

## Modification of saltmarsh for mosquito control in Australia alters habitat use by nekton

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### Abstract

The most common modification of saltmarsh habitat for mosquito control in Australia is runnelling, a system of shallow channels increasing seawater exchange in pools high on the marsh. Local effects within the immediate vicinity of runnels were examined by testing the hypothesis that fish assemblages immediately alongside runnels differ from those further away. Nekton assemblages were sampled using pop nets in winter (May) and summer (December) on a saltmarsh in southeast Queensland, alongside runnels and further (30 m) away, at two distances from a mangrove-lined intertidal creek. Nekton assemblages were dominated numerically (50–80%) by one or two species of small fish (*Ambassis marianus*, *Mugilogobius stigmaticus*), and a commercially important prawn, *Fenneropenaeus merguensis*. In winter, nekton assemblages alongside runnels were significantly different from those further away. Species richness, total nekton densities and densities of several individual species were higher away from runnels, but only at sites far from the creek. No differences in species richness or densities were found in summer. During both periods, nekton assemblages differed strongly with distance from the creek, with more species and higher densities of most species near the creek. For most species, the overall effect of runnelling appears to be a reduction in abundances in the immediate vicinity of runnels, at some times of year. This is probably related to lower prey availability near runnels. Given the extent of runnelling in some parts of Australia, even this local reduction in densities extending no more than 30 m from runnels means that nekton may be adversely affected over a large total area. The apparent influence of runnels on nekton densities highlights the potential effects of saltmarsh modification on non-target animals that should be considered as this management technique becomes more prevalent.

### Introduction

The majority of studies on saltmarsh nekton have been done on North American marshes (Connolly 1999). These have shown that saltmarshes can support very high densities of juvenile fish (Weinstein 1979; Talbot and Able 1984; Kneib 1997) and decapod crustaceans (Rozas and Zimmerman 2000).

In Australia, saltmarshes typically occur landward of mangrove forests, high in the intertidal zone, and have shorter and less frequent periods of inundation than marshes on the east and gulf coasts of the USA, which generally lack mangroves and extend down to the mid-intertidal zone (Adam 1990). The vegetation of Australian saltmarshes is dominated by succulent herbs and grasses that are

considerably shorter than the stands of cordgrass (*Spartina* spp.) dominating northern hemisphere saltmarshes (Adam 1990). These differences may affect the influence of marsh modification on the use of marshes by nekton.

The first descriptions of fish assemblages associated with unaltered Australian saltmarshes were of the fauna in tidal creeks that drained the saltmarsh flats or in semi-permanent pools that remain on the flats after the tide has receded. Nekton was sampled in a macro-tidal estuary in northern Australia (Davis 1988) and in estuaries with smaller tidal amplitudes in southeast Queensland and temperate NSW (Gibbs 1986; Morton et al. 1987, 1988). More recently, it has been shown that juvenile fish occur over the large expanses of saltmarsh flat inundated only on spring tides, in temperate and subtropical Australian waters (Connolly et al. 1997; Thomas and Connolly 2001). It is only on these spring tides that the distribution of nekton on the marsh can be measured, and this is therefore the most important time to examine effects of marsh modification.

Anthropogenic changes to saltmarsh topography affect the frequency and length of inundation (Rozas 1995), and can affect nekton abundances (Gilmore et al. 1982; Rozas and Minello 1999). Modification of saltmarshes for mosquito control in the USA and use of techniques such as Rotational Impoundment Management have been shown to alter fish (Balling et al. 1980; Harrington and Harrington 1982; Talbot et al. 1986; Poulakis et al. 2002) and invertebrate (Barnby et al. 1985) densities. In Australia, the main form of marsh modification used for mosquito control since 1980 has been runnelling. Runnels are shallow (<30 cm depth), spoon-shaped channels constructed to a maximum gradient of 1:1000 (Hulsman et al. 1989). They provide tidal access to isolated mosquito-breeding pools high on the marsh via small channels that link the tidal source to graded shore regions of the saltmarsh (Dale and Hulsman 1990). The position of a runnel on the shore is determined by the location of mosquito-breeding pools as well as elevation and topographic features of the saltmarsh (Dale et al. 1998). Runnels control mosquitoes by allowing water movement over the marsh during low-amplitude tides that would not normally flood higher regions of the marsh.

Runnels are thought to reduce mosquito abundances by reducing the number of oviposition sites, increasing marsh access for predatory fish (e.g., *Pseudomugil signifer*), and changing water quality which affects maturation and survival of larvae that are produced (Morton et al. 1987; Hulsman et al. 1989; Dale and Hulsman 1990). Although runnels are considered to have little effect on some non-target organisms (e.g., marsh herbs and grasses, Dale et al. 1993), they are known to increase the number of tides on which mangrove propagules are transported to the upper marsh (Breitfuss et al. 2003). Runnels have also been shown to increase substrate moisture content and decrease sediment consolidation immediately surrounding them (i.e. within 5–10 m, Breitfuss and Connolly in press), and alter the densities of shore crabs (Breitfuss et al. in press). The direct effect of runnels on nekton might be expected to be increased access to areas immediately adjacent to runnels, leading to higher densities. Indirect effects of runnels include the impact of changed benthic invertebrate abundances and therefore food availability for nekton, and diminution of predation protection that may be provided by very shallow water (Ruiz et al. 1993).

Runnelling is being used increasingly in southeast Queensland, even where marshes are notionally protected as fish habitat, because urban areas are encroaching on saltmarshes (Connolly and Bass 1996). Potential negative effects of runnelling on nekton might operate locally, within the vicinity of runnels, or on a broader scale, altering the whole of runnelled saltmarshes. This study examines the local scale on a saltmarsh in southeast Queensland, and tests the hypothesis that nekton assemblages immediately alongside runnels differ from those further away, when the marsh is inundated on spring tides. It also assesses the influence of distance from a mangrove-lined feeder creek on nekton during these conditions.

## Methods

### *Study site and timing of sampling*

I sampled nekton on a saltmarsh at Meldale in northern Moreton Bay, in southeast Queensland, Australia (Figure 1). Tidal flows occur through a

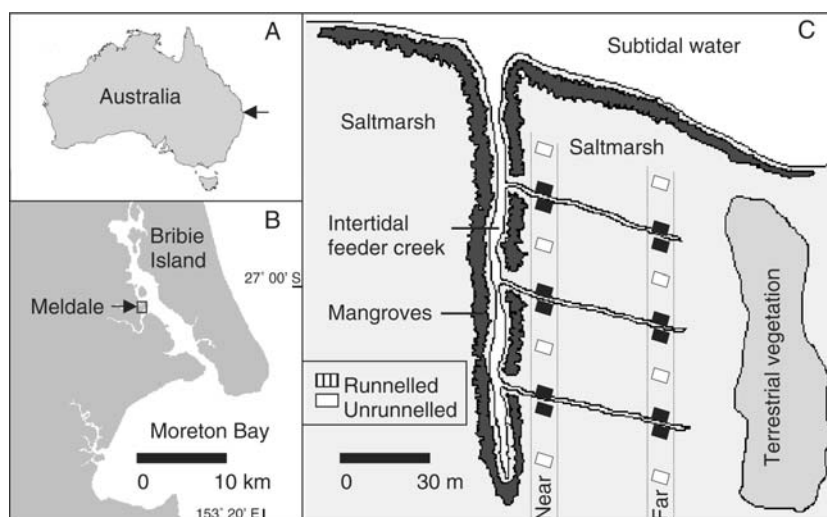


Figure 1. Map showing Meldale study location and configuration of habitats, runnels and stylistic sampling regime.

large creek which drains into Pumicestone Passage, an estuarine component of Moreton Bay. A short turf of glasswort, *Sarcocornia quinqueflora* (Bunge ex Ung-Sternb.), and patches of marsh grass, *Sporobolus virginicus* (L.) Kunth, dominate the vegetation. Mangrove forests of *Avicennia marina* (Forssk.) Vierh. and *Rhizophora stylosa* Griffith occupy the zone seaward of the marsh.

Tides at this marsh are asymmetrically semi-diurnal with amplitudes up to 2.1 m. The marsh is completely inundated at high tide for approximately four consecutive days on spring tides. During these periods it is completely inundated for an average of 17% of the time, and is partially inundated for longer. On a yearly basis, about 7% of tides totally inundate the marsh, although this varies across seasons, with more inundation in winter and summer than other seasons (Connolly 1999). The high tides that inundate the marsh in winter are at night, those in summer are during the day. Two periods were sampled, one in winter (May 1998) and one in summer (December 1998). I sampled nekton on the four consecutive days/nights over which the marsh was inundated in both periods. Given that a comparison between seasons would be confounded by any day/night differences in nekton use, and that such diel differences have not been investigated for Australian marshes, the intention was not to compare the two periods but to test for the effects of runnels at two different times.

#### Nekton collections

Nekton was captured using a series of floorless, buoyant pop nets, modified from Connolly (1994). Nets consisted of four walls of 1 mm diameter mesh (5 m long  $\times$  1 m high) which, when installed, formed a square sitting flush with the marsh surface. Disturbance to the marsh surface was minimised by compressing the substrate to form a shallow depression rather than digging a trench for the net. At high tide, the nets were released using remote trip wires, surfacing within two seconds of deployment, and sampled a 25 m<sup>2</sup> area. Nets were positioned so that nekton would be channelled toward the downstream corner as the tide receded. Nekton was collected initially with hand-held dip nets from the corner where they congregated as the tide receded; these frequent collections also mitigated against predation by scavengers. A final collection was made once the tide had fully receded, about 1 hr after high tide. All nektonic animals were identified (using Johnson (1999) taxonomy), counted, and measured to the nearest millimetre using total length for fish and carapace length for prawns.

Nekton assemblages were compared between: (1) vegetated areas of the marsh alongside runnels and vegetated areas distant from runnels, and (2) at near (5–10 m) and far (40–60 m) distances from an intertidal, mangrove-lined feeder creek that supplied water to and drained the marsh. Within these

near and far distance bands, nets were placed alongside (nearest edge 0.5 m) and distant from (30 m) several runnels that ran approximately perpendicular to the feeder creek (Figure 1). This orthogonal design allowed comparisons between the two habitats, alongside runnels (runnelled) and distant from runnels (unrunnelled), to be made at distances both near and far from the feeder creek. Water depths ( $\pm 1$  cm), temperatures ( $^{\circ}\text{C}$ ) and salinities ( $\text{‰}$ ) were recorded at the surface at each site.

Nets were placed at randomly selected sites at each deployment (i.e. they were moved every day), within the constraints of habitat type (runnelled or unrunnelled) and distance from creek (near and far). This random placement of sites is the best method for ensuring that samples are truly representative of the levels of each factor. In the winter sampling period, 44 pop nets were deployed, 11 in each combination of habitat type and distance from creek. In the summer sampling period, 48 nets were deployed, 12 in each combination. In both sampling periods, an even number of nets in each habitat/distance combination was deployed on each of the four days/nights.

#### Data analysis

Multivariate analyses were used to examine habitat use by nekton. Non-metric multidimensional scaling (NMDS) was used to ordinate habitat groups from biotic similarity matrices using Bray–Curtis similarities. Differences in nekton assemblages between the two habitats (runnelled/unrunnelled) and two distances from the creek (near/far) were tested for significance using a two-factor analysis of similarities (ANOSIM) separately for each period, using the PRIMER v. 4 package. Raw counts were transformed using  $x^{0.25}$  to emphasise the contribution of less common species in analyses. The intention of ANOSIM is to detect differences in average similarities within and among groups, but like other multivariate randomisation tests, it will give a significant result even if the only difference among groups is in dispersion (i.e., multivariate variances). After significant ANOSIM tests I therefore analysed the similarity matrix further using NPDISP (McArdle and Anderson 2001), to check whether dispersion of groups differed.

Univariate analyses were done separately for the two periods. A two-factor analysis of variance was used to test whether species richness and nekton densities differed between habitats and distance from the creek. The variables analysed were species richness (number of species  $25\text{ m}^{-2}$ ) and density (individuals  $\text{m}^{-2}$ ) of: (1) all species combined, (2) all species except the numerically dominant species at that period, and (3) selected species considered common enough to analyse (occurring in  $\geq 5$  nets, and shown in Tables 2 and 3). Data were  $\log(x + 1)$  transformed prior to ANOVA tests so that the interaction term provided a test of proportional rather than magnitudinal differences on raw data (Hurlbert and White 1993). If a significant interaction was found, *post hoc* Tukey tests were used to determine differences between levels of a factor for each level of the other factor separately. Length-frequency distributions between the levels of each of the two factors, habitat and distance, were compared using Kolmogorov–Smirnov tests. The criterion for testing was that at least 10 individuals must occur in the two levels being tested. The species analysed in winter were *Ambassis marianus*, *Mugilogobius stigmaticus* and *Pseudogobius* sp., and *Calamiana* sp., *Gerres subfasciatus* and *M. stigmaticus* in summer.

## Results

#### Environmental data

Although the mean water depth across all sampling sites was less in winter (21.8 cm) than summer (25.7 cm), the patterns in depths among treatments was the same. Sites near to the creek were always deeper than those far from the creek, and within each distance band, the water depths at runnelled and unrunnelled sites were similar. Depth ranges were: near creek, runnelled habitat 25–47 cm; near creek, unrunnelled habitat 21–48 cm; far from creek, runnelled habitat 6–26 cm; far from creek, unrunnelled habitat 8–26 cm. Mean temperatures and salinities were  $17.9\text{ }^{\circ}\text{C}$  (range = 15.4–18.9) and 28.4 $\text{‰}$  (25.6–30.6) in winter, and  $28.5\text{ }^{\circ}\text{C}$  (26.4–34.0) and 30.5 $\text{‰}$  (25.5–31.8) in summer. No differences among temperature or salinity means for different habitat/distance combinations could be detected using ANOVA, at either period.

Table 1. List of species caught, taxonomic authorities, extent of distribution into open marine waters, economic importance and size range.

Species	Family	Marine/ Estuarine	Economic importance	Size range (mm)	
				Winter	Summer
<i>Acanthopagrus australis</i> (Owen)	Sparidae	ME	C R	77–198	34–107
<i>Ambassis jacksoniensis</i> (Macleay)	Ambassidae	E	–	32–43	25
<i>Ambassis marianus</i> (Günther)	Ambassidae	E	–	13–65	30–43
<i>Arenigobius frenatus</i> (Günther)	Gobiidae	E	–		31
<i>Arrhamphus sclerolepis</i> (Steindachner)	Hemiramphidae	ME	C R	56–125	110
<i>Atherinomorus ogilbyi</i> (Whitely)	Atherinidae	E	–	65–66	
<i>Calamiana</i> sp.	Gobiidae	E	–	18–31	20–34
<i>Fenneropenaeus merguensis</i> (de Man)	Penaecidae	E	C R	8–34	
<i>Gerres subfasciatus</i> (Cuvier)	Gerridae	ME	B	40	11–68
<i>Gobiopterus semivestitus</i> (Munro)	Gobiidae	E	–	19–42	9–15
<i>Herklotsichthys castelnaui</i> (Ogilby)	Clupeidae	ME	B	53	
<i>Liza argentea</i> (Quoy and Gaimard)	Mugilidae	ME	C		43
<i>Mugilogobius stigmaticus</i> (De Vis)	Gobiidae	E	–	15–65	14–52
<i>Pseudogobius</i> sp.	Gobiidae	E	–	22–39	32–37
<i>Pseudomugil signifer</i> (Kner)	Atherinidae	E	–	25–31	
<i>Sillago maculata</i> (Quoy and Gaimard)	Sillaginidae	ME	C R	31–84	27–155
<i>Tetractenos hamiltoni</i> (Gray and Richardson)	Tetraodontidae	E	–	61–104	95–102
<i>Torquigener pleurosticta</i> (Günther)	Tetraodontidae	E	–	54–105	118–127
<i>Tylosurus gavialoides</i> (Castelnaui)	Belontiidae	ME	R		89–209
<i>Valamugil georgii</i> (Ogilby)	Mugilidae	ME	C	88–102	126–145

Species are arranged alphabetically. Extent of distribution into marine waters: E = solely estuarine species, ME = species in marine and estuarine waters. Economic importance: C = commercial species, R = recreational species, B = bait species. Size range is for all individuals caught in the current study, by period, total length for fish and carapace length for prawns.

### Species composition

Over the whole study, one prawn species and 19 fish species from 11 families were caught (Table 1), with 64% of nets catching nekton. Eight of the fish species were of economic importance, and five of these occurred in both sampling periods. Economically important fish species contributed about 15% to the total catch, and were represented by small juveniles, although mugilids, sparids and hemiramphids also occurred as larger juveniles or adults. In the winter sampling period, 701 fish (0.64 m<sup>-2</sup>) from 16 species were caught (Table 2), with the catch dominated numerically by *A. marianus* (55%) and *M. stigmaticus* (23%). The banana prawn, *Fenneropenaeus merguensis*, was also abundant in this sampling period (0.25 m<sup>-2</sup>). For the summer sampling period, 372 fish (0.31 m<sup>-2</sup>) of 16 species were caught, with the catch dominated numerically by *Calamiana* sp. (41%), *G. subfasciatus* (23%) and *M. stigmaticus* (18%). This is the first record of the genus *Calamiana* from southeast Queensland (Johnson 1999).

### Comparisons of species richness and composition

Clear differences were found in multivariate assemblages across the two factors. Stress for the ordinations was between 0.1–0.2 for both periods, implying that the distances reflect similarities among sites reasonably well in the two-dimensional solutions (Clarke 1993). In both periods, assemblages from sites near and far from the creek grouped separately (Figure 2a, shown for winter only), and were statistically different (2-way ANOSIM, Distance factor,  $R = 0.41$  (winter), 0.48 (summer),  $p < 0.001$  for both periods). Multivariate dispersion was not significant at either period (NPDISP:  $p = 0.220$  (winter), 0.301 (summer), so it is safe to conclude ANOSIM detected differences in average similarities among groups. Separation of runnelled and unrunnelled sites was also evident in winter (Figure 2b) and was statistically significant, but was not as distinct as the distance groupings (2-way ANOSIM, Habitat factor,  $R = 0.13$ ,  $p = 0.012$ ). Again, multivariate

Table 2. Summary of species richness (no. of species 25 m<sup>-2</sup>) and density (no. of individuals 100 m<sup>-2</sup>) in runnelled (R) and unrunnelled (U) habitat, and at near and far distances from creek.

Species	Winter						Summer					
	Overall		R	U	Near	Far	Overall		R	U	Near	Far
	D	%	D	D	D	D	D	%	D	D	D	D
<i>Acanthopagrus australis</i>	1.6	2.6	1.3	2	2.9	0.4	0.3	1.1	0.7	–	0.7	–
<i>Ambassis jacksoniensis</i>	0.9	1.4	–	1.8	0.9	0.9	0.1	0.3	0.2	–	0.2	–
<i>Ambassis marianus</i>	35.3	55.3	30.9	39.6	44.9	25.6	1.8	5.6	3.5	–	3.5	–
<i>Arenigobius frenatus</i>	–	–	–	–	–	–	0.1	0.3	0.2	–	0.2	–
<i>Arrhamphus sclerolepis</i>	1.4	2.1	1.1	1.6	2.7	–	0.1	0.3	–	0.2	0.2	–
<i>Atherinomorus ogilbyi</i>	0.2	0.3	–	0.4	0.4	–	–	–	–	–	–	–
<i>Calamiana</i> sp.	1.5	2.3	2	0.9	2.7	0.2	12.8	41.1	12	13.5	19.7	5.8
<i>Fenneropenaeus merguensis</i>	24.7	NA	17.1	32.4	48.9	0.5	–	–	–	–	–	–
<i>Gerres subfasciatus</i>	0.1	0.1	–	0.2	0.2	–	7.2	23.1	3	11.3	6.3	8
<i>Gobiopterus semivittatus</i>	0.2	0.3	–	0.4	0.4	–	0.4	1.3	0.3	0.5	–	0.8
<i>Herklotsichthys castelnaui</i>	0.1	0.1	–	0.2	0.2	–	–	–	–	–	–	–
<i>Liza argentea</i>	–	–	–	–	–	–	0.1	0.3	0.2	–	–	0.2
<i>Mugilogobius stigmaticus</i>	14.4	22.5	13.6	15.1	24.9	3.8	5.5	17.7	6.3	4.7	5.3	5.7
<i>Pseudogobius</i> sp.	4.7	7.4	6.4	3.1	9.5	–	1.1	3.5	1.5	0.7	1.7	0.5
<i>Pseudomugil signifer</i>	0.2	0.3	0.2	0.2	–	0.4	–	–	–	–	–	–
<i>Sillago maculata</i>	0.6	1.0	0.7	0.5	0.7	0.5	0.8	2.7	1.3	0.3	1.5	0.2
<i>Tetractenos hamiltoni</i>	1.1	1.7	1.6	0.5	1.6	0.5	0.3	0.8	0.5	–	0.2	0.3
<i>Torquigener pleurosticta</i>	1	1.6	0.4	1.6	0.4	1.6	0.2	0.5	0.2	0.2	–	0.3
<i>Tylosurus gvaloides</i>	–	–	–	–	–	–	0.3	0.8	–	0.5	0.2	0.3
<i>Valamugil georgii</i>	0.5	0.9	0.4	0.7	0.5	0.5	0.2	0.5	0.3	–	0.3	–
Total (all species)	63.7		58.5	68.9	92.9	34.5	31.0		30.2	31.8	39.8	22.2
Dominant species excluded	28.4		27.6	29.3	48.0	8.9	18.2		18.2	18.3	20.2	16.3
Species richness	3.3		2.7	4.0	4.6	2.1	2.2		2.5	2.0	2.7	1.8

Overall = habitats combined, D = mean density, % = % of total fish abundance. NA = not applicable.

dispersion was not significant (NPDISP:  $p = 0.198$ ), so I can safely conclude that assemblages are different on average between runnelled and unrunnelled sites. Runnelled and unrunnelled sites did not group separately in summer and there was not a statistically significant difference (2-way ANOSIM, Habitat factor,  $R = 0.08$ ,  $p = 0.291$ ).

For both sampling periods, irrespective of habitat type, a significantly higher number of species (i.e., species richness) occurred near to the creek, despite a significant interaction term (Table 3 for winter, Table 4 for summer). Differences in species richness between runnelled and unrunnelled habitat only occurred in winter and these differences depended on the distance from the creek. A significantly higher number of species was found in unrunnelled habitat, far from the creek. Near to the creek, however, species richness did not differ between the two habitat types (Table 3).

The distributions of individual species (presence/absence data) showed some interesting patterns,

although these were rarely consistent over both periods. *M. stigmaticus* was the only species to occur in all four combinations of distance from creek and habitat type in both sampling periods. Within each sampling period, some species were specific to particular combinations. Of the species that were abundant (occurring in  $\geq 10$  nets, combining periods), three never occurred in runnelled habitat far from the creek (*Sillago maculata*, *Arrhamphus sclerolepis* and *Acanthopagrus australis*).

#### Comparisons of densities and sizes

In both periods, the overwhelming pattern was that for most species, significantly higher densities were found near the creek (Table 3 for winter, Table 4 for summer). Only *Torquigener pleurosticta* was more common far from creek, and then only in unrunnelled habitat and only in winter. The other tetraodontid, *Tetractenos hamiltoni*, showed exactly the opposite pattern, being more common

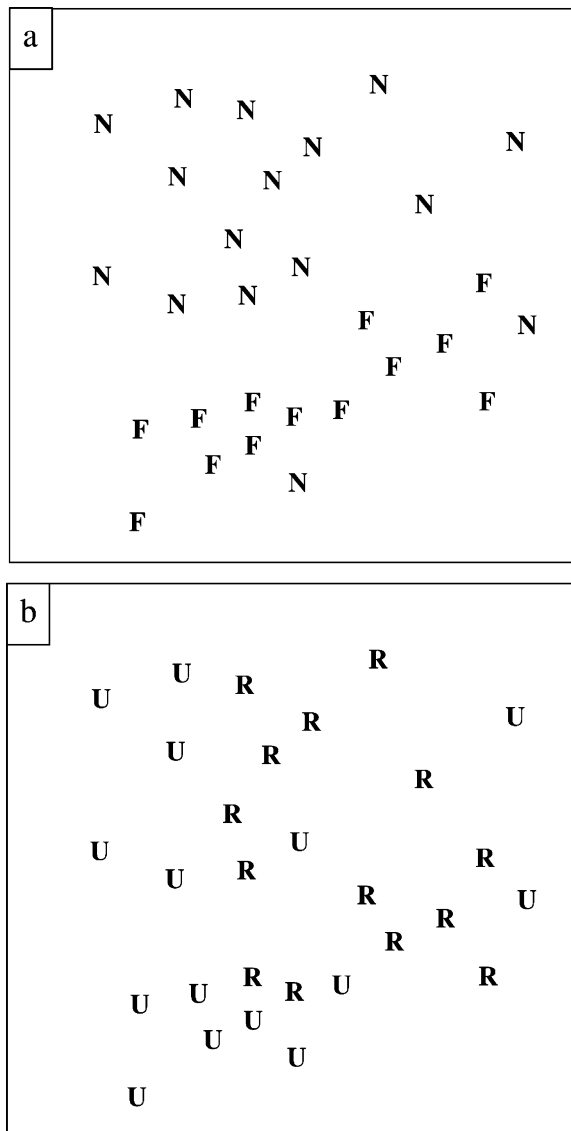


Figure 2. Ordination (NMDS) of nekton assemblages from the winter period (stress = 0.13), labelled to show: (a) near (N) and far (F) distances from creek, and (b) runnelled (R) and unrunnelled (U). Sites at which no nekton was caught (15 sites) are not shown. Axes are unitless.

near to the creek but only in runnelled habitat. No differences in densities of nekton species between runnelled and unrunnelled habitat were detected in summer, and in winter the differences were generally weaker than differences between distances from the creek. In winter, *Ambassis jacksoniensis* was more abundant in unrunnelled habitat, regardless of distance from creek. An interaction

between habitat type and distance from the creek was detected for total nekton densities and for densities excluding the dominant species in winter. Higher densities of nekton occurred in unrunnelled habitat far from the creek but not near the creek.

In winter, the size distributions of *A. marianus* and *M. stigmaticus* differed between runnelled and unrunnelled habitats, near to the creek. These species had lower proportions of small individuals in runnelled habitat. For example, the smallest (<30 mm) size classes of *M. stigmaticus* were absent in runnelled habitat, near to the creek (Figure 3, KS test:  $p < 0.001$ ). The size distributions of *A. marianus* also differed at distances close to and far from the creek in unrunnelled habitat (Figure 4, KS test:  $p < 0.001$ ). Individuals less than 24 mm did not occur far from the creek in this habitat. No other significant differences in size distributions were detected in winter.

In summer, *G. subfasciatus* was the only species to have different size distributions among habitat/distance combinations. In unrunnelled habitat, a much higher proportion of small individuals (8–14 mm) and fewer of the larger size classes (15–68 mm) occurred far from the creek, relative to near the creek (KS test:  $p < 0.001$ ,  $n = 21$  near,  $n = 47$  far).

## Discussion

The numerical dominance of the nekton assemblages by a small number of species is a common pattern in northern hemisphere marsh creeks and flats (e.g., Kneib and Wagner 1994), and is consistent with results from previous sampling on southeast Queensland marshes. The numerical dominance by a small number of species is also a widespread pattern. The same species that were dominant in the current study were dominant at a previous study at Meldale marsh (Thomas and Connolly 2001), although some differences were observed. For example, *A. marianus* was dominant in both winter and summer in the previous sampling whereas it was much less so in summer in the present study. The numerically dominant species in the current study were small fishes that are resident in estuarine waters at all times of year. Perhaps surprisingly, larger fishes of economic

Table 3. Summary of winter results of 2-way ANOVA and subsequent Tukey testing for differences between runnelled (R) and unrunnelled (U) habitat and distance from creek for species richness and densities.

Variable	Factor	<i>p</i>	Pairwise comparisons
Species richness	Habitat	0.003**	N: R = U F: U > R
	Distance	<0.001***	R: N > F U: N > F
	Interaction	0.004**	
Density			
	All species		
	Dominant species excluded		
All species	Habitat	0.019*	N: R = U F: U > R
	Distance	<0.001***	R: N > F U: N = F
	Interaction	0.021*	
Dominant species excluded	Habitat	0.006**	N: R = U F: U > R
	Distance	<0.001***	R: N > F U: N > F
	Interaction	<0.001***	
<i>Acanthopagrus australis</i>	Habitat	0.218	
	Distance	0.001**	N > F
	Interaction	0.965	
<i>Ambassis marianus</i> <sup>d</sup>	Habitat	0.052	
	Distance	0.040*	N > F
	Interaction	0.112	
<i>Arrhamphus sclerolepis</i>	Habitat	0.997	
	Distance	0.001**	N > F
	Interaction	0.997	
<i>Ambassis jacksoniensis</i>	Habitat	0.002**	U > R
	Distance	1.000	
	Interaction	1.000	
<i>Calamiana</i> sp.	Habitat	0.147	
	Distance	0.002**	N > F
	Interaction	0.408	
<i>Fenneropenaeus merguensis</i>	Habitat	0.284	
	Distance	<0.001***	N > F
	Interaction	0.354	
<i>Mugilogobius stigmaticus</i>	Habitat	0.245	
	Distance	<0.001***	N > F
	Interaction	0.677	
<i>Pseudogobius</i> sp.	Habitat	0.997	
	Distance	<0.001***	N > F
	Interaction	0.997	
<i>Tetractenos hamiltoni</i>	Habitat	0.384	N: R > U F: R = U
	Distance	0.384	R: N > F U: N = F
	Interaction	0.005**	
<i>Torquigener pleurosticta</i>	Habitat	0.048*	N: R = U F: U > R
	Distance	0.048*	R: N = F U: F > N
	Interaction	0.001**	

N = Near, F = Far. <sup>d</sup> = dominant species numerically in winter.

\* =  $p < 0.05$ , \*\* =  $p < 0.01$ , \*\*\* =  $p < 0.001$ .

importance were also caught at both sampling periods, including as large juveniles and adults. Some fraction of populations of these species was on the marsh throughout the year, and the marsh fauna lacked representatives of transient species spending only part of their life cycle in upper estuarine habitats. The abundance of *F. merguensis* caught in winter is an important new observation. Microhabitat preferences of this species in

mangrove forests and creeks in southeast Queensland have been studied in detail (Meager et al. 2003), but their occurrence on saltmarshes has not previously been recorded. There is a valuable fishery for *F. merguensis* along the east and north coasts of Queensland (Vance et al. 1998), and its occurrence on the marsh is an indication of the potential importance of this habitat for the species.



Table 4. Summary of summer results of 2-way ANOVA and subsequent Tukey testing for differences between runnelled and unrunnelled habitat and distance from creek for species richness and densities.

Variable	Factor	p	Pairwise comparisons	
Species richness	Habitat	0.285	N > F	
	Distance	0.040*		
	Interaction	0.240		
Density				
	All species	Habitat	0.872	N > F
		Distance	0.036*	
Interaction		0.927		
Dominant species excluded	Habitat	0.736		
	Distance	0.593		
	Interaction	0.633		
<i>Calamiana</i> sp. <sup>d</sup>	Habitat	0.709	N > F	
	Distance	0.002**		
	Interaction	0.576		
<i>Gerres subfasciatus</i>	Habitat	0.114		
	Distance	0.377		
	Interaction	0.169		
<i>Mugilogobius stigmaticus</i>	Habitat	0.399		
	Distance	0.777		
	Interaction	0.218		
<i>Pseudogobius</i> sp.	Habitat	0.325	N > F	
	Distance	0.047*		
	Interaction	0.451		
<i>Sillago maculata</i>	Habitat	0.280		
	Distance	0.071		
	Interaction	0.071		

N = Near, F = Far. <sup>d</sup> = dominant species numerically in summer.

\* =  $p < 0.05$ , \*\* =  $p < 0.01$ .

The differences between runnelled and unrunnelled habitat were found mainly away from the creek. In this high marsh zone, there were more species, higher overall densities with and without the dominant species, and higher densities of several individual species away from the runnels. The lower densities near runnels might result from rapid upstream movement within runnels to other habitats, but are more likely related to prey availability, discussed below. The pattern described above occurred only in the winter period. The lack of consistency across sampling periods may reflect a seasonal difference, although this study was not designed to test those. The difference may also be due to diel patterns of marsh use by nekton, since the marsh is inundated at night in winter and day in summer. Differences in nekton use of marshes between night and day have been shown

elsewhere (e.g., Rountree and Able 1993), but no information is available on diel influences on Australian marshes.

The lower densities near the runnels could be explained by secondary effects associated with runnels. The runnels might provide better access to the occasional large predator, too sparsely distributed themselves to be detected in this study (given the small total area sampled), but resulting in smaller fish avoiding the area. Although size frequency analysis showed that smaller individuals of some species were absent from runnels, this was only found near the creek, and thus does not provide evidence of smaller size classes avoiding the runnelled habitat on the upper marsh. Rather, it is apparent that certain species, regardless of size class, are less abundant near runnels. This difference in abundance is not related to water temperature or salinity, since there was no difference in these variables between runnelled and unrunnelled habitat. Furthermore, although it has previously been shown that water depth and fish abundances are positively correlated on southeast Queensland marshes (Thomas and Connolly 2001), this variable cannot explain differences between runnelled and unrunnelled habitat because water depths were similar in the two habitats. There is also a possibility that the pop nets are more effective in unrunnelled habitat, although I consider this unlikely given the similarity in water depth and vegetation cover in the two habitats.

The most likely impact of runnels on nekton is the effect they have on invertebrate prey. Runnels have been shown to increase substrate moisture on tides that do not fully inundate the marsh (Breitfuss and Connolly in press) and reduce benthic invertebrate densities within 5 m of runnels on some marshes (Breitfuss et al. 2004). Such invertebrates are prey for at least some of the nekton species on the marsh (Morton et al. 1988), and reductions in prey densities might therefore adversely affect fish densities. Benthic carnivores (e.g. *A. australis*) are most likely to be affected and mid-water species (e.g. *A. marianus*) least likely. The effect of changes in substrate moisture from runnelling on the abundance of benthic microalgae and particulate detritus is unknown, and I can therefore make no prediction about the likelihood of runnels affecting food availability for detritivores (e.g. *Valamugil georgii*). Fishes such as

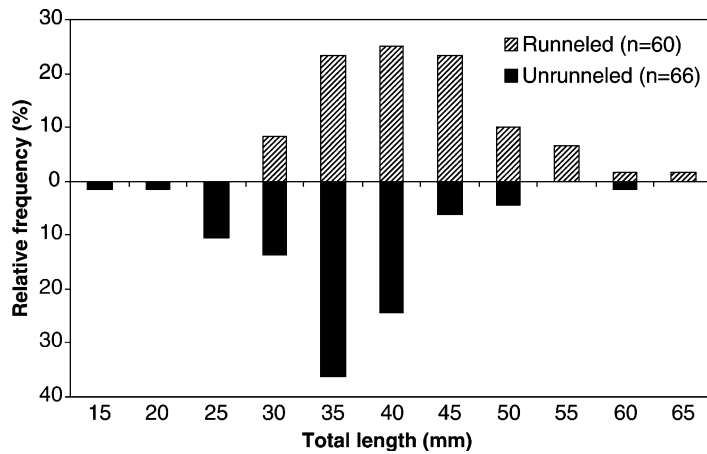


Figure 3. Length-frequency distributions in winter of: (a) *M. stigmaticus* in runnelled and unrunnelled habitats, near to the feeder creek, and (b) *A. marianus* at near and far distances from the feeder creek, in unrunnelled habitat.

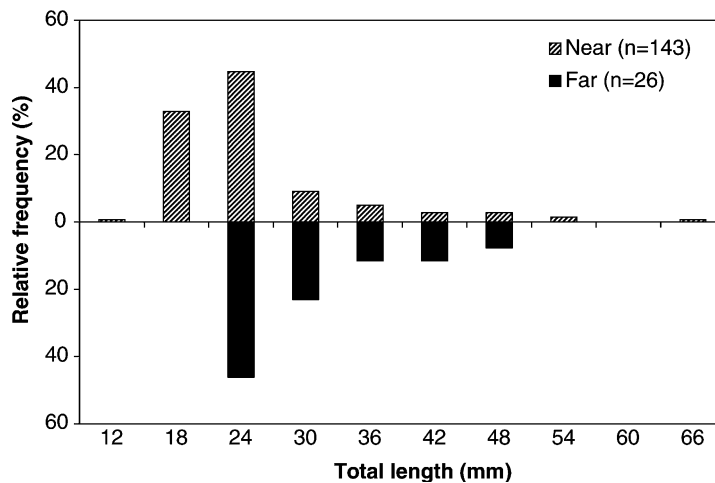


Figure 4.

*P. signifer* that are known to prey on mosquito larvae (Morton et al. 1988) might be affected by the reduction in mosquitoes resulting from runnelling. Given the demonstrated impacts of runnelling on potential prey items of some fishes, the potential link between nekton, their prey and runnelling should be a focus for researchers in the future.

It should be acknowledged that runnels might have been located in parts of the marsh that were topographically different to other parts prior to runnelling, in which case the differences in nekton assemblages between runnelled and unrunnelled habitat might not be due to the runnels at all.

However, the sampling sites immediately adjacent to runnels had the same elevation as unrunnelled sites, as evidenced by the similar water depths at high tide, and there were no discernible differences in vegetation type between runnelled and unrunnelled areas. It is therefore likely that differences between runnelled and unrunnelled habitat are due to runnelling. Another avenue for determining the effects of runnelling is to compare nekton use of marshes before and after runnelling. Comparisons of fauna before and after modification of marshes for mosquito control in the USA have been able to discern differences in nekton abundances (Gilmore et al. 1982;

Harrington and Harrington 1982; Poulakis et al. 2002). I have found no comparisons of nekton on marshes before and after runnelling, but this is an avenue that should be pursued.

The Meldale marsh is only inundated at spring tides, and the current study was deliberately focussed on this part of the tidal cycle. However, the use of the marsh by nekton might also be affected on lower amplitude tides. Runnels allow water movement on the marsh even when tides are not high enough to inundate the marsh flats. Fish probably occur in the runnels themselves at these times, and may also enter, via runnels, the semi-permanent pools that remain on the marsh flat after spring tides. Although these pools are small relative to the total marsh area, they are known to contain nekton (Morton et al. 1988). The influence of runnels on the ecology of nekton and their prey in the semi-permanent pools high on the marsh is an aspect of marsh modification that needs further work.

The strongest pattern shown in this study was that species richness and the densities of several species were higher near the mangrove-lined feeder creek, in both periods. The densities recorded near the creek in winter (nearly 1 individual  $m^{-2}$ ) are higher than any values recorded at this marsh in a previous study (Thomas and Connolly 2001). The focus of work elsewhere, particularly in the USA, has been more on distance onto the marsh from open water, a term describing subtidal channels or deeper open areas. Most studies show increasing dominance by smaller, estuarine fish and crustacean species at greater distances onto the marsh and different assemblages near and far onto the marsh (Talbot and Able 1984; Baltz et al. 1993; Kneib and Wagner 1994; Minello et al. 1994; Peterson and Turner 1994). Larger individuals mostly remain near the marsh edge close to deeper water, although occasionally small numbers of these species venture further onto marshes (Kneib 1991; Rozas 1992; Peterson and Turner 1994). Previous sampling at Meldale and another marsh in southeast Queensland demonstrated no change in fish densities with distance from subtidal water (Thomas and Connolly 2001). On subtropical marshes in Australia, the influence of intertidal creeks appears to be greater than that of open water adjacent to marsh edge. The higher species richness near the creek also occurred where water depth was greater; this is consistent with a strong

positive relationship between species richness and water depth shown previously at Meldale marsh (Thomas and Connolly, 2001). While intertidal creeks and open water adjacent to marshes both potentially provide access to the marsh, the structure provided by mangrove roots and low branches in intertidal creeks might offer higher levels of protection from predators than the open water. Kneib and Wagner (1994) suggested that proximity to the marsh edge lessens the chance of being stranded, and this would apply equally to proximity to creeks.

The tetraodontids showed interesting interspecific patterns but were also noteworthy for their abundances relative to those on North American marshes, where they are rare. *T. pleurosticta* was found at higher densities far from the creek. Individuals of this species move onto the marsh flat at the front of the incoming tide, pushing far onto the marsh in very shallow water. This strategy has the effect of increasing the likelihood of finding major prey items, such as snails and crabs (Hughes 1984), that become available while the marsh is emergent. The other tetraodontid, *T. hamiltoni*, has a similar diet, and the very dissimilar pattern of abundance for the two species might reflect spatial segregation through competition. The abundances of this family on southeast Queensland marshes relative to those on North American marshes might result from the lower frequency and shorter period of inundation in Queensland. With the marsh flat emergent for such long periods, marshes in Queensland lack small, resident nekton species, and this may create an opportunity for the tetraodontids to obtain prey relatively easily as they enter the marsh upon inundation.

A large proportion of saltmarshes in southeast Queensland, northern NSW and south-western Western Australia are runnelled or designated to be runnelled (Breitfuss 2001; Latchford et al. 2002). For most species, the overall effect of runnelling appears to be a reduction in abundances in the immediate vicinity of runnels, at some times of year, probably as a result of lower prey availability near runnels. Although the effects of runnels on nekton are local (e.g. <30 m in the current study), the total area affected by runnels would be considerable. The apparent influence of runnels on nekton densities highlights the potential effects of

saltmarsh modification on non-target animals that should be considered as this management technique becomes more prevalent.

In weighing up the consequences of runnelling it is important to compare the potential impacts on non-target organisms to those of other mosquito control methods. Runnelling probably has no greater impact than chemical larvacides, the main alternative in southeast Queensland. However, two further studies are required to be more certain of runnelling impacts. First, a regression design sampling nekton at a series of distances from runnels should be used to determine more precisely the extent of the influence of runnels. Second, comparisons of nekton before and after runnelling on marshes not yet runnelled are needed to detect any potential larger-scale, marsh-wide impacts of runnelling. Even existing data, however, highlight the caution necessary in modifying marsh habitat for mosquito control.

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### References

- Adam P. 1990. *Saltmarsh Ecology*. Cambridge University Press, Cambridge, UK.
- Balling S.S., Stoehr T. and Resh V.H. 1980. The effects of mosquito control recirculation ditches on the fish community of a San Francisco Bay salt marsh. *California Fish and Game* 66: 25–34.
- Baltz D.M., Rakocinski C. and Fleeger J.W. 1993. Microhabitat use by marsh-edge fishes in a Louisiana estuary. *Environmental Biology of Fishes* 36: 109–126.
- Barnby M.A., Collins J.N. and Resh V.H. 1985. Aquatic macro-invertebrate communities of natural and ditched potholes in a San Francisco Bay salt marsh. *Estuarine Coastal and Shelf Science* 20: 331–347.
- Breitfuss M.J. 2001. Predicting the effects of runnelling on non-target saltmarsh resources. *Arbovirus Research in Australia* 8: 23–29.
- Breitfuss M.J. and Connolly R.M. (in press). Consolidation and volumetric soil water content of saltmarsh soils following habitat modification for mosquito control. *Wetland Ecology and Management*.
- Breitfuss M.J., Connolly R.M. and Dale P.E.R. 2003. Mangrove distribution and mosquito control: transport of *Avicennia marina* propagules by mosquito-control runnels in southeast Queensland saltmarshes. *Estuarine Coastal and Shelf Science* 56: 573–579.
- Breitfuss M.J., Connolly R.M. and Dale P.E.R. 2004. Densities and aperture sizes of burrows constructed by *Helograpsus haswellianus* (Decapoda: Varunidae) in saltmarshes with and without mosquito control runnels. *Wetlands* 24: 14–22.
- Clarke K.R. 1993. Non-parametric multivariate analyses of changes structure. 18: 117–143.
- Connolly R.M. 1994. Comparison of fish catches from a buoyant pop net and a beach seine net in a shallow seagrass habitat. *Marine Ecology Progress Series* 109: 305–309.
- Connolly R.M. 1999. Saltmarsh as habitat for fish and nektonic crustaceans: challenges in sampling designs and methods. *Australian Journal of Ecology* 24: 422–430.
- Connolly R.M. and Bass D.A. 1996. Do fish actually use saltmarsh flats? Ecology and management of Australian saltmarshes. In: Harvey N. (ed.), *Proceedings of the Australian Coastal Management Conference*, Glenelg, South Australia, April 1996, University of Adelaide, Adelaide, South Australia, pp. 273–276.
- Connolly R.M., Dalton A. and Bass D.A. 1997. Fish use of an inundated saltmarsh flat in a temperate Australian estuary. *Australian Journal of Ecology* 22: 222–226.
- Dale P.E.R., Dale P.T., Hulsman K. and Kay B.H. 1993. Runnelling to control saltmarsh mosquitoes: Long-term efficacy and environmental impacts. *Journal of the American Mosquito Control Association* 9: 174–181.
- Dale P.E.R., Hapgood G., Kay B., Morris C. and Standfast H. 1998. *Australian Mosquito Control Manual*. Australian Mosquito Control Association Inc., Redland Bay, Queensland, Australia.
- Dale P.E.R. and Hulsman K. 1990. A critical review of salt marsh management methods for mosquito control. *Reviews in Aquatic Science* 3: 281–311.
- Davis T.L.O. 1988. Temporal changes in the fish fauna entering a tidal swamp system in tropical Australia. *Environmental Biology of Fishes* 21: 161–172.
- Gibbs P.J. 1986. The fauna and fishery of Wallis Lake. In: Gibbs P.J. (ed), *Wallis Lake: Present and Future*, Australian Marine Science Association, NSW, Australia, pp. 1–7.
- Gilmore R.G., Cooke D.W. and Donohoe C.J. 1982. A comparison of the fish populations and habitat in open and closed salt marsh impoundments in east-central Florida. *Northeast Gulf Science* 5: 25–37.
- Harrington R.W. and Harrington E.S. 1982. Effects on fishes and their forage organisms of impounding a Florida salt marsh to prevent breeding by salt marsh mosquitoes. *Bulletin of Marine Science* 32: 523–531.
- Hughes J.M. 1984. A note on predation by toadfish *Spheroides* spp. on the mangrove snail *Littorina scabra*. In: Coleman R.J., Covacevich J. and Davie P. (eds), *Focus on Stradbroke: new information on North Stradbroke Island and surrounding*

- areas 1974–1984, Boolarong Publications, Brisbane, Australia, pp. 312–314.
- Hulsman K., Dale P.E.R. and Kay B.H. 1989. The runnelling method of habitat modification: an environment-focused tool for salt marsh mosquito management. *Journal of the American Mosquito Control Association* 5: 226–234.
- Hurlbert S.H. and White M.D. 1993. Experiments with freshwater invertebrate zooplanktivores: quality of statistical analyses. *Bulletin of Marine Science* 53: 128–153.
- Johnson J.W. 1999. Annotated checklist of the fishes of Moreton Bay, Queensland, Australia. *Memoirs of the Queensland Museum* 43: 709–762.
- Kneib R.T. 1991. Flume weir for quantitative collection of nekton from vegetated intertidal habitats. *Marine Ecology Progress Series* 75: 29–38.
- Kneib R.T. 1997. The role of tidal marshes in the ecology of estuarine nekton. *Oceanography and Marine Biology Annual Review* 35: 163–220.
- Kneib R.T. and Wagner S.L. 1994. Nekton use of vegetated marsh habitats at different stages of tidal inundation. *Marine Ecology Progress Series* 106: 227–238.
- Latchford J., McComb A., Davis J. and Paling E. 2002. The effects of runnelling: a technique for controlling mosquitoes in saltmarshes in south-western Western Australia. Report to Department of Health, Western Australia, Murdoch University, Perth, Australia.
- McArdle B.H. and Anderson M.J. 2001. Fitting multivariate models to community data: a comment on distance-based redundancy analysis. *Ecology* 82: 290–297.
- Meager J.J., Vance D.J., Williamson I. and Loneragan N.R. 2003. Microhabitat distribution of juvenile *Penaeus merguensis* de Man and other epibenthic crustaceans within a mangrove forest in subtropical Australia. *Journal of Experimental Marine Biology and Ecology* 294: 127–144.
- Minello T.J., Zimmerman R.J. and Medina R. 1994. The importance of edge for natant macrofauna in a created salt marsh. *Wetlands* 14: 184–198.
- Morton R.M., Beumer J.P. and Pollock B.R. 1988. Fishes of a subtropical Australian saltmarsh and their predation upon mosquitoes. *Environmental Biology of Fishes* 21: 185–194.
- Morton R.M., Pollock B.R. and Beumer J.P. 1987. The occurrence and diet of fishes in a tidal inlet to a saltmarsh in southern Moreton Bay, Queensland. *Australian Journal of Ecology* 12: 217–237.
- Peterson G.W. and Turner R.E. 1994. The value of salt marsh edge vs interior as a habitat for fish and decapod crustaceans in a Louisiana tidal marsh. *Estuaries* 17: 235–262.
- Poulakis G.R., Shenker J.M. and Taylor D.S. 2002. Habitat use by fishes after tidal reconnection of an impounded estuarine wetland in the Indian River Lagoon, Florida (USA). *Wetlands Ecology and Management* 10: 51–69.
- Rountree R.A. and Able K.W. 1993. Diel variation in decapod crustacean and fish assemblages in New Jersey polyhaline marsh creeks. *Estuarine Coastal and Shelf Science* 37: 181–201.
- Rozas L.P. 1992. Comparison of nekton habitats associated with pipeline canals and natural channels in Louisiana salt marshes. *Wetlands* 12: 136–146.
- Rozas L.P. 1995. Hydroperiod and its influence on nekton use of the salt marsh: a pulsing ecosystem. *Estuaries* 18: 579–590.
- Rozas L.P. and Minello T.J. 1999. Effects of structural marsh management on fishery species and other nekton before and during a spring drawdown. *Wetlands Ecology and Management* 7: 121–139.
- Rozas L.P. and Zimmerman R.J. 2000. Small-scale patterns of nekton use among marsh and adjacent shallow nonvegetated areas of the Galveston Bay estuary, Texas (USA). *Marine Ecology Progress Series* 193: 217–239.
- Ruiz G.M., Hines A.H. and Posey M.H. 1993. Shallow water as a refuge habitat for fish and crustaceans in nonvegetated estuaries – an example from Chesapeake Bay. *Marine Ecology Progress Series* 99: 1–16.
- Talbot C.W. and Able K.W. 1984. Composition and distribution of larval fishes in New Jersey high marshes. *Estuaries* 7: 434–443.
- Talbot C.W., Able K.W. and Shisler J.K. 1986. Fish species composition in New Jersey salt marshes: effects of marsh alterations for mosquito control. *Transactions of the American Fisheries Society* 115: 269–278.
- Thomas B.E. and Connolly R.M. 2001. Fish use of subtropical saltmarshes in Queensland, Australia: relationships with vegetation, water depth and distance onto the marsh. *Marine Ecology Progress Series* 209: 275–288.
- Vance D.J., Haywood M.D.E., Heales D.S., Kenyon R.A. and Loneragan N.R. 1998. Seasonal and annual variation in abundance of postlarval and juvenile banana prawns, *Penaeus merguensis*, and environmental variation in two estuaries in tropical northeastern Australia: a six-year study. *Marine Ecology Progress Series* 163: 21–36.
- Weinstein M.P. 1979. Shallow marsh habitats as primary nurseries for fishes and shellfish, Cape Fear River, North Carolina. *Fishery Bulletin* 77: 339–357.