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Research article

A method to quantify and value floodplain sediment and nutrient retention ecosystem services

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ABSTRACT

Floodplains provide critical ecosystem services to local and downstream communities by retaining floodwaters, sediments, and nutrients. The dynamic nature of floodplains is such that these areas can both accumulate sediment and nutrients through deposition, and export material downstream through erosion. Therefore, estimating floodplain sediment and nutrient retention should consider the net flux of both depositional and erosive processes. An ecosystem services framework was used to quantify and value the sediment and nutrient ecosystem service provided by floodplains in the Difficult Run watershed, a small (151 km^2) suburban watershed located in the Piedmont of Virginia (USA). A sediment balance was developed for Difficult Run and two nested watersheds. The balance included upland sediment delivery to streams, stream bank flux, floodplain flux, and stream load. Upland sediment delivery was estimated using geospatial datasets and a modified Revised Universal Soil Loss Equation. Predictive models were developed to extrapolate field measurements of the flux of sediment, sediment-bound nitrogen (N), and sediment-bound phosphorus (P) from stream banks and floodplains to 3232 delineated stream segments in the study area. A replacement cost approach was used to estimate the economic value of the sediment and nutrient retention ecosystem service based on estimated net stream bank and floodplain flux of sediment-bound N for all streams in the study area. Results indicated the net fluvial fluxes of sediment, sediment-bound N, and sediment-bound P were $-10{,}439$ Mg yr^{-1} (net export), 57,300 kg-N yr^{-1} (net trapping), and 98 kg-P yr^{-1}(net trapping), respectively. For sediment, floodplain retention was offset by substantial losses from stream bank erosion, particularly in headwater catchments, resulting in a net export of sediment. Nutrient retention in the floodplain exceeded that lost through stream bank erosion resulting in net retention of nutrients (TN and TP). Using a conservative cost estimate of $12.69 (USD) per kilogram of nitrogen, derived from wastewater treatment costs, the estimated annual value for sediment and nutrient retention on Difficult Run floodplains was $727,226 ± 194,220 USD/yr. Values and differences in floodplain nitrogen retention among stream reaches can be used to target areas for floodplain conservation and stream restoration. The methods presented are scalable and transferable to other areas if appropriate datasets are available for validation.

1. Introduction

An ecosystem services framework has been increasingly used to link ecosystem functions to human benefits (Fisher et al., 2009). Ecosystem services are broadly defined as the benefits people obtain from ecosystems (MEA, 2005). Floodplains and wetlands provide a wide array of ecosystem services including the provisioning of food and water, the regulation of floodwaters, and the supporting service of nutrient cycling and sediment retention. Quantification and valuation of these services provides land and water resources managers with information to consider societal impacts and tradeoffs associated with management decisions. Providing information on the capacity of floodplains to retain sediment and nutrients is particularly important in settings where multiple jurisdictions are coordinating restoration efforts to address

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nutrient and sediment pollution upstream of estuaries such as the Chesapeake Bay. The restoration of the Chesapeake Bay and its tributaries is guided by a series of goals agreed upon by representatives from six states and the District of Columbia. While we focus on understanding the role and value of floodplains within the Chesapeake Bay watershed, the methods we employ can be transferred to inform management efforts in other settings. The U.S. Environmental Protection Agency (US EPA) has identified sediment and nutrients as two of the primary causes of impairment in assessed rivers and streams in the United States, leading to the creation of pollution limits expressed as Total Maximum Daily Loads (TMDLs) for rivers draining into large estuaries like the Chesapeake Bay (US EPA, 2010, 2016). The Chesapeake Bay TMDL is the largest ever developed by the US EPA, setting pollution limits for waterways in Delaware, Maryland, New York, Pennsylvania, Virginia, West Virginia and the District of Columbia. States are implementing best management practices (BMPs) to meet requirements to reduce sediment and nutrient loading to rivers and estuaries. BMPs include practices such as riparian forest buffer planting, stream channel and floodplain restoration, and forest conservation. Enhanced knowledge of sources and sinks for sediments and nutrients can help inform and optimize the placement and types of BMPs implemented to meet TMDLs in Chesapeake Bay and beyond. Reducing sediment and nutrient inputs in the watershed will improve water quality in the rivers, streams, and estuaries ultimately benefiting people via ecosystem services. Water quality improvements benefit people through improved recreation (Keeler et al., 2012; Lipton, 2004), commercial fishing (Keeler et al., 2012; Lellis-Dibble et al., 2008), and aesthetics (Keeler et al., 2012; Phaneuf et al., 2008).

Floodplains within the Chesapeake Bay watershed may provide substantial ecosystem services to local and downstream communities. Floodplains are located at the interface between aquatic and terrestrial environments in areas that can intercept upland sources of sediment and nutrients as well as retain sediment and nutrients from stormwater during overbank flood events, providing two paths to improve downstream water quality (Noe, 2013). For instance, Noe and Hupp (2009) found that Coastal Plain floodplains in the Chesapeake Bay watershed accumulated a substantial portion of the annual river sediment load (trapping the equivalent of 119% of river load), nitrogen load (22%), and phosphorus load (59%). Human actions and management interventions can both positively and negatively impact the types of ecosystem services supplied by floodplains (Schindler et al., 2014). Land use decisions to develop or preserve floodplain areas result in tradeoffs and synergies between society and the ecosystem that vary across temporal and spatial scales (e.g., site versus municipality) (Felipe-Lucia et al., 2014). However, there is limited understanding of the benefits provided by floodplains at the local-county scale where management decisions are made.

This study focuses on floodplain sediment and nutrient retention defined as the storage of nitrogen, phosphorus, and sediment via biological, chemical or geomorphic processes that make constituents not readily accessible to the water column. Floodplain sediment and nutrient retention was quantified as the net of both depositional and erosive processes, including both the accumulation of sediment and nutrients from vertical deposition or lateral accretion and the export of material from the lateral erosion of stream banks. Floodplains in the Chesapeake Bay watershed likely have appreciable value in terms of both sediment and nutrient retention. Estimates of sediment deposited in-channel and on floodplains in the Piedmont near Baltimore, Maryland, range from 45–455 Mg/km/yr (Donovan et al., 2015). The net sediment balance (floodplain deposition and bank erosion) ranges from −37 (net export) to 289 (net floodplain deposition) kg/m/yr in Linganore Creek an agricultural watershed in south central Maryland, −133 to 341 kg/m/yr in Little Conestoga Creek an agricultural watershed in southeast Pennsylvania, −136 to 1097 kg/m/yr in Difficult Run a suburban watershed in northern Virginia (Schenk et al., 2013), and 146 to 593 kg/m/yr in Smith Creek an agricultural watershed in central Virginia (Gillespie et al., 2018). Previous research related rates of bank erosion and floodplain deposition in these watersheds to simple geomorphic characteristics of stream valleys, such as floodplain width, bank height, and channel width, suggesting that these characteristics have the potential to predict floodplain and stream bank water quality functions (Schenk et al., 2013).

We aim to advance the state of knowledge on the ecosystem services that floodplains provide. We present an approach to quantify floodplain sediment and nutrient retention and place an economic value on the service that floodplains provide. Study objectives were to: 1) use existing field and geospatial datasets to develop a predictive model to estimate floodplain sediment (and associated N and P) deposition and bank erosion at each stream reach, and 2) quantify the economic value of the sediment and nutrient retention ecosystem service associated with floodplains in the Difficult Run watershed. Difficult Run was identified as the study location due to the unique availability of data on floodplain sediment and nutrient net retention within the watershed which allows for geospatial predictions of water quality functions and has relevance to Chesapeake Bay restoration efforts. The approach presented is flexible enough to be transferred to other study areas if similar datasets are available.

2. Methods

2.1. Study site

The Difficult Run watershed (151 km^2) is a suburban watershed located in Fairfax County, Virginia, to the west of Washington, D.C. (Fig. 1). Roughly 127,000 people live in the Difficult Run watershed according to area-weighted bock group statistics from the U.S. Census in 2010 (Manson et al., 2017). The area has seen substantial population growth in the past 40 years and population is expected to continue to increase (Hovland et al., 2016). The median household income is $115,717 (U.S. Census Bureau, 2017). Land use in the watershed is predominantly suburban and urban development in the uplands and a forested floodplain in the lowlands that includes parkland (Table 1). The mainstem of Difficult Run is a 6th order stream that drains into the Potomac River and eventually into the Chesapeake Bay. Streams in the watershed are typically pool-riffle systems on gravel to sand beds with substantial amounts of floodplain sediment storage and trapping along the mainstem and substantial bank erosion along headwater streams (Gellis et al., 2017; Hupp et al., 2013). The watershed is located in the crystalline Piedmont with bedrock dominated by gneiss and schist. The U.S. Geological Survey (USGS) operates three streamgages in the watershed, two located on the mainstem, Difficult Run (DIFF) and Fox Lake (FOX), and one located on a tributary draining to the mainstem, South Fork of Little Difficult Run (SFLD) (Fig. 1). The USGS monitored and estimated the total annual load of sediment, nitrogen, and phosphorus from FOX and SFLD during water years 2008–2012 (Jastram, 2014) and water year 2013 in DIFF (Hyer et al., 2016).

2.2. Mapping watershed, floodplain, and channel characteristics

Aerial light detection and ranging (LiDAR) imagery was collected over the study site between April 12–14, 2012 as part of the Fauquier, Fairfax, Frederick (MD), and Jefferson County acquisition for FEMA Region 3 FY12 VA LiDAR (Dewberry, 2012). LiDAR points classified as ground and water were used to create a 3-m digital elevation model (DEM) clipped to the Difficult Run watershed with a 500-m buffer in ArcGIS 10.3.1 (ESRI, Redlands, CA). The DEM was hydrologically conditioned by breaching through pits with no downslope neighboring cells to force surface flow to continuously move downslope using Whitebox Geospatial Analysis Tools (Lindsay and Dhun, 2015; Lindsay, 2016). Pits that were not properly breached (e.g., culverts) were manually adjusted using elevation information from the DEM and aerial imagery to locate culverts under roadways. The final breached DEM

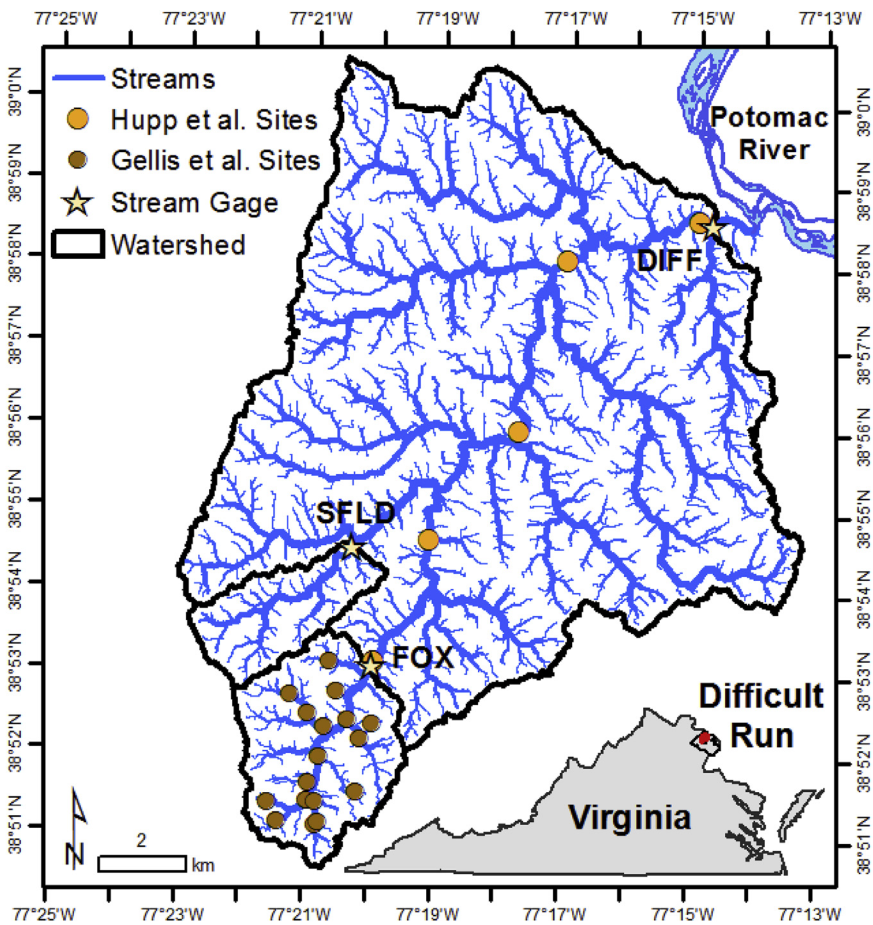

Fig. 1. Locations of streamgages and field sites in the Difficult Run watershed. The streamgages at Difficult Run (DIFF), Fox Lake (FOX), and the South Fork of Little Difficult Run (SFLD) are shown. Inset map shows the location of the Difficult Run watershed in Fairfax County, Virginia (USA).

was processed through Terrain Analysis Using Digital Elevation Models (TauDEM) to extract a stream channel network (Tarboton, 2003). A channel initiation threshold of 3000 pixels (0.027 km^2) of upstream contributing area was manually selected to best represent the true stream network based on inspection with aerial imagery and location of headwater sites from Gellis et al. (2017).

Streambank and floodplain geomorphic metrics were calculated for each stream reach in the watershed using the Stream Channel and Floodplain Metric Toolbox, hereafter referred to as the Toolbox (Hopkins et al., 2018a). The Toolbox requires four input layers from the TauDEM output: a hydrologically corrected DEM, a D8 flow direction grid, a catchment area grid for each delineated stream reach, and a shapefile of the delineated stream network. The user sets several parameters controlling cross-section spacing, sensitivity of slope break to locate stream banks, and floodplain inundation height threshold. The Toolbox output includes a floodplain extent grid, bank point locations, stream and valley cross-section locations, and fifteen metrics that are calculated for each cross-section (Table 2). If the Toolbox could not locate bank points, channel metrics were not calculated for that cross-section. Metrics were summarized for each stream reach using the median of all cross-sections on the reach.

2.3. Field estimates of floodplain deposition and bank erosion

Field measurements of floodplain deposition and bank erosion were provided by Hupp et al. (2013) at six sites along the mainstem of Difficult Run from 2008 to 2011 and two additional years of data collected from 2012 to 2013. Data from five of the six Hupp et al. (2013) sites were included in this analysis (Fig. 1). The most upstream site was excluded because of extensive bank riprap in the reach. Gellis et al. (2017) sampled an additional 24 sites in the catchment above the FOX streamgage from 2010 to 2013 (Fig. 1). Of the 24 bank erosion sites sampled by Gellis et al. (2017), five were excluded from this analysis due to lack of Toolbox estimates for stream channel metrics, and two

Table 1
Watershed characteristics for the study sites.

Name	USGS Gage No.	Drainage Area (km^2)	Floodplain Area (km^2)	Impervious Cover (%)	Natural and Forest	Turf Grass	Tree Canopy over Turf	Mixed Open (herbaceous)
DIFF	01646000	151	31.2	24%	36%	18%	15%	7%
FOX	01645704	14	3.3	32%	30%	17%	14%	7%
SFLD	01645762	7	1.2	16%	44%	19%	15%	5%
RUSLE2 C-Factor	–	–	–	–	0.001	0.03	0.01	0.08

Note: Floodplain area was determined from the output of the Stream Channel and Floodplain Metric Toolbox (Hopkins et al., 2018a). Land cover data were derived from the Phase 6 Land Classes for the Chesapeake Bay Land Change Model.

Table 2
Descriptions of stream channel and floodplain metrics.

Metric Name	Description
CHWID	Channel width (m)
BNKHT	Bank height (m)
BAREA	Cross-sectional area below bank points (m^2)
LFANG; RTANG	Angle from vertical of left and right banks, respectively (degrees)
OVERRATIO	Ratio of channel width to width just over the banks
AREARATIO	Ratio of total area under the stream transect to cross-sectional area below bank points
FPWIDTOT	Total floodplain width (m)
FPWIDNET	Total floodplain width minus channel width (m)
FPMIN; FPMAX	Minimum and maximum elevation along the valley transect (m)
FPRANGE; FPMEAN; FPSTD; FPSUM	Range, mean, standard deviation, and sum of elevation along the valley transect (m)

were excluded due to either overlap with the mainstem sites or lack of geospatial data on measurement locations. Gellis et al. (2017) measured floodplain deposition at 13 of their 24 sites, and in this study, only 10 were included because the remaining three lacked Toolbox output data on stream channel metrics.

The field approach employed in these studies is briefly described here but more detail is presented in Hupp et al. (2013) and Gellis et al. (2017). Floodplain deposition was measured at each site using a network of artificial marker layers (i.e., clay pads) located throughout the site. The depth of clay pad burial was measured annually during the study period. Floodplain vertical deposition (cm/yr) was estimated as the mean of all pads within each site. Bank erosion was measured using erosion pins, typically placed at three heights along both bank faces at three cross-sections per site. Pins were measured annually and after storm events during the study period. Lateral bank erosion (cm/yr) was estimated as the mean of all cross-sections with each site.

Floodplain deposition fluxes of sediment per meter of stream (kg/m/yr) for each site were calculated by multiplying mean vertical deposition rate by total floodplain width and the mean bulk density of floodplain sediment (Schenk et al., 2013). Bank erosion fluxes of sediment per meter of stream (kg/m/yr) for each site were calculated as the product of two times mean lateral erosion rate, bank height, and the mean bulk density of bank sediment (Schenk et al., 2013). Floodplain sites along the mainstem were located on only one side of the river and were normalized to both sides by comparing measured floodplain width on the measured side to the total floodplain width at the site (FPWIDNET) estimated by the Toolbox.

Sediment samples taken from floodplain clay pads and stream banks at the Hupp et al. (2013) mainstem sites were used to determine bulk density, total nitrogen content (TN), and total phosphorus (TP) content, see Batson et al. (2015). Floodplain and bank sediment was dried at 60 °C, weighed to calculate bulk density, ground to pass through a 1-mm sieve, and then measured for TN by an elemental analyzer (Flash 2000 CHNS analyzer, Thermo Scientific, Waltham, MA, USA) and for TP by microwave-assisted acid digestion (HF and HNO_3; Discover SPD, CEM Corporation, Matthews, NC, USA) followed by ICP-OES analysis (Optima 4300DV, Perkin-Elmer, Waltham, MA, USA). Fluxes of TN and TP were calculated by multiplying the floodplain and bank fluxes of sediment in each site by the average concentrations of TN and TP in floodplain and bank sediment, respectively. Mainstem sites were calculated using the mean nutrient concentration in that site; whereas headwater sites were calculated using the mean TN and TP of all mainstem sites.

2.4. Regression models to predict reach flux

The latitude and longitude of each field site in Difficult Run was used to select 3–5 of the surrounding cross-sections from the Toolbox spaced 9-m apart to represent the floodplain and streambank sampling locations at the field sites. The median values for the extracted geomorphic metrics for the floodplain, bank, and channel at each site were used as potential predictors for regression models of bank or floodplain fluxes of sediment and sediment-bound N and P. The upstream drainage area of each site was calculated using the Hydrology tools in ArcMap (Esri ArcMap version 10.3, Redlands, CA) and the 3-m DEM described previously.

Potential predictor variables and flux dependent variables were transformed when necessary. The best multiple linear regressions for each of the six regressions were determined using All Subsets Regression followed by ranking of models using the PRESS statistic in the R package smwrStats with the allreg.R procedure (Lorenz and De Cicco, 2017). PRESS is a cross-validation procedure that uses iterative leave-one-out computations, where included observations are used to predict the left out observation. PRESS evaluates the ability of a regression model to make predictions for unmeasured locations. The multiple linear regression with the lowest PRESS score was selected for the bank sediment, bank N, bank P, and floodplain P fluxes. However, both floodplain sediment and floodplain N fluxes regressions with the lowest PRESS score generated predictions with extremely large outliers at unmeasured locations in the watershed and the regressions with the second lowest PRESS score were used instead.

Predictions of bank and floodplain fluxes of sediment, N, and P were made for each of the 3232 delineated stream reach segments from the TauDEM output. The optimal regression for each of the six fluxes was applied to each stream reach using the median values of floodplain, bank, and channel geomorphic metrics of that reach estimated from the Toolbox. For stream reaches with drainage areas smaller than 0.05 km^2 (smaller than the streams where fluxes were measured in the field), floodplain fluxes were assumed to be zero and bank fluxes were calculated from the average bank sediment fluxes from first order streams in Difficult Run (Gellis et al., 2017). Regression confidence intervals (90%) for the predicted fluxes in each delineated stream reach were calculated.

2.5. Incorporating upland sediment into the sediment balance

A sediment balance approach was used to validate modeled floodplain and bank erosion fluxes against estimated upland delivery (U_d) and the measured stream loads (Ls). Balance calculations were made for sediment, sediment-bound N, and sediment-bound P. The balance accounted for inputs from upland delivery, floodplain flux (F_L), and bank flux (B_L) (Fig. 2). Upland sediment erosion and delivery to the stream was estimated using a modified Revised Universal Soil Loss Equation (RUSLE2) approach (USDA-ARS, 2013). The RUSLE2 erosion model is typically applied to agricultural watersheds to estimate soil erosion

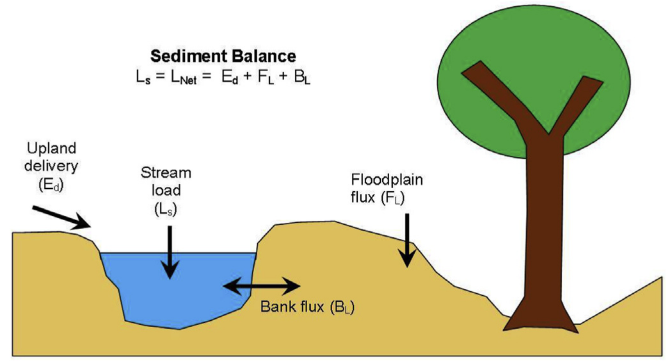

Fig. 2. Components of the sediment balance include upland delivery (E_d), floodplain flux (F_L), and bank flux (B_L). The sum of these three components should equal the in-stream load (L_s).

rates caused by rainfall and overland flow. Upland erosion (E_u) was calculated as,

$$E_u = \sum_m R_m * K * LS * C * P * Area\,(acres) \tag{1}$$

where upland erosion (E_u) in mass per area per year was calculated as the sum of the product of the erosivity factor (R_m) estimated for each month, the soil erodibility factor (K), the length-slope factor (LS), C-factor (C), farming support practices (P) assumed to be 1, and the area of each land cover grouping in acres. Rasters with 10-m resolution were created for these five variables in the modified RUSLE2 equation to create an upland erosion (E_u) raster for the study area.

An erosivity factor (R_m) raster was generated for each month of the year using 30-year (1981–2010) normal precipitation datasets from PRISM Climate Group at Oregon State University. The monthly precipitation rasters were resampled from 800-m to 10-m and units converted from mm to inches. The erosivity factor was then estimated using precipitation and power function developed by Cooper (2011) for the eastern United States,

$$R_m = 1.24{p_m}^{1.36} \tag{2}$$

where p_m is total monthly precipitation in inches. Soil erodibility factor (K) was determined from Soil Survey Geographic database (SSURGO) and the Natural Resources Conservation Service Digital General Soil Map of the United States (STATSGO2) K factor at 10-m resolution. A length-slope (LS) raster was created using the System for Automated Geoscientific Analyses (SAGA) application that estimates the effect of slope length and slope steepness on erosion (Conrad et al., 2015). SAGA provides several options for computing the LS factor. The algorithm outlined by Desmet and Govers (1997) was selected for use in calculating the LS factor due to its more generalized results compared to the alternative Unit Stream Power - based Erosion Deposition model. A vegetation C-factor (C) was estimated for each land use category in the study area based on data from Panagos et al. (2015) (Table 1).

The mass of eroded sediment delivered to the stream (E_d) was calculated with equation (3),

$$E_d = E_u(tons) * (0.083 * IC_{lc} + 0.764) \tag{3}$$

where E_u is upland erosion estimated from Equation (1) and IC_{lc} is the index of connectivity determined for each land cover class. An Index of Connectivity (IC) raster was created with a 10-m filled DEM obtained from the USGS National Elevation Dataset and processed with the SedAlp Connectivity Toolbox ArcGIS 10.1 (Cavalli et al., 2013). For every non-stream cell in a DEM, the IC represents the log-transformed ratio of upslope to downslope factors influencing the transport of upland sediment to streams. The upslope factor is composed of relative surface roughness, slope gradient, and contributing area metrics. The downslope factor is composed of path length (to a stream), relative surface roughness, and slope gradient metrics. All components of these factors were directly derived from the DEM using the SedAlp Connectivity Toolbox. Land cover cells with high connectivity were located on steep slopes near streams with large upslope contributing areas. Low connectivity cells were located along ridges, far from streams. Sediment E_u and E_d were converted to N and P fluxes using the average concentrations of TN and TP for upland soil (forest and turf) measured in Cashman et al. (In press).

The estimate of upland eroded sediment delivered to streams (E_d) and the estimates for floodplain (F_L) and bank flux (B_L) were summed and compared to the in-stream load (L_s) measured at the USGS streamgages (Fig. 2). Negative flux indicated an exporting source of sediment compared to positive flux values indicating a trapping sink for sediment. Individual stream segment fluxes were summed together to estimate the overall flux that accounted for all stream segments upstream of the three streamgages in Table 1. Uncertainty in the bank and floodplain fluxes for each reach and the sums for each of the streamgages were calculated from the reach-specific confidence intervals (Bevington and Robinson, 2003).

2.6. Economic analysis of sediment and nutrient retention ecosystem service

Sediment and nutrient retention on floodplains reduces the load of sediment and nutrients downstream. This leads to better water quality (under conditions of high nutrient loads) and ultimately benefits people by enhancing aesthetics, increasing recreational opportunities, and otherwise improving water quality. Water quality is not traded in traditional markets; therefore, deriving a value for this ecosystem service requires the use of non-market valuation techniques. Economic theory points to the quantification of consumer surplus or willingness to pay as the appropriate measure of benefits of a non-market good or service (Brown et al., 2007). To conduct this quantification, common approaches include stated preference (i.e., choice experiments) and revealed preference (i.e., hedonic property premium and travel costs) methods (Champ et al., 2017). An original survey to derive willingness to pay in the watershed would be very useful; however, that was outside of the scope of this research. A second-best alternative for valuation would be using the benefit transfer method, which applies an economic value estimated from a previously conducted study to a new policy context (Boyle and Bergstrom, 1992).

A number of studies have valued the benefits of increased water quality or clarity using aesthetics and property values (Leggett and Bockstael, 2000; Poor et al., 2007; Walsh et al., 2017), recreation benefits (Bockstael et al., 1989; Lipton, 2004; Lipton and Hicks, 2003, 1999), and commercial fisheries (Anderson, 1989; Kahn and Kemp, 1985; Mistiaen et al., 2003). However, using criteria from Richardson et al. (2015), it was determined that none of the studies could provide a credible benefits transfer value for water quality improvements for this study. Other studies used biophysical parameters as measures for improvement in water quality different than the final biophysical output for sediment and nutrient retention used in this study – kilograms of nitrogen, phosphorus, and sediment. Without data for a valid benefit transfer analysis, the next best approach would be to use the replacement cost method.

The replacement cost method is a cost-based valuation approach that monetized an ecosystem service based on the costs that would be incurred by society if the ecosystem function was degraded or ceased to exist. The ecosystem service would then presumably have to be replaced by a human-made alternative given society's reliance and demand for the service (Emerton and Bos, 2004; MEA, 2005). This is a well-documented approach for assessing the benefits of sediment and nutrient retention, with several meta-analyses demonstrating the approach (Brander et al., 2006; de Groot et al., 2012; Woodward and Wui, 2001).

Replacement costs do not directly assess consumer surplus or individuals' willingness to pay for an ecosystem service, rather it is a proxy of the value of the ecosystem service (de Groot et al., 2002; National Research Council, 2004). The approach assumes that the reduction of sediments and nutrients associated with floodplain protection is equal to or greater than the public or private cost of nutrient removal. The following conditions are required to reasonably use the replacement cost method: a) the human-made alternatives considered can provide similar services as the natural ecosystem; b) the alternative used for cost comparison should be the least cost alternative; and c) there should be significant evidence that the service would be demanded by society if it were provided by that least cost alternative (Brown, 2017; Shabman and Batie, 1978). For a more in-depth discussion of the replacement cost method see Bockstael et al. (2000), Sundberg (2004) and Conte et al. (2011).

A wastewater treatment plant (WWTP) was chosen as the human-engineered system to use as a proxy for water quality improvement via sediment and nutrient retention. WWTPs perform very similar treatment functions to that of floodplains, treating several constituents found in wastewater including N, P, and sediment (US EPA, 2004). The

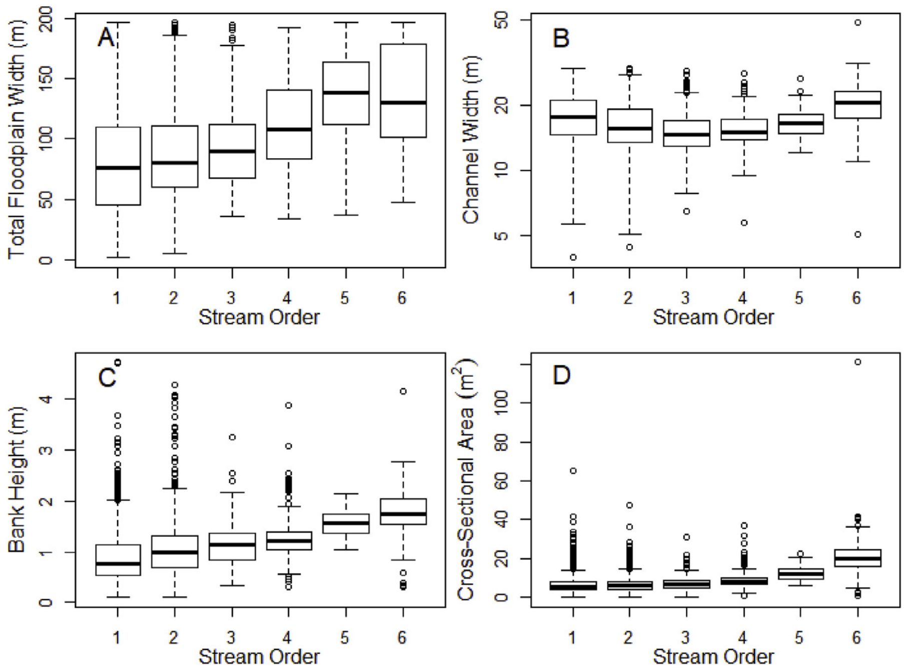

Fig. 3. Boxplots showing the distribution of Toolbox derived stream channel metrics by stream order for total floodplain width (A), channel width (B), bank height (C), and channel cross-sectional area (D). Boxes represent 25th and 75th percentile, whiskers are the highest value within 1.5 of the inter quartile range, and the solid line indicates the median.

Table 3

Optimal regressions to explain spatial variation in measured stream bank and floodplain fluxes, as determined by All Subsets Regression with validation using the PRESS statistic. Predictor variables were the same for all floodplain fluxes, and the same for all stream bank fluxes. See Table 1 for abbreviations for Toolbox output geomorphic metrics.

Flux	R^2	P	Predictor variables (see Table 2)
Floodplain sediment (log10)	0.85	< 0.001	Drainage area (log10), AREARATIO (log10), BNKHT
Floodplain sediment-N (log10)	0.85	< 0.001	
Floodplain sediment-P (log10)	0.84	< 0.001	
Streambank sediment	0.47	0.096	FPWIDNET, FPWIDTOT, BNKHT:FPWIDNET (log10), BFAREA, average (LFANG, RTANG) (sqrt), Drainage area (log10)
Streambank sediment-N	0.46	0.107	
Streambank sediment-P	0.46	0.106	

Blue Plains wastewater treatment plant, located in the District of Columbia, is responsible for treating the wastewater produced in the Difficult Run watershed. Since WWTPs treat several different contaminants simultaneously it is challenging to derive a value for individual pollutants. The treatment costs for nitrogen removal was used as a proxy for both nutrients (N and P) and sediment and the associated improvement in water quality to avoid double counting and to be conservative in the valuation.

Different technologies provide different levels of treatment at WWTPs. Data was available for three different levels of N removal achieved at Blue Plains: total nitrogen concentrations of treated effluent discharged to the river at 7.5 mg-N/L, 5.0 mg-N/L and 3.9 mg-N/L. The lower the outgoing N concentration, the higher the marginal cost of treatment due to increasingly expensive technologies. Data on treatment costs associated with different N concentrations limits for effluent were obtained from US EPA (2014). The capital expenditure for the Blue Plain's from 14 mg/L to 7.5 mg/L was 16 million USD (US EPA, 2014). The technology implemented to further reduce the concentration from 7.5 mg/L to 5.0 mg/L cost 130 million USD (US EPA, 2014). The Blue Plain Enhanced Nitrogen Removal project, put in place to achieve the US EPA nitrogen standards and reduce the N concentration from 5.0 mg/L to 3.9 mg/L, had an estimated cost of more than 1 billion USD (DC Water, 2012). To meet the second criteria for the use of replacement costs (e.g. least cost alternative, see Sundberg, 2004) lowest cost technology installed at the Blue Plains facility was used. For the third criteria related to demand, the service of water treatment was assumed to be in demand by society. The 2010 Chesapeake Bay TMDL was established by the US EPA and set into place regulatory limits on nutrient and sediment discharge in the watershed. Local jurisdiction are developing Watershed Implementation Plans detailing how and when they would meet pollution allocations, including specific practices such as stream and floodplain restoration. The TMDL sets a limit of 185.9 million pounds of nitrogen (25% reduction), 12.5 million pounds of phosphorus (24% reduction) and 6.45 billion pounds of sediment (20% reduction) per year (US EPA, 2010). Therefore, there is a "political willingness to pay" whereby society's willingness to pay is impacted by political decisions (e.g. regulatory requirements, laws, fees) regarding environmental goods and services such as water quality standards (Sundberg, 2004).

The total value of the floodplain sediment and nutrient retention ecosystem service was calculated using equation (4):

$$V_w = R_i \times C_i \tag{4}$$

where V_w is the value of sediment and nutrient retention (USD), R_i is the marginal quantity of net N retained by the floodplain (kg; floodplain – stream bank fluxes), and C_i is the marginal cost of N retention based on WWTP treatment costs (USD/kg). The source of total capital, operating, and maintenance costs were derived from the U.S. EPA (2014) and DC Water (2012). The cost per kg of N removed at the Blue Plains WWTP was determined with the following equations;

$$CNR = AC/NR \tag{5}$$

$$AC = CC + OM \tag{6}$$

where the cost of N removed in annual dollars per kg (CNR) equals the

Table 4
Results from predictions for sediment balance components for the three study sites. Values indicate the mean ± the uncertainty in parentheses which report the 90% confidence interval of predicted fluxes and the 95% confidence interval for monitored in-stream suspended load. Positive values indicate a net sink (deposition). Negative values indicate a net source (erosion). Net fluvial load is the sum of predicted floodplain load, predicted headwater bank load, and predicted non-headwater bank load. Net watershed load is the sum of upland delivery and net fluvial load.

Site	Upland Erosion (kg/yr)	Upland Delivery (kg/yr)	Upland Trapping (kg/yr)	Predicted Floodplain Load (kg/yr)	Predicted Headwater Bank Load (kg/yr)	Predicted Non-Headwater Bank Load (kg/yr)	Net Fluvial Load (kg/yr)	Net Watershed Load (kg/yr)	Monitored In-Stream Load (kg/yr)
Sediment									
DIFF	−2,567,494	−1,210,499	1,356,995	31,253,636 (± 15,522,305)	−12,376,039 (± 231,698)	−29,316,810 (± 1,471,295)	−10,439,212 (± 15,593,600)	−11,649,711	−7,576,809 (± 3,875,494)
FOX	−108,782	−51,111	57,671	1,067,481 (± 54,061)	−1,256,924 (± 74,798)	−2,744,220 (± 326,550)	−2,933,662 (± 339,341)	−2,984,773	−4,204,195 (± 1,521,655)
SFLD	−128,767	−60,483	68,283	514,632 (± 19,556)	−575,707 (± 50,520)	−1,330,070 (± 292,606)	−1,391,145 (± 297,579)	−1,451,628	−2,640,070 (± 1,066,704)
Sediment-bound N									
DIFF	−13,787	−6500	7287	85,820 (± 15,270)	−8478 (± 158)	−20,035 (± 1033)	57,307 (± 15,305)	50,807	−63,320 (± 2529)
FOX	−584	−274	310	2984 (± 53)	−861 (± 51)	−1856 (± 229)	267 (± 241)	−7	−13,331 (± 990)
SFLD	−691	−325	367	1442 (± 19)	−394 (± 34)	−911 (± 205)	136 (± 209)	−189	−8105 (± 481)
Sediment-bound P									
DIFF	−2157	−1017	1140	14,016 (± 20,330)	−4096 (± 76)	−9821 (± 505)	98 (± 20,323)	−919	−6229 (± 1327)
FOX	−91	−43	48	484 (± 61)	−416 (± 25)	−908 (± 112)	−840 (± 130)	−883	−1354 (± 309)
SFLD	−108	−51	57	232 (± 21)	−191 (± 16)	−444 (± 100)	−402 (± 104)	−453	−746 (± 216)

Notes: Export for FOX and SFLD are average annual export from WY 2008–2012 from (Jastram, 2014) and export for DIFF represents the load during WY 2013 (Hyer et al., 2016). Headwater streams are defined as reaches with a drainage area less than 0.05 km^2.

annual cost in USD (AC) divided by annual N removal (NR), i.e., the average cost of removing a kg of N. The AC was the sum of annualized capital costs (CC) and yearly operating and maintenance costs (OM). Estimates for CC and NR were calculated as;

$$CC = TC \times [1 \div (1 + i)^{UL}] \quad (7)$$

$$NR = DF \times CR \times 8.34 \times 365 \quad (8)$$

where CC equals the total project cost (TC) in dollars times a discount rate (i), set at 7%, to convert future costs into present dollars and the useful lifetime of the project, set at 20 years. Annual N removal was calculated as the WWTP design flow (DF) set at 370 mgd for Blue Plains times the reduction in N concentration attributable to technology set at 14 mg-N/L – 7 mg-N/L and unit conversion factors. The CNR was applied to the net retention of kilograms of N on the floodplains in Difficult Run to determine the ecosystem service value.

Data generated during this study are available at USGS ScienceBase, https://doi.org/10.5066/F7K936HB.

3. Results

3.1. LiDAR derived stream and floodplain metrics

Stream and floodplain metrics were calculated for all stream reaches in the Difficult Run watershed using the Toolbox and a 3-m DEM. Stream and floodplain metrics were compared across stream orders from first to sixth order reaches (Fig. 3). First order streams composed 52% of the total stream length followed by 2nd order (25% total length), 3rd order (11%), 4th order (8%), and 5–6th order (5%) streams. Median floodplain width, bank height, and channel cross-sectional area all increased with stream order (Fig. 3). For first and second order streams, the Toolbox generally estimated the width of the valley for floodplain width.

3.2. Regression models to predict reach flux

Stream reach geomorphic metrics successfully explained spatial variation in the measured fluxes of floodplain or stream bank sediment, sediment-bound N, and sediment-bound P throughout the Difficult Run watershed. The optimal multiple linear regression, identified by All Subsets Regression with PRESS validation, explained 47% of the variation in stream bank sediment fluxes, 85% of floodplain sediment fluxes, 46% of stream bank N fluxes, 85% of floodplain N fluxes, 46% of stream bank P fluxes, and 84% of floodplain P fluxes, using a variety of geomorphic metric predictor variables (Table 3). The optimal regression for bank fluxes may explain less of the variation in stream bank fluxes because of greater variability in Toolbox output for floodplain width and bank height that are used as predictor variables (Fig. 3). Each of the six flux regressions were then applied to each of the 3232 delineated stream reaches in the Difficult Run watershed to provide flux estimates for each stream reach.

3.3. Watershed sediment balance

Fluxes for each reach were summed to provide a watershed-scale estimate of floodplain trapping and stream bank flux in the Difficult Run watershed. Sediment balances were constructed for the three gaged study catchments using predictions for upland sediment erosion and upland sediment delivery derived from RUSLE2, stream bank flux, and floodplain flux to estimate the cumulative net watershed load to the stream (Fig. 2). Separate balances were constructed for each study catchment for sediment, sediment-bound N, and sediment-bound P. The net watershed sediment load for all three catchments was negative indicating a net source of sediment to the stream (Table 4). Results indicated the net fluvial load (balance of floodplain and stream bank) of sediment from DIFF was $-10{,}439\ Mg\ yr^{-1}$, with substantial floodplain

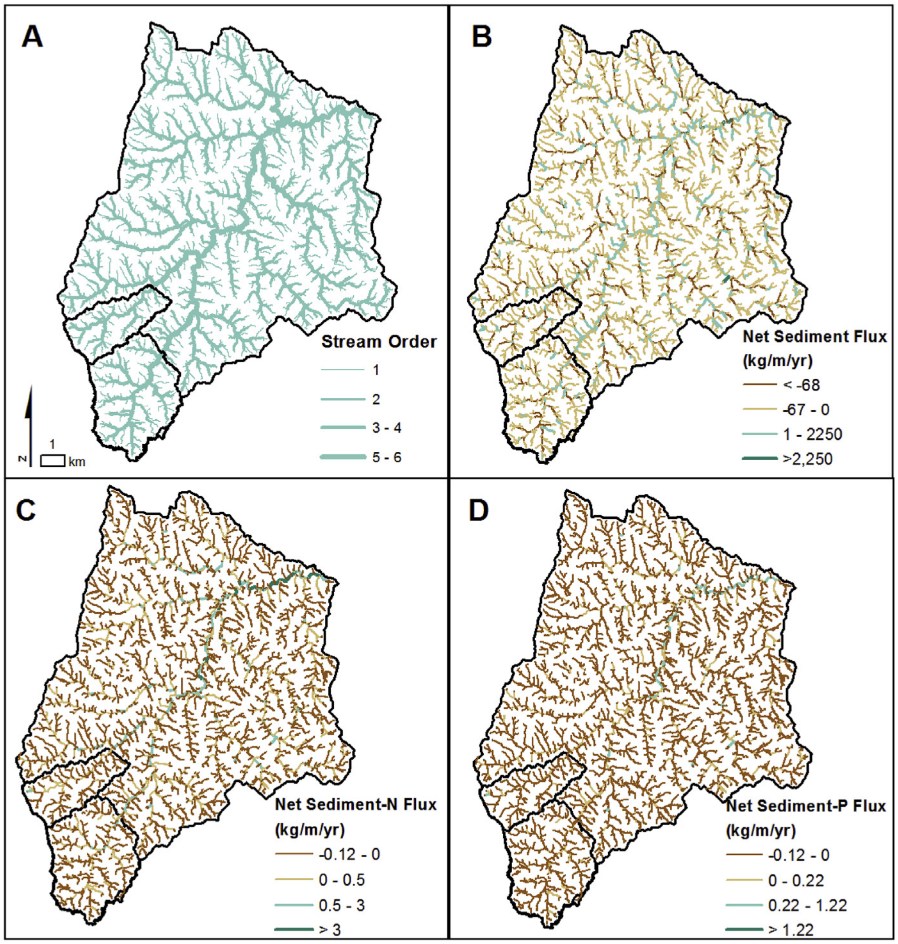

Fig. 4. Spatial patterns in stream order (A), reach estimates of net fluvial flux (stream bank plus floodplain flux) of sediment (B), sediment-N (C), and sediment-P (D) for streams in the Difficult Run watershed. Negative values indicate reaches that are net erosional (brown lines) whereas positive values indicate reaches that are net depositional (blue lines). Black outlines show the watershed for DIFF and sub-watershed boundaries for FOX and SFLD. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

trapping (31,254 Mg yr^{-1}) more than offset by substantial fluxes from stream bank erosion (−41,693 Mg yr^{-1}), and relatively little upland sediment delivery to streams (1211 Mg yr^{-1}), resulting in net export of sediment (Table 4). Results from FOX and SFLD also indicated that the much smaller amount of floodplain trapping was greatly offset by substantial bank erosion in the headwater catchments. Sediment bank erosion from larger streams (i.e., watershed area > 0.05 km^2; predicted using regression) was more than two times that of headwater streams (i.e., watershed area < 0.05 km^2; first order average) at all three sites (Table 4).

Estimates of upland sediment erosion and delivery for DIFF, FOX, and SFLD indicated that roughly 53% of upland sediment eroded was trapped in uplands before delivery to streams. Floodplain sediment flux roughly balanced bank sediment flux from non-headwater reaches in DIFF. Annual estimated sediment deposition on the floodplains was 4.12, 0.25, and 0.19 times the annual measured stream load in DIFF, FOX, and SFLD, respectively. Annual estimated bank erosion was 5.5, 0.95, and 0.72 times the annual measured stream load in DIFF, FOX, and SFLD, respectively (Table 4). The estimated watershed load sediment balance (i.e., upland sediment delivery to streams, floodplain flux, and bank flux) was 1.5 times the measured stream load in DIFF and 0.7 and 0.6 times the stream load in FOX and SFLT, respectively. The differences in net watershed loads versus stream loads for FOX and SFLT were within the ranges of uncertainty.

3.4. Watershed sediment-N and -P balances

The net watershed load for sediment bound-N was positive in DIFF indicating a net fluvial storage of particulate N, but negative in FOX and SFLD indicating a net export of particulate N (Table 4). Results indicated the net fluvial load of sediment bound-N from DIFF was 57,307 kg-N yr^{-1}, with substantial floodplain trapping (85,820 kg-N yr^{-1}) offset by smaller fluxes from stream bank erosion (−28,513 kg-N yr^{-1}), resulting a net trapping of sediment bound-N (Table 4). In contrast, floodplain N trapping was roughly equal to bank N erosion in FOX and SFLD. Bank sediment-N flux was greater from large streams than headwater streams in all three watersheds. Estimates of upland sediment bound-N erosion and delivery for DIFF, FOX, and SFLD indicated that roughly 53% of the upland sediment bound-N that was eroded was trapped in the upland (Table 4). Although the estimated net watershed sediment bound-N load was positive (net trapping) for DIFF, the measured in-stream loads indicate substantial export of N from all three catchments (Table 4). The measured in-stream load includes contributions from particulate and dissolved nitrogen, while the estimated net watershed load is just particulate N.

The net watershed balance for sediment-bound P was negative in all three watersheds, indicating a net source of particulate P (Table 4). Results indicated the net fluvial load of sediment bound-P from DIFF was 98 kg-P yr^{-1}, with substantial floodplain deposition (14,016 kg-P yr^{-1}) offset by slightly smaller fluxes from stream bank erosion (−13,918 kg-P yr^{-1}), resulting a net trapping of sediment bound-P

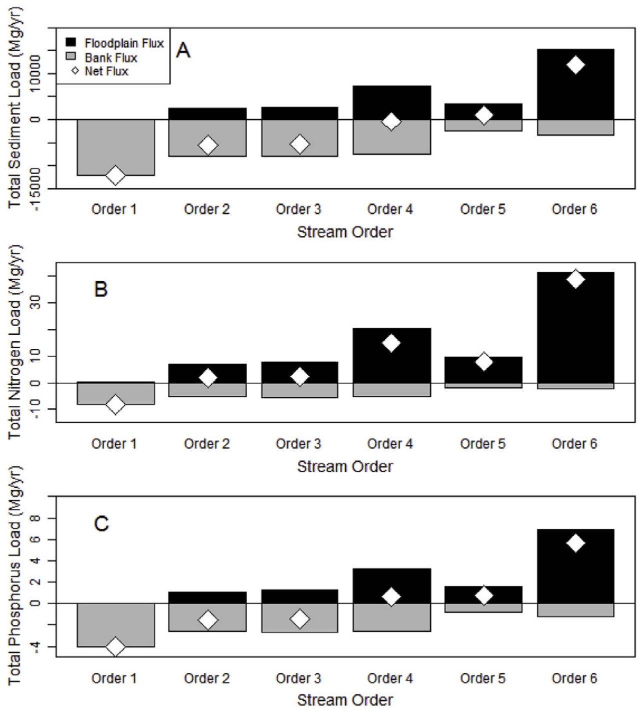

Fig. 5. Contributions from floodplain and stream bank flux by stream order for sediment (A), sediment-N (B), and sediment-P (C) in the Difficult Run watershed. Diamonds denote the net flux when both bank and floodplain flux are considered.

(Table 4). Bank P flux was roughly 0.9 times that of floodplain P deposition in FOX and SFLD. When upland delivery of sediment bound-P was added into the balance, the net watershed load indicated net export of P. Estimates of upland sediment bound-P erosion and delivery for DIFF, FOX, and SFLD indicated that roughly 53% of the upland sediment bound-P that was eroded was trapped in the upland (Table 4). The net watershed P load for FOX and SFLD was slightly below the monitored in-stream total P load, while the net watershed P load for DIFF was substantially less than the monitored in-stream total P load (Table 4). The measured in-stream load includes contributions from particulate and dissolved phosphorus, while the estimated net watershed load is just particulate P.

3.5. Spatial patterns in fluxes

Fluxes of sediment, sediment-N, and sediment-P were spatially variable in the Difficult Run watershed (Fig. 4). Reach-scale floodplain and bank fluxes were summed by stream order to estimate the overall flux of all stream reaches in each stream order category (Fig. 5). Headwater streams (catchment < 0.5 km^2) contributed the largest cumulative bank erosion flux of total sediment, sediment-bound N, and sediment-bound P, whereas floodplain depositional fluxes were assumed to be zero, indicating headwaters are a net source of sediment and nutrients (Fig. 5). Sixth order streams (3% total stream length) contributed the largest cumulative floodplain flux of total sediment, sediment-bound N, and sediment-bound P, indicating high-order floodplains are a large net sink. Streams shifted from net erosional to net depositional at fifth order streams for sediment, second order streams for N, and fourth order streams of P (Fig. 5).

3.6. Value of sediment and nutrient retention ecosystem services

The average cost of removing an additional kilogram of N is $12.69, $41.44, and $253.87 for treatment levels 7.5 mg/L, 5.0 mg/L, and 3.9 mg/L of TN, respectively. Table 5 provides details on the capital and operating and maintenance costs for the Blue Plains WWTP. The most conservative cost estimate, 12.69 USD/kg of N, was applied to the net fluvial load for sediment bound-N (accounting for floodplain deposition as well as stream bank erosion). For DIFF, the net fluvial N load was 57,307 ± 15,305 kg-N/yr (floodplain N trapping > stream bank N loss) resulting in an annual value for sediment-bound N retention of 727,226 ± 194,220 USD/yr for the entire floodplain area in the Difficult Run watershed (Table 6). The annual value of sediment-bound N retention, based on the net N fluvial load, for FOX and SFLF was $3388 and $1726 USD/yr, respectively (Table 6). If only floodplain flux was considered, neglecting losses from bank erosion, the annual value of sediment and nutrient retention for DIFF would be 1,089,056 USD/yr (± $193,776) (Table 6).

4. Discussion

Floodplains within the Difficult Run watershed provide appreciable ecosystem service benefits by trapping sediment, nitrogen, and phosphorus. Monetization of sediment and nutrient retention using the replacement cost approach indicted floodplains within the Difficult Run watershed provide net economic benefits valued at 727,226 ± 194,220 USD/yr for the entire floodplain area and 233 ± 61 USD per hectare of floodplain annually. This value is based on a conservative estimate using the least cost technology currently installed at the Blue Plains WWTP. Additional technologies for nitrogen removal are far more expensive on a per kg basis (Table 5). If we consider the upper bound value using a more costly technology (253.87 USD/kg-N) the total value would be 14,558,000 USD (4663 USD/ha/yr) – 20 times the estimated value based on the least cost technology. One limitation of basing the value on the cost of WWTP nitrogen removal is that we assume WWTP costs are representative of the demand for nitrogen removal from streams draining to Chesapeake Bay. The floodplain value we computed using a replacement cost approach likely represents the upper bound on floodplain value compared to the true willingness to pay for nutrient trapping on floodplains.

Estimates of the value of water quality improvements attributed to floodplains and wetlands are highly variable and typically do not account for both floodplain trapping and bank erosion. Our study provides a valuation that accounted for both depositional and erosive processes within the floodplain system (Table 6). The estimated value of Difficult Run floodplain sediment and nutrient retention (233 ± 61 USD/ha/yr) was within the range reported in other studies. Costs from other studies were adjusted based on an average inflation rate of 2.29% to represent the value in the year 2014 to match the year of our cost estimates. The value of floodplains in the Danube watershed in Europe

Table 5
Cost per kilogram of nitrogen removed at Blue Plains WWTP based on three treatment levels for total nitrogen (TN) reductions. All other costs are reported in millions of dollars.

Treatment Level Reduction in TN	Reduction in TN concentration (mg/L)	Wastewater flow treated (Mg/d)	TN load removed (kg/year)	Total Capital Costs (M$)	Total O&M Costs (M$/year)	Total Annual Costs (M$/year)	Cost per kg of N removed
14.0 to 7.5 mg/L	6.5	370	3,318,179	$16	$40.6	$42	$12.69
7.5 to 5.0 mg/L	2.5	370	1,276,223	$130	$40.6	$53	$41.44
5.0 to 3.9 mg/L	1.1	370	561,538	$1080	$40.6	$143	$253.87

Table 6
Ecosystem service value for floodplain sediment-bound N retention at each site. Values in parenthesis indicate error estimates based on the 90% confidence interval for sediment-nitrogen flux estimates.

Site	Value of Floodplain Flux (USD/yr)	Value of Stream Bank Flux (USD/yr)	Value of Fluvial Net Load (USD/yr)
DIFF	$1,089,056 (± $193,776)	-$361,830 (± $15,114)	$727,226 (± $194,220)
FOX	$37,867 (± $673)	-$34,479 (± $3553)	$3388 (± $3058)
SFLD	$18,299 (± $241)	-$16,561 (± $3033)	$1726 (± $2652)

was estimated using benefits transfer at 335 USD/ha/yr (162 ECU/ha/yr in 1991) for N trapping and 104 USD/ha/yr (50 ECU/ha/yr in 1991) for P trapping, which accounts for more than half the total value provided by Danube floodplains, estimated at 775 USD/ha (374 ECU/ha in 1991) (Gren et al., 1995). The value of floodplain wetlands in the Elbe Basin located in Germany was estimated using replacement costs at 2111 USD/ha/yr (1716 €/ha in 2012) based replacement cost methods and a 5% reduction in TN and TP loads to the river (Grossmann, 2012). The annual value of wetland treatment services was 67 USD/ha/year for wetlands in the Peace River watershed in British Columbia, Canada, which reflected the replacement costs to treat the excess of N and P from the water system (Wilson, 2014). In British Columbia, Olewiler (2004) used replacement costs to estimate wetland sediment and nutrient retention services are worth 567 USD/ha/year in the Lower Fraser Valley. A global assessment using a benefits transfer approach placed a 2666 USD/ha/year (1669/USD/ha/yr in 1994 USD) value on swamps and floodplains for waste treatment (Costanza et al., 1997). Our ecosystem services assessment in Difficult Run focused on one ecosystem service, valuing improvements in water quality, bringing to light only a portion of the total value and benefits provided by floodplains in this watershed. Floodplain ecosystems have been demonstrated to be very effective at delivering a multitude of ecosystem services from flood attenuation to water supply to habit and recreation (Schindler et al., 2016). The approach presented in our study provides a flexible and scalable method to quantify sediment and nutrient retention services provided by floodplain in other study areas.

The sediment and nutrient trapping services provided by floodplains are spatially variable within the Difficult Run watershed (Fig. 4). When the net balance (erosion and deposition) was considered, higher order streams (order > 4) provided the majority of net trapping for sediment and nutrients, and lower order streams contributed a substantial amount of sediment from stream bank erosion (Fig. 5). This finding is consist with sediment balance constructed by Schenk et al. (2013) for Difficult Run indicating that higher order streams provided substantially more total sediment deposition with net sediment accretion increasing with basin area. Colonial agricultural practices in the Difficult Run watershed likely resulted in substantial upland sediment erosion that was subsequently trapped in the lowlands as legacy sediment, with present day sediment dynamics influenced by rebound and equilibration from past agricultural practices and ongoing urbanization (Hupp et al., 2013). Even with the considerable bank erosion in the headwaters, the floodplains in the headwater catchments still trap substantial amounts of N and provide economic value (e.g., 38,000 USD/yr in FOX and 18,000 USD/yr in SFLT) because floodplains in these reaches are able to function as a net N sink despite less frequent overbank inundation and smaller sedimentation rates (Hupp et al., 2013).

While our net watershed balance reports a net sink for sediment-N in DIFF, the in-stream monitored total N load indicates substantial export from all watersheds (Table 4). This discrepancy may be attributed to our focus on particulate (sediment associated) nitrogen which does not account for contributions from diffuse sources of the dissolved constituents of nitrogen such as leaking septic systems and groundwater inputs from legacy agriculture. Dissolved N accounted for between 60 and 85% of the total annual nitrogen load exported from four small watersheds in Fairfax County, Virginia, USA, including FOX and SFLD (Jastram, 2014). Much of the dissolved N load is transported during baseflow conditions when floodplains are disconnected from the channel, therefore our balance does not account for potential diffuse N sources to the stream.

Modeled estimates of floodplain sediment deposition and bank erosion at FOX were 0.59 and 1.63 times that of sediment fluxes reported in Gellis et al. (2017), respectively. While our model estimated less floodplain retention and more bank erosion than Gellis et al. (2017), the final sediment balances reported in these two studies were within the range of error estimates. Gellis et al. (2017) accounted for additional sediment fluxes from changes to the streambed and bars, while our study accounted for upland sediment sources. Our estimate of the net watershed load would be 88% of the measured in-stream load if the Gellis et al. (2017) estimates of stream bed and bar changes ($-802\,\mathrm{Mg\,yr^{-1}}$ bed erosion and 74.7 Mg/yr bar deposition) were added into our balance, compared to 71% of the measured in-stream load without these estimates. Our estimate of upland sediment erosion at FOX ($109\,\mathrm{Mg\,yr^{-1}}$) was substantially lower than the unmeasured residual ($2130\,\mathrm{Mg\,yr^{-1}}$), assumed to sediment from upland sources, as reported in Gellis et al. (2017). Non-channel sediment sources can constitute a substantial (one-half to two-thirds) portion of the sediment load in the Maryland Piedmont, with the highest sediment yield (234 and 427 $\mathrm{Mg/km^2/yr}$) coming from suburban zero-to first-order watersheds (Smith and Wilcock, 2015). RUSLE2 calculations for DIFF indicated that roughly 47% of upland sediment load was delivered to the stream (Table 4) compared to other urban and suburban watersheds that have reported higher sediment delivery ratios from 60 to 76% in Maryland (Devereux et al., 2010).

5. Conclusions

Floodplains trap large volumes of sediment in the Difficult Run watershed providing substantial benefits in terms of sediment and nutrient retention, with higher order streams providing the bulk of the retention services. The budgets reported herein highlight the dynamic nature of fluvial systems moving from headwater streams dominated by upland sources of sediments and bank erosion, to larger mainstem reaches that can trap substantial volumes of sediment on floodplains. As development associated with population growth drives land use conversion in floodplain areas within the Difficult Run watershed and throughout the Chesapeake Bay region, natural resource managers and policy-makers need to carefully consider the benefits provided by floodplains. Primary biophysical assessments, such as the one included in this study, can aid in local land use decision making by providing spatially explicit information to enable identification of areas to conserve because of their high ecosystem service value. Spatial patterns in bank flux can also be used by managers to target restoration efforts in areas with greater bank erosion rates, while spatial patterns in floodplain flux highlights areas that could be conserved to maintain the valuable ecosystem services provided by floodplains. This ecosystem service approach can helps decision makers weigh how management intervention may influence the wide array of services floodplains provide and assess the economic, environmental, and social tradeoffs in natural resource management and decision making.

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