



Evaluating Impacts of Land-Use Change on Water Quality and Sedimentation in Downstream Estuarine Waters: a Comparative Approach

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Received: 28 October 2020 / Revised: 11 February 2022 / Accepted: 18 March 2022 / Published online: 5 April 2022
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Abstract

Watershed land use directly affects the supply of nutrients and sediments to adjacent waters. Higher loads are associated with urban and agricultural land use, often resulting in degraded water quality and increased sedimentation rates. Conservation practices can reduce these loads, but assessing the success of these practices remains challenging. This study evaluates the impacts of land-use conversion from intensive grain agriculture to conservation plantings in Trippe Creek watershed, within the Choptank basin (an estuarine tributary of Chesapeake Bay), compared to those in an adjacent reference watershed (Goldsborough Creek). Changes in both water quality constituents (total nitrogen (N) and phosphorus (P), chlorophyll *a*) and bottom sediment characteristics (particulate N and P, organic content, grain size) were compared, as well as seasonal- and decadal-scale sedimentation rates using ⁷Be (half-life 53.3 days) and ²¹⁰Pb (half-life 22.3 years), respectively. Results provide evidence for improving water quality and reduced sedimentation rates in Trippe Creek, where high concentrations of chlorophyll *a* decreased to concentrations similar to those in the reference Goldsborough Creek. Sedimentation rates remained fairly steady in Trippe Creek, but increased by ~50% in reference Goldsborough Creek. These changes are likely associated with 50% conversion of crop fields to conservation plantings and/or changes in the nature of agriculture over time, offering insight into the effects of land-use change and the difficulties of detecting them in downstream estuarine waters.

Keywords Sediment · Nutrient delivery · Nitrogen · Conservation · Water quality · Comparative analysis

Introduction

Since the 1970s, there has been concern about Chesapeake Bay water quality (EPA 1983; Boesch et al. 2001). Initial degradation was attributed to expanding human populations and sewage discharge, increasing use of fertilizers, and overharvesting of aquatic resources (Clune and Capel 2021). Extreme storm events, such as Hurricane Agnes in summer 1972, exacerbated increasing trends in surface water turbidity, while accelerating terrestrial nitrogen (N), phosphorus (P), and total suspended solids (TSS) inputs via rivers. This resulted in higher phytoplankton and epiphytic

algae, which contributed to losses of submersed aquatic vegetation (SAV) beds. In turn, shifting of primary production from SAV to phytoplankton led to increased turbidity and sedimentation that resulted in the depletion of bottom oxygen in summer and decreased fisheries yields (Orth and Moore 1984; Boesch et al. 2001; Kemp et al. 2005; Fisher et al. 2006a, 2021). As a result of these cascading ecological changes, the Chesapeake Bay Program Office (CBPO) of the US Environmental Protection Agency (EPA) was formed in 1982 to address water quality problems and tasked in 2010 to develop achievable remediation efforts as TMDL (total maximum daily load) goals (<https://www.epa.gov/chesapeake-bay-tmdl/chesapeake-bay-tmdl-fact-sheet>).

Achieving the TMDL goals requires an understanding of anthropogenic sources of N, P, and TSS. These sources are primarily related to agriculture, human waste, and lawn fertilizers (Lee et al. 2000; Kemp et al. 2005; Fisher et al. 2010). Point sources of nutrients, such as sewage and industrial discharges, are well documented because they are permitted and monitored at discrete outlets. In contrast, diffuse or nonpoint

Communicated by Dennis Swaney

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sources of nutrients from agriculture, lawn fertilizers, and septic systems are less well characterized, spatially heterogeneous, and more difficult to identify and quantify. As a result, diffuse sources of anthropogenic N, P, and TSS are less well documented (Boesch et al. 2001), although regional summaries of land-use-specific yield coefficients exist (Reckhow et al. 1980; Beaulac and Reckhow 1982; Clesceri et al. 1986; Clark et al. 2000; Schlesinger and Bernhardt 2013). These summaries generally indicate that the primary sources of N, P, and sediment are related to the production of human food and animal feed (agriculture) and the disposal of human and animal waste (Lee et al. 2001; Fisher et al. 2006b; Fisher et al. 2010; Clune and Capel 2021; Fisher et al. 2021).

Conservation practices are often used with the goal of reducing sediment and nutrient loads. For example, the Conservation Reserve Program (CRP) and Conservation Reserve Enhancement Program (CREP) were created in 1985 and 1998, respectively, to support conservation practices and voluntary retirement of agricultural lands (US Department of Agriculture, USDA 2001; Farm Service Agency, FSA 2020) and have been associated with a variety of ecosystem improvements in terrestrial and freshwater environments (Dunn et al. 1993; Johnson et al. 2016; Huang et al. 2019). As an example, Chesapeake watersheds with more CRP enrollment have been shown to have lower orthophosphate concentrations in summer (Fanelli et al. 2019). Still, assessing the success of these practices can be difficult (Palmer 2009; Clune and Capel 2021). For example, land-use change can reduce TN loading to the Chesapeake in model simulations, but models often do not consider several aspects of real-world application such as hydrologic variability and feasibility of implementation (Miller et al. 2019). Indeed, modeled effectiveness of conservation may not be realized in field measurements (Schilling et al. 2011).

Furthermore, impacts of conservation efforts on adjacent estuarine waters often remain elusive. The CBPO extensively monitors the main axis of Chesapeake Bay and its major tributaries, but the Chesapeake is a highly dissected estuary largely on the Mid-Atlantic Coastal Plain. The low topographic relief results in extensive tidal and salt intrusion into small tributaries with heterogeneous adjacent land uses resulting in variable water quality in their estuarine waters. For example, in a small tributary of the Chesapeake (Corsica River), decreases in sedimentation rates were consistent with known watershed restoration efforts, but direct linkage was hindered by lack of monitoring and appropriate reference sites for comparison (Palinkas 2013). While both known conservation efforts and monitored estuarine responses are available for some small systems (Schilling et al. 2011), most small tributaries are not monitored for sediments or nutrients, and estuarine responses to conservation efforts in a tributary can only be inferred based on comparative analysis if a suitable reference site is available.

This study evaluates the comparative approach (e.g., Geyer et al. 2000) as a tool to assess the effect of land-use changes on water quality in a downstream estuary. The study site is the watershed and estuary of Trippe Creek, located in the Choptank River, one of the tributaries to Chesapeake Bay on the Delmarva Peninsula. We used documented land-use changes from intensive grain agriculture (corn, wheat, soy) to conservation planting in a large portion of Trippe Creek watershed to evaluate downstream estuarine impacts. Previous work showed a relatively rapid decrease in groundwater nitrate after conversion to conservation planting (Bunnell-Young et al. 2017), but subsequent changes in the downstream estuary were unknown. Trippe Creek is not routinely monitored for sediments or nutrients, but it does have an adjacent watershed and estuary (Goldsborough Creek) with similar land-use history until land-use conversion in the Trippe Creek watershed enabled Goldsborough Creek to serve as a reference site. While the sizes of the watersheds differ, with the Trippe Creek watershed being > 5 times larger than the reference Goldsborough Creek watershed, land use in 1994 (see below) was quite similar except for a twofold difference in %forest cover (higher for the Trippe Creek watershed). This study compares recent estuarine water-column and sediment observations that capture differences between the two estuaries after ~ 50% conversion of grain agriculture to conservation land. Both watersheds are also likely impacted by ongoing development, increasing intensity of agriculture, and TMDL effects in the region. We assumed that the effect of these federal, state, and local programs would be similar in both watersheds, and we used the difference in responses between the two estuarine systems to estimate the effect of the land-use conversion.

The present study is driven by the over-arching hypothesis that the effect of partial land-use conversion from grain agriculture to conservation plantings is detectable in downstream estuarine waters and sediment. This over-arching hypothesis was divided into three testable hypotheses (Fig. 1):

1. The percentage of watershed land use was similar in both creeks before conversion, followed by increased development in both creeks but decreased agriculture in Trippe Creek and increased agriculture in Goldsborough Creek.
2. Nutrient (N and P) and chlorophyll *a* (chl_a) concentrations were lower in the estuary of Trippe Creek than in the reference estuary of Goldsborough Creek after partial conversion of grain production to conservation plantings.
3. Sediment and nutrient burial rates following the land-use change decreased in Trippe Creek due to reduced agriculture but increased in Goldsborough Creek due to continued development and increasing agricultural intensity.

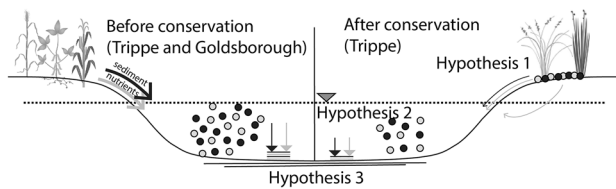


Fig. 1 Conceptual model of hypotheses. The left side shows pre-conservation conditions in both Trippe and Goldsborough Creek: primarily agricultural land use (wheat, corn, soy) leads to high runoff of sediment and nutrients from land into adjacent waters, resulting in high sediment and nutrient burial rates. These conditions are hypothesized to continue with continued agriculture and increased development in the Goldsborough Creek watershed, with increasing sedimentation rates (not shown). The right side shows expected changes in Trippe Creek with conversion to conservation plantings (e.g., grass, trees) in the Trippe Creek watershed. Symbols are from the Integration and Application Network, University of Maryland Center for Environmental Science (ian.umces.edu/symbols/)

To evaluate these hypotheses, we used geospatial analyses of land-use history in the watersheds (hypothesis 1), recent estuarine water-column observations of nutrients (hypothesis 2), and history of estuarine sediment and nutrient characteristics before and after the land-use conversion (hypothesis 3). As in many small tributaries in Chesapeake Bay with limited or no monitoring, our study is limited by available data to test these hypotheses. However, our study results provide insight into assessing land-use impacts on downstream estuarine ecosystems in areas lacking routine monitoring. Small-scale evaluations are essential for showing initial movement toward TMDL goals because larger scale responses are likely to be slower. Our results also provide an example and recommendations for other small-scale evaluations with limited data.

Methods

Study Sites—Trippe and Goldsborough Creeks

This study was motivated by documented land-use changes at Harleigh Farms, located in Talbot County on the eastern shore of Chesapeake Bay, primarily within the Trippe Creek watershed (Fig. 2). This region lies within the Coastal Plain physiographic province, which has broad, low-gradient floodplains underlain by unconsolidated sand and gravel that contribute little fine sediment to the Bay (Brush 2009; Gellis et al. 2009). Trippe and Goldsborough Creeks are adjacent tributaries of the Tred Avon River, which drains to the Choptank River—the largest tributary of Chesapeake Bay on its eastern shore. Choptank River discharge is highest in late winter and spring when the surficial groundwater aquifer approaches saturation and

is lowest in summer due to high evapotranspiration and a depleted surface aquifer (Novotny and Olem 1995; Fisher et al. 2010). Rainfall in the summer tends to occur during brief but intense thunderstorm events; N is delivered to the estuary primarily as nitrate (NO_3^-) during baseflow, whereas P and TSS are delivered primarily during stormflows (Koskelo et al. 2018).

Land use in the Choptank basin is primarily agriculture and forest (Fisher et al. 2006b), and the watershed is relatively flat, with the highest elevation < 30 m above sea level (Lee et al. 2000). Like many Chesapeake tributaries, sediment and nutrient loads were highly influenced by European colonization in the late 1600s–1800s that caused large-scale deforestation and implementation of agricultural practices (Fisher et al. 2006b; Brush 2009). In the Choptank watershed, human populations have been increasing, and agriculture has been decreasing in area but increasing in intensity since about 1900, with the net effect of increasing N and P loads from fertilizer use and human wastewater discharges (Fisher et al. 2006b).

Harleigh Farms is a collection of farms under a single owner. The farms were purchased sequentially for partial conversion from intensive grain production (corn, wheat, soy rotation) to conservation plantings such as the Conservation Reserve Program (CRP), Conservation Reserve Enhancement Program (CREP), and wetlands (Bunnell-Young et al. 2017). The earliest CRP contracts at Harleigh Farms began in October 1995 (Clay Robinson, property manager at Harleigh Farms, personal communication). Since then, agricultural fields have been successively put into conservation easements, and the property is split into $\frac{1}{3}$ forest, $\frac{1}{3}$ crop, and $\frac{1}{3}$ managed conservation to support game species such as quail. Conversion from agriculture to conservation land on individual fields from 1994 to 2006 began with the establishment or widening of a buffer strip that encircled an entire agricultural field or was installed directly adjacent to a wetland or waterway. Most of the conversion to conservation lands from 2006 to 2017 involved either the expansion of those buffers or a conversion of the whole field to conservation land. Previous work on the property focused on the recovery time of groundwater after cessation of intensive grain agriculture and found that concentrations of nitrate in waters of the surficial unconfined aquifer were reduced ~90% (~10 to ~1 mg N L^{-1}) within ~3–5 years after fertilizer applications ceased (Bunnell-Young et al. 2017). Another study examining near-surface ground- and surface-water interactions noted the strong influence of weather events and human activity on nutrient exchanges (TNC 2016). However, to our knowledge, no previous work assessed the land–estuary connection within either of the study creeks.

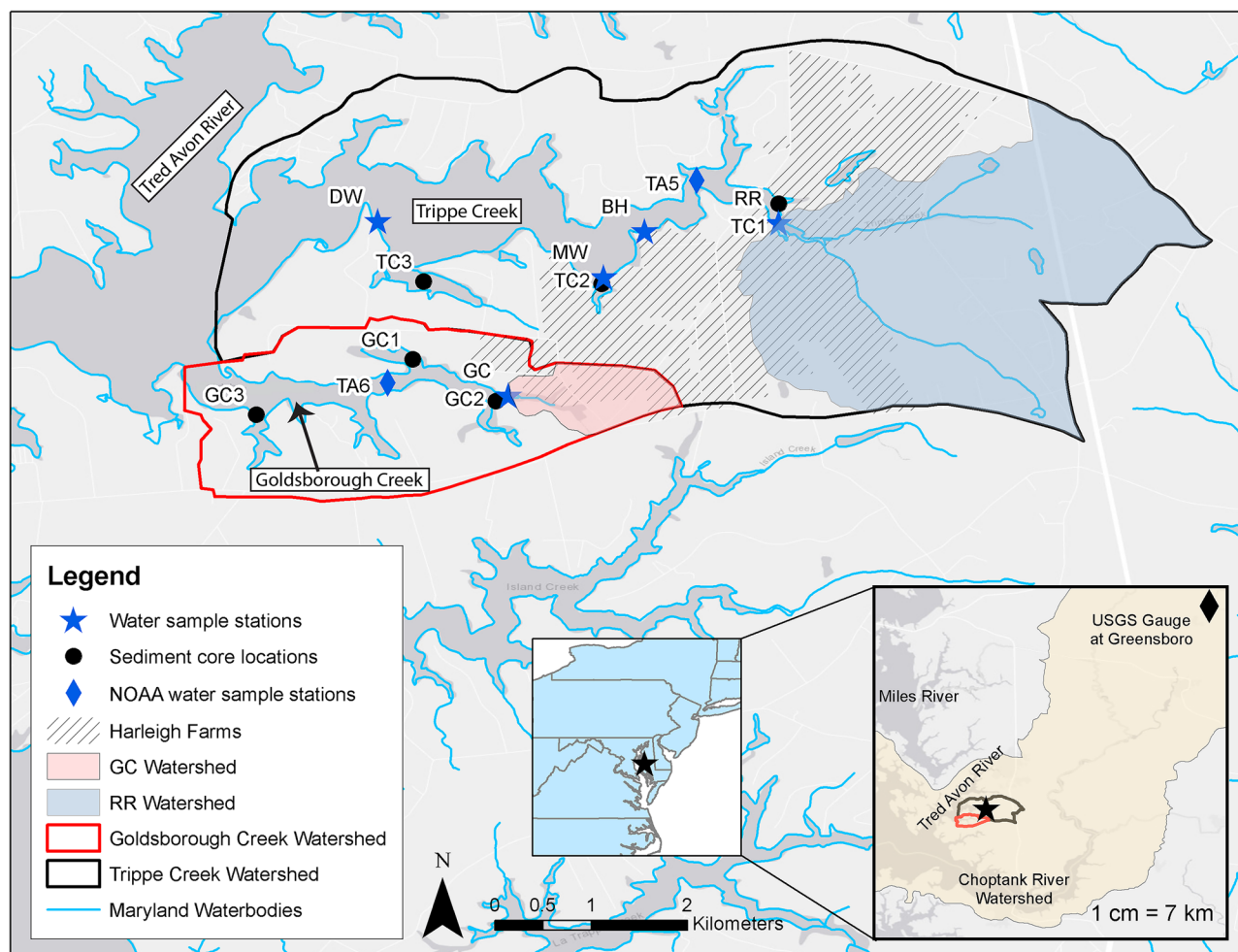


Fig. 2 Study area map showing the extent of the Trippe (black line) and Goldsborough (red line) watersheds. Sampling stations are indicated by blue stars for water-column and black circles for sediment samples, respectively

Field and Laboratory Methods

Water samples were collected in 2010 and 2016–2017 at five estuarine stations in Trippe Creek (Railroad Bridge, NOAA Trippe, Boat House, Millwood, and Deep Water Point), and at two stations in reference Goldsborough Creek (Goldsborough, NOAA Goldsborough, Fig. 2). The two NOAA stations were only collected during 2016–2017 as part of a fisheries survey (McLaughlin et al. 2018), whereas the other stations were collected in 2010 and 2016–2017 as part of a monitoring program by Harleigh Farms. Salinity of these water samples was measured by a laboratory refractometer for the Harleigh Farms monitoring program and an in situ YSI sonde at the two NOAA stations.

All water chemistry analyses were done by the Analytical Services Laboratory of the Horn Point Laboratory. Surface water samples were collected in 1-L acid-washed plastic bottles every 1–2 months for 8–17-month periods and transferred on ice within 24 h to the Analytical Services Laboratory.

Particulate nitrogen (PN) and phosphorus (PP) were analyzed on GFF filters using a CHN analyzer (Cornwell et al. 1996) and ashing/colorimetry (Aspila et al. 1976), respectively. Dissolved nutrients (NH_4^+ , NO_3^- , and PO_4^{3-}) were measured in filtered subsamples using automated colorimetric methods, and total N and P (TN, TP) or total dissolved N and P (TDN, TDP) were measured as PO_4^{3-} and NO_3^- following persulfate digestion of unfiltered or filtered subsamples in an autoclave (Valderama 1981). TN and TP in the NOAA samples were computed as the sum of TDN + PN and TDP + PP. To measure chl_a, water samples were filtered on Whatman GFF filters, which were frozen at -80°C and stored for <2 weeks prior to fluorometric analysis following standard procedures (Lane et al. 2000).

Six sediment cores (~100 cm long; 5 cm in diameter) were collected on 7 June 2016 via a hand-deployed piston corer. Three cores were located in Trippe Creek (denoted as TC1, 2, 3), and three were located in the reference Goldsborough Creek (denoted as GC1, 2, 3; see Fig. 2). Intact cores were

returned to the laboratory and sectioned into 1 cm (0–20 cm) and 2 cm (20 cm to base of the core) increments. Sediment samples from every ~2–5 cm down each core were analyzed for bulk density, grain size, organic and particulate nutrient content, and the naturally occurring radioisotopes ^7Be and ^{210}Pb . Finer-scale increments and analyses near the tops of cores were intended to improve resolution of sediment characteristics during recent years.

Bulk density was assumed to be a function of porosity, calculated from water loss after drying wet samples at 60 °C until constant sediment weight was reached, assuming sediment density of 2.65 g cm⁻³. Grain-size analyses were performed by wet sieving sediments at 64 µm to separate the mud (< 64 µm) and sand (> 64 µm) components. Organic content was determined via combustion at 450 °C for 4 h. Total particulate nitrogen (N) and phosphorus (P) analyses were conducted by the HPL Analytical Services Laboratory, as described above for water-column particulates.

^7Be (half-life 53.3 days) was used to quantify the sediment deposition rates in 2016 at all stations. ^7Be is generated naturally in the atmosphere and attaches to organic and inorganic sediment particles during wet and dry deposition (Olsen et al. 1986). The presence of ^7Be in bottom sediments of Trippe and Goldsborough Creeks requires that the material had been on land within the last ~250 days (5 half-lives, when < 5% of the original activity remains). ^7Be activities were measured via gamma spectroscopy of the 477.7 keV photopeak, using a Canberra germanium detector and following the procedure of Palinkas et al. (2005). Gamma emissions were counted for 24 h and measured activities were decay-corrected to the time of sample collection. For each core, analysis began with the topmost sample (0–1 cm increment) and proceeded with every 1-cm section down the core until no ^7Be activity was detected; one additional section was counted below this horizon. ^7Be was restricted to the top sample in all cores except the most upstream site in Trippe Creek (TC1), where it was also present in the 1–2-cm increment. Sediment deposition rates also were calculated following Palinkas et al. (2005), after normalizing activities to the corresponding mud content, since ^7Be preferentially attaches to fine-grained particles (Goodbred and Kuehl 1998).

^{210}Pb (half-life 22.3 years) was used to quantify longer-term decadal-scale sediment accumulation. ^{210}Pb is produced by the decay of ^{238}U and is supplied to the water column through precipitation, runoff, and decay of its parent ^{226}Ra (Nittrouer et al. 1979). ^{210}Pb has been used in many Chesapeake Bay environments (Brush 1989; Colman et al. 2002; Willard et al. 2003), including those with historically varying sedimentary environments (Palinkas and Koch 2012; Palinkas et al. 2017; Palinkas and Russ 2019). ^{210}Pb measurements were carried out via alpha spectroscopy following the procedure of Palinkas and Nittrouer (2007). Measured activities were decay-corrected to the

time of sample collection and normalized to the corresponding mud content of sampled depth horizons in cores. Depth-integrated ^{210}Pb inventories were calculated for each core and used to calculate sediment accumulation rates with the Constant Rate of Supply (CRS) model (Appleby and Oldfield 1978), also referred to as the Constant Flux (CF) model (Sanchez-Cabeza and Ruiz-Fernández 2012). This model does not require an assumption of steady sedimentation but rather calculates ages for discrete sediment horizons within the core (see Table S3). Supported ^{210}Pb activities were calculated via gamma spectroscopy from a weighted average of the ^{214}Pb energies (295 and 352 keV) and ^{214}Bi photopeak (609 keV) (Arias-Ortiz et al. 2018; Russ and Palinkas 2020). Several samples were analyzed for each core, and the activities were averaged. Down-core linear (cm year⁻¹) sedimentation rates were calculated by dividing the sediment age by its depth; mass accumulation rates (g cm⁻² year⁻¹) were then calculated by multiplying the linear sedimentation rate by the corresponding bulk density. The CRS model is relatively insensitive to the effects of mixing but may underestimate sedimentation rates if, as in our case, every sample down the core is not analyzed (Arias-Ortiz et al. 2018). The depth horizon corresponding to 1996 was calculated via the CRS model.

GIS Analyses

Changes in the potential contribution of shoreline-derived sediments were evaluated via GIS analyses of shoreline change, following Russ and Palinkas (2020). These methods are similar to those used in other coastal settings to quantify geomorphic changes at the land–water interface (Eulie et al. 2017; Smith and Terrano 2017; Hopkinson et al. 2018; Johnson et al. 2020). For these analyses, shoreline surveys in 1942 and 1994 by the Maryland Geological Survey (<https://data.imap.maryland.gov>) represented historical conditions, and LiDAR data collected in 2003 and 2015 were obtained via NOAA's Digital Coast (<http://coast.noaa.gov/digitalcoast>) to represent more current conditions. Horizontal accuracy of LiDAR data collected between 1998 and 2013 ranged over 0.7–1.5 m (Stockdon et al. 2002; Hapke et al. 2011; Johnson et al. 2020). Shapefiles of digitized shorelines were created as boundaries to calculate the area of water in each creek for each year. The rate of change in creek area was calculated by differencing these areas and normalizing by the time between surveys.

Individual watershed boundaries of Trippe and Goldsborough Creeks were not available in public datasets but rather were contained within a larger hydrologic unit (HUC-12) defined by the US Geological Survey (USGS) and available from the USGS National Map Viewer (<https://viewer.nationalmap.gov/basic/>). The watershed boundaries of the creeks and subwatershed boundaries of two water quality

monitoring stations within each creek, Railroad (RR) and Goldsborough (GC), were estimated using 1-m resolution LiDAR data of the HUC-12 area. Watershed boundaries were calculated from the LiDAR data with the ArcHydro toolbox in ArcGIS 10.4 (Saunders 1999; Callow et al. 2007; Li 2014). The resulting digital elevation model (DEM) was reconditioned via the stream-grid tool to burn-in known hydrology (<https://www.usgs.gov/national-hydrography/access-national-hydrography-products>) via the AGREE methodology (Saunders 1999; Baker et al. 2006; Callow et al. 2007).

Changes in land use and land cover over 23 years were evaluated at three time steps (1994, 2006, and 2017) using aerial imagery with a spatial resolution of 0.3–1 m. These years were chosen to coincide as closely in time as possible with the onset of land-use conversion at Harleigh Farms (1996) and water-column and sediment sampling (2010, 2016–2017). The imagery was acquired from Maryland iMAP (<https://imap.maryland.gov/Pages/data.aspx>). The 2017 imagery was acquired as part of the National Agriculture Imagery Program (NAIP; <https://imap.maryland.gov/Pages/imagery-download-files.aspx>). A land-cover change analysis was completed for the two intervals between the three years by digitizing the areas at a 1:20,000 map scale using the DEM-defined watershed boundaries. Land-use codes were derived from the Anderson Level 1 and Level 2 land-use and land-cover (LULC) classification system (Anderson et al. 1976). One such code, “conservation land,” is defined differently in this paper to be agricultural land that is no longer actively farmed for cash crops (e.g., corn, soybean, and wheat) and has been allowed to senesce and go fallow by the farmer and/or land owner. Differences in LULC between time steps were evaluated as percent change in area for each land use. The error rates for digitizing these watersheds were calculated by re-digitizing three polygons of varying land use and size. The percent differences were less than 1%.

Auxiliary Data

Like most small estuaries within the highly dissected lands of the lower Choptank basin, neither Trippe nor Goldsborough Creeks are routinely monitored for water discharge and chemistry. The closest USGS gauge is at Greensboro MD (USGS 1,491,000), which is ~40 km from the study site and measures discharge and water chemistry from a nearby portion of the Choptank Basin (Fig. 2). We obtained annual rainfall data (m year^{-1}) from several local meteorological stations in the Choptank Basin and water yields ($\text{m}^3 \text{ water m}^{-2} \text{ watershed area year}^{-1} = \text{m year}^{-1}$, units equivalent to rain) from the USGS gauge as proxies for local discharge.

Rainfall data were obtained from NOAA’s National Centers of Environmental Information (NCEI; <https://>

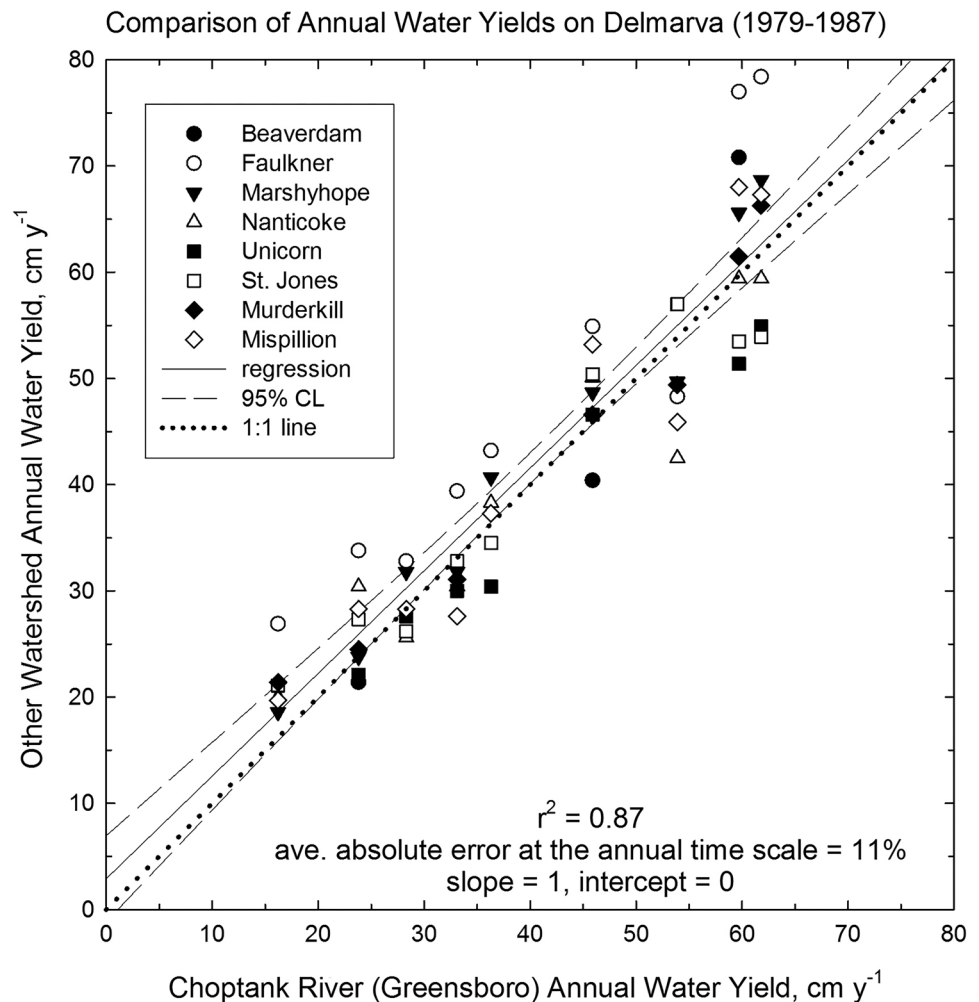
www.ncdc.noaa.gov). We requested data for the time period from 1 January 1949 (first complete year of data) to 5 May 2019 (most recent data), using the Global Historical Climatology Network (GHCN) database for stations at Royal Oak (GHCND:USC00187806), Easton (GHCND:US1MDTB0001), and Bellevue (US1MDTB0007). The Royal Oak station was the closest to the study site (~7 km) and had the longest and most complete dataset. Daily rainfall was averaged over the three stations to include spatial variability in rainfall and to cover gaps in data records, and the averaged daily values were summed over each calendar year to obtain total annual rainfall in 2010 and 2016 (m year^{-1}). Rain data were available from all stations during 2010 and 2016, and data were available from at least one station for every day of those two years.

We obtained daily water discharge ($\text{m}^3 \text{ day}^{-1}$) from the USGS Gauge near Greensboro MD (USGS 1,491,000) for 1949–2019. Discharge was normalized to watershed area (m^2) and summed by calendar year to obtain annual water yields (m year^{-1}), which are independent of watershed size, enabling comparisons between watersheds. In addition, water yields have the same units as rainfall and are typically ~40% of rainfall in the Mid-Atlantic region (Fisher et al. 2010). Figure 3 shows the relationship between water yields at various USGS gauges on the Delmarva Peninsula compared to water yields at Greensboro, MD. There is a strong correlation ($r^2 = 0.87$, $p < 0.0001$) between water yields at the eight stations, and the intercept (2.9) and slope (0.97) through the data are not different from 0 ($p = 0.16$) and 1 ($p = 0.96$), respectively. The average absolute difference between annual water yields at Greensboro and other USGS stations was 11% of the Greensboro value, which represents real spatial variations plus measurement errors. Approximately double the average absolute difference (22%) sets the limit of accuracy for the use of water yields to estimate discharge at any location on the Delmarva Peninsula at the annual time scale.

Statistical Analyses

Statistical tests were performed with R statistical software and SigmaPlot v12.5. *T* tests were used to reveal differences between measurements, grouped by location (Trippe versus Goldsborough Creek) and/or time period (< 1996, 1996–2016). ANOVA was used to assess within-creek variability. *P* values are reported for all tests, although *p* values do not measure the importance of results (Wasserstein and Lazar 2016), and management decisions should not be based solely on them (Smith 2020). Indeed, geological field data often preclude robust statistical analysis (Krumbein 1960), and the number of stations and samples in this study were relatively low. To that end, we highlight trends that may

Fig. 3 Comparison of annual water yields from USGS gauge sites on the Delmarva Peninsula. Large dashed lines are the 95% CIs of the slope (solid line), and the dotted line is a 1:1 relationship passing through zero. The intercept and slope of the linear regression are not different from 0 and 1, respectively



reflect physically meaningful processes, especially differences of > 10% in our change analysis. This value exceeds the average replication error in similar Chesapeake shallow-water environments (Palinkas and Koch 2012).

Results

Basin Descriptions and Land Uses

The watersheds of both Trippe and Goldsborough Creeks were rural in 1994, with 40–50% agricultural land use (Fig. 4). The Trippe Creek watershed area was 22.0 km², with a 5:1 ratio of land to water, and the Goldsborough Creek watershed area was smaller (6.4 km²), but with a similar ratio of land to water (4.5:1). Both contained portions of the Harleigh Farms property. However, the majority of Harleigh Farms was located in the Trippe Creek watershed (5.8 km² or 26.4% of the watershed), with only a small fraction of Harleigh Farms in the reference Goldsborough Creek watershed (0.55 km² or 8.6% of the watershed). In 1994, the year

with the highest agricultural land-use percentage, Harleigh Farms represented 63.0% and 20.4% of the agricultural lands in Trippe Creek watershed and the reference Goldsborough Creek watershed, respectively.

Changes in land use were observed between the three time steps (1994, 2006, 2017) in the Trippe Creek and reference Goldsborough Creek watersheds (Fig. 4). In 1994, both watersheds were dominated by agricultural land (40–50%). In 2006 and 2017, conservation areas in Trippe Creek watershed increased from 0.5 to 21%, and agricultural land decreased from 42 to 20%, resulting in > 50% reduction in agriculture in the Trippe Creek watershed. In the reference Goldsborough Creek watershed, conservation lands increased from 0 to 9% and agricultural land decreased from 53 to 42%, a ~20% decrease in agriculture. Most of these changes occurred during 1994–2006, but continued more slowly during 2006–2017. Areas of other land uses varied little in either watershed over time. The majority of conversions in both watersheds occurred on the Harleigh Farms property, accounting for 67% and 59% of the land-use change in Trippe and Goldsborough Creeks, respectively.

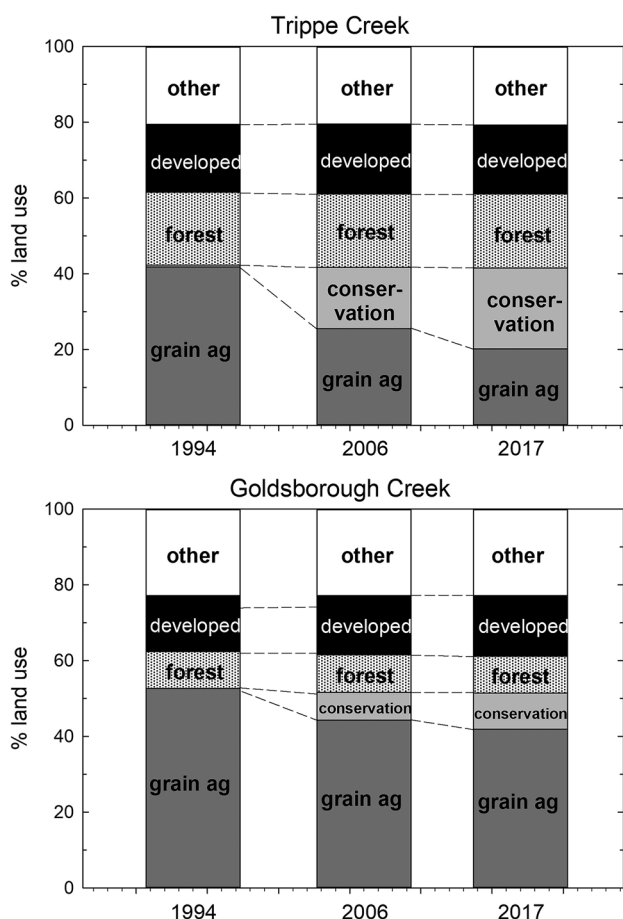


Fig. 4 Time series of land use in the watersheds of Trippe and Goldsborough Creeks based on aerial imagery taken in 1994, 2006, and 2017. Bars are centered on the year of the imagery, and dashed lines indicate linear extrapolation between analyzed imagery. The data show conversion of grain agriculture (“grain ag”) to conservation planting due to the presence of Harleigh Farm straddling the boundary between the two watersheds (see Fig. 2). Harleigh Farms lies primarily in the Trippe Creek watershed, where ~50% of grain agriculture was converted to conservation plantings. “Other” land use refers to water, wetlands, and transitional lands

These land-use changes were not distributed uniformly within each watershed. The two headwater sampling sites, RR and GC, exhibited changes on a smaller scale. For the full time period, the only land-use changes that occurred in the subwatersheds of RR and GC stations were transitions from cropland to conservation land; all other land uses were the same in total area and spatial location. The RR site subwatershed of Trippe Creek comprised 34% of the total watershed of Trippe Creek but was only 9% Harleigh Farms property. In contrast, the reference subwatershed of the GC site comprised 13% of the total watershed of Goldsborough Creek but was 56% Harleigh Farms property; i.e., land-use changes on Harleigh Farms have a potentially larger influence on the reference GC site in Goldsborough Creek

than on the RR site of Trippe Creek. However, the land-use changes by 2017 represented large changes in conservation lands compared to 1994: from 0 to 33% in the RR site subwatershed of Trippe Creek, and from 0 to 9% in the reference GC site subwatershed of Goldsborough Creek.

Shoreline Erosion

There was considerable evidence for shoreline erosion in both creeks. The total shoreline length was more than twice as long around Trippe Creek (28.8 km in 2015) than around the reference Goldsborough Creek (13.0 km in 2015). In Trippe Creek, the area between shorelines (i.e., area of water) expanded by $3.5 \times 10^3 \text{ m}^2 \text{ year}^{-1}$ between 1942 and 1994 (5.1% total change in area), and by an additional $2.6 \times 10^3 \text{ m}^2 \text{ year}^{-1}$ between 2003 and 2015 (0.75% total change in area). This represents land loss at the shoreline, with the recent (2003–2015) rate of loss ~25% lower than the historical rate (1942–1994). In reference Goldsborough Creek, the area between shorelines expanded by $1.3 \times 10^3 \text{ m}^2 \text{ year}^{-1}$ between 1942 and 1994 (8.5% total change in area), and by $0.45 \times 10^3 \text{ m}^2 \text{ year}^{-1}$ between 2003 and 2015 (0.55% total change in area). Like Trippe Creek, this represents land loss at the shoreline, with the recent rate of loss ~65% lower than the historical rate. Shoreline heights along both creeks were >0–1.5 m and were composed mostly of sandy, siliceous sediment.

Rainfall and Water Yields

Local rainfall and Greensboro water yields for the calendar years of water column sampling were used as indicators of transport from land to water. Average annual rainfall from 1949–2018 was $117.5 \pm 2.5 \text{ cm}$, highest in 2003 (167.3 cm) and lowest in 1965 (80.5 cm; Fig. 5). There is a long-term trend in annual rainfall, increasing at a rate of $0.23 \text{ cm year}^{-2}$ ($r^2 = 0.05, p = 0.059$). The highest daily rainfall was 22.6 cm on 2 November 1956, and precipitation was not measurable (0 cm) for ~65% of the days. For the two historical periods of interest for sediment transport, average \pm SE of annual rainfall was $114.4 \pm 20.5 \text{ cm}$ from 1949 to 1996 and $118.1 \pm 19.7 \text{ cm}$ from 1997 to 2018, with no differences in means between these time periods ($p = 0.27$). The additional degrees of freedom in the entire dataset were required to detect the increasing rainfall over the entire time period (Fig. 5). High-rainfall events occurred in both time periods, with similar magnitudes; maximum daily rainfall was 22.6 cm (2 November 1956, Hurricane Greta) before 1996 and 20.1 cm (16 September 1999, Hurricane Floyd) after 1996. These large storm events are likely to be the major drivers of terrestrial sediment delivery to the estuaries.

There were only small differences in annual rainfall during the years of water column sampling. For calendar year

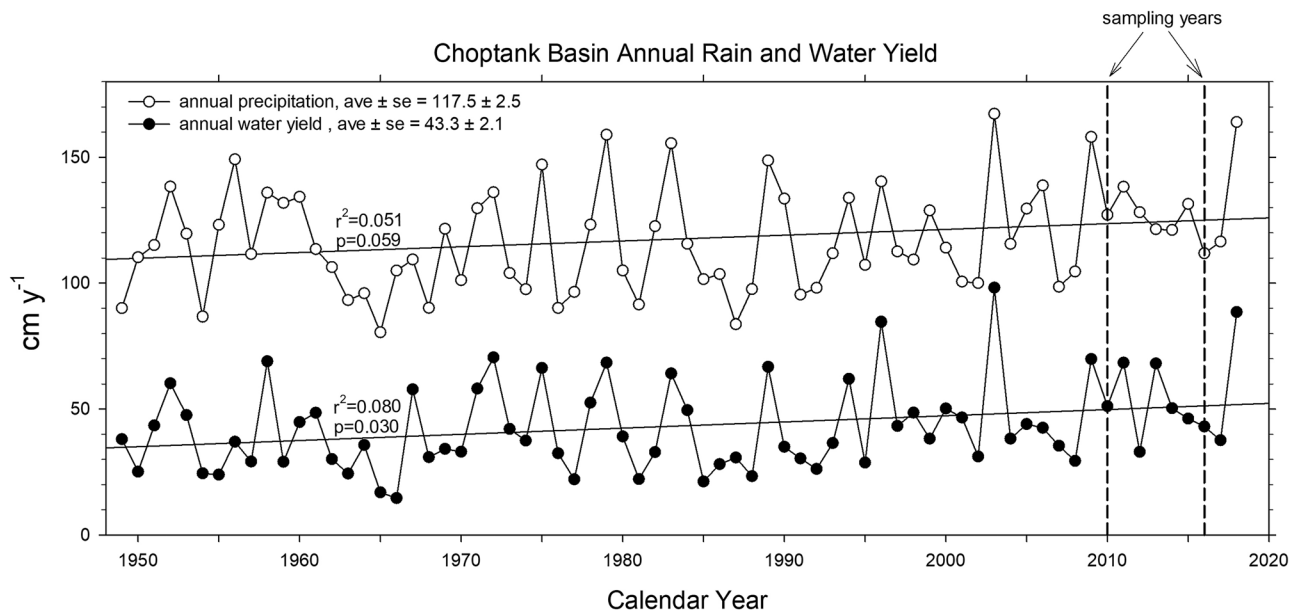


Fig. 5 Average annual precipitation data from 3 local NOAA rain gauges and annual water yields from the USGS gauging station near Greensboro MD. See Fig. 2 for locations. Water yields are discharge normalized to watershed area, with the same units as precipitation

2010, the annual precipitation was 127 cm year^{-1} , 10 cm above the long-term mean of 117 cm year^{-1} . For calendar year 2016, the total rainfall was 112 cm year^{-1} , 5 cm below the long-term mean rainfall. These data allow us to characterize 2010 as slightly (8.5%) wetter than average and 2016 as slightly drier than average (-4.3%). However, we note that December 2009, 1 month prior to the first water column sampling in January 2010, was one of the wettest Decembers in the entire rainfall record ($20.6 \text{ cm month}^{-1}$) and may have influenced the 2010 water quality samples. For the sediment sampling, especially ^7Be depositional data, rainfall data for other time scales are more relevant. For the time period coinciding with ^7Be observations in 2016 prior to core collection (77 days mean lifetime), total rainfall was 28.4 cm, which is similar to the long-term mean of $24.7 \pm 8.1 \text{ cm}$ over that same 77-day time period. Data from ^{210}Pb deposition span decades, and the long-term mean values for rainfall are most relevant.

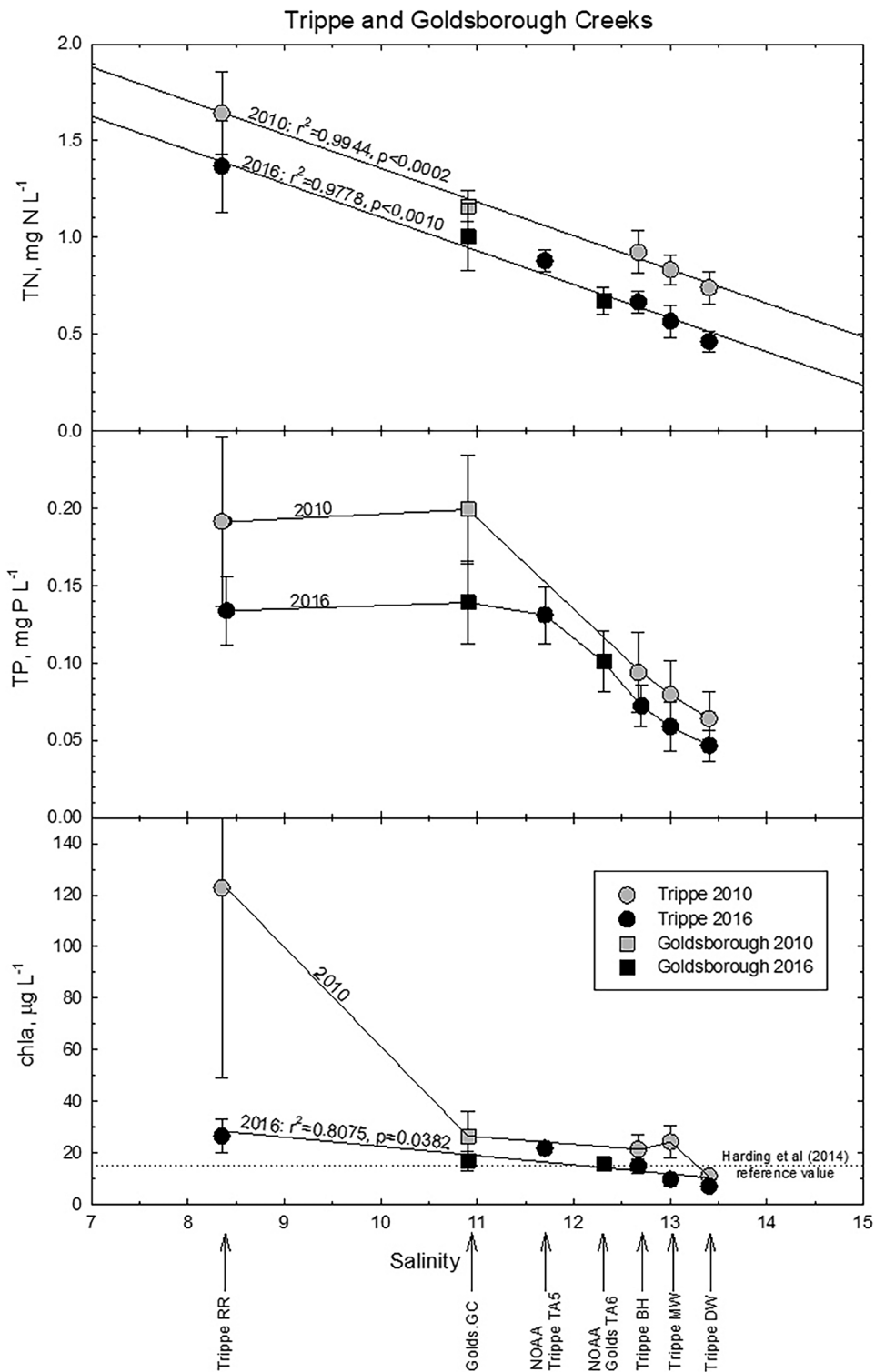
Annual water yields (cm year^{-1}) at the USGS gauge at Greensboro MD indicated conditions similar to annual precipitation (Fig. 5). The long-term mean of annual, calendar year, water yield was $43.3 \pm 2.1 \text{ cm year}^{-1}$, and there was a long-term increase in annual water yields of $0.25 \text{ cm year}^{-2}$ ($r^2=0.0803$, $p=0.0297$), largely parallel to the long-term trend in precipitation. Not surprisingly, annual water yields were correlated with annual precipitation ($r^2=0.60$, $p<0.0001$), supporting our use of average precipitation and annual water yield as measures of transfer of materials from the watersheds to the estuaries of both basins.

As for annual precipitation, there were small differences in annual water yields during the years of sampling. For calendar year 2010, the annual water yield was $51.1 \text{ cm year}^{-1}$, 7.8 cm above the long-term mean of $43.3 \text{ cm year}^{-1}$. For calendar year 2016, the annual water yield was $43.0 \text{ cm year}^{-1}$, 0.3 cm below the long-term mean. These data allow us to characterize 2010 as having 18% more discharge than average and 2016 as having only 0.01% discharge less than average. As noted above, precipitation in December 2009, prior to our initial sampling, was anomalously high and resulted in a record water yield of 19 cm month^{-1} for all Decembers since 1949. For the sediment sampling, especially ^7Be deposition, water yields for other time scales are more relevant. For the time period in 2016 represented by ^7Be (77 days mean lifetime prior to core collection), the mean water yield was 11.6 cm, similar to the long-term mean of $11.8 \pm 5.6 \text{ cm}$ over that the time period. Data on ^{210}Pb deposition spans decades, and the long-term mean water yield is most relevant.

Water-Column Observations

Salinity increased from east to west along the length of both creeks. TC1 and Railroad stations in Trippe Creek were closest to freshwater (salinity = 8.3), indicating that the salt gradient penetrated far inland in these small systems (see Fig. 2). The highest salinities were found at Deep Water Point in Trippe Creek (13.4) and the NOAA Goldsborough station (11.7) in Goldsborough Creek.

Fig. 6 Total N (TN), total P (TP), and chlorophyll *a* (chl_a) along the salinity gradients of Trippe Creek (circles) and Goldsborough Creek (squares) during the two time periods of sampling (2010 in gray, 2016–2017 in black). TN was mixed linearly along the salinity gradient in both years, and the y-axis intercepts for TN (salinity=0) were significantly different ($p < 0.0001$), indicating systematically lower concentrations of TN in 2016 compared to 2010. For TP, there were also systematically lower concentrations in 2016 compared to 2010, but both years showed a non-linear distribution with salinity, suggesting a P source near salinities of 10–12. Like TN, chlorophyll *a* declined with salinity, with systematically lower concentrations in 2016 compared to 2010. See Table S1 in the Supplemental Material for values of all water-column constituents, including the forms of nitrogen and phosphorus



Average (\pm SE) of water quality constituents (TN, TP, chl_a) at each station for 2010 and 2016 are shown in Fig. 6 and Table 1 (see Supplemental Material, Table S1 for observations at individual stations). TN, TP, and chl_a at the stations sampled along the salinity gradient of each creek are plotted against salinity to account for the effects of mixing of freshwater from

the watershed (high-nutrient, freshwater end-member) with estuarine water from the Tred Avon river (lower nutrient, more saline end-member). Note that we did not measure salinity in 2010; we assumed that average salinities in 2010 were similar to those in 2016, an assumption supported by the similar precipitation and water yields for both years reported above.

Table 1 Water-column observations averaged over the two study periods, given as mean \pm SE

	TC		GC		Trippe*		RR	
	2010	2016/17	2010	2016/17	2010	2016/17	2010	2016/17
Total Nitrogen, mg L ⁻¹	1.03 \pm 0.10	0.78 \pm 0.081	1.16 \pm 0.080	0.84 \pm 0.10	0.83 \pm 0.053	0.63 \pm 0.05	1.64 \pm 0.21	1.37 \pm 0.45
Total Phosphorus, mg L ⁻¹	0.11 \pm 0.019	0.088 \pm 0.010	0.20 \pm 0.035	0.12 \pm 0.017	0.079 \pm 0.012	0.076 \pm 0.010	0.19 \pm 0.054	0.134 \pm 0.022
Chlorophyll <i>a</i> , μ g L ⁻¹	44.9 \pm 19.9	15.7 \pm 2.1	26.6 \pm 9.33	15.8 \pm 2.3	18.9 \pm 3.1	13.0 \pm 1.7	122.7 \pm 73.4	31.5 \pm 12.2

TC Trippe Creek (all stations), GC Goldsborough Creek stations, RR Railroad station

*All TC stations except RR

These adjacent estuaries exhibited similar mixing patterns from local freshwaters to estuarine waters of the Tred Avon River, a tributary of the Choptank (see Fig. 2). Average TN values of both estuaries for each sampling period fit on one TN line in Fig. 6, indicating similar mixing of local freshwater with Tred Avon water. However, the TN data for 2016 were systematically lower than in 2010 (-0.19 mg N L⁻¹). Although TN had a negative linear relationship to salinity (dilution of freshwater TN by lower-TN, Tred Avon water), TP exhibited a convex pattern suggestive of a local P source at salinities of 10–12. Both years had parallel mixing patterns for TP, with 2016 concentrations systematically lower than those of 2010. Trippe Creek *chl a* in 2010 had a concave relationship with salinity due to unusually high algal biomass in summer 2010 in the lower salinity waters at the Railroad site. However, in 2016, *chl a* was linearly related to salinity in both creeks, with systematically lower values in 2016 compared to 2010. All *chl a* values were high, particularly in 2010, and most *chl a* values in both creeks exceeded the *chl a* criterion of 15 μ g *chl a* L⁻¹ developed by Harding et al. (2014) for Chesapeake Bay stations.

The composition of the TN and TP in both creeks was primarily organic. Average ammonium and nitrate were both ≤ 0.01 mg N L⁻¹, comprising $< 2\%$ of the TN. Total dissolved P, an analytical class that includes dissolved organic P, was ~ 0.03 mg P L⁻¹, representing $\sim 27\%$ of the TP in both creeks. Inorganic N and P were clearly small fractions of the TN and TP and likely close to limiting concentrations for phytoplankton growth (Fisher et al. 1992, 1999) due to biological interception of inorganic forms upstream of our lowest salinity stations (Trippe RR, Goldsborough GC).

Sediment Observations

Surficial (uppermost 1 cm) sediment characteristics were similar in Trippe and Goldsborough Creeks (Table 2). However, there was a notable outlier at the most upstream station in Trippe Creek (TC1, close to the Railroad water-column station, Fig. 2), where both particulate nitrogen (PN) and particulate phosphorus (PP) were higher than the other

stations. Seasonal (⁷Be-derived) sediment deposition rates in Trippe Creek were similar at the lower salinity head and middle of the estuary, with a decrease at the higher salinity mouth (Table 2). In contrast, seasonal deposition rates in reference Goldsborough Creek increased downstream from the lowest salinity station (GC2) to the highest salinity station (GC3).

Core-averaged sediment characteristics and ²¹⁰Pb-derived burial rates (Table 2) reflect changes over decades within the estuaries but may not include the same time span for all cores due to spatially varying burial rates. At the decadal time scale, sediment characteristics were similar ($p = 0.24$ to 0.33), but sediment burial rates in Goldsborough Creek were approximately twice those in Trippe Creek (1.0 vs. 0.49 g cm⁻² year⁻¹; $p = 0.11$). Core averages for mud, organic, PN, and PP content generally decreased downstream in reference Goldsborough Creek, but accumulation rates increased downstream (Fig. 7). In contrast, in Trippe Creek, average mud, organic, PN, and PP content generally decreased downstream, as did accumulation rates. Overall, mean mud and organic composition of sediments were somewhat higher in Trippe Creek than in reference Goldsborough Creek, whereas the accumulation rates were somewhat lower in Trippe Creek, suggesting a lower sediment supply. While spatial patterns may be explained by sample locations within the creeks (e.g., higher accumulation rates at sites in more embayed regions), the large variability around all averages in Table 2 resulted in p values > 0.10 .

Down-core changes in sediment characteristics reflect temporal variability of sedimentation (Supplemental Material, Fig. S1, Table S3). Geochronologies established from ²¹⁰Pb activities (see “Methods” section) were used to separate observations into two time periods above and below the depth horizon corresponding to 1996, the onset of land-use conversion at Harleigh Farms. Since excess ²¹⁰Pb may be present in sediments up to ~ 100 years old (5 half-lives), and cores were collected in 2016, the ²¹⁰Pb activities represent both 1916–1996 and 1996–2016 (Fig. 7; Table S2). Before 1996, stations in Trippe Creek had higher average PN and PP composition ($p = 0.07$ and $p < 0.001$, respectively), but

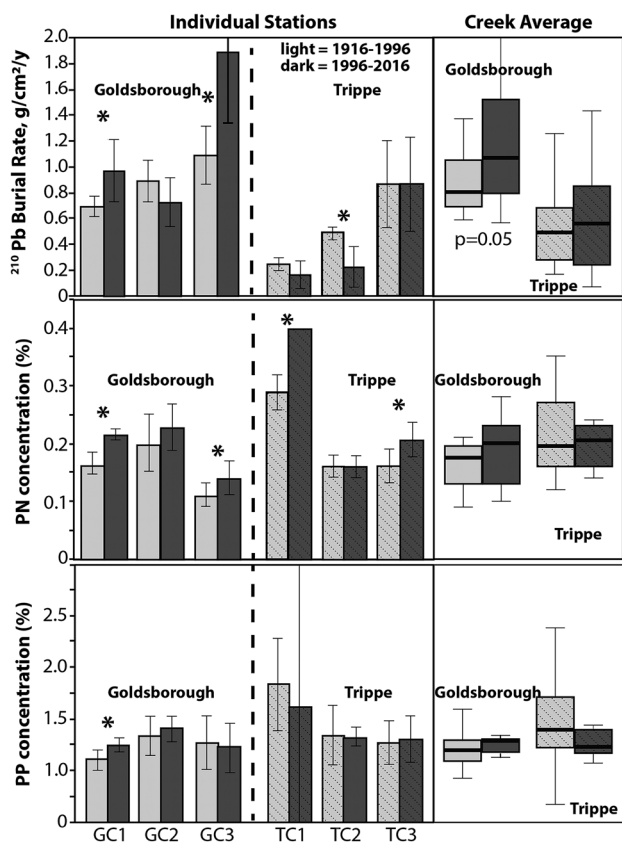


Fig. 7 Average values of (top to bottom) decadal (²¹⁰Pb) sediment burial rates, particulate nitrogen (PN), and particulate phosphorus (PP) in Goldsborough Creek (solid bars) and Trippe Creek (cross-hatched bars). For each parameter, the left panel shows averages for individual stations from 1916 to 1996 (light color) and 1996–2016 (dark color), and the right panel includes all down-core data for each creek, separated into the two time periods, with dark horizontal bars indicating the median. All error bars represent one standard deviation. *P* values for comparisons between the two time periods for each station are listed in Table 3; asterisks (*) indicate *p* values < 0.10

Table 2 Surficial-sediment (upper 1 cm) characteristics (PN=particulate nitrogen; PP=particulate phosphorus) and seasonal (⁷Be) sedimentation rates

Station	Mud content (%)	Organic content (%)	PN content (%)	PP content (%)	⁷ Be rate (g cm ⁻² year ⁻¹)
GC2	95.7	9.9	0.28	0.94	0.42
GC1	95.5	8.9	0.23	0.97	0.92
GC3	71.9	7.8	0.19	0.58	1.42
TC1	89.9	7.0	0.47	0.12	0.70
TC2	89.6	8.9	0.18	0.87	0.69
TC3	97.8	8.9	0.24	0.87	0.53
Goldsborough average	87.7 ± 13.7	8.8 ± 1.0	0.23 ± 0.05	0.83 ± 0.22	0.92 ± 0.50
Trippe average	92.4 ± 4.7	8.3 ± 1.1 (8.9 ± 0.04 w/out TC1)	0.30 ± 0.15 (0.21 ± 0.04 w/out TC1)	0.62 ± 0.43 (0.87 w/out TC1)	0.64 ± 0.09
	<i>p</i> = 0.61	<i>p</i> = 0.53	<i>p</i> = 0.55	<i>p</i> = 0.51	<i>p</i> = 0.44

P values for comparisons between Trippe and Goldsborough Creek are listed below the Trippe average. See Fig. 2 for station locations; stations are organized by position along the salinity gradient

lower average sedimentation rates than stations in reference Goldsborough Creek (*p* < 0.001). In addition, sedimentation rates increased downstream in both creeks. After 1996, this spatial pattern in accumulation rates persisted, and average accumulation rates in Trippe Creek were lower than in reference Goldsborough Creek (*p* = 0.006), but average PN and PP content were similar between the creeks (*p* = 0.17 and *p* = 0.65, respectively). Directions of change for each parameter were determined by calculating the percent change before and after 1996 (Table 3); for our purposes, a change of > 10% was considered notable, as described in “Methods” section. Averaging all down-core data within the creek, the only change > 10% in Trippe Creek was a decrease in PP composition. However, PN increased at the upstream site, PP decreased at the middle site, and accumulation rates decreased at the upstream and middle sites. For reference Goldsborough Creek, both average PN and average accumulation rate increased > 10%, with the latter being the only change in average values for either creek with *p* < 0.10 in *t* tests. At individual sites, all parameters either remained similar or increased after 1996, except for a decrease in accumulation rate at the upstream site.

Discussion

The testable hypotheses for the present study were evaluated through the lens of comparative analysis (Fig. 1). Our hypothesis about land-use change (hypothesis 1) was partially supported (Table 4). We found that land use in Trippe and Goldsborough Creeks was indeed similar prior to conversion of grain production fields to conservation plantings (Fig. 4, left bars in each graph). As conservation plantings increased during 1994 to 2017, there

Table 3 Sediment characteristics and decadal (^{210}Pb) sedimentation rates averaged for each entire core; superscripts indicate within-creek differences with $p < 0.05$ to highlight relatively large differences

Station	Mud content (%)	Organic content (%)	PN content (%)	PP content (%)	^{210}Pb rate ($\text{g cm}^{-2} \text{ year}^{-1}$)
GC2	91.8 ± 8.2 ^a	6.5 ± 1.9 ^{ab}	0.20 ± 0.05 ^b	1.0 ± 0.26 ^b	0.86 ± 0.18 ^a
GC1	89.6 ± 12.9 ^a	6.9 ± 1.3 ^a	0.17 ± 0.05 ^a	0.80 ± 0.13 ^a	0.80 ± 0.20 ^a
GC3	77.0 ± 8.2 ^b	5.5 ± 0.7 ^b	0.12 ± 0.03 ^c	0.90 ± 0.27 ^{ab}	1.4 ± 0.86 ^b
TC1	91.3 ± 4.1 ^a	11.3 ± 2.8 ^a	0.30 ± 0.05 ^a	1.70 ± 0.72 ^a	0.22 ± 0.06 ^a
TC2	91.7 ± 3.6 ^a	7.0 ± 0.95 ^b	0.15 ± 0.03 ^b	0.95 ± 0.30 ^b	0.44 ± 0.15 ^b
TC3	98.0 ± 0.69 ^b	7.4 ± 0.65 ^b	0.17 ± 0.04 ^b	0.98 ± 0.22 ^b	0.81 ± 0.24 ^c
Goldsborough average	86.1 ± 8.0	6.3 ± 0.7	0.16 ± 0.04	0.90 ± 0.11	1.0 ± 0.33
Trippe average	93.6 ± 3.6 $p = 0.24$	7.2 ± 4.1 $p = 0.24$	0.20 ± 0.08 $p = 0.47$	1.2 ± 0.36 $p = 0.33$	0.49 ± 0.29 $p = 0.11$

These station averages were used to calculate average values for Goldsborough and Trippe Creeks; p values for the comparison between creeks are listed below the Trippe average

was a net ~50% decrease in agriculture and corresponding increase in conservation in Trippe Creek, but only a 20% decrease in agriculture in the reference Goldsborough Creek. In contrast, developed lands remained similar in both watersheds over time. Most of the conversion from agriculture to conservation land in both watersheds occurred on the Harleigh Farms properties, accounting for ~85–90% of observed changes in both watersheds. These changes were concentrated in the headwaters of each watershed, which drained directly to the RR water quality station in Trippe Creek and the GC water quality station in reference Goldsborough Creek. While this potentially confounds the comparative analysis because a small portion of Harleigh Farms lies in the reference Goldsborough

Creek watershed, the reduction of agriculture was much larger in the Trippe Creek watershed (–50%) than in the Goldsborough watershed (–20%)(Table 5).

We also evaluated changes in estuarine N, P, and *chl a* concentrations (hypothesis 2) between two time points: 2010 and 2016. The 6-year time span between these time points is longer than observed decreases in groundwater nitrate in the Trippe Creek surficial aquifer (90% reduction within 3–5 years, Bunnell-Young et al. 2017), and hypothesis 2 predicts decreases in estuarine N, P, and *chl a* concentrations. All TN, TP, and *chl a* annual means in 2016 were in fact consistently lower than 2010 means. TN, TP, and *chl a* means were similarly distributed along the salinity gradient in both years, but with consistently lower concentrations in 2016

Table 4 Percent change from before and after 1996 (see Table S2; Fig. 6) at each station, with negative values indicating a decrease; p values are given for the corresponding t test

Station	Mud content (%)	Organic content (%)	PN content (%)	PP content (%)	^{210}Pb rate (%)
GC2	–0.36 $p = 0.43$	+24.7 $p = 0.16$	+9.4 $p = 0.26$	+12.9 $p = 0.23$	–17.9 $p = 0.15$
GC1	+1.5 $p = 0.11$	+13.6 $p = 0.009$	+30.8 $p < 0.001$	+17.4 $p = 0.03$	+41.7 $p = 0.05$
GC3	–8.9 $p = 0.06$	+0.52 $p = 0.95$	+23.1 $p = 0.04$	–6.0 $p = 0.71$	+79.2 $p = 0.07$
TC1	–1.6 $p = 0.46$	–10.2 $p = 0.78$	+41.2 $p < 0.001$	–25.4 $p = 0.77$	–33.0 $p = 0.47$
TC2	+0.73 $p = 0.65$	–0.59 $p = 0.95$	+2.7 $p = 0.67$	–1.9 $p = 0.87$	–53.8 $p = 0.04$
TC3	–0.07 $p = 0.89$	+0.47 $p = 0.87$	+1.0 $p = 0.87$	+1.2 $p = 0.93$	+1.3 $p = 0.95$
Goldsborough average	–6.1 $p = 0.13$	+7.4 $p = 0.20$	+11.8 $p = 0.23$	+5.9 $p = 0.45$	+50.7 $p = 0.05$
Trippe average	+1.5 $p = 0.20$	–7.0 $p = 0.33$	+6.1 $p = 0.60$	–15.6 $p = 0.30$	+10.1 $p = 0.68$

Values in the “average” row for each creek are calculated using all down-core data

Table 5 Results supporting or not supporting the 3 testable hypotheses

Hypothesis	Observation	Source	Supports?
1	Land use was similar in both watersheds in 1994	Figure 4	Yes
	Harleigh Farms represented 63% of ag in Trippe but only 20% of ag in Goldsborough watershed	Text	Partial
	50% reduction in grain ag in Trippe by 2017, but only 20% decrease in Golds, mostly due to Harleigh Farms	Figure 4	Partial
	Little change in developed land in Trippe and Golds	Figure 4	No
	Precipitation and discharge were similar in 2010 and 2016	Figure 5	Yes
2	Estuarine station means of TN, TP, and chl a lower in 2016/2017 compared to 2010	Figure 6	Yes
	Major algal blooms in 2010, none in 2016/2017	Figure 6	Yes
	Similar mixing behavior in both estuaries	Figure 6	Yes
	In both time periods, sed rates in Trippe > Golds	Figure 7	No
3	Sediment organic C, N, and P conc in Trippe estuary > Golds estuary	Figure 7	No
	N sed rate increased in both estuaries between 2010 and 2016	Figure 7	No
	2016 P sed rates in Trippe estuary were = or < 2010 rates	Figure 7	Yes
	2016 P sed rates in Golds estuary were > or = 2010 rates	Figure 7	Yes

(Fig. 6). While TN was essentially conservative, TP was elevated in the middle of the salinity gradient in both creeks, implying a local P source. Average chl a had the highest values at the Railroad station, with high variability from algal blooms in summer 2010. The systematically lower TN, TP, and chl a in 2016 compared to 2010, and the algal blooms in 2010 support hypothesis 2 (Table 4).

We have combined the TN, TP, and chl a data from all stations in both creeks by year in Fig. 6. We argue that the parallel distributions of TN, TP, and chl a along the salinity gradient imply similar mixing between freshwater inputs from both watersheds and higher salinity waters of the Tred Avon River (Fig. 2). RR and GC stations had the highest TN and chl a concentrations in each creek because they are located near the headwaters of their respective creeks, where land-use effects are most influential (Leight et al. 2014, 2015). We also note that TP was higher at intermediate salinities of 10–12 (stations Goldsborough GC and Trippe TA5) due to apparent P inputs from local land uses or desorption from local sediments as salinity increased.

The systematic decreases in TN, TP, and chl a between 2010 and 2016 suggest improving water quality in both creeks as a result of reductions in agricultural land use. These results are similar to expectations from model results in other studies (White et al. 2014; García et al. 2016; Muenich et al. 2016; Clune and Capel 2021). It is tempting to attribute decreases in TN, TP, and chl a to the effects of conservation plantings. However, there are other possible explanations, including seasonal differences in data collection periods. The 2010 dataset was collected during January–August of 2010 (winter to summer conditions), while the 2016–2017 dataset was collected from May 2016 to March 2017 (spring to winter conditions). Seasonal data in the Chesapeake typically

show blooms of phytoplankton in late winter to spring, associated with increasing light, nutrient availability, and freshwater flow, along with warming temperatures and increasing vertical stratification (Harding 1994; Harding et al. 2019). These seasonal increases in chl a would have been captured in the 2010 data but not necessarily in the 2016–2017 data. Also, while daily average rainfall and water yields were similar between the two time periods, the 2010 record is highly influenced by multiple tropical depressions and a single large event occurring just before the July sampling. Large runoff events can fuel algal blooms by providing freshwater and nutrients and promoting stratification, as observed in long-term datasets in the Chesapeake region (e.g., Fisher et al. 1988; Prasad et al. 2010). Indeed, chl a measurements at the Railroad station were anomalously high in both July and August 2010, with the August 2010 value being 5 times higher than any other chl a observation. However, chl a in reference Goldsborough Creek had no obvious response to this event, and there was no obvious response in TN or TP in either creek. Thus, while our observations are consistent with decreasing supplies of terrestrial nutrients due to land-use change, other environmental factors may have also influenced our results.

Sediment data support a more robust change analysis since they provide a continuous record of both pre- and post-conservation conditions (hypothesis 3). In both time periods, relative to reference Goldsborough Creek, sedimentation rates were lower in Trippe Creek, but organic content, PN, and PP were higher. However, the trajectory of change for some parameters between pre- and post-conservation time periods differed between the two creeks. In Trippe Creek, percent sediment PN increased and percent sediment PP decreased at the upstream site, and accumulation rates decreased at the middle and

upstream sites, with other parameters remaining fairly steady before and after conservation. In reference Goldsborough Creek, all parameters either remained similar or increased after conservation, except for a decrease in accumulation rate at the upstream site. Increases in PN observed in both creeks could reflect particle scavenging of dissolved N in the water column during settling and/or settling of primary and secondary producers from the water column (Boynton et al. 1995). Trends in PP and accumulation rates diverged, decreasing or remaining steady in Trippe Creek but increasing or remaining steady in Goldsborough Creek, except at the upstream site. On average, accumulation rates were similar in Trippe Creek but increased by 51% in Goldsborough Creek, providing evidence to support hypothesis 3 (Table 4).

Expected reductions in N and P loads from conversion can be estimated by comparing average export from agricultural and forested lands under baseline conditions in the Choptank River (Fisher et al. 2021). Conversion from grain production to forest would result in an estimated 94% reduction for N and 69% for P. Note that these estimates do not include the time for groundwater response to conservation (3–5 years; Bunnell-Young et al. 2017) and another 5–10 years for groundwater to reach a stream (Sutton et al. 2009). These estimated reductions are similar to those calculated from average loading rates used in the Chesapeake Bay Program (CBP) model for crop production and forests (Chesapeake Bay Program, CBP 2020). The CBP model also includes “agriculture open space” as a land use, which may better represent the conservation land in our study. Transitioning from crop production to agriculture open space land use in the CBP model results in slightly lower reductions, 85% for N and 57% for P, and a 96% reduction in sediment loads. Using these values as long-term, steady-state estimates, and given that only about half of the agricultural land was converted to conservation, expected reductions would be about half of these values (i.e., 42% for nitrogen, 28% for PP, and 48% for sediment). We did not observe a major difference in %PN between the two creeks, and the difference in %PP (decrease of 16% in Trippe Creek versus no change in Goldsborough Creek) was more modest than expected. The difference in sedimentation rates (no change in Trippe Creek versus an increase of 51% in Goldsborough Creek) was similar to expectations. This difference could reflect both differences in land-use change in the two watersheds and/or changes in the nature of agriculture. The intensity of agriculture has increased over the last ~50 years, with double cropping (winter wheat and soybeans harvested in the same years) and increased fertilizer use almost doubling corn yields (Fisher et al. 2006b) and potentially increasing sediment supply. However, there also has been increasing use of no-till agriculture and cover crops that would reduce sediment supply (Staver and Brinsfield 2001), and the net effect of these changes is not clear.

Land-use change is not the only factor influencing sediment and nutrient supply from terrestrial to aquatic environments. Climate variability and change are widely acknowledged (Kaushal et al. 2010), and rainfall is the main driver of watershed sediment erosion (Romkens et al. 2002), varying greatly over time scales from events to millenia (Cronin et al. 2000; Miller et al. 2006; Saenger et al. 2008; Wagena et al. 2018). The potential impact of this variability on sediment observations is more limited than for water-column observations for two reasons. First, the watersheds are small and located immediately adjacent to each other. While we do not have direct, local measurements of rainfall within each watershed, we can assume that precipitation reaching each watershed was similar and thus does not drive differences between the two creeks. Second, the long-term increases in average annual rainfall are expected to have increased annual discharge (Fig. 5) and sediment deposition, even though precipitation was similar in 2010 and 2016, and both time periods included high-rainfall events of similar magnitude. Thus, precipitation probably did not drive differences between pre- and post-conservation eras within each creek. Mean sea level has been measured in the Choptank River (at Cambridge; station 8,571,892; <http://tidesandcurrents.noaa.gov>) by NOAA since 1971. The rate of relative sea-level rise increased from 2.1 mm year⁻¹ between 1971 and 1996 to 4.2 mm year⁻¹ between 1996 and 2016. This increase in sea-level rise makes more bottom area and volume available for sediment deposition, facilitating higher sedimentation rates, but it would do so in both creeks. Thus, sea-level rise does not explain differences in trends between the two creeks.

The input of inorganic sediment is likely dominated by soil and shoreline erosion, with an unmeasured component of biogenic production of silica from diatoms in the estuary. Shoreline erosion dominates the sediment supply in some settings, including mid-Chesapeake Bay (Hobbs et al. 1992; Schilling et al. 2011; Sherriff et al. 2018), although rigorous quantification of shoreline sediment sources is hindered by availability of shoreline surveys and/or aerial imagery. Biogenic production of silica and organic matter is typically significant only where inorganic sediment sources are minimal (e.g., southern portion of Chesapeake Bay; Hobbs et al. 1992).

In this study, the annual average loss of shoreline area from 1942 to 1994 and 2003–2015 is assumed to represent conditions during pre- and post-conservation eras, respectively. Over both time periods, shoreline loss in Trippe Creek was much greater than in reference Goldsborough Creek. From 1942 to 1994, the shoreline-loss rate was ~2.5× higher in Trippe Creek ($3.5 \times 10^3 \text{ m}^2 \text{ year}^{-1}$) compared to Goldsborough Creek ($1.3 \times 10^3 \text{ m}^2 \text{ year}^{-1}$), approximately scaled with the shoreline length in each estuary (Trippe shoreline is approximately twice the Goldsborough shoreline length). However, during the later period 2003–2015, shoreline loss

in Trippe Creek ($2.6 \times 10^3 \text{ m}^2 \text{ year}^{-1}$) was nearly 6 times higher than in reference Goldsborough Creek ($0.45 \times 10^3 \text{ m}^2 \text{ year}^{-1}$), and shoreline-loss rates decreased by $\sim 25\%$ and $\sim 65\%$ in Trippe and reference Goldsborough Creeks, respectively. This translates into slightly lower sediment loads in Trippe Creek but much lower loads in Goldsborough now than in the past—the opposite trend observed for sedimentation rates (higher sedimentation rates imply higher sediment supply). For comparison, shoreline-loss rates for Talbot County where these creeks reside decreased by $\sim 70\%$ after 1990 (Russ and Palinkas 2020) likely due to shoreline stabilization. Differences in the degree and timing of stabilization could explain observed differences between Trippe and Goldsborough Creeks. The only available shoreline inventory for these creeks was completed in 2004 and shows that much of the Trippe Creek shoreline was already hardened (mostly rip rap and bulkheads), while the reference Goldsborough shoreline was not (MD Geological Survey; <https://data.imap.maryland.gov>). The diverging trends in sediment burial rates and shoreline-erosion rates in Goldsborough imply an increase in terrestrial sediment supply, consistent with hypothesis 3. While we did not measure terrestrial sediment supply, trends in sedimentation rates (no change in Trippe Creek versus an increase of 51% in Goldsborough Creek) was similar to model-derived expectations from land-use change discussed above.

The preceding discussion assumes that sediment delivered to each creek remains there, when in fact there could be some tidal exchange between the two creeks and/or with the Tred Avon River at their confluence, similar to exchange between the Choptank River and Chesapeake Bay (Sanford and Boicourt 1990). Other factors such as the presence of submersed aquatic vegetation (SAV; e.g., de Boer 2007) and nutrient uptake by phytoplankton (e.g., Prasad et al. 2010) can influence sediment and nutrient retention within each creek. Nonetheless, the most likely explanation for diverging trends in Trippe and Goldsborough Creeks is land-use change via conservation in Trippe Creek watershed.

Improvement of water-column and sediment conditions is critical for a healthy Bay ecosystem (Phillips and McGee 2016; Tango and Batiuk 2016; Zhang et al. 2018). While Johnson et al. (2016) found that the ecosystem benefits provided by CRP conservation lands exceed the cost of payments to farmers, this balance may change under future climate scenarios (Bosch et al. 2018). It is also interesting to speculate whether increased shoreline erosion and encroachment of wetlands into farm fields with continued environmental change (Stevenson et al. 1985; Garbisch and Garbisch 1994; Tully et al. 2019) will motivate shoreline stabilization and/or land conversion that reduces future terrestrial sediment and nutrient supply to adjacent waters. We recognize that the high rates of conversion from cropland to conservation land in these watersheds are atypical, especially

in this region where agriculture is the dominant land use. However, high percentages of conserved land in watersheds may become more typical as interest in solar farms grows throughout the region. Unlike the CRP and CREP programs, land-rental payments from solar companies are likely to be relatively high, providing economic incentives along with equivalent or perhaps even greater ecological benefits of retiring agricultural land for 15–20 years, a typical solar farm contract term. Regardless, trade-offs between food production and conservation should be carefully considered to support sustainable communities (Martinelli and Filoso 2009; Blanchard et al. 2017; McLellan et al. 2018).

Summary

This study evaluated the impacts of land-use change to downstream estuarine waters via comparative analysis of two adjacent, small watersheds with similar initial historical land uses. In 1996, conversion of land from grain agriculture to conservation plantings began in the Trippe Creek watershed, and about half of the agricultural land was converted by 2017. In the reference watershed (Goldsborough Creek), $\sim 20\%$ of the agricultural land was converted to conservation. Study results provide evidence for water quality improvement in the headwaters of Trippe Creek, where high and variable concentrations of chl a decreased to levels similar to those in Goldsborough Creek. Sedimentation rates remained fairly steady in Trippe Creek but increased by $\sim 50\%$ in our reference Goldsborough Creek. However, sediment particulate nitrogen (PN) concentrations increased in both creeks, suggesting particle scavenging during settling through the water column. While other factors also may influence these observations, land-use change is the most likely explanation for driving the observed trends. Our results illustrate the problems of detecting improvements in estuarine water quality in locations with little systematic monitoring even when there is documentation of significant land-use change that could result in improved water quality. We recommend that more efforts to identify improvements in estuarine water and sediment quality be undertaken when significant land-use changes are documented. There are many small datasets collected by citizen groups, students, and other scientists in Chesapeake Bay far from major monitoring stations that can potentially be used to document local responses. We also suggest that USDA and other agencies should use more targeted watershed approaches in the placement of conservation efforts to simplify the evaluation of their effects downstream. Improving sediment conditions and water quality are critical for a healthy Bay ecosystem, but documentation of estuarine improvements can be difficult to quantify.

Supplementary Information The online version contains supplementary material available at <https://doi.org/10.1007/s12237-022-01077-7>.

Acknowledgements The authors thank the many students and technicians who assisted in the field and laboratory, especially Anne Gustafson and Debbie Hinkle. AK Leight provided data for the two NOAA water-column stations, which greatly improved the manuscript. We also thank Jim Lewis for his insights into farming practices. Sediment data formed part of a Research Experience for Undergraduates project by Carl Polard supported by the National Science Foundation and administered by Maryland Sea Grant. The NSF Coastal SEES Program (grant #13-25553, Ecosystem Studies Program) provided partial support for TRF and ELK. We gratefully acknowledge Chip Akridge for funding the water-column and sediment data, as well as staff at Harleigh Farms for providing field support and property management records, especially Jimmy Greenhawk and Clay Robinson. Constructive comments from two anonymous reviewers greatly improved the original manuscript. This is UMCES contribution #5957.

References

- Anderson, J.R., E.E. Hardy, J.T. Roach, and R.E. Witmer. 1976. *A land use and land cover classification system for use with remote sensor data*. Professional Paper 964. Professional Paper. USGS.
- Appleby, P.G., and F. Oldfield. 1978. The calculation of lead-210 dates assuming a constant rate of supply of unsupported ^{210}Pb to the sediment. *CATENA* 5: 1–8. [https://doi.org/10.1016/S0341-8162\(78\)80002-2](https://doi.org/10.1016/S0341-8162(78)80002-2).
- Arias-Ortiz, A., P. Masqué, J. Garcia-Orellana, O. Serrano, I. Mazarrasa, N. Marbà, C.E. Lovelock, P.S. Lavery, and C.M. Duarte. 2018. Reviews and syntheses: ^{210}Pb -derived sediment and carbon accumulation rates in vegetated coastal ecosystems – setting the record straight. *Biogeosciences* 15: 6791–6818. <https://doi.org/10.5194/bg-15-6791-2018>.
- Aspila, K.I., H. Agemian, and A.S.Y. Chau. 1976. A semi-automated method for the determination of inorganic, organic and total phosphate in sediments. *The Analyst* 101: 187. <https://doi.org/10.1039/an9760100187>.
- Baker, M.E., D.E. Weller, and T.E. Jordan. 2006. Comparison of automated watershed delineations. *Photogrammetric Engineering & Remote Sensing* 72: 159–168. <https://doi.org/10.14358/PERS.72.2.159>.
- Beaulac, M.N., and K.H. Reckhow. 1982. An examination of land use - nutrient export relationships. *Journal of the American Water Resources Association* 18: 1013–1024. <https://doi.org/10.1111/j.1752-1688.1982.tb00109.x>.
- Blanchard, J.L., R.A. Watson, E.A. Fulton, R.S. Cottrell, K.L. Nash, A. Bryndum-Buchholz, M. Büchner, et al. 2017. Linked sustainability challenges and trade-offs among fisheries, aquaculture and agriculture. *Nature Ecology & Evolution* 1: 1240–1249. doi: <https://doi.org/10.1038/s41559-017-0258-8>.
- Boesch, D.F., R.B. Brinsfield, and R.E. Magnien. 2001. Chesapeake Bay eutrophication: Scientific understanding, ecosystem restoration, and challenges for agriculture. *Journal of Environmental Quality* 30: 303–320. <https://doi.org/10.2134/jeq2001.302303x>.
- Bosch, D.J., M.B. Wagena, A.C. Ross, A.S. Collick, and Z.M. Easton. 2018. Meeting water quality goals under climate change in Chesapeake Bay watershed, USA. *Journal of the American Water Resources Association* 54: 1239–1257. <https://doi.org/10.1111/1752-1688.12684>.
- Boynton, W.R., J.H. Garber, R. Summers, and W.M. Kemp. 1995. Inputs, transformations, and transport of nitrogen and phosphorus in Chesapeake Bay and selected tributaries. *Estuaries* 18: 285–314.
- Brush, G.S. 1989. Rates and patterns of estuarine sediment accumulation. *Limnology and Oceanography* 34: 1235–1246.
- Brush, G.S. 2009. Historical land use, nitrogen, and coastal eutrophication: A paleoecological perspective. *Estuaries and Coasts* 32: 18–28. <https://doi.org/10.1007/s12237-008-9106-z>.
- Bunnell-Young, D., T. Rosen, T.R. Fisher, T. Moorshead, and D. Koslow. 2017. Dynamics of nitrate and methane in shallow groundwater following land use conversion from agricultural grain production to conservation easement. *Agriculture, Ecosystems & Environment* 248: 200–214. <https://doi.org/10.1016/j.agee.2017.07.026>.
- Callow, J.N., K.P. Van Niel, and G.S. Boggs. 2007. How does modifying a DEM to reflect known hydrology affect subsequent terrain analysis? *Journal of Hydrology* 332: 30–39. <https://doi.org/10.1016/j.jhydrol.2006.06.020>.
- CBP, Chesapeake Bay Program. 2020. Chesapeake Assessment and Scenario Tool (CAST) Version 2019. Chesapeake Bay Program Office, Last accessed March 2021. <https://cast.chesapeakebay.net/documentation/modeldocumentation>
- Clark, G.M., D.K. Mueller, and M.A. Mast. 2000. Nutrient concentrations and yields in undeveloped stream basins of the United States. *Journal of the American Water Resources Association* 36: 849–860.
- Clesceri, N.L., S.J. Curran, and R.I. Sedlak. 1986. Nutrient loads to Wisconsin lakes: Part 1. Nitrogen and phosphorus export coefficients. *Journal of the American Water Resources Association* 22: 983–990. <https://doi.org/10.1111/j.1752-1688.1986.tb00769.x>.
- Clune, J.W., and P.D. Capel. (eds). 2021. Nitrogen in the Chesapeake Bay watershed - a century of change, 1950–2050. *US Geol Surv Circular* 1486-168. <https://doi.org/10.3133/cir1486>
- Colman, S.M., P.C. Baucom, J.F. Bratton, T.M. Cronin, J.P. McGeehin, D. Willard, A.R. Zimmerman, and P.R. Vogt. 2002. Radiocarbon dating, chronologic framework, and changes in accumulation rates of Holocene estuarine sediments from Chesapeake Bay. *Quaternary Research* 57: 58–70. <https://doi.org/10.1006/qres.2001.2285>.
- Cornwell, J.C., D.J. Conley, M. Owens, and J.C. Stevenson. 1996. A sediment chronology of the eutrophication of Chesapeake Bay. *Estuaries* 19: 488–499. <https://doi.org/10.2307/1352465>.
- Cronin, T., D. Willard, A. Karlsen, S. Ishman, S. Verardo, J. McGeehin, R. Kerhin, C. Holmes, S. Colman, and A.N.D.A. Zimmerman. 2000. Climatic variability in the eastern United States over the past millennium from Chesapeake Bay sediments. *Geology* 28: 3–6.
- de Boer, W.F. 2007. Seagrass–sediment interactions, positive feedbacks and critical thresholds for occurrence: a review. *Hydrobiologia* 591: 5–24. <https://doi.org/10.1007/s10750-007-0780-9>.
- Dunn, C.P., F. Stearns, G.R. Guntenspergen, and D.M. Sharpe. 1993. Ecological benefits of the Conservation Reserve Program. *Conservation Biology* 7: 132–139.
- EPA. 1983. *Chesapeake Bay: A Profile of Environmental Change*.
- Eulie, D.O., J.P. Walsh, D.R. Corbett, and R.P. Mulligan. 2017. Temporal and spatial dynamics of estuarine shoreline change in the Albemarle-Pamlico estuarine system, North Carolina, USA. *Estuaries and Coasts* 40: 741–757. <https://doi.org/10.1007/s12237-016-0143-8>.
- Fanelli, R.M., J.D. Blomquist, and R.M. Hirsch. 2019. Point sources and agricultural practices control spatial-temporal patterns of orthophosphate in tributaries to Chesapeake Bay. *Science of the Total Environment* 652: 422–433. <https://doi.org/10.1016/j.scitotenv.2018.10.062>.
- Fisher, T.R., A.B. Gustafson, K. Sellner, R. Lacuture, L.W. Haas, R. Magnien, R. Karrh, and B. Michael. 1999. Spatial and temporal

- variation in resource limitation in Chesapeake Bay. *Marine Biology* 133: 763–778.
- Fisher, T.R., E.R. Peele, J.A. Ammerman, and L.W. Harding. 1992. Nutrient limitation of phytoplankton in Chesapeake Bay. *Marine Ecology Progress Series* 82: 51–63.
- Fisher, T.R., J.A. Benitez, K.-Y. Lee, and A.J. Sutton. 2006b. History of land cover change and biogeochemical impacts in the Choptank River basin in the mid-Atlantic region of the US. *International Journal of Remote Sensing* 27: 3683–3703. <https://doi.org/10.1080/01431160500500383>.
- Fisher, T.R., J.D. Hagy, W.R. Boynton, and M.R. Williams. 2006a. Cultural eutrophication in the Choptank and Patuxent estuaries of Chesapeake Bay. *Limnology and Oceanography* 51: 435–447. https://doi.org/10.4319/lo.2006.51.1_part_2.0435.
- Fisher, T.R., L.W. Harding, D.W. Stanley, and L.G. Ward. 1988. Phytoplankton, nutrients, and turbidity in the Chesapeake, Delaware, and Hudson River estuaries. *Estuarine, Coastal and Shelf Science* 27: 61–93.
- Fisher, T.R., R.J. Fox, A.B. Gustafson, E. Koontz, M. Lepori-Bui, and J. Lewis. 2021. Improving water quality in the Choptank estuary, a tributary of Chesapeake Bay. *Estuaries and Coasts*. <https://doi.org/10.1007/s12237-020-00872-4>.
- Fisher, T.R., T.E. Jordan, K.W. Staver, A.B. Gustafson, A.I. Koskelo, R.J. Fox, A.J. Sutton, et al. 2010. The Choptank Basin in transition: intensifying agriculture, slow urbanization, and estuarine eutrophication. In *Coastal lagoons: critical habitats of environmental change*, ed. M. J. Kennish and H. W. Paerl, 135–165. New York, NY: Taylor and Francis Group.
- FSA, Farm Service Agency. 2020. Conservation Research Program Statistics.
- Garbisch, E.W., and J.L. Garbisch. 1994. Control of upland bank erosion through tidal marsh construction on restored shores: Application in the Maryland portion of Chesapeake Bay. *Environmental Management* 18: 677–691. <https://doi.org/10.1007/BF02394633>.
- García, A.M., R.B. Alexander, J.G. Arnold, L. Norfleet, M.J. White, D.M. Robertson, and G. Schwarz. 2016. Regional effects of agricultural conservation practices on nutrient transport in the upper Mississippi River basin. *Environmental Science & Technology* 50: 6991–7000. <https://doi.org/10.1021/acs.est.5b03543>.
- Gellis, A.C., C.R. Hupp, M.J. Pavich, J.M. Landwehr, W.S.L. Banks, B.E. Hubbard, M.J. Langland, J.C. Ritchie, and J.M. Reuter. 2009. *Sources, transport, and storage of sediment at selected sites in the Chesapeake Bay watershed*. 2008–5186. Scientific Investigations. US Geological Survey.
- Geyer, W.R., J.T. Morris, F.G. Prahl, and D.A. Jay. 2000. Interaction between physical processes and ecosystem structure: A comparative approach. In *Estuarine science: A synthetic approach to research and practice*, ed. J.E. Hobbie, 177–206. Washington, D.C.: Island Press.
- Goodbred, S.L., and S.A. Kuehl. 1998. Floodplain processes in the Bengal Basin and the storage of Ganges-Brahmaputra river sediment: An accretion study using ^{137}Cs and ^{210}Pb geochronology. *Sedimentary Geology* 121: 239–258. [https://doi.org/10.1016/S0037-0738\(98\)00082-7](https://doi.org/10.1016/S0037-0738(98)00082-7).
- Hapke, C.J., E.A. Himmelstoss, M.G. Kratzmann, J.H. List, and E.R. Thieler. 2011. *National assessment of shoreline change: historical shoreline change along the New England and Mid-Atlantic coasts*. 2010–1118. Open-File Report. US Geological Survey.
- Harding, L.W. 1994. Long-term trends in the distribution of phytoplankton in Chesapeake Bay: Roles of light, nutrients and streamflow. *Marine Ecology Progress Series* 104: 267–291. <https://doi.org/10.3354/meps104267>.
- Harding, L.W., Jr., R.A. Batiuk, T.R. Fisher, C.L. Gallegos, T.C. Malone, W.D. Miller, M.R. Mulholland, H.W. Paerl, and P. Tango. 2014. Scientific bases for numerical chlorophyll criteria in Chesapeake Bay. *Estuaries and Coasts* 37: 134–148.
- Harding, L.W., M.E. Mallonee, E.S. Perry, W.D. Miller, J.E. Adolf, C.L. Gallegos, and H.W. Paerl. 2019. Long-term trends, current status, and transitions of water quality in Chesapeake Bay. *Scientific Reports* 9: 6709. <https://doi.org/10.1038/s41598-019-43036-6>.
- Hobbs, C.H., J.P. Halka, R.T. Kerhin, and M.J. Carron. 1992. Chesapeake Bay sediment budget. *Journal of Coastal Research* 8: 292–300.
- Hopkinson, C.S., J.T. Morris, S. Fagherazzi, W.M. Wollheim, and P.A. Raymond. 2018. Lateral marsh edge erosion as a source of sediments for vertical marsh accretion. *Journal of Geophysical Research: Biogeosciences* 123: 2444–2465. <https://doi.org/10.1029/2017JG004358>.
- Huang, L., F.H. Liao, K.A. Lohse, D.M. Larson, M. Fragkias, D.L. Lybecker, and C.V. Baxter. 2019. Land conservation can mitigate freshwater ecosystem services degradation due to climate change in a semiarid catchment: The case of the Portneuf River catchment, Idaho, USA. *Science of the Total Environment* 651: 1796–1809.
- Johnson, C.L., Q. Chen, and C.E. Ozdemir. 2020. Lidar time-series analysis of a rapidly transgressing low-lying mainland barrier (Caminada Headlands, Louisiana, USA). *Geomorphology* 352: 106979. <https://doi.org/10.1016/j.geomorph.2019.106979>.
- Johnson, K.A., B.J. Dalzell, M. Donahue, J. Gourevitch, D.L. Johnson, G.S. Karlovits, B. Keeler, and J.T. Smith. 2016. Conservation Reserve Program (CRP) lands provide ecosystem service benefits that exceed land rental payment costs. *Ecosystem Services* 18: 175–185. <https://doi.org/10.1016/j.ecoser.2016.03.004>.
- Kaushal, S.S., M.L. Pace, P.M. Groffman, L.E. Band, K.T. Belt, P.M. Meyer, and C. Welty. 2010. Land use and climate variability amplify contaminant pulses. *Eos, Transactions American Geophysical Union* 91: 221–222. <https://doi.org/10.1029/2010EO250001>.
- Kemp, W.M., W.R. Boynton, J.E. Adolf, D.F. Boesch, W.C. Boicourt, G. Brush, J.C. Cornwell, et al. 2005. Eutrophication of Chesapeake Bay: Historical trends and ecological interactions. *Marine Ecology Progress Series* 303: 1–29.
- Koskelo, A.I., T.R. Fisher, A.J. Sutton, and A.B. Gustafson. 2018. Biogeochemical storm response in agricultural watersheds of the Choptank River Basin, Delmarva Peninsula, USA. *Biogeochemistry* 139: 215–239. <https://doi.org/10.1007/s10533-018-0464-8>.
- Krumbein, W.C. 1960. Some problems in applying statistics to geology. *Applied Statistics* 9: 82. <https://doi.org/10.2307/2985430>.
- Lane, L., L. Van Heukelem, and M. Maddox. 2000. A manual of standard operating procedures for analyses of dissolved inorganic nutrients, organic compounds, and particulates at the HPL-UMCES Analytical Services Laboratory, Cambridge MD. Internal Publication.
- Lee, K.Y., T.R. Fisher, E. Rochelle-Newall. 2001. Modeling the hydrochemistry of the Choptank River basin using GWLF and Arc/Info: 2 Model Validation and Application. *Biogeochemistry* 311–348. <https://doi.org/10.1023/A:1013169027082>.
- Lee, K.Y., T.R. Fisher, T.E. Jordan, D.L. Correll, D.E. Weller. 2000. Modeling the hydrochemistry of the Choptank River Basin using GWLF and Arc/Info: 1 Model Calibration and Validation. *Biogeochemistry* 143–173. <https://doi.org/10.1023/A:1006375530844>.
- Leight, A., J. Jacobs, L. Gonsalves, G. Messick, S. McLaughlin, J. Lewis, J. Brush, et al. 2014. *Coastal ecosystem assessment of Chesapeake Bay watersheds: a story of three rivers – the Corisca, Magothy, and Rhode*. Technical Memorandum 189. NOS NCCOS. NOAA.
- Leight, A., R. Trippe III, L. Gonsalves, J. Jacobs, S. McLaughlin, and G. Messick. 2015. *Coastal ecosystem assessment of Chesapeake*

- Bay watersheds: land use patterns and river conditions*. Technical Memorandum 207. NOS NCCOS. NOAA.
- Li, Z. 2014. Watershed modeling using arc hydro based on DEMs: A case study in Jackpine watershed. *Environmental Systems Research* 3: 11. <https://doi.org/10.1186/2193-2697-3-11>.
- Martinelli, L. Antonio., and S. Filoso. 2009. Balance between food production, biodiversity and ecosystem services in Brazil: A challenge and an opportunity. *Biota Neotropica* 9: 21–25. <https://doi.org/10.1590/S1676-06032009000400001>.
- McLaughlin, S.M., A.K. Leight, J.E. Spires, S.B. Bricker, J.M. Jacobs, G.A. Messick, S. Skelley. 2018. Coastal Ecological Assessment to Support NOAA's Choptank River Complex Habitat Focus Area: Tred Avon River. NOAA Technical Memorandum NOS NCCOS 251. Oxford, MD. 118 pp. <https://doi.org/10.25923/k2xr-5n27>
- McLellan, E.L., K.G. Cassman, A.J. Eagle, P.B. Woodbury, S. Sela, C. Tonitto, R.D. Marjerison, and H.M. Van Es. 2018. The nitrogen balancing act: Tracking the environmental performance of food production. *BioScience*. <https://doi.org/10.1093/biosci/bix164>.
- Miller, M.P., P.D. Capel, A.M. García, and S.W. Ator. 2019. Response of nitrogen loading to the Chesapeake Bay to source reduction and land use change scenarios: A SPARROW-informed analysis. *Journal of the American Water Resources Association* 1752–1688: 12807. <https://doi.org/10.1111/1752-1688.12807>.
- Miller, W.D., D.G. Kimmel, and L.W. Harding. 2006. Predicting spring discharge of the Susquehanna River from a winter synoptic climatology for the eastern United States. *Water Resources Research*. <https://doi.org/10.1029/2005WR004270>.
- Muenich, R., M. Kalcic, Logsdon, and D. Scavia. 2016. Evaluating the impact of legacy P and agricultural conservation practices on nutrient loads from the Maumee River watershed. *Environmental Science & Technology* 50: 8146–8154. <https://doi.org/10.1021/acs.est.6b01421>.
- Nittrouer, C.A., R.W. Sternberg, R. Carpenter, and J.T. Bennett. 1979. The use of Pb-210 geochronology as a sedimentological tool: Application to the Washington continental shelf. *Marine Geology* 31: 297–316. [https://doi.org/10.1016/0025-3227\(79\)90039-2](https://doi.org/10.1016/0025-3227(79)90039-2).
- Novotny, V., and H. Olem. 1995. *Water quality: prevention, identification, and management of diffuse pollution*. 3. [print.]. New York: Van Nostrand Reinhold.
- Olsen, C.R., I.L. Larsen, P.D. Lowry, N.H. Cutshall, and M.M. Nichols. 1986. Geochemistry and deposition of ⁷Be in river-estuarine and coastal waters. *Journal of Geophysical Research* 91: 896–908. <https://doi.org/10.1029/JC091iC01p00896>.
- Orth, R.J., and K.A. Moore. 1984. Distribution and abundance of submerged aquatic vegetation in Chesapeake Bay: An historical perspective. *Estuaries* 7: 531–540.
- Palinkas, C.M. 2013. Seasonal and interannual patterns of sedimentation in the Corsica River (MD): Evaluating the potential influence of watershed restoration. *Estuarine, Coastal and Shelf Science* 127: 37–45. <https://doi.org/10.1016/j.ecss.2013.04.015>.
- Palinkas, C.M., and C.A. Nittrouer. 2007. Modern sediment accumulation on the Po shelf, Adriatic Sea. *Continental Shelf Research* 27: 489–505. <https://doi.org/10.1016/j.csr.2006.11.006>.
- Palinkas, C.M., C.A. Nittrouer, R.A. Wheatcroft, and L. Langone. 2005. The use of ⁷Be to identify event and seasonal sedimentation near the Po River delta, Adriatic Sea. *Marine Geology* 222–223: 95–112. <https://doi.org/10.1016/j.margeo.2005.06.011>.
- Palinkas, C.M., and E.W. Koch. 2012. Sediment accumulation rates and submersed aquatic vegetation (SAV) distributions in the mesohaline Chesapeake Bay, USA. *Estuaries and Coasts* 35: 1416–1431. <https://doi.org/10.1007/s12237-012-9542-7>.
- Palinkas, C.M., and E. Russ. 2019. Spatial and temporal patterns of sedimentation in an infilling reservoir. *CATENA* 180: 120–131. <https://doi.org/10.1016/j.catena.2019.04.024>.
- Palinkas, C.M., L.P. Sanford, and E.W. Koch. 2017. Influence of shoreline stabilization structures on the nearshore sedimentary environment in mesohaline Chesapeake Bay. *Estuaries and Coasts* 41: 952–965. <https://doi.org/10.1007/s12237-017-0339-6>.
- Palmer, M.A. 2009. Reforming watershed restoration: Science in need of application and applications in need of science. *Estuaries and Coasts* 32: 1–17. <https://doi.org/10.1007/s12237-008-9129-5>.
- Phillips, S., and B. McGee. 2016. Ecosystem service benefits of a cleaner Chesapeake Bay. *Coastal Management* 44: 241–258. <https://doi.org/10.1080/08920753.2016.1160205>.
- Prasad, M.B.K., M.R.P. Sapiano, C.R. Anderson, W. Long, and R. Murtugudde. 2010. Long-term variability of nutrients and chlorophyll in the Chesapeake Bay: A retrospective analysis, 1985–2008. *Estuaries and Coasts* 33: 1128–1143. <https://doi.org/10.1007/s12237-010-9325-y>.
- Reckhow, K.H., M.N.B., and J.T. Simpson. 1980. *Modeling phosphorus loading and lake response under uncertainty: a manual compilation of export coefficients*. EPA 440/5–80–011. Washington, DC: US Environmental Protection Agency.
- Romkens, M.J.M., K. Helming, and S.N. Prasad. 2002. Soil erosion under different rainfall intensities, surface roughness, and soil water regimes. *CATENA* 46: 103–123. [https://doi.org/10.1016/S0341-8162\(01\)00161-8](https://doi.org/10.1016/S0341-8162(01)00161-8).
- Russ, E., and C. Palinkas. 2020. Evolving sediment dynamics due to anthropogenic processes in upper Chesapeake Bay. *Estuarine, Coastal and Shelf Science* 235: 106596. <https://doi.org/10.1016/j.ecss.2020.106596>.
- Saenger, C., T.M. Cronin, D. Willard, J. Halka, and R. Kerhin. 2008. Increased terrestrial to ocean sediment and carbon fluxes in the northern Chesapeake Bay associated with twentieth century land alteration. *Estuaries and Coasts* 31: 492–500. <https://doi.org/10.1007/s12237-008-9048-5>.
- Sanchez-Cabeza, J.A., and A.C. Ruiz-Fernández. 2012. ²¹⁰Pb sediment radiochronology: An integrated formulation and classification of dating models. *Geochimica Et Cosmochimica Acta* 82: 183–200. <https://doi.org/10.1016/j.gca.2010.12.024>.
- Sanford, L.P., and W.C. Boicourt. 1990. Wind-forced salt intrusion into a tributary estuary. *Journal of Geophysical Research* 95: 13357. <https://doi.org/10.1029/JC095iC08p13357>.
- Saunders, W. 1999. Preparation of DEMs for use in environmental modeling analysis. 1999 ESRI User Conference.
- Schilling, K.E., T.M. Isenhardt, J.A. Palmer, C.F. Wolter, and J. Spooner. 2011. Impacts of land-cover change on suspended sediment transport in two agricultural watersheds. *Journal of the American Water Resources Association* 47: 672–686. <https://doi.org/10.1111/j.1752-1688.2011.00533.x>.
- Schlesinger, W.H., and E.S. Bernhardt. 2013. *Biogeochemistry: an analysis of global change (3rd edition)*. Academic Press. <https://doi.org/10.1016/C2010-0-66291-2>.
- Sherriff, S.C., J.S. Rowan, O. Fenton, P. Jordan, and D. Ó hUallacháin. 2018. Sediment fingerprinting as a tool to identify temporal and spatial variability of sediment sources and transport pathways in agricultural catchments. *Agriculture, Ecosystems & Environment* 267: 188–200. <https://doi.org/10.1016/j.agee.2018.08.023>.
- Smith, E.P. 2020. Ending reliance on statistical significance will improve environmental inference and communication. *Estuaries and Coasts* 43: 1–6. <https://doi.org/10.1007/s12237-019-00679-y>.
- Smith, K.E.L., and J.F. Terrano. 2017. Shoreline change analysis of coastal and estuarine shorelines in Barnegat and Great Bays, NJ: 1839 to 2012. U.S. Geological Survey. <https://doi.org/10.5066/F75X275C>.
- Staver, K.W., and R.B. Brinsfield. 2001. Agriculture and water quality on the Maryland Eastern Shore: where do we go from here? *BioScience* 51: 859–868.
- Stevenson, J.C., M.S. Kearney, and E.C. Pendleton. 1985. Sedimentation and erosion in a Chesapeake Bay brackish marsh system. *Marine Geology* 67: 213–235. [https://doi.org/10.1016/0025-3227\(85\)90093-3](https://doi.org/10.1016/0025-3227(85)90093-3).

- Stockdon, H.F., A.H. Sallenger, J.H. List, and R.A. Holman. 2002. Estimation of shoreline position and change using airborne topographic Lidar data. *Journal of Coastal Research* 18: 502–513.
- Sutton, A.J., T.R. Fisher, and A.B. Gustafson. 2009. Historical changes in water quality at German Branch in the Choptank River basin. *Water, Air, & Soil Pollution* 199: 353–369.
- Tango, P.J., and R.A. Batiuk. 2016. Chesapeake Bay recovery and factors affecting trends: Long-term monitoring, indicators, and insights. *Regional Studies in Marine Science* 4: 12–20. <https://doi.org/10.1016/j.rsma.2015.11.010>.
- TNC, The Nature Conservancy. 2016. *Near-surface groundwater flow patterns across Akridge Farm's Duvall Tract, Talbot County, MD*.
- Tully, K., K. Gedan, R. Epanchin-Niell, A. Strong, E.S. Bernhardt, T. BenDor, M. Mitchell, et al. 2019. The invisible flood: The chemistry, ecology, and social implications of coastal saltwater intrusion. *BioScience* 69: 368–378. <https://doi.org/10.1093/biosci/biz027>.
- USDA, US Department of Agriculture. 2001. *The conservation reserve enhancement program*. PA-1701. Washington, DC: Farm Service Agency.
- Valderama, J.C. 1981. The simultaneous analysis of total N and P in natural waters. *Marine Chemistry* 10: 1009.
- Wagena, M.B., A.S. Collick, A.C. Ross, R.G. Najjar, B. Rau, A.R. Sommerlot, D.R. Fuka, P.J.A. Kleinman, and Z.M. Easton. 2018. Impact of climate change and climate anomalies on hydrologic and biogeochemical processes in an agricultural catchment of the Chesapeake Bay watershed, USA. *Science of the Total Environment* 637–638: 1443–1454. <https://doi.org/10.1016/j.scitotenv.2018.05.116>.
- Wasserstein, R.L., and N.A. Lazar. 2016. The ASA statement on p-values: Context, process, and purpose. *The American Statistician* 70: 129–133. <https://doi.org/10.1080/00031305.2016.1154108>.
- White, M.J., C. Santhi, N. Kannan, J.G. Arnold, D. Harmel, L. Norfleet, P. Allen, et al. 2014. Nutrient delivery from the Mississippi River to the Gulf of Mexico and effects of cropland conservation. *Journal of Soil and Water Conservation* 69: 26–40. <https://doi.org/10.2489/jswc.69.1.26>.
- Willard, D.A., T.M. Cronin, and S. Verardo. 2003. Late-Holocene climate and ecosystem history from Chesapeake Bay sediment cores, USA. *The Holocene* 13: 201–214. <https://doi.org/10.1191/0959683603hl607rp>.
- Zhang, Q., R.R. Murphy, R. Tian, M.K. Forsyth, E.M. Trentacoste, J. Keisman, and P.J. Tango. 2018. Chesapeake Bay's water quality condition has been recovering: Insights from a multimetric indicator assessment of thirty years of tidal monitoring data. *Science of the Total Environment* 637–638: 1617–1625. <https://doi.org/10.1016/j.scitotenv.2018.05.025>.