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Growth and Movements of Mummichogs (*Fundulus heteroclitus*) Along Armored and Vegetated Estuarine Shorelines

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Abstract Alteration of estuarine shorelines associated with increased urbanization can significantly impact biota and food webs. This study determined the impact of shoreline alteration on growth and movement of the estuarine fish *Fundulus heteroclitus* in a tributary of the Delaware Coastal Bays. *Fundulus heteroclitus* is abundant along the east coast of the USA, and is an important trophic link between marsh and subtidal estuary. The restricted home range of *F. heteroclitus* allowed discrete sampling, and fish growth comparisons, along 35–65-m long stretches of fringing *Spartina alterniflora* and *Phragmites australis* marsh, riprap, and bulkhead. *Fundulus heteroclitus* were tagged with decimal Coded Wire Tags. Of 725 tagged *F. heteroclitus*, 89 were recaptured 30–63 days later. Mean growth rate (0.06–0.15 mm day^{-1} across all shoreline types) was greatest at riprap, lowest at *Spartina* and *Phragmites*, and intermediate at bulkhead, where growth was not significantly different from any other shoreline. This suggests that discernible environments exist along different shoreline types, even at the scale of tens of meters. No difference in movement distance was detected at different shoreline types; most individuals displayed a high degree of site fidelity. Forty-seven percent were recaptured within 5 m of their tagging location, although alongshore movements up to 475 m were recorded. Estimates of relative *F. heteroclitus* productivity, using relative density data from a concurrent study, were highest along *Spartina* and *Phragmites*, intermediate at riprap, and lowest at bulkhead. Therefore, despite greater growth rates along riprap than at vegetated shores, armoring reduces abundance sufficiently to negatively impact localized productivity of *F. heteroclitus*.

Keywords *Spartina* · *Phragmites* · Riprap · Bulkhead · Mark-recapture · Decimal coded wire tag

Introduction

Since the mid-1900s, more than 70% of the estuaries in the USA have been at least moderately degraded (Mercer 1984; Peterson et al. 2000). Degradation in many estuaries can be attributed to urbanization and shoreline development (Able et al. 1999). These processes usually occur in concert, as shoreline development is implemented to protect upland development. Shoreline armoring, including bulkhead and riprap structures, can lead to modification of hydrology (Odum 1970), increases in nutrient loads (Bilkovic and Roggero 2008), decreases in allochthonous inputs (Jennings et al. 1999; Christensen et al. 1996; Bilkovic and Roggero 2008), alteration of shallow water habitat (Jennings et al. 1999; Peterson et al. 2000; Bilkovic et al. 2006; Bilkovic and Roggero 2008), and reduction in habitat complexity (Jennings et al. 1999; Scheuerell and Schindler 2004; Bilkovic and Roggero 2008).

Alterations to the physical habitat associated with armoring shorelines can impact the associated biological communities. A study along the James River in Virginia, USA, concluded that bulkheading was associated with a significantly lower Fish Community Index (a multimetric measurement of biotic

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integrity) than either natural marsh or riprap shorelines (Bilkovic and Roggero 2008). Able et al. (1998) found that fishes were less abundant and less diverse under piers in the New York-New Jersey Harbor Estuary than in nearby natural marsh habitat or uncovered pile fields. A subsequent study showed that winter flounder (*Pseudopleuronectes americanus*) and tautog (*Tautoga onitis*) confined under piers had lower growth rates (often negative) compared to fish confined in natural marsh and uncovered pile fields (Able et al. 1999). Other studies have demonstrated impacts of shoreline modification (bulkhead, riprap, riprap-sill, rock-crib pier, retaining wall) on various biota, including the following: benthic invertebrate communities (Bilkovic et al. 2006; Seitz et al. 2006), fish eggs (Rice 2006; Balouskus and Targett 2012), predator abundances (Seitz et al. 2006), and fish community structure (Beauchamp et al. 1994; Jennings et al. 1999; Bilkovic and Roggero 2008; Balouskus and Targett 2016). However, no research has been conducted on the effects of bulkhead or riprap structures on growth and movement of the abundant estuarine resident fish species, the mummichog (*Fundulus heteroclitus*).

Fundulus heteroclitus spends its entire life cycle in shallow estuarine waters, and it is the most abundant fish in tidal marshes on the east coast of the USA (Kneib 1997; Able and Hagan 2000, 2003; Able et al. 2003; Hagan et al. 2007) where it reaches a maximum size of approximately 130 mm total length (TL) (Able and Fahay 2010). This species uses salt marsh edge and surface habitat as refuge from predation (Kneib 1987; Hagan et al. 2007), feeding areas (Weisberg and Lotrich 1982; Hagan et al. 2007), spawning sites (Able and Castagna 1975; Taylor et al. 1977; Hagan et al. 2007), and juvenile fish habitat (Kneib 1984; Talbot and Able 1984; Hagan et al. 2007). Due to its reliance on marsh habitat, *F. heteroclitus* populations are likely impacted by shoreline hardening that alters the marsh edge and can eliminate access to the marsh surface. This expectation is supported by research into the impact of marsh alteration on *F. heteroclitus* by invasive plants (Hagan et al. 2007).

Although *F. heteroclitus* utilizes native smooth cordgrass (*Spartina alterniflora*; hereafter *Spartina*) marsh, the invasive common reed (*Phragmites australis*; hereafter *Phragmites*) has been shown to negatively affect *F. heteroclitus* populations. Early juvenile *F. heteroclitus* remain in marsh pools at low tide and little or no standing water remains in *Phragmites* marsh after tidal flooding has left the marsh surface (Able et al. 2003; Hagan et al. 2007). Numerous studies have found the abundance of juvenile *F. heteroclitus* to be significantly lower in *Phragmites* marsh than in naturally occurring *Spartina* marsh (Able and Hagan 2000, 2003; Osgood et al. 2003; Raichel et al. 2003; Hunter et al. 2006; Osgood et al. 2006; Hagan et al. 2007). In addition, lower growth rates of *F. heteroclitus* have been reported in *Phragmites* compared to *Spartina* marshes (Hagan et al. 2007).

Any impact of shoreline hardening or *Phragmites* invasion on *F. heteroclitus* growth or movement could have cascading effects on local estuarine environments. *Fundulus heteroclitus* are a major component of estuarine nekton (Kneib 1997) and a common prey for larger estuarine species. *Fundulus heteroclitus* has been shown to be prey for other fishes, such as striped bass (*Morone saxatilis*) (Tupper and Able 2000; Nemerson and Able 2003), weakfish (*Cynoscion regalis*) (Nemerson 2001; Nemerson and Able 2004), white perch (*Morone americana*) (Nemerson and Able 2004), and Atlantic croaker (*Micropogonias undulatus*) (Nemerson 2001), and is therefore an important trophic link between production on the marsh surface and subtidal estuarine habitats (Kneib 1997).

Individual *F. heteroclitus* exhibit a high degree of site fidelity and small feeding range (Lotrich 1975; Able et al. 2006), so their growth rate should reflect the local environmental conditions. This allows comparisons of growth, productivity, and movement along specific vegetated and armored shorelines. Alongshore movement of *F. heteroclitus*, based on tagging studies, has been reported to be 18 m or less over the course of a month (Lotrich 1975) and generally 30 m or less (but up to 300 m) over a 166-day recapture period (Able et al. 2006). This restricted home range, along with the observation of differing stable isotopes (δ^{13}C) between *F. heteroclitius* in nearby *Spartina* and *Phragmites* marshes (Currin et al. 2003), suggests that *F. heteroclitus* feeding and growth reflect localized habitat conditions. Furthermore, the opportunistic diet of *F. heteroclitus* (Kneib 1986) is largely determined by the available community of small benthic invertebrates (Baker-Dittus 1978), so *F. heteroclitus* growth and productivity can also serve as a reflection of benthic habitat quality (e.g., Goto and Wallace 2011) along localized shoreline types.

It is not known how *F. heteroclitus* growth, productivity, and movement compare among vegetated and armored shoreline types, nor is there information on movement between shoreline types. The objectives of this study were to (1) compare growth rates of *F. heteroclitus* along *Spartina*, *Phragmites*, riprap, and bulkhead shorelines, (2) estimate relative productivity of *F. heteroclitus* along these same shorelines, and (3) compare movement distance and patterns within and among shoreline types.

Methods

Study Area

This research was conducted in Pepper Creek, a tributary of Indian River Bay, in the Delaware Coastal Bays (Fig. 1). The Delaware Coastal Bays comprise a coastal lagoon estuary, with limited water exchange with the coastal ocean at Indian

Fig. 1 (**a**) Delaware Coastal Bay region of the USA. (**b**) The Delaware Coastal Bays, comprised of Rehoboth and Indian River Bays. (**c**) Pepper Creek study location off Indian River Bay. Dark circles indicate study sites, labeled with site ID and shoreline lengths in meters. *SP Spartina alterniflora*, *PH Phragmites australis*, *RR* riprap, *BU* bulkhead; numbers refer to replicates. Dark triangles show locations of multiparameter sondes
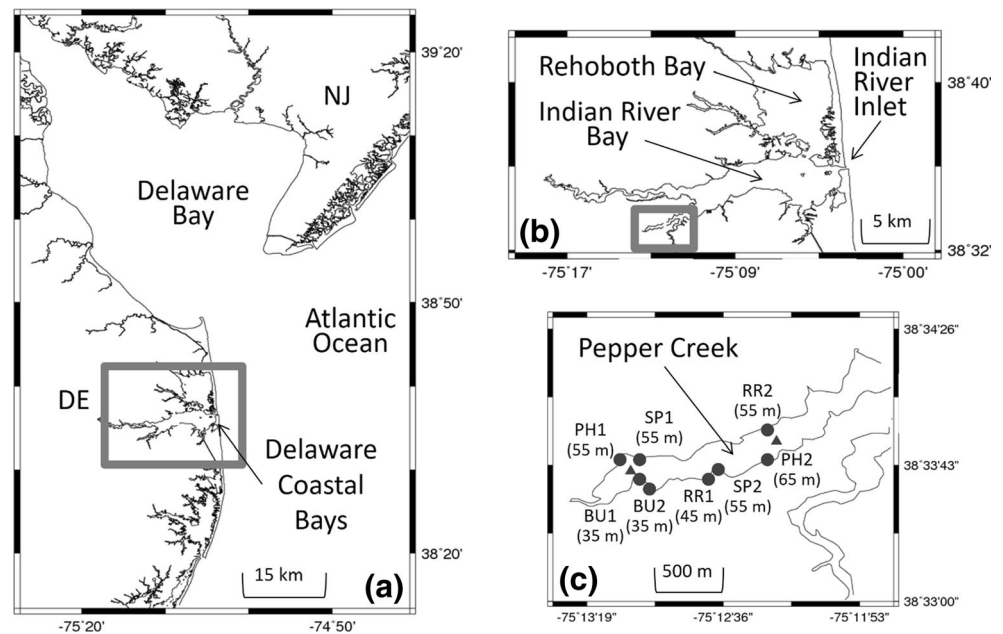
River Inlet (Tyler et al. 2009). Pepper Creek is 5-km long and varies in width from 0.75 km at the mouth to 0.2 km in the upstream portions. It has a tidal range of 0.75 m and a mean low water depth of 1.5 m in the channel and less than 0.25 m on the shoals. Summertime temperature and salinity typically range from 20 to 28 °C and 15 to 25, respectively (Tyler and Targett 2007; Tyler et al. 2009). The present study was conducted in the upper portions of the creek (Fig. 1c), where turbidity is high and sediments are primarily soft mud (Tyler and Targett 2007; Tyler et al. 2009).

Fundulus heteroclitus were sampled along four shoreline types: *Spartina alterniflora* marsh, *Phragmites australis* marsh, riprap, and bulkhead. *Spartina* and *Phragmites* sites were fringing marshes (less than 5-m wide) typical of Pepper Creek marsh shores. Two replicate sites of at least 35-m length were used at each shoreline type (Table 1; Fig. 1c). Site names used hereafter consist of shoreline type (SP for *Spartina* marsh, PH for *Phragmites* marsh, RR for riprap, or BU for bulkhead) paired with a replicate number (1 or 2). Shoreline sites were selected as close as possible to one another, within the constraints of availability, and all eight sites were within a 1.25-km stretch of Pepper Creek. Site proximity minimized the effect of environmental differences among sites, and allowed assessment of potential movement between shoreline types, but focused applicability of the results to fringing marshes.

Water temperature and dissolved oxygen (DO) were measured every 15 min by 2 YSI multiparameter sondes from July 7, 2010 to September 10, 2010. Sondes were located at the upstream and downstream edges of the study area   (Fig. 1c). Sondes were cleaned weekly to prevent biofouling and replaced every 3 weeks. Due to the sonde rotation schedule,

and occasional sonde biofouling, data were incomplete for 5 and 10 days at the upper and lower sonde, respectively. Incomplete records were rarely on adjacent days.

Tagging

Fundulus heteroclitus were tagged with sequential decimal Coded Wire Tags (CWTs, Northwest Marine Technology, Inc.) for individual fish identification, which allowed growth rates to be determined by mark-recapture. Numerous recent mark-recapture studies have successfully used CWT with *F. heteroclitus* and other fishes (e.g., Teo and Able 2003; Able et al. 2006; Brennan et al. 2007; Kneib 2009). The 1.1-mm long tags have been shown to have minimal effects on survival and behavior of *F. heteroclitus* larger than 30 mm TL, and retention rates are as high as 99% (Able et al. 2006). Prior to tagging, CWTs were examined under ×375 magnification and each tag was stored individually with its number for quick access in the field.

Tagging occurred on 5 days between July 6 and July 20, 2011 (Table 1). *Fundulus heteroclitus* were collected at two locations 15 m apart (Fig. 2) near the center of each site using 5 unbaited cylindrical minnow traps per location. Traps were set on flooding tide for at least 1 h, with trap opening initially just below the water surface to ensure similar depth of sampling at all sites. On shore, collected fish (40–70 mm TL) were weighed (±0.01 g) on a digital balance, measured (±0.1 mm TL) with calipers, tagged, and then released back at the collection location. Tags were inserted by syringe on the left side slightly dorsal and anterior to the anal fin. Ninety *F. heteroclitus* (45 at each of the two locations) were tagged per site (Fig. 1c), except at PH1 where 95 fish were tagged, for

Table 1 Shoreline types, lengths, and schedule of tagging and recapture for each replicate site. All dates are from 2011

Site	Shoreline type	Shoreline length (m)	Tagging dates	Recapture dates
SP1	*Spartina*	55	July 6, 11, 12	Aug 13, Sept 9
SP2	*Spartina*	55	July 6, 20	Aug 19, Sept 10
PH1	*Phragmites*	55	July 6, 11	Aug 10, Sept 7
PH2	*Phragmites*	65	July 6, 15	Aug 17, Sept 10
RR1	Riprap	45	July 6, 12	Aug 13, Sept 9
RR2	Riprap	55	July 6, 20	Aug 19, Sept 2, 20
BU1	Bulkhead	35	July 6, 11	Aug 10, Sept 7
BU2	Bulkhead	35	July 6, 20	Aug 17, Sept 2, 20

a total of 725 tagged fish. On the first day of tagging, 1 fish was weighed 10 times to assess the precision of the weighing procedure. Maximum and minimum weight measurements differed by 3.3%.

Recapture

A minimum of two rounds of recaptures were conducted at all sites between 1 and 1.5 months after tagging (Table 1). For the first round, two sites were sampled each day in the same order as tagging was done (to keep growing days similar for fish at each shoreline type). Sites RR2 and BU2 had lower numbers of recaptures during the first round, so those sites were sampled first in round two to maximize the possibility of recapturing fish; otherwise rounds one and two occurred in the same site-order. Sites RR2 and BU2 were sampled a third time because of low recapture numbers in the first two rounds (Table 1).

During recapture sampling, pairs of minnow traps were deployed at 5-m intervals, on either side of both tagging locations (Fig. 2). Trapping procedure was identical to the original collection protocol. Recaptured fish were identified using a V-detector (Northwest Marine Technology, Inc.) which detected the presence of a tag. These fish were reweighed (±0.01 g) and remeasured (±0.1 mm) at the same location used for initial tagging. They were then anesthetized with MS-

222, frozen on dry ice, and returned to the laboratory for tag extraction, number identification, and sex determination.

Extracted tags were cleaned with ethanol and identified under ×375 magnification. Sex of fish was determined based primarily on external characteristics (Murdy et al. 1997; Lopez et al. 2010). Female *F. heroclitus* have an ovipositor along the first spine of the anal fin and are more drab in color than males. Five fish were dissected and examined internally, which confirmed sex based on external characteristics.

Data Analysis

Temperature and Dissolved Oxygen

Temperature measurements from the two multiparameter sondes were used to determine the maximum, minimum, and mean temperature at each sonde location. Mean tributary temperature was calculated for each 15-min point as the average of the two values and correlations between individual sonde temperatures and mean temperatures were used to estimate means for times when only one sonde was active. Mean tributary temperature could thus be calculated over any time period, allowing temperature experienced by fish over different periods to be considered when comparing shoreline-specific growth rate results.

Fig. 2 Distribution of minnow traps for tagging and recapture of *Fundulus heteroclitus*. Fish were tagged at the 2 circled locations and recaptured at each vertical line. The illustration is for a 55-m section of shoreline. Recapture locations were added or subtracted as necessary, for longer or shorter shorelines, by adding or eliminating outer traps

Growth Rate

For each recaptured fish, linear growth rate (mm day^{-1}) and specific growth rate (% body mass day^{-1}) were calculated according to Ricker (1975) as:

$$\text{Linear growth rate} = \frac{L_f - L_i}{\Delta t}$$

$$\text{Specific growth rate} = (e^G - 1) \cdot 100$$

$$G = \text{Instantaneous growth rate} = \frac{ln(W_f) - ln(W_i)}{\Delta t}$$

where L_f is final length and L_i is initial length, W_f is final weight and W_i is initial weight, and Δt is time in days between final and initial measurements.

Fish recaptured at the same site were assumed to have not moved from that shoreline type. Our goal was to analyze these "non-mover" *F. heteroclitus* for the effects of 3 independent variables (shoreline type, replicate sites at each shoreline type, and sex) on 3 dependent variables (linear growth rate (LGR), specific growth rate (SGR), and movement distance). Site BU2 was excluded from the analyses because only one fish was recaptured there; results were similar with and without BU2. Growth data required pooling and transformation to be properly analyzed. Number of recaptures precluded analyzing the growth data by either partially nested ANOVA (replicate sites and sex nested within shoreline) or separate nested

ANOVAs (replicate sites nested within shoreline) for each sex. In such analyses, a third of the groups would have contained 3 or fewer samples and half would have contained 4 or fewer samples, so tests were run to see if data could be pooled by site or by sex. The appropriateness of pooling replicate sites was tested with nested ANOVAs (sites nested within shoreline) and pooling by sex of fish was tested with 2-way ANOVAs (shoreline + sex) with Tukey multiple comparisons (Table 2). Specific growth rate data were natural log transformed (referred to as tSGR) because of heteroscedasticity. Based on a cutoff at a critical value of 0.25 for pooling (Underwood 1997; Beninger et al. 2012), growth rate data could be pooled by replicate sites but not by sex. Initial length data did not meet the pooling criteria by replicate sites; however, because initial length was being considered as a confounding factor for the growth rate analyses, pooling by site was necessary.

With the replicate sites data pooled, growth rate was analyzed with 2-way ANOVAs, and Tukey multiple comparisons, to examine effects of shoreline and sex on LGR and tSGR (Table 2). Initial length data exhibited heteroscedasticity that could not be remedied with data transformations, so initial length data were analyzed with a 1-way ANOVA with a Welch's test (by shoreline, for which data were heteroscedastic) and Games-Howell multiple comparisons, and with a 1-way ANOVA (by sex, for which data was not heteroscedastic) and Tukey multiple comparisons. Welch's test and Games-Howell multiple comparisons are more appropriate for analysis of heteroscedastic data than an ANOVA

Table 2 Summary statistics for growth rate and length data analyses

Test	Factor (*df*)	Type III SS			*p* value, *F* statistic		
		LGR	tSGR	IL[b]	LGR	tSGR	IL[b]
Nested ANOVA	Shoreline (3)	0.095	0.829	378.481	0.11, 5.58	0.04, 66.63	0.45, 1.18
	Error (1.4–2.9)	0.016	0.006	308.239			
	Site[a] (3)	0.017	0.017	312.789	0.42, 0.96	0.94, 0.13	0.1, 2.2
	Error (73)	0.433	3.121	3459.07			
2-way ANOVA	Shoreline (3)	0.095	0.917	–	0.002, 5.40	<0.001, 7.47	–
	Sex (1)	0.027	0.052	–	0.03, 4.66	0.26, 1.28	–
	Interaction (3)	0.001	0.108	–	0.99, 0.03	0.45, 0.88	–
	Error (72)	0.420	2.946				
1-way ANOVA (Welch)	Shoreline (3, 39.412)						0.08, 2.47[c]
1-way ANOVA	Among sex (1)			46.043			0.34, 0.91
	Within sex (78)			3966.19			

LGR linear growth rate, *tSGR* natural log transformed specific growth rate (see Methods section), and *IL* initial length. Nested ANOVA justified pooling sites; 2-way ANOVA showed sex could not be pooled. Two 1-way ANOVAs (one with a Welch test) were used for initial length data because of heteroscedasticity (see Methods section)

[a] Site is nested within shoreline

[b] IL 2-way ANOVA could not be done due to heteroscedasticity

[c] Welch statistic

with Tukey multiple comparisons. ANOVAs were performed in SPSS and a critical value of 0.05 was considered significant for all tests.

Shoreline-Specific Relative Productivity

Relative productivity of *F. heteroclitus* along each shoreline type was estimated by multiplying relative shoreline-specific density by relative shoreline-specific instantaneous growth rate as: $\dot{P} = \dot{D} \cdot \dot{G}$; where $\dot{P}$ is relative productivity, $\dot{D}$ is relative density, and $\dot{G}$ is relative instantaneous growth rate. Shoreline-specific density of *F. heteroclitus* ($\# \text{ m}^{-2}$) data were available on 11 dates, between July 7 and September 15, 2010, from a concurrent seining study (Balouskus 2012; Balouskus and Targett 2017) in the same section of upper Pepper Creek. Density was considered a reasonable proxy for biomass because the average length of *F. heteroclitus* from each shoreline in the concurrent study was similar: 51, 51, 54, and 57 mm standard length (SL) at *Spartina*, *Phragmites*, riprap, and bulkhead, respectively. To establish relative productivity, growth rate, and density, values from each shoreline were normalized to the same base. Any shoreline could serve as this base and we used riprap, for which values were set at 1.

Movement Distance

Placement of traps at 5-m intervals during recapture sampling meant that fish at longer shoreline sites (Table 1; Fig. 1c) could move farther from their tagging locations and still be recaptured at that site/shoreline type. To make the movement distance metric comparable across all sites, only the fish recaptured at the 8 recapture locations nearest their site's tagging locations were considered in the within-site movement distance analysis. Movement distances within sites were categorized into recaptures ≤ 5 m from the tagging site and recaptures > 5 m from the tagging site. Fish counts in each category were compared among shoreline types with chi-square tests of independence for male and female fish. To determine if movement behavior was related to fish size, a Welch's *t* test was run on initial length of those fish that moved from the shoreline at which they were tagged vs all those that did not, for males and females.

Results

Overall, 89 (12%) of the 725 tagged *F. heteroclitus* were recaptured; 81 of which (91%) were recaptured at the same shoreline site at which they had been tagged (Table 3). Temperatures at the upstream and downstream sondes were similar; mean, maximum, and minimum temperatures were 28.8, 33.7, and 23.9 °C, respectively, at the upper sonde and

28.3, 33.9, and 23.3 °C, respectively, at the lower. Average temperature experienced by fish tagged on the first tagging day and recaptured on the first recapture day (29.5 °C ± 0.02 SE) was similar to that experienced by fish tagged on the last day of tagging and recaptured on the last day (28.0 °C ± 0.03 SE). Pepper Creek undergoes diel cycles of dissolved oxygen during summer (Tyler et al. 2009), and dissolved oxygen (DO) levels were similar at the two sondes. Average daily mean, maximum, and minimum DO was 5.29, 9.98, and 1.55 mg O_2 L^{-1}, respectively, at the upper sonde and 5.60, 10.66, and 2.41 mg O_2 L^{-1}, respectively, at the lower sonde. Minor spatial and temporal differences in temperature and DO conditions were considered negligible as potentially confounding the shoreline-specific growth rate results for *F. heteroclitus* within the study area.

Growth Rate

Fundulus heteroclitus recaptured at the same site where they had been tagged were assumed to have growth rates reflecting growth along that shoreline type. Data on 80 fish that were recaptured at their tagging site, excluding BU2, were available for analysis of specific growth rate (Table 3). After pooling growth rate data by replicate sites, 2-way ANOVA tests revealed that LGR ($F_{3,72} = 5.404$, $p = 0.002$) and tSGR ($F_{3,72} = 7.467$, $p < 0.001$) both differed significantly by shoreline type (Table 2). *Fundulus heteroclitus* along vegetated shorelines (*Spartina* and *Phragmites*) grew slower than those along riprap (0.15 mm day^{-1} and 0.65% body mass day^{-1}, Table 3), and fish from bulkhead grew at an intermediate rate that was not significantly different from that at other shoreline types (Tables 2 and 3, Fig. 3). Female *F. heteroclitus* had significantly greater linear growth rates, but not greater specific growth rates, than males along all shorelines (Fig. 3).

Initial length of recaptured *F. heteroclitus* did not differ significantly between shoreline types or between sexes (Table 2), so size did not have a confounding effect on any growth rate tests. The difference between mean initial length at the shoreline with the longest fish (bulkhead) and the shortest fish (riprap) was only 5.5 mm; and male and female fish differed by only 1.5 mm.

Shoreline-Specific Relative Productivity

Relative productivity of *F. heteroclitus* was highest along vegetated shorelines despite having lower growth rates (Table 4). Productivity of *F. heteroclitus* was greatest along *Phragmites* (114% greater than along riprap), due to a mean growth rate only 27% of that at riprap yet a nearly eight-fold greater mean density (Table 4). Fish along *Spartina* shoreline had the second highest relative productivity (97% greater than at riprap), with *F. heteroclitus* along *Spartina* having the lowest growth rate but occurring at the highest mean density (Table 4).

Table 3 Summary of mean initial lengths, growth rates, and movement distances of recaptured *Fundulus heteroclitus* by shoreline and sex

| Movement type	Shoreline	Sex	Recaptures	Mean initial total length (mm)	Mean linear growth rate (mm day^{-1})	Mean specific growth rate (% body mass day^{-1})	Percent of recaptures $\leq$ 5 m from tagging location[c]
Fish that did not move from tagging site	*Spartina*	female	11	63.7 ± 3.0	0.10 ± 0.03	0.29 ± 0.11	44
		male	9	61.1 ± 1.9	0.05 ± 0.02	0.06 ± 0.10	75
		total	20 (11%)	62.5 ± 1.7	0.07 ± 0.02	0.10 ± 0.08	59
	Phragmites	female	12	62.3 ± 1.0	0.07 ± 0.02	0.22 ± 0.08	90
		male	12	60.7 ± 2.2	0.04 ± 0.02	0.13 ± 0.06	50
		total	24 (13%)	61.5 ± 1.2	0.06 ± 0.01	0.18 ± 0.05	70
	Riprap	female	16	60.3 ± 2.3	0.16 ± 0.02	0.67 ± 0.18	67
		male	8	58.5 ± 3.7	0.12 ± 0.03	0.59 ± 0.22	14
		total	24 (13%)	59.7 ± 1.9	0.15 ± 0.02	0.65 ± 0.14	50
	Bulkhead[a]	female	8	65.0 ± 1.4	0.13 ± 0.04	0.27 ± 0.08	50
		male	4	65.0 ± 3.0	0.09 ± 0.05	0.35 ± 0.14	75
		total	12 (7%)	65.0 ± 1.2	0.12 ± 0.03	0.30 ± 0.06	58
	All	female	47	62.4 ± 1.0	0.12 ± 0.01	0.39 ± 0.07	64
		male	33	60.8 ± 1.3	0.07 ± 0.01	0.22 ± 0.07	52
		total	80[a] (11%)	61.7 ± 0.8	0.10 ± 0.01	0.32 ± 0.05	59
Fish that moved from tagging site	–	female	4	66.9 ± 1.8	0.09 ± 0.03	0.63 ± 0.28[b]	–
		male	4	58.9 ± 2.5	0.21 ± 0.16	0.31 ± 0.21	–
		total	8	62.9 ± 2.1	0.15 ± 0.07	0.44 ± 0.16[b]	–
Overall	–	female	51	62.7 ± 1.0	0.12 ± 0.01	0.40 ± 0.07	–
		male	37	60.6 ± 1.2	0.08 ± 0.02	0.23 ± 0.07	–
		total	88[a] (12%)	61.8 ± 0.7	0.10 ± 0.01	0.33 ± 0.05[b]	–

Values in parentheses under the "Recaptures" column are percentage of fish tagged at that shoreline type which were recaptured. Standard errors are given for all means

[a] The one fish recaptured at site BU2 was excluded from analyses; see Methods section. Thus, 89 of the 725 tagged *F. heteroclitus* (12%) were recaptured; 81 of which were recaptured at the same shoreline site where they were tagged

[b] The sample size is 1 less than indicated (initial weight of one fish was unavailable so specific growth rate was not calculated); see Methods section

[c] only fish recaptured at 8 recapture locations nearest the site's tagging locations are included; see Methods section

Fig. 3 (a) Linear growth rate (mm day^{-1}) and (b) specific growth rate (% body mass day^{-1}) of *Fundulus heteroclitus* by shoreline type and sex. Growth rates with the same letter above the bar are not significantly different from one another (Tukey multiple comparisons; critical value of 0.05). Error bars are standard errors
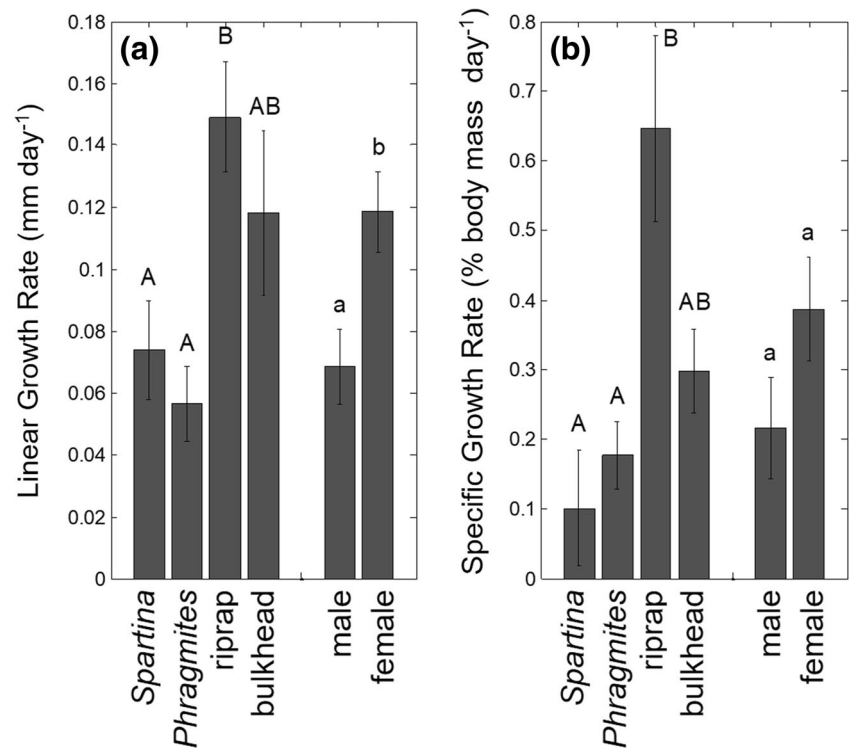
Fundulus heteroclitus along bulkhead had the lowest relative productivity, 75% less than at riprap (Table 4).

Movement Distance

Fundulus heteroclitus were recaptured 30–63 days after tagging (41 days average). Of the 89 recaptures, 22% were from their original tagging location and 47% were within 5 m. Of the 71 fish considered for the within-site movement comparison (fish recaptured at the 8 recapture locations nearest their site's tagging locations, Fig. 2), 28% were recaptured at their original tagging location and 59% were recaptured within 5 m (Table 3). Percentage of recaptures within 5 m of original tagging location varied from 14% for males at riprap to 90%

for females at *Phragmites* (Table 3; Fig. 4). Chi-square tests showed no significant difference in movement distance (≤ 5 m vs > 5 m) among shoreline types for male ($\chi^2 = 6.545$, $\chi^2_{df} = 3$, $p = 0.09$) or female ($\chi^2 = 5.171$, $\chi^2_{df} = 3$, $p = 0.16$) fish.

Of the 89 recaptured *F. heteroclitus*, 8 were recaptured at a site other than where they had been tagged (Table 3). Five fish moved 20–50 m between the contiguous RR1 and SP2 sites (Fig. 1c); 4 of which were tagged at riprap and recaptured at *Spartina*, and 1 was the opposite. The other 3 moved 450–475 m from RR1 to PH2, BU2 to RR1, and PH2 to SP2 (Fig. 1c). Only 1 of the 8 fish moved from a vegetated site (SP2) to an armored site (RR1). The female *F. heteroclitus* that moved from their tagging site were significantly larger ($t = 2.605$, $t_{df} = 6.52$, $p = 0.04$) than those that had not, whereas the males

Table 4 Shoreline-specific relative productivity of *Fundulus heteroclitus* calculated using instantaneous growth rates from the present study multiplied by aerial densities (biomass proxy) from a concurrent seining study in the same section of upper Pepper Creek (Balouskus 2012); see Balouskus and Targett (2017) for methods

Shoreline	Relative instantaneous growth rate	Relative biomass (density)	Relative productivity
Spartina	0.16	12.35 (37.67)	1.97
Phragmites	0.27	7.94 (24.23)	2.14
Riprap	1	1 (3.05)	1
Bulkhead	0.46	0.54 (1.64)	0.25

Growth rates and densities were normalized to those at riprap (where values were set at 1) to generate relative values (see Methods section). Unnormalized density values (fish m^{-3}) from which the relative biomass values were determined are included in parentheses for context
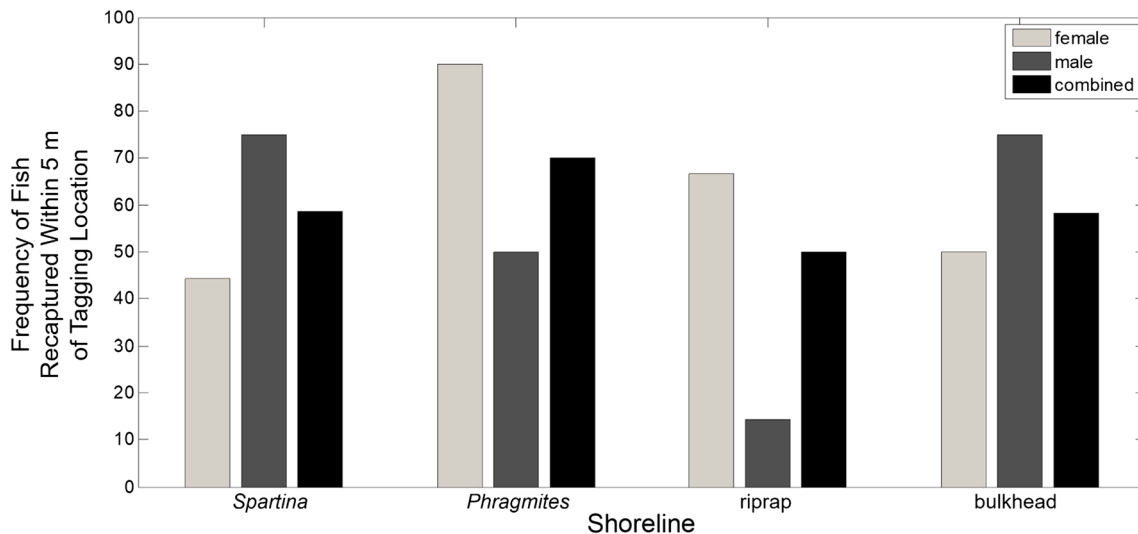
Fig. 4 Frequency of recaptured *Fundulus heteroclitus* that were recaptured ≤ 5 m from their tagging location, by shoreline type and sex. These numbers refer to the 71 fish considered in the movement distance analysis

in the two groups did not differ significantly in size ($t = -0.776$, $t_{df} = 5.35$, $p = 0.47$).

Discussion

Growth Rate

There have been no previous studies of *F. heteroclitus* growth along armored shorelines, or comparisons of growth along armored structures with growth at vegetated shores. Growth rate along riprap (0.15 mm day^{-1}) was significantly higher than along *Spartina* (0.07 mm day^{-1}) and *Phragmites* (0.06 mm day^{-1}) shorelines, whereas growth along bulkhead (0.12 mm day^{-1}) was intermediate and not significantly different from the other shorelines. Mean growth rate at *Spartina* sites in this study is lower than rates reported in previous studies in Delaware *Spartina* marsh habitat. Growth rates of 0.20–0.25 mm day^{-1} were reported for similar sized *F. heteroclitus* in *Spartina* marsh during summertime (Teo and Able 2003; Hagan et al. 2007). Hagan et al. (2007) also reported significantly lower summertime growth for *F. heteroclitus* in *Phragmites* than in *Spartina* marsh (0.13 and 0.24 mm day^{-1}, respectively), with the difference less pronounced during August–September.

Shoreline hardening is generally done to protect adjacent upland development, so shoreline hardening is most likely to occur at fringing marsh (Currin et al. 2008). Most of the marsh shorelines along Pepper Creek are fringing marshes, and the *Spartina* and *Phragmites* sites in this study were 2–5-m wide. Narrow marsh width can limit the amount of flooded marsh surface available to *F. heteroclitus*, which typically move onto the marsh surface at high tide to feed (Weisberg et al. 1981;

Weisberg and Lotrich 1982; Javonillo et al. 1997). This may be why observed growth rates at *Spartina* and *Phragmites* sites in this study are lower than those reported in previous studies on *F. heteroclitus* from the interior of more extensive marsh systems (Teo and Able 2003; Hagan et al. 2007), and it may also contribute to the similarity in growth rate at the two vegetated sites in the present study. Comparisons of the ecological benefits of fringing marsh versus more extensive interior marsh for feeding and growth of fishes are warranted.

The higher growth rate along riprap shorelines than at *Spartina* and *Phragmites* can potentially be attributed to density-dependent processes associated with absolute and per-fish prey availability. The available literature suggests that *F. heteroclitus* inhabit vegetated shorelines at sufficient densities to be food limited, since they exhibit density-dependent growth both on the marsh surface and in the adjacent subtidal zone (Kneib 1981; Weisberg and Lotrich 1982, 1986; Lockfield et al. 2013). *Fundulus heteroclitus* typically feed on benthic invertebrates, such as small crustaceans and polychaetes, and may consume large amounts of plant material when prey is scarce, although detritus is likely of little nutritional value (Prinslow et al. 1974; Baker-Dittus 1978; Kneib and Stiven 1978; Kneib 1986). Weisberg and Lotrich (1982) showed that *F. heteroclitus* caged without access to the marsh surface grew significantly slower than those caged with marsh surface access. Furthermore, the same study showed that *F. heteroclitus* without marsh surface access grew faster than those starved under laboratory conditions, demonstrating utilization of subtidal food resources. In fact, Weisberg and Lotrich (1982) estimated that as much as 75% of the energy requirements of the *F. heteroclitus* population in Canary Creek, DE could come from subtidal sources. These results suggest density-dependent growth processes along riprap, where *F. heteroclitus* density was < 10%

of that at *Spartina*. If riprap does not greatly reduce prey availability in the subtidal environment compared with *Spartina*, then riprap shoreline could have more food available per fish than *Spartina* shoreline, particularly if suitable prey are also associated with the riprap structure itself.

Seitz et al. (2006) found no significant differences in density or diversity of infauna between riprap and natural marsh shorelines in a relatively undeveloped system in Chesapeake Bay, which led to speculation that nearby marsh could "subsidize" developed shorelines in systems where proximate marsh was sufficiently abundant. In the present study, the riprap shorelines were short (45–55 m) and in close proximity to *Spartina* marsh. Different growth rates observed in a species with a high degree of site fidelity, such as *F. heteroclitus,* along different shoreline types in the present study suggest that limited stretches of shoreline (35–65-m long) have distinct microenvironments.

Female *F. heteroclitus* in the present study had significantly faster linear growth rates than males, but not faster specific growth rates. Sex-specific growth rate differences have been reported in other studies (e.g., Kneib 2009) and are consistent with the observation that females are longer than males of the same age, but males are heavier than females of the same length (Kneib and Stiven 1978).

Shoreline-Specific Relative Productivity

The estimates of shoreline-specific relative productivity were based on growth data from this study and density data from a concurrent study at the same location in upper Pepper Creek (Balouskus 2012; Balouskus and Targett 2017). Estimates showed that both *Spartina* and *Phragmites* shores supported approximately 95–115% higher relative productivity of *F. heteroclitus* than riprap, and that productivity was lowest along bulkhead. Higher growth rates of *F. heteroclitus* along armored shoreline types, particularly at riprap, was not enough to compensate for the lower densities that occur at these habitat types.

Geomorphology of the armored shorelines may have contributed to the lower *F. heteroclitus* densities there. Banks at the armored shorelines were steeper and reached deeper depths nearer to shore than at vegetated sites. McIvor and Odum (1988) showed that fishes, including *F. heteroclitus*, select shallow depositional habitats where invertebrate prey abundance is greater and predator pressure is less than along steeper sloped erosional banks. We speculate that armored shorelines in the present study function more like erosional tributary banks than depositional habitats, in terms of prey abundance and predation risk. Predation pressure would be further lessened at vegetated shorelines by greater access to the intertidal marsh where *F. heteroclitus* predators are less abundant (Banikas and Thompson 2012).

Potential changes in predation pressure and access to intertidal habitat due to armoring suggest that hybrid approaches

may be better shoreline stabilization options. One such option is riprap-sill, which consists of a rock sill with native vegetation planted between the sill and the shore. Riprap-sills have recently been shown to offer equivalent or superior protection from erosion and higher nekton abundance compared with traditional riprap and bulkhead (Gittman et al. 2014, 2016; Balouskus and Targett 2016). Further study of the physical and biological performance of such hybrid structures is warranted.

It should be noted that the relative productivity comparisons in this study apply to *F. heteroclitus* that inhabit the shoreline edge environment, along the adjacent channel. Marsh surface habitats were not sampled. Larvae and early juveniles (up to 10 mm SL) stay within shallow marsh pools that remain on *Spartina* marsh surface, and even to a lesser extent *Phragmites*, after the flood tide recedes (Kneib 1984; Able and Hagan 2003). Thus, *Spartina* marsh likely had greater larval and early juvenile *F. heteroclitus* abundances than the other three shorelines.

Movement Distance

Twenty-two percent of the *F. heteroclitus* recaptured in the present study were at their original tagging location and 47% were within 5 m, after 1–2 months. No significant differences in within-site movement distance were detected in male or female *F. heteroclitus* among shoreline types. These results are consistent with the high degree of site fidelity known to be a characteristic of this species, as shown by Lotrich (1975) who found that most *F. heteroclitus* were recaptured within 18 m of their tagging location in Canary Creek, Delaware. However, some individuals in the present study traveled 450–475 m between sites over 1–2 months. Other recent studies have also reported that *F. heteroclitus* can move such distances. Able et al. (2006) reported that some *F. heteroclitus* were recaptured as far as 299 m away up to 166 days after tagging in marshes along the Mullica River, a tributary of Great Bay in southern New Jersey, and Able et al. (2012) documented movements of 1000–1200 m over a 17-month study in marshes and a boat basin in Great Bay. Skinner et al. (2005) reported several instances of *F. heteroclitus* movement greater than 2000 m, and two movements of 3500–3600 m in just a few weeks during late spring and early summer along the shore of the upper Miramichi River estuary in northeastern New Brunswick, Canada. Meyer and Posey (2014) postulated a spring/summer spawning and recruitment phase and a late fall/winter dispersal-colonization phase for *F. heteroclitus*. We note that the 450–475 m maximum movement distances observed in the present study occurred during spring and summer when small-scale populations are postulated to be maintained, not during the larger-scale dispersal phase.

Movement of *F. heteroclitus* between shoreline types in this study was based on only 8 individuals, but 5 moved from armored to vegetated shorelines and 1 moved from vegetated to armored. The other 2 moved either between vegetated or armored shorelines. Additionally, more movement was observed between adjacent armored and *Spartina* shorelines than between adjacent vegetated shorelines. At the contiguous SP2 and RR1 sites, 34 tagged fish were recaptured and 5 of those fish switched between these sites (4 from riprap to *Spartina* and 1 in the opposite direction). However, at the contiguous vegetated sites SP1 and PH1, none of the 22 recaptured fish had moved between sites. Although sample size is limited, these results suggest that *F. heteroclitus*, at the size studied here, have a greater tendency to move from armored to vegetated shorelines, particularly to *Spartina*, than between *Spartina* and *Phragmites* fringing marshes.

Conclusions

This research shows that armoring shorelines along fringing marshes, where pressure to protect upland development is most likely, may negatively impact the localized productivity of *F. heteroclitus*. Although prey resources are sufficient to support greater growth rates along riprap than at *Spartina* and *Phragmites* shores, armoring reduces *F. heteroclitus* abundance sufficiently to result in reduced production. We conclude that along armored shorelines, reduced *F. heteroclitus* abundance appears to be driven by habitat characteristics other than prey limitation. The same impacts are possible for other fishes which feed opportunistically on benthic invertebrates such as small crustaceans and annelids. We also note that living shorelines, such as riprap-sills, provide shoreline protection and may mitigate the negative effects of traditional bulkheading and riprap observed in this study (Gittman et al. 2014, 2016; Balouskus and Targett 2016). The results presented here demonstrate details of the negative biological impacts associated with traditional armored shoreline structures and provide support for the importance of alternative shoreline stabilization approaches.

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