Queensland **the Smart State**

Environmental flows for sub-tropical estuaries:

understanding the freshwater needs of estuaries for sustainable fisheries production and assessing the impacts of water regulation

Compiled by Ian Halliday and Julie Robins

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Freshwater resources of Australia are limited and under increasing demand for human use (i.e. urban, industrial and agricultural). However, the environment is a legitimate water user and must have an allocation (= environmental flows) under state and Commonwealth legislation. Estuaries are the downstream section of rivers and streams, and freshwater flowing into estuaries is one of the key factors defining an estuary. However, how much water is required to sustain estuarine biota and the impact of changing freshwater flows on estuaries and the species contained within them is not fully understood and is rarely quantified (Chapter 1).

This project was initiated in response to increasing requests about the freshwater requirements of estuarine fisheries (e.g. for water management planning). In addition, the effects of extended droughts and climate change on estuarine fisheries species require greater understanding of environmental influences on estuarine-dependant stocks. Many estuarine fishery species are likely to be highly responsive to the flood-drought cycle of northern Australia, but the lack of quantitative information precludes the inclusion of such factors in stock assessment.

Results of the current work provide quantitative evidence that freshwater flows to estuaries are crucial in supporting fisheries production, through effects on fish movement (i.e. catchability and access to nursery habitats), recruitment (i.e. number of fish surviving the 1st year of life), and growth rates (of fish and prawns), where faster growth leads to better survival and more individuals in the population. As such, freshwater water flowing to estuaries is not wasted.

We adopted a structured approach to investigating the effects of freshwater flows on estuarine fisheries production. Firstly, we developed conceptual models (i.e. diagrams) of the life-cycle of selected fishery species (e.g. barramundi, banana prawns, mud crabs, king threadfin, blue threadfin and mullet) and super-imposed how and when freshwater flows might affect these species. This process was formalised into a generic framework that could be applied to any Australian estuary (Chapter 2).

Secondly, we analysed commercial catches of fishery species, in the Fitzroy River and Port Curtis estuaries of central Queensland as a case study (Chapter 3). Barramundi and banana prawns were significantly influenced by summer flow and rainfall. For both species, catch was positively correlated with flow in the same year. Barramundi catches were also significantly and positively correlated to flow lagged by three and four years, suggesting a recruitment effect. Correlations with larger scale climatic variables (e.g. Southern Oscillation Index, Madden Julian Oscillation and sea surface temperature), showed similar correlations with lags of four years significant (Chapter 4). As a comparison, catches of barramundi in Princess Charlotte Bay (north Queensland) were significantly correlated to summer flow and rainfall in the same year, as well as to flow and rainfall two years previous; again suggesting a recruitment effect. Recreational catches of summer whiting in central Queensland were positively influenced by summer freshwater flow two years previous, with evidence that this was a result of increased growth and recruitment in high flow years (Chapter 5).

Thirdly, we looked at the age-structure of barramundi, king threadfin and summer whiting catches to see if strong and weak year-classes (i.e. the number of fish surviving the first year of life = recruitment) persisted through time and were correlated with freshwater flow (Chapters 5 and 6). The year-class strength of barramundi, king threadfin and summer whiting were positively correlated with freshwater flow (and coastal rainfall) in spring and summer. We speculate that freshwater flows (and/or coastal rainfall) are important in delivering nutrients to the estuary, thereby creating environmental conditions favourable to species eaten by juvenile barramundi, king threadfin and summer whiting. Freshwater flows may also allow access to or from (or create) nursery habitats that are important for sustaining estuarine fish populations.

Fourthly, we analysed ANSA tag-recapture data of central Queensland barramundi to see if freshwater flows effected growth rates (Chapter 7). They did. Barramundi grow seasonally, being faster in summer, late spring and early autumn than in winter, when no growth occurs. After accounting for seasonal effects, growth rates varied significantly with freshwater flow, being faster at higher flow rates. However, there were flow thresholds beyond which growth rates did not change. We speculate that faster growth rates of barramundi may result from increased food availability that occurs when flows deliver nutrients (e.g. carbon and nitrogen) to the estuary. Additional analysis of otolith micro-chemistry of Fitzroy River barramundi (Chapter 8) showed that growth rates were enhanced in fish that accessed freshwater habitats as juveniles.

Fifthly, we analysed length-frequency data of juvenile banana prawns in the Fitzroy, Calliope and Boyne River estuaries (Chapter 9). Our results suggest that freshwater flows significantly increase the growth rates of juvenile banana prawns, which leads to a greater biomass of banana prawns in the estuary. In addition, the greater abundance of juvenile banana prawns was observed in years with increased freshwater flow, as were commercial catches of banana prawns. The migration and catchability of banana prawns is also known to be affected by freshwater flow.

In addition, we sampled the animals living near the seafloor (=substrate) of the estuary (i.e. the demersal community) to look at the effects of freshwater flows on the biodiversity of estuaries (Chapter 10). Estuarine demersal communities occur in three states: (i) a ‘before flow’ state, when planktonic feeders dominate the community; (ii) a ‘during flow’ state when most animals disappear, presumably leaving because of physiological stress, although demersal feeders remain; and (iii) an after flow state, which is a recovery stage when the estuary (and its animal residents) returns to the ‘before flow’ condition, as waters become less turbid and more salty, and the planktonic feeders reappear. This cyclicity of change occurs annually with the wet season, and indicates that there are animals that exploit all facets of the flow regime. Creating a steady state flow regime would cause major perturbations to the natural faunal community of the estuary. Less dramatic alterations of the freshwater flow regime would have more subtle impacts on the demersal community and the biodiversity of an estuary, but requires more work to elucidate.

Understanding the role of freshwater flow on fisheries species and their productivity (e.g. catch) has important implications for water and fisheries management. During extended low flow decades (e.g. 1960s, 1980s and 2000s), the size of estuarine fish populations are probably reduced as a consequence of successive years of low recruitment. During such times, fisheries populations are likely to be at greater risk from fishing pressure, water abstraction and other anthropogenic impacts. During extended high flow years, estuarine fish populations probably increase in size and are likely to be at lower risk of over-fishing and water abstraction may have lesser impacts, although this is unquantified. The most critical period to ensure freshwater flows to estuaries occur is probably during and just after extended dry periods. Only through greater quantification of the key aspects of freshwater flows and subsequent effects on estuarine fisheries, will we be able to sustainably manage water and fisheries resources, so that the maximum amount of water is available for human use, whilst minimizing impacts on the environment.

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Objectives

1. To review the current knowledge of the relationship between freshwater flows and estuarine fisheries production.
2. To develop a logical framework for investigating: (i) the role of freshwater flow; and (ii) the effects of modified flows, on estuarine fisheries production.
3. To correlate historical flow and fisheries production data of sub-tropical estuaries.
4. To develop procedures for assessing the changes in Queensland's estuarine fisheries production that result from water abstraction and regulation.
5. To develop and communicate guidelines on environmental flows for estuarine fisheries to water managers, water users, the fishing industry and the general community.

Outcomes achieved to date

The project has clearly shown that there are substantial benefits to fisheries production from allowing freshwater flows to reach the estuary. These can be viewed as increasing catch though the delivery of fish from freshwater reaches to estuarine sections, facilitating movement of newly recruited fish from marine spawning areas to estuarine and freshwater habitats, and increases in growth rates of fisheries species probably through the delivery of nutrients into the estuary. These results have been received with enthusiasm amongst the fishing community, and water resources managers but less so in the fisheries management sphere were it is thought that the effects of water (managed or natural changes) are outside their sphere of control. The work presented in this report will assist in developing monitoring programs for water resource plans with respect to measuring estuarine health as well as providing a framework for assessing the need for freshwater to flow to estuaries in areas outside the present study region. The current and increasing demand on Australian water resources requires that water efficiencies (i.e. achieving the same effect with less water) are made by all water users. In the case of the environment, if key aspects of the flow regime and the subsequent effects on estuarine fisheries are identified and understood, then this may allow the maximium amount of water to be available for human use, whilst minimizing impacts on the environment.

KEYWORDS: **environmental flows, freshwater flows, estuarine fisheries, water regulation**

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Background

Estuaries are semi-enclosed bodies of water that open to the sea and are supplied with freshwater draining from the land via rivers and streams. Freshwater inflow is one of the key factors that defines an estuary and contributes to the biological and physical attributes that make it an important spawning, nursery and feeding habitat for many commercial and recreational fish and crustacean species.

In estuaries and near-shore waters of tropical and sub-tropical Australia, there are commercial fisheries for penaeid prawns - banana prawns (*Penaeus merguiensis*, *P. indicus*), tiger prawns (*P. esculentus*, *P. semisulcatus*), endeavour prawns (*Metapenaeus ensis*, *M. endeavouri*), finfish - barramundi (*Lates calcarifer)*, king threadfin (*Polydactylus macrochir*), blue threadfin (*Eleutheronema tetradactylum*), golden snapper (*Lutjanus johnii*), black jewfish (*Protonibea diacanthus*), grunter (*Pomadasys kaakan*, *P. argenteus*), grey mackerel (*Scomberomorous semifasciatus*), mullet (*Mugil cephalus*, *Liza vaigiensis*, *L. argentea*), queenfish (*Scomberoides lysan*, *S. commersonianus*), sharks (*Carcharhinus tilstoni*, *C. sorrah*) and mud crabs (*Scylla serrata*), having a combined annual value of over $220 million AUD. Some species are also important recreationally and to indigenous communities (e.g. barramundi and mud crabs). Many of the above species are dependent on estuarine ecosystems during their life-cycle and are influenced by freshwater flowing into the estuary. In tropical and sub-tropical Australia, freshwater inflow to the estuary is influenced by the seasonal summer monsoon trough, with increased flow during late spring and summer, and decreased flow during winter and early spring. The timing, duration and magnitude of freshwater flowing to estuaries may change as a consequence of the development water resources, and such, these changes will impact upon estuarine species (Drinkwater and Frank 1994; Gillanders and Kingsford 2002).

Water resources are highly developed in some areas of tropical and sub-tropical Australia (e.g. Queensland east coast) and are being planned for ‘development’ in others (e.g. Gulf of Carpentaria and Northern Territory). Within Australia, and worldwide, there is increasing recognition of the need to allocate water for the environment as part of the sustainable use of water resources (Dyson *et al.* 2003; Davis and Hirji 2003 a, b, c). Water for the environment is referred to as environmental flow allocations (Tharme 2003), freshwater inflow needs (Powell *et al.* 2002), or freshwater inflow requirements (Adams *et al.* 2002). In Australia, environmental flows are allocated to maintain the health and viability of water-dependent ecosystems (including estuaries) in catchments where water resources are managed.

What does water do in estuaries?

Studies in Australia and overseas have documented the responses of estuarine communities to the inflow of freshwater by correlating increased river flow (or rainfall) with increased catches of prawns, crabs and fish. While the link between freshwater flows and fisheries catches has been documented in some areas for some species, the causal mechanisms are complex and poorly understood. In most cases, the amount of freshwater inflow required to sustain fisheries is unknown. There are three main proposed causal mechanisms: (i) that freshwater may enhance the overall biological productivity of estuaries by providing an input of organic and inorganic matter (including nutrients) that drives the lower end of the food chain (e.g. phytoplankton and bacteria), and which has flow-on effects for the growth and survival of species at higher trophic levels; (ii) that freshwater flows alter the accessibility of important nursery habitats such as coastal lagoons and floodplains, thereby improving recruitment (and subsequent abundance) of estuarine species; and (iii) that freshwater flows may affect catch rates in estuarine and coastal fisheries by triggering behavioural responses in some species to reduced salinity (e.g. movement and spawning). It is likely that all three mechanisms occur, but to what extent and how they interact to influence total fisheries catches is unclear.

Anthropogenic changes to freshwater flows into estuaries

Regulation and abstraction of water has been necessary to develop land-based activities such as agriculture, industry, and cities and towns. Consequently, most Australian river systems have been modified through the construction of dams, weirs and barrages. In Queensland, only a handful of coastal rivers south of Cooktown have no or only minor water regulation, and there are 23 major dams proposed for rivers throughout the state. Therefore, there is great potential for the further reduction of freshwater flows to estuaries in the future. The effects of water regulation and abstraction have already modified the timing, quantity and quality of freshwater flows sufficiently to pose a threat to the sustainability of estuaries. There is limited documentation on the extent of flow modification and the probable effects on the long-term productivity and biodiversity of Queensland estuaries. Despite this, there is an increasing awareness of the need to manage freshwater flows to ensure the sustainability of downstream environments.

How is water managed and how has water management changed recently?

Freshwater is a limited resource and there has been increasing allocation-related competition between water users. Freshwater resources are managed under State or Territory law, and until recently, the management process involved balancing the needs of traditional non-estuarine water users. In 1994, the Coalition of Australian Governments (COAG) signed the Water Reform Agreement and in the process identified the environment as a legitimate water user. The Agreement restricts trading rights associated with water allocations unless water has also been allocated to the environment. These allocations, referred to as ‘environmental flows’, are aimed at protecting the health of natural ecosystems, while providing security of supply to water users. The COAG Agreement also specifies that environmental flows must be determined through a structured, transparent process. Water management processes vary between states and territories, but generally consider available scientific advice and include community and stakeholder consultation.

Environmental flow allocations

In Queensland, Water Resource Plans (which are secondary legislation to the Queensland *Water Act 2000*) stipulate the ‘outcomes for sustainable management of water’ and include a subsection on ‘ecological outcomes’. The Plans also stipulate environmental flow objectives, water allocation security objectives and performance indicators of water allocations. The allocation of environmental flows has focused on the needs of freshwater ecosystems, despite flows to the estuary frequently being modified extensively. This largely has been due to the limited quantitative information on the needs of estuarine ecosystems and the lack of appropriate methods. This has frustrated estuary-based stakeholders (e.g. commercial and recreational fishers). The applicability of environmental flows determined for freshwater ecosystems to the estuary is unknown and untested. For example, first-post-winter-flows are thought to trigger spawning in freshwater fish, but no estuarine fish species is thought to spawn as a consequence of freshwater flow per se. Freshwater flows may trigger estuarine fish species to move downstream, with spawning cues related to other environmental parameters (e.g. day-length and lunar phase). Increasing the understanding of the role of freshwater in estuaries will assist in having the needs of estuaries and their stakeholders duly considered during water management processes.

Feasibility of allocating freshwater flows for estuaries

Allocation of freshwater for the specific purpose of sustaining estuaries is a new concept in water management that has resulted from legislative change. As such, there is no Australian example of a premeditated and reasoned allocation of freshwater flows for estuaries. However, there are several examples from overseas where environmental flows have been allocated specifically to ensure the sustainability of the estuary. Two notable examples are (i) Texas bays and estuaries where sustaining fisheries production is a legislative requirement of water management and (ii) San Francisco Bay. Project staff have gathered information and contacted personnel associated with these examples. The information requirements and feasibility of applying these methods to Australian estuaries was investigated.

What implication does the research have for fisheries management?

Management of fisheries resources aims to achieve ecologically sustainable development (ESD). An essential part of ESD-oriented management is to move beyond monitoring single-species by adopting whole-of-ecosystem approaches and considering effects additional to fishing (e.g. habitat degradation, pollution and altered flow regimes). However, efforts to move towards a whole-of-ecosystem management approach are hampered by our poor understanding of the ecosystem processes that underpin commercial and recreational fisheries production. Increased knowledge of freshwater flows as a major driver of fisheries production for some estuarine species will assist population modellers and fisheries managers to consider environmental factors and the impacts of non-fishing activities on fisheries catches.

Estuarine fisheries of Queensland and the CZCRC management study areas

About 75% of Queensland's commercial and recreational fisheries are comprised of species that rely on estuarine water conditions and habitats for at least part of their life-cycle. These species include barramundi, king threadfin, bream, whiting, mullet, tailor, prawns and mud crabs. They are important for commercial fishers in many sectors of the industry i.e. inshore net, crab and trawl. The Coastal Zone CRC focused on three main study estuaries: (1) Moreton Bay representing an urbanised catchment, (2) Port Curtis representing an industrialized catchment and (3) the Fitzroy estuary, representing an agricultural catchment.

The research presented within focused on the estuaries of the Port Curtis and Fitzroy regions in central Queensland as case studies for environmental flows, with an additional comparison to Princess Charlotte in far north Queensland. Although the studies were restricted in area, the species studied are widespread in tropical and sub-tropical waters and the results are important for decision making in many estuaries in Australia.

Need

Freshwater allocations to sustain fisheries

The 1994 COAG Water Reform Agreement and various state legislation (e.g. *Water Act 2000* in Queensland) require that managers allocate water to maintain downstream ecosystem health. One aspect of ecosystem health is estuarine and coastal fisheries production. Information on the role of freshwater in maintaining the productivity of commercial and recreational fisheries is needed to ensure that estuaries and their stakeholders are duly represented in water allocation processes. Managers of fisheries and water resources need to be made aware of the fishing industry's vulnerability to the impacts of non-fishing activities, such as water regulation. This issue has been identified as a challenge for the fishing industry in reaching sustainable production (see Challenge 1 of FRDC's R&D Plan 2000, page 59).

Logical frameworks for research leading to new procedures and methods

A logical framework needs to be developed for investigating the role of freshwater flows in estuaries. Procedures to assess the impacts of current and proposed water infrastructure in Australian coastal rivers on estuarine fisheries also need to be developed. Methods for monitoring biological responses to environmental flows are needed to provide feedback to managers as to whether desired fisheries-related outcomes are being achieved under current water allocations.

Enhancing the research outcomes - integrating across research disciplines

An integrated research program is needed to develop a robust sampling program that can investigate the role of freshwater flows in estuaries and the impacts of modified flows on fisheries production. The Coastal Zone CRC offered the opportunity to integrate flow-influenced fisheries data with other hydrological (i.e. coastal modelling) and primary production (i.e. nutrient cycling) research projects to provide greater insight into ecosystems processes.

Chapter 1. Review of rainfall/river flow and estuarine fisheries landings

J. Robins, I. Halliday, and J. Staunton-Smith [1]

Introduction

Estuarine fisheries species and their productivity (i.e. catch) have been proposed as tangible indicators of estuarine ecosystem health linked to environmental flow allocations. This is because fisheries species have economic and social value, and are often the only species for which sufficient life history information and/or long-term abundance data (in the form of catch) are available. Alber (2002) suggested that linking freshwater flows to important fishery species is a relatively simplistic relationship that can be easily understood by a range of stakeholders even if the underlying mechanisms are not certain.

Drinkwater and Frank (1994) reviewed the general effects that freshwater may have on fish and invertebrates in coastal and marine waters, concluding that the number and geographic extent of examples strongly supports a link between freshwater flow and production of certain estuarine and marine fish and shellfish. Proposed mechanisms for the connection between estuarine fishery species and freshwater flow include: (i) trophic linkages via changes to primary or secondary production that result from the addition of nutrients; (ii) changes in distribution as a consequence of altered (expanded, reduced or connected) habitats; and (iii) changes in population dynamics such as recruitment, growth, survival, and abundance (Copeland 1966; Aleem 1972; Peters 1982; Drinkwater 1986; Drinkwater and Frank 1994; Loneragan and Bunn 1999; Gillanders and Kingsford 2002). Most effects on estuarine fishery species are one or more steps removed (**Figure 1.1**) from the direct changes in physical parameters (e.g. water velocity, salinity, water temperature, turbidity) that result from freshwater flowing into estuaries (Drinkwater 1986; Hart and Finelli 1999; Alber 2002; Kimmerer 2002a).

Numerous studies have analysed the correlative relationship between fisheries productivity (e.g. catch or landings) and environmental variables related to rainfall or freshwater flow. Analyses of correlation between rainfall or freshwater flow and fisheries production can be grouped into: (i) single species analyses or (ii) multi-species analyses. Single species analyses are usually based upon hypotheses about the causal mechanisms between hydrological factors and the ecology/life history of an individual species, with results interpreted in relation to possible causal mechanisms. Multi-species analyses are focused on a region or estuary, and analyses are undertaken for a number of shellfish and finfish species. Results may or may not be interpreted in relation to species-specific mechanisms or to more generic ecological mechanisms (e.g. trophic responses of estuarine food webs). Multi-species analyses have been conducted for fisheries productivity in estuarine or coastal areas of the USA, Canada, Norway, Australia and the Mediterranean (**Table 1.1**).

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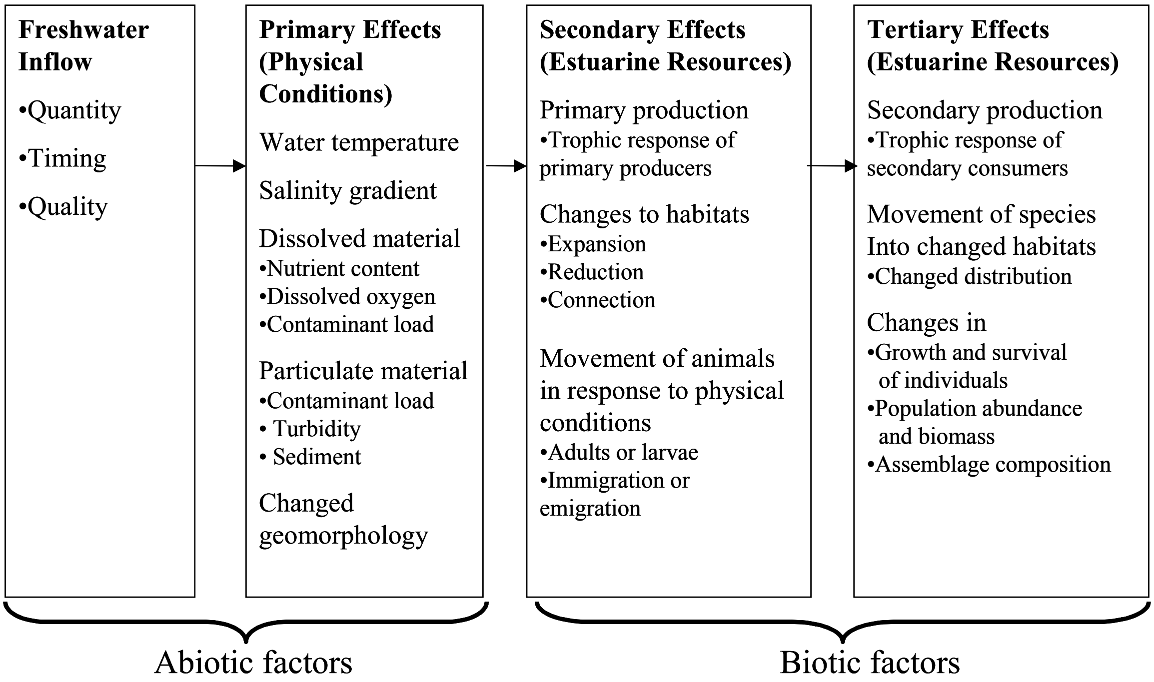
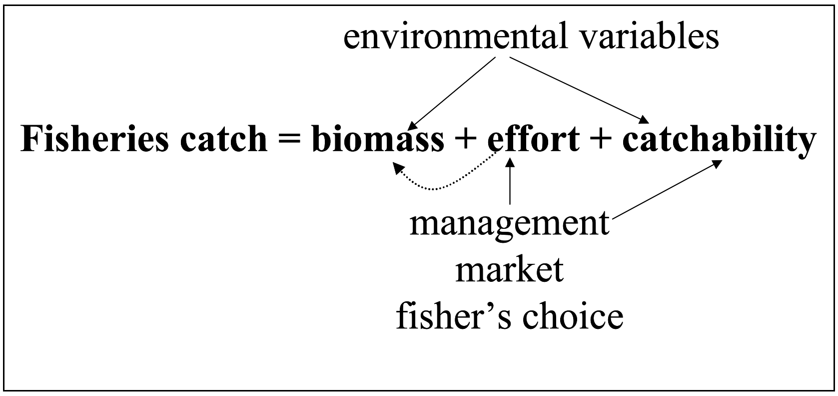
[1] Parts of this chapter are published in: Robins, J.B., Halliday, I. A., Staunton-Smith, J., Mayer, D.G. and Sellin, M.J. (2005). Freshwater flow requirements of estuarine fisheries in tropical Australia: a review of the state of knowledge and application of a suggested approach. *Marine and Freshwater Research* 56, 343-360.

Figure 1.1 Conceptual framework of the ‘order-of-effects’ relationship between freshwater inflow and ecological processes (modified from Hart and Finelli (1999) and Alber (2002))



Fisheries landings are the culmination of numerous factors acting upon the population of a fished species (Figure 1.2). Correlation analysis assumes that freshwater flow (as a specific environmental variable) is an appropriate index of the causal mechanism, despite the influence of other factors (e.g. effort). There is also an assumption that environmental variables have a dominant and over-riding effect on the variation in fisheries landings, i.e. greater than that of factors not considered in the analysis (e.g. effort, spawning stock recruitment relationship). The known or speculated mechanism(s) or processes that link the environmental variable to the fisheries landings should be stated prior to analysis, so that it can be clearly seen how many steps the environmental variable is removed from the causal mechanism(s). Environmental variables are often correlated with each other, creating difficulties in determining the effects of single variables.

Figure 1.2 Simplified representation of factors affecting variation in estuarine and coastal fisheries landings



Data considerations

Rainfall

Rainfall data are readily available for many areas and in many instances as a long time-series. Rainfall has been used as a surrogate for freshwater flow (= river flow), where such flow is not measured directly or is measured unreliably. Using rainfall as an index of runoff or freshwater flow assumes that the relationship between rainfall and freshwater flow is consistent over time and with differing amplitudes of rainfall. This assumption is usually invalid because individual rainfall events result in different amounts of river flow as a consequence of the degree of saturation of the catchment, which depends on the history of rainfall, the distribution of rainfall and rainfall intensity (Glaister 1978). Variability in the relationship between rainfall and runoff or river flow is an unquantified source of error which potentially creates inconsistencies that are difficult to interpret. Rainfall may not be a spatially consistent index of runoff or river flow between different catchments. Rainfall events of equivalent magnitude (i.e. intensity, duration) will result in different levels of freshwater inundation in catchments or estuaries of unequal area (i.e. km^2) (Vance *et al.* 1998). Rainfall scaled by catchment (or estuarine) area maybe a more useful parameter for correlation when only rainfall data are available and comparisons across different catchments are being made. The variability in how rainfall translates into freshwater flow and subsequent inundation of estuarine habitats may account for the inconsistency in relationships between rainfall and catch for some species across different areas (e.g. banana prawns).

Freshwater flow

Freshwater flow (= river flow) data can be gauged (i.e. raw data) or estimated (i.e. modelled data). Gauged flow is often available for locations upstream of the estuarine reach. Estimated flow can be modelled for points within the estuarine reach e.g. End-of-System. Ideally, gauged and estimated flow should include all sources of water input (i.e. runoff) and extraction, and any discrepancies to this should be constant.

Correlating flow with estuarine fisheries landings assumes that there is some relationship between freshwater flow and fisheries production. The consistency of this relationship over increasing volumes, rates or durations of freshwater flow will depend on the hydrology (i.e. tidal and mixing regime) and the geomorphology of the estuary. Like rainfall, freshwater flow may need to be scaled by an appropriate factor (e.g. volume of the estuary) if data from multiple estuaries/catchments are compared.

The appropriate flow statistic for correlation should be related to the proposed causal mechanism between hydrological and ecological patterns. Olden and Poff (2003) recommend selecting hydrological indices that describe unique and non-redundant patterns of variance (in relation to other indices) in hydro-ecological studies. They also note that temporal flow indices should be included because of their critical ecological importance. In the majority of correlation studies summarised (**Table 1.1**), the flow statistic used is magnitude, particularly total volume, at varying temporal scales (i.e. daily, monthly, seasonal or annual). Further consideration needs to be given to other hydrologic indices, such as frequency, timing, duration and rate of change (Olden and Poff 2003).

Water temperature

Water temperature is often included in correlative analyses because of its potential effect on the survival or growth of particular life stages of some species. Air temperature is often used as a surrogate of water temperature, because air temperate is more readily available.

Fisheries data

Fisheries landings are often used as an index of the abundance or biomass of a fished species or assemblage of fished species. Hilborn and Walters (1992) caution of the possibility that fishery-dependent data can be an 'imperfect representation of the actual fishery'.

Fisheries landings are often available for a number of different species. In choosing a species for analysis, consideration should be given to: (i) the likely link between environmental variables and fisheries landings, based on known biology or life history; and (ii) whether the data are reported for single species (e.g. sea mullet: *Mugil cephalus*) or a species assemblage that comprises a market-grouping (e.g. mullet: *M. cephalus + Liza vaigiensis + M. georgii* etc.). The pooling of a species assemblage is important because of the potential for individual species to have significantly different responses to environmental variables, such as freshwater flow events. This is particularly the case when species or taxa are pooled together in landings data (e.g. total prawns, total fish, total crabs or total landings). For example, penaeid prawn species (= shrimp) are unlikely to show the same response to variation in rainfall or river flow, as a consequence the different preferences of each species for varying levels of salinity. Floods may stimulate most species of prawn to migrate downstream and into coastal waters, resulting in increased catch rates (Ruello 1973; Glaister 1978; Vance *et al.* 1985). However, floods may have considerably different effects on the recruitment of different penaeids. Within Australian penaeids, the eastern king prawn (*Penaeus plebejus)* is more halophilic than *Metapenaeus macleayi* (Ruello 1973) and salinity levels within an estuary may vary from a low salinity habitat that favours that *M. macleayi* to a high salinity habitat that favours eastern king prawns. Therefore, where possible, single species should be analysed separately. If this is not possible, the limitations of the multi-species grouping should be acknowledged and implications discussed.

Measures of abundance: to use catch or catch per unit effort?

It is widely recognised that catch varies as a function of fishing effort for many species. The relationship may be predictable for species that are widely dispersed, such that Catch Per Unit Effort (CPUE) is a robust indicator of their abundance. However, this does not hold for aggregating species or where search time is not included in effort. Units of effort may vary in some fisheries (e.g. soak time for nets). In this case, CPUE is not a robust index.

Vance *et al.* (1985) argue that CPUE is the best indicator of abundance in a fishery with animals having a one-year life-cycle and where the fishery is not fully exploited, because fishing effort is the main cause of fluctuations in the total catch. However, they argue that catch is the best indicator of abundance in a fishery that is heavily exploited, because increases in fishing effort will not increase the total catch by very much. Therefore, the decision to use catch or CPUE depends on the known or assumed exploitation rate of the stock. In fisheries with changing levels of effort, other factors may be of greater importance in inter-annual yield variation at higher effort levels (Evans and Opnai 1995).

Stock-recruitment relationships may need to be included in analyses of fisheries landings data where recruitment overfishing has occurred and spawning stock size has a significant contribution to the inter-year variation in recruitment. This may be in the form of including some index of the remaining spawning stock biomass in the analysis (e.g. landings or CPUE from the previous year).

Analytical approaches

Choosing appropriate spatial and temporal scales

The appropriate spatial and temporal scale at which correlation analyses should be conducted probably depends on the question of interest and the species involved. Many of the papers in the literature conduct analyses at several temporal and/or spatial scales. We suggest that the scale of analyses selected should be based on biological understanding and the hypotheses to be tested.

Quiñones and Montes (2001) suggest that biological characteristics (e.g. spawning and recruitment) are affected at smaller than annual scales (e.g. monthly scale) and that annual scales (of freshwater flow and rainfall) are indicative of the 'cumulative' effect of monthly freshwater flow and rainfall.

Use of running means

Running means are commonly used in climatological studies, but are not commonly used in fisheries production studies. Running means may be appropriate in fisheries studies where the catch is comprised of several year-classes or where the hypothesised effect of freshwater flow is cumulative over several years. Running means are probably inappropriate in fisheries where there are single year-classes or where the hypothesised effect is confined within a single year or month. Quiñones and Montes (2001) used three-year running means for inter-annual analyses of róbalo (*Eleginops maclovinus*) catch with freshwater flow and rainfall to: (i) improve the detection of long-term trends; (ii) reduce high frequency variability in the time series; and (iii) diminish the effects of the age-structure in the annual landings data. However, auto-correlation between data points increases with the use of running means, reduces the effective number of independent data points in a time series (Drinkwater and Myers 1987) and should be adjusted for in the analysis (e.g. Pyper and Peterman 1998).

Lagged variables

Many correlative studies lag the fisheries landings from the environmental variables (e.g. Sutcliffe 1973; Wilber 1992; Skreslet 1997; Lloret *et al.* 2001; Powell *et al.* 2002). The use of lagged variables is justified on the basis that it takes a number of years for the effects of the environmental variables (i.e. on larval or juvenile survival) to be expressed in the fisheries landings and are usually based on the known or speculated age at which the fish or shellfish recruit to the fishery. While correlating fisheries landings with prior environmental variables fits within the biologically-based hypotheses, it tends to assume that the delay between cause (i.e. increased juvenile survival) and effect (i.e. increased fisheries landings) is constant. This is probably a valid assumption for species with very short life-cycles (i.e. one-year) but may not be valid for long-lived species because of variable timing in recruitment to the fishery which may be the consequence of variable growth rates. It is difficult to determine whether this is the case without some 'data snooping' (Potter *et al.* 2001) i.e. trialling a variety of lags then selecting that which gives the 'best fit' (e.g. Quiñones and Montes 2001; Powell *et al.* 2002; Salen-Picard *et al.* 2002). Alternatively, and preferably, lags should be based on other sources of information, such as serial monitoring of the age-structure of the fished population, to provide robust quantitative estimates (and variation) in the time to recruit to the fishery.

Assumed linearity of the relationship

Kimmerer (2002b) and Powell *et al.* (2002) raise the issue that estuarine responses are not always proportional to changes in freshwater flow, as a consequence of an estuary being a three dimensional structure, influenced by tides and other oceanographic features. It is likely that the nature (positive or negative) and form (linear, threshold, parabolic) of relationships between environmental variables and fisheries production varies with the different life history stages of estuarine and coastal fisheries species. To some degree, non-linearity and curvi-linear relationships between variables can be accounted for by transformation of data, often ln (Powell *et al.* 2002) or $\log_{10}$ (Galindo-Bect *et al.* 2000). However, the validity of assumed linearity or the possibility of thresholds should be kept in mind.

In summary

Despite the above limitations, which should be discussed in order to justify the validity of the resulting interpretation (Underwood 1997), correlation analysis can provide insights into potential cause and effect relationships, for guiding water management and associated monitoring (ideally within an adaptive management framework). Correlation analysis can also identify areas for further research on the relationship between the environmental and fisheries variables, provided the correlations are conducted within conceptual frameworks with clear objectives as to the reason for exploring the link between environmental and fisheries landings variation.

Modelling (e.g. Maunder and Watters 2003) can assist in the assessment of the relationship between freshwater flow and fisheries productivity, but modelling cannot make up for lack of appropriate data or system understanding (Sharp 1995). Linking freshwater flows to commercially (or recreationally) important species is a relatively simplistic relationship that can be easily understood by a range of stakeholders, even if the underlying mechanism is not certain. The disadvantage is that other non-fished components of the ecosystem may have differing needs for freshwater flows and are not considered (Alber 2002).

Review of published studies

Relationships between catch of estuarine or near-coastal fishery species and freshwater flow (or rainfall as a proxy of freshwater flow) have been reported for more than 20 tropical or sub-tropical species (**Table 1.1**). Relationships between freshwater flow and the commercial catch have also been investigated for temperate species (see Sutcliffe 1973; Sutcliffe *et al.* 1977; Drinkwater and Myers 1987; Ardisson and Bourget 1997; Skreslet 1997; Perry *et al.* 2000; Lloret *et al.* 2001; Salen-Picard *et al.* 2002). Temperate studies include a greater number of finfish and mollusc species, while tropical and sub-tropical studies include a greater number of crustacean species (i.e. penaeid prawns = shrimp).

Correlation or regression analysis to identify environmental variables that contribute to variation in fisheries catch can be criticised because of: (i) the confounding effects of stock size and fishing pressure (Walters and Collie 1988); (ii) the likely non-linearity of linking mechanisms (Baumann 1998) and the probability of multiple mechanisms; (iii) the possibility of type I errors (i.e. false significant correlations, Potter *et al.* 2001); (iv) the lack of ability to prove causality (Quiñones and Montes 2001); and (v) their uncertain predictive capability as a consequence of long-term climatic variation or human-induced changes (e.g. habitat loss or pollution). Whilst an experimental approach is needed to determine causality, manipulative experiments of freshwater flow are rarely practical at the scale of whole estuaries and it is difficult to obtain appropriate controls for 'Before-After-Control-Impact' experiments that utilise ongoing human manipulation of freshwater flows (i.e. water regulation). Therefore, in many instances, observational studies are used to derive insights into the factors driving the distribution and abundance of fisheries species at a whole of estuary scale, with the analysis of relationships between catch and environmental variables often being used for fishery species (Tyler 1992).

Most studies of tropical or sub-tropical species report positive relationships between (fishery) catch and increased freshwater flow (**Table 1.1**). The variables that explain the greatest amount of variation in catch are not consistent and the patterns differ for the same species in different areas and for different species in the same area. However, as discussed below, the proposed mechanisms underlying the observed relationships are relatively consistent within the species groups.

Penaeid prawns

Penaeid prawns (= shrimp) are targeted by major commercial fisheries in tropical and sub-tropical Australia, as well as many other areas of the world. Penaeid prawns are short-lived (i.e. one to two years), opportunistic omnivores (Chong and Sasekumar 1981). Many species of penaeid prawn are dependent on estuarine habitats for part of their life-cycle, but have different habitat preferences, tolerance to and degree of emigration from low salinity water. Freshwater flow (or rainfall) has been related to catch for ten species of penaeid prawn (**Table 1.1**). Most correlations between freshwater flow (or rainfall) and prawn catch have been reported for those species with the greatest tolerance or exploitation of brackish-water habitats. In general, significant positive relationships occur between annual catch and total freshwater flow (or rainfall) in the same or previous year (Gunter and Hildebrand 1954; Ruello 1973; Glaister 1978; Vance *et al.* 1985; Gammelsrød 1992; Galindo-Bect *et al.* 2000). Significant within-year correlations between catch and monthly or seasonal freshwater flow (or rainfall) have also been reported (Glaister 1978; Browder 1985; da Silva 1985; Vance *et al.* 1985; Gammelsrød 1992; Vance *et al.* 1998). The relationship between catch and freshwater flow (or rainfall) is not always consistent between areas, even for the same species. For example, negative correlations between prawn catch and rainfall have been reported in the Gulf of Papua, Papua New Guinea (Evans *et al.* 1997), whilst significant positive correlations between prawn catch and rainfall have been reported for some areas of the Gulf of Carpentaria, Australia (Vance *et al.* 1985). This example demonstrates that the relationship between the prawn catch and freshwater flow should be assessed for individual estuaries to account for hydrological and biological differences between catchments (Vance *et al.* 1998).

The relationships between prawn catch and freshwater flow (or rainfall) are potentially confounded by other factors such as fishing effort and spawning stock size (Browder 1985; da Silva 1985; Vance *et al.* 1985). The degree of influence of these factors depends on the level of exploitation of the population by the fishery (Vance *et al.* 1985), although most of the correlative studies for prawns do not account for these factors.

Suggested causal mechanisms for the observed relationships between the catch of penaeid prawns and increased freshwater flow (or rainfall) include: (i) enhanced emigration of prawns to areas accessible to the fishery, leading to increased catchability (Ruello 1973; Glaister 1978; da Silva 1985; Gammelsrød 1992; Evans *et al.* 1997; Vance *et al.* 1998); and (ii) enhanced growth and survival of various life stages leading to increased abundance or biomass (Ruello 1973; Evans *et al.* 1997), through enhanced recruitment from enlarged nursery areas (Browder 1985; Gammelsrød 1992; Galindo-Bect *et al.* 2000), and or enhanced food availability from increased primary and secondary production (Loneragan and Bunn 1999).

Whilst the conclusions of most correlative studies are speculative, other evidence supports the likelihood of these mechanisms. For example, emigration rates of juvenile banana prawns from estuaries to near shore areas is strongly linked to rainfall events (Staples and Vance 1986; Vance *et al.* 1998) and emigration rates are significantly correlated with commercial catches (Staples and Vance 1986; Staples and Vance 1987; Vance *et al.* 1998). Experimental studies demonstrate that the growth of penaeid prawns is influenced strongly by temperature and salinity, and that there are temperature-by-salinity optima for each species (Staples and Heales 1991; Haywood and Staples 1993; Vinod *et al.* 1996). The possibility of a salinity-by-temperature optimum that influences the growth and survival of prawn life stages potentially explains the parabolic relationship between prawn catch and rainfall in the Gulf of Papua suggested by Evans *et al.* (1997), where high rainfall (= high freshwater flow and low salinity) reduces the immigration and survival of larvae and post-larvae, intermediate rainfall (= intermediate freshwater flow and intermediate salinity) stimulates emigration to offshore waters, and low rainfall (= low freshwater flow and high salinity) reduces emigration to offshore waters. Enlarged nursery areas may occur as a result of freshwater flows providing favourable salinity ranges over the greatest amount of suitable and accessible areas of nursery habitat during critical life stages of pink shrimp (*Farfantepenaeus duorarum*) (Browder *et al.* 2002).

Kimmerer (2002b) recommends that greater knowledge of the causal mechanisms and the biology of the species-of-interest is needed for standards or indices of freshwater inflow to be biologically effective or to be used to make water efficiencies (i.e. achieve the same effect with less water). In the case of pink shrimp, Browder *et al.* (2002) recommended further research into the seasonal and annual patterns of the availability of post-larvae and the timing and magnitude of immigration and influencing factors. Whilst there is strong evidence of the influence of freshwater inflow on the emigration of prawns from an estuary, there is little supporting evidence that freshwater inflow to estuaries results in greater abundance or biomass of prawns as a consequence of improved growth and survival.

Finfish fisheries

Finfish are often selected for analyses of the relationship between environmental variables and fisheries production (Table 1.1) because many species are estuarine-dependent for part of their life-cycle. However, finfish can be short or 'long' lived (i.e. >10 years) and depending on the species, and may occupy various trophic levels. Finfish are commonly considered in multiple species analyses, pooled as total fish landings, or are considered as individual species with species-specific time lags (Table 1.1).

Relationships between commercial catch and freshwater flow have been published for seven tropical or sub-tropical finfish species, and are focused in estuarine or coastal areas of the USA, Mexico, South America and Australia (Table 1.1). Whilst not a quantitative study, Aleem (1972) described the dramatic decrease in the catch of *Sardinella* sp. in the Mediterranean Sea as a consequence of damming the Nile River and the elimination of floods. In sub-tropical Australia, significant positive correlations between catch and freshwater flow were found for mullet (*Mugil* spp.) and flathead (*Platycephalus* spp.) (Loneragan and Bunn 1999). In central-south Chile, significant negative correlations between catch and freshwater flow were reported for róbalo (*Eleginops maclovinus*) (Quiñones and Montes 2001); whilst in the USA, catches of red drum (*Scianeops ocellatus*), black drum (*Pogonias cromis*) and spotted seatrout (*Cynoscion nebulous*) have been both negatively and positively related to freshwater flow aggregated into two-monthly totals (Powell *et al.* 2002). Other analyses correlating finfish fisheries production with environmental variables related to rainfall and river flow include studies on barramundi (*Lates calcarifer*) (Sawynok 1988), black bream (*Acanthopagrus butcheri*) (Walker *et al.* 1998), and common sole (*Solea solea*) (Salen-Picard *et al.* 2002).

Suggested causal mechanisms for the observed relationships between finfish catch and freshwater flow include: (i) changes to catchability (Loneragan and Bunn 1999); (ii) changes to cohort or year-class strength during the first year of life (Quiñones and Montes 2001); and (iii) changes to food availability via productivity changes resulting from flow-borne nutrients (Aleem 1972; Salen-Picard *et al.* 2002). Effects on catchability are suggested where correlations between catch and freshwater flow occur within a relatively short period suggesting an immediate response (e.g. within the same year for annual correlations). In southern Queensland, increases in catchability are proposed to be the consequence of restricting the distribution or stimulating the movement of flathead species (Loneragan and Bunn 1999). Anecdotal reports from commercial fishers in tropical Australian estuaries suggests that barramundi may also be stimulated to move by freshwater flows, both from upstream habitats as well as moving around within estuarine habitats, thus increasing their catchability in passive fishing gear (i.e. set gill nets). However, the movement of barramundi from upstream habitats requires that individuals be abundant in such habitats.

Effects on year-class strength are suggested where significant correlations are between catch and prior freshwater flow. The proposed mechanisms include: (i) advection (negative effect) or retention (positive effect) of eggs and larvae in nursery areas; (ii) increased predation (negative effect) on young-of-the-year; (iii) expansion of suitable reproductive and nursery habitats; and (iv) improved food availability for larvae and juveniles (Quiñones and Montes 2001; Salen-Picard *et al.* 2002). Few studies have investigated the causal mechanisms in detail, although Kimmerer *et al.* (2001) reassessed the effects of freshwater flow on the early life history of striped bass (*Morone saxatilis*) in the San Francisco estuary. They found that strong relationships existed between freshwater flow and survival from eggs to young-of-the-year, but that recruitment of three-year-olds was not related to freshwater flow during the early life stages. North and Houde (2003) also studied the effects of freshwater flow on the early life history of striped bass and white perch (*M. americana*) in upper Chesapeake Bay (a temperate estuary). They reported that freshwater flow conditions were strongly related to physical conditions, prey concentrations and larval fish distributions associated with the estuarine turbidity maxima.

These three effects (i.e. catchability; recruitment in terms of abundance; and productivity) are not mutually exclusive. It is likely that each of these mechanisms contributes to fluctuations in catches of finfish species that are significantly correlated to freshwater flows to estuaries.

Other species – crabs, oysters and octopus

Crabs are common in tropical and sub-tropical estuaries throughout the world, and are the target species of many fisheries. Relationships between commercial catch and freshwater flow have been investigated for the mud crab (*Scylla serrata*), which is the main commercial crab species in tropical Australia, as well as for blue swimmer crabs (*Portunus pelagicus*) and blue crabs (*Callinectes sapidus*) (**Table 1.1**). Annual catches of mud crab in southern Queensland are positively correlated with summer freshwater flow. The causal mechanisms suggested are: (i) an increase in catchability, resulting from freshwater flow stimulating downstream movement; and (ii) an increase in the survival of juveniles through reduced cannibalism and competition for burrows as a consequence of the emigration of adult crabs (Loneragan and Bunn 1999). Blue swimmer crabs and blue crabs are con-specifics, occurring in estuaries of the southern and northern hemispheres respectively. No significant correlations between catch and seasonal freshwater flow were reported for blue swimmer crabs (Loneragan and Bunn 1999), but seasonal freshwater flow explained a significant proportion of the variation in the annual catch of blue crabs in numerous estuaries of the USA (Meeter *et al.* 1979; Funicelli 1984; Powell *et al.* 2002).

Oysters are an attractive species to use as an index of estuarine responses to freshwater flow because they are sedentary, individuals can be measured over time for aspects such as growth and mortality, and spatfall (i.e. recruitment) can be readily measured using artificial collectors (Livingston *et al.* 1997). Oyster harvest has been negatively correlated to freshwater flow in the same year (Meeter *et al.* 1979; Wilber 1992) and positively correlated to freshwater flow in previous years (Wilber 1992). Significant negative and positive relationships between oyster harvest and seasonal freshwater flow were also reported by Powell *et al.* (2002). Freshwater flow is suggested to negatively affect growth (Meeter *et al.* 1979), mortality and spawning (Wilber 1992). Livingston *et al.* (2000) suggested that freshwater flow affects oyster production through two mechanisms: (i) predation and disease related to changes in salinity; and (ii) growth effects related to changes in the trophic productivity of the estuary. Livingston *et al.* (1997) suggested that oysters may be an appropriate index of the response of estuaries to freshwater flow because they appear to need a delicate balance of freshwater flow, requiring lowered salinities to minimise predation and disease, but not so low as to impact on growth rates (i.e. salinities not too low). However, if using natural oyster populations (*c.f.* cultured populations in oyster racks) then changes in available habitat need to be considered (i.e. if suitable habitat is reduced as a consequence of turbidity etc.).

Significant negative correlations between catch and freshwater flow are reported for octopus (*Octopus vulgaris*) in the Gulf of Cadiz, Spain, but not for cuttlefish (*Sepia officinalis*, Sobrino *et al.* 2002). Fluctuations in octopus catch are suggested to be a consequence of environmentally driven variation in recruitment and that freshwater flow (= river flow) changed environmental conditions, stimulating the movement of octopus from their dens.

In summary

The additional studies reported in the literature since the review of Drinkwater and Frank (1994) reinforce the conclusion that the catch of some estuarine and marine finfish and shellfish is strongly linked to freshwater flow in tropical, sub-tropical and temperate areas. The correlative studies demonstrate that seasonality is often as important as volume (Loneragan and Bunn 1999), and that freshwater flow requirements of fished species need to be assessed on a species-by-estuary basis. Relationships between the catch of estuarine species and freshwater flow have been included in models designed to optimise freshwater inflows over specified physical, chemical and biological constraints in Texas, USA (Bao and Mays 1994; Powell *et al.* 2002). However, this method depends on large quantities of catch data, which are unlikely to be available for most estuaries.

Drinkwater and Frank (1994) recommended that more quantitative research is needed into the relationship between freshwater flows for fish and ecosystems, including multi-disciplinary studies and integrated physical-biological models. This would lead to greater knowledge of the mechanisms underlying the relationships between catch and freshwater flows and would help to determine which aspects of the flow regime were important. Kimmerer (2002a,b) and Browder *et al.* (2002) reiterated the need to understand the causal mechanisms in the relationship between freshwater flow and (the biology of) the species of interest. Conceptual models of the role of freshwater flow in estuarine ecosystems, and hypotheses developed from them, need to be explicitly considered in order to direct multi-disciplinary studies, and to provide the conceptual structure for integrated bio-physical models.

Table 1.1 Summary of studies correlating fisheries production with freshwater flows to estuaries

Source, Location, Data & Analysis	Results	Discussion points
Gunter and Hildebrand 1954. USA, Texas • **White shrimp,** *Penaeus setiferus* • Calendar year commercial catch for Texas • Average annual Texas rainfall • 1927 to 1952 (26 yrs) • *Analysis*: (i) correlation; 8 combinations of catch and rainfall	• 7 of 8 relationships tested were significant • Annual catch significantly correlated with: (i) previous yr's rain ($p<0.05$); (ii) same plus previous yr's rain ($p<0.05$); (iii) 2 previous yr's rain ($p<0.01$), but not with same yr's rain • 2-yr running average of catch significantly correlated with 2-yr running average of rain one-yr previous ($p<0.01$) • 3-yr running average of catch significantly correlated with: (i) 3-yr running average of rain, same yrs ($p<0.01$); (ii) 3-yr running average of rain, same yr & 2 previous yrs ($p<0.01$)	• Discusses the possible influence of other factors: technical improvements, growth of the fishery, economics, overfishing • Speculated mechanism is change in salinity • Lag in catch explained by the time it takes for rainfall to change the salinity of inshore waters
Ruello 1973. Australia, New South Wales • **School prawns,** *Metapenaeus macleayi* • Monthly research trawls estuarine & coastal waters 67/68 & 69/70 • Commercial landings, 1954 to 1971 (financial yrs) • Rainfall and Hunter River flow • *Analysis*: (i) correlation	• Catch not significantly correlated to same yr rainfall • Catch significantly correlated to preceding yr rainfall ($r=0.64$, $p<0.05$) • Small flows stimulate large prawns to move (emigrate), large flows stimulate all but very small prawns to move • Significant river flows led to the movement of school prawns from the estuary to the ocean, resulting in increased density & commercial catch.	• Raises the issue of "effective rainfall" • Speculates that flow has a cumulative effect on reproduction, recruitment, growth & survival of all life stages • Emigration may be a response to the disturbance of bottom sediments by flows, interfering with normal burrowing & respiratory activity
Glaister 1978. Australia, New South Wales • **School prawns,** *Metapenaeus macleayi* • Mean rainfall and Clarence River flow • Estuarine & oceanic catch, effort & catch per unit effort (CPUE) • Daily, weekly, monthly & yearly (financial) data, 1966/67 to 1975/76 • *Analysis*: (i) correlation, (ii) multiple linear regression	• 29 of 103 correlations tested were significant • Weekly oceanic catch significantly correlated with same week ($p<0.05$) & previous week flow ($p<0.05$). • Monthly estuarine catch significantly correlated with previous month flow ($p<0.01$) • Monthly oceanic catch significantly correlated with same month ($p<0.01$) & previous month flow ($p<0.01$) • Annual oceanic catch significantly correlated with same yr river flow ($p<0.01$), but estuarine catches were not	• Concludes that flows enhance seasonal emigration of school prawns to offshore waters, which determines the magnitude of oceanic production • Suggests that modification of flow could have a detrimental effect on the "normal harvest" of the fishery
Hunt *et al.* 1980. USA, North Carolina • **Brown shrimp,** *Penaeus aztecus* • Monthly commercial catch Pamlico Sound, 1972-77 (5 yrs) • Wind, Pamlico River flow, salinity, air & water temp., rainfall • *Analysis*: multivariate regression to develop a predictive harvest model	• Juvenile abundance highly correlated with commercial harvest ($r=0.94$) • Harvest correlated with water temperature & salinity • River flow was poorly predicted salinity, so used salinity as the predictor of catch	• Salinity of 10 & water temp. of 20°C are threshold values acting on post larval settlement & juvenile growth (April & May) • These levels determine subsequent production

Vance *et al.* 1985. Australia, Gulf of Carpentaria - **Banana prawns,** *Penaeus merguiensis* - Commercial catch 1970 to 1979 - Research data juveniles for Norman River - Rainfall, wind and air temp. - Gauged river flow - June to May grouping of data - Seasonal data (used in multiple regression) - *Analysis*: (i) single variable correlation analysis and (ii) forward-selection multiple regression analysis	- Commercial prawn catch not significantly correlated to juvenile abundance in Karumba region - 10 of 30 correlations between mean seasonal or total annual rainfall & regional prawn catch significant. Regional prawn catch significantly correlated with: (i) Mitchell winter rainfall ($r=0.89$, $p<0.01$); (ii) Karumba spring ($r=0.76$, $p<0.01$), summer ($r=0.80$, $p<0.01$) & annual rainfall ($r=0.86$, $p<0.01$); (iii) Mornington Is. winter ($r=0.78$, $p<0.01$) & annual rainfall ($r=0.74$, $p<0.01$); (iv) Limmen Bight winter ($r=0.82$, $p<0.01$) & annual rainfall ($r=0.77$, $p<0.01$); (v) Groote Eylandt winter ($r=0.75$, $p<0.01$), spring rainfall ($r=0.69$, $p<0.05$);(vi) Weipa spring rainfall ($r=0.67$, $p<0.01$) - Karumba region catch correlated with summer river flow ($r=0.80$, $p<0.01$) & annual flow ($r=0.82$, $p<0.01$) - 3 of 30 correlations between annual regional prawn catch & mean seasonal or annual temperature significant - 5 of 24 correlations between annual regional prawn catch & seasonal wind component significant - Rainfall models explained between 42% & 72% of the variation in regional banana prawn catches	- Discusses the problem of variables being significant correlated but having no supporting biological mechanism. They did not use these variables in further analyses or predictive models. - The issue of regional variation in environmental factors, possible reflecting the different timing of larval recruitment into GOC estuaries. - Raises the issue that the size of the spawning stock could be influencing juvenile recruitment and subsequent commercial catches. - No flow recommendations.
Gammelsrød 1992, (*Est. Coast. Shelf Sci.*), Mozambique - **Red-leg banana prawns,** *Penaeus indicus* - Commercial catch, effort, size classes, - Zambezi River flow, rainfall - 1974 to 1988 - Used climatic year: Oct to Sept - *Analysis*: (i) linear regression, (ii) stepwise multiple regression, (iii) predicted catch based on discharge see also da Silva 1985	- No significant correlations of catch & rainfall. - 11 out of 18 correlations significant for flow. - Annual CPUE significantly correlated to annual flow ($p<0.01$); annual scaled catch rate (CPUE/total effort i.e. kg per hour^2) significantly correlated to annual runoff ($p<0.01$); - Annual CPUE of big & med. prawns significantly correlated to annual flow ($p<0.05$, $p<0.01$); annual scaled catch rate of big, med. & small prawns significantly correlated to annual flow ($p<0.01$). - Effort most significant variable in the stepwise regressions, but for the last 9 yrs of data (when effort constant), flow was most significant variable, with effort not significant - Uses the regression results to provide predictive equations of catch rate (C) versus wet season (WS) flow: (i) C all species = 34 + 0.8*WS flow ($p<0.05$); (ii) C *P. indicus* = 13 + 0.5*WS flow ($p<0.05$); (iii) C nos. total prawns = 6687 + 249*WS flow ($p<0.01$); (iv) C nos. big prawns = 554 + 95*WS flow ($p<0.05$); (v) C nos. medium prawns = 2975 + 108*WS flow ($p<0.05$)	- Biological mode of wet season runoff, dry season runoff, wet season effort. - Points out that the results suggest hypotheses rather than "solid conclusions". - Postulated mechanisms: (i) lower salinities stimulate the recruitment to the fishery of a greater portion of the population; (ii) larger flows creates a greater amount of "activated" nursery areas for the recruitment process - Notes that timing of floods appears to be important with late floods resulting in greater production - Timing of flow with spring tides might be important
Evans *et al.* 1997. Papua New Guinea, Gulf of Papua - **Banana prawns,** *Penaeus merguiensis* - Catch and effort data 1974 to 1993 - Rainfall (Oct to Sept) - Flow for Fly, Kikori, Purari, Vailala & Lakekamu Rivers - *Analysis*: (i) CLIMPROD, (ii) linear regression	- Only significant negative Spearman Rank correlation for mean monthly rainfall at one station & banana prawn CPUE in the Gulf of Papua for the calendar year, ($r=-0.4434$, $p=0.057$) - Rainfall improved effort-catch models by at most 5%, i.e. annual rainfall added almost nothing to the variation explained by the CLIMPROD effort alone models	- The mechanism between rainfall & yield of banana prawns is speculated to be parabolic: high rainfall lowers the immigration & survival of larvae & post-larvae in estuaries, intermediate rainfall stimulates the emigration of sub-adults into offshore waters, low rainfall delays or reduces the offshore recruitment of juveniles.

Vance *et al.* 1998. Australia, north-eastern Gulf of Carpentaria • **Banana prawns,** *Penaeus merguiensis* • Research sampling post larvae and juveniles • Commercial catch in Albatross Bay • Spring 1986 to autumn 1992 • Temp., Embley River salinity, light penetration & rainfall • Data groupings: pre-wet (Oct-Dec), wet (Jan-Mar); early dry (Apr-June); dry (July-Sept) • *Analysis*: (i) regression, (ii) tested for autocorrelation	• Catch significantly correlated with (i) combined pre-wet & wet season emigrating prawn abundance ($R^2=0.82$, $p<0.01$); (ii) pre-wet & wet season benthic juvenile abundance ($R^2=0.71$, $p=0.03$); • Catch positively but not significantly correlated with (i) pre-wet & wet season total rainfall ($R^2=0.25$, $p=0.31$); or (ii) mean river salinity ($R^2=0.56$, $p=0.09$)	• Suggests that the size of the river catchment area is likely to be the most important factor determining the strength of the correlation between rainfall & offshore catch, as catchment area will affect the duration & volume of freshwater inundation in the estuary
Galindo-Bect *et al.* 2000. Mexico, Gulf of California • **Shrimp, *Litopenaeus stylirostris*** • Annual landings & CPUE (landings/number of trawlers) • Colorado River flow (log10 transformed) & number of trawlers (i.e. effort) • 1977 to 1996 • *Analysis*: (i) correlation, (ii) multiple regression	• Significant downward trend in landings between 1977 & 1996 • 5 of 6 correlations for landings significant • None of the 6 correlations for CPUE with flow & effort were significant • Landings significantly correlated with same yr river discharge ($r=0.54$, $p<0.05$), landings significantly correlated with previous yr river flow ($r=0.67$, $p<0.001$) • Landings significantly correlated with effort ($r=0.77$, $p<0.001$) • Best model was the product of lagged river discharge (log transformed) & effort ($r=0.80$, $p<0.001$): landings = 232 + 1.67*($\log_{10}$ lagged flow*number of trawlers)	• Unknown cause of link between flow & production of the shrimp fishery • Possibly through enhanced survival of early life stages in enlarged nursery areas, although *L. stylirostris* is a euryhaline species with post-larvae & juveniles occurring in hypersaline habitats
Browder 1985. USA, Gulf of Mexico • **Pink shrimp,** *Penaeus duorarum* • Landings of Tortugas shrimp grounds • Standardised effort (monthly) • Quarterly average water level at a station in the Shark River, principal tributary of the Florida Everglades • Air temp. & CPUE lagged by 3 months • *Analysis:* multiple regression	• Standardized effort, water level lagged one quarter & CPUE lagged four quarters explained 88% of the variation in the 14 years of combined quarterly data • Oct-Dec and July-Sept water levels had the greatest influence on annual landings	
da Silva 1985. Mozambique • **Red-leg banana prawns,** *Penaeus indicus* • Total annual catch, monthly fishing effort • Catch data from one company, 1974 to 1983 • Zambezi River flow • *Analysis*: abundance indices analysed by length groups, & related to river runoff	• Best correlations between total catch rate of *P. indicus* & Zambezi runoff were obtained for the recruitment period with a 1-month lag between runoff & catch rate, & were always better for smaller length groups (13.5 cm) than for the total number or catch weight • Average size of individuals has decreased, either due to high fishing pressure or movement of small prawns out of the river triggered by artificial flows	• Zambezi River runoff is inferred to affect either directly or indirectly, the recruitment strength of *P. indicus* to fishing areas
Staples and Vance 1986. Australia, south-eastern Gulf of Carpentaria • **Banana prawns,** *Penaeus merguiensis* • Research sampling of emigrating prawns • Commercial catch offshore of Karumba • *Analysis*: (i) correlation, (ii) multiple regression	• Strong correlation between rainfall & commercial catch due to increased emigration of juvenile prawns • Emigration rates significantly correlated to rainfall, juvenile numbers & tide range, but when pooled into monthly data, emigration rate was highly correlated with rainfall of same month ($R^2=0.74$)	• Annual juvenile emigration is a function of resident prawn density & rainfall. At low resident densities, emigration is related to juvenile numbers & rainfall. At high densities, rainfall increasingly determines emigration strength & subsequent commercial catch.
Staples and Vance 1987. Australia, Gulf of Carpentaria • **Banana prawns,** *Penaeus merguiensis* • Fortnightly research sampling post larvae, juveniles, emigrating juveniles & adolescents, Sept. 1978 to March 1979 • Tide, rainfall, juvenile abundance • *Analysis*: forward step-wise multiple regression	• (Not strictly a commercial catch correlation) • Multiple immigration of post-larvae into the estuaries, but was variable between estuaries • The number & size of emigrating juvenile & adolescent prawns significantly correlated with (i) juvenile numbers, (ii) tide phase & (iii) rainfall, but differed between estuaries	• Timing of rainfall compared to prawn development was important in influencing the contribution of an estuary to the offshore fishery

Wilber 1992. USA, Florida • **Oysters,** *Crassostrea virginica* • Total annual landings, CPUE (per registered oystermen per yr) • 1960 to 1984 (24 yrs) • Apalachicola River Flow: (i) mean low & high flow events for the 1,7, 30 & 60 consecutive days & 90 & 120 consecutive days for low flows; (ii) annual mean monthly-minimum, monthly-mean, & monthly-maximum flow • *Analysis*: (i) linear regression of annual CPUE & flow, with 2-year lag; (ii) regression analysis using the number of days where flow was above or below a threshold, (iii) predicted CPUE in 1983 & 1984 based on regression equations	• Significant negative correlation CPUE & same year flow (due to 3 particular yrs) • Significant positive correlation CPUE & min. flows with: (i) 2-yr lag [1 (r=0.69), 7 (r=0.65), 30 (r=0.64), 60 (r=0.60), 90 (r=0.62) and 120 (r=0.63) day flows]; and (ii) 3-yr lag of min. flows [1 (r=0.42) and 90 (r=0.61) day flows] • No proof of a cause & effect relationship, but "supports ecological based rationale that freshwater inflow affects oyster production"	• Significant positive correlation between min. flow & oyster productivity lagged by 2 & 3 yrs consistent with growth to marketable size. • Suggests that flow events influence oyster predation, most of which are excluded from estuaries at salinities of 15 • Concludes an inverse relationship between spat survival & estuarine salinities • More frequent low flows may be detrimental to oyster production • Notes that this scenario may be different in different estuaries
Livingston *et al.* 2000. USA, Gulf of Mexico, Apalachicola Bay • **Oysters** *Crassostrea virginica* • Larval density, spatfall, growth rates, biomass in • Measured water quality parameters • Modelled water quality parameters • *Analysis*: (i) stepwise linear regression	• Larvae significantly correlated with oyster density, secchi readings & mean bottom salinity, & inversely correlated to bottom salinity max., but low r-value • Oyster density significantly correlated with average surface current velocity & bottom min. salinity, & inversely correlated with bottom temp., surface colour & max. surface temp. • Average growth of new oysters was significantly correlated with turbidity • Growth of old oysters was inversely correlated with bottom temp., river flow, oyster density, & surface current velocities, but was positively correlated with surface salinity variation • "Bar growth" was positively correlated with surface water colour, secchi readings & average bottom salinities • Overall oyster mortality was positively correlated with max bottom salinity & surface residual current velocity	
Sawynok 1998. Australia, central Queensland • **Barramundi,** *Lates calcarifer* • Recaptures of tagged fish 1984-1997 • Fitzroy River flow, & rainfall • *Analysis*: (i) plots Oct-Feb flow & rainfall with % of juveniles tagged, (ii) average daily linear growth plotted against monthly flow	• (Not strictly a commercial catch correlation) • Higher levels of recruitment (i.e. % of juveniles tagged) occurred with high river flows (›1-million ML) in October to February, except for 87/88 • Average daily growth was 0.61 mm/day/fish for monthly flow ‹2.5 million ML, & was 0.90 mm/day/fish for monthly flow ›2.5 million ML	
Walker *et al.* 1998. Australia, Victoria • **Black bream,** *Acanthopagrus butcheri* • Research sampling Gippsland Lakes District • Otolith-based year-class strength estimates • River flow, SOI, rainfall, temperature • *Analysis*: (i) dynamic regression analysis, (ii) Durbin-Watson statistic for 1st and 2nd order autocorrelations, (iii) ARIMA	• Recruitment strength predicted by: (i) temperature prior to spawning (Oct.), (ii) temp. during spawning (Feb.), & (iii) rainfall after spawning (May) • No significant models were found using river flow or SOI • Strong year-classes corresponded with avg. summer air temperatures ›26•C; weak year-classes corresponded with avg. summer air temperatures ‹24 •C • Recruitment positively correlated with high May rainfall • Best models accounted for 41% of the variation in the relative year-class strength	• H_0: year-class strength related to larval survival, which depends on favourable environmental conditions for optimal food production & larval growth

Quiñones and Montes 2001. Central-south Chile • **Róbalo**, *Eleginops maclovinus* • Annual landings • Itata & Bio-Bio River flow, monthly & annual • Cumulative rainfall, monthly & annual • Itata River 1975-91, Bio-Bio River 1975-94 • Landings, flow & rainfall smoothed using 3-yr running means • *Analysis*: (i) correlation lagged (1-5 yrs), (ii) Pearson correlation coefficients, d.f. adjusted for autocorrelation, (iii) linear regression & (iv) stepwise regression	• Landings correlated with annual Bio-Bio River flow lagged by 4-yrs (r= -0.80, p<0.05) • Landings correlated with annual Itata River flow lagged by 3-yrs (r= -0.94, p<0.01) • Landings not significantly correlated with combined rivers annual flow • Landings correlated with annual rainfall, lagged by 4-yrs (r= -0.75, p=0.05), but correlated with mean March rainfall, lagged by 4-yrs (r=0.78, p<0.05). • Best stepwise linear regression model: Landings = 177.01 – 0.59*annual mean rainfall – annual mean Itata flow lagged by 3-yrs (r2 = 0.93, p<0.05) • No discussion as to why one river has a 4-yr lagged effect, while the other has a 3-yr lagged effect	• Proposed that (i) flow influences the survival rates of Róbalo in the first year (negative relationship) due to (a) washout of eggs & larvae or (b) a trophic effect that increases the predation on 0+ Róbalo year-class, (ii) rainfall in March leads to flow that expands the area of suitable reproductive & nursery habitat • No flow recommendations.
Salen-Picard *et al.* 2002. Mediterranean Sea, Gulf of Lions • **Common sole** *Solea solea* • Rhine River flow 1920-2000 • Seasonal polychaete sampling 1993-2000 • Seasonal dietary sampling of juvenile sole • Commercial landings 1972-1999 • *Analysis*: (i) cumulative mean deviation method, (ii) Spearman rank correlation coefficient, time lags: polychaetes 1 to 48 months, landings 1 to 8 years; d.f. adjusted for serial autocorrelation	• Annual commercial landings at two fishing ports positively correlated to mean annual flow (r=0.84, p<0.01 Port of Martigues; r=0.93, p<0.001 Port of Sete) lagged by 5- yrs	• Speculated a 2-step process: higher survival of larvae & juveniles in the yr of the flood, followed by an increase in the fecundity of the populations for a few years (3-yrs to maturity with fecundity increasing with size) • Both are the consequence of a trophic response to flood events, i.e. increases in short & long-lived polychaetes increased the survival & growth of different life history stages of sole.
Sutcliffe 1973. Canada, Gulf of St Lawrence • **Lobsters** • **Halibut** • St Lawrence River flow • Analysis: (i) correlation, 8- or 9-yr lag	• Quebec lobster catch correlated with April flow (r=0.831, p<0.001) • Prince Edward Is lobster catch correlated with June flow (r=0.853, p<0.001) • Quebec halibut landings correlated with March flow, 3-yr moving average, 10-yr lag	
Sutcliffe *et al.* 1977. USA, Gulf of Maine • Commercial catch 17 commercial species • Sea temperature • Temp. & catch data not independent because catch of 1-yr is made up of individuals of a number of year-classes • *Analysis*: (i) data smoothed using 2 or 3 yr running means (giving the results greater r indicating the importance of low frequencies & long-term trends, but decreases the statistical significance)	• Significant correlations between catch & sea temp for **Alewife** (r =0.850) 6-yr lag, **Butterfish** (r =0.735) 3 to 5-year lag, **Atlantic cod** (r =-0.661) 4-yr lag, **Atlantic herring** (r =0.604) 2-yr lag, **Atlantic menhaden** (r =0.872) 3-yr lag, **Redfish** (r =-0.728) 8-yr lag, **Silver hake** (r =0.781), 5-yr lag, **Striped bass** (r =-0.631) 3-yr lag, **Yellowtail flounder** (r =-0.792) 2-yr lag, **Hard clams** (r =0.805) 2-yr lag, **Soft-shell clams** (r =-0.798) 7-yr lag	
Powell 1977. USA, Texas, San Antonio Bay System • **Fish** harvest (7 species) • **Shellfish** harvest (crabs, oysters, brown & white shrimp) • 1962 to 1976 (15 yrs) • Gauged river flow, (1959-1976) • *Analysis*: (i) correlation, (ii) ranked white shrimp harvest from best to worst & looked for similarities/differences in the seasonal flows	• **Total finfish harvest** correlated with average spring flow 3-yr lag (p<0.01, r=0.64) • **Red drum harvest** correlated with average spring flow 3-yr lag (p<0.01, r=0.72) • **Shellfish** harvest correlated with spring flow (p<0.05, r=0.43.3) • **White shrimp**: (i) best years had >average inflows, (ii) worst years had <average inflows • Thresholds in spring flows were identified to be related to finfish & red drum harvests that were below the 15 yr average	• Developed an approx. gauged freshwater inflow regime (monthly % distribution) to meet minimum sustaining fishery requirements • No details are provided as to how the minimum sustaining fishery requirements were arrived at
Meeter *et al.* 1979. USA, Florida, Apalachicola estuary • Commercial landings, 1957-1977 • Monthly catch blue crabs & shrimp, 1972-1977 • Flow (log), rainfall (square-root) • *Analysis*: (i) spectral & cross-spectral analyses, (ii) correlation	• Annual **shrimp** catch positively correlated with flow (p>0.05) • Annual **blue crab** catch positively correlated with flow (p<0.05) • Annual **oyster harvest** negatively correlated with flow (p<0.05) • Monthly **shrimp** numbers negatively correlated with flow (p<0.05) • Monthly **crab** numbers positively correlated with flow (p<0.05)	

Funicelli 1984. USA, Texas estuaries • *Analysis*: (i) stepwise GLM	• Analysis yielded 19 statistically significant regression equations relating to eight species groups **Shellfish, Penaeid shrimp, White shrimp, Blue crab, Oyster, Finfish, Spotted seatrout, red drum**	• Study indicates that reduced flows will directly reduce river borne nutrients & freshwater inundation of deltaic marshes, negatively impacting on habitats & commercial fisheries production
Drinkwater and Myers 1987. Canada, Quebec • Annual catch landings • Monthly flow of St Lawrence, Ottawa & Saguenay Rivers • *Analysis*: (i) linear regression, (ii) regression equations for predicting catches of subsequent years, (iii) predicted catch correlated with observed landings	• Ho: Variability of invertebrate & fish abundance due to environmental factors • Correlations of $r>0.5$ for: **Lobster, Soft shell clams, Butter-fish, Redfish** and, **Flounder** • None significant due to high autocorrelation in the data	
Kimmerer and Schubel 1994 San Francisco Bay / Sacramento / San Joaquin Delta estuary (Workshop) • Goal: consensus among the scientific community regarding the flow needs of the estuary • Selected the location of a salinity of 2-psu at the bottom as an index on which a standard should be based	• Found significant relationships between the historical value of χ_2 & all trophic levels i.e. total input of organic carbon, abundance of mysid *Neomysis mercedis*, bay shrimp *Crangon franciscorum*, longfin smelt, starry flounder, survival of striped bass from egg to young-of-the-year, striped bass year-class strength, survival of salmon smolts, & biomass of molluscs • Biological responses to the optimum χ_2 were sigmoidal, with a region of rapid increase in response to decreasing χ_2, followed by a levelling off at high value of χ_2 • Most biological responses increased monotonically with decreasing χ_2; none showed a clear maximum at intermediate values of χ_2	
Powell and Malstaff 1994. USA, Texas, Guadalupe Estuary • **Red drum** (*Sciaenops ocellatus* n=19) • **Black drum** (*Pogonias cromis* n=24) • **Spotted seatrout** (*Cynoscion nebulous* n=19) • **Blue crabs** (*Callinectes sapidus,* 24 years) • **Eastern oyster** (*Crassostrea virginica* n=24) • **Brown shrimp** (*F. aztecus*, n=27) • **White shrimp** (*L. setiferus*, n=27) • Annual landings (+ effort for shrimp) • G(auged) & EOS flow (bimonthly) • Average minimum air temperature • *Analysis*: (i) all possible subsets regression to select the 10 best predictors (raw & ln transformed), (ii) serial correlation using runs test, autocorrelation using Durbin-Watson test; (iii) observed historic catches plotted against the predicted harvests	• Derived harvest-inflow equations for each species-group. • Harvest = linear function of ln transformed combined inflows for 2 month period (lagged to suit species life history). Equations for each species are provided in text. • **Red drum** ($R^2=0.69$, $p=0.0001$, Gflow & temp.), ($R^2=0.52$, $p=0.0011$, EOS flow & temp.) • **Black drum** ($R^2=0.66$, $p=0.0001$, Gflow & temp.), ($R^2=0.44$, $p=0.0024$, EOS flow & temp.) • **Blue crabs** ($R^2=0.78$, $p<0.0001$, Gflow & temp.), ($R^2=0.62$, $p=0.0001$, EOS flow & temp.) • **Seatrout** ($R^2=0.61$, $p=0.0002$, Gflow & temp.), ($R^2=0.60$, $p=0.0003$, EOS flow & temp.) • **Oysters** ($R^2=0.88$, $p<0.0001$, Gflow & temp.), ($R^2=0.83$, $p<0.0001$, EOS flow & temp.) • **Brown shrimp** ($R^2=0.89$, $p<0.0001$, Gflow, effort & temp.), ($R^2=0.88$, $p<0.0001$, EOS flow, effort & temp.) • **White shrimp** (adj. $r^2 = 0.77$, $p<0.0001$, G flow & temp.), ($R^2 = 0.73$, $p<0.0001$, EOS flow & temp.)	• Analysis of data for the other 4 estuaries gave similar results although the degree of fit varied considerable between estuaries for any one species
Skreslet 1997. Norway • **NE artic cod** *Gadus morhua* • Melt water discharge	• Year-class strength positively correlated with melt water discharge 1-yr in advance • Commercial landings of juvenile cod positively correlated with melt water discharge 3-yrs in advance	

Ardisson and Bourget 1997. Canada, Quebec • Gulf of St Lawrence total discharge • Maximum abundance, biomass, & mean weight of juveniles of 5 epibenthic species: (i) **Hydroid** *Obelia longissima* (ii) **Bivalves** *Hiatella artica, Mytilus edulis* **(iii) Cirripedes** *Semibalanus balanoides, Balanus crenatus* • Cumulative spring runoff, mean annual runoff • 1975 to 1985 (10 years) • *Analysis*: (i) simple linear regression, spearman correlation test; (ii) polynomial regression of 2nd and 3rd orders; (iii) GLM; (iv) GAM; (v) Bonferroni corrections & non-parametric correlations.	• No analyses adequately modelled the data • No significant relationships observed	
Loneragan and Bunn 1999. Australia, southeast Queensland • Calendar year commercial landings • (i) Queensland Fish Board landings, 3 ports, 1945 to 1975 (30 years); (ii) Commercial logbook records 1988 to 1995 (8 years) • Logan River gauged flow: (i) total monthly; (ii) total seasonal (iii) total calendar year • *Analysis*: (i) correlation: total catch with seasonal flows from preceding September to December; (ii) Pearson correlation coefficients; (iii) regressions equations fitted for significant correlations and used to predict catches as a function of minimum & maximum seasonal flows	• Summer flows significantly correlated with catches of **mud crabs** ($R^2 = 0.80$, $p<0.01$), **total crabs** ($p<0.01$), **king prawns** ($r^2=0.75$, $p<0.01$), **total prawn** ($r^2=0.74$, $p<0.01$), **bay prawns** ($p<0.05$), **tiger prawns** ($p<0.01$), **flathead** ($r^2=0.69$, $p<0.05$) • Total annual flow significantly correlated with catches of **school prawns** ($p<0.01$), **greasy prawns** ($p<0.05$), **tiger prawns** ($p<0.05$), **mullet** ($p<0.05$), **flathead** ($p<0.05$), **total fish** ($p<0.05$) • Winter flow significantly correlated with **mullet** ($p<0.05$), **school prawns** ($p<0.05$) • Long-term data (flow & total fish catch), after accounting for year, flow explained 25% of the variation in fish catch	
Ritter *et al.* 2000. USA, Texas • **White shrimp** (*P. setiferus*, n=35) • **Brown shrimp** (*P. aztecus*, *n=35*) • **Blue crabs** (*Callinectes sapidus,* n=33) • **Spotted seatrout** (*Cynoscion nebulous, n=*20) • **Red drum** (*Sciaenops ocellatus,* n=20) • **Black drum** (*Pogonias cromis,* n=33) • Commercial landings 1963 to 1998, species dependent • Salinity data (automated & non-automated) • Mission-Aransas Estuary EOS flow • Effort for brown shrimp • *Analysis*: (i) regression analysis, (Based on methods of TxEMP, Powell & Malstaff 1994)	• Derived a harvest-inflow equations for each species group i.e., harvest = some linear function of ln transformed combined inflows for a two month period. Equations are provided for each species, containing between 3 and 5 inflow parameters i.e., two months of inflow. • Significance levels were **white shrimp** (adj. $R^2 = 0.50$, $p=0.0001$), **brown shrimp** (adj. $R^2 = 0.75$, $p<0.0001$), **blue crabs** (adj. $R^2 = 0.58$, $p<0.0001$), **spotted seatrout** (adj. $R^2 = 0.63$, $p=0.0006$), **red drum** (adj. $R^2 = 0.59$, $p=0.0013$), **black drum** (adj. $R^2 = 0.41$, $p=0.0017$)	
Lloret *et al.* 2001. North-west Mediterranean Sea • 13 Commercial species that were dominated by fisheries for 1-& 2-yr old individuals. • Monthly catch, • Rhône & Muga Rivers flow & wind-mixing index • 1976 to 1998 (up to 32 years) • *Analysis*: (i) decomposed environmental time series into trend & seasonality, (ii) linear relationships analysed with annual data, (iii) transfer function time series analysis.	• H_o: recruitment is influenced by river discharge & wind condition during the spawning & recruitment season through enhanced fertilisation or larval retention • Catch & CPUE significantly positively correlated (at different time lags) with river flow & wind-mixing-index during the reproductive season • Significant time lags were identified for 2 to 6 months, 6 to 10 months, 16 to 18 months, 18 to 26 months across a number of species & sizes caught	

Powell *et al.* 2002. USA, Texas, Galveston Bay • **Red drum** (*Sciaenops ocellatus* n=19) • **Black drum** (*Pogonias cromis* n=24) • **Spotted seatrout** (*Cynoscion nebulous* n=19) • **Blue crabs** (*Callinectes sapidus,* 24 years) • **Eastern oyster** (*Crassostrea virginica* n=24) • **Brown shrimp** (*F. aztecus*, n=27) • **White shrimp** (*L. setiferus*, n=27) • **Flounder** *(Paralichthys lethostigma*) • Annual landings (effort for shrimp) • G(auged) & EOS flow (bimonthly) • Average minimum air temperature • *Analysis*: (i) all possible subset regression analysis to select the 10 best predictors (raw & ln transformed); (ii) serial correlation using runs test, autocorrelation using Durbin-Watson test; (iii) observed historic catches plotted against the predicted harvests, (Based on methods of TxEMP).	• Derived a harvest-inflow equations for each species group i.e., harvest = some linear function of ln transformed combined inflows for a two month period. Equations are provided for each species, containing between 3 and 5 inflow parameters i.e., two months of inflow. • Significance levels were **red drum** (adj. R^2 = 0.59, p=), **black drum** (adj. R^2 = 0.41, p=), **spotted seatrout** (adj. R^2 = 0.52, p=), **blue crabs** (adj. R^2 = 0.76, p), **eastern oyster** (adj. R^2 = 0.46, p=), **white shrimp** (adj. R^2 = 0.57, p=), **brown shrimp** (adj. R^2 = 0.49, p<), **flounder** (adj. R^2 = 0.54, p=)	• Removed up to 10% of the outliers in order to maximise the correlation, but number not used is specified for each species.
Sobrino *et al.* 2002. Spain, Gulf of Cadiz • **Common octopus** (*Octopus vulgaris*, n=18) • **Cuttlefish** (*Sepia officinalis,* n=18) • Landings (kg by month), effort • Monthly Sea Surface Temperature; Monthly river flow, mean rainfall rates – aggregated into 'pluvial year' = September to June • *Analysis:* (i) parametric correlation analysis, with Pearson correlation coefficient; (ii) stepwise multiple regression	• Significant negative correlations existed for octopus but not for cuttlefish • Landings of octopus (which comprise 28% of the cephalopod catch) were significantly correlated with the previous rain season coefficient (Rain $^{-1}$, r=-0.69, $p<0.01$, river discharge in January (River^1, r=-0.83, $p<0.01$), February (River^2, r=-0.67,$p<0.01$), & December (River^{12}, r=-0.65, $p<0.01$), & SST coefficients for months February r=-0.56, $p<0.05$, April r=-0.60, $p<0.05$, May r=-0.64, $p<0.01$, June r=-0.68, $p<0.01$. • Identified that Rain $^{-1}$ (~River^1 River^2), River^{12} & SST^5 & SST^6 should be used as the variables affecting abundance of octopus • Generated a significant multiple regression with an R^2 of 0.76	• Octopus abundance of depends annual recruitment, probably defined at early planktonic stages • Rainfall hypothesized to have a negative effect on egg-laying or paralarvae survival • SST affects abundance through (i) an indirect effect on cephalopod paralarvae via food availability or predation or (ii) as a direct effect on paralarvae mortality • River flow may change environmental conditions forcing octopus to leave relatively unsheltered dens • Cuttlefish a euryhaline species able to tolerate fluctuations in salinity

Chapter 2. Development of a conceptual framework

J. Robins, I. Halliday, J. Staunton-Smith, D. Mayer, and M. Sellin [2]

Introduction

Advice on the freshwater flow requirements of fisheries is currently sought by managers and the legal and political ramifications of the water allocation process prohibit delaying these allocations until greater understanding of ecosystem functioning is achieved. Therefore, a more structured and transparent method is needed to identify the freshwater flow requirements for fisheries using the available information.

Numerous correlative studies conclude that the catch of some estuarine and marine finfish and shellfish is strongly linked to freshwater flow, being equally applicable in tropical, sub-tropical and temperate areas (see Chapter 1; Drinkwater and Frank 1994; Robins *et al.* 2005). However, the causal mechanisms underpinning these correlative relationships need to be better understood in order to predict or minimise the impacts of altering freshwater flows (Browder *et al.* 2002; Kimmerer 2002a,b). Conceptual models of the role of freshwater flow in estuarine ecosystems, and hypotheses developed from them, need to be explicitly considered in order to direct multi-disciplinary studies, and to provide the conceptual structure for integrated bio-physical models.

We developed a conceptual framework to identify aspects of the freshwater flow regime that are potentially important for estuarine shellfish and finfish species (Figure 2.1). Conceptual models (with hypotheses) of key biological processes of estuarine fisheries resources that are influenced or dependent on freshwater flow were developed and refined using a combination of life history information (Chapter 2) and analysis of catch and freshwater flow data (Chapter 3). These results can be used to identify key flow events that are ecologically important (sensu Gippel 2002) and that contribute to estuarine ecosystem health. In addition, our approach identifies priority areas of research in order to 'test' or gather more evidence in support of any particular hypothesis.

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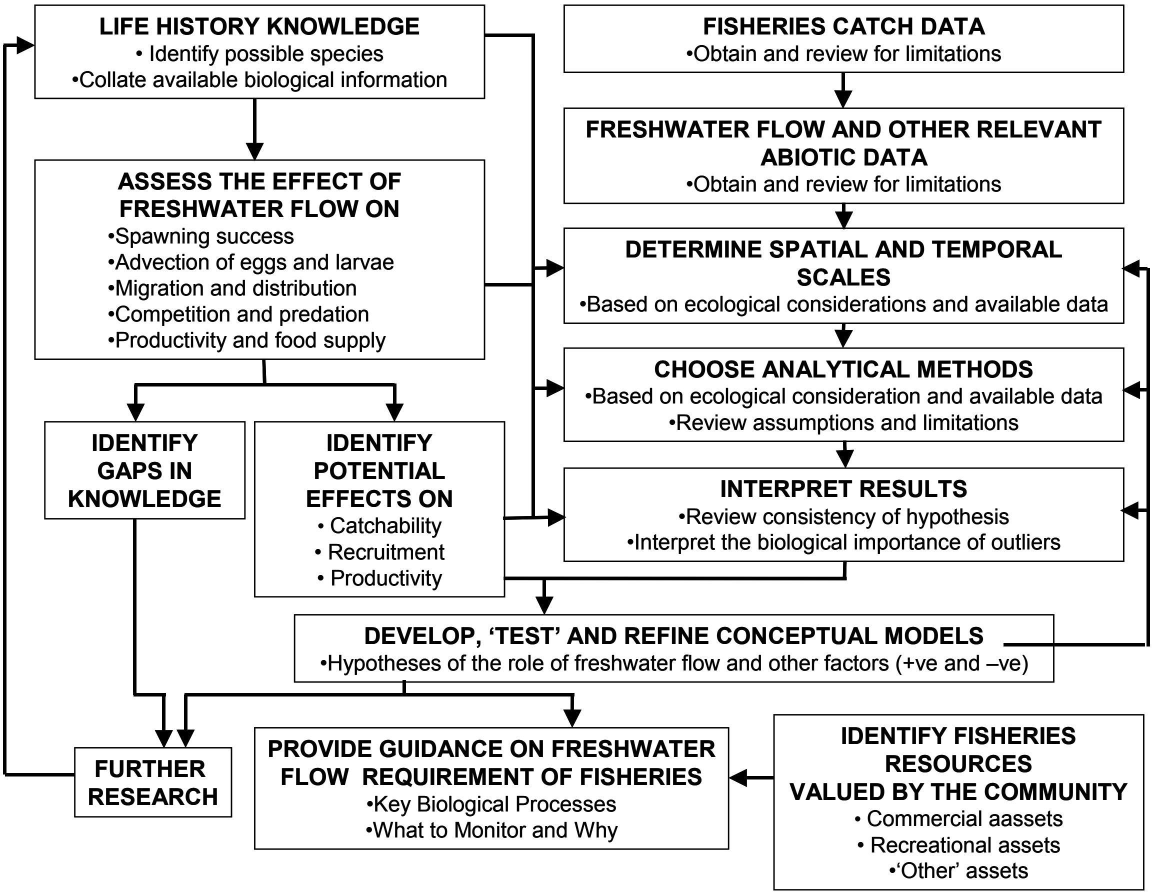
Figure 2.1 Generalised framework (logical approach) to identifying aspects of the freshwater flow regime which potentially influence estuarine fisheries production



Method and approach

Assessment of life history

Assessing life history information aims to identify the biological processes for estuarine or near-coastal fishery species that are affected by freshwater flows and also highlights gaps in the current knowledge. Our assessment criteria were based on categories of freshwater flow effects identified by Drinkwater and Frank (1994); i.e. spawning success, advection of eggs and larvae, migration, competition and distribution, general productivity and food supply, and water quality (Table 2.1). However, migration and distribution are considered together because of the similarity in the effects of freshwater flow on these aspects.

Table 2.1 Criteria for assessing the role of freshwater flows in the life history of estuarine species (from Drinkwater and Frank 1994)

1. Migration
Is variation in freshwater flow required to induce the (spawning) migration of adults?
Is variation in freshwater flow required for the migration of juveniles?
Is migration triggered or enabled by freshwater flow events? (i.e. change in flow velocity vs. connections caused by flooding)
What are the triggers for migration? (e.g. flow velocity, day-length, water temperature, lunar phase, chemical cues)
What are the directions of these migrations in relation to freshwater flows?
2. Spawning Success (& Survival of Eggs and Larvae)
Is the timing of spawning synchronised to peak freshwater flow?
What are the triggers for spawning? (e.g. flow velocity, day-length, water temperature, lunar phase, chemical cues)
Is there a single spawning event or are there multiple spawning events of the population?
Is the 'quality' of spawning habitat (=egg survival) reliant on freshwater flow?
What qualities of the spawning habitat are important? (e.g. structure, availability, flow velocity, dissolved oxygen, salinity)
3. Advection of Eggs and Larvae
Do eggs or larvae utilise currents resulting from freshwater flows?
Do eggs or larvae use flow velocities or water borne chemical cues to direct movement?
4. Species Competition and Distribution
Are nursery habitats modified by freshwater flows?
What is the relationship between the nursery habitats and freshwater flow? (e.g. positive, negative, inverse)
Is the distribution of the species effected by freshwater flow?
What life stages are effected in their distribution?
What are the causes of the changed distribution? (e.g. habitat access/connection, food availability, salinity gradients, turbidity)
Do flow-related changes in the distribution of the species result in the changed catchability of a fished species?
Is predation enhanced or reduced because of habitat access, turbidity or salinity levels associated with freshwater flows?
5. General Productivity and Food Supply
Are flow-related changes in primary productivity directly or indirectly translated to changes in the productivity of the species of interest? (e.g. what is the trophic level of the species? – primary consumer, secondary consumer).
How is the species-of-interest linked to the 'general productivity' of the estuary? (e.g. diet, dependence on bacterial or phytoplankton food chain).
6. Water Quality
Is the water quality reduced as a result of freshwater flow?
Is freshwater flow required to maintain water quality?
If so, what aspects of water quality are important? (e.g. flushing of pollutants)

Assessing the role of freshwater flows on spawning success includes impacts on the survival of eggs and larvae. Issues considered may include the timing of spawning in relation to freshwater flow events, triggers for the act of spawning, whether the quality of spawning habitat (including water quality parameters) is reliant or affected by freshwater flows, and if so what factors are effected. The role of freshwater flow on the advection of eggs and larvae should identify whether eggs and or larvae are influenced by currents or chemical cues resulting from freshwater flow events. Issues regarding the effects of freshwater flow on migration and distribution include how flow affects the distribution of each life history stage, causes of the change in distribution (i.e. as a consequence of habitat access, food availability, salinity gradients or turbidity), effects on nursery habitats (e.g. connections, longevity, and water quality), and triggers for the migration of adults (e.g. spawning migration) or juveniles including the relevance of other factors in stimulating migration (e.g. day-length, water temperature and lunar phase). Effects on the catchability of a species resulting from migration or changes in distribution patterns should be identified. Assessing the role of freshwater flows on competition includes effects on predation that may result from increased turbidity or changed distribution of predators. The trophic level of the species can be assessed to determine how changes in primary productivity occurring as a consequence of freshwater flows could translate to changes in the productivity of the species, including changes in growth rates.

Proposed mechanisms of the role of freshwater flow can be summarised into effects on: (i) movement and catchability; (ii) recruitment (i.e. survival during early stages of life that translates to the 'strength' or size of a cohort); and (iii) productivity, such as increased growth rates. Examination of the proposed mechanisms allow the *a priori* identification of freshwater flow variables to be used in the analysis of fisheries catch data, as recommended by Tyler (1992).

Analysis of catch data

The analysis of catch and freshwater flow data can be made more robust, and therefore more useful to determining the freshwater needs of estuarine fisheries, by critically selecting data for analysis, choosing analytical methods appropriate to the data limitations, and critically interpreting the results (Figure 2.1). Fisheries catch data is influenced by numerous factors (e.g. stock-recruitment relationships, levels of fishing effort, habitat changes, pollution impacts, and other density-dependent processes). The impacts of such factors on the catch should be considered, and where possible, species where these impacts are minimal, constant or measurable, be selected for analysis. The degree to which a species migrates should also be considered, as correlative analyses between freshwater flow and catch may not be appropriate for species that undertake extensive spawning migrations (e.g. mullet *Mugil* spp.) or which have broad-scale dispersal mechanisms of eggs or larvae (e.g. eastern king prawns *Penaeus plebejus*).

Freshwater flowing to estuaries is often based on gauged data from a station located upstream of the estuary. Freshwater flow data may need to be revised to include water extractions or additional freshwater flows from ungauged streams or tributaries occurring downstream of the gauging station. The spatial (and temporal) scales at which fisheries and freshwater flow data are aggregated should appropriately represent the area of influence of freshwater flow on the fished species.

Consideration should be given to the appropriate analytical method and whether auto-correlation is a feature of the data (Pyper and Peterman 1998). Results from the analysis should be reviewed for consistency with the theoretical mechanisms proposed before the analysis. We suggest that further consideration be given to interpreting anomalous observations (i.e. ‘outliers’ where the relationship between catch and freshwater flow does not conform to the analytical model). Such observations may provide insights into other factors (positively or negatively) influencing fisheries catch.

Results from the analysis of catch and freshwater flow data can be used to identify which of the proposed mechanisms are supported by the available evidence and therefore, (from the available evidence of life history assessment and analysis of data) identify the biological processes of estuarine fishery species that are likely to require freshwater flows.

Life history assessments

Detailed accounts of the life history of some species (e.g. banana prawns and barramundi) in Australian estuaries can be found in the published literature (e.g. Dunstan 1959; Davis 1985; Russell and Garrett 1985; Griffin 1987). However, we have included detailed reviews of the main estuarine fishery species of northern Australia to demonstrate the extent to which we consider information, both published and ‘here-say’, should be reviewed in order to determine the how, when and where freshwater flow may be influencing the life-cycle of a species.

Banana prawns (*Penaeus merguiensis*)

Banana prawns are an estuarine and coastal species associated as adults with waters up to 20 km from the coast (up to 45 m water depth, Grey *et al.* 1983). Banana prawns have a typical type-2 penaeid life-cycle (Dall *et al.* 1990). Adolescent banana prawns migrate downstream from estuarine habitats to marine waters. Here they mature and spawn benthic eggs, which become pelagic larvae. Nauplia, protozoeal and mysis stages of banana prawns have the highest survival in salinity of 30 to 35 (Nisa and Ahmed 2000 on the Practical Salinity Scale of 1978). Larvae and post-larvae migrate from offshore waters into estuaries using tidal currents, and settle as post-larvae into estuarine nursery habitats. Juvenile banana prawns remain in the estuary for several months, as benthic residents, before migrating out of the estuary to coastal marine waters.

Banana prawns spawn in spring[3], resulting in post-larval and juvenile prawns migrating into nursery habitats between November and May. Banana prawns are though to have a one-year life-cycle, with spring-spawned individuals contributing to individuals taken by the commercial trawl fleet between February and May. A complicating factor, reported by Dredge (1985), is the recruitment of juvenile banana prawns (>10 mm carapace length) into the Burnett River, (central Queensland) in December to March as well as May to June. Large banana prawns have been recorded in the estuaries of central and southern Queensland during winter and there is some speculation that this is indicative of banana prawns in this area having a six-monthly life-cycle with two generations per year (Dredge 1985).

Mangrove-lined creeks are the preferred habitats of juvenile banana prawns (Staples *et al.* 1985). The downstream migration of juvenile and sub-adult banana prawns to outside the estuary occurs from summer to autumn and coincides with seasonal rainfall in tropical and sub-tropical areas. The migration of juvenile banana prawns is associated with increased rainfall in the Gulf of Carpentaria (Staples and Vance 1986) and other parts of Australia (i.e. south east Queensland, Meager *et al.* 2003).

Whilst in the estuary, juvenile banana prawns are carnivorous detritivores, consuming a wide range of organisms and organic detritus (Chong and Sasekumar 1981). Gut-content studies report unidentified debris as well as live benthic and pelagic animals such as polychaetes, copepods, amphipods, isopods, mysids, carids, sergestids, foraminifera, molluscs, gastropods, nematodes, insects, diatoms, algae, bacteria, and epiphytes (Wassenberg and Hill 1993). Banana prawns feed while inside mangrove forests as well as in the shallows of creeks and rivers when the water levels are below the mangroves (Logan River – Sue Pillans , personal communication). Newly arrived pelagic post-larvae are carnivorous, feeding mostly on calanoid copepods, while epibenthic post-larvae and juveniles are carnivorous detritivores feeding on detritus, foraminiferans (Rhotallidae), copepods (calanoid and harpacticoid), larval bivalves, diatoms and brachyuran larvae (Chong and Sasekumar 1981). Sub-adults are detritivorous carnivores feeding on large crustaceans such as *Acetes* and mysids, with lesser amounts of detritus. Adults are detritivorous carnivores feeding on detritus and animals (e.g. large crustaceans such as *Acetes* sp., molluscs and fishes) in equal amounts. Plant material consumed by juveniles (in small but consistent amounts) included pieces of mangrove, filamentous algae (*Trichodesmium* and *Microcoleus* spp.), and diatoms (*Coscinodiscus* sp., *Cyclotella* sp., *Pleurosigma* sp. and *Gyrosigma* spp.).

Isotope studies are suggested to give a better indication of the relative importance of dietary items because results indicate a time-integrated, objective measure of carbon assimilated by the organism (Primavera 1996). Several authors have investigated the isotopic signature of banana prawns to identify the relative importance of the various organisms in their nutrition (Newell *et al.* 1995; Primavera 1996; Loneragan *et al.* 1997). Newell *et al.* (1995) reported that juvenile banana prawns living in tidal creeks derived nutrition from mangrove sources as well as benthic micro algae, although the greater relative abundance of mangrove detritus in tidal creeks resulted in its greater consumption by juvenile banana prawns. Primavera (1996) reported that δ^{13}C of banana prawns (-18) was closer to plankton and epiphytic algae (-22.6 and –24.3 respectively) than to mangroves (-28.6). He reported a similar finding for δ^{15}N, with banana prawns (<9 mm to 30 mm carapace length) having a signal (6.9) closer to epiphytic algae (6.0) than to decomposing mangrove leaves (3.8) or plankton (2.3). Primavera (1996) noted that the high δ^{15}N for epiphytic algae may be due to contamination by nematodes and meiofauna present in the samples. Primavera (1996) suggested that the enriched δ^{15}N signal of banana prawns suggests that prawns are two to three levels up the trophic chain from phytoplankton (assuming a 2.4‰ enrichment per trophic level). Primavera (1996) suggested the use of stable S isotopes to improve the understanding of plankton-penaeid shrimp connections. Loneragan *et al.* (1997) found similar

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[3] Banana prawns spawn throughout the year with a peak in spawning activity in spring and autumn. However, it is speculated that it is the spring-spawned individuals that contribute most to the commercial autumn fishery for banana prawns.

results to those of Primavera (1996). Banana prawns had δ^{13}C and δ^{15}N values closer to that of macroalgae/seston. Values of δ^{34}S were between the values of seagrass (*E. acoroides*) and mangrove (*C. tagal*). Loneragan *et al.* (1997) concluded that juvenile banana prawns were likely to obtain <10% of their nutrition from mangrove detritus.

Haywood and Staples (1993) used length-frequency analysis and modal progression to derive growth rates for banana prawns during the estuarine phase of their life-cycle. They found that growth rates ranged from 0.63 to 1.65 mm carapace length per week, and that a linear model could describe the relationship between growth, water temperature (a positive effect) and prawn density (a negative effect). Previously, Staples (1980) used polymodal frequency analysis (assuming negligible effects of size-selective mortality within a cohort) to derive the mean carapace length of different cohorts at weekly intervals and then estimates growth rates. Staples (1980) noted sexual dimorphism in size occurred at >10 mm carapace length, although the slight difference in growth rates between females and males was not of sufficient magnitude to include in growth equations.

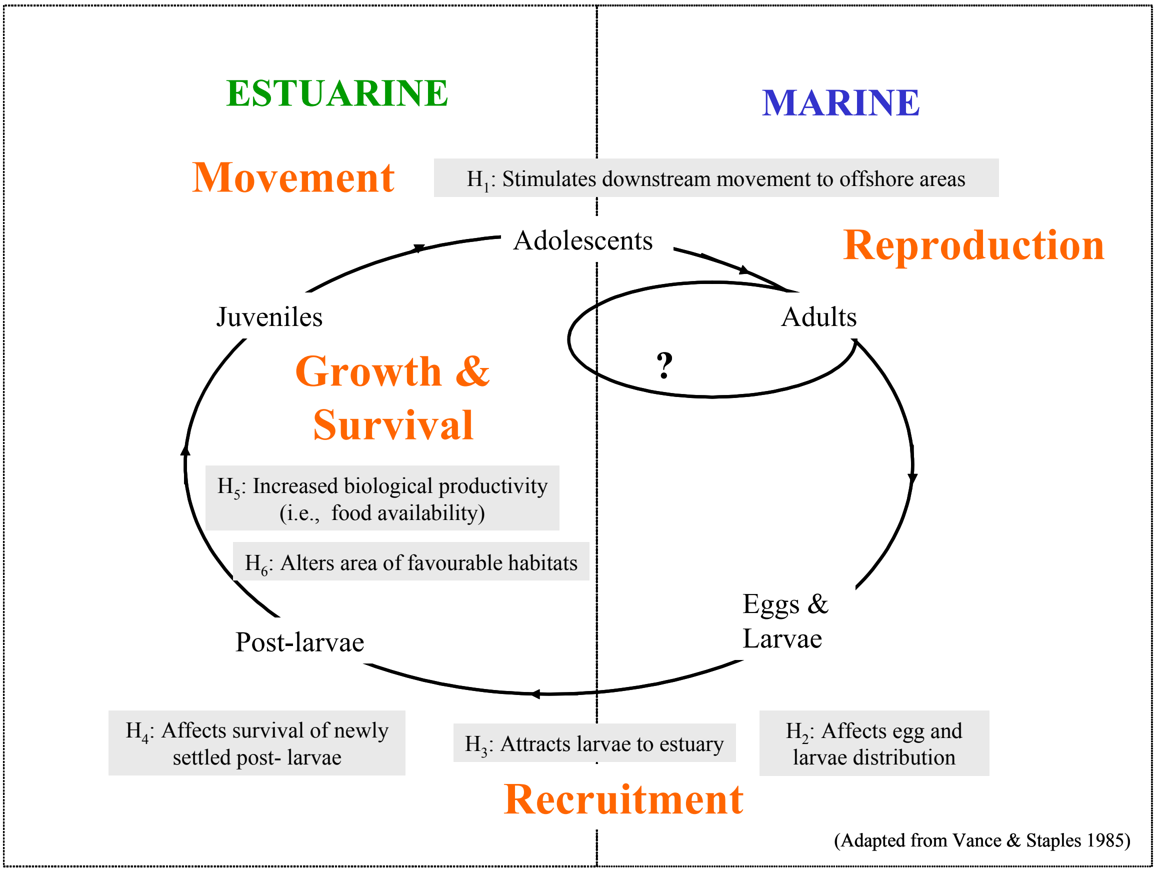
Haywood and Staples (1993) reported that salinity had no detectable effect on growth rates. However, experimentally, juvenile banana prawns were found to have optimal food consumption and production at a salinity of 20 (Vinod *et al.* 1996). At higher salinities, there was a considerable decrease in growth and food consumption (Vinod *et al.* 1996). In contrast, Saldanha and Achuthankutty (2000) report that the growth of juvenile banana prawns increased with salinity (up to 40‰), while Staples and Heales (1991) report that the optimum temperature and salinity for the growth in length of juvenile banana prawns (i.e. shortest inter-molt period and largest carapace length increment) was 31°C and a salinity of 30‰ (resulting in a weekly growth rate of ~1 mm per week). However, accounting for survival, increases in wet and dry weight, the optimum temperature and salinity for the greatest increase in biomass and production were 28°C and a salinity of 25‰. Staples and Heales (1991) concluded that deviations from the optimum of temperature have a greater effect on productivity than of salinity. Based on their experimental work on variation in the growth and survival with temperature and salinity, Staples and Heales (1991) predicted that in an estuary, post-larval prawns would grow quickly but suffer high mortality when temperature and salinity were high, but would grow slowly and remain in nursery areas if the salinity of the estuary fell below 20‰.

Offshore commercial catches of adult banana prawns are often higher in years when there is high coastal rainfall (Vance *et al.* 1985). This is probably related to the physical movement of juveniles out of the estuaries (Staples and Vance 1987), but also may be helped by the increase in nutrients to the coastal areas associated with freshwater runoff. Timing of the freshwater flow event in relation to the developmental stage of banana prawns will determine the contribution of an estuary to the offshore banana prawn population (Staples and Vance 1987). Evans and Opnai (1995) explored the correlation between rainfall and banana prawn CPUE and yield in Gulf of Papua trawl fishery and suggested that the recruitment of juvenile banana prawns into the fishery stock may be greatest in years with an intermediate level of rainfall in the Southern Highlands and the southern hills adjacent to the Gulf of Papua.

Freshwater flows may also have a negative effect on banana prawns. The immigration of post-larval banana prawns into estuaries of the southern Gulf of Carpentaria is hindered when estuarine waters are lowered in salinity to freshwater conditions (Staples and Vance 1987). Evans and Opnai (1995) postulated that in rivers adjacent to the Gulf of Papua, rainfall may have a negative effect on the commercial catches of banana prawns through reduced survival of newly settled post-larvae by high levels of rainfall and river flow. Evans *et al.* 1997 suggest that the relationship between rainfall and 'yield' of banana prawns is parabolic: positive where the effect of flow on the emigration of juveniles into offshore waters dominates the subsequence abundance of banana prawns (i.e. effect on post-larvae is relatively small) and is negative where the effect of flow on the immigration and survival of post-larvae into riverine waters dominates the subsequent abundance of banana prawns (i.e. effect on sub-adults is relatively small).

Populations of banana prawns are likely to be highly responsive to freshwater flows. Life history information was conceptualised into a life-cycle figure, with the potential influence of freshwater flow included (**Figure** 2.2).

Figure 2.2 Conceptual model of the influences of freshwater flows on the life-cycle of banana prawns



(Adapted from Vance & Staples 1985)

Specifically, we identified six hypotheses of the influence of freshwater flow on banana prawn populations. They were that freshwater flow:

1) Stimulates downstream movement of juvenile and sub-adult banana prawns to offshore areas, potentially increasing their catchability by the otter-trawl fishery. The stimulus is possibly salinity changes, although flushing may occur at high flow rates.
2) Affects the transport of larvae through currents. Large freshwater flows may prevent larval immigration to estuarine habitats either by washing eggs and larvae away from the estuary or dilution of chemical cues.
3) Creates chemical signals for larvae to enter the estuary.
4) Affects the survival of newly settled post-larvae in estuarine habitats, with freshwater (i.e. salinity of 0) being unsuitable habitat for post-larvae.
5) Enhances biological productivity of the estuary, thereby increasing the availability of food in the estuary for post-larvae, juveniles and adolescents which results in improved growth, survival and 'condition' of the banana prawn population, potentially leading to larger biomass and better reproduction.
6) Affects the area of favourable habitat for juveniles and adolescents, potentially through larger areas of decreased salinity, the creation of a salinity gradient (i.e. 5 to 30‰) or turbid conditions reducing predation, which increases the survival of juveniles and adolescents.

These potential causal mechanisms can be grouped into effects on: (i) recruitment, by a) washing away eggs and larvae thereby reducing larval immigration to the estuary (negative effect), or b) stimulating the larvae to enter the estuary in response to chemical cues; (ii) catchability, by stimulating the downstream movement of juvenile and sub-adult banana prawns; and (iii) productivity, through increased food availability resulting from enhanced biological productivity of the estuary, resulting in improved growth and survival of post-larvae, juveniles and adolescents.

Barramundi (*Lates calcarifer*)

Barramundi are the main finfish species caught in estuarine and coastal waters of the Queensland east coast and Gulf of Carpentaria. They are harvested using mesh nets (i.e. gill nets). No studies have published correlations between the catch of barramundi and freshwater flow. However, numerous aspects of barramundi life history suggest that populations are likely to be highly responsive to freshwater flows. Dunstan (1959) and Williams (2002) proposed that the catch of barramundi is closely associated with freshwater flows and or rainfall, through influencing adult spawning success, juvenile recruitment and catchability. Barramundi can move large distances between estuaries, but most individuals remain within a specified region (Davis 1986; Salini and Shaklee 1988; Keenan 1994). Tag-recapture information indicated that this is also true of barramundi in the Fitzroy River region (Australian National Sportfishing Association Queensland Inc., Infofish Services, Rockhampton, unpublished data), suggesting that changes in local freshwater flow conditions may be reflected in the local barramundi population. In Australia, commercial barramundi fisheries are restricted by fisheries regulations to estuarine and marine waters, despite barramundi using freshwater habitats as juveniles (i.e. barramundi is a diadromous species).

Detailed accounts of barramundi life history in Australian estuaries can be found in Dunstan (1959), Davis (1985), Russell and Garrett (1985), and Griffin (1987). Barramundi have a complex life history, are catadromous (i.e. migrating from freshwater to saltwater to spawn) and protandrous (changing sex from male to female). Pender and Griffin (1996) demonstrated that catadromy in barramundi is not obligatory, finding that a high proportion of barramundi in estuaries of the Northern Territory do not enter freshwater, based on the analysis of barium and strontium concentrations of barramundi scales (Griffin and Walters 1999). However, where access and habitats permits, barramundi will move into freshwater habitats as juveniles, returning to estuaries and coastal foreshores when mature. Barramundi along the Queensland east coast generally are not catadromous because of the ephemeral nature of most Queensland east coast rivers and the construction of barriers associated with water infrastructure that prevent the movement of fish upstream (Russell 1990).

Mature female barramundi are thought to reside in the lower reaches of estuaries and along the coastal foreshore (i.e. in saltwater habitats, Dunstan 1959). The life-cycle of barramundi generally results in the spatial separation of male and female fish, with smaller and younger male fish residing in the upper estuary or in freshwater reaches of the river (e.g. Gulf of Carpentaria and Papua New Guinea). Mature males must move downstream to the estuary in order to participate in spawning. Mature barramundi are though to be stimulated to move downstream to areas of higher salinity by the first freshwater flow in spring that lowers the salinity of estuarine waters (Rod Garrett pers. comm. 2000). This effect could be achieved by small freshwater flows that do not necessarily release landlocked individuals.

In Australia, barramundi spawn during spring and summer. The timing and duration of spawning varies between regions, rivers and years (**Table 2.2**). The commencement and completion of a major part of the breeding cycle before the onset of the wet season is possibly a strategy for avoiding low-salinity water (Russell and Garrett 1985). Davis (1985, p 188) commented on the variation in the timing and duration of barramundi spawning and concluded that “there is considerable variation in the timing and duration of breeding between regions, rivers and from year to year, but essentially it is synchronized so that juveniles can take advantage of the aquatic habitat that results from rains in the monsoon season". Davis (1985, p 189) goes onto say that "these vast aquatic habitats formed during the summer monsoons provide juvenile barramundi with an almost predator-free prey-rich environment promoting rapid growth and improved survival". Griffin and Kelly (2001, p 7) report that "monitoring of recruitment suggests that rainfall is an important influence, presumably through its effect on the availability and habitability of swamp habitat, particularly in the early part of the spawning season".

Table 2.2 Variation in spawning season of barramundi in Australia

Location	Spawning season	Source
Northern Territory	September to February	Davis (1985)
Southern Gulf of Carpentaria	November to March, peak in December	Davis (1985)
Northern Gulf of Carpentaria & Far northern east coast of Queensland	From October	Williams (2002)
Queensland east coast	November to February (peak)	Stuart (1997)
Southern Queensland east coast	Possibly as late as March/April	Williams (2002)
(e.g. Rockhampton)	October to January	Dunstan (1959)

Barramundi on the Queensland east coast are single spawners (Russell 1990), unlike barramundi in the Gulf of Carpentaria (Embley and Hay Rivers), Papua New Guinea and Thailand that are multiple spawners. Davis (1987) speculated that a prolonged spawning season might be the result of landlocked fish arriving at spawning grounds 'late'. Dunstan (1959) suggested that 'spawning of river barramundi occurs just prior to or during the wet season, usually October to January (spawning season 1)' and that 'the barramundi land-locked in coastal lagoons and swamps are released when the wet season floodwaters connect these areas with the estuaries or open sea, they usually spawn from January to March (spawning season 2)'. Garrett (1987) suggested that it was unlikely that late spawning resulted in significant recruitment during 'normal' wet seasons because the late-spawned recruits were unlikely to survive the high predation pressure from early-spawned recruits already occupying nursery areas. However, prolonged spawning, resulting in multiple recruitment, could be important in overcoming climatic variability such as late or interrupted monsoons (Garrett 1987). Localised spawning occurs in Australia (*c.f.* Papua New Guinea) because of the relatively small discharge of freshwater from northern Australian rivers (Russell and Garrett 1985). Dunstan (1959) suggested that 'in years when the wet season is not pronounced, and floodwaters are insufficient to release land-locked adult barramundi, the number of spawning fish at sea is greatly reduced'.

In general, spawning activity peaks during new and full moon periods (Grey 1987), i.e. the week following new and full moon in northern Gulf of Carpentaria estuaries. Spawning on the incoming tide to may help eggs to move into estuaries. Garrett (1987, p 39) suggests that 'the position of spawning grounds probably differ slightly from year to year, depending on coastal salinities' and that 'these in turn vary with the degree of river discharge'.

Gametogenesis in barramundi is initiated by the seasonal increase in water temperate and photoperiod (Russell 1990). Movement to spawning areas is triggered by the seasonal increase in water temperature (Grey 1987). High salinity appears to be the main requirement of spawning grounds (Davis 1987), i.e. 32 to 38‰ (Rod Garrett, pers. comm. 2000). However, Maneewong (1987) found that hatching of fertilised barramundi eggs occurs in salinities between 20 and 30‰, with lowered hatching success at salinities lower or higher (i.e. 5-15‰ or 35‰). Fertilized barramundi eggs are pelagic, average 0.7 mm in diameter (Russell and Garrett 1985), and hatch within a day at 28°C. The eggs then incubate for 17 to 18 hours at about 30°C (Rod Garrett, pers. comm. 2000). Barramundi larvae spend about three weeks in inshore waters and require brackish water for embryonic development (Kailola *et al.* 1993). Davis (1987) caught small barramundi larvae 77 km upstream in the South Alligator River in the Northern Territory.

Barramundi post-larvae move into nursery habitats and utilise freshwater habitats if available (Russell and Garrett 1985). Moore (1980) suggested that barramundi larvae are cued or attracted upstream by chemicals released from swamps. Peak spring tides and seasonal flooding assist barramundi post-larvae to enter supra-littoral habitats (Russell and Garrett 1985), coastal lagoons (Grey 1987) and other seasonal habitats that form during the monsoon season (Williams 2002). Coastal swamps (i.e. adjacent to the coast and estuary) form the predominant nursery habitat for post-larval barramundi in areas of north-eastern Queensland where large river systems are absent (Russell and Garrett 1985). Monsoon rains also create a variety of temporary nursery habitats for juvenile barramundi that are highly productive in food resources and are thought to offer some protection from larger predatory fish. These swamps rely on 'flood rains' to connect with more permanent waters (Russell and Garrett 1985). Juvenile barramundi were recorded moving into supra-littoral pools in the Fitzroy River estuary during March (Hyland 2002). Griffin (1985) suggests that rainfall replenishes the water levels in supra-littoral habitats between high tides (thereby maintaining these nursery habitats for longer periods) and that 'the amount of time that the young-of-the-year fish are able to utilise this safe and rich environment is limited by the amount and extent of rainfall during the wet season'. Griffin (1985) only considers rainfall, although it is possible that floods that inundate flood plains may have a similar effect in extending the spatial and temporal extent of these high quality nursery habitats. This relationship was further confirmed with an additional two years of data, when Griffin (1987) reported a significant correlation (r=0.81) between juvenile abundance (i.e. young-of-the-year) and early wet season rainfall. Griffin (1994) reports that strong and weak year-classes are correlated with the amount of rainfall during the spawning season, particularly the early wet season, based on sampling of Yellow Waters Billabong (Kakadu National Park, Northern Territory),

The lowering of water-levels and depletion of food in seasonal habitats is likely to stimulate juvenile barramundi to move to other habitats (Russell and Garrett 1985). For example, juvenile barramundi began moving from swamps in Trinity Inlet (Cairns) in April and remained in tidal creeks until December and January (Russell and Garrett 1985). In the Gulf of Carpentaria, floodwaters recede around March. Many juvenile barramundi move to permanent freshwater habitats when the seasonal coastal habitats dry-out (Russell and Garrett 1985). Juvenile barramundi in Papua New Guinea waters take more than one year to reach inland freshwater habitats because of the need to migrate along the coast from spawning areas.

Barramundi are suggested to move into freshwater habitats when they are between one and two years old (Williams 2002). Dunstan (1959) suggested that '1+ fish are found in deep holes of the upper reaches, with 1+ fish being plentiful below the falls on the Burdekin River that are about 120 miles from the mouth, 1+ fish are common in the Dawson River and other tributaries of the Fitzroy River'. Fishway studies have recorded juvenile barramundi (120 to 500 mm fork length (FL)) moving upstream through the Fitzroy River fishway primarily between spring and early summer i.e. October to December (Stuart 1997). The highest number of barramundi moving through the fishway were recorded during low flows when salinities below the barrage were high (i.e. October, November and December). However, barramundi were recorded moving through the Fitzroy River fishway (which flows at 18 ML per day) throughout the year (i.e. June, September, January, February, March, May). All barramundi ascending the Fitzroy River fishway were <600 mm FL and immature (Stuart and Mallen-Cooper 1999). They moved upstream in equal numbers in the day and night, being the only species to do so. Stuart and Mallen-Copper (1999) noted that most barramundi <310 mm FL were collected during the two days immediately after the barrage gates were closed and the only source of freshwater flow to the estuary was from the fishway.

Dunstan (1959) suggested that the 'abundance of barramundi in Queensland east coast rivers appears to be closely associated with the flow of freshwater (within the year)'. Annual wet-season rainfall is thought to influence adult spawning success and juvenile recruitment (Williams 2002). Breeding success is enhanced in high rainfall years and such events are often followed by a strong year-class evident in the fishery some years later as evidenced by higher catches (Williams 2002). Rainfall levels in the southern Gulf of Carpentaria are thought to influence catches four to seven years later (Williams 2002), with a direct correlation between monthly freshwater discharge and monthly catch (Williams 2002). In Queensland, commercial catches of barramundi are highest prior to, and immediately following, the breeding season (i.e. October/November and February to April) (Williams 2002), however, this also coincides with increased effort before and after the annual seasonal closure for commercial fishing.

Barramundi stocks in northern Australia are genetically different between the Gulf of Carpentaria and the Queensland east coast (Shaklee and Salini 1985; Salini and Shaklee 1988; Williams 2002). Davis (1985, p189) suggests that because of localised spawning and genetic evidence of stock heterogeneity that 'recruitment into major river systems would depend largely on the successful spawning of local populations' and that 'the populations in different river systems may be quite independent of each other, and it may be appropriate to manage them as separate stocks'.

Size-at-maturity also varies between areas (Table 2.3). Davis (1982) reported that size-at-maturity for males was 600 and 550 mm for fish in the Northern Territory and south-eastern Gulf of Carpentaria respectively and for females was 900 and 850 mm. Davis (1982) went on to speculate that these 'size differences were due to a slower growth rate of barramundi in the Gulf of Carpentaria, both processes being related to age rather than size'. Griffin (1988) also speculates that growth rates differ between the Daly and Liverpool Rivers (Northern Territory) based on differences in the size-at-age structure in the two rivers.

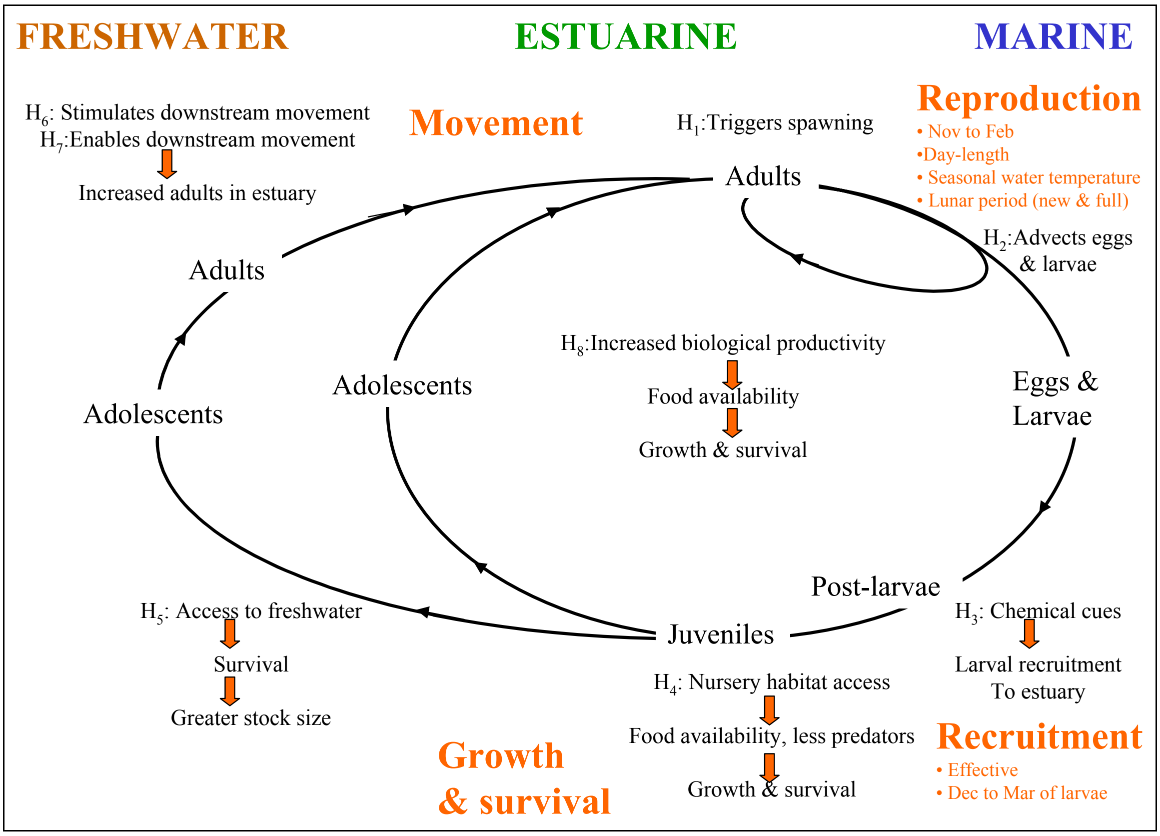
Table 2.3 Size and age to maturity of barramundi in Australia

Location	Males (TL mm)	Females (TL mm)
Northern Territory	600	900
Southern Gulf of Carpentaria	550 mm	850
Southern Gulf of Carpentaria	620 to 650 mm, 3 to 4 years	~900 mm, 7 to 8 years
Queensland east coast	620 to 650 mm, 3 to 4 years	~900 mm, 7 to 8 years
Cape York Peninsula	300 mm males	500 to 600 mm

Davis (1987) reported an ontogenetic change in diet for barramundi from micro-crustaceans to macro-crustaceans to fish.

Populations of barramundi are likely to be highly responsive to freshwater flows given their life history preference to utilise freshwater habitats where accessible or available, and the residency of most individuals within a region, thereby reflecting local freshwater flow conditions over time. Life history information was conceptualised into a life-cycle figure, with the potential influence of freshwater flow included (Figure 2.3).

Figure 2.3 Conceptual model of the influences of freshwater flows on the life-cycle of barramundi



Specifically, we identified eight hypotheses of the influence of freshwater flow on barramundi populations. They were that freshwater flow:

1) Triggers spawning of mature fish at the mouth of the estuary (although unlikely as high salinity water is required for the survival of barramundi eggs).
2) Affects the distribution of eggs and larvae; large flows may wash eggs and larvae away from the estuarine water (i.e. a negative effect).
3) Creates chemical signals that cue larvae to enter the estuary.
4) Connects the estuary and ephemeral supra-littoral nursery habitats (e.g. floodplains and coastal swamps); post-larval and small juveniles use these connections to enter these temporary nursery habits, which allow faster growth and better survival (i.e. enhances the available nursery areas).
5) Connects the estuary and perennial freshwater habitats; large juveniles use the connections to move into freshwater habitats, which allow good growth and survival.
6) Stimulates mature barramundi to move downstream in preparation for seasonal spawning, although the stimulus is unknown. Potentially it could be changes in salinity for estuary-based individual or flow rates for freshwater-based individuals.
7) Connects perennial freshwater habitats (including the overflow of impoundments) and the estuary, enabling mature individuals to move downstream and participate in seasonal spawning and enlarging the fished stock size.
8) Enhances biological productivity of the estuary, thereby increasing the availability of food for juveniles, adolescents and adults resident in the estuary, which potentially results in improved growth, survival and 'condition' of the estuarine population (e.g. fat barramundi syndrome, leading to faster age-at-maturity or greater reproductive output for a season).

These potential causal mechanisms can be grouped into effects on: (i) catchability in the estuary by a) stimulating the downstream migration of mature barramundi (in preparation for estuarine spawning), b) enabling downstream migration of mature individuals through the connection of habitats intermittently linked to the river or estuary, and c) changing the distribution of individuals within the estuary (by stimulating within-estuary movement), thereby increasing the chance of being caught in the set gill-net fishery; (ii) recruitment, by a) transporting eggs and larvae away from the estuary (negative effect), b) creating chemical signals for larvae to enter the estuary and locate nursery habitats (positive effect), c) enabling post-larvae and small juveniles to move into supra-littoral nursery habitats, and d) enabling large juveniles to migrate into freshwater habitats intermittently linked to the estuary; and (iii) on productivity, by increasing food availability as a consequence of enhanced biological productivity of the estuary, thereby improving the growth and survival of post-larvae, juveniles, adolescents and adults.

Mullet (*Mugil cephalus*)

Mullet is the common name applied to a group of species that have similar features, but most commonly refers to the sea (grey or stripped) mullet (*Mugil cephalus*). Other mullet species include diamond scale mullet (*Liza vaigiensis*) and flat-tailed mullet *(Liza argentea*), and are the main mullet species caught in the Gulf of Carpentaria. Sea mullet is the main species caught on the Queensland east coast (Williams 2002), having a minimum legal size of 300 mm total length in Queensland.

Mullet utilise fresh, estuarine and coastal waters (Kailola *et al.* 1993) and have a life-cycle similar to that of barramundi. Spawning must occur in marine waters; juveniles and adults use freshwater habitats where accessible, but can develop in estuarine habitats where access to freshwater habitats is restricted (i.e. opportunistically catadromous, Thomson 1963).

Spawning is preceded by the downstream movement of mature individuals from freshwater and upper reaches of estuaries in late summer, often coinciding with seasonal rain and freshwater flow. Schools of mullet aggregate in the lower reaches of estuaries, until movement outside the estuary to coastal waters is stimulated by offshore winds (i.e. westerly winds) in autumn or winter (Thomson 1963). Spawning schools of mullet predominately travel northwards along the east Australian coast (Kailola *et al.* 1993; Williams 2002), and may be caught a substantial distance away from the river system in which they initially developed. It is also thought that post-spawning adult mullet return to estuaries, but that individuals enter more northerly rivers after spawning than where they originated from (Kailola *et al.* 1993).

In addition to the movements associated with spawning, mullet commonly have a ‘hard-gut’ migration of immature fish along coastal beaches. These schools are though to originate from local rivers and may later return to estuarine or freshwater reaches. It is speculated that they may be stimulated to move as a consequence of flooding in rivers, often being referred to as ‘wash-out’ mullet in New South Wales and Queensland (Thomson 1963).

Mullet have a protracted spawning season (i.e. March to July) with most spawning occurring in winter (Williams 2002). Spawning occurs in marine waters (Thomson 1963), adjacent to the surf zone (Kailola *et al.* 1993). Eggs and larvae require high salinity water and drift from marine spawning grounds with prevailing currents for about two to three months, until they reach 20 to 30 mm FL. Post-larval mullet move into estuaries from September onwards (Thomson 1963). Lowered salinities are thought to attract mullet fry (Thomson 1963). Small juvenile mullet will disperse into freshwater habitats (where accessible) before the end of November. Mullet migrate through the fishway on the Fitzroy River barrage in most months (i.e. June, October, November, December, January, February, March, and May) with greatest numbers migrating in November (Stuart 1997). Small mullet (8 to 102 mm *Liza* sp. and 32 to 65 mm *Valamugil* sp.) were recorded in temporary supra-littoral pools in the Gulf of Carpentaria (Russell and Garrett 1983), suggesting that mullet species will opportunistically use available nursery habitats.

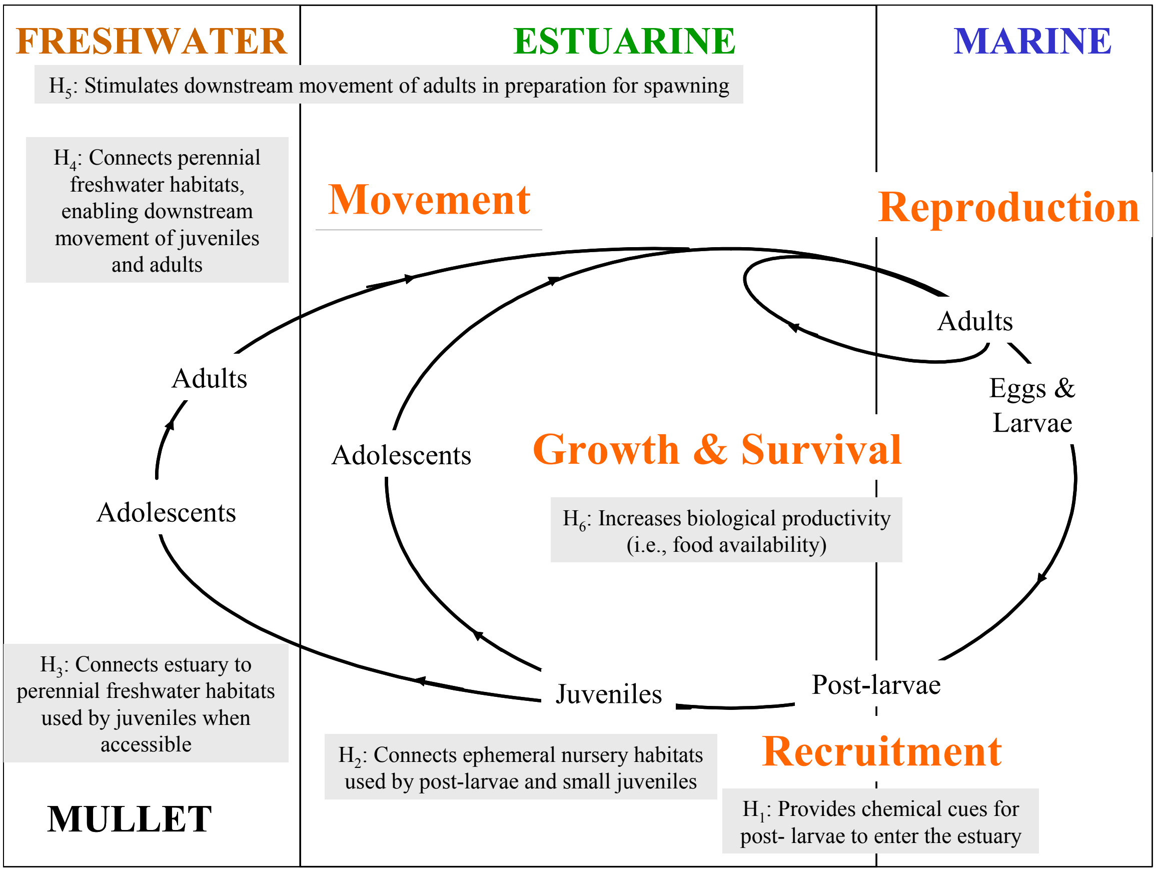
Estuaries are used by juvenile mullet, which often school during the ebb tide, but then disperse over sand and mud flats to feed during high tide (Thomson 1963). Mullet mature after about three years (Kailola *et al.* 1993), with individuals between three and six years participating in the spawning migrations (Virgona *et al.* 1998). Estuarine catches of sea mullet in southern Queensland are dominated by two-year olds, but a considerable proportion of the catch is comprised of fish ranging in age from three to seven years. The ocean beach fishery in southern Queensland is dominated by older fish, i.e. four- to six-year olds (Dichmont *et al.* 1999).

Cardona (2000), using experimental and wild studies, demonstrated that the growth of sea mullet was highly dependent on salinity values and that this changed ontogenetically. Cardona suggests that 'mullet with a total length between 40 and 300 mm are highly dependent on areas with low salinity and hence any human activity reducing the availability of such an environment will negatively affect the fisheries of this species'. However, Cardona cautioned that this may not apply to Indo-Pacific populations (i.e. those of Hawaii).

Mullet are opportunistic omnivores, feeing on detritus, diatoms, algae and micro-invertebrates filtered from muddy and sandy substrates in estuaries, but feed mainly on algae in freshwater habitats (Thomson 1963).

Populations of mullet are likely to be highly responsive to freshwater flows given their life history preference to utilise freshwater habitats where accessible or available, and the anecdotal belief that flows stimulate the movement of mullet downstream. However, relationships between freshwater flow and fisheries catch may be confounded by the migration patterns of mullet, which undertake substantial longshore movements. The presence of numerous species contributing to the commercial catch in some areas (particularly northern Australia) may also confound relationships. The alteration of river systems through the construction of dams and weirs reduces the availability and or accessibility of brackish and freshwater habitats for juvenile mullet. This could potentially be a factor limiting the productivity of the mullet population and have consequential impacts on the size of the resulting mullet fishery (Williams 2002). Life history information was conceptualised into a life-cycle figure, with the potential influence of freshwater flow included (**Figure 2.4**).

Figure 2.4 Conceptual model of the influences of freshwater flows on the life-cycle of sea mullet



We identified six hypotheses of the role of freshwater flow in the life-cycle of sea mullet. They were that freshwater flows:

1) Creates chemical signals that cue sea mullet larvae to enter the estuary.

2) Connects the estuary and ephemeral supra-littoral nursery habitats (e.g. floodplains and coastal swamps) that post-larval and small juveniles use opportunistically; these habitats may allow faster growth and higher survival of young-of-the-year (i.e. enhances the available nursery areas).

3) Connects the estuary and perennial freshwater habitats; large juvenile sea mullet use the connections to move into freshwater habitats, which allow faster growth and higher survival.

4) Stimulates mature sea mullet to move downstream in preparation for seasonal spawning, although the stimulus is unknown. Potentially it could be changes in salinity for estuary-based individual, or flow rates for freshwater-based individuals. Also may induce immature mullet to move downstream and undertake the 'hard-gut' migration as a consequence of 'washout' effects.

5) Connects perennial freshwater habitats (including the overflow of impoundments) and the estuary, enabling mature sea mullet to move downstream and participate in seasonal spawning and enlarge the fished stock size.

6) Enhances biological productivity of the estuary, thereby increasing the availability of food for juvenile, adolescent and adult sea mullet resident in the estuary, which potentially results in improved growth, survival and 'condition' of the estuarine population, leading to faster age-at-maturity or greater reproductive output.

King threadfin (*Polydactylus macrochir*)

King threadfin (previously *Polydactylus sheridani*) are another important fishery species that utilise estuaries in northern Australia. They have a complex life history, being protandrous hermaphrodites (i.e. males first then females), and utilise estuarine and associated coastal foreshore waters. King threadfin do not use freshwater during any life history stage, although adults can be found upstream during winter, as saline waters intrude up the estuary (Ian Halliday pers. obs.). Preliminary genetic tests indicate that distinct populations of king threadfin occur in Queensland (i.e. east coast versus Gulf of Carpentaria), but that there is little evidence to suggest genetic differences between stocks in the Gulf of Carpentaria (Garrett 1997).

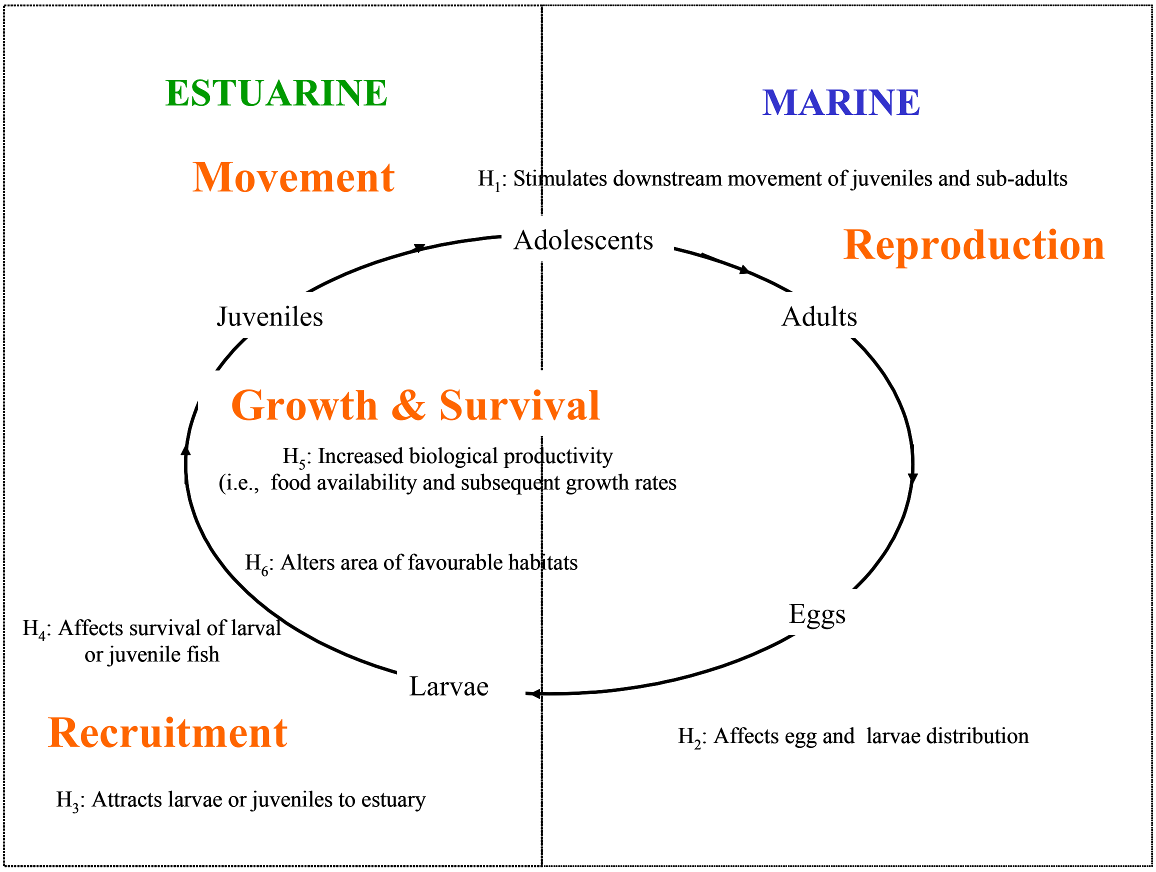
Adults spawn in inshore coastal waters away from river mouths (Williams 2002), and are likely to have an extended spawning season (Garrett 1997). Spawning peaks during late winter or early spring (Garrett 1997; Welch *et al.* 2002), occurring in September and October in ‘northerly stocks’ and November and December in more ‘southerly stocks’ (Garrett 1997). In the Northern Territory, king threadfin are reported to spawn from October to March, with a peak in December (Kailola *et al.* 1993).

Spawning is thought to occur in high salinity water (>32‰ Rod Garrett pers. comm.) and it is likely that the pelagic eggs require salinities near that of seawater for high survival rates. The early life history of this species is poorly quantified, although nursery areas are probably inshore, shallow and of lesser salinity than that of seawater (Kailola *et al.* 1993; Williams 2002). Juvenile fish (~100 mm FL) appear in estuaries of the Queensland east coast (i.e. between Townsville to Cairns) in January (Ian Halliday, pers. obs.). No king threadfin were recorded in temporary supra-littoral pools in the Gulf of Carpentaria (Russell and Garrett 1983), suggesting that king threadfin restrict their use of estuarine habitats to permanent water areas in the main channels and tributaries of creeks and rivers. The level of wet season rainfall is suggested to influence adult spawning success and juvenile survival (Williams 2002), but this has not been investigated or quantified. King threadfin can move large distances along the coastline (e.g. 550 km Kailola *et al.* 1993), potentially confounding relationships between freshwater flow and the species abundance and distribution.

King threadfin reach maturity at between two to five years and a size at 600 to 800 mm FL (Roelofs 2003). Most individuals change from males to females between the age of six and ten years and at a size of 750 to 1000 mm FL. Minimum legal size is 400 mm TL for Queensland east coast and 600 mm TL for the Gulf of Carpentaria (Welch *et al.* 2002). Peak inshore catches occur in late summer and autumn, coinciding with the movement of commercial fishers from riverine to foreshore areas (Williams 2002). This reflects a change in targeting by commercial fishers from barramundi to king threadfin.

King threadfin are carnivorous, eating a variety of the seasonally available small fish species and crustaceans, including penaeid pawns (Salini *et al.* 1998). Life history information was conceptualised into a life-cycle figure, with the potential influence of freshwater flow included (**Figure 2.5**).

Figure 2.5 Conceptual model of the influences of freshwater flows on the life-cycle of king threadfin



Based on the limited information in the literature, it is possible that freshwater flow influences the catchability and or the recruitment of king threadfin. We identified six hypotheses of the influence of freshwater flow on king threadfin populations. They were that freshwater flows:

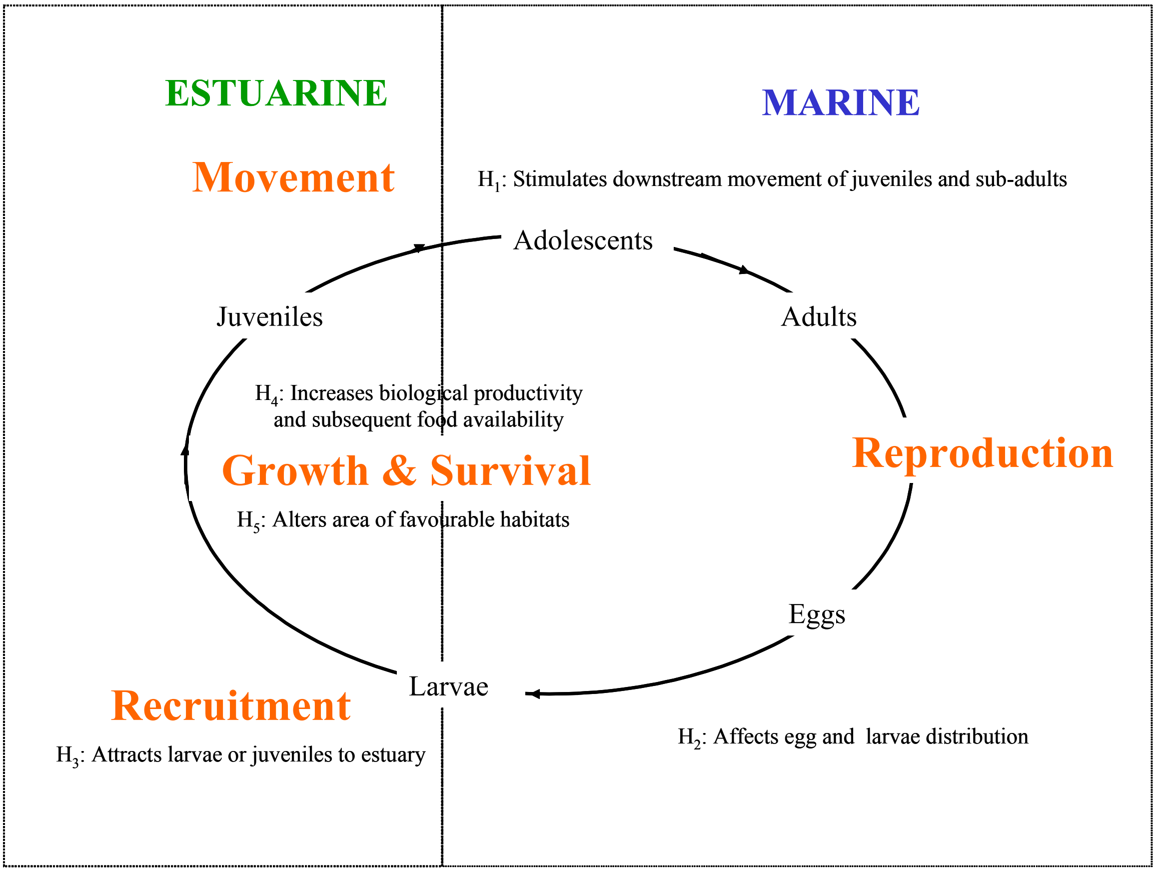
1) Stimulate downstream movement of juvenile and sub-adult king threadfin, potentially increasing their catchability in set-net fisheries through increased movement or increased densities in fished areas. The stimulus is possibly salinity changes.
2) Affects the transport of larvae through currents. Large freshwater flows may prevent larval immigration to estuarine nursery habitats by washing eggs and larvae away from the estuary.
3) Creates chemical signals for larvae or juveniles to enter the estuary.
4) Affects the survival of larval or juvenile fish in estuarine habitats, with freshwater (i.e. salinity of 0‰) being unsuitable habitat.
5) Enhances the biological productivity of the estuary, thereby increasing the availability of food in the estuary for larvae, juveniles and adolescents which results in improved growth and survival, potentially leading to greater year-class strength.
6) Affects the area of favourable habitat for juveniles and adolescents, potentially through larger areas of decreased salinity, the creation of a salinity gradient (i.e. 5 to 30‰) or turbid conditions reducing predation, which increases the survival of juveniles and adolescents.

Blue threadfin (*Eleutheronema tetradactylum*)

Blue threadfin (previously known as blue or Cooktown salmon) contribute to the estuarine fisheries of northern Australia (Kailola *et al.* 1993). Like barramundi and king threadfin they are protandrous hermaphrodites (i.e. males first then females, Garrett 1997). They utilise estuarine and associated coastal foreshore waters. Nursery areas of blue threadfin are thought to be inshore and shallow areas of low salinity (Kailola *et al.* 1993). Juveniles and adult blue threadfin occur in near shore habitats, usually away from the direct influence of freshwater discharge (Williams 2002). Juvenile blue threadfin (<100 mm FL) recruit into estuaries of north Queensland in January (Ian Halliday, per. obs.) and have been recorded as 'occasional occupants' of temporary supra-littoral pools in the Norman River (Gulf of Carpentaria, Russell and Garrett 1983), being captured in December. Blue threadfin in spawning condition are caught throughout most months, but are most numerous between August and May (Williams 2002). The breeding season for blue threadfin in waters of the Gulf of Carpentaria extends from early winter into summer, probably finishing before the annual wet season rains degrade near shore environmental conditions suitable for spawning and larval development (Garrett 1997; Williams 2002). Blue threadfin reach maturity at between one- to two-years old and at size at 200 to 300 mm FL (Garrett 1997; Roelofs 2003). Blue threadfin are males first, but are hermaphrodites or functional females by four-years of age (Garrett 1997).

Like king threadfin, blue threadfin also undertake longshore coastal migrations of at least 150 km (Kailola *et al.* 1993), potentially confounding relationships between freshwater flows and their abundance and distribution. Blue threadfin are opportunistic carnivores, eating small fish (mullids, mullets, and ponyfish) as well as penaeids (including banana prawns) alphaeids, crabs, mantis shrimp, bugs, gastropods, bivalves, octopus, cuttlefish and squid (Salini *et al.* 1998). Therefore, blue threadfin are likely to take advantage of any trophic bloom occurring as a result of a freshwater flow. We conceptualised life history information for blue threadfin into a life-cycle figure, and included the potential influence of freshwater flow (**Figure 2.6**).

Figure 2.6 Conceptual model of the influences of freshwater flows on the life-cycle of blue threadfin



Blue threadfin opportunistically use estuaries and based on the limited information in the literature, we identified five hypotheses of the potential influence of freshwater flow on blue threadfin populations. They were that freshwater flows:

1) Stimulates downstream movement of juvenile and sub-adult king threadfin, increasing their catchability in estuarine and coastal set-net fisheries through increased movement or increased densities in fished areas. The stimulus is possibly avoidance of very low salinity water.
2) Affects the transport of larvae through currents. Large freshwater flows may prevent larval immigration to estuarine nursery habitats by washing eggs and larvae away from the estuary.
3) Creates chemical signals for larvae or juveniles to enter the estuary.
4) Enhances biological productivity of the estuary, thereby increasing the availability of food for larvae, juveniles and adolescents which results in improved growth and survival, potentially leading to greater year-class strength. This occurs after the freshwater flow event as saline waters push back up the estuary.
5) Affects the area of favourable habitat for juveniles and adolescents, potentially through larger areas of decreased salinity habitat (but not freshwater = 0‰), the creation of a salinity gradient (i.e. 5 to 30‰) or turbid conditions reducing predation, which increases the survival of juveniles and adolescents.

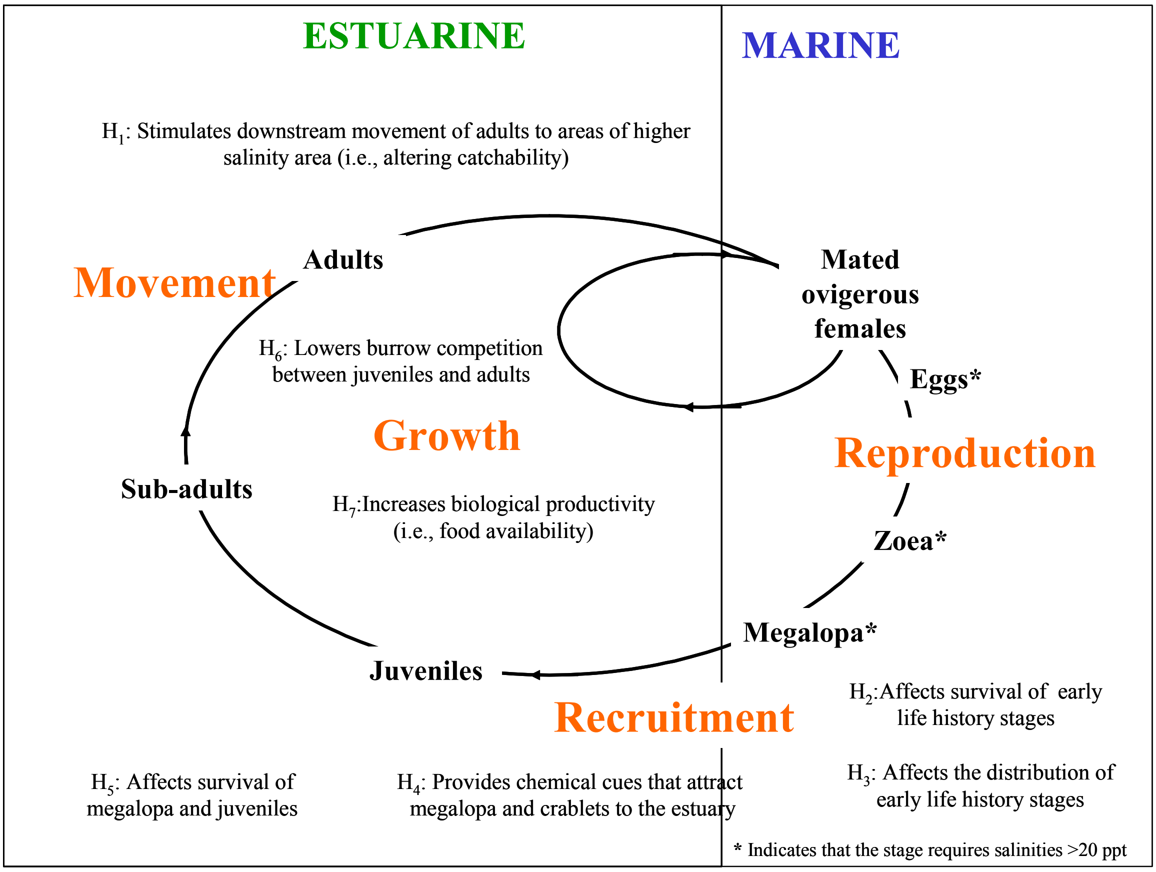
Mud crabs (*Scylla serrata*)

The mud crab, also known as the mangrove crab, is a portunid crab characteristically associated with mangrove areas. We found conflicting evidence in the available life history information about the influence of freshwater flows on mud crabs. The life-cycle of the mud crab involves several stages and uses both marine offshore areas and estuaries (Arriola 1940). In tropical Australia, mated ovigerous female mud crabs migrate to offshore waters, but the timing of the migration (i.e. before the monsoon season), suggests that the spawning migration is not triggered by low salinities in estuaries (Hill 1994). Early life history stages of mud crabs (i.e. eggs, zoeal and megalopal larval stages) require high salinities (i.e. >20‰), with considerable mortality occurring at salinities below 20‰ (Hill 1974; Quinn and Kojis 1987). Recruitment success of mud crabs in Madagascar was estimated to be seasonal and inversely related to rainfall (Le Reste *et al.* 1976). However, flooding associated with cyclones had little measurable effect on the recruitment of the megalopa larvae of mud crabs in the St Lucia estuary, South Africa (Forbes and Hay 1988). Under experimental conditions, adult mud crabs showed varying levels of mortality after exposure to different salinities, but did not show an ability to discriminate between salinities (Davenport and Wong 1987). However, during salinity preference experiments, mud crabs were only allowed 30 minutes to choose a salinity (Davenport and Wong 1987), which may not be a sufficient period for the crabs to react.

Fisheries for mud crabs are associated mostly with estuaries. Within-year variation in catch rates of mud crabs in sub-tropical Queensland were positively correlated with temperature and the incidence of moulting (Hill 1982; Williams and Hill 1982). Williams and Hill (1982) found that catches were positively correlated with daily water temperature (r=0.56, n=44), but not with salinity (r=0.09, n=44), where salinity ranged between 24 and 35‰. However, low catches of mud crabs in the Gulf of Carpentaria have been attributed by commercial fishers to high migration rates of mud crabs out of fishing areas and recruitment failure occurring as a consequence of extended periods of freshwater runoff (Helmke *et al.* 1998). The downstream movement of mud crabs following floods was also reported by Stephenson and Campbell (1960), and Hill (1975) reported that heavy floods (with salinity dropping to 2‰) eliminated or severely reduced the number of mud crabs in two South African estuaries.

Mud crab catch has been positively correlated with summer freshwater flow in a sub-tropical estuary (Loneragan and Bunn 1999). Loneragan and Bunn (1999) suggested that freshwater flow might influence the catchability of mud crabs by stimulating their downstream movement away from low salinity water (thereby increasing their density in fishing grounds), affecting recruitment by reducing the competition for burrows and increasing the survival of juveniles. Increased juvenile survival would suggest that enhanced catches of mud crab could occur in the following years (i.e. a lag effect), but lag correlations were not examined by Loneragan and Bunn (1999). Life history information was conceptualised into a life-cycle figure, with the potential influence of freshwater flow included (**Figure 2.7**).

Figure 2.7 Conceptual model of the influences of freshwater flows on the life-cycle of the mud crab



We identified seven hypotheses of the role of freshwater flow in the life-cycle of the mud crab. They were that freshwater flows:

1) Stimulate the movement of mud crabs downstream to higher salinity areas as a consequence of a dislike for low salinity waters possibly increasing catch rates.
2) Affects the survival of early life history stages; eggs, zoea and megalopa probably require salinity > 20‰.
3) Washes megalopa or crablets out of the estuary.
4) Creates chemical signals for megalopa or crablets to enter the estuary.
5) Affects the survival of megalopa and juvenile mud crabs via salinity preferences.
6) Displaces adult mud crabs downstream, leading to lowered burrow competition by juveniles.
7) Enhances biological productivity of the estuary, thereby increasing the availability of food for juveniles, adolescents and adults, which potentially results in improved growth, survival and 'condition'.

Based on the available information in the literature, we identified that freshwater flow might influence: i) the catchability and ii) the recruitment of mud crabs, but we can not speculate further in the exact mechanisms of this influence, given the conflicting evidence in the literature.

Chapter 3. Correlating catch and freshwater flow

J. Robins, I. Halliday, J. Staunton-Smith, D. Mayer, and M. Sellin [4]

Summary

Commercial catches of fishery species were analysed for the Fitzroy and Port Curtis estuaries of central Queensland as a case study of correlations between catch and freshwater flow. Barramundi and banana prawns were significantly correlated with flow in the same year. Barramundi catches were also significantly and positively correlated to flow lagged by three and four years. Correlations between mud crab catch and freshwater flow were ambiguous.

The correlations indicated that the total catch of banana prawns increased in proportion to summer freshwater flow. The temporal and spatial aspects of the correlation supported the proposed hypothesis that freshwater flow affects the catchability and recruitment of banana prawns. Freshwater flows may increase the abundance or biomass of banana prawns as a consequence of improved growth and survival through trophic cascading, but evidence for this is unlikely to come from within-year analyses of commercial catch data. Further investigation found that the growth rates of juvenile banana prawns were significantly and positively related to freshwater flow (see Chapter 9) and coincided with greater abundance of juvenile prawns in the estuary and increased commercial catches (see Chapter 9).

The within-year influence of freshwater flow on barramundi catches supports the proposed hypotheses for the effect of flow on catchability and concurs with anecdotal reports by commercial fishers. The significant correlations between catch and freshwater flow or rainfall lagged by three or four years provides consistent quantitative evidence in support of the positive influence of freshwater flow on the recruitment of barramundi. This effect is further supported by results from an age-based index of year-class strength that confirmed recruitment was strongly correlated to freshwater flow in summer and spring (see Chapter 6). In addition, analysis of tag-recapture information from the long-term ANSA tagging program supports hypotheses that growth is strongly influenced by freshwater flow (see Chapter 7) and that growth rates were enhanced in fish that accessed freshwater habitats as juveniles (see Chapter 8).

There was some evidence that freshwater flows in autumn may affect the recruitment (i.e. cohort or year-class) strength of mud crabs. However, the biological link between autumn flows and the life history of mud crabs requires further investigation, perhaps into the hypothesis that recruitment effects are related to burrow competition.

The relationships between estuarine fisheries production and freshwater flow reported in this chapter provide quantitative evidence that the productivity of estuarine fisheries in northern Australia is related to freshwater flow.

Introduction

Fisheries landings are often speculated to be linked to environmental variables, such as rainfall, freshwater flow, temperature etc. The postulation justifying such a relationship is often based on theoretical or known aspects of the life-cycle of fishery species, anecdotal reports from experienced commercial or recreational fishers and or significant correlations between environmental variables and catch statistics either demonstrated for a species in a location or a similar species in another location.

Fisheries landings are attractive in the investigation of relationships between environmental variables and biological production (of which fisheries landings is a component), because fisheries

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[4] Parts of this chapter are published in: Robins, J.B., Halliday, I. A., Staunton-Smith, J., Mayer, D.G. and Sellin, M.J. (2005). Freshwater flow requirements of estuarine fisheries in tropical Australia: a review of the state of knowledge and application of a suggested approach. *Marine and Freshwater Research* 56, 343-360.

data sets are often the only data available over a 'long' time series, which allows for an analysis across a range of environmental variation. Fisheries landings (i.e. recreational or commercial catch or landings) are also attractive because they have a tangible, economic and social value, making them useful when considering the effects of environmental variation that can be manipulated by humans i.e. freshwater flow or habitat availability.

Our analyses aimed to determine whether there was evidence of potential relationships between catch and freshwater flow for the major estuarine fishery species in central Queensland, and if so, which theoretical mechanisms generated from the life history assessments were supported most consistently by the correlative analyses.

Methods

Study areas

Estuaries in central Queensland were used as a case study to determine the effects of freshwater flow on commercial fisheries of tropical and sub-tropical estuaries and inshore areas of northern Australia. Central Queensland estuaries are typical of northern Australia in that they have associated trawl fisheries for penaeid prawns, net fisheries for estuarine finfish and trap fisheries for crabs. Some estuaries have a two component trawl fishery: a within-estuary (river) beam-trawl fishery and an offshore (and coastal foreshore) otter-trawl fishery (e.g. the Fitzroy River), whilst others only have an offshore and coastal foreshore otter-trawl fishery (e.g. Port Curtis). The latter is the typical arrangement in the Gulf of Carpentaria and Northern Territory (i.e. Northern Prawn Fishery), where there is only an offshore otter-trawl trawl fishery.

The Fitzroy River region

The Fitzroy River has a large, mangrove-fringed estuary that straddles the Tropic of Capricorn, and receives freshwater flow from the largest catchment (142,537 km^2) on the east coast of Australia. A tidal barrage was constructed in 1970, about 50 km upstream of the river mouth. The substratum is predominately mud and there is a complex of islands and channels in the delta. Some islands are covered in mangroves, others have mangrove fringes with intertidal salt marshes, and there are also extensive salt pans (Long and McKinnon 2002).

The Fitzroy catchment is south of the summer monsoon trough that occurs seasonally over tropical Australia. High variation in annual freshwater flow and rainfall are characteristic of the region. Mean annual (i.e. September to August) freshwater flow is 5.2 million ML (164.8 m^3s^{-1}), with recorded minimum and maximum annual flows of 0.08 million ML (2.5 m^3s^{-1}) and 37.3 million ML (1,182.7 m^3s^{-1}) respectively. The pattern of freshwater flow in the Fitzroy River is typical of estuaries in sub-tropical and tropical Australia, being dominated by summer floods and winter droughts, but varying seasonally as a consequence of rainfall patterns. In general, seasonal increases in freshwater flow occur between November and May, with the largest average monthly flows occurring in February. Between June and October, freshwater flow can drop to almost zero. Water resources in the Fitzroy River are highly regulated, via 19 dams (including weirs) and one tidal barrage. However, this infrastructure is unable to withhold seasonal episodic floods associated with low pressure systems.

The Port Curtis region

Port Curtis is the next major estuarine region south of the Fitzroy River estuary and is comprised of Gladstone Harbour, Curtis and Facing Island and The Narrows. The Port Curtis region had about 80 km^2 of mangroves plus 100 km^2 of salt marsh, salt flat and mudflat (Saenger 1996). Extensive areas of tidal flats are unvegetated due to underlying clay soils which have poor drainage and high evaporation rates. Since 1941, 650 hectares of mangroves and 950 hectares of coastal salt flat have been lost, mostly due to industrial and urban development (Arnold 1996). Most of the development occurred after 1960. An increase in the area of *Rhizophora* and *Avicennia* fringing mangroves has been recorded for the Boyne River (Arnold 1996). Arnold attributed this increase to reduced downstream flows and scouring activity, which are the consequence of Awoonga Dam (on the Boyne River). Two major rivers discharge into Port Curtis: the Boyne River in the south and the Calliope River in the centre, as well as two minor creeks: Auckland Creek and Munduran Creek.

THE CALLIOPE RIVER ESTUARY

The Calliope River is situated at the northern end of the town of Gladstone. It has a catchment area of about 2,150 km^2 and is about 100 km in length. The estuarine section of the Calliope River is distinctly defined via an artificial causeway at Calliope Crossing (about 10 km upstream from the mouth). The causeway is only flooded by tides over 4.9 m in height. This barrier creates a definite partition between the marine and brackish sections of the river and has a significant effect on the free movement of fish and crustaceans between the estuarine section and waters further upstream (McKinnon et al. 1995). The estuarine section is relatively shallow and muddy, with regular large rocky outcrops, and is bordered for most of its length by a narrow fringe of mangroves. Salt marsh and salt pans form patches of habitat in the lower estuarine reach of the Calliope River. Mean annual (September to August) rainfall for the Calliope River catchment is 885 mm. Blain et al. (1980) modelled storm surges in the vicinity of Port Curtis and the Narrows and found a 1-in-50-year flood in the Calliope River to have an 'insignificant effect on levels outside the river mouth'. They suggest that the stormwater runoff from the Calliope River is insignificant in comparison of the volume of water oscillating in Gladstone harbour. However, more recently Currie and Small (2005) found significant correlations between freshwater flow of the Calliope River and turbidity and chlorophyll a concentrations at the Calliope River mouth. There is no water impoundment infrastructure on the Calliope River, but limited pumping licences allow for the removal of some freshwater.

THE BOYNE RIVER ESTUARY

The Boyne River catchment covers an area of 2,590 km^2. The main river channel is 120 km long, whilst the estuary is 20 km long. The estuary has a very narrow fringe of riparian vegetation. The Boyne River has been impounded since the mid 1960's. Awoonga Dam was originally constructed as a weir, but was raised in 1984 (to 30 m FSL) and again in 2005 (to 40m FSL). Downstream of the dam is Mann's Weir (17.1 km AMTD) and Pike's Crossing (20 km AMTD), which is a low-level causeway (AHD 1.7) that incorporates a culvert. The environment downstream of Awoonga Dam is effectively two freshwater pools, separated by Pikes' Crossing and bounded on the downstream side by Mann's Weir. This privately registered weir is generally the upper tidal limit (Anonymous 2000). The annual rainfall during the 1990's has been below the long-term average of 900 mm. This combined with the Awoonga Dam has resulted in very few freshwater flows passing the dam and flowing to the estuary in recent times.

Fisheries catch data

The landings data includes estuarine, coastal, and offshore species. Three species were selected for analysis: barramundi; banana prawns and mud crabs. These species were selected because they are highly estuarine, commercially important or comparable to other species groups analysed in similar studies. There was also greater certainty that these were single species and that the catch trends were probably not confounded by long-shore migration (see Chapter 2). These issues are considerable for commercial catch data of mullet, blue threadfin, king threadfin and whiting and as such, correlations were not conducted for these species.

Catch data were obtained from: (i) the financial year reports of the Queensland Fish Board (QFB) from 1945 to 1980; and (ii) the daily commercial fisheries logbook (CFISH) of the Department of Primary Industries and Fisheries, Queensland from January 1988 to June 2003. The QFB data represent fisheries catch passing through regional depots, with the majority of the fisheries catch passing through a depot caught in the nearby area. Appendix 3 provides more details about QFB data and an assessment of its reliability. For the QFB data, the Fitzroy region combines landings from the Rockhampton, Yeppoon and Rosslyn Bay depots, while the Port Curtis region uses landings from the Gladstone depot. The CFISH data represent the catch (by weight or number) recorded in spatial grids of 30^2nm (=1,668 km^2). For the CFISH data, catch in the Fitzroy region was assumed to be represented by CFISH grids R28, R29, R30 and S29, while catch in the Port Curtis region was assumed to be represented by CFISH grids S30 and T30.

Catch was aggregated into annual totals to investigate inter-year trends. However, QFB data were only available for financial year totals. CFISH data were aggregated to represent the seasonal trends in landings (as per Sobrino *et al.* 2002). For CFISH data, the starting month was September for banana prawns, but July for barramundi and mud crabs (i.e. financial year, consistent with QFB data). An additional adjustment that we made to the barramundi data set was to exclude the 1988 financial year data point because this was the first year of the compulsory CFISH logbook and anecdotal reports from fishers suggested that the catch might have been under-reported.

Only CFISH data were used in the analysis of banana prawn catches because the QFB data pooled all prawn species (i.e. banana prawns as well as eastern king prawns) and occurred during a time when the trawl fishery was rapidly expanding in northern Australia. Therefore, it was uncertain if the QFB data were an appropriate index of abundance of banana prawns. Data from the QFB and CFISH were used in the analysis of barramundi, but the data sets were analysed separately, because data for additional factors (i.e. fishing effort and stocking of fingerlings) were available and these factors potentially influenced the catches reported in the CFISH data.

Additional factors considered

The number of fingerlings stocked annually into the Fitzroy River system was obtained from the Department of Primary Industries and Fisheries. Stocking of barramundi fingerlings has occurred in the Fitzroy River catchment since February 1990, to enhance recreational fisheries in freshwater impoundments. Barramundi stocked into freshwater impoundments can move downstream to the estuary when floods cause the impoundments to overflow (Australian National Sportfishing Association, Infofish Services, Rockhampton, unpublished data). The total number of fingerlings stocked into impoundments varied between years, ranging from about 1,000 to 78,000 but did not occur in every year (i.e. no stocking in 1991 or 1995). We only included stocking events into impoundments that have overflowed since 1990, as it was only individuals from these impoundments that could have contributed to the commercial catch.

Freshwater flow and other relevant abiotic data

Freshwater flow data were obtained from an Integrated Quantity and Quality Model (IQQM) of the estimated end of system flow maintained by the Department of Natural Resources and Water[5]. IQQM includes rainfall-runoff modelling and balances stream flow (both estimated and gauged) against water extractions to provide an estimate of the quantity of water flowing out of the 'End-of-System' (EOS) i.e. that flowing into the estuary. A single source of freshwater flow estimates was not available from 1945 to 2003. The IQQM data was only available until 1996, which is mid-way through the CFISH time series. Therefore, we used an alternative estimate of freshwater flow to the estuary for the CFISH time series; being gauged stream flow at the most downstream gauging station (i.e. at 'The Gap', 142.1 km Adopted Middle Thread Distance = AMTD), minus the estimated seasonal extractive uses provided by the Department of Natural Resources and Water and Fitzroy River Water. We considered the compatibility of the two methods and given that the differences in the estimated quantities by the two methods are inconsequential in comparison to the size of the flows in the Fitzroy River, we decided that it was better to have a relatively constant measure of the freshwater flowing to the Fitzroy River estuary for each data set. Therefore, we used IQQM flow estimates for the QFB data (i.e. 1945 to 1980) and gauged flow

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[5] The Department of Natural Resources and Water undertakes to provide data that is as accurate, up-to-date and reliable as prevailing conditions at the time of collection and current measuring methods permit, but is not responsible for any inaccuracies, misinterpretation or misuse of the data under any circumstances, and whether arising out of negligence on the part of the NRW or not.

minus extracted use for the CFISH data (i.e. 1985 to 2003). The same arrangement was adopted for freshwater flow data for the Port Curtis region. We used IQQM EOS flow data for the Calliope and Boyne Rivers to provide an estimate of total freshwater flow into Port Curtis for 1945 to 1980 and gauged flow at Castlehope gauging station (Calliope River) and past Awoonga Dam (Boyne River) for flow into Port Curtis for 1985 to 2003.

Selection of freshwater flow variables

The selection of freshwater flow variables was based on the proposed causal mechanisms derived from the life history assessment. Variables were also categorised for their effects on: (i) catchability (i.e. short-term, no lags, coinciding with the seasonality of the fishery); and (ii) recruitment (i.e. survival during the first year of life, beginning at the start of the spawning season for that species), lagged by the average time for a cohort (= year-class) to recruit to the fishery. Lags were one year for banana prawns (indicated as variable^{-1}), three and four years for barramundi (indicated as variable^{-3} or variable^{-4}), and one and two years for mud crabs (indicated as variable^{-1}or variable^{-2}). No variables were categorised for their effects on productivity because of the difficulty in differentiating this effect given that we only had catch data.

Flow variables were aggregated into seasonal totals, where spring = September to November, summer = December to February, autumn = March to May and winter = June to August. For barramundi, we also included ‘spawning season’ which is an aggregate of total freshwater flow in spring and summer i.e. September to February inclusive.

The additional abiotic variable of coastal rainfall was included in analyses because of the uncertainty of the proposed causal mechanisms. In the Fitzroy River estuary, ‘blue sky’ floods (i.e. increased freshwater flows to the estuary without coastal rainfall events) can occur because of the vast size of the catchment. Alternatively, heavy coastal rainfall can result in localised flooding of the estuarine floodplain, without increased freshwater flow occurring in the river. Rainfall variables were aggregated in the same manner as freshwater flow. Rainfall in the Fitzroy region was averaged for the nine coastal rainfall stations that were within 50 km of coast and on the seaward side of the coastal mountain ranges (i.e. that contributing to the estuarine catchment of the Fitzroy River estuary). Rainfall in the Port Curtis region was averaged for the five rainfall stations that occur within the Calliope and Boyne River catchments. Rainfall data were sourced from the historic monthly rainfall provided in *Rainman StreamFlow 4.3* ™ (Clewett *et al.* 2003).

Analysis of data

All data were transformed ($\log_{10}(X+1)$) prior to analysis to normalise the variances. Correlation coefficients were calculated between annual catch and freshwater flow and rainfall variables. Then, all sub-sets general linear models (GLM’s), which identify a number of ‘best’ models (GenStat 2005), were used to more thoroughly explore potential relationships between catch and the freshwater flow and rainfall variables. Correlations and the GLMs were checked for the degree of auto-correlation amongst the residuals and where significant, the degrees of freedom were adjusted to account for serial auto-correlation (Pyper and Peterman 1998). Following Staunton-Smith *et al.* (2004), ridge regressions were investigated. These methods adjust for any collinearity amongst the independent (X) variables. As this was not a notable feature of our data (R^2 between the Xs, averaged across all models, was 0.036), the ridge adjustments had very little effect on either the degree of fit or the coefficients, so the simpler GLMs were retained.

Results

Banana prawns

Fitzroy region

Annual catches of banana prawns fluctuated between 50 and 200 tonnes. Fishing effort explained a significant proportion of this variation for all sectors combined ($r=0.83$, $P<0.01$), the beam-trawl sector (r=0.90, P<0.01) and the otter-trawl sector (r=0.94, P<0.01). After adjusting for effort, summer flow, summer rainfall and spring^{-1} flow were significantly correlated with catch for all sectors combined (**Table 3.1**). Summer flow or rainfall accounted for 33% of the variation in the residuals for the catch and effort relationship (R^2 = 0.33 for the relationship between the standardised residuals and $\log_{10}$ summer flow or rainfall, n=14). Summer flow was highly correlated with summer rain (r=0.84). After adjusting for effort, only spring^{-1} rain was significantly correlated with catch in the beam-trawl sector and no variable was significantly correlated with catch in the otter-trawl sector (**Table 3.1**). Auto-correlation was not a feature of the banana prawn data in the Fitzroy River region, as the residuals of the correlations were not significantly auto-correlated.

Table 3.1 Correlation coefficients (r) between annual banana prawn catch (adjusted for effort) and freshwater flow and rainfall in the Fitzroy Region, based on annual catches reported to CFISH

Effect^{A}	Variable	All sectors combined (1989 to 2002)		Otter-trawl (1989 to 2002)		Beam-trawl (1989 to 2002)	
		Flow	Rain	Flow	Rain	Flow	Rain
R	Spring	0.07	-0.35	0.22	0.16	0.09	0.35
R&C	Summer	**0.58***	**0.58***	0.51	0.41	0.19	0.19
C	Autumn	-0.04	-0.03	-0.42	-0.49	-0.14	-0.35
R	Spring^{-1}	**-0.54***	-0.27	-0.29	0.09	0.35	**0.58***
R	Summer^{-1}	0.48	-0.36	-0.29	-0.37	0.05	-010
R	Autumn^{-1}	-0.15	-0.01	-0.41	0.02	0.19	-0.03

* = P < 0.05. ^{A} Variables are proposed to affect Recruitment (R), Catchability (C). Summer flow and summer rainfall are proposed to affect both Recruitment and Catchability (R&C). Recruitment of banana prawns occurs during spring and summer, whilst the fishery occurs in summer and autumn.

All sub-sets GLMs identified two alternative models that explained around 84% of the variation in the catch of banana prawns for all sectors combined (**Table 3.2**). The models contained summer flow (or rainfall) and spring^{-1} flow and explained around 91% and 84% of the variation in the catch of banana prawns for the otter-trawl and beam-trawl sectors respectively (**Table 3.2**). Summer freshwater flow was significant in models for all sectors combined and the offshore otter-trawl sector, but not for the estuarine-based beam-trawl sector, where spring^{-1} rain was a significant factor in the model. The spatial distribution of the beam-trawl sector compared to the otter-trawl sector adds to the complexity of interpreting these results, as the beam- and otter-trawl sectors target different life stages (i.e. juveniles in the estuary versus adults in offshore waters respectively).

Our results lend some support the theory that summer flows affect the distribution of banana prawns, with higher flows stimulating prawns to move downstream and out of estuaries, thereby increasing the catch of offshore fisheries (Glaister 1978; Vance *et al.* 1985). However, interpretation of our results are complicated by the conflicting direction of impact of spring^{-1} flow and spring^{-1} rain, which had both a negative and positive effect on total catches and beam-trawl catches respectively.

Table 3.2 Best all sub-sets regression models for banana prawn catch and freshwater flow and rainfall components for the Fitzroy region, based on annual catches reported to CFISH

Fishery sector	Regression model [A]	Percent variance accounted for (R^2)	Degree of auto-correlation
All sectors combined	Effort, summer flow, $spring^{-1}$ flow (-)	84.9	-0.033
	Effort, summer rain, $spring^{-1}$ flow (-)	84.3	-0.126
Otter-trawl	Effort, summer flow	90.7	-0.013
Beam-trawl	Effort, $spring^{-1}$ rain	84.3	-0.294

[A] Factors in the multiple regressions are positively related to catch unless otherwise indicated.

Port Curtis region

Fishing effort explained a significant proportion of the variation in banana prawn catch in the otter-trawl sector (r=0.824, P<0.01), which is the only sector to operate in Port Curtis. After adjusting for effort, no freshwater flow or rainfall variable was significantly correlated with catch (**Table 3.3**). Auto-correlation was not a feature of this dataset, as the residuals of the correlations were not significantly auto-correlated. All sub-sets GLM failed to identify any significant models for the variation in banana prawn catches in Port Curtis. This is in contrast to preliminary results of the analysis of banana prawn catch undertaken in 2001 for the experimental design workshop of this project.

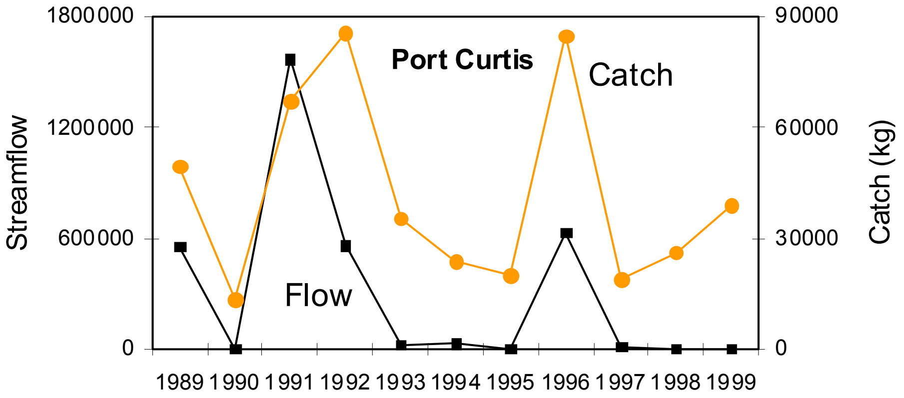
Table 3.3 Correlation coefficients (r) between annual banana prawn catch (adjusted for effort) and freshwater flow and rainfall in the Port Curtis Region, based on annual catches reported to CFISH

Effect[A]	Variable	Otter-trawl (1989 to 2002)	
		Flow	Rain
R	Spring	-0.45	-0.30
R&C	Summer	0.13	-0.03
C	Autumn	-0.38	-0.48
R	$Spring^{-1}$	0.22	0.04
R	$Summer^{-1}$	-0.24	-0.19
R	$Autumn^{-1}$	-0.10	0.03

* = P<0.05. [A] Variables are proposed to affect Recruitment (R), Catchability (C) or both Recruitment and Catchability (R&C), because recruitment of banana prawns occurs during spring and summer, whilst the fishery occurs in summer and autumn.

In the 2001 analysis, the strongest relationship between freshwater flow and CFISH banana prawn catch data occurred in the Port Curtis region. Inspection of the data showed that the years of highest commercial catch in this region were associated with years of increased freshwater flow (**Figure 3.1**). In years of virtually zero flow, there was a relatively low level of otter-trawl catch. The 2001 analysis used otter-trawl catches between 1989 and 1999, seasonal freshwater flow and seasonal rainfall. For Port Curtis catches, summer flow was the most significant variable in the regression, explaining 82% of the variation in catch, and was positively related to catch.

Figure 3.1 Annual banana prawn catch from otter-trawlers and summer freshwater flow for the Port Curtis region from 1989 to 1999



The main difference between the 2001 analysis and that presented in this chapter is the addition of CFISH data for a further three years (i.e. 2000, 2001 and 2002). The lack of model stability with the addition of three data points is concerning and suggests that results for the Port Curtis region, either preliminary or for the full data set, should be considered with caution.

Barramundi

Barramundi has been the consistent marketing name in Queensland for the single species *Lates calcarifer* over the past 70 years. Annual catch of barramundi has fluctuated between four and 40 tonnes between 1945 and 2002 in central Queensland, with a notable 15 to 20 year cycle in the data.

Fitzroy region

For the Queensland Fish Board (QFB) data (i.e. 1945 to 1980), the catch of barramundi in the Fitzroy region was significantly correlated with summer flow and rainfall lagged by three and four years (i.e. summer^{-3} and summer^{-4}) and autumn^{-3} flow (**Table 3.4**). Auto-correlation was a feature of the QFB data, so critical values for the correlation coefficient (r) were based on the adjusted degrees of freedom to account for serial auto-correlation (Pyper and Peterman 1998). For the CFISH data (i.e. 1989 to 2002), fishing effort explained a significant proportion of the variation in catch (r=0.76, P<0.01). After adjusting for effort, summer flow and rainfall in the same year as catch were significantly correlated with barramundi catch, but no lag variables were significant (**Table 3.4**). Auto-correlation was not a feature of the CFISH data, once barramundi catch had been adjusted for effort.

All sub-sets GLM identified a number of alternative models that explained about 38% of the variation in the QFB barramundi catch and about 87% of the variation in the CFISH barramundi catch (Table 3.5). The inclusion of effort in the models of the CFISH barramundi catch data was the main driver for the increased fit of the multiple linear models. Variables in the models from both data sets (i.e. QFB and CFISH) are consistent with the theoretical mechanisms proposed in the life history assessment (see Chapter 2), with summer rain having a positive effect on catchability and summer freshwater flow or rain lagged by three or four years having a positive effect on recruitment (i.e. year-class strength). As expected, the stocking of fingerlings in impoundments that have overflowed appears to have a significant lagged effect on commercial CFISH catches in the estuary as stocking is the fourth term included in both of the alternate best models (**Table 3.5**). The role of autumn^{-3} flow and winter^{-4} flow (QFB models **Table 3.5**) is not obvious from the life history assessment, although positive effects on recruitment are possible, albeit via an unknown mechanism. This might be a Fitzroy specific relationship, as this estuary is at the lower limit of the distribution of barramundi with fish kills being associated with unusually low winter temperatures.

Table 3.4 Correlation coefficients (r) between annual barramundi catch (financial year) and freshwater flow, rainfall and stocking for the Fitzroy region, based on annual catches reported to the Queensland Fish Board and CFISH

Effect[A]	Variable	Queensland Fish Board (1945 to 1980)		CFISH (1989 to 2002)	
		Flow	Rain	Flow	Rain
C	Spring	0.05	-0.10	-0.38	-0.47
C	Summer	0.19	0.36	**0.64***	**0.67***
R	Spring^{-3}	0.31	0.21	-0.11	0.13
R	Spring^{-4}	0.24	0.18	0.25	-0.03
R	Summer^{-3}	**0.49***	**0.46***	-0.39	-0.30
R	Summer^{-4}	**0.44***	**0.44***	0.31	0.25
R	Autumn^{-3}	**0.42***	0.09	0.44	0.05
R	Autumn^{-4}	0.23	0.05	0.05	-0.24
R	Winter^{-3}	0.21		0.33	
R	Winter^{-4}	0.27		-0.07	
R	Stocking^{-3}			0.45	
R	Stocking^{-4}			0.25	

* = P<0.05; Critical values for correlation coefficients (r) for the Queensland Fish Board data were based on the adjusted degrees of freedom to account for serial autocorrelation as per Pyper and Peterman (1998). Auto-correlation was not a feature of the CFISH data, after catch had been adjusted for effort. [A] Variables are proposed to affect Recruitment (R) or Catchability (C).

Table 3.5 Best all sub-sets regression for barramundi catch and freshwater flow and rainfall components for the Fitzroy Region based on annual catches reported to the Queensland Fish Board (QFB, 1945 to 1980) and CFISH (1989 to 2002)

Data source	Regression model [A]	Percent variance accounted for (R^2)	Degree of auto-correlation
QFB	Summer^{-3} flow, summer^{-4} flow, summer rain	38.65	0.216
	Autumn^{-3} flow, summer rain, winter^{-4} flow	38.37	0.125
	Autumn^{-3} flow, summer rain, summer^{-4} rain	38.79	0.157
	Summer^{-3} flow, summer^{-4} rain, summer rain	37.89	0.132
CFISH	Effort, summer rain, summer^{-4} rain, stock^{-4}	88.02	0.091
	Effort, summer rain, summer^{-4} flow, stock^{-4}	86.98	-0.068

[A] Factors in the multiple regressions are positively related to catch unless otherwise indicated.

Port Curtis region

For the QFB data (i.e. 1945 to 1980) catch of barramundi was not significantly correlated with any freshwater flow or rainfall variable (Table 3.6). Auto-correlation was not a feature of the CFISH data, once barramundi catch had been adjusted for effort. For the CFISH data (i.e. 1989 to 2002), fishing effort explained a significant proportion of the variation in catch ($r=0.73$, $P<0.01$). After adjusting for effort, winter flow^{-4}, stocking^{-3} and stocking^{-4} were significantly correlated with barramundi catch in the Port Curtis region (Table 3.6).

Table 3.6 Correlation coefficients (r) between annual barramundi catch (financial year) and freshwater flow, rainfall and stocking for Port Curtis, based on annual catches reported to the Queensland Fish Board and CFISH

Effect^{A}	Variable	Queensland Fish Board (1945 to 1980)		CFISH (1989 to 2002)	
		Flow	Rain	Flow	Rain
C	Spring	0.11	-0.01	-0.46	-0.15
C	Summer	-0.22	-0.01	0.19	-0.17
R	Spring^{-3}	0.19	0.16	-0.04	0.29
R	Spring^{-4}	0.10	0.16	-0.01	0.11
R	Summer^{-3}	0.09	0.04	-0.34	-0.02
R	Summer^{-4}	0.24	0.25	-0.09	0.02
R	Autumn^{-3}	-0.09	-0.16	-0.04	-0.04
R	Autumn^{-4}	-0.09	-0.15	-0.25	-0.43
R	Winter^{-3}	-0.24		-0.35	
R	Winter^{-4}	-0.13		**-0.57***	
R	Stocking^{-3}			**0.66****	
R	Stocking^{-4}			**0.65****	

* = $P<0.05$; ** $P<0.01$; Autocorrelation was not a feature of the CFISH data. ^{A} Variables are proposed to affect Recruitment (R) or Catchability (C).

All sub-sets GLM failed to identify any significant models for the variation in QFB barramundi catch in Port Curtis, but identified a number of alternative models for the variation in CFISH barramundi catch (**Table 3.7**). The inclusion of effort in the regression models of the barramundi catch (i.e. for the CFISH data) was the main driver for the increased fit of the multiple linear models. Results from the regression analysis for Port Curtis do not concur with those from the Fitzroy region, except for the indication that stocking of fingerlings in impoundments that have overflowed appears to have a lagged effect on commercial CFISH catches in the Port Curtis estuary.

Table 3.7 Best all sub-sets regression for barramundi catch and freshwater flow and rainfall components for Port Curtis based on annual catches reported to the Queensland Fish Board and CFISH

Data source	Regression model ^{A}	Percent variance accounted for (R^2)	Degree of auto-correlation
QFB	**No significant models**		
CFISH	Effort, stock^{-4}, autumn flow (-), autumn^{-4} flow (-)	87.65	0.085
	Effort, stock^{-4}, autumn rain (-)	84.08	0.080
	Effort, stock^{-3}, autumn rain (-)	82.83	0.080

^{A} Factors in the multiple regressions are positively related to catch unless otherwise indicated.

We suspect that barramundi of the Fitzroy and Port Curtis region have a large degree of interchange (via the Narrows) and that much of the barramundi landed in Gladstone could have been caught at the mouth of the Fitzroy or in the Narrows. While currently considered separately, evidence is building that barramundi landings currently allocated to Fitzroy and Port Curtis should be pooled into a ‘greater Fitzroy’.

Additional exploratory analyses

During our consideration of the barramundi data, we noted that there was a remarkable similarity in the trends of barramundi catch and freshwater flow for the Fitzroy and Port Curtis region (**Figure 3.2**). This suggested that there were decadal scale trends in fisheries catch that were related to the long-term climatic influences on rainfall and subsequent freshwater flows. In the study area, relatively high freshwater flows occurred in the 1950s and 1970s with associated high catches, while extended periods of low freshwater flow occurred in between these decades (i.e. 60s and 80s) with associated low catches.

We explored the use of running means to examine these long-term trends and to diminish the effects of age-structure in the annual catch data, as per Quiñones and Montes (2001, see Chapter 1). We acknowledge that running means increase the potential for auto-correlation between data points and reduce the effective number of independent data points in a time series (Drinkwater and Myers 1987). Barramundi landings for the financial years 1945 to 1980 were combined with those from 1987 to 1998 (i.e. CFISH) and compared against freshwater flow. Three-year moving-averages of barramundi catch and freshwater flow were calculated. Average barramundi catch was regressed against average freshwater flow three- or four-years prior (**Figure 3.3**). For example, the 1951 catch data point in **Figure 3.3** represents the three-year moving-average of catch for 1951, 1950 and 1949, while the corresponding freshwater flow data point represents the three-year moving-average of freshwater flow in 1947, 1946 and 1945.

For the QFB data, the three-year moving average of barramundi catch in the Fitzroy region was significantly correlated with summer flow and rainfall three or four years previous (i.e. summer^{-3} or summer^{-4}) and autumn^{-3} flow (**Table 3.8**), even after adjusting for serial auto-correlation. No significant correlations were found for the CFISH data, but this may be a function of the relatively short time series in this data set (i.e. two decades).

All sub-sets GLM identified several alternative models that explained between 80 and 90% of variation in the three-year moving average of QFB barramundi catch, and at best, 91% of the variation in the three-year moving average of the CFISH barramundi catch (**Table 3.9**). Effort accounted for 81% of the variation in the three-year moving average of CFISH barramundi catch.

Figure 3.2 Queensland Fish Board barramundi landings in the Fitzroy region and freshwater flow

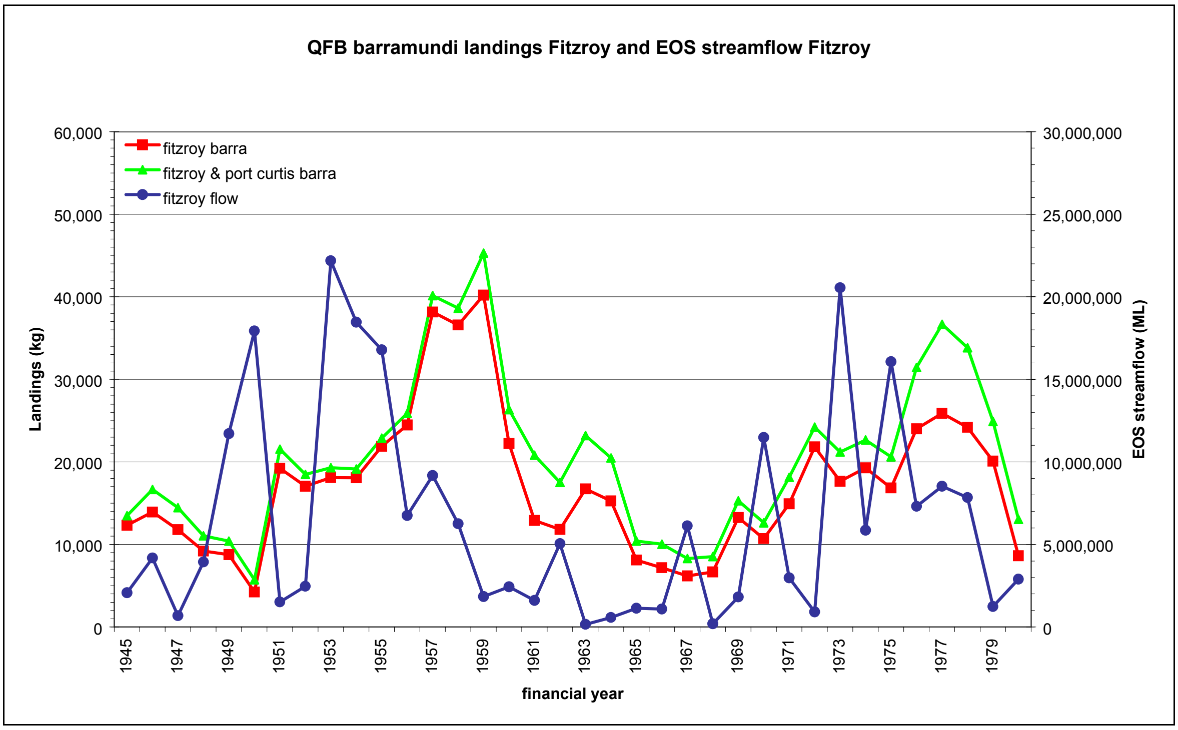


Figure 3.3 Barramundi catch (three-year moving-average) and freshwater flow (three-year moving-average) for the Fitzroy region

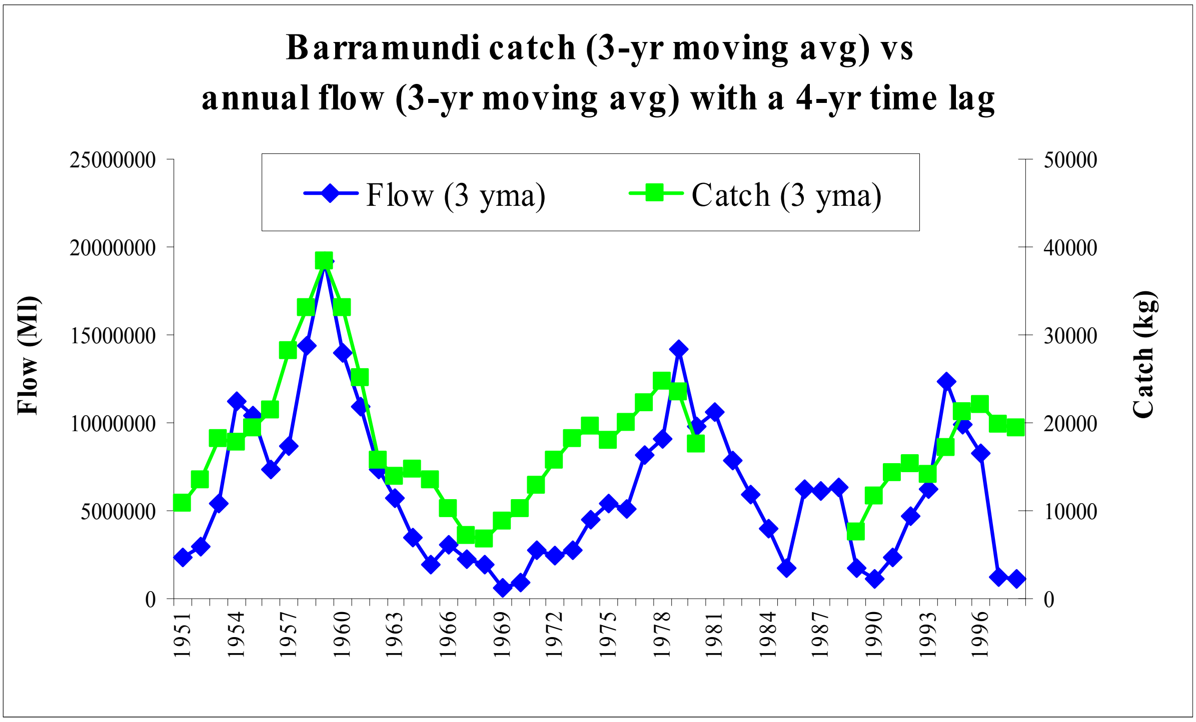


Table 3.8 Correlation coefficients (r) between the three-year moving average of annual barramundi catch (financial year) and the three-year moving average of freshwater flow or rainfall, plus stocking for the Fitzroy region, based on annual catches reported to the Queensland Fish Board and CFISH, the commercial logbook of the Department of Primary Industries and Fisheries

Effect^A	Variable	Queensland Fish Board (1945 to 1980)		CFISH (1989 to 2002)	
		Flow	Rain	Flow	Rain
C	Spring	-0.17	-0.16	-0.34	-0.03
C	Summer	0.27	0.56	-0.08	-0.02
R	Spring^{-3}	0.27	0.32	0.02	-0.04
R	Spring^{-4}	0.27	0.40	0.10	-0.16
R	Summer^{-3}	**0.85****	**0.79****	-0.08	-0.10
R	Summer^{-4}	**0.87****	**0.75****	0.30	0.27
R	Autumn^{-3}	**0.70****	0.31	-0.09	**-0.12****
R	Autumn $^{-4}$	0.57	0.33	0.28	0.01
R	Winter^{-3}	0.41		0.01	
R	Winter^{-4}	0.44		-0.10	
R	Stocking^{-3}			0.42	
R	Stocking^{-4}			0.22	

* = $P<0.05$; ** $P<0.01$; Critical values for correlation coefficients (r) adjusted to account for serial auto-correlation for the QFB data, but not for the CFISH data. ^{A} Variables are proposed to affect Recruitment (R) or Catchability (C).

Table 3.9 Best all sub-sets regression for the three-year moving average of annual barramundi catch (financial year) and the three-year moving average of freshwater flow, rainfall, plus stocking for the Fitzroy region, based on annual catches reported to the Queensland Fish Board and CFISH, the commercial logbook of the Department of Primary Industries and Fisheries

Data source	Regression model[A]	Percent variance accounted for (R^2)	Degree of auto-correlation
QFB	Spring flow (-), summer^{-3} flow, summer^{-4} flow	87.48	0.389*
	Spring flow (-), summer^{-3} flow, autumn^{-3} flow	86.15	0.143
	Spring flow (-), summer^{-4} flow, autumn^{-3} flow	83.77	0.253$^{\#}$
	Spring flow (-), summer^{-3} flow, autumn^{-4} flow	83.03	0.195
	Summer^{-4} flow, summer^{-3} flow, autumn^{-3} flow	82.28	0.462*
	Summer^{-4} flow, summer^{-3} flow, winter^{-4} flow	82.07	0.448*
	Spring flow, summer^{-3} flow, summer^{-4} rain	88.75	0.274$^{\#}$
	Spring flow, summer^{-4} flow, summer^{-3} rain	87.15	0.369*
	Summer^{-4} flow, summer^{-3} rain, winter^{-4} flow	86.26	0.307$^{\#}$
	Summer^{-4} flow, summer^{-3} rain, winter^{-3} flow	84.76	0.399*
	Spring flow, summer^{-4} flow, summer rain	83.49	0.290$^{\#}$
	Summer^{-3} flow, summer^{-4} rain, winter^{-4} flow	83.22	0.407*
	Summer^{-4} flow, summer^{-3} rain, autumn^{-3} flow	82.94	0.440*
CFISH	Effort, stock^{-4}, summer^{-3} rain (-), winter^{-4} flow	91.47	-0.086
	Effort, stock^{-4}, summer flow (-), winter^{-3} flow	80.81	-0.251$^{\#}$
	Effort, stock^{-4}, summer^{-3} flow (-), winter^{-3} flow	80.41	-0.018
	Effort, summer^{-3} rain (-), winter^{-4} flow, summer flow	77.80	-0.016

* = P<0.05, ** P<0.01. # = close to being significant. [A] Factors in the multiple regressions are positively related to catch unless otherwise indicated.

In summary

Results suggest that summer flow and or rainfall influences the catch of barramundi immediately and in subsequent years (i.e. lagged by three or four years) and, at least in the Fitzroy River estuary, these relationships are consistent over time (i.e. 1945 to 2002), despite several events in study area that may have caused major system changes. These include: (i) a major change to the hydrology of the Fitzroy River in 1991, when a 10 km loop of the river was eliminated during a 1-in-100 year flood; (ii) the stocking of barramundi fingerlings into various freshwater habitats in the catchment from 1992, although stocking is included in the analyses; (iii) upgrading of the fishway on the barrage in 1994 to enable migration of fish (Stuart and Mallen-Cooper 1999); and (iv) various changes to fisheries management and marketing arrangements, that may have altered patterns of fishing.

Mud crabs

Fitzroy region

Reported catches of mud crabs in the Fitzroy region have dramatically increased from around five tonnes in 1960 to ~100 tonnes in 2002. We speculate that this increase is probably a consequence of increased fishing effort in the region and more accurate reporting of catch data. Auto-correlation was not a feature of the QFB mud crab catch, once catch had been adjusted for year ($r=0.75$, $P<0.001$), as per Loneragan and Bunn (1999). The QFB catch of mud crabs adjusted for year was significantly and negatively correlated with autumn flow and summer rain, but was significantly and positively correlated with autumn rain lagged by two years (i.e. autumn^{-2} rain, **Table 3.10**). Fishing effort explained most of the variation in CFISH mud crab catches ($r=0.97$, $P<0.01$). After adjusting for effort, autumn flow lagged by two years (i.e. autumn^{-2} flow) was the only variable significantly correlated with mud crab catch (**Table 3.10**).

Table 3.10 Correlation coefficients (r) between annual mud crab catch and freshwater flow and rainfall in the Fitzroy region based on annual catches reported to the Queensland Fish Board and CFISH, the commercial logbook of the Department of Primary Industries and Fisheries

Effect ^{A}	Variable	Queensland Fish Board (1960 to 1980)		CFISH (1988 to 2002)	
		Flow	Rain	Flow	Rain
C	Spring	-0.15	-0.40	0.30	0.19
C	Summer	-0.38	**-0.46***	0.38	0.11
C	Autumn	**-0.63****	-0.10	-0.35	-0.36
R	Spring^{-1}	-0.32	-0.32	0.07	0.01
R	Spring^{-2}	0.36	0.17	0.47	0.26
R	Summer^{-1}	-0.07	0.05	0.02	-0.21
R	Summer^{-2}	0.08	-0.25	0.20	0.19
R	Autumn^{-1}	0.18	0.23	0.05	0.14
R	Autumn $^{-2}$	0.14	**0.46***	**0.70****	0.23

*= $P<0.05$, ** $P<0.01$; Auto-correlation was not a feature of the Queensland Fish Board data, after catch had been adjusted for year, or the CFISH data, after catch had been adjusted for effort. ^{A}Variables are proposed to affect Recruitment (R) or Catchability (C).

All sub-set GLM identified a number of alternative models that explained between 70% and 97% of the variation in mud crab catch for the QFB and CFISH data respectively (**Table 3.11**). Whilst multiple term models could be fitted, they only had a slight improvement on the 'simpler' models of year and autumn flow (QFB data, adjusted R^2=70.91) or effort and autumn^{-2} flow (CFISH data, adjusted R^2=96.70).

Table 3.11 Best all sub-sets regression model of mud crab catch and freshwater flow and rainfall components for the Fitzroy region, based on annual catches reported to the Queensland Fish Board (QFB) and CFISH, the commercial logbook of the Department of Primary Industries and Fisheries

Data source	Regression model ^{A}	Percent variance accounted for (R^2)	Degree of auto-correlation
QFB	Year, autumn flow	70.91	0.191
	Year, spring rain (-), spring^{-1} rain (-), spring^{-2} flow	77.13	-0.074
	Year, spring^{-1} rain (-), spring rain (-), autumn^{-1} flow	77.00	-0.019
	Year, spring rain (-), summer rain (-), spring^{-2} flow	73.71	-0.132
CFISH	Effort, autumn^{-2} flow	96.70	0.163
	Effort, autumn rain (-), spring^{-2} rain, spring rain	96.72	0.162

^{A} Factors in the multiple regressions are positively related to catch unless otherwise indicated.

Port Curtis region

Reported catches of mud crabs in the Port Curtis region have dramatically increased from around 17 tonnes in 1960 to ~143 tonnes in 2002, which is also probably a consequence of increased fishing effort and more accurate reporting of catch data.

QFB mud crab catch was not significantly correlated with year (r=-0.20), and auto-correlation was not a feature of this data. The QFB catch of mud crabs was significantly and negatively correlated with summer flow, autumn flow, and spring rain, as well as summer flow lagged by one and two years, autumn flow lagged by one year, spring rain lagged by two years and summer rain lagged by one year (**Table 3.12**).

Fishing effort explained 51% of the variation in CFISH mud crab catches ($P<0.01$). After adjusting for effort, only spring flow and spring rain lagged by two years (i.e. spring^{-2} flow and spring rain^{-2}) were significantly correlated with the CFISH mud crab catch in Port Curtis (**Table 3.12**). All sub-sets GLM identified several alternate models that explained between 60 and 65% of the variation in mud crab catch for the QFB data and at best 71% of variation for the CFISH data respectively (**Table 3.13**).

Table 3.12 Correlation coefficients (r) between annual mud crab catch and freshwater flow and rainfall in the Port Curtis region based on annual catches reported to the Queensland Fish Board and CFISH, the commercial logbook of the Department of Primary Industries and Fisheries

Effect [A]	Variable	Queensland Fish Board (1960 to 1980)		CFISH (1988 to 2002)	
		Flow	Rain	Flow	Rain
C	Spring	-0.27	**-0.44***	-0.43	-0.03
C	Summer	**-0.59****	-0.11	0.24	0.09
C	Autumn	**-0.65****	-0.07	-0.10	-0.08
R	Spring^{-1}	-0.15	-0.42	0.01	0.23
R	Spring^{-2}	-0.29	**-0.59****	**0.56***	**0.66****
R	Summer^{-1}	**-0.62****	**-0.52****	-0.38	-0.24
R	Summer^{-2}	-0.46*	-0.42	-0.18	-0.07
R	Autumn^{-1}	**-0.64****	-0.03	-0.46	-0.37
R	Autumn^{-2}	-0.01	0.41	-0.28	-0.26

*= P<0.05, ** P<0.01; Autocorrelation was not a feature of the Queensland Fish Board data, after catch had been adjusted for year, or the CFISH data, after catch had been adjusted for effort. [A] Variables are proposed to affect Recruitment (R) or Catchability (C).

Table 3.13 Best all subsets regression model of mud crab catch and freshwater flow and rainfall components for the Port Curtis region, based on annual catches reported to the Queensland Fish Board (QFB) and CFISH, the commercial logbook of the Department of Primary Industries and Fisheries

Data source	Regression model [A]	Percent variance accounted for (R^2)	Degree of auto-correlation
QFB	Summer^{-1} rain (-), summer^{-2} flow (-), spring rain (-)	65.95	0.108
	Summer^{-1} rain (-), summer^{-2} flow (-), autumn^{-2} rain	65.84	0.187
	Autumn flow (-), autumn rain, summer^{-2} rain (-)	62.20	0.428$^{\#}$
	Spring^{-2} rain (-), autumn^{-2} rain, summer^{-2} flow (-)	60.60	0.474*
CFISH	Effort, spring^{-2} rain	71.13	0.251
	Effort, spring^{-2} flow	63.50	0.332

[A] Factors in the multiple regressions are positively related to catch unless otherwise indicated.

Discussion

Implications of correlation analyses

Banana prawns

The results of the correlation analyses for the central Queensland area concur with other studies of banana prawns, in that catches, particularly in offshore otter-trawl fisheries are related to summer freshwater flows. The total catch of banana prawns increases in proportion to summer freshwater flow. The temporal and spatial aspects of the correlation support the proposed hypothesis that freshwater flow affects their catchability and possibly recruitment. Further within-year analyses of commercial catch data may provide stronger evidence of recruitment effects. Browder *et al.* (2002) recommended further research into the seasonal and annual patterns of the availability of post-larval prawns to provide greater evidence that freshwater flows increase the abundance or biomass of prawns as a consequence of improved growth and survival through trophic cascading. It is unlikely that correlative analyses would provide evidence of such productivity related effects. Therefore, further research into the freshwater flow requirements of banana prawn fisheries may need to use empirical data collected at a scale different to that of simply the quantity of commercial catch (see Chapter 9). Whether such data are fishery-dependent (i.e. size-class structure and length-frequency analysis) or fishery-independent (i.e. relative densities of post-larval and juvenile banana prawns) will depend on the theoretical mechanism (of the role of freshwater flow) investigated. Indeed, in further investigations of the effects of freshwater flow on banana prawns, we found strong evidence that growth rates of juvenile banana prawns were significantly and positively related to freshwater flow and that this potentially resulted in a three-fold increase in the biomass of juvenile banana prawns. This also coincided with a greater abundance of juvenile prawns in the estuary and increased commercial catches (see Chapter 9).

Barramundi

We cannot attribute causality to the observed relationships or to changes in the relationships between catch and freshwater flow because our assessment is based on correlation. However, the within-year influence of freshwater flow on barramundi catches supports the proposed hypotheses for the effect of flow on catchability and concurs with anecdotal reports by commercial fishers. Indeed, during sampling of landings at commercial seafood processors (for the age-structure work presented in Chapter 6), we observed a major difference in the number, size and age-structure of barramundi caught before and immediately after a moderate flood in the Fitzroy River in February 2003, which inundated coastal floodplains and released many land-locked barramundi that became available to the estuarine commercial fishery (see Chapter 8). The significant correlations between catch and freshwater flow or rainfall lagged by three or four years in Queensland Fish Board and CFISH data provides consistent quantitative evidence in support of the positive influence of freshwater flow on the recruitment for barramundi, as suggested by Dunstan (1959) and Williams (2002).

In the first year of life, post-larval barramundi exploit swamps, coastal lagoons and other supra-littoral habitats, which are thought to offer sheltered, highly productive nursery habitats that allow rapid growth and enhanced survival (Davis 1985; Russell and Garrett 1983, 1985). Access to these habitats is via tidal inundation and freshwater runoff from localised rainfall. Life history assessment (see Chapter 2) and some of the correlative analyses reported in this chapter (but see also Chapter 4), support previous assertions that barramundi in tropical Australian estuaries have freshwater flow requirements. However, barramundi populations are adaptable to the prevailing conditions, with the life-cycle varying across the species' distribution. For example, diadromy is facultative (Russell 1990; Pender and Griffin 1996) and cannibalism affects recruitment rates in some areas (Walters *et al.* 2001). Further research is needed to clarify how recruitment and productivity are enhanced as a consequence of increased freshwater flow, in particular the timing of these flows, their magnitude, the mechanism by which these effects occur (including the habitats affected) and whether the relationship is linear or threshold.

Outcomes of such research will need to be adapted for each estuary in tropical Australia to account for the specific hydro-geographic conditions and life-cycle adaptations. Our current findings led us to investigate the influence of freshwater flow on the year-class strength of barramundi in the Fitzroy River estuary (see Chapter 6), the influence of flow on the growth rates of barramundi using tag-recapture data (see Chapter 7), and otolith micro-chemistry (see Chapter 8). Results from an age-based index of year-class strength confirmed that recruitment was strongly correlated to freshwater flow in summer and spring (published as Staunton-Smith *et al.* 2004). In addition, analysis of tag-recapture information from the long-term ANSA tagging program supported our hypotheses that growth is strongly influenced by freshwater flow (published as Robins *et al.* 2006) and that growth rates are enhanced in fish that accessed freshwater habitats as juveniles (Milton *et al.* submitted).

Mud crabs

The observed correlations between catch (adjusted for year or effort, depending on the data set) provide ambiguous albeit preliminary results as to the effects of freshwater flow on mud crabs. This may be a consequence of the reported catch data not being an appropriate index of the abundance of mud crabs in central Queensland estuaries, particularly between 1960 and 1980 (i.e. the Queensland Fish Board data). From the CFISH data, there is evidence that freshwater flows in autumn may affect the recruitment (i.e. cohort or year-class) strength of mud crabs, which concurs with the suggestion of recruitment effects by Loneragan and Bunn (1999). However, the biological link between autumn flows and the life history of mud crabs requires further investigation, perhaps into the hypothesis that recruitment effects are related to burrow competition.

Productivity effects

It is difficult from the correlative analysis of annual mud crab catch data to identify productivity effects on mud crab populations that might result from increases in primary or secondary production occurring as a consequence of freshwater flows (Drinkwater and Frank 1994). Productivity effects (e.g. growth) are not independent of recruitment effects (e.g. survival), and occur at similar temporal scales. Investigation of trophic linkages hypothesised to occur as a consequence of nutrient input associated with freshwater inflow requires data for several levels of the estuarine food web, collected at appropriate spatial and temporal scales (e.g. Livingston *et al.* 1997; Kimmerer 2002a). As an example, Kimmerer (2002a) was able to conclude that variation in the abundance of organisms at higher trophic levels was more likely to be a consequence of changes in the physical habitat associated with freshwater flow, than with upward trophic transfer. An alternative to such detailed studies, maybe to correlate growth rates of individual species with freshwater flow. Our preliminary investigations of growth rates in the Fitzroy River estuary suggest a significant positive relationship between growth rates and freshwater flow to the estuary (see Chapter 7; Sawynok 1998; Robins *et al.* 2006) for barramundi and for banana prawns (see Chapter 9). These results provide some of the first quantitative evidence in support of the productivity hypothesis, for at least two major fishery species that are important within tropical and sub-tropical estuaries of northern Australia.

Freshwater flow requirements of estuarine fisheries in northern Australia

The relationships between estuarine fisheries production and freshwater flow reported in this chapter provide quantitative evidence that the productivity of estuarine fisheries in northern Australia is related to freshwater flow. Determining whether these relationships are consistent in other tropical estuaries is an issue requiring further research. Correlative analyses would be a starting point, as per the conceptual model presented in Chapter 2, but improved knowledge of the role of freshwater flow in tropical Australian estuaries would assist in determining how freshwater flows effects the catchability, recruitment or productivity of estuarine fisheries species.

It is uncertain whether other tropical estuarine species have similar freshwater flow requirements to that of banana prawns, barramundi or mud crabs. However, we suggest that for most fishery species, freshwater flow requirements could be considered in terms of the effects on catchability, recruitment (i.e. cohort or year-class strength) and productivity.

Catchability is likely to be related to the seasonality of the fishery, and therefore freshwater flows preceding or occurring during the main fishing season may have the most influential effect. We have grouped freshwater flow effects on recruitment as those occurring during the first year of life for species that live longer than one year, or its equivalent in shorter-lived species (e.g. banana prawns). There may be several critical times during this period when the survival of a species is enhanced by freshwater flow. However, parts of Australia have a highly variable climate, and consequentially, a variable flow regime. It may be that estuarine fishery species are able to take advantage of windows of opportunity when good conditions for recruitment occur. For a discussion as it pertains to barramundi see Davis (1985), Russell and Garrett (1985) and Griffin (1987). We speculate that freshwater flows that enhance recruitment may not be temporally limited, outside the need for such flows to coincide approximately with the spawning season, which is extended in many tropical species.

Effects of increased freshwater flow on estuarine productivity are likely to be observed as increased growth rates, which theoretically leads to greater survival of individuals and or increased biomass of the population. Another area requiring research is determining the mechanisms by which estuarine productivity is enhanced through increased freshwater flow, and how (and whether) this productivity is progressed up the food chain.

Knowledge of the freshwater flow requirements of estuarine fisheries has increased substantially since the issue was raised in the scientific literature in the 1950s, 1960s and 1970s. Although estuarine issues are being increasingly identified as important in the documentation associated with water planning and development, lack of knowledge and or availability of suitable methods are hampering the ability of managers to effectively include the needs of estuarine fisheries. Coalescence of available knowledge supported by further research and analysis is needed to better quantify the required freshwater flows and any impacts of altering natural flow regimes.

Chapter 4. Climate impacts on barramundi and banana prawn fisheries of Queensland tropical east coast

J. Balston

Summary

The commercial catch of barramundi (*Lates calcarifer*) and the banana prawn (*Penaeus merguiensis*) displays a high degree of inter-annual variation; a characteristic that many fishers believe is the result of climate variability. However, studies to examine the effects of the full range of climate variables are limited and the impact of some variables, such as evaporation, have never been considered. This chapter examined the effects of short-term (inter-annual) climate variability on the commercial catch of wild barramundi and banana prawn in Princess Charlotte Bay of north-east Queensland.

A life-cycle model of each species was developed to link climate variables with the developmental stages of the species from spawning through to catch. Fisheries (catch and effort) and climate (rainfall, freshwater flow, evaporation, sea surface temperature, terrestrial temperature and indices of the SOI and MJO) data were extracted from a variety of sources and compiled for analysis. A gamma-distributed logarithm-link function model was constructed to calculate total freshwater flow for those years when flow records were incomplete. Correlation analysis was used to identify significant relationships between climate variables and catch, and forward stepwise ridge regression was used to model catch using climate variables as predictors.

Warm sea surface temperatures in January – March, high rainfall in July – September and January - March, increased freshwater flow in January – March and low annual evaporation (all measures of an extensive and productive nursery habitat) were all significantly and positively correlated with barramundi catch two years later as recorded by the CFISH logbook system. These results suggest that early barramundi survival is enhanced in these conditions. Catchability of barramundi was significantly increased after high freshwater flow and rainfall from October – December in the year of catch, reinforcing the observation that mature fish in freshwater habitats are flushed into the commercial estuarine fishery. A forward stepwise ridge regression model built to predict commercial barramundi catch in Princess Charlotte Bay contained July – September rainfall, annual evaporation and January – March sea surface temperatures two years prior to catch and explained 63% of the variance in catch. The model had a cross validated predictive R^2 of 49%. A comparative analysis of data from the anthropogenically modified Fitzroy River area returned far fewer significant relationships, a result that indicated changes to habitat either affected or masked the relationship between other climate variables and barramundi catch in the area.

A correlation analysis between seasonal and annual climate variables and banana prawn catch in the Princess Charlotte Bay area in the year of catch did not return any significant relationships. Other studies of rainfall and banana prawn catch gave significant correlations with catch data from beam trawlers only. Results with catch data from otter trawlers were not significant – the primary method of harvest in Princess Charlotte Bay. Results from the current chapter indicate that a significant proportion of the variability seen in commercial barramundi catch in north-east Queensland is driven by variability in climate.

Evaluation of approach and methods

The objective of this section was to raise the awareness of natural climate variability and its impact on freshwater flows, dependant fisheries and water allocation management through the delivery of outputs relating to the CZCRC AF project research question (ii) i.e., What are the impacts of natural climate variability upon estuarine fisheries and therefore environmental flow allocations to sustain healthy estuarine communities? An initial analysis of the variability of rainfall and freshwater flow was undertaken using historical data from the Australian Rainman StreamFlow 4.3 TM package (Clewett et al. 2003). Rainfall across Queensland is highly variable as seen from the historical records for each of the two areas considered in this section of the study: the Fitzroy region of central Queensland and Princess Charlotte Bay (PCB) in the far north of the state (Figure 4.1). This variability is amplified in freshwater flows which are in some cases an order of magnitude higher than recorded rainfall for the catchment (Figure 4.2). Because of this, both inter-annual and inter-decadal climate variability needs to be taken into account when analysing data from a short time series in order to contextualise the results into a historical setting. In the rainfall graph (Figure 4.1), the study period considered in this analysis is overlayed onto the historical time series.

Figure 4.1 Annual rainfall variability at Yaamba station on the Fitzroy River and Musgrave station in Princess Charlotte Bay (1900-2004)

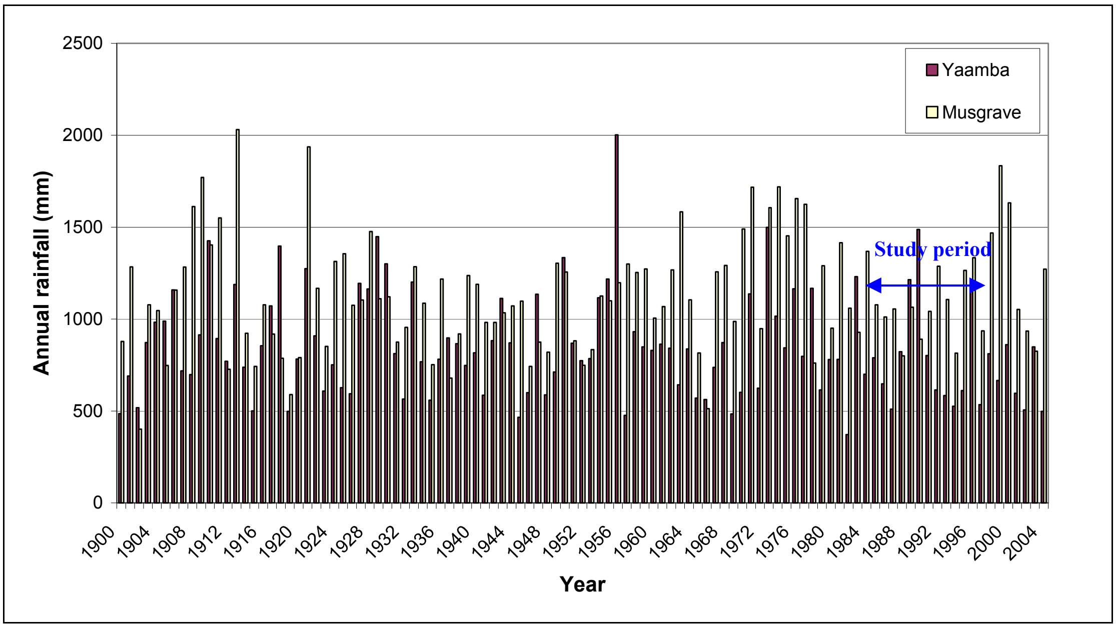
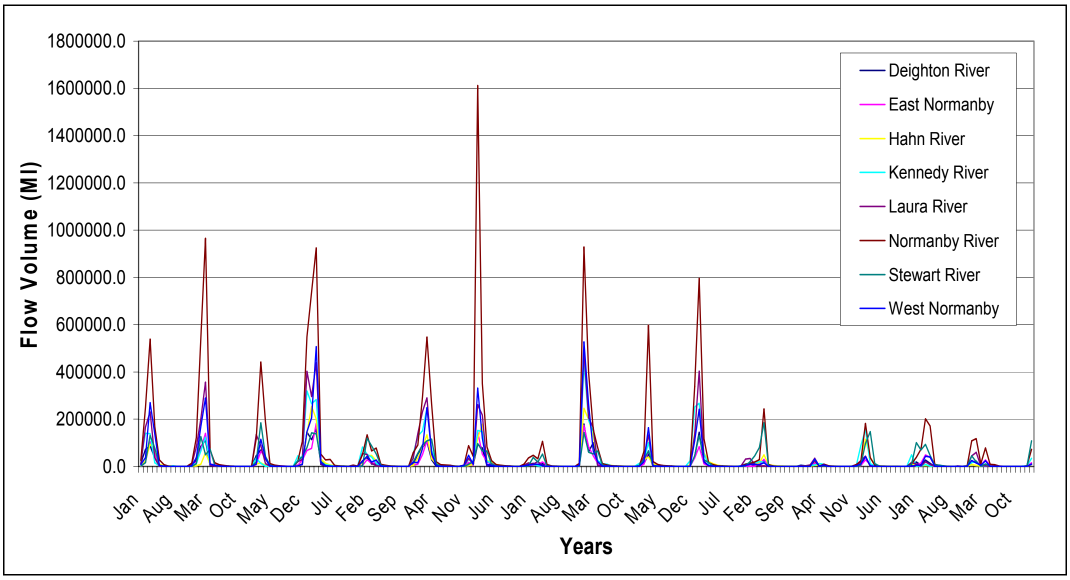


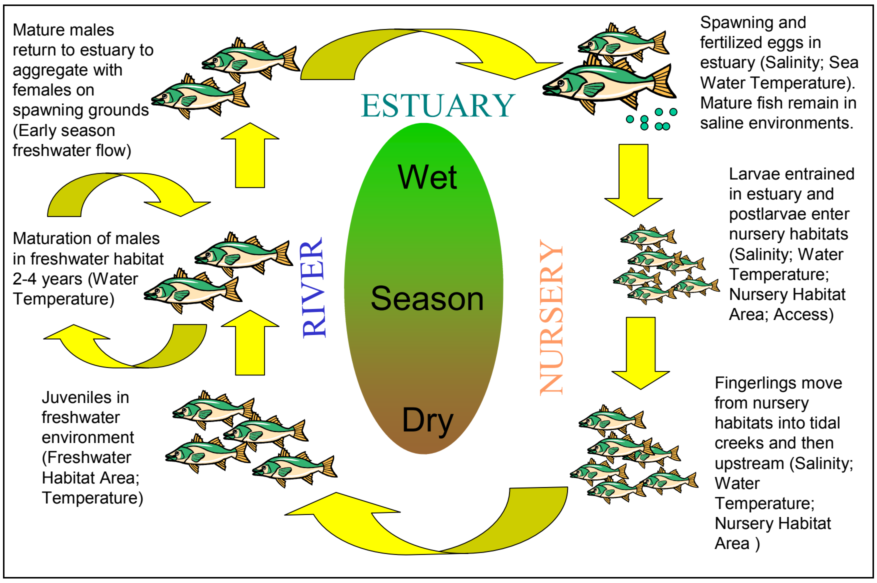
Figure 4.2 River flows entering Princess Charlotte Bay River between 1971-1987



A literature review of published papers and other reports was undertaken to determine climatic factors which might impact on each of the species studied: barramundi (*Lates calcarifer*) and banana prawn (*Penaeus merguiensis*). Based on this review, and as recommended by Vance *et al.*

(1985), the conceptual life history model (Figure 4.3) was extended to include climate influences known to affect the species with the timing of the life-cycle stages. Relevant climate variables, climate indices (Madden Julian Oscillation and Southern Oscillation Index), freshwater flow and fisheries catch data were collated for each region. Correlation and multiple linear regression analysis were then undertaken to provide an assessment of the predictive capacity of seasonal climate variables and forecasting indices in forecasting the productivity of each fishery.

Figure 4.3 Life-cycle model of the barramundi showing links to climate



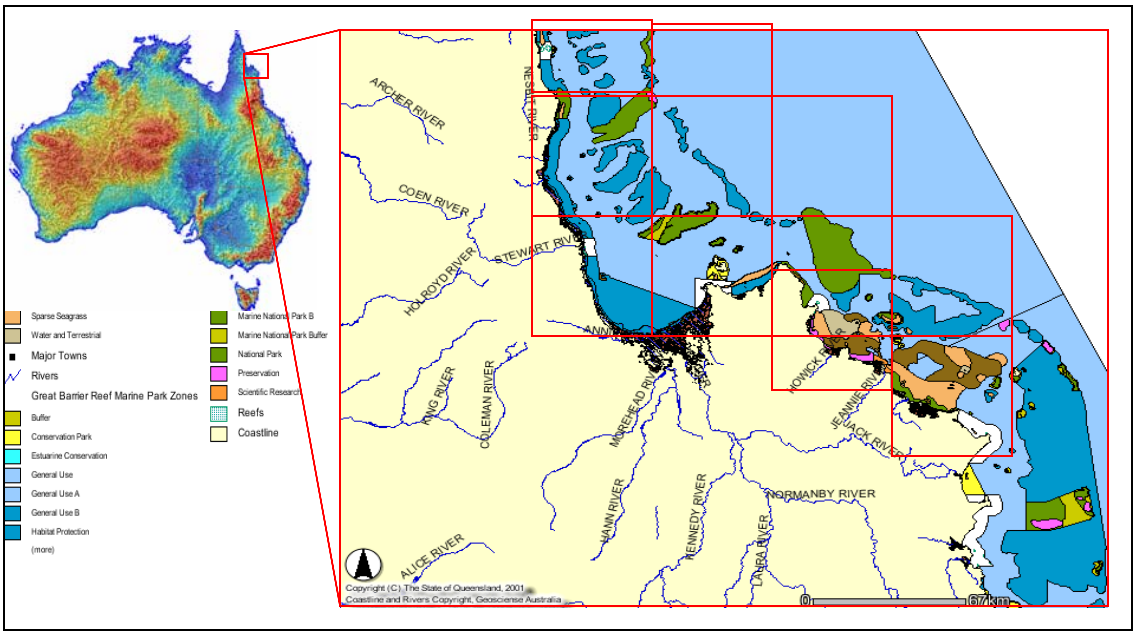
The Princess Charlotte Bay (PCB) region was selected as a comparison to the Fitzroy River region for a number of reasons. Lying about 340 km north of Cairns on the east coast of Cape York Peninsula (Figure 4.4), PCB has a climate typical of a tropical, monsoonal area and so varies from that of central Queensland which is sub-tropical. Original vegetation and ecosystems have remained relatively undisturbed and the catchment as a whole is classified as near pristine/relatively undisturbed. There are then, very few of the anthropogenic impacts seen in the Fitzroy River region and other areas along the east coast of Queensland such as habitat degradation, impediments to fresh water flow, stocking of fish fingerlings and urban/agricultural development. As a result, 'noise' from man-induced factors that may impact on the fishery is minimal. In addition, the barramundi populations in PCB are part of a discrete genetic strain (Shaklee *et al.* 1993) and so may be assumed to be homogeneous in biological aspects.

Variables

Fisheries data

Monthly catch and effort data recorded as part of the mandatory Commercial Logbook Data (CFISH) program of the Department of Primary Industries and Fisheries were sourced for grid squares from the northern edge of PCB to Cape Flattery, and for the Fitzroy River outflow area for all available years, batched by financial year and summed to give a total annual financial year catch for each study area. Data for grid squares inland from the coast represent fish caught in rivers and were included in the analysis where appropriate. Freshwater impoundments on the Fitzroy River catchment have been stocked with fingerlings since 1990 for recreational fishers. The impact of stocking fingerlings on commercial catch was considered by Staunton-Smith *et al.* (2004) and not found to be significant. For this reason stocking was not included in this analysis.

Figure 4.4 Princess Charlotte Bay region showing river systems, estuarine and near shore habitats, adjacent coral reefs and CFISH grid squares



Rainfall

For PCB splined rainfall data were sourced from the Bureau of Meteorology (BOM) spatially interpolated rainfall and climate database (SILO). A detailed explanation of how the rainfall surface was created can be found at the SILO website (http://www.bom.gov.au/silo). Total monthly rainfall for five locations in PCB corresponding to Annie River (14° 30'S; 143° 42'E), Port Stewart (14° 06'S; 143° 42'E), Normanby River (14° 24'S; 144° 12'E), Aloszville station (14° 24'S; 144° 00'E) and Lakefield (14° 57'S; 144° 12'E) were extracted from the database and used to calculate an average monthly local area rainfall data set. For the Fitzroy region, local area average total seasonal rainfall data was calculated from data extracted from Rainman StreamFlow 4.3 (Clewett *et al.* 2003) for stations within the coastal region of the Fitzroy River estuary (i.e. Bajool, Mt Morgan, Mt Larcom, Rockhampton, Port Alma, Langmorn, Raglan, Stanwell and Gracemere) as per Staunton-Smith *et al.* (2004).

Freshwater flows

Monthly freshwater flow data for each region were collected from the Department of Natural Resources and Water (NRW) stream gauge website database (http://www.nrm.qld.gov.au/watershed/index.html). This included for PCB total monthly flow for all rivers in the basin with gauges (eight in total) and for the Fitzroy region the Gap station (the most downstream gauging station) minus the estimated downstream extraction. For PCB, monthly flow from each river was summed in order to calculate total monthly flow into the bay, and missing data identified. Recent years were missing for many of the gauge stations and so total basin flow for years corresponding to CFISH data was modelled using the full set of data for the period January 1971 to February 1987 as the base period (a time series of 133 consecutive months with no missing values). A gamma distributed logarithm link function model which included each month as a variable was developed (i.e. the model changes depending on the month being calculated). As the Normanby, East Normanby and Laura rivers each showed high correlations with total basin flow and have recent data, they were selected for the generation of total flow for the years with missing data in other rivers.

Temperature

Maximum and minimum air temperature and evaporation were sourced from the BOM SILO database (http://www.bom.gov.au/silo) for Lakefield National Park in PCB (14° 57’S; 144° 12’E; 40 m) and for two grid points aligning with the mouth of the Fitzroy river (23 ° 30’S; 150 ° 48’E) and Keppel Bay (near Broadmount/Port Alma, 23° 30’S 150° 48’E) in the Fitzroy study area as representative points. Monthly and seasonal averages, and annual total temperature degree days (sum of daily temperatures) were calculated for both maximum and minimum air temperature.

Sea surface temperatures

As there are no in situ recordings for sea surface temperatures in PCB or the Fitzroy River region, data were sourced via the web from the Physical Oceanography Distributed Active Archive Center (PO.DAAC), generated by the National Aeronautics and Space Administration (NASA) Jet Propulsion Laboratory at a resolution of 1°C, and is accurate to within 0.5°C (http://podaac.jpl.nasa.gov/products/product119.html). Monthly averaged data for PCB were extracted for the point 14°S; 114°E in PCB and the point 23°S; 151°E for the Fitzroy River region. Monthly and seasonal average and annual total temperature degree days were calculated for analysis.

Southern Oscillation Index (SOI)

Monthly average values of Troup’s SOI (Troup 1965) were extracted from the Department of Natural Resources and Water ‘LongPaddock’ website which uses a base period from 1887 to 1989 (www.longpaddock.qld.gov.au). The index is derived from normalised Tahiti minus Darwin mean sea level pressure.

Madden Julian Oscillation (MJO)

All Season Real Time Multi-Variate MJO Indices (RMM) were sourced from the Bureau of Meteorology Research Centre website (http://www.apsru.gov.au/mjo/) and include both phase (as defined by a longitudinal position of the centre of the oscillation) and the number of days in each phase. The variable used in the analysis was a count of the number of days for each phase over the northern wet season (defined here as 1 November - 30 April). This corresponds to the time of the year when the MJO has the strongest influence on the region.

Wind

Mean monthly V-wind (meridional – north/south) and U-wind (zonal – east/west) data were extracted for the point 14°S; 114°E for PCB from the NCEP/NCAR Reanalysis Data: Derived Products data request web page at the NOAA Climate Diagnostics Centre (http://www.cdc.noaa.gov/cdc/data.ncep.reanalysis.derived.html#surface). Data is at sea level for a 2.5-degree latitude x 2.5-degree longitude grid. Wind data were used only for analysis with the banana prawn data and were averaged for each spawning season: March to May for the autumn spawning; August to October for the spring spawning. Annual total wind run (summed daily wind vector ms^{-1}) was also calculated for each vector.

Selected data were checked for normality using histograms and the Shapiro-Wilks test and transformed where necessary. In order to capture the monsoonal climate cycles, the seasons were defined as per Vance *et al.* (1998): pre-wet (October to December); wet (January to March); early dry (April to June) and dry (July to September). This varies from the seasons defined by Robins *et al.* (2005), and was selected in order to capture the northern monsoon wet in one season. Annual data, where used, was for the financial year (1st July to 30th June). A summary of data sets collated for analysis is shown in Table 4.1.

Table 4.1 Summary of climate and catch data collated for analysis in each region

Fitzroy region	N	Mean	Minimum	Maximum	Std.Dev.
Annual barramundi catch adjusted for effort	15	-0	-7689	10815	4976.80
Flow Jul-Sep (Ml)	20	257649	1220	3112225	704239.19
Flow Oct-Dec (Ml)	20	426839	0	2012083	525863.85
Flow Jan-Mar (Ml)	20	2347482	167536	21666046	4731624.77
Flow Apr-Jun (Ml)	20	634619	1945	5309242	1469671.02
Maximum temp December (°C)	20	30	28	31	1.02
Minimum temp July (°C)	20	11	9	14	1.36
Evaporation Annual (mm)	20	1903	1743	2229	117.70
Rain Jul-Sept (mm)	20	87	9	252	67.28
Rain Oct-Dec (mm)	20	238	95	512	99.72
Rain Jan-Mar (mm)	20	278	127	540	114.69
Rain Apr-Jun (mm)	20	123	29	351	86.24
MJO Phase 4	20	23	6	43	8.13
MJO Phase 5	20	22	6	43	9.51
MJO Phase 8	20	21	7	39	7.17
Average Jul-Sept SOI	20	-2	-16	14	8.31
Average Oct-Dec SOI	20	-1	-14	16	8.78
Average Jan-Mar SOI	20	-2	-24	10	8.99
Average Apr-Jun SOI	20	-4	-20	13	9.03
Average Jul-Sep SST (°C)	18	22	21	23	0.43
Average Oct-Dec SST (°C)	18	27	25	28	0.70
Average Jan-Mar SST (°C)	18	27	26	28	0.59
Average Apr-Jun SST (°C)	18	23	22	24	0.45
Princess Charlotte Bay	**N**	**Mean**	**Minimum**	**Maximum**	**Std. Dev.**
Annual barramundi catch adjusted for effort	13	0	-13902	22792	12813.85
Annual prawn catch adjusted for effort	15	0	-114	84	49.81
Average Jul-Sep Min temperature (°C)	19	17	15	20	1.28
Average Oct-Dec Min temperature (°C)	19	23	21	24	0.69
Average Jan-Mar Min temperature (°C)	19	24	22	25	0.41
Average Apr-Jun Min temperature (°C)	19	20	18	21	0.92
Average Jul-Sep Max temperature (°C)	19	30	29	31	0.59
Average Oct-Dec Max temperature (°C)	19	34	32	35	0.64
Average Jan-Mar Max temperature (°C)	19	32	32	34	0.65
Average Apr-Jun Max temperature (°C)	19	30	29	31	0.58
Annual Min temperature degree days	19	7599	7158	8087	244.13
Annual Max temperature degree days	19	11580	11250	11791	156.82
Rain Jul-Sept (mm)	19	6	0	21	6.23
Rain Oct-Dec (mm)	19	244	87	599	154.54
Rain Jan-Mar (mm)	19	787	436	1323	245.38
Rain Apr-Jun (mm)	19	81	7	253	67.27
Annual rain (mm)	19	1118	665	1736	322.50
Flow Jul-Sep (Ml)	19	11529	1002	22077	6779.46
Flow Oct-Dec (Ml)	19	48358	3481	170824	46697.58
Flow Jan-Mar (Ml)	19	1347443	279305	2204024	671831.24
Flow Apr-Jun (Ml)	19	163458	11792	456567	123583.60
Annual flow (Ml)	19	1431531	131218	4282468	1199091.76
Total Evaporation Jul-Dec (mm)	19	1209	1000	1322	80.96
MJO Phase 1	19	20	5	38	8.68
MJO Phase 4	19	24	14	43	7.25
MJO Phase 6	19	22	11	41	7.55
Average Jul-Sept SOI	19	-1.8	-16	14	8.48
Average Oct-Dec SOI	19	-1.3	-14	16	9.03
Average Jan-Mar SOI	19	-2.6	-23	10	9.21
Average Jul-Sept SST (°C)	19	25	24	27	0.46
Average Oct-Dec SST (°C)	19	28	28	29	0.43
Average Jan-Mar SST (°C)	19	29	27	29	0.45
Average Apr-Jun SST (°C)	19	26	25	27	0.51
Annual SST temperature degree days	19	9865	9639	10053	125.88
Average Vwind Sep/Oct (m/s)	19	4.7	3.8	5.6	0.56
Average Vwind Mar/Apr (m/s)	19	1.6	0.4	2.7	0.69
Average UwindSept/Oct (m/s)	19	-2.3	-3.8	-0.6	0.78
Average Uwind Mar/Apr (m/s)	19	-2.5	-5.2	-0.3	1.29
Annual Vwind run (m/s)	19	1008.1	786.3	1329.9	162.59
Annual Uwind run (m/s)	19	-779.7	-1138.5	-489.7	174.41

Fitzroy River region

Expanding on the analysis of barramundi and rainfall/freshwater flow interactions, additional climate variables and climate indices which were identified in the barramundi life-cycle were selected for analysis. A correlation matrix of all relevant climate variables (for barramundi analysis lagged up to five years) and catch adjusted for effort was generated to identify significant relationships and possible collinearity between independent variables. Some of the independent variables were found to be significantly correlated with each other; however, as each describes a mechanism which affects the fishery in a different biological way (e.g. freshwater flow in the river bed versus rainfall replenishing wetland habitat separate from the river) in the earlier stages of the analysis, it was considered valid to include them. Collinearity between independent variables was compensated for through the use of forward stepwise ridge regression (FSRR) modelling (StatSoft. Inc. 2005). As not all of the variables were collineated and correlations where they did exist were weak, the ridge regression constant l (lambda) was initially set at 0.1. Three different FSRR models were built. The first used each of the climate variables which showed a significant correlation to catch in the correlation matrix, including lagged variables (Climate Variables Model). The second model incorporated each of the indices of the SOI and MJO for all lags (Climate Indices Model), and the third model used only significantly correlated climate variables lagged by two and three years (Predictive Model) so as to capture impacts on early life-cycle stages of the fish. The first two allowed for a comparison between the use of climate variables and climate indices in describing catch. The third explored the possibility of generating predictions of future catch with sufficient time for a response from fisheries managers and/or operators. Models were limited to three steps, as the use of more variables in the model risks an artificially high level of the variance being accounted for, and a corresponding decrease in forecasting skill due to the increased degrees of freedom (Shepherd *et al.* 1984). Residuals were checked for normality using a normal probability plot and for auto-correlation using the Durbin-Watson statistic. Adjusted coefficients of determination (R^2) which take into account the degrees of freedom in the model were calculated (StatSoft. Inc. 2005).

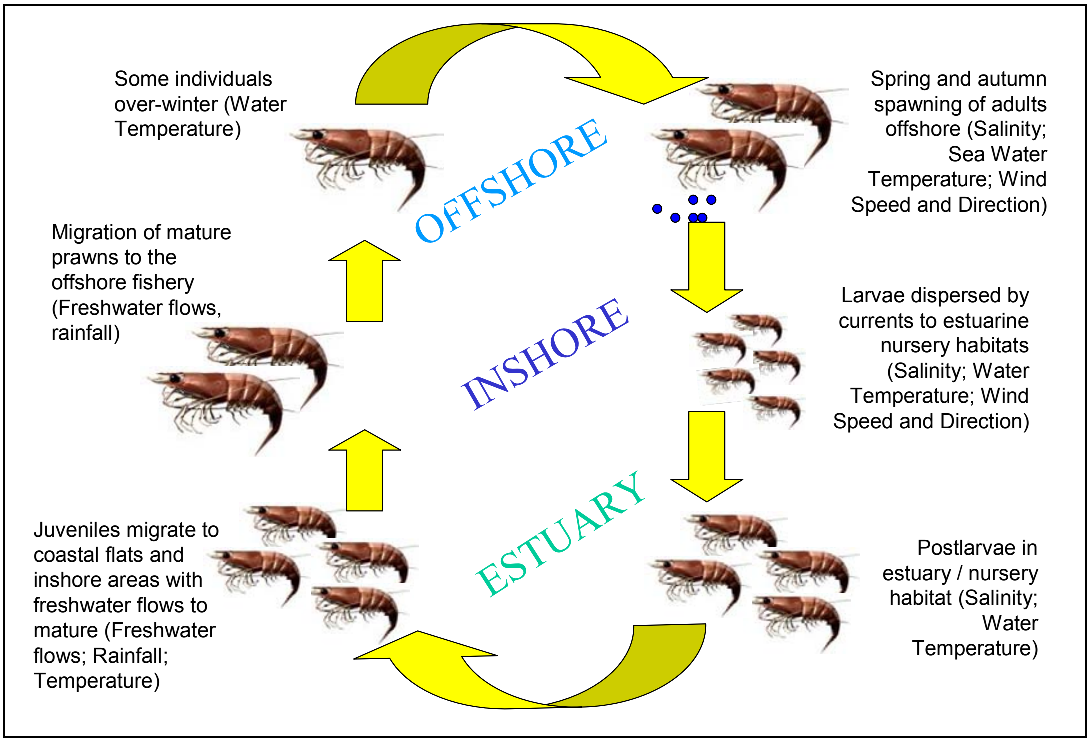
Princess Charlotte Bay region

Analysis for the PCB region followed the same methodology outlined for the Fitzroy analysis and determined the appropriateness of transferring methodologies and results spatially.

Banana prawn analysis

In order to determine the opportunity for transference of methodology and results to another species, an analysis of banana prawn and climate interactions followed the same methodology as that outlined for barramundi using climate variables significant in the banana prawn life-cycle. The PCB region was selected for the analysis as results from the analysis of barramundi indicated that a stronger signal might show up in this region as results from Robins *et al.* (2005) showed limited correlations for banana prawns in the Fitzroy River region. A literature review of previously published research was undertaken to determine possible linkages between climate influences and the species and a life-cycle model developed which includes these linkages (Figure 4.5). CFISH data for the species included additional grid squares off the coast to include banana prawn fishing grounds. Lag analysis was undertaken for a one year lag only due to the predominantly annual lifespan of the species.

Figure 4.5 Life-cycle model of the banana prawn showing links to climate



Predictive capacity of models developed

To explore the predictive capability of the models generated, the PCB barramundi Predictive model was considered. When used in a predictive tool, the capability of regression-based models has been questioned by several authors such as Stergiou and Christou (1996) and Myers (1998). However, the robustness of a regression model can be enhanced by using cross validation (Wilkes 1995) and data-splitting (Myers 1998). The PCB barramundi Predictive model outputs were cross validated against recorded catch using a 'leave-one-out' (LOO) technique (Wilkes 1995) which results in a Predictive R^2 value as per Equation 4.1. This cross validated R^2 value was used as a measure of the model's predictive capability.

$$\text{Predictive } R^2 = 1 - \frac{SSEdeleted}{SST} \qquad \text{(Equation 4.1)}$$

where $SSEdeleted = \sum_{i=1}^{n} (y_i - \hat{d}_i)^2$ and y_i is the i^{th} observed value and $\hat{d}_i$ is the predicted value when y_i is not included in the analysis.

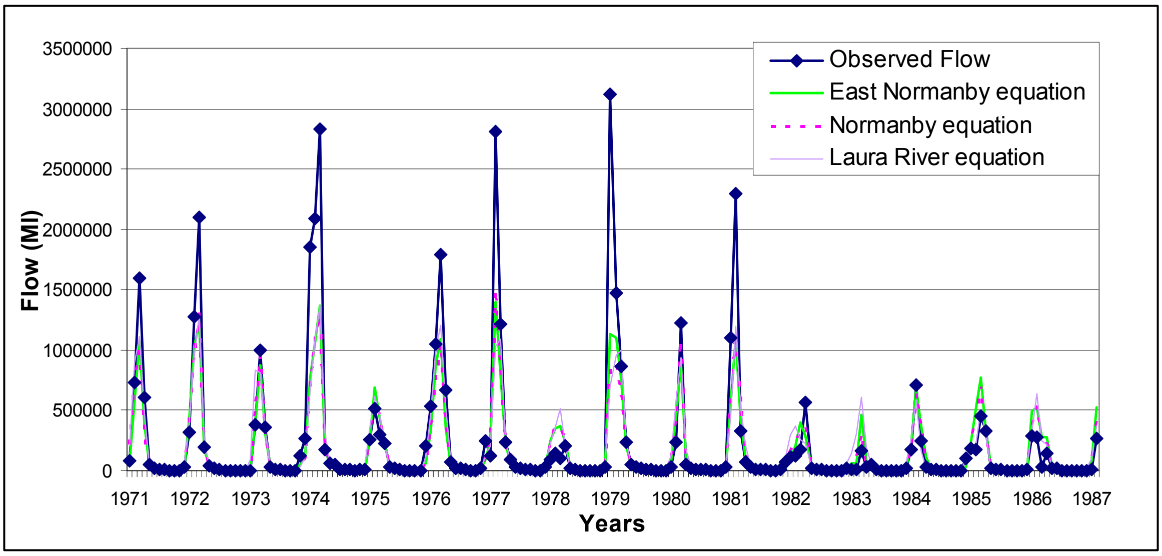
Evaluation of methodology

Improvements to future analysis of this kind that are suggested include:

i. Modelled PCB freshwater flow – Despite correlations with actual total basin flow returning R^2 values greater than 0.9 for each river used, flows generated to fill the missing years in total basin flow are underestimating peak flow events in extreme wet years (Figure 4.6). As threshold events may be the critical drivers of population, it is recommended that future analysis of freshwater flows in PCB are based either on a lower number of gauges for which there is actual data for the recent period, or improvements in modelled data.

ii. Modelled Climate Data - It is possible that the necessary use of spatially interpolated data (such as was sourced for many of the climate variables) will be inaccurate. In many cases extreme events are not modelled well and these events if representing significant thresholds in a population response would reduce the climate dependant signal in catch. Avoiding this is impossible unless real data can be sourced, and so the inclusion of a weather station on site for areas without data would be an improvement in any climatic analysis.

Figure 4.6 Actual versus modelled total freshwater flow into Princess Charlotte Bay



Results

Fitzroy River region – barramundi analysis

The correlation analysis of Fitzroy barramundi catch adjusted for effort and climate variables and indices only gave significant values in the year of catch for minimum temperature in July. This result varies from that found by Robins *et al.* (2005) and is explained by the different seasons used in the two analyses. Significant correlations were also found between catch adjusted for effort and the MJO phase four, average July-September and October-December SST all four years previous. Average January-March and April–June SSTs were significant at the five year lag period (Table 4.2).

As there was only one variable in the year of catch significantly correlated to barramundi catch adjusted for effort in the Fitzroy data set, FSRR of climate variables in the year of catch were not undertaken. However, a FSRR model was constructed for both climate variables and climate indices for data lagged by four and five years. Resulting models when adjusted for collinearity accounted for 67% of the variance for the climate variables (average July–September SST^{-4}; flow October–December^{-4} and rainfall January–March^{-5}) and 55% of the variance for climate indices (Table 4.3).

Table 4.2 Correlation coefficients (r) between the barramundi catch adjusted for effort and selected climate variables in the Fitzroy River region

Variable	0 lag	1 year lag	2 year lag	3 year lag	4 year lag	5 year lag
Maximum temp Dec (°C)	0.35	0.17	-0.51	-0.48	0.03	0.47
Minimum temp Jul (°C)	**-0.75***	-0.19	0.20	0.14	0.31	-0.29
Evaporation Annual (mm)	0.16	0.07	-0.35	-0.31	-0.48	0.04
Rain Jul-Sept (mm)	-0.46	-0.31	0.14	0.21	0.34	-0.41
Rain Oct-Dec (mm)	-0.19	0.04	0.46	-0.38	0.41	0.25
Rain Jan-Mar (mm)	0.26	0.11	0.03	-0.51	0.12	-0.11
Rain Apr-Jun (mm)	0.02	0.36	-0.25	0.16	-0.27	0.04
MJO Phase 4	0.04	0.28	0.10	0.05	**-0.64***	-0.22
MJO Phase 5	-0.06	0.20	-0.02	0.08	0.03	0.04
MJO Phase 8	-0.03	-0.13	-0.24	-0.14	0.28	-0.27
Average Jul-Sept SOI	-0.22	-0.13	0.18	0.00	0.44	-0.24
Average Oct-Dec SOI	-0.31	0.07	0.46	0.19	0.38	-0.34
Average Jan-Mar SOI	-0.09	0.23	0.42	-0.03	0.13	-0.46
Average Apr-Jun SOI	-0.14	0.08	0.26	0.20	0.05	0.02
Average Jul-Sep SST	-0.21	-0.13	-0.13	0.05	**0.77***	-0.37
Average Oct-Dec SST	0.11	0.26	-0.41	-0.55	**0.52***	0.20
Average Jan-Mar SST	0.16	0.11	-0.21	-0.22	-0.06	**0.59***
Average Apr-Jun SST	0.18	-0.11	0.12	-0.36	-0.07	**0.71***
Flow Jan-Mar (Ml)	0.18	-0.13	0.17	-0.41	0.18	-0.03
Flow Apr-Jun (Ml)	-0.25	0.04	-0.08	0.09	-0.07	0.23
Flow Oct-Dec (Ml)	-0.17	-0.12	0.51	-0.42	0.58	0.03
Flow Jul-Sep (Ml)	-0.37	0.00	0.20	-0.05	0.50	-0.07

(*P<0.05)

Table 4.3 Comparison of forward stepwise ridge regression models for the Fitzroy region CFISH barramundi catch

	Regression coefficient	Standard error	p-level
Climate Variables Model (adjusted R^2=0.67)			
Intercept	**-179142***	**44283.27***	**0.003***
Average Jul-Sep SST (4 year lag)	**7619***	**1886.14***	**0.003***
Flow Oct-Dec (4 year lag)	**5***	**2.14***	**0.038***
Rain Jan-Mar (5 year lag)	1302	893.33	0.179
Climate Indices Model (adjusted R^2=0.55)			
Intercept	**19115***	**7265.15***	**0.025***
MJO Phase 4 (4 year lag)	**-6017***	**1534.72***	**0.003***
Average Oct-Dec SOI (5 year lag)	**-366***	**126.02***	**0.016***
MJO Phase 5 (5 year lag)	1818	1064.05	0.118

(*P<0.05)

Residuals for the Climate Variables Model were normally distributed, independent (Durbin-Watson statistic; $P<0.05$) and fell within +2 standard deviations of the mean, indicating an absence of outliers. However, in the Climate Indices Model the financial year 1989/1990 was an outlier.

Princess Charlotte Bay region – barramundi analysis

The correlation matrix between selected climate variables and barramundi catch adjusted for effort identified 12 significant correlations (Table 4.4). The FSRR Climate Variables Model (Table 4.5) included rain July-September^{-2} (two years previous); evaporation annual^{-2} and average October-December SOI (no lag), and explained 68% of the variance in catch adjusted for effort. The Climate Index Model explained 53% of the variance with average October-December SOI (no lag), MJO Phase 4^{-1} and July-September SOI^{-2}. The Predictive Model contained rain July-

September^{-2}, evaporation annual^{-2} and average January-March SST^{-2} and explained 63% of the variance (**Table 4.4**). Once again, residuals for all models were normally distributed, independent (Durbin-Watson statistic; $P<0.05$) and fell within + 2 standard deviations of the mean.

Table 4.4 Correlation coefficients (r) between barramundi catch adjusted for effort and selected climate variables for the Princess Charlotte Bay region

Variable	0 lag	1 year lag	2 year lag	3 year lag	4 year lag
Minimum temp Jul (°C)	0.10	-0.16	-0.25	**-0.62***	-0.27
Maximum temp Dec (°C)	-0.55	-0.25	-0.02	0.06	0.21
Rain Jul-Sept (mm)	-0.01	0.02	**0.77***	0.46	0.48
Rain Oct-Dec (mm)	**0.56***	0.30	0.38	0.15	-0.06
Rain Jan-Mar (mm)	**0.56***	0.31	**0.62***	0.12	0.26
Rain Apr-Jun (mm)	0.37	0.40	-0.02	0.14	0.08
Flow Jul-Sep (Ml)	0.36	0.41	0.33	0.18	-0.34
Flow Oct-Dec (Ml)	**0.71***	0.37	0.29	-0.02	-0.13
Flow Jan-Mar (Ml)	0.52	0.35	**0.76***	0.36	0.43
Flow Apr-Jun (Ml)	0.33	0.33	0.13	0.27	0.08
Evaporation Annual (mm)	**-0.73***	-0.48	**-0.62***	-0.34	-0.15
Average Oct-Dec SST (°C)	0.12	0.25	0.40	0.46	**0.56***
Average Jan-Mar SST (°C)	0.32	0.17	**0.58***	0.03	0.29
MJO Phase 1	-0.55	0.00	-0.26	-0.08	0.17
MJO Phase 4	0.04	-0.39	-0.18	0.05	0.54
MJO Phase 6	0.38	0.50	0.16	0.21	-0.09
Average Jul-Sept SOI	0.47	0.12	0.29	0.13	0.06
Average Oct-Dec SOI	**0.62***	0.19	0.29	0.14	0.14
Average Jan-Mar SOI	0.47	-0.18	0.10	0.19	0.37

(*P<0.05)

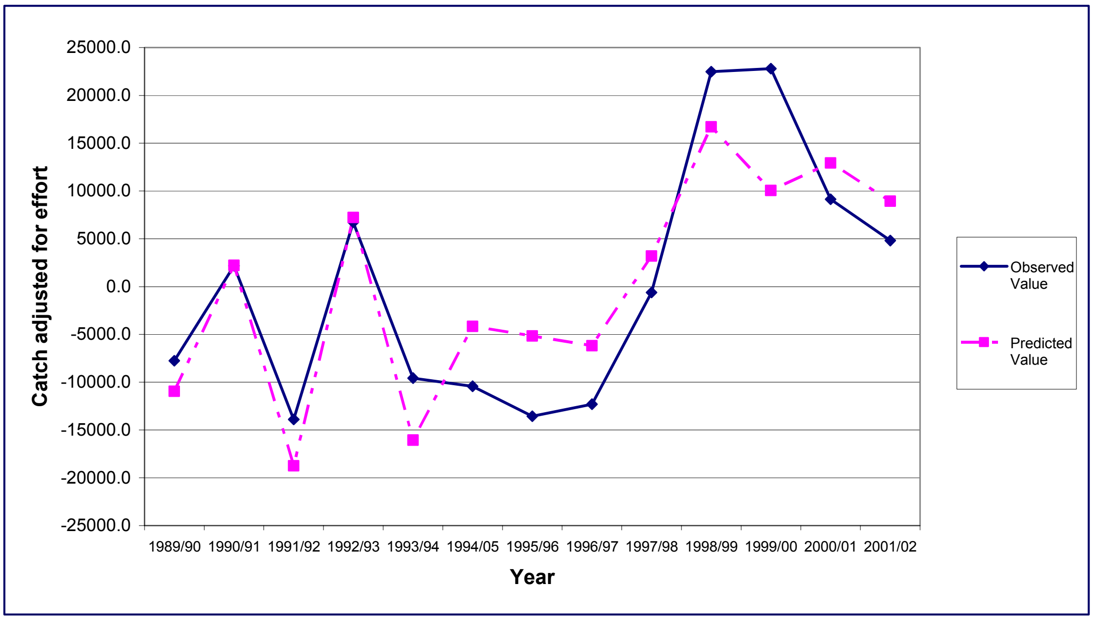
Predicted versus observed values of catch were plotted for the Predictive Model with all but four points falling within the 95% confidence limits (**Figure 4.7**). Cross validation of the Predictive Model returned an adjusted R^2 of 0.48 compared to an R^2 of 0.63 initially calculated. This shows the robustness of the model and indicates that even when used in a predictive capacity, the Predictive Model is explaining nearly half the variance in catch adjusted for effort in PCB.

Table 4.5 Comparison of forward stepwise ridge regression models

	Regression coefficient	Standard error	p-level
Climate Variables Model (adjusted R²= 0.68)			
Intercept	27524	19494.78	0.19
Rain Jul-Sept (2 year lag)	6453	2348.22	**0.02***
Evaporation Annual (2 year lag)	-0.01	0.00	0.07
Average Oct-Dec SOI (no lag)	377	234.08	0.14
Climate Indices Model (adjusted R²= 0.53)			
Intercept	18929	7768.18	0.04
Av Oct-Dec SOI (no lag)	814	255.66	**0.01***
MJO Phase 4 (1 year lag)	-669	288.47	**0.05***
Average Jul-Sept SOI (2 year lag)	463	246.61	0.09
Predictive Climate Variables Model (adjusted R²= 0.63)			
Intercept	-132042.32	156562.77	0.42
Rain Jul-Sept (2 year lag)	6552.84	2635.84	**0.03***
Evaporation Annual (2 year lag)	-0.01	0.00	0.07
Average Jan-Mar SST (2 year lag)	5671.57	5452.42	0.33

(*P<0.05)

Figure 4.7 Observed versus predicted value of catch adjusted for effort for the PCB barramundi fishery for financial years 1989/90 to 2001/02



Princess Charlotte Bay region – banana prawn analysis

The correlation coefficient between seasonal climate parameters and CFISH catch adjusted for effort was calculated for the year of catch but did not identify any significant relationships at the P<0.05 level (Table 4.6). Variables lagged by one year gave significant correlations with average October–December maximum temperature (r=-0.56; P<0.05) and total July–September evaporation (r=-0.53; P<0.05). Analysis of annual variables returned no significant correlations. It should be noted that the two significant variables (Average October-December maximum temperature and Total Evaporation July–December) are correlated with each other (r=0.74, P<0.05). A forward stepwise ridge regression analysis of both seasonal and annual climate parameters against catch adjusted for effort failed to include any parameters at the P<0.05 level.

Table 4.6 Correlation coefficients (r) between catch adjusted for effort and seasonal climate variables in the year of catch and lagged by one year

Variable	0 lag	1 year lag
Average Jul-Sep Minimum temperature (°C)	-0.03	-0.02
Average Oct-Dec Minimum temperature (°C)	0.07	0.13
Average Jan-Mar Minimum temperature (°C)	0.42	-0.04
Average Jan-Mar Minimum temperature (°C)	-0.19	0.04
Average Jul-Sep Maximum temperature (°C)	0.06	-0.18
Average Oct-Dec Maximum temperature (°C)	-0.40	**-0.56***
Average Jan-Mar Maximum temperature (°C)	-0.09	-0.22
Average Apr-Jun Maximum temperature (°C)	-0.24	-0.43
Rain Jul-Sept (mm)	-0.47	-0.25
Rain Oct-Dec (mm)	0.41	0.30
Rain Jan-Mar (mm)	0.00	0.06
Rain Apr-Jun (mm)	-0.42	0.03
MJO Phase 1	-0.27	-0.05
MJO Phase 4	-0.09	0.06
MJO Phase 6	-0.40	0.12
Average Jul-Sept SOI	-0.08	0.26
Average Oct-Dec SOI	-0.01	0.25
Average Jan-Mar SOI	-0.14	-0.05
Average Jul-Sept SST (°C)	-0.08	-0.06
Average Oct-Dec SST (°C)	-0.16	-0.41
Average Jan-Mar SST (°C)	0.18	-0.22
Average Apr-Jun SST (°C)	0.00	-0.34
Average Vwind Sep/Oct	0.18	0.05
Average Vwind Mar/Apr	0.10	-0.19
Average UwindSept/Oct	-0.40	-0.22
Average Uwind Mar/Apr	-0.49	0.12
Flow Jul-Sep (Ml)	0.02	0.33
Flow Oct-Dec (Ml)	-0.02	0.17
Flow Jan-Mar (Ml)	-0.02	0.20
Flow Apr-Jun (Ml)	-0.02	0.23
Total Evaporation Jul-Dec (mm)	-0.31	**-0.53***

(*P<0.05)

Discussion

Barramundi

In PCB, significant correlations in the year of catch support the theory that early wet season freshwater flow (r=0.71) affects the catchability of barramundi by enhancing fresh water connections to the commercial fishery. In early wet years, male fish residing in fresh water reaches return to the estuary in large numbers and are caught later that same year as rainfall and flow are high and connectivity to these areas is good. As would be expected, the October-December SOI as an indicator of seasonal rainfall, and hence flow, was also significantly correlated with catch in the same year (r=0.62). Results from the analysis of data in the Fitzroy region in this study vary from the findings of Robins *et al.* (2005). Although the direction of relationship between summer and spring rain in the year of catch is the same, in this analysis results were not significant. The discrepancy is most likely a result of the difference in seasonal definition which varied across the two studies.

The significant negative correlation with minimum temperature in July in the Fitzroy River region is puzzling as the Fitzroy River region is towards the southern end of the species geographical distribution and it would be expected that cold winters would have a negative impact on the fishery, a conclusion validated by recorded fish kills in cold winters (Robins *et al.* 2005). However, it could be describing some other more complex interaction such as the predator prey responses to cooler winters.

Lag correlations indicate that conditions which maintain optimum nursery habitat, and therefore improved survival of young-of-year barramundi such as high rainfall and freshwater flows in January-March, high rainfall in July-September, high January-March SSTs and low levels of evaporation in the PCB region, are significantly affecting catch two years later. Annual evaporation, a parameter that has not been considered in earlier studies, also gives a highly significant inverse relationship with barramundi catch in this region. The impact of this on the fishery may be explained by research in the Northern Territory which has shown that the size or area of available wetland nursery habitat appears to be the strongest measure of population fluctuations in barramundi (Griffin 1985).

Significant negative correlations with minimum July temperatures three years prior in the PCB region, significant positive correlations with sea surface temperature four years prior in the PCB region and five years prior in the Fitzroy may be identifying the effect of temperature on gonad activity and egg maturation, and hence spawning success in subsequent years (Rod Garrett pers. comm. August 2005, Principal Fisheries Biologist, DPI&F). Growth rates for barramundi vary considerably between genetic stocks and even from one river to the next (Shaklee *et al.* 1993). Male barramundi in river systems north of 15°S on both the east and west coasts of Cape York Peninsula have been found to be breeding at age one or two years (Davis and Kirkwood 1984; Garrett 1987), and barramundi as young as two and three years old are entering the commercial fishery in the Fitzroy River region (Staunton-Smith *et al.* 2004). However, without reliable age class data for each region it is not possible to validate the results.

Variables selected in the PCB Climate Variables Model capture this influence from climatic conditions two years prior to catch (rain July–September^{-2} (+), evaporation^{-2} (-)). Catchability of barramundi in the year of fishing is explained by the inclusion of the October–December SOI (+) (no lag). This is also the first parameter selected for the PCB Climate Indices Model, which includes phase 1 of the MJO^{-1} (-) a measure of suppressed rainfall, and a possible indicator of shallow habitat maintenance, and July-September SOI^{-2} (+) a predictor of early wet season rainfall. In the Fitzroy analysis, all the variables selected in the Climate Variables Model were lagged by four or five years and again describe conditions desirable for spawning and nursery habitats including warm SSTs^{-4}, high October–December flow^{-4} and January–March rainfall^{-5}.

The inclusion of rain from July–September^{-2} in both the PCB Climate Variables Model and PCB Predictive Model is somewhat surprising as rainfall at this time of year, although variable, is minimal (0.1 – 21.3 mm) in the region. It may be that this rain maintains juvenile habitats which would otherwise dry out, resulting in the death of all fish. There may also be the secondary benefit of establishing a suitable nursery habitat for the arrival of early spawned fish in September-October. Again, research in the Northern Territory has shown that spawning commences in the very early months of the wet season, before the regular monsoon rains, and that the success of this early spawning significantly depends on the amount of rain which falls to replenish water levels in supra-littoral nursery swamps (Griffin 1985).

At each site the use of climate indices as opposed to climate variables reduced the amount of variance explained by the models and so are not considered as useful a tool in forecasting impacts on the fishery. This is most likely because climate indices are a measure of large climate systems (such as the Madden Julian Oscillation in the case of the MJO and the El Nino/Southern Oscillation (ENSO) in the case of the SOI) and local conditions reduce their efficacy in describing what happens at more regional scales. This is demonstrated by the fact that the SOI is describing up to only 46% of rainfall or flow variability in the Fitzroy region.

The best opportunities for predictive management of the fishery is by identifying climate variables significant in creating optimal conditions for successful spawning and juvenile development. The importance of conditions at the time of spawning and early development is clearly shown by the PCB Predictive Model which includes rain July–September, annual evaporation and January–March SST (all two years previous) which explains 62.7% of the variation in catch. Use of the model in a predictive capacity gives a cross validated R^2 squared value of 0.48, i.e. nearly half of the variance in catch can be described by the selected variables two years prior.

Banana Prawns

According to the latest 'Queensland's Fisheries Resources: Current condition and recent trends' report (Williams 2002), commercial prawn fishers consider the banana prawn to be a highly variable commodity with catch affected on an annual basis by variations in climate including freshwater flows and district rainfall. Previous research in the Gulf of Carpentaria has shown significant correlations between climate variables and catch, with rainfall explaining up to 72% of the variance in commercial catch for some areas (Staples and Vance 1986). However, when annual landings of catch on the east coast were compared with summer (January–March) rainfall, correlations varied depending on location with significant relationships for Townsville, Bowen and Mackay, but not for grounds off Cairns, Yeppoon, Bundaberg and Moreton Bay (Williams 2002). Results of analysis between rainfall, freshwater flow and banana prawn catch by Robins *et al.* (2005) indicated that summer flow and summer rain were significantly correlated with banana prawn catch adjusted for effort when considering total catch from both otter- and beam-trawlers. However when otter-trawlers only were analysed there were no significant variables. Commercial fishing in PCB consists entirely of otter-trawlers, and so this may explain a lack of significant relationships in the year of catch for this region. Significant positive correlations with September–December maximum temperature and annual evaporation lagged by one year are difficult to explain in the context of the species life-cycle. Most prawns are caught in the year of spawning with probably only a few individuals over wintering. Of those that do over winter, it is possible that warm temperatures in their year of spawning may affect survival, although other studies have shown a negative correlation between summer temperatures and catch (e.g. Vance *et al.* 1985). This result was not spatially consistent.

Relative to other grounds on the east coast, PCB is a small banana prawn fishery and effort is variable and often very low. It could be that these limitations in the commercial catch data may reduce the signal from freshwater flow and other climate variables considered here on the population. Additionally, the seasonal scale of the analysis may mean a signal from a discrete but significant threshold event (e.g. a few days of flood) may be buffered, reducing the relationship.

Current modelling of banana prawn catch by the Department of Primary Industries and Fisheries along the east coast of Queensland south of Cairns allows for parameters of freshwater flow, rainfall and temperature to be included (Michael O'Neil, Fisheries Biologist, Deception Bay DPI&F pers. comm. June 2005). Preliminary results in this work show reasonably consistent significant correlations with freshwater flow and rainfall along the coast with stronger relationships in the north of the state. However, in order to detect this relationship, daily catch data for individual boats was needed so the model could standardise for boat success and compensate for improved success within individual boats over time, a level of analysis beyond the scope of this study.

Chapter 5. Effects of stream flows on selected recreational fisheries

J. Platten

Summary

This chapter examines the influence of freshwater flows on the catch of a recreational fishing club and discusses probably causes. It demonstrates the existence of a flow-volume threshold, above which catch rates are positively influenced. It also examines the implications of reduced freshwater flows both on the catch rates of recreational fishers and estuarine productivity.

The club's catch is based around the sand whiting (*Sillago ciliata*) (around 80-90% of the catch). Whiting are the most commonly captured species by recreational fishers in Queensland and Australia and are important in estuarine food chains. The Wanderers Fishing Club fishes set locations in central Queensland. Two sites visited were examined. A Gladstone Harbour site influenced by the Calliope and Boyne Rivers and a second influenced by the Fitzroy River. Median fish catch rates (fish/angler/trip) over thirteen years were positively influenced by annual flows above a flow threshold. For the site within Gladstone Harbour the correlation coefficient between catch rates and flow volume of greater than 150000 ML was 0.989. For flows below this magnitude the correlation was -0.134. For the site adjacent to the Fitzroy River, flows positively influenced catch rates when greater than 1 GL (r = 0.91) but not when below this volume (r = -0.134). The differences in the flow thresholds are likely to reflect the distance of the sites from the mouths of the estuaries. If these flows are not attained, a slow reduction in catch rate occurs over time until the next flood. The reasons behind these relationships appear related to fish growth rates and survival. The majority of whiting in the catch are close to the minimum legal size. However, their ages are highly variable (from 1- to 6-years old). The best catches correspond to wetter years with more numerous but small fish of younger age. Linked to this, fish spawned in wetter years tend to be more numerous in the catch than those spawned in dry years. It would seem that more fish survive and grow to legal size in high flow years.

Whiting are largely second order macro-benthos feeders. A comparison between catch rates in Gladstone harbour and macro-benthos abundance shows a close correlation (r = 0.967). This suggests that the increase in the numbers of whiting probably reflects an overall increase in food abundance and the productivity of an estuary. The results suggest that at any particular site there is a flow quantum that positively influences estuarine productivity, fish growth and abundance and consequently catch rates. Reduced flows are likely to result in a gradual reduction in catch rates and will have most impact for sites less directly influenced by rivers within the outer estuary and coastal zone.

Introduction

Why examine recreational catch rates?

Platten (2004, 2005) examines time series trends of catch rates of recreational fishing clubs in central Queensland and is the basis of this chapter. Staunton-Smith *et al.* (2004) and Robins *et al.* (2005) have examined evidence of freshwater flows influencing commercial fisheries in central Queensland. A range of commercial fisheries show a positive correlation between freshwater flows and fisheries production. This data provides a powerful insight into the relationship between flow and fishery production and clearly establishes that many fisheries are positively influenced by river discharge. Why then consider recreational fisheries? To establish a relationship between catches and flow is hardly unique.

The recreational fishing club catch information provides an opportunity to compare the influence of freshwater flow on catch information from a different source and provides evidence from different species and methods of fishing that may reinforce the trends established for commercial species. This in itself is important, but in addition, recreational fishing club data also provides some unique features that help further elucidate the influence of freshwater flow on fishery production.

Recreational fishing club data can be analysed at a much finer spatial resolution than commercial fisheries data. Club trips have been conducted at the same particular sites over an extended period. These sites are identified to within one kilometre. This is in contrast to much of the commercial data that cannot be resolved beyond a thirty-minute grid (i.e. 1668 km^2). This provides the opportunity to examine freshwater flow influence at a particular site and not over a broad region.

The recreational fishing club data also has a degree of control over some factors affecting catches. In particular the club fisher's fish:

- in set comparable locations;
- over set time periods;
- on days with similar tides;
- using similar methods;
- seeking similar species; and
- in every month of every year.

These issues are not necessarily controlled in the commercial fishery. Fishers are also motivated to maximise their catch at each site and to carefully record their results because of the competitive nature of the activity.

There are also some advantages in the analysis of recreational catch data used herein, in that the catch rates can be more easily corrected for effort than the commercial data. That is, it can be difficult to tell whether an increase in commercial catch is related to increased abundance of fish or whether increased catch is caused by more effort being generated by perceptions of abundance by the fishers. This issue can be resolved with the current recreational data, as the participants tend to fish despite the perceived chance of success. The data also provides an insight into the numbers of fish involved, as well as their size. This knowledge is useful in investigating the cause of any correlation between freshwater flow and catch. The commercial catch is based around the mass of fish landed. It can be difficult to determine whether an increase in commercial catch is related to increased numbers of fish or larger fish. Because club fishing is based around line fishing, the recreational data also provides some insights related to the causes of any freshwater flow/catch correlation. An increase in catch is highly likely to be related to more fish feeding on baits.

There are also disadvantages of the data. Line fishing is likely to be particularly influenced by experience and skill of the fishers. The skill level of competitors will vary and the group of individuals fishing from year to year does not remain the same. However, a core group of fishers have remained active throughout much of the study period. There are controls on the type of gear that can be used, but increasingly efficient gear (or experience in using it) could also influence results. The catch consists of a multi-species mix and the species composition is only available in more recent years. The implications of this are considered further below, however the fact that all are caught on particular baits (and hence tend to be at similar trophic levels) and the dominance of certain species (over 80 % of the catch is whiting in the club data examined) reduces the impact of this issue.

It is important to consider both commercial and recreational data to gain a more complete understanding of the influence of flow on catches, as both provide important contributions to understanding.

Aims

This chapter examines what can be deduced from the catch of recreational fishing clubs in central Queensland to inform flow/ catch correlations. It sets out both to examine evidence of flow/catch correlations and to attempt to postulate as to the cause of observed flow influences. This chapter is a summary of work in progress and a series of working hypotheses requiring further investigation, rather than definite findings. However, the information points to important issues deserving of consideration in relation to the influence of freshwater flows on fisheries.

This chapter examines evidence for three working hypotheses:

1) That the recreational fishing club catch rates are positively influenced by freshwater flows;
2) That the catch rates of the Wanderers Fishing Club at particular sites are influenced by freshwater flows above a certain threshold; and
3) That Wanderers Fishing Club catch rates respond relatively rapidly to threshold flows through changes in the growth rate of whiting close to the minimum legal size.

Hypothesis 1. Club catch rates are positively influenced by freshwater flows

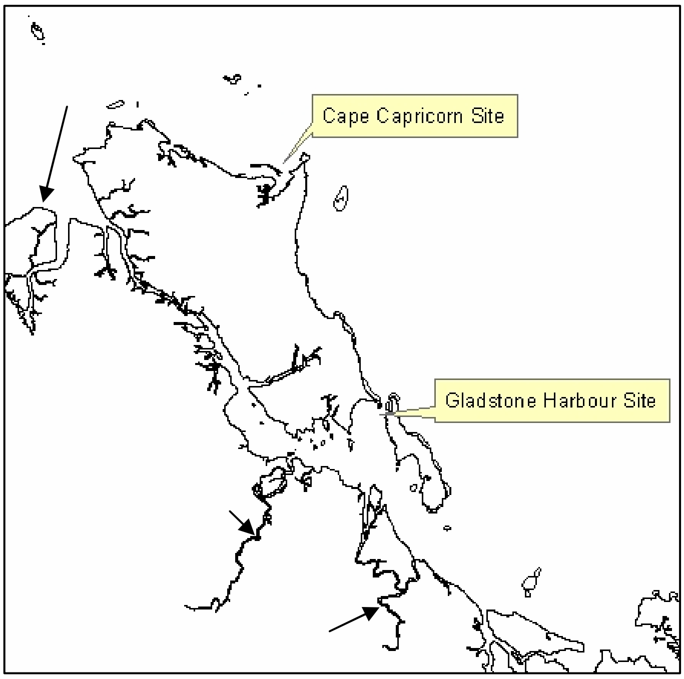
Methods

Catch rates

The catch rates of the Wanderers Fishing Club (see Platten 2004, 2005) were used to establish correlations. These fishers conduct monthly competitions in estuaries from Bustard Head to Cape Capricorn and hold reliable data from 1982/83, 1987/88 and from 1990/91 to 2001/02. Each club trip is held over five hours in conjunction with a spring tide. They fish in similar locations each year. Hence there are data available in each year from catches taken from the shores of Facing Island within Gladstone Harbour, and from the beaches close to Cape Capricorn (Figure 5.1). Fishing has always been conducted using rod and reel techniques; almost universally using natural baits of yabbies (*Carassiops* spp.). All competitors fish the same locality and for the same length of time (i.e. five hours).

Data were analysed to indicate the median number of fish caught per angler per trip for each financial year (July to June) at each locality. The median was chosen to reduce any bias of unusually low or high individual catches and variations in skill between anglers (see Mapstone *et al.* 1996).

Figure 5.1 Locations fished by the Wanderers Fishing Club



Freshwater flow

Similar flow measures were used as those by Robins *et al.* (2005). Boyne River flow was estimated using the Integrated Quantity and Quality Model (IQQM) developed by the Gladstone Area Water Board in conjunction with the Department of Natural Resources and Water (NRW). Estimated annual flow downstream of Awoonga Dam was used. Calliope River flows measured at 'Castlehope' gauging station by NRW were used for the Calliope River and Fitzroy River flows were based on flows measured at The Gap gauging station by NRW.

Correlations

Correlations were sought between catch rates in Gladstone Harbour, total annual Calliope River flows, annual Boyne River flows and combined Boyne and Calliope River annual flows. Cape Capricorn catches were compared with annual Fitzroy River flows. Both immediate (same year) responses and responses lagged by one to four years (as per Staunton-Smith *et al.* 2004) were sought. Correlations were calculated using the Pearson product moment correlation coefficient using the routines of SYSTAT 10 (SPSS Inc.).

Results

Catch rates of the club were positively correlated to river flow (Table 5.1and Table 5.2). The Gladstone Harbour catch rates were correlated with the flow of the Calliope River, the Boyne River and the combined flow of the two rivers. The closest correlation was with the combined flow of the two rivers for the same financial year but a significant correlation also exists for a two year lagged response (Table 5.1). Cape Capricorn catches were correlated with Fitzroy River annual flows of the same financial year (Pearson correlation coefficient r=0.73). A weak lagged response exists with flows two years previous (i.e. $Flow^{-2}$) (Table 5.2).

Table 5.1 Correlation coefficients (r) between catch rates of the Wanderers Fishing Club in Gladstone Harbour and the freshwater flow of the Calliope, Boyne and combined river flow

River	Flow	Flow $^{-1}$	Flow $^{-2}$	Flow $^{-3}$	Flow $^{-4}$
Calliope	**0.58***	0.26	**0.59***	0.10	0.36
Boyne	**0.68***	0.49	**0.67***	0.31	0.42
Total flow	**0.68***	0.40	**0.67***	0.23	0.42

Flow^{-1}, Flow^{-2}, Flow^{-3} and Flow^{-4} represent the correlations with flows from 1 to 4 years previous. (*$P<0.05$)

Table 5.2 Correlation coefficients (r) between catch rates of the Wanderers Fishing Club at Cape Capricorn and the freshwater flow of the Fitzroy River

River	Flow	Flow $^{-1}$	Flow $^{-2}$	Flow $^{-3}$	Flow $^{-4}$
Fitzroy	**0.74***	-0.40	0.45	-0.003	0.11

Flow^{-1}, Flow^{-2}, Flow^{-3} and Flow^{-4} represent the correlations with flows from 1 to 4 years previous. (*$P<0.05$)

Discussion

The results are similar to the evidence presented by Robins *et al.* (2005) for a range of Fitzroy River fish species and fisheries. Catch rates appear positively influenced by freshwater flows. What is more unusual is that the most significant correlation is immediate and not lagged. This appears unusual for most finfish catch data. For example Robins *et al.* (2005) and Staunton-Smith *et al.* (2004) note a three to four year lagged response between freshwater flows and barramundi catch. It should be noted that both freshwater flows and catch rates are for aggregated financial years. In almost all cases the major flows will have occurred during the period November to March so that a lagged response of up to seven months could be involved. This raises the issue as to whether the response observed may not represent a causative relationship but rather a coincidental response. This issue is further considered below, however that the same trend in correlation exists for two separate locations and freshwater flow from two separate systems does suggest a causative relationship.

The freshwater flow from the Calliope and Boyne Rivers and total flows into Gladstone Harbour are positively auto-correlated and the combined total freshwater flow is particularly dependent on Boyne River discharge because no water passes over the Awoonga Dam in a number of years. This could partially explain why a closer correlation exists with Boyne flow than Calliope flow.

Currie and Small (2005) and Platten (2005) note a very close relationship in trends of the catch rates of the Wanderers Fishing Club and macro-benthos abundance (r=0.97) over the period 1995 to 2000. However Currie and Small (2005) found no correlation between macro-benthos abundance and river flow, rainfall or other climatic measures. Examination of the Wanderers Fishing Club data over the same time period shows that there is no significant correlation over this period as well (r=-0.18 with combined flow, r=-0.37 with Calliope flow). However, over the longer time period available for the catch rates a positive correlation exists. This suggests that in some years (and not all) a particular flow regime could drive the flow/catch rate correlation. This is an issue that has not been considered in any literature reviewed.

The Wanderers Fishing Club data comes from defined locations. The Cape Capricorn and Gladstone Harbour locations are some distance from the mouths of the rivers; the Cape Capricorn site is around 35 km from the mouth of the Fitzroy River and the Gladstone Harbour site some 11 km from the Calliope River mouth and 8 km from the Boyne River mouth. Observations of satellite images and hydrodynamic modelling (J. Platten, pers. obs.) show that both sites are influenced by the flood plumes of the rivers, however not in every year.

It might be expected then that there may be a threshold flow above which catches may be beneficially affected. It is thus hypothesised that for a particular site outside of the mouth of a river, there will be a threshold river flow that could have a beneficial impact. Evidence for this hypothesis is considered in the following section.

Hypothesis 2. Club catch rates at particular sites are influenced by flows above a certain threshold

Methods

Scatter plots were made of the relationship between flow (combined Calliope and Boyne River flow for Gladstone Harbour, Fitzroy River flow for Cape Capricorn) and Wanderers Fishing Club catch data (see above). These were examined for evidence of threshold flow influence i.e. were there freshwater flows above which a positive flow/catch rate relationship was more readily visible. Correlations (Pearson correlation coefficients) were then prepared between catch rates and flows above and below this freshwater flow threshold.

Results

Gladstone Harbour

A defined linear relationship was visible in the scatter plot for years where combined flow was greater than 150,000 ML (Figure 5.2). For years where this flow threshold was exceeded (four in all), a very close linear relationship between freshwater flow and catch was evident

(r = 0.99, R^2 =0.98; Table 5.3b). That is, ~98% of variance in these years can be explained by freshwater flow, below this threshold the correlation was very weak (r=-0.13; Table 5.3a).

Figure 5.2 Scatter plot of total flow from the Boyne and Calliope Rivers (X axis) and the catch rate of the Wanderers Fishing Club in Gladstone Harbour (Y axis). Vertical line shows a flow of 150000 ML.

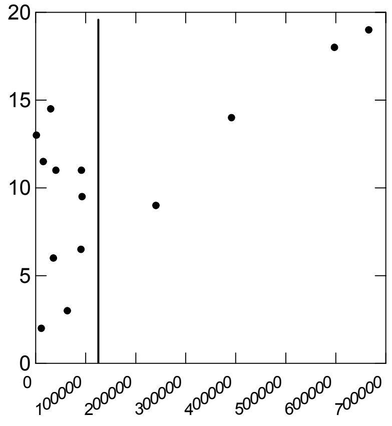


Table 5.3 Correlation coefficients between catch rate of the Wanderers Fishing Club in Gladstone Harbour and the combined annual freshwater flow from the Boyne and Calliope Rivers for years when: a) flow was <150000 ML; and b) flow was >150000 ML.

a) Combined flow <150000 ML

	Catch	Flow
Catch	1.00	-0.13
Flow	-0.13	1.00

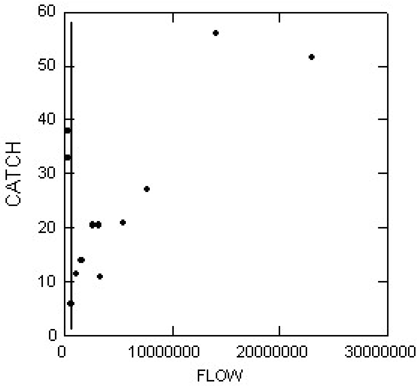
b) Combined flow >150000 ML

	Catch	Flow
Catch	1.00	0.99
Flow	0.99	1.00

Cape Capricorn

The scatter plot shows that years where annual Fitzroy River flow was greater than 1 GL were more closely related to catch rate than below this figure (Figure 5.3). Flow was more closely correlated with catch rate above 1 GL (r=0.91) than below (r=0.003).

Figure 5.3 Scatter plot of flow from the Fitzroy River (X Axis) and the catch rate of the Wanderers Fishing Club at Cape Capricorn (Y Axis). Vertical line shows a flow of 1GL



Discussion

The results strongly support the hypothesis that the recreational catch rates of the Wanderers Club at particular sites outside of the mouth of river systems are influenced by freshwater flows above a certain threshold. This provides further support for the view that freshwater flows beneficially influence fisheries. For locations not within the rivers influence in every year, it would be expected that only in those years where freshwater flow are above a threshold will a beneficial effect occur.

If this hypothesis holds for other fisheries and fish, then there are a number of important ramifications. If catch data are collected from a broad area outside of the mouth of a river, then it may be that a range of threshold flow levels apply. This could reduce any catch/flow correlation because only locations close to the river mouth may be beneficially affected by smaller river flows. Similarly if data are collected over a dry period where threshold flows are not attained, then there is a risk that no correlation will be observed. This could help explain why no correlation (or a weak one) is observed in some fisheries data.

This relationship is also of considerable importance because of the changing freshwater flow patterns in local coastal streams. Increasing flow regulation of streams may decrease the frequency of freshwater flow events greater than thresholds and the projections of the impact of climate change on river flows also points to decreasing freshwater flows. This may mean that there will be fewer threshold flows entering Gladstone Harbour or from the Fitzroy River. The impact of this is unknown but deserves close consideration. Additionally, if catch rates are indicative of estuarine productivity (see Platten 2005), then there may be important consequences for coastal ecosystems. The link between estuarine and inshore productivity and climate change has received little research attention and deserves much closer examination.

One important issue is that all correlations examined in this section are based on time lags of less than one year between freshwater flow changes and a catch response. Unless catch rates could respond with this immediacy, then the correlation between them would be coincidental. Robins *et al.* (2005) correctly emphasise the importance of considering the biology of a species and its fishery characteristics in determining the influence of freshwater flow.

The Wanderers Fishing Club catch consists mainly of whiting (around 80%, Platten 2005) and particularly sand whiting *Sillago ciliata*. The characteristics of the Wanderers Club catch and the biology of sand whiting are used to formulate an hypothesis in the next section that could account for the correlations observed.

Hypothesis 3. Club catch rates respond relatively rapidly to threshold flows through changes in growth rate of whiting close to the minimum legal size

Robins *et al.* (2005) review proposed mechanisms for freshwater flows influencing estuarine fishery production. These might be summarised as:

1) Changes in food availability related to nutrient enrichment associated with freshwater flows;
2) Changes in distribution related to translocation or alteration of habitat. This may cause fish to become more available to fisheries; and
3) Changes to population dynamics associated with factors such as growth, recruitment and survival.

They propose an integrated approach for examining the flow/catch relationship that involves both (i) an examination of the fishery characteristics, life history, and biology of the species to attempt to hypothesise how flow could influence these factors; and (ii) examination of catch and flow statistics for evidence to support this view.

In order to examine whether the catch rates of the Wanderers Fishing club could respond in the manner suggested from the correlations of catch and flow, it is thus necessary to briefly examine the characteristics of the clubs catch and the biology of the key species. The catch of the Wanderers Fishing Club is dominated by whiting, *Sillago* spp. (around 80% of the catch, Platten 2004, 2005). Observations of the catch from several outings suggest that all of the whiting caught were sand whiting (*Sillago ciliata*). Any rapid changes in catch rates are likely to be based around changes in the catch of this species.

Sand whiting feed on benthic macro-invertebrates (Burchmore *et al.* 1988). They grow relatively rapidly and may mature in their first year of life at around 24 cm in length (Burchmore *et al.* 1988). Sand whiting are most commonly found on sandy substrates or associated with seagrass (Burchmore *et al.* 1988). Currie and Small (2005) examined macro-benthos abundance (the principal food group used by sand whiting) at a number of sites in Port Curtis. This abundance is closely correlated (r=0.967) to the catch rates of the Wanderers Club (see Platten 2005 for a more detailed examination), suggesting that it is at least possible that changes in catch rate could be related to changes in food abundance.

Whiting are the most common species of fish taken by recreational fishers in Queensland (Higgs 1998). The catch of the species was regulated by a minimum size limit of 23 cm total length (TL) at the time of this study.

There is no information available that links sand whiting catch or biology to freshwater flows. However, examination of the three causal mechanisms suggests some possibilities. These are summarised as follows as a conceptual framework for further examination:

1) Changes in food availability: Increased nutrient input related to larger freshwater flows could provide the basis for improved primary production and ultimately increased food availability via the food chain. This could increase growth rates and or result in increased survival. This could increase the numbers of whiting available to the fishery;
2) Changes in movement patterns or habitat resulting from increased freshwater flow: Whiting could move away from locations of lower salinity or in response to other changes in habitat related to increased freshwater flow. This could result in more fish being available at the fishing locations (at least temporarily);
3) Changes to population dynamics associated with factors such as growth, recruitment and survival: Increased growth rate related to increased food availability linked to increases in nutrient, resulting in more fish reaching or surviving to legal size and hence becoming available to the fishery.

The first and third causal mechanisms are obviously linked. They might be aggregated to a single causal mechanism suggesting increased nutrient supply could result in increased growth, survival and recruitment to the fishery. Thus, this causal mechanism requires a biological population dynamics response to freshwater flow inputs (positive growth and or survival). To test this hypothesis, information related to the age, growth and survival of sand whiting at the sites and the age structure of the catch would assist. Further information linking changes in food availability to catch rates and changes in the size structure of the catch related to freshwater flow would also assist. The second causal mechanism requires no positive growth or survival response. Increased catches should be based around increased abundance of individuals with similar size structures.

In order to examine possible causes for a catch rate/flow correlation, a limited analysis of the age structure of the catch was conducted.

Methods

A random sample of 257 whiting frames was collected from the catch of Wanderers Fishing Club at Cape Capricorn. The length (cm, length to caudal fork (FL)) and sex of the fish were recorded and the otoliths removed from the heads. The otoliths were aged by the staff of the Department of Primary Industries and Fisheries at the Southern Fisheries Centre Deception Bay (see Chapter 6; Staunton-Smith *et al.* 2004).

Estimated age and measured length were compared and the catch curve of the percentage frequency of fish at age used to illustrate mortality. A model was constructed using least squares regression between log(e) transformed percentage frequency and age. From this, the residuals between the fitted and observed values were calculated and compared with freshwater flow data from the Fitzroy River for the back-calculated year of spawning i.e. year of sampling = predicted age (see Chapter 6; Staunton-Smith *et al.* 2004).

Results

The relationship between estimated age and fish length is illustrated in Figure 5.5. This plot shows that 87% of sand whiting captured were between 210 and 250 mm FL. This is clustered around the legal minimum length of 230 mm total length (= approximately 210 mm FL). In fact, the largest size class of fish was between 210 and 220 mm FL (30.4% of fish) and 55.6% of fish were less than 230 mm FL (Figure 5.4). This indicates that the number of fish just over legal length particularly affects the catch rates.

Figure 5.4 Size distribution of sand whiting within the sample of fish from the catch of the Wanderers Fishing Club

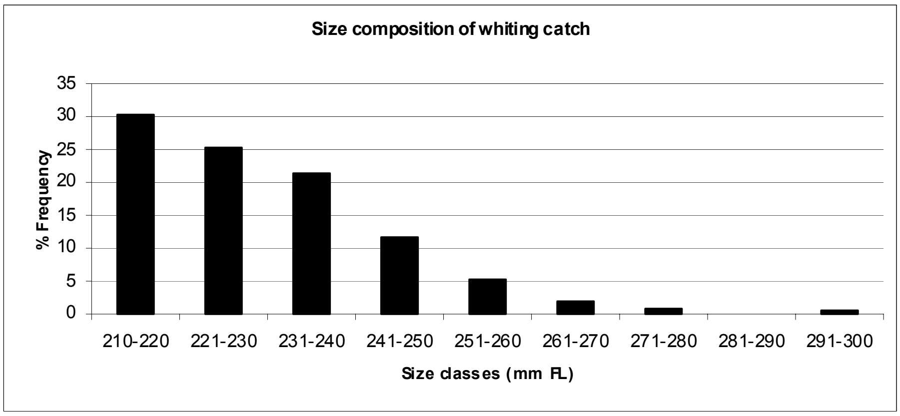
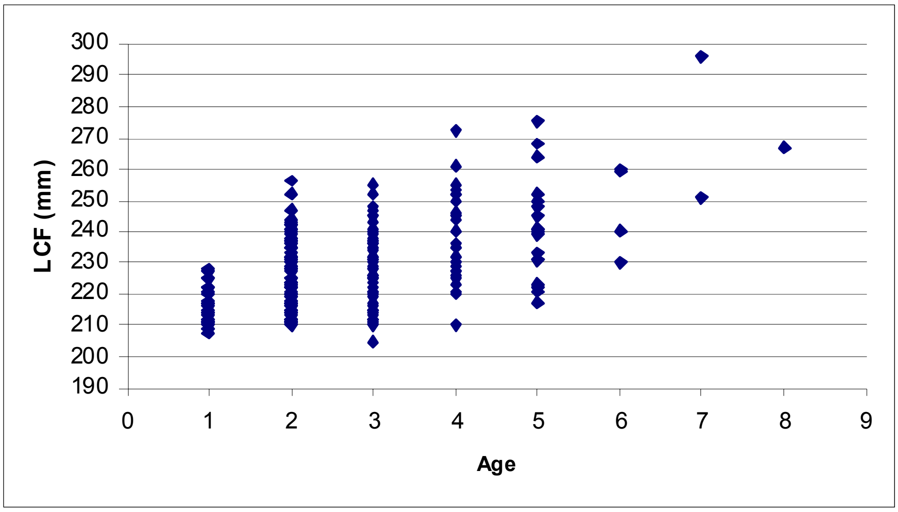


Figure 5.5 Age/length distribution of sand whiting within the sample of fish from the catch of the Wanderers Fishing Club. Age in years, size in mm length to caudal fork



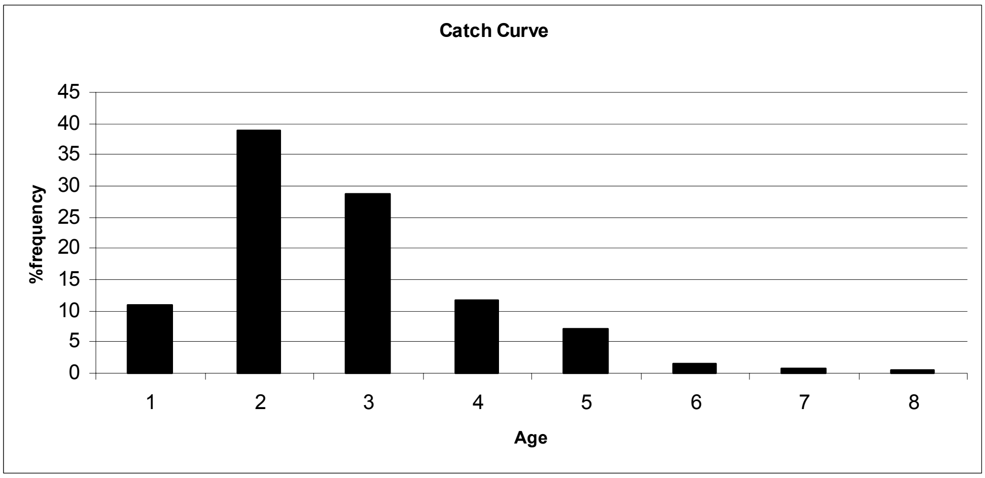
Growth is extremely variable amongst individual, with sand whiting between 210 and 220 mm FL being between one and four years old. Sand whiting less than 250 mm FL were up to seven years old (Figure 5.5). There was also considerable variability in size of sand whiting between the 25th and 75th percentile of length in each age class (**Table** 5.4). This suggests that a change in the percentage of fish that have fast growth (the 75th percentile) could significantly increase the number of fish above a particular size (for example the minimum legal length).

Table 5.4 Percentiles (25th, 50th, 75th) of lengths-at-age (in mm) of sand whiting within the sample of fish from the catch of the Wanderers Fishing Club

	Age 1	Age 2	Age 3	Age 4	Age 5
25th	211.00	215.25	215.00	227.50	231.00
50th (=Median)	214.00	223.00	226.00	235.50	240.50
75th	220.25	232.00	236.00	249.00	249.50
Difference between 75th & 25th	9.25	16.75	21.00	21.50	18.500

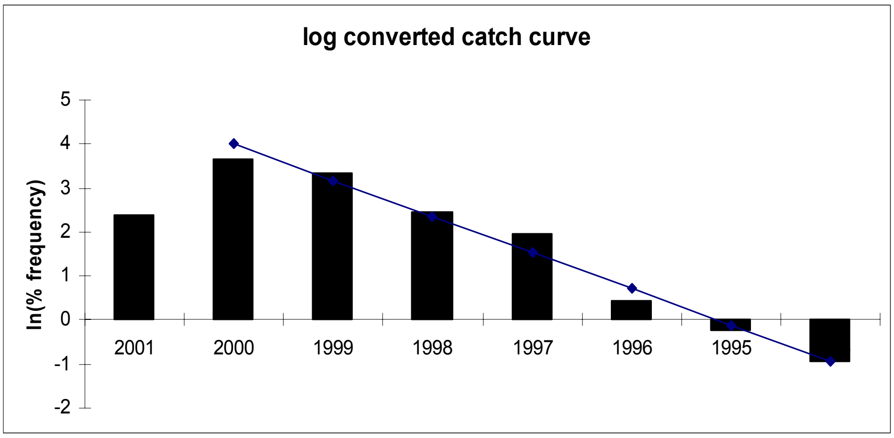
According to plots of the catch curve (i.e. percent frequency of fish in each age class), age-1 whiting were not fully susceptible to capture (Figure 5.6); that is, only the faster growing fish in the age-1 class are taken. This means that if a group of fish in age-1 class grew faster, then more fish would be available to the fishery.

Figure 5.6 Catch curve (percent frequency at age) of sand whiting within the sample of fish from the catch of the Wanderers Fishing Club



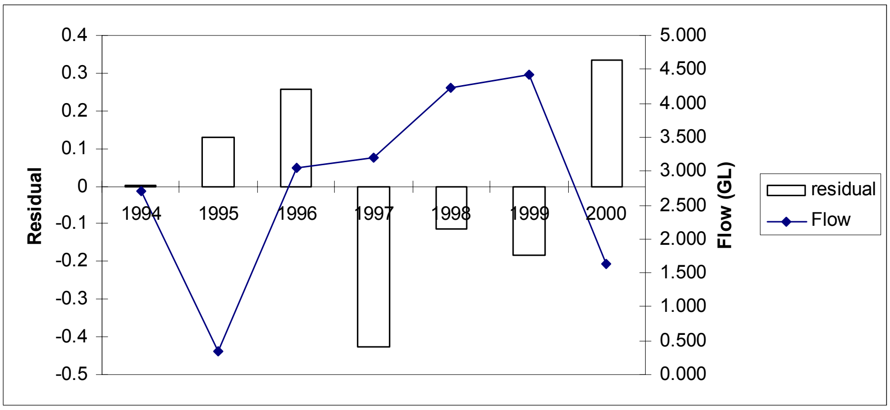
A fitted mortality curve for the Cape Capricorn fish was calculated using the log(e) transformed percent frequency of fish from two to eight years old i.e. those fish of size-classes that were fully recruited to the recreational fishery (Figure 5.7). This shows that both positive and negative residuals exist between observed and predicted log(e) transformed percent frequency.

Figure 5.7 Log transformed catch curve (Log(e) percent frequency at estimated year of spawning) of sand whiting within the sample of fish from the catch of the Wanderers Club Straight line was fitted by least squares regression (R^2= 0.98).



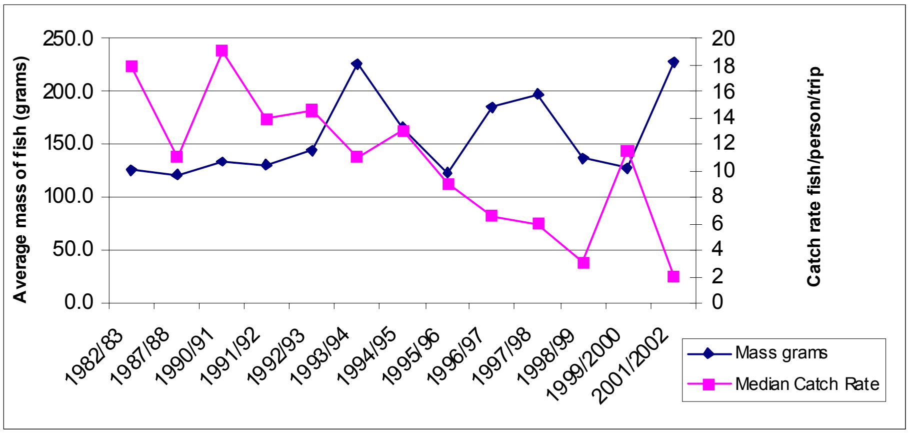
A comparison between these residuals (modelled versus observed) and the measured freshwater flow of the Fitzroy River in the predicted year of spawning shows a correlation of 0.56 (Figure 5.8). That is, the number of fish in an age-class where large freshwater flows occurred in the year of spawning tended to be greater than the predicted number, and the number of fish in an age-class spawned in years of low freshwater flow were smaller than predicted (Figure 5.8). This supports the view that recruitment of sand whiting was higher in age-classes spawned in years of high freshwater flow.

Figure 5.8 Residuals between modelled and observed log-transformed percent-frequency plotted with flow (GL) of the Fitzroy River (y axis) for the estimated year of spawning (x axis)



A comparison of the catch rate of the Wanderers Fishing Club and the average mass of the fish taken (Figure 5.9) shows that average size (weight) of fish tends to decrease in years where catch rates increase (r = -0.512).

Figure 5.9 Median catch and average mass of fish caught by the Wanderers Club in Gladstone Harbour. Note the data are not continuous except between 1990/91 and 2001/02



Discussion

The data support the hypothesis that: (i) catch rates could increase rapidly enough to demonstrate a response to increased freshwater flow within one year and that changes in catch rate are influenced by growth, survival and recruitment responses in fish close to or at the minimum legal length; and (ii) increases in freshwater flow may be associated with increased food availability and this may result in increased recruitment. These issues are considered in turn.

Catch rates could increase rapidly enough to demonstrate a response to increased freshwater flow within one year. The rates are influenced by increasing growth, survival and recruitment responses.

The sand whiting caught during this study were predominantly just over legal size. This means that a small increase in length amongst fish below legal size could significantly affect catch rates. Similarly the highly variable growth rates of individual fish indicate that a change in the growth patterns of fish is certainly biologically possible. The catch curve shows that fish of age-1 can be found within the fishery but were under represented. This evidence in combination suggests that an increase in growth of age-1 fish could increase the numbers of fish available to the fishery significantly and within one year.

Another piece of supporting evidence can be seen from the average size of fish in the catch. Figure 5.9 shows that average size (weight) tends to decrease as catch rate increases. That is, it is likely that more small fish are caught in years where catch rate increases. This data is also consistent with higher catches occurring in times when greater numbers of just legal fish are available. This evidence is circumstantial; data from a number of years across several flow scenarios would be needed to further test the hypothesis. However all data available supports the working hypothesis.

The data do not entirely support the alternative hypothesis that the increase in catch rate is related to fish being translocated and becoming more available to the fishery (see Loneragan and Bunn 1999). If this mechanism were to hold, then a change in the average size of fish would not be expected. However, the data available cannot dismiss this mechanism and in fact it may be that both mechanisms operate. More information is needed to clarify the issue.

Increases in freshwater flow may be associated with increased food availability and this may result in increased recruitment.

The data (for example Figure 5.2 and Figure 5.3) suggest that freshwater flows above a threshold value increase the numbers of fish caught. The discussion above examines the evidence that this increase in numbers could be related to an increase in growth rate of fish close to the legal minimum size. That is increased freshwater flow causes increased growth and availability of legal sized fish. There is also some information to support the view that this is related to food availability. Sand whiting are largely second order consumers (Burchmore *et al.* 1988) feeding on benthic macro-invertebrates. Currie and Small (2005) and Platten (2005) demonstrate that the abundance of macro-benthos is closely related to catch rates in Gladstone Harbour. It is possible that the response in catch rates is reflecting changes in food availability. Currie and Small (2005) propose that changes in nutrient supply are associated with changes in macro-benthos abundance. Thus, it is likely that increased nutrient supply associated with freshwater flows could also positively influence both macro-benthos abundance and recreational fishing club catch rates and requires further research.

Discussion

To establish a link between freshwater flow and the catch of whiting is important because whiting are the most frequently captured recreational fishing species in Queensland, Australia (Higgs 1998; Henry and Lyle 2003). In addition, the findings of this chapter add to the knowledge relating to the influence of freshwater flow on fisheries in several ways.

There is strong evidence that freshwater flows positively influence the line fishing catch of whiting in both Gladstone Harbour and at Cape Capricorn. Demonstrating the relationship between whiting catches and freshwater flow adds another species at another trophic level to the body of evidence that demonstrates the positive influence of freshwater flows on fisheries.

The findings are also the first to demonstrate that at a particular site, a freshwater flow threshold may be required before a defined flow/catch correlation exists. This evidence is important because it offers one explanation as to why a clear flow/catch relationship may not be obvious. For example, Gartside *et al.* (1999) found no correlation between recreational catch rates and river floods, but their data was from a broad area and only an overall correlation between flood volumes and catches was sought i.e. no analysis using a technique likely to demonstrate threshold effects was conducted. It is suggested that an analysis across a broad range of flow regimes over a long time frame is necessary to clearly establish any freshwater flow/catch relationship. An approach that seeks possible threshold freshwater flow values is also warranted. Similarly, the location of fishing appears to influence any freshwater flow/catch correlation. It seems likely that data from a closely defined fishing location will provide the best opportunity to establish a definite correlation.

The chapter also points to another likely causal mechanism to explain why catch rates could be influenced by freshwater flow. Unlike several other fisheries, the response to freshwater flow seems relatively immediate. That is, the strongest correlation exists with freshwater flow of the same year. This does not mean there is no lagged effect, but that the lag period is less than one year. This fits with the biology of the species (i.e. sand whiting) and the characteristics of the recreational club fishery. Catch rates are particularly influenced by the numbers of fish just over legal minimum size. This combined with the variability in growth rates provides a likely explanation for why the catch rate could respond in the way observed. There is strong evidence to suggest that the catch response is related to increased recruitment of whiting related to more fish just above legal minimum size becoming available. There is also evidence to suggest that this may be related to food availability and associated changes in growth rate. The view proposed by Robins *et al.* (2005) that it is important to examine the biology of fish species and fishery characteristics to gain an understanding as to why a correlation exists is supported.

The information examined supports the view that the freshwater flow/catch relationship for sand whiting is influenced by the number of fish caught, rather than an increase in the size of fish. This aspect is not always obvious from commercial catch data where catch records are based on the weight of landed product rather than the numbers of fish. It is possible that a positive influence of freshwater flow on the numbers of fish caught may be masked within records that only record landed weights since catch weight may not increase (or even decrease) in response to more small fish being captured. This could be a further factor masking a freshwater flow/catch correlation. It is suggested that wherever possible both landed weight and the numbers of fish caught should be analysed.

The implications of the freshwater flow/catch correlation may extend well beyond a positive influence on catch rates. The close relationship between macro-benthos abundance and catch rates suggests that catch trends could be indicative of broader ecological productivity changes (see Platten 2004, 2005). Whiting are likely to be integral to a range of trophic inter-relationships. If this is so, then freshwater flows may influence the productivity of most of the estuarine ecosystem. This is of particular significance given both the predicted changes in rainfall patterns related to climate change and the increasing regulation of streams. This aspect is deserving of much greater research effort.

Chapter 6. Using age-structure of commercial catch to investigate the importance of freshwater flows in maintaining barramundi and king threadfin populations

I. Halliday, J. Staunton-Smith, J. Robins, D. Mayer and M. Sellin [6]

Summary

The age-structures of the commercial catch of barramundi (*Lates calcarifer*) and king threadfin (*Polydactylus macrochir*) were examined over five consecutive years in a dry tropical estuary and used to estimate the year-class strength of each species. Variation in year-class strength was compared with variability in freshwater flowing to the estuary. The year-class strength of both barramundi (an opportunistically catadromous species) and king threadfin (a non-diadromous estuarine species) fluctuated and was significantly and positively correlated with freshwater flow and coastal rainfall in spring and summer. All sub-sets general linear models were used to screen potential relationships between year-class strength of each species and freshwater variables. Several alternate models were identified that explained ~85 to 90% of the variation in the abundance of age classes of barramundi and ~76 to 80% of the variation in the abundance of age classes of king threadfin.

The survival of young-of the year barramundi may be enhanced by freshwater flows by: (a) enhancing the access of larvae, post-larvae and juvenile barramundi to suitable nursery habitats in the estuary, such as temporary supra-littoral pools; (b) enabling a proportion of juvenile barramundi to access freshwater habitats intermittently linked to the estuary; and (c) increasing the productivity of the estuary (i.e. increased prey species abundance), with increased growth rates of post-larval and very young barramundi increasing the number of individuals that survive the first year of life.

We hypothesised that freshwater flows may influence the survival of larval and/or juvenile king threadfin in estuaries leading to increased year-class strength by: (a) enhancing the biological productivity of the estuary, thereby increasing prey species abundance resulting in improved growth rates of king threadfin; (b) affecting the area of favourable habitat through larger areas of decreased salinity, with lowered salinities affecting the energy budgets; and/or (c) creating turbid conditions reducing predation on juvenile king threadfin.

Freshwater flows in spring and summer are important drivers of the year-class strength of at least two important estuarine finfish species, and reduction in these flows, through the development of water infrastructure and abstraction or long-term climate change, will potentially reduce the size of the population of estuarine fish available for human harvest. In addition, these results suggest that stock assessments should consider the impacts of freshwater flows and or coastal rainfall on the annual recruitment of barramundi and king threadfin.

Introduction

The use of freshwater resources (e.g. for cities, industry and agriculture) has altered the magnitude, duration and timing of freshwater flowing into estuaries and impacted estuarine species (Drinkwater and Frank 1994; Gillanders and Kingsford 2002). Sustainable management of freshwater requires an understanding of the role (or importance) of freshwater flows in downstream biological processes, and how changing natural river flows impacts uponestuarine populations. A number of estuarine species are exploited by commercial, recreational or subsistence fisheries, and the management of these fisheries would also benefit from an

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[6] Parts of this chapter are published in: Staunton-Smith, J., Robins J.B., Mayer, D.G., Sellin, M.J., Halliday, I.A. (2004). Does the quantity and timing of freshwater flowing into a dry tropical estuary affect year-class strength of barramundi (*Lates calcarifer*). *Marine and Freshwater Research*, 55: 787-797.

understanding of factors other than fishing effort (i.e. environmental factors) that influence the population size (Shepard *et al.* 1984; Hilborn and Waters 1992).

The link between freshwater flows and estuarine organisms has often been investigated by comparing catch data from commercial fisheries with patterns in naturally variable, or highly altered, freshwater flows (e.g. Sutcliffe *et al.* 1977; Lloret *et al.* 2001). Significant covariation between freshwater flow and catch has been reported for numerous marine and estuarine species, often with time lags equalling the approximate age at which a species enters the fishery (e.g. Lloret *et al.* 2001; Quiñones and Montes 2001). This pattern has been used to generate or support hypotheses that freshwater flows influence the spawning, survival and growth of fish during their first year of life (Drinkwater and Frank 1994). Although some studies have undertaken further investigation into the variation in abundance and distribution of early life history stages (e.g. North and Houde 2003), few have been able to confirm the speculated causal mechanisms.

Through conceptual models and review of life history's (see Chapter 2), we identified that freshwater flows may affect the recruitment of estuarine fish species. There are a number of mechanisms by which enhanced recruitment may occur:

1) Transporting eggs and larvae away from the estuary (negative effect);
2) Creating chemical signals for larvae to enter the estuary and locate nursery habitats (positive effect);
3) Enabling post-larvae and small juveniles to move into supra-littoral nursery habitats; and
4) Enabling large juveniles to migrate into freshwater habitats intermittently linked to the estuary.

If freshwater flows do affect recruitment by influencing the survival of young fish, then year-class strength (YCS) should vary with freshwater flow and persist through time in order to affect the subsequent abundance of adult fish and associated commercial catch. Studies of YCS frequently quantify the abundance of specific young age-classes on an annual basis (e.g. Helle *et al.* 2000; DiCenzo and Duval 2002; Sutela *et al.* 2002). However, strong and weak year-classes can persist through time and are often detected in the age-structure of adult populations of freshwater and marine species (e.g. Mills and Mann 1985; Maceina 1997; Morison *et al.* 1998; McGlennon *et al.* 2000; DiCenzo and Duval 2002). Therefore, examining the age-structure of an adult population of fish provides an opportunity to examine the relative strength of a number of consecutive year-classes, especially for long-lived species. Such an approach can be completed with relatively few years of sampling compared with surveys of early life history stages (e.g. 0+ or 1+ age-classes), which yield one estimate of year-class strength per year, and therefore require many years of sampling.

The aim this chapter was to determine whether there was evidence of variation in YCS in the adult population of long-lived estuarine fish species for which large scale variability in YCS has not yet been demonstrated, and if so, whether the variation could be related to freshwater flowing into the estuary. Life history review identified two estuarine fish species, barramundi (*Lates calcarifer*) and king threadfin (*Polydactylus macrochir*), whose YCS may be influenced by freshwater flow. We examined the age-structure of the adult population because of the time efficiency of this approach, and because the scale of the estuarine commercial fishery was sufficient to sample the adult populations adequately, thus negating the need to kill large numbers of fish during fishery-independent surveys.

The specific objectives of the study were to:

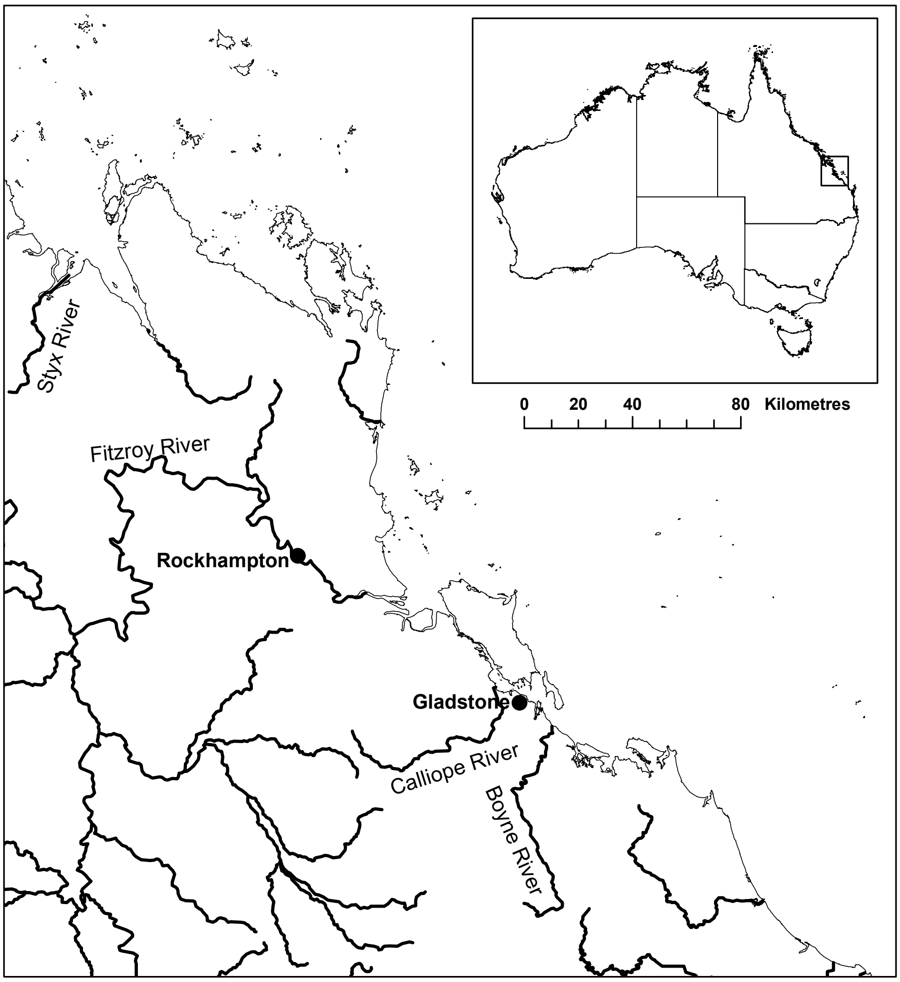
1) Estimate the age-structure of barramundi and king threadfin populations in the Fitzroy River region using samples from commercial catches;
2) Estimate YCS for barramundi and king threadfin populations;
3) Investigate whether variation in YCS of each species was related to patterns in the freshwater flowing into the estuary; and
4) Propose possible causal mechanisms for the observed relationships.

Methods

Sampling of commercial catch

Sampling of the commercial catch was scheduled to coincide with peaks in commercial fishing effort that occur in the weeks preceding and following the annual seasonal closure for barramundi (i.e. 1 November to 31 January). The commercial catch was sampled for five consecutive 'sampling years', with year-1 = 2000/01, year-2 = 2001/02, year-3 = 2002/03, year-4 = 2003/04 and year-5 = 2004/05. We sampled twice each 'sampling year', once in the week(s) preceding the seasonal fishing closure (i.e. October) and then again in the week(s) after the opening of the fishery (i.e. February). Sampling was concentrated at three seafood processors (two in Gladstone and one in Rockhampton, **Figure 6.1**), as these seafood wholesalers had previously been identified as consistently purchasing most of the estuarine fish caught by commercial fishers in the Fitzroy River region. Where possible, barramundi and king threadfin purchased by the processors during the sampling period were measured (total length (TL) for barramundi and caudal fork length (FL) for king threadfin) and otoliths were removed from as many of these fish as time permitted. We also identified approximately where each fish was caught, by questioning the processors or fishers. Sex of fish was not recorded, as the fish were 'cleaned' (i.e. gills and viscera removed) prior to arriving at the seafood processors.

Figure 6.1 Locality map of the Fitzroy River estuary (downstream of the city of Rockhampton) in central Queensland, Australia



Otoliths were blocked in resin, then sectioned at 300 μm. Sections were viewed using a microscope and reflected light. In general, there was very clear differentiation between the slow (narrow, opaque, light) and fast (broad, translucent, dark) growth zones in barramundi otoliths from the Fitzroy River region (Stuart and McKillup 2002). Distinct increments probably occur because feeding and growth rates slow down greatly during the colder months in the Fitzroy River region, which is near the southern extremity of the distribution of barramundi and king threadfin in eastern Australia. Counts of the narrow, opaque increments were validated as having been laid down yearly and were counted to determine the fish age. However, an extra year was added to fish caught in October if an increment was not visible on the otolith's edge (i.e. if they had a wide marginal increment). The justification for this procedure was that preliminary marginal increment analysis suggested increments are laid (or become visible increments around the margin) in or around October (see Appendix 6). Therefore, when an increment was not visible on the otolith's edge in October, it was assumed that this was a visual artefact due to the curved nature of the otolith.

Once ages were estimated, age-length keys were constructed and used to convert length-frequencies into age-frequencies. Age-length keys and length-frequency distributions were constructed for each sampling trip (i.e. two trips per sampling year). A single age-structure was constructed for each sampling year. Year-classes were assigned on the basis of spawning year, when spawning occurs over spring and summer, with an assumed 1 January birthday (i.e. fish born in November 1990 and February 1991 are in the 1991 year-class).

Only barramundi from three- to eleven-years-old were included in the analysis as it was considered that these best represented the year-classes most effectively sampled by the net size restrictions in the commercial fishery (see Staunton-Smith *et al.* 2004; or Appendix 6 for details). For similar reasons, only king threadfin from two- to eleven-years-old were included in the analyses.

Estimating year-class strength

We used the method described by Maceina (1997) to estimate year-class strength (YCS) objectively from population age-structure using catch-curve regressions (i.e. regression of the natural log of the number of fish in each year-class against age). Deviation from an expected abundance of each year-class, given its age and the catch-curve regression equation, is assumed to reflect variable recruitment. Therefore, residuals from the catch-curve regressions are indices of YCS, with large positive and negative residuals representing strong and weak year-classes respectively.

Correlating year-class strength with environmental variables

We investigated the relationship between YCS and freshwater flowing into the estuary by: i) correlation analyses of YCS and freshwater flow variables (river flow and coastal rainfall, plus stocking); and ii) all sub-sets general linear modelling (GenStat 2005) with year-class strength as the response, and age, sample year, freshwater flow variables and stocking as independent variables. Age was forced into the model, as was sampling year, because the abundance of individual age-classes is not comparable between years. Significant models with three additional independent terms were reported. We examined serial auto-correlation of regression residuals using residual maximum likelihood (REML, GenStat 2005), to investigate whether there was evidence that recruitment success was auto-correlated (e.g. due to stock-recruitment relationships). The results showed no significant auto-correlations, thus standard general linear modeling methods were used. The general linear models were used for data exploration and model screening. Ridge regression (GenStat 2005) was used to account for any lack of independence between freshwater flow variables (i.e. river flow and coastal rainfall) in the final models. This statistical method identifies and adjusts for observed levels of collinearity, and provides adjusted regression coefficients that are the expected values had the X-variables been independent.

We used two variables as indices of freshwater flowing into the estuary: river flow (i.e. water flowing down the Fitzroy River and through the barrage) and coastal rainfall (i.e. rainfall in the coastal catchment of the estuary and in catchments of creeks entering the river below the barrage). River flow and coastal rainfall were expressed as seasonal totals (i.e. total flow for the Fitzroy River and total rainfall, averaged across stations within the coastal region of the Fitzroy River estuary). We obtained river flow data from the Department of Natural Resources and Water, Queensland, and rainfall data from *Rainman StreamFlow 4.3* (Clewett *et al.* 2003). River flow equalled gauged flow at the most downstream gauging station (i.e. at 'The Gap', 142.1 km Adopted Middle Thread Distance), minus the estimated downstream extraction. Seasons are defined as: spring (September to November), summer (December to February), autumn (March to May) and winter (June to August). We also included a 'spawning season' aggregate of flow or rainfall, which is the sum of flow or rainfall during spring and summer i.e. September to February inclusive. Fish stocking events were factored into analyses using the total number of barramundi fingerlings stocked in the Fitzroy River catchment per year between September and the following August. These totals did not include fish stocked into upstream impoundments that had not overflowed since being stocked. Analyses of king threadfin data did not include stocking, as this species is not stocked in the Fitzroy River. River flow, rainfall and stocking data were transformed ($\log_{10}$+1) to normalise data and stabilise variances.

We made the following assumptions in our analysis:

1) Age-structure is determined mainly by recruitment;
2) Migration rates between estuaries were low;
3) Fish stocking does not bias the results; and
4) Age-structure of the adult population was estimated accurately.

The details and validity of these assumptions are discussed in Staunton-Smith *et al.* (2004), Halliday *et al.* (submitted) and Appendix 6.

Results

Length-frequencies of samples

A total of 2690 barramundi and 1185 king threadfin from the Fitzroy River region were measured during the five 'sampling years', with 2112 and 716 aged respectively (**Table 6.1**). There were consistent differences between the samples collected in October compared with those of the following February. Differences included the number of fish measured, as well as size-structure. The most notable difference was the large number barramundi <800 mm TL in February that were not present in the October samples (i.e. 78 and 69% of fish measured in February 2001 and 2002 *c.f.* 32 and 35% of fish measured in October 2000 and 2001).

Table 6.1 Number of barramundi and king threadfin caught in the Fitzroy River estuary that were measured and aged

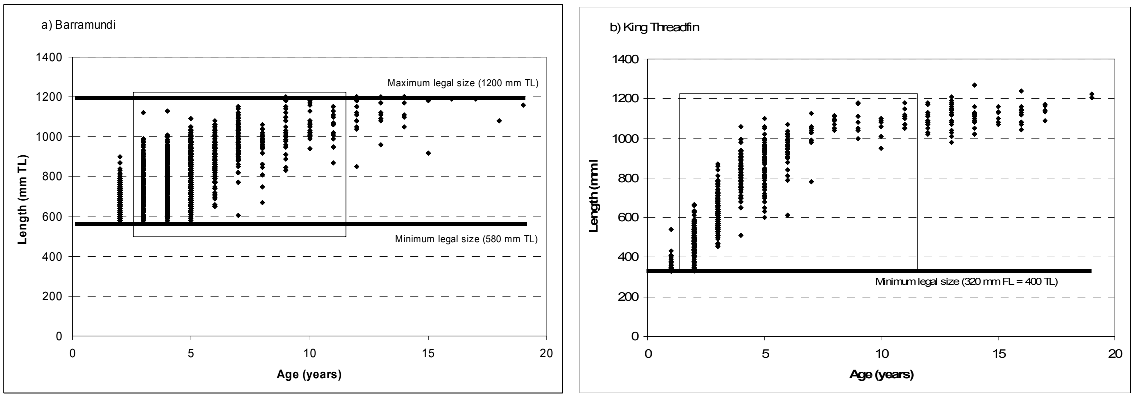
Sample year	Sample trip	Barramundi		King threadfin	
		No. Measured	No. Aged	No. Measured	No. Aged
Year-1	October 2000	155	154	50	47
	February 2001	495	413	157	133
Year-2	October 2001	155	128	168	96
	February 2002	488	197	512	143
Year-3	October 2002	125	124	48	47
	February 2003	480	319	4	4
Year-4	October 2003	236	236	45	45
	February 2004	240	237	8	8
Year-5	October 2004	72	69	83	83
	February 2005	244	235	110	110
	Total	2690	2112	1185	716

The catch of barramundi ranged in age from two-years-old to 32-years-old (two individuals), with 8.3% of the sampled catch aged as two-years-old, 88.4% of the catch aged between three-and eleven-years-old, 2.0% of the catch aged between 12- and 20-years-old and 1.3% of the sampled catch aged as greater than 20-years-old.

The catch of king threadfin ranged in age from one-year-old to 19-years-old (two individuals), with 3.1% of the sampled catch aged as one-year-old, 86.3% of the catch aged between two- and eleven-years-old, and 10.6% of the sampled catch aged between 12- and 19-years-old.

There was large variation in the length-at-age of both barramundi and king threadfin (Figure 6.2). For example, three-year-old barramundi may range in length range from 580 mm to 1000 mm TL. Three-year-old king threadfin may range in length from 500 mm to 850 mm FL. This shows that fish length is not a reliable indicator of fish age for barramundi or king threadfin. Similarly, a 1000 mm TL barramundi can be between three- and 13-years-old, whilst a 1000 mm FL king threadfin can be between four- and 16-years-old.

Figure 6.2 Length-at-age plots for a) barramundi and b) king threadfin sampled from the Fitzroy River estuary

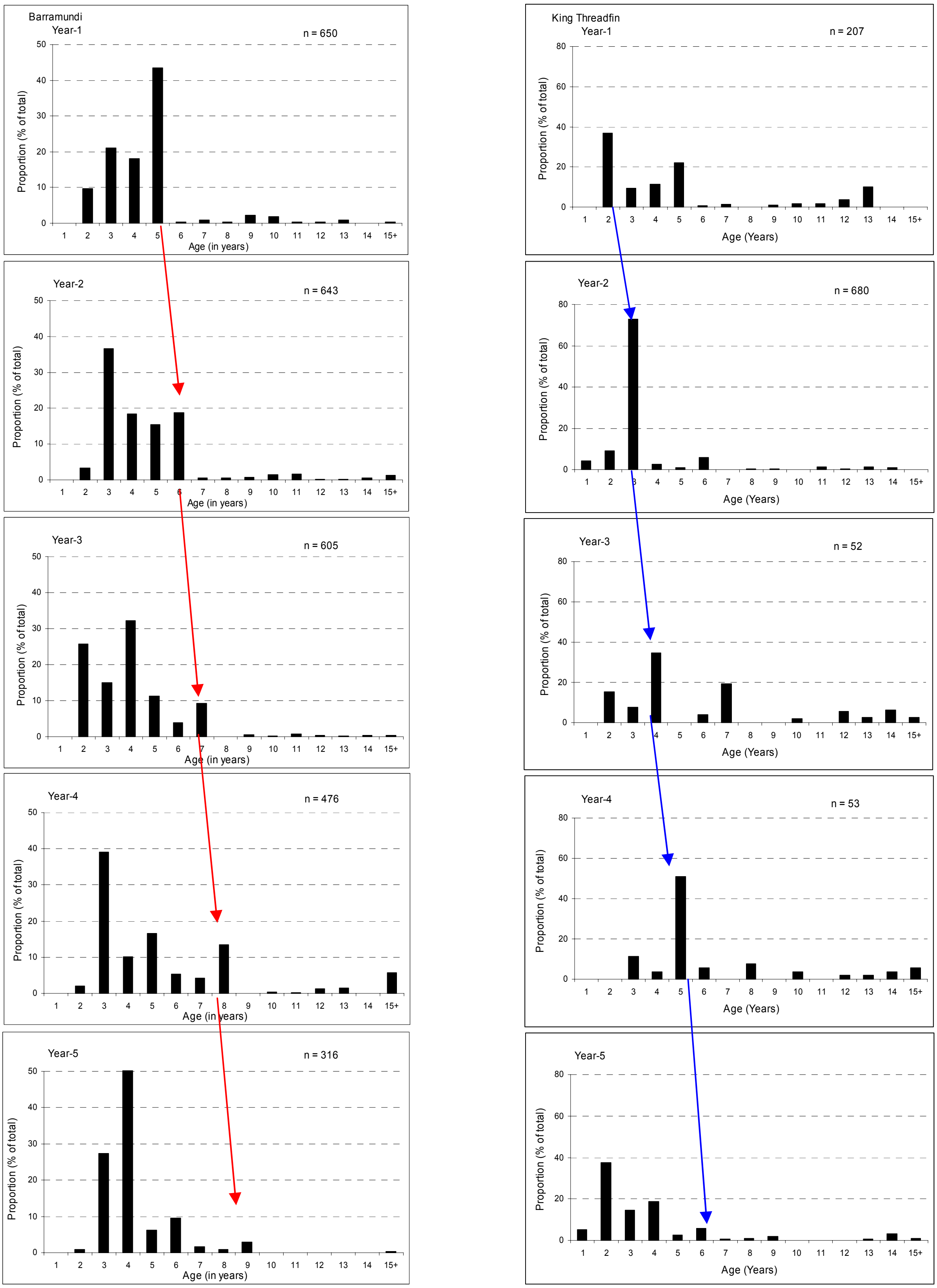


Boxed area shows data used in the analyses of year-class strength (ages three- to 11-years-old for barramundi and two- to 11-years-old for king threadfin).

Age-structure

There was a systematic change in the age-structure of samples from year-1 to year-5 (**Figure 6.3**). In year-1, the five-year-old age-class was 'strong' for both species and could be followed in the yearly age structures over the five sample years. The two-year-old age-class for king threadfin was also strong and can be followed through the five sample years, becoming the six-year-old age-class in sample year-5 (**Figure 6.3**). For barramundi the six-, seven- and eight-year-old age-classes were 'weak' in year-1 and this pattern persisted throughout the sampling (**Figure 6.3**).

Figure 6.3 Proportional age-structures for a) barramundi and b) king threadfin collected from the Fitzroy River estuary for five consecutive years. Year-1: October 2000 + February 2001, Year-2: October 2001 + February 2002, Year-3: October 2002 + February 2003, Year-4: October 2003 + February 2004, and Year-5: October 2004 + February 2005.



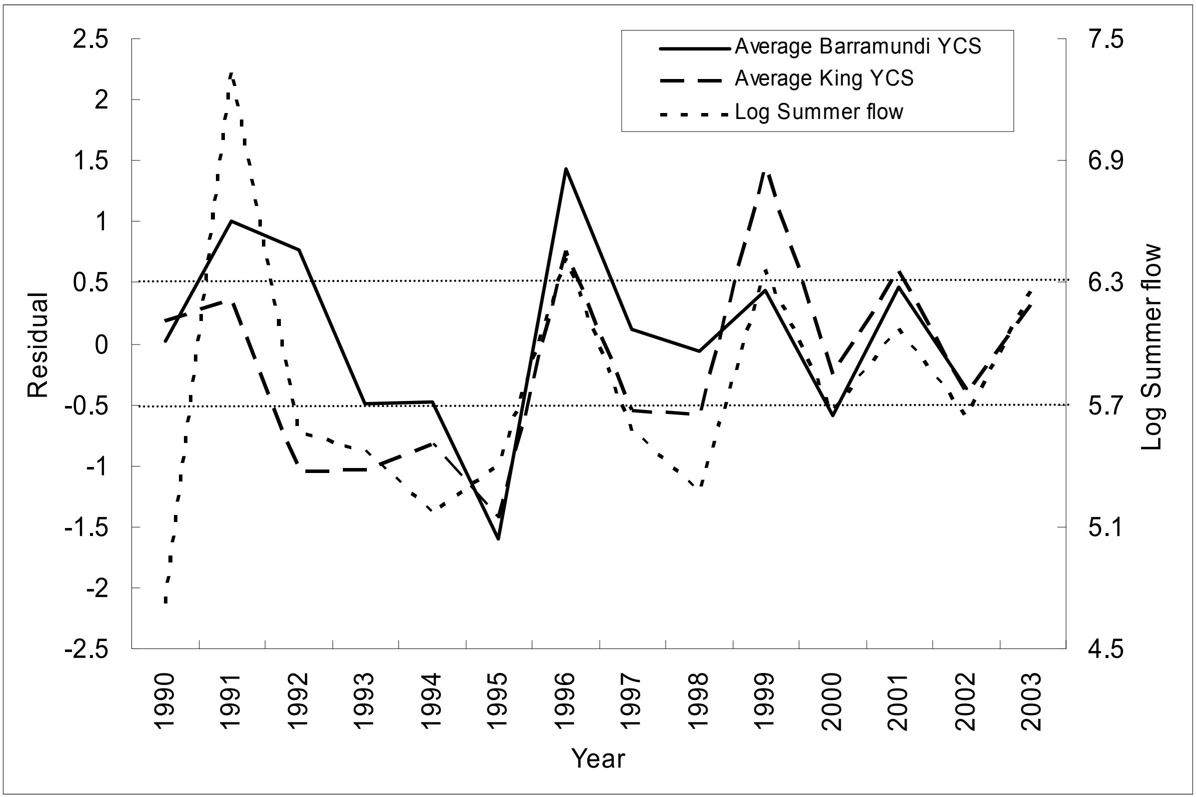
Using residuals from the catch-curve regression

The standardised residuals from the catch-curve regressions give an indication of relative year-class strength (YCS).

The barramundi year-classes 'born' in 1991, 1992 and 1996 had large, positive residuals, indicating strong recruitment, whilst those 'born' in 1995 and 2000 had large, negative residuals, indicative of weak recruitment (Figure 6.4). The 2001 year-class had only two data points (and thus estimates of YCS), both of which were positive, but only one could be classed as indicative of strong recruitment i.e. a catch curve residual of 0.86 from the year-5 sample . Of the remaining year-classes, 1990, 1997 and 1999 had small, positive residuals (i.e. 0.0 to 0.5) while 1993, 1994, 1998 and 2002 had small, negative residuals (i.e. -0.5 to 0.0) and thus could not be classified as either 'strong' or 'weak' (Figure 6.4).

King threadfin year-classes 'born' in 1996, 1999 and 2001 had large, positive residuals, indicating strong recruitment (**Figure 6.4**), whilst those 'born' in 1992, 1993, 1994, 1995, 1997, 1998 and 2000 had large negative residuals, indicative of weak recruitment.

Figure 6.4 Residuals from catch-curve regressions of barramundi and king threadfin against summer freshwater flows from the Fitzroy River estuary



Correlating environmental variables with year-class strength

River flow and coastal rainfall were significantly correlated in spring, summer and autumn (r= 0.73, 0.78 and 0.63 respectively, $P<0.05$, $n=14$), but not in winter. They were also significantly correlated during the spawning season (i.e. September to February inclusive). These results suggest that seasonal river flow and coastal rainfall in summer, autumn and spawning season should have similar correlations with the measures of YCS. However, this was not always the case.

YCS of barramundi was significantly and positively correlated with freshwater flow and coastal rainfall annually, during summer and the spawning season (i.e. spring + summer), and was significantly and positively correlated with spring flow but not spring rain (Table 6.2). YCS was not significantly correlated with freshwater flow or coastal rain in autumn or winter. Stocking of barramundi fingerlings in the freshwater reaches of the Fitzroy River was significantly correlated with YCS, although a strong year-class occurred when stocking was low (1991).

YCS of king threadfin was significantly and positively correlated with freshwater flow and coastal rainfall for annual totals, during spring and summer, as well as with the spawning season aggregate (Table 6.2). YCS of king threadfin was not significantly correlated with freshwater flow or coastal rainfall in autumn or winter. Correlation coefficients tended to be at least equal or of greater value for freshwater flow variables than for rainfall variables.

Table 6.2 Correlation coefficients (r) between estimated year-class strength (YCS) and freshwater flow and rainfall variables for barramundi and king threadfin in the Fitzroy River estuary

	Barramundi YCS	King threadfin YCS
Annual flow	**0.55*** **	**0.60*** **
Annual rain	**0.67*** **	**0.60*** **
Spawning season flow	**0.61*** **	**0.72*** **
Spawning season rain	**0.63*** **	**0.60*** **
Spring flow	**0.66*** **	**0.64*** **
Spring rain	0.17	**0.44*** **
Summer flow	**0.60*** **	**0.61*** **
Summer rain	**0.50** **	**0.32* **
Autumn flow	0.16	0.21
Autumn rain	0.09	-0.06
Winter flow	-0.01	0.03
Winter rain	-0.24	-0.02
Stocking	**0.37* **	n/a

*= P<0.05, ** = P<0.01, *** =P<0.001; n/a = not applicable for king threadfin i.e. no stocking

All sub-sets general linear modelling identified several alternate models that explained ~85 to 90% of the variation in the abundance of age classes of barramundi and around 75% of the variation in the abundance of age classes of king threadfin (Table 6.3).

For barramundi, the base model of age and sample year (i.e. forced variables) explained 62.0% of the variation in the abundance of age classes. This is slightly greater that reported in Staunton-Smith *et al.* (2004) and is a consequence of an additional two years of data (i.e. sample year-4 and year-5). The additional terms of summer river flow, stocking and autumn rainfall (all positive) was the ‘best’ model (adjusted R^2 = 90.4%) and is consistent with that of Staunton-Smith *et al.* (2004). The other ‘best’ model reported by Staunton-Smith *et al.* (2004; i.e. summer rain, spring flow and winter flow, overall adjusted R^2 = 85.9%), was not significantly better for the five years of data than the two term model of summer rain and spring flow (overall adjusted R^2 = 86.3%). This was consequence of the winter flow term not significantly improving the two term model. Other models with only two environmental terms also explained a high degree of the variation in age class abundance, with all terms in the models being positive in direction (Table 6.3). The models explained between 73.7% and 62.9% of the variation in abundance of barramundi age classes not explained by age and sample year alone. Flow terms appeared in all four of these ‘best’ all sub-set models, whilst rainfall terms appeared in three of the four models. Spring and summer flow or rainfall was included in all ‘best’ models.

Interestingly, stocking of barramundi in the freshwater reaches of the Fitzroy River system influences YCS as well as the estuarine catch (Robins *et al.* 2005). Mature barramundi that were stocked as fingerling move from freshwater reaches with flows to estuarine waters (unpulished tag-recapture information, Australian National Sportfing Association) and join the breeding population. The impact of stocked fish on the abundance, genetic diversity and sustainability of the commercial fishery with respect to stocked vs. wild recruits is poorly understood and requires further examination.

Table 6.3 Best all sub-set regression models for the abundance of age-classes of barramundi and king threadfin sampled from the Fitzroy River estuary

Species	Regression model[A]	Percent variance accounted for (adjusted R^2)
Barramundi	Age, sample year, summer flow, stocking, autumn rain	90.4
	Age, sample year, summer flow, stocking	88.2
	Age, sample year, spring flow, summer flow	86.3
	Age, sample year, spring flow, summer rain	85.9
King threadfin	Age, sample year, summer flow, spring flow	76.4
	Age, sample year, summer flow, spring rain	73.3

[A] Factors in the multiple regression are positively related to age class abundance unless otherwise indicated.

The base model of age and sample year (i.e. forced variables) explained 54.6% of the variation in abundance of year-classes of king threadfin. The best final two term model of summer flow and spring flow (R^2 = 74.6%) explained 44% of the residual variation from the base model. The only other significant two term model included summer flow and spring rain (R^2 = 73.3%) explained 42% of the residual variation from the base model. The relative influence of each term (summer flow and spring flow) on YCS was determined from the estimates in the 'table of effects' from the REML analysis (i.e. summer flow = 0.78, spring flow = 0.21; multiplied by the flow range). This indicated that summer flow as having an influence of approximately double that of spring flow on the strength of any given year-class of king threadfin. Significant positive serial auto-correlation of AR1 (i.e. a one year lag) was detected (REML, GenStat 2005) indicating that for king threadfin there was an increased likelihood of strong year-classes following strong year-classes and weak year-classes following weak year classes. Coefficients adjusted for AR1 were not significantly different than those derived from the standard GLM. Ridge regression correlations amongst the independent variables were not pronounced averaging 0.26 with a maximum of 0.36. These values indicated a low adjustment factor (k-coefficient) for the ridge regressions and in these ranges the ridge traces (i.e. the coefficient values for the independent variables) were stable. Therefore, the low degree of correlations amongst the independent variables was dismissed and standard general linear models adopted throughout. Significant correlation of the YCS of king threadfin and barramundi (r = 0.57, P <0.05) indicated that freshwater flows influenced the YCS of both species in a similar way.

Discussion

Using commercial catches to estimate the age-structure of the fish populations

Sampling commercial catches of barramundi and king threadfin via carefully selected seafood processors provided a reasonable estimate of the age-structure of the populations of these species in the Fitzroy River region. This approach was cost and time efficient, and allowed project staff to build a strong rapport with local seafood stakeholders (including commercial fishers). The value of this approach for the central Queensland region was recognised by the Long Term Monitoring Program of the DPI&F, who have adopted a sampling program for barramundi based on this approach (i.e. via seafood processors), the information from which is used in the ongoing stock assessment of barramundi.

Estimating year-class strength

By using the method of Maceina (1997) to estimate YCS we have been able to objectively, quickly, and cheaply obtain information on large-scale variability in the recruitment of barramundi and king threadfin in the Fitzroy River region. The success of the method, i.e. using otoliths to estimate age-structure and hindcast year-class strength is dependent on being able to: (i) representatively sample the population; and (ii) confidently estimate the absolute age of individual fish (compared to estimating relative age) so that a birth-year can be assigned; and (iii) repeatedly sample the population for a number of years, with three years being the minimum.

The ability to sample a species adequately is a potential problem when estimating relative rates of annual recruitment, especially if measuring the abundance of a single life history stage. The distribution of small and juvenile barramundi and king threadfin is not known in the Fitzroy River region, and as such would be difficult to sample representatively. In contrast, sampling the adult life history stage of barramundi and king threadfin in the estuary (e.g. from the commercial catch) represents a convenient method for comparing the relative abundance of numerous year-classes (see Staunton-Smith *et al.* 2004).

Otoliths from barramundi and king threadfin sampled from the Fitzroy River estuary had clearly defined opaque and translucent bands (Figure 6.5). In addition, ages of known-age barramundi were estimated from otoliths and used to validate our ability to estimate absolute age (see Staunton-Smith *et al.* 2004). Not only are increments laid annually, but the first increment was also identified accurately, which is often a problem (Morison *et al.* 1998).

Figure 6.5 Transverse sections of otoliths from a) barramundi and b) king threadfin sampled from the commercial catch of the Fitzroy River estuary

Possible causal mechanisms of relationships between YCS and freshwater

In general, long-lived species, which have numerous age-classes in the population, are buffered against vast recruitment-based variation in stock size, unless there is a series of strong or weak year-classes (McGlennon *et al.* 2000). However, the large variability in YCS we observed, and persistence of strong and weak year-classes in the barramundi and king threadfin populations over five consecutive years, suggests that recruitment variability has the potential to influence adult stock size in at least these species.

Variation in year-class strength (YCS), as an indicator of the overall recruitment and survival of juvenile barramundi and king threadfin, was consistently and positively related to the amount of freshwater flowing into the Fitzroy River estuary during spring or summer. We would not expect barramundi and king threadfin to show the same patterns in the strength of recruitment (i.e. year-class strength), as barramundi are more tolerant of (and use) freshwater (habitats) whilst king threadfin do not tolerate (or use) 0 salinity water (or habitats). However, we would expect some overlap in years of strong (or weak) recruitment as both species are carnivorous and would probably exploit any trophic blooms occurring in the estuary proper. Thus, although there may be some overlap in the causal mechanisms between year-class strength and freshwater flow for barramundi and king threadfin, we will consider each species separately.

Barramundi – a diadromous species

The significant correlations between spring and summer freshwater flows and year-class strength supports the suggestion that flows at that time of year affect the abundance and survival of very young barramundi (i.e. juvenile recruitment), which occur in estuaries during spring and summer (Dunstan 1959; Russell and Garrett 1983, 1985; Davis 1985). Life history assessment identified four causal mechanisms by which the recruitment (=YCS) of barramundi could be related to freshwater flow (see Chapter 2). These were:

1) transporting eggs and larvae away from the estuary (negative effect);
2) creating chemical signals for larvae to enter the estuary and locate nursery habitats (positive effect);
3) enabling post-larvae and small juveniles to move into supra-littoral nursery habitats, and
4) enabling large juveniles to migrate into freshwater habitats intermittently linked to the estuary.

Life history assessment also identified a casual mechanism affecting the productivity of the estuary, which through increased food availability, could improve the growth and survival of post-larvae, juveniles, adolescents and adults.

The above mechanisms are not mutually exclusive, and it is unlikely that they represent the only mechanisms by which freshwater flows affect the survival of juvenile barramundi. For example, YCS may be related to the size of the spawning population. This causal mechanism is based on increased egg-production and subsequent numbers of juvenile fish in years when floodwaters release land-locked fish, as proposed by Dunstan (1959). However, most fish migrating from freshwater habitats are likely to be males and it is uncertain whether enough mature females arrive from freshwater habitats to increase egg production greatly.

Our results provide no evidence that freshwater flows transport the eggs and larvae of barramundi away from the estuary (i.e. mechanism (a) above). The remaining causal mechanisms relate to access or attraction to estuarine nursery habitats (i.e. mechanisms (b) and (c) above), access to freshwater habitats (i.e. mechanism (d) above) and enhanced productivity (i.e. a trophic response) within the estuary. As the method reported herein is correlative, the following discussion is speculative and requires further investigation.

Freshwater flows may enhance the survival of barramundi during their first months of life, by enhancing the access of larvae, post-larvae and juveniles to suitable nursery habitats. The larvae and very young juveniles of barramundi have been observed in ephemeral supra-littoral nursery habitats in close proximity to spawning areas, such as coastal swamps and lagoons and supra-littoral pools on salt pans (Dunstan 1959; Moore 1982; Russell and Garrett 1983, 1985; Davis 1985; Griffin 1987). The spatial and temporal extent of many of these nursery habitats is affected by the amount of freshwater runoff, and access to and from them is dependent on seasonally high tides and or freshwater flows (Russell and Garrett 1983; Davis 1985; Griffin 1987). The locations of habitats used by very small barramundi (e.g. <50 mm TL) in the Fitzroy River region are unknown. However, they could include the substantial supra-littoral habitats that occur close to the mouth of the estuary (Dunstan 1959; Long and McKinnon 2002). It is possible that high coastal rainfall and freshwater flows in the Fitzroy River enhance survival of the early life history stages of barramundi by generating, and improving access (e.g. frequency, extent and duration of access routes) to supra-littoral nursery areas, extending the spatial or temporal extent of ephemeral nursery areas, increasing their productivity and carrying capacity (i.e. abundance of prey species), and or increasing their suitability in some other way (e.g. physico-chemical characteristics). However, otolith microchemistry indicates that barramundi are not using freshwater habitats until they are at least three-months-old (see Chapter 8; Milton *et al.* submitted) and spring tides are probably more important in allowing juvenile barramundi (i.e. <50 mm TL) access to important supra-littoral nursery habitats (Russell and Garrett 1985).

Freshwater flows may enhance the survival of juvenile barramundi and the subsequent size of the adult population by enabling a proportion of juvenile barramundi to access freshwater habitats intermittently linked to the estuary. Migration of juvenile barramundi into upstream habitats typically occurs at the end of the wet season (Russell and Garrett 1983, 1985). However, only a proportion of juvenile barramundi migrate upstream as individual barramundi can complete their entire life-cycle in estuaries and coastal waters (Griffin 1987; Russell 1990). In the Fitzroy River, juvenile barramundi, mainly between 250 and 400 mm TL and about one-year-old, migrate upstream using the fishway on the tidal barrage (Stuart and Mallen-Cooper 1999; Stuart and McKillup 2002). However, access to, and suitable conditions within, other significant off-stream freshwater habitats (e.g. swamps, lagoons and billabongs), up and downstream of the barrage, might be restricted to times of high river flows or flooding rain. Our results do not provide evidence either for or against this mechanism.

Current results do not support the theory that freshwater flows allow access to wetland areas for juvenile barramundi. Instead large king tides are the probably the drivers of juvenile habitat access especially when barramundi are very small (i.e. <50 mm TL). Movement into and out of supra-littoral habitats on each tide has been reported (Russell 1987).

Freshwater flows may enhance the productivity of the estuary, with increased growth rates of post-larval and juvenile barramundi increasing the number of individuals that survive the first year of life. If this occurs, then our results suggest that it is productivity resulting from spring and summer freshwater flows that is important in driving the growth and survival of young-of-the-year barramundi. In Chapter 7, we report on the analysis of tag-recapture data that shows significant positive correlations between growth rates of juvenile and adolescent barramundi and the size and timing of freshwater flows. If the results from Chapter 7 hold true for post-larval and young-of-the year barramundi, then it is highly likely that year-class strength of barramundi is being driven by the effects of freshwater flow on trophic productivity. There is probably a time lag between the occurrence of a freshwater flow and a bloom in trophic productivity of the estuary, especially at the trophic level of barramundi, which even as a juvenile is a higher order carnivore.

Barramundi has an ontogenetic change in diet, from micro-crustaceans to macro-crustaceans to fish (Davis 1987). The abundance and biomass of *Acetes* and banana prawns, which are likely to be major prey items of young-of-the-year barramundi, are coincidental and positively related to freshwater flow (see Chapters 9 and 10). This provides further evidence in support of a productivity mechanism, where a trophic cascade (or bloom) occurs as a result of freshwater flows delivering nutrients to the estuary and improving the 'quality' and or 'quantity' of nursery habitats of estuarine fishery species.

King threadfin

Variation in year-class strength (YCS), as an indicator of the overall recruitment and survival of juvenile king threadfin, was consistently and positively related to the amount of freshwater flowing or coastal rainfall delivered into the Fitzroy River estuary during spring and summer. These results are the first to provide quantitative evidence in support of increased survival of juvenile king threadfin with increased wet season freshwater flow (or rainfall).

Significantly less detailed information is available about the life history of king threadfin in northern Australia than for barramundi. King threadfin does not access freshwater habitats and will avoid waters of 0 salinity, but may utilise the salinity gradient in estuaries created by freshwater flows. We identified six possible ways in which freshwater flow may affect king threadfin (see Chapter 2). From the limited quantitative information available on the life history of king threadfin, we hypothesised that freshwater flows may influence the survival of larval and/or juvenile king threadfin in estuarine habitats leading to increased year-class strength by:

1) enhancing the biological productivity of the estuary, thereby increasing the availability of food resulting in improved growth; (Whitfield 2005; Robins *et al.* 2006) and/or
2) affecting the area of favourable habitat potentially through larger areas of decreased salinity, with lowered salinities affecting the energy budgets (Cardona 2000); and/or
3) creating turbid conditions reducing predation, enhancing survival rates (Hecht and van der Lingen 1992).

While none of these hypotheses have been directly addressed for king threadfin, the significant positive correlations between spring and summer freshwater flow and YCS provides evidence for the hypothesis that freshwater flows positively influence the survival of larval and juvenile king threadfin which are reflected in the age structure of the commercial fishery years later.

Commonalities between species

The YCS of barramundi and king threadfin have a similar positive response to summer flows, despite different tolerances by juveniles of each species for lower salinity water that occurs as a consequence of freshwater flows.

The significant correlation between the YCS of barramundi and king threadfin suggest that young-of-the-year of both species have similar levels of survival in response high and low spring and summer freshwater flows. Recruitment patterns for both species were similar in 1996 and 1999 (i.e. strong) and 1993, 1994, 1995, 1997 and 1998 (i.e. weak). In 1992, the estimate of YCS differs, with king threadfin having a weak YCS and barramundi having a strong YCS. Coastal rainfall was above average in 1992, and was the only year when there was high coastal rainfall but low freshwater flow into the Fitzroy River estuary. The high coastal rainfall may have allowed young-of-th-year barramundi to exploit low salinity habitats adjacent or connected to the main estuary that are not available to young-of-the-year king threadfin because of salinity preference limitations.

Overlap in recruitment variation would be expected as both species are carnivorous and would probably exploit any trophic blooms occurring in the estuary as a consequence of flow events. In addition, both species have a spawning season that extends over several months, thus allowing early life-history stages to exploit the benefits of freshwater flows that vary in timing from year to year in northern Australia.

The current paradigm suggests that juvenile barramundi access supra-littoral and freshwater wetlands where ever possible, whilst juvenile king threadfin occur in the estuary before freshwater flows, leave once the freshwater flow has lowered the salinity, and return on the intruding salinity wedge. Banana prawns are a major component in the diet of both these species of fish, and variability in banana prawn abundance within the estuary may significantly affect the availability of prey for young-of-the-year barramundi and king threadfin. The abundance of juvenile banana prawns varies seasonally, generally peaking in autumn and probably being related to the volume and timing of freshwater flows (see Chapter 9), and may be a function of increased growth rates of juvenile banana prawns (see Chapter 9). *Acetes* is another macro-crustacean whose populations are highly responsive to freshwater flow (see Chapter 10) and who are a major prey item for barramundi and king threadfin.

Implications for fisheries production (e.g. commercial catch)

Fisheries catch (commercial and recreational) is affected by the cumulative history of everything that has happened to all year-classes of the fished stock plus factors that effect fisheries, such as gear selectivity, effort, and management restrictions. The results of the present chapter (i.e.

variation in YCS related to freshwater flow) have implications for the fisheries management of barramundi and king threadfin. In particular, these results suggest that stock assessments should consider the impacts of freshwater flows and or coastal rainfall on the annual recruitment of barramundi and king threadfin. Furthermore, because freshwater flows affect recruitment on a catchment basis, it is likely that stock assessments should be conducted regionally rather than at large spatial scales, such as the whole of the Queensland east coast. This is particularly relevant when a significant proportion of regional populations do not undertake long-shore migration.

Implications for water management

For at least two of the main estuarine commercial fisheries species in tropical Australia, spring and summer flows are important in determining the size of year-classes of fish, which subsequently mature and enter the fishery. Whilst current water infrastructure is unable to impede the flow of very large floods, it is important that managers (and politicians and the general public) are made aware that freshwater flowing to estuaries is not wasted, but rather supports the maintenance of estuarine fish populations.

Further research and numerical modelling is required before a robust estimate can be made of the extent to which the quantity, duration or frequency (e.g. number per decade) of freshwater flows could be modified and what effects these modified flows would subsequently have on estuarine fish populations. The aim of such work would be to achieve water efficiencies in environmental flow allocations i.e. how to achieve the same effect with less water. While the underlying mechanisms for strong YCS in years of high spring and summer flows are yet to be determined it appears likely that if freshwater flows to the estuary are reduced in spring and summer there will be a reduction in the abundance of these fish species.

Chapter 7. Effects of freshwater flow on growth rates of estuarine fish

J. Robins, D. Mayer, I. Halliday and J. Staunton-Smith[11]

Summary

Relationships between freshwater flows and growth rates of an opportunistic predatory finfish (barramundi, *Lates calcarifer*) in a dry tropical estuary were examined using data from a long-term tag-recapture program. Lagged effects were not investigated.

After accounting for length-at-release, time-at-liberty and seasonal variation (e.g., winter, spring summer, autumn), growth rates were significantly and positively related to freshwater flowing to the estuary. Effects were present at relatively low levels of freshwater flow (i.e. 2.15 m^3s^{-1}, the 5th percentile of the mean flow rate experienced by fish in the study during time-at-liberty).

The analysis, although correlative, provides quantitative evidence to support the hypothesis that freshwater flows are important in driving the productivity of estuaries and can improve growth rates of species high in the trophic chain.

Introduction

One of the hypotheses about freshwater flows to estuaries is that they stimulate the productivity of the estuary. In general, this refers to the nutrients being brought to the estuary that stimulate a bloom in primary productivity (i.e. phytoplankton) and have flow-on effects for higher levels in the trophic chain. Fish may benefit from the increase in productivity because of greater availability of food. An increased abundance of food (and its subsequent consumption) has been suggested to result in faster growth rates of finfish and shellfish (Aleem 1972; Turek *et al.* 1987; Drinkwater and Frank 1994; Quiñones and Montes 2001; Salen-Picard *et al.* 2002; see also Chapter 5). Although widely speculated upon, few studies have quantified the relationship between freshwater flow and growth rates of estuary finfish (but see De Graaf 2003).

Sawynok (1998) investigated the relationship between freshwater flow and growth rates of barramundi in central Queensland. Sawynok (1998) suggested that the average linear daily growth of barramundi in the Fitzroy River system was lower for flows less than 2.5 million Ml $year^{-1}$ than for flows greater than 2.5 million Ml $year^{-1}$ (i.e. 0.61 mm/day/fish compared to 0.90 mm/day/fish). Average linear daily growth of barramundi was based on 181 individuals tagged and recaptured within the same 'tagging' year, which had an initial total length of between 300 and 450 mm, were at liberty for >30 days and whose growth was positive.

Freshwater flow is only one of many environmental factors likely to influence the growth rates of fish within and between years. Season, particularly related to temperature, is a well-documented environmental effect on the growth rates of finfish. For barramundi, growth is strongly seasonal (i.e. like a cosine wave) with rates increasing from mid-spring (October) to the start of autumn (March) and decreasing from autumn to mid-spring (Dunstan 1959; Davis and Kirkwood 1984; Xiao 1999, 2000). Seasonal growth is also evident in sectioned otoliths of barramundi, where annular 'bands' are present in the otoliths, with the opaque zone representing slow growth being laid down before October (see Chapter 6; Staunton-Smith *et al.* 2004), and coincides when water temperatures are lowest. Xiao (1999) suggested that strong seasonal growth was potentially related to seasonal availability of food and seasonal changes in water temperature.

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Our aim in the present chapter was to re-assess the length differences in ANSA tagged-recaptured barramundi, include additional recapture data, and after accounting for seasonal growth, quantify the relationships between growth and freshwater flow conditions.

Methods

Tagged fish data

Tag-recapture data were obtained from the Suntag Program of the Australian National Sport Fishing Association Queensland Inc. (ANSA Qld). This catch and release tagging program aims to generate data for resource management purposes and has been ongoing in the Fitzroy River catchment and adjacent areas since 1984. A variety of freshwater, estuarine and marine species are caught, tagged and released by mostly recreational anglers. Recapture of tagged fish may be by ANSA Qld members, other recreational anglers, commercial fishers or researchers. Capture methods include limited cast-netting, line fishing with lures and bait (i.e. by recreational fishers) and gill-netting with mesh sizes ranging between 135 mm to 180 mm (i.e. by commercial fishers).

At the time of accessing, the Suntag database included 58 000 records of tagged fish of 154 ‘species’ tagged and recaptured between October 1984 and November 2004. Information available from tag and recapture events included tag number, date, total length in mm (TL), location of capture, and fisher. From this information, time-at-liberty, length-at-release, length-at-recapture, release year, release month, recapture year, recapture month, release location type, and recapture location type were calculated. From the available data, records that met the following criteria were selected: (i) species = barramundi; (ii) region of release and recapture = Fitzroy River catchment; (iii) time-at-liberty (in days) >30 and <366; and (iv) change in total length >0 mm. Selection of fish with a least 30 days between release (i.e. tagging) and recapture negated the need to consider tagging effects on growth rates (Reynolds and Moore 1982; Wang and Jackson 2000). Release and recapture locations were classified to account for the possibility that habitat type influences growth rates, as suggested by Davis and Kirkwood (1984). Anecdotal reports indicate juvenile barramundi grow faster in aquatic habitats with salinities less than that of seawater, suggesting that the habitat type in which a barramundi is tagged and recaptured should be considered as a cofactor during growth analyses. Release and recapture locations were classified into three types: (i) estuarine creek; (ii) estuarine river; and (iii) flood plain lagoon, which are freshwater areas intermittently connected to the estuary by floods. Movement of individuals between habitat types was examined. The majority of movement (~50%) involved individuals released in estuarine river habitats moving to estuarine creek habitats, but remaining within the estuary of the Fitzroy River.

Freshwater flow data

Freshwater flow data were obtained from the Department of Natural Resources and Water. The freshwater flowing to the estuary (end-of-system, EOS) was estimated as being gauged stream flow at the most downstream gauging station (i.e. at ‘The Gap’, 142.1 km Adopted Middle Thread Distance, ~23°5’S, 150°6’E), minus the estimated seasonal extractive uses provided by the Department of Natural Resources and Water and Fitzroy River Water. Flow-during-liberty was calculated as the sum of estimated EOS flows of the Fitzroy River for the period during which individual tagged barramundi were at liberty i.e. flow-during-liberty is specific for each tagged individual.

Analysis

The data were explored using graphical methods and general linear models. Data patterns were noted, and three outliers identified and excluded from subsequent analyses.

The non-linear von Bertalanffy equation (Ratkowsky 1986) remains the cornerstone for analyses of finfish growth data, primarily because it has a single parameter (*K*) representing growth rate. In alternate equations, such as the Schnute model (Ratkowsky 1986) or the function of Xiao (2000), growth rate is incorporated into two or three parameters, with a diminished capacity to incorporate environmental effects. Problems with the von Bertalanffy equation include a postulated relationship between the variance of the residuals and the fitted values (Sainsbury 1980), but this did not occur with our data. Also, the estimates of K (the growth rate) and L_∞ (the population's average asymptotic length) can be correlated, but this is more of a problem when comparing analyses between different data sets. The current analysis fits a single data set.

Seasonality of growth is a known and major effect with barramundi (Xiao 2000), so the seasonal form of the von Bertalanffy equation (Somers 1988) was adopted as the starting model using GenStat (2005). This form is

$$\mathrm{L}_{\mathrm{t}} = \mathrm{L}_{\mathrm{t}\text{-}\delta} + (L_\infty - \mathrm{L}_{\mathrm{t}\text{-}\delta})(1 - e^{-K\delta + S(t-\delta) - S(t)})$$

where L_{t} is the length-at-recapture, t is the day of recapture,
δ is time-at-liberty (days), $\mathrm{L}_{\mathrm{t}\text{-}\delta}$ is length-at-release,
L_∞ is the asymptotic length, K is the average exponential growth rate, and

$$\mathrm{S}_{(\mathrm{i})} = \frac{CK \sin[2\pi(i - t_s)]}{2\pi}$$

where C measures the magnitude of the seasonal oscillation and t_{s} is the time shift for the annual cycle (Somers 1988).

Parameters to be estimated are L_∞, K, C, and t_s. This equation directly incorporates the effects of time-at-liberty, seasonality, and length-at-release. The effects of freshwater flows on growth rates were incorporated into the growth coefficient, K, as described below. A log-linear relationship of flow was superior to linear, and a threshold value (below which flow had no effect) was also found to be significant ($P<0.05$). Thus, in the equation above, K was expanded to be K_a + (K_b {flow – threshold} {if flow > threshold}). Other factors were screened for their effects on growth using general linear models. Recapture-year was significant, but year was correlated with freshwater flow, so the latter was inferred as the more likely cause of growth differences. Release and recapture habitat type (i.e. estuarine creek, estuarine river, and flood plain lagoon) were not significant ($P>0.05$) and so were not included in the final model. The final model had an adjusted R^2 of 91%.

Assumptions and limitations

We assumed that: (i) lag effects between flow and growth rates were not significant; and (ii) the effects of measurement errors on growth rates were insignificant. The details and validity of these assumptions is discussed further in Robins *et al.* (2006), see Appendix for Chapter 7.

Results

A total of 1168 tagged and recaptured barramundi satisfied the selection criteria and were used in the analysis. Length-at-release ranged from 170 to 910 mm TL (**Figure 7.1**), although ~90% of fish were <600 mm TL and were probably juveniles or young adults. Barramundi were tagged throughout the year, with ~85% tagged between spring (September) and mid-autumn (April). The data were distributed within and across seasons, with 346 fish at-liberty during one season, 273 fish at-liberty over two seasons, 273 fish at-liberty over three seasons and 276 fish at-liberty over four seasons (**Table 7.1**). Mean daily growth rates (i.e. change in TL/days-at-liberty) varied between individuals of the same length-at-release (**Figure 7.2**). The greatest change in length (i.e. 380 mm by a 200 mm TL length-at-release individual at-liberty for 352 days) is similar to the greatest change in length within one year reported for barramundi tagged and recaptured by researchers in Papua New Guinea (Reynolds and Moore 1982).

There was considerable variety in the range of freshwater flow during-liberty experienced by barramundi included in the current study (**Figure 7.3**), including a 1-in-100 year flood. In terms of release-location-type, 93% of barramundi were released in estuarine river habitats, and 4% and 3% were release in flood plain lagoon and estuarine creek habitats respectively. About 46% and 52% of barramundi were recaptured in estuarine river and estuarine creek habitats, respectively, with only 2% being recaptured in flood plain lagoons.

Figure 7.1 Size of barramundi tagged, released and recaptured within the Fitzroy River estuary (n=1168)

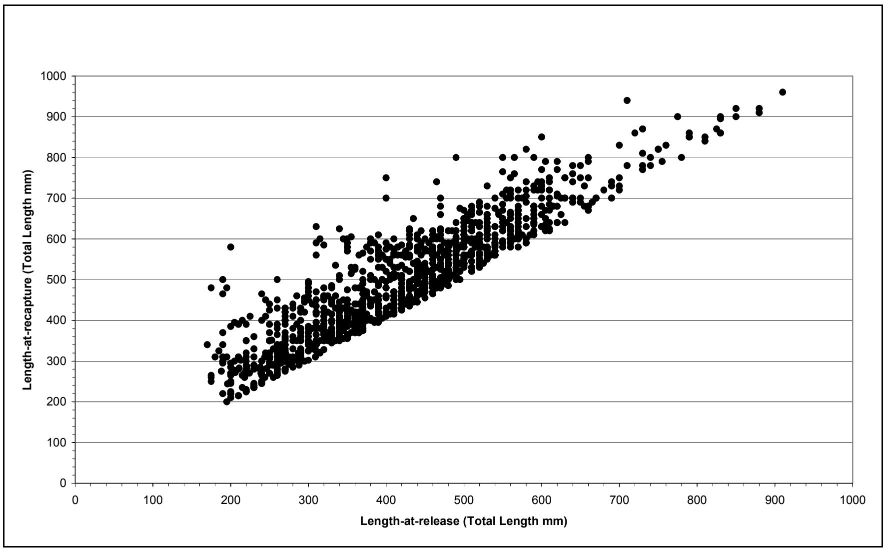


Figure 7.2 Growth rates (mm/day) of barramundi tagged, released and recaptured within the Fitzroy River estuary (n=1168)

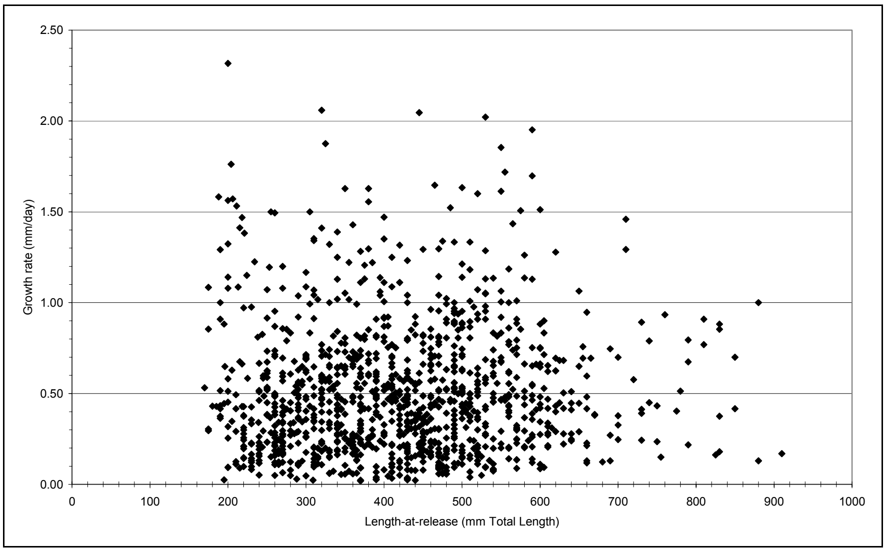
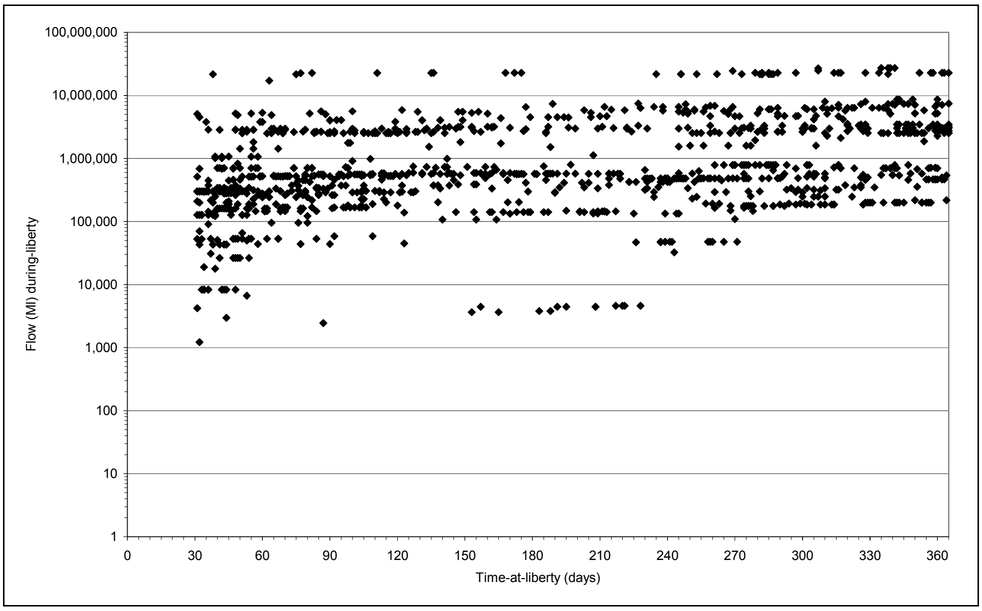


Table 7.1 Duration and seasonal distribution of time-at-liberty of barramundi tagged, released and recaptured in the Fitzroy River estuary (n=number of fish)

One season		Two seasons		Three seasons		Four seasons	
	n		n		n		n
spring	35	spring summer	95	spring summer autumn	34	spring summer autumn winter	25
summer	200	summer autumn	110	summer autumn winter	78	summer autumn winter spring	119
autumn	103	autumn winter	36	autumn winter spring	70	autumn winter spring summer	131
winter	8	winter spring	32	winter spring summer	91	winter spring summer autumn	1
Total	346		273		273		276

Figure 7.3 Freshwater flow (Ml) experienced by barramundi tagged, released and recaptured within the Fitzroy River estuary during their time-at-liberty (days), y-axis presented on a $\log_{10}$ scale, (n=1168)



Relationships between growth and freshwater flow

The estimated biological parameters (i.e. L_∞ = 1829 mm, C = 1.045 and t_s = 19.4) and average growth rates of different sized fish (**Table 7.2**) were all in agreement with other sources (e.g. Reynolds and Moore 1982; Griffin 1988; Fishbase 2004). As expected, growth rates of barramundi in the Fitzroy River estuary were strongly seasonal, and were fastest in summer and slowest in winter. Freshwater flow had a major interaction with season (**Figure 7.4**), having minimal effect on growth rates in winter and greatest effect on growth rates in summer (i.e. compared to average seasonal *K*, **Table 7.2**).

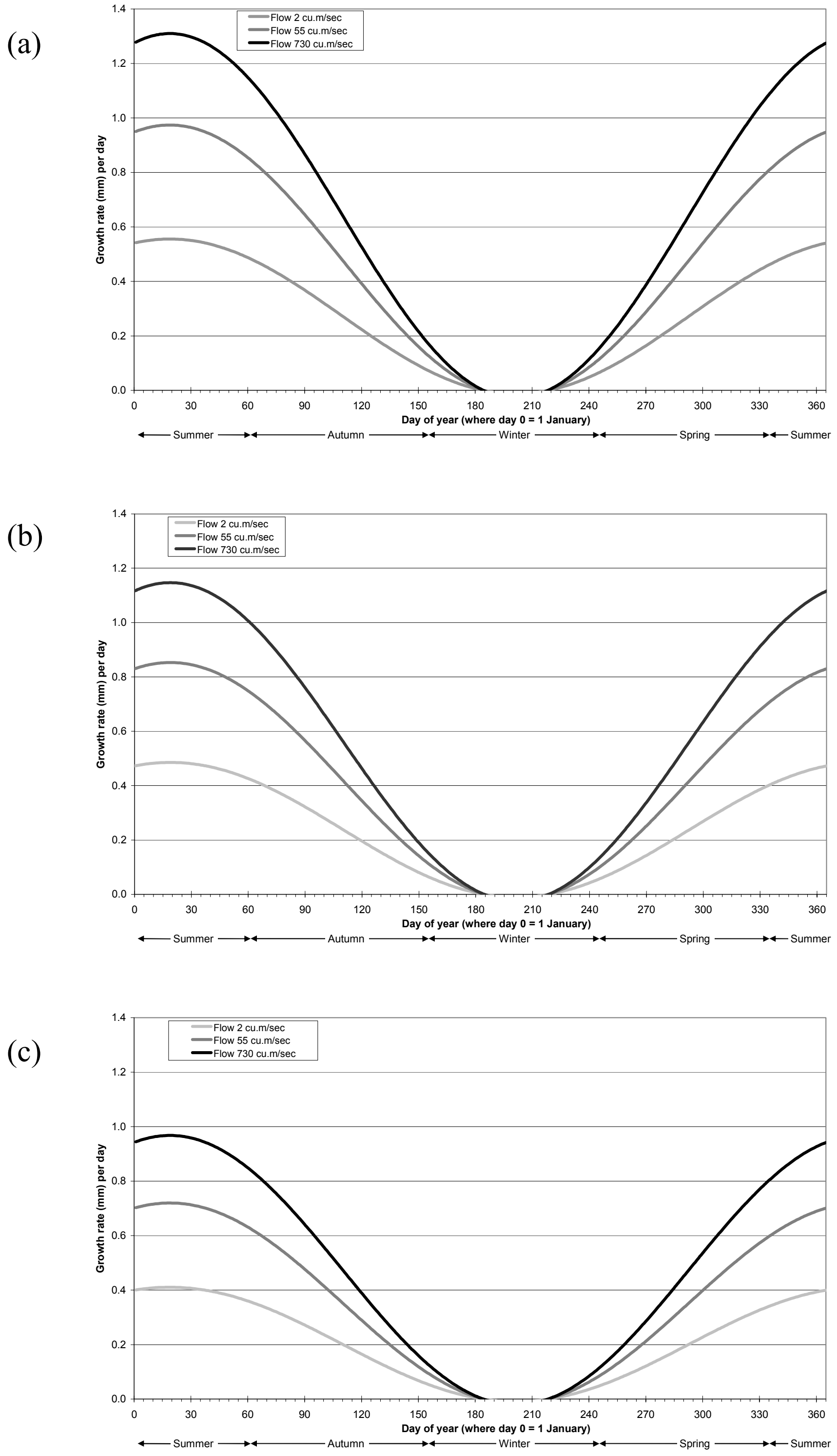
Table 7.2 Growth parameters for barramundi in the Fitzroy River estuary estimated under varying length-at-release and freshwater flow conditions

Length-at-release (TL in mm)	Freshwater flow (m3sec-1)	Annual *K* (yr^{-1})	Average Seasonal *K*			
			Summer	Autumn	Winter	Spring
420^A	2.15^D	0.238	0.461	0.255	0.015	0.223
420^A	54.55^A	0.418	0.810	0.448	0.026	0.393
420^A	730.63^C	0.562	1.089	0.603	0.035	0.528
220^B	54.55^A	0.478	0.925	0.512	0.030	0.448
640^C	54.55^A	0.353	0.683	0.378	0.022	0.331

^{A}= 50th percentile (median), ^{B} = 5th percentile, ^{C} = 95th percentile, ^{D} = break-point (critical threshold)

During summer, median or greater freshwater flows resulted in about twice the growth rates of minimal flows. For example, at the peak of the seasonal growth curve, growth for a 420 mm TL barramundi is 0.81 mm/day compared to 0.46 mm/day for flows of ~55 m^3sec^{-1} and ~2 m^3sec^{-1} respectively. The threshold at which freshwater flows affected seasonal growth (i.e. below which freshwater flow had no effect) was ~2 m^3sec^{-1}, which is equivalent to the 5th percentile of the freshwater flow during-liberty experienced by fish in the current study (**Figure 7.5**).

Figure 7.4 Seasonally adjusted growth curves for barramundi in the Fitzroy River estuary under various freshwater flow conditions*, modelled for: (a) 220 mm; (b) 420 mm and (c) 640 mm (total length) fish, which are the 5th, 50th and 95th percentiles of the sizes of barramundi tagged, released and recaptured



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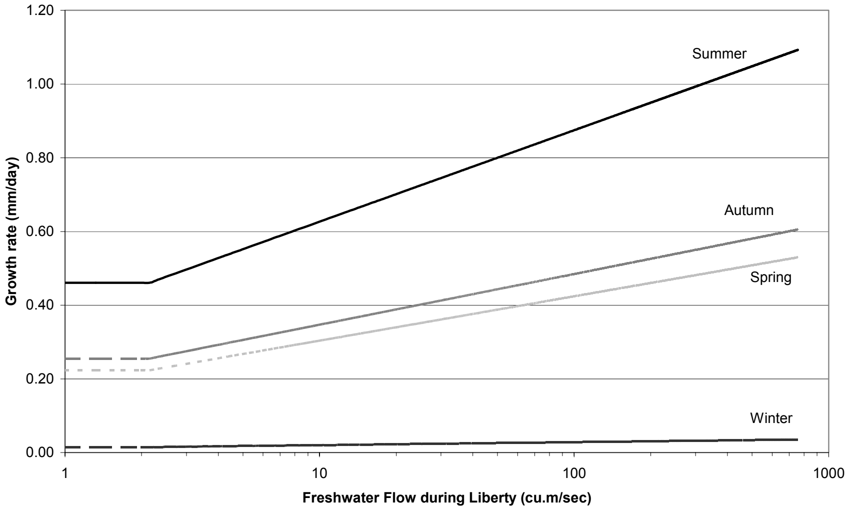
[*] 2 m^3sec^{-1}, 55 m^3sec^{-1} and 730 m^3sec^{-1} are the 5th, 50th and 95th percentiles of flow-during-liberty experienced by barramundi tagged, released and recaptured in the Fitzroy River estuary between 1984 and 2004.

Figure 7.5 Thresholds of the effect of freshwater flow-during-liberty on seasonal growth rates of barramundi within the Fitzroy River estuary, x-axis presented on a $\log_{10}$ scale



Discussion

The analysis of tag-recapture data for barramundi in the Fitzroy River estuary, collected over almost 20 years, clearly demonstrates that: (a) growth rates vary seasonally; and (b) variable growth of barramundi is significantly related to the freshwater flows experienced by individuals, although other factors (e.g. genetic variation) are also likely to be important. The analyses does not account for lag effects between freshwater flows and growth, although such effects may exist.

In the current study, the threshold at which freshwater flows affected seasonal growth (i.e. 2 m^3sec^{-1}) was an order-of-magnitude lower than the threshold identified by Sawynok (1998) of 79.3 m^3sec^{-1} (= 2.5 million Ml $year^{-1}$), who undertook more simplistic and less quantitative analyses using annual flow. This result may reflect the greater complexity of the current analysis, with the inclusion of seasonality allowing greater resolution of the effects of freshwater flow. The value identified by Sawynok (1998) is similar to the median annual flow in the Fitzroy River, above and below which seasonal growth rates (particular summer growth) are faster or slower respectively.

Variation in growth with freshwater flow might explain the inability of Davis (1987) to determine a reliable overall growth curve for barramundi from six rivers in northern Australia. He suggested that variability in growth was a reflection of different environmental conditions. We suggest that freshwater flow is probably a major factor of the environmental condition that Davis (1987) postulated to influence growth. Davis (1987) goes onto say that “these vast aquatic habitats formed during the summer monsoons provide juvenile barramundi with an almost predator-free prey-rich environment promoting rapid growth and improved survival”. The probability of highly variable growth resulting from adaptive phenotypic plasticity, dependent on the conditions to which an individual is exposed was also suggested by Morita and Morita (2002). This probably applies to barramundi, which is a euryhaline, opportunistic predator, with a non-obligatory catadromous life-cycle.

Numerous aspects of the barramundi life-cycle are likely to have evolved to take advantage of the flood-drought conditions of northern Australia, with growth being one example. Recruitment of barramundi has been quantitatively related to freshwater flow (Staunton-Smith *et al.* 2004; see Chapter 3 and 4), as has its catchability (Robins *et al.* 2005; see Chapter 3 and 4). Indeed, faster growth (as a consequence of higher freshwater flows) may lead to increased survival of young-of-the-year barramundi (i.e. < ~450 mm TL) which would enhance the strength of the year-class during higher flow conditions. As such, the results of the current chapter are consistent with those of Chapter 6 (published as Staunton-Smith *et al.* 2004), where significant correlations were found between age-based assessments of year-class strength of barramundi and freshwater flow into the Fitzroy River estuary.

Possible causal mechanisms of relationships between fish growth and freshwater flow

Commercial and recreational fishers anecdotally report that barramundi grow faster in freshwater habitats than saltwater habitats, and Dunstan (1959) commented that for fish of the same length, barramundi from freshwater habitats are heavier than those from saltwater. However, in the current analysis, the inclusion of release and recapture location type (i.e. estuarine creek; estuarine river; and flood plain lagoon) did not significantly improve the model. It may also be that more fish from different habitats, especially flood plain lagoons are needed to detect habitat effects or we may need to assess weight rather than length data.

Freshwater flows are speculated to affect the growth rates of estuary-associated fish such as barramundi by a number of mechanisms. Freshwater flows to estuaries lower salinities for varying periods, with the lowered salinity then affecting the energy budgets of estuarine fish species. For example, Cardona (2000) examined the effects of salinity on the habitat selection and growth of mullet (*Mugil cephalus* Linnaeus), which is a euryhaline and non-obligatory catadromous species. Cardona (2000) speculated that the growth of euryhaline species was affected by salinity because the energy used for osmo-regulation is not available for growth. Results in the present study of higher growth rates at higher freshwater flows (= lowered salinities in the estuary), and the anecdotal reports of higher growth in freshwater habitats, would be consistent with this hypothesis.

Trophic linkages are another mechanism by which freshwater flows (i.e. floods) may influence the growth rates of finfish. This is a common speculation (Aleem 1972; Turek *et al.* 1987; Drinkwater and Frank 1994; Quiñones and Montes 2001; Salen-Picard *et al.* 2002; see also Chapter 5), with the theory that the abundance and or accessibility of prey affect the growth rates of predators (Zimmerman *et al.* 1990). Darnaude *et al.* (2004) linked the flood-related pulses in short and long-lived polychaete species to increased growth and survival of sole (*Solea solea* Linaeus) using stable isotope analysis, supporting previously reported relationships between river runoff and the coastal fishery production of sole in the Gulf of Lions (Salen-Picard *et al.* 2002). The result of Darnaude *et al.* (2004) provides quantitative evidence of the existence of trophic links between freshwater flow and fisheries production through enhanced growth and survival. Barramundi is an opportunistic ambush predator that is likely to exploit any pulses in prey availability related to trophic responses to flood events, in either estuarine or freshwater habitats. Further research would be needed to determine if relationships between freshwater flow and growth rates are common phenomena for other estuarine-dependent finfish species. Platten (2005; see chapter 5) presents further evidence in support of trophic linkages resulting in increased growth rates of estuarine-associated finfish.

Implications for water management

Climatic variation in the strength of the wet season in northern Australia is the primary driver of variation in freshwater flow, although the development of water resources in northern Australia is increasingly altering the quantity, quality, timing, duration and frequency of freshwater flows to estuaries. There is increasing recognition of the need to allocate water for the environment as part of the sustainable use of water resources and to consider in a quantitative manner the impacts of water abstraction and regulation on downstream ecosystems (Davis and Hirji 2003 a,b,c; Dyson *et al.* 2003).

Results from the current study indicate that the impact of altering freshwater flows varies seasonally, with the greatest impacts on the growth rates of barramundi occurring during summer, late spring and early autumn. This is an obvious, albeit previously unquantified, outcome for a species that has seasonal growth. In addition, altering freshwater flows will have greater consequences on the growth rates of smaller (younger) fish, whose instantaneous rate of growth is faster than that of larger (older) fish. It is difficult to make more detailed speculations of the consequences of altering flow on the production (i.e. fisheries catch) of estuary-associated species such as barramundi without further, more detailed modelling and the exploration of potential lagged effects of freshwater flow. However, more detailed modelling is an area of research that would potentially provide quantitative answers to the continuing questions that fisheries scientists are frequently asked by water managers; namely: (a) How much water do estuaries require? and (b) What are the consequences of altering the freshwater flow regime by a certain quantity at a certain time of year?

Chapter 8. Otolith micro-chemistry of barramundi

D. Milton, I. Halliday, J. Staunton-Smith, M. Sellin, R. Marsh, D. Smith, M. Norman and J. Woodhead[12]

Summary

The microchemistry of otoliths of cohorts of a fished population of the large catadromous fish, barramundi *Lates calcarifer* were examined to help understand the role of freshwater river flows in maintaining connectivity among key habitats. Barramundi from the estuary of the large, heavily regulated Fitzroy River, north-eastern Australia were analysed by making transects of ^{87}Sr/^{86}Sr isotope and trace metal/Ca ratios from the core to the outer edge.

Firstly, we examined the Sr/Ca, Ba/Ca, Mg/Ca and ^{87}Sr/^{86}Sr isotope ratios in otoliths of barramundi tagged in freshwater and recaptured by commercial fishers in the estuary. We used ^{87}Sr/^{86}Sr isotope ratios to identify periods of freshwater residency and assess whether trace metal/Ca ratios varied between habitats. Only Sr/Ca ratios consistently varied between known periods of estuarine or freshwater residency. The relationship between trace metal/Ca ratios and river flow, salinity, temperature or age were examined in fish tagged and recaptured in the estuary with partial correlations.

We found most variables correlated with each metal/Ca ratio, but few correlations were consistent among fish. These results suggest that both individual movement history within the estuary and seasonal changes in flow patterns were influencing the correlations. Fish that had accessed more productive coastal freshwater habitats as juveniles had enhanced growth rates, highlighting the need to maintain freshwater flows in rivers. About half the fish examined had accessed freshwater habitats before capture. Of these, all had spent at lest their first two months in marine salinity waters before entering freshwater and some did not enter freshwater until four years of age. This contrasts with previous studies in other parts of the range of barramundi where larval access to freshwater swamps is important to enhanced barramundi population productivity.

Introduction

Mechanisms by which freshwater flow enhances estuarine fish populations varies between species and depends on their life history (Robins *et al.* 2005). For anadromous and catadromous fish species, freshwater flow is required to maintain natural migrations around dams and barrages. Catadromous fish species, such as barramundi, may have enhanced survival and growth in years when coastal spawning grounds receive good flooding rains (Staunton-Smith *et al.* 2004; Robins *et al.* 2006).

In northern Australia, barramundi reach up to 30 kg and occur in both the estuaries and accessible freshwater reaches of most rivers. However, in many rivers, water infrastructure, such as barrages, weirs and dams have limited upstream access by barramundi (and other species) to the extent that they no longer occur in freshwater reaches where they were previously abundant (Hogan and Vallance 2005). A stocking program for barramundi in many natural and artificial water bodies in Queensland has occurred to help overcome this disruption to natural migration.

Many river systems in north-eastern Queensland, such as the large Fitzroy River have numerous dams and weirs that restrict or even halt freshwater flows during the austral winter when rainfall is minimal. Barramundi stocked in these impoundments can only move (up and downstream) during floods that cause sufficient flow to fill these impoundments, allowing overflow (or drown-

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[12] Parts of this chapter are published in: Milton, D., Halliday, I., Staunton-Smith., Sellin, M., Marsh, R., Norman, M. and Woodhead, J. (submitted). Otolith chemistry of barramundi *Lates calcarifer* can provide insight about the role of freshwater flows in maintaining estuarine populations of regulated rivers. *Estuarine and Coastal Shelf Science*.

outs) and thus connectivity. Therefore, there needs to be a minimum flow in these regulated rivers before fish can migrate.

Besides movement of freshwater-resident fishes, moderate to large floods also fill temporary coastal nursery swamps and provide large amounts of additional habitat for larval and juvenile barramundi (Moore 1982; Davis 1987). These temporary coastal supra-littoral swamps are believed to enhance both survival (Russell and Garrett 1983, 1985; Staunton-Smith *et al.* 2004) and growth (Robins *et al.* 2006) of larval and small juvenile barramundi. However, a recent study of otolith chemistry of barramundi in Papua New Guinea found little evidence that fish were spending extended periods in freshwater during the first year of their life (Milton and Chenery 2005). This suggests that freshwater swamps may not be important habitats, but rather the saline temporary estuarine supra-littoral swamps among coastal mangroves may be more important habitats for larval and early juvenile life history stages (Moore 1982; Russell and Garrett 1983; Davis 1987).

Otolith chemistry methods are being increasingly used to improve understanding of fish habitat use and movements. This is largely due changes in water chemistry being detectable in otolith concentrations of several metals and their isotopes (Kennedy *et al.* 1997; Campana 1999; Milton and Chenery 2001). Otoliths are primarily aragonitic calcium carbonate and the concentrations of common metals, such as Strontium (Sr), Barium (Ba), Magnesium (Mn) and Manganese (Mg), change in response to environmental conditions (Campana 1999; Gillanders 2005). Strontium has been the most widely used trace metal to infer movements between freshwater and marine habitats (Secor and Rooker 2000). However, temperature, growth and stress have all been shown to influence Sr concentrations in fishes (Kalish 1992; Sadovy and Severin 1994).

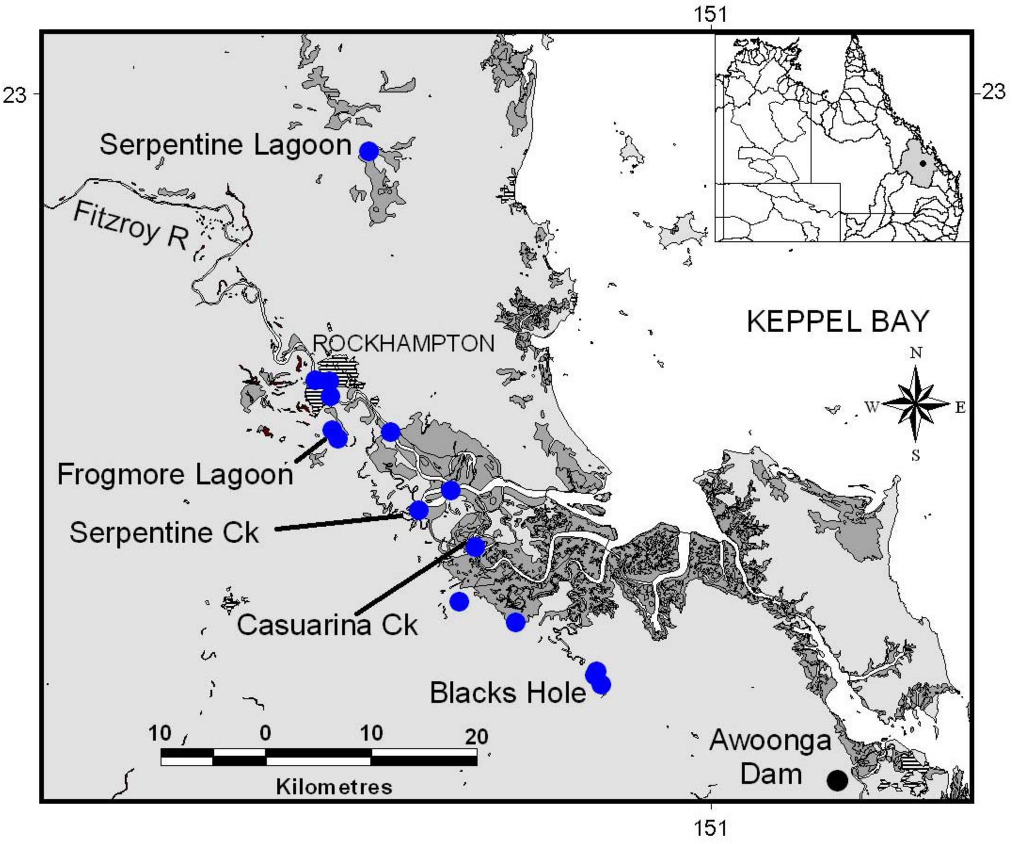
The aims of this chapter were to: (i) examine ^{87}Sr/^{86}Sr isotope ratios and trace element concentrations in fish with known habitat history to verify that otolith chemistry can detect differences in fish from freshwater and estuarine habitats; (ii) assess the scale of variation in transects of ^{87}Sr/^{86}Sr isotope ratios and trace element concentrations in barramundi otoliths; in order to (iii) determine their linkage with temperature, salinity, growth and freshwater flow volumes in the Fitzroy River at similar scales; (iv) assess the role of freshwater flows for providing connectivity for barramundi that have accessed freshwater by analysing fish from the fished estuarine population during a flood event; and (v) compare the habitat history of fast and slow-growing individuals of two strong age-classes to determine whether enhanced growth or survival were related to freshwater access.

Methods

Description of study area

Barramundi were collected from several locations within the Fitzroy River basin (Figure 8.1) and impoundments in the neighbouring Boyne River. A 396 m long tidal barrage was built on the Fitzroy River about 59 km from the estuary mouth in 1970 and now defines the upper limit of saline tidal water intrusion. This barrage has a system of 18 x 3.4 m high lift gates that maintain water levels between 3.1 and 3.4 m above the barrage (Kowarsky and Ross 1981). Three lift gates are usually raised 0.3 m during non-flood conditions to maintain some flow. A fixed spillway 0.3 m higher than the lift gates also allows some flow during non-flood conditions. Since 1996, fish have been able to access the upper Fitzroy River through a fishway that operates effectively for about half the tidal cycle. Runoff from the mostly monsoonal rainfall in the catchment flows into the mangrove-lined estuary through the barrage. The lift gates are normally raised during heavy flow periods. Tidal flushing of the estuary is minimal under non-flooding conditions with a water residence time of up to eight months (Connell *et al.* 1981). However, this can reduce to as little as a week under extreme flood events.

Figure 8.1 Location of fish tagging and recapture sites within the Fitzroy River estuary. Inset shows the extent of the Fitzroy River basin in eastern Australia. The darker shaded region of the lower estuary represents the extent of ephemeral habitat flooded during moderate floods. The dot on the inset represents the tagging location of fish DR1 on the upper Dawson River, a tributary of the Fitzroy River.



Sample collection

Two groups of samples were collected for otolith analysis to address specific questions. Firstly, to confirm that otolith chemistry could be used to infer fish habitat use, we obtained samples of barramundi with known habitat histories. These included fish tagged in freshwater and recaptured in the estuarine commercial fishery, fish tagged and recaptured in the estuary and fish from two freshwater impoundments. Tagging studies of barramundi and other recreationally-important species has been undertaken by recreational fishers in the Fitzroy River basin for over 25 years. We obtained most tagged fish from the local fish processors (see Chapter 6; Staunton-Smith *et al.* 2004).

Secondly, we obtained a large sample of otoliths from fish caught by commercial fishers in the Fitzroy River estuary before and during a flood event in February 2003. Capture locations of all fish were verified with the fisher before processing. Most samples were obtained from fish frames that were available after filleting. Fish were measured (total length TL in cm) and their otoliths removed, cleaned of adhering tissue and dried with tissue paper before storing in plastic vials for return to the laboratory for sectioning.

In the laboratory, otoliths were weighed (± 0.001 g), blocked in clear casting resin. All plasticware used was washed with 10% nitric acid for 24 hrs and rinsed with millipore water. Blocked otoliths were sectioned transversely through the core using a low speed saw lubricated with millipore water. Each 400 μm section was polished with 1500 grit wet and dry paper moistened with millipore water. Polished sections were stored in acid washed plastic vial until mounted. Sections were mounted on clear Perspex blocks using clear casting resin. The exposed polished surfaces of otoliths were wiped with tissue and 0.5M nitric acid. Up to eight sections were randomly mounted on a single perspex block. Annuli were counted and compared with their period of release to verify the rate of increment formation. Staunton-Smith *et al.* (2004) also validated that increment checks were formed annually in fish of known age. Dried blocks were returned to plastic bags before analysis by LA-ICPMS (Laser Ablation - Inductively Coupled Plasma Mass Spectrometer) or MC-ICPMS (Multi Cupped - Inductively Coupled Plasma Mass Spectrometer).

Trace metals

Elemental concentrations in otoliths were determined by making ablations with a Resonetics LPX120i ARF 193-nm excimer laser system coupled with an Agilent 7500s Series quadrupole ICP-MS (Eggins *et al.* 1998; Eggins and Shelley 2002). The ICPMS was operated in time-resolved mode and with a dwell time of 20 ms per mass. The LA-ICPMS system was calibrated with NIST 610 glass. For both the NIST glass and otoliths, we used Ca as an internal standard to control for variation in ablation and aerosol transfer efficiency. Otoliths were analysed randomly in blocks of eight with standards run before and after each block. Prior to analysis, samples were wiped with concentrated nitric acid to remove any surface contamination. For each sample, we acquired a 60 sec. blank to correct for background. During ablation, we sampled each otolith continuously along the transverse axis from the core to the edge. This resulted in a trench of ablated material, 40 µm wide and 10-15 µm deep at a laser repetition rate of 5 Hz. Sixty-second acquisitions of standards were made along transects with similar methods to those used for otolith analysis. Based on this analytical procedure, we were able to quantify the concentrations of ^{24}Mg, ^{55}Mn, ^{86}Sr, ^{88}Sr, ^{137}Ba and ^{138}Ba. The detection limits (in µg.g^{-1}) for each element during the otolith analysis were typically ^{24}Mg: 0.1 – 0.4, ^{55}Mn: 0.2 – 0.5, ^{86}Sr: 0.8 – 1.3, ^{88}Sr: 0.2 – 0.9, ^{137}Ba: 0.1 – 0.5 and ^{138}Ba: 0.1 – 0.4. Overall Sr and Ba concentrations were calculated as the mean of the values obtained from the two isotopes measured for each element.

Sr isotopes

^{87}Sr/^{86}Sr isotope ratios were measured on otoliths of 14 fish that could define the isotope signatures of particular freshwater tributaries of the Fitzroy River, or had patterns of Sr and Ba concentrations across transects that indicated the fish had spent periods in reduced salinity. A further 35 fish caught before (10) and during the February 2003 flood were also examined to determine whether they had recently entered the estuary from freshwater habitats. These fish were further selected to be from two single age cohorts that showed a strongly bi-modal size distribution (4 and 7 yr olds). Barramundi show an extreme range in growth rates (Staunton-Smith *et al.* 2004), so we examined fast and slow-growing fish within each cohort to assess whether habitat use was influencing growth rate or survival.

Otoliths were analysed with a 'Nu Plasma' MC-ICPMS (Nu Instruments, Wrexham, UK) coupled with a Helium excimer laser ablation system containing a Compex 110 laser operating at 193 nm and instrument operational conditions similar to those of Woodhead *et al.* (2005). The laser and ICPMS operating systems have been described in greater detail by Eggins *et al.* (1998) and Woodhead *et al.* (2004). The ICPMS has 12 Faraday cups available and eight of these were used to collect ion currents at masses 82 – 89. Mass bias was corrected by reference to the ^{86}Sr/^{88}Sr ratio of 0.1194 after appropriate interference correction (Woodhead *et al.* 2005). The system has been demonstrated to have high levels of precision and accuracy due to instrument sensitivity and a more robust correction for Kr and Rb (Woodhead *et al.* 2004, 2005).

Otolith ablation tracks were marked with the Nu Plasma software prior to analysis and the tracks were made perpendicular to the longest growth axis on each otolith. The laser system was operated in the routine 'spot' analysis mode as transects were not linear and thus the alternate 'slit' mode (Woodhead *et al.* 2005) was not available. We aimed to obtain a Sr signal between 3 and 7 V and this corresponded to fluence at the target of ~ 10 J.cm^{-2} and a repetition rate of 5 Hz. This corresponded to a spot size of approximately 50 µm.

For any otolith, peak centering was undertaken prior to analysis at a point on the otolith away from the transect track. This was followed by 60 sec. of blank before the otolith transect began. Most otoliths took about 500 sec. to ablate with the laser moving at about 12 $\mu m.s^{-1}$. Data validation was made after every four otoliths by undertaking spot analyses of a modern marine mollusc shell (*Donax*). This shell has fine-scale growth zonation in Sr content and been shown to be in equilibrium with seawater (Woodhead *et al.* 2005). Dwell times for the validation spots were 60 sec. after a 60 sec. blank acquisition.

Environmental variables

Daily river flow volume (Ml day^{-1}) measured at the most downstream gauging station 'The Gap', minus the estimated downstream extraction was obtained from the Department of Natural Resources and Water. Flow volume was chosen as the proxy for flood events as Staunton-Smith *et al.* (2004) found that seasonal river flow and coastal rainfall totals were highly correlated and both explained similar amounts of the variation in observed barramundi year-class strength of commercial catches in the Fitzroy River estuary. Monthly water temperatures and salinities were also measured mid-river at six depths at three sites in the estuary (20, 36.2 and 50 km upstream from the river mouth) by the Department of Natural Resources and Water. These sites covered the major part of the estuary where barramundi were most abundant.

Data analysis

Trace metals

The recreational fishing association ANSA (Infofish Services, Rockhampton) provided data on dates, size (TL in mm) and location of initial capture. Ablation tracks were matched to fish ages by calibrating the ablation rate with measured distances along the track. Distances between otolith annuli along the ablation track were measured using Optimas image analysis software with a microscope at 100x power (± 0.01 mm). Distances of each ablation along the transect were then converted to a fraction of the year. These were then adjusted for the length of the growing season (October – April) (see Chapter 7; Robins *et al.* 2006). Thus, the translucent zone of each annulus was assumed to have been laid down during this period. Ablations made across the opaque zone were assumed to have been laid down between May – September when growth is minimal (see Chapter 7; Robins *et al.* 2006).

In order to understand the scale of patterns in environmental and otolith chemistry, we analysed the patterns in both data sets and explored the scale of these patterns with wavelet transforms. These have the advantage in that they decompose a longitudinal pattern into a hierarchy of different scales (Keitt 2000; Keitt and Urban 2005). They offer distinct advantages over similar Fourier transforms in the analysis of complex, non-stationary patterns (Keitt and Urban 2005). This is due to their being local transforms and thus can be used to examine the intensity of patterns at different scales (Daubechies 1992; Walker 1999). Prior to application of the discrete wavelet transform, log-transformed otolith microchemistry data was generated on an equispaced temporal mesh of 2^n points. This mesh spans the age range of the fish. The interpolated data are generated via a two stage process utilising piecewise cubic Hermite interpolation (Ralston and Rabinowitz 2001). The first stage yields a set of ablation values on the new temporal mesh from a reduced number of available (ablation, time) values, onto which the given larger set of regularly spaced (ablation, concentration) data is finally interpolated. Daubechies D_2 transforms (Strang and Nguyen 1996), including boundary correction (Cohen *et al.* 1993), are applied to the interpolated data with bottom resolution level 3.

Coefficients for river flow, temperature and salinity were correlated with the coefficients of Mg, Mn, Sr and Ba by partial correlations as river flow and salinity covaried. We analysed all environmental variables for transforms at resolution levels 3 to 7. This was because there were rarely more than 128 salinity and temperature measurements within the lifetime of the fish.

Sr isotopes

Examination of the $^{87}Sr/^{86}Sr$ isotope ratios showed that all tagged fish with known freshwater history had ratios less than 0.7080 during their periods in freshwater. Thus, we used this ratio as the threshold for identification of freshwater residency. $^{87}Sr/^{86}Sr$ isotope ratio transects across the otoliths of barramundi caught in the estuary after the 2003 flood were examined. For fish that had been in freshwater, the age at migration into and out of freshwater were identified and the period in freshwater estimated. Based on fish age-at-capture, the year that the fish migrated into the estuary was also calculated. Differences in use of freshwater habitats among age-classes and growth rates ('fast growers' vs. 'slow growers') (**Table 8.1**) were made with tests of proportions (Hoel 1976).

Table 8.1 The total length (in mm ± se) of barramundi collected in the Fitzroy River estuary before and during the February 2003 flood and analysed for $^{87}Sr/^{86}Sr$ isotope ratios

Collection period	Age-class (years)	Growth cohort	Total Length ± se (mm)	N
Before flood	4	fast	912 ± 11	5
	4	slow	667 ± 27	7
After flood	4	fast	926 ± 10	10
	4	slow	663 ± 17	10
After flood	7	fast	1093 ± 42	3
	7	slow	844 ± 27	5

Results

A total of 29 tagged adult fish and 20 juveniles were examined for trace metals and 14 of these were reanalysed for Sr isotopes (Table 8.2). Adult fish were mostly chosen from two distinct groups, those caught and tagged in freshwater and subsequently caught in the estuary (ten fish) and those that were tagged and recaptured in the estuary (ten fish). Other fish analysed had been tagged and recaptured in freshwater impoundments and were known to be originally stocked (seven fish) or caught in super-saline water in a saltworks drying pond (two fish).

Table 8.2 Mean size (TL ± se in mm) and age (increments ± se) of tagged barramundi from different locations examined for otolith chemistry. Fish were grouped according to their known movements based on tagging and recapture locations.

Capture location classification	Size at capture (TL in mm)	Recapture location classification	Size at recapture (TL in mm)	Age at recapture (increments)	N
Freshwater, adult	174 ± –	Freshwater	833 ± 54	3 ± 0.5	7
Freshwater, adult	486 ± 28	Estuary	806 ± 47	4.3 ± 0.7	10
Estuary, adult	540 ± 29	Estuary	754 ± 40	4.9 ± 0.6	10
Estuary, adult	–	Hypersaline	755 ± -	4.5 ± –	2
Estuary, juvenile	–	Freshwater	140 ± 50	0	4
Freshwater, juvenile	–	Freshwater	79 ± 5	0	16

Trace metal concentrations at known times

The concentrations of trace metals and Sr isotopes in otoliths of tagged fish during periods of known freshwater and estuarine residency showed considerable variation (Table 8.3). The greatest differences in trace metal concentrations were between fish caught in freshwater and estuarine habitats. However, there were significant differences ($P<0.05$) among fish caught at the same site and in the same fish as it moved between habitats. Sr was the only trace metal that showed a consistent pattern of higher concentration in fish from known saline habitats. Mean Sr/Ca ratios were about 1 $mM.M^{-1}$ in freshwater and greater than 2 $mM.M^{-1}$ in the estuary (Table 8.3, Figure 8.2). Among fish that were tagged in freshwater and then recaptured in the estuary, mean Ba/Ca, Mg/Ca and Mn/Ca concentrations were generally lower at the time of capture ($P<0.05$; Table 8.3). However, the overall mean Ba/Ca, Mg/Ca and Mn/Ca variation among fish tagged and recaptured in the estuary were similar to that found in fish from freshwater.

The $^{87}Sr/^{86}Sr$ isotope ratios of fish from most freshwater habitats had similar mean ratios (Table 8.3). Fish tagged in estuarine habitats had $^{87}Sr/^{86}Sr$ isotope ratios similar to marine values (0.70918) except during floods when the isotope ratios were lower. The mean $^{87}Sr/^{86}Sr$ isotope ratio of fine particulate sediments from the peak of the 2003 flood was 0.70996 ± 0.00015) (Table 8.3). Replicate analyses of *Donax* shell had a mean $^{87}Sr/^{86}Sr$ isotope ratio identical to recorded marine values, giving confidence that the MC-ICPMS was giving precise and accurate data.

Otolith chemistry transects

Transects of Sr/Ca ratios of barramundi tagged in freshwater and recaptured in the estuary showed different patterns (Figure 8.2). Most fish had moved into low Sr/Ca ratio waters by one year old and most had only recently returned to the estuary. Among fish tagged and recaptured in the estuary, the patterns were more similar, varying annually (Figure 8.2). Only one fish tagged in the estuary (EST 5), appeared to have moved into freshwater after tagging and stayed there for about 2.5 years.

Figure 8.2 The Sr/Ca ratios (mM.M-1) of barramundi tagged in freshwater and caught in the estuary (a – e) and barramundi tagged and recaptured in the estuary (f – k)

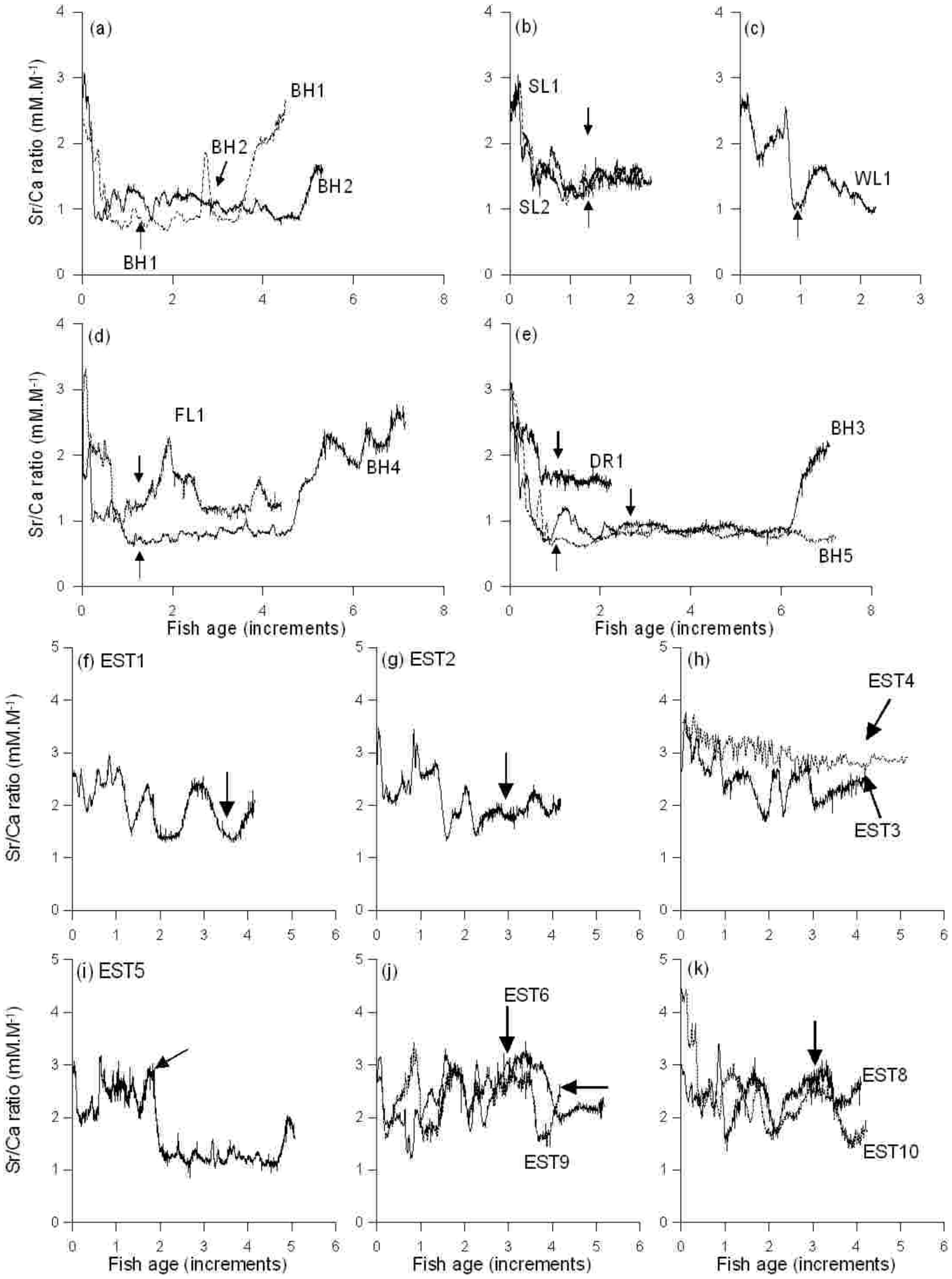


Figure 8.3 Ba/Ca ratios (µM.M-1) of barramundi tagged in freshwater and caught in the estuary (a – e) and barramundi tagged and recaptured in the estuary (f – k)

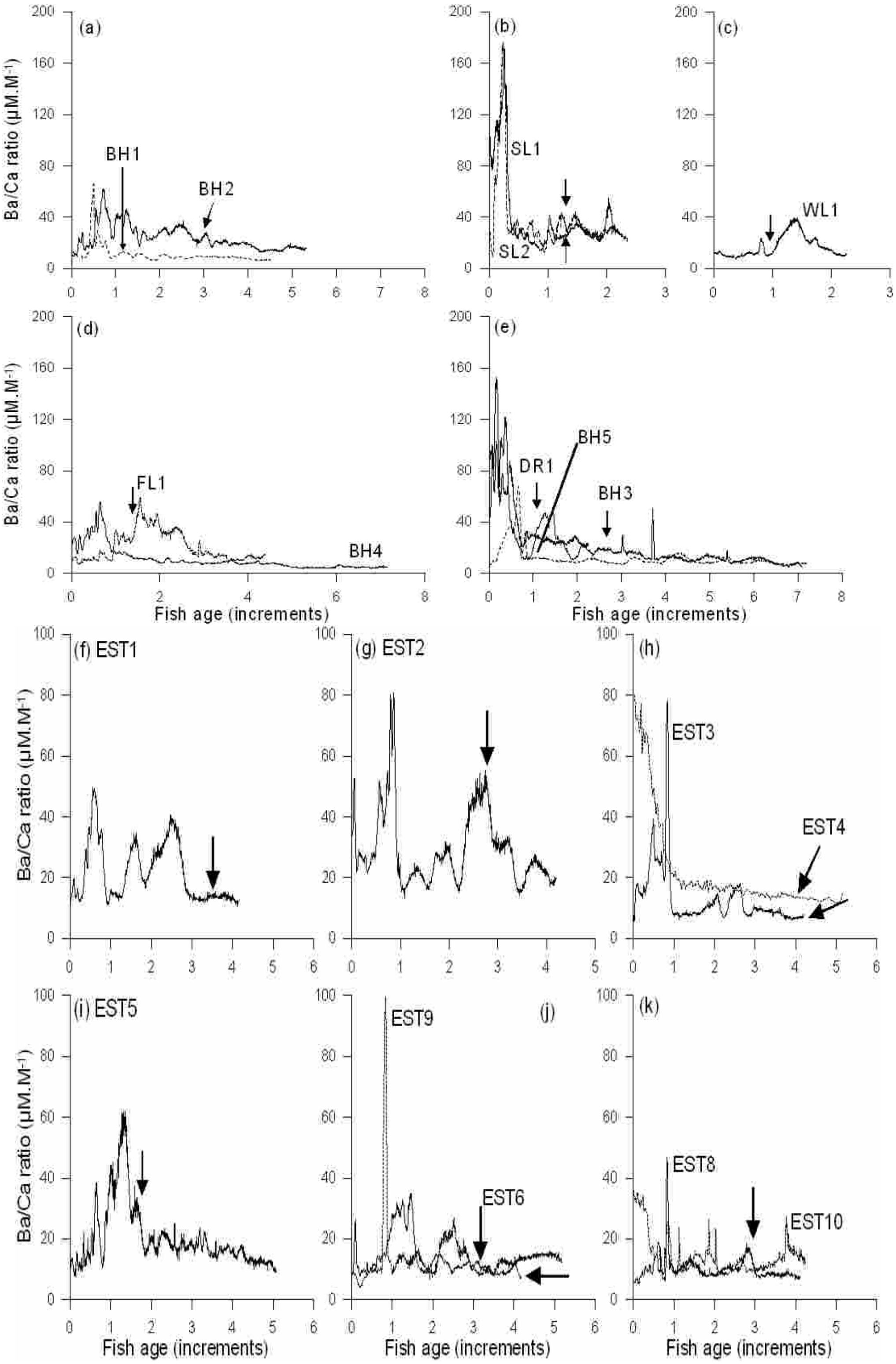


Figure 8.4 The Mg/Ca ratios (µM.M-1) of barramundi tagged in freshwater and caught in the estuary (a – e) and barramundi tagged and recaptured in the estuary (f – k)

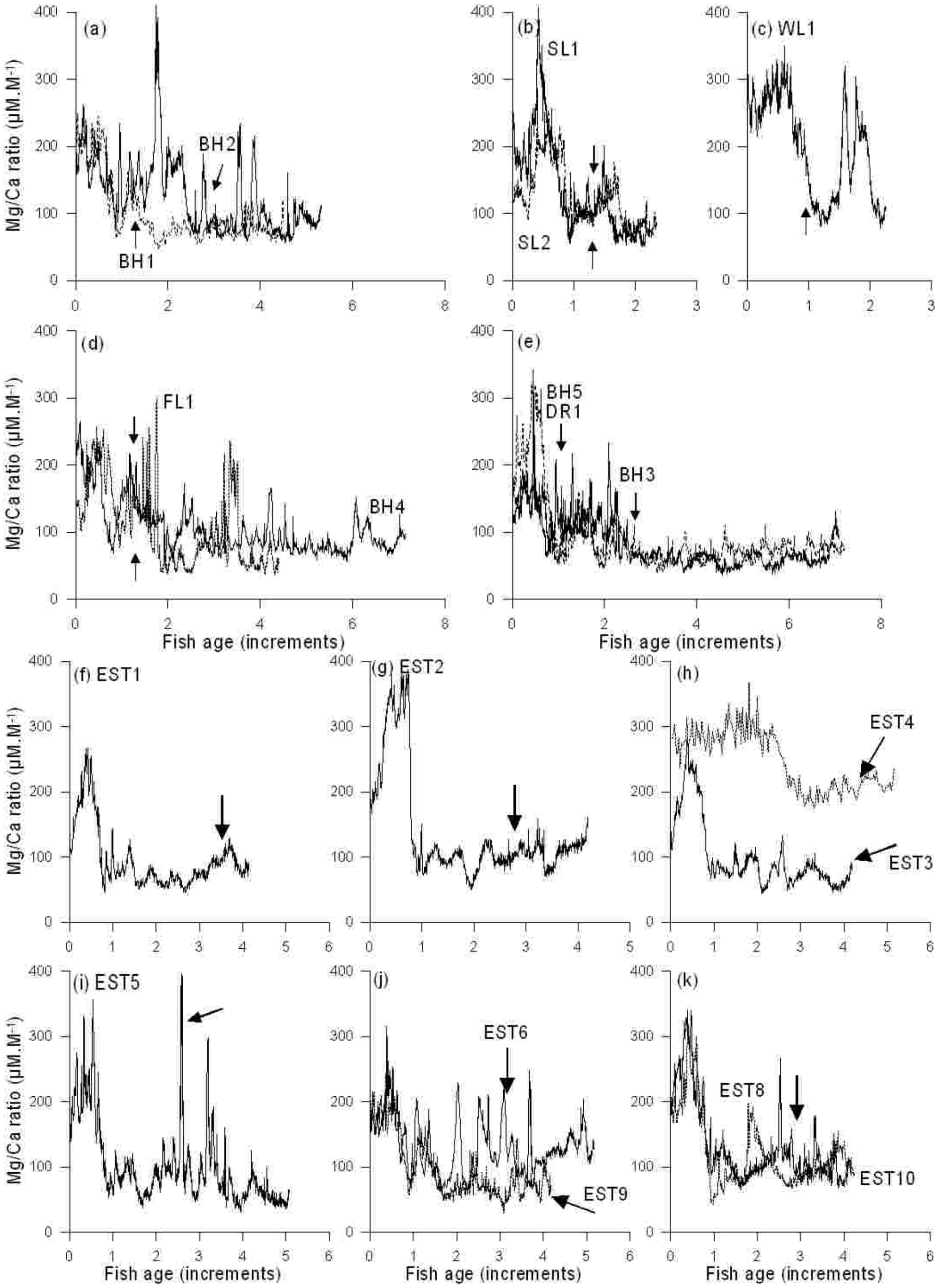
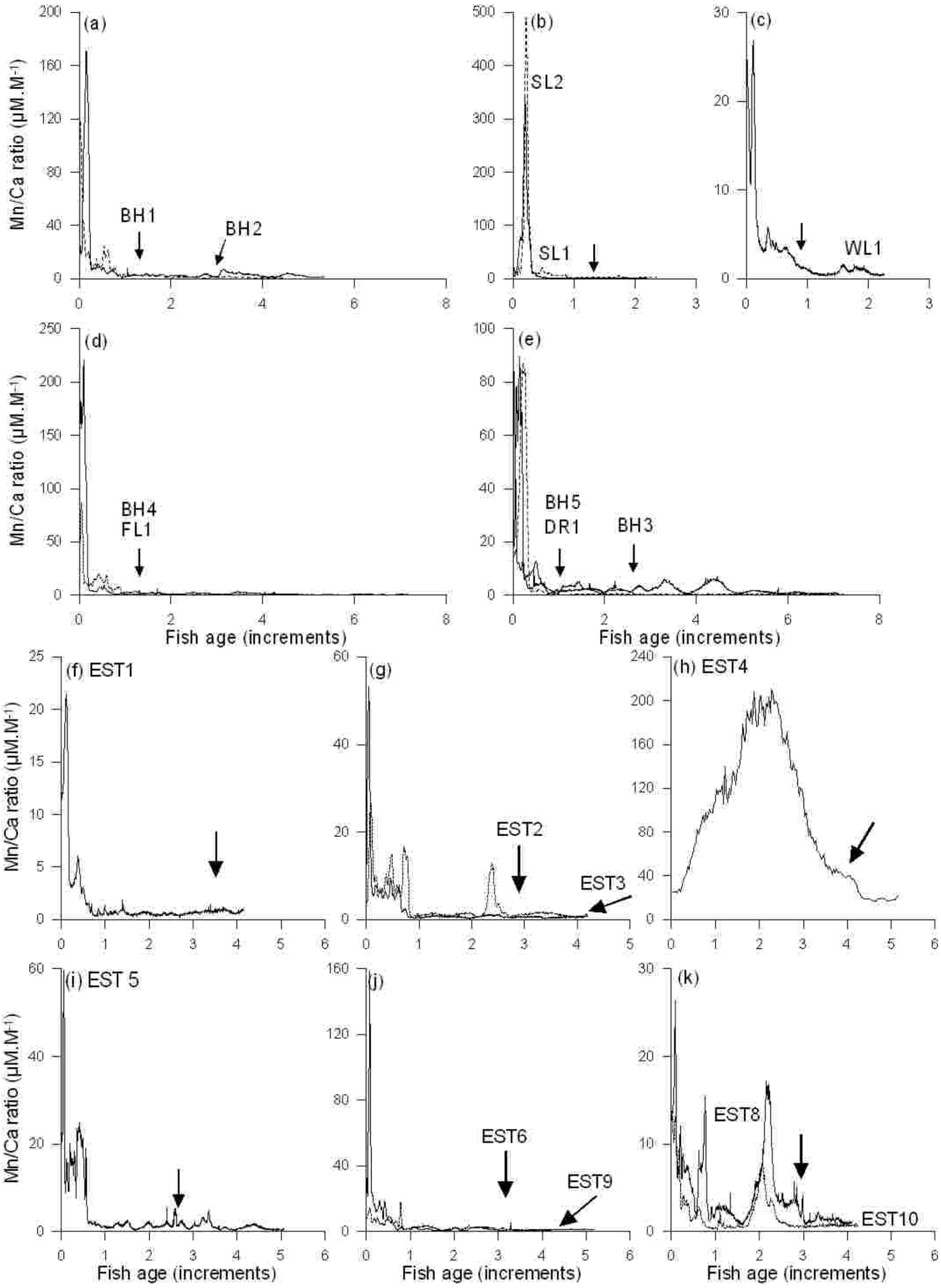


Figure 8.5 The Mn/Ca ratios (µM.M-1) of barramundi tagged in freshwater and caught in the estuary (a – e) and barramundi tagged and recaptured in the estuary (f – k)



The Ba/Ca, Mg/Ca and Mn/Ca transects appear to vary independently from Sr/Ca (Figure 8.3 – 8.5). All showed consistency between fish that had been tagged at the same site and recaptured in the estuary around the same time. For example, fishes SL1 and SL2 that were tagged the same day in Serpentine Lagoon (Figure 8.1) and recaptured the same month from the estuary. They had almost identical patterns in Ba/Ca, Mg/Ca and Mn/Ca ratios (Figure 8.3b – 8.5b).

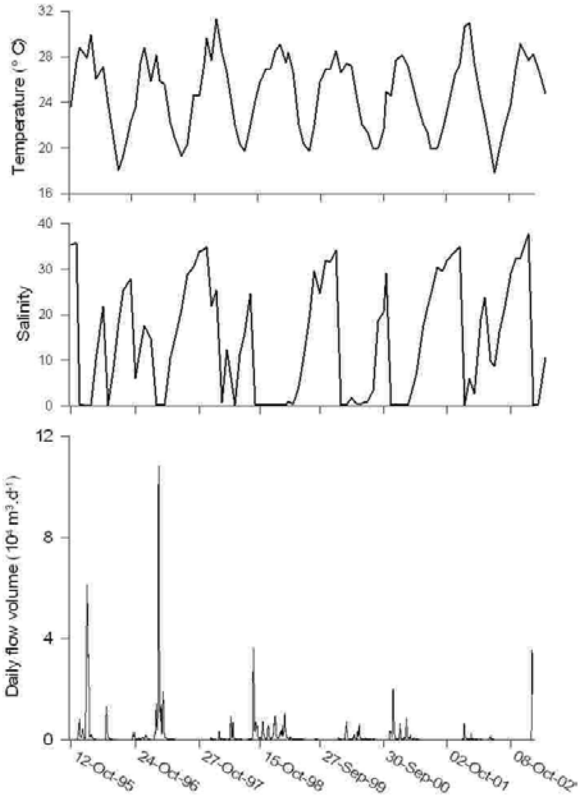
Unlike the other trace metals, Mn/Ca ratios showed a similar pattern among the majority of fish and the pattern was independent of habitat (Figure 8.5). Most fish had a spike in Mn/Ca ratios during the first three months of life, but the peak Mn/Ca ratio varied widely among fish. The only fishes that did not show this pattern were among those tagged in the estuary (EST 4, EST 8 and

EST 10). These fish had a peak in Mn/Ca ratios when the all three fish were aged two years (Figure 8.5).

Environmental variation

Temperature and salinity data were available from March 1993 until May 2003 and showed a similar pattern at the three sites within the estuary. The temperatures at the sites were highly correlated (all $R^2>0.96$; $P<0.0001$), but the salinities varied more and values changed depending on the distance from the sea (Figure 8.6). Both show a strong seasonal pattern with mid-year (autumn – winter) temperature minima coinciding with higher salinities. Low salinity corresponded with higher flow volumes (Figure 8.6). The period that salinities remained low varied with the extent of flow rather than actual flow volume. The timing of flood events and increased flow varied between years. It most commonly occurred in December-January, but could occur as early as in September (Figure 8.6).

Figure 8.6 Mean monthly salinity and temperature variation at a site in mid-estuary and daily flow volume measured in the upper estuary from September 1995 to February 2003



Correlations between otolith and environmental parameters at different scales

There were few strong correlations in the wavelet coefficients at the lower domains for any of the trace metals (Table 8.3). Only Sr/Ca coefficients showed any patterns that were consistent for all fish analysed. All fish had a significant positive correlation ($P<0.05$) between their Sr/Ca coefficients and temperature in domain 6 (Table 8.3).

Temperature was also the most commonly correlated environmental parameter with Ba/Ca and Mg/Ca coefficients in level 6 (Table 8.3). However, unlike the other trace metals, only the coefficients of a few fish tagged in freshwater were correlated with temperature and half were negatively correlated. Mn/Ca coefficients of few fish correlated significantly with any environmental variables. Unlike the other trace metals, there were few significant correlations between Mn/Ca coefficients and temperature in level 6.

Figure 8.7 The proportion of the tagged barramundi that showed statistically significant correlations ($P<0.05$) between Sr/Ca ratios in their otoliths and river flow ($m^3.d^{-1}$), temperature (° C), salinity and fish age (y) at scales from level 3 to 7 (plots a – e)

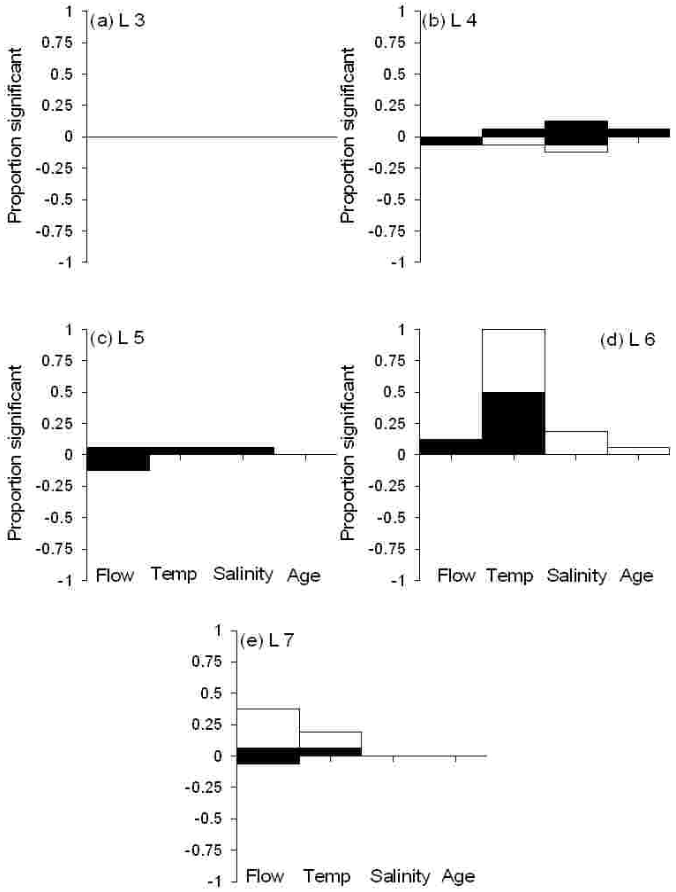


Table 8.3 The chemistry of barramundi otoliths (mean ± se) in freshwater and estuarine habitats during periods of known residency based on ^{87}Sr/^{86}Sr isotope ratios. Numbers in brackets are sample sizes.

Specimen	Site	Habitat	Capture date	Age	Sr/Ca (mM.M^{-1})	^{87}Sr/^{86}Sr	Mg/Ca (µM.M^{-1})	Mn/Ca (µM.M^{-1})	Ba/Ca (µM.M^{-1})
AW2	Awoonga Dam	Freshwater	9 Jun 01	1	1.0 ± 0.003	0.70634 ± 0.00002, (1010)	164.8 ± 1.7	0.8 ± 0.01	10.0 ± 0.09
AW4	Awoonga Dam	Freshwater	2 Sep 00	0	1.0 ± 0.010	0.70611 ± 0.00002, (953)	219.1 ± 2.4	2.1 ± 0.09	7.6 ± 0.07
	Awoonga Dam	Freshwater	11 Dec 04	5	1.1 ± 0.008	0.70586 ± 0.00001, (610)	92.4 ± 1.0	0.7 ± 0.11	6.8 ± 0.07
FL1	Frogmore Lagoon	Freshwater	13 Jan 98	2	1.2 ± 0.004	0.70592 ± 0.00001, (358)	144.6 ± 2.3	1.1 ± 0.02	26.1 ± 0.20
	Fitzroy River	Estuary	4 Feb 01	4	1.2 ± 0.008	0.70908 ± 0.00001, (403)	47.6 ± 1.9	0.6 ± 0.03	13.5 ± 0.20
FL2	Frogmore Lagoon	Freshwater	22 Mar 04	3	1.0 ± 0.003	0.70572 ± 0.00001, (366)	90.8 ± 2.7	0.5 ± 0.04	8.6 ± 0.09
FL4	Frogmore Lagoon	Freshwater	22 Mar 04	3	1.5 ± 0.010	0.70660 ± 0.00001, (133)	126.6 ± 1.8	0.5 ± 0.02	10.1 ± 0.10
DR1	Dawson River	Freshwater	28 Jan 02	1	1.6 ± 0.007	0.70998 ± 0.00001, (928)	109.5 ± 1.5	1.8 ± 0.02	25.8 ± 0.10
	Fitzroy River	Estuary	11 Feb 03	2	1.6 ± 0.005	0.70735 ± 0.00005, (10)	138.7 ± 4.5	3.0 ± 0.06	22.4 ± 0.20
BH1	Blacks Hole, Raglan Ck	Freshwater	28 Dec 96	2	0.9 ± 0.030	0.70652 ± 0.00001 (777)	122.9 ± 3.6	2.0 ± 0.1	11.8 ± 0.40
	Port Alma	Marine	23 Oct 00	5	2.4 ± 0.030	0.70913 ± 0.00006, (10)	77.4 ± 4.6	0.8 ± 0.02	6.9 ± 0.10
PL1	Powerline Ck	Estuary	14 May 03	0	2.9 ± 0.010	0.70917 ± 0.00002, (10)	187.2 ± 5.5	6.1 ± 0.1	23.7 ± 0.80
EST1	Fitzroy River	Estuary	8 Jan 02	3	1.6 ± 0.020	0.70763 ± 0.00002, (26)	92.4 ± 1.1	0.8 ± 0.04	13.1 ± 0.20
	Fitzroy River	Estuary	29 Oct 02	4	2.0 ± 0.050	0.70898 ± 0.00004, (10)	83.6 ± 2.0	0.8 ± 0.03	11.9 ± 0.20
EST2	Fitzroy River	Estuary	14 Aug 01	2	1.8 ± 0.010	0.70889 ± 0.00002, (58)	113.1 ± 1.8	0.6 ± 0.02	31.0 ± 0.40
	Fitzroy River	Estuary	29 Oct 02	4	2.0 ± 0.020	0.70922 ± 0.00003, (10)	138.9 ± 3.1	0.7 ± 0.02	18.8 ± 0.20
EST3	Fitzroy River	Estuary	12 Oct 02	4	2.4 ± 0.020	0.70895 ± 0.00004, (10)	74.2 ± 2.2	0.9 ± 0.04	7.0 ± 0.07
	Fitzroy River	Estuary	29 Oct 02	4	2.5 ± 0.020	0.70903 ± 0.00002, (10)	83.6 ± 2.1	1.1 ± 0.05	7.2 ± 0.10
EST8	Fitzroy River	Estuary	12 Oct 01	3	2.5 ± 0.030	0.70898 ± 0.00002, (50)	88.5 ± 2.2	0.6 ± 0.02	10.4 ± 0.20
	Fitzroy River	Estuary	29 Oct 02	4	1.8 ± 0.020	0.70914 ± 0.00002, (10)	96.6 ± 2.7	0.7 ± 0.05	13.2 ± 0.30
EST9	Fitzroy River	Estuary	11 Oct 02	4	2.8 ± 0.030	0.70921 ± 0.00002, (7)	44.5 ± 3.0	0.5 ± 0.03	8.8 ± 0.08
	Fitzroy River	Estuary	29 Oct 02	4	2.4 ± 0.020	0.70919 ± 0.00002, (6)	67.3 ± 2.1	0.5 ± 0.03	7.6 ± 0.10
EST10	Serpentine Ck	Estuary	13 Oct 01	3	2.7 ± 0.030	0.70873 ± 0.00003, (10)	92.6 ± 4.8	1.0 ± 0.05	7.8 ± 0.08
	Fitzroy River	Estuary	29 Oct 02	4	2.6 ± 0.040	0.70915 ± 0.00002, (10)	89.5 ± 2.5	1.0 ± 0.03	7.3 ± 0.10
Flood sediment (< 10 µ)	Fitzroy River	Freshwater	10-13 Feb 03	–	7.8 ± 0.700	0.70996 ± 0.00015, (12)	–	–	9.5 ± 1.30
Donax shell	–	Marine	–	–	–	0.70918 ± 0.00001, (3921)	–	–	–

Use of freshwater habitats

Overall, almost 60% of the fish analysed from the estuary with both trace metal/Ca and ^{87}Sr/^{86}Sr isotope ratios had spent at least two months in freshwater prior to capture. This was not significantly different from half the population ($P>0.10$). About half (47%) of 40 barramundi analysed before and after the 2003 flood had spent time in freshwater habitats. There was no difference ($P>0.70$) in the percentage of fish that had migrated to freshwater habitats within each age-class (4 yr olds: before flood – 45%, after – 49%; 7 yr olds: after – 47%; based on both trace metal/Ca and Sr isotope ratios). Thus, for subsequent analyses of freshwater migration, all fish were pooled.

Of those that had migrated to freshwater, most had done so during their first year (Figure 8.8). About 30% of these fish had only recently migrated back into the estuary during the flood as mature adults (4 yr olds). However, the age when barramundi migrated back into the estuary varied from young-of-the-year (i.e. the 0+ age-class) to 7 yr olds (Figure 8.8). Thus, barramundi that had migrated into freshwater spent varying time in that habitat (**Figure 8.9**), but most had returned to the estuary by sexual maturity (> 80%). Among the age-classes examined (4 and 7 yr olds), there were no significant differences in the year that fish migrate into the estuary (P>0.10). However, the largest percentage had migrated into the estuary during the 2003 flood (**Figure 8.9**).

Figure 8.8 The percentage (± se) of those barramundi caught in the estuary (that had accessed freshwater) that migrated into this habitat at each age (a) and then returned to the estuary (b)

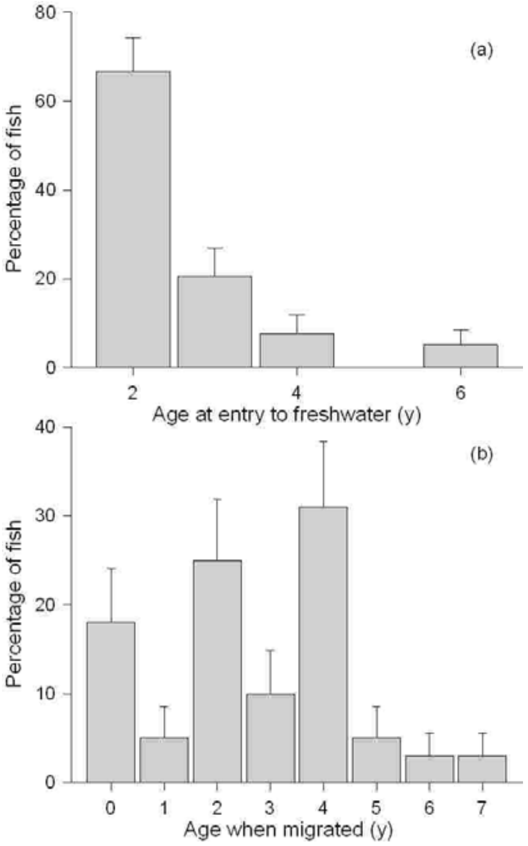


Figure 8.9 The percentage (± se) of those barramundi caught in the estuary (that had accessed freshwater) that had (a) spent different periods in freshwater and (b) the year that each fish returned to the estuary

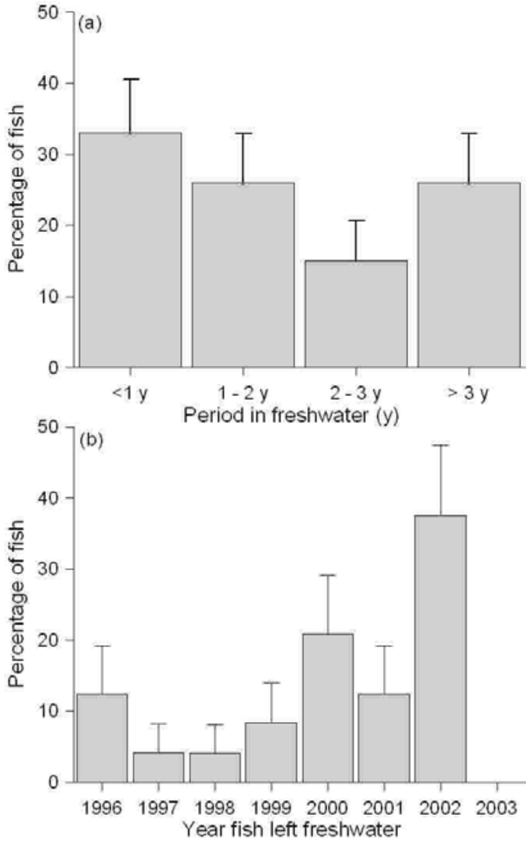
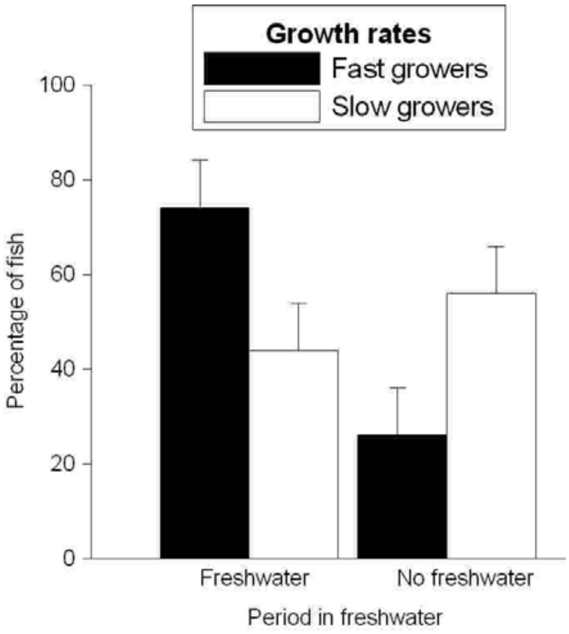


Figure 8.10 The percentage (± se) of fast and slow growing barramundi that had accessed freshwater or remained in the estuary throughout their life



A significantly greater proportion of the fast-growing four-year old fish analysed had spent a period in freshwater (P<0.05) (Figure 8.10). Among the slow-growing four-year olds examined, there were no difference in the percentage that had remained in the estuary, or spent time in freshwater (P>0.40).

Discussion

Validation of metal/Ca ratios

The marked variation in $^{87}Sr/^{86}Sr$ isotope ratios in most freshwater bodies from the stable ratio in estuarine/marine waters (0.70918) offers a powerful tool to identify periods of freshwater and estuarine residency by fish (Kennedy *et al.* 1997). We have used these differences in the Fitzroy River to confirm periods when tagged barramundi were in estuarine or freshwater habitats. This has then enabled us to assess the value of the trace metal/Ca ratios of fish for identifying fish habitat history. Our results show that Mn/Ca, Mg/Ca and Ba/Ca ratios in barramundi varied as much between individuals from the same known location as between freshwater habitats and the Fitzroy River estuary. Only Sr/Ca showed a pattern consistent with the expected pattern of higher Sr/Ca ratios in estuarine and marine waters (Milton and Chenery 2001).

Most studies of fish migration with otolith chemistry have relied on Sr/Ca ratios. However, recent studies have also found Ba/Ca ratios may be a sensitive indicator of freshwater residency (Elsdon and Gillanders 2005; Martin and Thorrold 2005). Ba/Ca ratios in barramundi otoliths reflect Ba/Ca ratios of water in which the fish is resident (Milton and Chenery 2001). Thus, our results suggest that the large variation in Ba/Ca ratios in the otoliths of barramundi (and often high values) when in the estuary may be related to the location of fish relative to the estuarine Ba maxima (Coffey *et al.* 1997; Colbert and McManus 2005).

Patterns of variation in all trace metal/Ca ratios across the otoliths of barramundi tagged in freshwater and the estuary had few significant correlations with the environmental variables at any scale. Previous otolith studies have found that salinity, temperature and growth rate (age) can influence otolith chemistry of some species (Campana 1999). The only consistent correlation we found occurred between otolith Sr/Ca and temperature at level 6. This corresponds with approximately monthly variation in most of the fish examined and was independent of fish habitat history.

Most fish migration studies usually examine annual or semi-annual changes in otolith chemistry (Secor and Rooker 2000). The variations in otolith chemistry at this scale were not influenced by temperature. This was despite temperature having a strong influence on the length of the growing season of barramundi in the Fitzroy River estuary (Robins *et al.* 2006). The finer scale correlations between temperature and otolith Sr/Ca ratios are likely to magnify the effects on otolith Sr/Ca of fish movement into higher saline water during summer spawning.

There was considerable fine-scale variation in otolith Ba/Ca and Mg/Ca ratios in barramundi otoliths that were not significantly correlated with any of the measured environmental variables. Some of the barramundi had a significant correlation between Mg/Ca ratios and river flow volume at level 4. This scale would approximate quarterly variation in most of the fish analysed. Of the barramundi that had a significant relationship at this level, fish tagged in the estuary had a positive relationship with flow, while freshwater fish had a negative relationship. This is consistent with the higher concentration of Mg in marine waters and follows the gradients in Mg found in other estuaries (e.g. Dorval *et al.* 2005). A lack of a consistent relationship for all fish may reflect the annual variable intensity or length of freshwater flow.

Role of freshwater flows

The utilisation of a combination of trace metal/Ca and $^{87}Sr/^{86}Sr$ isotope ratios to trace fish movements and habitat history greatly increases the potential to accurately identify critical habitats of migratory fish species (Bacon *et al.* 2004; Milton and Chenery 2005). $^{87}Sr/^{86}Sr$ isotope ratios can avoid many of the potential confounding effects detected in trace metal/Ca ratios (Barnett-Johnson *et al.* 2005), but their effectiveness is generally limited to waters with salinities below 5 and distinctive geology (Milton and Chenery 2003, 2005).

The $^{87}Sr/^{86}Sr$ isotope ratios in all barramundi analysed from the Fitzroy River estuary differed greatly from that measured in the 2003 flood (0.70996 vs. ~ 0.706). Tagged barramundi that were initially caught in coastal freshwater lagoons and creeks also had similar $^{87}Sr/^{86}Sr$ isotope ratios. This strongly suggests that these coastal freshwater systems are the major freshwater habitat for barramundi and their flooding has a significant role in maintaining barramundi recruitment. This is not surprising; despite the additions of a fish ladder on the main barrage in the Fitzroy River and studies demonstrating that barramundi do migrate upstream when river flow is sufficient (Kowarsky and Ross 1981; Stuart and Mallen-Cooper 1999). Our data indicate that only a very small proportion of the estuarine barramundi population have spent time in the Fitzroy River above the barrage. Few of the fish that had migrated upstream, or of the large number of hatchery-reared fish stocked in upstream impoundments appear to be returning to the estuary. Both Davis (1986) and Milton and Chenery (2005) have found that most barramundi from the middle and upper reaches of other unregulated rivers also do not migrate regularly to the sea to spawn.

The $^{87}Sr/^{86}Sr$ isotope ratios of barramundi caught after the 2003 flood that had spent time in freshwater habitats were all similar to those of tagged barramundi from the coastal lagoons. We also found that most barramundi that migrate into freshwater do so during their first year of life. This appears to be mostly after three months of age and these fish have all migrated into the estuary by sexual maturity at three to four years old (Dunstan 1959; Davis 1982). Our otolith chemistry (both trace metal/Ca and Sr isotopes) demonstrates that barramundi in the Fitzroy River use habitats of marine salinity during their larval and early juvenile phases. This habitat could include the extensive areas of supra-littoral pools and saltpans in the lower Fitzroy River estuary directly behind the mangrove fringe (Moore 1982; Russell and Garrett 1983, 1985). These types of habitats fill during the large spring tides that occur in between the October to February spawning season. Barramundi are known to time their spawning to coincide with periods of maximum tidal height (Moore 1982). Thus, these habitats would be saline during early growth, except following a large annual flood or heavy coastal rains. At that time, the whole lower coastal plain of the Fitzroy River can become inundated with freshwater and many barramundi would be able to redistribute to more permanent freshwater habitats.

Staunton-Smith *et al.* (2004) hypothesized three possible mechanisms to explain how barramundi year-class strength correlated with river flows and coastal rainfall. Firstly, by increasing the spawning population with migrants from land-locked freshwaters. Secondly, by increased survival of larvae and early juveniles through improved access to, or conditions in inundated supra-littoral nursery habitats. And lastly, by increased survival (and growth) of older juveniles and sub-adults by enabling them to access more productive permanent freshwater habitats (Salen-Picard *et al.* 2002). Our data and that of Milton and Chenery (2005) suggest that barramundi do not access freshwater during their larval and early juvenile stages. This would support the third hypothesis that access to more permanent freshwater lagoons by older juveniles is contributing to increased survival and growth of barramundi during years of high flow.

The return migration of barramundi from these more permanent local creeks into the estuary probably occurs during annual floods when fish activity greatly increases (Dunstan 1959) and the permanent lagoons reconnect to the main river. Barramundi that had spent time in these freshwater systems were significantly more likely to be fast-growing than fish that had remained in the estuary throughout their life. Thus, barramundi that mature in freshwater appear to take advantage of the higher productivity in these ephemeral systems (Gillanders and Kingsford 2002).

Barramundi born during years with above average floods and coastal rainfall had better survival than fish from poorer years (Staunton-Smith *et al.* 2004). Robins *et al.* (2006) have taken this further and shown that increased river flow also increases barramundi growth rates. Our study supports these findings and suggests that the growth of larger juveniles and immature fish is enhanced by improved access to productive coastal freshwater habitats during more substantial flood years.

In order to better manage the scarce freshwater resources in tropical Australia and provide advice on the optimal freshwater (including environmental flow allocations) to maintain estuarine fish populations, we need a better understanding of the relationship between flood characteristics (timing, magnitude and duration) and freshwater habitat use. We found that barramundi moved into freshwater habitats during most years and this did not appear to be correlated with flow volume. However, our sampling design was planned to explore differences in habitat use and movement of two strong year-classes (1996 and 1999) (Staunton-Smith *et al.* 2004). In order to estimate the freshwater flow that is associated with access to freshwater habitats by juvenile barramundi, a larger, more random sub-sampling of the entire barramundi population across years would be required, preferably with a greater contrast in flood flow conditions.

Chapter 9. Effects of freshwater flows on banana prawns

J. Robins, I. Halliday, D. Mayer, A. Colefax, M. Sellin and D. Vance

Summary

Freshwater flows are speculated to affect the biological productivity of estuaries, with one impact being the improved growth and survival of finfish and shellfish species. Penaeid prawns are a group of estuarine-dependent shellfish species, whose fishery catch production is frequently correlated to freshwater flow. The catch of banana prawns (*Penaeus merguiensis*) is highly correlated with rainfall or freshwater flow and this species is stimulated to emigrate from estuaries as a consequence of reduced salinity resulting from rainfall events and the subsequent freshwater flows. However, little attention has been given to the possible effects of freshwater flow on the growth rates of this species.

The effect of freshwater flows on the growth rates of banana prawns was examined for juveniles sampled from the Fitzroy, Calliope and Boyne River estuaries. Growth rates of juvenile banana prawns were estimated from length-frequency data and analysed using FiSAT II software. This chapter also examines annual patterns in the recruitment of juvenile banana prawns into estuaries and compares abundance estimates under different flow conditions.

The basic von Bertalanffy growth equation for juvenile banana prawns was significantly improved by expanding *K* (the growth rate parameter) to be a function of freshwater flow (or rainfall where flow data were not available) and temperature. Flow four-weeks prior to sampling (i.e. pre-period flow$_4$) significantly improved the fit of the basic von Bertalanffy growth equation in two of the three estuaries. Final models differed between estuaries and were a mix of quadratic and linear functions of flow (or rain) and temperature. Modelling prawn growth under hypothetical environmental scenarios (i.e. high flows *c.f.* low flows) suggested that the growth and weight (=biomass) of juvenile banana prawns was significantly greater during high flow conditions compared to low flow conditions, with major increases in biomass (i.e. up to a three-fold increase), with similar effects occurring at low, average and high temperature conditions.

The recruitment of juvenile banana prawns into the three sample estuaries varied between years, but the numerical abundance of juvenile banana prawns was estimated to be highest in the sampling years with the greatest flow. The results of the current chapter provide quantitative evidence that freshwater flows do affect the growth rates of juvenile banana prawns. Whether this is as a result of a trophic cascade, increased time for feeding through decreased predation or the provision of a salinity gradient is uncertain from the current research. The increased abundance of juvenile banana prawns in years of higher freshwater flow provides further evidence that freshwater flows to estuaries result in greater abundance and greater biomass of juvenile banana prawns, prior to their recruitment to coastal and offshore fisheries. Our ability to demonstrate multiple effects of freshwater flow on juvenile banana prawns provides strong evidence that improved growth rates probably led to greater survival and thus higher abundance of juvenile prawns in estuaries, as suggested by Browder *et al.* (2002).

Introduction

Banana prawns (*Penaeus merguiensis*) were selected as a species in which to investigate the relationship between freshwater flow (=river runoff) and production because: (i) previous studies indicate a highly significant correlation between the fisheries catch and summer rainfall (see Chapter 3; Gunter and Hildebrand 1954; Ruello 1973; Glaister 1978; Browder 1985; Vance *et al.* 1985; Gammelsrød 1992; Evans *et al.* 1997; Vance *et al.* 1998; Galindo-Bect *et al.* 2000); (ii) banana prawns have a one-year life-cycle simplifying the effects of lag-times between freshwater flows and changes in production; and (iii) banana prawns are opportunistic omnivores, being relatively low in the estuarine trophic chain and may show a relatively quick response to increased nutrients brought to the estuary by freshwater flows (see Chapter 2; Robins *et al.* 2005).

Production related to freshwater flow, as evidenced by changes in growth rates, has been demonstrated for an estuarine shellfish (i.e. oysters in Livingston *et al.* 2000), two bivalve mollusc (*Chione cortezi* and *C. fluctifraya* in Schöne *et al.* 2003) and some estuarine fish species (e.g. whiting see Chapter 6 and barramundi see Chapter 7, published as Robins *et al.* 2006).

The aim of the current chapter was to investigate the effects of freshwater flows on: (i) growth rates of juvenile banana prawns; and (ii) the recruitment of juvenile banana prawns into estuaries on an annual basis.

Life history of banana prawns

The life history of banana prawns is reviewed in detail in Chapter 2. From the review, we identified six hypotheses of the influence of freshwater flow on banana prawn populations. They were:

1) Stimulates downstream movement of juvenile and sub-adult banana prawns to offshore areas, potentially increasing their catchability by the otter-trawl fishery. The stimulus is possibly salinity changes, although flushing may occur at high flow rates.
2) Affects the transport of larvae through currents. Large freshwater flows may prevent larval immigration to estuarine habitats either by washing eggs and larvae away from the estuary or dilution of chemical cues.
3) Creates chemical signals for larvae to enter the estuary.
4) Affects the survival of newly settled post-larvae in estuarine habitats, with freshwater (i.e. salinity of 0‰) being unsuitable habitat for post-larvae.
5) Enhances biological productivity of the estuary, thereby increasing the availability of food in the estuary for post-larvae, juveniles and adolescents which results in improved growth, survival and ‘condition’ of the banana prawn population, potentially leading to larger biomass and better reproduction.
6) Affects the area of favourable habitat for juveniles and adolescents, potentially through larger areas of decreased salinity, the creation of a favourable salinity gradient (i.e. 5‰ to 30‰) or turbid conditions reducing predation, which increases the survival of juveniles and adolescents.

We grouped the above potential causal mechanism into effects on:

1) recruitment, by (a) washing away eggs and larvae thereby reducing larval immigration to the estuary (negative effect), or (b) stimulating the larvae to enter the estuary in response to chemical cues (positive effect);
2) catchability, by stimulating the downstream movement of juvenile and sub-adult banana prawns; and
3) productivity, through increased food availability from enhanced biological productivity of the estuary, resulting in improved growth and survival of post-larvae, juveniles and sub-adults.

Other factors also influence the abundance of banana prawns at various points throughout their life-cycle, including the relationship between spawning stock-size and larval abundance (i.e. stock-recruitment relationship); the effects of water temperature on spawning success, survival and growth of larvae, and post-larvae; and the effects of wind and tidal currents on the migration of larvae into estuaries. Therefore, although freshwater flows may have an impact on populations of banana prawns, it is likely that other factors will also contribute to variation in fisheries catch. Indeed, it is likely that these other factors may confound trends in banana prawn populations that are being driven by freshwater flows.

Commercial fishery

Banana prawns are a significant target species in the trawl fisheries of northern Australia. They are caught by otter-trawlers fishing in near-coastal waters along the east coast of Queensland, in the Gulf of Carpentaria and in Joseph Boneparte Gulf. Offshore otter-trawlers, especially those in the northern gulfs, catch primarily adult banana prawns. However, in southern districts of the Queensland east coast, juvenile banana prawns are also caught by beam-trawlers fishing in the downstream reaches of estuaries, during the ontogenetic migration of banana prawns from estuaries to near-shore coastal waters.

Methods

We were guided in the design of our sampling program by input from a steering committee and selected stakeholders (see Chapter 17). An experimental design workshop was held in November 2001, with a subsequent review of data collected-to-date in March 2003. All research activities were carried out under 'General Fisheries Permit No. PRM02360D' and 'Animal Ethics Approval No. Bribie54/11/02'.

Design of sampling strategies

Various methods of sampling juvenile prawns were discussed at the 2001 experimental design workshop. Fyke netting might be a method of estimating banana prawn population abundances, but investigations of sampling sites revealed that fyke netting would not be efficient because of logistical problems (i.e. a large tidal range of ~5 m, the presence of crocodiles and deep, soft, muddy substrates). Beam-trawling was used by the CSIRO during work on the banana prawn populations in the Gulf of Carpentaria. Vance and Staples (1992) reported that catches of banana prawns were influenced primarily by the tidal cycle, with a secondary influence by the day-night cycle. Banana prawns do not bury and Vance and Staples (1992) reported that catches were highest when water levels were low, with banana prawns being concentrated in the shallows adjacent to the waters edge.

Site investigations of the Fitzroy, Calliope and Boyne River estuaries suggested that beam-trawling for juvenile banana prawns was the only practical method of sampling that would allow sufficient samples to be taken within the four hour window when tides were at their lowest. Sampling needed to occur at low tide because juvenile banana prawns disperse into the mangrove forest as the tide floods the mangroves, making the prawns less catchable.

Equipment

Fishery-independent samples of juvenile banana prawns were collected using a beam-trawl with an opening of 1 m wide by 0.5 m high and a mesh size of 6 mm (Figure 9.1). The beam-trawl was towed in water ~0.5 to 1.0 m deep for two minutes (~50 m in distance) and the volume of water flowing through the mouth of the net was measured using a flow meter.

Figure 9.1 Fishery-independent sampling of banana prawns, showing: (a) beam-trawl equipment (1 m wide by 0.5 m high); and (b) types of sites sampled

Locations

Fitzroy River estuary

The Fitzroy River estuary is a large (~50 km AMTD in length), muddy, mangrove-lined estuary, and by necessity was divided into four regions to facilitate sampling over two days (Figure 9.2). Sampling sites were selected on the basis of: (i) presence of mangroves; (ii) presence of a runoff channel; (iii) depth; and (iv) suitability for beam-trawling (i.e. no snags).

Port Curtis estuaries

Sampling for banana prawns also occurred in Port Curtis estuaries. Port Curtis is a major fishing ground with significant and regular otter-trawl catches of banana prawns. Preliminary analyses of historic catch data, reported that Port Curtis had the strongest correlation between stream flow, rainfall and total banana prawn catch (see Chapter 3). Site investigations lead to eight sites being selected in the Calliope River estuary and eight sites being selected in Boyne River estuary in a section known as South Trees Inlet (Figure 9.3).

Figure 9.2 Location of beam-trawl sites in the Fitzroy River estuary

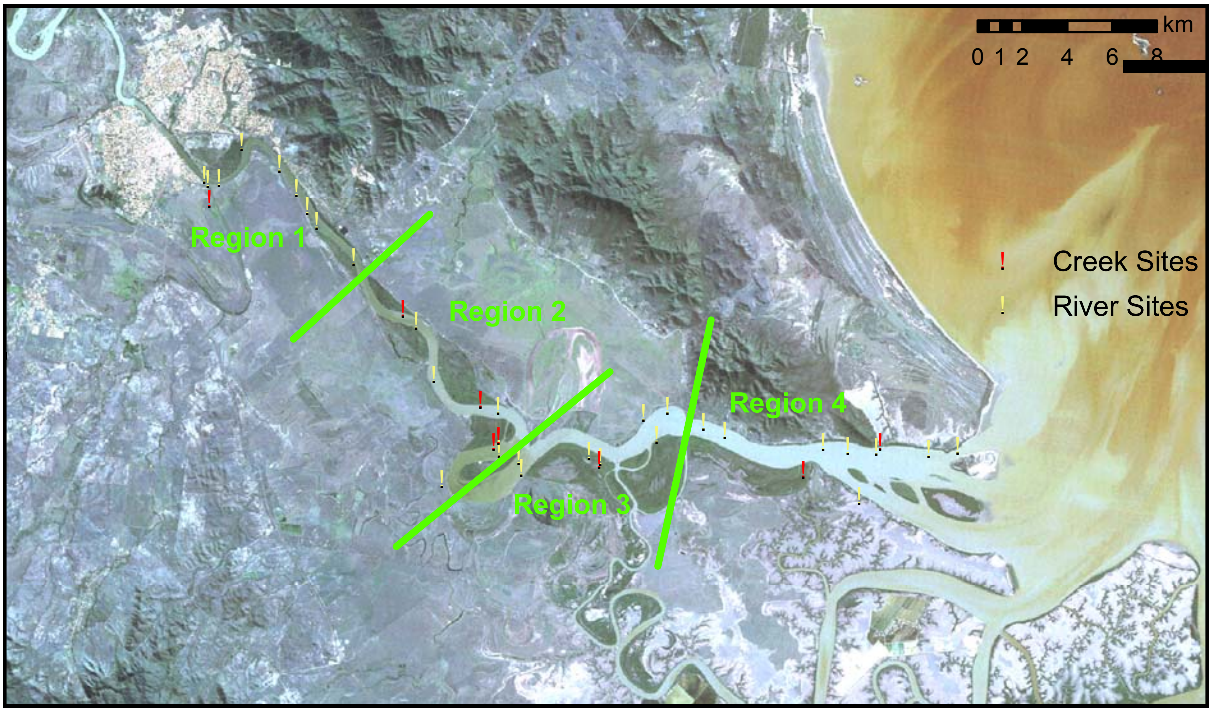
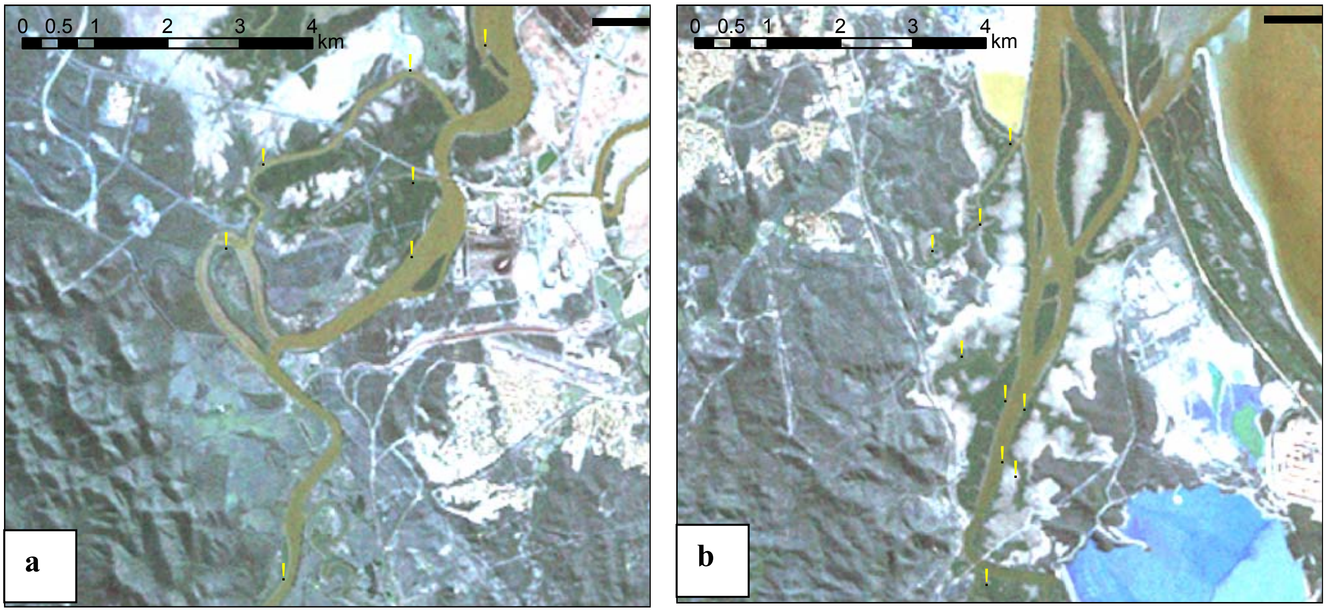


Figure 9.3 Location of beam-trawl sites in: (a) the Calliope River estuary; and (b) Boyne River estuary



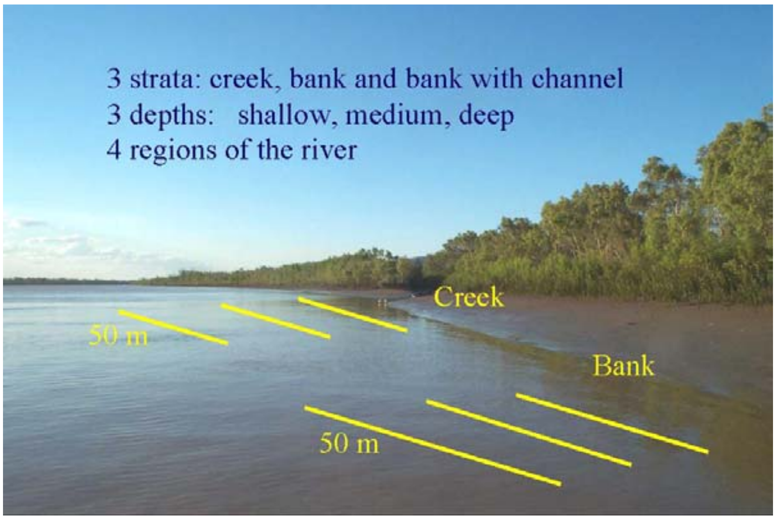
Frequency of sampling

The constraints of distance between sites, day-length and lag times of low tide within the Fitzroy River estuary resulted in regions 1 and 2 of the Fitzroy River estuary (i.e. upstream sites) being sampled on day-one of a trip and regions 3 and 4 of the Fitzroy River estuary (i.e. downstream sites) being sampled on day-two of a trip. Sites in the Calliope and Boyne River estuaries were sampled before or after sampling of the Fitzroy River estuary. Sampling occurred for a number of purposes (**Table 9.1**) and this dictated the frequency and location of sampling.

Spatial pilot

The spatial pilot used samples collected along transects of the Fitzroy River estuary to determine the most effective areas for the capture of juvenile banana prawns (**Figure 9.4**), and was investigated during the 1st and 2nd trips in year-1 (**Table 9.1**). Spatial strata thought to potentially influence the density of juvenile banana prawns were geomorphology (i.e. creeks, banks, and drainage channels of a bank) and water depth (i.e. shallow, medium, and deep). The relative density of juvenile banana prawns for each strata combination (i.e. adjacent geomorphology by water depth) was sampled within each of the four regions of the Fitzroy River estuary. Juvenile banana prawns were only captured at shallow water depths (i.e. <1 m). The relative density of juvenile banana prawns was greatest at sites adjacent to creeks or drainage channels of a bank. The preference of juvenile banana prawns for shallow water adjacent to creeks or drainage channels was consistent with experiences in the Gulf of Carpentaria (Vance and Staples 1992; Dave Vance, pers. comm. 2002). Therefore, we decided to sample only in shallow water (i.e. <1 m deep) adjacent to creeks and drainage channels of banks in the main river. In addition, we chose to sample in the major creeks adjoining the main arm of the Fitzroy, Calliope and Boyne River estuaries. Within a creek system, samples were collected from an upstream and downstream site.

Figure 9.4 Sites of spatial pilot sampling



Temporal pilot

The short-term temporal variability in the presence of juvenile banana prawns before or after the ebb tide was investigated during the 3rd, 4th and 5th trips (**Table 9.1**).

Standard sample

Based on the results of the spatial and temporal pilot studies, standard sampling involved sampling a set series of at least 23 sites. In sample year-1, we sampled the Fitzroy River estuary on a fortnightly basis (Table 9.1). In sample year-2, we expanded fortnightly sampling to the Calliope River and Boyne River estuaries. In sample year-3 and -4, we sampled the Fitzroy, Calliope and Boyne River estuaries, but sampled on a monthly basis, (i.e. new moon) to allow for annual comparisons, whilst adding sampling for temporal variation (see below).

Table 9.1 Date, location and purpose of beam-trawl sampling in the Fitzroy, Calliope and Boyne River estuaries (n.s. = no sampling)

Sample year	Start date	End date	Trip number	Fitzroy River	Calliope River	Boyne River	Purpose	Moon phase
1	11/01/2002	12/01/2002	1	√	n.s.	n.s.	Spatial pilot	New moon
1	13/01/2002	14/01/2002	2	√	n.s.	n.s.	Spatial pilot	New moon
1	28/01/2002	29/01/2002	3	√	n.s.	n.s.	Temporal Pilot	Full moon
1	30/01/2002	31/01/2002	4	√	n.s.	n.s.	Temporal Pilot	Full moon
1	02/02/2002	03/02/2002	5	√	n.s.	n.s.	Temporal Pilot	Last quarter
1	08/02/2002	11/02/2002	6	√	n.s.	n.s.	Standard sample	New moon
1	23/02/2002	25/02/2002	7	√	n.s.	n.s.	Standard sample	Full moon
1	10/03/2002	12/03/2002	8	√	n.s.	n.s.	Standard sample	New moon
1	24/03/2002	26/03/2002	9	√	n.s.	n.s.	Standard sample	Full moon
1	09/04/2002	11/04/2002	10	√	n.s.	n.s.	Standard sample	New moon
1	23/04/2002	25/04/2002	11	√	n.s.	n.s.	Standard sample	Full moon
1	08/05/2002	10/05/2002	12	√	n.s.	n.s.	Standard sample	New moon
1	22/05/2002	24/05/2002	13	√	n.s.	n.s.	Standard sample	Full moon
2	17/10/2002	19/10/2002	14	√	√	√	Standard sample	Full moon
2	02/11/2002	04/11/2002	15	√	√	√	Standard sample	New moon
2	15/11/2002	18/11/2002	16	√	√	√	Standard sample	Full moon
2	01/12/2002	04/12/2002	17	√	√	√	Standard sample	New moon
2	16/12/2002	19/12/2002	18	√	√	√	Standard sample	Full moon
2	03/01/2003	06/01/2003	19	√	√	√	Standard sample	New moon
2	16/01/2003	19/01/2003	20	√	√	√	Standard sample	Full moon
2	30/01/2003	02/02/2003	21	√	√	√	Standard sample	New moon
2	14/02/2003	17/02/2003	22	√	√	√	Standard sample	Full moon
2	27/02/2003	02/03/2003	23	√	√	√	Standard sample	New moon
2	15/03/2003	18/03/2003	24	√	√	√	Standard sample	Full moon
2	30/03/2003	02/04/2003	25	√	√	√	Standard sample	New moon
2	14/04/2003	17/04/2003	26	√	√	√	Standard sample	Full moon
2	29/04/2003	02/05/2003	27	√	√	√	Standard sample	New moon
2	13/05/2003	16/05/2003	28	√	√	√	Standard sample	Full moon
2	28/05/2003	31/05/2003	29	√	√	√	Standard sample	New moon
2	09/07/2003	12/07/2003	30	√	√	√	Standard sample	Full moon
3	23/10/2003	26/10/2003	31	√	√	√	Standard sample	New moon
3	22/11/2003	25/11/2003	32	√	√	√	Standard sample	New moon
3	20/12/2003	23/12/2003	33	√	√	√	Standard sample	New moon
3	17/01/2004	18/01/2004	34	√	√	n.s.	Temporal variation	New moon
3	19/01/2004	22/01/2004	35	√	√	√	Standard sample	New moon
3	23/01/2004	24/01/2004	36	√	n.s.	√	Temporal variation	New moon
3	16/02/2004	16/02/2004	37	n.s.	√	√	Temporal variation	New moon
3	17/02/2004	20/02/2004	38	√	√	√	Standard sample	New moon
3	18/03/2004	21/03/2004	39	√	√	√	Standard sample	New moon
3	22/03/2004	22/03/2004	40	n.s.	√	√	Temporal variation	New moon
3	15/04/2004	15/04/2004	41	√	n.s.	n.s.	Temporal variation	New moon
3	16/04/2004	19/04/2004	42	√	√	√	Standard sample	New moon
3	20/04/2004	20/04/2004	43	√	n.s.	n.s.	Temporal variation	New moon
3	16/05/2004	19/05/2004	44	√	√	√	Standard sample	New moon
4	11/10/2004	14/10/2004	45	√	√	√	Standard sample	New Moon
4	09/11/2004	12/11/2004	46	√	√	√	Standard sample	New Moon
4	10/12/2004	13/12/2004	47	√	√	√	Standard sample	New Moon
4	5/01/2005	6/01/2005	48	√	√	n.s.	Temporal variation	New Moon
4	7/01/2005	10/01/2005	49	√	√	√	Standard sample	New Moon
4	11/01/2005	11/01/2005	50	n.s.	n.s.	√	Temporal variation	New Moon
4	5/02/2005	5/02/2005	51	n.s.	√	n.s.	Temporal variation	New Moon
4	6/02/2005	9/02/2005	52	√	√	√	Standard sample	New Moon
4	9/03/2005	12/03/2005	53	√	√	√	Standard sample	New Moon
4	13/03/2005	13/03/2005	54	n.s.	√	n.s.	Temporal variation	New Moon
4	6/04/2005	9/04/2005	55	√	√	√	Standard sample	New Moon
4	5/05/2005	8/05/2005	56	√	√	√	Standard sample	New Moon

Temporal variation

In sample year-3 and year-4, we re-sampled sites within days to determine short-term variability in the abundance and spatial distribution of juvenile banana prawns within the estuaries. This sampling protocol (referred to as temporal variation in the 'Purpose' column of Table 9.1) was undertaken after advice from the steering committee. The aim of sampling temporal variation was to include random temporal replication into the hierarchy of our experimental design.

Measurement of prawns

All banana prawns caught were measured to a truncated 1 mm Carapace Length (CL) size-class (i.e. 1 mm CL = 1.00 to 1.99 mm CL). Other penaeid species caught were identified and measured to a truncated 1 mm CL size-class. Metapenaeids were grouped, owing to the difficulty of separating species at small size-classes (i.e. <5 mm CL). Length-frequency data for male and female juvenile banana prawns were pooled because juveniles <13 mm CL could not be reliably sexed by external examination and those that could be sexed comprised <8% of all individuals caught.

Analyses

Length-frequency analysis

The total catch of banana prawns for each size-class within each region (i.e. four within the Fitzroy River estuary and one each from the Calliope and Boyne River estuaries) was calculated and standardized to density per 1000 m trawled. The catches for each size-class group were averaged across the four regions of the Fitzroy River estuary, and for the Calliope and Boyne River estuaries respectively. Average catches per trip were used in the final cohort analyses.

Growth rates of juvenile banana prawns sampled between January 2002 and May 2005 were estimated using Modal Progression Analysis by measuring the shift of modes or means in the time series of length-frequency samples. There are three main stages in this procedure. Firstly, the decomposition of composite length-frequency distributions into their components to identify means. Secondly, the subjective identification and 'linking' of the means perceived to belong to the same cohorts. And thirdly, the use of growth increments resulting from the linking of estimated growth parameters.

We used the FISAT II (FAO-ICLARM) software package to carry out the length-frequency analysis and used Bhattacharya's method to decompose the composite length-frequency distributions into their components. This is an iterative process involving visual identification of cohorts and subtraction of each component from the remainder of the sample using a Gaussian function. Mean lengths, population sizes (in numbers), standard deviations and separation index (SI) for the cohorts identified were calculated.

Sometimes, only one or two components seemed to be present in a sample but inspection of the samples taken before and after suggested more cohorts. In this case, the procedure was carried out again but with extra cohorts fitted as suggested by the adjacent samples. After calculation of the mean lengths of each cohort, the appropriate means from each cohort were linked to enable calculation of growth rates between each pair of means. A mean growth rate was then calculated for all pairs of samples. A linear model of growth in CL is a satisfactory model for the growth of juvenile banana prawns because increases in the moult increment offsets slower moulting rates associated with larger size-classes of prawns (Staples and Heales 1991).

Estimating the effect of flow on growth rates

The von Bertalanffy growth equation remains a key method of analysing growth data. Alternate methods such as the Schnute model (Ratkowsky 1986) rely on the estimation of growth rates at two points on the growth curve, and as such are not well suited to an analysis of factors which affect growth. Known problems with the von Bertalanffy include a postulated relationship between the variance of the residuals and the fitted values (Sainsbury 1980 – but this did not occur with our data), and the correlation between K (the estimated growth rate) and L_∞ (the population's average asymptotic length).

Initially, data exploration was conducted using graphical methods and general linear models of growth rate. The nonlinear von Bertalanffy model was adopted as the base model for growth, and provided a good fit for each of the estuaries. The form of the von Bertalanffy model used was:

$$\text{Length}_2 = \text{Length}_1 + (L_\infty - \text{Length}_1) * [1 - \exp (-K * \text{Days} + b)]$$

where the parameters to be estimated were L_∞ (the asymptotic length), K (the growth rate), and b (an intercept coefficient).

This form is preferred on statistical grounds to the more usual version of the von Bertalanffy model (which has 'growth' as the dependent variable), as in that form Length_1 appears on both sides of the equation.

However, in many of the exploratory models for the different data sets, convergence problems were experienced due to the correlations between K and L_∞. This indicates that there is a range of combinations of these parameters which fit each data set approximately equally well. As any direct and valid comparisons of alternate models for K need to be made at a consistent value of L_∞, we adopted an L_∞ for banana prawns of 38 mm (Lucas *et al.* 1979; Haywood and Staples 1993; Tanimoto *et al.* 2006) for all analyses.

Next, the growth rate (K) was expanded to become a function of the other independent terms, namely freshwater flows (or rainfall for the Boyne River estuary), temperature, salinity, pH and turbidity. Daily freshwater flow (ML) to the Fitzroy River estuary was estimated as the gauged streamflow at the most downstream gauging station (i.e. Wattlebank, *c.* 23.100°S; 150.133°E, 139 km adopted middle thread distance) minus the estimated seasonal downstream extractive uses provided by the Department of Natural Resources and Water and Fitzroy River Water, whilst daily freshwater flow to the Calliope River estuary was estimated as the gauged streamflow at the most downstream gauging station (i.e. Castlehope, *c.* 23.983°S; 150.083°E, 32.8 km adopted middle thread distance). Freshwater flow data were summed to give total freshwater inflow to the estuary for the period between dates when each cohort was sampled (indicated as flow_0).

There is possibly a delay between the timing of a freshwater flow and subsequent effects on the banana prawn growth rates (i.e. a lagged effect) as a consequence of nutrient input and trophic transfer (Drinkwater and Frank 1994). To include potential lagged effects in model screening, freshwater flows were aggregated for the two-, four- and six-weeks prior to the first sampling date of a cohort (indicated as flow_2, flow_4, and flow_6 respectively), and were included as potential terms in the model. They variables are referred to hereafter as pre-period flows.

Rainfall was used as an index of freshwater input to the Boyne River estuary because freshwater flow data were limited and did not cover the full sampling period. Daily rainfall was obtained from the Australian Bureau of Meteorology for two rainfall stations (i.e. station no. 39123, *c.* 23.850°S; 151.267°E and station no. 39297, *c.* 24.533°S; 151.383°E) and averaged to give an estimate of mean daily rainfall within the Boyne River catchment. Mean daily rainfall was summed to give total rainfall for the period between dates when each cohort was sampled and log-transformed to normalise the data (indicated as $\log_{10}\text{rain}_0$). Totals were aggregated for two-, four- and six-weeks prior to sampling (indicated as $\log_{10}\text{rain}_2$, $\log_{10}\text{rain}_4$, $\log_{10}\text{rain}_6$) and referred to hereafter as pre-period rain. Water quality parameters (i.e. temperature, salinity, pH and turbidity) were averaged from values measured during each sampling trip.

Step-forward general linear modelling (GenStat 2005) was utilised, with terms being included as quadratics to allow for curvature. At each step, the most significant term was added, and then the other terms re-screened. One-tailed significance testing was used for flows and rainfall (due to their expected positive effect on growth), with two-tailed being used for all other terms. Thus, the final models for each estuary were in the form of:

$$\text{Length}_2 = \text{Length}_1 + (L_{\infty} - \text{Length}_1) * [1 - \exp(-\{a + c * \text{independent term}_1 + d * \text{independent term}_1^2 + f * \text{independent term}_2 + g * \text{independent term}_2^2\} * \text{Days} + b)].$$

There are problems associated with attempting to measure growth rates using length-frequency analysis. The accurate identification of cohorts is made difficult if migration of prawns is occurring into or out of the population (Haywood and Staples 1993). In the present study, the problem was minimized because samples were taken over a large area and multiple sites within the each estuary. Although there would undoubtedly have been movement of juvenile banana prawns downstream during the sampling period, and eventually out of the sampling area (i.e. emigration), they would have remained within the sampling area for much of the sampling period. In addition, sample sites were distributed extensively throughout the estuaries (**Figure 9.2**; **Figure 9.3**) to provide large spatial coverage of estuarine resident juveniles. Although it is likely that the natural mortality rate of banana prawns changes with age or size (Wang and Haywood 1999), the current method assumed that there was negligible effects of size-selective mortality over the range of length-classes (as per Staples 1980).

Calculation of an annual index of abundance

Banana prawns recruit to an estuary over a period of time greater than a single sampling trip. Therefore, we took the approach of generating an annual index of abundance of juvenile banana prawns by calculating the area under of curve of the predicted mean prawn densities (stratified in various ways) accumulated over the full sampling season. We used a general linear model (GLM) of $\log_{10}$(prawn density+1) with the factors of time (i.e. trip number), spatial location within the estuary (i.e. region), and habitat-type (i.e. creeks versus rivers) to provide estimates of the annual mean density (and standard errors) of juvenile banana prawns per estuary. Preliminary analysis of fortnightly and monthly data indicated that estimates of the annual index of abundance differed. In general, indices based on monthly new moon data were greater than indices based on monthly full moon data, with indices based on fortnightly data (new and full moon) lying in-between (see Appendix 9). This was consistent regardless of the factors included in the GLM used to estimate the mean. As a consequence and considering practical limitations, sampling from year-3 onwards occurred at monthly intervals on the new moon. Thus, annual indices of the abundance of juvenile banana prawns presented in this chapter use monthly new moon data. We considered that the annual index of abundance needed to be adjusted to account for different duration of sampling seasons. Therefore, the annual index of abundance of juvenile banana prawns per estuary was calculated as the sum of estimated monthly mean prawn density weighted by the number of days between sampling trips.

Results

A total of 56316 banana prawns were caught across the three study estuaries between January 2002 and June 2005. A further 33884 prawns were caught, with ~62% of these being metapenaeids. Banana prawns ranged in size from 1 mm truncated CL size-class (i.e. 1 mm CL = 1.00 to 1.99 mm CL) to 33 mm truncated CL size-class. The vast majority of banana prawns caught were juveniles (i.e. >3 mm CL as per Meager *et al.* 2003), with only 365 being post-larvae.

Within the Fitzroy River estuary, a total of 946 trawl shots were conducted over the four years, with 603 occurring on the new moon. In the Calliope River estuary, a total of 283 shots were conducted over three years, with 188 occurring on the new moon. Similarly, a total of 270 trawl shots were conducted in the Boyne River estuary, with 186 occurring on the new moon. Juvenile banana prawns were patchily distributed throughout the study estuaries in time and space, with mean abundance (standardized to number per 1000 m trawled) being highly variable between trips and amongst sites, as evidenced by the large standard errors associated with trip means (**Figure 9.5**).

Length-frequency analysis

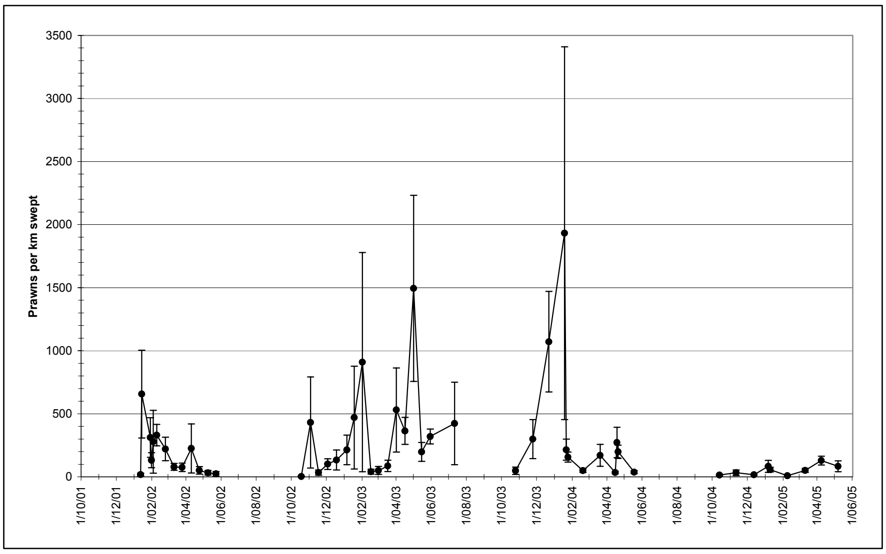
Length-frequency histograms were plotted in FISAT II and mean carapace length of each cohort identified (Figure 9.6, **Figure 9.7**, **Figure 9.8**). From the length-frequency histograms, 31 pairs of mean lengths were used to calculate fortnightly or monthly growth rates of juvenile banana prawns in the Fitzroy River estuary (**Figure 9.9**). A similar process was applied to the length frequency histograms for the Calliope and Boyne River estuaries, resulting in 11 pairs of cohort mean lengths for each estuary (**Figure 9.9**).

Catch data were used to identify the model progression with the least possible number of cohorts per sample year. This was done to limit the number of subjective cohorts that were forced into the data. Although sampling was carried out in the Calliope River estuary and Boyne River estuary for three years, only sufficient banana prawns were collected in year-2 and year-3 to develop growth curves for these rivers. The variance of the residuals was randomly related to the fitted values (see Appendix 9) and was not an issue that needed to be considered for our data.

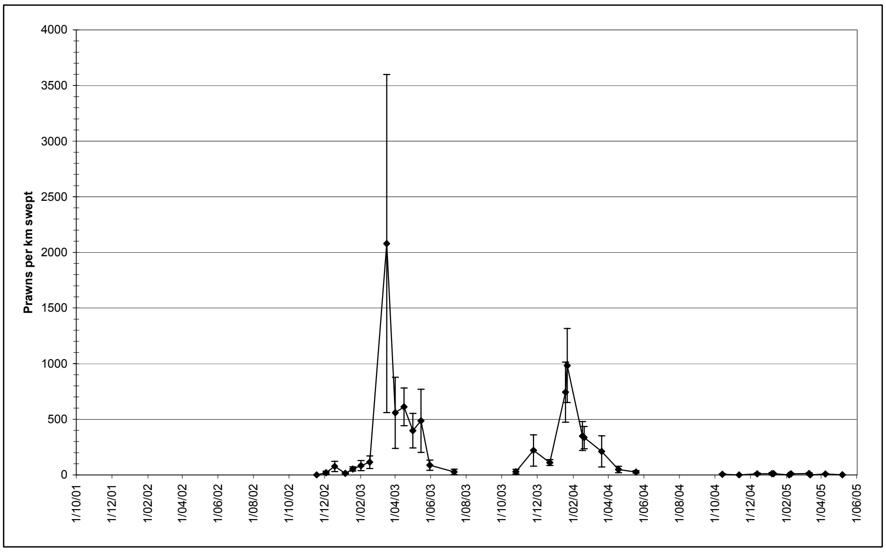
Mean growth rates of cohorts ranged between 0.29 mm CL wk^{-1} and 1.76 mm CL wk^{-1} (**Table 9.2**), while the overall mean growth rate for all cohorts across all estuaries was 0.727 mm/week (± 0.04 standard error).

Figure 9.5 Mean catch (and standard errors) of banana prawns per trip sampled from: (a) the Fitzroy River estuary, (b) the Calliope River estuary; and (c) the Boyne River estuary

(a) Fitzroy River estuary



(b) Calliope River estuary



(c) Boyne River estuary

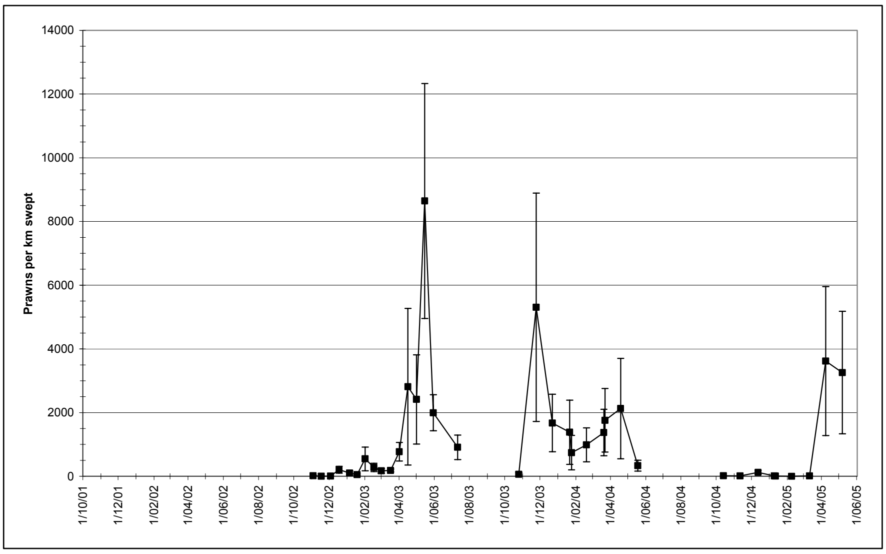
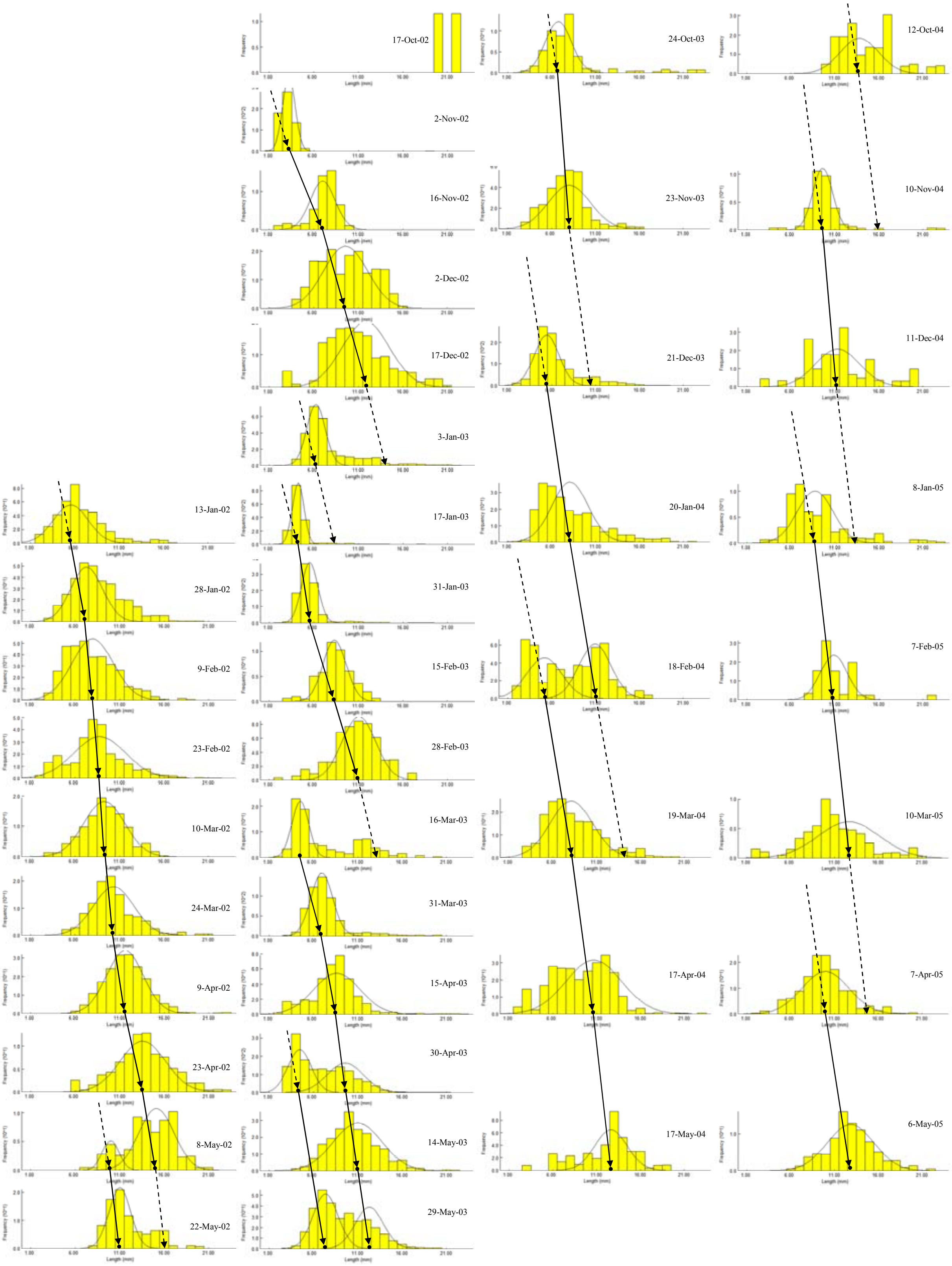
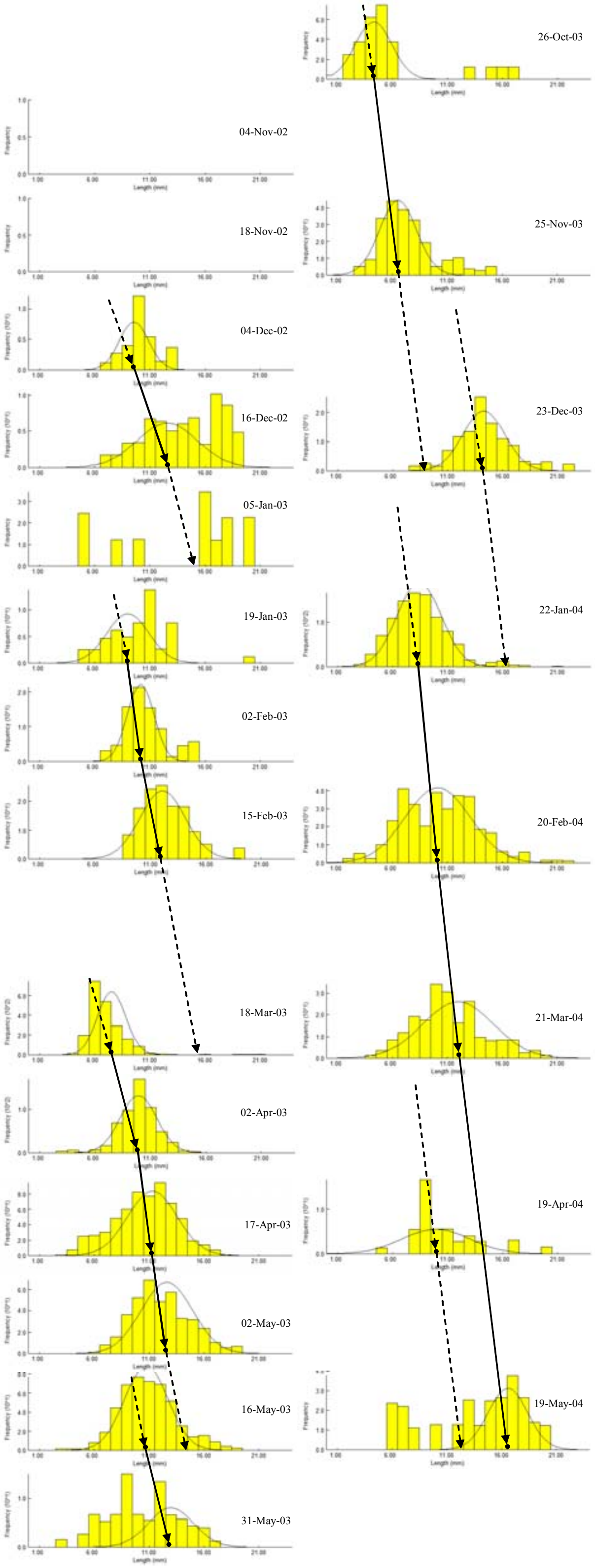


Figure 9.6 Length-frequency histograms for juvenile banana prawns sampled from the Fitzroy River estuary. x-axis is carapace length (mm) and y-axis is frequency.



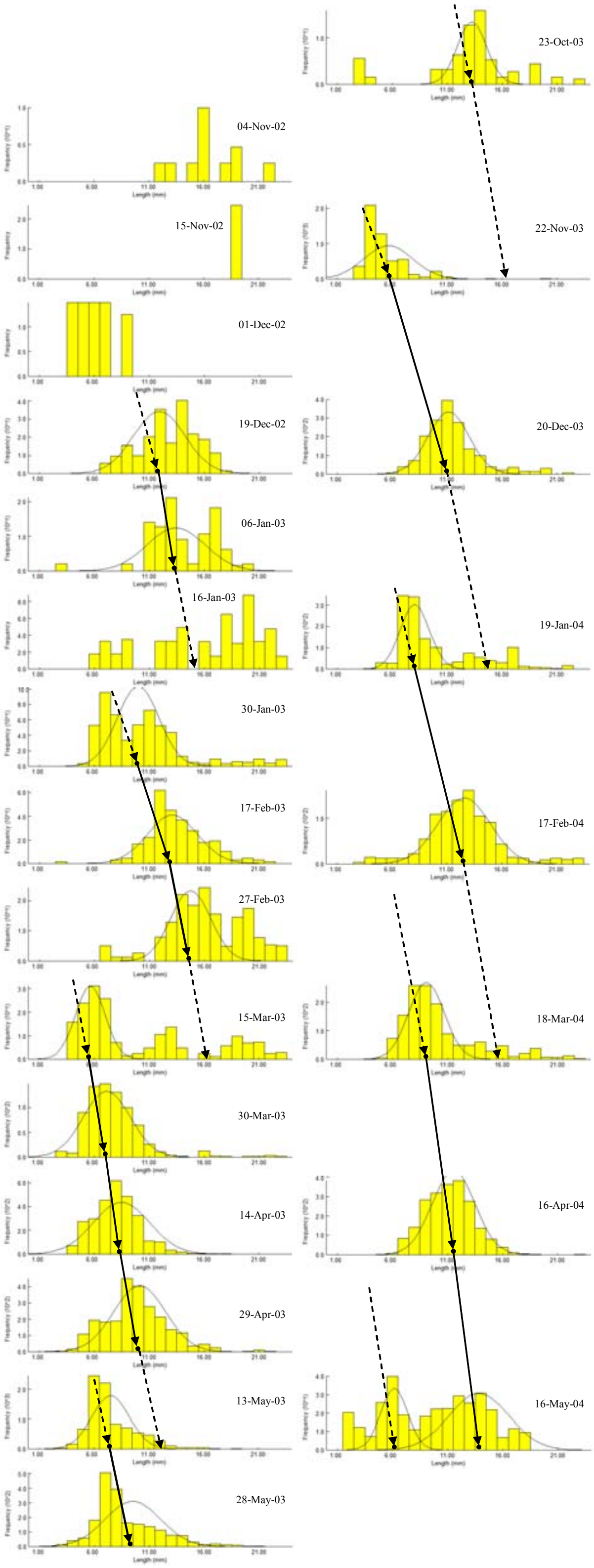
Fortnightly samples collected from January 2002 to May 2003 and monthly samples from October 2003 to May 2005. Mean carapace length of each cohort indicated by (●). Arrows link successive cohorts.

Figure 9.7 Length-frequency histograms for juvenile banana prawns sampled from the Calliope River estuary. x-axis is carapace length (mm) and y-axis is frequency.



Fortnightly samples collected from October 2002 to May 2003 and monthly samples from October 2003 to May 2005. Mean carapace length of each cohort indicated by (●). Arrows link successive cohorts.

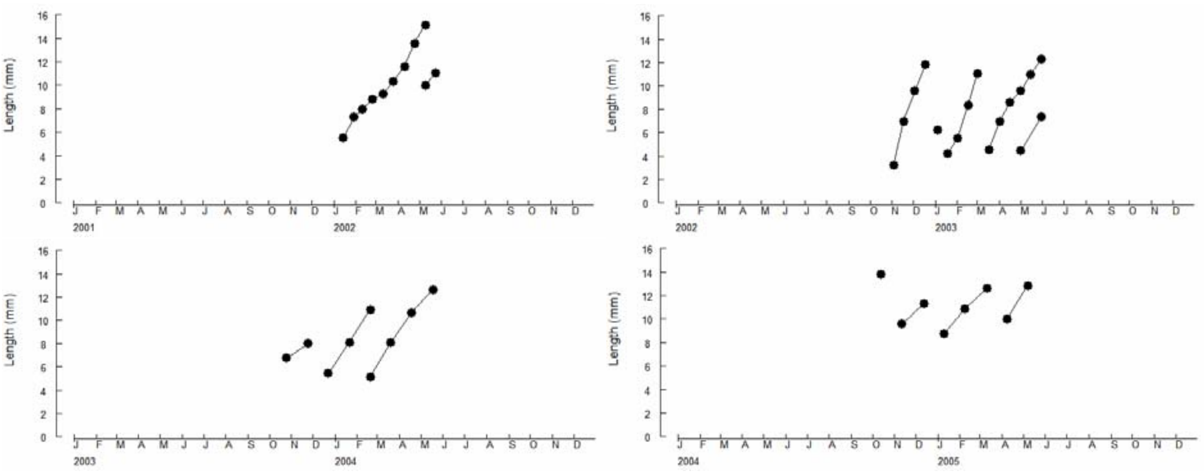
Figure 9.8 Length-frequency histograms for juvenile banana prawns sampled from the Boyne River estuary. x-axis is carapace length (mm), y-axis is frequency.



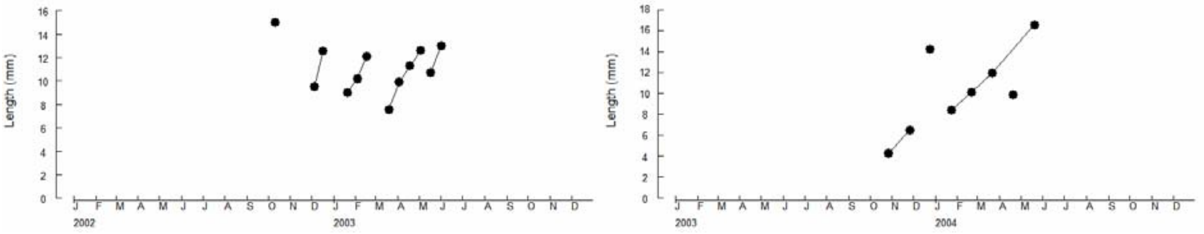
Fortnightly samples collected from October 2002 to May 2003 and monthly samples from October 2003 to May 2005. Mean carapace length of each cohort indicated by (●). Arrows link successive cohorts.

Figure 9.9 Mean carapace lengths for samples of banana prawns in the: (a) Fitzroy River estuary, (b), Calliope River estuary and (c) Boyne River estuary

(a) Fitzroy River estuary



(b) Calliope River estuary



(c) Boyne River estuary

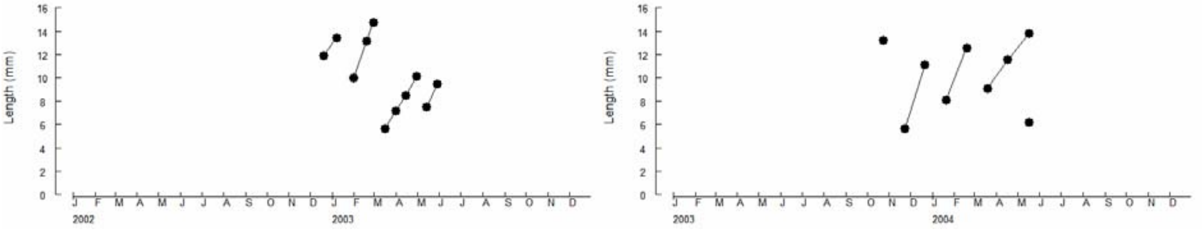


Table 9.2 Estimated mean growth rates of cohorts of juvenile banana prawns sampled between January 2002 and May 2005 from three estuaries on the east coast of Queensland. n = number of times a cohort was sampled.

Estuary	Sample year	Cohort no.	Settlement date	Initial mean carapace length (mm)	Last date sampled	Final mean carapace length (mm)	(*n*)	Growth rate (mm/week)
Fitzroy River	Yr-1	1.1	13/01/2002	5.53	08/05/2002	7.29	9	0.58
		1.2	08/05/2002	9.98	22/05/2002	11.02	2	0.52
	Yr-2	2.1	02/11/2002	3.19	17/12/2002	11.85	4	1.36
		2.2	17/01/2003	4.23	28/02/2003	11.05	4	1.14
		2.3	16/03/2003	4.51	29/05/2003	12.3	6	0.74
		2.4	30/04/2003	4.44	29/05/2003	7.37	2	0.71
	Yr-3	3.1	24/10/2003	6.76	23/11/2003	8.01	2	0.29
		3.2	21/12/2003	5.43	18/02/2004	10.94	3	0.53
		3.3	18/02/2004	5.15	17/05/2004	12.64	4	0.59
	Yr-4	4.1	12/10/2004	14.2	-	-	1	-
		4.2	10/11/2004	9.60	11/12/2004	11.30	2	0.38
		4.3	8/01/2005	8.74	10/03/2005	12.64	3	0.45
		4.4	07/04/2005	9.97	6/05/2005	12.83	2	0.69
Calliope River	Yr-2	2.1	18/10/2002	14.4	-	-	1	-
		2.2	04/12/2002	9.56	16/12/2002	12.57	2	1.76
		2.3	19/01/2003	9.01	15/02/2003	12.09	3	0.81
		2.4	18/03/2003	7.54	31/05/2003	12.99	5	0.85
	Yr-3	3.1	26/10/2003	4.28	19/05/2004	16.58	5	0.48
		3.2	22/01/2004	13.9	-	-	1	-
		3.3	19/04/2004	6.2	-	-	0	-
Boyne River	Yr-2	2.1	19/12/2002	11.92	6/01/2003	13.44	2	0.59
		2.2	30/01/2003	10	27/02/2003	14.74	3	1.18
		2.3	15/03/2003	5.64	29/04/2003	10.13	4	0.70
		2.4	13/05/2003	7.52	28/05/2003	9.49	2	0.92
	Yr-3	3.1	23/10/2004	13.2	-	-	1	-
		3.2	22/11/2003	5.66	20/12/2003	11.13	2	1.37
		3.3	19/01/2004	8.07	17/02/2004	12.59	2	1.09
		3.4	18/03/2004	9.09	16/05/2004	13.83	3	0.57
		3.5	16/05/2004	6.1	-	-	1	-

Effects of flow on growth rates

Results of the step-forward general linear modelling indicated that the basic von Bertalanffy growth equation could be significantly improved by expanding *K* (the growth rate parameter) to be a function of other independent terms (**Table 9.3**). Of the pre-period flows (or rain), flow$_4$ (or log$_{10}$rain4) was used in the final models, as it provided a greater improvement in fit over the basic von Bertalanffy model than flow$_2$ or flow$_6$ (or log$_{10}$rain$_2$ or log$_{10}$rain$_6$ in the Boyne River estuary). Of the water quality parameters, temperature was a consistent and dominant effect, and when this term was included in the models, precluded salinity, pH and turbidity from significantly improving the models. Estimates of the parameters in the final models for each estuary are presented in **Table 9.4**

Table 9.3 Results from the step-forward general linear modelling of the effects of abiotic parameters on the growth rates of juvenile banana prawns sampled between January 2002 and May 2005 from three estuaries on the east coast of Queensland. (Final model in bold)

Estuary	Terms included in the general linear model (being quadratic, unless specified as linear=1)	Percent variance accounted for (R^2)	Improvement on base model (% of residual SS explained)
Fitzroy River	Base model - standard von Bertalanffy	92.2	-
	plus flow$_4$	93.8	20.5
	plus flow$_4$, plus flow$_0$	94.9	35.2
	plus flow$_4$, plus flow$_0$, plus temperature	95.8	46.1
	plus flow4 (l), flowo, temperature	**95.7**	**45.2**
Calliope River	Base model – standard von Bertalanffy	94.2	-
	plus temperature	97.4	55.7
	plus temperature, plus flow$_0$	98.2	69.7
	plus temperature, plus flow$_0$, plus flow$_4$	99.9	99.1
	plus flow4 (l), flowo (l), temperature	**99.0**	**83.7**
Boyne River	Base model – standard von Bertalanffy	86.2	-
	plus log$_{10}$rain$_0$ (l)	**94.3**	**58.0**

The most extensive model screening was conducted on data from the Fitzroy River estuary, as it had the highest number of observations. Even for the most complex models (**Table 9.3**), the residual degrees of freedom were adequate at 24. The three term quadratic model of $flow_4$, $flow_0$ and temperature was simplified to a linear term for $flow_4$, as the quadratic term for $flow_4$ did not significantly improve the model (P=0.27) and the residuals of the simplified model were still randomly distributed. For the Calliope River estuary, the three term quadratic model of temperature, $flow_0$ and $flow_4$ was simplified to a linear terms for $flow_0$ (P=0.04) and $flow_4$ (P=0.05), as realistically the data set for the Calliope River estuary was too small (and resulting residual degrees of freedom too few) to adequately test for the significance of quadratic terms for flow variables. For the Boyne River estuary, the three term quadratic model of $\log_{10}rain_0$, temperature and $\log_{10}rain_4$ was simplified to a linear of $\log_{10}rain_0$ because the additional terms of temperature and $\log_{10}rain_4$ did not significantly improve the model (P>0.05).

Table 9.4 Estimated parameters of the expanded form of the von Bertalanffy growth equation [A], where the growth rate (*K*) was expanded to become a function of the other independent terms, namely freshwater flow (or rainfall for the Boyne River estuary)

Parameter	Estimate (± standard error)		
	Fitzroy River Estuary	Calliope River Estuary	Boyne River Estuary
a	-3.44E-02 (±2.53E-02)	-1.65E-02 (±3.00E-02)	3.58E-03 (±1.15E-03)
b	-2.49E-02 (±1.07E-02)	-2.21E-02 (±1.65E-02)	-1.60E-03 (±2.28E-02)
C (temperature)	2.97E-03 (±1.92E-03)	1.91E-03 (±2.05E-03)	-
D ($temperature^2$)	-6.04E-05 (±3.66E-05)	-4.42E-05 (±3.59E-05)	-
E ($flow_0$)	9.32E-08 (±4.01E-08)	1.50E-07 (±0.65E-07)	-
F ($flow_0^2$)	-5.79E-13 (±3.93E-13)	-	-
G ($flow_4$	5.57E-08 (±1.78E-08)	-1.19E-06 (±0.57E-06)	-
H ($rain_0$)	-	-	2.29E-03 (±0.686E-03)

[A] The expanded form of the Bertalanffy equation used was:
$Length_2 = Length_1 + (L_{\cdot} - Length_1) * [1 – \exp (-\{ a + C * \text{independent term}_1 + D * \text{independent term}_1^{\ 2} + E * \text{independent term}_2 + F * \text{independent term}_2^{\ 2} + G * \text{independent term}_3 + H * \text{independent term}_4 \} * \text{Days} + b)]$

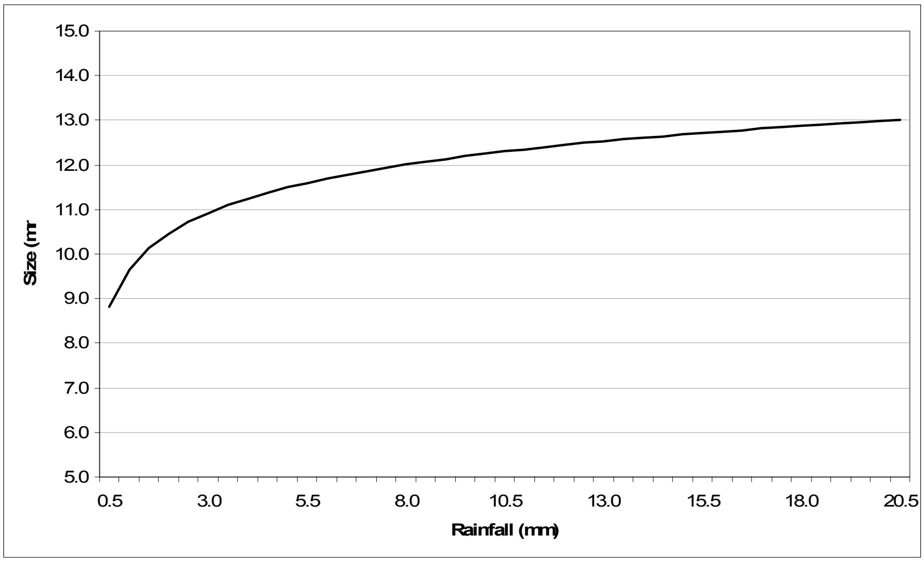
Simulating growth rates at various flow conditions

In order to better understand the implications of the fitted models, we estimated the growth and final length ($Length_2 = L_2$) of a juvenile banana prawn ($Length_1 = L_1$ = 5mm CL) after a set period of time (i.e. 42 days ~6 weeks) but varied the environmental parameters in the model between the 5th and 95th percentile of the observed range in these parameters (i.e. temperature, $flow_0$, $flow_4$ or $\log_{10}rain_0$). We consider this to be a hypothetical simulation as it would be unlikely that a particular set of environmental parameters (especially the extreme percentiles of all parameters) would apply continuously for 42 days. However, the simulation allowed theoretical insights to be gained into the implications of the fitted models.

Although the final model for each estuary was different, trends in growth rates were similar. In all three estuaries, growth rates of juvenile banana prawns, as indicated by greater L_2 after 42 days, increased with increasing freshwater flow (or rainfall). The trend of increasing growth rate with increasing flow was consistent at low, average and high temperature in Fitzroy and Calliope River estuaries, where temperature was also included in the final model (see below). Growth rates were slightly faster in the Boyne River estuary.

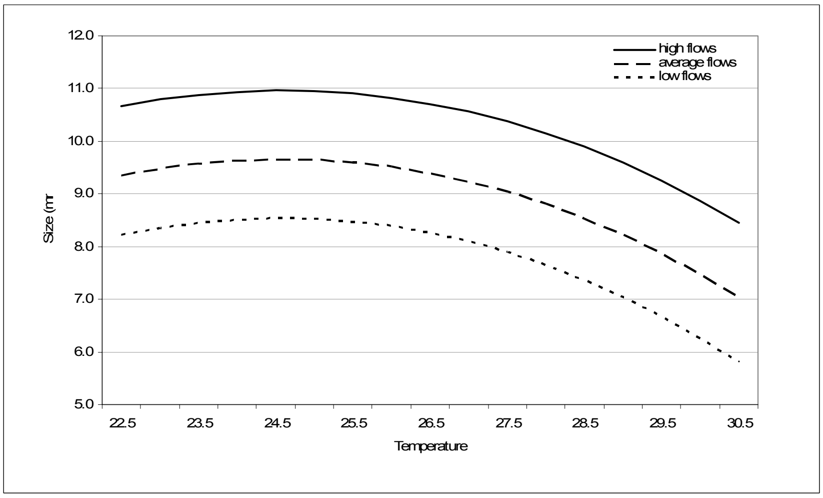
For the simplest model - the Boyne River estuary, which was a von Bertalanffy model expanded to include a linear effect of $\log_{10}rain_0$ – the L_2 of juvenile bananas prawns after 42 days increased with increasing rainfall (**Figure 9.10**). The estimated L_2 under low rainfall conditions (i.e. 5th percentile) was 8.8 mm CL, compared to an L_2 of 11.6 mm and 12.9 mm CL under average and high (i.e. 95th percentile) rainfall conditions.

Figure 9.10 Modelled growth of juvenile banana prawns (Length$_1$ = 5mm, days=42) for variable daily rainfall in the Boyne River estuary



For the more complex models developed for the Fitzroy and Calliope River estuaries (**Table 9.3**), models of prawn growth were influenced by temperature as well as flow. Our results suggested that there was an optimal temperature for growth (e.g. **Figure 9.11**) and that temperatures above or below this resulted in slower growth rates. This result is consistent with previous work (Staples and Heales 1991). However, for simplicity and because we were interested in elucidating the effects of freshwater flow, temperature was set at the observed average for each estuary. Flow$_4$ and flow$_0$ were varied between the 5th percentile (= low flow condition) and the 95th percentile (= high flow condition) to further explore the effects of flow on growth rates of juvenile banana prawns in the study estuaries.

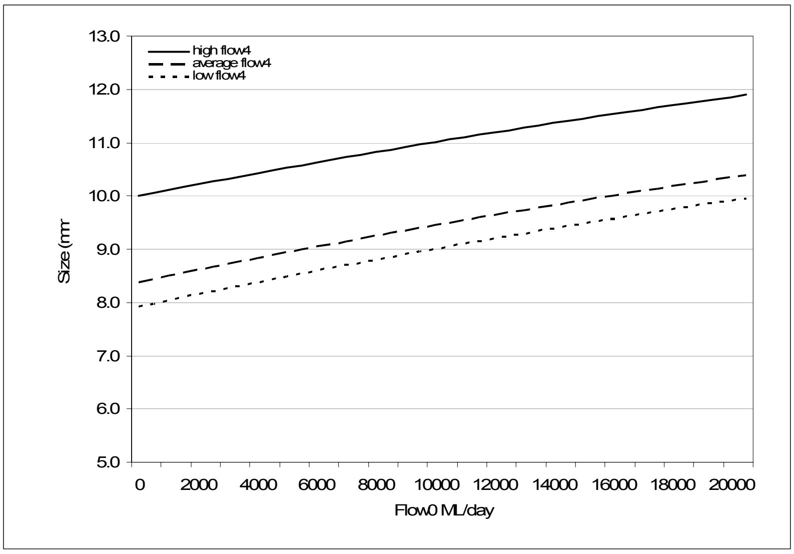
Figure 9.11 Modelled growth of juvenile banana prawns (Length$_1$ = 5mm, days=42) for variable temperature (°C) at low, average and high flows[A] in the Fitzroy River estuary



[A] low flows = 5th percentile of flow$_0$ and flow$_4$, average = average of flow$_0$ and flow$_4$, high flows = 95th percentile of flow$_0$ and flow$_4$.

There were major differences in L$_2$ of juvenile bananas prawns between low and high flow conditions. For example, in the Fitzroy River estuary (Figure 9.12), the estimated L$_2$ of banana prawns was greatest when both flow$_0$ and flow$_4$ were high (L$_2$=11.9 mm CL) and smallest when flow$_0$ and flow$_4$ were low (L$_2$=7.9 mm CL). However, a similar L$_2$ (i.e. 10 mm CL) could be achieved under low flow$_0$ but high flow$_4$ conditions as that under high flow$_0$ and low flow$_4$ conditions (Figure 9.12).

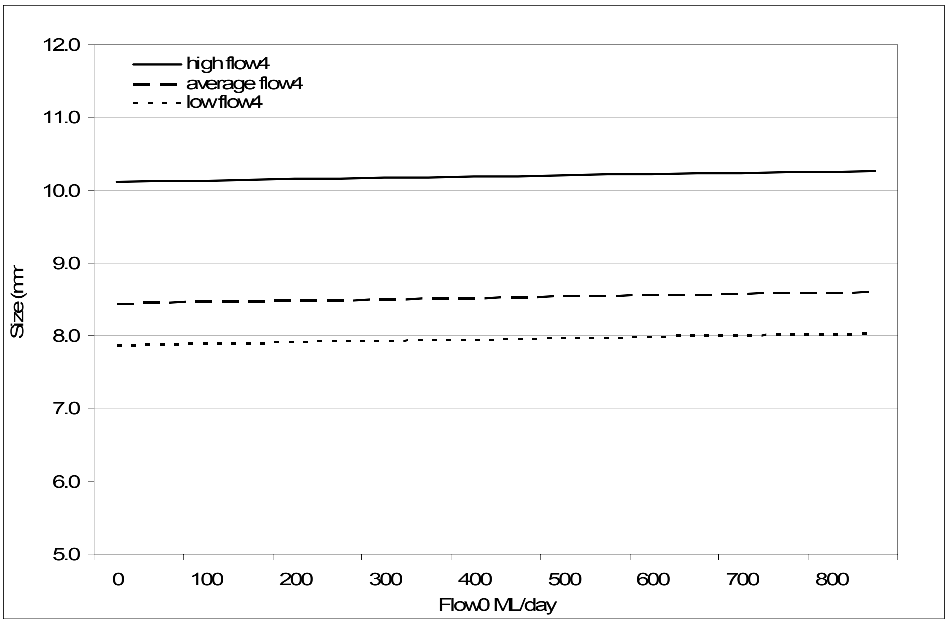
Figure 9.12 Modelled growth of juvenile banana prawns ($Length_1$=5mm, days=42; temperature=27.5°C=average) for variable $Flow_0$ at low, average and high $Flow_4^A$ in the Fitzroy River estuary



^{A}low flows = 5th percentile of $flow_4$; high flows = 95th percentile of $flow_4$.

In the Calliope River estuary, the estimated L_2 was greatest under high flow conditions. However, in this estuary, the estimated L_2 of juvenile banana prawns varied little with $flow_0$ (Figure 9.13), suggesting that the pre-period flows (i.e. $flow_4$) had a greater influence on the estimated L_2 than the flow at the time of capture (i.e. $flow_0$).

Figure 9.13 Modelled growth of juvenile banana prawns ($Length_1$=5mm, days=42; temperature=29.1°C=average) for variable $Flow_0$ at low, average and high $Flow_4^A$ in the Calliope River estuary



^{A} low flows=5th percentile of $flow_4$; high flows=95th percentile of $flow_4$.

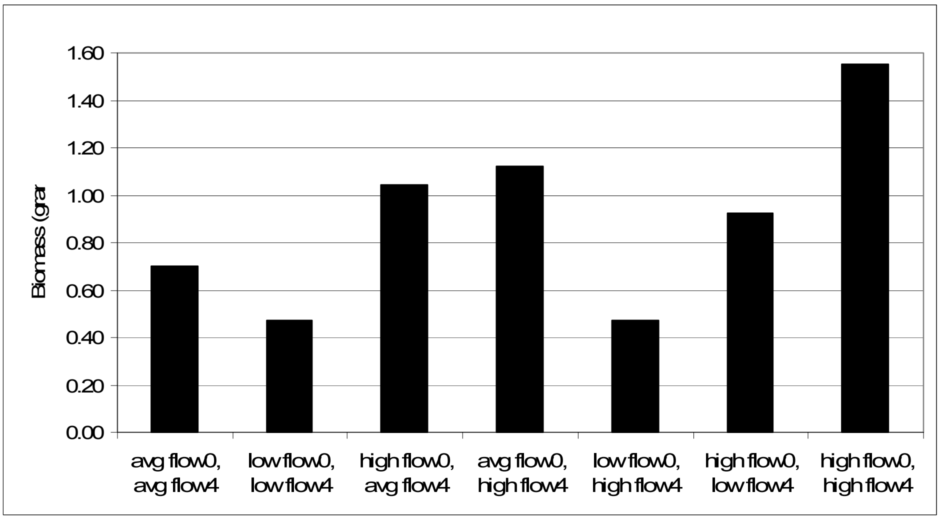
We extended the simulation by converting the L_2 achieved under different flow condition to a weight, based on the following relationship observed for juvenile banana prawns in the study estuaries: weight = 0.0011*carapace $length^{2.9287}$ (n=15379; R^2 = 0.97). Weight (=biomass) is frequently considered as a measure of productivity and different flow conditions have the potential to have major impacts on the weight of individuals and thus the biomass of the estuarine population of banana prawns.

In the Boyne River estuary (with the simplest model), the weight a juvenile prawn was estimated to be 0.65, 1.44 or 1.98 grams for low, average and high rainfall conditions, respectively. In the Fitzroy River estuary, the weight a juvenile prawn was estimated to be 0.47, 0.70 or 1.55 grams for low, average and high flow conditions respectively (**Figure 9.14**). Whilst in the Calliope River estuary, the weight would be 0.46, 0.57 and 1.01 grams respectively. Our results imply that freshwater flows can double or triple the weight of individual juvenile banana prawns. When considered at a population scale (i.e. millions of individuals in an estuary per season), this would have major effect on the biomass of th juvenile banana prawn population pre-recruitment to commercial fisheries.

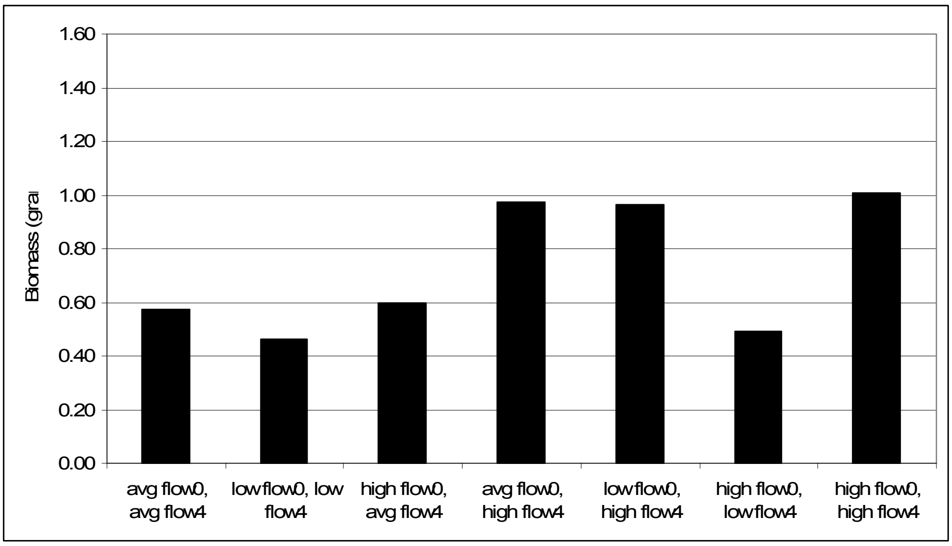
It should be noted that the data analysed in the present study was collected between 2002 and 2005, when freshwater flow did not exceed the average annual freshwater flows in the Fitzroy River (i.e. conditions were drier than average). As such, it is uncertain whether we could extend the results to include the implications of extremely large flow events (such as a 1-in-100 year flood).

Figure 9.14 Modelled biomass (in grams) of juvenile banana prawns (Length$_1$=5mm, days=42; temperature=average) under various flow conditions^A in two of the study estuaries

(a) Fitzroy River estuary (average temperature = 27.5°C)



(b) Calliope River estuary (average temperature = 29.1°C)



^{A}low flow = 5th percentile of observed flow$_0$ or flow$_4$, avg. = average of observed flow$_0$ or flow$_4$, high flow = 95th percentile of observed flow$_0$ or flow$_4$; and where flow$_0$ = total freshwater inflow to the estuary for the period between dates when each cohort was sampled, and flow$_4$ = total freshwater inflow to the estuary for four-weeks prior to the first date a cohort was sampled.

Annual index of abundance

Fitzroy River estuary

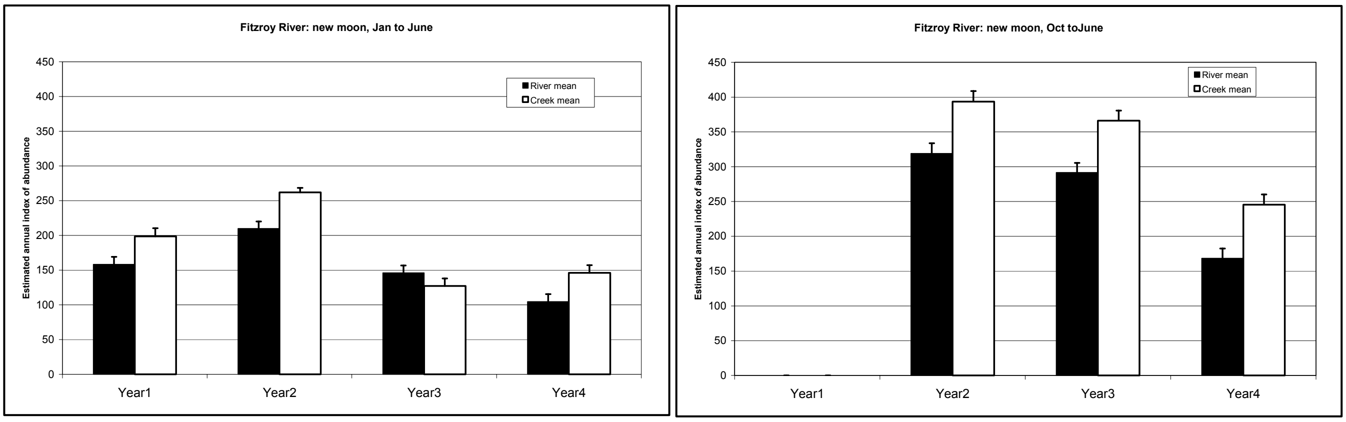
Trip, region and habitat-type were significant factors accounting for variation in the abundance of juvenile banana prawns within the estuaries (**Table 9.5**). These factors were used in the final model to estimate the annual index of abundance, which accounted for ~40% of the variation in juvenile banana prawn abundance per site.

Table 9.5 Output (accumulated analysis of variance) from the GLM analysis of juvenile banana prawns abundance in the Fitzroy River estuary

Change	d.f.	s.s.	m.s.	v.r.	F pr.
+ Trip	27	186.6644	6.9135	10.42	<0.001
+ Region	3	10.3849	3.4616	5.22	0.001
+ Habitat	1	11.4097	11.4097	17.19	<0.001
+ Trip.Region	81	128.0824	1.5813	2.38	<0.001
Residual	490	325.1624	0.6636		
Total	602	661.7039	1.0992		

Annual indices of abundance were calculated separately for creek and river habitats, because of the significant variability in the abundance of juvenile banana prawns within these two habitat types. However, similar inter-annual trends in the annual index of abundance were observed in both creeks and river habitats. When comparing the estimates based on January to June samples over four years, year-2 (i.e. 2003) was estimated to have the highest mean annual abundance of juvenile banana prawns in the Fitzroy River estuary (**Figure 9.15**a). The same trend was observed when using estimates based on a longer sampling year (i.e. October to June), but the difference between year-2 estimates and year-3 estimates was not significant (**Figure 9.15**b).

Figure 9.15 Estimated mean (+ variance) annual index of abundance of juvenile banana prawns, based on the area under the curve from the weighted predicted trip means for creek and river habitats in the Fitzroy River estuary, for: (a) between January and June (a 4-year comparison); and (b) October to June (a 3-year comparison)



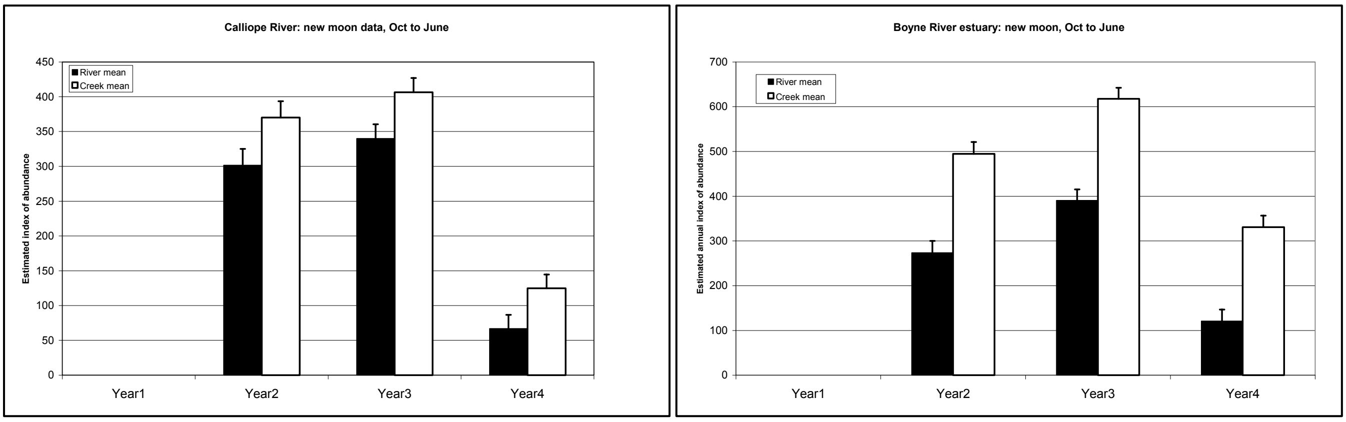
(a) 4-year comparison

(b) 3-year comparison

Calliope and Boyne River estuaries

In the Calliope and Boyne River estuaries, trip and habitat-type were significant factors accounting for variation in the abundance of juvenile banana prawns, accounting for 56.1% and 56.6% of the variation in banana prawn abundance per site in each estuary respectively. A regional term was not applicable to the Calliope or Boyne River estuaries because these estuaries were sufficiently small that all of the estuary could be sampled on one day. In the Calliope River estuary, juvenile banana prawns were significantly more abundant in year-2 and year-3 of the study compared with year-4. Similarly, in the Boyne River estuary, year-4 abundance estimates were significantly lower than those of year-2 and year-3 (**Figure 9.16**).

Figure 9.16 Estimated mean (+ variance) annual index of abundance of juvenile banana prawns, based on the area under the curve from the weighted predicted trip means for creek and river habitats in: (a) the Calliope River estuary; and (b) the Boyne River estuary


(a) Calliope River estuary (b) Boyne River estuary

Trends in the inter-annual abundance of juvenile banana prawns were mirrored in the commercial catches of banana prawns reported in the compulsory logbook (CFISH) of the Department of Primary Industries & Fisheries for the same period (**Table 9.6**).

Table 9.6 Commercial catch of prawns (tonnes) in the Fitzroy and Port Curtis regions for January to June for 2002 to 2005

	Fitzroy[A]		Port Curtis[B]	
	Banana prawns	Total Prawns	Banana prawns	Total Prawns
Year-1 (2002)	57.47	69.69	71.98	78.71
Year-2 (2003)	122.43	144.27	103.90	110.75
Year-3 (2004)	111.91	126.84	79.02	81.16
Year-4 (2005)	48.68	61.81	25.02	27.26

[A] comprised of catch within CFISH grids R28, R29, R30, S28 and S29; [B] comprised of catch within CFISH grids S30 and T30

Best all-subsets general linear models (GLMs), which identify a number of 'best' models, was used to thoroughly explore potential relationships between juvenile banana prawn abundance and physical parameters (i.e. salinity, secchi depth, water temperature and pH) measured at the time of each trawl sample (in a manner similar to that of Meager *et al.* 2003).

Across all years, but separate for each estuary, best all sub-sets GLM identified several alternate models that explained ~26%, ~62% and ~59% of the variation in catch rates of juvenile banana prawns in the Fitzroy River, Calliope River and Boyne River estuaries respectively (**Table 9.7**). Trip was a significant factor in almost all models, with the physical parameters included in the models varying between estuaries and between all years and individual years. Variation in R^2 values between years was also reported for relationships between catches of juvenile banana prawns and physical parameters in the Logan River estuary (Meager *et al.* 2003).

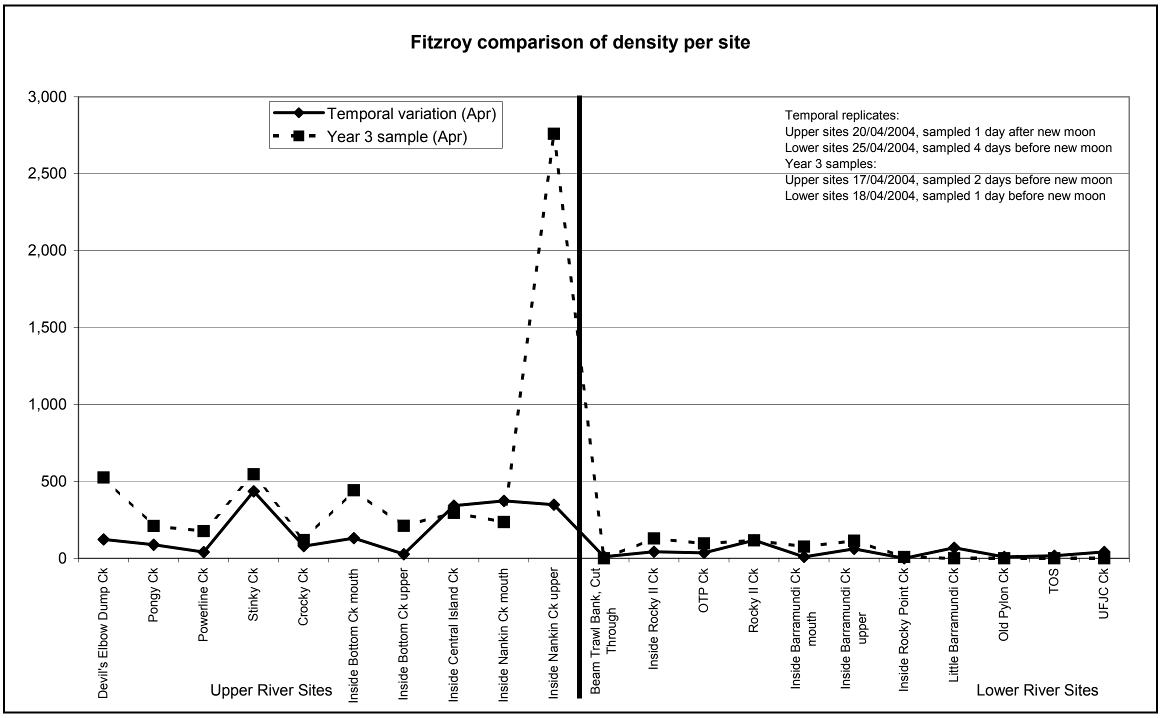
Table 9.7 Results from best all sub-sets GLM of juvenile banana prawn abundance and physical parameters in three estuaries of central Queensland

	Years included in analysis	Model	Overall R^2 (%)	df	v.r.	F pr.
Fitzroy River estuary	1,2,3,4	Trip + Habitat + Salinity + pH	26.1	53, 884	7.27	<0.001
	1,2,3,4	Trip + pH + Salinity	25.5	52, 885	7.18	<0.001
	1,2,3,4	Trip + pH + Habitat	26.1	52, 885	7.35	<0.001
	1,2,3,4	Trip + Habitat + Salinity	25.7	52, 885	7.24	<0.001
	1	Trip + Habitat + Secchi depth	36.4	11, 169	10.36	<0.001
	1	Habitat + Secchi depth + Temp.	35.2	3, 177	33.62	<0.001
	2	Trip + pH	33.7	17, 309	10.86	<0.001
	2	Trip + pH + Salinity	34.6	18, 308	10.55	<0.001
	2	Trip + pH + Secchi depth	34.2	18, 307	10.38	<0.001
	3	Trip + Habitat + Secchi depth	17.7	13, 205	4.60	<0.001
	3	Trip + Habitat	14.8	12, 206	4.16	<0.001
	4	Trip	11.9	8, 182	4.08	<0.001
Calliope River estuary	2,3,4	Trip + Habitat + Temp.	62.6	37, 241	13.59	<0.001
	2,3,4	Trip + Habitat + Salinity	62.5	37, 241	13.50	<0.001
	2,3,4	Trip + Temp. + Secchi depth	62.5	37, 241	13.49	<0.001
	2	Trip + Temp. + pH	53.40	16, 95	8.95	<0.001
	2	Trip + Temp. + Secchi depth	51.70	16, 95	8.42	<0.001
	3	Trip + Habitat + Salinity	51.5	11, 76	8.52	<0.001
	4	Trip + Salinity	16.5	11, 76	2.56	0.008
Boyne River estuary	2,3,4	Trip + Habitat + Secchi depth +pH	59.2	37,231	11.51	<0.001
	2,3,4	Trip + Secchi depth +pH	58.6	36, 232	11.54	<0.001
	2,3,4	Trip + Habitat + Secchi depth	58.3	36, 232	11.39	<0.001
	2,3,4	Trip + Habitat + pH	57.4	36, 232	11.05	<0.001
	2	Trip + pH	57.7	16, 104	11.23	<0.001
	2	Trip + Secchi depth	55.4	16, 104	10.31	<0.001
	2	Trip + Habitat	55.3	16, 104	10.27	<0.001
	3	Trip + Secchi depth + pH	50.4	11, 67	8.20	<0.001
	3	Trip + Habitat + pH	48.9	11, 67	7.79	<0.001
	4	Trip + Salinity + Secchi depth	53.6	10 , 58	8.87	<0.001
	4	Trip + Secchi depth + pH	51.9	10, 58	8.34	<0.001

Temporal variation

Densities of prawns in the replicate samples are likely to be influenced by the timing of the sampling in relation to the days before or after the new moon as this relates to the degree of tidal draining of mangrove habitats. Preliminary summaries of juvenile banana prawn densities showed that there was short-term temporal variation. In the Fitzroy River estuary, temporal replicates were collected on five occasions (**Table 9.1**). Timing of the replicate samples was randomly allocated to before or after the standard sampling times. Of note is the large variation in banana prawn densities at one particular site in the upper Fitzroy River (i.e. Inside Nankin Creek upper), which occurred in temporal replicates (e.g. see **Figure 9.17**).

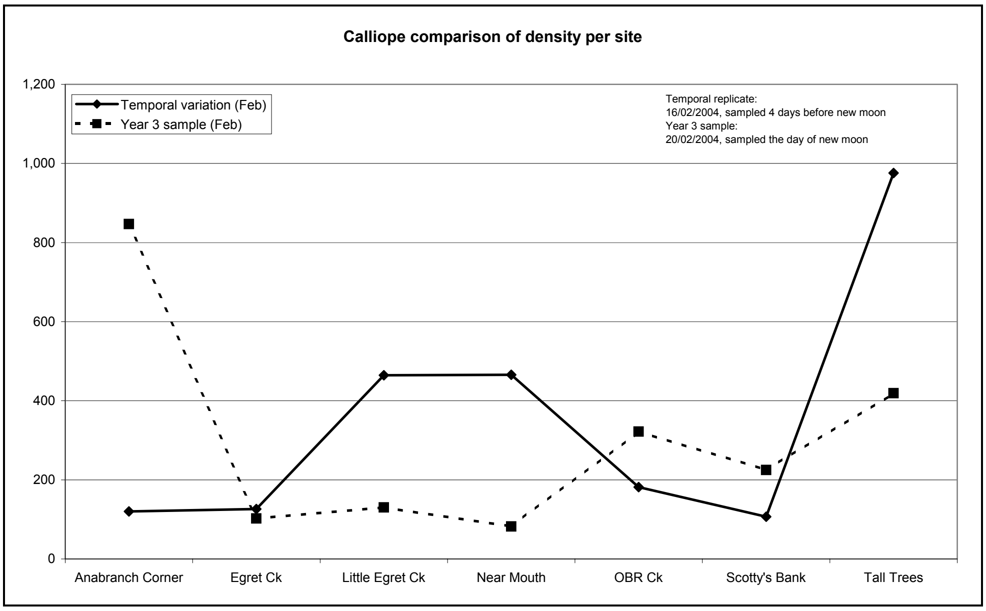
Figure 9.17 Banana prawn density per site: temporal replicate compared to standard sample for January 2003 in the Fitzroy River estuary



For the January 2003 sample, one-way ANOVA showed significant differences ($P<0.05$) in prawn densities in the temporal replicate (higher) compared to the standard sample (lower), when salinity, turbidity and both salinity and turbidity were included as covariates, but not pH or temperature. No differences were significant in the other temporal replicate samples.

In the Calliope River, temporal replicates were collected on six occasions (Table 9.1). Densities of banana prawns varied in the temporal replicates from that of the standard samples (e.g. see Figure 9.18) but the variation was much less pronounced than that observed in the Fitzroy River estuary. In the Boyne River estuary, temporal replicates were collected on four occasions (Table 9.1), with prawn densities being relatively consistent between temporal replicates and standard samples. Prior to the standard sampling in January 2003 (i.e. 21st) heavy coastal rain occurred with slightly lowered salinities (i.e. ~30). From the 22 to 28 January 2003, an environmental flow of 3124 ML/day was released from Awoonga Dam, although this appeared to have little effect on the salinities in the Boyne River estuary as they had returned to 32 to 36 by the temporal sampling on the 24 January 2003.

Figure 9.18 Banana prawn density per site: temporal replicate compared to standard sample for February 2003 in the Calliope River estuary



Discussion

Length-frequency analyses

The estimated average growth rates of juvenile banana prawns in central Queensland estuaries (Table 9.2) are consistent with estimates for this species from the Embley (Haywood and Staples 1993) and Norman Rivers (Staples 1980) in the Gulf of Carpentaria, where reported minimum and maximum growth rates were 0.63 to 1.65 mm CL wk^{-1} respectively in the Embley River and averaged 1.2 mm CL wk^{-1} in the Norman River. Haywood and Staples (1993) found that growth rates of juvenile banana prawns in the Embley River were significantly and positively correlated with mean water temperatures, but not with mean salinity.

The present study differs from the Embley River study in that: (i) there was a greater temperature range (i.e. 19.9°C to 35.15°C compared to 24.3°C and 31.4°C); and (ii) the salinity at the time of sampling was used in the present analysis compared to salinity averaged over the estuarine life of each cohort (Haywood and Staples, 1993). The temperature and salinity range experienced by juvenile banana prawns sampled by the present study is more consistent with the temperature and salinity regime used in laboratory experiments on growth by Staples and Heales (1991). Previous field studies report that salinity has little or no detectable effect on growth rates of juvenile banana prawns (Staples 1980; Haywood and Staples 1993). However, experimentally, juvenile banana prawns were found to have optimal food consumption and production at a salinity of 20‰ (Vinod *et al.* 1996), including the best food conversion ratio. At higher salinities, there was a considerable decrease in growth and food consumption (Vinod *et al.* 1996). In contrast, Saldanha and Achuthankutty (2000) report that growth of juvenile banana prawns increases with salinity (up to 40‰), while Staples and Heales (1991) report that the optimum temperature and salinity for the growth in length of juvenile banana prawns (i.e. shortest inter-molt period and largest carapace length increment) is 31°C and a salinity of 30‰ (resulting in a weekly growth rate of ~1 mm per week). However, after Staples and Heales (1991) accounted for survival, increases in wet and dry weight, the optimum temperature and salinity for the greatest increase in biomass and production were 28°C and a salinity of 25‰. Staples and Heales (1991) concluded that deviations from the optimum temperature have a greater effect on productivity than of salinity. Based on their experimental work on variation in growth and survival with temperature and salinity, Staples and Heales (1991) predicted that in an estuary, post-larval prawns would grow quickly but suffer high mortality when temperature and salinity were high, but would grow slowly and remain in nursery areas if the salinity of the estuary fell below 20‰.

Effects of freshwater flows on the growth of banana prawns

The effect of freshwater flows (or rainfall) on the growth of banana prawns was reasonably consistent over the three estuaries sampled, having a positive influence on growth rates in all three estuaries. Where temperature was a significant factor in the final model, the quadratic fit of temperature supports previous findings of an optimum value for growth (Staples and Heales 1991; Haywood and Staples 1993).

The effects of flow on growth rates of juvenile banana prawns have not been included in any other published analysis. The inclusion of freshwater flow as an independent variable has been shown to significantly affect growth rates of the finfish barramundi (*Lates calcarifer*) in the Fitzroy River estuary (see Chapter 7; Robins *et al.* 2006). This is not surprising as there are significant changes in the abiotic variables of turbidity and salinity within the estuary when flows occur (see Chapter 10), and consequently the water in which fisheries production occurs.

Increased growth of estuarine biota in response to freshwater flow suggests that there are processes occurring within the estuary on a shorter than seasonal scale that significantly effect fisheries productivity. We speculate that the increased availability of nutrients to primary producers provided by freshwater flows instigates a trophic bloom in the estuary that cascades up the food chain, which results in the increased growth rates of higher order consumers (e.g. juvenile banana prawns and then barramundi). Alternatively, or in addition, increases in turbidity resulting from freshwater flows decrease the ability of visual predators to search for prey (Hecht and van der Lingen 1992). Reduced predation pressure may allow juvenile banana prawns more time to feed and translate to increased growth rates (Currie and Small 2005).

Moriarty and Barclay (1981) proposed that food might be one of the limiting factors in the growth rates of juvenile banana prawns in the Gulf of Carpentaria, based on inter-year differences in mean concentrations of nitrogen and organic carbon in the gut and intestine contents juvenile banana prawns. They highlighted that many insect populations are regulated by the supply of nitrogen to juveniles and suggested that a similar effect might occur for prawns which select food with high nitrogen content. Loneragan and Bunn (1999) speculated that catchment nutrients from high summer flows might stimulate primary productivity leading to increased production of prawns via increased survivorship and /or growth of juvenile stages. Further research is required to quantify mechanisms linking growth rates of juvenile banana prawns to freshwater flow conditions.

Recruitment of juvenile banana prawns

The recruitment of juvenile banana prawns varied between years, with the greatest abundance occurring in year-2 in the Fitzroy River estuary, and years -2 and -3 in Port Curtis estuaries. In general, trends in the annual index of abundance were reflected in the total annual catch of prawns of the commercial trawl fishery.

Implications for management

The results of the current chapter provide quantitative evidence that freshwater flows affect the growth rates of juvenile banana prawns. Although based on simulation, the present results suggest that freshwater flows (or rainfall) have significant implications on the growth rates and subsequent weight of juvenile bananas prawns in an estuary i.e. the biomass pre-recruitment to commercial fisheries. Therefore, if freshwater flows to an estuary are reduced as a consequence of water infrastructure and abstraction or long-term climate change, then there is likely to be a negative impact on the growth rates of juvenile banana prawns in the downstream estuary and a probably reduction in banana prawn biomass and the subsequent commercial fisheries catch.

The results of the current chapter also provide strong evidence that environmental factors probably play an important role in determining the size of the estuarine population of juvenile banana prawns (i.e. biomass pre-recruitment to commercial fishery) and probably should be considered in the modelling and stock assessment of banana prawn populations.

In addition, the greater relative abundance of juvenile banana prawns in years of higher freshwater flow provides further evidence that freshwater flows to estuaries result in greater abundance as well as greater biomass of banana prawns. Our ability to demonstrate effects on both growth and abundance of juvenile banana prawns in the estuaries of central Queensland provides strong evidence that improved growth rates potentially leads to greater survival and thus higher abundance of juvenile prawns in estuaries, as suggested by Browder *et a*l. (2002).

Further analysis of the extensive data set collected within the current project and other Coastal Zone CRC projects (i.e. Webster *et al.* 2003; Margvelashvili *et al.* 2003) may provide further insights and evidence into the causal mechanisms linking enhanced banana prawn production with freshwater flows. Better understanding of the mechanism(s) linking freshwater flow to increased growth and biomass of estuarine populations of juvenile banana prawns is required before robust estimates can be made of the effects of: (i) specific water management scenarios, and (ii) climatic variability on biomass pre-recruitment to the fishery and subsequent implications for stock assessment. However, results of this chapter clearly show that freshwater flowing to estuaries is not wasted, but rather supports estuarine fisheries production, in this case banana prawns.

Chapter 10. Effects of flow on demersal community structure

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Summary

Demersal communities were sampled using a beam-trawl for a period of three years in three central Queensland estuaries. Over 160 species from 66 families were collected from the three estuaries with over 80 species occurring in all three estuaries. Significant differences were found in the community structures between all rivers, between some years within rivers and between months within years within rivers. Community structure was dominated by similar species in all estuaries, with *Acetes* sp., banana prawns, metapenaeid prawns and planktonic fishes being the most common. All other species were found to be less than 0.5% of the total number of animals caught.

Three different phases were identified in the community structure within each estuary (i.e. 'before', 'during' and 'after' a freshwater flow). The onset of freshwater flows significantly changed the community structure by initially decreasing the abundance of all species that were in the estuary prior to the flow and introducing numerous species of freshwater fish into the estuary. These changes in community structure indicate that for estuaries to be considered 'healthy' instability in the community structure is desirable attribute. The ability of estuaries to support a wide range of faunas is enhanced by the changes brought about by of freshwater flows. These include physical changes to salinity, turbidity, pH, the delivery considerable quantities of nutrients, extension of aquatic habitats and connection between aquatic habitats. The diversity of estuarine faunas enables some species to exploit this combination of effects making them extremely dynamic in terms of the community structure at any given point in time. No single physical parameter could be used as a predictor of the abundance and species composition of any of the estuaries sampled.

The instability of estuarine communities in response to freshwater flows indicates that a healthy estuary is one that is inherently dynamic with different species exploiting the estuarine environment at different times.

Introduction

The aim of this chapter was to assess the changes in demersal communities of estuaries in central Queensland in response to freshwater flows.

Methods

The collection of demersal community data occurred during beam-trawl sampling for banana prawns (see Chapter 9). Demersal samples were collected from January 2003 until May 2005 for the Fitzroy, Calliope and Boyne River estuaries. All samples were identified to species level, weighed and measured. Abiotic data (secchi depth = turbidity, pH, salinity and water temperature) were also collected.

The initial sampling method (i.e. beam-trawling) was designed to collect juvenile banana prawns and net selectivity will have influenced our ability to sample other types of animals, particularly free swimming juvenile fish.

Statistical analyses

Univariate analysis

All univariate analyses were conducted using GenStat (2005). Step-forward generalized linear models (GLM, McCullagh and Nelder 1989) were used. Accumulated analysis of variance (or deviance) tables were developed, utilising Type-I sums of squares and testing. Preliminary analyses identified the important effects of the physical parameters (i.e. temperature, salinity, turbidity, and pH) on catch rates. However, these tended to interact strongly with each other, and were not independent. Also, they were correlated with flow regimes. As such, the 'flows' factor (as defined below) was taken as a more direct measure of this effect on catch rates.

The model was fitted throughout, with no model refinement (i.e. no deletion of non-significant terms), as analyses had ample degrees of freedom. The 'fixed model' then provided a consistent basis from which all means could be directly compared. The factors and variates included in the analyses of catch rates were:

- River – Boyne, Calliope, Fitzroy (upper), and Fitzroy (lower), as these constituted different environments.
- Location – a binary variable, defining whether the sample was in the main river channel or in a side-creek. This term was included in all analyses along with its interaction with 'River', as it may well have a varying effect, depending both on species behaviour and which river was sampled.
- Month – to allow estimation of the seasonal pattern for each species. It was preferred to temperature, as these terms were highly correlated.
- Flow – each trip was defined as falling 'before', 'during', or 'after' the major seasonal flow event for that year.
- Year – 'year -1' (January to May 2002 samples), to 'year-4' (October 2004 to May 2005), see Table 9.1. The 3-way interaction, of 'Year' with 'River' and 'Flow' was included, despite not all combinations being sampled, to allow for possible differential responses amongst these factors.
- Trip type – the majority of the sampling trips were 'new moon', but full moon samples were also included. This binary factor allows for this contrast, should it exist.
- Flow meter – this was a measure of the amount water passing through the net, being a measure of effort. It was expected to have a positive effect on abundances and was included in all models, even if not a significant term for consistency between models.

Only main effects were analysed and interpreted for, as we discounted all other interactions due to the complexity of interpretation. In addition, interaction effects between variables, whilst likely to be significant, tended to be of an order-of-magnitude below their respective main effects. Overall year by flow by river means was the only three-way interaction term included in the models. For the analyses of the physical parameters, which were measured prior to each trawl, the flow meter term was not included. A three-way interaction table of means was produced to determine how freshwater flows affected catch rates across years and river systems.

Multivariate data analysis

Multivariate statistics using Primer 5.2.2 were used to identify patterns between rivers, years and flow regimes based on the demersal community caught in the beam-trawl. The $\log_{10}$ abundance and $\log_{10}$ biomass data (standardised to N per 1000 m trawled) was used in the analysis. A similarity matrix based on Bray-Curtis indices representing species abundances and biomasses was calculated and used in cluster analyses, with group average linking, to separate spatial groups. Significance of these spatial groups was tested using a one way analysis of similarities (ANOSIM). SIMPER analysis was carried out to identify the species determining the spatial groups. For each species, this analysis calculates the ratio of average contribution to similarity between groups to the standard deviation of similarity between groups. The higher the value of the discrimination index the more useful the species is for discriminating between groups.

Multi-dimensional scaling (MDS) was used to ordinate and view spatial relationships identified in the cluster analysis. MDS is a procedure that iteratively rearranges points from the similarity matrix to generate graphs that can be visually inspected for gradients along the axis and clusters within the data set. The MDS procedure uses established techniques of numerical optimisation to compress coordinates from higher to lower multidimensional space while maintaining the relationships in the original higher dimensional matrix to the maximum degree possible (Clarke and Warwick 1994). A matrix representing 160 species and 20 trips would generate a plot in 160 dimensions, and thus be impossible to visually inspect and evaluate for patterns. The MDS procedure produces a 2 – or 3 – dimensional graph based on the original 160 dimensional space and then calculates a measure of fit (referred to as a 'stress value') between the original and reduced configurations of points. The lower the stress value the better the fit. If the stress value was less than 0.20, the plots were inspected for the possible influences of the two main effects (i.e. estuary and flow).

Results

Univariate analyses

Physical parameters

For each parameter, residual diagnostics and plots indicated the normal distribution as appropriate. **Table 10.1** shows an example of an analysis of variance table. Due to the high level of replication, almost all terms were significant ($P<0.05$), but it was the three-way interaction that was of most interest. It was highly significant for salinity ($P<0.01$; see **Table 10.1**), turbidity (F-value of 3.1) and pH (F-value of 6.3).

Table 10.1 Analysis of variance table for salinity

Term	d.f.	s.s.	m.s.	F-value	F-prob.
River	3	97355	32452	1287.6	<.001
Location	1	5235	5235	207.7	<.001
Month	8	48828	6104	242.2	<.001
Flow	2	45423	22711	901.1	<.001
Year	3	3655	1218	48.3	<.001
Trip type	1	19	19	0.8	0.38
River. Location	3	1034	345	13.7	<.001
Flow.Year	5	3214	643	25.5	<.001
River.Flow	6	19765	3294	130.7	<.001
River.Year	8	815	102	4.0	<.001
River.Flow.Year	13	1476	114	4.5	<.001
Residual	1566	39469	25		

The adjusted means for the physical parameters (adjusted for all other contrasts and terms in the model) were calculated and listed in **Table 10.2**, **Table 10.3** and **Table 10.4**. The small flows that were encountered during the sampling period were reflected in the salinity measures with only small decreases in the mean value of salinity 'during' the flow periods in the Calliope and Boyne River estuaries with larger decreases in the Fitzroy River estuary.

Table 10.2 Adjusted mean salinity (‰), by estuary, flow state and year. (n.s. = not sampled)

Estuary	Flow	Year-1	Year-2	Year-3	Year-4
Boyne River	Before	n.s.	39.3	39.4	34.8
	During	n.s.	29.8	33.0	34.7
	After	n.s.	34.9	34.1	37.0
Calliope River	Before	n.s.	38.1	36.7	34.8
	During	30.2	25.4	23.7	32.1
	After	n.s.	33.3	32.5	35.9
Fitzroy River (upper)	Before	n.s.	33.3	34.2	32.1
	During	5.8	3.5	4.0	7.6
	After	17.1	16.9	10.3	12.2
Fitzroy River (lower)	Before	n.s.	36.7	37.8	34.5
	During	16.1	10.7	15.5	24.3
	After	29.0	27.9	23.0	25.8

Table 10.3 Adjusted turbidity (secchi depth) means, by estuary, flow state and year. (n.s. = not sampled)

Estuary	Flow	Year-1	Year-2	Year-3	Year-4
Boyne River	Before	n.s.	0.413	0.328	0.367
	During	n.s.	0.337	0.338	0.323
	After	n.s.	0.447	0.322	0.424
Calliope River	Before	n.s.	0.497	0.450	0.391
	During	0.488	0.458	0.351	0.555
	After	n.s.	0.656	0.505	0.523
Fitzroy River (upper)	Before	n.s.	0.349	0.302	0.340
	During	0.195	0.131	0.160	0.181
	After	0.430	0.316	0.242	0.255
Fitzroy River (lower)	Before	n.s.	0.184	0.125	0.124
	During	0.203	0.066	0.118	0.072
	After	0.237	0.102	0.118	0.122

Table 10.4 Adjusted pH means, by estuary, flow state and year. (n.s. = not sampled)

Estuary	Flow	Year-1	Year-2	Year-3	Year-4
Boyne River	Before	n.s.	7.70	7.65	8.13
	During	n.s.	7.80	7.64	7.69
	After	n.s.	7.56	7.54	7.53
Calliope River	Before	n.s.	8.02	7.83	8.38
	During	n.s.	8.02	7.67	7.87
	After	n.s.	7.89	7.95	7.70
Fitzroy River (upper)	Before	n.s.	7.88	7.83	8.39
	During	7.96	7.97	7.82	7.90
	After	8.21	7.74	7.93	7.58
Fitzroy River (lower)	Before	n.s.	7.97	7.84	8.46
	During	8.03	7.80	7.83	8.09
	After	8.11	7.83	7.81	7.72

Catch rates

The inflated zero-class of these data invalidates the assumptions of many of the standard analytical models. Through our thorough investigative study (Mayer *et al.* 2005), we identified that conditional distributions, in particular the binomial and lognormal combination, were the most appropriate method of analysis. A selection of individual species of commercial and recreational importance was used to determine the effects of flow on their abundances.

Table 10.5 Correlations against annual flows (or log) 'during' flow events

	Boyne River estuary		Calliope River estuary		Fitzroy River estuary (upper)		Fitzroy River estuary (lower)	
	flow	log	flow	log	flow	log	flow	log
Acetes sp.	-0.056	0.295	-0.434	-0.094	-0.966	-0.962	-0.916	-0.909
Metapenaeus spp.	0.562	0.814	-0.392	-0.048	-0.585	-0.497	-0.130	-0.025
Penaeus merguiensis	-0.129	0.223	-0.224	0.129	-0.603	-0.616	-0.339	-0.294
Parapen. sculptilis	0.128	0.464	-0.389	-0.045	-0.761	-0.819	0.278	0.170
Thryssa hamiltoni	0.809	0.963	-0.849	-0.613	-0.722	-0.734	**-0.992***	**-0.994***
Leiognathus decorus	-0.143	-0.477	-0.881	-0.662	-0.841	-0.850	**-0.992***	**-0.994***
Pomadasys kaakan	-0.844	-0.978	-0.873	-0.649	-0.955	-0.950	-0.974	-0.970
Sillago spp.	-0.560	-0.837	-0.970	**-0.994***	-0.921	-0.928	-0.327	-0.343
E. tetradactylum	-	-	-	-	-0.602	-0.615	0.847	0.838

(* = significant at P<0.10)

The majority of commercial and recreational species captured in the three estuaries showed negative effects on their abundance during a freshwater flow event when compared with 'before the flow' (**Table 10.5**), but these were mostly not significant. With only five (out of 68) significant at $P<0.10$ (as per Bonvechio and Allen 2005), this is less than expected from random chance but this result may be largely due to the low number of years and small range of flows that were measured for these relationships. Examples of this are *Thryssa hamiltoni*, *Leiognathus decorus*, *Pomadasys kaakan*, and *Sillago* spp. in the Calliope River estuary, where the correlation coefficients (r) were between -0.85 and -0.97, but none were statistically significant.

Table 10.6 Correlations against annual flows (or log), 'after' flow events

Species	Boyne River estuary		Calliope River estuary		Fitzroy River estuary (upper)		Fitzroy River estuary (lower)	
	flow	log	flow	log	flow	log	flow	log
Acetes sp.	**0.997***	0.907	0.751	0.933	**0.999***	**0.999***	**0.999***	**0.999***
Metapenaeus spp.	**0.994***	0.971	0.479	0.710	0.886	0.830	**0.921***	0.872
Penaeus merguiensis	**0.999***	0.949	**0.973***	0.642	0.895	0.842	**0.991***	**0.971***
P. sculptilis	0.945	**0.999**	**0.989***	0.710	**0.988***	**0.965***	**0.947***	**0.907***
Thryssa hamiltoni	**0.999***	0.931	0.959	**0.998***	**0.999***	**0.999***	**0.996***	**0.994***
Leiognathus decorus	-0.576	-0.824	0.934	0.751	**0.994***	**0.992***	0.749	0.760
Pomadasys kaakan	-0.932	**-0.999***	0.446	0.729	0.334	0.318	0.815	0.805
Sillago spp.	-0.957	**-0.998***	0.311	0.621	-0.131	-0.147	0.773	0.784
E. tetradactylum	-	-	-	-	-0.339	-0.355	0.789	0.799

(* = significant at P<0.10)

For the same set of species, the majority of species showed positive correlations in abundance with 'after' flow events (**Table 10.6**). For most species, abundance was significantly and positively correlated with flow or $\log_{10}$(flow). Fifteen out of a total of 34 species were more abundant after a flow, being a greater number than would be expected from by chance.

Multivariate analysis of community structure

Over 160 species from 66 families were collected from the three estuaries, with 85 species occurring in all estuaries. *Acetes* sp. dominated the catches in all estuaries representing ~ 90% of all animals caught in the Fitzroy River and Boyne River estuaries and ~ 60% in the Calliope River estuary. In the Fitzroy River estuary, three prawn species, *Penaeus merguiensis*, *Metapenaeus* spp., and *Parapenaeopsis sculptilis* were the next most abundant (Table 10.7). A full species list appears in Appendix 10. In the smaller catchments of the Calliope and Boyne River estuaries (Table 10.7), *Ambassis gymnocephalus* was the most abundant finfish species, but was absent from the demersal community of the Fitzroy River estuary. Juvenile fish of commercial and recreational importance were not common in any of the estuaries except for whiting (*Sillago* spp.) which was present at low levels in both the Calliope and Boyne River estuaries.

Significant differences occurred between estuaries in the composition, abundance and biomass of species sampled (**Figure 10.1**) and in the hydrology driven abiotic variables (**Figure 10.2**). This is expected as there are differences in catchment size, habitat composition, extent of barriers and substrate type of the estuaries sampled.

The Boyne River estuary sustained the greatest number of individuals per unit of area. This is an interesting result as this system is the most modified in terms of freshwater flow, receiving no freshwater except that provided by rainfall events and two small environmental flow releases (that were undetected in our abiotic sampling regime, see p171). The results suggest that the estuarine habitats of the Boyne River estuary were functioning in a 'desirable way' even though there has been extensive modification to the freshwater flow regime. The long-term consequences of the reduced flow on the function of these habitats are unknown.

Stable isotope analysis of banana prawns collected from the Fitzroy and Boyne River estuaries showed that different food sources were being exploited by banana prawns in each estuary (Halliday and Revell unpublished data). This is important as the effect of freshwater flow on turbidity has ramifications for organisms in the lower trophic levels. Increased turbidity resulting from freshwater flows can cause a reduction in the light regime, limiting planktonic growth in the water column, while increasing the available nutrients and stimulating benthic primary production (micro-phyto-benthos). The ability of banana prawns to switch between abundant food sources is an adaptation that allows them to survive and grow in estuaries under a range of freshwater flow regimes. We suggest that it is likely that juvenile banana prawns are exploiting whatever food sources are available. Any blooms in their food items, such as micro-phytobenthos, that result from nutrients delivered by flows may allow juvenile banana prawns to consume more food and grow quickly between size classes, thus enhancing their survival rate.

Table 10.7 Ten most common species caught in the Fitzroy, Calliope and Boyne River estuaries

Species	Fitzroy River estuary		Calliope River estuary		Boyne River estuary	
	Total number	% of catch	Total number	% of catch	Total number	% of catch
Acetes sp.	772339	91.4	29692	58.8	489395	89.3
Penaeus merguiensis	23118	2.7	5944	11.8	21363	3.9
Ambassis gymnocephalus	-	-	4749	9.48	12900	2.3
Ambassis vachelli	-	-	1863	3.7	3651	0.7
Metapenaeus spp.	14415	1.7	1119	2.2	3684	0.7
Parapenaeopsis sculptilis	11527	1.4	-	-	-	-
Thryssa hamiltoni	2281	0.3	1754	3.5	3217	0.6
Leiognathus decorus	-	-	910	1.8	2632	0.5
Stolephorus commersonii	-	-	877	1.7	-	-
Sillago spp.	-	-	586	1.2	660	0.1
Leandrites celebensis	-	-	379	0.7	-	-
Valamugil sp.	-	-	-	-	2712	0.5
Pseudomugil signifer	-	-	-	-	1577	0.3
Larval fish 4	2205	0.3	-	-	-	-
Philypnodon grandiceps	1822	0.2	-	-	-	-
Prionobutis microps	1637	0.2	-	-	-	-
Periclimenes sp.	1565	0.2	-	-	-	-
Macrobrachium sp.	1324	0.16	-	-	-	-

Figure 10.1 MDS plot of species: (a) abundance and (b) biomass data for three years of sampling in the Fitzroy, Calliope and Boyne river estuaries

Figure 10.2 PCA plot of abiotic data collected over three years of sampling in the Fitzroy, Calliope and Boyne River estuaries

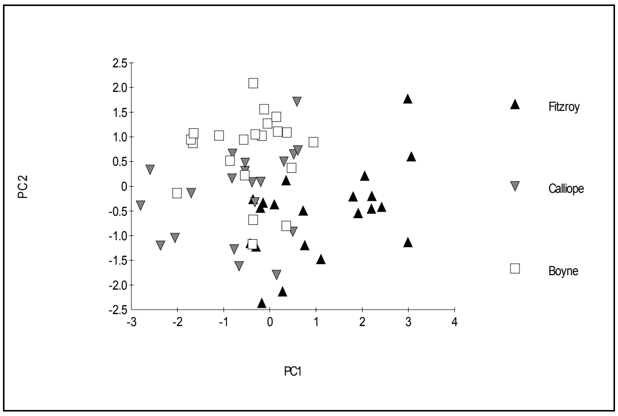

Table 10.8. ANOSIM and SIMPER results comparing species abundances and biomass between the Fitzroy, Calliope and Boyne River estuaries

Comparison of rivers		ANOSIM test probability	SIMPER main contributing species (discrimination value + or – indicates direction of change)
Species abundance	Fitzroy vs. Calliope	0.001	*Parapenaeopsis sculptilis* (2.09-), *Acetes* sp. (1.36-), *Johnius australis* (1.60-), *Ambassis gymnocephalus* (1.37+)
	Fitzroy vs. Boyne	0.001	*Ambassis gymnocephalus* (2.02+), *Leiognathus decorus* (1.69+) *Parapenaeopsis sculptilis* (1.71-), *Ambassis vachelli* (1.01+)
	Calliope vs. Boyne	0.001	*Acetes* sp. (1.67), *Penaeus merguiensis* (1.38), *Ambassis gymnocephalus* (1.31), *Ambassis vachelli* (1.09)
Species biomass	Fitzroy vs. Calliope	0.001	*Aseraggodes rautheri* (1.88), *Acetes* sp. (1.51), *Ambassis gymnocephalus* (1.39), *Leiognathus decorus* (1.79)
	Fitzroy vs. Boyne	0.001	*Ambassis gymnocephalus* (2.16), *Leiognathus decorus* (2.28), *Aseraggodes rautheri* (1.80), *Thryssa hamiltoni* (2.15)
	Calliope vs. Boyne	0.001	*Acetes* sp. (1.83), *Ambassis gymnocephalus* (1.36) *Ambassis vachelli* (1.14), *Penaeus merguiensis* (1.29)

The species abundance and biomass was significantly difference between rivers (Table 10.8). SIMPER analysis identified that the species primarily responsible for the differences between the Fitzroy and Calliope River estuaries were *Parapenaeopsis sculptilis*, *Acetes* sp. and *Johnius australis* which were more abundant in the Fitzroy River estuary and *Ambassis gymnocephalus*, which was only caught in the Calliope River estuary. Based on biomass, *Aseraggodes rautheri*, *Acetes* sp., *Ambassis gymnocephalus* and *Leiognathus decorus* caused the main difference between estuaries (Table 10.8). The demersal community of the Fitzroy and Boyne River estuaries were significantly different (Table 10.8), with *Ambassis gymnocephalus*, *Leiognathus decorus*, and *Ambassis vachelli* being more abundant in the Boyne River estuary and *Parapenaeopsis sculptilis* being more abundant in the Fitzroy River estuary. The demersal community of the Calliope and Boyne River estuaries were significantly different from each other, with *Acetes* sp., *Penaeus merguiensis*, *Ambassis gymnocephalus* and *Ambassis vachelli* being more abundant in the Boyne River estuary (Table 10.8).

Linking MDS plots of abundance and biomass with the abiotic data using the BIOENV analysis indicated that although there was relatively good matching of the abundance and biomass plots with abiotic variables, no single variable fully explained the patterns observed (**Table 10.9**). Of the four abiotic variables measured, at least three were required to represent the patterning seen in the PCA. This indicates that it was a combination of factors that determined the demersal community structure and that no single variable associated with freshwater flow was driving the abundance and structure of the demersal community within each estuary.

Table 10.9 BIOENV results for matching MDS abundances plots with PCA abiotic data

Comparison of rivers	BIOENV correlation	Environmental variables
Species abundance	0.42	Temperature, salinity, turbidity
	0.42	Temperature, salinity, pH, turbidity
	0.38	Salinity, pH, turbidity
Species biomass	0.39	Salinity, pH, turbidity
	0.38	Temperature, salinity, pH, turbidity
	0.34	pH, turbidity

Yearly differences in the Fitzroy River estuary

There were no significant differences between years in the demersal community structure of the Fitzroy River estuary (**Table 10.10**). This indicates that although there are changes in abundance and species present at different times of the year, the general pattern of community structure remains similar. Although there were different flow regimes during different years, the demersal community structure did not change substantially between years. Comparing the matching between species abundances and abiotic variables required the use of all four abiotic variables (i.e. temperature, salinity, pH, and turbidity) to achieve a 58.3% correlation. No single abiotic variable was identified that could be used to predict the likely community structure. The results of the cyclicity test (P=0.009) showed that for species abundance there was a significant patterning, with samples collected one after the other being most similar and those collected more than one month apart being significantly different from each other.

Figure 10.3 (a) MDS plot of species abundance and)b) PCA plot of associated abiotic data for three years of sampling in the Fitzroy River estuary

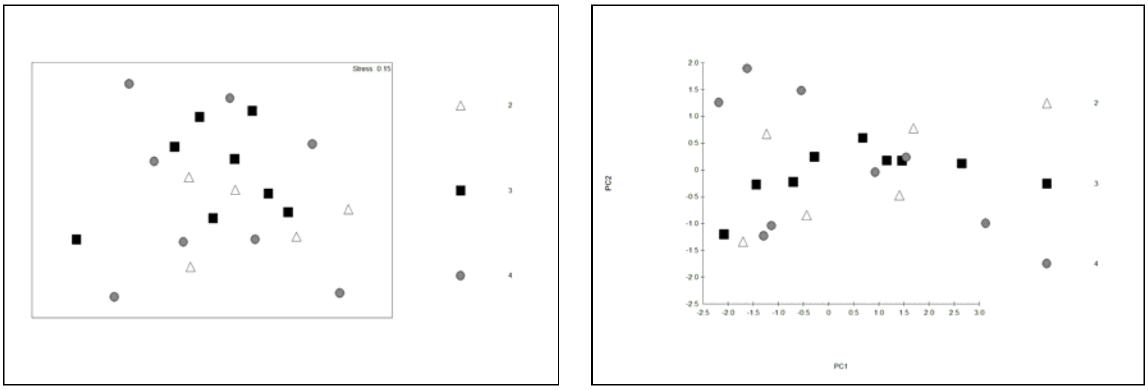

Table 10.10 ANOSIM and SIMPER results comparing species abundances between years for the Fitzroy River estuary

Comparison of year	ANOSIM test Probability	SIMPER Main Contributing Species (discrimination value + or – indicates direction of change)
Year-2 vs. Year-3	0.12	*Periclimenes* sp. (1.96-), *Leandrites celebensis* (1.93+), *Acetes* sp. (1.10-), *Parapenaeopsis sculptilis* (1.52-)
Year-2 vs. Year-4	0.11	*Acetes* sp. (1.41-), *Periclimenes* sp. (1.73-), *Penaeus merguiensis* (1.90-), *Parapenaeopsis sculptilis* (1.31-)
Year-3 vs. Year-4	0.14	*Prionobutis microps* (1.29-), *Johnius australis* (1.28+), *Acetes* sp. (1.47-), *Stolephorus commersonii* (1.60-)

Effects of flow on community structure in the Fitzroy River estuary

Using the same data but coding freshwater flow into the categories of ‘before’, ‘during’ (i.e. within six weeks of the flow beginning) and ‘after’, a general pattern emerged, indicating that the response of the demersal community to freshwater flow events was similar in each year. The MDS of community structure and PCA of abiotic data (Figure 10.4) showed a similar patterning, with samples collected ‘before’, ‘during’ and ‘after’ the freshwater flows being relatively discrete.

Figure 10.4 (a) MDS plots of species abundance and (b) PCA plot of abiotic data for ‘before’, ‘during’ and ‘after’ flow events over three years in the Fitzroy River estuary

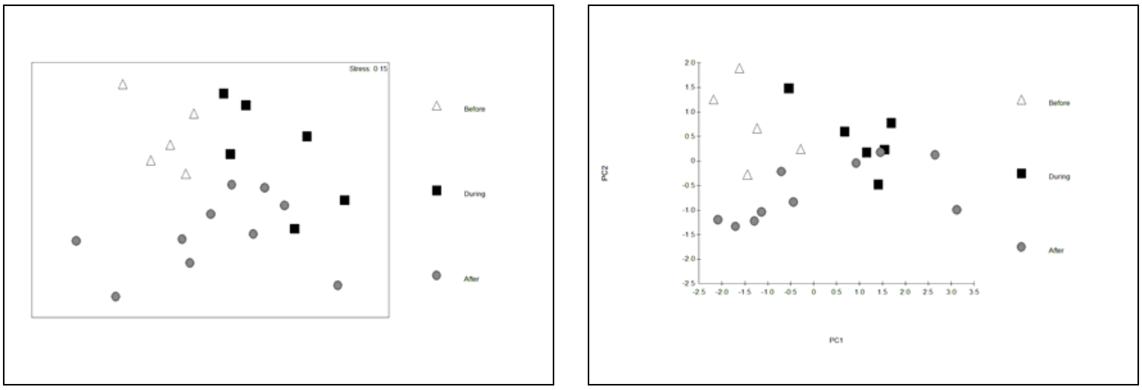

Table 10.11 ANOSIM and SIMPER results comparing species abundances ‘before’, ‘during’ and ‘after’ flow events for the Fitzroy River estuary

Comparison of flow	ANOSIM test probability	SIMPER main contributing species (discrimination value + or – indicates direction of change)
Before vs. During	0.03*	*Parapenaeopsis sculptilis* (1.28+), *Nematalosa erebi* (1.21+), *Philypnodon grandiceps* (1.10+), *Acetes* sp. (1.58-)
Before vs. After	0.01*	*Prionobutis microps* (1.65+), *Thryssa hamiltoni* (2.30-), *Acetes* sp. (1.34+)
During vs. After	0.06	*Thryssa hamiltoni* (1.62-), *Periclimenes* sp. (1.09-), *Nematalosa erebi* (1.29+), *Philypnodon grandiceps* (1.10-)

(*P<0.05)

The demersal community structure of the estuary defined *a priori* to analysis into either ‘before’, ‘during’ and ‘after’ flow events showed significant differences in the Fitzroy River estuary between ‘before’ versus ‘during’ flow events (**Table 10.11**). This was a consequence of the prawn *Parapenaeopsis sculptilis,* the freshwater bony bream, *Nematalosa erebi*, and *Philypnodon grandiceps* being more abundant ‘during’ flow events, and *Acetes* sp. being less abundant ‘during’ flow events. Significant differences between the ‘before’ and ‘after’ flow samples were primarily driven by *Prionobutis microps* and *Acetes* sp. becoming more abundant ‘after’ flow events and *Thryssa hamiltoni* becoming less abundant ‘after’ flow events than ‘before’ flow events.

Yearly differences in the Calliope River estuary

The three years of data from the Calliope River estuary showed that there were significant differences in the community structure between some years. While year-2 and year-4 were not significantly different from one another, year-3 was significantly different from both year-2 and year-4 (**Table 10.12**). The MDS plot of species abundance (**Figure 10.5**) shows that there was unclear separation between the three years of data. The PCA plot of abiotic data shows similar patterning to the MDS of species abundance with BIOENV analysis needing three of the four abiotic variables to achieve a 35.6% matching with the community structure. The results of the cyclicity test (P=0.001) showed that for species abundance there was significant patterning, with samples collected one after the other being most similar to each other and those collected more than one month apart being significantly different from each other.

Figure 10.5 (a) MDS plot of species abundance and (b) PCA plot of abiotic data for 'before', 'during' and 'after' flow events over three years in the Calliope River estuary

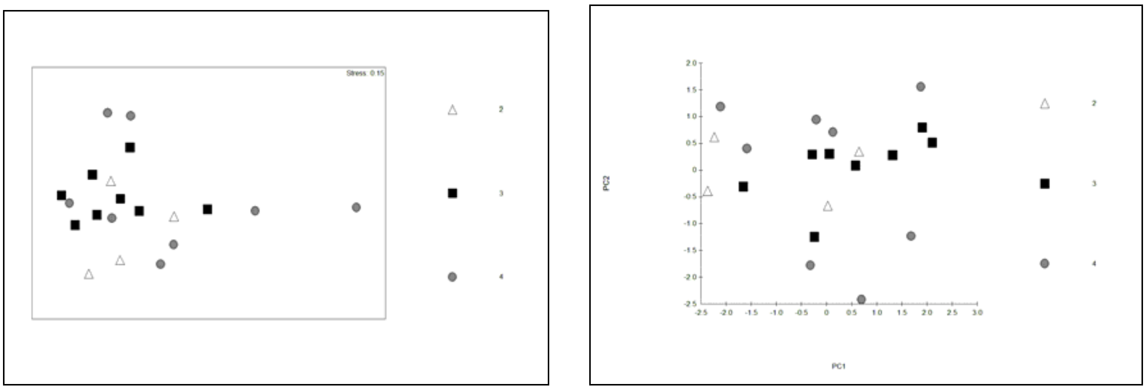

Table 10.12 ANOSIM and SIMPER results comparing species abundances between years for the Calliope River estuary

Comparison of year	ANOSIM test probability	SIMPER main contributing species (discrimination value + or – indicates direction of change)
Year-2 vs. Year-3	0.05*	*Leandrites celebensis* (1.98+), *Ambassis gymnocephalus* (1.34-), *Ambassis vachelli* (1.10+), *Valamugil* sp. (1.27-)
Year-2 vs. Year-4	0.17	*Ambassis gymnocephalus* (1.60-), *Penaeus merguiensis* (1.93-), *Valamugil* sp. (1.23-), *Stolephorus commersonii* (1.50+)
Year-3 vs. Year-4	0.04*	*Penaeus merguiensis* (2.02-), *Ambassis gymnocephalus* (1.41-), *Ambassis vachelli* (1.24-), *Leandrites celebensis* (1.51-)

(*P<0.05)

SIMPER analysis showed that the species most responsible for the differences between years of sampling were *Leandrites celebensis* and *Ambassis vachelli* which were less abundant in year-2 than in year-3 and *Ambassis gymnocephalus* and the mullet, *Valamugil* sp. which were more abundant in year-2 than in year-3. The significant differences between year-3 and year-4 samples were primarily related to the abundance of banana prawns *Penaeus merguiensis*, *Ambassis gymnocephalus*, *Ambassis vachelli* and *Leandrites celebensis,* all of which were significantly more abundant in year-3 than in year-4.

Effects of flow on community structure in the Calliope River estuary

Significant differences were found for the effect of flow on the community structure of the Calliope River estuary between 'before' and 'during' flow events and 'before' and 'after' flow events (Table 10.13). There was no significant difference between samples taken 'during' and 'after'flows. SIMPER analysis indicated that the abundances of *Ambassis gymnocephalus, Penaeus merguiensis, Stolephorus commersonii,* and *Herklotsichthys castelnaui* were all higher 'during' flow events than 'before' flow events. Significant differences between 'before' and 'after' flow events were primarily driven by *Favonigobius exquisitus* and *Leandrites celebensis* which were more abundant 'before' flow events than 'after' and *Ambassis gymnocephalus* and *Penaeus merguiensis* which were less abundant 'before' flow events than 'after'.

Figure 10.6 (a) MDS plot of species abundance and (b) PCA plot of abiotic data for 'before', 'during' and 'after' flow events over three years in the Calliope River estuary

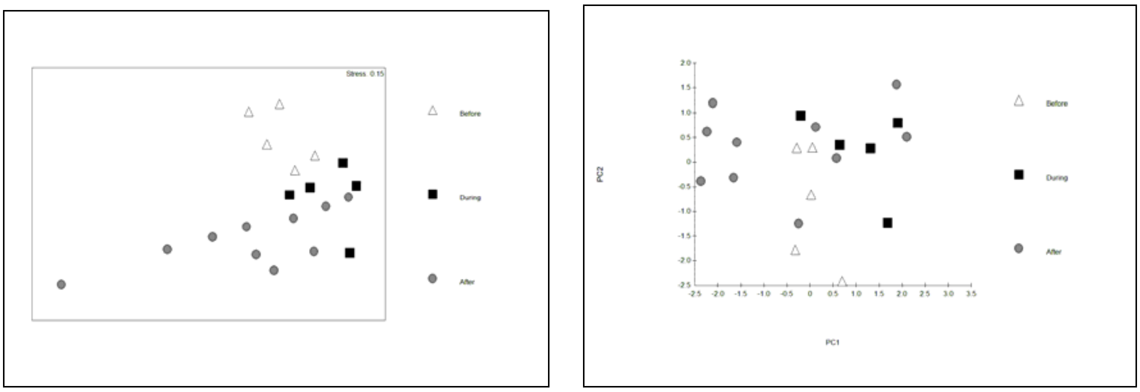

Table 10.13 ANOSIM and SIMPER results comparing species abundances 'before', 'during' and 'after' flow events for the Calliope River estuary

Comparison of flow	ANOSIM test probability	SIMPER main contributing species (discrimination value + or – indicates direction of change)
Before vs. During	0.01*	*Ambassis gymnocephalus* (1.35+), *Penaeus merguiensis* (1.29+), *Stolephorus commersonii* (1.86+), *Herklotsichthys castelnaui* (1.16+)
Before vs. After	0.01*	*Favonigobius exquisitus* (2.03-), *Ambassis gymnocephalus* (1.35+), *Penaeus merguiensis* (1.33+), *Leandrites celebensis* (1.66-)
During vs. After	0.37	*Ambassis gymnocephalus* (1.23-), *Thryssa hamiltoni* (1.25-), *Stolephorus commersonii* (1.55-), *Herklotsichthys castelnaui* (1.08-)

(*P<0.05)

Yearly differences in the Boyne River estuary

The demersal community structure of the Boyne River estuary showed no significant differences between years (**Table 10.14**). The MDS plot of species abundance and PCA of abiotic data (**Figure 10.7**) showed no clear separation between years, with a large amount of scatter in the data. While there were changes in the abundance of most of the species between years (**Table 10.14**), these were not sufficiently different to create significant changes in the community structure. The PCA plot of abiotic data showed little similarity to the MDS of species abundance, with BIOENV analysis indicating that pH was the only variable to show matching at 16.4%. The cyclicity test of species abundance showed significant patterning (P=0.03), with samples collected one after the other being most similar and those collected more than one month apart being significantly different.

Figure 10.7 (a) MDS plot of species abundance and (b) PCA plot of abiotic data for 'before', 'during' and 'after' flow events over three years in the Boyne River estuary

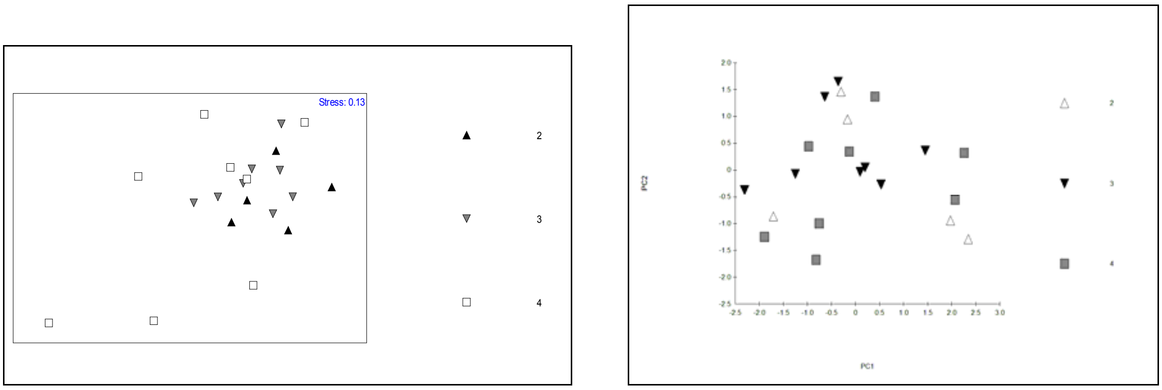

Table 10.14 ANOSIM results comparing species abundances between years for the Boyne River estuary

Comparison of year	ANOSIM test probability	SIMPER main contributing species (discrimination value + or – indicates direction of change)
Year-2 vs. Year-3	0.28	*Ambassis vachelli* (1.08+), *Pseudomugil signifier* (1.58+), *Periclimenes* sp (1.71-), *Leandrites celebensis* (1.29+)
Year-2 vs. Year-4	0.19	*Penaeus merguiensis* (1.33-), *Acetes* sp. (1.23-), *Ambassis vachelli* (1.20+), *Ambassis gymnocephalus* (1.02-)
Year-3 vs. Year-4	0.06	*Penaeus merguiensis* (1.55-), *Ambassis vachelli* (1.16-), *Ambassis gymnocephalus* (1.18-), *Acetes* sp. (1.33-)

Effects of flow on community structure in the Boyne River estuary

Effect of flow resulting from rainfall events on the community structure of the Boyne River estuary was significantly different only for the samples taken 'before' and 'after' rainfall events. These were characterised by an increase in the abundance of *Ambassis vachelli, Penaeus merguiensis*, and *Acetes* sp. 'after' rainfall events and a decrease in the abundance of *Pseudomugil signifier* 'after' rainfall events.

Figure 10.8 (a) MDS plot of species abundance and (b) PCA plot of abiotic data for 'before', 'during' and 'after' rainfall events over three years in the Boyne River estuary

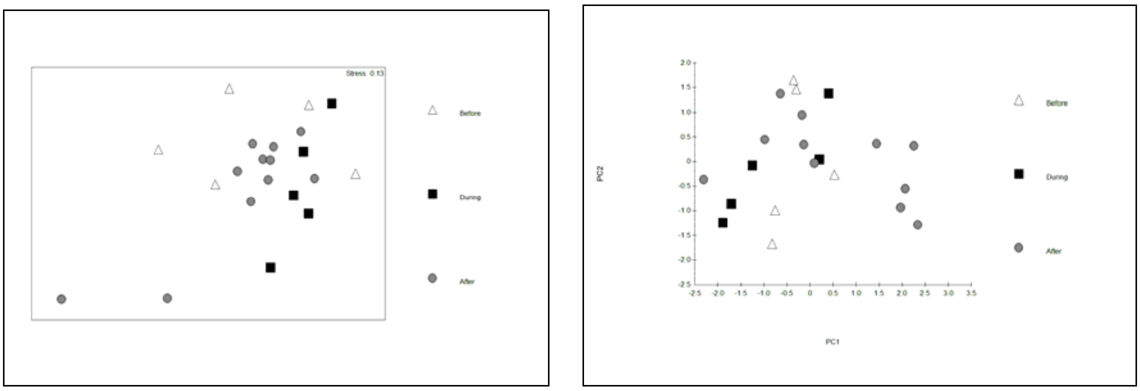

Table 10.15 ANOSIM results comparing species abundances 'before', 'during' and 'after' rainfall events for the Boyne River estuary

Comparison of flow	ANOSIM test probability	SIMPER main contributing species (discrimination value + or – indicates direction of change)
Before vs. During	0.07	*Pseudomugil signifier* (1.65-), *Thryssa setirostris* (2.52+), *Herklotsichthys castelnaui* (1.43+), *Penaeus merguiensis* (1.37-)
Before vs. After	**0.041***	*Ambassis vachelli* (1.23+), *Penaeus merguiensis* (1.46+), *Pseudomugil signifier* (1.37-), *Acetes* sp. (1.53+)
During vs. After	0.09	*Ambassis vachelli* (1.16+), *Herklotsichthys castelnaui* (2.05-), *Penaeus merguiensis* (1.14+), *Ambassis gymnocephalus* (1.01-)

(*P<0.05)

Discussion

The Fitzroy, Calliope and Boyne River estuaries showed significant differences in their demersal community structures separating them from one another at an estuary scale. These differences in community structure were expected given the different catchment and estuary sizes, geomorphologies, estuarine habitat types and quantity of freshwater inputs. The differences were persistent over time as indicated by the similarity of the demersal community structure over years within each estuary.

However, the demersal communities within each estuary did react to freshwater flows (or rainfall) in a similar way with significant differences in the demersal community structure 'before', 'during' and 'after' freshwater flow events. Generally, there was a decrease in the abundance of species or complete absence of many species 'during' a flow event with an increase in abundance once the estuary started to recover from the freshwater input. One explanation for this is the movement of fauna away from the poor water quality of the first flush of fresh water associated with flow events, which usually has elevated nutrient and sediment loads, decreased oxygen and suddenly decrease the salinity of the estuary (Eyre 1997; Eyre and Twigg 1997). This would result in considerable physiological stress to fauna inhabiting the estuary prior to the flow, stimulating their movement away from poor water quality towards more benign conditions. Many of the smaller species with limited swimming abilities may be physically flushed from the estuary by increased flow rates. We speculate that in the short term, the estuary is re-colonised by individuals recruited from the marine environment as well as individuals swept down from freshwater reaches. These individuals acclimatise to the hydrological condition of the estuary and slowly move upstream as the saline wedge gradually intrudes upstream.

Although, sampling for the present study occurred during three years of drought, there was sufficient freshwater input to impact the demersal community structure in all three estuaries. We would expect that under higher flow conditions the demersal community structure would show greater variation in response to the timing and duration of the freshwater flows and the length of the recovery time within each of the estuaries.

Demersal communities of estuaries have evolved to accommodate large changes in environmental conditions, the largest of which is freshwater flow input and its subsequent effects. The changing nature of estuaries provides for a large diversity of species with different hydrological tolerances to inhabit estuaries and take advantage of suitable conditions when they arise. One of the main impacts of reducing freshwater flows into estuaries would be the stabilization of the demersal community structure favouring predominantly marine species, while making it less favourable for freshwater and brackish species that may need access to marine waters for reproduction. This would effectively decrease the biodiversity of the estuary compared to its pre-regulation condition.

No single abiotic parameter adequately predicted the demersal community structure in the three study estuaries, indicating that a range of water quality parameters alone are unlikely to be an effective predictor of the biodiversity of an estuary. This is compounded by the difficulty in selecting appropriate references sites within an estuary because of the changing nature of the environment. It is also difficult to define the optimal attributes an estuarine community structure, given the individual nature of the communities found in the Fitzroy, Calliope and Boyne River estuaries. While distinctly different communities inhabit each estuary, these communities can change significantly under the influence of freshwater flows. The idea that stability in estuarine community structures is good has to be dispelled. Variability in demersal community structure is part of what makes estuaries such dynamic places with changes to the extent of available habitats and water quality parameters a fundamental characteristic of estuaries.

Chapter 11. Benefits

The main benefits of the project have been:

- To provide a logical conceptual framework for addressing the issue of determining the role of freshwater in estuarine fisheries production. This has been used by other scientists to develop research projects on the role of freshwater for fisheries production in temperate Australian estuaries and by climatologists reviewing the potential impacts of climate change on fisheries production.
- To provide quantitative evidence of the role of freshwater flows in the recruitment, growth and abundance of estuarine fisheries species (e.g. banana prawns, barramundi, king threadfin, whiting and mud crabs). Quantitative and scientifically robust evidence is important in showing that freshwater flowing to the ocean via estuaries is not a waste of water.
- To raise the profile of the need for freshwater to flow to estuaries with water resource managers, the wider scientific community and to a lesser degree with the general public.
- To assist in a paradigm shift in (at least in Queensland water management) as to reasons why environmental flows should be allocated for estuaries (e.g. for improved growth and survival of young-of-the year fish rather than to trigger spawning with the first-post-winter-flow) and to identify biological indictors that may be suitable to monitor in order to determine that environmental flow allocations are achieving the desired management outcome (i.e. what you might monitor).
- To demonstrate that annual monitoring of the barramundi population (in the Fitzroy region) can be achieved by sampling fish at commercial seafood processors, and this method has been subsequently adopted as the standard yearly sampling protocol of the Long Term Monitoring Program for barramundi in central Queensland by the Department of Primary Industries & Fisheries.
- To provide further evidence that barramundi populations are influenced by regional environmental factors and as such, stock assessment should be carried out at regional spatial scales. Regional stock assessment has been recently adopted by the Department of Primary Industries & Fisheries, partly in response to results presented within.

Dissemination of research results

Research results from the project have been made available to a wide variety of audiences.

Public media articles

A press release on the project aims was published in Queensland Country Life, 28th February 2002.

WIN news (Channel 9 central Queensland) ran a segment on the project headlined "Groundbreaking" on the 11th February 2002 discussing the importance of the project. This was also broadcast in other regional WIN stations in northern Queensland.

Rockhampton ABC radio interviewed Ian Halliday in March 2002 about the project aims and desired outcomes, which was aired in the Rockhampton district.

A full page feature on the environmental flows for estuaries project was published in the Saturday edition of the Rockhampton Morning Bulletin, on the 3rd August 2002.

Channel 7 Mackay interviewed Ian Halliday regarding "The effects of droughts on estuarine fisheries production and the need for freshwater flows to estuaries" which aired on 30th January 2003.

ABC radio (regional – central Queensland) interviewed Jonathan Staunton-Smith about the research being undertaken by the environmental flows project and the likely effects of the February 2003 flood that occurred in the Rockhampton and Gladstone area.

An article on the research being undertaken by the project appeared in the Rockhampton Morning Bulletin (March 2003), as well as appearing in the outdoors section of the Queensland Courier Mail (March 2003).

"The Travelling Fisherman", a fishing show broadcast in northern Australia, interviewed Julie Robins on the effects of floods on barramundi, in January 2004.

CZCRC articles e.g., newsletter

A research update on the project was published in the Coastal CRC monthly newsletter Flotsam and Jetsam February 2002 entitled "Estuarine fish help identify past river flows".

A one-page double-sided handout on the preliminary findings of the barramundi work was distributed in hard copy to commercial and recreational fishers in the Fitzroy and Port Curtis region; water management agencies, schools and public libraries.

Industry articles

Research focus on river flows to boost fish stocks, The Queensland Fisherman, March 2002.

Fitzroy flooding helps research, The Queensland Fisherman, April 2003.

Milestone reports submitted to the Finfish Management Advisory Committee, Fisheries Business Group of the Queensland Department of Primary Industries & Fisheries.

Presentations and seminars

Fitzroy region stakeholders, 19th March 2002.

CRC project review committee, 20th March 2002.

Port Curtis stakeholders, 30th April 2002.

Adaptive Environmental Assessment and Management (AEAM) workshop hosted by the Catchment Hydrology CRC and the Coastal Zone CRC, 14 and 15th May 2002 - contributed fisheries production models relating to flow for the Fitzroy region to the Fitzroy Basin System Model.

Recreational fishers meeting Yeppoon and Mackay, 2002.

Australian Society for Fish Biology Conference, August 2002.

State of Port Curtis Conference Gladstone, 23rd to 25th October 2002.

Finfish Management Advisory Committee of the Queensland Fisheries Service, March 2003.

Land & Water Australia workshop on research into environmental flows for estuaries, April 2003 Tasmania.

Australian Institute of Marine Science, April 2003.

CZCRC seminar morning for water policy and management personal (e.g. QDNR&M head office), fisheries policy and management personal (e.g. the Queensland Fisheries Service) and other key stakeholders in natural resource management (e.g., EPA personal), June 2003.

International River Symposium, Brisbane September 2003.

CZCRC Review Panel, 27th October 2003.

CZCRC Stakeholder Workshop in Yeppoon, 28th October 2003.

National Estuaries Network (NEN) meeting, November 2003, Hobart.

2004 Sustainable Futures for Australia's Tropical Rivers Forum, Darwin, 1-3rd Feb 2004.

Estuarine and Coastal Sciences Association and Estuarine Research Federation International Conference on "Estuaries and Change", Ballina, June 2004.

Calliope River Water Planning Technical Advisory Panel, 23rd August 2004.

Water Planning Directors of QDRN&M, 6th September 2004.

Capricorn Shire Council (assessing the fisheries requirements of environmental flows for the Water Park Creek and Sandy Creek Catchments in the Yeppoon area).

Calliope/Boyne NRM Group planning meeting.

Australian Society for Fish Biology Symposium, August 2004.

CZCRC Fitzroy Stakeholders meeting, 8th November 2004.

Coastal Zone CRC Seminar, 12th November 2004.

DPI&F Southern Fisheries Seminar, 19th November 2004.

MODSIM 2005 International Congress on Modelling and Simulation, December 2005.

Internet information

The website (http://www.dpi.qld.gov.au/far/13055.html) was developed, with pages for the presentations and links from the QDPI&F page to FRDC page.

Published papers (alphabetically) – available upon request

Balston, J. (2005). Modelling impact of climate variability on the commercial wild barramundi (*Lates calcarifer*) fishery of north-eastern Queensland. In Zerger, A. and Argent, R.M. (eds.) MODSIM 2005 International Congress on Modelling and Simulation. Modelling and Simulation Society of Australia and New Zealand, December 2005, pp. 475-481. ISBN: 0-9758400-2-9. Available at: http://www.mssanz.org.au/modsim05/proceedings/papers/balston.pdf

Halliday, I.A., Robins, J.B. and Staunton-Smith, J. (2003). A conceptual framework for investigating environmental flows for estuarine fisheries production. Selected papers from the 2003 International River Symposium.

Halliday, I.A., Robins, J.B., Mayer, D.G., Staunton-Smith, J. and Sellin, M.J. (in preparation). Freshwater flows and year-class strength of barramundi (*Lates calcarifer*) in a dry tropical estuary; an update.

Halliday, I.A., Robins, J.B., Mayer, D.G., Staunton-Smith, J. and Sellin, M.J. (submitted). Effects of freshwater flow on the year-class strength of a non-diadromous estuarine finfish, *Polydactylus macrochir*, in a dry tropical estuary. *Marine and Freshwater Research.*

Milton, D., Halliday, I., Staunton-Smith, J., Sellin, M., Marsh, R., Smith, D., Norman, M. and Woodhead, J. (submitted). Otolith chemistry of barramundi *Lates calcarifer* can provide insight about the role of freshwater flows in maintaining estuarine populations of regulated rivers. *Estuarine, Coastal and Shelf Science.*

Mayer, D., Roy, D., Robins, J., Halliday, I. and Sellin, M. (2005). Modelling zero-inflated fish counts in estuaries – A comparison of alternate statistical distributions. In Zerger, A. and Argent, R.M. (eds.) MODSIM 2005 International Congress on Modelling and Simulation. Modelling and Simulation Society of Australia and New Zealand, December 2005, pp. 2581-2587. ISBN: 0-9758400-2-9. Available at: http://www.mssanz.org.au/modsim05/proceedings/papers/mayer.pdf

Robins, J.B., Halliday, I.A., Staunton-Smith, J., Mayer, D.G. and Sellin, M.J. (2005). Freshwater-flow requirements of estuarine fisheries in tropical Australia: a review of the state of

knowledge and application of a suggested approach. *Marine and Freshwater Research* **56**: 343-360.

Robins, J., Mayer, D., Staunton-Smith, J., Halliday, I., Sawynok, B. and Sellin, M. (2006). Variable growth rates of the tropical estuarine fish species barramundi, *Lates calcarifer* (Bloch) under different freshwater flow conditions. *Journal of Fish Biology* **69**: 379-391.

Staunton-Smith, J., Robins, J.B., Mayer, D.G., Sellin, M.J. and Halliday, I.A. (2004). Does the quantity and timing of fresh water flowing into a dry tropical estuary affect year-class strength of barramundi (*Lates calcarifer*)? *Marine and Freshwater Research* **55**: 787-797.

Information from the Flows for Estuaries Project has been passed onto participants of the national barramundi modelling workshop held in Darwin, July 2005. This is predominately information and data supporting the scientific basis for the inclusion of environmental variables (such as river flow or rainfall) in stock assessment models.

Published papers and other information have been passed on to the Bureau of Rural Sciences for an article on drought and fisheries.

Chapter 12. Further development

Understanding of the role of freshwater flows in estuarine fisheries production and the effect of modifying such flows through water abstraction and management (or as a consequence of long-term climate change) would benefit from knowledge gained by:

- Determining whether relationships between freshwater flow and fisheries production (i.e. recruitment, growth and abundance) observed for fishery species in central Queensland estuaries are consistent in other estuaries.
- Disentangling the effects of the physical aspects of freshwater flows (e.g. volume, duration, frequency) from the nutrient delivery aspects.
- Further correlation of fisheries catch data with selected hydrological indices that describe unique and non-redundant patterns of variance such as frequency, timing, duration, rate of change (as per Olden and Poff 2003) and threshold effects.
- Determining the mechanisms by which estuarine productivity is enhanced through increased freshwater flow, and how (and whether) this productivity is progressed up the food chain.
- Determining whether the annual trends in juvenile abundance and commercial catch of banana prawns are related to the salinity profiles through increased areas of favourable habitat.
- Further biological research on important estuarine fishery species such as king threadfin, blue threadfin and mud crabs, that would clarify hypotheses on the effects of flow on these species, such as the biological link between autumn flows and their effect on mud crab recruitment as related to burrow competition.
- Further research to clarify how barramundi recruitment and productivity are enhanced as a consequence of increased freshwater flow, in particular the timing of these flows, their magnitude, the mechanism by which the effects occur (including habitats access) and whether the relationship is linear or threshold.
- Integration of flow related effects on the recruitment and growth of barramundi into the national generic barramundi population model to allow exploration of water abstraction scenarios e.g. the extent to which the quantity, duration or frequency of summer freshwater flows could be modified and what effects these modified flows would subsequently have on estuarine fish populations.
- Further exploration and modelling of the impacts of freshwater flow on the growth rates of juvenile banana prawns and the consequences of this on the abundance and biomass of juvenile banana prawns pre-recruitment to offshore commercial fisheries.
- Determining the link between estuarine and inshore productivity, and exploring the consequences of changed flow scenarios as a result of predicted climate change.
- Determining the impact of stocked barramundi on the abundance, genetic diversity and sustainability of the commercial estuarine fisheries.

Chapter 13. Planned outcomes

In 2001, when the project proposal was written, we planned to ‘provide scientific advice to water managers, fishing industry representatives, and the community in general, about the importance of freshwater flows to estuarine fisheries production.’

To some extent, this outcome has been achieved by raising the awareness of natural resource managers, of both fisheries and water, as to the need to consider the role of freshwater in sustaining estuaries. There has been a change in the paradigm of government agencies responsible for water management, where prior to the current project estuaries were mentioned but not explicitly considered in water resource plans, particularly in Queensland. We argue that project results and presentations by project staff to water resource managers has contributed to this change, which can be seen in more recent water resource plans, where estuaries are being considered more explicitly. However, the greatest difficulty is the quantification of the freshwater needs of estuaries.

Results from the current project have begun to quantify the timing and quantity of water required to flow to estuaries to sustain fisheries production, particularly in terms of the catchability, recruitment strength (i.e. year-class strength) and growth rates of some (high profile) estuarine species. Raised awareness and better quantification of the freshwater needs of estuaries will assist water managers on behalf of the community to balance agricultural, industrial and urban needs for water with those of estuaries and the fisheries species contained within them.

We developed and/or applied methods for determining the role of freshwater flow in estuarine fisheries production (e.g. year-class strength Chapter 6, seasonal growth curves adjusted for freshwater flow, Chapters 7 and 9) and developed a logical framework for approaching the issue (Chapter 2). However, further assessment and modelling of the data collected within this project is required to quantitatively address the challenging issue of assessing changes in fisheries production due to changes in freshwater flow, be they a consequence of water abstraction or as a consequence of climate change. Such information is critical as the water resources of Australia, particularly in sub-tropical and temperate Australia, are stretched to the limit as a consequence of climatic variability and human use.

Chapter 14. Conclusions

Freshwater flowing to estuaries (and the ocean) is not wasted. Rather, as quantitatively demonstrated by the results of the present work, freshwater plays an important role in the recruitment of juvenile fish and crustaceans, enhances growth rates (and probably the subsequent survival and abundance) of estuarine fish and crustaceans, and is an essential element driving changes in the salinity state of the estuary and the cyclicity of demersal assemblages. Freshwater flow affects estuarine fauna on daily, monthly, yearly and decadal scales. The challenge for humans, as users of both water and estuarine resources, is to understand the complex role of freshwater in estuaries and then make water efficiencies (i.e. achieve the same effect with less water) so that there is sufficient water for human use, as well as environmental allocations to sustain estuaries and their dependent fisheries.

Without freshwater flowing to the ocean, there would be no estuaries, nor estuarine-dependent fish and crustaceans, nor commercial, recreational or indigenous fisheries that exploit these estuarine-dependent fishery species.

Extent to which objectives have been achieved

1. To review the current knowledge of the relationship between freshwater flows and estuarine fisheries production.

We reviewed the current knowledge by focusing on papers correlating catch with flow, although there were also numerous papers dealing with the theoretical relationships between flow and production. This provided a strong hypothetical background developing the work schedule of the project. Most papers report positive correlations between catch and flow, with finfish, crustaceans and bivalves being affected. The review highlighted the need for greater understanding of the causal mechanisms of the relationships between flow and fisheries production. Correlation and modelling can assist in assessing these relationships, but cannot make up for a lack of appropriate data or system understanding (Sharp 1995). Understanding causal mechanisms allows greater certainty in answering questions in regards to how much, when, and of what duration freshwater is needed to flow to estuaries in order sustain fishery populations.

2. To develop a logical framework for investigating (i) the role of freshwater flow, and (ii) the effects of modified flows, on estuarine fisheries production.

We developed a logical framework that incorporates fisheries production data (i.e. catch, growth, recruitment) over varying timeframes with hydrological considerations while including aspects of the species biology. Conceptual models of species-of-interest were essential in determining at which stages of a species life-cycle freshwater flows were most likely to influence. The framework provides a robust and logical starting point from which knowledge gaps can be identified and hypotheses developed prior to instigating a research or monitoring program relating to the effects of freshwater flow. It can be applied to any estuary, but lack of knowledge or quantitative data for the biology of a species or hydrological data of the estuary in question will limit the outcomes from the framework.

3. To correlate historical flow and fisheries production data of sub-tropical estuaries.

We correlated catches of the major commercial species of central Queensland (i.e. barramundi, banana prawns and mud crabs) with freshwater flow. This was the first step in understanding the role of freshwater flows in the life-cycles of these species in the study area. Determining that there were positive effects on catch led us to the questions of why and directed our subsequent research efforts.

4. To develop procedures for assessing the changes in Queensland's estuarine fisheries production that result from water abstraction and regulation.

Using the framework and procedures outlined above, we investigated methods that would assist in estimating the potential impacts of altered freshwater flows to estuaries on fisheries production. Useful procedures were: (i) assessing year-class strength based on hindcasted estimates from otolith based age-assessment of estuarine fish (e.g. barramundi, king threadfin and sand whiting); (ii) assessing growth rates, as related to flow, using: (a) tag-recapture information (e.g. barramundi) (b) otolith micro-chemistry correlated to otolith increment analysis, and (c) serial length-frequency analysis (e.g. banana prawns); (iii) assessing changes in demersal community structure before, during and after flows; (iv) correlative analyses between catch and flow (and climatic) variables; and (v) assessing freshwater access by estuarine fish (e.g. barramundi) using otolith micro-chemistry. Results from these procedures provided quantitative evidence that the catchability, recruitment (in terms of year-class strength) and growth of selected fishery species (e.g. barramundi, banana prawns and sand whiting) were positively influenced by freshwater flows. As such, changes in fisheries production resulting from water abstraction and regulation will vary with the level of impact (and with the presence of potential thresholds), but are likely to be greater (and negative) with increasing development and abstraction.

5. To develop and communicate guidelines on environmental flows for estuarine fisheries to water managers, water users, the fishing industry and the general community.

Guidelines for the monitoring of environmental flows were difficult to develop as the duration of water resource plans, their desired outcomes and the level of monitoring have not been specified in sufficient biological detail. However, our results support the following generalisations:

- Environmental flows affecting the catchability of a species will probably be related to the seasonality of the fishery, with flows preceding or occurring during the main fishing season having the most influential effects on any particular species.
- Environmental flows affecting the recruitment of a species may not be temporally limited, outside the need for such flows to coincide approximately with the spawning season, which is extended in many tropical and some sub-tropical species. However, the number of times such flows occur within a decade may need to be explicitly considered so that recruitment pulses necessary to sustain populations of long-lived estuarine fish species can occur.
- Environmental flows affecting the growth rates of estuarine species are likely to vary seasonally, with the impact of altering flows being greatest probably in summer, late spring and early autumn for long-lived species which have annual seasonal growth, such as barramundi. The issue is more complex for short-lived species such as banana prawns.
- Environmental flows affecting the connectivity of habitats for estuarine finfish species is an important reason for allocating freshwater, but requires further understanding before guidelines could be made.
- Any monitoring of the consequences of environmental flow allocations to estuaries should be based on explicit aspects of the biology or life history of a number of estuarine species, so that a range of measures are collected and used for assessment.

Chapter 15. References

Adams, J.B., Bate, G.C., Harrison, T.D., Huizinga, P., Taljaard, S., van Niekerk, L., Plumstead, E.E., Whitfield, A.K., and Wooldridge, T.H. (2002). A method to assess the freshwater inflow requirements of estuaries and application to the Mtata Estuary, South Africa. *Estuaries* **25**, 1382-1393.

Alber, M. (2002). A conceptual model of estuarine freshwater inflow management. *Estuaries* **25**, 1246-1261.

Aleem, A.A. (1972). Effect of river outflow management on marine life. *Marine Biology* **15**, 200-208.

Ardisson, P-L., and Bourget, E. (1997). A study of the relationship between freshwater runoff and benthos abundance: a scale-oriented approach. *Estuarine, Coastal and Shelf Science* **45**, 535-545.

Arnold, D.B. (1996). Changes to mangrove ecosystem distribution: Port Curtis 1941 to 1989. In 'Mangroves - a resource under threat? An issue for the central Queensland coast'. (Eds D. Hopley and L. Warner.) pp. 41-58.

Arriola, F.J. (1940). A preliminary study of the life history of *Scylla serrata* (Forskal). *Philippines Journal of Science* **73**, 437-456.

Bacon, C.R., Weber, P.K., Larsen, K.A., Reisenbichler, R., Fitzpatrick, J.A., and Wooden, J.L. (2004). Migration and rearing histories of chinook salmon (*Oncorhynchus tshawytscha*) determined by ion microprobe Sr isotope and Sr/Ca transects of otoliths. *Canadian Journal of Fisheries and Aquatic Sciences* **61**, 2425-2439.

Bao, Y., and Mays, L.W. (1994). New methodology for optimization of freshwater inflows to estuaries. *Journal of Water Resources Planning and Management* **120**, 199-217.

Barnett-Johnson, R., Ramos, F.C., Grimes, C.B., and MacFarlane, R.B. (2005).Validation of Sr isotopes in otoliths by laser ablation multicollector inductively coupled plasma mass spectrometry (LA-ICPMS): opening avenues in fisheries science applications. *Canadian Journal of Fisheries and Aquatic Sciences* **62**, 2425-2430.

Baumann, M. (1998). The fallacy of the missing middle: physics →···→ fisheries. *Fisheries Oceanography* **7**, 63-65.

Blain, Bremner, and Williams (1980). Report of a study of extreme water level frequencies in the vicinity of Port Curtis and the Narrows. Blain, Bremner and Williams, Brisbane, Australia.

Bonvechio, T.F., and Allen, M.S. (2005). Relations between hydrological variables and year-class strength of sportfish in eight Florida waterbodies. *Hydrobiologia* **532**, 193-207.

Browder, J.A. (1985). Relationship between pink shrimp production on the Tortugas grounds and water flow patterns in the Florida Everglades. *Bulletin of Marine Science* **37**, 839-856.

Browder, J.A., Zein-Eldin, Z., Criales, M.M., Robblee, M.B., Wong, S., Jackson, T. L., and Johnson, D. (2002). Dynamics of pink shrimp (*Farfantepenaeus duorarum*) recruitment potential in relation to salinity and temperature in Florida Bay. *Estuaries* **25**, 1355-1371.

Burchmore, J.J., Pollard, D.A., Middleton, M.J., Bell, J.D., and Pease, B.C. (1988). Biology of four species of whiting (Pisces: Sillaginidae) in Botany Bay, NSW. *Australian Journal of Marine and Freshwater Research* **39**, 709-727.

Campana, S.E. (1999). Chemistry and composition of fish otoliths: pathways, mechanisms and applications. *Marine Ecology Progress Series* **188**, 263-297.

Cardona, L. (2000). Effects of salinity on the habitat selection and growth performance of Mediterranean flathead grey mullet *Mugil cephalus* (Osteichthyes, Mugilidae). *Estuarine, Coastal and Shelf Science* **50**, 727-737.

Chong, V.C., and Sasekumar, A. (1981). Food and feeding habits of the white prawn *Penaeus merguiensis. Marine Ecology Progress Series* **5**, 185-191.

Clarke, K.R., and Warwick, R.M. (1994). 'Change in marine communities: an approach to statistical analysis and interpretation.' National Environmental Research Council, Plymouth Marine Laboratory, U.K. 144 pp.

Clewett, J.F., Clarkson, N.M., George, D.A., Ooi, S.H., Owens, D.T., Partridge, I.J., and Simpson, G.B. (2003). Rainman StreamFlow version 4.3: a comprehensive climate and streamflow analysis package on CD to assess seasonal forecasts and manage climate risk. Information series QI03040, Queensland Department of Primary Industries, Brisbane, Australia.

Coffey, M., Dehairs, F., Collette, O., Luther, G., Church, T., and Jickells, T. (1997). The behaviour of dissolved barium in estuaries. *Estuarine, Coastal and Shelf Science* **45**, 113-121.

Cohen, A., Daubechies, I., and Vial, P. (1993). Wavelets on the interval and fast wavelet transforms. *Applied and Computational Harmonic Analysis* **1**, 54-81.

Colbert, D., and McManus, J. (2005). Importance of seasonal variability and coastal processes on estuarine manganese and barium cycling in a Pacific Northwest estuary. *Continental Shelf Science* **25**, 1395-1414.

Connell, D.W., Bycroft, B.M., Miller, G.J., and Lather, P. (1981). Effects of a barrage on flushing and water quality in the Fitzroy River estuary, Queensland. *Australian Journal of Marine and Freshwater Research* **32**, 57-63.

Copeland, B.J. (1966). Effects of decreased river flow on estuarine ecology. *Journal - Water Pollution Control Federation* **38**, 1831-1839.

Currie, D.R., and Small, K.J. (2005). Macrobenthic community responses to long-term environmental change in an east Australian sub-tropical estuary. *Estuarine, Coastal and Shelf Science* **63**, 315-331.

da Silva, J. (1985). River runoff and shrimp abundance in a tropical coastal ecosystem - the example of the Sofala bank (central Mozambique). In 'Proceedings of the NATO advanced research workshop on The Role of Freshwater Outflow in Coastal Marine Ecosystems'. (Ed. S. Skreslet.) pp. 330-344. (Springer-Verlag: Heidelberg, Germany.)

Dall, W., Hill, B.J., Rothlisberg, P.C., and Staples, D.J. (1990). The Biology of the Penaeidae. *Advances in Marine Biology* **27**, 1-489.

Darnaude, A.M., Salen-Picard, C., Polunin, N.V.C., and Harmelin-Vivien, M.L. (2004). Trophodynamic linkage between river runoff and coastal fishery yield elucidated by stable isotope data in the Gulf of Lions (NW Mediterranean). *Oecologia* **138**, 325-332.

Daubechies, I. (1992). Ten lectures on wavelets. CBMS-NSF Regional conference series in applied mathematics. *Society for Industrial and Applied Mathematics* Philadelphia, PA. USA. 357 pp.

Davenport, J., and Wong, T.M. (1987). Responses of adult mud crabs (*Scylla serrata*) (Forskal) to salinity and low oxygen tension. *Comparative Biochemistry and Physiology* **86**A, 43-47.

Davis, R., and Hirji, R. (2003a). Environmental flows: Concepts and methods. Water Resources and Environment Technical Note C.1. The World Bank, Washington D.C. 27pp.

Davis, R., and Hirji, R. (2003b). Environmental flows: Case studies. Water Resources and Environment Technical Note C.2. The World Bank, Washington D.C. 28pp.

Davis, R., and Hirji, R. (2003c). Environmental flows: Flood flows. Water Resources and Environment Technical Note C.3. The World Bank, Washington D.C. 21pp.

Davis, T.L.O. (1982). Maturity and sexuality in barramundi, *Lates calcarifer* (Bloch), in the Northern Territory and South-eastern Gulf of Carpentaria. *Australian Journal of Marine and Freshwater Research* 33, 529-545.

Davis, T.L.O. (1985). Seasonal changes in gonad maturity, and abundance of larvae and early juveniles of barramundi, *Lates calcarifer* (Bloch), in Van Diemen Gulf and the Gulf of Carpentaria. *Australian Journal of Marine and Freshwater Research* 36, 177-190.

Davis, T.L.O. (1986). Migration patterns in barramundi, *Lates calcarifer* (Bloch), in Van Diemen Gulf, Australia, with estimates of fishing mortality in specific areas. *Fisheries Research* 4, 243-258.

Davis, T.L.O. (1987). Biology of wildstock, *Lates calcarifer* in northern Australia. In 'Management of Wild and Cultured Sea Bass/Barramundi (*Lates calcarifer*): proceedings of an international workshop held at Darwin, N. T. Australia, 24-30 September 1986'. (Eds J.W. Copland and D.L. Grey.) pp. 22-29. (Australian Centre for International Agricultural Research: Canberra.)

Davis, T.L.O., and Kirkwood, G.P. (1984). Age and growth studies on barramundi, *Lates calcarifer* (Bloch), in Northern Australia. *Australian Journal of Marine and Freshwater Research* 35, 673-689.

De Graaf, G. (2003). The flood pulse and growth of floodplain fish in Bangladesh. *Fisheries Management and Ecology* 10, 241-247.

DiCenzo, V.J., and Duval, M.C. (2002). Importance of reservoir inflow in determining white bass year-class strength in three Virginia reservoirs. *North American Journal of Fisheries Management* 22, 620-626.

Dichmont, C.M., Haddon, M., Yeomans, K., and Kelly, K. (1999). Proceedings of the South-East Queensland Stock Assessment Review Workshop: Southern Fisheries Centre, Deception Bay, Queensland, 16-28 August 1998. Conference and workshop series QC99003, Queensland Department of Primary Industries, Brisbane, Australia. 179 pp.

Dorval, E., Jones, C.M., and Hannigan, R. (2005).Chemistry of surface waters: distinguishing fine-scale differences in sea grass habitats of Chesapeake Bay. *Limnology and Oceanography* 50, 1073-1083.

Dredge, M.C.L. (1985). Importance of estuarine overwintering in the life cycle of the banana prawn, *Penaeus merguiensis*. In 'Second Australian National Prawn Seminar, Kooralbyn, Queensland, 22-26 October 1984'. (Eds P.C. Rothlisberg, B.J. Hill and D.J. Staples.) pp. 115-124.

Drinkwater, K.F. (1986). On the role of freshwater outflow on coastal marine ecosystems - a workshop summary. In 'Proceedings of the NATO advanced research workshop on The Role of Freshwater Outflow in Coastal Marine Ecosystems'. (Ed. S. Skreslet.) pp. 429-438. (Springer-Verlag: Heidelberg, Germany.)

Drinkwater, K.F., and Frank, K.T. (1994). Effects of river regulation and diversion on marine fish and invertebrates. *Aquatic Conservation: Freshwater and Marine Ecosystems* 4, 135-151.

Drinkwater, K.F., and Myers, R.A. (1987). Testing predictions of marine fish and shellfish landings from environmental variables. *Canadian Journal of Fisheries and Aquatic Sciences* 44, 1568-1573.

Dunstan, D.J. (1959). The barramundi, *Lates calcarifer* (Bloch) in Queensland waters. Division of Fisheries and Oceanography Technical Paper Number 5, Commonwealth Scientific and Industrial Research Organization, Melbourne, Australia. 22pp.

Dyson, M., Bergkamp, G., and Scanlon, J. (Eds) (2003). Flow: The essentials of environmental flows. (IUCN: Gland, Switzerland and Cambridge, United Kingdom.). 118pp.

Eggins, S.M., and Shelley, J.M.G. (2002). Compositional heterogeneity in NIST SRM 610-617 glasses. *Geostandards Newsletter: The Journal of Geostandards and Geoanalysis* **26**, 269-286.

Eggins, S.M., Kinsley, L.P.J., and Shelley, J.M.G. (1998). Deposition and element fractionation processes during atmospheric pressure laser sampling for analysis by ICP-MS. *Applied Surface Science* **127-129**, 278-286.

Elsdon, T.S., and Gillanders, B.M. (2005). Alternative life-history patterns of estuarine fish: barium in otoliths elucidates freshwater residency. *Canadian Journal of Fisheries and Aquatic Sciences* **62**, 1143-1152.

Evans, C.R., and Opnai, L.J. (1995). Fisheries ecology of the white banana prawn *Penaeus merguiensis* in the Gulf of Papua: estimates of sustainable yields and observations on trends in abundance. In 'South Pacific Commission and Forum Fisheries Agency Workshop on the Management of South Pacific Inshore Fisheries: Manuscript Collection of Country Statements and Background Papers'. Integrated Coastal Fisheries Management Project Technical Document Number 11, Volume 1, Background Paper Number 24. pp. 557-607. (South Pacific Commission: Nouméa.)

Evans, C.R., Opnai, L.J., and Kare, B.D. (1997). Fishery ecology and oceanography of the prawn *Penaeus merguiensis* (de Man) in the Gulf of Papua: estimation of maximum sustainable yield and modelling of yield, effort and rainfall. *Marine and Freshwater Research* **48**, 219-228.

Eyre, B. (1997). Water quality changes in an episodically flushed sub-tropical Australian estuary: a 50 year perspective. *Marine Chemistry* **59**, 177-187.

Eyre, B., and Twigg, C. (1997). Nutrient behaviour during post-flood recovery of the Richmond River Estuary Northern NSW, Australia. *Estuarine, Coastal and Shelf Science* **44**, 311-326.

Fishbase (2004). http://www.fishbase.org

Forbes, A.T., and Hay, D.G. (1988). Effects of a major cyclone on the abundance and larval recruitment of the portunid crab *Scylla serrata* (Forskal) in the St Lucia Estuary, Natal, South Africa. *South African Journal of Marine Science* **7**, 219-225.

Funicelli, N.A. (1984). Assessing and managing effects of reduced freshwater inflow to two Texas estuaries. In 'The Estuary as a Filter'. (Ed. V.S. Kennedy.) pp. 435-446. (Academic Press: Orlando.)

Galindo-Bect, M.S., Glenn, E.P., Page, H.M., Fitzsimmons, K., Galindo-Bect, L.A., Hernandez-Ayon, J.M., Petty, R.L., Garcia-Hernandez, J., and Moore, D. (2000). Penaeid shrimp landings in the upper Gulf of California in relation to Colorado River freshwater discharge. *Fishery Bulletin* **98**, 222-225.

Gammelsrød, T. (1992). Variation in shrimp abundance on the Sofala Bank, Mozambique, and its relation to the Zambezi River runoff. *Estuarine, Coastal and Shelf Science* **35**, 91-103.

Garrett, R.N. (1987). Reproduction in Queensland barramundi (*Lates calcarifer*). In 'Management of Wild and Cultured Sea Bass/Barramundi (*Lates calcarifer*): proceedings of an international workshop held at Darwin, N. T. Australia, 24-30 September 1986'. (Eds J.W. Copland and D.L. Grey.) pp. 38-43. (Australian Centre for International Agricultural Research: Canberra.)

Garrett, R.N. (1997). Biology and harvest of tropical fishes in the Queensland Gulf of Carpentaria gillnet fishery. Information series QI98018, Queensland Department of Primary Industries, Brisbane, Australia. 119 pp.

Gartside, D.F., Harrison, B., and Ryan, B.L. (1999). An evaluation of the use of fishing club records in the management of marine recreational fisheries. *Fisheries Research* **41**, 47-61.

GenStat. (2005). GenStat for Windows, Release 8.1, 8th Edn. (VSN International Ltd.: Oxford.)

Gillanders, B.M. (2005). Using elemental chemistry of fish otoliths to determine connectivity between estuarine and coastal habitats. *Estuarine, Coastal and Shelf Science* **64**, 47-57.

Gillanders, B.M., and Kingsford, M.J. (2002). Impact of changes in flow of freshwater on estuarine and open coastal habitats and the associated organisms. *Oceanography and Marine Biology: an Annual Review* 2002 **40**, 233-309.

Gippel, C. J. (2002). Workshop on environmental water requirements for Australian estuaries. Outcomes and further directions. Environment Australia, Canberra.

Glaister, J. P. (1978). The impact of river discharge on distribution and production of the school prawn *Metapenaeus macleayi* (Haswell) (Crustacea: Penaeidae) in the Clarence River region, northern New South Wales. *Australian Journal of Marine and Freshwater Research* **29**, 311-323.

Grey, D.L. (1987). An overview of *Lates calcarifer* in Australia and Asia. In 'Management of Wild and Cultured Sea Bass/Barramundi (*Lates calcarifer*): proceedings of an international workshop held at Darwin, N. T. Australia, 24-30 September 1986'. (Eds J.W. Copland and D.L. Grey.) pp. 15-21. (Australian Centre for International Agricultural Research: Canberra.)

Grey, D.L., Dall, W., and Baker, A. (1983). 'A guide to the Australian penaeid prawns.' (Northern Territory Government Printing Office.)

Griffin, R.K. (1985). The importance of mangrove/coastal wetland to three commercial fisheries in the Northern Territory; particularly for barramundi (*Lates calcarifer*). In 'Proceedings of Conference on Coasts and Tidal Wetlands of the Australian Monsoon region'. (Eds K.N. Bardsley, J.D.S. Davie and C.D. Woodroffe.) pp.277-284. (Australian National University: Darwin, Australia.)

Griffin, R.K. (1987). Barramundi (*Lates calcarifer*) research in the Northern Territory, Australia. In 'Management of Wild and Cultured Sea Bass/Barramundi (*Lates calcarifer*): proceedings of an international workshop held at Darwin, N. T. Australia, 24-30 September 1986'. (Eds J.W. Copland and D.L. Grey.) pp. 87-91. (Australian Centre for International Agricultural Research: Canberra.)

Griffin, R.K. (1988). A comparison of exploited and unexploited seabass *Lates calcarifer* populations in two rivers in the Northern Territory, Australia. *Asian Fisheries Science* **1**, 107-115.

Griffin, R.K. (1994). The barramundi population of Yellow Waters Billabong, Kakadu National Park. Fishery Report Number 33, Northern Territory Department of Primary Industry and Fisheries, Darwin, Australia. 9 pp.

Griffin, R.K., and Kelly, M. (2001). Fishery Assessment Report, Northern Territory Barramundi Fishery - 1999. Summary of Assessment Information. Fishery Report Number 56, Northern Territory Department of Primary Industry and Fisheries, Darwin, Australia. 15 pp.

Griffin, R.K., and Walters, C.J. (1999). Recreational fishery response to rebuilding and reallocation of the barramundi (*Lates calcarifer*) stocks in Australia's Northern Territory. In 'Evaluating the Benefits of Recreational Fisheries'. (Ed. T.J. Pitcher.) pp. 100-104. Fisheries Centre, University of British Columbia, Canada.

Gunter, G., and Hildebrand, H.H. (1954). The relation of total rainfall of the state and catch of the marine shrimp (*Penaeus setiferus*) in Texas waters. *Bulletin of Marine Science of the Gulf and Caribbean* **4**, 95-103.

Halliday, I.A., Robins, J.B., Mayer, D.G., Staunton-Smith, J., and Sellin, M.J. (submitted). Effects of freshwater flow on the year-class strength of a non-diadromous estuarine finfish, *Polydactylus macrochir*, in a dry tropical estuary. *Marine and Freshwater Research.*

Hart, D.D., and Finelli, C.M. (1999). Physical-biological coupling in streams: the pervasive effects of flow on benthic organisms. *Annual Review of Ecology and Systematics* **30**, 363-395.

Haywood, M.D.E., and Staples, D.J. (1993). Field estimates of growth and mortality of juvenile banana prawns (*Penaeus merguiensis*). *Marine Biology* **116**, 407-416.

Hecht, T., and van der Lingen, C.D. (1992). Turbidity-induced changes in feeding strategies of fish in estuaries. *South African Journal of Zoology* **27**, 95-107.

Helle, K., Bogstad, B., Marshall, C.T., Michalsen, K., Ottersen, G., and Pennington, M. (2000). An evaluation of recruitment indices for Arcto-Norwegian cod (*Gadus morhua* L.). *Fisheries Research* **48**, 55-67.

Helmke, S.A., Gribble, N.A., and Gould, S.F. (1998). Status of Gulf of Carpentaria mud crab stocks: establishing a benchmark for Albatross Bay, Weipa. Project report QO98017, Queensland Department of Primary Industries, Cairns, Australia. 22pp.

Henry, G.W., and Lyle, J.M. (Eds) (2003).The national recreational and indigenous fishing survey. New South Wales Fisheries Final Report Series Number 48, New South Wales Fisheries, Cronulla, Australia. 188 pp.

Higgs, J.B. (1998). Experimental recreational catch estimates for Queensland residents: RFISH Technical Report 2, results from the 1997 diary round. Queensland Fisheries Management Authority, Brisbane, Australia. 55 pp.

Hilborn, R., and Walters, C.J. (1992). ‘Quantitative Fisheries Stock Assessment: Choice, Dynamics and Uncertainty.’ (Chapman and Hall: New York.)

Hill, B.J. (1974). Salinity and temperature tolerance of zoeae of the portunid crab *Scylla serrata. Marine Biology* **25**, 21-24.

Hill, B.J. (1975). Abundance, breeding and growth of the crab *Scylla serrata* in two South African estuaries. *Marine Biology* **32**, 119-126.

Hill, B.J. (1982). The Queensland mud crab fishery. Queensland Fisheries Information series FI8201, Queensland Department of Primary Industries, Brisbane, Australia. 54pp.

Hill, B.J. (1994). Offshore spawning by the portunid crab *Scylla serrata* (Crustacea: Decapoda). *Marine Biology* **120**, 379-384.

Hoel, P.G. (1976). ‘Elementary Statistics.’ 4th Edn. (John Wiley and Sons: New York.)

Hogan, A.E., and Vallance, T.D. (2005). Rapid assessment of fish biodiversity in southern Gulf of Carpentaria catchments. Project report QI04074, Queensland Department of Primary Industries and Fisheries, Walkamin, Australia. 148pp.

Hunt, J.H., Carroll, R.J., Chinchilli, V., and Frankenberg, D. (1980). Relationship between environmental factors and brown shrimp production in Pamlico Sound, North Carolina. Special Scientific Report Number 33, University of North Carolina, North Carolina, USA.

Hyland, S.J. (2002). An investigation of the impacts of ponded pastures on barramundi and other finfish populations in tropical coastal wetlands. FRDC Final Report Project Number 97/201, Queensland Department of Primary Industries, Rockhampton, Australia. 146 pp.

Kailola, P.J., Williams, M.J., Stewart, P.C., Reichelt, R.E., McNee, A., Grieve, C. (1993). ‘Australian Fisheries Resources.’ (Bureau of Resource Sciences: Canberra.)

Kalish, J.M. (1992). Formation of a stress-induced chemical check in fish otoliths. *Journal of Experimental Marine Biology and Ecology* **162**, 265-277.

Keenan, C.P. (1994). Recent evolution of population structure in Australian barramundi, *Lates calcarifer* (Bloch): an example of isolation by distance in one dimension. *Australian Journal of Marine and Freshwater Research* **45**, 1123-1148.

Keitt, T.H. (2000). Spectral representation of neutral landscapes. *Landscape Ecology* **15**, 479-494.

Keitt, T.H., and Urban, D.L. (2005). Scale-specific inference using wavelets. *Ecology* **86**, 2497-2504.

Kennedy, B.P., Folt, C.L., Blum, J.D., and Chamberlain, C.P. (1997). Natural isotopes markers in salmon. *Nature (London)* **387**, 766-767.

Kimmerer, W.J. (2002a). Effects of freshwater flow on abundance of estuarine organisms: physical effects or trophic linkages? *Marine Ecology Progress Series* **243**, 39-55.

Kimmerer, W.J. (2002b). Physical, biological and management responses to variable freshwater flow into the San Francisco Estuary. *Estuaries* **25**, 1275-1290.

Kimmerer, W.J., and Schubel, J.R. (1994). Managing freshwater flows into San Francisco Bay using a salinity standard: results of a workshop. In 'Changes in fluxes in estuaries: implications from science to management. Proceedings of ECSA22/ERF Symposium'. (Eds K.R. Dyer and R.J.Orth.) pp. 411-416. (Olsen and Olsen: Fredensborg.)

Kimmerer, W.J., Cowan, J.H. Jr., Miller, L.W., and Rose, K.A. (2001). Analysis of an estuarine striped bass population: effects of environmental conditions during early life. *Estuaries* **24**, 557-575.

Kowarsky, J., and Ross, A.H. (1981). Fish movement upstream through a Central Queensland (Fitzroy River) coastal fishway. *Australian Journal of Marine and Freshwater Research* **32**, 93-109.

Le Reste, L., Feno, L., and Ramueloson, A. (1976). Information on the biology and ecology of the crab *Scylla serrata* (Forskal) in Madagascar. Oceanography and Biology Report, ORSTROM Paris, France.

Livingston, R.J., Lewis, F.G., Woodsum, G.C., Niu, X.-F., Galperin, B., Huang, W., Christensen, J.D., Monaco, M.E., Battista, T.A., Klein, C.J., Howell IV, R.L., and Ray, G.L. (2000). Modelling oyster population response to variation in freshwater input. *Estuarine, Coastal and Shelf Science* **50**, 655-672.

Livingston, R.J., Niu, X., Lewis III, F.G., and Woodsum, G.C. (1997). Freshwater input to a Gulf estuary: long-term control of trophic organization. *Ecological Applications* **7**, 277-299.

Lloret, J., Lleonart, J., Solé, I., and Fromentin, J-M. (2001). Fluctuations of landings and environmental conditions in the north-western Mediterranean Sea. *Fisheries Oceanography* **10**, 33-50.

Loneragan, N.R., and Bunn, S.E. (1999). River flows and estuarine ecosystems: implications for coastal fisheries from a review and a case study of the Logan River, southeast Queensland. *Australian Journal of Ecology* **24**, 431-440.

Loneragan, N.R., Bunn, S.E., and Kellaway, D.M. (1997). Are mangroves and seagrasses sources of organic carbon for penaeid prawns in a tropical Australian estuary? A multiple stable-isotope study. *Marine Biology* **130**, 289-300.

Long, P.E., and McKinnon, S.G. (2002). Habitats and fisheries resources of the Fitzroy River estuary (Central Queensland). Information series QI02104, Queensland Department of Primary Industries, Brisbane, Australia. 104pp.

Lucas, C., Kirkwood, G., and Somers, I. (1979). An assessment of the stocks of the banana prawn *Penaeus merguiensis* in the Gulf of Carpentaria. *Australian Journal of Marine and Freshwater Research* **30**, 639-652.

Maceina, M.J. (1997). Simple application of using residuals from catch-curve regressions to assess year-class strength in fish. *Fisheries Research* **32**, 115-121.

Maneewong, S. (1987). Induction of spawning of sea bass (*Lates calcarifer*) in Thailand. In 'Management of Wild and Cultured Sea Bass/Barramundi (*Lates calcarifer*): proceedings of an international workshop held at Darwin, N. T., Australia, 24-30 September 1986'. (Eds J.W. Copland and D.L. Grey.) pp. 116-119. (Australian Centre for International Agricultural Research: Canberra.)

Mapstone, B.D., McKinlay, J.P., and Davies, C.R. (1996). A description of commercial reef line fishery logbook data held by the Queensland Fisheries Management Authority. Report to the Queensland Fisheries Management Authority from the Cooperative Research Centre for the Ecologically Sustainable Development of the Great Barrier Reef and the Department of Tropical Environment Studies and Geography, James Cook University, Townsville, Australia. (Queensland Government Press: Brisbane.)

Margvelashvili, N., Robson, B., Sakov, P., Webster, I.T., Parslow, J., Herzfeld, M., and Andrewartha, J. (2003). Numerical modelling of hydrodynamics, sediment transport and biogeochemistry in the Fitzroy Estuary. Technical Report No. 9, Cooperative Research Centre for Coastal Zone, Estuary and Waterway Management, Indooroopilly, Queensland, Australia.

Martin, G.B., and Thorrold, S.R. (2005). Temperature and salinity effects on magnesium, manganese, and barium incorporation in otoliths of larval and early juvenile spot *Leiostomus xanthurus*. *Marine Ecology Progress Series* **293**, 223-232.

Maunder, M.N., and Watters, G.M. (2003). A general framework for integrating environmental time series into stock assessment models: model description, simulation testing and example. *Fishery Bulletin* **101**, 89-99.

Mayer, D., Roy, D., Robins, J., Halliday, I., and Sellin, M. (2005). Modelling zero-inflated fish counts in estuaries - a comparison of alternate statistical distributions. In 'MODSIM 2005 International Congress on Modelling and Simulation'. Modelling and Simulation Society of Australia and New Zealand, December 2005. (Eds A. Zerger and R.M.Argent.) pp. 2581-2587.

McCullagh, P., and Nelder, J.A. (1989). 'Generalized Linear Models.' 2nd Edn. (Chapman and Hall: London.)

McGlennon, D., Jones, G.K., Baker, J., Jackson, W.B., and Kinloch, M.A. (2000). Ageing, catch-at-age and relative year-class strength for snapper (*Pagrus auratus*) in northern Spencer Gulf, South Australia. *Marine and Freshwater Research* **51**, 669-677.

McKinnon, S.G., Lupton, C.J., and Long, P.E. (1995). A fisheries resource assessment of the Calliope River system in Central Queensland 1994. Information series QI95001, Queensland Department of Primary Industries, Brisbane, Australia. 153pp.

Meager, J.J., Vance, D.J., Loneragan, N.R., and Williamson, I. (2003). Seasonal variation and environmental influences on juvenile banana prawn (*Penaeus merguiensis*) abundance in a subtropical estuary (Logan River) of eastern Australia. *Estuarine, Coastal and Shelf Science* **57**, 569-576.

Meeter, D.A., Livingston, R.J., and Woodsum, G.C. (1979). Long-term climatological cycles and population changes in a river-dominated estuarine system. In 'Ecological processes in coastal and marine systems. Marine Science 10'. (Ed. R. Livingston.) pp. 315-338. (Plenum Press: London.)

Mills, C.A., and Mann, R.H.K. (1985). Environmentally-induced fluctuations in year-class strength and their implications for management. *Journal of Fish Biology* **27**, 209-226.

Milton, D.A., and Chenery, S.R. (2001). Sources and uptake of trace metals in otoliths of juvenile barramundi (*Lates calcarifer*). *Journal of Experimental Marine Biology and Ecology* **264**, 47-65.

Milton, D.A., and Chenery, S.R. (2003). Movement patterns of the tropical shad hilsa (*Tenualosa ilisha*) inferred from transects of ^{87}Sr/ ^{86}Sr isotope ratios in their otoliths. *Canadian Journal of Fisheries and Aquatic Sciences* **60**, 1376-1385.

Milton, D.A., and Chenery, S.R. (2005). Movement patterns of barramundi *Lates calcarifer*, inferred from ^{87}Sr/^{86}Sr and Sr/Ca ratios in otoliths, indicate non-participation in spawning. *Marine Ecology Progress Series* **301**, 279-291.

Milton, D., Halliday, I., Staunton-Smith, J., Sellin, M., Marsh, R., Smith, D., Norman, M., and Woodhead, J. (submitted). Otolith chemistry of barramundi *Lates calcarifer* can provide insight about the role of freshwater flows in maintaining estuarine populations of regulated rivers. *Estuarine, Coastal and Shelf Science.*

Moore, R. (1980). Reproduction and migration in *Lates calcarifer* (Bloch). PhD Thesis, University of London, UK. 213 pp.

Moore, R. (1982). Spawning and early life history of barramundi, *Lates calcarifer* (Bloch), in Papua New Guinea. *Australian Journal of Marine and Freshwater Research* **33**, 647-661.

Moriarty, D.J.W., and Barclay, M.C. (1981). Carbon and nitrogen content of food and the assimilation efficiencies of penaeid prawns in the Gulf of Carpentaria. *Australian Journal of Marine and Freshwater Research* **32**, 245-251.

Morison, A.K., Coutin, P.C., and Robertson, S.G. (1998). Age determination of black bream, *Acanthopagrus butcheri* (Sparidae), from the Gippsland Lakes of south-eastern Australia indicates slow growth and episodic recruitment. *Marine and Freshwater Research* **49**, 491-498.

Morita, K., and Morita, S.H. (2002). Rule of age and size at maturity: individual variation in the maturation history of resident white-spotted charr. *Journal of Fish Biology* **61**, 1230-1238.

Myers, R.A. (1998). When do environment-recruitment correlations work? *Reviews in Fish Biology and Fisheries* **8**, 285-305.

Newell, R.I.E., Marshall, N., Sasekumar, A., and Chong, V.C. (1995). Relative importance of benthic microalgae, phytoplankton, and mangroves as sources of nutrition for penaeid prawns and other coastal invertebrates from Malaysia. *Marine Biology* **123**, 595-606.

Nisa, Z., and Ahmed, M. (2000). Hatching and larval survival of important Penaeid shrimps of Pakistan in different salinities. *Pakistan Journal of Zoology* **32**, 139-143.

North, E.W., and Houde, E.D. (2003). Linking ETM physics, zooplankton prey, and fish early-life histories to striped bass *Morone saxatilis* and white perch *M. americana* recruitment. *Marine Ecology Progress Series* **260**, 219-236.

Olden, J.D., and Poff, N.L. (2003). Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. *River Research and Applications* **19**, 101-121.

Pender, P.J., and Griffin, R.K. (1996). Habitat history of barramundi *Lates calcarifer* in a north Australian river system based on barium and strontium levels in scales. *Transactions of the American Fisheries Society* **125**, 679-689.

Perry, I.R., Boutillier, J.A., and Foreman, M.G.G. (2000). Environmental influences on the availability of smooth pink shrimp, *Pandalus jordani*, to commerical fishing gear off Vancouver Island, Canada. *Fisheries Oceanography* **9**, 50-61.

Peters, J.C. (1982). Effects of river and streamflow alteration on fishery resources. *Fisheries* **7**, 20-22.

Platten, J.R. (2004). Historical trends in recreational fishing catches in the Gladstone region. An indicator of the state of Port Curtis? State of Port Curtis Conference, 23-25th October 2002, Central Queensland University, Gladstone, Queensland, Australia.

Platten, J.R. (2005). Historical trends in recreational fishing catches in the Gladstone region. Coastal CRC report, Cooperative Research Centre for Coastal Zone, Estuary and Waterway Management, Indooroopilly, Queensland, Australia.

Potter, I.C., Bird, D.J., Claridge, P.N., Clarke, K.R., Hyndes, G.A., and Newton, L.C. (2001). Fish fauna of the Severn Estuary. Are there long-term changes in abundance and species composition and are the recruitment patterns of the main marine species correlated? *Journal of Experimental Marine Biology and Ecology* 258, 15-37.

Powell, G.L. (1977). Estuarine fishery dynamics and freshwater inflow fluctuations in the San Antonio Bay systems, Texas. *Proceedings of the Annual Conference Southeastern Association of Fish and Wildlife Agencies* 31, 498-504.

Powell, G.L., and Malstaff, G. (1994). Inflow effects on fisheries harvest. In: ‘Freshwater inflows to Texas bays and estuaries: ecological relationships and methods for determination of needs’. (Ed. W.L. Longley.). pp 192-203. Texas Water Development Board and Texas Parks and Wildlife Department, Austin, Texas, USA.

Powell, G.L., Matsumoto, J., and Brock, D.A. (2002). Methods for determining minimum freshwater inflow needs of Texas Bays and estuaries. *Estuaries* 25, 1262-1274.

Primavera, J.H. (1996). Stable carbon and nitrogen isotope ratios of penaeid juveniles and primary producers in a riverine mangrove in Guimaras, Phillippines. *Bulletin of Marine Science* 58, 675-683.

Pyper, B.J., and Peterman, R.M. (1998). Comparison of methods to account for autocorrelation in correlation analyses of fish data. *Canadian Journal of Fisheries and Aquatic Sciences* 55, 2127-2140.

Quinn, N.J., and Kojis, B.L. (1987). Reproductive biology of *Scylla sp.* (Crustacea: Portunidae) from the Labu Estuary in Papua New Guinea. *Bulletin of Marine Science* 41, 234-241.

Quiñones, R.A., and Montes, R.M. (2001). Relationship between freshwater input to the coastal zone and the historical landings of the benthic/demersal fish *Eleginops maclovinus* in central-south Chile. *Fisheries Oceanography* 10, 311-328.

Ralston, A., and Rabinowitz, P. (2001). ‘A first course in numerical analysis.’ (Courier Dover Publications.)

Ratkowsky, D.A. (1986). Statistical properties of alternative parameterizations of the von Bertalanffy growth curve. *Canadian Journal of Fisheries and Aquatic Sciences* 43, 742-747.

Reynolds, L.F., and Moore, R. (1982). Growth rates of barramundi, *Lates calcarifer* (Bloch), in Papua New Guinea. *Australian Journal of Marine and Freshwater Research* 33, 663-670.

Ritter, C., Brock, D., Wentzel, M., Austin, B., Matsumoto, J., Malstaff, G., Solis, R., Jia, Z., and Powell, G. (2000). Values and constraints for the TXEMP model used in the freshwater inflow analysis of the Mission-Aransas Estuary. Texas Water Development Board, Austin, Texas, USA.

Robins, J., Mayer, D., Staunton-Smith, J., Halliday, I., Sawynok, B., and Sellin, M. (2006). Variable growth rates of the tropical estuarine fish barramundi *Lates calcarifer* (Bloch) under different freshwater flow conditions. *Journal of Fish Biology* 69, 379-391.

Robins, J.B., Halliday, I.A., Staunton-Smith, J., Mayer, D.G., and Sellin, M.J. (2005). Freshwater-flow requirements of estuarine fisheries in tropical Australia: a review of the state of knowledge and application of a suggested approach. *Marine and Freshwater Research* 56, 343-360.

Roelofs, A. (2003). Ecological assessment of the Gulf of Carpentaria inshore finfish fishery: a report to Environment Australia on the sustainable management of a multi-species tropical gillnet fishery. Information series QI03045, Queensland Department of Primary Industries, Brisbane, Australia. 184 pp.

Ruello, N.V. (1973). The influence of rainfall on the distribution and abundance of the school prawn *Metapenaeus macleayi* in the Hunter River region (Australia). *Marine Biology* **23**, 221-228.

Russell, D.J. (1987). Review of juvenile barramundi (*Lates calcarifer*) wildstocks in Australia. In 'Management of Wild and Cultured Sea Bass/Barramundi (*Lates calcarifer*): proceedings of an international workshop held at Darwin, N. T. Australia, 24-30 September 1986'. (Eds J.W. Copland and D.L. Grey.) pp. 44-49. (Australian Centre for International Agricultural Research: Canberra.)

Russell, D.J. (1990). Reproduction, migration and growth in *Lates calcarifer*. M.App. Sc. Thesis, Queensland University of Technology, Brisbane, Australia.

Russell, D.J., and Garrett, R.N. (1983). Use by juvenile barramundi, *Lates calcarifer* (Bloch), and other fishes of temporary supralittoral habitats in a tropical estuary in northern Australia. *Australian Journal of Marine and Freshwater Research* **34**, 805-811.

Russell, D.J., and Garrett, R.N. (1985). Early life history of barramundi, *Lates calcarifer* (Bloch), in north-eastern Queensland. *Australian Journal of Marine and Freshwater Research* **36**, 191-201.

Sadovy, Y., and Severin, K.P. (1994). Elemental patterns in red hind (*Epinephelus guttatus*) otoliths from Bermuda and Puerto Rico reflect growth rate, not temperature. *Canadian Journal of Fisheries and Aquatic Sciences* **51**, 133-141.

Saenger, P. (1996). Ecology of mangroves of Port Curtis: a regional biogeography. In 'Mangroves - a resource under threat? An issue for the central Queensland coast'. (Eds D. Hopley and L. Warner.) pp. 23-38.

Sainsbury, K.J. (1980). Effect of individual variability on the von Bertalanffy growth equation. *Canadian Journal of Fisheries and Aquatic Sciences* **37**, 241-247.

Saldanha, C.M., and Achuthankutty, C.T. (2000). Growth of hatchery raised banana shrimp *Penaeus merguiensis* (de Man) (Crustacea: Decapoda) juveniles under different salinity. *Indian Journal of Marine Sciences* **29**, 179-180.

Salen-Picard, C., Darnaude, A.M., Arlhac, D., and Harmelin-Vivien, M.L. (2002). Fluctuations of macrobenthic populations: a link between climate-driven river run-off and sole fishery yields in the Gulf of Lions. *Oecologia* **133**, 380-388.

Salini, J., and Shaklee, J.B. (1988). Genetic structure of barramundi (*Lates calcarifer*) stocks from Northern Australia. *Australian Journal of Marine and Freshwater Research* **39**, 317-329.

Salini, J.P., Brewer, D.T., and Blaber, S.J.M. (1998). Dietary Studies on the Predatory Fishes of the Norman River Estuary, with particular reference to Penaeid Prawns. *Estuarine, Coastal and Shelf Science* **46**, 837-847.

Sawynok, B. (1998). Fitzroy River: effects of freshwater flows on fish: impact on barramundi recruitment, movement and growth. National Fishcare Report Project 97/003753, Infofish Services, Rockhampton, Queensland, Australia. 59pp.

Schöne, B.R., Flessa, K.W., Dettman, D.L., and Goodwin, D.H. (2003). Upstream dams and downstream clams: growth rates of bivalve mollusks unveil impact of river management on estuarine ecosystems (Colorado River Delta, Mexico). *Estuarine, Coastal and Shelf Science* **58**, 715-726.

Secor, D.H., and Rooker, J.R. (2000). Is otolith strontium a useful scalar of life cycles in estuarine fishes? *Fisheries Research* **46**, 359-371.

Shaklee, J.B., and Salini, J.P. (1985). Genetic variation and population subdivision in Australian barramudi, *Lates calcarifer* (Bloch). *Australian Journal of Marine and Freshwater Research* **36**, 203-218.

Shaklee, J.B., Salini, J., and Garrett, R.N. (1993). Electrophoretic characterization of multiple genetic stocks of barramundi perch in Queensland, Australia. *Transactions of the American Fisheries Society* **122**, 685-701.

Sharp, G.D. (1995). It's about time: new beginnings and old good ideas in fisheries science. *Fisheries Oceanography* **4**, 324-341.

Shepherd, J.G., Pope, J.G., and Cousens, R.D. (1984). Variations in fish stocks and hypotheses concerning their links with climate. *Rapports et Procès-Verbaux des Réunions du Conseil International pour l'Exploration de la Mer* **185**, 255-267.

Skreslet, S. (1997). A conceptual model of the trophodynamical response to river discharge in a large marine ecosystem. *Journal of Marine Systems* **12**, 187-198.

Sobrino, I., Silva, L., Bellido, J.M., and Ramos, F. (2002). Rainfall, river discharges and sea temperature as factors affecting abundance of two coastal benthic cephalopod species in the Gulf of Cádiz (SW Spain). *Bulletin of Marine Science* **71**, 851-865.

Somers, I.F. (1988). On a seasonally oscillating growth function. *Fishbyte* **6**, 8-11.

Staples, D.J. (1980). Ecology of juvenile and adolescent banana prawns, *Penaeus merguiensis*, in a mangrove estuary and adjacent off-shore area of the Gulf of Carpentaria. II. Emigration, population structure and growth of juveniles. *Australian Journal of Marine and Freshwater Research* **31**, 653-665.

Staples, D.J., and Heales, D.S. (1991). Temperature and salinity optima for growth and survival of juvenile banana prawns *Penaeus merguiensis*. *Journal of Experimental Marine Biology and Ecology* **154**, 251-274.

Staples, D.J., and Vance, D.J. (1986). Emigration of juvenile banana prawns *Penaeus merguiensis* from a mangrove estuary and recruitment to offshore areas in the wet-dry tropics of the Gulf of Carpentaria, Australia. *Marine Ecology Progress Series* **27**, 239-252.

Staples, D.J., and Vance, D.J. (1987). Comparative recruitment of the banana prawn, *Penaeus merguiensis*, in five estuaries of the south-eastern Gulf of Carpentaria, Australia. *Australian Journal of Marine and Freshwater Research* **38**, 29-45.

Staples, D.J., Vance, D.J., and Heales, D.S. (1985). Habitat requirements of juvenile penaeid prawns and their relationship to offshore fisheries. In 'Second Australian National Prawn Seminar, Kooralbyn, Queensland, 22-26 October 1984'. (Eds P.C. Rothlisberg, B.J. Hill and D.J. Staples.) pp. 47-54.

StatSoft. (2005). 'STATISTICA.' (StatSoft Incorporated: Tulsa, USA.)

Staunton-Smith, J., Robins, J.B., Mayer, D.G., Sellin, M.J., and Halliday, I.A. (2004). Does the quantity and timing of freshwater flowing into a dry tropical estuary affect year-class strength of barramundi (*Lates calcarifer*)? *Marine and Freshwater Research* **55**, 787-797.

Stephenson, W., and Campbell, B. (1960). The Australian Portunids (Crustacea: Portunidae) IV. Remaining Genera. *Australian Journal of Marine and Freshwater Research* **11**, 73-122.

Stergiou, K.I., and Christou, E.D. (1996). Modelling and forecasting annual fisheries catches: comparison of regression, univariate and multivariate time series methods. *Fisheries Research* **25(2)**, 105-138.

Strang, G., and Nguyen, T. (1996). 'Wavelets and Filter Banks.' (Wellesley Cambridge Press.)

Stuart, I.G. (1997). Assessment of a modified vertical-slot fishway, Fitzroy River, Queensland. Project report QO97023, Queensland Department of Primary Industries, Bundaberg, Australia. 82 pp.

Stuart, I.G., and Mallen-Cooper, M. (1999). An assessment of the effectiveness of a vertical-slot fishway for non-salmonid fish at a tidal barrier on a large tropical/subtropical river. *Regulated Rivers: Research & Management* **15**, 575-590.

Stuart, I.G., and McKillup, S. (2002). The use of sectioned otoliths to age barramundi (*Lates calcarifer*) (Bloch, 1790) [Centropomidae]. *Hydrobiologia* **479**, 231-236.

Sutcliffe, W.H. Jr. (1973). Correlations between seasonal river discharge and local landings of American lobster (*Homarus americanus*) and Altantic halibut (*Hippoglossus hippoglossus*) in the Gulf of St. Lawrence. *Journal of the Fisheries Research Board of Canada* **30**, 856-859.

Sutcliffe, W.H. Jr., Drinkwater, K., and Muir, B.S. (1977). Correlations of fish catch and environmental factors in the Gulf of Maine. *Journal of the Fisheries Research Board of Canada* **34**, 19-30.

Sutela, T., Mutenia, A., and Salonen, E. (2002). Relationship between annual variation in reservoir conditions and year-class strength of peled (*Coregonus peled*) and whitefish (*C. lavaretus*). *Hydrobiologia* **485**, 213-221.

Tanimoto, M., Courtney, A.J., O'Neill, M.F., and Leigh, G.M. (2006). Stock assessment of the Queensland (Australia) east coast banana prawn (*Penaeus merguiensis*) fishery. Queensland Department of Primary Industries and Fisheries, Brisbane, Australia. 89 pps.

Tharme, R.E. (2003). A global perspective on environmental flow assessment: emerging trends in the development and application of environmental flow methodologies of rivers. *River Research and Applications* **19**, 397-441.

Thomson, J.M. (1963). Synopsis of the Biological Data on the Grey Mullet *Mugil cephalus* (Linnaeus). Division of Fisheries and Oceanography Fishery Synopsis Number 1, Commonwealth Scientific and Industrial Research Organization, Canberra, Australia.

Troup, A.J. (1965). The southern oscillation. *Quarterly Journal of Royal Meteorological Society* **91**, 490-506.

Turek, J.G., Goodger, T.E., Bigford, T.E., and Nichols, J.S. (1987). Influence of freshwater inflows on estuarine productivity. NOAA Technical Memorandum NMFS-F/NEC-46, Woods Hole, Massachusetts, USA. 26 pp.

Tyler, A.V. (1992). A context for recruitment correlations: why marine fisheries biologists should still look for them. *Fisheries Oceanography* **1**, 97-107.

Underwood, A.J. (1997). 'Experiments in ecology: their logical design and interpretation using analysis of variance.' (Cambridge University Press: Cambridge.)

Vance, D.J., and Staples, D.J. (1992). Catchability and sampling of three species of juvenile penaeid prawns in the Embley River, Gulf of Carpentaria, Australia. *Marine Ecology Progress Series* **87**, 201-213.

Vance, D.J., Haywood, M.D.E., Heales, D.S., Kenyon, R.A., and Loneragan, N.R. (1998). Seasonal and annual variation in abundance of postlarval and juvenile banana prawns *Penaeus merguiensis* and environmental variation in two estuaries in tropical northeastern Australia: a six year study. *Marine Ecology Progress Series* **163**, 21-36.

Vance, D.J., Staples, D.J., and Kerr, J.D. (1985). Factors affecting year-to-year variation in the catch of banana prawns (*Penaeus merguiensis*) in the Gulf of Carpentaria, Australia. *Journal du Conseil International pour L'Exploration de la Mer* **42**, 83-97.

Vinod, K., Bhat, U.G., and Neelakantan, B. (1996). Effect of salinity on food intake, growth and conversion efficiencies of juvenile *Penaeus merguiensis* (de Man). *Environment and Ecology* **14**, 71-75.

Virgona, J., Deguara, K., Sullings, D., Halliday, I., and Kelly, K. (1998). Assessment of the stocks of sea mullet in New South Wales and Queensland waters. FRDC Final Report Project Number 94/024, New South Wales Fisheries Research Institute, Cronulla, Australia and Queensland Department of Primary Industries, Southern Fisheries Centre, Deception Bay, Australia. 104 pp.

Walker, J.S. (1999). 'A primer on wavelets and their scientific applications.' (Chapman Hall: New York.)

Walker, S., Sporic, M., and Coutin, P. (1998). Development of an environment-recruitment model for black bream: a case study for estuarine fisheries management. FRDC Final Report Project Number 96/102, Marine and Freshwater Resources Institute, Queenscliff, Victoria, Australia.

Walters, C., Buckworth, R., Gribble, N., Griffin, R., Johnson, G., Loneragan, N., and Ramm, D. (2001). Status and management options for the barramundi fishery. In 'Towards the sustainable use of Northern Territory fishery resources: review workshops led by Carl J. Walters'. (Ed. D.C. Ramm.) Appendix 1, Fishery Report Number 39, Northern Territory Department of Primary Industry and Fisheries, Darwin, Australia.

Walters, C.J., and Collie, J.S. (1988). Is research on environmental factors useful to fisheries management? *Canadian Journal of Fisheries and Aquatic Sciences* **45**, 1848-1854.

Wang, Y-G., and Haywood, M.D.E. (1999). Size-dependent natural mortality of juvenile banana prawns *Penaeus merguiensis* in the Gulf of Carpentaria. *Marine and Freshwater Research* **50**, 313-317.

Wang, Y-G., and Jackson, C.J. (2000). Growth curves with time-dependent explanatory variables. *Environmetrics* **11**, 597-605.

Wassenberg, T.J., and Hill, B.J. (1993). Diet and feeding behaviour of juvenile and adult banana prawns *Penaeus merguiensis* in the Gulf of Carpentaria, Australia. *Marine Ecology Progress Series* **94**, 287-295.

Webster, I.T., Ford, P.W., Robson, B., Margvelashvili, N., and Parslow, J. (2003). Conceptual models of the hydrodynamics, fine sediment dynamics, biogeochemistry and primary production in the Fitzroy Estuary. Technical Report No. 8, Cooperative Research Centre for Coastal Zone, Estuary and Waterway Management, Indooroopilly, Queensland, Australia.

Welch, D.J., Gribble, N.A., and Garrett, R.N. (2002). Assessment of the threadfin salmon fishery in Queensland - 2002. Information series QI02115, Queensland Department of Primary Industries, Cairns, Australia. 20 pp.

Whitfield, A.K. (2005). Fishes and freshwater in southern African estuaries - a review. *Aquatic Living Resources* **18**, 275-289.

Wilber, D.H. (1992). Associations between freshwater inflows and oyster productivity in Apalachicola Bay, Florida. *Estuarine, Coastal and Shelf Science* **35**, 179-190.

Wilkes, D.S. (1995). 'Statistical Methods in Atmospheric Science.' (Academic Press: San Diego.)

Williams, L. (2002). Queensland's fisheries resources. Current condition and recent trends 1988 - 2000. Information series QI02012, Queensland Department of Primary Industries, Brisbane, Australia. 180pp.

Williams, M.J., and Hill, B.J. (1982). Factors influencing pot catches and population estimates of the portunid crab *Scylla serrata*. *Marine Biology* **71**, 187-192.

Woodhead, J., Hergt, J., Shelley, M., Eggins, S., and Kemp, R. (2004). Zircon Hf-isotope analysis with an excimer laser, depth profiling, ablation of complex geometries, and concomitant age estimation. *Chemical Geology* **209**, 121-135.

Woodhead, J., Swearer, S., Hergt, J., and Maas, R. (2005). *In situ* Sr-isotope analysis of carbonates by LA-MC-ICP-MS: interference corrections, high spatial resolution and an example from otolith studies. *Journal of Analytical Atomic Spectrometry* **20**, 22-27.

Xiao, Y. (1999). General age- and time-dependent growth models for animals. *Fishery Bulletin* **97**, 690-701.

Xiao, Y. (2000). Use of the original von Bertalanffy growth model to describe the growth of barramundi, *Lates calcarifer* (Bloch). *Fishery Bulletin* **98**, 835-841.

Zimmerman, R.J., Minello, T.J., Baumer, T.J., and Castigione, M.C. (1990). Utilization of nursery habitats in San Antonio Bay in relation to annual salinity variation. NOAA Technical Report, Austin, Texas, USA. 50 pp.

Chapter 16. Appendix 1. Intellectual property

Intellectual property resulting from this study relates to information that was collected as part of the research project. The data have been summarised, analysed and interpreted to provide the Coastal Zone CRC and the Fisheries Research and Development Corporation with this Final Report. Published papers will allow access by industry and other interested persons to the summarised data.

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Chapter 18. Appendix 3. Data archives

Appendix for Chapter 3. Queensland fish board data

General Information

Fishing has occurred for a long time in Queensland, being used for subsistence by indigeneous and early European inhabitants. The first fishery to be developed after colonisation was the inshore fishery. The annual reports by harbours, which date back to the late 1800s comment that 'commercial fishermen disposed of their catch at the nearest population centres'. 'Fish caught in Moreton Bay about 1891 by the regular fishermen were brought up the Brisbane River to Breakfast Creek, fishermen from vessels working out of the Noosa River sent their fish to Gympie and Maryborough fishermen sent theirs to Gympie and Maryborough for distribution to the surrounding district. Some of the Morton Bay fisherman forwarded their catch by train to the Brisbane Markets from such convenient places as Sandgate, Pinkenba, Wynnum, Nambour, Stapylton, Coomera and Southport.' The Inspector of Fisheries James H. Stevens advocated the starting of a central fish market for Brisbane in 1901 as a means of controlling the size of fish being marketed. Regulations in force in 1902 had substituted 'length' for controlling fish size instead of 'weight'. Stevens advocated that all fish hawkers should be licensed for food safety reasons and to assist in the control of selling fish only caught by legal means. The Fisheries Act of 1904 provided for the marketing of fish and by 1907 the Brisbane Fish Market began operating.

Documentation of the scale and extent of fishing in Queensland is somewhat patchy. The main source of fishery catch data between 1930 and 1980 are the figures published in the annual reports of the fish board responsible for marketing and distributing fish in Queensland during this period. It is uncertain what proportion of the total Queensland fisheries landings these figures represent. Fisheries landings destined for interstate or international export were not required to pass through the fish board. Anecdotal reports suggest that a number of private companies handled fisheries landings independent of the fish board. However, there is some evidence to suggest that private handling of fisheries landings increased as the processing and marketing industry expanded with the introduction and growth the prawn trawlingindustry. There are numerous reports of black market sales of fisheries landings throughout Queensland, the timing and scale of which is difficult to quantify. Comparisons of Queensland Fish Board landings of with that reported by the Australian Bureau of Census & Statistics sheds little light on the proportion of total Queensland fisheries landings passing through the Queensland Fish Board, as the figures are very similar. This is not surprising as the Commonwealth Bureau of Census & Statistics derived fisheries statistics mainly from annual returns supplied by State fisheries authorities.

No effort data is available to compliment the landings data. From 1954 onwards, the trawling industry of the Queensland east coast and Gulf of Carpentaria was rapidly expanding. Anecdotal reports suggest that many individuals who previously net fished were drawn to the expanding trawl industry, resulting in unknown changes to fishing effort within net fisheries. The above factors impose limitations that should be considered in any analysis and interpretation of the Queensland Fish Board database (held in full electronic form by the Department of Primary Industries & Fisheries).

The Queensland Fish Board reports record 61 market 'categories'. These market categories have been assigned into the most likely current species or species group. It is difficult to confirm exactly which species comprised some of the market categories, but the interpretation has been checked against available Commonwealth and State reports that mention species names, as well as discussion with fishers and DPI&F staff. The species reported in the Queensland Fish Board Landings are listed in Table A.3.1.

Table A.3.1 Species occurring in the Queensland Fish Board landings data

Current CFISH name	QFB name	Other names	Scientific name
Barramundi	Barramundi	Giant Perch	*Lates calcarifer*
Blue swimmer crab	Crabs, sand, meat		*Portunus pelagicus*
Blue swimmer crab	Crabs, sand, bodies		*Portunus pelagicus*
Blue threadfin	Salmon	Cooktown salmon	*Eleutheronema tetradactylum*
Bream, mixed	Bream black	Pikey bream	*Acanthopagrus berda*
Bream, mixed	Bream		*Acanthopagrus australis*
Butterfish	Johnny dory	dory, john dory, old maid, scat	*Scatophagus multifasciatus*
Cod	Cod		*Epinephelus spp.*
Coral trout	Coral trout		*Plectropomus spp.*
Dart	Dart		*Trachinotus spp.*
Flathead, mixed	Flathead		*Platycephalus arenarius, inducus,*
Gar	Gar		*Hyporhamphus spp.; Arrhamphus sp.*
Grunter bream	Trumpeter	Javelin fish	*Pomadasys* sp.
Jew	Jew		
King threadfin	Salmon Burnett	Threadfin salmon	*Polydactylus sheridani*
King threadfin	King	Threadfin salmon, Burnett Salmon	*Polydactylus sheridani*
Leatherjacket	Leather jackets	Monacanthidae	
Lethrinid	Emperor		*Lethrinius spp.*
Lethrinid	Sweet lip		*Lethrinius spp.*
Longtom	Long toms	Needlefish	*Tylosurus spp.*
Mackerel, mixed	Mackerel		*Scomberomorus* sp.
Mackerel, mixed	Mackerel school		*Scomberomorus queenslandicus*
Mixed fish	Mixed		
Mud crab	Crabs, mud, meat		*Scylla serrata*
Mud crab	Crabs, mud, bodies		*Scylla serrata*
Mullet	Mullet		*Mugil cephalus, Liza* sp.
Oysters	Oysters, bottles		
Oysters	Oysters, bags		
Parrot	Parrot	Scarids	
Pearl perch	Pearl perch		*Glaucosma scapulare*
Prawns	Prawns mixed		*Penaeus* sp., *Metapenaeus* sp.
Ray	Ray		*Penaeus* sp., *Metapenaeus* sp.
Samson	Sampson	Amberjack, pool with Yellowtail	*Seriola hippos*
Sea perch	Nanygai	Saddletail, Fingermark	*Lutjanus spp.*
Sea pike	Pike	Barracuda	*Sphyraena spp.*
Sea pike	Barracouta	Barracuda	*Sphyraena spp.*
Shark	Shark		Carcharhinidae
Snapper	Squire		*Pagrus auratus*
Snapper	Snapper	Red bream	*Pagrus auratus*
Spanner crab	Crabs, spanners, bodies		*Ranina ranina*
Squid	Squid		
Sweetlip bream	Morwong	Haemulidae	*Diagramma* sp, *Plectorhinchus* sp
Tailor	Tailer		*Pomatomus saltatrix*
Whiting mixed	Whiting		*Sillago* sp.
Yellowtail kingfish	Yellow tail	Yellowtail Kingfish	*Seriola lalandi*

Monthly fisheries landings that passed through the Brisbane Fish Market from 1936 to 1945 are recorded by the annual reports, but do not indicate the point of origin of the landings. More location specific information is available from 1945, where the annual financial year landings from each depot are reported. Fisheries landings were recorded from 46 districts (=depots). The depot landings do not guarantee that landed fish was sourced from a particular area. For the purposes of analyses, we assume that the majority of the fish landed at a depot were caught in the nearby area.

Fisheries landings were recorded in pounds (lbs) until 1973/1974, and then in kilograms (kg) for most market species. All landings data were converted to kg. An additional conversion was required to convert fish fillets to an equivalent gilled and gutted weight (2 x fillet weight), sand crabs and spanner crab bodies (i.e. numbers) to kg (3 crabs per kg) and mud crab bodies (i.e.

numbers) to kg (1 crab per kg). Landings or crab meat, mud crab meat and sand crab meat were converted to a whole green weight using the following meat recovery rates: crab meat (unspecified) 30%, mud crabs 25% and sand crabs 35%. Dr Ian Brown supplied meat recovery figures from research work.

Barramundi

Barramundi is the consistent marketing name for *Lates calcarifer*. Few references are made to barramundi in the Queensland Fish Board annual reports. Dunstan (1959), in his review of barramundi in Queensland waters remarks that 'a large part of the barramundi catch in eastern Australia is made by part-time fishermen who generally do not market through the fish board'. However, he also states that 'the total weight of fish marketed (by the fish board) represents only part of the total catch, but it is a constantly related part and the figures can thus be used as an accurate indication of overall yearly trends'. We suspect that the barramundi of the Fitzroy and Port Curtis region have a large degree of interchange (via the Narrows) and that much of the barramundi landed in Gladstone could have been caught at the mouth of the Fitzroy or in the Narrows. While currently considered separately, evidence is building that barramundi landings currently allocated to Fitzroy and Port Curtis should be pooled into a 'greater Fitzroy'.

Mullet

Mullet is the consistent marketing name of a number of species, including *Mugil cephalus*. Mullet was a major component of the inshore net fisheries, suppling the local fish and chip trade with fish fillets. Mullet landings were often in excess of demand, leading to over-supply. To provide a means of return on glut quantities, the Queensland Fish Board bought excess landings and these were then sold to Government institutions (e.g. several hospitals, Peel Island Lazeret, and Brisbane Goal). An example of the scale of government purchase of mullet can be seen in 1936/1937, where a reported 2½ tons of sea mullet per week was supplied during the mullet season. This arrangement ceased in the mid 1970s. Mullet landings suffered from 'kerosene taint' in which fish affected by the taint could not be sold. This problem caused a change in fishing practices e.g. fishos normally taking fish from north of the Brisbane River estuary during the months of May and June were active elsewhere. Kerosene taint is likely to have changed landings for the Moreton Bay region, but its impact on mullet landings in the Fitzroy and Port Curtis regions is unknown. Export of mullet is first reported upon in the 1959/1960 report. The Queensland Fish Board mullet landings data are not the best representation of annual mullet landings because financial year is half way through the winter season fishery for sea mullet.

Blue threadfin

Blue threadfin is the current marketing name for *Eleutheronema tetradactylum* previously known as 'Cooktown salmon', 'Blue salmon' and 'salmon'. In all annual reports of the Queensland Fish Board, there are the categories 'salmon' and 'king', and on occasion, an additional category of 'salmon Burnett'. This lead to the interpretation of the QFB annual reports that 'salmon' referred to blue threadfin and 'king' or 'salmon Burnett' referred to king threadfin (*Polydactylus macrochir*). This validity of this assumption is unknown, and as such, the use of blue threadfin and king threadfin versus threadfin combined should be used with caution.

King threadfin

King threadfin is the current marketing name for *Polydactylus macrochir (=sheridani*) previously known as 'Burnett salmon' and 'king salmon'. The term 'salmon, Burnett' appears in the annual reports from 1945/19460 to 1949/1950, but thereafter only the categories 'salmon' and 'king' appear. The current interpretation of threadfin landings in the annual reports of the Queensland Fish Board is that 'salmon' referred to blue threadfin and 'king' or 'salmon Burnett' referred to king threadfin. This validity of this assumption is unknown, and as such, the use of data for blue threadfin and king threadfin versus threadfin combined should be used with caution.

Mud crabs

Hill (1982) reports that the Queensland landings data for mud crabs are unreliable, mostly because of the possibility of mud crabs being shipped directly to interstate markets. For example, crabs caught in Princess Charlotte Bay (about 20,000 per year) were not handled by any fish board. In 1980, the Sydney Fish Market handled more crabs than the Queensland Fish Board, despite the small size of the NSW mud crab fishery. However, Hill (1982) suggests that the Queensland Fish Board data give a relative index of the production of various areas.

Whiting

This market category is a combination of several species of estuarine whiting. It is unknown whether whiting were a target species in the Port Curtis and Fitzroy regions or if they were incidental catch to other estuarine netting operations (e.g. fish traps).

Appendix for Chapter 6. Year class strength of estuarine fish

Marginal Increment Analysis

We counted the opaque increments, and assigned the marginal increment to one of three categories: (i) 'new', when the opaque increment was on the margin; (ii) 'plus', when the opaque increment was separated from the margin by a narrow translucent increment; and (iii) 'due', when the distance from the outer opaque increment to the margin was almost equal to the width of the previous translucent increment.

There was very clear differentiation between fast and slow growth zones on barramundi otoliths from the Fitzroy River estuary, as found by Stuart and McKillup (2002). Opaque (light, narrow) increments were visible on the margin of most otoliths collected in October, but were rarely on the margin of otoliths collected in February (Table A.6.1) suggesting that increments form by October. Therefore, when estimating the age of a fish collected in October, and when an increment was not visible on or near the margin of the otolith, we assumed that one should have been present. Consequently, an extra year was added to the estimated ages of 46 fish that were caught in October but had an otolith increment classed as 'due'.

Table A.6.1 Results of marginal increment assessment for barramundi in the Fitzroy River region. Marginal increments defined as: 'new', when the opaque increment is on the margin; 'plus', when the opaque increment was separated from the margin by a narrow translucent increment; and 'due', when the distance from the outer opaque increment to the margin was almost equal to the width of the previous translucent increment.

		Marginal Increment			
	Sample Time	New	Plus	Due	Total
Year–1					
	October 2000	154			154
	February 2001	2	412		414
Year–2					
	October 2001	85		44	129
	February 2002	1	192	3	196
Year–3					
	October 2002	122		2	124
	February 2003		319		319

Counts of opaque increments were converted to ages, taking into account the assessment of marginal increment and the date of capture. We assigned 1 January as the birthday of each fish, as the spawning season for barramundi extends from approximately October to March on the east coast of Queensland (Russell 1990). Therefore, all fish born during the same spawning season were assigned the same birthday and identified as belonging to the same year-class. Once ages were estimated, age-length keys were constructed and used to convert length-frequencies into age-frequencies. Age-length keys and length-frequency distributions were constructed for each of sampling trips and a single age-structure was constructed for each spawning season sampled (October plus February).

Year-classes recruited to the fishery

Barramundi has minimum (580 mm) and maximum (1200 mm) legal size-limits on the east coast of Queensland. Therefore, the size-structure of the commercial catch is not representative of the whole population. To account for this potential bias, we restricted analyses to a range of age-classes that were likely to be least biased by the restricted size-structure. We selected age-classes for inclusion in analyses after we examined the size-distribution of each age-class, in both October and February samples (pooled for all years). The youngest age-class we included was the youngest one for which >90% of the individuals measured were larger than the minimum size-class of fish sampled (580–599 mm). This criterion was used as an approximate indication that most fish in this age-class were likely to have reached the minimum legal size-limit. Likewise, the oldest age-class we included in our analyses was the oldest one for which >90% of the individuals we measured were smaller than the maximum size-class of fish sampled (1180–1200 mm). This criterion was used as an approximate indication that most fish in this age-class were unlikely to have exceeded the maximum legal size-limit.

The size and age structure of sample fish were examined to determine which age-classes to include in the analysis. At the lower end of the size and age ranges, 92.5% of two year-old fish were larger than the smallest size-class (580–599 mm, with 580 mm being the minimum legal size), but only three of these were caught during October trips. Therefore, we believed this age-class had not recruited sufficiently to the commercial fishery. Of the three year-old fish, 96.1% were larger than the 580–599 mm size-class, and three year-olds were common in October and February trips, so we decided this age-class was the youngest one that should be included in our analyses. At the upper end of the size and age ranges, the youngest fish to have reached the maximum legal size-limit was 10 years-old, although most 10 year-old fish were smaller than 1090 mm. Similarly, most 11 year-old fish were smaller than 1080 mm, with 93.3% being smaller than the largest 20 mm size-class (1180–1200 mm). While most 12 year-old fish were smaller than 1060 mm, 28.6% were in the 1180–1200 mm size-class. Therefore, we decided that the 11 year-old age-class was the oldest one that should be included in our analyses.

A similar approach was applied to king threadfin, whose minimum legal size is 350 mm TL on the east coast of Queensland.

Migration of king threadfin

Our current analyses assume that migration king threadfin between estuaries is low. Individual king threadfin can move large distances along the coastline (e.g. 550 km Kailola *et al.* 1993), potentially confounding relationships between freshwater flow and the abundance of king threadfin. However, the frequency of these movements and proportion of the population that moves such large distances is unknown. We evaluated tag-recapture data from the Suntag Program of the Australian National Sport Fishing Association Queensland Inc. (ANSQ Qld) for evidence of migration of king threadfin. In the Fitzroy River region, 148 king threadfin were tagged and recaptured between 1986 and 2005, being at-liberty for between 0 and 2599 days (median = 227 days), at sizes between 340 and 1424 mm FL. Although individuals had moved within the Fitzroy River estuary and adjacent surrounds, none had been recapture more than 68 km from their release location supporting our assumption that that migration rates between estuaries were low for king threadfin in the Fitzroy River region.

Our method also assumes that the bands on the otoliths of king threadfin from the Fitzroy River estuary are indicative of annuli. We could not find any published aging studies on king threadfin, nor any known age individuals. However, otoliths of king threadfin sampled from the Fitzroy River estuary had clearly defined opaque and translucent bands (Fig. 4). Over the five years of sampling, we observed consistent differences in the marginal increment of king threadfin otoliths collected in October to those collected in following February. Therefore, we assumed that like barramundi (see Staunton-Smith *et al.* 2004), otolith increments of king threadfin in the Fitzroy River estuary were laid annually and that the first increment could be accurately identified. The Fitzroy River estuary is towards the southern limit of the distribution of king threadfin in Australia, and water temperatures (and food availability) drop considerably over winter. The assumption of annuli may not be valid throughout the distribution of king threadfin, but appears to be reasonable for the Fitzroy River estuary, until otolith annuli can be validated in known age fish.

Appendix for Chapter 7. Assumptions of barramundi growth rates

Only the short-term effects of freshwater flow on growth rates were investigated, after accounting for length-at-release, seasonality of growth rates, and time-at-liberty. Possible lag effects of freshwater flows on growth rates were not investigated i.e. flow conditions that occurred in the river prior to the tagging and release of an individual were not considered. It is possible that pre-existing conditions affected the observed growth rates. Other sources of variability that are likely to affect growth rates, such as individual genetic variation, were also not investigated. Whilst the data included information on habitat type at release and recapture, there was no information on whether these habitats were consistent throughout the time-at-liberty. However, as only fish at liberty for between 30 and 366 days were used, unquantified habitat effects are likely to be small.

The ANSA Qld tag-recapture data are collected mostly from recreational and commercial fishers. There is potential for error in measurement of the total length of a fish and subsequent error in calculated growth rates. However, tagging and measurement of fish has been ongoing in the Fitzroy River region for over 20 years, and ANSA Qld has numerous members in this region that have many years of experience tagging and measuring fish. In addition, where possible, data (e.g. changes in total length) and estimated parameters (e.g. L_∞, C, t_s and K) were compared for similarity with that reported in the literature.

Appendix for Chapter 9. Banana prawn data

Table A.9.1 Estimated annual index of banana prawn abundance in the Fitzroy River estuary using fortnightly and monthly data, as well as different factors in the GLM model

Data included in GLM analysis	Periodicity	GLM model and predicted means for annual index	Estimated means for calculating annual index	Annual index of banana prawn abundance[1]	s.e. of annual index[1]
All trips	Fortnightly	Trip	Trip	158.019	8.6720
New moon v1	Monthly	Trip	Trip	176.986	14.4644
New moon v2	Monthly	Trip	Trip	203.312	12.7372
Full moon v1	Monthly	Trip	Trip	132.115	13.3337
Full moon v2	Monthly	Trip	Trip	126.421	13.2404
All trips	Fortnightly	Trip, Region, Trip*Region	Trip	155.524	8.1207
New moon v1	Monthly	Trip, Region, Trip*Region	Trip	183.157	16.2440
New moon v2	Monthly	Trip, Region, Trip*Region	Trip	198.990	12.8707
Full moon v1	Monthly	Trip, Region, Trip*Region	Trip	127.955	11.4668
Full moon v1	Monthly	Trip, Region, Trip*Region	Trip	131.930	12.9547
Full moon v2	Monthly	Trip, Region	Trip	126.505	12.6576
Full moon v2	Monthly	Trip, Region	Trip	127.201	11.3618
All trips	Fortnightly	Trip, Region, Trip*Region	Trip by region	624.648	32.5057
New moon v1	Monthly	Trip, Region, Trip*Region	Trip by region	734.880	59.9274
New moon v2	Monthly	Trip, Region, Trip*Region	Trip by region	809.514	47.9371
Full moon v1	Monthly	Trip, Region, Trip*Region	Trip by region	501.056	49.8390
Full moon v2	Monthly	Trip, Region, Trip*Region	Trip by region	504.878	49.4648

[1] The index and s.e. are based on the cumulative mean prawn abundance per trip x number of days between trips) across the sampling period.

New moon trips v1 = trips 1,6,8,10 and 12; new moon trips v2 = 2,6,8,10 and 12; full moon trips v1 = trips 3,7,9,11 and 13; full moon trips v2 = 4,7,9,11 and 13.

As expected, fortnightly sampling provided greater power to detect between year differences in annual mean prawn abundance than monthly sampling at all hypothetical levels of negative differences. The results of the power analysis suggest that if the level of variation in year-2 is similar to that in year-1 sampling, then it is likely we will have a >80% probability of detecting a between year difference in prawn abundance that is 40% less than the year-1 abundance (assuming a 95% significance level i.e., α level). Monthly sampling (at either of the full or new moon) would provide a much lower power (~50% probability) to detect a between year difference in prawn abundance that is 40% less than the year-1 abundance.

Table A.9.2 Dates, mean lengths, growth increments and growth rates for each pair of samples of banana prawns from the Fitzroy River derived from Figure 8.5 and underlying Figure 8.6

Sample year	Cohort	First date	Mean carapace length (mm)	Second date	Mean carapace length (mm)	Growth increment (mm)	Time interval (days)	Growth rate (mm/week)	Total flow (ML)
Year-1	1.1	13/01/2002	5.53	28/01/2002	7.29	1.76	15	0.82	189,243
		28/01/2002	7.29	9/02/2002	7.96	0.67	12	0.39	17,316
		09/02/2002	7.96	23/02/2002	8.78	0.82	14	0.41	49,193
		23/02/2002	8.78	10/03/2002	9.29	0.51	15	0.24	36,130
		10/03/2002	9.29	24/03/2002	10.35	1.06	14	0.53	9,181
		24/03/2002	10.35	9/04/2002	11.59	1.24	16	0.54	997
		09/04/2002	11.59	23/04/2002	13.58	1.99	14	1.00	2,615
		23/04/2002	13.58	8/05/2002	15.13	1.55	15	0.72	315
	1.2	08/05/2002	9.98	22/05/2002	11.02	1.04	14	0.52	252.
Year-2	2.1	02/11/2002	3.19	16/11/2002	7.00	3.81	14	1.91	0
		16/11/2002	7.00	2/12/2002	9.57	2.57	16	1.12	0
		02/12/2002	9.57	17/12/2002	11.85	2.28	15	1.06	0
	2.2	17/01/2003	4.23	31/01/2003	5.52	1.29	14	0.65	0
		31/01/2003	5.52	15/02/2003	8.34	2.82	15	1.32	1,580,532
		15/02/2003	8.34	28/02/2003	11.05	2.71	13	1.46	123,798
	2.3	16/03/2003	4.51	31/03/2003	6.95	2.44	15	1.14	20,490
		31/03/2003	6.95	15/04/2003	8.64	1.69	15	0.79	11,7600
		15/04/2003	8.64	30/04/2003	9.58	0.94	15	0.44	6,2840
		30/04/2003	9.58	14/05/2003	11.00	1.42	14	0.71	4,101
		14/05/2003	11.00	29/05/2003	12.3	1.3	15	0.61	335
	2.4	30/04/2003	4.44	29/05/2003	7.37	2.93	29	0.71	4,436
Year-3	3.1	24/10/2003	6.76	23/11/2003	8.01	1.25	30	0.29	306
	3.2	21/12/2003	5.43	20/01/2004	8.09	2.66	30	0.62	101,970
		20/01/2004	8.09	18/02/2004	10.94	2.85	29	0.69	766,373
	3.3	18/02/2004	5.15	19/03/2004	8.10	2.95	30	0.69	111,6420
		19/03/2004	8.10	17/04/2004	10.67	2.57	29	0.62	11,095
		17/04/2004	10.67	17/05/2004	12.64	1.97	30	0.46	540
Year-4	4.1	10/11/2004	9.60	11/12/2004	11.30	1.70	31	0.38	27,001
	4.2	8/01/2005	8.74	7/02/2005	10.87	2.13	30	0.50	443,384
		07/02/2005	10.87	10/03/2005	12.64	1.77	31	0.40	70425
	4.3	07/04/2005	9.97	6/05/2005	12.83	2.86	29	0.69	522

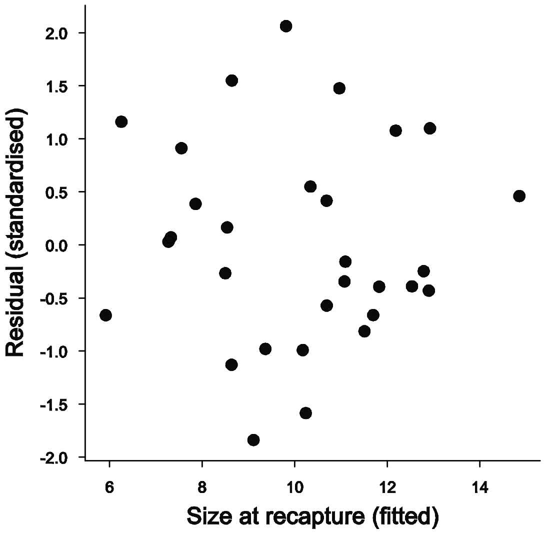
Table A.9.3 Dates, mean lengths, growth increments and growth rates for each pair of samples of banana prawns from the Calliope River underlying Figure 9.7

Sample year	Cohort	First date	Mean carapace length (mm)	Second date	Mean carapace length (mm)	Growth increment (mm)	Time interval (days)	Growth rate (mm/week)	Total flow (ML)
Year-2	2.1	04/12/2002	9.56	16/12/2002	12.57	3.01	12	1.76	0
	2.2	19/01/2003	9.01	2/02/2003	10.17	1.16	14	0.58	0
		02/02/2003	10.17	15/02/2003	12.09	1.92	13	1.03	262,921
	2.3	18/03/2003	7.54	2/04/2003	9.91	2.37	15	1.11	417
		02/04/2003	9.91	17/04/2003	11.31	1.40	15	0.65	301
		17/04/2003	11.31	2/05/2003	12.62	1.31	15	0.61	227
		16/05/2003	10.75	31/05/2003	12.99	2.24	15	1.05	190
Year-3	3.1	26/10/2003	4.28	25/11/2003	6.51	2.23	30	0.52	786
		22/01/2004	8.41	20/02/2004	10.14	1.73	29	0.42	49,342
		20/02/2004	10.14	21/03/2004	11.99	1.85	30	0.43	5,202
		21/03/2004	11.99	19/05/2004	16.58	4.59	59	0.54	632

Table A.9.4 Dates, mean lengths, growth increments and growth rates for each pair of samples of banana prawns from the Boyne River estuary underlying from Figure 9.8.

Sample year	Cohort	First date	Mean carapace length (mm)	Second date	Mean carapace length (mm)	Growth increment (mm)	Time interval (days)	Growth rate (mm/week)	Mean daily rainfall (mm)
Year-2	2.1	19/12/2002	11.92	6/01/2003	13.44	1.52	18	0.59	5.18
	2.2	30/01/2003	10	17/02/2003	13.13	3.13	18	1.22	27.19
		17/02/2003	13.13	27/02/2003	14.74	1.61	10	1.13	10.94
	2.3	15/03/2003	5.64	30/03/2003	7.14	1.5	15	0.70	1.37
		30/03/2003	7.14	14/04/2003	8.49	1.35	15	0.63	1.15
		14/04/2003	8.49	29/04/2003	10.13	1.64	15	0.77	0.71
	2.4	13/05/2003	7.52	28/05/2003	9.49	1.97	15	0.92	0.23
Year-3	3.1	22/11/2003	5.66	20/12/2003	11.13	5.47	28	1.37	6.30
	3.2	19/01/2004	8.07	17/02/2004	12.59	4.52	29	1.09	5.61
	3.3	18/03/2004	9.09	16/04/2004	11.56	2.47	29	0.60	0.82
		16/04/2004	11.56	16/05/2004	13.83	2.27	30	0.53	1.22

Figure A.9.1 Size at recapture and residual variance of banana prawn cohorts for Fitzroy River estuary data



Appendix for Chapter 10. Species lists of demersal communities

Table A.10.1 List of species caught by beam-trawl in the Fitzroy River estuary, with overall mean numbers per sample, and frequency of capture (i.e. percentage of trawls which caught at least one individual), ranked by species.

Species	Mean number per sample	Frequency of capture	Species	Mean number per sample	Frequency of capture	Species	Mean number per sample	Frequency of capture
Acetes spp.	1249.97	77.88%	*Latreutes cf pymoeus*	0.05	2.76%	*Megalops cyprinoides*	0.00	0.46%
Metapenaeus spp.	11.76	71.43%	*Leiognathus equulus*	0.05	2.61%	*Oratosquillina* sp. 1	0.00	0.46%
Penaeus merguiensis	27.04	69.59%	*Acanthopagrus berda*	0.04	2.61%	*Saurida tumbil*	0.00	0.46%
Thryssa hamiltoni	3.54	36.56%	Gobiidae sp. 2	0.04	2.61%	*Craterocephalus stercusmuscarum*	0.02	0.31%
Parapenaeopsis sculptilis	16.95	36.25%	Platycephalidae sp.	0.03	2.61%	*Drombus cf ocyurus*	0.01	0.31%
Johnius (Johnius) australis	4.45	33.18%	*Gerres subfasciatus*	0.09	2.46%	Unidentified gudgeon 2	0.01	0.31%
Prionobutis microps	3.59	29.34%	*Polydactylus macrochir*	0.04	2.46%	*Acentrogobius caninus*	0.00	0.31%
Pomadasys kaakan	1.16	29.19%	*Charybdis anisodon*	0.04	2.46%	*Pseudogobius* sp.	0.00	0.31%
Macrobrachium sp.	2.12	25.96%	*Pseudorhombus arsius*	0.04	2.00%	*Scomberoides* sp.	0.00	0.31%
Eleutheronema tetradactylum	0.79	23.50%	*Glossamia aprion*	0.04	1.84%	*Dexillichthys muelleri*	0.00	0.31%
Leiognathus decorus	1.36	20.43%	*Redigobius* sp. 2	0.03	1.84%	*Gobiopterus macrostoma*	0.00	0.31%
Thryssa setirostris	0.78	18.59%	*Platycephalus indicus*	0.02	1.84%	Larval fish 9	0.00	0.31%
Leandrites celebensis	2.14	18.28%	*Ambassis agassizii*	0.03	1.69%	*Monacanthus chinensis*	0.00	0.31%
Brachyamblyopus coecus	0.64	16.44%	*Paradicula setifer*	0.02	1.69%	*Muraenosox bagio*	0.00	0.31%
Stolephorus commersonii	0.80	16.28%	*Drombus* sp.	0.03	1.54%	*Amoya* sp.	0.00	0.15%
Aseraggodes rautheri	0.72	14.44%	Larval fish 1	0.02	1.54%	Carid sp. 4	0.00	0.15%
Periclimenes sp.	3.72	14.13%	*Paraplagusia sinerama*	0.02	1.54%	Larval fish 6	0.00	0.15%
Atypopenaeus formosus	0.86	13.21%	*Arenigobius frenatus*	0.02	1.54%	*Terapon* sp.	0.00	0.15%
Ambassis gymnocephalus	1.08	12.60%	Unidentified fish 3	0.06	1.38%	Blennidae sp.	0.00	0.15%
Marilyna pleurosticta	0.26	12.44%	*Callianassa australiensis*	0.03	1.38%	Carid sp. 1	0.00	0.15%
Loliolus noctiluca	0.25	12.14%	*Redigobius bikolanus*	0.03	1.23%	*Caridina longirostris*	0.00	0.15%
Valamugil sp.	0.42	11.98%	*Pelates quadrililineatus*	0.02	1.23%	*Caridina nilotica*	0.00	0.15%
Cynoglossus sp. 2	0.45	11.52%	*Enigmaplax littoralis*	0.02	1.23%	*Ctenotrypauchen microcephalus*	0.00	0.15%
Sillago spp.	0.26	11.52%	*Lutjanus russelli*	0.01	1.23%	*Drepane punctata*	0.00	0.15%
Nematalosa erebi	1.63	10.75%	*Cynoglossus* sp. 1	0.04	1.08%	*Epinephelus coioides*	0.00	0.15%
Escualosa thoracata	0.28	10.29%	*Philocheras cf angustirostris*	0.04	1.08%	Gobiidae sp. 1	0.00	0.15%
Selenotoca multifasciata	0.47	9.68%	*Polydactylus multiradiatus*	0.01	1.08%	*Hyporhamphus quoyi*	0.00	0.15%
Thryssa sp.	0.47	9.68%	Carid sp. 2	0.03	0.92%	*Hypseleotris sp*	0.00	0.15%
Favonigobius exquisitus	0.38	8.91%	*Aseraggodes* sp.	0.02	0.92%	*Latreutes mucronatus*	0.00	0.15%
Herklotsichthys castelnaui	0.18	7.68%	*Macropthalmus latreillei*	0.01	0.92%	*Leptobrama muelleri*	0.00	0.15%
Philypnodon grandiceps	2.83	7.07%	*Pantolabus radiatus*	0.01	0.92%	*Monodactylus argenteus*	0.00	0.15%
Hypseleotris compressa	0.66	6.91%	Palaemon sp.	0.06	0.77%	*Mugilogobius* sp.	0.00	0.15%
Alpheus sp. 1	0.10	6.91%	*Harpadon transluscens*	0.01	0.77%	*Nematalosa come*	0.00	0.15%
Larval fish 4	3.56	6.45%	*Pelates* sp.	0.01	0.77%	*Oratosquillina interupta*	0.00	0.15%
Butis butis	0.26	6.45%	*Acanthopagrus australis*	0.01	0.77%	*Platycephalus fuscus*	0.00	0.15%
Palaemon serrifer	0.34	6.14%	*Lates calcarifer*	0.01	0.77%	*Portunus* sp.	0.00	0.15%
Hyporhamphus sp 1	0.10	5.84%	*Australoplax tridentata*	0.01	0.77%	*Scylla serrata*	0.00	0.15%
Chelonodon patoca	0.07	5.68%	*Liza subviridis*	0.01	0.77%	*Siganus guttatus*	0.00	0.15%
Ambassis vachelli	0.55	5.53%	*Metapenaeopsis palmensis*	0.01	0.61%	*Siganus* sp.	0.00	0.15%
Redigobius sp. 1	0.15	5.38%	*Craterocephalus mugiloides*	0.01	0.61%	*Suggrundus* sp.	0.00	0.15%
Glossogobius biocellatus	0.07	3.53%	*Macropthalmus* sp.	0.01	0.61%	*Torquigener pleurogramma*	0.00	0.15%
Pseudomugil signifer	0.39	3.38%	*Dorripe* sp.	0.01	0.61%	Unidentified fish 5	0.00	0.15%
Arius graeffei	0.13	3.23%	Gobiidae sp. 3	0.01	0.61%	Unidentified fish 6	0.00	0.15%
Portunus pelagicus	0.11	3.23%	*Sphyraena putnamae*	0.01	0.61%	Xanthidae sp.	0.00	0.15%
Alpheus sp. 2	0.05	3.23%	*Gerres oyena*	0.09	0.46%	*Metapenaeopsis novoguineae*	0.00	0.15%
Terapon theraps	0.42	3.07%	*Aliaporcellana* sp.	0.01	0.46%	*Trachypenaeus fulvus*	0.00	0.15%
Apocryptodon mandurensis	0.03	3.07%	*Urocaridella urocaridella*	0.01	0.46%			
Metapenaeus ensis	0.06	2.76%	*Callionymus russelli*	0.00	0.46%			

Table A.10.2 List of species caught by beam-trawl in the Calliope River estuary, with overall mean numbers per sample, and hit rate (i.e. percentage of trawls which caught at least one individual), ranked by species.

Species	Mean number per sample	Frequency of capture	Species	Mean number per sample	Frequency of capture	Species	Mean number per sample	Frequency of capture
Acetes spp.	127.95	74.90	*Penaeus esculentus*	0.05	3.64	*Redigobius* sp. 2	0.01	1.21
Thryssa hamiltoni	7.84	63.56	*Enigmaplax littoralis*	0.05	3.24	*Sphyraena* sp.	0.01	1.21
Penaeus merguiensis	23.80	63.16	*Latreutes mucronatus*	0.13	3.24	*Strongylura* sp.	0.01	1.21
Leiognathus decorus	5.68	52.63	*Oratosquillina* sp. 1	0.11	3.24	*Parapenaeopsis sculptilis*	0.01	1.21
Metapenaeus sp.	4.72	52.23	*Pseudomugil signifer*	0.11	3.24	*Penaeus plebejus*	0.01	1.21
Sillago spp.	3.23	51.42	*Thryssa setirostris*	0.07	3.24	*Ambassis agassizii*	0.00	0.81
Stolephorus commersonii	6.09	43.32	*Acanthopagrus australis*	0.03	2.83	*Ambassis urotaenia*	0.00	0.81
Favonigobius exquisitus	1.03	32.79	*Butis butis*	0.02	2.83	*Apogon fasciatus*	0.00	0.81
Ambassis gymnocephalus	20.17	31.58	*Drombus* sp.	0.03	2.83	*Apogon nigripinnis*	0.00	0.81
Loliolus noctiluca	0.66	30.36	Gobiidae sp. 3	0.04	2.83	*Aseraggodes* sp.	0.00	0.81
Pomadasys kaakan	0.64	28.34	*Nematalosa erebi*	0.29	2.83	Carid sp. 1	0.00	0.81
Leandrites celebensis	1.73	24.70	*Redigobius sp. 1*	0.04	2.83	Carid sp. 3	0.00	0.81
Herklotsichthys castelnaui	3.46	21.46	*Siganus sp.*	0.04	2.83	Carid sp. 4	0.00	0.81
Marilyna pleurosticta	0.50	19.43	*Metapenaeopsis palmensis*	0.05	2.83	*Caridina longirostris*	0.00	0.81
Glossogobius biocellatus	0.33	18.22	*Arenigobius frenatus*	0.02	2.43	*Charybdis anisodon*	0.00	0.81
Ambassis vachelli	7.78	14.17	*Cynoglossus* sp. 2	0.02	2.43	*Cynoglossus* sp. 1	0.00	0.81
Callionymus russelli	0.21	14.17	*Leiognathus equulus*	0.05	2.43	*Dexillichthys muelleri*	0.00	0.81
Periclimenes sp.	1.11	14.17	*Monacanthus chinensis*	0.02	2.43	*Drepane punctata*	0.00	0.81
Hyporhamphus sp. 1	0.18	12.96	*Sepia* sp.	0.02	2.43	*Eleutheronema tetradactylum*	0.00	0.81
Drombus cf ocyurus	0.56	12.55	*Siganus guttatus*	0.04	2.43	Gobiidae sp. 2	0.00	0.81
Gerres subfasciatus	0.36	12.55	*Urocaridella urocaridella*	0.06	2.43	*Hyporhamphus* sp. 2	0.00	0.81
Valamugil sp.	0.71	12.55	*Aliaporcellana* sp.	0.02	2.02	*Hypseleotris compressa*	0.01	0.81
Acanthopagrus berda	0.22	10.93	*Amoya* sp.	0.03	2.02	*Lutjanus argentimaculatus*	0.00	0.81
Alpheus sp. 1	0.19	10.93	Larval fish 1	0.06	2.02	*Macropthalmus latreillei*	0.00	0.81
Alpheus sp. 2	0.34	10.53	*Oratosquillina* sp. 2	0.02	2.02	*Macropthalmus* sp.	0.00	0.81
Macrobrachium sp.	0.23	9.31	*Metapenaeopsis novoguineae*	0.02	2.02	*Oratosquillina interupta*	0.00	0.81
Selenotoca multifasciata	0.29	8.91	*Metapenaeus ensis*	0.02	2.02	*Palaemon* sp.	0.00	0.81
Latreutes cf pymoeus	0.30	8.50	*Brachyamblyopus coecus*	0.02	1.62	*Paradicula setifer*	0.00	0.81
Thryssa sp.	0.26	8.50	Carid sp. 2	0.02	1.62	*Pelates quadrililineatus*	0.00	0.81
Saurida tumbil	0.10	7.69	*Dorripe sp.*	0.02	1.62	*Pelates* sp.	0.00	0.81
Acentrogobius caninus	0.06	6.07	*Epinephelus coioides*	0.01	1.62	*Platycephalus arenarius*	0.00	0.81
Lutjanus russelli	0.07	6.07	*Liza subviridis*	0.02	1.62	*Platycephalus fuscus*	0.00	0.81
Portunus pelagicus	0.09	6.07	*Megalops cyprinoides*	0.02	1.62	*Pomadasys maculatum*	0.00	0.81
Chelonodon patoca	0.06	5.67	*Monodactylus argenteus*	0.01	1.62	Scomberoides sp.	0.02	0.81
Benthopanope estuarius	0.10	5.26	Platycephalidae sp.	0.01	1.62	*Siganus fuscesens*	0.01	0.81
Escualosa thoracata	0.23	5.26	*Portunus* sp.	0.01	1.62	*Suggrundus* sp.	0.00	0.81
Prionobutis microps	0.06	5.26	*Sphyraena putnamae*	0.01	1.62	*Torquigener pleurogramma*	0.00	0.81
Terapon theraps	0.15	5.26	*Callianassa australiensis*	0.01	1.21	Unidentified fish 4	0.00	0.81
Tripodichthys angustifrons	0.06	4.86	*Euprymna sp*	0.01	1.21	Xanthidae sp.	0.01	0.81
Apocryptodon mandurensis	0.05	4.45	*Gerres oyena*	0.01	1.21	*Atypopenaeus formosus*	0.00	0.81
Palaemon serrifer	0.17	4.05	*Hyporhamphus quoyi*	0.01	1.21	*Penaeus monodon*	0.00	0.81
Philocheras cf angustirostris	0.06	4.05	*Johnius (Johnius) australis*	0.01	1.21	*Trachypenaeus fulvus*	0.00	0.81
Philypnodon grandiceps	0.20	4.05	*Mugilogobius* sp.	0.01	1.21	*Petroscirtes lupus*	0.00	0.40
Pseudorhombus arsius	0.05	4.05	*Polydactylus multiradiatus*	0.01	1.21			
Platycephalus indicus	0.04	3.64	*Redigobius bikolanus*	0.01	1.21			

Table A.10.3 List of species caught by beam-trawl in the Boyne River estuary, with overall mean numbers per sample, and hit rate (i.e. percentage of trawls which caught at least one individual), ranked by species.

Species	Mean number per sample	Frequency of capture	Species	Mean number per sample	Frequency of capture	Species	Mean number per sample	Frequency of capture
Acetes spp.	2191.61	89.32	*Siganus* sp.	0.27	5.13	Carid sp. 2	0.06	1.28
Penaeus merguiensis	111.26	79.49	*Acanthopagrus australis*	0.07	4.70	*Ctenotrypauchen microcephalus*	0.01	0.85
Leiognathus decorus	19.66	76.50	*Gerres oyena*	0.07	4.70	*Epinephelus coioides*	0.01	0.85
Metapenaeus sp.	16.59	75.21	*Prionobutis microps*	0.04	4.27	*Leiognathus moretoniensis*	0.01	0.85
Thryssa hamiltoni	19.58	70.94	*Palaemon serrifer*	0.07	4.27	*Platycephalidae sp*	0.01	0.85
Ambassis gymnocephalus	57.77	53.42	*Polydactylus multiradiatus*	0.09	4.27	*Penaeus plebejus*	0.01	0.85
Sillago sp.	4.02	47.86	*Escualosa thoracata*	0.05	3.85	*Centrogenys vaigiensis*	0.01	0.85
Favonigobius exquisitus	1.59	42.31	*Sphyraena putnamae*	0.03	3.42	*Glossamia aprion*	0.01	0.85
Loliolus noctiluca	1.30	41.88	*Metapenaeopsis novoguineae*	0.03	3.42	Unidentified fish 7	0.01	0.85
Pomadasys kaakan	1.60	35.04	*Strongylura strongylura*	0.04	3.42	*Liza melinoptera*	0.02	0.85
Hyporhamphus sp 1	1.43	33.33	*Lutjanus russelli*	0.03	2.99	*Craterocephalus mugiloides*	0.03	0.85
Stolephorus commersonii	3.04	33.33	*Platycephalus indicus*	0.03	2.99	*Hyporhamphus quoyi*	0.03	0.85
Valamugil sp.	15.31	30.77	*Urocaridella urocaridella*	0.05	2.99	*Drepane punctata*	0.00	0.48
Glossogobius biocellatus	0.60	28.21	*Parapenaeopsis sculptilis*	0.09	2.99	*Ambassis* sp.	0.00	0.43
Macrobrachium sp.	2.30	26.07	*Callianassa australiensis*	0.14	2.99	*Amoya* sp.	0.00	0.43
Callionymus russelli	0.65	25.64	*Drombus* sp.	0.03	2.56	*Apogon poecilopterus*	0.00	0.43
Periclimenes sp.	1.67	20.94	*Metapenaeus ensis*	0.03	2.56	*Aseraggodes* sp.	0.00	0.43
Herklotsichthys castelnaui	2.63	20.94	*Benthopanope estuarius*	0.03	2.56	Carid sp. 1	0.00	0.43
Marilyna pleurosticta	0.97	20.51	*Charybdis anisodon*	0.03	2.56	*Caridina longirostris*	0.00	0.43
Leandrites celebensis	2.35	20.51	*Monodactylus argenteus*	0.03	2.56	*Eleutheronema tetradactylum*	0.00	0.43
Pseudomugil signifer	7.78	20.51	Gobiidae sp. 2	0.03	2.56	Larval fish 1	0.00	0.43
Ambassis vachelli	21.03	20.09	*Portunus* sp.	0.03	2.56	Larval fish 6	0.00	0.43
Thryssa sp.	1.38	17.09	*Torquigener pleurogramma*	0.03	2.56	*Lates calcarifer*	0.00	0.43
Drombus cf ocyurus	0.32	15.38	*Monacanthus chinensis*	0.07	2.56	*Leucosia ocellata*	0.00	0.43
Acanthopagrus berda	0.36	13.68	*Siganus fuscesens*	0.13	2.56	*Macropthalmus latreillei*	0.00	0.43
Selenotoca multifasciata	0.19	11.54	*Liza subviridis*	0.64	2.56	*Mugilogobius* sp.	0.00	0.43
Gerres subfasciatus	0.36	11.54	*Dexillichthys muelleri*	0.02	2.14	*Nematalosa come*	0.00	0.43
Philocheras cf angustirostris	0.26	10.68	*Redigobius* sp. 2	0.02	2.14	*Palaemon* sp.	0.00	0.43
Alpheus sp. 1	0.23	10.26	*Dorripe* sp.	0.03	2.14	*Paradicula setifer*	0.00	0.43
Leiognathus equulus	0.65	10.26	*Macropthalmus* sp.	0.03	2.14	*Pelates* sp.	0.00	0.43
Portunus pelagicus	0.18	9.40	*Sillago burrus*	0.03	2.14	*Petroscirtes lupus*	0.00	0.43
Chelonodon patoca	0.15	8.97	*Enigmaplax littoralis*	0.05	2.14	*Platycephalus fuscus*	0.00	0.43
Apocryptodon mandurensis	0.18	8.55	*Terapon theraps*	0.09	2.14	*Pomadasys maculatum*	0.00	0.43
Brachyamblyopus coecus	0.28	8.55	*Latreutes mucronatus*	0.27	2.14	*Pseudogobius* sp.	0.00	0.43
Alpheus sp. 2	0.12	8.12	*Oratosquillina* sp. 2	0.02	1.71	*Strongylura* sp.	0.00	0.43
Saurida tumbil	0.14	8.12	*Johnius (Johnius) australis*	0.02	1.71	*Suggrundus* sp.	0.00	0.43
Metapenaeopsis palmensis	0.21	8.12	*Lethrinus* sp. 1	0.02	1.71	*Synanceia horrida*	0.00	0.43
Nematalosa erebi	0.18	7.69	*Oratosquillina* sp. 1	0.02	1.71	*Tetractenos hamiltoni*	0.00	0.43
Philypnodon grandiceps	1.19	7.69	*Trachypenaeus fulvus*	0.02	1.71	Unidentified crab 1	0.00	0.43
Siganus guttatus	0.10	6.84	*Atypopenaeus formosus*	0.03	1.71	Unidentified crab 2	0.00	0.43
Pseudorhombus arsius	0.13	6.84	*Euprymna* sp.	0.03	1.71	*Metapenaeopsis* sp.	0.00	0.43
Redigobius sp. 1	0.13	6.84	Gobiidae sp. 3	0.04	1.71	*Alpheus* sp. 3	0.01	0.43
Thryssa setirostris	0.56	6.84	*Arenigobius frenatus*	0.01	1.28	Gobiidae sp. 1	0.01	0.43
Penaeus esculentus	0.34	6.41	*Sepia* sp.	0.01	1.28	Larval fish 4	0.01	0.43
Latreutes cf pymoeus	0.33	5.98	*Siphamia rosiegaster*	0.01	1.28	*Oratosquillina interupta*	0.01	0.43
Tripodichthys angustifrons	0.06	5.56	*Apogon fasciatus*	0.02	1.28	Scomberoides sp.	0.01	0.43
Butis butis	0.08	5.56	*Thalamita admete*	0.02	1.28	*Australoplax tridentata*	0.02	0.43
Cynoglossus sp. 2	0.07	5.13	*Pelates quadrililineatus*	0.03	1.28	Carid sp. 4	0.02	0.43
Acentrogobius caninus	0.13	5.13	*Redigobius bikolanus*	0.03	1.28			