

The effects of restored hydrologic connectivity on floodplain trapping vs. release of phosphorus, nitrogen, and sediment along the Pocomoke River, Maryland USA

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ABSTRACT

River channelization and artificial levees have decreased the hydrologic connectivity of river-floodplain systems around the world. In response, restoration through enhancing connectivity has been advocated to improve the functions of floodplains, but uncertain benefits and the possibility of phosphate release from re-flooded soils has limited implementation. In this study, we measured change in floodplain P, N, and sediment mass balances after restoration along channelized reaches in the lowland Pocomoke River, Maryland USA. Two floodplains (one headwater, one mainstem) restored through partial levee breaches were compared to two additional mainstem floodplains (one natural unchannelized, one unrestored channelized). Potential soluble reactive P (SRP) release from soil cores during experimental laboratory floods; soil P, Fe, and Al fractionation; and deposition and P and N content of sediment were measured before and after the restoration period, as well as in situ inputs and release of SRP and dissolved inorganic N from soils after restorations. Potential SRP release, during both the before and after restoration period, was greatest at the channelized mainstem and restored mainstem sites, lower at the restored headwater site, and small at the natural mainstem site. Both restored sites had smaller potential SRP release after restoration compared to before restoration. In situ SRP release slightly exceeded inputs to soils at connected sites during the post-restoration period, with less net release at the restored sites compared to the natural mainstem site. The magnitude of gross and net SRP release from soils in the field was smaller than, and uncorrelated with, potential SRP release estimated from laboratory experimental floods. Gross soil SRP release rates in the field were predictable using the ratio of soil oxalate-extractable P/Al. Sedimentation inputs of P and N increased at all sites during the post-restoration period, with rates at restored sites intermediate compared to the much higher rates at the natural mainstem site and somewhat lower rates at the channelized mainstem site. These sediment inputs of nutrients were much larger than rates of inorganic P and N release from soils, indicating net trapping of P and N after restoration. Restoring floodplain hydrologic connectivity showed moderate success at increasing the trapping of P, N, and sediment, with relatively little phosphate release, and therefore improving water quality.

1. Introduction

Increased nutrient and sediment loading in rivers due to elevated anthropogenic inputs is a worldwide phenomenon (Meybeck, 1982; Howarth et al., 1996). Increased river loads can have negative impacts on downstream ecosystems by decreasing water clarity (Cloern and

Jassby, 2012) and stimulating algal blooms (Smith et al., 1999), thus altering primary and secondary productivity (Wood and Armitage, 1997) and generating more frequent hypoxia (Conley et al., 2009; Murphy et al., 2011). Floodplains provide important regional water quality benefits by removing sediment and nutrients from river water during floods (Mitsch et al., 2001; Noe and Hupp, 2009; Vidon et al.,

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2010). Extensive channelization and ditching of rivers, combined with artificial berms or other riverbank hardening structures, now limit these benefits by disconnecting river channels from adjacent floodplains (Yarbro et al., 1984; Hupp and Bazemore, 1993; Kroes and Hupp, 2010). Floodplain reconnection is a management practice that has potential to improve downstream water quality by increasing the hydrologic connectivity of degraded rivers (Mitsch et al., 2001; Bernhardt et al., 2005; Craig et al., 2008; Kaushal et al., 2008; Opperman et al., 2009); however, insufficient data on the efficacy and benefits of restoration have limited its implementation.

The water quality functions of floodplain wetlands are highly dependent upon hydrologic connectivity with the river channel (Ward and Stanford, 1995; Hupp et al., 2009). Inputs of nitrogen (N), phosphorus (P), and sediment to the floodplain greatly increase with flood inputs from the river channel and runoff from uplands (Forsberg et al., 1988; Heiler et al., 1995). The act of restoring hydrologic connectivity between channel and floodplain also results in increased P, N, and sediment inputs and trapping by the floodplain (Sheibley et al., 2006; Kronvang et al., 2007; Hoffmann et al., 2009). Biogeochemical cycling rates of those N and P inputs, as well as soil nutrient storage, are also influenced by connectivity through increases in soil moisture content that leads to lowered soil redox (Orr et al., 2007; Noe et al., 2013). For example, rates of N mineralization and denitrification increase with inputs of particulate and dissolved N and wetter soils in floodplains (Forshay and Stanley, 2005; Roley et al., 2012; Wolf et al., 2013; McMillan and Noe, 2017).

The potential for phosphate release, however, from soils of floodplains with restored wetland hydrology is a large concern. The reduced conditions typical of wetlands often are presumed to enhance P mobility, but subtle environmental controls on P cycling complicate efforts to evaluate and predict floodplain water quality benefits (Reddy et al., 1999). Soil phosphate dynamics in floodplains are known to be influenced by many factors, including P inputs, forms of soil P and metal geochemistry, flood dynamics and soil moisture and their influence on microbially-controlled redox conditions, organic decomposition, and plant species composition (Walbridge and Struthers, 1993; Baldwin and Mitchell, 2000; Noe et al., 2013). In particular, the form of P, whether sorbed to minerals and organic matter molecular surfaces (amorphous P) or embedded within a mineral structure (crystalline P) strongly affects P reactivity, bioavailability, and potential transport to downstream water resources (Reddy et al., 1998). Further, the distribution of soil P among available labile and more recalcitrant soil pools is influenced by the source and chemistry of floodwaters (Smolders et al., 2006). Several studies have evaluated the potential for phosphate release from restored wetland soils after reintroduction of flooding by using experimental inundation of soil cores, and identified the role of metal and P geochemical fractionation, organic matter quality, and microbial dynamics (Aldous et al., 2007; Loeb et al., 2008; Schönbrunner et al., 2012; Surridge et al., 2012).

Phosphate P release rates and availability are expected to be greater and last longer in landscapes with a history of anthropogenic P inputs and an accumulation of P-enriched soils (Reddy et al., 2005; Sharpley et al., 2013). Enhanced phosphate release likely continues after hydrologic restoration until microbial processing exhausts reducible forms of P fractions, such as phosphate sorbed to iron hydroxides, that are exported downstream as dissolved P or is transferred to long-term biologically unavailable soil sinks such as carbonate precipitates or recalcitrant soil organic matter. To date, measurement of actual phosphate release after floodplain hydrologic reconnection in a landscape context has been limited (Ardón et al., 2010).

Globally, floodplain habitat has been greatly reduced through river channelization, artificial levees, and conversion to agriculture and urbanization (Tockner and Stanford, 2002). Floodplain reconnections potentially provide one of the most effective opportunities to enhance in-stream water quality and habitat conditions (Bayley, 1985; Opperman et al., 2009), but only a relatively small fraction of funds

spent for river restoration projects in the U.S. has been directed at floodplain reconnections (Bernhardt et al., 2005). In part, insufficient measurement of the benefits of floodplain restoration may limit its implementation, and there remains a critical need to quantify long-term nutrient and sediment storage benefits of reconnected floodplains. Identification of the predictors of phosphate release could provide a tool for restoration managers to screen potential restoration sites for their likelihood to release phosphate. Furthermore, a mass balance comparison of both P release as well as P inputs after restoration would provide a more holistic perspective on the net impacts of floodplain restoration on water quality.

Our goal was to measure changes in P input and release from floodplain soil due to restoration. Because water quality concerns include excess N and sediment, in addition to P, and because these three constituents are inextricably related, we also investigated N and sediment exchange. We used a combination of field measurements and laboratory soil core experiments to evaluate nutrient fluxes before and after floodplains were reconnected to the adjacent river and to compare results with naturally connected and channelized floodplains. We expected that reconnection of floodplain would 1) increase the number of flood pulses and hydroperiod; and 2) increase deposition of P-enriched sediments and dissolved-P inputs; but also 3) potentially increase P release from wetland soils due to greater flood durations. However, we also expected that 4) rates of P inputs would exceed rates of soil P release, resulting in a net retention of P after restoration. Finally, we expected that 5) sediment P fluxes are predictable based on soil geochemical characteristics that are easier to measure, thus potentially providing a simpler indicator of potential water quality function. Accordingly, we measured floodplain inundation, sediment deposition, N and P inputs to soils, N and P release from soils, and change in mass balance (inputs minus outputs) of soil N and P, and geochemical controls of phosphate release in channelized, restored, and naturally connected floodplains of a lowland river system.

2. Methods

2.1. Location

We compared restored, channelized, and natural floodplain along the Pocomoke River system, located on the eastern shore of Maryland, USA, in the Coastal Plain physiographic province (Fig. 1). The flat topography of the Pocomoke River watershed (1300 km²) results in a very low river gradient (0.15 m/km) and formation of floodplains with broad back-swamp areas along the mainstem and its major tributaries. Surficial deposits consist mostly of sandy, organic-rich sediments, and surface waters tend to be dilute with low pH (less than 6; Hamilton et al., 1991). Inundation of naturally connected mainstem floodplains of the Pocomoke River typically occurred more than 255 days/year historically (Kroes and Hupp, 2010). The upper Pocomoke River watershed area (157 km²) has an average annual discharge of 2.1 m³ s⁻¹ at the U.S. Geological Survey (USGS) stream gage at Willards, Maryland (1951 to 2016, USGS gage 01485000) and an average annual precipitation of 1096 mm (NOAA Climate Data: MD CD1) evenly distributed throughout the year.

Agriculture accounts for about 45% of land use in the watershed above the USGS stream gage (Ator et al., 2004), largely comprised of row crops with a corn and soybean rotation and a high concentration of commercial poultry houses. Extensive ditching of wetlands and river channelization increased substantially during the 20th century to support agricultural and lumber industries by enhancing drainage. As a result, stream length of the Pocomoke River system expanded nearly 2000 km (Bricker et al., 2003). Channelization along the mainstem increased channel bed depth and cross-sectional area, which further increased watershed drainage during storm events and reduced floodplain inundation (Kroes and Hupp, 2010). Spoil bank levees, material excavated from the channel and deposited parallel to the stream, nearly

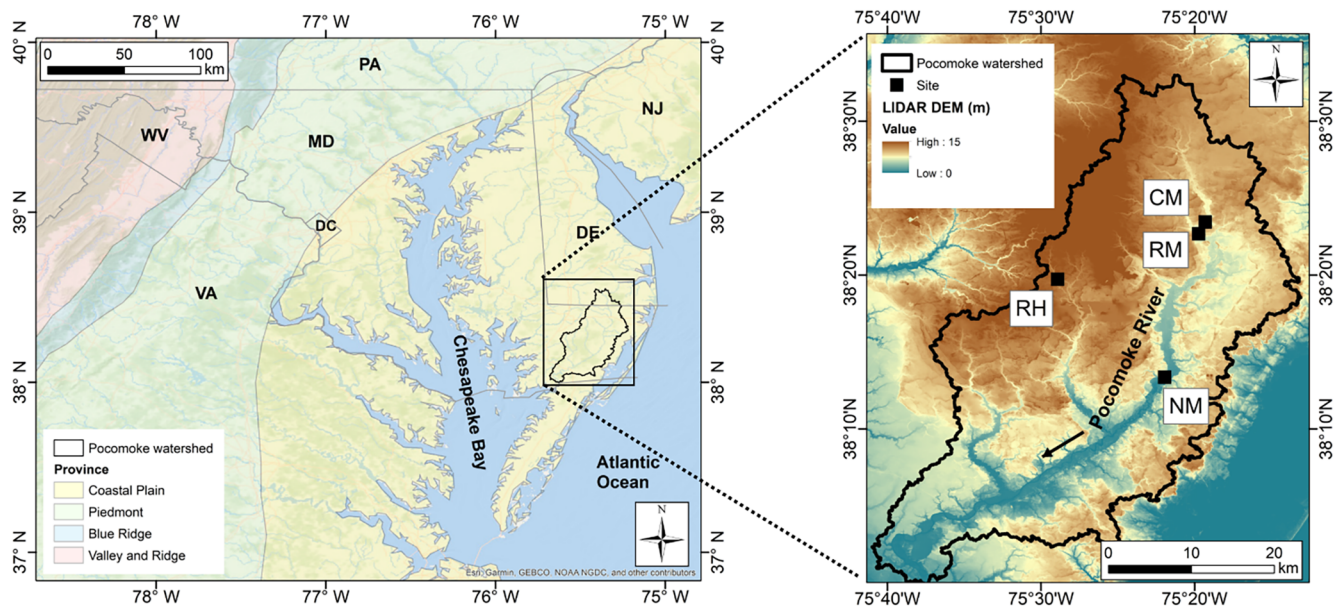


Fig. 1. Map of the Chesapeake Bay along the mid-Atlantic coast of the United States with the location of the Pocomoke River watershed (left) and an inset of the Pocomoke River watershed showing locations of floodplain study sites (right). CM = channelized mainstem, RH = restored headwater, RM = restored mainstem, and NM = natural mainstem.

eliminated hydrologic connectivity and nutrient and sediment inputs from the channelized river floodplains (Noe and Hupp, 2005).

We compared four floodplain sites along the Pocomoke River system (Fig. 1, Table 1): channelized mainstem (CM, a control site) that was not restored, located at the USGS stream gage; restored headwater (RH) on a tributary to the Pocomoke River; restored mainstem (RM) just downstream from the gage; and natural mainstem (NM, a control site) on the lower Pocomoke River, much further downstream from the gage. Channelization ends roughly five km upstream of NM. The river channel at NM is microtidal but the floodplain is not inundated by tides; tidally inundated floodplain currently occurs several km downstream of NM before the river discharges to the Chesapeake Bay. Each of the four sites were forested floodplain without known agricultural land use prior to restoration.

2.2. Restoration

Sites RH and RM were restored using a partial levee breaching technique to restore hydrologic connectivity to the floodplains during river flooding events. Breaches 10–45 m in length were excavated in levees to a depth of 0.15 m above the surface of adjacent floodplain. Preliminary hydrodynamic modeling of the breaches indicated river flow would connect to floodplain during flood events with an annual occurrence probability of between 1 in 2 years to 10 years. RH had 4 breaches installed (Fig. 2, Appendix Photo 1), and RM had 5 breaches installed (Fig. 2, Appendix Photo 2). Due to scheduling constraints,

construction of breaches was completed one year apart, in December 2014 at RH and September 2015 at RM (Fig. 3).

2.3. Site layout

Each site was instrumented with coordinated networks of water table wells, sedimentation plots, and integration plots that spanned the spatial extent of each floodplain. The plots sampled gradients of elevation and geomorphic features within the site, laterally from levees and proposed breach locations to the back swamp or edge of active floodplain (as delineated by changes in vegetation and slope; Fig. 2). Each sedimentation plot had an artificial marker horizon of feldspar clay. The integration plots refer to a subset of six sedimentation marker locations in each site where additional nutrient and sediment fluxes, soil physico-chemical characteristics, and assays of soil nutrient release potential were measured.

2.4. Floodplain hydrology (Pre- and post-restoration)

At each site, six to twelve water table wells were installed in a triangular configuration across the floodplain complex (Fig. 2). Wells were constructed of 3.2 cm PVC pipe, fully slotted (0.25 cm slits) from 5 cm above the land surface to the underlying aquitard or to the maximum depth penetrable by hand. Well depths ranged from 0.5 to 2 m below the land surface. Continuous pressure transducer loggers were deployed to record hourly readings. Recordings were converted to

Table 1

Pocomoke River floodplain site characteristics. “Watershed” includes the entire area of watershed upstream from the floodplain site, “Local contributing area” includes only immediately adjacent uplands that directly drain to the floodplain site.

Site	Floodplain Area (km ²)	Length of Stream Reach (km)	Drainage area (km ²): Watershed/ Local contributing area	Agriculture land use (%): Watershed/Local contributing area	Forest land use (%): Watershed/ Local contributing area
Channelized mainstem	0.2	0.5	152/1	41/17	52/51
Restored headwater	0.2	1.3	7/0.6	48/0	32/100
Restored mainstem	0.9	1.4	213/4.3	41/33	49/63
Natural mainstem	0.1	0.3	505/0.2	37/100	54/0

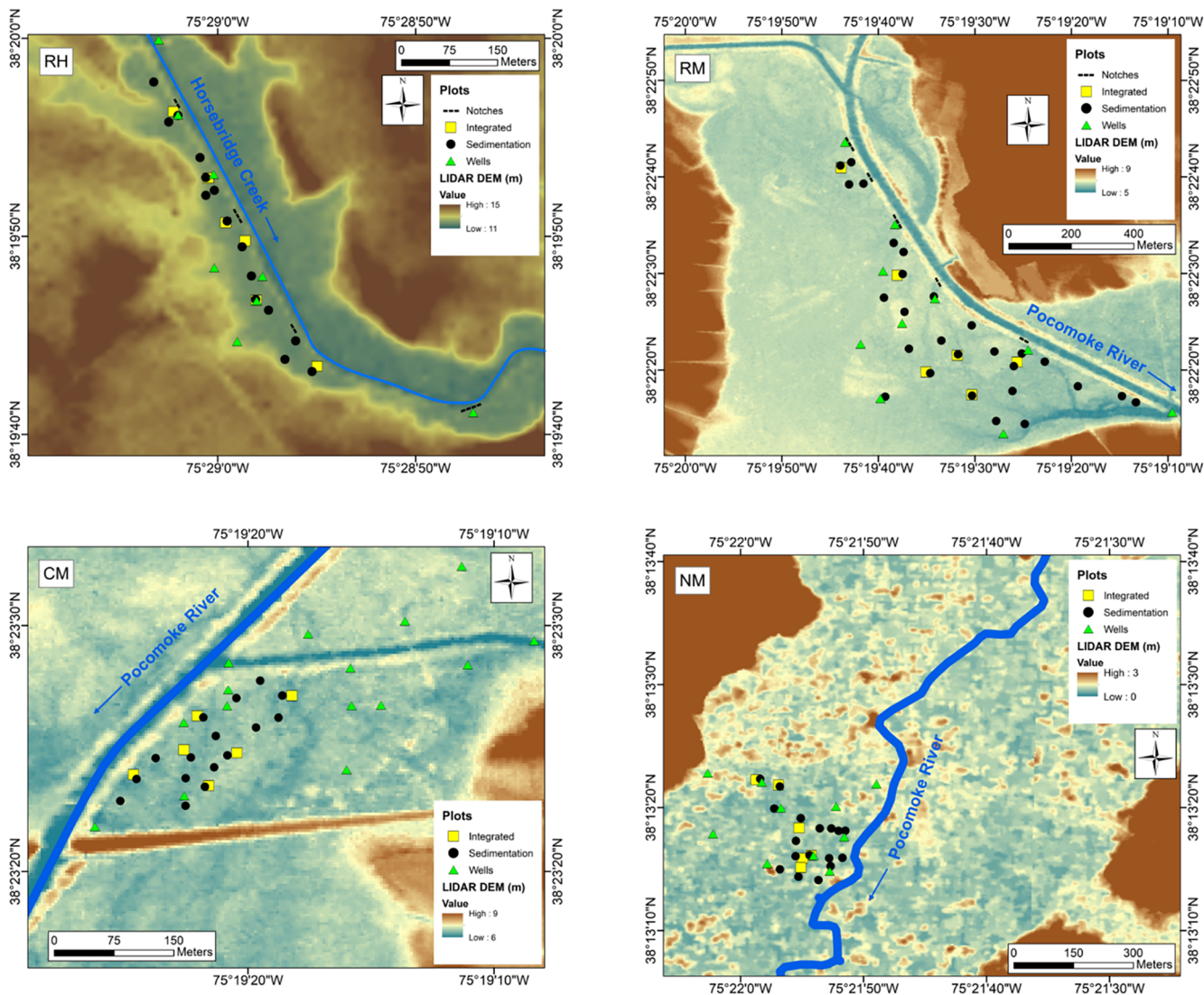


Fig. 2. Map of floodplain study sites, showing location of sedimentation and integration plots, hydrology wells, and levee breaches (notches) at restored sites. CM = channelized mainstem, RH = restored headwater, RM = restored mainstem, and NM = natural mainstem.

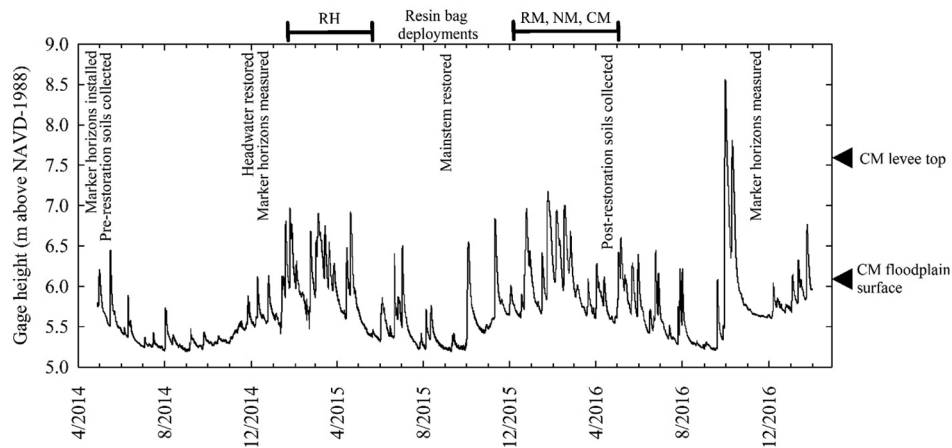


Fig. 3. Pocomoke river stage (m above NAVD 1988) recorded at the USGS Willards gage (#01485000) during the study, with timing of sampling and restorations noted. The adjacent CM floodplain surface elevation is 6.1 m and the top of spoil levee is approximately 7.6 m.

absolute water levels after correcting for atmospheric air pressure and accounting for casing heights surveyed to a common benchmark (m above mean sea level). Water level measurements are within 2 cm accuracy. Transformed water level data were compared to the USGS Willards gage data and to precipitation data collected on-site or measured within three-mile radius from the site to evaluate the response of each wetland system to changes in river stage and precipitation events (NASA Global Precipitation Mission Wallops-PRF Station ID NASA006).

2.5. Soil P release and physico-chemical characteristics (Pre- and post-restoration)

To evaluate the effects of floodplain reconnection on the potential for soil chemistry changes and P release during flooding, soil cores were collected from each of the six integration plots within each site prior to restoration, in April 2014, and post-restoration, in April 2016. Two 10 cm deep soil cores were taken each time with a 15 cm × 7.6 cm diameter PVC pipe, one for determining bulk density and one for physico-chemical analyses of soil characteristics and P and metal fractions that could likely explain rates of phosphate release from soils. An additional 10 cm deep soil core was collected each time using a 30 cm × 7.6 cm diameter PVC pipe for short-term flood experiments to contrast potential short-term release of phosphorus from soils upon reflooding between sites. Sediment cores were retained only if compaction was less than 1 cm, otherwise the plot was re-cored.

2.5.1. Soil physico-chemistry

Soil bulk density, water content, organic content, particle size, pH, and total elemental content were analyzed from soil cores. Soil bulk density was estimated using total wet mass of soil, soil moisture content, and dimension of the soil core. Soil moisture was calculated by drying a soil subsample to a constant weight in a drying oven at 60 °C; this temperature was chosen to minimize organic matter and N volatilization losses at higher temperatures (Mayland, 1968). Water filled pore-space (WFPS) was calculated from gravimetric moisture content, bulk density, and an assumed particle density of 2.65 Mg m⁻³ (Robertson et al., 1999). A roughly 50 g dry-weight equivalent subsample of the core was dried, ground, and sieved to < 1 mm for subsequent analysis of organic content, soil particle size, total nitrogen (TN) and total carbon (TC), and total phosphorus (TP) and total bulk metal concentrations. Organic content of an approximately 4 g soil subsample was measured by loss on ignition at 550 °C for 4 h (Heiri et al., 2001). Soil particle size of mineral matter in the combusted subsample was measured using a laser diffraction particle size analyzer following dispersion with sodium hexametaphosphate and sonication (Sperazza et al., 2004; LS 13 320, Beckman Coulter, Brea, CA, USA). Metal content, including total P, Al, and Fe, was determined using microwave acid digestion (Sandroni and Smith, 2002; using HF and HNO₃; Discover SPD, CEM Corporation, Matthews, NC, USA) and analysis on an ICP-OES (Optima 4300DV, Perkin-Elmer, Waltham, MA, USA). Total N and total C were measured using a CHNS analyzer (Yeomans and Bremner, 1991; Flash 2000, Thermo Scientific, Waltham, MA, USA). Soil pH was determined from a slurry mixture of 5 g of air-dried soil and 10 ml of deionized water (Robertson et al., 1999; pHi 460, Beckman Coulter, Brea, CA, USA).

2.5.2. Soil P and metal chemistry

Soil P pools of differing potential reactivity were operationally determined using a combination of sequential fractionation schemes developed by Pacini and Gächter (1999) and by Darke and Walbridge (1994; see Appendix Fig. 1). A subsample of approximately 1 g-dry weight equivalent of homogenized soil from the second core collected from each integration plot was extracted sequentially with the following reagents: 1) NH₄Cl to determine bioavailable P; 2) DCB (sodium dithionite, sodium citrate and sodium bicarbonate) to measure redox-sensitive, labile soil P; 3) NaOH to measure less mobile Al-OM soil P and

humic/microbial P; and 4) HCl to measure more recalcitrant soil carbonate P pools and recalcitrant organic P. Resulting supernatant mixtures were analyzed for both soluble reactive P (SRP) and soluble non-reactive P (NRP) concentrations (see below).

A second fractionation scheme was used to characterize the size and reactivities of soil Al and Fe pools (Darke and Walbridge, 1994; U.S. Department of Agriculture 1996; see Appendix Fig. 1). Soil subsamples were extracted with the following reagents: 1) ammonium oxalate (ammonox) to characterize amorphous (more reactive) Al and Fe soil pools, followed by 2) NaOH to measure less reactive crystalline Al (0.4 g-dry weight equivalent soil subsamples); 3) DCB to measure redox-sensitive amorphous and crystalline Fe (0.8 g-dry weight equivalent soil); and 4) pyrophosphate to measure organic Fe and Al pools (0.25 g-dry weight equivalent soil). Crystalline Fe was calculated as the difference between DCB-Fe and ammonox-Fe.

Reagents were added to 50 ml centrifuge tubes with soil subsamples, vortexed for 5–10 s, and mixed with a tube rotator at ambient temperature for 0.5 h (NaOH following ammonium oxalate), 1 h (NH₄Cl, DCB), 4 h (ammonium oxalate), or 16 h (NaOH, HCl, DCB-metals, pyrophosphate) hours. Tubes were then centrifuged for 10 min and the resulting supernatant was poured into a 60 ml syringe and filtered (1.2 µm GD/X glass microfiber filter; the larger porosity was necessary to avoid clogging) for SRP, NRP, and metals analyses. Half of the centrifuged supernatant from sequential P extracts was reserved for total dissolved P (TDP) analysis. Concentrations of SRP and TDP (alkaline persulfate digestion in an autoclave for 1 h; Patton and Kryskall, 2003) in sequential P extracts were analyzed on a discrete analyzer along with reagent blanks (Seal AQ2, SEAL Analytical Inc., Mequon, Wisconsin, USA). NRP was calculated as TDP minus SRP. The soil that remained after sequential phosphorus fractionation extractions was dried, ground, and digested using microwave assisted strong acid digestion (HF and HNO₃; Discover SPD, CEM Corporation, USA) to measure the residual P fraction. Each metals extract and the residual P digest were analyzed for total P, Fe, and Al on an ICP-OES (Optima 4300DV, Perkin-Elmer, Waltham, Massachusetts, USA).

2.6. Short-term flooding experiments

We used controlled laboratory microcosm experiments to compare the effects of inundation on P release from soils of restored, natural, and channelized floodplains, independently of variability in weather, flood hydrology, and flood chemistry. Potential phosphate release from soil cores was measured using experimental floods of river water in a laboratory aquarium following Dunne et al. (2010). Flood experiments were conducted twice, with soil cores and river water collected in April 2014 (pre-restoration) and again in May 2016 (post-restoration). Soil cores used for each flood experiment were capped and sealed on the bottom with PVC caps with cement and then stored upright to prevent infiltration of water from outside the core. Soils were kept cool after collection and during transport to the laboratory where they were stored at 4 °C until processing began the next day. For each experiment, approximately 30 L of Pocumoke River surface water was collected adjacent to the CM site and filtered (0.45 µm Supor polyethersulfone, Pall Life Sciences, Port Washington, New York, USA; 0.2 µm filters clogged rapidly).

For each flood experiment, soil cores were placed in an opaque aquarium (dimensions: 46 × 63 cm, 63 cm deep) with flow-around tap water of constant temperature (mean_{pre}: 21.0 °C, mean_{post}: 18.9 °C) maintained at a constant water level roughly 1 cm below the open tops of cores (Appendix Photo 3). Cores were arranged in random order in a 4 × 6 grid, and filtered river water was added inside each core tube until the soil core was saturated and then flooded to a depth of 15 cm. Three blanks of filtered river water with no soil were kept inside 50 ml centrifuge tubes and placed in the aquarium tank for the duration of the experiment. The aquarium was fitted with a loose opaque lid between sampling events to limit light and algal growth in the cores.

Surface water in flooded cores and blanks were sampled for SRP concentration at 1 h, and after 1, 2, 3, 4, 5, 7, 14, and 21 days after initiating the experiment. Samples were collected with a 3 ml syringe following gentle mixing of the water column, filtered (0.2 µm Supor polyethersulfone, Pall Life Sciences, Port Washington, New York, USA) into a polypropylene scintillation vial, and immediately stored at 4 °C. Additional samples of filtered river water from the start of the experiment and filtered surface water in flooded soil cores and blanks on day 21 were stored at –20 °C until dissolved NRP measurement could be completed. All surface water samples collected for SRP analysis were analyzed on the day of collection. NRP samples were thawed, digested (TDP; [Patton and Kryskall, 2003](#)), stored at 4 °C, and analyzed for SRP within one week of digest. All SRP and NRP analyses were completed using a discrete analyzer (Seal AQ2, SEAL Analytical Inc., Mequon, Wisconsin, USA). SRP and NRP fluxes in each soil core were calculated using a regression of mass of P released over time, corrected for volumes of water removed by sampling over the experiment and for river-water blank concentrations.

2.7. Field flux measurements

2.7.1. Sedimentation

A network of 15 to 27 feldspar clay artificial marker horizons (dimensions: 30 cm × 30 cm, 1 cm depth; [Hupp and Bazemore, 1993](#)) were installed at each site ([Fig. 2](#)) in April 2014 prior to scheduled floodplain restorations ([Fig. 3](#)). Net vertical accretion of sediment above each marker horizon was measured pre-restoration in December 2014 (length of deployment approximately 7 months) and post-restoration in November 2016 (length of deployment approximately 2.5 years) by slicing out a portion of the marker horizon and deposited sediment and measuring accumulation of sediment above the feldspar. Unconsolidated leaf litter without attached sediment, roots, and woody debris was removed from the top of the marker horizon prior to measurement of accretion or coring. In addition, sediment deposited above the marker horizon at each integration plot was cored in November 2016 using a 7.6 cm diameter PVC pipe. The sediment core was extruded onto a dry, clean cutting board in the field. Sediment above the feldspar marker horizon was then separated from the core, collected, and stored at ambient temperature until returned to the laboratory, stored at 4 °C, and then dried at 60 °C, weighed, ground and sieved < 1 mm, and analyzed for TP and TN. Vertical accretion rates were calculated over two time periods: pre-restoration (April to December 2014) and post-restoration (December 2014 to November 2016). The December 2014 sampling occurred immediately after restoration at RH but prior to any flooding.

2.7.2. Inorganic nutrient flux

Within each integration plot, vertically stacked resin bags were deployed to measure dissolved inorganic P and N inputs to, and release from, soils in situ (design modified from [McMillan and Noe, 2017](#)). Upper resin bags trapped SRP, NH_4^+ , NO_2^- , and NO_3^- inputs to the soil from surface water and atmospheric deposition; lower bags trapped the same ions released upwards from the soil. Each resin bag was constructed with 20 g-ww mixed-bed ion exchange resin beads (Rexyn I-300, Fisher Scientific, Pittsburgh, Pennsylvania) in acid washed, undyed nylon stocking with a nitrile rubber O-ring, and closed at both ends with a stainless-steel staple. To install resin bags, a 5 cm long section of 7.8 cm diameter thin-walled PVC pipe was inserted 4 cm deep into the soil, leaving 1 cm of pipe above the soil surface. Two stacked, labelled resin bags were placed inside the PVC pipe, above the soil, and each sealed in place with silicone caulk free of antimicrobial agents. Deployment was approximately 30 days for each stacked resin bag set; bags were retrieved and replaced following each deployment. RH had 5 months of measurement over the flood season from January to May 2015, while CM, RM, and NM had 6 months of measurement from December 2015 to May 2016. Upon retrieval, each resin bag was placed

in a sealed polyethylene bag, kept cool, and transported to or shipped overnight to the laboratory. If shipped, samples were immediately extracted upon receipt; if transported directly from the field, samples were stored at 4 °C overnight until extraction the following day. Approximately 2 g-ww of resin was subsampled from each bag and placed in a 50 ml centrifuge tube, 40 ml of 2 M KCl was added to each centrifuge tube, and tubes were vortexed for 10 s and then agitated on a shaker table for 1 h at 250 RPM. The supernatant was poured off into a 60 ml syringe and filtered (0.2 µm Supor polyethersulfone, Pall Life Sciences, Port Washington, New York, USA). Filtered extractant and 2 M KCl blanks were analyzed for SRP, NO_3^- , NO_2^- , and NH_4^+ on a discrete analyzer (Seal AQ2, SEAL Analytical Inc., Mequon, Wisconsin, USA).

2.8. Statistics

Differences among sites and between pre- vs. post-restoration periods were tested using non-parametric Kruskal-Wallis tests. When appropriate, non-parametric post-hoc comparisons of individual sites were tested using Mann-Whitney tests. The ability of soil physico-chemical characteristics to explain spatial variation in pre-restoration potential SRP and NRP release rates (flood experiments) and in post-restoration measured field SRP release rates (lower resin bags) were evaluated using stepwise linear multiple regressions. All pre-restoration soil characteristics were added to stepwise regressions, as well as ratios of amnox-P to amnox-Al, amnox-Fe, and amnox-Al + Fe. All tests were evaluated at alpha = 0.05.

3. Results

3.1. Hydrologic response

The study was complicated by rather extreme weather conditions throughout the study period. In the first water year (2014, October 2013 to September 2014), eight months fell below the 25th percentile of long-term monthly precipitation records; whereas in 2016, six months exceeded the 75th percentile. Further, the second largest flood event recorded at the Pocomoke River gage occurred on September 30, 2016, after restorations were completed. Following a large rain event of 18.80 cm in less than 48 h, river discharge peaked at 69.1 cubic meters per second, resulting in stage heights that overtopped most spoil levees along the channelized Pocomoke River, including CM ([Fig. 3](#)). Based on long term records of precipitation and river discharge, the likelihood of the storm was classified as a 200–300 year return interval ([Ries and Dillow, 2006](#)).

Across all sites, floodplain water table elevations increased 25 to 40 cm from water year 2014 to 2016, reflecting increasing precipitation

Table 2
Hydrology of each floodplain site over time.

Site	Water Year (Oct 1–Sept 30)	Mean Flood Depth (cm)	Number of Flood Events	Percent of Time Flooded
Channelized mainstem	2014	< 0	3	0
	2015	24	7	9
	2016	32	5	13
Restored headwater	2014	7	3	4
	2015	6	2	0
	2016	10	6	1
Restored mainstem	2014	< 0	0	0
	2015	10	1	4
	2016	20	1	8
Natural mainstem	2014	4	2	5
	2015	24	7	39
	2016	23	8	67

Table 3
Release of soluble reactive phosphorus (SRP) and soluble non-reactive phosphorus (NRP) from floodplain soils of each site to river water during experimental floods of pre- and post-restoration soils. Mean $\pm$ one standard error of the six integration plots in each site.

Site	Pre-restoration SRP flux (mg-P m ⁻² d ⁻¹)	Post-restoration SRP flux (mg-P m ⁻² d ⁻¹)	Change in SRP flux (mg-P m ⁻² d ⁻¹)	Pre-restoration NRP flux (mg-P m ⁻² d ⁻¹)	Post-restoration NRP flux (mg-P m ⁻² d ⁻¹)	Change in NRP flux (mg-P m ⁻² d ⁻¹)
Channelized mainstem	5.0 $\pm$ 2.2	5.8 $\pm$ 2.7	0.76 $\pm$ 1.4	0.47 $\pm$ 0.27	2.8 $\pm$ 1.2	2.3 $\pm$ 0.9
Restored headwater	1.2 $\pm$ 0.8	0.62 $\pm$ 0.46	-0.58 $\pm$ 1.0	0.22 $\pm$ 0.09	0.31 $\pm$ 0.34	0.090 $\pm$ 0.3
Restored mainstem	4.7 $\pm$ 1.6	3.1 $\pm$ 0.6	-1.6 $\pm$ 1.3	0.67 $\pm$ 0.33	1.8 $\pm$ 0.3	1.2 $\pm$ 0.3
Natural mainstem	0.19 $\pm$ 0.14	0.07 $\pm$ 0.05	-0.12 $\pm$ 0.13	-0.021 $\pm$ 0.025	0.14 $\pm$ 0.04	-0.12 $\pm$ 0.05

during the three-year study. The frequency and magnitude of flooding evident in the floodplain water level record also increased but did not show a consistent pattern across sites in relation to floodplain condition (Table 2). CM flooded almost as frequently as NM. However, the water table in CM fluctuated more at the upland-wetland edge rather than adjacent to the river as observed at NM, indicating local water contributions rather than riverine inputs strongly influenced CM floodplain hydroperiod. Although the water table increased at RH, flooding patterns did not change significantly before and after restoration in December 2014 and flood frequency remained less at RH than observed at NM or CM. The RM site had more changes in hydrology following restoration in September 2015 with deeper water over a longer duration of flooding compared to before restoration, but with no detectable change in flood frequency. To summarize, annual metrics of floodplain inundation show inconsistent patterns in response to floodplain restoration, largely due to changing precipitation and an extremely large flood during the post-restoration period that inundated all sites. Importantly, these hydrologic statistics do not directly evaluate for changes in water sources to the floodplain and whether channel-floodplain connectivity increased after restoration in contrast to changes in local water sources associated with greater precipitation and upland runoff into floodplains.

3.2. Pre-restoration potential P release and soil characteristics

3.2.1. Potential P release

Soil cores collected during the pre-restoration period, and experimentally flooded with river water for 21 days in the laboratory (potential release), released SRP from soil to water at flux rates ranging from averages of 0.2 to 5.0 mg-P m⁻² y⁻¹ among sites (Table 3, Appendix Fig. 2). Differences in these potential SRP release among sites were not significant (Kruskal-Wallis test, $p = 0.078$), but average potential pre-restoration SRP flux was lowest at NM and (to be restored) RH and highest at CM and (to be restored) RM (Table 3). Potential NRP release fluxes significantly differed among sites ($p = 0.025$), with uptake by soil at NM that was statistically less than the rates of NRP release measured elsewhere. Rates of NRP release were intermediate at (to be restored) RH, but not statistically different than CM and (to be restored) RM.

3.2.2. Soil characteristics

Soils everywhere were wet (site averages > 74% WFPS), organic (> 28%), and acidic (pH < 4.9) before the restoration period; soils at NM were significantly wetter (Kruskal-Wallis test, $p = 0.003$) but also less acidic ($p = 0.008$) than other sites (Table 4). Soils everywhere were also generally fine textured and moderately P-enriched. Elemental TP, TC, and TN content, bulk density, and particle size did not differ among sites. Results of sequential P extractions of soils indicated labile reactive (NH₄Cl-SRP) and labile non-reactive (NH₄Cl-NRP) P were larger at CM (Kruskal-Wallis test, $p = 0.038$ and 0.032, respectively), whereas redox-sensitive reactive Fe and Mn (DCB-SRP) P fraction was largest at NM ($p = 0.018$). The non-reactive microbial, humic, organic P pool (NaOH-NRP) was larger at (to be restored) RM ($p = 0.020$). Finally, carbonate reactive (HCl-SRP) P was smaller at (to be restored) RH ($p = 0.002$). Generally, Al- and Fe-oxide (NaOH-SRP and DCB-SRP extractable, respectively) P fractions and residual organic, refractory P were the largest P fractions (Fig. 4). Among metal extractions, soil total Al and crystalline (ammonox + NaOH) Al were greater at (to be restored) RH ($p = 0.026$ and $p = 0.013$, respectively), and organic (pyrophosphate) Al and amorphous plus crystalline (DCB) Al were greater at CM ($p = 0.031$ and 0.011, respectively). Total Fe, organic (pyrophosphate) Fe, and amorphous plus crystalline (DCB-extractable) Fe and P were greatest at NM ($p = 0.014$, 0.049, 0.046, and 0.041, respectively). Amorphous, non-crystalline (ammonox) P concentrations were smaller at both (to be restored) RH and RM ($p = 0.042$; Table 4).

Table 4Soil physico-chemistry of each site measured pre- and post-restoration. Mean $\pm$ one standard error of six soil cores collected in each site.

	Channelized mainstem	Restored headwater	Restored mainstem	Natural mainstem
PRE-RESTORATION				
WFPS (%)	73.7 $\pm$ 5.35	75.4 $\pm$ 4.92	83.2 $\pm$ 4.27	97.1 $\pm$ 1.41
Bulk density (g cm ⁻³)	0.363 $\pm$ 0.0370	0.333 $\pm$ 0.0150	0.513 $\pm$ 0.0655	0.427 $\pm$ 0.0535
Organic content (%)	46.4 $\pm$ 8.99	40.6 $\pm$ 1.59	33.1 $\pm$ 6.31	27.8 $\pm$ 3.75
pH	4.28 $\pm$ 0.2	4.35 $\pm$ 0.1	4.53 $\pm$ 0.0	4.89 $\pm$ 0.1
Mean particle size (μ m)	71 $\pm$ 5	97 $\pm$ 9	102 $\pm$ 4	115 $\pm$ 19
Clay and silt (< 63 μ m, %)	63.9 $\pm$ 4.48	61.8 $\pm$ 2.72	49.3 $\pm$ 6.44	52.7 $\pm$ 4.40
Total N (%)	1.18 $\pm$ 0.113	1.12 $\pm$ 0.0367	0.943 $\pm$ 0.146	0.795 $\pm$ 0.132
Total C (%)	20.6 $\pm$ 2.48	18.3 $\pm$ 0.556	16.9 $\pm$ 2.38	12.6 $\pm$ 2.33
Total P (mg g ⁻¹)	1.60 $\pm$ 0.328	0.963 $\pm$ 0.131	1.03 $\pm$ 0.201	1.51 $\pm$ 0.307
Total Al (mg g ⁻¹)	18.6 $\pm$ 2.79	34.9 $\pm$ 5.09	15.9 $\pm$ 2.71	29.7 $\pm$ 4.74
Total Fe (mg g ⁻¹)	7.10 $\pm$ 1.18	9.69 $\pm$ 1.29	7.27 $\pm$ 1.35	23.0 $\pm$ 4.41
Ammon-P (mg g ⁻¹)	1.05 $\pm$ 0.23	0.45 $\pm$ 0.05	0.67 $\pm$ 0.07	1.18 $\pm$ 0.28
Ammon-Al (mg g ⁻¹)	8.00 $\pm$ 2.81	4.50 $\pm$ 0.23	3.10 $\pm$ 0.43	3.06 $\pm$ 0.41
Ammon-Fe (mg g ⁻¹)	3.66 $\pm$ 0.52	3.37 $\pm$ 0.23	4.56 $\pm$ 0.80	17.37 $\pm$ 3.65
POST-RESTORATION				
WFPS (%)	66.6 $\pm$ 2.88	79.9 $\pm$ 4.77	73.2 $\pm$ 4.28	100 $\pm$ 3.14
Bulk density (g cm ⁻³)	0.377 $\pm$ 0.0312	0.364 $\pm$ 0.0283	0.511 $\pm$ 0.0453	0.549 $\pm$ 0.0967
Organic content (%)	39.2 $\pm$ 5.61	39.1 $\pm$ 2.11	32.0 $\pm$ 4.61	26.8 $\pm$ 5.53
pH	6.09 $\pm$ 0.4	5.65 $\pm$ 0.4	4.78 $\pm$ 0.0	5.42 $\pm$ 0.1
Mean particle size (μ m)	57 $\pm$ 5	67 $\pm$ 7	92 $\pm$ 13	68 $\pm$ 10
Clay and silt (< 63 μ m, %)	68.2 $\pm$ 3.44	66.1 $\pm$ 2.42	51.0 $\pm$ 5.66	64.0 $\pm$ 4.04
Total N (%)	1.07 $\pm$ 0.129	1.11 $\pm$ 0.0707	0.877 $\pm$ 0.115	0.788 $\pm$ 0.120
Total C (%)	19.2 $\pm$ 3.04	17.7 $\pm$ 1.06	16.4 $\pm$ 2.24	12.4 $\pm$ 2.59
Total P (mg g ⁻¹)	1.82 $\pm$ 0.338	0.941 $\pm$ 0.0449	1.12 $\pm$ 0.0989	1.60 $\pm$ 0.311
Total Al (mg g ⁻¹)	29.7 $\pm$ 5.31	44.3 $\pm$ 6.63	20.8 $\pm$ 1.39	42.9 $\pm$ 3.90
Total Fe (mg g ⁻¹)	7.91 $\pm$ 1.14	9.71 $\pm$ 1.03	7.76 $\pm$ 0.834	23.8 $\pm$ 4.11
Ammon-P (mg g ⁻¹)	1.30 $\pm$ 0.31	0.40 $\pm$ 0.04	0.71 $\pm$ 0.08	0.94 $\pm$ 0.21
Ammon-Al (mg g ⁻¹)	12.82 $\pm$ 4.44	4.99 $\pm$ 0.32	3.45 $\pm$ 0.44	3.11 $\pm$ 0.42
Ammon-Fe (mg g ⁻¹)	4.58 $\pm$ 0.83	3.42 $\pm$ 0.54	5.22 $\pm$ 1.01	14.85 $\pm$ 3.74

3.2.3. Controls of spatial variation of potential P release

Potential SRP release flux rates (pre-restoration) were highly predictable (stepwise multiple linear regression, $R^2 = 0.81$, $P < 0.001$) by pre-restoration soil labile reactive P (NH_4Cl -SRP, $\beta = 0.53$), total Al ($\beta = -0.39$), and total P ($\beta = -0.26$). Potential NRP release flux rate also was highly predictable ($R^2 = 0.73$, $P < 0.001$) by soil amnox-P/ amnox-Al + Fe ratio ($\beta = 0.54$), total P ($\beta = -0.97$), and carbonate reactive P (HCl -SRP, $\beta = 0.55$). The other measured soil physico-chemical characteristics were not selected by regression models of pre-restoration potential SRP or NRP release rates.

3.3. Post-restoration potential P release and soil characteristics

3.3.1. Potential P release

Potential SRP release from soil cores collected during the post-restoration period, and experimentally flooded with river water for 21 days, differed among sites (Kruskal-Wallis test, $p = 0.016$). Potential SRP flux to flood waters was similar and low at NM and RH, and similar and high at RM and CM (Table 3). Patterns of potential NRP release from the same post-restoration cores also similarly differed among sites ($p = 0.004$), with highest NRP fluxes at RM and CM, and lowest NRP

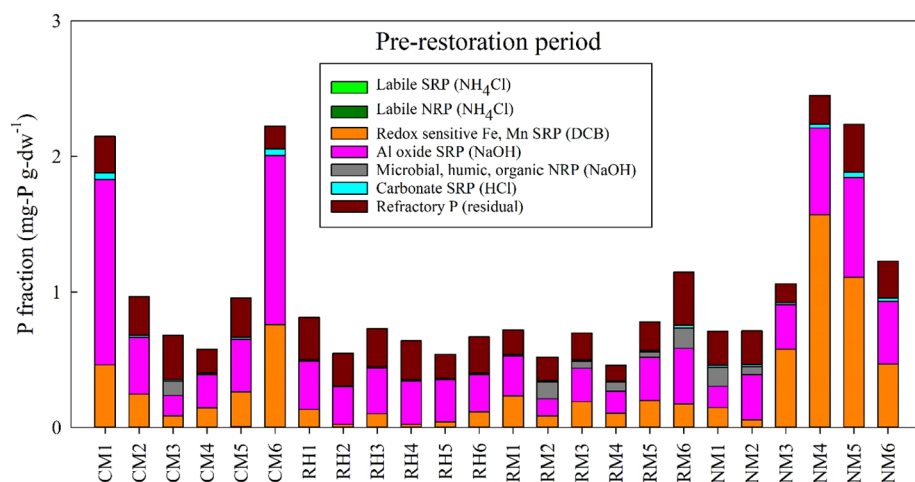


Fig. 4. Pre-restoration sequential P fractionation of floodplain soils. The soil cores from each integration plot at each site (numbered 1–6) are shown individually. CM = channelized mainstem, RH = (to be) restored headwater, RM = (to be) restored mainstem, and NM = natural mainstem. See Appendix Fig. 1 for a description of each extraction.

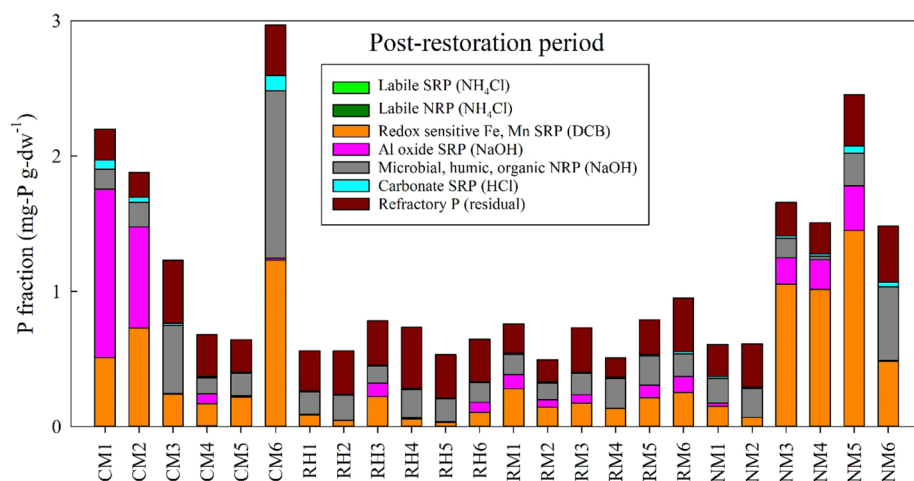


Fig. 5. Post-restoration sequential P fractionation of floodplain soils. Soil cores from each integration plot at each site (numbered 1–6) are shown individually. CM = channelized mainstem, RH = restored headwater, RM = restored mainstem, and NM = natural mainstem. See Appendix Fig. 1 for a description of each extraction.

fluxes at NM (uptake) and RH (Table 3).

3.3.2. Soil characteristics

In the period after restoration, floodplain soils were generally similar to pre-restoration conditions (Table 4). Soils at the two restored sites were wetter compared to CM but drier than NM. Sequential P fractionation generally was similar to pre-restoration, except the microbial, humic, organic P (NaOH-NRP) fraction increased to a relatively large fraction and Al-oxide reactive P (NaOH-SRP) diminished across sites (Fig. 5). Spatial differences and similarities in metal fractionation during the post-restoration period also were similar to the pre-restoration period.

3.4. Changes in potential SRP release and soil characteristics after restoration

3.4.1. Potential P release changes

Comparing pre- and post-restoration rates of potential SRP release from floodplain soils indicate a decrease at both RH and RM as well as NM, whereas potential SRP release increased at CM after restoration (Table 3). Therefore, soils from sites with natural or restored connectivity of floodplain to the channel released less SRP over time. However, spatial variation in changes of potential SRP flux were large within sites and rates of change were not significantly different among sites (Kruskal-Wallis test, $p = 0.929$). In contrast, pre-to-post restoration changes in potential NRP flux did differ among sites ($p = 0.003$). CM released much more NRP in the post-restoration period, as did RM to a lesser degree, whereas NM and RH had little change over time. Although average rates of SRP release decreased over time in the restored floodplains, there was not a statistically significant change in release rate from pre- to post-restoration for either individual RH cores (Wilcoxon Signed Rank Test, $p = 0.917$) or individual RM cores ($p = 0.600$). In contrast, individual cores at RM released more NRP over time ($p = 0.046$), but not at RH ($p = 0.463$).

3.4.2. Soil characteristics changes

None of the sequential P extraction fractions significantly changed from the post- compared to pre-restoration periods; neither did TP, TN, TC, OM, total Al or Fe, or bulk density. Soil pH increased after the restoration period, particularly at RH and CM (Kruskal-Wallis test, $p = 0.003$), whereas soil texture was finer after the restoration period at NM ($p = 0.022$). Most metal extractions were stable over time, with

the exception of decreases in amorphous (ammonox) -Fe and -P fractions at NM (both $p = 0.020$).

3.4.3. Controls of spatial variation of changes in potential P release

Change in potential SRP release rate (post minus pre-restoration) from individual plots' soil was explained by the concurrent change in soil labile reactive P ($\text{NH}_4\text{Cl-SRP}$, $\beta = -0.43$) and change in total P ($\beta = -0.40$) during the same time period (stepwise multiple linear regression, $R^2 = 0.48$, $p = 0.002$). In other words, soils with greater gain of available labile and total P after restoration lost less SRP during experimental flooding after restoration. However, the average decrease in SRP release after restoration (0.037 mg-P less per core) did not match the magnitudes of increase in soil $\text{NH}_4\text{Cl-SRP}$ (0.004 mg-P more per core) or TP (48.1 mg-P more per core) after restoration, suggesting correlation and not direct causation. The change in potential NRP release rate could not be statistically explained using changes in physico-chemical characteristics as possible predictors ($p > 0.05$).

3.5. Field flux measurements

3.5.1. Sedimentation

Vertical deposition of sediment (mm yr^{-1}) on floodplains was relatively low in the pre-restoration period and did not vary significantly among sites (Kruskal-Wallis test, $p = 0.313$, Table 5). However, deposition rates in the post-restoration period did significantly differ among sites ($P < 0.001$, Table 5). Average deposition rates at CM, RM, and RH were all significantly less than NM during the post-restoration period. Change in deposition rates from the post- compared to the pre-restoration period was positive (increasing) at all sites but also differed among sites ($P < 0.001$) with the largest increases at NM compared to the other statistically similar sites.

Distinct spatial patterns of sediment accumulation within each site emerged. In particular, spatial variation in deposition in relation to breach locations at restored sites informs the effects of restoration on floodplain connectivity using sediment trapping as an indicator. At both RM and RH, the largest increases in deposition rates after restoration (increases of 12.7 and 9.2 mm/yr, respectively) were recorded at marker horizons near levee breaches (Fig. 6). Further, changes in deposition rates were highest along upstream portions of both restored sites, whereas lowest sediment accumulation occurred in interior floodplain distant from river and along the downstream portion of sites.

Table 5

Sediment deposition and sedimentation rates measured at marker horizon plots (former) and the subset of integration plots (latter, n = 6) of each site. Mean $\pm$ one standard error.

Site	Deposition rates (mm yr^{-1})				Sedimentation rates ($\text{g m}^{-2} \text{yr}^{-1}$)		
	Number of marker horizons	Pre-restoration	Post-restoration	Change after restoration	Pre-restoration	Post-restoration	Change after restoration
Channelized mainstem	16	1.3 ± 0.2	5.0 ± 1.4	4.0 ± 1.6	227 ± 92	386 ± 71	160 ± 124
Restored headwater	15	1.0 ± 0.2	3.5 ± 0.7	2.5 ± 0.7	114 ± 31	514 ± 83	400 ± 77
Restored mainstem	27	1.8 ± 0.5	5.3 ± 0.7	3.0 ± 0.7	332 ± 197	694 ± 169	362 ± 150
Natural mainstem	17	1.0 ± 0.3	12.2 ± 1.2	11.2 ± 1.3	448 ± 133	5403 ± 1481	4955 ± 1429

This pattern was most pronounced at RH, which is located farthest upstream within Pocomoke River system. At RM, deposition was moderate or low throughout the downstream portion of the floodplain associated with the three most downstream levee breaches. Changes in deposition rates varied widely at CM, with greatest increases in sediment accumulation occurring along preferred flow channels. NM had consistent substantial increases in deposition during the study period with little spatial variation.

Coring marker horizons of the six integration plots at each site at the end of the study provides estimates of the mass flux and characteristics of deposited sediment. Though sampled sediment represents material deposited both before and after restoration, most sediment was deposited after the restoration period (prior paragraph). Sediment deposited on NM was more mineral, with less TC (Kruskal-Wallis test, $p = 0.009$) and TN ($p = 0.014$) and organic content ($p = 0.041$), greater total Fe concentration ($p = 0.004$), and coarser mean particle size ($p = 0.009$) than other sites. Organic content of sediment deposited at CM and RH was high, and intermediate at RM, otherwise, characteristics of deposited sediment were similar among sites.

Sedimentation rates ($\text{g m}^{-2} \text{yr}^{-1}$) did not significantly differ among sites before restoration (Kruskal-Wallis test, $p = 0.326$), but NM had significantly greater sedimentation rate in the post-restoration period compared to other sites ($p = 0.002$; Table 5). Change in sedimentation rate after restoration was greatest at NM, intermediate at RH and RM, and least at CM ($p = 0.003$), suggesting that restoration had moderate effect (less than natural floodplain, more than channelized floodplain). However, changes in sedimentation rates were not statistically different between RH and CM (Mann-Whitney test, $p = 0.200$) and RM and CM ($p = 0.262$). Similar patterns occurred for P-sedimentation and N-sedimentation across sites both pre- (Kruskal-Wallis test, $p = 0.093$ and $p = 0.784$) and post-restoration ($p = 0.002$ and $p = 0.003$), and for change in rates after restoration ($p = 0.004$ and $p = 0.003$, respectively; Table 6).

3.5.2. Inorganic nutrient flux

Stacked bags of ion exchange resin beads measured in situ rates of vertical SRP and DIN input (upper bag) and release from floodplain soils (lower bag) over the flood season following restoration periods.

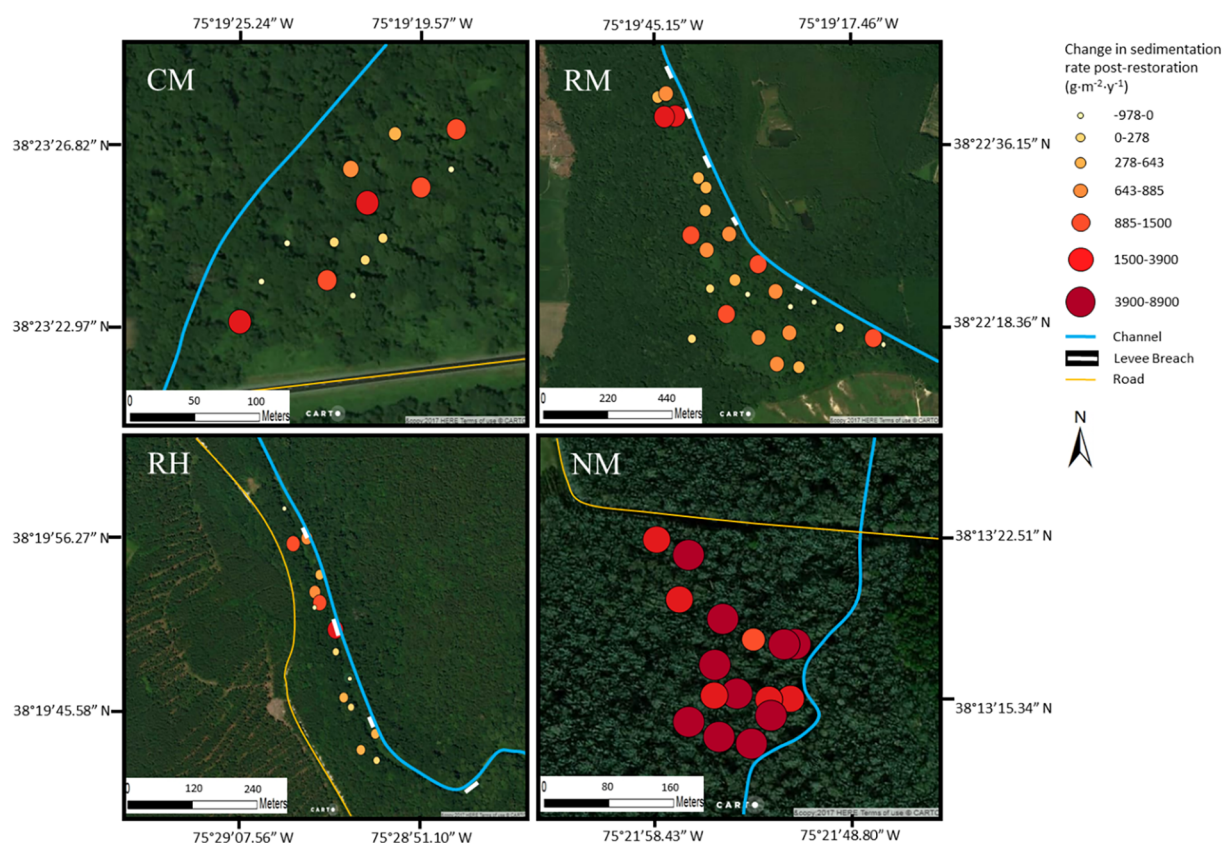


Fig. 6. Change in sedimentation rates (post-restoration minus pre-restoration rates) within each floodplain site.

Table 6
Mass balance of P and N fluxes before and after restoration period. Change in fluxes is calculated as post- minus pre-restoration fluxes. Sum of fluxes assumes net SRP flux measurements during the flood season are all that occur over the year. Mean $\pm$ one standard error of the six integration plots in each site. DIN = dissolved inorganic N (NH_4^+ + NO_2^- + NO_3^-).

Site	Pre-restoration P- sedimentation flux ($\text{g-P m}^{-2} \text{y}^{-1}$)	Post-restoration P- sedimentation flux ($\text{g-P m}^{-2} \text{y}^{-1}$)	Change in P- sedimentation flux ($\text{g-P m}^{-2} \text{y}^{-1}$)	Input SRP flux (g-P m^{-2})	Release SRP flux (g-P m^{-2})	Net SRP flux (g-P m^{-2})	Sum post-restoration P fluxes ($\text{g-P m}^{-2} \text{y}^{-1}$)	Sum change in P fluxes ($\text{g-P m}^{-2} \text{y}^{-1}$)
Channelized mainstem	0.17 $\pm$ 0.059	0.37 $\pm$ 0.13	0.20 $\pm$ 0.16	0.095 $\pm$ 0.011	-0.082 $\pm$ 0.013	0.013 $\pm$ 0.024	0.38 $\pm$ 0.15	0.21 $\pm$ 0.18
Restored headwater	0.085 $\pm$ 0.029	0.36 $\pm$ 0.047	0.28 $\pm$ 0.041	0.017 $\pm$ 0.0042	-0.027 $\pm$ 0.015	-0.010 $\pm$ 0.019	0.35 $\pm$ 0.066	0.27 $\pm$ 0.060
Restored mainstem	0.36 $\pm$ 0.22	0.64 $\pm$ 0.18	0.28 $\pm$ 0.11	0.078 $\pm$ 0.028	-0.091 $\pm$ 0.025	-0.013 $\pm$ 0.053	0.63 $\pm$ 0.23	0.27 $\pm$ 0.16
Natural mainstem	0.44 $\pm$ 0.11	6.0 $\pm$ 1.5	5.5 $\pm$ 1.4	0.098 $\pm$ 0.017	-0.12 $\pm$ 0.033	-0.024 $\pm$ 0.050	6.0 $\pm$ 1.6	5.5 $\pm$ 1.5
Pre-restoration N- sedimentation flux ($\text{g-N m}^{-2} \text{y}^{-1}$)		Post-restoration N- sedimentation flux ($\text{g-N m}^{-2} \text{y}^{-1}$)	Change in N- sedimentation flux ($\text{g-N m}^{-2} \text{y}^{-1}$)	Input DIN flux (g-N m^{-2})	Release DIN flux (g-N m^{-2})	Net DIN flux (g-N m^{-2})	Sum post-restoration N fluxes ($\text{g-N m}^{-2} \text{y}^{-1}$)	Change in N fluxes ($\text{g-N m}^{-2} \text{y}^{-1}$)
Channelized mainstem	2.4 $\pm$ 0.88	4.6 $\pm$ 1.1	2.2 $\pm$ 1.6	1.6 $\pm$ 0.41	-1.5 $\pm$ 0.64	0.10 $\pm$ 1.1	4.7 $\pm$ 2.2	2.3 $\pm$ 2.7
Restored headwater	1.4 $\pm$ 0.39	6.0 $\pm$ 0.66	4.6 $\pm$ 0.71	0.28 $\pm$ 0.038	-0.18 $\pm$ 0.044	0.10 $\pm$ 0.082	6.1 $\pm$ 0.74	4.7 $\pm$ 0.79
Restored mainstem	3.6 $\pm$ 2.0	7.2 $\pm$ 1.6	3.7 $\pm$ 1.4	2.5 $\pm$ 1.0	-1.1 $\pm$ 0.37	1.4 $\pm$ 1.4	8.6 $\pm$ 3.0	5.1 $\pm$ 2.8
Natural mainstem	2.7 $\pm$ 1.0	27 $\pm$ 2.6	24 $\pm$ 1.6	5.0 $\pm$ 1.5	-1.2 $\pm$ 0.27	3.8 $\pm$ 1.7	31 $\pm$ 4.3	28 $\pm$ 3.3

Three sites had greater average rates of soil SRP release than SRP input, or a net loss of soil SRP (RH, RM, and NM), whereas CM had greater SRP input than release (Table 7). However, net (uptake minus release) SRP uptake did not significantly differ among sites (Kruskal-Wallis test, $p = 0.200$). Although soils of both naturally and restored connected floodplains had a net release of SRP, input fluxes were only slightly smaller than release fluxes. Both input and release fluxes of SRP were smaller at RH. The stacked resin bags also measured net uptake of DIN (NH_4^+ , NO_3^- , plus NO_2^-) at all sites, with average inputs exceeding release from soils at each site (Table 7). Net flux of DIN was greatest at NM, intermediate at RM, and low at RH and CM ($p = 0.026$). Inputs of DIN, particularly NO_3^- , to floodplain soils were much greater at NM and RM. Finally, gross SRP release rates (bottom bag only) after restoration were predictable using pre-restoration soil amorphous amox-P/amox-Al ratio ($\beta = 0.78$) and total N ($\beta = 0.30$, $R^2 = 0.48$, $p = 0.001$). Other pre-restoration soil physico-chemical characteristics were excluded by stepwise regression.

These rates of actual SRP release from floodplain soils after restoration, measured in situ (resin bags), were generally much smaller than and uncorrelated with potential SRP release rates measured from pre-restoration soils (experimental flood, Fig. 7). Average potential SRP release rates were 5 times greater than actual SRP release at RH, 8 times greater at RM, and 9 times greater at CM, whereas the opposite pattern occurred at NM with 4 times greater actual than potential SRP release.

3.5.3. Mass balance of phosphorus and nitrogen

Combined vertical inputs (sedimentation and dissolved inorganic loading) of P and N greatly exceeded floodplain soil upward release to floodwater (dissolved inorganic discharge) at all sites. Sedimentation fluxes of P and N greatly exceeded dissolved inorganic nutrient inputs (SRP and DIN) to upper resin bags (Table 6). As previously mentioned, gross release of SRP from floodplain soils (lower resin bags) slightly exceeded dissolved inorganic inputs to soil at all connected sites (natural or restored; net release) and was less than inputs at CM site (net input). Phosphorus sedimentation inputs, estimated as either the rate after the restoration period or the change in rate (post- minus pre-restoration period), were roughly 50 times greater than SRP gross release rates from soils at NM, 10 times greater at RH, 5 times greater at RM, and 3 times greater at CM; N sedimentation inputs were roughly 20, 30, 5, and 2 times greater than DIN release, respectively (Table 6). Net P mass balance, estimated from all measured fluxes measured after restoration, was much larger at NM, intermediate at RM, and low but still positive (net retention) at RH and CM (Kruskal-Wallis test, $p = 0.003$). When calculated as change after restoration ($p = 0.004$), RH and CM (Mann-Whitney test, $p = 0.173$) and RM and CM sites ($p = 0.631$) net P mass balances were statistically similar with larger mean changes at the two restored sites. The different results for net P mass balance at RM, estimated as post-restoration vs. change after restoration, is due to relatively large rates of P sedimentation that occurred pre-restoration at that site. Net N mass balances (post-restoration) were generally similar to patterns of P mass balance among sites (Kruskal-Wallis test, $p = 0.001$), although CM had lowest rates of N retention. Although change in net N mass balance after restoration differed among sites ($p = 0.003$), it was statistically similar between RH and CM (Mann-Whitney test, $p = 0.262$) and RM and CM ($p = 0.055$); however, restored sites had larger mean values.

4. Discussion

Do partial berm excavations designed to reconnect floodplain-river systems significantly increase water quality benefits of floodplain wetlands? The goal of this study was to compare P, N, and sediment inputs and release from floodplain soils before and after restoration of disconnected floodplains along the channelized river of an agricultural, lowland watershed. Multiple partial breaches 10–45 m in length were constructed in artificial spoil levees along each of two reaches of the

Table 7

Fluxes of inorganic P and N to stacked resin bags deployed in situ during the post-restoration period at each site. Upper (resin bag) represents input fluxes to floodplain soils, Lower (resin bag) represents release fluxes from floodplain soils, Upper-Lower is the net flux. Fluxes ($\text{mg-P m}^{-2} \text{d}^{-1}$ or $\text{mg-N m}^{-2} \text{d}^{-1}$) are mean $\pm$ one standard error of the six integration plots in each site. < MDL = below machine detection limit.

Resin bag	Channelized mainstem	Restored headwater	Restored mainstem	Natural mainstem
<i>SRP</i>				
Upper	0.65 ± 0.073	0.14 ± 0.036	0.54 ± 0.19	0.67 ± 0.12
Lower	0.56 ± 0.092	0.23 ± 0.13	0.62 ± 0.17	0.84 ± 0.23
Upper-Lower (net)	0.087 ± 0.10	-0.087 ± 0.12	-0.088 ± 0.17	-0.16 ± 0.13
NH_4^+				
Upper	1.5 ± 0.21	1.1 ± 0.090	1.4 ± 0.29	2.8 ± 0.64
Lower	1.1 ± 0.15	1.0 ± 0.32	1.5 ± 0.33	2.4 ± 0.59
Upper-Lower (net)	0.31 ± 0.18	0.031 ± 0.33	-0.046 ± 0.30	0.39 ± 0.63
NO_2^-				
Upper	< MDL	< MDL	< MDL	0.082 ± 0.050
Lower	< MDL	< MDL	< MDL	0.072 ± 0.057
Upper-Lower (net)	< MDL	< MDL	< MDL	0.010 ± 0.023
NO_3^-				
Upper	9.2 ± 2.6	1.3 ± 0.23	16 ± 6.6	31 ± 9.4
Lower	8.8 ± 4.2	0.52 ± 0.051	5.9 ± 2.2	5.9 ± 1.2
Upper-Lower (net)	0.38 ± 2.1	0.81 ± 0.24	9.9 ± 5.6	25 ± 9.2

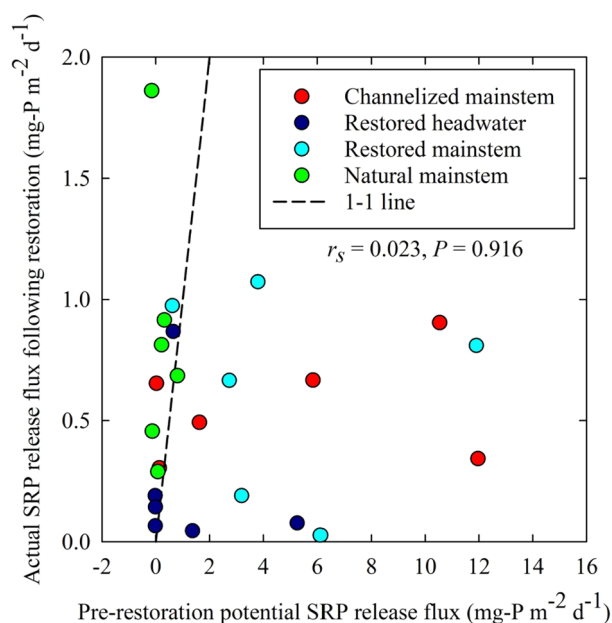


Fig. 7. Measured (in situ capture by resin bag after restoration) vs. potential (short-term experimental flood in laboratory of pre-restoration soil cores) SRP release flux from floodplain soils. Spearman rank correlation statistics and 1-to-1 line are included.

channelized Pocomoke River system. Using a combination of laboratory and field measurements, we found that flood frequency, P, N, and sediment retention increased after restoration, however, results were not statistically different from those observed at unrestored channelized floodplain. One rare, very large flood (200- to 300-year return interval) resulted in river stage heights that exceeded spoil berm elevations along the river's entire length and flooded unrestored channelized floodplain with river water. Natural floodplain had the greatest rates of increased nutrient and sediment trapping during this period. Because levee

breaches at restored floodplains connected more frequently during smaller floods, presumably restorations would have yielded larger increases in trapping capacity compared to unrestored channelized floodplain during average weather conditions. Next, we discuss key findings about the studied floodplain restorations.

4.1. Hydrologic results demonstrate the effects of watershed position

Was the hydrology of floodplains changed as a result of restoration through levee breaching? Overall, reconnection did not appreciably change floodplain inundation statistics at two restored sites, RH and RM, compared to unrestored CM and did not lead to hydrology similar to natural NM. Presumably more differences would have been found in years with more typical flooding (and not like 2016) when river stage would not exceed levee tops at channelized floodplains but would have connected restored floodplains through their levee breaches (see below). We have observed hydrologic connection at the restored floodplains during moderate floods, with water from the river channel entering floodplains through levee breaches ([Appendix Photo 2](#)).

The study sites were located at different positions within the Pocomoke River system and water quality results were consistent with river-floodplain classifications that highlight systematic changes in hydrologic function along a river corridor, from headwater reaches to delta ecosystems ([Brinson, 1993](#); [Rosgen, 1994](#); [Church, 2002](#); [Batzer et al., 2018](#)). Lowest sediment accumulation rates and least evidence of riverine influence occurred at RH farthest up-gradient within the river system. RM, located at in intermediate longitudinal river location, reflected significant influence by both the Pocomoke River and local tributaries. The most frequent flooding and highest sediment delivery into floodplain, and thus evidence of riverine influence, was observed farthest downstream at NM. Indeed, channelization ends 5 km upstream of the site perhaps reflecting an engineering decision on the limited effectiveness of further downstream channelization due to much greater frequency, duration, and magnitude of flooding at this position in the watershed.

4.2. Floodplain P and N inputs exceed release

Field measurements of plot-scale vertical P and N input fluxes were greater than losses at all floodplain sites after the period of restoration. These measurements included sedimentation and dissolved inorganic nutrient inputs to soils and dissolved inorganic nutrient losses from soils. Both RH and RM floodplains had positive net retention after restoration: 0.35 and 0.63 g-P m⁻² y⁻¹, respectively; and 6.1 and 8.6 g-N m⁻² y⁻¹, respectively. Positive net retention of P and N occurred because sedimentation inputs of nutrients were large, and dissolved inorganic nutrient loss was small and only slightly larger than dissolved inorganic nutrient input. In comparison, the reviews by [Craft et al. \(2018\)](#) calculated a mean P burial rate of 2.2 g-P m⁻² y⁻¹ for floodplains, and [Mitsch et al. \(2005\)](#) calculated a median nitrate removal rate of 29 g-N m⁻² y⁻¹ for river diversion wetlands in agricultural watersheds.

Phosphorus and N enriched sediment were deposited on floodplains at rates greatly exceeding SRP or DIN inputs to soils and releases from soils. The P- and N-sedimentation rates in restored sites in this study were generally low compared to other studies, whereas rates in the natural floodplain were intermediate ([Yarbro, 1983](#); [Owens and Walling, 2002](#); [Noe and Hupp, 2005](#); [Anderson and Mitsch, 2006](#); [Wassen and Olde Venterink, 2006](#); [Olde Venterink et al., 2006](#); [Wolf et al., 2013](#)). However, not all sediment bound P and N is bioavailable. Estimates of the proportion of sediment- and soil-bound nutrients that is mineralized or bioavailable for aquatic primary productivity (roughly one quarter of P; [Sharpley et al., 1992](#); [Ellison and Brett, 2006](#); [Kronvang et al., 2009](#); roughly one half of N; [Mayer et al., 1998](#)) suggest that net mass balance of bioavailable P and N was still positive (net retention) in this study, with sediment inputs of bioavailable P and N exceeding long-term phosphate and DIN release from soils. Thus, the restored floodplains, as well as natural and channelized floodplains (to a smaller degree), are likely reducing loading of total and bioavailable nutrients to downstream waters.

Rates of in situ phosphate release from floodplain soils after restoration of RH and RM or in NM were slightly larger than rates of phosphate inputs to soil. However, DIN inputs were much greater than release from floodplain soils at RH, RM, and NM, mostly due to larger inputs than release of NO₃⁻, while NH₄⁺ inputs were balanced with release. These Pocomoke River floodplain soils had much lower P release rates than other rewetted floodplain soils from agricultural settings ([Aldous et al., 2007](#); [Kronvang et al., 2009](#)). The floodplains in this study have remained forested with no cropping or grazing in an otherwise agricultural watershed, and as a result have moderate to somewhat above average P and typical N content of soils compared to other wetlands ([Bedford et al., 1999](#)). Hydrologic reconnection to floodplains that have very high P content, such as those fertilized during agricultural land use, are more likely to release greater amounts of dissolved inorganic nutrients upon re-flooding that can persist over many years ([Surridge et al., 2012](#)).

4.3. Partial levee breaches were somewhat effective at restoring hydrologic connectivity and water quality functions

Both RH and RM had intermediate P and N retention fluxes compared to NM and CM. Change in net P retention (after restoration relative to pre-restoration mass balance) at both RH and RM was 0.06 g-P m⁻² y⁻¹ greater compared to CM. However, these rates of increase in P and N, after restoration were not statistically different from CM. The NM floodplain, however, had significantly larger increases in net P

retention during the same time period than either RH or RM or CM (an increase of 5.3 g-P m⁻² y⁻¹ over the change at CM). Post-restoration increases in net N mass balance at RH and RM were 2.4 and 2.8 g-N m⁻² y⁻¹, respectively, above that of CM, and 26 g-N m⁻² y⁻¹ at NM compared to CM. Similarly, sedimentation rates had larger increases after restoration at RH and RM compared to CM but differences were not statistically different, whereas NM had very large increases in sedimentation. In conclusion, restored floodplains had better water quality functioning than unrestored, channelized floodplain but did not function as well as natural floodplain. Next, we describe several possible explanations for the observed responses in restored floodplains.

The meager increases in retention at RH and RM relative to CM are likely in part due to hydrologic extremes during the study. These extremes included weather of drought, followed by extreme flooding, which limited our capacity to observe differences in water quality benefits among our restored, channelized, and natural floodplain sites. Specifically, the large flood overtopped the artificial levee at CM and likely increased nutrient inputs from the river channel to its floodplain. Historically, flooding by river water is unlikely to occur at channelized floodplain ([Kroes and Hupp, 2010](#)) but two to six overbank flow events are expected to occur in areas lacking artificial spoil levees, and stronger differentiation of restored sites from channelized floodplain is expected to occur. For example, sediment deposition rates at CM averaged 0.6 mm/yr compared to 2.9 mm/yr at NM during a period with typical flooding from 1998 to 2002 ([Kroes and Hupp, 2010](#)). In this study, CM had an average deposition rate of 1.3 mm/yr in 2014, increasing to 5.0 mm/yr from 2015 to 2016 with the large flood.

The spatial patterning of sediment deposition indicated that delivery of sediment-rich river water is limited to areas near the partial levee breaches of restored floodplains. NM had a homogenous pattern of increased deposition across the network of marker horizons throughout the site, whereas CM and both RH and RM had heterogeneous patterns. The largest increases in sedimentation within each of the restored floodplains occurred immediately down gradient from their upstream levee breaches, with smaller increases or net erosion occurring at interior or areas further downstream of breaches. These observations also suggest that the river is the source of sediment and nutrients to areas of the floodplain with high trapping rates.

Sediment transport from the river channel and sedimentation within floodplains is a function of river load, internal floodplain channels, and distance from channel in most natural and restored floodplains ([Mertes, 1997](#); [Ross et al., 2004](#); [McMillan and Noe, 2017](#)). Sedimentation patterns in both restored Pocomoke River floodplains suggests that loading and distribution of river sediment throughout these floodplains are limited and that additional levee breaches could increase sediment loading through, and trapping within, restored floodplains. The relatively short breaches in spoil levees of the Pocomoke River restoration, typically less than 45 m in length, may still limit river-floodplain interactions, and a greater portion of the dredge spoil berm excavation may be needed to reestablish natural floodplain hydrologic regimes ([Orr et al., 2007](#); [Nichols and Viers, 2017](#)).

Alternatively, the simple conceptual model of overbank flow inundating an entire floodplain complex may not adequately capture floodplain hydrodynamics and describe floodplain capacity to improve riverine water quality. Previous studies show that while floodplain inundation is strongly controlled by river stage and antecedent conditions, the source of flood waters to the floodplain zone largely depends on watershed position and local microtopographic controls on the intersection of multiple, mixing water sources, including groundwater, local discharge, direct precipitation, antecedent surface water storage,

as well as river water (Mertes, 1997; Jones et al., 2015). Even in large floodplain systems where river flood pulses are obvious, non-riverine water can be the primary contributor to the wetland water budget for portions of the floodplain (Mertes, 1997; Jones et al., 2014; Lewin and Ashworth, 2014; Batzer et al., 2018).

Another potential explanation for limited restoration response may be that the relatively simple restoration technique of breach excavations may not adequately overcome the full range of human impacts affecting floodplain function. A more comprehensive floodplain restoration, including raising the river bed and limiting artificial drainage from adjacent uplands directly to the river, likely is needed to overcome extensive hydrologic alterations throughout the river corridor (Rheinhardt et al., 1999; Bathurst et al., 2002; Wyżga et al., 2016). Restoring natural floodplain hydrologic regime likely requires addressing comprehensive hydrologic alterations that short-circuit floodplain wetland water quality benefits.

4.4. Laboratory flooding experiment did not accurately predict phosphate release after restoration

Experimental flooding of soil cores is a common approach for predicting the potential for phosphate release from restored wetland and floodplain soils under controlled conditions (Aldous et al., 2007; Loeb et al., 2008; Dunne et al., 2010; Schönbrunner et al., 2012). We measured potential phosphate release during 21-d experimental floods of river water in soil cores collected both before and after the field restorations. However, our estimates of phosphate release flux from pre-restoration soil cores to surface floodwater were uncorrelated with measured rates of release in the field after restoration. Other studies also have found that experimental flooding of soil cores underestimates SRP release in situ after floodplain rewetting (Surridge et al., 2012). Our experimental laboratory floods did correctly predict greater phosphate release from floodplain soils at RM compared to RH. Thus, experimental flooding of soil cores in this study was useful for coarsely estimating the relative magnitude of phosphate release from soils to floodwater among different floodplains, but not quantitatively and accurately estimating flux.

The most likely explanation for discrepancies between potential and measured phosphate release rates is the hydrology of the experimentally inundated soil cores in the laboratory. Soil porewater phosphate can be transported to surface waters through both advection and diffusion (Reddy and DeLaune, 2008). The experimental flood method added a batch of river water to closed soil cores at the beginning of the experiment, generating a static hydraulic head. These static water inundation conditions differ from inundation of floodplains where, for example, surface water floods can infiltrate into soil and recharge groundwater (Ramberg et al., 2006). Groundwater levels were typically below the soil surface at all sites, indicating the likelihood of flood water infiltration into floodplain soils. Downward advection would transport most soil porewater phosphate deeper into soil (Reddy et al., 2005) instead of upwards out of the soil to be captured by resin bags or quickly exported to surface waters downstream. Some of this groundwater phosphate can eventually be transported to surface water through groundwater flowpaths to floodplain distributary channels (Surridge et al., 2012) or the river channel (Gu et al., 2017).

4.5. Phosphate release was predictable by soil geochemistry

Half of the spatial variation in measured field fluxes of soil SRP release to the surface after restoration was predictable using the

ammox-P/ammox-Al ratio and total N content of pre-restoration soils. Notably, no other pre-restoration soil physico-chemical characteristics, including both sequential P fractionations and individual metal fractionations, could significantly explain field flux. Ammox-Al or -Fe and their ratio with ammox-P are useful indexes of P sorption capacity and saturation in soils and sediments to predict P availability and loss to surface waters (Reddy and DeLaune, 2008; Maguire and Sims, 2002). Experimental flooding of wetland soils can increase ammox-Al and the sorption capacity for P (Darke et al., 1997). We did not find a change in ammox-Al over time from the pre- to post-restoration period, whereas we found a decrease in ammox-Fe and ammox-P in natural floodplain during the wetter post-restoration period. In this study, the ratio of ammox-P to ammox-Al was a useful index to screen potential floodplain restoration sites for their likelihood of releasing soil phosphate.

4.6. Phosphate release decreased over time after restoration

Soils of restored floodplains released less SRP to surface water over time during laboratory experimental floods. Compared to rates measured before restoration, rates measured 16 months after reconnection at RH and 7 months after reconnection at RM were on average 48% and 34% smaller, respectively. These changes in rates were highly spatially variable across each site and the changes in SRP release were not statistically significant in either site. Rates of potential SRP release after restoration of RH were similar to NM, but RM continued to have somewhat elevated SRP release after restoration. The changes in potential SRP release over time could be due to either export of P from soils after restoration or changing P and metal fractionation due to restoration. However, no sequential P extraction fractions significantly changed, and only ammox-Fe and ammox-P (amorphous forms) changed slightly at restored sites after restoration, suggesting that altered fractionation did not decrease reducible or desorbable forms of P. We did observe increases in potential NRP release after restoration, indicating that some of the SRP that was releasable pre-restoration was instead released as NRP post-restoration. The change from SRP to NRP release is possibly explained by the general but not statistically significant increase in soil NaOH-NRP (microbial, humic, and organic P) fraction after restoration. Nevertheless, the large decrease in average SRP potential release from restored floodplain soils over time suggests that SRP release will decrease over time following hydrologic restoration as labile and reducible forms of P are exported from soils.

5. Conclusions

Restored floodplains had slightly greater increases in P, N, and sediment retention after reconnection with the river than unrestored channelized floodplain, and in comparison, natural floodplain had much larger retention rates. We found that reconnection of floodplains through partial breaches of spoil levees along a channelized river led to: 1) visible inputs of river water into floodplains but that post-restoration trends were obfuscated by a large flood that also connected unrestored, channelized mainstem floodplain; 2) increased sedimentation and attached P and N inputs to floodplains; 3) no increase in SRP release from floodplains after restoration compared to unrestored, channelized floodplain; and 4) rates of SRP release from floodplain soils were predictable by soil geochemistry, namely, the ratio of ammonium oxalate extractable P and Al and the concentration of TN; 5) SRP release from restored floodplain soils decreased over time; and 6) P and N inputs greatly exceeded P and N release from soils at all sites. However, the large decreases in sediment deposition rates downgradient from partial

levee breaches and in the interior of restored floodplains suggest that removal of additional length of artificial levee could increase the water quality functions of these restored floodplains. In addition, approaches that utilize static inundation of soil cores to measure potential phosphate release after re-flooding are likely not accurate predictors of actual in situ release following floodplain restorations. In summary, restoring floodplain hydrologic connectivity was moderately successful at trapping additional P, N, and sediment, most likely from river water, and therefore improving water quality in the Pocomoke River and downstream estuary. Other ecosystem services, beyond the water quality processes we measured, may also have been influenced by floodplain restoration, including biodiversity, carbon sequestration, recreation, and flood attenuation (Opperman et al., 2009).

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

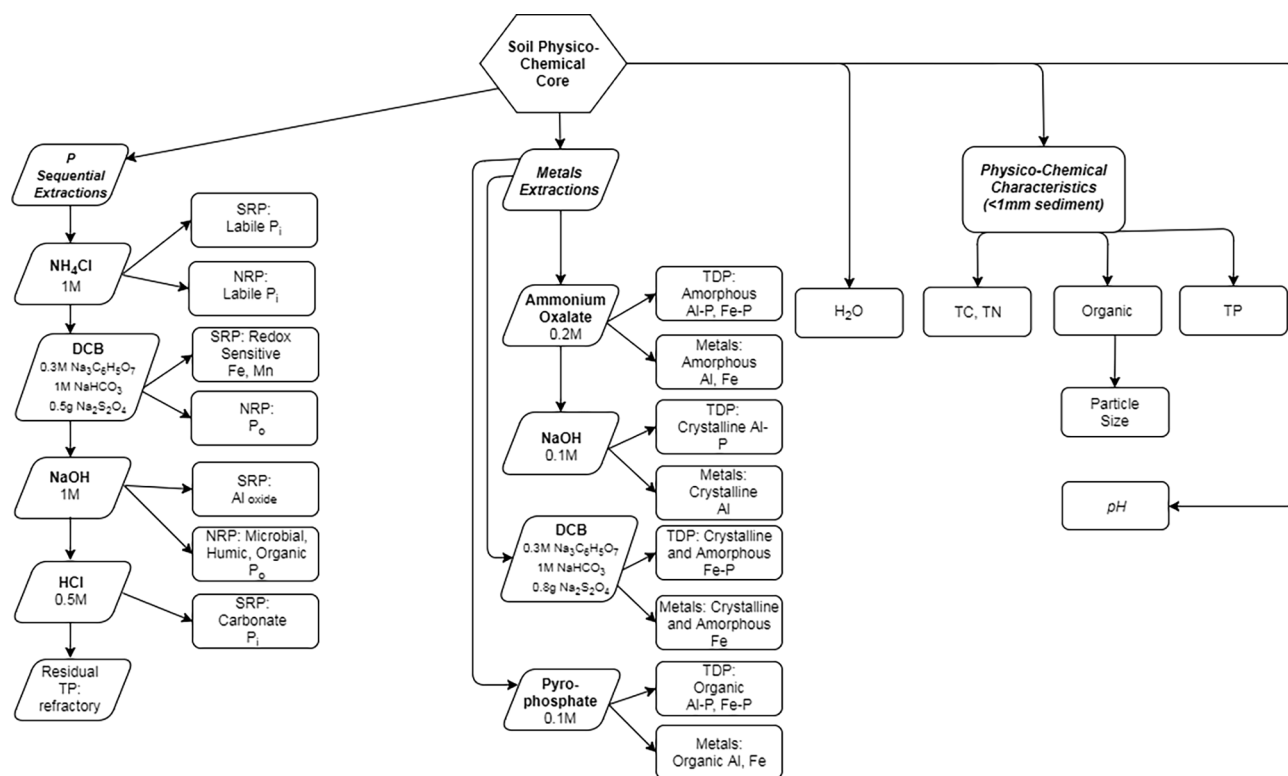
Appendix

Acknowledgements

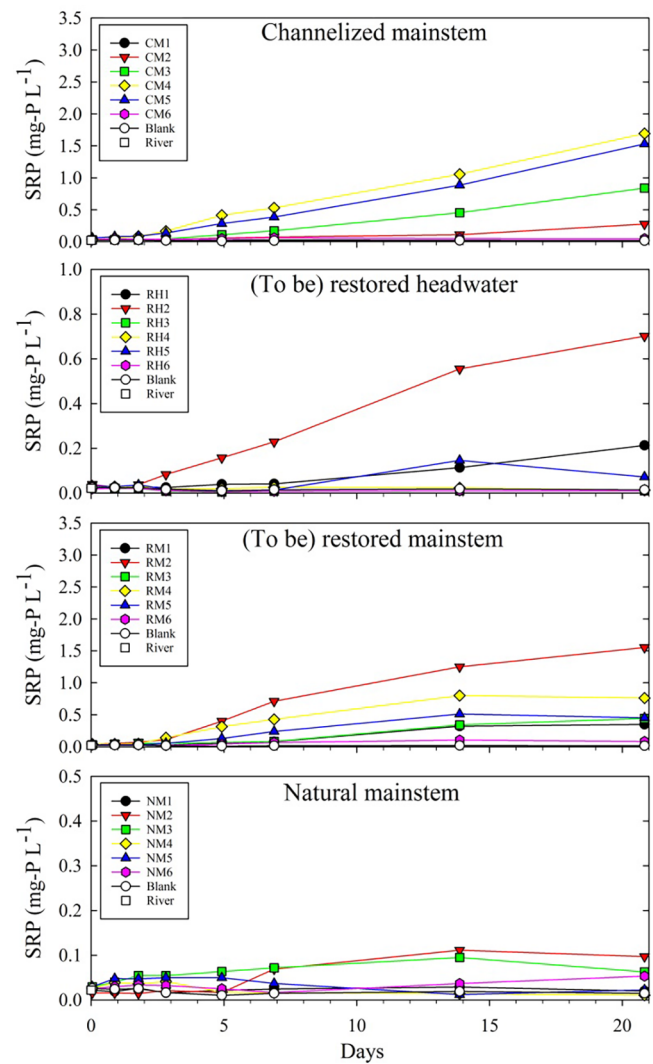
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Data accessibility statement

All data presented in the study have been published by Noe et al. (2019) at <https://doi.org/10.5066/P9G2LP0Z>.



Appendix Fig. 1. Laboratory analyses of floodplain soil physico-chemical characteristics core collected at each integration plot of each site, both pre- and post-restoration. An additional flood simulation core and bulk density core were also collected at each integration plot (not shown).



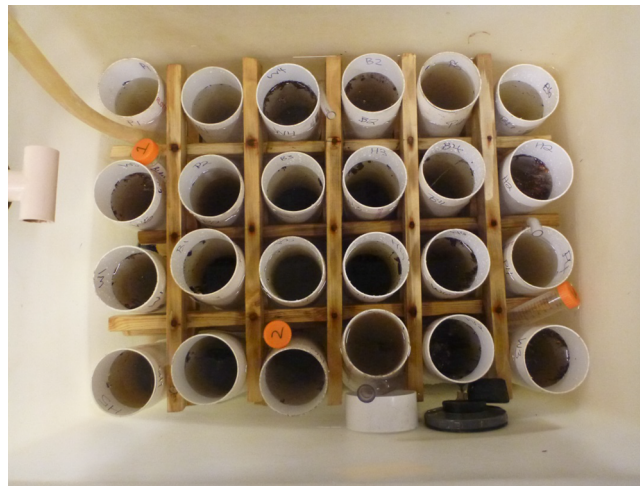
Appendix Fig. 2. Concentrations of SRP in surface water over time during short-term experimental floods of pre-restoration soil cores in the laboratory. The soil cores from each integration plot at each site (numbered 1–6) are shown individually, as well as the average of the three river water blanks and the initial river water concentration.



Appendix Photo 1. A levee breach at the restored headwater site during a low discharge period. Water from the river channel (left) flows across the breach (middle, with person standing in low spot) into the floodplain (right) during floods. Photo taken 2 December 2014. Credit: G. NOE, USGS.



Appendix Photo 2. A levee breach at the restored mainstem site during a flood. Water from the river channel (right) flows across the breach (middle) into the floodplain (left). View is looking across from one end of the breach to the other where the spoil levee continues upstream. Photo taken 8 February 2016. Credit: G. NOE, USGS.



Appendix Photo 3. Overhead photo of the short-term laboratory flooding experiment with the post-restoration soil cores. Fresh tap water was circulated around the core tubes that were inundated with filtered river water. Tap water was added in the tubing at the left and drained from the outlet on the side wall of the aquarium tank that also maintained constant tap water level. Blanks of filtered river water are incubating in orange-topped centrifuge tubes. Credit: G. NOE, USGS.

Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.ecoleng.2019.08.002>.

References

- Aldous, A.R., Craft, C.B., Stevens, C.J., Barry, M.J., Bach, L.B., 2007. Soil phosphorus release from a restoration wetland, Upper Klamath Lake, Oregon. *Wetlands* 27, 1025–1035.
- Anderson, C.J., Mitsch, W.J., 2006. Sediment, carbon, and nutrient accumulation at two 10-year-old created riverine marshes. *Wetlands* 26, 779–792.
- Ardón, M., Montanari, S., Morse, J.L., Doyle, M.W., Bernhardt, E.S., 2010. Phosphorus export from a restored wetland ecosystem in response to natural and experimental hydrologic fluctuations. *J. Geophys. Res.* 115. <https://doi.org/10.1029/2009JG001169>.
- Ator, S.W., Denver, J.M., Brayton, M.J., 2004. Hydrologic and geochemical controls on pesticide and nutrient transport to two streams on the Delmarva Peninsula. U.S. Geological Survey, Scientific Investigations Report 2004-5051.
- 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 Res. Appl.* 16, 457–467.
- Bathurst, J.C., Benson, I.A., Valentine, E.M., Nalluri, C., 2002. Overbank sediment deposition patterns for straight and meandering flume channels. *Earth Surf. Proc. Land* 27, 659–665. <https://doi.org/10.1002/esp.346>.
- Batzer, D.P., Noe, G.B., Lee, L., Galatowitsch, M., 2018. A floodplain continuum for Atlantic Coast rivers of the Southeastern US: predictable changes in floodplain biota along a river's length. *Wetlands* 38, 1–13.
- Bayley, S.E., 1985. Effect of natural hydroperiod fluctuations on freshwater wetlands receiving added nutrients. In: *Ecological Considerations in Wetlands Treatment of Municipal Wastewaters*. Van Nostrand Reinhold Company, New York, pp. 180–189.
- Bedford, B.L., Walbridge, M.R., Aldous, A., 1999. Patterns in nutrient availability and plant diversity of temperate North American wetlands. *Ecology* 80, 2151–2169.
- 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* 308, 636–637.
- Bricker, O.P., Newell, W.L., Simon, N.S., 2003. Bog iron formation in the Nassawango watershed, Maryland. U.S. Geological Survey Open-File Report 2003-346.
- Brinson, M.M., 1993. Changes in the functioning of wetlands along environmental gradients. *Wetlands* 13, 65–74.
- Church, M., 2002. Geomorphic thresholds in riverine landscapes. *Freshw. Biol.* 47, 541–557. <https://doi.org/10.1046/j.1365-2427.2002.00919.x>.
- Cloern, J.E., Jassby, A.D., 2012. Drivers of change in estuarine-coastal ecosystems: discoveries from four decades of study in San Francisco Bay. *Rev. Geophys.* 50. <https://doi.org/10.1029/2012RG000397>.
- Conley, D.J., Paerl, H.W., Howarth, R.W., Boesch, D.F., Seitzinger, S.P., Havens, K.E., Lancelot, C., Likens, G.E., 2009. Controlling eutrophication: nitrogen and phosphorus. *Science* 123, 1014–1015.
- Craft, C., Vymazal, J., Kröpfelová, L., 2018. Carbon sequestration and nutrient accumulation in floodplain and depressional wetlands. *Ecol. Eng.* 114, 137–145.
- Craig, L.S., Palmer, M.A., Richardson, D.C., Filoso, S., Bernhardt, E.S., Bledsoe, B.P., Doyle, M.W., Groffman, P.M., Hassett, B.A., Kaushal, S.S., Mayer, P.M., Smith, S.M., Wilcock, P.R., 2008. Stream restoration strategies for reducing river nitrogen loads. *Front. Ecol. Environ.* 6, 529–538.
- Darke, A.K., Walbridge, M.R., 1994. Estimating non-crystalline and crystalline aluminum and iron by selective dissolution in a riparian forest soil. *Commun. Soil Sci. Plant Anal.* 25, 2089–2101.
- Darke, A.K., Walbridge, M.R., Lockaby, B.G., 1997. Changes in Al and Fe crystallinity and P sorption capacity in a floodplain forest soil subjected to artificially manipulated flooding regimes in field mesocosms. *Wetlands Ecol. Manage.* 4, 235–244.
- Dunne, E.J., Clark, M.W., Mitchell, J., Jawitz, J.W., Reddy, K.R., 2010. Soil phosphorus flux from emergent marsh wetlands and surrounding grazed pasture uplands. *Ecol. Eng.* 36, 1392–1400.
- Ellison, M.E., Brett, M.T., 2006. Particulate phosphorus bioavailability as a function of stream flow and land cover. *Water Res.* 40, 1258–1268.
- Forsberg, B.R., Devol, A.H., Richey, J.E., Martinelli, L.A., dos Santos, H., 1988. Factors controlling nutrient concentrations in Amazon floodplain lakes. *Limnol. Oceanogr.* 33, 41–56.
- Forshay, K.J., Stanley, E.H., 2005. Rapid nitrate loss and denitrification in a temperate river floodplain. *Biogeochemistry* 75, 43–64.
- Gu, S., Gruau, G., Dupas, R., Rumpel, C., Crème, A., Fovet, O., Gascuel-Oudoux, C., Jeanneau, L., Humbert, G., Petitjean, P., 2017. Release of dissolved phosphorus from riparian wetlands: evidence for complex interactions among hydroclimate variability, topography and soil properties. *Sci. Total Environ.* 598, 421–431.
- Hamilton, P., Shedlock, R.J., Phillips, P.J., 1991. Water-quality assessment of the Delmarva Peninsula, Delaware, Maryland, and Virginia—Analysis of available Groundwater-quality data through 1987: U.S. Geological Survey Water-Supply Paper 2355-B, 65 p.
- Heiler, G., Hein, T., Schiemer, F., Bornette, G., 1995. Hydrological connectivity and flood pulses as the central aspects for the integrity of a river–floodplain system. *River Res. Appl.* 11, 351–361.
- Heiri, O., Lotter, A.F., Lemcke, G., 2001. Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. *J. Paleolimnol.* 25, 101–110.
- Hoffmann, C.C., Kjaergaard, C., Uusi-Kämpä, J., Brunn Hansen, H.C., Kronvang, B., 2009. Phosphorus retention in riparian buffers: review of their efficiency. *J. Environ. Qual.* 38, 1942–1955.
- Howarth, R.W., Billen, G., Swaney, D., Townsend, A., Jaworski, N., Lajtha, K., Downing, J.A., Elmgren, R., Caraco, N., Jordan, T., Berendse, F., Frenay, J., Kudeyarov, V., Murdoch, P., Zhao-Liang, Z., 1996. Regional nitrogen budgets and riverine N & P fluxes for the drainages to the North Atlantic Ocean: natural and human influences. *Biogeochemistry* 35, 75–139.
- Hupp, C.R., Bazemore, D.E., 1993. Temporal and spatial patterns of wetland sedimentation, West Tennessee. *J. Hydrol.* 141, 179–196.
- Hupp, C.R., Pierce, A.R., Noe, G.B., 2009. Floodplain geomorphic processes and environmental impacts of human alteration along Coastal Plain rivers, USA. *Wetlands* 29, 413–429.
- Jones, C.N., Scott, D.T., Edwards, B.L., Keim, R.F., 2014. Perirheic mixing and biogeochemical processing in flow-through and backwater floodplain wetlands. *Water Resour. Res.* 3794–7405. <https://doi.org/10.1002/2014WR015647>.

- Jones, C.N., Scott, D.T., Guth, C., Hester, E.T., Hession, W.C., 2015. Seasonal variation in floodplain biogeochemical processing in a restored headwater stream. *Environ. Sci. Technol.* 49, 13190–13198. <https://doi.org/10.1021/acs.est.5b02426>.
- Kaushal, S.S., Groffman, P.M., Mayer, P.M., Striz, E., Gold, A.J., 2008. Effects of stream restoration on denitrification in an urbanizing watershed. *Ecol. Appl.* 18, 789–804.
- Kroes, D.E., Hupp, C.R., 2010. The effect of channelization on floodplain sediment deposition and subsidence along the Potomac River, Maryland. *J. Am. Water Resour. Assoc.* 46, 686–699.
- Kronvang, B., Andersen, I.K., Hoffmann, C.C., Pedersen, M.L., Ovesen, N.B., Andersen, H.E., 2007. Water exchange and deposition of sediment and phosphorus during inundation of natural and restored lowland floodplains. *Water Air Soil Pollut.* 181, 115–121.
- Kronvang, B., Hoffmann, C.C., Drøge, R., 2009. Sediment deposition and net phosphorus retention in a hydraulically restored lowland river floodplain in Denmark: combining field and laboratory experiments. *Mar. Freshw. Res.* 60, 638–646.
- Lewin, J., Ashworth, P.J., 2014. The negative relief of large river floodplains. *Earth Sci. Rev.* 129, 1–23. <https://doi.org/10.1016/j.earscirev.2013.10.014>.
- Loeb, R., Lamers, L.P.M., Roelofs, J.G.M., 2008. Prediction of phosphorus mobilisation in inundated floodplain soils. *Environ. Pollut.* 156, 325–331.
- Maguire, R.O., Sims, J.T., 2002. Soil testing to predict phosphorus leaching. *J. Environ. Qual.* 31, 1601–1609.
- Mayer, L.M., Keil, R.G., Macko, S.A., Joye, S.B., Ruttenberg, K.C., Aller, R.C., 1998. Importance of suspended particulates in riverine delivery of bioavailable nitrogen to coastal zones. *Global Biogeochem. Cycles* 12, 573–579.
- Mayland, H.F., 1968. Effect of drying methods on losses of carbon, nitrogen and dry matter from alfalfa. *Agron. J.* 60, 658–659.
- McMillan, S.K.; Noe, G.B., 2017. Increasing floodplain connectivity through urban stream restoration increases nutrient and sediment retention. *Ecological Engineering*. In press.
- Mertes, L.A.K., 1997. Documentation and significance of the perirheic zone on inundated floodplains. *Water Resour. Res.* 33, 1749–1762.
- Meybeck, M., 1982. Carbon, nitrogen, and phosphorus transport by world rivers. *Am. J. Sci.* 282, 401–450.
- Mitsch, W.J., Day Jr., J.W., Gilliam, W.J., Groffman, P.M., Hey, D.L., Randall, G.W., Wang, N., 2001. Reducing nitrogen loading to the Gulf of Mexico from the Mississippi River basin: strategies to counter a persistent ecological problem. *Bioscience* 51, 373–388.
- Mitsch, W.J., Day, J.W., Zhang, L., Lane, R., 2005. Nitrate-nitrogen retention in wetlands in the Mississippi River basin. *Ecol. Eng.* 24, 267–278.
- Murphy, R.R., Kemp, W.M., Ball, W.P., 2011. Long-term trends in Chesapeake Bay seasonal hypoxia, stratification, and nutrient loading. *Estuaries Coasts* 34, 1293–1309.
- Nichols, A.L., Viers, J.H., 2017. Not all breaks are equal: Variable hydrologic and geomorphic responses to intentional levee breaches along the lower Cosumnes River, California. *River Res. Appl.* 33, 1143–1155. <https://doi.org/10.1002/rra.3159>.
- Noe, G.B.; Boomer, K.; Gillespie, J.L.; Hupp, C.R.; Martin-Alciati, M.; Floro, K.; Schenk, E.R.; Jacobs, A.; Strano, S., 2019. Data supporting the study of the effectiveness of floodplain reconnection on water quality functions along Potomac River, Maryland, USA, 2014–2016: U.S. Geological Survey data release. DOI: 10.5066/P9G2LP0Z.
- Noe, G.B., Hupp, C.R., 2005. Carbon, nitrogen, and phosphorus accumulation in floodplains of Atlantic Coastal Plain rivers, USA. *Ecol. Appl.* 15, 1178–1190.
- Noe, G.B., Hupp, C.R., 2009. Retention of riverine sediment and nutrient loads by Coastal Plain floodplains. *Ecosystems* 12, 728–746.
- Noe, G.B., Hupp, C.R., Rybicki, N.B., 2013. Hydrogeomorphology influences soil nitrogen and phosphorus mineralization in floodplain wetlands. *Ecosystems* 16, 75–94.
- Olde Venterink, H., Vermaat, J.E., Pronk, M., Wiegman, F., van der Lee, G.E.M., van den Hoorn, M.W., Higler, L.W.G., Verhoeven, J.T.A., 2006. Importance of sediment deposition and denitrification for nutrient retention in floodplain wetlands. *Appl. Veg. Sci.* 9, 163–174.
- Opperman, J.J., Galloway, G.E., Fargione, J., Mount, J.F., Richter, B.D., Secchi, S., 2009. Sustainable floodplains through large-scale reconnection to rivers. *Science* 326, 1487–1488.
- Orr, C.H., Stanley, E.H., Wilson, K.A., Finlay, J.C., 2007. Effects of restoration and re-flooding on soil denitrification in a leveed midwestern floodplain. *Ecol. Appl.* 17, 2365–2376.
- Owens, P.N., Walling, D.E., 2002. Changes in sediment sources and floodplain deposition rates in the catchment of the River Tweed, Scotland, over the last 100 years: the impact of climate and land use change. *Earth Surf. Proc. Land* 27, 430–443.
- Pacini, N., Gächter, R., 1999. Speciation of riverine particulate phosphorus during rain events. *Biogeochemistry* 47, 87–109.
- Patton, C.J., Kryskall, J.R., 2003. Methods of analysis by the U.S. Geological Survey National Water Quality Laboratory—Evaluation of alkaline persulfate digestion as an alternative to Kjeldahl digestion for determination of total and dissolved nitrogen and phosphorus in water. U.S. Geological Survey Water, Resources Investigation Report 03-4174.
- Ramberg, L., Wolski, P., Krah, M., 2006. Water balance and infiltration in a seasonal floodplain in the Okavango Delta, Botswana. *Wetlands* 26, 677–690.
- Reddy, K.R., DeLaune, R.D., 2008. *Biogeochemistry of Wetlands: Science and Applications*. CRC Press.
- Reddy, K.R., Kadlec, R.H., Flaig, E., Gale, P.M., 1999. Phosphorus retention in streams and wetlands: a review. *Crit. Rev. Environ. Sci. Technol.* 29, 83–146.
- Reddy, K.R., O'Connor, G., Gale, P.M., 1998. Phosphorus sorption capacities of wetland soils and stream sediments impacted by dairy effluent. *J. Environ. Qual.* 27, 438–473.
- Reddy, K.R., Wetzel, R.G., Kadlec, R.H., 2005. *Biogeochemistry of phosphorus in wetlands*. In: Sims, J.T., Sharpley, A.N. (Eds.), *Phosphorus: Agriculture and the Environment*. Agron. Monogr. ASA, CSSA, and SSSA, Madison WI, pp. 263–316.
- Rheinhardt, R.D., Rheinhardt, M.C., Brinson, M.M., Faser, K.E., 1999. Application of reference data for assessing and restoring headwater ecosystems. *Restor. Ecol.* 7, 241–251. <https://doi.org/10.1046/j.1526-100X.1999.72017.x>.
- Ries, K.G., Dillow J.J.A., 2006. Magnitude and frequency of floods on nontidal streams in Delaware. U.S. Geological Survey Scientific Investigations Report 2006-5146.
- Robertson, G.P., Coleman, D.C., Bledsoe, C.S., Sollins, P., 1999. *Standard Soil Methods for Long-Term Ecological Research*. Oxford University Press, USA.
- Roley, S.S., Tank, J.L., Stephen, M.L., Johnson, L.T., Beaulieu, J.J., Witter, J.D., 2012. Floodplain restoration enhances denitrification and reach-scale nitrogen removal in an agricultural stream. *Ecol. Appl.* 22, 281–297.
- Rosgen, D.L., 1994. A classification of natural rivers. *Catena* 22, 169–199.
- Ross, K.M., Hupp, C.R., Howard, A.D., 2004. Sedimentation in floodplains of selected tributaries of the Chesapeake Bay. In: Bennett, S.J., Simon, A. (Eds.), *Riparian Vegetation and Fluvial Geomorphology*. American Geophysical Union, Washington, D.C. <https://doi.org/10.1029/008WSA14>.
- Sandroni, V.R., Smith, C.M.M., 2002. Microwave digestion of sludge, soil and sediment samples for metal analysis by inductively coupled plasma-atomic emission spectrometry. *Anal. Chim. Acta* 468, 335–344.
- Schönbrunner, I.M., Preiner, S., Hein, T., 2012. Impact of drying and re-flooding of sediment on phosphorus dynamics of river-floodplain systems. *Sci. Total Environ.* 432, 329–337.
- Sharpley, A., Jarvie, H.P., Buda, A., May, L., Spears, B., Kleinman, P., 2013. Phosphorus legacy: overcoming the effects of past management practices to mitigate future water quality impairment. *J. Environ. Qual.* 42, 1308–1326.
- Sharpley, A.N., Smith, S.J., Jones, O.R., Berg, W.A., Coleman, G.A., 1992. The transport of bioavailable phosphorus in agricultural runoff. *J. Environ. Qual.* 21, 30–35.
- Sheibley, R.W., Ahearn, D.S., Dahlgren, R.A., 2006. Nitrate loss from a restored floodplain in the Lower Cosumnes River, California. *Hydrobiologia* 571, 261–272.
- Smith, V.H., Tilman, G.D., Nekola, J.C., 1999. Eutrophication: impacts of excess nutrient inputs on freshwater, marine, and terrestrial ecosystems. *Environ. Pollut.* 100, 179–196.
- Smolders, A.J.P., Lamers, L.P.M., Lucassen, E., Van der Velde, G., Roelofs, J.G.M., 2006. Internal eutrophication: how it works and what to do about it – a review. *Chem. Ecol.* 22, 93–111.
- Sperazza, M., Moore, J.N., Hendrix, M.S., 2004. High-resolution particle size analysis of naturally occurring very fine-grained sediment through laser diffractometry. *J. Sediment. Res.* 74, 736–743.
- Surridge, B.W.J., Heathwaite, A.L., Baird, A.J., 2012. Phosphorus mobilisation and transport within a long-restored floodplain wetland. *Ecol. Eng.* 44, 348–359.
- Tockner, K., Stanford, J.A., 2002. Riverine flood plains: present state and future trends. *Environ. Conserv.* 29, 308–330.
- U.S. Department of Agriculture, 1996. Soil survey laboratory methods manual. Soil Survey Investigations Report No. 42, Version 3.0, January 1996.
- Vidon, P., Allan, C., Burns, D., Duval, T.P., Gurwick, N., Inamdar, S., Lowrance, R., Okay, J., Scott, D., Sebestyen, S., 2010. Hot spots and hot moments in riparian zones: potential for improved water quality management. *J. Am. Water Resour. Assoc.* 46, 278–298.
- Walbridge, M.R., Struthers, J.P., 1993. Phosphorus retention in non-tidal palustrine forested wetlands of the mid-Atlantic region. *Wetlands* 13, 84–94.
- Ward, J.V., Stanford, J.A., 1995. Ecological connectivity in alluvial river ecosystems and its disruption by flow regulation. *River Res. Appl.* 11, 105–119.
- Wassen, M.J., Olde Venterink, H., 2006. Comparison of nitrogen and phosphorus fluxes in some European fens and floodplains. *Appl. Veg. Sci.* 9, 213–222.
- 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. *J. Environ. Qual.* 42, 1245–1255.
- Wood, P.J., Armitage, P.D., 1997. Biological effects of fine sediment in the lotic environment. *Environ. Manage.* 21, 203–217.
- Wyżga, B., Zawiejska, J., Radecki-Pawlik, A., 2016. Impact of channel incision on the hydraulics of flood flows: examples from Polish Carpathian rivers. *Geomorphology* 272, 10–20. <https://doi.org/10.1016/j.geomorph.2015.05.017>.
- Yarbro, L.A., 1983. The influence of hydrologic variations on phosphorus cycling and retention in a swamp stream ecosystem. In: Fontaine, T.D.I., Bartell, S.M. (Eds.), *Dynamics of Lotic Ecosystems*. Ann Arbor Science, Ann Arbor, Michigan, pp. 223–245.
- Yarbro, L.A., Kuenzler, E.J., Mulholland, P.J., Sniffen, R.P., 1984. Effects of stream channelization on exports of nitrogen and phosphorus from North Carolina Coastal Plain watersheds. *Environ. Manage.* 8, 151–160.
- Yeomans, J.C., Bremner, J.M., 1991. Carbon and nitrogen analysis of soils by automated combustion techniques. *Commun. Soil Sci. Plant Anal.* 22, 843–850.