3.1 Water Quality

(*This section was adapted from the Schoharie SMP and is meant to provide a comparison of Manor Kill water quality to other major streams in the upper Schoharie basin*)

Introduction

The purpose of this section is to provide a general understanding of water quality in the upper Schoharie basin. To further this understanding the authors have included a synopsis of the research that has been conducted in the basin, a general discussion of the various parameters that are routinely monitored and conclusions that can be extrapolated from the various research projects. The following text is meant as an educational primer of water quality designed to broaden the general understanding of watershed residents. It is not meant to be used in a legal or regulatory context.

Determining whether a stream has good or bad water quality depends largely upon the end user. For example, defining what constitutes good water quality for the supply of drinking water may be different from defining good water quality for maintaining a cold water fishery. The water quality parameters researchers would analyze would differ based upon the different end-users (people versus trout). Overall, the Schoharie Watershed plays an important role in the delivery of high quality water to the approximately 9 million end-users in New York City and the surrounding region. The high quality of this drinking water is demonstrated by its lack of need of filtration before consumption. From the trout's perspective, research indicates that the Schoharie Creek from the Village of Hunter upstream to the Dale Lane area (where the 2006 assessment started) the creek supports a healthy aquatic community (Novak et al., 1989; Bode et al., 1995; Arscott et al., 2004). The creek below the Village of Hunter has shown some impacts to biota in the past and requires continued monitoring (Bode et al., 2004).

This good water quality supporting multiple uses can most likely be attributed to the watershed's high percentage of forest cover (Figure 3.1.1). There have been many studies that demonstrate the effects of land use/land cover on water quality. For example, there has been a vast array of research demonstrating that as land use becomes more urbanized, biotic communities decline in health (Schueler and Holland, 2000; Limburg and Schmidt, 2000; May et al., 2000; Wang et al., 2001; Potter et al. 2005 and Kratzer et al. 2006). Concentrations of selected chemical constituents, including nitrate, in stream base-flow were strongly affected by the predominant land use in a large Hudson Valley study (Heisig, 2000). The decline of watershed forest cover below 65% percent marked a transition to degraded water quality (Booth, 2000). Based upon these results, it is safe to theorize land use/land cover is a major factor of water quality. Maintaining the land use/land cover conditions that allow for good quality water should be a priority.

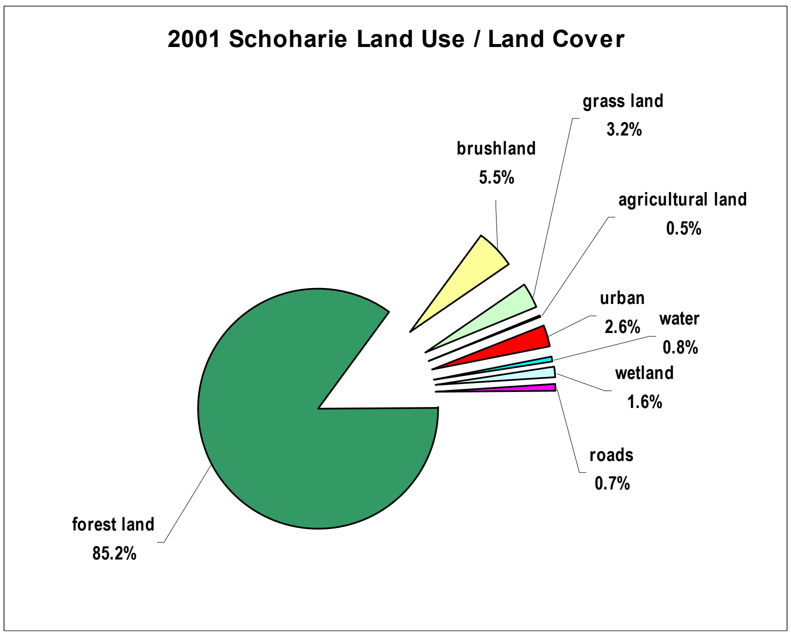


Figure 3.1.1. Land use in the Schoharie basin based upon 2001 satellite imagery (NYCDEP, 2001).

Water quality threats in the Schoharie basin are more abstract than the classic piped outfall containing pollutants. For example, silt and clay – buried in ice age deposits – are easily eroded into the stream and often, after a major storm, the streams run with a characteristic reddish brown color, which elevates in-stream

Turbid water draining from a steep sloped, developed landscape following a summer downpour. Note the clean water entering from the other tributary.

turbidity. Although temporary, this increase in turbidity can act to degrade fish habitat (Newcombe and Jensen, 1996; Henley et al., 2000; Newcombe, 2003), act as a transport mechanism for other pollutants and *pathogens* (LeChevallier et al., 1981) and cause changes in the operations of the NYC water supply (NYCDEP, 2004). Although a certain percentage of this erosion is natural, disturbances to the steep slopes in the basin and/or other human interventions add to the problem and constitute the percentage of the problem that may be identified and addressed more easily. The multitude of interventions designed to protect infrastructure (bridges, roads and buildings) in the Schoharie watershed can exacerbate the rates of erosion, thus releasing turbidity causing materials into the stream (Fischenich, 2003). In addition, this infrastructure protection is often constructed of rock, or the infrastructure itself of blacktop, and the presence of these surfaces typically means the native vegetation had been removed, possibly adding stress to the stream biota (Sweeney, 1993; Jones III et al., 1999). This stream management plan offers recommendations for minimizing these efforts in a collaborative effort. The following text will describe many of the water quality parameters of interest and offer a chemical snapshot for the Schoharie Creek and its major tributaries.

NYSDEC Stream Classification and Impaired Water Body List

All waters in New York State are given a class and standard designation based on best usage for that water body (NYSDEC, 2004). The New York State DEC stream classification system includes the following designations:

Stream Classifications

Class	Best Use
AA	Drinking (after disinfection), Bathing and Fishing
A	Drinking (after disinfection and approved treatment), Bathing and Fishing
B	Bathing and Fishing
C	Fishing – Propagation and Survival
D	Fishing - Survival

New York Codes, Rules, and Regulations (“NYCRR”), Title 6, Section 701.

Additional designations of “T” or “TS” can be added if a water body has sufficient amounts of dissolved oxygen to support trout (T) and/or trout spawning (TS). Water bodies that are designated as “C (T)” or higher (e.g., “C (TS)”, “B”, or “A”) are collectively referred to as "protected streams," and are subject to additional regulations and require a State permit for disturbance of the bed or banks. Periodically, the DEC publishes the Priority Water bodies List (PWL), which includes a list of water bodies that do not meet their designated "best use" classification. A data sheet that describes the conditions, causes, and sources of water quality degradation for each of the respective listings is also included in the PWL. The PWL is used by the DEC and other agencies as a primary resource for water resources management and funding. In 1998, the Schoharie Reservoir was listed on the PWL for silt and sediment from construction activities and for atmospheric deposition of mercury. Mercury bioaccumulates in the fatty tissue of fish, particularly predatory species, and is passed on to the consumer. In the Schoharie, smallmouth bass over 15" and walleye over 18" should not be eaten; and smallmouth bass under 15" and walleye under 18" should be eaten only once per month (NYSDOH, 2006).

In 2008, the Manor Kill was classified C(ts) from its headwaters to near its confluence with the Schoharie Reservoir. A small portion (~ half of MU 10) of the stream close to the reservoir was classified as A(ts). The Bear Kill was also classified C(ts). Many of the smaller tributaries to the Manor Kill were classified "C".

Water Quality Record

In the United States (USEPA, 2005) and New York State (NYSDEC, 2004) nonpoint sources of pollution are the cause of the majority of water quality impairments. In New York State, nonpoint sources of pollution accounted for 90% of impacts on the water quality of rivers and streams and 92% for lakes and reservoirs, including the Schoharie (NYSDEC, 2004). There are many ways to measure water quality, from direct laboratory analysis of water samples for various analytes to indirect measures such as aquatic insect surveys as indicators of water quality. Water samples collected from the stream and analyzed for a suite of chemical, biological and physical parameters provide us with a good picture of the constituents that are carried by the Schoharie's waters. Between the NYCDEP, USGS, NYSDEC and other researchers the large quantity of these water quality data necessary to draw conclusions is available.   Biological indicators, such as fish and macroinvertebrates, are also monitored to determine surface water quality and nonpoint source pollution impacts (Barbour et al., 1999; Murray et al., 2002). For example, biological assessment models have been tested with field data and the results suggested that macroinvertebrate data collected for establishing the degree of water quality impairment can also be used to identify the impairment source with reasonable accuracy (Murray et al., 2002). There is a relatively extensive set of data for both direct and indirect measures on the streams in the Schoharie basin, including the Manor Kill.

Direct Water Quality Measurements

There are several sources for direct water quality measurements for Schoharie basin streams. The following sources provide the bulk of available information:

- The most extensive and comprehensive set of available data is from NYCDEP as part of its long-term water quality monitoring of the NYC drinking water supply (NYCDEP, 2006). NYCDEP has been sampling and analyzing the Schoharie since the early 1900's.
- The United States Geological Survey (USGS) collected water quality data near the Prattsville gage (# 01350000) from 1966 to 1992. The water quality data is available on the USGS website: http://nwis.waterdata.usgs.gov/ny/nwis/qwdata/?site_no=01350000&amp – there doesn't appear to be a water quality component to the Manor Kill gage.
- The USGS, under contract to NYC DEP, has collected water quality at 2 locations in the Schoharie Creek Watershed: Batavia Kill near Maplecrest (1997 – present), Batavia Kill at Red Falls (1999 – present): http://ny.cf.er.usgs.gov/nyc/unoono.cfm. The two sites are designed upstream and downstream to document changes in water quality from land use changes in between the two stations. USGS also completed an in-depth study on the Batavia Kill (Heisig, 1998): http://ny.water.usgs.gov/pubs/wri/wri984036/.
- In 2000, Stroud Water Research Center located in Pennsylvania was awarded a Safe Drinking Water Act (SDWA) grant funded by the New York State Department of Environmental Conservation and the USEPA to conduct a six-year study to monitor and evaluate water quality and sources of pollution in the streams, rivers, and reservoirs that provide New York City's (NYC) drinking water. There were ten sites in the Schoharie Creek watershed that have been

variably sampled since 2000. Copies of the reports for the first five years can be found at: (http://www.stroudcenter.org/research/newyorkproject.htm).

- Upstate Freshwater Institute: UFI is currently under contract to NYCDEP to develop "Integrated Programs of Monitoring, Process Studies, and Modeling in Support of Rehabilitation Initiatives for Turbidity Problems in Schoharie Reservoir and Esopus Creek". As a consequence, a vast amount of very detailed data (e.g. water temperature, conductivity, beam attenuation coefficient, turbidity) has been collected for the Schoharie Creek and Reservoir. The data have been presented at numerous meetings with regulators, and are being published in peer-reviewed international literature.
- NYSDEC, Routine Statewide Monitoring Program provides for the routine monitoring of the waters of the State to allow for the determination of the overall quality of waters, trends in water quality, and identification of water quality problems and issues. This monitoring effort is coordinated through the Rotating Integrated Basin Studies (RIBS) Program which typically operates on a 5-year cycle. Contacts for the program staff, which can provide relevant reports, are available at their website: http://www.dec.ny.gov/chemical/30951.html.
- "Conine" Water Quality Monitoring Project - The objective of this monitoring project is to quantify the effectiveness of natural channel design at reducing turbidity and suspended sediment in the Batavia Kill. Observations and sampling have documented that the Batavia Kill delivers a significant quantity of suspended sediment and turbid water to Schoharie Creek, the main inflow to Schoharie Reservoir. Major sediment source areas are known immediately above and below Red Falls. Through a contract with the DEP's Stream Management Program, the Greene County Soil and Water Conservation District is designing and implementing a natural channel design restoration project to reduce the sediment and turbidity originating in the Red Falls area, specifically the DEP-owned property, located just downstream of Red Falls. DEP has been monitoring water quality at several sites on the Batavia Kill prior to BMP implementation (currently scheduled for 2007), and will continue to do so for several years after implementation. The monitoring project is based on collecting samples during

storm events both upstream and downstream of the project area before and after implementation of the project. The goal is to sample about ten events each year with about 15 samples collected at each site over the course of the event. By quantifying the turbidity and suspended sediment loads in the Batavia Kill before and after restoration, DEP will be able to evaluate the effectiveness of the approach used in mitigating turbidity, which can then help guide restoration design for other problem sites in the watershed.

- In 2008, researchers from SUNY Cobleskill conducted macroinvertebrate and fish surveys along the Manor Kill. See the macroinvertebrate and fish reports (Appendix F) for more detailed information regarding the surveys and their findings.

NYCDEP has a long-term water quality sampling program of streams in the NYC water supply watersheds. Water quality samples are collected at a fixed frequency from a network of sampling sites throughout the watershed. Grab samples are generally collected once a month (twice a month at selected sites). Storm event sampling is also performed at selected sites. While the analyses performed on samples from a specific site vary somewhat based on the objectives for the site, in general, samples are tested for temperature, pH, alkalinity, specific conductivity, dissolved oxygen, turbidity, nutrients, dissolved organic carbon, total organic carbon, silica, chloride, suspended solids (selected sites), major cations (Ca, Mg, Na, K, Fe, Mn, Al, Cu) (analyzed monthly), trace metals (Ag, As, Ba, Cd. Also included here are Cr, Hg, Pb, Se, Zn) (collected at selected sites quarterly), and total and fecal coliform (most sites). The current monitoring system was re-designed in 2002 and was based on multiple objectives (NYCDEP, 2002), with several sampling sites located in the Schoharie Basin (Figure 3.1.2). Results are presented in annual water quality monitoring reports (e.g. NYCDEP, 2006).

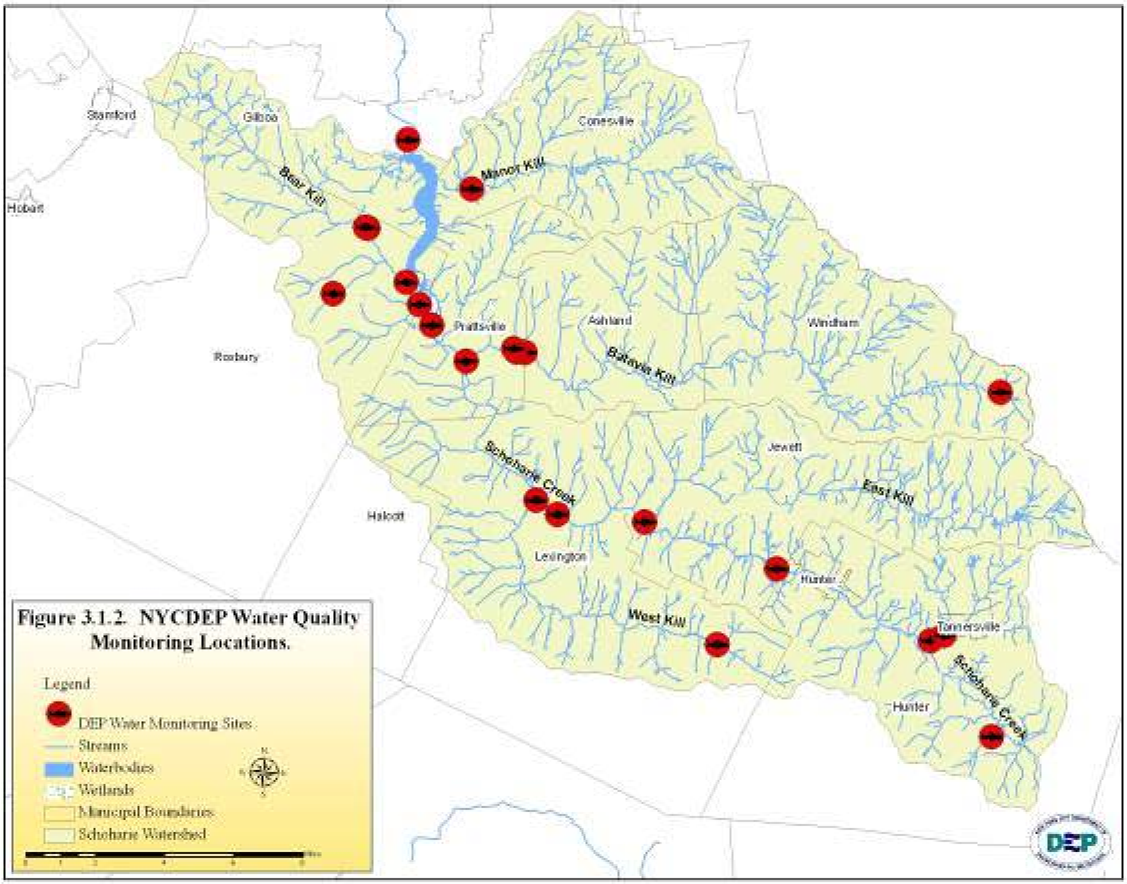


Figure 3.1.2. NYCDEP Water Quality Monitoring Locations.

Constituents of Schoharie Creek Water

The following section provides a summary of the major parameters that are tracked by NYCDEP in the Schoharie basin. Combined, these parameters provide a basic overview of water quality, while potentially allowing for a general understanding of human-induced changes to water quality. The NYCDEP data reported here are annual medians for selected water quality variables. The median is a statistic that expresses the "typical" condition of something. The median is simply the value in the center of a data set, i.e. half of the samples are higher, and half lower. One characteristic of the median is that it is not overly influenced by data from extreme events. Also, the results are based on routine grab samples, and do not specifically target extreme events.

Turbidity and Total Suspended Solids

Turbidity, an index of water clarity, is a concern in this watershed for two regulatory reasons: Safe Drinking Water Act oversight of NYC water supply and a State Pollution

Discharge Elimination System (SPDES) Permit for the Shandaken Tunnel. The Safe Drinking Water Act and associated regulations are concerned with turbidity levels entering the distribution systems for public water systems; accordingly, from a Safe Drinking Water Act perspective, DEP's primary concern is the level of turbidity in water leaving the Kensico reservoir (Westchester County). For purposes of drinking water, turbidity is of concern because it has the potential to mask pathogens and interfere with disinfection. In contrast, the focus of the Shandaken SPDES permit is on turbidity at the Esopus Creek outfall of the Shandaken Tunnel, which diverts water from the Schoharie reservoir to the Esopus Creek. Turbidity is a concern for the ecologic, recreational and aesthetic use of the Esopus Creek (CCEUC, 2007).

Since 1977, the Shandaken Tunnel has been operated under the guidelines of Part 670 of the NYS DEC Rules and Regulations (http://www.dec.ny.gov/regs/2485.html). As of September 2006, the Shandaken Tunnel turbidity discharges have been regulated under a NYSDEC issued SPDES permit. The SPDES permit sets limits on the turbidity of the water that can be discharged from the tunnel (Appendix C). Following extensive analysis, NYCDEP decided to focus their efforts on meeting the permit requirements through modified reservoir operations (e.g. reducing or eliminating Schoharie diversions during times when the water is not needed because Ashokan is likely to refill on its own) (Joint Venture, 2006).

Turbidity is an optical measurement of the light-scattering at 90^{o} caused by particles suspended in water (Figure 3.1.3). Turbidity is measured in arbitrary "nephelometric turbidity units" (NTUs) by a "nephelometer". The higher the NTU value, the lower the water clarity. Turbidity can be influenced not only by the amount of particles in suspension, but also by the shape and size of the particles. There is no single, fixed relationship between turbidity and total suspended solids. Total suspended solids are a measure of suspended solids concentration, expressed as a mass per

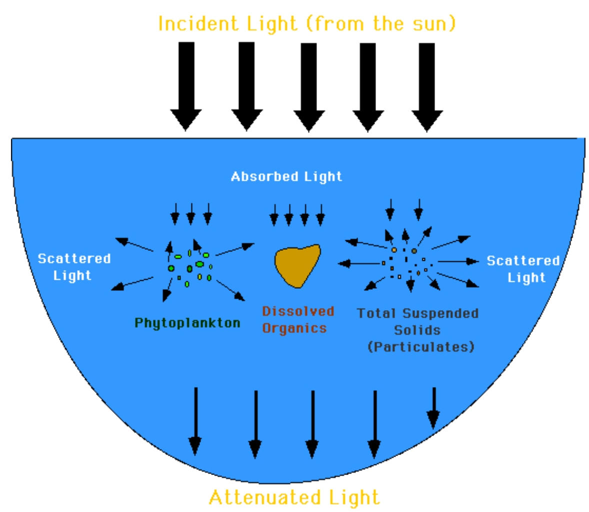


Figure 3.1.3 Illustration of light scattering caused by suspended particles in water.

volume (mg/L) obtained by physically separating the liquid and solid phases by filtration. Further, it is important to note that there is no universal, usable, fixed turbidity/clarity relationship.

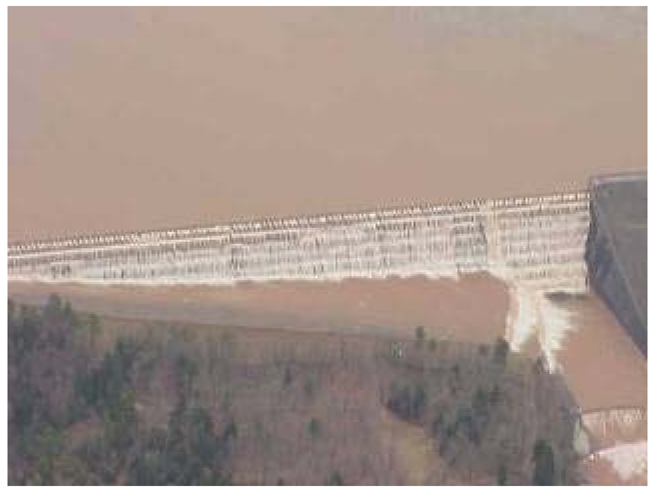
Turbid discharge from a road ditch as it enters the clear Schoharie Creek

Suspended solids in Catskill streams are predominantly fine sediment. It does not take much of the fine suspended sediment to reduce water clarity. Water clarity can range from clear to an opalescent red-brown following a significant high water event. Sediment gets in the stream primarily from two sources: (1) runoff from the landscape carries fine sediment (silt and clay) into the stream through ditches and culverts; and (2) from entrainment in the stream. Due to the large amount of forested landscape in the Schoharie system it is safe to speculate that the main sources of sediment are erosion within the stream channel and banks, and not the landscape. Exposed "clays" that the stream has cut into, and the mobilization of fine sediments mixed in the stream bed deposits, are the major sources of turbidity at times when turbidity reaches levels of concern for drinking water purposes (NYCDEP, 2006). However, landscape sources should not be ignored because they may assist in the development of a watershed stewardship ethic, and left untreated may cause further instability within the bed and banks.

Turbid water in the East Kill following a large chunk of bank with a high clay content falling into the stream. Upstream of this bank the stream was clear

The regulatory water quality standard for turbidity in New York State is a narrative standard: "no increase that will cause a substantial visible contrast to natural conditions" (NYCRR, Title 6, Section 703.2). There is also a narrative water quality standard for suspended, colloidal, and settleable solids: "None from sewage, industrial wastes or other wastes that will cause deposition or impair the waters for their best usages." Although there are no numerical standards for turbidity or suspended sediment, these constituents are of concern in streams because the presence of fine-grain sediments such as clay particles suspended in the water column can affect stream biota. These fine sediments can settle on substrates used by colonizing algae and invertebrates and can fill the small spaces between gravel where fish lay their eggs. Transmission of light through the water can be reduced, which can affect stream productivity through decreased photosynthesis. Turbid waters also become warmer as suspended particles absorb heat from sunlight, which can also cause oxygen levels to fall.

Turbidity in Catskills is not a new phenomenon. The design of the Catskill Water Supply System (in service from, Ashokan (1915) and Schoharie (1926)) included components, such as the ability to stop water transfers during flood events that reflect concern for turbidity on the part of the design engineers. Water in the Schoharie Reservoir can remain turbid for extended periods after flood events due to characteristics of the reservoir and its watershed (Joint Venture, 2004 and Joint Venture, 2006). It remains to be seen what the effects of global climate change will be on the frequency of large storms, and the related spikes in turbidity. The function of the Catskill water supply system and turbidity is discussed in more detail in the Upper Esopus Creek Management Plan (CCEUC, 2007).



Schoharie reservoir spillway with turbid water following a storm.

The characteristics that lead to these extended periods of high turbidity include the exposure of the "clays", which are actually ice age deposits from when the landscape was covered by glaciers, and afterwards by their melt water lakes. The glaciers left glacial till, a dense mixed "hardpan" of clay and rocks. The legacy of the glacial lakes in the Schoharie watershed is the thick blanket of layered silt and clay that settled out while the glacial lakes were in place.

Glacial till with clay content exposed in streambank on Schoharie Creek

Many of these deposits are locked in place by vegetation and a hardened rock stream-bottom. However, when erosion into the banks or downcutting into the bottom occurs some of these glacial lake deposits are remobilized (Figure 3.1.4). Some of the silt and clay entrained from the glacial sources settle out along the stream course and get incorporated into the stream bed material. This material is often resuspended following storms.

Figure 3.1.4. Example of the clay exposures (yellow) mapped during the Manor Kill 2008 stream feature inventory. This is Management Unit eight of ten and is located approximately 0.6 miles upstream of the Schoharie Reservoir.

On January 18-19, 1996 heavy rains fell on a substantial snow pack, which, along with unseasonably mild temperatures, resulted in widespread flooding in the Schoharie basin. Compared to preflood levels, turbidity levels remained elevated dramatically affecting water quality (Figure 3.1.5). The storm, and associated mitigation measures (channelization, berming, etc.), apparently damaged the Schoharie watershed resulting in an enhanced ability to entrain turbidity-causing material. This temporarily enhanced ability to mobilize turbidity-causing material under all flow conditions resulted in sustained elevated turbidity levels in the Schoharie Reservoir, and the Shandaken Tunnel. It appeared that beginning in 2001 and continuing into 2002, turbidity levels in the Schoharie watershed had returned to pre-1996 levels (Figure 3.1.5).

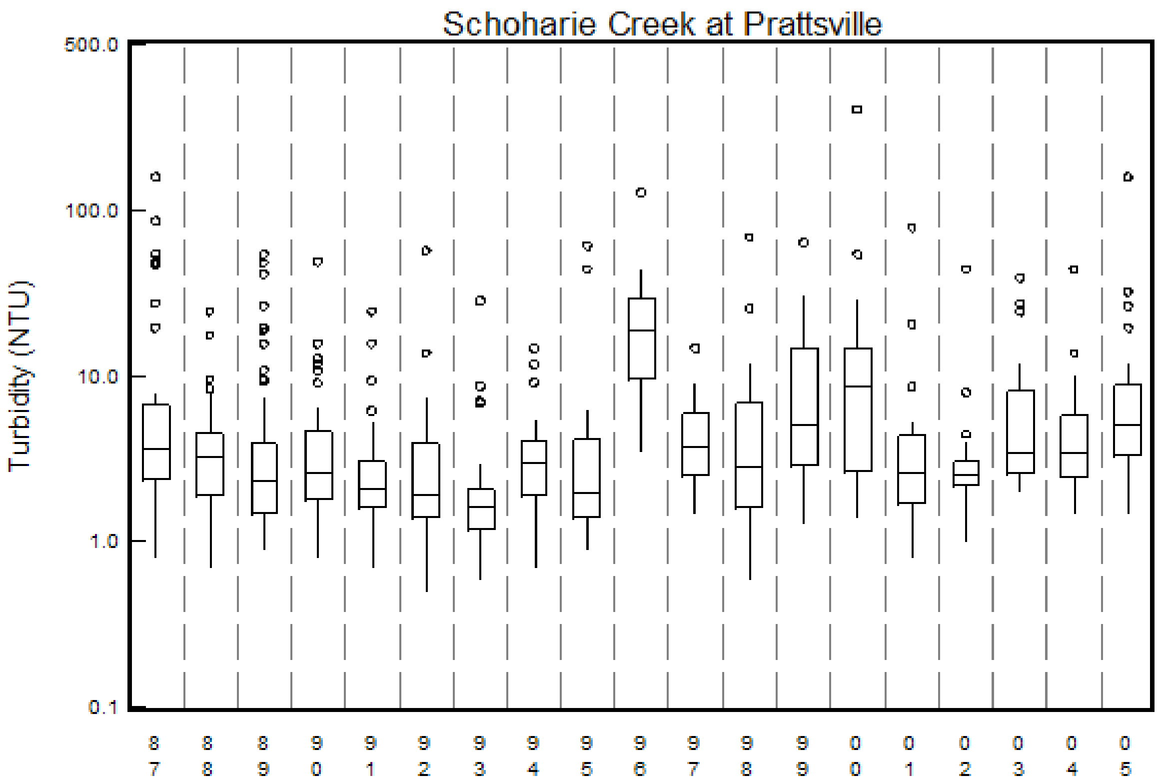


Figure 3.1.5. Box plots of turbidity values by year (1989-2002) for Schoharie Creek at Prattsville. The rectangular part of the plot extends from the lower to the upper quartile, covering the center half of each sample. The centerline in each box shows the location of the sample medians, and the horizontal lines (whiskers), extend from the box to the interquartile range values in the sample. Outliers that lie more than 1.5 times outside the interquartile range above or below the box are shown as small circles (Source – NYCDEP).

Tributaries to the Schoharie Creek also contribute significant quantities of turbidity/TSS. They provide variable sediment loads depending upon their geology/geomorphology, recent flood history and storm conditions. For example, the median annual turbidity for the tributaries and main stem sites combined was 2.5 for the period of record, but 11.3 for 1996. This demonstrates the system-wide effects on turbidity that storms the magnitude of 1996s can have. Through the period of record, Batavia Kill and West Kill have had the largest contribution of turbidity/TSS to the Schoharie Creek (Figure 3.1.6). Each of these tributaries has a Stream Management Plan detailing their conditions and offering recommendations for remediation to the extent it is possible (GCSWCD, 2003; GCSWCD, 2005).

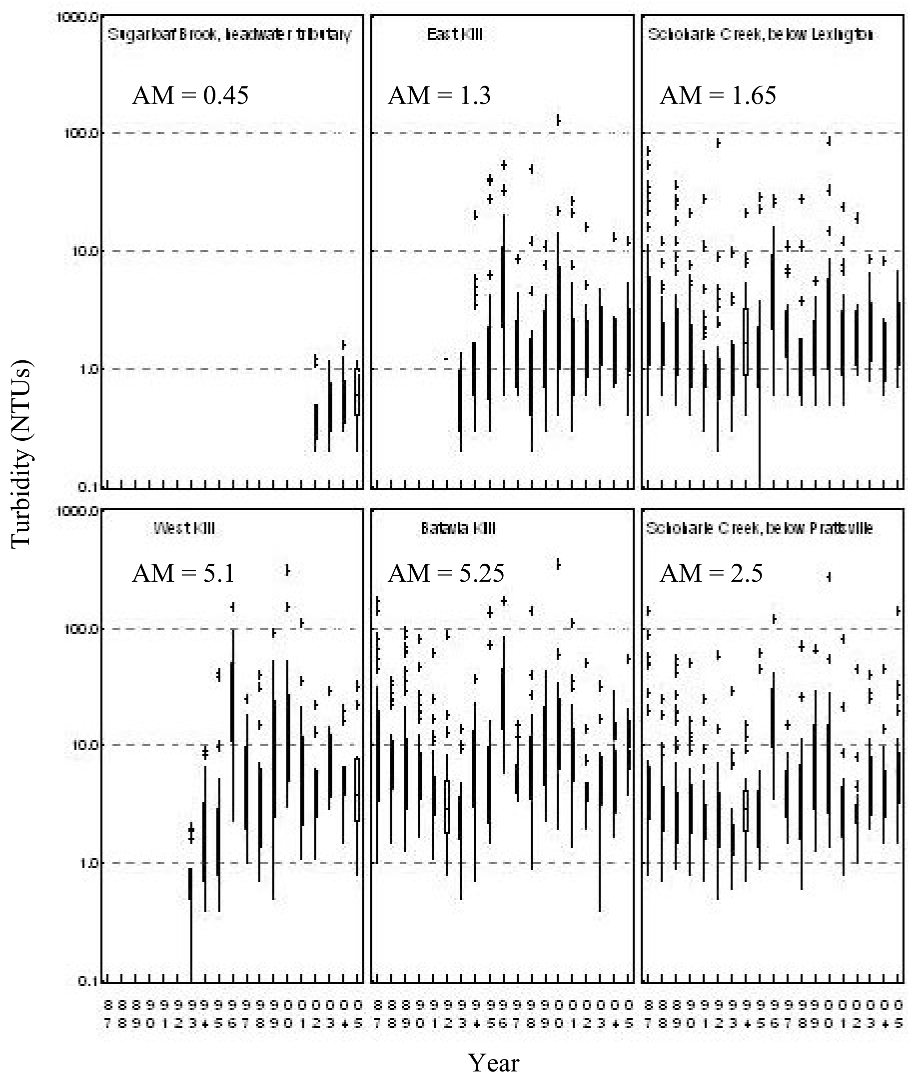
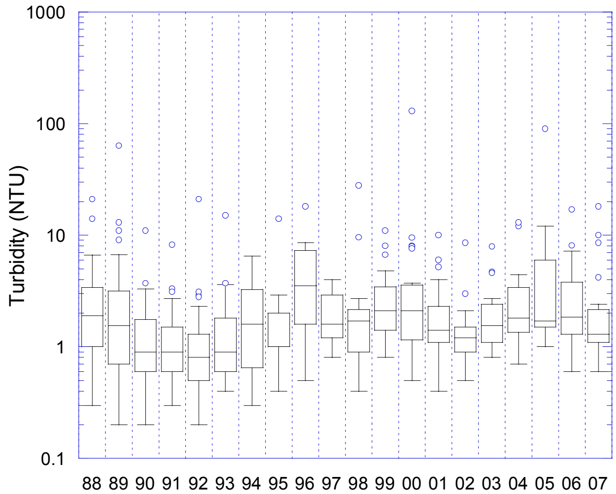


Figure 3.1.6. Turbidity of the Schoharie Creek and its major tributaries from 1987 through 2005. Note that Batavia Kill and West Kill have the most points above 100 NTUs and their annual median (AM) turbidity levels are higher than the others (Source – NYCDEP).

The Manor Kill had an annual median turbidity of 1.4 NTUs through the period 1998 through 2007 (Figure 3.1.7). It also showed the effects of the 1996 flooding with an annual median turbidity of 3.5 NTUs, which was much higher than other years. In the case of



Catskill stream turbidity, both hydrology (storm events) and geology are important determining factors. The hydrology and geology are natural factors that cannot be effectively managed. Therefore, management efforts should be focused on preventing further human-induced water quality degradation through implementation of best management practices designed to reduce/minimize sediment impacts. These efforts should be both direct (e.g. planting a riparian buffer) and indirect (e.g. reducing stormwater inputs and/or properly installing new infrastructure so it doesn't destabilize the stream).

Pathogens

NYCDEP monitors for pathogens, specifically giardia and cryptosporidium, in a large number of Catskill mountain streams (Figure 3.1.8). Specifically, NYCDEP's Pathogen Program monitors fourteen sampling location sites within the Schoharie Creek Watershed (Figure 3.1.8), twelve stream locations and two waste water treatment plants (WWTP) for, among other water quality parameters, protozoa; *Cryptosporidium spp.* oocysts and *Giardia spp*. cysts. While there are no regulatory thresholds for these protozoa in surface waters, NYCDEP maintains a monitoring program for them due to their potential negative effects on public health. These protozoa are of concern to public health for two reasons: 1) if consumed, certain strains of these protozoa can cause disease in humans, and 2) the presence of these protozoa indicates that the water has been contaminated with fecal matter (animal or human) and; therefore, may be carrying other pathogens that have the potential to cause disease in humans.

DEP's monitoring data has shown the presence of these (oo)cysts in ambient water, and during high flow conditions related to runoff events; however concentrations have been at low levels. In any event, since certain strains have the potential to cause disease in humans, determining their source, transport properties, and fate are of utmost importance to DEP. DEP maintains a surveillance program designed to narrow down source locations and trends of (oo)cysts throughout New York City's water supply watersheds. Additional tools used by DEP to ultimately assess the public health risk associated with these protozoa in the watershed include: 1) PCR (polymerase chain reaction) source tracking to identify anthropogenic (human) and autochthonous (natural) sources, 2) landuse/landcover which also indirectly identifies potential human sources such as failing septic systems and wildlife sources, 3) and watershed physiographic characteristics such as percent area of contribution to a site, slope and elevation which may affect transport and fate.

From 2003 to 2006 average concentrations of Cryptosporidium in Schoharie watershed streams were very low all of which were <1 oocyst 50 L-1 except for one site which averaged 1.1 oocysts 50 L-1 (Figure 3.1.9). Giardia was found in higher concentrations than Cryptosporidium throughout the watershed averaging from <1 cyst 50 L-1 to 140.7 cysts 50 L-1 (Figure 3.1.9).

A breakdown of the 2003 to 2006 data is as follows, the average concentration of Cryptosporidium from downstream to upstream locations along the Schoharie Creek was 0.64 oocyst 50 L-1 near Prattsville, 0.54 oocyst 50 L-1 at Lexington, 0.16 oocyst 50 L-1 near the Village of Hunter and 0 oocyst 50 L-1 at the headwaters near Elka park. Giardia concentrations were higher at the same four sites with 33.87, 53.58, 49.0, and 0.77 cysts 50 L-1 respectively. Tributary confluences to the Schoharie also monitored include the Manor kill, Bear kill, Toad Hollow, Batavia kill, West kill, and East kill. Their average Cryptosporidium concentrations are as follows; 0.7, 0.81, 1.19, 0.44, 0.0, and 0.11 oocyst 50 L-1 respectively. Giardia concentrations were higher at these stream sites, and are as follows; 140.7, 27.72, 4.9, 23.11, 80.4, and 20.44 cysts 50 L-1 respectively. Headwater locations for the Batavia kill and West kill were also monitored and were among the locations with the lowest average concentrations of Cryptosporidium found with 0.33 and 0 oocyst 50 L-1 respectively, and lowest average Giardia concentrations with 3.66 and 0.75 cysts 50 L-1 respectively.

The two WWTPs monitored within the watershed are Hunter Highlands which processes waterwater for the Village of Hunter and the other is Tannersville for the Village of Tannersville. All but two samples collected at these sites did not contain Cryptosporidium or Giardia. On those occasions both samples were collected at Hunter Highlands, of which one sample contained 3 oocysts 50 L-1 and the other contained 7 cysts 50 L-1.

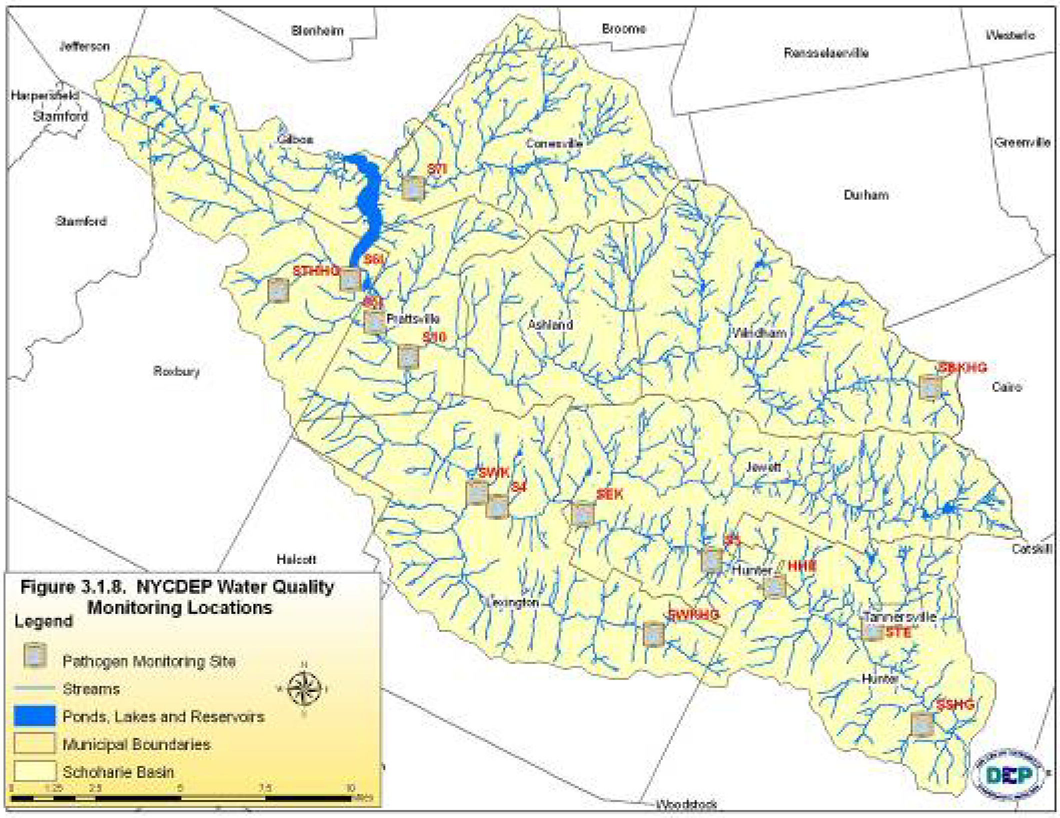
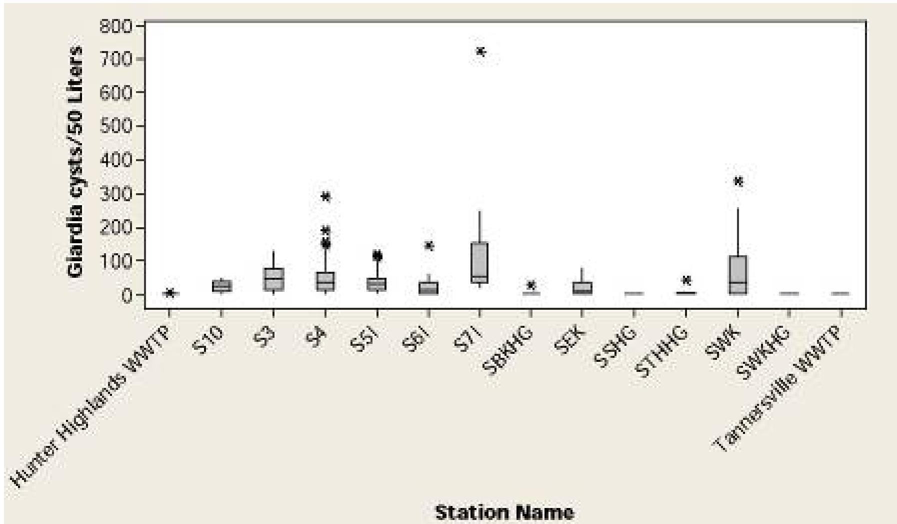


Figure 3.1.8. NYCDEP Water Quality Monitoring Locations

DEP scientists analyzed storm water from streams East-of-Hudson in an attempt to identify the sources of *Cryptosporidium* oocysts. Samples were analyzed using a small-subunit rRNA based diagnostic tool utilizing polymerase chain reaction technology to identify the genetic patterns of the oocysts. Results indicated that all of the oocysts genotyped in 2003 originated primarily from non-human sources, and deer, muskrat, and skunks topped the list of sources (NYCDEP, 2004). This does not mean that these results automatically translate to streams west-of-Hudson, but offers a glimpse into potential sources.



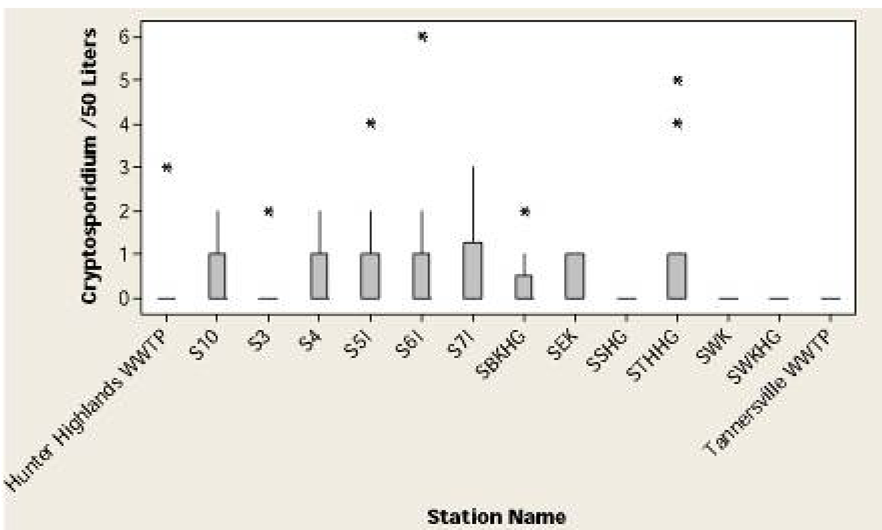


Figure 3.1.9. Boxplots of Giardia and Cryptosporidium at Schoharie Watershed Site Locations. See Figure 3.1.8 for site locations (Source – NYCDEP).

Temperature

Water temperature is one of the most important variables in aquatic ecology. Temperature affects movement of molecules, fluid dynamics, and metabolic rates of organisms as well as a host of other processes. In addition to having its own potential "toxic" effect (i.e. when temperature is too high), temperature affects the solubility and, in turn, the toxicity of many other parameters. Generally the solubility of solids increases with increasing temperature, while gases tend to be more soluble in cold water (i.e. available O_2 to fish).

In densely wooded areas where the majority of the streambed is shaded, heat transferred from the air and groundwater inputs drive in-stream temperature dynamics. However, in areas that aren't shaded the water temperatures can rise much more quickly due to the direct exposure to the sun's radiation. Rock and blacktop also hold heat and can transfer the heat to the water (like hot coals in a grill). Annual fluctuation of temperature in a stream may drive many biological processes, for example, the emergence of aquatic insects and spawning of fish. Even at a given air temperature, stream temperature may be variable over short distances depending on plant cover, stream flow dynamics, stream depth and groundwater inflow. Water temperatures exceeding 77˚ Fahrenheit cannot be tolerated by brook trout, and they prefer water temperatures less than 68˚ Fahrenheit (TU, 2006).

The annual median water temperature of the Manor Kill from 1988 to 2007 varied from 4.4 degrees C (40˚F) in 1988 to 11.7 degrees C (53˚ F) in 1995 (Table 3.1.1). The median temperature during the summer months was 17.25 degrees C (63˚F), which was similar to the East and West Kills (Table 3.1.1).

Table 3.1.1. Annual and summer median temperatures for the Schoharie Creek and major tributaries (Source – NYCDEP) .

Site	Annual Median Temperature	Summer Median Temperature
Sugarloaf Brook, south of Tannersville (headwater trib. of Schoharie Creek) (n=4)	6.8˚ C (44˚F)	14.5˚ C (58˚F)
East Kill, at Schoharie Creek confluence (n=13)	7.9˚ C (46˚F)	17.4˚ C (63˚F)
West Kill, at private bridge upstream of Schoharie Creek confluence (n=13)	8.3˚ C (47˚F)	17˚ C (63˚F)
Batavia Kill, 1st bridge above Schoharie Creek confluence (n=19)	8.6˚ C (47˚F)	18.3˚ C (65˚F)
Schoharie Creek, just downstream of Lexington bridge (n=19)	8.4˚ C (47˚F)	18.8˚ C (66˚F)
Schoharie Creek, below Prattsville bridge (n=19)	8.8˚ C (48˚F)	19.2˚ C (66˚F)
Manor Kill at West Conesville (n=604)	7.98˚ C (46.4˚F)	17.25˚ C (63˚F)

Phosphorus

Phosphorus is a nutrient essential to plant growth. In aquatic ecosystems phosphorus occurs primarily in the form of organic phosphorus. Organic phosphorus is bound in plant and animal tissue and is unavailable for plant uptake. Phosphate (PO_4^{3-}) is a form that is available and needed by plants. Plants assimilate phosphate from the surrounding water and convert it to organic phosphorus. In freshwater ecosystems phosphate tends to be the nutrient that is least available for plant growth. Consequently, phosphate is often the limiting factor, and small additions to surface waters can result in large amounts of plant growth and eutrophication.

Phosphate binds to soil particles, which act to slow its transport. The soil-attached phosphate will often settle out in standing water (ponds/lakes/reservoirs), which once disturbed and resuspended, or due to anoxic conditions, can lead to excessive vegetation growth. The most likely sources of phosphate inputs include animal wastes, human wastes, fertilizer, detergents, disturbed land, road salts (anticaking agent), and storm water runoff. Based upon the average concentrations found in water samples from 85 sites across the United States in relatively undeveloped watersheds, the median concentrations of total phosphorus (P) and orthophosphate were 0.022 and 0.010 mg/L respectively (Clark et al., 2000). In general, any concentration over 0.05 mg/L of phosphate will likely have an impact on surface waters (Behar, 1996). However, in many streams and lakes concentrations of phosphate as low as 0.01 mg/L can have a significant impact on water resources by causing a proliferation of aquatic vegetation and phytoplankton. In order to control eutrophication, the USEPA recommended limiting phosphate concentrations to 0.05 mg/L in waters that drain to lakes, ponds and reservoirs, and 0.1 mg/L in free flowing rivers and streams (USEPA, 1996). DEP considers the 0.05 mg/L as a guidance value for streams. However, the critical guidance value for the Schoharie reservoir is 0.02 mg/L (NYCDEP, 1999).

The disturbances associated with the 1996 flooding elevated total annual median phosphorus concentrations at Prattsville to the highest for the period of record (1987-2005) at 0.036 mg/L. However, much of the total phosphorus is not biologically available. Gooseberry Creek contained high annual median total phosphorus concentrations (highest - 0.083), but the levels dropped significantly following the upgrades to the Tannersville

Wastewater Treatment facility. This trend holds true throughout the watershed. Table 3.1.2 provides a summary of annual median total phosphorus over the period of record (n), and is useful for comparison of basins against each other. However, since total phosphorus is often storm driven, the annual medians should not be compared to the guidance values for rivers and reservoirs.

Table 3.1.2. Annual median total phosphorus concentrations for the Schoharie Creek and major tributaries (Source – NYCDEP).

Site	Annual Median Total Phosphorus Concentrations (mg/L)
Sugarloaf Brook, south of Tannersville (headwater trib. of Schoharie Creek) (n=4)	0.003
East Kill, at Schoharie Creek confluence (n=13)	0.0075
West Kill, at private bridge upstream of Schoharie Creek confluence (n=13)	0.011
Batavia Kill, 1st bridge above Schoharie Creek confluence (n=19)	0.016
Schoharie Creek, just downstream of Lexington bridge (n=19)	0.0085
Schoharie Creek, below Prattsville bridge (n=19)	0.011
Manor Kill at West Conesville (n=431)	0.011

Nitrogen

Nitrogen is found in various forms in ecosystems including organic forms, nitrate (NO_{3-}), nitrite (NO_{2-}) and ammonium (NH_4+). The majority of nitrogen is in the form of a gas (N_2), which makes up approximately 80% of our air. It is converted into inorganic forms by some types of terrestrial plants (legumes) with nitrogen-fixing bacteria, lightning and microbes in the water and soil. Nitrate, the most mobile form of nitrogen, can either be assimilated by vegetation to make protein, leached into groundwater or surface water, or converted to nitrogen gas in the process of denitrification (Welsch et al. 1995). Nitrites, ammonia and ammonium are intermediate forms of nitrogen in aquatic systems and are quickly removed from the system by being converted to another form of nitrogen (NO_{3-} or N_2) (Behar, 1996). Ammonium is released into the system during animal or plant decomposition or when animals excrete their wastes. Through the process of nitrification, ammonium is oxidized to nitrates by nitrifying bacteria. Nitrate concentrations in water can serve as an indicator of sewage or fertilizer in surface or ground water.

Based upon average concentrations found in water samples from 85 sites across the United States in relatively undeveloped watersheds, the median concentrations of nitrate-nitrogen and total nitrogen were 0.087 and 0.26 mg/L respectively (Clark et al., 2000). Due to land uses and atmospheric deposition, the undeveloped watershed concentrations (below 0.087 mg/L) of in-stream NO_{3-} rarely occur in the Hudson Valley and Schoharie basin. Major sources of nitrate (most mobile form of nitrogen) in streams are municipal and industrial wastewater discharges and agricultural and urban runoff. Deposition from the atmosphere of the nitrogenous material in automobile exhaust and industrial emissions are also a source (Smith et al., 1991).

Nitrate in excessive amounts can accelerate eutrophication of surface waters, and can present a human health concern in drinking water. Any water that contains nitrate concentrations of 44 mg/L (equivalent to 10 mg/L nitrate-nitrogen for EPA and NYSDOH standards) or higher has the potential to cause methemoglobinemia, or "blue baby" disease in children, and the excess nitrate can indicate serious residential or agricultural contaminants (McCasland et al., 1998). Although the human health standard for nitrate consumption has little correlation with stream health, high levels of nitrate in both surface and ground water typically indicate widespread nonpoint source pollution.

The headwaters of the Schoharie Creek contained the highest annual median concentration of nitrate-nitrite as nitrogen (Table 3.1.3). This may be due to less stream flow in the headwaters available for dilution of nitrate concentrations, nonpoint source pollution, wastewater discharges and/or atmospheric deposition. Heisig (1998) found that the Batavia Kill (Schoharie tributary) had low nitrate concentrations during the growing season, when uptake by plants was greatest, and highest concentrations during the nongrowing season. This trend was also evident in DEP data and may reflect the effects of a heavily forested watershed on nutrients levels (Figure 3.1.10). Table 3.1.3 provides a summary of annual median nitrate-nitrite as nitrogen concentrations over the period of record (n), and is useful for comparison of basins against each other. However, since nitrogen is often storm driven, the annual medians should not be compared to the guidance values for rivers and reservoirs.

Table 3.1.3. Annual median nitrate-nitrite as nitrogen concentrations for the Schoharie Creek and major tributaries (Source – NYCDEP).	
Site	Annual Median nitrate-nitrite as nitrogen Concentrations (mg/L)
Sugarloaf Brook, south of Tannersville (headwater trib. of Schoharie Creek) (n=4)	0.37
East Kill, at Schoharie Creek confluence (n=13)	0.21
West Kill, at private bridge upstream of Schoharie Creek confluence (n=13)	0.15
Batavia Kill, 1st bridge above Schoharie Creek confluence (n=19)	0.14
Schoharie Creek, just downstream of Lexington bridge (n=19)	0.21
Schoharie Creek, below Prattsville bridge (n=19)	0.16
Manor Kill at West Conesville (n=430)	0.14

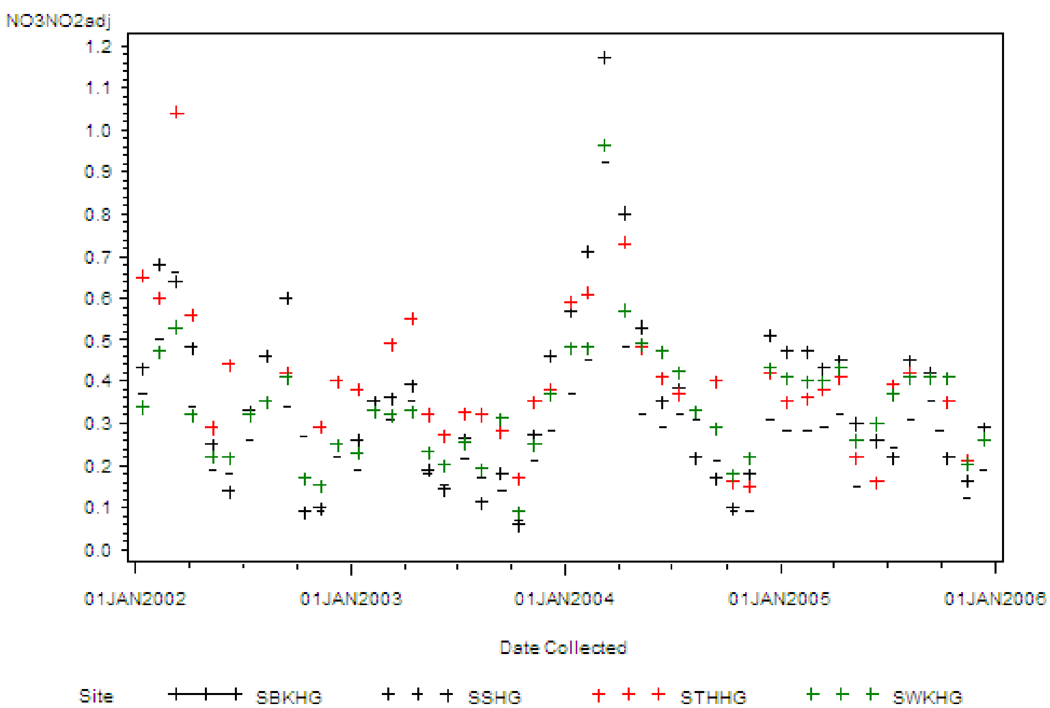


Figure 3.1.10. Nitrate-nitrite fluctuations by season in the Schoharie basin. Nitrate concentrations peak in the nongrowing season (SBKHG = Batavia Kill, SSHG = Sugarloaf Brook, STHHG = Toad Hollow Brook , and SWKHG = West Kill) (Source – NYCDEP).

Fecal Coliform

Fecal coliform bacteria are used as an indicator of possible sewage contamination because they are commonly found in human and animal feces. Although coliform bacteria are generally not harmful themselves, they indicate the possible presence of pathogenic bacteria, viruses, and protozoa that also live in the digestive tract. Therefore, the greater the numbers of fecal coliform bacteria colonies present the greater the human health risk for other pathogens. In addition to the human health risk, excess fecal coliform bacteria can cause increased oxygen demand, cloudy water, and unpleasant odors. Common sources of fecal coliform bacteria in waterways include poorly functioning sewage treatment plants, on-site septic systems, domestic and wild animal manure, and storm water runoff.

Testing for all bacteria, viruses and protozoa is very costly and time consuming. Therefore it is common practice to test for fecal coliform bacteria as an indicator of pathogens. The New York State Department of Health standard for contact recreation (swimming) is as follows: the fecal coliform bacteria density should not exceed 200 colonies per 100 ml, based on a logarithmic mean from a series of five or more samples over a thirty-day period.

Although not comparable to the Department of Health standard, annual median values from the Schoharie Creek at Prattsville for the period of record show that median fecal coliform colonies peaked around 26 CFU/100 mL in 1990 with lesser peaks of around 22 in 1992 and 1996. The highest annual median value of 35 CFU/100 mL was found at the Schoharie Creek at Lexington in 1997; and the highest summer median value was 400/100 mL above the Gooseberry Creek sewage treatment plant in 2002. Fecal coliform bacteria can survive longer in warmer water temperatures, so higher levels typically are found in the summer months. Table 3.1.4 provides a summary of annual and summer median fecal coliform bacteria levels over the period of record (n), and is useful for comparison of basins against each other. However, the annual medians should not be compared to the New York State standard due to the frequency of sampling.

Table 3.1.4. *Annual and summer median fecal coliform bacteria levels for the Schoharie Creek and major tributaries (Source – NYCDEP).*		
Site	Annual Median Fecal Coliform Bacteria (CFU/100 mL)	Summer Median Fecal Coliform Bacteria (CFU/100 mL)
Sugarloaf Brook, south of Tannersville (headwater trib. of Schoharie Creek) (n=4)	1	6
East Kill, at Schoharie Creek confluence (n=13)	4	8
West Kill, at private bridge upstream of Schoharie Creek confluence (n=13)	4	12
Batavia Kill, 1st bridge above Schoharie Creek confluence (n=19)	18	36
Schoharie Creek, just downstream of Lexington bridge (n=19)	14	60
Schoharie Creek, below Prattsville bridge (n=19)	13	28
Manor Kill at West Conesville (n=517)	10	14

Specific Conductivity

Specific conductivity describes the ability of water to conduct an electric current, and is an index of the concentration of chemical ions in solution. An ion is an atom of an element that has gained or lost an electron which will create a negative or positive state. High conductivity is created by the presence of anions such as chloride, nitrate, sulfate, and phosphate or cations such as sodium, magnesium, calcium, iron, and aluminum. The natural conductivity in streams and rivers is affected primarily by the geology of the area through which the water flows. Conductivity is often used to compare different streams because it is a cheap and easy measurement that can indicate when and where a site is being influenced by a source of contamination. Often when wastewater treatment plant effluent constitutes the majority of flow in a stream, it can be seen in water quality data due to its higher conductivity signature. Road salting practices can also impact conductivity.

Studies of inland fresh waters indicated that streams supporting good mixed fisheries had a conductivity range between 150 to 500 μmhos/cm (USEPA, 1997). The Schoharie at Lexington and Prattsville had a relatively low annual median conductivity (Table 3.1.5). The major contributor of the annual medians most likely reflects the geologic contribution to the total. Storm events would need to be monitored to pick up a nonpoint source pollutant signature.

Table 3.1.5. Annual median specific conductivity for the Schoharie Creek and major tributaries (Source – NYCDEP).

Site	Annual Median specific conductivity (µmhos/cm)
Sugarloaf Brook, south of Tannersville (headwater trib. of Schoharie Creek) (n=4)	20.7
East Kill, at Schoharie Creek confluence (n=13)	52
West Kill, at private bridge upstream of Schoharie Creek confluence (n=13)	54
Batavia Kill, 1st bridge above Schoharie Creek confluence (n=19)	86
Schoharie Creek, just downstream of Lexington bridge (n=19)	64.5
Schoharie Creek, below Prattsville bridge (n=19)	76.5
Manor Kill at West Conesville (n=580)	83

Dissolved Oxygen

Dissolved oxygen refers to oxygen gas (O_2) molecules in the water. The molecules are naturally consumed and produced in aquatic systems, and necessary for almost all aquatic organisms. If dissolved oxygen levels fall below a certain threshold, biologic integrity will be compromised. For example, on a scale of 0 to 14 mg/L, a concentration of 7 mg/L to 11 mg/L is ideal for most stream fish (Behar, 1996). Dissolved oxygen can be measured as the concentration of milligrams O_2 per liter (mg/L) or as percent saturation of O_2. Percent saturation is the amount of oxygen in a liter of water relative to the total amount of oxygen the water can hold at a given temperature. In cold water systems, a percent saturation of 60% to 79% is acceptable for most stream animals (Behar, 1996).

The New York State regulations for a stream designating as supporting trout spawning states that the DO should not be less than 7.0 mg/L from other than natural conditions. Data from 1987 to 2005 indicated that the annual median DO for the Schoharie and its tributaries ranged from about 10 to 12 mg/L and may dip down into the 9 mg/L range during hot summer months (Table 3.1.6). Dissolved oxygen concentrations may dip below 9 mg/L, particularly in the mornings of the summer months, but that level of analysis was outside the scope of this plan. The annual medians allow for a comparison between basins and seasons.

Table 3.1.6. Annual and summer median dissolved oxygen concentrations for the Schoharie Creek and major tributaries (Source – NYCDEP).

Site	Annual Median dissolved oxygen concentration (mg/L)	Summer Median dissolved oxygen concentration (mg/L)
Sugarloaf Brook, south of Tannersville (headwater trib. of Schoharie Creek) (n=4)	11.65	9.9
East Kill, at Schoharie Creek confluence (n=13)	11.5	9.5
West Kill, at private bridge upstream of Schoharie Creek confluence (n=13)	11.6	9.5
Batavia Kill, 1st bridge above Schoharie Creek confluence (n=19)	11.55	9.2
Schoharie Creek, just downstream of Lexington bridge (n=19)	11.65	9.4
Schoharie Creek, below Prattsville bridge (n=19)	11.3	9.0
Manor Kill at West Conesville (n=419)	11.8	9.4

<u>*Sulfur*</u>

Sulfur in natural waters is essential in the life processes of plants and animals. Although the largest Earth fraction of sulfur occurs in reduced form in igneous and metamorphic rock, there is significant sulfur in sedimentary rock as well. When sulfide minerals undergo weathering in contact with oxygenated water, the sulfur is oxidized to yield stable sulfate ions that become mobile in solution. Another major source of sulfate in the environment is the combustion of coal, petroleum and other industrial processes such as smelting of sulfide ores. Atmospheric deposition both as dry particulates and entrained in precipitation can cause acid rain that can alter stream chemistry. Sulfate is highly mobile and often ends up in our local streams, lakes and reservoirs. Sulfate is classified under the EPA secondary maximum contaminant level (SMCL) standards. The SMCL for sulfate in drinking water is 250 milligrams per liter (mg/l). Sulfate was not monitored by DEP until 1994. Since that time, annual median concentrations found in the Schoharie Creek varied from 4 to 5 mg/L in the headwaters area to around 5 to 6 mg/L at Prattsville and 7 mg/l in the Manor

Kill. Sulfate values basinwide have dropped since 1994, and despite a brief rise in 2002, have remained at a lower level, possibly due to reduced sulfur emissions throughout the US.

pH

For optimal growth, most species of aquatic organisms require a pH in the range of 6.5 to 8.0, and variance outside of this range can stress or kill organisms. Due to the acidity of rainfall in the northeast, maintaining this range is of concern. According to the NYSDEC (2004a), average pH of rainfall in New York ranges from 4.0 to 4.5. To understand the drivers of pH in the Schoharie basin we can look at one of its tributaries. The Batavia Kill basin contains an abundant carbonate source in the till and glacial melt water deposits in the upland areas from the north and south, which act to raise the pH of the Batavia Kill through tributary inputs (Heisig, 1998). This carbonate source is not present in the bedrock aquifer, or glacial deposits near or within the Batavia Kill valley or uplands to the east (Heisig, 1998). Basically, the carbonate materials (limestone fragments) were imported to the basin and deposited by glaciers in an uneven distribution, primarily in the uplands located north and south. Since most of the Schoharie basin water has a similar pH, this phenomenon is likely true for the entire basin. This carbonate material provides a buffer for acidic inputs, but remains in a delicate balance as observed in the Batavia Kill when tributary inputs drop during the hot summer months, and the stream is primarily fed by acidic groundwater, the instream pH becomes more acidic (Heisig, 1998).

Annual median pH values for the period of record for the Schoharie Creek and tributaries range from 7.2 to 7.4, with one headwater location at 6.3 (Table 3.1.7). The annual medians are similar to the pH neutral of 7.0, but annual medians are too course to differentiate between seasons and flow regimes.

Table 3.1.7. Annual median pH for the Schoharie Creek and major tributaries (Source – NYCDEP).	
Site	Annual Median pH
Sugarloaf Brook, south of Tannersville (headwater trib. of Schoharie Creek) (n=4)	6.3
Gooseberry Creek, above Tannersville STP (n=19)	7.2
East Kill, at Schoharie Creek confluence (n=13)	7.25
West Kill, at private bridge upstream of Schoharie Creek confluence (n=13)	7.4
Batavia Kill, 1st bridge above Schoharie Creek confluence (n=19)	7.3
Schoharie Creek, just downstream of Lexington bridge (n=19)	7.2
Schoharie Creek, below Prattsville bridge (n=19)	7.3
Manor Kill at West Conesville	7.5

Chloride

Chlorides are salts resulting from the combination of chlorine gas with a metal. Chlorine as a gas is highly toxic, but in combination with a metal such as sodium it becomes useful to plants and animals. Small amounts of chlorides are required for normal cell function in plants and animals. Common chlorides include sodium chloride (NaCl), calcium chloride ($CaCl_2$) and magnesium chloride ($MgCl_2$). Chlorides can get into surface water from several sources including geologic formations containing chlorides, agricultural runoff, industrial wastewater, effluent from wastewater treatment plants, and the salting of roads. Excess chloride can contaminate fresh water streams and lakes, negatively affecting aquatic communities.

Concentrations of chloride of approximately 140 mg/L should be protective of freshwater organisms for short-term exposure; concentrations less than 35 mg/L are likely protective during long-term exposures (Environment Canada, 2001). Overall, approximately 5 percent of species would experience effects from chronic exposure to concentrations of chloride of 210 mg/L, while 10 percent of species would be affected at concentrations of 240 mg/L (Environment Canada, 2001). According to the United States Environmental Protection Agency, biota on average should not be affected if the four-day average concentration of chloride does not exceed 230 mg/L more than once every three years (USEPA, 2005a). Biotic impacts would be minimal if the one-hour average chloride concentration did not exceed 860 mg/L more than once every three years (USEPA, 2005a).

The major sources of chloride in the Schoharie watershed are most likely geology, road salting and wastewater treatment plants. The annual median chloride concentrations are low across the board (Table 3.1.8). Annual medians are too coarse to tease out specific contributors. However, it is interesting that the highest annual median chloride concentrations occurred in the year 2001, which coincided with low annual stream flows (median annual stream at Prattsville for period of record ~103 years is 462.5 CFS; in 2001 median annual streamflow was 178.3 CFS). This may have been due to geology, landscape or a combination of the two sources of chloride becoming more concentrated in lower flows.

Table 3.1.8. Annual median chloride concentrations for the Schoharie Creek and major tributaries (Source – NYCDEP).

Site	Annual Median Chloride (mg/L)
Sugarloaf Brook, south of Tannersville (headwater trib. of Schoharie Creek) (n=4)	1.7
Gooseberry Creek, above Tannersville STP (n=19)	14.9
Gooseberry Creek, below Tannersville STP (n=19)	17.2
East Kill, at Schoharie Creek confluence (n=13)	5.3
West Kill, at private bridge upstream of Schoharie Creek confluence (n=13)	4.1
Batavia Kill, 1st bridge above Schoharie Creek confluence (n=19)	12.1
Schoharie Creek, just downstream of Lexington bridge (n=19)	7.8
Schoharie Creek, below Prattsville bridge (n=19)	8.2
Manor Kill at West Conesville (n=440)	5.5

Biomonitoring

Benthic macroinvertebrates (BMI) can be simply defined as animals without backbones that are larger than 1 millimeter and live at least a portion of their life cycles in or on the bottom of a body of water. In freshwater systems these animals may live on rocks, logs, sediments, debris and aquatic plants during their various life stages. A few common examples of BMIs include crustaceans such as crayfish, mollusks such as clams and snails, aquatic worms, and the immature forms of aquatic insects such as stonefly, caddisfly and mayfly nymphs.

BMIs function at the lower levels of the aquatic food chain, with many feeding on algae, detritus, and bacteria. Some shred and eat leaves and other organic matter that enters the water, and others are predators. Because of their abundance and position in the aquatic food chain, BMIs play a critical role in the natural flow of energy and nutrients through the aquatic system (Covich et al., 1997). For example, Sweeney (1993) demonstrated in a second order stream, that leaf litter and woody debris were primarily consumed in the forested woodlot where the debris originated. Also, as benthos die, they decay, leaving behind nutrients that are reused by aquatic plants and other animals in the food chain. Insects fill the roles of predators, parasites, herbivores, saprophages, and pollinators, among others, which indicate the pervasive ecological and economic importance of this group of animals in both aquatic and terrestrial ecosystems (Rosenberg et al., 1986).

Biological assessments have been used by many states to evaluate the effectiveness of water quality programs, particularly for nonpoint source impact determinations (USEPA, 2002). For example, biological assessment models have been tested with field data and the results suggested that macroinvertebrate data collected for establishing the degree of water quality impairment can also be used to identify the impairment source with reasonable accuracy (Murray et al., 2002). In addition, it has been suggested that the percentage of chironomids in samples may be a useful index of heavy metal pollution (Winner et al., 1980). Furthermore, the Ohio EPA employs biological response signatures, based on biological, chemical, physical, bioassay, pollution source, and watershed characteristic, that consist of key response components of the biological data that consistently indicate one type of impact over another (Yoder, 1991). In New York State, the first recorded biological monitoring effort dates from 1926-1939, but the regulatory role of stream biomonitoring did not begin in New York until after the passage of the Federal Water Pollution Control Act Amendments of 1972 (Clean Water Act). The primary objective of New York State's program was to evaluate the relative biological health of the state's streams and rivers through the collection and analysis of macroinvertebrate communities (Bode et al, 2002).

Biological monitoring appears to be an attractive methodology for documenting water quality for several reasons. First, the community collected at a given site reflects the water quality at that site over several weeks, months, or years. The alternative methodology of grabbing a water sample reflects the water quality at the instant the sample is collected (i.e. a snap shot image). Second, the community-based approach focuses on the biological integrity of the water body, and not a limited number of chemical parameters. Third, samples can be preserved in reference collections for future application; this provides a convenient routine of summer collection and winter analysis. Finally, biological assessments tend to be much more cost effective than chemical analysis. Table 3.1.9 lists the rationale for biomonitoring in New York State (Bode et al, 2002).

Table 3.1.9. Rationale for the analysis of macroinvertebrate communities to determine water quality of streams and rivers in New York State (Bode et. al., 2002).

1. BMIs are sensitive to environmental impacts;
2. BMIs are less mobile than fish, and thus can avoid discharges;
3. They can indicate the effects of spills, intermittent discharges, and lapses in treatment;
4. They are indicators of overall, integrated water quality, including synergistic effects and substances lower than detectable limits;
5. They are abundant in most streams, and are relatively easy and inexpensive to sample;
6. They are able to detect non-chemical impacts to the habitat, such as siltation or thermal change;
7. They are readily perceived by the public as tangible indicators of water quality;
8. They can often provide an on-site estimate of water quality;
9. They bioaccumulate many contaminants to concentrations that analysis of their tissues is a good monitor of toxic substances in the aquatic food chain;
10. They provide a suitable endpoint to water quality objectives.

Standardized protocols for benthic macroinvertebrate monitoring were developed in the mid-1980s due to the need for cost-effective habitat and biological survey techniques (Plafkin et al., 1989). The primary driver of the development was limited economic resources available to states with miles of unassessed streams. It was also recognized that it was crucial to collect, compile, analyze, and interpret environmental data rapidly to facilitate management decisions and resulting actions for control and/or mitigation of impairment. Therefore, the conceptual principles of rapid bioassessment protocols (RBPs) were as follows: cost-effective, yet scientifically valid procedures; provisions for multiple site investigations in a field season; quick turn-around of results for management decisions, easily translated to management and the public; and environmentally benign procedures (Barbour et al. 1999). Finally, in order to save time, it was recognized that a certain degree of accuracy would need to be sacrificed, and a field-based assessment would be necessary (Hisenhoff, 1988). Therefore, a family based assessment was developed that could be calculated in the field by professionals (Hilsenhoff, 1988). This field based assessment allows professionals to focus their time and efforts on the more in-depth analysis of areas that indicated degradation in the rapid field assessment.

Schoharie Creek and its tributaries exhibit good water quality based on BMI community structure (Figure 3.1.11). All the sites sampled have assessed as non-impaired

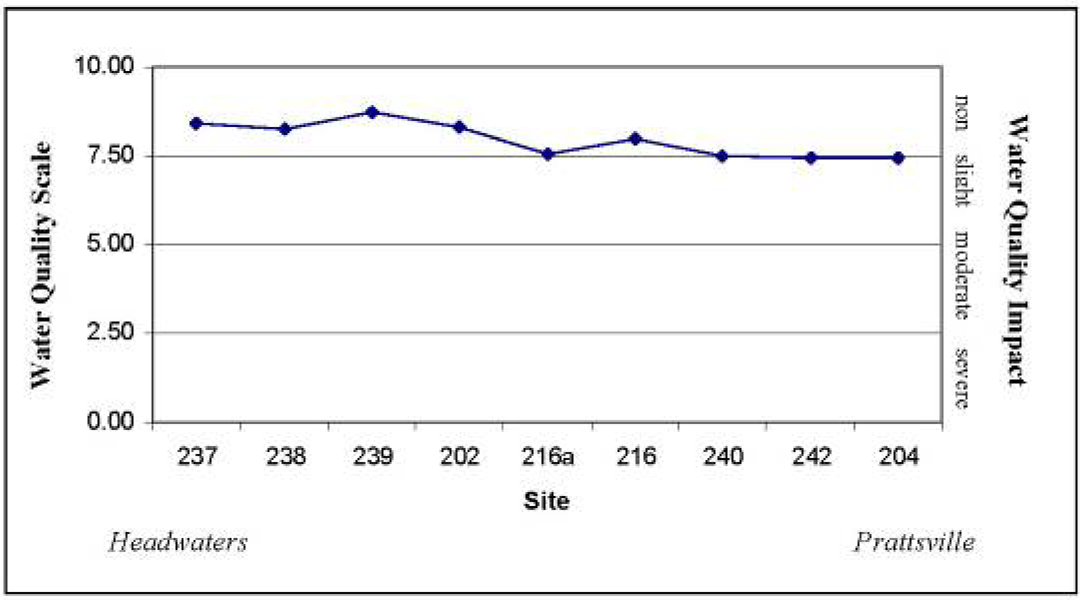


Figure 3.1.11. Mean water quality assessment values demonstrating water quality impact for sample years 1994 through 2005 for the Schoharie Creek. The mean value for all the years sampled is represented by the line which runs from upstream (237) to downstream (204). The stream was nonimpacted for much of its length, but does show a slight impact in the downstream stretch (240 to 204). N = 237 (4), 238 (4), 239 (3), 202 (12), 216a (2), 216 (9), 240 (4), 242 (2), 204 (11).

in at least one year of sampling (Figure 3.1.12). However, in general, the macroinvertebrate data indicate a slight decline in water quality from the headwaters (site 237) to Prattsville (site 204), particularly downstream of Lexington (site 216). DEP researchers employed impact source determination analysis in an attempt to explain this trend, but the results were inconclusive. Monitoring efforts will continue in order to determine if this downstream trend is a reaction to in-stream factors such as flood or low flows, water temperature and/or habitat disturbance or if there is a more insidious cause. In 2008, researchers from SUNY Cobleskill conducted macroinvertebrate and fish surveys along the Manor Kill. See the macroinvertebrate and fish reports (Appendix F) for more detailed information regarding the surveys and their findings. The results of these findings also have important implications to the viability of the Schoharie as a cold water fishery resource.

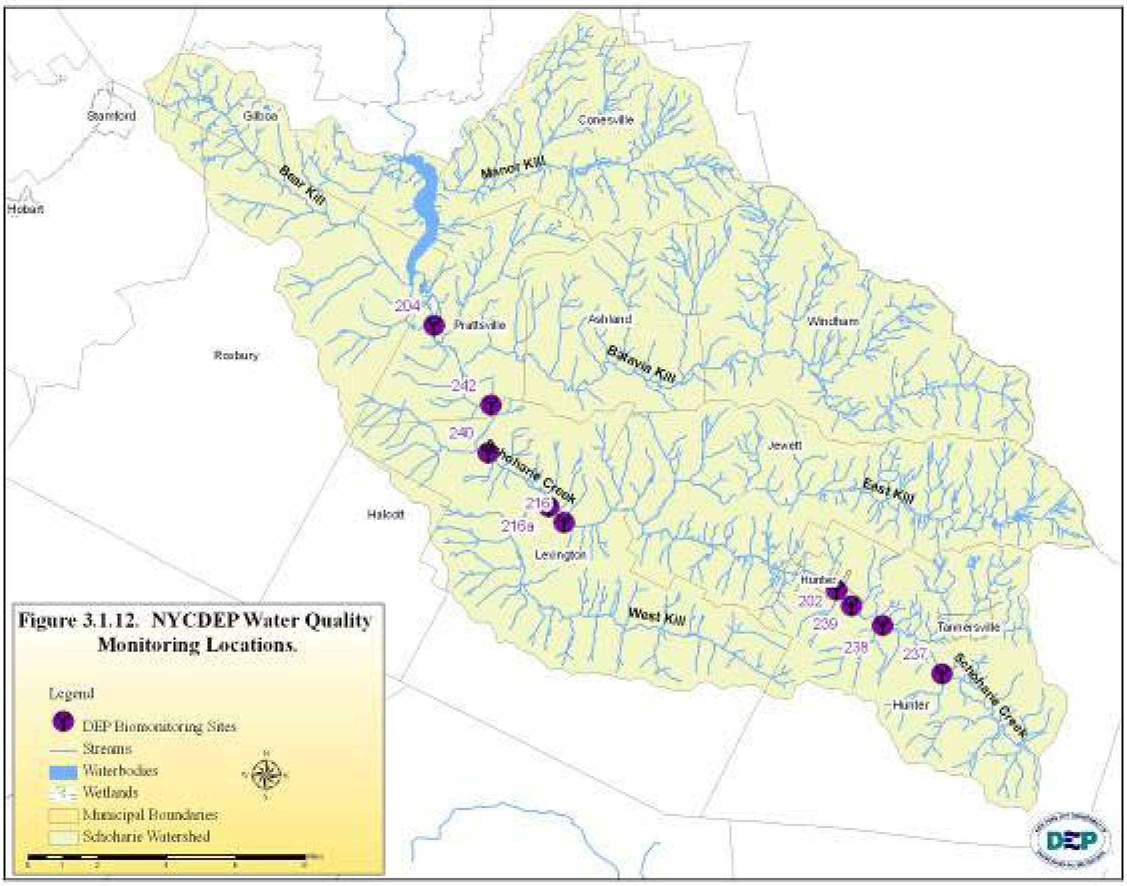


Figure 3.1.12. NYCDEP Water Quality Monitoring Locations.

Stream Management Implications

Determining whether a stream has good or bad water quality often depends largely upon the end user. For the purposes of the NYC water supply, the Schoharie watershed supplies good quality water with the exception of the time period following large storms in which in-stream turbidity and suspended solids are high. For water supply purposes, DEP believes these temporary spikes in turbidity can best be controlled through operational changes in the Catskill water supply system. Streams in the Catskills have moved large amounts of suspended sediment during storms for thousands of years; and will continue to for thousands of years until all the glacial lake sediments and glacial till have been removed from the stream network. That being said, watershed landowners do have direct influence over land uses in the watershed and there are other, more local reasons for watershed protections measures to be implemented. For example, protecting and enhancing the fishery could also benefit the economy and aesthetic values of the region. Proper watershed management can also assist in protecting infrastructure, reducing flood damages and help to develop a stream stewardship ethic. Taken together, all these benefits can increase the quality of life of watershed residents, while providing high quality drinking water to the residents of New York City into the future.

In 2001, approximately 85% of the Schoharie basin was forested. However, this is somewhat deceptive since much of the developed land is on the gentle slopes adjacent to the stream, particularly roads. Although, in general, water quality appears to be good, there also seems to be specific areas where water quality may be impacted; and late summer water temperatures are high for a cold-water fishery. Future development in the stream corridor, with a resulting increase in impervious surface, may increase runoff and impair water quality. Therefore, management efforts should be focused on preventing further human-induced degradation through implementation of best management practices designed to reduce/minimize impacts. These efforts should be both direct measures such as remediating failing septic systems and upgrading sewer treatment plants (point sources of pollution); and indirect measures such as reducing stormwater inputs, properly installing new infrastructure and planting riparian buffers. In areas where existing infrastructure is acting to destabilize the stream, or is threatened by erosion, stabilization techniques incorporating natural channel design should be employed. Reforesting the banks of the Schoharie and its tributaries, coupled with the protection of cold groundwater seeps, may help to lower summer water temperatures and enhance the fishery.

References

Arscott, D.B., Aufdenkampe, A.K., Bott, T.L., Dow, C.L., Jackson, J.K., Kaplan, L.A., Newbold, J.D., and Sweeney, B.W. 2004. Water Quality Monitoring in the Source Water Areas for New York City: An Integrative Watershed Approach: A Report on Year 4 (2003) Monitoring Activities. Stroud Water Research Center, Avondale, PA: 76.

Behar, Sharon. 1996. Testing the Waters Chemical and Physical Vital Signs of a River. River Watch Network, Montpelier, VT. 147 P.

Barbour, M.T., Gerritsen, J., Snyder, B.D., Stribling, J.B. 1999. Rapid Bioassessment Protocols for Use in Wadeable Streams and Rivers: Periphyton, Benthic Macroinvertebrates, and Fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington, D.C.

Bode, R.W., Novak, M.A., Abele, L.E. 1995. Biological Stream Assessment, Schoharie Creek. New York State Department of Environmental Conservation Division of Water, Stream Biomonitoring Unit, Albany, NY.

Bode, R.W., Novak, M.A., Abele, L.E., Heitzman, D.L., and Smith, A.J. 2002. Quality Assurance Work Plan For Biological Stream Monitoring in New York State. NYS Department of Environmental Conservation, Division of Water, Stream Biomonitoring Unit, Albany, NY.

Bode, R.W., Novak, M.A., Abele, L.E., Heitzman, D.L., and Smith, A.J. 2004. 30-Year Trends in Water Quality of Streams and Rivers of New York State based on Macroinvertebrate Data 1972-2002. NYS Department of Environmental Conservation, Division of Water, Stream Biomonitoring Unit, Albany, NY.

Booth, D. 2000. Forest Cover, Impervious Surface Area, and the Mitigation of Urbanization Impacts in King County, Washington. Center for Urban Water Resources Management, University of Washington, Seattle, WA.

CCEUC. 2007. Upper Esopus Creek Management Plan. Cornell Cooperative Extension of Ulster County, Kingston, NY.

Clark, G.M., Mueller, D.K., Mast, M.A. August 2000. Nutrient Concentrations and Yields in Undeveloped Stream Basins of the United States. Journal of the American Water Resource Association, Vol 36(4): 849-860.

Covich, A.P., Palmer, M.A., and Crowl, T.A. 1997. The role of benthic invertebrate species in freshwater ecosystems, zoobenthic species influence energy flows and nutrient cycles. BioScience 49(2):119-127.

Environment Canada. 2001. The Canadian Environmental Protection Act Assessment Report on Road Salts. Available http://www.ec.gc.ca/substances/ese/eng/psap/final/roadsalts.cfm (Accessed June 3, 2005).

Fischenich, J.C. 2003. Effects of Riprap on Riverine and Riparian Ecosystems. United States Army Corps of Engineers ERDC, Vicksburg, MS: publication # ERDC/EL TR-03-4GCSWCD, 2003.

GCSWCD, 2003. Batavia Kill Stream Management Plan. Greene County Soil and Water Conservation District, Cairo, NY.

GCSWCD, 2005. West Kill Stream Management Plan. Greene County Soil and Water Conservation District, Cairo, NY.

Heisig, P. 1998. Water Resources of the Batavia Kill Basin at Windham, Greene County, NY. Publication # WRIR 98-4036. United States Geological Survey, Troy, NY.

Heisig, P. 2000. Effects of Residential and Agricultural Land Uses on the Chemical Quality of Baseflow of Small Streams in the Croton Watershed, Southeastern New York. Publication # WRIR 99-4173. United States Geological Survey, Troy, NY.

Henley, W.F., Patterson, M.A., Neves, R.J. and Lemly, A.D. 2000. Effects of Sedimentation and Turbidity on Lotic Food Webs: A Concise Review for Natural Resource Managers. Reviews in Fisheries Science 8(2): 125-139.

Hilsenhoff, W.L. 1988. Rapid field assessment of organic pollution with a family-level biotic index. J.N. Am Benthol. Soc. 7(1): 65-68.

Joint Venture, 2004. *Catskill Turbidity Control Study: Phase 1 Final Report*. Prepared by Gannet-Flemming and Hazen and Sawyer for NYCDEP.

Joint Venture, 2006. *Catskill Turbidity Control Study: Phase I1 Final Report*. Prepared by Gannet-Flemming and Hazen and Sawyer for NYCDEP.

Jones III, D.J., Helfman, G.S., Harper, J.O. and Bolstad, P.V. 1999. Effects of Riparian Forest Removal on Fish Assemblages in Southern Appalachian Streams. Conservation Biology 13(6): 1454-1465.

Kratzer, E.B., Jackson, J.K, Arscott, D.B., Aufdenkampe, A.K., Dow, C.L., Kaplan, L.A., Newbold, J.D., and Sweeney, B.W. 2006. Macroinvertebrate distribution in relation to land use and water chemistry in New York City drinking water supply watersheds. Journal of the North American Benthological Society, 25(4):954-976.

LeChevallier, M.W., Evans, T.M. and Seidler, R.J. 1981. Effect of Turbidity on Chlorination Efficiency and Bacterial Persistence in Drinking Water. Applied and Environmental Microbiology Vol 42(1): 159-167.

Limburg, K.E. and Schmidt, R.E. 2000. Patterns of Fish Spawning in Hudson River Tributaries: Response to an Urban Gradient?. Ecology Volume 71 (4): 1238 – 1245.

May, C.W., Horner, R.R., Karr, J.R., Mar, B.W. and Welch, E.B. 2000. Effects of Urbanization on Small Streams in the Puget Sound Ecoregion. Watershed Protection Techniques, 2(4): 483-494.

McCasland, M., Trautmann, N. M., Wagenet, R. J., Porter, K.S. 1998. Nitrate: Health Effects in Drinking Water. Natural Resources, Cornell Cooperative Extension, 5123 Comstock Hall Cornell University, Ithaca, New York. On Internet: http://pmep.cce.cornell.edu/facts-slides-self/facts/nit-heef-grw85.html.

Murray, K.R., Bode, R.W., Phillips, P.J., Wall, G.L. 2002. Impact Source Determination with Biomonitoring Data in New York State: Concordance with Environmental Data. Northeastern Naturalist 9(2): 127-162.

Newcombe, C.P. and Jensen, J.O. 1996. Channel Suspended Sediment and Fisheries: A Synthesis for Quantitative Assessment of Risk and Impact. North American Journal of Fisheries Management 16(4): 693-727.

Newcombe, C.P. 2003. Impact Assessment Model For Clear Water Fishes Exposed to Excessively Cloudy Water. J. of the American Water Resources Association (JAWRA) 39(3):529-544.

Novak, M.A., Bode, R.W., Abele, L.E. 1989. Schoharie Creek Biological Assessment. New York State Department of Environmental Conservation Division of Water, Stream Biomonitoring Unit, Albany, NY.

NYCDEP, 1999. Proposed Phase II Phosphorus TMDL Calculations for Schoharie Reservoir. New York City Department of Environmental Protection, Division of Drinking Water Quality Control. Valhalla, NY.

NYC DEP, 2002. Integrated Monitoring Report, New York City Department of Environmental Protection, Bureau of Water Supply, Division of Drinking Water Quality Control, Valhalla, NY 10595.

NYCDEP. 2004. New York City Department of Environmental Protection 2003 Watershed Water Quality Annual Report. NYCDEP, Division of Drinking Water Quality Control, Valhalla, NY.

NYC DEP, 2006. 2005 Watershed Water Quality Annual Report. New York City Department of Environmental Protection, Division of Drinking Water Quality Control. Valhalla, NY: 123 pp.

NYSDEC. 2004. New York State Water Quality Section 305b Report (2004). New York State Department of Environmental Conservation, Bureau of Watershed Assessment and Management, Division of Water, Albany, NY. Available on web: http://www.dec.state.ny.us/website/dow/bwam/305b.html.

NYSDEC. 2004a. Some Questions and Answers on Acid Rain. New York State Department of Environmental Conservation, 625 Broadway, Albany, NY. http://www.dec.state.ny.us/website/dar/ood/acidrain.html (Accessed October 8, 2004).

NYSDOH. 2006. Chemicals in Sportfish and Game – 2006 – 2007 Health Advisories. New York State Department of Health, Albany, NY. Available on web: www.nyhealth.gov/nysdoh/fish/fish.

Plafkin, J.L., Barbour M.T., Porter K.D., Gross S.K. & Hughes R.M. 1989. Rapid Bioassessment Protocols for Use in Streams and Rivers: Benthic Macroinvertebrates & Fish. US Environmental Protection Agency Assessment and Watershed Protection Division, EPA/440/4-89/001,Washington.

Potter, P.M., Cubbage, F.W., and Schaberg, R.H. 2005. Multiple-scale landscape predictors of benthic macroinvertebrate community structure in North Carolina. Landscape and Urban Planning 71: 77-90.

Rosenberg, D.M., Danks, H.V., Lehmkuhl, D.M. 1986. Importance of Insects in Environmental Impact Assessment. Environmental Management 10(6): 773-783.

Scheuler, T.R. and Holland, H.K. 2000. Housing Density and Urban Land Use as Indicators of Stream Quality. In: The Practice of Watershed Protection 2(4): 735-739.

Smith, R.A., Alexander R.B., and Lanfear, K. J. 1991. Stream Water Quality in the Conterminous United States -- Status and Trends of Selected Indicators During the 1980's. National Water Summary 1990-91 -- Stream Water Quality, U.S. Geological Survey Water-Supply Paper 2400. U.S. Geological Survey, 410 National Center, Reston, VA.

Sweeney, B.W. 1993. Effects of streamside vegetation on macroinvertebrate communities in White Clay Creek in Eastern North America. Proceedings of the Academy of Natural Sciences of Philadelphia 144: 291-340.

TU, 2006. Back the Brookie Education, Biology and Habitat Needs. Trout Unlimited. Available on web: http://www.brookie.org/site/pp.asp?c=liKVL3POLvF&b=1595121.

USEPA. 1996. Environmental Indicators of Water Quality in the United States: United States EPA Report # EPA 841-R-96-002: 25 p.

USEPA. 1997. Volunteer Stream Monitoring: A Methods Manual. EPA 841-B-97-003. United States Environmental Protection Agency Office of Water, Washington, DC. 173 P.

USEPA. 2002. Summary of biological assessment programs and biocriteria development for states, tribes, territories, and interstate commissions: Streams and wadeable rivers. EPA-822-R-02-048. U.S. Environmental Protection Agency Office of Water, Washington, D.C.

USEPA. 2005. National Management Measures to Protect and Restore Wetlands and Riparian Areas for the Abatement of Nonpoint Source Pollution. EPA 841-B-05-003. U.S. Environmental Protection Agency; Office of Water; Washington, D.C.

USEPA. 2005a. Current National Water Quality Criteria, chloride value is based on a 304(a) aquatic life criterion that was derived using the 1985 Guidelines (Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses, PB85-227049, January 1985) and was issued in (EPA 440/5-88-001). Available http://www.epa.gov/waterscience/criteria/wqcriteria.html#G2 (Accessed June 3, 2005).

Wang, L., Lyons, J., Kanehl, P. and Bannerman, R. 2001. Impacts of Urbanization on Stream Habitat and Fish Across Multiple Spatial Scales. Environmental Management Vol. 28(2): 255-266.

Welsch, David J., Smart, David L., Boyer, James N., Minkin, Paul, Smith, Howard C., and McCandless, Tamara L. 1995. Forested Wetlands, Functions, Benefits, and the Use of Best Management Practices. United States Department of Agriculture Forest Service, Radnor, PA: 30-31 P.

Winner, R.W., Boesel, M.W., Farrell, M.P. 1980. Insect Community Structure as an index of heavy-metal pollution in lotic ecosystems. Can. J. Fish. Aquat. Sci. 37: 647-655.

Yoder, C.O. 1991. The integrated biosurvey as a tool for evaluation of aquatic life use attainment and impairment in Ohio surface waters. *In*: Biological Criteria: Research and Regulation. U.S. Environmental Protection Agency, Office of Science and Technology, Washington, D.C.: 110-122.

3.2 Introduction to Stream Processes

"You cannot step twice into the same river; for fresh waters are ever flowing in upon you." -Heraclitus, 2500 B.P

Ask anyone who lives streamside, and they'll tell you that living around streams carries both benefits and risks; to enjoy the benefits, we accept the risks. Both the pleasures as well as the dangers of living near streams stem in part from their ever-changing nature. Icy spring flood-flows are exciting and beautiful as long as they don't creep up over their banks and run across your yard into the basement window, or suddenly tear out a stream bank and begin flowing down the only access road to your house. For many reasons, the relatively flat land in the floodplain of a stream may be an inviting place to build a home or road –in fact it may be the only place– but as long-time residents of floodplains know only too well, it's not a matter of *if* they will see floodwaters, but of *when*.

As changeable as streams are, though, there is also something consistent about the way they change through the seasons, or even through an individual storm. As unpredictable as streams can be, they are also predictable in many ways. If we take the time to observe them carefully, we can begin to understand the patterns in the way streams behave and, more importantly, what we might do in our individual roles as stream stewards and managers to increase their benefits to us, and to reduce the risks they pose.

This section of the management plan is provided to offer the reader a basic explanation of what stream scientists know about how streams "make themselves": why they take different forms in different settings, what makes them evolve, and how we can manage them effectively to increase the benefits and reduce the risks they offer.

It's obvious that streams drain water off the landscape, but they also have to carry *bedload* –gravel, cobble, and even boulders– eroded from streambeds and banks upstream.

If you stand near the bank of a mountain stream during a large flood event, you can feel the ground beneath your feet vibrate as gravel, cobble and boulders tumble against each other as they are pushed along by the force of the floodwaters down the streambed. As the water begins to rise in the channel during a major storm, at some point the force of the water begins to move the material on the bottom of the channel. As the stormwaters recede, the force falls and the gravel and cobble stop moving. The amount of water moving through the channel determines the amount of *bedload* moving through it as well.

To effectively manage the stream, managers first need to understand how much water is delivered from the landscape to the stream, at any particular point in the system. The amount of water any stream will carry off the landscape is primarily determined by four characteristics of the region:

- the climate, specifically the amount of rainfall and the temperatures the region typically sees throughout the course of a year;
- the topography of the region;
- the soils and bedrock geology; and
- the type of vegetation (or other land cover like roads and buildings) and its distribution across the landscape.

These characteristics also play key roles in determining the type and frequency of flood hazards in the region, the quality of the water, and the health of the stream and floodplain ecosystems.

The shape and size of a stream channel adapts itself to the amount of water and bedload it needs to carry. Within certain limits, the form, or *morphology*, of a stream is self-adjusting, self-stabilizing, self-sustaining. If stream managers exceed those limits, however, the stream may remain unstable for a long time.

Over the period since the last glaciers retreated some 12,000 years ago, the Catskills streams have adapted their shape to these regional conditions. Because the climate, topography, geology and vegetation of a region usually change only very slowly over time, the amount of water moving through a stream from year to year, or *streamflow regime*, is fairly consistent at any given location.[1] This stream flow regime, in turn, defines when and how much bedload will be moving through the stream channel from year to year. Together, the movement of water and bedload carve the form of the stream channel into the landscape. Because the streamflow regime is fairly consistent year after year, then, the form of the stream channel also changes relatively slowly, at least in the absence of human influence. Over the 120 centuries since glaciers covered the region, the stream and the landscape conditions evolved a dynamic balance.

However, as we made our mark on the landscape –clearing forest for pastures, or straightening a stream channel to avoid having to build yet another bridge– we unintentionally changed that balance between the stream and its landscape. We may notice that some parts of the stream seem to be changing very quickly, while others remain much the same year after year, even after great floods. Why is this? Streams that are in dynamic balance with their landscape adapt a form that can pass the water and bedload associated with both small and large floods, regaining their previous form after the flood passes. This is the definition of stability. In many situations, however, stream reaches become unstable when some management activity has upset that balance, and altered the stream's ability to move its water and bedload effectively.

[1]One exception is when the vegetation changes quickly, such as can happen during forest fires, volcanic eruptions or even rapid commercial or residential development.

The amount of potential force the water has to move its rock is determined by its **slope** –the steeper the slope, the more force– and its **depth** –the deeper the stream, the more force. So, for example, if changes made to a stable reach of stream reduce its slope and/or depth, the stream may not be able to move effectively the bedload supplied to it from upstream. The likely result will be that the material will deposit out in that section, and the streambed will start building up, or *aggrading*.

On the other hand, when we straighten a stream, we shorten it; this means that its slope is increased, and likewise its potential force to move its bedload. Road encroachment has narrowed and deepened many streams, with the same result: too much force, causing the bed of the stream to *degrade* and, ultimately, to become *incised,* like a gully in its valley. Both situations, aggrading and degrading, mean that the stream reach has become unstable, and both can lead to rapid bank erosion, as well as impairment of water quality and stream health. Worse yet, these local changes can spread upstream and downstream, causing great lengths of stream to become unstable.

The lay of the land determines the pattern and grade of the stream, but the stream also shapes the lay of the land. The stable form for a particular stream depends on the larger form of the valley it flows through.

The stream pattern we now see throughout the Catskills is the result of millions of years of landscape evolution: fractured bedrock, chiseled repeatedly by rivers, and then glaciers, and then rivers again, as glacial ages came and went, as valleys were eroded out of the mountains and washed out to sea. In the broader valleys like the Esopus or the Delaware, floodplains formed as they filled with cobble and gravel, sand and silt carved away from the steeper mountainsides by roaring meltwater. The material often settled out as the streams entered into local lakes, created where notches at the lower end of the valley were dammed by glacial ice. When the ice dams melted, the lakebed remained a fairly flat valley floor, poorly vegetated initially, through which the stream could meander from one side of the valley to the other.

As the streams, century by century, shaped these flatter valleys they flowed through, the resulting shape of the valleys in turn changed the streams. As valleys developed floodplains, the streams flowing through them became less steep, and their pattern and shape progressively adjusted to assume new stable forms, in balance with the new landscape.

In many settings, the story is even more complicated. The main valleys were widened out by glacial scouring, while in many small pockets, soil materials melting out of glaciers created complex local deposits of clay, sand, gravel, cobble and boulders, and leaving diverse terrace forms throughout the valley. As the steeper streams coming off the mountainsides joined into a more gently sloped main channel running through the main valley, the stream became wider, and less deep.

The stable form that a stream takes in balance with the steep, mountain notches will be different from the one it takes in medium-gradient valleys, and this will be different still from the stable form in a relatively gently-sloping, broad floodplain like the West Branch of the Delaware.

As our climate warmed, grasses and then trees recolonized the evolving valley floor. As vegetation returned to the floodplains, the conditions that determine the balance between stream shape and the landscape changed once again. Stream banks that have a dense network of tree and shrub roots adding strength to the soil can better resist the erosive power of flood flows, and consequently a new stable stream form emerges; a new balance is struck between resistive and erosive forces. A dense mat of woody roots is essential if we want to maintain a stable stream bank. If streamside trees and shrubs are removed, we can expect the bank to soon begin eroding.

In the Catskills, a naturally stable stream will have trees and shrubs all along the stream bank to help hold the soil together. If you remove the trees and shrubs, and mow right down to the edge of the stream, you may be risking big-time erosion problems.

If we want to maintain healthy, stable streams, then, we need to maintain a stable stream *morphology* and vigorous streamside, or *riparian*, vegetation. Stable streams are less likely to experience bank erosion, water quality and habitat problems. The management plans being developed by the Stream Management Program and their partners generally describe the current condition of the stream form and streamside vegetation throughout the watersheds they address, and then make recommendations for protecting healthy sections of stream and for restoring the stability of those sections that are at risk.

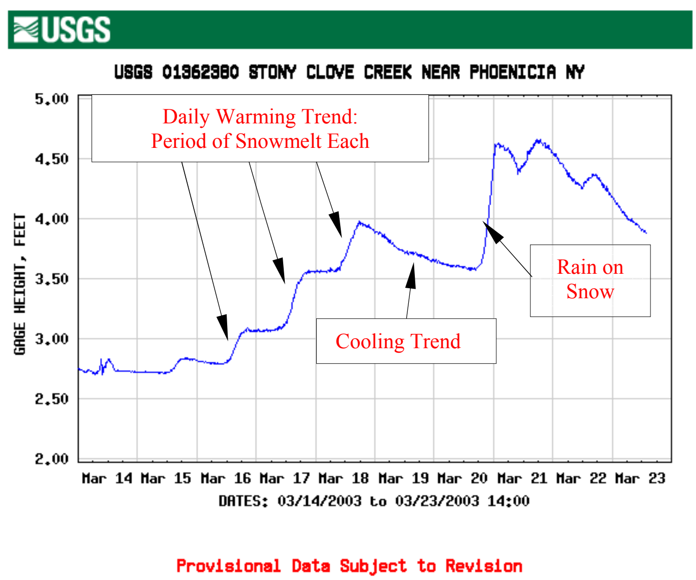
Stream Morphology and Classification

This section provides more technical information for the curious about the relationship between stream ***form*** (or ***morphology***) and physical stream ***function*** (i.e., flood behavior, sediment transport).

The last section described how a stream's form --slope and depth-- determine its function --how much potential force the stream has to move the silt, sand, gravel, cobble and boulders that make up its *bedload*. We focused on slope and depth because they are often changed --intentionally or unintentionally-- by stream managers. There are, however, many characteristics that come together to influence how a stream "makes itself", and whether it is stable or unstable in a given valley. These characteristics[2] include:

Stream flow (Q)

Usually represented as cubic feet or cubic meters per second, streamflow is also called stream ***discharge***. Stream flow changes from hour to hour, from day to day, from season to season, and from year to year.



The typical pattern of streamflow over the course of a year is called the streamflow regime. Some streamflows play a more significant role than others in determining the shape of the stream. In alluvial streams, the "*bankfull flow*" is considered most responsible for defining the stream form, and for this reason, bankfull flow is also sometimes called the *channel-forming flow*. This flow typically recurs every 1-2 years. It may seem surprising that very large floods aren't more important in forming the channel, but while they may induce catastrophic changes in a stream—severely eroding banks and

[2] Each characteristic is followed, in parentheses, by the variable used to represent it in formulae.

washing countless trees into the channel—these major floods are more rare, occurring on the average every decade or so. The flows that have the most effect on channel shape are those that come more frequently, but which are still powerful enough to mobilize the gravel and cobble on the streambed: the smaller, bankfull flows.

The height of the water in the channel is called the ***stage***. When a stream overtops its banks, it's in ***floodstage***. ***Bankfull stage*** –when the stream is just about to top its banks-- is used as a benchmark for measuring stream dimensions for classifying different stream types (see ***Rosgen Classification System***, below).

Slope (S)

We already mentioned slope as one of the two main determinants of a stream's potential force for erosion of the streambed and banks. The slope of a stream usually refers to the average slope of the water surface when the stream is running at bankfull flow.

Channel average depth (d)

Depth is the other primary determinant of potential force, and is measured from the streambed to the water's surface. Again, this will depend on the level of the streamflow. When used to compare one stream reach to another in ***stream classification systems*** (see below), the average depth of the stream during a bankfull flow is used.

Channel width (w)

Together with average depth, the ***width*** of the channel determines the ***cross-sectional area*** (Area = width x depth). If a roadway encroaches on a stream, its width is reduced. To pass the same sized flood, the stream is going to have to be deeper, that is, floodstage is increased, or move the water faster through it.

Channel roughness (n)

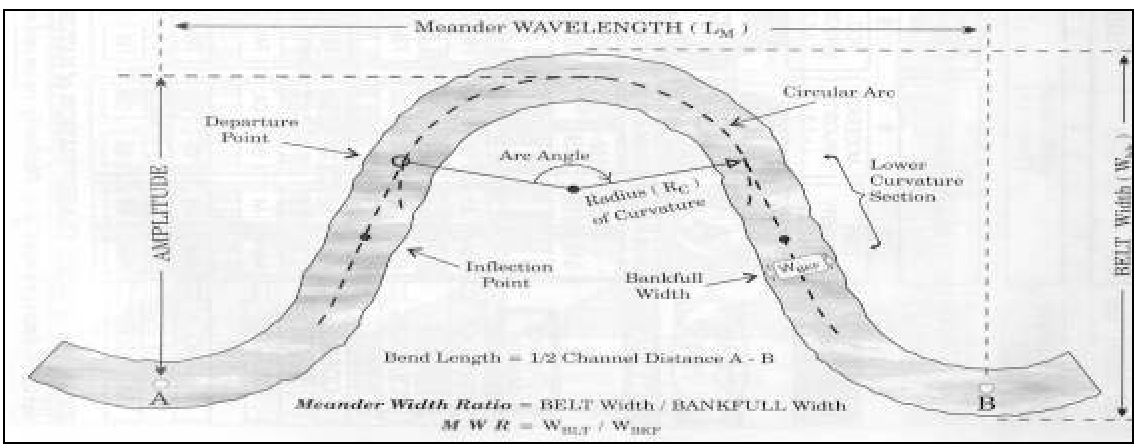
So far we've only talked about what gives the water its potential force to erode the streambed and banks. There are also characteristics of the stream that slow the water down, or resist the flow. One of these is the channel ***roughness***: it's harder for the stream to flow through a section of stream filled with boulders than through a stream with a silt-bottomed bed, and no obstructions. Water flows more slowly across a floodplain filled with trees and dense brush than it does across a smooth, newly mown lawn or parking lot, and so is less likely to cause erosion. Within streams, this is also sometimes referred to as ***bed roughness***.

Sinuosity (k)

A different kind of roughness that slows water down has to do with whether the channel runs straight, or curves. When the flow of a stream is slowed as it moves around a bend in the stream, we say that the flow is encountering ***form roughness***. The curviness of a stream is called its ***sinuosity***, and is measured as the stream length divided by the valley length. That is, if a stream runs completely straight down a mile long valley, both the valley and the stream are the same length, or 1 mile / 1 mile = a sinuosity of 1. If the stream snakes, or ***meanders***, down the same valley, it might be two miles long, or 2 miles / 1 mile = a sinuosity of 2. As a rule of thumb, we find that, in natural channels, the lower the slope, the more sinuous the stream.

Radius of curvature (Rc)

Radius of curvature is another measure the "curviness" of the stream, but at a single curve, and is measured as in this illustration:



Adapted from The Reference Reach Field Book, D. Rosgen.

Belt width

Meander Beltwidth describes the width of a stream's meander through its valley (see figure above). It is measured from the outside of one meander to the outside of the next, perpendicular to valley fall. This is also sometimes referred to as the floodway, and during large floods, the entire meander beltwidth is often inudated, as the stream takes a "shortcut" on its way downvalley. Homes and roads in this region are at greater risk for flooding and damage from erosion.

Sediment size (D50)

It takes more force for a stream to move the material on the bed of the stream if it is made up of large cobble than if it is sand or silt; the smaller the particles, the more easily they will be moved. To characterize a reach of stream, 100-300 particles are randomly selected and measured, and the median size particle gives the ***D50*** of the reach: meaning that 50% of the particles in the stream are smaller, and 50% are larger.

Name	Size
Silts	< 0.062mm
Sands	0.064mm - 2mm
Gravels	2mm - 64mm
Cobbles	64mm – 256 mm
Boulders	256mm – 512mm

Bed and Bank Cohesiveness

Due to the glacial history of the region, soils in the Catskills are extremely variable from place to place, and some soil types hold together better than others, or are more ***cohesive***. Some streambeds have their gravel and cobble locked together in a form that resists movement by streamflow, and others "unzip" easily. The roots of trees and shrubs can reach deep into the soil of a stream bank, and the web of fine root fibers can add a tremendous amount of cohesiveness to the soil.

The "balance" that streams develop over time when they aren't disturbed is the balance between the erosive forces of floodwaters, and the strength of the bed and banks to resist that erosive power. This balance develops because streams will erode away their banks until, eventually, the lengthening of their meanders reduces the slope, or the stream is widened and depth is decreased sufficiently, such that the cohesiveness of the soil and vegetation together just equal the erosive potential of the floodwaters. If the vegetation on the stream bank is changed, the soil cohesiveness will change, and that balance point will change. Likewise, if a stream bank gradually migrates into a less cohesive soil type, it can suddenly begin eroding very quickly.

Sediment discharge (Qs)

When silts and sands, gravels, cobbles or even boulders have been moved by the streamflow, we call them ***sediment***. ***Sediment discharge*** is the amount of sediment moving past a particular point over some interval of time, and is usually measured in tons per year. ***Bedload*** is sediment that moves along the bottom of the channel, while ***washload*** is sediment that is suspended up in the water. Measuring sediment discharge is one way to determine if a stream is stable or not. If the amount of sediment coming into a reach of stream doesn't roughly equal the amount leaving the reach in the same time period, the form of the reach will have to change.

Entrenchment

When a reach of stream is straightened or narrowed, the power of the streamflow is increased, and it may cut down into its bed, so that flood flows can't spill out into the floodplain. When this happens, we say that the reach has ***incised***, and that the channel has become ***entrenched***. Entrenchment can be low, moderate or extreme. When even large flood flows are confined to the narrow channel of the stream, they can become very deep, and therefore very erosive. The result may be that the stream gullies down even deeper into the bed. Eventually the banks may become too high and steep, and they may erode away on one or both sides, widening the channel. Eventually, the channel may widen enough to allow a new floodplain to develop inside the entrenched banks (see the figure below). This is one way that streams evolve over time.

Entrenchment may also occur as a result of building berms that prevent the stream from using its natural floodplain during large flows, or if the amount of water the stream is forced to carry increases significantly as a result of storm drainage associated with land development.

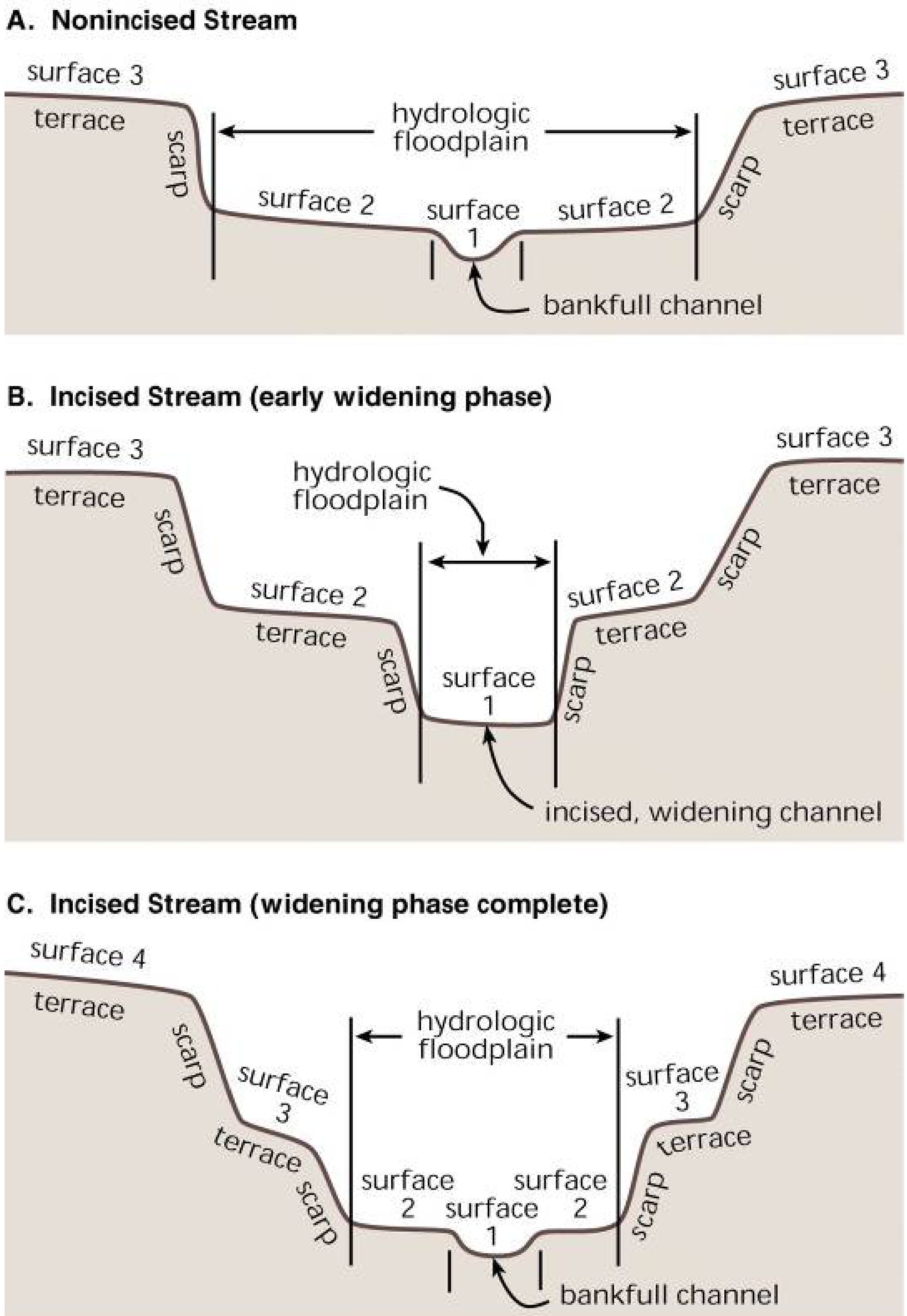
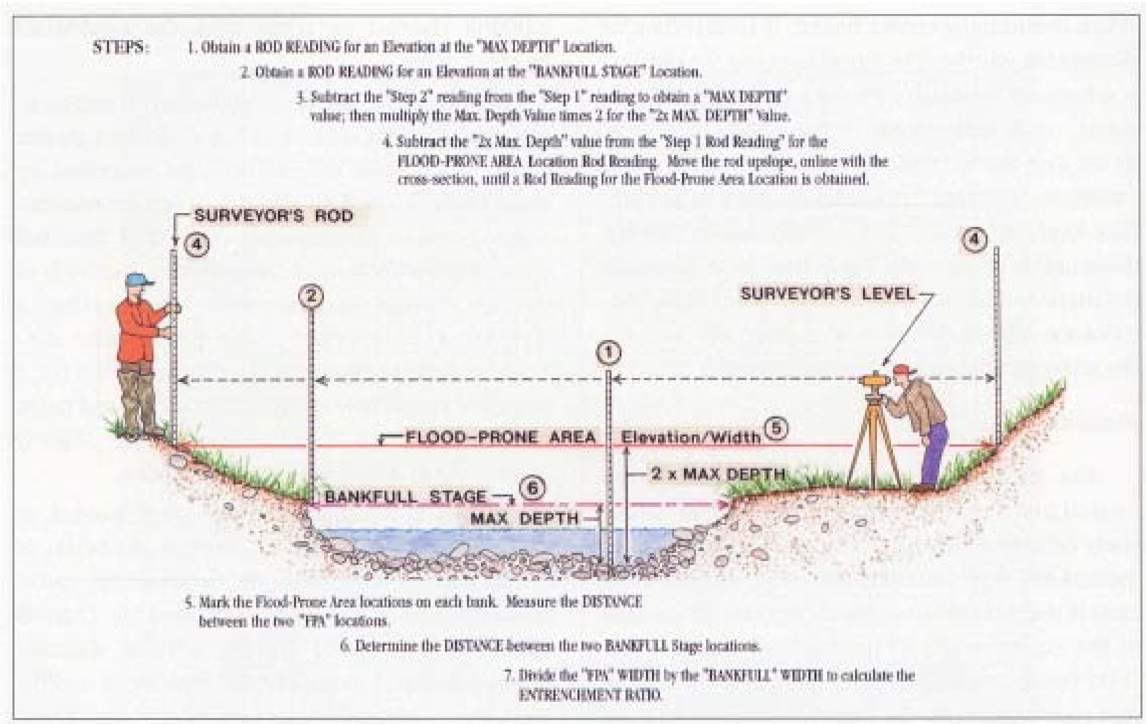


Fig. 1.24 -- Terraces in (A) nonincised and (B and C) incised streams. Terraces are abandoned floodplains, formed through the interplay of incising and floodplain widening.
In Stream Corridor Restoration: Principles, Processes, and Practices (10/98).
Interagency Stream Restoration Working Group (15 federal agencies)(FISRWG).

One method of measuring entrenchment was developed by hydrologist David Rosgen. His ***Entrenchment Ratio*** compares the stream's width at bankfull flow with its width at twice the maximum depth at bankfull flow:



D. Rosgen's measure of entrenchment from Rosgen 1996.

Applying the Science of Stream Form and Function to Stream Management

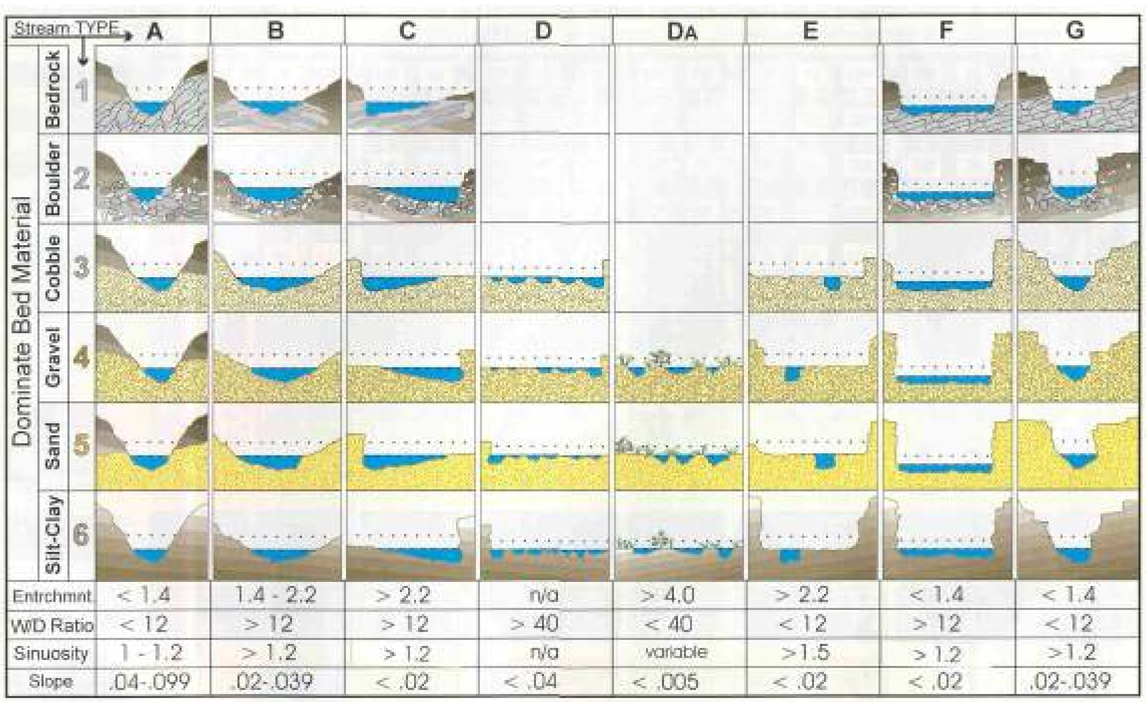
By carefully analyzing all these characteristics of stream form, stream managers can get a fairly good idea about the relative stability of a stream, reach by reach, over its whole length. By understanding the relationship between the stream's form and its function, managers can prioritize severely unstable stream reaches for treatment, and can apply different management strategies appropriately, and more cost effectively. Analysis of stream morphology can also make for more successful design of stream restoration projects; designers identify and survey stable stream reaches and then use their form characteristics as a design template for restoration projects.

Classifying Streams by their Form

One useful tool for stream managers, also developed by Dave Rosgen, is a system for classification of different stream reaches on the basis of their form. Rosgen's system gives letter and number designations to different stream types, depending on their combination of five characteristics:

1) Entrenchment ratio
2) Ratio of width to depth
3) Slope
4) Sinuosity
5) Bed material size (D_{50})

Different combinations of these characteristics result in a great number of different stream types, from A1 through G6 (see Figure XX; read letter designation across the top, number down the left side). These letter/number designations provide a sort of shorthand for summing up the form of a stream reach.



From Rosgen 1996.

So, for example, a B3 stream type has a cobble dominated bed, has a moderate amount of accessible floodplain, is more than 12 times as wide as it is deep, is moderately sinuous, and drops between 2 and 4 feet for every 100 feet of stream length. How does a B3 differ from an F3? An F3 is more entrenched, meaning that it can't spill out onto its floodplain during storm flows, and it's also less steep, dropping less than 2 feet for every 100 feet of stream length. How is a B3 different from a G4? Not only is the G4 more entrenched, like the F3, but also has a smaller width-to-depth ratio than a B3, and a finer, gravel-dominated bed.

As we have discussed above, each of these different forms functions a little differently from the next, especially with regard to the stream's ability to transport its sediment effectively. By classifying the different stream types in a watershed, then, different management strategies can be targeted to each section of stream. Rosgen (1994) has created a table (see Table 2), which suggests how the different stream forms can be interpreted with regard to a number of management issues.

Throughout this management plan you will find references to these stream types. It is important to emphasize that these are only very general management interpretations, and that the stream types are included as a convenient, "shorthand" summary of the morphology of a reach. To produce reasonably reliable conclusions about how a stream reach is likely to behave in the future, the actual surveyed conditions at each reach must also be considered in the context of the conditions found in adjoining reaches upstream and downstream, historical information taken from aerial photography and additional field studies of soils, vegetation and watershed land use.

Table 2. Stream forms and their associated management issues (Rosgen, 1994).

Stream type	Sensitivity to disturbance[a]	Recovery potential[b]	Sediment supply[c]	Streambank erosion potential	Vegetation controlling influence[d]
A1	very low	excellent	very low	very low	negligible
A2	very low	excellent	very low	very low	negligible
A3	very high	very poor	very high	very high	negligible
A4	extreme	very poor	very high	very high	negligible
A5	extreme	very poor	very high	very high	negligible
A6	high	poor	high	high	negligible
B1	very low	excellent	very low	very low	negligible
B2	very low	excellent	very low	very low	negligible
B3	low	excellent	low	low	moderate
B4	moderate	excellent	moderate	low	moderate
B5	moderate	excellent	moderate	moderate	moderate
B6	moderate	excellent	moderate	low	moderate
C1	low	very good	very low	low	moderate
C2	low	very good	low	low	moderate
C3	moderate	good	moderate	moderate	very high
C4	very high	good	high	very high	very high
C5	very high	fair	very high	very high	very high
C6	very high	good	high	high	very high
D3	very high	poor	very high	very high	moderate
D4	very high	poor	very high	very high	moderate
D5	very high	poor	very high	very high	moderate
D6	high	poor	high	high	moderate
Da4	moderate	good	very low	low	very high
DA5	moderate	good	low	low	very high
DA6	moderate	good	very low	very low	very high
E3	high	good	low	moderate	very high
E4	very high	good	moderate	high	very high
E5	very high	good	moderate	high	very high
E6	very high	good	low	moderate	very high
F1	low	fair	low	moderate	low
F2	low	fair	moderate	moderate	low
F3	moderate	poor	very high	very high	moderate
F4	extreme	poor	very high	very high	moderate
F5	very high	poor	very high	very high	moderate
F6	very high	fair	high	very high	moderate
G1	low	good	low	low	low
G2	moderate	fair	moderate	moderate	low
G3	very high	poor	very high	very high	high
G4	extreme	very poor	very high	very high	high
G5	extreme	very poor	very high	very high	high
G6	very high	poor	high	high	high

a Includes increases in streamflow magnitude and timing and/or sediment increases.
b Assumes natural recovery once cause of instability is corrected.
c Includes suspended and bedload from channel derived sources and/or from stream adjacent slopes.
d Vegetation that influences width/depth ratio-stability.

References

Rosgen, D.L. 1994. A classification of Natural Rivers. Catena 22: 169-199.

Rosgen, D.L. 1996. Applied River Morphology. Wildland Hydrology, Pagosa Springs, Colorado.

3.3 Watershed Assessment and Inventory

A watershed assessment protocol was prepared to support the development of the Manor Kill Stream Management Plan. The protocol was meant to provide the project team with a general baseline inventory of conditions throughout the stream corridor.

Schoharie County SWCD staff collecting data on an eroding bank in the Manor Kill, 2008.

Stream Feature Inventory

In the initial stages of a watershed assessment and planning effort, it is necessary to gain a basic familiarity with the stream corridor and surrounding watershed. An inventory of stream features can reveal trends important to understanding the stream system. The stream feature inventory protocol provided an inventory of the following features:

1) Conditions that affect hydraulic function, particularly sediment transport function such as bedrock sills and banks, cultural and natural grade controls, berms, and rip-rap or other revetment, and inadequate riparian vegetation;

2) Potential sources of water quality impairment in the corridor, especially eroding banks, clay exposures, road runoff outfalls, dumps sites, and exposed septic leach fields or other hazards;

3) Locations of bank erosion monitoring sites to be monumented and surveyed for study of bank erosion rates;

4) Infrastructure, including road crossings, bridge abutments, culverts and outfalls, and utility lines or poles;

5) Other features such as tributary confluences, water intakes, springs, wells, diversions, and invasive species.

This stream feature inventory was also used to help define and prioritize further assessment, and scope the issues to be addressed in the management plan. Most of the data presented in the Management Unit Descriptions in Section 4 was derived from the stream feature inventory walkover conducted during the summer of 2008. The Trimble GeoXH Global Positioning System (GPS) was used to map locations of features described above. Photographs and attribute data were also taken at each feature. Protocol used for attribute collection is detailed in Appendix E, Stream Management Data Dictionary Guide.

GCSWCD personnel assess a stretch of Schoharie Creek that includes many stream features examples.

Following collection, all data was integrated into a common geodatabase using the Stream Analyst ArcGIS extension. The geodatabase is the common data storage and management framework for ArcGIS. It supports all the different types of data that can be used by ArcGIS such as; attribute tables, geographic features, and survey measurements. Utilizing GPS coordinates, each feature was then linked to the management unit in which it was located creating an improved organizational structure and allowing for the reporting of stream feature statistics based on management unit. The first page of each of the Management Unit Descriptions in Section 4 presents the results of this data for each individual management unit. The following are summary statistics for the Manor Kill mainstem.

<u>2008 Manor Kill Stream Feature Statistics</u>

3.7 miles or 36% of streambanks had experienced erosion
2,217 feet or 4% of streambanks had been stabilized
1,385 feet or 2.6% of streambanks had been bermed
1,504 feet of clay exposures
480 acres of inadequate vegetation within 300 ft of the Manor Kill
4.5 miles of road within 300 ft of the Manor Kill
21 structures located in 100-year floodplain

Riparian Vegetation

Riparian vegetation mapping of a 300-foot stream corridor was conducted to identify the status of the vegetative community, and identify areas in need of enhancement. This protocol provided a characterization of the vegetative community (physiognomic) structure of riparian areas from remotely-sensed data. Characterizing riparian vegetation supported the assessment of the capacity of the riparian buffer to mitigate potentially deleterious water quality impacts from upland land uses. In addition, riparian classification defines the role of vegetation in the cohesiveness of stream bank soils and the integrity of the stream and riparian ecosystems. This analysis will lead to recommendations of where improvements to the riparian buffer may be most critical and/or effective, and locations of reference riparian vegetative communities within the watershed. The mapping also provided the area of impervious surfaces (e.g. roofs, driveways, roads) within the 300 foot buffer. The complete protocol is located in Appendix B. Planting recommendations, descriptions and maps of the existing riparian community are presented in each management unit (Section 4).

Japanese Knotweed Mapping

As part of stream feature inventory, locations of Japanese knotweed (*Fallopia japonica*) along the streambank were identified. This invasive species has become a widespread problem in recent years, shading out other species and not providing adequate root structure to stabilize the soil in streambanks. The results can include rapid streambank erosion and decreased community richness. Japanese knotweed occurrences are discussed in each management unit and locations are included on the riparian vegetation maps (Section 4).

Historical Channel Alignments

A series of historical stream channel alignments from 1959, 1967, 1980, 2001, and 2004 was used to determine the frequency and magnitude of historical channel avulsions. ArcGIS 9.2 was used to georeference aerial photographs, when necessary, and then used to digitize each stream channel alignment. The alignment from each flight series was compared to locate areas of historic instability. This characterization was also one criteria used to divide the stream corridor into management units. Historic stream channel alignments from

1959, 1980, and 2001, overlayed with the 2006 aerial photographs, can be viewed in each management unit under historic conditions.

Management Unit Delineation

To describe the current conditions and recommendations for the stream corridor, the 10.1 miles of the Manor Kill was divided into ten management units based on the following criteria:

1) Valley Slope - A profile of the valley slope was created using United States Geologic Survey contour data. This profile was divided into segments based on common slope characteristics.

2) Valley Confinement - The width of the 100-year floodplain was measured perpendicular to the valley fall line at each of the cross-sections along the mainstream, and the ratio of the width to bankfull and floodprone width at each was determined. A graph of these ratios was generated and analyzed to identify segments exhibiting common valley confinement characteristics.

3) Historical Channel Alignment - Stream alignments were created from 1959, 1967, 1980, 2001 and 2004 aerial photographs (as described above). These alignments were overlaid to determine segments of historical stream instability.

4) Vertical and Lateral Controls - Bedrock channels and banks, revetments, bridges and berm locations were documented in the 2006 GPS walkover. Frequency of occurrence of these controls influenced management segment breaks.

The resulting 10 management units are described in Section 4 and depicted in Figure 4.0.1. The data were then compiled by management unit to facilitate interpretation of conditions, trends and to make recommendations.

Bank Erosion Monitoring Sites (BEMS)

Using data collected from the 2008 stream feature inventory, four bank erosion monitoring sites were chosen along the Manor Kill. These banks may be monumented, and have cross-section and longitudinal profile surveys conducted for the purpose of long-term monitoring. To determine the distribution of bed material at each cross-section a pebble count in accordance with the Modified Wolman Pebble Counts Procedure should also be performed.

Bank erosion sites should also be evaluated using Rosgen's Bank Erodibility Hazard Index (BEHI) (Figure 3.3.1). BEHI is a means of measuring the potential for significant bank erosion at specific locations. The BEHI method evaluates bank erosion potential by measuring seven criteria; bank height versus the *bankfull stage*, ratio of riparian vegetation rooting depth to stream bank height, bank angle, percentage of root density, composition of stream bank materials, soil stratification, and bank surface protection afforded by debris and vegetation. As the ratio of bank height to bankfull depth increases, the potential for bank erosion increases. Steep bank angle, low root density, high soil stratification and homogeneous particle distribution contribute to a higher potential for bank erosion. Values of these seven criteria are calculated and each assigned an index number, which are totaled to determine bank erosion potential.

BANK EROSION HAZARD INDEX

SITE: Stony Clove BEHI# DATE:
DATA COLLECTED BY: LOCATION:

Notes:

CRITERIA		VERY LOW		LOW		MODERATE		HIGH		VERY HIGH		EXTREME	
		VALUE	INDEX	VALUE	INDEX	VALUE	INDEX	VALUE	INDEX	VALUE	INDEX	VALUE	INDEX
BANK HT/ BKF HT		1.0 - 1.1	1.0 - 1.9	1.1 - 1.19	2.0 - 3.9	1.2 - 1.5	4.0 - 5.9	1.6 - 2.0	6.0 - 7.9	2.1 - 2.8	8.0 - 9.0	> 2.8	10
ROOT DEPTH / BANK HEIGHT		1.0 - 0.9	1.0 - 1.9	0.89 - 0.50	2.0 - 3.9	0.49 - 0.30	4.0 - 5.9	0.29 - 0.15	6.0 - 7.9	0.14 - 0.05	8.0 - 9.0	< 0.05	10
ROOT DENSITY (%)		100 - 80	1.0 - 1.9	79 - 55	2.0 - 3.9	54 - 30	4.0 - 5.9	29 - 15	6.0 - 7.9	14 - 5	8.0 - 9.0	< 5	10
BANK ANGLE (DEGREES)		0 - 20	1.0 - 1.9	21 - 60	2.0 - 3.9	61 - 80	4.0 - 5.9	81 - 90	6.0 - 7.9	91 - 119	8.0 - 9.0	> 119	10
SURFACE PROTECTION (%)		100 - 80	1.0 - 1.9	79 - 55	2.0 - 3.9	54 - 30	4.0 - 5.9	29 - 15	6.0 - 7.9	15 - 10	8.0 - 9.0	< 10	10
TOTALS													
NUMERICAL ADJUSTMENTS	None												
TOTAL ADJUSTED SCORE													**0.0**

BANK MATERIALS:
BEDROCK: BANK EROSION POTENTIAL ALWAYS VERY LOW
BOULDERS: BANK EROSION POTENTIAL ALWAYS LOW
COBBLE: DECREASE BY ONE CATEGORY UNLESS MIXTURE OF GRAVEL/SAND IS OVER 50%
GRAVEL: ADJUST VALUES UP BY 5 - 10 POINTS DEPENDING ON COMPOSITION OF SAND
SAND: ADJUST VALUES UP BY 10 POINTS
SILT/CLAY: NO ADJUSTMENT

STRATIFICATION: 5 - 10 POINTS (UPWARD) DEPENDING ON POSITION OF UNSTABLE LAYERS IN RELATION TO BANKFULL STAGE

BANK EROSION POTENTIAL

Very Low	5-9.25
Low	10-19.5
Moderate	20-29.5
High	30-39.5
Very High	40-45
Extreme	46-50

Figure 3.3.1. Sample BEHI data collection sheet from Rosgen, Dave "Applied River Morphology." Wildland Hydrology Books, Pagosa Springs, Colorado, Table 6-8, pg. 6-41, 1996 revised 2001