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Response of Tidal Creek Fish Communities to Dredging and Coastal Development Pressures in a Shallow-Water Estuary

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Abstract To investigate the effects of dredging and associated development pressures (i.e., shoreline armoring, developed land use) on fish, three sets of paired dredged and undredged tidal creeks were surveyed within Lynnhaven River, Virginia. Fish species diversity, community abundance, biomass, and size structure were compared among creeks and related to watershed, shoreline, and physicochemical characteristics. Mean fish community characteristics (e.g., abundance) were similar among creeks; however, species-specific analysis revealed subtle differences. Species biomass differed between dredged and undredged creeks, though species abundance was similar. Turbidity highly influenced differences in species abundance among creeks, while organic matter, dissolved oxygen, turbidity, and shoreline hardening may be influencing biomass patterns. The most recently dredged creek appeared to provide less suitable nursery habitat for some species than historically dredged creeks, suggesting initial adverse effects with eventual recovery. Protective measures, such as preservation of marshes, dredge depth, and time-of-year restrictions, may be moderating development and dredging pressures.

Keywords Tidal creeks · Chesapeake Bay · Dredging · Fish community · Development · Shallow-water habitats · Fringe marsh

Introduction

Coastal development pressures including pollution, commercial and recreational fishing, land use changes, shoreline modification, and dredging have contributed to significant alterations to estuarine systems. Development affects aquatic resources and associated habitats in complex and diverse ways, severing land–water linkages and disrupting critical functions. Shallow-water habitats, such as tidal flats, creeks, and shallow subtidal bottom, positioned in the landscape at the land–water interface are highly susceptible to development stressors. These highly productive habitats are established essential nursery areas for nekton, providing protection from predators and foraging opportunities for numerous fish, shellfish, and crustacean species (e.g., McIvor and Odum 1988; Ruiz et al. 1993). In the Chesapeake Bay, this critical resource area is under intense and increasing pressure from a variety of uses and users and generally exists without an operative comprehensive management plan. Tidal shoreline systems are managed by a complex framework of regulatory agencies that are each responsible for an individual resource, rather than the coastal zone as a whole ecosystem. Local governments implementing the Chesapeake Bay Preservation Act manage the riparian zone, intertidal areas fall under the purview of local wetland boards, and the subaqueous environment is the responsibility of the Virginia Marine Resources Commission. A further complication is that cumulative effects of the development on the coastal zone are often not fully incorporated into regulatory decisions due in part to difficulties in separating habitat and faunal responses driven by natural environmental variability versus human-induced stressors.

Watershed land development and shoreline alteration have been demonstrated to negatively affect biological communities and their habitats in many places (Beauchamp et al. 1994; Jennings et al. 1999; Dauer et al. 2000; Lerberg et al. 2000; DeLuca et al. 2004; Kiffney 2004; Scheuerell and Schindler 2004; Bilkovic et al. 2006; Seitz et al. 2006; Storry et al. 2006; Bilkovic and Roggero 2008). Coastal

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dredging commonly accompanies watershed and shoreline development. Adverse effects of dredging in coastal systems have been generally defined to include habitat removal, increased turbidity, alteration to current patterns, sediment, water quality, changes in salinity, and decreased flushing (e.g., Morton 1977; Johnston 1981; Newell et al. 1998; Wilber and Clarke 2001).

Whether these effects intensify or vary in shallow-water systems (≤ 2 m) is uncertain. Typically, shallow creeks are dredged to provide residential boat access, and maintenance (i.e., repeated) dredging is required to prevent siltation, allowing acute disturbances to become chronic stressors in the ecosystem. In the short term, the physical acts of dredging may impact food sources, causing reductions in the primary and secondary productivity of macrobenthic, microalgal, oyster reef, and vascular plant communities (Johnston 1981). Recurring physical disturbances caused by anthropogenic activities such as dredging could result in the loss of benthic prey community abundance and diversity through mechanisms such as the alteration of sediment chemistry and subsequent reduction of infauna recruitment (Marinelli and Woodin 2002). Alterations to topography and bathymetry may also change the accessibility to these systems and subsequently influence the interactions of biotic communities. In the longer term, predators may have enhanced access to areas that previously served as prey refuge habitat (Ruiz et al. 1993).

Critical to understanding the long and short-term effects of dredging on aquatic communities is an approximation of their rate of recovery. Primarily, efforts have examined the effects on and recovery rates of macrobenthic communities from dredging. Brooks et al. (2006) noted that no consistent pattern of macrobenthos response to dredging was found in the literature. However, other studies have shown that recovery is faster for benthic assemblages in lower versus higher salinity habitats (e.g., oligohaline vs. euhaline) or those associated with fine-grained sediments versus coarse-grained sediments (Newell et al. 1998). For nekton, after the immediate exposure to dredging disturbances is suspended (e.g., entrainment, elevated suspended sediments, noise level), recovery may be strongly influenced by the availability of suitable prey or habitats. In other words, nekton recovery rates may track the recolonization of food sources or the preservation of essential habitats.

Resource managers of shallow-water tidal systems are often forced to make rapid decisions on the potential impact from an activity (e.g., dredging) with inadequate information on expected ecosystem responses. Individual projects, such as shoreline alterations, are typically permitted with minimal consideration of the effects that multiple and cumulative stressors may be having on the system. Residential dredging is increasingly requested by individuals and community groups that desire navigable access to maintained municipal channels from their residences on small tidal creeks. Limited empirical information is available regarding the cumulative effects that converting shallow water or tidal flat habitats (≤2 m depth) to deeper open waters may have on marine resources.

Lynnhaven River, located in the southernmost extent of the Chesapeake Bay, is an example of a shallow-water tidal system under intense development pressure that is confronted with multiple and often conflicting coastal management issues. The watershed, located in the City of Virginia Beach, covers 206 km^2 or approximately one quarter of the area of Virginia Beach. Lynnhaven River has approximately 272 km of shoreline, and Broad Bay has an additional 138 km. Rapid development in and around the City of Virginia Beach over the past few decades has led to the loss of natural buffers and habitat (e.g., oyster, wetlands, and sea grasses), increased sedimentation, and degraded water quality (VDEQ 2004). Current management efforts include *The Lynnhaven Ecosystem Restoration Project*, led by US Army Corps of Engineers with State and Federal partners, which aims to identify and implement the most effective strategies for improving water quality, restoring oysters, marsh, and sea grasses, and managing siltation.

As in many similar systems, limited information exists on the effects that coastal development is having on tidal creek ecosystems within the Lynnhaven River restoration area. In particular, fish responses to and recovery from dredging in shallow-water tidal creeks are currently unknown. The main objective of this study was to investigate residential dredging effects on fish communities and secondarily to examine the influence and interaction of multiple anthropogenic stressors in tidal creeks.

Materials and Methods

Study Area

The Lynnhaven River, fed by the Western and Eastern Branches, flows into the Chesapeake Bay through a very narrow inlet characterized by fast tidal currents. This feature, in combination with the shallowness of the estuary (average depth < 0.75 m outside of navigation channels) and convoluted shoreline results in complex hydrodynamics often characterized by extreme fluctuations in physicochemical conditions. For instance, during the survey period (August–October 2006), salinity ranged from 2 to 36 in the Western Branch. Biological communities must either be adapted to tolerate exposures to intense and rapid shifts in physical condition or able to move to alternate habitats.

Three sets of paired dredged and undredged tidal creeks (North, Hebden, and Buchanan) were surveyed from the Western Branch of the Lynnhaven River, Virginia (Fig. 1). The creeks are all located on the eastern shore of the

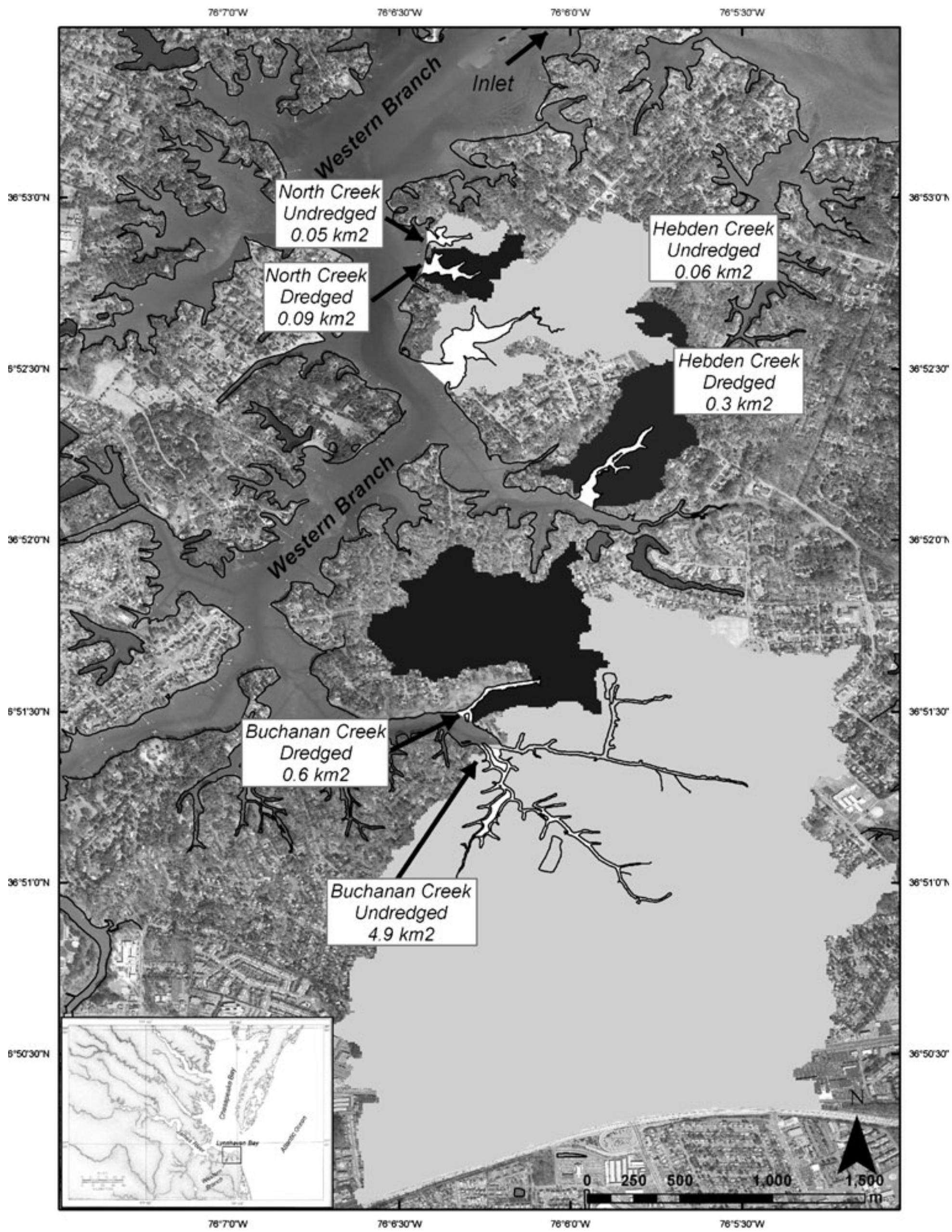


Fig. 1 Lynnhaven River tidal creek drainage boundaries and areas

Western Branch with mean salinity of 18–22. In order to consider both short- and long-term effects of dredging, a recently dredged creek (within 6 months of survey) and two creeks dredged several years ago were included as sites.

Watershed Characteristics

The linear distance of each creek from the inlet of the Lynnhaven River was estimated in ARCGIS. Dredge history for each creek was obtained from the Virginia Institute of Marine Science (VIMS) and Virginia Marine Resources Commission (VMRC) permit database (http://ccrm.vims.edu/perms/newpermits.html). The drainage basin was delineated for each creek with the ARCGIS Watershed Function and USGS National Elevation Dataset (10-m resolution). For each watershed, land cover data were obtained from the National Land Cover Dataset (30-m raster coverage, NLCD 2001), and impervious surface estimates were extracted from the dataset RESAC 2000 CBW Impervious Surface Product.

Shoreline and Riparian Characteristics

Shoreline and riparian characteristics within each tidal creek were determined by a comprehensive inventory of shoreline condition. The inventory protocol was specifically developed for Virginia and Maryland coastlines and included a spatially explicit method for collecting, classifying, mapping, and reporting conditions along the shore. The inventory employed a continuous three-tiered shoreline assessment approach, dividing the shore zone into three regions: (1) immediate riparian zone, evaluated for land use; (2) bank, evaluated for height, stability, cover, and natural protection; and (3) shoreline, describing the presence of shoreline structures for shore protection and recreational purposes. Data collection was performed in the field from a small shoal draft vessel, navigating at slow speeds parallel to the shoreline. A complete set of geographically referenced shoreline data was acquired using a preprogrammed data dictionary in a handheld Trimble Global Positioning System (GPS) GeoExplorer receiver that included a suite of characteristics describing the shoreland's land use, bank condition, and shoreline features. GeoExplorers were accurate to within 10 cm of true position with extended observations and differential correction. GPS field data were converted to GIS spatial coverages which were corrected to reflect true shoreline geometry (for additional details, see Berman et al. 2007). Descriptors from the inventory may be used as indicators of shoreline disturbance and potential habitat degradation for both pelagic and benthic organisms. Percentage of shoreline land use types (developed, forested, grass, scrub-shrub), hardened structure (bulkhead, riprap revetment, dilapidated bulkhead, unconventional, and debris), and marsh (fringe, extensive, and invasive *Phragmites australis*) were summarized for each of the six surveyed tidal creeks.

Physicochemical Characteristics

During each fish collection event, auxiliary data were collected on variables with the potential to influence fish distributions, including water depth, dissolved oxygen (DO), salinity, conductivity, pH, turbidity, tides, and water temperature. Water depth in channel and at channel edge was recorded for every sampling event (n=129). Two dedicated water quality YSI sondes were placed in Buchanan dredged and undredged creeks, which had relatively low boat traffic and a reduced likelihood of instrument disturbance, for the entire survey period (August–October) to continuously record dissolved oxygen, salinity, conductivity, pH, turbidity, and temperature. Precipitation data were obtained from *Wunderground.com*, Station KVAVIRGI14, located at the North end of the Beach in Virginia Beach, Virginia (Lat: N 36° 51′ 57 ″ (36.866°); Lon: W 75° 59′ 47 ″ (−75.996°)). Precipitation were measured and recorded with Rainwise MK III hardware and Weather View 32 v60 software. Within each creek, three surface sediment samples were obtained along a randomly selected upstream–downstream transect in the littoral zone and assessed for grain size and organic content.

Physicochemical measures (salinity, temperature, DO, pH, turbidity, sediment composition (i.e., percent clay), sediment organic matter, and water depth) were compared among creeks and between creeks grouped as dredged or undredged with one-way analyses of variance (ANOVAs). Variables were transformed when necessary to correct for nonnormality. If assumption of homogeneity of variance was violated, the Brown–Forsythe F ratio was reported which corrects for violations and is preferable to the F statistic when the assumption of equal variances does not hold (Brown and Forsythe 1974). Post hoc comparisons were completed with Tukey honestly significant differential (HSD), except in cases where unequal variances occurred where Dunnett's T3 pairwise comparison statistic was applied. Statistical tests were performed in SPSS 17.0.

Fish Community Survey

Fish were sampled during the day with multiple gear types (gill net, haul, and beach seines) once per month for 3 months (August, September, October) in 2006. To capture representative tidal creek assemblages, surveys were conducted during the time of the year when abundance and diversity are generally highest in temperate estuaries (e.g., Richards and Castagna 1970; Hoff and Ibara 1977; Rountree and Able 1992). Each gear type selects for various components of a fish community in a given location. Experimental gill nets had five panels of varying mesh size to target juvenile to adult fish, beach seines selected for nearshore juvenile species, and haul seines selectively captured small pelagic nekton. Monthly sampling occurred during 8–17 August, 18–21 September, and 16–19 October, 2006. Alternate survey methods were utilized in an individual creek on separate days only, for example, gill nets might be deployed on the first sample date and haul seines conducted in the same creek on the second sample date to reduce potential disturbance.

Gill nets were deployed on high ebb tide and extended across the creek mouth to block fish passage; nets were retrieved at low tide (approximately 2–3-h sets). Each monofilament gill net (38 m long × 2.4 m deep) consists of five panels that are 7.6 m in length with the mesh sizes: 25.4 (#4 twine size), 38.1, 50.8, 63.5, and 76.2 mm (#6 twine size). An additional block net was necessary in Hebden undredged creek in order to obstruct the entire creek mouth. Data from capture on the auxiliary wing net were not included in analyses.

Haul seines were conducted with a bow-mounted net of 1.83 m wide × 0.6 m high × 3.7 m in length. The funnel was 2.4 m in length (6.4 mm delta) with a 1.2-m-long cod end (3.2-mm delta). Funnel mouth opening to cod end was 508 mm in diameter. The net was attached to a frame, and the cod end was cinched close with rope. Four replicate hauls were conducted for 2 min (to meet a lower limit of 50 m^3 of water filtered) in each tidal creek during monthly sampling events. The volume of the tow was determined by multiplying the area of the net by the distance towed (based on flowmeter revolutions).

Two beach seine replicate hauls (30.5 m × 1.22 m bagless seine of 6.4 mmbar mesh) were conducted near the mouth of each creek. One end of the seine was held on shore or as close to shore as possible. The other was fully stretched perpendicular to the shore and swept with the current over a 729-m^2 quarter circle quadrant. When depths of 1.22 m or greater were encountered, the offshore end was deployed along this depth contour. After encircling an area, the mouth of the seine was closed by crossing over the lead lines of each wing of the net. The seine was slowly hauled closed and the lead line continually checked to ensure contact with the bottom.

Fish and blue crab (*Callinectes sapidus*) species captured were enumerated, and a subset of 25 individuals was measured (total length to the nearest 1.0 mm), weighed when possible (wet weight to the nearest 0.1 g), and released. Weights for all measured fish were estimated using species-specific length–weight regressions either generated from data collected or reported in the literature. In instances when greater than 25 fish of a species were collected in a given sampling event, the average weight of a species (based on the subset of fish measured at that time) was applied to those specimens not weighed to estimate a total biomass.

Fish Community Structure and Dredging Effects

To assess the effect of dredging on fish, community structure (abundance, biomass, diversity, and size distribution) was estimated and compared among surveyed creeks (three dredged, three undredged). Individual gear-type catches were combined to assess the overall community structure in each creek. While no gear can inclusively sample every habitat or life stage of fish species, the combination of gear types allows the relative characterization of predator and prey use of the tidal creeks during the period surveyed. Total abundance, total biomass, and diversity measures (e.g., Shannon–Weiner diversity index and Pielou's evenness) were estimated for each creek. Monthly samples were treated as replicates, and among-creek differences in total abundance biomass and diversity (Shannon–Weiner and Pielou's evenness) were assessed with one-way ANOVAs, and post hoc comparisons were completed with Tukey HSD. Abundance, biomass, and length data were log-transformed to meet normality requirements.

Length frequency distributions among (1) tidal creeks and (2) tidal creeks for select abundant species were graphed and compared using a distribution-free two-sample Kolmogorov–Smirnov (KS) goodness-of-fit test (Hollander and Wolfe 1999). Since pairwise tests rather than multiple comparisons are preformed with the KS procedure, the significance level for each subtest was adjusted using Bonferroni correction (Neumann and Allen 2007). The experiment-wise error rate (p=0.05) was divided by the number of subtests preformed (0.05/14 for tidal creek comparisons). Pooled fish length data across gear type and month (comprised of the subset of fish collected that were measured; n=3,130) were examined to determine community structure differences among tidal creeks. Species that were determined to be influencing statistical differences among tidal creeks or with known habitat associations (i.e., tidal marsh) were examined in more detail to ascertain patterns. For select abundant species (Atlantic silverside (*Menidia menidia*), bay anchovy (*Anchoa mitchilli*), and mummichog (*Fundulus heteroclitus*)), the percentage of total catch that were young of year (YOY) was estimated for each creek. Size cutoff values used to distinguish YOY individuals from adults were obtained from the scientific literature (e.g., Rountree and Able 1992; Teo and Able 2003; Jung and Houde 2004).

Fish abundance and biomass similarities among creeks were examined independently with hierarchical cluster analysis, nonparametric multidimensional scaling (nMDS), and analysis of similarities (ANOSIM) in PRIMER 6.0. Prior to the MDS ordination and hierarchical cluster analysis, species abundances were square-root-transformed to moderately down weight the effect of dominant species, and a Bray–Curtis coefficient was used to calculate the similarity matrix. Hierarchical cluster analysis implements hierarchical agglomerative clustering, which is plotted on a dendrogram. The applied cluster mode algorithm was "group average" which means the new node takes the average similarity of the individual nodes to calculate the distance between clusters. MDS ordinates sites based on similarities in species makeup, using rank order of distances to map out relationships. A stress coefficient represents the goodness of fit of the data to a nonparametric regression, and acceptable ordinations of data occur when stress values are <0.2 (Clarke and Warwick 2001). Factors were overlaid on the MDS plot to visualize community groupings in relation to and ANOSIM was used to test relationships among the following: (1) dredged state, dredged or undredged; (2) month of survey, August, September, and October; and (3) paired creek group, North, Hebden, or Buchanan.

Exploration of species contributions to describing similarities within and dissimilarities among groups was completed with Similarity Percentages (SIMPER) procedure (PRIMER 6.0). This method uses relative abundance or biomass, represented by Bray–Curtis similarities, to determine those species contributing the most to overall dissimilarity between pairs of groups (Clarke and Warwick 2001).

Fish Community Structure and Other Environmental Variables

In order to detect possible reasons for the distributional pattern of fish among creeks, a comparison between the Bray–Curtis similarity matrices of square-root-transformed fish abundance and biomass and the normalized matrix of environmental parameters was performed and tested by the standard BIO-ENV routine (PRIMER 6.0; Clarke and Warwick 2001). Watershed, shoreline, riparian, and physicochemical characteristics described in Table 1 were evaluated as candidate environmental parameters potentially influencing fish community patterns. An initial environmental matrix was constructed from parameters anticipated to directly influence fish distribution (e.g., minimum mean DO, mean NTU) or to represent other nondredging development pressures (e.g., shoreline alteration, land conversion) that may indirectly affect fish communities. Prior to BIO-ENV testing, correlations between $\log(x+1)$-transformed candidate parameters in the initial matrix were analyzed by a Draftsman plot routine to reduce redundant variables with mutual correlations ($p \geq 0.95$). After colinearity inspection, 13 factors were excluded because of their strong correlation (Spearman rank correlation ≥ 0.95) with other variables: distance to inlet, mean minimum and maximum depth, pH, turbidity, and salinity, percentage of watershed residential land use, mean temperature, and mean and mean maximum DO. Final representative environmental factors (15) retained in the BIO-ENV analysis were watershed area (km^2), impervious surface (%), shoreline hardening (%), residential riparian land use (%), fringe marsh (%), fringe *Phragmites* (%), mean depth (m), mean minimum and maximum temperature (°C), mean minimum DO (mg/L), mean salinity, mean turbidity (NTU), mean pH, clay/silt (%), and mean organic matter (%) (Table 1). A dissimilarity matrix was calculated for creek environmental data using Euclidian distance on $\log(x+1)$-transformed data that were normalized to eliminate differences in measurement scale. The BIO-ENV procedure maximizes rank correlations between biotic and environmental matrices expressed as weighted Spearman rank correlation ρ. The maximum number of trial environmental variables to try in combination was set at four. A significance test was calculated based on random 999 permutations of sample names.

Results

Watershed Characteristics

Paired creeks North undredged (NU) and North dredged (ND) were the closest to the inlet of the surveyed creeks. NU is an unnamed creek that is undredged and adjacent to the recently dredged ND (permitted activity occurred in February 2006). These creeks have the smallest drainage area of the three pairs (0.06 and 0.09 km^2, respectively). Based on the National Land Cover Dataset (NLCD 2001), NU and ND are located in mixed land use watersheds with relatively low impervious surface (Table 1; Fig. 1).

Paired creeks Hebden undredged (HU) and Hebden dredged (HD) are located near and within Hebden Creek at a moderate distance from the inlet. HU is a tidal creek with a 0.9-km^2 watershed draining into the mainstem of Hebden Creek. HD (dredged ~March 2000) is adjacent to the mainstem of Hebden Creek and has a drainage area of 0.3 km^2. HU is located in a mixed land use watershed, HD watershed is predominantly forested, and both watersheds have relatively low impervious surface (Table 1; Fig. 1).

Paired creeks Buchanan dredged (BD) and Buchanan undredged (BU) are the furthest from the inlet and located within Buchanan Creek. BU has the largest drainage basin of 4.9 km^2, while BD (dredging activity occurred ~July 2000) is similar in watershed area to Hebden creeks (0.6 km^2). Both watersheds are predominantly residential, with relatively moderate levels of impervious surface (Table 1; Fig. 1). For all creeks, impervious surface was highly correlated to the amount of residential land use in the watershed (Pearson correlation=0.94, $p=0.006$); accordingly, Buchanan creek watersheds contained the highest amounts of impervious surface (BU=36%; BD=19%).

Shoreline and Riparian Characteristics

All six tidal creek watersheds consisted of equal to or greater than 50% residential riparian land use, with total armored shorelines ranging from 0% to 32% (HU had 0% armored shoreline and the other creeks had typically ≥20%). While the tidal creeks were heavily developed, fringe marsh dominated shorelines in all creeks (55–94%). The amount of *P. australis* ranged from 0% to 22% with the highest in the most developed shoreline of BD creek (91% residential riparian land use and 22% *P. australis*); however, residential riparian land use and *P. australis* were not significantly correlated (Pearson correlation=0.68, $p=0.1$; Table 1).

Physicochemical Characteristics

Select physicochemical measures followed an expected gradient with increasing distance from the inlet, e.g., salinity and pH decreased moving away from the inlet (North, Hebden > Buchanan paired creek group, one-way ANOVA, salinity: $F(2, 49)=5.9$, $p=0.005$; pH: $F(2, 49)=10.5$, $p<0.0001$; Table 1). Dissolved oxygen ($\mathrm{mg\,L^{-1}} \pm$SE) was significantly lower in Buchanan creeks (5.7±0.2) versus Hebden and North creeks (6.7±0.2; 7.0±0.2, respectively; one-way ANOVA, $F(2, 49)=3.3$, $p=0.05$;

Table 1 Watershed, shoreline, riparian, physicochemical, and fish community characteristics of tidal creeks in the Western Branch of the Lynnhaven River

	North undredged	North dredged	Hebden undredged	Hebden dredged	Buchanan undredged	Buchanan dredged
Watershed characteristics						
Watershed area (km^2)[a]	0.05	0.09	0.6	0.3	4.9	0.6
Distance from inlet (m)	3,820	4,025	4,550	5,700	7,620	7,475
Watershed land cover (%)						
Residential	25.0	12.8	27.7	19.7	82.8	78.0
Barren	28.6	14.9	6.0	7.0	1.6	2.0
Forest	19.6	42.6	38.8	52.5	7.0	12.5
Pasture/hay/crops	16.1	18.1	17.1	12.4	5.3	5.8
Wetlands	10.7	11.7	10.3	8.4	3.3	1.7
Impervious surface (%)[a]	2.2	0.8	3.3	2.9	36.0	19.2
Dredge date	–	Feb-06	–	Mar-00	–	Jul-00
Shoreline and riparian characteristics						
Shoreline structure (%)						
Bulkhead	21.07	12.89	0.00	9.71	0.31	32.20
Riprap	8.88	9.55	0.00	8.78	3.03	0.00
Total hardening[a]	29.96	24.43	0.00	20.20	8.85	32.20
Riparian land cover (%)						
Residential[a]	49.56	71.84	63.05	50.50	65.22	91.13
Forest	33.00	12.16	22.28	48.19	28.37	8.87
Grass	2.26	5.05	4.05	1.31	2.72	0.00
Scrub-shrub	15.18	10.94	10.63	0.00	3.69	0.00
Marsh (%)						
Fringe[a]	79.09	59.72	71.28	54.81	77.55	93.52
Extensive	0.00	0.00	0.00	0.00	18.93	0.00
Phragmites[a]	0.00	1.81	6.30	9.42	1.07	22.17
Fringe marsh (%) associated with hardened shoreline	38.1	27.8	0.0	6.3	10.7	27.4
Physicochemical parameters						
Mean depth (m)[a]	0.9 (0.2)	1.4 (0.4)	1.0 (0.2)	1.3 (0.3)	1.2 (0.3)	1.1 (0.3)
Mean temperature (°C)	23.2 (4.2)	23.2 (4.3)	24.4 (5.3)	23.4 (5.4)	23.7 (5.1)	23.3 (4.6)
Temperature range (°C)[a]	18.4–28.0	18.4–28.2	17.2–30.0	16.8–29.5	17.6–29.4	17.7–28.3
Mean DO ($mg L^{-1}$)	6.9 (1.2)	7.1 (1.3)	6.9 (1.4)	6.6 (1.6)	5.7 (0.8)	5.8 (1.8)
DO range ($mg L^{-1}$)[a]	6.3–8.2	6.2–8.3	5.4–7.6	4.5–7.7	5.1–6.5	3.8–7.0
Mean salinity[a]	21.2 (1.0)	21.1 (1.2)	21.2 (1.7)	20.9 (1.6)	18.6 (3.5)	19.4 (2.7)
Salinity range	20.4–22.5	20.1–22.7	19.3–23.1	19.2–22.9	15.0–22.0	16.4–22.2
Mean NTU[a]	18.0 (10.0)	18.4 (9.4)	38.3 (39.2)	54.7 (44.9)	34.4 (16.6)	48.1 (49.6)
NTU range	8.0–26.2	13.0–27.7	11.3–80.4	25.1–112.3	18.7–44.3	18.4–65.4
Mean pH[a]	7.9 (0.2)	7.9 (0.2)	7.8 (0.3)	7.7 (0.3)	7.4 (0.6)	7.5 (0.2)
pH range	7.7–8.0	7.7–8.1	7.4–8.0	7.3–8.0	6.8–7.9	7.3–7.7
Sediment characteristics						
Mean clay (%)[a]	37.6 (3.8)	36.8 (3.0)	38.2 (3.1)	40.7 (1.7)	45.7 (4.9)	48.6 (3.3)
Mean silt (%)[a]	53.8 (3.6)	41.8 (6.3)	54.4 (4.0)	43.9 (3.5)	36.8 (1.8)	44.8 (1.8)
Mean sand (%)	8.6 (2.5)	21.4 (8.5)	7.4 (1.3)	15.4 (5.0)	17.4 (3.8)	6.6 (2.2)
Mean gravel (%)	0.0	0.0	0.0	0.0	0.2 (0.2)	0.0
Mean moisture (%)	57.8 (3.5)	56.7 (2.5)	56.0 (4.9)	66.1 (4.2)	68.2 (4.1)	67.2 (1.7)
Mean organic matter (%)[a]	7.0 (0.7)	7.8 (1.7)	6.7 (1.7)	11.4 (2.0)	13.9 (2.1)	11.0 (1.3)
Fish community characteristics						
Total abundance	409	292	1,032	1,074	915	2,105
Total biomass (kg)	66.2	36.9	101.0	15.5	26.1	24.7

Table 1 (continued)

	North undredged	North dredged	Hebden undredged	Hebden dredged	Buchanan undredged	Buchanan dredged
Total number of species	14	13	20	16	18	18
Number species=99%	10	10	12	10	10	8
Shannon–Weiner	2.16	2.11	2.73	2.14	2.47	2.22
Pielou's evenness	0.71	0.66	0.61	0.56	0.53	0.50
Average length (cm)	14.2 (0.8)	13.1 (0.8)	12.1 (0.5)	6.6 (0.2)	8.4 (0.4)	7.2 (0.2)
Range length (cm)	2.6–50	2.7–46.4	2.5–50.0	2.3–48.0	2.3–47.3	1.9–47.5
Average weight (g)	163.0 (17.4)	137.3(16.8)	122.0 (10.7)	20.2 (3.9)	53.2 (8.2)	20.0 (3.5)
Range weight (g)	1.0–1,502.5	1.0–1,304.1	1.0–1,474.2	1.0–1,162.3	1.0–1,134.0	1.0–1,247.4

Physicochemical ranges (e.g., temperature range) represent the mean minimum and mean maximum values recorded. Values in parenthesis represent standard error

[a] Variables included in the environmental matrix that was compared with biotic patterns (fish abundance and biomass similarity matrices)

Table 1). However, DO values met the Chesapeake Bay criterion for larval, juvenile, and adult fish growth of ≥ 5.0 $mg L^{-1}$ (US EPA 2003). Turbidity (NTU±SE) was lower in North creeks (18.2±1.5) compared to Hebden and Buchanan creeks (45.9±6.3; 41.2±5.7, respectively; one-way ANOVA, $F(2, 49)=3.1$, $p=0.05$). While Virginia does not have specific turbidity standards for marine aquatic life, North Carolina estuarine water quality criteria is 25 NTU (NC-DENR 2003), which is exceeded in Hebden and Buchanan creeks (Table 1). No significant difference was observed between creeks when grouped as dredged or undredged for salinity, temperature, DO, pH, and turbidity (one-way ANOVA, $F(1, 50)=0.002$, 0.8, 0.5, 0.2, 2.5, respectively, $p>0.05$).

Mean depths ranged from 0.9 to 1.4 m. Between pairs, dredged creeks were on average ≥0.3 m deeper than undredged creeks, with the exception of the Buchanan pair where the dredged creek was shallower than the undredged (Table 1). Among the creeks, NU and HU had similar average sampled depths that were shallower than all other creeks (one-way ANOVA, $F(5, 123)=9.97$, $p=0.05$; Brown–Forsythe F ratio $F(5, 103)=9.89$, $p<0.0001$).

Continuous water quality stations in BD and BU creeks indicated that from August to October 2006, the creeks responded similarly to precipitation events including the extreme Hurricane Ernesto event on September 1, 2006. Creeks were experiencing a drought as sampling began in August as indicated by salinities in excess of 30; the high precipitation Hurricane event led to drops in salinity to near zero. Salinity typically remained between 10 and 20 following the storm. During the sampling season, conditions fluctuated between extremes; salinity ranged from approximately 2 to 36 and dissolved oxygen from 2 to 11 $mg L^{-1}$ (Fig. 2). Dissolved oxygen fluctuated dramatically with the diel cycle following an established pattern in shallow-water tidal creeks of minimum levels in the early morning due to local production and respiration patterns and/or incursion of bottom waters through winds and tides with maximums in the afternoon (Breitburg 1990; D'Avanzo and Kremer 1994).

The tidal creeks were clay/silt-dominated with high levels of organic matter (7–14%). Sediment composition consisted of higher amounts of clay and organic matter in Hebden dredged creek and Buchanan creeks (one-way ANOVA, organic matter: $F(1, 16)=14.8$, $p=0.001$; clay: $F(1, 16)=7.6$, $p=0.01$; Table 1). Significant differences in sediment composition did not occur between dredged and undredged creeks; however, in North and Hebden creeks, silt and sand exhibited opposite trends, with higher amounts of silt and lower amounts of sand in the undredged versus the paired dredged creek. An opposite pattern was observed for the Buchanan creeks (Table 1).

Fish Community Structure and Dredging Effects

A total of 5,732 fish, 93 blue crab, and 30 species were collected from the six tidal creeks surveyed in August–October. The combined catches were dominated by Atlantic silverside, bay anchovy, gizzard shad (*Dorosoma cepedianum*), silver perch (*Bairdiella chrysoura*), Atlantic menhaden (*Brevoortia tyrannus*), and mummichog (> 90% of catch; Table 2). Gizzard shad, red drum (*Sciaenops ocellatus*), and Atlantic menhaden made up 90% of total biomass. The beach seine captured the largest number of species (26, eight unique species observed in seine), and the haul seine and gill net captured equal numbers of species (13) with one unique species in the haul seine and two in the gill net. Generally, larger fish occupied NU, ND, and HU compared to HD, BD, BU (KS test, $p<0.0001$, $n=310$, 267, 662, 698, 475, and 718, respectively).

Mean fish community characteristics were similar among the tidal creeks surveyed (one-way ANOVA: total

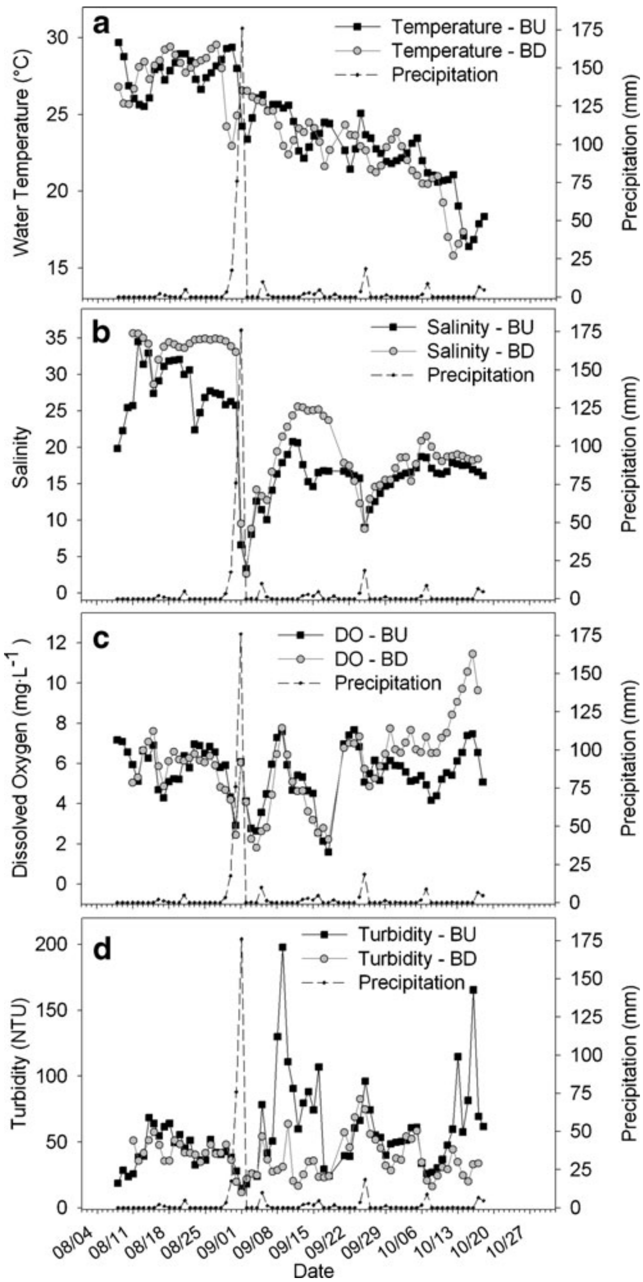


Fig. 2 Similar physicochemical patterns were observed in mean daily **a** salinity, **b** water temperature, **c** dissolved oxygen, and **d** turbidity for Buchanan undredged (*BU*) and Buchanan dredged (*BD*) tidal creeks from August to October 2006. Creeks were experiencing a drought as sampling began in August as indicated by salinities in excess of 30; a high precipitation Hurricane event on September 1, 2006, led to drops in salinity to near zero. Salinity typically remained between 10 and 20 following the storm. The magnitude of fluctuations in dissolved oxygen and turbidity increased after the storm event

abundance ($F(5, 17)=0.66$, $p=0.13$); total biomass ($F(5, 17)=2.42$, $p=0.10$); diversity (Shannon–Weiner; $F(5, 17)=0.74$, $p=0.61$); and evenness ($F(5, 17)=0.57$, $p=0.72$); Fig. 3, Table 1). However, differences between North creeks in relation to Hebden and Buchanan creeks were noted in multivariate analyses of communities (hierarchical cluster analysis and MDS) which examined similarity in abundance by species (global $R=0.18$, $p=0.02$; F(pairwise ANOSIM: North vs. Hebden $R=0.4$, $p=0.004$; North vs. Buchanan $R=0.2$, $p=0.03$); Fig. 4). With the exception of Atlantic croaker (*Micropogonias undulatus*), contributing species (including Atlantic silverside, bay anchovy, mummichog, silver perch, and blue crab) were in higher abundance in Hebden and Buchanan creeks as compared to North creeks (SIMPER, Diss/SD>1.1). No differences in species abundance were observed between creeks grouped as dredged or undredged (ANOSIM, global $R=0.03$, $p=0.62$) or by months pooled across sites (ANOSIM, global $R=0.14$, $p=0.06$).

Conversely, multivariate comparison of biomass among creeks indicated a difference in dredged from undredged creeks (ANOSIM, global $R=0.12$, $p=0.05$; Fig. 5). Major species contributing to biomass differences included Atlantic menhaden, Atlantic silverside, bay anchovy, gizzard shad, mummichog, mullet spp. (*Mugil* spp.), red drum, silver perch, and spot (*Leiostomus xanthurus*; SIMPER analysis, Diss/SD>1.1). Atlantic menhaden, red drum, and silver perch each had a similar size distribution for dredged and undredged creeks (KS test: $p=0.2$, $n=6$, 17; $p=0.2$, $n=83$, 97; $p=0.6$, $n=138$, 175, respectively) but had higher biomass in undredged creeks (Table 2). The size distribution of the most abundant species, Atlantic silverside, varied between dredged and undredged creeks (KS test: $p<0.001$, $n=451$, 398, respectively), as well as between paired creek groups (KS test; North: $p<0.001$, $n=63$ (NU), $n=9$ (ND); Hebden: $p<0.0001$, $n=163$ (HU), $n=240$ (HD); Buchanan: $p<0.0001$, $n=172$ (BU), $n=202$ (BD)), with higher biomass in undredged creeks (Fig. 6a, Table 2). The YOY of Atlantic silverside (20–60 mm TL) were absent in ND as opposed to all other creeks (Fig. 6a). Large adult gizzard shad were more prevalent in NU, ND, and HU versus HD, BD, and BU, and size distribution varied between dredged and undredged creeks (KS test: $p<0.0001$, $n=193$, 225, respectively; Fig. 6b). Conversely, spot, mullet spp., and bay anchovy exhibited similar size distributions in dredged and undredged creeks (KS test: $p=0.3$, $n=34$, 31; $p=0.6$, $n=32$, 12; $p=0.3$, $n=559$, 379, respectively) with higher biomass in dredged creeks. The YOY of the second most abundant species, bay anchovy (< 40 mm TL), was observed in the lowest proportion in ND (46.4%) compared to other creeks (55.7–70%) (Fig. 6c). Bay anchovy size distributions differed between each paired creek group (KS test; North: $p=0.02$, $n=86$ (NU), $n=110$ (ND); Hebden: $p=0.05$, $n=169$ (HU), $n=239$ (HD); Buchanan: $p<0.0001$, $n=124$ (BU), $n=210$ (BD)). Mummichog size distribution varied between dredged and undredged creeks (KS test: $p<0.0001$, $n=133$, 27, respectively), and higher biomass was observed in dredged creeks (Fig. 6d, Table 2). The YOY of mummichog (< 50 mm total length) were only collected in the three dredged creeks surveyed, with the highest proportion of the

Table 2 Summary statistics for fish species by site

Scientific name	Common name	North creeks								Hebden creeks			
		Undredged				Dredged				Undredged			
		Total	*n*	Ave TL (SE)	Ave Wt (SE)	Total	*n*	Ave TL (SE)	Ave Wt (SE)	Total	*n*	Ave TL (SE)	Ave Wt (SE)
Anguilla rostrata	American eel	–	–	–	–	–	–	–	–	3	3	45.0 (3.2)	193 (54)
Micropogonias undulatus	Atlantic croaker	7	7	16.7 (5.7)	267 (128)	4	4	15 (6.3)	239 (189)	9	9	19.3 (2.8)	148 (43)
Brevoortia tyrannus	Atlantic menhaden	117	37	14.7 (0.2)	35.6 (1.7)	18	18	13.7 (0.5)	30.0 (3.6)	36	36	13.1 (0.2)	24.6 (1.6)
Menidia menidia	Atlantic Silverside	73	63	5.9 (0.2)	2.8 (0.2)	9	9	8.4 (0.2)	6.2 (0.6)	307	163	5.2 (0.1)	2.3 (0.2)
Anchoa mitchilli	Bay anchovy	87	87	3.9 (0.1)	1.3 (0.1)	135	110	4.1 (0.1)	1.3 (0.1)	325	169	4.1 (0.1)	1.2 (0.03)
Symphurus plagiusa	Black cheek tonguefish	–	–	–	–	–	–	–	–	–	–	–	–
Callinectes sapidus	Blue crab	6	–	–	–	2	–	–	–	18	–	–	–
Pomatomus saltatrix	Bluefish	–	–	–	–	–	–	–	–	1	1	14.1	25.8
Caranx hippos	Crevalle jack	–	–	–	–	–	–	–	–	–	–	–	–
Dorosoma cepedianum	Gizzard shad	69	67	37.2 (1.0)	657 (32)	53	53	37.0 (0.9)	626 (33)	111	111	34.1 (1.1)	616 (34)
Goby spp	Goby spp	–	–	–	–	–	–	–	–	–	–	–	–
Trinectes maculatus	Hogchoker	1	1	14.8	6.5	–	–	–	–	–	–	–	–
Elops saurus	Ladyfish	–	–	–	–	1	1	11.7	8.6	–	–	–	–
Mugil spp	Mullet spp	10	6	12.1 (1.9)	15.3 (7.8)	7	7	15.3 (1.9)	46.7 (0.3)	8	5	19.6 (2.0)	83.5 (17.7)
Fundulus heteroclitus	Mummichog	–	–	–	–	18	18	7.1 (0.3)	5.9 (0.7)	12	12	6.6 (0.2)	4.5 (0.6)
Gobiosoma bosc	Naked goby	1	1	4.4	4.1	–	–	–	–	–	–	–	–
Trachinotus falcatus	Permit	–	–	–	–	–	–	–	–	2	2	6.4 (0.6)	3.2 (0.5)
Sciaenops ocellatus	Red drum	2	2	38.2 (5.6)	524 (297)	1	1	32.2	821	9	9	34.6 (0.1)	526 (34)
Cyprinodon variegatus	Sheepshead minnow	–	–	–	–	–	–	–	–	–	–	–	–
Bairdiella chrysoura	Silver perch	29	29	8.1 (0.3)	6.6 (1.0)	41	41	9.1 (0.4)	13.2 (2.8)	157	90	8.3 (0.2)	9.5 (2.4)
Leiostomus xanthurus	Spot	–	–	–	–	–	–	–	–	30	30	16.3 (0.3)	73.5 (6.7)
Eucinostomus argenteus	Spotfin mojarra	–	–	–	–	–	–	–	–	2	2	6.4 (0.1)	2.8 (0.1)
Cynoscion nebulosus	Spotted sea trout	–	–	–	–	–	–	–	–	1	1	43.0	739.4
Anchoa hepsetus	Striped anchovy	10	10	6.7 (0.3)	1.8 (0.4)	2	2	6.8 (0.7)	1.8 (0.6)	16	16	6.5 (0.1)	1.6 (.01)
Morone saxatilis	Striped bass	–	–	–	–	–	–	–	–	–	–	–	–
Fundulus majalis	Striped killifish	1	1	6.9	3.9	3	3	6.0 (0.3)	2.1 (0.3)	1	1	6.0	2.4
Paralichthys dentatus	Summer flounder	2	2	43.9 (6.2)	1,063 (439)	–	–	–	–	–	–	–	–
Lobotes surinamensis	Tripletail	–	–	–	–	–	–	–	–	1	1	3.6	1.2
Cynoscion regalis	Weakfish	–	–	–	–	–	–	–	–	1	1	8.0	3.7
Morone americana	White perch	–	–	–	–	–	–	–	–	–	–	–	–
All Species		415	313	14.2 (0.8)	163 (17.4)	294	267	13.1 (0.8)	137 (0.8)	1,050	662	12.1 (0.5)	122 (10.7)

Scientific name	Common name	Hebden creeks				Buchanan creeks							
		Dredged				Undredged				Dredged			
		Total	*n*	Ave TL (SE)	Ave Wt (SE)	Total	*n*	Ave TL (SE)	Ave Wt (SE)	Total	*n*	Ave TL (SE)	Ave Wt (SE)
Anguilla rostrata	American eel	–	–	–	–	1	1	21.0	6.7	–	–	–	–
Micropogonias undulatus	Atlantic croaker	–	–	–	–	1	1	4.1	1.0	11	11	4.8 (0.3)	1.4 (0.2)
Brevoortia tyrannus	Atlantic menhaden	5	5	12.4 (0.4)	20.2 (2.0)	24	24	14.4 (0.3)	33.3 (1.9)	191	60	13.5 (0.3)	28.7 (1.7)
Menidia menidia	Atlantic Silverside	509	240	4.6 (0.1)	1.4 (0.1)	316	172	4.3 (0.1)	1.2 (0.1)	1,153	202	4.5 (0.1)	1.4 (0.1)
Anchoa mitchilli	Bay anchovy	332	239	3.8 (0.05)	1.1 (0.03)	402	124	3.7 (0.1)	1.2 (0.0)	452	210	3.8 (0.1)	1.3 (0.0)
Symphurus plagiusa	Black cheek tonguefish	–	–	–	–	–	–	–	–	2	2	6.1 (1.6)	2.5 (1.5)
Callinectes sapidus	Blue crab	45	–	–	–	12	–	–	–	10	–	–	–

Pomatomus saltatrix	Bluefish	–	–	–	–	1	1	29.9	226.8	–	–	–	–
Caranx hippos	Crevalle jack	–	–	–	–	1	1	10.5	13.1	–	–	–	–
Dorosoma cepedianum	Gizzard shad	65	61	18.4 (1.5)	173 (39)	47	47	28.6 (2.0)	413 (53)	107	80	16.6 (1.1)	110 (28)
Goby spp	Goby spp	–	–	–	–	–	–	–	–	1	1	13.0	11.3
Trinectes maculatus	Hogchoker	–	–	–	–	–	–	–	–	–	–	–	–
Elops saurus	Ladyfish	–	–	–	–	–	–	–	–	–	–	–	–
Mugil spp	Mullet spp	29	19	15.1 (0.5)	37.3 (2.9)	3	3	15.9 (3.8)	54.4 (32.9)	6	6	14.5 (1.2)	35.8 (7.7)
Fundulus heteroclitus	Mummichog	44	44	6.2 (0.2)	4.0 (0.3)	15	15	6.8 (0.2)	4.9 (0.5)	94	71	4.4 (0.1)	1.6 (0.2)
Gobiosoma bosc	Naked goby	–	–	–	–	2	2	2.9 (0.6)	1.0 (0.0)	–	–	–	–
Trachinotus falcatus	Permit	–	–	–	–	–	–	–	–	–	–	–	–
Sciaenops ocellatus	Red drum	1	1	30.8	368.5	6	6	36.4 (1.8)	612 (76)	4	4	31.5 (2.1)	312 (69)
Cyprinodon variegatus	Sheepshead minnow	1	1	5.0	1.7	–	–	–	–	1	1	4.1	2.1
Bairdiella chrysoura	Silver perch	57	57	8.6 (0.2)	7.7 (0.6)	74	56	8.3 (0.2)	6.6 (0.6)	52	40	8.3 (0.2)	7.0 (0.9)
Leiostomus xanthurus	Spot	12	12	15.2 (0.5)	60.0 (8.8)	1	1	14.8	42.3	22	22	15.3 (0.3)	57.5 (3.6)
Eucinostomus argenteus	Spotfin mojarra	–	–	–	–	5	5	6.8 (0.3)	6.5 (0.5)	–	–	–	–
Cynoscion nebulosus	Spotted sea trout	–	–	–	–	–	–	–	–	–	–	–	–
Anchoa hepsetus	Striped anchovy	2	2	5.9 (0.3)	1.2 (0.2)	3	3	6.6 (0.4)	1.8 (0.4)	6	6	6.3 (0.2)	1.3 (0.1)
Morone saxatilis	Striped bass	3	3	6.0 (2.0)	3.3 (1.5)	–	–	–	–	1	1	20.5	85
Fundulus majalis	Striped killifish	7	7	4.0 (0.3)	1.1 (0.1)	–	–	–	–	–	–	–	–
Paralichthys dentatus	Summer flounder	1	1	29.3	252.3	–	–	–	–	1	1	12.8	19.5
Lobotes surinamensis	Tripletail	–	–	–	–	–	–	–	–	–	–	–	–
Cynoscion regalis	Weakfish	–	–	–	–	–	–	–	–	–	–	–	–
Morone americana	White perch	6	6	11.1 (1.1)	21.0 (7.6)	13	13	9.1 (0.3)	9.0 (1.0)	1	1	9.0	8.7
All Species		1,119	698	6.6 (0.2)	20.2 (3.9)	927	475	8.4 (0.4)	53.2 (8.2)	2,115	719	7.2 (0.2)	20.0 (3.5)

Average length and weight are based on the subset (*n*) of field-measured specimens

Total abundance, *Ave TL* average total length (cm), *Ave Wt* average weight (g), *SE* standard error

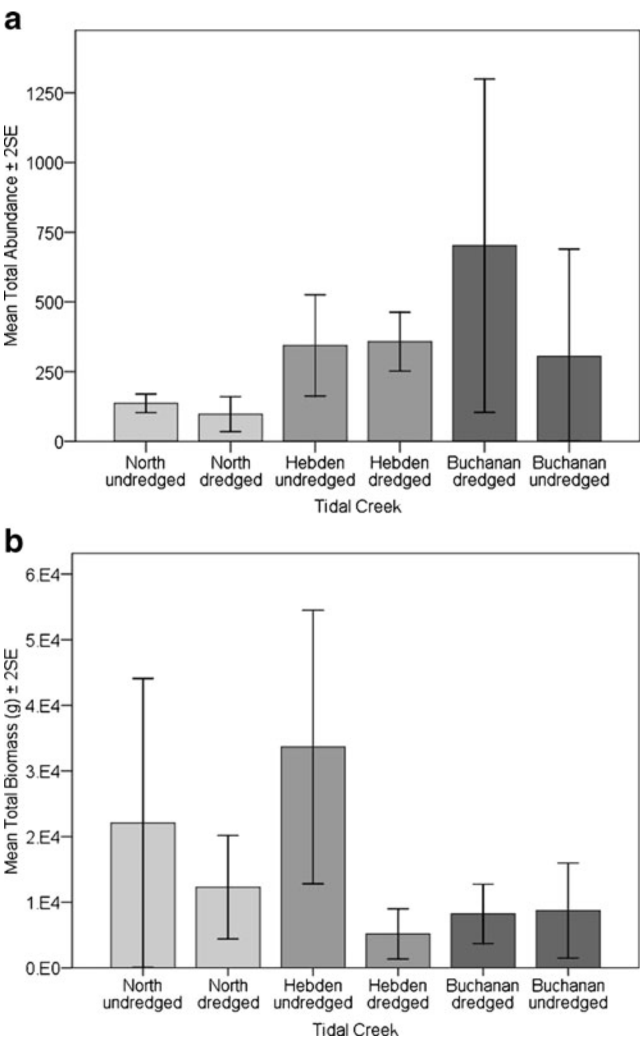


Fig. 3 Mean total **a** abundance and **b** biomass by tidal creek (±2 SE, representing 95% confidence interval). No significant difference in abundance or biomass among creeks was observed in ANOVA analyses

catch in BD and the smallest in newly dredged North creek (5.6%; Fig. 6d). No differences were observed in species biomass by months pooled across sites (ANOSIM, global R= 0.05, p=0.25) or by paired creek group (ANOSIM, global R= 0.12, p=0.07), although North and Buchanan creeks were near significance in pairwise ANOSIM (R=0.21, p=0.06). Cluster groupings for biomass were similar to patterns observed for species abundance, with the exception of HU where the highest biomass was collected (Fig. 5).

Fish Community Structure and Other Environmental Variables

The single variable that grouped the tidal creeks best, in a manner consistent with the fish patterns based on abundance, was turbidity (mean NTU; BIO-ENV, weighted Spearman rank correlation ρ=0.921; global R=3.4%; Table 3). None of

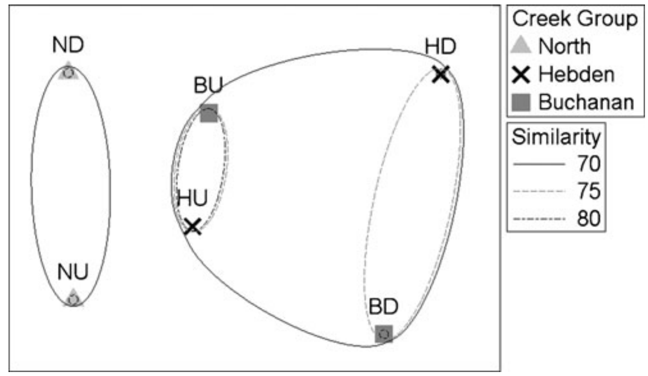


Fig. 4 MDS ordination of species average abundance with hierarchical clusters superimposed. Paired dredged and undredged creek groups are also superimposed: (1) North, (2) Hebden creeks, and (3) Buchanan creeks. North creeks (NU and ND) are distinct from Hebden and Buchanan creeks. Further clustering occurs between Hebden and Buchanan undredged versus dredged creeks. 2D stress=0.001

the excluded collinear factors were strongly correlated with turbidity. Cluster groupings of creeks based on turbidity were similar to patterns observed with species abundance (Fig. 7). Considering ordination by biomass, a combination of four factors delivered the best correlation but was not significant: organic matter percentage, mean minimum DO, total shoreline hardening, and turbidity (BIO-ENV, weighted Spearman rank correlation ρ=0.811; global R=18%; Table 3). None of the excluded collinear factors were strongly correlated with these four factors. Cluster groupings of creeks for the four environmental parameters were similar to patterns observed with species biomass (Fig. 8).

Discussion

Dredging activity appeared to have little effect on overall fish community structure as measured by mean community

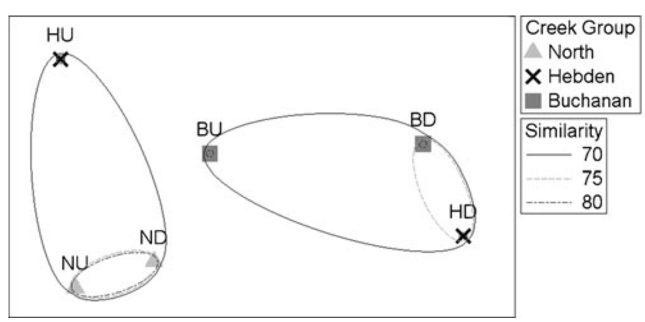


Fig. 5 MDS ordination of species average biomass with hierarchical clusters superimposed. Paired dredged and undredged creek groups are also superimposed: (1) North, (2) Hebden, and (3) Buchanan. ANOSIM analysis suggests that dredged creeks are distinct from undredged creeks. Cluster group are similar to patterns observed for species abundance, with the exception of Hebden undredged where the highest biomass was collected. 2D stress=0.001

abundance, biomass, and diversity; however, more subtle effects were observed from size structure and species-specific comparisons. At the community level, several factors may be at play, ameliorating or masking effects including estuarine variability, potential sampling resolution or frequency deficiencies, and the regular influx of migratory and transient species to the Lynnhaven River due in part to its location near the mouth of the Chesapeake Bay. Other factors promoting diversity are the presence of a variety of critical habitats, including mudflats, marsh, and shallow water. Numerous studies have documented the utilization of shallow-water habitats, tidal creeks, and marshes by nekton for nursery areas, including those observed in this study. For example, shallow water has been described as important nursery habitat for spot, silver perch, spotted sea trout, and Atlantic croaker in the Chesapeake Bay (Chao and Musick 1977). Tidal creeks and marshes (vegetated and nonvegetated) are reported nursery habitats for several species including spot, spotted sea trout, silver perch, pinfish, striped mullet, Atlantic

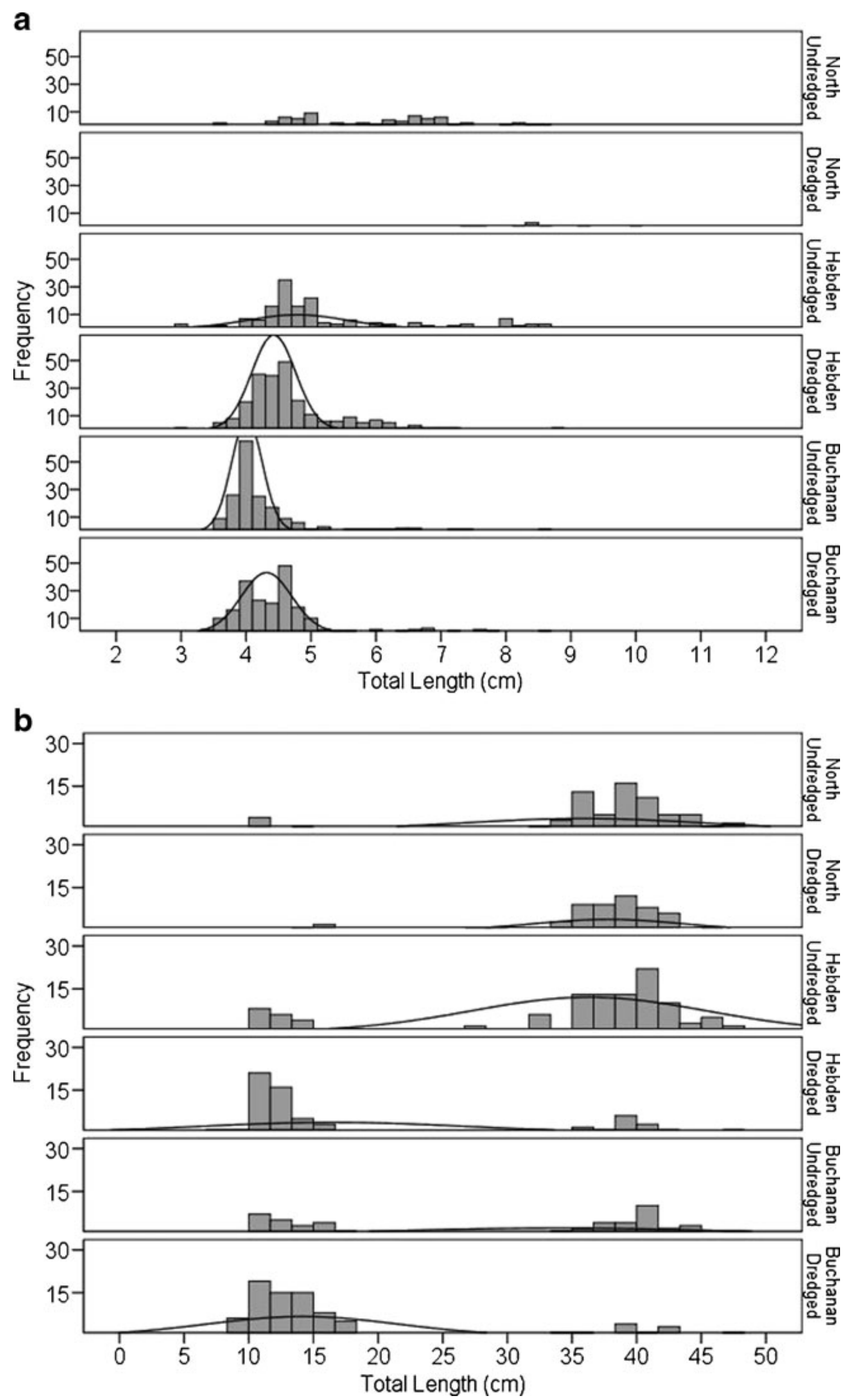
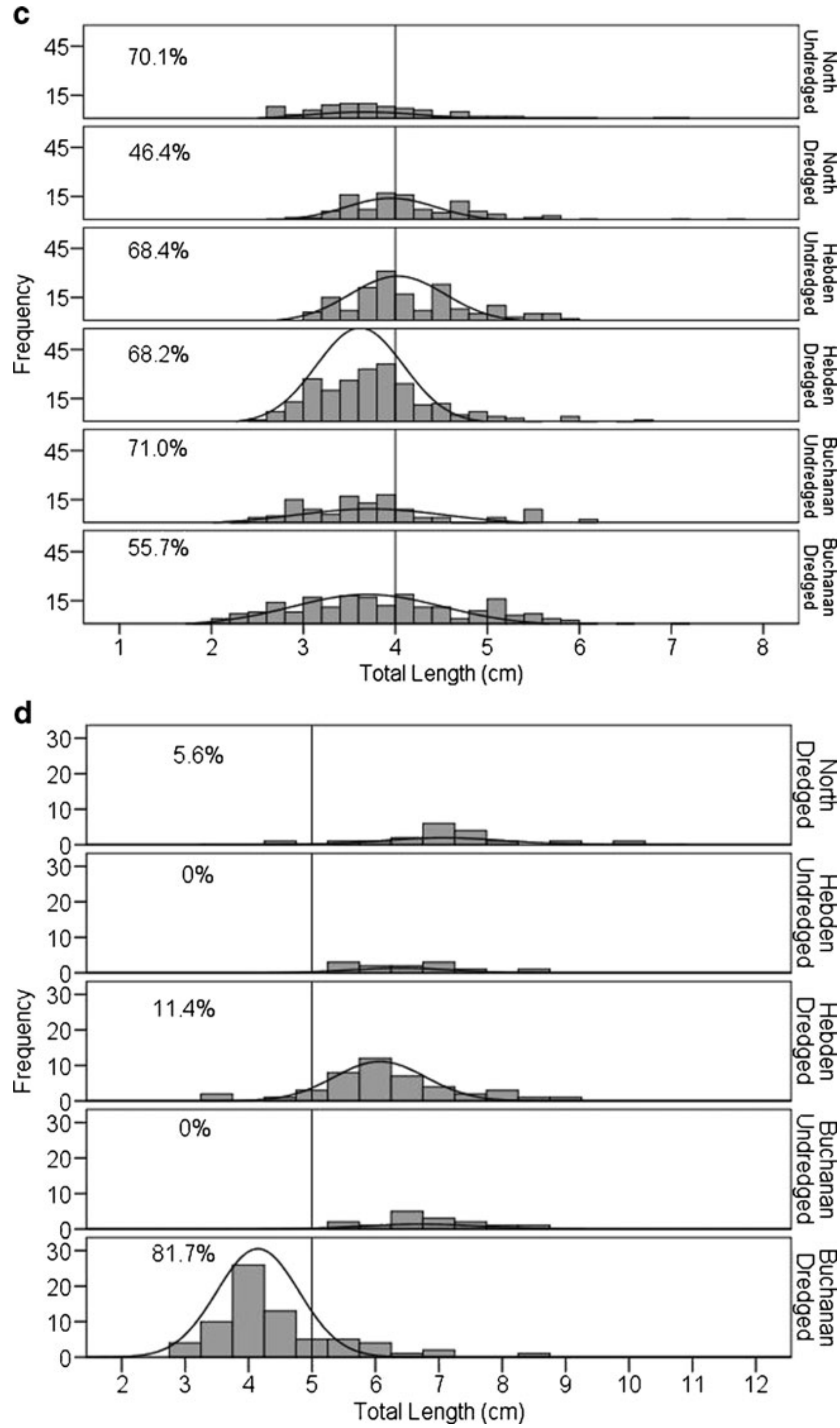


Fig. 6 Length frequency distribution by tidal creek for **a** Atlantic silverside, **b** gizzard shad, **c** bay anchovy, and **d** mummichog. Proportion of YOY by creek is indicated as a percentage placed on the left of the size demarcation. YOY Atlantic silverside (20–60 mm), which prefer shallow-water habitats, were only absent in the newly dredged creek (ND). After 60 mm, YOY begin to utilize deeper estuary waters and reduce movement in and out of marsh creeks. This may indicate a reduction in available nursery habitat for YOY ≤ 60 mm in the newly deepened creek ND. Large adult gizzard shad were primarily observed in NU, ND, and HU creeks, while YOY predominantly utilized BU, BU, and HD creeks. This pattern of habitat use influenced biomass differences noted in multivariate analysis. YOY bay anchovies were present in all creeks (<4 cm) with the lowest proportion found in the newly dredged creek (ND; 46.4%). By comparison, the adjacent NU creek supported 70% YOY. This suggests a reduction in available nursery habitat for YOY bay anchovy in the newly deepened creek. No mummichog were collected in North undredged creek. YOY (<5 cm total length) were evident in three creeks with the highest catch in BD (BD>> HD > ND). YOY mummichog observations suggest that spawning is occurring in these three creeks

Fig. 6 (continued)



menhaden, spotfin mojarra, red drum, and blue crab (Weinstein 1979; Weinstein and Brooks 1983; O'Neil and Weinstein 1987; Minello et al. 2003; King et al. 2005), and subtidal creek habitats may be critical to larval spot, gobies, bay anchovy, and Atlantic croaker (Allen and Barker 1990). Detrimental effects of dredging on individual species or life stages may be obscured in mean fish community measures by the high diversity and regular influx of species.

The minimal detected impacts of dredging activity may have been due to preventative measures imposed by the regulatory agency (VMRC) to minimize dredging impact on tidal creeks in the form of mandatory buffer widths for the preservation of wetlands, time-of-year restrictions (March 1–Sept 30) to protect fish and shellfish during critical spawning and nursery periods, and dredging depth limitations. Specifically, depth of dredging to these ancil-

Table 3 BIO-ENV results from square-root-transformed abundance and biomass data

Biotic metric	# var.	R	Variables
Abundance	1	0.921	Mean NTU
	4	0.889	% Silt/clay, mean min DO, mean min temp, mean NTU
	3	0.879	Mean min DO, mean min temp, mean NTU
	4	0.854	Mean depth, mean min DO, mean min temp, mean NTU
	2	0.850	Mean min temp, mean NTU
Biomass	4	0.811	Organic matter, mean min DO, shoreline hardening, mean NTU
	3	0.729	Organic matter, mean min DO, shoreline hardening
	4	0.700	Organic matter, mean min DO, mean min temp, shoreline hardening
	3	0.696	Organic matter, mean min DO, mean min temp
	4	0.675	Organic matter, *Phragmites* percent, shoreline hardening, mean NTU

Bray–Curtis similarity was used for biotic data and Euclidean distance for normalized abiotic data. Correlation (R) based on spearman rank coefficient. Combinations with maximal four factors are showed. Significance test was calculated based on 999 random permutations of sample names. Top five best results shown for each biotic metric

lary creeks was restricted to ≤1.5 m (5 ft) MLW with a minimum setback width of four times the channel depth to the edge of emergent wetlands (VIMS and VMRC permit database). Evidence of the effectiveness of this practice may be seen in the presence of YOY mummichog (<5 cm total length) in all three dredged creeks surveyed (Fig. 6d). This suggests that spawning is occurring in these creeks, since mummichog are marsh-dependent species with a reportedly small migratory range in the summer (<400 m) that exhibit strong site fidelity (Teo and Able 2003). The three creeks, while dredged, retain 55–94% fringe marsh (Table 1). Although YOY mummichog were absent from undredged creeks, sampling did not occur on the marsh surface and thus was inadequate to completely quantify spawning site utilization. Intensive sampling of fringe marshes is necessary to definitely preclude spawning in a creek; however, the presence of YOY is a reasonable indication of spawning occurrence. While the presence of YOY mummichog does indicate that spawning is occurring, it does not eliminate the possibility that the connection between marsh habitat and shallow water may be compromised or marsh inundation patterns may be altered by dredging with potential long-term implications. To more fully understand the effectiveness of preventative measures in preserving tidal creek ecosystem functions, additional research examining varying spatial and temporal patterns of marsh utilization by fish between dredged and undredged creeks is imperative.

The overall fish assemblage in the most recently dredged creek (dredged 6 months prior to sampling) resembled those in the adjacent undredged creek (North creeks), which may indicate a quick overall recovery rate for fish communities postdredging. Immediate adverse effects from dredging may include entrainment mortalities, behavioral changes, noise effects, and fish gill injury from exposure to high suspended sediment loads which are expected to be localized and

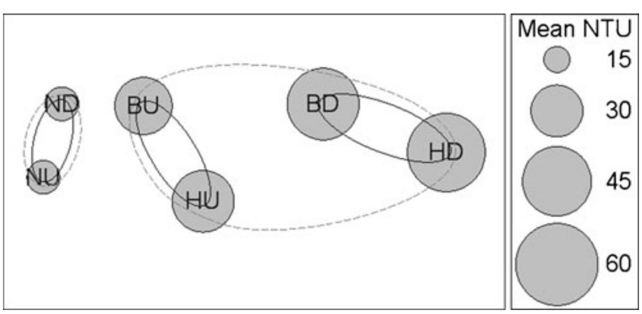


Fig. 7 MDS ordination of Euclidean distance similarities from creek turbidity data (2D stress=0.001), which were significantly associated with fish abundance patterns (BIO-ENV, ρ=0.81, global R=18%). Group-average clustering from Euclidean distances is superimposed. *Circles* of *increasing size* represent increasing turbidity (NTU). Relatively high fish abundance, particularly YOY or small prey species, was observed in creeks with relatively high turbidity (Hebden and Buchanan creeks)

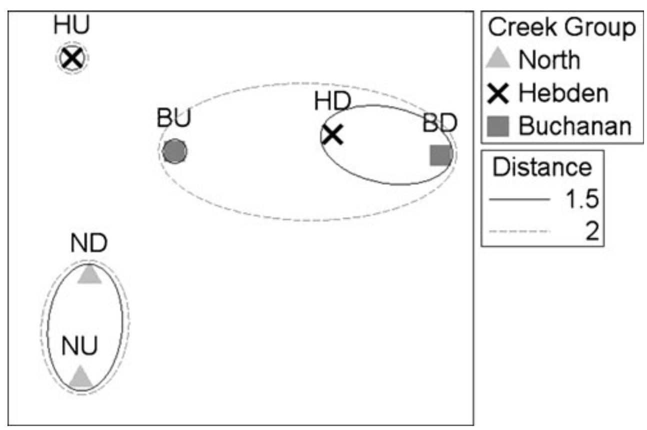


Fig. 8 MDS ordination of Euclidean distance similarities from environmental data (organic matter percent, mean minimum DO, total shoreline hardening, and turbidity; 2D stress=0.001), which were correlated with fish biomass patterns (BIO-ENV, ρ=0.81, global R= 18%). Group-average clustering from Euclidean distances is superimposed. Creeks with the highest biomass (NU, ND, HU) also had relatively low organic matter and high mean minimum DO. All creeks exhibited elevated levels of organic matter, suggesting a stressful environment for the benthos; therefore, relatively low levels associated with high fish biomass may indicate a relatively healthier benthic community as a source of prey. Turbidity was relatively low in North creeks and moderate in HU. Highest biomass and diversity was observed in HU, which was the only creek without hardened shoreline

temporary (Nightingale and Simenstad 2001). Nonetheless, the presence of fish does not mean that ecosystem alterations do not occur from dredging activities. Information on prey communities is necessary, as well as long-term empirical studies to fully evaluate ecosystem changes. Many studies assessing the negative effects of dredging on nearshore fish fauna have primarily focused on the effects of sediment disposal (e.g., Lindeman and Snyder 1999). In Lynnhaven River, dredged sediment from small tidal creeks is removed from the creek, minimizing burial impact, though sedimentation may occur due to the input of fine sediments postdredging over a longer time span. Similarly, long-term and cumulative effects on habitats and biota may occur that have yet to be measured. For example, sustained elevated turbidities could hinder primary productivity and larval feeding with negative implications to higher trophic levels (Wilber and Clarke 2001), and conversion of habitats (e.g. shallow intertidal–subtidal to deeper subtidal) could result in a shift in ecosystem dynamics with unknown cumulative effects. While instantaneous turbidity measures were not significantly different between dredged and undredged creeks in this study, over longer time span (months–years) trends in elevated and more variable turbidity may become apparent in dredged creeks as is intimated in Hebden and Buchanan paired creek comparisons (Table 1). Seasonal restrictions on dredge activities may minimize direct physical impact to fish species with sensitive early life stages in the estuary during the restricted period (often spring–summer when migration, peak spawning, and nursery use occurs). However, those species with early life stages in the estuary during dredge activities, such as Atlantic croaker and spot (winter spawners), may still experience direct losses from sedimentation effects, entrainment, smothering, and reduced feeding.

There is evidence that the recently dredged creek (ND) was providing less suitable nursery habitat for the young of year of abundant prey species, Atlantic silverside and bay anchovy. For example, Atlantic silverside were generally abundant in all creeks except for ND, and YOY (20–60 mm TL) were absent from this creek (Fig. 6a). Length frequency distributions also differed between the paired North creeks, indicating that Atlantic silverside size patterns were not solely due to the location of North creeks near the Lynnhaven inlet. When Atlantic silverside YOY reach 60–80 mm, they begin to move to deeper estuary waters and reduce diel movement in and out of marsh creeks (Rountree and Able 1992). The newly deepened North creek may have been providing less nursery habitat for small-sized YOY Atlantic silverside than the other creeks. However, HD supported a similar percentage of YOY Atlantic silverside as the paired HU; thus, depth may not be the sole factor dictating absence of YOY in the newly dredged creek. This may indicate an initial adverse response of these species life stages to dredging, with an eventual recovery in the absence of additional dredging activity.

Species biomass differences observed between dredged and undredged creeks were in part driven by high biomass of gizzard shad, red drum, and Atlantic menhaden (which make up 90% of the total biomass captured) in undredged creeks (Table 2). This difference may be influenced by the pattern of higher use by large adult gizzard shad of North and Hebden undredged creeks (NU, ND, and HU), two of which are undredged (Fig. 6b). Juvenile gizzard shad occupied creeks at a greater distance from the inlet compared to adults. However, red drum and Atlantic menhaden captured had a similar size distribution for dredged and undredged creeks and displayed higher utilization of undredged than that of dredged creeks. This suggests that undredged creeks may be associated with relatively high fish biomass for select species. Further research is needed to determine the exact mechanisms driving these patterns (e.g., enhanced prey availability compared to dredged creeks).

Species abundance differences were not observed between dredged and undredged creeks, though North creeks had lower abundance than Hebden and Buchanan creeks. Species-specific trends influencing abundance differences include the preferential use of Hebden and Buchanan creeks by Atlantic silverside, bay anchovy, juvenile silver perch, and blue crab as suggested by higher abundance in those creeks (Table 2). For example, 94% of Atlantic silverside, 84% of bay anchovy, and 91% of blue crabs captured were located in Hebden and Buchanan creeks. Additionally, the majority (72%; 61 of 85) of blue crabs was young of year (in the first year, typically early juvenile stages). This signifies that these creeks may be providing relatively important nursery habitat for small prey and juvenile species, irrespective of dredge status.

The single environmental parameter of watershed, shoreline, physicochemical, and sediment measures that highly influenced observed differences in species abundance among creeks was turbidity. Higher turbidity levels were observed within Hebden and Buchanan creeks than within North creeks, as was higher abundance, most notably, of Atlantic silverside, bay anchovy, silver perch, spot, blue crab, and red drum (Table 2). In this study, dredging-induced turbidity may not have increased the background turbidity level beyond environmentally acceptable limits for a sustained period, since significant differences between levels in dredged and undredged creeks were not observed. Instead, species distributions may be reflective of turbidity tolerances, particularly of juveniles which may not be as susceptible as adult predators to high turbidity effects (i.e., reduced foraging efficiency; Deegan et al. 2000). Subsequently, turbid waters may reduce predation risk by visual predators and confer an advantage to juvenile fish. Higher numbers of juvenile prey species in Hebden and Buchanan creeks in relation to North creeks may indicate a positive effect of turbidity in the form of enhanced nursery habitat quality for these species.

Environmental parameters did not significantly influence fish biomass patterns among creeks; however, a high correlation was observed for four variables: organic matter percent, mean minimum DO, total shoreline hardening, and mean turbidity. Creeks with the highest biomass (NU, ND, HU) also had relatively low organic matter and high mean minimum DO. North creeks had relatively low turbidity (~18 NTU) and high shoreline hardening unlike HU which had moderate turbidity (38.3 NTU) and no shoreline hardening (Table 1, Fig. 7). Hyland et al. (2005) found that extreme concentrations of total organic carbon (TOC) in sediments can have adverse effects on benthic communities. TOC levels below 0.5 mg/g (0.05%) and above 30 mg/g (3.0%) were related to reduced benthic abundance and biomass. In this study, TOC (assuming organic matter contains 58% organic C; Nelson and Sommers 1996) ranged from approximately 4% to 8%, surpassing the threshold for negative benthic impacts. The proximity to decomposing salt marsh plants and upland runoff, and possibly reduced flushing rates, most likely explains the high organic content in the tidal creeks. Elevated TOC suggests a stressful environment for the benthos. Dauer (2007) reported that the areal extent of degraded benthic habitat (benthic index of biotic integrity < 3.0) was extremely high in the Lynnhaven River (81.1±5.8%) compared to all of Virginia tidal waters (59.0±9.6%). Relatively low organic matter and high mean minimum DO evident in NU, ND, and HU could be indicative of a more productive and healthier benthic community compared to the other surveyed creeks, which may be supporting the higher biomass of fish observed. This also suggests that undredged creeks support healthier benthic communities than dredged creeks, as reflected in relatively low organic matter, mean minimum DO, and high fish biomass in two undredged creeks (NU, HU). However, the pattern is not unequivocal since ND and BU exhibit conflicting trends. Without additional comparative examination of benthic community structure, the influence of dredging on the benthos cannot be definitively stated.

While shoreline hardening may be affecting biomass trends, the limited number of tidal creeks surveyed along with varying watershed and shoreline conditions (i.e., most creeks had high residential development and high levels of shoreline armoring) prevented the robust examination of a gradient of development stressors influencing fish community structure. For example, thresholds of impervious surface area have been reported between 10% and 20% for fish responses in streams (Paul and Meyer 2001), and in this study, only the Buchanan creek watersheds had in excess of 10% impervious surface hindering the establishment of an unequivocal relationship with fish assemblages. Lack of predevelopment fish community data also prevents definitive determination of community shifts in relation to cumulative watershed and shoreline alterations. However, relative comparisons of current assemblages may help target promising habitats and regions for future restoration and conservation efforts. The only creek without shoreline armoring (HU) possessed the highest fish biomass and diversity of all creeks surveyed, supporting previous research on the importance of land–water linkages to shallow-water communities (Wang et al. 1997; Paul and Meyer 2001; Seitz et al. 2006; Bilkovic and Roggero 2008). In this creek, the preservation of marshes, as well as the connection between marsh habitat and shallow water may be contributing to the maintenance of a diverse fish community and perhaps subsidizing adjacent highly developed areas.

Limited quantitative historical information on fish communities in Lynnhaven River exists. However, Schauss, Jr. (1977), reported similar levels of species diversity as this study (31 species observed in February 1973–January 1974 beach seine and plankton collections) and that it was an important nursery ground for several species including bay anchovy, spot, white mullet, *Gobiosoma* spp., and green goby. While exact comparisons of abundance cannot be conducted due to varying sampling effort and some gear differences, general trends can be informative. In this study, several species were prevalent that were absent or in low abundance in the historic survey including Atlantic menhaden, gizzard shad, white perch, and silver perch. In contrast, in 1973, sheepshead minnow, spotfin mojarra, striped killifish, naked goby, and black cheek tonguefish, which have established associations with structural habitat (i.e., marsh, oyster reefs; Weinstein and Brooks 1983; Rountree and Able 1992; Able and Fahey 1998; Breitburg 1999; Layman and Smith 2001), were more prevalent in surveys than presently observed. This may indicate that a shift in fish community structure has occurred, possibly due to further reduction in marsh and oyster reef habitats in the intervening years.

Remaining intertidal and shallow-water habitats (e.g., marsh) in Lynnhaven Bay are increasingly vulnerable to a combination of anthropogenic (dredging, shoreline armoring, land conversion) and natural (sea level rise) stressors. Although tidal creeks surveyed (including dredged creeks) currently retain narrow fringing marshes, these were often found in conjunction with shoreline hardening and thus are threatened as sea level rise rates increase in the Chesapeake Bay (Pyke et al. 2008). Of the creeks with currently hardened shoreline, between 6% and 38% of fringe marsh was associated with shoreline structures which will prevent marsh transgression. These trends were prevalent over the entire Lynnhaven River (Eastern and Western Branches), with 29% of the inventoried shoreline (181.2 km) armored and an estimated average annual rate of hardening of 0.32% (based permit activity from 1993 to 2007, VIMS and VMRC Permit Database). The presence of fringe marshes may be an essential ecoscape element in the estuary, moderating development pressures by providing habitat corridors for juvenile fishes and crabs. As sea level rises, shallow water and intertidal habitats will likely become limited or fragmented. Further loss of the remaining habitat from dredging and shoreline alterations may surpass ecological thresholds, leading to irreversible productivity losses (Limburg and Schmidt 1990; Wang et al. 1997; Paul and Meyer 2001; Bilkovic et al. 2006; Bilkovic and Roggero 2008). When faced with development decisions, coastal managers should take into consideration not only preservation of current shallow-water habitats but also expected redistribution of these habitats in the future from climate change pressures. Allowance for landward transgression of these habitats as sea level rises will be a necessary component of coastal management plans if tidal creek nursery habitat functions are to be preserved.

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