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Differences in trophodynamics of commercially important fish between artificial waterways and natural coastal wetlands

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Abstract

Extensive artificial waterways have replaced natural wetlands and created new estuarine habitats on the southern Queensland coast, Australia. Economically important fish species found in adjacent natural wetlands of mangrove, saltmarsh and seagrass also occur in the artificial waterways. Stable isotope analyses ($\delta^{13}C$, $\delta^{15}N$) were used to test whether the relative importance of basal sources of energy varied for foodwebs found in artificial (canals and tidal lakes) and natural waterways. None of the fish species differed in their isotope values between artificial waterways. In contrast, isotopic signatures of snub-nosed garfish (*Arrhamphus sclerolepis*; Hemiramphidae) varied greatly between natural and artificial waterways, having highly enriched $\delta^{13}C$ values (−10.5‰) in natural wetlands, demonstrating reliance on seagrass (−11.4‰), and significantly less enriched values (−19.0‰) in artificial waterways, consistent with either local algal sources (−19.8 to −20.4‰) or a mixture of seagrass and other less enriched autotrophs from adjacent natural wetlands. Isotopic signatures of sand whiting (*Sillago ciliata*; Sillaginidae) were also significantly more enriched in natural (−18.2‰) than artificial (−21.0‰) habitats, but means were not far enough apart to distinguish between different sources of nutrition. $\delta^{13}C$ values of yellowfin bream (*Acanthopagrus australis*; Sparidae) did not differ between artificial and natural habitats (about −20‰ in both). $\delta^{15}N$ values of fish varied among habitats only for *A. sclerolepis*, which in artificial waterways had values enriched by 2‰ over those in natural waterways. This was consistent with a shift from seagrass (relatively depleted $\delta^{15}N$) as a source in natural habitat to algal sources (relatively enriched $\delta^{15}N$) in artificial habitats. This study provides some of the first evidence that at least some fish species rely on different autotrophs in artificial waterways than in adjacent natural wetlands.

Keywords: artificial lake; Australia; canal; fisheries; foodweb; stable isotope analysis

1. Introduction

Increasing human population in coastal regions has caused extensive modification of the coastline for urban purposes in many places (Mitsch and Gosselink, 1993). This has led to concerns about the ecological function of urban waterways and their role as fish habitat (Wolter, 2001). Comparisons of distribution and abundance patterns of animals have been used as an indirect means of assessing whether ecological processes differ between artificial and natural waterways (Lindall et al., 1973; Lindall and Trent, 1975; Baird et al., 1981; Neill and Turner, 1987; Rozas and Reed, 1994). These comparisons, however, neither necessarily elucidate the underlying ecological role of urban wetlands, nor do they show the interaction between artificial and natural waterways.

The creation, since 1970, of hundreds of kilometres of artificial waterways in southern Queensland, designed to maximise the number of dwellings with a waterfront, has resulted in the loss of natural coastal wetlands supporting mangroves, saltmarsh and seagrass (Morton, 1992). Most of this development was in the form of canals created in the wetlands themselves or dug from adjacent land. Over the past 10 years there has been a shift away from the

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construction of narrow canals leading directly off estuarine stretches of coastal rivers to the construction of artificial tidal lakes with restricted exchange with natural estuarine waters (Zigic et al., 2002). Comparisons of the fish fauna in these artificial waterways and in adjacent natural wetlands have shown almost complete overlap in the species present (Morton, 1989, 1992). Although differences in the relative proportions of species are detectable, all of the economically important species found in adjacent estuarine waters have also been recorded in the artificial waterways (Morton, 1989, 1992). This is also true of canals developed elsewhere in the world (Baird et al., 1981; Lincoln-Smith et al., 1995). Furthermore, there is no compelling evidence that trophic structure (e.g. proportion of piscivores) differs between artificial and natural waterways.

Fish in coastal wetlands are mostly carnivorous (Klumpp et al., 1989; Edgar and Shaw, 1995) but, like all animals, they ultimately rely on autotrophic sources for energy (carbon) and nutrients. They are supported by foodwebs based on conspicuous autotrophs such as mangroves, saltmarsh plants and seagrass (e.g. Kwak and Zedler, 1997; Kirsch et al., 2002), sometimes supplemented with a contribution from inconspicuous algal sources such as benthic microalgae (BMA), microalgae epiphytic on seagrass, and phytoplankton (e.g. Moncreiff and Sullivan, 2001). Autotrophic production in artificial waterways in southern Queensland is restricted to algal sources (BMA and phytoplankton), although lawns of terrestrial grass occur at the edge of the waterways along much of their length. Organic matter is considered to be highly mobile in estuarine systems (Odum, 1984), and foodwebs supporting fish in artificial waterways might be driven either by local sources or allochthonous inputs of detrital material from adjacent natural wetlands.

Examination of the stomach contents of fish cannot determine the ultimate autotrophic source supporting them. The contents of not only the fish but also those of their prey would need to be determined and, with decreasing resolution, so on down to lower trophic levels, becoming impossible at the level of tiny animals or microbes. Furthermore, stomach contents show only what is ingested in the last meal, not necessarily what is assimilated. Stable isotope analysis is a more direct, albeit still qualitative, measure of the autotrophic source(s) supporting fish growth. Different autotrophs often have different ratios of the rare to common isotope of elements such as carbon and nitrogen, the two elements employed most frequently in estuarine foodweb studies, because of different photosynthetic pathways in the case of carbon (Hemminga and Mateo, 1996) and different sources for nitrogen (Peterson, 1999). The isotope ratio of sources is reflected in the tissue of consumers at whatever trophic level they occur, albeit with some fractionation shift, especially for nitrogen (Peterson, 1999).

This paper attempts to identify the autotrophic sources at the base of foodwebs supporting fish in artificial and natural wetlands. Stable isotope values were examined for three commercially important fish species found in canals and wetlands of southern Queensland. Two levels of comparisons were made: (1) differences in isotope values between artificial habitats (canals and lakes); and (2) differences between artificial and natural habitats. For each species of fish, the test was between four models:

1. fish have different source(s) in different habitat types, and this is reflected in different isotope values of fish;
2. fish have different source(s), but the same isotope values (e.g. because, by chance, different combinations of sources give the same mean value);
3. fish have the same source(s), reflected in isotope values of fish being the same; and
4. fish have the same source(s), yet have different isotope values (e.g. because of different fractionation rates in the different habitats).

If habitat modification has altered the autotrophic source for consumers then model 1 applies, if not, then model 3 applies. Models 2 and 4 are options that may confound the interpretation of the results. There are no records for differential fractionation among habitats of the same autotroph (model 4) but the possibility cannot be excluded because, for example, fish may obtain nutrition via different numbers of trophic levels despite having the same autotrophic source.

2. Methods

2.1. Sample collection and processing

Autotrophs and fish were collected from three habitat types: artificial canals, artificial tidal lakes, and natural wetlands adjacent to the artificial habitats. Collections were made from three separate locations in each habitat type in southern Moreton Bay, southeast Queensland (Fig. 1). These locations for different habitats were kilometres apart, and were interspersed wherever possible. Samples were collected from three randomly selected sites separated by hundreds of metres within each location, in August 2001. The natural wetland collections came from intertidal mudflats, mangrove forests, seagrass meadows and saltmarsh; only wetlands >1 km from artificial habitats were chosen, to minimise the likelihood of artificial habitats influencing trophic pathways.

I sampled all of the dominant autotrophs (Table 1) within each natural and artificial location by collecting a composite sample at each site in each location (total of nine samples in each habitat in which it occurred). Suspended organic material (seston) was sampled by

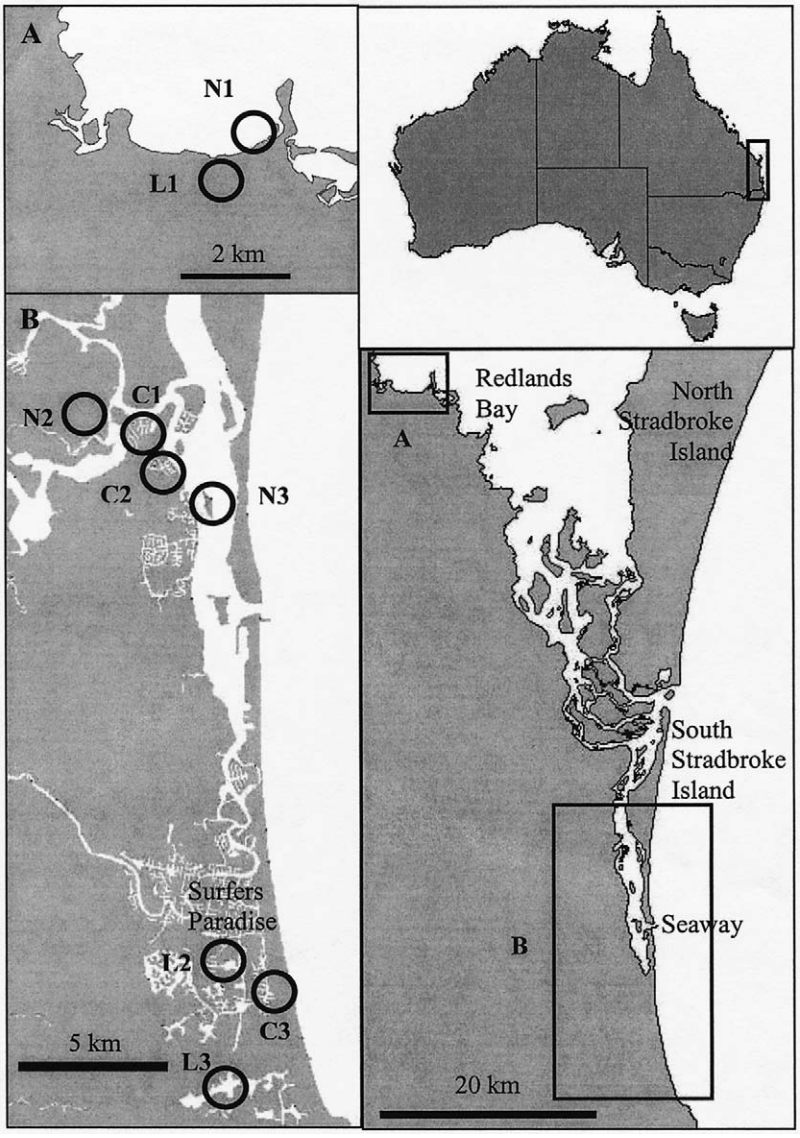

Fig. 1. Map of sampling locations. C, canal; L, artificial lake; N, natural wetland.

collecting material passing through 125 μm mesh (to remove the majority of zooplankton) but retained on 37 μm mesh (a mixture of diatoms and non-living organic material). Samples were frozen until processing. BMA were collected by scraping the upper centimetres of sediments from mud banks. About 100 mL of sediment was washed through 53 μm mesh to remove infauna. Material passing through the mesh was then washed through 5 μm mesh. Material retained on this mesh was added (~9 mL) to a centrifuge tube containing 21 mL colloidal silica (LUDOX™ AM30, density = 1.21) and centrifuged at 10 000 rpm for 10 min. A band of diatoms, some organic matter and silica particles was suspended at the top of the centrifuge tube. This band was removed and again washed in 5 μm mesh to remove the silica and any remaining microbes. Material retained on this mesh was then dried at 60 °C to constant weight. Inspection of samples showed that they consisted predominantly of microalgae (mainly diatoms) with occasional contamination by very fine detrital fragments.

Three fish species occurring in all the three habitat types were selected: sand whiting (*Sillago ciliata* Cuvier; Sillaginidae), snub-nosed garfish (*Arrhamphus sclerolepis* Günther; Hemiramphidae) and yellowfin bream (*Acanthopagrus australis* (Owen); Sparidae). These species were selected because, firstly, replicate individuals of similar sizes were caught at each site in each location across all habitat types. Secondly, there is great interest from coastal managers in species such as those that have major economic value in southeast Queensland. Thirdly, the species have different feeding strategies. *Arrhamphus sclerolepis* is omnivorous, *S. ciliata* strictly a benthic carnivore, and *A. australis* more of

Table 1
Autotrophs analysed from artificial and natural habitats

Location	Taxon	Common name	Wetland type	
			Artificial	Natural
Water column	Seston, including phytoplankton		Yes	Yes
Mud or rock surface	Benthic microalgae (diatoms)		Yes	Yes
	Caulerpa racemosa	Green macroalgae	Yes	Yes
Mangroves	*Avicennia marina*	Grey mangrove		Yes
	Aegiceras corniculatum	River mangrove		Yes
	Rhizophora stylosa	Red mangrove		Yes
Seagrass	*Zostera capricorni*	Eelgrass		Yes
	Halophila ovalis	Dugong grass		Yes
	Halophila spinulosa	Spiny dugong grass		Yes
Saltmarsh	*Sarcocornia quinqueflora*	Glasswort		Yes
	Sporobolus virginicus	Marine couch		Yes
	Suaeda australis	Seablite		Yes
Terrestrial	Poaceae	Grass	Yes	

a bentho-pelagic carnivore. No firm data exist on the dietary breadth of these species but I consider *A. australis* to have a broader diet than the other two species. Three individuals of each of these species were collected at each site using seine nets (three per site, three sites per location, three locations per habitat, total of 27 individuals per habitat per species). Individuals within the same size range in all habitats were selected. White muscle tissue collected from immediately below the anterior end of the dorsal fin was used for isotope analyses (Pinnegar et al., 2001). Stable isotope values of fish muscle tissue are an indicator of diet over periods of several weeks to months (Hesslein et al., 1993). As such my present results are indicative of diet over autumn and early winter for these species.

Frozen autotroph and fish samples were thawed, dried at 60 °C to constant weight, ground to a fine powder and analysed on an Isoprime Isotope Ratio Mass Spectrometer. Isotopic compositions of C and N were expressed in δ notation, as ‰ differences from an international standard (Vienna Pee Dee Belemnite for carbon, air for nitrogen):

$$\delta X = [(R_{\text{sample}}/R_{\text{standard}}) - 1] \times 10^3,$$

where X is ^{13}C or ^{15}N, and R is the corresponding ratio $^{13}C/^{12}C$ or $^{15}N/^{14}N$.

2.2. Data analysis

Biplots of δ^{13}C and δ^{15}N were used to examine the proximity of fish from different habitat types to potential source signatures. The main focus was on δ^{13}C, because the ratios of consumers follow closely to their source. Although consumer δ^{15}N values can be confounded with fractionation shifts at increasing trophic levels (Peterson, 1999), δ^{15}N nevertheless offers the possibility of resolving sources where δ^{13}C cannot (Peterson and Fry, 1987), and was used here for that purpose.

Autotroph taxa occurring in more than one habitat type (Table 1) were analysed using nested ANOVAs for differences in isotopic values among locations within a habitat type (e.g. different canal locations against each other), between artificial habitats (i.e. lake against canal locations), and between artificial and natural habitats (canals and lakes pooled if not significantly different). The same ANOVA models were used to test for differences in fish isotope values. All data were assessed for homogeneity of variances and normality prior to analysis. Data that failed to meet these assumptions were $\log_{10}(x)$ transformed and reassessed.

3. Results

3.1. Autotroph isotope values

Autotrophs were assigned to three main groups based on their δ^{13}C values: (1) those with enriched δ^{13}C values of −12 to −15‰, viz. seagrasses and saltmarsh grass (*Sporobolus virginicus*) in natural habitats and terrestrial grass from the edges of artificial waterways; (2) those with values of −18 to −23‰, viz. algal sources, including BMA, seston and macroalgae from both natural and artificial waterways; and (3) those with depleted δ^{13}C values around −28‰, viz. mangroves and saltmarsh succulents (*Sarcocornia quinqueflora* and *Suaeda australis*) (Table 2, Fig. 2).

δ^{13}C values varied among canal locations for only one autotroph taxon (BMA) and among lake locations for two taxa (BMA, seston; Table 2). Even so, there were no significant differences for δ^{13}C or δ^{15}N between canal and lake habitats (nested ANOVA: all $p > 0.05$). Values for the two artificial habitats were therefore pooled, and there were no significant differences between these and values for natural habitat for either C or N for any autotroph. However, all forms of algae

(BMA, macroalgae and seston, which includes phytoplankton) showed slight differences in either $\delta^{13}C$, $\delta^{15}N$ or both between natural and artificial waterways, with enriched $\delta^{15}N$ values and slightly depleted $\delta^{13}C$ values in artificial waterways (Fig. 2).

3.2. Fish isotope values

$\delta^{13}C$ values for fish fell in a narrower range than for autotrophs; fish values across all habitat types were between −22.0 and −18.2‰, except *Arrhamphus sclerolepis* collected from natural habitat which were highly enriched (−10.5‰). $\delta^{15}N$ values for fish were more enriched than for all of the autotrophs, typically by about 6–8‰ (Fig. 2). An assumption of a mean enrichment in $\delta^{15}N$ of about 3‰ for each trophic level (Peterson and Fry, 1987) would place the fish 2–3 trophic levels removed from autotroph sources.

Significant differences in $\delta^{13}C$ and $\delta^{15}N$ values were found among different canal locations and among different lake locations for *Acanthopagrus australis*, and in $\delta^{15}N$ among canal locations for *Sillago ciliata* (Table 3). This demonstrates the effectiveness of the procedure for detecting differences where they exist. Yet no differences were detected between canal and lake habitats for any of the fish species for either element (nested ANOVA: all $p > 0.05$), and values were therefore pooled for comparison with those from natural habitat.

The results of comparisons between artificial habitats (canals and lakes pooled) and natural habitat varied among the three fish species. *Arrhamphus sclerolepis* had a significantly more enriched mean $\delta^{13}C$ value in natural than in artificial habitats (Table 3, Fig. 2a). The highly

Table 2
Mean $\delta^{13}C$ and $\delta^{15}N$ values for autotrophs. Significant differences among replicate locations within an artificial habitat type are shown alongside the means. There were no significant differences between canal and lake habitats, or between artificial and natural habitats

	$\delta^{13}C$			$\delta^{15}N$		
Taxa	Canal	Lake	Natural	Canal	Lake	Natural
Aegiceras corniculatum	–	–	−28.2	–	–	2.5
Avicennia marina	–	–	−27.8	–	–	5.2
Benthic microalgae	−23.1**	−20.9**	−20.4	4.6	5.2	3.8
Caulerpa racemosa	−17.1	−20.1	−19.8	5.7	5.0	7.9
Halophila ovalis	–	–	−11.7	–	–	2.6
Halophila spinulosa	–	–	−13.9	–	–	3.3
Rhizophora stylosa	–	–	−28.1	–	–	0.7
Sarcocornia quinqueflora	–	–	−27.4	–	–	2.4
Seston, including phytoplankton	−23.4	−24.8*	−22.6	7.0	7.0	4.8
Sporobolus virginicus	–	–	−14.6	–	–	1.5
Suaeda australis	–	–	−28.6	–	–	3.6
Zostera capricorni	–	–	−10.9	–	–	3.7

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

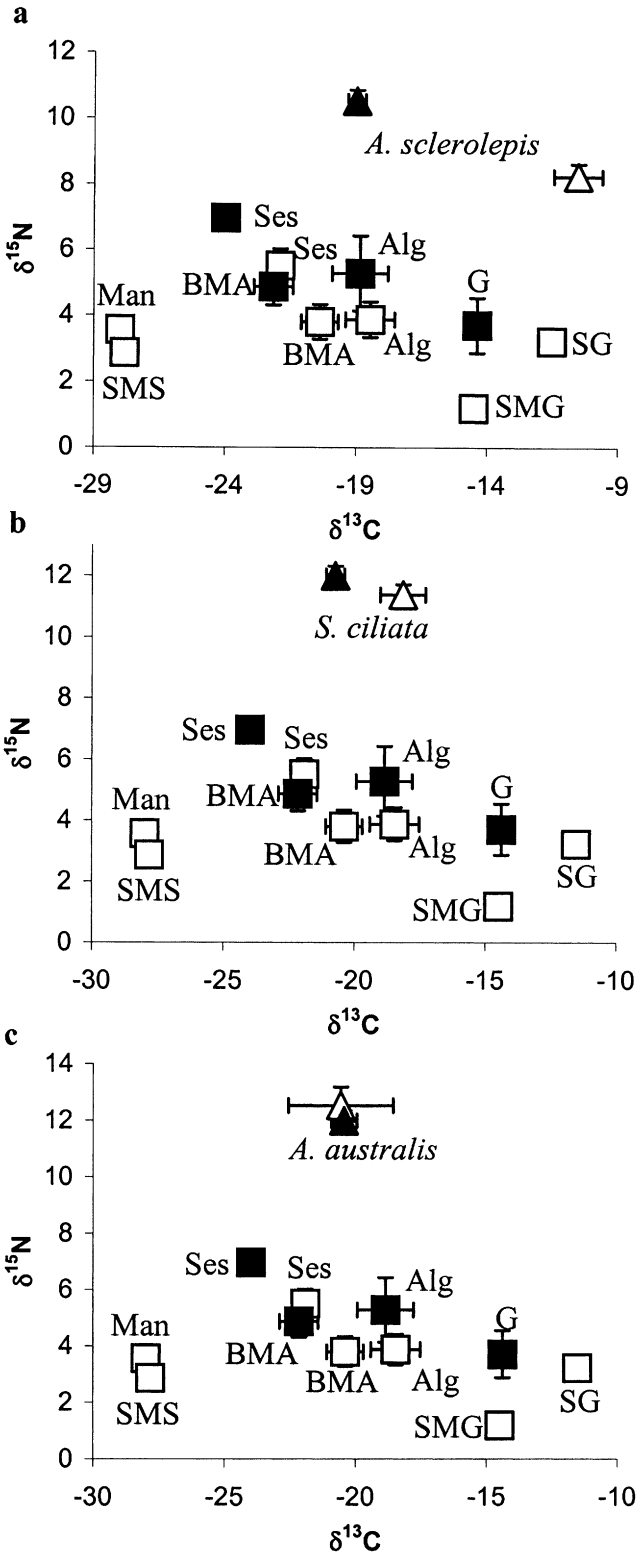

Fig. 2. $\delta^{13}C$ and $\delta^{15}N$ values for fish (a) *Arrhamphus sclerolepis*, (b) *Sillago ciliata*, and (c) *Acanthopagrus australis* (shown as triangles) and autotrophs (shown as squares). Results from artificial waterways (filled symbols) are distinguished from those of natural waterways (open symbols). All values are means ±SE, although some SE values are too small to show. Alg, macroalgae; BMA, benthic microalgae; G, terrestrial grass; Man, mangroves; SG, seagrass; SMG, saltmarsh grass; SMS, saltmarsh succulents; Ses, seston.

Table 3
Mean $\delta^{13}C$ and $\delta^{15}N$ values for fish. Sizes are total lengths, with a similar range for all habitat types. Significant differences among replicate locations within an artificial habitat type are shown alongside the means. Significant differences between artificial (Art) and natural (Nat) habitats are shown in a separate column. There were no significant differences between canal and lake habitats

			$\delta^{13}C$				$\delta^{15}N$			
Scientific name	Common name	Size range (cm)	Canal	Lake	Natural	Art vs Nat	Canal	Lake	Natural	Art vs Nat
Acanthopagrus australis	Yellowfin bream	9–17	−19.8**	−21.2**	−20.6		11.7**	12.4*	12.5	
Arrhamphus sclerolepis	Snub-nosed garfish	11–16	−18.6	−19.4	−10.5	***	9.8	11.3	8.3	*
Sillago ciliata	Sand whiting	13–29	−20.0	−22.0	−18.2	**	11.5**	12.9	11.4	

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

enriched $\delta^{13}C$ values in natural habitat demonstrated a reliance on seagrass, the only source with values as enriched. The more depleted $\delta^{13}C$ values (−19.0‰) in artificial waterways indicated a dietary shift either local algal sources or a mixture of seagrass and other less enriched autotrophs (e.g. mangroves) from adjacent natural wetlands. *Arrhamphus sclerolepis* had significantly higher $\delta^{15}N$ values, by about 2‰, in artificial than in natural habitats. *Sillago ciliata* also had $\delta^{13}C$ values significantly more enriched in natural (−18.2‰) than artificial (−21.0‰) habitats (Fig. 2b). However, mean $\delta^{13}C$ values were not far enough apart to clearly distinguish different sources of nutrition. $\delta^{13}C$ values of *Acanthopagrus australis* did not differ between artificial and natural habitats (Table 3, Fig. 2c). Fish $\delta^{15}N$ values were not significantly different (nested ANOVA: all $p > 0.05$) between artificial and natural habitats for *S. ciliata* or *A. australis*.

4. Discussion

Spatial variation in the isotopic signatures of fish, evident even in natural habitats across large scales (Jennings et al., 1997; Melville and Connolly, 2003), was successfully employed here to demonstrate that whilst fish species successfully colonise artificial waterways, they may utilise different trophic pathways in doing so. The three fish species analysed showed different patterns of variation across the habitat types and the models explaining the supply of energy and nutrients to foodwebs supporting these species therefore need to be considered separately.

For *Arrhamphus sclerolepis*, there was a large difference in $\delta^{13}C$ values (8.5‰) between artificial and natural habitats. This result is consistent with two of the four proposed models, namely 1 and 4. The different $\delta^{13}C$ values are most likely the result of *A. sclerolepis* relying on very different autotrophic sources in artificial and natural waterways (model 1). *Arrhamphus sclerolepis* caught in natural habitat have been shown to be omnivorous, ingesting large quantities of green seagrass leaves as well as zooplankton (Blaber and Blaber, 1980). Plant material can be assimilated even though the species lacks a gut microbial flora capable of digesting it. Instead, *A. sclerolepis* seems to rely on mechanical maceration to extract cell contents and then uptake via a mucus facilitated process (Tibbetts, 1997). The confamilial *Hyporhamphus melanochir* common in southern Australian waters also consumes green seagrass and zooplankton (Klumpp and Nichols, 1983), obtaining up to two-thirds of its carbon directly from seagrass and the remainder from zooplankton (Nichols et al., 1986). It is therefore not surprising that *A. sclerolepis* individuals caught in natural habitat in the present study had $\delta^{13}C$ values very similar to that of seagrass. Although seagrass is absent from artificial waterways in southern Queensland, making direct grazing there by *A. sclerolepis* impossible, the relatively mobile nature of energy and nutrients in estuaries means that seagrass material passively carried from adjacent natural habitat might still have been important to *A. sclerolepis*. Instead, *A. sclerolepis* in artificial waterways utilise a different trophic pathway, probably relying on microalgae, either through direct consumption or more likely via zooplankton prey. They may still utilise seagrass material, but as a smaller proportion of their ultimate autotroph sources, in conjunction with $\delta^{13}C$ depleted material from other wetland autotrophs such as mangroves. Examination of stomach contents would be useful in confirming among these possibilities.

The possibility exists that the differences in $\delta^{13}C$ values for *Arrhamphus sclerolepis* between artificial and natural habitats are not the result of different sources (model 4). Physiological processes that result in a consumer having a different isotope value from its assimilated food (fractionation), for example, might operate differently in different places. Fractionation rates vary depending on fish size, growth rates and food quality (Hesslein et al., 1993; Adams and Sterner, 2000; Overman and Parrish, 2001; Vander Zanden and Rasmussen, 2001), so differences in any of these factors between natural and artificial habitats could result in differential fractionation rates. However, $\delta^{13}C$ fractionation is typically small (0–2‰, Peterson and Fry, 1987), and has never been observed at anything like 8.5‰. Further, *A. sclerolepis* individuals were of a similar size in all habitats. It is very unlikely that the striking pattern in

δ^{13}C values of *A. sclerolepis* could be explained by anything other than different sources.

The elevation of δ^{15}N values for *Arrhamphus sclerolepis* in artificial relative to natural habitats was too slight (about 2‰) to make assertions about differences in sources between habitats. However, given that all artificial sources were more enriched than seagrass in δ^{15}N, the higher values for *A. sclerolepis* in artificial waterways is consistent with them obtaining their nutrition there from local sources.

Although *Sillago ciliata* δ^{13}C values were more depleted in artificial waterways, the difference of 2.8‰ between artificial and natural habitats is slight when interpreting foodwebs. The result is consistent with models 1 and 4, and either model could reasonably explain the small difference. *Sillago ciliata* are caught mainly over shallow, unvegetated areas in natural wetlands (Gray et al., 1996), and feed on polychaetes and benthic crustaceans (Burchmore et al., 1988); both are presumably available in artificial canals and lakes although the species and abundances present are unknown. A higher proportion of enriched δ^{13}C material from seagrass or saltmarsh grass in natural than in artificial habitats would explain the more enriched δ^{13}C value there. However, differential fractionation between the two habitat types could also result in a difference of 2.8‰. The size range of fish was similar, so size alone cannot be part of the explanation, but differences in food quality or growth rates might exist.

δ^{13}C values of *Acanthopagrus australis* did not differ between artificial and natural habitats, consistent with models 2 and 3. *Acanthopagrus australis* in natural wetlands in southern Queensland feeds mainly on crustaceans and other invertebrates (Morton et al., 1987), and these types of prey are available in canals and lakes. If these invertebrates obtain their nutrition from the same autotrophic sources in artificial and natural habitats then *A. australis* values would also be similar (model 3). Model 3 also includes the possibility that *A. australis* individuals move frequently enough among the locations sampled that the fish analysed were essentially part of a single, well mixed group across southern Moreton Bay over the period leading up to our sampling. However, juvenile *A. australis* of the size analysed in the present study have only ever been recorded as moving very small distances in tagging studies in Moreton Bay (Pollock, 1982). Furthermore, *A. australis* individuals were shown to have some fidelity to locations in the present study, in that significant differences in δ^{13}C and δ^{15}N values among locations of the same habitat type were found. It is possible that *A. australis* utilised different sources in artificial and natural habitats but that the source or combination of sources in the two habitat types had the same mean δ^{13}C and δ^{15}N values (model 2). For example, *A. australis* may utilise, via invertebrate intermediaries, a mixture of enriched and depleted autotrophs in natural wetlands but algal sources in artificial waterways; both scenarios would give a δ^{13}C value towards the middle of the range of autotroph values.

The explanation for the similar isotope values for *Acanthopagrus australis* across all habitats might lie in the greater breadth of diet for this species compared to the other two species studied. If the different prey types that it potentially feeds upon have a range of different feeding strategies, it is more likely that the prey themselves will, taken as a whole, have a range of autotroph sources. *Acanthopagrus australis* individuals will, under this scenario, tend to have isotope values in the centre of the range of autotroph sources, which was the case in the present study, regardless of the habitat from which they were caught. Further work is warranted on measuring isotope values in prey for the fish species studied here.

Whereas significant differences in fish isotope values were found between artificial and natural habitats, no such differences were detected between canals and lakes. Both types of systems lack macrophytes, and autochthonous primary productivity is limited to benthic and water column microalgal sources. Present results indicate that the two types of artificial systems have similar trophic pathways for the three fish species analysed. The shift from canal to lake construction over the last decade in southern Queensland will probably make little difference to trophic processes leading to production of these three species.

The creation of extensive artificial estuarine waterways in southern Queensland has provided altered and new aquatic habitats for fish. Some species may utilise trophic pathways in the artificial habitats similar to those they use in adjacent natural habitats (e.g. *Acanthopagrus australis*). Other species, e.g. *Arrhamphus sclerolepis*, rely on entirely different autotrophic sources in the artificial waterways. The determination of trophic processes for fish species offers direct insights into the ecological functioning of the artificial waterways that goes beyond comparisons of abundances of animals in different habitats.

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