PA DEP FINAL REPORT

BIG SPRING RUN NATURAL FLOODPLAIN, STREAM, AND RIPARIAN WETLAND - AQUATIC RESOURCE RESTORATION PROJECT MONITORING

2013

Project Investigators:

Dr. Robert Walter, Franklin & Marshall College
Dr. Dorothy Merritts, Franklin & Marshall College
Michael Rahnis, Franklin & Marshall College
Dr. Michael Langland, US Geological Survey
Dr. Daniel Galeone, US Geological Survey
Dr. Allen Gellis, US Geological Survey
Dr. William Hilgartner, The Johns Hopkins University; The Friends School, Baltimore
Dr. David Bowne, Elizabethtown College
Dr. John Wallace, Millersville University
Dr. Paul Mayer, US Environmental Protection Agency
Dr. Ken Forshay, US Environmental Protection Agency

Deliverables:

Deliverable 1: Quantify volumes of historic (legacy) sediment in stream corridor and associated nutrients in sediments. (Merritts/Walter/Rahnis/F&M Staff)

Deliverable 2: Monitor surface water and shallow ground water and quantify sediment and nutrient loads. (Galeone/Langland/Walter)

Deliverable 3: Identify sources of sediment (upland vs. stream corridor) in stream water via geochemical fingerprinting. (Gellis/Walter/Rahnis)

Deliverable 4: Quantify rates of stream bank erosion, stream corridor deposition, sediment storage in the stream corridor, and soil erosion from uplands (sediment budget) (Gellis/Merritts/Rahnis/F&M Staff)

Deliverable 5: Biological indicators of ecosystem services (Hilgartner/ Bowne/ Wallace)

OUTLINE

Executive Summary

A. Synopsis of the Big Spring Run Aquatic Ecosystem Restoration Project
   - A. 1. Location and Background
   - A. 2. Motivation for Floodplain, Stream, and Riparian Wetland Aquatic Resource Restoration
   - A. 3. Restoration Target: Pre-settlement Anastomosing Channels and Emergent Wetlands
   - A. 4. Restoration Summary
   - A. 5. Design Criteria and Goals
   - A. 6. Description of Restoration Construction Process

B. Deliverable 1: Quantify volumes of historic (legacy) sediment in stream corridor and associated nutrients in sediments. (Merritts/Walter/Rahnis/F&M Staff)
   - B. 1. Volume of historic (legacy) sediment in stream corridor
   - B. 2. Amount of nutrients in historic (legacy) sediment
     - a. Phosphorous
     - b. Nitrogen

C. Deliverable 2: Monitor surface water and shallow ground water and quantify sediment and nutrient loads. (Galeone/Langland/Walter/Merritts)
   - C. 1. Surface water sediment loads
   - C. 2. Nutrients in surface water and groundwater

D. Deliverable 3: Identify sources of sediment (upland vs. stream corridor) in stream water via geochemical fingerprinting. (Gellis/Walter/Rahnis)
   - D. 1. Sediment sources from fingerprinting

E. Deliverable 4: Quantify rates of stream bank erosion, stream corridor deposition, sediment storage in the stream corridor, and soil erosion from uplands (sediment budget) (Gellis/Merritts/Rahnis/F&M Staff)
   - E. 1. Rates of stream bank erosion pre-restoration
   - E. 2. Analysis of bank erosion and its contribution to suspended sediment load
   - E. 3. Post-restoration topographic surveys
   - E. 4. Analysis of net loss of fine sediment from bank erosion
   - E. 5. Sediment budget

F. Deliverable 5: Biological indicators of ecosystem services (Hilgartner/ /Bowne/Wallace)
   - F. 1. Vegetation assessment—Hilgartner
   - F. 2. Herpetological assessment--Bowne
   - F. 3. Macroinvertebrate assessment—Wallace

Appendices

Appendix 1. Project design and construction details
Appendix 2. Survey methods
Appendix 3. Analytical methods (sediment fingerprinting)
Appendix 4. Analytical methods (nutrient analysis)
Appendix 5. Data
- 5a. USGS gage data
- 5b. Survey data
- 5c. Trace-element geochemistry (fingerprinting)
- 5d. Vegetation data
- 5e. Macroinvertebrate data

Appendix 6: Annotated list of videos of the Big Spring Run restoration and monitoring
Appendix 7: Pre-Restoration and Post-Restoration Photos of Reach with Cross Sections 6 and 7 to Illustrate Erosion and Deposition in Stream Corridor with Time (contained in Figure File)
Appendix 8: References Cited

Executive Summary

Deliverable 1 – Quantify Legacy Sediment Volumes and Nutrients

- The depth of excavation during restoration was based on reconstruction of the pre-settlement ground surface elevation in the valley bottom. This ground surface, marked by a distinctive dark A-horizon of a paleo-soil, was buried beneath historic (i.e., legacy) sediment. The buried soil and legacy sediment were exposed along incised stream banks prior to restoration.
- We estimate that 21,704 cubic yards of legacy sediment was removed from the restoration reach, which equates to a mass of ~21,955 tons when volume is multiplied by the average bulk density of legacy sediment throughout the restoration reach (bulk density = 1.2 g/cm^3 = 75 lb/ft^3). This estimate of volume removed is the difference between (1) the ground surface obtained from pre-restoration (2008) airborne lidar, and (2) the ground surface obtained from a post-restoration (2011) as-built survey done with a total geodetic station.
- The average concentration of total phosphorus (TP) in Big Spring Run (BSR) legacy sediment is 1161 ppm, which is equivalent to 2.3 lbs-P/ton of sediment. Therefore, the mass of total phosphorus removed with the sediment during restoration is ~50,498 lbs.
- The average content of sorbed P (TPi) in BSR legacy sediment is 811 ppm, or 1.6 lbs/ton, which equals ~35,128 lbs of sorbed P removed during restoration. Sorbed P is attached to particle surfaces and can be released to surface waters under reducing redox conditions.
- The average amounts of labile (bioavailable) and water extractable P in BSR legacy sediment are 16.2 and 6.0 ppm, respectively, which equate to 0.032 and 0.012 lbs/ton, respectively. Thus, 703 lbs of labile P and 263 lbs of WEP were removed from the valley bottom ecosystem during restoration.
- The average concentration of total nitrogen (TN) in BSR legacy sediment is 1441 ppm, or 2.9 lb-N/ton of sediment. This equates to ~63,669 lbs of TN removed from the BSR valley bottom ecosystem during restoration.
- The average amount of bioavailable nitrate-N in BSR legacy sediment is 6.1 ppm, which is 0.012 lbs-NO_3/ton of sediment. Thus, ~263 pounds of soluble NO_3-N was permanently removed from the valley bottom ecosystem as a result of the restoration.

Deliverable 2 – Quantifying Sediment and Nutrient Loads in Surface and Groundwater

- This deliverable is a partnership between: (1) the US Geological Survey (USGS), which installed and maintained three gage stations at BSR and which installed 18 valley bottom piezometers (via this grant and other grants to F&M); (2) US Environmental Protection Agency (US EPA), which installed 29 shallow ground water wells in the uplands surrounding the BSR restoration reach, and which

analyzed all but the initial SW and GW samples for this project; and (3) F&M, whose role was to assist USGS and EPA scientists when necessary.

- Despite acquiring extensive surface water (SW) and groundwater (GW) data during the pre-restoration phase of the project (2008-2011), due to delays in implementing the restoration project that went beyond the sampling timeline, no GW samples and few SW samples were collected during the post-restoration phase (2012 to present).
- As a result of the delays that were beyond the control of the monitoring project, we are not yet able to estimate nutrient load reductions and efficiencies. However, the data collected during this project provides a baseline of nutrient loads that will be used for future estimates of the load reductions that resulted from the restoration efforts.
- New funding is available and monitoring is underway for 2014-2016 that is intended to fill this data gap.
- We expect to have pre- to post-restoration nutrient load reductions by the end of 2016.

Deliverable 3 – Sources of Sediment and Sediment Fingerprinting

- The average contribution of sediment supplied to Big Spring Run from bank erosion prior to restoration can be deduced using mass balance calculations of ^{137}Cs concentrations in stream bank sediments (Walter et al., 2006). These calculations show that 30-65% of the sediment supplied to the upper BSR watershed prior to restoration can be attributed to bank erosion. These values reflect a minimum estimate of the contributions from bank erosion, as the stream banks themselves contain an appreciable amount of ^{137}Cs in the upper ca. 30 cm of the stream bank section.
- We conducted a methodological test of our trace element sediment fingerprint procedures by analyzing samples from the Mill Stream Branch (MSB) in the Maryland Coastal Plain that had been examined previously and independently by an analytical laboratory at the USGS. Our investigation on the same samples that were analyzed by the USGS yielded identical results to those presented in Banks et al 2005 and Massoudieh et al 2012. These results indicate that stream banks contributed ca. 100% of the suspended sediment load to MSB, even though the USGS and F&M labs used different sample preparation methods and analytical methods. This methods test adds a high level of confidence and quality assurance to the trace element fingerprinting methods employed by our group at BSR.
- Applying the same analytical procedure and mixing model calculations that we used to demonstrate the similarity between USGS and F&M results for the Mill Stream Branch sediment fingerprint study, we find that 60-70% (average of 63%) of the suspended sediment load sampled at the downstream gage at BSR (located just below the restoration reach), was derived from stream bank erosion.
- Conducting the same mixing model calculations for the fluvial sediments collected at the East Branch (Sweeney) and West Branch (Fry) gages at BSR, we estimate that

stream banks contributed up to 33% of the East Branch (Sweeney) gage sediments and up to 54% of the West Branch (Fry) gage sediments.
- We observe a 63% stream bank contribution (i.e., from bank erosion) at the downstream Keener gage, which indicates from mass balance calculations that bank erosion between the upstream and downstream gages at BSR must be ca. 80-90%.

Deliverable 4 – Quantify Bank Erosion, Sediment Storage and Upland Erosion

- After restoration, the sediment flux out of the restoration reach is much less than prior to restoration (decreased from 3-year pre-restoration average of 218 tons per year to 1-yr post-restoration value of 109 tons per year). More years of monitoring will be necessary to identify trends with time and to calculate long-term averages for comparison with longer-term averages of pre-restoration data. Data for October 2012 September 2013, the 2nd year of post-restoration monitoring, will be available in 2014.
- USGS gage data indicate that an average of 94 tons/yr of fine suspended sediment was contributed from bank erosion within the restoration reach for 3 years prior to restoration (average annual value for 2008-09, 2009-10, and 2010-11).
- Our repeat topographic surveying and grain size analysis show that ~91 tons/year was eroded from stream banks that consisted of nearly 100% fine sediment (average annual value of survey results from 2004-2011).
- Based on data from the gage stations for suspended sediment load (with a measured mean grain size of 16 microns), there has been a substantial increase in the # of days when the amount of suspended sediment entering the restoration reach is greater than that measured at the downstream terminus of the restoration reach since restoration in 2011. Hence, not only must erosion within the restoration reach be small (if not 0), but also some deposition must be occurring within the restoration reach.
- The amount of deposition of fine sediment (clay, silt, and fine sand) within the restoration reach is small, and is estimated to have been approximately 15 tons during the first year after restoration. This amount is insignificant in comparison to the amount of deposition of legacy sediment that buried the original wetland and was removed during restoration, which was ~23,000 tons.
- We conclude from gage data and surveying that bank erosion in the restoration reach no longer is a source of fine sediment, hence explaining the reduction from 218 to 109 tons of suspended sediment per year measured at the downstream USGS gage station. Both the USGS gage data and long-term monitoring of surveyed cross sections independently support this conclusion.
- Prior to restoration the percentage of days in a year during which deposition occurred ranged from as low as 2% to as high as 25%. After restoration, deposition occurred during 49% of days.
- The % of days in which there was net loss of sediment from erosion in the restoration reach (i.e., suspended sediment load measured at downstream gage divided by the sum of sediment loads measured at both upstream gages is > 1)

decreased from a range of 75-98% from 2009-2011 (pre-restoration) to 51% in 2012, the first post-restoration year.

- In sum, the source of the load of sediment from bank erosion within the restoration reach no longer exists, and since restoration the restored reach now traps and stores some fine sediment. The result of these two factors is that the net loss of sediment is substantially reduced by removal of legacy sediment. Analysis of data from subsequent water years will be useful to further evaluate this trend.
- The increase in fine sediment trapping may have begun after the plant community was established during the early part of the 1st post-restoration growing season in May, 2012.

Deliverable 5 – Biological Indicators of Ecosystem Services

- The shift in vegetation characteristics between pre- and post- restoration time periods was definitive from a plant community dominated by facultative upland, non-native species to a plant community dominated by obligate/facultative wetland, primarily native wetland plants.
- The immediate reduction in relative percent cover of reed canary grass, an invasive species, is a positive outcome and indicates the initial success in reducing undesirable plant species in the riparian zone.
- The restoration target palustrine emergent vegetation was successfully established during the first growing season after construction.
- The current trajectory of the developing palustrine emergent marsh plant community, given its early establishment and stability under a variety of hydrologic conditions since restoration, is expected to continue.
- Given the major ecosystem disruption during construction, the anticipated decline of amphibian populations was confirmed during the first six months after sediment excavation and channel reforming.
- Preferred habitat for northern red salamander has increased over pre-restoration conditions and is expected to result in long term increases in these populations as the site and aquatic ecosystems continue to mature.
- The unrestored section of BSR may have provided refugia for amphibians during construction.
- As the restored aquatic ecosystems continue to mature, we expect a shift in the amphibian community from primarily streamside species to those more common in palustrine emergent wetlands.
- The current results from the macroinvertebrate study include data from only one sample date eight months post construction, and it is likely that at least several years will be required to fully assess macroinvertebrate responses to restoration.
- Both diversity and densities of macroinvertebrates in the restoration reach declined in the first year after construction as is commonly observed following restoration activities.
- A positive macroinvertebrate community response, from an ecological perspective, might be achieved if the goal of the restoration is for multi-functional anastomosing channel and palustrine emergent wetland ecosystems.

A. Synopsis of the Big Spring Run Aquatic Ecosystem Restoration Project

A. 1. Location and Background

Location: The Big Spring Run (BSR) Watershed is a sub-basin of the Conestoga River Watershed in Lancaster County, PA (Figure A1). The Conestoga River sub-basin is part of the Lower Susquehanna River flowing to the Chesapeake Bay (CB). BSR is typical of many headwater watersheds in the Piedmont Physiographic Province with low valley slopes (~0.005) and relief (~30 m). The focus of this study is a restoration site along 1st- and 2nd-order sections of BSR (drainage area 15 km^2, Figure A2).

The restoration site is located about 1.7 km upstream of a ~2.5-m high milldam that is breached. The breached milldam is near the confluence of BSR with Mill Creek, the latter of which flows directly into the Conestoga River. The milldam is identified on the 1864 Atlas of Lancaster County (Bridgens, 1864) and was used to operate a machine shop. A local farmer (H. Keener 2008, personal communication) who knew the miller reported that the mill operated on waterpower until the early 1900s.

Today, stone remnants of the structure are found at the location where the dam once spanned the entire valley. Fine-grained, laminated sediment, typically referred to as legacy sediment, is stacked to the level of the top of the milldam on the upstream side. The milldam appears breached on early historic aerial photographs (late 1930s, 1940, 1957) and a 2005 digital orthoimage. These images also reveal that the stream was incised into legacy sediment within the restoration reach during this time period. Other structures and channelization/relocation activities along Big Spring Run during the 1800s and early 1900s led to constrictions and slope changes that caused additional sedimentation upstream of the reaches where the structures were built (Figure A3).

After centuries of legacy sediment deposition along BSR, grade-control structures that breached sometime in the early 20th c. led to deep channel incision. The channel incision, in turn, led to lateral channel migration and bank erosion. Channel incision in the restoration reach was possibly augmented by channel relocation and straightening just downstream of the restoration area (see Figure A3).

Channel incision, often to bedrock and groundwater levels, created flow conditions that remobilized Pleistocene gravel from beneath buried Holocene wetland soils. This gravel can be seen in Figure A4a as an exhumed Pleistocene surface (see below) that was exposed as the channel bank retreated during bank erosion. Within the restoration reach, clasts up to ~5-cm in diameter were mobilized from this underlying gravel during high-flow events and were associated with gravel bar formation along the incised stream corridor (Ahamed, 2012). The heights of these gravel bars are shown in Figure A3.

From historic photos and landowner accounts, we estimate that channel incision into legacy sediment began at the restoration site sometime between the early 1900s and 1930s. Historic photos taken by landowners show small dams and ponds in the uppermost part of the restoration site as recently as ~1930 (photos owned by R. Houser). Multiple

structures that include culverts and small (<3-ft) dams were built along the incised channel in the 20th century. Culverts with dirt road crossings sometimes caused localized backwater effects even after channel incision, keeping the bed elevation higher than the original (i.e., pre-European settlement) valley bottom surface at some locations (Figure A3).

Background: Because of nearly 8 years of pre-existing scientific research and hydrologic (surface and groundwater) monitoring data by the United States Geological Services (USGS), the BSR site was selected by PA Department of Environmental Protection (PADEP) in 2007 to evaluate a new approach to aquatic resources restoration. The *Natural Floodplain, Stream, and Riparian Wetland Restoration Best Management Practice (NFSRWR-BMP)* proposed by PADEP was included in the recently developed CB Watershed Implementation Plan, part of EPA's Total Maximum Daily Load (TMDL) assessment for the CB. The previous USGS study was a 7-yr paired-watershed investigation at BSR from 1993-2001 (Galeone et al, 2006). This USGS study documented stream flow, nutrient and sediment loads from several gaging stations, 17 piezometers, and 2 wells in both "treated" and control basins over this time. The current restoration experiment at BSR is located in the same basin that was used as the "control" basin in the USGS paired watershed study. Information from the previous study provides a valuable addition to our pre-monitoring study.

A. 2. Motivation for Floodplain, Stream, and Riparian Wetland Aquatic Resource Restoration

Stream bank erosion, in general, was not widely viewed as a major source or delivery mechanism for either sediment or nutrients to streams in the Chesapeake Bay watershed until recently (c.f., Walter and Merritts, 2008). Most watershed models still consider upland hill slope erosion as the dominant source of suspended sediment and particulate-bound P (c.f., Boomer et al., 2013). Although Pionke et al. (2000) found that 98% of the algal available P in an agricultural watershed in Pennsylvania came from just 6% of the watershed that was nearest a stream, they did not consider bank erosion as a contributor. Even when bank erosion is recognized, it is "balanced" against storage, implying that there is no or little net contribution from bank erosion (c.f., Smith et al., 2011). Such reasoning implies that the eroded upland soil that enters a stream can somehow preferentially avoid being stored.

This view is paradoxical because many, if not most, streams in the Chesapeake Bay watershed are characterized by highly erosive vertical cut-banks that expose many feet of fine sediment. These banks have the potential to deliver substantial loads of fine sediments and sediment-derived nutrients directly to streams (Wolman, 1958; Merritts et al, 2011, 2013). A common misconception, however, is that substantial bank erosion is a "natural" fluvial process associated with self-formed meandering stream systems. As such, its contribution to contemporary nutrient and suspended sediment loads is considered to be negligible because bank erosion is balanced by storage in the channel, or it is thought that bank erosion can be minimized by restoration strategies that "protect" banks from eroding (c.f., Smith et al., 2011).

Our previous work (Walter and Merritts, 2008, Merritts et al., 2011, and Merritts et al, 2013) demonstrates that bank erosion in many parts of the Chesapeake Bay watershed is not a "natural" fluvial process, but rather it is largely an artifact of centuries of human manipulation of regional valley bottoms for waterpower. Immense amounts of historic sediment are stored in valley bottoms as a consequence of these human activities. As a result, bank erosion contributes substantial sediment and nutrient loads that have negative impacts on downstream ecosystems. Based on these insights and the work of other researchers (Banks et al., 2010; Mukundan et al., 2010; Devereux et al., 2011; Massoudieh et al., 2012; Voli et al., 2013), views on the relative importance of bank erosion of legacy sediments to suspended sediment loads are beginning to change (c.f., Boomer et al., 2013).

Earlier researchers observed and described post-European settlement valley-bottom sedimentation (e.g., Bennett, 1931; Happ, 1945, Overstreet et al., 1968). Costa (1975), for example, documented that stream bank sediments in the Maryland Piedmont contain substantial amounts of transported agricultural soil eroded from uplands during the peak agricultural period of the 18th and 19th Centuries, which is described as ca. 1 to 1.5 m thick massive, red-brown loams. Costa (1975) and previous authors attribute the deposition of these silt loam sediments to vast amounts of eroded soil overwhelming natural stream systems: "The slow rates of channel migration of Piedmont streams (Wolman and Leopold, 1957; Overstreet and others, 1968) imply that the agricultural sediment was spread over the surface of flood plains by overbank deposition, because streams would not have had sufficient time to rework valley fills since the initiation of large-scale agricultural land use."

This scenario is untenable, however, because it is a physically unlikely that fine-grained sediment would have spread over 100- to 300-foot-wide valley bottom surfaces (the typical valley bottom widths) while accumulating the thicknesses (3 to 20 feet) that are observed in stream banks throughout the mid-Atlantic Piedmont, without acquiring any of the sedimentary structures associated with fluvial deposition. For example, within typical legacy sediment deposits there are no point bars, no fining upward sequences, no lateral accretion surfaces, and no cross bed sets that are essential indicators of deposition by moving water. Instead, we observe stream banks composed of massive, horizontally layered and thinly bedded silts and clays, which are hallmarks of deltaic deposition in slackwater reservoirs.

To the best of our knowledge, no one has successfully modeled or shown mechanistically how a meandering stream can build a stack of massive loamy sediments above a gravel bed in the manner described by Leopold and Wolman, 1957; Costa, 1975; or Jacobson and Coleman, 1987. However, it is a straightforward process to model and demonstrate how meandering cut banks can be generated from base-level lowering resulting from dam breaching and incision into a sediment-filled reservoir (Cantelli, 2004). Occum's Razor requires us to accept the simplest solution (i.e., the profound and documented widespread influence of base level change, after Merritts et al., 2011 and 2013) until more complex hypotheses (i.e., thick floodplain deposition of massive silt loam without a mechanism associated with a grade control structure, as articulated by Costa, 1975) can be successfully demonstrated.

Our studies document that stream banks in most 1st to 3rd order stream valleys in the Mid-Atlantic Piedmont were formed within the last few centuries from the following sequence of events: (1) the construction of tens of thousands of milldams for water-power (base level rise); (2) complete infilling of millpond reservoirs during the intense agricultural period of the 18th and 19th Centuries; and (3) the subsequent breaching of milldams (base level fall) in the 20th Century (Walter and Merritts, 2008, Merritts et al., 2011). Widespread evidence for millpond sedimentation is irrefutable (c.f., Walter and Merritts, 2008, in Pennsylvania and Maryland; Pizzuto and O'Neal 2010, in Virginia; Wegman et al., 2013, in North Carolina; and Strouse, 2013, in New England).

Of significance to these findings is the compelling evidence of widespread millpond sedimentation from airborne lidar data (Walter and Merritts, 2008, Merritts et al, 2011). High-resolution topographic data from lidar reveals clearly that valley bottom sediment is graded to the crests of dams that span valley bottoms, as it typical of modern reservoir sedimentation. One difference between older milldams and more modern structures is that the older dams rarely had a spillway. Instead, a race delivered water from one end of the dam to a mill, and water spilled over the remainder of the dam. Some of the longer dams (up to hundreds of meters in length across valleys) had a higher part in the middle and water spilled over a lower crest on the side of the valley opposite the race. When these older dams breach, failure is typically at one of the two endpoints. As a consequence, modern incised streams are commonly on one side of the valley or the other at the locations of the thickest parts of the wedges of historic sediment. Lidar data and historic maps reveal that the upstream parts of these reservoirs of sediment pinch out, and this is where the next milldam is located. The result is a long series of sediment-filled wedges along the lengths of streams, each wedge indicating a former millpond and millseat that supplied power for a host of purposes to the local area.

Millpond sediments might not be everywhere on small order Mid-Atlantic streams, but so far we have found only a few local examples where they do not exist. One such area occurs near the headwaters of Marsh Creek in Chester County. Here, at a site called Great Marsh (Martin, 1958), we observe broad, flat palustrine emergent wetlands (Cowardin, 1979). The palustrine emergent wetlands in the Great Marsh include an abundance of wet meadow wetlands often dominated by tussock sedges. Marsh Creek flows through the palustrine emergent wetlands; drainage ditches dug by farmers over the past few centuries, and more recently the NRCS, augment its flow (J. Moore, personal communication). Radiocarbon dates and pollen analyses of the wetland soil, by Martin (1958) and us (Grand Pre et al., 2012), reveal that this palustrine emergent wetland formed ca. 10,000 years ago in response to warming after the Last Glacial Maximum. The palustrine emergent marsh has continued to function as a resilient wetland ecosystem up to the present day without changing wetland types or successional stages to palustrine forested wetlands.

It is important to note that the valley bottom palustrine emergent marsh at Great Marsh was never covered by more than a few inches of agricultural sediment, despite undergoing the same land use history and intense agricultural erosion as the rest of the surrounding

Piedmont region of Pennsylvania, and despite the fact that Marsh Creek has flowed through it for centuries. In fact, we observe soil from upland soil erosion stacked at the edge of the marsh and the base of the adjacent hill slopes, indicating that there was no mechanism to bring the sediments into or spread onto or over the valley bottom aquatic ecosystems. The reason that Great Marsh never accumulated legacy sediment is that the valley bottom immediately downstream of Great Marsh was never dammed for milling.

We assert that this is incontrovertible proof that damming and reservoir filling are required to create broad valley flats filled with massive, fine-grained legacy sediment. If this were not the case, if -- for example -- legacy sediments were deposited entirely by overbank fluvial processes, then Great Marsh should have stacks of legacy sediments deposited in the manner suggested by Costa (1975), or Jacobson and Coleman (1986), and this sediment should contain evidence of fluvial sedimentary structures. The fact that it does not, combined with the observation that there is no geological or historical evidence for a milldam, leads us to consider this is as *prima facie* evidence in support of base-level control for stream bank formation, based on pervasive (but not ubiquitous) construction of milldams (Walter and Merritts, 2008; Merritts et al., 2011).

Again, Occum's Razor dictates the simplest interpretation that modern cut-banks formed by incision (via post-dam breach base level fall) into stored millpond sediment. Furthermore, the cut banks are not incising into sediment deposited on floodplains by fluvial processes. It is worth emphasizing that we do observe some sedimentary structures indicative of fluvial processes, but they are found only under specific spatial and temporal conditions, that is as: (a) modern, inset point bar deposits that formed after the milldam breached and after incision cut deep enough into the to stack of stored millpond sediments to mobilize coarser material; or (b) thin, lenticular, sandy strata at the top of some sediment stacks far upstream of the breached milldam impoundment.

These observations demonstrate that: (i) sandy-gravel point bar deposits form only in the modern meandering stream, as modern inset features spatially lower and temporally later than the silt loam cut banks of older legacy sediments; (ii) sediment clasts in these point bar deposits are all locally derived, range from gravel to clay, and fine upward; (iii) the absence of fluvially derived sediment structures in older stream bank legacy sediments is proof that they were not deposited by flowing water; and (iv) occasional thin, sandy deposits that cap upstream legacy sediment surfaces were deposited after the millpond reservoir filled with sediment. The coarsening upward sequence noted in iii and iv above is characteristic of sedimentation in a delta, in which "legacy sediments" comprise the fine-grained bottomset beds (suspended load) and occasional thin, lenticular sandy caps as the coarser topset beds (bedload) of a shoaling delta sequence. Also, just after a dam breaches and incision begins, the meandering stream banks are relatively shallow and overbank deposition of coarser bedload material can spread out on a transient "floodplain surface". As incision cuts more deeply in the millpond sediments, such over bank deposition becomes less frequent until the cut is so deep that some banks (especially those between 6 and 20 ft high) have never experience overbank deposition again. For example, in the 10+ years we have studied the Denlinger's Mill site, which has legacy sediment cut banks up to 20 feet high, no storm flow has gone higher that half the bank height.

Wegman et al. (2013) document two stages of Post-Settlement legacy sediment deposition in Piedmont valley bottoms of North Carolina: (1) thinner (ca. 1 ft) pre-dam sediments consist primarily of fluvial sands, and are interpreted as channel aggradation in response to soil erosion from upland land clearing prior to dam construction; and (2) thicker (up to 10 ft) Post-dam sediments are characterized by finer grain size and sedimentology consistent with slack water deposition. It is noted that intense milldam construction began ca. 100 years later in the southern Piedmont compared to the Mid-Atlantic region (Walter and Merritts, 2008: supplementary information), indicating that during the period of intense agriculture and soil erosion that some pre-dam fluvial deposits had time to form in the southern Piedmont but not in the Mid-Atlantic region were intense erosion and milldam construction were contemporaneous. Further, Wegman et al (2013) accentuate the necessity of milldam impoundments for trapping fine-grained legacy sediments, which comprise the majority of stream banks in the North Carolina Piedmont.

Understanding the correct mechanism by which post-settlement, agricultural-era "legacy" sediments were deposited, as well as understanding the reason for modern incision and bank erosion, are crucial for determining what to do about stream bank erosion. In other words, diagnosing the problems correctly is crucial to any efforts at restoration. For example, although Wolman (1958) and Costa (1975) recognized that bank erosion rates in the Maryland Piedmont are high, this erosion was interpreted to be the result of urban development and storm water runoff (c.f., Walsh et al., 2005). Subsequent restoration strategies were developed in a targeted effort to stop or slow stream banks from eroding (e.g., riparian buffers and bank rip rap). Incised streams with eroding banks were characterized as having "urban bank syndrome", yet our work shows that such conditions are found wherever a historic milldam has breached, regardless of upstream land use. For example, Mountain Creek, PA, has nearly 100% forest cover that dates to the late 1800s, but 11 of its 12 milldams are breached, and all but the unbreached dam site have incised, eroding banks.

Bank armoring often fails because these strategies are rooted in a paradigm accepting stream banks as part of a natural fluvial system that was simply "overloaded" with agricultural sediment. When it becomes clear, however, that modern meandering streams are as much artifacts of the post-settlement agricultural era as the legacy sediments themselves, then one must recognize that new restoration methods are needed that use this knowledge to its fullest potential.

Unlike upland soil erosion, where the sediment delivery to the stream is a function of a number of complex variables (e.g., geomorphologic, hydrologic, landuse, soil texture, vegetation, location, distance to stream, and extent of sediment sources), the delivery of bank sediments to stream water is as uncomplicated as it is efficient: 100% of eroded stream bank sediment will be delivered to the stream, perhaps not instantaneously but over a few seasons. Stream bank erosion is a highly seasonal and stochastic process, but it can yield high long-term average suspended sediment loads (Merritts et al., 2011). Here we document the nutrient content of stream bank sediments at Big Spring Run. In

subsequent sections we relate these concentrations to nutrient loads from stream bank erosion.

Brantley et al. (2011) outline twelve testable hypotheses on the geobiology of weathering, in which Hypothesis 11 states: "In many severely altered settings, restoration of hydrological processes is possible in decades or less, whereas restoration of biodiversity and biogeochemical processes requires longer timescales". The Big Spring Run restoration experiment is testing this hypothesis through the removal of ca. 21,955 tons of legacy sediment and the subsequent reconnection of groundwater and surface water in a low, valley bottom aquatic ecosystem. How long will it take to establish the hydrology, biodiversity and biogeochemical processes of a functioning wetland ecosystem? Here we quantify the reduction of nutrient loads as a result of legacy sediment removal, and discuss the impacts this is having and will have on the rejuvenated and restored valley bottom aquatic ecosystems.

A. 3. Restoration Target: Pre-settlement Anastomosing Channels and Palustrine Emergent Wetlands

The template for valley bottom restoration in the Mid-Atlantic region lies beneath the legacy sediments, and extends back in time to a deeper understanding of the geological trajectory and development of these buried aquatic ecosystems. Essential is an understanding of the functions provided by pre-settlement valley bottom aquatic ecosystems. Previous workers identified older strata beneath the post-settlement agricultural sediments, and many described seeing dark, organic rich sediment directly underlying the oxidized legacy sediment. For example, Costa (1975) and Jacobson and Coleman (1987) note a "not uncommon" dark organic horizon below agricultural sediment, but attributed it to a buried soil of unknown age, origin or relevance. However, the age and origin of this organic rich pre-settlement soil are relevant, and are part of the solution to developing sustainable restoration targets.

We now recognize that Mid-Atlantic pre-settlement, Holocene valley bottoms were characterized by pervasive wetland ecosystems (Voli et al., 2010; Merritts et al., 2011), with no evidence of significant single thread meandering fluvial processes during the Holocene (Walter et al., 2008). Based on this knowledge, we propose that the restoration approach should be to restore the valley bottom wetland ecosystems that predominantly consisted of palustrine emergent marshes, similar to that which exists today at Great Marsh. Wetlands are highly valued for their ability to trap sediment and filter or remove nutrients from surface water (Tiner, 1987), and their widespread restoration could substantially and markedly improve surface water quality (Mitsch and Gosselink, 2000).

Following Martin's (1958) seminal paper on the age and ecological persistence of Great Marsh for millennia, Bricker and Moss (1958) struggled to understand why Great Marsh was there at all, noting how unusual such valley bottom wetlands were in the Mid-Atlantic Piedmont. What Bricker and Moss could not realize at the time was that these valley bottom wetlands were actually pervasive, but that they were buried and hidden from view for two centuries under stacks of millpond sediments. By the late 20th and early 21st

Centuries, millpond sediment stacks had been eroded deeply enough to expose these widespread, buried wetland soils, providing us with the evidence necessary to reconstruct their origin and evolution (Walter and Merritts, 2008; Voli et al., 2009; Merritts et al., 2011).

Beneath these pervasive valley bottom wetlands is an older yet equally pervasive gravel deposit, which has been consistently misidentified as fluvial point bar deposits (see Costa, 1975; Jacobson and Coleman, 1986). These deposits contain clasts that range in size from boulder to fine sand: they are usually quartz, are typically angular to subangular in texture, and they directly overly bedrock (Walter and Merritts, 2008; Merritts et al, 2013). Typically, these clasts are matrix-supported with no clear bedding structures. In aggregate, these observations highlight: (1) the improbability of small-order Piedmont streams carrying clasts the size of boulders; (2) the fact that large angular clasts would become rounded if transported by water even a short distance; and (3) that the lack of bedding structures and the presence of matrix-supported clasts are indicators of transport by mass movement, and not by water.

Instead, mounting evidence (Merritts et al., in prep) suggests that these poorly sorted gravels moved downslope into valley bottoms by periglacial mass wasting process that occurred during one or more Pleistocene glacial periods, when much of the Mid-Atlantic Piedmont was experiencing permafrost conditions, and remained essentially in place (except for winnowing of matrix fines) ever since. These Pleistocene gravels served to concentrate and direct shallow groundwater flow in the valley bottom, and formed the substrate on which Holocene wetlands eventually developed and evolved. Prior to European settlement, the gravel/hydric soil complex formed the hyporheic zone, where critical surface water/groundwater interactions occurred.

Burial in wet silt deposited upstream of small dams and other valley grade control structures provided ideal conditions to preserve paleo-records at the BSR restoration site. Fossil seeds of water plantain (*Alisma plantago-aquatica*), for instance, in silt near the base of legacy sediment indicate that ponding began circa 1730 (Neugebauer, 2011). The paleo-record at BSR also includes extensive, dark, organic-rich hydric soil that was buried—and preserved—by historic sedimentation in slackwater environments (see (a) in Figure A4). Portions of this buried hydric soil were exposed along most of the length of the incised channel prior to restoration, and still are exposed along incised channels of BSR downstream of the restoration reach. Two exceptions included locations where (1) a sewer line was installed below the level of the hydric soil in the mid-20th c., and (2) at point bars formed in the wake of stream incision and eroding channel banks that formed during the 20th c.

At BSR, the paleosol that is a hydric soil varies in thickness from 20 to 50 cm and is composed of dark gray to black (typically 10YR 2/1) organic matter, sand and locally abundant angular to sub-angular quartz gravel derived from long-term weathering of the Paleozoic limestone bedrock with quartz veins. At some valley margin locations, the underlying quartz rubble is contained within toe-of-slope colluvial deposits.

The paleo-record at BSR indicates at least 3000 years of aquatic ecosystem stability persisted prior to European settlement in 1709 (Merritts et al, 2009; discussed in detail in section F). Analysis of fossil seeds and 18 radiocarbon dates acquired from the extensive, organic-rich hydric soil reveals that a palustrine emergent marsh that was predominantly a sedge *(Carex spp.)* dominated wet meadow wetland persisted from ca. 3500 BP to 300 BP (ca. A.D. 1700). The wet meadow wetland surface intersected the ground water table at the level of multiple seeps and springs that control base flow. Notably absent from the paleo-seed record are woody plants typical of palustrine forested and shrub-scrub dominated plant communities that are considered in many riparian plant community restorations. During this period of thousands of years of aquatic ecosystem stability, the long-term sedimentation rate in the valley bottom was low, only ~0.01 cm/yr, resulting in just several tens of cm of deposition during the last three millennia.

Paleo-geomorphic analysis revealed that small, low-energy channels with minimal bedload sediment transport existed throughout the wet meadow at BSR prior to European settlement. We propose that the reason we have not found distinct palaeochannel forms buried beneath historic millpond sediment at BSR, or other mid-Atlantic Piedmont 1st-3rd-order streams, is that channels in wet meadows were likely to be similar to those characterized by Nanson & Knighton (1996) as the cohesive-sediment anabranching type. Such laterally stable channels would have been multiple and small, with low stream power (<10 W m^{-2}) and with cohesive banks bounded by wetland soils. These streams transported little sediment, a finding that is consistent with the low relief in the BSR watershed, the limestone bedrock, and the paleo-record of low sediment loads to the Chesapeake Bay prior to Colonial settlement [Brush, 1989].

Prior to mill damming and post-dam breach incision into legacy sediment, the valley bottoms described here were substantially different throughout the Holocene Epoch (11,500 years ago to present); with a key pre-settlement difference being the predominance and ecological persistence of sedge dominated wet meadows for thousands of years. We propose that as wetlands developed upon a low-relief periglacial rubble substrate during the Holocene, plants that populated the spring-fed valley bottoms might have increased resistance to, and hence attenuated, water flow. Some of the species listed above, particularly *Carex spp*., form prominent mounds or hummocks that add microtopographical flow resistance as well as bed and bank roughness elements.

Although stable anastomosing channels are considered relatively uncommon today (Knighton, 1998), a review of archaeological, historic and geomorphological evidence indicated that anastomosing channels and floodplain wetlands 'were formerly of considerable significance' in lowlands of England and Wales [Lewin, 2010, p. 267]. We posit that they were similarly of considerable significance in low-relief areas of the mid-Atlantic region prior to European settlement and anthropogenic impacts.

The target post-restoration plant community type at BSR is similar to the pre-settlement palustrine emergent marsh that is predominantly an open canopy wet meadow (Figure A4). The depth of excavation during restoration was based on our estimates of the location of the pre-settlement ground surface in the valley bottom. This ground surface is a hydric

soil that merges along valley margins with colluvium derived from hillslopes (Kratz, 2011). Most of the colluvium is coarse gravel-to boulder-sized material that formed during the Pleistocene Epoch (11,500 to ~2 million years ago), a time of cyclical cold and warm intervals. During some of the cold intervals, temperatures in the region were low enough for permafrost to form. During annual warm periods (e.g., summer months), sediment moved downslope as the uppermost part of the permafrost thawed. The past 11,500 years, known as the Holocene, is the most recent of the warm episodes. Warming began ~15,000 years ago, leading to the complete demise of permafrost and deep seasonal frost in the region. As a result, large amounts of colluvium moved downslope during this period of extensive permafrost thawing, much as is happening today in Arctic permafrost regions as a result of modern global warming.

It is possible that fluvial processes in the valley floor prior to Holocene wetland formation and stabilization reworked some Pleistocene colluvium, but we find no evidence of sediment transport of cobble or larger sized sediment in the sedimentary record. This conclusion is not surprising, considering that BSR is a low-gradient headwater site (<2 km^2 upstream drainage area). The Holocene hydric soil formed on this Pleistocene periglacial rubble as a result of groundwater flow from bedrock to the gravel substrate at springs.

A. 4. Restoration Summary

The engineering firm LandStudies, Inc. removed ~23,000 tons of legacy sediment, most of which was fine-grained silt, during restoration in September-October 2011 (Figures A5-A8)[1]. The post-restoration stream and floodplain wetland ecosystem established at the level of the original wetland hydric soil consists of small channels with low banks that frequently flow overbank (Figures A9-A11). Discharge greater than spring base flow is conveyed through both the channel and floodplain, with variable channel depth of 0.1-0.2 m and floodplain boundary shear stresses <1.5 N/m^2 to maximize channel stability. The channel planform increases flow retention and promotes exchange between the stream channel and hyporheic zone across the entire valley bottom.

Ongoing scientific research and monitoring at BSR includes multiple ecological, hydrological, and geomorphic strategies, as follows:

(1) Identify pre- and post-restoration plant communities, herpetological communities, and aquatic macroinvertebrate communities and their characteristics,

(2) Determine sediment sources and quantify pre- and post-restoration sediment transport (suspended load) and fluxes,

(3) Characterize bedload transport,

[1] See Appendix 1 for engineering and construction details.

(4) Quantify nutrient fluxes, and

(5) Model wetland hydroecological dynamics (lead investigator Dr. Laurel Larsen, University of California at Berkeley).

Three USGS stream gaging stations installed for this project (with turbidity and temperature sensors and suspended sediment sampling) are located at the upstream end of each tributary entering the restoration reach and on the main stem downstream of the restoration reach. USGS data are collected with respect to water years, which begin on October 1 and end on September 30 each year. Groundwater is monitored and sampled with18 USGS piezometers that were installed in October 2008.

Three years of pre-restoration and 2 years of post-restoration data have been collected from these gage stations and piezometers as of this report (October 2013). Only the first year of post-restoration data has been analyzed to date and is presented here. An update to this report will be provided when USGS Water Year 2013 data, the 2nd year of post-restoration monitoring, are analyzed.

A. 5. Design Criteria and Goals

General criteria that guided the restoration engineering design and construction were based on the geomorphic and paleo-ecologic assessments of the pre-restoration condition and ecological functions and services provided by similar aquatic ecosystems. The criteria are as follows:

1. Flows greater than normal spring base flows are conveyed through the floodplain.

2. Woody material placed within the channel increases the water surface elevation during normal base flow and promotes base flow exchange within the hyporheic zone.

3. The channel and floodplain are designed to allow for extremely wet areas and dry areas depending upon annual precipitation amounts and fluctuating groundwater levels. The legacy sediments were excavated and removed from the valley bottom to the elevation of the pre-settlement floodplain material that remains as is and in place.

4. The channel plan form is designed to increase flow retention and flow exchange from the stream channel into the adjacent hyporheic zone and across the valley bottom. This involved creating a variety of stream reaches (short steep run/riffles, long pools, wide pools, narrow pools, etc.) and slight depression areas across the valley bottom and along toes of hill slopes near the valley margins.

5. In order to provide additional denitrification potential and other habitat and base flow grade control benefits, stumps and other woody material are placed frequently within the channel and on the floodplain surfaces. These materials control the base flow direction and water surface elevations within the channel and promote nutrient removal and increase roughness within the floodplain.

6. The constructed channel has a variety of widths and depths with a varying streambed profile, creating a low gradient slope overall. Vertical drops through the restoration occur primarily over woody material or larger rock that provides both base flow and invert grade control and represents debris jams or colluvial material deposited along a hill slope.

A. 6. Description of Restoration Construction Process

A John Deere 750J-LGP dozer primarily was used to establish the restored floodplain grade by removing legacy sediment to the pre-settlement floodplain elevation. Track hoes also were used to establish the floodplain elevations in places where using the bulldozer was not possible, as well as to excavate the restored channel. Restored channel inverts were established by the elevations of bedrock and gravel layers that were encountered during the initial channel excavation.

Next, log and brush piles were used to establish grade control for restored channel base flow (see Figures A6 and A9). The log and brush pile structures were established by excavating a trench to bedrock depth and perpendicular to the restored channel, then installing the structures and backfilling over the structures to the desired channel width.

Erosion control matting/blankets were placed along restored channels per standard application of erosion/sediment control techniques (see Figure A6b). The erosion matting/blankets consisted of an interwoven biodegradable material that enhances seed germination by absorbing water, regulating soil temperature, adding soil erosive resistance strength, etc. The material also permits installation of woody vegetation within the interwoven and flexible strands of netting material.

The exposed soils were seeded with a temporary cover crop of cool season winter rye (grain) annuals due to the time of year and likelihood that a permanent cover of perennial wetland species was not expected given the short growing season after sediment excavation was completed in December 2011. The annual cover crop also acted as a nurse crop that germinates and grows quickly, keeping soil and supplemental wetland seeds in place. In addition, a commercial wetland seed mixture with predominantly perennial plants was applied to the entire site immediately after sediment excavation. This commercial seed mixture was custom designed to include species found in the paleo-seed record at this site. Not all species representing the wet meadow herbaceous paleo-seed record are available from commercial suppliers, so the seed mixture included additional species indicative of wet-meadow herbaceous plant communities typical for this region.

In the spring of 2012, after as-built contour plans were developed and groundwater/surface water observations were made for several months, additional native wetland seed mixtures were applied. In addition, approximately 10,000 commercially available container grown herbaceous plants, commonly referred to as plugs and predominantly sedges and rushes typical of tussock forming sedge meadows, were installed. Some woody species (shrubs and trees) were added to the perimeter of the wet

meadow, forming a riparian buffer for the restored palustrine emergent wetland (see Figure A9b). Since that time, no significant planting activities have occurred.

B. Deliverable 1: Quantify volumes of historic (legacy) sediment in stream corridor and associated nutrients in sediments. (Merritts/Walter/Rahnis/F&M Staff)

B. 1. Volume of legacy sediment in stream corridor at BSR

During 2004-2011, we examined Pleistocene, Holocene, and historic stratigraphy in stream cut banks, in the walls of trenches dug with a backhoe, and in cores extracted by drilling. From these investigations, we determined the elevation of the pre-settlement valley bottom surface throughout the restoration area. The depth of excavation during restoration was based on measurements and estimates of these pre-settlement ground surface elevations in the valley bottom.

This original floodplain surface is a relatively shallow hydric (wetland) soil that merges along valley margins with older colluvium derived from hillslopes. We distinguished historic (~1710-1930 AD) sediment in the valley bottom from this older (Pleistocene) colluvium and from the dark hydric Holocene wetland soil sandwiched between the two.

We estimate the volume of legacy sediment in the restoration reach from the difference between (1) the ground surface obtained from pre-restoration (2008) lidar, and (2) the ground surface obtained from a post-restoration (2011) as-built survey done with a total geodetic station by LandStudies, Inc. (Table B1). This estimate includes only the legacy sediment removed from the area shown within the black lines in Figure B1.

The red lines in Figure B1 identify the total area of legacy sediment, including unrestored portions of the study area that are up and downstream. The white dashed lines in Figures B2 and B3 represent the areal extent of legacy sediment accumulated in the valley bottom and the construction zone area, respectively. Figures B2 and B3 are provided with two different types of base maps for viewing.

Table B1. Estimate of amount of legacy sediment excavated (*cut*) and used to fill low spots in the valley bottom (*fill*) during restoration over an area of ~362,748 ft^2 (8.3 acres). Sediment also was used as fill to bring low areas up to design elevation where the stream had cut into the hill slopes along valley margins.

	Cubic ft (ft^3)	**Cubic yards (yd^3)**
Legacy sediment removed (cut)	612,252	22,680
Sediment used as fill	26,352	976
Net sediment removed*	589,900	21,704

*Net = Cut – Fill

These estimates of volume of sediment cut (removed) and used for fill do not take into account changes in density and volume that might occur as a result of excavation. The difference in volume between compacted versus loose fine-grained sediment can be as high as 35%.

B. 2. Nutrients in Sediments Exposed along Stream Banks at Big Spring Run (Merritts/Walter/Rahnis/F&M Staff)

Introduction

Anthropogenic processes can cause an increase in the discharge of nutrients into fluvial systems, but the magnitudes of these increases are difficult to predict (Jordan et al., 1997a). For example, watersheds with greater proportions of agricultural land might discharge excess N, excess P, or excess N and P (Hill, 1978; Neill, 1989; Mason et al., 1990; Dillon and Kirchner, 1975; Rekolainen, 1990; Correll et al., 1992; Nearing et al., 1993), while some smaller watersheds show no correlation at all between the proportion of agricultural land and fluvial discharges of N or P (Thomas et al., 1992). For some larger drainage basins, nitrate discharge shows little correlation with the proportion of cropland, but is highly correlated with anthropogenic input from atmospheric deposition, fertilizer application, cultivation of N_2-fixing crops, and net import of agricultural products (c.f., Jordan and Weller, 1996). It is well known that fluvial discharge of P is more strongly influenced by erosion rates and transportation distances of fine sediment, rather than by a metric linked to anthropogenic inputs, such as the application rates of P-rich fertilizer (Vighi et al., 1991; Jordan et al., 1997).

Despite the multitude of nutrient pathways, discharge of both N & P is strongly correlated with precipitation and to the type of water delivery mechanism that transports each nutrient from source to sink (Jordan et al., 1997b): (1) Sheet wash and rilling can cause soil erosion that enhances the transport of particulate-bound P (Dillon and Kirchner, 1975; Grobler and Silberbauer, 1985): (2) In contrast, infiltration of groundwater enhances transport of nitrate, which is highly soluble and readily leachable from soils under specific biogeochemical conditions. For example, groundwater contamination from nitrate (via nitrification processes) is common beneath well-drained, oxidized soils (e.g., Spalding and Exner, 1993), whereas nitrate removal (via denitrification processes) occurs in poorly drained, reduced soils (Gambrell et al., 1975).

Understanding delivery mechanisms and pathways that influence nutrient discharge is critical to understanding how lakes, estuaries, and coastal waters become degraded by nutrient inputs (Nixon, 1995, Boesch et al., 2001). Discharge of nitrate to streams is related to a groundwater delivery mechanism that reflects the leaching potential of soils in the watersheds, the oxidation state of soils and sediments it encounters during transport, the hydraulic pathways of groundwater movement (Brenner and Mondok, 1995), and biogeochemical processes acting on ground and surface water mixing in the hyporheic zone, where, for example, gradients in redox potential control chemical and microbially mediated nutrient transformations occurring on particle surfaces. (Boulton et al., 1998). Such studies shine considerable light on the fact that nitrogen transformations in soils and sediments can strongly influence the availability of soluble forms of nitrogen available to ecosystems, and emphasize the importance of understanding nitrogen contents in soils and

the biogeochemical processes that may cause soluble reactive N to increase or decrease in surface waters.

B. 2. a. Phosphorus (P)

Systematics

Phosphorus is relatively insoluble, and moves through watersheds primarily by electrostatic attachments to the surfaces of fine particles via inner and outer sphere sorption bonds (Brady and Weill, 2008). The processes by which P desorbs and becomes a free ion in water are complex and incompletely understood: (1) phosphorus occurs primarily as the P^{+5} cation in soil organic matter and as the PO_4^{-3} molecule associated with inorganic matter (i.e., orthophosphate or ortho-P: c.f., Essington, 2003); and (2) phosphorus chemistry in soils and sediments is strongly influenced by redox potential; (a) under oxidized conditions, ferric and manganic oxides and hydroxides are important adsorption sites for P (Moore and Coale, 2000); and (b) under reducing conditions these minerals are unstable, resulting in dissolution and release of P into the soil solution (Patrick et al., 1973; Emerson, 1976; Emerson and Widmer, 1978; Boyle and Lindsay, 1986; Moore and Reddy, 1994).

Background

The transfer of phosphorus from nonpoint-source soils to freshwater bodies and estuaries contributes to accelerated eutrophication in receiving waterways (Sharpley et al., 1999; Pierzynski 2000; Bennett et al., 2001; and Vadas et al., 2005). Nutrient-induced eutrophication restricts water use for fisheries, recreation, industry, and human consumption due to increased blooms of undesirable algae and aquatic weeds, followed by oxygen depletion in bottom waters as this unwanted biomass decomposes. In recent years, the relative contribution of P from nonpoint sources to surface waters has increased substantially, as point sources of P have been identified and reduced. Renewed attention on nonpoint source P has increased the demand for analysis for soil, water, and residual materials for environmentally relevant forms of P (Pierzynski 2000), and for knowledge about the methods used in making these determinations.

A practical approach to address nonpoint source P pollution is to: (1) identify nonpoint source areas in watersheds with high potential for soil-derived P export; (2) quantify the soil-P export; and (3) assess the ability of management practices to minimize this export (Coale et al., 2002). Part of our monitoring program for the Big Spring Run (BSR) restoration experiment was to accomplish these three goals by: (a) measuring the forms of phosphorus in stream bank sediments (Deliverable 1); and (b) documenting the contribution of P to downstream waterways from stream bank erosion (Deliverable 3&4). Our working hypothesis was that bank erosion along the Big Spring Run restoration reach contributed large sediment and sediment-derived P loads to surface waters, and that these contributing loads would be substantially reduced by removal of the stream bank

sediments (the impairment) and by the creation of a valley bottom aquatic ecosystems that will store sediment and process nutrients.

The Big Spring Run watershed, in the temperate Piedmont Physiographic Province, is underlain by the Cambro-Ordovician Conestoga Formation, a light gray limestone containing abundant white quartz veins and interbedded with dark gray phyllite. Long-term weathering of the Conestoga Formation, over the past 50 to 100 million years, yielded thick residual soils that are composed of quartz fragments (clay to cobble-size) and clay and colloidal Fe-Mn-Al oxyhydroxides. No silicate clay minerals (e.g., vermiculite or montmorillinite) have been found in Lancaster County soils above the Conestoga Formation, indicating the dominance of non-silicate weathering processes. The end product of long-term weather of the limestone bedrock is a quasi-laterite soil rich in iron, manganese and aluminum nodules, and residual (lag) quartz fragments ranging from clay-size particles to boulders. Our research demonstrates that the P in soils at BSR is likely bound to Fe-Mn oxyhydroxides (Voynova 2006; Weitzman 2008; and Fullinwider 2010), which are prone to dissolution under reducing conditions (see P Systematics, above).

Analytical Methods

The principle method of P analysis is colorimetry (e.g., Murphy and Riley, 1962). Colorimetric methods produce sensitive and reproducible results, and lend themselves to automated analysis (Pierzynski and Sharpley, 2000). Recently, Inductively coupled plasma (ICP) spectrophotometry has been used for P determination (Pierzynski and Sharpley 2000), but results from ICP analyses are not always comparable to those from colorimetric methods. ICP measures the total amount of P in the solution, whereas colorimetry measures only the P that reacts with the color-developing reagent. Typically, ICP values are higher than colorimetric values measured on the same extractant (Pittman et al., 2005), suggesting that ICP values are a more accurate representation of the P concentration in the solution. For this report, we used a combination of colorimetry (via Flow Injection Analysis, FIA) and spectroscopy, namely Inductively Coupled Plasma-Optical Emission Spectroscopy (ICP-OES) and X-Ray Fluorescence (XRF) to obtain quantitative information regarding different forms of P present in soils and sediments (see below).

Phosphorus forms in soils and sediments are difficult to standardize (Pierzynski and Sharpley 2000), due to the variety of disciplines involved (e.g., soil scientists, agronomists, limnologists, hydrologists, and geoscientists), and the variety of goals and objectives desired by each discipline. For example, agronomists might be interested in understanding the concentration of bioavailable P (BAP) in soils in order to determine if the soils need to be amended to sustain agricultural fields. Geoscientists, on the other hand, might be more interested in how rocks weather to soils, and the pathways that various minor and trace elements, such as P, take in the transformation from bedrock and saprolite to residuum and soil. Therefore, geoscientists might be more interested in the total rock phosphorus (TP, bound in and on constituent minerals) and how this P gets released during weathering to become part of the more labile and soluble forms of P in the environment.

To avoid confusion of terms, Pierzynski and Sharpley (2000) recommend using chemical

identifiers when referring to various forms of P (e.g., water extractable P, Mehlich-3 extractable P, etc.), which have procedures that are well known and clearly defined. In addition, other terms used to describe and interpret P data (e.g., desorbable P, bioavailable P, sorbed P etc.) can be used as long as they are clearly defined. The following terms highlight the complexities in characterizing soil-derived P (modified from Pierzynski and Sharpley, 2000), and define the forms of P discussed in this report:

Total Phosphorus (TPx) = Total concentration of P in the soil or sediment (organic and inorganic). (Here measured using XRF lithium tetraborate fusion methods to obtain all forms of P in the internal structures of soil minerals and organic compounds, as well as P sorbed onto surfaces of organic and inorganic compounds.)

Total Sorbed Phosphorus (TPi) = Inorganic and organic P bound to surfaces of eroded sediment particles. (Here measured by inductively coupled plasma optical emission spectroscopy (ICP-OES) using a microwave-assisted HNO_3 digestion protocol (EPA 3051), which is a partial digestion that liberates sorbed and environmentally available elements from particle surfaces. This sorbed TPi is typically 60-70% of TPx, and represents essentially an "infinite" reserve of labile and water extractable forms of P, as defined below. Measured TPi concentrations are ca. 2-3 orders of magnitude greater than measured labile and water extractable forms of P).

Labile P (LP) = Bioavailable ortho-P and a portion of particulate P that is algal available. (Here measured with FIA colorimetric methods, or by ICP-OES analyses, using a Mehlich-3 digestion (see Baker 2011).)

Water Extractable P (WEP) = Dissolved inorganic (ortho-P) and organic P. (Here measure by ICP-OES using a deionized water extraction method.)

Early interests in examining soil P were primarily based on determining the quantity of supplemental P needed to adequately meet the needs of crops (Self-Davis et al., 2000). The method for using distilled water as an extractant to determine P needs of plants was examined in a paper by Luscombe et al. (1979), which – we believe -- is a more realistic measure of determining the soluble P fraction in soil, and the availability of that soil to yield soluble P to stream water. There is now a national focus on examining excessive P buildup in the soil and consequent excessive P concentrations in runoff from agricultural land. A study conducted by Pote et al. (1996) found an excellent correlation between water extractable soil test P and dissolved reactive P concentrations in runoff.

Results (Figures 1-29) ***[File: D1_BSR Stream Bank Nutrients]***

Total Phosphorus (TPx)

Table A shows measured concentrations for Total Phosphorus (TPx) in upland agricultural soils and in valley bottom stream bank sediments at Big Spring Run (see also Figures B5-B9). Surface soils (0-5 cm depth) from an active pasture yield a TPx concentration of 1792 mg-P/kg-sediment (see Appendix 4 for details of sampling and analytical methods). Note that TPx concentrations decrease with depth (from 1792 mg/kg at 0-5 cm to 1165 mg/kg

at >20 cm), presumably due to less biomass with depth, and to the enrichment of manure and biomass near the soil surface. Phosphorus concentrations (TPx) measured for limestone bedrock and saprolite yield two key points: (1) the Conestoga Limestone itself is enriched in phosphorus (ca. 450 mg/kg); and (2) the process of chemical weathering enriches saprolite in P (ca. 1050 mg/kg) roughly 2.3 times over the bedrock concentration, suggesting that substantial amounts of P are mobilized during bedrock weathering.

Chemo stratigraphic results from individual stream bank sections (Figure B4) are shown in Figures B5-B9. The highest TPx concentrations are typically in the upper 20 cm (0-20 cm), lowest concentrations in mid-section (ca. 20-100 cm), and again increased TPx concentrations in the lower 20 cm (ca. 120-140 cm). The bulk of stream bank sections (ca. 0-120 cm) are composed of legacy sediments, the upper 20 cm of which is the active root zone. The abundant biomass in this root zone accounts for its elevated TPx concentrations. The lower part of these sections (ca. 120-140 cm) are usually composed of dark, buried hydric soil with high organic matter content, which explains the high P concentrations in these stratigraphic units (see Sites 1, 2, 4, 5, 8, and 13). Where the hydric soils is absent (Sites 6, 12, and 14), the sections are composed of recent point bar deposits, dating from the early 20th C after the milldam on BSR was breached (these point bar deposits contain occasional historical artifacts such as tractor chains, cement blocks and bricks that date the deposits to no earlier than the early 20th C). The base of each measured section represents the base of the Holocene sediment sequence above the basal Pleistocene gravels.

Average Total Phosphorus (TPx) concentrations measured on ten stream bank sections along the BSR restoration reach yield values ranging from 874 mg/kg (Site 14) to 2792 mg/kg (Site 13), for an average of 1161 +/- 577 mg/kg (Table x). This average concentration is equivalent P concentrations in the saprolite, and which equates to a mass of ca. 2.3 pounds of P per ton of stream bank sediment. In terms of area, this average stream bank sediment value equates to 2321 pounds of phosphorus per acre. Note that the TPx concentrations for Site 13 are unusually high, with average of 2792 mg/k and with the highest values reaching concentrations in excess of 7000 mg/kg. We cannot explain these anomalous concentrations at this time, except to point out that the high P values occur in the lower part of the section, and might indicate a discrete mass of organic matter that was subsequently buried by sediment, and is not representative of the sediment as a whole.

Table A - Total Phosphorus (TPx) in Soils and Stream Bank Sediments at Big Spring Run by XRF Total Fusion

BSR Sediment Type	TPx (mg/kg)	TPx (lb/ton)	TPx (lb/ac)
Active Pasture			
Upland Soil A (0-5 cm)	1792	3.6	3583
Upland Soil A (5-10 cm)	1434	2.9	2869
Upland Soil A (10-15 cm)	1230	2.5	2460
Upland Soil A (15-20 cm)	1259	2.5	2517
Upland Soil A (20-80 cm)	1165	2.3	2329
Saprolite below Soil A	1047	2.1	2093
Bedrock below Soil A	458	0.9	916
Stream Banks			
Site 1 (n = 28)	1007	2.0	2014
Site 2 (n = 12)	1019	2.0	2038
Site 4 (n = 27)	941	1.9	1882
Site 5 (n = 12)	944	1.9	1888
Site 6 (n = 17)	935	1.9	1870
Site 7 (n = 23)	992	2.0	1984
Site 8 (n = 58)	1129	2.3	2258
Site 12 (n = 8)	974	1.9	1948
Site 13 (n = 18)	2792	5.6	5584
Site 14 (n = 15)	874	1.7	1748
Avg. (Bank)	**1161**	**2.3**	**2321**
Std. Dev. (Bank)	**577**	**1.2**	**1154**

Total Sorbed Phosphorus (TPi)

Table B shows measured concentrations for Total Sorbed Phosphorus (TPi) in upland agricultural soils and in valley bottom stream bank sediments at Big Spring Run. Surface soils (0-5 cm depth) from an active pasture yield a TPi concentration of 1377 mg-P/kg-sediment (or 76.8% of the TPx value). Note that like TPx concentrations. TPi decreases with depth (from 1377 mg/kg at 0-5 cm to 788 mg/kg at >20 cm), presumably due to less biomass with depth, and to the enrichment of manure and biomass at and near the soil surface. Sorbed phosphorus concentrations (TPi) measured for limestone bedrock and saprolite yield two additional points: (1) the Conestoga Limestone contains substantial amounts of sorbed P (257 mg/kg, or 56.1% of the TPx value); and (2) the saprolite in P (ca. 1178 mg/kg) yields 100% of the TPx value (within error), suggesting that all the P in the saprolite is potentially mobile and accessible.

Total Sorbed Phosphorus (TPi) concentrations measured on eleven stream bank sections along the BSR restoration reach yield average values ranging from 518 mg/kg (Site 14) to 2921 mg/kg (Site 13), for an average of 811 +/- 703 mg/kg (Table B). Chemo stratigraphic relationships for TPi in the eleven stream bank sections (Figures B10-B14) mirrors that observed above for TPx, but with the TPi concentrations usually about 70-80% of the TPx values, suggesting that roughly ¾ of the total phosphorus in the silt-loam stream bank sediments is sorbed onto the surfaces of fine sediment particles.

This average sorbed P concentration equates to a mass of ca. 1.6 pounds of P per ton of stream bank sediment. In terms of area, this average value equates to 1623 pounds of phosphorus per acre.

Table B – Total Sorbed Phosphorus (TPi) in Soils and Stream Bank Sediments at Big Spring Run by ICP-OES and EPA 3051 Partial Dissolution

Sediment Type	TPi (mg/kg)	TPi (lb/ton)	TPi (lb/ac)	% of TPx
Active Pasture				
Upland Soil A (0-5 cm)	1377	2.75	2753	76.8
Upland Soil A (5-10 cm)	1078	2.16	2156	75.2
Upland Soil A (10-15 cm)	848	1.70	1696	68.9
Upland Soil A (15-20 cm)	741	1.48	1483	58.9
Upland Soil A (20-80 cm)	788	1.576	1576	67.7
Saprolite below Soil A	1178	2.356	2356	113
Bedrock below Soil A	257	0.514	514	56.1
Stream Banks				
Site 1 (n = 28)	589	1.2	1178	58.5
Site 2 (n = 12)	615	1.2	1230	60.4
Site 4 (n = 27)	602	1.2	1204	64.0
Site NYT (n = 26)	539	1.1	1078	
Site 5 (n = 12)	615	1.2	1230	65.1
Site 6 (n = 17)	501	1.0	1002	53.6
Site 7 (n = 23)	678	1.4	1356	68.3
Site 8 (n = 58)	732	1.5	1464	64.8
Site 12 (n = 8)	615	1.2	1230	63.1
Site 13 (n = 18)	2921	5.8	5842	104.5
Site 14 (n = 15)	518	1.0	1036	59.3
Avg. (Bank)	**811**	1.6	1623	65.1
Std. Dev. (Bank)	**703**	1.4	1406	14.1

Labile Phosphorus (LPf and LPi)

Labile (bioavailable) Phosphorus (LPf and LPi) concentrations are shown in Table C. The active pasture soils are highly enriched in bioavailable P, with LPf concentrations near the surface (0-5 cm) of 239 mg/kg. Concentrations decrease with depth, with the lowest values at <20 cm of 15 mg/kg. The saprolite yields a value of 10.7 mg/kg. No labile P measurements were made on bedrock.

Five stream bank section yield LPf and LPi concentrations ranging from 12.7 mg/kg (Site 2) to 18.9 mg/kg (Site 4), with detailed Labile P chemo stratigraphy shown in Figures B15-B18. Repeat analyses of Site 4 by Flow Injection Analyses (colorimetry) and by ICP-OES (spectroscopy) yield values of 18.9 mg/kg (LPf) and 17.4 mg/kg (LPi), respectively. Note that these two concentrations are within 2σ analytical error (ca. +/- 1.5 mg/kg), indicating that the two analytical techniques yield comparable results. The average of all LPf results is 16.2 +/- 3.2 mg/kg, and the average of all LPi results is comparable at 15.6 +/- 2.3 mg/kg. These averages yield mass and area equivalents of ca. 0.03 pounds per ton and 31.5 pounds per acre of bioavailable P, respectively.

Table C – Labile Phosphorus (LPf and LPi) in Soils and Stream Bank Sediments at Big Spring Run by FIA and ICP Methods via Mehlich-3 Extractions

Sediment Type	LPf (mg/kg)	LPf (lb/ton)	LPf (lb/ac)	Lpi (mg/kg)	Lpi (lb/ton)	Lpi (lb/ac)
Active Pasture						
Upland Soil A (0-5 cm)	239	0.48	478			
Upland Soil A (5-10 cm)	104	0.21	208			
Upland Soil A (10-15 cm)	65	0.13	130			
Upland Soil A (15-20 cm)	39	0.08	78			
Upland Soil A (20-80 cm)	15	0.030	30			
Saprolite below Soil A	10.7	0.021	21			
Bedrock below Soil A	N/A	N/A	N/A			
Stream Banks						
Site 1 (n = 28)				13.1	0.026	26
Site 2 (n = 12)	12.7	0.025	25			
Site 4 (n = 27)	18.9	0.038	38	17.4	0.035	35
Site NYT (n = 26)				13.7	0.027	27
Site 5 (n = 12)	17.1	0.034	34			
Avg. (Bank)	**16.2**	**0.032**	**32**	**15.6**	**0.031**	**31**
Std. Dev. (Bank)	**3.2**	**0.01**	**6.4**	**2.3**	**0.00**	**4.7**

Water Extractable Phosphorus

Water Extractable P (WEP) was measured at one stream bank site (Site NYT) as a test of the methodology. The measured WEP concentration was 6.1 mg/kg, or 44.5% of the P released by the Mehlich 3 extraction method and measured by ICP-OES (13.7 mg/kg, Table C). This equates to 0.012 pounds of Water Extractable P per ton of sediment, or 12 pounds per acre.

Table D – Water Extractable P (WEP) in Soils and Stream Bank Sediments at Big Spring Run by ICP-OES

Sediment Type	WEP (mg/kg)	WEP (lb/ton)	WEP (lb/ac)
Stream Banks			
Site NYT (n = 4)	6.01	0.012	12

Needle Ice Experiment (Figures 18-25 in D1 PPT File).

During the winter months in the mid-latitudes, many exposed soils and unconsolidated sediments experience diurnal freeze-thaw cycles. These cycles occur when ambient air temperatures alternate from below freezing at night to above freezing during the day. During a nightly freezing phase, when air temperatures at exposed soil surfaces falls below freezing, a freezing front of frozen pore water is created while at some depth below the surface temperatures remain above freezing and pore water remains in liquid form. As freezing progresses, interior liquid pore water is "wicked" by capillary action toward the freezing front due to pressure gradients at the soil/atmosphere interface. Active wicking brings interior liquid pore water to the surface, which freezes and pushed outward in long, thin needles as new ice forms from behind. Capillary action– the ability of a liquid to flow in narrow spaces against the force of gravity – occurs most readily in soils dominated by silt-size particles, as the pore spaces between silt grains are of the right dimension to promote capillarity. Depending on temperature gradients, the duration over which freezing occurs, and the texture of the soil, the depth of penetration of the freezing front may vary from a few millimeters to a few centimeters below the surface.

Over one winter season (2009-2010) prior to restoration, BSR experienced roughly 90 freeze-thaw cycles (Becker 2010). During several prior winter seasons, we observed the formation of ubiquitous ice needles protruding from vertical, silt-rich stream banks at BSR and elsewhere in the region. It struck us that needle ice is a sample of pore water, and that such a sample would provide valuable information regarding the dissolved nutrients in stream bank pore water (c.f., Zhao, 2010). The purpose of the study discussed here was to recreate needle ice formation in stream banks using an experimental apparatus designed and built by our research group during the summer of 2011(Figure B19a;B19b;B20). The objectives of this study were to measure P (PO_4) mobility between soils and pore water (needle ice) before and after multiple freeze-thaw cycles. Previous research suggested high concentrations of nitrates in the needle ice in the natural environment (Zhao 2010).

Starting with a sample of stream bank sediments collected from Site NYT, which has a WEP concentration of 6.0 mg/kg, we: (1) subjected the soil to multiple experimental freeze-thaw events(Figure B21-B25); (2) measured the WEP concentration in the thawed sediment (4.3 mg/kg); and (3) measured the concentration of P in the needle ice (1.2 mg/L). The results of this freeze-thaw study indicate that: (a) needle ice mobilizes roughly 2% of the water extractable phosphorus; and (b) the concentration of P lost from the soil (0.17 mg/kg) after a freeze-thaw cycle closely correlates with that gained in the needle ice (1.2 mg/L), suggesting a mass balance relationship between the two processes(Figure B26). This work shows that freeze-thaw processes during the winter season could be an important daily source of soluble orthophosphate to surface water as melting needle ice at the bank edge flows into the adjoining stream (Figure B27).

Conclusions – Phosphorus

Stream bank sediment contain high concentrations of total, total sorbed, bioavailable and water extractable P. Total Sorbed P (TPi) and Water Extractable P (WEP) are critical parameters in this investigation. Total Sorbed P represents the pool of soil P that can become available theoretically under reducing conditions, which for stream bank sediment at BSR averages roughly 740 mg/kg (or 1.5 pounds of sorbed P per ton of sediment). Water extractable P represents the pool of soil P that readily goes into solution, which is roughly 6 mg/kg (or 0.012 pounds of WEP per ton of sediment).

Approximately 21,955 tons of stream bank sediment was removed from the restoration ca. 3000-foot long reach, which equates to 26,346 pounds of sorbed P and 263 pounds of water extractable P permanently removed from the watershed.

In this report (see Deliverable C) we document that bank erosion contributes to roughly 63% of the annual suspended sediment load, indicating that stream banks at BSR are a major non-point source for suspended sediment and sediment-derived nutrients. Here, we quantify the phosphorus (P) composition in stream bank sediments at BSR.

B. 2. b. Nitrogen (N)

Systematics

Nitrogen is found mainly in organic forms in soils, and is an essential nutrient for plant growth. Nitrogen (N) moves through the environment mostly as anions and undergoes complex oxidation-reduction reactions to form multiple ionic species, some of which are soluble, while others are gaseous. The movement of excess soluble N compounds can disrupt the balance of aquatic ecosystems, leading to algal blooms, declining oxygen levels, and degradation of aquatic life (Brady and Weil, 2008).

Earth's atmosphere contains 78% nitrogen in the form of N_2 gas. Because of its strong triple bonds, however, this vast pool of N_2 is relatively inert and would not be directly available for plant or animal life if not for microbial nitrogen fixation in oxygen rich soils (nitrification), which along with lightening in the atmosphere are processes that can break N_2 triple bonds to form reactive nitrogen. Reactive N is any form of organic or inorganic N that is readily available to living organism (Brady and Weil, 2008), where the N is usually bonded to H, O, or C (e.g., NH_4^+, NO_3^-, and amino acids, respectively). On one hand, excess concentrations of reactive N can cause eutrophication and degrade water quality in receiving waters. On the other hand, microbes in reducing wetland soils can transform soluble NO_3^- into inert N_2 gas through a process called de-nitrification. Thus, an atom of N can appear in many different chemical forms, each with its own properties, behaviors and consequences for ecosystems. The transformation of soluble nitrogen to inert N_2 gas is a highly valued attribute of wetland ecosystems, and is becoming a critical restoration target in watersheds where ecosystems that support active nitrification can be rehabilitated to become ecosystems that support active denitrification (Mitsch and Gosselink, 2007).

Soil organic matter and nitrogen concentrations in soils are inextricably linked, which implies that any restoration intending to decrease reactive forms of nitrogen in surface and ground water must consider the correlation between C and N. This is a critical observation since most of the nitrogen in terrestrial ecosystems is found in soil (Brady and Weill, 2008), which leads to the logical conclusion that soil health is a key to ecosystem recovery and water quality improvements. The bulk of soil nitrogen occurs as constituents in organic molecules, which are thought to degrade slowly and are relatively insoluble. These recalcitrant organic forms of N, however, can undergo microbial biogeochemical transformation (mineralization) to produce highly soluble inorganic N in the form of ammonium (NH_4) and nitrate (NO_3). The vast majority of soil N resides in insoluble organic compounds (90-95%) that protect it from loss, but which makes it largely unavailable to higher plants unless mineralized. In this report, we focus on nitrates, as ammonium concentrations in BSR soils are negligible.

Taylor and Townsend (2012) observed that ecosystem nitrate accumulation exhibits consistent and negative nonlinear correlations with organic carbon availability along a hydrologic continuum from soils, through freshwater systems to coastal margins, and into the open ocean. This trend persists even in ecosystems subjected to substantial human disturbance. This global-scale insight adds credence to restoration designs intended to improve water quality through the reduction in nitrates that place an emphasis on adding organic carbon to the ecosystem (Mayer et al., 2003).

Background

Nitrogen is a limiting nutrient in Chesapeake Bay, where excess nitrogen, especially soluble forms such as nitrate-nitrogen (NO_3-N), can lead to algal blooms and low oxygen concentrations in the Bay's bottom waters. Based on land use analyses, the current EPA Chesapeake Bay Watershed Model designates that roughly 70% of the nitrogen load to the Bay comes from non-point sources such as farm fields and suburban lawns via the application of fertilizers, and that 69% of this non-point source (and non-atmospheric)

nitrogen pollution is presumed to be from agricultural sources. Nitrate nitrogen is highly soluble, and Lancaster County is regarded as a hot spot for dissolved nitrate in groundwater and surface water due to the large land area used for agriculture.

The research results presented here focuses on an analysis of potential sources of nitrogen derived from stream bank erosion at Big Spring Run, a small agricultural watershed in southern Lancaster County. The main objective of this study is to measure total nitrogen and nitrate concentrations in stream bank sediments, and to elucidate the processes by which soluble nitrate can be released to surface water due to bank erosion. An additional objective is to provide insights into possible mechanisms for nitrate production and mobility in upland soils and stream bank sediments in the Big Spring Run (BSR) Watershed. As we develop the tools to understand the nitrogen contribution from stream bank erosion at Big Spring Run, this understanding can help regulatory agencies such as PA DEP and the US EPA to scale up to larger watersheds.

As is common in many streams in the mid-Atlantic Piedmont region, stream banks at BSR consist of three principle stratigraphic units, which from youngest to oldest are: 1) post-settlement "legacy sediments"; 2) pre-settlement hydric soils; and 3) basal gravels (Walter and Merritts, 2008). Stream bank samples of legacy sediments and hydric soils were collected at ten stratigraphic sections along Big Spring Run. Samples were collected in 10 cm increments, from top to bottom of each site. Samples were analyzed for Total N by elemental combustion analyses (ECA) and for nitrate-N by flow injection analysis (FIA).

Results (Figures 30-49) ***[File: D1_BSR Stream Bank Nutrients]***

Total Nitrogen (TN) and Total Carbon (TC)

Table E shows measured concentrations for Total Nitrogen (TN) and Total Carbon (TC) from upland agricultural soils and valley bottom stream bank sediments at Big Spring Run (see Figures B28-B33 for detailed TN and TC chemostratigraphy in stream bank sediments). Observe that TN concentrations co-vary with TC concentrations, even though TC is roughly an order of magnitude greater in concentration than TN. Average TC/TN ratios for active pasture soils (12.1), fallow pasture soil (11.0), and stream bank sediments (14.7) are similar to each other and similar to a global soil C:N ratio of 14.3 observed by Cleveland and Liptzin (2007). Fixed soil C:N ratios across large geographical distances indicate that plants are the major source of total soil C and N in terrestrial ecosystems (Cleveland and Liptzin, 2007), which suggests a close biogeochemical link between organic matter and nitrogen in soils (Taylor and Townsend, 2012). For example, high N requirements during photosynthesis, combined with low N availability in many terrestrial ecosystems, means that increases in primary production are dependent on the availability of N (Vitousek and Howarth 1991; Asner et al. 1997; Cleveland and Liptzin, 2007).

Surface soils (0-5 cm depth) from an active upland hill slope pasture yield TN concentrations of 2558 mg-N/kg-sediment (Table E; see Appendix 4 for details of sampling and analytical methods). The TN concentrations in these soils decrease gradationally with depth to 844 mg-N/kg at >20 cm (Table E), presumably due to the gradual loss of biomass

with depth, and to the enrichment of manure and biomass at and near the soil surface. TN concentrations measured in soils from a fallow field yield NO_3 concentrations that are slightly higher than measured in the active field, yielding values ranging from 3925 mg-N/kg sediment at the surface (0-5 cm) (Table E), and which also steadily decline to 1150 mg-N/kg below 15 cm depth.

Results from individual stream bank sections are shown in Table E and Figures B28-B33, which typically indicate the highest TN concentrations in the upper 20 cm (0-20 cm), lowest levels in mid-section (ca. 20-100 cm), and increased TN concentrations in the lower 20 cm (ca. 120-140 cm), similar to what is observed for P (c.f., Figures B28-B31). The bulk of stream bank sections (ca. 0-120 cm) are composed of legacy sediments, the upper 20 cm of which is the active root zone. The abundant biomass in this root zone accounts for the elevated TN concentrations. The lower part of these sections (ca. 120-140 cm) are usually composed of dark, buried hydric soil with high organic matter content, which explains the high N concentrations in these stratigraphic units (see Sites 1, 4, 7, and 8; Figures B28-B31). Where the hydric soils are absent (Sites 13, and 14; Figures B32 and B33), the sections are composed of recent point bar deposits, dating from the early 20th C after the channel incised into legacy sediment causing channel banks to erode and leading to meander formation with alternating inset point bar deposits(these point bar deposits contain occasional historical artifacts such as tractor chains, cement blocks and bricks that date the deposits to no earlier than the early 20th C). The base of each measured section represents the base of the Holocene sediment sequence above the basal Pleistocene gravels.

Average Total Nitrogen (TN) concentrations measured on six stream bank sections along the BSR restoration area yield values ranging from 916 mg/kg (Site 13) to 2472 mg/kg (Site 8), for an average of 1411 +/- 559 mg/kg (Table E). This average concentration equates to a mass of ca. 3.0 pounds of TN per ton of stream bank sediment (range = 1.8 to 4.9 lb/ton). In terms of area, this average stream bank sediment value equates to a value of 2881 pounds of nitrogen per acre (range = 1832-4924 lb/ac).

The average carbon content of BSR stream bank sections is ca. 18,000 mg/kg, which equates to roughly 36 lbs-C/ton. The carbon content of the active and fallow pastures are 18,000 and 23,000 mg-C/ton, respectively, showing that TC in the fallow pasture is significantly higher than the active pasture, presumably due to the relative increase in plant biomass in soils, even down to >15 cm depth) since the cessation of cattle grazing on this field a decade ago. Still, the TC/TN ratio for the active and fallow pastures are similar but significantly different at 12.1 (range = 11.5-12.6) and 11.0 (range = 10.7-11.3), respectively. Carbon and nitrogen contents in the stream bank sediments are lower than either the active or fallow pastures, while their average TC/TN ratio of 14.7 (range = 11.7-23.3) is higher, suggesting that carbon is relatively enriched or nitrogen is relatively depleted in stream bank sediments compared to upland soils.

Table E – Total Carbon and Total Nitrogen in Soils and Stream Bank Sediments at Big Spring Run by Elemental Combustion Analysis

Sediment Type	TC (mg/kg)	TC (lb/ton)	TN (mg/kg)	TN (lb/ton)	TC/TN Molar Ratio
Active Pasture					
Upland Soil A (0-5 cm)	27678	55.4	2558	5.1	12.6
Upland Soil A (5-10 cm)	17085	34.2	1660	3.3	12.0
Upland Soil A (10-15 cm)	18497	37.0	1760	3.5	12.3
Upland Soil A (15-20 cm)	16193	32.4	1557	3.1	12.1
Upland Soil A (20-80 cm)	8347	16.7	844	1.7	11.5
Avg. (Pasture)	**17560**	**35.1**	**1676**	**3**	**12.1**
Saprolite below Soil A	N/A	N/A	N/A	N/A	N/A
Bedrock below Soil A	N/A	N/A	N/A	N/A	N/A
Fallow Field					
Upland Soil B (0-5 cm)	38735	77.5	3925	7.9	11.5
Upland Soil B (5-10 cm)	25060	50.1	2745	5.5	10.7
Upland Soil B (10-15 cm)	18330	36.7	2015	4.0	10.6
Upland Soil B (15-110 cm)	11169	22.3	1150	2.3	11.3
Avg. (Fallow Field)	**23324**	**46.7**	**2459**	**5**	**11.0**
Stream Bank					
Site 1 (n = 28)	19544	39.1	1071	2.1	21.3
Site 4 (n = 27)	15927	31.9	1653	3.3	11.2
Site 7 (n = 23)	12309	24.6	1228	2.5	11.7
Site 8 (n = 58)	32630	65.3	2462	4.9	15.5
Site 13 (n = 18)	12816	25.6	916	1.8	16.3
Site 14 (n = 15)	13645	27.3	1313	2.6	12.1
Avg. (Bank)	**17812**	**35.6**	**1441**	**2.9**	**14.7**
Std. Dev. (Bank)	**7730**	**15.5**	**559**	**1.1**	**3.9**

Nitrate-Nitrogen (NO_3)

Average nitrate concentrations of upland agricultural soils and valley bottom stream bank sediments in the Big Spring Run restoration reach watershed are shown in Table F, and with NO_3 chemostratigraphy shown in Figures B34-B37. The surface soil of an active upland pasture yields concentrations of 122 mg-NO_3/kg (0 to 5 cm depth), but these concentrations drop steadily and markedly in lower soils to 7.9 mg-NO_3/kg at >20 cm depth. In the fallow field (Table F), nitrate concentrations range from 23.4 mg-NO_3/kg (0-5 cm) to 3.7 mg-NO_3/kg (>15 cm depth). Nitrate-N measured on saprolite (weathered bedrock) found below these upland soils yields a concentration of 3.2 mg-NO_3/kg. These results show that: (1) reactive NO_3 is enriched in surface soils and decreases monotonically with depth in upland soils to values <4 mg-NO_3/kg at depths below 15 cm; (2) nitrate contents are higher in the active pasture versus the fallow field, which is opposite the trend observed for TN concentrations; and (3) NO_3 concentrations in saprolite (3 mg/kg) are similar to the lowest subsoil value (3.7 mg/kg), indicating that limestone bedrock weathering might impose a substantial baseline value of labile NO_3 in surface soils and possible to groundwater.

Relatively high NO_3 concentrations are found in the root-zone in the upper 15-20 cm of each of the eight stream bank section studied (ranging from ca. 25 mg-NO_3/kg to ca. 5 mg-NO_3/kg: Figures B34-B37). These high NO_3 concentrations are likely the result of accumulation of plant roots, decaying plant matter, and other organic remains in the upper part of the soil. In addition, active bacterial nitrification processes might be occurring in the root-zone as well (Brady and Weil, 2002; Weitzman, 2011). Nitrate concentrations generally drop below 5 mg-NO3/kg in legacy sediments below 20 cm, and frequently remain low to the bottom of the sections, even though the pre-settlement hydric soils observed at Sites 4 and 8 (Figures B34 and B36) that possess high TN concentrations (c.f., Figures B29 and B31). On the contrary, we note an increase in Labile NO_3 at the base of sections that are composed of recent point bar deposits, where the hydric soil had been eroded prior to its deposition (Sites 6, 12, and 13; Figures B35-B37), and which paradoxically show decreasing TN values (c.f., Figure B32). These results indicate the stream bank sediments at Big Spring Run contribute an average of 6.1 mg-NO_3/kg to surface waters system via bank erosion. Under the correct redox conditions these sediments can continue to release labile NO_3 via microbial nitrification processes as oxidized sediments are transported through stream networks to the Chesapeake Bay.

Table F Nitrate-N (NO_3-N) in Soils and Stream Bank Sediments at Big Spring Run by Flow Injection Analysis via KCl extraction

Sediment Type	NO3k (mg/kg)	NO3k (lb/ton)	NO3k (lb/ac)
Active Pasture			
Upland Soil A (0-5 cm)	122.4	0.24	244.8
Upland Soil A (5-10 cm)	53.9	0.11	107.8
Upland Soil A (10-15 cm)	26.5	0.05	53
Upland Soil A (15-20 cm)	19.6	0.04	39.2
Upland Soil A (20-80 cm)	7.9	0.016	15.8
Avg. (Pasture)	**46.1**	**0.1**	**92.1**
Saprolite below Soil A	3.2	0.006	6.4
Bedrock below Soil A	N/A	N/A	N/A
Fallow Field			
Upland Soil B (0-5 cm)	23.4	0.05	46.7
Upland Soil B (5-10 cm)	8.5	0.017	17.0
Upland Soil B (10-15 cm)	6.8	0.014	13.5
Upland Soil B (15-110 cm)	3.7	0.007	7.4
Avg. (Fallow Field)	**10.6**	**0.0**	**21.2**
Stream Banks			
Site 1 (n = 28)	7.0	0.014	14.0
Site 4 (n = 27)	9.8	0.020	19.6
Site NYT (n = 26)	2.9	0.006	5.8
Site 5 (n = 12)	5.3	0.011	10.6
Site 6 (n = 17)	7.5	0.015	15.0
Site 7 (n = 23)	6.5	0.013	13.0
Site 8 (n = 58)	5.5	0.011	11.0
Site 12 (n = 8)	7.2	0.014	14.4
Site 13 (n = 18)	3.3	0.007	6.6
Site 14 (n = 15)	6.1	0.012	12.2
Avg. (Bank)	**6.1**	**0.012**	**12.2**
Std. Dev. (Bank)	**2.0**	**0.004**	**4.1**

Water Extractable Nitrogen

Table G shows the results of a water extraction experiment employed on stream bank sediments at BSR (Site NYT). This experiment was designed to compare the viability of different water solutions (low-NO3 surface and groundwater, and de-ionized water) for extracting reactive NO_3 from stream bank sediments. Duplicate KCl extractions yielded NO_3 concentrations of 2.5 mg/kg and 3.5 mg/kg, which are identical within 2 sigma analytical error (+/- 1.0 mg/kg). The three water extraction solutions yield NO_3 concentrations of 3.9, 3.0 and 3.1 mg/kg for surface water, groundwater, and de-ionized water, respectively, which also are identical within 2 sigma analytical error (+/- 1.0 mg/kg).

The range in concentrations measured by water extraction (3.0 to 3.9 mg/kg) encompasses the range measured by the KCl extraction method (2.9 to 3.5 mg/kg), suggesting that this series of stream bank sediments release the same concentrations of nitrate-N whether extracted by KCl, surface water, groundwater or di-ionized water. This experiment demonstrates that water is as efficient as KCl for extracting reactive NO_3 from stream bank sediments, and that different compositions of water have no net effect on the release of reactive NO_3-N from the bank sediments.

Table G Water Extractable Nitrate-N versus KCl Extractions in Stream Bank Sediments at Big Spring Run

Sediment Type	NO3k (mg/kg) KCl	NO3k (mg/kg) KCl	NO3sw (mg/kg) Surface Water	NO3gw (mg/kg) Ground Water	NO3dw (mg/kg) DI Water
Stream Bank Site NYT (n=20)	2.9	3.5	3.9	3.0	3.1

Discussion: Nutrient Limited vs. Nutrient Saturated Soils and Sediments

There is mounting evidence that soil organic matter (especially soil microbes) possess an optimum C:N:P ratio, similar in concept -- but different in value -- to the C:N:P stoichiometry of marine phytoplankton (e.g., the Redfield Ratio). Similar to marine phytoplankton, element concentrations of individual phylogenetic groups within the soil microbial community may vary, but on average, atomic C:N:P ratios in both soil (186:13:1, derived from C:N = 14.3; C:P = 186; and N:P = 13.1) and the soil microbial biomass (60:7:1, derived from C:N = 8.6; C:P = 59.5; and N:P = 6.9) are well-constrained at the global scale (Cleveland and Liptzin, 2007), suggesting that close interactions between organisms and the environment drive the observed similarities in their element ratios. It is interesting to note that upland soils at BSR possess atomic C:N:P ratios of 32:3:1 (using TPx) and 47:4:1

(using TPi), whereas legacy sediments at BSR possess atomic C:N:P ratios of 42:3:1 (using TPx) and 67:5:1 (using TPi).

The measured C:N:P ratios at BSR for soils and legacy sediments are similar to each other, and they are similar to the global average C:N:P ratio of 60:7:1 for soil microbial biomass determined by Cleveland and Liptzin (2007), suggesting that carbon, nitrogen and phosphorus measured in soils and legacy sediments at BSR are governed by microbial biomass stoichiometry. Soils and legacy sediment C:N:P ratios at BSR are, however, substantially less that observed for global soils, suggesting that BSR soils are P limited, and that N and C are relatively enriched. The relatively strict nutrient requirements of the soil microbial biomass—combined with the relative P-poor status of some soils—provides on explanation for the observation that P often limits both microbial biomass and activity in these soil ecosystems (e.g., Gallardo and Schlesinger 1994; Cleveland et al. 2002; Cleveland and Townsend 2006).

Fixed C:P and N:P ratios in soil are surprising. In contrast to total soil C and N, weathering of primary rock minerals provides the dominant source of total P in terrestrial ecosystems (Walker and Syers 1976; Chadwick et al. 1999), which is observed also for the soils at BSR (see Tables A and B, which show that weathered limestone saprolite contributes up to 100% of the sorbed P). Even though organisms may not directly regulate total soil P, total soil P ultimately influences the amount of biologically active P that is available for plant productivity, thus indirectly linking the abundance of total P to the abundances of total C and N in soil (Cleveland and Liptzin, 2007).

Nutrient limitation is a term that indicates when N or P deviate from an ideal ecological stoichiometry, and that the system in question has a stoichiometric excess or depletion in one nutrient over the other. For example, phosphorus limitation occurs when there is proportionally less phosphorus than nitrogen required to meet the nutrient demands of the primary producers in the ecosystem (e.g., phytoplankton in marine environments and microbial communities in soils). Phosphorus limitation in surface water occurs in some locations in the spring season when abundant nitrogen is available from stormwater flow. Nitrogen limitation occurs when there is proportionally less nitrogen than phosphorus (i.e. excess phosphorus). Nitrogen limitation often happens in the summer and fall when stormwater flows are lower (so less nitrogen is being added to the water) and some of the nitrogen has been used up by phytoplankton growth during the spring.

Conclusions -- Nitrogen

The stream banks at BSR contain an average of 1441 mg-N/kg of total nitrogen (equivalent to a load of 2.9 lbs-N/ton and a yield of 2881 lbs-N/ac) and an average of 6.1 mg-NO_3/kg of nitrate-N (equivalent a load of 0.012 lbs-NO_3/ton and yield of 12.2 lbs-NO_3/ac). The average nitrate pool is thus roughly 0.4 % of the total nitrogen pool, confirming that the reactive N pool is a small fraction of the total N pool, and suggesting (along with the correspondence between TC and TN in chemo stratigraphic sections) that most of the total N pool is locked up in recalcitrant organic molecules. However, it also suggests that under the appropriate redox conditions that microbial nitrification processes have a potentially large pool of N from which to produce reactive forms of N over a long period of time (c.f., Weitzman, 2011).

Roughly 21,955 tons of stream bank sediment was removed during the restoration at BSR. This equates to a permanent removal of 63,670 pounds of total nitrogen, and 263 pounds of nitrate-N from the restoration reach of the BSR watershed.

C. Deliverable 2: Monitor surface water and shallow ground water and quantify sediment and nutrient loads at BSR. (Galeone/Langland/Walter/Merritts)

C. 1a. Surface water sediment loads

All discharge, turbidity, and suspended sediment load data are summarized in a report from USGS scientists in Appendix 5. Figure C0 is a map of the study site and identifies the locations of each gage station. Details regarding the locations, upstream drainage areas, and USGS station numbers are presented in the table below.

USGS 01576516 -- Eastern tributary, informally named the Sweeney gage Big Spring Run Tributary near Willow Street, PA http://waterdata.usgs.gov/usa/nwis/uv?01576516 Latitude 39°59'29.56", Longitude 76°15'39.35" NAD83 Lancaster County, Pennsylvania, Hydrologic Unit 02050306 Drainage area: 0.36 square miles Datum of gage: 315 feet above NGVD29. Also referred to as # 015765159.
USGS 015765185 -- Western tributary, informally named the Fry gage Unnamed Tributary to Big Spring Run near Willow Street PA http://waterdata.usgs.gov/usa/nwis/uv?015765185 Latitude 39°59'28.29", Longitude 76°15'50.23" NAD83 Lancaster County, Pennsylvania, Hydrologic Unit 02050306 Drainage area: 1.05 square miles Datum of gage: 325 feet above NGVD29. Also referred to as #015765184.
USGS 015765195 -- Main stem, downstream, informally named the Keener gage. Big Spring Run near Mylin Corners, PA http://waterdata.usgs.gov/pa/nwis/uv?site_no=015765195 STATION.--015765195 BIG SPRING RUN NEAR MYLIN CORNERS, PA LOCATION.--Lat 39`59'45.37", long 76`15'50.54", Lancaster County, Hydrologic Unit 02050306. DRAINAGE AREA.--1.68 mi2. GAGE.--Water-stage recorder. Elevation of gage is 315 ft above National Geodetic Vertical Datum of 1929, from topographic map. REMARKS.--Satellite telemetry at station.

Each station is essentially a sampling point we measure water quality parameters, and its suspended sediment and nutrients. The upstream gages/sampling points measure incoming discharge and sediment and are identified as the Sweeney and Fry gages. The downstream gage/sampling point is referred to as the Keener gage and this gage measures outgoing discharge and sediment parameters on the main stem of Big Spring Run below the restoration site.

Suspended sediment sampling during both base flow and storm flow events are used to construct a calibration curve for the correlation between turbidity and suspended sediment concentration. The product of suspended sediment concentration and discharge provides the estimate of load. Load values are presented as daily values, and from these values the annual loads are calculated by summing all daily values.

Daily values for discharge, sediment load, and the ratio of the two (sediment load per unit discharge) are shown in Figure C1 for the Keener gage (main stem, downstream of restoration reach). Note that the y-axis is logarithmic on all graphs in Figure C2.

After restoration, which began approximately at day 1065 (September 1, 2011) on the x-axis and ended on approximately day 1126 on December 1, 2011, a marked reduction in sediment load occurred. A marked reduction occurred in May-June 2012, about days 1250-1300. The marked reduction occurred during the time that emergent wetland and submerged aquatic vascular plants grew vigorously during spring to early summer. When shown as load per unit discharge (bottom chart in Figure C1), the order of magnitude reduction after vegetation increase is prominent.

Daily sediment loads for all gages are shown in Figure C2. Whereas the sediment loads decreased after restoration at the downstream Keener gage, at the upstream gages--which represent sediment flux into the restoration reach—daily sediment load was increasing during the same time period. This is especially evident in the middle graph of Figure C, which illustrates the sum of sediment loads from the two upstream gages in comparison with the sediment load at the downstream gage.

Nine files present all the data from 2008-2012. The file names and brief descriptions are as follows:

1) Big_Spring_Sed_Loads_new_2009.xls
2) Big_Spring_Sed_Loads_new_2010.xls
3) Big_Spring_Sed_Loads_new_2012.xls
4) Big_Spring_Sed_Loads_new_2013.xls
5) BS_Q_T_2009.xlsx
6) BS_Q_T_2010.xlsx
7) BS_Q_T_2011.xlsx
8) BS_Q_T_2012.xlsx
9) Big_Spring_Run_Monitoring_Summary_2012.doc

C. 2. Nutrients in Surface Water and Groundwater

Principal EPA Investigators: Paul Mayer, Ken Forshay, Bart Faulkner, USEPA, Office of Research and Development, National Risk Management Research Laboratory, GWERD, Ada, OK 74820 (mayer.paul@epa.gov, forshay.ken@epa.gov, faulkner.bart@epa.gov)
Collaborators: Dorothy Merritts and Robert Walter, Franklin and Marshall College; Dan Galeone, Mike Langland, and Allen Gellis, USGS

Background: Excess sediments and anthropogenic nutrients, especially nitrogen (N) and phosphorous (P), are leading causes of water quality impairment in streams and wetlands throughout the Mid-Atlantic Region of the US. Legacy sediments, deposited as a function of historic mill dam construction, may contribute significantly to the sediment and nutrient load of streams and estuaries including the Chesapeake Bay. Removing legacy sediments may be a cost-effective, sustainable means to reduce sediment and nutrient pollution in watersheds. Therefore, identifying Best Management Practices for streams and wetlands to mitigate the impacts of legacy sediments is an important goal for resource managers in the Mid-Atlantic Region. Big Spring Run (BSR), a rural stream in Lancaster Co, PA is impacted by legacy sediments from past mill pond dams. BSR has been the subject of long-term nutrient and sediment studies (Galeone et al. 2006; Walter and Merritts 2008).

In the Fall 2011, legacy sediments were removed throughout a portion of the BSR watershed to expose buried wetlands and reconnect floodplain hydrology. This restoration effort represents a unique opportunity to assess the effects of watershed restoration on ecological function in a watershed, especially sediment and nutrient reduction.

Objectives: 1) Assess ecosystem benefits of restoration; 2) Identify stream restoration methods that enhance nitrogen control; 3) Develop predictive models of stream hydrology and sediment movement; 4) Develop ecologically-based guidelines for stream restoration.

Approach: Examine BSR before and after restoration to measure surface and ground water hydrology, nutrient dynamics, and microbial denitrification, a natural subsurface process that removes bioreactive nitrogen by transformation to a biologically inactive gas form. Employ isotope tracer techniques such as membrane inlet mass spectrometry (MIMS) and stable isotopes of ^{18}O of nitrate to quantify denitrification and determine nitrogen source. Establish stream flow gages; characterize stream geomorphology; track sediment movement; monitor surface water and ground water chemistry, and measure ground water level, temperature, and hydraulic head among a network of piezometers established throughout the restoration and in control locations. Construct mass balance models of nitrate fate and transport in the watershed.

Expected Results: Based on previous studies, geomorphic stability of restored streams may be greatly improved after restoration; far less sediment is transported and lateral migration of streams are halted. Significant denitrification activity occurs in the stream channel and hyporheic zone, especially where carbon concentration is high and the stream is connected with the floodplain. Thus, not only do we expect the source of sediments and

nutrients to be removed after restoration, but we expect more bioreactive nitrogen to be removed in the stream channel and associated floodplain wetlands due to better hyporheic connection and retention, and increased organic matter supply. We expect that restoration involving legacy sediment removal will be a sustainable means of improving water quality in watersheds.

Results

Appendix 5 includes a link to the compilation of surface water and groundwater data collected and analyzed during the pre-restoration phase, between 2008 and 2011, and for surface water only during the post-restoration phase from 2012 to 2013. The first set of analyses in 2008 were collected and analyzed by the USGS; all subsequent water quality data, 2008-2013, were collected by both USGS and the USEPA and all of these samples were analyzed by the USEPA in the National Groundwater Laboratory in Ada, OK.

Surface water samples were collected at regular intervals by the USGS at each of the three gage stations at the BSR study site (Figure C0). In addition, the USGS collected groundwater samples from eighteen shallow piezometers that were installed in 2008. The piezometers were installed in a nested configuration of three that included one within the channel and one on each side of the channel (Figure C3). Subsequently, the US EPA installed shallow groundwater wells both on the upland hill slopes adjacent to the restoration reach and within the restored area(Figure C3).

Due to construction delays for the restoration project, funding for surface water and groundwater expired in 2010. When funding expired the USGS collaborators discontinued groundwater and surface water sampling. Supplemental funds provided by a grant from the National Science Foundation included support for the USGS to continue operating the gages to acquire hydrological and sediment load data, but these funds did not extend to water quality sampling. Therefore, we lack critical post-restoration water quality data to calculate changes and estimate the predicted nutrient load reductions.

New grant funds from the US EPA has provided resources necessary for USGS to renew their water quality sampling efforts and analyses. As before, US EPA is committed to these efforts and is continuing to analyze all groundwater and surface water samples collected by USGS.

D. Deliverable 3: Identify sources of suspended sediment (upland vs. stream corridor) in stream water via geochemical fingerprinting. (Gellis/Walter/Rahnis)

I. Background and Previous Work

In order to successfully implement strategies to mitigate sediment delivery to the Chesapeake Bay, resource managers must be able to confidently apportion sources that contribute suspended sediment to the Bay and its tributaries. Traditional methods for identifying suspended sediment sources involves one or more of the following: (1) aerial photographic interpretations of landscape and land-use change; (2) field-based erosion studies at the plot or stream bank scale; and (3) the application of mathematical models to "scale-up" the field-based studies. Walling (2005) points out that while traditional tools such as aerial photography, erosion pins, and erosion plots can document bank loss and sediment mobilization, these methods cannot accurately quantify which source is actually contributing to the stream network.

Sediment fingerprinting uses geochemical properties inherent in soil and sediment particles, and it offers a direct means to quantify suspended-sediment sources. In this approach, identifiable geochemical properties are used to characterize and uniquely "fingerprint" sediment sources: "uniqueness" is usually defined statistically. Such sediment fingerprint techniques are analogous to well-known geochemical fingerprinting methods that have been applied for decades to study sources and distributions of volcanic ash (tephra) in the sedimentary record (c.f., Westgate et al., 1985: Walter et al., 1987).

Suspended sediments collected under a variety of flow conditions contain a host of source material properties that, taken together, create a characteristic "fingerprint" that reflects the relative contribution of each sediment source to the suspended sediment load during any particular flow event (Collins and Walling, 2002; Motha and others, 2003; Walling, 2005; Gellis and Landwehr, 2006). The most common sediment sources are associated with erosion from agriculture fields, pastures, forested hill slopes, channel banks and beds, and drainage ditches, and to a lesser extent construction sites, dirt roads, and urban lawns. Classification of source types for most studies usually involves a simple distinction between upslope (hill slope) areas (sediment mobilized by sheet and rill erosion) and sediment mobilized from the channel system by channel erosion (stream bank erosion) (Walling and Woodward, 1995; Collins et al., 2001; Gellis et al., 2009; Gellis and Walling, 2011). Based on the application of sediment fingerprint studies, it is becoming clear that bank erosion can be a major contributor to the suspended sediment load (Banks et al., 2010; Mukundan et al., 2010; Devereux et al., 2010; Massoudieh et al., 2012).

Land-Use Change in the Big Spring Run Watershed

A previous land-use change study integrating historical research, personal interviews of landowners, GIS database development, and aerial photograph analysis (1940-2005), and the application of the Revised Universal Soil Loss Equation (RUSLE) investigated how upland erosion patterns and rates have changed in the Big Spring Run watershed (Sullivan

2006; Walter et al., 2006). These results suggest that upland erosion in the agricultural fields around Big Spring Run has been decreasing since 1940, and has stabilized since the 1970s to rates below 5 tons per acre per year. These relatively low and apparently sustainability upland soil erosion rates are attributed to increased soil conservation practices in the watershed (Figure D1), including contour plowing that began in the 1950s and no-till practices that were implemented in the BSR restoration reach watershed in the 1990s (Joe Sweeney, personal communication).

^{137}Cs Inventory of Soil Erosion in the Big Spring Run Watershed

Previous studies (Sullivan 2006; Walter et al., 2006) used a ^{137}Cs inventory to document the relative contributions of sediment from two main landscape sources at BSR: (1) upland soil erosion from agricultural fields; and (2) stream bank erosion in valley bottoms. An inventory of fallout ^{137}Cs activity from two hill slope transects adjacent to Big Spring Run yield average post-1963 erosion rates of 1.8 t/ha/yr (3.9 t/acre/yr) and 0.3 t/ha/yr (0.7 t/acre/yr) (Figure D2), both of which are significantly less than the presumed average county-wide erosion rate of 4 t/ha/yr (8 t/acre/yr). These values are consistent with our calculation of erosion rates in the watershed using the revised universal soil loss equation, and from our GIS interpretation of land use change in the watershed from aerial photographs flown over the past 60 years (#1 above). This ^{137}Cs study indicates a reduction in soil erosion rates from ca. 25 t/acre/yr in 1940 to ca. 5 t/acre/yr in 1988, and which remained under 5 t/acre/yr to 2005, when that study ended.

In addition, the average contribution of sediment supplied to Big Spring Run from bank erosion can be deduced using mass balance calculations of the ^{137}Cs data (Walter et al., 2006). These calculations show that ~30-65% of the sediment supplied to this watershed can be attributed to bank erosion (Figure D3). These values reflect a minimum estimate of the contributions from bank erosion, as the stream banks themselves contain an appreciable amount of ^{137}Cs in the upper ca. 30 cm of the stream bank section(Figure D4a-c). Correcting for the contribution of ^{137}Cs in stream banks increases the relative contribution of stream bank erosion at BSR to between 50 and 80%.

II. Trace Element Geochemical Fingerprinting of Upland Soil and Stream Bank Sediments in the Big Spring Run Watershed

The purpose of this study is to determine the sources of suspended sediment loads in the Big Spring Run watershed. To document sediment sources, geochemical analyses were performed using x-ray fluorescent spectrometry (XRF) and inductively coupled plasma-optical emission spectroscopy (ICP-OES) on the silt and clay fraction of stream banks (Figure D4a and D14), upland sample sites (Figure D9), and suspended sediment samples collected during storms (via USGS gage stations: Figure D10) or from flood deposits on inset point bars after storms (suspended sediment proxies) Figure D23.

III. Results

Test of Methodology – Mill Stream Branch Comparison Study:

The bulk of our trace element geochemistry used for this sediment fingerprint study was obtained by ICP-OES techniques using the EPA 3051 partial dissolution method to prepare the samples (Figure D6 and Appendix 4). The EPA 3051 partial dissolution method was used initially by us to measure the concentrations of phosphorus sorbed onto particle surfaces, while at the same time and using the same dilution we obtained a host or trace elements that are also sorbed onto particle surfaces. We opted to continue using the ICP-OES/EPA-3051 method for our trace element fingerprinting study for the following reasons: (1) we had already acquired hundreds of soil and sediment analyses from BSR using the method for our sorbed-P study; and (2) the ICP-OES technique requires only 0.25 g of sample mass to be digested for analysis. In comparison, the XRF trace element method requires 8 to 15 times the sample mass (4-7 grams) as ICP-OES. The issue of sample mass is important when analyzing suspended sediment samples collected from the USGS gage stations at BSR. It is typical that the mass of suspended sediment collected per storm event is much less than one gram, making the application of XRF impractical.

Despite our reliance on the ICP-OES method, to the best of our knowledge, the EPA 3051 partial dissolution method has never been used for a sediment fingerprint study. Therefore, as part of Deliverable 3, we determined to test this methodology by comparing our ICP-OES/EPA-3051 method to well-established and published sediment fingerprinting results. We chose for our test to analyze samples from the Mill Stream Branch, in the Maryland coastal plain (Figure D5), which has been thoroughly investigated for its sediment source properties by the US Geological Survey (Banks et al., 2010; Maussdieh et al., 2012).

Outline of Methodology

(a) Samples of upland Crop soils (CR), Forest soils (FR) and Stream Corridor (e.g., Stream Bank) sediments (SC) from the Mill Stream Branch (Maryland) were analyzed by ICP-MS at USGS (using a total digestion method: c.f., Banks et al., 2010 for summary of USGS results and sample nomenclature) and subsequently by ICP-OES at F&M (using the EPA 3051 partial dissolution method);

(b) Results from both ICP-MS (USGS) and ICP-OES (F&M) were compared with trace element concentrations measured on suspended sediment (fluvial) samples (FL) collected at the downstream end of the Mill Stream Branch study reach;

(c) The objective of this study was to determine the relative contributions of the three potential source areas, CR, FR and SC, to the FL suspended sediment load using trace element geochemical fingerprinting, and to determine if the F&M ICP-OES methods yielded similar results as those obtained by the USGS ICP-MS methods (Figure D7).

Summary of USGS ICP-OES Results for Mill Stream Branch (MSB)

The trace element compositions measured by the USGS were subsequently analyzed by two different statistical tests used to determine relative source contributions; the results of an inverse mixing modeling approach were published by Banks et al., 2009, and the results of a Bayesian Inference statistical approach was published by Massoudieh et al., 2012. Both statistical tests (Banks et al., 2010; Massoudieh et al., 2012) indicate that stream bank erosion along Mill Stream Branch (SC) accounts for 100% of the suspended sediment load in all six fluvial storm samples collected.

Summary of F&M ICP-OES Results for Mill Stream Branch (MSB)

As a test of methodology, we requested that the USGS send splits of the same MSB samples analyzed by Banks et al., 2009 and Massoudieh et al., 2012, which then were prepared at F&M using the EPA 3051 method and analyzed by ICP-OES, with the aim of comparing F&M's fingerprint methods against the published results cited in the USGS studies (Bank et al., 2010: Massoudieh et al., 2012). If the F&M and USGS fingerprinting studies agree, it would add a high level of confidence and quality assurance to the F&M methodology, and suggest that the EPA 3051 protocol is reliable for suspended sediment fingerprinting analyses.

Analyses by the F&M method (partial dissolution + ICP-OES) indicates that bank erosion accounts for 93-100% of the suspended sediment load at MSB, comparable to the results derived by the USGS method (Figures D8).

Conclusions of Mill Stream Branch Comparison Study

In summary, the F&M ICP-OES results yield essentially identical results as presented in Banks et al 2005 and Massoudieh et al 2012, that stream banks contribute to ca. 100% of the suspended sediment load, despite the fact that different sample preparation methods and analytical methods were used.

This test of methodology demonstrates that the sample preparation and analytical methods employed at F&M and applied to Big Spring Run are suitable for sediment fingerprinting studies, which adds a high level of confidence to the results for BSR outline below.

Big Spring Run Sediment Fingerprinting Results

Figure D9 shows the location of the BSR sediment fingerprint study, Figures D10a-b show the stream bank and upland soil sample locations used in this study, and Figure D11 shows probable pathways for upland soil erosion in the watershed. Figures D12-D15 show sampling locations and methods for representative upland (Figures D11) and stream bank (Figures D12 and D15) sections. Over one thousand analyses were incorporated into this study, representing an even distribution between upland, stream bank, and suspended sediment samples. Samples from adjacent roads were sampled, and they were determined

to have no contribution to the suspended sediment load and will not be considered further here.

Visual inspection of dot and bivariate plots (e.g., Figure D16) and the application of multivariate statistics to an array of trace element data (Figures D17 and D18) demonstrate that stream bank sediments are geochemically similar to suspended sediment samples at Big Spring Run. Applying the same mixing model that we used to demonstrate the similarity between USGS and F&M results for the Mill Stream Branch sediment fingerprint study, we find that 60-70% of the suspended sediment load sampled at the downstream gage below the restoration reach was derived from stream bank erosion (Figures D19 and D20). Compositional data for cobalt (Co) and phosphorus (P), when plotted on a bivariate diagram, demonstrate that the fluvial suspended sediment load at the downstream (Keener) gage at BSR is a mixture of mean upland and mean stream bank sediments (Figure D19), and that compositional data measured on the suspended sediments fall on a mixing line between these two end-member components. Further, the mean value of the plotted Co vs. P compositional data lies along the mixing line at a value that is equivalent to a ratio of 63% stream bank sediment to 37% upland soil.

Conducting the same mixing model calculations for the fluvial sediments collected at the East Branch gage (Sweeney) and West Branch gage (Fry), we estimate that stream bank sediments contribute to 33% of the East Branch (Sweeney) gage sediments and 54% of the West Branch (Fry) gage sediments (Figure D21). These results indicate that the two tributaries within the study area are sampling a larger fraction of upland soils than is sampled between the upstream and downstream gages. In other words, a simple mixing of suspended sediment entering from the two upstream gage tributaries, we expect to see an average of 43% contribution from bank erosion. The fact that we see a 63% stream bank contribution at the downstream Keener gage means--from mass balance calculations--that bank erosion between the upstream and downstream gages must be between ca. 80-90%.

A sediment fingerprint study of sediments collected from tile pads within the BSR study area (Figure D22), indicates that 80-100% of the deposition on these pads is derived from stream bank sediments. This result is compatible with our mass balance calculations noted above that 80-90% of the suspended sediment load between the upstream and downstream gages must be from stream bank erosion. Furthermore, this tile pad study revealed that there was no deposition on the floodplain legacy sediment surface during the three years that the tile pad study was conducted (2009-2011). Instead, deposition occurred only on the inset point bars, which are topographically lower than the "floodplain" surface. These inset point bar features formed as a result of base level lowering associated with the removal a milldam and/or other downstream grade control such as a culvert or road crossing. Subsequent incision into stored millpond sediment results in the construction of a meandering stream system, with actively eroding stream banks composed mostly of legacy sediments, and inset point bars composed mostly of gravels and sands, but which fine upward to silts and clays (Figure D23). This observation highlights the textural disconnect between actively eroding stream banks (Figure D15) that are composed of silt loams, and the accretion of inset point bars that are made up of mostly gravels and sand (Figure D23).

After Tropical Storm Hanna in September 2008, we observed large blocks of silt loam stream bank sediments that broke free of the bank surface and collapsed into the stream (Figure D16). We continued to observe these slump blocks (> 1 ton each), and noted that they took more than a year to winnow away. These observations lead us to the following conclusions: (1) stream bank erosion by slumping is a highly stochastic process related to infrequent large storms; (2) large slump blocks can remain in the channel for more than a year; and (3) the storage of silt in the channel is temporary, and that it eventually works through the system.

Conclusions: Big Spring Run Sediment Fingerprint Study

Our results indicate that the primary source of suspended sediment at Big Spring Run is stream bank sediment erosion, which contributes at least 63% of the suspended sediment load sampled at the downstream (Keener) gage.

Summary of Sediment Fingerprint Results

Gage Station	Percent Load from Stream Bank Erosion
East Branch (Sweeney Gage)	33%
West Branch (Fry Gage)	54%
Main Stem (Keener Gage)	63%

This interpretation is somewhat complicated by a number of factors: BSR is a small headwater tributary (e.g., the watershed area above the restoration reach is less that 5 mi^2) and that the stream bank deposits themselves (e.g., legacy sediments) were derived from erosion and deposition of soils from hill slopes directly adjacent to the study area. Therefore, given the close proximity of stream bank sediments to their source soils, the two potential source groups (uplands and stream banks) are geologically and geochemically similar.

Despite these complications, we are confident in our analyses and interpretation that stream bank erosion is a major source of suspended sediment to Big Spring Run, for the following key reasons: (1) The results of our comprehensive trace element fingerprint study, indicating that >60% of the suspended sediment load is from bank erosion, is consistent with the estimates obtained in an earlier ^{137}Cs study (Walter et al., 2006; Figure D4a); and (2) In our comparison study of the Mill Stream Branch, we obtained comparable results as those acquired by the US Geological Survey, using a substantially different methodology. This establishes an independent check on our methodology, and suggests that the trace element fingerprint analysis at BSR is robust.

E. Deliverable 4: Quantify rates of stream bank erosion, stream corridor deposition, sediment storage in the stream corridor, and soil erosion from uplands (sediment budget) (Gellis/Merritts/Rahnis/F&M Staff)

E. 1. Rates of stream bank erosion pre-restoration: Topographic surveys and bank erosion pins

The information needed to evaluate the contribution of stream bank erosion to total suspended sediment load in a sediment budget analysis for the restoration reach at BSR includes the following:

1) Total suspended sediment load entering the study reach (i.e., *Flux In*);

2) Total suspended sediment load exiting the reach (i.e., *Flux Out*);

3) Rate of bank erosion; and

4) Rate of deposition (storage) within the study reach.

The first two fluxes were measured at 3 USGS stream gage stations (the "Fry" gage on the western tributary, the "Sweeney" gage on the eastern tributary, and the "Keener" gage on the main stem; see Figure A1 for gage locations). At all three stations, the USGS has collected continuous data on turbidity from October, 2008, to present, and during storm flow has sampled stream water in order to determine the flux of fine sediment into and out of the study reach. No tributaries enter the 1.5-km reach between these gage stations (i.e., the restoration reach).

Topographic Surveys

The third contributor to the sediment budget, rate of bank erosion, was determined by repeat topographic surveying over a 5.5-year duration, from 2004-2010 (Figure E1). Twelve cross sections perpendicular to channel flow were installed and surveyed along the study reach with a total geodetic station in 2004. These were surveyed with a total geodetic station again in 2006, 2008, and 2009. Three additional cross sections, XS's 13-15, were installed with a laser level by A. Gellis, D. Merritts, and M. Rahnis in 2008. All cross sections were surveyed with a real-time kinematic (RTK) GPS survey unit in 2010. (Note: Post-restoration surveys are discussed below).

Results of repeat channel surveys for the 15 cross sections are shown in Figures E2-E6. Areas of erosion and deposition are identified on these cross sections. Numbers along horizontal axis above distance markings indicate polygon numbers. Each polygon is an area of erosion or deposition, with eroded areas shown as hachures and depositional areas shown as shading. Our repeat surveying of the cross sections yielded rates of channel-normalized sediment production from lateral bank erosion that ranged from 0.04 to 0.27 m/yr between 2004 and 2010.

The fourth contributor to the sediment budget, rate of deposition, was determined in two ways: first, by repeat topographic surveying (i.e., the 12 cross sections), and second, by taking the difference between #3, #2 and #1, as discussed below.

Bank Erosion Pins:

In 2008–2009, we installed 20 sets of erosion pins to monitor bank erosion, with three to four pins aligned vertically on the bank at each site. The pin data yield lateral bank-erosion rates that ranged from 0.05 to 0.64 m/yr during the period of 2008–2010. The equivalent rates of channel-normalized sediment production are 0.04–0.65 m/yr.

The bank pins were useful to observe the spatial and temporal variability (seasonality in the latter case) of bank erosion, but have multiple limitations for estimating the contribution of fine sediment to the stream from bank erosion. The major limitation is that they are not useful for monitoring deposition at point bars. They do provide an estimate of maximum rates of bank erosion, and the range in rates of bank erosion throughout the investigation area. We found, however, that the swelling and shrinking of banks during repeated winter freeze-thaw sometimes extruded the pins, leading to pin measurement errors. Videos of needle ice and freeze-thaw processes can be viewed at

http://www.youtube.com/channel/UCorgwKIsH03jLRuTSF3Wxzg/about
(See also Appendix 6)

At two locations (referred to as the "Type Locality" and the "Houser Grid" sites) a grid of pins was installed to more closely monitor bank erosion. The Type Locality site had seven columns of pins with three pins in each column (see Figure A7 for the location of the "Type Locality"). The columns and pins were installed at equally spaced, regular intervals (~1 m apart). At the Houser Grid site, located ~300 m upstream of the study reach on the eastern tributary of BSR, five equally spaced columns of pins were installed across the bank face (~1 m apart) and the pins within each column were spaced equidistant vertically on the face (Figure E7).

The bank pins at Big Spring Run reveal that more lateral bank erosion occurs in the winter than in other seasons (Merritts et al, 2011, 2013). This same phenomenon was observed in the 1950s at Watts Branch in Maryland (Wolman, 1959), and in studies of stream banks in the UK (Lawler, 1986). Detailed monitoring of sites along the Ilston River, UK (Lawler, 1986, 1993) and Strouble Creek, Virginia (Wynn et al, 2008) established that freeze–thaw processes significantly lower the critical shear strength and increase the erodibility of cohesive stream-bank sediment. We observed freeze–thaw processes and needle ice in the stream banks of Big Spring Run during the winters of 2008–2009 and 2009–2010. An apron of debris from freeze–thaw processes accumulated during winter months; occasional high-water events notch into this apron (see Figure E7), and typically higher spring-summer stormflow events removed the apron.

In addition to the role of freeze–thaw in bank erosion, we noted that bank slumping and calving are frequent and typically occur immediately after a rise and fall in stage. Other workers have noted that high stages cause banks to be wetted, and the rapid drop in pore pressure in wet banks after a high-stage event is conducive to failure, particularly in banks composed of large amounts of silt and clay (Simon et al, 2000). For Big Spring, we documented that such failure occurred throughout the year, but was especially notable in the autumn during long-duration high-flow events that wetted the banks over a period of days. For stream banks consisting of more than 40 per cent silt and clay, as is the case at Big Spring Run (and all other sites discussed here), flow duration is more important than flow depth in bank erosion (Julian and Torres, 2006).

During summer months, we observed that the stream banks desiccate and fracture (see panoramas in Appendix 8 figures). Grasses and other vegetation growing on the fill terrace accelerate stream-bank desiccation as the growing season progresses. Summer drying and fracturing prime the banks for failure during the autumn, when the decline in vegetation activity and the increased frequency of storms in the mid-Atlantic region increase bank moisture content.

E. 2. Analysis of bank erosion and its contribution to suspended sediment load

To calculate the load of fine sediment from bank erosion, we used the following procedure:

1) Survey cross sections over time, with longer periods better for estimating long-term average rates of erosion and deposition (see Figures E2-E6).

2) Map bank types, identifying those with erosion, deposition, and no change (Figures E8-E11). Determine length of each bank type in study reach.

3) Measure grain size and bulk density for each bank type and determine the % fine sediment (clay, silt, very fine sand, and fine sand) in each that contributes to suspended sediment load (Figures E12-13). We use this range in sizes because that is the range of the suspended sediment sampled at the USGS gage stations (see file name “Grain Size Analysis DEP Report”). Note that nearly all sediment that is historic fill (Bank Types 1a, 1b, and 2) is fine sand or finer, and hence able to be transported as suspended sediment.

4) Identify areas of erosion and deposition on cross sections (see polygons identified along top of x-axis in Figures E2-E6).

5) Calculate net sediment change in fine sediment from all polygons at all cross sections (Figure E14).

6) Sum all data for erosion and deposition at each polygon and calculate net change in fine sediment from the stream corridor in the entire study area (Figure E15).

Bank Type	Length, ft	Bank erosion total, lbs	Deposition total, lbs	Average annual bank erosion, lbs	Average annual deposition, lbs
1	3123	-1561290	364761	-283871	66320
2	3261	-161663	228593	-29393	41562
3	2201	-84839	214028	-15425	38914

Using data from the table above, the total mass of fine sediment (fine sand or finer) eroded on an annual basis (average of 5.5 years of data) is 164.3 tons. The total mass deposition on an annual basis (average of 5.5 years of data) is 73.4 tons. The difference between these is net change from the restoration reach, and is 90.9 tons of erosion of fine sediment, per year (on average during 5.5 years of monitoring).

E. 3. Post-restoration topographic surveys

All but one of the 15 cross sections is re-surveyed at least once a year to monitor post-restoration change (Figures E16-22). Cross sections 8 and 9 were closely spaced and redundant, so we discontinued XS 9. Since 2010, surveys have been done with RTK GPS. All but XS's 1 and 2, which are in the area that was not restored, were extended after restoration. Coordinate points for all surveys, including end points, are provided at an on-line repository:

https://github.com/mrahnis/orangery

E. 4. Analysis of net loss of fine sediment from bank erosion

In this section, we compare the ~91 tons of fine sediment contributed from bank erosion (previous section, E2) with that determined from analysis of suspended load data at the USGS gage stations.

The flux out of the restoration reach, measured at the Keener gage, is referred to as *Flux Out*, or just *Out*. The flux into the restoration reach is measured at both the Sweeney and Fry gages, and is referred to as *Flux In,* or just *In*. The following net balance equation shows the relation between these fluxes, bank erosion, and deposition:

Flux Out = Flux In + Bank Erosion Contribution – Deposition

(Note: Bed scour and erosion measured during this study were insignificant, and are not considered here.)

For example, if annual load out is 200 tons, annual load in is 100 tons, and annual contribution from bank erosion is 120 tons, then 20 tons of deposition must occur within the reach between the stations measuring flux in and flux out.

The following conditions can be established (Figures E23-26):

- When *Out* – *In* > 0, bank erosion must be occurring.
- When *Out* – *In* < 0, deposition must be occurring.
- When *Out/In* > 1, bank erosion must be occurring.
- When *Out/In* = 1, bank erosion must equal deposition.
- When *Out/In* < 1, deposition must be occurring and net erosion cannot occur. I.e., if local erosion occurs, an equivalent amount of sediment must be deposited before exiting the restoration reach.

Based on these conditions and the data from the gage stations for suspended sediment load (with a measured mean grain size of 16 microns), there has been a significant increase in the # of days when *Out* – *In* < 0 and deposition must be occurring. In addition, the # of days when bank erosion occurs has diminished substantially. After restoration, deposition occurred during 49% of days, whereas prior to restoration the percentage of days in a year with deposition ranged from as low as 2 to as high as 25%. Finally, the % of days in which there was net loss of sediment from erosion in the restoration reach (*Out/In* > 1) decreased from a range of 75-98% from 2009-2011 (pre-restoration) to 51% in 2012, post-restoration.

[Data discussed in this section (E. 2.) are provided in file named "Load Time Series All 3 Gages_DJM_12092013".

E.5. Sediment budget

Data from the previous sections are combined in Figure E27 to show the following:

- Data from 3 USGS gages indicate that an average of 94 tons/yr of fine suspended sediment was contributed from bank erosion during each of the 3 years prior to restoration (2008-09, 2009-10, and 2010-11).
- Our repeat topographic surveying results prior to restoration demonstrate that ~91 tons/year was eroded from stream banks that consisted of nearly 100% fine sediment that could be transported by suspension.
- During the year after restoration began, the sediment flux out of the restoration reach was much less than prior to restoration. It decreased from 3-year pre-restoration average of 218 tons to post-restoration value of 109 tons. More years of

monitoring will be necessary to identify trends with time and to calculate long-term averages for comparison with longer-term averages of pre-restoration data.

- We estimate that not only is bank erosion no longer a source of sediment, hence explaining the reduction from 218 to 109 tons of sediment per year, but also 15 tons of fine sediment deposition is occurring during the first year after restoration began. We determine this value as the difference between the Flux In and the Flux Out from the gage data, as follows:

  Flux in = 124 tons
  Flux out = 109 tons
  Difference = -15 tons

  In sum, the load of sediment from bank erosion within the restoration reach no longer exists, and the restored aquatic ecosystems are trapping sediment. The result of these two factors is that the net loss of sediment is substantially reduced by removal of legacy sediment and restoration of natural valley bottom morphology characteristics and aquatic ecosystems.

F. Deliverable 5: Biological indicators of ecosystem services

F. 1. Vegetation Assessment

Introduction

Vascular plants are perhaps one of the most conspicuous components of a wetland ecosystem and frequently are used to identify wetlands and classify them into discreet categories (Cowardin, 1979; USACE, 1987; Fike, 1999). Plants primarily are immobile and their presence represents a manifestation of temporal, spatial, chemical, physical, and biological attributes of aquatic ecosystems, making them ideal proxies for long term and intensive monitoring efforts. Individual plant species may occupy narrow ecosystem ranges and may reasonably predict ecosystem characteristics or changes that have occurred over time (USEPA, 2002).

Vascular plants are primary producers that affect the energy flow in aquatic ecosystems both directly and indirectly through interactions with other biological components like algae, animals, macroinvertebrates and fish (Mitsch and Gosselink, 2000). Plants may affect water quality through nutrient uptake and sediment trapping and may act as pumps moving nutrients from sediment and groundwater into surface water. Plants also may affect hydrology, particularly during flooding events. There are many links between hydrology and plant community structure and other dynamics (Mitsch and Gosselink, 2000). Likewise, plant communities often are used as indicators for a variety of aquatic ecosystem characteristics (USEPA, 2002).

The pre-restoration vegetation at this site was growing on legacy sediment accumulated within the last ~300 years (Walter and Merritts, 2008). Analysis of pre-restoration floristic and plant community characteristics provides a baseline of vegetation to be compared with post-restoration vegetation over the next several years. We hypothesize that by removing legacy sediment to restore the natural valley morphology, and thereby restore natural hydrologic and hydric soil conditions, we may be able to restore natural wetland plant communities at the Big Spring Run restoration site. Furthermore, we will evaluate the restored plant community to determine if it is representative of a natural palustrine emergent marsh plant community that persisted for millennia at this site in the paleo-environment and prior to European settlement (Neugebauer, 2011; Hilgartner, et al., 2012). Finally, we will use plant communities to determine the temporal recovery of natural wetland plant communities and infer the resulting effects on chemical and physical components of the restored natural ecosystem characteristics.

Methods

We conducted plant cover plot surveys along transects to characterize the vegetation before aquatic ecosystem restoration and repeated these surveys during the first growing season after restoration. Transect lines oriented perpendicular to the fall of the valley and spanning the restoration zones were sampled in July 2009, August 2010, and June 2011 prior to restoration. The transect end points generally were established at fixed locations, like fence posts, for ease of re-establishing each transect during future monitoring and the locations were verified using various GPS units. Transects extended beyond the restoration area and the endpoints generally are located beyond the limits of accumulated legacy sediment (Figure F1). A tape measure was used to establish a linear transect reference between the endpoints and to locate fixed interval plot locations. A square meter frame was used to delineate the boundaries of each sampling plot (Figure F2). A general survey of the transect positions and vegetation was noted as preparation for transect analysis.

The samples included plant species percent cover in one square meter plots at 5 meter intervals along each transect. A control (Transect 1) spanning the valley width of a direct tributary to the restored reach was sampled in August 2010 and May 2013. The control reach remained unrestored with legacy sediment elevations unchanged during the sampling period. Two pre-restoration samples are reported for three transects, including the control. Post-restoration monitoring was conducted during the first growing season following excavation of legacy sediment in August 2012. Because some transects extended beyond the restoration zone for the restored reach, only plots located within the restored zone, where legacy sediment was excavated, were compared to pre-restoration survey data.

The cumulative percent cover for each species was divided by the total number of one square meter plots to derive relative percent cover for each species. Relative frequency for each species was calculated by dividing the individual species frequency by the cumulative frequency of all species. Importance value for each species is represented as the sum of the relative percent cover and relative frequency.

Results

Pre-restoration

A total of 42 species were recorded prior to restoration along three transects (Transects 1, 4 and 6) during August 2010 and June 2011 sampling (Figure F1). The pre-restoration vegetation included a total of 27 upland species comprising a total mean percent cover of approximately 80 percent. All species representing the dominant upland plant cover, except yellow foxtail, have a wetland indicator status of facultative upland (FACU). FACU indicates that the plant rarely is a hydrophyte, and almost always occurs in upland environments.

Prior to restoration, upland grasses and forbs were determined to be the most important plant species. Dominant species included *Poa pratensis* (Kentucky bluegrass), *Festuca elatior* (tall festuca), *Agropyron repens* (quackgrass), *Setaria glauca* (yellow foxtail), *Dactylus glomerata* (orchard grass), *Lolium perenne* (ryegrass), *Secale cereale* (rye) and *Cirsium arvense* (Canada thistle). These species are overwhelmingly non-native and typical of old-fields, dry meadows, pastures and often described as occupying "roadsides and waste places/ground" in the botany literature (Rhoads and Block, 2000; Gleason and Cronquist, 1991). The dominant *Cirsium arvense* is a Pennsylvania noxious weed and Pennsylvania legislation dating from as early as 1862 mandated control of this troublesome plant (Hill, 1983a; PA Dept. of Agriculture, 2013). *Rosa multiflora* (multiflora rose) is another Pennsylvania noxious weed present in the pre-restoration vegetation, although not a dominant species (Hill, 1983b; PA Dept. of Agriculture, 2013).

Prior to restoration, wetland species comprised approximately 27 percent of the pre-restoration plant community along three transects. These wetland species were dominated by *Phalaris arundinacea* (reed canarygrass), with lesser occurrence of *Impatiens capensis* (jewelweed) and *Conium maculatum* (poison hemlock). Nine other species were identified in the wetland component of the pre-restoration plant community. With the exception of reed canarygrass and poison hemlock, the wetland species were growing on or within the banks of the channel incised into legacy sediment along each transect. Reed canarygrass and poison hemlock grew outside of the incised channel boundaries, often in monotypic patches. Ten species identified in the pre-restoration sampling are designated as facultative (FAC) or facultative wetland (FACW). FAC plants commonly occur as either a hydrophyte or non-hydrophyte and FACW plans usually are hydrophytes but occasionally are found in uplands. *Alliaria petiolata* (garlic mustard) is listed as FACU but personal observations (Hilgartner) have led to the conclusion that this species does thrive along riparian and floodplain zones in the Maryland Piedmont, so is listed here under wetland plants.

Acorus calamus (sweetflag) was the one obligate (OBL) wetland species observed in the pre-restoration samplings. This species was isolated only along transect 1 in a wet swale located along the valley margin. The valley margin characteristically is where legacy sediment depth is shallow, much less than the 3-4 foot depth throughout the remainder of the site, and in a location presumably near a spring seep. While no other OBL species were

observed along Transect 1, several OBL species were recorded in another nearby swale during reconnaissance in August 2010.

The cumulative total percent cover of all species equals 107% and appears to represent observation or calculation errors. However the results from cover estimates for species that represented <1% were rounded up to 1% at a given transect. Since "1 percent species" were found among both upland and wetland components, the relative percentages may be represented as 75% upland and 25% wetland species.

We summarize that upland species, primarily pasture/old-field grasses and Canada thistle, comprised 3 times as much ground cover as common riparian or wetland vegetation during the pre-restoration monitoring period. Wetland species occurred only in the immediate channel bank incised into legacy sediment and a wet swale located at the valley margin. Observations of vegetation in other parts of the study area located outside of transects during this pre-restoration monitoring period confirm that upland species were widespread over much greater area of the riparian zone and adjacent farmland. These predominantly upland species included plants on the Pennsylvania noxious plants list.

Post-restoration

Legacy sediment excavation began in early September 2011 and was completed by late November 2011, marking the first stage of site restoration. The re-exposed and restored hydric soils were immediately seeded with a specialized wetland seed mix containing a variety of plant seeds typical of wet meadow palustrine emergent wetland plant communities. Additional wetland seed mix applications occurred during March and April 2012. During the seed application in the spring of 2012, container grown wetland plants(plugs) also were installed throughout the restored areas. The species included in the plugs and specialized seed mix was developed by inference of the natural plant communities from the paleo-seed investigations at this site. Some species present and dominant in the paleo-seed record are not available commercially and were not included in the custom wetland seed mix. In particular, *Carex prasina* (drooping sedge) was collected from a buried paleosol at a side slope spring seep and this species not included in the custom wetland seed mix.

General observations of plants growing throughout the restored natural aquatic ecosystem during the first growing season following restoration reported a total of 86 species. Approximately 85 percent of the species richness identified in 2012 are native plants and wetland indicators (FAC, FACW, and OBL). In addition, greater than 40 percent of the post-restoration species richness are OBL wetland plants, indicating successful restoration of wetland hydrology and soils conditions (Figure F3).

Post-restoration vegetation survey of the restored areas along two transects identified 37 species. Approximately 80 percent of the species identified along these transects are hydrophytes. *Panicum rigidulum* (redtop panicgrass), a FACW species, provided 30 percent of the relative percent cover in the post restoration transect sampling. Other important species included *Leersia oryzoides* (rice cutgrass), *Nasturtium officinale* (watercress), and

Phalaris arundinacea (reed canarygrass) comprising 13, 8, and 7 percent of the relative cover, respectively(Figure F4). *Cyperus strigosus* (strawcolored flatsedge), *Ludwigia palustris* (water purslane) and *Juncus effusus* (soft rush) combined provided a total of 14 percent of the relative cover. Together, wetland plant species surveyed along transects accounted for 72 percent of the relative cover in all plots. *Trifolium arvense* (clover) and *Setaria spp.* (foxtail) accounted for 4 percent and 2 percent of the relative cover, but primarily were found along the edge of the restored wetland demarcating the transition to uplands.

Comparing the two compositions of pre- and post- restoration vegetation along Transect 6 using the Sorensen's Index of Similarity reveals a similarity of only 10.7 percent (2C/A+B = 6/56). This finding demonstrates that very little similarity exists between pre- and post-restoration plant communities. *Phalaris arundinacea* (reed canarygrass), one of the most abundant wetland species in the pre-restoration sampling of Transect 6 with 33 percent of relative cover, was present in much smaller coverage at 4 percent in the post-restoration vegetation. The U.S. Department of Agriculture (2002) reports that reed canarygrass may become weedy or invasive in wetland habitats.

A comparison of vegetation along the control Transect 1 indicates that upland grasses have decreased in importance over time. Because the pre-restoration survey was conducted in August 2010, grass species were difficult to differentiate since most of them had already past their blooming period. For this reason, upland grasses were grouped and included *Festuca spp., Poa spp., Dactylus sp., Agropyron sp., and Setaria spp*. These grasses plus Canada thistle provided 65 percent of the relative cover for Transect 1 in August 2010. These grasses declined in the June 2013 survey, comprising just 46 percent of the relative cover. However, when the upland grasses are combined with Canada thistle from the June 2013 sample they comprise 53 percent of the relative cover and indicate that a predominantly upland habitat has been maintained throughout the three year monitoring period.

A Sorensen's Index of Similarity analysis reveals similarity of 43 percent for pre and post restoration samples and suggests that the samples are not comparable. Increasing in importance within the control Transect 1 over the three year sampling period is reed canarygrass and poison hemlock that combined provided 34 percent of the relative cover in the June 2013 survey. However, the increase in poison hemlock during the June 2013 sampling may simply be due to the fact that it blooms between from May through July (Rhoads and Block, 2000) and rapidly dies back in early August (Hartranft, personal observation). The pre-restoration samples for the control Transect 1 were completed in August 2010, during a particularly dry period. The additional 34 percent cover also may indicate an increase in soil moisture over the three year period, perhaps due to effects of hydrologic change brought about by the close proximity to the restoration area. Further investigation is needed before an explanation of the shift is conclusive.

Conclusions

The shift in vegetation characteristics in the restoration area and between pre- and post-restoration time periods was decisive. The pre-restoration plant community dominated but FACU non-native species shifted to one dominated by OBL/FACW and primarily native species(Figure F4). along Transect 6 post-restoration. This conclusion is supported with additional post-restoration observations throughout the site where the plant community is comprised of primarily native wetland plants. The immediate reduction in relative percent cover of reed canarygrass is a positive indication that desirable wet meadow palustrine emergent marsh conditions have been restored.

Our investigations and the transect analysis so far indicate that the presence of many, if not a majority of species, are the combined result of applying a specialized wetland mix and plant plug installation during restoration. The sedge (*Carex spp.*) dominated wet meadow that was prevalent before the 18th century has not yet been re-established during the first growing season following restoration. Instead, a palustrine emergent marsh dominated by a wet meadow plant community as described by Fike (1999) has been restored at this site.

That a natural wet meadow plant community was established within the first growing season after restoration is a positive indicator of restored wetland hydrology and soil conditions. As the site matures and plant species continue to grow both above ground and below ground biomass, the immediate establishment of wet meadow vegetation is likely to give way to increased dominance of sedges. The anticipated maturation and trajectory of the plant community will be more similar to the paleo-environment at this site that was described by Neugebauer (2011) and Hilgartner, et al. (2012).

One reason why a sedge meadow may be take time to establish is rooted in the fact that some species, particularly *Carex prasina*, were not available for seeding after legacy sediment was removed. We have not observed the previously dominant *Carex prasina* growing in the immediate vicinity of where it was collected from soils buried under the legacy sediment that has now been removed. Our initial hypothesis, based on the apparently remarkable condition of these buried seeds, was that they may be viable given restored hydrology and soil conditions. So far, we have not supported this hypothesis but we will continue to monitor for the emergence of *Carex prasina.* We also will continue to monitor for the presence of several other wetland plant species identified in the buried seed record and not included in the specialized wetland seed mix or plant plug installations.

Future monitoring of the immediate and dramatic changes in the plant community at this site will be essential to determine the long term establishment of natural wetland plant communities. Years or decades of monitoring may be necessary to conclude that a natural and stable wetland plant community has been restored at this site. However, the early indications and trajectory of the restored vegetation at this site supports the hypothesis that a natural wetland plant community may be restored by removing legacy sediment. Furthermore, the immediate establishment of a plant community typical of wet meadow

palustrine emergent marsh is a positive indicator that removing legacy sediment has successfully restored wetland hydrology and hydric soils.

F. 2. Herpetological assessment – Adapted from a Technical Report titled: Amphibian surveys at Big Spring Run, 2010-2012 by David R. Bowne, Chelsea Payne, Samantha Hartzell, Aleah Miller, Amanda Harris, Matthew Doeing, Alexandra Doran and Ryan Conway.

The surveys for salamanders focused on relative abundance of larval stages at Big Spring Run and an upstream reference location that we refer to as the Kennel Branch. The cryptic nature and small size of salamanders make them difficult to study. We used a variety of techniques to increase the detection probability. These techniques included litter bags, kick nets, and dip nets. Litter bags create a site of suitable habitat for salamanders to colonize while not restricting their movements. Each litter bag was a 0 .7 X 0.7 m piece of plastic netting with a mesh size of 1.75 cm filled with rocks and leaf litter and bundled with a plastic twist tie. The large mesh size allowed for unobstructed access by salamanders to the interior of each bag. We placed fifteen litter bags into each branch each season. The kick net technique consisted of disturbing approximately 1.0 m^2 of upstream substrate for one minute while catching the disturbed material in a fine mesh net. We performed 30 kick nets per stream branch, with one upstream and one downstream sample around each litter bag. We also used a D-frame dip net to haphazardly sample the aquatic environment. We performed 10 dips upstream and 10 dips downstream of each litter bag for a total of 300 dips per branch. These three techniques were employed once in late May and early June of 2011 and 2012. In July of 2010, we used only litter bags and kick nets, but performed each techniques two times. Within each year, we expended the same capture effort in each branch (Figure F5).

Our survey for anurans was based on vocalizations. We installed a Song Meter (Wildlife Acoustics, Cambridge, MA) within the area that was restored at Big Spring Run and programmed it to record for 30 minutes at sunset each night from mid-March to late July. The recorder captured pre-restoration data in 2010 and 2011. The post restoration data was collected in 2012 beginning only approximately 4 months after sediment excavation and prior to one full growing season of plant community development. We then used the program Song Scope (Wildlife Acoustics, Cambridge, MA) to analyze the recordings for specific anuran species. We also documented anuran sightings and vocalizations when we were on site during the day.

Results

Eurycea bislineata (northern two-lined salamander) was the most common species captured at Big Spring Run. All of the captured *E. bislineata* were larvae, as identified by presence of external gills. In July 2010, the greatest numbers of captures were in the Main branch, followed by the West branch (Figure F6). Main and West branches also had the greatest captures in May 2011 (Figure F7). The first spring following the restoration, capture numbers in the Main and West branches decreased while the numbers in East branch increased (Figure F6). Captures in Kennel Run, the reference site, were approximately equal between May 2011 and May 2012 (Figure F6).
Pseudotriton ruber (northern red salamander) was detected at Big Spring Run but not at Kennel Run (Figure F7). One individual was captured in a litter bag in July 2010 and

another in May 2011 in East branch. Before the restoration, individuals were not detected in either the Main or West branches. Post-restoration (2012), two individuals were captured in litter bags in the Main branch and one individual in East branch.

We have conducted preliminary analyses of the audio recordings for two common anuran species, the green frog (*Lithobates clamitans*) and the bullfrog (*Lithobates catesbeianus*). Neither species has been detected via an analysis of the recordings taken in 2010, 2011, or 2012. During field surveys for salamanders in May 2012, we did hear at least one *L. clamitans* calling from a clump of vegetation in an area of standing water adjacent to the West branch. We did not hear or see any anurans in our pre-restoration work. We are continuing to analyze the recordings for other anuran species.

Discussion

Prior to restoration, the amphibian community at Big Spring Run appeared to be restricted to *E. bislineata* and *P. ruber*. *Eurycea bislineata* was most abundant in the Main and West branches. The frequency of capture of immature individuals suggests local recruitment is occurring. In the first field season following the restoration, *E. bislineata* was still detected at Big Spring Run but its numbers had decreased. Captures in the Main and West branches were very low but captures increased in the East branch. Our finding that captures increased in the East branch after the restoration suggests this branch served as refugia for *E. bislineata* since no sediment excavation or site disturbance occurred in this area. The immediate impact of the restoration activities on herpetofaunal populations is evidenced by the lack of change in *E. bislineata* captures in Kennel Run. Given the restoration was a major disruption to the system, it was expected that captures of *E. bislineata* would decrease in the restored Main and West branches in the short-term. What will happen in the long-term is less certain. *Eurycea bislineata* is a streamside salamander preferring fast-flowing, sediment-free water with abundant rocks in which to hide and forage (Petranka 1988). The restoration did not increase *E. bislineata* preferred habitat and this species may continue to have reduced abundance in the restored system. In contrast, *P. ruber* is a species preferring slow-moving water and an abundance of wetland vegetation (Petranka 1988) and the restoration increased preferred habitat for this species. We have tentative evidence that P. ruber abundance will increase over time since its numbers and spatial range increased following the restoration. Given its rarity (n = 5), however, a modest increase for the six months following the restoration is not conclusive evidence.

As the restored wetland ecosystem continues to mature, we expect a shift in the amphibian community from streamside species to those more common in wetlands. The rate at which this shift occurs will depend both on the maturation of the restored ecosystem and on the surrounding landscape. The Big Spring Run restoration site resides within a matrix dominated by agriculture landuse in the adjacent uplands. Agricultural landuses generally do not provide high quality habitat for amphibian species. It is thus possible that species that may be supported by the restoration at Big Spring Run will not colonize the site in the immediate future because of the lack of source populations to supply immigrants. When colonization does occur, it may take additional time to establish a self-sustaining population.

F. 3. Macroinvertebrate Assessment – Adapted from a Technical Report Titled: Macroinvertebrate Community Response to Stream Restoration Big Spring Restoration Project (2010-2012) by John Wallace

Introduction

Anthropogenic environmental impacts may negatively impact aquatic insect communities via homogenizing in-stream habitat, altering trophic structures, decreasing diversity, and ultimately reflecting poor water quality and loss of ecosystem services (Paul and Meyer 2001, Sweeney et al. 2004, Meyer et al. 2005). A range of macroinvertebrate taxa in stream ecosystems are affected by land-use land change (LULC) patterns and associated impairments, including those connected to urban development (Lenat and Crawford 1994; Wang et al. 1997; Sponseller et al. 2001; Paul and Meyer 2001; Wang et al. 2001; Roy et al. 2003; Allan 2004; Cuffney et al. 2005; Roy et al. 2005; Cuffney et al. 2010). Decreased macroinvertebrate species richness and higher community tolerance is exhibited with increased impervious surface area due to urbanization (Morse et al. 2003, Roy et al. 2003, Moore and Palmer 2005, Cuffney et al. 2010) as well as an overall degradation of macroinvertebrate community structure due to changes in stream water chemistry (Roy et al. 2003), sediment particle size (Roy et al. 2003, Violin et al. *in press*), hydrological changes (Walsh et al. 2005, Cuffney et al. 2010), and increased sedimentation (Minshall 1984, Roy et al. 2003).

Table F1. Symptoms associated with the 'urban stream syndrome'. Consistent response are those observed in multiple studies, inconsistent responses are those that have mixed responses from different studies with increased urbanization (as modified from Walsh et al., 2005.)

Feature	Consistent response	Inconsistent response	Limited research
Hydrology	↑ Frequency of overland flow ↑ Frequency of erosive flow ↑ Magnitude of high flow ↓ Lag time to peak flow ↑ Rise and fall of storm hydrograph	Baseflow magnitude	
Water chemistry	↑ Nutrients (N, P) ↑ Toxicants ↑ Temperature	Suspended sediments	
Channel morphology	↑ Channel width ↑ Pool depth ↑ Scour ↓ Channel complexity	Sedimentation	
Organic matter	↓ Retention	Standing stock/inputs	
Fishes	↓ Sensitive fishes	Tolerant fishes Fish abundance/biomass	
Invertebrates	↑ Tolerant invertebrates ↓ Sensitive invertebrates		Secondary production
Algae	↑ Eutrophic diatoms ↓ Oligotrophic diatoms	Algal biomass	
Ecosystem processes	↓ Nutrient uptake	Leaf breakdown	Net ecosystem metabolism Nutrient retention

Of the 86,000 stream/river miles in Pennsylvania, nearly 85,000 are assessed for aquatic life use and approximately 20% do not meet these designated uses (PADEP, 2012). During attempts to reverse the effects of LULC, degraded streams are often targeted for restoration. Restoration generally seeks to return degraded streams to as close to un-impacted conditions as possible (National Research Council 1992). Over the last twenty years, this concern to improve impaired or degraded streams in Pennsylvania has led to boom of stream restoration projects (Thomas, http://www.ansp.org/research/environmental-research/projects/restoration/).

Many times, stream restoration practices operate on the assumption that by reconfiguring channel geomorphology to pre-degradation status or condition, this change will foster the recovery of native aquatic organisms. This assumption relies on the idea that stream and riparian habitat rehabilitation will facilitate the overall aquatic community recovery (Brooks et al. 2002). While this assumption is based on empirically demonstrated correlations between increased habitat diversity and increased fish and macroinvertebrate diversity (Angermeier and Winston 1998, Brown 2003), some underlying questions may remain. First, there is conflicting evidence to support this 'field of dreams' (i.e., if you 'build' it, they will come) concept that physical habitat restoration is sufficient for community restoration (Palmer et al. 1997, 2010; Purcell et al. 2002; Gerard and Hellenthal, 2003). Second, extensive monitoring programs post restoration are scant or simply lacking (Moerke et al. 2004). Some studies of stream restoration success have shown limited or mixed success with regard to geomorphological improvement (Jähnig et al. 2010), and other studies report that it may take several years before new species begin to populate a restored stream channel (Rinkevich and Wallace 2001; Reppert et al. 2005; Palmer et al. 2010).

Project Purpose Summary

In this study, the BSR was deeply incised into ca. 1.5m of historic sediment for nearly a century prior to restoration (Figures F8 a,b). As part of a team of 20+ scientists, the purpose of this part of the study was to examine the effects of aquatic ecosystem restoration on in-stream macroinvertebrate community structure. Specific objectives included: 1) rapid habitat and stream assessment before/after restoration and; 2) characterization of the macroinvertebrate fauna using a BACI (Before/After, Control/Impact) sampling design over a three year period. An assessment of in/out of stream habitat also was assessed using Rapid Habitat and Visual Stream Assessment (RHVSA*)* protocols.

As described by Hartranft et al (2011), the BSR channel is a second order Piedmont stream with a link magnitude of 3. The aquatic ecosystem restoration reach described here includes the main stem that previously was the subject of a long-term USGS study. In the USGS study, this part of the main stem channel served as a control reach to a treatment site that flows into the BSR approximately 1km downstream (Galeone et al. 2006). The USGS report provided valuable historical data for the macroinvertebrate community at the restoration site and a control stream selected for this project. For this project, it was important to understand the upstream input of macroinvertebrates from the contributing

tributaries to BSR. Therefore, five sites were selected from both the West and East Branches of BSR, both 1st order streams that flow through residential and agricultural land use areas (see Figure F5).

The West Branch originates approximately 4 km upstream of the confluence with the main stem near a retirement community, several small businesses, and residential housing. This stream is influenced by overland flow from impervious surface and has very limited riparian buffer. The East Branch originates approximately 1-1.5 km upstream from where it joins the main stem of BSR. This branch is dominated by groundwater flow and adjacent agricultural land use that also limits the riparian buffer.

Three reference streams were used in this study to compare to the restored main stem of BSR. These control sites primarily flow through agricultural fields with poor riparian buffer characteristics. Each reference stream, as well as the restored reach of BSR, consisted of five sample sites. All sample site locations are listed in Table 2 with GPS coordinates. Where two sites are located too close for individual GPS coordinates, the sample sites are represented by one coordinate. A third reference stream with similar adjacent land use dominated by agriculture is located on a tributary to BSR approximately 1 km downstream of the restoration reach.

Table F2. List of sites and corresponding GPS coordinates, in some cases, only one coordinate designates site location e.g., restoration and control streams.

East Branch (Reference)	Lat/Long Coordinates
Site 1	N39˚ 59.091; W 076˚ 16.226
Site 2	N39˚ 59.485; W 076˚ 15.816
Site 3	N39˚ 59.369; W 076˚ 15.981
Site 4	N39˚ 59.475; W 076˚ 15.826
Site 5	N39˚ 59.471; W 076˚ 16.842
West Branch (Reference)	
Site 1	N39˚ 59.085; W 076˚ 16.226
Site 2	N39˚ 59.085; W 076˚ 16.225
Site 3	N39˚ 59.447; W 076˚ 15.505
Site 4	N39˚ 59.494; W 076˚ 15.654
Site 5	N39˚ 59.544; W 076˚ 15.730
Main stem (Restoration Site)	
All riffles near	N39˚ 59.602; W 076˚ 15.737
Reference Stream Long Rifle Road	
All riffles near	N39˚ 59.954; W 076˚ 15.874

Rapid Habitat and Visual Stream Assessment

An assessment of in/out of stream habitat was determined using modified procedures of EPA's Rapid Bioassessment Protocols revised by Barbour et al. (1999) to include parameters for high gradient streams and improved modifications for low gradient streams. Ten parameters were visually evaluated between May 25 – June 6, 2010 – and 2012. These parameters were used to rate the overall quality of the sampling reach on a scale of 0 (lowest score) to 20 (highest score), with representing "optimal" conditions. The first five parameters represented in-stream habitat features and were rated specifically for the sampling reach in this study. These parameters included: Epifaunal Substrate/Available Cover, Embeddedness, Pool Substrate Characterization, Pool Variability, Sediment Deposition. The next five parameters, were related to large-scale effects and required visual assessments beyond the sampling reach, included: Channel Alteration, Channel Sinuosity, Bank Stability (Condition of Banks), Vegetation Protection, and Riparian Vegetation Zone Width. The scores were summed for each sampling date. Scores represent overall averages for each stream reach studied. For example, a 100-m reach of each reference stream was walked and assessed with this protocol.

Macroinvertebrate Sampling

Because of the singular nature of this restored site, collections were conducted using the Before/After/Control/Impact method as described by Steward-Oaten, et al. (1992) to assess the impact of stream restoration on macroinvertebrate community structure. Because of their propensity to display maximal diversity of macroinvertebrates sensitive to water quality changes, macroinvertebrates from riffle areas were collected at each site using a Surber Sampler designed for water less than 25cm in depth and appropriate for quantitative analysis for a variety of metrics (Figure F8). Each collection was standardized with timed 30 second sampling bouts of each area with the Surber sampler.

Approximately 5 samples from each reference stream (West/East Branches and Control reference) as well as the main stem section of the BSR were collected May 25 –June 6, 2010-2012. Years 2010 and 2011 represented pre-restoration samples. Sampling in May, 2012 was approximately eight months after construction, at the beginning of the first growing season following disturbance. Sampling in early spring was based on macroinvertebrate size and diversity advantages compared to other time periods during the year when aquatic insects may be too small to identify or absent from the streams as larvae.

Macroinvertebrate samples were preserved in 95% ETOH, stored in 16oz. Whirl-Pac® bags, and returned to the laboratory for identification and enumeration. Macroinvertebrates were identified to the generic level except for midges (Diptera: Chironomidae) that were identified to family level using identification keys from Merritt et al (2008) and Peckarsky et al. (1990).

Statistical Analyses

Rapid habitat and visual stream assessment average scores were compared among all sites. Macroinvertebrate assemblages were compared using the Macroinvertebrate Aggregate Index for Streams (MAIS) for the four stream reaches as described by Moeykens (2002). While the index correlates well with acid mine drainage impacts, it was designed to respond to other forms of anthropogenic impacts on stream systems such as urbanization, sedimentation and nutrient loading (Johnson 2007). Therefore, MAIS is a standard approach to compare these stressors in the BSR restoration project. The MAIS score includes ten different metrics combined to create one aggregate score for each site. The ten parameters included: 1) Ephemeroptera richness, 2) EPT richness, 3) Intolerant taxa richness, 4) % Ephemeroptera, 5) %EPT, 6) % 5 dominant taxa, 7) Simpson Diversity, 8) Hilsenhoff Biotic Index, 9) % scrapers, and 10) % haptobenthos (Table 3).

Table F3. The nine biological metrics that comprise the final MAIS index score (modified from Johnson, 2007).

	METRIC	DEFINITION
1	EPT richness	Number of caddisfly, stonefly and mayfly families
2	# Emphemeroptera	Number of mayfly families
3	% Ephemeroptera	% abundance of mayflies
4	% 5 dominant taxa	5 most dominant taxa combined
5	Simpson Diversity Index	Integrates richness and evenness
6	Modified Hilsenhoff Biotic Index	Taxa are weighted by pollution tolerance
7	# intolerant taxa	# of families with tolerance values of 5 or less (very sensitive)
8	% scrapers	Abundance of macros that feed on periphyton
9	% haptobenthos	Abundance of macros that require clean, coarse, firm substrates

*10 %EPT % of mayflies, stoneflies and caddisflies

* Absent from this table.

Each parameter was scored 0,1, or 2. These scores were summed for a total single metric score (Table F4). Scores were compared to determine a rating between Very Poor and Very Good (Table F5).

Table F4. Cutoff levels used during index development to assign scores to each of the metrics of the MAIS (modified from Smith and Voshell 1997).

Metric	0	1	2
EPT	≤2	>2 – 7	>7
% 5 most dominant taxa	100	79.13 - <100	<79.13
HBI	≥5.56	4.22 - <5.56	<4.22
# Ephemeroptera taxa	0	>0 - 3	>3
% haptobenthos	≤51.98	>51.98 – 83.26	>83.26
% Ephemeroptera	≤0.1	>0.1 – 17.515	>17.515
# intolerant taxa	≤1	>1 - 9	>9
% scrapers	≤0.1	>0.1 – 10.7	>10.7
Simpson Diversity Index	≤0.656	>0.656 – 0.8225	>0.8225

Table F5. Biocriteria and levels for the MAIS in the Blue Ridge Mountains ecoregion using only reference sites (modified from Smith and Voshell 1997).

	Unacceptable		Acceptable	
	Very Poor	Poor	Good	Very Good
Reference site quartiles	≤5%	6-24%	25-49%	≥50%
MAIS score	<11	11-14	15-16	≥17

Because of the single restored site and potential pseudo-replication in comparing MAIS scores across years, MAIS scores were compared among years with nonmetric multidimensional scaling. Analysis of the macroinvertebrate communities between locations for each year was done using R (v2.15.2, The R Foundation for Statistical Computing) with the *metaMDS* function from the *Vegan* package. Analysis was done using Bray-curtis dissimilarities, with 2 *apriori* defined axes, and without transforming the abundance data to preserve the importance of abundance differences between samples. Spacing between data points on the NMDS plot reflect degree of dissimilarity; the further apart two points were on the biplot was correlated to increasing dissimilarity. Conversely, the closer two points occur to each other reflects less dissimilarity.

RESULTS

Rapid Habitat and Visual Stream Assessment (RHVSA)

Scores for the RHVSA were quite variable both pre and 8-months post construction for all sites (Table F6). Data were not statistically compared due to minor interpretational issues among some parameters, but are displayed to give a superficial evaluation of the changes that occurred within the first year after construction began. As stated previously, the first 5 metrics represent in-stream parameters and the last 5 represent values for riparian habitat adjacent to the stream channel.

It appears that the West Branch Reference scores were considerably higher than all other streams both pre and 8 months post-construction. The main stem (restored reach of BSR) score was greater pre-restoration compared to 8 months after construction. Of special note however, were the changes for the main stem. We observed that scores for both parameters of Epifaunal Substrate/Available Cover and Pool Substrate Characterization declined 8 months after construction (-3 and -7 points, respectively). However, Pool Variability and Sediment Deposition both improved (+9 and +2 points, respectively). No change was observed between years with Channel Flow Status. Riparian habitat scores for the main stem overall improved 8 months post construction. Specifically, Channel Alteration, Channel Sinuosity, and Bank Stability increased in score(+8, +4, and +15 points, respectively). In contrast, the parameters for Vegetative Protection and Riparian Vegetative Zone Width declined (-16 and -18 points, respectively).

Table F6. Summary scores from the Rapid Habitat and Visual Stream Assessment for 2010 (Pre- restoration) and 2012 (eight months post-restoration) for all sites.

	Main stem		Control Ref		East Br. Ref		West Br. Ref	
Metric	2010	2012	2010	2012	2010	2012	2010	2012
Epi.Sub/Av.Cov.	5	2	11	8	8	9	7	12
Pool Sub Charac.	13	6	11	11	11	10	6	13
Pool Variabil.	7	16	5	11	8	7	10	14
Sed. Dep.	6	8	10	8	6	10	6	11
Chan. Flow Stat.	13	13	7	15	8	15	15	12
Chan. Alteration	8	16	6	18	15	20	16	18
Chan. Sinuosity	16	20	6	6	9	6	7	14
Bank Stability	0	15	0	12	6	8	12	9
Veg. Protection	20	4	20	10	10	6	20	13
Rip. Zone Width	20	2	20	6	10	4	20	12
Total Scores	108	102	96	102	91	95	119	128

Macroinvertebrate Analyses

Approximately 12,758 macroinvertebrates were collected and identified to genus level during this study (2010: 8227 organisms; 2011: 2259 organisms; 2012: 2272 organisms) representing 15 different insect families and five non-insect invertebrate groups. The macroinvertebrate families or orders collected from all four sites pre and post construction are listed in Table F7. The macroinvertebrates identified to generic level for all sites pre and post restoration are in Appendix . The most abundant families of macroinvertebrates across all streams and years included Gammaridae (Amphipods/Scuds), Chironomidae (midges), Simuliidae (black flies), Baetidae (mayflies), Hydrospychidae (net-spinning caddisflies), and Elmidae (riffle beetles).

Table F7. List of macroinvertebrate families/orders collected in 2010, 2011 (pre-restoration) and 2012 (8 months after construction) for all study streams. (X indicates presence of group)

	Main stem (BSR)			Control Ref			East Br Ref.			West Br. Ref		
Group	'10	'11	'12	'10	'11	'12	'10	'11	'12	'10	'11	'12
Planaridae			X									
Oligochaeta							X			X		
Hirudinea	X			X			X			X		
Cambaridae	X				X							
Gammaridae	X	X	X	X	X		X	X	X	X	X	X
Coenagrionidae					X	X						
Aeshnidae		X										
Chironomidae	X	X	X	X	X	X	X	X	X	X	X	X
Simuliidae	X	X	X	X	X	X	X	X	X	X	X	X
Tipulidae		X			X						X	
Muscidae				X			X		X	X		
Athericidae											X	
Baetidae	X		X	X	X	X	X			X	X	X
Philopotamidae					X					X	X	
Hydroptilidae											X	
Hydropsychidae	X	X	X	X	X	X		X	X	X	X	X
Dytiscidae							X		X			
Hydrophilidae	X			X						X		
Elmidae	X	X	X	X	X	X	X	X	X	X	X	X
Psephenidae						X						

Macroinvertebrate density/m^2 was greatest both pre (2010-11) and 8 months post construction (2012) in the main stem BSR (Figure F9). A substantial portion of the biomass found in the main stem BSR consisted of Amphipods (Gammaridae). MAIS scores for the Control reference reach were significantly higher than all other sites with average values ranging from 8.8 to 10, suggesting a 'fair' stream (Figure F10) ($F=36.73$; $P < 0.05$). East Branch values were the lowest, with average values ranging from 1.4-2.6, suggesting a

'poor' stream. There was not a statistical difference in MAIS scores among sites over the three years.

NMDS analyses demonstrated some degree of dissimilarity among sites. The closer two points were located to each other, the less degree of dissimilarity. Using 2 axes for the analyses (Figure F11) resulted in a stress value of 0.071. A Shepard plot of ordination distances against Bray-curtis dissimilarities found R^2's for non-metric and linear fits of 0.995 and 0.974 respectively, indicating high fit of ordination distances to measured dissimilarities. Communities for each year in the Control site were generally similar to each other and generally different from the communities found at the other sites. The main stem restored reach of BSR eight months post construction was more similar to the East Branch reference stream and significantly departed from NMDS point samples in 2010 and 2011.

DISCUSSION

Water quality in lotic (running waters, e.g., streams) and lentic (standing waters, e.g., ponds) water bodies may be assessed via physicochemical properties, such as the level of dissolved oxygen, pollutants, suspended sediment, and water clarity (Kenney et al., 2009). In this project, some key restoration objectives are to reduce specific physicochemical parameters, such as nutrients and suspended sediments. However, while such parameters may provide a snapshot of certain conditions of a stream, Kenney et al. (2009) contend that they do not provide the integrative measure of overall health of a stream and may at times inadequately identify impaired waters (United States Environmental Protection Agency, USEPA, 2005). Rather, biological measures or criteria provide the necessary integrative and comprehensive assessment of the health of a water body over time. Biocriteria utilize attributes of the biological community such as lower trophic level organisms (e.g. algae or benthic macroinvertebrates) and upper trophic level species(e.g. fish) fish to encapsulate the 'health' status of a given aquatic resource (Kenney et al 2009).

The full suite of natural aquatic ecosystem restoration, particularly the combination of stream and palustrine emergent wetland restoration, is inherently more complex for either lotic or lentic water bodies per se. Physical, chemical and biological conditions in palustrine emergent marshes are often dependent on a combination of stream and groundwater hydrologic regimes that fluctuate seasonally or annually, for instance (Cowardin, 1979). Subsequently, natural aquatic ecosystem restoration projects for such complex systems also vary in type and scope in order to meet pre-determined goals and objectives (Palmer and Wainger, 2011).

Paleogeographical and paleobotanical evidence suggests that the BSR valley bottom in the restoration area consisted of anastomosing channels with adjacent palustrine emergent wetlands largely comprised of sedge dominated wet meadows during the pre-settlement period and prior to the accumulation of legacy sediment (Hartranft, et al., 2011). The objective of this project from a geomorphology and engineering perspective was to restore valley bottom geomorphic characteristics by removing legacy (i.e. historic) sediment. Broader objectives were to restore anastomosing stream channels with adjacent palustrine

emergent wetlands largely comprised of wet meadows. These broader restoration objectives are analogous to the Holocene paleoenvironment that was documented at this site.

Because of the multi-faceted goals and objectives for this project, a la Palmer and Wainger (2011), the macroinvertebrate component of this study becomes a considerable challenge to evaluate by relying on traditional stream ecological perspectives. This insight is important in the sense that the goal of this project was not solely to improve stream channel habitat for the sake of macroinvertebrate biodiversity, or some other instream metric, but to achieve a multitude of functioning natural aquatic ecosystem benefits (e.g. water quality, habitat, biodiversity, etc.).

Increasing habitat heterogeneity has been the benchmark in terms of stream restoration paradigms essential to promote macroinvertebrate diversity (Palmer et al 2010). A review of stream restoration projects that spanned four decades indicates that a common stream restoration practice was to re-configure channels by creating meanders that increased sinuosity, as well as adding physical structures such as boulders, root wads, and log vanes to recreate artificial riffles. The introduction of a reconfigured channel and physical structures to enhance channel heterogeneity and, as a consequence, biodiversity is analogous to what was accomplished in this project. In other words, assessing how the BSR stream channel responds in terms of biodiversity depends upon whether it is treated as a lentic or lotic system. Because the resulting aquatic ecosystem at BSR that is an anastomosing channel within a wet meadow floodplain ecosystem, it is not quite either.

The RHVSA results indicate that the restoration reach of BSR has improved in terms of in-stream habitat parameters like pool variability and sediment deposition and declined in others e.g., epifaunal substrate and pool substrate characteristics. From a biological perspective, both diversity and densities of macroinvertebrates in the restoration reach declined eight months after construction was completed. Similar studies suggest little to no improvement, or perhaps even a decline in macroinvertebrate biodiversity, perhaps because of the disturbance resulting from construction (Palmer et al. 2010).

In general, riparian habitat scores improved post-construction at BSR. About eight years prior to restoration, the landowner excluded livestock access to the stream and riparian zone from a neighboring farm and replanted the riparian zone with trees as part of a CREP project. Sediment excavation during construction resulted in an immediate decline in riparian habitat scores post restoration. The width of the riparian zone has not changed, however, the maturity of the woody species as well as the herbaceous plant community has been reset. After eight months, plant diversity typical of palustrine emergent wetlands is increasing in the riparian zone with some taxa reestablishing anew (Jeffrey Hartranft, pers. communication). In addition, submerged aquatic vegetation also has increased and now includes species representative freshwater marshes. The emergence of instream submerged aquatic vascular plants may provide refugia or habitat for macroinvertebrate faunal assemblages typical of slower moving, lentic ecosystems, such as anastomosing channels with adjacent palustrine emergent wet meadows.

While instream habitat heterogeneity has been discussed in the literature as the gold standard to improve instream biodiversity (Bond and Lake 2003; Palmer et al 2010), riparian habitat heterogeneity may be just as critical. For example, plant cover along the stream corridor during adult aquatic insect dispersal flights may aid macroinvertebrate and plant recolonization. A lack of riparian heterogeneity may be a 'soft' barrier to the adult aquatic insect colonization process (Bond and Lake 2003; McIntyre 2000). The magnitude of how local habitat manipulations may impact biological targets, e.g., macroinvertebrates is a function of the life stage of the target taxa (Beck 1995), however, this question cannot be answered with one post construction sample.

Long-term abundance, diversity, and richness of macroinvertebrate communities have been shown to increase following remediation (Hoiland et al. 1994, Nelson and Roline 1996, Adams et al. 2005). However, sediment removal projects can often result in an immediate degradation of macroinvertebrate communities (Bonvincini et al. 1985, Quigley and Hall 1999, and Gilkinson et al. 2005). In the BSR study, the dominant groups of macroinvertebrates, e.g., Chironomidae and Baetidae are indicators of stream systems impacted by suspended sediment loads (Relyea et al. 2000). However, net-spinning Hydropsychidae, filter-feeding Simuliidae and scraping Elmidae found to be common and abundant pre and post restoration have been shown to be highly influenced by fine sediments via clogging feeding structures and nets (Lemly 1982). Macroinvertebrate densities declined over the three year period but not significantly among the reference streams. Macroinvertebrate density in the main stem (restored) reach significantly declined eight months post construction. Friberg et al. (1998) cited several studies that found where the creation of new meanders was priority in the restoration, macroinvertebrate diversity and density decreased for a short period following construction as a result of the mechanical disturbance and excess sediment transport associated with construction work (Doeg and Koehn, 1994; Kronvang et al., 1998).

MAIS index scores were significantly higher in the Control reference stream compared to the other references streams and the main stem BSR. This difference may be explained in terms of the heterogeneity of the stream substrate in the Control reference reach. This reach consists of few sediment-laden pools with primarily gravel to small cobble size substrates(Wallace, unpublished data). While the decline in MAIS scores over time was not significantly different among stream reaches, this type of decline has been observed in previous studies (Rinkevich and Wallace 2001; Reppert et al 2005; Palmer et al 2010). In general, the NMDS plot showed little variation in the main stem during the three study years suggesting that there was not much change in community structure; however, the 2012 point for the main stem eight months post construction is more similar to the East Branch reference stream, which is significantly impacted by agricultural runoff and higher suspended sediment loads. This also corresponds with the MAIS scores, which demonstrate very little variability over the course of the study period. The lack of variability in community structure from pre to post restoration is not surprising due to the similarity in families found in 2010 and 2011 compared to 2012 post restoration, egg. Gammaridae, Hydropsychidae, Elmidae, Simuliidae, and Chironomidae.

As indicated in earlier work, the macroinvertebrate community is expected to recover rapidly (within 1-4 years) reflecting a high resiliency (Reice, 1985; Niemi et al. 1990) and in agreement with the findings of Biggs et al. (1998) and Reppert et al. (2005) in other restored streams. Moreover, both diversity and density may increase and surpass estimates prior to restoration as a result of either the creation of more habitat or the synergistic effects of habitat heterogeneity and improved water quality (e.g. Minshall, 1984; Reppert et al. 2005). Milner (1996) found that macroinvertebrate community diversity generally became stable three to four years post restoration but continued to slowly increase due to longer recolonization rates for taxa with low dispersal abilities. A three to four year time frame was corroborated on the local scale by Reppert et al (2005) on the Hammer Creek, Snavely Mill restoration site.

Dispersal limitations or barriers for adult aquatic insects can significantly affect the rate of diversity and density increase in restored streams (Smith et al. 2009). Anthropogenic alteration of natural landscapes can create such barriers and affect both in-stream and terrestrial stages of aquatic insects, thus changing dispersal capabilities of adult aquatic insects (Smith et al. 2009). Adult insect diversity and abundance were not examined in this study, however, their contribution to the in-stream faunal assemblage should not be overlooked and will be included in future monitoring efforts at BSR. In terms of how the organisms process leaf litter, a source of nitrogen to the stream, preliminary data indicate that after 18 months post construction, macroinvertebrates were processing leaf packs in the main stem at a faster rate than the control reference stream (Wallace, unpublished data). In fact, we have noticed a shredder caddisfly (Limnephilidae) in the main stem during these leaf pack experiments. The shredder caddisfly was not observed in any of the study streams prior to restoration (Wallace, unpublished data). We plan to assess how macroinvertebrates have responded to restoration in terms of how fast they process allochthonous organic matter.

CONCLUSIONS

In 2011, the BSR restoration reach was converted from a single thread meandering channel system to an anastomosing channel system bounded by a palustrine emergent wetland via removal of fine sediment that had buried the valley bottom natural aquatic ecosystem. We continue to monitor how this ecosystem responds (i.e., changes in population and community composition) with regards to macroinvertebrate community dynamics following restoration. The current study illustrates that a positive macroinvertebrate community response, from an ecological perspective, might be achieved if the goal of the restoration is for multi-functional anastomosing channel and palustrine emergent wetland ecosystems. The difference in comparison with a standard stream restoration assessment is the assemblage of macroinvertebrates. In order to detect changes in population/community composition, rigorous and lengthy biomonitoring surveys are required. The current results from this macroinvertebrate study include data from only one sample date eight months post construction, and it is likely that at least several years will be required to fully assess ecosystem response to restoration.

Appendix 1. Project Design and Construction Details

Construction Dates		September to November, 2011
Equipment Used	John Deere 750J-LGP	**L**ow **G**round **P**ressure Bulldozer: The LGP machine has a 4.45psi(30.7 kPa) rating vs. the standard 750J that is rated at 7.24psi(49.9kPa). The ground contact area is almost double for LGP than standard 750J.
		http://www.deere.com/en_US/docs/construction/crawler_dozers/750j/DKAJGDZR.pdf
		The 750J LGP dozer was used to set the grade of the restored floodplain by pushing the legacy sediment into a pile.
	Pan (pull-type scraper)	The pull-type scraper, commonly referred to as a "pan", was used to move the legacy sediment to another pile.
	John Deere 9R/9RT	The pan is pulled by a John Deere 9R/9RT Series four wheel drive tractor.
	Trackhoes	When space became limited and the pile was too high, track backhoes were used to excavate historic ("legacy") sediment.
	Haul Trucks	The excavated sediment was "live loaded" into "haul trucks". The haul trucks back into position, then the track hoe puts the excavated material right into the bed. The haul trucks are articulated and four-wheel drive, but they also had some difficulty with wet conditions and got stuck frequently.
Watershed Properties	Drainage Area	BSR Above the downstream (main stem) USGS Gage = 1.68 mi^2
		http://waterdata.usgs.gov/usa/nwis/uv?015765195
	Length Restored	2,900 - 3,000 linear feet (total)
		1,240 ft (West Branch (Fry Branch), Reach 2
		220 ft (East Branch, (Sweeney Branch), Reach 1
		1440 ft (Main Stem below confluence)

	Total Area of Riparian Wetland/Flood plain Created	4 acres
Design Criteria	Objectives	The goal of the Big Spring Run project is to restore the natural floodplain, stream and riparian wetlands in a portion of this watershed to its natural ecological potential. Establishing the ecological potential at Big Spring Run requires knowledge of the range of physical, biological, and chemical conditions and processes that existed prior to legacy sediment accumulation, as well as an understanding of existing and future watershed conditions that might constrain the ecological potential of the restoration.
	Design Features	Design features include stream stability, nutrient removal efficiency, aquatic habitat, and other ecological functions and values.
	Maximize Stream stability	Based upon the pre-settlement valley morphology and stratigraphy, modern bed material analysis (pebble counts) and lack of sustainable bedrock, watershed area, and other hydrologic and hydraulic design considerations, the allowable boundary shear stress within the channel will be less than 0.3 lbs per square ft. Therefore, the design depth from normal water surface to floodplain will vary between 0.3 and 0.7 feet, depending upon the local water surface slope.
	Maximizing nutrient removal efficiency	*Nitrogen* – The key factors to improve nitrogen removal include:
		(1) increasing the amount and availability of carbon based material,
		(2) increasing the retention time and flow contact with the carbon based material
		(3) increasing the base flow channel water elevation to promote hyporheic exchange and increase hyporheic zone ecosystem function
		Sediment and phosphorus – The key factors to improve phosphorus and sediment removal include:

		(1) frequent overtopping of channel flows into the floodplain and
		(2) high floodplain roughness as well as locations where flow direction changes quickly and depression areas increase flow retention time.
Design Numbers	Channel Design Q	2-4 cfs
	Mean Depth	0.6 ft
	Radius of Curvature	~15
	Sinuosity	1.6
	Boundary Shear	0.15 lbs/ft2
	Entrenchment	>12
Restoration Costs	Design & Permitting	$55,000.00
	Construction, Management, and Monitoring	$445,000.00
	Sewer Line Relocation	$140,000.00

Appendix 2. Topographic and Geomorphic Survey methods

2004 Survey, Pre-Restoration

LandStudies, Inc., conducted initial stream cross-section surveys at the Big Spring Run restoration site using a Total Geodetic Station in the fall of 2004. Five survey control points were installed (iron rods with yellow caps placed flush with the ground). Data were provided by LandStudies in a text file with local grid coordinates and codes for both the cross-sections and the benchmarks. Twelve cross sections were surveyed. In Fall 2009 LandStudies and Franklin and Marshall College staked out the cross section end-points with a total geodetic station and installed rebar with survey caps in the ground at those locations.

2010 Survey, Pre-Restoration

Franklin and Marshall staff resurveyed the cross-sections during April 2010 using a RTK-GPS. The RTK base was configured to log static data during survey sessions. Northing and Easting values were recorded in Pennsylvania state plane, US-feet. Elevation was recorded in NAVD88, US-feet. The static data was differentially corrected using the OPUS service offered by the National Geodetic Survey at NOAA. All data were adjusted to account for the differential correction of the base location.

The RTK rover was used to observe the control points and the ground surface on the lines of cross-section. All five benchmarks and the nail were located and observed for 180 1-second epochs with an RTK-fixed solution. Topographic points were observed for four 1-second epochs with an RTK-fixed solution.

2012-13 Surveys, Post-Restoration

Franklin and Marshall staff extended and resurveyed the cross-sections during December 2012-November 2013 using an RTK-GPS. The RTK base was configured to log static data during survey sessions. Northing and Easting values were recorded in Pennsylvania state plane, US-feet. Elevation was recorded in NAVD88, US-feet. The static data was differentially corrected using the OPUS service offered by the National Geodetic Survey at NOAA. All data were adjusted to account for the differential correction of the base location.

Surveys in December 2012 used the same base location as in 2010. A new, monumented base station was installed and surveyed with RTK GPS by UNAVCO surveyor Brendan Hodges in June 2013. Hodges also installed a 2nd monumented control point that we use as a check on surveys. Both survey monuments are installed in bedrock.

Cross-sections surveys in 2013 use the new UNAVCO benchmarks as the base location. Coordinates (in PA Plane South) for the new base station benchmark are

Northing 243717.170
Easting 2385650.347

Orthometric height (NAVD88, GEOID12A) 338.802

The RTK rover was used to observe the control points and the ground surface on the lines of cross-section. Topographic points were observed for four 1-second epochs with an RTK-fixed solution.

Data Adjustments

Survey data were exported from the job file for import into Excel and ArcGIS. Data from 2004 was adjusted from its local grid coordinates to Pennsylvania state plane using the similarity method with the five benchmarks in ArcGIS" Spatial Adjustment" tools. The similarity transform scales, rotates and translates the data; it does not change the aspect ratio or skew the data. The adjustment yields a horizontal RMS of 0.02 feet.

For the 2004 and 2010 pre-restoration surveys, and for post-restoration surveys done from December 2012-June 2013, repeated observation of a mag-nail control point produced results that were consistently within 0.02 vertical feet of the value noted on the restoration design plan sheet. Repeated observation of the benchmarks used in the original cross-section survey produced elevations that were consistently higher than those from the original survey, with the exception of one benchmark from which the yellow cap had broken off.

The mag-nail used prior to June 2013 is believed to have represented the best available local elevation datum at the time, as it was in pavement, whereas the iron rods were driven into soil in a pasture. Therefore a vertical offset of 0.088 feet was added to the 2004 cross-section survey to bring it into close agreement with current survey control, which includes the mag-nail. The vertical offset is the average vertical difference between benchmark elevations observed in 2004 and 2010 for four of five benchmarks, excluding the benchmark with the missing yellow cap.

Elevations presented in this report are orthometric heights relative to NAVD88 and calculated using GEOID03. The 2013 survey data were surveyed using the UNAVCO benchmark which has an orthometric height calculated using GEOID12A. Therefore a correction was added to the 2013 data to make it consistent with earlier data. The correction is equal to the difference between the height of the two geoids above the ellipsoid as determined using NOAA GEOID height calculators:

http://www.ngs.noaa.gov/cgi-bin/GEOID_STUFF/geoid03_prompt1.prl
http://www.ngs.noaa.gov/cgi-bin/GEOID_STUFF/geoid12A_prompt1.prl

At the base station location, GEOID12A is 3.8 cm lower than GEOID03, and therefore an adjustment of 0.125 ft was added to 2013 surveys. In the future we plan to transition to GEOID12A.

Appendix 3. Analytical methods (grain-size analysis and sediment fingerprinting)

Grain Size Analysis

Air-dried, lightly crushed (for disaggregation) samples were sieved with a RoTap to 0.6 mm grain size. Particle size for the fraction finer than 0.6 mm (coarse sand) was analyzed with a Micromeritics Saturn laser diffraction particle size analyzer. Sieve data were merged with laser diffraction data to produce a complete grain-size distribution.

Grain size data for stream bank sediment are available in the file named "Bank Sediment Grain Size DEP Report 2013".

Grain size data for suspended sediment (USGS gage stations) and deposition pad samples are available in the file named "Grain Size Analysis DEP Report 2013".

Appendix 4. Analytical methods (nutrient analysis)

Appendix IV – Protocols for Sediment and Soil Sampling and Geochemical Analyses

Stream Bank Sampling

(1) Cut back vertical stream bank ca. 10 to 20 cm to assure a clean, uncontaminated surface (using a pick, hoe, shovel or trowel).
(2) Measure stratigraphic sections, delineate sedimentary units, describe sediments, and collect samples for geochemical and geophysical studies, including nutrient and trace metal contents, radiocarbon dating and magnetic susceptibility.
(3) Sampling Methodology -- Beginning at the upper (topmost) surface, cut a vertical channel or mini-trench with a clean trowel in approximately 15 cm long (along the face of the cut bank), by 10 cm wide (perpendicular to face) and 5 cm deep increments. If the sediment is moist and consolidated, the sediment can be extracted in oriented 5x10x15 cm bricks (which has some advantages in subsequent analyses), but for most cases simply scoop the material into a zip-lock bag one trowel tip at a time until a representative sampling for each 5x10x15 cm increment has been achieved (ca. 300 – 500 g).
(4) Typically, the stream surface is intersected before bedrock is reached, so this will be an incomplete stratigraphy unless it is possible to reach below the water line and extract sediment samples by hand.
(5) In the lab: (a) open ziplock bags and allow samples to air dry (usually 2-3 weeks), (b) select subsample of field moist sample from moisture content and low-temperature oven drying (<60°C).

Phosphorus

X-Ray Fluorescence (XRF)

Major Elements: Crushed rock powder (.4 grams) is mixed with lithiumtetraborate (3.6 grams), placed in a platinum crucible and heated with a meeker until molten. The molten material is transferred to a platinum casting dish and quenched. This produces a glass disk that is used for XRF analysis of SiO_2, Al_2O_3, CaO, K_2O, P_2O_5, TiO_2, Fe_2O_3, MnO, Na_2O and MgO.

Working curves for each element are determined by analyzing geochemical rock standards (Abbey (1983) and Govindaraju (1994).) Between 30 and 50 data points are gathered for each working curve; various elemental interferences are also taken into account. Results are calculated and presented as percent oxide.

Inductively Coupled Plasma – Optical Emission Spectroscopy (ICP-OES)

Sample Preparation and Microwave Digestions

The EPA Method 3051 is used to partially digest 0.2500 +/- 0.0002 g of crushed soil samples in 10 mL of concentrated nitric acid heated to 175°C for (insert time) in a CEM

MARSX press microwave unit. The resulting digestion solution is filtered and diluted to 50 mL, resulting in a 20% HNO3 solution. Finally, the amounts of 14 trace elements (Al, Ba, Be, Cd, Co, Cr, Cu, Fe, Mn, Mo, Ni, P, Pb, and Zn) in these solutions are measured by ICP-OES.

An ANALYTICAL *Plus* balance is used to weigh out 0.2500 +/- 0.0002 g of soil sample onto Fisherbrand 4"x 4" weighing paper. This sample is then poured into a numbered digestion vessel. The Fisherbrand 2-10mL Finnipipette is used to add 10 mL of concentrated nitric acid to the digestion vessels containing the soil samples. The vessels are then weighed on a Sartorius balance before placing them in the microwave carousel, which is in turn placed in the CEM MARSXpress microwave unit. The samples are rotated to ensure homogenous distribution of heat in the microwave using a pre-programmed heating and cooling cycle. After removal from the microwave, the samples are reweighed on the Sartorius balance to check that no significant weight change has occurred during heating, which would indicate the loss of material by venting of vapors from the digestion vessels.

The next step in the procedure is the filtering of the nitric acid and soil sample solution. Because this procedure only partially digests the soil samples, there still remains much solid material after heating which must be separated from the nitric acid containing digested ions in solution. The glass funnels containing the conically folded filter papers are placed on a stand with their outlets running into the 50 mL glass volumetric flasks. The inner caps of the digestion vessels are covered in droplets of condensed vapor from the heating and so are rinsed into the funnel with distilled water. The contents of the digestion vessel are then poured into the filter with distilled water (used to keep the filter paper from tearing). The liquid portion of the vessel's contents filters through the paper and drips into the volumetric flask and the solid portion remain in the filter. The digestion vessel is rinsed twice with distilled water, which is then poured into the filter.

Once all of the digestion solution has been filtered, the filter paper is removed from the funnel and placed aside to dry. The glass funnel is rinsed into the volumetric flask with distilled water. Subsequently, the digestion solution is diluted to 50 mL with distilled water. Occasionally the samples must be diluted to 100 mL due to tearing of the filter paper during initial pouring or to overfilling of the 50 mL volumetric flasks.

The ICP spectrometer is calibrated with five calibration standards, and a blank (20% HNO3) each time it is used to measure the trace elements present in the digested soil samples. The standards are made from the initial solutions. The standards are made in Pyrex glass volumetric flasks of various sizes using ~6 Mega-Ohm distilled water. Volumes are measured using Fisher brand Finnipipettes of 2-10mL, 1-5mL, and 100-1000?L sizes and Finntip 10mL and 5 mL and Fisher brand 101-1000 mL pipette tips. Standards are stored in Nalgene bottles and used within 10 days of preparation. The samples are analyzed by ICP-OES on a Spectro Ciros CCD machine connected to a Hewlett Packard computer running Smart Analyzer Ciros CCD software. This ICP spectrometer runs using Argon plasma.

Flow Injection Analyses

Flow injection analysis, or FIA, is a continuous flow method for rapidly processing samples. A peristaltic pump draws sample from the sampler into the injection valve. Simultaneously, reagents are continuously pumped through the system. The sample is loaded into the sample loop of one or more injection valves. The injection valve is then switched to connect the sample loop in line with the carrier stream. This sweeps the sample out of the sample loop and onto the manifold. The sample and reagents then merge in the manifold (reaction module) where the sample can be diluted, dialyzed, extracted, incubated and derivatized. Mixing occurs in the narrow bore tubing under laminar flow conditions. For each method, the operating parameters are optimized to address high sample throughput, high precision and high accuracy. The samples are passed over a spectrophotometer that is set to specific wavelengths for reading N or P.

Nitrogen

Analytical Methods

The major objective of this study was to explore appropriate laboratory techniques for the analysis of Total N and nitrate concentrations of stream bank sediments, in order to elucidate the natural process of nutrient release in the surface water due to bank erosion.

Total N

Total N and Total C in stream bank sediments were measured by elemental combustion analyses and gas chromatography using a Costech ECS 4010 system according to the following analytical procedures:
1) Soil were oven-dried (80 degrees C, 24 hours).

2) Dried samples are ground to a fine powder (250 um or less) using a ball mill or ceramic mortar and pestle before being sealed into 5 x 9 mm tin capsules. Thorough sample homogenization in the grinder stage is required, to make certain that the subsample taken for analysis is representative of the total sample.

3) Roughly 25 micrograms of soil samples are weighed into pure tin capsules using an automated and computer controlled microbalance.

4) In addition to the calibration standards (see below), a certified standard reference soil (ECA 542) and an internal soils standard (BS-1 13) were analyzed every ten unknowns.

5) All samples, standards and blanks (empty tin capsules) we loaded in a 50-slot auto-changer carousel.

6) Automated analyses were controlled by Windows-based EAS Software with a multichannel 24 bit A/D interface connected to the electronic detection system in the ECA.

7) The ECS software compares the elemental peak to the calibration standard data, and generates a report for each element on a weight basis.

All carbon in the sample is converted CO_2 during flash combustion. Nitrogen-bearing combustion products include N_2 and various oxides of nitrogen NO_x, which pass through a reduction column filled with chopped Cu wire at heated to 600 degrees C, which transforms NO_x to N_2. Water vapor released from the sample is removed by a gas trap (d) that contains magnesium perchlorate. The cleaned sample gases pass through a gas chromatograph column, which separates the N_2 and CO_2. N_2 elutes from the GC column first, then CO_2.

The sample gas pulses and a separate reference stream of helium pass through a detector; differences in thermal conductivity between the two streams are displayed as visible peaks and recorded as numerically integrated areas. Linear regression applied to combustion of known standard materials yields a regression line by means of which peak areas from unknowns are converted into total element values for each sample.

The Costech ECS 4010 is calibrated by including five solid-phase certified reference materials in the tin capsule stage at the beginning of each run, and at fixed intervals thereafter (usually one reference standard per ten unknowns.) Ultra-high purity acetanilide (four samples in ca. 0.25, 0.50, 0.70 and 1.00 micrograms increments) and atropine (at ca. 0.1 micrograms) were used to generate the calibration curve; total C and total N contents of these materials are calculated from their chemical formulae.

Empty tin-capsule blanks are included ever tenth sample, and any detectable N or C in these blanks was subtracted from the sample and standard values to give a true zero baseline. Blanks allow correction for traces of C originating from the tin capsules and for the small amount of N_2 gas introduced as an impurity in the oxygen pulse.

Nitrate-N

Nitrate is soluble in water, thus water has the potential to be an effective solution to extract nitrate from soil. Yet, conventional soil nitrate analysis procedures rely heavily on using 2M KCl solution as the extractant, because KCl can effectively extract both nitrate and ammonium (Bremner and Keeney, 1966). We compared deionized (DI) water and 2M KCl as extractants to test lab methods for soil nitrate analysis and to model the quantities of soil nitrate released when bank sediments erode into the stream. We also used natural stream water and groundwater as extractants to determine how natural extractant that already has nitrate may influence the results of soil-nitrate extraction.

We divided the soil samples for extraction into the following three categories: field moist samples (FM), bulk air-dried samples (AD), and bulk oven-dried samples (OD) (Figure 11). Low-temperature oven-dried samples were often used in laboratory procedures for soil nitrate analysis, but we decided to incorporate field moist samples, which would better reflect the natural condition of stream bank sediment, and bulk air-dried samples, which would represent the exposed surface of bank soils that are baked dry by solar radiation.

Soil sampling and preparation

Stream bank sampling

Samples were collected from five stream bank sites along Big Spring Run (BSR) and their positions were mapped with high-precision GPS. Four of the sites are located within the restoration reach of BSR on the farm of Joseph Sweeney (now owned by the Kirchner Family (Sites 1, 4, 8, NYT) (Figure 1b), the fifth site is located on the Robert Houser property, upstream of the proposed restoration reach along the east branch of BSR (termed Houser Grid; Site HG) (Figure 1c). Before sampling, stream bank surfaces were cleared of debris and plant roots, and scraped clean with a trowel to remove all loose surface sediment. This process exposed fresh stream bank sediments. Samples were collected in 10 cm increments, approximately 300 - 500 g of sediment per interval, from the top of the bank downward to the top of the basal gravels (Sites 1, 4, NYT, and 8) or to the stream surface (Site HG). Sediments were collected using a stainless steel hand trowel, cleaned between each use, and the samples were stored in clean, unused "Ziploc" storage freezer bags. Each sample bag was labeled with the following information: (1) the sample location/site, (2) depth of the soil sample interval, (3) date, and (4) time of sampling. Each sample was mixed by hand from outside the bag to promote mixing and sample homogeneity. Sealed sample bags were stored in ice-filled coolers in the field – to slow down possible microbial activities that might alter the nitrate composition. In the laboratory, all samples were refrigerated at 4 °C (Weitzman, 2008).

Needle Ice sampling

We collected samples of needle ice from the sub-root zone at two sites (HG and NYT) along with ice blocks as field blank. Needle ice is pervasive in stream banks from December through February (Appendix A6). They were also stored and transported in individual "Ziploc" storage bags in ice-filled coolers, then refrigerated at 4°C until further analysis in the lab.

Sample preparation

Homogenized samples from each collection bag were subdivided into three preparation categories: field moist, bulk air-dried, and bulk oven-dried. Each category was subsequently analyzed for nitrate-nitrogen using a flow injection analyzer to test the effects of sample preparation methodologies on nitrate composition. Nitrate is highly reactive and its concentration in soils can be altered by heat and microbial activity. Because of the possibility of nitrate volatilization, only field moist and bulk air-dried samples were analyzed for Sites 1, 4, and HG.
Field moist samples were processed and analyzed directly from refrigerated samples within 24 hours of collection.

Roughly one-third of the material from each sample was transferred onto a labeled paper plate, which was covered by a large Kimwipe – to permit air circulation but reduce dust

contamination – and left for ca. one week to air dry. These air-dried samples were either processed immediately by flow injection analysis, or stored in air-tight plastic bottles. Soil nitrite may oxidize in the air to nitrate in the interim; in addition, aerobic microbes may either consume or produce nitrate (J. P. Kaye, PSU, personal communication, 2010).

The final one-third of each sample increment was transferred into an aluminum container and placed in a drying oven at 55 °C for 1 to 3 days. Upon completion, the samples were either processed immediately by flow injection analysis, or stored in air-tight plastic bottles for later analysis. Drying temperatures higher than 60 °C may result in the loss of nitrate due to volatilization (P. M. Mayer, EPA, personal communication, 2010).

Soil Nitrate Extraction

In general, four extractants were used to extract inorganic nitrate from soil samples: 1) 2M Potassium Chloride (KCl) solution, 2) deionized (DI) water, 3) low-nitrate BSR groundwater (GW), and 4) BSR surface water (SW). All samples were extracted using the KCl method, which is a standard procedure for agronomic soil analytical methods for determining plant available and labile NO_3. Various water extractants were used on selected samples as a test of the efficacy of these methods.

2M KCl Extraction

Dissolving 894 g KCl in 6 L DI water in a 10-L water jug made a 2M KCl solution. Approximately 5.0-g of soil sample was weighed into a capped centrifuge bottle. Visible contaminants such as plant roots, rocks, or small living organisms were extracted by hand. Each soil sample was vigorously shaken with 50 mL of 2M KCl solution for 30 minutes on a wrist-action shaker. The extracts were then filtered through Whatman no. 4 filter paper (qualitative). The filtered solution was then transferred to a 10-mL test-tube to be analyzed by flow injection spectrometry. Though rarely happened, if the solution contained visible sediment remnants, the filtering process would be repeated until a clear extraction was obtained (Krista, 2002).

DI Water/Groundwater/Stream Water Extraction

In this method, 50 mL of DI water was added to a 5.0-g soil sample in a capped centrifuge bottle. Each sample was vigorously shaken for 30 minutes on a wrist-action shaker, after which a few drops of $CaCl_2$ – a flocculent – was added to the solution to aggregate suspended colloids to precipitate (J. P. Kaye, personal communication, 2010). Prior to the addition of this flocculent, filtered samples remained cloudy from suspended colloids. The clear sample was then centrifuged (using an IEC Centrifuge model K) at 3,000 rpm for 5 minutes. Subsequently, the solvent was transferred by pipette to a 10-mL test-tube for flow injection analysis.

The groundwater extraction of soil nitrate followed the same procedures as the DI water extraction, with the exception that refrigerated, low nitrate-N groundwater from BSR was used as the extractant. Groundwater, collected from a shallow EPA monitoring well #200

(contained 0.6 mg/L Nitrate-N), was used to mimic the natural chemical matrix of stream water. These water extraction experiments were conducted to determine the amount of water-soluble nitrate in stream bank sediments, and to quantify the nitrate loading to surface water from bank erosion at Big Spring Run. Stream water collected from BSR (contained 7.5 mg/L Nitrate-N) was also used as an extractant to compare with the results of groundwater and DI water extractions.

A number of tests were performed to assess the effectiveness of using water as an extractant for nitrate in soil. First, DI water and 2M KCl solution were both used to extract soil nitrate from a separate set of field moist samples from site 1. Second, 5 air-dried samples (0-10 cm, 40-50 cm, 80-90 cm, 100-110 cm, 130-138 cm) were selected from Site 1 to be extracted with 2M KCl, DI water, groundwater, and stream water, respectively. Third, five 5.0-g air-dried samples (80-90 cm from site 1) were extracted with different volumes of DI water: 5 mL, 10 mL, 25 mL, 50 mL, and 100 mL.

Upon collection, GW and SW samples were preserved on ice and filtered in the laboratory within 24 hours. A portion of each sample was poured to a 60-mL disposable syringe and pushed through a 0.45 μm nylon filter into a 60 mL plastic bottle. The filtered solution was then acidified to pH = 2 with 2M sulfuric acid before it was refrigerated at 4 °C. The holding time of acidified water samples for nitrate analysis was 1-month.

Flow Injection Analysis

Two reagents were prepared prior to the analysis as calibration standards. Standards and samples were then loaded on the Lachat XYZ Auto Sampler ASX-40 series (Hach Company, Loveland, CO) and analyzed by the Lachat Flow Injection Analyzer Quickchem 2000 (Hach Company, Loveland, CO) via Lachat Reagent Pump RP-150 series 8500 (Hach Company, Loveland, Co). Each nitrate sample was converted to nitrite solution before it reacted with reagents to form a highly colored azo dye, which was then analyzed calorimetrically using a 520 nm photospectrometer.

Appendix 5. Data Files and URL Links

5a. USGS gage data

All original (including raw) USGS data are available in 9 Excel files:

1) Big_Spring_Sed_Loads_new_2009.xls
2) Big_Spring_Sed_Loads_new_2010.xls
3) Big_Spring_Sed_Loads_new_2012.xls
4) Big_Spring_Sed_Loads_new_2013.xls
5) BS_Q_T_2009.xlsx
6) BS_Q_T_2010.xlsx
7) BS_Q_T_2011.xlsx
8) BS_Q_T_2012.xlsx
9) Big_Spring_Run_Monitoring_Summary_2012.doc

Analysis of USGS data presented in Section E is provided in file named "Load Time Series All 3 Gages_DJM_12092013".

5b. Survey data

All survey data for channel cross sections, as well as code for doing erosion/deposition analysis, are available at

https://github.com/mrahnis/orangery

5c. Water Quality Data files are available in the following file folders

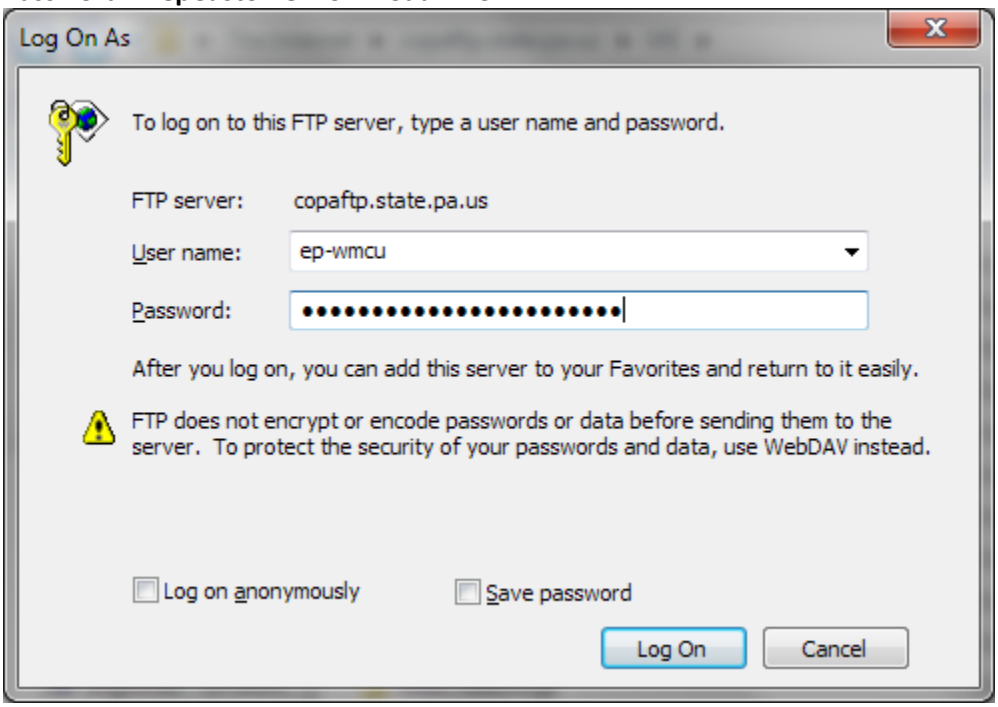
1) Big Spring Run EPA-F&M WQ Data_2009_031809
2) Big Spring Run EPA-F&M WQ Data_2009_080609
3) Big Spring Run EPA-F&M WQ Data_2009_111709
4) Big Spring Run EPA-F&M WQ Data_2010_011310
5) Big Spring Run EPA-F&M WQ Data_2010_072110
6) Big Spring Run EPA-F&M WQ Data_2011_060111
7) Big Spring Run EPA-F&M WQ Data_2012_071012
8) Big Spring Run EPA-F&M WQ Data_2012_091712
9) Big Spring Run EPA-F&M WQ Data_2012_110512
10) Big Spring Run EPA-F&M WQ Data_2013_011313
11) Big Spring Run EPA-F&M WQ Data_Sample Slam_ 4-7-10

5d. Trace-element geochemistry (fingerprinting)

Data files for 5a and 5c above may be accessed by following the ftp site retrieval directions below

Instructions to retrieve files are as follows:

1) Right-click the **Start** button and left-click **Open Windows Explorer**. Enter ftp://copaftp.state.pa.us in the address bar at the top. Hit enter.
2) Click the **File** menu and click **Login As**. If you do not have a File menu, left-click the **Organize** button, left-click **Layout** , and left-click **Menu bar**. The bar should now be displayed.
3) Enter the log on credentials as follows, and then click **Log On**.
   User name: ep-wmcu
   Password: DepCustomerDownload#143



4) Double-click the **WE** file folder.
5) Locate and double-click the Big_Spring_Run_Report_Documents file folder
6) Locate and double-click the Appendix 5 Data Files file folder
7) Locate and double-click file folders or files to access data

Appendix 6. Annotated list of videos of BSR restoration and monitoring

A. The Big Spring Run Restoration Project – Youtube Channel

Videos can be obtained or viewed at:
http://www.youtube.com/channel/UCorgwKIsH03jLRuTSF3Wxzg/about

This channel showcases active research conducted by the US EPA, PA DEP, USGS and scientists at Franklin & Marshall College and other academic institutions in regards to the floodplain-wetland restoration project with newly developing ecosystem at Big Spring Run in Lancaster County, Pennsylvania.

Videos:

BSR Restoration Process – Stream Channel Revealed September, 2011
- 1 image per minute (sunrise to sunset)
- Game Camera positioned on fence along edge of the restoration, facing west

Monitoring the Restoration at BSR
- Time-lapse of Franklin & Marshall students and Land Studies collecting data post-restoration at Big Spring Run June 21st, 2012

Time-lapse of Restoration – 9/16/2011 to 6/5/2013
- 1 image per day (taken between 10AM and 4PM)
- Game camera positioned at upper end of the restoration facing north

Time-lapse of Restoration (slow motion) – 9/16/2011 to 6/5/2013
- 1 image per day (taken between 10AM and 4PM)
- Game camera positioned at upper end of the restoration facing north

Storm Even at BSR 10/11/2013 (5:00PM - 6:30PM)
- 1 image per minute
- Images shot with mounted webcam at lowermost edge of restoration area, with camera facing south
- Water level rises quickly above channel banks, vegetation is submerged and flattened as water flows down valley.

Time-lapse Storm Event at BSR (web camera)—September 9th, 2012
- 1 image per minute (sunrise to sunset)

Using Helium Weather Balloon
- A clip of the team walking with a helium weather balloon fitted with a GoPro camera used to obtain aerial footage of the restoration at Big Spring Run on November 22nd 2013

B. Affiliated Videos:

Affiliated videos also can be viewed at:
http://www.youtube.com/channel/UCorgwKIsH03jLRuTSF3Wxzg/about

Diurnal needle ice formation, growth, and melting--Winter to Spring, 2011.

- 1 image per hour, including night images.
- With associated bank erosion.
- Includes flood in April 2012 that washes away detritus produced by previous freeze-thaw cycles.
- Uploaded by halomaster1989

Restoring Big Spring Run

- Uploaded on Aug 4, 2009 by Franklin and Marshall College
- About: This short film by Peter Cutler, a senior English major and film minor at Franklin & Marshall College, follows the summer research project of two Hackman Scholars, Katie Datin '11 and Erik Ohlson '10, who are working with earth and environment professors Dorothy Merritts and Robert Walter to restore a Lancaster County watershed

Big Spring Run Restoration – Trial 3D Model

- Published on Apr 19, 2012 by James Dietrich
- About: A Structure-from-Motion model of the Big Spring Run (PA) Restoration Project. Model by Mark Fonstad and James Dietrich, Univ. or Oregon. Restoration by Dorothy Merritts and Bob Walter at Franklin and Marshal College.

Fly over Big Spring Run restoration site

- Published on Mar 25, 2012 by Eugene Potapov
- About: You can see Dr. Robert Walter and Dr. Candace Grand Pre with students and visitors at the Big Spring Run restoration site.

Big Spring Run stream restoration project in West Lampeter Township

- Uploaded on Oct 17, 2011 by LancasterOnline
- About: What may be the future of stream cleanup in the Chesapeake Bay is currently a wide swath of exposed dirt on a scenic West Lampeter Township farm.

Appendix 7: Pre-Restoration and Post-Restoration Photos of Reach with Cross Sections 6 and 7 to Illustrate Erosion and Deposition in Stream Corridor with Time (contained in Figure File)

Appendix 8. References Cited

Adams, S.M., Ryon, M.G., and Smith, J.G. 2005. Recovery in diversity of fish and invertebrate communities following remediation of a polluted stream: investigating causal relationships. In: Developments in Hydrobiology, Aquatic Biodiversity II (Segers, H., and Martens, K., editors). Springer Netherlands 180: 77-93.

Ahamed, Aakash, 2012, Gravel Transport in a Floodplain Wetland Restoration Experiment, Big Spring Run, Lancaster PA: Senior thesis, Franklin and Marshall College, 45 p.

Allan, J.D. 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. *Annu. Rev. Ecol. Evol. Syst*., 35:257-284.

Angermeier, P.L. and M.R. Winston. 1998. Local vs. regional influences on local diversity in stream fish communities in Virginia. *Ecology*, 79(3):911-927.

Asner, Gregory P., Timothy R. Seastedt, and Alan R. Townsend. 1997. "The Decoupling of Terrestrial Carbon and Nitrogen Cycles." *BioScience* 47 (4): 226–34.

Banks, W. S. L., A. C. Gellis, and G. Noe. 2010. "Sources of Fine-Grained Suspended Sediment in Mill Stream Branch Watershed, Corsica River Basin, a Tributary to the Chesapeake Bay, Maryland, 2009." *Proceedings of the 2nd Joint Federal Interagency, Las Vegas, NV* 27.

Banks, W. S. L., A. C. Gellis, and G. Noe. 2010. "Sources of Fine-Grained Suspended Sediment in Mill Stream Branch Watershed, Corsica River Basin, a Tributary to the Chesapeake Bay, Maryland, 2009." *Proceedings of the 2nd Joint Federal Interagency, Las Vegas, NV* 27.

Barbour, M.T., J. Gerritsen, B.D. Snyder and J.B. Stribling. 1999. Rapid bioassessment protocols For use in streams and wadeable rivers: Periphyton, Benthic Macroinvertebrates, and Fish, 2nd Edition. http://www.epa.gov/OWOW/monitoring/techmon.html.

Beck M. W. (1995) Size-specific shelter limitation in stone crabs: a test of the demographic bottleneck hypothesis. *Ecology* 76, 968–980.

Becker, Steven, 2011, Needle Ice and its contribution to stream bank erosion at Big Spring Run Lancaster, PA, Senior thesis, Franklin and Marshall College, 44 p.

Bennett, Elena M., Stephen R. Carpenter, and Nina F. Caraco. 2001. "Human Impact on Erodable Phosphorus and Eutrophication: A Global Perspective Increasing Accumulation of Phosphorus in Soil Threatens Rivers, Lakes, and Coastal Oceans with Eutrophication." *BioScience* 51 (3): 227–34.

Bennett, H. H. 1931. "Cultural Changes in Soils from the Standpoint of Erosion." *Journal of the American Society of Agronomy*.

Biggs, J., A. Corfield, P. Grøn, H.O. Hansen, D. Walker, M. Whitfield, and P. Williams. 1998. Restoration of the rivers Brede, Cole and Skerne: a joint Danish and British EU-LIFE demonstration project, V—short-term impacts on the conservation value of aquatic macroinvertebrate and macrophyte assemblages', *Aquatic ConserTation: Marine and Freshwater Ecosystems,* 8, 241–254.

Boesch, D. F, R. B Brinsfield, and R. E Magnien. 2001. "Chesapeake Bay Eutrophication: Scientific Understanding, Ecosystem Restoration, and Challenges for Agriculture." *J. Environ. Qual* 30 (2): 303–20.

Bond, N.R. and P.S. Lake. 2003. Local habitat restoration in streams: constraints on the effectiveness of restoration for stream biota. *Ecol. Mgmt & Restor*., 4(3): 193-198.

Bonvincini, A.M., A.M. Cognetti Varriale, R. Crema, M. Curini Galetti, and R. Vandini Zunarelli. 1985. Environmental impact of extensive dredging in a coastal marina. *Marine Pollution Bulletin* 16: 483-488.

Boomer, Kathleen B., Donald E. Weller, and Thomas E. Jordan. 2008. "Empirical Models Based on the Universal Soil Loss Equation Fail to Predict Sediment Discharges from Chesapeake Bay Catchments." *Journal of Environmental Quality* 37 (1): 79–89. doi:10.2134/jeq2007.0094.

Boomer, Kathleen, Donald E. Weller, Thomas E. Jordan, Lewis Linker, Zhi-Jun Liu, James Reilly, Gary Shenk, and Alexey A. Voinov. 2013. "Using Multiple Watershed Models to Predict Water, Nitrogen, and Phosphorus Discharges to the Patuxent Estuary1." *JAWRA Journal of the American Water Resources Association* 49 (1): 15–39.

Boulton, Andrew J., Stuart Findlay, Pierre Marmonier, Emily H. Stanley, and H. Maurice Valett. 1998. "The Functional Significance of the Hyporheic Zone in Streams and Rivers." *Annual Review of Ecology and Systematics* 29 (1): 59–81.

Boyle, F. W., and W. L. Lindsay. 1986. "Manganese Phosphate Equilibrium Relationships in Soils." *Soil Science Society of America Journal* 50 (3): 588–93.

Brady, N. C., and R. R. Weil. *The Nature and Properties of Soils . 2002*. Upper Saddle River, Prentice Hall, NJ.

Brantley, SL, J.P. Megonigal, FN Scatena, Z. BALOGH-BRUNSTAD, RT Barnes, MA Bruns, P. Van Cappellen, et al. 2011. "Twelve Testable Hypotheses on the Geobiology of Weathering." *Geobiology*.

Brenner, Fred J., and James J. Mondok. 1995. "Nonpoint source pollution potential in an agricultural watershed in northwestern Pennsylvania." *JAWRA Journal of the American Water Resources Association* 31 (6): 1101–12.

Bricker, Owen P., and John H. Moss. 1958. "Origin of the Marsh, East Nantmeal Township, Chester County, Pennsylvania." *Proceedings of the Pennsylvania Academy of Science* 32: 168–71.

Bridgens, H. F., 1864, Bridgen's atlas of Lancaster County, Pennsylvania: From actual surveys by H. F. Bridgens and assistant. Lancaster, PA: D. S. Bare.

Brooks S.S., M.A. Palmer, B.J. Cardinale, C.M. Swan and S. Ribblett. 2002. Assessing stream ecosystem rehabilitation: limitations of community structure data. *Restoration Ecology*, 10, 156–168.

Brown B.L. 2003. Spatial heterogeneity reduces temporal variability in stream insect communities. *Ecology Letters*, 6, 316–325.

Brush, G. S., 1989, Rates and patterns of estuarine sediment accumulation: Limnology and Oceanography, v. 34, p. 1235–1246.

Cantelli, Alessandro, Chris Paola, and Gary Parker. 2004. "Experiments on Upstream-Migrating Erosional Narrowing and Widening of an Incisional Channel Caused by Dam Removal." *Water Resources Research* 40: W03304. doi:10.1029/2003WR002940.

Cleveland, Cory C., and Daniel Liptzin. 2007. "C: N: P Stoichiometry in Soil: Is There a 'Redfield Ratio' for the Microbial Biomass?" *Biogeochemistry* 85 (3): 235–52.

Coale, Frank J., J. Thomas Sims, and April B. Leytem. 2002. "Accelerated Deployment of an Agricultural Nutrient Management Tool." *Journal of Environmental Quality* 31 (5): 1471–76.

Collins, A. L., and D. E. Walling. 2002. "Selecting Fingerprint Properties for Discriminating Potential Suspended Sediment Sources in River Basins." *Journal of Hydrology* 261 (1-4): 218–44. doi:10.1016/S0022-1694(02)00011-2.
Correll, David L., Thomas E. Jordan, and Donald E. Weller. 1992. "Nutrient Flux in a Landscape: Effects of Coastal Land Use and Terrestrial Community Mosaic on Nutrient Transport to Coastal Waters." *Estuaries* 15 (4): 431–42.
Costa, John E. 1975. "Effects of Agriculture on Erosion and Sedimentation in the Piedmont Province, Maryland." *Geological Society of America Bulletin* 86 (9): 1281–86.
Cowardin, L.M., V. Carter, F.C. Golet and E.T. LaRoe, 1979. Classification of wetlands and deepwater habitats of the United States. U.S. Department of the Interior, Washington D.C. FWS/OBS-79/31.
Cuffney, T.F., H. Zappia, E.M.P. Giddings and J.F. Coles. 2005. Assemblages in contrasting environmental settings: Boston, Massachusetts; Birmingham, Alabama; and Salt Lake City, Utah. American Fisheries Society Symposium, 47:361-407.
Cuffney, T.F., R.A. Brightbill, J.T. May, and I.R. Waite. 2010. Responses of benthic macroinvertebrates to environmental changes associated with urbanization in nine metropolitan areas. *Ecol. Appl.,* 20(5): 1384-1401.
Department of Environmental Protection. 2012. 2012 Pennsylvania integrated water quality monitoring and assessment report. Clean Water Act, Section 305(b) Report and 303(d) list. 67pp.
Devereux, Olivia H., Karen L. Prestegaard, Brian A. Needelman, and Allen C. Gellis. 2010. "Suspended-sediment Sources in an Urban Watershed, Northeast Branch Anacostia River, Maryland." *Hydrological Processes* 24 (11): 1391–1403.
Devereux, Olivia H., Karen L. Prestegaard, Brian A. Needelman, and Allen C. Gellis. 2010. "Suspended-sediment Sources in an Urban Watershed, Northeast Branch Anacostia River, Maryland." *Hydrological Processes* 24 (11): 1391–1403.
Dillon, P. J., and W. B. Kirchner. 1975. "The Effects of Geology and Land Use on the Export of Phosphorus from Watersheds." *Water Research* 9 (2): 135–48.
Doeg, T.J. and Koehn, J.D. 1994. 'Effects of draining and desiltation of a small weir on downstream fish and macroinvertebrates', *Regulated RiTers: Research and Management,* 9, 263–277.
Emerson, Steve. 1976. "Early Diagenesis in Anaerobic Lake Sediments: Chemical Equilibria in Interstitial Waters." *Geochimica et Cosmochimica Acta* 40 (8): 925–34.
Emerson, Steven, and Gerhard Widmer. 1978. "Early Diagenesis in Anaerobic Lake sediments—II. Thermodynamic and Kinetic Factors Controlling the Formation of Iron Phosphate." *Geochimica et Cosmochimica Acta* 42 (9): 1307–16.
Essington, Michael E. 2004. *Soil and Water Chemistry: An Integrative Approach*. CRC press.
Fike, J. 1999. Terrestrial and palustrine plant communities of Pennsylvania. Harrisburg, PA: Pennsylvania Department of Conservation and Natural Resources, Bureau of Forestry. See http://www.dcnr.state.pa.us/forestry/sfrmp/plants.htm.
Friberg, N., B. Kronvang, H.O. Hansen and L.M. Svendsen. 1998. Long-term, habitat-specific response of a macroinvertebrate community to river restoration. *Aquatic Conserv: Mar. Freshw. Ecosyst.* 8: 87 – 99.
Galeone, Daniel, Brightbill, Robin, Low, Dennis, O'Brien, David, 2006, Effects of streambank fencing of pasture land on benthic macroinvertebrates and the quality of surface water and shallow ground water in the Big Spring Run Basin of Mill Creek

watershed, Lancaster County, Pennsylvania, 1993-2001: U.S. Geological Survey Scientific Investigations Report 2006-5141, 184 p.
Gambrell, R. P., J. W. Gilliam, and S. B. Weed. 1975. "Nitrogen Losses from Soils of the North Carolina Coastal Plain." *Journal of Environmental Quality* 4 (3): 317–23.
Gellis, A. C., and D. E. Walling. 2011. "Sediment Source Fingerprinting (tracing) and Sediment Budgets as Tools in Targeting River and Watershed Restoration Programs." *Geophysical Monograph Series* 194: 263–91.
Gellis, A. C., and J. M. Landwehr. 2006. "Identifying Sources of Fine-Grained Suspended Sediment in the Pocomoke River, an Eastern Shore Tributary to the Chesapeake Bay." In *Proceedings of the 8th Federal Interagency Sedimentation Conference, Reno, Nevada*, 2–6.
Gellis, Allen C., Cliff R. Hupp, Milan J. Pavich, Jurate M. Landwehr, William SL Banks, Bernard E. Hubbard, Michael J. Langland, Jerry C. Ritchie, and Joanna M. Reuter. 2009. *Sources, Transport, and Storage of Sediment at Selected Sites in the Chesapeake Bay Watershed*. US Geological Survey.
Gerard, K.J. and R.A. Hellenthal. 2003. Response of aquatic invertebrates to a stream restoration in northern Indiana. *Bulletin of the North American Benthological Society*. 20(1):364.
Gilkinson, K.D., D.C. Gordon Jr., K.G. MacIsaac, D.L. McKeown, E.L.R. Kenchington, C. Bourbonnais, and W.P. Vass. 2005. Immediate impacts and recovery trajectories of macrofaunal communities following hydraulic clam dredging on Banquereau, eastern Canada. ICES. *J. of Mar. Sci*., 62(5): 925-947.
Gleason, H.A. and A.C. Cronquist, 1991. Manual of vascular plants of Northeaster United States and Adjacent Canada, 2nd edition. The New York Botanical Garden, Bronx, New York 10458, USA.
Grand Pre, Candace A., Robert C. Walter, Dorothy J. Merritts, Amy C. Moser, Christopher Bernhardt, Marina Potopova, and William B. Hilgartner. 2012. "A High-Resolution Multi-proxy Record of Late Pleistocene and Holocene Climate Change from Great Marsh, Southeastern PA." In *2012 GSA Annual Meeting in Charlotte*. *North Carolina.*
Grobler, D. C., and M. J. Silberbauer. 1985. "The Combined Effect of Geology, Phosphate Sources and Runoff on Phosphate Export from Drainage Basins." *Water Research* 19 (8): 975–81.
Happ, Stafford C. 1945. "Sedimentation in South Carolina Piedmont Valleys." *American Journal of Science* 243 (3): 113–26.
Hartranft, Jeffrey, Merritts, Dorothy, Walter, Robert, Rahnis, Michael, 2011, The Big Spring Run restoration experiment: Policy, geomorphology, and aquatic ecosystems in the Big Spring Run watershed, Lancaster County, PA: Sustain: A Journal of Environmental and Sustainability Issues: Issue 24, p. 24-30. http://louisville.edu/kiesd/sustain-magazine.
Hilgartner, W.B., Dorothy Merritts, Robert Walter, Michael Rahnis, Christopher Bernhardt, Jeff Hartranft, Ali Neugebauer, Mark Voli, Hanna Jantzi, Amy Moser, and Candace Grand Pre. 2012. Millennial Pre-settlement Stability of Sedge Meadow Habitats in Two Piedmont River Valleys. 9th INTECOL International Wetlands Conference, Orlando, FL, June 3-8, 2012. (Presentation 6/7)).
Hill, A. R. 1978. "Factors Affecting the Export of Nitrate-Nitrogen from Drainage Basins in Southern Ontario." *Water Research* 12 (12): 1045–57.

Hill, R.J., 1983a. Regulatory horticulture weed circular number 2. Pennsylvania Department of Agriculture, Vol. 9, No. 1-2.
Hill, R.J., 1983b. Regulatory horticulture weed circular number 6. Pennsylvania Department of Agriculture, Vol. 9, No. 1-2.
Hoiland, W.K., R.W. Rabe, and R.C.Biggam. 1994. Recovery of macroinvertebrate communities from metal pollution in the South Fork and main stem of the Coeur d'Alene River, Idaho. *Water Environment Research* 66(1): 84-88.
Jacobson, Robert B., and Derrick J. Coleman. 1986. "Stratigraphy and Recent Evolution of Maryland Piedmont Flood Plains." *American Journal of Science* 286 (8): 617–37.
Johnson, K. S. 2007. Field and Laboratory Methods for using the MAIS (Macroinvertebrate Aggregated Index for Streams) in Rapid Bioassessment of Ohio Streams. Ohio University Department of Biological Sciences. Athens, OH. 18 pp.
Jordan, Thomas E., and Donald E. Weller. 1996. "Human Contributions to Terrestrial Nitrogen Flux." *BioScience*: 655–64.
Jordan, Thomas E., David L. Correll, and Donald E. Weller. 1997a. "Relating Nutrient Discharges from Watersheds to Land Use and Streamflow Variability." *Water Resources Research* 33 (11): 2579–90.
Jordan, Thomas E., David L. Correll, and Donald E. Weller. 1997b. "Effects of Agriculture on Discharges of Nutrients from Coastal Plain Watersheds of Chesapeake Bay." *Journal of Environmental Quality* 26 (3): 836–48.
Julian, J. P. & Torres, R., 2006, Hydraulic erosion of cohesive riverbanks: Geomorphology, v. 76, p. 193–206.
Kenney, M.A., A.E. Sutton-Grier, Smith, and S.E. Gresens. 2009. Benthic macroinvertebrates as indicators of water quality: the intersection of science and policy. *Terrestrial Arthropod Reviews* 2: 99-128.
Knighton, D., 1998, Fluvial forms and processes. London, UK: Hodder Arnold.
Kratz, Laura, 2012, Unraveling the Geomorphic Evolution of Enigmatic Valley Bottom Deposits in Lancaster County, Pennsylvania: Senior thesis, Franklin and Marshall College, 55 p.
Kronvang, B., Svendsen, L.M., Ottosen, O., Nielsen, M.B. and Johannesen, L. 1998. 'Re-meandering of rivers: short-term implications for sediment and nutrient transport', *Verhandlungen der Internationalen Vereinigung fu¨rtheoretische und angewandte Limnologie.*
Lemly, A.D. 1982. Modification of benthic insect communities in polluted streams: combined effects of sedimentation and nutrient enrichment. *Hydrobiologia*, 87:229-245.
Lenat, D.R. and J.K. Crawford. 1994. Effects of land-use on water quality and aquatic biota of 3 North Carolina Piedmont streams. *Hydrobiologia*, 294:185-199.
Lawler, D. 1993 Needle ice processes and sediment mobilization on river banks: the River Ilston, West Glamorgan, UK. J. Hydrol. 150, 81–114.
Lawler, D. M., 1986, River bank erosion and the influence of frost: A statistical examination: Transactions of the Institute of British Geography: v. 11, p. 227–242.
Leopold, Luna B., and M. Gordon Wolman. 1957. *River Channel Patterns: Braided, Meandering, and Straight*. USGS Professional Paper 282-B. Washington DC: U.S. Government Printing Office. http://pubs.er.usgs.gov/usgspubs/pp/pp282B.

Lewin, J., 2010, Medieval environmental impacts and feedbacks: The lowland floodplains of England and Wales: Geoarchaeology, v. 25, p. 267–311.
Luscombe, P. C., J. K. Syers, and P. E. H. Gregg. 1979. "Water Extraction as a Soil-testing Procedure for Phosphate." *Communications in Soil Science & Plant Analysis* 10 (11): 1361–69.
Martin, Paul Schultz. 1958. "Taiga-Tundra and the Full-Glacial Period in Chester County, Pennsylvania." *American Journal of Science* 256 (7): 470–502.
Mason, John W., Gerald D. Wegner, Gregory I. Quinn, and Eugene L. Lange. 1990. "Nutrient Loss via Groundwater Discharge from Small Watersheds in Southwestern and South Central Wisconsin." *Journal of Soil and Water Conservation* 45 (2): 327–31.
Massoudieh, Arash, Allen Gellis, William S. Banks, and Michael E. Wieczorek. 2013. "Suspended Sediment Source Apportionment in Chesapeake Bay Watershed Using Bayesian Chemical Mass Balance Receptor Modeling." *Hydrological Processes* 27 (24): 3363–74.
Massoudieh, Arash, Allen Gellis, William S. Banks, and Michael E. Wieczorek. 2013. "Suspended Sediment Source Apportionment in Chesapeake Bay Watershed Using Bayesian Chemical Mass Balance Receptor Modeling." *Hydrological Processes* 27 (24): 3363–74.
Mayer, Paul M., Peter M. Groffman, Elise A. Striz, and Sujay S. Kaushal. 2010. "Nitrogen Dynamics at the Groundwater-Surface Water Interface of a Degraded Urban Stream." *Journal of Environmental Quality* 39 (3): 810–23. doi:10.2134/jeq2009.0012.
McIntyre, N. E. 2000. Ecology of urban arthropods: a review and a call to action. *Annals Entomol. Soc. Am.,* 93:825–835.
Merritt, R.W., K.W. Cummins, and M.B. Berg, (eds.) 2008. An. Introduction to aquatic insects of North America, (4th ed). Kendall Hunt Publishing, Dubuque, IA. 1158pp.
Merritts, Dorothy, Robert Walter, Michael Rahnis, Jeff Hartranft, Scott Cox, Allen Gellis, Noel Potter, William Hilgartner, Michael Langland, and Lauren Manion. 2011. "Anthropocene Streams and Base-Level Controls from Historic Dams in the Unglaciated Mid-Atlantic Region, USA." *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences* 369 (1938): 976–1009.
Merritts, Dorothy, Robert Walter, Michael Rahnis, Scott Cox, Jeffrey Hartranft, Chris Scheid, Noel Potter, et al, 2013, The rise and fall of Mid-Atlantic streams: Millpond sedimentation, milldam breaching, channel incision, and stream bank erosion." In *Reviews in Engineering Geology,* v. 21, p. 183-203.
Merritts, Dorothy, Walter, Robert, Rahnis, Michael, Hartranft, Jeff, Cox, Scott, Gellis, Allen, Potter, Noel, Hilgartner, William, Langland, Michael, Manion, Lauren, Lippincott, Caitlin, Siddiqui, Sauleh, Rehman, Zain, Scheid, Chris, Kratz, Laura, Shilling, Andrea, Jenschke, Matthew, Reed, Austin, Matuszewski, Derek, Voli, Mark, Datin, Katherine, Ohlson, Erik, Neugebauer, Ali, Ahamed, Aakash, Neal, Conor, Winter, Allison, and Becker, Steven, 2011, Anthropocene streams and base-level controls from historic dams in the unglaciated mid-Atlantic region, USA: Phil. Trans. R. Soc. A, v. 369, p. 1–34. (One contribution of 13 to a Theme Issue 'The Anthropocene: a new epoch of geological time?') http://rsta.royalsocietypublishing.org/content/369/1938.toc
Meyer, J. L., M. J. Paul, and W. K. Taulbee. 2005. Stream ecosystem function in urbanizing landscapes. *J. North Am. Benth. Soc*., 24:602-612.

Milner, A.M. 1996. System recovery, in Petts, G. and Calow P. (Eds), *RiTer Restoration,* Blackwell Science, Oxford, 205 – 226.
Minshall, W.A. 1984. Aquatic insect-substratum relationships. Ecology of Aquatic Insects. Praeger Press, N.Y. p. 358.
Mitsch, William J. and James G. Gosselink. 2007. *Wetlands, 4th ed.*, John Wiley & Sons, Inc., New York, 582 pp
Mitsch, William J. and James G. Gosselink.. 2000. "The Value of Wetlands: Importance of Scale and Landscape Setting." *Ecological Economics* 35 (1): 25–33.
Moerke, A.H., K.J. Gerard, J.A. Latimore, R.A. Hellenthal and G.A. Lamberti. 2004. Restoration of an Indiana, USA, stream: bridging the gap between basic and applied lotic ecology. *J. North Am. Benthol. Soc.,*23, 647–660.
Moeykens, M.D. 2002. Studies of benthic macroinvertebrate use for biomonitoring of Mid-Atlantic Highland streams. Ph.D. Dissertation, Department of Entomology, Virginia Tech University. 131 pp.
Moore, A., and K. R. Reddy. 1994. "Role of Eh and pH on Phosphorus Geochemistry in Sediments of Lake Okeechobee, Florida." *Journal of Environmental Quality* 23 (5): 955–64.
Moore A. A. and M.A. Palmer, 2005. Invertebrate biodiversity in agricultural and urban head-water streams: implications for conservation and management. *Ecol. Appl*., 15(4):1169-1177.
Moore, P., and F. Coale. 2009. "Phosphorus Fractionation in Flooded Soils and Sediments." *Methods of Phosphorus Analysis for Soils, Sediments, Residuals, and Waters Second Edition*: 61.
Morse, C. C., A. D. Huryn, and C. Cronan. 2003. Impervious surface area as a predictor of the effects of urbanization on stream insect communities in Maine, U.S.A. Environmental Monitoring and Assessment 89:95–127.
Motha, J. A., P. J. Wallbrink, P. B. Hairsine, and R. B. Grayson. 2003. "Determining the Sources of Suspended Sediment in a Forested Catchment in Southeastern Australia." *Water Resources Research* 39 (3).
Mukundan, Rajith, Desmond E. Walling, Allen C. Gellis, Michael C. Slattery, and David E. Radcliffe. 2012. "Sediment Source Fingerprinting: Transforming From a Research Tool to a Management Tool1." *JAWRA Journal of the American Water Resources Association* 48 (6): 1241–57.
Murphy, James, and J. P. Riley. 1962. "A Modified Single Solution Method for the Determination of Phosphate in Natural Waters." *Analytica Chimica Acta* 27: 31–36.
Nanson, G. C. & Knighton, A. D., 1996, Anabranching rivers: Their cause, character and classification: Earth Surface Processes and Landforms, v. 21, p. 217–239.
National Research Council. 1992. Restoration of aquatic ecosystems: science, technology and public policy. National Academy Press, Washington, DC. 552pp.
Nearing, M. A., R. M. Risse, and L. F. Rogers. 1993. "Estimating Daily Nutrient Fluxes to a Large Piedmont Reservoir from Limited Tributary Data." *Journal of Environmental Quality* 22 (4): 666–71.
Neill, Michael. 1989. "Nitrate Concentrations in River Waters in the South-East of Ireland and Their Relationship with Agricultural Practice." *Water Research* 23 (11): 1339–55.

Nederveld, L.B. 2009. Sediment remediation impacts on macroinvertebrate community structure: Assessing the success of urban stream restoration. Master's Thesis, Biology Department, Grand Valley State University, Allendale, MI. 77 pp.
Nelson, S.M. and Roline, R.A. 1996. Recovery of a stream macroinvertebrate community from mine drainage disturbance. *Hydrobiologia* 339: 73-84.
Neugebauer, Ali, 2011, Paleoecological Reconstruction of Big Spring Run, Lancaster, PA: Senior thesis, 43 p.
Niemi, G.J., P. DeVore, N. Detenbeck, J.D. Yount, A. Lima, J. Pastor, and R.J. Naiman. 1990. Overview of case studies on recovery of aquatic systems from disturbance', *Environmental Management,* 14, 571–588.
Nixon, Scott W. 1995. "Coastal Marine Eutrophication: A Definition, Social Causes, and Future Concerns." *Ophelia* 41 (1): 199–219.
Overstreet, W.C., A. M. White, J.W. Whitlow,, P.K. Theobald,, D.W. Caldwell, and N.P. Cuppels,. 1968. *Fluvial Monazite Deposits in the Southeastern United States*. US Government Printing Office.
Palmer M.A., C.C. Hakenkamp, K. Nelson-Baker. (1997) Ecological heterogeneity in streams: why variance matters. *J. North Am. Benthol. Soc*., 16, 189–202.
Palmer, M.A., HL. Menninger, and E. Bernhardt. 2010. River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? *Freshwater Biology* 55: (1-18).
Palmer, M.A. and L. Wainger. 2011. Promoting successful restoration through effective monitoring in the Chesapeake Bay Watershed. National Fish & Wildlife Federation, Wash. DC., FinalReport, 24pp.
Patrick, W. H., S. Gotoh, and B. G. Williams. 1973. "Strengite Dissolution in Flooded Soils and Sediments." *Science* 179 (4073): 564–65.
Paul, M. J., and J. L. Meyer. 2001. Streams in the urban landscape. *Annu. Rev. of Ecol. and Syst*., 32:333-365.
Peckarsky, B.L., P.R. Fraissinet, M.A. Penton, D.J. Conklin, JR. 1990. Freshwater macro-invertebrates of northeastern North America. Cornell University Press, Ithaca, NY. 442 pp.
Pennsylvania Department of Agriculture, 2013. Pennsylvania noxious weeds control list. http://www.agriculture.state.pa.us
Petranka, J. 1988. Salamanders of the United States and Canada. Smithsonian Institution Press.
Pierzynski, Gary M. 2000. *Methods of Phosphorus Analysis for Soils, Sediments, Residuals, and Waters*. North Carolina State University Raleigh.
Pierzynski, Gary M., and Andrew N. Sharpley. 2000. *Methods of Phosphorus Analysis for Soils, Sediments, Residuals, and Waters*. North Carolina State University Raleigh.
Pionke, Harry B., William J. Gburek, and Andrew N. Sharpley. 2000. "Critical Source Area Controls on Water Quality in an Agricultural Watershed Located in the Chesapeake Basin." *Ecological Engineering* 14 (4): 325–35.
Pittman, J. J., H. Zhang, J. L. Schroder, and M. E. Payton. 2005. "Differences of Phosphorus in Mehlich 3 Extracts Determined by Colorimetric and Spectroscopic Methods." *Communications in Soil Science and Plant Analysis* 36 (11-12): 1641–59.

Pizzuto, Jim, and Michael O'Neal. 2009. "Increased Mid-Twentieth Century Riverbank Erosion Rates Related to the Demise of Mill Dams, South River, Virginia." *Geology* 37 (1): 19–22.

Pote, D. H., T. C. Daniel, P. A. Moore, D. J. Nichols, A. N. Sharpley, and D. R. Edwards. 1996. "Relating Extractable Soil Phosphorus to Phosphorus Losses in Runoff." *Soil Science Society of America Journal* 60 (3): 855–59.

Purcell, A.H., C. Friedrich, V.H. Resh. 2002. An assessment of a small urban stream restoration project in northern California. Restoration Ecology. 10(4):685-694.

Quigley, M.P. and J.A. Hall. 1999. Recovery of macrobenthic communities after maintenance dredging in the Blyth estuary, north-east England. *Aquatic Conservation: Mar. and Freshw. Ecosys.*, 9(1): 63-73.

Reppert, J, B. Kondrikoff and J.R. Wallace. 2005. Impact of stream and floodplain rehabilitation Abstract, North American Benthological Society, New Orleans, LA.

Relyea, C.D., G.W. Minshall, R.J. Danehy. 2000. Stream insects as bio-indicators of fine sediment. Proceedings from Waste Management Conference. Water Environment Federation. 19pp.

Reice, S.R. 1985. 'Experimental disturbance and the maintenance of species diversity in a stream community', *Oecologia,* 67, 191–195.

Rhoads, A.F. & T. Block, 2000. The Plants of Pennsylvania. Philadelphia: University of Pennsylvania Press.

Rinkevich, JD and JR Wallace. 2001, Community analysis of the impacts of stream channel restoration on aquatic insects and fish in southeastern Pennsylvania. Entomological Society of America Poster Abstract from San Diego, CA

Roy A.H., A.D. Rosemond, M.J. Paul, D.S. Leigh, and J.B. Wallace. 2003. Stream macroinvertebrate response to catchment urbanization (Georgia, USA). *Freshw. Biol.* 48:329–46.

Scarlett, L. and J. Boyd. 2011. Ecosystem Services: quantification, policy applications and current federal capabilities. RFF DP: 75pp.

Self-Davis, M. L., P. A. Moore Jr., and B. C. Joern. 2000. "Determination of Water-And/or Dilute Salt-Extractable Phosphorus." *Methods of Phosphorus Analysis for Soils, Sediments, Residuals, and Waters. Southern Coop. Ser. Bull* 396: 24–26.

Sharpley, Andrew N. 2003. *Agricultural Phosphorus and Eutrophication*. US Department of Agriculture, Agricultural Research Service.

Shilling, Andrea, 2010, Rates and processes of stream bank erosion and its contribution to suspended sediment load, Big Spring Run, PA, Senior thesis, Franklin and Marshall College, 82 p.

Simon, A., Curini, A., Darby, S. E. & Langendoen, E. J., 2000, Bank and near-bank processes in an incised channel: Geomorphology, v. 35, p. 193–217.

Smith, E.P., and J.R. Voshell, Jr. 1997. Final Report - Studies of Benthic Macroinvertebrates and Fish in Streams Within EPA Region 3 for Development of Biological Indicators of Ecological Condition. Part I - Benthic Macroinvertebrates. Department of Statistics and Department of Entomology, Virginia Polytechnic Institute and State University, Blacksburg, Virginia 24061. July 24, 1997.

Smith, R.F., L.C. Alexander, W.O. Lamp. 2009. Dispersal by terrestrial stages of stream insects in Urban watersheds: a synthesis of current knowledge. J. North Am. Benthol. Soc., 28(4):1022-1037.

Smith, Sean MC, Patrick Belmont, and Peter R. Wilcock. 2011. "Closing the Gap between Watershed Modeling, Sediment Budgeting, and Stream Restoration." *Geophysical Monograph Series* 194: 293–317.
Spalding, R. F, and M. E Exner. 1993. "Occurrence of Nitrate in Groundwater- a Review." *Journal of Environmental Quality* 22 (3): 392–402.
Sponseller, R.A., E.F. Benfield, and H.M. Valett. 2001. Relationships between landuse, spatial scale and stream macroinvertebrate communities. *Freshwater Biology* 46:1409-1424.
Stewart-Oaten, A., J.R. Bence and C.W. Osenberg. 1992. Assessing the effects of unreplicated perturbations: no simple solutions. *Ecology*, 73:1396-1404 al. 1993.
Strouse, Stephanie R. 2013. *The Effect of Millponds on Sedimentation in a Post-Glacial Mid-Coast Maine River Valley*. Boston College.
Sullivan, Alexandra. 2006. "Geomorphology in the Agricultural Watershed of Big Spring Run, Lancaster, PA: An Integrative Approach to Erosion Research and Remediation". Bachelor's Thesis, Franklin & Marshall College.
Sweeney, B. W., T. L. Bott, J. K. Jackson, L. A. Kaplan, J. D. Newbold, L. J. Standley, W. C. Hession, and R. J. Horwitz. 2004. Riparian deforestation, stream narrowing, and loss of stream ecosystem services. *Proc. Natl. Acad. Sci. of the USA*, 101:14132-14137.
Taylor, Philip G., and Alan R. Townsend. 2010. "Stoichiometric Control of Organic Carbon–nitrate Relationships from Soils to the Sea." *Nature* 464 (7292): 1178–81. doi:10.1038/nature08985.
Thomas, Grant W., Gerald R. Haszler, and James D. Crutchfield. 1992. "Nitrate-Nitrogen and Phosphate-Phosphorus in Seven Kentucky Streams Draining Small Agricultural Watersheds: Eighteen Years Later." *Journal of Environmental Quality* 21 (1): 147–50.
Thomas, R. (no date). Compilation and evaluation of stream restoration projects: learning from past projects to improve future success. http://www.ansp.org/research/environmental-research/projects/restoration/.
Tiner, Ralph W. 1987. *Mid-Atlantic Wetlands: A Disappearing Natural Treasure*. US Fish and Wildlife Service, Fish and Wildlife Enhancement, National Wetlands Inventory Project.
U.S. Army Corps of Engineers, 1987. Corps of Engineers wetlands delineation manual. Vicksburg, MS. Wetlands Research Program Technical Report Y-87-1.
U.S. Department of Agriculture, 2002. Plant fact sheet – reed canarygrass. NRCS plant materials program.
U.S. EPA, 2002. Methods for evaluating wetland condition: Using vegetation to assess environmental conditions in wetlands. Office of Water, U.S. Environmental Protection Agency, Washington DC. EPA-822-R-02-020. Vadas, P. A., P. J. A. Kleinman, A. N. Sharpley, and B. L. Turner. 2005. "Relating Soil Phosphorus to Dissolved Phosphorus in Runoff." *Journal of Environmental Quality* 34 (2): 572–80.
Vighi, M., S. Soprani, P. Puzzarini, and G. Menghi. 1991. "Phosphorus Loads from Selected Watersheds in the Drainage Area of the Northern Adriatic Sea." *Journal of Environmental Quality* 20 (2): 439–44.
Violin, C.R., P. Cada, E.B. Sudduth, B.A. Hassett, D.L. Penrose, E.S. Bernhardt. 2011. Effects of urbanization and urban stream restoration on the physical and biological structure of stream ecosystems. *Ecol. Appl.,* 21:1932-1949.

Vitousek, Peter M., and Robert W. Howarth. 1991. "Nitrogen Limitation on Land and in the Sea: How Can It Occur?" *Biogeochemistry* 13 (2): 87–115.
Voli, M. et al. 2009 Preliminary reconstruction of a Pre-European settlement valley bottom wetland, Southeastern Pennsylvania. Water Resources Impact, v. 11, p. 11–13.
Voli, Mark T., Karl W. Wegmann, DelWayne R. Bohnenstiehl, Elana Leithold, Christopher L. Osburn, and Viktor Polyakov. 2013. "Fingerprinting the Sources of Suspended Sediment Delivery to a Large Municipal Drinking Water Reservoir: Falls Lake, Neuse River, North Carolina, USA." *Journal of Soils and Sediments* 13 (10): 1692–1707.
Voli, Mark, Dorothy Merritts, Robert Walter, Erik Ohlson, Katherine Datin, Michael Rahnis, Laura Kratz, Wanlin Deng, William Hilgartner, and Jeffrey Hartranft. 2009. "Preliminary Reconstruction of a Pre-European Settlement Valley Bottom Wetland, Southeastern Pennsylvania." *Water Resources Impact* 11 (5): 11–12.
Voynova, Yoana. 2006. "Phosphorus Content of Legacy Sediments in Lancaster County: Significance for Non-Point Source Pollution to the Chesapeake Bay."
Walling, D. E., and J. C. Woodward. 1995. "Tracing Sources of Suspended Sediment in River Basins: A Case Study of the River Culm, Devon, UK." *Marine and Freshwater Research* 46 (1): 327–36.
Walling, D.E. 2005. "Tracing Suspended Sediment Sources in Catchments and River Systems." *Science of The Total Environment* 344 (1-3): 159–84. doi:10.1016/j.scitotenv.2005.02.011.
Walsh, Christopher J., Allison H. Roy, Jack W. Feminella, Peter D. Cottingham, Peter M. Groffman, and Raymond P. Morgan II. 2005. "The Urban Stream Syndrome: Current Knowledge and the Search for a Cure." *Journal Information* 24 (3).
Walter, Robert C., and Dorothy J. Merritts. 2008. "Natural Streams and the Legacy of Water-Powered Mills." *Science* 319 (5861): 299–304. doi:10.1126/science.1151716.
Walter, Robert C., William K. Hart, and John A. Westgate. 1987. "Petrogenesis of a Basalt-Rhyolite Tephra from the West-Central Afar, Ethiopia." *Contributions to Mineralogy and Petrology* 95 (4): 462–80. doi:10.1007/BF00402206.
Walter, Robert, Alexandra Sullivan, Yoana Voynova, Andrew Stubblefield, Dorothy Merritts, and Jerry Ritchie. 2006. "A 137Cs Inventory of a Small Agricultural Watershed in Lancaster County, PA, with Implications for Sediment and Nutrient Loads to the Chesapeake Bay." In *Abstracts with Programs*, 38:7.
Wang, L.Z., J. Lyons, P. Kanehl, and R. Gatti. 1997. Influences of watershed land use on habitat quality and biotic integrity in Wisconsin streams. *Fisheries* 22(6):6–12.
Wang L.Z., J. Lyons, and P. Kanehl. 2001. Impacts of urbanization on stream habitat and fish across multiple spatial scales. *Environ. Mgmt.*, 28:255 - 266.
Westgate, J. A., R. C. Walter, G. W. Pearce, and M. P. Gorton. 1985. "Distribution, Stratigraphy, Petrochemistry, and Palaeomagnetism of the Late Pleistocene Old Crow Tephra in Alaska and the Yukon." *Canadian Journal of Earth Sciences* 22 (6): 893–906.
Wolman, M. G., 1959, Factors influencing erosion of a cohesive river bank: American Journal of Science, v. 257, p. 204–216.
Wynn, T., Henderson, M. & Vaughan, D., 2008, Changes in streambank erodibility and critical shear stress due to subaerial processes along a headwater stream, southwestern Virginia, USA: Geomorphology, v. 97, p. 260–273.