Artificial waterway design affects fish assemblages in urban estuaries

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Fishes were collected from residential canal estates leading directly off natural estuaries and artificial lakes with control tidal exchange in south-east Queensland, Australia, to test the model that the difference in artificial waterway design affects fish assemblages. A total of 17 779 fish representing 52 species was caught, including 23 species of economic importance (45% of the catch). Total fish abundance and species richness differed little among sites within or between the two artificial habitats (canals and lakes) in either of the seasons sampled (winter and spring). Multivariate analysis showed, however, that assemblages differed among sites within the same artificial habitat. The differences were best explained by the distance sites were from open water, while salinity, water temperature and dissolved oxygen explained little of the variability. Most species were found in canals and lakes, but there were enough differences in composition between the habitats to detect a difference in both seasons (significant ANOSIM tests). Salinity was lower in lakes because of the tidal restrictions, and while this was only weakly correlated with differences in fish assemblages, it had the most explanatory power of any environmental variable. New recruits arrived later in lakes than canals, perhaps because of the barriers to tidal flow. A survey the following year showed that differences among individual lakes were consistent through time, offering insights into the influence of different tidal barriers on fish assemblages. The design change from canals to lakes has a minor influence on fish assemblages and alters the timing of recruitment. © 2007 The Authors

INTRODUCTION

Increasing human population is placing pressure on coastal fish habitat in many places around the world (Cohen *et al.*, 1997). Among the many anthro-pogenic perturbations affecting estuarine environments (*e.g.* overfishing, fresh-water diversion, sedimentation and dredging), the most tangible is the uncontrolled expansion of residential development in coastal areas (Kennish,

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2002). In particular, property developers are excavating or reclaiming areas of natural wetland habitat to extend the limited areas of usable waterfront land by constructing artificial waterway developments (Baird *et al.*, 1981; Lincoln Smith *et al.*, 1995; Maxted *et al.*, 1997). An obvious consequence of these created urban waterways is the fragmentation or loss of coastal wetland habitats (Lee *et al.*, 2006). This raises concerns about the ecological value of artificial waterways and their role as fish habitat in the coastal zone (Rozas, 1992; Connolly, 2003).

Artificial waterways differ from the shallow estuarine habitats they replace. Water quality is generally poorer in artificial waterways because of their greater depth, and because they usually receive untreated urban stormwater runoff and have limited water circulation (Maxted *et al.*, 1997; Waltham, 2002). They also lack the conspicuous macrophytes of natural vegetated wetlands. Artificial waterways are considered to offer some habitat value for local fisheries by extending the amount of available estuarine habitat (Baird *et al.*, 1981; Morton, 1992). This extent is becoming increasingly evident in situations where they are created in terrestrial rather than aquatic environments (Zigic *et al.*, 2005). A survey of a canal system in south-east Queensland shortly after it was constructed showed that the assemblage of fishes in canals was similar to that of the unvegetated channel in a natural estuary (Morton, 1989). In spite of the extent of artificial waterway systems in certain locations, however, differences in fish assemblages between natural and artificial estuarine habitats have rarely been assessed, and the usefulness of artificial waterways cannot be stated with any confidence.

In many places, these created urban waterways can provide an order of magnitude extension to the amount of coastal wetland habitats available to fishes. Australia has the greatest expanse of residential canal estates in the world, with *c.* 440 km of linear urban waterways. By comparison, Florida Keys residential estates provide 175 km of linear habitat (Mactec, 2003). Residential canals have also been built in other states of the U.S.A. (*e.g.* California and Texas), but the linear extent is smaller there, typically <15 km, as it is in other places such as South Africa (Baird *et al.*, 1981), United Arab Emirates (unpubl. data), and even Venice [40 km linear habitat (Dabala *et al.*, 2005)]. In Australia, the Gold Coast in south-east Queensland has the most canals. For example, canal estates have increased the linear extent of the Nerang River estuary from an original 20 km to >150 km (Waltham & Connolly, 2006).

In Queensland, in recent years, it has been recognized that canal estates have caused hydraulic problems. The expansion of the Nerang River estuary, for example, has increased the tidal prism, causing erosion to downstream residential properties (Zigic *et al.*, 2005). In order to circumvent similar hydraulic problems on other waterways in south-east Queensland where waterfront developments are planned, property developers have been forced by legislation to build lake developments with a tidal barrier (*e.g.* locks, weirs, gates and pipes; Environmental Protection Agency, 2005). The design change to artificial lakes allows further property developments, while only marginally increasing the tidal prism of local waterways (Zigic *et al.*, 2005). The quality of these lakes as fish habitat might be affected by restrictions on fish access due to tidal barriers or other hydrological differences such as the greater depth of lakes. Fish

assemblages in the lakes have not previously been studied, so to date no comparison with canal fish assemblages has been possible.

The construction of barriers can prevent fish movement between different habitats in fresh (Morita & Yamamoto, 2002) and estuarine waters (Rozas, 1992). In coastal areas, the type of fish barrier design is important (Cattrijsse *et al.*, 2002). This has been shown in Louisiana, U.S.A., where permanently open or partially open channels dug for oil and gas pipes allow fishes access to adjacent marsh habitats, while closed channels totally restrict fish movement (Neill & Turner, 1987). Another implication is that water quality in upstream reaches can be adversely affected by reduced tidal exchange. Pollard & Hannan (1994) examined estuarine fish assemblages upstream and downstream of flood mitigation barriers in the Clarence River, Australia. They found that fish assemblages above the gates were dominated by freshwater species, because of lower salinity, compared to typical estuarine assemblages below the gates. In this example, Pollard & Hannan (1994) recommended structural modifications to barriers to improve water quality in the upper estuary and enable greater access to the previously unavailable estuarine fish habitat. Similar efforts to reconnect these modified wetlands to the coastal zone have occurred in U.S.A. (Burdick *et al.*, 1997; Haltiner *et al.*, 1997) and Europe (Wolter & Arlinghaus, 2003).

In the current study, fishes were collected from the two artificial estuarine habitats, canals and lakes, to test the model that the design change from canals to lakes results in different fish assemblages, for example through restricted access. In the following year, fishes were collected from several of the same artificial lakes to investigate whether patterns in fish assemblages were consistent from year to year. Ultimately, the study is intended to provide coastal managers with information to assist in the management and planning of developments in coastal urban waterways.

MATERIALS AND METHODS

SITES AND SAMPLE COLLECTION

Fishes were collected during the day in austral winter (July) and spring (October) 2001, from five canals and five lakes, each with four replicate sites, in south-east Queensland, Australia (Fig. 1). Each location (canal or lake) was a different distance from the open waters of Moreton Bay (from 0·7 to 18·0 km, measured using GIS-based software on digital maps, and taken as the shortest route by water between each location and the open waters of the bay). Canals and lakes have a homogeneous shoreline and benthic substratum of sand and mud and they are deeper (7–8 m) than adjacent natural estuaries (1–3 m), though lakes are far deeper (7–28 m) and wider (60–700 m) than canals (30–150 m) because of floodwater storage requirements (Waltham, 2002). The lakes had different mechanisms for reducing tidal exchange (concrete pipes of *c.* 1 m diameter, shallow concrete sills and modification of soft-sediment entrance width and depth). In a second study in 2002, fishes were collected from three of the five lakes, again in July and October. These three lakes are part of the Burleigh Lakes system, the largest and most extensive artificial lake system in south-east Queensland (280 ha of open water). Fishes were collected at all sites over unvegetated, soft-sediment edges of artificial waterways using a large (70 m × 4 m, 18 mm stretch mesh) and small (2 m × 1 m, 1 mm stretch mesh) seine; catches from the two nets were pooled to make

FIG. 1. Southern Moreton Bay showing extent of artificial waterways (■) and natural estuaries (□). Canals (△): P, Paradise Point; H, Hollywell; R, Runaway Bay; S, Sorrento Waters; M, Mermaid Waters. Lakes (○): V, Sovereign Waters; L, Lake Intrepid; A, Miami Lake; B, Heron Lake; W, Swan Lake.

the sample. Fishes were identified, counted and a random selection (up to 20 individuals) of each species was measured for total length (L_T) to the nearest mm. Salinity, temperature and dissolved oxygen were measured at each site with a water quality multiprobe, 1 m below the surface [edge areas are generally well mixed compared to deeper areas of artificial waterways which are stratified (Morton, 1992)].

DATA ANALYSIS

Abundances (all species combined, and species for which >20 individuals were caught) and species richness were analysed using a nested ANOVA to test for differences between habitats and among locations within habitats, in each season. Data were tested for homogeneity of variance and normality prior to analysis and transformed using $\log_{10}x$ where necessary. Non-metric multidimensional scaling (NMDS) was used to ordinate groups from biotic similarity matrices using the Bray–Curtis index, on both

raw and presence and absence (p/a) data. Differences in fish assemblages (1) among locations in canals and lakes separately, (2) between the two habitats and (3) between years in the Burleigh Lakes system, across the two seasons in each case, were tested for significance using a two-factor analysis of similarities [ANOSIM (Clarke, 1993)]. When significant differences were detected, the R-statistic was used to determine the extent of the difference, and similarity percentages (SIMPER) elucidated which species contributed most to the difference [based on having a high mean:S.D. ratio (Clarke, 1993)]. BIOENV was used to assess relationships for single or combinations of environmental factors (the three physicochemical factors plus distance to open water) with the composition of the fish assemblage, using the weighted Spearman coefficient (ρ_w) recommended by Clarke & Ainsworth (1993). The L_T frequency distributions for the common (present at >5 sites), economically important species in canals and lakes were compared using Kolmogorov–Smirnov tests.

RESULTS

ENVIRONMENTAL CONDITIONS

Water was warmer in spring than winter but was similar in canals and lakes (Table I). Canals had higher salinity than lakes, with a difference of 11 and 7 in winter and spring, respectively. Dissolved oxygen concentrations were higher in winter than spring for both habitats, and canals had consistently lower concentrations than lakes.

SPECIES COMPOSITION

In the first study, 17 779 fishes representing 52 species were caught (Table II). Twenty-three of the species are of economic importance in the region, representing c. 45% of the total catch. The most abundant species were *Herklotsichthys castelnaui* (Ogilby), *Gobiopterus semivestitus* (Munro), *Gerres subfasciatus* Cuvier and *Ambassis jacksoniensis* (Macleay) with respective contributions of 15·4, 13·4, 11·7 and 11·4%.

COMPARISON OF FISH ASSEMBLAGES AMONG LOCATIONS WITHIN HABITATS

Fish assemblages differed among canals, with a consistent pattern across seasons (same pattern for raw data and p/a, only p/a shown: ANOSIM, season,

TABLE I. Water quality conditions (mean $\pm$ S.E.) in canals and lakes in 2001 during winter and spring ($n = 20$). Values within rows having different superscript lower case letters are significantly different according to Tukey's *post hoc* test ($P < 0·05$)

Variable	Winter		Spring	
	Canal	Lake	Canal	Lake
Temperature (° C)	18·3 $\pm$ 0·4[b]	18·7 $\pm$ 0·2[b]	22·2 $\pm$ 0·2[a]	23·5 $\pm$ 0·2[a]
Salinity	30·1 $\pm$ 0·6[a]	19·6 $\pm$ 1·2[c]	29·9 $\pm$ 0·6[a]	22·3 $\pm$ 1·0[b]
Dissolved oxygen (mg l^{-1})	7·3 $\pm$ 0·2[b]	8·1 $\pm$ 0·2[a]	6·9 $\pm$ 0·2[c]	7·5 $\pm$ 0·2[b]

TABLE II. Relative abundance (%) of the common fish species, total abundance and species richness in canals and lakes caught during winter and spring in 2001

Family/species name	Common name	Canal		Lake	
		Winter	Spring	Winter	Spring
AMBASSIDAE					
Ambassis jacksoniensis	Port Jackson glassfish	18	28	2	1
Ambassis marianus	Yellow perchlet	2	3	3	2
CARANGIDAE					
Scomberoides lysan	Queenfish	<1	<1	1	—
CLUPEIDAE					
*Herklotsichthys castelnaui**	Southern herring	24	8	1	25
*Nematalosa erebi**	Bony bream	2	<1	10	1
ELEOTRIDAE					
Philypnodon grandiceps	Flat headed gudgeon	<1	<1	3	5
GERREIDAE					
*Gerres subfasciatus**	Common silver belly	18	24	4	2
GOBIIDAE					
Favonigobius exquisitus	Exquisite sand-goby	3	5	20	28
Favonigobius lateralis	Long finned goby	5	—	2	—
Gobiopterus semivestitus	Glass goby	16	7	8	20
Redigobius macrostoma	na	<1	—	5	1
HEMIRAMPHIDAE					
*Arrhamphus sclerolepis**	Snub nosed garfish	2	<1	<1	<1
PSEUDOMUGILIDAE					
Pseudomugil signifer	Pacific blue eye	<1	—	14	3
MUGILIDAE					
*Liza argentea**	Tiger mullet	1	7	4	1
*Mugil cephalus**	Sea mullet	2	3	6	2
*Myxus elongates**	Sand mullet	<1	1	<1	—
*Valamugil georgii**	Fantail mullet	1	<1	<1	<1
PLATYCEPHALIDAE					
*Platycephalus fuscus**	Dusky flathead	<1	<1	<1	<1
SCATOPHAGIDAE					
Selenotoca multifasciata	Striped butterfish	<1	1	<1	<1
SILLAGINIDAE					
*Sillago ciliata**	Summer whiting	<1	3	<1	2
*Sillago maculata**	Trumpeter whiting	<1	1	<1	—
SPARIDAE					
*Acanthopagrus australis**	Yellowfin bream	2	5	6	2
*Rhabdosargus sarba**	Tarwhine	<1	<1	1	<1
TETRAODONTIDAE					
Tetractenos hamiltonii	Common toadfish	<1	1	<1	<1
	Total number of fishes	5651	3314	4598	4216
	Total number of fish species	43	38	35	33
	Mean ± s.e. total abundance	278 ± 31	209 ± 50	205 ± 47	164 ± 22
	Mean ± s.e. species richness	10 ± 0	9 ± 1	10 ± 0	7 ± 1

—, no catch; na, no common name.

*, economically important species.

global $R = 0.07$, $P > 0.05$, location, $R = 0.46$, $P < 0.01$) [Fig. 2(a)]. Pair-wise comparisons showed Mermaid and Sorrento to be well separated from other canals (ANOSIM, $P < 0.01$ in all cases), with Runaway Bay, Hollywell and Paradise Point supporting similar fish assemblages (all $P > 0.05$). *Ambassis jacksoniensis* was a good separator of canals with higher abundance in Mermaid than all other canals (SIMPER). Distance from open water was the best predictor of differences among canals, with slighter fewer species and total abundance of fishes in canals furthest from open water compared to those located adjacent to open water ($\rho_w = 0.52$).

Fish assemblages also differed among lakes, with a consistent pattern across seasons (same patterns for raw and p/a data, only p/a shown: ANOSIM, season, global $R = 0.06$, $P > 0.05$, location, $R = 0.58$, $P < 0.01$) [Fig. 2(b)]. Pair-wise comparisons showed Sovereign Waters and Swan Lake to be well separated from other lakes and each other (ANOSIM, $P < 0.05$ in all cases). Assemblages in Heron Lake were also significantly different from those of other lakes, but
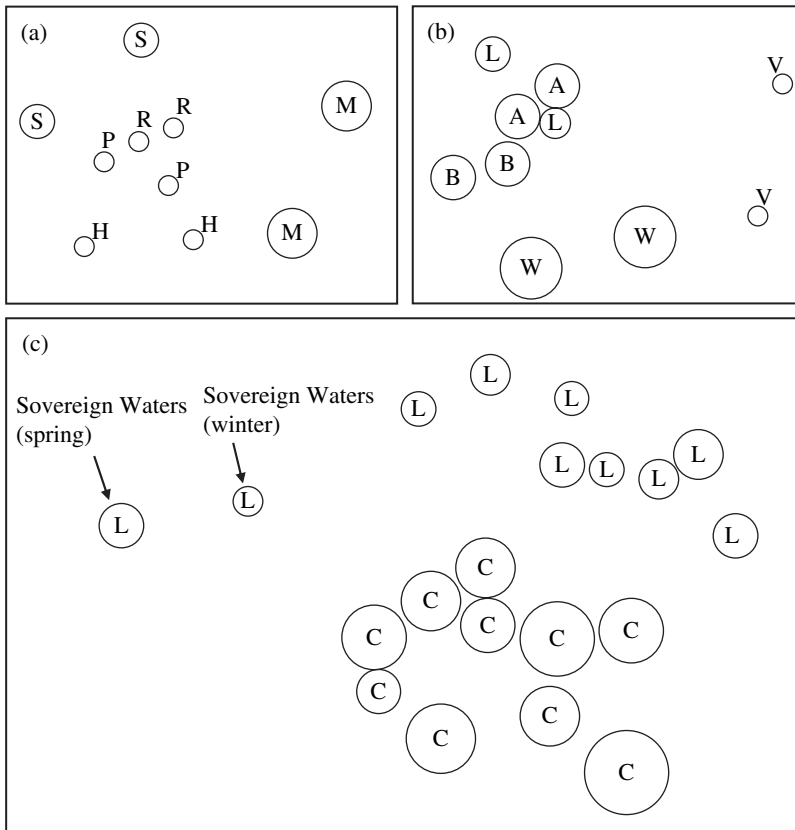

FIG. 2. Two-dimensional MDS ordination plot based on presence and absence data in 2001 for (a) canals (see Fig. 1); stress 0·04, (b) lakes (see Fig. 1); stress 0·06 and (c) canals (C) and lakes (L); stress 0·17. Seasons combined in all plots. Diameter of circle is proportional to distance from open water in (a) and (b) with smallest circle = 0·7 km, largest = 18 km; for (c) diameter of circle is salinity with smallest circle = 10, largest = 33.

were closer to Lake Intrepid and Miami Lake, two lakes with similar assemblages ($P > 0.05$), while Heron Lake was intermediate among lakes (ANOSIM, $P < 0.05$ in all cases). *Gobiopterus semivestitus* was a good separator of lakes with higher abundance in all lakes except Sovereign and Swan Lake (SIMPER). Distance from open water was the best predictor of differences among lakes, with slightly fewer species and total abundance of fishes in lakes furthest from open water compared to those located adjacent to open water ($\rho_w = 0.49$).

COMPARISON OF FISH ASSEMBLAGES AND FISH SIZES BETWEEN HABITATS

There was a consistent difference in fish assemblages between canals and lakes across seasons (similar pattern for raw and p/a data, only p/a shown: ANOSIM, seasons, global $R = 0.08$, $P > 0.05$, habitat, $R = 0.47$, $P < 0.05$) [Fig. 2(c)]. The difference between habitats was attributable to greater abundances of *G. subfasciatus* and *Myxus elongatus* Günther in canals, and greater abundances of *Redigobius macrostoma* (Günther) and *Nematalosa erebi* (Günther) in lakes [SIMPER (Fig. 3)]. Differences among locations were best explained by salinity ($\rho_w = 0.69$), and given that salinity was usually lower in lakes, this also underlies the difference between habitats.

Overall, fish abundances and species richness did not vary strongly between habitats, but locations within habitats sometimes varied significantly. In winter, total abundance and species richness did not differ significantly between habitats or among locations within habitats [total abundance nested ANOVA: habitat, d.f. = 1 and 8, $P > 0.05$; location (within habitat) d.f. = 8 and 30, $P > 0.05$; species richness: habitat d.f. = 1 and 8, $P > 0.05$; location (within habitat) d.f. = 8 and 30, $P > 0.05$]. In spring, total abundance and species richness differed among locations but not between habitats [total abundance nested ANOVA: habitat d.f. = 1 and 8, $P > 0.05$; location (within habitat) d.f. = 8 and 30, $P < 0.001$; species richness: habitat d.f. = 1 and 8, $P > 0.05$; location (within habitat) d.f. = 8 and 30, $P < 0.001$]. Differences in total abundances and species richness among locations within habitats in spring were due to an overall lower catch (abundance and richness) in Sovereign Waters than other lakes. This resulted in the obvious separation of this location on the ordination plot [Fig. 2(c)]. Univariate differences were not significant in winter, and although Sovereign Waters was separate again on the ordination plot, it was closer to other locations than in spring.

The L_T of fishes varied between habitats in many instances (Table III). In winter, there was a lower proportion of small individuals (new recruits) in lakes than canals for several species, resulting in a larger average size in lakes [*G. subfasciatus*, *H. castelnaui*, *Rhabdosargus sarba* (Forsskål), *Liza argentea* (Quoy & Gaimard) and *N. erebi*]. By spring, the smallest cohorts absent in winter were evident in lakes, and several species were therefore larger on average in canals [*Acanthopagrus australis* (Günther), *Sillago ciliata* Cuvier, *H. castelnaui*, *L. argentea* and *N. erebi*]. The later recruitment of the smallest cohort is shown for *L. argentea* (Fig. 4).
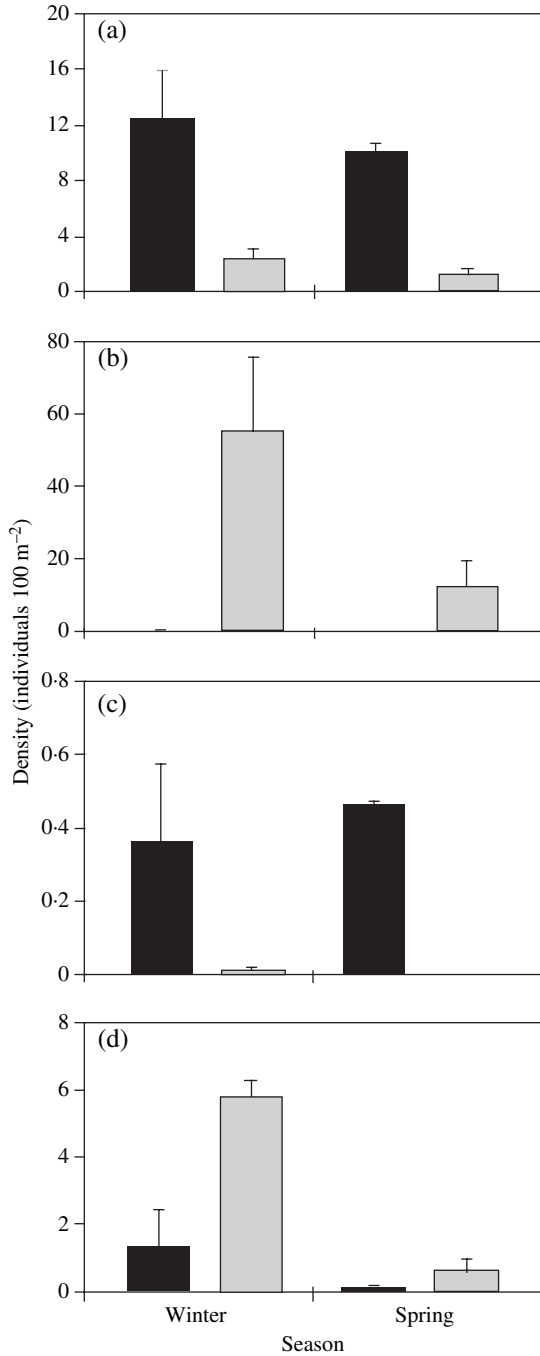
FIG. 3. Mean + s.e. densities of key species in 2001 in canals and lakes of (a) *Gerres subfasciatus*, (b) *Redigobius macrostoma*, (c) *Myxus elongatus* and (d) *Nematalosa erebi* in canals (■) and lakes (□) in 2001. Seasons are presented separately to illustrate the consistent difference between habitats. $n = 5$ locations for each habitat.

TABLE III. Results of Kolmogorov–Smirnov tests comparing the total length distributions between canals (C) and lakes (L) in 2001 of the most common economically important species. Letters in parenthesis indicate the artificial habitat with longer fish

Species	Winter	Spring
Acanthopagrus australis	NS	***(C)
Sillago ciliata	NS	*(C)
Mugil cephalus	NS	***(L)
Gerres subfasciatus	***(L)	***(L)
Herklotsichthys castelnaui	***(L)	***(C)
Arrhamphus sclerolepis	NS	—
Sillago maculata	NS	—
Rhabdosargus sarba	***(L)	—
Liza argentea	*(L)	*(C)
Nematalosa erebi	*(L)	*(C)

—, too few fish; NS, not significant.
*, <0.05; **, <0.01; ***, <0.001.

INTER-ANNUAL PATTERNS IN FISH ASSEMBLAGES IN BURLEIGH LAKES SYSTEM

In the Burleigh Lakes system, there was no significant multivariate difference between seasons, but there was a clear difference between years (two-way
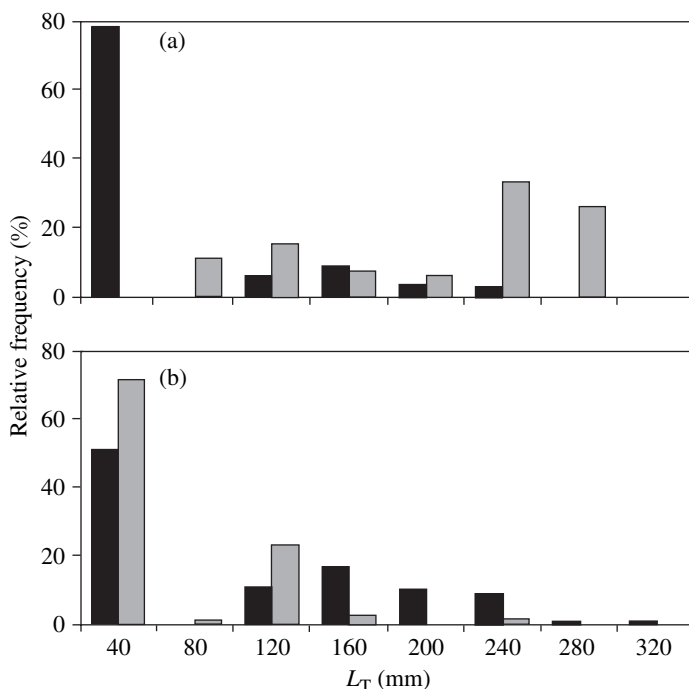

FIG. 4. Total length-frequency distributions for *Liza argentea*, one of the species showing differences between habitats, in 2001, in canals (■) and lakes (▨) for (a) winter and (b) spring.

ANOSIM: season, global $R = 0.06$, $P > 0.05$; year, $R = 0.46$, $P < 0.05$) (Fig. 5). The separation of years was due to a higher catch rate of *Philypnodon grandiceps* (Krefft) in 2001, while *Pandaka lidwilli* (McCulloch) was more prevalent in 2002 (Fig. 6). Salinity best explained patterns among locations ($\rho_w = 0.28$), and given that salinity was higher in 2002 (range 21–31) across locations than 2001 (10–24), this also explains differences between years.

DISCUSSION

FISH ASSEMBLAGES IN ARTIFICIAL URBAN WATERWAYS

Artificial canals and lakes both supported a wide variety of fishes, including many estuarine species of economic importance that also occur within natural wetlands in the region. Although differences in the fish assemblages of artificial and natural systems have not been quantified, the obvious difference is that artificial waterways lack those species that are specific to certain natural, vegetated habitats. For example, several species of pipefishes (Syngnathidae) are common in seagrass meadows of Moreton Bay adjacent to the artificial waterways studied here (Johnson, 1999; Takahashi *et al.*, 2003), yet none has been recorded in the artificial waterways. This supports the conclusion of several studies that artificial waterways probably contribute to fisheries production in much the same way as unvegetated areas of natural estuarine reaches (Trent *et al.*, 1972; Weinstein *et al.*, 1977; Baird *et al.*, 1981; Morton, 1989). Where artificial waterways have replaced shallow, vegetated wetlands, the outcome is more likely to be one of a net loss in fisheries production (Maxted *et al.*, 1997). Early developments in south-east Queensland were constructed from such shallow wetlands. In south-east Queensland, this practice is no longer condoned, however, and artificial waterways are now, in the main, required to be
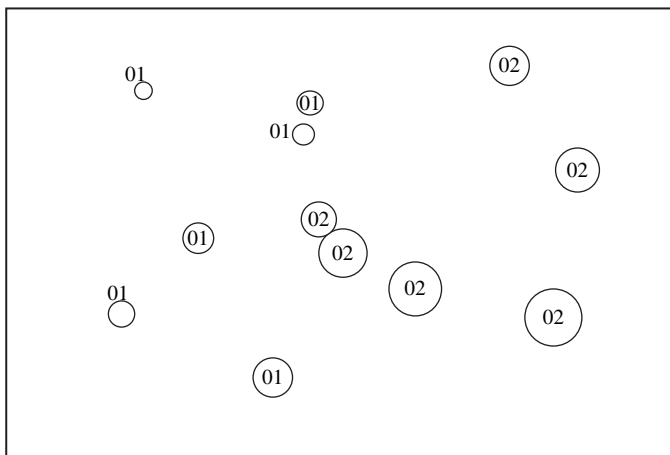
FIG. 5. Two dimensional MDS ordination plot based on presence and absence data for Burleigh Lakes system in 2001 (01) and 2002 (02); stress 0·1. Seasons are combined within years. Diameter of circle is proportional to salinity with smallest circle = 11, largest = 33.
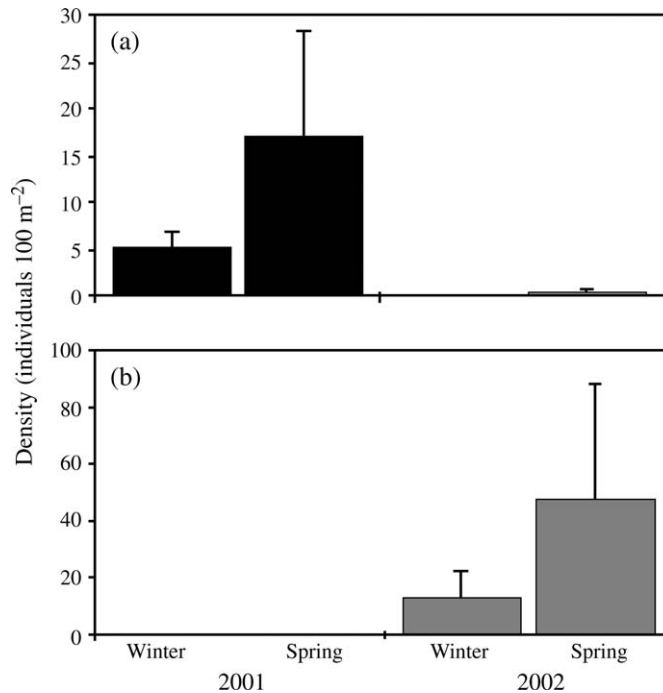
FIG. 6. Mean + S.E. densities of key species in Burleigh Lakes system (a) *Philypnodon grandiceps* and (b) *Pandaka lidwilli* in 2001 (■) and 2002 (■).

constructed from terrestrial habitat. Estuarine systems created in this fashion presumably provide an extension to the amount of unvegetated estuarine habitat for fishes.

VARIABILITY WITHIN THE SAME ARTIFICIAL HABITAT TYPE

Clear differences occurred in fish assemblages among different locations within the same artificial habitat type. Patterns in differences among locations were best explained by the distance from open water. Canals and lakes located similar distances from the open waters of Moreton Bay grouped together on ordination plots because they had a similar suite and total abundance of fishes, while systems located much further or closer to open water supported different assemblages. Variability among fish assemblages at different sites within an artificial waterway type has been shown previously. In the study of a single artificial canal system in New South Wales, Australia, fish assemblages varied among sites, and in that study, too, the differences were considered to be related to the distance from open water (Lincoln Smith *et al.*, 1995). This apparent effect of distance from open waters on fish assemblages has implications for fish utilization. Evidence from natural estuaries shows that with increasing distance from open coastal waters, the contribution of marine fishes to the assemblage is minimal, instead it is dominated by a few, typically freshwater, fish species (West &

King, 1996; Wagner, 1999). A similar trend is evident in lakes and canal in the present study, and will probably become more pronounced because of the trend in south-east Queensland to connect new systems to the furthest reaches of existing networks (Zigic *et al.*, 2005).

Several water-quality variables have been shown to influence fish distributions in estuaries, with the most commonly reported being water temperature, salinity and dissolved oxygen (Blaber & Blaber, 1980; Loneragan *et al.*, 1987; Marshall & Elliott, 1998; Whitfield, 1999; Martino & Able, 2003; Barletta *et al.*, 2005). In this study, water-quality variables explained little of the variability among locations within lakes and canals. This is in itself an interesting finding, given that water quality in artificial waterways is highly variable (Maxted *et al.*, 1997; Kruczynski, 1999). Locations in the current study were placed haphazardly within the overall extent of available habitat, and locations such as dead-end areas that have reduced water exchange and therefore likely to have the poorest water quality were not well represented. Future focus on the differences among locations within canal and lake systems is warranted, and it is recommended that locations with poor water quality and circulation be specifically included.

IMPLICATIONS OF DESIGN CHANGE ON ARTIFICIAL URBAN WATERWAYS

While the design change from canals to lakes has resolved the hydraulic concerns associated with extending these created waterways (Zigic *et al.*, 2005), it would seem also to have resulted in a change in the fish assemblages populating these urban waterways. While many of the fish species present in canals were also found in lakes, some fish species were present in one habitat and not the other. Whilst it is possible that the pattern of fish assemblages is independent of connection because, for example, fishes spawn in lakes, it seems more likely that this pattern is a response to the limited tidal exchange of lakes, which causes salinity in lakes to be more strongly affected than canals. Although many estuarine fishes are able to tolerate the full salinity range of an estuary, only a small sub-set of these species can tolerate extended periods of very high or low salinity (Kennish, 1990). In the local context, for example, *N. erebi* is found mostly over backwaters of coastal floodplains with low salinity (Allen *et al.*, 2003). It is not surprising, therefore, that this species occurred predominantly in lakes in the current study, and contributed strongly to the difference in assemblages between canals and lakes. Certain estuarine fishes are also known to occur in ponded rather than flowing waters (Strydom *et al.*, 2003), and a difference in current might directly affect fish assemblages in the two habitats. One aspect of the fish assemblage requiring further research is the differential distribution of piscivorous fishes between canals and lakes, particularly given that the gear used here would not effectively include this component of the assemblage.

Despite the restricted tidal exchange between lakes and downstream reaches of the estuary, during prolonged periods of low freshwater runoff from rainfall, lakes can experience higher salinities more similar to canals (Waltham, 2002). Salinity in Burleigh Lakes was higher in 2002 than in 2001, and while many

of the fish species were present in both years under the different salinity regimes, some species were absent in 2002, leading to a detectable separation of years in ordination plots. This supports the understanding that subtle differences occur among assemblages in these artificial waterways. The majority of fishes are able to tolerate the range of salinity conditions encountered in these urban waterways. A few species are sensitive to prolonged periods of either high or low salinities, however, and it is the presence and absence of these species that drives differences in assemblages. The salinity of waterways with urbanized catchments potentially changes rapidly after rainfall (Walsh, 2004), a factor no doubt underlying the rapid salinity changes in lakes previously measured in south-east Queensland (Waltham, 2002). Rapid salinity changes following high freshwater input have been shown to cause sudden reductions in the diversity and abundance of fishes in the Swan River estuary, Western Australia, followed by the slow return of many fish species as salinity again increased during dry periods (Kanandjembo *et al.*, 2001). Further research is required to determine whether this same pattern occurs in urban lakes following heavy rainfall events, in particular focusing on whether fishes are absent because of migration or mortality.

Sovereign Waters Lake was an exception to the overall pattern because it consistently supported fewer fishes than any other lake or canal system. Tidal flow to Sovereign Waters Lake is particularly restricted, occurring *via* a single, relatively small concrete pipe (1 m diameter). While this tidal control device achieves the intended hydraulic outcome (Development Consulting, 1999), it might be having a greater effect on fish passage than the barriers used in other artificial lakes studied here. After many years of tidal modifications in coastal wetlands areas of the U.S.A., attempts have been made to restore these once natural wetlands by partially or completely removing tidal barriers (Burdick *et al.*, 1997; Callaway & Zedler, 2004). This process of habitat restoration has been shown to improve the tidal exchange and subsequently water quality in upstream areas, and it has also given estuarine fishes greater access to previously unavailable habitats. The extent of tidal restriction, and therefore also of fish access, should be taken into consideration in the design of artificial waterways, with a view to maximizing the utilization of artificial waterways by fishes.

Perhaps the most significant influence of the design change is the apparent delay in the arrival of new recruits for several key species. Each lake examined here has a different mechanism for controlling tidal exchange but, compared to canals, all showed a consistent pattern of delayed arrival of new recruits. This bottleneck effect on fish movement has been shown in the inland channels of Europe where flow restriction devices have held back the arrival of fish species migrating to upstream habitats (Wolter & Arlinghaus, 2003). This study was not specifically designed to determine whether the timing of recruitment was influenced by the tidal barrier itself or a greater distance for new recruits to traverse before reaching lakes. The former is more likely, however, given that recruits were just as scarce in Sovereign Waters Lake as in other lakes, even though Sovereign Waters Lake is immediately adjacent to the open waters of Moreton Bay. The delayed recruitment evident in lakes is not necessarily detrimental to the utilization of these artificial systems as fish habitat, but

presumably leads to a different temporal pattern in the interaction between fish stocks of artificial and natural estuarine waters than exists for canals.

The construction of artificial waterways in south-east Queensland has provided new and altered habitats for many local fish species. The change from canal systems to lake developments with tidal-control devices results in minor changes to fish assemblages, and perhaps more importantly appears to alter the timing of recruitment of fishes to lakes by delaying arrival. Salinity tends to be lower in lakes than canals, and possibly fluctuates more rapidly in response to rainfall runoff, apparently influencing a small sub-set of species in the waterways. Overall, lakes, like canals, are used by a wide variety of fishes, including many species of economic importance. With planning regulations requiring that they be constructed from terrestrial rather than aquatic environments, the outcome is a massive extension in the amount of estuarine habitat available to fishes.

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