Influence of Climate and Estuarine Habitat Characteristics on Coastal Fisheries - A Case Study for Queensland, Australia

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INFLUENCE OF CLIMATE AND ESTUARINE HABITAT CHARACTERISTICS ON COASTAL FISHERIES - A CASE STUDY FOR QUEENSLAND, AUSTRALIA

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A thesis submitted in fulfilment of the requirements of the

degree of Doctor of Philosophy.

Australian Rivers Institute

School of Environment

Griffith University

Gold Coast

December 2007

STATEMENT OF ORIGINALITY

This thesis contains no material extracted in whole or in part from a thesis by which I have qualified for or been awarded another degree or diploma.

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Jan-Olaf Meynecke

31 December 2007

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„Es ist nicht genug zu wissen, man muß es auch anwenden. Es ist nicht genug zu wollen, man muß es auch tun" (Goethe).

ABSTRACT

Estuarine habitats such as seagrass beds, mangrove forests, salt marshes, mud and sandflats are considered important habitats supporting a diverse range of fish and crustaceans of ecological and economic importance. Many commercial fish species use estuarine habitats as nurseries or breeding grounds and have lifecycles correlated to rainfall and temperature variations. Despite the globally recognised ecological value of estuaries, there is little direct evidence to support the estuary–fisheries linkage paradigm. Traditionally, analyses of the importance of estuarine habitat characteristics to commercial fish catch have taken the single-habitat approach, i.e., assessing the value of individual habitat types. Different estuarine habitats offer different, and arguably, complementary functions and services to nekton during their life cycle. Connectivity of habitats is therefore likely to play a major role in fish movement and abundance. Little is known of movement patterns of estuary-dependent fish species within the estuarine landscape and few if any studies have applied the idea of spatial habitat metrics, in particular, connectivity, to estuarine environments. Elements of climate, such as temperature and rainfall, may impact on sustainable coastal fish production. How these factors, which are strongly related to global climate change, may influence future coastal fish production will have significant implications for sustainable fishery management.

This thesis examines relationships between estuarine habitat characteristics, climate and commercial fish species in Queensland, Australia, with the objective of gaining clearer insight into how estuarine habitats may support commercial fisheries. The following specific questions were addressed:

(1) What knowledge gaps exist in the literature regarding the role of climate and estuarine habitats in sustaining coastal fisheries?

(2) What deficiencies exist in studies that addressed the relationship between estuarine habitats, climate (e.g., rainfall and temperature) and fisheries and how may existing data sets be refined to allow better analyses of this relationship?

(3) What are the key structural features of estuarine habitats (e.g., connectivity) and the role of temperature and rainfall influencing coastal fishery production?

(4) What are the implications of the relationships between fish catch, estuarine habitat and climate, for fisheries management?

ABSTRACT

The primary data sources for this study were (1) 17 years of commercial fish catch data from over 30 selected estuarine fish species or species groups; and (2) estuarine habitat maps for Queensland, Australia. Queensland appears to be the best study region as it had the largest extent of soft sediment estuarine habitats, comprising seagrass, salt marsh and mangroves, and offers long-term commercial fish catch data compared to other states in Australia. As a first step, the relationship between catch (both as catch-per-unit-effort (CPUE) and total catch) for commercially caught species in their dominant fisheries (trawl, line, net or pot fisheries) and estuarine geomorphic spatial metrics were extracted from digital habitat maps in geographic information systems (GIS). For preliminary analyses, I examined 13 geographical regions in Queensland based on their importance to commercial fisheries. Spatial metric characteristics such as Euclidean distance, patch density and landscape connectivity for 273 estuaries along the Queensland coast were then calculated within 90 fish catch grids, which provided inshore fish catch data from 21 species groups.

The collective spatial characteristics of estuarine habitats such as size and structural connectivity showed significant correlation with fish catch, with r^2 values > 0.7 for 17 commercial species groups. More detailed multiple regression analysis and non-metric multidimensional scaling (nMDS) plots showed significant links between geomorphic coastal features such as structural connectivity and nearshore fisheries production. The relationship was best explained by connectivity indices for mangroves, salt marsh and channels, further suggesting the fundamental importance of connected tidal wetlands to the fish catch.

Climate-driven dependencies of fish catch were explored by using CPUE, rainfall, coastal air temperature and the Southern Oscillation Index (SOI) and catch time-series for specific combinations of climate, seasons and regions. Surplus production models where applied to the commercial fish catch data using the program CLIMPROD. In addition to habitat configuration, climate, particularly rainfall, played a major role in the species composition of the fish catch. Up to 30 % of Queensland's total fish catch and up to 80 % of the barramundi catch variation for specific regions were explained by rainfall, often with a time-lag response to rainfall events. Temperature had similar

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influences on fish and prawn catches, with high temperatures having a negative effect on the prawn catch in the Gulf of Carpentaria.

The importance of estuary-scale habitat connectivity to fish populations was assessed by studying fish movement in relation to the extent of flooding for several commercially and recreationally important species in and out of a small mangrove creek using a stationary passive integrated transponder (PIT) system, augmented by underwater digital video recorders (DVRs) for visual sensing. A high-resolution digital elevation model derived from airborne light detection and ranging (LIDAR) and aerial imagery was used to estimate inundation pattern of intertidal habitats to compare with movement of PIT tagged fish. Investigations of fish movement at a detailed scale confirmed the broad-scale findings that habitat connectivity and close spatial proximity of resources are key factors for estuary-dependent fish species in tide-dominated systems to access different habitats. Access to habitat resources such as mangroves is more limited in time than previously thought and is dependent on fish species and size, thus suggesting importance of other adjoining habitats. Fish species and size also affected time of residency and diel pattern of habitat use.

The analyses presented in this thesis also allow an evaluation of the economic consequences of climate parameters on estuarine fisheries. Species-specificity in climate-catch relationships suggested a strong need to develop forecast models and manage estuaries for future climate change by adjusting the quota for the more sensitive species. In addition, my findings on the role of estuarine habitat connectivity could guide the construction of a network of protected marine areas of various structural configurations that can optimise ecosystem services. The demonstrated value of habitat connectivity further warns against future reduction of connectivity by habitat destruction and climate change. The new techniques developed for assessing the spatial ecology of estuarine fish and their habitat utilisation and site fidelity behaviour will assist evaluation of the role of estuarine habitat inter-dependencies in supporting coastal fish populations.

The fish catch data set used for fisheries management is valid for broad-scale investigations. However, it can be insufficient for managing certain stocks, e.g., fish targeted by recreational fishers. Data sets can be significantly improved by in-

ABSTRACT

corporating locally collected recreational fish catch data, which can provide detailed information for particular river systems and allow better assessments of potential overfishing.

The results of this study suggest that sound ecosystem-based fisheries management requires a diversity of information, an improved catch record system, a species-specific approach to managing the impact of climate change, and ecologically meaningful protection areas with high habitat connectivity.

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TERMINOLOGY AND ABBREVIATION

In the context of this thesis the following terms and abbreviations are used:

- AR – Auto-Regression
- ARIMA - Autoregressive Integrated Moving Average
- BIO-ENV - Biological environmental gradients procedure
- BNC - Barron North Coast
- BOM - Bureau of Meteorology
- CA - Total Area
- CHRIS - Coastal Habitat Resources Information System
- CLIMLAB 2000 - Climate Modelling Laboratory 2000
- CONNECT - Connectivity Index
- CPUE – Catch-Per-Unit-Effort
- CSIRO - Commonwealth Scientific and Industrial Research Organisation
- CYPLUS - Cape York Peninsula Land Use Study
- DEM - Digital Elevation Model
- DPI&F - Department of Primary Industries and Fisheries
- DVR - Digital Video Recorder
- EC - East Coast
- ECC - East Central Coast
- ENN - Euclidean Nearest-Neighbour Distance
- ENSO - El Niño-Southern Oscillation
- EPA - Environmental Protection Agency
- FAO - The Food and Agricultural Organisation of the United Nations
- GAM - Generalised Additive Modelling
- GIS - Geographical Information System
- GLM - General Linear Models
- GOC - Gulf of Carpentaria
- GVP - Gross Value Production
- HNC - Herbert North Coast
- IPCC - Intergovernmental Panel on Climate Change
- LA - Latitude

- LC - Length of Coastline
- LCE - Length Connected Edge
- LCPA - Length Connected Edge to Area Ratio
- LIDAR - Light Detection and Ranging
- MPB - Microphytobenthos
- MSC - Moreton South Coast
- MSY - Maximum Sustainable Yield
- NatMIS - National Marine Information System
- NE - Number of Estuaries
- *n*MDS – Non-Metric Multidimensional Scaling
- NP - Number of Patches
- NRM - Department for Natural Resources and Mines
- PARA - Mean Perimeter to Area Ratio
- PC - Principal Component
- PCA - Principal Component Analyses
- PD - Patch Density
- Peri - Perimeter
- PIT - Passive Integrated Transponder
- PSC - Port Curtis South Coast
- RL - River Length
- SOI - Southern Oscillation Index
- SP - South Peninsula

CHAPTER 1

GENERAL INTRODUCTION

1.1   The value of estuaries to coastal fisheries

Despite the wide recognition of estuaries as an important driver of nearshore fish productivity (Anon, 1992; Leadbitter and Doohan, 1992), a direct link between estuarine habitats and fisheries is yet to be found (Duke et al., 2007). Estuaries have been assigned the highest economic value per hectare of any ecosystem, estimated at A\$39,000 y^{-1} (Costanza et al., 1997). Important commercial fisheries exist in estuaries and near-shore waters of Australia for penaeid prawns (A\$~360 million per year) and finfish (A\$~55 million per year) (CRC Reef, 2005). Recreational fishing is an especially important activity for Australians, involving an estimated 3.4 million anglers (~20 % of population) fishing each year (DAFF, 2000). In Queensland, Australia, up to 75 % by weight and 80 % by value of the total commercial fish catch (A\$300 million per year) is composed of estuary-dependent species (Quinn, 1992).

The role of estuarine habitats as nursery grounds is widely accepted: mangroves in particular are believed to provide important nursery grounds to coastal nekton (Blaber, 2000; Beck et al., 2001; Kathiresan and Bingham, 2001). In addition, the export of organic matter from productive estuarine wetlands may be an important contribution to sustaining nearshore fishery production (Odum, 1980; Lee, 1995). There is also an assumption that the area of tidal wetland habitats in an estuary numerically translates into the secondary production and catch of commercial fisheries (Baran and Hambrey, 1998). However, not all habitats function equally as nurseries (Beck et al., 2001) and their effectiveness is likely to depend on other factors such as climate, but a comparison of their values has rarely been undertaken. It may, in fact, be a particular combination of habitats and their accessibility (i.e., the 'seascape') that explains the importance of estuaries to nekton (Cappo et al., 1998). The connectivity of habitats may be a key factor supporting estuarine fisheries by allowing exchange of energy and organisms

between estuarine habitats (Merriam, 1984; Chong et al., 1990; Nagelkerken et al., 2001).

1.2 Factors influencing estuarine fish assemblages

Multiple factors can influence fisheries species within an estuary. However, the definition of estuaries appears often vague and unclear. For this study I adopt the definition of Pritchard (1967), i.e., an estuary as a semi-enclosed coastal body of water which has a free connection with the open sea and within which sea water is measurably diluted with fresh water derived from land drainage, often extending to the tidal influence (Perillo, 1995). An estuary forms a collective, complex system containing various habitats (Romao, 1996). Multiple factors within an estuary can influence the abundance of fish and crustacean fisheries species assemblages. These include both physical and biological variables, such as (1) vegetation (Cappo et al., 1998; Beck et al., 2001); (2) currents (Flierl et al., 1999); (3) tides (Bucher and Saenger, 1994); (4) wind and wave energy (Kailola et al., 1993); (5) geomorphology (Gribble, 1997); (6) geology (Scanlon et al., 2002); (7) biogeographic units (McAllister et al., 1994); and (8) climate (Robertson and Duke, 1990; Smith, 1990b; Staples et al., 1995; Furnas et al., 1997). This study will focus especially on the influence of habitat distribution, and the climatic factors rainfall and temperature on fisheries species production.

1.2.1 Climate and its influence on estuarine fish

Climate, in particular temperature and rainfall interact with ocean currents, larval supply and survival of coastal nekton through, for example, initiating the breeding cycles of grapsid crabs (Smith, 1990a). The variability in rainfall governs flood runoff (Furnas et al., 1997) and can have a strong influence on fish catch (Staunton-Smith et al., 2004). The effect of rainfall on banana prawn (*Penaeus merguiensis*) recruitment is one of the earliest examples of use of environmental data in fisheries production models (Staples et al., 1995), with heat and light governing primary production, resulting in zooplankton (copepod genera such as: *Oithona*, *Pseudodiaptomus*, *Acartia*, *Paracalamus*) showing high abundance during the wet season (Robertson and Duke, 1990). Long-term temperature and rainfall effects are also altering the distribution of estuarine habitats

(Saintilan and Williams, 1999), indicating a sensitivity of estuary-dependent fish species to climate change.

The climate along the Queensland coast ranges from tropical in the far north, to sub-tropical in the south-east. During the sub-tropical wet season (November-April) average maximum monthly air temperature is 28°C and average minimum is 20°C. In the dry season these temperatures drop to 20°C and 9°C, respectively. During the tropical wet season average maximum monthly temperature is 33°C, and average minimum of 25°C. In dry season these temperatures drop to 26°C and 17°C, respectively (BOM, 2007b). Mean annual coastal rainfall ranges from 1,200 mm in Brisbane to > 4,000 mm in Babinda and is distinctly seasonal, with 75 - 90 % occurring between November and April (BOM, 2007b). Widely differing rainfall regimes are characteristic in Queensland and may vary considerably even over short distances because of changes in local topography, including height and orientation of mountain ranges, and the direction of the coastline with respect to the prevailing moist south-east airstream. In general, rainfall is high in tropical Queensland and lower in sub-tropical Queensland but conjoining areas of high and low rainfall are present along the tropical Queensland coast – the 'wet' and 'dry' tropics.

The climatic analysis of the Intergovernmental Panel on Climate Change (IPCC) report for Australia predicted a trend to greater dryness in the next 50 to 100 years and a clear increase in temperature (IPCC, 2001). In accordance, CSIRO regional projections predicted an increase in annual average temperatures of $0.4 - 2.0°C$ by 2030 (relative to 1990) and $1.0 - 6.0°C$ by 2070. Considerable uncertainty remains with future changes in rainfall for 2070, which may vary between +10 % and -35 % for the east coast of Australia (Walsh et al., 2000; Hughes, 2003). Such variations are closely linked to Southern Oscillation fluctuations and are similar to Northern Pacific atmospheric forcing (Norton and McLain, 1994). Over the last decades, El Niño events increased in frequency and intensity with the Southern Oscillation Index (SOI) mirroring the rise in global temperatures. The SOI is in general in phase with Australian rainfall on inter-annual time scales with the strongest correlations in south-east Australia (Power et al., 1999). There is a clear dependence between El Niño-Southern Oscillation (ENSO), rainfall and freshwater runoff (Power et al., 1999). The ability to forecast El Niño can provide a longer lead-time for developing strategies to deal with expected impacts - including those on fisheries economy.

1.2.2 Importance of estuarine habitats to fish

A number of paradigms have been developed to explain the importance and role of estuarine habitats for fish. Food and shelter have been cited as important services of mangroves, salt marsh and seagrass to fish (Duarte, 2000), as has been the diversity of habitats present in estuaries. Evidence for the protective role of estuarine habitats comes from studies showing that few large piscivorous fish enter dense vegetated areas at high tide (Blaber et al., 1989; Vance et al., 1996a; Connolly, 1999; Rönnbäck, 1999; Vance et al., 2002; Meager et al., 2005). This means, that smaller animals are therefore able to escape their predators by entering vegetated habitats. The structural complexity provided by submerged or inundated vegetation, shallow water and/or high turbidity waters can provide significant refuges from predators, especially for small nekton (Robertson and Duke, 1990; Robertson and Blaber, 1992; Rönnbäck, 1999; Nagelkerken et al., 2001). These characteristics are commonly found in a number of estuary-associated habitats, particularly mangroves, seagrass beds and salt marsh. Some details of the mechanism of how these factors, e.g., turbidity, work remain in debate (Baker and Sheaves, 2007). In addition to shelter it is thought that food for juvenile fish and crustaceans is more abundant in estuaries than in other coastal habitats (Hutchings and Saenger, 1987; Robertson and Blaber, 1992; Laegdsgaard and Johnson, 2001). Nutrients are brought into estuarine ecosystems from upstream terrestrial (freshwater inflows) and from marine (tidal mixing) sources and are concentrated by lateral trapping particularly within the vegetated areas (Wolanski et al., 2001). Sheltered, calm water leads to the retention of planktonic larvae and post-larvae (Rönnbäck, 1999), thereby increasing their chances of settlement.

The combination of food and shelter available makes estuaries important habitats sustaining fisheries. However, the extent of fish movement and detailed trophic linkages in estuaries remain unknown and the debate about the trophic role of estuarine habitat types for nekton continues (Beck et al., 2001; Dahlgren et al., 2006; Sheaves et al., 2006).

1.2.3   Estuarine habitats in Queensland

Queensland offers in abundance a variety of estuarine habitats for fish (Table 1), with significant contributions to the areas of mangrove, salt marsh and seagrass along the Australian coast. There are 39 mangrove species in Queensland, dominated in areal coverage by *Avicennia marina, Rhizophora stylosa* and *Bruguiera parviflora* (Tomlinson, 1986; Hogarth, 1999; Duke, 2006). Mangroves provide habitat for a large diversity of macrobenthos species, for example, sesarmid crabs (Frusher et al., 1994), prawns (Chong et al., 1990) and fish (Laegdsgaard and Johnson, 1995). Mangroves often share their habitat with salt marshes (low growing herbs, shrubs and grasses) (Bucher and Saenger, 1994). The habitat of Australian marshes is generally more stressful than that of their counterparts in the northern hemisphere, with greater fluctuations in salinity and higher maximum values, resulting in a much shorter stature of the marsh vegetation in Australia. Australian salt marshes nevertheless support a significant diversity of fauna dominated by molluscs and crustaceans such as the ocypodid crabs *Heloecius cordiformis* and *Uca* spp. (Morton et al., 1987). Seagrass beds are often found in close proximity to mangroves. Seagrass supports infaunal, epifaunal and epibenthic species with high variability in assemblage structure (Hemminga and Duarte, 2000). Common seagrass species in Queensland are *Halophila ovalis, H. spinulosa, H. ovata, H. decipiens, Halodule uninervis, H. pinifolia, Zostera capricorni* and *Cymodocea serrulata* (Lee Long et al., 1993).

Table 1: The approximate area of mangroves, salt marsh and seagrass along the Australian coast in km^2 (Bucher and Saenger, 1991; Hamdorf and Kirkman, 1995; Saenger, 1996)

State	NSW	VIC	TAS	SA	WA	NT	QLD	Total
Mangrove	107	41	0	111	1561	2952	3424	8195
Salt marsh	57	125	37	84	2965	5005	5322	>13 595
Seagrass	153	364	500	5,000	22,000	No data	23,200	>51,217

A few other important habitat types found in estuaries also contribute to the estuarine food webs and act as additional habitats for fish. These are, from the high to low tidal levels, (1) melaleuca swamps; (2) unvegetated mud flats; (3) submerged muddy and

sandy basins; and (4) open waters. These habitats are often closely connected both hydrologically and spatially to mangroves, salt marshes and seagrass.

1.2.4   *Estuarine habitat connectivity*

Estuarine habitats often coexist in the same catchment and are connected by water, creating a mosaic of interconnected habitats. Merriam (1984) first introduced the concept of landscape connectivity for terrestrial habitats, emphasizing the interaction between species attributes and landscape structure in determining movements of biota among habitat patches. As pointed out by Moilanen and Nieminen (2002), connectivity (or its inverse, isolation) has long been recognized as a fundamental factor in determining the distribution of species (MacArthur and Wilson, 1967; Fahrig and Merriam, 1985), but application of the concept to marine ecology has been limited. To date, there are no studies that have created a connectivity index for tidal wetlands that can facilitate analyses of their services to fish and fisheries. Habitat connectivity is important as it impacts on other ecological processes such as population dynamics, with strong implications for individuals, populations, and communities in heterogeneous landscapes.

Habitats within estuaries are connected primarily by water flow that drives the exchange of material and biomass and therefore, energy. The connectivity of estuarine habitats may influence the nearshore productivity by influencing availability of the services provided by various habitats to different life history stages of animals. There is broad functional interdependence among seagrass beds, marshes, mangroves and, if present, adjacent coral reefs (Kitheka, 1996; Marguillier et al., 1997). The correlation of mangrove and salt marsh area in estuaries also indicates a possible link between these habitats (Saintilan, 2004). The estuary-fish relationship should be investigated with a scale and perspective incorporating the significance of such interactions (Lee, 2004; Sheaves, 2005).

1.3   Fishery species and their life-history

How coastal fishery species utilise and are dependent on estuarine habitats is determined by their life-history patterns. Fish and invertebrates use estuarine and inshore habitats in many ways: some are only occasional visitors, some use them only at certain life-stages, while others reside permanently in the estuaries (Potter et al., 1990; Potter and Hyndes, 1999; Whitfield, 1999a; Whitfield, 1999b). These differences in life-history patterns may influence the nature of any interactions between species and their habitats. Some species use a range of environments, including offshore, inshore and estuarine regions. From an estuarine perspective, those that are found only occasionally in estuaries have been termed 'marine stragglers' (Potter et al., 1990; Whitfield, 1999a) and are regarded as having no direct dependence on estuaries. 'Marine stragglers' will not be further reviewed in this thesis. Examples of this group include mackerel (e.g., *Scomberomorus queenslandicus*) and coral trout (e.g., *Plectropomus maculatus*).

A second group of species, termed marine-estuarine species, use inshore areas and estuaries for significant periods of time, often during their juvenile phase. Several marine-estuarine species have juveniles that are only found amongst estuarine habitats (e.g., banana prawn - *Penaeus merguiensis*, (Staples et al., 1985; Vance et al., 1996b). Catadromous species travelling between freshwater and marine habitats use estuarine habitats at certain life-stages (e.g., barramundi - *Lates calcarifer*, (Russell and Garrett, 1983). Their dependence on only one type of habitat during one life stage is unlikely (Robins et al., 2005). Typically, the adults of these species spawn offshore but the eggs and juveniles develop in inshore and estuarine waters. The length of time spent in these habitats varies between species, between regions, and even between individuals, and also depends on environmental factors such as temperature, season, salinity and rainfall. This generalised life-cycle applies to a number of fisheries taxa, e.g., banana prawns (*Penaeus merguiensis*) (Dall et al., 1990), sea mullet (*Mugil cephalus*), whiting (*Sillago* spp.), flathead (*Platycephalus* spp.), leatherjackets (family Monacanthidae), black trevally (*Siganus* spp.), black bream (*Acanthopagrus butcheri*) and yellowfin bream (*A. australis*). Most economically important species are estuary-dependent but also mobile amongst habitat sites, localities and regions (Cappo et al., 1998).

A final grouping is the true estuarine species that complete their entire life-cycle within estuaries. These species are clearly estuary-dependent, but many are small and

short-lived, such as members of the Gobiidae and Atherinidae, (Potter and Hyndes, 1999; Whitfield, 1999a; Blaber, 2000). Few of them contribute significantly to fisheries and are therefore not represented in fish catch data. Nevertheless, they are important as a food source for many predatory fisheries species (Baker and Sheaves, 2007).

1.4   Research questions and framework

The main purpose of this study is to improve our understanding on the importance of habitat characteristics of estuaries and climate parameters on commercial fish catch, using Queensland as a case study. From both fisheries management and biodiversity conservation perspectives, it is essential to better understand how climate and habitat affect the distribution of fish within various estuary types (e.g., estuaries with different habitat connectivity) and broad geographic areas (Cappo et al., 1998; Blaber, 2000). My research will address the importance of estuaries for commercial fish species at the landscape scale and clarify the role of connectivity among habitats, an attribute that has been largely overlooked by most past studies (Beck et al., 2001; Dahlgren et al., 2006).

Identifying 'critical' or 'essential' habitats and climate dependencies is important, given the rate of anthropogenic coastal habitat modification and predicted climate change. The search for relations between variations in fisheries catches and variation in the physical and biological environment will generate testable hypotheses on the influence of ecosystem processes on fishery production. Furthermore, understanding such dependencies will allow assessment of their sensitivity to particular drivers such as climate change (Adam, 1990) and will help identify important attributes of estuaries for nektonic production along the coast of Queensland and beyond. This will help to ensure that an integrated approach of estuarine habitat conservation and the impact of climate change is considered in fishery management, supporting the move from conventional single-species-based to ecosystem-based management (NRC, 1999). This thesis will also provide guidance to estuarine habitat research along the coast of Queensland, by defining the estuarine habitat configuration that supports the highest fish catch. Specifically this study will address will address the following questions and issues:

(1) Knowledge gaps in the literature with respect to climate, habitat and fish catch relationships and how existing data sets may be refined to allow better estuarine-habitats fisheries analyses (Chapter 2);

(2) Key features of coastal habitats governing production (Chapters 3, 4);

(3) Whether some geographical regions are more strongly linked with fisheries production and offshore waters than others (Chapter 3, 4);

(4) The role of habitat connectivity in sustaining commercial fish species (Chapters 3, 4, 5);

(5) The influence of rainfall and temperature on fish catch (Chapters 6, 7); and

(6) Defining estuarine characteristics to be incorporated in the selection of protected areas (Chapter 7).

1.5   General methodology and data sources

There are different ways to approach the relationships between habitat wetlands characteristics of coastal ecosystems and fisheries production. The problem is the large amount of data, and reviews from which to make valid judgements. The following data packages will be used in the analyses (Appendix 1):

(1) Fishery catchment data (Department of Primary Industries & Fisheries – DPI&F);

(2) Distribution maps for mangroves, seagrass, salt marshes, channels, mud and sandflats and open water for Queensland (Environmental Protection Agency - EPA, DPI&F, Geoscience Australia, Ozestuaries) (Appendix 2);

(3) Rainfall, temperature and freshwater flow data (Bureau of Metereology - BOM, Department for Natural Resources and Mines - NRM);

(4) Aerial imagery, Light Detection and Ranging (LIDAR) data, pressure sensor data and fish movement data.

Starting with a literature review, Australian and worldwide studies presenting dependence of certain fishery species on climate variables and estuarine habitats are identified and validated (Chapter 2). On the basis of previous studies, preliminary analyses will be carried out using information on estuary characteristics (e.g., extend of mangrove, seagrass and salt marsh), which are available from existing survey data from the DPI&F in Queensland, and the Ozestuaries database (Digby and Ferguson, 1996; Heap et al., 2001).

The DPI&F Coastal Habitat Resources Information System (CHRISweb) database (http://chrisweb.dpi.qld.gov.au/chris/) was used to extract information on fish catch.

Monthly average catch values from 17 years (1988–2004) were used to minimise potential bias due to fluctuations and uncertainties in catch volume. Fish species were selected based on their biology and life history characteristics. The major focus of this investigation was on the marine-estuarine category that includes a number of economically important species. Non-estuarine species were included in the analyses as controls (Chapter 3). Estuarine habitat variables were then defined and factors influencing on their relation to fish catch were undertaken (Chapter 3). This was followed by a more detailed study with specific considerations of habitat connectivity and the development of various connectivity indices including the complete coastal fish catch data (Chapter 4). The geographical information system (GIS) software ArcGIS 9.0/9.1/9.2 were used to analyse, calculate, compare and visualise the databases, imagery and aerial photographs. Multivariate statistical techniques, including multiple regression analysis and non-metric multidimensional scaling (nMDS; Clarke and Warwick, 2001, biological environmental gradients (BIO-ENV) procedure (Clarke and Ainsworth, 1993) and principal components analysis (PCA) (see also linear PCA, Reyment and Joreskog, 1996) were carried out using the PRIMER 5.0 (Clarke and Ainsworth, 1993), and SPSS 12.01/14.01 and Fragstats 3.0 (McGarigal et al., 2002) were used to derive habitat parameters.

The outcomes of the fish catch data analyses identified estuary sites with high value for commercial fisheries. One of these estuarine systems was selected to investigate the role of habitat availability for a group of commercial and recreational estuarine fish species in detail. A study on small-scale movement of commercial important fish species among estuarine habitats, based on a simplified inundation model as well as artificial tagging, provided insights into the importance of connectivity within an estuarine scale. LIDAR-derived digital elevation model (DEM). This model, with vertical accuracy between 20-30 cm, was used to create an elevation model to model tidal wetland inundation by water. This was then linked with the fish movement data collected by a modified antenna system (Chapter 5). Finally in Chapter 6, coastal rainfall, temperature and river flow data from (BOM and NRM) were extracted and relationships with regional fish catch tested using regression and correlation analyses. This provided insights into factors other than habitat characteristics that drive fisheries production. The thesis close with considerations for fisheries management, and recommendations drawn based on the studies outcomes (Chapter 7) (Fig. 1, see also project framework Fig. 2).
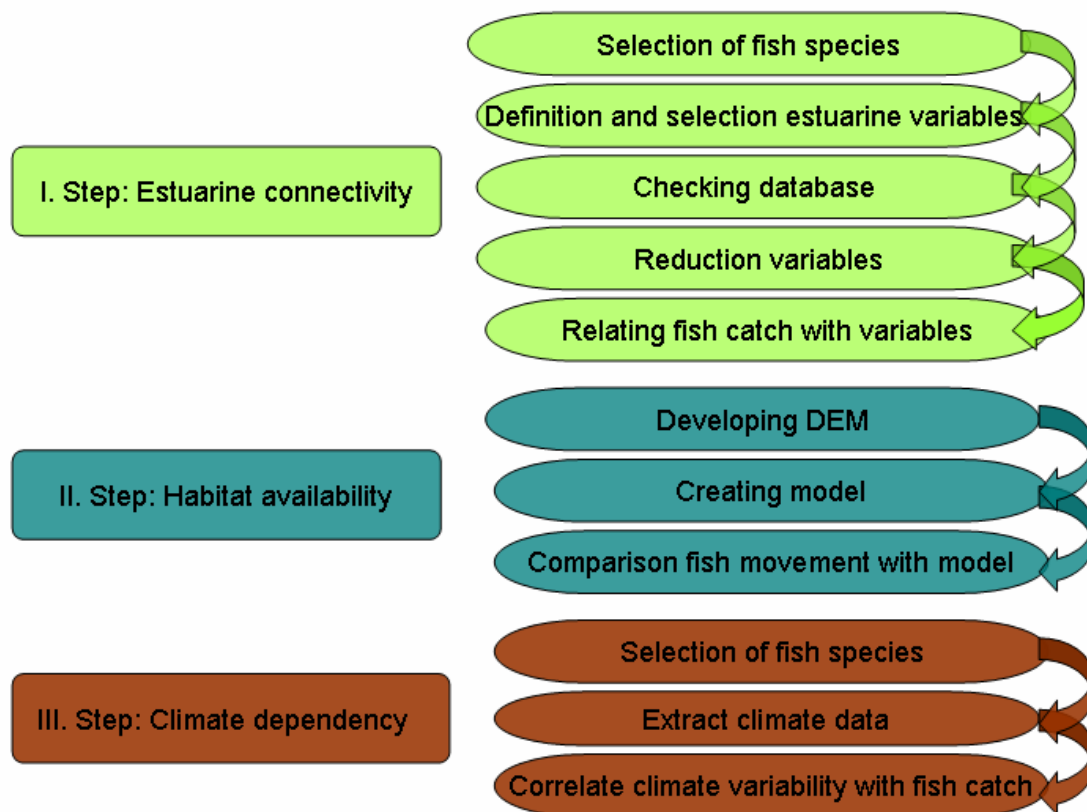
Figure 1: The three elements of this study and the tasks involved in progressing the investigation

Previous studies:
- correlation analyses
- classification analyses
- measurements of fish abundance regionalisation

"Relationship"

Estuaries ⟷ **Fisheries**

Fisheries species and estuaries
- which species use estuaries?
- what immigrations and life stages occur?
- are they dependent on estuaries?
- what habitats within estuaries are used?

Data needs:
- accurate habitat distribution data
- reliable fisheries data at estuaries scale and for certain species

Research needs:
- interactions between ecologists, fisheries scientists, spatial scientists and governmental agents

Research and data needs
- where are the gaps in knowledge?
- what spatial and temporal scales should be used?
- what are the data requirements to make the link between estuaries and fisheries?
- what are the research requirements?
- can predictions be made about catch?

Important values of estuaries for fish
- why do fish species use estuaries?
- what are the important drivers for fish catch?
- what type of estuary supports most fish production
- what is the role of habitat and habitat connectivity?

Management of estuarine habitats
- where are future protection zones?
- required type of protection

Estuarine habitat values for fish:
- refuge
- food availability
- shelter from physical disturbance

Potential responses to climate changes for fish catch
- is fish catch influenced by rainfall and temperature?
- what are potential effects of climate induced habitat changes?

Habitat changes:
- mangrove and saltmarsh shift
- saltmarsh loss and mangrove gain
- climate induced or not?

Figure 2: A schematic illustration of the project framework

CHAPTER 2

ESTUARINE HABITATS AND CLIMATE AS DRIVERS OF NEARSHORE FISHERY PRODUCTION – A REVIEW

2.1   The role of estuaries in nearshore fisheries

In the last four decades, many studies have demonstrated a strong empirical relationship between estuarine habitats (in particular, mangrove abundance) and fish catch (Turner, 1977; Pauly and Ingles, 1986; Yáñez-Arancibia et al., 1993; Lee, 2004; Manson et al., 2005b), with catch being influenced by the abundance of estuarine habitats in a region. Numerical correlations have also been found between the extent (area or linear extent) of mangroves and the catch of prawns (e.g., banana prawn – *Penaeus merguiensis*) in fisheries adjacent to mangroves (Turner, 1977; Staples et al., 1985; Pauly and Ingles, 1986; Baran et al., 1999). Such studies provide important information on the fisheries–mangrove relationship and often form the basis for the economic valuation of mangroves (e.g., Barbier and Strand, 1998; Grasso, 1998; Barbier, 2000).

The observed relationship is mainly derived from the group of economically important fishery species classified as either estuary-dependent (Cappo et al., 1998) or (non-estuarine) bay-habitat-dependent (Nagelkerken et al., 2000b). Mangroves and other estuarine habitats are the principal habitat for at least a part of their life cycle (Blaber et al., 1989; Nagelkerken et al., 2000b). Typically, the adults spawn offshore, producing eggs that disperse in the water column for varying lengths of time. The eggs then develop into planktonic larvae which migrate, or are carried by currents, into inshore and estuarine waters. The subadults or adults migrate out of the estuary or lagoon, and back towards the offshore areas or adjacent coral reefs upon sexual maturity to complete their life cycle. Therefore, estuarine habitats could function as an important link in the chain of habitats that provide essential resources and benefits, e.g., as nursery areas for fish, prawns and crabs (Sheridan and Hays, 2003; Crona and Ronnbäck, 2005), with spatial complexity at a scale that provides refuge from predators, and abundant food for commercial species at certain stages in their life cycle (Chong et al., 1990).

2.2   The origin of the estuary-nearshore fisheries link

It was once believed that estuarine habitats, in particular, mangrove and/or salt marsh, fuelled offshore fisheries production through the 'outwelling' of nutrients (Odum and Heald, 1972; Robertson and Blaber, 1992; Lee, 2005). This theory laid the foundation for the estuarine habitat-fisheries link. Almost 40 years ago Odum (1968) first presented the idea of outwelling, following from the hypotheses of Schelske and Odum (1962) that estuaries are highly productive, and that temperate salt marshes might export significant percentages of their organic production offshore to support secondary production (Teal, 1962). A comprehensive review of the theory of outwelling and the empirical data from flux measurements conducted in salt marsh ecosystems was given by Nixon (1980). He also compared coastal marine fisheries in North America for evidence of an energy subsidy from estuaries (of which 10 % of the area were intertidal marshes) but found only a weak relationship.

The movement of nutrients between estuarine and coastal systems is a complex and dynamic process (Ford et al., 2005; Webster et al., 2005). The general model of outwelling is too simplistic (Kneib, 1997; Ford et al., 2005) as a mechanism explaining the diversity and variation of coastal fisheries. Recent studies using stable isotopes and other trophic tracers indicate that offshore fisheries may not be driven through the outwelling of nutrients from estuaries (Loneragan et al., 1997; Connolly, 1999; Chong et al., 2001). For example, mangrove-derived nutrients only contribute directly to the food webs within highly restricted local areas, e.g., mangrove-lined creeks (Loneragan et al., 1997). Separating the contribution from estuarine producers, i.e., mangroves, seagrass and salt marshes, using casually deployed tracers, such as natural-abundance stable isotope analysis, is still problematic because of the large number of potential sources, some with overlapping 'signatures' (Lee, 2005).

Although some studies have documented greater abundances of juvenile fish and crustaceans in mangroves than in other estuarine and inshore habitats (Robertson and Duke, 1990; Nagelkerken et al., 2002), other studies found a significant contribution by salt marshes (Connolly, 1999). Controversy also surrounds the measurement of nursery ground values (Sheridan and Hays, 2003). Beck et al. (2001) and Adams et al. (2006a) gave detailed discussions of the nursery habitat concept, where the value of nursery grounds is measured in terms of numbers of juveniles raised in the nursery contributing

to adult populations: either the average number of individuals per unit area (Beck et al., 2001) or the number of individuals per habitat (Dahlgren et al., 2006; Layman et al., 2006), as an attempt to simplify the measurement of nursery function and to identify core components (e.g., habitats) that are most important in maintaining overall ecosystem function. Sheaves et al. (2006) argued that this approach is over-simplistic and relates to the value of a nursery from a short-term, fisheries perspective. Since the value of estuaries, whether acting as a source of organic production or nursery, rests upon the collective presence of the associated habitats, it is important to develop methods that would improve the measurement of habitat connection, how these connections function, and how they can be maintained. There is no single habitat that can adequately explain the contribution of estuarine productivity to nearshore fisheries.

2.3   Linking fish catches with estuaries

Numerous studies on the estuary-fishery connection are based on understanding estuarine productivity and how estuaries function as nursery grounds for fish. It is estimated that over two-thirds of the world's harvest of fish and shellfish are directly linked to estuarine habitats (Robertson and Blaber, 1992). Rönnbäck (1999) listed the proportion of estuary-related species in fisheries around the world: e.g., Florida (80 %), Fiji and India (60 %), eastern Australia (67 %), Malacca Strait (49 %) and south-east Asian countries (fish catch – 30 %; prawn catch – nearly 100 %). In Malaysia, it was estimated that 32 % of the 1981 fish biomass (total fish catch) could be linked to mangroves, while in the Philippines, ~ 72 % of the catch between 1982 and 1986 was associated with mangroves (Paw and Chua, 1991). In Australia, estuarine habitats, such as mangroves, seagrasses and salt-water channels, are critical to ~ 75 % and 70 % of commercial fish and crustacean species in Queensland (Quinn, 1992) and New South Wales (Pollard, 1981), respectively. The values are lower for south-western Australia (20 %) and Australia as a whole (32 %) (Lenanton and Potter, 1987). A review of the literature by Baran and Hambrey (1998) on estuaries and fish catch demonstrated the dependence of fish harvests on estuarine environments, and showed the importance of these systems in terms of sustainable management of the coastal resource. Rönnbäck (1999) and Barbier (2000) identified and synthesised ecological and biophysical links of mangroves that sustain seafood production from an economic perspective.

The most direct approach to searching for links between estuaries and fisheries is to correlate fish catch data with estuarine parameters. Manson et al. (2005b) found a significant influence of mangrove forest characteristics (e.g., perimeter and area) on the abundance of mangrove-related and estuarine fishery species, while latitude was the only variable influencing catch of offshore species along the north-eastern Australian coast. The earliest study in this field was Macnae (1974) who showed that inshore fish production in Malaysia was related to mangrove area. Turner (1977) found a positive correlation between penaeid shrimp catches and surface area of vegetated estuaries in the Gulf of Mexico. Martosubroto and Naamin (1977), working in Indonesia, showed a positive correlation between annual fish landings and mangrove area. These authors also opined that a certain minimum mangrove area is necessary for high production, a point also noted by Pauly and Ingles (1986) who suggested that the impact of destroying a mangrove area might be greater if the area is small and residual. Through a global analysis Lee (2004), however, suggested that the amount of intertidal habitats, rather than just the area of mangroves functions as a major driver for prawn production. As tidal amplitude increases, the extent of intertidal habitats such as mudflats increases, so will the flux of material (Lee 2004). The linear extent of mangrove-lined estuaries has been used as an alternative index of the available mangrove nursery for penaeid prawns, rather than total area (Staples et al., 1985). Linear extent was regarded as a better index of available habitat than total area, because prawns access the mangrove forest through the mangrove-water interface as they move in and out with the tide (Vance et al., 1996a; Vance et al., 2002). Recent studies undertaken in Australia by Saintilan (2004), similar to Pease (1999), found a strong correlation between fish catch and seagrass as well as mangroves and salt marsh area for New South Wales, Australia (Potter et al., 1990; Pollard, 1994). Strong dependencies on estuarine habitat characteristics for some commercial species have been reported in Queensland, Australia (Manson et al., 2005b).

The review presented here found a total of 27 studies (Fig. 3 and Table 2) quantifying the relationship between commercial catch and estuarine habitats, in particular mangroves in tropical and subtropical zones, over the last four decades. Most of the analyses in these studies have resulted in positive correlations, suggesting that the area of mangroves or tidal wetland habitats directly translates to commercial fish catch (Manson et al., 2005b). Estimates of the amount of variance in commercial catch explained by the presence of mangroves or estuaries range from 20 % to 90 % (Fig. 3).

The most common predictor used was mangrove area, followed by linear extent and intertidal area or estuary size. Over 15 studies used mangroves as a predictor and seven studies used the extent of estuaries, coastal vegetation or shallow water. Most studies were undertaken in Australia, Asia and the U.S.A., whereas West Africa and South America were conspicuously under-represented (see also Faunce and Serafy, 2006, in their review on mangroves–fish studies). The r^2 values reported in studies covering worldwide
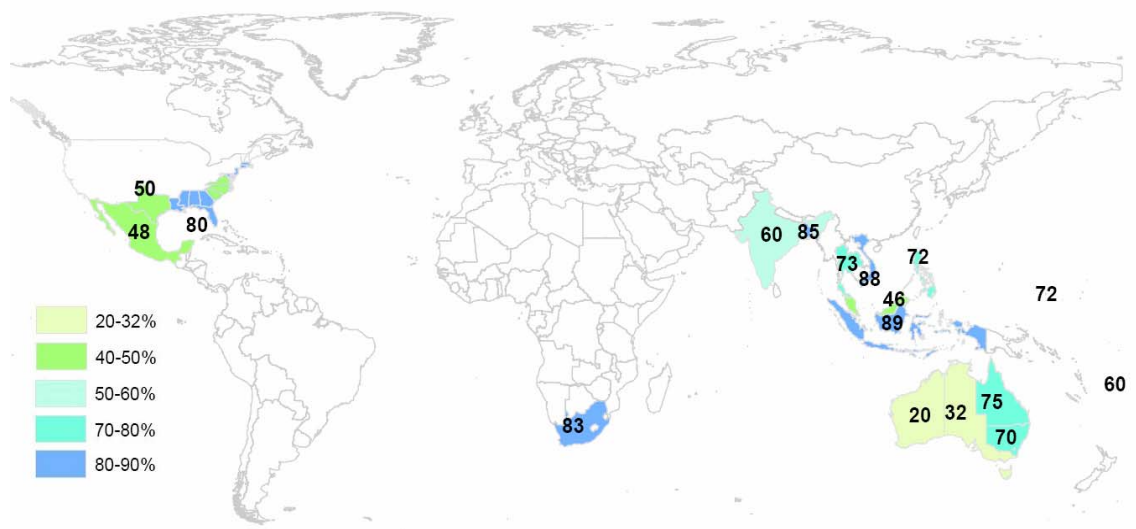
Figure 3: **World map showing the percentage of variance in commercial fish catch explained by either mangroves or estuarine habitats based on 27 reviewed studies, of which 22 provided quantitative data.** For countries where more than one study was conducted the conservative value is chosen for display (for a detailed listing see Table 2)

tropical commercial catch ranged between 21 % (Houde and Rutherford, 1993) and 30 % (Naylor et al., 2000) for an estuarine–fish catch relationship, 38 % for a mangrove-prawn catch relationship (Lee, 2004), 53 % for a mangrove area–fish catch relationship (Pauly and Ingles, 1986), and 54 % for an intertidal wetlands–prawn catch relationship (Turner, 1977). The strength of the correlation usually bears an inverse relationship with the geographical extent covered by the analyses. Most studies on estuarine fish communities and their linkages with nearshore fisheries have been done on mangrove

habitats (Table 2). Estuarine or lagoonal habitats and the strong links between different habitat types have been neglected in past studies (Sheridan and Hays, 2003). A certain combination of habitats and their accessibility is likely to explain the importance of estuaries to nekton (Cappo et al., 1998). Only in the last decade have studies started focusing in more detail on the connectivity between mangroves, seagrass beds, salt

Table 2: Overview of studies investigating relationships between estuarine habitat abundance and fisheries production for prawn (+), or fish (++), or both (with no superscript) in the last three decades. Nr - Not reported. The r^2 values indicate goodness of fit of a linear model between habitat and catch (Baran, 1999; Manson et al., 2005a)

r^2 (n)	Region	Variable	Reference
Positive	Malaysia	mangrove area	Macnae, 1974
0.54 (27)[+]; 0.64 (14)[+]	World, tropical	intertidal area	Turner, 1977
0.89 (nr)[+]	Indonesia	mangrove area	Martosubroto and Naamin, 1977
0.58 (6)[+]	Gulf of Carpentaria, Australia	linear extent of mangroves	Staples et al., 1985
0.48 (10)[++]	Gulf of Mexico	coastal vegetation area	Yáñez-Arancibia, 1985
0.53 (nr)[++]	World, tropical	mangrove area	Pauly and Ingles, 1986
Positive	Philippines	mangrove area	Camacho and Bagarinao, 1987
0.89 (10)[+]	Peninsular Malaysia	mangrove area	Sasekumar and Chong, 1987
0.32 (nr)[+]	U.S.A	salt marsh length interface	Browder et al., 1989
0.61 (18)[+]; 0.66 (18)[+]; 0.34 (15)[++]; 0.88 (5)[++] 0.53 (18); 0.40 (20); 0.66 (12); 0.40 (18); 0.4 (34); 0.45 (39); 0.95 (nr)	Philippines	mangrove area	Paw and Chua, 1991
0.95 (nr); 0.88 (5)[++]	Vietnam	mangrove area	de Graaf and Xuan, 1998
Positive[++]	Philippines	mangrove area	Gilbert and Janssen, 1998
0.38 (37)[++]	World, tropical	coastline length, mangrove area, tidal amplitude	Lee, 2004
0.32-0.75 (49)	NewSouthWales Australia	total area of mangrove, salt marsh, seagrass	Saintilan, 2004
0.46-0.63 (8)	Malaysia	mangrove area	Loneragan et al, 2005
0.37-0.70 (36)[+]; 0.57-0.77 (36)[++]	East coast Queensland, Australia	mangrove perimeter, area shallow water, mangrove area/length coast line	Manson et al., 2005b

marsh, tidal channels and coral reefs with regard to fish movement (Nagelkerken et al., 2000c). This has resulted in the identification of several (commercial) reef fish species that appear to depend on mangroves as juveniles (Cocheret de la Morinière et al., 2002; Christensen et al., 2003; Serafy et al., 2003; Eggleston et al., 2004; Dorenbosch et al., 2007). Otolith microchemistry studies have also suggested a linkage between mangroves and coral reefs (Chittaro et al., 2004) through ontogenetic fish movement.

Further evidence came from studies undertaken in the Caribbean where permanently inundated fringing mangroves primarily serve as daytime refugia for a major component of fishes occupying various habitats in lagoons or bays (Nagelkerken et al., 2000b; Valdés-Muñoz and Mochek, 2001), while fishes from adjacent habitats feed in large intertidal mangroves at high tide (Sheaves, 2005; Lugendo et al., 2007). This suggests for some species that fish production attributed to mangroves may not necessarily be derived from single habitats alone. Evidence exists, for example, that fish abundance and species richness are higher when mangroves and seagrass beds occur together rather than in isolation (Robertson and Blaber, 1992; Nagelkerken et al., 2001; Dorenbosch et al., 2006; Jelbart et al., 2007). Many fish species occupying estuaries appear to show long-term ontogenetic and short-term behavioural movements (Chapter 5) between seagrass beds, salt marsh, tidal channels and mangroves, making the individual contribution of habitats difficult to determine (Adams et al., 2006a).

My review showed that the relationships between commercial catch and estuarine habitats vary widely, depending on species, location and time scale, indicating that the link is more complex than a linear function. The predictors used in the regression analyses are themselves strongly correlated, and catch statistics are often not well delineated. Logbook information is being collected at different spatial and temporal resolutions and species groupings depending on the country. There is high variation within the data sets (mangrove forest distribution, commercial records, effect of stock size and fishing pressure) and difficulty in distinguishing links against a background of highly variable temperature, rainfall, ocean currents, and fishing effort. A number of gaps in regards to fish catch–habitat studies were identified from the literature review, which should be considered in future studies: (1) mapping of intertidal vegetation was done at broad spatial scales (often greater than 1:100,000), which are likely to over-estimate vegetation occurence; (2) the scale of fisheries data generally did not match the spatial scale of estuaries, or the data have not been available; (3) the collection method

of fisheries catch data has been prone to under-reporting and, occasionally, over-reporting; (4) misleading statistical methods; (5) groups of species tended to be included in the relationship irrespective of whether their life-histories were linked to estuaries; and (6) focusing on only one habitat type in the analysis. These observations suggest that future studies need to be conducted at suitable scales, require better data definition and a holistic approach to measuring and presenting estuarine habitat characteristics, such as habitat connectivity.

2.4   Linking estuary-dependent fish species with climate variables

One problem in linking fisheries production to estuarine productivity is the difficulty in distinguishing links against a background of large variability in temperature, humidity and other variables. For instance, tropical estuaries in Australia experience distinct wet and dry seasons. Extreme land-estuary-sea variations and connections have rarely been examined, but the life cycles of many estuary-dependent organisms, such as penaeid prawns, are cued to weather changes (Vance et al., 1996a; Vance et al., 1996b). Prawn yields are highest just before and during the wet season, but correlations with environmental cues (e.g., rainfall and temperature) are not simple (Staples et al., 1995).

Efforts have been made to understand linkages between physical phenomena such as El Niño events in the ocean environment and biological processes that influence commercially important fish stocks (Evans et al., 1995; Schwartzlose et al., 1999; Lea, 2000; Byrne et al., 2002; Eide and Heen, 2002; Currie and Small, 2005). Evans et al. (1995), for example, found links between catch-per-unit-effort (CPUE) and sea temperature for the Bermuda spiny lobster (*Panulirus argus*). Eide and Heen (2002) discussed possible economic effects of global warming on the fish resources in the Barent Sea. Byrne et al. (2002) examined environmental parameters and variation in Sea Trout smolt (*Salmo trutta*) migration using multiple regressions, showing that *S. trutta* catch was related to water level, temperature and hours of sunshine. It is also known that El Niño events in the eastern Pacific have a negative impact on shrimp farming (Rosenberry, 2004).

In the last 30 years, studies in Australia and elsewhere predominantly on prawns and estuary-dependent species suggested a sensitivity of these species to freshwater runoff, leading to fluctuation in fish catch (Table 3). Available evidence demonstrated that river

flow is a critical factor in maintaining nutrient and detrital input to estuaries, as well as preventing the development of hypersaline conditions within these systems (Forbes and Cyrus, 1993; Whitfield, 1994). Knowledge of the freshwater flow requirements of fisheries was based on the analysis of catch (= landings) and freshwater flow data. Loneragan and Bunn (1999) for example showed that total fish catch corresponded with freshwater runoff ($r^2 = 0.72$; $P < 0.01$) for the Logan River, Queensland, Australia, in particular for mullet (*Mugil* spp.) and flathead (*Platycephalus* spp.). Furthermore, there is much anecdotal evidence that higher rainfall in the wet season translates to good prawn seasons. Simple models based on rainfall, monthly run off data and monthly sea surface temperature (SST) are used for predictions in the Gulf of Carpentaria prawn fishery, as emigration rates of juvenile banana prawns from estuaries to nearshore areas are 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). Most correlations between freshwater flow, rainfall and prawn catch have been reported for species with the greatest tolerance or exploitation of brackish-water habitats. In general, significant positive relationships occur between annual catch and total rainfall (or freshwater flow) in the same or previous year (Gunter and Hildebrand, 1954; Ruello, 1973; Glaister, 1978; Vance et al., 1985; Gammelsrod, 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; Vance et al., 1985; Gammelsrod, 1992; Evans et al., 1997; Vance et al., 1998). Other studies found a correlation between evaporation and barramundi (*Lates calcarifer*) fish catch for Cairns, Queensland, Australia (Jaqueline Bolston, pers. communication) and between freshwater runoff and barramundi catch (Staunton-Smith et al., 2004). Genetically discrete stocks for barramundi were found to exist in different groups of river systems in northern Australian waters (Quinn, 1987; Watts and Johnson, 2004) and of mud crabs (*Scylla serrata*) between the Gulf of Carpentaria and the Queensland east coast (Gopurenko and Hughes, 2002). Such differences could be related back to climate adaptation.

The effects of temperature on fisheries appear more complex. Underlying mechanisms (e.g., initiation of breeding cycles in mud crabs) may be controlled by trigger values, which are more difficult to detect in broad spatial and temporal analyses (Smith, 1990b). Strong links between temperature and fish catch have not been reported

for tropical Australia so far. However, similar dependencies as for rainfall may be expected.

This review shows that the relationships between commercial catch and climate variables are highly variable depending on the region and species selected. However, most studies were able to detect dependencies using simple linear regressions. Similar problems and issues arise from this statistical method as for the fisheries and estuarine habitat studies. The relationship between fish catch and rainfall and/or freshwater runoff were often carried out on single species for single estuaries or regions, suggesting that broad-scale investigations could give further insights on the relationships.

Table 3: Studies showing a positive relationship between catch of fishery species and environmental factors

Author	Factor	Fishery species	Region
Gunter and Hildebrand (1954)	Rainfall	White shrimp (*Penaeus setiferus*)	USA, Texas
Ruello (1973); Glaister (1978)	Rainfall	School prawn (*Metapenaeus macleayi*)	Australia, Hunter River
Vance et al. (1985); Staples and Vance (1986); Vance et al. (1998)	rainfall, freshwater runoff	Banana prawn (*Penaeus merguiensis*)	Australia, Gulf of Carpentaria
Smith (1990)	rainfall, temperature	Blue crab (*Callinectes sapidus*)	USA, Louisiana
Gammelsrod (1992)	Rainfall	Red-legged banana prawn (*Penaeus indicus*)	Mozambique
Loneragan and Bunn (1999)	freshwater runoff	Mullet *(Mugil* spp.); flathead (*Platycephalus* spp.); tiger (*Penaeus semisulcatus*), school (*Metapenaeus macleayi)*, greasy (*Metapenaeus bennettae)* and king prawns (*Penaeus esculentus, P. plebejus*)*;* mud crab *(Scylla serrata)*	Australia, Logan River
Galindo-Bect et al. (2000)	Rainfall	Blue shrimp (*Litopenaeus stylirostris*)	Mexico, Gulf of Calif.
Browder et al. (2002)	Rainfall	Pink shrimp (*Penaeus duorarum*)	USA, Gulf of Mexico
Powell et al. (2002)	freshwater runoff	Brown shrimp (*Penaeus aztecus*)	USA, Texas
Staunton-Smith et al. (2004)	freshwater runoff	Barramundi (*Lates calcarifer*), School prawns (*Metapenaeus macleayi*)	Australia, Fitzroy River
Growns and James (2005)	freshwater runoff	Australian bass (*Macquaria novemaculeata*)	Australia, Hawkesbury River

2.5   Discussion and conclusion

There are strong links between nearshore fish catch and estuarine habitat parameters, as well as climate elements such as rainfall and temperature. Studies indicated that climate and habitat variables were the strongest drivers influencing fish catch and with it, fish abundance. However, there are as yet few attempts to relate widespread collection of intertidal and subtidal habitat data in inventories, for example, of the Australian coast (e.g., the Australian coast: NatMIS, CYPLUS, Ozestuaries) or long-term climate data (e.g., from the BOM) to fisheries production. There is also little effort in trying to turn survey data into biologically meaningful patterns that would benefit coastal management and fisheries research.

A number of gaps in regards to fish catch–habitat and climate studies were identified from the literature review. The review reveals that studies of estuarine-fisheries links have used a variety of data sources, including topographic survey maps (Staples et al., 1985), forestry and soil-use maps (Macnae, 1974), and navigational charts (Macnae, 1974). These data sources ranged in scale from 1:100,000 to 1:2,500,000, risking misinterpretation of vegetation distribution. Advances in pursuing habitat inventories include the development of methods for classification and assessment (Blackman et al., 1993), and the use of GIS to store information and make interpolations (Johnston and Barson, 1993). GIS has been a key platform in mapping estuarine habitats in some areas of Australia (Ahmad and Hill, 1994; Derbyshire et al., 1995), further suggesting increased use of GIS for any further estuaries–fish catch studies.

Another problem identified in this review is that logbook collection methods vary among countries, with information being collected at different spatial and temporal resolutions, species groupings and level of reliability of recording. In some cases, catches are likely to be under-reported by fishers in order to avoid tax payments, to circumvent quotas and/or to conceal fishing in protected or closed areas (Turner, 1977; Watson and Pauly, 2001). Production considerations of effort and changes in efficiency (e.g., monofilament nets, electronic navigation and fish-finding devices) are often difficult to quantify. Over time most fisheries experienced increasing legislative restrictions on gears, areas and times of fishing (Stergiou and Pollard, 1994; Pease, 1999). Other factors such as stock recruitment relationships, habitat changes, pollution impacts, competition and predation influence the fish catch data and are not only

difficult to quantify but they can also change quickly over time (Allen and McGlade, 1986). A difficulty in interrogating Australian production figures in isolation is the lack of knowledge about the harvest by recreational anglers. The harvest by anglers is at least as large as commercial production in some important estuaries and bays (Hancock, 1995; Cappo et al., 1998). All this has led to the assumption that commercial fisheries data do not generally reflect actual fish distribution. Starr and Fox (1996), however, found that at least for individual species, the data recorded by commercial fishers via logbooks was no more biased in terms of reflecting actual fish distributions than those collected through scientific fishery surveys (Duffy et al., 2003). Recent analyses of estuaries and fish catch data have demonstrated the potential of data sets available for Australia (Pease, 1999; Saintilan, 2004).
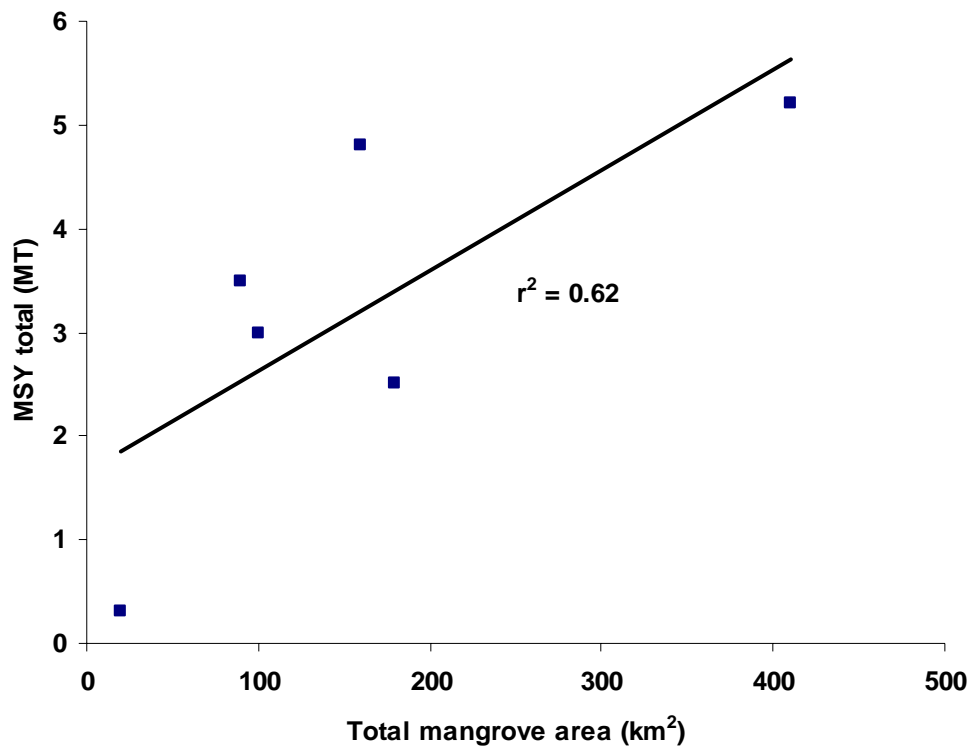
Figure 4: **An example of the traditional view on the relationship between mangrove area and fishery catch: annual maximum sustainable yield (MSY) of penaeid shrimp and mangrove area in the Philippines (redrawn from Pauly and Ingles, 1986)**

Other confounding factors in fish catch studies include spurious correlation in the data sets. Most estuary-fisheries studies used linear correlation analyses (Fig. 4), which require a number of considerations, such as (1) non-linearity of linking mechanisms (Baumann, 1998) and the probability of multiple mechanisms, for example, rainfall events can trigger fish catch but my have negative influence on recruitment in the long term; (2) possibility of type I errors (i.e., false significant correlations) (Potter et al., 2001); and (3) uncertain predictive capability as a consequence of long-term climatic variation or human-induced changes that counteract linear relationships (Robins et al., 2005).

Misleading outcomes from estuary-fish catch studies also came from the inclusion of groups of species without reference to the degree of dependence of their life-histories on estuaries (Pauly and Ingles, 1986). With species composition of the catch being available for some data sets, this problem may be overcome. Analyses between estuarine characteristics and catch may not be appropriate for species that undertake extensive migrations (e.g., mackerel – *Scomberomorus commerson*) (Robins et al., 2005).

A further significant problem in previous studies was the attempt to assess the estuary-fishery relationship with catch data concentrating on single habitats, such as mangroves (Sheridan and Hays, 2003) (see also Table 2, Fig 3). However, in order to maintain fisheries production, the combination of estuarine habitats that are utilised by different species and their various life history stages must be taken into account to ensure that all the resources required by these species are included in analysis (Lee, 2004). Further, attempts to investigate the ecological services of estuarine ecosystems must be conducted at the correct spatial and time scales. Current research on individual estuarine habitats, i.e., mangroves, salt marsh and seagrass is important but their contributions are likely linked rather than being offered to nektonic species in isolation. There is little direct evidence of a positive relationship between estuaries and their habitats and fishery catch on a national and global scale (Lee, 2004) suggesting the need for further research. Future investigations of the tidal wetlands–fisheries link should be based on an understanding of habitat connectivity and should employ sound data collection methods. In addition, more research is required to document the movement and utilisation of different estuarine habitats by fishes on the scale of estuaries to better understand their utilisation of estuarine habitats.

CHAPTER 3

PRELIMNARY INVESTIGATIONS ON THE RELATIONSHIPS BETWEEN ESTUARINE HABITATS AND COASTAL FISHERIES IN QUEENSLAND

3.1   Introduction

Estuaries play an important, often essential, role in the life histories of many aquatic organisms (Blaber, 2000), including fish species of importance to indigenous, commercial and recreational fishers (Dunning et al., 2001). Australia's total annual economic value of the fishery industry in 2002 was A\$7.4 billion (Williams, 2002) including the gross value of production (GVP) of A\$2.3 billion (ABARE, 2005).

The importance of estuarine habitats rests not only with their extent, but also their combined occurrence and relation to each other. Attributes of the estuarine 'seascape' may also be critical for sustaining productivity of coastal fisheries. I expect that estuary-dependent species catch would increase with connected estuarine habitats. Previous studies have found positive correlations between mangrove extent and fisheries catch (Baran, 1999; Manson et al., 2005b) only a few have examined multiple habitats (Saintilan, 2004; Lee, 2005). Most estuary-dependent species use a wide range of habitats during their life history. Not all habitats contribute equally as nurseries (Beck et al., 2001), but comparison of their values has rarely been rigorously undertaken.

The purpose of this chapter is to demonstrate the importance of estuarine habitat characteristics on commercial fish catch, with particular focus on the role of mangroves, salt marsh, seagrass and mud- and sandflats as connected habitats driving observed fluctuations in fish catch. The connectivity of habitats can be a key factor in fish production by allowing exchange of energy and organisms (Merriam, 1984) among estuarine habitats. This also includes 'edge effects', which reflect changes in ecological factors at the boundary between habitats, thus supporting the 'chain of habitats' concept as a useful management system for Australian and worldwide species (Nagelkerken et al., 2001). Here I: (a) present information available for assessing the linkage between

estuaries and commercially important fish species; (b) provide a case study relating combined occurrence of Queensland estuarine habitats with fish catch data using multiple regression models and nMDS; and (c) discuss ways of assessing the data and benefits for fisheries management.

3.2   Methods

3.2.1   Data collection

Data on catch, effort (number of days and boats) and gross value of production for estuary-dependent species or species groups were provided by DPI&F Assessment and Monitoring Unit (Table 4). This data set is based on daily logbook records reported by commercial fishers providing details of their catch and effort, covering the years 1988-2004, and recorded in half-degree grids (30-nautical-mile) for the entire coast of Queensland. Data from each of the four fisheries (trawl, line, net and pot) can be distinguished. The trawl fishery for prawns has components: a within-estuary (river) beam-trawl fishery and an offshore (and coastal foreshore) otter-trawl fishery. However, the Gulf of Carpentaria has only an offshore trawl fishery (Staunton-Smith et al., 2004). The fish catch data have an estimated error of at least 10 % due to the type of recording, market fluctuations, policies and management changes (L. Olyott, DPI&F, pers. com.).

Data on Queensland coastal wetland vegetation were obtained from DPI&F Assessment and Monitoring Unit. The 1:100,000 coastal wetland vegetation map includes information on mangrove communities, salt marsh and open water based on Landsat TM images and ground truthing (1987-1999). Data on channels, intertidal flats and sandflats for estuaries were taken from Geoscience Australia (Geoscience Australia, 2004). This information was primarily sourced from the 1:100,000 scale National Topographic Map series produced by Geoscience Australia.

3.2.2   Fish catch data

The coast of Queensland was divided into 13 separate sections for the purpose of this study. These sections were selected because of their importance to the commercial

Table 4: Selected species for the analyses with the total catch of 13 selected geographical areas along the coast of Queensland from 1988-2004.

Habitat dependence as described in Williams (2002). Channels and Salt marsh are not included. FL - Mud- and Sandflats, SG - Seagrass, MG - Mangrove

Fish catch class	Taxa	Habitat	Catch (t)
Barramundi	*Lates calcarifer* (Bloch, 1790)	MG	8 212
Bream	*Monodactylus argenteus* (Linnaeus, 1758), *Pomadasys maculatum* (Bloch ,1793), *Acanthopagrus australis* (Akazaki, 1984), *A. berda* (Forsskal, 1775), *Plectorhinchus gibbosus* (Lacepède, 1802), Sparidae spp.	MG, FL, SG	10 870
Bugs	*Thenus indicus* (Leach, 1815), *T. orientalis* (Lund, 1793)	FL	2 921
Blue Swimmer Crab	*Portunus pelagicus* (Fox, 1924)	FL	5 277
Mud Crab	*Scylla serrata* (de Haan, 1833)	MG	513
Dart	*Trachinotus anak* (Ogilby, 1909), *T. blochii* (Lacepède, 1801), *T. botla* (Shaw, 1803), *Trachinotus* spp.	FL	1 025
Flathead	*Platycephalus fuscus (*Cuvier, 1829*), Platycephalus* spp.	SG, FL	6
Flounder	*Pseudorhombus jenynsii* (Bleeker, 1855), *P. arsius* (Amaoka, 1969)*, P. spinosus* (McCulloch, 1914)	FL	465
Grunter	*Hephaestus fuliginosus* (Macleay, 1883), *Pomadasys* spp.	FL	14
Milkfish	*Chanos chanos* (Forsskål, 1775)	MG	7 954
Mullet	*Liza vaigiensis* (Quoy & Gaimard, 1825), *L. subviridis* (Valenciennes, 1836), *L. argentea* (Quoy and Gaimard, 1825), *Valamugil georgii* (Bleeker, 1858)*, Valamugil seheli* (Forsskål, 1775*), Mugil cephalus* (Linnaeus, 1758), *Parupenaeus* spp. *Trachystoma petardi* (Castelnau, 1875), Mugilidae spp.	FL	28 447
Prawns-Bait	Family Penaeidae	-	152
Prawns-Banana	*Fenneropenaeus indicus* (Milne-Edwards, 1837)*, F. merguiensis* (de Man, 1888)	MG, FL	7 801
Prawns-Bay	*Metapenaeus macleayi* (Haswell, 1879)*, M. insolitus* (Racek & Dall, 1965)	-	5 906
Prawns-Endeavour	*Metapenaeus endeavouri* (Schmitt, 1926)*, M. ensis* (De Haan, 1844)	SG	11 803
Prawns-Greasy	*Metapenaeus bennettae* (Racek & Dall, 1965)	MG, FL	1 415
Prawns-King	*Penaeus esculentus* (Haswell, 1879)*, P. plebejus* (Hess, 1865)	SG	27 389
Prawns-School	*Metapenaeus macleayi* (Haswell, 1879)	SG	654
Prawns-Tiger	*Penaeus monodon* (Fabricius, 1798)*, P. semisulcatus* (De Haan, 1844)	SG	23 438
Prawns-Unspecified	Penaeidae spp.	-	69
Rays	*Gymnura australis* (Ramsay & Ogilby, 1886)*, Himantura toshi* (Whitley, 1939), *Myliobatis australis* (Macleay, 1881), *Urolophus paucimaculatus* (Dixon, 1969), *Rhynchobatus djiddensis* (Forsskål, 1775), *Dasyatis kuhlii* (Müller and Henle, 1841), *Aetobatus narinari* (Euphrasen, 1790, *Trygonorrhina* sp., Dasyatidae spp.	FL	142
Sawfish-Unspecified	*Pristis zijsron* (Bleeker, 1851), Pristidae spp.	FL	13
Sea Perch-Mixed	Lutjanidae spp.	MG	27
Sea Perch-Mangrove jack	*Lutjanus argentimaculatus* (Forsskål, 1775)	MG	10 46
Snapper	*Pagrus auratus* (Foster, 1801), *Etelis carbunculus* (Cuvier, 1828), Caesionidae spp., Lutjanidae spp.	SG, FL	726
Tailor	*Pomatomus saltatrix* (Linneo, 1766)	FL	2 599
Tarwhine	*Rhabdosargus sarba* (Forsskål, 1775)	-	10
Threadfin-Blue	*Eleutheronema tetradactylum* (Shaw, 1804)	FL	33
Threadfin-King	*Polydactylus macrochir sheridani* (Macleay, 1884)	MG	2 045
Threadfin -Unspecified	Polynemidae spp.	-	5 226
Whiting	*Sillago ciliata* (Cuvier, 1829), *S. analis* (Whitley, 1943), *S. maculata* (Quoy & Gaimard, 1824) *, S. burrus* (Richardson, 1842), *S. ingenuua* (McKay, 1985)*, S. sihama* (Forsskål, 1775), *S. robusta* (Stead, 1908), Sillaginidae spp.	MG, FL	17 739

fisheries (L. Olyott, DPI&F, pers. com.), likely independence between the catches, and their value which represented two thirds of Queensland's total fish catch (or 65 % of the total fish catch in 2004 (DPI&F, 2005a). Selection criteria for species were that they should be/have (1) relatively constant and high market values; (2) well known; (3) estuary-dependent; and (4) wide spread throughout Queensland (based on Yearsley et al., 1999); L. Williams, DPI&F, pers. com.).

Annual summaries of the fish catch data were calculated together with the following variables: latitudinal section, species or species group, fishery type, total catch (tonnes) and catch-per-unit-effort (CPUE, kg/day). Daily catch data recorded from compulsory commercial fishing logbooks were standardised for fishing effort by dividing the catch of each species by the number of days fished in each section (Tanner and Liggins, 2000). I separated the four fisheries (trawl, net, pot, line) but also used total CPUE and catch, as the majority of the techniques are considered passive and their effort is measured in days with one technique dominating the fish catch for a species group. I compared total fish catch with the habitat parameters and did the same for CPUE for all fisheries and for individual fisheries to see whether different combinations would give similar outcomes. Yearly catch values for a total of 31 species or species groups were used. The fish species were selected according to their estuary-dependence (Table 4). These species are mainly 'marine-estuarine species' (see also Whitfield, 1999b for a detailed classification), which use inshore areas and estuaries for significant periods of time, often (but not limited to) during their juvenile phase. Several marine-estuarine species have juveniles that are only found within estuarine habitats (e.g., *Penaeus merguiensis*) (Staples et al., 1985; Vance et al., 1996a). Some catadromous species travelling between freshwater and marine habitats use estuarine habitats at certain life-stages (e.g., *Lates calcarifer*) (Russell and Garrett, 1983). The fish species groups have been divided into different categories according to their known or suggested habitat requirements (Appendix 3). These groups were then tested for their dependence using forward stepwise regression with the respective fish species as the dependent variable to identify any relationship between the geomorphic variables and fisheries catches.

3.2.3   Geomorphic data

The GIS software ArcGis 9.0 was used to access and visualise the estuarine habitat data and to produce raster maps of the habitats with a 10x10 m grid size. The DPI&F

1:100,000 vector data of the coastal wetland vegetation were combined with intertidal vector data from Geoscience Australia. The resulting vector data layer was converted to a 10x10 m grid for each for the 13 regions applying the assigned Geocentric Datum of Australia (GDA) 94 projection for each section using ArcGis 9.0. Each grid layer was analysed with the spatial metrics analysis package Fragstats 3.3 (McGarigal et al., 2002) to calculate the total area (CA), perimeter (Peri), mean perimeter to area ratio (PARA), number of patches (NP) and a connectivity index (CONNECT), giving a value for the distances between patches (threshold 100m) within the 13 separate sections (Fig. 5 and Fig. 6). A patch is defined as a small unit of the same habitat type. Perimeter includes all edges e.g. holes in the patch whereas edge only refers to the true edge of a patch.
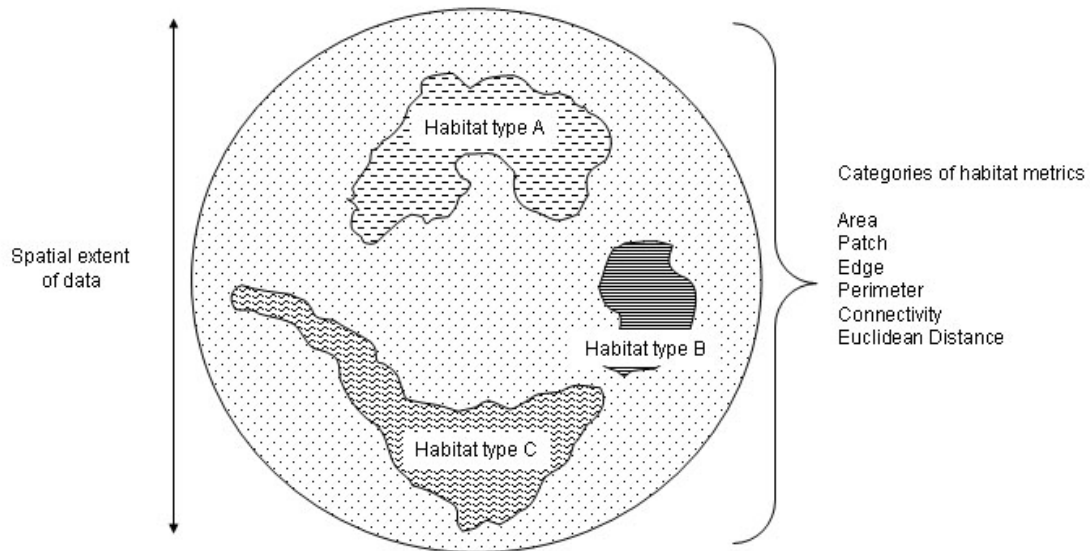
Figure 5: Categories of habitat metrics for land and seascape units

The connectivity index (CONNECT) is defined as:

$$CONNECT = \left[\frac{\sum_{j=k}^{n} c_{ijk}}{\frac{n_i(n_i-1)}{2}} \right] \times 100$$

The index is equal to the number of functional connections between all patches of a particular patch type ($\Sigma\ c_{ijk}$, where $c_{ijk} = 0$ if patches j and k are not within the specified

Figure 6: **Location of investigated area showing major cities and river systems in Queensland and selected geographical areas with fish catch grids for preliminary analyses**

distance of 100 m of each other and $c_{ijk} = 1$ if patch j and k are within the specified distance of 100 m), divided by the total number of possible connections between all patches of the corresponding patch type, and converted to a percentage (Clarke and Warwick, 2001). I used the lowest possible threshold to avoid the inclusion of habitat patches that were divided by a barrier (e.g., terrestrial vegetation). The perimeter to area ratio was measured for each habitat as a mean which equals the sum, across all patches of the corresponding patch type and patch metric values, divided by the number of patches (McGarigal et al., 2002). Over 140 estuaries from > 300 recognised Queensland estuaries were covered, ranging from the Gulf of Carpentaria (17° S 141° E) to Coolangatta (28° S 153° E). The following classes were used: channels, mangroves, salt marsh, seagrass, combined mud and sandflats, total number of estuaries and latitude per section. Mangroves, salt marsh, channel, mud and sandflats were calculated separately for total wetland parameters to look at the overall effect of the wetland on fish catch data. Seagrass was not included in the total wetland category due to its overlapping with the channel habitats on the digital map, which did not allow its separate calculation. The habitat parameter of each geographical area was assigned the corresponding grid code from the DPI&F fishery grid (Fig. 7).

3.2.4 Data analysis

Statistical analyses were carried out using the PRIMER 5.0 (Clarke and Ainsworth, 1993) and SPSS 12.01 software packages. Correlation analyses and nMDS were used to explore dependency and similarity between perimeter, area, mean perimeter to area ratio and connectivity of mangroves, salt marsh, seagrass, mud and sandflats, using Bray Curtis similarity and Euclidean distance. Data were square root transformed prior to analysis to normalise the variances. CPUE variables were not transformed whereas other variables were square root, fourth root or $\log_{10}$ transformed to reduce the right-skewness of the data. nMDS was used to represent the similarity of estuaries on the basis of habitats using Euclidean distance as a common method and explaining the variability in fish catch and CPUE was determined using the BIO-ENV procedure (Clarke and Ainsworth, 1993). This procedure generated various similarity matrices from subsets of the environmental variables and displayed the best of these various correlations (Mantel's tests). In addition, multiple regression models were used to
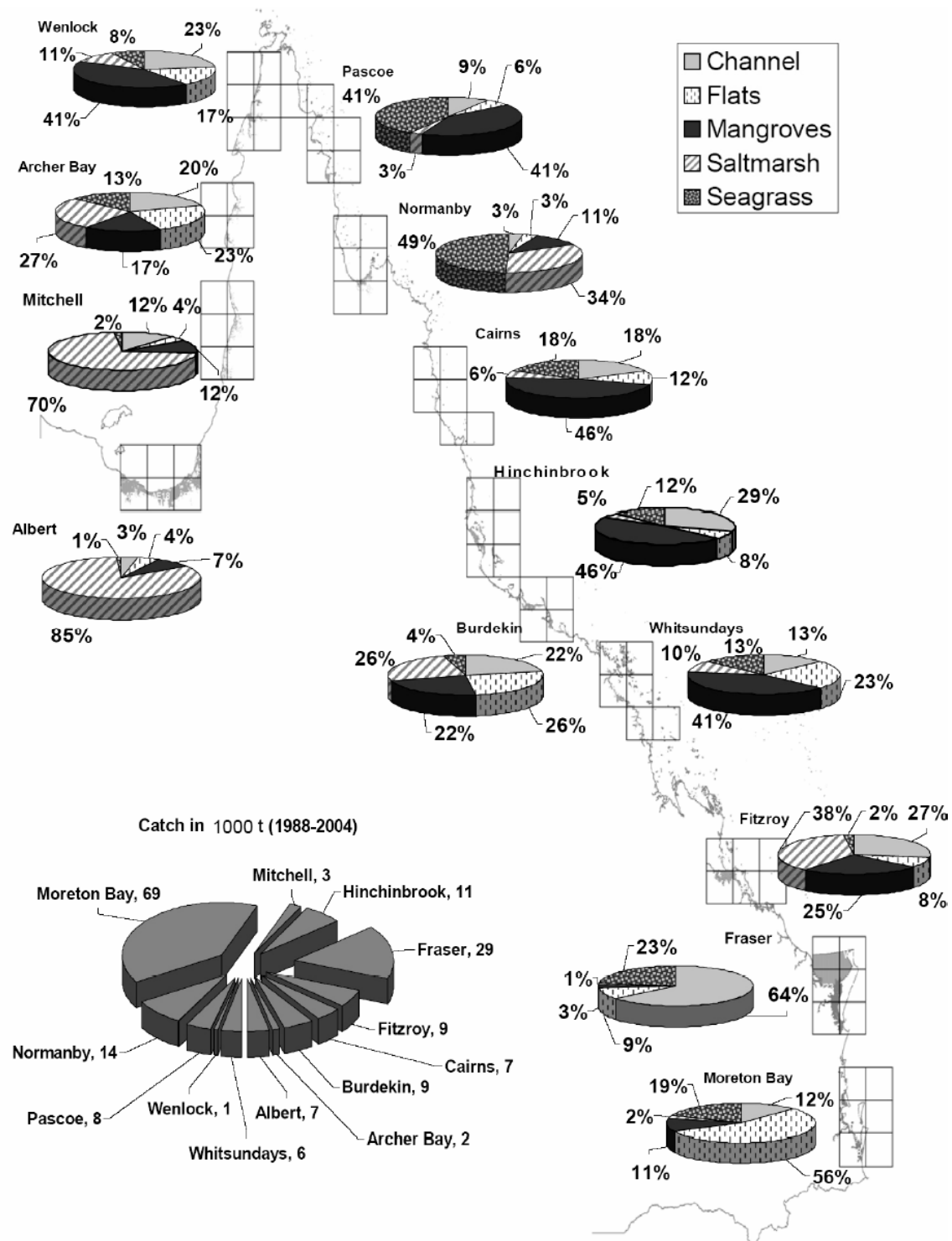
Figure 7: Total catch of 31 estuary-dependent species groups for 13 selected regions in Queensland between 1988-2004 and their extent of estuarine habitat

investigate which of the estuarine habitats accounted for the variation in fisheries production throughout the region. For this approach the CPUE for each fishery as well

as an averaged CPUE for all fisheries was applied. In order to exclude auto-correlation within the data (Pyper and Peterman, 1998), results from correlation analysis have been reviewed for consistency with the theoretical mechanisms proposed before the analysis, with the expectation that high correlation exists between habitats themselves and estuarine fish catch. Further considerations were given to "outliers", where species groups had less than 100 t catch in more than half of the investigated sections. These groups were excluded from the later analysis.

3.3   Results

3.3.1   Fisheries and environmental data

The total estuarine fish catch from all 13 geographical areas between 1988 and 2004 was 174,000 t compared to 255,000 t for the whole coast of Queensland (Fig. 7). There was little overlap in the species caught in the four fisheries, except for blue swimmer crabs (*Portunus pelagicus*), which were caught in both the pot (> 60 % of the catch) and trawl fisheries, and whiting (*Sillago* spp.) which were caught in the trawl (> 50 % of the catch) specifically directed towards stout whiting (*S. robusta*) and net fisheries. There have been differences in catches throughout the regions with the highest catch in Moreton Bay and Fraser Island, which also had the highest effort. The highest CPUE for all fisheries (pot, line, net and trawl) was in the Moreton Bay and Fraser region, where mullet (e.g., *Mugil cephalus*) contributed 28 % and whiting (*Sillago* spp.) 12 % towards the total catch. The CPUE in the line fishery was high in the central region (e.g., Hinchinbrook) and also in the north where effort was in general low and for some species (e.g., flatheads - *Platycephalus* spp.), too low to provide a meaningful CPUE. The net fishery dominated in the Gulf regions where the total catch and effort were low. The prawn fishery is not included within inshore grids as the trawl fishery in the Gulf of Carpentaria is undertaken only offshore. Barramundi (*Lates calcarifer*) and mud crab (*Scylla serrata*) catches accounted for more than half of the total catch in these regions.

    A large range of different estuarine habitat characteristics was covered by the selected geographical areas (Fig. 7). One extreme occurred in the north with the Albert River section having a large area of salt marsh/saltpan and high number of mangrove patches and the other extreme at the southern end of Queensland with Moreton Bay

providing large areas of flats, channel perimeter and seagrass perimeter. Large areas of mangrove occurred in the Hinchinbrook region whereas the Burdekin region had a high number of channel and patches of flats. The channel category dominated in the Fraser

Table 5: **Data on selected environmental variables for 13 locations for wetland components of six habitat types: channels, flats (mud and sand), mangroves, salt marsh, seagrass and total wetlands.** Variables include: area in km^2, perimeter in km, number of patches and estuaries, mean perimeter to area ratio (Mn_P:A) in km, and connectivity in %. Locations include: Al - Albert River, Ar - Archer Bay, Bu - Burdekin River, Ca - Cairns, Fi - Fitzroy River, Fr - Fraser, Hi - Hinchinbrook, Mi - Mitchell River, Mo - Moreton Bay, No - Normanby, Pa – Pascoe, We - Wenlock, Wh – Whitsundays

Parameters	Al	Ar	Bu	Ca	Fi	Fr	Hi	Mi	Mo	No	Pa	We	Wh
Latitude	18	14	20	16	24	25	19	16	27	15	12	12	21
Estuaries	7	5	11	13	12	6	21	4	21	6	6	8	22
Channel													
Area	11	6	20	3	26	366	19	6	16	3	3	15	7
Perimeter	1412	813	2132	483	1659	1828	329	1300	1910	627	441	1344	1177
Patches	27	16	431	22	23	186	114	8	129	14	10	12	55
Mn_P:A	2193	276	333	1167	779	115	2882	1407	1299	392	899	331	1078
Connectivity	3.70	0.83	0.07	3.03	1.58	0.00	1.41	10.8	0.57	0.00	2.22	0.00	1.28
Flats													
Area	15	7	23	2	8	51	5	2	73	3	2	11	12
Perimeter	400	888	2409	231	629	2737	70	314	3290	175	154	645	1109
Patches	33	104	486	58	117	138	133	64	344	20	42	82	240
Mn_P:A	321	262	355	719	503	443	527	555	680	250	540	337	445
Connectivity	0.57	0.15	0.07	0.36	0.22	0.30	0.21	0.25	0.15	1.05	0.46	0.39	0.24
Mangrove													
Area	25	5	20	8	24	15	30	6	15	10	14	27	21
Perimeter	5720	763	2132	776	4478	2038	2584	1128	2026	1647	1114	3341	2420
Patches	2471	312	431	126	1281	544	280	618	539	688	186	725	512
Mn_P:A	447	342	333	256	432	367	290	344	315	335	282	281	359
Connectivity	0.01	0.05	0.07	0.46	0.03	0.06	0.17	0.02	0.06	0.03	0.16	0.04	0.08
Saltmarsh													
Area	280	8	23	1	35	6	3	37	3	32	1	7	5
Perimeter	11760	1221	2409	132	4081	1421	520	4842	632	3315	350	1526	1040
Patches	1016	458	486	75	1032	726	251	1157	447	833	196	838	601
Mn_P:A	463	331	355	334	404	411	360	310	422	336	339	347	425
Connectivity	0.07	0.08	0.07	0.14	0.03	0.03	0.05	0.03	0.04	0.04	0.14	0.03	0.03
Seagrass													
Area	4	4	4	3	2	132	8	0	25	47	14	5	7
Perimeter	123	112	157	95	126	292	400	0	1307	409	250	129	409
Patches	8	6	28	18	29	9	59	0	277	14	15	14	105
Mn_P:A	36	49	140	248	158	146	96	0	289	87	32	61	155
Connectivity	0	0	0	0	0	0	0	0	0.05	1.10	0	0	0.05
Wetlands													
Area	358	27	66	13	92	434	56	53	208	51	20	61	44
Perimeter	5622	2175	3320	954	3841	2623	2660	5810	2664	3432	1194	4369	2575
Patches	514	368	328	70	210	113	108	914	73	755	125	626	242
Mn_P:A	461	344	382	323	443	433	355	337	504	343	293	342	385
Connectivity	0.09	0.12	0.08	0.54	0.29	0.51	0.33	0.04	0.76	0.04	0.19	0.07	0.11

region with Hervey Bay having large areas of seagrass. The Mitchell River section had the highest channel connectivity index and Normanby River, the highest flats connectivity index (Table 5). Furthermore, I detected significant positive correlation between estuarine habitat variables. Out of 304 estuaries and their adjacent habitats in Queensland, I found significant positive correlation between mangroves and salt marsh area ($r = 0.41$, $P < 0.01$, $n = 304$) and between seagrass and mangrove area ($r = 0.40$, $P < 0.01$, $n = 304$) (Table 6), suggesting that these habitats mostly occur together and may be regarded as a network of adjacent ecosystems.

Table 6: Pearson correlation of different estuarine habitat types in Queensland for 304 estuaries. $P < 0.05*$, $P < 0.01**$.

Habitats	Salt marsh	Seagrass	Flats	Channels
Mangroves	.411**	.403**	.576**	.627**
	Salt marsh	-	.120*	.180*
		Seagrass	.224**	.897**
			Flats	.499**

3.3.2   Relationships between habitats, fish catch and CPUE data

I compared total catch with different variables of the data set and found the best fit between section data of total fish catch and the wetland connectivity index ($r^2 = 0.62$) (Fig. 8). Furthermore, total catch correlated with the wetland mean perimeter to area ratio ($r^2 = 0.45$) and averaged CPUE for all fisheries correlated with total wetland area ($r^2 = 0.55$).

An *n*MDS based on square root transformed total fish catch data using Bray Curtis similarity showed a clear grouping of three classes for combined CPUE: (1) South-eastern Queensland; (2) Central and Northern Queensland; and (3) the Gulf of Carpentaria (Fig. 9a). The *n*MDS using standardised habitat parameters and applying Euclidean distance as a similarity measure resulted in some overlap with the *n*MDS based on standardised catch with the following groups: (1) Fraser, Moreton Bay (2) Whitsundays, Hinchinbrook, Cairns, Pascoe and Wenlock (3) Albert, Mitchell and Normanby. The Fitzroy, Archer Bay and Burdekin sections did not show clear grouping. The distribution of wetland connectivity is an important factor separating the regions, with the Fraser and Moreton Bay region having the highest connectivity index (Fig. 9b). The highest correlation was found in BIO-ENV ($r = 0.729$) between CPUE for

all fisheries and the wetland mean perimeter to area ratio, followed in rank order by the wetlands and flats connectivity index, and the total number of estuaries.
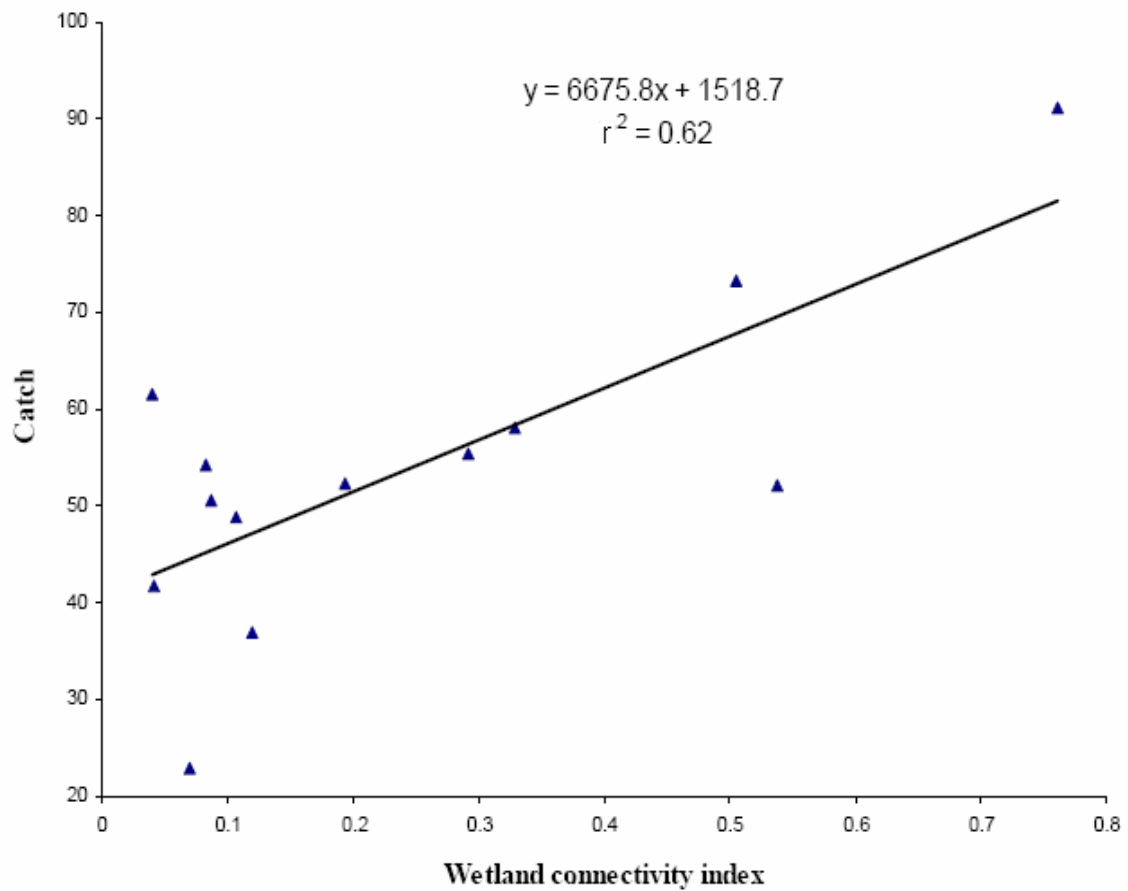
Figure 8: Relationship between wetland connectivity and total catch from 13 geographical areas along the coast of Queensland. The catch data were fourth root transformed
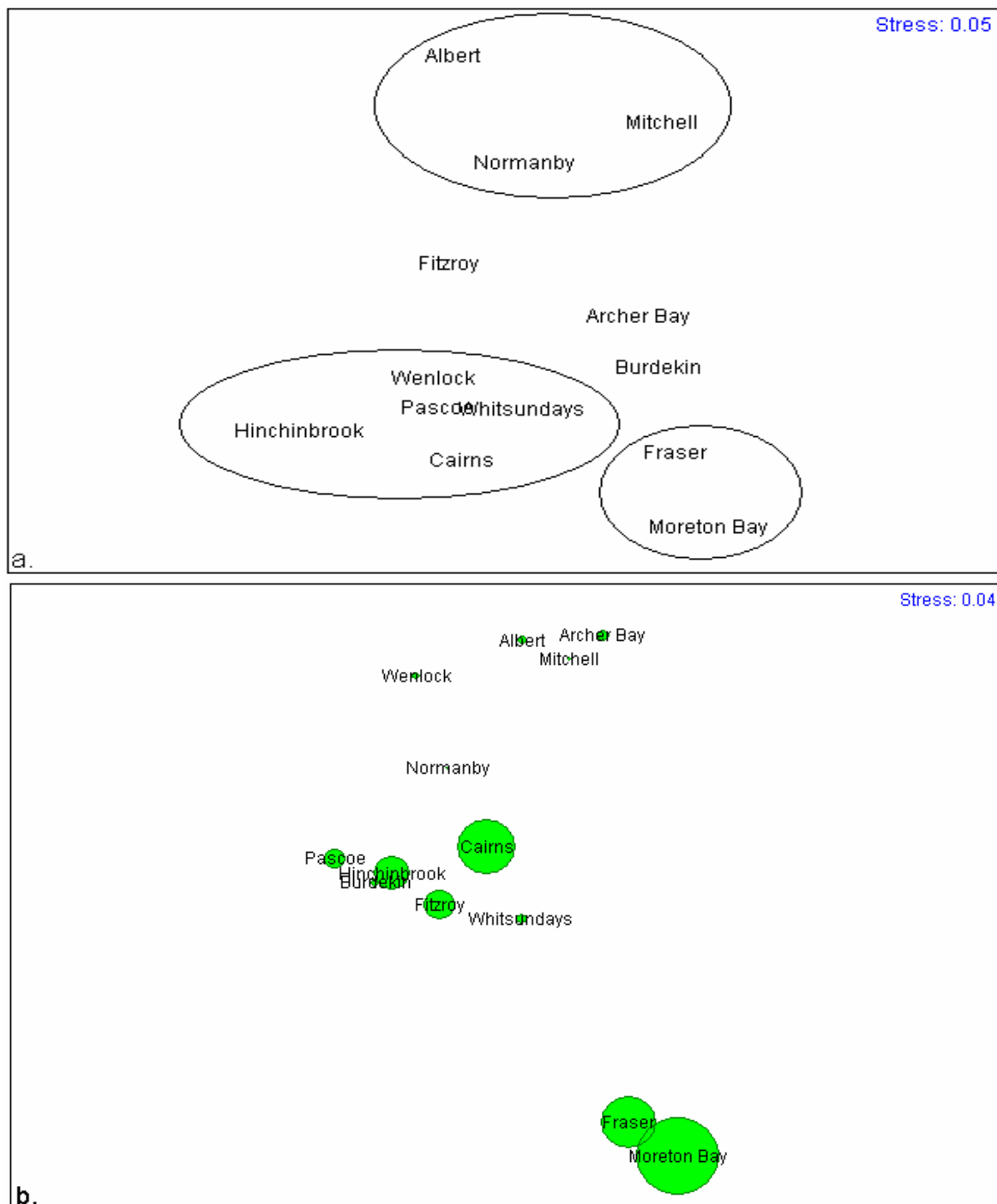
Figure 9: *n*MDS plots for 13 selected geographical areas. (a) based on untransformed standardised habitat parameters and latitude using Euclidean distance showing the three groups Southern Queensland, Central and Northern Queensland, the Gulf of Carpentaria; and (b) *n*MDS for 13 selected geographical areas (based on square root transformed catch data and Bray Curtis similarity) in Queensland, Australia. The value of the wetland connectivity index is indicated by the size of circles

3.3.3   Stepwise multiple regression analysis of key species

The most important environmental parameter for nine out of 24 species or species groups for predicting CPUE with stepwise regression was the total area of mud and sandflats as well as patches and their connectedness, followed by the wetlands parameters, which gave good CPUE prediction for another four species. The fitted model for flats, for example, accounted for 70-90 % of the variation in CPUE of mullet (e.g., *M. cephalus*), blue swimmer crab (*Portunus pelagicus*) and dart (*Trachinotus* spp.) (Table 7). Mangrove parameters were less important than expected for the mangrove related species such as barramundi (*L. calcarifer*) and mud crabs (*S. serrata*) (for comparison see Manson et al., 2005b), for which total wetland and seagrass parameters gave the best fit, albeit weak. Prawn CPUE data were in general best explained by the presence of flats, channels and total wetland parameter. For the seagrass related group it was wetlands or other parameters. Seagrass parameters did not emerge as significant.

The models identified mangrove parameters (connectivity) as being important in predicting CPUE for whiting (*Sillago* spp.) and greasy prawns (*Metapenaeus bennettae*). Latitude was the only parameter fitted to the model for king prawns (*Penaeus esculentus, P. plebejus*) that explained the variation in CPUE of this species group, accounting for 56 % of variation (Table 7). The total number of estuaries per geographical area was only important as a predictor for CPUE for banana prawns (e.g., *Fenneropenaeus indicus*). Some species groups did not show any significant relationship with any of the variables (grunter, *Pomadasys* spp.; snapper, *Pagrus auratus*) or had no significant r^2 value (flathead, *Platycephalus* spp.; flounder, *Pseudorhombus* spp.; rays, e.g., *Gymnura australis*; threadfin unspecified, Family Polynemidae; sawfish, Family Pristidae). Their total catch throughout the 13 geographical areas was in general low. When leaving out areas with no catch, I found fewer differences between CPUE for all fisheries and CPUE for the most important individual fisheries. The habitat to CPUE relation was relatively consistent throughout the different data sets.

Table 7: Significant r² values for the most important variables in stepwise multiple regressions predicting CPUE for 24 species groups.

A – all fisheries, N – net, T – trawl, C – pot, L – line. * P < 0.05, ** P < 0.01.
MG = mangroves, FL = flats, SG = seagrass, PERI = perimeter, CA = total area, NP = number of patches, PARA = perimeter to area ratio, Connect = connectivity

Habitat	Species	Parameters	Type	Adjusted r²	D.f.
MG	Barramundi	Seagrass NP, Wetlands NP	A CPUE	0.684*	12
		Seagrass Peri, Salt marsh Peri	L CPUE	0.648*	10
FL	Blue Swimmer crab	Flats CA, Channel CA	A CPUE	0.896**	12
		Channel CA, Seagrass NP	C CPUE	0.935**	11
		Wetlands CA, Salt marsh PARA	T CPUE	0.817*	8
MG, FL, SG	Bream	Flats CA, Channel NP	A CPUE	0.861**	12
		Flats CA, Channel connect	N CPUE	0.900**	12
FL	Bugs	Flats NP	A CPUE	0.535*	12
		Flats NP, Seagrass Peri	T CPUE	0.734*	8
FL	Dart	Flats CA, Wetlands CA	A CPUE	0.944**	12
		Flats CA, Wetlands CA	N CPUE	0.913**	12
MG	Milkfish	Flats connect, Seagrass PARA,	A CPUE	0.649*	12
MG	Mud Crabs	Flats connect, Mangrove CA, Wetlands NP	A CPUE	0.543*	12
		Wetlands NP, Mangrove CA, Flats connect	C CPUE	0.546*	12
FL	Mullet	Flats CA	A CPUE	0.743**	12
		Flats CA	N CPUE	0.789**	12
-	Prawns, Bait	Flats connect, Salt marsh Peri	A CPUE	0.553*	12
		Flats connect	T CPUE	0.658*	8
MG, FL	Prawns, Banana	Flats PARA, number of estuaries	A CPUE	0.653*	11
		Flats PARA	T CPUE	0.522	8
-	Prawns, Bay	Flats NP	A CPUE	0.634*	12
		Flats NP	T CPUE	0.915**	8
SG	Prawns, Endeavour	Wetlands Peri, Flats connect	A CPUE	0.578*	12
		Latitude	T CPUE	0.591	8
MG, SG, FL	Prawns, Greasy	Mangrove connect, Salt marsh connect	A CPUE	0.701**	12
		Channel Peri, Salt marsh Peri	T CPUE	0.958**	6
SG	Prawns, King	Latitude	A CPUE	0.563*	12
		Latitude	T CPUE	0.492	8
SG	Prawns, School	Channel CA, Wetlands connect	A CPUE	0.880**	12
		Wetlands all CA, Salt marsh connect	T CPUE	0.956**	8
SG	Prawns, Tiger	Wetlands Peri	A CPUE	0.293	12
		Wetlands PARA, Salt marsh CA	T CPUE	0.938**	8
-	Prawns unsp.	Channel NP, Flats NP	A CPUE	0.867**	12
		Channel NP, Flats NP	T CPUE	0.844*	8
MG	Sea perch	Wetlands NP	A CPUE	0.750**	12
		Channel CA, Channel PARA	L CPUE	0.924**	12
MG	Mangrove Jack	Channel connect, Wetlands NP	A CPUE	0.849**	12
		Channel connect, Wetlands NP	N CPUE	0.854**	12
FL	Tailor	Channel connect, Salt marsh connect	A CPUE	0.858**	12
		Channel connect, Salt marsh connect	N CPUE	0.898**	12
-	Tarwhine	Flats CA	A CPUE	0.622*	12
FL	Threadfin Blue	Seagrass NP	A CPUE	0.885**	12
		Channel Peri	N CPUE	0.566*	12
MG	Threadfin King	Salt marsh CA	A CPUE	0.828**	12
		Salt marsh CA	N CPUE	0.935**	12
MG, FL	Whiting	Wetlands all CA, Mangrove connect	A CPUE	0.959**	10
		Wetlands PARA, Seagrass CA	N CPUE	0.773**	12
		Channel CA, Wetlands connect	T CPUE	0.995**	8

3.4 Discussion and conclusion

This study has shown an empirical link between estuarine habitat and fishery production for estuary-dependent species. Intertidal flats were one of the most important variables, beside tidal wetland connectivity, explaining fish catch variation in Queensland according to regression analyses for single species. These outcomes are similar to findings by Saintilan (2004) for New South Wales, Australia. The BIO-ENV results with CPUE for all fisheries further demonstrated that flats were one of the most important variables. This suggests that the significance of mud and sandflats has been greatly underestimated in broad scale analyses to date. Mud and sandflats are often in proximity to mangroves, salt marshes and seagrass suggesting connectivity making Queensland's coastline a complex mosaic of habitat types (Banks and Skilleter, 2002). Larger fish of the same species use the flats and unvegetated areas in inshore and open waters of estuaries as feeding grounds (Chong et al., 2001; Laegdsgaard and Johnson, 2001). These ecosystems are an important habitat for larger fish species, have high microbial activity and large quantities of microphytobenthos (MPB). Mullet (Mugilidae spp.) for example feed on detritus, diatoms, algae and small invertebrates that they filter from mud and sand (Williams, 2002). According to my results, their catch is best explained by the size of mud and sandflats. Mudflats are common in tropical Australia with extensive occurrence in northern Australia due to large tidal ranges. However, defining just one habitat as the major driver of CPUE from the data is misleading. My results support the model that fish species depend on a number of habitats with the overall catch and CPUE being dependent on the whole estuarine wetland habitat suite rather than any single habitat. For example, mangrove jacks (*Lutjanus argentimaculatus*) are known to utilise all types of tidal wetlands as juveniles (Williams, 2002) and prefer sheltered areas in channels. Therefore, areas with high channel connectivity and relatively large wetland patch perimeter are likely to promote higher mangrove jack catches.

Although mangroves were well represented in the 13 geographical areas they were not the only important habitat. One reason is that many fish and crustaceans only use mangrove forests for a part of the tidal cycle (Vance et al., 2002). Noting that mangroves are flooded less than 50 % of the time (Duke, 2006), the availability of adjacent habitats must be important as well. Some estuarine species move and migrate

between habitat types, localities and regions, e.g., sea mullet (*M. cephalus*) (Cappo et al., 1998). This implies, that it is difficult and possibly misleading to separate the value of each habitat type from the broader estuarine values when looking at fish catch data.

The connectivity index calculated with the program 'Fragstats' appeared as a useful parameter that did not strongly correlate with other habitat parameters but combined their values in biologically meaningful ways. The index was positively related to total catch and combined CPUE of all fisheries. One explanation for this relationship may be the importance of easily accessible estuarine habitats to fish for the provision of food and shelter. The connectivity index reflects the proximity of habitat patches and therefore their potential accessibility. The differences between CPUE in the north and the south may also be related to a reduction in habitat connectivity in the northern habitats. Studies on habitat connectivity are important; as connectivity influences other ecological processes such as species distribution, food availability and population dynamics.

3.4.1   Extraneous Influences

Associated with the habitats are a number of abiotic factors such as rainfall, which may influence fish catch (Robins et al., 2005) but have not been considered in this study. Data on environmental factors could be useful in regression analyses to explain more of the variation in catch and is essential to better understand how the interactions of such factors affect the distribution of fish within various estuary types and broad geographic areas (Blaber, 2000). Other factors such as stock-recruitment relationships, habitat changes, pollution impacts, competition and predation, management plans, fuel prices and market forces are also influencing the fish catch data and may be considered in any such future analyses when quantification is possible.

The analyses are limited by the resolution of the data set, the type of fish selected and spatial classification of habitats. An *a priori* classification of the data is essential as unclassified aggregation can lead to misinterpretation. Assessments may be improved by: (1) refining parameters; (2) including species that are not estuary-dependent; and (3) conducting analyses over a range of spatial and temporal scales since the spatial scale of data collection can affect covariance and correlation statistics (Dungan et al., 2002). Analyses, for example, between the fish catch data and small areas of estuarine habitats

are not meaningful, as the spatial resolution of fish catch data does not support such a spatial scale. Some data in Queensland are now collected at the 6-nautical-mile scale, which can significantly improve the usefulness of the data. Additional limitations in fish catch data include the recording of fish by common names, which can often result in confusion of specific identity.

The correlations in my study were generally strong, with r^2 values usually above 0.7. However, this cannot be used to assume causality because of possible non-linearity of linking mechanisms (Baumann, 1998) and the small size of the data sets. In general, I found that species or species groups with an overall low catch record had weak or non-significant results when nMDS and stepwise regression analyses were used. Significance test for most of the adjusted r^2 values resulted in significance at 0.01 probability levels. A standard experiment wise error test showed a 0.26 probability that the significance tests resulted in a Type I error. The actual experiment wise error rate will range between the computed experiment wise error rate (0.26) and the test wise error rate (0.01). The computed experiment wise error rate can be reduced further, by reducing the size of the allowable error (significance level) for each comparison but may fail to identify an unnecessarily high percentage of actual significant differences in the data (Olejnik et al., 1997). Another way of assessing such correlated parameters can be achieved with the application of multivariate decision trees (Breiman et al., 1984; De'ath and Fabricius, 2000).

Unfortunately, there is a lack of essential life-history information for most of the major fishery species in Australia. There is a need for additional information on "critical" habitat requirements and processes such as recruitment, post-recruitment mortality and competition, spawning, and species interactions: information that is important in assessing the value of habitats for fishery species (Beck et al., 2001) and further modelling.

3.4.2 *Conclusion*

In conclusion, healthy and functional estuarine wetlands are fundamental to the health of fish stocks and for optimising sustainable yield for commercial and recreational fishers. The protection of one habitat type will only benefit a minority of commercially important fish species. To optimise habitat and fisheries management a combination of

estuarine habitats should be considered in fisheries management, supporting the move from conventional single-species or single-habitat management to ecosystem-based management (NRC, 1999). Fishery systems are complex, and the management systems needed to optimise the benefits accruing from fisheries require institutions and knowledge systems that are able to cope with the multi-disciplinary requirements of the fisheries management function. Fisheries managers need to broaden their knowledge base in order to make informed decisions. The controversy over estuarine outwelling has benefited from recent tracer techniques, such as stable isotope analyses, which enable the tracking of food sources for individual species (Lee, 2005). In many cases, the results question the extent of the outwelling hypothesis (Odum, 1968), suggesting the need for further studies and refinement of current knowledge. Deficiencies of studies that address the relationship between estuarine habitats and fish catch are mainly due to their single-habitat approach. The present data on fish catch and estuarine habitat distribution allowed for modelling but has its spatial limitations. The analyses showed a strong dependency by 24 estuarine species or species groups to different estuarine wetland habitats with mud and sandflats being most important for 25 % of the species. Overall, the results showed a broad diversity of coastal habitats, other than those receiving most attention, were essential also to the completion of fish life cycles. My investigation contributed to the broader knowledge of coastal habitats and how they influence fisheries catch and productivity. Future studies concerning fish catch and estuarine habitats could benefit from: (1) refined collection of fisheries-dependent data; (2) considerations of linked processes between estuarine and coastal habitat types; and (3) enhanced knowledge on life history stages of estuary-dependent fish species.

CHAPTER 4

LINKING SPATIAL METRICS AND FISH CATCH REVEALS THE IMPORTANCE OF TIDAL WETLAND CONNECTIVITY TO INSHORE FISHERIES IN QUEENSLAND, AUSTRALIA

4.1   Introduction

Estuary-dependent or opportunistic fish species are the main target of inshore fisheries that sustain many coastal communities (Feierabend and Zelazny, 1987; Houde and Rutherford, 1993). Of the ~11,300 extant fish species, the largest proportion (46 %) are coastal, ranging from estuaries, lagoons, and deltas to the outer continental shelf (Nelson, 1994), providing > 90 % of the global fish catch (Pernetta and Milliman, 1995). These fish species have different resource requirements depending on their life cycle (Pittman and McAlpine, 2003; Pittman et al., 2007). However, increased fishing pressure and the alteration of coastal habitats cast doubts on the future of nearshore fisheries. More than 50 % of the world human population live within 50 km of the coast (Hinrichsen, 1998), impacting on important nursery habitats for fish. Understanding of the value of these nursery habitats for estuarine fish species is vital for their effective protection. In order to establish functional marine protected areas (MPA) to prevent fisheries habitat destruction, an appraisal of the most effective spatial combination of habitats is necessary (Lenanton and Potter, 1987).

There has been extensive search for estuarine habitat values in the past few decades for fish species (Odum, 1968; Macnae, 1974; Blaber et al., 1995; Barbier, 2000; Beck et al., 2001; Gillanders et al., 2003) helping to define the most effective marine protected area design. Many studies have shown linkages between estuarine habitats and nearshore fisheries (Pauly and Ingles, 1986; Manson et al., 2005b; see also Chapter 3). Analyses of estuaries and fish catch data have demonstrated the potential of data sets already available for Australia (Pease, 1999; Duffy et al., 2003; Saintilan, 2004; Manson et al., 2005b; see also Chapter 3). Several studies have found correlations between the area of mangroves and the catch in nearby fisheries (Staples et al., 1985; Manson et al.,

2005b). However, little is known about the relative influence of coastal geomorphic characteristics, in particular, the importance of habitat connectivity, on the spatial distribution and abundance of commercial estuarine fish species. Studies suggested that the 'edge' of tidal wetlands provides the greatest attraction to aquatic organisms such as crustaceans and fish (Minello et al., 1994; Vance et al., 2002; Haas et al., 2004). Any direct or indirect (e.g., change in edge to area ratio due to fragmentation) alteration of the habitat edge will therefore affect the abundance of aquatic organisms. For example, Browder et al. (1989) showed that salt marsh fragmentation was related to prawn abundance. On scales of a few meters predation rates of shellfish in seagrass beds were dependent on the patch size to perimeter ratio (Irlandi et al., 1999) and densities of juveniles of larger fish species showed a negative relationship with increased fragmentation of seagrass beds (Jackson et al., 2006). Non-species-specific connectivity measures have previously been used in terrestrial reserve design (e.g., buffer measure: Araújo et al., 2002; distance-dependent connectivity measures: Briers, 2002) but currently there are no examples of work on spatial metrics addressing coastal benthic habitat connectivity at appropriately broad spatial scales to support resource management decision making (Schumaker, 1996; Fortin et al., 2003; Ray, 2005; Sheaves, 2005).

A landscape approach to the ecological study of marine animals is still in its infancy (Kneib and Wagner, 1994; Robbins and Bell, 1994; Pittman and McAlpine, 2003) with few studies quantifying seascape structure at spatial scales appropriate to fisheries management. Investigations have been dominated by studies in single habitat types, predominantly mangroves (Faunce and Serafy, 2006) or seagrass (Pittman and McAlpine, 2003). For highly mobile species, particularly those that use more than one habitat type through daily home range movements or ontogenetic shifts, a broader scale approach that incorporates multiple habitat types is appropriate (Irlandi and Crawford, 1997; Nagelkerken et al., 2001; Faulkner, 2004; Mumby et al., 2004; Faunce and Serafy, 2006). Landscape connectivity is thus an important concept that has rarely been explored for marine environments but is now subject to increasing attention (Hemminga, 1998; Frost et al., 1999; Kneib, 2000; Sheaves, 2005; Guest and Connolly, 2006). A seascape connectivity index for estuarine habitats could be a useful instrument for the implementation of an ecosystem approach to fisheries management.
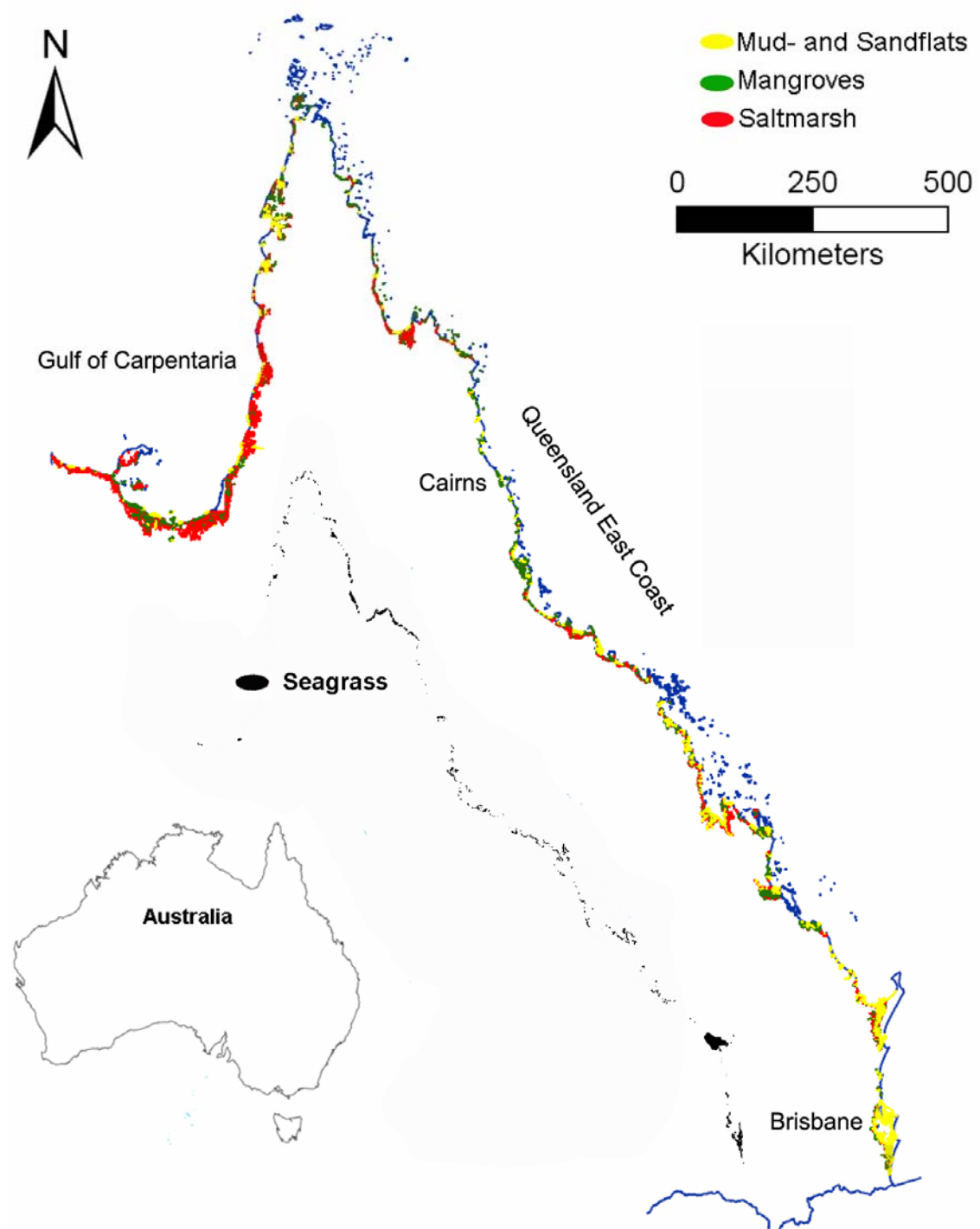
Figure 10: Study area of Queensland showing the coastline, tidal wetlands and seagrass distribution (data based on BOM, 2003; Geoscience Australia, 2004)

I used structural 'connectivity' in this study to investigate the proximity of important fish habitat types and develop indicators for spatially characterising coastal ecosystems

of Queensland, Australia. This study was exploratory and focuses on spatial associations between faunal patterns and environmental patterns. The coastline of Queensland is covered with an estimated 32,000 km^2 of mangroves, salt marsh and seagrass (Bucher and Saenger, 1994) and extending from Moreton Bay (27° S 153° E) in south-east Queensland with a subtropical climate and average rainfall of 1100 mm y^{-1}, to the Gulf of Carpentaria (14° S 139° E) in the north-west of Queensland with a tropical climate and an average rainfall ranging from 600 mm y^{-1} in the south to 1600 mm y^{-1} in the north (BOM, 2003). This stretch of coastline provides a large variety of different habitat types varying in spatial configuration along the coast (Fig. 10).

I investigated the linkages between seascape structure and the spatial distributions and abundance of estuary-dependent fish species based on fisheries species catch records using landscape ecology concepts and methods together with the long term fish catch data from Queensland. The analysis of this chapter had the following objectives: (1) evaluate the relative importance of connectivity in comparison to other habitat metrics in determining coastal fish catch in Queensland, Australia; and (2) identify geographical regions in coastal Queensland with high estuarine habitat connectivity. The results will help generate testable hypotheses concerning the value of habitat connectivity to nearshore secondary production, and assist prioritisation of estuaries for future ecosystem-based fishery management.

4.2   Material and methods

4.2.1   Fishery and tidal wetland data collection

Data on catch, effort (number of days and boats) and gross value of production for estuary- dependent species or species groups were provided by DPI&F Assessment and Monitoring Unit (Table 8) (see also Chapter 3 for a detailed description). All data collected were assigned to cells, resulting in about 400,000 records over the 17 years. Data unevenness between grid cells was minimised by calculating catch-per-unit-effort (CPUE) values based on the raw catch data. Queensland coastal waters provided an annual estuarine fish catch of 27,000 t in 2004 (DPI&F, 2005b) with a continuous increase in inshore fish catch (Fig. 10).

Data on Queensland coastal wetland vegetation were obtained from DPI&F Assessment and Monitoring Unit. The 1:100,000 coastal wetland vegetation map includes information on mangrove communities, salt marsh and seagrass based on Landsat TM images and ground truthing (currency 1987-1999). Data on channels, intertidal flats and sandflats for the whole coastline were taken from Geoscience Australia (Geoscience Australia, 2004). This information was primarily sourced from the 1:100,000 scale National Topographic Map series produced by Geoscience Australia, with the minimum mapping unit being 1:15,000. Other details on the maps, such as accuracy and depth limits, can be found from Geoscience Australia (2004).
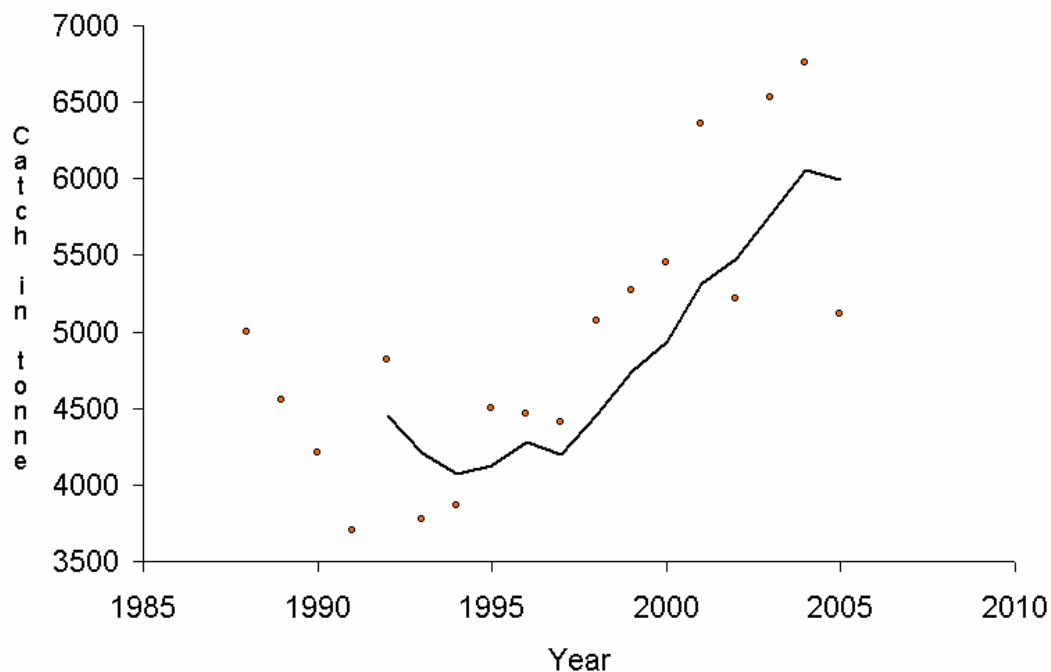
Figure 11: Annual catch of all inshore fisheries in Queensland, Australia.

The line represents the 5-year running mean (data from DPI&F, 2005)

I divided the coastline into two regions: the east coast of Queensland (EC) and the Gulf of Carpentaria (GOC). Preliminary studies showed that habitat and fish catch characteristics broadly fall into these two groups (Chapter 3). Some species are genetically different e.g., *Lates calcarifer* (Shaklee and Salini, 1985) between the two regions. Ninety 30-nautical-mile grids within a 30 km radius to the coastline were selected (Fig. 12), representing almost two-thirds of Queensland's total fish catch in 2004. I also tested available 6 nm catch grid records (2001-2005) for 41 grids where

single estuaries could be assigned to one particular grid. Only numerically dominant species were examined (Table 8). Additional selection criteria for species were their relatively constant and high market values, estuary-dependence and occurrence throughout Queensland (based on Yearsley et al. (1999); and L. Williams pers. comm., DPI&F; Table 8). Two offshore species groups mackerel (Scombridae) and coral trout

Table 8: Species selected for the analyses (a total catch of 0.17 M t from 21 estuarine associated fish species or species groups) from 90 fish catch grids along the coast of Queensland from 1988-2004. The dominant type of fishery selected for the analyses is shown in brackets and two control offshore species groups are shown in bold

Common Name	Taxa or dominant taxa of species group	Catch in 1000 t
Barramundi (net)	*Lates calcarifer*	11
Bream (net)	*Monodactylus argenteus, Acanthopagrus australis, A. berda, Plectorhinchus gibbosus*, Sparidae spp.	2.9
Blue Swimmer Crab (pot)	*Portunus pelagicus*	8.3
Mud Crab (pot)	*Scylla serrata*	10.6
Dart (net)	*Trachinotus anak, T. blochii, T. botla, Trachinotus* spp.	0.5
Grunter (net)	*Pomadasys maculatum*	0.6
Milkfish (net)	*Chanos chanos*	0.2
Mullet (net)	*Liza vaigiensis, L. subviridis, L. argentea, Valamugil georgii, Valamugil seheli, Mugil cephalus, Trachystoma petardi*, Mugilidae spp.	2.5
Bait Prawns (trawl)	Family Penaeidae	1.1
Banana-Prawns (trawl)	*Fenneropenaeus indicus, F. Merguiensis*	10.2
Bay-Prawns (trawl)	*Metapenaeus macleayi, M. insolitus*	6.1
Endeavour-Prawns (trawl)	*Metapenaeus endeavouri, M. ensis*	16.5
Greasy-Prawns (trawl)	*Metapenaeus bennettae*	1.5
King-Prawns (trawl)	*Penaeus esculentu, P. plebejus*	14.2
School-Prawns (trawl)	*Metapenaeus macleayi*	1.3
Tiger-Prawns (trawl)	*Penaeus monodon, P. semisulcatus*	29.4
Mangrove Jack (net)	*Lutjanus argentimaculatus*	0.1
Tailor (net)	*Pomatomus saltatrix*	2.4
Blue-Threadfin (net)	*Eleutheronema tetradactylum*	3.1
King-Threadfin	*Polydactylus macrochir*	7.1
Whiting (net)	*Sillago ciliata, S. analis, S. maculata, S. burrus, S. ingenuua, S. sihama, S. robusta*, Sillaginidae spp.	22.8
Mackerel (line)	*Scomberomorus commerso,* **Family Scombridae**	**4.3**
Coral–Trout (line)	*Plectropomus leopardus, P. maculatus*	**2.1**

(Serranidae) were selected as "control" species. I expected that catch of the control species would not be related to estuarine characteristics. Annual summaries of the fish catch data were calculated with latitudinal bands, species or species group, fishery type
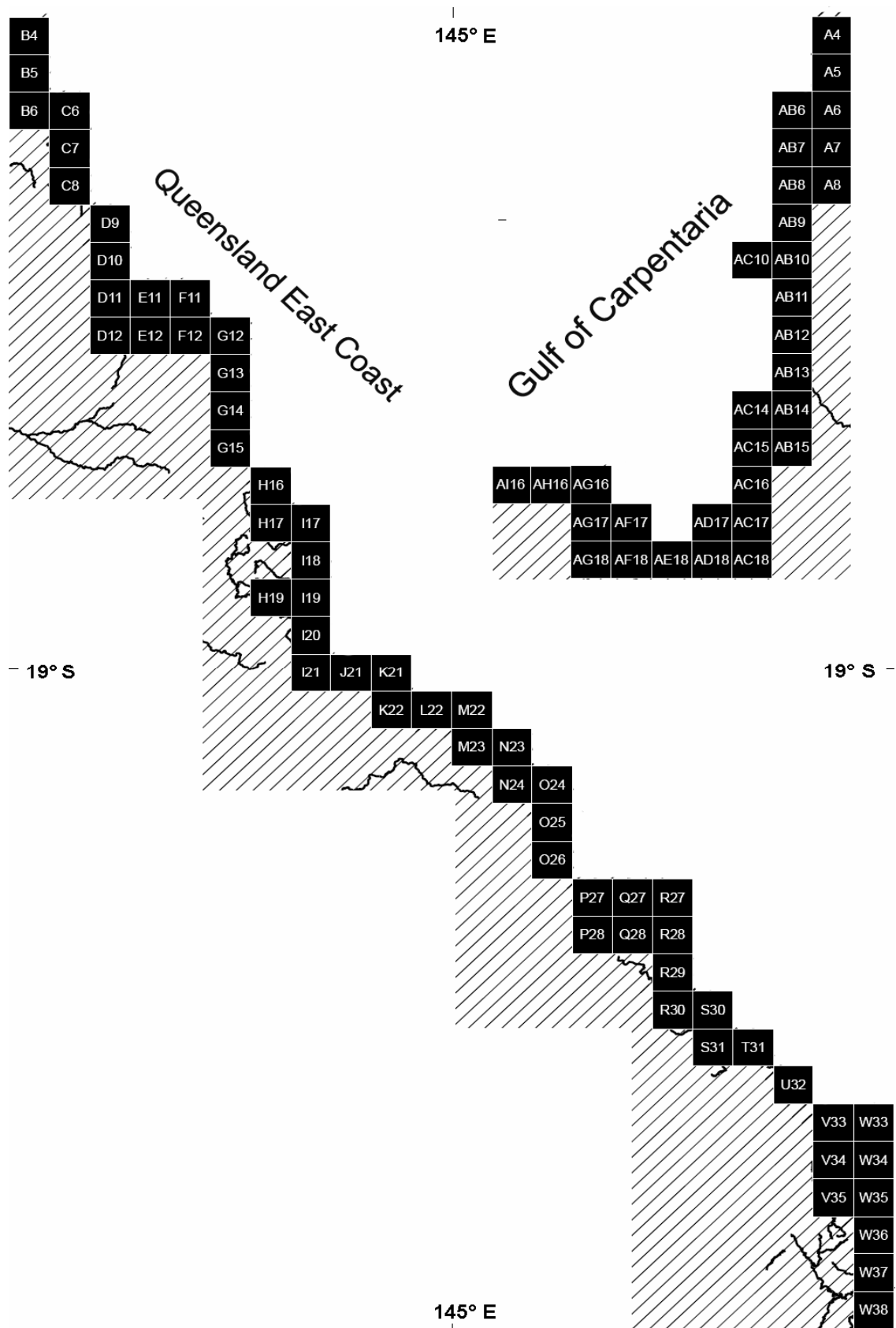
Figure 12: Distribution of 90 selected fish catch grids showing fish catch record identification number

and catch-per-unit-effort (CPUE, kgd^{-1}). Fish landing data were standardised for fishing effort by dividing the catch of each species by the number of days fished in each section (Tanner and Liggins, 2000). I separated trawl, net, pot and line fisheries and used the dominant type of fishery of each species group for further analyses.

I compared the CPUE and total catch for all fisheries and for individual fisheries with the spatial pattern metrics (i.e., those of Table 9). Yearly catch values for a total of 21 species or species groups were used. These species were mainly 'marine-estuarine species' (see Whitfield, 1999b for a detailed classification), which use inshore areas and estuaries for significant periods of time, often during their juvenile phase. Two grids at the north east coast, AB10 and A8, where the fish catch was close to zero were excluded from further analyses. The grids V33, W33, V34, W34 and V35, W35 were combined to two grids in each case because they shared significant areas of contiguous estuarine habitats. Fish catch data were checked for records of low catch and entries with less than five days of fishing effort were removed.

Geomorphic characteristics of estuarine habitats can be derived from physical attributes of the landscape measured by the quantification of geometric and spatial properties of categorical maps, such as area, number of patches, patch area, patch perimeter, perimeter-area ratio, shape, and location of habitat patches, describing a landscape matrix (Hargis et al., 1998). I used the GIS software ArcGIS 9.1 to access and visualise the estuarine habitat data and to produce maps of the habitats (Fig. 10). The DPI&F 1:100,000 vector data of the coastal wetland vegetation were combined with intertidal vector data from Geoscience Australia. The resulting vector data layer was converted to a 10x10 m grid for each of the 90 grids. I selected a small pixel size (10x10 m) in comparison to the data resolution of 1:100,000 thus minimising the error associated with pixel size. The characteristics of the spatial configuration of estuarine landscape mosaic (mangroves, salt marsh, seagrass, channels, mud and sandflats and total tidal wetlands) for each of the 90 fish catch grids were quantified, with 12 different landscape metrics selected as the most appropriate parameters to measure composition and spatial pattern (Bunn et al., 2000; Manson et al., 2005b) resulting in a total of 38 different parameters. Mangroves, salt marsh, wetted channels and mud and sandflats were combined to wetland parameters (WL) based on class level metrics representing the area (proportion of the landscape) and spatial pattern of a single habitat type within a landscape unit (McGarigal et al., 2002). Seagrass was not included in the total wetland parameter due to its overlap with the channel habitat on the digital map, thus not

allowing its separate calculation. To measure structural connectivity (Calabrese and Fagan, 2004), I used patch density (PD), Euclidean nearest-neighbour distance (ENN), and the number of connections within 100 m and 1000 m between patches divided by the number of all possible connections between these patches (CONNECT) (McGarigal et al., 2002). In addition, I calculated the length of shared edge between different habitat type patches (LCE). For this, the polygon layers were transformed into a single line layer, the length for each line calculated and shared edges between the habitat types mangroves, salt marsh, and flats selected manually (Table 9). The number of rivers was counted and estuary length as well as coastline length measured with the ArcGIS distance measurement tool. Patch-dependent metrics (ENN, CONNECT and PD) have been excluded from analyses where fewer than two habitat patches were present in a fish catch grid. Connectivity metrics for channels were not calculated as all channels were assumed to be spatially connected with the number of channels being a better predictor of connectivity.

Table 9: Metrics computed with Fragstats software and ArcGis 9.1.

Metric	Abbreviation	Description
Total area (ha)	CA	Total area of all patches
Number of patches	NP	Total number of patches
Patch density	PD	Number of patches per 100 ha divided by total landscape area
Length of coastline	LC	Total length of coastline in km
Number of estuaries	NE	Total number of estuaries
Latitude	LA	Latitude in degree
River length	RL	Total river length in km
Connectivity index	CONN-ECT	Percentage of the number of functional connections between all patches of the same patch type within 100 or 1000 m, divided by the total number of possible connections between these patches. Lowest possible threshold to avoid the inclusion of habitat patches that are divided by a barrier.
Euclidean distance	ENN	Distance (m) from a patch to nearest neighbouring patch of the same type (class), based on patch edge-to-edge distance, computed from cell centre to cell centre
Length connected edge	LCE	Total length of shared edge between habitat patches in km
Mean perimeter to area ratio	PARA	Mean that equals the sum, across all patches of the same patch type and patch metric values, divided by the number of patches
Length connected edge to area ratio	LCPA	Length of shared edge line between mangroves, salt marsh and flats patch types, divided by the total area of all patches

4.2.2   Statistical analyses

Statistical analyses were carried out using the PRIMER 5.0 (Clarke and Ainsworth, 1993) and SPSS 15.0 software packages. Pearson correlation analyses were used to explore colinearity between some of the variables. *n*MDS was used to visually represent the between cell similarity of tidal wetlands on the basis of habitat connectivity (Conner et al., 2002). The extent to which habitat variables explain the variability in fish catch and CPUE was determined using the BIO-ENV procedure (Clarke and Ainsworth, 1993). This procedure generates various similarity matrices from subsets of the environmental variables and displays the best of these various correlations (Mantel's test). The biological environmental gradients procedure carries out tests of the relationship between species catch and environmental variables and is comparable to multiple regressions.

Using several landscape metrics to measure similar phenomena can result in a high degree of multicolinearity among variables (Li and Reynolds, 1993), which can cause difficulties in later interpretation. Pearson correlation coefficients were calculated and relationships were detected for the landscape variables. However, since the chosen metrics measured different aspects of the estuarine landscape configuration, the landscape composition and landscape context, and were meaningful to the questions being asked, they were not considered to be redundant. To overcome these problems, a PCA was conducted (Johnson and Cage, 1997). Only principal components (PCs) accounting for > 10 % of the data variation were considered. All metrics have been linearly transformed using PCA into a set of uncorrelated, i.e., orthogonal, principal components. The PCs were then entered into forward stepwise regression models with the respective fish species as the dependent variable to identify any relationship between the environmental variables and fisheries catches. In addition, relationships between CPUE and the proportion of individual habitat types in the seascape were investigated using linear regression on square-root transformed variables.

4.3   Results

4.3.1   *Habitat distribiution, spatial characteristics and overall results*

The habitat analyses reflected large areas of tidal wetlands in south-east Queensland and the GOC where saltpans are dominating. Wetland edge to area ratio was highest in the south-east (V35W35), north of Queensland (G14) and the GOC (AG18, AD18); wetland CONNECT (100 m) (see also Table 9) was highest in south-east Queensland, near Cairns (H17, I17) and in some grids in the GOC (AC10, AG18, AC18) (Appendix 4).
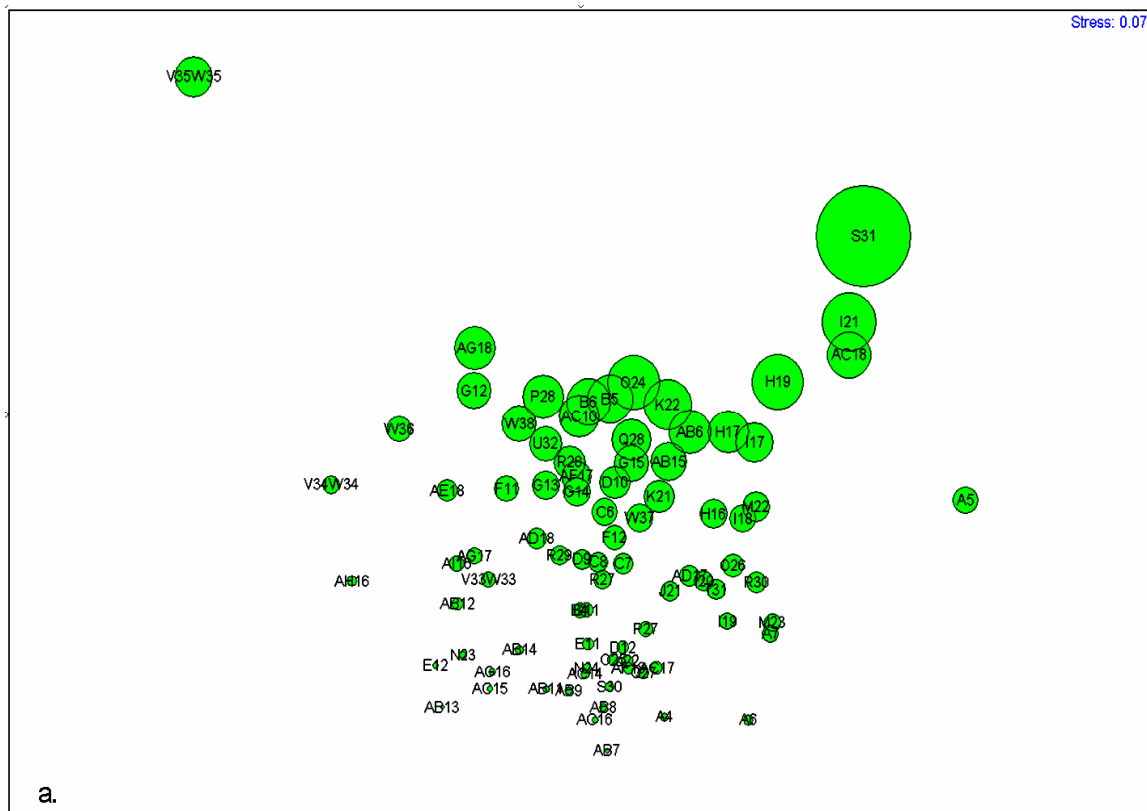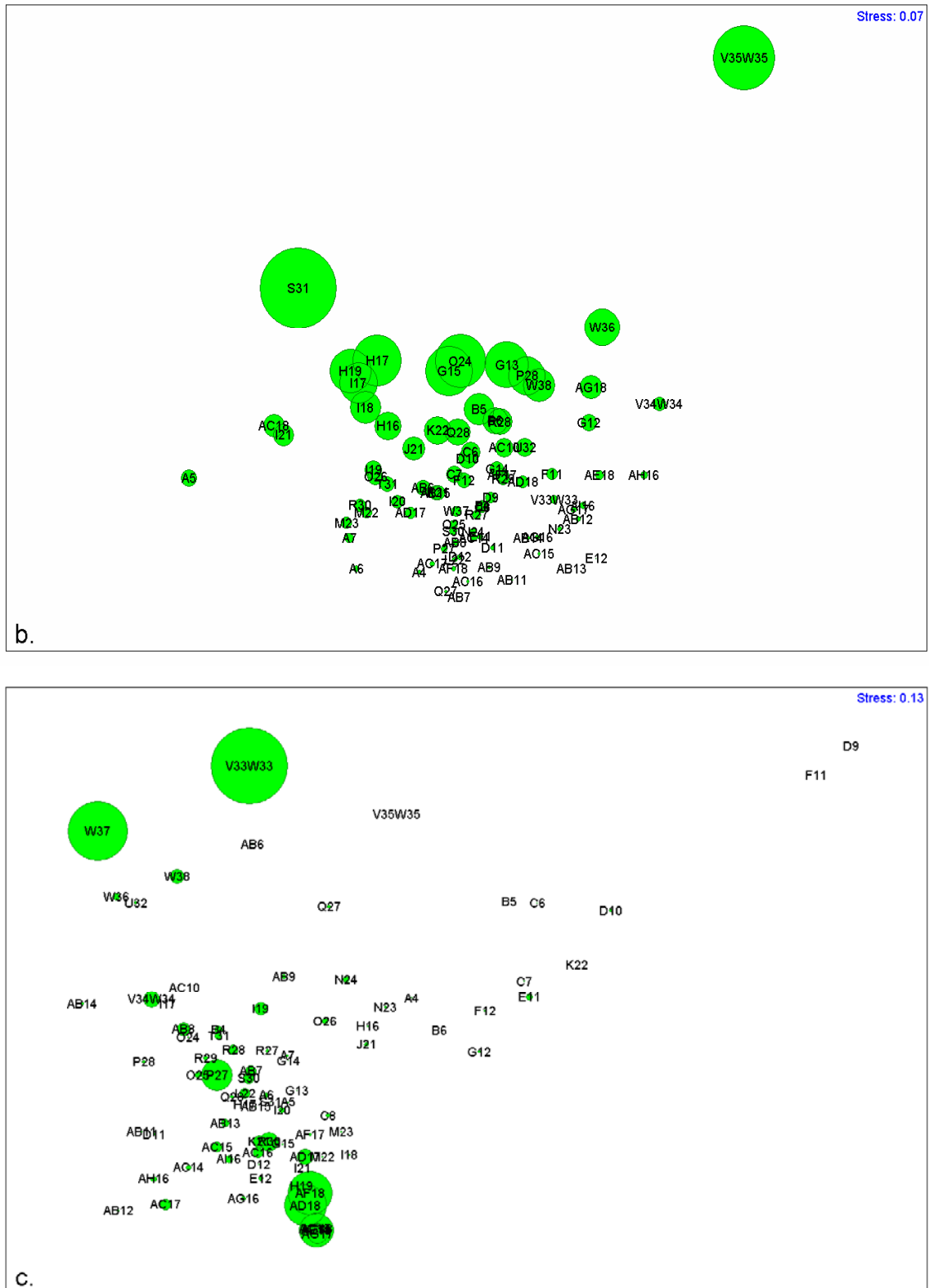
Figure 13: Results of *n*MDS for the total CPUE of 21 estuarine fish species groups.

(a) wetland connectivity index (CONNECT 100 m), (b) CONNECT 1000 m, and (c) total wetland habitat area from 90 selected fish catch grids (based

I found that high connectivity of landscape generally matched areas with high annual rainfall. There is a known link between long term average rainfall along the coast and the distribution of mangroves (Fosberg, 1961) and salt marsh areas, with large areas of saltpans in the drier areas such as the Gulf of Carpentaria and larger areas of mangroves in the wetter areas like Cairns (Fig. 10).

nMDS plots showed clear separation between grids of high CPUE and high wetland CONNECT (100 m) values (Fig. 13) with grids along the EC, in particular, S31, V35W35, W36, H19 and H17 having high CPUE and high connectivity values (Fig. 13a). Similar outcomes were obtained for wetland CONNECT (1000 m), (Fig. 13b). A PCA with and without CONNECT (1000 m) revealed no change in the eigenvalue loadings for the EC but reduced cumulative variation explained by the eigenvalues for the GOC by 10 %. I chose CONNECT (100 m) for further analyses. An nMDS plot based on CPUE distribution against total area of tidal wetlands showed no clear trend between total CPUE and wetland area for the 90 investigated fish catch grids (Fig. 13c). Notwithstanding, some grids with high CPUE values had large areas of tidal wetlands.

4.3.2   Fish-habitat linkages for the east coast

Mangrove patch density, wetland patch density and salt marsh patch density were the best contributing factors for the east coast of Queensland, using the total catch-per-unit-effort and 38 parameters resulted in a forward stepwise regression procedure ($r^2 = 0.68$, $P < 0.01$, n = 54). The link between the habitat characteristics and estuarine fish CPUE was further explored using BIO-ENV. The BIO-ENV procedure resulted in five parameters, namely, wetland CONNECT (100 m), wetland patch density, flats (number of patches), seagrass perimeter to area ratio and mangrove (number of patches) providing the strongest relation (r = 0.36) between CPUE and the 38 parameters. All BIO-ENV results were based on Euclidean distance and no transformation of the entered parameters.

PCA analyses provided three PCs for the EC habitat parameter with eigenvalues > 1.7 and together accounted for 69 % of the standardised variance of the data set (Table 10). The first principal component (PC 1) had an eigenvalue of 5.7 and contributed to 34 % of the variation of the whole matrix. It was dominated by weights of total wetland area followed by channel area, channel perimeter to area ratio, flats perimeter to area

Table 10: Results of principal component analysis of estuarine habitats, physical variables and latitude including coefficients for generating linear combination of variables for orthogonal principal components

Eigenvalues	PC1	PC2	PC3
East Coast			
Eigenvalues	5.75	4.09	1.78
% Variation	33.8	24.1	10.5
Cumulative % variation	33.8	57.9	68.4
Pearson correlation for			
PC1-PC2 and PC1-PC3		-0.13	0.04
Gulf			
Eigenvalues	6.69	5.25	2.62
% Variation	31.9	25.0	12.5
Cumulative % variation	31.9	56.9	69.4
Pearson correlation for			
PC1-PC2 and PC1-PC3		0.3	0.02

ratio and seagrass area. This axis is representative of the actual amount of wetland habitat in the landscape and its configuration. The second principal component (PC 2) comprised wetland (number of patches), salt marsh (number of patches) and wetland perimeter to area ratio, which accounted for an additional 24 % of the variance and had an eigenvalue of 2.16. PC 2 was therefore identified as representing the number of wetland habitat patches in the landscape. Finally, the third principal component (PC 3) (eigenvalue 0.7) was dominated by mangrove ENN, length of connected edge to area ratio and latitude representing connectivity of mangroves and other coastal habitat types (Fig. 14). Detailed species or species group specific analyses with the PC scores showed that barramundi (*Lates calcarifer*) CPUE from the EC was best explained by PC 2 and therefore the number of wetland patches, mangrove connectivity and length of connected habitat ($r^2 = 0.34$, P < 0.01). The number of wetland habitat patches fitted best with the category bay prawns CPUE. Bream (e.g., *Acanthopragus australis*) CPUE and whiting (*Sillago* spp.) total catch was best explained by PC 1 and therefore by the

amount of wetland habitat. Dart (*Trachinotus* spp.) CPUE, mullet (Mugilidae) total catch, king prawns (*Penaeus esculentus, P. plebejus)* total catch and tailor (*Pomatomus*
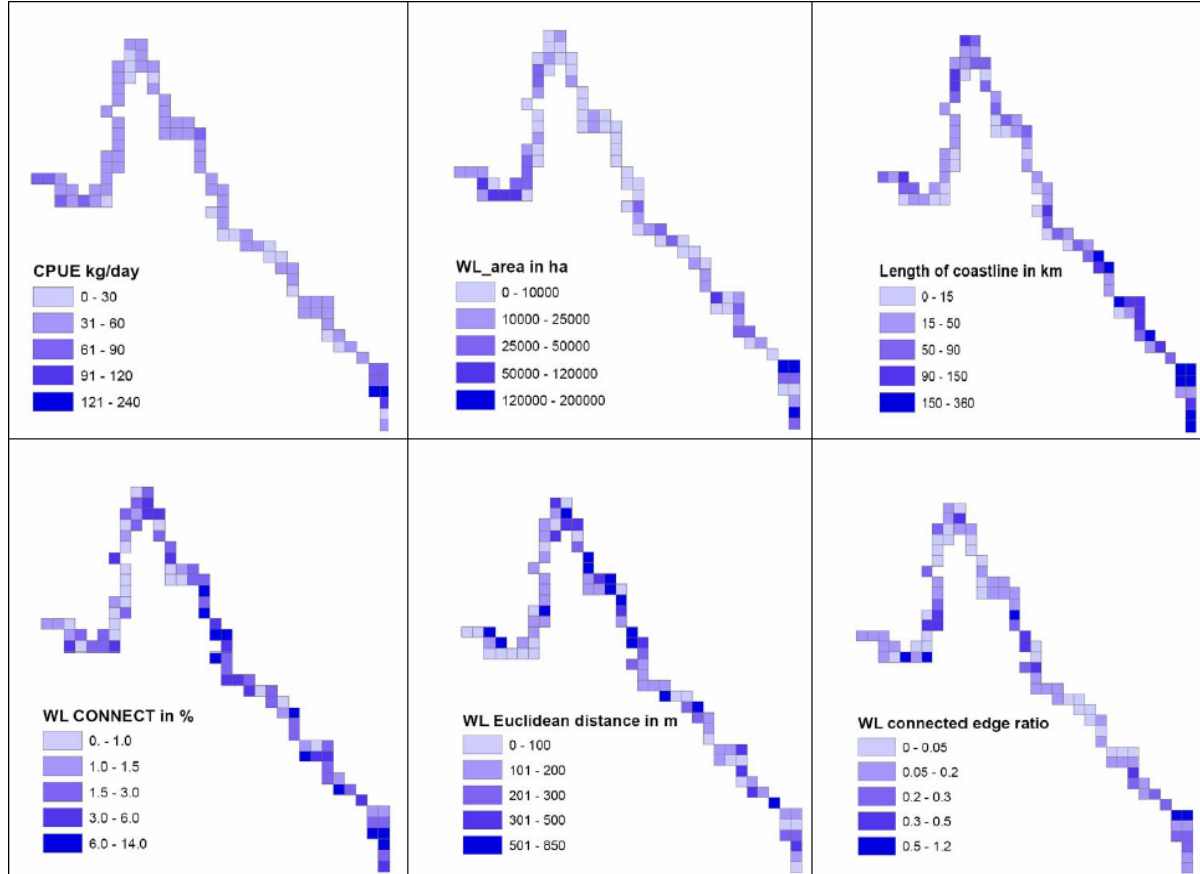
Figure 14: Distribution of CPUE, wetland area (mangroves, salt marsh, mud- and sandflats and channels), length of coastline, wetland CONNECT, Euclidean distance between patches in m, ratio between length of shared edges between patches and total area within 90 selected sections

saltatrix) total catch were best explained by wetland habitat patches and mangrove ENN, length of connected edge to area ratio and latitude (PCs 1 and 3). Bait prawns (Penaeidae) CPUE and blue swimmer crab (*Portunus pelagicus)* CPUE showed the best correlation with scores from PC 1, 2 and 3. A weak non-significant positive correlation was identified for grunter (*Pomadasys maculatum*) CPUE with PC 2 score values ($r = 26$, $P < 0.8$). Mackerel (Scombridae) CPUE and total catch as well as coral trout (*Plectropomus leopardus*) CPUE and total catch were not explained by any of the score

values, nor were correlation identified with the score values. This further demonstrated that the PC score values were robust and able to exclude spurious correlation.

4.3.3   Fish-habitat linkages for the Gulf of Carpentaria

Forward stepwise regression for the total Gulf of Carpentaria CPUE and 38 parameters gave the best explanation for CPUE distribution by seagrass patch density, the number of estuaries and salt marsh CONNECT (100 m) ($r^2 = 0.60$, $P < 0.01$, $n = 28$).

Results of BIO-ENV for the GOC showed the best Spearman's rank correlation between CPUE and 38 parameters ($r = 0.58$) for: (1) length of connected edge to area ratio; (2) number of estuaries; and (3) seagrass patch density, in this rank order. A separate PCA for the GOC derived three PCs from habitat attributes with eigenvalues > 2, which together accounted for 70 % of the standardised variance (Table 10). The first principal component (PC 1) had an eigenvalue of 6.7 and contributed to 32 % of the variation of the whole matrix. It was dominated by river length, wetland CONNECT, wetland perimeter to area ratio, length of connected edge to area ratio. This suggests that this axis is representative of the actual amount of connected wetland habitat in the landscape and its configuration. Salt marsh (number of patches) and perimeter to area ratio and latitude dominated the second principal component (PC 2), which accounted for an additional 25 % of the variance and had an eigenvalue of 5.5. PC 2 was therefore identified as being representative of salt marsh habitat patches of the landscape and latitude. Finally, the third PC (eigenvalue 2.6) was dominated by mangrove number of patches, wetland and seagrass perimeter to area ratio, mangrove CONNECT and wetland CONNECT (100 m) thus interpreted to reflect number, shape and connectivity of wetland patches (Fig. 15a). The species or species group analyses using the PCs showed that mud crab (*Scylla serrata*) CPUE from the GOC was best explained by wetland connectivity, number of salt marsh. patches and latitude. Salt marsh perimeter to area ratio and salt marsh number of patches were most important to explain mullet (Mugilidae) CPUE, blue threadfin (*Eleutheronema tetradactylum*) total catch and king threadfin (*Polydactylus macrochir*) total catch distribution. Barramundi CPUE was best explained by mangrove (number of patches), mangrove CONNECT, wetland CONNECT (100 m), wetland perimeter to area ratio and seagrass perimeter to area

ratio. Results were similar for the EC where barramundi was best explained by wetland patches (score 2) and connectivity parameters (score 3) (Fig. 15b). All other groups
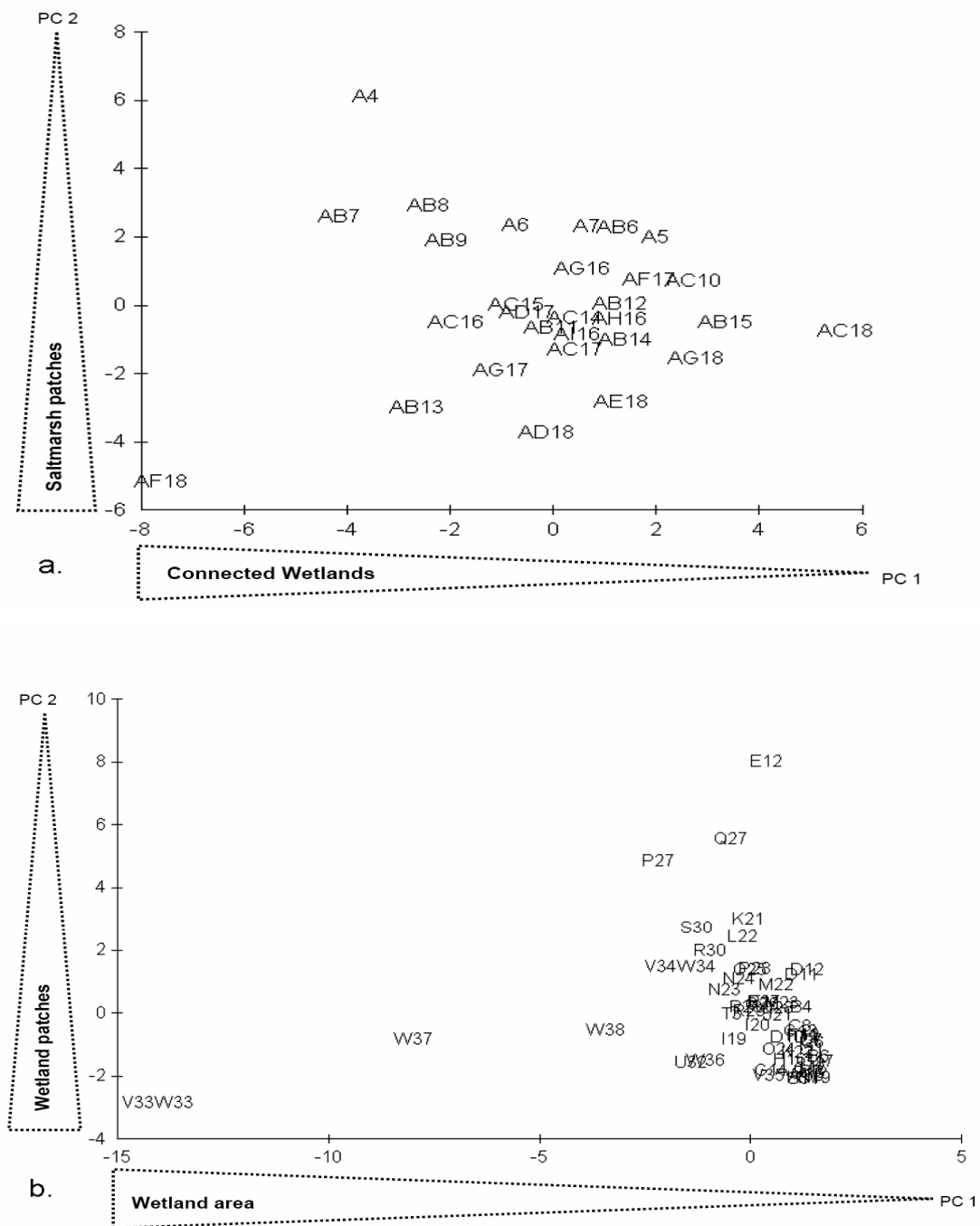
Figure 15: The distribution of fish catch grids according to 38 coastal parameters.
For the Gulf of Carpentaria (a), with PC 1 representing wetland connectivity and PC 2 salt marsh patches and latitude; and for the east coast of Queensland (b) PC 1 represents wetland area and PC 2, wetland patches. Values increase from right to left and from top to bottom

Table 11: Significant r^2 values (*Coast, [†]Gulf) for fish catch and CPUE based on forward stepwise regression analyses for three scores derived from principal component analyses. Score 1 = wetland habitat area; Score 2 = wetland patches, Score 3 = length of connected edge to area ratio and latitude; Score 1 = wetland connectivity, Score 2 = salt marsh patches and latitude; Score 3 = number, shape and connectivity of wetland patches

Species group	r^2	Multiple r^2	P	n
Bream (CPUE)*	.34 (Score 1)	-	0.001	38
Whiting (catch)*	.64 (Score 1)	-	0.001	40
Dart (CPUE)*	.20 (Score 1)	.44 (Score 3)	0.001	36
Mullet (catch)*	.24 (Score 1)	.35 (Score 3)	0.001	49
Prawns King (catch)*	.30 (Score 1)	.36 (Score 3)	0.001	53
Tailor (catch)*	.31 (Score 1)	.56 (Score 3)	0.05	16
Barramundi (CPUE)*	.21 (Score 2)	.34 (Score 3)	0.01	49
Prawns Bay (CPUE)*	.27 (Score 2)	-	0.001	41
Prawns Bait (CPUE)*	-	.40 (Score 1,2,3)	0.05	18
Blue Swimmer (CPUE)*	-	.22 (Score 1,2,3)	0.05	39
Mud Crabs (CPUE)[†]	.19 (Score 1)	.43 (Score 2)	0.01	27
Mullet (CPUE)[†]	.20 (Score 2)	-	0.05	20
Threadfin Blue (catch)[†]	.22 (Score 2)	-	0.01	28
Threadfin King (catch)[†]	.22 (Score 2)	-	0.01	27
Barramundi (CPUE)[†]	.25 (Score 1)	.38 (Score 3)	0.05	28

showed no significant r^2 values for the PC scores (Table 11). I found no correlation for mackerel (Scombridae) CPUE and total catch for PC score values in the GOC. A coral trout fishery was not present.

4.4   Discussion

4.4.1   Connectivity - indicator for estuarine habitat value

This study applied an explanatory landscape approach to identify the major environmental variables influencing the observed spatial patterns in fish and prawn catches in Queensland, Australia. I was able to demonstrate an association between broad-scale seascape connectivity and fish catch. The study supports the paradigm that heterogeneity and connectivity of coastal marine habitats at a broad spatial scale appears

to explain a significant proportion of the spatial variability in fish catch (MacArthur and Wilson, 1967; Kotliar and Wiens, 1990; McCoy and Bell, 1991). More specifically, patch density and connectivity metrics explained a large and significant proportion of the variation in fish, crab and penaeid prawn CPUE distribution. Such seascape characteristics were equally important to size and shape of habitats, indicating that the role of estuarine habitats is not independent of their configuration (Guest and Connolly, 2006). The findings could guide the construction of marine protected area networks of various sizes and spacing to maintain ecosystem services and avoid further reduction of connectivity by habitat destruction. Regions with high connectivity scenarios should receive most attention when establishing habitat protection zones e.g., grid S31 and V35W35 (Fig. 13a and 13b).

Optimal configurations of habitat types may be identified for particular species and assemblages, which function to offer a chain of essential resources supporting species through tidal movements and ontogenetic shifts (Nagelkerken et al., 2001; Cocheret de la Morinière et al., 2003; Gillanders et al., 2003; Mumby et al., 2004). When the close spatial proximity of resources (e.g., settlement substratum, food and refuge) results in higher densities of resources, 'landscape complementation' occurs (Dunning et al., 1992; Schlosser and Angermeier, 1995). McIvor and Odum (1988) found the largest fish populations in areas of a salt marsh that were opposite tidal creeks, where pools offered refuge at low tide. The CONNECT and length to connected area ratio appeared to be helpful in finding optimal configuration of habitat types on a broader scale. However, similar to Pittman et al. (2004a) the total area of coastal habitats was still an important parameter explaining fish catch distribution. In comparison, terrestrial studies have found similar results where abundance of birds was best explained by the degree of patch connectivity (Uezu et al., 2005).

For some species, a preference for patchy structure may be a trade-off between minimising the risk of predation, whilst maximising foraging efficiency (Holt et al., 1983). In my study, *Sillago* spp. total catch was well explained by the amount of wetland habitat ($r^2 = 0.64$, $P < 0.001$). These fish are widely distributed generalists with fast-swimming demersal juveniles and adults that readily move across a wide range of estuarine habitat types (McKay, 1992). Prawns (Penaeidae*)* are also known to use a variety of estuarine habitats (Dall et al., 1990; Halliday, 1995; Loneragan et al., 2005). Their CPUE was best explained by the number of wetland habitat patches, mangrove ENN and length of connected edge to area ratio suggesting significant edge effects

(Minello et al., 1994). The catch of blue (*Eleutheronema tetradactylum)* and king threadfin (*Polydactylus macrochir*) in the GOC was best explained by the number of salt marsh patches. The species are known to forage in shallow water habitats on crustaceans and nematodes (Blaber et al., 1995). Shallow waters are also attributed to waters adjacent to salt marsh/salt pan which is the dominating habitat type in the GOC. A link to salt marsh patch characteristics is therefore not surprising.

A significant relationship between tiger prawns (*Penaeus monodon, P. semisulcatus*) and any of the PC score values was not identified. Tiger prawns prefer seagrass for shelter and foraging as juveniles (Haywood et al., 1995). But, seagrass area was underrepresented in the wetland metrics due to the overlap with channel area.

I speculate that high fish catch may have been maintained through diversity of different estuarine habitats arranged with a high number of different patch types in close proximity. Thus representing supplementary resources throughout the home range of the investigated species (Dunning et al., 1992) and a concomitant increase in connectivity where suitable habitat types are adjacent to one another (Jonsen and Fahrig, 1997). The distribution of tidal wetlands partly reflects physical attributes of an estuary including tidal range, rainfall, temperature, ocean currents and water quality which may also alter fish abundance and distribution (e.g., large tidal flats and salt pans in areas with high tidal energy and low relief energy). Ley (2005) found that higher fish abundance was evident in tide dominated estuarine systems in north Queensland, which are associated with larger mangrove areas and a wider entrance compared to wave dominated systems.

4.4.2   *Limitations of data and spatial pattern metrics*

There are clear limitations associated with the use of structural connectivity metrics. Nearest neighbour measures, for instance, are less likely to detect a significant effect of connectivity and are more sensitive to sample size and spatial resolution than other, more complex connectivity metrics (Moilanen and Nieminen, 2002). Such "area-sensitive" metrics generally perform better than those that lack area considerations (Moilanen and Nieminen, 2002; Tischendorf et al., 2003). CONNECT is one such useful metric for measuring connectivity in landscapes (Pfister, 2004). CONNECT is most realistic for organisms that utilise a broad range of habitats, perceive few boundaries and are limited by dispersal distance, such as some generalists, e.g., bream

(*Acanthopagrus* spp.) and whiting (*Sillago* spp.). In this chapter, I developed a connected habitat edge to area ratio reflecting the importance of edge and habitat patch distance as a useful metric. However, it is prone to errors due to a manual selection procedure within ArcGIS 9.1. Other algorithms and indices are available to measure landscape connectivity (Mumby, 2006).

Landscape connectivity changes with spatial scale of investigation (Moilanen and Hanski, 2001) and is therefore a relative value. Care must be taken in comparisons, however, as the interpretation of habitat metrics from mapped data may vary significantly between maps depending on mapping accuracy (Loehle and Wein, 1994; Mumby, 2001). Calculations of mangrove perimeter for instance are highly sensitive to the pixel size employed; the fractal geometry of a structure leads to larger estimates of perimeter as the unit of measurement (pixel size) decreases (Sugihara and May, 1990). Another limitation of the current study is that data were used at a single spatial resolution and quantified within a grid cell selected at a scale determined by ecological criteria but instead based on available data and management units.

In the analysis, a total of eigth species groups did not show any significant relation with the PCA scores. This suggests that a linear model may not be appropriate in all cases and that the fish catch for a number of species groups were too low. For example, the catch for mangrove jack (*Lutjanus argentimaculatus*) and milkfish (*Chanos chanos*) were too low for further analyses. Splitting the fish catch data reduced the catch per grid to a level where analyses for some grids were not possible - a problem also identified in other studies (Manson et al., 2005b).

The use of 30-nautical-mile catch grids comes to its limitations when trying to define values for particular estuaries. I tested available 6-nautical-mile catch grid records (2001-2005) for 41 grids where single estuaries could be assigned to one particular grid. However, due to fewer records per grid the amount of catch was too low for further analyses.

4.4.3 Management and research implications

For the coast of Queensland I have identified the most relevant areas and habitat configuration explaining fish catch distribution, thereby providing guidance where future fisheries research and protection efforts should focus. A similar approach is

required for analyses at finer spatial scales to enable catch information to be related to particular estuarine habitats within an estuary and to allow for a full understanding of the importance of habitat connectivity for fisheries. As a further step, a fine scale investigation of fish species movement, abundance and assemblages in these locations is necessary. Telemetry studies (McCutcheon et al., 1994) may provide useful information on movement behaviour leading to the determination of species-specific (or life-stage specific) functional connectivity and allow investigation into the ability of species to adjust the scale of their movements to the spatial configuration of their habitat (Lyons and Lucas, 2002; Taylor et al., 2006) (see Chapter 5). Furthermore empirical studies of isolated habitats may help to shed light on the extent to which larval dispersal and movement of individuals throughout their lives limit the distribution of organisms, e.g., how well can any single combination of size and spacing of protected areas serve all the target species? Such data are critical to discussions of connectivity between potential marine protected areas, persistence of fish species within areas, and determination of functional connections between habitat types. Little is known about the ability or tendency of individuals to traverse unvegetated substrata between patches of estuarine habitats such as mangroves, or seagrass patches.

The demonstrated difference in habitat connectivity and other ecosystem-level habitat characteristics for various coastal Queensland regions, and their implications for fishery production in these areas, should be further investigated through long-term monitoring. The following questions require investigation and/or clarification: (1) What is the role of structural connectivity at different spatial scales? (2) To what extent can refined metrics improve the understanding of habitat connectivity for fisheries? (3) What are times of residency of fishery species in different habitats? One of the greatest challenges in marine conservation management remains the definition and establishment of fisheries habitat protection zones at appropriate scales for regional, biogeographic to metaestuarine systems (Ray, 2005), and how to achieve this in the shortest possible time frame.

CHAPTER 5

MOVEMENT OF ESTUARINE FISH IN A SUBTROPICAL MANGROVE CREEK – THE IMPORTANCE OF HABITAT CONNECTIVITY

5.1   Introduction

Knowledge of patterns, drivers, and mechanisms of nekton movements in estuaries is essential to our understanding of how fish use estuaries for nurseries, feeding and other purposes (Beck et al., 2001) as well as understanding the importance of tidal wetland connectivity (Sheaves, 2005). Current research on estuarine ecosystems suggests broad interdependence between habitats (e.g., seagrass and mangroves) (Nagelkerken et al., 2001; Jelbart et al., 2007). Mangrove habitats support diverse and abundant assemblages of fish. For example, studies of mangrove fish assemblages have recorded at least 79 species in Taiwan (Kuo et al., 1999), 49 in Brazil (Barletta et al., 2003), and 42 species in Australia (Morton, 1990; Clynick and Chapman, 2002). Many (up to 73 %) of the species of fish sampled in mangrove habitats are of commercial interest (Laroche et al., 1997), although patterns vary strongly with season (Laroche et al., 1997) or patch size (Clynick and Chapman, 2002).

   Mangroves are considered important habitats for juvenile fish (Robertson and Duke, 1990). Juvenile fish are attracted to the spatial refuge from predators provided by the structural complexity of the roots (Robertson and Duke, 1990), the shade and turbidity (Cyrus and Blaber, 1992) as well as abundance of food (Chong et al., 1990). Due to tidal ranges between 1 and 3 m along the coast of Queensland, few fish can use mangroves exclusively as habitat for either feeding or predator avoidance, and most are forced to use alternative nearby habitats during low tide, establishing a temporally variable habitat mosaic (Sheaves, 2005). I therefore expect that creeks connecting mangrove and salt marsh habitats to the main estuary channel function as "highways" for fish, connecting different habitat patches with short-term semi-daily migrations in response to changing tide levels (Vance et al., 1996a). The value of connectedness (specifically "hydrologic connectivity" of surface waters (Pringle, 2003)) has already

been shown to explain fish catch distribution on larger scales in Queensland, Australia (Chapters 3 and 4). Studies on subtidal movements of a number of species have revealed tidal, diel, and/or seasonal migrations as well as residency in certain areas for days to months, e.g., for sciaenids (Weinstein and O'Neil, 1986), pinfish (*Lagodon rhomboides)* (Potthoff and Allen, 2003), glassfish (*Ambassis jacksoniensis*) (Thomas and Connolly, 2001) and haemulids (Verweij and Nagelkerken, 2007).

Connectivity is mediated through tidal inundation (Jelbart et al., 2007). Caribbean studies confirm the importance of tidal movements of fish among habitats (Nagelkerken et al., 2001). Other work shows movement patterns between estuarine habitats at different times of day and diel pattern of habitat use (Krumme, 2004; Krumme et al., 2004; Cocheret de la Morinière et al., 2004; Verweij and Nagelkerken, 2007). Such information on habitat use and movement is also essential for many ecological studies and usually requires individual identification of organisms to assess the processes that influence small-scale distribution patterns (McCormick and Smith, 2004).

The methods used for investigating fish movement in tidal wetlands have included visual sensing of tagged fish along transects (Cocheret de la Morinière et al., 2002), tagging and recapture with nets and traps (Sheaves, 1996), repeated net sampling for abundance and assemblages (Jelbart et al., 2007), acoustic tagging (Humston et al., 2005) and by split-beam echosounders (Krumme, 2004). A common problem in using tagging as a tool for estimating habitat connectivity are sparse data, due to generally low recapture probability (Pine et al., 2003). This may be overcome by simultaneous use of new techniques such as underwater digital video cameras (DVCs) (Jan et al., 2007) and automated fish detection systems using passive integrated transponder (PIT) (Zydlewski et al., 2001). PIT tags have been used extensively in freshwater ichthyological research (Harvey and Campbell, 1989; Peterson et al., 1994) for measuring fish movement (Prentice and Park, 1983; Roussel et al., 2000; Achord et al., 2003). Zydlewski et al. (2001) pioneered a pass-through design in a natural stream 8 m in cross section and achieved detection efficiencies averaging 93 %. Remote detection of PIT tags enables accurate recording of relative abundance, habitat use and movements (McCormick and Smith, 2004). In many cases, the use of PIT tags in marine environments has been restricted to manual recapture (Quartararo and Bell, 1992) to determine demographic parameters (Achord et al., 2003). However, a pilot study in a mangrove creek in Florida, U.S.A., successfully used PIT tags for recording the transit of individuals past a specifically placed sensor gateway (Adams et al., 2006b).

The purpose of this study was to quantify movements of fish in and out of a subtropical mangrove creek in Queensland, Australia that provided access to an area of intertidal mangrove forest. I used a modified PIT sensory gateway for saline conditions and combined with simultaneous deployment of underwater DVCs to quantify key habitat use and time of residency of common estuarine species in relation to tidal condition and, using remote sensing data, to determine the dynamics of habitat availability within the creek during the tidal cycle. Associated fish movement should be species and size specific as well as day and night dependent (Sheaves, 1996). The specific aims of the study were to compare juvenile and sub-adult yellowfin bream (*Acanthopagrus australis*), Moses perch (*Lutjanus russelli*), grunter (*Pomadasys kaakan*) and sea mullet (*Mugil cephalus*) in terms of (1) semidiurnal migration; (2) site fidelity; and (3) differences in day/night activity patterns. I also assessed species composition and its relationships with habitat availability based on inundation patterns.

5.2   Methods

5.2.1   Study Area

The study was undertaken in a tidal mangrove creek ~ 6 km upstream from the mouth of the Burrum River (25° 10' S, 152° 37' E) on the east coast of Queensland, Australia (Fig. 16). The Burrum River contains a mixture of both tropical and temperate fish faunas (Kirkwood and Hooper, 2004). Mean fish catch for the Burrum River region was 53 t for 2001-2005, of which > 50 % were accounted by mullets (Mugilidae) (DPI&F, 2007). A declared Fish Habitat Area (McKinnon et al., 2002), the Burrum remains one of south-east Queensland's most undisturbed river systems, and the region has shown high estuarine habitat connectivity (Chapters 3 and 4). The climate of the catchment is subtropical with distinct wet (November to April) and dry (May to October) seasons. Annually, daily water temperatures between 1994 - 2005 at the study site ranged from 16 - 30ºC. Ongoing urban development within the 935 km^2 catchment (BOM, 2007a) threatens the estuarine system; however, substantial areas of seagrass (47 ha), salt marsh (147 ha), mangrove (515 ha) and open channel (2013 ha) habitats are present within the estuary (Meynecke, unpublished data) (Appendix 5). Two weirs and a dam currently

restrict natural freshwater flow from upstream into the estuary and oceanic water dominates the system.
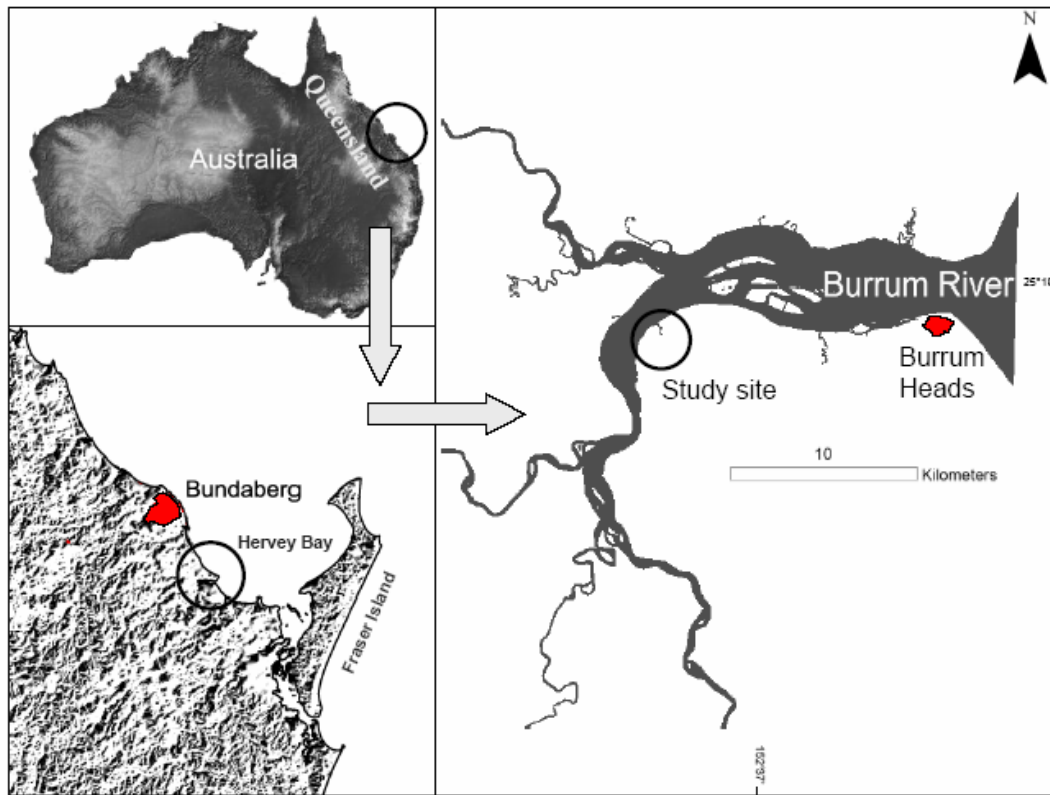
Figure 16: Location of the Burrum river study site (based on Geoscience Australia, 2005)

The mangrove creek selected for this study drains a 20 ha depression with about 10 ha of mangroves and salt marsh and is surrounded by terrestrial vegetation (Fig. 17). The creek is ~ 800 m long with width ranging from 8 m to > 20 m. The substrate is mostly fine muddy sediment. At the mouth of the creek, the mean water depth averages 1.5 m, but carries no water at low tide and reaches 2.5 m in depth at high tide. The creek is fringed by the mangroves *Aegiceras corniculatum* and *Avicennia marina* and in higher tidal areas by *Ceriops tagal*. The system has high flushing rates when exchanging water from the Burrum River; current velocities can reach 1 m s^{-1} (Meynecke, unpublished data). This exchange maintains salinities around 35 ppt during most times of the year. The tidal cycle is semidiurnal with the highest astronomical tide at ~ 2 m above Australian Height Datum (AHD, 0 m equals Mean Sea Level) (MSQ, 2000).

5.2.2   Habitat mapping and elevation data

The habitat distribution of the study site was derived from rectified three-band, true colour (RGB) aerial imagery (May 2006, Fugro Spatial Solutions Pty Ltd) with a pixel size of 1x1 m. Habitats (mangroves, salt marsh, channel and terrestrial vegetation) were digitized on screen in ArcGIS 9.1 based on a 1:2000 scale followed by ground truthing.
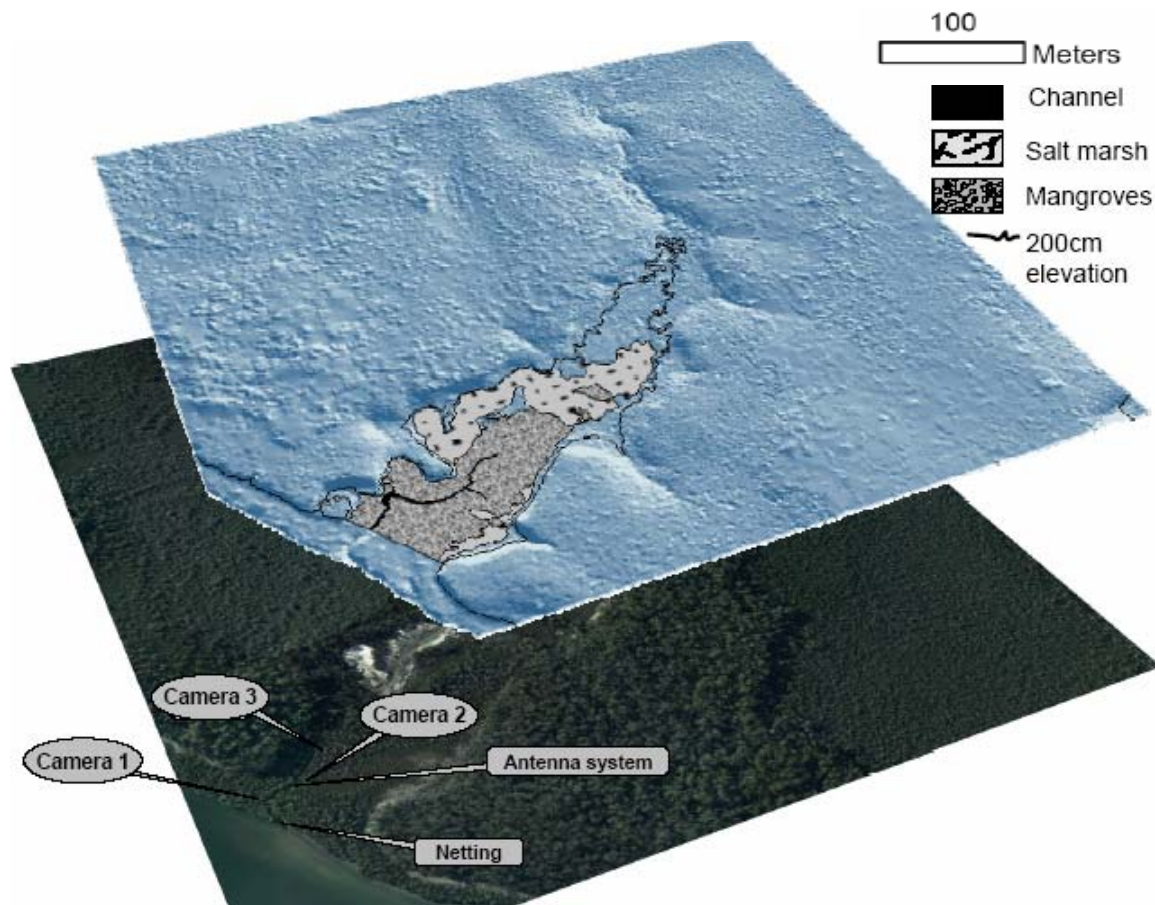
Figure 17: Digital elevation model of the mangrove creek showing salt marsh and mangrove distribution as well as the 200 cm elevation contour line for the watershed

DEMs represent topography by recording the mean elevation for each cell in a grid overlain atop a landscape. High-resolution DEMs, generated from LIDAR remote sensing data, have been used successfully in wetlands to study surface hydrology (Bates and De Roo, 2000). I obtained LIDAR data for the site from Fugro Spatial Solutions Pty

Ltd (flying height 2000 m, vertical accuracy of 0.2-0.3 m) (Zhang et al., 2003). The data were collected on 28th May 2006 at low tide, when the channel was dry. Using bare earth returns from the LIDAR sensor, which were projected in Geocentric Datum of Australia (GDA) 94 (Zone 56), I created a DEM of the Burrum River estuary at a resolution of 2.5x2.5 m (pixel size). The resulting grid had occasional areas where the LIDAR signal did not return ground-surface data. These areas were generally either small (typically < 0.01 km^2), or interspersed with successful LIDAR returns such that missing data were seldom > 30 m horizontally from successful LIDAR readings. Such areas of missing data were filled using a "Natural Neighbour" interpolation (ArcGIS 9.1, ESRI) using the LIDAR data points immediately adjacent to the areas of missing values. The study creek's catchment was extracted from the resulting contiguous digital elevation model (Fig. 17). I estimated the timing and duration of habitat inundation within the creek's catchment by classifying the DEM into 15 elevation categories between -0.3 and 2.3 m (the approximate range of water surface elevation at the stream's mouth during the maximum annual tidal cycle). Using these categories, I estimated patterns of inundation for each of the 15 water levels, and determined the fraction of each habitat in the creek basin (stream channel, salt marsh, and mangrove) that would be inundated at each water level. This analysis provided estimates of intertidal fish habitat availability across the observed range of water levels at the mouth of the creek associate with fish passage.

5.2.3   Fish catch data

Long-term recreational fish catch data provided by the Burrum Heads Fishing Club were used to determine seasonal species composition and their relative abundance in the river. The data set provided important insights to whether the wet season covered during my survey reflected the species composition of the river. All data were based on fish caught within the Burrum River and catch in kg, species, date and number of anglers have been continually recorded from 20 Sept 1997 to 06 Jan 2007 every three weeks. The data were entered into a database and analysed for total abundance and seasonal species composition using catch per day per angler. The standardised data should be sufficiently robust to reflect the rank order of abundance of fish species (Terceiro, 2003). As the correct species name and weight were important for the anglers'

competition the error for misreporting of the dominant species is expected to be minor. Mullets (Mugilidae) were not an accredited angler species and were therefore not included in the data set.

5.2.4   PIT tag reader design and operation

An autonomous antenna system, consisting of an open loop inductor coil antenna connected to tuning circuits (which were connected to a reader unit and a data logger, Fig. 18), was set ~ 100 m upstream from the mouth of the mangrove creek (see also Fig. 17). The electronics were housed in weatherproof enclosures mounted on a platform in a
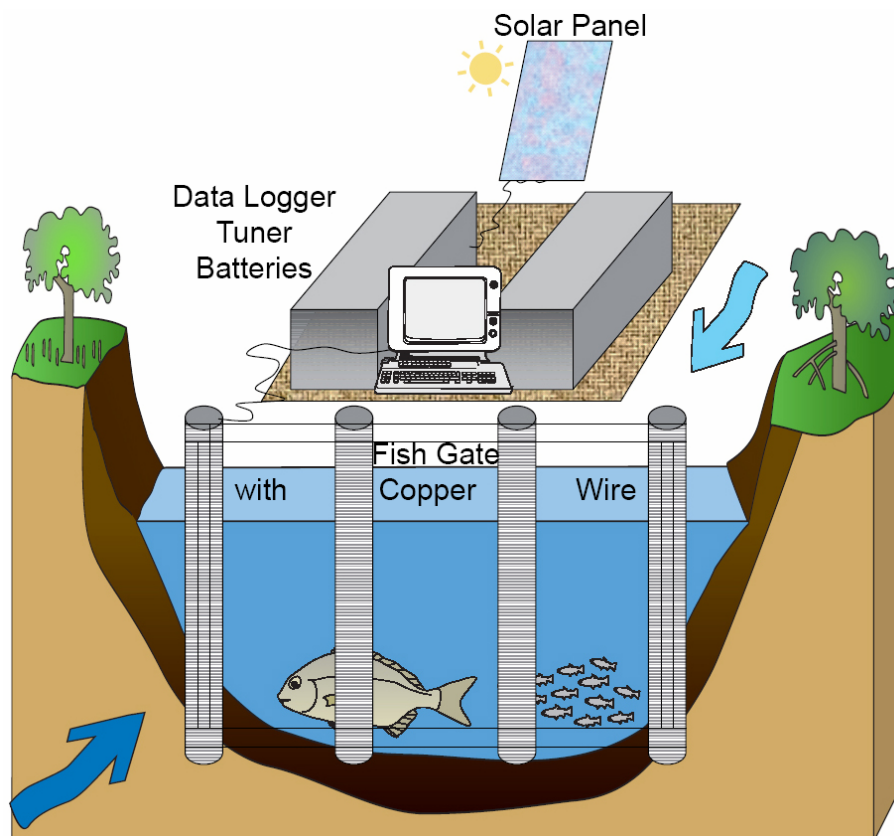
Figure 18: A schematic illustration of the fish gate. The platform was mounted on a nearby mangrove tree. Arrows indicate tidal movement

nearby mangrove tree. The system was powered by a 12 V battery and charged by a 165 W solar panel (BP Solar Pty Ltd) allowing continuous use without maintenance. The antenna stretched across the entire mangrove fringed creek width of 8 m (suspended by

wooden stakes) so that fish swimming in and out of the creek from the main river had to pass through the antenna system to enter or leave the mangrove and salt marsh area. The antenna operated on a half-duplex system at 134.2 kHz. If a PIT tag is present, the reader generates a close-range, electromagnetic field that immediately activates the tag, which transmits a unique alphanumeric code (Gibbons and Kimberly, 2004) to the data logger (Appendix 6 and 7).

To overcome the problem of a reduced detection radius of the antenna due to saline waters (35-38 ppt) the gate voltage was set between 300 and 1000 V to achieve maximum detection range from the antenna (Steig et al., 2003). A second gate was established 2 m from the first gate to (a) increase detection rate and (b) monitor travel speed and direction. PIT tags can be read even while an animal is moving with speeds up to 3.6 m s^{-1} (Prentice et al., 1990). The second gate was connected to the same system. The gates were tested for efficiency with PIT tags held within the magnetic field to check for gaps in the detection area which resulted in 100 % coverage. In addition 10 PIT-tagged fish were released at a peak tide event upstream of the gate.

I operated the gate for 16 weeks, between Jan 2007 and Apr 2007. Fish were first captured using gill nets with a mesh size of 3-4 cm, as well as hook and line, at the mouth of the mangrove creek to capture fish from mangroves. These are common methods to capture fish around mangrove habitats, because of their largely impenetrable nature (Hindell and Jenkins, 2004). Fish were measured, identified, and time, location and the physical parameters of (1) temperature; (2) salinity; and (3) turbidity (Secchi disc) of the water column recorded before release. For tagging, fish were placed on a foam board, measured, and a PIT tag (Texas Instruments) inserted into the abdominal cavity with a tag gun and the function of the tag tested with a handheld antenna. Experience on other species (Baras et al., 2000; Jepsen et al., 2002) indicate that post-tagging sutures were not necessary. Because of the size of the tag (23 mm, 0.6 g in air), only individuals > ~ 120 mm fork length (FL) were tagged, with the tag weight always being < 2 % of their total body weight, the approximate threshold for effects on growth or survival (Begout et al., 2003; Skov et al., 2005). I caught a total of 101 fish with gillnets and hook and line at the entrance of the mangrove creek. I released 26 juvenile (< 120 mm FL) yellowfin bream and Moses perch without PIT tags. The remaining 75 fish consisting of ten different species were PIT tagged.

Continuous tidal range data at the fish gate were captured by a pressure sensor (DR 1050 Richard Brancker Research Pty Ltd) and later overlayed with fish detection at the

gate. The elevation of the gate was determined relative to the LIDAR-derived digital elevation model. Multiple detections of an individual PIT tag within a 2 s time period were removed from the data. To estimate travel speed of fish, I used detections at the second gate only if they occurred within 5 min of detection at the first gate. I also determined time spent upstream of the gate (in the tidal creek) using detections at the gate. Detections were only considered valid when there were at least two detections, one at incoming tide and one at outgoing tide during one tidal cycle (assuming that fish had to leave the creek at least once to avoid being stranded in the dry creek). In addition to PIT tags, I also applied T-bar anchor tags for external identification.

5.2.5   Underwater Digital Video Cameras

Three commercially available DVCs (Sony DC handycam N50) were deployed underwater (in custom-made waterproof DVCs housings) during out going tides beneath mangrove stands. Because the DVCs required high light and high tides, I

Figure 19: Snap shot from DVC of juvenile yellowfin bream (*Acanthopagrus australis*) in mangroves

deployed the DVCs during the day between 9 am and 1 pm. Three sites were chosen for DVC deployment in the lower, mid and upper channel and located ~ 25 m from the

channel into the mangroves (Fig. 17). Cameras were deployed 3 times, each for 1 h, over a period of three months. The cameras were pointing towards mangrove roots at a 90° angle from the ground (Fig. 19). The captured video data were later visually analysed. I recorded the presence of fish, direction of fish movement relative to water flow defined as against or with the current, species and species composition to compare with data from the fish gate. The primary objective was to confirm that fish were present in mangroves throughout the study site during the period of use of monitoring using the fish gate. Additional observations from DVC video included active feeding and fast versus slow movement.

5.3   Results

5.3.1   Inundation model and habitat map

The habitat mapping showed a clear distribution of mangroves on lower land surfaces between -30 and 200 cm elevation (Australian Height Datum (AHD)). Salt marsh occurred in higher elevations between 110 and 200 cm above AHD. The tidal creek's watershed contained ~14 ha of land < 200 cm in elevation. Of that, 5.2 ha were covered by mangroves, 4.4 ha by salt marsh, 4.2 ha by terrestrial vegetation and 0.1 ha by creek channels. The 200 cm elevation contour line roughly marked the highest astronomical tide (HAT) (MSQ, 2000). The inundation model based on elevation showed a threshold level for initial mangrove inundation (110 cm) and initial salt marsh inundation (140 cm), above which inundation rates increased rapidly (Fig. 20). The channel was fully inundated at 90 cm above AHD. These thresholds mark the water surface elevation at which associated habitats become available to fish.

5.3.2   Recreational fish catch data

The angler fish catch data recorded a total of 4127 fish from 60 species during 288 fishing days, with a total weight of ~2620 kg. About 90 % of the individuals were from ten species. Almost one third of the catch was from yellowfin bream and 13 % from sand whiting (*Sillago* spp.), independent of the season. On the basis of frequency of

occurrence in angler catches for seasons between 20 Sept 1997 and 06 Jan 2007, the fish community of the Burrum River during the wet season (November to April) showed distinct differences in abundances for flathead (*Platycephalus fuscus*), tailor (*Pomatomus saltatrix*), flounder (*Pseudorhombus* spp.), pike (*Dinolestes lewini*), school mackerel (*Scomberomorus queenslandicus*), grunter.
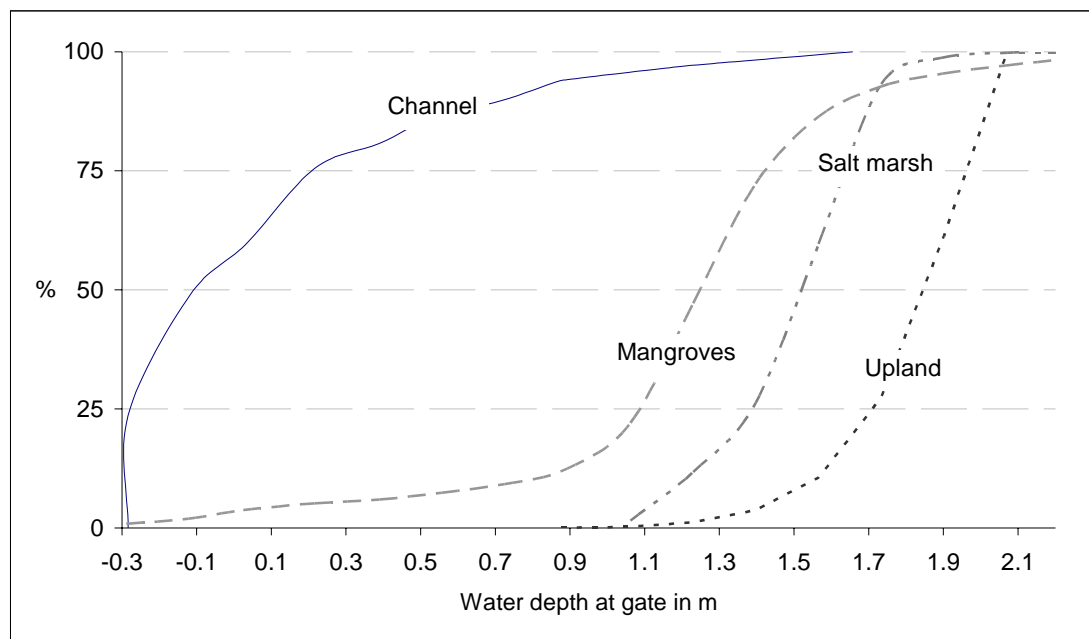
Figure 20: Percent inundation for mangroves, salt marsh, channel and terrestrial vegetation with increasing water height in m at fish gate

(*Pomadasys kaakan*), black bream (*Acanthopagrus berda*), and cod (*Epinephelus coioides*) (Fig. 21). No changes in catch abundance between seasons were observed for yellowfin bream and sand whiting (*Sillago* spp.) suggesting that these species resided in the estuary. More species were caught during the wet season (15 vs 12) (Fig. 21). In addition, there was a pattern over a period of 10 years from less estuarine/brackish water species (e.g., estuary perch - *Macquaria novemaculeata*, grinner - *Trachinocephalus myops*, garfish - *Hemiramphus* spp.) to more oceanic species independent of the season (e.g., school mackerel - *Scomberomorus queenslandicus*, parrot fish – Scaridae) (Fig. 21).

a.
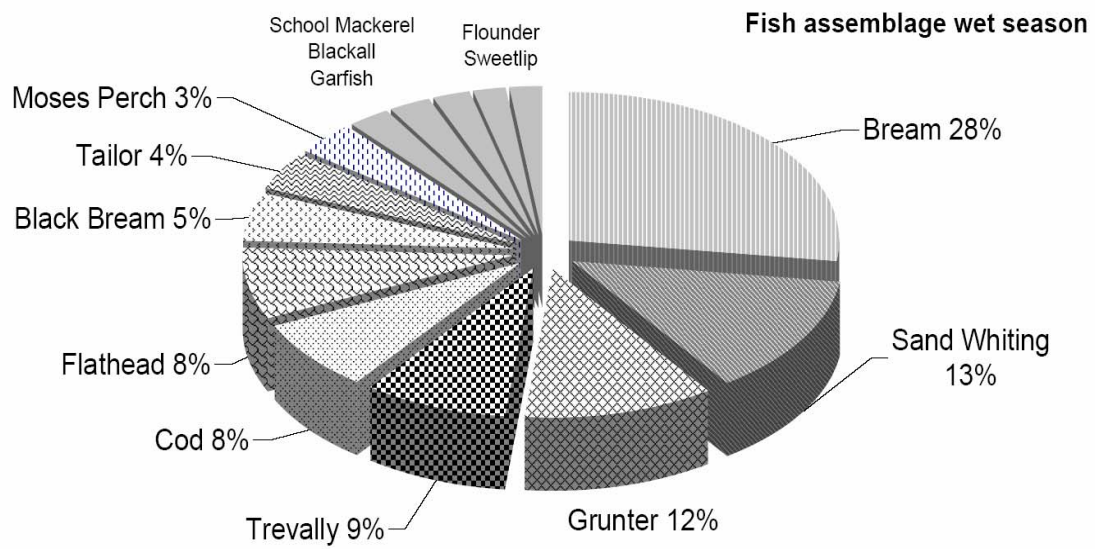
b.
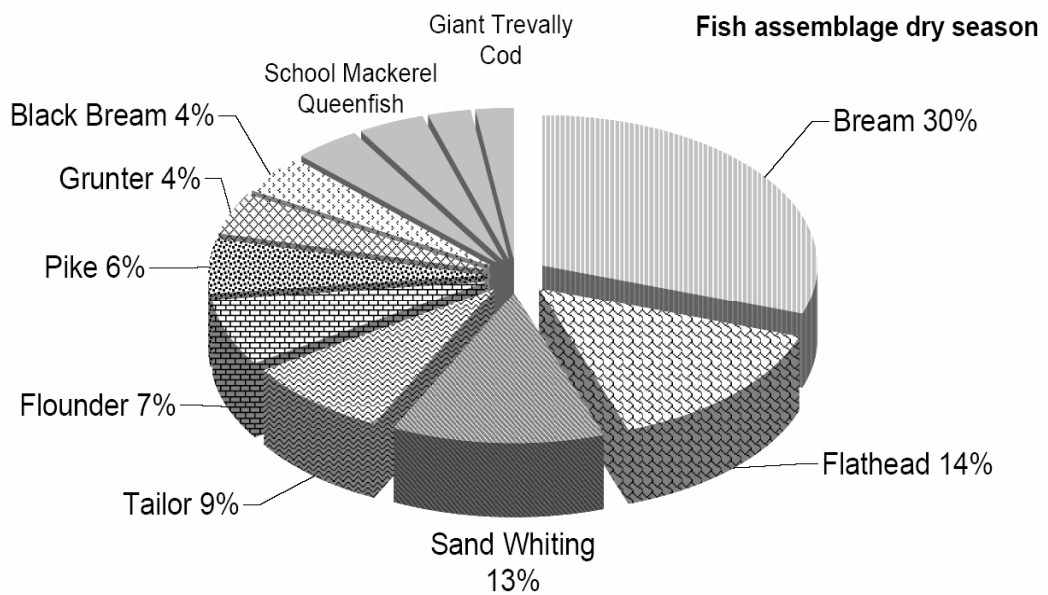
Figure 21: Percentage of species caught by anglers in the Burrum River during wet (a) and dry season (b) from 1997 until 2006
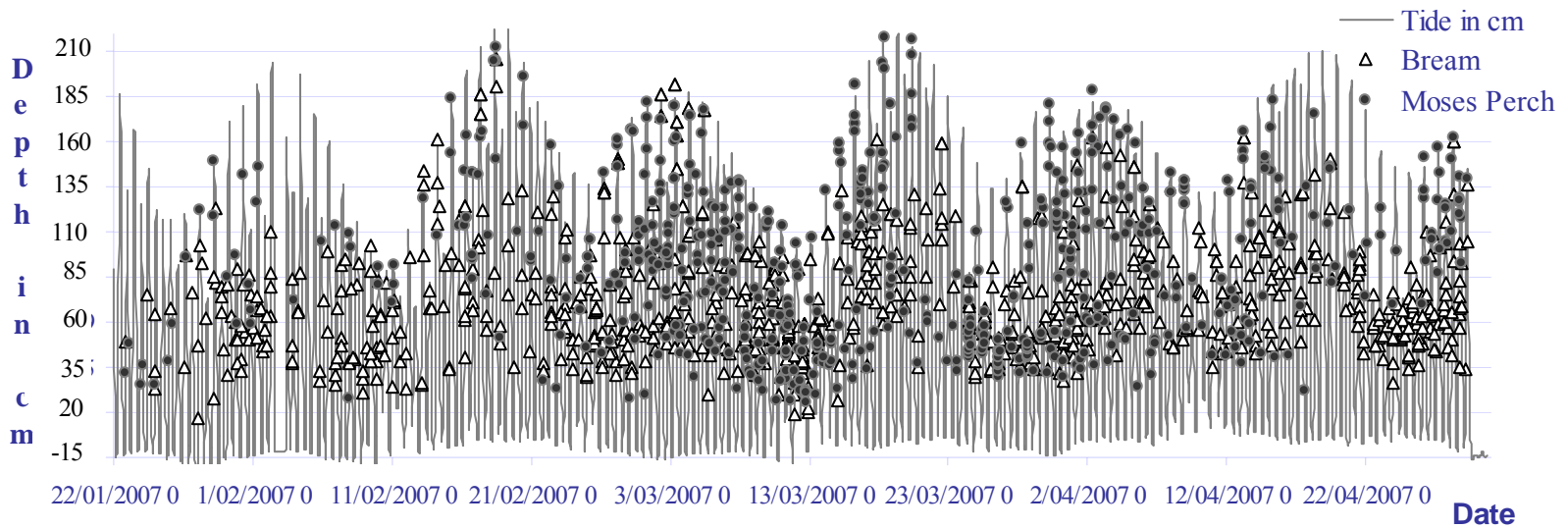
Figure 22: Detection of yellowfin bream (*Acanthopagrus australis*) and Moses perch (*Lutjanus russelli*) at the fish gate in relation to water depth. Yellowfin bream was mainly detected between 30 - 110 cm water height where as Moses perch was detected between 50 - 160 cm

5.3.3 Captured fish

Yellowfin bream and Moses perch accounted for most of the catch (27 and 17) followed by sea mullet or blue tailed mullet (*Valamugil buchanani*) (13), grunter (5), cod (*Epinephelus coioides*) (4), banded scat (*Selenotoca multifasciata)* (3), mangrove jack (*Lutjanus argentimaculatus*) (2), ponyfish (*Leiognathus equulus*) (1) and flathead (1). The average size and weight of PIT tagged fish was 16 cm and 120 g, with only two individuals (yellowfin bream) regarded as adults (Pollock, 1982).

5.3.4 Antenna efficiency

When PIT tags were passed manually through the gates' electromagnetic fields, the maximum detection range of the stationary system antennae was 45 cm from the antenna coil. Reading range varied with position and orientation of the tag with respect to the antenna coil. If a tag was passed through the centre with speed exceeding 3 m s^{-1}, it could not be read. Variation of the magnetic field was due to variation in the voltage ranging from 300 to 1000 V which can reduce efficiency of the gate. The high voltage was necessary to overcome higher resistance of saltwater compared to freshwater and to allow for an 8 m crossing width.

Initially ten PIT tagged fish (five yellowfin bream, five Moses perch) were released behind the gate at a peak tide, of which six were detected at the out-going tide on the same day indicating that 100 % efficiency was not always possible. In particular at peak tides the total number of entry events (fish entering creek) without corresponding exit events, and "good" readings (e.g., where entry events could be matched to exit events) occurred. The gate was set in place for 104 days from 15th January until 30th April 2007 and recorded > 5000 detections. During this time, 31 out of 75 PIT tagged individuals were detected, giving a recapture rate of 41 %. The average period from mark to last recapture for each fish was 47 days but ranged from one to 97 days.

Table 12: List of individual fish detected by the fish gate from 15th Jan 2007 until 30th Apr 2007. Age estimation based on Griffith (2001) and Department of Primary Industries and Fisheries (2006). Maximum and minimum water depths in cm refer to heights relative to AHD; for actual water depth 15 cm need to be added. SP = average speed estimated from multiple detection between the two fish gates in m s-1, LT = lowest tidal height of detection in cm, HT = highest tidal height of detection in cm above AHD, DM = total number of days of detection, FL = fork length in cm, WG = mass in gram, TG = average time upstream of the gate, n.d. = no data, AC = age class, j = juvenile, m = mature, sa = sub-adult.

Species	FL	WG	AC	HT	LT	SP	TG	DM	ID	Date tagged	Last recap
Acanthopagrus australis	135	70	J	75	30	0.4	3:35	18	490	09.04.07	28.04
Acanthopagrus australis	215	n.d.	M	60	30	n.d.	n.d.	48	835	18.12.06	27.04
Acanthopagrus australis	205	n.d.	M	125	30	n.d.	2:35	95	847	18.12.06	29.04
Epinephelus coioides	210	75	J	175	105	0.5	2:49	57	310	26.01.07	28.04
Platycephalus fuscus	350	600	Sa	0	0	0.5	n.d.	1	552	16.02.07	16.02
Pomadasys kaakan	260	300	J	210	15	0.3	n.d.	86	372	22.01.07	28.04
Pomadasys kaakan	295	525	Sa	200	15	0.33	4:45	76	475	02.02.07	29.04
Pomadasys kaakan	235	200	J	190	45	0.33	4:12	87	588	21.01.07	28.04
Lutjanus argentimaculatus	245	300	J	105	70	0.7	n.d.	22	467	03.02.07	04.03
Lutjanus russelli	140	50	J	120	85	0.25	3:53	9	339	09.04.07	27.04
Lutjanus russelli	205	n.d.	J	205	30	0.2	3:04	97	386	16.01.07	29.04
Lutjanus russelli	150	100	J	210	45	0.3	2:52	62	424	17.02.07	28.04
Lutjanus russelli	250	135	J	0	0	0.4	n.d.	1	453	03.02.07	06.02
Lutjanus russelli	120	20	J	0	0	n.d.	n.d.	1	576	06.02.07	06.02
Lutjanus russelli	235	250	J	190	45	0.4	2:33	71	631	16.02.07	28.04
Lutjanus russelli	230	225	J	210	60	0.3	2:14	72	748	03.02.07	28.04
Mugil cephalus	205	100	J	65	10	0.3	n.d.	1	434	29.01.07	29.01
Mugil cephalus	292	n.d.	M	175	60	0.2	4:01	39	825	19.12.06	02.03
Acanthopagrus australis	185	n.d.	J	0	0	n.d.	n.d.	1	832	18.12.06	15.02
Selenotoca multifasciata	125	100	J	215	80	0.3	2:31	76	405	27.01.07	18.04
Acanthopagrus australis	150	100	J	145	15	0.13	4:48	94	295	21.01.07	28.04
Acanthopagrus australis	110	50	J	120	60	0.02	n.d.	2	385	25.01.07	29.01
Acanthopagrus australis	125	40	J	160	45	0.02	5:41	6	395	08.02.07	15.02
Acanthopagrus australis	135	75	J	120	60	0.08	4:04	60	408	08.02.07	11.04
Acanthopagrus australis	135	75	J	120	10	0.02	4:30	90	410	21.01.07	28.04
Acanthopagrus australis	135	75	J	0	0	0.22	n.d.	1	470	12.02.07	12.02
Acanthopagrus australis	170	150	J	160	15	0.3	5:47	55	517	22.01.07	17.04
Acanthopagrus australis	0	0	J	110	40	0.2	4:37	65	523	22.01.07	28.04
Acanthopagrus australis	135	75	J	135	60	n.d.	n.d.	5	553	08.02.07	14.02
Acanthopagrus australis	155	75	J	185	30	0.33	4:00	97	615	21.01.07	29.04
Acanthopagrus australis	185	150	J	215	90	0.18	1:37	71	618	04.02.07	28.04

5.3.5   Fish detection and tidal range

Estimated swimming speed for fish ranged from 0.7 to < 0.02 m s^{-1}. The slowest travelling speed was recorded for fork length sizes between 110 and 150 mm with an average speed passing through the gate of more than 0.13 m s^{-1}. Individuals > 200 mm FL had travel speeds between 1.5 and 0.3 m s^{-1} (Table 12).

The continuously-recorded tidal data at the gate successfully captured the semidiurnal pattern of the estuary with a two-week cycle of spring tide events and a four-week cycle of peak tides of ~ 190 cm AHD (Fig. 22). Sea mullet emigrated from the creek while water depths were > 90 cm of water at the gate but often immigrated in shallower water (~ 60 cm depth). Grunter entered the creek during incoming and left during outgoing tides at water levels of ~ 30 cm. While different species of fish entered the creek at different water depths on the incoming tide, there was a general trend for most individuals to leave early in the ebb tide with very few departing once water levels were < 30 - 40 cm deep at the gate. A water depth of 110 cm at the gate during the incoming tide was a threshold level for the onset of inundation of mangroves. Based on fish detections at the gate, 50 % of fish movement into or out of the creek occurred when water depths were >110 cm. Differences in fishes species detection were reflecting variation in activity patterns, with 70 % of all activity for yellowfin bream, 60 % for grunter, 40 % for mullet and 35 % for Moses perch occurring when water level was < 110 cm water level (Table 13).

Despite a low number of recaptures, there was a significant positive correlation between fork length and water depth at the gate for Moses perch (r^2=0.85, n=5, p < 0.05) and yellowfin bream (r^2=0.63, n=8, p < 0.05). Moses perch of sizes between 140 and 230 mm tended to enter the creek at higher water levels (~ 80 cm) (Fig. 23).

The average time spent behind the gate for individuals ranged from 2 to 6h depending on species and size. Larger individuals of Moses perch and yellowfin bream with fork length > 180 mm spend on average < 3 h behind the gate, whereas smaller sized individuals of these species spent between 3 and 6 h behind the gate. Sea mullet spent 4 h and grunter between 4 and 5 h behind the gate. The time behind the gate could not be calculated for all tagged fish because the number of monitored days was too small or only one detection per tidal cycle was recorded due to detection variation (Table 13).
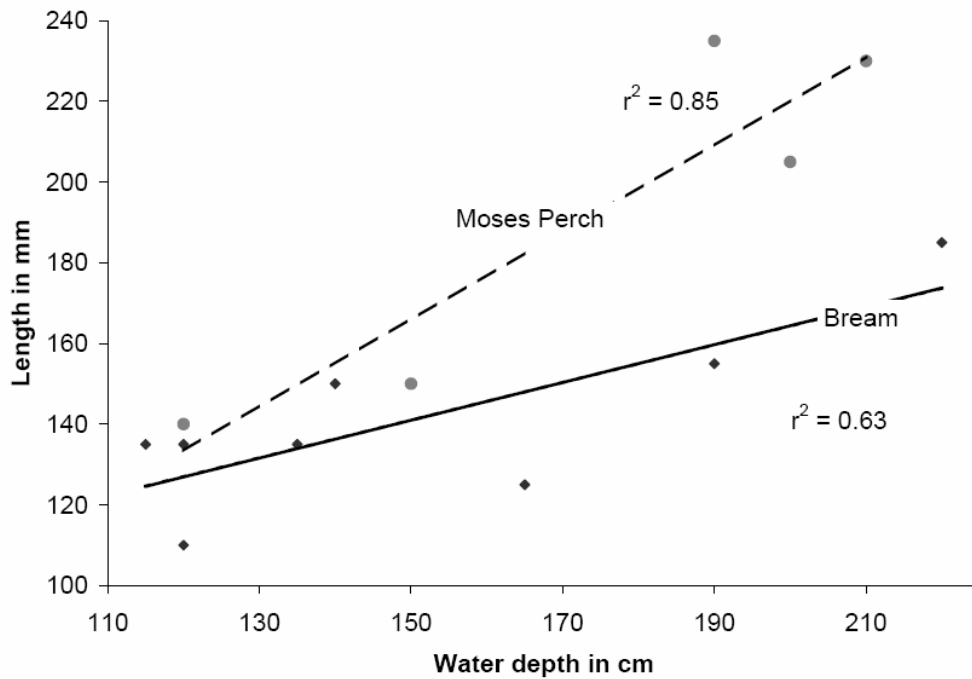
Figure 23: **Relationship between FL and maximum water depth at average detection for Moses perch (*Lutjanus russelli*) and yellowfin bream (*Acanthopagrus australis*)**

Table 13: **Percentage of detection < 40 cm water height and < 110 cm water height (marking threshold level for mangroves inundation) at the gate**

Species	% detection < 40 cm	% detection < 110 cm	total detection
Acanthopagrus australis	12	74	893
Lutjanus russelli	10	45	694
Pomadasys kaakan	7	60	214
Mugil cephalus	0	39	31
Selenotoca multifasciata	0	2	46
Epinephelus coioides	0	0	20

Diel activity patterns were shown for all 4 target species (Fig. 24). I detected yellowfin bream most commonly in the early mornings and during the day, whereas Moses perch, grunter and sea mullet were never detected between 12 - 5 pm. Their activity peaked during night times with Moses perch being more often detected in the evenings between 6 - 8 pm, grunter between 6 pm and 5 am and mullet between 7 pm - 9 am (Fig. 22, 23).

During the field survey I observed a number (~10) of T-bar tagged yellowfin bream and Moses perch in the water column at the mouth of the creek at low tide, suggesting a small home range for these species, which was also reflected by daily detection of these species at the gate. One recapture of a PIT-tagged individual was reported by local angler for a yellowfin bream 178 days after initial capture (1st August 2007) at ~ 6 km downstream near the river's mouth.

5.3.6   Underwater Digital Video Cameras

A total of 27 deployments resulted in over 25 h of video recording. Analysis of the DVCs images yielded mainly yellowfin bream, which supported the results from the gate (Fig. 25). However, small (< 10 cm FL) Moses perch and yellowfin bream estimated at 4 - 10 cm FL were often seen together, swimming in the mangroves in shallow water between 30 - 100 cm.

I sighted < 4 species per DVC deployment and identified a total of ten taxa (Gobiidae, *L. russelli*, *Mugil cephalus*, *A. australis*, Gerridae, *Monodactylus argenteus*, *Herklotsichthys castelnaui*, *Rhabdosargus sarba*, *Tetractenos hamiltoni*, Oxudercinae) (Fig. 25).

The observation of fish movement relative to tidal flow and against the current resulted in no clear direction of movement with 356 movements with the current and 357 movements against the current. I found no significant dependency between number of fish sightings, water temperature and salinity. However, these parameters did not vary greatly during the field study. As expected, visibility measured in meters with a Secchi depth (m) and number of sighted fish correlated significantly ($r^2 = 0.79$, n = 10, p < 0.05).
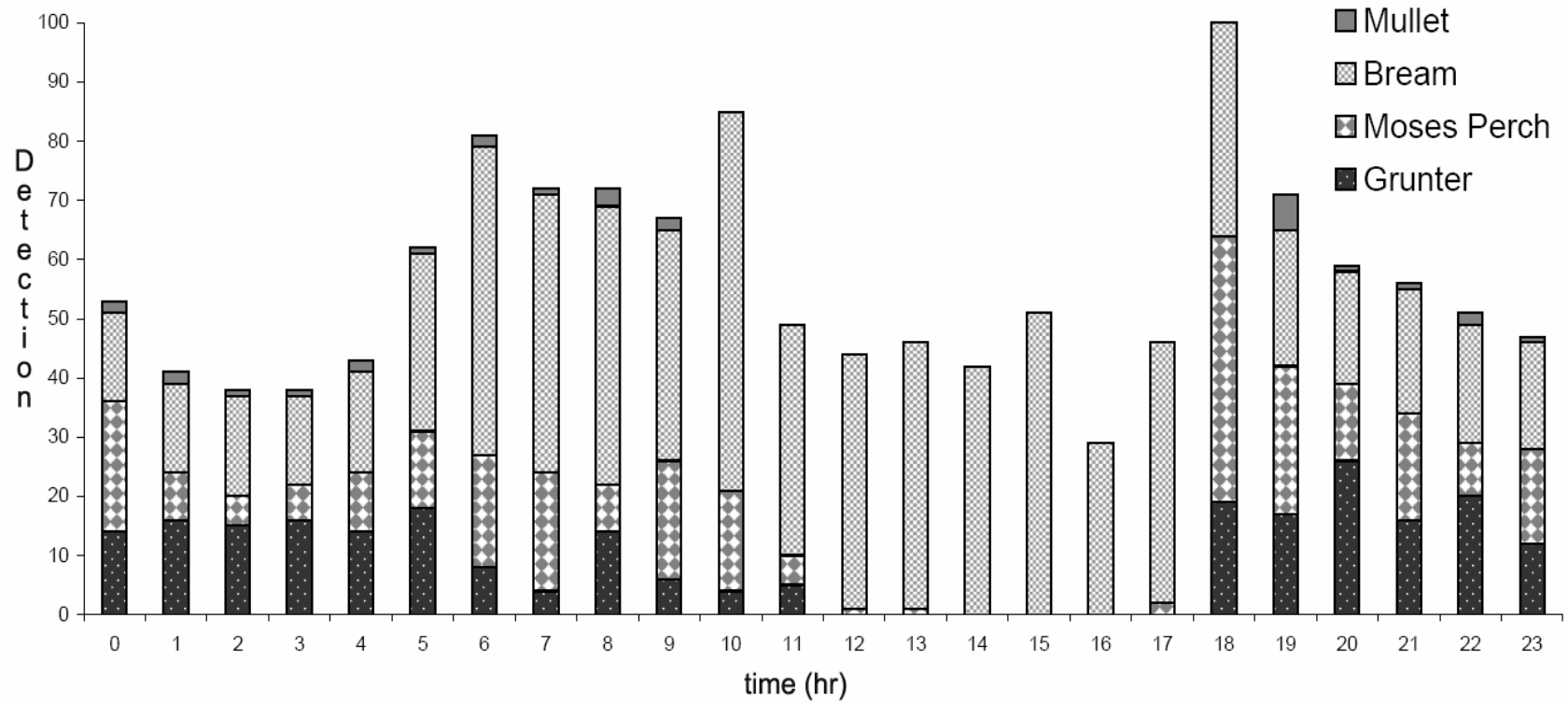
Figure 24: Total number of fish detections for bream (*Acanthopagrus australis)*, Moses perch (*Lutjanus russelli)*, grunter (*Pomadasys. Kaakan*) and mullet (*Mugil cephalus*) during each hour of the day between 15th Jan and 30th Apr 07
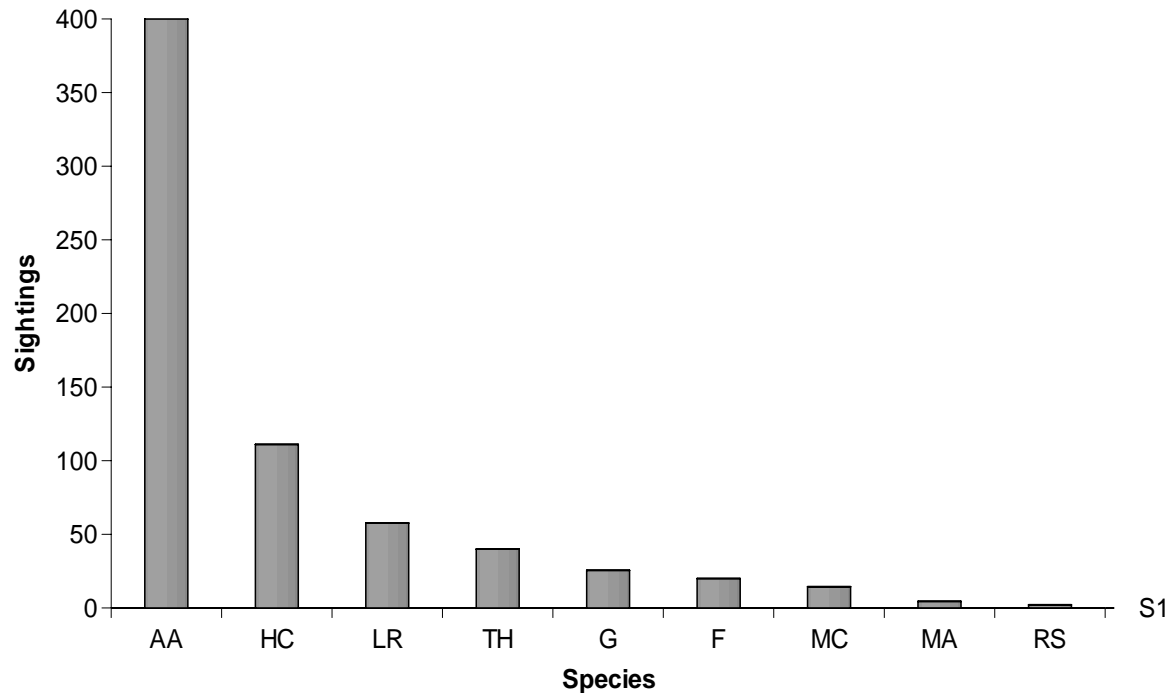
Figure 25: **Number of individual sightings for species during the visual survey using underwater DVCs**. AA = *Acanthopagrus australis*, HC = *Herklotsichthys castelnaui*, LR = *Lutjanus russelli*, TH = *Tetractenos hamiltoni*, G = Gobiidae, F = unidentified, MC = *Mugil cephalus*, MA = *Monodactylus argenteus*, RS = *Rhabdosargus sarba*

5.4   Discussion

5.4.1   *Specific consideration for species biology*

Links between habitat inundation and species detection at different tide phases indicated species-specific patterns in the timing of utilisation of the mangrove creek (Fig. 22, 23). Yellowfin bream an endemic species to Australia, is abundant in estuaries along the east coast of Australia (Blaber and Blaber, 1980). Up to 400 ty^{-1} are caught by commercial and recreational fishers along the coast of Queensland (Quinn, 1992). It is therefore not surprising that this species dominated the Burrum River anglers catch. Most of the captured yellowfin bream caught during my study in the mangrove creek were juveniles with large numbers of the 0+ year class (mean lengths between 60 mm and 80 mm; Griffiths, 2001) which were too small for tagging. Yellowfin bream were also the most

commonly sighted fish captured on the DVC footage (Fig. 25). However, care must be taken as the gate detection showed that other species were not active during the day when the DVCs were deployed, and yellowfin bream was particularly active in the early mornings. Therefore, observations from the DVCs were limited to those periods with visibility conditions suitable for deployment, but gave insights on species that moved together. Small sized (< 120 mm FL) Moses perch and yellowfin bream were often seen together, swimming in shallow water between 30 - 100 cm in mangrove habitat where they most likely foraged on shrimps or zooplankton (Baker and Sheaves, 2005). Observed fish movements with in and outgoing tides did not result in a clear movement of fish with the outgoing tide. Such fish movement against the flow could be due to foraging and related to optimised energetics when facing against the flow while trying to remain in a certain area (McLaughlin and Noakes, 1998) or perhaps orientation for catching prey. A certain amount of movement against the tide is common in tidally migrating fish. This behaviour has also been observed by Krumme (2004).

Isotopic analyses of yellowfin bream by Melville and Connolly (2003) suggested a 33 % dependence on mangroves as a carbon source (Melville and Connolly, 2003). Lugendo et al. (2007) showed that feeding by fishes within mangroves is more common when mangroves are permanently (vs. temporarily) accessible due to the tidal regime. Considering that some of the juveniles spend between 5 and 6 h behind the gate in a mangrove dominated environment, a similar dependence based on indirect intake mangrove derived carbon via phytoplankton and MPB (Layman, 2007) is possible. Notwithstanding, the gate detection showed that 70 % of all juvenile yellowfin bream detections occurred < 110 cm water height, a water depth where few mangroves were accessible (Fig. 22). All tagged yellowfin bream were detected on a daily basis suggesting that the juveniles had a small home range using the mangrove creek regularly for shelter and foraging. Small home ranges are also known for other common sparids e.g., pinfish (*Lagodon rhombiodes*) (Irlandi and Crawford, 1997). A mark and recapture study by Sheaves (1993) revealed similar results for Moses perch and yellowfin bream, demonstrating a strong tendency to be recaptured on the side of the creek on which they were initially captured. When exposed to predators, smaller fish tend to avoid open water habitats and try to seek shelter in areas of submerged vegetation that provides the structurally complexity in which their predators are less successful (Werner et al., 1983).

The individuals of larger Moses perch showed a very different behaviour to smaller Moses perch and yellowfin bream. They only entered the mangrove creek at high water levels (only 35 % of the detection were under the 110 cm water mark), and remained for short periods of time behind the gate. They travelled quickly, preferring the evening between 6 and 8 pm. Smaller juveniles during the day are likely to forage on shrimps whereas larger individuals are also likely to prey on small fish in the night reflecting ontogenetic shifts in habitat utilisation. Thayer et al. (1987) reported similar shifts for grey snapper (*Lutjanus griseus*) where adults were moving to adjacent habitats. For a number of mangrove fish species e.g., mangrove jack, the importance of mangrove food sources, such as sesarmid crabs and other invertebrates, to their diet has been indicated by a number of studies (Thong and Sasekumar, 1984; Sheaves and Molony, 2000). Grunter was only detected at night with half of their detections by the gate being over 110 cm water depth. Beside the relatively large size of the grunter of around 25 cm FL (size and age at maturity 30 cm/2 years, (DPI&F, 2006)) they remained behind the gate between 4 - 5 h. Grunter and Moses perch are known as nocturnal zoobenthivores (Laprise and Blaber, 1992). Foraging of individuals at night could be an advantage when searching for prey (Smith and McKay, 1986). Day–night migrations between resting and feeding habitats (Ogden and Ehrlich, 1977; Nagelkerken et al., 2000a), in particular, twilight migrations of Haemulidae (grunts) have been studied in detail (Dorenbosch et al., 2004). Behaviour of the herbivorous sea mullet (Jobling, 1995) was different from grunter. Sea mullet only returned on a 4-week cycle coming in with the high tides in the evenings around sunset (7 pm). The average time spent behind the gate was 4 h. They were not recorded at outgoing tides with < 90 cm of water at the gate but recorded at incoming tides ~ 60 cm. They left at high water levels when the mangroves were still inundated, allowing the sea mullet to enter the salt marsh area behind the mangrove. As indicated by Krumme et al. (2004) tide and day of time were the most important trigger factors for movements of a number of species in a study undertaken in a tidal mangrove creek in Brazil. My results support these findings.

Overall, there was a trend of fish being detected on a higher water level at the outgoing tide than at the incoming tide. I suggest that this is due to the hydrodynamics of the system as the water penetrates into mangroves and the salt marsh slower than it is running out. Fish would therefore have to leave the mangroves at higher water levels to avoid being stranded by the fast retreating ebb tide.

5.4.2 Recommendations

I was able to identify a total of 15 species within the mangrove creek, which is relatively low compared to other studies (Clynick and Chapman, 2002). By using 3 - 4 cm mesh size gill nets and hook and line as well as fishing only during daylight hours I was excluding smaller and nocturnal species. To increase the number of species being observed or captured by DVCs and fishing, the activities need to be undertaken at different times including night (e.g., using artificial red light, sensu Weiss et al., 2006). The effect of turbidity can be reduced by increasing the number of deployments. Application of stereo DVCs and the use of video software to estimate fish size would improve DVC observation (White et al., 2006). The sample size of tagged fish needs to be increased and field studies need to be extended to investigate seasonal differences. One sub-adult *Acanthropagus australis* has been recaptured by an angler at the mouth of the Burrum River on the 1st August 2007 (178 days after initial capture), perhaps on its way to its first spawning aggregation (Pollock, 1982). This individual was continuously recaptured by the gate over 71 days. It would be interesting to see, through a more elaborate tagging operation, if individual fish returns to the same location after a spawning migration.

Investigating fish growth or survival rate was not an objective of this study. Therefore, I did not control for survival. I expected survival rate for individuals >120 mm being close to 100 % (Zydlewski et al., 2001; Adams et al., 2006b). For larger and long-term application aiming to investigate survival, and for species with high site fidelity, it would be possible to extract emigration from survival over a number of seasons. Such investigations would be important to better understand the full extent of population dynamics and habitat dependency (see also habitat discussion Beck et al., 2001).

My results demonstrated the functionality of the developed autonomous antenna system with a recapture rate of 41 % and detection efficiency ~ 60 %. This is in line with observed PIT tag detection efficiencies in freshwater systems for similar settings (55–100 % (Zydlewski et al., 2001) or saline environments 61 % (Adams et al., 2006b)). However, fish were sometimes able to go around the gate at higher water levels. This problem could be overcome by establishing more gates further upstream

which would also give better resolution of the movement and spatial occurrence of fish within the mangrove channel.

My inundation model was based on elevation only and did not provide a dynamic reflection of water movement. I expect a delay of inundation for areas further away from the channel. A hydrodynamic model would enable the prediction of fine-scale connectivity for the entire river system and more realistic habitat utilisation of fish for various inundation scenarios.

5.4.3 Conclusion

I was able to quantify movements of four commercially and recreationally important estuarine associated fish species in and out of a mangrove creek and identified habitat use and time of residency. All four target species continuously used the mangrove creek during the four-month survey, showing close association with the site. Temporal trends for presence and absence of fish in relation to habitat availability were also demonstrated. In particular, I found interspecific and ontogenetic differences in daily movement and diel activity, with more species and larger individuals being active at night.

Such assessment of movement of estuarine fish species is critical information for understanding their population dynamics and how habitat connectivity sustains fish populations. The autonomous antenna system represents a valuable addition to the conventional mark and recapture and radio telemetry methods for studying the spatial habitat ecology of estuarine fish. It is an accurate method for examining how abiotic factors, e.g., tidal range, may influence nursery function, thus assisting the identification and management of nursery habitats. The combined use of a LIDAR derived elevation model with fish movement data provided essential information on habitat inundation and allowed for a detailed assessment of individual fish behaviour in relation to habitat availability. Observations from underwater DVCs supported the results from the fish gate and proved to be a qualitative, non-invasive method to study fish behaviour and movement but will need further improvement.

CHAPTER 6

EFFECT OF RAINFALL AND TEMPERATURE ON ESTUARINE FISH PRODUCTION IN QUEENSLAND

6.1 Introduction

When investigating climate effects on fish stocks, the focus has been on commercially important pelagic fish species such as tuna (*Thunnus albacares, T. thynnus*), mackerel (*Trachurus declivis, T. novaezelandiae*) and sardines (*Sardinops sagax, S. pilchardus*) (Klyashtorin, 1998; Yáñez et al., 2002). In particular, there has been a number of studies demonstrating sensitivity of small pelagic species to ENSO fluctuations (Schwartzlose et al., 1999; Ghofar and Mathews, 2000; Lea, 2000; Lehodey et al., 2003). Potential climate effects on estuary-dependent fisheries species have largely been neglected so far. It is reasonable to expect that on-going climatic changes may also affect a number of estuary-dependent fish species such as mullet, which is one of the most important animal protein sources in many regions of the world (Hill, 2004), and some prawn species (e.g., *Penaeus esculentus* and *P. plejebus*), of significant commercial value. There are commercial fisheries in estuaries and near-shore waters of tropical Australia (Dent, 2002; Robins et al., 2005). However, potential economic impacts on fish catch caused by climate change are unclear so far (Abbs, 2002). This gap is mainly caused by the difficulty in establishing a clear link between commercial estuarine fish species and climate variables on a broad scale. Large variability in climate, the catch and biology of these species make generalisations difficult and the local nature of most studies restricts the usefulness of modelling approaches. It remains an important task to develop climatic indices capable of improving fisheries management (Roessig et al., 2004). Studies in parts of Queensland, Australia (Glaister, 1978; Vance et al., 1998; Loneragan and Bunn, 1999; Staunton-Smith et al., 2004; Growns and James, 2005) indicated that there is a strong relation between freshwater runoff and catch of some important commercial fisheries species (e.g., mullet (*Mugil* spp.), flathead, whiting (*Sillago* spp.), prawns (Family Penaeidae), mud crabs (*Scylla serrata*)) but it remains unclear whether these

links also apply to the whole coast of Queensland and if a global climate indicator such as the SOI has measurable influence on fish catch. The Queensland coastline offers a wide range of rainfall, temperature and high records of fish catch that would enable a broad scale modelling approach (Fig. 26).

A number of different models have been developed worldwide to study links of fisheries to climate and their economic consequences. Modified simple surplus production models including both environmental and bio-economic diversity variables such as the Gordon-Schaefer model or the Gulland-Fox model to relate banana prawn *(Penaeus merguiensis)* commercial catch with rainfall have been used (Polovina and Opnai, 1989; Kasulo and Perrings, 2004). Furthermore, bio-economic models were established for more detailed analysis where information on stock dynamics (e.g., mortality) was available (Hannesson, 1993; Seijo et al., 1994; King, 1995; Pascoe, 1997; Seijo et al., 1997; Sparre and Venema, 1997; Ulrich et al., 2001). However, if the population dynamics are largely unknown, simple models (e.g., biomass dynamic population model with environmental parameters) such as those available in CLIMPROD (FAO, 1994) can provide useful outcomes for preliminary investigations (Chen et al., 1997; Evans et al., 1997; Ghofar and Mathews, 2000).

The main objectives of this chapter are (1) to establish whether there is any historical association between selected elements of climate, namely, rainfall, SOI and coastal air temperature, and the catch of selected Queensland estuary-dependent fisheries species and (2) to provide a broad-scale case to model climatic effects on fish landings and potential economic losses in response to climate change. I do not expect temperature to have the same measurable influence on fish catch data and have therefore focused on rainfall and SOI for estuarine fish species, particularly, mullet (*Mugil* spp.), penaeid prawns (Family Penaeidae) and barramundi (*Lates calcarifer*) in Queensland. However, monthly average air temperature from selected stations has been considered in the analyses for the Queensland coast.
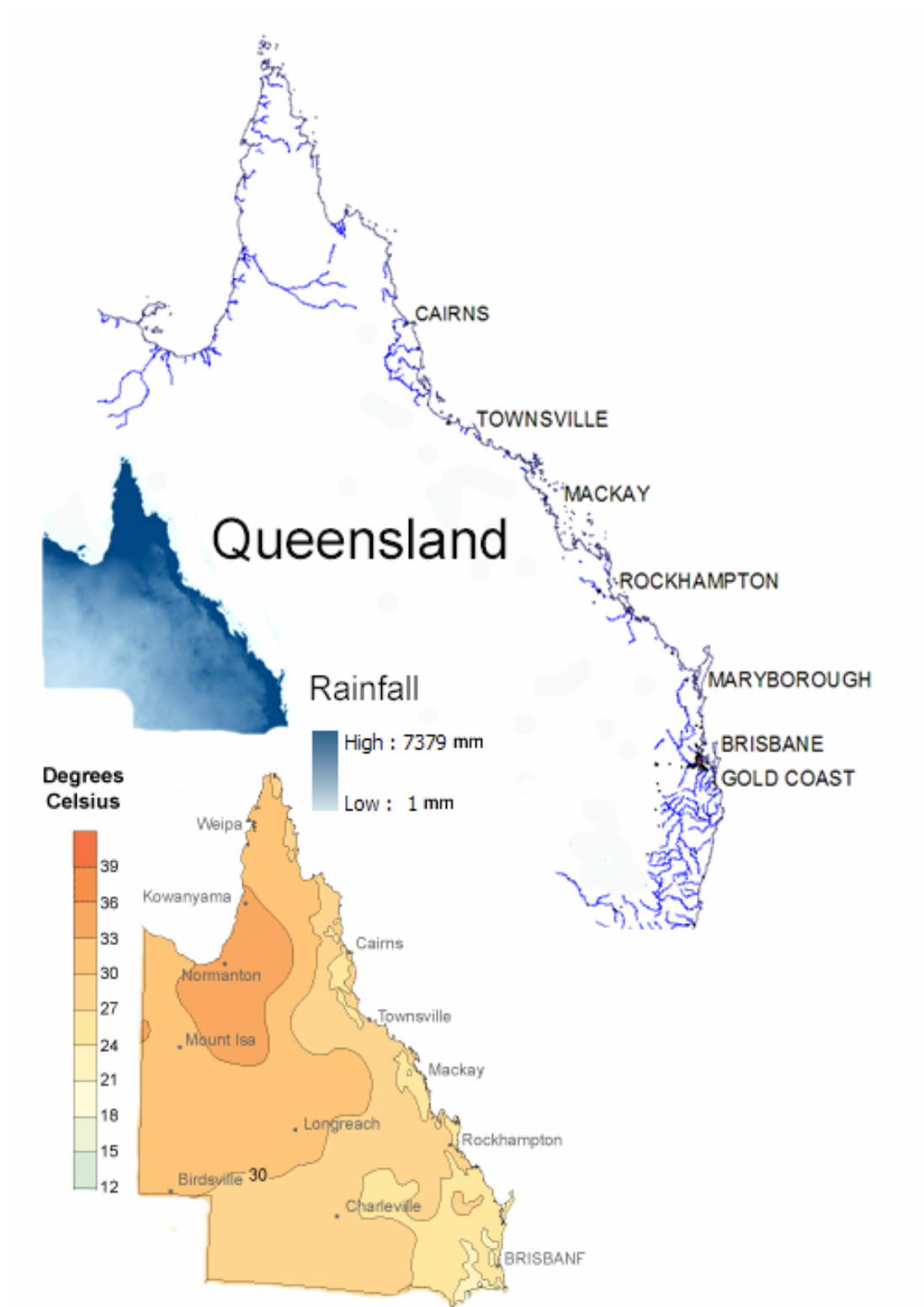
Figure 26: Map showing major river systems, cities, rainfall and mean maximum temperature distribution for Queensland (Geoscience Australia, 2005)

6.2   Methods

6.2.1   Estuary-dependent species and climate – a conceptual model

The relevant processes related to commercial fish catch in estuaries are described in a model to allow better understanding of the complex interactions between climate parameter and estuary-dependent fish stock.
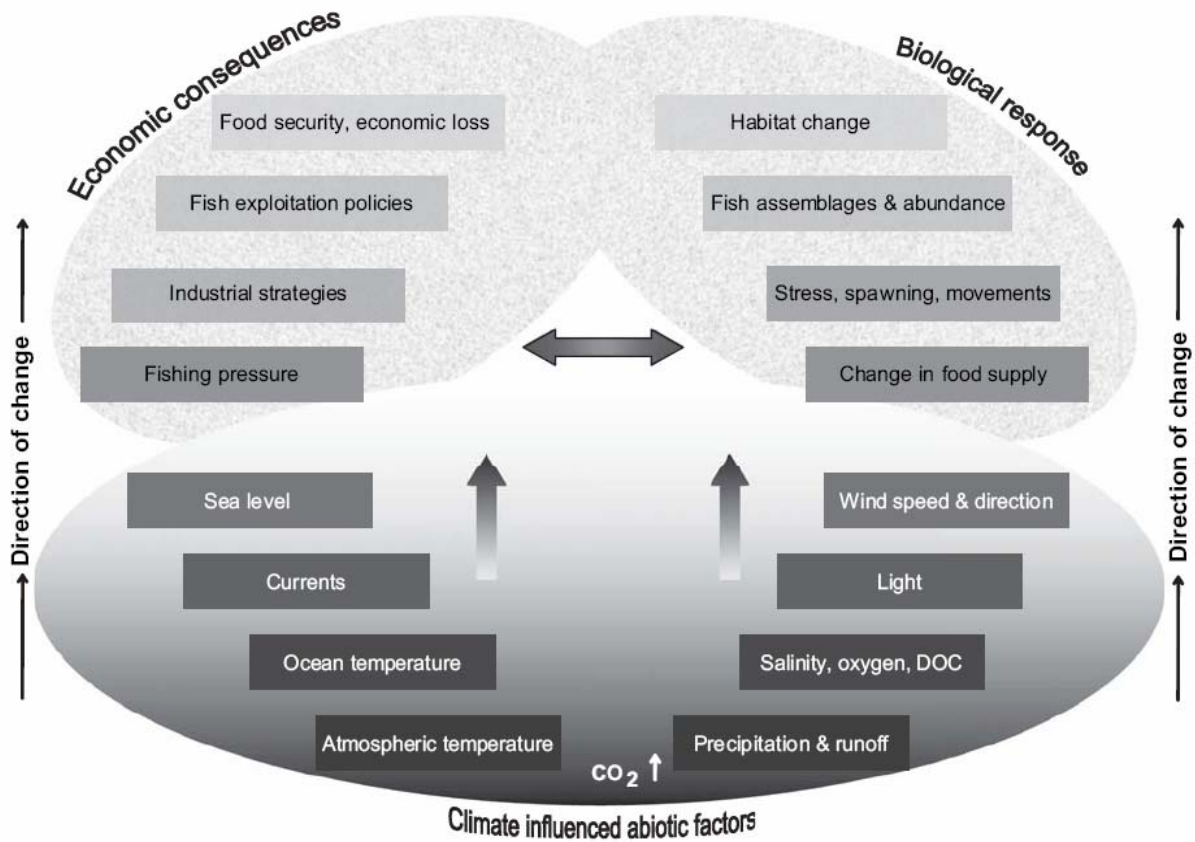
Figure 27: A conceptual model of the complex interaction between estuary-dependent species and climate-influenced abiotic factors.

The effects of factors are expected to increase from bottom to top of figure

For example, several models have been used to conceptualise the role of rainfall in estuarine ecosystems. An 'order of effects' model has been suggested, which distinguishes between the direct (salinity, erosion) and indirect effects (abiotic and

biotic variables) of flow (rainfall) on ecological processes (Hart and Finelli, 1999). A simple conceptual model, where rainfall and, with it, freshwater inflow to estuaries result in physical, chemical and biological consequences was proposed by Kimmerer (2002). A similar approach was given by Alber (2002) who suggested that freshwater inflow affects estuarine conditions (e.g., physical and chemical conditions), which in turn affect estuarine resources (e.g., plants and animals) and ecological processes. Sklar and Browder (1998) also included the effects of landscape modifications, tidal actions and solar activity. Following these approaches a conceptual model for this study was developed, in which climate-relevant processes and parameters are related to economic consequences in a temporal dimension (Fig. 27). Three categories representing (1) the abiotic factors; (2) the biological response; and (3) the economic consequences are suggested for the model. Starting with an increase in atmospheric temperature a number of additional abiotic factors such as rainfall experience a significant shift, leading to biological response of sensitive fish species that, for example, actively avoid the estuary or indirectly through an increased mortality rate of juveniles causing recruitment failure (Loneragan and Bunn, 1999; Robins et al., 2005) or a change in fish assemblages (Whitfield, 2005). In the long term, temperature increase may induce a rise in sea level, causing an ecotone shift of important nursery habitats and resulting in reduced habitat availability for estuarine fish species. As a consequence, fish catch may be reduced and fishing pressure increased (e.g., increased number of fishing days), risking overexploitation and economic loss. A decline in income for the fishermen may then lead to conflicts with protection zones and current management strategies.

6.2.2   Data

Data on catch, effort (number of days and boats) and gross value of production for estuary-dependent species or species groups from 1988-2004 were provided by DPI&F Assessment and Monitoring Unit (Table 16; see Chapter 3 for detailed information). In addition, available data on mullet catch for the years 1945-1980 were obtained from DPI&F reports but have not been pooled with recent data sets due to major differences in the methodology of the census. The Queensland fisheries report for 1988-2000 (Williams, 2002) provided additional data for some species mainly for the east coast of Queensland. Effort was calculated for the individual fisheries (line, trawl, net and pot)

and pooled for each species. Data on annual and monthly rainfall and temperature for the time period 1988-2004 and values for the SOI (1945–2004) were obtained from the BOM, as well as from weather stations in Cairns and Moreton Bay. Temperature data were provided as point data whereas rainfall data were provided as average values for 8 coastal regions (Fig. 28). The Cairns and Moreton Bay regions were selected for a detailed study since the weather data are consistent over a long time and intensive fishing activity within Trinity Inlet and Moreton Bay provides high number of catch records. Data from the Barron River gauging station (Cairns, Queensland) on freshwater runoff was provided by the NRM. This gauging station represents a major part of the freshwater runoff into Trinity Inlet. Other gauging stations were found to have significant data gaps.

Table 14: Major estuary-dependent fish species or species groups selected for analysing the relationship between catch data and rainfall and temperature. Criteria for species selection: The species (1) should have relative constant and high market value; (2) should be estuary-dependent; (3) widespread throughout Queensland; and (4) reported to be climate-sensitive. Source: Yearsley et al. (1999)

Common Name and fish catch class	Dominant taxa
Barramundi	*Lates calcarifer*
Bream	*Nematalosa* sp., *Monodactylus argenteus, Pomadasys maculatum, Acanthopagrus australis, A. berda*
Bugs	*Thenus indicus, T. orientalis*
Mud Crab	*Scylla serrata*
Flathead – Dusky	*Platycephalus fuscus*
Grunter	*Pomadasys* spp., *P. kaakan*
Mullet	*Liza vaigiensis, Valamugil georgii, Mugil cephalus, L. argentea*
Prawns – Bait	Family Penaeidae
Prawns – Banana	*Penaeus merguiensis*
Prawns – Bay	*Metapenaeus bennettae, M. insolitus*
Prawns - Endeavour	*Metapenaeus endeavouri, M. ensis*
Prawns – Greasy	*Metapenaeus macleayi*
Prawns – King	*Penaeus plejebus*
Prawns – School	*Metapenaeus macleayi*
Prawns – Tiger	*Penaeus monodon, P. semisulcatus*
Sea Perch - Mixed	Family Lutjanidae
Mangrove Jack	*Lutjanus argentimaculatus*
Threadfin King/Blue	*Eleutheronema tetradactylum, Polydactylus macrochir sheridani*
Whiting	*Sillago ciliata, S. analis, S. maculata, S. burrus, S. ingenuua, S. sibama, S. robusta*
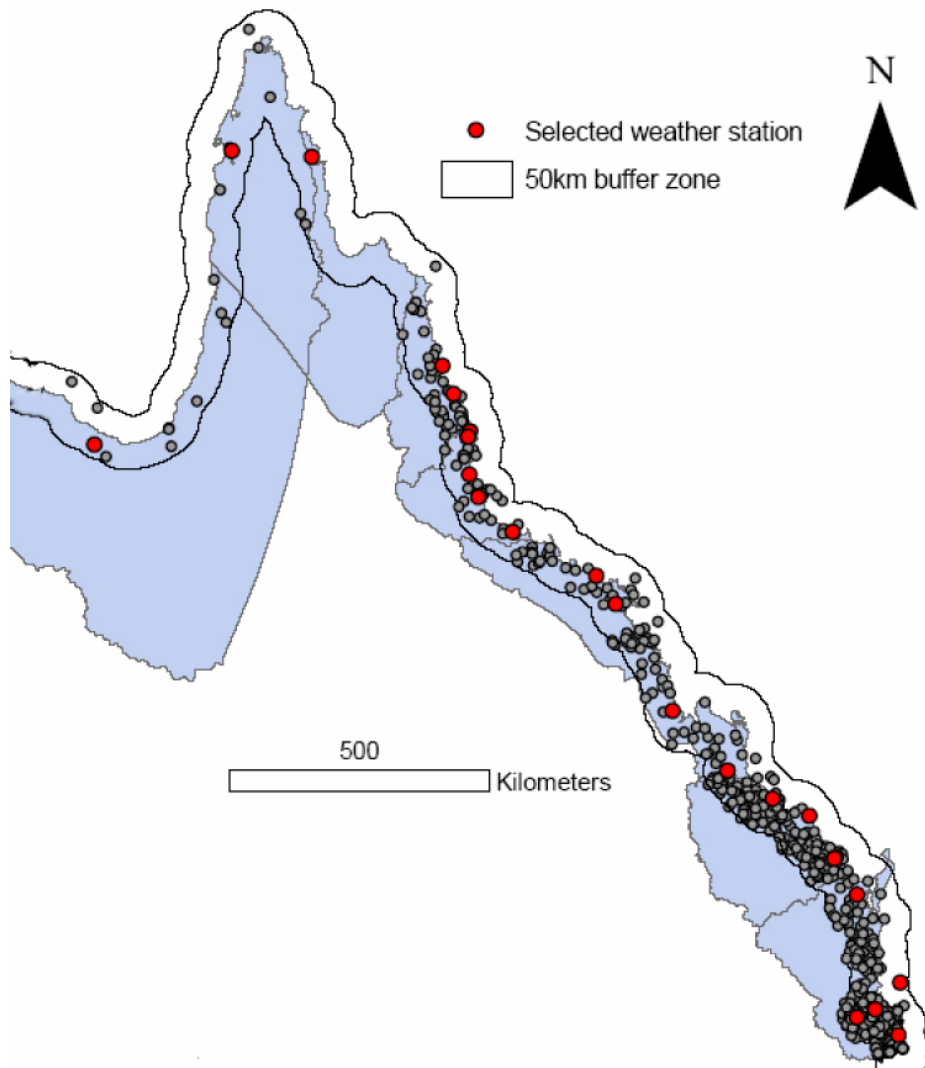
Figure 28: Selected weather stations (in red) for temperature data within a 50 km buffer zone along the coast of Queensland. Only selected weather stations had continuous data from 1988-2004

For rainfall, regional station data as well as monthly average coastal rainfall and temperature data were used. Because of the known seasonality, with the maximum rainfall in Queensland occurring from December to February, and the minimum during May and October, the 'annual' rainfall, SOI and catch data were grouped into wet (November to April) and dry seasons (May to October). Furthermore, I divided the coast of Queensland into 8 different regions according to the Bureau of Meteorology rainfall districts (BOM, 2004) and used coastal catch data from 13 different species groups (Table 14) to observe for regional differences in the correlation between rainfall and fish catch (Fig. 29). These 8 regions also present some of the differences in species

composition between coastal regions of Queensland (e.g., Gulf of Carpentaria, North Queensland, Central Queensland and Southern Queensland).
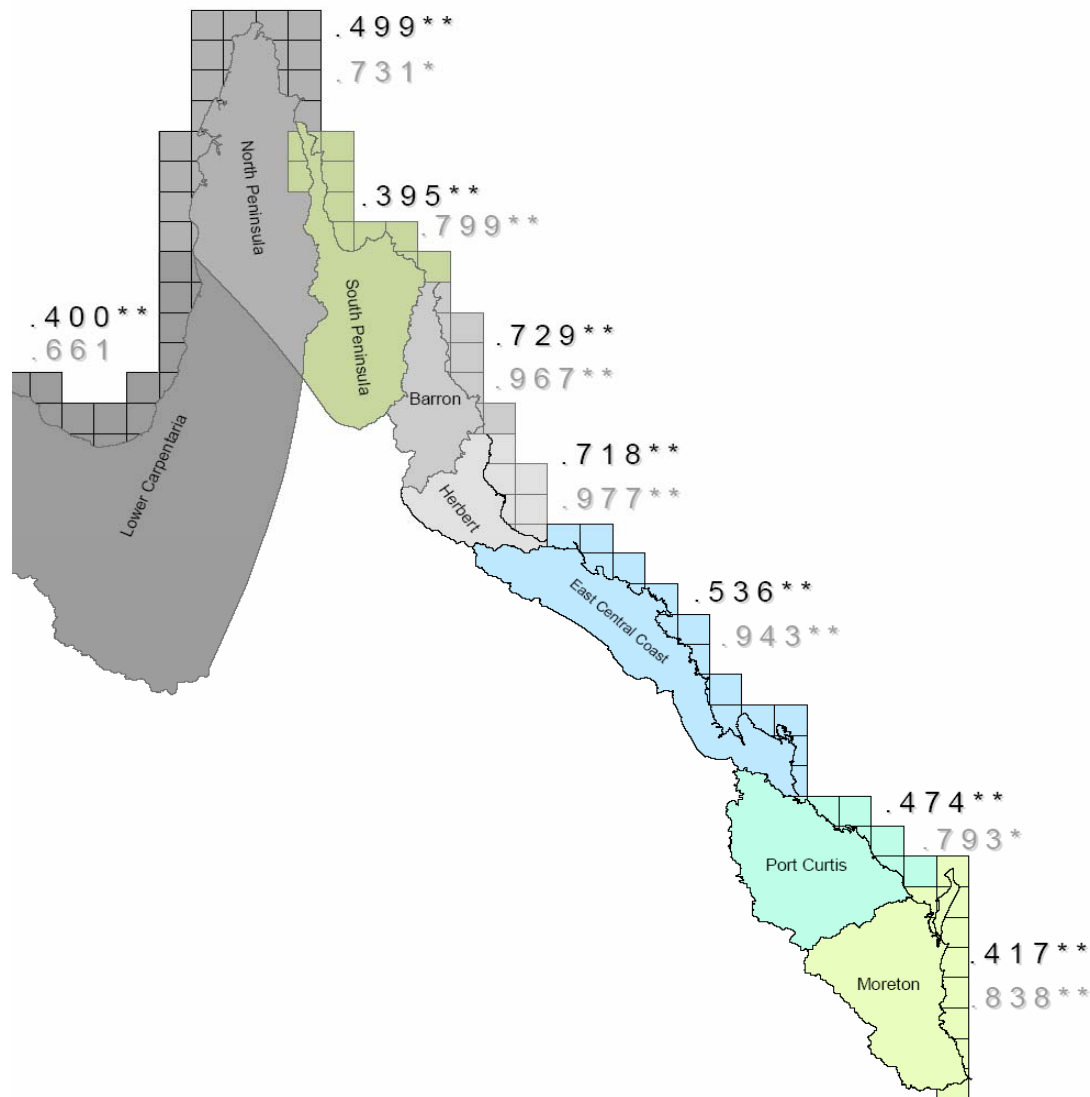
Figure 29: Releationships between eight different rainfall regions and their selected fish catch grids.

Positive Pearson correlation values for monthly barramundi catch and rainfall (black, n = 153) as well as seasonal correlation values (grey, n = 9) for the period 1988-2004 are shown for each region (* P < 0.05, ** P < 0.01). There were no data on barramundi catch during the barramundi closure period from November 1 (October 1 in the Gulf of Carpentaria) to February 1

6.2.3 Data analyses

As a first attempt, El Niño events (1991-92, 1993-95, 1997-98) and related rainfall events have been compared with fish catch data from Queensland, Moreton Bay and Cairns. Relationships between catch, temperature, rainfall, and SOI were explored using single linear regression models and correlation analysis. I used this approach instead of the time series models (e.g., transfer function) because the dependent variable (catch) in the time series has a maximum length of 17 years, which is less than the minimum necessary (i.e., ~ 50-60 data points in time) to properly use Box-Jenkins approaches (i.e., autoregression (AR), autoregressive integrated moving average (ARIMA) and transfer functions models) (Box and Jenkins, 1976; Rothschild et al., 1996).

Predominantly flathead (*Platycephalus* sp.), mud crab (*Scylla serrata*), penaeid prawns (e.g., *Penaeus esculentus* and *P. plejebus*), barramundi (*Lates calcarifer*) and mullet (e.g., *Mugil cephalus*) were target species for the analyses. Where appropriate SOI and rainfall have been lagged as a common method for such analyses (Sutcliffe, 1973; Drinkwater et al., 1991; Wilber, 1992) but this was done only when such a lag was biologically meaningful (e.g., age of sexual maturity). Basic activities of fish life cycles such as feeding and movement are most likely affected by environmental variability at shorter than annual or seasonal time scales. Therefore, monthly rainfall, monthly average air temperature and monthly catch data have also been included in the analyses.

Following a simple regression model the deviation of the mean GVP of Queensland's total estuary-dependent fish catch was compared with average annual coastal rainfall and El Niño events. This resulted in an estimation of potential loss caused by these events providing information for future model development.

6.2.4 CLIMPROD program

CLIMPROD (FAO, 1994; Fréon et al., 1998) was used to explore whether simple surplus production models could adequately incorporate the links between environmental factors and fish catch. A fit assessment for the chosen models based on jackknife estimation of the parameters and of r^2 (Efron and Gong, 1983) allowed a qualified test. Initial attempts were made to fit the Fox (1970), Schaefer (1957) and

Pella and Tomlinson (1969) models using catch total effort for mullet (*Mugil* spp.), mud crab (*Scylla serrata*), barramundi (*Lates calcarifer*), king (*Penaeus plejebus*) and tiger prawns (*Penaeus monodon, P. semisulcatus)* and commercial estuarine fish catch. The Schaeffer model is based on the Verhulst population growth equation. Fox's approach uses a logarithmic population growth equation and Pella & Tomlinson's approach uses a generalized population growth equation.

Catch, effort and rainfall data were entered into the CLIMPROD program and CLIMPROD's expert system was used to select an appropriate model based upon the known biology of the investigated species. This was done to see whether there is a simple surplus production model available to predict fish catch under environmental influence. Only simple modelling was possible as there is a lack of detailed ecological knowledge on the investigated species (e.g., mortality rate, predators). CLIMPROD requires annual data series on catch and effort of a fishery on a single stock, and annual (or seasonal) data series on an environmental variable. Standardisation of effort was not used in the CLIMPROD analyses (Evans et al., 1997). The appropriate model is chosen, using a nonlinear regression routine and results are assessed with parametric and nonparametric tests. The coefficient of determination (r^2) was given by the program and the jackknife method is used to give an indication of the robustness of the model in which the percentage variation in r^2 resulting from the elimination of a single data point from the set is calculated for all of the points (Efron and Gong, 1983). The Food and Agricultural Organisation of the United Nations (FAO) suggests a conservative r^2 value of > 90 % for bivariate models and > 70 % for multivariate models (Fréon et al., 1993). The fit assessment is also based on residual analysis and on data set characteristics.

6.3   Results

6.3.1   El Niño events and estuarine fish catch

The preliminary analyses on total fish catch suggested that between 20 % and 30 % of the catch variability might be explained by El Niño events and the associated reduced rainfall. The r^2 values for total fish catch with yearly local rainfall records between 1988-2004 were 0.31 and 0.27, respectively, for Moreton Bay and Cairns. Comparison

between penaeid prawn CPUE, rainfall and El Niño events indicated some further links. Three major El Niño periods can be identified for the last 17 years, showing significant correlation with Queensland coastal rainfall (Fig. 30). These periods were coincident with a reduced CPUE for penaeid prawns (Family Penaeidae), mullet (e.g., *Mugil cephalus*), flathead (*Platycephalus* spp.) and the total catch of estuary-dependent species, thus suggesting a link between the SOI and some estuary-dependent species. The same holds for periods of very low average rainfall. However, analyses of the data showed strong influence of effort especially on prawn catch. Correlation between average monthly temperature and annual temperature in Cairns and Moreton Bay and estuary-dependent fish catch were not significant.

The major fluctuations in Queensland's capture fisheries were contemporaneous with El Niño and La Niña events and reduced rainfall. If the loss is calculated from yearly GVP during dry events against the mean GVP of the study period, a total loss of A$ 65 million during nine dry years versus a gain of A$61 million in seven wet years was estimated. This seems reasonable as the fish catch explained about 66 % of the GVP variability for the period 1988-2003. On a smaller scale the regional data for Cairns showed a similar pattern with an estimated loss during dry events relative to mean GVP (1988-2004) of A$3.24 million. The south-east Queensland king (*Penaeus plejebus*), tiger (*Penaeus monodon, P. semisulcatus*) and bay prawn (*Metapenaeus bennettae, M. insolitus)* catches showed a strong reduction during El Niño events (1990, 93, 94, 97) of up to 15 kg day^{-1} year^{-1}, accounting for a total loss of A$70 000 during 1988-2004 (Fig. 31).

6.3.2 *Relationship between catch, rainfall and SOI*

Separate correlation analyses were undertaken for flathead (*Platycephalus* spp.), mud crab (*Scylla serrata*), penaeid prawns (Family Penaeidae), barramundi (*Lates calcarifer*) and mullet (e.g., *Mugil cephalus*) to explore differences of the correlation between the fish catch data and rainfall and SOI. Results obtained for annual barramundi catch and freshwater runoff showed positive but non-significant correlation. There was also no significant correlation between yearly maximum or minimum temperature and catch for any of the investigated species for the subregion Cairns. However, further analyses for

temperature dependency have been undertaken for the eight selected regions along the coast.
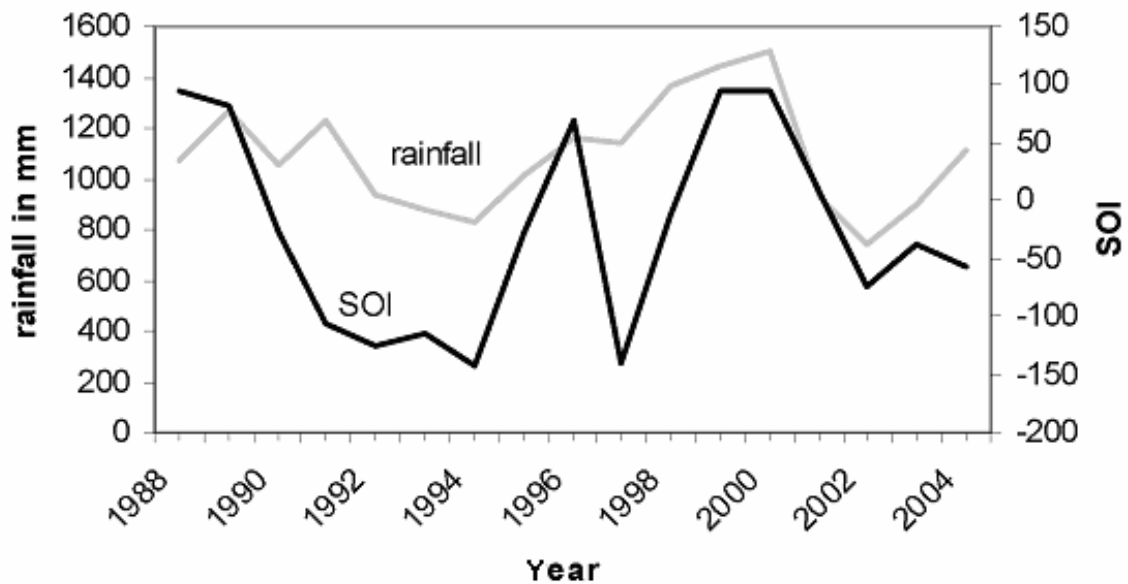
Figure 30: Average coastal rainfall in Queensland and SOI from 1988-2004.
r = 0.62; P < 0.01, along the east coast and r = 0.59; P < 0.05, for the whole coast (n=17). El Niño events occurred in 91/92, 94/95 and 97/98
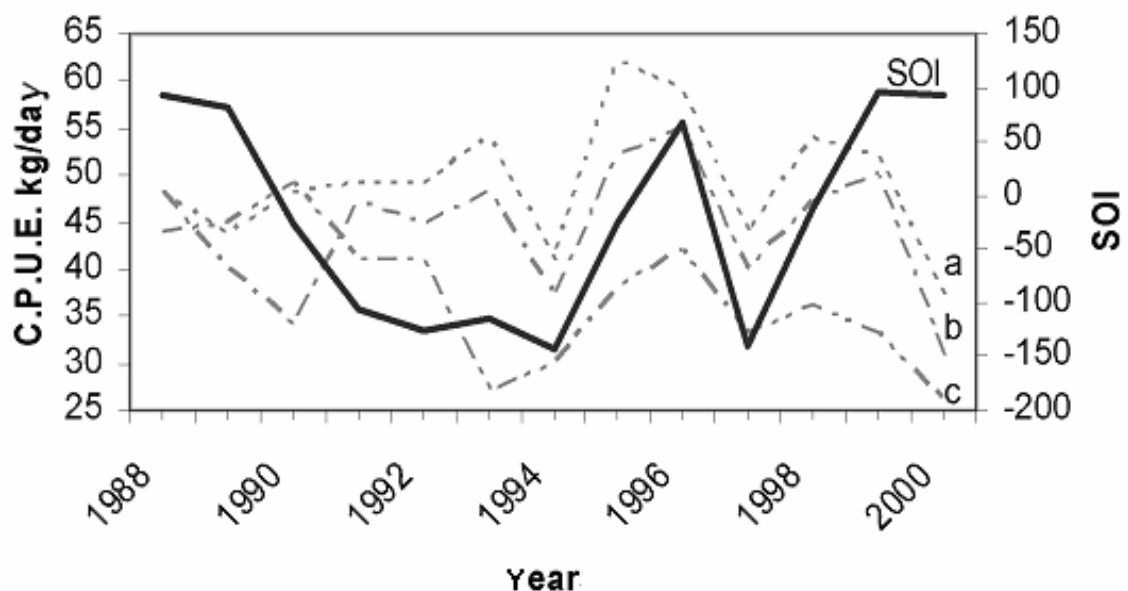
Figure 31: SOI and catch of (a) tiger (*Penaeus esculentus, P. semisulcatus*), (b) king (*Penaeus plejebus*) and (c) bay prawn (*Metapenaeus bennettae, M. insolitus*) in south-east Queensland
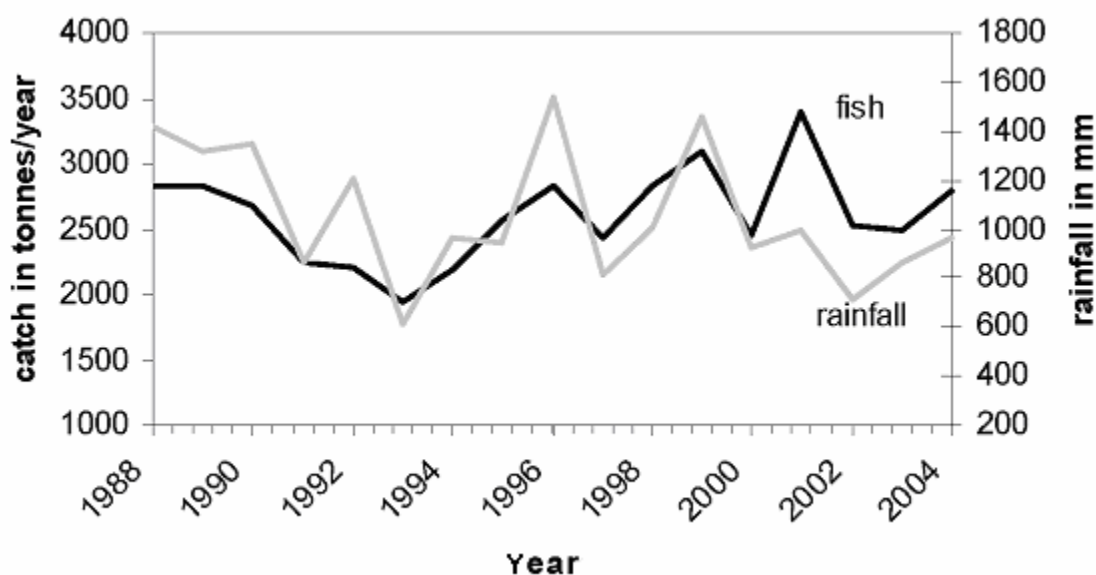
Figure 32: Rainfall and commercial fish catch (1988-2004) in Moreton Bay
(r = 0.54; P < 0.05, n = 17)

The Moreton Bay and Cairns subregions both showed significant positive correlation between annual rainfall and total fish catch (r = 0.54 and 0.52, respectively; P < 0.05, n = 17) (Fig. 32). Significant positive correlations resulted from analysis between mean annual coastal rainfall, SOI and total annual commercial catches of mullet (*Mugil* spp.), barramundi (*Lates calcarifer*), flathead (*Platycephalus* spp.) and mud crabs (*Scylla serrata*). Monthly rainfall and monthly barramundi catch were found to correlate significantly in all, (Fig. 29) and in particular, the Herbert and Barron regions. Correlations for penaeid prawns, mud crabs and rainfall shifted from significant positive in the south to negative correlations in the north (Table 15). Monthly air temperature and monthly mullet, flathead and whiting (*Sillago* spp.) catch showed negative relationships in south-east Queensland and in the central east coast whereas barramundi catch was positive related to average air temperature in the central east coast. Mud crabs and prawns (family Penaeidae) commercial catch was also positively correlated to average air temperature in sub-tropical south-east Queensland but an opposite trend was recorded in tropical north Queensland (Table 16).

Furthermore, regression analyses have been performed for rainfall, SOI and selected species groups, where the variation in SOI accounted for between 41 % and 49 % of the variance (r^2) in the catches of total mullet (e.g., *Mugil cepalus*) and barramundi (*Lates*

calcarifer) in the Gulf of Carpentaria whereas regional rainfall accounted for between 55 % and 81 % of the variance (Fig. 33, Appendix 8 and 9).

Long-term data for annual mullet catch (1945-1980) gave no significant correlation with annual SOI (r = 0.23; P > 0.05, n = 35). However, the mullet CPUE from 1988-2004 showed a positive correlation with SOI (r = 0.57; P < 0.05, n = 17). Monthly mullet fish catch data for the period 1988 to 2004 were pooled into wet (Nov-Apr) and dry seasons (May-Oct) using annual coastal rainfall, seasonal rainfall for the Moreton South Coast region and seasonal SOI values. Mullet catch and CPUE correlated with SOI in May-Oct (range of r = 0.61 to 0.65; P < 0.01; n = 17). This correlation was weaker and non-significant for the wet season (r = 0.36 to 0.39; P > 0.05, n = 17) but still led to an overall significant correlation between annual SOI and catch (r = 0.49 to 0.57; P < 0.05; n = 17), which was contrary to the rainfall-CPUE relation. Therefore, the SOI (May-Oct) could be a useful environmental parameter for mullet catch prediction.
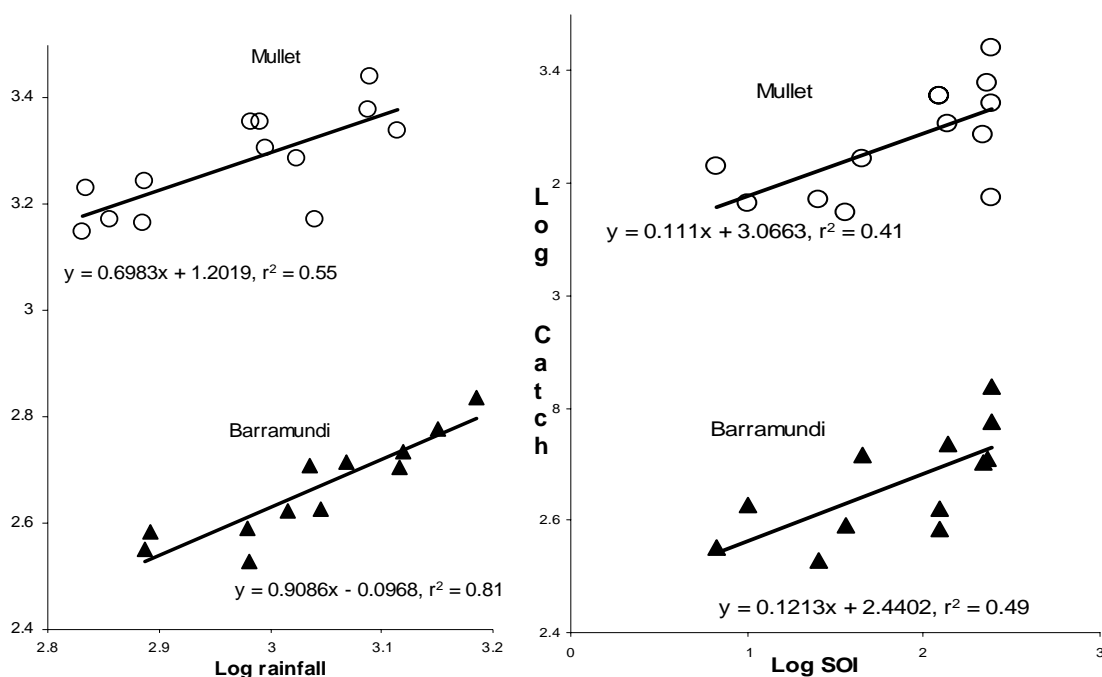
Figure 33: Relationship between (a) annual coastal rainfall and (b) SOI in Queensland and annual commercial catch (1988-2000) of mullet and barramundi from the Gulf of Carpentaria. Data for catch and flow are both on $\log_{10}$ axes. Lines are the least-square regression lines

Table 15: Significant Pearson correlation coefficients (r) between commercial catches and rainfall in Queensland. Categories: annual catch based on 1988 – 2000 data (n = 13), summarised monthly catch based on 1988-2004 data (n = 12), annual catch based on 1988-2004 data (n = 17), monthly catch based on 1988 – 2004 (n = 204) (n = 153 Barramundi open season only) (* P < 0.05; ** P < 0.01). NP = North Peninsula, SP = South Peninsula, LC = Lower Carpentaria, HNC = Herbert North Coast, ECC = East Central Coast, PSC = Port Curtis South Coast, MSC = Moreton South Coast, SOI = Southern Oscillation Index. An asterix indicates the use of total catch in kg.

Species Catch	NP	LC	SP	BNC	HNC	ECC	PSC	MSC	SOI	n
Barramundi East Coast*	-	-	.64**	.63**	.77**	-	-	-	-	13
kg d⁻¹	.68**	.72**	.71**	.54*	-	-	-	-	-	13
Gulf *	.85**	.73**	-	-	-	-	-	-	.77**	13
kg d⁻¹	.89**	.78**	-	-	-	-	-	-	.57*	13
Annual*		.58*	-	-	-	-	-	-	-	17
Seasonal*	-	-	-	.58*	.61*	-	-	.84**	-	12
kg d⁻¹	-	-	-		.67**	-	-	.79**	-	12
Monthly*	.16*			.45**	.42**	.24**	.21**	.21**	-	204
kg d⁻¹	.27**	.33**	.25**	.38**	.39**	.22**	.17*	-	-	153
Mullet*	-	-	-	-	-	-	.54*	.74**	.63**	13
Annual*	-	-	-	-	-	-	-	.78**	.51**	17
kg d⁻¹	-	-	-	-	-	-	-	.67**	.57*	17
Seasonal*	-	-	-	.-	-.	-	-.61*	-	-	12
kg d⁻¹	-	-	-	-	-	-	-.62*	-	-	12
Monthly*	-	-	-	-	-	-	-.18**	-17**	-	204
kg d⁻¹	-	-	-	-	-	-	-.15**	-.15**	-	204
Flathead*	-	-	-	-	-	.54*	-	.49*	-	17
kg d⁻¹	-	-	-	-	-	-	-	.55*	-	17
Mud Crab*	.74**	.75**	.79**	.77**	.72**	-	-	-	-	13
kg d⁻¹	-	-	-	.50	0.63**	.55*		-	-	13
Annual*	.55*	-	.56*	-	-	-	-	-	-	17
kg d⁻¹	.52*	-	-	-	.57*	-	-	-	-	17
Seasonal*	-.74**	-	-	-	-	.69*	.70**	.86**	-	12
kg d⁻¹	-	.72**			.63*	.63*	.62*	.80**	-	12
Monthly*	-.24**	-.18**	-	-	.19**	.19**	.22**	.29**	-	204
kg d⁻¹	-.18*		.14*	-	.22**	.26**	.19**	.34**	-	204
Tiger Prawn*	-	-	-	.63*	.51*	-	.53*	-	-	17
Seasonal*	-	-	-	-	-	-.58*	-	.63*	-	12
Monthly*	-.29**	-	-.19**	-.16**	-.14*	-.27**		.33**	-	204
kg d⁻¹	-.38**	-	-.45**	-	-	-	-	.31**	-	204
Endeavour Prawn*	-	-	-	.58*	-	-	-	-	.61*	17
Seasonal*	-.68*	-	-.62*	-	-	-.69*	.77**	.89**	-	12
kg d⁻¹	-.61*	-	-.64*	-	-	-.63*	.87*	.91**	-	12
Monthly*	-.37**	-	-.41**	-.22**	-	-.21**	.25**	.38**	-	204
kg d⁻¹	-.42**	-	-.39**	-.26**	-	-.15**	.28**	.27**	-	204
Greasy Prawn*	-	-	-	-	-	-	-	.49*	-	17
Seasonal*	-	-	-	-	-	-	.75**	.91**	-	12
kg/day	-	-	-	-	-	-	-	.91**	-	12
Monthly*	-	-	-	-	-	-	.28**	.35**	-	204
kg d⁻¹	-	-	-	-	-	-	.18**	.49**	-	204

Table 16: Significant Pearson correlation coefficients (r) between commercial catches and average coastal air temperature in Queensland. Sample sizes, abbrevations and significance as indicated in Table 15

Species	Catch/CPUE	N P	S P	B N C	H N C	E C C	P S C	M S C	n
Barramundi	Annual*	-	-	-	-	.83**	.47*	-	17
	Monthly*	-.34**	-	-	.23**	-	-	-	204
	Monthly kg d⁻¹	-.27**	-.25**	-	.38**	-	.61*	-	153
Mullet	Seasonal*	-	-	-	-	-.75**	-	-.59*	12
	Seasonal kg d⁻¹	-	-	-	-	-.76**	-	-.60*	12
	Monthly*	-	-	-	-	-.36**	-.57**	-.52**	204
	Monthly kg d⁻¹	-	-	-	-	-.31**	-.59**	-.54**	204
Flathead	Seasonal*	-	-	-	-	-.59*	-	-.90**	12
	Seasonal kg d⁻¹	-	.66*	-	-	-	-	-.82**	12
	Monthly*	-	-	-	-	-33**	-.39**	-.65**	204
	Monthly kg d⁻¹	-	-	-	-	-	-.25**	-.55**	204
Mud Crab	Annual*	-	-	-	-	.87**	.55*	.66*	17
	Seasonal*	-	-	-	-	-	.66*	.76**	12
	Seasonal kg d⁻¹	-		-	-	-	-	.64*	12
	Monthly*	-	-	-	-	.31**	.44**	.58**	204
	Monthly kg d⁻¹	-	-	-	-	.18*	.34**	.46**	204
Tiger Prawn	Annual*	-	-	-	-	-.35	-	.43*	17
	Seasonal*	-	-	-.58*	-	-.84**	-	.59*	12
	Monthly*	-	-.23**	-.46**	-.25**	-.64**	-	.51**	204
	Monthly kg d⁻¹	-	-.19**	-.31**	-.22**	-.28**	-	.35**	204
Endeavour Prawn	Annual*	-	-	-.38	-.36	-.74**	−.55*	-.36	17
	Seasonal*	-	-.70**	-.68*	-	-.83**	.77**	.92**	12
	Seasonal kg d⁻¹	-	-.82**	-.60*	-	.58*	.87**	.91**	12
	Monthly*	-	-.52**	-.51**	-.24**	-.47**	.32**	.70**	204
	Monthly kg d⁻¹	-	-.31**	-.49**	-	.25**	.45**	.48**	204
Greasy Prawn	Annual*	-	-	-	-	-	.54*	-	17
	Seasonal*	-	-	-	-	-	.71**	.95**	12
	Seasonal kg d⁻¹	-	-	-	-	-	.59*	.93**	12
	Monthly*	-	-	-	-	-	.29**	.53**	204
	Monthly kg d⁻¹	-	-	-	-	-	.17*	.69**	204
Bay Prawn	Seasonal*	-	-	-	-	-.61*	-	.97**	12
	Seasonal kg d⁻¹	-	-	-	-.76**	-	-	.96**	12
	Monthly*	-	-	-	-	-.17*	-	.73**	204
	Monthly kg d⁻¹	-	-	-	-	-.16*	-	.79**	204
Whiting	Annual*	-	-	-	-.36	-	-	-.35	17
	Seasonal*	-	-	-	-	-	.44	-.91**	12
	Seasonal kg d⁻¹	-	-	-	-	-	.73**	-.87**	12
	Monthly*	-	-	-	-	-	-	-.50**	204
	Monthly kg d⁻¹	-	-	-	-	-	-	-.46**	204
School Prawn	Seasonal*	-	-	-	-	-	.61**	.61**	12
	Monthly*	-	-	-	-	-	.53**	.34**	204

6.3.3 CLIMPROD model

Analyses with CLIMPROD using the Schaefer model gave poor results with values of the goodness of fit parameter r^2 varying from 0.02 – 0.20. When the SOI and rainfall data were incorporated into landings and effort data, different models were obtained that had medium to high r^2 values (0.30 – 0.87), and a wide range of jacknife r^2 (0.001 –

0.85). Further statistical analyses showed that there were small variations in the jacknife r^2 over the time series for barramundi (*Lates calcarifer*) east coast and mud crab (*Scylla serrata*) catches suggesting that the model for these two species were robust. Mullet (*Mugil* spp.) also presented a strong case, with a conventional r^2 of 0.75 (Table 17).

Table 17: Production models suggested by CLIMPROD that include rainfall or SOI. The maximum sustainable yield (MSY) for mud crabs (*Scylla serrata*) may be overestimated due to a constant increasing effort. CPUE is landing per unit effort kg d^{-1} y^{-1}; E is fishing effort; V is the environmental variable, and a, b, c and d are constants. The MSY was estimated by the median of the MSY. [a] 1988-2000; [b] 1988-2004; [c] rainfall; [d] SOI

Species	Conventional r^2	Jackknife r^2	MSY in t (+ SE)	CPUE model	Modification
Barramundi East coast [a; c]	0.93	0.85	246 (±40)	$aV^b \exp(EcV^d)$	Age at recruit. 4yr
Barramundi [b; c]	0.52	0.35	45 (±15)	$aV + bE$	Age at recruit. 4yr
Mullet [a; c]	0.75	0.40	2076 (±101)	$a + bV + cV^2 + dE$	Age at recruit. 3yr
Mullet [a; d]	0.61	0.35	1909 (±135)	$(aV + bV^2) \exp(cE)$	Age at recruit. 3yr
Mullet May-Oct [b; d]	0.34	0.09	1708 (±433)	$aV(1 + bV) \exp(cV(1 + bV)E)$	-
Mud Crab [b; c]	0.87	0.75	221 (±63)	$a + bV + cE$	Exploited year classes 3
Prawn King [b; c]	0.41	0.19	3176 (±281)	$(a + bE)^{(1/c-1)}$	Exploited year classes 3
Prawn Tiger [b; c]	0.61	0.39	2464 (±313)	$(a + bV) \exp(cE)$	Age at recruit. 1yr
Catch, all [b; c]	0.65	0.47	n.a.	$aV + bV^{2+c}$	Exploited year classes 3

6.4   Discussion and conclusions

6.4.1   *Mechanisms underlying the climate-fishery relationships*

Understanding the biological mechanisms underpinning relationships between climate parameters and fish production is essential to interpreting the outcomes. Proposed mechanisms for the connection between estuarine fishery catch and rainfall include: (1) trophic linkages - changes to primary or secondary production that result from the

increased nutrient runoffs from terrestrial sources (Aleem, 1972; Salen-Picard et al., 2002); (2) changes in distribution as a consequence of altered (expanded, reduced or connected) habitats (Loneragan and Bunn, 1999) and with it, changes of catchability; and (3) changes in population dynamics such as recruitment, growth, survival, abundance, assemblages and migration behaviour (Copeland, 1966; Peters, 1982; Drinkwater, 1986; Drinkwater and Frank, 1994; Gillanders and Kingsford, 2002) as well as cohort or year-class strength during the first year of life (Quiñones and Montes, 2001). The latter point (3) is also relevant for the effects of temperature. The relationships between estuarine catch, rainfall and temperature are potentially confounded by other factors such as fishing effort (Browder, 1985; da Silva, 1985). The level of exploitation of the population by the fishery has influence on the catch and therefore the climate-fish catch analyses (Vance et al., 1985).

The analysis of the Queensland wild capture data between 1988 and 2004 indicated that dry years, often associated with El Niño, lead to reduced overall fish catch and wet years translated to higher catches. This dependence may also reflect the known influence of SOI on some important non estuary-dependent commercial fish species, e.g., mackerels (Yáñez et al., 2002), and tuna (Scombridae) (Lehodey et al., 2003). Similarly, on a smaller scale the regional data for south-east Queensland king, tiger and bay prawn catches showed a strong reduction during El Niño events for the study period (Fig. 31). However, a detailed analysis of the prawn catch data demonstrated that effects on penaeid prawns often have contrasting regional differences – rainfall was positively correlated with catch for south-east Queensland but negatively for north Queensland (Table 15). Prawn life history stages respond to rainfall differently in time, resulting in either positive or negative impact (Vance et al., 1998). Rainfall stimulates the movement of juvenile banana prawns (*Penaeus merguiensis*) and school prawns (*Metapenaeus macleayi*) from their nursery habitats (Glaister, 1978; Vance et al., 1998). However, at an early life stage of prawns heavy rainfall can force the juvenile prawns to leave their nursery habitat and result in lower biomass and therefore lower prawn catch (Vance et al., 1998). Beside hydrological and biological differences, there are also differences in the prawn trawler fleet in Far North Queensland and the Gulf of Carpentaria, which mainly operates from April to May (Kailola et al., 1993). The strong influence of effort on tiger and king prawn catches makes a Queensland-wide simulation of climate effects on these species difficult. On the other hand, the positive relationship between rainfall, SOI, catches, CPUE and CLIMPROD outcomes for mud

crabs (*Scylla serrata*), barramundi (*Lates calcarifer*) and mullet (e.g., *Mugil cephalus*) on a regional as well as state-wide scale suggests that these environmental parameters may be used for further modelling. Monthly and yearly average temperatures did not have a significant influence on the fish catch for two selected sites but patterns were detected for prawns (Penaeidae) and mud crabs for which higher monthly air temperature resulted in higher catch rates in south-east Queensland. A shift from positive temperature effects towards a negative relationship for the northern regions of Queensland may be related to adaptation of crustaceans to temperature (Kutty et al., 1971; Deering et al., 1995; Ruscoe et al., 2004). There are temperature threshold limits for crustacean zoea and megalopa larval stages. Staples and Heales (1991) found the lowest survival rates of juvenile *P. merguiensis* at high temperature (30°C) and low salinity (< 10 ppt). The temperature data were only available for a number of stations along the coast and differences between air and water temperature can be significant, the temperature analyses are therefore somehow compromised.

Regional differences in the rainfall–catch relationship were significant and exemplified the major fishing areas for mullet (e.g., *Mugil cephalus*) in the south-east, barramundi (*Lates calcarifer*) in the north, flathead (*Platycephalus* spp.) in the south-east and mud crab (*Scylla serrata*) on the east coast of Queensland and in the Gulf of Carpentaria. For instance, almost 80 % of the Queensland mullet catch is harvested from April to August (dry season) with ~ 40 % of the mullet-harvesting effort. Monthly correlation between mullet, rainfall and temperature is therefore negative (Table 15 and 16). The Moreton Bay region in the south-east of Queensland dominates harvest with ~ 70 % contribution, and the Fraser–Burnett region provides ~ 25 % of the harvest (Williams, 2002). This is reflected by a significant relationship between mullet catch and annual rainfall in the Moreton Bay region (Table 15). The general reproductive pattern of *M. cephalus* involves migration from either fresh or estuarine waters to offshore waters where spawning takes place in large schools. Larvae and fry then migrate to inshore estuaries where they inhabit shallow, warm water in the intertidal zone. This reproductive cycle suggests that wetter periods between May and August may stimulate the migration of mullet (*Mugil* spp.) out of the estuaries and therefore increase their catchability, since most of the mullet is caught along beaches and not in estuaries. Dryer but warmer wet seasons would, however, increase algal productivity in the estuaries and therefore strengthen the 0+ year-class with increased catches, separated by the period it takes to 'recruit' to the fishery, demonstrating a time-lagged

response. A similar mechanism operates for the increased catches of mud crabs (*Scylla serrata*), which are not caught until they are > 12–24 months old because of legal size limit regulations. Higher rainfall and hence river flow stimulates the downstream movement of mud crabs (Hill et al., 1982) and this could increase their catchability in the lower estuary and bay. The reduction in numbers of subadult and adult crabs in the river systems may also enhance the survival of juveniles because of reduced cannibalism and competition for burrows, which may be the explanation for the strong 1-year time-lagged correlation between rainfall and mud crab (*Scylla serrata*) catch. Such a lagged effect of river flow on catches has also been recorded for the relationship between annual rainfall and barramundi catches in the Gulf of Carpentaria and some estuaries on the east coast, due to enhanced productivity and increased survival and/or growth of the juvenile stages (Staunton-Smith et al., 2004). The lagged response of fish catch to rainfall is an important characteristic for a number of species. Overall, there can be positive effects of rainfall (e.g., availability of food) or negative effects (e.g., higher mortality of juveniles, reduction of usable nursery habitat) because of changes in salinity and/or turbidity (Staples, 1980; Potter et al., 1991). These considerations collectively suggest mechanisms linking productivity, rainfall and catchability (Kimmerer, 2002).

However, to prove any underlying correlations between environmental condition and recruitment is extremely difficult. There is as yet insufficient knowledge about impacts of climate changes on regional ocean currents and physical-biological linkages. The argument for including climate factors in assessments is clearly strengthened if the mechanistic links are better understood (see also Fig. 27). I expect rainfall and temperature to alter fish species composition in nearshore and estuarine waters. Further analyses in this direction are necessary. Analysis of relationships between catch and environmental variables are often used in observational studies to derive insights into the factors driving the distribution and abundance of fisheries species at the whole-estuary scale (Tyler, 1992) and provide directions for research and help establish modelling approaches (Planque et al., 2003).

6.4.2   Predicting catch limits using different modelling approaches

Simple modelling of the economic impacts of climate change can be performed by regressing SOI and rainfall with fish catch over time (Xie and Hsieh, 1989; Beamish and Bouillon, 1993; Aaheim and Sygna, 2000). This method has been applied successfully to a number of studies (Vance et al., 1985; Loneragan and Bunn, 1999; Growns and James, 2005) and can provide first ideas on the relation between rainfall and fish catch. However, the regression approach has been criticised because of: (1) the confounding effects of stock size and fishing pressure (Walters and Collie, 1988); (2) the likely non-linearity of linking mechanisms (Baumann, 1998) and the probability of multiple mechanisms; (3) the lack of ability to prove causality (Quiñones and Montes, 2001); and (4) the uncertain predictive capability as a consequence of long-term climatic variation or human-induced changes (e.g., habitat loss, pollution).

The results from the CLIMPROD program were overall encouraging and will allow incorporation, for example, into bioeconomic models. The application of the CLIMPROD program to Queensland yield and effort and seasonal rainfall inputs resulted in validated modelling of the fishery for some estuary-dependent species and the analyses can assist in refining hypotheses developed from life history assessment, and provide guidance as to where further research should be focused. The case studies suggested that regional and seasonal differences in rainfall and landings have to be considered but that the positive relationship of certain fish species with rainfall along the coast is valid on a broad scale and that there are trends for temperature–catch relationships. However, the models including rainfall or SOI as environmental variables (Aksnes et al., 1995) also showed that more refined modelling tools are required to provide better outcomes. Simple surplus production models have been criticized because they lack biological realism, resulting in poor predictive ability. One of the weaknesses of the biomass production modelling approach is that changes in community composition will not be detected. For example, they do not consider the structure of the stock by age, size or species. In many cases, however, more sophisticated age-structured models do not necessarily perform better, owing to difficulties in additional parameters estimation (Ludwig and Walters, 1985). Dynamic software can achieve valid outcomes for the efficient management of a fishery as they include several dynamic, i.e., non-equilibrium, models (Fréon et al., 1993; Punt and

Hilborn, 1996). The limited availability fish catch data set and information on species biology restricts the use of other models.

Some further problems exist with analysing the data including: (1) the failure to account for autocorrelation in the time-series; (2) over-fitting of a function due to unknown behaviour outside of the data sample; and (3) the identification of spurious correlations where the variables are statistically related but are not causally linked (Hilborn and Walters, 1992). A number of statistical techniques are available to deal with autocorrelations although they tend to reduce the test power (Drinkwater, 1987). The risks of over-fitting can be reduced by careful exploratory correlation analyses and applying corrections to threshold probabilities for multiple comparisons (Carscadden et al., 2000). The choice of the environmental variables is often the key factor to avoid spurious correlation. In general, even in the case of surplus production models, a minimum knowledge of the stock and of the species biology is required. Further analysis of the data may be via low and high frequency analysis (e.g., ARIMA) (Pyper and Peterman, 1998). ARIMA models can remove the autocorrelation and/or move average structure of the series transforming it into 'white noise'. White noise can be defined as a stochastic process by a marginal distribution function. Transfer functions allow ARIMA outputs to be associated with environmental or other external variables to better explain seasonality and noise, or an assessment of changes of time series (intervention analysis) may help to detect anomalous events (Rothschild et al., 1996). However, consistent long-term data sets for seasonally-based analysis with > 50 data points for effective time-series analysis are not available for Queensland.

A more practical modelling approach alongside the use of other statistical tools (e.g., CLIMLAB 2000, GAM & GLM) has been demonstrated by Kasulo and Perrings (2004), who studied the influence of changes in environmental conditions for Lake Malawi. They suggested a modified Gordon-Schaefer model (Gordon, 1954; Schaefer, 1957) to include both environmental and bio-economic diversity variables (Simonit and Perrings, 2005). Since economics are an important aspect driving fishing activities, bio-economic modelling is much more realistic than purely biological (or purely economic) approaches (Sparre and Willmann, 1993; Seijo et al., 1994; King, 1995; Pascoe, 1997; Seijo et al., 1997; Helstad, 2000; Ulrich et al., 2001). However, a single model cannot describe all aspects of the environment that are relevant. A coupled model, in which different types of models are used to explain different parts of the system, can provide better outcomes. One important part of this approach is a trophic model (Pauly et al.,

1996). This summarises the species present in an ecosystem, their abundances, and the quantities of one species consumed by another. For example, estuarine fish such as barramundi (*Lates calcarifer*), king threadfin (*Polydactylus sheridani*) and catfish (Ariidae) are major predators of juvenile tiger prawns (*Penaeus esculentus, P. semisulcatus*) (Salini et al., 1990). Such a model should include movements, feeding and survival rate. The integrated energy balance ECOPATH with ECOSIM software (Walters et al., 1997) for dynamic simulation modelling can determine the long-term catch rates and biomass that result under varying degrees of fishing mortality (Christensen and Walters, 2004; Pitcher, 2005). ECOSPACE, the spatial version of ECOSIM can also dynamically allocate biomass across a grid map (Walters et al., 1997; Okey and Pauly, 1999). However, the lack of local information for the modelling would create fatal uncertainty for the simulations with ECOPATH and ECOSIM. Better fitting can be achieved with time-series data on biomass, fishing mortality and survey information when available.

In this chapter, I identified a clear link between estuarine fish catch and climate variables on a broad scale but also showed that there are differences between species and regions to be considered in modelling approaches. In conclusion, the fisheries should not be managed without including proactive responses to changes in rainfall, temperature and the SOI and their predicted future trends. The SOI can be predicted with some accuracy for up to six months including some estimation for rainfall and temperature. Such prediction could be a significant element in an environmentally-sensitive management policy. It is necessary to identify a more comprehensive management policy that will ensure sustainability even under conditions of higher environmental pressure, e.g., from more frequent and more extreme El Niño conditions resulting in a reduction of rainfall. Predictions of SOI, rainfall and temperature are possible and should not only be used in agricultural forecasts. Overexploitation and economic losses can be avoided when planning ahead by targeting fish species which are less influenced by El Niño events than others, e.g., mullet (*Mugil* spp.) versus whiting (*Sillago* spp.). The multi-species fishery in Queensland would have the opportunity and capacity to undergo such shifts to increase both stable fish stocks and economic outcome.

CHAPTER 7

GENERAL DISCUSSION – IMPLICATIONS FOR SUSTAINABLE COASTAL FISHERIES

7.1. Tidal wetland connectivity and coastal fisheries

The passive transport of dissolved and particulate nutrients and organic matter from productive estuarine marshes to coastal and offshore waters, termed "outwelling", was considered critical in supporting secondary production of many coastal fisheries (Odum, 1968). This concept of trophic linkage between productive estuaries and coastal fisheris has since strongly influenced estuarine research. Regardless of the direction of the overall net flux of energy across ecosystem boundaries, estuaries are now widely recognised as key drivers of coastal fisheries (Cappo et al., 1998; Blaber, 2000; Dunning et al., 2001; Able, 2005). Fish migrating from estuaries to coastal areas are exporting biomass from estuaries (Deegan, 1993) as most estuarine fish faunas around the world are dominated by 'transient' species that spawn offshore and utilise estuarine systems as nurseries before returning to nearshore adult habitats (Yáñez-Arancibia, 1985; Deegan, 1993; Kneib, 1997). Tidal wetland habitats in an estuary thus support secondary production and catch of commercial fisheries (Baran and Hambrey, 1998), but not all habitats contribute equally as nurseries (Beck et al., 2001). Outcomes in this thesis support the hypothesis that a combination of mangroves, seagrass and saltmarsh and their connectivity determined the importance of estuaries to nekton (Chapters 3 and 4). This is in line with the "chain of habitats" concept (Chong et al., 1990; Nagelkerken et al., 2001). Our present understanding of drivers of coastal fisheries within tropical and sub-tropical Indo-Pacific estuarine systems was largely based on single estuary or single habitat studies, and on simple area-fish catch relationships. The role of various seascape characteristics of estuaries was not a central focus of these studies (Baran and Hambrey, 1998; Manson et al., 2005b). Habitat connectivity (or its inverse, isolation) has long been recognized as a fundamental factor in determining the distribution of terrestrial species (MacArthur and Wilson, 1967; Fahrig and Merriam, 1985). Surprisingly, to date, there were no studies that devised a connectivity index for tidal

wetlands or assessed fish movement in relation to availability of connected tidal wetlands in Australia. Some fish species utilising estuarine habitats are opportunists (Potter and Hyndes, 1999) which is probably a survival strategy for a dynamic and fast changing environment. It is therefore not surprising that past studies were unable to identify the habitat contributing most significantly to estuarine fish production. Geomorphic and hydrologic conditions determine where habitats occur and their extent in an estuary. As changes in such conditions are fundamentally gradual, estuarine habitats are usually inter-connected and should be regarded as a network of adjacent ecosystems. Studies on landscape connectivity are therefore important as connectivity influences a range of ecological processes (e.g., dispersal of larvae) that impact on individuals, populations, and communities in heterogeneous landscapes.

In addition to the configuration of habitats (Unsworth and Smith, 2007), climate has been shown to significantly influence fish abundance (Peltonen et al., 2007), their movement (Lehodey et al., 2003) and biology (Hart and Reynolds, 2002). Relationships between rainfall and fish catch based on single estuaries were found for Australia (Ruello, 1973; Gillanders and Kingsford, 2002). Some studies have demonstrated a dependence of prawns (Vance et al., 1996b) and fish (Whitfield, 2005) on temperature and rainfall in the Gulf of Carpentaria. This thesis found that rainfall and temperature were significantly related to a number of estuarine fish species catch and that this relationship varied accross geographical regions.

The Fisheries Act 1994 of Queensland protects fisheries resources in declared fish habitat areas and allows for regulations and the use of quota. Current management plans in Queensland include annual quota reviews on fisheries (e.g., Gulf of Carpentaria Inshore Fin Fish Management Plan 1999 and Fisheries Coral Reef Fin Fish Management Plan 2003). Changes to current quota are only considered when for instance a significant decline in CPUE (standardised in the same way) for a target species in a sector of the fishery occurs over three consecutive years or when there is a significant decline in the commercial catch of major fish species. However, such information is based on logbook data or surveys and does not take natural fluctuations into account. A fish population may collapse under high fishing pressure. This risk is increased when additional stressors such as higher maximum temperature or reduced rainfall are not taken into account in fisheries management. Furthermore, intensive fishing in nursery habitats or spawning aggregation sites can drive population to unsustainable levels.

The aim of this thesis was to gain a clear insight into the most important potential drivers of fish catch, and by doing so, to significantly enhance our understanding of the functioning of these important systems. This thesis has provided a different conceptual view to the 'traditional' single habitat perspective and evaluated the importance of estuarine habitats, and their characteristics, such as connectivity, for fisheries for 5,000 km of the Australian coast line. A landscape approach identified the major environmental variables influencing observed spatial patterns in fish and prawn catches for Queensland. The outcomes of this thesis support the notion that heterogeneity and connectivity of habitats at a large scale play a key role in determining the spatial distribution and abundance of fish species (MacArthur and Wilson, 1967; Kotliar and Wiens, 1990; McCoy and Bell, 1991). Furthermore, new methods to investigate the role of small-scale connectivity using a modified fish gate system showed the importance of proximity of habitats and strong site fidelity for four commercially and recreationally important fish species. The investigation contributed to the broader knowledge of how coastal habitats and climate influence fisheries catch and productivity, with significant implications for fisheries management.

7.2   Estuarine habitat connectivity

This thesis investigated relationships between connectivity and estuarine fish species on a broad scale using fish catch grids (Chapters 3 and 4) and on a small scale using fish movement data (Chapter 5). A number of estuarine characteristics were expressed as metrics and the fish catch data were split into ecologically meaningful species groups and regions for further analyses. For preliminary analyses, landscape connectivity was measured using the number of patches, perimeter, total area and the connectivity index CONNECT and for refined analyses, Euclidean distance, patch density, and length of shared habitat edges were used including all grids along the coast of Queensland (Chapter 4). Patch density and connectivity metrics explained a large and significant proportion of the variation in fish, crab and penaeid prawn CPUE distribution. Such landscape characteristics were equally important to size (wetland habitat area) and shape of habitats (wetland mean perimeter to area ratio). For individual habitat types, mud and sandflats were most important with large areas of mud and sandflats associated with tide-dominated, shallow estuarine systems providing habitat for a number of
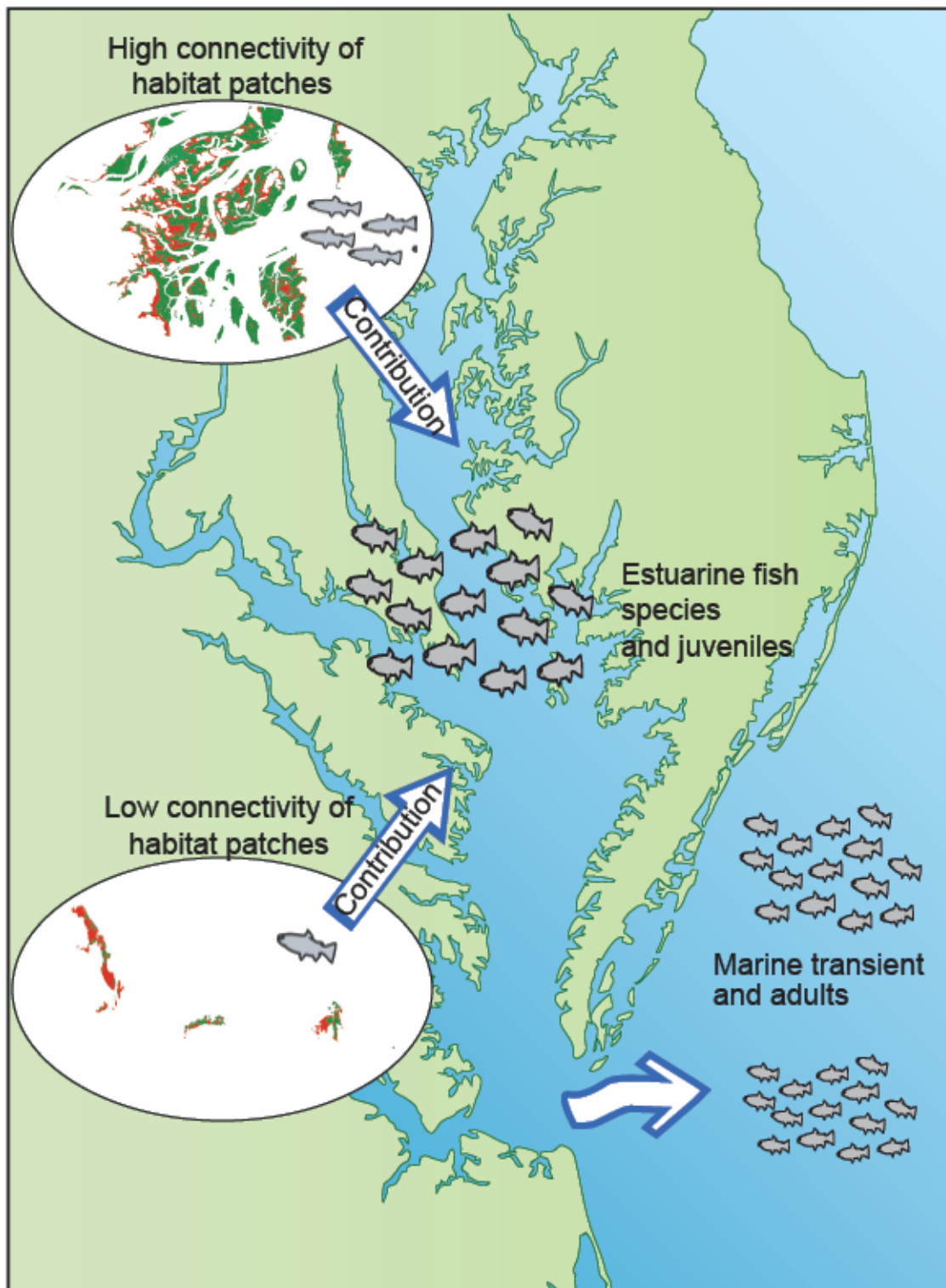
Figure 34: A schematic diagram illustrating the contribution of tidal wetland connectivity to estuarine fishes (symbols courtesy of Integration and Application Network, University of Maryland)

species (Ley et al., 1999; Saintilan, 2004). Overall, the results support the model that fish species depend on multiple habitats, with the overall catch and CPUE being dependent on the whole estuarine wetland habitat suite rather than any single habitat. For example, many fish and crustaceans only use mangrove forests for a part of the tidal cycle (Vance et al., 2002) since mangroves are flooded less than 50 % of the time (Duke, 2006), thus underlining the importance of easily accessible alternative estuarine habitats for the provision of food and shelter. Overall results suggested that high fish catch may have been maintained through a diversity of different estuarine habitats arranged with a high number of different patch types in close proximity. Provided there is parallel hydrolic connectivity (Jonsen and Fahrig, 1997), this arrangement then presents complementary resources throughout the home range of fish species (Dunning et al., 1992) .

Detailed analyses of movement for four commercially important fish species supported the outcomes of the broad-scale findings for connectivity. All four target species continuously used the mangrove creek during a four-month survey, showing close association with the site. Trends for presence and absence of fish in relation to habitat availability and connectivity were also demonstrated (Chapter 5). Daily use of mangroves by nekton was only for a maximum of six hours with at least 18 hours spent in other habitats. Therefore fish require other habitats (e.g., seagrass beds) in close proximity to reduce predation risk and energy costs before returning to the same site. The distribution and abundance of fishes in estuarine waters may be related to the trade-off between the costs and benefits of accessing or moving between alternate feeding or refuge habitats (Sheaves, 2005). Ultimately this will also decide whether the individual can successfully contribute to the adult population and take part at spawning events (Fig. 34).

7.3   Influence of climate on fisheries

Associated with the habitats are a number of abiotic factors such as rainfall and temperature, which influence fish and prawn catch (Chapter 6). Climatic factors such as temperature and rainfall explain significant variations in catch data, and their interactions affect the distribution of fish within various estuary types and broad geographic areas (Blaber, 2000). Such influences are particularly relevant for

consideration of fishery management in the face of anticipated future climate change. This thesis found that rainfall and temperature modulate catchability and/or vulnerability of estuarine fish species and affect estuarine fish catch. Analyses on total fish catch suggested that 20-30 % of the catch variability was explained by El Niño events. Significant correlations also resulted from analyses between mean annual coastal rainfall, SOI, mean air temperature and total annual commercial catches of mullet (*Mugil* spp.), barramundi (*Lates calcarifer*), flathead (*Platycephalus* spp.), whiting (*Sillago* spp.), mud crabs (*Scylla serrata*) and prawns (Penaidae spp.). These
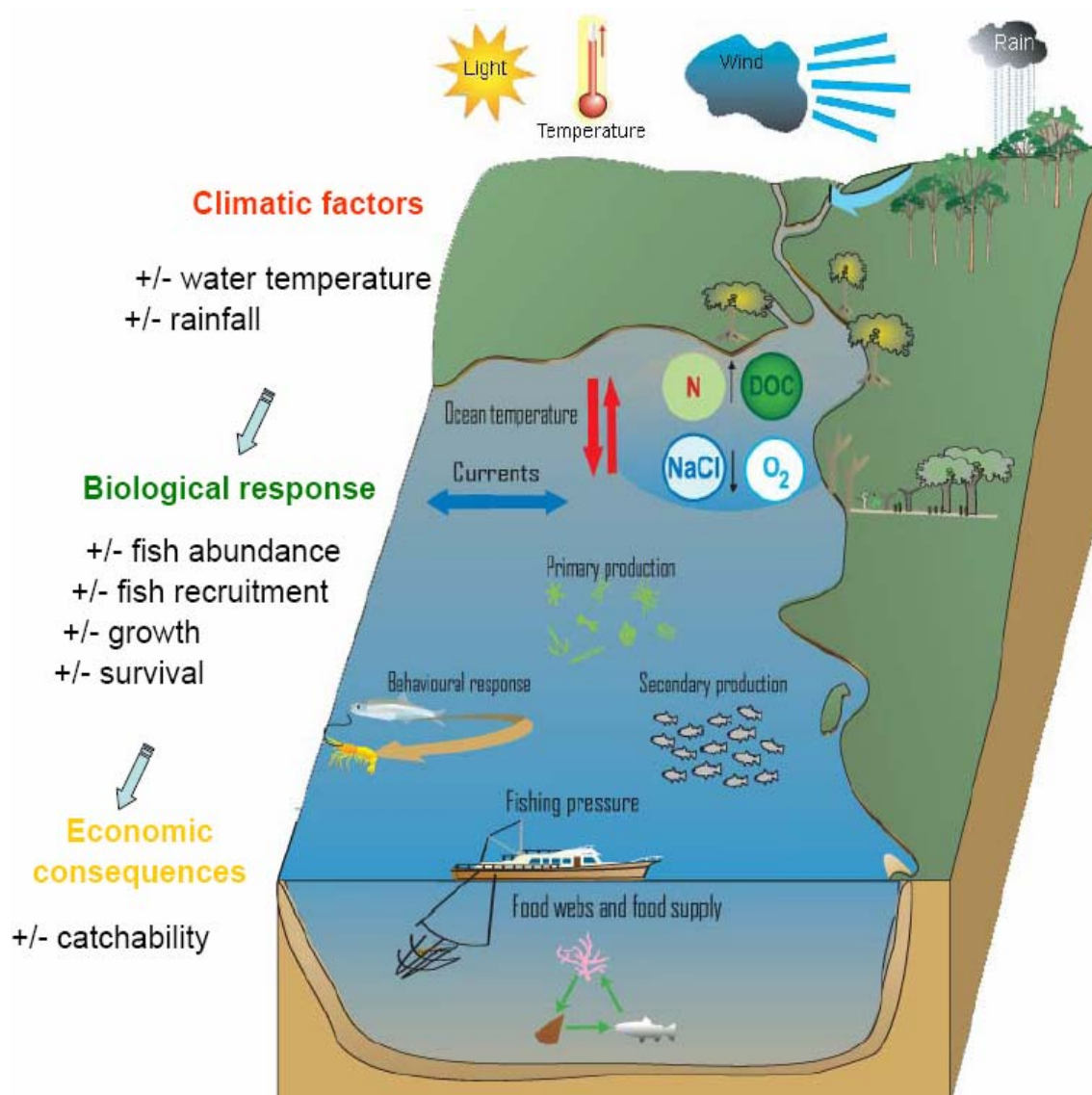
Figure 35: Influence of climatic factors on estuarine fisheries (symbols courtesy of Integration and Application Network, University of Maryland)

species constitute highly significant components of the commercial fish catch in Australia. Regional differences were evident between south-east Queensland, the tropical east coast and Gulf of Carpentaria, with rainfall and temperature changes exerting opposite effects in different regions. This divergence in response may be the result of genetic adaptation (Shaklee and Salini, 1985) and/or different behaviour/movement between regions. Varying temperature and rainfall can alter primary production, regional currents and water quality which may cause a change in fish migration, abundance, growth and survival. This has consequences for the catchability of commercial fish species and the quality or size of the catch (Copeland, 1966; Peters, 1982; Drinkwater, 1986; Drinkwater and Frank, 1994; Loneragan and Bunn, 1999; Quiñones and Montes, 2001; Gillanders and Kingsford, 2002) (Fig. 35). The relationships described are long term and it may take years or decades until climatic changes come into affect. However, some physical attributes in estuaries can change within days or month such as water quality and restricted freshwater flow by dams and are of cryptic influence in fish catch.

7.4   Drawbacks of current reporting system

The current reporting system for Queensland fisheries provides catch and effort information on 170 fish species or species groups from 1988 to the present. However, the CPUE sometimes does not necessarily reflect fish abundance properly (Klaer, 1993) when the fishery of interest is targeting fish aggregation sites. I have therefore used both CPUE and catch data in the analyses in order to compare the results between CPUE and catch. My findings demonstrated that large-scale analyses of this data set can provide meaningful results for fisheries managements and a better understanding of the function of estuarine habitats for fish. However, analyses on the scale of estuaries with the current data set were not possible or meaningful due to the low number of data points and the capture of data in 30-nautical-miles grids. As shown in Chapters 3 and 4, some species groups did not show any significant relationship with any of the variables (milkfish, *Chanos chanos;* mangrove jack, *Lutjanus argentimaculatus;* grunter, *Pomadasys* spp.; snapper, *Pagrus auratus*) or only weak positive r^2 values (flathead, *Platycephalus* spp.; flounder, *Pseudorhombus* spp.; rays, e.g., *Gymnura australis*;

threadfin unspecified, Family Polynemidae; sawfish, Family Pristidae) which was largely due to their low total catch throughout the fish catch grids. Splitting the fish catch data reduced the catch per grid to a level where analyses for some grids were impossible. Researchers cannot distinguish catches in some estuaries and bays because this would clearly identify an individual's activities, potentially breaching privacy requirements. A similar problem was also identified in other studies (Manson et al., 2005b).

Some data in Queensland have been collected at the 6-nm scale since 2001, which can significantly improve the usefulness of the data, but due to fewer records per grid the amount of catch has been too low for analyses in this study. The spatial scale problem may also be overcome by applying novel computational-intensive spatial analysis techniques such as neural networks, heuristic search and cellular automata for spatial data analysis (Andrienko and Andrienko, 2006). There are widespread efforts at improving the collection of production information (e.g., realtime tracking of fishing vessels), which may also improve data quality for analysis. For this study the use of the fish catch data was improved by: (1) refining parameters and using 1:100,000 mapping scale for all habitats (Chapters 3, 4 and 6); (2) classification of fish species based on known biology (Chapters 3,4 and 6); (3) conducting analyses over a range of spatial scales and regions (Dungan et al., 2002) with different statistical methods (Chapters 3); and (4) inclusion of all major estuarine habitats (Chapters 3, 4).

An *a priori* classification of fish species groups based on species distribution and importance for fisheries helped to reduce the error in the data set. A quantification of the data set error was not possible due to various unknown error sources. For example the quantification of fluctuating market requirements and extraction from fish catch is a complicated matter and has not been achieved for most fisheries in the world (FAO, 1997). Additional limitations in fish catch data included the recording of fish by common names, which can often result in confusion of species identity. Other factors such as stock-recruitment relationships, habitat changes, pollution impacts, competition and predation, management plans, fuel prices, also influence the fish catch data and may be considered in future analyses when such temporal changes can be quantified. The current data set can already be significantly improved by fish catch data from harvest by recreational anglers and charter operators (Hancock, 1995). A recent study undertaken by Greiner and Patterson (2007) estimated that 200 t of fish were caught by recreational anglers during six months out of a river system in the Gulf of Carpentaria in 2006.

Furthermore, commercial fish catch data can be improved by information on species-specific fish movement from the InfoFish database, a Queensland wide monitoring system of tagged fish (Chapter 5).

The overall lack of biological information, in particular, scale of movement for a number of commercially important species, e.g., sharks, is concerning (Camhi et al., 1998). There is a risk of overfishing if the biology of a species is either not well known or not considered in management decisions. The shark fishery provides an example showing increased catch rates by five times in 17 years but with inadequate management response. I will explore the issues of potential fishery mis-management by two examples in the following section.

7.4.1   Example 1: Shark fishery in Queensland

Recent declines in the populations and catches of many species of sharks have sparked considerable interest in the conservation of these elasmobranchs. Fishery-induced mortality of sharks has already dramatically reduced the abundance of some species in other parts of the world (Castro et al., 1999; Baum et al., 2003; Clarke et al., 2006). Sharks are likely to routinely move and migrate over large spatial scales and the effects of fishing in one area will be manifest in another. Catch records from 1988 to 2004 showed that the Queensland shark fishery increased beside the cascading effect of top predator removal in food chains resulting in significant changes in the composition and structure of fish communities (Myers et al., 2007) (Fig. 36). There is little information regarding habitat use and foraging behaviour of sharks. Large sharks are regarded as keystone predators that can influence the structure of ecosystems through predator-prey interactions (Heithaus, 2001).

Knowledge of the population size of exploited shark species is currently very limited and an estimation of carrying capacity not available. In consideration of the slow reproduction rates of sharks, the sustainability of the Queensland shark fishery is questionable and this biological 'time bomb' may explode within the next 10 years (Gallucci et al., 2006). The fishery is regulated under Queensland's Fisheries Act 1994 and Fisheries Regulations 1995 but there are no size or take and possession limits for most shark species. To significantly improve information on shark stocks, the catch

records should be species-specific and linked with biological information for each species.
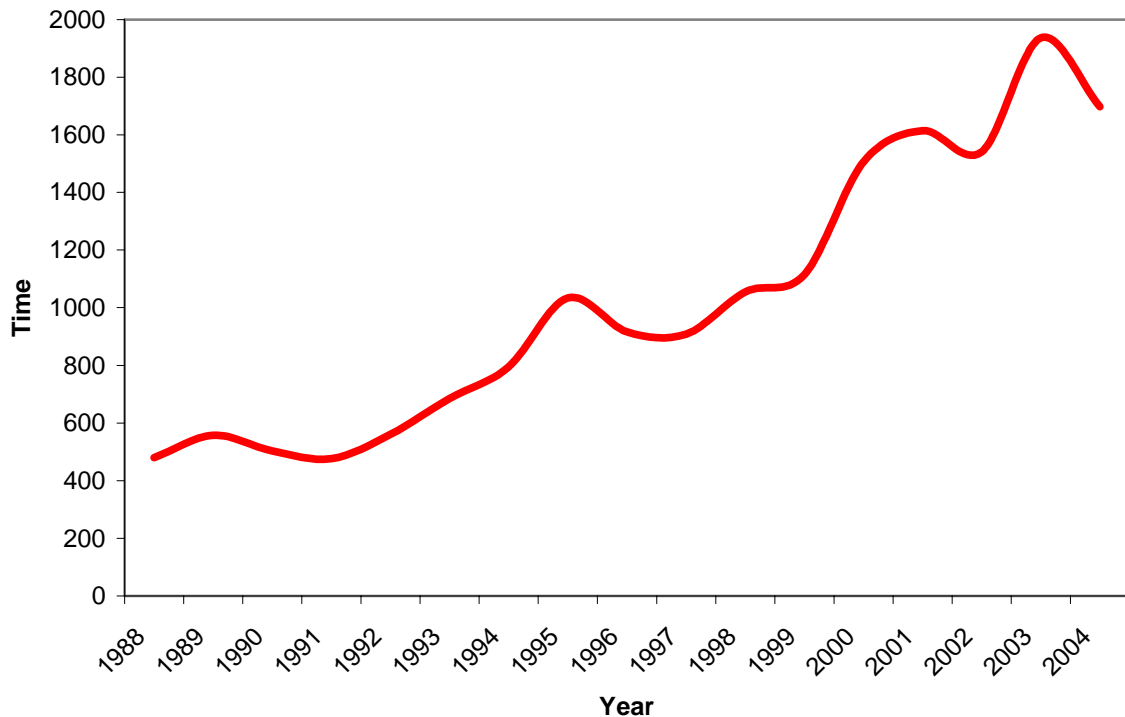
Figure 36: Total annual shark catch in Queensland during the period 1988-2004.

Shark species include black tip sharks (*Carcharhinus limbatus*, *C. tilstoni*), scalloped hammerhead shark (*Sphyrna lewini*), milky sharks (*Rhizoprionodon acutus*), sharpnose shark (*Rhizoprionodon taylori*), bull shark (*C. leucas*), spot tail shark (*C. sorrah*), white cheek shark (*C. dussumieri*), creek whaler (*C. fitzroyensis*), silky shark (*C. falciformes*), spinner shark (*C. brevipinna*), pigeye shark (*C. ambiosensis*), tiger shark (*Galeocerdo cuvier*), lemon shark (*Negaprion acutidens*), graceful shark (*C. amblyrhynchoides*), hardnose shark (*C. macloti*), grey reef shark (*C. amblyrhynchos*), white tip reef shark (*Triaenodon obesus*) (CRC Reef, 2005; DPI&F, 2005b)

7.4.2   Example 2: Recreational fish catch data

Recreational fish catch data sets from local and regional competitions held by fishing
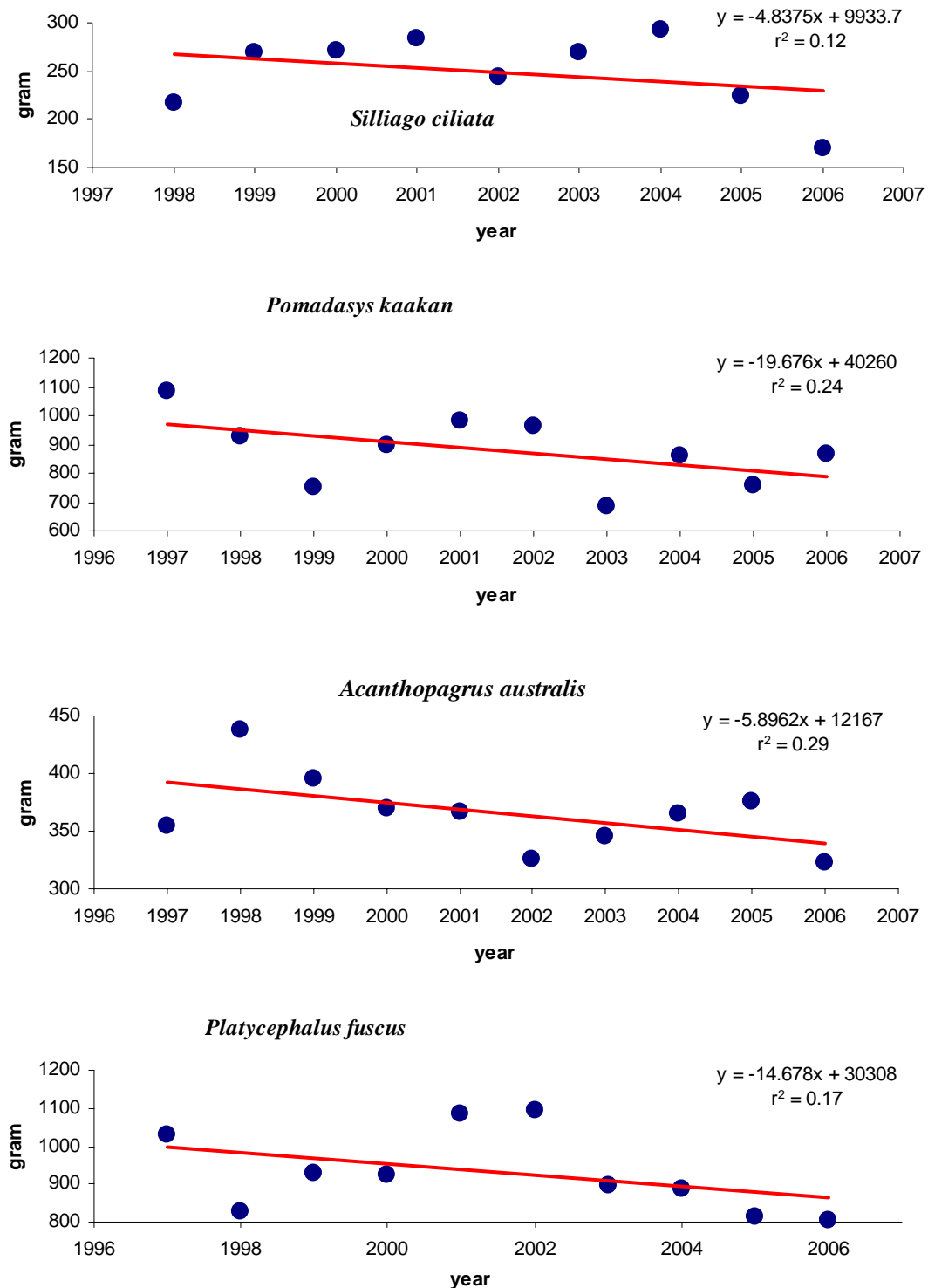
Figure 37: Trends of average annual weight of captured a) sand whiting (*S. ciliate*), b) grunter (*P. kaakan*), c) bream (*A. australis*), and d) flathead (*P. fuscus*) by anglers in the Burrum River from 1997-2006

clubs are largely unexplored. Recreational fish catch data are available for some regions (CFISH data base at http://chrisweb.dpi.qld.gov.au/chris/) but are generally based on return of survey forms provided by various fishing clubs on a voluntary basis. However, most fishing clubs keep their own records of fish species catch and weight. Such records often represent the only long-term data for a particular estuary or region. Anglers will usually target the dominant species and large individuals of fish to receive the most credit points. The data are less driven by external forces than commercial fish catch and can be a more reliable source of information for single estuaries. In Chapter 5 a nine-year data set from a recreational fishing club was used to investigate seasonal differences of species assemblages in the Burrum River system. Analyses of the data set also showed downward trends in weight per fish per year for the four dominant species (Fig. 37). Changes in body weight are likely to be a strong and universal indicator of fishing effects (Piet and Jennings, 2005). Recreational fish catch data can provide information on regional trends and allow comparison over various temporal and spatial scales. For example, whiting (*Sillago* spp.) experienced a significant decline in weight and total catch in the Burrum River, which is supported by anecdotal reports from local commercial fisher who claim a reduction of catch with increased efforts. Such information is important for improving current commercial catch data and for better stock management. If a species experiences weight decline over a number of years, a sufficient recovery time should be enforced.

7.5 Implications from estuary-fish catch relationships for fisheries management

My findings suggest a change of current fish stock management practices that focus on refined regional and species-specific management plans. For example, some regions have higher carrying capacities than others. The selection of fish habitat areas and fishing zones should be based on the quality of the area for fisheries. A proactive approach is needed to manage fisheries sustainably. This includes the calculation of quota on the basis of expected future events (e.g., rainfall and temperature) and on the area being fished. It is necessary to identify a more comprehensive management policy that will ensure sustainability even under conditions of higher environmental pressure, e.g., from more frequent and more extreme El Niño conditions. Overexploitation and

economic losses can be avoided if future planning adopts a flexible species-specific management strategy in which fish species which are less influenced by El Niño events than others, e.g., mullet (*Mugil* spp.) versus whiting (*Sillago* spp.), are targeted in different years. The multi-species fishery in Queensland would have the opportunity to undergo such shifts to increase both stable fish stocks and economic outcome. However, this would require a very flexible regulatory framework adapted to the dynamics of marine ecosystems.

7.5.1 Species-specific considerations

This thesis found species-specific dependencies on climate and habitat configuration, which should be considered in any future management plans and establishment of marine protected areas (MPAs). The outcomes reflect the biology of individual species and regional variations in climate and habitat characteristics. For barramundi (*L. calcarifer*), habitat and protection zones should be established in areas with high mangrove connectivity and total wetland area. The effect of river flow in the Gulf of Carpentaria and some estuaries on the east coast needs to be considered as it enhances productivity and increases survival and/or growth of the juvenile stages (Staunton-Smith et al., 2004). There are also region-specific climate influences, e.g., positive effects of higher temperatures in the East Coast Central (ECC) and Port Curtis South Coast (PSC) regions but negative effects in the North Peninsula (NP) and South Peninsula (SP). Such regional differences suggest the need for implementation of regional management plans for barramundi stocks. Similar outcomes were evident for the mud crab (*Scylla serrata*) and it is suggested that management efforts should be concentrated in areas with high wetland connectivity, high tidal wetland areas and seagrass beds along the EC but areas with high number of salt marsh patches, well connected mud and sandflats and large areas of mangroves in the GOC. Relationships with rainfall suggest to target mud crabs in wet years in the Moreton South Coast (MSC), PSC, Herbert North Coast (HNC), ECC but to avoid seasons with very low temperatures in the North Peninsula. River flow stimulates the downstream movement of mud crabs (Hill et al., 1982) and this could increase their catchability in the lower estuary and bay. The reduction in numbers of subadult and adult crabs in the river systems may also enhance the survival of juveniles because of reduced cannibalism and

competition for burrows, which may be the explanation for the strong correlation between 1-year-lagged rainfall and mud crab (*Scylla serrata*) catch. Warm temperature, however, increases mud crab catch, which is likely a result of enhanced feeding activity. High numbers of channels and mangrove patches were important for mangrove jack (*Lutjanus argentimaculatus)*, which is known to utilise all types of tidal wetlands as juveniles (Williams, 2002) and prefers sheltered areas in channels. Whiting (*Sillago* spp.) also falls into this group. These species are widely distributed generalists with fast-swimming demersal juveniles and adults that readily move across a wide range of estuarine habitat types (McKay, 1992). My analyses suggest protection zones in areas with high mangrove and tidal wetland connectivity. Whiting (*Sillago* spp.) should not be targeted during hot years on the EC. Protection efforts for mullet (Mugilidae spp.) should be concentrated in the MSC and PSC in areas with high number of wetland and salt marsh patches as well as mud and sandflats. Their reproductive cycle suggests that wetter periods between May and August may stimulate their migration out of the estuaries. Wetter periods increase the density of mullet (Marais, 1982) and therefore increase their catchability, since most of the mullet are caught along beaches and not in estuaries. Dryer but warmer wet seasons would, however, increase algal productivity in the estuaries and therefore strengthen the 0+ year-class with increased catches lagged by the period it takes individuals of the species to 'recruit' to the fishery. In seasons with high average temperature, mullet should not be targeted to allow enhanced growth to larger sizes. To strengthen bream (e.g., *Acanthopragus australis*) as well as blue swimmer crab (*Portunus pelagicus)* stocks, areas with tidal wetlands, large areas of mud and sandflats and high number of channels should receive particular attention, whereas wetland habitat patches and connectivity of mangroves and salt marsh are important for dart (*Trachinotus* spp.) and tailor (*Pomatomus saltatrix*) (Table 18).

Areas of high penaeid prawn production are characterised by the number of wetland habitat patches, wetland and mangrove connectivity, wetlands and mud and sandflats perimeter. Prawns (Penaeidae) are known to use a variety of estuarine (Dall et al., 1990; Halliday, 1995; Loneragan et al., 2005) habitats with a significant positive role of edge effects (Minello et al., 1994). While a range of juvenile prawn species have been caught either within, or (more often) in the river or creek channels adjacent to mangroves, only a few species have been found to be exclusively associated with mangroves as juveniles *Penaeus merguiensis* (Staples et al., 1985), *P. indicus* (Rönnbäck et al., 2002; Kenyon et al., 2004) and *P. penicillatus* (Chong et al., 1990)). Other species are less specific

Table 18: Summary of selected taxa showing significant relationships (P < 0.01 or P < 0.05) between estuarine habitat characteristics, average air temperature and rainfall. '+' indicates a positive relationship and '−' a negative relationship; superscripts: m = monthly, s = seasonal, a = annual. WA = wetland area; MA = mangrove area; SA = seagrass area; FA = mud and sandflat area; WPA = number of wetland patches; MPA = number of mangrove patches; SPA = salt marsh number of patches; MC = mangrove connectivity; LCE = length of connected habitat edge; MPA = number of mangrove patches; MCONNECT (100m) = mangrove connectivity; WCONNECT = wetland connectivity; SCONNECT = salt marsh connectivity; FCONNECT = mud and sandflat connectivity; WPARA = wetland mean perimeter to area ratio; SPARA = salt marsh mean perimeter to area ratio; MENN = mangrove Euclidean Distance; LCPA = length of shared edge line between habitats; CH = number of channels; WPERI = wetland perimeter; NR = no relationship detected; EC = East Coast, GOC = Gulf of Carpentaria; NP = North Peninsula; SP = South Peninsula; LC = Lower Carpentaria; HNC = Herbert North Coast; ECC = East Central Coast; PSC = Port Curtis South Coast; MSC = Moreton South Coast

Species	Habitat	Rainfall	Temperature
Lates calcarifer	**EC**: WPA, MCONNECT, LCE, WA, SA **GOC**: MPA, MCONNECT, WCONNECT,WPARA	+ GOC^{a,m} and EC^{a,m}	+ HNC^m, ECC^a, PSC^{a,m} - NP^m, SP^m
Mugilidae spp.	**EC**: WPA, MENN, LCPA **GOC**: SPARA, SPA	+ MSC^a - PSC^{m,s}, MSC^m	- MSC^{m,s}, PSC^m, ECC^{m,s}
Eleutheronema tetradactylum Polydactylus macrochir	**GOC**: SPARA, SPA		
Platycephalus spp.	NR	+ MSC^a and ECC^a	- MSC^{m,s}, PSC^m, ECC^{m,s}
Lutjanus argentimaculatus	MPA, CH	NR	NR
Lutjanidae spp.	WPA	NR	NR
Sillago spp.	MCONNECT, TW, WCONNECT	NR	- MSC^{a,s,m} + PSC^s
Acanthopragus australis	WA, FWA, CH	NR	NR
Trachinotus spp.	WPA, MENN, LCPA, FA, WA	NR	NR
Pomatomus saltatrix	WPA, MENN, LCPA, CH, SCONNECT	NR	NR
Portunus pelagicus	FA, CH	NR	NR
Scylla serrata	**EC**: WA, SA **GOC**: WCONNECT, SPA, FCONNECT, MA	+ MSC^{s,m}, PSC^{s,m}, HNC^{a,s,m}, ECC^{a,m,s}, BNC^a, Sp^{a,m}, LC^{a,s}, NP^a, - LC^m, NP^{s,m}	+ MSC^{a,s,m}, PSC^{a,s,m}, ECC^{a,m}
Penaeus esculentus, P. Plebejus	WPA, MENN, LCPA	NR	NR
Metapenaeus bennettae	MCONNECT, WPA, MENN, LCE	+ MSC^{a,s,m} PSC^{a,m}	+ MSC^{s,m} PSC^{a,s,m}
Metapenaeus endeavouri, M. ensis	FCONNECT, WPERI	+ MSC^{s,m}, PSC^{s,m}, - ECC^{s,m}, BNC^m, SP^{s,m}, NP^{s,m}	+ MSC^{s,m}, PSC^{s,m} - MSC^a, PSC^a, ECC^{a,s,m}, HNC^{a,m}, BNC^{a,m,s}, SP^{s,m}
Penaeus monodon, P. semisulcatus	WPARA, SA	+ MSC^{s,m}, PSC^a, HNC^a, BNC^a - ECC^{s,m}, HNC^m, BNC^m, SP^m, NP^m,	+ MSC^{a,s,m} - ECC^{a,s,m}, HNC^m, BNC^{s,m}, SP^m
Metapenaeus macleayi	WCONNECT, CH, WA	NR	+ MSC^{s,m}, PSC^{s,m}

in their choice of nursery habitat, e.g., *Metapenaeus ensis* (Staples et al., 1985), *M. monoceros* (Rönnbäck et al., 2002), *M. brevicornis* and *M. affinis* (Chong et al., 1990). Prawn species responded to rainfall and temperature with different discordance in time, either positively or negatively influencing their ontogenetic stages (Vance et al., 1998). Rainfall and temperature inlfuences were reflected in positive correlations for MSC and PSC and negative correlations for north Queensland in North Peninsula, SP, BNC (Barron North Coast), HNC and ECC, again suggesting regionally based management.

7.5.2   Implications for marine protected areas

The findings on habitat connectivity could guide the construction of marine protected area networks of various sizes and configuration to maintain ecosystem services and avoid further reduction of connectivity by habitat destruction. Regions with high connectivity should receive most attention when establishing habitat protection zones. Other important habitat indicators explaining CPUE and fish catch were the number of patches, perimeter to area ratio and wetland area, and these metrics should also be duly considered.

Fish catch grids along the EC, in particular, S31, V35W35, W36, H19 and H17 have high CPUE and high CONNECT (100m) values (Chapter 4, Fig. 13a), while the Fraser and Moreton Bay regions have the highest percentage of connected wetlands (Chapter 4, Fig. 13b). Other grids of similar value were H17 and I17 near Cairns and for some grids in the GOC (AC10, AG18, AC18). Large areas of wetlands are present in south-east Queensland with large areas of mud- and sand flats and the GOC, where saltpans dominate. Wetland edge to area ratio was highest in the south-east (V35W35), north of Queensland (G14) and the GOC (AG18, AD18). The size of habitat was not the best characteristic explaining fish catch but still important to explain fish catch variation. The analyses of CPUE, catch and habitat distribution revealed at least three broad regional groups, which should be managed separately: (1) south-east Queensland; (2) Central and Northern Queensland; and (3) the Gulf of Carpentaria (Chapter 3, Fig. 9a). Currently declared fish habitat areas only partly include areas of high wetland connectivity (Fig. 38) and only provides sufficient protection for marine flora - not fauna (Queensland's Fisheries Act 1994). To optimise habitat and fisheries management, a combination of estuarine habitats should be considered in fisheries

management, supporting the move from conventional single-species or single-habitat management to ecosystem-based management (NRC, 1999). Marine reserve networks can serve biodiversity conservation objectives by protecting regionally representative
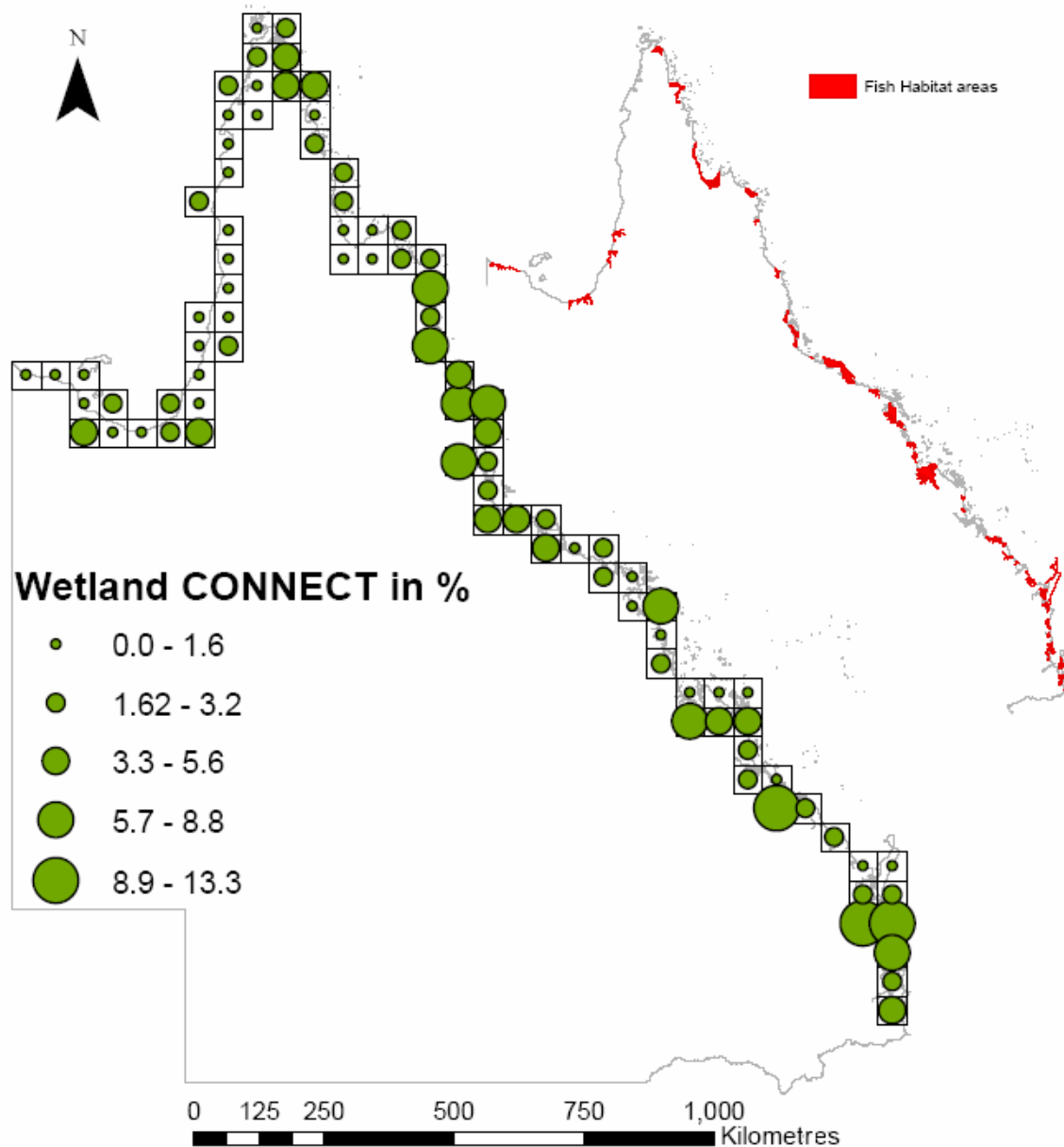
Figure 38: Distribution of the tidal wetland connectivity index (CONNECT 100 m) for 90 fish catch grids and declared fish habitat areas along the coast of Queensland

habitats at large spatial scales (from 100s to 1000s of km) (Sala et al., 2002) and excluding fishing. Such networks serve fisheries management objectives by conserving populations of target species within refugia and by providing sources of replenishment through spillover and recruitment (Russ, 2002). Ideally, both sets of objectives can be addressed when designating marine reserves by applying a rigorous selection process. Most of the theoretical benefits of marine reserves are contingent upon the increase of adult fish biomass within their boundaries. For this to occur, the reserve should be designed to encompass all or the majority of a species' home range and be placed in critical habitats. Most reserves have been implemented on an *ad hoc* basis. Poorly designed marine reserves may generate over-confidence about resource protection and misplaced optimism concerning the benefits of marine reserves (Ley, 2005). "No take zones" have proved effective in increasing stocks of commercially exploited species (see also Halpern, 2003 for a review), with the majority of studies recording greater abundance, biomass, size and diversity of fish (e.g., Babcock et al., 1999). Such zones need to be placed in strategic locations (spawning sites, nursery habitats, areas of high connectivity, proximity of resources) to gain maximum benefit.

7.6   Challenges and directions for future research

This thesis has identified for the coast of Queensland the most relevant areas and habitat configuration explaining fish catch distribution, thereby providing guidance where future fisheries research and protection efforts should be targeted. A similar approach incorporating landscape connectivity is required for analyses at finer spatial scales to enable catch information to be related to particular estuarine habitat configuration within an estuary, and to allow for a better understanding of the importance of habitat connectivity to fisheries. Due to increasing habitat fragmentation by human activities (Ray, 2005), connectivity of estuarine habitats will be a critical management issue in the near future. Fragmentation of estuarine habitats can be caused by (1) habitat destruction (Duke et al., 2007); (2) climate change such as through rainfall (reduced freshwater runoff) and temperature alteration or sea level rise causing habitat shift, resulting in the disruption of ecological processes, such as dispersal or migration (Hess, 1994; McCallum and Dobson, 2002; Lee et al., 2006; Bierwagen, 2007); (3) sedimentation (Hovel and Lipcius, 2001), and (4) pollution (Schaaf et al., 1987).

Habitat destruction can be caused by direct removal of habitat but more likely, by unknown habitat degradation, e.g., mangrove dieback caused by herbicides (Duke et al., 2005). More information is needed on effects of water quality on estuarine habitat degradation. In addition, increased sedimentation due to land use changes in the catchments coupled with reduced freshwater run off are causing blocked waterways and alteration of hydrodynamics in estuaries. Estuarine variability through reduced river flow fluctuations is becoming increasingly evident in many estuaries that have heavily impounded catchments, with both plant and animal communities responding accordingly (Whitfield and Paterson, 2003). These changes have major consequences for the migrations and population sizes of diadromous fish species (Bruton et al., 1987). In regard to climate change, alteration of rainfall along the coast of Queensland (Hughes, 2003) is expected to significantly change tidal wetland distribution and with it, the connectivity of estuarine habitats. This thesis also found that high connectivity of landscape often occurs in areas with high annual rainfall. There is, for example, a known link between long-term average rainfall along the coast and the distribution of mangroves (Duke, unpubl. data). Changes to base tidal levels or the magnitude or regularity of freshwater flows will be a major factor in determining the future of wetland connectivity (Lovelock and Ellison, 2007). Sea level rise will also contribute to a shift of habitats to higher elevations, which are often limited by natural or anthropogenic boundaries, thus resulting in overall loss of habitat. Changes in the extent and proximity of habitats will also alter the nature and extent of connecting corridors (Nicholls, 2004). Benefits from a landward progression of mangrove forests by enhancing the benefits of an increase in habitat to shelter will only be short-term. Landward intrusion of mangroves was recorded from 41 photogrammetric studies (1930-1999) showing a salt marsh decline of 45 % and mangrove increase of 55 % (e.g., *Sarcocornia* spp. replaced by *Avicennia marina*, mainly in New South Wales) (Meynecke, unpublished data, Appendix 10). Insights on the effect of sea level rise, rainfall and temperature shifts are needed to identify risks for estuarine fish habitats. Furthermore, studies of the ranging and migratory behaviour of estuarine fish species, not only of individual species but also of specific populations, are required to inform the design of networks of marine protected areas. Strong site fidelity of estuarine fish species (Chapter 5) indicates the importance of habitat protection to avoid that fish has to move to less suitable habitats. Data on foraging, social and reproductive behaviour of fish are important for predicting the effectiveness of "no take" zones. Understanding the

processes that drive the patterns of distribution and habitat use of juvenile fishes within estuarine nurseries is not only critical to the understanding of particular habitats in the overall functioning of estuarine systems, but is also vital information for catch modelling and stock assessment. In order to further assess the value of habitats for fishery species and to go beyond simple linear models, it is necessary to gain additional information on "critical" habitat requirements and processes such as recruitment, post-recruitment mortality and competition, spawning, and species interactions (Beck et al., 2001). This is necessary with the ever-increasing pressure on fish stocks. Seascape ecology and modelling approaches can be used to recommend conservation strategies and actions based on fish movement and trophic linkages. Further information based on structural and functional connectivity between habitat types are required to contribute to the development of a hierarchical seascape approach based on geomorphic features of the marine environment and contribute to marine protection as well as resource management for coastal communities. Finally, the ecosystem services provided by tidal wetlands need to be recognized as natural capital (Barbier and Heal, 2006) to prevent undue destruction and long-term ecological and economic costs.

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