Comparison of salt marsh creeks and ditches as habitat for nekton

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ABSTRACT: Salt marshes are dynamic systems supporting a diverse assemblage of resident and transient nekton (free-swimming fish and decapod crustaceans). Within a marsh, many sub-habitats are used by nekton, including ditches and natural creeks. While the use of natural creeks is well documented, the role of ditches as habitat for nekton remains less well known. The present study describes the nekton-support function of this prolific marsh sub-habitat, and compares it to tidal creeks. Sampling was conducted in the summer of 2008 in a microtidal, polyhaline salt marsh on Fire Island, NY, USA. Ditches and creeks were found to have different nekton communities. Creeks had significantly higher species richness and represented nearly all ditch species. Prolonged anoxia (up to 18 h) in ditches is suggested as a significant factor contributing to differences in the creek and ditch nekton community. Upper portions of ditches and creeks differed for resident nekton species, with higher species richness and density of the mummichog *Fundulus heteroclitus* in the upper portion of creeks than in the upper portion of ditches. Additional differences were seen in the nekton community composition of open and naturally plugged ditches, where the densities of individual species (such as *F. heteroclitus*, which was more abundant in open ditches) also differed. Ditches are a major habitat type in salt marshes of the northeastern USA, such as at the Fire Island marsh site studied here, and as efforts are considered to remove ditches from marshes as a habitat restoration technique, their nekton support function should be considered.

KEY WORDS: Ditches · Mosquito ditches · Salt marsh creeks · Nekton · *Fundulus heteroclitus* · Plugged ditches · Great South Bay

INTRODUCTION

Within a salt marsh, many sub-habitats exist, including the marsh surface, pools, pannes, creeks, and ditches. These sub-habitats are often used preferentially by different nekton species, presumably due to some intrinsic advantage of that type of habitat (Halpin 1997, Minello et al. 2003, Allen et al. 2007). Refuge from predation (Boesch & Turner 1984, Deegan et al. 2000), access to the marsh surface for foraging (Rozas et al. 1988), competitive interaction (Peterson & Turner 1994), and suitable water quality (Smith & Able 2003) are all drivers in the distribution of these species. While the use of many of these sub-habitats by nekton

has been studied (e.g. marsh surface by Hettler 1989, Kneib & Wagner 1994, Peterson & Turner 1994; pools by Raposa et al. 2003, Able et al. 2005; creeks by Weinstein & Brooks 1983, Rozas et al. 1988, Rountree & Able 1992, Allen et al. 2007, Granados-Dieseldorff & Baltz 2008), the differences between marsh creeks and ditches have not been significantly addressed.

Many species use marsh creeks as a conduit to access the vegetated marsh surface at high tide for foraging (e.g. Weisberg & Lotrich 1982, Rozas et al. 1988). In this way, tidal creeks are involved in the trophic transfer of energy between the marsh surface and adjacent open water (Kneib 2002). The suitability of creeks for nekton utilization is reportedly dependent

on specific physical parameters, including flow, depth, location, and other geomorphologial factors (Rozas 1992, Williams & Zedler 1999, Visintainer et al. 2006, Allen et al. 2007). For example, sloped banks of marsh creeks are conducive to more abundant nekton populations (McIvor & Odum 1988, Rozas 1992, Allen et al. 2007). Despite the fact that they both transfer water tidally through the marsh, ditches and creeks are physically very different. The present study will assess the capacity of ditches to support nekton, given that they lack the gently sloping sides and other physical attributes that make creeks attractive to nekton (McIvor & Odum 1988, Allen et al. 2007).

Ditches were installed in salt marshes throughout the northeastern United States in the 1930s for mosquito control purposes, creating an extensive and conspicuous grid network with altered hydrology and ecological structure (Taylor 1938, Bourn & Cottam 1950, Daiber 1986, Talbot et al. 1986, Wolfe 1996, Lathrop et al. 2000, Adamowicz and Roman 2005, Crain et al. 2009). Some restoration efforts are underway or are being considered to remove ditches from the marsh landscape by filling, plugging at the mouth to encourage natural filling, or other techniques (e.g. Adamowicz et al. 2004, Baustian & Turner 2006, Rochlin et al. 2009). Also Open Marsh Water Management (OMWM), an alternative method to grid-ditching, is now widely implemented for mosquito control and has been offered as a technique to restore hydrology of grid-ditched marshes (Meredith et al. 1985, Meredith & Lesser 2007). In brief, the intent of OMWM is to promote mosquito control through restoring the hydrology of grid-ditched marshes. For example, standing water (e.g. pond) is created in areas of high mosquito-breeding potential, thus establishing habitat that is unsuitable for salt-marsh mosquito breeding. Radial ditches are also installed, targeting drainage of specific breeding areas and allowing access to the marsh surface by larvivorous fishes.

In light of these restoration efforts, this study sought to answer two questions. (1) Are there distinct ditch and creek nekton communities, and (2) would the removal of ditches impact the overall marsh nekton community? Resource managers will require information from multiple disciplines (e.g. ecology, hydrology, geomorphology, biogeochemistry) to support decisions regarding marsh restoration by ditch removal. This study sought to assess one of these important factors, an evaluation of the nekton habitat function of salt-marsh ditches. At a study site on Fire Island, NY, comparisons were made between (1) ditches and creeks, (2) upper ditch (portion toward the upland) and lower ditch (portion toward the mouth), (3) upper creek and lower creek, and (4) open-mouthed and naturally plugged ditches.

MATERIALS AND METHODS

Study area. This study was conducted in the Otis Pike High Dunes Wilderness Area of Fire Island National Seashore, Brookhaven, NY (40.7° N, 73.2° W). Fire Island, a barrier island system, is located off the south shore of Long Island, about 50 km east of New York City (Fig. 1). The salt marshes of the Wilderness Area span roughly 10 km, and are bordered to the north by Great South Bay and to the south by the shrub community of the barrier island. The salt marshes are dominated by *Spartina alterniflora* and *S. patens*. The tides are semidiurnal with a mean range of <0.5 m in the vicinity of the Wilderness marshes. This area of the barrier island is undeveloped, lacking roads or other interior access. Because it is a Wilderness Area, the National Park Service is considering options for restoring the marsh system to as natural a condition as possible, without evidence of mosquito control ditches. The ditches of the Wilderness Area marshes, installed for mosquito control purposes in the 1930s, are typically oriented perpendicular to the open water of Great South Bay and are spaced about 50 m apart. The ditches are extensive within the 200 ha Wilderness Area marshes, with a cumulative length of about 74 km (Corman 2009).

Study design. Nekton sampling was conducted in both natural marsh creeks and human-made ditches (Fig. 1). The sampled ditches were on average 0.8 m wide, 0.4 m deep and 90 to 360 m long. Sampled creeks were on average 25 m wide and 130 to 300 m long. There are 8 natural creeks within the Wilderness Area, 6 of which were accessible for sampling. Nine ditches were randomly selected (from within accessible parts of the marsh) for sampling. Ditches were excluded that were of insufficient length to allow establishment of 2 nekton sampling stations with at least 25 m between stations. Half the sampled ditches were open at the mouth and half were plugged at the mouth by natural sediment transport processes. Both open ditches and creeks had a permanent connection to the open water, while, for the naturally plugged ditches, hydrologic connection was limited to high tide.

The 9 ditches and 6 creeks were each sampled 4 times during July and August 2008. This is the period of maximum nekton abundance and species richness for salt marsh nekton (Rountree & Able 1992, Raposa et al. 2003, Allen et al. 2007). To account and test for spatial variability within ditches and creeks, each ditch and creek was divided in half. One station was then located randomly between the middle of the creek or ditch and the mouth, and the other between the middle and the upland end of the ditch or creek (hereinafter referred to as 'lower' ditch or creek and 'upper' ditch or creek).
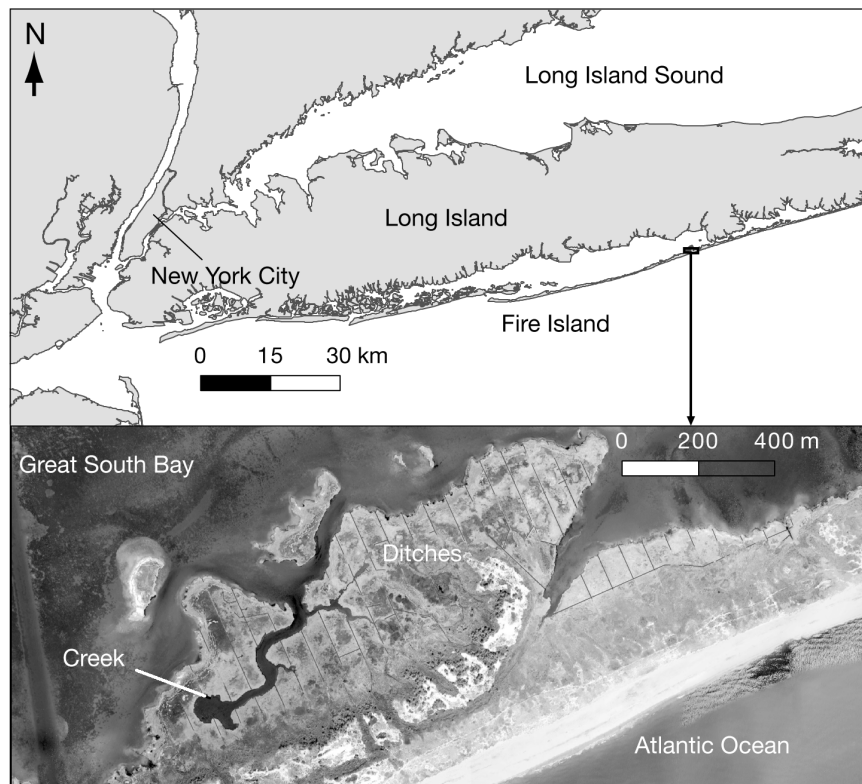
Fig. 1. Location of study marsh, Fire Island, NY. Aerial photograph shows meandering creek and linear ditches, some of which intersect the creeks. Data sources: base map, NOAA; aerial photograph, NYS GIS

Field methods. A total of 72 ditch and 48 creek samples were collected. All nekton sampling took place near daytime low tide. Because nekton forage on the marsh surface when it is accessible, sampling was never conducted when the marsh surface was flooded (Peterson & Turner 1994).

Creek sampling was conducted using throw traps (Raposa et al. 2003). Each 1 m^2 trap was made of a square aluminum frame with a side length of 1 m and a height of 0.5 m. The side panels were made of 3 mm mesh. The trap was deployed by quietly approaching the creek from the marsh surface, then throwing the trap a distance of about 4 m through the air such that it landed near the middle of the creek and settled into the bottom sediment.

After deployment, a 1 × 0.5 m dip net with 1 mm mesh was passed through the trap from one end to the other to retrieve organisms from the trap, skimming the sediment surface. Each time the net was brought to the surface, all organisms collected on the net were identified, counted, and then returned to the creek (outside the trap). The dip net was passed through the throw trap from each direction until 3 consecutive passes yielded zero individuals. Temperature (°C) and salinity (ppt) were then measured using a handheld YSI Model 85 meter. Time, tide direction, water depth, and GPS coordinates were also recorded.

Ditch sampling was conducted using ditch nets (James-Pirri et al. 2010). The ditch net is a quantitative enclosure trap designed to sample intertidal and subtidal salt marsh ditches. Ditch nets cover the walls and bottom of ditches with mesh net, allowing nekton to freely swim through the ditch. After a minimum acclimation period of 20 min (James-Pirri et al. 2010), the ditch was quietly approached from the marsh surface, and mesh doors at each end of the ditch net, hidden within the bottom sediment of the ditch, were rapidly pulled shut. The net with trapped nekton was then pulled from the ditch and placed on the marsh surface. Each organism within the net was identified, counted, and then returned to the ditch. The surface area sampled by the net was variable depending on the dimensions of each ditch, with an average area captured by a ditch net in this study of 0.4 m^2. Like the throw traps, these nets were constructed of 3 mm mesh. As with the creeks, temperature (°C) and salinity (ppt) were then measured using a handheld YSI Model 85, and time, tide direction, water depth, and GPS coordinates were recorded.

Quantitative sampling of salt marsh nekton can be conducted using any of numerous methods, including flume weirs, block nets, bottomless lift nets, throw traps, and ditch nets (Rozas & Minello 1997, James-Pirri et al. 2010). By sampling a known area, the quantity of nekton captured in the sample can be used to estimate the overall density or abundance of that organism. In the present study, due to the physical differences between creeks and ditches, different sam-

pling techniques were necessary. At Fire Island, the majority of ditches did not drain at low tide (prohibiting the use of several methods, such as flume weirs) and many were too narrow to allow for the use of a seine (also precluding other methods, such as block nets). Throw traps are an excellent quantitative gear type for creek sampling, but because many of the ditches were narrow (<1 m across) and had uneven bottom contours, a throw trap would be unlikely to form a seal on the ditch bottom to prevent nekton from escaping. Ditch nets were not feasible for use in creeks because the method requires that the net span the entire width of the creek. Except for the far upstream origin of marsh creeks, they are too wide for the ditch net method. Simply put, no sampling method is appropriate for use in both habitat types. However, ditch nets and throw traps are both quantitative enclosure methods that sample a known area with high capture and recovery efficiencies (Kushlan 1981, Rozas & Minello 1997, James-Pirri et al. 2010), thus producing comparable data regarding the nekton community of this system.

To consider potential tidal and diel fluctuations, water quality and physical data (temperature, salinity, dissolved oxygen, water depth) were monitored in one creek and in 3 ditches for periods of 2 to 12 d using a YSI Model 6600EDS instrument recording at 15 min intervals. While the replicates with this instrument were limited, their deployment was designed to illustrate trends in each habitat type and to provide a basis for comparison, rather than to provide any statistically interpreted conclusions. Due to the small tidal range at the Wilderness Area marshes and to the very shallow standing water (often <10 cm) in portions of many ditches and creeks at low tide, sample locations were selected that had sufficient standing water at low tide that the probe, placed near the creek or ditch bottom, would always be submerged.

Data analysis. Nekton data were analyzed for differences in species richness, individual species density, and community composition between groups (ditches and creeks, upper and lower portions of creeks, upper and lower ditch, open and naturally plugged ditches). Environmental data were also compared. Except for species richness, the data did not meet the assumptions of normality and homogeneity of variance required to conduct parametric statistical analyses, even after several data transformations were applied, including $\log(x + 1)$, square root and 4th root.

To evaluate species richness, a jackknife estimate was calculated, and then a t-test was performed to compare different marsh habitat types. Individual species density comparisons were performed using a Mann-Whitney U-test for the 7 most abundant species collected during the field sampling (*Cyprinodon varie-*

gatus, Fundulus heteroclitus, F. luciae, F. majalis, Lucania parva, Menidia menidia and *Palaemonetes pugio*). Significance was assessed for multiple comparisons as needed using a sequential Bonferroni adjustment as described by Rice (1989), ranking comparisons in order of their p-values and then applying different levels of significance to each. This allowed multiple comparisons to be made without biasing the results toward increased incidence of significance with increasing numbers of comparisons.

Total community composition was determined using Analysis of Similarity (ANOSIM) and Similarity Percentage (SIMPER) with a Bray-Curtis similarity matrix. For significant differences determined through ANOSIM, SIMPER identified the contribution of individual species that defined the nekton community differences. ANOSIM was used for 4 habitat type comparisons (ditch versus creek, upper versus lower ditch, upper versus lower creek, and open versus plugged ditches) and then a sequential Bonferroni adjustment was considered at the 0.05 p-value and at the less robust, but still meaningful, 0.10 p-value. The environmental data collected at each site (temperature and salinity) were compared by habitat types as indicated above, also using Mann-Whitney U-tests and Bonferroni-adjusted p-values. Mann-Whitney analyses were conducted in SPSS (version 11.0 for Windows), t-tests in Microsoft Excel 2007, and ANOSIM and SIMPER in Primer-E (version 6; Clarke & Warwick 2001).

RESULTS

Records of environmental data

The tidal range in ditches and creeks was similar, between 30 and 40 cm. During the continuous record, a peak in dissolved oxygen (DO) was typically recorded in the late afternoon (Fig. 2). After this peak the DO in the ditch decreased to near 0% or 0% saturation, although, for the creek, levels only declined to 10 to 40% saturation. Duration of the low-DO period was also more prolonged in the ditches than in the creek, with anoxia observed for up to 18 h in some ditches. No anoxic events were recorded in the creek during this summer sampling.

The continuous-recording data were in good agreement with the discrete measurements from the ditches and creeks. The mean (±SD) pooled temperature from the 3 continuous ditch deployments was 22.9 ± 2.5°C, while the average from all ditch discrete values was 22.6 ± 4.3°C. The salinity from the continuous-recording data in the ditch (mean ± SD) was 22.9 ± 4.6 ppt, while the discrete measurements averaged 24.2 ± 5.0 ppt. In the creeks, only one deployment was con-

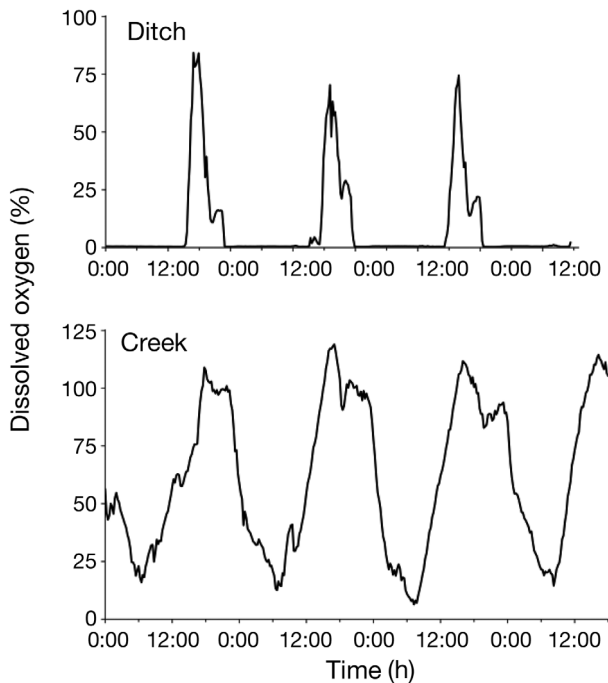Fig. 2. Dissolved oxygen (% saturation) in a creek and ditch over independent 4 d periods

ducted, and the continuous-recorded temperature (mean ± SD) during that deployment was 24.7 ± 2.3°C, while the discrete measurement was 26.4 ± 2.1°C. Average creek salinities were 27.2 ± 0.5 and 24.8 ± 3.6 ppt for continuous-recording data and discrete values, respectively. It should be noted that discrete measurements were all taken during daylight and near low water, as opposed to the continuous record, which was taken day and night and through all tidal stages, a factor presumed to contribute to the observed differences.

Average temperature, derived from the discrete measurements, showed a significant difference (p < 0.001) between ditches (mean ± SD: 22.6 ± 4.3°C) and creeks (26.4 ± 2.1°C), although no difference was seen in discrete salinity between these two habitat types. No significant differences were observed between the upper (26.1 ± 1.6°C, 26.0 ± 1.8 ppt) and lower (26.7 ± 2.2°C, 23.8 ± 4.4 ppt) portions of creeks, or between the upper (21.7 ± 4.1°C, 23.1 ± 5.8 ppt) and lower ditch (23.5 ± 4.4°C, 25.4 ± 3.8 ppt), when the more stringent Bonferroni adjustments were applied (at the p = 0.05 level). There was a significant difference in temperature between the upper and lower portion of ditches only at the Bonferroni-adjusted p = 0.10 level. Open ditches were shown to have both significantly lower average discrete temperatures (20.6 ± 2.9°C) and salinities (21.5 ± 4.0 ppt) than plugged ditches (25.1 ± 4.6°C, 27.7 ± 3.8 ppt) at both the 0.05 and 0.10 Bonferroni-adjusted p-value levels.

Ditches versus creeks

Nine species were captured in ditches and 13 in creeks (Table 1). All ditch species were also found in creeks, with the exception of *Callinectes sapidus*. However, only 1 individual of this species was captured during the study, representing <0.1% of the total nekton abundance. Several species found in creeks were not found in ditches, including *Anchoa mitchilli, Anguilla rostrata, Carcinus maenas, Crangon septemspinosa* and *Mugil curema*. Jackknifed species richness showed a difference between ditches (11.0 species) and creeks (16.0 species) (p < 0.001; Table 2). The nekton community was significantly different between ditches and creeks at the 0.10 level (ANOSIM, p = 0.081; Table 3). When considered independently, the densities of 3 species were significantly different: *Cyprinodon variegatus* (p = 0.006), *Menidia menidia* (p = 0.002) and *Palaemonetes pugio* (p < 0.001) (Tables 4 & 5).

Table 1. All nekton species captured (X) in ditches or creeks at least once

Species	Common name	Ditches	Creeks
Anchoa mitchilli	Bay anchovy		X
Apeltes quadracus	Fourspine stickleback	X	X
Anguilla rostrata	American eel		X
Crangon septemspinosa	Sand shrimp		X
Carcinus maenas	Green crab		X
Callinectes sapidus	Blue crab	X	
Cyprinodon variegatus	Sheepshead minnow	X	X
Fundulus heteroclitus	Mummichog	X	X
Fundulus luciae	Spotfin killifish	X	X
Fundulus majalis	Striped killifish	X	X
Lucania parva	Rainwater killifish	X	X
Mugil curema	White mullet		X
Menidia menidia	Silverside	X	X
Palaemonetes pugio	Grass shrimp	X	X

Table 2. Jackknife estimates of species richness (mean ± SD). Comparisons between habitats were made using a *t*-test with a significance level of p < 0.05

Habitat	Species richness	p
Creek	16.0 ± 1.4	
Ditch	11.0 ± 1.1	<0.001
Upper creek	14.9 ± 2.1	
Lower creek	13.9 ± 1.3	0.065
Upper ditch	4.0 ± 0.0	
Lower ditch	11.0 ± 1.4	<0.001
Plugged ditch	9.9 ± 1.3	
Open ditch	9.9 ± 2.2	0.977

Table 3. Nekton community comparison for the 4 primary habitat groups of interest. The p-value (derived from ANOSIM) was tested for significance with a Bonferroni adjustment of both p = 0.05 and p = 0.10. Multiple comparisons were significant at the 0.10 level

Habitat	Global R	p	Bonferroni p = 0.05	p = 0.10
Ditches vs. creeks	0.022	0.081	0.05	0.1
Upper vs. lower creek	0.053	0.063	0.025	0.05
Upper vs. lower ditch	0.07	0.013	0.0125	0.025
Open vs. plugged ditches	0.061	0.019	0.0167	0.033

Upper versus lower creek

When comparing the upper and lower portions of creeks, no significant differences were noted in species richness (Table 2), community composition (ANOSIM; Table 3), or discrete measurements of temperature or salinity (p > 0.05). There was a significant difference in the density of *Fundulus heteroclitus* with 25.6 ± 46.5 ind. m^{-2} (mean ± SD) in the upper creek and 2.4 ± 3.7 ind. m^{-2} in the lower creek (Table 5).

Upper versus lower ditch

The upper and lower portions of ditches represented 2 distinct communities (ANOSIM, p = 0.013, significant at the 0.10 level; Table 3). SIMPER results suggest that 79% of the variability between these portions of ditches was related to differences in the density of *Fundulus heteroclitus, F. luciae,* and *Lucania parva.* The average density in the lower ditch for *F. heteroclitus* was 28.2 ind. m^{-2}, while in the upper portion of ditches it was only 8.5 ind. m^{-2}. In contrast, *F. luciae* and *L. parva* both showed higher densities in the upper portion of ditches (2.7 and 3.0 ind. m^{-2}, respectively) than in the lower ditch (1.1 and 0.2 ind. m^{-2}, respectively; Table 4). Species richness was significantly different (p < 0.001; Table 2) for upper (4.0 species) and lower (11.0 species) ditch areas. When analyses were conducted on each species independently, the density of only 1 species (*Cyprinodon variegatus*) was significantly different between the upper and lower ditch (p = 0.011; Table 5). Again, since the Mann-Whitney test is more rigorous than the SIMPER, these results can differ from SIMPER results.

Open versus plugged ditches

Open ditches and naturally plugged ditches contained different nekton communities (ANOSIM, p = 0.019, significant at the 0.10 level; Table 3). Open and

Table 4. Mean (± SD) density m^{-2} of each species by habitat type

Species	Ditch	Creek	Upper ditch	Lower ditch	Upper creek	Lower creek	Open ditch	Plugged ditch
Anchoa mitchilli	0	0.19 ± 1.04	0	0	0.29 ± 1.43	0.08 ± 0.41	0	0
Anguilla rostrata	0	0.17 ± 0.38	0	0	0.25 ± 0.44	0.08 ± 0.28	0	0
Apeltes quadracus	0.02 ± 0.14	0.1 ± 0.72	0	0.03 ± 0.20	0.21 ± 1.02	0	0.03 ± 0.19	0
Callinectes sapidus	0.03 ± 0.22	0	0	0.05 ± 0.32	0	0	0	0.06 ± 0.34
Carcinus maenas	0	0.02 ± 0.14	0	0	0	0.04 ± 0.20	0	0
Crangon septemspinosa	0	0.83 ± 4.64	0	0	0.04 ± 0.20	1.63 ± 6.53	0	0
Cyprinodon variegatus	0.34 ± 1.45	3.96 ± 19.56	0	0.69 ± 2.00	7.38 ± 27.49	0.54 ± 1.14	0.06 ± 0.38	0.7 ± 2.09
Fundulus heteroclitus	18.34 ± 50.88	13.98 ± 34.67	8.51 ± 20.50	28.18 ± 68.07	25.58 ± 46.49	2.38 ± 3.73	27.03 ± 63.91	7.48 ± 23.92
Fundulus luciae	1.9 ± 9.03	0.23 ± 0.88	2.7 ± 12.62	1.09 ± 2.21	0.29 ± 1.08	0.17 ± 0.64	1.27 ± 3.15	2.68 ± 13.16
Fundulus majalis	1.03 ± 3.25	0.75 ± 2.07	0.63 ± 2.29	1.42 ± 3.99	0.92 ± 2.64	0.58 ± 1.32	1.37 ± 3.79	0.6 ± 2.42
Lucania parva	1.58 ± 7.67	0.83 ± 3.86	3.01 ± 10.72	0.16 ± 0.56	1.5 ± 5.41	0.17 ± 0.48	0.23 ± 0.83	3.28 ± 11.34
Menidia menidia	0.53 ± 3.17	1.77 ± 10.24	0	1.05 ± 4.46	3.25 ± 14.47	0.29 ± 0.55	0.4 ± 2.52	0.69 ± 3.88
Mugil curema	0	0.58 ± 1.92	0	0	0.92 ± 2.50	0.25 ± 1.03	0	0
Palaemonetes pugio	2.82 ± 23.73	6.38 ± 21.51	0	5.65 ± 33.55	5.38 ± 15.92	7.38 ± 26.27	0	6.35 ± 35.59

Table 5. p-values from a Mann-Whitney *U*-test comparing the average densities of the 7 most abundant individual species in different habitat types. Sequential Bonferroni adjustment of p-values was used following the method described in Rice (1989) at the p = 0.05 and p = 0.10 level

	Cyprinodon variegatus	*Fundulus heteroclitus*	*Fundulus luciae*	*Fundulus majalis*	*Lucania parva*	*Menidia menidia*	*Palaemonetes pugio*
Ditches vs. creeks	0.006	0.957	0.152	0.416	0.751	0.002	<0.001
Upper vs. lower creek	0.501	0.017	0.966	0.432	0.928	0.580	0.924
Upper vs. lower ditch	0.011	0.081	0.098	0.118	0.122	0.154	0.154
Open vs. plugged ditches	0.046	0.010	0.402	0.076	0.069	0.858	0.111

plugged ditches showed similar species richness (both 9.9 species; Table 2). However, ANOSIM showed 2 distinct communities because, despite having similar species richness, they did not have all species in common. Additionally, ANOSIM accounts for variability in the density of species. The SIMPER results revealed that >80% of the variability between the open and plugged ditches resulted from differences in the density of *Fundulus heteroclitus, F. luciae,* and *Lucania parva.* The average density in open ditches for *F. heteroclitus* was 27.0 ind. m^{-2}, while in plugged ditches it was only 7.5 ind. m^{-2}. *F. luciae* and *L. parva* both showed higher densities in the plugged ditches (2.7 and 3.3 ind. m^{-2}, respectively) than in the open ditches (1.3 and 0.2 ind. m^{-2}, respectively; Table 4). *F. heteroclitus* and *Cyprinodon variegatus* showed significantly different densities when considered independently (Table 5).

DISCUSSION

Ditches and creeks

Creeks support both a more diverse community and also represent all species found in the ditches. However, both open and naturally plugged ditches are also an important habitat, as they support many marsh resident species; 5 species (*Anchoa mitchilli, Anguilla rostrata, Carcinus maenas, Crangon septemspinosa, Mugil curema*) were captured in creeks but not in ditches (Table 1). All of these except *C. septemspinosa* are marsh-transient species that spend only part of their life cycle in the marsh (Crothers 1967, Ayvazian et al. 1992). Thus creeks may be a more highly utilized habitat than ditches for these transient estuarine species.

One important consideration with ditch-filling restoration is whether the current creek system on Fire Island would be capable of supporting the many nekton individuals that would be displaced by ditch removal. The results presented here show that the average nekton density in ditches was 26.6 ind. m^{-2} and the density in creeks was 30.3 ind. m^{-2}, representing a total of 1.6 million ind. in the 59 600 m^2 ditch area

within the Fire Island Wilderness Area and 1.4 million ind. in the 44 800 m^2 creek area (Corman 2009). Previous studies from the southern New England region have shown that salt marsh creeks have summer nekton densities similar to the creek and ditch nekton densities from the present study (e.g. 22.6 ind. m^{-2} and 27.95 ind. m^{-2} in Massachusetts, Raposa et al. 2003; 26.83 ind. m^{-2} in Rhode Island, Roman et al. 2002). We did not attempt to evaluate the nekton carrying capacity of ditches or creeks, but based on similar density estimates from other marshes in the northeastern USA, it seems unlikely that existing creeks could support a substantial increase in nekton density. However, Raposa et al. (2003) do report a creek nekton density of 89.90 ind. m^{-2} within a Rhode Island marsh.

The ditches were colder on average than the creeks, perhaps influencing the distribution of species within the 2 habitats. Moreover, the ditches sampled had prolonged periods (up to 18 h) of water column anoxia, while the creek never reached such low levels (Fig. 2). At all times, except for the late afternoon when DO in the ditches is high, nekton must either leave the ditches in search of a more oxygen-rich environment, or adapt to these low levels, such as *Fundulus heteroclitus* is known to do (Layman et al. 2000, Smith & Able 2003). The creeks were wider than the narrow and steep-bank protected ditches, which may have impacted water flow, wind mixing, and light availability for surface-water photosynthesis. These physical differences in the dimensions of the habitat types may have thus contributed to the observed differences in DO levels.

Hettler (1989) determined that marshes that are highly reticulated (i.e. have a high abundance of creeks and rivulets) could be more productive for fisheries than marshes without these sub-habitats. It is unknown whether ditches contribute in this same manner to marsh secondary productivity. It is essential to understand the role of such marsh microhabitats in order to determine what response, if any, there would be in the productivity of the greater estuary were some of these sub-habitats removed through restoration efforts.

Within-ditch and within-creek nekton patterns

The patterns of use between the upper and lower portions of the 2 habitat types in the present study suggest that nekton use of ditches may be different than nekton use of creeks. Rozas et al. (1988) showed that intertidal creek banks serve as conduits for nekton to access the marsh surface for foraging. Halpin (1997) demonstrated that *Fundulus heteroclitus* congregates in the upper portion of creeks, using this habitat as a residence during low tide in addition to a corridor to the marsh surface at high tide. Our creek data show patterns consistent with these prior studies, but our ditch data do not, suggesting that ditches are utilized differently by the local nekton community.

Comparing the upper portions of both ditches and creeks showed that species richness was lower in the upper ditch (4.0 species) than the upper creek (14.9 species) (Table 2). For creeks, a trend similar to Halpin (1997) was noted for *Fundulus heteroclitus*, with a significantly higher density in the upper portion of creeks (25.6 ind. m^{-2}) than in the lower portion (2.4 ind. m^{-2}) (Tables 4 & 5). However, the average density in lower ditches for *F. heteroclitus* was 28.2 ind. m^{-2}, while in the upper ditches it was only 8.5 ind. m^{-2}, an opposite trend from what we observed in creeks. These findings suggest that the upper reaches of creeks (i.e. those furthest from the creek mouth) are used when the marsh surface is not flooded. Studies have also shown that *F. heteroclitus, F. luciae* and *Palaemonetes pugio* remain near the marsh surface at low tide (Kneib & Wagner 1994). The congregation of *F. heteroclitus* in the upper creek noted in the present study supports these findings. Fish that remained in the creeks, rather than those that left the creeks on the ebb tide, would have a shorter travel distance (requiring less energy expenditure) to access the marsh surface at the next flood tide.

This fundamental difference between nekton utilization of the upper reaches of creeks and ditches could be explained by differences in habitat characteristics. Although conducted in a southern US marsh with some physical differences from Fire Island, a study by Allen et al. (2007) showed that nekton use of marsh creeks was related to geomorphological traits, including average depth, flow, steepness of banks, and proximity to uplands, but was not related to creek size (volume, length, width, or bottom area). Thus, the fact that the ditches we studied were narrower than creeks may not be a reason for the observed differences, but one major geomorphologic difference between creeks and ditches is bank steepness. In general, the ditches on Fire Island have very steep banks, while the creeks have more gently sloping banks. The ditches are also linear, while the creeks meander. In addition, the observed anoxia in ditches, as previously noted, probably influenced the nekton species composition and abundance.

The environmental data showed differences in temperature and salinity between the upper portion of ditches (21.7°C, 23.1 ppt) and the lower portion of ditches (23.5°C, 25.4 ppt), although these differences were not significant with the Bonferroni adjustment, with the exception of upper and lower ditch temperature at the p = 0.10 level. The within-ditch differences of <2°C and <2.5 ppt may not have been ecologically significant to these species in determining their distribution within ditches. Furthermore, there was a significant difference (p < 0.001) in temperature between the upper portion of creeks (26.7°C) and the upper portion of ditches (21.7°C). This slightly larger difference in temperature (5°C) between the upper portion of ditches and the upper portion of creeks could be a contributing factor to the observed nekton differences between ditches and creeks as a whole in less tolerant marsh species. However, killifish (and perhaps other marsh residents) are known to tolerate a wide range of both temperature and salinity, especially *Fundulus heteroclitus* (Garside & Chin-Yuen-Kee 1972, Umminger 1972, Griffith 1974).

Open and naturally plugged ditches

Open and naturally plugged ditches had different nekton communities (Table 3), and most notably, the density of *Fundulus heteroclitus* was more than 3 times greater in open ditches (Tables 4 & 5). These results could be related to differences in flow between open and plugged ditches, as water flow rate influences nekton use of salt marsh creeks (Cattrijsse et al. 1994, Allen et al. 2007). Plugged ditches do not flush to the same extent, perhaps causing them to function somewhat like a marsh pool (e.g. Raposa 2003, Smith & Able 2003, Able et al. 2005). This inherent difference in hydrology between the open and plugged ditches likely caused some difference in the resident species, such as in *F. heteroclitus* and *Cyprinodon variegates*. Differences between open and naturally plugged ditches have important ecological relevance because ditch plugging is a possible restoration technique (Adamowicz et al. 2004, James-Pirri et al. 2011).

Management perspective

Filling of ditches and application of various techniques associated with OMWM are used to restore the hydrology and ecological function of grid-ditched salt marshes. As restoration efforts are considered and

designed, careful consideration must be given to the nekton support function of ditches, as well as consideration of how other functions and processes (e.g. hydrology, sedimentation, biogeochemistry, vegetation, fauna) will respond to ditch removal. The ditches of the Fire Island Wilderness marshes support a diverse nekton community, although a different one from creeks. Since no species exclusively use the ditches as habitat, ditch removal would, from this perspective, likely not cause a decrease in the overall nekton species composition. However, if the Fire Island salt marsh ditches were filled, it is reasonable to think that the nekton would be displaced or significantly reduced in abundance, although the nekton carrying capacity of the existing creeks remains uncertain. Perhaps a good restoration approach would be to fill some ditches and leave others open to serve a nekton support function. Although not feasible at the federally designated Wilderness Area, another restoration scenario might be to fill the linear ditches, while also creating meandering tidal creeks and marsh pools, providing essential nekton habitat. This was recently done at a nearby salt marsh at Great South Bay, NY (Rochlin et al. 2009). Often, historic tidal channels that had been lost in response to grid ditching (Adamowicz & Roman 2005) can be relocated and restored.

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