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Community structure, density and standing crop of fishes in a subtropical Australian mangrove area

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Abstract. The fishes occurring in a subtropical mangrove (*Avicennia marina*) area in Moreton Bay, Australia, were studied for one year (November 1987 to November 1988, inclusive). Fishes within the mangroves were sampled using a block net, whilst those in adjacent waters were sampled using seine and gill nets. Forty six percent of the species, 75% of the number of fishes and 94% of the biomass taken during the study (all methods combined) were of direct importance to regional fisheries. The fish community utilising the habitat within the mangrove forest differed from that occurring in adjacent waters in terms of density, standing crop, species composition and diversity-index values. Standing-crop estimates for the fishes occurring within the mangroves (study period mean $\pm SD = 25.3 \pm 20.4$ g m^{-2}) were amongst the highest recorded values for estuarine areas whilst those for adjacent waters (2.9 ± 2.3 g m^{-2}) were comparable to those of other estuarine studies.

Introduction

The disturbance of estuaries in Eastern Australia is occurring at a rapid rate as a result of an upsurge in the demand for waterfront developments (e.g. residential canal estates and marinas; Morton 1988). Many existing and proposed developments involve removal of intertidal habitats, especially mangroves. Mangroves are a prominent intertidal feature of sheltered subtropical and tropical estuaries and are particularly extensive along the east coast of Australia (Hutchings and Saenger 1987). The recent upsurge in development pressure on mangroves has lead to conflict with fishing interests. Mangrove-lined estuaries are generally accepted as providing a habitat for aquatic animals, many of which are economically important (Newell and Barber 1975, Pollard 1976, Staples 1980). However, our knowledge of the utilisation of mangrove areas by economically important fishes is limited and it is probable that the importance of such habitats has been underestimated in previous estuarine studies because of the sampling techniques used.

Most previous studies of mangrove ichthyofauna have relied upon sampling techniques that are unlikely to take fishes (especially large mobile individuals) of direct fisheries value. Sample methods have involved trawling (otter or beam) or hauling seines in water adjacent to mangroves. The limitations and poor catch-efficiency of otter trawls are well known (Taylor 1953, Kjelson and Colby 1977). Similar problems apply to beam trawls, which principally catch small fish (Ruello 1975, Young 1975). Fast-swimming and pelagic fishes and fishes which live in very shallow water are often under-represented in trawl samples (Yanez-Arancibia et al. 1980). Seines are likely to provide a more accurate indication of the true abundance and composition of the fish community, providing nets are long enough and deployed in such a manner as to avoid bias due to avoidance by large fish. However, few studies (Stephenson and Dredge 1976, Blaber and Blaber 1980, Bell et al. 1984, Blaber et al. 1985) have used techniques which overcome the effects of large-fish avoidance (e.g. large block, drop or seine nets), most studies providing data only on juvenile or benthic fishes.

Mangroves are considered to provide shelter and food (either directly or indirectly) for fishes (Odum and Heald 1975, Wallace and van der Elst 1975). Yet, the common sampling methods of trawling or seining techniques can only be used adjacent to mangroves and not within the actual mangrove forests. Seagrass beds, sand banks and mud flats occur near mangroves, and thus samples taken in waters adjacent to mangroves may contain fishes which prefer one or a combination of these habitats. In a recent North American study, Thayer et al. (1987) provided the only description of the ichthyofauna within mangrove (*Rhizophora mangle*) forests. Quantitative samples were taken using a block net, although sample areas were small (21 to 68 m^2), and it is unlikely that

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block nets set in small areas can efficiently sample fast-swimming or large fishes.

Present knowledge on the functioning of mangrove ecosystems is mostly based upon research conducted in North America and the Caribbean region (e.g. Austin and Austin 1971, Odum and Heald 1975, Weinstein et al. 1977, Thayer et al. 1987). Recent studies in Eastern Australia have indicated some differences in trophic relations that may affect the fish community compositions (Robertson 1986, Robertson and Duke 1987). Unfortunately, few studies of finfish in Australia have sampled efficiently for large mobile fishes or provided estimates of relative species abundance, to enable calculation of community diversity and identification of complete food chains within mangrove systems. No study has described utilisation by fishes, or provided quantitative estimates of fish density and standing crop, for the habitat within mangrove forests.

The following study was undertaken to assess the utilisation by fishes of a typical mangrove area in the subtropical region of Eastern Australia. Given the current debate on the importance of mangroves to regional finfisheries, emphasis was placed on sampling techniques that would capture economically important species. The study aims to quantitatively describe the ichthyofauna within a mangrove forest and to contrast it with fish populations occurring in adjacent habitats.

Materials and methods

Study area

This study was conducted in Moreton Bay, a large estuarine embayment located in the subtropical region of Eastern Australia (Fig. 1 a, b). Moreton Bay supports major commercial and recreational fisheries and has extensive areas of mangroves, seagrasses and saltmarshes (Hyland and Butler 1988). The study area was located at the confluent mouths of two creeks (Lota Creek and Tingalpa Creek, Fig. 1 c) on the western shores of Moreton Bay. Large forests (185 ha) of mangroves (*Avicennia marina, Rhizophora stylosa* and *Ceriops* spp.) occur along both creeks and on foreshores (Fig. 1 c). Saltmarsh vegetation (54 ha, mainly *Salicornia quinqueflora* and *Sporobolus virginicus*) occurs landward of mangroves in many areas along the creek. Large areas of seagrass (*Zostera capricorni* and *Halophila ovalis*) occur on the foreshore in the lower littoral and sublittoral zones of this area. Commercial fishing operations using block nets are prohibited in the study area although gill (mesh)-netting operations occur infrequently.

The selected study site at the mouths of the Lota and Tingalpa Creeks was chosen, as the land form provided a unique area to examine mangrove ichthyofauna. A 100 m long, 3 m wide, 1 m high ridge separated mangroves from saltmarsh/mudflat areas and provided a landward border for the block net (see subsection "Sampling methodology" below). The ridge was completely surrounded by water on high tides, yet provided a dry area suitable for block net and seine deployment. The mangrove forest studied was mainly *Avicennia marina* (<1 to 7 m high) although a few small (<5 m high) *Rhizophora stylosa* and *Ceriops tagal* were present. Pneumatophores (0.5 to 14.0 cm high) completely covered the substrate in the middle of the forest (density approx 430 m^{-2}), but gradually thinned towards the forest margins. Areas immediately adjacent to the mangroves were composed of bare sand/mud, although seagrasses (*Zostera capricorni*) were common in the lower littoral zone.

Tides in the area are unequal semi-diurnal, with the spring tidal range being 2.3 and 1.1 m on neap tides (Anonymous 1988). Mangrove areas are exposed on all low tides. During spring tidal periods, high water inundates mangrove areas whereas during most neap periods high tidal water is rarely present in mangroves or, if present, is shallow (<20 cm). In summer (December–February), spring high-tides occurring during the day are large and inundate mangrove areas, whereas night high tides are small, rarely reaching mangrove areas. In winter (June–August), the opposite diel tidal regime occurs. Both day and night spring high-tides inundate man-

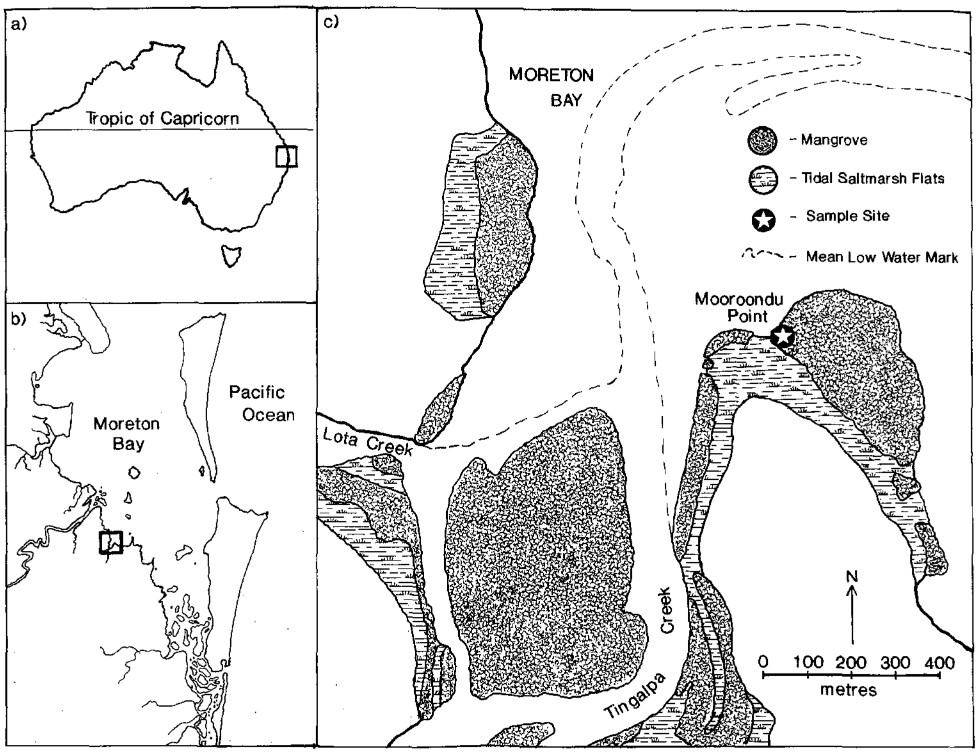

Fig. 1. (a) Study region on east coast of Australia; (b) location of study site within Moreton Bay; (c) study area at mouths of Lota and Tingalpa Creeks, showing sample site and mangrove distribution

grove areas in autumn (March–May) and spring (September–November), although tidal ranges are less than in summer or winter. This tidal regime dictated the sampling strategy described below.

Site preparation

A single site was prepared where all within-mangrove forest sampling (see "Sampling methodology" below) was undertaken. Stakes (50 mm diam, 3.5 m long) were driven into the soft substrate 15 to 20 m apart so as to surround a mangrove area of 3 340 $m^2 \pm 50$ m^2, the area enclosed being determined by aerial survey techniques. The height of each stake was adjusted so that >0.5 m of the stake was above water levels on all high tides sampled. A 1.0 m wide path, adjacent to the stakes, was cleared through the mangroves. This involved cutting all pneumatophores to substrate level, removing some small seedlings, and trimming the occasional overhanging large tree-limb.

Sampling methodology

Sampling was scheduled when the predicted high-tide would result in >300 mm of seawater over mangrove pneumatophores at the landward edge of the sample area. Samples were taken on each full moon from November 1987 to November 1988, inclusive. Two replicate samples were taken 25 h apart. Day samples were taken from November to April and August to October, inclusive. Night samples were taken from April to September, with two sampling periods occurring in May (M1 and M2). When day and night samples were scheduled within the same lunar period (April, August, September) at least 96 h separated sampling periods. The second sample of the April sampling period could not be taken due to extreme floods in the area which prevented effective sampling. Prior to fish-sampling, subsurface (150 mm) temperature (± 0.1 C°), salinity (± 1‰) and turbidity (± 0.1 nephlometric turbidity units; NTU) were measured using a mercury-in-glass thermometer, optical refractometer, and HACH Portalab Turbimeter, respectively. Tidal height (± 5 mm) was measured from a benchmark established at the upper littoral region of the blocked within-mangrove sample area.

Fish sampling was undertaken in three areas within the mangrove forest, in deep (2 to 3 m) water adjacent to the mangroves, and in shallow (1.5 m) shoreline water between mangrove forests. Fish were sampled within the mangrove forest using a multi-filament block net, 240 m long, 2 m deep, tapering to 4 m deep in the centre, with 18 mm stretch-mesh and a leadcore leadline. At slack high-water, the net was anchored to the shoreline ridge, and then deployed from a 3 m punt fitted with an electric motor. The floatline was attached to each stake and pulled taut so that it was above the water, the process continuing until the mangrove area was enclosed by the block net and the landward ridge. The block net was left in place until the tide had completely ebbed from the enclosed area (approximately 4 h), at which time all fish were collected. A haul seine, 70 m long, 4.5 m deep, with 18 mm stretch-mesh was used to sample fish occurring in shoreline water adjacent to the mangrove forest at high water. The seine was deployed from the punt and a single haul was taken for each sample. Two gill nets (30 m long, 3 m deep, 100 mm mesh; 30 m long, 4 m deep, 150 mm mesh) were set at high water approximately 50 m offshore from the mangroves over a mud/seagrass (*Zostera capricorni*) area and retrieved after 1.5 h.

Collected fish were identified to species, counted, and the total wet weight (± 2 g) of each species was recorded. The fork (FL) or total length (TL) of each individual was recorded, as appropriate. Mark and recapture studies were carried out to estimate the efficiency of the block-net method. Fish were collected by seine, fin-clipped, and released into the enclosed block area 1 h after high water. Species were selected for fin-clipping on the basis of their vulnerability to seining techniques and similarity to species commonly collected with the block-net.

The Shannon-Wiener diversity index (Cox 1980), H', was calculated for each sample [$H' = -\sum_{i=n}^{n} P(i) \ln P(i)$, where $P(i)$ is the ratio of a single species to the total number of individuals collected]. The Wilcoxon test for paired observations (Sokal and Rohlf 1981) was used to compare replicate samples, and samples taken within and adjacent to the mangroves in terms of fish density, standing crop, number of species and diversity. The degree of similarity between replicate samples and samples taken within and adjacent to the mangroves was calculated using the Jaccard (1908) similarity index described by Clifford and Stephenson (1975); [$J = x/(s+x)$, where x is the number of co-occurrences of an attribute, s is the sum of non-co-occurring attributes in both samples, and J is the co-efficient of similarity; note that co-absences are ignored].

Fish were placed in the following groups (see also Table 2) on the basis of earlier studies by Thomson (1954, 1959), Blaber and Blaber (1980), Bell et al. (1984), Morton et al. (1987), and own personal field observations; detritivores (feed almost exclusively on detritus and other plant remains); planktivores/microbenthic carnivores (feed on plankton, small epibenthic animals and occasionally small amounts of detritus); herbivores (feed almost exclusively on plant material); intermediary carnivores (feed primarily on macrobenthos, but often consume small amounts of plant material or small fishes); and top-level predators (fishes that are exclusively carnivorous and feed mainly upon fishes, although macrobenthic animals are taken occasionally).

Results

Abiotic variables

High-water tide-levels measured at the benchmark position ranged from 145 to 590 mm (mean $\pm$ SD $= 370 \pm 120$ mm) above the benchmark, and were generally higher on the second replicate sample than on the first, the largest difference being 165 mm. Water at the deepest seaward edge of the block net (fish-collection area of the net) was 660 mm deeper than the benchmark. Water salinities were relatively stable during the study, ranging from 24 to 36‰ (33 $\pm$ 3‰). Little difference was observed between replicate samples (mean difference $=1.2$‰), the greatest difference being 7‰ (February) following heavy rainfall. Water temperature ranged from 16.0 °C in August to 26.5 °C in January (21.9 C° $\pm$ 3.1 C°). Turbidity values ranged widely during the survey from a minimum of 2.3 NTU to a maximum of 25.0 NTU (10.8 $\pm$ 6.2 NTU), apparently varying primarily as a result of changing wind and wave action. Substantial fluctuations in turbidity also occurred between replicate samples, the largest difference being 20.0 NTU (mean difference $=9.5$ NTU). Turbidity values exceeded 100 NTU and salinities were <2‰ during the April floods, which forced cancellation of a scheduled second sampling.

Fishes within the mangroves

Eight species were released into the blocked area on at least three occasions (Table 1). Estimates of the catch efficiency (% recapture) of the block net ranged from 66 to 100%. Demersal fishes (e.g. *Sillago analis*) were more likely to escape from the enclosed area than

Table 1. Mean capture efficiency of block net for selected species in mangrove area, Moreton Bay, Australia

Species	No. of trials	Total no. of fish released	Mean recapture (%)
Mugil georgii	7	159	91
Gerres ovatus	5	14	100
Pranesus ogilbyi	7	33	73
Arrhamphus sclerolepis	7	25	92
Hyporhamphus ardelio	7	26	96
Acanthopagrus australis	9	45	87
Sillago analis	11	83	66
Tylosaurus macleayanus	3	6	100

mid-water or surface species (e.g. *Mugil georgii, Tylosaurus macleayanus*). The mean catch efficiency calculated over all species in all trials was 88.1%. All data presented are unadjusted unless otherwise stated, when an abundance correction factor of 1.14 was applied.

A total of 21 683 individuals weighing 1 950 469 g of at least 42 species were taken in the block net. Numerically, six species comprised 79.6% of the fishes taken: *Mugil georgii* (29.8%), *Ambassis marianus* (6.5%), *Girella tricuspidata* (10.9%), *Acanthopagrus australis* (13.2%), *Sillago analis* (12.4%), *Torquigener hamiltoni* (6.8%). None of the remaining species constituted individually $>3.5\%$ of the total fishes collected. It is probable that the first replicate sample adversely affected the second replicate sample taken 25 h later. The Wilcoxon test for paired observations indicated that the second replicate sample was significantly lower than the first sample in terms of fish density ($p<0.01$), standing crop ($p<0.01$), and Shannon-Wiener diversity values ($p<0.01$), irrespective of season. However, no significant difference ($p>0.05$) in the number of species was recorded between the first and second replicate sample. The species composition of the first and second samples was similar, with Jaccard similarity-index values ranging from 0.60 to 0.83 (mean = 0.71). All data presented for fishes within mangroves refer only to the first of the replicate samples.

A total of 15432 individuals weighing 1439016 g of 40 species were taken in the first of the replicate samples. Seven species numerically comprised 86.48% of the fishes taken: *Mugil georgii, Ambassis marianus, Girella tricuspidata, Acanthopagrus australis, Plotosus anguillaris, Sillago analis* and *Torquigener hamiltoni* (Table 2). None of the remaining species constituted individually $>2.7\%$ of the total fishes collected. In terms of economic value, 67.5% of the species, 79.6% of the number and 93.1% of the biomass taken are of direct fisheries importance. *Girella tricuspidata* accounted for 67.5% of the total biomass taken (Table 2), most fishes individually weighing ~ 750 g. Other species which contributed substantially to the total biomass included *Liza dussumeri, Mugil georgii, Selenotoca multifasciata, Acanthopagrus australis* and *Sillago analis*. All other species accounted for $<1.6\%$ of the total weight of fishes taken.

Mugil georgii, Acanthopagrus australis, Sillago analis, and *Torquigener hamiltoni* were recorded in all samples taken. *M. cephalus, Ambassis marianus, Gerres ovatus, Pranesus ogilbyi, Arrhamphus sclerolepis, Girella tricuspidata, Hyporhamphus ardelio, Selenotoca multifasciata, Platycephalus fuscus* and *Sphyraenilla obtusata* were also common (occurring in at least 64.7% of samples taken). *Liza dussumeri, Harengula abbreviata, Pelates quadrilineatus, Plotosus anguillaris, Pseudorhombus arsius, Rhabdosargus sarba, Sillago ciliata, S. maculata, Torquigener pleurostictus* and *Tylosaurus macleayanus* occurred regularly in samples (26.5 to 58.8%). All other species can be termed occasional visitors (occurring in 2.9 to 23.5% of samples), with the exception of *Myxus elongatus, Scatophagus argus, Euristhmus lepturus, Pomadasys opercularis, Rhinobatus batillum, Terapon jarbua, Agriopophyraena barracuda* and *Pomatomus saltatrix,* which were each taken on only one occasion.

Most species (57.5%), particularly those of economic value, were present as both juveniles and adults (Table 2). Species represented exclusively by juveniles accounted for 30% of those taken. *Girella tricuspidata* were represented exclusively by adult fish. *Acanthopagrus australis* were represented by both juveniles and adults, although juvenile fishes comprised the majority (82.5%) of the total number taken. Most individuals of *Mugil* spp. and *Sillago* spp. taken were juveniles.

Numerically, detritivores (35.9%) and intermediary carnivores (38.5%) dominated, although herbivores (15.9%) also formed a major proportion of the fish community (Table 2). Herbivores dominated the fish community in terms of biomass (72.4%), due to the predominance of *Girella tricuspidata,* although intermediary carnivores were also important (15.9%). Two top-level predators (*Platycephalus fuscus* and *Sphyraenella obtusata*) commonly occurred within the mangroves, although relatively few (0.22%) numbers of most other species were taken.

Fish numerical density ranged from a minimum of $0.07\ m^{-2}$ during the day in December to a maximum of $0.64\ m^{-2}$ during the day in August (Fig. 2). The overall mean ($\pm$SD) density for all samples combined was $0.27 \pm 0.14\ m^{-2}$ ($0.31\ m^{-2}$ when data are adjusted for fish escapement). Standing-crop values ranged from a minimum of $1.90\ g\ m^{-2}$ during the day in December to a maximum of $61.55\ g\ m^{-2}$ during the night in April. The mean standing crop ($\pm$SD) for all samples from within the mangroves combined was $25.34 \pm 20.36\ g\ m^{-2}$. Shannon-Wiener diversity values (Fig. 2) ranged from 0.76 to 2.61 (mean $\pm$ SD $= 1.79 \pm 0.44$).

Fishes occurring adjacent to mangroves

Comparison of replicate samples for fish taken in the seine (Wilcoxon paired-observations test) indicated that there was no significant difference ($p<0.01$) between samples in terms of fish density, standing crop, number of species, and diversity. Replicate samples were combined

Table 2. Percentage abundance and biomass of fishes taken within (block net) and adjacent to (seine and gill net) mangroves in Moreton Bay. J: juvenile; A: adult; C: commercial; R: recreational; +: present, but constituting <0.01% of total number or weight; –: no direct value

Trophic status and species	% of total number			% of total weight (g)			Life-history stage	Fisheries importance
	block	seine	gill	block	seine	gill		
Detritivores								
Liza dussumeri	1.37	0.32	0.0	2.39	2.85	0.0	J/A	C/R
Mugil cephalus	0.99	1.02	33.0	2.94	5.79	30.6	J/A	C/R
Mugil georgii	33.24	17.50	0.0	4.39	14.49	0.0	J/A	C/R
Myxus elongatus	0.25	0.00	0.0	0.14	0.00	0.0	J	C/R
Planktivores/microcarnivores								
Ambassis marianus	5.39	17.35	0.0	0.23	3.83	0.0	J/A	–
Engraulis australis	0.00	0.41	0.0	0.00	0.02	0.0	A	–
Gerres ovatus	0.65	0.38	0.0	0.14	0.71	0.0	J/A	–
Harengula abbreviata	0.18	0.03	0.0	0.04	0.27	0.0	J/A	–
Liza argentea	0.08	0.17	0.0	0.12	1.53	0.0	J/A	C
Pranesus ogylbyi	2.42	24.26	0.0	0.25	7.71	0.0	J/A	–
Thryssa hamiltoni	0.05	0.26	0.0	0.01	0.19	0.0	A	–
Herbivores								
Arrhamphus sclerolepis	0.71	6.91	0.0	0.50	18.81	0.0	J/A	C/R
Girella tricuspidata	11.61	0.03	26.8	67.49	0.70	23.1	J/A	C/R
Hyporhamphus ardelio	0.95	6.55	0.0	0.19	6.70	0.0	J/A	C/R
Scatophagus argus	+	0.00	0.0	+	0.00	0.0	J/A	C
Selenotoca multifasciata	2.66	0.26	26.8	4.21	2.08	11.4	J/A	C
Intermediary carnivores								
Acanthopagrus australis	12.40	2.73	0.9	9.55	7.46	1.0	J/A	C/R
Acentrogobius sp.	0.01	0.00	0.0	+	0.00	0.0	A	–
Caranx ignoblis	0.01	0.03	0.9	+	0.06	1.1	J	R
Dasyatis fluviorum	0.01	0.03	2.6	0.14	7.57	0.0	A	–
Dicotylichthys myersi	0.00	0.00	+	+	0.00	6.6	A	–
Euristhmus lepturus	0.05	0.00	0.0	0.07	0.00	0.0	J/A	–
Himantura uarnak	0.00	0.00	1.7	0.00	0.00	2.4	A	–
Lutjanus russelli	0.03	0.00	0.0	0.03	0.00	0.0	J	C/R
Monodactylus argentus	0.01	0.00	0.0	+	0.00	0.0	J/A	–
Pelates quadrilinatus	0.95	0.00	0.0	0.05	0.00	0.0	A	R
Plotosus anguillaris	4.15	0.00	0.0	1.58	0.00	0.0	J/A	–
Pomadasys opercularis	+	0.00	0.0	+	0.00	0.0	J	C/R
Pseudorhombus arsius	0.14	0.75	0.0	0.05	0.63	0.0	J/A	C/R
Rhabdosargus sarba	0.79	0.44	0.0	0.12	0.31	0.0	J	C/R
Rhinobatus batillum	+	0.06	2.6	0.02	1.01	3.1	J	C
Sillago analis	12.60	10.21	0.0	2.85	8.46	0.0	J/A	C/R
Sillago ciliata	0.08	2.32	0.0	0.03	3.31	0.0	J	C/R
Sillago maculata	0.03	0.06	0.0	0.01	0.01	0.0	J	C/R
Scomberoides commersonianus	0.07	6.15	0.0	0.01	2.16	0.0	J	R
Terapon jarbua	+	0.00	0.0	+	0.00	0.0	J	–
Torquigener hamiltoni	7.09	0.55	0.0	1.39	0.43	0.0	J/A	–
Torquigener pleurostictus	0.11	0.00	0.0	0.03	0.00	0.0	J/A	–
Top-level predators								
Agrioposphyraena barracuda	0.01	0.00	0.0	+	0.00	0.0	J	R
Carcharhinus obscurus	0.00	0.00	1.7	0.00	0.00	13.8	A	–
Megalops cyprinoides	0.00	0.00	0.9	0.00	0.00	0.7	A	R
Platycephalus fuscus	0.37	0.06	0.9	0.23	0.14	2.9	J/A	C/R
Pomatomus saltatrix	0.01	0.03	0.0	+	0.10	0.0	J	C/R
Sphyraenella optusata	0.28	0.15	0.0	0.20	0.54	0.2	J/A	R
Strongylura leiura	0.00	0.03	0.9	0.00	0.51	0.5	J/A	–
Tylosaurus macleayanus	0.20	0.61	0.0	0.62	1.68	0.0	J/A	–
Total	15 432	3 438	112	143 906	67 028	70 380		

for estimates of fish density, standing crop and data presentation.

A total of 3 438 individuals from 30 species weighing 67 082 g was taken using the seine net. Seven species numerically comprised 88.9% of the fishes taken: *Mugil georgii, Ambassis marianus, Pranesus ogilby, Arrhamphus sclerolepis, Hyporhamphus ardelio, Sillago analis* and *Scomberoides commersonianus*. None of the remaining species constituted individually >2.8% of the total fishes collected. In terms of economic value, 66% of the spe-

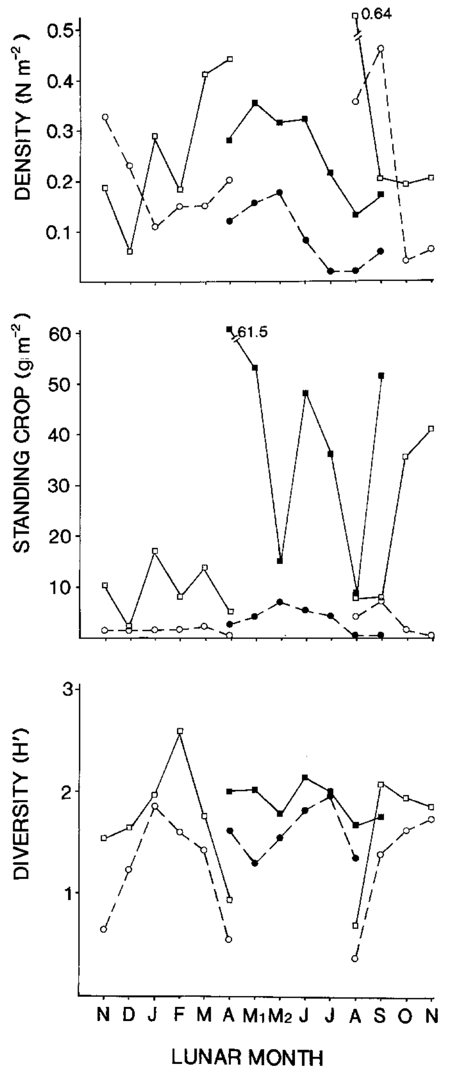

Fig. 2. Fish density (N), standing crop and diversity (*H'*) taken within mangroves and adjacent waters, November 1987–November 1988. Continous line indicates samples taken within mangroves (block net); dashed line, samples taken in adjacent waters (seine net). Open symbols, day samples; filled symbols, night samples. Two samples taken in May

cies, 57% of the number and 77% of the biomass taken were of direct fisheries importance. *M. georgii, Pranesus ogilbyi, A. sclerolepis, H. ardelio, Acanthopagrus australis* and *Sillago analis* were the dominant species in terms of weight (63.6% of total).

Pranesus ogilbyi, Arrhamphus sclerolepis, Hyporhamphus ardelio and *Sillago analis* were common in seine catches (occurring in at least 61.7% of samples). *Mugil georgii, Acanthopagrus australis, Torquigener hamiltoni* and *Tylosaurus macleayanus* regularly occurred in samples (26.5 to 58.8%). All other species can be termed occasional visitors (occurring in 2.9 to 23.5% of samples), with the exception of *Engraulis australis, Harengula abbreviata, Girella tricuspidata, Caranx ignoblis, Dasyatis fluviorum, Rhinobatus batillum, Pomatomus saltatrix* and *Strongylura leiura,* which were all taken on only one occasion.

Seine net samples contained a greater proportion of planktivores/microcarnivores (42.86% numerically) and fewer intermediary carnivores (23.33%) than samples taken within the mangroves. The proportion of top-level predators (0.88%) was similar to samples taken with the block net. In terms of biomass, intermediary carnivores were the dominant group, although detritivores and herbivores were also of substantial importance. Most species taken in seine samples were present as juveniles and adults.

Fish density in seine samples ranged from 0.02 m^{-2} in July to 0.46 m^{-2} in November 1987 (Fig. 2), with a mean value of 0.15 ($\pm$SD=0.13) m^{-2}. Standing-crop values ranged from a minimum of 0.21 g m^{-2} during the night in August to a maximum of 7.07 g m^{-2} during day in September. The mean ($\pm$SD) standing crop for all samples combined was 2.85 ($\pm$2.26) g m^{-2}. Shannon-Wiener diversity values ranged from 0.37 to 2.14 (1.39$\pm$0.43). No diversity index value was calculated for the September night-sample period as only one species was taken in the first replicate sample.

Stingray (*Himantura uarnak*), shark (*Carcharhinus obscurus*) and giant herring (*Megalops cyprinoides*) were taken exclusively by the gill nets. Other than these species, the gill nets caught species of larger fishes similar to those taken by the block net. It is probable that some fish captured in the gill nets may have been present in the mangroves at high water and were taken as they moved offshore with the ebbing tide. The gill nets fished inefficiently on several occasions due to large quantities of jellyfish (*Catostylis mosaicus*) which clogged the net meshes and caused the nets to be highly visible.

Comparisons between ichthyofauna within and adjacent to mangroves

The Jaccard similarity measure indicated that the fish communities within and adjacent to the mangrove forest (seine catches only) differed markedly in their species composition. Jaccard values for comparisons between the block net (first replicate sample) and seine (first replicate sample) were low, ranging from 0.06 to 0.53 (mean$\pm$SD=0.35$\pm$2.1). Comparisons between the block net (first replicate sample) and second replicate seine sample indicated Jaccard values ranging from 0.20 to 0.75 (0.36 $\pm$3.2).

The numerically dominant species occurring adjacent to the mangroves were planktivores (*Ambassis marianus* and *Pranesus ogilbyi*), in contrast to samples taken within mangroves where detritivores (*Mugil georgi*) or intermediary carnivores (*Acanthopagrus australis, Sillago analis, Torquigener* spp.) were more important. Whilst herbivores formed numerically similar proportions of the ichthyofauna both within and adjacent to the mangroves, the species comprising this trophic level differed. *Arrhampus sclerolepis* and *Hyporhamphus ardelio* occurred with similar frequency in both areas, but formed a small proportion of the within-mangrove ichthyofauna. *Girella tricuspidata* were abundant within the mangroves, but were rarely taken in adjacent waters. Top-level predators formed a numerically similar proportion of the ichthyofauna within and adjacent to the mangroves. However, *Platycephalus fuscus, Sphyraenella obtusata* and *Tylosaurus macleayanus* were common or occurred regularly within mangroves but were rare in adjacent waters. Large top-level predators were recorded exclusively in adjacent waters (*Carcharhinus obscurus, Megalops cyprinoides, Strongylura leiura*), albeit in low numbers.

Fish density, standing crop and diversity within the mangrove forest was significantly higher than in adjacent waters (Fig. 2) for almost all samples, and particularly for winter night-samples [Wilcoxon paired-observations test; fish density ($p<0.05$), standing crop ($p<0.01$), number of species ($p<0.01$) and diversity ($p<0.01$)].

Discussion

The sampling technique used within the mangroves was efficient at taking both pelagic and demersal fishes. Samples contained only fishes that were present in the mangroves at high tide. Fishes leaving landward saltmarsh areas on ebbing tides were prevented from entering the enclosed mangrove area by the block net and the ridge at the mangrove/saltmarsh interface. Fish escapement was lessened due to the large area enclosed (the top of the net being above the water), the close substrate contact of the leadline (as a result of pneumatophore removal), and the relatively large depth of the net combined with water depth creating a large "belly". It is probable that block-net sampling adversely influenced the ichthyofauna moving into the mangroves on the next high tide (25 h later). This may have been due to substrate disturbance during fish collection causing the release of repelling substances (e.g. hydrogen sulphide), or the effect of fish removal from the resident population.

The fish community in the present study area was dominated by Chandidae, Atherinidae, Mugilidae, Sillaginidae, Kyphosidae, Hemirhamphidae, Tetradontidae, and Sparidae. These families are commonly associated with undisturbed estuarine habitats in the study region (Stephenson and Dredge 1976, Quinn 1980, Morton et al. 1987). The major families numerically, in terms of weight, and frequency of occurrence are of direct economic importance (Sparidae, Mugilidae, Sillaginidae, Kyphosidae, Hemirhamphidae), and form the basis of regional finfisheries. Most economically important species were present as both juveniles and adults. The importance of the mangrove habitat to fisheries can be gauged by the statistics that 46% of the species, 75% of the number of fishes and 94% of the biomass taken during the study (all methods combined) were of direct economic importance. A conservative estimate of the monetary value, based on these catch data, shows that A $ 2800 of marketable fish (1988 values) were captured in the block-netted area ($3340\ m^{-2}$ = A $ 8380 ha^{-1}) during the study. It must be stressed that this value only indicates marketable fish value, it does not take into account the numerous commerically important juveniles captured or non-commercial values (e.g. recreation, education etc.). The above values indicate a substantially greater importance of mangroves to fisheries than indicated by other fish-community studies conducted in the study region (Stephenson and Dredge 1976, Quinn 1980). The importance of mangroves to fisheries may have been previously underestimated as a result of inappropriate sampling techniques and the absence of sampling within mangrove forests. Alternatively, it is possible that the present study site, although typical of mangrove areas in Moreton Bay in terms of vegetation and substrates, has a unique ichthyofauna.

In particular, these results contrast with those of Robertson and Duke (1987), who concluded that mangrove areas of Northern Australia do not appear to be utilised to a large extent by fishes of direct economic importance, the dominant families being Chandidae, Atherinidae, Leiognathidae, Engraulidae, Gobiidae, and Clupeidae. There are differences in latitude and dominance of mangrove/seagrass species between the study areas, however, it is possible that the sampling techniques used by Robertson and Duke contribute to the differences observed in the fish communities. The net deployment methods, mesh size and length (30 m) used by Robertson and Duke would be biased against the capture of fast-swimming fishes, many of which are of economic importance (e.g. Sparidae, Mugilidae, Sillaginidae, Polynemidae). Additionally, mangrove ecosystems in tropical Australia may function differently from those in subtropical areas, since Blaber (1980) also concluded that tropical mangrove-lined estuaries are dominated by planktivores.

The ichthyofauna utilizing the habitat within the mangrove forest differed in community composition from that in adjacent waters and was of greater density, standing crop and diversity. It is acknowledged that the block net was likely to be more efficient at taking fishes, given the larger area enclosed and hence the lesser likelihood of fish escapement during net deployment. Compared with other studies of waters adjacent to mangroves in the general study region (Stephenson and Dredge 1976, Quinn 1980), intermediary carnivores were more abundant numerically and herbivores more important in terms of biomass. *Girella tricuspidata,* 67% of total block-net biomass taken, have previously been recorded in Australian estuarine studies (Bell et al. 1984); however, none of these have recorded the presence of this species in such large numbers. This may result from a "lack of sampling" within mangroves or because *G. tricuspidata* is particularly abundant at the present study site. *G. tricuspidata* has previously been recorded as feeding primarily upon filamentous algae (Thomson 1959, Bell et al. 1984), which were observed during field studies to be abundant on mangrove pneumatophores within the mangrove stand. As noted above, the families comprising these trophic levels (Sillaginidae, Kyphosidae, Hemirhamphidae, Sparidae) are of major importance to regional fisheries. Diversity indices (annual mean = 1.79) were higher than in adjacent waters, and were in the upper range of reported values for other Australian estuarine studies (Stephenson and Dredge 1976, Beumer 1978, Quinn 1980, Bell et al. 1984). The habitat within the mangroves may provide greater protection and food supply compared to that in subtidal regional creeks.

The species composition and dominance of trophic levels of the fish community adjacent to mangroves was similar to that recorded in studies of subtidal mangrove-lined creeks in subtropical Eastern Australia (Stephenson and Dredge 1976, Quinn 1980, Bell et al. 1984). The only substantial difference between these studies and the present study is the low abundance of Gerridae, Clupeidae and Tetradontidae in the present study. Diversity indices (study period mean = 1.39) were comparable to those found in other studies conducted along the east coast of Australia. Morton et al. (1987) recorded an annual mean diversity value of 0.94 for saltmarsh areas, whilst values for studies conducted in mangrove-lined estuaries range from 1.33 (Bell et al. 1984) to 1.90 (Quinn 1980).

The increased proportion of intermediary carnivores in the fish community within mangroves compared to that of adjacent waters may reflect the presence of large macrobenthic populations, suitable as food, that occur within mangrove forests. Most of the present concepts of energy flow in mangrove-dominated systems are based on research undertaken in Florida (Odum and Heald 1972, 1975, Odum et al. 1972), and basically surmise that there is little retention of mangrove leaf-litter within the forest (most being subject to tidal flushing). Energy is considered to mainly flow along the link, mangrove leaf detritus → offshore decomposition → detritivores → lower carnivores → higher carnivores. However, Robertson (1986) concluded that hypotheses about mangrove food-chains in the Indo-West Pacific should be altered to account for the presence of substantial sesarmid-crab populations, which are responsible for the consumption of large quantities of mangrove-leaf litter within mangrove forests. It is probable that these crab populations represent a major food source for intermediary carnivores (most of which are of economic importance) within the mangrove ecosystem considered. Morton et al. (1987) noted that sesarmid crabs constitute a dominant dietary component of intermediary carnivorous fishes (e.g. *Acanthopagrus australis* or *Torquigener* spp.) in intertidal areas in Moreton Bay. It is probable that one of the major trophic chains in Moreton Bay mangrove communities is leaf litter → sesarmid crab → lower carnivore, and that this chain occurs within mangrove forests.

Mangroves are generally considered to be one of the most productive estuarine habitats. Standing-crop values for finfish samples recorded within the mangroves in the present study are amongst the highest values reported for estuarine habitats, whilst those in adjacent waters are comparable to those of other studies conducted in estuarine areas (Table 3). Only one study has reported standing-crop estimates higher than those recorded in the present study (44.1 g m^{-2}; Perry 1976, as reported in Ross et al. 1987). Those results were based on a relatively short sample-period, and were taken in relatively deep (> 3 m) water, hence sampling a greater volume of enclosed water. Although standing-crop values in the present study were high, fish densities were low in comparison to those reported for similar studies in other aquatic environments (Table 3). To some extent, this difference may reflect different sampling techniques, in that most other studies have used methods (small mesh-size and sample area) which would be unlikely to take large fish, but would retain substantial numbers of fish smaller than those taken in the present study.

Management implications

The data in this study result from a single sample-area with limited temporal replication. However, the results indicate that the direct importance of mangroves to fishes in Moreton Bay has previously been underestimated due

Table 3. Comparison of mean density and standing crop (over total study period) for studies of ichthyofauna in estuarine areas. –: not reported

Density (fish m^{-2})	Standing crop (g m^{-2})	Sampling gear	Habitat	Area sampled (m^2)	Location	Source
0.27	25.3	block net	within mangrove	3 340	Queensland, Australia	Present study
–	22.3	drop net	mud/sand bay	–	Texas, USA	Jones (1965)
9.00	15.0	drop net	seagrass	10	Texas, USA	Gilmore et al. (1978)
8.00	15.0	block net	within mangrove	21.7–58.2	Florida, USA	Thayer et al. (1987)
–	13.8	drop net	mud/sand bay	100	Texas, USA	Jones et al. (1963)
4.53	9.3	seine	saltmarsh creek	102–403	Florida, USA	Carr and Giesel (1975)
~0.94	~6.4	block net	mangrove	~1 000	New South Wales, Australia	Bell et al. (1984)
0.15	2.9	seine	adjacent to mangroves	1 385	Queensland, Australia	Present study
0.53	2.0	seine	seagrass	1 160	Texas, USA	Gilmore et al. (1978)
0.22	0.8	trawl	seagrass	260–540	Florida, USA	Thayer et al. (1987)
0.04	–	trawl	seagrass	20 000	New South Wales, Australia	Gibbs and Matthews (1982)

to an absence of sampling within mangrove forests and inappropriate sampling techniques. The mangrove habitat functions as both a nursery area and a feeding area for adult fishes, most of which are of direct economic importance. The richness of the fish community which moves into mangrove forests on flooding tides reflects the high productivity of such areas. At present, it is difficult to quantify losses to fisheries when wetland communities are disturbed for coastal development. The effects of even relatively large developments on regional fisheries are unclear. Future research should be directed towards quantitative studies that will permit estimates of fisheries losses and enable comparison of habitat value.

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