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ESTUARIES AS JUVENILE HABITATS FOR LUTJANID AND SERRANID FISHES IN TROPICAL NORTHEASTERN AUSTRALIA.

Thesis submitted by Marcus John SHEAVES BSc(Hons) (*JCU*) in January 1995

for the degree of Doctor of Philosophy in

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Lutjanus argentimaculatus sheltering amongst *Rhizophora stylosa* prop-roots.

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ABSTRACT

A number of lutjanid and serranid fishes are thought to utilise tropical estuaries as juvenile habitats. However, little detailed biological or life-history information exists for any species, and the species compositions of these families inhabiting tropical estuaries, are poorly known. During this study, the species compositions and life-histories of lutjanid and serranid fishes inhabiting estuaries along the north-eastern coast of tropical Australia were investigated. Using fish-traps, estuary faunas were compared to those inhabiting near-shore reefs. Additional samples were donated by anglers and collected from estuaries by angling. The reproductive statuses, and size and age structures of two serranids *(Epinephelus coioides* and *E. malabaricus)* and two lutjanids (*Lutjanus russelli* and *L. argentimaculatus*), in estuaries were compared to those of the same species from offshore. Fish-traps were also used to obtain data on the distribution and abundance by size of *L. russelli*, *E. coioides* and *E. malabaricus* from three estuaries - Cattle, Barramundi and Alligator Creeks - over a two year period.

Far fewer species of lutjanids and serranids were trapped from estuaries than from nearshore reefs. While fish-trap and angling collections from estuaries produced 9 species of serranids and 5 species of lutjanids, most were collected in low numbers. Only two serranids *(Epinephelus coioides* and *E. malabaricus)* and two lutjanids (*Lutjanus russelli* and *L. argentimaculatus*) were common in either trap or angling catches. All fish of each of these species from estuaries were found to be much smaller and younger than the largest and oldest fishes of these species from offshore waters. Furthermore, all fish of each species from estuaries were found to be in pre-reproductive condition. This implies that the estuarine populations of these species consist of juveniles, and that they undergo migrations to offshore adult habitats. Thus they possess three distinct life-history stages (pelagic larvae, estuarine juveniles, offshore adults) that correspond to major habitat shifts .

L. russelli were common in all estuaries and distributed throughout the three estuaries studied in detail. The probability of capturing *L. russelli* was similar in seaward areas of Cattle, Barramundi and Alligator Creeks, remained similar upstream areas of Barramundi Creek, but fell markedly in upstream parts of Cattle Creek. Spatial differences in the size of *L. russelli* (both within and between estuaries) were small, however, there was a strong pattern of seasonal change in the size of *L. russelli* in all estuaries. This seasonal pattern was apparently a product of the interaction between recruitment, mortality and migration. Studies in Alligator Creek showed that *L. russelli* demonstrated a strong preference for structurally complex habitats provided by fallen timbers and mangrove roots.

The numbers of both species of *Epinephelus* in trap catches declined in upstream areas of the three estuaries considered in detail. However, this reduction was much more marked for *E. coioides* than *E. malabaricus*, and while in downstream areas, the numbers of *E. coioides* were similar to or greater than those of *E. malabaricus*, in upstream areas *E. malabaricus* dominated. This suggests that the two species differ in their abilities to access or remain in upstream areas of estuaries. There was a strong negative correlation between the maximum deviation of salinity from 'normal' seawater levels and catches of both species of *Epinephelus*, suggesting that long-term salinity variation may be important in determining the distribution and abundance of *Epinephelus* spp. within estuaries.

The occupation of specific habitats during particular periods of development must be considered in the development of management strategies for these fishes. The use of estuaries as juvenile nursery grounds underlines the importance of maintaining the quality of estuarine habitats. More data from localities further afield (both within Australia and overseas) are needed to determine if the reproductive patterns found here apply to these species in other areas or to related species.

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DECLARATION

I declare that this thesis is my own work and has not been submitted in any form for another degree or diploma at any university or other institution of tertiary education. Information derived from the published or unpublished work of others has been acknowledged in the text and a list of references is given.

M. J. Sheaves

15 January 1995

CHAPTER 1.

GENERAL INTRODUCTION

It is generally considered that estuaries in tropical Indo-Pacific waters are dominated by small and juvenile fishes (Blaber, 1980; Blaber *et al.*, 1985; Blaber *et al.*, 1989; Robertson and Duke, 1987). While a number of studies have considered estuarine fish assemblages in the Indo-Pacific, the biology of individual species has received little attention. This is particularly true of habitat associated predators such as lutjanids and serranids. The structurally complex habitats used extensively by these species (Sheaves, 1992) inhibit sampling with seine and gill nets, the gears employed in most studies. In consequence members of these families have usually been sampled in low numbers only. Despite this, these fish comprised a major part of angler's catches from an estuary in tropical Australia (Sheaves, 1992), and fish trapping data (Sheaves, 1992; 1994) suggests that gears which target appropriate habitats may produce quite high catch rates of lutjanids and serranids.

1.1 IMPORTANCE OF TROPICAL ESTUARIES AS HABITATS FOR JUVENILE FISHES

In the northern-hemisphere, estuaries are known to function as nursery grounds for many species of fishes (e.g. Elliott *et al.*, 1990; McBride and Conover, 1991). However, there has been considerable debate about the status of Indo-Pacific estuaries as habitats for juvenile marine fishes. Generally, the discussion has centred on the questions of whether estuaries are "nursery grounds" or if a large proportion of fishes in estuaries are "estuarine dependent". The conclusions reached vary widely and to a large extent are dependent on the definition of the terms "nursery ground" and "estuarine dependent" used.

Bell *et al.* (1984) found clear evidence that mangrove creek habitats in temperate Australia were nursery grounds for "economically important species", with mangrove creeks being beneficial in providing shelter and feeding sites for juvenile fishes. Similarly South African estuaries are important nursery grounds for more than 80 species of fishes (Whitfield, 1983), and are dominated by the juveniles of marine species (Whitfield, 1990). For example, the 6 species of carangids that are important components of estuary communities in South Africa only utilise estuaries as juveniles

(Blaber and Cyrus, 1983). While a judgement of the extent of estuarine dependence in these species is difficult, the fact that carangids were the most important piscivorous fishes in a number of estuaries, emphasises their importance in the overall trophic structure of the estuarine communities (Blaber and Cyrus, 1983). Studies in tropical Australia have also highlighted the importance of estuaries as juvenile habitats. The fish faunas of Trinity Inlet (Blaber, 1980) and embayments in north-western Australia (Blaber *et al.*, 1985) were found to be dominated by juvenile marine fishes, and at least one-third of the species in the Embley estuary were estuarine dependent [defined as "those species for which estuaries form an essential habitat for at least one stage of the life cycle"] (Blaber *et al.*, 1989). Forty-five percent of the twenty most dominant species in Alligator Creek, in north-eastern Australia, were dependent on estuarine habitats (Robertson and Duke, 1990b).

On the other hand, many studies have suggested that estuaries were not important as juvenile habitats for marine species. In temperate south-western Australia, few species are entirely dependent on estuaries (Lenanton and Potter, 1987). Similarly, it has been found that in tropical Solomon Islands' estuaries only 8-9% of fish numbers were juveniles of reef species (Blaber and Milton, 1990), and that estuaries in equatorial Papua New Guinea were important nursery areas for only a few species of fish from coral reef habitats (Quinn and Kojis, 1985a). Robertson and Duke (1987) considered the value of estuaries in north-eastern tropical Australia as nursery grounds for commercially important species. On this basis they suggested that the estuaries they studied were not major nursery sites for fish species of direct commercial importance within Australia. However, they pointed out that these estuaries were nursery sites for species of commercial importance in other countries, and contain many species that are the prey of major commercial fishes such as the barramundi, *Lates calcarifer* (Robertson and Duke, 1987).

Judging the value of estuaries as habitats for juvenile fishes on the basis of strict estuarine-dependence requires estuaries to provide an essential habitat for juveniles (Blaber *et al.*, 1989). On this basis, if juveniles of a species occur in other habitats it is generally considered that estuaries are not vital for juveniles of the species (Lenanton and Potter, 1987). This is probably too simplistic a view and undervalues the importance of estuaries to juvenile fishes. Simply determining if juveniles of a species occur elsewhere ignores three important and possibly vital points. Firstly, there may be far greater numbers of the species in estuary habitats. If so there may be too little alternate habitat to support a viable group of juveniles, or the juveniles occurring in the other habitats may simply be stragglers that failed to find and settle in preferred estuarine habitats. Secondly, mortality rates may be far greater in the alternative habitats. If so the alternative habitats may contribute little to the adult population. Thirdly, growth rates of fishes may be greater in estuaries than in alternate inshore nursery areas (Lenanton and Potter, 1987). Where this is the case juveniles from alternative nursery grounds may be at a disadvantage in intra-specific and even inter-specific competition. Thus there may be clear advantages that accrue to those individuals accessing estuarine nursery areas, and the simple presence of a species in other habitats does not mean that these habitats are important or even useful juvenile habitats. The judgement of the nursery ground value should not simply come down to a question of distribution, but should include comparisons of abundance, growth rates and mortality rates in the different habitats.

It has been suggested that as estuaries are "transitory features in a geological sense" (Hedgpeth, 1982), from a long-term point of view they cannot be critically important habitats for fishes. This is a narrow view. As Blaber *et al.* (1989) state "individual estuaries may be short-lived, as are coral reefs, estuarine-dependent juvenile fish are no more dependent on individual estuaries than coral reef fishes are on a particular coral reef".

Notwithstanding the various points of view, most studies of tropical Australian estuaries have reported catches to be dominated by large numbers of juvenile fishes (Blaber, 1980; Blaber *et al.*, 1985; Blaber *et al.*, 1989; Robertson and Duke, 1987). Clearly, whether or not a particular species is totally dependent on estuaries, the dominance of juvenile fishes in estuaries in tropical Australia demonstrates their importance as nursery areas. On this basis alone there is little doubt that, as with estuaries in temperate Australia and South Africa (Lenanton, 1977; Blaber and Cyrus, 1983; Potter *et al.*, 1990; Whitfield, 1990), estuaries in tropical Australia must play a very important role in the life cycles of many fish species.

1.2 LUTJANIDS IN ESTUARIES

Most studies of Indo-Pacific estuaries, in and adjacent to tropical Australia, have reported the presence of lutjanid fishes. The main species recorded from estuarine localities are *Lutjanus argentimaculatus* and *L. russelli* (Table 1.1). However, a number of other species have been reported less frequently (Table 1.2).

Table 1.1: Studies of Indo-Pacific estuaries, in and adjacent to tropical Australia, which have reported *L. argentimaculatus* and/or *L. russelli*.

Study	Locality	Species
Blaber, 1980	Trinity Inlet, N.E. Australia	*L. argentimaculatus*
Blaber, 1986	Dampier region, N.W. Australia	*L. argentimaculatus* *L. russelli*
Blaber and Milton, 1990	Solomon Islands	*L. argentimaculatus* *L. russelli*
Blaber *et al.*, 1985	Dampier region, N.W. Australia	*L. argentimaculatus* *L. russelli*
Blaber *et al.*, 1989; 1990b	Embley estuary, N. Australia	*L. argentimaculatus* *L. russelli*
Coles *et al.*, 1993	Trinity Inlet, N.E. Australia	*L. russelli*
Collette, 1983	N. Australia	*L. argentimaculatus* *L. russelli*
Collette, 1983	Papua New Guinea/Irian Jaya	*L. argentimaculatus* *L. russelli*
Davis, 1988	Leanyer Swamp, N. Australia	*L. argentimaculatus* *L. russelli*
Haines, 1979	Papua New Guinea	*L. argentimaculatus*
Quinn and Kojis, 1985a,b	Papua New Guinea	*L. argentimaculatus* *L. russelli*
Robertson and Duke, 1987; 1990a	4 N.E. Australian estuaries	*L. argentimaculatus* *L. russelli*
Sheaves, 1992	Alligator Creek, N.E. Australia	*L. argentimaculatus* *L. russelli*
Sheaves, 1994	Alligator Creek, N.E. Australia	*L. argentimaculatus* *L. russelli*
Thollot *et al.*, 1990	New Caledonia	*L. argentimaculatus* *L. russelli*

Table 1.2: Studies of Indo-Pacific estuaries, in and adjacent to tropical Australia, where lutjanid species other than *L. argentimaculatus* and *L. russelli* have been reported.

Study	Locality	Species
Blaber, 1980	Trinity Inlet, N.E. Australia	*L. fulviflamma*
Blaber and Milton, 1990	Solomon Islands	*L. ehrenbergii* *L. fulviflamma* *L. fulvus* *L. maxweberi* *L. rivulatus*
Coles *et al.*, 1993	Trinity Inlet, N.E. Australia	*L. fulviflamma* *L. erythopterus*
Collette, 1983	Papua New Guinea/Irian Jaya	*L. ehrenbergii* *L. fulviflamma* *L. fulvus* *L. fuscescens*
Quinn and Kojis, 1985a,b	Papua New Guinea	*L. ehrenbergii* *L. johnii* *L. maxweberi*
Sheaves, 1994	Alligator Creek, N.E. Australia	*L. johnii*
Thollot *et al.*, 1990	New Caledonia	*L. fulvus*

1.2.1 *Lutjanus argentimaculatus*

Lutjanus argentimaculatus is among the largest of the snappers reaching a total length of about 1200mm (Allen, 1985). It is a commercially (Allen, 1985; Thollot *et al.*, 1990) and recreationally (Robertson and Duke, 1990a) important species, with a wide Indo-West Pacific distribution (Allen and Talbot, 1985). While *L. argentimaculatus* is abundant in estuaries it commonly penetrates considerable distances up coastal freshwater streams (Haines, 1979; Allen and Talbot, 1985), and has been found entering tidal swamps in northern Australia (Davis, 1988). In offshore waters *L. argentimaculatus* is found in reef habitats to a depth of at least 100m (Allen, 1985; Allen and Talbot, 1985).

Distribution:

L. argentimaculatus is widespread in the tropical Indo-Pacific, occuring from East Africa through South East Asia to the Central Pacific (Allen and Talbot, 1985). *L. argentimaculatus* has been recorded from most studies in tropical Australia (Table 1.1). As well, *L. argentimaculatus* has been recorded from estuaries in sub-tropical Australia (Ellway and Hegerl, 1972; Shine *et al.*, 1973), and in Papua New Guinea, the Solomon Islands and New Caledonia (Table 1.1).

In coastal waters in the Australian region, *L. argentimaculatus* has been recorded from Albatross Bay in northern Australia (Blaber *et al.*, 1990b; Blaber *et al.*, 1994). It has also been recorded from offshore waters in New Caledonia (Thollot *et al.*, 1990).

Habitats:

Although *L. argentimaculatus* has been reported in most studies of estuaries in tropical Australia, it has usually been recorded in low numbers only (e.g. Blaber, 1980; Blaber, 1986). A number of species of estuarine fishes occur in much greater abundances in the structurally complex habitats afforded by fallen timber and mangrove prop-roots than in other estuarine habitats (Sheaves, 1992). *L. argentimaculatus* seems to be a species that has a strong preference for structurally complex habitats. Although Sheaves (1992) recorded only low numbers of *L. argentimaculatus*, all came from such habitats. Furthermore, anglers specialising in the capture of *L. argentimaculatus* fish almost exclusively close to fallen timber and prop-roots (pers. obs.). Thus *L. argentimaculatus* is likely to be poorly represented in studies employing nets as sampling tools. Sheaves (1992) employed fish traps to sample structurally heterogeneous habitats but recorded low catch rates of *L. argentimaculatus*. As *L. argentimaculatus* was consistently a major component of anglers' catches from the same locality it seems that fish traps were not efficient tools to sample this species. In northern Australia Blaber *et al.* (1992) found small numbers of *L. argentimaculatus* in coastal seagrass beds, another structurally complex habitat.

Life-history status:

The life-history status of *L. argentimaculatus* residing in estuaries is unclear. Allen and Talbot (1985) state that juveniles and adults are common in estuaries and coastal freshwater streams and suggest that larger *L. argentimaculatus* migrate offshore. Both Johannes (1978) and Thollot *et al.* (1990) assert that *L. argentimaculatus* migrates offshore to spawn. However, neither Johannes (1978) nor Thollot *et al.* (1990) present any evidence to support this premise.

Most studies that have recorded the presence of *L. argentimaculatus* in estuaries have not reported life-history status. Those studies that have, often recorded the presence of both juvenile and adult fish (Blaber and Milton, 1990; Thollot *et al.*, 1990) or adult fish only (Shine *et al.*, 1973; Blaber, 1980). With one exception (Blaber, 1980) no criteria for the determination of life-history status were presented. Blaber (1980) caught a small number (1 or 2) *L. argentimaculatus* in Trinity Inlet in north-eastern Australia and classed it/them as adult on the grounds that "...all reproductively immature fish were classed as juveniles..." (Blaber, 1980, p.139). In other studies no adults were reported. Blaber (1986) found small numbers of *L. argentimaculatus* in mangrove creeks in north-western Australia and classed them as juveniles, while Robertson and Duke (1990a) sampled *L. argentimaculatus* from mangrove forest habitats in Alligator Creek in north-eastern Australia and classified them as sub-adults.

Feeding:

As with many other fishes of the genus *Lutjanus*, *L. argentimaculatus* feeds mainly on fish and crustaceans (Allen, 1985). Robertson and Duke (1990a) found *L. argentimaculatus* using intertidal mangrove forests in Alligator Creek in north-eastern Australia as feeding sites at high tide. The diet of *L. argentimaculatus* in Alligator Creek consisted mainly of fish [47.5% {percentage volume}], crabs [40.2%] and shrimps [10.8%] (Robertson and Duke, 1990b), while *L. argentimaculatus* from the Embley Estuary in northern Australia, consumed mainly crabs [55% {percentage frequency of occurrence}], other crustaceans [27%], teleost fish [45%] and molluscs [27%] (Salini *et al.*, 1990).

1.2.2 *Lutjanus russelli*

Lutjanus russelli is a medium sized snapper reaching a maximum total length of about 45cm (Allen, 1985). It is common on offshore coral reefs (Allen, 1985) and in estuaries (Allen, 1985). It also penetrates into freshwater streams (Allen, 1985). *L. russelli* has an extensive Indo-West Pacific distribution (Allen, 1985; Allen and Talbot, 1985) and is an important commercial species throughout its range (Allen, 1985).

Distribution:

As with *L. argentimaculatus*, *L. russelli* is widespread in the tropical Indo-Pacific, occuring from East Africa through South East Asia to the Central Pacific (Allen and Talbot, 1985). *L. russelli* has been reported from all parts of tropical Australia, Papua New Guinea, Irian Jaya, the Solomon Islands, and New Caledonia (Table 1.1). *L. russelli* is also found in offshore waters in Australia (Allen and Talbot, 1985; Blaber *et al.*, 1990b), and in the Solomon Islands (Thollot *et al.*, 1990).

Habitats and Movements:

L. russelli has been captured in the majority of studies of fishes of estuaries in tropical Australia, but as with *L. argentimaculatus* it is generally caught in small numbers only. Like *L. argentimaculatus*, *L. russelli* seems to favour habitats of high structural heterogeneity (Sheaves, 1990). Thus the low catch rates of *L. russelli* reported in most studies probably reflects the difficulties of recovering a representative sample from these habitats using standard netting techniques. However, unlike *L. argentimaculatus*, *L. russelli* is amenable to fish trapping. In a study in Alligator Creek, in tropical Australia, *L. russelli* was the largest component of the catch, contributing 31.4% of the total numbers of fishes trapped (Sheaves, 1994). *L. russelli* also occur in seagrass beds in estuaries in northern Australia (Blaber *et al.*, 1989; Coles *et al.*, 1993).

From mark-recapture data Sheaves (1993) [see Appendix II] found *L. russelli* within a tropical estuary showed a strong tendency to remain at one site for a considerable length of time. Most individuals were recaptured at the site of original capture, and showed no tendency to move further from the site of capture with increasing time at liberty, no preferred direction of movement and little tendency to cross to the opposite bank of the estuary.

Life-history status:

While it is generally considered that estuarine populations of *L. russelli* consist of juveniles (Allen and Talbot, 1985; Iwatsuki *et al.*, 1989), this has not been demonstrated. The studies that have reported the presence of *L. russelli* and reported life-history status have invariably classed them as juveniles (Blaber, 1986; Blaber and Milton, 1990; Davis, 1988; Thollot *et al.*, 1990). However, none of these studies reported the basis for their classification.

Feeding:

There seems to be a shift in the dietary preferences of *L. russelli* between estuarine and offshore locations. In Albatross Bay, in tropical northern Australia, teleosts were the major component of the diet of *Lutjanus russelli* [frequency of occurrence: 74.3%] (Brewer *et al.*, 1991). Other important dietary components were molluscs [frequency of occurrence: 15.4%], Brachyura [15.4%], Penaeidae [5.1%] and other crustaceans [17.9%]. A second study in Albatross Bay (Salini *et al.*, 1994) reported similar results. Salini *et al.* (1990) investigated the diet of *L. russelli* within the Embley Estuary, which opens to Albatross Bay. Within the estuary penaeid prawns were the most frequently occurring dietary item [52% of stomaches] followed by other Crustacea [48%] and teleosts [35%].

1.3 EPINEPHELINE SERRANIDS IN TROPICAL ESTUARIES

The serranid fishes reported from tropical estuaries in the Indo-Pacific region almost invariably belong to the sub-family Epinephelinae. A number of species, all from the genus *Epinephelus*, have been reported from estuaries in Australia, the Solomon Islands, Papua New Guinea and New Caledonia (Table 1.3).

Table 1.3: Studies of Indo-Pacific estuaries, in and adjacent to tropical Australia, which have reported fish of the genus *Epinephelus*.

Study	Locality	Species
Blaber and Milton, 1990	Solomon Islands	*E. ongus* *E. polystigma* *E. sexfasciatus* *E. spilotoceps* *E. suillus* *E. tauvina*
Blaber *et al.*, 1985	Dampier region, N.W. Australia	*E. tauvina*
Blaber *et al.*, 1989; 1990b	Embley estuary, N. Australia	*E. malabaricus* *E. merra* *E. suillus*
Collette, 1983	N. Australia	*E. corallicola* *E. tauvina*
Haines, 1979	Papua New Guinea	*E. tauvina*
Quinn and Kojis, 1985a,b	Papua New Guinea	*E. tauvina*
Sheaves, 1992	Alligator Creek, N.E. Australia	*E. coioides* *E. malabaricus*
Sheaves, 1994	Alligator Creek, N.E. Australia	*E. coioides* *E. malabaricus*
Thollot *et al.*, 1990	New Caledonia	*E. malabaricus*

1.3.1 Confusion as to identity of *Epinephelus coioides* and *E. malabaricus*.

For the present study the classification of epinepheline serranids presented in Randall and Heemstra (1991) is adopted.

In the past much confusion has existed in the classification of serranid fishes of the genus *Epinephelus* (Randall and Ben-Tuvia, 1983). This is particularly true of species inhabiting estuaries in tropical Australia.

The species reported most commonly from estuaries in tropical Australia are *E. malabaricus*, *E. tauvina*, *E. coioides* and *E. suillus*. Unfortunately the identification of these species has been very uncertain. Gloerfelt-Tarp and Kailola (1984) show a colour plate (p.132) that is labelled *E. malabaricus* but appears to be *E. coioides* [having orange-brown spots rather than black], while a plate that is labelled *E.*

tauvina (p.134) also depicts *E. coioides* (Randall and Heemstra, 1991). Masuda *et al.* (1984) provide two photos labelled *E. malabaricus* (plate 116, figures A,B) both of which show *E. coioides* (Randall and Heemstra, 1991), and two photos labelled *E. salmonoides* (plate 115, figures J,K) that are both *E. malabaricus* (Randall, 1987; Randall and Heemstra, 1991). Grant (1978) grouped *E. coioides* and *E. malabaricus* from northern Australia as a single species of "estuary rock-cod" under the name of *E. tauvina*. However, *E. tauvina* is primarily a clear-water, coral-reef species (Randall and Ben-Tuvia, 1983). In subsequent work Grant (1987) recognised three species of estuary cod; *E. tauvina* [the photograph (p.142) of which is clearly *E. coioides*], *E. chewa* [the photographs (p.143) of which are clearly *E. malabaricus*], and *E. malabaricus* [of which there is no photograph but which is reported by Grant (1987) to have orange-brown spots over the back - *E. malabaricus* re: Randall and Heemstra (1991) has black spots]. Randall *et al.* (1990) consider that *E. suillus* is a synonym of *E. coioides.*

It appears, therefore, that the reports of *E. malabaricus*, *E. tauvina*, *E. coioides* and *E. suillus* from estuaries, particularly those in tropical Australia, refer to only two species, *E. coioides* and *E. malabaricus*. Both *E. coioides* and *E. malabaricus* are often found in estuaries (Randall *et al.*, 1990) and both are common in at least one estuary on the northeastern coast of tropical Australia (Sheaves, 1992; 1993; 1994). Unfortunately, due to the confusion in the literature, in most cases it is unclear to which species various studies refer.

1.3.2 *Epinephelus coioides* and *Epinephelus malabaricus*

Where it is unclear whether a particular study is referring to *E. malabaricus* or *E. coioides* the term "estuary cod" will be used.

Distribution:

Estuary cod are widely distributed within tropical Australia. *E. malabaricus*, *E. tauvina*, *E. coioides* or *E. suillus* have been reported from estuaries in the Dampier region of north-western Australia (Blaber *et al.*, 1985), on the northern coast of Australia (Collette, 1983; Blaber *et al.*, 1989) and on the north-eastern coast of Australia (Sheaves, 1992; 1994).

The within estuary distribution of estuary cod has received little study. Blaber *et al.* (1989) used gill nets to sample lower, middle and upper reaches of the Embley estuary. Estuary cod were collected only in the upper reaches. However, as the data are presented as biomass per hour per metre of net it is unclear whether or not the catch of estuary cod in the upper reaches represents any more than a few fish. Thus the biological importance of these data is unclear. Sheaves (1992) used fish traps to compare catches of *E. coioides* and *E. malabaricus* in three regions of Alligator Creek in north-eastern Australia and found quite different patterns of abundance for the two species. Substantial catches of *E. coioides* were made in the lower and middle regions of the creek but few were caught in the upper region. *E. malabaricus* on the other hand showed a steady increase in numbers from the lower parts of the creek to upstream areas. Thus the pattern of change of abundance of the two species was quite different, to the extent that *E. coioides* was the numerically dominant serranid in the lower parts of the creek while *E. malabaricus* was numerically dominant in upstream parts. This suggests underlying differences in the biology of the two species, possibly in the way they respond to physical conditions of estuaries.

Habitats and Movements:

Blaber *et al.* (1989) sampled in five habitat types in the Embley estuary in northern Australia. Estuary cod were recorded from mid-channel and seagrass habitats, but not from sandy mud beaches, intertidal mudflats or small mangrove creeks and inlets. This suggests that estuary cod tend to be associated with particular types of habitats. Data from fish trapping in Alligator Creek in north-eastern Australia (Sheaves, 1992) supports this. Both *E. coioides* and *E. malabaricus* were caught in significantly greater numbers in areas of submerged timber and mangrove prop-roots than in open areas. Thus high abundances of *E. coioides* and *E. malabaricus* occurred in habitats unlikely to be sampled efficiently by nets. Because of this it is likely that these species have been under-represented in studies using nets as their main sampling gear.

Mark-recapture data (Sheaves, 1993) show that in a tropical estuary both *E. coioides* and *E. malabaricus* demonstrate strong tendencies to remain at one site for considerable periods of time. As with *L. russelli,* most individuals were recaptured at the site of original capture, showed no tendency to move further from the site of capture with increasing time at liberty, no preferred direction of movement and little tendency to cross to the opposite bank of the estuary.

Life-history status:

As with tropical estuarine lutjanids, few studies have investigated or commented on the life-history status of serranids in estuaries. Where life history information has been presented both juveniles and adults have been reported (Blaber and Milton, 1990; Thollot *et al.*, 1990). Once again no criteria for the determination of life-history status were presented in either work.

Feeding:

Estuary cod from the Embley estuary have been found to feed heavily on crustaceans, with Brachyura the most commonly occurring item of prey (40% of stomachs), and Penaeidae (20%), Alpheidae (15%) and other crustaceans (25%) being important also (Salini *et al.*, 1990). Teleosts, the only other prey category reported occurred in only 15% of stomachs. The weight that can be placed on these data is unclear as only 20 individuals were examined. There is no information on the diet of estuary cod in other areas.

1.4 Conclusion

Due to their roles as large benthic predators (Parrish, 1987), and their commercial (Allen and Talbot, 1985; Gloerfelt-Tarp and Kailola, 1984) and recreational importance (Robertson and Duke, 1990a; Sheaves, 1993), serranids and lutjanids are important members of estuarine fish assemblages. Within the Indo-Pacific, estuarine habitats are under increasing pressure from forestry, agriculture, aquaculture and tourist related developments (Hatcher *et al.*, 1989). As these anthropogenic pressures are likely to impact on estuarine fish populations (Robertson and Duke, 1990a) a detailed understanding of the importance of estuaries to fishes is clearly of great importance.

A full understanding of the way in which lutjanids and serranids use estuaries is particularly important, as a number of authors have suggested that at least some members of these families utilise estuaries as juvenile habitats and nursery areas. Despite this suspected nursery role, hard evidence is not available for any species and a number of important questions remain unanswered.

This study first investigates the range of lutjanid and serranid species occurring in estuaries on a section of the north-eastern tropical coast of Australia. The study then considers in detail four species, *L. russelli*, *L. argentimaculatus*, *E. coioides* and *E. malabaricus*, that are common in estuaries on this section of coast, and addresses two questions:

1. *What life history stages occur in estuaries?*
2. *How are these lutjanids and serranids distributed within and between estuaries and how does this distribution vary over time?*

CHAPTER 2.

COMPARISON OF LUTJANID AND SERRANID FAUNAS IN ESTUARIES TO THOSE ON NEARSHORE REEFS

2.1 INTRODUCTION

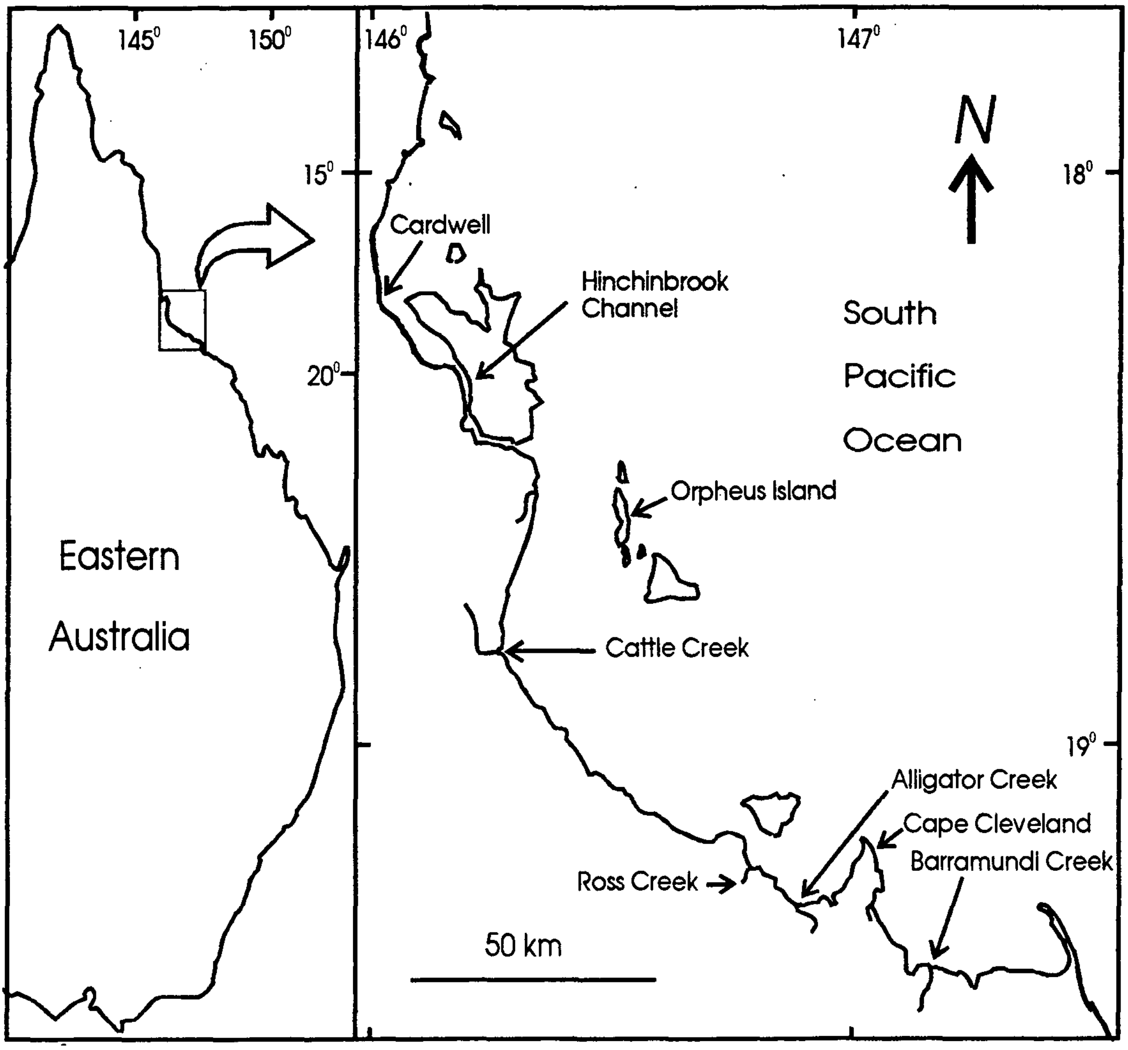
It is generally accepted that estuaries harbour qualitatively different fish faunas to coastal, near-shore and offshore habitats. While studies in tropical Australia have highlighted differences between estuaries and nearby areas (Blaber, 1980; Robertson and Duke, 1987), the use of nets as sampling tools has usually restricted sampling to areas of shallow water and prevented sampling in areas of high structural heterogeneity. In estuaries such areas, often composed of submerged mangrove prop roots or fallen timber, may contain species assemblages quite different to those found in shallow, clear areas (Sheaves, 1992). In particular, these areas may be important habitats for lutjanid and serranid fishes (Thayer *et al.*, 1987; Sheaves, 1992). Clearly, without considering these important habitats, any description of the lutjanid and serranid faunas of estuaries in tropical Australia would be incomplete.

In this study the lutjanid and serranid faunas of estuaries on the north-eastern tropical coast of Australia were compared to the faunas inhabiting near-shore rocky and coral reefs; areas that are important habitats for many species of lutjanids and serranids (Randall *et al.*, 1990). To allow useful comparisons to be made between habitats, one gear type - fish traps - was used in both areas. Fish traps have been employed successfully to sample lutjanid and serranid fishes both in estuaries (Sheaves, 1992; Sheaves, 1993; Sheaves, 1994) and on coral reefs (Davies, 1989) of north-eastern Australia. Furthermore, fish traps can be employed over a wide range of environmental conditions and in all depths of water.

2.2 METHODS

Between October 14, 1991 and August 25, 1993, samples of fish from the families Serranidae and Lutjanidae were collected from Cattle, Barramundi and Alligator Creeks (Fig. 2.1), three mangrove-lined estuaries on the north-eastern coast of tropical Australia (between 18° 15'S, 146° 15'E and 19° 25'S, 147° 10'E). The distance from the most northerly estuary (Cattle Creek) to the most southerly estuary

Figure 2.1


145⁰
150⁰
15⁰
20⁰
Eastern
Australia
146⁰
147⁰
18⁰
19⁰
N
Cardwell
Hinchinbrook
Channel
South
Pacific
Ocean
Orpheus Island
Cattle Creek
Alligator Creek
Cape Cleveland
Barramundi Creek
Ross Creek
50 km

(Barramundi Creek), is about 121km. Sampling was carried out using 12 Antillean Z fish traps (1800 X 1100 X 600mm, volume approx. 0.92m^3) with straight entrance funnels and 11.5mm square galvanised steel mesh. Details of the design of these traps can be found in Sheaves (1994) and chapter 6.2.2 (see Fig. 6.3).

At each set all traps were baited with 500g of Western Australian Blue Pilchards (*Sardinops neopilchardus*) as described in Sheaves (1994) and chapter 6.2.2. Sampling was carried out over the navigable length of these estuaries with a total of 1440, 3½ hour daytime trap soaks and 720 overnight trap soaks being completed. As this chapter is only a simple description of lutjanid and serranid fauna the sampling design will not be presented in detail. A detailed description of the sampling methodology is presented in chapter 6.2.2. Sampling was confined to the areas of high structural heterogeneity afforded by mangrove prop-roots and fallen timber. In a smaller study (Chapter 5) in Alligator Creek the same traps were also used to sample three other major habitat types - (i) areas of bank devoid of structural heterogeneity, (ii) mud banks and (iii) mid channels. As no species apart from those trapped in the larger study were captured the composition of the catch in the smaller study is not reported here.

The same traps were used to sample serranid and lutjanid fishes from fringing reefs around small rocky islands within 1 km of the shore at Cape Cleveland (240 trap soaks) and at Orpheus Island some 19 km offshore (192 trap soaks). These sampling sites were located offshore of the estuaries sampled (Fig. 2.1). In sampling reef habitats the traps were set-up, baited and managed in the same way as in the estuary, with the exception of the selection of trapping sites and the lengths of soak. The traps were placed haphazardly on the reefs. However, as trapping was carried out on fringing reefs close to islands different sites could be chosen for each trap soak. Traps were set with the aid of an echo sounder. Thus it was possible to ensure that traps were placed on reef. Most traps were set between 3m and 9m but some were set as deep as 18m. Two soak times were used; all day (from about 0600-0730hrs to about 1630-1800hrs) and overnight (from about 1630-1800hrs to about 0600-0730hrs).

Whenever possible additional samples of fish were collected from estuaries by hook and line fishing using either bait (fish, squid or crustacean) or artificial lures. At every opportunity the catches of anglers, professional fishers and crab trappers operating in estuaries were examined and any serranids and lutjanids identified. Data from both these samples were collected from many estuaries between Barramundi Creek and Cardwell (Fig. 2.1).

At least one fish of each species was retained and identified with reference to Allen and Talbot (1985) [lutjanids] or Randall and Heemstra (1991). The fork length of most individuals (particularly small and large individuals) was measured to the nearest millimetre.

2.3 RESULTS

During the study 614 serranids were trapped in the three estuaries, with two species - *Epinephelus coioides* and *E. malabaricus* - from a single genus, making up the entire catch (Table 2.1). In contrast, the 61 serranids trapped from near-shore reefs comprised 11 species from 4 genera (Table 2.1).

Table 2.1: Summary of catches of family Serranidae from near-shore reefs and estuaries using Antillean-Z fish traps.

Location codes: Or = Orpheus Island, CC = Cape Cleveland, Ca = Cattle Creek, Ba = Barramundi Creek, Al = Alligator Creek.

	Reefs			Estuaries		
Species	Fork length range (mm)	N°	Location	Fork length range (mm)	N°	Location
Cephalopholis boenak	123-140	3	Or	-	-	-
Cephalopholis cyanostigma	215-258	4	Or	-	-	-
Cromileptes altivelis	176-445	7	Or	-	-	-
Epinephelus coioides	443-915	8	Or, CC	120-500	280	Ca, Ba, Al
Epinephelus fuscoguttatus	450	1	Or	-	-	-
Epinephelus malabaricus	582-762	5	Or, CC	122-619	334	Ca, Ba, Al
Epinephelus merra	198-286	8	Or, CC	-	-	-
Epinephelus caeruleopunctatus	184-338	10	Or	-	-	-
Epinephelus quoyanus	255-291	3	Or, CC	-	-	-
Plectropomus leopardus	510-628	8	Or	-	-	-
Plectropomus maculatus	298-631	12	Or	-	-	-

Similarly, while the catch of 2,252 lutjanids from estuaries was composed of only 4 species (predominantly *Lutjanus russelli*), the 196 lutjanids from near-shore reefs came from 9 species (Table 2.2).

Table 2.2: Summary of catches of family Lutjanidae from near-shore reefs and estuaries using Antillean-Z fish traps.

Location codes: Or = Orpheus Island, CC = Cape Cleveland, Ca = Cattle Creek, Ba = Barramundi Creek, Al = Alligator Creek.

	Reefs			Estuaries		
Species	Fork length range (mm)	N°	Location	Fork length range (mm)	N°	Location
Lutjanus argentimaculatus	-	-	-	71-212	5	Ca, Al
Lutjanus carponotatus	161-339	101	Or, CC	-	-	-
Lutjanus fulviflamma	223	1	Or	54-110	19	Ba, Al
Lutjanus johnii	-	-	-	97-225	5	Ca, Ba
Lutjanus lemniscatus	312-314	2	Or	-	-	-
Lutjanus lutjanus	239	1	Or	-	-	-
Lutjanus malabaricus	264-320	4	Or	-	-	-
Lutjanus quinquelineatus	189-195	2	Or	-	-	-
Lutjanus russelli	99-321	30	CC	28-220	2223	Ca, Ba, Al
Lutjanus sebae	177-333	28	Or	-	-	-
Lutjanus vitta	194-265	27	Or	-	-	-

Angling catches provided another 5 species of serranids and 1 species of lutjanid from estuaries (Table 2.3). One individual of 1 additional species of serranid, *Epinephelus lanceolatus* (415mm FL), was seen in the catch of a crab trapper. Of the additional species of serranids all the *Cephalopholis boenak*, *Epinephelus quoyanus* and *Epinephelus sexfasciatus* were caught in the Hinchinbrook Channel (Fig. 2.1), and 10 of the 15 *Lutjanus fulviflamma* and all of the *Lutjanus lemniscatus* were captured from breakwaters in the mouth of Ross Creek (Fig. 2.1) which has been developed into the port of Townsville. With the exception of *Lutjanus argentimaculatus*, the species that were trapped in the largest numbers in estuaries also comprised the bulk of the angling catches.

Table 2.3: Summary of catches of families Serranidae and Lutjanidae from estuaries using hook and line.

The number of each species, their range of fork lengths and the number of estuaries from which they were sampled are presented.

! Species not captured in estuaries by fish traps during this study.

* All *Cephalopholis boenak*, *Epinephelus quoyanus* and *Epinephelus sexfasciatus* were caught in the Hinchinbrook Channel.

Ten of the *Lutjanus fulviflamma* and all of the *Lutjanus lemniscatus* were captured from breakwaters in the mouth of Ross Creek.

Species	N° of estuaries	N° of fish	Range of fork lengths (mm)
Cephalopholis boenak !	1	1*	102
Epinephelus coioides	10	59	171 - 386
Epinephelus corallicola !	2	2	180 - 277
Epinephelus malabaricus	13	146	180 - 515
Epinephelus caeruleopunctatus !	1	2	213
Epinephelus quoyanus !	1	2 *	268
Epinephelus sexfasciatus !	1	8 *	107 - 154
Lutjanus argentimaculatus	15	244	147 - 541
Lutjanus fulviflamma	3	15 #	120 - 142
Lutjanus johnii	1	10	105 - 450
Lutjanus lemniscatus !	1	6 #	95 - 107
Lutjanus russelli	7	114	130 - 232

2.4 DISCUSSION

Extensive fish trapping (2160 soaks) in 3 estuaries in tropical north-eastern Australia produced only 2 species of serranids and 4 species of lutjanids. This contrasted with the 11 species of serranids and 9 species of lutjanids collected with the same fish traps from much more restricted sampling (432 soaks) of reefal areas offshore of the estuaries. Thus it seems that many species common on near-shore reefs were absent from the estuaries. It appears evident that the serranid and lutjanid species found in these estuaries constitute a restricted sub-set of the species assemblage found in offshore areas. Even though 6 additional species of serranids and 1 extra species of lutjanid were collected by angling or observed in commercial or recreational catches from estuaries, all were represented by one or a few individuals only.

Previous studies of estuaries in tropical northeastern Australia (Blaber, 1980; Blaber *et al.*, 1989; Blaber *et al.*, 1990b; Robertson, 1988; Robertson and Duke, 1990a; Robertson and Duke, 1990b; Russell and Garrett, 1983; Sheaves, 1992; Sheaves, 1993; Sheaves, 1994) have together reported seven species recorded in the present study. These are *L. argentimaculatus, L. fulviflamma, L. johnii, L. russelli, E. coioides, E. malabaricus* and *E. lanceolatus.* The only species reported from estuaries in these studies not recorded in the present study were *Centrogenys vaigiensis* and *Epinephelus merra* both reported from the Embley estuary (Blaber *et al.*, 1990b). *Epinephelus suillus* was also reported in some studies (Blaber *et al.*, 1989; Blaber *et al.*, 1990b) but this is a synonym of *E. coioides* (Randall *et al.*, 1990). The seven species recorded in the present study that were not reported in the previous studies, *C. boenak, E. corallicola, E. caeruleopunctatus, E. quoyanus, E. sexfasciatus* and *L. lemniscatus*, were all caught only occasionally during this study (Table 2.3). Moreover, *C. boenak, E. quoyanus,* and *E. sexfasciatus* were only captured in the Hinchinbrook channel, while *L. lemniscatus* was only captured from breakwaters in the mouth of Ross Creek. Both of these sites could not be considered typical estuaries. The Hinchinbrook Channel (Fig. 2.1) is a narrow body of deep water, open to the ocean at both ends, and is flushed through with ex-estuarine water on each tidal cycle. So, although bordered by mangroves, the physical environment of the Hinchinbrook Channel probably tends to be closer to offshore areas than that of many estuaries. The breakwaters at the mouth of Ross Creek form part of the shore of a dredged harbour basin which probably acts as a reservoir of water more characteristic of near-shore water than of estuaries. It seems probable, therefore, that the presence of at least some of the species newly recorded from estuaries in the present study, may be the result of sampling in atypical locations. This together with the scarcity of these species in both this and previous studies suggests that these are probably not typical estuary species.

Each of the species recorded in the highest numbers in the present study, *L. argentimaculatus, L. russelli, E. coioides* and *E. malabaricus*, was also reported from an extensive study of the Embley estuary (Blaber 1980; Blaber *et al.*, 1989). Given the large spatial separation of the Embley estuary and the estuaries in the present study these species are apparently common inhabitants of estuaries over a large part of tropical north-eastern Australia.

When comparing different areas or habitats it is important to employ the same sampling methodology in each area. While the same traps, baited in the same way, were used both in estuaries and on reefs there were some methodological differences. Firstly, due to the different physical natures of the two environments different methods of selecting trapping sites were employed. Secondly, only a single all-day soak was used compared to two 3½ hour soaks in the estuaries. As species compositions rather than numbers were considered in this study it would seem unlikely that these methodological differences had any great bearing on the results.

From the results of this and previous studies it appears that, compared to nearby reef waters, estuaries in tropical north-eastern Australia contain characteristic, but depauperate serranid and lutjanid faunas. This restricted group of "typical" estuarine species is bolstered by a number of other species that occur infrequently and in some cases may be confined mainly to estuarine areas exhibiting "atypical" conditions. In sub-tropical waters of the southeastern United States a comparable situation exists. While at least eight species of lutjanids occur in inshore waters only one, *Lutjanus griseus*, is common in most seagrass and mangrove areas (Starck and Schroeder, 1970).

CHAPTER 3.0

SIZE, AGE AND GROWTH OF *LUTJANUS ARGENTIMACULATUS*, *L. RUSSELLI*, *EPINEPHELUS COIOIDES* AND *E. MALABARICUS* IN ESTUARIES IN TROPICAL NORTH-EASTERN AUSTRALIA.

3.1 INTRODUCTION

A range of methods are available to estimate the age of fishes. All are subjective and most have been criticised for one reason or another. While mark-recapture studies have been used to produce age estimates, such estimates are often considered unreliable. Information from recaptured fish is usually available over part of the size range only, and tagging has the potential to affect fish growth (Francis, 1988). The modal structure in length-frequency data is often used to assign ages to fish (Pollock, 1982; Wright *et al.*, 1986). However, age modes can be very difficult to identify, especially for larger sizes of long lived fishes (due to variations in growth between fish), and in fish with extended reproductive and settlement periods. This may lead to poor estimates of age and growth (Manooch, 1987; Ferreira and Vooren, 1991). Direct estimates of age can be obtained from check marks on scales. However, ring structure at the periphery of scales can be difficult to interpret and scales may be lost and replaced or partially reabsorbed during times of starvation (Campana and Neilson, 1985; Talbot and Doyle, 1992). Otoliths often also display interpretable ring structures. Annual rings may be counted in whole otoliths. For older fish with larger, thicker otoliths the rings may become more difficult to differentiate, leading to underestimates of the ages of larger fish (Beamish, 1979; Boehlert, 1985). Where possible, counts of rings in sectioned otoliths are usually the preferred option, generally giving the most reliable results (Beamish, 1979; Hoyer *et al.*, 1985). While counts of rings in sectioned otoliths are still subjective (for instance; what is to be considered a ring?) this technique has been applied widely to serranid and lutjanid fishes with reliable results (Manooch and Drennon, 1987; Bullock *et al.* 1992; Ferreira and Russ, 1992).

In recent years determining the age of fish has become a common practice (Paul, 1992). Indeed there are many reasons why knowing the age of fishes may be important. For example, estimates of age are important in fisheries biology. Whether fisheries managers are dealing with a short or long lived species will determine the range of management strategies they can employ (Hilborn and Walters, 1992). At another level, many fisheries models require detailed information on age, growth and mortality rates (Pauly, 1984).

A knowledge of age is also important in a more general biological context. Many aspects of the life-histories of fishes implicitly involve questions of age. At what age does a species reproduce? How many years do fish remain reproductively active? How long do fish remain in one location or one habitat? The answers to such questions are basic to the understanding of the biology of any species. It is in the context of life-histories that the question of age is important to the present study of estuarine lutjanids and serranids.

In this study, the age and size structures of *E. coioides*, *E. malabaricus*, *L. argentimaculatus* and *L. russelli* in estuaries in tropical north-eastern Australia are investigated and compared to age and size structures of these species in near-shore waters.

3.2 METHODS

3.2.1 Collection of samples

Between October 1990 and November 1994 specimens of *E. coioides*, *E. malabaricus*, *L. argentimaculatus* and *L. russelli* were collected to provide size structure and otolith samples. Data for each month from different years were pooled.

Estuary samples were collected from fish trap catches supplemented by line caught fish, either captured during research or supplied by anglers. Between October 1991 and August 1993 extensive fish trapping was conducted in Cattle, Barramundi and Alligator Creeks using Antillean-Z fish traps (Fig. 2.1). This sampling spanned about 145km of coast and consisted of 2,736 trap sets. Trap sampling details are given in chapters 5 and 6. Line caught fish originated from a large number of estuaries between Hinchinbrook Island (approximately 18° 10'S, 146° 10'E) and Barramundi Creek (19° 25'S, 147° 10'E), a length of coast extending approximately

220km. In an attempt to ensure that fish of the largest sizes available in estuaries were obtained, all line fishing was aimed at catching large individuals, and pamphlets requesting large fish of the required species were distributed through angling clubs and displayed in fishing tackle shops. As well, requests for samples were made directly to anglers encountered in the field.

Samples were also collected from waters offshore of the estuaries studied. Sampling was conducted on offshore reefs around Cape Cleveland [a coastal headland] and Orpheus Island [a near-shore island] (Fig. 2.1), utilising the same fish traps used for estuary sampling. These samples were supplemented by fish from angler donations, trawl samples from Cleveland Bay (a coastal embayment), and fish collected during James Cook University and Australian Institute of Marine Science field trips.

3.2.2 Comparison of size structures estuary/offshore

Fork and standard lengths of all fish were measured to the nearest millimetre. For fish with truncate caudal fins (all *Epinephelus* spp. and some *Lutjanus* spp. individuals), fork length (FL) was defined as the length from the snout to the posterior edge of the centre of the spread caudal fin. For fish with emarginate caudal fins (some *Lutjanus* spp. individuals), fork length (FL) was defined as the length from the snout to the posterior edge of the caudal fork. The standard length (SL) was defined as the length from the snout to the posterior end of the vertebral column.

3.2.3 Growth from mark-recapture

In conjunction with fish trapping studies in Cattle, Barramundi and Alligator Creeks (Sheaves 1992, 1993, 1994, Chapter 6 and 7) between 14 October 1991 and 25 August 1993, *E. coioides*, *E. malabaricus*, *L. argentimaculatus* and *L. russelli* were tagged and released. Apart from fish required for dissection all *Epinephelus* spp. and *L. argentimaculatus* over 150mm and all *L. russelli* over 100mm were tagged and released. This pool of tagged fish was supplemented with fish captured by hook and line fishing. Upon capture and recapture the fork length of each fish was recorded to the nearest millimetre. To prevent the use of non-independent samples, where fish were recaptured on more than one occasion only data from the longest period at liberty was used in the analysis of growth.

As growth must be zero at time zero, for all species the regressions of growth on days at liberty were forced through the origin. Before calculating the regression equations any outlying points [externally studentized residuals >2 (Belsley *et al.*, 1980; p.28)] were omitted. These points may have represented fish growing at faster or slower rates than the bulk of the population. Alternately they may have represented measurement or recording errors or, in the case of slow growth rates, fish whose growth was adversely affected by the processes of handling and tagging. Whatever the reason, such atypical data were probably unrepresentative of the population as a whole. Because of their potential to greatly influence the regression relationships, any data points with the potential to be extremely influential [leverage > 2p/n; where p=number of explanatory variables and n=sample size (Belsley *et al.*, 1980; p.17)] were omitted from the analyses also. These were mainly fish recaptured after an unusually long period at liberty. The omitted points were plotted with the data actually used in the analyses. As the variable "days at liberty" was not under the control of the investigator, Model II regression was appropriate (Sokal and Rohlf, 1981; p.459). Thus, the reduced major axis (geometric mean) regression coefficient (b') and its standard error were calculated (Sokal and Rohlf, 1981; p.550) and used to calculate growth rates in preference to the simple least squares linear regression coefficient. However, as it is not valid to test the significance of the reduced major axis regression coefficient (Sokal and Rohlf, 1981; p.550) the results of significance tests on the simple least squares linear regression coefficient are presented.

3.2.4 Age determination using otoliths

During this study the sagittae of 108 *E. coioides*, 193 *E. malabaricus*, 300 *L. argentimaculatus*, and 427 *L. russelli* were used for age determination. Immediately upon capture the fish were placed on ice where they were kept until dissected (less than 12 hours). The sagittae of each fish were removed, washed and stored dry.

All otolith reading was conducted on sectioned sagittae. Right or left otoliths were selected at random for processing. Sectioning was carried out as described by Ferreira and Russ (1992). The number of opaque bands present in the sectioned otoliths were counted independently by two readers under a microscope at 40X magnification using transmitted light. If the counts of both readers agreed the otoliths were used for analysis. If the counts differed the counts were repeated by both original readers and a third reader. If the counts of at least two of the readers agreed this count was also accepted if the count of the third reader differed by no more than ± 1ring.

To determine if the opaque bands were annual marks, tetracycline marking of otoliths of tagged fish of all species was carried out at Alligator Creek (approx. 19° 20'S, 146° 55'E) between January 1991 and March 1994. The fish were collected while fish-trapping or caught on hook and line. Selecting only fish greater than 120mm FL, 310 *E. coioides*, 219 *E. malabaricus*, 193 *L. argentimaculatus*, and 560 *L. russelli* were marked with T-bar anchor tags, injected with oxytetracycline (dosage of 50mg per kilogram of body weight) into the coelomic cavity, and released. Recaptured tetracycline marked fish were processed and their otoliths prepared as described above for non-marked fish. The sectioned sagittae from these fish where observed microscopically under white light, ultraviolet light (UV) and a combination of both. Distances between the fluorescent tetracycline bands, the otolith margin and the outer edge of the opaque band were measured. Measurements were made from the outer edge of the opaque bands because at this point the discontinuity between opaque and translucent zones provided a sharp contrast (Ferreira and Russ, 1992).

3.3 RESULTS

3.3.1 Comparison of size structures estuary/offshore

It is assumed that for all four species the samples collected included fish of the largest sizes usually encountered in estuaries in the study area. It was clear from the angler collections that most anglers tended to donate fish when they caught what they considered to be a large fish of that species. Thus it is likely that few fish larger than those collected were caught from estuaries and not reported. During the study period no *L. argentimaculatus* or *L. russelli* larger than those caught in fish traps or donated by anglers were reported from estuaries. Similarly, no *E. coioides* or *E. malabaricus* of length greater than those captured in fish traps were reported to have been caught by anglers in estuaries. Notwithstanding this, during the study period anglers occasionally reported seeing very large *Epinephelus* spp. (estimated as > 1m) in various estuaries. Furthermore, while angling in Cattle Creek, the author observed a fish of approximately 1m in length, which was almost certainly either an *E. coioides* or an *E. malabaricus*, swallow an *L. argentimaculatus* of approximately 350mm that was at the surface on a fishing line.

Samples were much more difficult to obtaine from offshore areas than from estuaries. This is reflected in small offshore sample sizes (Table 3.1). Despite the small offshore samples, for *Epinephelus coioides*, *E. malabaricus* and *Lutjanus russelli*, fish of considerably larger sizes were trapped from offshore waters than from estuaries using the same fish traps. Apparently if larger fish of these species were available in these estuaries they could have been trapped. Thus, given the spatially and temporally extensive trapping undertaken in estuaries during this study, it seems likely that the maximum sizes trapped reflect the normal maximum sizes of these species inhabiting estuaries in tropical northeastern Australia. This is supported by the fact that no *E. coioides*, *E. malabaricus* or *L. russelli* larger than those trapped were obtained from anglers fishing in the estuaries where trapping was carried out. Furthermore, of these three species only one individual larger than the largest trapped (a line caught *L. russelli* [232mm FL] slightly largest than the largest trapped [220mm FL]) was obtained from any estuary within the study area. During the study *L. argentimaculatus* were trapped only occasionally in estuaries and not at all in off-shore waters. However, the maximum size recorded for line caught fish from estuaries was substantially smaller than the maximum size from offshore (Table 3.1).

Table 3.1: Size ranges of *E. coioides*, *E. malabaricus*, *L. argentimaculatus* and *L. russelli* caught in Antillean-Z fish traps, and supplied by anglers.

	Estuary		Off-shore	
	FL range (mm)	N°	FL range (mm)	N°
E. coioides				
Trap	120-500	280	443-915	8
Angler collection	171-386	72	645-1085	7
E. malabaricus				
Trap	122-619	334	582-762	5
Angler collection	180-515	153	523-1199	5
L. argentimaculatus				
Trap	71-212	5	-	-
Angler collection	147-541	301	412-890	26
L. russelli				
Trap	28-220	2223	99-321	30
Angler collection	130-232	154	189-445	202

3.3.2 Growth from mark-recapture

Capture and recapture data for *E. coioides*, *E. malabaricus*, *L. argentimaculatus* and *L. russelli* are summarised in table 3.2. For each of the four species, growth rate (size increment per unit time) and mean fork length were not significantly correlated (Table 3.3). Thus for all species growth rate was independent of fork length.

Table 3.2: Summary of tagging and recapture data showing numbers of fish tagged and recaptured, the maximum days at liberty (DAL), and maximum growth between recaptures.

Species	N° tagged	N° recaptured	Max. DAL	Max. growth (mm)
Epinephelus coioides	398	104	619	129
Epinephelus malabaricus	293	63	728	202
Lutjanus argentimaculatus	120	10	395	66
Lutjanus russelli	1070	43	238	44

Table 3.3: Summary of the correlation between growth rate (size increment per unit time) and mean fork length (for the increment in question) for *Epinephelus* spp. and *Lutjanus* spp.

Species	r	n	p
Epinephelus coioides	-0.1018	104	0.3036
Epinephelus malabaricus	-0.0189	63	0.3957
Lutjanus argentimaculatus	-0.4478	10	0.1944
Lutjanus russelli	0.0448	43	0.7754

For all four species there was a strong linear relationship between growth and period at liberty (Table 3.4). In each case, both the reduced major axis regression coefficient (b′) and the simple least squares linear regression coefficient were very similar. This is to be expected as in each case the two variables were highly correlated (Sokal and Rohlf, 1981; p.550).

Table 3.4: Summary of the regression between growth rate and days at liberty for *Epinephelus* spp. and *Lutjanus* spp.
For each species the reduced major axis (geometric mean) regression coefficient (b′) and its standard error are presented together with the simple least squares linear regression coefficient and its associated significance tests, and the coefficient of determination (r^2).
The probability associated with the F statistic was < 0.0001 for all species.

Species	b′	S.E.	b	r^2	d.f.	F
Epinephelus coioides	0.2253	0.0072	0.2142	0.9042	95	896.4
Epinephelus malabaricus	0.2565	0.0100	0.2443	0.9070	61	595.1
Lutjanus argentimaculatus	0.1780	0.0074	0.1770	0.9896	5	568.7
Lutjanus russelli	0.1911	0.0107	0.1971	0.8783	38	281.4

For *E. coioides* interpolation suggested a growth rate of approximately 82mm per year. Eight points were not used in the regression analysis (Fig. 3.1). Five of these points had large standard residuals but still occurred reasonably close above or below the regression line. One fish exhibited apparent extremely rapid growth

Figure 3.1: The relationship between growth of *E.coioides* and time at liberty. The data used to compute the regression are represented by hollow circles and data omitted from the analysis by filled diamonds. The line represents the regression of growth on days at liberty using the reduced major axis regression coefficient (b').

Figure 3.2: The relationship between growth of *E.malabaricus* and time at liberty. The data used to compute the regression are represented by hollow circles and data omitted from the analysis by filled diamonds. The line represents the regression of growth on days at liberty using the reduced major axis regression coefficient (b').

Figure 3.1

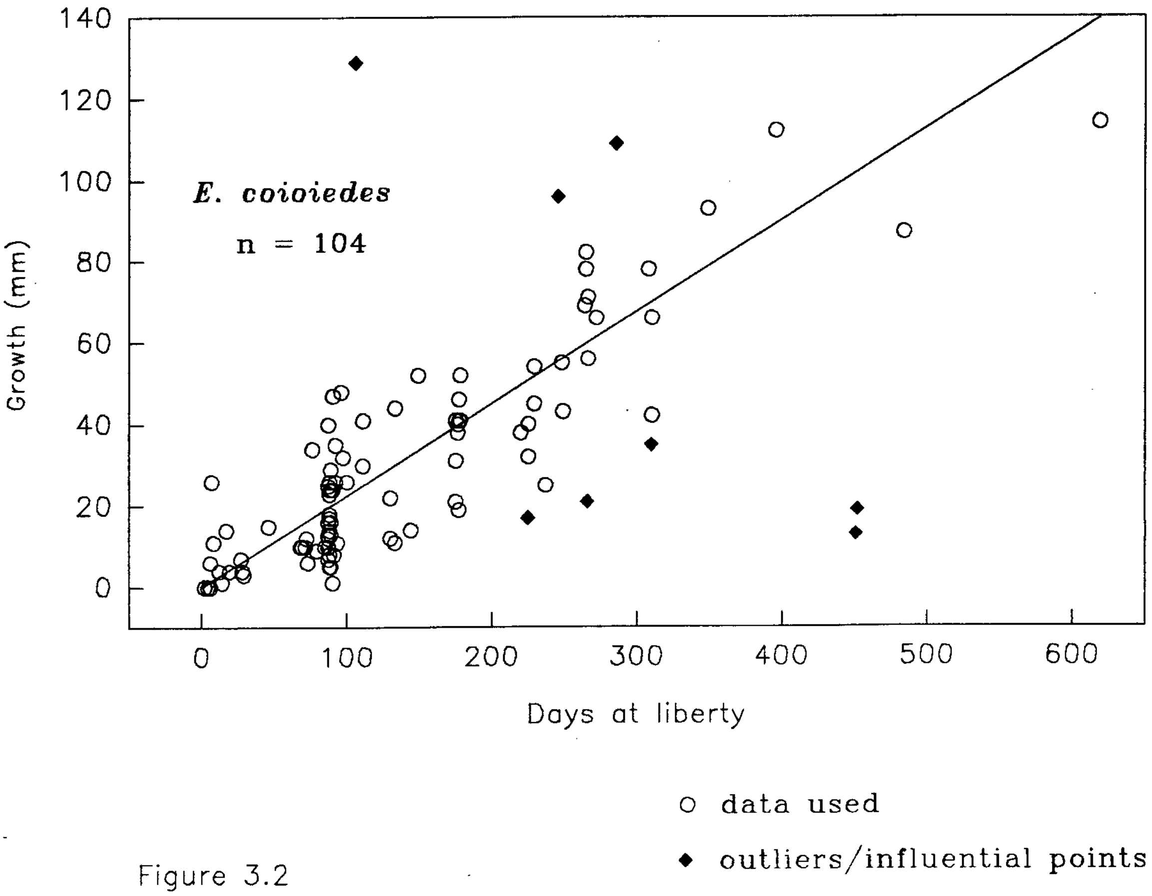
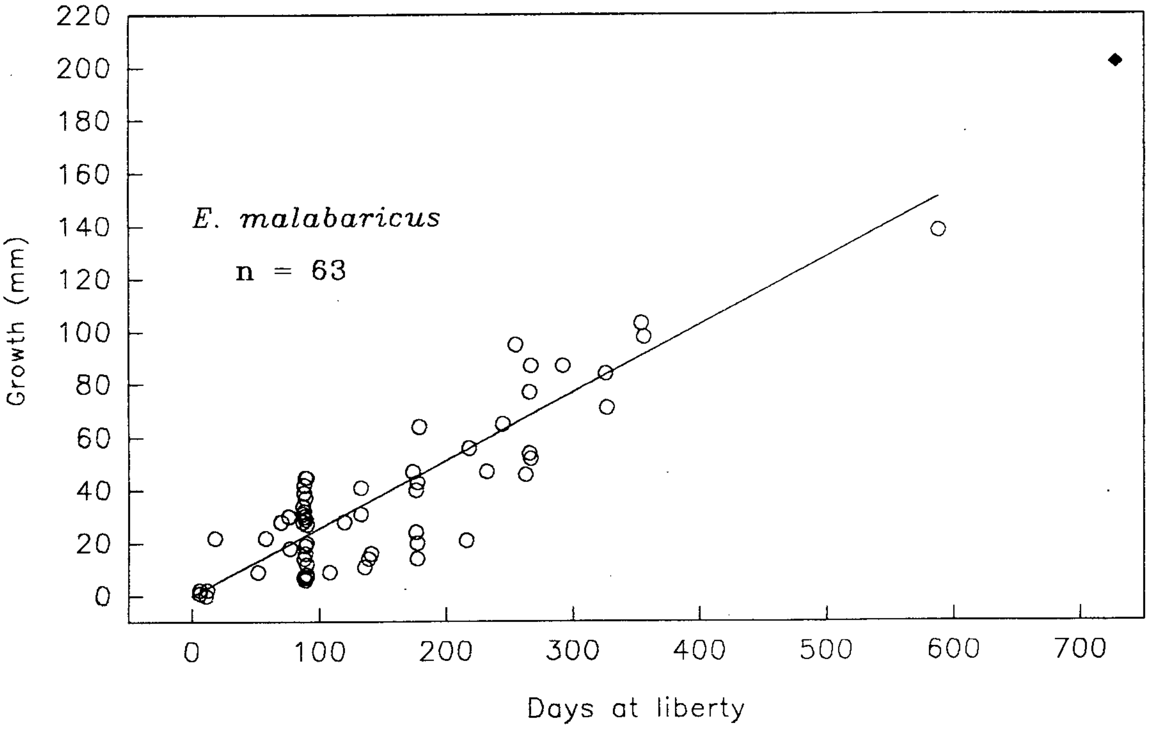


Figure 3.2



occurring far above the regression line. It seems most probable that this extreme growth rate was the result of faulty data recording. Two points occur close together well below the regression line and apparently represent very slow growth rates. While these points may be the result of faulty recording they may also represent fish that were adversely affected by the process of handling and tagging.

For *E. malabaricus* the interpolated growth was approximately 94mm per year. Only one point was omitted from the analysis (Fig. 3.2). This fish was recaptured after an unusually long period at liberty (728 days) and thus had the potential to be very influential in the analysis. When plotted with the rest of the data this point agreed well with the extrapolated regression line.

For *L. argentimaculatus* few recapture data (n = 10) were available. Three points were apparent outliers (low growth rates) and were omitted from the analysis (Fig. 3.3). A fourth point was potentially an influential point and so also omitted from the analysis. This point did, however, fit well with the extrapolated regression, suggesting that despite the small sample size the regression relationship may be reasonable. From the regression equation the growth of *L. argentimaculatus* was calculated at approximately 64.9mm per year.

From the regression relation for *L. russelli* growth was calculated at approximately 69.8mm per year. Three outlying points (2 above and 1 below the regression line) were omitted from the analysis (Fig. 3.4). A potentially influential data point for a fish at liberty for 238 days was also omitted from the analysis. When this point was plotted with the rest of the data it closely coincided with the extrapolated regression line.

3.3.3 Age determination using otoliths

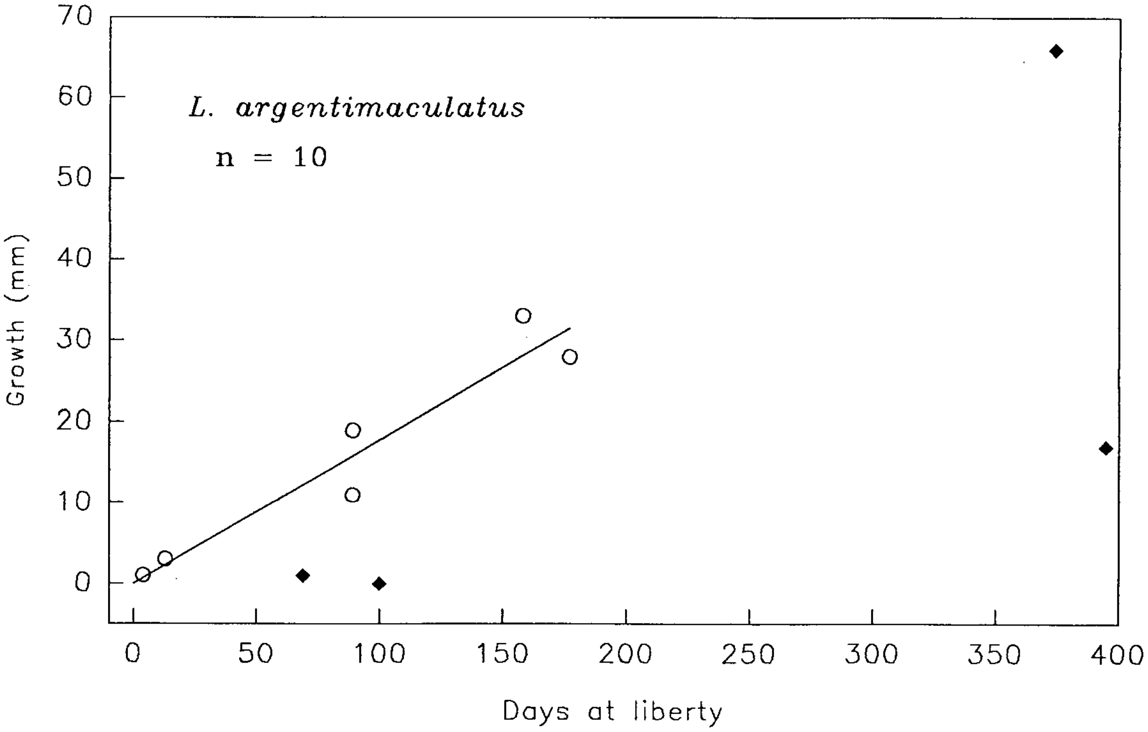
Of the sagittae examined 92 of the 108 *E. coioides* otoliths, 174 of the 193 *E. malabaricus* otoliths, 298 of the 300 *L. argentimaculatus* otoliths, and 423 of the 427 *L. russelli* otoliths were accepted for age determination.

The sectioned sagittae of each of the species displayed a pattern of alternating narrow opaque and broader translucent zones. Under transmitted light the opaque bands appeared as dark rings on a lighter (translucent) background. The clarity of definition of the dark, opaque bands varied between the species with those of *L.*

Figure 3.3: The relationship between growth of *L. argentimaculatus* and time at liberty. The data used to compute the regression are represented by hollow circles and data omitted from the analysis by filled diamonds. The line represents the regression of growth on days at liberty using the reduced major axis regression coefficient (b').

Figure 3.4: The relationship between growth of *L. russelli* and time at liberty. The data used to compute the regression are represented by hollow circles and data omitted from the analysis by filled diamonds. The line represents the regression of growth on days at liberty using the reduced major axis regression coefficient (b').

Figure 3.3



○ data used
◆ outliers/influential points

Figure 3.4

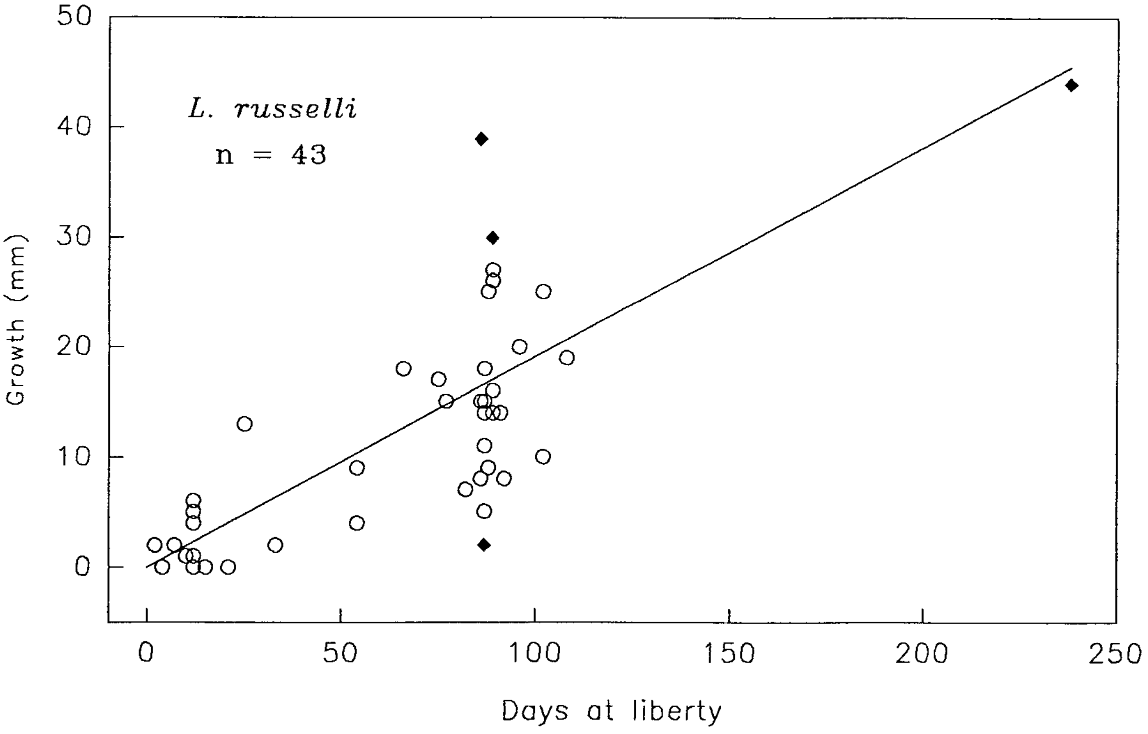
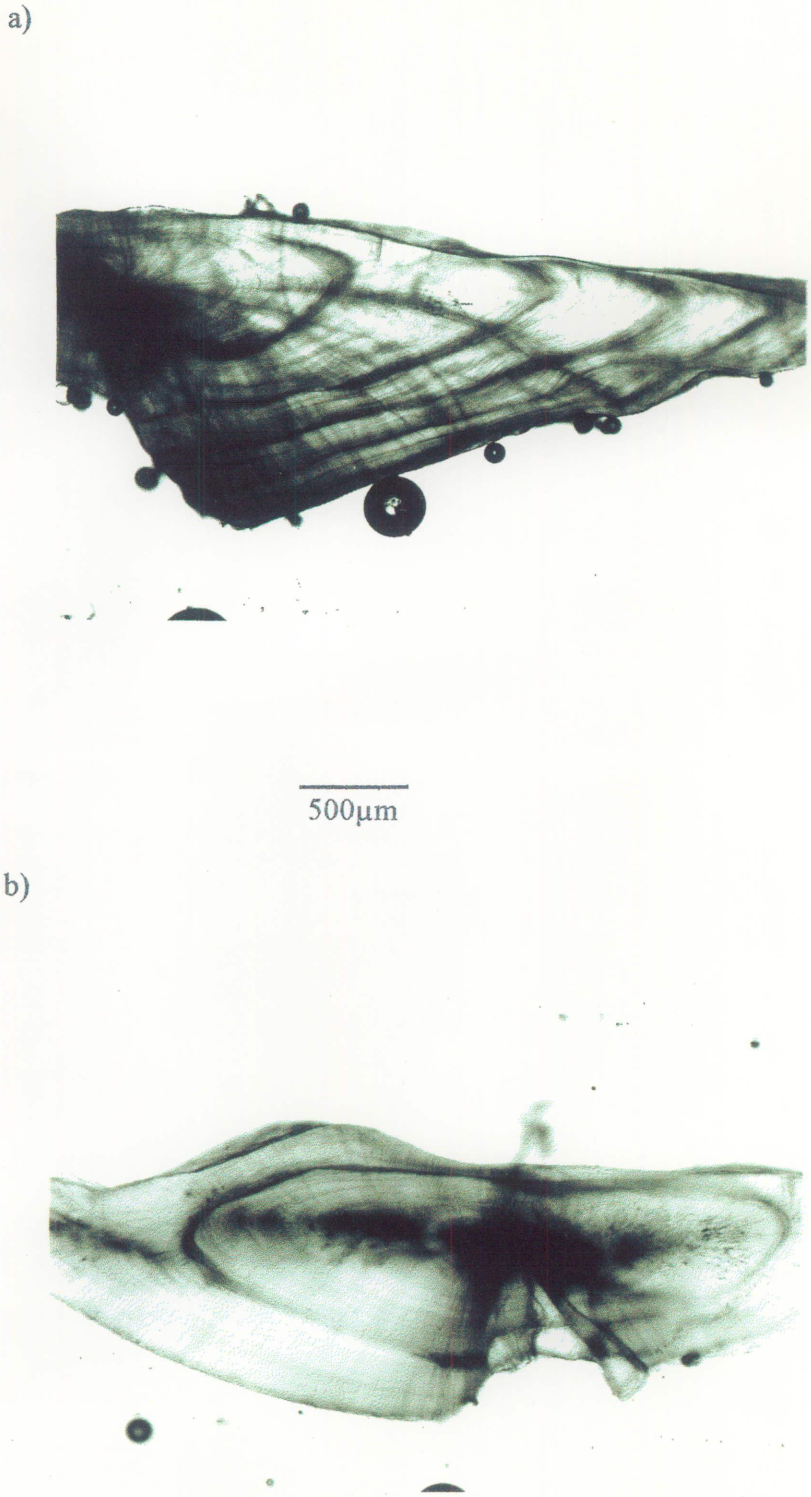


Figure 3.5: Transverse sections of sagittal otoliths of a) *L. argentimaculatus* [397mm FL, 4 rings] and b) *L. russelli* [172mm FL, 1 ring] showing annual banding.

Figure 3.5:



argentimaculatus (Fig. 3.5a) being very clearly defined, and *L. russelli* (Fig. 3.5b), *E. malabaricus* (Fig. 3.6a) and *E. coioides* (Fig. 3.6b) each in turn slightly less well defined.

Of the tetracycline marked fish 4 *E. coioides*, 5 *E. malabaricus*, 2 *L. argentimaculatus* and 1 *L. russelli* had been at liberty for more than a year. In section the sagittae from each of the *Epinephelus* spp. showed a single opaque band outside of the fluorescent tetracycline mark (Fig. 3.7 a,b), except for one *E. coioides* marked in October 1991 where the tetracycline mark coincided with an opaque band. Due to the positioning of the opaque bands relative to the tetracycline marks and the edge of the otolith it was concluded that both species of *Epinephelus* deposited the opaque bands between about June and December and that the opaque bands represented annuli. Although the sample sizes of *Lutjanus* spp. were small they too suggest that the opaque bands are annuli layed down between about June and December (Fig. 3.8 a,b). In fact, a *L. argentimaculatus* tetracycline marked both in October 1991 and October 1992 showed a single opaque band between the two fluorescent marks with both tetracycline marks situated close outside opaque bands. It should be noted that with the tetracyclined sagittae only two dates are known, the date of tetracycline marking and the date of recapture. Other structures can only be ascribed relative positions as actual measurements are distorted by such things as variations in growth, cutting and polishing angles.

Further evidence that the opaque bands represent annuli comes from growth rates from mark-recapture. For each species, when a regression line derived from growth from mark-recapture data and centred on the mean size and count from otolith data, is plotted with the otolith count data, the growth rate (slope) from mark-recapture corresponds well with the growth between age classes from otolith ageing (Figs. 3.9,10,11,12). The production of the opaque annuli in the June to December period (particularly the later September to December part) is supported by the observation that for each species the most peripheral annulus was at or close to the edge of the otolith for most fish collected between September and December.

Figure 3.6: Transverse sections of sagittal otoliths of a) *E. malabaricus* [410mm FL, 5 rings] and b) *E. coioides* [393mm FL, 4 rings] showing annual banding.

Figure 3.6:

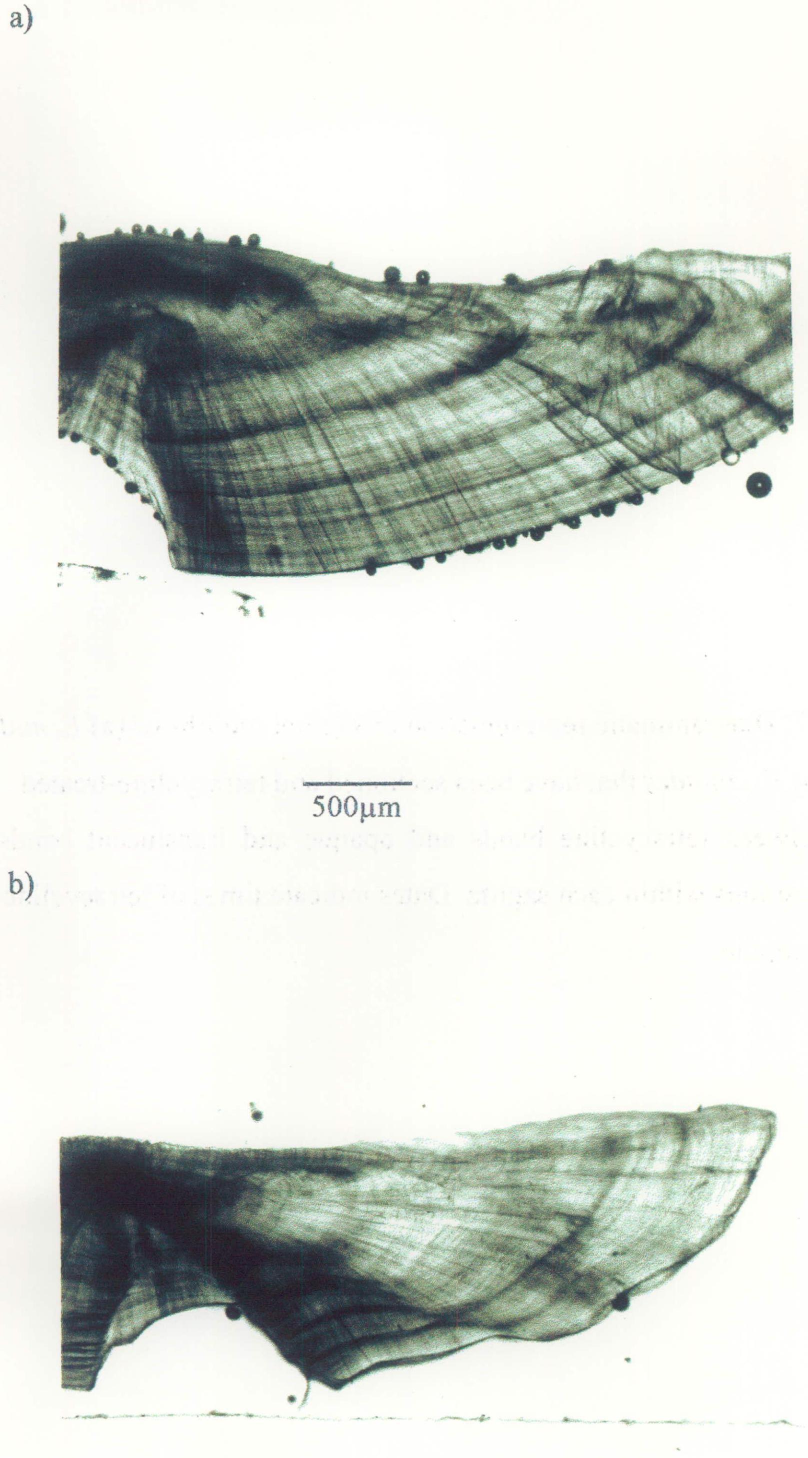
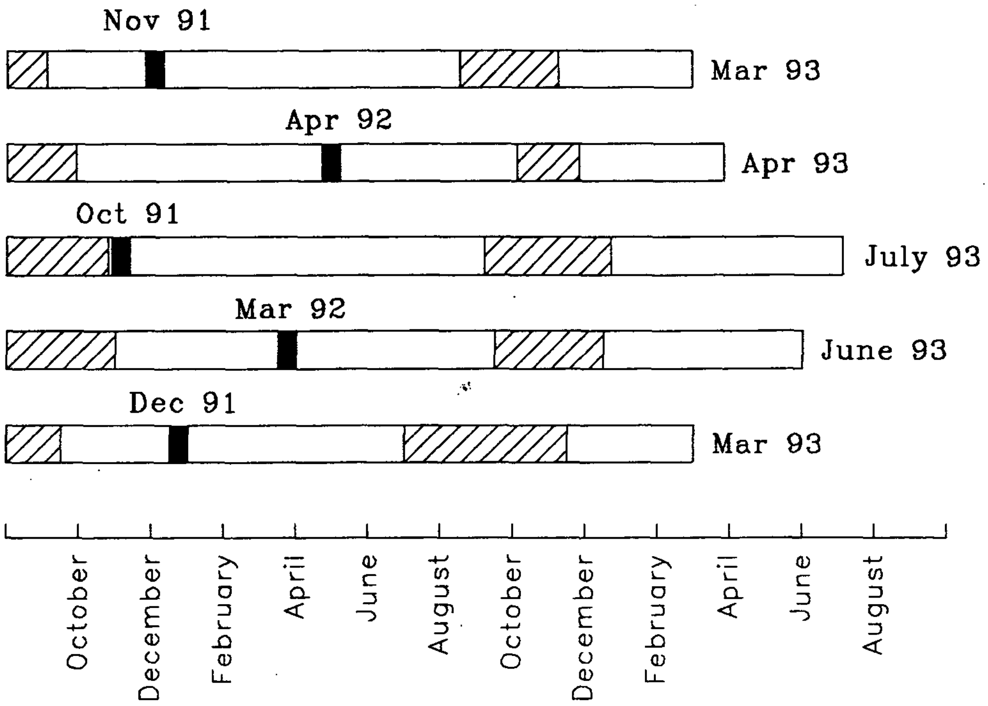


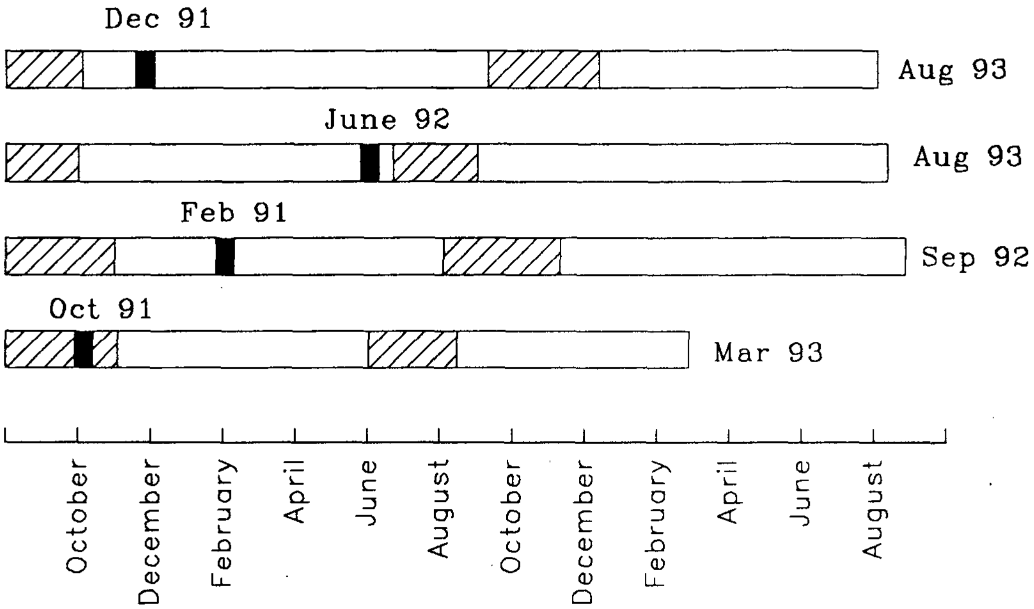
Figure 3.7: Diagrammatic representation of sagittal otoliths of (a) *E. malabaricus* and (b) *E. coioides* that have been sectioned and tetracycline-treated. The distances between tetracycline bands and opaque and translucent bands are relative positions within each sagitta. Dates indicate times of tetracycline marking and recapture.

Figure 3.7:

(a) *Epinephelus malabaricus*



(b) *Epinephelus coioides*

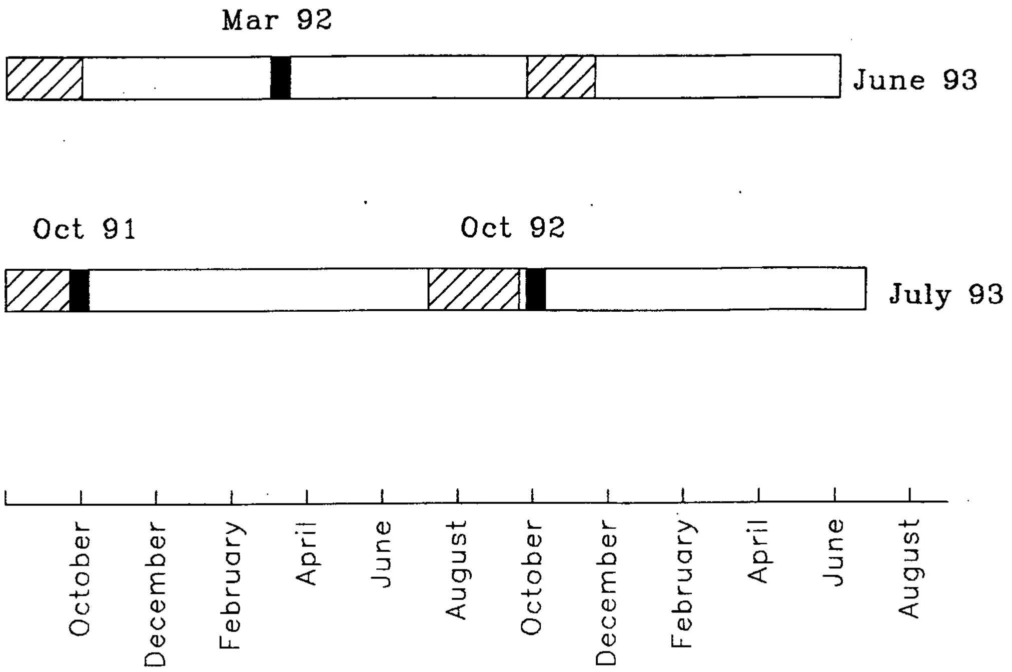


▨ opaque □ transluscent ■ tetracycline

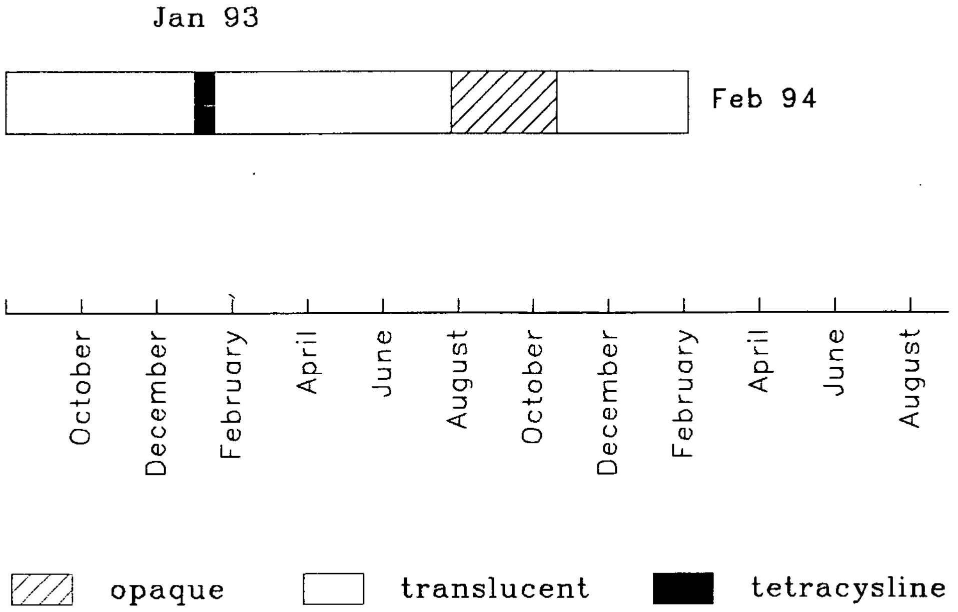
Figure 3.8: Diagrammatic representation of tetracycline-treated sectioned sagittal otoliths of (a) *L. argentimaculatus* and (b) *L. russelli*. The distances between tetracycline bands and opaque and translucent bands are relative positions within each sagitta. Dates indicate time of tetracycline marking and recapture.

Figure 3.8:

(a) *Lutuanus argentimaculatus*



(b) *Lutjanus russelli*



3.3.3.1 Estuary samples

From estuary samples, *E. coioides* (n = 87) displayed between 1 and 5 annual rings (Fig. 3.9), *E. malabaricus* (n = 171) between 0 and 7 rings (Fig. 3.10), *L. argentimaculatus* (n = 276) between 0 and 8 rings (Fig. 3.11), and *L. russelli* (n = 196) between 0 and 2 rings (Fig. 3.12). When fork length of *L. russelli* is plotted against otolith count for each month an increase in size for the age 1class from October to August is apparent, suggesting ring formation about September (Fig. 3.13). For each species there was a broad variation in sizes at each count, however, there was a general trend for the number of annuli to increase with increasing fish size.

3.3.3.2 Offshore samples

Few samples of *E. coioides* [n=5] (Fig. 3.9) or *E. malabaricus* [n=3] (Fig. 3.10) were available from offshore. However, for each species the largest individual from offshore displayed more rings (*E. coioides* 16; *E. malabaricus* 8) than did the largest estuary fish. More extensive samples of *L. argentimaculatus* [n=22] (Fig. 3.11) displayed ages up to 32 years with growth appearing to slow at greater ages. *L. russelli*, with an extensive offshore sample [n=227] (Fig. 3.12), showed a maximum age of approximately 17 years with growth appearing to slow and begin to asymptote after approximately 3 years. For both *L. argentimaculatus* and particularly *L. russelli* growth appears to slow shortly after the transition from estuarine to offshore habitats.

3.4 DISCUSSION

Even though fish trapping in estuaries in tropical north Queensland was extensive, both spatially and temporally, the maximum sizes of *E. coioides*, *E. malabaricus* and *L. russelli* trapped were considerably smaller than the largest sizes captured in the same traps from offshore waters. Similarly, collections from anglers fishing in estuaries produced relatively small individuals (Table 3.1). While *L. argentimaculatus* trapped poorly, considerable numbers of specimens were collected by angling or donated by anglers. All the fish comprising these samples were relatively small compared to fish obtained from offshore waters. As *L. argentimaculatus* is an important recreational sportfish in north Queensland estuaries and close contact was kept with local sportfishing clubs, it seems likely that few if

Figure 3.9: The relationship between fork length and sagittal otolith counts for *E. coioides*. Filled circles represent fish from estuaries while hollow triangles represent fish from offshore. The line represents the growth of *E. coioides* from mark-recapture data. The slope of the line is derived from the regression of fork length on time at liberty with the range representing the range of fork lengths of recaptured fish. The line is centred on the mean fork length and count (of estuary fish) derived from the otolith data.

Figure 3.10: The relationship between fork length and sagittal otolith counts for *E. malabaricus*. Filled circles represent fish from estuaries while hollow triangles represent fish from offshore. The line represents the growth of *E. malabaricus* from mark-recapture data. The slope of the line is derived from the regression of fork length on time at liberty with the range representing the range of fork lengths of recaptured fish. The line is centred on the mean fork length and count (of estuary fish) derived from the otolith data.

Figure 3.9

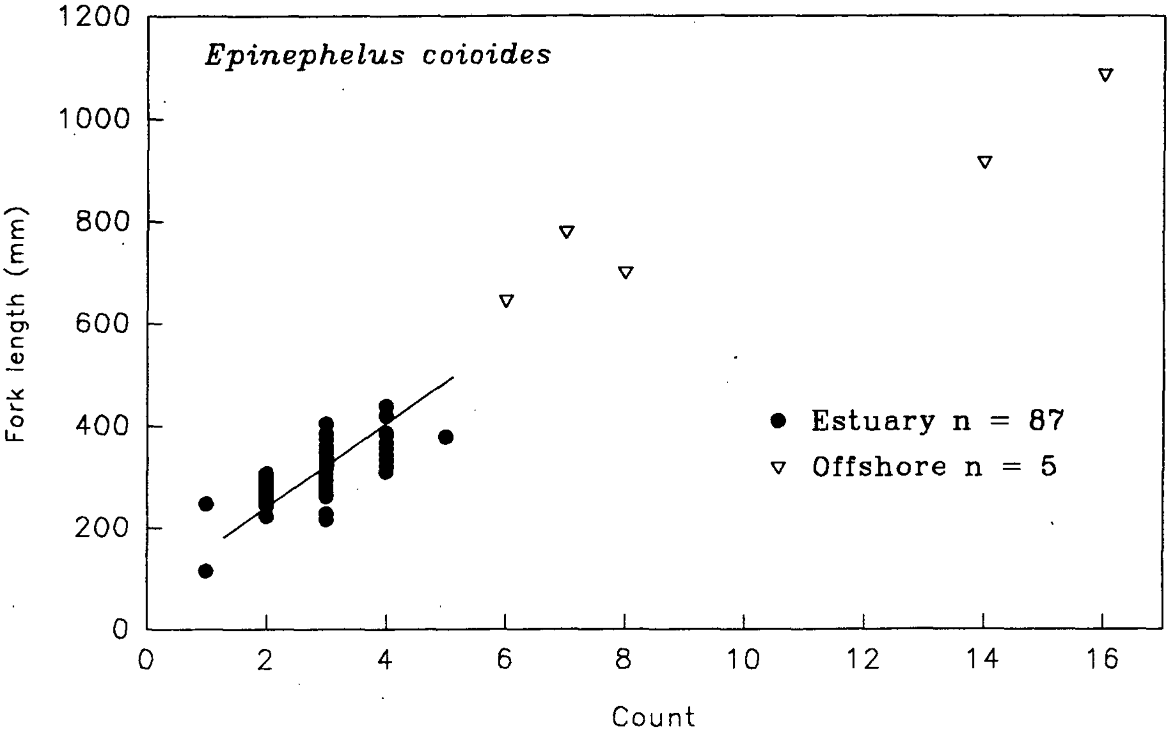


Figure 3.10

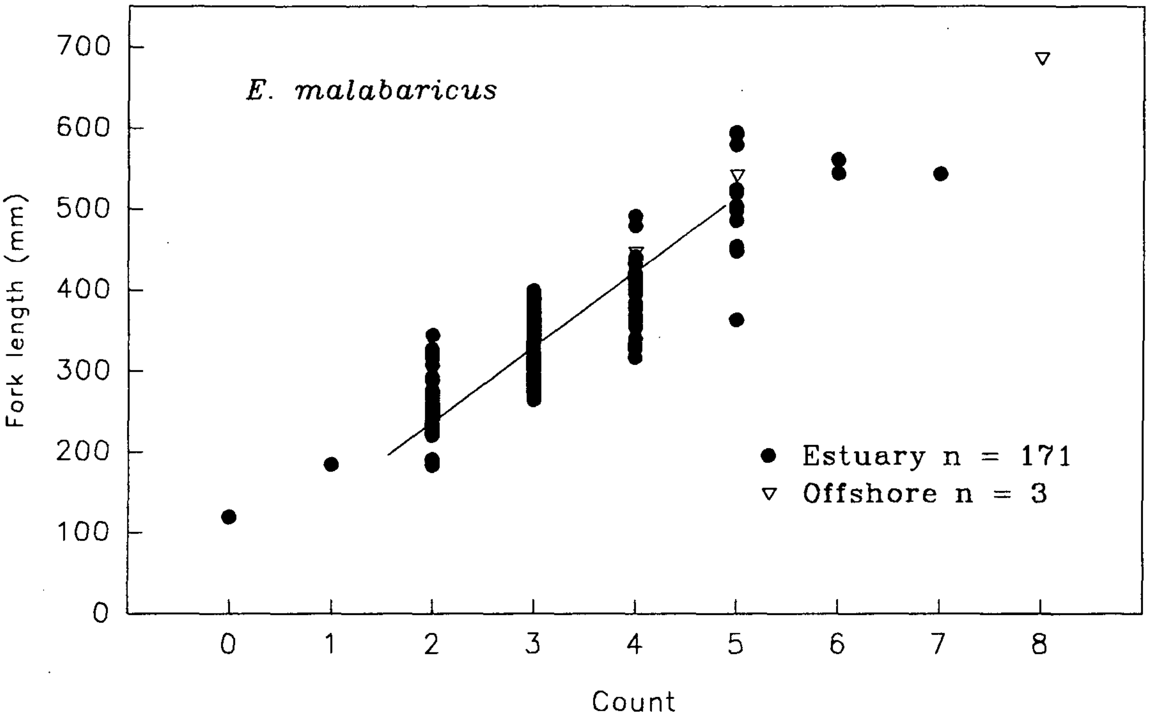


Figure 3.11: The relationship between fork length and sagittal otolith counts for *L. argentimaculatus*. Filled circles represent fish from estuaries while hollow triangles represent fish from offshore. The line represents the growth of *L. argentimaculatus* from mark-recapture data. The slope of the line is derived from the regression of fork length on time at liberty with the range representing the range of fork lengths of recaptured fish. The line is centred on the mean fork length and count (of estuary fish) derived from the otolith data.

Figure 3.12: The relationship between fork length and sagittal otolith counts for *L. russelli*. Filled circles represent fish from estuaries while hollow triangles represent fish from offshore. The line represents the growth of *L. russelli* from mark-recapture data. The slope of the line is derived from the regression of fork length on time at liberty with the range representing the range of fork lengths of recaptured fish. The line is centred on the mean fork length and count (of estuary fish) derived from the otolith data.

Figure 3.11:

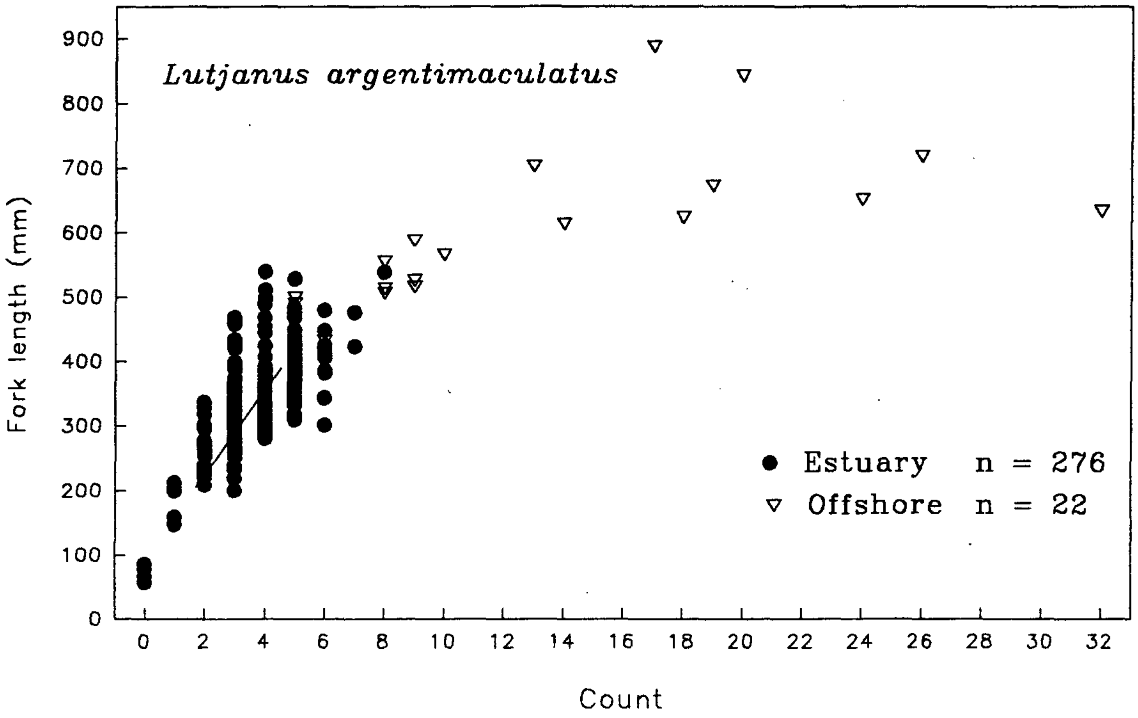


Figure 3.12:

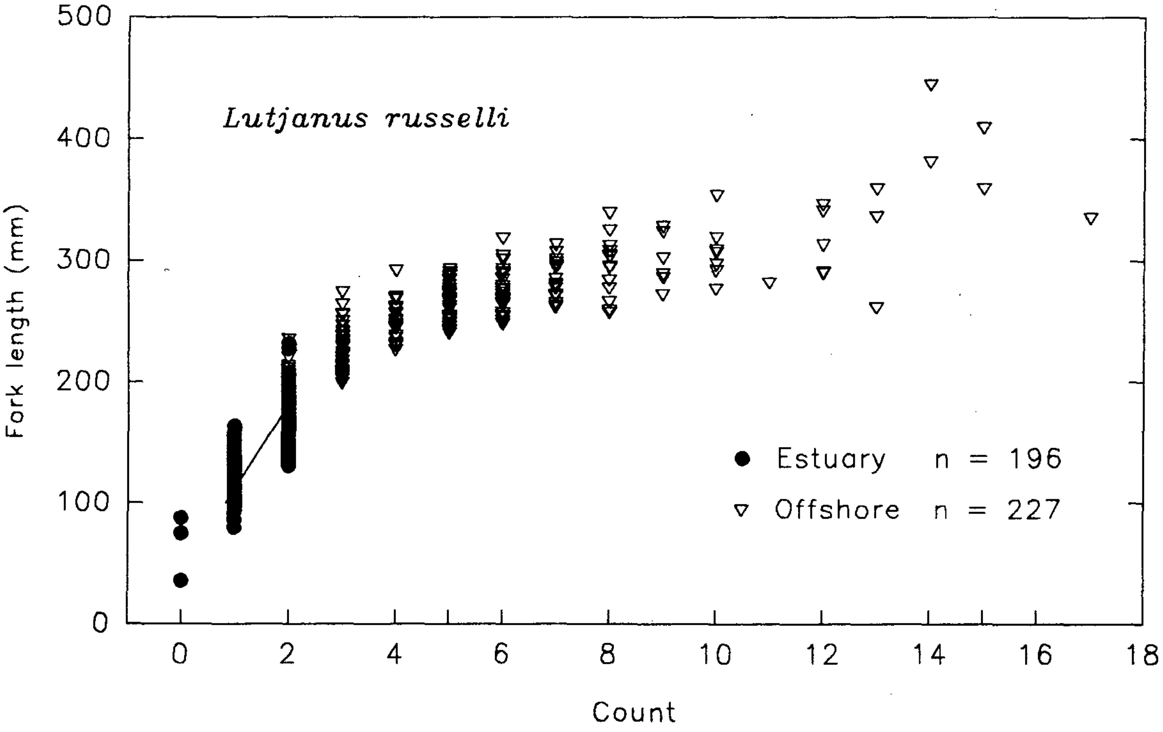
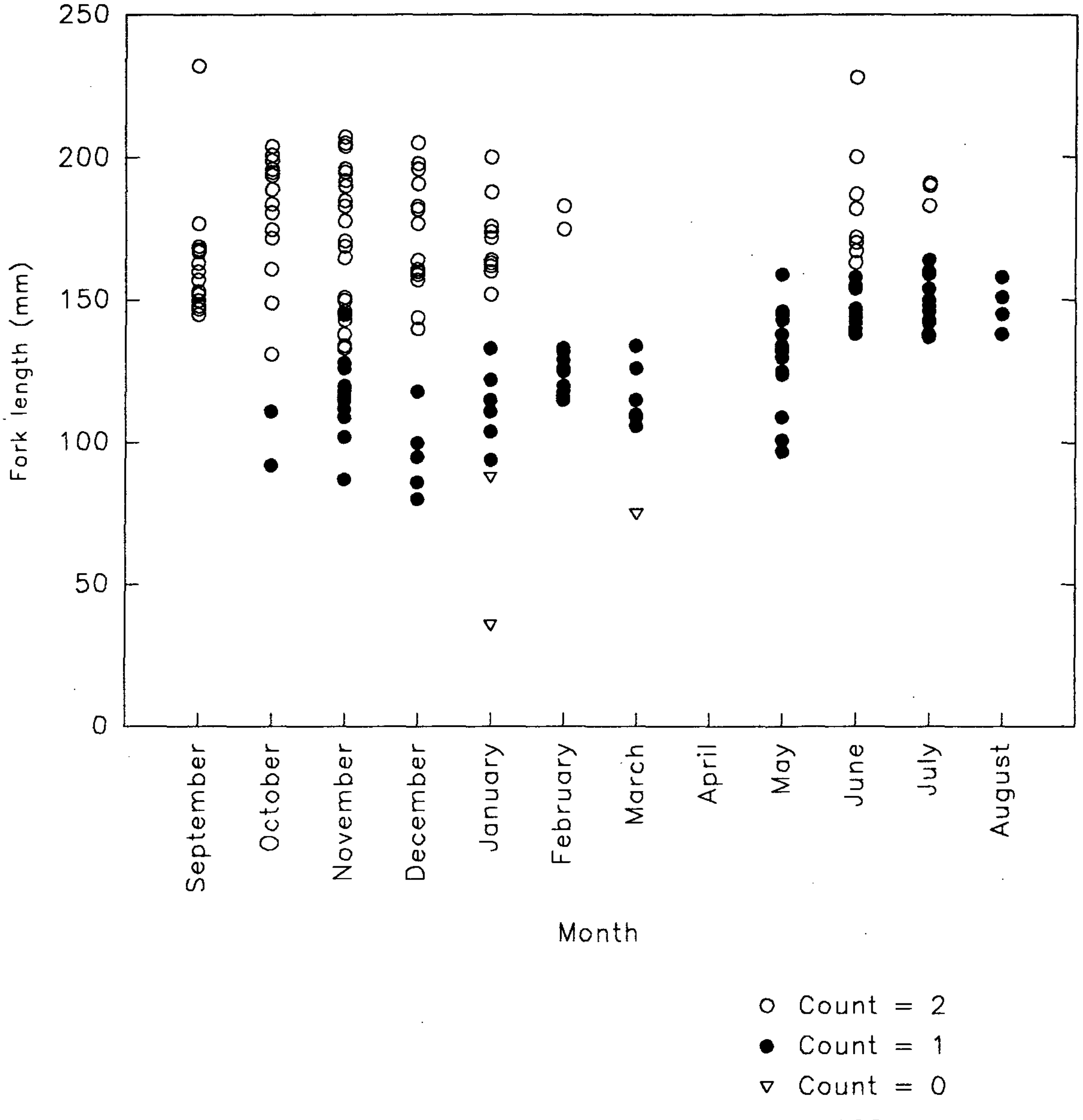


Figure 3.13: The relationship between fork length and sagittal otolith counts for *L. russelli* from estuaries for each month from September to August. Data for all years are combined. Clear triangles represent fish with count = 0, filled circles represent fish with count = 1, and clear circles represent fish with count = 2.

Figure 3.13:


250
200
150
100
50
0
Fork length (mm)
September
October
November
December
January
February
March
April
May
June
July
August
Month
○ Count = 2
● Count = 1
▽ Count = 0
n = 196

any larger *L. argentimaculatus* were caught and not reported. The largest *E. coioides* [387mm F.L. (Sheaves, 1992)], *E. malabaricus* [440mm F.L. (Sheaves, 1992)] and *L. argentimaculatus* [320mm S.L. (Blaber *et al.*, 1989)] previously reported from estuaries in tropical Australia are all smaller than the largest fish from estuaries in this study. The largest *L. russelli* [237mm F.L. (Sheaves, 1992)] previously reported from Australian estuaries was of similar size to the largest individual captured during the present study (232mm F.L.).

While many studies both in estuarine (Blaber, 1980; Blaber and Milton, 1990; Blaber *et al.*, 1989; Collette, 1983; Haines, 1979; Inger, 1955; Quinn and Kojis, 1985b; Robertson, 1988; Robertson and Duke, 1990a; Sheaves, 1992; Thollot *et al.*, 1990) and off-shore (Blaber *et al.*, 1990a; Blaber *et al.*, 1992; Blaber *et al.*, 1994; Coles *et al.*, 1993) habitats in the tropics have reported at least one of these species, sample sizes have generally been small and sizes rarely quoted. Where sizes are available for *L. argentimaculatus* and *L. russelli* (Blaber *et al.*, 1989) the same pattern as in the present study is seen; larger sizes from offshore locations than from estuaries.

There is some anecdotal and observational evidence (Results 3.3.1) of large *Epinephelus* occurring in estuaries in tropical north Queensland. However, the discrepancy between the maximum sizes trapped in estuaries and the maximum sizes able to be caught in the same traps in offshore waters suggests that if larger fish were common in estuaries they should have been captured. It appears that if large *Epinephelus* do occur in estuaries they probably constitute a minor part of the main estuary population only.

In this study validation of otolith check marks as annuli was successfully accomplished for each species using tetracycline marking techniques. Despite this the validation cannot be considered complete. Validation could only be achieved over a small size range, meaning that only a few of the check marks were validated. Furthermore, no validation was accomplished for more than a single year and the sample sizes of validated fish were small. As extensive validation is required to produce accurate estimates of age (Boehlert, 1985) the level of validation achieved here may be insufficient for definitive age studies. Nevertheless, the level of validation achieved does seem sufficient to establish the approximate maximum time the fish remain in estuaries and to establish that the maximum age in estuaries is considerably less than the maximum attained in offshore waters.

From the otolith data it seems that the maximum period spent in estuaries was about 5 years for *E. coioides*, 7 years for *E. malabaricus*, 8 years for *L. argentimaculatus* and 2 years for *L. russelli*. The exact period of residency of these species in estuaries is difficult to determine. The process of reading sectioned otoliths is subjective (Campana and Neilson, 1985; Thresher, 1988; Talbot and Doyle, 1992) particularly for young fish as it is difficult to determine exactly what period of time has elapsed prior to the deposition of the first annulus. While the counting of daily rings may overcome this, a second problem remains. The ageing of a fish from an estuary only tells the age of a fish that was still resident within the estuary. Unless fish can be sampled immediately on exiting an estuary and it can be demonstrated clearly that those fish have originated from an estuary, it is impossible to determine the range of ages at which fish actually migrate from estuaries.

For each of the four species, maximum ages determined for fish from estuaries were considerably younger than those for fish from offshore waters. Additionally, for each species the size at age determined from sectioned otoliths agreed well with the growth data from mark-recapture (Figs. 3.9,10,11,12). Furthermore, as the growth of fishes is expected to follow a curvilinear trajectory, with reduced growth at larger sizes (Pauly, 1984), the linear growth of all four species in estuaries, implied by the mark-recapture data, is consistent with rapid growth in the rising part of an asymptotic growth curve. This suggests that the fish from estuaries were sampled over a restricted section of the lower part of the growth curve only.

Taken together the size, age and growth data suggest strongly that not only were *E. coioides*, *E. malabaricus*, *L. argentimaculatus* and *L. russelli* from estuaries smaller than from offshore, but they were considerably younger. Apparently, estuary populations of all four species are comprised of small young individuals.

CHAPTER 4.0

REPRODUCTIVE STATUS OF *L.ARGENTIMACULATUS*, *L.RUSSELLI*, *E.COIOIDES* AND *E.MALABARICUS* IN ESTUARIES OF TROPICAL NORTH-EASTERN AUSTRALIA

4.1 INTRODUCTION

Recent reviews (Randall and Ben-Tuvia, 1983; Allen, 1985; Allen and Talbot, 1985; Randall, 1987; Randall *et al.*, 1990) list *L. argentimaculatus* and *L. russelli* as inhabitants of coastal and estuarine waters. These works suggest that within estuaries *L. argentimaculatus* occurs as juveniles and sub-adults while *L. russelli* is present as juveniles only. However, few studies of Indo-Pacific estuarine fishes have reported the reproductive status or life-history stages of these species. Where such classification has been carried out *L. russelli* populations have usually been classified as juveniles (Blaber, 1986; Blaber and Milton, 1990; Thollot *et al.*, 1990). While most studies that have recorded *L. argentimaculatus* from estuaries have reported the presence of juveniles, a number of studies have also reported the presence of adults (Shine *et al.*, 1973; Blaber, 1980; Blaber *et al.*, 1989; Blaber and Milton, 1990; Thollot *et al.*, 1990). While Blaber (1980) stated that "all reproductively immature fish were classed as juvenile" it is unclear how the assessment of adult status was made in other studies.

The life-history of two serranids in Indo-Pacific estuaries is even less clear. *E. coioides* and *E. malabaricus* are both known to inhabit coastal and estuarine waters (Randall and Ben-Tuvia, 1983; Randall *et al.*, 1990). Where life-history stage has been recorded, both juvenile and adult *E. coioides* [recorded as *E. suillus*] (Blaber and Milton, 1990) and *E. malabaricus* (Thollot *et al.*, 1990) have been reported. However, the criteria used for these determinations were not presented. Thus there is little direct evidence of the reproductive and life-history status of any of these lutjanids and serranids in Indo-Pacific estuaries.

In the present study the reproductive statuses of *E. coioides*, *E. malabaricus*, *L. argentimaculatus* and *L. russelli* within estuaries in tropical north-eastern Australia were determined. Assessment was made in terms of gonad maturity (macroscopic and histological) and relative size. Comparison was also made to the maturity and relative sizes of the gonads of fish from off-shore waters.

4.2 METHODS

4.2.1 Collection of samples

Between October 1990 and March 1994 specimens of *E. coioides*, *E. malabaricus*, *L. argentimaculatus* and *L. russelli* were collected to provide gonad samples. Data for each month from different years were pooled.

Estuary samples were collected from fish trap catches supplemented by line fishing and samples donated by anglers (see Chapter 3.2.1).

Immediately upon capture or receipt of donated samples, the fish were placed on ice where they were kept until dissected (less than 12 hours). Fish were weighed (total and cleaned weight) and measured (fork and standard length), and the gonads removed. The gonads were preserved in FAAC fixative [Formaldehyde 4%, Acetic Acid 5%, Calcium chloride 1.3%; (McCormick and Molony, 1992)]. The gonads were weighed within two days of preservation and returned to the fixative where they were stored until histological preparation was carried out. The proportion of cleaned body weight that each gonad represented was recorded.

Fish were initially classified as females, males or indeterminate sex from the macroscopic appearance of the gonads. Histological examination was carried out on all gonads greater than approximately 1mm diameter. Smaller gonads were not examined histologically as very small gonads were difficult to process and it was considered unlikely that they would be sexually active. In support of this premise, most of the smallest gonads sampled contained only oogonia or spermatogonia (precursors of oocytes or spermatocytes respectively) rather than more developed gonadial tissue.

A section from each of the anterior, central and posterior regions of one gonad from each fish was embedded in paraffin wax and transverse sections cut at 6 μm.

The sections were stained with haematoxylin and eosin, and mounted in DePeX. The sex of each gonad was determined by microscopic examination. For each female the stage of the most advanced oocyte recorded.

Ovarian tissue was staged according to the scheme described by West (1990) as follows:-

Stage 1 - Chromatin nucleolar stage,
Stage 2 - Perinucleolar stage,
Stage 3 - Yolk vesicle formation,
Stage 4 - Vitellogenic stage,
Stage 5 - Ripe.

For each male the most advanced stage of spermatic tissue present (i.e. spermatogonia, primary and secondary spermatocytes, spermatids, spermatozoa) was recorded.

4.3 RESULTS

It is assumed that for all four species the samples collected included fish of the largest sizes usually encountered in estuaries in the study area (see Chapter 3.3.1). It is likely that few fish larger than those collected were caught from estuaries and not reported as it was clear from the angler collections that most anglers tended to donate fish when they caught what they considered to be a large fish of that species. During the study, no *L. argentimaculatus* or *L. russelli* larger than those collected were reported from estuaries. Similarly, no *Epinephelus* spp. larger than those collected were reported to have been caught from estuaries. There were, however, occasional reports of sightings of very large *Epinephelus* spp. individuals from estuaries (see Chapter 3.3.1).

Few large *E. coioides*, *E. malabaricus* or *L. argentimaculatus* were collected from offshore. Large individuals of these species were not common components of anglers catches, and *E. coioides* and *E. malabaricus* larger than 1200mm are protected in Queensland waters. No attempt was made to specifically target large *Epinephelus* spp. from offshore. It was considered that, in terms of this project, obtaining gonads of large *Epinephelus* spp. was not important enough to justify killing comparatively rare, large individuals.

4.3.1 Fish from estuaries

Except for *L. russelli* in April, fish of each species were collected from estuaries for gonad examination in all months (Table 4.1) [in April no *L. russelli* large enough for sex determination were collected]. Compared to fish from offshore, the gonads of fish of all four species from estuaries were small relative to body weight (Table 4.2; Fig. 4.1).

Table 4.1: Summary of *Epinephelus* spp. and *Lutjanus* spp. collected for sex status determination.

		E. coioides	*E. malabaricus*	*L. argentimaculatus*	*L.russelli*
		Max Fl.(mm); [n]	Max Fl.(mm); [n]	Max Fl.(mm); [n]	Max Fl.(mm); [n]
January	Estuary	262 [16]	433 [27]	490 [25]	200 [23]
	Offshore	780 [3]	-	592 [5]	337 [1]
February	Estuary	384 [14]	544 [21]	475 [23]	183 [17]
	Offshore	915 [3]	-	-	382 [23]
March	Estuary	377 [11]	498 [12]	466 [39]	136 [9]
	Offshore	645 [1]	-	-	365 [20]
April	Estuary	387 [9]	545 [11]	411 [12]	-
	Offshore	436 [2]	-	-	445[31]
May	Estuary	405 [18]	525 [13]	422 [16]	159 [16]
	Offshore	-	687 [2]	636 [5]	362 [37]
June	Estuary	471 [10]	400 [13]	482 [29]	228 [23]
	Offshore	1085 [1]	-	615 [1]	296 [10]
July	Estuary	315 [8]	433 [12]	410 [17]	230 [27]
	Offshore	-	-	476 [2]	298 [23]
August	Estuary	333 [11]	595 [16]	288 [11]	158 [14]
	Offshore	-	-	-	360 [5]
September	Estuary	387 [18]	422 [18]	540 [31]	232 [18]
	Offshore	-	-	-	309 [27]
October	Estuary	394 [13]	594 [26]	541 [44]	204 [23]
	Offshore	-	-	654 [7]	405 [13]
November	Estuary	368 [17]	499 [20]	487 [28]	209 [38]
	Offshore	-	523 [3]	890 [4]	329 [28]
December	Estuary	420 [11]	562 [19]	450 [23]	205 [21]
	Offshore	-	-	528 [1]	410 [8]

For all species a number of fish had gonads too small (many gonads less than about 0.1g and most less than about 0.01g) to enable confident macroscopic sex determination (Table 4.2). Of the fish from estuaries that could be sexed, all *Epinephelus* spp. were identified as females, while for *Lutjanus* spp. both females and males were collected in similar numbers. Histological examination of the larger gonads from estuary samples supported the results of macroscopic examination (Table 4.3). All *Epinephelus* spp. with gonads sufficiently developed to allow sex determination were females, while for *Lutjanus* spp. both females and males were identified.

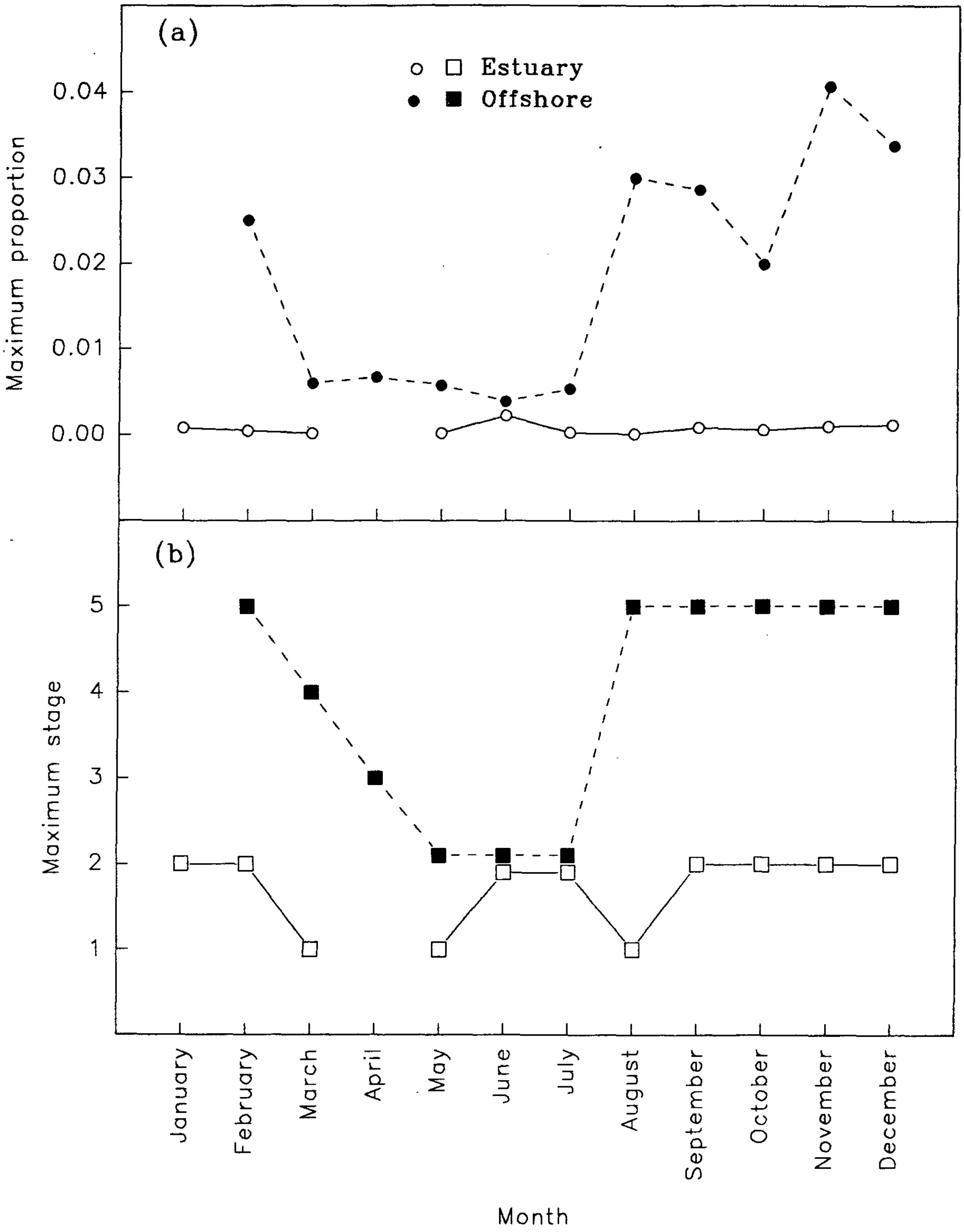
Table 4.2: Summary of sexes and gonad sizes of *Epinephelus* spp. and *Lutjanus* spp. determined by macroscopic examination.
The numbers of fish and the maximum proportion of body weight contributed by the female or male gonad is shown.
ind. = sex indeterminate

		Estuary		Offshore	
		n	Max. proportion of body weight ($*10^{-4}$)	n	Max. proportion of body weight ($*10^{-4}$)
E. coioides	♀	144	2.7	7	8.3
	♂	-	-	3	7.9
	ind.	12	-		-
E. malabaricus	♀	185	8.9	-	-
	♂	-	-	3	6.8
	ind.	23	-	2	5.0
L. argentimaculatus	♀	141	21.2	12	178.0
	♂	132	9.0	13	27.5
	ind.	25	-	-	-
L. russelli	♀	53	22.4	108	406.0
	♂	55	7.5	114	40.3
	ind.	121	-	4	-

The ovaries of all female fish from estuaries were thin (max. diameter: 7mm *L. argentimaculatus* [FL 540mm]; 3mm *L. russelli* [FL 228mm], *E. coioides* [FL 471mm], *E. malabaricus* [FL 595mm]) and less than 1/4 the length of the body

Figure 4.1: Changes in relative gonad size and stage over time for *L. russelli* from estuaries and offshore waters. Data presented are maxima of (a) gonad weight as a proportion of cleaned weight and (b) maximum gonad stage for each month.

Figure 4.1


(a)
○ □ Estuary
● ■ Offshore
Maximum proportion
0.00
0.01
0.02
0.03
0.04
(b)
Maximum stage
1
2
3
4
5
January
February
March
April
May
June
July
August
September
October
November
December
Month

cavity. No oocytes were visible to the naked eye. For all species, histological examination showed the most advanced oocytes to be chromatin nucleolar stage [stage 1] or perinucleolar stage [stage 2] (Fig. 4.2). Neither α or β stage atretic structures (Hunter and Macewicz, 1985; Hunter *et al.*, 1986; Kjesbu and Klungsoyr, 1991) were found in the ovaries of any fish. Thus on the basis of gonad size, and macroscopic and histological examination of ovaries, all female fish of all four species from estuaries were classified as reproductively immature.

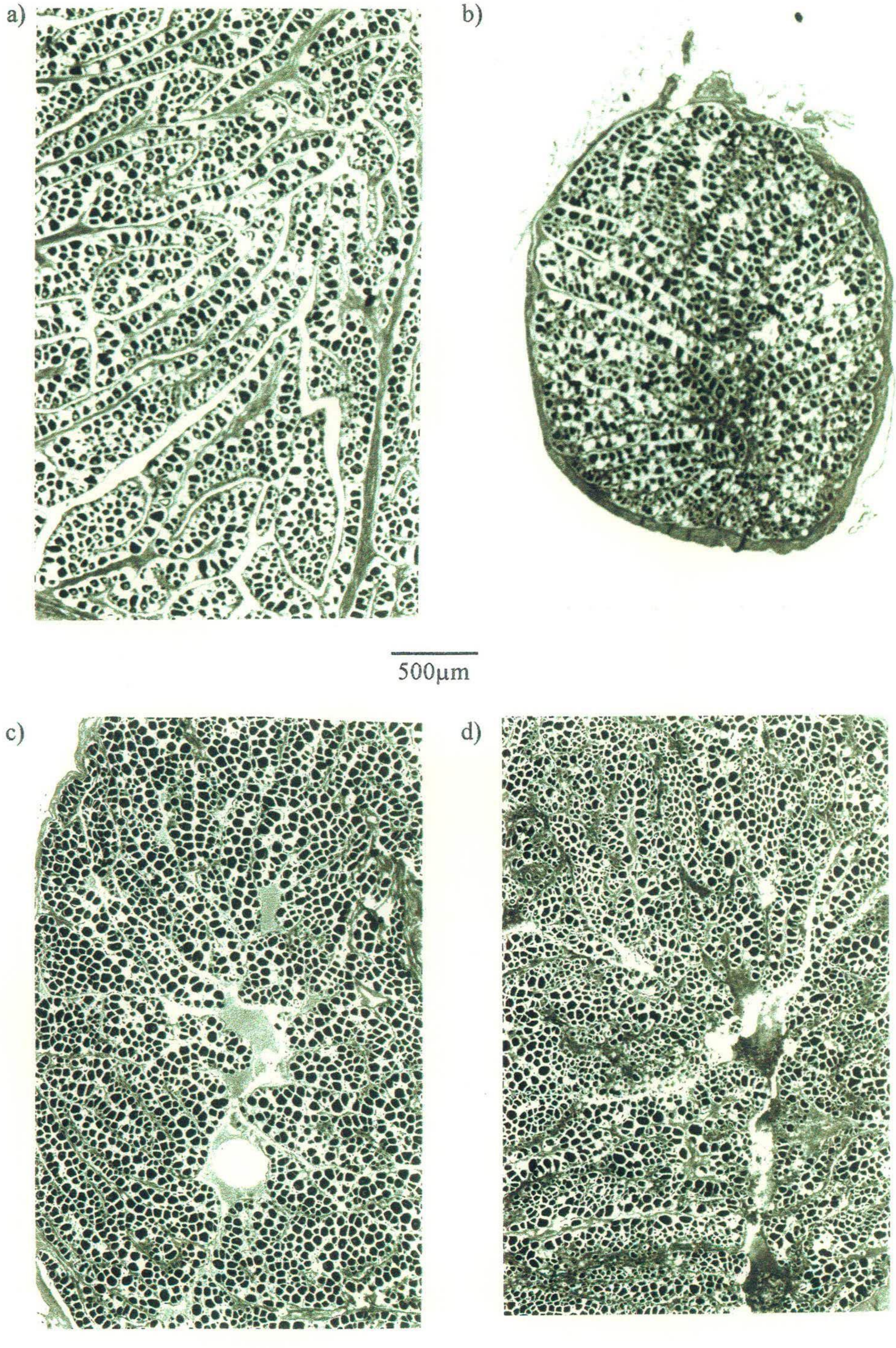
Table 4.3: Summary of histological examination of gonads of *Epinephelus* spp. and *Lutjanus* spp. collected from estuary and offshore waters.
ind. = indeterminate sex

		Estuary		Offshore	
		n	Maximum stage (♀)	n	Maximum stage (♀)
E. coioides	♀	149	2	7	2
	♂	0	-	3	-
	ind.	7	-	0	-
E. malabaricus	♀	199	2	2	2
	♂	0	-	3	-
	ind.	9	-	0	-
L. argentimaculatus	♀	159	2	12	5
	♂	134	-	13	-
	ind.	5	-	0	-
L. russelli	♀	93	2	108	5
	♂	108	-	116	-
	ind.	28	-	2	-

The testes of all male *L. argentimaculatus* and *L. russelli* from estuaries were firm, narrow and strap-like, and comprised only a small proportion of body weight (Table 4.2). The most advanced spermatic tissue present in most *L. argentimaculatus* and *L. russelli* testes from estuaries were primary and secondary spermatocytes. However, as well as containing large areas of immature spermatic tissue, the testes of a sample of 3 male *L. argentimaculatus* (432-541mm) collected from the mouth

Figure 4.2: Transverse sections of the ovaries of the largest female a) *L. argentimaculatus* [FL 540mm], b) *L. russelli* [FL 228mm], c) *E. coioides* [FL 471mm] and d) *E. malabaricus* [FL 595mm] collected from estuaries.

Figure 4.2:



of Ross Creek (Fig. 2.1) in October 1993 contained some spermatids and spermatozoa. In these fish, the spermatids and spermatozoa were confined to the proximal (posterior) parts of the testes. Four large female *L. argentimaculatus* sampled at the same time were all reproductively immature. This included the largest female *L. argentimaculatus* (540mm) obtained from estuaries during the study (Fig. 4.2). As the testes of these males were very small and the area of reproductive development limited it was assumed that they represented fish in early stages of reproductive maturity.

4.3.2 Fish from offshore

Limited samples of other species besides *L. russelli* were available from offshore waters (Table 4.1 & 4.2). However, for *L. argentimaculatus* and *L. russelli* of both sexes from offshore, the maximum proportion of body weight contributed by the gonad was an order of magnitude greater than for estuary fish (Table 4.2). Histologically, mature oocytes [ripe (stage 5) and/or vitellogenic (stage 4)] occurred in offshore samples of *L. argentimaculatus* from December and *L. russelli* from August, September, October, November, December, February and March (Fig. 4.1) [no females were collected in January]. During these months all individuals of these species sampled from estuaries were reproductively inactive. Small *L. russelli* (<210mm) collected from Cape Cleveland and Cleveland Bay were all reproductively immature. Throughout the year, the maximum stage and maximum proportion of body weight of ovaries of *L. russelli* from estuaries varied little (Fig. 4.1), and during the reproductively active period (August to about February) were consistently smaller than those from offshore.

4.4 DISCUSSION

All female *L. argentimaculatus*, *L. russelli*, *E. coioides* and *E. malabaricus* collected from estuaries were found to be reproductively immature. No reproductively active~~ly~~ individuals were found despite the examination of gonads from the largest fish available throughout the year. This included periods when reproductively active females were present in samples from offshore. As the juvenile period lasts until maturation of the first gametes (Balon, 1984), all the female fish sampled from estuaries can be considered juveniles. Although a sample of three male *L. argentimaculatus* from one estuary showed some degree of reproductive maturity, it was assumed that these were fish in the early stages of reproductive development only. The testes of these males were very small, and mature spermatic tissue was limited to proximal regions of the gonads. Furthermore, four large females sampled at the same time showed no sign of reproductive development, suggesting that the spermatic development was not related to spawning at that time. It seems likely that these were a group of large fish preparing to migrate from the estuary. Of the *E. coioides* and *E. malabaricus* possessing gonads advanced enough for sex determination, all individuals of both species were found to be females. As *Epinephelus* are recognised as protogynous hermaphrodites (Bannerot *et al.*, 1987; Randall *et al.*, 1990) the presence of only non-reproductive females with poorly developed gonads suggests that the populations consisted of pre-reproductive individuals. Taken together, the data presented here suggest strongly that *L. argentimaculatus*, *L. russelli*, *E. coioides* and *E. malabaricus* do not become reproductively active within the estuaries studied. This implies that the estuarine populations of these species consist of juveniles. In South Africa a large number of fish species use estuaries as juvenile habitats and return to the sea before attaining sexual maturity (Whitfield, 1990). For example, for six species of carangids only juveniles and sub-adults utilise estuaries (Blaber and Cyrus, 1983), and three species of *Gerres* move offshore from estuarine nursery grounds before spawning (Cyrus and Blaber, 1984). Similarly, in Florida juvenile *Sciaenops ocellatus* are found in estuaries but adults spawn in nearshore waters (Peters and McMichael, 1987) and while juvenile *Lutjanus griseus* are found in mangrove areas adults spawn on offshore reefs (Starck and Schroeder, 1970).

Most previous studies have classified *L. russelli* from the tropical Indo-Pacific estuaries as juveniles (Allen, 1985; Allen and Talbot, 1985; Blaber *et al.*, 1985; Blaber, 1986; Blaber and Milton, 1990; Thollot *et al.*, 1990). The results of the present study agree with this. Where life history stages have been recorded, previous studies (Shine *et al.*, 1973; Blaber, 1980; Blaber *et al.*, 1985; Blaber and Milton, 1990; Thollot *et al.*, 1990) have classified estuary populations of *E. coioides*, *E. malabaricus* and *L. argentimaculatus* in the Indo-Pacific as being comprised of both juveniles and adults. The present study does not support this. Despite extensive collections, no female fish of any of the four species collected from estuaries possessed mature gonads. This, together with the fact that in this study the fish collected from estuaries were smaller and younger than those from offshore (Chapter 3), indicates that the populations of these species within the estuaries studied probably contain few adults, if any. If large *Epinephelus* do occur in estuaries, as anecdotal and observational evidence suggests (Chapter 3), the lack of large fish during this study suggests that they probably constitute a minor portion of the estuary population only. These fish may represent a small number of individuals that have returned to the estuaries after spawning offshore or that have failed to migrate out of the estuaries.

With one exception, previous studies did not present the criteria used to classify life history stage. If reproductive status was not evaluated it seems likely that these judgements were based on size. Given the large sizes attained by *L. argentimaculatus*, *E. coioides* and *E. malabaricus* in estuaries (in the present study fish ≥ 500mm of all three species were collected), such judgements would be understandable. It is possible that more detailed investigations of the reproductive status of these three species in the populations previously studied may have revealed that the populations were in fact composed of pre-reproductive fish. Indeed in the studies where *E. coioides, E. malabaricus* and *L. argentimaculatus* from estuaries were recorded as adults the maximum sizes reported were considerably smaller than the largest fish of each species collected during the present study. Blaber (1980) sampled one or two *L. argentimaculatus*, 370mm in length, from Trinity Inlet in north-eastern Australia and classified it/them as reproductively mature. As Trinity Inlet is some 200km north of the estuaries studied here, this may represent a spatial difference in life-history. However, *L. argentimaculatus* from estuaries throughout north-eastern tropical Australia, frequently posses a large fat body closely associated with, and often surrounding the gonads (pers. obs.). On cursory examination these fat bodies may be mistaken for mature gonadial tissue. Indeed, during the present study many anglers supplied fish with what they believed to be mature gonads - invariably these proved to be fat bodies.

The pre-reproductive status of *L. argentimaculatus, L. russelli, E. coioides* and

E. malabaricus in the estuaries studied implies that they move offshore to spawn. The lack of large, old (Chapter 3), reproductively active individuals (with the possible exception of a few *Epinephelus*) suggests that adults of these species generally do not return to estuaries. In South Africa, three species of *Gerres* that leave estuarine juvenile habiats to spawn offshore usually do not return to estuaries (Cyrus and Blaber, 1984). This is the case for most fish in South Africa that spawn offshore but use estuaries as juvenile habitats (Day *et al.*, 1981). Previous workers have asserted that *L. argentimaculatus* (Johannes, 1978; Thollot *et al.*, 1990), *E. malabaricus* and *L. russelli* (Thollot *et al.*, 1990) migrate offshore. Other studies have suggested that *L. argentimaculatus* and *L. russelli* (Allen, 1985; Allen and Talbot, 1985; Randall *et al.*, 1990) occur as juveniles in estuaries and adults in offshore habitats, inferring migration away from the estuaries. Despite the necessity of a spawning migration for these fish few offshore movements have been documented for any of these species. One *L. argentimaculatus* tagged during a sportfish tagging program moved from an estuary on Hinchinbrook Island (Fig. 2.1) to a reef some 80km to the north-east (ANSA, 1991). An *E. malabaricus* tagged during the present study (length at release 480mm FL) also moved offshore. It was tagged in Barramundi Creek (Fig. 2.1) and captured by an angler 17 months later on Lodestone Reef, some 75km to the north. The paucity of direct evidence of movement from estuaries to offshore habitats needs to be rectified. It may be that, while recapture rates are reasonably high within the estuary where the fish are tagged, when tagged fish move out of estuaries they mix with fish from other estuaries and spread out over large areas of offshore reefal water resulting in low probabilities of recapture. If so a substantial tagging and recapture effort would be required to demonstrate movement offshore.

Alternative approaches to demonstrating migration away from estuaries exist. Estuarine and offshore waters differ in physical variables, such as temperature regimes. Such changes may be recorded in otolith microchemistry (Radtke *et al.*, 1990). Thus an alternate approach to demonstrate offshore migration from estuaries may be to investigate changes in microchemistry across the otoliths of these species. Such changes could be compared to the chemical profiles of the otoliths of congenerics known to complete their life cycles in offshore waters. Microchemical techniques have produced promising results for *Mugil cephalus* from New Zealand

(Meyer-Rochow *et al.*, 1992) and *Morone saxatilis* from the Atlantic coast of the United States (Coutant and Chen, 1993). However, in order to demonstrate that the particular chemical changes observed are related to movement between estuarine and offshore habitats, considerable validation would be necessary (Kalish, 1989).

Estuaries are not the only habitats where juvenile *L. russelli* and *L. argentimaculatus* are found. During the present study small, reproductively immature *L. russelli* were sampled from coastal embayments and headlands (Chapter 2), and have been reported in low numbers from coastal seagrass beds (Coles *et al.*, 1993; S. Kramer, pers. comm.). Small numbers of juvenile *L. russelli* and *L. argentimaculatus* were reported from coastal seagrass beds in north-western Australia (Blaber *et al.*, 1992). Other studies of coastal seagrass beds (Robertson and Duke, 1987) and seagrass beds further offshore (G. Wilson, pers. comm.) in tropical northern Australia failed to sample any *L. russelli* or *L. argentimaculatus*. As juvenile *L. argentimaculatus* and *L. russelli* have been reported from offshore habitats only sporadically and in low numbers, the importance of these areas as juvenile habitats is unclear. It is uncertain if juvenile *E. coioides* or *E. malabaricus* occur in habitats other than estuaries. Studies that have reported these species from coastal or reef habitats (e.g. Blaber *et al.*, 1994) have not clearly identified the life-history stages present. Notwithstanding the potential for alternative habitats to provide nursery grounds for these four species, the presence of large numbers of functionally juvenile *E. coioides*, *E. malabaricus* and *L. russelli* in trap catches during this study (Chapter 6 & 7), together with the fact that all four species are common components of estuary angling catches (pers. obs), suggests that these species are common in estuaries of north-eastern Australia. This implies that estuaries are important nursery areas for these species.

The presence of a shallow-water, estuarine, juvenile habitat spatially distinct from that of the adults, indicates an offshore ontogenetic movement and the possession of a tripartite life cycle ~ planktonic larvae, estuarine juvenile and offshore adult. How the possession of such a life-history might benefit these species can only be speculated upon. It has been suggested that turbid conditions reduce the effectiveness of large predators so reducing predation on juvenile fishes (Blaber and Blaber, 1980; Cyrus and Blaber, 1987). This explanation seems to be self contradictory in the case of the four species considered here. While it is plausible that turbid conditions may reduce predation on small juveniles, large juveniles should be negatively affected by high turbidity. These species are themselves predators on smaller fish (Parish, 1987). As such their feeding should be hampered by turbid conditions. It may be that estuarine habitats provide enhanced feeding opportunities for fish (Chong *et al.*, 1990). Structurally complex habitats of tropical estuaries contain more fish than less complex areas (Blaber and Milton, 1990; Sheaves, 1992), and contain high densities of lutjanids and serranids (Sheaves, 1992; Chapter 5). These habitats may be beneficial for predators such as *E. coioides*, *E. malabaricus*, *L. argentimaculatus* and *L. russelli* by providing high densities of prey, as well as appropriate sites from which to ambush those prey.

CHAPTER 5.

HABITAT PREFERENCES OF *L. RUSSELLI* AND *E. COIOIDES* IN ALLIGATOR CREEK

5.1 INTRODUCTION

Differences in the distribution and abundance of estuarine fishes between habitats are well documented (Thayer *et al.*, 1987; Robertson and Duke, 1990a; Morton, 1990). Habitat type influences the relative abundances of particular species in many ways. For example, habitats with high levels of structural heterogeneity can provide both ambush sites for predators and refuges for prey fishes (Weinstein, 1979; Boesch and Turner, 1984; Orth *et al.*, 1984; Blaber, 1986; Felley and Felley, 1986; Rozas and Odum, 1988; Laprise and Blaber, 1992). On the other hand, areas of clear bottom may provide forage areas for fish which feed on benthic invertebrates.

Epinephelus suillus (probably *E. coioides*), *Lutjanus argentimaculatus* and *Lutjanus russelli* were captured from a range of habitats in the Embley estuary on the eastern side of Cape York in tropical Australia (Blaber *et al.*, 1989). However, due to the use of different sampling methodologies in each habitat no comparison of the distribution and abundance of these fishes between habitats was possible. The difficulties of sampling the range of habitats available in estuaries - from areas of clear bottom to structurally complex habitats composed of mangrove prop roots and fallen timber (snags) - has forced most workers to utilise different sampling methods in different habitats. Using fish traps, Sheaves (1992) compared snaggy banks and clear banks (i.e. free of snags), and found that both *E. coioides* and *Epinephelus malabaricus* were more abundant along snaggy banks. However, in this study two major habitat types - mud banks and mid-channels - were not sampled. Furthermore, too few lutjanids were trapped to allow meaningful comparisons to be made. Considering the limited movement of *E. coioides*, *E. malabaricus* and *L. russelli* in estuaries in tropical Australia (Sheaves, 1993) [see Appendix II], a knowledge of the extent to which fish occupy the range of habitats available to them is important in determining overall patterns of distribution and abundance.

In this study fish traps were used to compare the distributions of *E. coioides* and *L. russelli* between four major habitat types (snaggy banks, clear banks, mud banks and mid channels) recognisable in sub-tidal areas of Alligator Creek, an estuary in tropical north-eastern Australia. Differences in catches between day and night, and between samples collected at four different times of the year are also examined. By using a mesh size (12.5mm) smaller than that used in previous work (42mm) this study considers a wider size range than previous fish trap studies (Sheaves, 1992).

5.2 METHODS

5.2.1 Study area

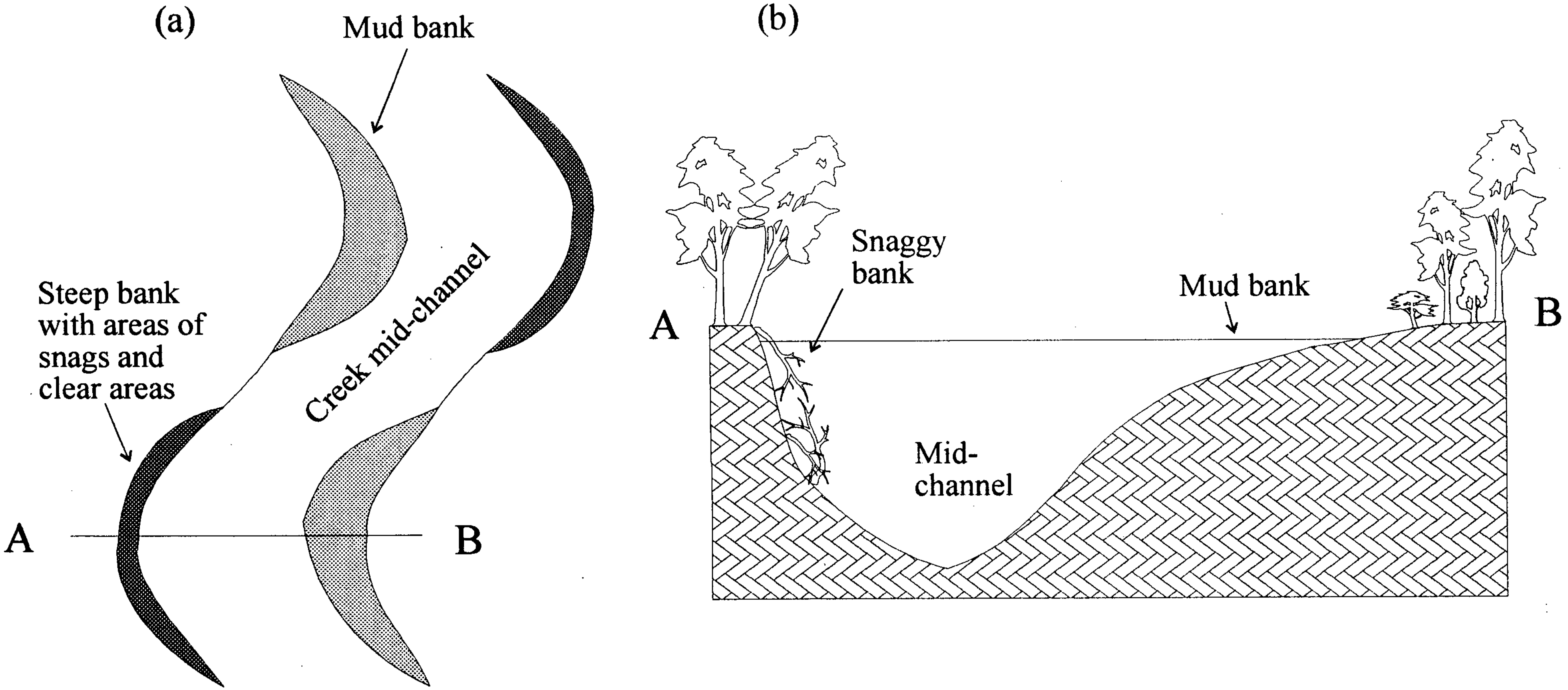
Sampling was carried out between December 1992 and August 1993 at Alligator Creek, a small estuary lying on the north-eastern coast of Queensland, in tropical Australia [approx. 19° 21'S, 146° 57'E] (Fig. 2.1). More detailed descriptions of Alligator Creek can be found in Sheaves (1992) and Chapter 6.2.1. Due to the shallow nature of the upper reaches of the estuary sampling was restricted to the seaward 10km.

5.2.2 Habitat definition

Within the sub-tidal areas of Alligator Creek four main habitat types could be identified (Fig. 5.1). Along the banks of the creek in areas of rapid stream flow (such as the outside radii of bends) the banks shelved steeply and reasonably deep water existed adjacent to such banks. Here two types of habitats occurred. Where the banks were forested, fallen timbers and mangrove prop roots formed areas of high structural heterogeneity (snaggy banks = habitat 1). Where the banks were not timbered, areas devoid of structural heterogeneity in the form of snags occurred (clear banks = habitat 2). Along banks in areas of slow stream flow, mud banks of shallow slope occurred (mud banks = habitat 3). These were also devoid of structural heterogeneity created by snags. Away from the banks towards the centre of the stream a central deep water channel existed (mid channel = habitat 4). At spring low tides the water depth in this channel varied between about 1.2m in upstream areas to about 5m near the creek mouth.

Figure 5.1: Diagrammatic representations of four habitat types in Alligator Creek a) the relative positions of snaggy and clear banks [outsides of bends], mud banks [insides of bends], and mid-channel habitats, b) cross section showing a steep bank [in this case with snags], mid-channel and mud bank habitats.

Figure 5.1


(a)
Mud bank
Steep bank with areas of snags and clear areas
Creek mid-channel
A
B
(b)
Snaggy bank
Mud bank
A
B
Mid-channel

5.2.3 Trap design and sampling protocol

The traps used in this study were modified Antillean Z-traps (Fig. 6.3). They were approximately 1800mm long, 1100mm wide and 600mm high with a plan area of approximately 1.53m^2, and a volume of approx. 0.92m^3. These traps had 12.5mm square wire mesh and had two straight, tapered entrance funnels, one set into each concave angle of the "Z". At each set all traps were baited with approximately 500g of Western Australian Blue Pilchards (*Sardinops neopilchardus*). All traps were oriented in such a way that one entrance faced down-current and the other up-current regardless of the direction of tidal flow. The traps, the trapping protocol and the baiting method are described in more detail in Chapter 6.2.2 and Sheaves (1994).

5.2.4 Sampling design

Sampling was carried out on four occasions three lunar months apart [01,02,03 December 1992; 01,02,03 March 1993; 28,29,30 May 1993; 23,24,25 August 1993]. Trapping was carried out over 3 consecutive days. In each of the months sampling occurred on the day of the first lunar quarter and one day either side. Thus sampling was carried out over the same part of each third lunar cycle. As sampling occurred only during one year with only one arbitrarily selected sampling period every three months, these samples cannot be seen as truly representing four seasons.

Within the lower 10km of the creek 300m long sites which contained all four habitats were identified. For each trap set a different site was selected at random. Within each site eight traps were set, two in randomly selected positions within each habitat type. In the case of the snaggy bank habitat, traps were set as close as possible to the largest individual snag within a particular area of habitat. For clear banks and mud banks traps were set in the centre of the area of the particular habitat type (at least 50m from any of the other bank habitat types). Traps in mid channel were set in the deepest part of the creek channel as judged from echo sounder readings. To ensure that all traps were set no less than 50m apart, where necessary alternate locations were chosen for the mid channel traps.

Traps were censused in the same order in which they were set to ensure approximately equal soak times. On each day setting and censusing traps was carried out twice, once between 0630 and 0730 hrs, and again between 1700 and 1800 hrs. The traps were relocated for each set. Thus three sites (each 300m long) were sampled on three seperate days and nights every three months. However, as daylight was necessary to enable safe navigation and the locating of trap sites, the day soak was only about 10.5 hrs while the night soak was about 13.5 hrs. Thus the factor of day/night was confounded with soaktime.

5.2.5 Data analysis

Both *E. coioides,* and *L. russelli,* were caught in sufficient numbers to allow useful analysis. The data for these species contained many zeros (> 60% for both species). As a result the data were first analysed on the basis of presence/absence using logistic regression (Appendix I.1) and backwards elimination (Collett, 1991). The data analysed were the proportion of trap soaks containing fish out of the six traps in each habitat/time/trip combination (4 habitats; 2 times; 4 trips). These analyses allowed comparison of the probability of trapping an individual in a particular habitat/time/trip combination. For factor levels where sufficient non-zero data were available the quantitative component of the data was then examined using factorial analyses of variance. In these analyses the mean numbers were compared between treatment levels for treatment combinations in which fish were trapped. For *E. coioides* sufficient data ~~was~~ were available for night samples in all habitats over the first three sampling trips. The factors habitats (4 levels) and trips (3 levels) were orthogonal and fixed. For *L. russelli* data for both day and night over all four trips were analysed for the snag habitat. The factors time (2 levels) and trips (4 levels) were orthogonal and fixed.

5.3 RESULTS

During the study 48 *E. coioides*, 8 *E. malabaricus* and 104 *L. russelli* were trapped (Table 5.1).

Table 5.1: Summary of catches of *E. coioides, E. malabaricus* and *L. russelli* from four habitats in Alligator Creek. Data are total numbers caught in each habitat/time combination.

		E. coioides	*E. malabaricus*	*L. russelli*
Snag	Day	5	2	53
	Night	10	2	33
Clear bank	Day	2	1	1
	Night	10	3	8
Mud bank	Day	3	0	0
	Night	7	0	5
Mid channel	Day	1	0	0
	Night	10	0	4

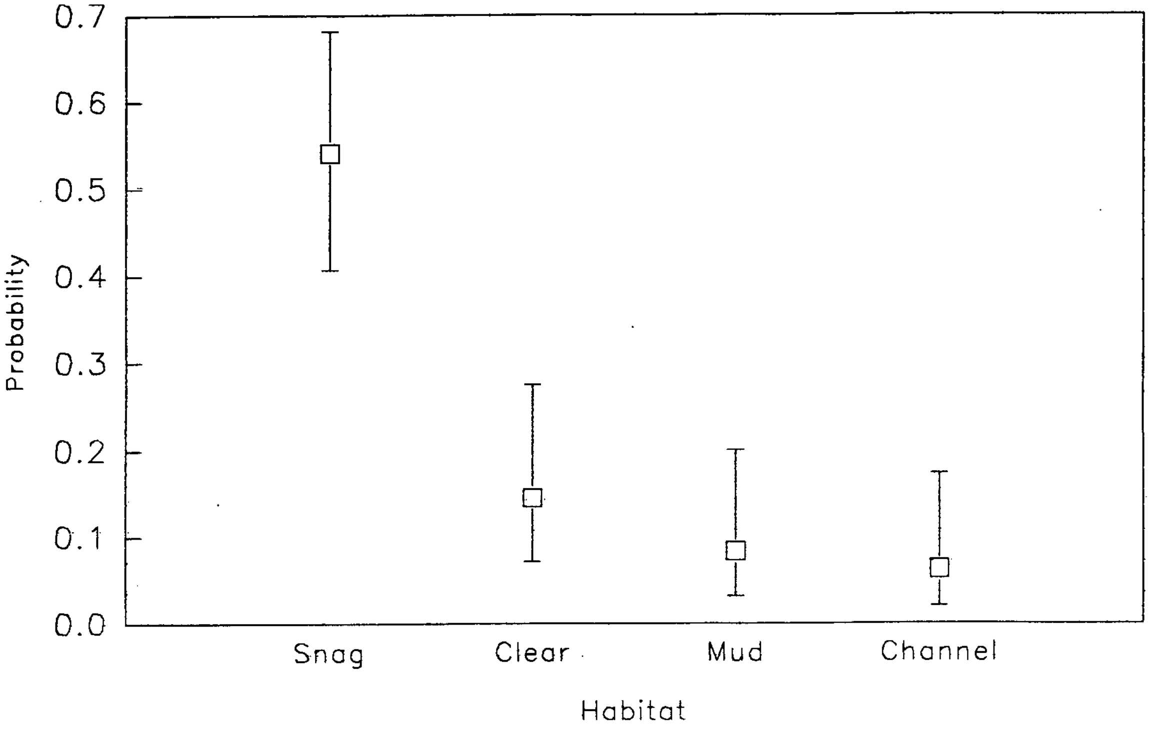
Too few *E. malabaricus* were trapped to allow statistical analysis. However, it is notable that all the *E. malabaricus* trapped came either from snag or clear bank habitats.

None of the factors (habitat/time/trip) were found to interact in the logistic regression analysis of the presence/absence data for either *E. coioides* or *L. russelli* (Table 5.2). Both habitat and time were independently important for *L. russelli* while both time and trip were independently important for *E. coioides*. *L. russelli* demonstrated a strong habitat specific pattern (Fig. 5.2a). The probability of trapping *L. russelli* was substantially greater in snags than in the other three habitats where the probability differed little. While the probability of capture of *E. coioides* was greater from snags than from the other habitats (Fig. 5.2b) the difference was not great and the factor habitat was not important in the logistic regression model. There were differences between day and night catches for both species. In each case the probability of capture was greater at night than in the day

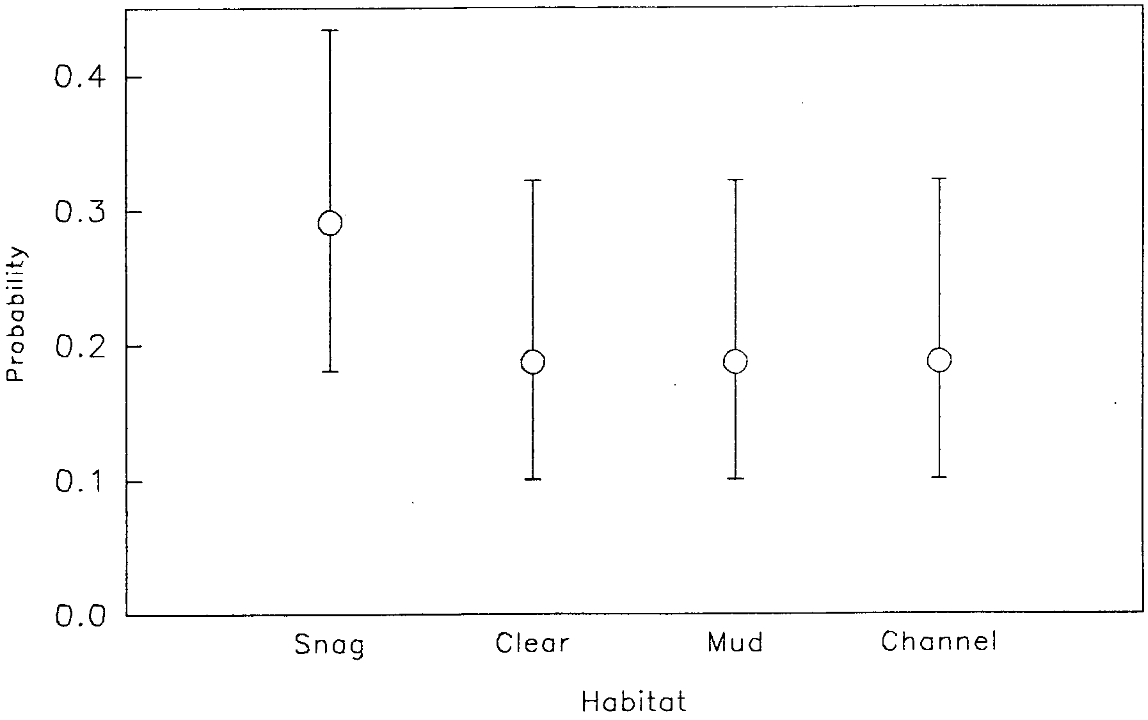
Figure 5.2: The probability of capture of a) *L. russelli* and b) *E. coioides* in four habitats of Alligator Creek. Error bars are 95% confidence limits.

Figure 5.2

(a) *L. russelli*



(b) *E. coioides*



(Fig. 5.3), although this effect was much stronger for *E. coioides*. There was a difference between trips for *E. coioides* resulting from a low probability of capture during the fourth trip.

Table 5.2: Analysis of deviance for the logistic regression of the presence/absence of (a) *E. coioides* and (b) *L. russelli* in trap catches from four habitats in Alligator Creek. The three left hand columns show the model fitted, and the associated deviance and degrees of freedom. The three right hand columns show the codes for the current model and the more complex model being compared together with the deviance of the comparison (Δ deviance), the degrees of freedom for the comparison (Δ d.f.), and the associated probability. The final model is the model from which no term can be removed without a significant increase in deviance (see Appendix I).

"0" is the code for the full 3-way model (deviance = 0, d.f. = 0)
signifies the final model

(a)

Model	Deviance	d.f.	Δ Deviance	Δ d.f.	Probability
A) habitat*time + habitat*trip + time*trip	7.66	9	A - 0 7.66	9	0.5687
B) habitat + time + trip	24.02	24	B - A 16.36	15	0.3585
C) habitat + time	33.41	27	C - B 9.39	3	0.0245
D) habitat + trip	39.07	25	D - B 15.05	1	0.0001
E) time + trip ##	**26.56**	**27**	**E - B 2.54**	**3**	**0.4681**
F) time	35.81	30	F - E 9.25	3	0.0262
G) trip	41.38	28	G - E 14.82	1	0.0001

why are these bold

(b)

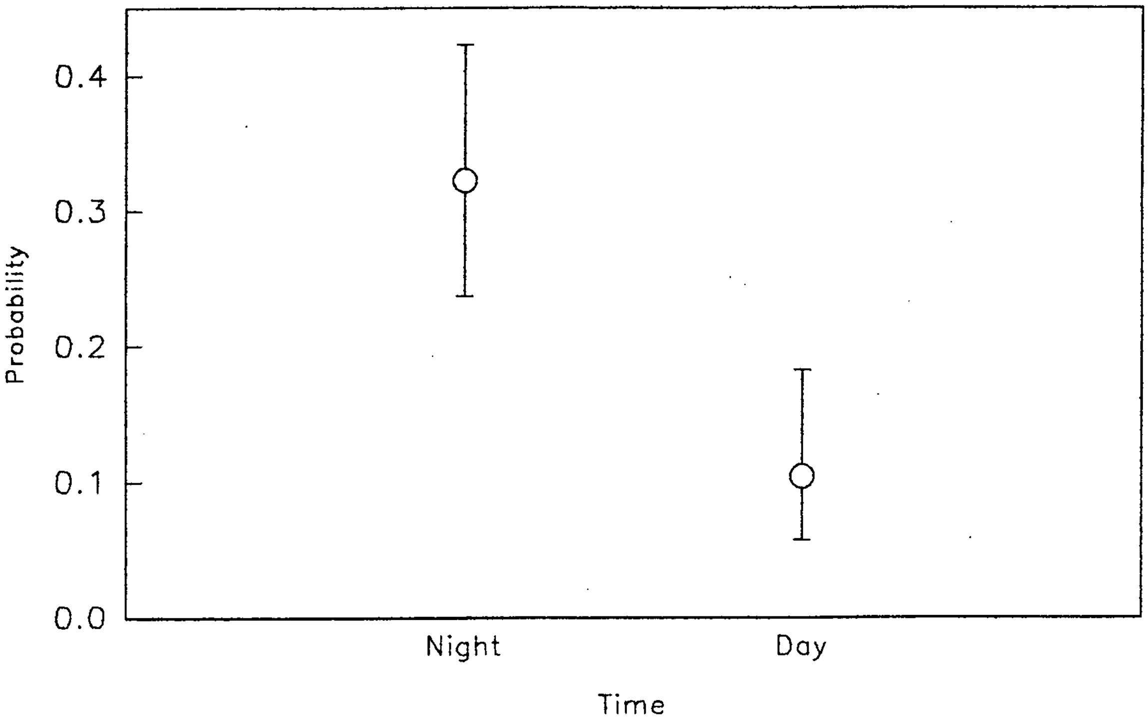
Model	Deviance	d.f.	Δ Deviance	Δ d.f.	Probability
A) habitat*time + habitat*trip + time*trip	2.52	9	A - 0 2.52	9	0.9803
B) habitat + time +trip	23.07	24	B - A 20.55	15	0.1518
C) habitat + time ##	**29.29**	**27**	**C - B 6.22**	**3**	**0.1014**
D) habitat + trip	27.42	25	D - B 4.35	1	0.0370
E) time + trip	66.02	27	E - B 42.95	3	>0.00001
F) habitat	33.45	28	F - C 4.16	1	0.0414
G) time	70.70	30	G - C 41.41	3	>0.00001

bold.??

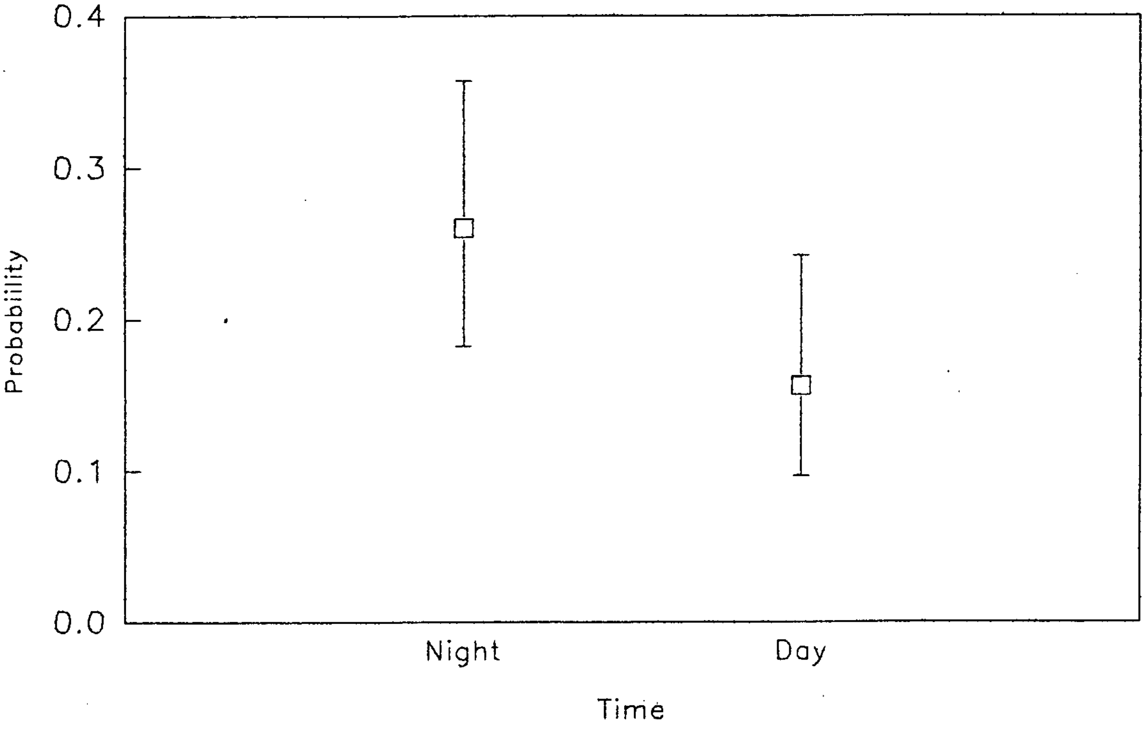
Figure 5.3: The probability of capture of a) *E. coioides* and b) *L. russelli* in Alligator Creek during day and night trap soaks. Error bars are 95% confidence limits.

Figure 5.3

(a) *E. coioides*



(b) *L. russelli*



For both *E. coioides* and *L. russelli* no significant differences were found in the catch per trap (for traps containing fish) between any of the factor levels for which analysis of variance could be used (Table 5.3).

Table 5.3: Analysis of variance of the numbers of (a) *E. coioides* and (b) *L. russelli* in trap catches from Alligator Creek for comparisons for which enough non-zero replicates were available for meaningful analysis. For *E. coioides* data from night samples in all habitats over the first three sampling trips were analysed, for *L. russelli* data for both day and night over the four trips were analysed for the snag habitat. Type I sums of squares are presented.

(a)

Source of variation	Sum of squares	d.f.	Mean square	F-ratio	Probability
Habitat	0.86349	3	0.28783	1.057	0.3964
Trip	0.45634	2	0.22817	0.838	0.4518
Habitat X Trip	1.26350	6	0.21058	0.774	0.6027
Residual	4.08333	15	0.27222		

(b)

Source of variation	Sum of squares	d.f.	Mean square	F-ratio	Probability
Time	0.15054	1	0.15054	1.648	0.2155
Trip	0.18754	3	0.06252	0.684	0.5731
Time X Trip	0.15324	3	0.05108	0.559	0.6487
Residual	1.64390	18	0.09133		

5.4 DISCUSSION

The lutjanid, *L. russelli* exhibited striking habitat specific differences in the probability of capture, with the probability of occurrence in catches from the structurally complex snag habitat over 3 times that for any other habitat type. This is consistent with the behaviour of snappers of the genus *Lutjanus* on reefs where they are frequently found associated with structures such as caves and ledges (Allen and Talbot, 1985). Two lutjanids, *Lutjanus apodus* and *Ocyurus chrysurus*, are common in mangrove prop-root habitats in Puerto Rico (Rooker and Dennis 1991), and in Florida *Lutjanus griseus* is common in structurally complex mangrove prop-root (Thayer *et al.*, 1987) and seagrass habitats (Chester and Thayer, 1990).

In many previous studies of estuaries, areas of high structural complexity have been found to contain greater numbers of fish of many species than do other habitats (Felley and Felley, 1986; Thayer *et al.*, 1987; Robertson and Duke, 1987; Blaber and Milton, 1990; Sheaves, 1992). Apparently snags in estuaries have similar habitat values to highly heterogeneous reef habitats, probably providing refuges form predators and sites from which prey can be ambushed. In a previous study of Alligator Creek the numbers *E. coioides* were found to be higher in snags than along clear banks (Sheaves, 1992). Although a similar pattern was seen in the probability of capture in the present study the effect was not substantial. While the sample size of *E. coioides* was small (48) the lack of any clear habitat differences suggests no strong habitat specificity such as that seen for *L. russelli*. While few (8) *E. malabaricus* were trapped, all were captured from either snaggy or clear banks. Both these habitat types consist of steeply shelving banks (in contrast to the shallowly sloping mud banks). So, despite this small sample size, some degree of habitat preference seems likely.

The factor of time (i.e. day/night) is inextricably confounded with the length of trap soak. However, given the magnitude of the differences - particularly for *E. coioides* - (Fig. 5.3a), it seems likely that the observed contrasts probably reflect higher catches at night than in the day, rather than simply being a product of variations in soak time. Given that sampling occurred in all the major habitats in Alligator Creek and that *E. coioides* and *L. russelli* tend to remain in a small area for extended periods (Sheaves, 1993) [see Appendix II], it is unlikely that these differences relate to changes in habitat use. Rather, it seems likely that such day/night differences reflect changes in the behaviour of the fishes between day and night, leading to variations in vulnerability to capture.

CHAPTER 6.

SPATIAL AND TEMPORAL PATTERNS IN THE DISTRIBUTION, ABUNDANCE AND SIZE OF *L. RUSSELLI* IN THREE ESTUARIES IN TROPICAL AUSTRALIA

6.1 INTRODUCTION

There are few cases where the patterns of spatial and temporal abundance and size of a tropical estuarine fish have been studied in detail. The author knows of no case where this has been done for any lutjanid.

Although *Lutjanus russelli* has been reported from most of the estuaries in tropical Australia that have been studied, it has usually been captured in low numbers only. The low catches probably reflect the difficulties of sampling in the structurally complex habitats preferred by *L. russelli* (Sheaves, 1992; Chapter 5) using the various netting methods usually employed in estuaries. Sheaves (1994) developed a fish trap design that produced high catch rates of *L. russelli*. For fish that are amenable to trapping, fish traps have a number of desirable features. For instance, they can be employed in almost any type of habitat and can be employed at a number of locations simultaneously. Thus fish traps lend themselves to studies of distribution and abundance.

An ability to cope with large-scale physical variability is an obvious prerequisite for fish inhabiting tropical estuaries. The north-eastern coasts of tropical Australia are characterised by highly seasonal rainfall patterns (Oliver, 1978; Bonell 1983). Rainfall is largely confined to the summer monsoon season (Pringle, 1986) with little rainfall during the rest of the year. Rainfall is often very heavy causing considerable run-off ~ in excess of 1 000mm annually in many areas (Bonell, 1983). Such large volumes of freshwater run-off entering estuary systems causes substantial changes in salinity (Wolanski *et al.*, 1992), and may result in considerable changes in temperature. Physical variability on such a large-scale clearly has the potential to influence the patterns of distribution and size of fishes substantially.

In this study fish traps were used to investigate patterns of change in the distribution, abundance and size of *L. russelli* within and between three tropical estuaries over a two year period. In addition, these patterns were related to the salinity and temperature regimes prevailing in the three creeks.

6.2 METHODS

6.2.1 Study sites

Cattle, Barramundi and Alligator Creeks are mangrove-lined estuaries on the north-eastern coast of tropical Australia (Fig. 6.1). Cattle Creek (18° 15'S, 146° 15'E) (Fig. 6.2a) is the most northerly and flows into Halifax Bay. Alligator Creek (19° 20'S, 146° 55'E) (Fig. 6.2b), some 85km to the south-east, enters Cleveland Bay. Barramundi Creek (about 19° 25'S, 147° 10'E) (Fig. 6.2c), 36km south-east of Alligator Creek opens to Bowling Green Bay.

The three creeks have quite different geographies. Cattle Creek lies at the southern extremity of the "wet tropics" (Bonell, 1983). The hinterland surrounding Cattle Creek receives a considerably higher annual rainfall than areas around the other creeks further south (Oliver, 1978), with an average annual run-off in excess of 1 000mm (Bonell, 1983). Thus, Cattle Creek regularly receives substantial freshwater inflow from many tributaries that drain the highlands of the Seaview Range lying to the west. This high level of freshwater input is reflected in the mangrove flora lining the banks of Cattle Creek.

Both Alligator and Barramundi Creeks lie in the "dry tropics" (Bonell, 1983), each receiving considerably lower annual rainfall than Cattle Creek (Oliver, 1978). The main tributary of Alligator Creek drains the highlands of Mount Elliot to the west, providing a low level of freshwater inflow for at least part of the year. The catchment area of Alligator Creek around Mount Elliot receives an average annual run-off in excess of 500mm (Bonell, 1983). Barramundi Creek is surrounded by coastal lowlands with no adjacent highlands and no obvious, permanent freshwater tributaries. Therefore, the major freshwater input into Barramundi Creek is from seasonal rainfall or from general flooding of the lowlands during high rainfall years. Average annual run-off in the catchment area of Barramundi Creek is less than 375mm (Bonell, 1983).

For much of their lengths, all three creeks are lined with mangroves. The mangrove community at Cattle Creek shows a greater species richness than at the other two study sites (pers. obs.). Species such as *Heritiera littoralis*, a species typical of wet tropical streams influenced substantially by freshwater (Dowling and McDonald, 1982; Lovelock, 1993), occur commonly in upstream areas. Downstream

Figure 6.1. Map showing locations of Cattle, Barramundi and Alligator Creeks.

Figure 6.1

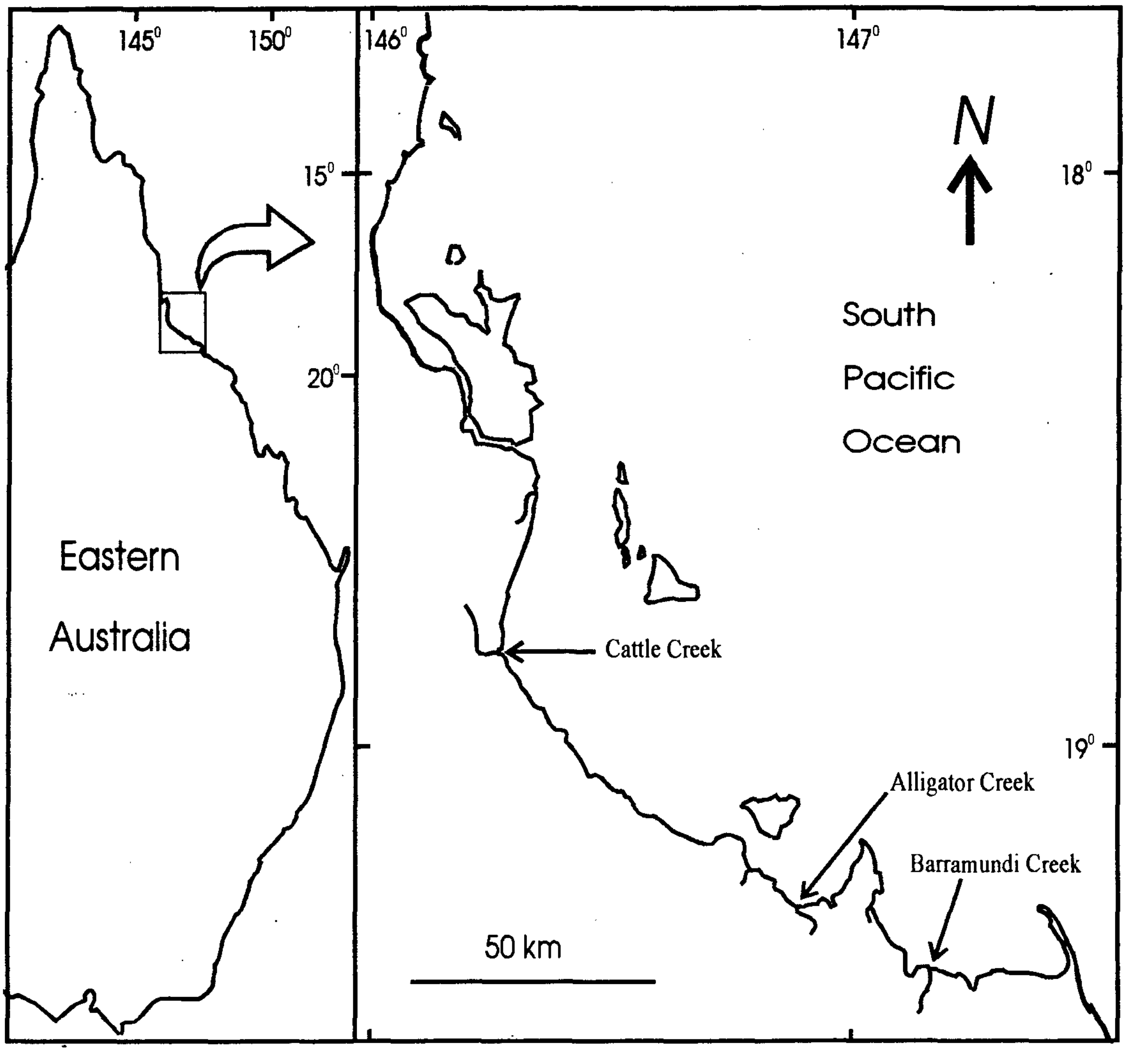

145°
150°
15°
20°
Eastern
Australia
146°
147°
18°
19°
N
South
Pacific
Ocean
Cattle Creek
Alligator Creek
Barramundi Creek
50 km

Figure 6.2. Aerial photographs of the seaward parts of a) Cattle, b) Barramundi Creeks and c) Alligator.

Figure 6.2:

a)

b)

c)

areas are dominated by *Rhizophora* spp., *Bruguiera* spp., *Avicennia marina,* and *Ceriops* spp. The mangrove flora of Alligator Creek is much less species rich than that of Cattle Creek and is dominated by *Rhizophora stylosa, Avicennia marina* and *Ceriops tagal* (Robertson and Duke, 1987). The flora of Barramundi Creek is broadly similar to that of Alligator Creek. Along the banks of all three creeks mangrove prop-roots and fallen mangrove trees provide large areas of structurally complex habitats (Sheaves, 1992). These areas are particularly prevalent along erosional banks on the outside radii of bends (see Fig. 5.1). Extensive salt pan areas occur adjacent to both Barramundi and Alligator Creeks. While salt pans do occur at Cattle Creek they are less extensive and are confined to lower parts of the creek.

The estuaries of both Cattle and Barramundi Creeks are some 15km in length, most of which is navigable by dinghy at low tide. While the influence of the tide extends some 14km along Alligator Creek only about 8km of this is deep enough to be navigable at low tide. All three creeks vary in maximum depth at low tide, from some 5m in downstream areas to less than 2m upstream.

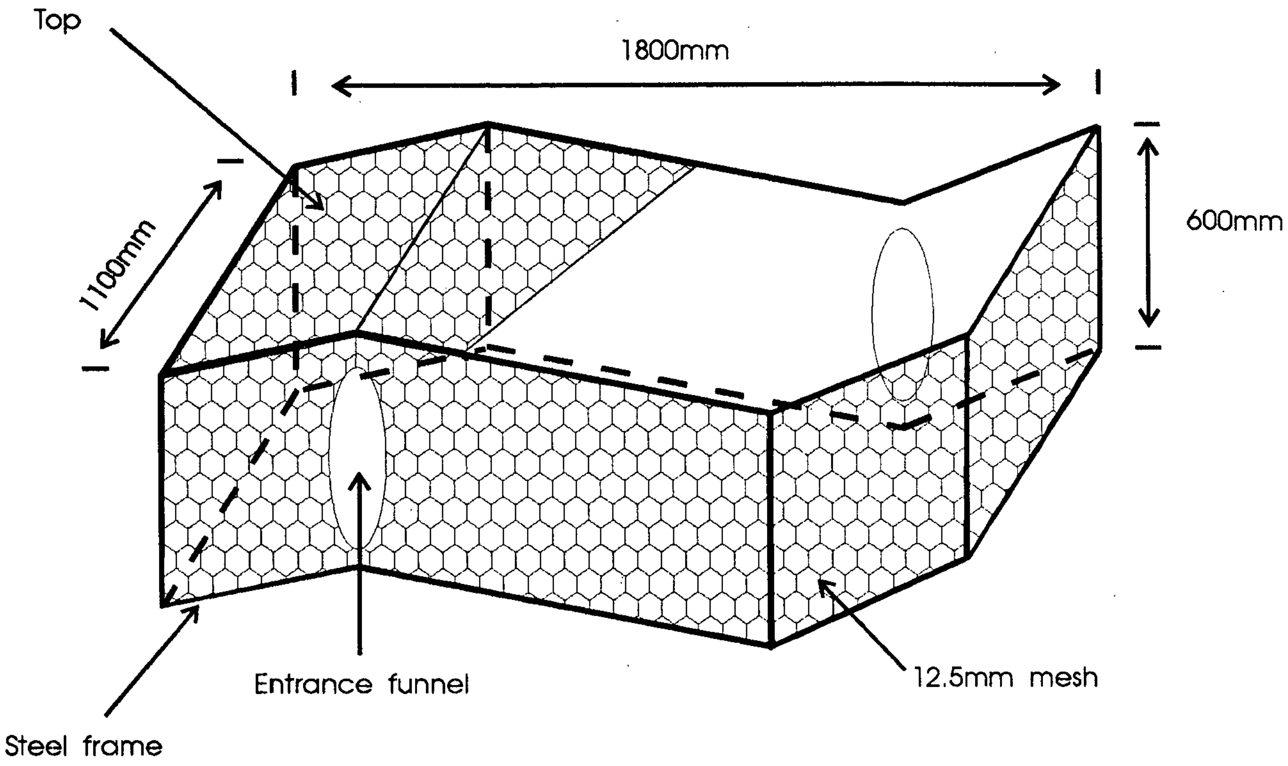
6.2.2 Biological sampling protocols

For 24 consecutive lunar cycles, between October 14, 1991 and August 25, 1993, samples were collected over the first quarter of the lunar cycle. Sampling occurred on the day ~~previous to~~ prior the first quarter, on the day of the first quarter and on the day following the first quarter. One creek was sampled in each lunar cycle. To enable data to be collected from each creek at a constant interval of 3 lunar cycles the creeks were always sampled in the same order, first Cattle Creek then Barramundi Creek then Alligator Creek.

Sampling was carried out using 12 Antillean-Z fish traps (1800mm long, 1100mm wide, 600mm high, plan area approx. 1.53m^2, volume approx. 0.92m^3) with 12.5mm square galvanised steel mesh and straight entrance funnels (Fig. 6.3). The straight entrance funnels were simple tapering, laterally compressed, conical tubes of mesh. Each trap had two funnels of the same design, one set into the concave angle of the "Z" on each side. The funnels were placed in such a way that the outer openings were flush with the vertical sides of the traps and the inner openings were towards the mid-lines of the traps with the inner entrance vertical and perpendicular

Figure 6.3: Diagrammatic representation of the Antillean-Z fish traps used for sampling.

Figure 6.3


Top
1800mm
1100mm
600mm
Entrance funnel
12.5mm mesh
Steel frame

to the trap floor. The funnels had the following approximate dimensions: 400x180mm outer opening, 260x150mm inner opening and 280mm long. Further details of the design of these traps can be found in Sheaves (1994). At each set, all traps were baited with 500g of Western Australian Blue Pilchards (*Sardinops neopilchardus*) as described in Sheaves (1994). Three trap soaks were made on each day; two, 3½ hour day time soaks and one overnight soak. Sampling was confined to the areas of high structural heterogeneity afforded by mangrove prop-roots and fallen timber (snags) as these areas are the major habitat of *Lutjanus russelli* (Sheaves, 1992, Chapter 5).

The lower 15kms of Cattle Creek (Fig. 6.4a) and Barramundi Creek (Fig. 6.4b) (measured along the creek mid-line from a 1:50,000 topographic map) were each divided arbitrarily into four regions (Bottom, Lower, Upper, Top) representing increasing distance upstream. Each of these regions was approximately 3 240m in length and separated from the adjacent region by a gap of about 500m. As Alligator Creek (Fig. 6.4c) had a shorter navigable length only the Bottom and Lower regions were defined.

Each region was divided into 3 sites approximately 1080m in length. One trap was used to sample within each site on each sampling trip. Each site was divided into seventy two, 15m long sections (Fig. 6.5), marked with surveyors tape tied to mangrove trees. During each 3-day trip, 9 of these sections was sampled in each site (6 day and 3 overnight trap soaks). On a particular trip, starting from either the top or bottom of the site (randomly selected) each trap was moved systematically along its site with every eighth section being sampled on a particular trip. On each trip a different starting point was used so that each possible position was used only once during the study. Each trap was placed close to the up-current side of the snag closest to the centre of the section so that the bait plume emanating from the trap (Whitelaw *et al.*, 1991) was directed into the snag.

For this study systematic sampling was preferred to random sampling. Caughley (1977) points out that systematic sampling often has many practical advantages over random sampling. For example, in the present study it greatly reduced the chance of accidentally resampling the same site, and by reducing the time needed to locate sites allowed a more efficient use of time. When systematic sampling is not biased with respect to the distribution of the target species it will

Figure 6.4. Maps of a) Cattle, b) Barramundi and c) Alligator Creeks showing the regions defined for the study. B = Bottom, L = Lower, U = Upper, T = Top.

Figure 6.4

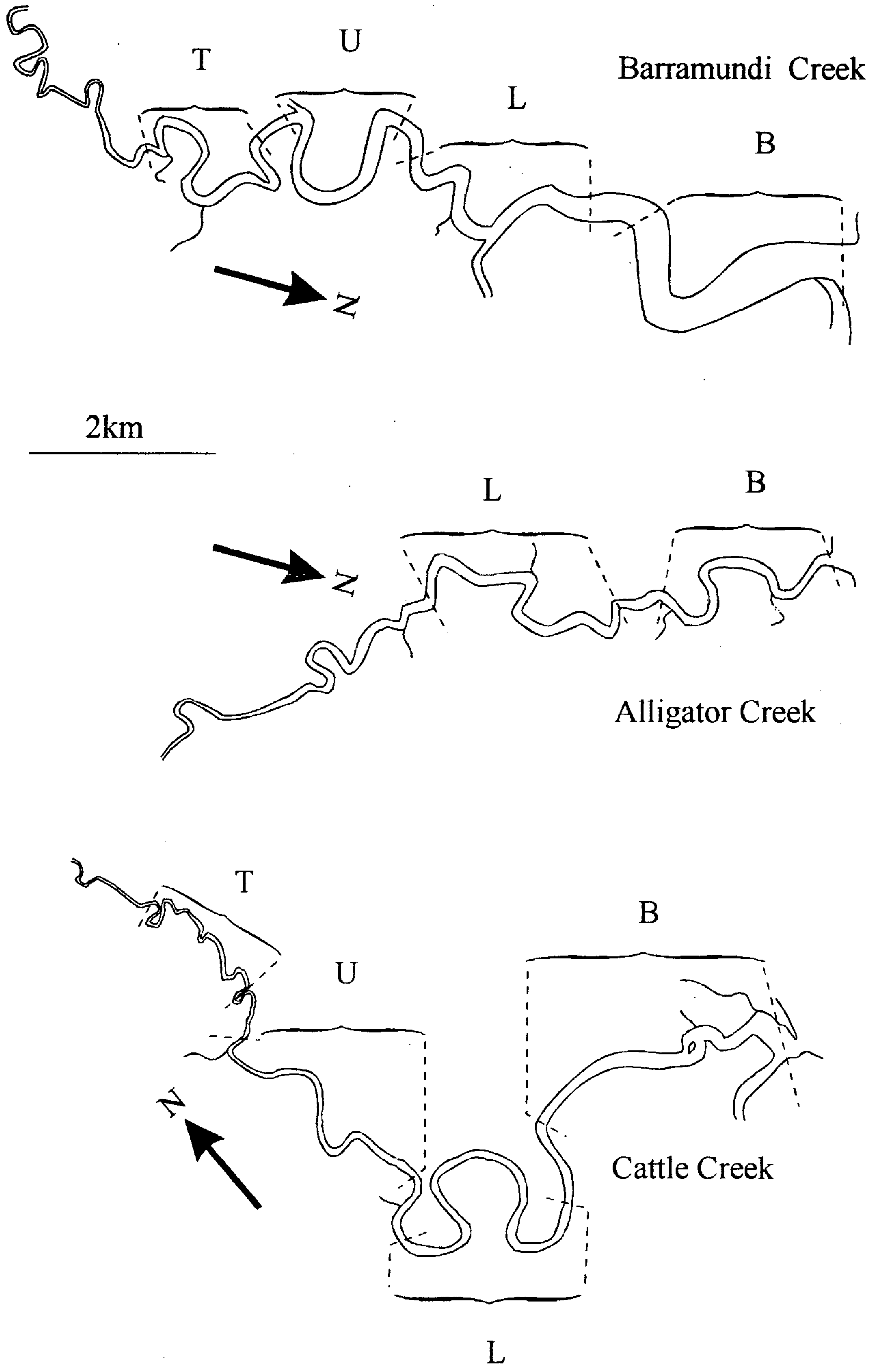
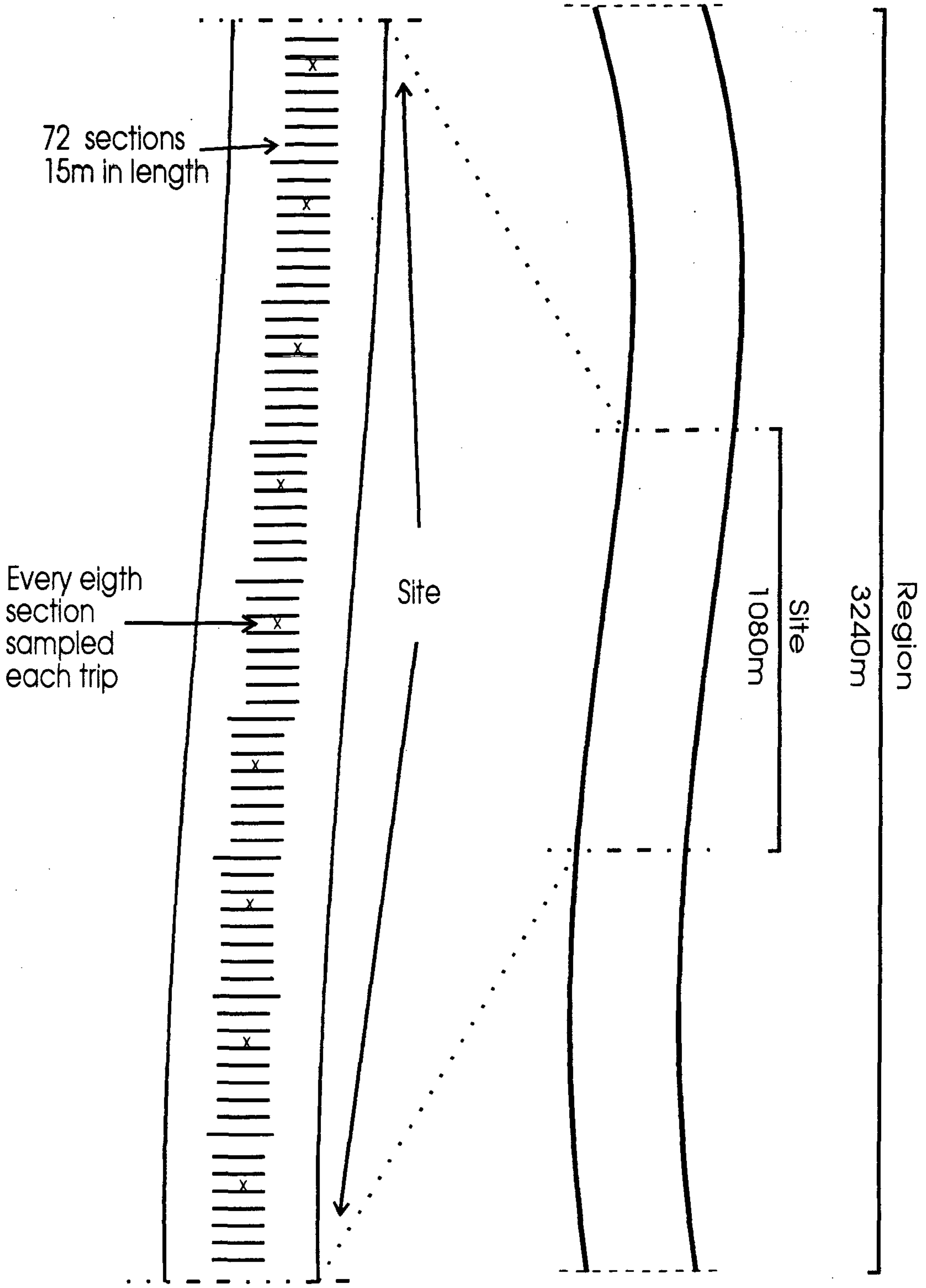

T
U
Barramundi Creek
L
B
N
2km
L
B
N
Alligator Creek
T
B
U
N
Cattle Creek
L

Figure 6.5. Diagrammatic representation of the division of the estuaries into sampling units [regions, sites and sections].

Figure 6.5


72 sections
15m in length
Every eigth
section
sampled
each trip
Site
Site
1080m
Region
3240m

provide a representative sample that does not violate the assumptions of statistical models grossly (Caughley, 1977).

To ensure that all samples were as independent as possible each trap was placed at a different site for each soak. However, although trap locations were kept 120m apart during any one trip, over the course of the study locations could be only 15m from a location used at a different time. To evaluate if the assumption of independence was valid all the *L. russelli* over 100mm FL that were released were tagged. Of 963 fish tagged only 9 fish [0.9%] were recaptured during the study. In previous studies where the same locations were resampled, recapture rates were much higher [11.4%] (Sheaves, 1993) [see Appendix II]. Thus it appears that a spacing of 15m was sufficient to ensure that different groups of fish were sampled with each trap. Therefore, in the present study the assumption of independence seems justified.

6.2.3 Physical sampling

On each sampling day from November 1991 (i.e. the 3 days of each field trip), surface and bottom temperatures and salinities were measured in the creek where trapping was in progress. Samples were taken as close to low tide as possible. On each occasion samples were taken at the same sites. These sites were located at the upper and lower ends of the sampling area and between each of the regions defined above (i.e. 5 physical sampling sites in Cattle and Barramundi Creeks and 3 in Alligator Creek). For each 3-day field trip the mean value of each parameter was calculated for each physical sampling site. These data were used to provide a summary of the physical variability for each of the 3 creeks during the study period. To allow investigation of the relationships between physical variables and catch rates of *L. russelli*, the mean of each of the physical variables for each region of each creek on each day was calculated. This was calculated as the mean of the values for the variable at the physical sampling sites at the upper and lower end of each region.

6.2.4 Data analysis

The primary study sites were Cattle and Barramundi Creeks where 4 regions (Bottom, Lower, Upper, Top) were defined. Sampling was conducted first at Cattle Creek then one lunar month later at Barramundi Creek followed by a gap of 1 lunar month before the pattern was repeated. This pattern was continued for 8 cycles extending over 24 lunar months.

In each of the lunar months that separated sampling at Cattle and Barramundi Creeks, sampling was carried out at Alligator Creek (the 2 most seaward regions only). This sampling was supplementary to the Cattle and Barramundi Creeks data set and served to extend the sampling spatially while completing the temporal sequence. Thus each creek was sampled on 8 occasions, 3 months apart. On each occasion, 3 traps were used in each region for 3 days with 2 daytime and 1 overnight sample collected from each trap on each day. Because only the two most seaward regions (Bottom and Lower) could be defined in Alligator Creek the data were analysed in two phases; all regions of Cattle and Barramundi Creeks (analysis I), and the lower 2 regions of all three creeks (analysis II).

The groups of successive samples from the three creeks over a three month period (i.e. Cattle, Barramundi then Alligator) were considered to constitute a season for the purposes of these analyses. However, as there was no gap separating the "seasons" this division was wholly arbitrary and served only to group the samples broadly into units for temporal comparisons.

The seasons were designated as follows: Spring 1991 (October/November/December 1991), Summer 1992 (January/February/March 1992), Autumn 1992 (April/May/June 1992), Winter 1992 (July /August/September 1992), Spring 1992 (October/November/December 1992), Summer 1993 (January/February/ early March 1993), Autumn 1993 (late March/April/May 1993), Winter 1993 (June/July/August 1993). Due to the differences in lengths of lunar and calendar months each sampling period in the second year occurred about 11 days earlier than the corresponding sampling period in the first year.

A significance level of α=0.05 was used for all hypothesis tests. Where analysis of variance (ANOVA) was employed, when appropriate, *a posteriori* comparisons of means were conducted using Tukey's test or orthogonal polynomial contrasts.

6.2.4.1 Analysis of the Distribution of *L. russelli*

To determine if the catch in a trap could be considered a representative sample from the region in which it was situated, the mean and variance of numbers of *L. russelli* in traps set in the same region on the same day (daytime soaks only) were plotted against each other. The results of this preliminary analysis [see results; 6.3.1] suggested that the numbers of fish in a treatment combination could be influenced substantially by large catches in one or a few traps. Because of the potential for a few large catches to bias the data numbers of fish was considered too unreliable a variable to be used in comparisons between treatment levels. In consequence, a two-stage analytical strategy was employed.

Firstly, the data from each trap was considered on a presence/absence basis and modelled using logistic regression (Appendix I.1) and backward elimination (Collett, 1991). This approach compared the probability of trapping at least one *L. russelli* between treatment levels. As the variate of interest was the presence or absence of *L. russelli* in each trap, separate analyses were carried out for the day and night samples to overcome any confoundment arising from differences in the probability of capture between day and night soaks. The variables of interest were seasons, creeks and regions. Thus the data consisted of the presence or absence of *L. russelli* in each trap (18 for daytime or 9 for night-time soaks) in each season*creek*region combination. From the fitted logistic regression models the probabilities of traps in particular treatments containing at least one *L. russelli* individual were calculated. The use of presence/absence data also overcame problems inherent in analysing data sets containing a high proportion of zeros.

Secondly, to determine if the number of fish caught per trap differed between seasons, creeks or regions the numbers of fish in those traps in which fish were caught (i.e. traps with 0 numbers excluded) was analysed for daytime samples in Cattle and Barramundi Creeks using analysis of variance. The variables analysed (Seasons, Creeks, Regions) were considered fixed and orthogonal. As differences between treatment levels of count data tend to be multiplicative rather than additive a $\log_{10}$ transformation was used. This transformation resulted in residuals displaying greatly improved homoscedasticity and normality compared to residuals from the raw data.

6.2.4.2 Analysis of the Fork Length of *L. russelli*

The preferred option for the analysis of the fork length of *L. russelli* was log-linear modelling of the data grouped into size classes. However, there was a lack of any fish of small sizes in a number of samples. Thus, to avoid a contingency table with a number of marginal zeros, the smaller size class would need to be pooled to such an extent that real trends in the data could be obscured. Therefore ANOVA was used to investigate changes in the mean fork length of *L. russelli*.

If *L. russelli* form size specific schools, fish within one trap would tend to be more similar in size than fish from a region as a whole. In this case the fish within one trap would not be independent replicates. To test this a subset of the data including all the instances where at least 2 fish were caught in each of at least 2 traps in 1 region at 1 soak on 1 day was analysed. The lengths of 2 fish were selected at random from each trap catch and the intraclass correlation coefficient (Appendix I.2) calculated (Zar, 1984; p.323). The intraclass correlation coefficient for pairs of fish selected at random from all traps in each region/soak/sampling-day combination was calculated also. These two intraclass correlation coefficients were then compared (Zar, 1984; p.315; Sokal and Rohlf, 1981, p.588). Because of strong evidence from this analysis that lengths of fish within traps were highly correlated [see results; 6.3.2] and thus not independent, the mean fork length (FL) of fish within a trap was used as the variate in the analyses of variance.

Many individual trap hauls caught no *L. russelli*. This meant that the data matrix contained many cells with no data. To overcome this the data were pooled over 'Sites' and 'Days', factors considered to be of little interest. This produced a model containing the three factors of interest (Seasons, Creeks, Regions) in which all cells contained data.

The data were transformed by $\log_{10}(x+1)$ to achieve homogeneity of variance as judged by plots of residuals against predicted values. Normal probability plots suggested that the transformed data did not deviate greatly from normality.

There were differences in mean lengths between regions (Table 6.5). To determine the components of the length distributions responsible for these differences the length frequencies in the 4 regions were compared by log-linear modelling.

In the data for the 8 seasons a strong periodic pattern that suggested an annual pattern of variation in size was apparent. To investigate the nature of this pattern the periodogram (Appendix I.3) of the sequence of mean fork lengths for the 2 lower regions for each of the 24 consecutive lunar cycles (disregarding the fact that the data came from 3 creeks) was calculated. To test if the cyclic pattern observed was attributable to white noise alone a cumulative periodogram of the data was constructed together with the 5% critical limit of the Bartlett D statistic (Diggle, 1992). If the cumulative periodogram deviates from linearity to a greater extent than the critical value the deviation in the cumulative periodogram is greater than can be explained by white noise alone.

6.3 RESULTS

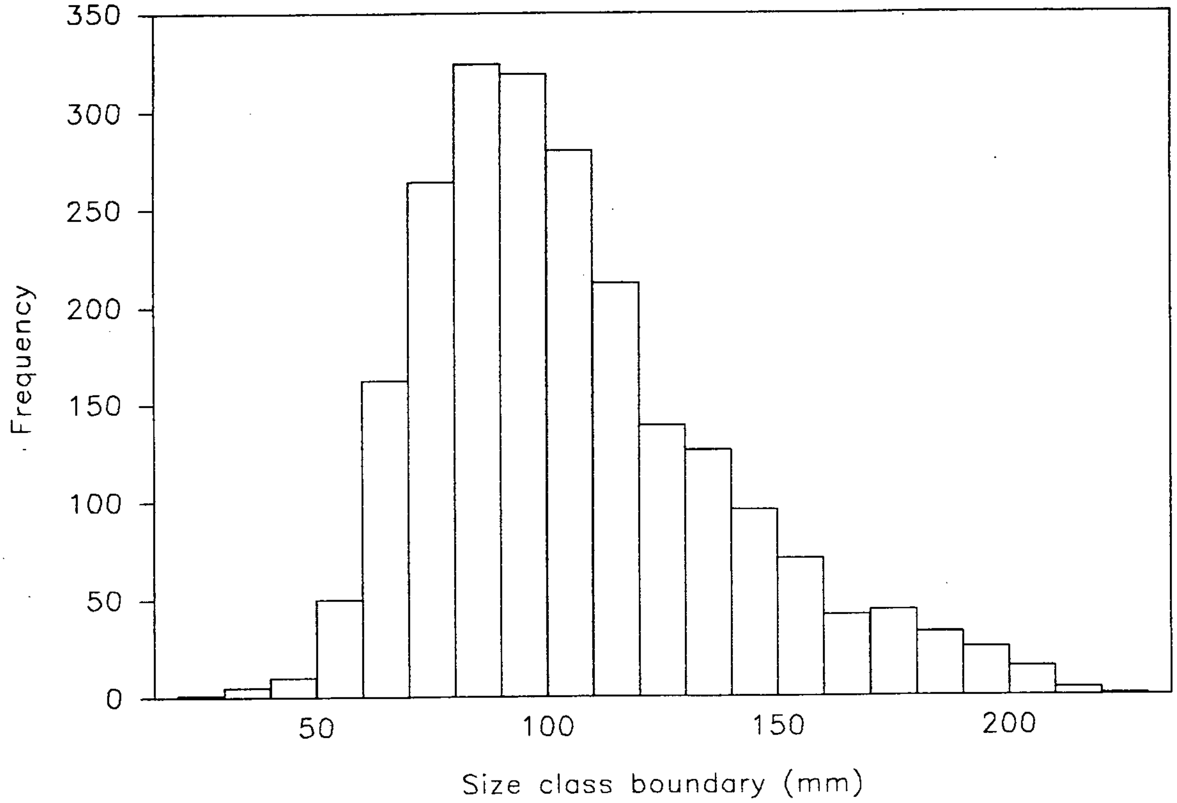
Over the course of the study a total of 2 223 *L. russelli* were trapped in Cattle, Alligator and Barramundi Creeks (Table 6.1). Fork lengths ranged from 28mm to 220mm with a single mode at 80-100mm (Fig 6.6). As it was possible for fish below about 60-70mm to squeeze through the trap mesh (pers. obs.), the sharp decline in numbers at smaller sizes can be attributed largely to the retention characteristics of the trap mesh.

Table 6.1: Summary of fish trap catches of *L. russelli* in Cattle, Barramundi and Alligator Creeks between 1991 and 1993.

Season	Creek	Numbers
Spring 1991	Cattle	36
	Barramundi	63
	Alligator	60
Summer 1992	Cattle	207
	Barramundi	131
	Alligator	85
Autumn 1992	Cattle	20
	Barramundi	148
	Alligator	159
Winter 1992	Cattle	37
	Barramundi	82
	Alligator	100
Spring 1992	Cattle	52
	Barramundi	70
	Alligator	24
Summer 1993	Cattle	47
	Barramundi	98
	Alligator	122
Autumn 1993	Cattle	89
	Barramundi	215
	Alligator	162
Winter 1993	Cattle	70
	Barramundi	69
	Alligator	77
Total		2223

Figure 6.6. Size frequency distribution of 2 223 *L. russelli* trapped in Cattle, Barramundi and Alligator Creeks.

Figure 6.6


350
300
250
200
150
100
50
0
Frequency
50
100
150
200
Size class boundary (mm)

6.3.1 Analysis of the Distribution of *L. russelli*

When the mean and variance of numbers in traps from regions at particular times were plotted against each other (Fig. 6.7) the variance tended to increase greatly with the mean. In particular, high mean numbers per trap within a region were associated with very high variances. The major reason for this was that in many cases high mean numbers of fish in a treatment combination resulted from large catches in one or a few traps while most of the traps in the combination contained few fish. It appeared therefore that in many cases the catch in a trap was not a truly representative sample from the region in which it was set.

For both the analysis of all regions of Cattle and Barramundi Creeks (analysis I) and the analysis of data for the 2 lower regions of all three creeks (analysis II), logistic regression analyses (presence/absence data) produced models for day and night samples that differed little in any of their main features. Consequently only results for the larger daytime data sets are presented.

For analysis I (4 regions of Cattle and Barramundi Creeks) the final logistic model contained both the Creek X Region and the Creek X Season interactions (Table 6.2a). There was a trend across all regions for the probability of trapping at least 1 *L. russelli* to be considerably lower in Cattle Creek than in Barramundi Creek (Fig. 6.8). For the two upstream regions the probability of capture in Barramundi Creek was clearly higher than in Cattle Creek. Furthermore, while the probability did not differ substantially between regions in Barramundi Creek, this was not the case in Cattle Creek. In the Bottom and Lower regions of Cattle Creek the probability was similar but dropped sharply in the Upper region and again in the Top region (Fig. 6.8).

Figure 6.7. Relationship between the mean and variance for numbers of *L. russelli* caught in traps from regions at particular times. The variance data are plotted on a $\log_{10}$ scale to allow the identification of small values.

Figure 6.7

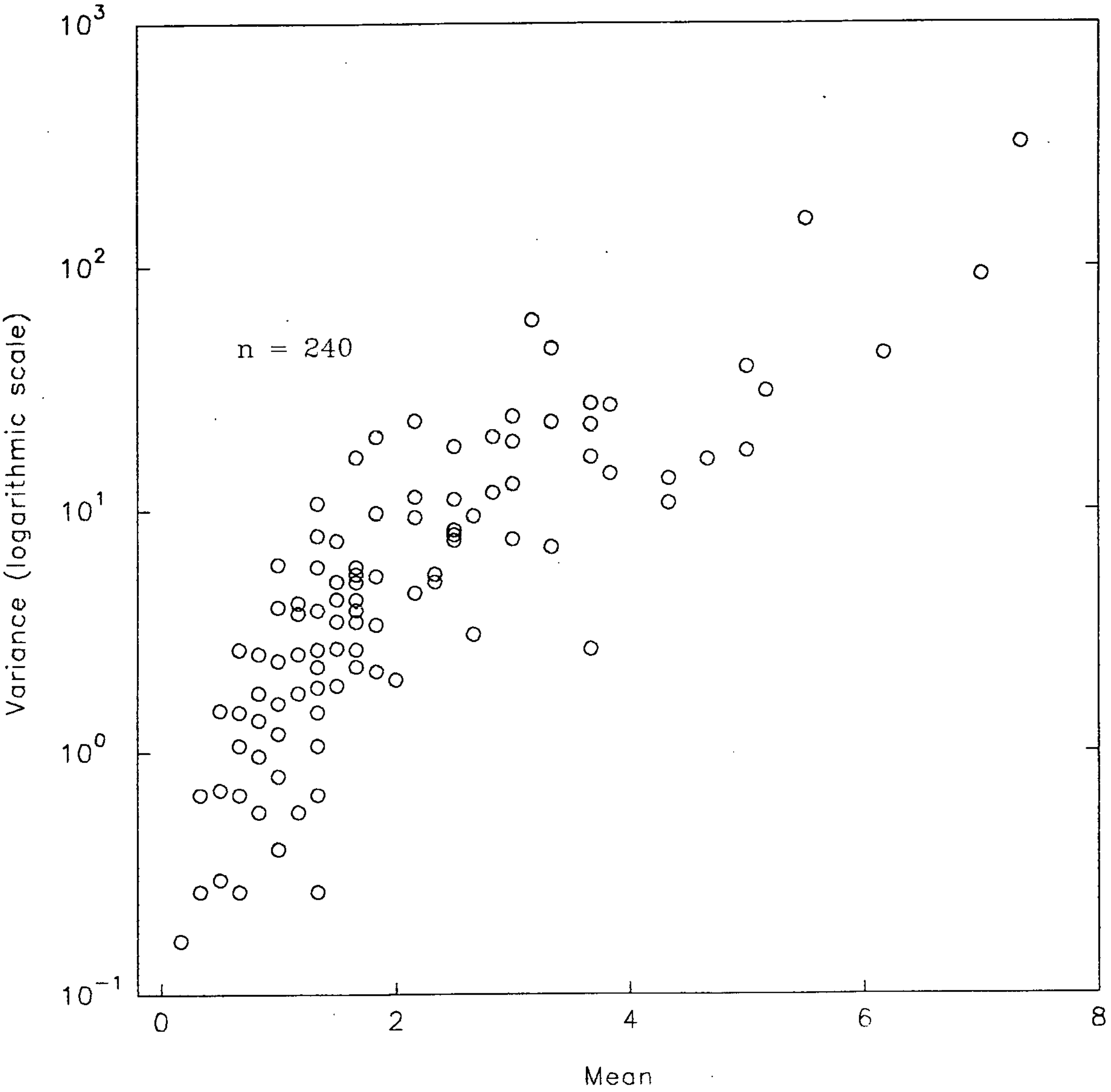

10^3
10^2
10^1
10^0
10^{-1}
Variance (logarithmic scale)
n = 240
0
2
4
6
8
Mean

Figure 6.8. Probabilities of trapping at least 1 *L. russelli* in 4 regions of Cattle and Barramundi Creeks. Error bars are 95% confidence intervals.

Figure 6.9. Probabilities of trapping at least 1 *L. russelli* over 8 seasons in Cattle and Barramundi Creeks. Error bars are 95% confidence intervals.

Figure 6.8

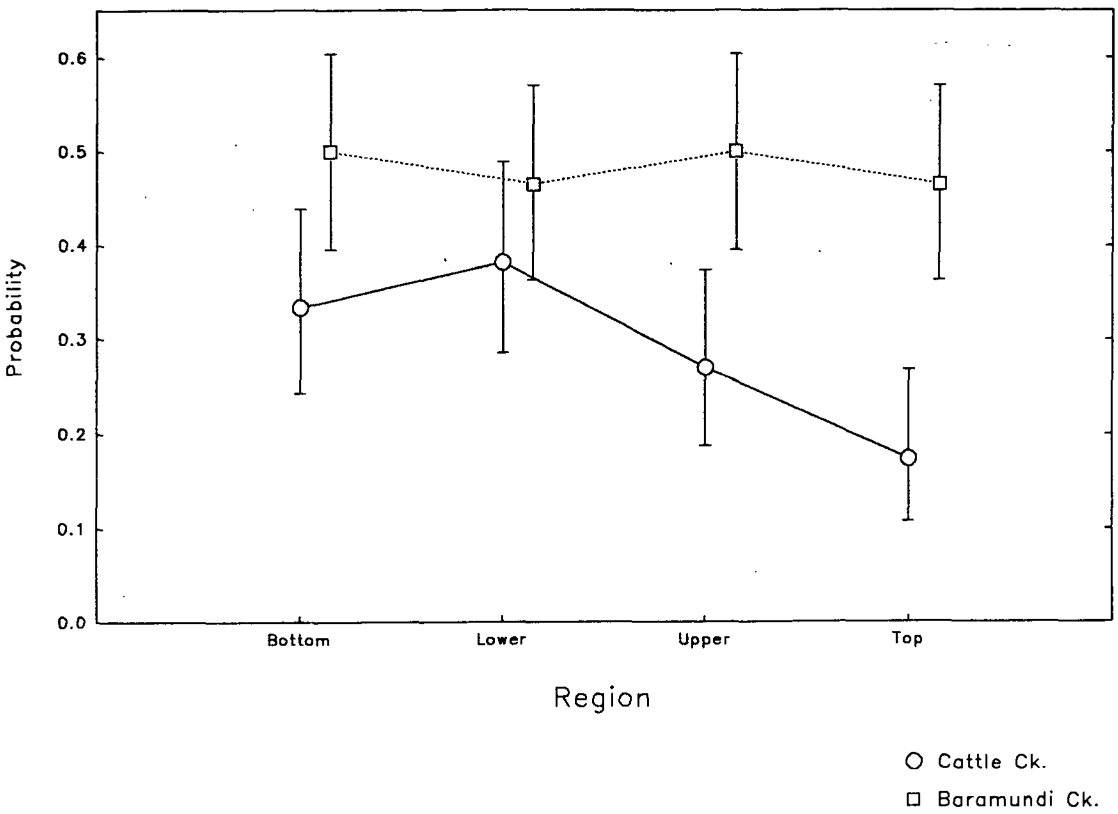


Figure 6.9

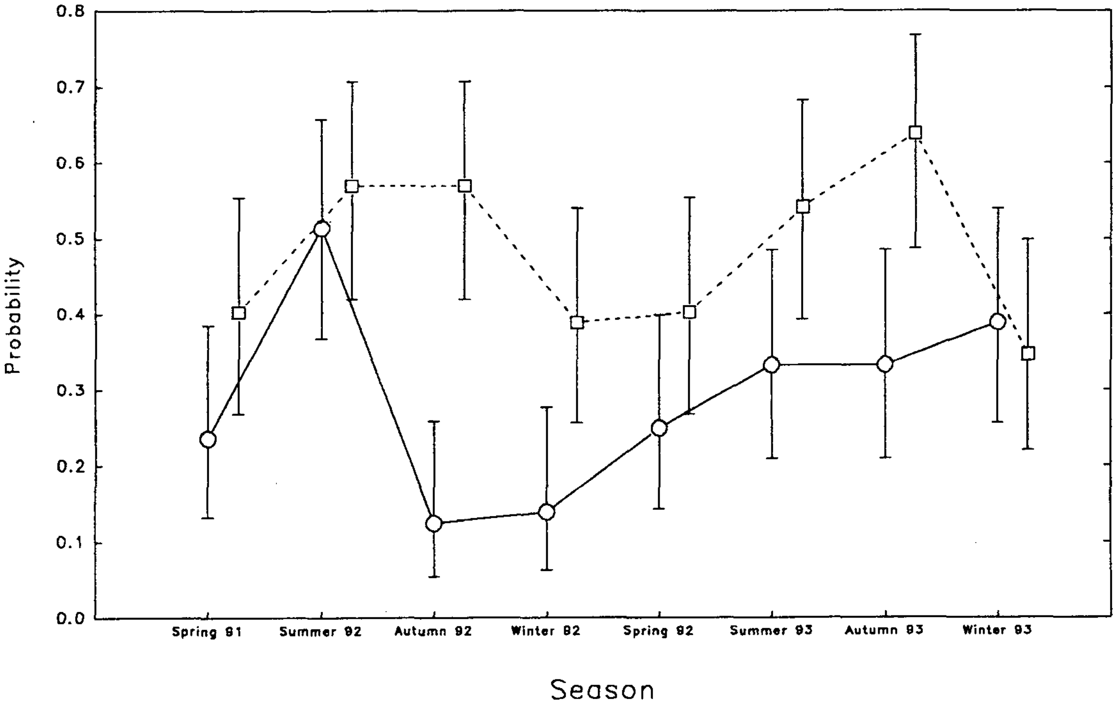


Table 6.2: Stepwise fitting (backwards elimination) of logistic regression models for the presence/absence of *L. russelli* in three estuaries. (a) Four regions in Cattle and Barramundi Creeks (analysis I), (b) Two regions in Cattle, Barramundi and Alligator Creeks (analysis II).

The three left hand columns show the models fitted, and the associated deviance and degrees of freedom. The three right hand columns show the codes for the current model and the more complex being compared together with the associated changes in deviance (Δdeviance), the changes in the degrees of freedom (Δd.f) and the probability (P) that the more simple model fits the data adequately. Models containing interactions also contain the main effects comprising the interaction. The final model is the model from which no term can be removed without a significant increase in deviance (see Appendix I).

"O" is the code for the full 3-way model (deviance = 0, d.f. = 0)
denotes the final model.

Model	Deviance	d.f.	Comparison & Δdeviance	Δd.f	P
(a)					
A) No 3-way interaction	27.20	21	A - O 27.20	21	0.164
B) No 2-way interactions	87.29	52	B - A 60.09	31	0.001
C) Creek*Season + Creek*Region ##	**57.83**	**42**	**C - A 30.63**	**21**	**0.081**
D) Creek*Season + Region	65.94	45	D - C 8.11	3	0.044
(b)					
A) No 3-way interaction	14.00	14	A - O 14.00	14	0.449
B) No 2-way interactions	32.35	37	B - A 18.35	23	0.738
C) Season + Creek	32.79	38	C - B 0.44	1	0.505
D) Season ##	**38.64**	**40**	**D - C 5.85**	**2**	**0.054**
E) Creek	65.52	45	E - C 32.73	7	0.000

The probability of capturing a *L. russelli* tended to be higher in Barramundi Creek for most seasons also (Fig. 6.9) and for Autumn 1992 and Autumn 1993 the differences were clear. For Barramundi Creek there was some suggestion of an annual cycle. The probability of capture increased from Spring 1991 to a peak in Summer 1992 and Autumn 1992, decreased to low levels in Winter 1992, then increased steadily to high levels in Autumn 1993 before falling away in Winter 1993 (Fig. 6.9). Overall probabilities of capture tended to be high in summer and autumn and low in winter and spring. The pattern for Cattle Creek showed some marked differences to that of Barramundi Creek. One of these, the low probability for Cattle Creek in Autumn 1993 coincided with a large influx of freshwater from major flooding (Fig. 6.23).

In the analysis of the 2 seaward regions of all 3 creeks (analysis II), none of the factors were found to interact significantly, and only the factor season was important in the model (Table 6.2b). The pattern of seasonal change closely followed that described above for analysis I. The factor creeks was not significant indicating that when only the lower regions of the creeks were considered, differences between the 3 creeks were not substantial.

The analysis of variance of the number of fish caught per trap (when zero data were omitted) indicated that there was a significant effect of region and a significant interaction between creeks and seasons (Table 6.3).

Table 6.3: Analysis of variance table for the number of *L. russelli* caught per trap in Cattle and Barramundi Creeks (zero catch data omitted). ## indicates significant effects

Source of variation	d.f. of F ratio	F	P
Season ##	**7/382**	**5.7930**	**0.000002**
Creek	1/382	2.5497	0.111135
Region ##	**3/382**	**4.1562**	**0.006467**
Season X Creek ##	**7/382**	**2.7251**	**0.009057**
Season X Region	21/382	1.1307	0.312647
Creek X Region	3/382	2.1190	0.097338
Season X Creek X Region	21/382	1.4317	0.099176

Larger numbers were caught per trap in bottom regions than in either lower or top regions [respective Tukey's probability levels; $p=0.03387$, $p=0.00012$] (Fig. 6.10). Trend analysis using orthogonal polynomial contrasts showed a significant linear decline in catch per trap with distance upstream (Table 6.4).

Table 6.4: Summary of orthogonal polynomial contrasts for the effect of region in the analysis of variance in table 6.3.

indicates significant effect

Source of variation	d.f. of F ratio	F	P
Linear ##	**1/382**	**9.954836**	**0.001731**
Quadratic	1/382	0.123052	0.725942
Cubic	1/382	1.668560	0.197232

Over the course of the study there was no consistent pattern for the size of catches in traps to be larger in either Barramundi or Cattle Creek (Fig. 6.11). The pattern of catch sizes in Barramundi Creek follows the pattern of probability of capture (Fig. 6.9) closely. In contrast, for Cattle Creek the pattern of catch sizes and the pattern of probability of capture (Fig. 6.9) show a marked difference. Both the probability of capture and catch size reached very low levels in Autumn 1992. This coincided with flooding in Cattle Creek (Fig. 6.23). However, while the probability of capture of *L. russelli* in Cattle Creek remained low in Winter 1992, the mean size of catches increased greatly.

6.3.2 **Spatial and Temporal Comparisons of Size of *L. russelli***

The intraclass correlation coefficient for fish randomly selected from individual traps [$r_I=0.5177$; $n=231$] was compared to the intraclass correlation coefficient for pairs of fish selected at random from all traps in each region/soak/sampling-day combination [$r_I=0.2295$; $n=103$]. This comparison produced evidence [$X^2=8.0134$; 1df; $p=0.0046$] that fish length was much more highly correlated within traps than within regions as a whole. Thus the lengths of *L. russelli* within traps were not independent. Because of this non-independence the mean fork length of fish for each trap was used as the variate in the analyses of variance.

Figure 6.10. Trend in the number of *L. russelli* caught per trap in 4 regions of Cattle and Barramundi Creeks. Data are means and 95% confidence intervals. Zero catches ignored.

Figure 6.11. Effect of the interaction of creeks and seasons on the number of *L. russelli* caught per trap in Cattle and Barramundi Creeks. Data are means and 95% confidence intervals.

Figure 6.10

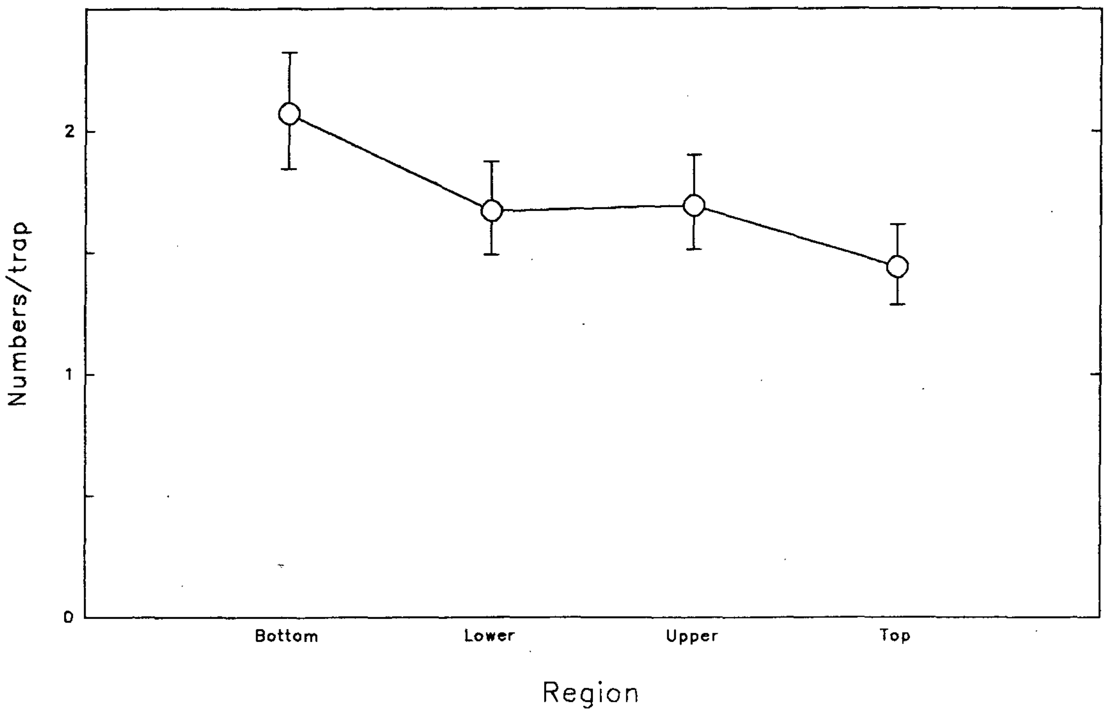
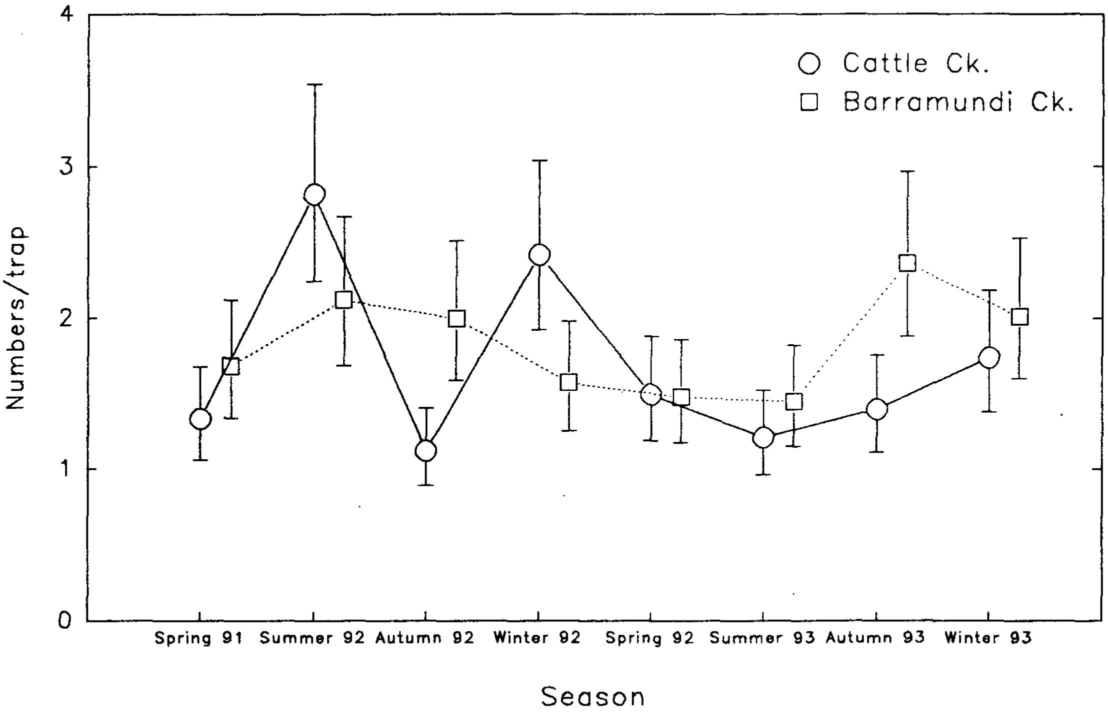


Figure 6.11



Both analysis I [4 regions of Cattle and Barramundi Creeks] (Table 6.5) and analysis II [2 lower regions of all 3 creeks] (Table 6.6) yielded the same significant factors. The mean fork length of *L. russelli* was influenced significantly by the interaction between season and location, and independently by region.

Table 6.5: Analysis of variance tables for the mean fork length of *L. russelli* from four regions of Cattle and Barramundi Creeks.
The data were transformed by $\log_{10}(x+1)$ prior to analysis. Factors marked with ## are significant at a significance level of α =0.05.

Source of variation	d.f.	Mean square	F-ratio	Significance level
Season (S) ##	**7**	**1.531860**	**44.62318**	**0.000000**
Location (L) ##	**1**	**0.206446**	**6.01378**	**0.014698**
Region (R) ##	**3**	**0.162172**	**4.72408**	**0.003038**
S*L	**7**	**0.129369**	**3.76852**	**0.000596**
S*R	21	0.043944	1.208008	0.184999
L*R	3	0.010177	0.29645	0.827966
S*L*R	21	0.051281	1.49382	0.076491
Residual	339	0.034329		

Table 6.6: Analysis of variance tables for the mean fork length of *L. russelli* from two regions of Cattle, Barramundi and Alligator Creeks.
The data were transformed by $\log_{10}(x+1)$ prior to analysis. Factors marked with ## are significant at a significance level of α =0.05..

Source of variation	d.f.	Mean square	F-ratio	Significance level
Season (S) ##	**7**	**1.065226**	**24.77756**	**0.000000**
Location (L) ##	**2**	**0.186980**	**4.34923**	**0.013814**
Region (R) ##	**1**	**0.426218**	**9.91400**	**0.001820**
S*L	**14**	**0.201316**	**4.68268**	**0.000000**
S*R	7	0.069882	1.62548	0.128062
L*R	2	0.037552	0.87346	0.418654
S*L*R	14	0.057539	1.33838	0.184164
Residual	276	0.042992		

The effect of regions was consistent across both analyses. For analysis II (Fig. 6.12), *L. russelli* from the bottom regions had a mean length 7.7mm greater than those from the lower regions. For analysis I *L. russelli* from the bottom region had a mean length 9.4mm larger than those from the lower region and 7.6mm larger than those from the upper region [respective Tukey's probability levels; p=0.001321, p=0.027293] but were not significantly larger than fish from the top region. While these difference were quite small they represented between about 8% and 10% of fork length. The mean length in the top region was also significantly greater (6.8mm) than that in the lower region [Tukey's probability level; p=0.041212] but was not significantly different from that in the upper region.

Although for analysis I the relative size frequencies in the 4 regions were very similar (Fig. 6.13) there were some statistical differences. A partitioning of the deviance from a log-linear model fitted to a 2-way contingency table of these size frequency data (4 regions X 5 size classes) suggested that the distribution of sizes was similar for the Lower and Upper regions [G^2=6.5477, 4df, p=0.1618] and that all 4 regions had similar proportions in the three middle size classes (70mm ≥ FL < 110mm, 110mm ≥ FL < 150mm, 150mm ≥ FL < 190mm) [G^2=2.5946, 4df, p=0.6278]. The bulk of the deviance in the 2-way table [G^2=30.6070, 4df, p=0.0000] was contained in the partitioning illustrated in Table 6.7.

Table 6.7: Column proportions for a 2-way contingency table of size classes of *L. russelli* by region after partitioning of deviance.
(Small <70mm; Medium 70 - <190mm; Large > 190mm).

	Bottom	Lower/Upper	Top
Small	0.103	0.115	0.041
Medium	0.861	0.873	0.951
Large	0.036	0.011	0.008

In the Bottom region small and medium size classes occurred in similar proportions as in the Lower/Upper regions [G^2=0.4191, 1df, p=0.5174] but there were a higher proportion of fish in the large size class in the Bottom region [G^2=13.2704, 1df, p=0.0003]. For the Top region medium and large size classes occurred in similar

Figure 6.12. Comparison of the fork lengths of *L. russelli* in four regions of Cattle and Barramundi Creeks and the two lower regions of Cattle, Barramundi and Alligator Creeks. Data are means and 95% confidence intervals.

Figure 6.12

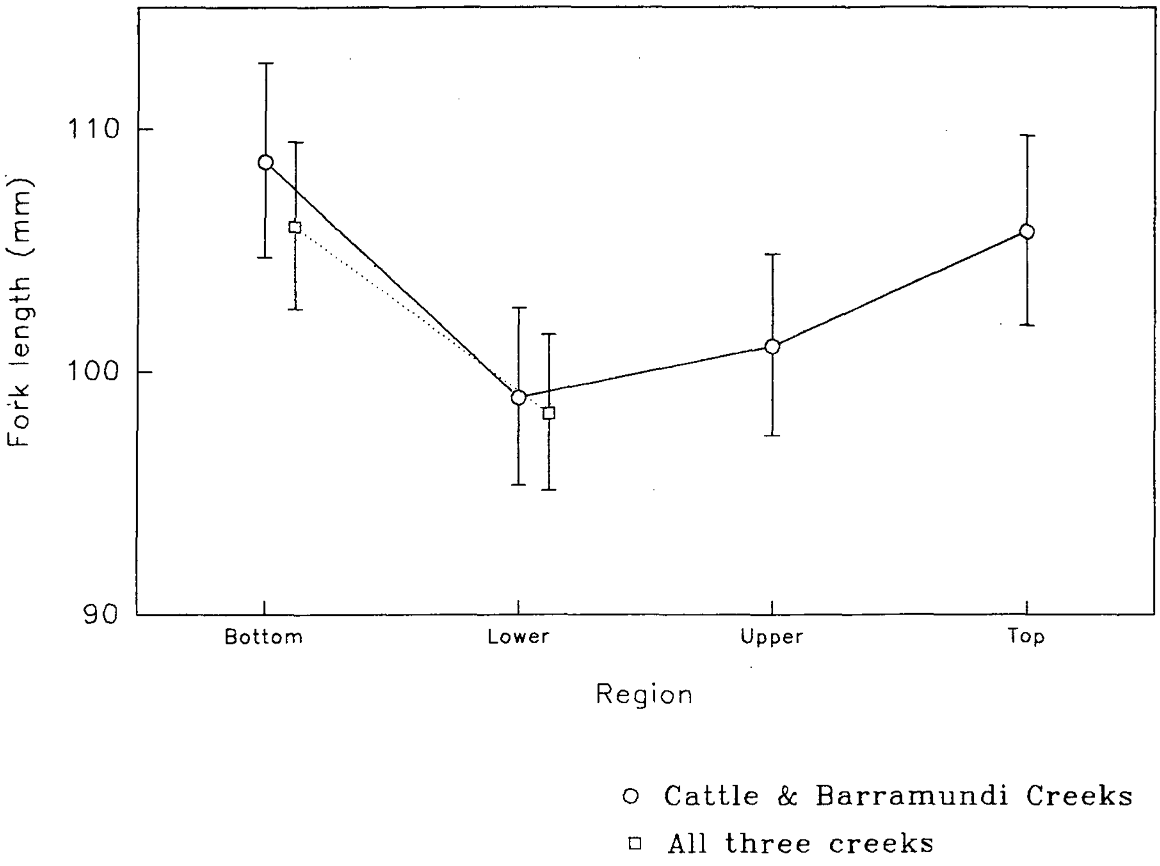
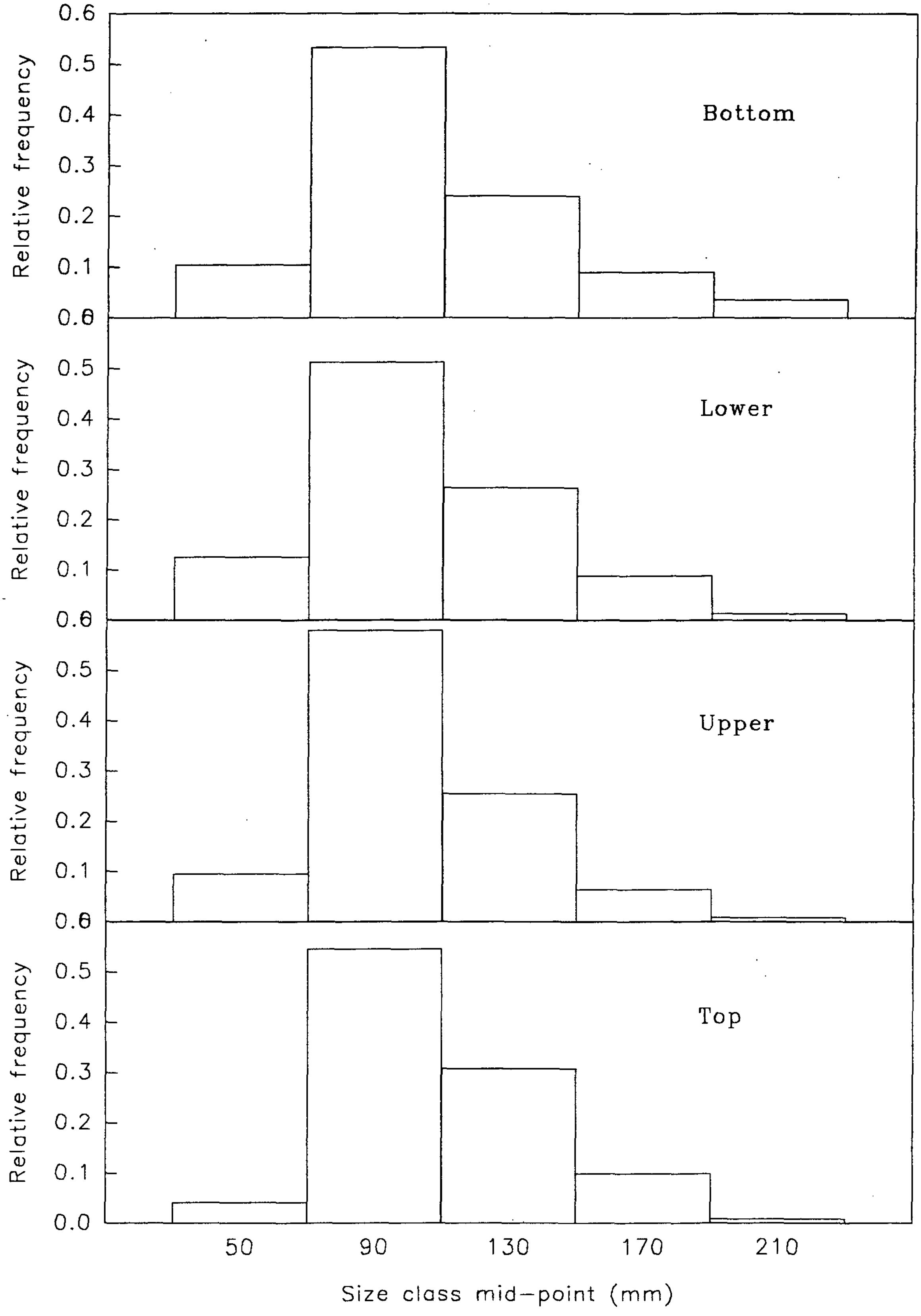

Fork length (mm)
110
100
90
Bottom
Lower
Upper
Top
Region
○ Cattle & Barramundi Creeks
□ All three creeks

Figure 6.13. Relative size-frequencies of *L. russelli* in four regions of Cattle and Barramundi Creeks.

Figure 6.13


Relative frequency
0.0
0.1
0.2
0.3
0.4
0.5
0.6
Bottom
Lower
Upper
Top
50
90
130
170
210
Size class mid-point (mm)

proportions as in the lower/upper regions [G^2=0.0.3310, 1df, p=0.5651] but a lower proportion of fish were in the small size class in the Top region [G^2=14.6193, 1df, p=0.0001]. Thus the larger mean sizes in the Bottom and Top regions had different causes - a greater proportion of large fish in the Bottom region and a lesser proportion of smaller fish in the Top region (Fig. 6.13).

While there was a strong season by location interaction in both analyses, significant differences between the two creeks were confined to three pairs of means. For analysis I (Fig. 6.14) the only significant difference was in Autumn 1992 when the mean size in Cattle Creek was 26.7mm smaller than that in Barramundi [Tukey's test; p=0.000133]. This relates to a period of flooding in Cattle Creek when the probability of capture was very low in the two upstream regions (Fig. 6.9). For analysis II (Fig. 6.15), the mean size of *L. russelli* in Spring 1991 was 49.8mm larger in Barramundi Creek than in Alligator Creek [Tukey's test; p=0.002023]. The mean size in Cattle Creek was intermediate and did not differ significantly from that in the other creeks. In Autumn 1993 the mean size in Alligator Creek was 28.1mm larger than in Cattle Creek [Tukey's test; p=0.001653]. The mean size in Barramundi Creek was intermediate and did not differ significantly from that in the other creeks. As the data for these "seasons" were collected in sequential months rather than contemporaneously differences between creeks arbitrarily grouped into "seasons" are to be expected. Consequently, these differences must be viewed with caution.

Despite these differences, the overall patterns of change in mean length per season over the course of the study, was very similar for all three creeks and for both analyses (Figs. 6.14, 6.15). The mean fork length of *L. russelli*, initially high in Spring 1991, declined to low levels in Summer and Autumn 1992, increased again to peak in Spring 1992, fell away to low levels in Summer and Autumn 1993, then rose again in Winter 1993. Overall, these data suggest an annual pattern of variation in size.

The changes in size structure producing the pattern of temporal change in mean fork length can be clearly seen in the size frequency distributions for each creek (Figs. 6.16, 6.17, 6.18). This pattern was similar for all three creeks and is repeated twice over the 8 seasons. In Spring 1991 and Spring 1992 there were low proportions of small fish (50 & 90mm) and high proportions of large fish (170 & 210mm)

Figure 6.14. The effect of the interaction of creeks and seasons on the fork length of *L. russelli* in Cattle and Barramundi Creeks. Data are means and 95% confidence intervals.

Figure 6.15. The effect of the interaction of creeks and seasons on the fork length of *L. russelli* in Cattle, Barramundi and Alligator Creeks. Data are means and 95% confidence intervals.

Figure 6.14

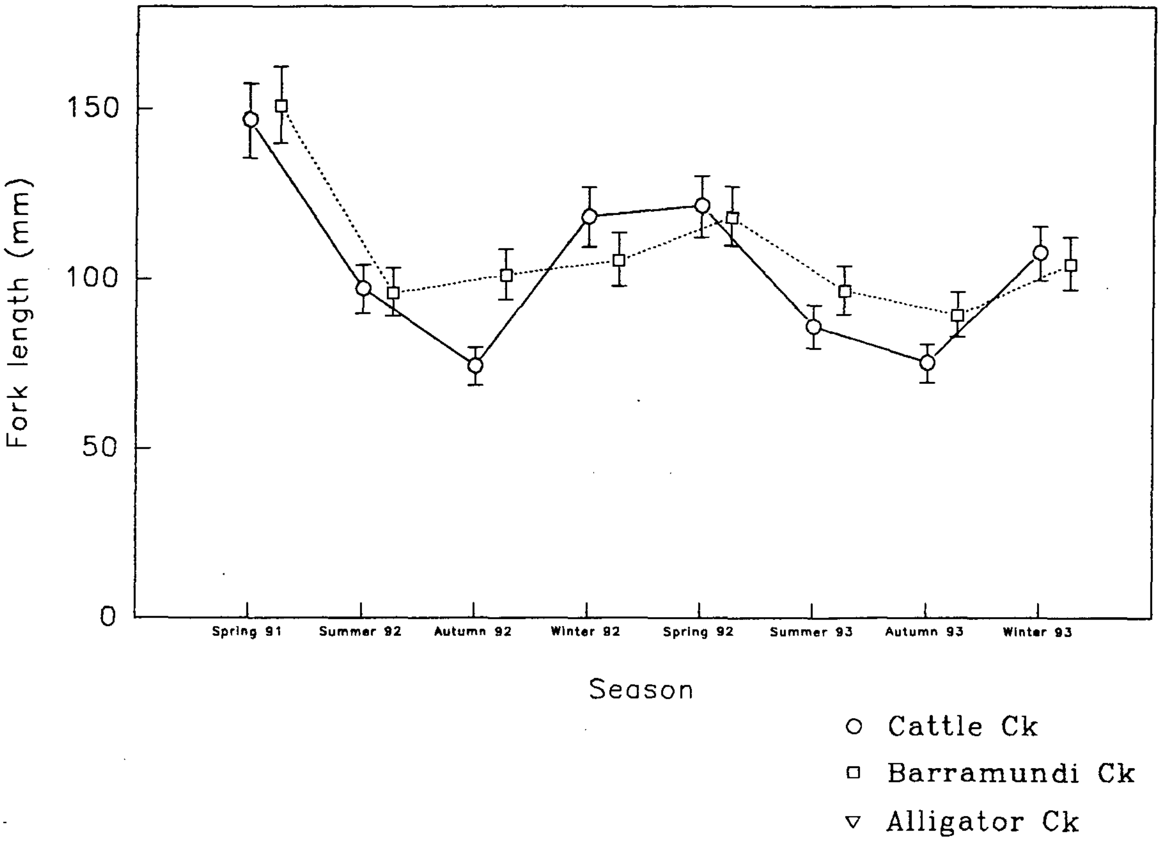


Figure 6.15

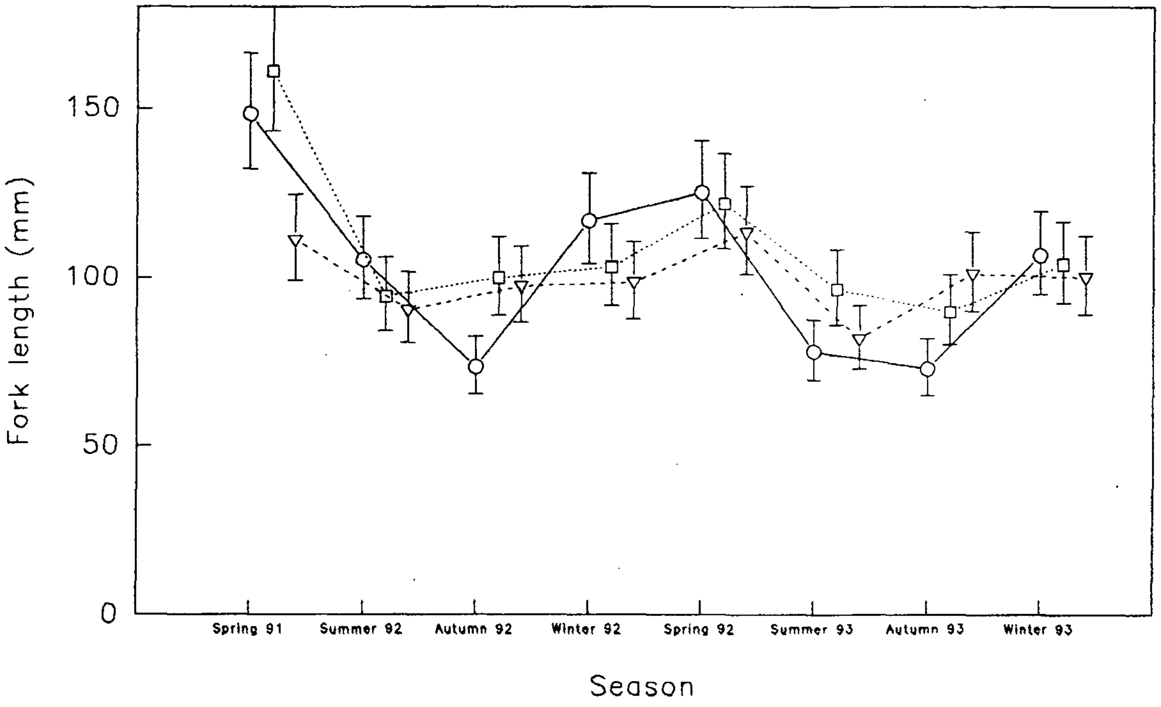


Figure 6.16. Relative size frequency of *L. russelli* in Cattle Creek over eight seasons.

Figure 6.16: Cattle Creek

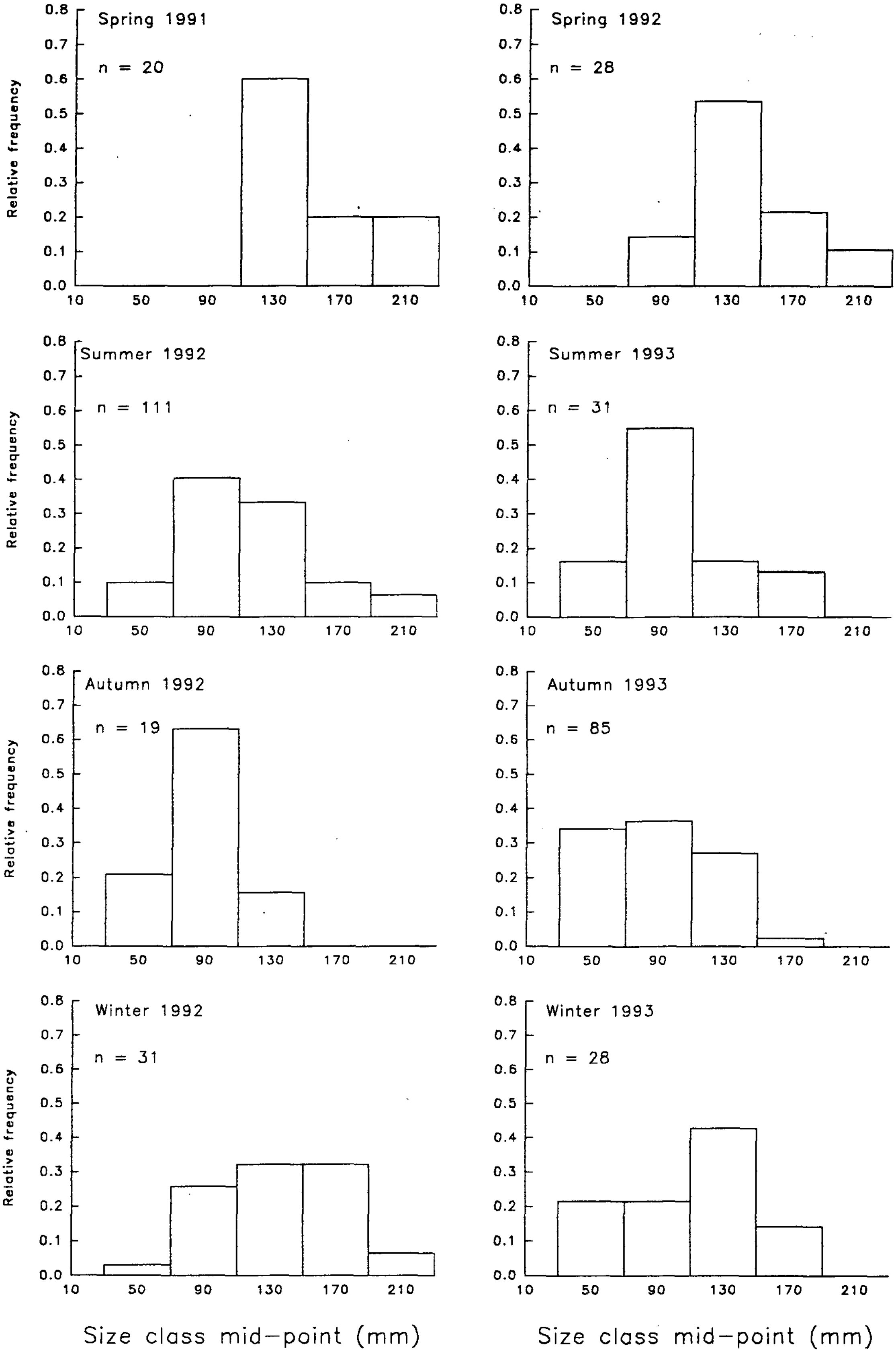


Figure 6.17. Relative size frequency of *L. russelli* in Barramundi Creek over eight seasons.

Figure 6.17: Barramundi Creek

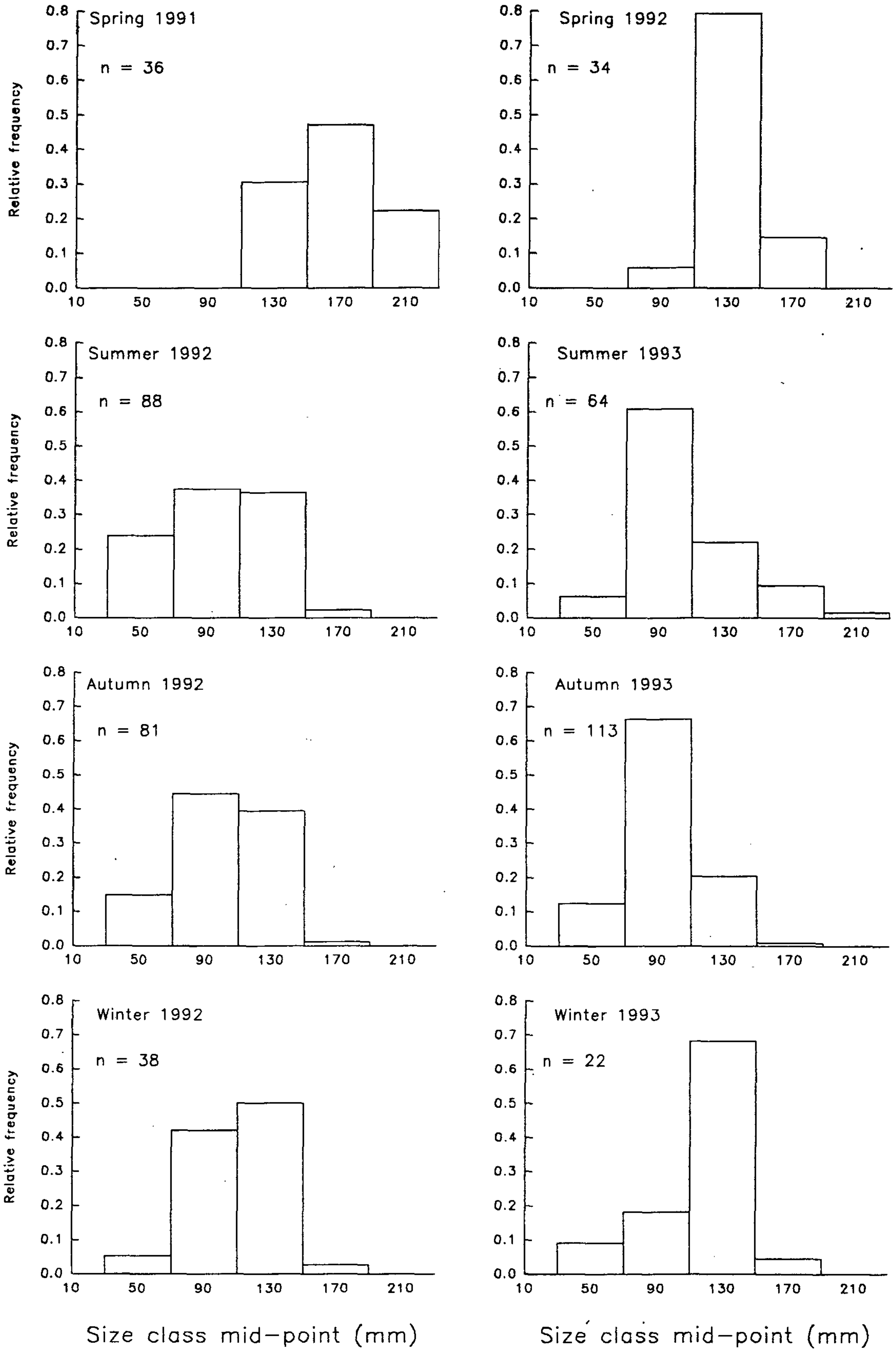
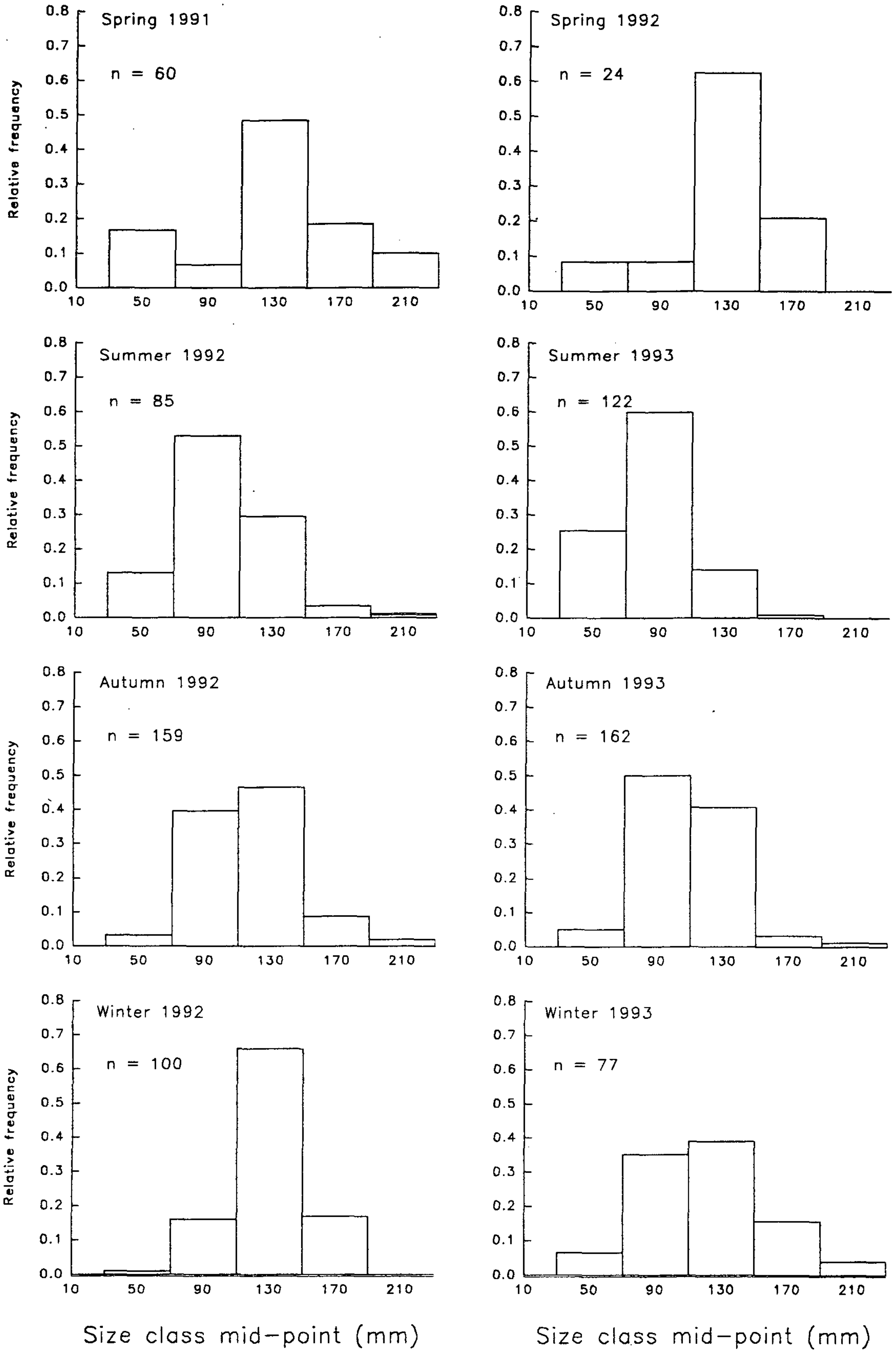


Figure 6.18. Relative size frequency of *L. russelli* in Alligator Creek over eight seasons.

Figure 6.18: Alligator Creek



generally with a strong mode in the 130mm size class. In Summer 1992 and Summer 1993 there was a sharp change with an increased proportion of small fish, a decreased proportion of large fish and the mode shifted back to the 90mm size class. In Autumn 1992 and Autumn 1993 the proportion of large fish remained low while the proportion of small fish remained high, with the mode in the 90 to 130mm size classes. The high proportions of small fish in Summer and Autumn correlate with high probabilities of capture in those months in Barramundi Creek (Fig. 6.9) and in the seaward regions of Cattle and Alligator Creeks. In Winter 1992 and Winter 1993 the distributions had moved back towards the situation in Spring 1991 and Spring 1992. A higher proportion of fish were in the large size classes although reasonable proportions of small fish were still present and the mode had shifted into the 130mm size class.

While the patterns of change are obvious and similar for the three creeks, details, such as the timing of the appearance of fish in the smaller size classes, differed. However, these "seasons" were actually offset by one month for the 3 creeks. Consequently, if these frequency distributions (Bottom and Lower regions only) are viewed as a 24 month sequence, disregarding creeks, the pattern of change becomes much more consistent (Fig. 6.19).

The 24 months of data constitute a very short time series only. Furthermore, the inclusion of data from three different creeks undoubtedly introduces additional variability. However despite this, the periodogram of mean fork lengths for the two lower regions for each month (Fig. 6.20) shows a pronounced peak corresponding to a 12 month cycle. The cumulative periodogram of the same data (Fig. 6.21) shows the maximum deviation of the data exceeds the 5% significance level of the Bartlett D statistic. This indicates that the cyclic pattern in the data is significant and not simply attributable to white noise.

6.3.3 **Physical data**

In all the creeks temperature varied by at least 10°C over the course of the study (Table 6.8a). Surface and bottom temperatures were always very similar with a maximum difference of about 2°C for any sampling site at any one time. The

Figure 6.19. Relative size frequency of *L. russelli* in Cattle, Barramundi and Alligator Creeks over two years (Bottom and Lower regions only).

Figure 6.19

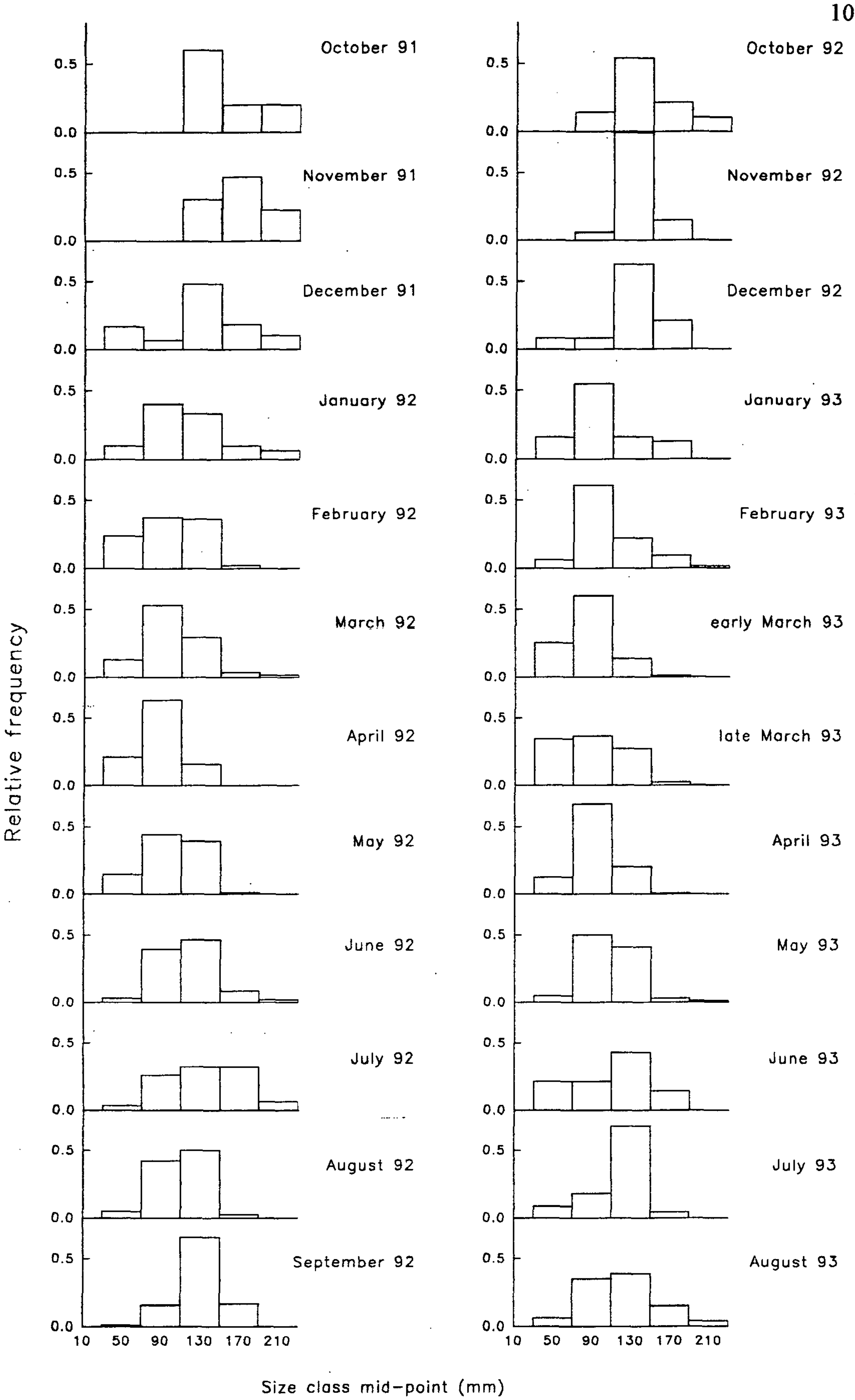

October 91
November 91
December 91
January 92
February 92
March 92
April 92
May 92
June 92
July 92
August 92
September 92
October 92
November 92
December 92
January 93
February 93
early March 93
late March 93
April 93
May 93
June 93
July 93
August 93
0.5
0.0
10
50
90
130
170
210
Relative frequency
Size class mid-point (mm)

Figure 6.20. Periodogram of mean fork length data for 24 consecutive lunar months, showing periodogram ordinates plotted against cycle length.

Figure 6.21. Cumulative periodogram of mean fork length data for 24 consecutive lunar months (solid line) with 5% critical value for the D statistic (dashed line).

Figure 6.20

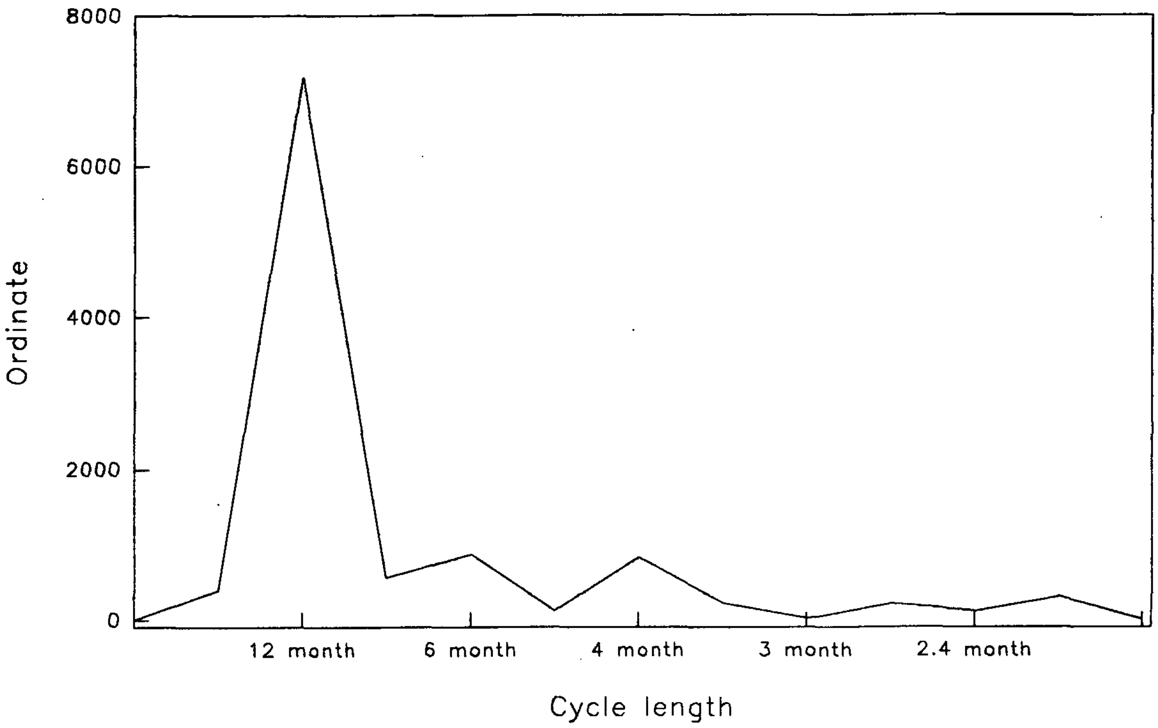
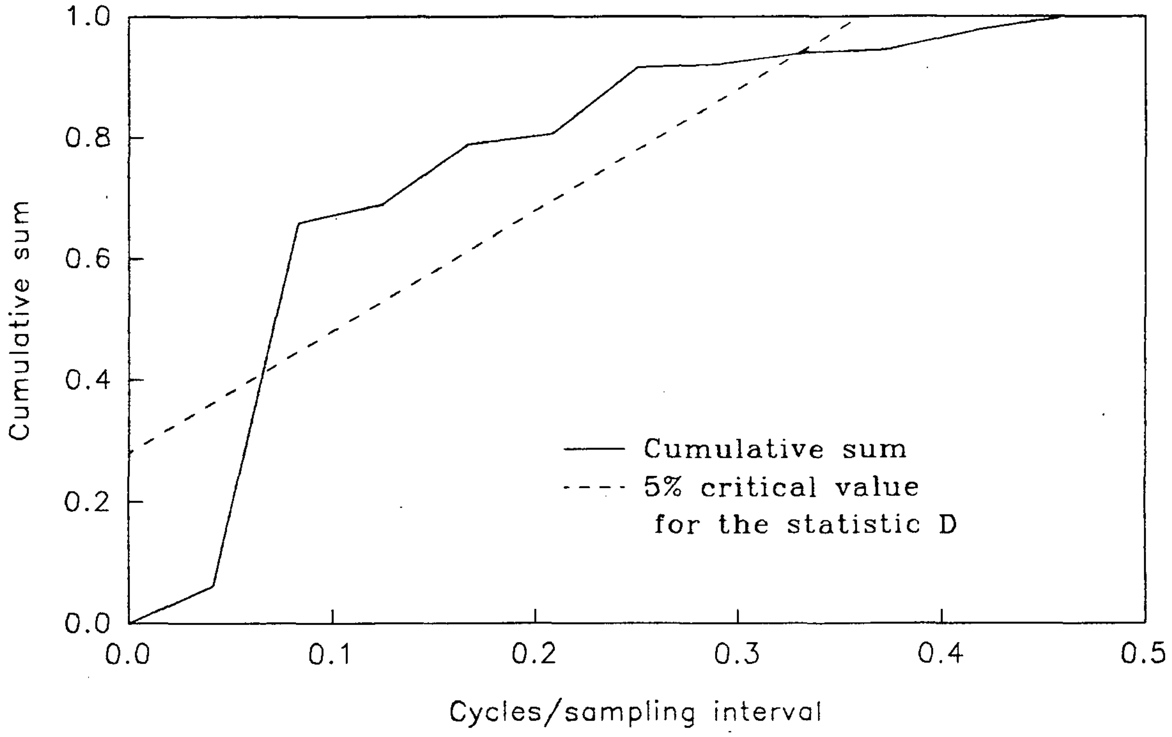


Figure 6.21



overall mean, minimum and maximum bottom and surface temperatures were similar for all three creeks.

Bottom temperatures were highest in Summer, and lowest in Winter. At any one time the mean temperature differed relatively little between sampling sites within an estuary (Fig. 6.22b)). The greatest differences in temperature generally occurred in the warmer months, with temperatures generally slightly higher in upper parts of the estuaries than towards their seaward ends.

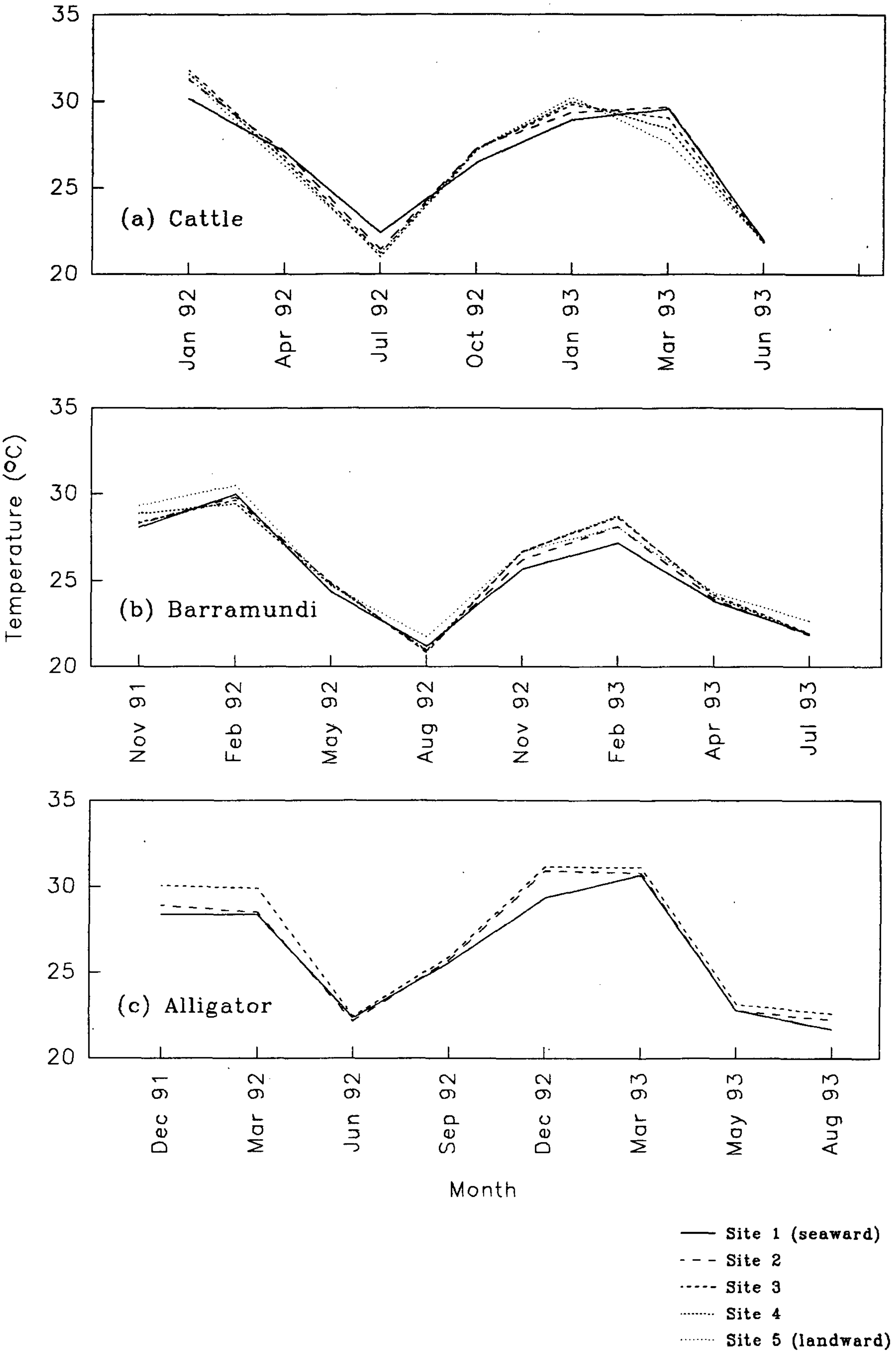
Table 6.8: Overall mean, minimum and maximum (a) temperatures and (b) salinities for Cattle, Barramundi and Alligator Creeks at daytime low tides during sampling trips over the course of the study.

(a)		Mean ± SE (°C)	Minimum (°C)	Maximum (°C)
Cattle				
	Bottom	26.6 ± 0.4	20.7	32.0
	Surface	26.8 ± 0.4	20.8	32.1
Barramundi				
	Bottom	25.2 ± 0.3	20.4	31.9
	Surface	26.0 ± 0.3	20.4	33.7
Alligator				
	Bottom	26.7 ± 0.4	21.7	31.7
	Surface	27.2 ± 0.5	21.8	32.2

(b)		Mean ± SE (‰)	Minimum (‰)	Maximum (‰)
Cattle				
	Bottom	27.6 ± 1.1	0.0	39.0
	Surface	23.9 ± 1.3	0.0	39.0
Barramundi				
	Bottom	41.6 ± 0.6	27.0	55.5
	Surface	41.1 ± 0.6	22.0	55.5
Alligator				
	Bottom	35.6 ± 0.8	14.0	45.0
	Surface	33.2 ± 1.1	8.5	45.0

Figure 6.22. Mean bottom temperatures in a) Cattle Creek, b) Barramundi Creek and c) Alligator Creek over the course of the study.

Figure 6.22


35
30
25
20
(a) Cattle
Jan 92
Apr 92
Jul 92
Oct 92
Jan 93
Mar 93
Jun 93
Temperature (°C)
(b) Barramundi
Nov 91
Feb 92
May 92
Aug 92
Nov 92
Feb 93
Apr 93
Jul 93
(c) Alligator
Dec 91
Mar 92
Jun 92
Sep 92
Dec 92
Mar 93
May 93
Aug 93
Month
Site 1 (seaward)
Site 2
Site 3
Site 4
Site 5 (landward)

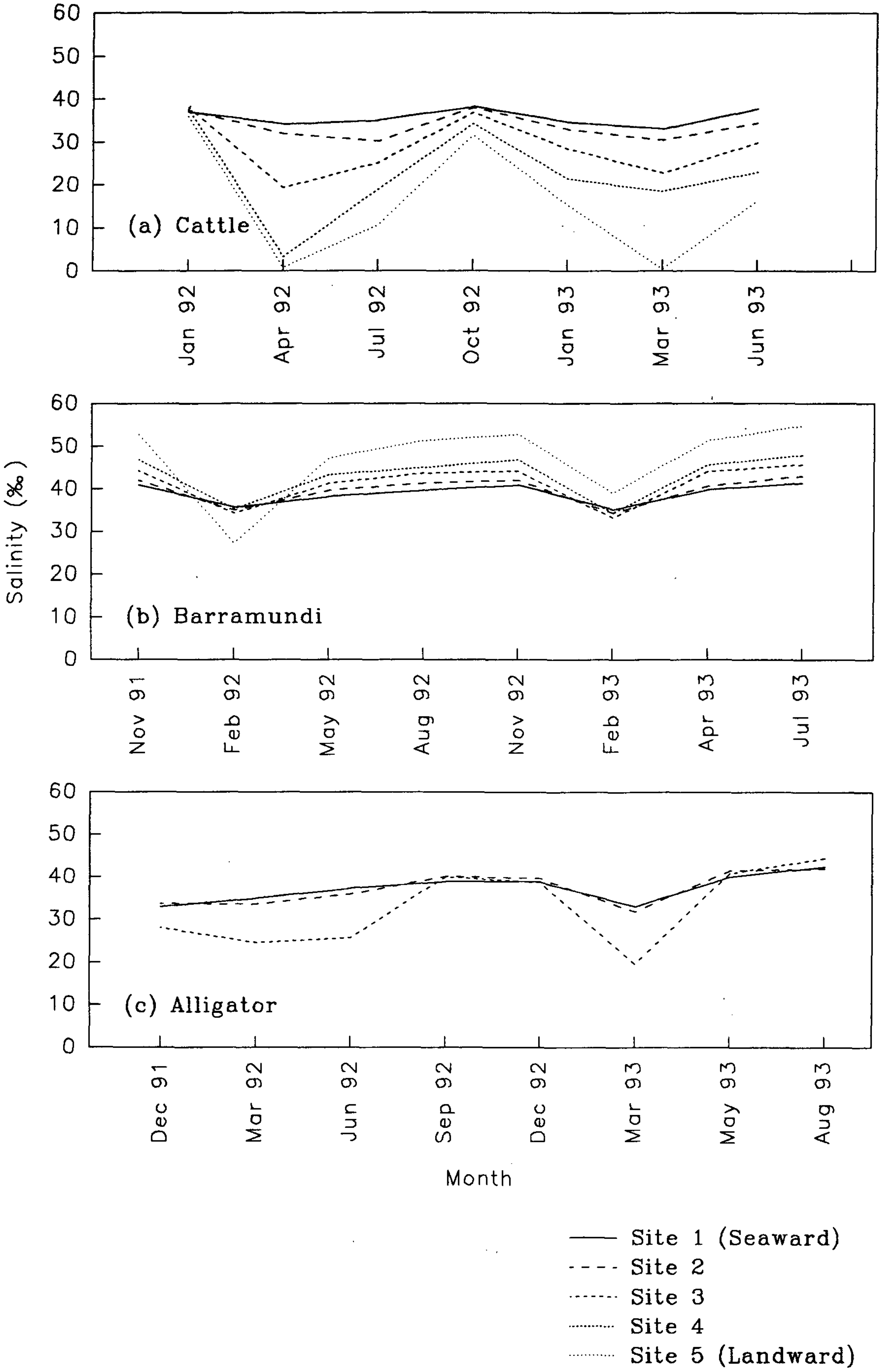
At each creek overall mean, minimum and maximum bottom and surface salinities (Table 6.8b) were similar, although surface means and minima tended to be somewhat lower than bottom values. Unlike temperature, salinity varied greatly between creeks. Cattle Creek exhibited low salinities, with both mean and minimum salinities well below 'normal' seawater levels [approx. 32‰ - 38‰; mean 35‰ (Kalle, 1971)]. This was due to continual freshwater inputs from stream flow and flooding in Autumn of each year. Lacking major freshwater inputs during the study period, Barramundi Creek exhibited high salinities, with both mean and maximum salinities well above 'normal' seawater levels. Alligator Creek's salinities fell between these extremes. While salinities in the three creeks differed substantially it should be noted that each demonstrated a wide range of salinities that extended both above and below 'normal' seawater salinity.

Cattle Creek demonstrated a marked pattern of seasonal change in bottom salinity (Fig. 6.23a). Highest salinities occurred at all sites in Summer and Spring 1992. Lowest salinities occurred in Autumn 1992 and 1993 and coincided with flooding subsequent to monsoon rainfall. During flooding both bottom and surface salinity levels in the upper parts of Cattle Creek fell to very low levels (close to 0‰). While surface salinities in the lower reaches fell to quite low levels (between about 10‰ and 20‰), bottom salinities (Fig. 6.23a) were only slightly depressed. With the exception of Summer 1992 when salinity was constant over the whole creek, throughout the study salinity levels were highest at the most seaward sampling sites and fell with movement upstream (Fig. 6.23a).

Barramundi Creek (Fig. 6.23b) demonstrated patterns of seasonal change and differences between sampling sites that were quite different to those of Cattle Creek. Barramundi did show falls in salinity in Summer 1992 and 1993, but these were much less pronounced than the changes seen in Cattle Creek. These reduced salinity levels did not coincide with periods of rainfall or flooding. Unlike Cattle Creek where salinity levels consistently fell in an upstream direction, in Barramundi Creek salinities consistently increased in an upstream direction, with the exception of the two periods of reduced salinity. Alligator Creek showed similar but less extreme patterns of salinity change to Cattle Creek (Fig. 6.23c). Salinities were generally lower in upstream areas and freshwater input from upstream areas caused reduced

Figure 6.23. Mean bottom salinities in a) Cattle Creek, b) Barramundi Creek and c) Alligator Creek over the course of the study.

Figure 6.23


Salinity (‰)
(a) Cattle
0
10
20
30
40
50
60
Jan 92
Apr 92
Jul 92
Oct 92
Jan 93
Mar 93
Jun 93
(b) Barramundi
0
10
20
30
40
50
60
Nov 91
Feb 92
May 92
Aug 92
Nov 92
Feb 93
Apr 93
Jul 93
(c) Alligator
0
10
20
30
40
50
60
Dec 91
Mar 92
Jun 92
Sep 92
Dec 92
Mar 93
May 93
Aug 93
Month
Site 1 (Seaward)
Site 2
Site 3
Site 4
Site 5 (Landward)

salinities during Summer of 1992 and 1993.

A comparison of the mean bottom salinity levels of the most seaward and most landward sampling sites in Cattle and Barramundi Creeks (Fig. 6.24) highlights the contrasting salinity regimes of the two estuaries. Salinity levels at the seaward sites of both creeks were quite similar, remaining reasonably constant at levels close to 'normal' seawater. These stable salinities, similar across creeks, correlates with similar probabilities of capture data of *L. russelli* in the seaward regions of the two creeks (Fig. 6.8). However, while the furthest upstream sites in Cattle Creek generally showed salinities markedly lower than normal seawater, upstream areas of Barramundi Creek generally showed salinities substantially above normal seawater levels (Fig. 6.24). These differences in salinity were greatest in Autumn 1992 and 1993 when catches in Cattle Creek were significantly lower than those in Barramundi Creek.

As bottom and surface values for both temperature and salinity were similar only the correlation of numbers and bottom salinity were investigated. There was no relationship between either numbers or $\log_{10}(n+1)$ and water temperature (Table 6.9). Their was a significant relationship between salinity and numbers trapped and an even stronger one when numbers were transformed to $\log_{10}(n+1)$. The significant relationship was largely the result of a lack of large catches at low (>20‰) salinities (Fig. 6.25).

Figure 6.24. Comparison of bottom salinities at the most seaward and most landward sampling sites in Cattle and Barramundi Creeks.

Figure 6.24

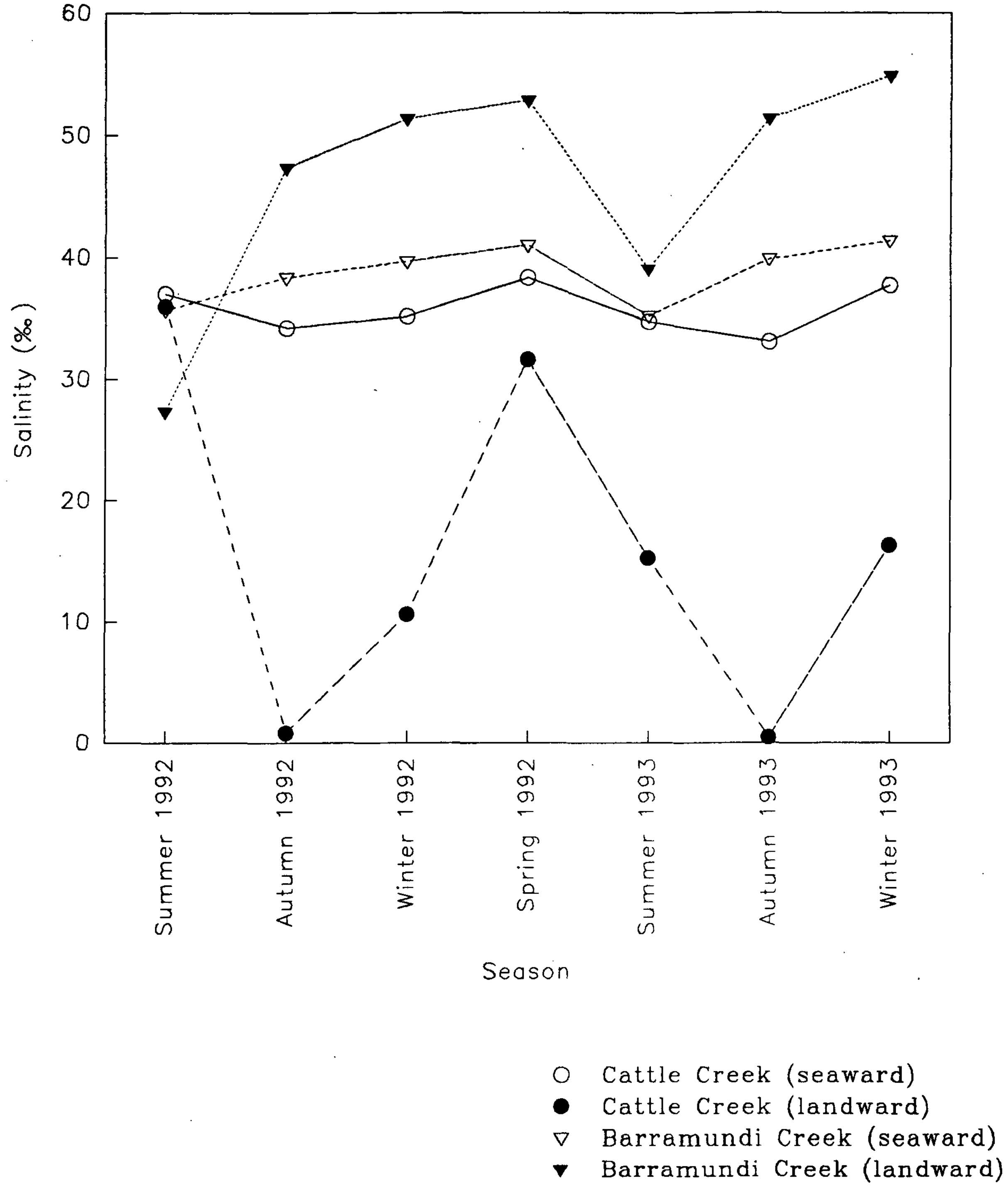

Salinity (‰)
0
10
20
30
40
50
60
Summer 1992
Autumn 1992
Winter 1992
Spring 1992
Summer 1993
Autumn 1993
Winter 1993
Season
○ Cattle Creek (seaward)
● Cattle Creek (landward)
▽ Barramundi Creek (seaward)
▼ Barramundi Creek (landward)

Figure 6.25. The correlation between numbers of *L. russelli* captured in fish-traps and bottom salinity. Numbers of *L. russelli* have been transformed by $\log_{10}(x + 1)$.

Figure 6.25

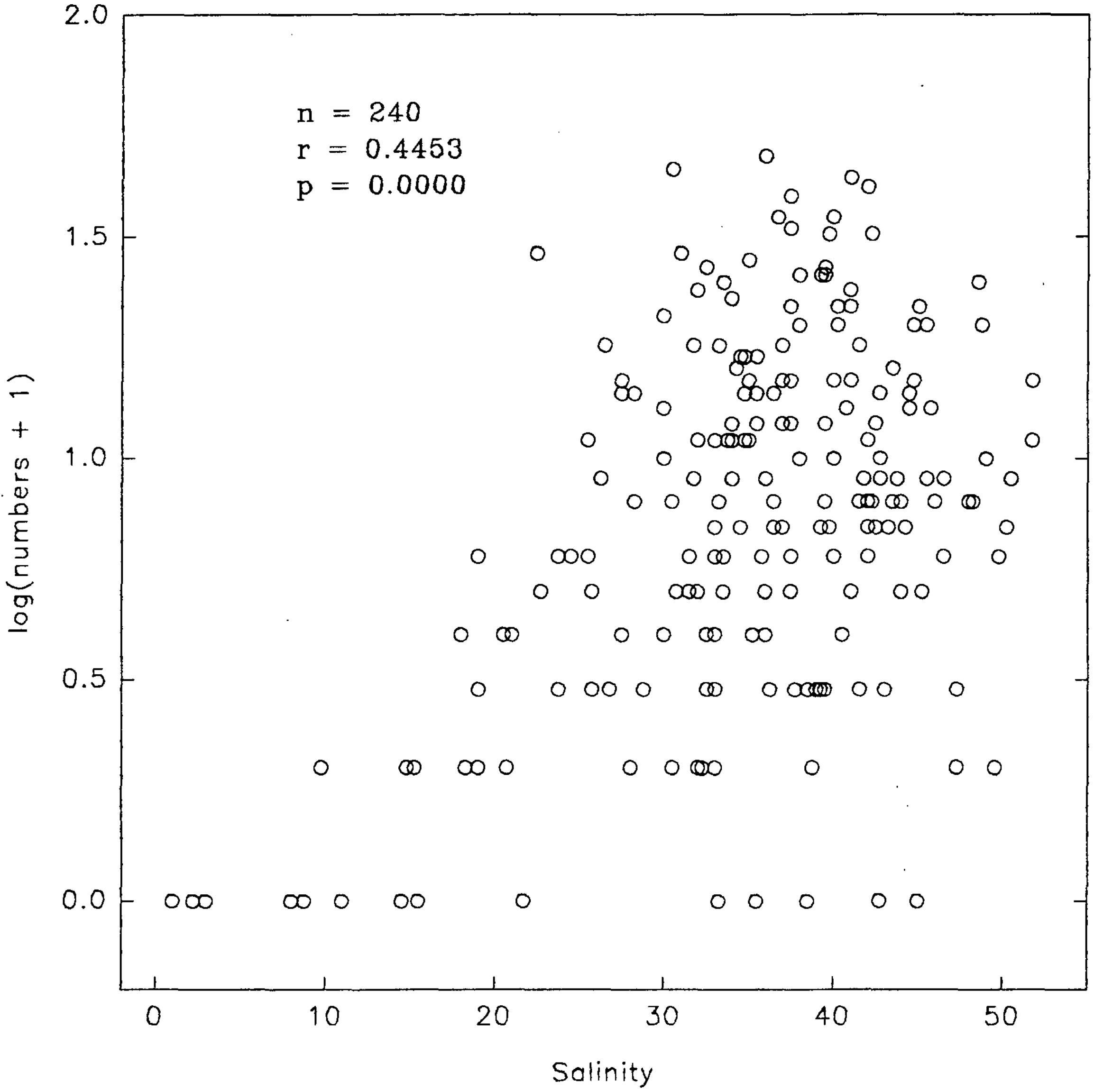

n = 240
r = 0.4453
p = 0.0000
2.0
1.5
1.0
0.5
0.0
log(numbers + 1)
0
10
20
30
40
50
Salinity

Table 6.9: Correlation between numbers [n] trapped and $\log_{10}(n+1)$ of *L. russelli*, and bottom salinity and temperature in Cattle, Barramundi and Alligator Creeks over the course of the study.

r = Pearson's product-moment correlation coefficient, n = sample size, p = significance level of Pearson's product-moment correlation coefficient.

		n	$\log_{10}(n + 1)$
Bottom temperature	r	0.0285	0.0496
	N	240	240
	p	0.66044	0.44436
Bottom salinity	r	0.2634	0.4453
	N	240	240
	p	0.00004	0.00000

6.4 DISCUSSION

Lutjanus russelli were the largest components of trap catches in the three estuaries sampled. They were trapped over a broad range of sizes, and were distributed throughout all three estuaries. The size range obtained from trapping probably reflects the range of sizes (above the minimum retention size of the traps) available in the estuaries. The largest *L. russelli* trapped (220mm FL) is comparable with the largest specimen obtained from any local estuaries (231mm FL) by any method (Chapter 3). Furthermore, when used in offshore waters, the same traps caught fish considerably larger than any trapped from estuaries (Table 2.2).

High mean numbers of *L. russelli* per trap within a region were associated with very high variances (Fig. 6.7). This suggested that the catch in a trap was not a random, representative sample from the region in which it was set. This was probably a result of schooling behaviour. Lutjanids, such as *L. russelli*, frequently form schools (Allen and Talbot, 1985). It is recognised that large catches in individual traps may occur due to conspecific behavioural interactions (Munro *et al.*, 1971). If one member of a school enters a trap a number of conspecifics often follow. This process has been termed conspecific attraction (Munro *et al.*, 1971). Thus, a numerically large catch may reflect the presence of a large school in the region of the particular trap rather than truly reflecting a greater abundance within an area as a whole. Because of this a mean, based on the numbers of *L. russelli* caught per trap, would probably be a biased estimator of the actual mean numbers in a region.

To overcome this a two-step analytical approach was used. Firstly, using logistic regression on the presence/absence component of the data, relative abundance was compared between treatment levels in terms of the probability of encountering at least one *L. russelli*. Secondly, for the traps that contained fish (i.e. the presence component), the number of fish caught per trap was compared between treatment levels using analysis of variance.

If the fish in each trap came from a single school and assuming that there was a positive correlation between school size and the numbers trapped, the analysis of numbers in traps with *L. russelli* present would provide direct information on school size. These assumptions are obviously very difficult to test. However, if they are reasonably well justified, analysing data by this two step process had clear advantages over a simple comparison of means. Firstly, if as was probably the case, catches of *L. russelli* were influenced by conspecific attraction, simply analysing mean numbers would clearly produce biased estimates. Trapping many fish from one large school could produce a high estimate of the mean number in an area, even though *L. russelli* might actually be in low abundance there. Secondly, with appropriate validation, the two step process has the potential to provide information on school size that is not available from a conventional comparison of means. If the catch in one trap was composed of fish from more than one school, interpretation of the analysis of mean numbers in traps that contained fish would be ambiguous. Notwithstanding this, if conspecific attraction did occur, due to its robustness the presence/absence approach would appear preferable to a standard analysis of variance comparison of means.

6.4.1 Spatial differences in abundance and size of *L. russelli*

The probability of capturing *L. russelli* (i.e. the probability of being present in a trap catch) was similar in the seaward regions of all three creeks (Fig. 6.8). However, probability of capture in upstream areas was markedly higher in Barramundi Creek than Cattle Creek. Furthermore, while the probability was similar for all regions of Barramundi Creek it dropped off markedly in the two upstream regions of Cattle Creek. The low probabilities of capture in the upper parts of Cattle Creek, may relate to the salinity regime there. In the summer of each year of the study, heavy rainfall caused the salinity of the upper parts of Cattle Creek to fall to very low levels (Fig. 6.23a). In fact during a major flood in Autumn 1992 when the lowest probabilities of capture were recorded (Fig. 6.9) salinities close to zero occurred. As *L. russelli* can be found in freshwater streams (Allen, 1985; Allen and Talbot, 1985) direct mortality due to low salinities seems unlikely. Despite this there are a number of ways in which freshwater input into upper parts of Cattle Creek might have reduced the probability of capture of *L. russelli*. Flood waters might have physically displaced *L. russelli*, in particular very small fish, to lower regions of the creek or reduced the number of recruits able to reach upper parts of the creek. Alternatively, low salinities may have made upper reaches less attractive to settling fish or reduced the availability of food.

An entirely different explanation for the decline in probability of capture in upstream areas of Cattle Creek must also be considered: The abundance of *L. russelli* may not have changed and low probabilities of capture may simply reflect behavioural differences at low salinities. This possibility is emphasised by the sharp decline in catch rates at low salinities (Fig. 2.25). However, as low salinities only occurred in the upper parts of Cattle Creek is not possible to determine if depressed catch rates and low probabilities of capture reflected changes in abundance or changes in behaviour. All the available methods of capturing *L. russelli* from deep-water areas of estuaries also suffer from the same potential for confoundment. Thus these two types of explanation cannot be differentiated. As a consequence, caution must be exercised when inferring changes in abundance from changes in the probability of capture.

The numbers of *L. russelli* trapped (in those traps containing fish) declined in an upstream direction (Fig. 6.10). This paralleled the reduction in probability of capture in upstream parts of Cattle Creek. Assuming that the fish in each trap came from a single school and that there was a positive correlation between school size and the numbers trapped, this reduction suggests that school sizes tended to become smaller towards upper parts of the estuaries.

While the probability of capture showed considerable spatial change, spatial differences in size were relatively minor. In general, *L. russelli* showed little indication of spatial segregation by size. However, mean fork lengths were clearly larger in the Bottom (most seaward) regions than the two middle regions of Cattle and Barramundi Creeks (Fig. 6.12). As the differences between regions was as much as 10% of the mean size it would seem to be biologically significant. This difference stemmed from a higher proportion of large fish in the bottom region (Fig. 6.13). While the reasons for this difference are unclear there are a number of possibilities. For example, the differences may have been due to the capture of large fish passing through the Bottom region during offshore migration. While mark-recapture studies suggest that *L. russelli* show little tendency to move within an estuary (Sheaves, 1993) [see Appendix II], the lack of large adult fish within the estuaries and the presence of large adults offshore (Chapter 3) means that *L. russelli* must undergo an offshore migration. It seems likely that the fish moving offshore would be those in larger size classes. Thus it may be that the increased mean length of fish in the bottom region was due to the capture of large fish from further upstream moving through the lower part of the estuaries during offshore migration. Alternatively, the differences may reflect responses to variations in the physical environment between regions. Upstream areas of Cattle and Barramundi Creeks tended to be hyposaline or hypersaline respectively compared to areas close to the sea, and thus probably present more challenging environments. The mean lengths in the top (most landward) regions of the Cattle and Barramundi Creeks were also larger than those in the lower region (Fig. 6.12). This difference had a different cause. Proportionally less small *L. russelli* were caught in the top than lower region. Apparently a different processes produced this difference. What this process might be is unclear.

6.4.2 Temporal differences in abundance and size of *L. russelli*

High probabilities of capture of *L. russelli* during summer and autumn in the whole of Barramundi Creek (Fig. 6.9) and in the two most seaward regions of all three creeks correlates with the time of smallest mean sizes (Fig. 6.13). This reflects the entry of large numbers of small recruits into the trappable population. Low probabilities of capture in winter and spring probably reflect a reduction in the size of the trappable population due to a combination of losses due to mortality and migration away from the estuaries.

Temporal patterns in mean length were consistent for both years and very similar for all creeks (Fig. 6.15). The major feature of the size data was a clear pattern of seasonal change in the mean fork length of *L. russelli*, with the largest mean sizes occurring in spring and the smallest mean sizes in summer and autumn. This pattern reflected the interaction of two processes - the growth of new recruits into the trappable population and migration of large fish from the estuaries. High proportions of large fish and low proportions of small fish occurred in catches in spring of both years (Fig. 6.19). This situation was apparently due to the growth of most of the previous year's recruits out of the smallest size classes and the presence of many large fish that had not yet migrated out of the estuaries. This pattern was reversed in summer and autumn when catches consisted of large proportions of small fish but small proportions of large fish. Apparently small fish from the most recent recruitment had reached a large enough size to be retained by the traps, while most of the larger fish present in summer had migrated out of the estuaries. Throughout the year reasonable proportions of fish in the middle size classes (about 130mm FL) were trapped. This probably stems from two factors. Firstly, as *L. russelli* appear to spend up to two years in the estuaries (Chapter 3), it would be expected that at any time fish of intermediate age and size would be present. Secondly, the long duration of the spawning period of *L. russelli* (Chapter 4) suggests that recruitment may occur over a large part of the year. As fish growth tends to be highly variable (Weatherly and Gill, 1987), differences in the timing of settlement would interact with variability in growth to make modes corresponding to different year classes difficult to distinguish. Furthermore, at particular times either fast growing early recruits or slow growing late recruits could contribute to the maintenance of reasonable numbers of fish in the middle size classes.

For Barramundi Creek both the patterns of probability of capture (Fig. 6.9) and size of catch in traps with *L. russelli* present were similar (Fig. 6.11). After flooding in autumn 1992 in Cattle Creek, both the probability of capture and the catch size fell to low levels. Subsequently, the catch size increased but the probability of capture remained low. If the catch size did reflect school size this suggests that *L. russelli* may have responded to flooding by forming larger schools, perhaps in areas where conditions were locally more favourable.

To simplify interpretation, data from three creeks sampled in consecutive months were grouped together as a season. It must be noted that this arbitrary grouping implicitly assumes that factors such as physical variation, habitat occupation and recruitment variability are equivalent over the three months.

6.4.3 **The physical environment of Cattle, Barramundi and Alligator Creeks**

The three creeks studied provided *L. russelli* with a harsh and extremely variable environment (Table 6.8). Temperatures varied more than 10°C during the study. Over the three creeks salinities ranged from very hyposaline (0‰) to extremely hypersaline (55.5‰) and varied up to 39‰ within a location (the upper parts of Cattle Creek). These levels of variation are similar to those reported from previous studies of estuaries in tropical Australia (Robertson and Blaber, 1992). Despite the obvious physiological demands of existing under such varying conditions, *L. russelli* were trapped over the full range of variation. Furthermore, given the restricted movements of *L. russelli* within estuaries (Sheaves, 1993) [see Appendix II], individual fish must have faced major salinity variations. Thus at the individual, population and species levels, *L. russelli* must possess formidable abilities to meet the demands of such a challenging environment.

Even though Cattle and Barramundi Creeks had very contrasting salinity regimes, these differences were mainly confined to upstream areas (Fig. 6.24). Salinities in the most seaward locations differed little. These differences were reflected in the probabilities of capture with clear differences between the two creeks confined to upstream areas. This suggests that salinity, or some correlated variable such as flushing by flood waters, is important in influencing the population dynamics of *L. russelli*. Salinity is thought to be a major factor structuring fish communities in both temperate and tropical estuaries (Quinn, 1980).

6.4.3 **Conclusion**

During the study major spatio-temporal variations in physical parameters, particularly salinity, occurred. To some extent the size structure and overall probability of capture of *L. russelli* seemed to reflect this variability. The data for Cattle Creek, particularly for the low salinity upper reaches, suggested reduced abundances and some modification in size structure compared to other areas. However, despite substantial physical variability, the major feature of the data was a clear pattern of annual change in the size structure of *L. russelli* that seemed to relate more to biological processes of recruitment, mortality and migration than to responses to physical variability.

Clearly, *L. russelli* populations must deal with considerable fluctuations in salinity and temperature as facts of life in estuaries in tropical northern Australia. Investigation of the stresses imposed on *L. russelli* by these levels of environmental fluctuation and the processes and mechanisms used to accommodate such physical variability would seem a fruitful line of future research. Of particular interest would be a comparison of the physical tolerances of *L. russelli*, *L. argentimaculatus* and other estuarine lutjanids to the tolerances of lutjanids inhabiting offshore waters exclusively.

CHAPTER 7

SPATIAL AND TEMPORAL PATTERNS IN THE DISTRIBUTION, ABUNDANCE AND SIZE OF *E. COIOIDES* AND *E. MALABARICUS* IN THREE ESTUARIES IN TROPICAL AUSTRALIA

7.1 INTRODUCTION

Serranid fishes of the genus *Epinephelus* occur in estuaries throughout tropical Australia (Blaber *et al.*, 1985; Collette, 1983; Blaber *et al.*, 1989; Sheaves, 1992; 1993; 1994). Although much confusion has existed in the classification of fishes of this genus (Randall and Ben-Tuvia, 1983; Randall, 1987; Randall and Heemstra, 1991) it appears that most reports of *Epinephelus* from this area refer to two species, *E. coioides* and *E. malabaricus* (Chapter 1.3.1). These species occur to sizes of 400mm FL or more within estuaries in tropical Australia (Sheaves, 1992; 1993; 1994).

The distribution of *E. coioides* and *E. malabaricus* within and between estuaries has received little study. However, Sheaves (1992) used fish traps to compare catches of *E. coioides* and *E. malabaricus* in three regions of Alligator Creek in north-eastern Australia. He found quite different patterns of abundance for the two species. Catches of *E. coioides*, the dominant serranid in downstream areas, decreased in upper parts of the creek. *E. malabaricus* showed the opposite pattern with numbers increasing from the lower parts of the creek to upstream areas, where this species was the dominant serranid. If this change in dominance is a general pattern in estuaries in tropical Australia, it suggests underlying differences in the biology of the two species, possibly in the way they respond to physical conditions in estuaries.

In this study the distribution, abundance and size structure of *E. coioides* and *E. malabaricus* within and between three estuaries in tropical north-eastern Australia was investigated. The distributions and abundances of the two species were compared and related to variations in temperature and salinity.

7.2 METHODS

The size structure and distribution of *Epinephelus coioides* and *Epinephelus malabaricus* were studied at Cattle, Barramundi and Alligator Creeks between October 14, 1991 and August 25, 1993. Data were collected using Antillean Z fish traps in the sampling design described for *L. russelli* (Chapter 6.2).

7.2.1 Data analysis

Previous studies (Sheaves 1992, 1993, 1994) suggested that *E. coioides* and *E. malabaricus* would be trapped in low numbers only. Thus it was decided *a priori* to pool data over sites, soaks and days. Data from day and night soaks were pooled also.

As with *L. russelli* the data for *E. coioides* and *E. malabaricus* were analysed in two phases; all regions of Cattle and Barramundi Creeks (analysis I), and the lower 2 regions of all three creeks (analysis II).

7.2.1.1 Size

Again, as with *L. russelli*, the preferred option for the analysis of the size of *E. coioides* and *E. malabaricus* was log-linear modelling of the data grouped into size classes. However, the pattern of zero data meant that, to avoid a contingency table with a number of marginal zeros, size classes would need to be pooled to such an extent that real trends in the data could be obscured. Therefore, as with *L. russelli*, analysis of variance (ANOVA) was used to investigate changes in the mean size (fork length) of *E. coioides* and *E. malabaricus*. The factors analysed (Seasons, Creeks, Regions) were considered fixed and orthogonal to each other.

As expected, many catch rates were low. Over the whole study only 20 *E. coioides* were trapped in Cattle Creek with only 1 of those coming from the Top and Upper regions. For the winter season in 1992 only 1 *E. coioides* was trapped in the whole of Barramundi Creek and for 2 other seasons (Summer 1992; Spring 1992) no individuals were trapped in the Top region. For the rest of the study only 10 *E.*

coioides were caught in the Top region of Barramundi Creek. Due to this paucity of data, Cattle Creek and Winter season in 1992 were omitted from analysis I, and the size of *E. coioides* analysed for the 3 lower regions of Barramundi Creek only. For analysis II data were analysed from Barramundi and Alligator Creeks for all seasons except Winter 1992.

On 4 sampling trips to Cattle Creek and 2 trips to Barramundi Creek, no *E. malabaricus* were trapped in the Top region. Because of this the Top region was excluded from analysis I. In Winter 1993 no *E. malabaricus* were trapped in the Bottom region of Barramundi Creek or the Lower region of Alligator Creek so this season was omitted from analysis II.

As *E. coioides* and *E. malabaricus* were usually caught singly no test for independence of fish size within traps was appropriate (see chapter 6).

Plots of residuals against predicted values and normal probability plots showed no heterogeneity of variance or non-normality in the data for *E. coioides* or the data for *E. malabaricus* from Cattle and Barramundi Creeks. However, in the data for *E. malabaricus* from the 2 lower regions of all creeks there was a tendency for the residuals to increase with the mean indicating heterogeneity of variance. Therefore a square root transformation was applied to these *E. malabaricus* data and this resulted in a distribution of residuals that showed no evidence of heterogeneity of variance.

7.2.1.2 Distribution and Abundance

To enable the abundances of *E. coioides* and *E. malabaricus* to be compared the numbers of the two species were analysed together. In a number of regions in a number of months no fish of one or other of the species was trapped, therefore the data were pooled over seasons and only the spatial factors (Creek and Region) analysed. The data in the form of a 3-way contingency table (Species*Creek*Region) were analysed using log-linear models (Appendix I.1).

7.2.1.3 Physical variables

Bottom and surface temperatures and salinities were collected as described in Chapter 6.2.3. As bottom and surface values were highly correlated only bottom values were investigated. These were thought to better represent the physical conditions the fish would have been exposed to. The mean bottom temperature and salinity for each region of each creek on each day were calculated as described in Chapter 6.2.3. The absolute value of the maximum deviation in salinity (MSD) away from mean 'normal' seawater salinity [35‰ (Kalle, 1971)] was calculated for each region of each creek. Pearson's product-moment correlation coefficients and their associated significance levels were calculated to investigate the relationship between the catches of *E. coioides* and *E. malabaricus* (in each region of each creek on each day), and bottom temperature, bottom salinity and MSD. Graphical analysis was used to investigate the relationship between the mean numbers of *E. coioides* and *E. malabaricus* trapped per day in each region of each creek and the bottom salinity level associated with the MSD.

7.3 RESULTS

7.3.1: Size

During the course of the study 280 *E. coioides* between 120mm and 500mm fork length and 334 *E. malabaricus* between 122mm and 619mm fork length were trapped in Cattle, Barramundi and Alligator Creeks (Table 7.1).

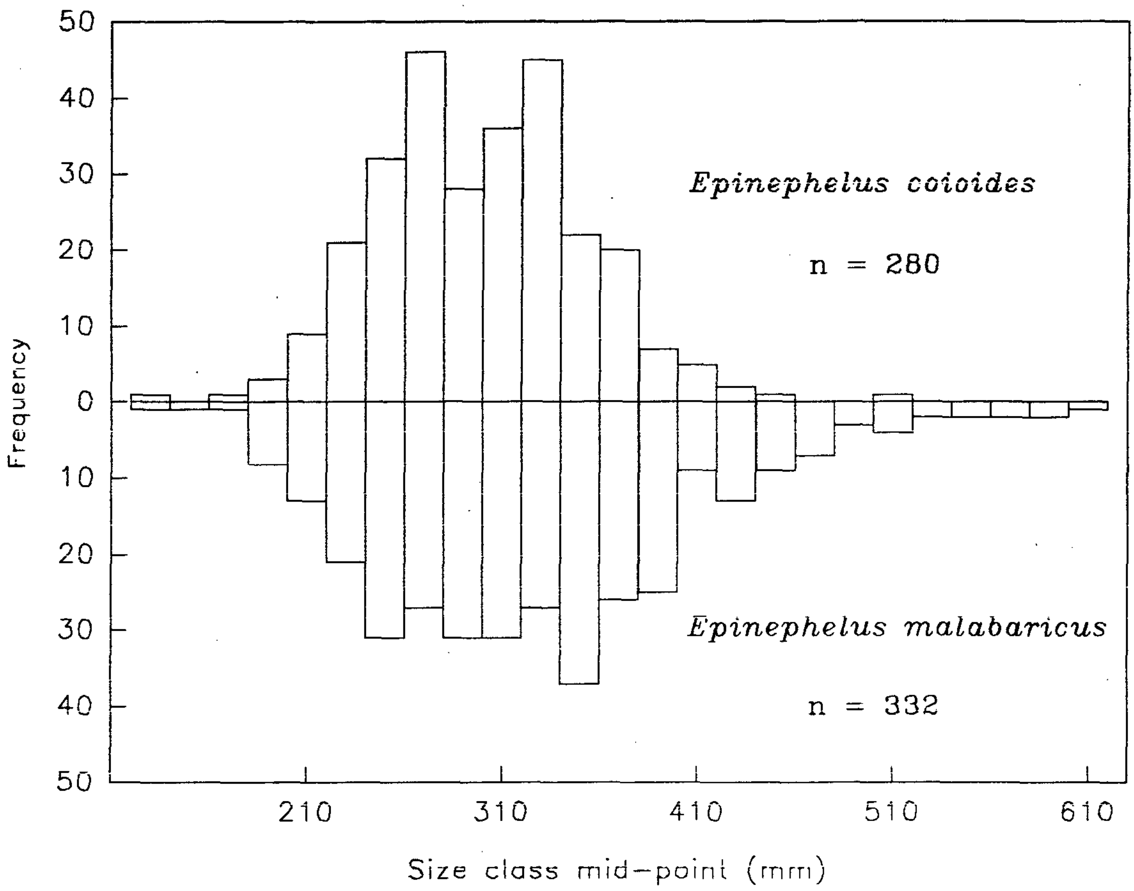
Table 7.1: Summary of fish trap catches of *E. coioides* and *E. malabaricus* in Cattle, Barramundi and Alligator Creeks between 1991 and 1993.

Season	Creek	*E. coioides*	*E. malabaricus*
Spring 1991	Cattle	1	7
	Barramundi	15	26
	Alligator	24	13
Summer 1992	Cattle	2	17
	Barramundi	13	21
	Alligator	26	16
Autumn 1992	Cattle	1	12
	Barramundi	15	16
	Alligator	19	10
Winter 1992	Cattle	4	12
	Barramundi	1	18
	Alligator	9	9
Spring 1992	Cattle	2	4
	Barramundi	11	10
	Alligator	12	9
Summer 1993	Cattle	4	12
	Barramundi	25	18
	Alligator	16	14
Autumn 1993	Cattle	5	19
	Barramundi	31	44
	Alligator	10	10
Winter 1993	Cattle	1	6
	Barramundi	22	10
	Alligator	11	1
Total		280	334

The overall size structures of *E. coioides* and *E. malabaricus* (Fig. 7.1) displayed some differences. The maximum size of *E. coioides* (500mm) was much smaller than that of *E. malabaricus* (619mm). Furthermore, for *E. coioides* only 3.2% of fish were greater than 400m and only one fish over 460mm, while for *E.*

Figure 7.1: Comparison of size structures (fork length) of *E. coioides* and *E. malabaricus* from estuaries.

Figure 7.1


50
40
30
20
10
0
10
20
30
40
50
Frequency
Epinephelus coioides
n = 280
Epinephelus malabaricus
n = 332
210
310
410
510
610
Size class mid-point (mm)

malabaricus 16.2% of fish were greater than 400mm. While both species showed a broad mode beginning at around 250mm, the mode for *E. malabaricus* extended to about 390mm, while for *E. coioides* the mode extended to about 330mm only. The lower ends of the distributions were similar for both species falling off rapidly from about 250mm to a minimum of about 120mm.

There were no clear differences between the fork lengths of *E. coioides* for any of the variables (Table 7.2) for analysis I [3 seaward regions of Barramundi Creek] or analysis II [the two seaward regions of Barramundi and Alligator Creeks], although there was weak evidence of differences between seasons in analysis II.

Table 7.2: Analysis of variance tables for the mean fork length of *E. coioides* in two estuaries. (a) Three regions in Barramundi Creek [analysis I], (b) Two regions in Barramundi and Alligator Creeks [analysis II]. [-] indicates factors not analysed.

	(a)			(b)		
.Source of variation	d.f of F ratio	F	P	d.f. of F ratio	F	P
Creek	-	-	-	1/190	1.32	0.252
Region	2/101	0.46	0.634	1/190	0.01	0.938
Season	6/101	1.78	0.111	6/190	2.07	0.059
Creek*Region	-	-	-	1/190	2.04	0.155
Creek*Season	-	-	-	6/190	0.52	0.796
Region*Season	12/101	1.16	0.321	6/190	1.94	0.076
Creek*Region*Season				6/190	0.61	0.722

For *E. malabaricus* season had an independent effect on fork length for both analysis I [3 seaward regions of Cattle and Barramundi Creeks] and analysis II [2 seaward regions of all 3 creeks] (Table 7.3). Creek and region interacted strongly in analysis I. While there was a very strong creek effect in analysis II there was no evidence of differences due to region (Table 7.3).

Table 7.3: Analysis of variance tables for the fork length of *E. malabaricus* in three estuaries. (a) Three regions in Cattle and Barramundi Creeks [analysis I], (b) Two regions in Cattle, Barramundi and Alligator Creeks [analysis II].

Shading indicates significant effects.

	(a)			(b)		
Source of variation	d.f of F ratio	F	P	d.f. of F ratio	F	P
Creek	1/171	0.35	0.556	2/189	14.77	0.000
Region	2/171	0.95	0.389	1/189	0.321	0.578
Season	7/171	2.46	0.020	6/189	2.865	0.011
Creek*Region	2/171	7.72	0.001	2/189	1.336	0.265
Creek*Season	7/171	1.34	0.234	12/189	1.263	0.244
Region*Season	14/171	1.42	0.148	6/189	0.592	0.726
Creek*Region*Season	14/171	1.09	0.370	12/189	0.695	0.755

While the mean fork length of *E. malabaricus* in Cattle and Barramundi Creeks differed with season (Fig. 7.2), there was no clear pattern; each of the three peaks of fork length occurred in different seasons. Size frequency plots of *E. malabaricus* over the 8 seasons also lacked any consistent patterns, as did fork lengths in the 2 lower regions of Cattle, Alligator and Barramundi Creeks.

The mean fork length of *E. malabaricus* was clearly larger in the Bottom and Lower regions of Barramundi Creek than in the corresponding regions of Cattle Creek (Table 7.3a; Fig. 7.3). However, in the Upper region the mean fork length was greater in Cattle Creek than in Barramundi Creek [Tukey's Test: $q_{171,6} = 4.247$, $p < 0.05$]. Size frequency plots for this interaction (Fig. 7.4) showed that in the regions with small mean sizes [Bottom and Lower regions of Cattle Creek and Upper region of Barramundi Creek] a small proportion of large *E. malabaricus* were trapped, compared to the Bottom and Lower regions of Barramundi Creek and the Upper region of Cattle Creek. When the 2 lower regions of Cattle, Alligator and Barramundi Creeks (Table 7.3b) were compared the mean fork length was larger in Barramundi Creek than in either Cattle Creek or Alligator Creek where the mean sizes were similar (Fig. 7.5). Size frequency comparisons between creeks (Fig. 7.6) showed that

Figure 7.2: Differences in fork length of *E. malabaricus* over 2 years. Data are means ±95% confidence intervals.

Figure 7.3: The effect of the interaction between creek and region on the fork length of *E. malabaricus* in Cattle and Barramundi Creeks. Data are means ±95% confidence intervals.

Figure 7.2

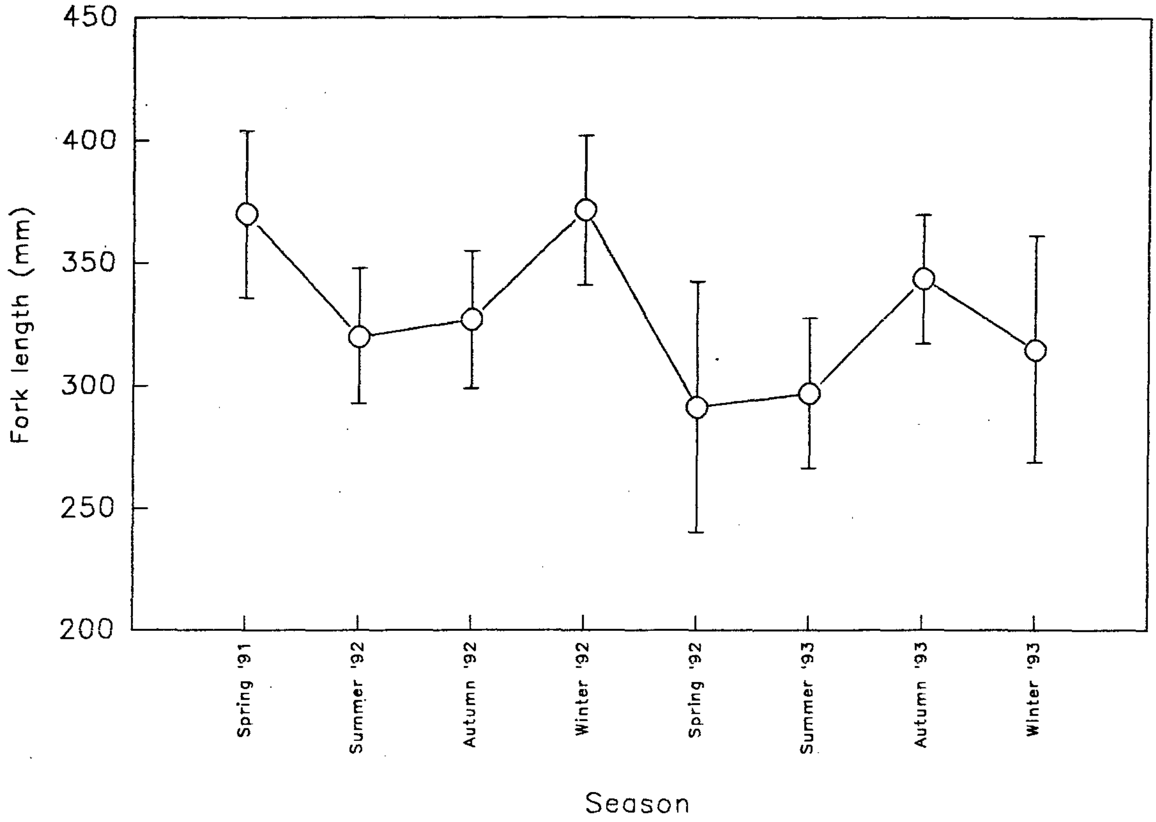


Figure 7.3

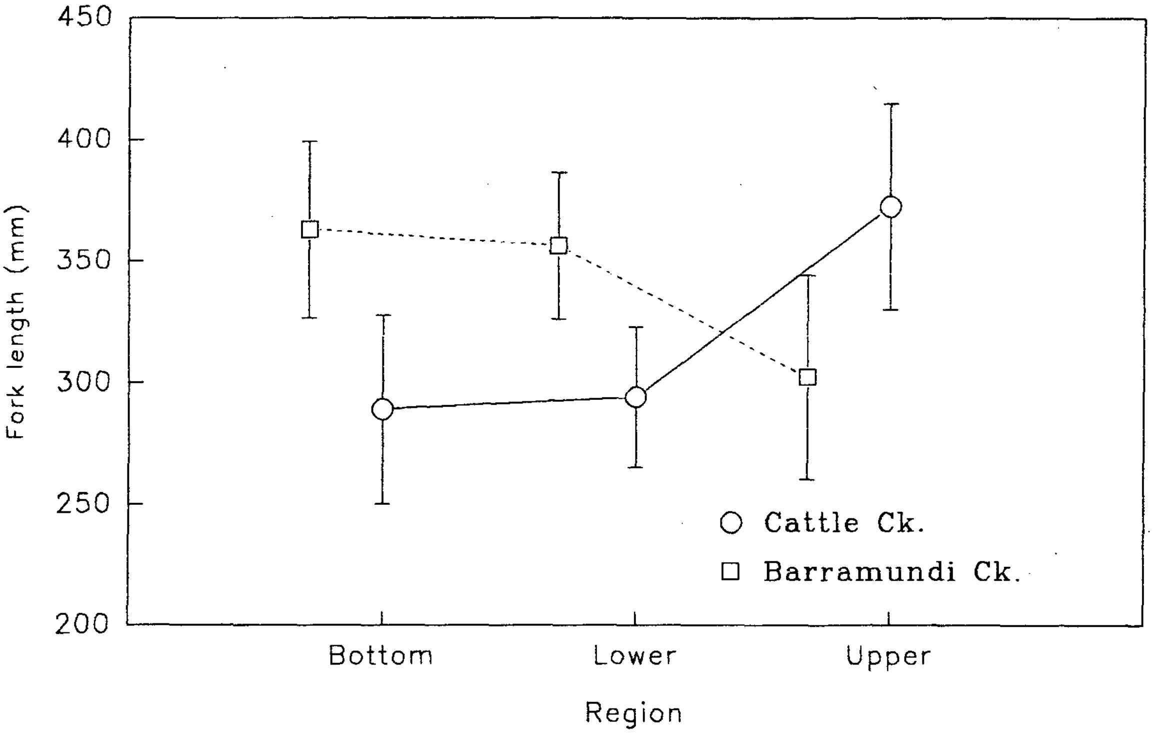


Figure 7.4: Size frequency plots for the interaction between creek and region on the fork length of *E. malabaricus* shown in figure 7.3.

Figure 7.4

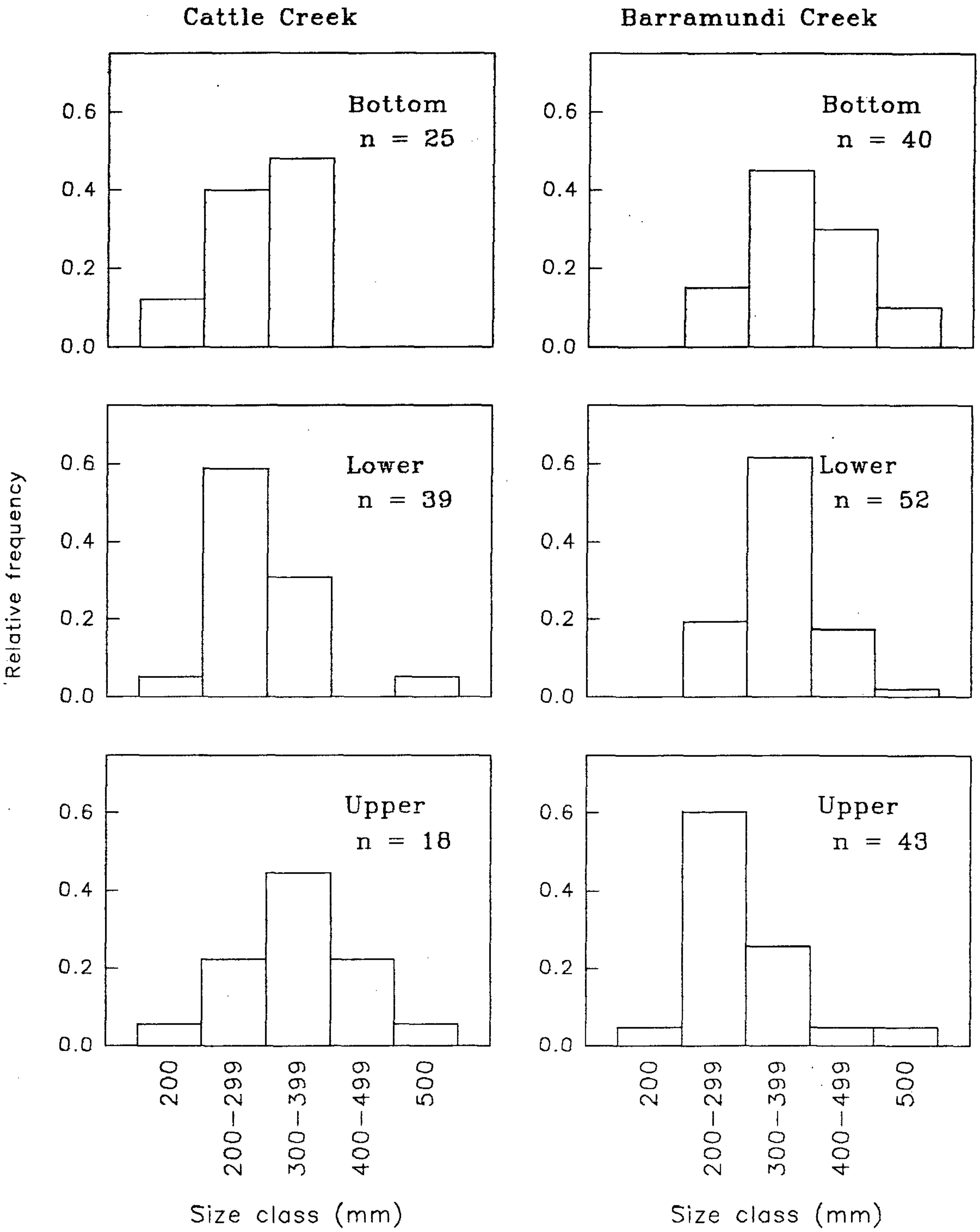

Cattle Creek
Barramundi Creek
Bottom
n = 25
Bottom
n = 40
Lower
n = 39
Lower
n = 52
Upper
n = 18
Upper
n = 43
Relative frequency
0.0
0.2
0.4
0.6
200
200–299
300–399
400–499
500
Size class (mm)

Figure 7.5: Comparison of the fork lengths of *E. malabaricus* in the two seaward regions of Cattle, Barramundi and Alligator Creeks. Data are means ±95% confidence intervals.

Figure 7.5

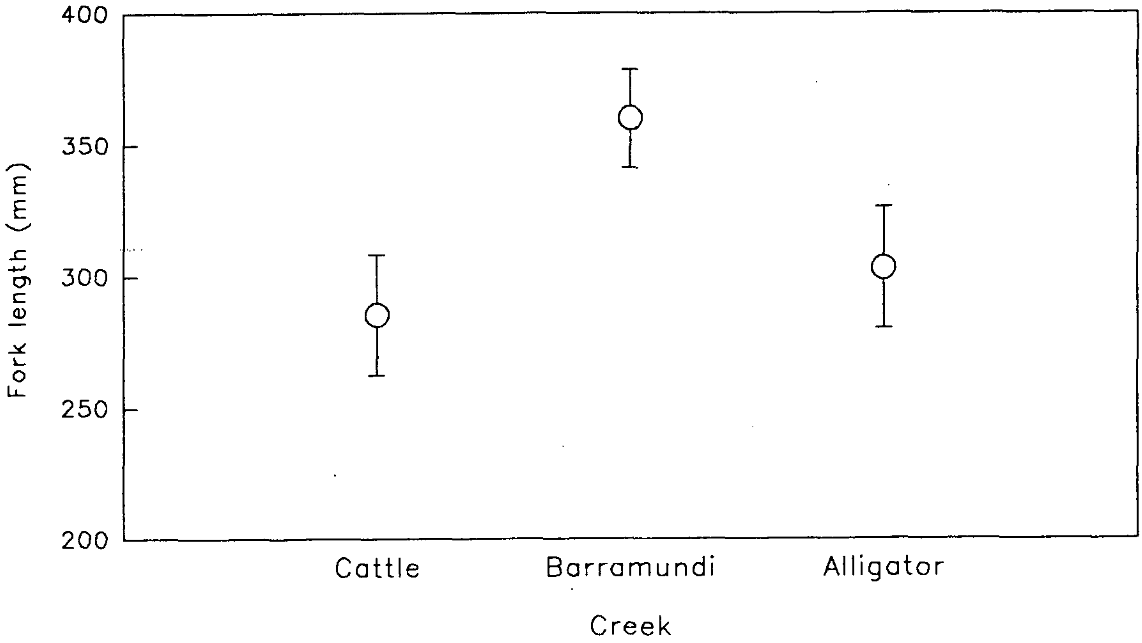
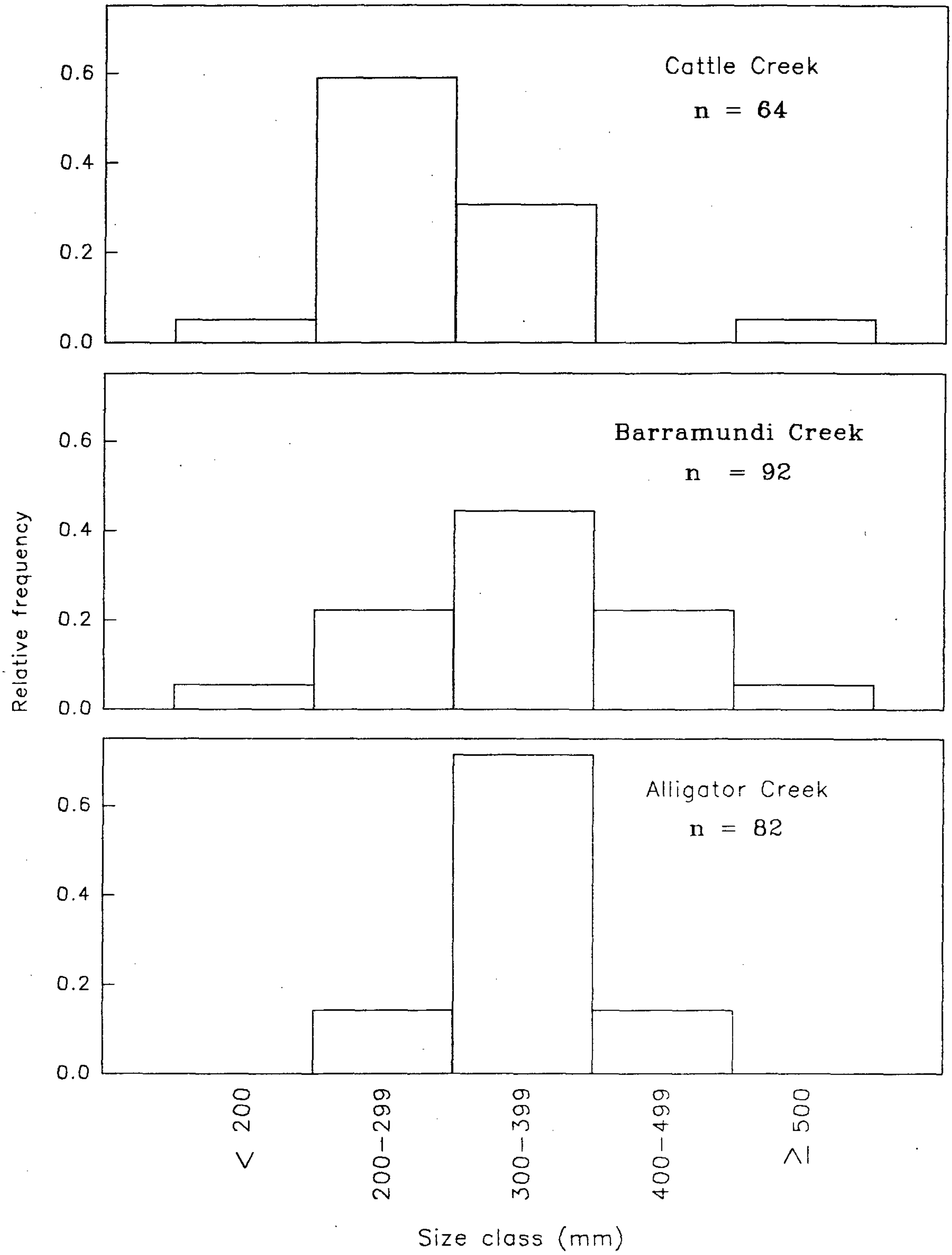

400
350
300
250
200
Fork length (mm)
Cattle
Barramundi
Alligator
Creek

Figure 7.6: Size frequency plots for the comparison of the fork lengths of *E. malabaricus* shown in figure 7.5.

Figure 7.6


Cattle Creek
n = 64
Barramundi Creek
n = 92
Alligator Creek
n = 82
0.6
0.4
0.2
0.0
Relative frequency
< 200
200–299
300–399
400–499
≥ 500
Size class (mm)

while Cattle and Barramundi Creeks produced a similar range of sizes the modal size was larger in Barramundi Creek with a higher proportion of fish in the larger size classes. While the modal fork length was similar in Barramundi and Alligator Creeks, Barramundi Creek had a greater proportion of large fish.

7.3.2 Analysis of the Distribution and Abundance of *E. coioides* and *E. malabaricus*

For both the comparison of numbers in 4 regions of Cattle and Barramundi Creeks [analysis I] and the comparison of numbers in 2 regions of Cattle, Barramundi and Alligator Creeks [analysis II] log-linear modelling yielded the same final model (Table 7.4). This was composed of 2-way interactions between each of the 3 factors. The numbers of the 2 species differed between both Creeks and Regions independently, and Creeks and Regions interacted in the same way for both species.

For the Species by Creek interaction in the analysis I (Table 7.4a) the numbers of both *E. coioides* and *E. malabaricus* were greater in Barramundi Creek than Cattle Creek (Fig. 7.7). Furthermore, in Cattle Creek the numbers of *E. malabaricus* were significantly greater than the numbers of *E. coioides*, while in Barramundi Creek there was no clear evidence that numbers of the two species were different (Fig. 7.7). For analysis II (Table 7.4b), while more *E. malabaricus* than *E. coioides* were trapped in Cattle Creek, *E. coioides* was trapped in greater numbers than *E. malabaricus* in Alligator Creek (Fig. 7.8). Both were trapped in similar numbers in Barramundi Creek. While numbers of *E. malabaricus* were similar between the three creeks, the numbers of *E. coioides* were significantly lower in Cattle Creek than in the other two creeks (Fig. 7.8).

Figure 7.7: The effect of the interaction between species and creek on the numbers of *E. coioides* and *E. malabaricus* in the four regions of Cattle and Barramundi Creeks. Data are estimates from the log-linear model ±95% confidence intervals.

Figure 7.8: The effect of the interaction between species and creek on the numbers of *E. coioides* and *E. malabaricus* in the two seaward regions of Cattle, Barramundi and Alligator Creeks. Data are estimates from the log-linear model ±95% confidence intervals.

Figure 7.7

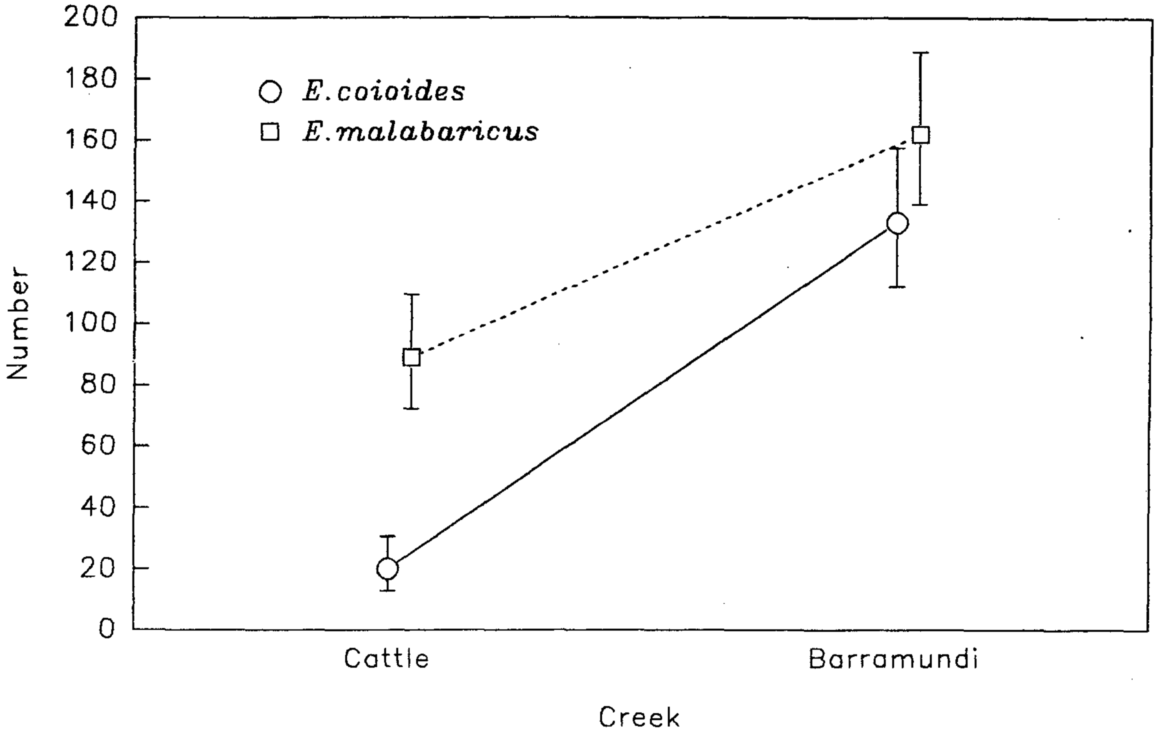


Figure 7.8

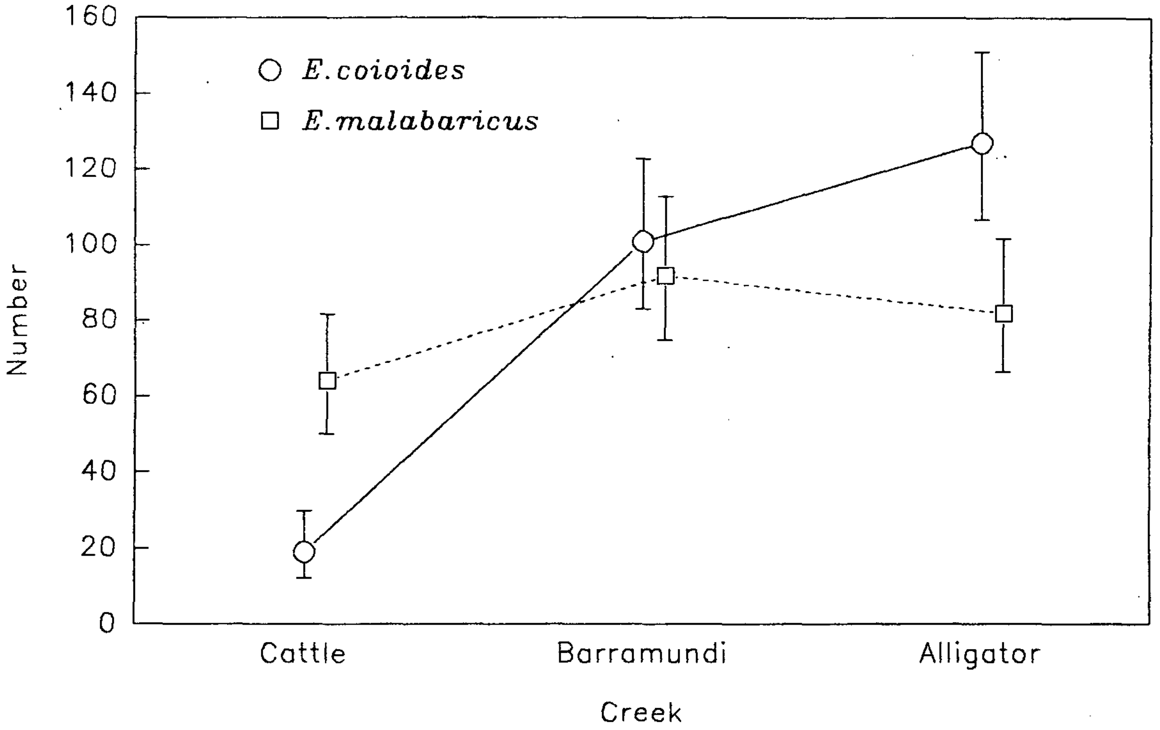


Table 7.4: Stepwise fitting (backwards elimination) of log-linear models for the numbers of *E. coioides* and *E. malabaricus* in three estuaries. (a) Four regions in Cattle and Barramundi Creeks [analysis I], (b) Two regions in Cattle, Barramundi and Alligator Creeks [analysis II]. The three left hand columns show the models fitted, and the associated deviance and degrees of freedom. The centre column shows the codes for the current model and the more complex model being compared. The three right hand columns show changes in deviance (Δ deviance), changes in the degrees of freedom (Δd.f) and the probability that the more simple model fits the data adequately. Models containing interactions also contain the main effects composing the interaction. The final model is the model from which no term can be removed without a significant increase in deviance (see Appendix I).

"O" is the code for the full 3-way model (deviance = 0, d.f. = 0)
denotes the final model.

(a)

Model	deviance	d.f.	Comparison	Δ deviance	Δd.f.	Probability
A) Species*Creek + Species*Region + Creek*Region ##	**2.09**	**3**	**A - O**	**2.09**	**3**	**0.5549**
B) Species*Creek + Species*Region	12.62	6	B - A	10.53	3	0.0145
C) Species*Creek + Creek*Region	26.64	6	C - A	24.55	3	0.0000
D) Species*Region + Creek*Region	32.01	4	D - A	29.92	1	0.0000
E) No 2-way interaction	59.33	10	E - A	57.24	7	0.0000

(b)

Model	deviance	d.f.	Comparison	Δ deviance	Δd.f.	Probability
A) Species*Creek + Species*Region + Creek*Region ##	**2.87**	**2**	**A - O**	**2**	**2.87**	**0.2377**
B) Species*Creek + Species*Region	14.17	4	B - A	2	11.30	0.0035
C) Species*Creek + Creek*Region	26.28	3	C - A	1	23.41	0.0000
D) Species*Region + Creek*Region	42.54	4	D - A	2	39.67	0.0000
E) No 2-way interaction	69.47	7	E - A	5	66.60	0.0000

For the Species by Region interaction, in analysis I (Table 7.4a) partitioning of deviance showed that the proportions of the two species trapped did not differ significantly between the Lower, Upper and Top regions (deviance = 3.295; 2 d.f.; p = 0.19253) with 2-3.5 times more *E. malabaricus* than *E. coioides* trapped in these regions (Fig. 7.9). The bulk of the deviance in the interaction was between these 3 regions (Lower, Upper, Top) and the Bottom region (deviance = 17.371; 1 d.f.; p = 0.00003) were the numbers of the 2 species were similar (Fig. 7.9). The numbers of *E. coioides* fell consistently from downstream parts of the creeks to upstream parts, however, after an initial increase in numbers from the Bottom region to the Lower region, the numbers of *E. malabaricus* also declined with distance upstream (Fig. 7.9). In analysis II (Table 7.4b), the numbers of *E. coioides* were similar in both regions and in the Bottom region were greater than the numbers of *E. malabaricus* (Fig. 7.10). The numbers of *E. malabaricus* increased substantially from the Bottom region to the Lower region and where they were greater than the numbers of *E. coioides* (Fig. 7.10).

For analysis I (Table 7.4a) the interaction between creeks and regions [independent of species] reflects differences between regions in proportions of *Epinephelus* spp. in the two creeks. Notwithstanding this interaction, the numbers of *Epinephelus* spp. in each region of Cattle Creek was consistently lower than in the same region of Barramundi Creek (Fig. 7.11). Furthermore, there is an overall similarity in the pattern of change between regions for both creeks. In both cases the numbers of *Epinephelus* spp. tended to be similar in the Bottom and Lower regions before falling away successively in the Upper and Top regions (Fig. 7.11). In analysis II (Table 7.4b) the numbers of *Epinephelus* spp. in Cattle Creek were similar in both seaward regions and substantially lower than in either Barramundi or Alligator Creeks (Fig. 7.12). The numbers of *Epinephelus* spp. were not clearly different between the Bottom regions of Barramundi or Alligator Creeks, and while there was some evidence that the numbers in the Lower regions were higher for Alligator Creek than Barramundi Creek, the difference was not clear (Fig. 7.12). As in Cattle Creek the numbers of *Epinephelus* spp. were very similar in the two regions of Barramundi Creek, but in Alligator Creek were greater in the Lower region.

Figure 7.9: The effect of the interaction between species and region on the numbers of *E. coioides* and *E. malabaricus* in the four regions of Cattle and Barramundi Creeks. Data are estimates from the log-linear model ±95% confidence intervals.

Figure 7.10: The effect of the interaction between species and region on the numbers of *E. coioides* and *E. malabaricus* in the two seaward regions of Cattle, Barramundi and Alligator Creeks. Data are estimates from the log-linear model ±95% confidence intervals.

Figure 7.9

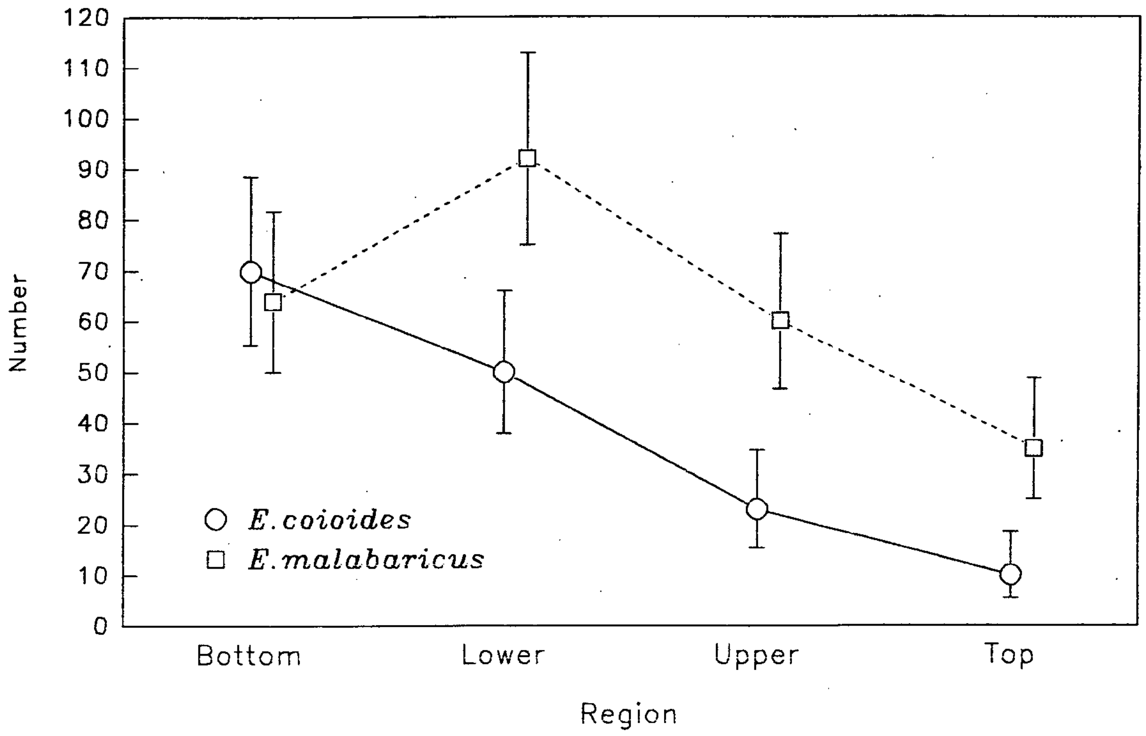


Figure 7.10

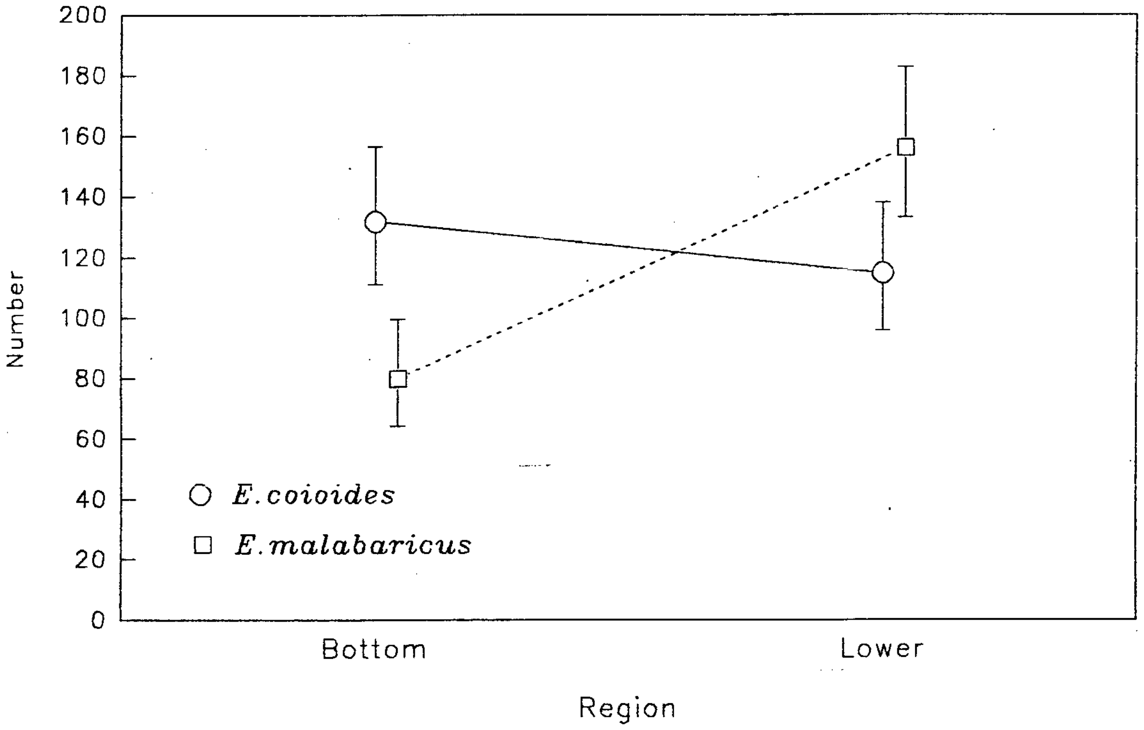


Figure 7.11: The effect of the interaction between creek and region on the numbers of *Epinephelus* spp. in the four regions of Cattle and Barramundi Creeks. Data are estimates from the log-linear model ±95% confidence intervals.

Figure 7.12: The effect of the interaction between creek and region on the numbers of *Epinephelus* spp. in the two seaward regions of Cattle, Barramundi and Alligator Creeks. Data are estimates from the log-linear model ±95% confidence intervals.

Figure 7.11

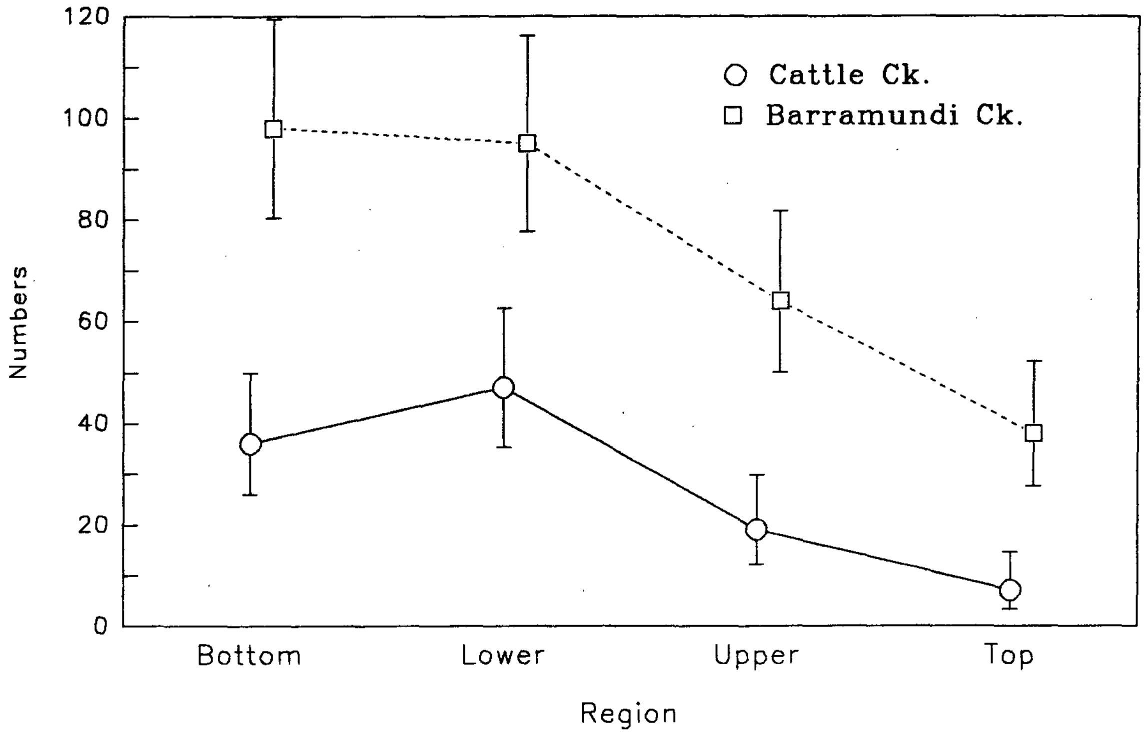
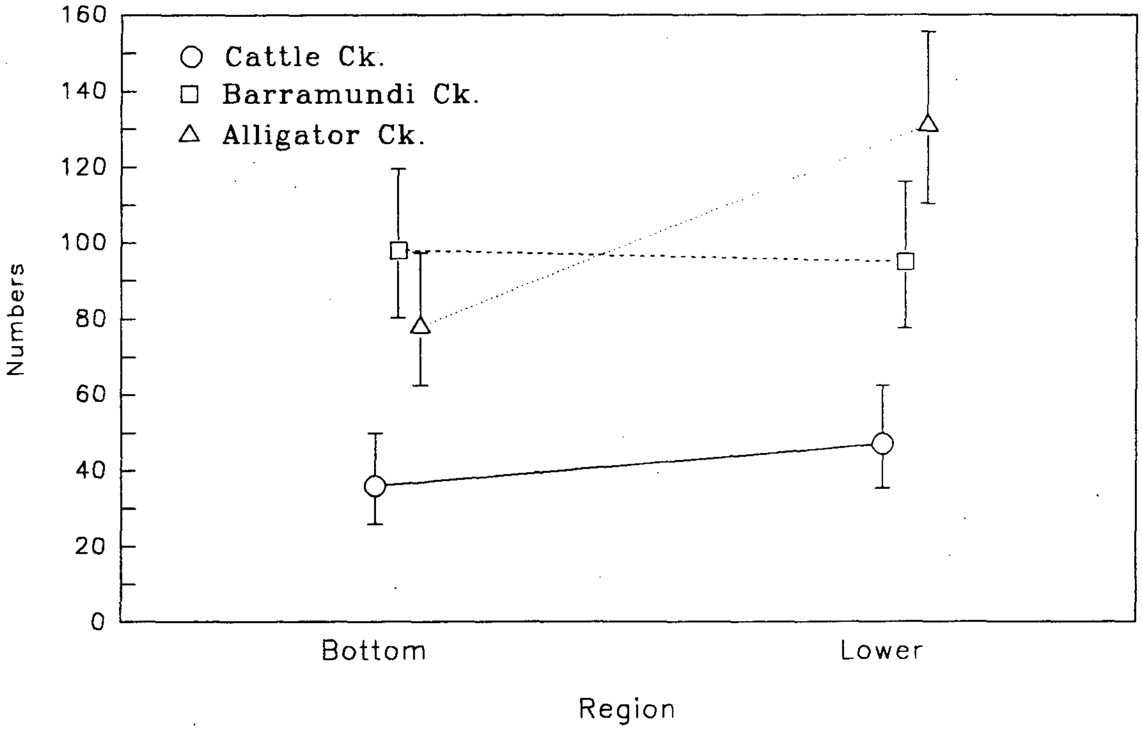


Figure 7.12



7.3.3 Relationship between physical variability and catches of *E. coioides* and *E. malabaricus*

There was no significant correlation between catch rates of either *E. coioides* or *E. malabaricus* and water temperature (Table 7.5). While catches of *E. coioides* per region of each creek per day were significantly correlated with bottom salinity catches of *E. malabaricus* were not (Table 7.5). However, the correlation between catches of both species and the maximum deviation in salinity away from normal seawater (MSD) were much stronger (Table 7.5). The correlation of MSD with catches of *E. coioides* per region produced a much higher significance level while catches of *E. malabaricus* per region were highly correlated where the simple correlation with salinity was not.

Table 7.5 Correlations between numbers of *E. coioides* and *E. malabaricus* caught per trap and bottom temperature (C^0), bottom salinity (‰) and maximum deviation in bottom salinity per region. Data presented are the Pearson product moment correlation coefficient (r) and its associated probability level (p); n = 240.

	Temperature r [p]	Salinity r [p]	Salinity deviation r [p]
E. coioides	0.0423 [0.5143]	0.1572 [0.0148]	- 0.3623 [0.0000]
E. malabaricus	0.1038 [0.1087]	0.0930 [0.1509]	- 0.1824 [0.0046]

Catches of *E. coioides* per region declined with MSD both above [Barramundi Creek] and below [Cattle Creek] normal seawater salinity (Fig. 7.13). For both positive [Barramundi Creek] and negative [Cattle Creek] MSDs catches of *E. malabaricus* first increased then decreased as MSD became greater (Fig. 7.14). This pattern was clearer for Cattle Creek than Barramundi Creek where the catch rates for the 4 regions were similar. In Alligator Creek MSD in the two regions deviated from normal seawater in opposite directions so the relation between MSD and catch rates is unclear.

Figure 7.13: The relationship between maximum salinity deviation [MSD] and numbers of *E. coioides* trapped per region in Cattle, Barramundi and Alligator Creeks. Data are mean numbers per region per day ±95% confidence intervals.

Figure 7.14: The relationship between maximum salinity deviation [MSD] and numbers of *E. malabaricus* trapped per region in Cattle, Barramundi and Alligator Creeks. Data are mean numbers per region per day ±95% confidence intervals.

Figure 7.13

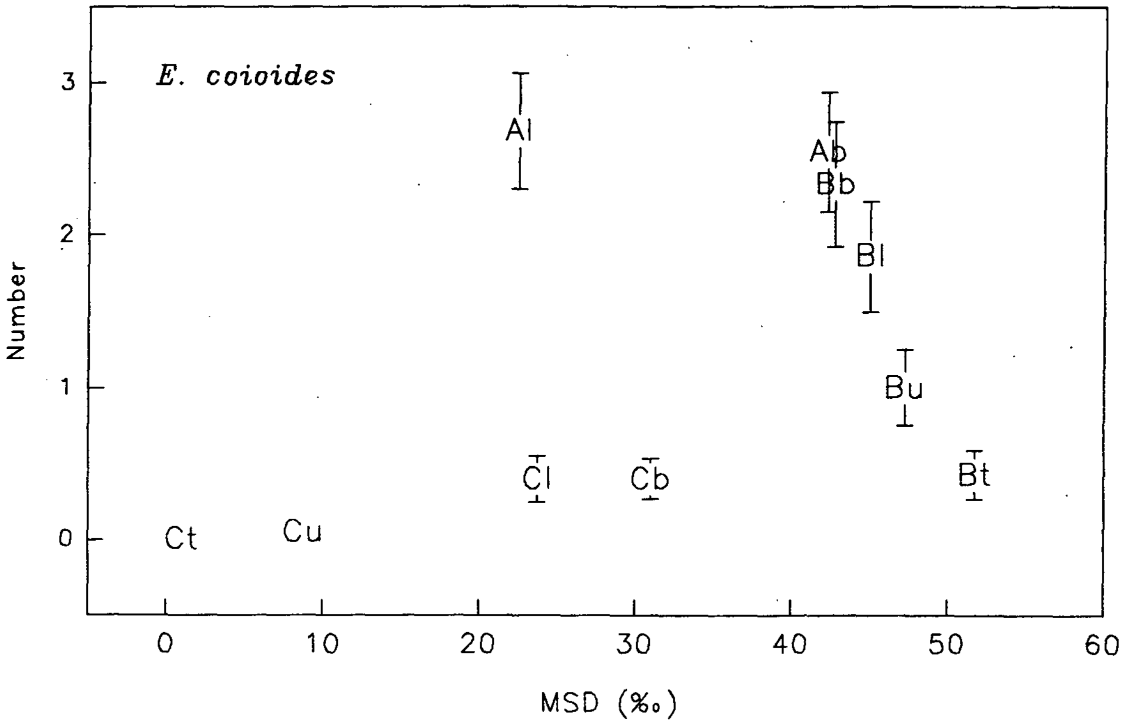
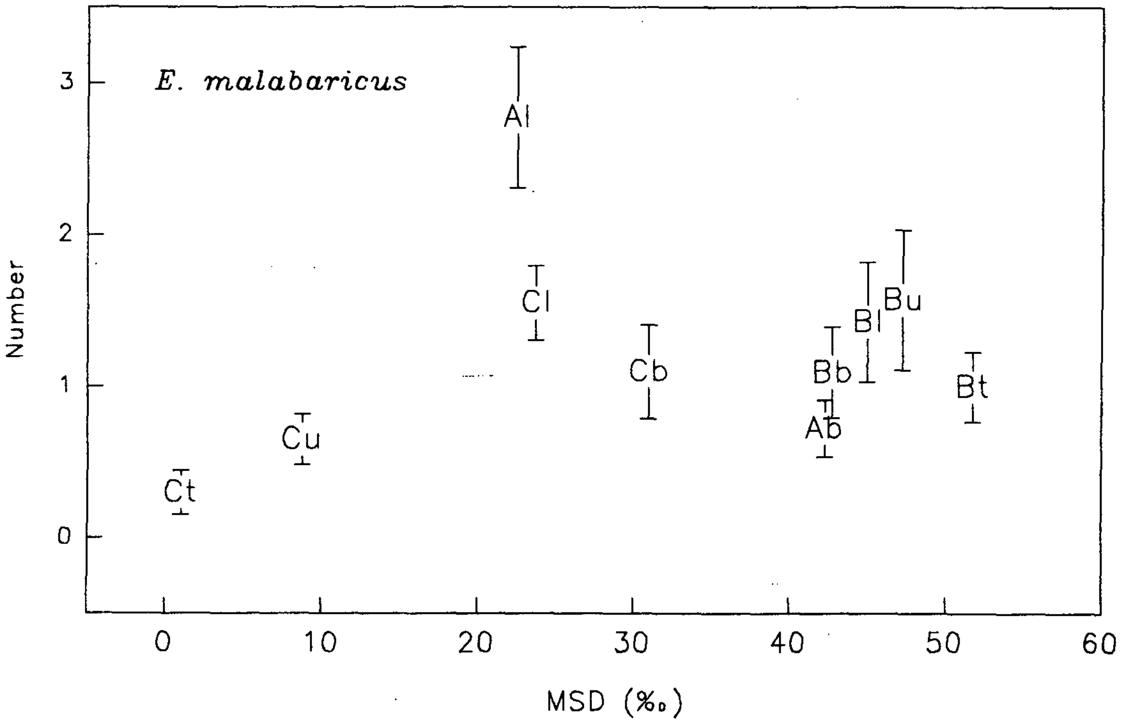


Figure 7.14



7.4 DISCUSSION

7.4.1 Comparison of size structures of *E. coioides* and *E. malabaricus*

Over the three creeks sampled the size structures of the two *Epinephelus* species demonstrated some marked differences. Both the modal and maximum size of *E. malabaricus* were larger than those of *E. coioides*. However, for both species there was a sharp decline in numbers caught at the lower ends of the size distributions, with few fish below 120mm trapped. This size was well above the maximum size that could have been retained by the 12.5mm mesh. For instance the largest *Lutjanus russelli* able to squeeze through the trap mesh was about 70mm (pers. obs.). Thus the traps should have been able to catch much smaller *Epinephelus* spp. than they did. It appears therefore, that either the smaller fish occupied habitats that were not sampled, or if present in the sampling area, that they exhibited behaviours that meant that they did not enter the traps. If small, juvenile *Epinephelus* spp. do use different habitats to the larger fish it is not clear what these may be. A number of studies have reported *Epinephelus* spp. from estuarine or coastal seagrass habitats. Blaber *et al.*, (1989) caught small (<10cm SL) *E. suillus* (probably *E. coioides*) using a beam trawl (28mm mesh) in seagrass beds in the Embley estuary in northern Australia. *E. suillus* was common in these samples contributing 13.32% to the total biomass. Although no sizes were presented, Blaber *et al.*, (1992) reported *E. malabaricus* (n = 2) and *E. suillus* (n < 26) from coastal seagrass beds in north-western Australia. Conversely, no species of *Epinephelus* were recorded from beam trawl samples (2mm mesh net) from seagrass beds adjacent to Trinity Inlet in north-eastern Australia (Coles *et al.*, 1993). Thus it is unclear if small juvenile *E. coioides* or *E. malabaricus* utilise seagrass habitats to a substantial extent. While there are other habitats that could be used be small, juvenile *E. coioides* and *E. malabaricus*, no study of other habitats in tropical Australia has reported significant numbers of small juveniles of either species.

No clear spatial of temporal differences were found in the size of *E. coioides*. This was probably due to a lack of data preventing viable analysis of *E. coioides* size for many spatio-temporal combinations.

The mean size of *E. malabaricus* was greater in the two seaward regions of Barramundi Creek than in the two seaward regions of either Cattle or Alligator

Creeks. The reasons for this difference are not clear. These differences in size structure may be a reflection of different settlement histories. For instance, settlement may have been relatively more successful in Barramundi Creek than in the other two creeks in the years when the large individuals now present were recruited. However, the mean size of *E. malabaricus* for the Upper region of Cattle Creek was greater than for the Upper region of Barramundi Creek, making arguments of differential settlement success between creeks (as well as most other simple explanations) difficult to sustain. Apparently more complex explanations, including mechanisms producing within-creek differences, are needed to account for the patterns of spatial differences in size of *E. malabaricus*.

There were no clear patterns of temporal change in the size or *E. malabaricus* that could be related to biological events.

7.4.2 Comparison of the distribution and abundance of *E. coioides* and *E. malabaricus*

The abundances of *Epinephelus* spp. varied between estuary systems. Greater numbers of both *E. coioides* and *E. malabaricus* were trapped in Barramundi Creek than in Cattle Creek (Fig. 7.7), and catches of *E. coioides* in the two seaward regions of Cattle Creek were substantially lower than in these regions in the other two creeks (Fig. 7.8). It is possible that the observed low abundances in Cattle Creek may reflect a fundamental difference between this and the other two creeks. If so, it would appear that in some way Cattle Creek provides a poorer quality of habitat than Barramundi and Alligator Creeks. One factor that could contribute to the low abundances in Cattle Creek is the occurrence of regular, annual floods. Flooding can influence abundances of estuarine fish populations directly (Marais, 1982), and by its influence on salinity (Hoff and Ibara, 1977; Marais, 1988; Moser and Gerry, 1989) which is important in structuring fish communities in tropical estuaries (Quinn, 1980). A negative correlation was found between catch rates and flooding in an estuary in South Africa (Marais, 1983). Annual flooding of Cattle Creek caused a marked depression of salinity compared to the other two creeks (Fig. 6.23). This annual influx of fresh water and the associated salinity reduction could work in a variety of ways to reduce numbers of the two species. Hyposaline conditions would cause osmoregulatory problems (Lasserre, 1975; Martin, 1988) perhaps increasing mortality rates (Lockwood, 1976) or causing fish to move to other areas (Hoese, 1960). Low salinity and increased stream flow during flooding may prevent many recruits entering the creek or at least penetrating upstream areas, or cause newly settled fish to be flushed from the creek. Increased outflow from the creek and associated reduction in salinity may reduce the abundance of prey organisms (Cyrus, 1988), thereby reducing the amount of food available to *Epinephelus* spp. If the extra freshwater input to Cattle Creek did affect the abundance of *Epinephelus* spp., the fact that *E. malabaricus* was trapped in greater abundance than *E. coioides* in Cattle Creek but not in the other creeks, suggests that *E. malabaricus* may possess greater tolerances to reduced salinity levels. *Leiostomus xanthurus* and *Micropogonias undulatus*, two species of fish inhabiting estuaries on the Atlantic coast of the United States, show different behavioural responses to salinity changes, which lead to different patterns of distribution (Moser and Gerry, 1989).

Despite the plausibility of explanations involving differences in salinity or flooding regimes there are other possible reasons for the changes in abundance of *Epinephelus* spp. between estuaries. It may be that the differences in abundance were specific to the time window in which the study was conducted and not a reflection of long term patterns. For instance, unusually poor recruitment in Cattle Creek over a number of years while the other creeks received normal or better than normal recruitment could have produced the observed differences.

The numbers of *E. coioides* and *E. malabaricus* were very similar in the seaward regions of Cattle and Barramundi Creeks. However, while the numbers of *E. coioides* fell consistently with distance upstream, the numbers of *E. malabaricus* increased substantially in the Lower region, and while they did fall consistently in the two upstream regions remained significantly higher than the numbers of *E. coioides* (Fig. 7.9). When data for Alligator Creek were included and only the two seaward regions considered, a change in the dominance of the two species was seen, with *E. coioides* more abundant in the most seaward [Bottom] regions and *E. malabaricus* more abundant in the Lower region (Fig. 7.10). Although using somewhat different spatial divisions, previous studies in Alligator Creek (Sheaves, 1992) also found that *E. coioides* was more abundant in downstream parts and *E. malabaricus* more abundant in upstream areas. While there are clear differences between the spatial abundances of the two species the reasons for these differences are unclear. It may be that *E. malabaricus* is more tolerant to variations in salinity away from 'normal' seawater levels than is *E. coioides*. Differences in physical tolerances can be important in causing closely related species of estuary fish to exhibit different patterns of distribution and abundance (Martin, 1988). An alternative explanation is that *E. malabaricus* recruits may be better able to access upstream areas than recruits of *E. coioides*.

Overall, in both Cattle and Barramundi Creeks, there was a strong trend for the numbers of both species of *Epinephelus* to decline in an upstream direction (Fig. 7.11). There are two possible explanations for this. Firstly, fewer recruits may penetrate to upstream areas leading to lower recruitment. Alternatively, habitat types, food availability or physical conditions may be less favourable in upstream areas. For instance, while the seaward regions of Cattle and Barramundi Creeks exhibited salinity levels close to that of 'normal' seawater (about 35‰), upstream parts of Cattle Creek often showed depressed salinities (Fig. 6.23). Conversely, upstream parts of Barramundi Creek tended to be hypersaline. The fact that numbers of *Epinephelus* spp. occurred in upstream areas of both creeks suggests that both hypersaline and hyposaline conditions may lead to reduced numbers of *Epinephelus* spp., as is the case for many other species (Gunter, 1961).

While there was some correlation between catch rates of *E. coioides* and salinity, there was a much stronger relationship between catch rates of both species of *Epinephelus* and the maximum deviation in salinity (MSD). This suggests that if salinity, or some correlated variable, does influence the abundance of the two species it is through long-term effects rather than short term changes. Clearly, if a long-lived fish cannot cope with particular salinity levels, a short period of adverse conditions may kill the fish or cause it to migrate away, thereby influencing patterns of abundance for a considerable time into the future. The link between catch rates and MSD does not establish causality, only correlation. Many other variables (e.g. water depth and distance from the ocean) also correlate with MSD. Even if the long term pattern of salinity deviation is a major factor controlling the abundance of *Epinephelus* spp. the relationship is unlikely to be as simple as a direct correlation with MSD. Factors such as the duration and regularity of salinity variation (Lockwood, 1976) would probably be as important as the actual deviation from 'normal' salinity.

CHAPTER 8.0

GENERAL DISCUSSION

8.1 THE COMPOSITION OF LUTJANID AND SERRANID FAUNAS OF ESTUARIES IN TROPICAL NORTH-EASTERN AUSTRALIA

Lutjanids and serranids are wide spread in tropical waters (Allen and Talbot, 1985; Randall and Heemstra, 1991). However, compared to other marine habitats in tropical north-eastern Australia, the lutjanid and serranid faunas of estuaries are depauperate; they contain only a small number of the species inhabiting nearshore waters. Collections carried out during the present study (Chapter 2) showed that while as many as 5 species of lutjanids and 7 species of serranids can be found in tropical north-eastern Australian estuaries, most occur only irregularly, and catches are dominated by a few species. Extensive sampling of estuaries across tropical Australia employing a diverse range of gear types including seine nets and gill nets (Blaber, 1980; Robertson and Duke, 1987; Blaber, *et al.*, 1989), rotenone (Blaber, *et al.*, 1989) and fish traps (Sheaves, 1992; 1993; 1994), have only reported two additional species of serranids. Populations of at least four of the most common species, *L. russelli*, *L. argentimaculatus*, *E. coioides* and *E. malabaricus*, appear to be composed entirely of pre-reproductive juveniles. These four species are probably wide-spread in estuaries in tropical Australia as they are common in the present study area and are also found in other areas of tropical Australia (Blaber 1980; Blaber *et al.*, 1989).

The fact that few lutjanid and serranid species utilise estuaries suggests a number of questions.

Why are the serranid and lutjanid faunas of tropical north-eastern Australian estuaries depauperate? Most offshore species occur in estuaries very irregularly, if at all. Thus it appears that estuaries are unsuitable or unattractive habitats for many species. The reasons for this are unclear. Physical variability is a major factor determining the distribution and abundance of fishes within estuaries (Gunter, 1961; Blaber and Blaber, 1980, Cyrus and Blaber, 1987; Martin 1988), thus it seems likely that many species not found in estuaries are unable to deal with the extreme physical variability that characterises tropical estuarine systems (Cyrus and Blaber, 1987; Cyrus and Blaber, 1992; Mazda *et al.*, 1990).

What is the mechanism or mechanisms that determine what species are found in estuaries? Marine habitats only a short distance offshore contain many species of lutjanids and serranids not found in estuaries (Chapter 2). Thus the larvae of these species must be present in waters close to the mouths of estuaries. There are therefore three possibilities; the larvae of offshore species may not enter estuaries; the larvae may enter estuaries but not settle; the larvae may settle in estuaries but only remain for a short time. If the larvae do not enter estuaries there must be some mechanism preventing their entry. This could involve adults spawning at times and locations that place the larvae in nearshore waters at times when they will not be washed into estuaries. Alternatively, the larvae may possess behaviours that enable them to remain outside estuaries. Such behaviours would parallel those that prevent estuary fish larvae from being washed out of the estuary. For example, barramundi (*Lates calcarifer*), a large predatory fish co-occurring in estuaries in tropical Australia, spawn in the lower reaches of estuaries (Griffin, 1987) and their larvae enter saline (Russell and Garrett, 1983), brackish and freshwater swamps (Russell and Garrett, 1985) within the estuary. Thus *L. calcarifer* larvae are faced with the problem of maintaining themselves within the estuary. This may be achieved by a mechanism similar to that used by *Gilchristella aesturaius* in South African estuaries. *G. aesturaius* larvae remain close to the bottom throughout the tidal cycle thus utilising flood-tide bottom-water currents and avoiding ebb-tide surface currents (Melville-Smith *et al.*, 1981). If the larvae enter estuaries but don't settle further questions present themselves. What prevents them from settling? Are there negative environmental cues or is it simply a lack of the positive cues needed to initiate settlement? If the larvae settle for a short time and then die or migrate out of the estuary what causes this? Three possibilities are inappropriate physical conditions, a lack of appropriate food or an absence of appropriate habitat. Whatever the mechanism determining what species are found in estuaries, the three main alternatives are testable, as are most of the subsidiary reasons.

8.2 LUTJANIDS AND SERRANIDS: JUVENILES IN TROPICAL ESTUARIES

During this study *L. russelli*, *L. argentimaculatus*, *E. coioides* and *E. malabaricus* were collected from estuaries over a broad section of the coast of north-eastern tropical Australia. Despite the collection spanning more than two years, all the fish collected from estuaries were much smaller and younger than the largest fish of the same species from offshore. Furthermore, it appears that none of the species were reproductively active within the estuaries studied. Thus estuarine populations of the four species appear to consist of functional juveniles.

The pre-reproductive status of *L. argentimaculatus*, *L. russelli, E. coioides* and *E. malabaricus* in the estuaries studied necessitates an offshore migration to spawning areas. Previous workers have suggested or implied offshore spawning migrations for *L. argentimaculatus* (Johannes, 1978; Thollot *et al.*, 1990), *E. malabaricus* (Thollot *et al.*, 1990) and *L. russelli* (Allen, 1985; Allen and Talbot, 1985; Randall *et al.*, 1990; Thollot *et al.*, 1990). The lack of adult fish of these species in estuaries during this study suggests that offshore migration is usually permanent; with adult fish remaining offshore. This is a common situation for estuary fish in South Africa, where most species that have estuarine juveniles but spawn offshore don't subsequently return to estuary habitats (Day *et al.*, 1981). For example, three species of *Gerres* remain in estuaries until sexual maturity is reached, then move offshore to spawning areas where they usually remain (Cyrus and Blaber, 1984).

Despite the necessity of a spawning migration for these fish few actual movements have been documented. This paucity of direct evidence of movement from estuaries to offshore habitats needs to be rectified. However, given the large areas of reefal habitat off the coast of north-eastern Australia, a substantial tagging and recapture effort would be required to demonstrate movement offshore.

These species either can't or don't spawn in estuaries. However, the reasons why they migrate offshore and the cues that initiate migration are unknown. There are many possible reasons for spawning offshore rather than in estuaries. Perhaps the sperm or ova of these species are inviable in the extreme physical conditions prevalent in estuaries. Alternatively an offshore migration may represent a mechanism for increasing genetic mixing. This would occur in two ways. Fish migrating offshore would be likely to breed with fish from other estuaries, and, after a planktonic larval stage followed by an offshore spawning migration, the offspring they produced would be unlikely to return to the location where they were spawned.

Most tropical marine demersal fishes have two-part life-histories; pelagic larvae and demersal adults (Doherty and Williams, 1988). However, the two lutjanids and two serranids studied here posses three distinct life-history stages; pelagic larvae, estuarine juveniles, offshore adults. These life-history stages correspond to major habitat shifts. The possession of a three-part life-history is not unique among tropical marine fishes. After a pelagic larval stage, newly settled juvenile scarids pass through an initial carnivorous stage before taking up a herbivorous life-style (Bellwood, 1988). Although necessitating no habitat shift, the possession of an obligate [carnivorous] stage between pelagic larvae and [herbivorous] adult, parallels the three-part life-cycle of the estuarine lutjanids and serranids.

It is unknown if the life-cycle described here for *L. russelli, L. argentimaculatus, E. coioides* and *E. malabaricus* is followed by these species in areas other than north-eastern Australia. The extent to which related species follow this life-history pattern is unclear also. It is possible that many other species of lutjanids and serranids use juvenile habitats outside estuaries and undergo a similar migration to adult habitats. For example, *L. griseus*, one of the most common lutjanids in sub-tropical areas of the southeastern United States, spawns on offshore reefs and uses inshore seagrass and mangrove areas as nursery grounds (Starck and Schroeder, 1970; Chester and Thayer, 1990). Similarly, in Papua New Guinea waters, *L. bohar* migrates into deep waters at large sizes (Wright *et al.*, 1986), and in Guadeloupe, French West Indies, large *Ocyurus chrysurus* migrate to reefs from primary nursery grounds in coastal seagrass beds (Baelde, 1990). Species with estuarine juveniles from a number of other families also move offshore to spawn. Six species of carangids that are common in South Africa occur only as juveniles and sub-adults in estuaries (Blaber and Cyrus, 1983), while three species of *Gerres* are found in South African estuaries only in juvenile and pre-spawning condition (Cyrus and Blaber, 1984). Similarly, in the southeastern United States the sciaenid, *Sciaenops ocellatus*, spawns offshore after occupying estuarine habitats during juvenile stages (Peters and McMichael, 1987), and on the Atlantic coast the juvenile bluefish, *Pomatomus saltatrix*, move to juvenile habitats in estuarine and nearshore areas (McBride and Conover, 1991).

While juvenile *L. russelli, L. argentimaculatus, E. coioides* and *E. malabaricus* are abundant and widespread in estuaries of tropical north-eastern

Australia, few juveniles of these species have been reported from other environments. This clearly indicates that estuaries fulfil an important nursery ground role for these species; probably supplying a large part of the adult population found in offshore waters. Estuaries perform a similar nursery ground role for fishes in all parts of the world. For example, many species of fish in Nigerian estuaries are represented by juveniles (Amadi, 1990), in the West Indies mangrove areas are nurseries for sparids, clupeids and gerreids (Baelde, 1990), in Scotland the Forth estuary provides nursery grounds for North Sea fishes including cod [*Gadus morhua*] (Elliott *et al.*, 1990), in Taiwan (Tzeng and Wang, 1993) and the southern Gulf of Mexico (Yâñez-Arancibia *et al.*, 1988) estuaries provide juvenile habitats for a broad range of species, and in South Africa estuaries are nursery grounds for more than eighty species of fishes (Day *et al.*, 1981; Whitfield, 1983).

What benefits do these species gain by utilising estuarine nursery grounds? It is likely that for any species there are a number of potential advantages. Small fish require more food per unit body weight than large fish (Pauly, 1986), thus juvenile fishes must be placed in habitats with high carrying capacities (Yâñez-Arancibia *et al.*, 1994). As estuary studies from around the world almost invariably report an abundance of small fishes, particularly juveniles (e.g. Austin 1971; Lenanton, 1977; Shenker and Dean, 1979; Bell *et al.*, 1984; Yâñez-Arancibia *et al.*, 1988; Robertson and Duke, 1990; Ayvazian *et al.*, 1992; Tzeng and Wang, 1992), it seems that estuaries must provide enhanced feeding opportunities for fish (Chong *et al.*, 1990). There are other possible advantages that may accrue to juveniles in estuaries. It has been suggested that turbid conditions reduce the effectiveness of large predators so reducing predation on juvenile fishes (Blaber and Blaber, 1980; Cyrus and Blaber, 1987), or that the structurally complex habitats prevalent in tropical estuaries (Sheaves, 1992) provide areas of refuge for small fish. Alternatively, estuaries may be of no direct benefit, but rather provide juveniles with an area free from species with superior competitive abilities.

8.3 SPATIAL AND TEMPORAL PATTERNS OF DISTRIBUTION OF *L. RUSSELLI*, *E. COIOIDES* AND *E. MALABARICUS* IN ESTUARIES

L. russelli, *E. coioides* and *E. malabaricus* were broadly distributed within the estuaries studied. However, there was a tendency for trap catches of all three species, particularly the two serranids, to decline in upstream areas. This pattern of decline in numbers may have been related to inherent differences between upstream and seaward regions of the estuaries. Upstream areas differed in many ways to seaward parts. For example, in upstream areas the water was generally shallower, the cross-sectional area of the stream less, and over the year physical conditions, particularly salinity, deviated further from "normal" seawater levels. One important factor in the present study appeared to be long-term patterns of salinity variability. Physical conditions are generally seen as the main forces structuring estuarine fish communities (Gunter, 1961; Lasserre, 1975; Marais, 1983; Cyrus, 1988; Martin, 1988), and in the present study the regional abundance of *E. coioides* and *E. malabaricus* was correlated with long term patterns of salinity deviation. An alternate explanation for reduced numbers in upstream areas is that less larvae may be able to penetrate into upstream areas, so less are available to settle. The need for larvae to penetrate upstream against adverse currents is a common problem for all estuarine animals (Melville-Smith *et al.*, 1981).

There were also differences in numbers trapped between the three estuaries studied in detail. In general catches of all three species tended to be lower in Cattle Creek [particularly in upstream parts] than in Barramundi and Alligator Creeks. These patterns of difference were consistent so the differences in catches probably reflect differences in abundance rather than differences in catchability. These differences may relate to differences in the physical natures of the creeks. For example, Cattle Creek received regular floods during the study while Barramundi and Alligator Creeks did not. These floods depressed the salinity and increased the outflow of Cattle Creek. Conversely, the differences in abundances between the creeks may not reflect intrinsic differences between the creeks, but rather short to moderate term differences in recruitment histories. While both physical differences and recruitment variability may contribute to the observed patterns, one is probably more important than the other. If so the most important determining factor should not be too difficult to identify. If intrinsic physical differences are the main structuring forces, then given a continuation of the physical differences, a future study should find lower abundances in Cattle Creek. If recruitment variability is the major factor, future studies should find different patterns of abundance between the estuaries.

Differences i~~s~~n distribution and abundance occurred on a finer spatial scale also. *L. russelli* showed a strong preference for structurally complex habitats provided by submerged timber and mangrove roots. In offshore areas many lutjanids have strong affinities for structurally complex reef habitats. Thus by utilising submerged timbers and roots in estuaries *L. russelli* is utilising parallel habitats to those used by lutjanids on reefs. *E. coioides* was more common in the structurally complex snag habitats, although not significantly so, however, previous work (Sheaves, 1992) indicated that this species does prefer complex habitats. Preferences for structurally, complex habitats in mangrove systems have been reported from other parts of the world (Thayer *et al.*, 1987) and probably relate to shelter from predators or enhanced feeding opportunities (Chong *et al.*, 1990).

There was a clear difference in the patterns of abundance of *E. coioides* and *E. malabaricus*. While numbers of the two species were similar in the seaward regions, the numbers of *E. coioides* declined much more rapidly in upstream areas than did those of *E. malabaricus*. A similar result has been reported previously (Sheaves, 1992). It seems likely that these differences reflect differences in the physiological tolerances of the two species, as this has been shown to cause differences in the distribution and abundance of closely related species of estuary fish (Martin, 1988). Despite the attractiveness of this interpretation there are other possible explanations, such as differences in the ability of recruits of the two species to access upstream areas.

While both spatial and temporal differences were found in sizes of *L. russelli*, *E. coioides* and *E. malabaricus* within estuaries, with one exception the biological importance of these were minor or unclear. The one exception was a clear, strong pattern of seasonal change in the size of *L. russelli*. This pattern reflected the competing influences of recruitment increasing numbers in the smaller size classes, and mortality and migration reducing numbers in the larger size classes. Apparently due to relatively small sample sizes and long residence times in estuaries, clear patterns of seasonal change in the sizes of *E. coioides* and *E. malabaricus* were not apparent.

8.4 IMPLICATIONS

L. russelli, *L. argentimaculatus*, *E. coioides* and *E. malabaricus* are widespread in estuaries of tropical north-eastern Australia and despite reaching quite large sizes within these estuaries are apparently all juveniles. Moreover these juveniles occupy estuaries for periods up to at least seven years. Clearly neither estuarine or offshore populations can be managed in isolation. The long-term occupation of estuary nursery grounds will need to be considered in the development of any future management plans for these species. The destruction of estuarine habitats is a major problem world-wide (Saenger *et al.*, 1983). Clearly any factors that impact adversely on estuarine habitats may have major down-stream effects for offshore populations of these species. Similarly, anything that adversely affects offshore breeding populations or diverts larval supply would have major effects on estuarine populations. Thus the results of this study have implications for both fisheries management and estuary protection.

As with most ecological studies this work poses many more questions than it answers. Why are the serranid and lutjanid faunas of tropical north-eastern Australian estuaries depauperate? What is the mechanism that determines what species are found in estuaries? What benefits do these species gain by utilising estuarine nursery grounds? How did the reproductive pattern evolve? Are there physiological differences that allow these species to utilise estuaries while congenerics cannot? All these questions have the potential to provide fruitful directions for future research.

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APPENDIX I

NON-STANDARD STATISTICAL TECHNIQUES

A number of non-standard statistical techniques were used in this thesis. These include; logistic regression (Chapters 5 and 6), intraclass correlation (Chapter 6), periodogram and cumulative periodogram (Chapter 6), and log-linear model (Chapter 7).

I.1 LOGISTIC AND LOG-LINEAR MODELS

Logistic and log-linear models are members of the class of models known as generalised linear models [GLMs] (Nelder and Wedderburn, 1972).

The logistic model is appropriate for binomial data (Collett, 1991), and takes the form:

$$p_i = \frac{\exp(\hat{\eta}_i)}{(1 + \exp(\hat{\eta}_i))} + \varepsilon \quad ; \quad \varepsilon \sim \text{Binomial} \ ; p_i \in [0,1]$$

Where p_i = probability of i; $\hat{\eta}_i$ = the **linear predictor** (the estimated value of the linear systematic component (Collett, 1991)).

The log-linear model is appropriate for count data (Fienberg, 1991), and takes the form:

$$y_i = \exp(\hat{\eta}_i) + \varepsilon \quad ; \quad \varepsilon \sim \text{Poisson} \ ; \ y \in [0,+\infty]$$

Where $\hat{\eta}_i$ = the linear predictor

I.1.1 SELECTING THE APPROPRIATE LOGISTIC OR LOG-LINEAR MODEL

The most complex model for a set of data will contain as many unknown parameters as there are observations (Collett, 1991). Thus this model, when fitted, coincides with the actual observations. Such a model is termed a **saturated model** (Collett, 1991). As the saturated model fits the data perfectly it is of limited usefulness; it does not provide a more simple summary of the data than the observations themselves (Bishop *et al.*, 1975). More simple models fit the data less well, but as they contain less parameters provide a simplified summary of the data. Such a simplified model provides the opportunity for clear biological interpretation of the main features of a data set. Thus the aim of linear modelling is to choose a model that adequately describes the data in question, but contains as few parameters as possible. Economical parameterisations, if justified, enable more efficient inferences to be drawn (Altham, 1984).

Generally, both log-linear models and logistic [regression] models are fitted using the method of maximum likelihood (Bishop *et al.*, 1975). Deviance [G^2] is used as the measure of goodness-of-fit. Deviance is a measure of the extent to which the current model deviates from the saturated model [$G^2 = 0$] (Collett, 1991; Fienberg, 1991). In comparing two models the difference in deviance [ΔG^2] between the two models can be used to judge the importance of additional terms. G^2 has an asymptotic χ^2 distribution. Thus the significance of ΔG^2 can be determined by comparison to the χ^2 distribution with degrees of freedom [d.f] equal to the difference between the d.f.s for the two G^2 terms [i.e. Δd.f.].

The general approach to selecting a model to describe a set of data is to compare a more complex model to a less complex model for the same data (Fienberg, 1991). The aim is to determine if there is a significant difference in the fit of the two models. If not the more simple model is adopted. This model can then be compared to a more simple model. This procedure continues until the most simple model that fits the data adequately is determined.

The model selection procedure used here is a stepwise backward elimination procedure described by Goodman (1971). First a significance level (say 0.05) is chosen as the criterion for determining a significant difference between models. Thus a significant ΔG^2 indicates an unacceptable reduction in the fit of the model. For example, for a model containing four variables the procedure might be as follows:

i) A model containing all 3-way interactions (and all 2-way interactions and main effects) is compared to the saturated model (i.e. including the 4-way interaction also).

ii) If the model containing all 3-way interactions is adequate it is compared to the model containing only 2-way interactions.

iii) If the model containing all 3-way interactions is adequate but the model containing only 2-way interactions is not, then the model containing all 3-way interactions is taken as a starting point and compared to a model containing all 3-way interactions except the least significant.

iv) This procedure is continued until the simplest model that fits the data adequately is found. At any point terms that significantly improve the fit of the model are re-entered into the model.

In general, terms are removed from models in groups (Bishop *et al.*, 1975). This reduces the number of significance tests performed, thereby lessening the chance of committing a type-I error. The procedure of selection is quite flexible. For instance, if 3-way interactions were of little interest, the first comparison might be between the saturated model and a model containing all 2-way interactions. Thus the probability of a type-I error is reduced by not performing unnecessary significance tests.

I.1.2 ANALYSIS OF SUB-TABLES

In an analogous way to *a posteriori* testing after analysis of variance, where significant factors in a logistic or log-linear model have more than two levels, the levels that are responsible for the significant effect can be ascertained. This is achieved by breaking the contingency table into sub-tables and partitioning the deviance between these sub-tables (Bishop *et al.*, 1975). Typically, this results in a group of factor levels that do not differ significantly from each other, but do differ significantly from a second group of factor levels. Thus the bulk of the deviance would be between the two groups of factors, with little deviance between members of the homogeneous groups.

I.2 INTRACLASS CORRELATION

Correlation is usually investigated between pairs of observations measured on distinctly different variables, for example salinity and temperature. In this case, it is clear that all salinity data should be designated to one axis and all temperature data to the other. Intraclass correlation (Zar, 1984; p.323) is used in the situation where the correlation between pairs of observations are of interest, but there is no reason why the members of a pair of observations should be designated to one axis or the other. For example in the present study (Chapter 6) the correlation between the lengths of pairs of fish from each of a number of fish traps was investigated. Clearly, there is no reason to allocate either member of a pair to either axis in preference to the other.

Each pair of observations is treated as a group in analysis of variance and the mean squares both within and between groups calculated. The intraclass correlation coefficient (r_i) is then defined as:

$$r_i = \frac{\text{group MS - error MS}}{\text{group MS + error MS}} \qquad \text{(Zar, 1984, p. 323)}$$

Correlation coefficients can be compared by calculating a weighted sum of squares (X^2) of the z-values corresponding to the correlation coefficients (Sokal and Rohlf, 1981). The significance of X^2 can be calculated by comparison to χ^2 tables with k-1 d.f.

I.3 PERIODOGRAMS AND CUMULATIVE PERIODOGRAMS

A periodogram is a time series summary tool that examines cyclic patterns more directly than other methods such as correlograms (Diggle, 1992, p. 43). A periodogram summarises a set of time series data as a superposition of sinusoidal waves of various Fourier frequencies (Diggle, 1992). Periodograms consist of plots of **periodogram ordinates** (ordinate) against the **frequency of cyclic fluctuations** (abscissa). To aid interpretation, for the periodogram presented in this thesis (Fig. 6.20) the frequency of cyclic fluctuations is replaced by cycle length. Large peaks in the periodogram correspond to cyclic fluctuations at particular cycle lengths.

If the observed peaks in a periodogram are due to **white noise** (i.e. random variation) alone, the periodogram ordinates should differ only because of sampling variation (Diggle, 1992, p.54). In this case a cumulative periodogram should increase in an approximately linear fashion. A test for departure from white noise is significant horizontal deviation of the cumulative periodogram from the y = x equivalence line. If the deviation from the equivalence line is greater than the 5% critical of Bartletts D statistic the null hypothesis of white noise is rejected (Diggle, 1992, p.55).

APPENDIX II

Patterns of Movement of Some Fishes Within an Estuary in Tropical Australia.

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