJECT: FINAL REPORT

Iowa Geological Survey
Technical Information Series 48



Iowa Department of Natural Resources
Jeffrey R. Vonk, Director
May 2005

COVER

Sny Magill Creek
near the SNI monitoring site.

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ABSTRACT

This report summarizes the major findings concerning the implementation of nonpoint source pollution controls and associated water quality changes from the Sny Magill Nonpoint Source Pollution Monitoring Project. From 1991 until 1999, selected best management practices (BMPs) were implemented throughout the Sny Magill watershed with the purpose of improving stream water and habitat quality. Many different BMPs were installed in the watershed, including tiled terraces, catchment basins, nutrient and pest management plans, and streambank protection structures. All of the structures and plans were implemented with the objective of decreasing sediment, nutrients, and fecal contamination in Sny Magill Creek.

From October 1991 until September 2001, a consortium of state and federal agencies attempted to document the effectiveness of the installed BMPs through monitoring of water quality, stream habitat, fish, and benthic macroinvertebrates. The adjacent Bloody Run watershed underwent no funded BMP application, but was monitored to serve as a control and statistically verify water quality changes in Sny Magill Creek. Monitoring in both watersheds was done on an even-interval (daily, weekly, and monthly) basis on three paired main channel and three tributary stations. Monitoring was continuous throughout the 10-year study period.

The primary objective of reducing sediment in Sny Magill Creek was observed by the significant decrease in turbidity documented in two of the three main channel monitoring locations in Sny Magill Creek relative to Bloody Run creek. Though less pronounced and significant, suspended sediment also declined during the study period. These two water quality parameters indicate that the primary objective of the study, sediment reduction, was partially fulfilled by the installation of the selected BMPs.

Nitrate+nitrite-N measurements indicated a statistically significant increase during the study in all paired monitoring stations. This increase is contrary to the projected decrease in nitrate+nitrite-N and indicates that some of the installed BMPs may have unforeseen effects on water quality. It should be noted that concentrations of nitrate+nitrite-N were very low in Sny Magill Creek at the beginning of the study, and even with the increase, were relatively low for the state of Iowa at the end of the study.

Fecal coliform levels had no overall trends in Sny Magill Creek. The downstream station noted a decrease in levels throughout the study period, while the midstream and upstream monitoring station noted no significant change and an increase, respectively.

The habitat assessments conducted during the fall of each year indicated monitoring sites with similar drainage areas had similar habitat characteristics for most years. Stream flow velocity was an important factor affecting habitat parameters. Substrate composition varied

from site to site, but mainly consisted of silt for both Sny Magill and Bloody Run creeks through the monitoring period.

The benthic macroinvertebrate communities in Sny Magill and Bloody Run watersheds showed the same dominant taxa. Based on the results from taxa richness, Hilsenhoff Biotic Index (HBI), EPT index, and percent dominant taxon analyses, site SNWF consistently had the best overall water quality ranking while site BR2 ranked at or near the bottom for most years. Analyses indicated trends towards improving water quality at the Sny Magill sites for taxa richness, EPT, and percent dominant taxa, but for HBI, regressions showed a declining water quality through time for these sites.

Fish monitoring indicated that the population in the study creeks was typical of Iowa coldwater streams. Each creek was dominated by different species: fantail darter in Sny Magill and slimy sculpin in Bloody Run. Reduced coldwater fish species diversity and the occurrence of slimy sculpin indicated that water quality in the Sny Magill Creek was improving during the later years of the study.

INTRODUCTION

The Clean Water Act (CWA), passed in 1972, directed that all of the nation's waters be made safe for fish, shellfish, wildlife, and humans. Since its establishment, numerous amendments have expanded and refined the initial charge of the CWA. Among these was the addition of Section 319 in 1987. Section 319 directs each state to institute monitoring and control of nonpoint source pollution in its waters. Unlike more readily quantifiable 'point' sources, nonpoint source pollution is diffuse in origin, with no specific location or identifiable sources to monitor or gage. Instead, nonpoint source pollution arises from rainwater mobilizing and transporting contaminants from widespread sources such as agriculture, urban areas, forestry, construction, and mining. Today, nonpoint source pollution is the leading cause of contamination in our continental rivers and lakes.

The National Monitoring Program was developed under Section 319 to evaluate the effectiveness of technologies designed to reduce nonpoint source pollution, and to improve our understanding of nonpoint source pollution. The National Monitoring Program is designed to monitor watersheds across the country for a period of six to ten years and evaluate how effectively "Best Management Practices," or BMPs, reduce nonpoint pollution on a watershed scale. Currently there are 23 national monitoring watershed projects (Figure 1). Two of these locations address urban nonpoint sources, while the remainder concentrate on agricultural areas.

The Sny Magill Project was designed to monitor and assess improvements in water quality associated with the implementation of BMPs. During the study, a variety of BMPs were implemented as part of a land treatment projects throughout the Sny Magill watershed. The goal of the land treatment project was to reduce the amount of sediment, pesticides, nutrients, and animal waste entering Sny Magill Creek.

Sny Magill Creek is a heavily fished cold water stream located in northeast Iowa. The stream is part of the Mississippi River Basin and is managed by the Iowa Department of Natural Resources (IDNR) for "put and take" trout fishing, along with other recreational activities. Sny Magill Creek drains a 35.6 mi^2 agricultural watershed in Clayton County (Figure 2). Land use in the watershed is primarily forest, forested pasture, rowcrop, cover crop, and pasture. The Sny Magill watershed is impacted by pollutants, including sediment, nutrients, pesticides, and bacteria, originating from intensive agricultural practices in the region. The project study area includes the entire watershed region of Sny Magill Creek and its major tributaries.

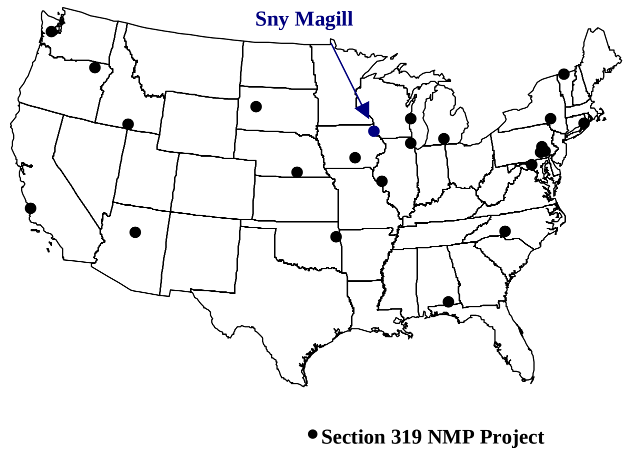


Figure 1. National Monitoring Program locations.

A paired-watershed approach was used to monitor improvements in water quality in the Sny Magill watershed. The paired watershed approach is the most appropriate monitoring design when evaluating the impact of a BMP or system of BMPs at the watershed scale (Spooner et al., 1985). The adjacent Bloody Run watershed serves as the control watershed (Figure 2) and was selected because of its similarities to Sny Magill Creek in size, climate, soils, hydrogeology, and topography.

The project objectives were to:

- Quantify changes, resulting from BMP implementation, in sediment, nitrate, pesticide, turbidity, and fecal coliform concentrations in Sny Magill Creek relative to Bloody Run Creek.
- Document improvements in the biological habitat through monitoring of the benthic

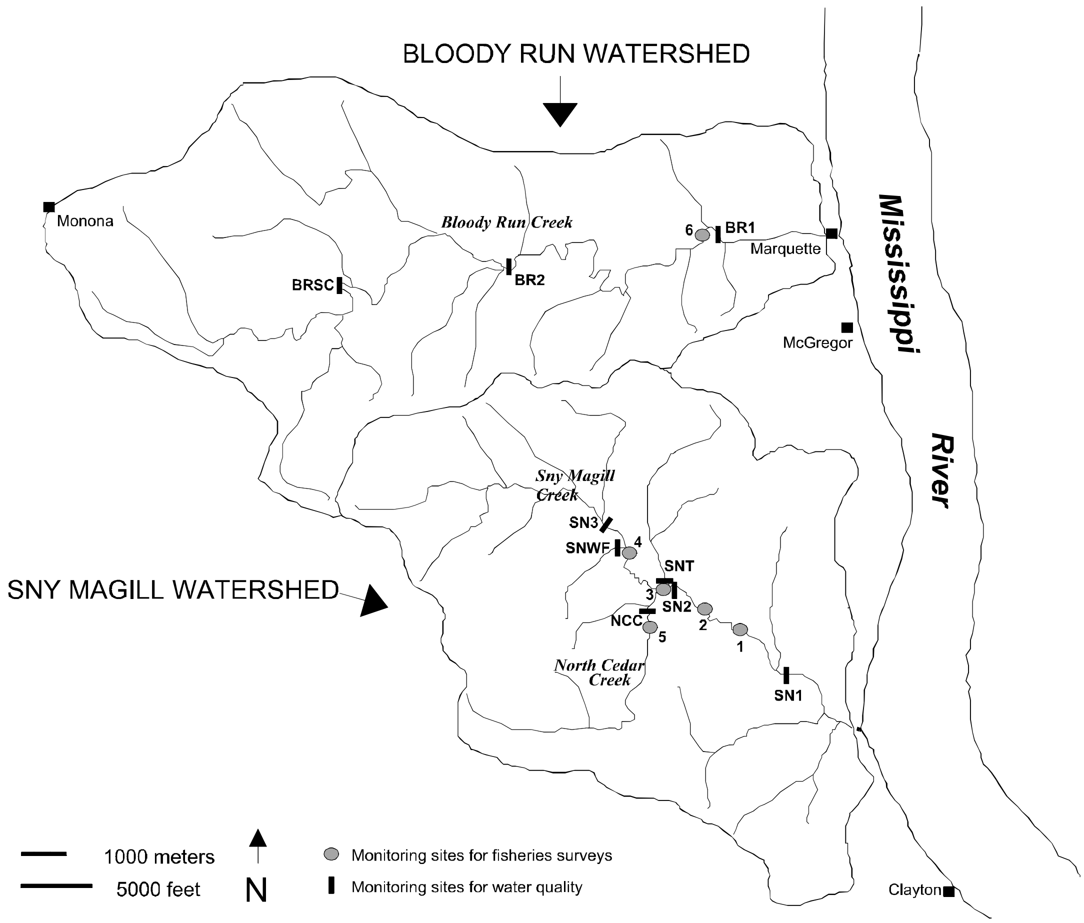


Figure 2. Sny Magill (treatment) and Bloody Run (control) watersheds.

macroinvertebrate populations, fish populations, and habitat assessments of the stream corridor.

Major monitoring components in this project were:

- Continuous stream discharge measurements near the mouths of both Sny Magill and Bloody Run creeks.
- Daily suspended sediment measurements at the stream gages on both Sny Magill and Bloody Run creeks.
- Weekly to monthly sampling for chemical and physical parameters at primary locations and several other sites on both creeks.
- Annual habitat assessments conducted along stretches of both stream corridors.
- Seasonal bi-monthly monitoring of benthic macroinvertebrates (April-October).
- Annual fisheries surveys conducted on the main channel of both streams.

The Sny Magill Nonpoint Source Pollution Monitoring Project was an interagency effort supported, in part, by a Section 319 grant managed by the Iowa Department of Natural Resources. In addition to project administration and management, IDNR conducted annual stream habitat assessment, fisheries surveys, and weekly water quality sampling. The University of Iowa Hygienic Lab conducted water quality analyses and performed monitoring of benthic macroinvertebrates. The U.S. Geological Survey (USGS) maintained gaging stations for stream discharge and sediment.

THE SNY MAGILL AND BLOODY RUN WATERSHEDS

Geologic Setting

Sny Magill and Bloody Run are third-order streams in northeast Iowa that discharge directly into the Mississippi River. The Sny Magill watershed is 22,780 acres, of which the water monitoring project studies the upper three-quarters, or 17,680 acres. The topography of the watershed is characterized by narrow, gently sloping uplands that break into steep slopes with abundant rock outcrops. Elevation in the watershed ranges from a high of 1160 ft. above sea level to a low of 650 ft. at the discharge monitoring station. Of the 24,220 total acres in the Bloody Run watershed, 21,620 acres were included in this study. Bloody Run elevations range from 1200 ft. to 650 ft. Both streams exhibit dendritic drainage patterns.

The Sny Magill and Bloody Run watersheds are located in the Paleozoic Plateau landform region in northeast Iowa (Prior, 1991). The landscape in this region is dominated by Paleozoic age bedrock. The oldest bedrock includes the sandstone and sandy dolostones of the Jordan Sandstone, found at the eastern edge of Bloody Run Creek. The youngest bedrock in the area is the Upper Ordovician-age Maquoketa Formation, located in the headwaters of the streams. The sandstone and limestone units that Sny Magill and Bloody Run creeks cut through form important aquifers throughout much of the region. Karst topography, including caves, sinkholes, and springs, is prevalent in the area, providing direct routes for surface pollutants to enter aquifers and streams.

The surficial deposits in the watersheds owe their origin to the direct or indirect effects of glacial advances. Upland surficial deposits include loam-textured pre-Illinioian glacial till, Sangamon and Farmdale paleosols, and Peoria Loess. Wedges of colluvium are present along most of the steeply sloping valley walls. Valley fills include slack-water deposits associated with glacial melting during the late Wisconsian, and loamy, sandy and silty late-Wisconsinan and Holocene alluvial deposits of the DeForest Formation (Bettis et al., 1994).

Climate

Northeast Iowa is characterized by a midcontinental subhumid climate, with distinct seasonal temperature fluctuations and relatively high precipitation. Although no long-term weather station is located within the watersheds, the Elkader recording station, located approximately 20 miles southwest of the Sny Magill watershed, has a mean annual precipitation rate of about 33 inches. Mean annual temperature for the Elkader station is 44^{o}F. Winter average temperature is 22^{o}F, and the summer average is 72^{o}F. During the study, daily precipitation was measured at recording stations lo-

A view of the rolling hills that make up Sny Magill and Bloody Run.

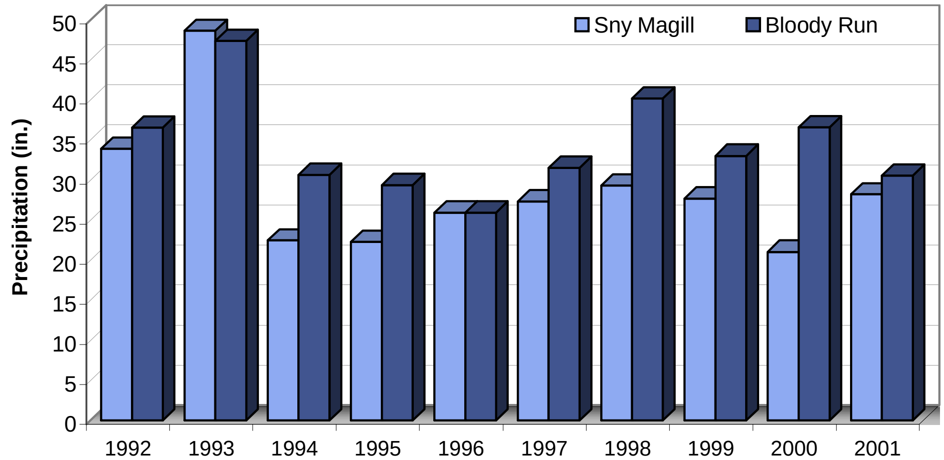


Figure 3. Total annual rainfall for Bloody Run and Sny Magill.

cated in each watershed. When these rain gages malfunctioned, rainfall data from nearby Prairie Du Chien, Wisconsin were used to supplement missing values. With the exception of Water Year 1993, annual precipitation in the Bloody Run watershed was higher than Sny Magill (Figure 3). The 10-year mean rainfall for Bloody Run was about 34 inches, 15% higher than the 10-year average of 29 inches for Sny Magill. The cause for the disparity between rainfall amounts in the watersheds is unclear. It might be attributed to faultiness of the rain gages, gage location, or a true difference in rainfall amount between the two watersheds. In both watersheds most rainfall occurred between April and September.

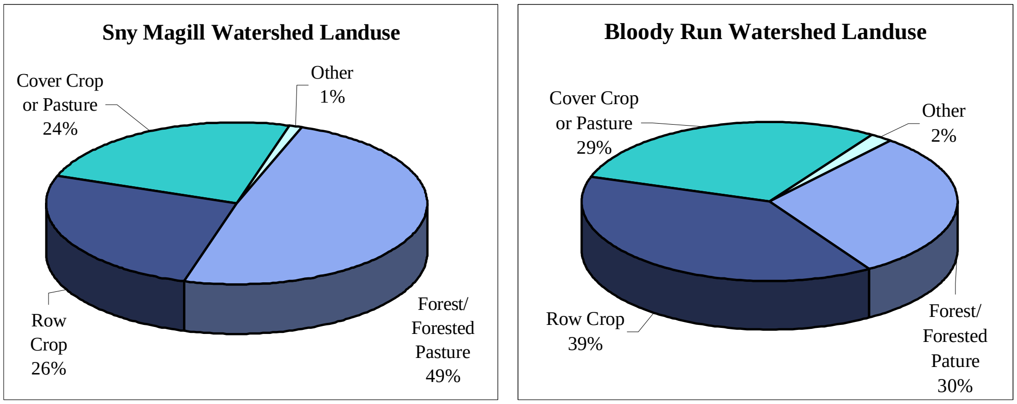
Hydrology

Documenting stream discharge is very important in providing a context for pollutant transport. Monitoring locations SN1 and BR1 were equipped with USGS streamflow gages for the duration of the project. During the study, daily discharge for Sny Magill averaged 21.0 cubic feet per second (cfs), 15% lower than Bloody Run's average daily discharge of 24.8 cfs. However, when average discharge is normalized for unit area in the watershed, Sny Magill has a slightly higher value of 0.76 cfs/mi^2 compared to 0.74 cfs/mi^2 for Bloody Run. Mean discharge varied considerably, both during the year and from year to year. The greatest mean discharge in both streams occurred during Water Year (WY) 1993, coinciding with the greatest amount of precipitation (a Water Year is defined as the October 1st through September 30th period). The Midwest experienced sustained flooding during WY 1993, as both rainfall and discharge levels were consistently high. Water Year 1997 had the lowest mean discharge for both streams.

The two primary components in stream discharge are direct surface runoff and baseflow. Surface runoff is immediately attributable to precipitation and thawing events. Baseflow is composed of several different sources, including tile lines, soils, and bedrock aquifers. Both Sny Magill and Bloody Run creeks receive significant baseflow from the Ordovician Galena aquifer. During the 10 years of this study, the mean baseflow component was estimated to be 85% for Sny Magill and 86% for Bloody Run. Analysis on Water Year 1992 data estimated a baseflow component of 78% for Sny Magill and 80% for Bloody Run (Bettis et al., 1994). The baseflow component in these two streams is greater than similarly sized streams in Iowa, a likely result of both streams cutting through the highly productive Cambrian-Ordovician aquifer.

Many small ephemeral and perennial streams feed into the main channel of both Sny Magill and Bloody Run creeks. The largest major tributary contributing to Sny Magill is North Cedar Creek (NCC). North Cedar Creek is estimated to contribute 20% of the discharge measured at the SN1

Figure 4. Landuse in the watersheds.



station. Another important tributary of Sny Magill is the West Fork of Sny Magill (SNWF), which contributes an estimated 15% of the flow at site SN1. Although Bloody Run has a number of tributaries, all monitoring stations were located on the main channel. Although there are no large lakes in the area, Sny Magill and Bloody Run watersheds contain numerous ponds, which typically are artificial structures that also serve as sediment traps.

Land Use

Land use in both watersheds is dominated by agriculture. Deciduous trees and riparian vegetation are located on steep hillsides and in close proximity to stream channels, but much of the watershed area is pastured or cultivated. Land use for the watersheds was interpreted from 1991 Landsat satellite imagery. Five land use classes were delineated from color infrared aerial photographs. Those classes include rowcrop, cover crop/ pasture, forest/forested pasture, farmstead/urban, and other. Most of Sny Magill watershed is forest/ forested pasture (49%), row crop (26%), and cover crop/pasture (24%). Bloody Run watershed is dominated by row crop (39%), followed by forest/ forested pasture (30%), and cover crop/pasture (29%). Figure 4 displays the land uses in the watersheds. The only urban area found in either watershed is the small town of Monona located on the western edge of the Bloody Run watershed.

BMP Implementation

Paired watershed environmental monitoring projects require two analogous watersheds to be measured simultaneously and compared to each other. One watershed, called the control watershed, undergoes limited changes in land use or management practices for the duration of the project. The other watershed, called the treatment watershed, undergoes BMP implementation during the study. The period before BMP installation is called the "calibration period," and is used to determine the relationship between the two watersheds before major land management changes. After the influence of new BMPs takes effect, a new relationship will exist; this is called the "treatment period." Although the paired watershed study design does not assume that the control and treatment watersheds are completely uniform in their physical characteristics, it does assume that both watersheds respond to environmental changes in a similar manner. This minimizes the effects of environmental variation on the data (Clausen and Spooner, 1993).

North Cedar Creek Water-Quality Special Project

In 1989 the State of Iowa designated certain streams and lakes as high-quality waters. These water bodies were chosen based on their exceptional quality and economical or ecological importance to the state. One of those chosen was North Cedar Creek, a tributary to Sny Magill, with a 3,220-acre watershed.

Funded by the U.S. Department of Agriculture (USDA), the North Cedar Creek Water-Quality Special Project was designed to control agricultural waste pollution and sediment erosion in the creek by the implementation of structural land-treatment practices. The primary focus of the project was to reduce the upland soil erosion in the watershed and the negative effects that in-stream sediment pollution had on the cold water fish such as brown and brook trout.

From 1988 to 1994 the project successfully installed many BMPs, including terraces, grade stabilization structures, and agricultural waste structures (Table 1). The estimated landowner participation for this project was 80-85% (Tisl and Palas, 1998). However water quality was not monitored on North Cedar Creek Project prior to implementation of these BMPs. After completion of this project in 1994, landowners in the North Cedar Creek watershed could receive additional BMP funds through the Sny Magill Hydrologic Unit Area (HUA) and Sny Magill Watershed Project.

Sny Magill HUA and Watershed Project

The Sny Magill HUA project began in 1991 and continued until 1999. The overall goal of the HUA was to coordinate efforts among the Natural Resources Conservation Service (NRCS), Iowa State University Extension (ISUE), and the Farm Service Agency (FSA) in providing services and technical assistance for landowners to install BMPs. This project covered 19,560 acres in the Sny Magill watershed, 10,468 acres of which were classified as Highly Erodible Land (HEL). Out of the 98 landowners in the Sny Magill HUA, 81% chose to participate in the project.

In 1994, funding for further BMPs in the Sny Magill HUA was eliminated by a shifting of priorities at the FSA. In response, the Clayton County Soil and Water Conservation District submitted a watershed project application with the INDR and the Iowa Department of Agriculture and Land Stewardship – Division of Soil Conservation (IDALS-DSC), creating a new project that worked supportively with the existing efforts and supplied lost BMP funding. This initiative, the Sny Magill Creek Watershed Project, began in Fiscal Year 1995 and continued until 1999.

The Sny Magill HUA and the Sny Magill Creek Watershed Project shared four major objectives. These objectives were chosen and prioritized based upon known water quality deficiencies in the creek.

Objective 1: Reduce sediment delivery to Sny Magill Creek by 50 percent. This objective focused on improving stream habitat by decreasing siltation and allowing greater areas of exposed rock and cobble in the stream bed, therefore providing increased protection and shelter for aquatic organisms. To meet this objective the projects relied heavily on structural and land management BMPs. Table 2 lists eight of the most extensively applied practices in the watershed. One of the more cost-effective and accepted practices implemented during the study was the construction of tile-outlet terraces. These structures effectively dealt with the high-topography landscape, reducing both the energy and quantity of surface runoff from row-cropped areas. Other widely used and effective methods of sediment reduction included water and

Table 1. Selected BMPs installed by the NCC Water-Quality Special Project (1988-1994).

Conservation Practice	Number Applied	Units
Terraces	109,720	ft.
Outlet	64,906	ft.
Agricultural Waste Structure	2	no.
Grade Stabilization Structure	6	no.
Old Terrace Repair	200	ft.

Conservation Practice	Number Applied	Units
Terraces	269,585	ft.
Streambank Protection	1,140	ft.
Water and Sediment Control Basin	61	no.
Underground Outlet	150,529	ft.
Grade Stabilization Structure	90	no.
Tree Planting	25	ac.
Field Border	26,700	ft.
Contouring	1,907	ac.

Table 2. Selected BMPs installed by the Sny Magill Watershed Project and HUA (1991-1999).

sediment control basins, grade stabilization structures, and contouring. The sediment control implementation efforts were judged successful and the number of installed structures exceeded the project goals. Based on the NRCS Universal Soil Loss Equation (USLE) model estimates, the project reached its goal of reducing sediment delivery by 50%. It is important to note that this number is based on a model for sediment delivery and does not necessarily reflect sediment concentration reductions in the stream.

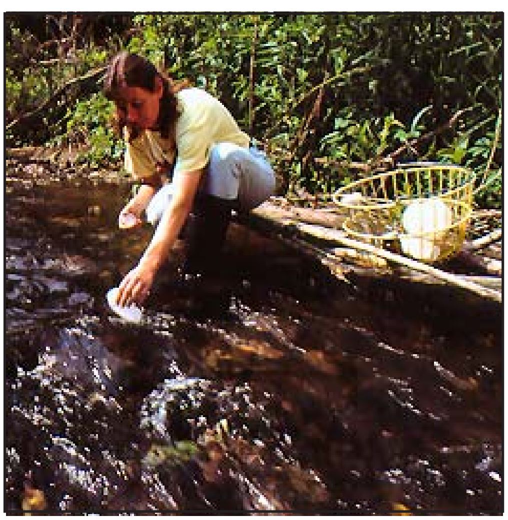
Objective 2: Reduce manure runoff to Sny Magill Creek by helping producers implement 30 animal manure management systems. Unfortunately, only one manure management system was constructed during the project period. This occurred in part because the number of feeder hog and dairy cattle operations in the watershed dropped substantially after the project began. Also, although approximately the same number of hogs were being raised in the watershed, the number of operations producing them decreased, concentrating hog production in high-population confinement sheds. The high costs of implementing manure management structures compared to the relative small size of the farms in the watershed also limited the number of producers who were willing to install the systems. However, the project did have success in working with producers to develop more effective procedures for manure application on their crops. In particular, manure applications were better matched to crop nutrient uptake potential, and application to steeper slopes were avoided.

Objective 3: Accelerate the adoption of refined crop and manure management practices that reduce agricultural pollution potential in the watershed. This objective was partly achieved through the implementation of an Integrated Crop Management (ICM) assistance program for the producers. Between 1992 and 1995, individual ICM assistance was provided to 14 producers with a combined 3,000 row-crop acres in the watershed. Although three producers dropped out after the first year, eleven continued and completed the program in either 1994 or 1995. It is estimated that in the 1995 crop season alone, producers in the program reduced their nitrogen application by 26,220 lbs, used 11,435 lbs less phosphate, 11,210 lbs less potash, 525 lbs less atrazine, and 145 lbs less cyanazine than in the beginning of the project, saving an estimated $11.15 per acre (Tisl and Palas, 1998).

After the ICM program was completed it was determined that continued producer education was essential in achieving the long-term goal of reducing nutrient and pesticide inputs. This resulted in the Nutrient and Pest Management Incentive Education Program. Through a series of workshops, producers were provided information on the benefits of reduced nutrient and pesticide inputs on the land, and shown how to maximize the economic benefit of the fertilizer and pest management products used. At the end of the study a survey was sent to 44 producers in the watershed. A total of 36 responded that they had reduced nitrogen inputs, 30 said they would change their manure management, and 37 planned on changing their pesticide inputs in the future (Tisl and Palas, 1998). Future surveys

Streambank stabilization structure.

A pesticide management demonstration.



Sampling water quality in Sny Magill Creek.

were planned to ensure that publicly funded education programs continue to have a positive effect on practices.

Objective 4: Develop a series of demonstrations to educate the watershed's producers and the public at large about water quality issues and provide additional data and learning experience for the participating agencies. One of the most important components of water quality protection efforts is the education of the public, as it is they who own the land and make the final land management decisions. Newspapers and other outlets were frequently used to publicize the project and its accomplishments. One of the more effective publications was the bi-monthly newsletter *Water Watch*. Iowa State University Extension and the Northeast Iowa Demonstration Project developed this newsletter cooperatively and released it on a bi-monthly basis to over 1,750 people, including producers, project coordinators, agribusiness leaders, and staff of the participating agencies in the watershed and across northeast Iowa. Special events were also conducted to provide more information on the monitoring program or assorted BMPs, including field tours highlighting streambank stabilization, nutrient management, and timber stand improvement activities in the watershed.

The installation of BMPs did not end with the North Cedar Creek Agricultural Conservation Program in 1994, or with the Sny Magill HUA and Watershed project in 1999. Many producers independently found other resources for project money and support, and continued to add to the list of BMPs installed in the watershed.

WATER MONITORING RESULTS

Sny Magill Creek and Bloody Run Creek were monitored for water quality changes throughout the 10-year study. Discharge was measured continuously by USGS monitoring stations located at sites SN1 and BR1 (Figure 5). Suspended sediment samples were taken daily at sites SN1 and BR1 and fitted to a hydrograph (normalized for discharge during the day). All sites (SN1, BR1, SN2, BR2,

Tiled terraces were used extensively as a BMP practice in the watershed.

SN3, BRSC, NCC, SNT, and SNWF) were monitored on an even-interval basis for water quality variables such as nitrate, nitrite, total phosphorus, and other parameters (Table 3). Annual water quality and stream discharge results are summarized in Seigley and others, 1994(a), 1994(b); Langel and others, 2001; Liu and others, (*in press)*.

Statistical Analysis

The Sny Magill Watershed National Monitoring Program Project was initially designed as a pre/post paired watershed study between downstream paired monitoring sites SN1 and BR1. The calibration period was to be Water Year (WY) 1992, and the treatment period was WYs 1999-2001. However the conclusion was reached that one year was insufficient for the calibration period; a firm relationship between the two streams couldn't be established with a single year of monitoring data.

As a result, a different pre/post implementation period was defined, and a gradual change model was also used to analyze for water quality changes. In both of these regression models the dependent variable (Y) was the Sny Magill water quality parameter, and one independent variable (X_1) was the matching Bloody Run water quality parameter. Other independent variables changed in accordance with the model design.

Pre/post Model

This regression model uses a pre/post design with the middle years of the project (1995-1998) removed. Water-years 1992-1994 became the pre-BMP calibration period (indicated by a 0 in the model) and 1999-2001 the post-BMP treatment period (indicated by a 1 in the model). Water-years 1992-1994 were chosen as the calibration period to establish a long enough period for a good relationship between the two streams; and because the majority of BMPs had not yet been implemented during this time period. The period WYs 1999-2001 was chosen as the treatment period because both the Sny Magill HUA and Sny Magill Watershed Project ended major funding for BMP implementations in 1999, causing a major decline in BMP adoption.

The general equation used for the pre/post model is:

$$Y = \beta_0 + \beta_1 X_1 + \beta_2 X_2 + \beta_3 X_1 X_2$$

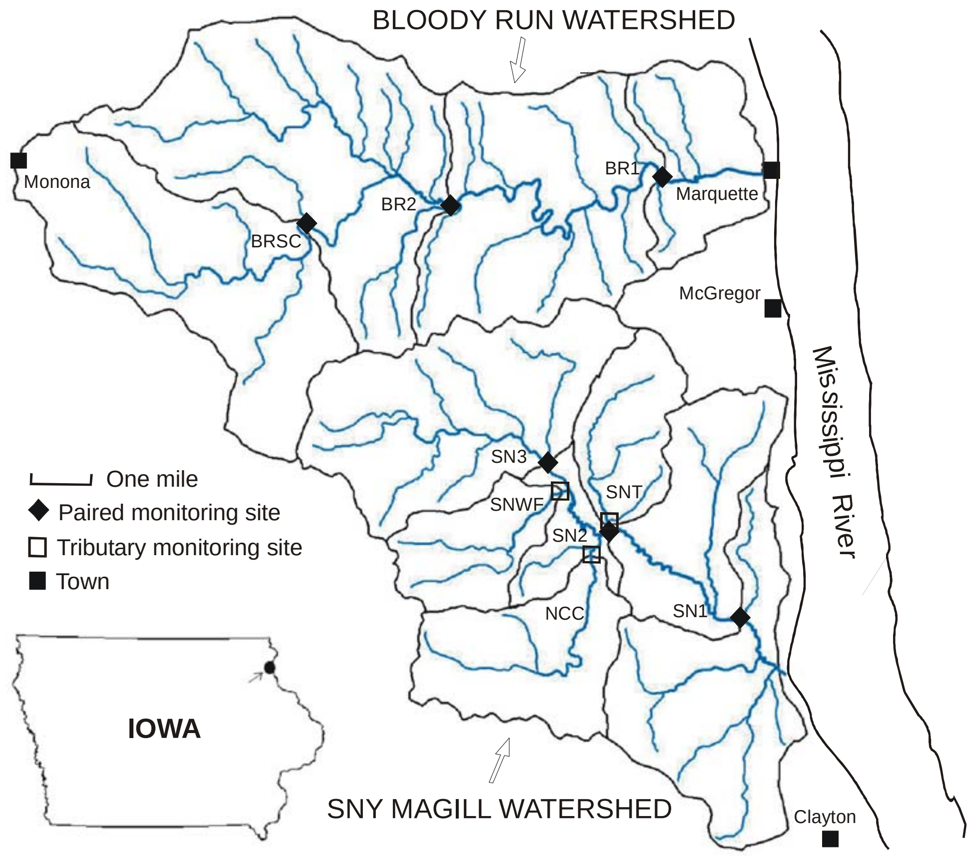


Figure 5. Water quality monitoring locations on Sny Magill and Bloody Run creeks.

The dependent variable (*Y*) is the Sny Magill water quality parameter in question. The first independent variable (X_1) is the Bloody Run water quality parameter; the second (X_2) indicates the calibration period (0) and the treatment period (1). $\beta_{1\text{-}3}$ are regression coefficients provided by the model.

During the calibration period the equation reduces to:

$$Y = \beta_0 + \beta_1 X_1$$

The full treatment period equation is:

$$Y = \beta_0 + \beta_2 + (\beta_1 + \beta_3) X_1$$

This equation gives both a different slope and y-intercept from the calibration period. In some cases the full treatment equation is not statistically different from the calibration equation. In this case a reduced model can be used:

$$Y = \beta_0 + \beta_2 + (\beta_1) X_1$$

The reduced equation drops out the interaction term (X_1X_2), yielding a regression with the same slope as the calibration period, but with a different y-intercept.

The advantage of the pre/post model is that it divides the project temporally into two distinct periods (calibration, treatment), establishing a strong relationship between the datasets. Therefore, trends found in the pre/post model are generally statistically significant. The disadvantage for this model is that actual BMP implementation in Sny Magill did not closely resemble the 1992-1994 period. A better calibration would have been obtained with a more extensive monitoring period prior to BMP implementation. Although a true calibration period did not occur for this project, many BMPs likely have a lag time between actual implementation and

Table 3. Monitoring sites and parameters used for paired analysis.

Site	Area	Frequency	Water Quality Parameters
SN1	17,680ac	Weekly	Temperature, Cond., NOx, Fecal Coliform, Ammonia, Dissolved Oxygen, Triazines, Chloride, Turbidity, Total Phosphorus
		Daily	Discharge (Q) and Suspended Sediment
BR1	21,620ac	Weekly	Temperature, Cond., NOx, Fecal Coliform, Ammonia, Dissolved Oxygen, Triazines, Chloride, Turbidity, Total Phosphorus
		Daily	Discharge (Q) and Suspended Sediment
SN2	14,550ac	Monthly	Temperature, Cond., NOx, Fecal Coliform, Ammonia, Dissolved Oxygen, Chloride, Turbidity
BR2	15,160ac	Weekly*	Temperature, Cond., NOx, Fecal Coliform, Ammonia, Dissolved Oxygen, Chloride, Turbidity
SN3	6,073ac	Weekly*	Temperature, Cond., NOx, Fecal Coliform, Ammonia, Dissolved Oxygen, Chloride, Turbidity
BRSC	6,773ac	Monthly	Temperature, Cond., NOx, Fecal Coliform, Ammonia, Dissolved Oxygen, Chloride, Turbidity

*Only used corresponding monthly date in analysis

measured water quality results. Therefore, although the pre/post model fit was not ideal, the use of the model will still provide significant trends and changes in water quality from the project.

Gradual Change Model

The second model, called the gradual change model, provides a better analog to actual BMP implementation and associated environmental effects during the project. Instead of having calibration and treatment periods, the independent variable is units of time passed (either in weeks or months, depending on the sampling of the parameter) to estimate a linear relationship and amount of increase or decrease in the variable for each unit of time.

$$Y = \beta_0 + \beta_1 X_1 + \beta_2 X_2$$

This model has the same dependent (Y) and first independent (X_1) variable as the pre/post model, but instead of calibration (pre) and treatment (post) dividing units used in the first equation, X_2 is units of time (days, weeks or months). The equation remains the same throughout the dataset.

Both of these statistical tests are parametric and require normal data distribution before being incorporated into the model (Grabow et al., 1998). If the data were highly skewed (not fitting the normal distribution curve), the dataset was log-transformed. Log transformation usually decreases skewness in a dataset. After the model analysis, the log-transformed data was converted back to the initial values.

Multiple linear regression also requires that raw data values be independent of each other. If the data are collected in short time intervals, there is a good chance that it will be autocorrelated, or dependent on the previous and subsequent values. The most important issue in regression analysis is that the residuals are not autocorrelated. The easiest way to remove autocorrelation is to either average or aggregate the data into larger time intervals. When this did not work, an autoregressive model was used (SAS® - proc autoreg) to try to eliminate the correlation in the data.

Three pairs of stream monitoring stations in

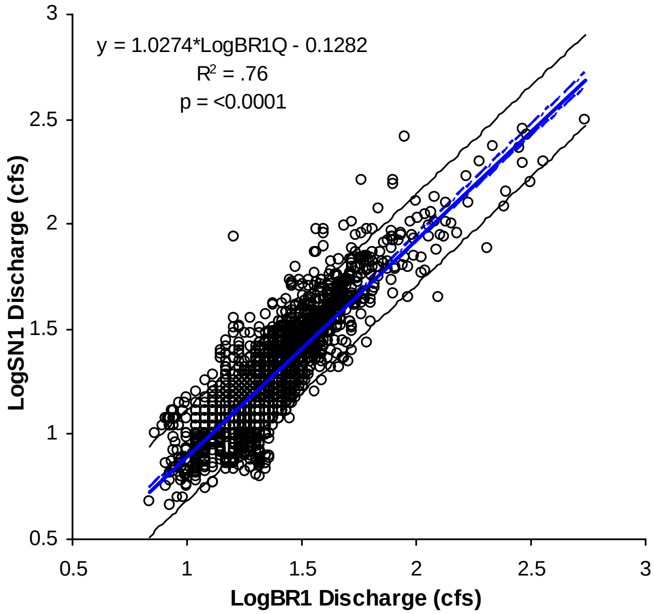


Figure 6. Regression between logs of Sny Magill and Bloody Run discharge during water years 1992-2001.

Sny Magill and Bloody Run extending from the upper reaches to the lower reaches of both watersheds were compared, (SN3-BRSC, upstream sites; SN2-BR2, midstream sites; and SN1-BR1, downstream sites). These sites were chosen based on their similarity in size and location in the watershed.

Water Quality Changes and Trends

Discharge

During the extent of the study, USGS monitoring stations continuously measured stream discharge at sites SN1 and BR1. A scatterplot (Figure 6) of average daily stream flow at both stations indicates that the ratio of discharge between the streams was highly correlated (R^2=0.76, p=<0.0001). A slope approaching 1.0 (1.027) indicates that the quantity of flow in the streams was also similar.

Although flows between the streams were similar, Sny Magill usually had less discharge than Bloody Run. Figure 7 illustrates the average daily discharge in cubic feet per second (cfs) for both SN1 and BR1 in boxplot form. For both locations, Water Year (WY) 1993 had the greatest median daily discharge, with 31 cfs measured at SN1 and 35 cfs at BR1. This correlates with record rainfall and flooding that occurred throughout much of the Midwest in 1993. The lowest median discharge for both streams occurred during WY 1997.

Pre/Post Model. Pre/post multiple linear regression on daily log transformed discharge data indicated that discharge increased at SN1 by 8%, or from 21.2 cfs in the calibration period to 22.9 cfs in the treatment period (p=<0. 0001). An autoregressive model with a lag of 1 was used to remove autocorrelation in the data.

Gradual Change Model. Gradual change multiple linear regression on daily log transformed data indicated that during the project, discharge at SN1 increased by 12%, or from a daily average of 16.8 cfs to 18.8 cfs. The confidence level is 97% (p=0.0327). An autoregressive model with a lag of 1 was used to remove autocorrelation in the data.

It is unclear why discharge in Sny Magill showed an increase during the study. Multiple regression analysis indicated no significant change on cumulative monthly precipitation during the project. Hydrograph separation indicated no significant change in either the baseflow or surface

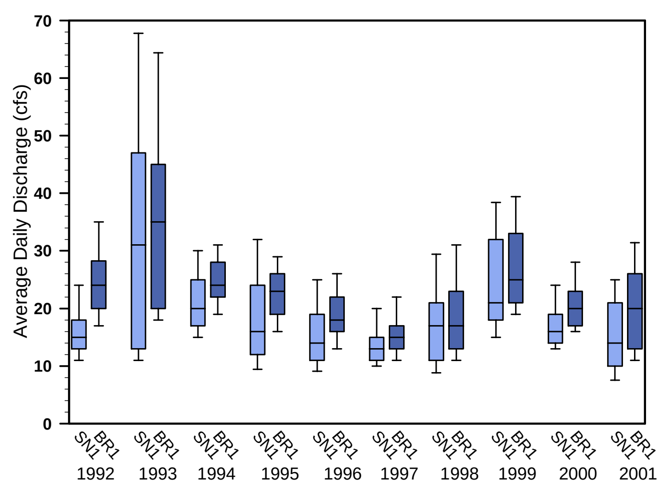


Figure 7. Average daily discharge at SN1-BR1.

flow components of discharge at SN1. A possibility for the increase in discharge could be the increased number of tiled terraces and tile drainage in the watershed. Tiles quickly remove water pooling on the surface and pipe it to the stream channel. In a high topography area the tile discharge response time to rainfall could be nearly immediate. Although tile line discharge is typically categorized as baseflow, rainfall runoff in tiled terraces may be relatively fast, and contribute to both surface and baseflow components of discharge. Tile discharge response to rainfall is also highly dependent on both the topography and tile structure in the watershed.

Suspended Sediment and Turbidity

Sediment, both suspended in the water column and transported in the bedload of the stream channel, is the major nonpoint source pollutant by volume in Midwestern streams (Iowa Dept. Natural Resources, 1997). Sediment transport in streams is a function of surface runoff, streambed erosion, and stream bank erosion. Row crop agriculture and pasture are the primary land-use practices that contribute to sediment flux in streams nationwide, accounting for 38 and 25% of the total sediment loss, respectively (Welsch, 1991).

Total suspended solids (TSS) was measured daily at monitoring sites SN1 and BR1. The sites alternated having the highest and lowest annual median concentrations during the 10-year project (Figure 8). TSS trends followed discharge trends in both watersheds and showed a steady decline in values from 1993 until 1996-1997, with an upward trend during the last three years. The highest yearly TSS median for SN1 was the last year of the study (2001) at 35 mg/L. The second highest median concentration occurred during the flood of 1993 at 28 mg/L. 1993 also had the highest single-day concentrations measured during the study. The lowest median TSS level for SN1 was 15 mg/L during WY 1996. Suspended sediment concentration at BR1 closely resembled SN1. The highest median value measured at BR1 was 42 mg/L during 1993. The lowest was 13 mg/L in 1996.

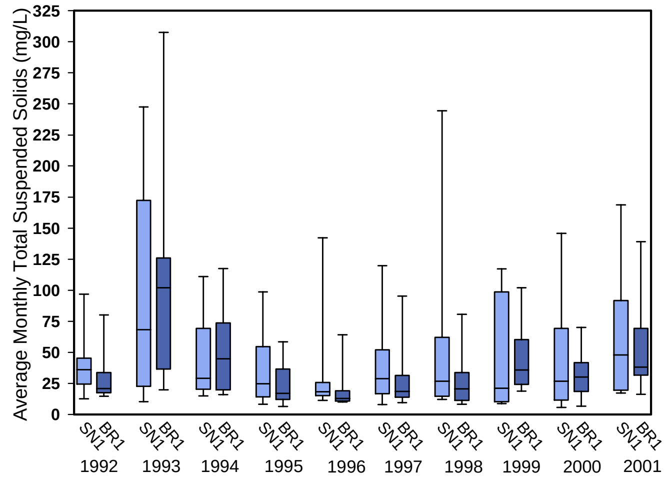


Figure 8. Total suspended solids at SN1-BR1.

Pre/post model. Pre/post multiple linear regression model indicated that TSS levels decreased at SN1 by 7%, or from 37.8 mg/L during the calibration to 35.1 mg/L during the treatment period (p=<0.0001).

Gradual change model. The gradual change multiple linear regression indicated no statistically significant change in the suspended sediment concentration at SN1 (p=0.6883).

Analysis of the monitoring data suggests a significantly more modest reduction in TSS concentrations than would seem to be implied by the estimated 50% reduction in sediment delivery suggested by the Universal Soil Loss Equation (USLE). However, the USLE doesn’t consider stream bed and bank erosion, which will continue even if upland erosion-control structures have decreased sediment delivery. Therefore, after sediment reducing structures are put in place, there will be a certain amount of time before the reduction in sediment delivery is observed in the stream itself. Depending on the size of the watershed, stream water will continue to scour out previously deposited sediments along the stream channel for many years. Compounding the issue of suspended sediment transport may be the significant increase in discharge in Sny Magill during the study. Because suspended sediment concentration in the stream is highly dependent on stream flow, any significant

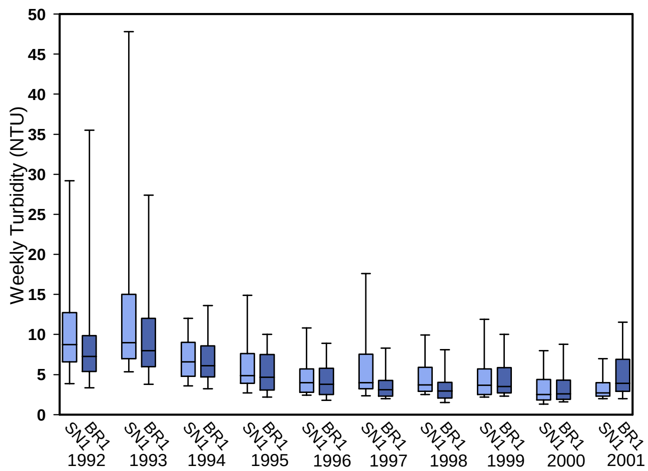


Figure 9. Turbidity measured at SN1-BR1.

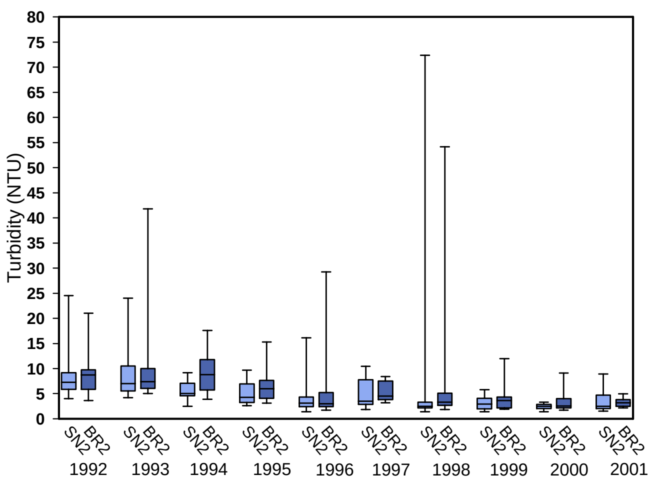


Figure 10. Turbidity measured at SN2-BR2.

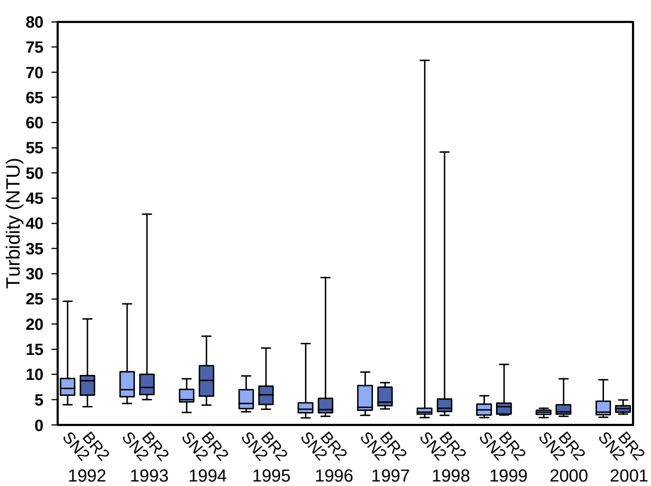


Figure 11. Turbidity measured at SN3-BRSC.

increase in discharge will increase the stream's sediment carrying capacity.

Turbidity, like suspended sediment, is a measure of particles carried in the water column. However, turbidity is a qualitative measure of scattered and absorbed light, not a quantitative measure of concentration. Turbidity measurements indicate the amount of both suspended and dissolved solids in the water sample. Turbidity measurements are taken by transmitting a beam of light through the water sample and measuring the amount of light transmitted and comparing it to the amount of light scattered at a 90° angle. This ratio determines the amount of nephelometric turbidity units, or NTU.

All three paired monitoring stations were monitored for turbidity. Weekly samples were used for the downstream stations (SN1-BR1), whereas matched monthly samples were used for the midstream (SN2-BR2) and upstream (SN3-BRSC) stations. Like suspended solids, turbidity is heavily dependent on discharge. However, turbidity levels at all three stations did not follow the same general trend as discharge. Instead, values from both streams showed a steady downward trend throughout all ten years (Figures 9-11). Median SN1 turbidity levels were higher than BR1 levels from WY 1992 through 1999. In 2000-2001, concentrations shifted and median BR1 turbidity levels became higher than SN1. The highest median turbidity for both streams was 1993 during the high flows and repeated flooding.

Pre/Post Model. Pre/post multiple linear regression indicated that turbidity decreased at SN1 by 41%, or from 7.3 NTU during the calibration period to 4.3 NTU during the treatment period (p=<0.0001). A reduced autoregressive model with a lag of 2 was used to eliminate autocorrelation in the data.

Gradual Change Model. Gradual change multiple linear regression indicated that at a statistically significant level of 99%, turbidity at SN1 decreased 46%, or from 7.1 NTU to 3.9 NTU during the project period. An autoregressive model with a lag of 1 was used to reduce autocorrelation in the data.

Turbidity values at the midstream stations were generally lower than corresponding values at the downstream stations. The mean value for SN2 was 5.9 NTU, 26% lower than the SN1 average of 8.0 NTU. A similar, but smaller, trend is visible in Bloody Run Creek, with an increase from 7 NTU at BR2 to 7.3 NTU at BR1 (4%). Turbidity trends followed the same trend seen in the downstream paired stations, with the highest values occurring during the first three years followed by a steady decline (Figure 10). WY 1998 had a few extremely high values from samples collected during a large flood event in the spring of that year.

Pre/Post Model. Pre/post multiple linear regression (reduced model) on monthly data from SN2 indicated that turbidity dropped from 5.3 NTU during the calibration period to 3.5 NTU during the treatment period, or by 34% (p=0.0042).

Gradual Change Model. Gradual change multiple linear regression indicated that turbidity decreased from 4.7 NTU at the beginning of the study to 3.6 NTU at the end, or by 24%. The level of confidence for this is only 92% (p=0.0822).

Temporal turbidity trends at the upstream sites closely followed the patterns seen in the midstream and downstream stations (Figure 11). Average turbidity levels, however, were lower. The SN3 stations had an average of 5.4 NTU, lower than both SN2 (5.9 NTU) and SN1 (8.0 NTU). This is most likely the result of less discharge in the upper region of the watershed. Bloody Run Creek turbidity followed a downstream trend similar to Sny Magill. BRSC has a lower average value (6.6 NTU) than BR2 (7.0 NTU) and BR1 (7.3 NTU). As seen at SN2, extremely elevated turbidity levels were noted during 1998. This was the result of sampling during a high flood period.

Both pre-post model and gradual change models indicated no statistically significant change in turbidity values at SN3 during the study. Turbidity values at this station were already very low at the beginning of the study, potentially limiting the measurement of any decrease in levels.

There is no national standard turbidity level for trout streams, and Iowa does not currently have a statewide turbidity standard. However, many states around the country, including Minnesota, have established 10 NTU as a water quality standard for cold water trout streams. Yearly mean turbidity levels in all three paired monitoring locations were never above the 10 NTU level. The large decrease in the mean turbidity levels seen at the midstream and downstream monitoring stations (SN2 and SN1) indicate that water clarity significantly improved in Sny Magill.

The decrease in turbidity is much larger then the decrease in suspended sediment, indicating that a decrease in a variable not measured in suspended sediment had an impact on turbidity levels. This could be smaller particles flushing out of the system before larger particles. These particles could be lighter weight organics, or colloidal sized particles. Particles smaller than 45 µm are considered total dissolved solids (TDS), rather than TSS, and may contribute to turbidity.

Nutrients

Nutrients, primarily nitrogen and phosphorus, are essential for plant and animal growth. However, when found at elevated levels nutrients can degrade the quality of water and cause a condition called eutrophication. Algal blooms and excessive growth in other aquatic plants are associated with eutrophication of a water body. The subsequent death and decay of plant matter can augment environmental problems by reducing dissolved oxygen levels, leading to hypoxia, anoxia, fish kills, foul odors, and bad water taste. Usually, streams themselves are minimally affected by elevated nutrients. Instead, eutrophic streams contribute to the pollution of stagnant water bodies such as ponds, lakes, or oceans. Nutrient concentrations are given in parts per million (ppm) or milligrams per liter (mg/L).

Nitrate+nitrite-N (NOx-N) was measured at all three paired locations. SN1-BR1 measurements were taken weekly; paired SN2-BR2 and SN3-BRSC measurements were taken monthly. Figure 12 illustrates annual NOx-N concentrations at SN1 and BR1. Both the treatment (Sny Magill) and control (Bloody Run) streams had an upward trend throughout all 10 years, with SN1 having around

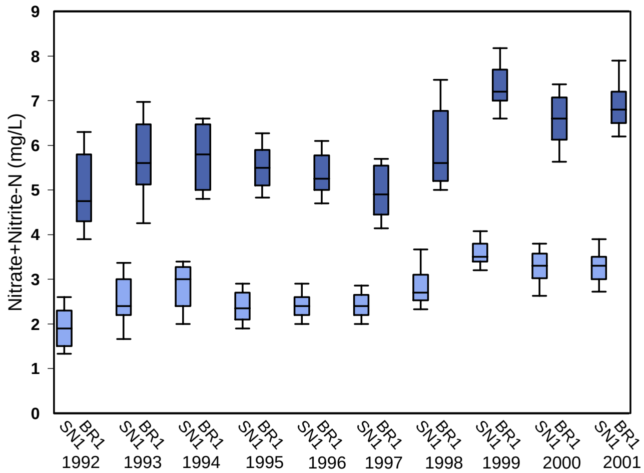


Figure 12. NOx-N concentrations measured at SN1-BR1.

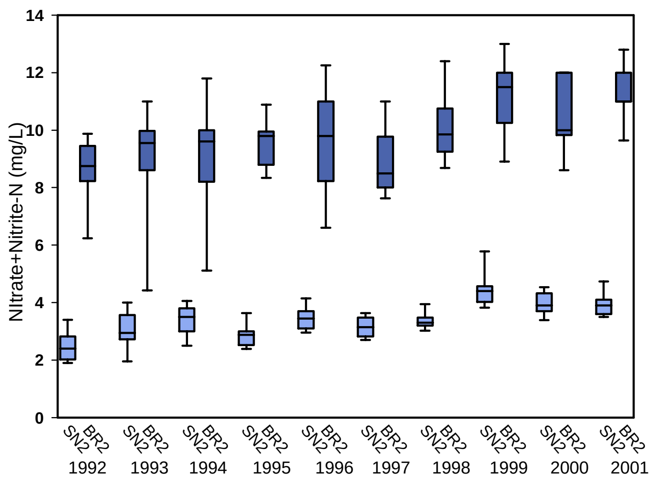


Figure 13. NOx-N concentrations measured at SN2-BR2.

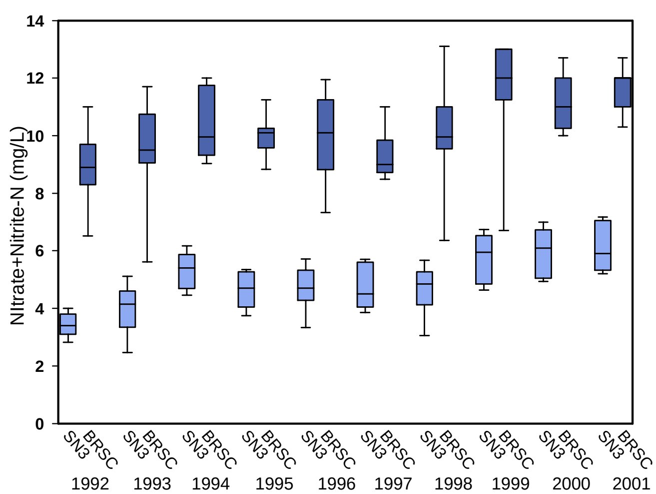


Figure 14. NOx-N concentrations measured at SN3-BRSC.

half of the concentration of BR1. This lower concentration measured at SN1 likely results from the Sny Magill watershed having less row crop acres than Bloody Run. Like suspended solids, NOx-N trends closely resembled discharge. Medians at SN1 ranged from a low of 1.9 mg/L in 1992 to a high of 3.5 mg/L in 1999, while BR1 ranged from 4.75 mg/L in 1992 to 7.2 mg/L in 1999. Annual NOx-N concentrations showed a relationship to annual discharge, with a gently downward trend until 1998, when concentrations increased and leveled off during the last three years of the study.

Pre/Post Model. Pre/post multiple linear regression indicated that NOx-N concentrations at SN1 increased 15%, or from 2.7 to 3.1 mg/L from the calibration period to the treatment period ($p=<0.0001$). A reduced autoregressive model with a lag of 1 was used to decrease autocorrelation in the data.

Gradual Change Model. Gradual change linear regression on the data indicated that mean NOx-N concentrations at SN1 increased 39% (from 2.3 mg/L to 3.2 mg/L) during the study period ($p=<0.0001$). A reduced autoregressive model with a lag of 1 was used to decrease autocorrelation in the data.

The large difference between the two models (increase of 15% in the pre/post model compared to 39% in the gradual change model) is due to the extremely low values seen at SN1 during WY 1992. The pre-post model averaged these values with the higher concentrations during WY 1993-94. The gradual change model incorporated the values for the estimation of the initial concentration.

NOx levels at the midstream paired sites were generally higher than the downstream sites (Figure 13). The mean concentration of nitrate+nitrite-N at BR2 was 9.76 mg/L, 66% higher than the average of 5.89 mg/L at BR1. The difference between SN2 and SN1 was less. The average value at SN2 throughout all ten years was 3.4 mg/L, only 24% higher than the 2.75 mg/L average at SN1. The trends between both streams are similar however, with a steady increase in NOx-N values throughout all ten years. As in SN1-BR1, SN2 had a lower

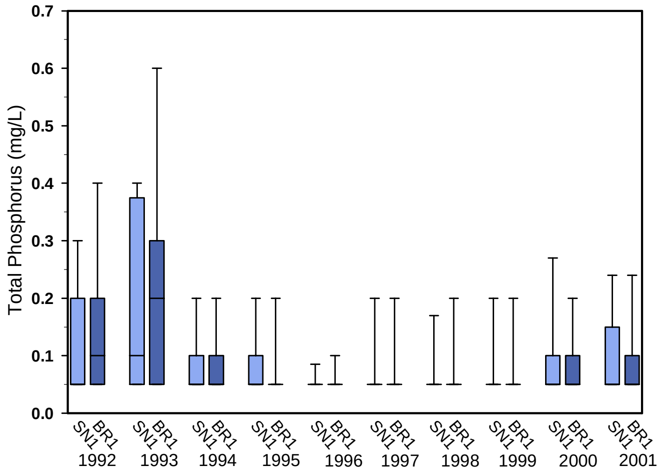


Figure 15. TP concentrations measured at SN1-BR1.

concentration than BR2. The difference was a little more accentuated, however, as SN2 had one-third the concentration of BR2. Concentration values at SN2 did not fluctuate as much as BR2, indicating a more stable source of nitrate+nitrite-N concentration in the stream.

Pre/Post Model. Pre/post multiple linear regression on SN2 (reduced model) indicated that NOx levels increased 26% between the calibration and treatment periods, or from 3.1 mg/L in the calibration period to 3.9 during the treatment ($p=<0.0001$) *Gradual Change Model.* Gradual change multiple linear regression indicated that from the beginning of the study period until the end, NOx concentrations at SN2 increased by 43%, or from 2.8 mg/L to 4.0 mg/L ($p=<0.001$).

The general upward trend of nitrate+nitrite-N at SN3-BRSC during the study was the same as noted at the downstream monitoring sites (Figure 14). However, the 10-year average NOx-N value at SN3 (4.5 mg/L) was 33% higher than SN2 (3.4 mg/L) and 61% higher then SN1 (2.8 mg/L). The same general trend is also seen in Bloody Run Creek. The lowest median NOx concentration in SN3 was 3.4 mg/L during 1992. The highest median was 6.1 mg/L in 2000. The lowest BRSC NOx value was also in 1992 at 8.9 mg/L. The highest median value was in 1999 at 12 mg/L.

`*Pre/Post Model.* Reduced pre/post multiple linear regression indicated an increase of NOx-N at SN3 by 39%, from 4.0 mg/L during the calibration period to 5.5 mg/L during the treatment period ($p=<0.0001$). *Gradual Change Model.* Gradual change multiple linear regression indicated an increase in NOx-N at SN3 by 37%, or from 4.1 mg/L to 5.6 mg/L to the end of the study ($p=<0.0001$).

The increase in NOx-N in Sny Magill, relative Bloody Run, is problematic. Part of the increase might be attributed to the increased use of drainage tiles in the watershed. Tiles added as part of terrace systems may be more efficiently transporting NOx-N from the soil column to the stream channel, circumventing plant uptake or transformations such as denitrification. Another possibility is the transport of NOx-N from areas not influenced by the extensive BMP implementation. The groundwater capture zone for both Sny Magill Creek and Bloody Run is also poorly understood. It is known that the Big Spring groundwater basin undercuts part of Bloody Run's surface watershed; but the true dimensions of both groundwater basins have not been mapped. NOx-N could potentially be transported from outside the surface watershed area, where extensive BMPs were not put in place.

Total Phosphorus (TP) was measured at the SN1-BR1 paired monitoring stations. Throughout the study, TP concentrations in Sny Magill Creek were normally lower than the detection limit (Figure 15). This limited the assessment of changes in total phosphorus concentrations. Except for Water Year 1993, median concentrations in both streams were below the recommended EPA level of 0.1 mg/L. In general, TP levels decreased in both streams during the mid to latter part of the project, with concentrations continuously below detection limits from 1996 until 1999.

Fecal Coliform Bacteria

Fecal coliform bacteria serve as an indicator of fecal contamination. Fecal coliform levels are highly variable, as the organisms are dependent on storm events and temperature for transport and survival.

Figure 16 lists the fecal levels at monitoring

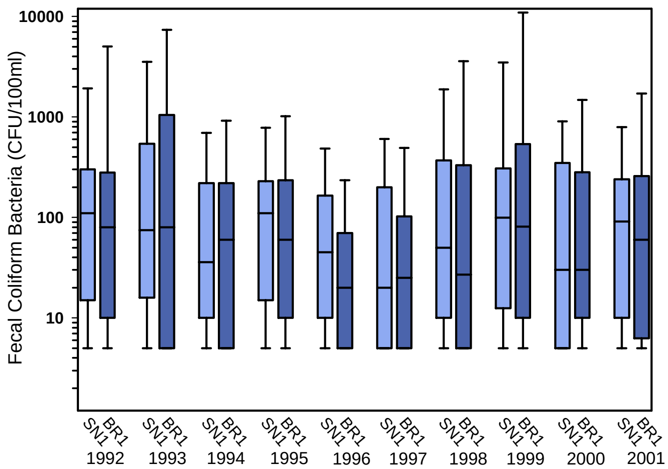


Figure 16. Fecal coliform measured at SN1-BR1.

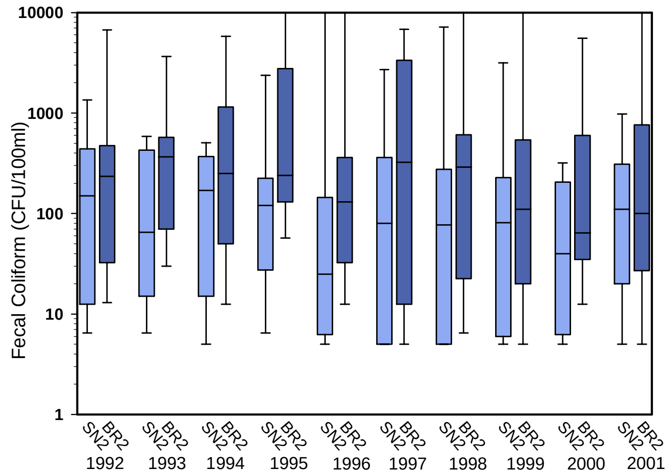


Figure 17. Fecal coliform measured at SN2-BR2.

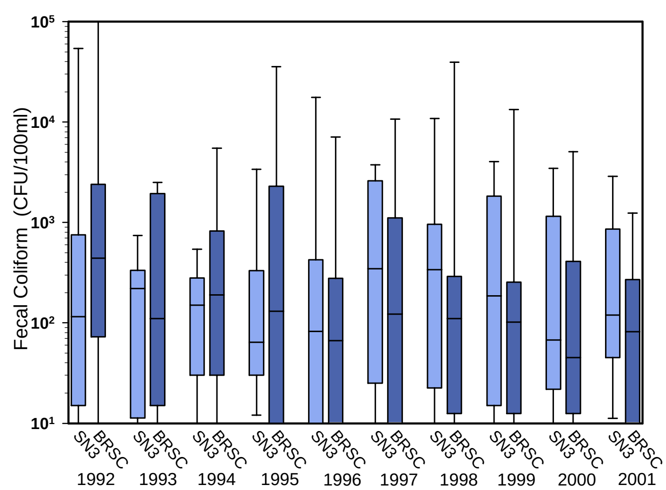


Figure 18. Fecal coliform measured at SN3-BRSC.

sites SN1-BR1 in Colony Forming Units (CFU) per 100 ml (CFU/100 ml) on a log scale. Median fecal coliform levels fluctuated, with SN1 having higher levels some years and BR1 having higher levels other years. The highest median concentration for SN1 was 110 CFU/100 ml in 1992. BR1 had the highest median level at 81 CFU/100 ml in 1999. 1997 had the lowest measured median values for both stream, with SN1 having 20 CFU/100 ml and BR1 having 25 CFU/100 ml.

Pre/Post Model. Pre/post multiple linear regression on SN1-BR1 indicated a decrease of 12%, or from 83 CFU/100 ml during the calibration period to 73 CFU/100 ml during the treatment period (p=0.0186)
Gradual Change Model. Gradual change multiple linear regression indicated no statistically significant change in Fecal Coliform throughout the study period (p=0.52)

Fecal coliform at SN2 was consistently lower than at BR2 (Figure 17). The average count of fecal coliform at SN2 was 546 CFU/100 ml, one sixth of the BR2 fecal count of 4649 CFU/100 ml. Sny Magill's fecal count doubled downstream, with station SN1 having an average of 875 CFU/100 ml. Conversely, fecal coliform in Bloody Run decreased downstream, having a ten-year average of 2337 CFU/100 ml. The highly erratic nature of fecal coliform survival and transport in water limits the speculation on reasons for the large shift in fecal counts downstream.

Both pre/post and gradual change models indicate no statistical change in fecal coliform levels at SN2 during the study period.

Fecal coliform levels at SN3 and BRSC varied greatly each year (Figure 18). SN3 had generally higher values than BRSC, except for WYs 1992, 1994 and 1995. BRSC had an extremely high value of 1,100,000 CFU/100ml on October 29, 1991. Other results were less than half of that value. The highest median value for SN3 occurred in 1997 at 345 CFU/100 ml. After the peak in 1997, median levels declined until 2001. The lowest median value for fecal coliform at SN3 was 64 CFU/100 ml during 1995. BRSC had its highest median during 1992 with 440 CFU/100 ml, lowest value in 2000 at 45 CFU/100 ml. Fecal contamination of water greatly decreased downstream. SN3's average value of 1530 CFU/100 ml was almost 3 times higher than the SN2 value of 546 CFU/100 ml, and twice as high as SN1 at 875 CFU/100 ml. BRSC's average fecal coliform level of 11,174 CFU/100 ml was over twice as large as the BR2 value of 4,649 CFU/100 ml, and four times larger then the BR1 value of 2337 CFU/100 ml.

Pre/Post Model. Reduced pre/post multiple linear regression indicated that the treatment fecal coliform level at SN3 was 192% larger than the calibration level, and increased from 75 to 218 CFU/100 ml (p=0.0049).

Gradual Change Model. Gradual change multiple linear regression indicated that from the beginning of the study period until the end fecal coliform levels at SN3 increased from 56 CFU/100 ml to 316 CFU/100 ml, or 464% (p=0.00005).

Fecal coliform levels in Sny Magill did not show a large decrease during the study period. In fact, fecal coliform levels at SN3 increased dramatically. This most likely is due to no significant decrease in animal feeding operations and lack of manure management structures in Sny Magill.

Pesticides

Figure 19 shows triazine herbicide concentrations at sites SN1 and BR1. Concentrations below the detection limit and high seasonality impeded both proper gradual change and pre/post regression analysis. Triazines were usually found at measurable concentrations during the last two weeks of May and the first two weeks of June. This was almost certainly due to application of triazine herbicides during this time. BR1 had consistently higher concentrations than SN1, most likely resulting from the greater number of rowcrop acres in Bloody Run. Like NOx-N, concentrations tended to follow annual discharge patterns, indicating a high reliance on flow for transport.

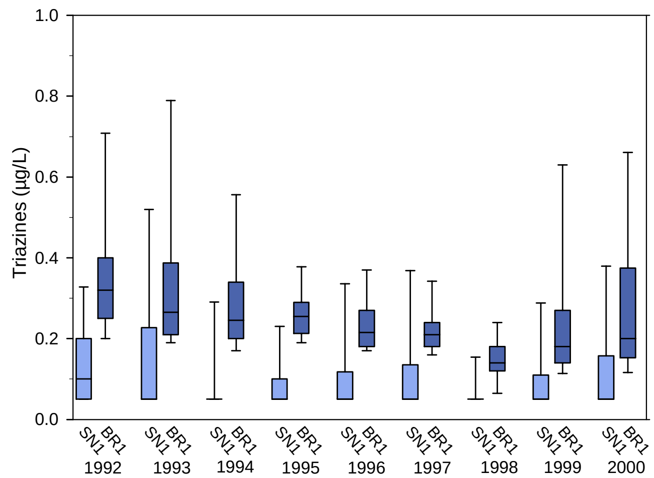


Figure 19. Triazine concentrations at SN1-BR1.

Temperature and Dissolved Oxygen

Both temperature and dissolved oxygen are extremely important factors for the survival of aquatic life. This is especially true of trout streams, where cold water temperatures and high dissolved oxygen levels are a prerequisite for sustained trout populations.

Temperature at both SN1 and BR1 monitoring sites stayed in the same range throughout all ten years (Figure 20). The lowest median temperature measured in the streams was during WY 1993. The median temperature at SN1 was 8°C and 7.5°C at BR1. This was also the only year in which BR1 had a lower median temperature than SN1. The highest median temperature occurred during water year 1995 and was 11°C for both sites. Temperature measurements at SN1 and BR1 were typically taken within the same hour to limit diurnal variation.

Pre/Post Model. Pre/post multiple linear regression indicated that temperature decreased at SN1 by 4%, or from 9.9°C to 9.5°C (p=<0.0002).

Gradual Change Model. Gradual change multiple linear regression indicated that temperature at SN1 decreased from 9.9°C at the beginning of the study to 9.4°C at the end of the study, or approximately 5% (p=0.0087).

Temperature at SN2 was generally higher than

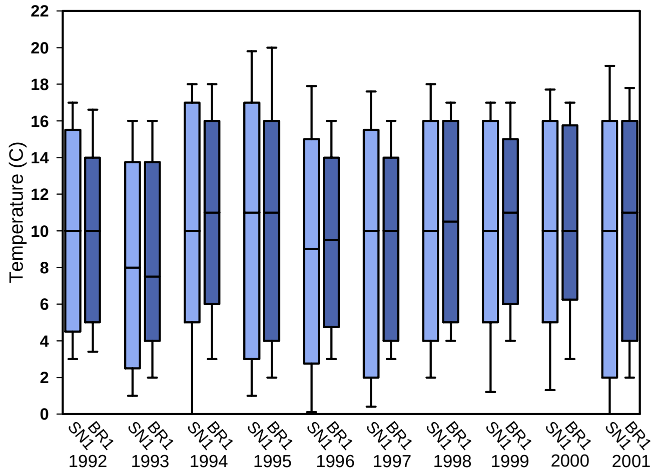


Figure 20. Temperature at SN1-BR1.

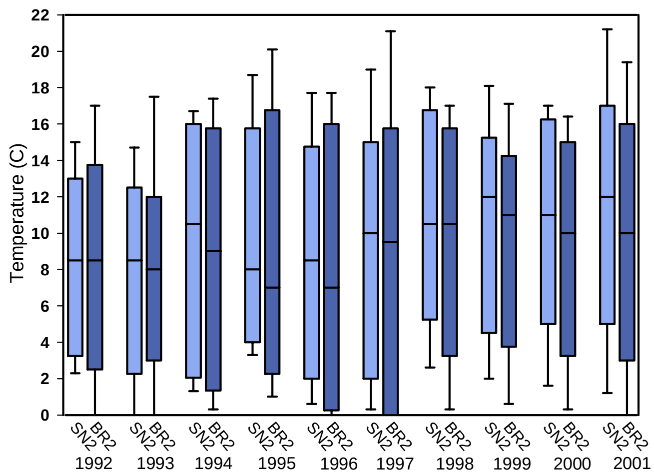


Figure 21. Temperature at SN2-BR2.

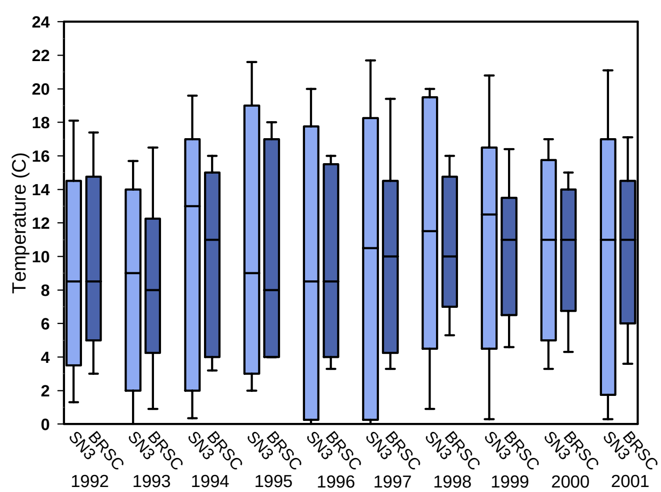


Figure 22. Temperature at SN3-BRSC.

at BR2 (Figure 21). This was most likely due to the time of measurement rather than an increase in water temperature. SN2 water temperature was generally taken in the mid afternoon (2-3 pm), while BR2 temperature was usually taken during the late morning hours (10-11 am).

Pre/Post Model. Pre/post multiple linear regression at SN2 indicated that the calibration temperature was 9.0°C, and increased by 10% to the treatment period to 9.9°C, (p=0.0249).
Gradual Change Model. Gradual change multiple linear regression indicated that temperature increased at SN2 from 8.8°C to 10.3°C or 17% from the beginning of the study until the end (p=0.0003).

Temperatures at the SN3-BRSC paired stations were comparable to the rest of the stations, averaging 10°C (Figure 22). Temperature was usually measured at both stations during the early afternoon (1-2 pm).

Pre/Post Model. Pre/post multiple linear regression indicated that temperature decreased at SN3 by 1%, from 9.7°C to 9.6°C (p=0.0001).
Gradual Change Model. Gradual change multiple linear regression indicated no significant increase or decrease in temperature at SN3 (p=0.7696).

Temperature fluctuated in Sny Magill Creek, with an increase at SN2 and decrease at both SN1 and SN3. This reflects the high variability of temperature in a stream, and also shows how sampling times can affect temperature readings. Since there is no general trend between all the monitoring sites, it is difficult to explain the variations. Part of the reason for the measured decrease in the stations could be attributed to greater riparian vegetation along the stream channel. Greater vegetation cover leads to more shade and less heating from the sun. Temperature fluctuations should not always be attributed to changes in the stream channel however, as time of sampling also plays an important role.

Dissolved oxygen (DO) levels steadily increased at both streams throughout the study (Figure 23). The lowest median DO values in both monitoring sites was 10 mg/L in WYs 1992 and 1993. After these first two years, DO increased steadily at SN1 until the end of the study, with WY 2001 having the highest concentration at 14 mg/L. Higher concentration of dissolved oxygen levels are, in general, good for aquatic organisms.

Pre/Post Model. Pre/post multiple linear regression indicated that DO at SN1 increased by 11%, from 10.7 mg/L during the calibration period to 11.9 mg/L during the treatment period (p=<0.0001).
Gradual Change Model. Gradual change multiple linear regression at SN1 indicated that DO increased during the study by 16%, from 10.9 mg/L to 12.6 mg/L at the end (p=<0.0001).

DO levels at SN2 and BR2 both increased relative to SN1 and BR1 (Figure 24). WY 1992 had the lowest DO level for SN2, with 9.0 mg/L. BR2 had the lowest value in WY 1993 with 8.5 mg/L. The highest median value for SN2 was 13 mg/L in 2001. BR2 had high values in 1998, 1999, and 2000 with an average of 13.5 mg/L. Average DO concentrations between SN2 and SN1 are very similar, with 11 mg/L at SN2 and 12 mg/L at SN1. BR1 and BR2 also have similar DO concentrations, at 12 mg/L for each.

Pre/Post Model. Pre/post multiple linear regression indicated that DO at SN2 increased 2%, or from 10.4 mg/l during the calibration period to 10.6 mg/L during the treatment period (p=0.0151).
Gradual Change Model. Gradual change multiple linear regression indicated that DO increased 26% at site SN2, from 9.8 to 12.3 mg/L (p=0.02).

DO levels at SN3-BRSC increased substantially during the study (Figure 25). WY 1992 and 1993 had the lowest levels found in SN3, with a median of 9 mg/L both years. DO steadily increased during the study to its highest median of 14.5 mg/L in 2001. BRSC showed a similar trend. The lowest yearly median was 10 mg/L during 1992 and levels increased to 14 mg/L at the end of the study.

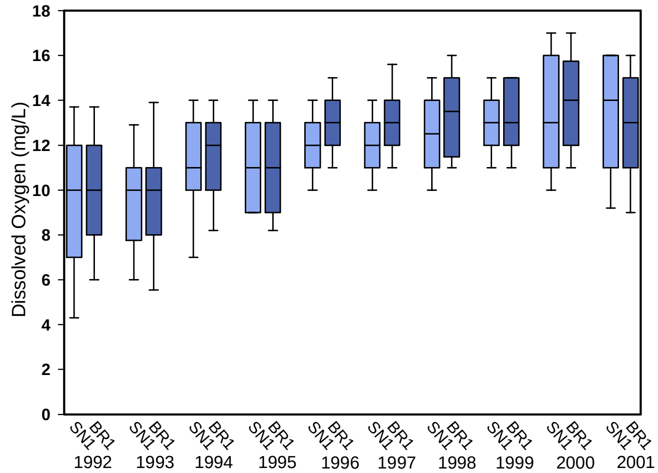


Figure 23. Dissolved oxygen at SN1-BR1.

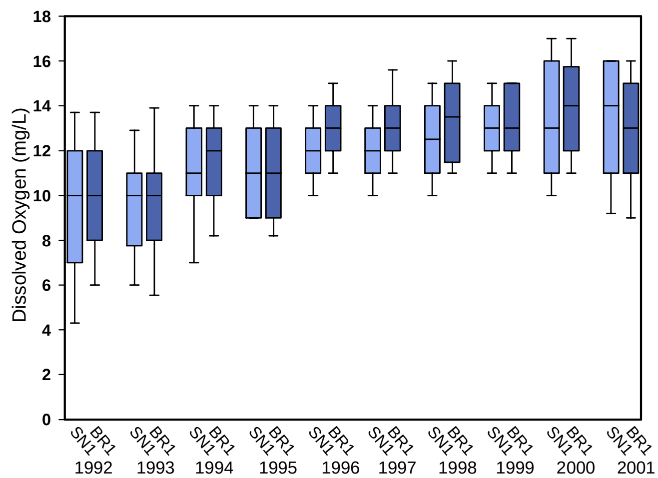


Figure 24. Dissolved oxygen at SN2-BR2.

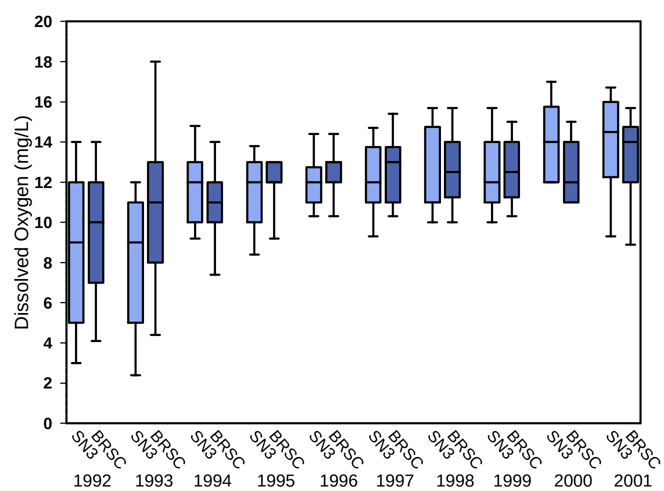


Figure 25. Dissolved oxygen at SN3-BRSC.

Table 4. Overall changes in water quality for Sny Magill sites when compared to their Bloody Run monitoring site.

Location	Discharge		Sediment		Turbidity		NOx		Fecal Coliform		Temperature		DO	
	P/P	Grad.	P/P	Grad.	P/P	Grad.	P/P	Grad.	P/P	Grad.	P/P	Grad.	P/P	Grad.
SN1	↑ 8%	↑ 12%	↓ 7%	NS	↓ 41%	↓ 46%	↑ 15%	↑ 39%	↓ 12%	NS	↓ 4%	↓ 5%	↑ 11%	↑ 16%
SN2	-	-	-	-	↓ 34%	↓ 24%	↑ 26%	↑ 43%	NS	NS	↑ 10%	↑ 17%	↑ 2%	↑ 26%
SN3	-	-	-	-	NS	NS	↑ 39%	↑ 37%	↑ 192%	↑ 464%	↓ 1%	NS	↑ 28%	↑ 36%

P/P = Pre/Post model; Grad. = Gradual Change model; NS = Not Significant

Pre/Post Model. Pre/post multiple linear regression indicated that DO at SN3 increased 28%, from 9.3 mg/l during the calibration period to 11.9 mg/L during the treatment period (p=0.0294).
Gradual Change Model. Gradual change multiple linear regression at SN3 indicated from the beginning until the end of the project DO increased 36%, from 9.7 to 13.2 mg/L (p=<0.0001).

The increase in DO levels can be attributed to two distinct factors. The first is that temperature at SN1 and SN3 had measurable decreases during the study, which in turn increased the level at which water becomes saturated with oxygen. The other factor is the lower turbidity levels in the stream. The low turbidity makes sunlight able to reach the aquatic plants, causing photosynthesis and the production of oxygen in the stream. DO and temperature levels indicate that water in Sny Magill Creek is almost always at or near the saturation point.

Paired Monitoring Summary

Table 4 summarizes changes in water quality that were observed in all paired monitoring stations at Sny Magill. Overall, the large decrease in turbidity and slight decrease in suspended sediment suggest that BMP implementation can measurably improve water quality, even in a relatively healthy trout stream. The significant increase in discharge and nitrate-nitrite-N, however, indicates that there were other effects associated with BMP implementation that are not fully understood.

A possibility for the lack of conclusive results is a greater lag time between BMP installation and water quality improvements than was measured during this study. After BMP implementation there is a certain amount of time needed for the practices to reach their maximum level of effectiveness. Deposits of sediment in the stream channel could also take a while to leave the watershed. Although USLE estimates a 50% reduction in sediment transport from the uplands, sediment already deposited in the stream channel can be re-suspended many times before leaving the system. It is not yet known how long this takes.

Lag time of groundwater travel through the watershed is also not completely understood. Water-soluble nutrients, such as nitrate and nitrite, can infiltrate deep into the groundwater system before entering the stream channel. Groundwater usually moves more slowly than surface water and has a greater residence time before entering the stream. Although groundwater residence times were not measured for this project, studies have indicated baseflow can reside up to decades before resurfacing and entering a surface water body (Katz et al., 1999; Phillips et al., 1999). Although karst regions and tile drainage can greatly reduce groundwater travel time, surface influence on groundwater can still last an extended period of time. In future studies it is essential to understand the groundwater capture zone, residence time and transport, especially in baseflow dominated streams.

Tributary Monitoring

Along with the three paired locations on the main channels of Sny Magill and Bloody Run Creeks, even-interval water monitoring was also done on Sny Magill tributaries. These streams included North Cedar Creek (NCC), West Fork

(SNWF), and an unnamed tributary (SNT). Data collection at these sites were limited; therefore only turbidity and NOx-N are provided in this report.

Nitrate+nitrite-N values (Figure 26) tended to be lower in the tributaries than in the main channel of the stream. Of the sites monitored as part of this study, NCC had the lowest levels of NOx (2.8 mg/L). SNWF and SNT had comparable concentrations at 3.5 and 3.8 mg/L, respectively. Timeplots indicate that NOx increased in the tributaries, just as in the paired stations. However, NCC had much less of an increase than any other stream, and values did not dramatically increase the last three years of the study, as in the other stations. This could be attributed to the longer time that NCC was in a BMP implementation phase (since 1988) compared to the rest of the watershed.

Turbidity levels in the tributaries also closely resembled levels found in the main channel (Figure 27). The lowest turbidity readings were at SNWF (5.7 NTU). NCC and SNT had 6.5 and 6.1 NTU, respectively. This is most likely due to much lower discharge than actual BMP implementation. In all sites, however, there was a dramatic decrease in the turbidity levels throughout the ten-year period. This is surprising given the general increase in flow during the last few years of the study (1998-2001). The drop in turbidity levels must be attributed to something other than discharge, most likely the reduction of dissolved solids in the water column due to BMP implementation.

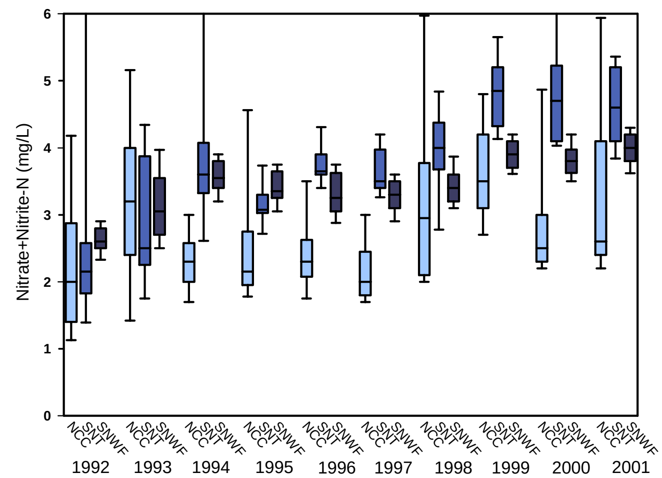


Figure 26. NOx-N in Sny Magill tributaries.

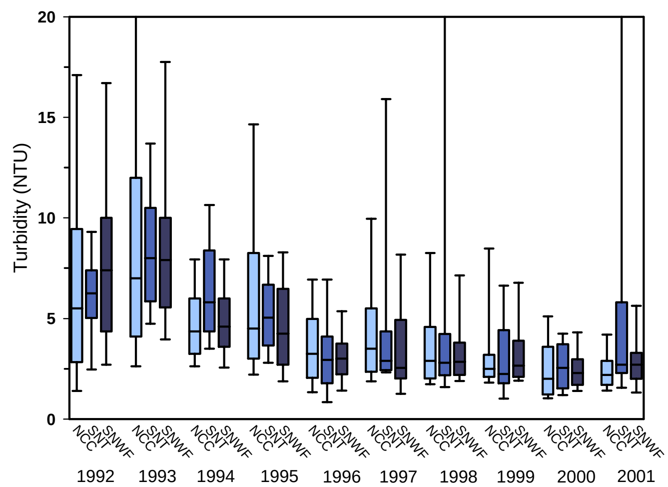


Figure 27. Turbidity in Sny Magill tributaries.

PHYSICAL HABITAT, BENTHIC MACROINVERTEBRATES, AND FISH ASSESSMENT

In addition to water quality monitoring, Sny Magill and Bloody Run watersheds were also tested on an annual basis for changes in habitat, benthic macroinvertebrates, and fish assessment. These parameters were measured to quantify any changes in natural habitat to Sny Magill Creek during the study period. Detailed yearly summaries of physical habitat, benthic macroinvertebrates, and fish assessments are available from Seigley and others, 1994(a), 1994(b); Langel and others, 2001; Liu and others, (*in press*).

Physical Habitat Assessment

The primary objectives of the physical habitat assessment were to:

- Characterize stream physical habitat conditions.
- Examine for temporal trends in physical habitat characteristics.
- Evaluate and refine aquatic habitat evaluation methods.

Habitat evaluations were conducted in the fall

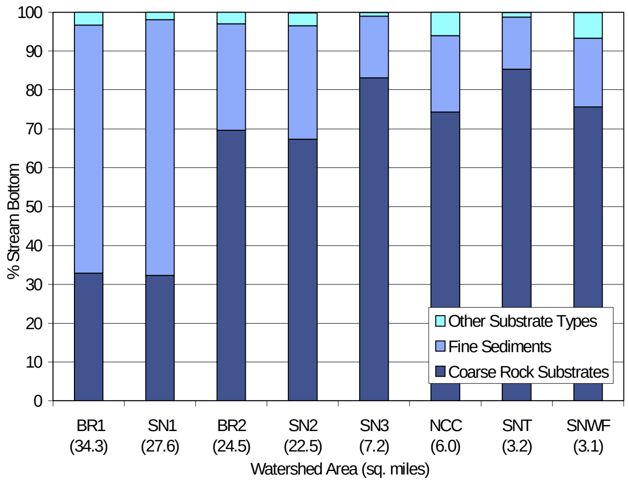


Figure 28. Relationship of watershed area and substrate composition at project monitoring sites (1991-2001).

during baseflow conditions. A suite of instream and streamside habitat variables were systematically measured or estimated at channel cross-section transects in a pre-defined stream reach. Consistent data collection and summarization methods were used throughout the ten-year monitoring project. In most years, duplicate evaluations were conducted by separate evaluators to estimate observational variability.

Physical Habitat Characterization

Stream habitat in the Sny Magill Creek and Bloody Run Creek watersheds is largely marked by sequences of riffles and pools. Riffles occur more frequently in the steeper, upper areas of the watersheds, while larger and deeper pools are more common in the lower areas of the watersheds. In the middle and upper areas of the watersheds, stream bottom composition is comprised mostly of gravel and cobble substrates (Figure 28) originating from limestone bedrock. Accumulations of fine sediment are more common in the lower reaches of Sny Magill Creek and Bloody Run Creek where the channel slope is less steep. Stream corridors generally meander underneath a canopy of mixed deciduous shrubs and trees. A relatively small amount of channel straightening has occurred adjacent to road bridges and agricultural fields. Stream bank slopes are moderate to steep and reasonably well anchored by rock and vegetation. Eroding banks are present, particularly along the outside of channel meanders formed in alluvial deposits.

Stream gradient (channel slope) at monitoring sites ranges from 28-121 feet per mile, which is relatively steep for the Midwest. Among the Sny Magill Creek watershed monitoring sites, there is a strong relationship between watershed area and stream gradient (Figure 29). The relationship is less pronounced among Bloody Run Creek monitoring sites.

As previously reported (Wilton 1994; Langel et al. 2001), instream and riparian habitat characteristics of Sny Magill and Bloody Run monitoring sites were strongly related with watershed size. Channel slope, landscape position, and watershed area can can explain many of the differences and similarities in habitat characteristics among monitoring sites, and may also explain patterns in benthic macroinvertebrate and fish species composition. Sites located in the smaller sub-watersheds (SNT, SNWF, NCC, SN3) of Sny Magill Creek shared greater similarity with each other than with sites located near the Sny Magill or Bloody Run watershed outlets (BR1, SN1). Small watershed sites tended to have more gradient (channel slope), riffles, coarse rock substrate, and shading, with less water depth and pool area compared to downstream sites. Figure 28 shows a strong gradient in substrate composition from large to small watershed sites.

Habitat assessment models were utilized to evaluate physical habitat for biomonitoring purposes and suitability to support trout species. Based on the U.S. EPA Rapid Bioassessment Protocol (RBP) (Barbour and Stribling 1991) habitat assessment model, ratings for physical habitat quality at project monitoring sites ranged from "good" to "very good." According to the RBP assessment framework, habitat differences of this magnitude are not large enough to account for more than slight differences in biological community indicators among project monitoring sites. U.S. Fish and Wildlife Service Habitat Suitability Index (HSI) models (Raleigh and others, 1984; Raleigh and others, 1986) were used to evaluate habitat quality for adult brown trout and rainbow trout. The differences in trout habitat suitability among project monitoring sites were quite large. The level of habitat suitability for adult brown trout ranged from 0.0 (unsuitable) to 1.0 (optimal). Adult rainbow trout habitat suitability ranged from 0.37 – 1.0. According to the models, the amount of instream cover (e.g., vegetation, woody debris, undercut banks), and the amount and quality of pool habitat were the variables most limiting for brown and rainbow trout habitat suitability.

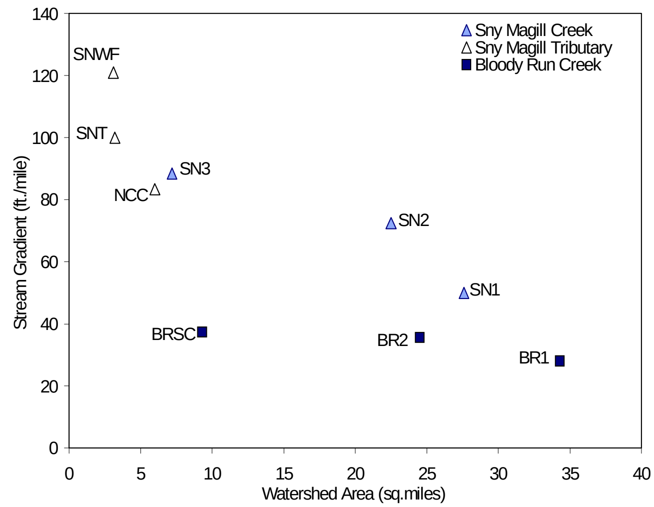


Figure 29. Relationship of watershed area and stream gradient at project monitoring sites.

Physical Habitat Trends

Habitat assessment data were examined for changes in habitat characteristics over the course of the ten-year monitoring project. Trends in physical habitat conditions are important to recognize since they can provide insight into long-term changes in stream water quality and biological assemblages. Because stream habitat characteristics are dynamic and affected by season and flow conditions, habitat assessments conducted on an annual basis are not ideal for trend monitoring purposes. From a positive standpoint, the habitat assessments were conducted using consistent methods at the same time of year throughout the project.

While conducting annual habitat assessments, relatively little change in human and wildlife activities was observed in the immediate vicinity of project monitoring sites. A few activities, which impacted small areas of the sample reaches, were bridge construction (SN3), riparian vegetation removal (SN3, SNWF), and beaver dam/stream impoundment (SN2). None of these disturbances appeared to alter habitat conditions beyond the adjacent stream areas.

At the watershed scale, it might be possible to detect changes in physical habitat characteristics in response to implementation of watershed BMPs. For example, many of the BMPs implemented in the Sny Magill Creek watershed were designed to reduce the volume of sediment delivered to stream channels. Reductions in sediment delivery over time might result in less sedimentation and improved habitat for benthic (bottom-dwelling) organisms. Potentially confounding the ability to detect trends would be the lag time between BMP implementation and corresponding reductions in sedimentation. This lag effect, as well as the ongoing contributions of sediment from stream bed and banks, could make it difficult to detect an improvement in sediment conditions during the relatively short life of the monitoring project.

The gradual change regression approach used to analyze for trends in water quality parameters was also applied to test for trends in quantitative physical habitat parameters. The regression equation was structured such that the Sny Magill Creek habitat parameter was the dependent (treatment) variable and the independent variables in the model were the corresponding Bloody Run Creek (control) habitat parameter and the number of years elapsed since initiation of monitoring. Based on the similarity in habitat characteristics and watershed position, site SN1 was paired with site BR1 for regression analysis and site SN2 was paired with BR2. Regression analysis results are discussed below. Trends in water depth and substrate composition were the most noteworthy.

Water Depth. Average depth was calculated annually from 50 individual measurements taken at ten cross-section transects in the sampling reach. During the 1991-2001 monitoring period, there was a significant decrease ($p<0.01$) in average water depth at sites SN1 and BR1 (Figure 30). The rate of change at SN1 was not significantly different than the rate of change at BR1. From the linear regression equation, average depth at SN1 decreased 27% from 63 cm to 46 cm during the ten-year monitoring period. BR1 average depth decreased (36%) from 56 cm to 36 cm in the same period.

Similar trend results for average thalweg were observed. The thalweg is the deepest line of flow through the stream channel. It was estimated by measuring the deepest point at each cross-section transect in the sample reach. Unlike average depth, however, the rate of thalweg depth reduction was significantly greater at BR1 than SN1 (Figure 31). From the linear regression equation, average thalweg depth at BR1 decreased 32% from 95 cm to 65 cm during the 1991-2001 monitoring period. The trend was smaller, but also significant ($p=0.02$) at SN1, which decreased 19% from 91 cm to 74 cm thalweg depth.

The reduction in water depth at SN1 was correlated with a significant reduction in the amount of instream cover from 70% in 1991 to 30% in 2001. Reduced pool depth, particularly in the upper one-half of the sample reach, was the most noticeable type of instream cover lost. The BR1 site also had a significant trend toward decreasing instream cover that was correlated with the SN1 trend.

Unlike BR1 and SN1, there were no trend relationships between SN2 and BR2 for either average depth or thalweg depth. Both average depth and thalweg depth levels at BR2 decreased during the monitoring period, while levels at SN2 did not change significantly.

Substrate. Despite a decreasing trend in water depth, the amount of stream bottom covered by silt did not change significantly at SN1 (Figure 32). In contrast, the stream bottom area covered by silt substrate at BR1 increased significantly from 32% in 1991 to 65% in 2001 (Figure 33). Upstream from BR1 and SN1, there were marginally significant decreasing trends in silt amount at BR2 ($p=0.07$) and SN2 ($p=0.09$). Graphical illustrations of annual silt levels at BR1 and SN1 suggest that levels fluctuate in a multi-year cycle.

One key characteristic distinguishing BR1 from SN1 was the greater percentage of stream area covered by submersed vascular plants (macrophytes). Additionally, the percent silt coverage at BR1 was correlated with percent aquatic macrophyte coverage, while these variables were not correlated at SN1 (Figure 34). Fluctuations in silt and aquatic macrophyte levels seemed to parallel each other at BR1 (Figure 35), but don't seem to correspond very well with the more constant linear trend toward decreased thalweg depth.

Sny Magill Upper Watershed and Tributary Sites. The upstream Sny Magill Creek site (SN3) and Sny Magill Tributary sites (NCC, SNT, SNWF) were grouped together for trend analysis. As discussed earlier, these sites have similar habitat characteristics and do not have corresponding sites in the Bloody Run watershed where annual habitat assessments were performed. Analysis of covariance (ANCOVA) was used to examine for temporal trends among the sites as a group instead of individually. In this statistical test, the habitat characteristic of interest was defined as the dependent variable, the independent (main effect) categorical variable was sample site, and years elapsed (time) was the covariable. The model tested for a significant linear relationship between the dependent habitat variable and time after accounting for the variability associated with between-site differences.

ANCOVA results indicated small, but statistically significant ($p<0.05$) increasing trends in average width, thalweg depth, and average depth during the monitoring period. These trends might indicate a connection between increased channel erosion and sediment export in the upper areas of the watershed and decreased water depth resulting from sediment deposition in the lower watershed (SN1).

Increasing trends in bank instability rating and tree canopy coverage were also observed at the upper watershed sites. No significant trends in substrate composition (e.g., % silt, % total fines, % coarse rock substrate) were observed, which is

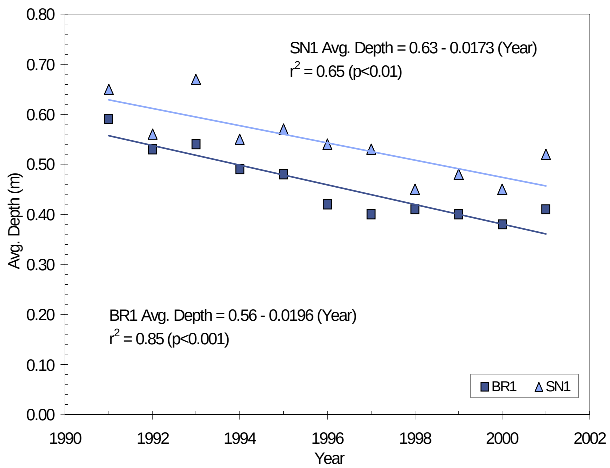


Figure 30. Time trends in BR1 and SN1 water depth.

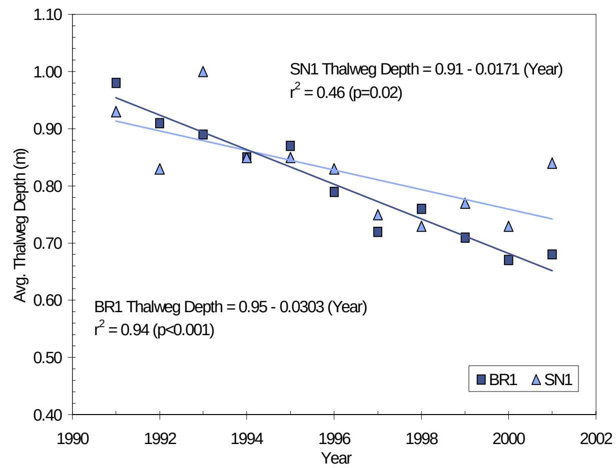


Figure 31. Time trends in BR1 and SN1 average thalweg depth.

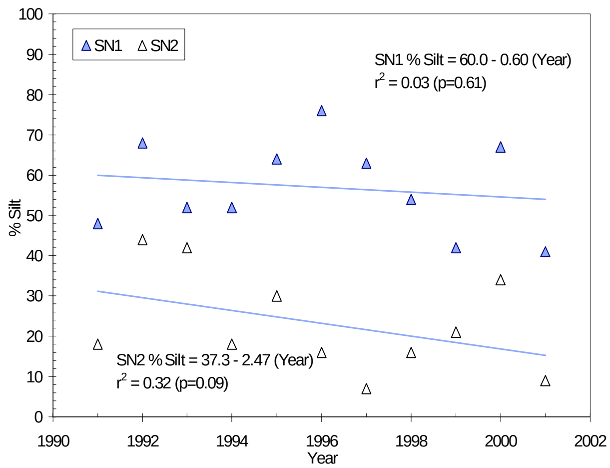


Figure 32. Percent of stream bottom covered by silt at SN1 and SN2.

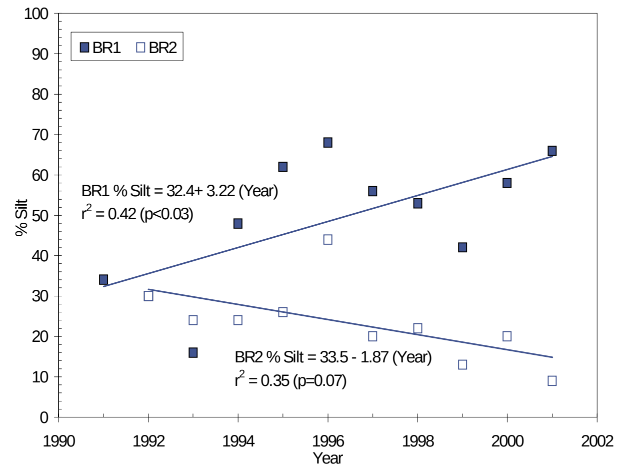


Figure 33. Percent of stream bottom covered by silt at BR1 and BR2.

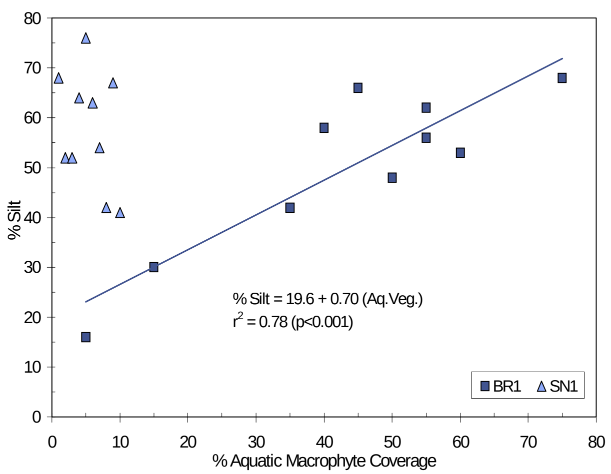


Figure 34. Relationship of % silt bottom and % aquatic macrophyte coverage at SN1 and BR1.

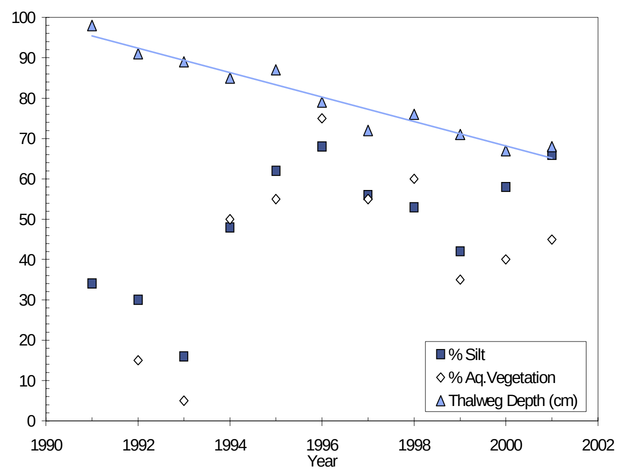


Figure 35. Relationship of % silt bottom, % aquatic macrophyte, and thalweg depth at BR1.

Table 5. Benthic biomonitoring results.

Metric	Stream	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
HBI	Sny Magill	2.21	1.99	1.9	2	2.05	2.2	2.07	2.12	2.49	2.49
	Bloody Run	2.24	2.17	2.17	2.22	2.22	2.1	2.31	2.57	2.57	2.45
Taxa Richness	Sny Magill	11.4	12.9	12.4	13.4	13	15.8	14.3	13.6	13.2	14.2
	Bloody Run	11.9	13.4	13.3	12.7	12.5	10.6	12.9	11.6	12.1	12.1
EPT	Sny Magill	6.4	7.3	7.5	7.8	8.2	7.7	7.7	7.8	6.9	7.2
	Bloody Run	6.9	6.8	7.6	6.8	7.2	6	6.4	5.3	5.6	4.9
% Dominant Taxa	Sny Magill	43.9	39.9	35.1	34.5	32.2	32.8	31.6	34.4	39.4	34
	Bloody Run	34.6	41.4	36.6	39.2	40.5	49.4	41.9	48.8	46.8	35.5

Modified from Birmingham (2002).

consistent with the lack of trends in sediment composition at SN1 and SN2.

Benthic Macroinvertebrates

Benthic macroinvertebrates (bottom-dwelling organisms) can be excellent indicator organisms because their life cycles are completed over an extended period of time in the stream and because they have varying degrees of tolerance for organic pollutants (Hilsenhoff, 1982). Personnel from the University Hygenic Laboratory (UHL) completed benthic macroinvertebrate monitoring for Sny Magill and Bloody Run creeks. Samples were collected from eight water quality sites at Sny Magill and Bloody Run creeks (Figure 2, p. 4) in April, June, August, and October from 1992-2001. Benthic macroinvertebrate monitoring was not conducted at site BRSC.

The benthic biomonitoring results suggest that water quality metrics at the study sites changed through time. Table 5 shows the average benthic biomonitoring results for Sny Magill and Bloody Run creeks. Taxa richness, a measure of the number of taxa in a sample, generally increases with higher water quality. During the 1992 through 2001 monitoring period, the average taxa richness in Sny Magill watershed generally increased from 11.4 in 1992 to 15.8 in 1997. Taxa richness declined from 1998 to 2000, and then increased to 14.2 in 2001. The percent dominant taxa gradually decreased from 43.9% in 1992 to 31.6% in 1998. The value then increased to 34.4% in 1999 and to 39.4% in 2000, and then declined to 34.0% in 2001. The taxa richness of the more intolerant species (EPT Index) increased from 6.4 in 1992 to 8.2 in 1996, suggesting water quality improvement in the Sny Magill watershed during this period. Since 1997, the EPT Index values fluctuated between 6.9 in 2000 and 7.8 in 1999. The Hilsenhoff Biotic Index (HBI) measures the overall pollution tolerance of the benthic community and increases as water quality decreases. The HBI did not show any water quality improvement trend in Sny Magill watershed through time. The years 2000 and 2001 had the highest HBI average value of 2.49, which resulted in several sites having their water quality ratings diminish in these years. For the Bloody Run watershed, results from the benthic biomonitoring study did not show obvious water quality changes, and all the values fluctuated through time.

Multiple regressions (Table 6) on means from the control sites and the treatment sites over time indicate trends towards improving water quality in the Sny Magill sites relative to the Bloody Run sites for taxa richness, EPT, and percent dominant taxa. The other metric, HBI, showed a significant increase toward declining water quality. Since sites SN3, SNT, SNWF, and NCC do not have corresponding counterparts in the Bloody Run water-

Table 6. Multiple regression on paired mean benthic macroinvertebrate indices.

Metric	Independent Variable	Coefficient Sign	r^2	Significance[2]
	control and time		.46	p<.00001
HBI	control	+		NS
	time	+		p=.0001
	time only	+	.45	p<.00001
Taxa Richness	control and time		.50	p<.00001
	control	+		p=.004
	time	+		p<.00001
	time only	+	.37	p<.00001
	control and time		.26	p=.004
EPT	control	+		p=.002
	time	+		p=.006
	time only	+	.05	NS
	control and time		.30	p=.001
% Dominant Taxa	control	+		p=.05
	time	-		p=.0007
	time only	-	.22	p=.002

[1]40 paired observations
[2]NS = not significant (p>.05)

Table 7. Simple regression on unpaired tributary mean benthic macroinvertebrate indices.

Metric	Independent Variable	Coefficient Sign	r^2	Significance[2]
HBI	time	+	.09	p=.0001
Taxa Richness	time	+	.06	p=.001
EPT	time	+	.0004	NS
% Dominant Taxa	time	-	.005	NS

[1]160 observations
[2]NS = not significant (p>.05)

shed, they were combined and simple regressions were conducted. These regressions of the combined Sny Magill tributary sites indicate trends towards improving water quality for taxa richness. However, the HBI showed a significant decline in water quality (Table 7).

To measure the taxon composition differences between samples, the detrended correspondence analysis (DCA) was performed on the taxon data sampled from Sny Magill and Bloody Run watersheds during 1992-2001. Results from DCA indicate that community composition did not vary substantially. However, differences in taxa composition between BR2 and other sites are consistently reflected from year to year, likely related to chironomid abundance. Additionally, DCA shows a steady unidirectional temporal trend for all sites, which may reflect the temporal trends in

Table 8. Index of biotic integrity in Sny Magill Creek.

	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
Sny Magill Creek #1										
IBI Score	30	15	5	-----	20	-----	-----	35	-----	-----
Integrity Rating	Poor	Poor	Very Poor	-----	Poor	-----	-----	Fair	-----	-----
Sny Magill Creek #2										
IBI Score	30	10	15	-----	35	15	-----	20	-----	20
Integrity Rating	Poor	Poor	Poor	-----	Fair	Poor	-----	Poor	-----	Poor
Sny Magill Creek #3										
IBI Score	25	30	20	-----	55	5	15	35	15	45
Integrity Rating	Poor	Poor	Poor	-----	Fair	Very Poor	Poor	Fair	Poor	Fair
Sny Magill Creek #4										
IBI Score	30	15	10	-----	35	20	15	15	10	-----
Integrity Rating	Poor	Poor	Poor	-----	Fair	Poor	Poor	Poor	Poor	-----
North Cedar Creek										
IBI Score	30	25	50	-----	-----	-----	40	-----	-----	-----
Integrity Rating	Poor	Poor	Fair	-----	-----	-----	Fair	-----	-----	-----
Bloody Run Creek										
IBI Score	60	35	60	70	70	60	40	85	60	60
Integrity Rating	Fair	Fair	Fair	Good	Good	Fair	Fair	Good	Fair	Fair

----- Fish numbers were less than 25, therefore, an IBI was not calculated.

Ceratopsyche slossonae and *Hydropsyche betteni* abundance common to both watersheds. For details in DCA, see Birmingham (2002).

Based on the results, no dramatic changes have been observed in the benthic communities of either watershed during the monitoring period. Though some metrics show statistically significant trends toward improving water quality, they are weak and results from regressions on the HBI metric indicate a significant trend toward declining water quality. Therefore, water quality changes in the study area based on benthic macroinvertebrate monitoring can not be directly linked to land treatment changes, or the results should be considered with caution. BMP implementation in the treatment watershed may have had no measurable effects, or the necessary attributes of the benthic community may not have been measured. The results could also be shadowed by other factors. For example, climatic effects and natural population fluctuations of individual species may have obscured any subtle changes that occurred in the Sny Magill watershed.

Fish Assessment

The IDNR-Fisheries Bureau inventoried the forage fish population at five sites in the Sny Magill watershed (four sites on the main channel of Sny Magill and one site on a tributary, North Cedar Creek) and one site on Bloody Run Creek (Figure 2). These six sites were sampled annually from 1992-2001. A baseline assessment conducted in

1991 included only the four sites on Sny Magill Creek (Wunder and Stahl, 1994).

All fish species collected in Sny Magill Creek, North Cedar Creek and Bloody Run Creek during the project were indicative of typical Iowa coldwater streams. Each creek was usually dominated by a single or few species of the stream forage fish population. At the Sny Magill sites, the fish populations were dominated by the fantail darter, white sucker, blacknose dace, and longnose dace. At North Cedar Creek, the fantail darter and blacknose dace were the most abundant species. Slimy sculpin dominated the fish population at Bloody Run Creek every year of monitoring, and accounted for more than 95% of the population in 1994 through 1997, 1999, and 2000.

Although limited in value, the results of the fish assessment and fish Index of Biotic Integrity (IBI) studies have shown that the environmental quality of Sny Magill Creek and North Cedar Creek has slowly improved during recent years. Research shows that high-quality coldwater streams generally have few fish species (Moyle and Herbold, 1987; Lyons, 1992; Lyons et al., 1996; Mundahl and Simon, 1999). The fish assessment data show that fish species richness decreased at almost all the sites on Sny Magill and North Cedar creeks during the monitoring period.

The IBI score slightly increased at several sites of Sny Magill and North Cedar creeks during the last few years of the project (Table 8). At Sny Magill Site 1, the IBI achieved the site's highest score in 1999. At Sny Magill Site 3, the IBI score was 45 in 2001, which was the second highest score obtained since 1992, and was significantly above the average of 27 for this site. For North Cedar Creek, only 4 years of data were available to do the IBI calculation. The last IBI score obtained in 1998, 40 points, was also higher than the average of 36 for this site. However, the IBI analysis did not show obvious achievement at Sny Magill Site 2, and even decreased at Sny Magill Site 4. That may result from several factors, such as local environmental variance, IBI analysis limitations for this study, or fish sampling bias.

In 2001 the slimy sculpin was first sampled in the Sny Magill Creek. This species was classified as intolerant of environmental degradation and as a stenothermal cold water species by Lyons and others (1996) and Mundahl and Simon (1999). A return of the slimy sculpin to Sny Magill may have indicated an improvement in stream quality.

LESSONS LEARNED

The following observations were made during the Sny Magill Project:

- An adequate period of time is needed to collect baseline data before best management practices are put in place. This time period should be, at minimum, 3-4 years in length to firmly establish pre-treatment relationships between the two watersheds.
- A forward thinking, flexible monitoring strategy is required at the outset of the project. Resources could be more efficiently used and conserved if the end goal and statistical methods of the project are kept in mind throughout the study period. If certain water quality parameters consistently measure below the detection limit, either change the analytical method or sample less frequently.
- Some BMPs installed can have unintended consequences on water quality. For example, tiled terraces might reduce sediment transport to the stream, but in the stream channel the high-energy water can re-suspend previously settled sediment particles. In addition, tiled terraces might increase stream discharge and NOx-N concentrations by piping water directly to the stream channel. This bypasses the evapotranspiration and denitrification processes that take place in a natural system.
- The lag time between initial BMP installation and measured changes in stream water quality might take many years, perhaps even decades. This is especially true of watersheds that are highly groundwater dependent, or that have significant pre-existing sediment deposits in the stream system.
- Turbidity and sediment, although sometimes similar, measure different parameters in the water column. A decrease in one does not

necessarily indicate a decrease in the other.

- Streams with relatively pristine conditions may add to the difficulty in accurately documenting water quality changes. Both sediment and nutrient levels in Sny Magill started out relatively low, limiting the amount at which levels could decrease. Many other streams in Iowa have two to three times the nutrient and sediment concentrations of Sny Magill Creek.
- When sediment concentrations varies drastically over a short period of time (<hourly), such as in a watershed the size of Sny Magill, daily sampling might not be adequate to quantify true concentration and load. It might be necessary to sample with higher frequency.
- Given the uncertainty of the contribution of sediment from stream beds, stream banks, crop lands, etc. it is important for future projects to address source tracking of sediments.
- A reduction in the size of the study area would have made it easier to target the implementation of BMPs. Targeting one or two impairments would have also focused the water monitoring design, better quantifying the changes to impairments in the stream.
- Without having a control to test against, it is difficult to prove a parameter did not shift due solely to a change in precipitation or ambient conditions during the study. A monitoring design that used more paired monitoring sites instead of single unpaired stations would have been better able to significantly prove increases or decreases in water quality.

CONCLUSIONS

The Sny Magill watershed underwent significant BMP implementation during the 1990s. These practices were put in place to reduce the amount of sediment, nutrients, and pesticides in Sny Magill Creek for improved fish and stream habitat. During the project, turbidity, and to a lesser extent suspended sediment, had significant reductions in the water column. Other important water quality variables such as nutrients, pesticides, fish, and benthic macroinvertebrate monitoring indicated either no change or a significant increase during the study. Part of the reason for these mixed results might be that the stream was already relatively healthy prior to project implementation. In fact, two important variables in the project (total phosphorus and triazines) had concentrations below the detection limit for the majority of the project.

Another possibility for the lack of conclusive results is a greater lag-time between BMP implementation and water quality improvements than was measured during this study. After BMP implementation, there is a certain amount of time needed for the practices to reach their maximum level of effectiveness. Deposits of sediment in the stream channel could also take a while to leave the watershed. Although USLE estimates a 50% reduction in sediment transport from the uplands, sediment already deposited in the stream channel can be resuspended many times before completely leaving the system. In addition, streams with significant groundwater inputs may face lag times related to groundwater transport rates for parameters such as nitrate and leachable herbicides. The length of these lag times is not clearly known, but appear to be longer that the period of monitoring.

The relatively large size of the Sny Magill watershed may have contributed to the lack of more immediate responses to BMP implementation. In a watershed the size of Sny Magill it may be unrealistic to assume that water quality will have a real-time change after BMP implementation. Effective results in larger watersheds will likely take more time to observe. More intensive land use changes may accelerate the response.

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This final report summarizes the water-quality data collected by various agencies during water years 1992 through 2001 from both Sny Magill and Bloody Run watersheds. Fish assessments were completed by Gaige Wunder, Jim Jansen, Vance Polton, Caleb Schnitzler, Ryan Doorenbos and Van Sterner of the IDNR-Fisheries Bureau; the habitat assessment was directed by Tom Wilton of the IDNR-Water Quality Bureau, with assistance from Mike Birmingham, Matt Coleman, Dennis Heimdal, Todd Hubbard, Jim Luzier, and Mike Schueller of the UHL Limnology Section, Connie Dou, John Olson, and Ralph Turkle of the IDNR-Water Quality Bureau, and Maggie Clover of the IDNR-Compliance and Enforcement Bureau; stream discharge and suspended sediment data was collected and compiled by Greg Nalley of the U.S. Geological Survey (USGS) in Iowa City, and by local observers Robert Davis and Travis Kruse; benthic work was conducted by Mike Birmingham and Mike Schueller; and water-quality sampling for physical and chemical parameters was completed by Rodney Rovang, Jennette Muller, and Chris Harmon of Effigy Mounds National Monument, and Lynette Seigley, Richard Langel, Bob Rowden, Deb Quade, Bob Libra, Carol Thompson, Mary Skopec, and Huaibao Liu of the IDNR-Geological Survey.

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PROJECT EXPENDITURES/COSTS

Funding restrictions in the Sny Magill HUA for FY94 affected cost-share funding to assist cooperating producers in installing BMPs. The HUA was able to operate in FY94 on limited funding that remained from previous years. The project applied for alternate funding to meet the unmet needs of producers to install BMPs. Funding for BMP implementation for 1995 through 1998 was provided by the Iowa Department of Agriculture and Land Stewardship – Division of Soil Conservation and the Iowa Department of Natural Resources.

Federal funding from the Agricultural Conservation Program to encourage BMP implementation was lost in 1993; however, applications for alternative funding sources were filed in 1994. Funding for sediment reducing practices, such as terraces, was secured through the Iowa Department of Agriculture and Land Stewardship, Division of Soil Conservation, for Fiscal Years 1995-1998. An application for funding was filed through the USEPA Section 319(h) Program for animal manure structures, Integrated Crop Management (ICM), and streambank stabilization practices. The USEPA Section 319(h) funding became available in 1995, and continued through 1998. Extended funding for the Sny Magill Hydrologic Unit Area was requested and received through 1999.

Project Element	**Funding Source ($)**			
	Federal	**State**	**Local**	**Sum**
I&E	445,000	233,550	NA	678,550
LT (cost share)	374,000	333,634	NA	707,634
LT (technical assist.)	874,000	NA	NA	874,000
WQ Monit	1,133,910	NA	NA	1,133,910
TOTALS	2,826,910	567,184	NA	3,394,094