Urban Stream Restorations Increase Floodplain Soil Carbon and Nutrient Retention along a Chronosequence

A Thesis submitted in partial fulfillment of the requirements for the degree of Master of Science at George Mason University

by

Katrina Napora
Bachelor of Science
Christopher Newport University, 2020

Director: Changwoo Ahn, Professor Wetlands Ecosystems
Environmental Science & Policy Department

Spring Semester 2023
George Mason University
Fairfax, VA

DEDICATION

I dedicate this work to my parents, Gail and Nick Napora, for letting me play in the mud at three years old in my pretty dresses, and hosed me off on the front porch.

ACKNOWLEDGEMENTS

I wish to thank my advisor, Dr. Changwoo Ahn, for his guidance and support throughout this process. I also want to thank the members of my committee for their guidance and wisdom: Dr. Gregory Noe and Dr. Youngsung Kim, as well as Meghan Fellows for her significant contribution to this project. I would like to acknowledge the Fairfax County Department of Public Works, Stormwater Planning Division, Fairfax County Park Authority, the National Park Service, the Truro Subdivision, the Reston Association, and the Poplar Heights Recreation Association for access to study sites. This work was supported by USGS Chesapeake Bay Activities. We thank Thomas Doody and Molly Huber for their help in conducting the research and improving the thesis.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATIONS AND SYMBOLS

C .... Carbon
N .... Nitrogen
P .... Phosphorus
TMDL .... Total Maximum Daily Load
Al .... Aluminum
Fe .... Iron
PO_4^{3-} .... ortho-Phosphate
TC .... Total Carbon
TN .... Total Nitrogen
TP .... Total Phosphorus
CO_2 .... Carbon Dioxide
EPC_0 .... Equillibrium Phosphorus Concentration
DPS .... Degree of Phosphorus Saturation
D50 .... 50^{th} percentile of particle size
ISC .... Impervious Surface Cover
OM .... Organic Matter
L .... Liter
cm .... Centimeter
°C .... Degrees Celsius
LOI .... Loss on Ignition
M3 .... Mehlich-3
ICP-OES .... Inductively Couple Plasma- Optical Emission Spectroscopy
ANOVA .... Analysis of Variance
GLM .... Generalized Linear Model
AICc .... corrected Akaike's information criterion

ABSTRACT

URBAN STREAM RESTORATIONS INCREASE FLOODPLAIN SOIL CARBON AND NUTRIENT RETENTION ALONG A CHRONOSEQUENCE

Katrina Napora, M.S.

George Mason University, 2023

Thesis Director: Dr. Changwoo Ahn

Stream restoration is a common management practice to meet regulatory or voluntary efforts to improve water quality via carbon and nutrient retention, including in the Chesapeake Bay watershed. However, restoration projects have few quantifiable measures of project success, no standard metrics, and rarely collect pre-restoration data. Storage of nutrients, such as phosphorus (P), and carbon (C), in floodplain soils of restored streams can act as an easily quantifiable indicator of restoration success, particularly when the project goals include improved water quality. To determine how floodplains of restored streams change in their phosphorus and carbon storage as time since restoration increases, floodplain surficial soil samples (10 cm depth) were collected from 18 streams in the urbanized Piedmont region of northern Virginia, representing a chronosequence of time (1-10+ yrs.) since restoration as well as unrestored and reference streams. The samples were analyzed for total carbon (TC), total nitrogen (TN) and total

phosphorus (TP) storage, whereas CO2 mineralization potential and equilibrium phosphorus concentration (EPC_0) were measured as metrics of nutrient and carbon loss. These metrics were compared to time since restoration and potential environmental drivers, including soil moisture, pH, particle size, organic matter content, and degree of phosphorus saturation. These stream restorations demonstrated increasing nutrient storage for TC, TN, and TP along the chronosequence to values greater than both unrestored or reference streams, as well as decreasing C mineralization turnover and no significant changes in EPC_0. Soil wetness and organic matter, key drivers in nutrient retention, also increased as restoration projects aged increasing nutrient and C storage. Overall, stream restoration did improve carbon and nutrient retention in floodplains as compared to unrestored sites and exceeded those of low urbanization ‘reference’ sites.

MANUSCRIPT ONE

Introduction

Nutrient enrichment of stream systems is linked to the proliferation of harmful algal blooms, hypoxic or anoxic dead zones, and fish kills in downstream aquatic and estuarine environments (Kiedrzyńska et al., 2008). Streams and their floodplains have the potential to either store or transport excess quantities of carbon and nutrients downstream (Correll, 1998). Natural floodplains are among critical biologically productive ecosystems, and their degradation is linked with the rapid decline in freshwater biodiversity (Tockner and Stanford, 2002). Stream restoration and the reconnection of urban streams to their floodplains has become a key management technique to improve several ecosystem services, including protection of human infrastructure, habitat improvement, and water quality (Bernhardt and Palmer, 2007; Palmer et al., 2014; Briggs and Osterkamp, 2021). In the Chesapeake Bay watershed, stream restoration has become increasingly used as a tool to improve water quality and meet total maximum daily load (TMDL) goals that place limits on the amount of sediment, N, and P transported downstream (Thompson et al., 2018).

In the past few years, stream restoration projects have focused on increased channel complexity and increased lateral hydrological connectivity of the floodplain, and reestablishment of riparian vegetation to target biogeochemical cycling (Berg et al., 2014). Channel complexity along a river or stream increases the distribution and concentration of organic carbon, as there are increased residence times of water and sediment (Sutfin et al., 2016). The newer restoration practice of increasing lateral hydrologic connectivity of floodplains with the channel increases flooding (Berg et al., 2014), which creates anaerobic conditions which can slow the rate of decomposition and influences soil chemistry through reduction-oxidation (redox) processes (Wolf et al., 2013). The enhancement or reestablishment of riparian vegetation directly influences carbon development through both aboveground biomass on the floodplain and within the soil through below-ground biomass and litterfall (Giese et al., 2000). Soil carbon content increased with time since restoration in South Carolina forested floodplains (Wigginton et al., 2000). However, since floodplains are not permanently inundated, their soils go through cycles of oxidation and reduction (Baldwin & Mitchell, 2000). Oscillating periods of inundation and drying create several pulses of stress on the microbial and plant community, which can increase the rates of carbon mineralization in reconnected floodplains (Yin et al., 2019). The floodplain restoration that incorporates hydrologic connectivity could increase soil wetness and thereby increase C storage, but the balance of C inputs and C loss is uncertain.

Hydrologic connectivity also enhances the delivery of stream loads to floodplains to allow for the retention of flood waters during high-flow events and promotes nutrient retention on the floodplain through particle deposition and soil denitrification, which decreases storage but improves overall water quality (McMillan & Noe, 2017). Wolf et al. (2013) found that sedimentation during high-flow events increased floodplain soil nitrification and was associated with greater denitrification. Likewise, dissolved orthophosphate (PO_4^{3-}) can bind to the floodplain soil during flooding events, allowing P to accumulate in the soil over time (Kronvang et al, 2006). These flooding events impact both aluminum (Al) and iron (Fe), which adsorb phosphates via covalent bonds upon the metals' surface (Walbridge & Struthers, 1993). Iron phosphates are sensitive to reduction during soil saturation, as Fe^{3+} is converted into Fe^{2+} and releases the associated phosphates (Richardson, 1985; Walbridge and Struthers, 1993). Likewise, Al has been measured to decline during seasonal flooding (Darke & Walbridge, 1999). When P accumulates within the floodplain, less is transferred downstream, mitigating the available P for algal blooms (Kiedrzyńska et al., 2008). The degree of phosphorus saturation (DPS) is often used to determine the amount of phosphorus present in the soil relative to the available binding sites, which informs the sorption potential and subsequent P retention capacity of the soil (Kleinman & Sharpley, 2002; Inamdar et al., 2020).

Likewise, P sorption depends on the relative concentrations of P in the stream water compared to the available binding sites in the soil. In degraded urban streams of Maryland (Inamdar et al., 2020), contact with lower concentrations of dissolved orthophosphate in the channel allowed the leaching of excess P in nutrient-rich legacy sediments into stream waters. Rather than sequester P as intended, higher orthophosphate concentrations in floodplain soils combined with reduced runoff can cause soils exposed by stream incision to release orthophosphate back into the stream. Equilibrium phosphorus concentration (EPC_0) represents the concentration at which there is no net sorption or desorption of P from the soil (Haggard & Sharpley, 2007; Inamdar et al., 2020). If the value of sediment EPC_0 exceeds that of the stream water phosphate, P will desorb from soil to water to be carried downstream. Conversely, if the value of EPC_0 is less than stream water phosphate concentration, then the soil will sorb and remove P from stream water. Enhanced nutrient mobility has been reported following the rewetting of some floodplain sediments over the timescale of 20 years (Surridge et al., 2012). However, other studies (Wolf et al., 2013; Hopkins et al., 2018; Noe et al., 2019; Gordon et al., 2020) have found that restored floodplains promote P deposition. A variety of environmental factors attribute to differences in P and C behavior, and many of these variables all have the potential to be influenced by restoration.

Over time, these targeted carbon and nutrient dynamics in restored stream sites change. When restoration is completed, there is the potential for biogeochemical time lags, delaying ecosystem services such as nutrient retention (Hamilton, 2012). In addition, restorations may follow a smooth trajectory of increasing ecosystem function that will eventually approach reference ecosystems (Orzetti et al., 2010) or may stagnate and fail to reach restoration targets (Zedler & Callaway, 1999; Violin et al., 2019) or fall somewhere in between over the course of the many target metrics. In worst-case scenarios, ecosystem function could degrade as the stream restoration project age even as they pass visual inspection (Hill et al., 2013), or is unable to handle increased pollutant loads in larger-volume storms due to climate change (William et al., 2017), or upstream land-use conversion.

Based on the 2005 National River Restoration Scientific Synthesis, an estimated 1 billion U.S. dollars are spent annually on stream restoration in the United States (Bernhardt et al., 2005); However, despite attempts to condense smaller-scale or partial databases of stream restoration projects (Jenkinson et al., 2006), there is a significant gap in accessible stream restoration records of the past twenty years despite calls for a centralized tracking system (Hasset et al., 2005). Furthermore, many individual stream or floodplain restoration projects do not have quantifiable measures of nutrient retention and rarely collect pre-restoration data (Rumps et al., 2007; Bernhardt and Palmer, 2011; Burch et al., 2019). Without an agreed-upon baseline of stream quality beforehand, the success of stream restorations may not be adequately valued. Furthermore, McMahon et al. (2020) caution that the effects of restoration can be delayed by years and may not be observable at larger scales. A few studies have examined if functions change as stream restorations age (Orzetti et al, 2010; Hasselquist et al, 2015), but the C and nutrient response to stream restoration over time is understudied (Pander & Geist, 2013). Because stream restoration outcomes have not been effectively quantified and aquatic habitat responses may be minimal, there is growing criticism of stream restoration practices (Stranko et al., 2012; Kenney et al., 2012). Increased nutrient storage in floodplain soils of restored streams can act as an easily quantifiable indicator of restoration success, particularly when the project goals include improved water quality downstream (Wolf et al., 2013). The source-sink behavior is critical to understanding floodplains' role in stream biogeochemistry (Lammers & Bledsoe, 2017).

The goal of this study is to determine if stream restoration increases the nutrient retention of floodplain soils and to identify the mechanisms driving changes in floodplain soils. Together these inform our understanding of the effectiveness of stream-floodplain restoration as a management technique for nutrient reduction. Our analyses tested metrics of C, N, and P storage, including total C (TC), total N (TN), and total P (TP) concentrations, as well as C and P biogeochemical fluxes (carbon potential mineralization and equilibrium phosphorus concentration). These C and nutrient metrics were compared over a chronosequence of time since restoration and compared to unrestored disturbed streams and less disturbed reference streams to assess if water quality functions change as a result of restoration. Key environmental variables and soil physicochemical attributes known to control C and nutrient dynamics were used to isolate which parameters explain any changes in floodplain nutrient and carbon storage.

Methods

Site Description

Fairfax County, VA, USA is one of the most urbanized and densely populated counties in the United States due to its proximity to Washington D.C. (Han et al., 2019), and this population density has affected the 1600 miles of stream reaches within the county borders (Carinci, n.d). Many of these reaches are in some state of degradation due to 400+ years of land use from deforestation, agriculture, civil war and urbanization. Unintentional channelization has also occurred through stormwater runoff from impervious surfaces that have vertically incised and laterally eroded channels, which can then disconnect and dry floodplains (Klein, 1979; Booth, 1990; Bledsoe & Watson, 2001). Increasing urbanization and climate change are likely to exacerbate existing degradation, preventing functional stream equilibrium within the scale of the next 100 to 1000 years (Nelson et al., 2009). There are three physiographic provinces found within Fairfax County, all of which have distinct parent materials and geologic characteristics: a sub-province of the Triassic basin known as the Culpeper Basin, the Coastal Plain, and the largest province the Piedmont (Froelich & Zenone, 1985). As source material and underlying geology can impact the overlying biology and can confound analysis of trends of restorative action success, sites were selected from the Piedmont physiographic province only. The Piedmont region has quaternary geology with crystalline metamorphic bedrock and alluvium and terrace deposits along the margins of some streams (Pavlides, 1990).

Between 2003 and 2020, over 60 stream restoration projects were completed in the Piedmont basin by Fairfax County, and an additional 30 are under some degree of project design. Fairfax County has implemented stream restoration to reduce stream nitrogen, phosphorus, and sediment loads downstream, improve habitat, and stabilize channels to protect infrastructure in streams identified as degraded. Almost universally, streams in an unrestored condition are incised (average over 4 feet), have become disconnected from any original floodplain, have experienced development in the watershed within the last 10 years, are generally forested but have a high percentage (>35% cover) of invasive shrub and ground layer, have aquatic macroinvertebrate IBI scores in the very poor, poor or fair category, and have housing or structures within 300 horizontal feet and private property within the 100-year floodplain; these constraints are factored into all stream restoration project designs.

Over the past decade, stream restoration engineering has moved away from more traditional grade control structures and bank armoring to incorporate more floodplain reconnection through grading of the slopes and incorporating floodplain benches (Berg et al., 2014, Altland et al., 2020). Some stream restorations remove legacy sediment for floodplain reconnection, but others instead raise the stream bed and convert legacy sediment terraces into floodplains (Berg et al., 2014). However, there are limits placed upon these restorations in urban systems due to public acceptance of the projects (Altland et al., 2020). With limited undeveloped land, floodplains often must be much narrower and inset, and the floodplain wetness is often tempered to limit mosquitoes (Altland et al., 2020). In restored floodplains, the widths of the floodplain can vary from less than a meter to over thirty meters, but most floodplains on smaller channels are under five meters wide.

To minimize confounding influences when evaluating the effect of time since restoration on stream-floodplain functions, stream restoration projects considered for this research were chosen to have similar area and impervious surface cover (ISC) percentage of their drainage areas. Using 3 m digital elevation models (DEM, Hopkins et al., 2020), a stream network for Fairfax County was used to determine the drainage basins and percent ISC of the restoration projects using StreamStats software (https://streamstats.usgs.gov/ss/). Only stream restorations with drainage basins larger than 0.5 km^2 (to ensure floodplain presence) and smaller than 3.25 km^2 were selected to limit confounding variables associated with larger streams.

From the qualifying watersheds, a total of 18 sites across five different classes of time since restoration completion (three sites per age class/restoration phase) were chosen in Fairfax County, VA (Figure 1): unrestored, minimum disturbed streams ("reference"), unrestored disturbed streams ("unrestored"), < 3 years since restoration, 3-5 years since restoration, 6-9 years since restoration, and 10+ years since restoration. A pre-restoration age class provides a similar comparison of the restored streams to unrestored disturbed systems in a similar urban setting. The under 3 years since restoration age class evaluates the initial adjustment period after the restoration is completed when the disturbance of construction may suppress ecological function (Brown, 2000). The 3-5 years since restoration age class is based upon recommendations for urban stream restorations with TMDL credits to require evaluation for renewal after 5 years (Berg et al., 2014). In addition, the U.S. Army Corps of Engineers monitoring requirements for mitigation permits expire at 5 years (USACE No. 08-03, 2008). The 6-9 years since restoration age class demonstrated noticeable improvement in water quality after buffer restoration in previous studies (Orzetti et al., 2010). Finally, the oldest age class (9.7-10.5 yrs, based upon the number of days since restoration completion, simplified to 10+ based on rounding) was chosen because previous research shows 10 years as a significant benchmark for improvements in stream function (Orzetti et al, 2010), and projects from this timeframe coincide with U.S. Executive Order EO-13508 (2009) when pollution reduction became a key goal in Chesapeake stream restoration projects. In addition, 10 years is often when the most robust of monitoring requirements expire for mitigation permits (USACE No. 08-03, 2008). An additional three stream sites with ISC under 3% within the Piedmont physiographic province were chosen as a baseline for a healthier stream-floodplain complex with minimal disturbance from urbanization. These sites, though still impacted by legacy sediment (Noe et al., 2020), are the closest equivalent to “reference conditions” available in the same geological subregion (Society for Ecological Restoration, 2004).

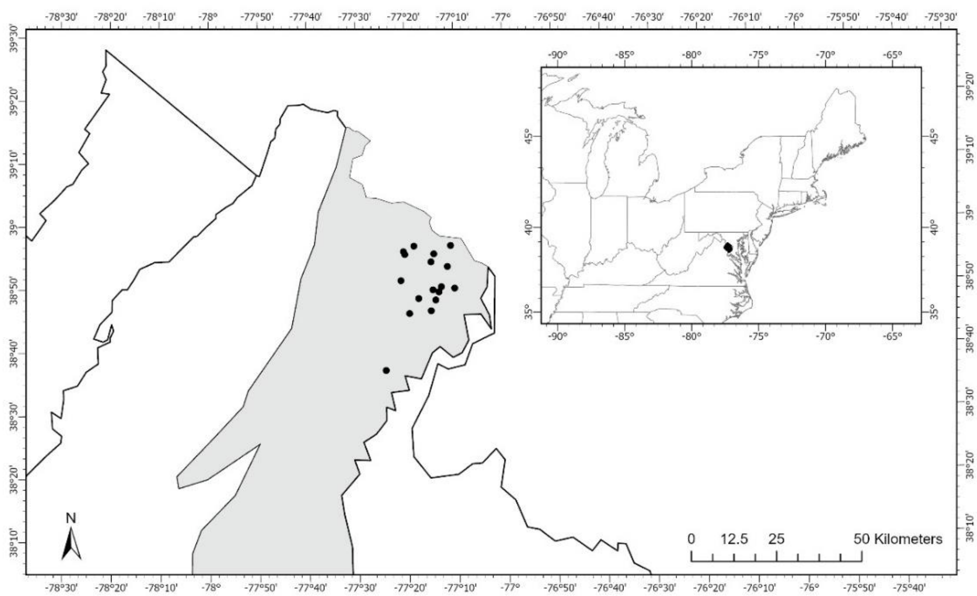

Figure 1: Locations of stream sites within the Piedmont Physiographic Province of Virginia. The Piedmont is denoted with grey shading, and the black dots denote site locations. Study location noted as a box on the inset map of the mid-Atlantic United States.

Floodplain Soil Sampling and Processing

Six sampling locations were chosen along the active floodplain of each of the 18 sites. Three cross-sections were established per site, each placed at a crossover where the thalweg shifts from one bank to the other (Kondolf & Micheli, 1995). Two floodplain soil core sampling locations along each cross-section were haphazardly selected from within the back swamp zone of the floodplain. This sampling design encompassed the longitudinal and lateral spatial heterogeneity within the dominant geomorphic zone of the natural and restored stream-floodplains. The average width of the floodplains measured was 2.7 m (range: 0.7 to 4.8 m).

In March 2022, two soil samples per sampling location were collected using a thread-on slide bulk-density hammer (5.7 cm diameter) to a depth of 10 cm (McKenzie et al., 2000; Yin et al., 2019). One core was collected for analyses requiring dried soil, and the other for analyses requiring field-moist soil. Each core was placed in a Ziploc bag and chilled in the dark until processing. In addition, 0.5 L of representative stream water samples were collected, passed through 1.5 µm porosity glass fiber vacuum filters in the lab, and stored at 4°C in the dark until analyzed for dissolved ortho-phosphate.

Additional soil cores were collected in May of 2022 at the start of the growing season for the bottle incubations of carbon potential mineralization, and in October of 2022 for bioavailable phosphorus analysis using Mehlich-3 extractions. Although seasonal variations impact both phosphorus and carbon dynamics in soil (Shi et al., 2013; Trentman et al., 2020), the relative effect of the season was constant across all sites, having limited influence on comparisons across the chronosequence.

Bulk Density Analysis and soil pH

Each core designated for ‘dry analyses’ was homogenized, weighed, dried at 60°C overnight, and reweighed to calculate bulk density as well as gravimetric moisture content. Samples were then ground with a mortar and pestle and sieved to remove particles greater than 1 mm and coarse organics. This process allows for more uniform subsamples to be taken for additional “dry weight” analyses, including organic matter, particle size, and total carbon.

In addition, the pH of a subsample of each ‘dry’ soil core was measured by creating soil slurries of 10 g of air-dried soil with 20 mL DI water. After 10 minutes rest, each slurry’s pH was measured using a pH meter (Beckman Coulter Series 500) with two-point calibration (pH 4 and 7 buffers).

In addition, the pH of a subsample of each ‘dry’ soil core was measured by creating soil slurries of 10 g of air-dried soil with 20 mL DI water. After 10 minutes rest, each slurry’s pH was measured using a pH meter (Beckman Coulter Series 500)with two-point calibration (pH 4 and 7 buffers).

Carbon Analyses

A subsample of each dried sample underwent combustion to calculate loss on ignition (LOI) as a proxy for organic matter (OM) content (Karam, 1993). Samples were burned at 550°C for four hours in a muffle furnace and cooled in a desiccator before reweighing. The post-combustion mineral matter was then used for particle size analysis using a laser diffraction analyzer that used hexametaphosphate and sonication to disaggregate particles (Beckman Coulter LS 13 320 Laser Diffraction Particle Size Analyzer). The total carbon (TC) of dried subsamples was measured on a Thermo Scientific FlashSmart CHNS/O Elemental Analyzer.

In addition, carbon potential mineralization rates were measured. Following the modified protocols of Paul et al. (1999), laboratory incubations were conducted on homogenized field-moist samples of soil adjusted to 60% of water holding capacity. The soils were incubated in glass canning jars covered with Kimwipes at 25°C in the dark for seven days and then measured in triplicate for each sample using a soil CO_2 flux system (LiCOR-8150). LI-8150 measures the change of CO_2 mole fraction of the air headspace over time in a closed chamber to calculate soil mineralization. The carbon potential mineralization was then normalized for available carbon by dividing the rate of potential mineralization by measured soil TC concentration from the same coring location. This metric of carbon turnover rate estimates the relative carbon potential mineralization of the floodplain independent of the amount of material available for mineralization.

Phosphorus Analyses

Bioavailable P, Al, and Fe components of the samples were determined using Mehlich-3 (M3) extractions. The use of ammonium oxalate ($C_2H_8N_2O_4$) as a measurement of phosphorus sorption in soils is an established practice (Schoumans, in Pierzynski, 2000); however, M3 has increasingly been used as an alternative method to test for bioavailable P concentrations within soil samples, as well as Fe and Al (Inamdar et al., 2020; Lammers & Bledsoe 2017). Samples taken from the October core were homogenized and air-dried over 3 days at room temperature (Kleinman & Sharpley, 2002; Odhiambo et al., 2016). These samples were then sent to the Penn State Extension Laboratory to be extracted with M3 solution and analyzed using Inductively Couple

Plasma- Optical Emission Spectroscopy (ICP-OES) (Odhiambo et al., 2016; Baker et al., 2005). ICP-OES analyzes both aqueous and organic liquid and samples through measurements of wavelength spectra to determine elemental composition (EPA Method 6010). From this, the degree of phosphorus saturation (DPS; Delaune et al., 2013) of the soil can be calculated as the amount of M3 phosphorus divided by the amount of M3 iron and aluminum present in the soil sample. In addition, to measure total phosphorus (TP), a 0.25g representative subsample of each oven dried and sieved sample was fully digested by microwave-assisted digestion (EPA Method 3052) in a SCP MultiVIEW Microwave Digestion System, and then diluted and analyzed using ICP-OES.

In addition, EPC_0 was assayed for each of the samples. For each soil sample, five incubations were made using 1 g of fresh soil and 20 mL of filtered stream water (Dead Run, background dissolved orthophosphate concentration of 0.042 mg-P L^{-1}) treated with an additional 0, 0.25, 0.5, 1, and 2 mg P L^{-1} using dissolved potassium phosphate (KH_2PO_4). Two drops of chloroform were added to each solution to inhibit microbial activity to focus on mineral uptake or release of P. After 24 hrs., incubations were filtered using 0.45 µm PTFE filters and measured colorimetrically for dissolved phosphate concentration using a AQ2 discrete analyzer (EPA-155-A Rev 0). The P sorbed on the soil after 24 hours (*S*, mg L g^{-1}) was calculated (Inamdar et al., 2020, developed from Haggard et al., 2007), and then graphed with P spike concentrations on the x-axis and *S* on the y-axis. EPC_0 was found as the x-intercept through logarithmic regression.

$$S = \frac{v}{m}(C_0 - C_{24})$$

Equation 1: equation to calculate the P sorbed onto soil (S, mg L g-1) after 24 hr. incubations, developed from Haggard et al., 2007, where v is the volume of the solution (L), m the mass of the dry soil (g), C0 is the initial solution P concentration, including the spike and the stream background concentration, and C24 is the P concentration of solution after 24 hrs incubation.

Stream water samples from each site were analyzed for dissolved orthophosphate (o-P) by the direct colorimetric analysis procedure (EPA Method 365.3) on a Seal AQ2 discrete analyzer. These values were compared to the EPC_0 values of the floodplain soils at the closest initial P concentration to predict the sink/source behavior of the floodplain at each site.

All data from this study is available at https://doi.org/10.5066/P9Y4ETDD (Napora et al. 2023).

Statistical Analyses

For each site, the six subsamples were averaged together. In one subsample, there was an outlier for the ratio of organic matter to total carbon that was not included in that site’s average for those individual parameters. Four separate linear models were run to directly test the influence of age since restoration (in years) on floodplain soil TC, TP, carbon turnover, and EPC_0, excluding the unrestored reference sites but including the unrestored sites as age zero. The reference sites were then incorporated in an ANOVA to compare with the restored age classes using a post-hoc Tukey HSD. These analyses were performed on-site average values from the multiple soil cores taken within each site. Another set of linear models was constructed to test the influence of age since restoration (in years) on the floodplain soil physicochemical attributes, also excluding reference sites. Again, an ANOVA and post hoc Tukey HSD test were run to see if reference sites were significantly different from those of the chronosequence of restored streams. Significance was accepted in all ANOVA tests at $p < 0.05$.

After determining the effects of time since restoration, generalized linear models (GLMs) were then used to identify the effects of watershed attributes and soil physicochemistry on soil TC, TP, carbon turnover, and EPC_0. Explanatory variables evaluated in the models included average streambank height, drainage basin area, impervious surface cover in the drainage basin, soil moisture, soil organic matter, soil pH, DPS, and median soil particle size. Soil bulk density and iron concentrations were not included in the model due to the multicollinearity with soil moisture, and aluminum and were not included due to multicollinearity and with DPS, respectively. The selection of explanatory variables was conducted based on corrected Akaike's information criterion (AICc) to account for smaller sample sizes (Burnham et al., 2010). These analyses were performed on values from individual soil cores. All statistical analyses were performed using R software (R version 4.0.4).

Results

Time since restoration influences TC and TN, as well as potential C mineralization

Time since restoration had a significant effect on floodplain soil TC and TN, as well as carbon turnover ($p < 0.05$, Table 1). Total C and N present in the soil increased as time since restoration increased with no initial drops immediately following construction (Figure 2). In fact, although not significantly different in pairwise comparisons, the youngest restoration sites typically had 3 x greater floodplain soil TC than unrestored sites. The TC and TN present was significantly different and greater than those of the unrestored sites in the 6-9 and 10+ year age classes (Tukey HSD, $p < 0.05$). Soil moisture was the most explanatory environmental metric for TC (based upon AICc values, Table 2). TN behaved very similarly to TC, increasing with time since restoration ($p < 0.05$, Table 1) and with the same age class differences to unrestored sites. Soil organic matter was the best model for TN, which helps to explain this linked relationship (Table 2). Soil moisture and OM also increased with time since restoration ($p < 0.05$, Figure 3).

Table 1*:* **Summary of results for simple linear models comparing soil biogeochemical metrics including total carbon (TC), total nitrogen (TN), total phosphorus (TP), carbon turnover (carbon potential mineralization normalized to available carbon), and the equilibrium phosphorus concentration (EPC_0; the concentration at which P sorbed onto the soil equals the P that goes into solution). Asterisks note significant relationships.**

Linear Model	DF	Adjusted R^2	F-statistic	p-value
TC ~ years since restoration	13	0.730	38.9	3.02e-05*
TN ~ years since restoration	13	0.694	32.7	7.07e-05*
TP ~ years since restoration	13	0.596	21.6	0.000456*
C Turnover~ years since restoration	13	0.429	11.5	0.00480*
EPC_0 ~ years since restoration	13	-0.0752	0.0205	0.888

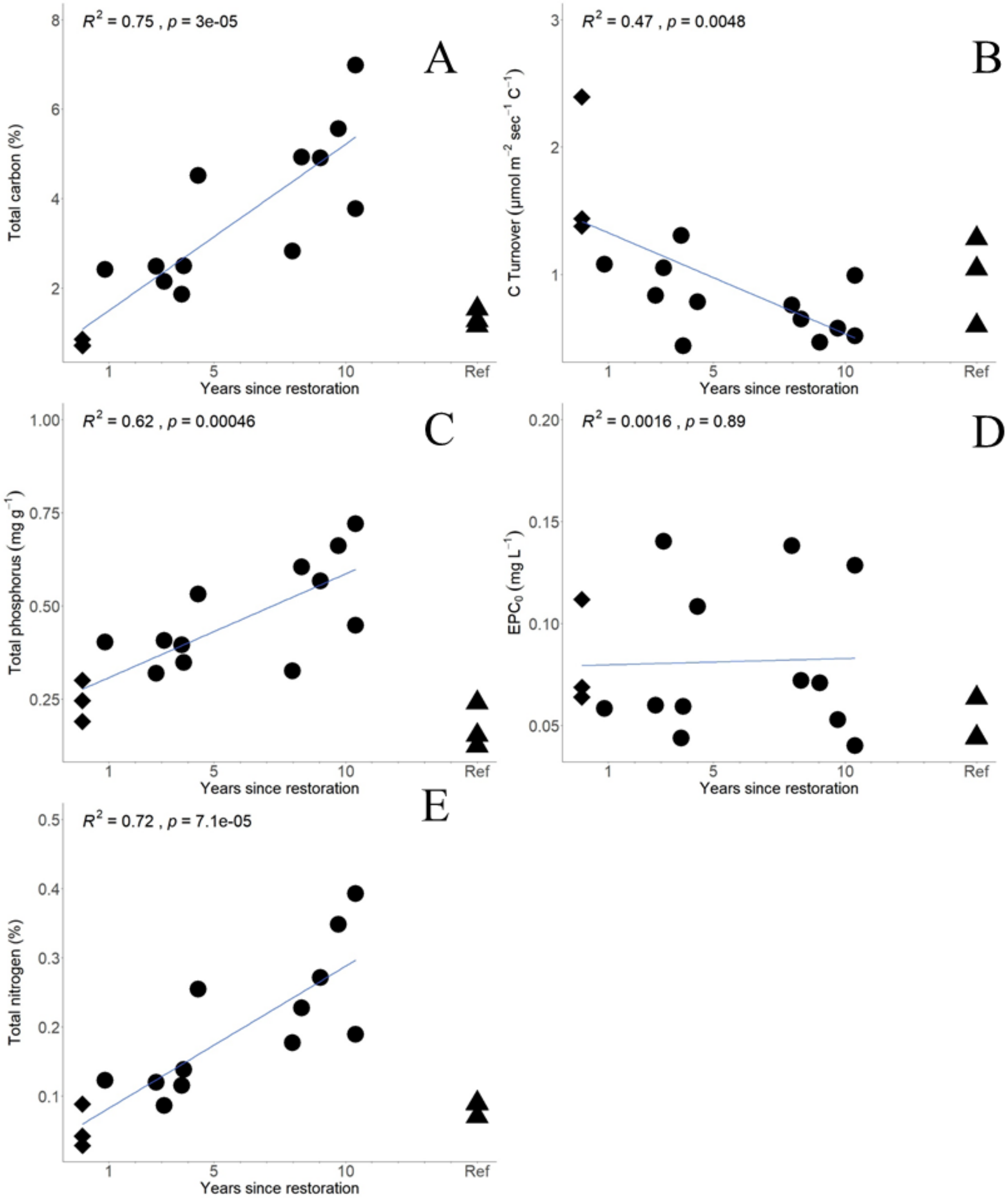

Figure 2: Relationship between years since stream restoration and the site average floodplain soil biogeochemical metrics: A) total carbon (%); B) carbon turnover (µmol m^{-2} sec^{-1} c^{-1}); C) total phosphorus (mg g^{-1}); D) equilibrium phosphorus concentration, or the concentration at which P sorbed onto the soil and P released into solution are equal (mg L^{-1}); and E) total nitrogen (%). Unrestored sites are designated by the diamond symbol, while restoration sites are designated by the circle symbol. Reference sites are designated by the triangle symbol, and not included in the trendline. R^2 and p-values are included in each graph.

In contrast, the carbon potential mineralization normalized for total carbon (carbon turnover) decreased (Figure 2). Although the absolute carbon potential mineralization correlated with TC along the chronosequence, increasing with time since restoration (data not shown), the carbon turnover was found to decrease the older the restoration project, with no spike in the younger restoration sites (Figure 2). Like TC and TN, the carbon turnover rate differed significantly from the pre-restoration sites at the 6-9 and 10+ age classes (Tukey HSD, $p < 0.05$). Soil moisture was the best available model for predicting the carbon potential mineralization, but particle size (reported as the 50th percentile, or d50) was the dominant environmental variable for carbon turnover (Table 2) and decreased with increasing time since restoration ($p < 0.05$, Figure 3). The reference sites had higher rates of carbon turnover compared to the oldest restored sites, but this could be attributed to the larger D50 values present in the reference sites.

Table 2*:* Top-ranking models predicting soil biogeochemical metrics, including total carbon (TC), total nitrogen (TN), total phosphorus (TP), carbon turnover (carbon potential mineralization normalized to available carbon), and the equilibrium phosphorus concentration (EPC_0; the concentration at which P sorbed onto the soil equals the P that goes into solution), as assessed with Akaike's information criterion corrected for small sample size (AICc). The number of parameters estimated, including the intercept and random effect (k), AICc, and AICc weight (W) are provided.

Variable	Top-ranked Model	K	AICc	W
TC	Soil Moisture	3	54.87	0.80
TN	Organic Matter	3	-68.60	0.72
TP	Organic Matter	3	-40.86	0.78
C Turnover	D50	3	18.34	0.54
EPC_0	Soil pH	3	-73.52	0.51

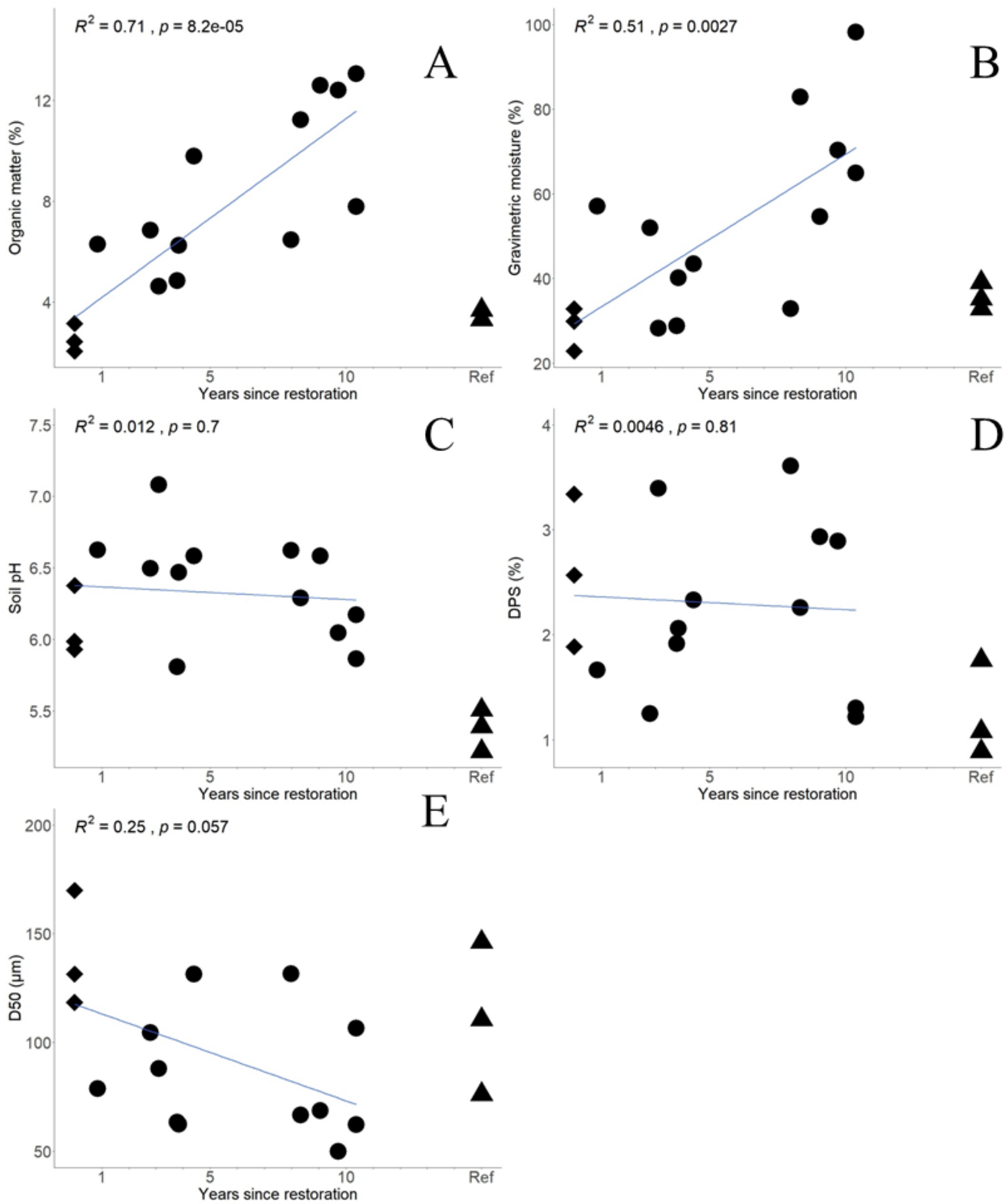

Figure 3: Relationship between years since stream restoration and the site average floodplain soil characteristics: A) organic matter (%); B) gravimetric moisture (%); C) soil pH (%); D) degree of phosphorus saturation, or the relative amount of P to available Al and Fe binding sites; and E) particle size (reported as the 50th percentile, or d50, µm). Unrestored sites are designated by the diamond symbol, while restoration sites are designated by the circle symbol. Reference sites are designated by the triangle symbol and not included in the trendline. R2 and p-values are included in each graph.

Time since restoration increases TP but does not drive its sorption/desorption behavior

TP increased with time since restoration ($p < 0.05$, Table 1). The relationship follows a linear pattern with no initial drop immediately following construction, and instead a nonsignificant tendency of higher TP in the youngest restorations relative to unrestored sites (Figure 2). However, TP differed significantly from the pre-restoration sites in the 10+ years age class (Tukey HSD, $p < 0.05$). Soil organic matter was the best model for explaining TP (Table 2).

The EPC_0 metric of phosphate sorption capacity did not change with time following stream restoration ($p > 0.05$, Figure 2), with no difference between the unrestored sites and those along the chronosequence, nor the reference sites. Rather, soil pH was the best predictor of soil EPC_0 (Table 2). Excluding the reference sites, the measured orthophosphate concentrations of the stream water were lower than the average EPC_0 values of the soil, suggesting floodplain soil releases phosphates into the water during overbank flow, but the overall increases in TP suggest that inputs P exceeds losses due to desorption. Based on the cumulative weight of potential models, DPS could also play an important role in determining EPC_0 values, but although both soil pH and DPS were important drivers for EPC_0, neither characteristic was influenced by time since restoration ($p < 0.05$, Figure 3).

Reference streams deviate from chronosequence trends

The unrestored, minimum disturbed stream reference sites had lower values of the floodplain soil biogeochemical metrics compared to the older restored streams. Reference sites differed significantly with less soil TC, TN, and TP storage than the two oldest restoration age classes (6-9 years since restoration, and 10+ years since restoration) but were similar to the younger restoration age classes and unrestored disturbed streams. Reference stream soils had similar carbon turnover and EPC_0 values as restored and unrestored disturbed streams (Figure 2). Furthermore, the environmental characteristics of the reference sites were more similar to pre-restoration sites than older stream restorations (10+ years old), with lower soil moisture and OM, as well as higher D50 (Figure 3).

Discussion

Time since restoration influences carbon increases and potential losses

Floodplain soil TC increased linearly as stream restoration projects aged, with no drop after construction. Soil organic matter and soil moisture both increased along the chronosequence, suggesting the underlying mechanism of how time since stream restoration affects carbon dynamics. Carbon pools develop over time, as there are more opportunities for deposition through flooding events as well as leaf litter inputs and belowground productivity with multiple growing seasons (Giese et al., 2000). Fairfax County’s restoration practices in particular often reestablish vegetation immediately after construction is complete, which could help prevent the potential time lag (Altland et al., 2020). In addition, wetland carbon is dependent on the saturation of the soil to increase the soil storage potential, as typically saturated soils have depressed rates of soil loss by decomposition (Wolf et al., 2013). Often one design goal of stream restoration projects is to reconnect the floodplain banks alongside incised channels, increasing the hydrologic connectivity to stream overbank flooding and increasing soil moisture. The stream restoration projects often raised the beds of the stream channels, increasing groundwater table elevations and soil moisture. Soil TN behaved similarly, and although the mechanisms are not identical to TC, the underlying drivers are the same. Stream restoration increased floodplain soil moisture, increasing the potential for C and nutrient accumulation.

Soil potential mineralization, when normalized by the amount of C present, decreased as restoration projects aged. Due to the increased carbon available for microbial decomposition, the older restoration projects demonstrated higher rates of potential mineralization (Wood and Yarwood, 2022). However, per unit of carbon, the organic carbon in older sites emitted less CO_2. As the restoration project ages, it has more opportunities for carbon inputs, such as litterfall and stream sediment deposition. Early restorations will be dominated by annuals and herbaceous material; older restorations will have deciduous leaves (Wood and Yarwood, 2022). Newer inputs of cellulose-rich labile carbon will be decomposed first, leaving an increasing pool of more lignocellulose-rich refractory and recalcitrant carbon pools (Melillo et al., 1989). Newer designs often incorporate more floodplain wood to have immediate contributions, although these may be localized (Altland et al., 2020). Furthermore, environmental characteristics like the higher moisture content in the soil can suppress microbial decomposition due to anoxia, allowing more organic matter to accumulate (Wolf et al., 2013). Soil texture (particle size, reported as d50) plays a significant role in the carbon turnover: per unit carbon, potential mineralization increased with increasing median particle size. These results are corroborated by Harrison-Kirk et al. (2013), where the effect of increasing SOC on carbon potential mineralization was much greater for silt loam than for clay loam soils undergoing dry/wet cycles. Butterly et al. (2010) also found that increases in mineralizable C were explained by the clay content of the soil. The finer textured soils can stabilize carbon pools with clay surfaces, promoting the physical occlusion of organic matter within aggregates and protecting organic matter from decomposition (Butterly et al., 2010). Sandier soils allow for the accessibility of microbes to the present organic matter due to the larger pore spaces and susceptibility of soil microbes to changes in water potential due to the periodic drying and rewetting (Butterly et al., 2010). Particle size also explains in part why the reference sites demonstrated such high normalized potential mineralization: all three reference sites demonstrated sandier soil textures compared to those of the restoration chronosequence. Overall, the growing pool of soil total carbon in floodplain soils of restored streams is due to increasing C inputs from both belowground production and surficial soil inputs from plant litter and sediment deposition, exceeding the decomposition rates (Wood and Yarwood, 2022).

Time since restoration increases total phosphorus, but not its sorption/desorption

Much like TC, soil TP concentration increased along the chronosequence due to inputs exceeding losses over time in the soil. Younger restored sites do not have as many opportunities for nutrient deposition with fewer cumulative flooding events compared to older sites. The correlation between TP and the organic matter present in the soil may correlate with the number of available binding sites present in the soil. Hogan et al. (2004) found that in forested wetlands in Maryland, a considerable proportion of soil Al was organically bound. As soil OM accumulates in restored stream floodplains, the organically bound Al may become more important in binding P (Hogan and Walbridge, 2007). Restored floodplains also demonstrated increasing TP along the chronosequence, with no evidence of a plateau. However, the oldest restoration sites included in this study were under 15 years old, so TP accumulation beyond that time frame may slow as binding sites become saturated with phosphate. Future work could incorporate a longer timescale, depending on the location and availability of older stream restoration sites, as well as same-site comparisons using repeated measures.

The floodplain soil phosphate sorption potential (EPC_0 values) did not correlate with TP or exhibit any trend along the chronosequence. Total phosphorus includes recalcitrant pools of phosphorus unavailable for sorption or desorption, which would not influence EPC_0. Instead, factors inherent to the soil geochemistry play a more prominent role compared to restoration age, such as the soil pH (Penn and Camberato, 2019) and available Fe and Al present in the soil (Walbridge & Struthers 1993).

Soil pH determines the P solubility based on the impacts on binding sites. Higher soil pH values were found to have higher EPC_0 values in turn. Due to the anion exchange at the surface binding sites of minerals, in soils ranging from 3-5 pH, phosphorus fixation is dominated by Fe, whereas soils in the pH range of 4-7 have Al phosphorus fixation (Penn and Camberato, 2019). Soil DPS demonstrated a weaker but positive relationship with EPC_0, but DPS did not change along the chronosequence. The Fe and Al concentrations of floodplains are not static: soils that are saturated for shorter periods of time have a greater capacity to sorb P due to higher Al and Fe concentrations; but chronic flooding could result in a gradual loss of both Al and Fe over time, with a concomitant loss in the capacity to retain P (Hogan et al., 2004). Due to the episodic nature of flooding events in streams, the available pools of potential Fe and Al binding sites (and thus EPC0) can change with restoration age. In addition, new sediment is deposited on floodplains over time. That new sediment, which changed soil texture to smaller particles, may also have undersaturated phosphate sorbed compared to Al and Fe sorption sites. Therefore, the development of additional phosphate sorption capacity over time since restoration may have resulted in no change in net sorption/desorption dynamics. This suggests that, like the C dynamics, the pool of TP in floodplain soils of restored streams grows from increasing P inputs outweighing potential losses due to desorption into stream water.

The reference stream sites have less P and C storage than restored sites

The three reference streams were chosen based on minimal ISC of the drainage area (<3%) to mitigate the largest known source of negative effects of urbanization on stream systems (Graf, 1975; Paul and Meyer, 2001). Surprisingly, these reference sites exhibited lower TP and TC concentrations than the older (6-9 yrs and 10+ yrs) restoration sites. Although not exposed to the same issues of urbanization that can degrade streams, these sites still exhibit altered hydrologic connectivity due to the legacy sediment of the colonial era (Mattern et al., 2020). Legacy sediment accumulated in valley bottoms of the eastern U.S. during European settlement due to erosion from land-clearing agriculture (James, 2013). Without active intervention to remove this buildup, the accumulation of sediment can disconnect the floodplain from the stream and lead to incised channels and a lack of hydrologic connectivity (Johnson et al., 2019), explaining the comparatively lower soil moisture content and lower concentrations of TP and TC of these reference sites. As to whether the dynamics present in older restored sites are more similar to a 'truly undisturbed stream system', to our knowledge the northern Virginia Piedmont does not have an equivalent stream system for comparison. Nonetheless, stream restoration has led to wetter, more organic floodplain soil that supports greater storage of P and C, and greater stability of that C, compared to both unrestored disturbed streams and less disturbed streams.

Conclusion

Stream restoration increased the amount of carbon and nutrient storage by floodplain soils without comparable increasing rates of loss. Furthermore, accumulation of carbon and nutrients by restored stream-floodplains was immediate, experienced no initial dip and increased with restoration age. Although we did not measure streambank erosion, which could counterbalance floodplain retention (Noe et al., 2022), the stream restorations were designed to minimize erosion. These findings suggest net retention of phosphorus, nitrogen, and carbon, and that the goal of improving water quality functions was successful for these stream restoration projects. These functions improved compared to unrestored, disturbed streams and with time since restoration due to increasing the wetness of floodplain soils through stream restoration, and thereby restoring water quality functions. Impervious surface cover was not a significant predictor of storage or rates of loss in any of the biogeochemical metrics measured, suggesting that potential success of water quality improvements is not hindered by the surrounding drainage basin. Furthermore, the closest analogues to healthy streams present in the surrounding area were less effective in their storage rates compared to the older (> 6 yr old) stream restoration sites. Although more modern degradation from urbanization has exacerbated the effects of downstream carbon and nutrient pollution, streams in the U.S. Mid-Atlantic have been negatively impacted by European colonization for centuries. Active intervention in the form of stream restoration can be extensive and alter the landscape, but these alterations improve water quality and ecological functions as they age. Although success is difficult to quantify, water quality goals and overall biogeochemical function can be aided by stream restoration, and positive ecological effects increase over time.

References for First Manuscript

Altland, D., Becraft, C., Berg, J. Brown, T., Burch, J., Clearwater, D., Coleman, J., Crawford, S., Doll, B., Geratz, J., Hanson, J., Hartranft, J., Hottenstein, J., Kaushal, S., Lowe, S. Mayer, P., Noe, G., Oberholzer, W., Parola, A., Scott, D., Stack, B., Sweeney, J., White, J. 2020. Consensus Recommendations to Improve Protocols 2 and 3 for Defining Stream Restoration Pollutant Removal Credits. Chesapeake Stormwater Network.

Baker, W. H., Herron, C. G., Caroll, S. D., Henslee, M. A., Lafex, D. C., Evans, E. E. 2002. A Comparative Summary of the Standard Mehlich 3 Soil Test with a Modified Mehlich 3 Dilution Ratio Procedure. Arkansas Agricultural Experiment Station Special Report 205.

Baldwin, D. S., Mitchell, A. M. 2000. The effects of drying and re-flooding on the sediment and soil nutrient dynamics of lowland river-floodplain systems: A synthesis. River Research and Applications, 16(5), 457–467. https://doi.org/10.1002/1099-1646(200009/10)16:5<457::aid-rrr597>3.0.co;2-b

Berg, J., Burch, J., Cappuccitti, D., Filoso, S., Fraley-McNeal, L., Goerman, D., Hardman, N., Kaushal, S., Medina, D., Meyers, M., Kerr, B., Stewart, S., Sullivan, B., Walter, R., Winters, J. 2014. Recommendations of the Expert Panel to Define Removal Rates for Individual Stream Restoration Projects. Urban Stormwater Workgroup

Bernhardt, E. S., Palmer, M.A. 2007. Restoring streams in an urbanizing world, Freshwater Biol., 52, 738–751.

Bernhardt, E. S., Palmer, M.A. 2011. River restoration—The fuzzy logic of repairing reaches to reverse watershed scale degradation, Ecol. Appl., 21, 1926–1931.

Bernhardt, E.S., Palmer, M.A., Allan, J.D., Alexander, G., Barnas, K, Brooks, S., Carr, J., Clayton, S., Dahm, C., Follstad-Shah, J., Galat, D., Gloss, S., Goodwin, P., Hart, D., Hassett, B., Jenkinson, R., Katz, S., Kondolf, G.M., Lake, P.S., Lave, R., Meyer, J.L., O'Donnell, T.K., Pagano, L., Powell, B., Sudduth, E. 2005. Synthesizing U.S. River Restoration Efforts. Science Policy Forum Ecology 308, 636-637

Bledsoe, B. P., Watson, C. C. 2001. Effects of urbanization on channel instability. Journal of the American Water Resources Association, 37(2), 255–270. https://doi.org/10.1111/j.1752-1688.2001.tb00966.x

Booth, D. B. 1990. Stream-channel incision following drainage-basin urbanization. Journal of the American Water Resources Association, 26, 407–417. https://doi.org/10.1111/j.1752-1688.1990.tb01380.

Briggs, M.K., Osterkamp, W.R. (Eds.), 2021. Renewing Our Rivers: Stream Corridor Restoration in Dryland Regions. University of Arizona Press, Tucson, AZ, U.S.A. ASIN:B08PK92QY4.

Brown, K. B. 2000. Urban stream restoration practices: an initial assessment. Center for Watershed Protection, Ellicott City, Maryland.

Burch, J., Cox, S., Davis, S., Fellows, M., Hoverman K., Law, N., Mumaw, K., Rauhofer, J., Schueler, T., Starr, R. 2019. Recommended Methods to Verify Stream Restoration Practices Built for Pollutant Crediting in the Chesapeake Bay Watershed. Urban Stormwater Work Group of the Chesapeake Bay Program.

Burnham K.P., Anderson, D.R., Huyvaert, K.P. 2010. AICc model selection in the ecological and behavioral sciences: some background, observations and comparisons. Behavior Ecology Sociobiology doi:10.1007/s0

Butterly, C.R., Marschner, P., McNeill, A.M., Baldock, J.A., 2010. Rewetting CO2 pulses in Australian agricultural soils and the influence of soil properties. Biology and Fertility of Soils 46, 739e753.

Carinci, C. n.d.. Stream Physical Assessment Program. Retrieved March 3, 2021, from https://www.fairfaxcounty.gov/publicworks/stormwater/stream-physical-assessment

Correll, D.L. 1998. The Role of Phosphorus in the Eutrophication of Receiving Waters. Journal of Environmental Quality 27, 261-266

Darke, A.K., and M.R. Walbridge. 2000. Al and Fe biogeochemistry in a fl oodplain forest: Implications for P retention. Biogeochemistry 51:1–32

DeLaune, R. D., Reddy, K. R., Richardson, C. J. 2013. Phosphorus Sorption and Desorption in Wetland Soils. In Methods in biogeochemistry of wetlands (Vol. 10). essay, John Wiley & Sons, Inc.

EPA, U.S. 1983) Chesapeake Bay Program: Findings and Recommendations. Region 3, Sixth and Walnut Streets, Philadelphia, PA 19106.

Exec. Order No. 13508, 74 Fed. Reg. 57675 (November 9, 2009).

Fairfax County. n.d. Stormwater Projects of Fairfax County. Retrieved August 17, 2021, from https://fairfaxcountygis.maps.arcgis.com/apps/webappviewer/index.html?id=aff7282578d1454ba5fea6a09f0a3559

Fairfax County Department of Public Works. 2013. Description & Interpretive Guide to Soils in (Issue August 2011).

Froelich, A.J., Zenone, C. 1985. Maps showing geologic terrane, drainage basins, overburden, and low flow of streams in Fairfax County, and vicinity, Virginia: U.S. Geological Survey Miscellaneous Investigations Series Map I-1534, scale 1:24,000.

Giese, L. A., Aust, W. M., Trettin, C. C., Kolka, R. K. 2000. Spatial and temporal patterns of carbon storage and species richness in three South Carolina coastal plain riparian forests. Ecological Engineering, 15. doi:10.1016/s0925-8574(99)00081-6

Gordon, B. A., Dorothy, O., Lenhart, C. F. 2020. Nutrient retention in ecologically functional floodplains: A review. Water (Switzerland), 12(10), 15–17. https://doi.org/10.3390/w12102762

Graf, W.L. 1975. The Impact of Suburbanization on Fluvial Geomorphology. Water Resources Research 11(5): 690-692.

Haggard, B.E., Smith, D.R. Brye, K.R. 2007, Variations in Stream Water and Sediment Phosphorus among Select Ozark Catchments. J. Environ. Qual., 36: 1725-1734. https://doi.org/10.2134/jeq2006.0517

Haggard, B., Sharpley, A. 2007. Phosphorus Transport in Streams: Processes and Modeling Considerations. In Modeling Phosphorus in the Environment; Radcliffe, D., Carbrera, M., Eds.; CRC Press: Boca Raton, FL, USA, 105–130.

Hamilton, S.K. 2012. Biogeochemical time lags may delay responses of streams to ecological restoration: time lags in stream restoration. Freshwater Biol. 57, 43–57

Han, X., Khaja, F., Lamsal, M. 2019. Demographic Reports 2019: County of Fairfax, Virginia. Economic, Demographic, and Statistical Research.

Harrison-Kirk, T., Beare, M.H., Meenken, E.D., Condron, L.M., 2013. Soil organic matter and texture affect responses to dry/wet cycles: effects on carbon dioxide and nitrous oxide emissions. Soil Biology and Biochemistry 57, 43-55

Hasselquist, E.M., Nilsson, C., Hjältén, J., Jørgensen, D., Lind, L., & Polvi, L. E. 2015. Time for recovery of riparian plants in restored northern Swedish streams: A chronosequence study. Ecological Applications, 25(5), 1373–1389. https://doi.org/10.1890/14-1102.1.sm

Hassett, B., Palmer, M., Bernhardt, E., Smith, S., Carr, J., Hart, D. 2005. Restoring watersheds project by project: trends in Chesapeake Bay tributary restoration. Frontiers in Ecology and Environment 3(5), 259-267

Hill, T., Kulz, E., Munoz, B., Dorney, J.R. 2013. Compensatory Stream and Wetland Mitigation in North Carolina: An Evaluation of Regulatory Success. Environmental Management 51, 1077-1091

Hogan, D.M., Walbridge, M.R. 2007. Best Management Practices for Nutrient and Sediment Retention in Urban Stormwater Runoff. Journal of Environmental Quality 36, 386-395

Hogan, D.M., Jordan, T.E., Walbridge, M.R. 2004. Phosphorus Retention and Soil Organic Carbon in Restored and Natural Freshwater Wetlands. Wetlands 24(3), 573-585

Hopkins, K.G., Ahmed, L., Metes, M.J., Claggett, P.R., Lamont, S., Noe, G.B. 2020. Geomorphometry for Streams and Floodplains in the Chesapeake and Delaware Watersheds: U.S. Geological Survey data release, https://doi.org/10.5066/P9RQJPT1.

Hopkins, K. G., Noe, G. B., Franco, F., Pindilli, E. J., Gordon, S., Metes, M. J., Claggett, P. R., Gellis, A. C., Hupp, C. R., Hogan, D. M. 2018. A method to quantify and value floodplain sediment and nutrient retention ecosystem services. Journal of Environmental Management, 220(December 2017), 65–76. https://doi.org/10.1016/j.jenvman.2018.05.013

Inamdar, S., Sienkiewicz, N., Lutgen, A., Jian, G., Kan, Jinjun 2020. Streambank Legacy Sediments in Surface Waters: Phosphorus Sources or Sinks? Soil Systems 4:30

James, L.A. 2013. Legacy sediment: Definitions and processes of episodically produced anthropogenic sediment. Anthropocene, 2, 16–26.

Jenkinson, R.G., Barnas, K.A., Braatne, J.H., Bernhardt, E.S., Palmer, M.A., Allan, J.D. 2006. Stream restoration databases and case studies: a guide to information resources and their utility in advancing the science and practice of restoration. Restoration Ecology 14, 177–186.

Johnson, K.M., Snyder, N.P., Castle, S., Hopkins, A.J., Waltner, M., Merritts, D.J., Walter, R.C. 2019. Legacy sediment storage in New England river valleys: Anthropogenic processes in a postglacial landscape. Geomorphology, 327, 417–437.

Karam. A. 1993. Chemical properties of organic soils. P. 459-471. In M.R> Carter (ed.) Soil sampling and methods of analysis. Canadian Society of Soil Science, Ottawa, ON.

Kenney, M.A., Wilcock, P.R., Hobbs, B.F. Flores, N.E., Martínez, D.C. 2012. Is Urban Stream Restoration Worth It? Journal of the American Water Resources Association (JAWRA) 48(3), 603-615. DOI: 10.1111/j.1752-1688.2011.00635.x

Kiedrzyńska, E., Wagner, I., Zalewski, M. 2008. Quantification of phosphorus retention efficiency by floodplain vegetation and a management strategy for a eutrophic reservoir restoration. Ecological Engineering 33, 15-25

Klein, R. D. 1979. Urbanization and Stream Quality Impairment. JAWRA Journal of the American Water Resources Association, 15(4), 948–963. https://doi.org/10.1111/j.1752-1688.1979.tb01074.x

Kleinman, P.J.A., and A.N. Sharpley. 2002. Estimating soil phosphorus sorption saturation from Mehlich-3 data. Commun. Soil Sci. Plant Anal. 33:1825–1839. doi:10.1081/CSS-120004825

Kondolf, G. M., Micheli, E. R. 1995. Evaluating stream restoration projects. Environmental Management, 19(1), 1-15. doi:10.1007/bf02471999

Kronvang B., Grant R., Hoffmann C.C., Ovesen N.B., Pedersen M.L. 2006. Hydrology, sediment transport and water chemistry. In: Running Waters – Historical Development and Restoration of Lowland Danish Streams (Eds. K. Sand-Jensen, N. Friberg & J. Murphy), 26–43. National Environmental Research Institute, Silkeborg, Denmark.

Lammers, R. W., Bledsoe, B. P. 2017. What role does stream restoration play in nutrient management? Critical Reviews in Environmental Science and Technology, 47(6), 335–371. https://doi.org/10.1080/10643389.2017.1318618

Mattern, K., Lutgen, A., Sienkiewicz, N., Jiang, G., Kan, J., Peipoch, M., Inamdar, S. 2020. Stream Restoration for Legacy Sediments at Gramies Run, Maryland: Early Lessons from Implementation, Water Quality Monitoring, and Soil Health. Water 12, 2164; doi:10.3390/w12082164

McKenzie, N. J., Ryan, P., Fogarty, P., and Wood, J. (2000). Sampling, measurement and analytical protocols for carbon estimation in soils litter and coarse woody debris. National Carbon Accounting System Technical Report No. 14, Australian Greenhouse Office, Canberra

McMahon, P., Beauchamp, V. B., Casey, R. E., Salice, C. J., Bucher, K., Marsh, M., Moore, J. 2021. Effects of stream restoration by legacy sediment removal and floodplain reconnection on water quality. Environmental Research Letters, 16(3). https://doi.org/10.1088/1748-9326/abe007

McMillan, S. K., Noe, G. B. 2017. Increasing floodplain connectivity through urban stream restoration increases nutrient and sediment retention. Ecological Engineering, 108(March), 284–295. https://doi.org/10.1016/j.ecoleng.2017.08.006

Melillo, J. M., J. D. Aber, A. E. Linkins, A. Ricca, B. Fry, K. J. Nadelhoffer. 1989. Carbon and nitrogen dynamics along the decay continuum: Plant litter to soil organic matter. Plant Soil 115:189-198.

Nelson, K. C., M. A. Palmer, J. E. Pizzuto, G. E. Moglen, P. L. Angermeier, R. H. Hilderbrand, M. Dettinger, Hayhoe, K. 2009. Forecasting the combined effects of urbanization and climate change on stream ecosystems: from impacts to management options. Journal of Applied Ecology 46:154–163.

Noe, G. B., Hopkins, K. G., Claggett, P. R., Schenk, E. R., Metes, M. J., Ahmed, L., et al. (2022). Streambank and floodplain geomorphic change and contribution to watershed material budgets. Environmental Research Letters, 17(6), 064015. https://doi.org/10.1088/1748-9326/ac6e47

Noe, G.B., M.J. Cashman, K. Skalak, A.C. Gellis, K.G. Hopkins, D. Moyer, J. Webber et al. 2020. "Sediment Dynamics and Implications for Management: State of the Science from Long-Term Research in the Chesapeake Bay Watershed, USA." WIREs Water 7: 28. https://doi.org/10.1002/wat2.1454.

Noe, G.B., Boomer, K., Gillespie, J.L., Hupp, C.R., Martin-Alciati, M., Floro, K., Strano, S., 2019. The effects of restored hydrologic connectivity on floodplain trapping vs. release of phosphorus, nitrogen, and sediment along the Pocomoke River, Maryland USA. Ecol. Eng. 138, 334–352. https://doi.org/10.1016/j.ecoleng.2019.08.002.

Odhiambo, B. K., Ricker, M. C., Le Blanc, L. M., Moxey, K. A. 2016. Effects of forested floodplain soil properties on phosphorous concentrations in two Chesapeake Bay sub-watersheds, Virginia, USA. Retrieved from https://link.springer.com/article/10.1007/s11356-016-6668-3

Orzetti, L. L., Jones, R. C., Murphy, R. F. 2010. Stream condition in piedmont streams with restored riparian buffers in the Chesapeake bay watershed1. Journal of the American Water Resources Association, 46(3), 473–485. https://doi.org/10.1111/j.1752-1688.2009.00414.x

Pander, J., and J. Geist (2013), Ecological indicators for stream restoration success, Ecol. Indicators, 30, 106–118.

Paul, E., Harris, D., Collins, H., Schulthess, U., Robertson, G. 1999. Evolution of CO2 and soil carbon dynamics in biologically managed, row-crop

agroecosystems. Applied Soil Ecology, 11(1), 53-65. doi:10.1016/s0929-1393(98)00130-9
Paul. M.J., Meyer, J.L. 2001. Streams in the Urban Landscape. Annual Review of Ecology Systems 32, 333-365
Pavlides, L. 1990. Geology of part of the northern Virginia Piedmont. Open-File Report OF-90-548, U.S. Geological Survey
Penn, C.J., Camberato, J.J. 2019. A Critical Review on Soil Chemical Processes that Control How Soil pH Affects Phosphorus Availability to Plants. Agriculture 2019, 9, 120; doi:10.3390/agriculture9060120
Pierzynski, G.M. 2000. Methods of phosphorus analysis for soils, sediments, residuals, and waters. Southern Cooperative Series Bulletin #396
Richardson, C. J. 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. Science 228:1424–1426.
Rumps, J. M., Katz, S. L., Barnas, K., Morehead, M. D., Jenkinson, R., Clayton, S. R., Goodwin, P. 2007. Stream restoration in the Pacific Northwest: Analysis of interviews with project managers. Restoration Ecology, 15(3), 506–515. https://doi.org/10.1111/j.1526-100X.2007.00246.x
Shi Y, Lalande R, Hamel C, Ziadi N, Gagnon B, Hu Z. 2013. Seasonal variation of microbial biomass, activity, and community structure in soil under different tillage and phosphorus management practices. Biol Fertil Soils 49:803– 818. https://doi.org/10.1007/s00374-013-0773-y.
Society for Ecological Restoration International Science & Policy Working Group. 2004. The SER International Primer on Ecological Restoration. www.ser.org & Tucson: Society for Ecological Restoration International.
Stranko, S.A., Hilderbrand, R.H. and Palmer, M.A. (2012), Comparing the Fish and Benthic Macroinvertebrate Diversity of Restored Urban Streams to Reference Streams. Restoration Ecology, 20: 747-755. https://doi.org/10.1111/j.1526-100X.2011.00824.x
Surridge, B. W. J., Heathwaite, A. L., Baird, A. J. 2012. Phosphorus mobilisation and transport within a long-restored floodplain wetland. Ecological Engineering, 44, 348–359. https://doi.org/10.1016/j.ecoleng.2012.02.009
Sutfin, N. A., Wohl, E. E., Dwire, K. A. 2015. Banking carbon: A review of organic carbon storage and physical factors influencing retention in floodplains and riparian ecosystems. Earth Surface Processes and Landforms, 41(1), 38-60. doi:10.1002/esp.3857
Tockner, K., Stanford, J.A. 2002. Riverine flood plains: present state and future trends. Environ. Conserv. 29, 308–33
Thompson, J., Pelc, C. E., Brogan, W. R., Jordan, T. E. 2018. The multiscale effects of stream restoration on water quality. Ecological Engineering, 124, 7–18. https://doi. org/10.1016/j.ecoleng.2018.09.016
Trentman, M. T., Tank, J. L., Jones, S. E., McMillan, S. K., Royer, T. V. 2020. Seasonal evaluation of biotic and abiotic factors suggests phosphorus retention in constructed floodplains in three agricultural streams. The Science of the Total Environment, 729, 138744

U.S. Army Corps of Engineers. 2008. Regulatory Guidance Letter No. 08-03: Minimum Monitoring Requirements for Compensatory Mitigation Projects Involving the Restoration, Establishment, and/or Enhancement of Aquatic Resources.

U.S. Environmental Protection Agency. 2010. Chesapeake Bay TMDL Executive Summary. 1–14.

Violin, C. R., Cada, P., Sudduth, E. B., Hassett, B. A., Penrose, D. L., Bernhardt, E. S. 2011. Effects of urbanization and urban stream restoration on the physical and biological structure of stream ecosystems. Ecological Applications, 21(6), 1932–1949. https://doi.org/10.1890/10-1551.1

Walbridge, M. R., & Struthers, J. P. 1993. Phosphorus Retention in Non-Tidal Palustrine Forested Wetlands of the Mid-Atlantic Region. Department of Biology. 13(2), 84-94

Wigginton JD, Lockaby BG, Trettin CC. 2000. Soil organic matter formation and sequestration across a forested floodplain chronosequence. Ecological Engineering 15(Supplement 1): S141–S155. DOI: 10.1016/S0925-8574(99)00080-4

Williams, M.R., Bhatt, G., Filoso, S., Yactayo, G. 2017. Stream Restoration Performance and its Contribution to the Chesapeake Bay TMDL: Challenges Posed by Climate Change in Urban Areas. Estuaries and Coasts 40, 1227-1246

Wolf, K. L., Noe, G. B., Ahn, C. 2013. Hydrologic Connectivity to Streams Increases Nitrogen and Phosphorus Inputs and Cycling in Soils of Created and Natural Floodplain Wetlands. Journal of Environmental Quality, 42(4), 1245–1255. https://doi.org/10.2134/jeq2012.0466

Wood, L. I., Yarwood, S. 2022. Soil Microbial Communities In Urban Stream Restorations And Forested Sites In Fairfax County, VA (thesis, University of Maryland College Park).

Yin, S., Bai, J., Wang, W., Zhang, G., Jia, J., Cui, B., Liu, X. 2019. Effects of soil moisture on carbon mineralization in floodplain wetlands with different flooding frequencies. Journal of Hydrology, 574, 1074-1084. doi:10.1016/j.jhydrol.2019.05.007

Zedler, J. B., Callaway, J. C. 1999. Tracking wetland restoration: Do mitigation sites follow desired trajectories? Restoration Ecology, 7(1), 69–73. https://doi.org/10.1046/j.1526-100x.1999.07108.x

BIOGRAPHY

Katrina Napora received her Bachelor of Arts in Environmental Biology from Christopher Newport University in 2020. After receiving her Masters of Science in Environmental Science and Policy from George Mason University in 2023, she will continue to work for the federal government.