Small creeks in a big lagoon: the importance of marginal habitats for fish populations

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Abstract

Temperate transitional water systems, as in the case of the Venice Lagoon, are characterised by many different shallow-water habitats. The availability of trophic resources and the low predator pressure make salt marshes one of the most important habitats for many fish species, both resident and marine migrant, but several anthropogenic pressures, erosion and relative sea level rise in particular, are causing a significant loss of this habitat. A part from natural habitats, in many small lagoon islands of the Venice lagoon, artificial creeks of different size and morphology are present, once used in traditional aquaculture activities or built up as defence lines. Aims of this study is to analyse and compare the structure and composition of fish communities inhabiting small-sized creeks, considering both the natural and artificial ones, in order to evaluate the ecological importance of these marginal habitats for fish populations. A particular attention was given to artificial sites, assessing their ecological value as alternative refuge habitats to natural salt marsh creeks. One year samplings conducted in four sites (two natural salt marshes and two artificial creeks) allowed to describe the local fish communities, which comprised 20 species overall. Influence of water parameters and habitat structure were considered in analysing the fish communities observed. In some cases, these habitats hosted high abundances of resident fish species listed in the Annex II of the Habitat Directive. Furthermore, juveniles of eight species of marine migrant fish were found, some of which are of economic importance. Results of this study underline the ecological importance of these marginal habitats for many fish species, of both conservation and economic importance, and thus a proper management and restoration strategy of these sites is needed to maintain their

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functionality and to buffer the disappearance of natural salt marshes.

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1. Introduction

Salt marshes constitute one of the most important habitats in temperate transitional water ecosystems (Kennish, 2002; Airoldi and Beck, 2007; Lowe and Peterson, 2014). They play several ecological roles for animal populations, and in particular for fish species (Mathieson et al., 2000; Kneib, 2000). Indeed these habitats host abundant populations of both estuarine resident and migrant species, with higher fish densities relative to adjacent unvegetated open-water habitats (Franco et al. 2006a, b). The ecological importance of these habitats is mainly due to the high level of trophic resources available and to the refuge function from predation deriving from a complex morphological structure (Rountree and Able, 2007). Furthermore, many fish species living in these areas are of conservation or commercial relevance (Franco et al., 2010, 2012). The importance of these habitats for aquatic fauna derives from their high spatial heterogeneity; salt marshes consist of a complex mosaic of microhabitats such as vegetated edges, subtidal and intertidal creeks, pools and ponds (Minello et al., 2003). Among them, salt marsh creeks in particular host high fish density (Desmond et al, 2000; Franco et al., 2006a, b).

In the last decades, many anthropic pressures determined a substantial loss of salt marsh habitats worldwide (Airoldi and Beck, 2007; Fagherazzi, 2013). Land claim, erosion, pollution, aquaculture and relative sea level rise determined the alteration or the complete destruction of these habitats. A major loss of salt marsh habitats occurred also within the Venice lagoon, the largest Mediterranean coastal lagoon, with salt marsh surface reduction from about 149 km^2 in 1912 to 37 km^2 in 2003 (Cucchini, 1928;

Silvestri et al., 2003), mostly due to the effects of anthropogenic-induced erosion (Sarretta et al., 2010). Furthermore, a strong reduction of the extent of salt marshes (Carniello et al., 2009) or even their complete disappearance (Cola et al., 2008) are predicted over the next 50 years. Even if in the last decades many salt marsh restoration activities have been carried out to counterbalance this negative trend (Carniello et al., 2009), some of the causes of marsh loss has not been, or cannot be, easily prevented. Subsidence and relative sea level rise have been addressed as among the major causes of salt marsh loss (Bock et al., 2012; Kirwan and Megonigal, 2013), particularly in situation with a lower income of sediments such as in the case of the Venice lagoon. For the Adriatic Sea, during the XX century, sea level rose at a rate of 1.3 mm y^{-1}, and by the year 2100 it could rise 14-49 cm (Scarascia and Lionello, 2013).

In the Venice lagoon, small inter/subtidal creeks a few meters wide only, with a mean depth of 0.5-1.5 m, may be also found within lagoon islands. They are mainly man-made artificial creeks, once used for traditional fish farming, as small marinas or as defence line during the two world wars.

While navigation channels are subjected to periodic maintenance and a certain degree of pollution and disturbance due to boat traffic, the other two types of artificial creeks are now mainly abandoned and partly renaturalised. These artificial habitats were built mostly during the XIX-XX centuries and can be divided into two main categories. The first one is composed of closed systems, once regulated by water gates but now mostly abandoned; in some cases, water infiltration by nearby channels allows for a partial water renewal. These habitats, as part of an integrated form of land use, are composed of a network of shallow water creeks used for fish farming, crossing through fields and orchards. In most of the cases, these traditional activities are now strongly diminished or completely abandoned, resulting in a renaturalisation of both terrestrial and aquatic habitats. A second group of artificial creeks comprises more open systems, mainly used as marinas or to defend military buildings. These creeks often present step stone banks and a greater depth, up to two meters, allowing navigation to small boats. Thus, for their morphology these are tide influenced habitats, but even during low tide a complete drainage of the water does not occur, allowing the permanence of fish within the creek. On the whole, Venice lagoon hosts about 100 km of these artificial creeks. Waltham and Connolly (2013) proposed the maintenance and restoration of artificial tidal lakes along the Gold Coasts in Queensland (Australia) as a process in some way inverse to land claim. Similarly, within the Venice lagoon, despite a simpler morphology, the network of artificial creeks could in part buffer the biodiversity loss due to salt marsh disappearance by providing suitable habitats for many fish species, even of conservation and commercial interest. Most of these artificial creeks are located in marginal and scarcely populated areas, thus avoiding impacts from the many anthropogenic pressures that affect other lagoon habitats, perhaps with the exception for those deriving from small-scale agricultural practices. Conversely, the lack of a regular control and maintaining of many sites determined, in some cases, a progressive burial due to sediment and detritus accumulation.

Considering the increasing loss rate of salt marshes, in a context of local and global pressures, this study focused on small artificial creeks within the Venice lagoon, to assess if these canals can be a habitat for fish fauna, even through the comparison with the most similar natural habitats, such as salt marsh creeks. Over a one-year period, the two main typologies of small-sized artificial creeks were investigated: one is a closed system, isolated from lagoon circulation, while the other one is an open creek, strongly influenced by sea water. Moreover, two natural salt marsh creeks were chosen as reference points and sampled at the same time, in order to: (1) investigate the ecological role of artificial creeks as habitat for fish fauna, (2) assess how the degree of connection with the open lagoon influence the fish community and (3) compare the fish fauna and the environmental conditions between artificial and natural habitats.

We expected that the differences in habitat structures between artificial and natural sites, especially as a result of the weak/absent tide regime in the former, would influence the composition of fish communities. In particular, the partial to complete isolation from lagoon open waters would increase the refuge function from aquatic predators typical of shallow water creeks, thus resulting in higher fish density in artificial sites. The assessment of environmental and fish community characteristics of artificial creeks, in comparison to natural salt marshes, is here provided as a baseline for the development of future management and restoration plans of these sites in the largest Mediterranean coastal lagoon, in order to buffer the biodiversity loss expected to occur as a consequence of natural salt marsh loss.

2. Materials and methods

The Venice lagoon is one of the largest lagoons along the coasts of the Mediterranean Sea, with a surface of about 540 km^2. It is a microtidal transitional water ecosystem, where tides can reach 1 m of excursion, characterised by wide extensions of shallow brackish water interrupted by a network of deeper channels and salt marsh habitats. Four sampling sites were chosen within the Venice lagoon (Figure 1): two creeks within natural salt marsh habitats (N1 in the northern basin and N2 in the southern basin) were used as reference sites and were compared to two artificial creeks (A1 and A2) within two islands of the lagoon. The natural sites comprised small-sized intertidal creeks (200-250 m long, 2-4 m width), with a maximum depth of 0.7 m, which completely drain during the low tide phase. Among the artificial aquatic habitats present within the Venice lagoons, two sites were chosen in order to represent two "extreme" situations.

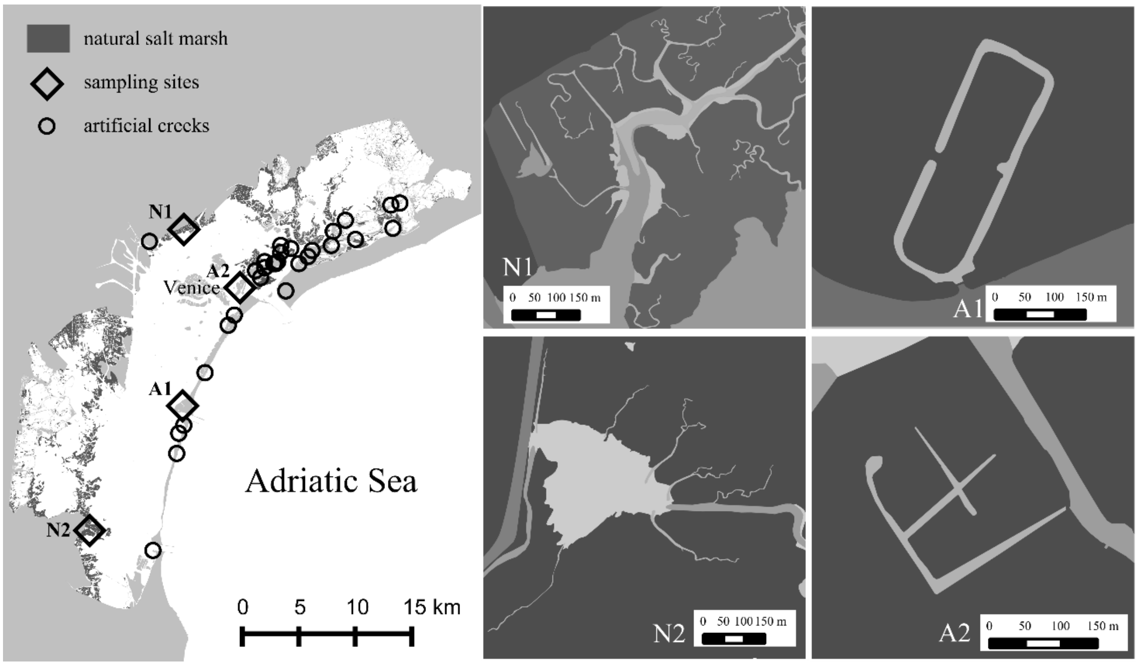

Figure 1. Map of the Venice lagoon showing the locations (left) and the different structural complexity (right) of the four sampling.

A1 was a ring-shaped ditch (about 700 m long, 8 m width) situated next to one of the lagoon sea-inlet and thus strongly influenced by seawater. A small connection allowed partial water exchanges with the lagoon, but a maximum depth of 2 m, together with the height of the connecting opening (about 1 m below the mean sea level), prevent from the complete drainage during low tide phase. A2 was a closed system of inland small-sized creeks (about 500 m long, 5 m width) isolated from lagoon waters, with a mean depth of 0.5 m and no tide excursion. In each sampling site, 11 sampling campaigns were performed from March 2010 to March 2011 with a beach seine net (8 m long, knot-to-knot distance of 2 mm). Three net tows were conducted during each campaign: in order to determine the explored area, length and width of each tow were measured. Sampled fish were photographed on millimeter paper and then released. Only when necessary, a representative subsample of fish was sacrificed with an excess of 2-phenoxyethanol and preserved in 8% buffered formaldeid. During each sampling event, the main chemico-physical water parameters were recorded: temperature (digital thermometer, ±0.1 °C), salinity (optical refractometer, ±1), dissolved oxygen (Winkler method, ±0.1 mg L^{-1} subsequently converted in percentage of saturation) and turbidity (portable nephelometer, ±0.1 ftu). In June three cores of sediment (Ø 3 cm) were collected in each site to determine the content of organic matter in the upper sediment layer (10 cm), estimated as loss on ignition (Loi 550).

Each specimen was identified to species level and density of each species was estimated by dividing the total abundance for the sampled surface. Each species was then assigned to a functional guild according to Franzoi et al. (2010). For data analysis, sampling dates were grouped on a seasonal basis (spring: March, April and May; summer: June, August and September; autumn: October and November; winter: December and February). To test for differences of composition and densities of fish communities among seasons and sites a two-way factorial ANOVA (followed by Tukey HSD post hoc test) was performed on total fish density, species richness, density of the two most abundant functional guilds (lagoon resident and marine migrant, which were the only regularly present in all sites with relevant densities) and density of the seven most abundant species, which represented more than 90% of the entire community. After a preliminary screening for homogeneity of variance and normality of residuals, all density data were log-transformed prior performing the ANOVA to meet variance assumptions.

The study of the fish community has been refined through the analysis of the β-diversity among sites, and its breakdown into the two components of turnover and nestedness-resultant (Baselga, 2012). We calculated the taxonomic and functional β-diversity on the presence/absence matrix of the fish community, using the R function provided in Villéger et al. (2013). To estimate the functional β-diversity we selected three traits based on the functional guilds adapted from the works of Franco et al. (2008) and Franzoi et al. (2010): the estuarine use (estuarine species, marine migrants, marine stragglers and freshwater species), the feeding mode (micro/macrobenthivores, hyperbenthivores, detritivores, herbivores and omnivores) and the reproductive mode (viviparous, oviparous with pelagic, benthic or adhesive eggs, oviparous guarders and oviparous shelterers).

A Principal Component Analysis (PCA) was performed on the environmental data collected in the four sampling sites, while to evaluate the influence of environmental parameters on the fish assemblages, a redundancy analysis (RDA, Legendre and Legendre 1998) was performed. For a set of response variables (i.e. fish density data) this method allows to quantify the amount of variance explained by a table of explanatory variables (i.e. the environmental parameters). RDA was performed on the Euclidean distance matrix of species densities after the Hellinger transformation of data (Rao, 1995); the first two canonical axes were considered to build a tri-plot of the constrained ordination of the response variables.

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3. Results

Overall, 20 species of fish belonging to 10 families were identified in the four sites (Table 1). Four functional guilds were assigned to the fish community: lagoon resident, marine migrant, freshwater species and marine occasional. The two most important guilds were the lagoon resident (nine species) and the marine migrant (seven species). Three species of lagoon resident fish sampled (Table 1) are of conservation interest: the south european toothcarp *Aphanius fasciatus* and the lagoon goby *Knipowitschia panizzae*, found with medium to high densities in all sites; the third species, the Canestrini's goby *Pomatoschistus canestrinii*, was present only with a few specimens in one of the two salt marsh creeks (N2). These species are listed within the Annex II of the European Directive 92/43, as "species of community interest, whose conservation requires the designation of special areas of conservation". Furthermore, two species of lagoon residents and six species of marine migrants were of commercial interest, mainly for the local traditional fisheries (Table 1).

Table 1. List of fish species sampled in the four sites and mean seasonal density in each site (Sp = spring, Su = summer, Au = autumn, Wi = winter). Functional guilds: LR = lagoon resident, MM = marine migrant, MO = marine occasional, FW = freshwater. Cons = species of conservation interest listed in Annex II of Directive 92/43/CEE, Comm = species of commercial interest. Light grey: less than 0.2 ind m^{-2}; medium grey: from 0.2 to 0.5 ind m^{-2}; dark grey: from 0.5 to 1 ind m^{-2}; black: over 1 ind m^{-2}.

Species	Code	Guild	Importance		natural - N1				natural - N2				artificial - A1				artificial - A2			
			Cons	Comm	Sp	Su	Au	Wi	Sp	Su	Au	Wi	Sp	Su	Au	Wi	Sp	Su	Au	Wi
Atherina boyeri	ABO	LR		X	▓	▓	░		░	▒	░		░	▒	░	▒				
Aphanius fasciatus	APFA	LR	X		▒	█		░	░	▓	░	░	░	▓	░	░	█	█	▓	▓
Chelon labrosus	CLA	MM		X									░	░						
Gobius cobitis	GCO	MO											░	░	░					
Gambusia holbrooki	GHO	FW				░				░							▓	█	█	▓
Knipowitschia panizzae	KPA	LR	X		░	░	░		░	░	░	░	░	░	░		▓	█	▓	░
Liza aurata	LAU	MM		X	░	░	░		░				▒	░	▒	░				
Liza ramada	LRA	MM		X	▒	░							░	░						
Liza saliens	LSA	MM			░	▒	▒		░	░	░	░	▒	▓	▒	░				
Nerophis ophidion	NOP	LR											░	░						
Pomatoschistus canestrinii	PCA	LR	X								░									
Platichthys flesus	PFL	MM		X					░											
Pomatoschistus minutus	PMI	MM		X																
Pseudorasbora parva	PPA	FW								░										
Parablennius sanguinolentus	PSA	MO											░	░						
Syngnathus abaster	SAB	LR										░	░	▓	░	░				
Sparus aurata	SAU	MM		X	░								░							
Salaria pavo	SPA	LR											░	░	░					
Syngnathus typhle	STY	LR											░	░	░					
Zosterisessor ophiocephalus	ZOP	LR		X									░	░						

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ANOVA found significant differences among sites and among seasons for species richness and total density. In the two natural creeks these two variables showed similar values (Figure 3), except for the number of species during winter, when only one species (*A. fasciatus*) was found in N1. Conversely, the two artificial creeks differentiated from natural sites, particularly during summer, when one site (A2) showed significantly higher fish densities, mainly due to the recruitment of *A. fasciatus* and *Gambusia holbrooki*, while the other (A1) hosted the most diverse fish community, with a peak of 11 species observed in August (figure 3). Differences among sites decreased during winter, with all sites showing lower densities and a lower number of species. Considering the two most abundant guilds, ANOVA found in all sites significant seasonal differences for lagoon resident densities, which were higher in summer relative to the other seasons, while for marine migrants differences among sites were found.

Table 2. Results of ANOVA (mean sq.) performed on the four variables describing the fish community and the mean density of the seven most abundant species. Asterisks marks significant effects for $P < 0.05$

	Site	Season	Site X Season	Residuals
d.f.	3	3	9	28
Species richness	24.527*	15.120*	3.366*	1.009
Fish density	2.355*	3.234*	0.327	0.498
Lagoon resident	0.743	2.688*	0.226	0.378
Marine migrant	4.815*	0.731	0.197	0.356
Atherina boyeri	1.855*	2.116*	0.503	0.318
Aphanius fasciatus	1.128	2.525*	0.486	0.442
Gambusia holbrooki	11.594*	0.934*	0.672*	0.281
Knipowitschia panizzae	2.167*	1.019*	0.421*	0.185
Liza aurata	1.892*	0.191	0.181	0.146
Liza ramada	0.215	0.571	0.177	0.281
Liza saliens	2.546*	1.295*	0.243	0.312

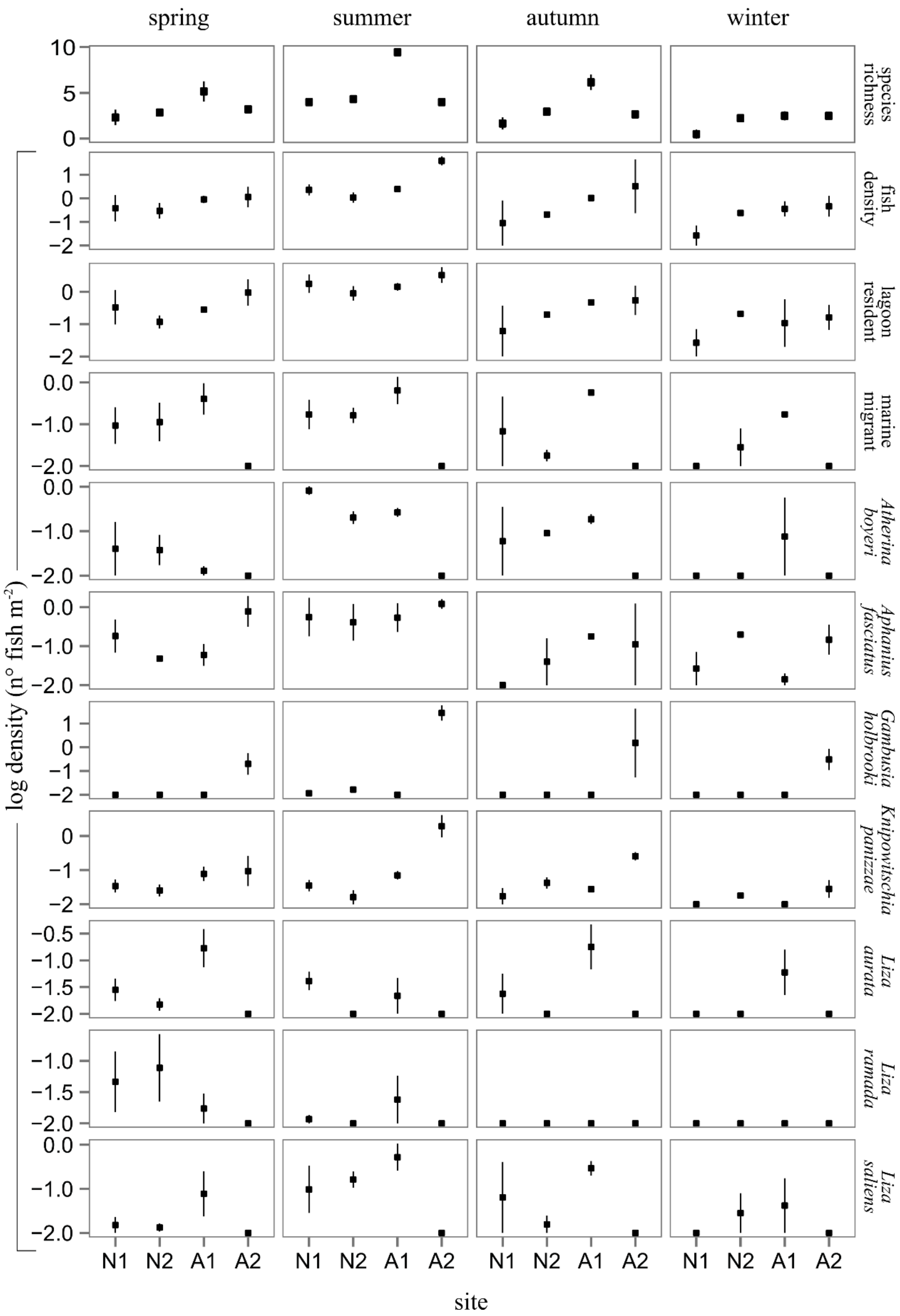

Figure 3. Characterisation of the fish community in the four sampling sites (N1, N2, A1, A2) on a seasonal basis. All values are mean ± S.E.

Marine migrants showed significantly higher densities in the artificial creek A1 relative to other sites, while it was completely absent in the closed artificial site A2. Among the 20 species identified, a pool of seven species represented more than the 98% of the total density (Table 1). ANOVA performed on these seven species (Table 2) showed significant differences for both factors considered except for the juveniles of one species (*Liza ramada*). The other two species of mullets (*L. aurata* and *L. saliens*) were found with significantly higher juvenile densities in A1 relative to the other sites, except for *L. aurata* in summer, when a higher density was recorded in N1. For this species no differences were found among seasons, while *L. saliens* was significantly more abundant in summer relative to spring and winter. The sand smelt *Atherina boyeri* showed the same seasonal pattern, and all these four species (the three mullets and the sand smelt) were completely absent from site A2 (Table 1). This site showed, on the other hand, significantly higher densities of *K. panizzae* relative to other sites, particularly during summer, when also the highest densities of *A. fasciatus* were found in all the four stations considered. In this creek, also the mosquitofish *G. holbrooki*, a freshwater species, was extremely frequent and abundant. It showed marked seasonal fluctuations in density, with significantly higher values observed in summer relative to spring and winter, while it was only an occasional caught in the natural salt marsh during summer.

The analysis of β-diversity among the four sites showed how the number of unique species in the four sites nearly doubled the number of shared species (Table 3). On average, similar levels of taxonomic and functional diversity were found (0.65 and 0.74

respectively), with the turnover being lower than the nestedness-resultant component (0.26 vs 0.40 for the taxonomic diversity, 0.31 vs 0.43 for the functional diversity).

Table 3. Summary of the taxonomic and functional β-diversity and its two components, turnover and nestedness (mean±S.D., range in parenthesis) for the four sites.

	Taxonomic		Functional	
β-diversity	0.65±0.14	(0.46-0.88)	0.74±0.32	(0.18-0.99)
Turnover	0.26±0.25	(0-0.59)	0.31±0.33	(0-0.64)
Nestedness	0.40±0.25	(0.06-0.75)	0.43±0.45	(0-0.99)
Shared richness	4.83±2.40	(2-7)	0.04±0.05	(0-0.12)
Unique richness	9.33±3.61	(5-14)	0.11±0.04	(0.03-0.14)

Considering the pairwise comparisons among the four sites (Table 4), the highest taxonomic diversity (0.88) has been found between the two artificial sites, with a species turnover contribute of 0.50. Considering the functional diversity, site A2 clearly differed from the other three sites, even if also A1 showed high diversity values relative to the natural creeks (0.63-0.64). The lowest taxonomic (0.46) and functional (0.18) β-diversities were found between the two natural sites. Comparing artificial versus natural creeks, the taxonomic diversities between the two habitat types (artificial vs natural) were comparable (0.56-0.75), even if for A2 they were due only to the nestedness-resultant component, while for A1 there was a significant contribute of the taxonomic diversity, particularly in the comparison A1-N2. Also for the functional β-diversity, the two artificial creeks showed a different pattern: the functional diversity between A1 and the two natural sites ascribed nearly only to the turnover component, while the strong differences between A2 and the two natural sites derived exclusively from the nestedness-resultant component.

Table 4. Pairwise comparisons of the taxonomic and functional β-diversity for the four sites

	A1-N1	A1-N2	A1-A2	N1-N2	N1-A2	N2-A2
Taxonomic						
β-diversity	0.56	0.65	0.88	0.46	0.63	0.75
Turnover	0.22	0.59	0.50	0.22	0.00	0.00
Nestedness	0.34	0.06	0.38	0.24	0.63	0.75
Shared richness	7	7	2	7	3	3
Unique richness	9	13	14	6	5	9
Functional						
β-diversity	0.63	0.64	0.99	0.18	0.99	0.99
Turnover	0.57	0.64	0.63	0.02	0.00	0.00
Nestedness	0.06	0.00	0.36	0.16	0.99	0.99
Shared richness	0.07	0.08	0.00	0.12	0.00	0.00
Unique richness	0.12	0.13	0.14	0.03	0.12	0.14

Principal component analysis of environmental data (Appendix A) did not show marked differences among the four sites. Anyway, a spatial gradient could be recognized along the first axis, which explains 30% of the overall variance. Higher values of salinity and dissolved oxygen concentration, coupled with a lower water turbidity, separate A1 from the other three sites. From this picture, the environmental characteristics of A2 showed to be comparable with those observed in the two salt marshes, which clustered together despite the geographical distance. Another 23% of variance is explained by the second axis, mainly due to seasonal variations in environmental parameters, in particular water temperature.

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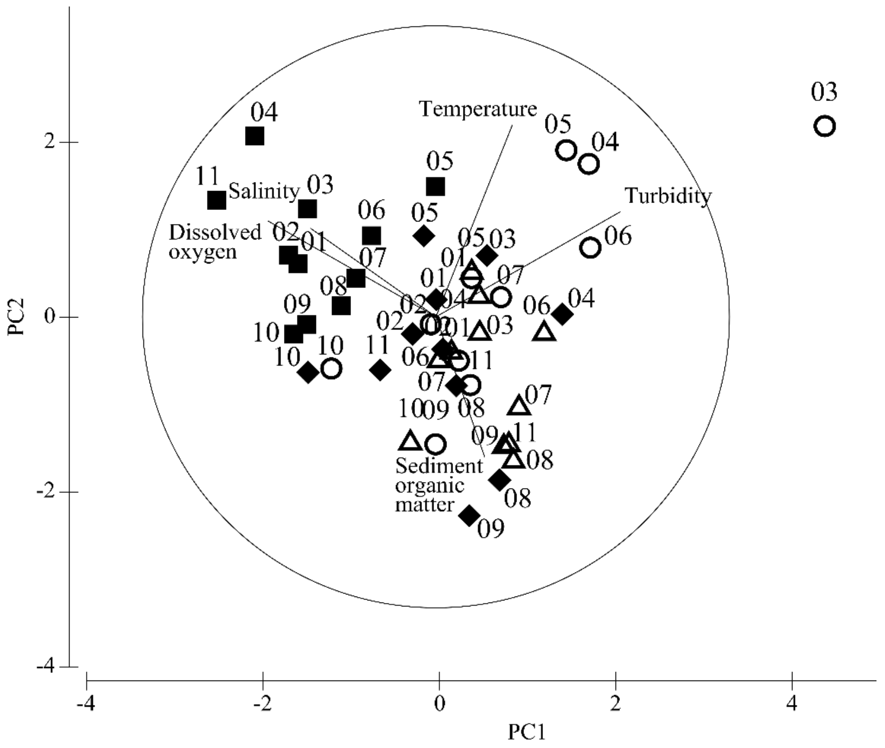

Figure 4. Ordination of the samples collected in the four sites according to the results of Principal Component Analysis (PCA). N1 = open triangles, N2 = open circles, A1 = solid squares, A2 = solid diamonds. Numbers indicate the sampling dates.

Samples ordination by means of redundancy analysis (Figure 4) explained 25% of total variance. Most of this variance (82%) is explained by the environmental variables measured, considering the first two axes. The analysis ordered the samples into three main groups, characterised by a different composition of the fish community. Along the first axis, which account for 60% of variability, A2 separated from the other sites for the high densities of *G. holbrooki*, without showing a substantial variability along the second axis. The second axis (explaining 22% of the variance) showed a marked seasonal variation of the other four environmental parameters for the two natural creeks, characterised by turbid waters and influenced by freshwater inputs from the mainland, and for A1, characterised by high densities of both *L. aurata* - *L. saliens* juveniles and by a strong influence of sea water, which resulted in higher values of salinity and dissolved oxygen.

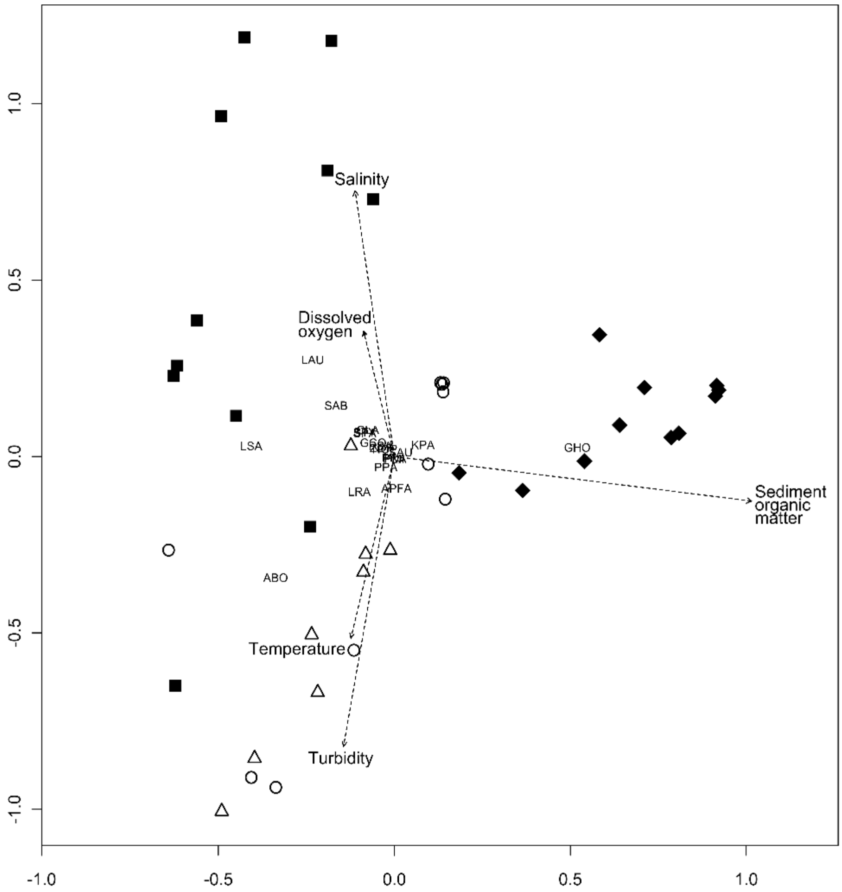

Figure 4. Ordination of the samples collected in the four sites according to the results of Redundancy Analysis. N1 = open triangles, N2 = open circles, A1 = solid squares, A2 = solid diamonds. See table 1 for species abbreviations.

The analyses highlighted some differences among the four sites, regarding the environmental conditions and the taxonomic/functional traits of the fish community. In particular, A1 with more marine environmental conditions clearly differentiated from the close system A2 in the principal component analysis. Indeed, environmental parameters registered in A2 were similar to those characterising the two salt marshes. Furthermore, the second multivariate analysis showed a stronger separation of the sampling sites, due to the differences in the species composition of the fish community. As in the PCA, the RDA showed a clear separation of A1 relative to the other three more confined creeks for the more diverse fish species hosted, as suggested also by the β-diversity analysis. Conversely, the isolated cluster of A2 may derive from the simplified fish community, that showed taxonomic and functional differences from the other three sites. Despite the differences with the two natural creeks, both artificial sites hosted high densities of resident species and, in the case of A1, also of marine migrants, in some cases even higher than in salt marshes.

4. Discussion

Results of the present study showed that the artificial creeks here selected, and compared with two natural salt marshes, hosted high densities of estuarine fish species in one site (A2), and, in the other one (A1), a diversified and well-structured fish community composed also of an abundant marine migrant component.

Precedent studies on the Venice lagoon considered many salt marsh habitats, but overlooked the smallest-sized creeks, that revealed to host an abundant and diversified fish community. In these small creeks high densities of resident fish species were observed, such as *A. boyeri* and *A. fasciatus*, and, to a lesser extent, of two marine migrant, *L. ramada* and *L. saliens*. The values registered for these species in the present study revealed to be comparable to those reported by Franco et al. (2006a, b) for larger creeks. However the resident *A. fasciatus* and the marine migrant *L. saliens*, showed, in the small creeks analysed in this work, a mean density one order of magnitude greater.

Two of the major indicators that are considered important in habitat selection by small and juvenile fish are risk of predation and foraging profitability (Werner and Hall, 1976; Holbrook and Schmitt, 1984; Schmitt and Holbrook, 1985). Indeed, the shallow water of these systems plays an important refuge function for small fish to avoid large aquatic predators (Rozas and Odum, 1988; Ruiz et al., 1993; Paterson and Whitfield, 2000). It is also well known that salt marsh habitats provide high trophic resources in a competition-limited habitat (Koutsogiannopoulou and Wilson, 2007; Maci and Basset, 2009), due to the high productivity and the low number of species able to afford the physiological stress deriving from the extremely variable physico-chemical water characteristics (Elliott et al., 2007). The very small size of the creeks sampled during this work may enhance these ecological functions, further preventing the access to predator or other competitor fish, making them particularly suitable for small fish, both resident and juvenile migrant.

The two natural salt marsh creeks were compared to two artificial creeks with opposite characteristics: a sea-influenced creek (A1) versus a completely closed system (A2). Both sites showed a simpler morphology relative to salt marsh creeks. Furthermore, the cyclic drainage occurring in natural habitats during low tide was partially (A1) or totally (A2) altered by physical barriers. Allen et al. (2007) found a strong relationship between nekton densities and certain hydro-geomorphological features. In particular, small-sized, slow flowing creeks supported the highest nekton densities. This could also be the case of artificial habitats, where the absent/altered connection with bigger channels and open lagoon waters may turn them into elective habitats for fish fauna. Natural intertidal creeks offer refuge from predator and the access to trophic resources only temporarily, when submerged by water during the high tide phase. On the contrary, small artificial creeks may complement the ecological functions offered by natural habitats since they never drain completely, thus allowing the permanence of fish even during the low tide phase. Indeed, despite the lack of structural complexity, the highest values of species diversity were found in artificial site A1. The seaward position alone did not explain the values of taxonomic and functional diversity registered here relative to the other three sites, since only two marine straggler species were sampled: the rest of the fish community was formed mainly by resident and marine migrant species usually found with higher densities in inner lagoon waters. Conversely, tidal flow restriction would determine the absence, or significant low abundances, of some species, with a reduction in richness and a change in composition of fish and decapods assemblages of tidal creeks, particularly of marine species (Boys et al., 2012). This would be the case of the other artificial site (A2). Its complete isolation prevented the access to the creeks by juvenile marine migrant and, at the same time, of aquatic predators. Indeed, this site showed the highest values of both taxonomic and functional β-diversity relative to the other three sites, hosting a few species that form a subset of the typical salt marsh fish community. This peculiar situation allowed two resident species (*A. fasciatus* and *K. panizzae*) to reach the highest densities registered, even in the presence of a possible competitor such as the alloctonous mosquitofish *G. holbrooki*. This is of particular relevance since these two resident species are listed in the Annex II of Council Directive 92/43/EEC. Even if locally widespread and abundant, these species, together with *P.*

canestrinii, “requires the designation of special areas of conservation”, due to the close ecological link with endangered habitats such as salt marsh (Franco et al., 2012). Thus these artificial creeks, despite the lack of proper maintenance since the end of fish farming activities, could play an ecological role as biological reservoir for some small fish species of conservation interest. In the study of Havens et al. (1995), total fish and commercial fish populations showed lower densities in constructed marsh relative to natural wetlands and observed differences between natural and artificial salt marsh often derive from the young age of artificial sites (Minello and Zimmerman, 1992; Minello and Webb, 1997; Larkin et al., 2009). In the present study, we do not have a time series long enough to assess the temporal evolution of the artificial sites. Anyway, the two creeks explored were not newly created, but they were consolidated systems, that had the time necessary to reach an ecological stability and functionality, thus being able to host well-structured fish populations (Cavraro et al., 2013, 2014).

Despite the similarities found between natural and artificial sites, the simplified morphology, the relative height of the banks and, in one case (A2), the complete lack of tide cycle could have altered the functioning of artificial systems. As expected, the last factor, in particular, proved to be the most important in shaping the fish community, as testified by the high values of β-diversity observed in A2 respect to the other sites. Salt marshes are well known as dynamic and complex systems, characterised by a high spatial and temporal variability in abiotic parameters. Thus, the ecological processes occurring can be influenced even by slight variations in habitat heterogeneity (Adam, 2002; Larkin et al., 2008). For example, some authors (Vivian-Smith, 1997; Larkin et al., 2006) compared created and natural pools within salt marshes, underlying the importance of morphology in influencing algal, invertebrate and fish responses. Most of all, the different topography and tidal regime of artificial sites may influence the behaviour of fish species by preventing marsh access during high tides. McIvor and Odum (1988) observed the importance of creek sinuosity, channel depth and bank stability in influencing fish utilisation of salt marsh creeks and flooded surface. The possibility to use the flooded marsh surface is of particular relevance for some fish species, in particular killifish. West and Zedler (2000) found that *Fundulus parvipinnis* accessing to the flooded marsh surface fed more different prey items relative to fish in the subtidal channels. Furthermore, Madon et al. (2001) underlined the crucial role of food resources, deriving from the temporary access to the marsh surface, in influencing bioenergetics and reproductive traits for the same fish species. Even if the altered hydrological regime may influence some ecological processes in artificial sites, resident populations showed to be able to manage this situation. This is the case, for example, of *A. fasciatus*, which could be considered, for the Mediterranean basin, ecologically equivalent to North American fundulids. Precedent studies conducted in the same sampling stations (Cavraro et al., 2013) showed a higher secondary production of this species in the artificial sites relative to the natural ones. In particular, the highest productivity was found exactly where the effect of tide cycles was absent (A2). In this site, the complete isolation from lagoon open waters excluded the presence of aquatic predators and shaped a fish community composed by only three species. This situation probably favoured mechanisms of resource partitioning among the few species present, thus resulting in plenty of resources for resident fish in a predator-free habitat. If it is so, the access to marsh surface would be more relevant within a diversified fish community in natural conditions, where a significant tide range let some species (i.e. killifish) to avoid inter-specific competition and predation by moving to the vegetated inundated marsh surface and using them as feeding ground (Baltz et al., 1993; Rountree and Able, 2007).

This study underlined the ecological importance of small-sized canalisation for fish fauna, highlighting significant differences in composition and density of fish community among the natural and artificial shallow water creeks analysed. In particular, also artificial sites proved to host relevant fish densities, depending on the level of connection with lagoon waters.

These habitats are mainly located in the Central and Northern part of the Venice lagoon (Figure 5), and most of them result to be near to the main deep channels directly connected with the sea inlets. Likely, open artificial habitats would present environmental characteristics and fish assemblages similar to those found in A1. Conversely, close artificial systems would be probably more similar to A2 independently from their geographical location. On the other hand, in many cases artificial sites lack of maintenance, which could cause a general habitat degradation. Close systems in particular can be subjected to water quality deterioration or landfilling of the creeks due to the accumulation of detritus and sediment, becoming unsuitable for the fish community. Another threat to close artificial creeks derives from the salinity reduction due to superficial run-off. As observed in S. Erasmo island (Northern basin of the Venice lagoon) this process determines a freshening of the creeks, with a consequent substitution of *A. fasciatus* by *G. holbrooki* (Cavraro, pers. obs.). Superficial run-off can

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also enrich creeks water with chemicals and nutrients from the nearby fields, even if only small-scale agricultural activities are carried out in the area.

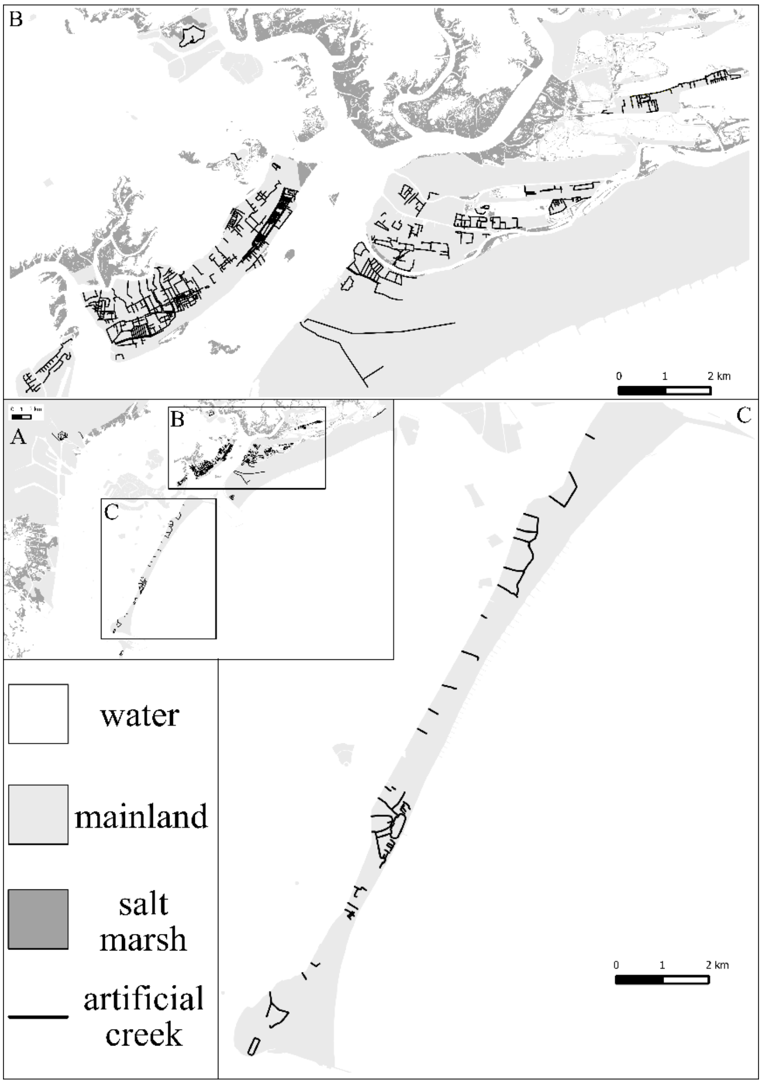

Figure 5. The network of the artificial creeks in the Venice lagoon (A); focus on the northern basin (B); focus on Lido island (C).

In the future, an exhaustive census of these sites would be desirable, in order to quantify the overall contribution to fish population at a lagoon scale and to assess their conservation status. Information about the ecological role of these habitats and their functioning provided by this study could constitute a first baseline in planning restoration and management activities, aimed to maintain or enhance the hosted fish populations. Indeed, these sites, both natural and artificial, are threatened, even if by different pressures. Worldwide, salt marsh extension decreased during the XX century, due to land-claim, erosion and sea level rise. This was also the case of the Venice lagoon, where a loss of about three quarter of marsh surface occurred in the last century. In the past 20 years many restoration activities have been carried out within the Venice lagoon, leading to the construction of several square km of artificial marshes over existing mudflats. These works were mainly focused on morphological and hydrodynamic aspects of the lagoon ecosystem, thus often overlooking the restoration of salt marsh considering the ecological point of view. The main biases in this kind of works were the use of sandy sediments, deriving from channel dredging (D'Alpaos et al., 2007), and the lack of morphological complexity, since the complex network of different microhabitats characterising natural salt marsh were rarely recreated. Results of the present study seem to highlight, in particular, the importance of small-sized canalisation for fish fauna. Anyway, further data should be collected to understand the key morphological and structural elements that characterise small-size creeks and that make them suitable for fish fauna. The acquisition of such information would be of particular interest in the context of planning new artificial salt marshes. On the other hand, these data would help specific interventions on artificial habitats, in order to restore their full ecological potential for fish fauna.

5. Conclusions

Many actions have been taken to counterbalance the loss of salt marsh habitats occurred in the last decades along temperate coasts worldwide. Since the processes involved are numerous and acting on different space and time scales, it is often not possible to eliminate the causes. Restoration activities on degraded wetlands, or creation of new habitats, became a widespread approach, with many studies focusing on the results of these programmes by comparison with natural systems (Havens et al., 2002; Raposa, 2002; Larkin et al., 2009; Boys and Williams, 2012). Also within the Venice lagoon salt marsh extension dramatically decreased in the last century, while many man-made creeks within lagoon islands were abandoned and could now contribute to the maintaining of lagoon fish populations. Starting from the results presented in this study, further efforts, such as a detailed census of all these artificial systems, would provide a complete picture of the status of these marginal systems, in order to maintain or enhance their ecological values. Anyway, data collected showed high densities of juvenile marine migrant and lagoon resident fish species in two examples of these artificial habitats, with species richness and density, in some cases, higher than in natural salt marsh systems. Furthermore, the fish communities studied hosted species of some conservation and economic value as well.

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References

Adam P., 2002. Saltmarshes in a time of change. Environmental Conservation, 26: 39-61.

Airoldi L. and Beck M.W., 2007. Loss, status and trends for coastal marine habitats of Europe. Oceanography and Marine Biology: An Annual Review, 45: 345-405.

Allen D.M., Haertel-Borer S.S., Milan B.J., Bushek D., Dame R.F., 2007. Geomorphological determinants of nekton use of intertidal salt marsh creeks. Marine Ecology Progress Series, 329: 57–71.

Baltz D.M., Rakocinski C., Fleeger J.W., 1993. Microhabitat use by marsh-edge fishes in a Louisiana estuary. Environmental Biology of Fishes, 36: 109-126.

Baselga A., 2012. The relationship between species replacement and dissimilarity derived from turnover and nestedness. Global Ecology and Biogeography, 9: 134-143.

Bock Y., Wdowinski S., Ferretti A., Novali F. and Fumagalli A., 2012. Recent subsidence of the Venice Lagoon from continuous GPS and interferometric synthetic aperture radar. Geochemistry, Geophysics, Geosystems, 13, Q03023.

Boys C.A. and Williams R.J., 2012. Succession of fish and crustacean assemblages following reinstatement of tidal flow in a temperate coastal wetland. Ecological Engineering, 49: 221-232.

Boys C.A., Frederieke J.K., Glasby T.M. and Wilkinson K., 2012. Improved fish and crustacean passage in tidal creeks following floodgate remediation. Journal of Applied Ecology, 49: 223-233.

Published version: Ecological Engineering 99 (2017) 228–237
DOI: 10.1016/j.ecoleng.2016.11.045

Carniello L., Defina A. and D'Alpaos L., 2009. Morphological evolution of the Venice lagoon: Evidence from the past and trend for the future. Journal of Geophysical Research, 114: 1–10.

Cavraro F., Torricelli P., Franzoi P. and Malavasi S., 2013. Productivity in natural and artificial habitats in brackish water systems: an example from *Aphanius fasciatus* populations. Transitional Waters Bulletin, 7: 23-31.

Cavraro F., Daouti I., Leonardos I., Torricelli P. and Malavasi S., 2014. Linking habitat structure to life history strategy: Insights from a Mediterranean killifish. Journal of Sea Research, 85: 205-213.

Cola S., Sanavia L., Simonini P., Schrefler B.A., 2008. Coupled thermohydromechanical analysis of Venice lagoon salt marshes. Water Resource Research, 44: 1-16.

Cucchini, E., 1928. Le Acque dolci che si versano in Laguna di Venezia, Ufficio Idrografico del Magistrato alle Acque. Venezia.

Desmond J.S., Zedler J.B. e G.D. Williams, 2000. Fish use of tidal creek habitats in two southern California salt marshes. Ecological Engineering, 14: 233-252.

D'Alpaos A., Lanzoni S., Marani M., Bonometto A., Cecconi G., Rinaldo A., 2007. Spontaneous tidal network formation within a constructed salt marsh: Observations and morphodynamic modelling. Geomorphology, 91: 186-197.

Elliott M, Whitfield A.K., Potter I.C., Blaber S.J.M., Cyrus D.P., Nordlie F.G. and Harrison T.D., 2007. The guild approach to categorizing estuarine fish assemblages: a global review. Fish and Fisheries, 8: 241-268.

Fagherazzi S., 2013. The ephemeral life of a salt marsh. Geology 41: 943-944.

Published version: Ecological Engineering 99 (2017) 228–237
DOI: 10.1016/j.ecoleng.2016.11.045

Franco A, Franzoi P, Malavasi S, Riccato F & Torricelli P, 2006a. Fish assemblages in different shallow water habitats of the Venice Lagoon. Hydrobiologia, 555: 159-174.

Franco A., Franzoi P., Malavasi S., Riccato F., Torricelli P. and Mainardi D., 2006b. Use of shallow water habitats by fish assemblages in a Mediterranean coastal lagoon. Estuarine, Coastal and Shelf Science, 66: 67-83.

Franco A., Elliott M., Franzoi P. and Torricelli P., 2008. Life strategies of fishes in European estuaries: the functional guild approach. Marine Ecology Progress Series, 354: 219-228.

Franco A., Fiorin R., Zucchetta M., Torricelli P. and Franzoi P., 2010. Flounder growth and production as indicators of the nursery value of marsh habitats in a Mediterranean lagoon. Journal of Sea Research, 64, 457-464.

Franco A., Franzoi P., Malavasi S., Zucchetta M., Torricelli P., 2012. Population and habitat status of two endemic sand gobies in lagoon marshes - Impliations for conservation. Estuarine, Coastal and Shelf Science, 114: 31-40.

Franzoi, P., Franco, A., Torricelli, P., 2010. Fish assemblage diversity and dynamics in the Venice lagoon. Rend Fis Lincei 21, 269-281.

Havens K.J., Varnell L.M. and Bradshaw J.G., 1995. An assessment of ecological conditions in a constructed tidal marsh and two natural reference tidal marshes in coastal Virginia. Ecological Engineering, 4: 117-141.

Havens K.J., Varnell L.M. and Watts B.D., 2002. Maturation of a constructed tidal marsh relative to two natural reference tidal marshes over 12 years. Ecological Engineering, 18: 305-315.

Holbrook S.J. and Schmitt R.J., 1984. Experimental analysis of patch selection by foraging black surfperch (*Embiotoca jacksoni* Agazzi). Journal of Experimental Marine Biology and Ecology, 79: 39-64.

Kennish M.J., 2002. Environmental threats and environmental future of estuaries. Environmental Conservation, 29: 78-107.

Kirwan M.L. and Megonigal J.P., 2013. Tidal wetland stability in the face of human impacts and sea-level rise. Nature, 504: 53-60.

Kneib R.T., 2000. Salt marsh ecoscapes and production transfers by estuarine nekton in the Southeastern United States. In: Concepts and controversies in tidal marsh ecology. Eds: Weistein M.P. and Kreeger D.A., Springer, pp. 267-291.

Koutsogiannopoulou V. and Wilson J.G., 2007. The fish assemblage of the intertidal salt marsh creeks in North Bull Island, Dublin Bay: seasonal and tidal changes in composition, distribution and abundance. Hydrobiologia, 588: 213-224.

Larkin D.J., Vivian-Smith G., Zedler J.B., 2006. Topographic heterogeneity theory and ecological restoration. In: Falk D.A., Palmer M.A., Zedler J.B. (eds) Foundation of Restoration Ecology. Island Press, Washington, DC, pp 142-164.

Larkin D.J., Madon S.P., West J.M., Zedler J.B., 2008. Topographic heterogeneity influences fish use of an experimentally restored tidal marsh. Ecological Application, 18: 483-496.

Larkin D.J., West J.M. and Zedler J.B., 2009. Created pools and food availability for fishes in a restored salt marsh. Ecological Engineering, 35: 65-74.

Legendre P. and Legendre L., 1998. Numerical ecology. Elsevier Science, Amsterdam.

Lowe M.R. and Peterson M.S., 2014. Body condition and foraging patterns of nekton from salt marsh habitats arrayed along a gradient of urbanization. Estuaries and Coasts, 38: 800-812.

Maci S. and Basset A., 2009. Composition, structural characteristics and temporal assemblages in non-tidal Mediterranean lagoons: a case study. Estuarine, Coastal and Shelf Science, 83: 602-612.

Madon S.P., Williams G.D., West J.M., Zedler J.B., 2001. The importance of marsh access to growth of the California killifish, *Fundulus parvipinnis*, evaluated through bioenergetics modelling. Ecological Modelling, 136: 149-165.

Mathieson S., Cattrijsse A., Costa M.J., Drake P., Elliott M., Gardner J., Marchand J., 2000. Fish assemblages of European tidal marshes: a comparison based on species, families and functional guilds. Mar Ecol Progr Ser, 204:225-242.

McIvor C.C. and Odum W.E., 1988. Food, predation risk, and microhabitat selection in a marsh fish assemblage. Ecology, 69: 1341-1351.

Minello T.J. and Zimmerman R.J., 1992. Utilization of natural and transplanted Texas salt marshes by fish and decapod crustaceans. Marine Ecology Progress Series, 90: 273-273.

Minello T.J. and Webb J.W., 1997. Use of natural and created *Spartina alterniflora* salt marshes by fishery species and other aquatic fauna in Galveston Bay, Texas, USA. Marine Ecology Progress Series 151: 165-179.

Minello T.J., Able K.W., Weinstein M.P. and Hays C.G., 2003. Salt marshes as nurseries for nekton: testing hypothesis on density, growth and survival through meta-analysis. Marine Ecology Progress Series, 246: 39-59.

Paterson A.W. and Whitfield A.K., 2000. Do shallow-water habitats function as refugia for juveniles fishes? Estuarine Coastal and Shelf Science, 51: 359-364.

Rao C.R., 1995 A review of canonical coordinates and an alternative to correspondence analysis using Hellinger distance. Qüestiió, 19: 23-63.

Raposa K., 2002. Early responses of fishes and crustaceans to restoration of a tidally restricted New England salt marsh. Restoration Ecology, 10: 665-676.

Rountree R.A. and Able K.W., 2007. Spatial and temporal habitat use patterns for salt marsh nekton: implications for ecological functions. Aquatic Ecology, 41: 25-45.

Rozas L.P. and Odum W.E., 1988. Food, predation risk, and microhabitat selection in a marsh fish assemblage. Ecology, 69: 1341-1351.

Ruiz G.M., Hines A.H. ad Posey M.H., 1993. Shallow water as a refuge habitat for fish and crustaceans in non-vegetated estuaries: an example from Chesapeake Bay. Marine Ecology Progress Series, 99: 1-16.

Sarretta A., Pillon S., Molinaroli E., Guerzoni S. and Fontolan G., 2010. Sediment budget in the Lagoon of Venice, Italy. Continental Shelf Research 30: 934–949.

Scarascia L. and Lionello P., 2013. Global and regional factors contributing to the past and future sea level rise in the Northern Adriatic Sea. Global and Planetary Change, 106: 51-63

Silvestri, S., Marani, M., Marani, A., 2003. Hyperspectral remote sensing of salt marsh vegetation, Morphology and soil topography. Phys. Chem. Earth 28, 15–25.

Schmitt R.J. and Holbrook S.J. 1985. Patch selection by juvenile black surfperch (Embiotocidae) under variable risk: interactive influence of food quality and structural complexity. Journal of Experimental Marine Biology and Ecology, 85: 269-285.

Villeger S., Grenouillet G. and Brosse S., 2013. Decomposing functional β-diversity reveals that low functional β-diversity is driven by low functional turnover in European fish assemblages. Global Ecology and Biogeography, 22: 671-681.

Vivian-Smith G., 1997. Microtopographic heterogeneity and floristic diversity in experimental wetland communities. Journal of Ecology, 85: 71-82.

Waltham N.J. and Connolly R.M., 2013. Artificial tidal lakes: Built for humans, home for fish. Ecological Engineering, 60: 414-420.

Werner E.E. and Hall D.J. 1976. Niche shifts in sunfishes: experimental evidence and significance. Science 191: 404-406.

West J.M. and Zedler J.B., 2000. Marsh-creek connectivity: fish use of a tidal salt marsh in southern California. Estuaries, 23: 699-710.

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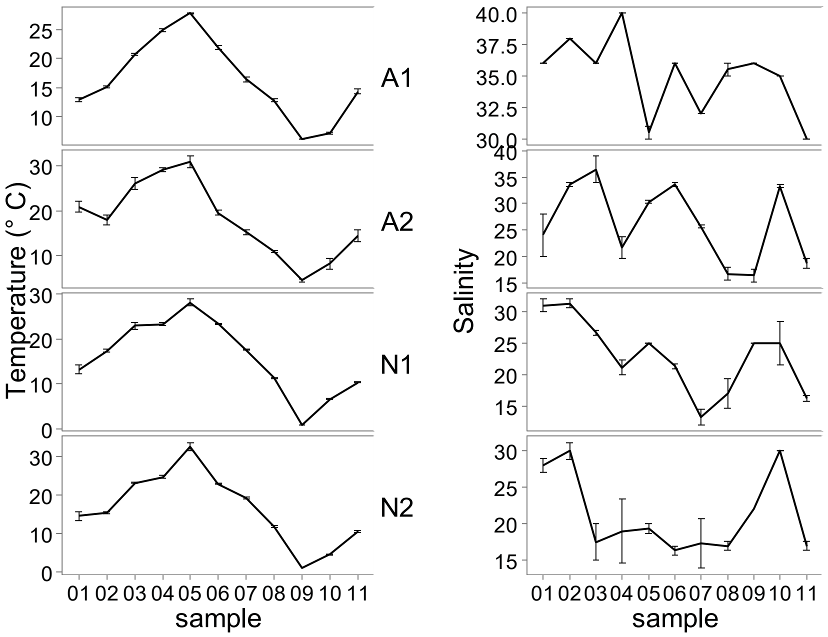
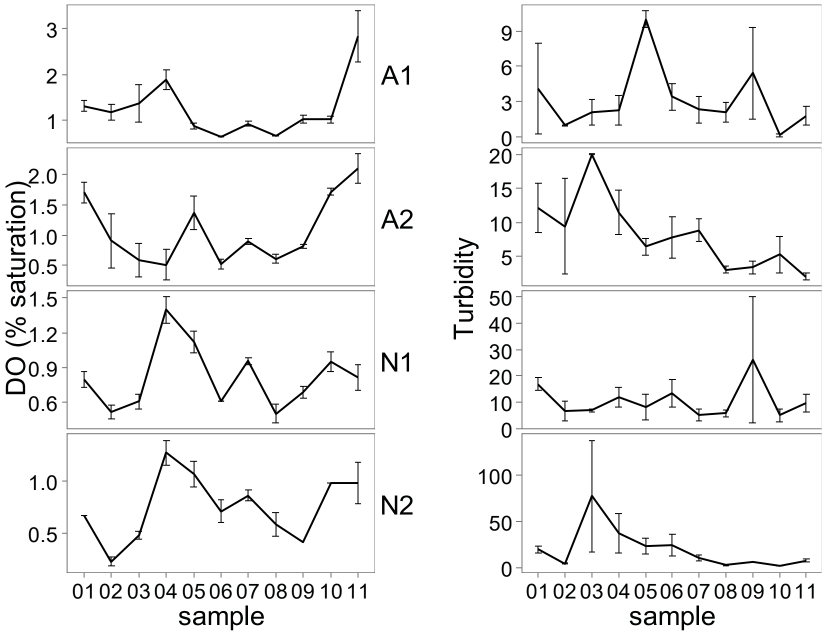
Appendix A

Sampling site	Sediment organic matter content
N1	14%
N2	7%
A1	7%
A2	15%





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