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Stream Restoration for Legacy Sediments at Gramies Run, Maryland: Early Lessons from Implementation, Water Quality Monitoring, and Soil Health

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Abstract: While stream restorations are increasingly being adopted to mitigate sediment and nutrient inputs and to meet water quality regulatory targets, less information is available on the drivers behind the design, implementation, effectiveness, and cost of restorations. We address these issues for a $4.2 million stream restoration for legacy sediments implemented for a rural Piedmont stream in Maryland, USA. A total of 1668 m of stream was restored in three phases, which included the partial removal of legacy sediments, the grading of streambanks, floodplain creation, channel reshaping with meanders and pool-riffle forms, the raising of the stream bed, and the planting of riparian vegetation. The sediment, nitrogen, and phosphorus concentrations and fluxes were monitored before- and during the restoration phases. The sites selected for restoration had legacy sediments vulnerable to erosion and were on state-owned land. The restoration design was based on the need to maintain mature riparian trees and preserve existing sensitive wetland habitats. Water quality monitoring indicated that the sediment and nutrient fluxes increased during the restoration phase and were attributed to disturbance associated with construction activities and increased runoff. We also recommend that soil health needs to be included as an integral component to enhance the effectiveness and resilience of stream restorations.

Keywords: sediment; nitrogen; phosphorus; stream restoration; nonpoint source pollution; water quality; riparian; floodplain

1. Introduction

Sediment, followed by nutrients—i.e., nitrogen (N) and phosphorus (P)—are the leading causes of water quality impairment in rivers and streams in the United States [1]. High sediment concentrations can increase water turbidity and cause harm by impeding sunlight and preventing the growth of aquatic life [2]. Excess nutrients can cause eutrophication and lower dissolved oxygen, which can be detrimental to fish and other species [2]. Historically, sediment pollution has been attributed to upland soil erosion; however, recent studies suggest that valley-bottom legacy sediments [3–5] can contribute a substantial portion of sediment loads to streams and rivers [6–9]. Valley-bottom legacy sediment deposits, especially in the mid-Atlantic US, have been attributed to the coupled effects of ubiquitous mill dams and extensive agricultural erosion in the 17th to early 20th century [10]. Mill dams raised base levels, reduced the flow velocities, and resulted in the deposition and accumulation of sediments in the drainage network [10–13]. Many milldams spanned the full extent of the valley bottoms, substantially altering the original floodplains [12,13], and the millpond sediment thicknesses were greatest near the dams and progressively decreased upstream [10]. Many of these milldams have since breached or been removed [14], resulting in highly incised contemporary streams with exposed vertical streambanks that are vulnerable to erosion [12,13,15–17].

The erosion and exports of legacy sediment from valley bottoms (streambanks and terraces) occur due to a variety of physical processes [13]. Fluvial erosion with large storm flows [18], freeze-thaw activity [19,20] during winter, and/or desiccation and cracking in summer and mass wasting [21] have been identified as important mechanisms for stream-bank erosion. Freeze-thaw cycles cause bank sediments to lose their cohesive strength with subsequent detachment and slumping/collapse at the base of the streambank [22]. The loose, fine, detached sediment is then flushed out by streamflow and subsequent storms and transported downstream [12]. Not surprisingly then, anomalously elevated rates of bank erosion and sediment exports have been reported for watersheds in the eastern US [6–8,23]. Stream-bank erosion has been found to contribute as much as 50–100% of the suspended sediment loads in Piedmont watersheds [6–8,23].

The US Environmental Protection Agency (US EPA) has taken initiative to reduce sediment and nutrient pollution in waterbodies such as the mid-Atlantic Chesapeake Bay through a regulatory "pollution diet" program, also known as total maximum daily load (TMDL; [24]). A TMDL is "the maximum amount of pollution a body of water can receive and still meet state water quality standards" [2]. For the Chesapeake Bay, this has amounted to a targeted decrease (by 2025 from 2009 levels) in N, P, and sediments of 25%, 24%, and 20%, respectively [2].

A rapidly growing practice to reduce sediments and nutrients in waterbodies and meet TMDL targets is stream restoration [24–27]. Stream restoration includes a variety of instream and/or floodplain activities, such as stream bank grading to reduce erosion and enhance floodplain connection, alterations in stream slope and shape including raising stream beds, the addition of woody debris and rock, wetland creation, and bank sediment removal [28–30]. The costs of these projects can vary considerably depending on the extent and the types of techniques employed [24,31]. For example, the Chesapeake and Atlantic Coastal Bays Trust Fund has funded over 200 stream restoration projects, and in the 2019–2020 fiscal year almost $20 million from this fund was dedicated to stream restoration projects [32]. Over the last 5 years, both the number of permits for stream restoration and the length of projects have increased, and the average restoration length has almost tripled, from 233 to 670 m (765 to 2200 linear feet) [32].

Public utilities and/or state agencies such as the Maryland (MD) State Highway Administration (SHA) are required to implement measures to help meet the Bay TMDL, since impervious surfaces within the SHA's Right-of-Ways contribute nutrients, sediment, and other types of pollution to waterways [33]. To discharge any municipal water into surface waters, the MD SHA needs a National Pollutant Discharge Elimination System (NPDES) permit or, more specifically, a Municipal Separate Storm Sewer System (MS4) permit, which is issued by the Maryland Department of the Environment (MDE) [33]. As an alternative to treating runoff from its impervious surfaces, SHA has the option to offset this area by investing in innovative approaches to improve water quality. One of these innovative, growing practices is stream restoration, which will account for about 25% of their impervious restoration credits [33].

While the use of stream restoration projects to improve water quality and meet regulatory targets has increased sharply in recent years, one of the big challenges is that very little is known about their effectiveness for water quality improvement. Many of the stream restoration projects have no monitoring component, pre- or post-restoration, and there is no way of knowing if the restorations achieved their water quality objectives [34]. For example, [35] reported that only 5.4% of the 4700 projects in their synthesis of studies in the Chesapeake Bay watershed report monitoring. Additionally, typically very little documentation exists on why and how various stream restoration projects are selected, designed, and implemented; the challenges and constraints driving implementation and monitoring; the costs of restoration; and the potential for the recovery of ecosystem processes and functions following restoration [35]. Addressing these knowledge gaps is critical for the continued and successful use of stream restorations as a water quality and watershed management practice.

We address some of these knowledge gaps for a large, $4.2 million stream restoration project that was recently (June 2020) completed on Gramies Run in Cecil County, Maryland [36]. A total of 1668 m (5473 linear feet) of stream was restored for proposed reductions of 912 kg/year (2010 lb/year) of total N, 175 kg/year (386 lb/year) of total P, and 367 tons/year of total suspended sediment (TSS) erosion [36]. Protocols 1 and 2 defined by the Expert Panel [37] were used to estimate nutrient reductions which will be used to help meet SHA's TMDL requirements [36]. The restoration included streambank grading and floodplain reconnection, streambed uplift, and the reshaping of the stream channel to reduce flow stresses and bank erosion [36]. Due to the site-specific constraints and selected design approach, the construction period extended from June 2018 through to June 2020 (approximately a 2-year period) and was spread across three restoration reaches or phases. Similar to other restoration projects, no routine/continuous water quality monitoring was conducted by the sponsor (MD SHA). Using seed grant funding from the University of Delaware, we conducted limited weekly and selected stormflow water quality monitoring (sediment and nutrients) for the pre- (7 September 2017 through to 14 June 2018) and during construction (15 June 2018 to 30 September 2019) periods. This resulted in approximately nine months of pre-restoration and 15 months of during-restoration data. Post restoration monitoring after June 2020 is planned for 3–4 years. It should be noted that, since restoration was staggered across three individual reaches/phases, some post-restoration effects also occurred for the "during-restoration" period. Here, we highlight some of the important early lessons and insights from this study that advance our understanding of stream restorations. Key questions that we address are: What were the key drivers, challenges, and constraints associated with site selection, design, and restoration implementation at the Gramies Run site? How did water sediment and nutrient concentrations change over the pre- and during-restoration phases of the project? How does the Gramies Run restoration compare with other studies in terms of the cost and nutrient reductions? How can we further enhance the ecological integrity and water quality services of stream restoration?

2. Materials and Methods

2.1. Study Site

Gramies Run is a second order tributary (approximately 5.8 km or 3.6 miles long, Figure 1) of the Big Elk Creek located in Cecil County, Maryland [36]. With a drainage area of 7.89 km^2 (3.05 mi^2), Gramies Run drains into the Big Elk creek, which empties into the Chesapeake Bay via the Elk River. Most of the stream is located within the Fair Hill Natural Resource Management Area, part of the Maryland Department of Natural Resources (DNR). The Big Elk Creek watershed is underlain by the Mt. Cuba Wissahickon formation and includes pelitic gneiss and pelitic schist, with subordinate amphibolite and pegmatite [38]. The current land use in the Gramies Run watershed is about 46% shrubs and low vegetation, 48% forest, and 5% developed (Figure 1). The soils are moderately eroded and excessively to moderately well drained and are primarily composed of Glenelg loam in a 3% to 8% slope, Glenelg loam in a 8% to 15% slope, and Glenville silt loam in a 3% to 8% slope [39].

Gramies Run is designated as Use Class I-P [40] for water recreation, the protection of aquatic life, and public water supply. Gramies Run is also a part of the Maryland Biological Stream Survey (MBSS) for benthic and fish monitoring that were conducted in 1996, 1997, and 2003. The benthic index of biotic integrity (IBI) scores were 4.67 for both 1996 and 2003, indicating that the benthic population in Gramies Run was in the "Good" range [36]. The fish IBI scores in 1996 and 2003 were 4.0 and 5.0, respectively, supporting that the fish population within Gramies Run was also in the "Good" range [36]. Both these benthic and fish IBI scores are considered relatively high for warm water Use I-P stream systems, indicating that Gramies Run provides good water quality and habitat conditions to support populations of less pollution-tolerant species [29]. Not surprisingly then, in 2007, MDE designated Gramies Run a Tier II High Quality Stream Segment [36]. The Tier II High

Quality Waters designations have been established for streams and watersheds where baselines have been established using biological community metrics that provide a cumulative assessment which indicates that the water quality exceeds the minimum conditions required to fully support the stream's designated uses.

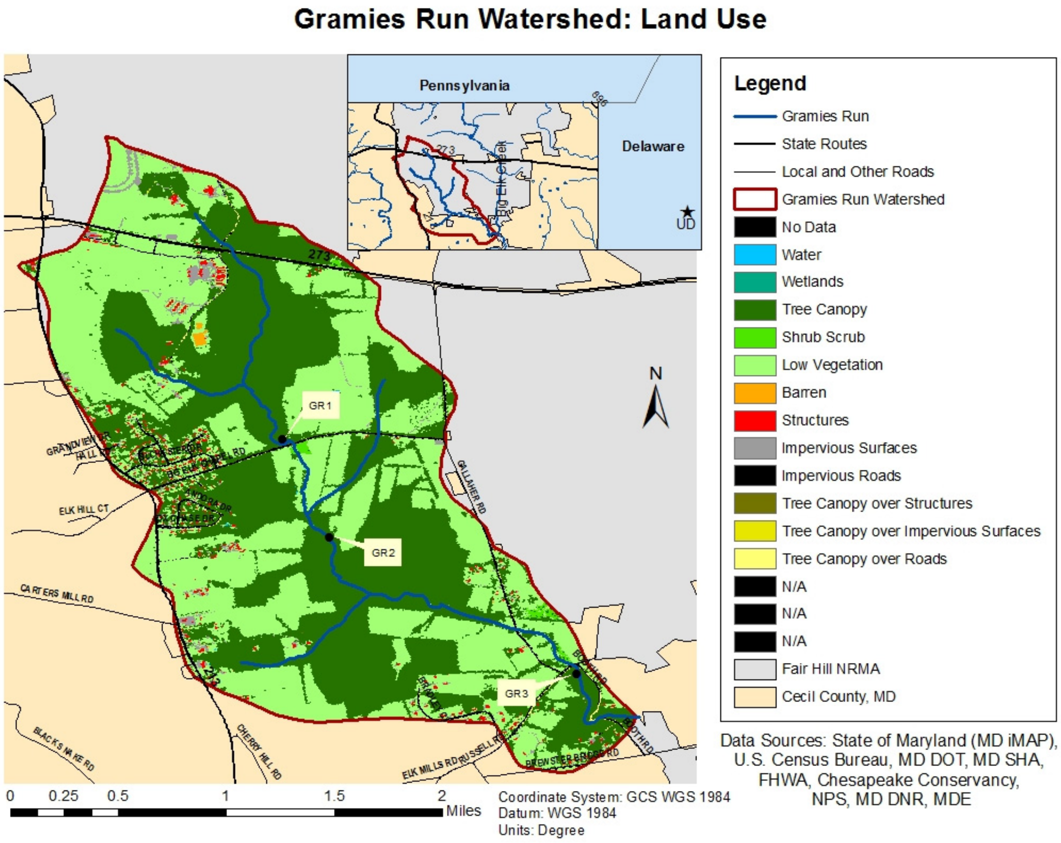


Figure 1. Location of Gramies Run watershed within the Big Elk Creek drainage basin (inset) and land use within the Gramies Run watershed (high resolution land cover data obtained from Chesapeake Conservancy's Conservation Innovation Center). The grey shaded area (inset) represents state-owned (Maryland DNR) property. Location of the stream water sampling points GR1, GR2, and GR3 are also indicated.

Gramies Run, and particularly Big Elk creek, have a long legacy of milling and agriculture. Historic maps [41] indicate numerous mill dams every few kilometers/miles along these creeks. Gramies Run had at least two mapped mill dams (see Supplementary Figure S1 [41]) prior to its confluence with the Big Elk creek (there could have been other dams that were likely not mapped). These small (<7 m height) dams were common across the Mid-Atlantic Piedmont region from the 1700s to the early 1900s, and in combination with poor, erosive agricultural practices resulted in a large accumulation of legacy sediments in valley bottoms and along stream banks [10,12,13,42]. Our recent studies in Gramies Run and Big Elk Creek [9,43,44] indicate the significant accumulation and depth of light-colored silt and clay-rich legacy sediments (with bank heights of up to 2 m), occasionally overlying a dark, precolonial, organic soil layer (Figure 2). Along one restoration reach in Gramies Run, the precolonial, organic-rich sediment layer (30 or more cm thick; Figure 2) was (total bank height was 170 cm) confirmed via ^{14}C radiocarbon dating [44] to be a mean age of 950 ± 30 BP (Beta Lab # 510411: 95.4%: 926–795 cal BP). Radiocarbon dating was performed for the fibrous organic material in this layer. This dark organic horizon, underlain by a white, Pleistocene-era gravel, was visible for about 500 m along this stretch of Gramies Run (Figure 2 and along Phase II in Figure 3), potentially suggesting the presence of a large valley-bottom swamp/bog or beaver-wetland complex during precolonial times at this location [10].

Figure 2. Light-colored legacy sediments overlying a dark, organic-rich precolonial horizon along Gramies Run. Fibrous organic material in the dark organic horizon was radiocarbon dated to a mean age of 950 ± 30 BP. The bank height (indicated by the red bar) was about 1.7 m, and the thickness of the organic horizon at this location was 30 cm or more. The organic horizon was visible for a length of 500 m along this reach of Gramies Run.

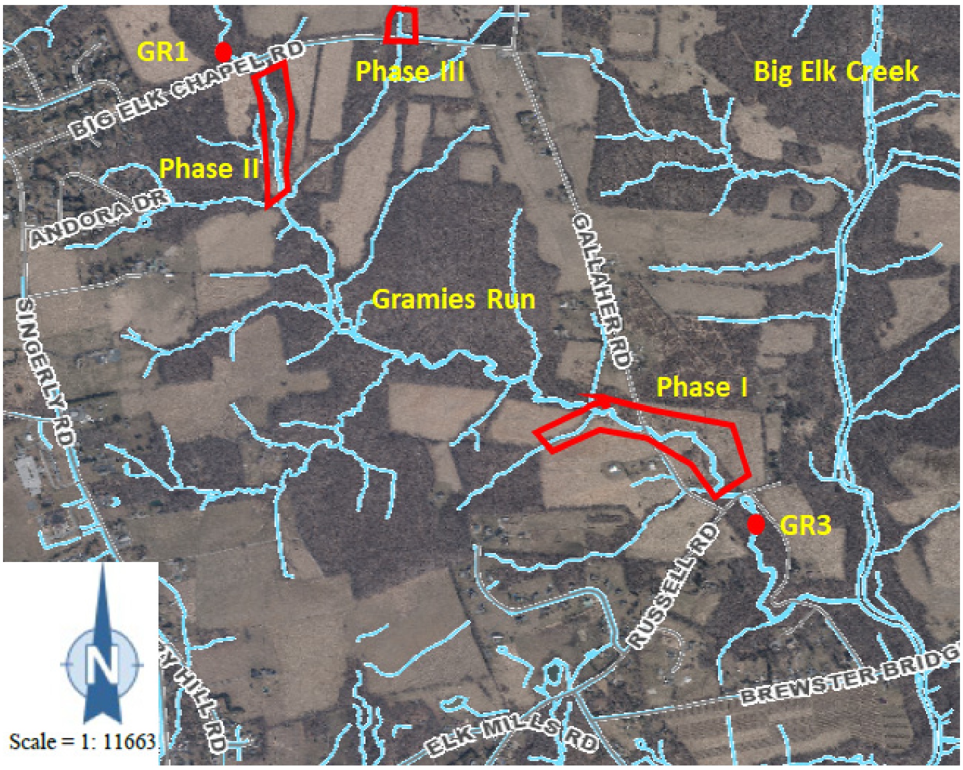


Figure 3. Aerial map indicating restoration phases (I–III) on the Gramies Run stream network. Sampling locations GR1 and GR3 (indicated by the filled red circles) are located upstream and downstream of the restoration reaches.

2.2. Gramies Run Restoration Approach and Stream Reaches Selected for Restoration

Full documentation of the Gramies Run design is available in [36], and only the key details are included here. A total of 1668 m (5473 linear feet) of stream was restored for proposed reductions of 912 kg/year (2010 lb/year) of total N, 175 kg/year (386 lb/year) of total P, and 367 tons/year of total suspended sediment (TSS) erosion (design engineers have indicated that the proposed reductions could be revised, if necessary, post construction). The pre-restoration bank erosion rate was measured, and it was assumed that restoration would mitigate 50% of that sediment loading. Using Protocol 1 [37], the N and P reductions were computed by multiplying the TSS reduction of 367 tons/year by 2.28 lb N/ton (1140 mg N/kg) and 1.05 lb P/ton (525 mg P/kg), respectively. Protocol 2 [37] was used to compute the N reduction due to hyporheic denitrification, which was calculated by finding the volume of the hyphoreic box and multiplying by a denitrification rate of 1.06×10^{-4} pounds/ton/day of soil derived from [26].

Stream restoration at Gramies Run included the partial removal of near-stream legacy sediments to decrease the height of the banks, reduce bank erosion, widen the floodplains (note Supplementary Figure S2), and enhance the hydraulic connectivity of the floodplains with the stream. The streambanks were graded to achieve an approximate 3:1 bank height to length ratio. The stream bank legacy sediment removal and grading also resulted in the removal of the underlying precolonial organic soil horizon at some locations. Following the Natural Channel Design (NCD) approach [45], the stream channels were reshaped with the introduction of meanders and pool-riffle geomorphology to reduce the flow velocities and shear stresses [36]. Invasive vegetation on the banks was cleared, and the regraded floodplains were planted with new trees. This restoration approach was selected to have minimal impact on the natural resources at the site, which included wetlands and mature riparian trees, particularly sycamores. In addition, a portion of the restoration site was also in the vicinity of wetlands designated by Maryland Department of Natural Resources (DNR) as a Sensitive Species Project Review Area (SSPRA). Thus, streams adjacent to the SSPRA were left at approximately the same elevation to minimize any impacts to the regional groundwater hydrology and SSPRA habitat.

Restoration occurred in three phases (I to III, Figure 3) between spring 2018 and spring 2020. Phase I was the most downstream section stretching across Gallagher road and was immediately upstream of the former (now breached; lower dam in Figure S1) mill dam location just above Russell road (Figure 3). Phase I began on 15 June 2018, and work in this phase continued until the summer of 2019. Phase II was south of Big Elk Chapel road and had the buried, organic soil horizon shown in Figure 2 (Figure 3). A historic mill dam (1700–1800s) was located upstream of this phase and above Big Elk Chapel road (upper dam in Figure S1). Phase II and III began in November 2018 and continued in varying extents until June 2020. Field surveys by restoration designers [36] and our own previous work [46] (see Figure 4b in [46] and Supplementary Figure S3) indicated the substantial accumulation of legacy sediments along phases I and II and which were vulnerable to fluvial and subaerial (freeze-thaw) erosion.

During in-stream construction, water in the stream channel was dammed and pumped around those stream reaches. Filter bags were used downstream to catch sediment while letting the stream water go through. Younger trees along the banks and on the floodplain were cut down for construction, although mature trees such as Sycamores were saved where possible [36]. In the tributaries, cascade material was added and rock sills were installed, as well as clay channel blocks, imbricated rock bank protection, and a riprap plunge pool on the downstream side of a new farm road. In addition, riffle material was added to create pool and riffle sections [36]. This also included in-stream structures such as j-hooks and cross-vanes to direct the flow away from the streambanks [36].

No in-stream construction occurred at any of the sites from March 1st until June 15th due to fish spawning [36]. These three restoration sections were selected for stream restoration because they are mostly located within the Fair Hill Natural Resources Management Area, a Maryland DNR property. A few segments are located on privately owned land, but landowner permission was obtained for the project. After final stabilization at the sites, landscape coir fiber matting was installed and held in place with dead wood stakes (Figure 4, top right) and riparian plantings were completed—i.e., trees, shrubs, tubelings/live stakes, and wetland vegetation (see Figure 4 for a sequence of restoration stages for portion of Phase II). The trees planted include black willow and alder.

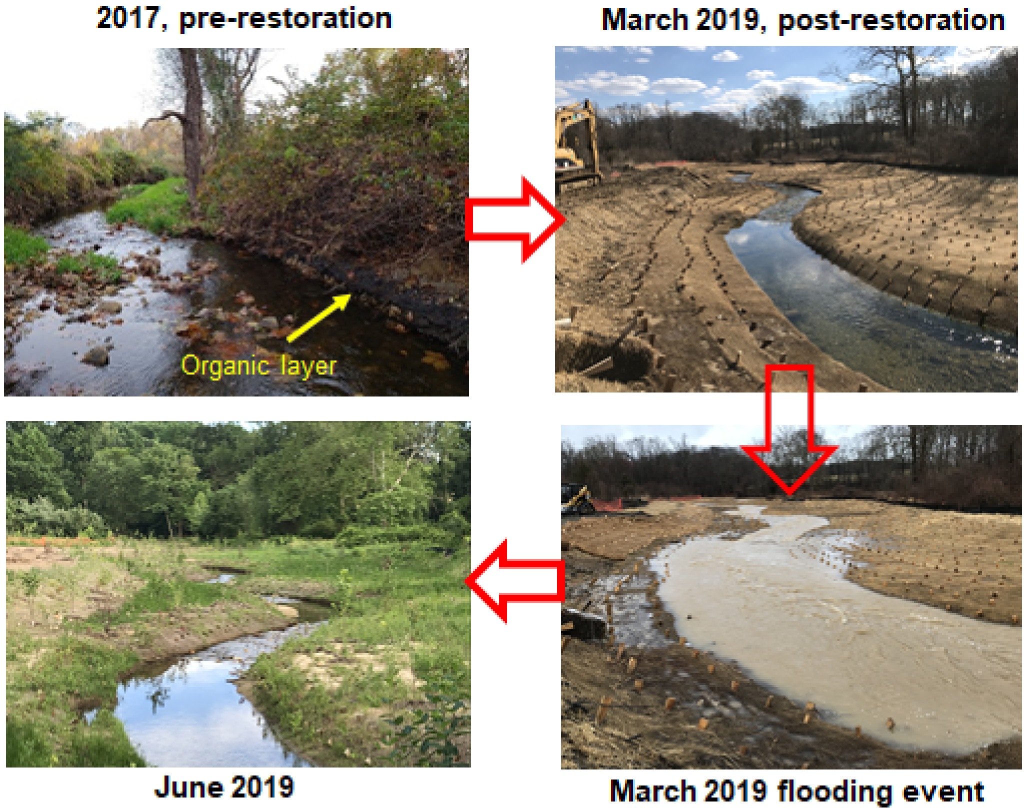


Figure 4. Sequence of photos for a reach in Phase II indicating the progression of restoration at Gramies Run. Note the pre-restoration height of the near-vertical stream banks and the location of the precolonial organic layer (top, left; photo by Brian Tetrick) and the new wider floodplains which allowed bank overflows during storms (bottom right).

2.3. Hydrologic Monitoring

Complete details on monitoring are available in [47], and only brief information is reported here. Pre- and during-stream restoration water quality sampling was performed at three locations: GR1, GR2, and GR3 (Figures 1 and 3). However, since more data at a higher frequency were available for sites GR1 and GR3, and since these sites bookended the restoration phases along the main stem, only data from these two sites are used. The drainage areas for GR1 and GR3 are 310.8 ha and 764 ha, respectively, which were estimated using the USGS Streamstats website [48]. The pre-restoration period was defined as 7 September 2017 to 14 June 2018 (nine months) while the during-restoration was characterized as 15 June 2018 to 30 September 2019 (15 months).

The stream water levels at GR1 and GR3 were recorded every 15 min using non-vented HOBO (Onset Inc.) water level transducers, which were corrected using a non-vented sensor exposed to the air (to record atmospheric pressure). A depth-discharge rating curve was developed for both sites by measuring the stream stage and velocity (Global Water Flow Probe) on a weekly basis. A rectangular cross section was assumed for GR3, while at GR1 two existing circular concrete culverts (below Big Elk Chapel Road) through which the flow passed (1.232 m, 48.5 inch diameter) were used for the flow cross section. The streamflow discharge computed for the two sites was normalized by the corresponding drainage areas. Precipitation data (5 min tipping bucket and hourly GEONOR gage) for the study period were available from the Delaware Environmental Observing System (DEOS) weather station [49] at Fair Hill (less than 4000 m away from Gramies Run sampling locations).

2.4. Water Quality Monitoring

Turbidity (NTU) was recorded at a frequency of 30 min at both GR1 and GR3 using water quality sondes (InSitu Inc.) placed in the stream. The sensors were calibrated approximately every two months and the data were downloaded periodically throughout the study period. Weekly stream water grab samples were collected at GR1 and GR3 from September 2017 to 2019 (data have been collected since September 2019, but were not available at the time of this analysis). Grab water samples were collected in 250 mL HDPE bottles and kept in a cooler on ice until they were filtered in the lab using 0.7 µm glass fiber filters. The filtered water was then stored in amber glass vials and these were either refrigerated (for total dissolved N, TDN) or frozen (for nitrate-N, ammonium-N, and ortho-P) until analysis. The nitrate-N (NO_3-N) concentration was analyzed using an absorbance based spectrolyser (S::CAN, Inc.) following calibrations and QAQC checks with standard solutions. The concentrations of ortho-P (PO_4) and ammonium-N were determined using a SEAL AQ2 discrete analyzer and the EPA methods EPA-118A and EPA-148-A, respectively. Ultimately, ammonium-N was not included in further analysis due to very low concentrations and non-detect results. The total dissolved nitrogen (TDN) concentrations were measured by combustion on a Shimadzu TN analyzer.

Similarly, stream water grab samples were also collected in 250 mL HDPE bottles at GR1 and GR3 during selected storm events. The sampling was timed to collect the highest flows during the storm hydrograph. For pre-restoration, four storm events were sampled at GR1 and GR3, while during restoration, four storm events were sampled at GR1 and nine at GR3. The storm samples were filtered using 0.7 µm glass fiber filters, and the suspended sediment concentrations (SSC) on the filters were determined gravimetrically [50]. Post filtration, the filters were dried in an oven at 40 °C for 12 h and weighted to compute the sediment mass to get the SSC. The sediments were then analyzed by combustion for the particulate N (PN) content (mg/kg). The filtered storm water samples were analyzed for the dissolved nutrients (N and P) following the protocols described above for non-stormflow samples.

2.5. Data Analysis

The storm sample SSC were plotted against the turbidity values to develop an SSC–turbidity relationship (Supplementary Figure S4). This relationship was used to convert the 30-min turbidity measurements to SSC data. Nitrate-N, TDN, and ortho-P concentrations were plotted as a time series for the upstream (GR1) and downstream (GR3) sites to compare the pre-restoration and during restoration levels and account for seasonal variability. Non-stormflow was defined as <0.025 mm runoff/15 min. (<86 L/s) at GR1 or <0.032 mm/15 min. (<275 L/s) at GR3. Stormflow was defined as >0.025 mm/15 min. (>86 L/s) at GR1 or >0.032 mm/15 min. (>275 L/s) at GR3. Concentration–discharge (C–Q) plots were created between the measured nutrients and discharge, and this relationship was extended for periods when sampling was not conducted [47]. However, generally, the relationship between concentration and discharge at lower flows was found to be weaker than during higher flows. Therefore, for nitrate-N and TDN, non-stormflow flux was calculated using methods found in [51]. In contrast, the C–Q plots were extended to calculate the discharge during stormflow. For ortho-P, a weak relationship was found between concentration and discharge, so the methods described in [51] were used. The streamflow and nutrient flux were compared between GR1 and GR3 pre- and during restoration, and the suspended sediment concentration data at GR3 were compared pre- and during restoration.

To investigate the specific influence of the restoration, nutrient fluxes measured at GR3 (downstream site) were subtracted from those measured at the upstream reference site GR1. To remove the effect of the temporal variation of rainfall-runoff between the pre- and during-restoration periods, the net flux (GR3-GR1) was then divided by the flux measured at GR1. Thus, the normalized net flux was given as (GR3-GR1)/GR1 and provided a way to determine if the changes in flux at the downstream site were a result of the stream restoration. Finally, ANOVA tests were performed on the monthly totals of the normalized net fluxes to determine if the differences were statistically significant. All the statistical analyses were completed using JMP Pro 14 software.

3. Results

3.1. Annual and Monthly Precipitation and Streamflow Totals Pre- and During Restoration

For the pre-restoration period (7 September 2017–14 June 2018; nine months), the annual precipitation (total for the duration divided by the duration time in years) was about 1101 mm/year, whereas during restoration (15 June 2018–30 September 2019; 15 months) it was 1521 mm/year (Figure 5). This was primarily because of the elevated annual precipitation for 2018 of 1721 mm, which was greater than the average annual precipitation for the region at approximately 1200 mm [49]. Months with the highest precipitation totals over the monitoring period include May 2018, July 2018, September 2018, November 2018, and December 2018 (monthly totals in Figure 6). The variation in the precipitation and runoff amounts over the monitoring periods makes the comparison and interpretation of the restoration effects more difficult.

The annual area-normalized streamflow at GR1 decreased from 721.4 mm/year before the restoration to 580.9 mm/year during the restoration, while the corresponding values for GR3 were 624.1 mm/year and 859.8 mm/year, respectively (Figure 5). Pre-restoration, the monthly runoff ranged from 31.9 to 66.7 mm at GR1 and 21.1 to 85.4 mm at GR3 (Figure 6). During the stream restoration, the monthly runoff ranged from 34.9 to 59.0 mm at GR1 and 36.8 to 103.6 mm at GR3 (Figure 6). In wetter months such as September, November, and December 2018, GR3 had more runoff than GR1 (Figure 6). Examining the monthly runoff at GR1 and GR3, the runoff is higher most months at GR3 during the restoration, whereas the monthly runoff was higher at GR1 most months pre-restoration. An ANOVA analysis indicated that the difference in monthly runoff between the downstream and upstream sites (GR3-GR1) was significantly higher during the restoration versus the pre-restoration period ($p = 0.0003$).

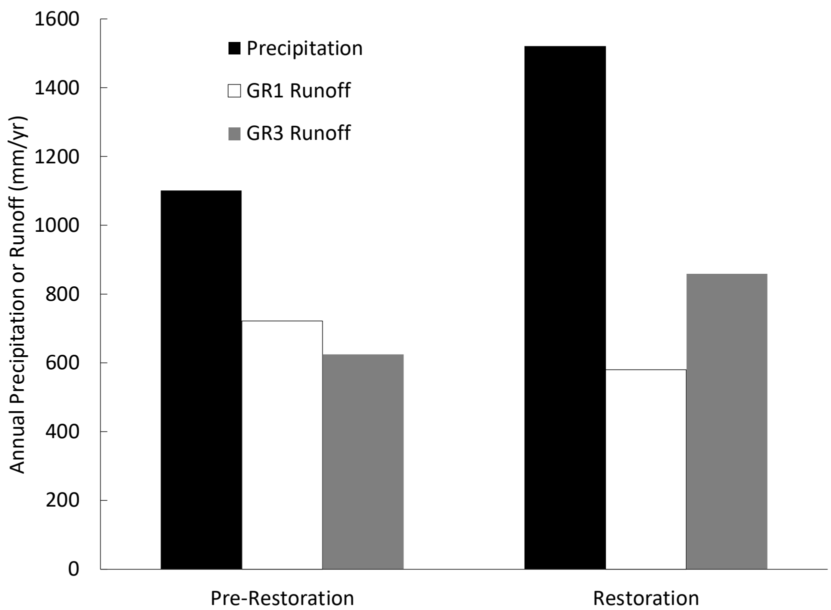


Figure 5. Annual precipitation and runoff/streamflow for the pre- and during-restoration phases of the project for upstream (GR1) and downstream (GR3) sites.

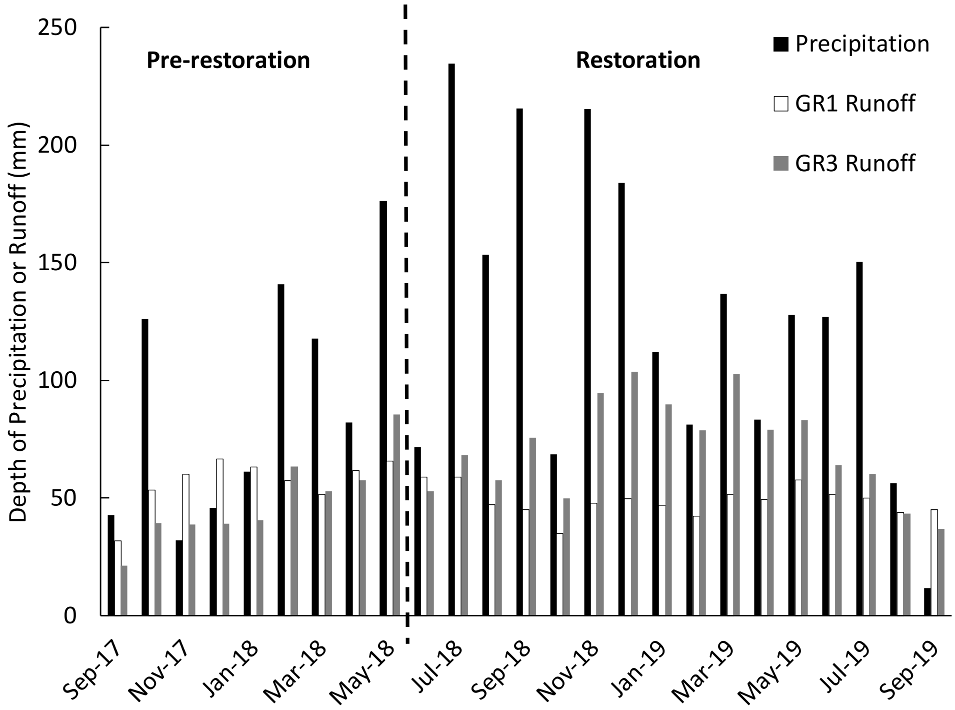


Figure 6. Monthly precipitation and runoff/streamflow for the pre- and during-restoration phases of the project for upstream (GR1) and downstream (GR3) sites.

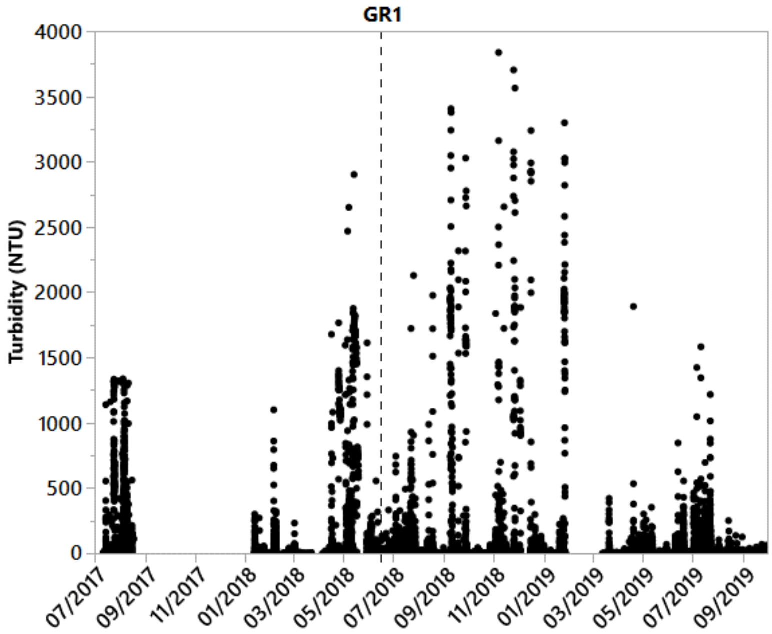
At GR1, approximately 90% of the flow was non-stormflow, both pre-restoration and during restoration (Table 1). However, non-stormflow only accounted for about 80% of runoff at GR3 pre-restoration and only about 72% during the restoration (Table 1). The relative decrease in non-stormflow at GR3 during the restoration resulted from the increased stormflow, which increased by about 8% for the restoration period. The increased stormflow at GR3 is attributed to a greater areal proportion of wetlands and contributing tributaries (Figures 1 and 3) above GR3 than those for GR1.

Table 1. Stormflow versus non-stormflow proportions for streamflow at GR1 and GR3.

	GR1	GR3
Watershed area (ha)	310.8	764.0
% Non-stormflow Pre- During	<0.025 mm/15 min. 92.7% 91.2%	<0.032 mm/15 min. 80.5% 72.3%
% Stormflow Pre- During	>0.025 mm/15 min. 7.3% 8.8%	>0.032 mm/15 min. 19.5% 27.7%

3.2. Turbidity and Nutrient Concentrations in Stream Waters

Over the study period, peak stream water turbidity values occurred during storm events (Figure 7). The pre-restoration turbidity values were between 0 and 2901 NTU, with a median of 6.2 NTU at GR1, while during restoration the values were between 0 and 3836 NTU, with a median of 1.7 NTU. At GR3, the pre-restoration values were between 0 and 774 NTU, with a median of 0.94 NTU, while during restoration the values were between 0 and 3859 NTU, with a median of 2.15 NTU. The maximum turbidity value at GR1 occurred as a result of a storm event on Nov 6, 2018, and the second highest turbidity value at this site occurred on 25 November 2018 after receiving approximately 57.2 mm of rain in an 8-hour period. The maximum turbidity value for GR3 was recorded on 25 November 2018.



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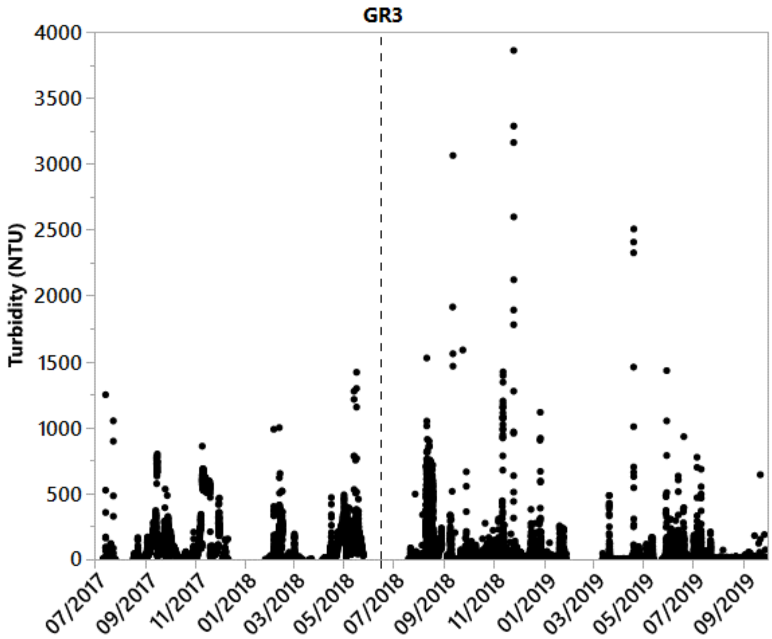

(**b**)

Figure 7. Time series plot of turbidity at (**a**) GR1 and (**b**) GR3. Gaps in the series are times when fouling occurred or the sonde was in for maintenance. The dashed line represents when in-stream construction began on 15 June 2018.

No seasonal trend was apparent in the stream water nitrate-N or TDN concentrations upstream or downstream of the restoration reaches (Figure 8). Decreases in the concentrations of N at both sites occur during storms (dilution). The C–Q plots (not included here, [47]) revealed that the concentrations decreased with increasing streamflow discharge. In general, the nitrate-N and TDN concentrations were lower at GR3 than at GR1 (Figure 8). Prior to the restoration, the nitrate-N concentration ranged from 0.6 to 2.09 mg/L at GR1, with corresponding values of 0.55–1.9 mg/L during restoration. For

GR3, the pre-restoration nitrate-N values were 0.53–1.44 mg/L, with during-restoration values of 0.5–1.86 mg/L. For TDN, the pre-restoration concentration at GR1 varied from 0.99 to 3.48 mg/L, while the during-restoration values ranged from 0.89 to 2.69 mg/L. At GR3, the pre-restoration TDN concentration ranged from 0.97 to 2.40 mg/L, while during restoration the concentration ranged 0.84 to 2.01 mg/L.

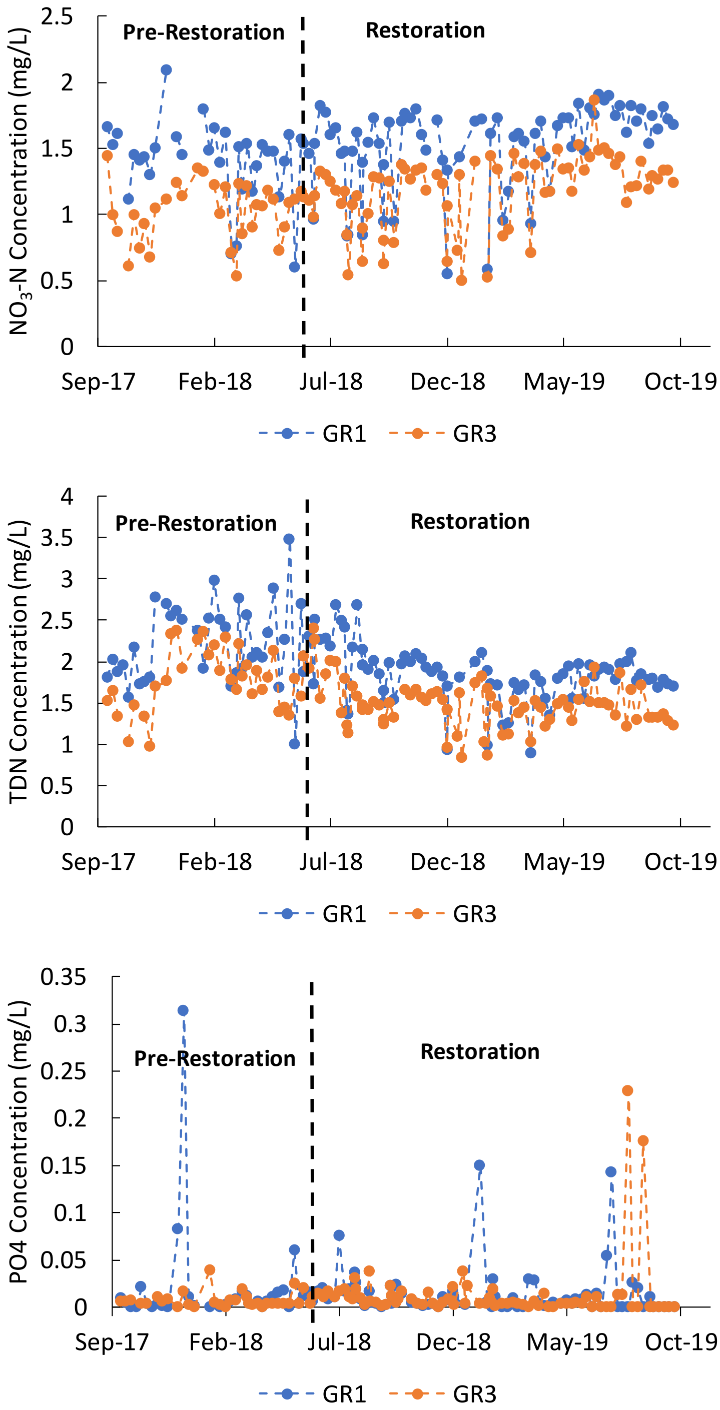

Figure 8. Stream water concentrations at the upstream (GR1) and downstream (GR3) monitoring sites for nitrate-N (NO_3-N), total dissolved N (TDN), and ortho-P (PO4). The dashed vertical line represents when in-stream construction began on 15 June 2018.

Similar to N, no seasonal pattern was observed in the dissolved ortho-P concentrations. The pre-restoration concentrations of ortho-P varied from 0 to 0.314 mg/L at GR1, with corresponding values of 0–0.15 mg/L for the during-restoration period (Figure 8). At GR3, the pre-restoration values ranged from 0 to 0.038 mg/L, with corresponding during-restoration values of 0–0.23 mg/L.

3.3. Dissolved N and P and Suspended Sediment Flux Pre- and During Restoration

At GR1, the nitrate-N flux decreased slightly from the pre-restoration value of 10 kg/ha/year to the during-restoration value of 8.8 kg/ha/year, whereas at GR3 the nitrate-N flux increased from 6.3 to 9.7 kg/ha/year over the same period (Figure 9). The same response was observed for TDN flux, dropping from 16.1 to 10.4 kg/ha/year for GR1, but increasing at GR3 from 10.6 to 12.1 kg/ha/year (Figure 9). Stormflows contributed a greater amount of these fluxes at GR3 (Figure 9). The mean monthly nitrate-N flux at GR1 was 0.79 ± 0.19 kg/ha and 0.73 ± 0.11 kg/ha for the pre-restoration and during restoration periods, respectively (Table 2). At GR3, the mean was 0.49 ± 0.18 kg/ha pre-restoration and 0.82 ± 0.23 kg/ha during restoration (Table 2). The pre-restoration mean monthly TDN flux was 1.24 ± 0.31 kg/ha at GR1, while during restoration the mean was 0.85 ± 0.14 kg/ha (Table 2). At GR3, the mean was 0.82 ± 0.31 kg/ha pre-restoration and 1.02 ± 0.29 kg/ha during restoration (Table 2).

A comparison of the normalized net flux ((GR3-GR1)/GR1) for the pre and during-restoration periods indicated a significant increase for both nitrate-N ($p = 0.0028$) and TDN ($p = 0.0006$) (Table 2). The positive values of the normalized flux indicate an increase in flux downstream of the restoration reaches. Prior to restoration, the average monthly contribution of nitrate-N to the TDN flux was about 65% at GR1, while during restoration the contribution was 86% (not shown). At GR3, the contribution was 61% pre-restoration and 81% during the restoration.

The dissolved ortho-P flux was very low at both monitoring sites. The flux increased slightly at both the upstream and downstream sites from the pre-restoration to the restoration period. At GR1, the flux rose from about 0.063 to 0.075 kg/ha/year from pre-restoration to during restoration, and at GR3 the flux rose from 0.053 to 0.084 kg/ha/year. (Figure 9). In fact, the mean monthly ortho-P flux prior to the restoration was 0.0047 ± 0.0040 kg/ha at GR1, while during restoration the mean was 0.0063 ± 0.0070 kg/ha (Table 2). At GR3, the pre-restoration mean monthly flux was 0.0040 ± 0.0041 kg/ha, while during restoration the mean was 0.0070 ± 0.0072 kg/ha (Table 2). The standard deviation for the average monthly flux is very high at both sites, and there do not appear to be any seasonal or precipitation-related patterns for the ortho-P flux. No significant difference ($p = 0.452$) was found for the net normalized flux ((GR3-GR1)/GR1) between the pre-restoration and during-restoration periods (Table 2).

The suspended sediment flux at the downstream site (GR3) nearly doubled from the pre-restoration (1275 kg/ha/year) to the restoration period (2510 kg/ha/year). With an average measured concentration of 0.575% for PN in suspended sediments, the PN flux at GR3 for the pre-restoration period was 7.3 kg/ha/year, while that for the during-restoration period was 14.4 kg/ha/year.

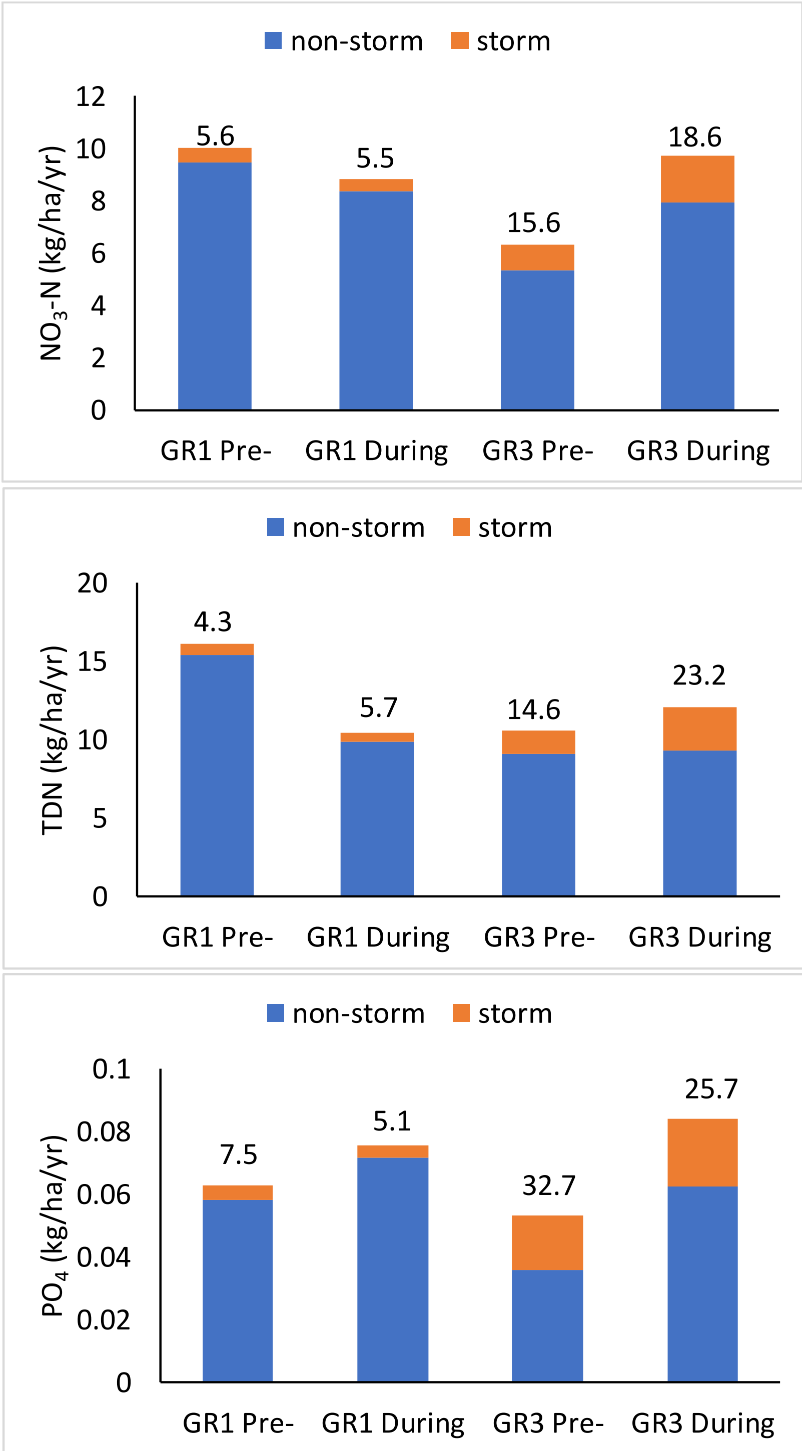

Figure 9. Stream water nutrient fluxes pre- and during restoration at the upstream (GR1) and downstream (GR3) sites for nitrate-N (NO_3-N), TDN, and ortho-P (PO4). Non-storm and storm values are reported with the % storm contributions listed above the bars. Storm event contributions were greater at the downstream site.

Table 2. Monthly nutrient flux means with one standard deviation. ANOVA *p*-values are listed for normalized net flux at GR3 comparisons ((GR3 flux-GR1 flux)/GR1 flux). $p < 0.05$ is significant, and N.S. is not significant.

Monthly Flux (kg/ha)	GR1 Mean ± s.d.	GR3 Mean ± s.d.	(GR3-GR1)/GR1 Mean ± s.d.	ANOVA p and Change Pre- vs. During
NO_3-N				
Pre-	0.79 ± 0.19	0.49 ± 0.18	−0.34 ± 0.35	$p = 0.0028$; Significant Increase
During-	0.73 ± 0.11	0.82 ± 0.23	0.15 ± 0.35	
TN				
Pre-	1.24 ± 0.31	0.82 ± 0.31	−0.35 ± 0.18	$p = 0.0006$ Significant Increase
During-	0.85 ± 0.14	1.02 ± 0.29	0.22 ± 0.39	
PO_4				
Pre-	0.0047 ± 0.0040	0.0040 ± 0.0041	0.59 ± 2.28	$p = 0.4520$ N.S.
During-	0.0063 ± 0.0070	0.0070 ± 0.0072	1.65 ± 3.68	

4. Discussion

While water quality monitoring was limited to the pre- and during-restoration periods, this large restoration project provided important insights into the drivers, challenges, implementation and other issues associated with stream restoration. We elaborate on these aspects following the key questions posed in the introduction.

4.1. Key Drivers, Challenges, and Constraints Associated with Site Selection, Design, and Restoration Implementation

There is an ongoing debate in the stream restoration community on how to identify, select, and implement streams for restoration [35]. Natural resource agencies and watershed managers are trying to decide if restorations should be implemented for the most polluted or degraded streams; if they should be implemented for streams with the greatest potential for nutrient reductions and/or habitat gains; or if we should select sites that are the most cost-effective, easy to access, and easy to implement for restoration. Addressing these issues and making appropriate decisions is important for advancing the practice of stream restoration. Of the 4700 projects in their database, [35] reported that 64 projects in Maryland and 67 in Pennsylvania were for streams that were listed on the state's 303d list (impaired waterways) and which had a TMDL associated with them.

The Gramies Run restoration site was located in a rural watershed, and the dissolved stream water nutrient (N and P) concentrations were generally on the low to moderate side. The nitrate-N concentrations in Gramies Run were in the neighborhood of nitrate-N concentrations recorded previously for a forested subwatershed of Big Elk Creek (0–2 mgN/L; [52]), and lower than those for the agriculturally influenced main stem of Big Elk Creek, where the baseflow concentrations varied around 4–5 mgN/L [53]. The benthic and fish IBI scores for Gramies Run were also relatively high for warm water Use I-P stream systems, suggesting that Gramies Run provided good water quality and habitat conditions to support less pollution-tolerant aquatic species. Given these generally favorable or acceptable water quality and habitat conditions, why was Gramies Run selected for a $4.2 million stream restoration?

Gramies Run did have legacy sediment deposits (Figure 2), particularly along streambanks in phases I and II, which were subject to fluvial and subaerial (freeze-thaw; Supplementary Figure S3) erosion [46], which contributed fine sediments to the stream waters. Furthermore, other than Phase I (Figure 3), which was on private land, all of the proposed restoration reaches were under state ownership (Maryland DNR), and acquiring permissions for construction was not a big hurdle. The private land owners in Phase I were also accommodating and provided the requisite permissions. The rural location of the site with no major highways or roads and/or commercial and residential communities also meant that construction could proceed without major negative impacts to transportation and local communities. Thus, in terms of logistics, this site represented a "low hanging fruit" for SHA for reducing sediment pollution and acquiring the required TMDL credits for their activities. This study site is thus a good example of the various factors and motivations that could come into play with regard to selecting sites for stream restorations.

Since Gramies Run was classified as a Use I-P stream, one important logistical hurdle that the restoration contractors faced was the instream closure period from 1 March to 15 June. This time window corresponded with the spawning of fish and other aquatic biota, and thus no instream construction was allowed during this time for their protection. This constraint was one of the factors, in addition to others, that likely contributed to the long 2-year construction period for this restoration. The contractors worked around this constraint by moving between the three construction phases and to activities that did not involve instream disturbance during this time window, such as tree planting on the floodplains. However, this meant that various portions of the restoration were in a continuous state of construction with implications for stream water quality (our water quality monitoring continued through the closure periods).

With regard to restoration design, multiple options were evaluated, ranging from the excavation and offsite removal of legacy sediments (greatest site disturbance) to a combination of partial legacy sediment removal and Natural Channel Design (NCD) [36]. Eventually, the latter option was implemented in an effort to protect mature riparian sycamores and avoid the substantial alteration of regional hydrology to preserve the Sensitive Species Project Review Area (SSPRA) in the vicinity of Phase II. In their preliminary surveys, the design engineers did identify the buried, precolonial organic horizon along the stream reach in Phase II [36]. However, they were not aware of its age, which we identified later using ^{14}C dating to a mean age of 950 ± 30 years BP [44]. The thickness (30 or more cm), length (~500 m), and depth near the baseflow water surface of the precolonial organic horizon (Figure 2) suggested there was likely a large swamp or a beaver-wetland complex in the vicinity of Phase II during the precolonial era. This assessment, if true, raises an interesting question —would a stream-wetland complex restoration design have been more appropriate at Phase II versus the currently implemented NCD design? These are important and difficult questions, but need to be addressed if we are to make ecologically appropriate restoration choices. We argue that rural stream sites such as Gramies Run, with low to moderate nutrient concentrations/loadings, likely provide the best opportunity to implement or experiment with restoration designs that mimic precolonial or pre-disturbance conditions (more on this in the last discussion section).

4.2. Changes in Water Quality Parameters and Fluxes Pre- and During Restoration

The pre-restoration suspended sediment flux for Gramies Run at the downstream site (1275 kg/ha/year) was slightly less than that reported by [9] for the Big Elk Creek (1348 kg/ha/year for the August 2017 to July 2018 period). The average flux for the Piedmont watersheds reported by [54] was 1037 kg/ha/year. The suspended sediment flux for Gramies Run, however, almost doubled from the pre-restoration to the restoration period at the downstream site (GR3—1275 to 2510 kg/ha/year). Not surprisingly, this change was also reflected in the sediment-bound N flux, which increased from 7.3 to 14.4 kg/ha/year. This could be due to a combination of increased sediment erosion associated with construction activities and/or the greater precipitation and runoff/streamflow amounts during the restoration period that likely enhanced the transport of sediments downstream. While precautions were taken at the site, such as the use of filter bags in reaches undergoing construction to prevent sediment from leaving the site, it is possible that sediments could have still escaped and flowed downstream. Several past studies support the possibility that the increased suspended sediment was the result of in-stream construction and bank destabilization if vegetation and trees were removed [34,55]. The study by [55] even reported increased SSC concentrations downstream of restored stream reaches post-restoration. In contrast, [27] found that the total suspended sediment flux decreased from the pre-restoration to the post-restoration period for Cyprus Creek in Anne Arundel County, MD. At the Big Spring Run restoration in PA [56], where much of the legacy sediments were excavated and removed offsite, a 50% decrease in sediment flux was observed post-restoration [57].

In comparison, while some legacy sediments were removed offsite at Gramies Run, the remaining sediments were used onsite for floodplain creation and restoration. In addition, given the long construction period and the staggered nature of restoration phases, there were always some stream reaches that were under construction. This meant that some sediment was always available for transport, particularly during the largest storms which occurred during the 2018 restoration period. Eventually, however, we expect that the sediment flux at Gramies Run should decrease post restoration as the floodplains stabilize and riparian grasses and trees take root and decrease the potential for erosion and sediment transport.

The annual dissolved nitrate-N and TDN fluxes increased at the downstream site during the restoration period. The stormflow contributions of nutrients, in particular, increased at the downstream GR3 site during restoration (Figure 9). Again, this could be the result of the stream restoration activities and/or the increased precipitation and runoff during the restoration period. The difference in monthly runoff between the downstream and upstream sites (GR3–GR1 runoff) was significantly higher during the restoration versus the pre-restoration period. In addition, the % of nitrate-N composing TDN flux was greater during the restoration period. This increased proportion of nitrate-N could be due to a number of factors or processes. For one, it could be a result of reduced in-stream processing—i.e., denitrification—since the stream water was pumped around the reaches undergoing restoration. It could also be due to the soil disturbance during in-stream construction, resulting in the mineralization and subsequent nitrification of organic N in the soils. This is supported by observations by [58], who concluded that conditions conducive to denitrification were not established until approximately 4 years post-restoration at Big Spring Run. Overall, they found that the nitrate-N load decreased by about 10% after the restoration [47]. Therefore, it is possible that the denitrification benefits of stream restoration projects might not be apparent for several years post-restoration and certainly not during the restoration period. It should be noted that floodplain reconnection (Protocol 3, [37]) was not counted toward the Gramies Run nutrient/sediment reduction credits. The N reduction credits via instream denitrification were only calculated for Protocol 2—i.e., hyporheic exchange. Thus, the TMDL-computed N credits for Gramies Run are likely an underestimate, since they do not include floodplain denitrification.

Other studies have reported mixed results with non-stormflow and stormflow conditions. For some stream restoration projects, the nitrate-N and total dissolved nitrogen increased from the upstream to downstream portions of restored reaches, but for others the N concentrations decreased from the upstream to downstream locations [51]. It should be noted that in the [51] study, pre-restoration data was not collected, so these values cannot be compared to the pre-restoration conditions. In addition, a reduction in the total nitrogen flux (including PN) through restored reaches was only reported for a third of the streams studied [51]. In another study, [27] did not monitor during the construction period or upstream of the restored stream reach on Cypress Creek, but pre- and post-restoration monitoring was conducted. They observed that the nitrate-N and total dissolved nitrogen flux decreased in the post-restoration period; however, the runoff also decreased slightly [27], which is the opposite of what occurred downstream at Gramies Run. The nitrate-N and total dissolved nitrogen decreased from the pre-restoration to post-restoration period during both baseflow and stormflow [27].

It is recommended that stream restoration projects should add in-stream features to store nitrogen and promote denitrification under both baseflow and stormflow in order to be effective [51]. In fact, [59] examined nitrogen load reduction for storm events of varying magnitudes at a stream restored using a sand-seepage wetland design. They found that the restored stream reduced nitrogen loads the most during low to moderate storms, ranging from 6.6 to 19 mm (0.26–0.75 inches). With streams that obtain most of their nitrogen input during stormflow, [29] suggests that streams should be restored to reconnect the stream and floodplain. Alternatively, streams that receive more nitrogen during periods of lower discharge should be restored to increase the hyporheic zone exchange, reconnect the stream and floodplain, and promote the storage of organic matter [29], which the Gramies Run restoration was designed to achieve.

There was no change in the ortho-P fluxes or concentrations at Gramies Run due to the restoration. In contrast to the observations from this study, [27] observed an increase in ortho-P flux following stream restoration, which was attributed to sediment desorption in anoxic areas. In comparison, soluble reactive phosphorus (SRP) flux decreased following restoration at Big Spring Run [60]. The SRP concentration fluctuated in the years following the restoration, decreasing but then increasing for several years, until finally decreasing again [60]. This result underscores that it could take a few years after the restoration for nutrient concentrations to decrease in runoff.

We implemented an upstream-downstream and pre- and during-restoration scheme to monitor the changes in stream water quality for this restoration study. We plan to continue this monitoring for the next 3–4 years to assess the effectiveness post-restoration. Sampling was performed for non-stormflow and stormflow conditions, and the work of [51] has highlighted the need for assessing both types of flow conditions. High flows and loads associated with storms could result in the reduced retention of sediments and nutrients [51]. Variability in precipitation over the study periods, as in this study, or other hydrologic perturbations could be complicating factors that should also be recognized and addressed. The sampling in this study was limited to a second-order stream where the cross-sectional flow variability was small; however, larger streams with greater flow and sediment variability would likely require a more rigorous sampling design and plan—e.g., [61].

4.3. Effectiveness of Restoration as a Function of Watershed Landuse and Location within Watersheds

Stream restoration credits for nutrient and sediment reductions [24] necessitates that the estimated reductions are being achieved, especially in Maryland, which is investing heavily in stream restoration to help meet the Bay TMDL goals [32]. The size and cost of projects is also increasing, but mixed results have been obtained in terms of their effectiveness to meet sediment and nutrient reductions [32,35]. This mixed response has been influenced by the land use in the watersheds and the consequent loadings, size, and location (headwater or lowland stream locations) of the restoration sites within the watershed [29,51]. Stream restoration in highly degraded urban watersheds has not been found to be very effective, since the watershed loadings are either too high or flashy/variable for the restorations to effectively manage and mitigate [62]. Similarly, [51] suggest that stream restoration projects in the Coastal Plain that are located closer to the bay tend to be better at reducing nitrogen loads than streams further up in the bay watershed due to their lower gradient and longer water residence times, which promote particulate N retention and also enhance in-stream nitrogen processing.

The Gramies Run restoration site was located in a small, rural, Piedmont watershed with low to moderate upland loadings primarily from pasture and some equestrian facilities in the headwaters of Gramies Run. The concentrations of N and P in Gramies Run were on the low to moderate side and much less than those typically observed for agricultural and urban watersheds in the region. The size of the watershed at 7.89 km^2 was also relatively small for the length (1668 m) of restoration. The restoration also occurred lower in the watershed and close to its outlet into Big Elk Creek. Given these favorable watershed conditions, we expect that the Gramies Run restoration should be effective in reducing the watershed N and P loadings if the restoration features and proposed stream and floodplains function as designed.

4.4. Stream Restoration Cost Effectiveness and Implications for the Chesapeake Bay

An important factor when assessing the efficacy of stream restoration projects is the cost of restoration. In the stream restoration community in the US, these costs are typically expressed in terms of dollars per linear foot of restoration or per pound of N or P reduced. Of the stream restoration projects examined with cost data available (Table 3), the Gramies Run restoration had the highest cost per linear foot ($767) of streams restored. This cost, however, lies within the range of costs ($500–1200 per linear foot) for urban stream restoration projects [31]. Granted, Gramies Run also had the highest estimated nutrient and TSS removal of the projects evaluated (Table 3), but these reductions are yet to be seen since the project has just been completed. If the costs are estimated for the amount of N and

P reduced (project cost in $ divided by the annual rate of N and P reduced in Table 3), for Gramies Run we get cost estimates of $2089/lb of N and $10,880/lb of P. In terms of N costs, the Gramies Run values are not far off the cost estimates for other restoration projects listed in Table 3 and the typical value of ~$2000/lb of N for projects implemented in the Chesapeake Bay watershed [63,64]. The cost estimates per pound of P, however, vary considerably across the projects ($2493 to $10,880/lb of P in Table 3 and [63]) and Gramies Run had the highest cost per pound of P.

Table 3. Comparisons of various stream restorations for costs, restoration techniques, and nutrient and sediment reductions.

Stream Name and Location	Restoration Technique	Length of Stream Restored (Meter & Feet)	Pre- or Post-Restoration Monitoring?	Nutrient/ Sediment Reduction?	Total Cost, Cost Per Feet and Per Pound of N and P	Citation
Gramies Run, Cecil County, MD (rural/agricultural, forested)	Partial legacy sediment removal, bank grading and stabilization, in-stream structures, and riparian vegetation planted.	1668 m 5473 linear feet (lf)	Pre- & during, no post restoration	Proposed reductions: TN *—2010 lb/year, TP *—386 lb/year, TSS *—367 tons/year. No reduction in nutrient or sediment flux during the restoration (this study).	$4.2 million; $767; $2089/lb N; $10,880/lb P	[36]
Big Spring Run, Lancaster County, PA (agricultural)	Legacy sediment removal, woody debris added to stream, riparian vegetation planted.	1950 m 6400 lf	Both	TN *—316 lb/year, TP *—174 lb/year. Sed. flux—109 tons/year, NO_3-N concentration —11% reduction, SRP flux—37% reduction.	$650,000 $102; $2056/lb N; $3735/lb P	[57,60]
First Mine Branch, Baltimore County, MD (agricultural)	Streambank stabilization, riffle areas created, wetlands created.	658 m 2160 lf	Both	N—216.75 lb/year, P—196.53 lb/year, TSS—64.85 tons/year. No change observed in the total dissolved nitrogen or DOC.	$489,958 $227; $2260/lb N; $2493/lb P	[65,66]

Table 3. *Cont.*

Stream Name and Location	Restoration Technique	Length of Stream Restored (Meter & Feet)	Pre- or Post-Restoration Monitoring?	Nutrient/ Sediment Reduction?	Total Cost, Cost Per Feet and Per Pound of N and P	Citation
Plumtree Run, Harford County, MD (urban)	Streambank graded, streambank stabilization, wetlands created.	378 m 1240 lf	Both	N—97.13 lb/year, P—85.01 lb/year, TSS—64.85 tons/year. No change observed in total dissolved nitrogen or DOC.	$501,600 $405; $5164/lb N; $5900/lb P	[65,66]
Bear Cabin Branch, Harford County, MD (suburban)	Streambanks graded to create microtopography, wetlands created.	1147 m 3675 lf	Both	N—216.75 lb/year, P—31.96 lb/year, TSS—10.5 tons/year. No change observed in total dissolved nitrogen or DOC.	$165,608 $45; $764/lb N; $5181/lb P	[65,66]
Red Hill Branch, Brampton Hills stream restoration project, Howard County, MD (urban)	Streambank and bed stabilization and grading, grade controls added, riparian vegetation planted.	964 m 3165 lf	Both	TN—619 lb/year, TP—644 lb/year, TSS—116 tons/year.	NA	[67]
Sandy Creek, Durham County, NC (urban)	Three restoration phases: new stream channel formation, earthen dam and weirs, stormwater treatment wetland.	600 m 1969 lf	Both	Significantly lower NO3-N, TN, and SRP concentration (downstream-upstream) after the final restoration phase than pre-restoration.	NA	[68]
Accotink Creek, Fairfax County, VA (urban)	Bank stabilization, riparian vegetation planted.	548 m 1800 lf	Both	Stormflow suspended sediment concentration increased post-restoration. No reduction in nutrient concentration post-restoration.	NA	[55]

* TN–total nitrogen; TP–total phosphorus, TSS–total suspended sediment.

Due to the variety of stream restoration projects, the differences in watershed land use (urban, agricultural, forested), the low availability of projects with pre-restoration data, and the short post-restoration monitoring periods, it is difficult to determine if one restoration technique is more cost-effective at nutrient and sediment removal than another. However, [69] examined the cost effectiveness of varying techniques to reduce nutrients and sediment. They found that "legacy sediment mitigation"—i.e., legacy sediment removal at areas with high erosion rates—was the least expensive method for sediment and phosphorus reduction to implement compared to wetland restoration, creating a forest riparian buffer or creating a grass riparian buffer [69]. Nitrogen reduction for "legacy sediment mitigation" costs a little bit more than for the riparian buffers and cover crops but is much less than wetland restoration [69].

Meeting the Chesapeake Bay TMDL becomes more urgent as the target year of 2025 approaches, and companies are thinking of creative ways to help organizations such as MD SHA looking to meet their individual TMDL goals and credits [64]. For example, Ecosystem Investment Partners (EIP) is investing in large-scale stream restoration projects and is seeking to sell stream restoration credits [64]. Two such projects are in Cecil County, Maryland, and are being completed for SHA [64]. EIP is restoring about 16 km (~10 miles) of stream for $23 million, which SHA does not have to pay for until the project is finished and nutrient reductions are established [64]. By restoring larger sections of streams, the cost per pound of nutrient and sediment declines [64]. For these two projects, the cost per linear foot is only about $436 [64], which is on par with the Plum Tree Run restoration and over $300 less per linear foot than the Gramies Run restoration (Table 3). New innovative approaches such as these increase the importance of water quality monitoring for the projects to ensure that estimated reductions are being achieved.

4.5. Enhancing Stream Restoration by Leveraging Pre-Disturbance Soils and Emphasizing Soil Health

Like the Gramies Run restoration, most stream restorations are primarily focused on geomorphic form and function to mitigate the effects of flow velocity and shear and reduce bank erosion [28,34,70–72]. However, the focus on geomorphic form has occasionally come at the cost of favorable biogeochemical conditions and services. For example, the loss of fine sediments and organic matter in stream beds following restoration has been shown to decrease the potential for the denitrification loss of N [73,74]. Not surprisingly then, many of these restoration efforts are not achieving their maximum potential for improving the ecological health of floodplains and streams [70,75]. Recently, there have been increasing calls from within and outside the stream community to go beyond geomorphic form and truly integrate the ecology and biology of streams and plant communities in the design and implementation process of restorations [75,76]. Similarly, there is an ongoing debate in the stream restoration community if pre-disturbance analogues of stream and floodplains exist, and if and how they can be used to guide stream and wetland restorations [75–77]. We argue that not only do we need to include pre-disturbance ecosystem attributes in stream restorations, but we also need to place a greater emphasis on enhancing soil health and microbial functions in these restorations.

While the restoration at Gramies Run included detailed plans for the design of the channel, no consideration was given to the 950-year-old precolonial organic horizon that was buried below the legacy sediments (Figure 2). Given its thickness and vertical position on the bank (the organic horizon was at or just above the stream base level), we speculated that, prior to legacy sediment deposition and accumulation, this organic horizon likely constituted the "floodplain" and provided valuable ecological services for this ecosystem. Pursuing that thinking, we requested that the restoration engineers and contractors preserve and maintain the historic organic layer to the maximum extent possible. While the restoration engineers and contractors agreed in principle, given their pre-set design constraints of stream base levels and bank grades they could only preserve or maintain the ancient organic soils at a few locations on the floodplain in Phase II.

Our recent work has revealed that the microbial composition of the relict, organic horizon at Gramies Run was different from the overlying legacy sediments and surficial horizons [43]. Furthermore, while denitrifying functional genes (e.g., nosZ) indicated very high abundance in the relict organic soils [43], the denitrification enzyme assay (DEA; a measure of denitrification rate, [78]) indicated that the denitrification potential for the organic soils was very low or zero (unpublished data). While the nosZ gene abundance and DEA results appeared to contradict, the low DEA values for relict, hydric soils are in agreement with the handful of studies that have investigated this response [79,80]. This suggests that although denitrifying microorganisms may be present in the buried, relict, hydric soil layers, they may be inactive or dormant. Some studies suggest that the denitrification rates in relict, buried soils may be initially very slow and significant recovery could take 1–2 years or more [81,82]. We speculate that the low denitrification rates in buried organic layers could be due to disconnection from the surface and lack of fresh organic carbon and microbial communities. We plan to address this recovery and the role that relict, organic soils play on restored floodplains at Gramies Run through a newly funded project. In addition to investigating the nutrient and microbial recovery of relict soils on the new floodplains, we will also monitor the stream water quality at the Gramies Run restoration site for the next 3–4 years to assess the post-restoration effectiveness of this restoration project. We expect that the water quality gains from the restoration may not be immediate and could take a year or more.

These observations underscore that the effectiveness and ecological recovery of restored streams extends beyond geomorphic form and function. In addition to geomorphic, hydraulic, and hydrologic changes, we need to account for the recovery of soil and microbial health and the newly planted riparian vegetative communities. In particular, historic soils could provide unique, native, microbial communities and associated functions that may be missing in contemporary, highly disturbed, polluted, and potentially microbially compromised soils [83] (e.g., by the excessive use of agricultural pharmaceuticals and other chemicals). Microbiomes from historic soils could also be used to “inoculate” or “seed” restored floodplains where such microbiomes are missing. These native organic soils and their microbiomes could also be preferable to the non-native, commercially mixed “topsoils” or “soil conditioners” that are routinely used by contractors in restorations to “polish” floodplain soils. If ecosystem components such as soils and microbes are considered at the outset and strongly integrated in stream and floodplain restoration designs, it is likely that we could have restorations with shorter recovery times and that are more ecologically resilient and environmentally sustainable.

5. Conclusions

While water quality monitoring was performed for only the pre- and during-restoration phases, a large $4.2 million restoration for a rural stream in Maryland Piedmont provided important insights into a variety of issues associated with stream restoration. Site selection for stream restorations is driven by numerous factors and site access and permissions for restorations could be important determinants, in addition to the potential mitigation of legacy sediment inputs. Similarly, in addition to reducing sediment erosion rates, the design approach for restoration could be dictated by the natural resources and site hydrologic conditions that need to be preserved or maintained at the site. We do recommend though that pre-disturbance site conditions (if known or ascertained by site features) should be given careful consideration while deciding or designing the restoration approach. Pre- and during-restoration water quality monitoring revealed that sediment and nutrient (nitrate-N and TN) loads increased during the restoration—likely due to the construction activities and increased rainfall runoff during the restoration period. We expect, however, given the removal of erodible legacy sediment stream banks, reduced flow velocities and shear, and increased hydrologic connection between the streams and newly created floodplains, that sediment and nutrient exports will decline with time. How quickly this decline will happen and whether it meets the design reductions of 912 kg/year (2010 lb/year) of total N, 175 kg/year (386 lb/year) of total P, and 367 tons/year of total suspended sediment (TSS) erosion will have to be determined through post-restoration monitoring for at least 3–4 years. We also recommend that stream restorations place a greater emphasis on improving soil health and include soil nutrient and microbiome attributes in floodplain design. This should especially be the case for sites that possess historic, pre-disturbance, or pre-settlement soils and native microbiomes. The inclusion of such native soils and microbiomes will result in restorations that are more environmentally sustainable and resilient.

Supplementary Materials: The following are available online at http://www.mdpi.com/2073-4441/12/8/2164/s1. Figure S1: 1858 Martenet map showing historic mill dams (circled in yellow) along Gramies Run (previously known as Fulling Mill Run), Figure S2: Partial construction along Phase 2 showing restored (left) and unrestored (right) reaches. The junction represents a construction pause for the no-instream work period—1 March to 15 June. Note the buried organic horizon the right and the corresponding level of the newly created floodplain on the left, Figure S3: Streambank at Gramies Run (along Phase II restoration) showing freeze-thaw drool and subaerial erosion, Figure S4: SSC-Turbidity relationship to convert turbidity values to SSC concentrations.

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