Nutrient Release Potential during Floodplain Reconnection: Comparison of Conventional and Ecological Stream Restoration Approaches

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Scholarly Abstract

In the last few centuries, many streams in the eastern United States have been severely disturbed by land use change and are now disconnected from their original floodplain due to the aggradation of legacy sediment. Currently, stream-floodplain reconnection is advocated as a stream restoration practice to take advantage of ecosystem services. The objective of this study is to compare two current stream restoration approaches for their nutrient flushing ability: 1) a conventional approach leaves legacy sediment on the floodplain; and 2) an ecological approach that involves removing the accumulated legacy sediment in order to restore the original floodplain surface wetland, revealing a buried A soil horizon. Soil cores were taken from the surficial legacy sediment layer and the buried A soil horizon in the floodplain of a 550-meter reach of Stroubles Creek in the Valley and Ridge province near Blacksburg, VA, to evaluate potential for flushable DOC, TDN, NO_3^-, NH_4^+, and SRP content. In addition, an inundation model was developed to evaluate the extent of flooding under the two restoration scenarios. The inundation model results and nutrient flushability levels were then used to simulate the release of nutrients as a function of stream restoration approach. Results indicate that the buried A horizon contained less flushable nutrients, but the ecological restoration would have a higher frequency of inundation that allows for more flushable nutrient release at the annual scale. Understanding the nutrient release potential from the floodplain will provide the ability to estimate net nutrient retention in different stream-floodplain reconnection strategies.

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General Abstract

Stream restoration is a popular practice in the United States used to fix a degraded stream to have the functions, habitat, and characteristics of a natural stream. Done correctly, the practice can be beneficial to stream health by slowing flows and allowing for a decrease in nutrient loads to downstream waters. The idea of a natural stream is widely debated because there are few streams left in the United States that have not been impacted by agriculture and urbanization. Man has significantly changed most streams and the land around them, while leaving little record of what the original stream looked like. This research was conducted on Stroubles Creek near Blacksburg, VA, and it compares two common methods of restoring a stream. One method designs the stream channel to have a specific pattern and shape and disregards the soils around the stream. The second method looks in the soils for clues to bring the stream and its floodplain back to its original level. By examining the soils around Stroubles Creek, we found evidence of the original channel and floodplain. We tested those soils to find out which restoration method would provide the maximum decrease in nutrient loads and then built a model to simulate the differences in flooding between the two methods. After comparing the two restoration methods, our findings indicated that restoring a stream to its original level would deliver a greater benefit of slowing floodwaters, but it would provide a disadvantage in an increase of nutrient loads.

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List of Abbreviations

BMPs	Best Management Practices
C	Carbon
Fe	Iron
N	Nitrogen
NH_4^+	Ammonium
NO_3^-	Nitrate
OM	Organic matter
P	Phosphorous
PO_4^{3-}	Orthophosphate
SOC	Soil organic carbon
SOM	Soil organic matter
SRP	Soluble Reactive Phosphorous

1 Introduction

1.1 Background and Motivation

Many streams in the eastern U.S. have been severely impacted by human activities in the last few centuries. Land use change has altered the flow regime, sediment budget, and physical characteristics of streams across the mid-Atlantic region. These streams are now hydrologically disconnected from their original floodplain because of the aggradation of legacy sediments from upland erosion, downcutting, streambank erosion, or upstream inputs. Currently, stream-floodplain reconnection is advocated during stream restoration to take advantage of floodplain ecosystem services such as sediment removal and flood peak attenuation. Two current floodplain reconnection strategies have been advocated: 1) a conventional stream restoration approach involves streambanks that are graded back or a streambed that is raised to allow for more connection with legacy sediments left on the floodplain; and 2) an ecological restoration approach involves removing the accumulated legacy sediments in order to restore the pre-disturbed floodplain. The pre-disturbed floodplain level may be found in the existing soil profile by identifying a buried A horizon (previous land surface) indicative of a wetland surrounding the stream before settlement disturbance.

In this study, we seek to evaluate the potential for nutrient release from the floodplain soils for each restoration approach by sampling from the surficial legacy sediment and the buried A horizon. Background data already show that legacy sediments are high in nutrients during certain seasons. Nutrient release from floodplain soil during periods of inundation may allow for an unintended nutrient loading to the stream. Nutrient loading to downstream surface waters can cause water quality problems such as eutrophication. Our hypotheses include: 1) the buried A horizon at Stroubles Creek contains less flushable nutrients than the surficial legacy sediment; 2) an ecological restoration would provide for a decrease in flushable nutrient loading to the stream; and 3) an ecological restoration would increase inundation frequency and area which may lead to added water quality benefits such as increased sedimentation, enhanced denitrification, and regrowth of native vegetation.

1.2 Literature Review

1.2.1 Historic Floodplains and Legacy Sediment

Legacy sediment is soil that eroded from upland areas under intensive land use change that has accumulated in alluvial areas over a relict surface soil horizon (Niemitz et al., 2013). The legacy sediment is most easily found in sites directly upstream from existing or breached milldams that were originally constructed in small-order streams to provide hydropower for milling processes during early European colonization of eastern North America and continued through the 19^{th} century (Walter and Merritts, 2008). Legacy sediment is not strictly associated with only historic milldam sites. A buried surface soil horizon in many floodplains of eastern North America can indicate the time period when European settlement began in an area. Before European settlement, most of the ridge-and-valley and piedmont areas were described as having streams that "ran clear under non-storm conditions" surrounded by hardwood and mixed forests (Jackson et al., 2005). Following settlement, widespread deforestation and agricultural land use caused accelerated upland soil erosion and resulted in valley-bottom sedimentation (Hill et al., 2004). The complex stream-riparian-wetland systems were then buried by the legacy sediment, which then formed floodplain terraces that limited stream-floodplain interaction (Walter and Merritts, 2008).

Some of these buried wetland soil horizons occur naturally. Where alluvial processes have dominated the formation of the riparian zone, the soil subsurface often contains buried horizons that can be thousands of years old. Ancient glacial processes have also caused the burial of wetland surface soils in areas of the northeastern United States (Gurwick et al., 2008b). These are considered relict processes, whereas this paper will mainly focus on the historic process of wetland soil burial.

The characterization of this buried wetland soil has been common across several studies in different areas of eastern North America. Hill et al. (2004) found buried wetland soils at depths of 0.4 m to 3 m in agricultural headwater riparian zones in southern Ontario, Canada. Walter and Merritts (2008) found a buried wetland soil evidenced by a 0.5 to 1 m thick dark (Munsell color 10 YR 2/1), organic-rich silt loam at depths of 1.4 to 4 m below the surface in floodplains of first- through third-order streams in southeastern Pennsylvania. In the piedmont of Georgia, Jackson et al. (2005) identified the barrier between the pre-historic floodplain and the historic floodplain sediments by a stratigraphic layer of sands and loamy sands with high mica content over a boundary layer of reduced clay with low mica content. They found the depth to the buried wetland soil to be an average of 12.2 cm over an entire watershed.

The nutrient composition of legacy sediment and buried wetland soils is an important factor in soil and water quality. Wetlands in floodplains serve as sinks for the SOC being lost from the upland landscape. This is occurring when stable forests are cleared for agriculture or construction, causing increased C mineralization and the disruption and erosion of soil aggregates, carrying SOC with it. Hydric soils are long-term sinks for SOC because the anaerobic conditions slow C mineralization, allowing for the buildup of SOC (Ricker et al., 2012).

A study in a riparian zone of an agricultural headwater stream showed that nitrate mobilization and mineralization in the buried A horizon was dependent on the water table (Hill, 2011). When the water table dropped below the buried wetland soil for several months because of a drought, they saw a 16-fold increase in the soil nitrate pool at that depth. This showed that buried organic-rich soils in riparian areas can serve as a nitrogen source during periods of water table drawdown.

Weitzman et al. (2014) hypothesized that legacy sediment would have high nitrification rates due to high agricultural nitrogen (N) inputs and high oxygen (O_2) levels, while the buried relict hydric soils would be nitrate (NO_3^-) sinks by way of denitrification due to high carbon (C), low O_2 levels, and longer hydrologic residence times. They did find net nitrification rates ranging from 9.2 to 77.9 g m^{-2} yr^{-1} for legacy sediment, meaning it serves as a NO_3^- source as expected. Contrary to their hypothesis, the relict hydric soil did not have sufficient denitrification rates comparable to the nitrification rates of the surface legacy sediment. This means that a buried wetland soil does not perform the ecological services expected from a surface wetland soil that could offset the mobilization of NO_3^- from surficial legacy sediment (Weitzman et al. 2014). This was further evidenced by Koval (2012) in observations of depressed denitrification rates in buried hydric soil when compared to those in natural or restored wetlands.

It is proposed that when wetlands are restored with increased denitrification as a goal, soil phosphorous (P) export can be a consequence of those conditions. This is because inorganic P is usually bound to oxidized iron (Fe) in a wetland soil, but when the Fe becomes reduced under long periods of inundation, P is released from that complex. The available P can also increase if that soil is left dry for a while, such as during the summer water table drawdown or a drought, due to enhanced organic P mineralization (Ardon et al., 2010). Olila et al. (1997) simulated the effects of wetland soil drainage on available P. Two treatments were done on peat soil cores where one set was drained for six weeks and the other set was drained for three weeks.

Dissolved reactive P flux from the soil drained for six weeks was 10-fold higher than the soil drained for three weeks. There was also an increase in bioavailable inorganic P and reduction in labile organic P. In another study, Dunne et al. (2011) looked at available P in wetland soils within different pasture types, an important location of study since there has been a large conversion of wetlands to land suitable for agricultural productivity. They found that surface soils had much greater concentrations of organic and inorganic P than subsurface soils. The conclusion was that the legacy P stored in wetland soils from dairies, when combined with best management practices to reduce nutrient loading to these systems, can result in a sort-term release of wetland soil P to the wetland water. Another study was done to compare P flux to hydroperiod prior to flooding by Dunne et al. (2010). They found similar release of soil P between upland, deep marsh, and shallow marsh soils. They did see a significant negative relationship between P release in deep marsh and hydroperiod. Therefore, to decrease P release from wetland soils, the soils should stay wet rather than dry, prior to flooding. Olila et al. (1997) also showed that P at the surface was more easily released than subsurface P due to environmental stress and instability. Ardon et al. (2010) studied a 440 hectare hydrologically-restored riverine wetland complex in the coastal plain of North Carolina and compared it to two disturbed reference wetlands and the adjacent agricultural field. They observed a release of soluble reactive phosphorus (SRP) from the restored wetland that was two to eight times higher than that of pre-restoration conditions, the reference forested wetland, and the reference adjacent agricultural field. They expect that this restored wetland has the ability to release legacy fertilizer P for up to a decade following hydrologic restoration. Other common factors that increase labile soil P include decreasing particle size (Makarov, 2004) and increasing SOM (Young and Ross, 2016).

A buried wetland soil often contains a seedbank from the native wetland vegetation. This has an added benefit in that if a restoration were to occur, the native vegetation would be easily established on its own. The O'Donnell et al. (2014) study showed that when sediment was sampled from a bar, a bench, and a floodplain in Australia, a total of 9456 seedlings emerged with 131 different species (83 native and 47 exotic), with most of these coming from a depth of 25-30 cm. They also saw that species richness decreased with depth into the floodplain soil. In conclusion, buried wetland soils can be hotspots for microbial activity, provide denitrification potential, and be sinks for organic carbon (Hill, 2011).

1.2.2 A Brief History of Stroubles Creek

The Stroubles Creek watershed has seen a long history of land use change since it was first settled in 1740. The headwaters were chosen for development because of the abundant springs and seeps. Urbanization continued to shape this part of the watershed when a land-grant university was founded in 1872 that later became Virginia Polytechnic Institute and State University (Parece et al., 2010). Below the headwaters, the watershed saw land use change in a different way. There were substantial agricultural practices spanning from the University's agricultural experimental station along the creek to vast pasture fields for various types of livestock with the creek as their water source (Beck, Jr., 1951).

Significant effort was made to determine if the sediment trapping effect of mill dams was a factor on Stroubles Creek. Thorough investigations of archived maps, reports, and photos, found that there has been only one known mill on Stroubles Creek. That mill belonged to the Millers near where Smithfield Plantation Road currently crosses Stroubles Creek. The mill was deconstructed in the early 20 century, but the Millers' cabin and outbuilding are still standing today. This location is approximately 1.6 km upstream of our study reach. Therefore, the aggradation of legacy sediment in the floodplain of the study reach was independent of any mill dam effects.

One of the earliest reports on the physical condition of Stroubles Creek can be found in a Master's thesis written in 1951 by V.P.I. Sanitary Engineering student William McKinley Beck, Jr. In the summers of 1948 and 1949, he was studying the creek for effect of the sewage discharge from a treatment plant near the Duck Pond. He did a physical assessment of the entire creek below the Duck Pond. In the stretch of where our current study site is, Beck (1951) reported seeing "rolling farm land and herds of cattle were kept in this area" and "the few trees were well back from the creek." This shows that by the late 1940's, this floodplain of Stroubles Creek had been completely deforested and turned into land for agriculture. His observations of the stream conditions included that "the bottom was almost completely carpeted with deep sludge deposits" and "the water was clear, though numerous particles were being swept along in suspension," so it is evident that the accumulation of legacy sediment had already occurred at that time (Beck, Jr., 1951).

In 1996, Stroubles Creek was first found to have a benthic macroinvertebrate impairment, and in 1998, five miles of the stream, including our study reach, was listed on Virginia's 303(d) TMDL Priority List. Possible stressors included organic matter, nutrients, and sediment, with sediment being selected for the TMDL development study. Likely sources of sediment for this watershed include animal grazing, eroded banks, hardened channels, impervious cover, and other human-induced land-disturbing activities (BSE, 2003).

During 2008-2010, Stroubles Creek underwent restoration of a 2.1 km reach to reduce sediment loading from eroding banks. This reach is now called the Stream Research, Education, and Management Lab, or StREAM Lab, and includes the study site for this thesis project (http://www.bse.vt.edu/site/streamlab/). Three stream restoration methods were implemented, and this study looks at one of those. The restoration method used on our study reach included livestock exclusion, reshaping of the banks with a 3:1 slope, and replanting, otherwise known as a Priority 4 restoration. This was done on a 550 m portion of the stream. The stream slope and baseflow channel went unchanged (Thompson et al., 2012).

1.3 Research Objectives

The legacy sediments that cover buried wetland soils are high in nutrients, which can increase exports to downstream ecosystems and contribute to eutrophication, among other environment impairments (Niemitz et al., 2013). A stream restoration method is proposed to remove the legacy sediment and restore the wetland soil to the surface in order to have increased ecosystem services. Few studies have looked at the nutrient release side-effects of exposing this buried A horizon in a stream-floodplain reconnection. Therefore, the objectives of this study are to evaluate the flushable nutrient availability of floodplain soils in surficial legacy sediments and the buried A soil horizon of Stroubles Creek in order to answer the following questions:

1. How do the flushable nutrient levels vary with depth in the floodplain soil?
2. How do the two restoration approaches differ in river-floodplain connection?
3. How do the two restoration approaches differ in nutrient flushing potential to the stream?

1.4 Organization of Thesis

This document is organized around a journal article that will be submitted for publication in *Ecological Engineering*. This article is located in Section 2 of this thesis.

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2 Nutrient release potential during floodplain reconnection: Comparison of conventional and ecological stream restoration approaches

In Preparation for *Ecological Engineering*

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2.1 Introduction

Natural floodplains perform a host of ecosystem services such as maintenance of native aquatic biota, habitat for terrestrial species, and biogeochemical processing, as well as the economic services of flood control, nutrient removal, and groundwater recharge (Tockner and Stanford, 2002). Since the 17^{th} century, anthropogenic management practices including logging, agriculture, urbanization, and other resource exploitation, have left a lasting, or legacy, effect on floodplains in the U.S. (Grove, 1992). Along the streams of the mid-Atlantic, Wolman and Leopold (1957) noted that these human activities lead to, among other geomorphic consequences, levels of vertical accretion of sediment that were higher than natural processes could deposit in floodplains. This lateral disconnection of the stream channel from the floodplain has resulted in deteriorated ecological communities, accelerated streambank erosion, and increased downstream sedimentation (Rosgen, 1997; Ecological Restoration, 2009). Anthropogenic legacy management practices also serve as a long-term source for nutrient loading within a transport-limited watershed (Basu et al., 2010). Floodplain reconnection during stream restoration has been an increasingly common tactic to mitigate these effects; the ecological benefits and successes of which have been widely disputed (Bernhardt and Palmer, 2011).

The restoration of riparian floodplains has shown to provide many important nutrient removal and sediment trapping functions. Nutrient dynamics have been studied in large floodplains, such as the Atchafalaya River Basin, where Scott et al. (2014) found a 16.6% reduction in nitrate (NO_3^-) of floodwater, but a slight release of ammonium (NH_4^+) and soluble reactive phosphorus (SRP) from the floodplain. Similar effects have been seen in small floodplains, such as Stroubles Creek where this study takes place, by Jones et al. (2015) where they found a mean NO_3^- storm load removal of 10% during simulated floods across all seasons.

Filoso and Palmer (2011) found that restoring streams to a wetland-stream complex can reduce N export by 13% annually and by 20% during storm flows. Despite floodplain nutrient removal mechanisms and increased use of BMPs, some managed watersheds have not shown significant decrease in nitrogen (N) and phosphorus (P) export likely due to the long-term biogeochemical stationarity of legacy nutrient sources (Basu et al., 2010).

Here, we propose two separate opposing ideologies on how to achieve the proper ecological state of a restored stream. Most stream restoration techniques have a common element: reshape the degraded channel based on fluvial geomorphological elements of the stream (Palmer et al., 2014a). We use the term “conventional” restoration to describe these common, form-based techniques that focus on restoring the hydrologic and sediment regimes by reshaping the channel and restoring sinuosity. The end-goal is a self-regulating, stable channel that the practitioner assumes would, in turn, restore species assemblages, nutrient cycling, and other ecological processes (Palmer et al., 2014b). In contrast, we declare an “ecological” restoration one that is process-based, where the primary focus is restoring the natural ecological features (e.g. low-lying riparian wetland) and processes (e.g. nutrient dynamics) of the stream corridor (Palmer et al., 2014b). In this setting, the floodplain is restored to its natural condition by removing accumulated legacy sediment and the stream channel is left in a dynamic equilibrium in which it can freely change over time.

The main focus of this study was an analysis of flushable nutrients in order to quantify downstream water quality impacts. Here we aimed to (i) characterize the extent of the legacy sediment and buried A horizon in a section of the StREAM Lab at Stroubles Creek, Blacksburg, VA, (ii) quantify the release of dissolved organic carbon (DOC), SRP, NO_3^-, NH_4^+, and total dissolved nitrogen (TDN) from the two soil layers, and (iii) model the inundation area using the existing surface as a conventional restoration and the surface of the buried A horizon as the ecological restoration. This information allowed us to evaluate the two restoration techniques based on nutrient flushing throughout different inundation scenarios and timescales. Our hypotheses include: 1) the buried A (Ab) horizon at Stroubles Creek contains less flushable nutrients than the surficial legacy sediment (LS); 2) an ecological restoration would provide for a decrease in flushable nutrient loading to the stream; and 3) an ecological restoration would increase inundation frequency and area which may lead to added water quality benefits such as increased sedimentation, enhanced denitrification, and regrowth of native vegetation.

2.2 Materials and methods

2.2.1 Site description

Stroubles Creek is a third order stream at the 1:24,000 topographic scale and is located in the Ridge and Valley physiographic province of southwest Virginia. Its headwaters consist of springs that start in the Town of Blacksburg and are mostly conveyed through underground pipes and culverts as the stream flows through the Town and the Virginia Tech campus (Parece et al., 2010). Downstream of the town, the stream is surrounded by agricultural fields. The study site has a contributing drainage area of approximately 15 km^2 with 84% urban, 13% agriculture, and 3% forest landcover (Jin et al., 2013). The area was settled around the year 1740, so we assume this is when the majority of the land clearing began. (Parece et al., 2010). Prior to 2008, the floodplain of the study site was livestock grazing with direct access to the stream. It is in this agricultural reach that, in 2008, a 2.1 km portion of Stroubles Creek was restored to reduce sediment loading from eroding banks. Three stream restoration methods were implemented: 1) livestock exclusion and natural re-vegetation, 2) livestock exclusion, reshaping of the banks with a 3:1 slope, and stabilization using bioengineering techniques, and 3) livestock exclusion, reshaping of the banks with a 3:1 slope, and inset floodplains within the main channel where the channel slope and baseflow channel went unchanged similar to a Priority 4 stream restoration (Rosgen et al., 1997, Thompson et al., 2012). The second restoration method was investigated in this study and was implemented on a 550 m portion of the restoration reach, (Thompson et al., 2012). The floodplain within the restored reach is now inundated 2-3 times per year and is predominantly covered by unmanaged *Phalaris arundinacea* (reed canary grass) (Jones et al., 2015).

2.2.2 Field methods

We conducted a sampling campaign geared toward characterizing the entire floodplain soil profile of the study reach at the StREAM Lab. Sampling cross-sections spaced approximately 75 m apart were selected to measure both (1) depth to buried A horizon and (2) soil flushing potential within the surficial legacy and buried A horizon soils. The toeslope, mid-floodplain, and behind the natural levee were selected as sampling areas within each cross-section. Using eight cross-sections, there were a total of 48 soil cores taken in the floodplain reach (figure 1). Background data from the site in 2014 showed that in the months of September through November, the water table was lowest and the flushable soil nutrients were at a peak due to senescence. The samples for our project were collected between September 22, 2015 and November 12, 2015. A Giddings (Giddings Machine Company, Inc., Windsor, CO) probe deep soil corer was used to take whole, intact soil cores from surface to underlying rock (typically 1-2 m). Cores were stored intact in sealed, air-tight sleeves and field-moist at 4°C until further processing in the lab.

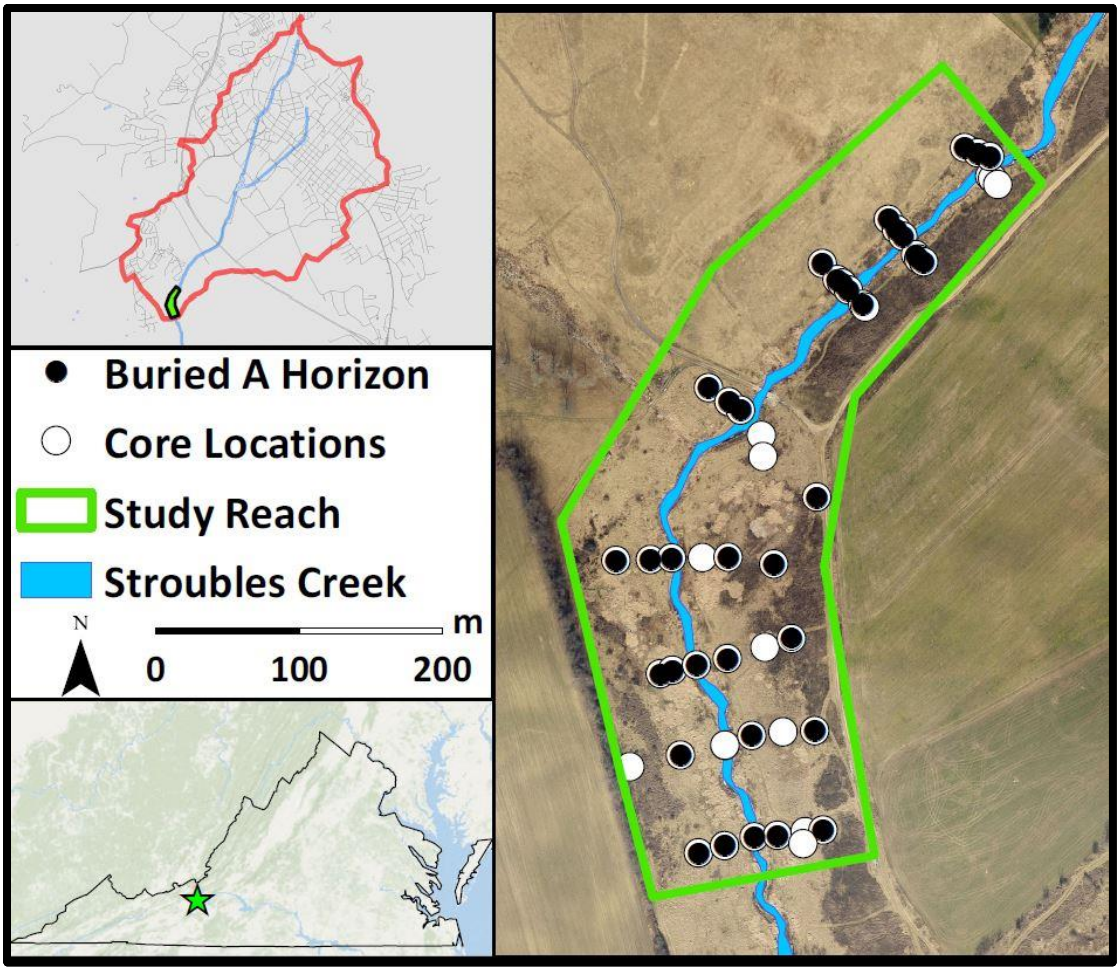

Figure 1. Map of study site at the StREAM Lab, Stroubles Creek, Blacksburg, VA. Location in Virginia is shown with a green star in the bottom left inset. In the top left inset, the map of the contributing drainage area (15 km^2) is outlined in red and the study reach is outlined in green. The main map shows the study area outlined in green and white circles where soil cores were taken at the points along each cross-section number. A core that was positive for a buried A horizon is represented by a black circle. The eight cross sections are numbered from top of the figure (1) to the bottom of the figure (8).

2.2.3 Lab Methods

Soil cores were processed in the lab at room temperature. The soil profile for each core was characterized by recording Munsell soil color, texture via the "feel method," and depth for each horizon present (www.munsell.com; Thein, 1979). When a buried A horizon was present, the entire horizon was taken as a sample, which usually consisted of at least 10 cm of soil. This represented the soil sample for the buried A horizon, which represents the surface an ecological restoration that could interact with flood water. For cores with a buried A present, a corresponding sample was taken from the top 10 cm of the soil core, which corresponded to the legacy sediment sample that has the ability to interact with flood water.

Flushable soil nutrients were measured following the methods outlined in Jones and Willett (2006) for a gentle extraction of dissolved nutrients using deionized water. Soil samples were initially homogenized. In a clean Pyrex 1 liter bottle, 120.0 +/- 0.1 g of each soil sample was added to 600.0 +/- 0.1 mL of deionized water. Deionized water was chosen over stream water for its homogeneity and use in previous studies, although we can assume the resulting nutrient mobilization is slightly higher than would naturally occur (Woodward et al., 2015). The bottles were then placed on a lab-scale shaker table (New Brunswick Scientific Excella™ E24R Temperature-Controlled Benchtop Shaker) for 1 hr at 200 rpm and room temperature. Then, a sterilized pipette was used to siphon off 59.0 +/- 0.1 g of the water from the top of the mixture. This sample was run in a centrifuge (Thermo Scientific™ HIGHConic III Fixed Angle Rotor) for 10 min at 6000 rpm and room temperature. To capture only the dissolved nutrient portion of the sample, the sample was then run through a 0.45 μm Geotech capsule filter using a geopump (Geotech Geopump™ Peristaltic Pump Series II) before being stored in a sterile 20 mL HDPE scintillation vial and frozen for later analysis.

The filtered samples were thawed and brought to room temperature directly prior to lab analysis in the VT-BSE Water Quality Lab. Levels of PO_4^{3-}, NH_4^+, NO_3^- were found with a SEAL nutrient analyzer (SEAL AutoAnalyzer 3 HR) using colorimetry. DOC and TDN levels were analyzed using a high-temperature combustion unit with chemiluminescence detection (Shimadzu TOC-V).

Soil subsamples were also sent to the VT Soils Lab for basic soil chemistry testing. Only 20 subsamples were tested, 10 from the legacy sediment and 10 from the buried A horizon. These were tested for pH via Water pH method, organic matter by method of Weight Loss on Ignition, as well as calcium, magnesium, iron, and plant-extractable P via Mehlich I extraction and analyzed by an ICP (inductively coupled plasma atomic emission spectrometer).

2.2.3 Analysis of Nutrient Data

Extractable nutrient concentrations were converted to a mass per area basis by normalizing to the area of the soil core.

$$\frac{c_{solute} * V_{solvent}}{SA_{sample}} = m_{solute}/SA_{soil}$$

where c_{solute}= concentration of solute (mg/L), $V_{solvent}$=volume of water solvent (120 mL), $SA_{samplel}$= surface area of sample soil core (0.00113 m^2), m_{solute}=mass of nutrient solute (mg), and SA_{soil}= surface area of soil (m^2). This simulated what mass of nutrient extract we would expect to be flushed from that area of the floodplain. Our initial analysis compared the flushable nutrient levels of the legacy sediment sample to the corresponding buried A horizon sample for each core. Because not all data followed a normal distribution, non-parametric tests were used in all analyses. We tested for correlation other than just location between pairs of samples using Spearman's correlation. To test for significant differences between the two layers, a paired, one-sided Wilcoxon Signed Rank test was performed based on EPA (2006) guidelines. One-sided tests were conducted with an alternative hypothesis where the larger mean was assumed to be greater than the smaller mean.

2.2.4 Inundation Analysis

An inundation model following the approach of Jones et al. (2015) was adapted in this study. We used a method of bathymetry interpolation by Merwade (2009) to create a 3D mesh of the entire channel and part of the immediate floodplain using eight surveyed cross sections of the stream topography. LiDAR data (2013 date of collection, 1 m resolution) was used to represent the remaining floodplain surface outside of the surveyed cross sections. The mesh and the LiDAR were welded together to form a complete digital elevation model (DEM) surface of the existing stream-floodplain system. A stream slope raster was developed by using inverse-distance-weighting interpolation of surface-water elevations. We subtracted the stream slope from the DEM which made to normalize the surface to the stream slope while retaining all of the 3D features of the stream corridor.

To compare inundation of the existing surface to one of an "ecological restoration," the surface model was manipulated simulate the removal of legacy sediment down to the buried A horizon. This was done by changing the elevations of the surveyed cross-section data to the elevations of the buried A horizon for each core location. Linear interpolation was used on each cross-section between the core sample locations to complete the topography of the buried A horizon at each cross-section (figure 2) providing a new surface of the "ecological restoration" of the channel and immediate floodplain which was then used to create a new 3D mesh. This was again welded with the surrounding LiDAR data to make the new surface for use in the inundation model. From this, we treated the stream-floodplain surface like the bed of a reservoir where we could model the area of inundation coverage based on the rise in stage in the stream and into the floodplain. A constant stage throughout the study reach was assumed for this model. Conditional raster algebra was utilized to identify inundated areas for a given stage. For both models, stage was varied at increments of 0.1 m and the corresponding inundated area was calculated.

To model nutrient flux from the floodplain during inundation, nutrient mass per area values were interpolated across the floodplain using IDW interpolation. This assumes a common depth of 10 cm (Woodward et al., 2015) interaction for floodwater to pick up flushable nutrients from the surficial soil for either restoration method. The nutrient mass per area values were applied to the area inundated per increase in stage under the different circumstances of peak flow, exceedance event, and annual flow to provide for a predicted total expect mass export.

$$\frac{m_{solute}}{SA_{soil}} x\, A_{inundation} = m_{inundation\ release}$$

where m_{solute} =mass of nutrient solute (mg), and SA_{soil}= surface area of soil (m^2), $A_{inundation}$= inundation area (m^2), and $m_{inundation\ release}$= mass of nutrient released from inundated area. Stage-discharge relationships were developed for each restoration surface using Manning's equation with existing channel characteristics and the area of cross-section 2 (figures A-2 and A-3). Following a method by Ssegane et al (2013), annual exceedances were calculated by using a regression analysis of regional runoff curves from the USGS stream gaining network and watershed attributes from the NHDplus database (Horizon Systems, 2007; USGS, 2014). This resulted in the flow duration curve used to calculate flows for a 1000 year synthetic flow record developed by Jones et al. (2015). From this record, we were able to simulate peak flows from storm events and an expected annual flow record. Using this relationship and the previous inundation model, we were able to estimate the areas inundated for different flow events for which to calculate nutrient mass loadings from.

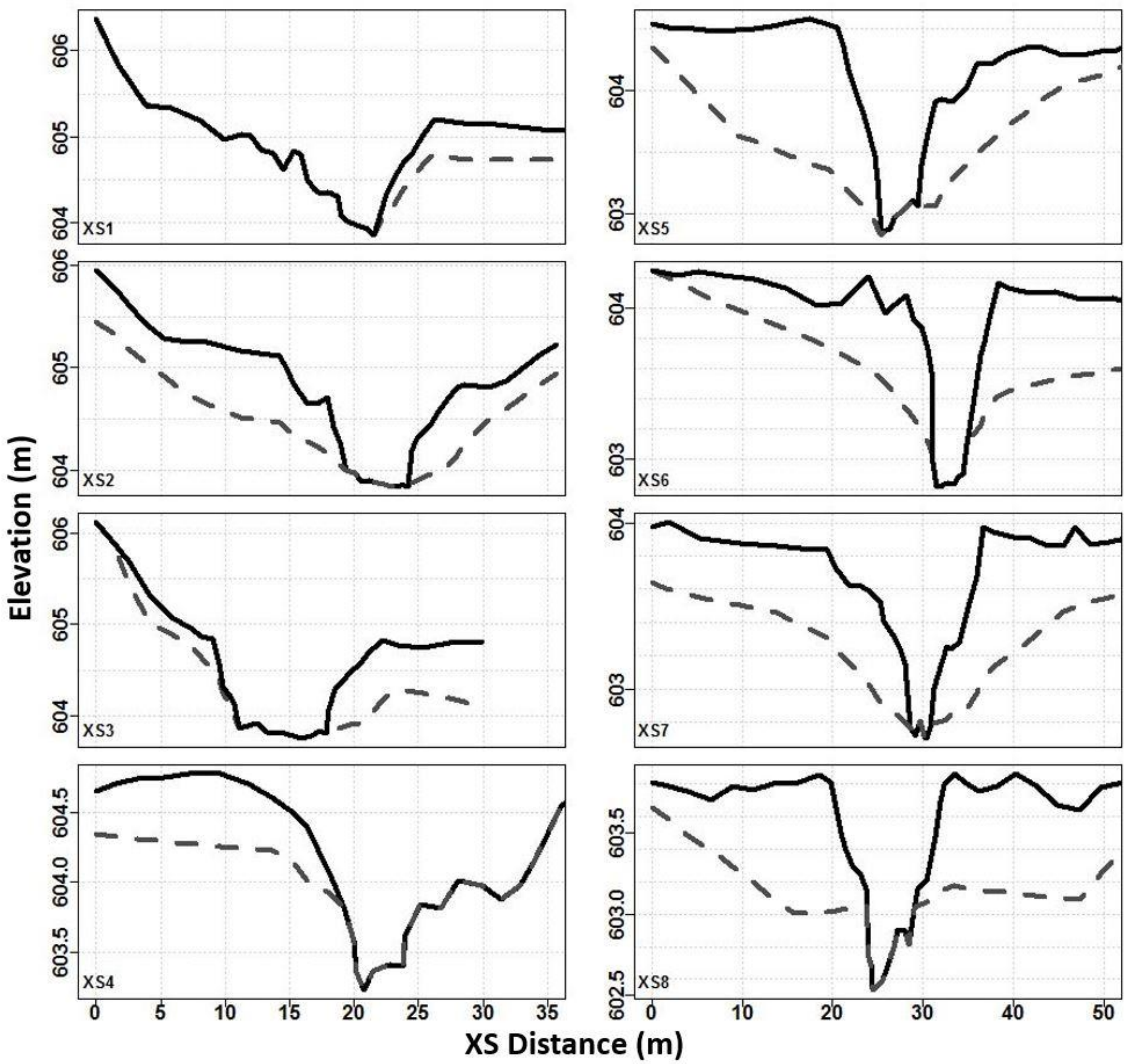

Figure 2. Cross section view of all sampled cross-sections looking downstream at the StREAM Lab, Stroubles Creek, Blacksburg, VA. Elevations are elevations in m above sea level. XS Distance (m) is the distance from the start of the cross section on the left bank. The buried A horizon surface (ecological restoration) is represented by the dashed grey line. The existing surface of legacy sediment (conventional restoration) is represented by the solid black line.

2.3 Results

2.3.1 Soil Characteristics

Out of the 48 soil cores taken, 39 of these had a buried A horizon which resulted in pairs of legacy sediment and buried A samples for 39 locations (Table A-1). Surficial legacy sediment found at the StREAM Lab had a silt loam texture and a Munsell color of 2.5 Y 3/2 (dark brown), which can be common for an upland soil with high organic matter (OM) input deposited in a floodplain. The buried A horizon found in the 39 out of 48 cores taken was generally characterized by a clay texture, a Munsell color of 5 Y 2.5/1 (black), and most often found at depths of 40-55 cm deep. If this soil were at the surface, as we are assuming it once was, this soil color would be indicative of a wetland soil based on the F3 indicator (depleted matrix) from the Regional Supplement for the Eastern Mountains and Piedmont to the U.S. Army Corps of Engineers 1987 Wetland Delineation Manual (USACE, 2010). Other floodplain wetland hydrology indicators found in the buried A horizon included rounded pebbles and mollusk shells. A significant spatial pattern ($p<0.001$) was found occurring across each floodplain cross section, where the average depth to the buried A horizon varied from 0.3 m at the toeslope to 0.6 m behind the natural levee (figure 3a). We can infer that the historic floodplain was a slope wetland, providing a hydrologic connection from the water table surfacing at the toeslope to the baseflow elevation of the stream. However, there was not a significant difference in depth to buried A between cross sections throughout the reach (figure 3b). Because we did not see a pattern of increasing depth to buried A from upstream to downstream, this is more evidence that the legacy sediment buildup was not caused by a downstream mill dam and more of a result from natural factors of floodplain sediment accumulation.

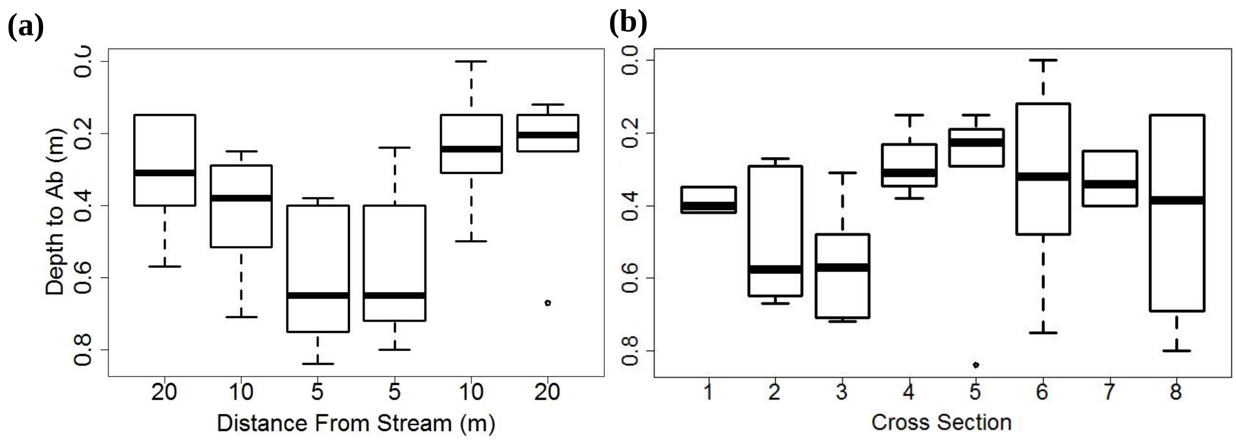

Figure 3. Boxplots showing the range of depths to the buried A horizon at the StREAM Lab, Stroubles Creek, Blacksburg, VA. Boxes and error bars represent the quartiles of observed field data. (a) Plot of the depths to the buried A horizon (m) across the floodplain, where the stream is in the "center" of the graph, with right and left banks on their respective sides. (b) Plot of the depths to the buried A horizon (m) across all floodplain cross sections. Cross section numbers increase with downstream distance through the reach.

The depth to buried A for each soil core was used to map the buried A horizon for that cross-section by using linear interpolation between the depth locations, as shown in figure 2. There were "gaps" in the buried A horizon on some cross sections, which we assume was a relict channel that once formed through the floodplain wetland, but was later filled in with legacy sediment. In these locations, we linearly interpolated from the positive buried A locations on either side of the "gap". These eight cross-sections with new elevation data provided for the new surface of what we termed the "ecological restoration" by effectively removing the legacy sediment to expose the buried A horizon to the surface. Because the existing stream restoration disturbed the natural banks by grading them back, soil cores were not taken within the natural levee or the streambank. Here we assumed that the buried A horizon would connect in to the historical baseflow elevation.

Fe, Ca, and Mg are well-known mineral complexes that sorb PO_4^{3-} (Ardon et al., 2010). As seen in table 1, similar levels of Ca and Mg were found in the two layers of floodplain soil. Table 1 shows that there was a significantly higher level of Fe in the legacy soil suggesting it can retain PO_4^{3-} under drier conditions, but also serve as a mechanism for release of PO_4^{3-} during longer periods of inundation under Fe-reducing conditions (Ardon et al., 2010). We also saw significantly higher OM levels in the legacy soil, which is unexpected given that the buried A horizon was once a wetland that retained high amounts of OM. The percent OM results were slightly compromised by the lab technique that tends to burn off Fe with the OM, which would explain similar patterns between OM and Fe in the two soil layers. This would explain why the OM levels are slightly higher than natural levels expected between 2-4 percent. We also expected OM levels at the surface to be high from the reed canary grass that tends to add nutrients and OM to the soil at a higher rate (Ehrenfield, 2003)

Table 1. Soil test results for legacy sediment and buried A horizon at the StREAM Lab, Stroubles Creek, VA. The mean value for 20 locations is given and its corresponding standard error in given in parentheses. The p-value is given for the test for significance ($\alpha<0.01$) using a paired Wilcoxon rank-sum test (where H_o: LS≤Ab).

Analyte	Legacy	Buried A	p
pH	7.2 (0.1)	7.6 (0.03)	0.065
P (ppm)	7.2 (0.3)	4.6 (0.2)	0.015
Ca (ppm)	2473 (59)	2353 (31)	0.298
Mg (ppm)	521 (16)	502 (16)	0.325
Fe (ppm)	4.8 (0.3)	2.70 (0.3)	0.018
OM (%)	7.2 (0.2)	4.1 (1.0)	0.001

2.3.2 Flushable Nutrients

Flushable nutrients represent the potential for soils to release nutrients upon inundation, assuming a depth of interaction of 10 cm into the subsurface. This assumption is an approximation of the depth that floodwater from the stream will mix with the soil in the time that it inundates the floodplain surface. In order to test for significance, the correlation between the two layers was first found to be significant for each analyte and ranged from weak (DOC: rho=0.119) to strong correlations (NH_4^+: rho=0.707) (table 2). Using the Wilcoxon test for significance for each analyte, we saw significant differences ($p<0.01$) between the legacy sediment and the buried A horizon. Our result show the legacy sediment contained more flushable SRP, NO_3^-, NH_4^+, TDN, and DOC than the buried A horizon. This confirms that the legacy sediment left on the floodplain during the conventional restoration is indeed higher in flushable nutrients and can likely pose a long-term water quality problem. In order to compare the potential release between the 2 restoration approaches, the frequency and area of inundation are required.

Table 2. Flushable nutrient test results for legacy sediment and buried A horizon at the StREAM Lab, Stroubles Creek, VA. The mean value for all locations is given and its corresponding standard error is given in parentheses. The p-values are given for the test for significance (α<0.01) using a paired Wilcoxon rank-sum test (where H_o: LS≤Ab). The rho values are results from a Spearman correlation test (0=no correlation, 1=true correlation).

Analyte	Legacy	Buried A	p	rho
SRP(mg/m^2)	9.7 (0.2)	6.0 (0.1)	<0.001	0.396
NO_3^-(mg N/m^2)	433 (8)	105 (1)	<0.001	0.402
NH_4^+(mg N/m^2)	12.9 (0.3)	9.2 (0.3)	<0.01	0.707
TDN(mg/m^2)	524 (8)	158 (2)	<0.001	0.284
DOC(mg/m^2)	980 (12)	557 (9)	<0.001	0.119

2.3.3 Inundation Area

As expected, the inundation model showed that the ecological restoration has more inundation area than the conventional restoration at each stage elevation as shown in figures 4a and 4b. Figure 5a shows this relationship across a range of typical stage values. On average, there was a 130% increase in inundation area for a given stage going from the conventional restoration to the ecological restoration. Stroubles Creek is a “flashy” stream because of its urban headwaters, therefore true differences in inundation area need to be normalized to the expected frequency of inundation based on the flow regime. On average, there was a 500% increase in inundation area across the range of annual exceedances (figure 3b). The ecological restoration has more inundation area than the conventional restoration for all storm events.

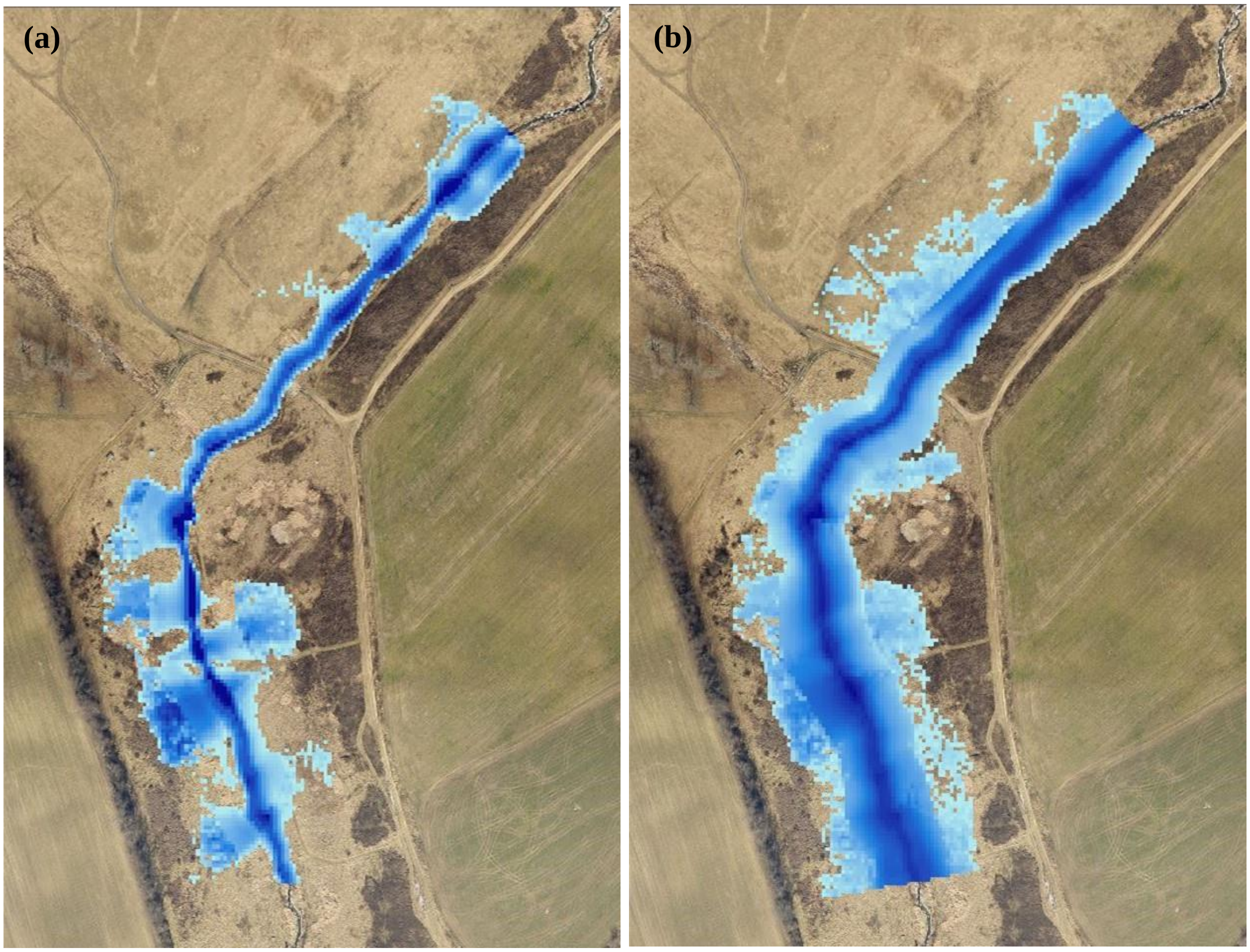

Figure 4. Inundation model for StREAM Lab, Stroubles Creek, VA. Modeled inundation for conventional (a) and ecological restoration (b) at 1.5 m stage.

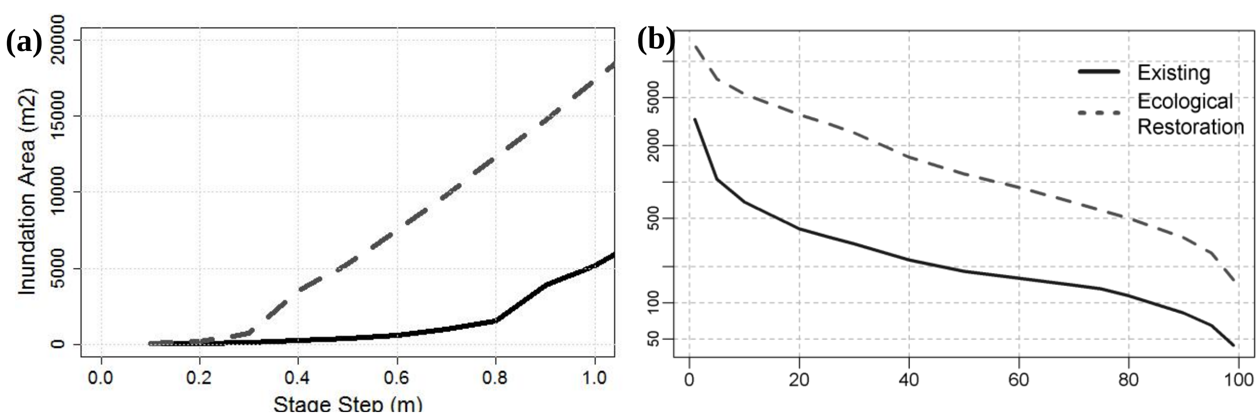

Figure 5. Inundation area relationships for StREAM Lab, Stroubles Creek, VA: (a) Inundation area (m^2) vs stage (m); (b) Inundation area (m^2) vs probability of annual exceedance of a given flow rate. Ecological restoration is represented by a dashed gray line. Conventional restoration is represented by a solid black line.

2.3.4 Nutrient Export Potential

To get an estimate of potential nutrient export from the floodplain, we need to look at the nutrient flushing and inundation patterns across different flow events. Figures 6a and 6b shows the same inundation area as figures 4a and 4b with the nutrient mass per area values interpolated across the surface. Figure 7 shows what nutrient export we would expect out of the two restorations for a range of peak flows. During most peak flows, there is a higher export of SRP, $NH4^+$, DOC, NO_3^-, and TDN flushing in the ecological restoration than the conventional restoration. This is likely because the stream can easily overtop the banks in the ecological restoration, resulting in more inundated area and more available nutrients to be flushed. At the same time, because the stream is barely above bankfull in a conventional restoration, there is little opportunity for the floodwater to reach the nutrients in the legacy sediment built up on the floodplain. The breakpoint can be seen in both restorations for all nutrients where the flows overtop the streambank, suggesting that the floodplain of both restorations is a higher source of nutrients. Looking back at Table 2, we see that the NO_3^- and TDN levels in the legacy sediment are three to four times higher than the buried A horizon, but the difference in the levels of SRP, NH_4^+, and DOC are much smaller (1.3-1.8 times the buried A layer). This explains why, for NO_3^- and TDN, we see the conventional restoration have smaller differences in export than the ecological restoration at all peak flows than the trends for SRP, NH_4^+, and DOC.

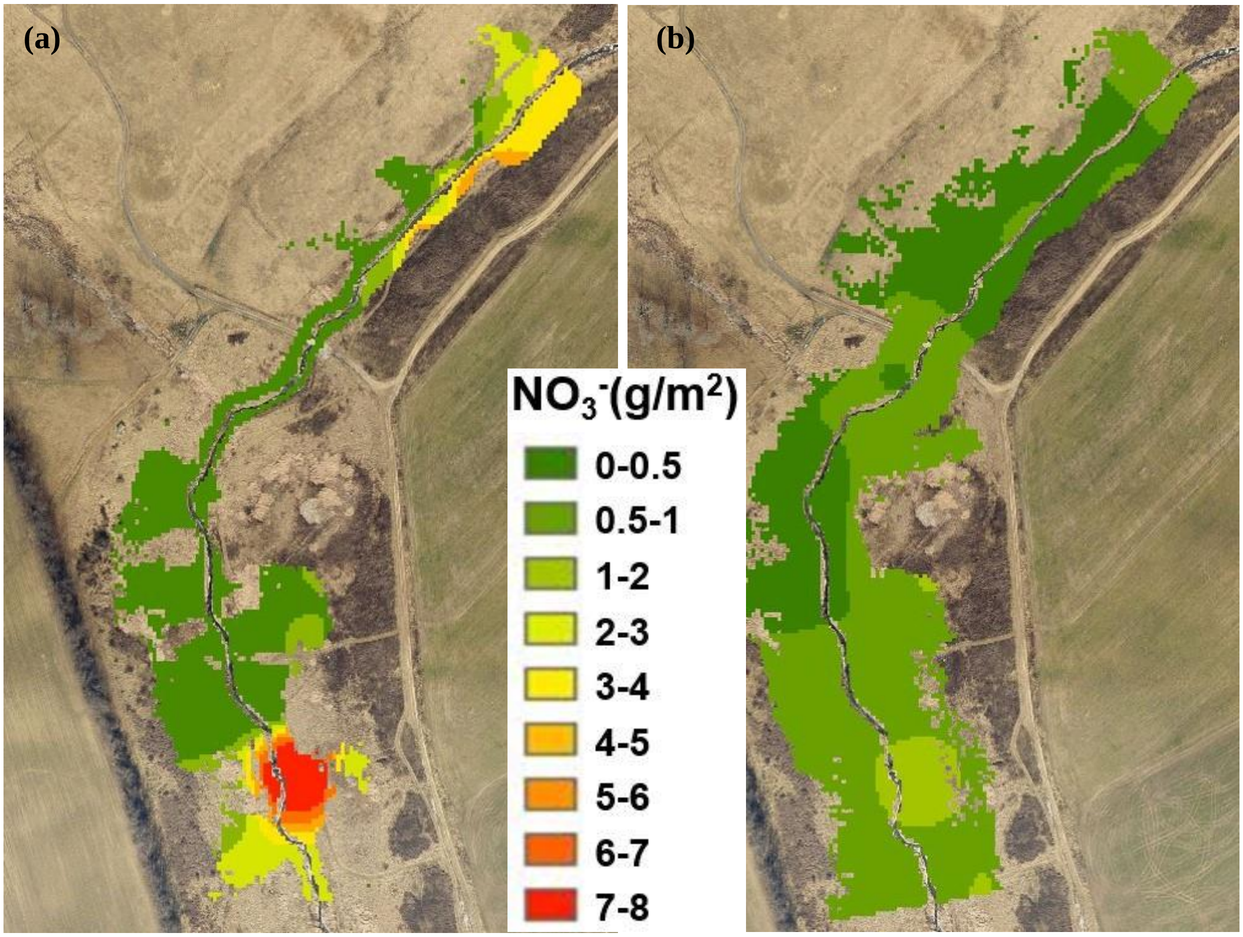

Figure 6. Nitrate export model for the StREAM Lab, Stroubles Creek, VA. Expected flushable nitrate levels (g m^{-2}) are given for the conventional restoration (a) and the ecological restoration (b) for 1.5 m of stage.

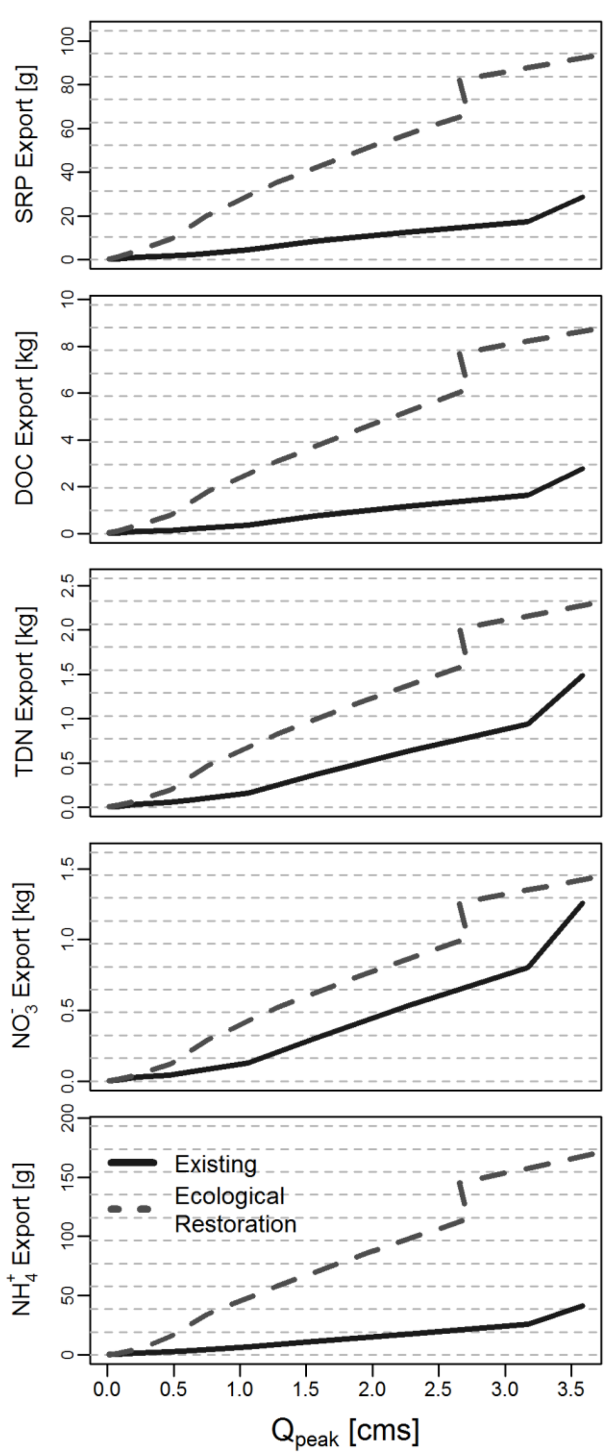

Figure 7. Nutrient export across peak flows at StREAM Lab, Stroubles Creek, VA. Peak flow ($m^3\ s^{-1}$) vs nutrient export (g or kg) during inundation. Existing surface is the conventional restoration represented by a solid black line. Buried A surface is the ecological restoration represented by a dashed gray line.

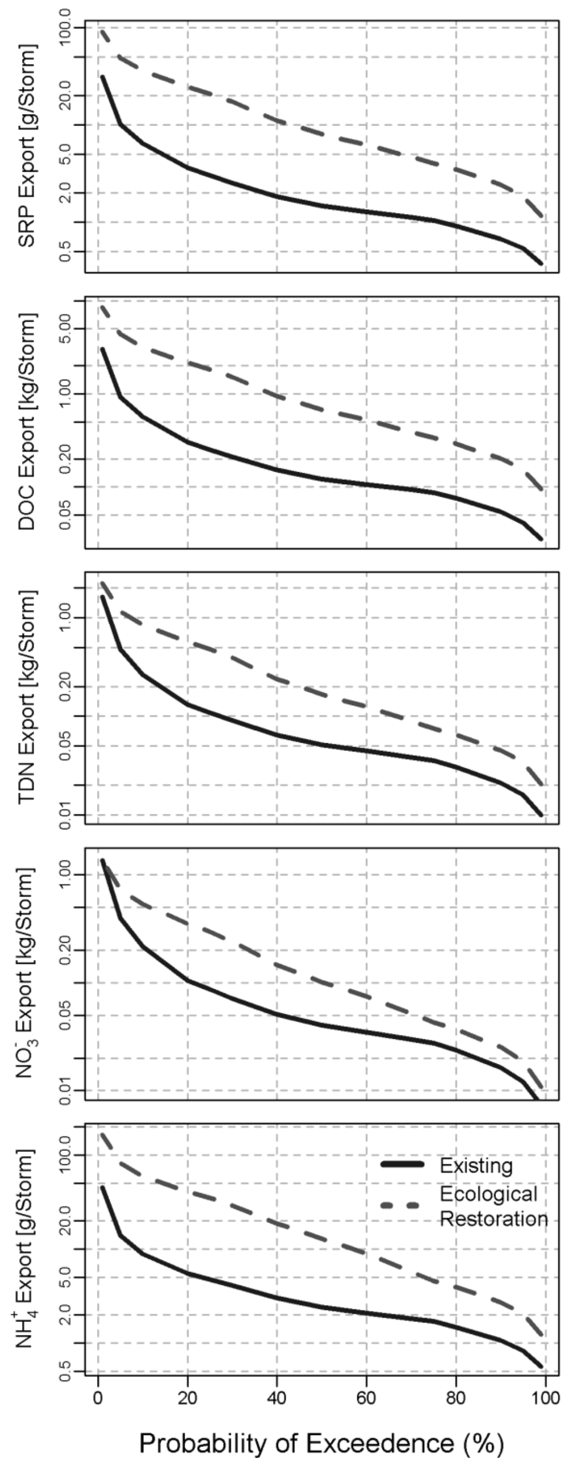

Figure 8. Nutrient export per storm event at StREAM Lab, Stoubles Creek, VA. Exceedance (%) vs nutrient export per storm (g/storm or kg/storm). Existing surface is the conventional restoration represented by a solid black line. The buried A surface is the ecological restoration represented by a dashed gray line.

When we took the nutrient export per area for each flowrate and applied it to our synthetic flow record, we saw that the ecological restoration was also expected to export more flushable nutrients over the entire range of exceedance events (figure 7). The increased time of inundation in the ecological restoration allows for more flushing of the floodplain soil. Even at smaller events near 100% exceedance, the ecological restoration allows floodwater to spread out more to come in contact with available nutrients in the floodplain soil. The conventional restoration begins to export more nutrients for the most extreme events with low probability of exceedance likely because the legacy soils that are less frequently inundated are acting as a large storage of flushable nutrients.

The picture becomes even clearer when we look at estimated export of nutrients on the annual scale (figure 8). Again, the ecological restoration is higher in nutrient export across all analytes. This analysis assumes biogeochemical stationarity or that the flushable nutrients are available for flushing at the same levels and are not depleted following events (Basu et al., 2010). It is worthy to note that these export rates are what we expect when the ecological restoration is first completed, and that the nutrients are not depleted immediately. The annual rate of NO_3^- export is much smaller than we expected based on a seasonal study of experimental floods in a slough of the same floodplain by Jones et al. (2015) where they saw a NO_3^- removal of 0 to 139 kg N/yr. This means that during certain times of the year, we can expect the NO_3^- removal to offset the NO_3^- flush we see from either restoration approach.

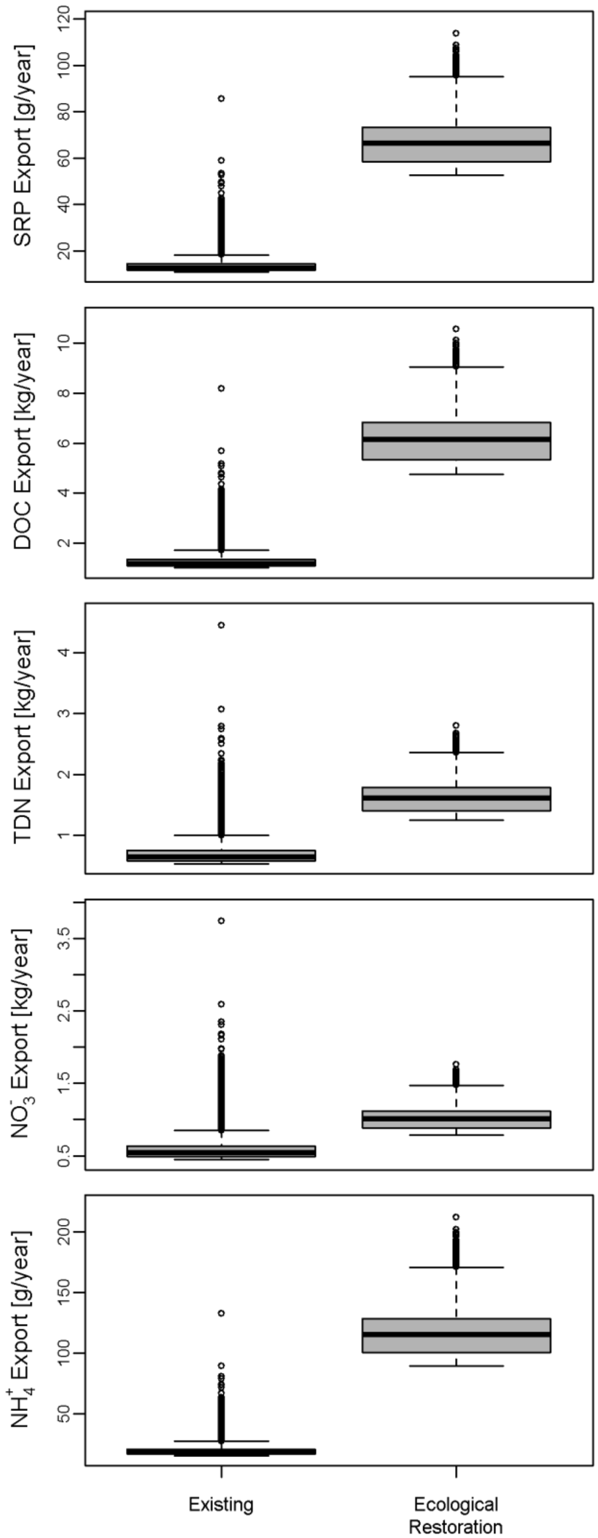

Figure 9. Annual nutrient export for StREAM Lab, Stroubles Creek, VA. Nutrient export per year (g yr^{-1}) by restoration method, where existing is the conventional restoration. Boxplots show the quartiles and outliers for 1000 years in the synthetic flow record.

2.4 Discussion

Our sampling campaign took place during the fall senescence period which occurs at a peak in the nutrient cycle. Thus, to model nutrient flushing temporally, we had to assume this rate of export per area stays constant throughout the year and through different storm events and flow rates. This likely resulted in the model predicting higher exports than what is naturally occurring. We also recognize these rates can very due to extraneous factors such as vegetation, antecedent moisture, etc. and thus assume the reservoir of soil nutrients is unable to deplete over time.

Stream restoration is a young science compared to forest or soil restoration, thus restoration strategies are still controversial (Palmer et al., 2014b). Restoration goals have shifted from recovering the natural ecosystem state to one of restoring desirable ecosystem services that can benefit humans (Palmer et al., 2014a).

We found a thick deposit of legacy sediment over a buried A horizon from a relict hydric soil throughout the floodplain of Stroubles Creek. We assume this legacy sediment accumulated in the floodplain over the last 270 years and was independent of any mill dam effects. The buried A horizon had significantly lower flushable nutrients per area than the surficial legacy sediment. When concentrations were converted to an area-weighted basis, we saw comparable numbers to Woodward et al. (2015) where they had 185 to 991mg N-NO_3^-/m^{-2} for and 12 to 148 mg P m^{-2}.

Removing the legacy soil from the floodplain to expose the buried A horizon, what we call an ecological restoration, can provide for more inundation area compared to the existing surface from a conventional restoration. We saw a 130% average increase in inundation area over all representative stream stages, and a 500% average increase in inundation area when modeled using exceedance intervals. This can provide hydraulic benefits such as more floodplain storage and roughness that would reduce downstream erosion.

Because of the increase in inundation area and frequency with an ecological restoration, our model predicts higher exports of flushable nutrients at the event and annual scales. We saw an average of 300% increase in SRP and NO_3^- export over all flowrates from the conventional restoration to the ecological restoration. On an annual scale, this equates to a 100% increase in NO_3^- and a 700% increase in SRP. This shows that by doing an ecological restoration, this floodplain will initially export more nutrients than its current restoration state due to the increased hydrologic connectivity. We do know from Jones et al. (2015) that the annual removal rate of the existing floodplain is about 0.5 to 1.5% of the stream load. Actual export and removal rates would need to be measured upon doing such a restoration in order to see what effects the natural processes would have on the buried hydric soil is restored to the surface.

Although our analysis shows nutrient flushing will be higher for the ecological restoration, there are a variety of positive benefits that further offset nutrient flushing. Restoring the buried hydric soil would reestablish native wetland plants in the floodplain (O'Donnell, 2014) and provide more wetland habitat (Ecological Restoration, 2009) leading to more biodiversity in the floodplain. Restoring wetlands has proven to provide an increase in biogeochemical processes such as nutrient removal (Jones et al., 2015; Scott et al., 2014; Filoso and Palmer, 2011) that could offset the nutrient flushing seen from the buried hydric soil. Weitzman et al. (2014) showed that leaving a hydric soil buried under legacy sediment still leads to more nutrient export than uptake, and Koval (2012) saw lower denitrification rates in the buried hydric soil than a wetland that is restored.

By removing erodible legacy sediment from the floodplain, the streambank erosion and floodplain sediment resuspension could be greatly reduced as a factor for the sediment impairment problems in the adjacent stream channel. This would allow for more sedimentation to occur on the floodplain, a benefit for the downstream waters. This also has its drawbacks. Performing stream restoration in order to encourage sediment retention will result in a finite lifespan of the project (Palmer et al, 2014a). Without restoring the natural hydrology of the watershed and thus the natural land cover, an impossible task, obvious maintenance issues arise with this design in order to keep the sediment accumulation from burying the restored hydric soil again. This practice should be coupled with an extensive effort at implementing upstream BMPs to slow runoff rates to predevelopment levels, keep existing upland soil intact, and reduce nutrient and sediment loading to streams. Economic incentives and environmental regulations have started to address these key watershed components, as well as the practice of stream restoration, but more effort needs to be put into assessing these practices once implemented and reporting the results so that sound science can back the decisions we make as engineers (Palmer et al., 2014b).

2.5 Conclusions

The buried A horizon at Stroubles Creek has a significantly less amounts of flushable nutrients than the legacy sediment above it. An ecological restoration of reconnecting the stream with its relict wetland floodplain can lead to an increase in initial flushable nutrient export. At the annual scale, NO_3^- loading may be offset by floodplain removal processes but still may lead to an export of other reactive nutrients (SRP, NH4, and DOC). At the event scale, the ecological restoration can be expected to release more flushable nutrients for most storms. The added ecological benefits may outweigh the flushable nutrient exports we measured, but we realize an ecological restoration needs to be implemented on the stream to evaluate the biogeochemical effects of reconnecting Stroubles Creek to its relict floodplain wetland.

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2.8 Appendices

A.1: Table of Buried A Soil Core Locations at StREAM Lab. XS is cross section number corresponding to figure 1. FL is the approximate floodplain location where A= river-right (RR) toeslope, B= RR mid-floodplain, C= RR behind natural levee, D= river-left (RL) behind natural levee, E= RL mid-floodplain, and F= RL toeslope. Latitude, longitude, and elevation (above sea level) correspond to the VA State Plane South coordinate system in m.

XS	FL	Longitude(m)	Latitude(m)	Elevation(m)
1	A	3327305.113	1099075.133	605.075
1	B	3327309.229	1099071.029	605.115
1	C	3327313.589	1099066.132	605.206
2	A	3327252.929	1099029.41	604.257119
2	B	3327258.4	1099022.393	604.31252
2	C	3327262.853	1099017.797	604.029119
2	D	3327271.936	1099004.926	604.286719
2	E	3327274.183	1099001.863	604.290819
2	F	3327277.423	1098999.766	604.394019
3	A	3327206.694	1098998.616	603.892619
3	B	3327217.063	1098987.599	603.911819
3	C	3327220.51	1098984.91	603.91972
3	D	3327224.887	1098980.266	603.968619
3	E	3327234.991	1098970.133	604.008619
4	A	3327127.184	1098912.016	603.663319
4	B	3327141.397	1098902.242	603.640819
4	C	3327149.681	1098896.165	603.837019
4	F	3327202.844	1098835.526	604.010519
5	A	3327062.603	1098791.279	603.581719
5	B	3327086.6	1098791.976	603.443419
5	C	3327101.446	1098792.861	603.369819
5	E	3327140.964	1098793.553	603.630819
5	F	3327172.788	1098788.871	603.592019
6	A	3327093.576	1098711.98	603.026819
6	B	3327101.983	1098714.453	603.104019
6	C	3327118.786	1098718.223	603.202119
6	D	3327140.214	1098722.763	603.260319
6	E	3327166.585	1098730.669	603.340219
6	F	3327185.038	1098736.724	603.547019
7	B	3327107.634	1098655.222	607.481319
7	D	3327157.258	1098668.83	607.31392
7	F	3327201.351	1098672.303	607.51032
8	A	3327120.091	1098586.799	603.016419
8	B	3327138.227	1098591.959	602.956419
8	C	3327158.968	1098598.032	602.916619
8	D	3327175.266	1098599.339	602.941019
8	E	3327194.697	1098601.679	602.836219
8	F	3327207.355	1098602.889	603.174319

A.2 Example Cross Section.

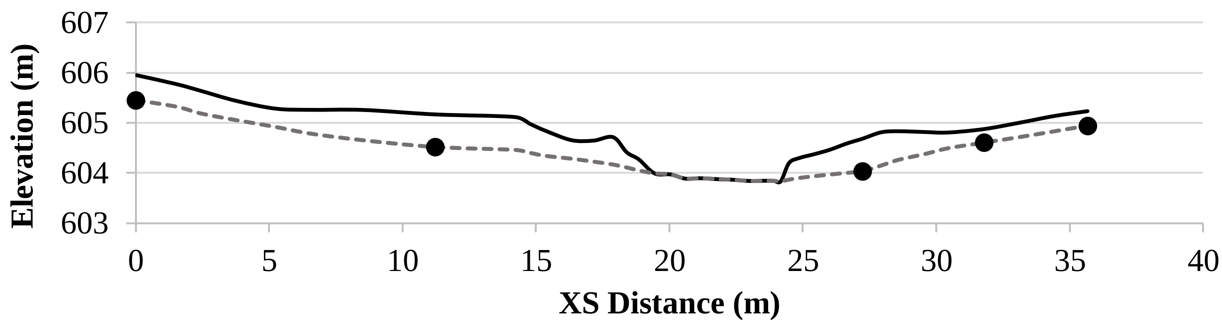

Figure A.2. Cross section 2 at StREAM Lab, Stroubles Creek, VA. Cross section view of cross section 2 looking downstream. Elevations are elevations in m above sea level. XS distance is from the start of the XS on left bank. The buried A horizon surface (ecological restoration) is represented by the dashed grey line. The existing surface (conventional restoration) is represented by black line.

A.3 Stage-Discharge Relationship

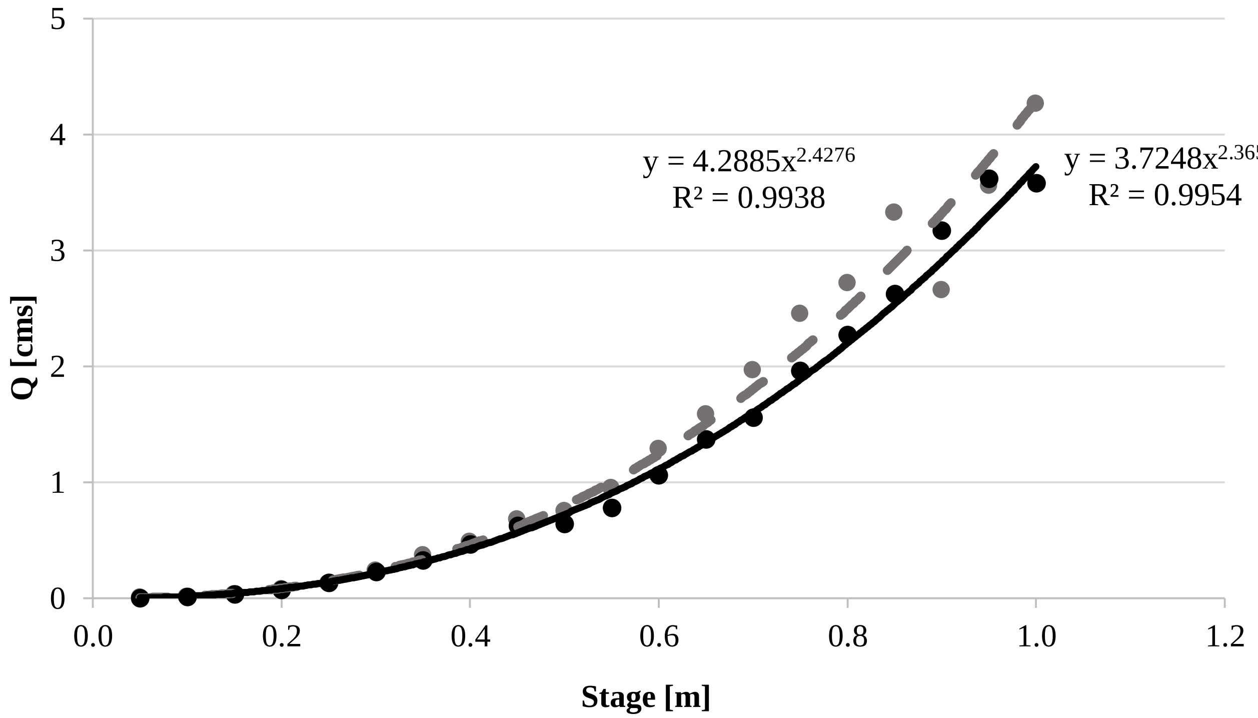

Figure A.3. Stage-Discharge relationship developed for the StREAM Lab, Stroubles Creek, VA. The relationship of stage (m) vs discharge(q, m^3s^{-1}) for the two restoration surfaces using cross-section 2. Ecological restoration surface is represented by grey points and the power equation of best fit by a grey dashed line. Conventional restoration surface is represented by black points and the power equation of best fit by a solid black line.

3 Conclusions and Future Work

3.1 Directions for Future Research

Despite nutrients being a huge concern in receiving waters, the sediment impairment on Stroubles Creek could be further addressed by the removal of legacy sediment within the floodplain. More research would need to be done to understand how the restored wetland floodplain would handle the new flow regime of an urban watershed with its increased nutrient and sediment loads. The relict floodplain wetland did not assimilate the abrupt changes in flow regime and sediment load seen with the accelerated land use change in the area hundreds of years ago. With more BMPs in place and the easier erodible soil already missing from the uplands, the restored wetland floodplain might be better suited for human impacts.

The flushable nutrient analytes chosen for this project only capture one piece of the biogeochemical puzzle that is river-floodplain exchange. Dissolved organic nitrogen would be another analyte of interest for its biologically reactive properties. Total nitrogen and total phosphorus are common nutrient measures used in the engineering world to assess nutrient removal projects such as stream restorations. TN and TP were measure for a portion of the floodplain but were excluded based on the minimal dataset that we had. Having that full dataset would be one way to make this project more applicable to the engineering and regulatory realm, as well as getting a better picture of the nutrient processes going on in both legacy and buried hydric soils.

Dating of floodplain sediments has been a useful way to prove when legacy sediment began to accumulate on the floodplain and at what rate (Walter and Merritts, 2008). More samples would need to be taken at different depths in order to estimate the age of the layers of legacy sediment and the underlying buried A horizon.

We also wish we could see more of the buried A horizon at the StREAM lab. One way to get a clearer picture of what the buried A horizon looks like across the floodplain is to excavate a trench perpendicular to the flow of the stream. We recommend this be done in the area on the river-left bank between cross sections.

To address the broader picture of this research, we recommend a study be done on the total sediment and nutrient budget of the study reach to find out what is in the stream, what is happening on the existing floodplain, what is the net downstream effect, and then ultimately, perform an ecological restoration on part of the study reach and compare the differences between the two restoration methods.